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WRRI Report No. 082

TERRESTRIAL CONTRIBUTION OF N TO STREAM WATER IN MANAGED AND UNDISTURBED FORESTED WATERSHEDS

Technical Completion Report

Project No. B-053-NMEX

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Technical Completion Report

Project No. B-053-NMEX

New Mexico Water Resources Research Institute
in cooperation with
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INTRODUCTION

Forests, typically occupying steep topography and supplied with abundant precipitation, are the source of a large proportion of the water that reaches streams and lakes. The drainage from forests ordinarily carries a relatively low concentration of nutrient elements, nevertheless, its aggregate amount is sufficient to transport large quantities of dissolved and suspended material into streams and lakes each year. This material plays a significant role in the productivity, diversity, and overall health of these aquatic ecosystems. The same forested areas which supply stream water are also subject to a relatively large number of pressures (e.g. recreation, timber harvesting) which involve some degree of disturbance. Numerous studies have shown that the quality of streams can change as a result of man's use causing large increases in the quantities of dissolved and suspended material entering streams and lakes, possibly to the point of eutrophication (Likens and Bormann 1970, Taylor 1971, Wang and Evans 1970, Timmons et al. 1970, Kurtz 1970, Holt et al. 1970, Fredricksen 1971, Edmondson 1970, Ryther and Dunstan 1971). One element which can be significantly affected by forest disturbance and which plays a major role in eutrophication is nitrogen; however, the relationship is complex.

Most of the nitrogen in a forest originated from atmospheric sources (e.g. precipitation, sedimentation, fixation of gaseous N_2); however, most of the nitrogen in a stream system comes from the terrestrial portion of the watershed. The loss of nitrogen to stream water seems to be regulated primarily by biological processes in the soil system and normally this loss is small, in many cases less than the precipitation input of nitrogen to the forest (Likens & Bormann, 1970). As a result of forest disturbance, the soil environment may be significantly changed resulting in complex patterns of nitrogen loss. In the hardwood forests of the Northeast, timber harvesting caused significant losses of nitrogen as NO_3-N to the point where the stream water was unsafe for consumption (Likens and Bormann 1970). In the coniferous forests of the West, losses of nitrogen following timber harvest have occurred but in general they have been much less drastic (Gessel and Cole 1965). There does not seem to be information on the deciduous forests of the West (e.g. aspen) which may respond more like the eastern decidous forests.

OBJECTIVES

The overall objective of this study was to understand the physical mechanisms responsible for the release of organic N, NO_3-N, and NH_4-N from a variety of terrestrial ecosystems to streams. Specific objectives were:

1) quantify the output of nitrogen forms from gaged watersheds in relation to precipitation, temperature, soils, and vegetation type. This information is necessary to understand natural levels and variation in streams;

2) using laboratory experiments, identify the influence of temperature and precipitation on changes in the forms of nitrogen in soil and the flushing of nitrogen from the soil;

3) evaluate man-caused disturbances (i.e. ski area development, timber cutting in aspen) in terms of increased nitrogen levels in streams.

Figure 1 shows a generalized flow diagram indicating the possible fates of nitrogen in a terrestrial ecosystem. The amount of soluble nitrogen forms which can enter a stream is dependent on a number of factors:

1) nitrogen contained in precipitation entering the system;

2) mineralization of organic nitrogen during tissue decomposition;

3) uptake (and immobilization) of nitrogen primarily by plants and microbes;

4) volatilization (NH_3) and denitrification losses;

5) and the presence of sufficient water to transport nitrogen into the drainage system of the watershed.

These factors all represent variables which can be expected to vary between ecosystems and with time in a particular ecosystem. The complexity of reactions which can, and no doubt, do occur among these factors in the soil system preclude a detailed analysis; however, estimates of the "free" nitrogen of the soil solution plus precipitation and temperature patterns should allow an explanation of the variability of nitrogen levels in watershed drainage systems.

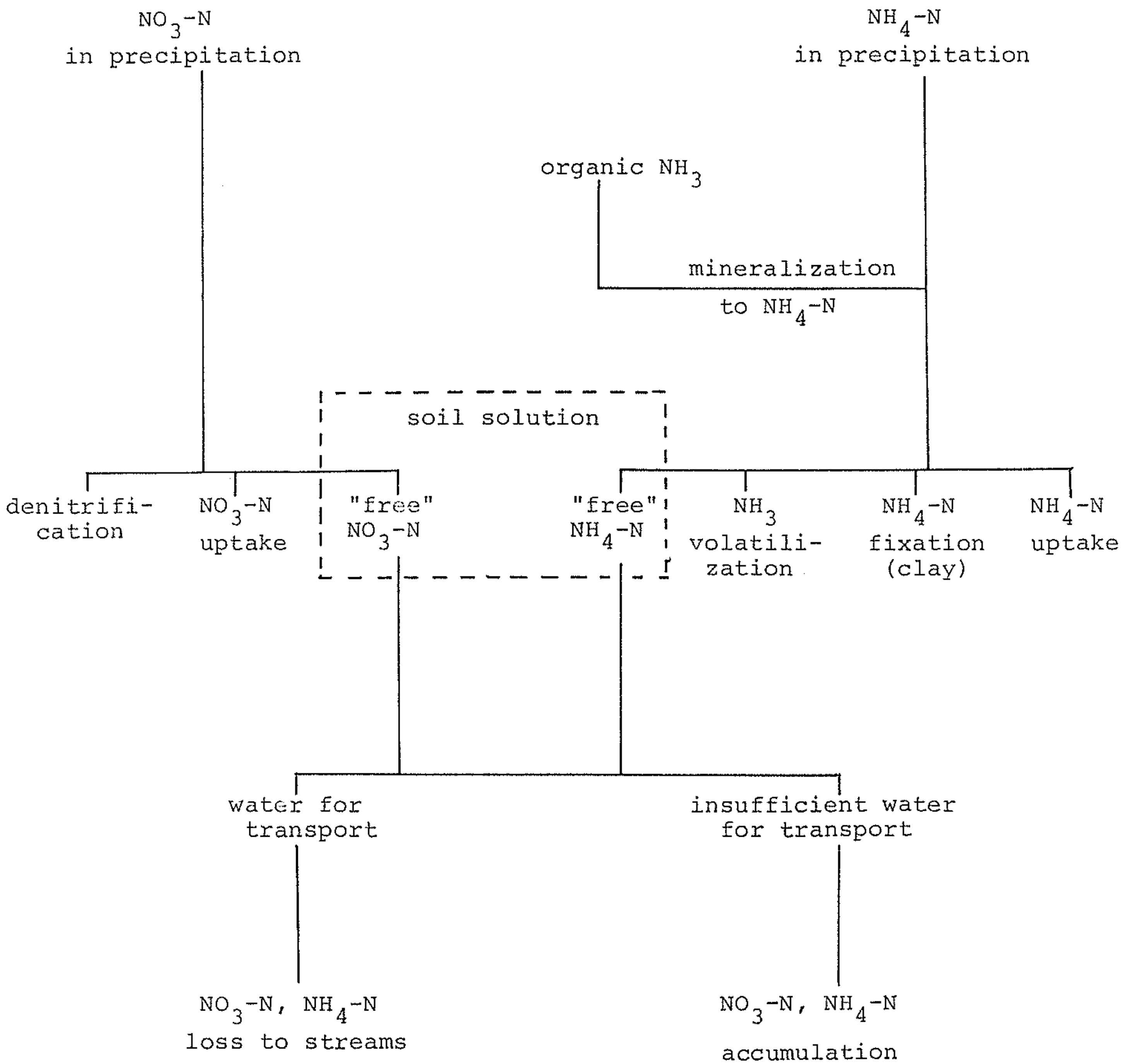


Figure 1. The major pathways and sinks of nitrogen in a watershed ecosystem.

PROCEDURES

Both the ecosystem concept and laboratory experiments were used to evaluate the N transfer from terrestrial to aquatic systems. The ecosystem is a part of the surrounding biosphere connected by a system of inputs and outputs. Knowledge of input-output relationships is necessary if we are to fully understand the nitrogen balance of both natural and manipulated ecosystems as well as aquatic ecosystems intimately linked to the terrestrial systems. Measurement of these critical input-output relationships presents difficulties particularly in studies of nutrient budgets. Nutrient flow is strongly geared to the hydrologic cycle and as a consequence, measurement of nutrient input and output requires simultaneous measurement of hydrologic input and output.

The small watershed approach to ecosystem studies is a concept which used the nutrient-cycle, hydrologic-cycle interaction to its advantage (Bormann and Likens, 1967). This is especially so if the watershed is underlain by a tight bedrock. From continuous measurement and analysis of precipitation and sedimentation entering a watershed the temporal input of N forms can be calculated in terms of grams per hectare. Geologic input, alluvium and colluvium, would not occur because there would be no transfer between adjacent watersheds. The major output of N would be geologic output, dissolved and particulate matter in either stream water or seepage water moving downhill above the impermeable base. A weir, anchored to the bedrock, will force all drainage water from the watershed to flow over the notch where it can be accurately measured. Chemical analyses of dissolved and particulate matter in the outflowing water provide an estimate of geologic output expressed as grams of an element lost per hectare of watershed. This is also the measure of input to the stream ecosystem.

The study site contains 8 gaged watersheds underlain by a tight bedrock (E. Cobb, W. Dien, U.S.G.S., personal communication) which were used to evaluate the contribution of N to streams from a number of terrestrial ecosystems. An additional ungaged watershed (P-J) also was used. These watersheds represent vegetational communities ranging from pinon-juniper (2365 m) to alpine tundra (3734 m) (see fig. 2, table 1).

Nitrogen inputs

Seven precipitation stations have now been established over the elevational gradient to measure inputs. These contain recording gages, storage gages, and polyethylene gages for chemical analysis. Weekly collections (biweekly or monthly during the winter) were made when possible and analyzed for NO_3^-, NH_4^+ and organic N. Organic N was analyzed by the microkjeldahl procedure, NH_4^+ by distilling into boric acid and titration (Bremner 1965), and NO_3^- by the ultraviolet procedure, (Standard Methods, 1965). In January of 1976 a Technicon Autoanalyzer was purchased and the above analyses were performed on subsequent samples using that equipment.

Nitrogen outputs via stream water

Weekly stream samples were taken from all watersheds and analyzed for NO_3^- and NH_4^+. Organic N analyses were made on samples from 4 watersheds. The analytical procedures were the

Figure 2. Study watersheds in the Santa Fe National Forest near Santa Fe, New Mexico. These watersheds represent vegetational communities ranging from pinon-juniper to alpine tundra (2365 m to 3734 m).

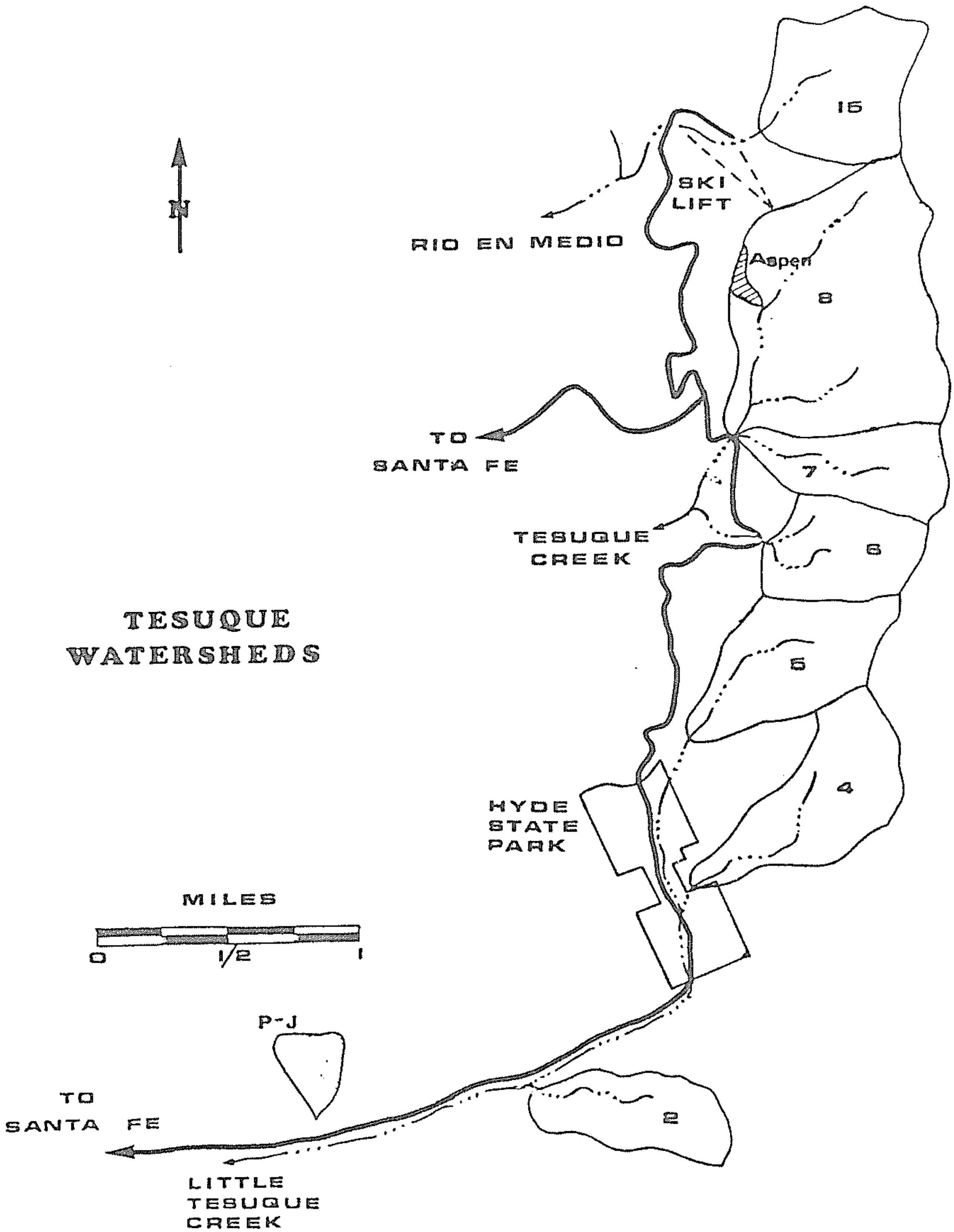


Table 1. Characteristics of the Tesuque watersheds; vegetation, area, elevation. Vegetation data from U.S. Forest Service

	Watershed								
Vegetation (ha)	P-J	2	4	5	6	7	AW-1	8	15
Pinon-Juniper	17	10	0	0	0	0	0	0	0
Pine	0	106	0	11	0	0	0	0	0
Mixed Conifer	0	0	100	18	0	0	0	0	0
Spruce-Fir	0	0	80	112	103	64	0	46	123
Aspen	0	0	0	23	19	13	3.4	203	0
Subalpine grassland	0	0	0	0	0	23	0	162	0
Alpine tundra	0	0	0	0	0	0	0	4	40
Total (ha)	17	116	180	164	122	100	3.4	415	163
Elevation (m)									
maximum	2460	2850	3383	3444	3520	3490	3231	3658	3734
minimum	2365	2423	2621	2804	2972	2987	3109	2941	3231

same as those used for analyzing precipitation. This data quantifies N leaving the terrestrial ecosystem via stream-flow and allows us to calculate N budgets for the watersheds. The budgets for the ungaged watershed (P-J) were calculated using an estimate of runoff based on the precipitation-evapotranspiration relationship for the elevational gradient (see Gosz, 1975). In addition to the weekly collections, collections were attempted at 4 hour intervals before, during, and after a number of rains.

Nitrogen content of the soil

In addition to inputs and outputs, the concentration of N in the soil pool and changes in this pool are important in evaluating mechanisms of loss to streams. Four vegetational communities were studied: pinon-juniper, mixed conifer, aspen, and spruce-fir. A representative area was sampled during the summer of 1974 for each community. During 1975 only the mixed conifer and aspen communities were intensively sampled. Core samples (2 cm dia) of the soil were taken for the 0-15 cm, 15-30 cm and 30-45 cm strata for each community. Organic layer samples were taken as described by Gosz et al. (1973). Analyses were made for total N (Kjeldahl procedure), an extractable NH_4^+ and NO_3^- (see Bremner, 1965).

Laboratory experiments

To aid interpretation of field data regarding N buildup in the soil a number of laboratory experiments were performed. Replicate blocks of soil (14 cm x 20 cm surface area, 10 cm deep) were collected in polyethylene containers from aspen, mixed conifer and spruce-fir communities. These blocks of soil were flushed with deionized water (a hole was drilled into the bottom of the container to allow drainage) until virtually all of the NO_3-N was removed. They were then subjected to a particular combination of temperature and drying regime for a two week interval and again flushed to estimate the potential for producing NH_4^+ and NO_3^- forms of nitrogen.

Management effects

The above procedures were designed to quantify N budgets for undisturbed vegetational types, quantify changes in the forms of N in soil as affected by temperature, vegetation type, precipitation, and explain the quantities and variation of N in stream water draining these vegetated watersheds. They also provide base line data for evaluating the effects of land management on levels of N in streams draining these areas. Two management operations have been performed on the study area which can be analyzed in terms of increased nitrogen contributions to streams:

a. Aspen thinning to increase water yield - an aspen watershed adjacent to a gaged aspen watershed had a 25% thinning cut in late October of 1975. Through the use of discharge measurements on a small flume at the base of this watershed and the gage data of the adjacent watershed we were able to estimate any change in water yield. Weekly chemical analyses were made on the stream water to allow us to quantify any change in the amount of nitrogen lost from the watershed.

b. Ski trail timber cuts - Watershed 15 is adjacent to the Santa Fe Ski Area. The ski area has been expanded into W-15 with additional ski lifts and trails developed. This involved timber cutting and some soil disturbance. Since watershed 15 has had weekly chemical analyses since June of 1971, chemical analyses of stream water for total N, NH_4^+, and NO_3^- during and after the timber cutting were used to quantify any difference in loss of N from the watershed.

RESULTS

The average concentrations and variability of various N forms in stream water both among watersheds and overtime within a watershed, are shown in Table 2. The pattern for NO_3-N is rather complex showing relatively high concentrations in both high and low elevation streams. The streams of intermediate elevations have lower concentrations which do not differ significantly ($P > .05$) although there seems to be a trend of higher average concentrations in streams draining mixed conifer watersheds (W-4, W-5) for 2 of the 3 years. There also is a trend toward increased variability in NO_3-N concentrations (larger standard errors) with a decrease in elevation. This trend parallels the variability of other inorganic water quality parameters (Gosz 1975, 1977a). The average concentrations of NH_4-N show a more pronounced relationship with elevation which follows patterns of inorganic cations; however, the variability is nearly uniform among the different watersheds ($P > .05$). Organic N analyses were made on 4 of the streams during the study period. Although the average concentrations among watersheds were not significantly different during a given year ($P > .05$), there was a consistently higher level of organic N in the stream water of the small aspen watershed. It is obvious that there is a number of

Table 2. Average annual concentration (mg/l) and variation (standard error) of nitrogen forms in the stream water of the Tesuque Watersheds during a 3 year period.

Watershed	P-J	W-2	W-4	W-5	W-6	W-7	W-8	AW-1	W-15
					NO_3-N				
1973-74		0.05 (0.01)	0.07 (0.02)	0.09 (0.04)	0.03 (0.01)	0.06 (0.01)	0.04 (0.01)	0.03 (0.01)	0.14 (0.01)
1974-75	0.39 (0.04)	0.03 (0.01)	0.08 (0.04)	0.05 (0.02)	0.05 (0.01)	0.07 (0.01)	0.03 (0.01)	0.07 (0.01)	0.11 (0.01)
1975-76	0.39 (0.04)	0.02 (0.00)	0.01 (0.00)	0.01 (0.00)	0.02 (0.00)	0.04 (0.01)	0.02 (0.00)	0.02 (0.00)	0.11 (0.01)
					NH_4-N				
1973-74		0.29 (0.03)	0.32 (0.02)	0.30 (0.03)	0.12 (0.02)	0.13 (0.02)	0.12 (0.01)	0.18 (0.02)	0.14 (0.01)
1974-75	0.13 (0.01)	0.10 (0.01)	0.12 (0.01)	0.10 (0.01)	0.09 (0.01)	0.08 (0.01)	0.08 (0.01)	0.08 (0.01)	0.07 (0.01)
1975-76	0.12 (0.03)	0.05 (0.01)	0.04 (0.01)	0.04 (0.01)	0.04 (0.01)	0.03 (0.01)	0.03 (0.01)	0.03 (0.01)	0.04 (0.01)
					Organic N				
1973-74		0.06 (0.02)	0.16 (0.05)					0.21 (0.02)	0.13 (0.02)
1974-75		0.20 (0.02)	0.18 (0.02)					0.25 (0.04)	0.21 (0.03)
1975-76		0.34 (0.06)	0.30 (0.06)					0.36 (0.06)	0.22 (0.04)

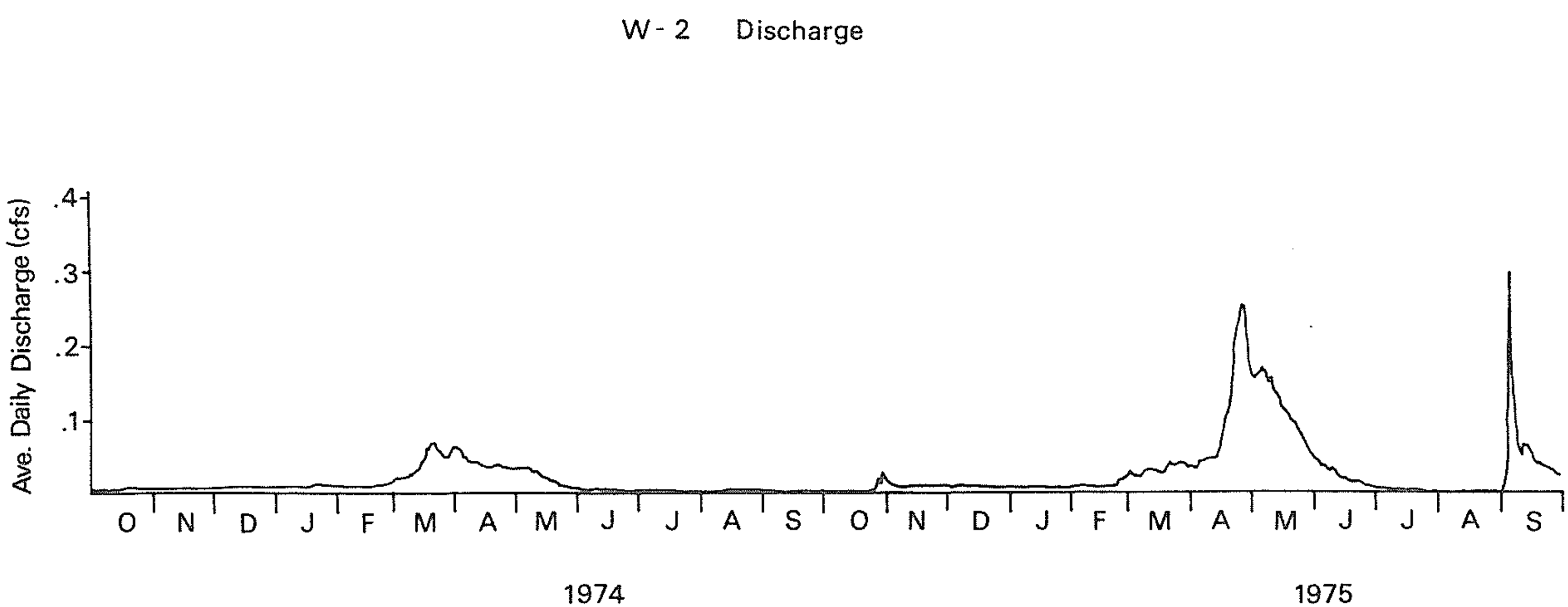
factors involved in controlling the concentrations of nitrogen forms in stream water. Those which appear most important are discussed below.

Variation with stream discharge

The pattern and quantity of stream discharge vary greatly among the study watersheds and it is important to identify any effect on nitrogen concentrations. The discharge rates vary by more than one order of magnitude between the low and high elevation watersheds (Fig. 3,4); however, this degree of difference is not reflected in the nitrogen concentrations of stream water from these watersheds (table 2, fig. 5,6). There are pronounced seasonal differences in discharge rates among all of the watersheds with highest flows occurring during spring months following snow melt and during the late summer months following heavy rain storms. It is difficult to see any resemblance to that pattern for either NO_3-N or NH_4-N.

On an annual basis NH_4-N was not significantly correlated with the log of the instantaneous discharge rate on any watershed during the two years of study. The log transformation was used on discharge rates to normalize the data and satisfy the assumptions of parametric tests. For NO_3-N only one watershed (W-15) showed a significant correlation ($P < .01$) which was negative. Figure 6 does show elevated NO_3-N concentrations in the stream of W-15 during the winter months, a time of low discharge, and low concentrations during the spring and summer months, a time of increased discharge. The other watersheds also showed a negative relationship

Figure 3. Average daily discharge rate (cfs) during a 2-year period on a low elevation watershed (W-2).



WATERSHED 2

Figure 5. NO_3-N, NH_4-N, and organic N concentrations of stream samples from a low elevation watershed during a 2-year period.

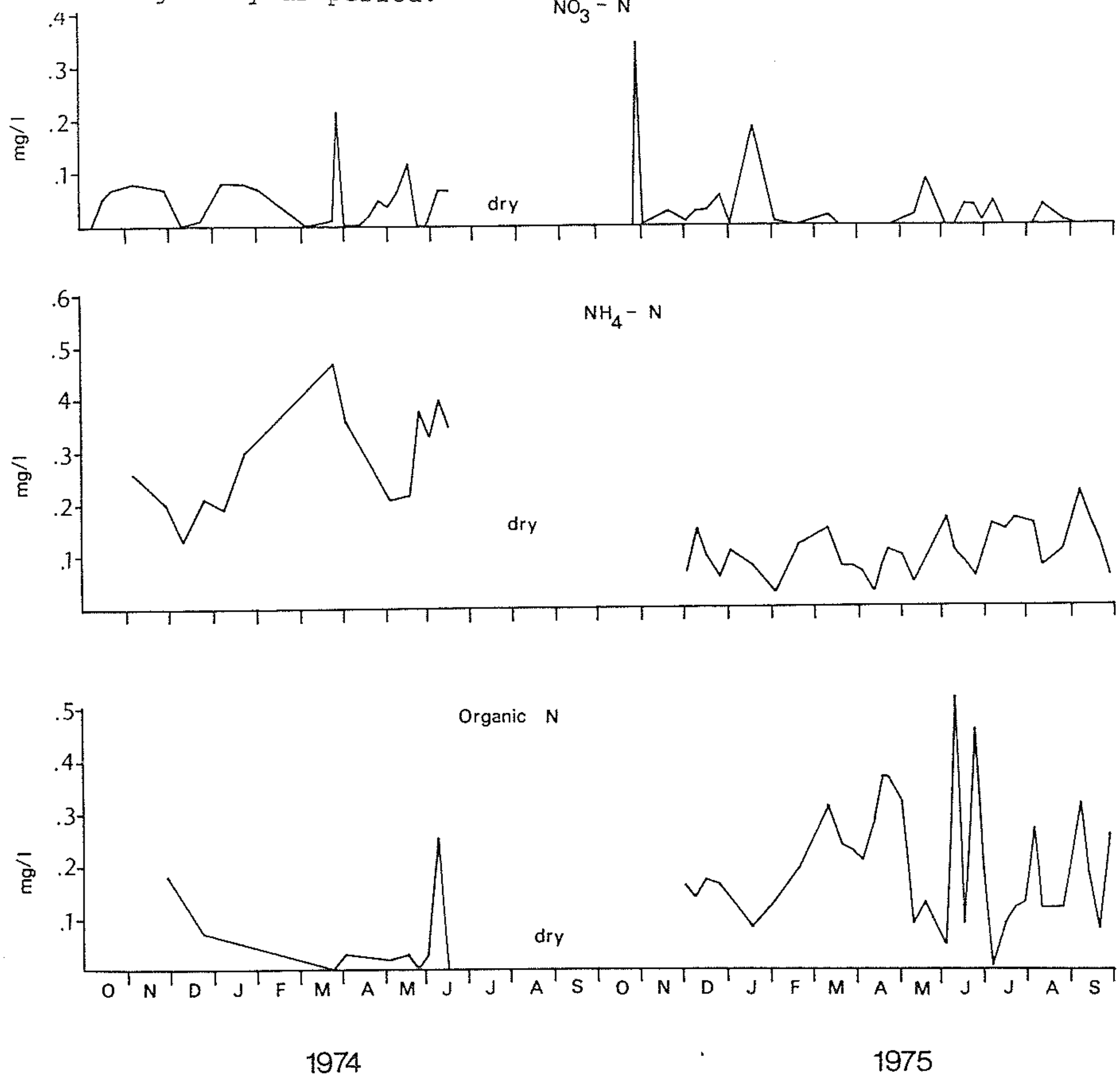
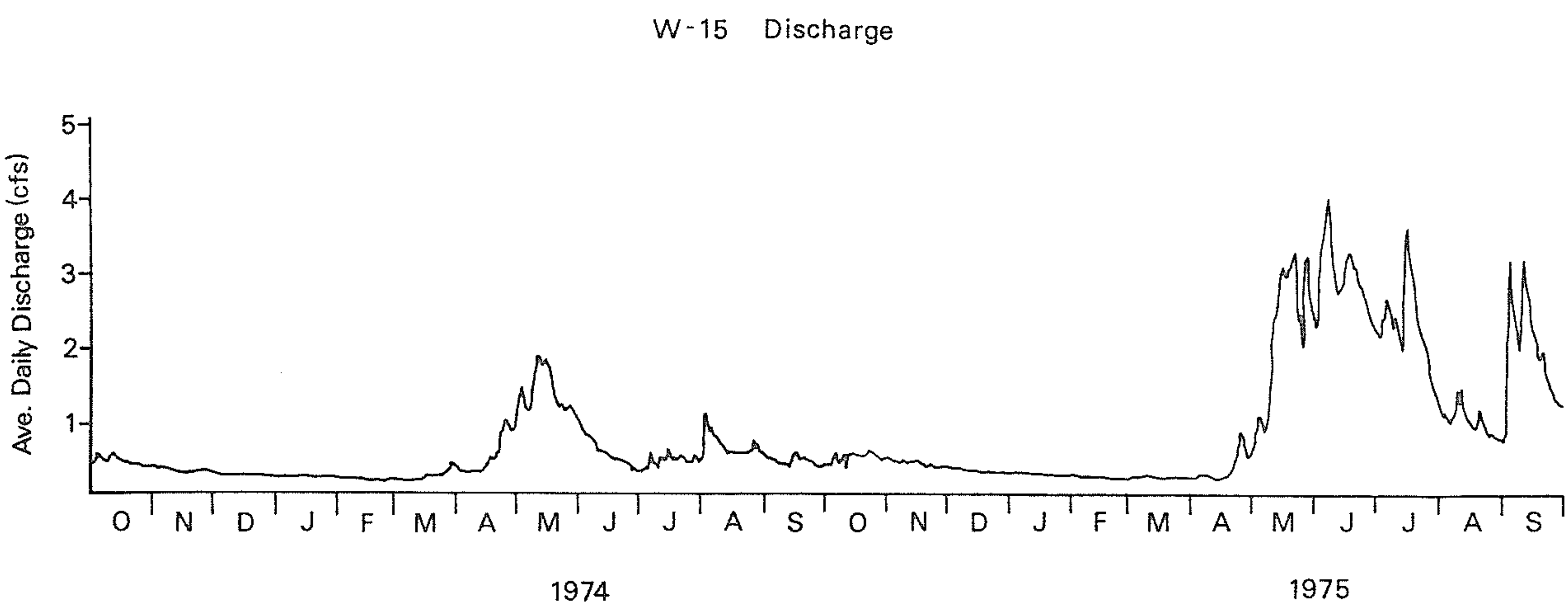
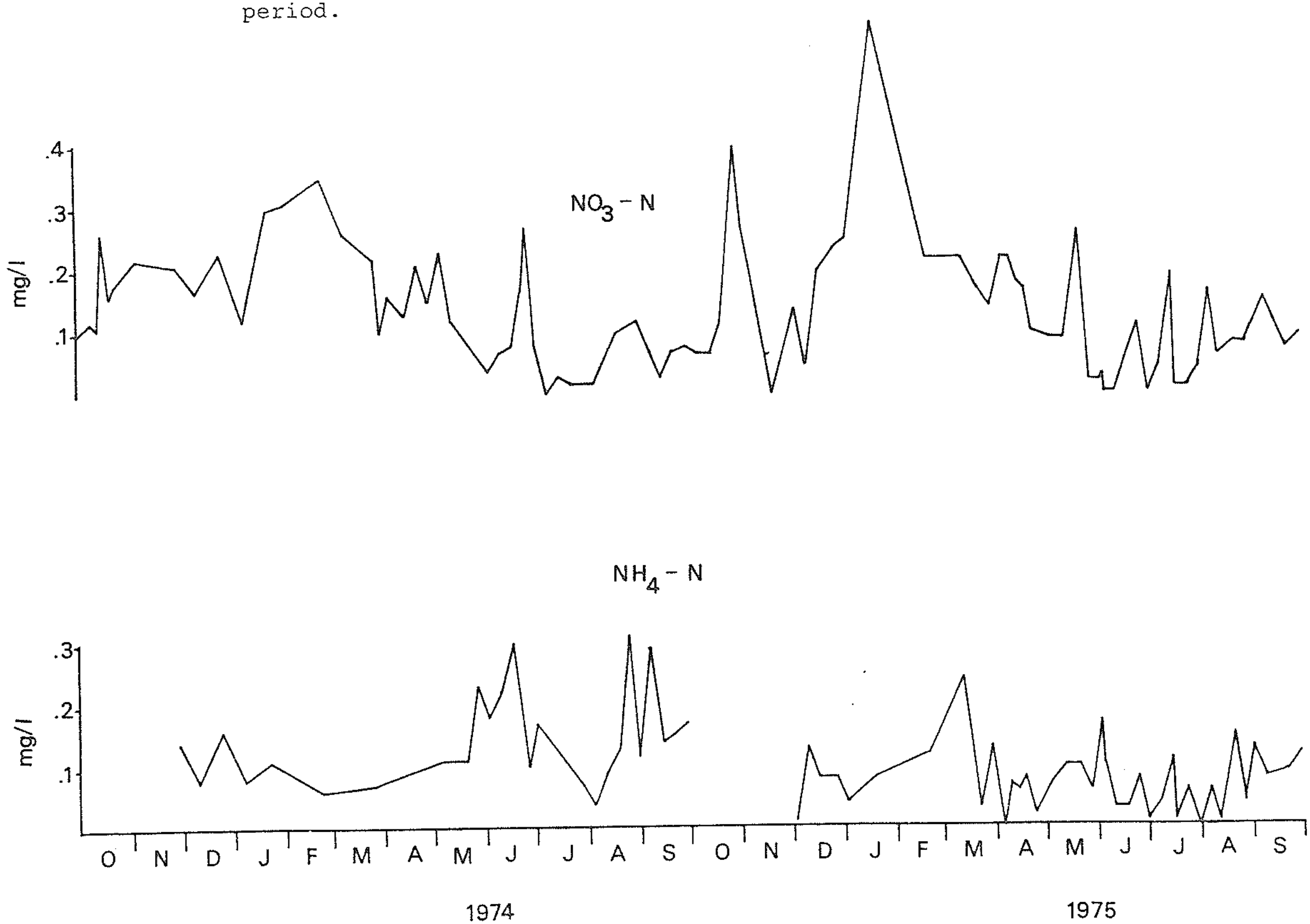


Figure 4. Average daily discharge rate (cfs) during a 2-year period on a high elevation watershed (W-15).



WATERSHED 15

Figure 6. NO_3-N and NH_4-N concentrations of stream samples from a high elevation watershed during a 2-year period.



between NO_3-N and discharge but the correlations were not significant ($P > .05$). Table 3 shows why the correlations of watersheds other than W-15 were not significant. The lower watersheds showed low NO_3-N concentrations during both late winter and spring months, a time when discharge ranged from very low to very high. With increasing elevation, progressively higher concentrations occurred in stream water during the late winter, low discharge months, but only at the highest elevations was the variation enough to have a significant amount explained by discharge. The NH_4-N concentrations generally did not vary much during different seasons and therefore were not correlated with discharge (Table 3).

Organic N demonstrated a variable relationship with discharge. During the 1973-74 water year, when discharge rates were relatively low, organic N was not correlated with discharge on any of the watersheds. However, during the 1974-75 water year, stream flow was much higher and more variable (Figs. 3,4) and organic N was significantly, positively correlated ($P < .01$) with the log of the discharge rate.

Clearly, discharge rate has a variable effect on concentrations of N in stream water, sometimes being associated positively, sometimes negatively, and other times not at all. Frequent analyses of N concentrations during periods of varying discharge at different times of the year can help to demonstrate this. Table 4 shows analyses of stream collections from the high elevation watershed (W-15) during three intervals in 1975. Collections made

Table 3. Weighted average concentrations of N forms in stream water during the four quarters of the 1974-75 water year. Values are in mg/l

	W-2	W-4	W-5	W-6	W-7	W-8	AW-1	W-15
				NO_3-N				
Oct.-Dec.	0.04	0.16	0.07	0.07	0.07	0.03	0.04	0.15
Jan.-Mar.	0.02	0.04	0.06	0.11	0.15	0.07	0.14	0.28
Apr.-Jun.	0.02	0.07	0.03	0.02	0.05	0.03	0.01	0.08
Jul.-Sept.	0.01	0.04	0.04	0.03	0.05	0.04	0.07	0.07
				NH_4-N				
Oct.-Dec.	0.37	0.11	0.08	0.09	0.12	0.08	0.07	0.08
Jan.-Mar.	0.10	0.12	0.07	0.09	0.07	0.09	0.11	0.12
Apr.-Jun.	0.09	0.11	0.09	0.09	0.11	0.07	0.07	0.08
Jul.-Sept.	0.16	0.20	0.11	0.09	0.06	0.09	0.08	0.06

Table 4. Analyses of stream water collections from watershed 15 during several periods in 1975. Correlation coefficients are for concentration and log of discharge rate

Date	Time (hrs)	NO_3-N (mg/l)	NH_4-N (mg/l)	Org. N (mg/l)	Instantaneous Discharge (cfs)
3 June	1350	0.02	0.16	0.07	2.637
	1745	0.00	0.17	0.38	3.280
	2145	0.04	0.07	0.24	3.236
4 June	0550	0.03	0.10	0.40	3.110
	1000	0.00	0.16	0.39	3.010
	1400	0.02	0.10	0.26	2.990
	1820	0.01	0.08	0.24	3.470
	2100	0.02	0.11	0.72	3.515
5 June	0610	0.00	0.10	0.22	3.375
	corr. coef (r) =	$-.16^{ns}$	$-.49^{ns}$	$.56^{ns}$	
25 June	0710	0.11	0.08	0.20	2.775
	1000	0.02	0.02	0.28	2.718
	1215	0.05	0.04	0.24	2.700
	1425	0.21	0.02	0.28	2.605
	1745	0.09	0.08	0.66	2.680
	2040	0.02	0.01	0.33	2.730
	corr. coef (r) =	$-.65^{ns}$	$.32^{ns}$	$-.26^{ns}$	
15 July	1630	0.19	0.11	5.10	10.590
	1800	0.08	0.06	0.48	5.128
	2000	0.02	0.01	0.25	4.117
17 July	0940	0.01	0.01	0.11	3.375
	corr. coef (r) =	0.99**	0.97**	0.96**	

$^{ns}P > .05$
$^{**}P < .01$

during June are representative of the high discharge rates associated with spring runoff (Fig. 4). This represents conditions of snow melt, percolation into the soil and subsequently into the stream. The collection during July represents high discharge following an intense rain storm. During the June runoff conditions, discharge rate was not significantly correlated ($P > .05$) with NO_3-N, NH_4-N, or organic N. During the July collections discharge rate was highly significantly and positively correlated with all three N forms. This array of data suggests that the amount and availability of N forms in the soils of the watershed must be included in the analysis to be able to explain the N concentrations in streams.

Soil Nitrogen

Concentrations of NO_3-N, NH_4-N, organic N, and total N in the soils of 4 communities are shown in table 5. These analyses are for samples taken from a number of depths during the summer and fall of 1974. There is an obvious trend which persists in all of the soils: a significant decrease ($P < .01$) in organic and total N between the forest floor and upper mineral soil and generally smaller and more variable decreases between the upper and lower mineral soil levels. This is a well documented pattern resulting from differences in the organic matter (and organic N) content at various soil depths (Brady 1974). The concentrations of NO_3-N and NH_4-N also show decreases between the forest floor and upper mineral soil and in most cases a continued decrease at lower soil depths. This pattern also is

Table 5: Average concentrations and (standard errors) of N at different soil depths in 4 plant communities during 1974. Data are expressed in % of dry weight. n = 5

Spruce - Fir

	Forest Floor	0-15 cm	15-30 cm	30-45 cm
		NO_3-N		
6 June	.00185 (.00008)	.00052 (.00014)	.00033 (.00011)	.00030 (.00006)
16 July	.00276 (.00047)	.00095 (.00016)	.00046 (.00007)	.00026 (.00007)
31 Aug.	.00225 (.00099)	.00103 (.00008)	.00067 (.00005)	.00062 (.00009)
17 Oct.	.00197 (.00064)	.00065 (.00009)	.00031 (.00010)	.00037 (.00002)
average	.00221	.00079	.00044	.00040
		NH_4-N		
6 June	.01141 (.00111)	.00070 (.00013)	.00042 (.00004)	.00027 (.00003)
16 July	.00890 (.00236)	.00103 (.00020)	.00069 (.00015)	.00052 (.00017)
31 Aug.	.00520 (.00056)	.00095 (.00012)	.00051 (.00011)	.00076 (.00021)
17 Oct.	.00592 (.00130)	.00073 (.00018)	.00024 (.00005)	.00025 (.00003)
average	.00786	.00085	.00046	.00045
		Organic N		
6 June	1.52377 (.11286)	.19395 (.04408)	.20197 (.00295)	.11192 (.02681)
16 July	1.74291 (.07994)	.35285 (.05070)	.23505 (.02381)	.14735 (.01122)
31 Aug.	1.45101 (.17066)	.29762 (.05825)	.23507 (.01751)	.13701 (.01418)
17 Oct.	1.26218 (.07468)	.31207 (.04983)	.18603 (.01360)	.18356 (.02389)
average	1.49497	.28912	.21453	.14496
		Total N		
6 June	1.53703	.19517	.20272	.11249
16 July	1.75457	.35483	.23620	.14813
31 Aug.	1.45846	.29960	.23625	.13839
17 Oct.	1.27007	.31345	.18658	.18418
average	1.50503	.29076	.21543	.14580

Table 5: Continued

Aspen

	Forest Floor	0-15 cm	15-30 cm	30-45 cm
		NO_3-N		
6 June	.00292 (.00018)	.00055 (.00010)	.00060 (.00007)	.00041 (.00006)
16 July	.00183 (.00027)	.00125 (.00054)	.00041 (.00010)	.00028 (.00008)
1 Sept.	.00286 (.00116)	.00076 (.00010)	.00022 (.00008)	.00050 (.00015)
17 Oct.	.00093 (.00029)	.00130 (.00029)	.00106 (.00023)	.00063 (.00023)
average	.00214	.00096	.00057	.00046
		NH_4-N		
6 June	.00461 (.00041)	.00114 (.00032)	.00039 (.00004)	.00024 (.00003)
16 July	.00765 (.00203)	.00061 (.00004)	.00029 (.00004)	.00019 (.00003)
1 Sept.	.00750 (.00156)	.00101 (.00027)	.00053 (.00020)	.00033 (.00004)
17 Oct.	.00673 (.00081)	.00205 (.00089)	.00047 (.00007)	.00043 (.00012)
average	.00662	.00120	.00042	.00030
		Organic N		
6 June	1.95849 (.05673)	.16488 (.04249)	.11125 (.01115)	.11850 (.02990)
16 July	1.40699 (.38786)	.21900 (.03846)	.11949 (.01702)	.09306 (.02155)
1 Sept.	1.65919 (.24686)	.20051 (.01753)	.11661 (.03083)	.12475 (.02734)
17 Oct.	2.12813 (.26863)	.34725 (.07870)	.12686 (.01126)	.10232 (.00679)
average	1.78820	.23291	.11855	.10966
		Total N		
6 June	1.96602	.16657	.11224	.11915
16 July	1.41647	.22086	.12019	.09353
1 Sept.	1.66955	.20228	.11736	.12558
17 Oct.	2.13579	.35060	.12839	.10338
average	1.79696	.23508	.11954	.11041

Table 5: Continued

Mixed Conifer

	Forest Floor	0-15 cm	15-30 cm	30-45 cm
		NO_3-N		
6 June	.00076 (.00036)	.00041 (.00011)	.00033 (.00015)	.00041 (.00012)
1 Sept.	.00193 (.00100)	.00084 (.00014)	.00086 (.00021)	rock
18 Oct.	.00030 (.00019)	.00057 (.00014)	.00063 (.00011)	.00060 (.00015)
average	.00100	.00061	.00061	.00050
		NH_4-N		
6 June	.00145 (.00036)	.00036 (.00007)	.00026 (.00007)	.00021 (.00002)
1 Sept.	.00318 (.00054)	.00036 (.00003)	.00022 (.00002)	rock
18 Oct.	.00464 (.00007)	.00042 (.00009)	.00042 (.00009)	.00017 (.00002)
average	.00309	.00038	.00030	.00019
		Organic N		
6 June	1.20928 (.23623)	.07474 (.00383)	.06216 (.00207)	.05220 (.00538)
1 Sept.	1.04447 (.13676)	.09280 (.00885)	.13159 (.06595)	rock
18 Oct.	1.01814 (.06750)	.08905 (.01127)	.05236 (.01126)	.04638 (.00568)
average	1.09063	.08553	.08203	.04929
		Total N		
6 June	1.21149	.07551	.06275	.05282
1 Sept.	1.04958	.09400	.13267	rock
18 Oct.	1.02308	.09004	.05341	.04715
average	1.09472	.08652	.08294	.04998

Table 5: Continued

	Pinon - Juniper	
	Forest Floor	0-15 cm
	NO_3-N	
7 June	.00103 (.00072)	.00028 (.00004)
1 Sept.	.00208 (.00115)	.00030 (.00005)
11 Oct.	.00303 (.00203)	.00029 (.00005)
average	.00205	.00029
	NH_4-N	
7 June	.00259 (.00105)	.00037 (.00008)
1 Sept.	.00210 (.00066)	.00027 (.00005)
11 Oct.	.00249 (.00072)	.00042 (.00008)
average	.00239	.00035
	Organic N	
7 June	.93588 (.25030)	.10986 (.01964)
1 Sept.	1.11870 (.15050)	.12824 (.01715)
11 Oct.	1.10299 (.08651)	.09651 (.01804)
average	1.05252	.11154
	Total N	
7 June	.93950	.11051
1 Sept.	1.12288	.12881
11 Oct.	1.10851	.09722
average	1.05696	.11218

well documented and the result of the surface soil layers being the major source of organic N and microbes responsible for mineralization and nitrification processes (Brady 1974).

The analyses of samples collected throughout the summer and fall months of 1974 did not demonstrate a consistent pattern through time for N-forms, soil depths, or soil types. There was a great deal of variation among the sampling dates but the number of collections was not large enough to show a correlation with other temporal events in each community. This stimulated several laboratory experiments to identify several of the more significant factors involved in patterns of NH_4-N and NO_3-N availability.

The factors evaluated were temperature and soil moisture conditions. Replicate blocks of soil were leached to remove available NH_4-N and NO_3-N and subjected to either freezing ($-5^{\circ}C$) or warm ($20^{\circ}C$) temperatures for a period of two weeks. In addition, under warm temperature conditions the soils were subjected to either moist or drying conditions. Each soil block was then leached (frozen soils were thawed first) and the quantity of inorganic N calculated from analyses of the leachate. For soils of the spruce-fir and mixed conifer communities the NH_4-N or NO_3-N production was not measureable under freezing soil conditions. Significantly more ($P < .01$) NO_3-N was produced at $20^{\circ}C$ with an additional significant difference ($P < .01$) occurring between moist and drying conditions (table 6, fig. 7). The soil of the aspen community showed the same pattern; however, the production

Table 6. Average NO_3-N production in soils from three communities subject to moist and drying conditions. Containers were closed or left open at 20°C for 14 days to create the moist and drying conditions. Data are expressed as mg of NO_3-N produced per m^2 of soil surface per day

	Spruce-fir	Mixed Conifer	Aspen
Open drying	44.3 mg/m^2-day	32.4 mg/m^2-day	193.7 mg/m^2-day
Moist	0.50 mg/m^2-day	11.7 mg/m^2-day	13.3 mg/m^2-day

Figure 7. NO_3-N concentrations of leachate from conifer soils. Soils were placed in polyethylene containers and either closed or left open at 20°C for a 2-week period. The NO_3-N concentrations and total leachate volume allow a calculation of nitrification.

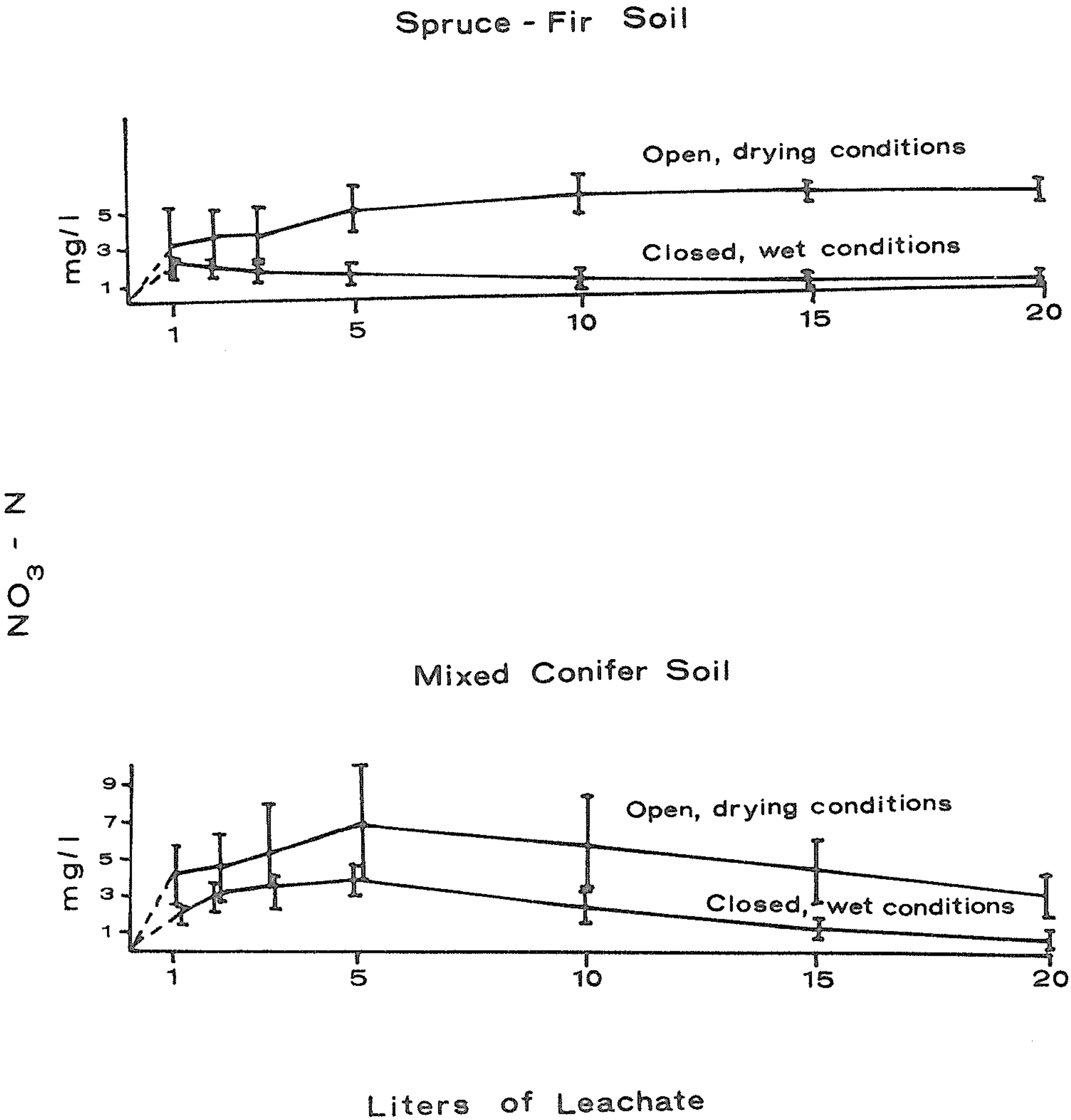
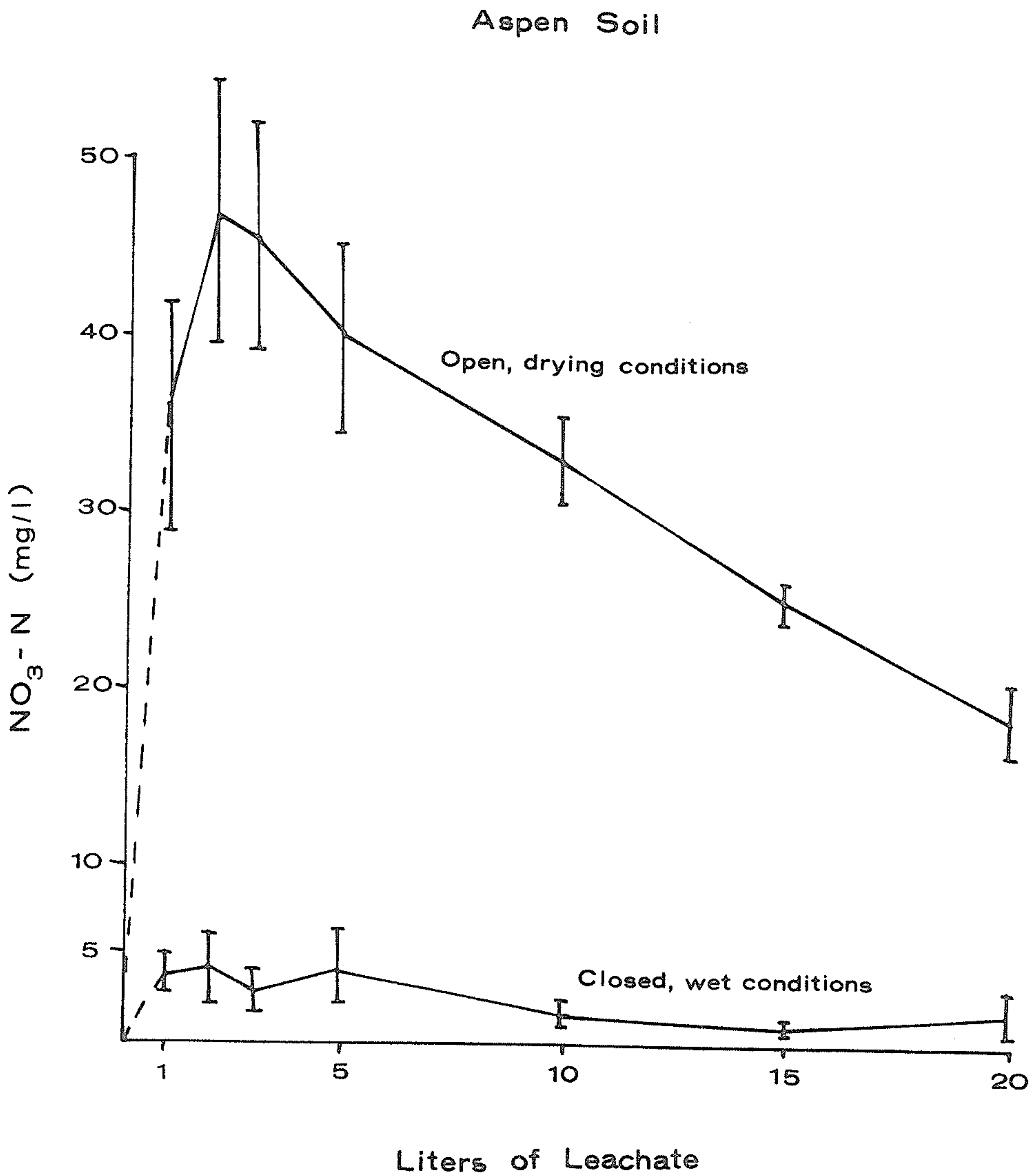


Figure 8. NO_3-N concentrations of leachate from aspen soil. Soils were placed in polyethylene containers and either closed or left open at 20°C for a 2-week period. The NO_3-N concentrations and total leach volume allow a calculation of nitrification.



of NO_3-N under warm and drying conditions was about 5 times greater than in conifer soils (table 6, fig. 8). The NH_4-N production in soils of the three communities was less than the respective NO_3-N production and there was no significant difference between moist and drying conditions (fig. 9). These patterns agree with the literature on the influence of microorganisms and the biogeochemistry of nitrogen (Alexander 1971), in that mineralization leads to an accumulation of NO_3-N under aerobic conditions. Some of the difference between NO_3-N and NH_4-N levels shown in figure 7, 8 and 9 may be a result of the greater difficulty in leaching NH_4^+ from soil because of it's cationic nature. This means that the NH_4-N production from these soils has been underestimated. Nevertheless, these simple experiments suggest that the soil temperatures and moisture levels (influencing O_2 levels) over time will influence the available NH_4-N and NO_3-N present. They also suggest that the nitrification potential in the soil of the aspen community is much larger than in soils of conifer communities and may approach that of the northeastern deciduous forests.

Soil samples were again collected during the summer of 1975 and analyzed for available NH_4-N and NO_3-N. Only two communities were sampled (mixed conifer, aspen) but samples were collected more frequently and before and after precipitation events.

The soil temperature data for 1975 clearly shows the variability which occurs in the upper soil levels (fig. 10). There is an overall warming trend during the summer months but

Figure 9. NH_4-N concentrations of leachate from conifer and aspen soils. Soils were placed in polyethylene containers and either closed or left open at 20°C for a 2-week period.

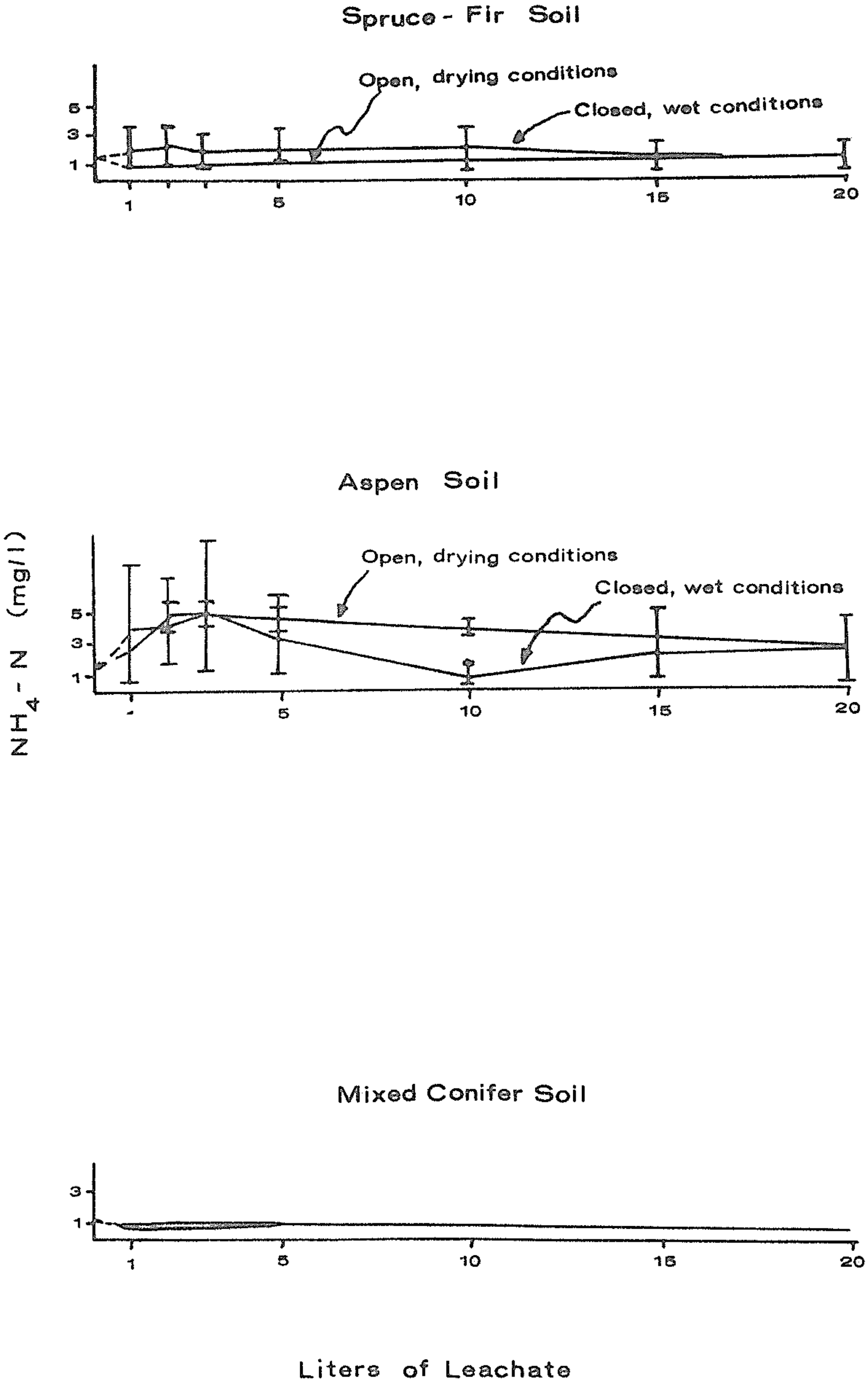
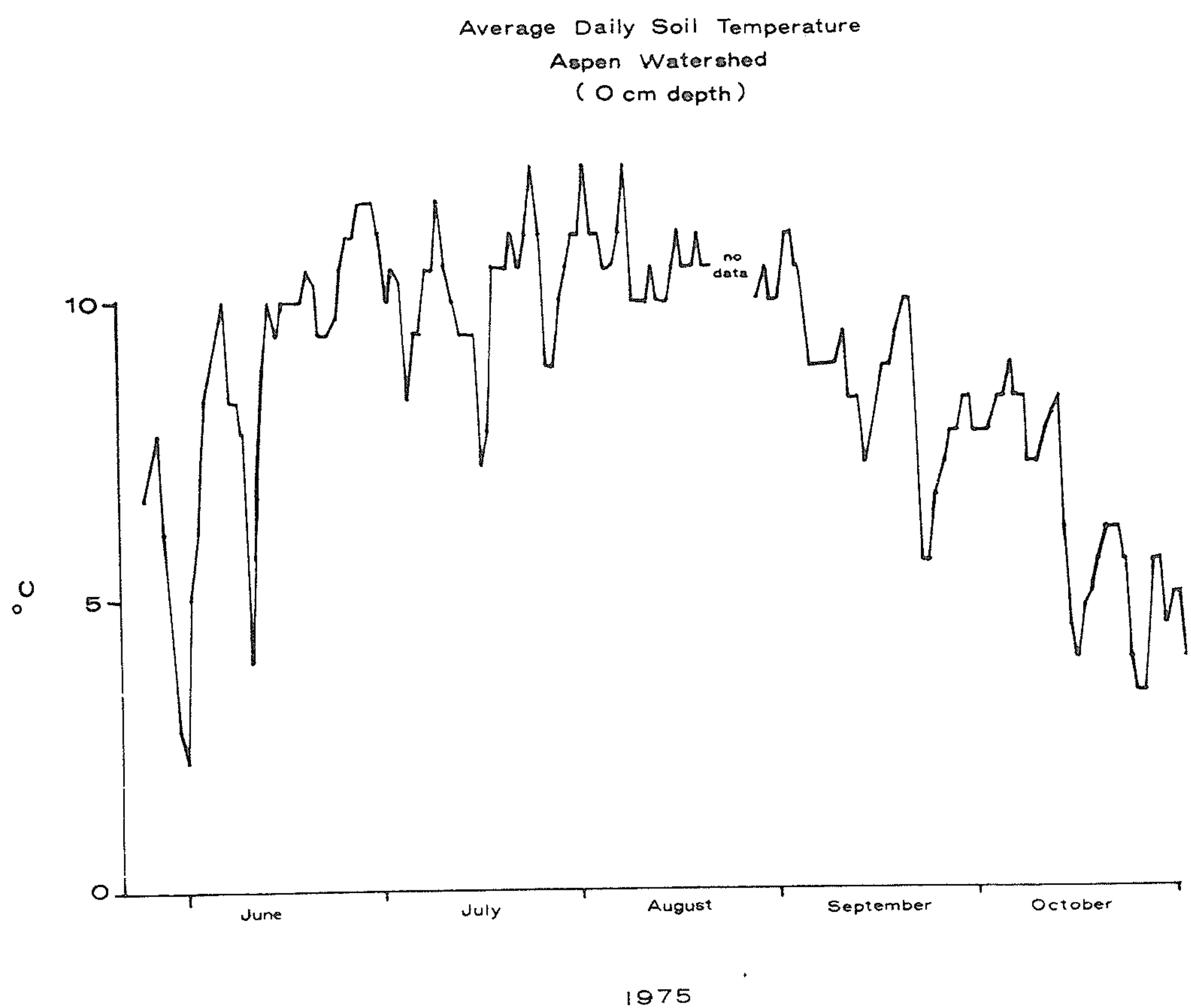


Figure 10. Average daily soil temperature at the A_1-organic layer interface (0 cm depth) of the aspen soil.



there are numerous temperature depressions which are caused by precipitation events. The variability not only represents warm and cool periods but also varying soil moisture and O_2 levels as well as irregular leaching of the available N forms.

The results from the laboratory experiments suggest that these various soil conditions should produce varying NO_3-N levels in the soil. Figure 11 shows the NO_3-N levels of the aspen forest floor in relation to precipitation events. The majority of the NO_3-N production occurs in the forest floor, therefore, that layer should have concentrations which correlate with environmental changes.

In early June when the soil was relatively cool and moist as a result of recent snow melt the NO_3-N concentration was low. After a warming and drying period (late June) the NO_3-N level increased greatly followed by a large decrease after the early July rains. These results are what we would expect, namely, that the production and accumulation of NO_3-N occurs primarily during dry and warm soil conditions. Heavy rains not only flush out the accumulated NO_3-N but cause cool, moist litter conditions which retard NO_3-N production. A break in this trend occurred in mid-July when 6.4 cm of precipitation (mostly as hail) did not seem to remove the NO_3-N pool. An analysis of the nature of the storm and the moisture content of the forest floor helps explain the situation. The moisture level decreased after the June dry period and increased after the early July rains (fig. 12). Several days of warm dry conditions preceeded

Figure 11. NO_3-N concentrations (% of dry weight) in the forest floor of the aspen community and daily precipitation volumes for the summer months of 1975.

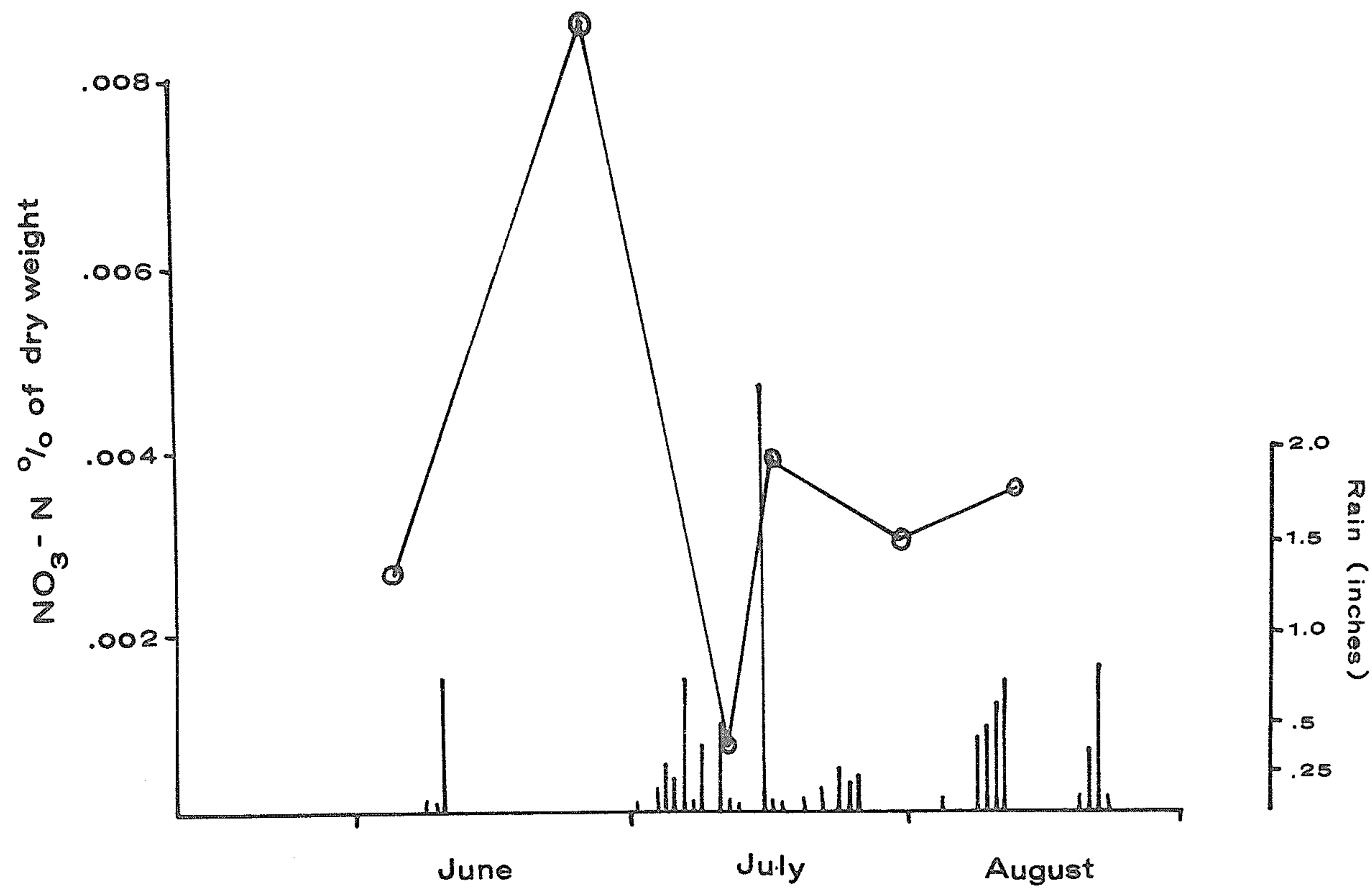
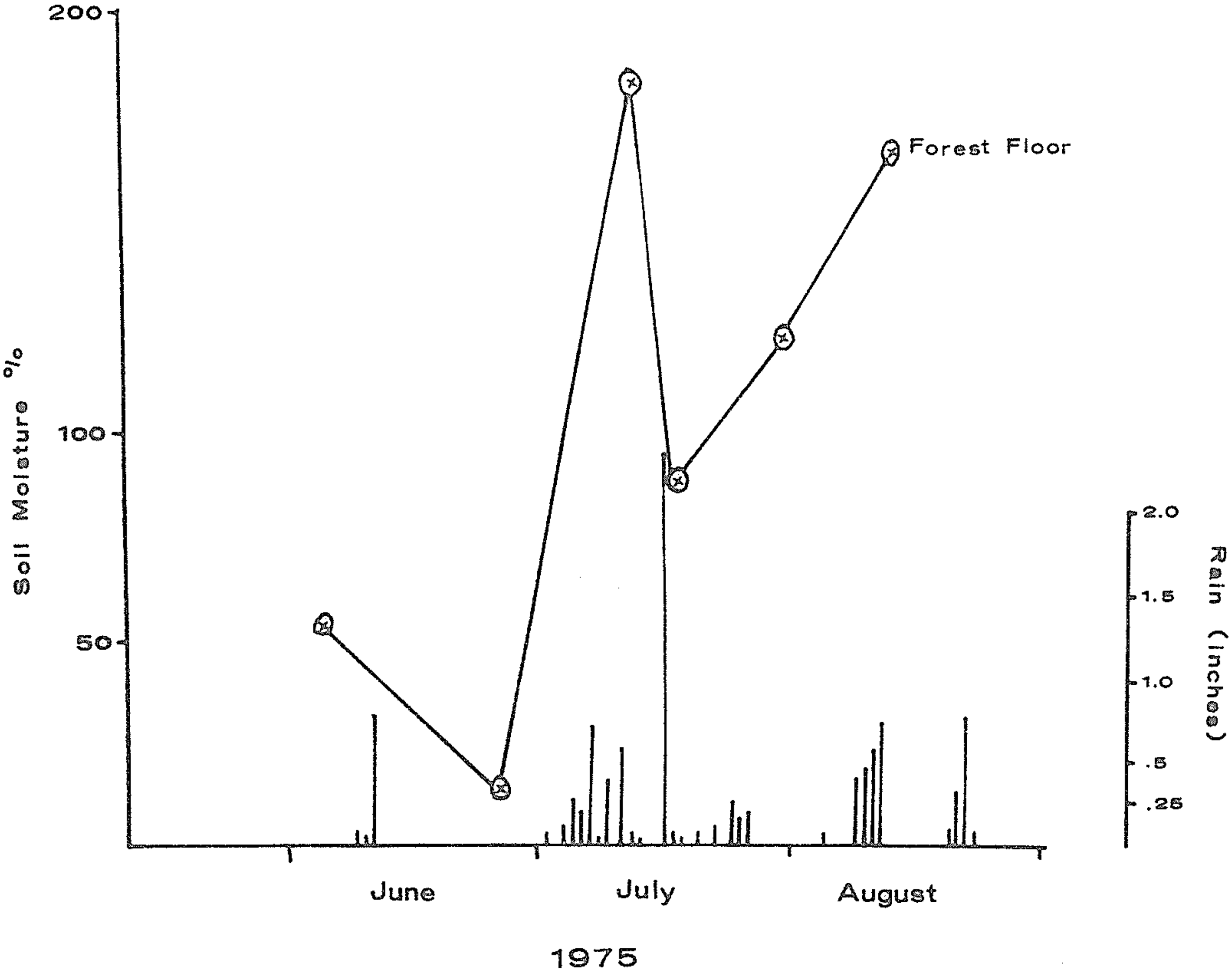


Figure 12. Moisture content (% of dry weight) of the forest floor of the aspen community and daily precipitation volumes for the summer months of 1975.



the large hail storm and the moisture content of the forest floor reflected the drying conditions more than the influence of the storm. The explanation seems to be in the nature of the storm, a truly exceptional event. The total amount of 6.4 cm (2.5 inches) of moisture fell in about ½ hour. Heavy rain fell during the initial few minutes followed by hail (up to 2 cm dia.) for the remainder of the ½ hour. At the time the soil sample was taken, very little of the hail had melted and entered the soil. This helps explain why the moisture content of the forest floor remained low and why the NO_3-N pool was not completely flushed out. A calculation of the discharge during the initial 3 hours after the start of the storm showed only 0.17 cm out of the total of 6.4 cm left the watershed. This was most likely the initial rain falling on or along the stream bed. The periodic rains which occurred during the remainder of the summer and poorer drying conditions (increased cloud cover during the afternoon hours) caused relatively high moisture levels and precluded a large accumulation of NO_3-N in the soil.

Our data shows that NO_3-N production can occur to varying degrees in the upper layers of the soil; however, it is not yet apparent that the NO_3-N level of the stream is closely related to soil NO_3-N levels. Referring again to figure 1 it is clear that the water available for the transport of N to streams, a function of soil moisture content, precipitation amount and intensity, is but one of the factors which influence levels of inorganic N. Nitrogen may also be fixed or taken up by various organisms or lost by volatilization and denitrification. The conditions necessary for a large input of NO_3-N into the stream appear to be those in which the upper soil horizons remain warm and relatively dry for a several week period followed by a heavy rainfall. Under these conditions the NO_3-N concentration in the stream can be expected to increase. The length of time the concentration remains above normal is a function of the pool of NO_3-N and the amount of moisture leaching the soil. NO_3-N levels can be expected to drop rapidly once the pool has been depleted even though the discharge remains high. If the NO_3-N pool is not completely leached out but merely leached down to the water table in the lower soil horizons the NO_3-N concentrations in the stream may stay above normal even though the stream discharge drops following the storm. Both of these situations have occurred in our watersheds which explains the non-significant correlations between NO_3-N and discharge on an annual basis and highly significant correlations on a short time interval following a large storm (table 4).

One additional factor which can influence N levels in streams is the N content of the precipitation which enters a watershed. Precipitation falling directly into the stream or on stream bank areas may cause an increased concentration of N in the stream if the level of N in precipitation is relatively high. Also, the accumulation of snow during the winter months allows a build-up of N in the snow pack. The freezing and thawing conditions in the late winter and spring allow a movement of soluble salts down to the base of the snowpack allowing an increased concentration. The first snow melt water which enters the stream can cause above normal concentrations followed by below normal concentrations when the remainder of the snow melt water enters the stream. The degree to which this pattern is seen in stream chemistry of N depends on the concentration in the snow pack and the rate of snow melt.

Nitrogen Inputs and Outputs

Calculating total N inputs in precipitation and sedimentation and outputs in stream water can indicate the ability of various communities to modify the N levels in streams. It also indicates the total amount of N transported to downstream ecosystems. Table 7 shows inputs, outputs, and net changes for our watersheds in kg/ha-year. The most striking feature is the low total output of all N forms causing a net accumulation. Virtually all of the NO_3-N which enters the watershed seems to stay there, or at least is not lost from the watershed as NO_3-N in stream water. Organic and NH_4-N are lost in somewhat larger quantities but there is still a large net accumulation. The only visible trend among the watersheds occurs for NH_4-N which seems to be accumulated to a greater extent at the intermediate elevations in the mixed conifer communities. This pattern is also found for other cationic species (Gosz 1975, 1977); however, it is not yet known whether this pattern is caused by features of the plant community (productivity, diversity), soil conditions, or modification of the hydrological cycle. The net accumulation

Table 7. Input, output, and net change for various forms of nitrogen for the Tesuque Watersheds during the 1974-75 water year. Data are expressed in kg/ha

	P-J	W-2	W-4	W-5	W-6	W-7	W-8	AW-1	W-15
					NO_3-N				
Input	1.4	2.7	2.7	2.7	2.7	2.8	2.8	2.8	3.2
Output	0.04	0.004	0.06	0.05	0.08	0.25	0.14	0.13	0.55
Net Change	+1.4	+2.7	+2.6	+2.6	+2.6	+2.6	+2.7	+2.7	+2.6
					NH_4-N				
Input	1.4	1.8	1.8	1.7	1.7	1.7	1.8	1.7	2.1
Output	0.02	0.03	0.13	0.12	0.25	0.40	0.32	0.28	0.43
Net Change	+1.4	+1.8	+1.7	+1.6	+1.4	+1.3	+1.5	+1.4	+1.7
					Organic N				
Input	1.9	1.7	1.7	2.1	2.4	2.8	2.8	2.8	3.3
Output	-	0.06	0.23	-	-	-	-	0.82	0.99
Net Change	-	+1.6	+1.5	-	-	-	-	+2.0	+2.3

of N by all communities does support other studies of N budgets indicating the ability of all communities to trap various forms of atmospheric N.

The small output of NO_3-N from the aspen watershed (AW-1) raises additional questions concerning the ability of precipitation to leach the NO_3-N pool from the soil. Figure 11 shows that the NO_3-N concentration of the aspen forest floor increases and decreases greatly as a result of moisture and temperature conditions. It would seem as though the significant drop in concentration between June 25 and July 11 would result in a large increase in the NO_3-N concentration of the stream. The decrease in the forest floor from .0087% to .0007%, and the weight of the forest floor (50,000 kg/ha) predict that about 4 kg/ha of NO_3-N should have been leached from each hectare of the watershed. Table 7 shows that only about 0.1 kg/ha of NO_3-N was lost for the entire year. Considering the production of NO_3-N which could have occurred during other portions of the summer, it is obvious that the community has the ability to greatly regulate the loss of N which is present in the NO_3^- form. There are three process which seem to be the most logical for explaining the ability of the aspen community to regulate NO_3-N loss:

1) the dissimilatory pathway of denitrification:

$$NO_3^- \longrightarrow NO_2^- \longrightarrow NO \longrightarrow N_2O\uparrow \longrightarrow N_2\uparrow$$

2) the electron sink pathway:

$$NO_3^- \longrightarrow NO_2^- \longrightarrow NH_4^+$$

3) plant uptake.

While there may be others (e.g. assimilatory pathway), these would appear to be the major ones which could operate best given the forest floor conditions of the aspen forest. The dominant pattern is a reduction in the NO_3-N concentration following precipitation. The cool, wet conditions following precipitation create anaerobic conditions which stimulate the synthesis of enzymes required in the dissimilatory and electron sink pathways (J. Tiedje, Michigan State Univ., personal communication). This increased activity would result in an increased volatilization of N_2O, N_2 and/or an increased concentration of NH_4-N (Stanford et al. 1975). Table 8 shows that the concentrations of NO_3-N and NH_4-N in the aspen forest floor during 1975 are inversely correlated ($P < .01$). The large increase in NH_4-N following the early July rains amounts to an increase of 6.2 kg/ha. This is much larger than the 4 kg/ha decrease in NO_3-N meaning that mineralization must have occurred in addition to any conversion of NO_3-N to NH_4-N by the electron sink pathway. The moisture which causes anaerobic conditions also decreases the soil temperature (see fig. 10); however, this does not seem to adversely affect denitrifying organisms as one might expect. A study of isolates of denitrifying organisms from soils of temporate latitudes showed that 68% of the isolates could grow at $4^{o}C$, 22% at $28^{o}C$ and only 10% at $41^{o}C$ (Tiedje, personal communication). Thus, the cool, moist conditions following precipitation would seem to provide favorable

Table 8. Concentrations of NO_3-N and NH_4-N in the forest floor of the aspen community during the summer of 1975. Values are % of oven-dry weight

	NO_3-N	NH_4-N
4 June	.00260	.00795
25 June	.00870	.00565
11 July	.00070	.01802
15 July	.00390	.01257
29 July	.00300	.01125

conditions for the conversion of NO_3-N to other forms. The NO_3-N which is leached to the lower soil horizons may suffer the same fate since the moist, cool conditions of these horizons also are conducive to denitrification.

The third process which may be involved in minimizing N loss, plant uptake, can be estimated from productivity values and N concentrations. Our studies have shown that the above ground productivity (understory plus overstory) of the aspen community is about 5,000 kg/ha (Gosz, unpublished data). Assuming a N concentration of 2% in this production, the annual requirement for N is about 100 kg/ha. It is not known how much of this requirement comes from the soil (versus translocation out of senescent tissues) and whether the requirement is for NO_3-N and/or NH_4-N; however, it represents a relatively large capacity to prevent N loss (Vitousek 1977).

Land Management Effects

The data on input-output relationships identify an ability of the terrestrial ecosystem to regulate the amount of N entering stream ecosystems. This seems to be a characteristic of most, if not all, undisturbed terrestrial ecosystems and it becomes important to ascertain whether various land management activities modify this ability. Two land management activities were studied during the course of this project; ski area development at the Santa Fe Ski Basin and timber cutting in an aspen forest community.

Ski Area Development:

The Santa Fe Ski Basin is located about 15 km northeast of Santa Fe, New Mexico in the Sangre de Cristo Mountains. The ski area is located at the headwaters of the Rio en Medio, a perennial stream. Prior to 1974, skiing activity occurred primarily on the slopes adjacent to watershed 15 (fig. 2). Only limited skiing occurred in the watershed itself by the more adventuresome individuals. During September and October of 1972 a poma lift along the south boundary of W-15 was replaced by a poma lift with larger capacity. Since no additional trees were cut the major impact was that of constructing new concrete piers for the towers. During the summers of 1973 and 1974, an additional poma lift was constructed in the center of W-15, near and parallel to the stream. The major construction activity consisted of blasting to remove boulders in the lift path and pouring 14 concrete footings for the towers. The lift required very little tree removal since it was placed in a relatively open area. During the summer of 1974, two ski runs were cleared in W-15 to serve the new poma lift. Again, because of the rather open terrain concerned, the number and basal area of the trees removed was small. A total of 907 stems ranging from 2 to 33 cm (1 to 13 inches) diameter (stump height) were cut amounting to a total basal area of 7.3 m^2 (68.5 sq. ft.) on a land area of about 2 ha.

Throughout all of the activity during these years, stream chemistry appeared normal (table 2, fig. 6). While some variation in annual average concentrations did occur it was not large enough to indicate anything other than hydrological differences (e.g. total precipitation, storm patterns) between years. This agrees well with analyses of other inorganic cations during this same period (Gosz 1977b).

A number of studies have demonstrated that NO_3-N, as well as other cations, increase greatly in stream water after vegetation clearing (Likens and Bormann 1972, Likens et al. 1970, Bormann et al. 1974, Fredriksen 1971). The reason NO_3-N did not increase after tree removal in the study watershed may be related to the small area cleared, the low density of trees with grass cover in open areas, and the small size of individual trees. Another factor may have been the cutting practice. Since most of the trees were growing in the open, they had branches down to the ground. The stems were cut above the lowest whorl of branches for the purpose of holding the snow pack; however, this also had the effect of keeping the root system alive and functioning.

Construction of the poma lift in the center of the watershed represents a greater impact since it disturbed the soil surface, required blasting to remove boulders, and is close to the stream. The reason that this activity did not influence stream chemistry appears to be the very low rainfall during the construction period. The soils were relatively dry and the majority of the moisture which did fall was retained by the soil. More precipitation events or more intense precipitation would probably have caused greater erosion of the disturbed area and increased stream concentrations of at least the organic form of nitrogen.

Timber Cutting in Aspen:

The studies on soil N and the laboratory experiments on the nitrification potential of aspen forest soils suggest that timber cutting in this community may cause increased nutrient loading of streams. Studies on the deciduous forests of the Northeast hypothesize that the high nutrient loading following clearcutting was related to the high production of NO_3-N (Likens and Bormann 1970). Living, actively growing vegetation is thought to inhibit nitrifying organisms in some way (Rice and Pancholy 1972, 1973) and only when the dominance of the vegetation is removed can the nitrification process increase to a level which results in stream eutrophication. Our laboratory studies (see p. 14) were made on blocks of soil without growing vegetation and the high NO_3-N production in aspen soil supports this hypothesis. The low levels of NO_3-N in stream water draining the natural aspen forest further support the idea that nitrifying organisms are being inhibited, or that the NO_3-N which is produced is being immobilized by plants or microbes or converted to a gaseous form and lost to the atmosphere.

One test of the hypothesis is to initiate cutting practices to decrease the dominance of the vegetation and moniter subsequent stream chemistry and nutrient budget changes. Our objective in this study was not to cause increased nutrient loading but to identify whether various timber cutting practices could be performed without seriously modifying water quality. With this in mind the first cutting made was very modest; a 25% thinning in which trees were felled and left in place. The thinning was performed in late October, 1975, after normal leaf fall when presumably most growth processes had stopped. The thinning occurred on a 3.39 ha watershed (AW-2) adjacent to the undisturbed, gaged watershed (AW-1) which was used as a control. The two watersheds have statistically equivalent aspen communities (stem density, basal area, leaf production), are the same size (3.39 and 3.44 ha), and have similar slopes, elevations, and aspects. These conditions predict that the hydrology of the two watersheds will be the same. The thinned watershed (AW-2) does not have a weir so an actual measurement of stream discharge is not possible. A small metal flume was installed in 1974, however, which does allow the measurement of instantaneous discharge rates of surface runoff. These measurements on both watersheds were used to develop regression equations with the gage height measurements of the gaged watershed to allow an analysis of the influence of the thinning operation on discharge. Table 9 shows these equations for periods before and after the thinning operation. Although all of the regression equations were highly significant ($P < .001$) there was a significant difference ($P < .05$) between equations for AW-1 and AW-2 for the same period as well as a significant difference ($P < .01$) between equations for AW-2 during 1974 and 1975. The difference which occurs between the watersheds appears to be the result of two factors:

Table 9. Regression equations of instantaneous discharge (log of liters/minute) on AW-1 and AW-2 with gage height on AW-1.

<u>June - September 1974</u>

a* $\log$ AW-1 l/min = 7.49886 (gage ht.) - 7.40485
F value = 104***, d.f. 1, 22
$r^2 = 0.83$
std. error of regression coef. = 0.735

b $\log$ AW-2 l/min = 18.36496 (gage ht.) - 19.52428
F value = 37.99***, d.f. 1, 22
$r^2 = 0.63$
std. error of regression coef. = 2.980

<u>June - September 1975</u>

c $\log$ AW-2 l/min = 5.31093 (gage ht.) - 4.90591
F value = 284.5***, d.f. 1, 18
$r^2 = 0.94$
std. error of regression coef. = 0.315

<u>June - September 1976</u>

b $\log$ AW-2 l/min = 13.08237 (gage ht.) - 13.5628
F value = 74.7***, d.f. 1, 10
$r^2 = 0.88$
std. error of regression coef. = 1.514

* equations with the same letter are not significantly different ($P > .05$).

*** significant at .001 level.

1) AW-2 does not have a weir while AW-1 does, and

2) AW-2 has a larger stream bed area than AW-1.

Since AW-2 does not have a weir which intercepts ground water as well as surface water, it can be expected that discharge rates on the two watersheds would be different. During moist conditions the surface runoff is a larger proportion of the total water discharge than during dry conditions and this will cause different regression equations for the two watersheds. The larger stream bed area for AW-2 adds to this difference since during a precipitation event, rain which falls on the stream bed and stream bank area runs off immediately. Rain falling on the remainder of the watershed normally percolates into the soil causing a delay in runoff.

These factors also explain the different regression equations for AW-2 during 1974 and 1975. The summer of 1975 was much moister than that of 1974 resulting in a higher base flow, both total and surface runoff. In spite of the differences between these watersheds the highly significant regression equations indicate that a given set of moisture conditions will cause a predictable discharge pattern on each watershed.

The moisture conditions during the summer of 1976 proved to be the same as those of 1974. Gage height measurements on AW-1 were not significantly different ($P > .05$) during those two periods while gage height measurements during 1975 were significantly higher ($P < .01$) than both 1974 and 1976. The

The nonsignificant difference ($P > .05$) between the regression equations for AW-2 during 1974 and 1976 verify this point (table 9). Furthermore, this must be interpreted to mean that the thinning operation in 1975 did not significantly alter the hydrology of the watershed (e.g. by reduced evapotranspiration). This assumption allowed the use of the total discharge data of AW-1 to calculate nutrient budgets for AW-2 to ascertain whether the thinning operation affected the nutrient loading of the stream.

The figure of NO_3-N concentrations before and after the thinning operation on AW-2 shows no significant difference ($P > .05$, fig. 13). Likewise, the pattern of NO_3-N concentrations on AW-2 was very similar to that on AW-1 (fig. 14) which suggests that the thinning operation did not alter stream chemistry. Comparison of other cation concentrations before and after thinning on AW-2 and with those on AW-1 further support the contention that stream chemistry was not altered (figs. 15, 16, 17, 18).

Table 10 shows NH_4-N and NO_3-N budgets for AW-1 and AW-2 which again demonstrate that the nutrient loading of the stream was not changed appreciably by the thinning operation. The largest difference between the two years seems to be a reduction in the input of both NO_3-N and NH_4-N during 1975-76. This caused a smaller net accumulation for the year.

It is not known if the thinning activity increased nitrification in the forest floor. Since aspen is a clonal species, the

Figure 13. NO_3-N concentrations of the stream water from an aspen watershed (AW-2) before and after a 25% thinning operation.

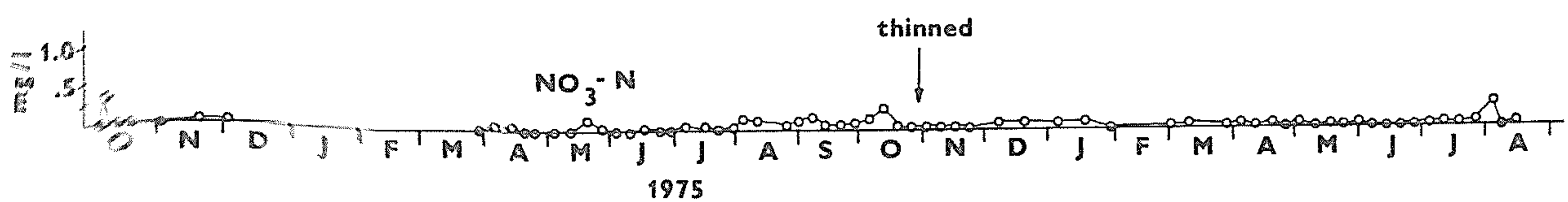


Figure 14. NO_3-N concentrations of the stream water from the control aspen watershed (AW-1).

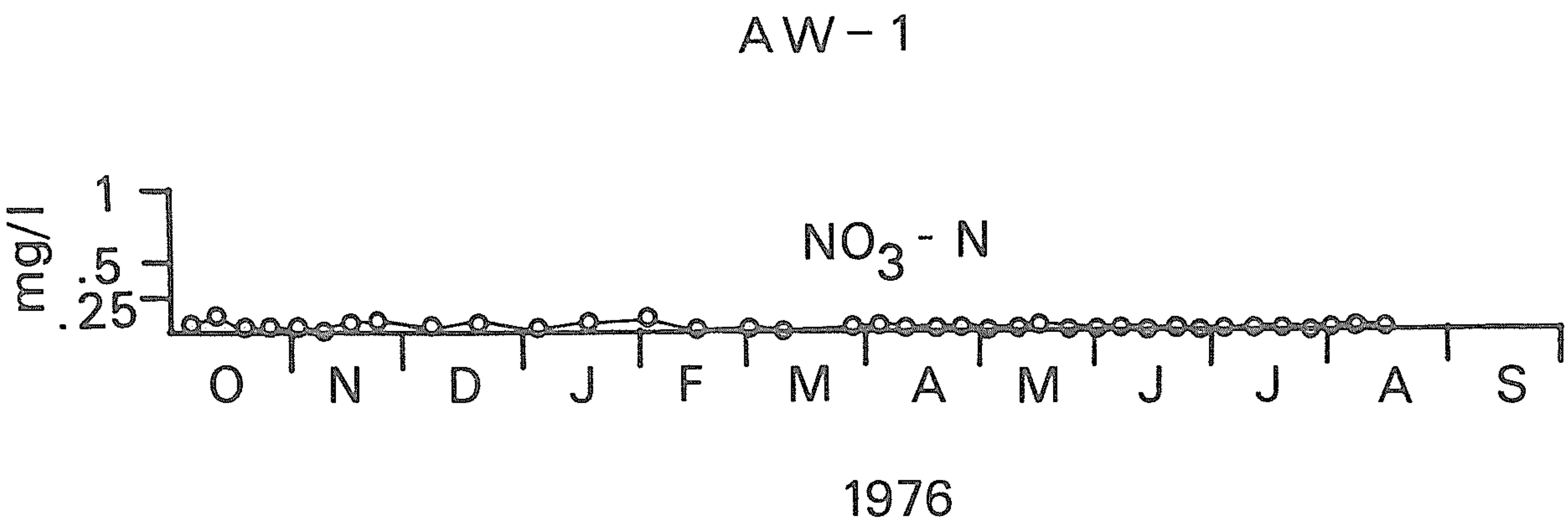


Figure 15. Calcium and Mg^{++} concentrations of the stream water from an aspen watershed (AW-2) before and after a 25% thinning operation.

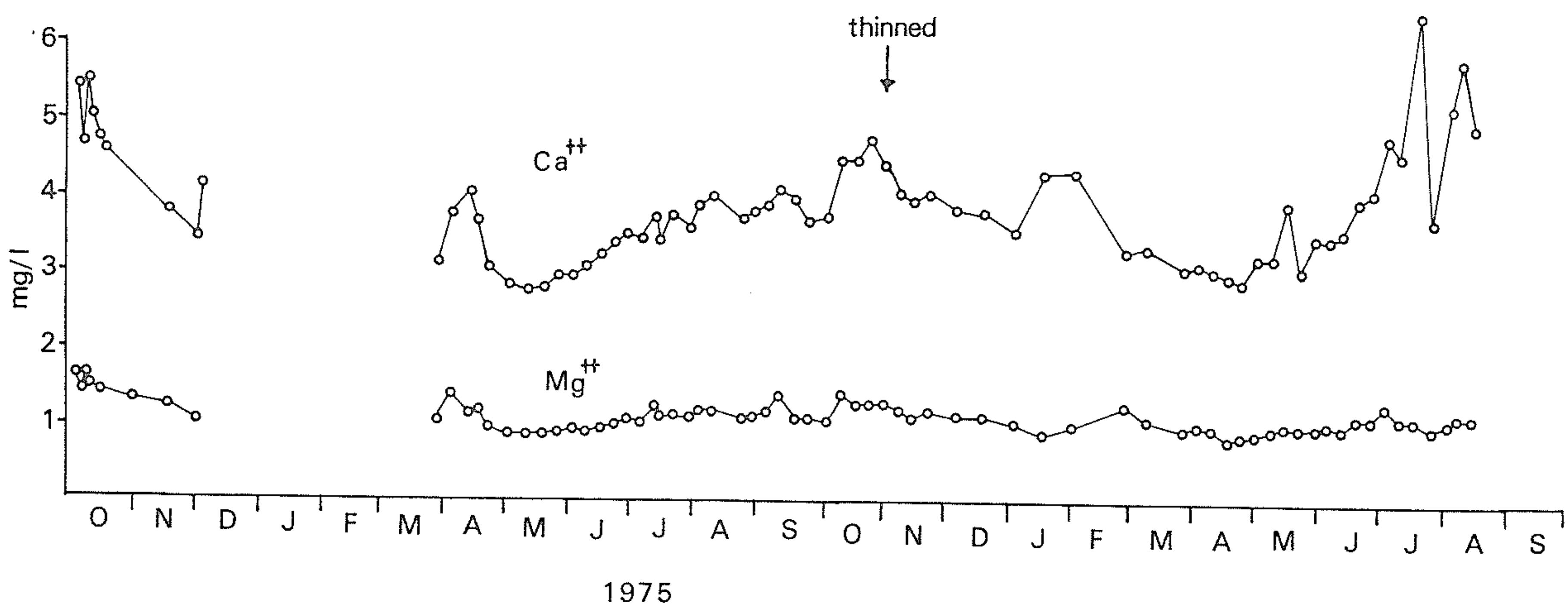


Figure 16. Calcium and Mg^{++} concentrations of the stream water from the control aspen watershed (AW-1).

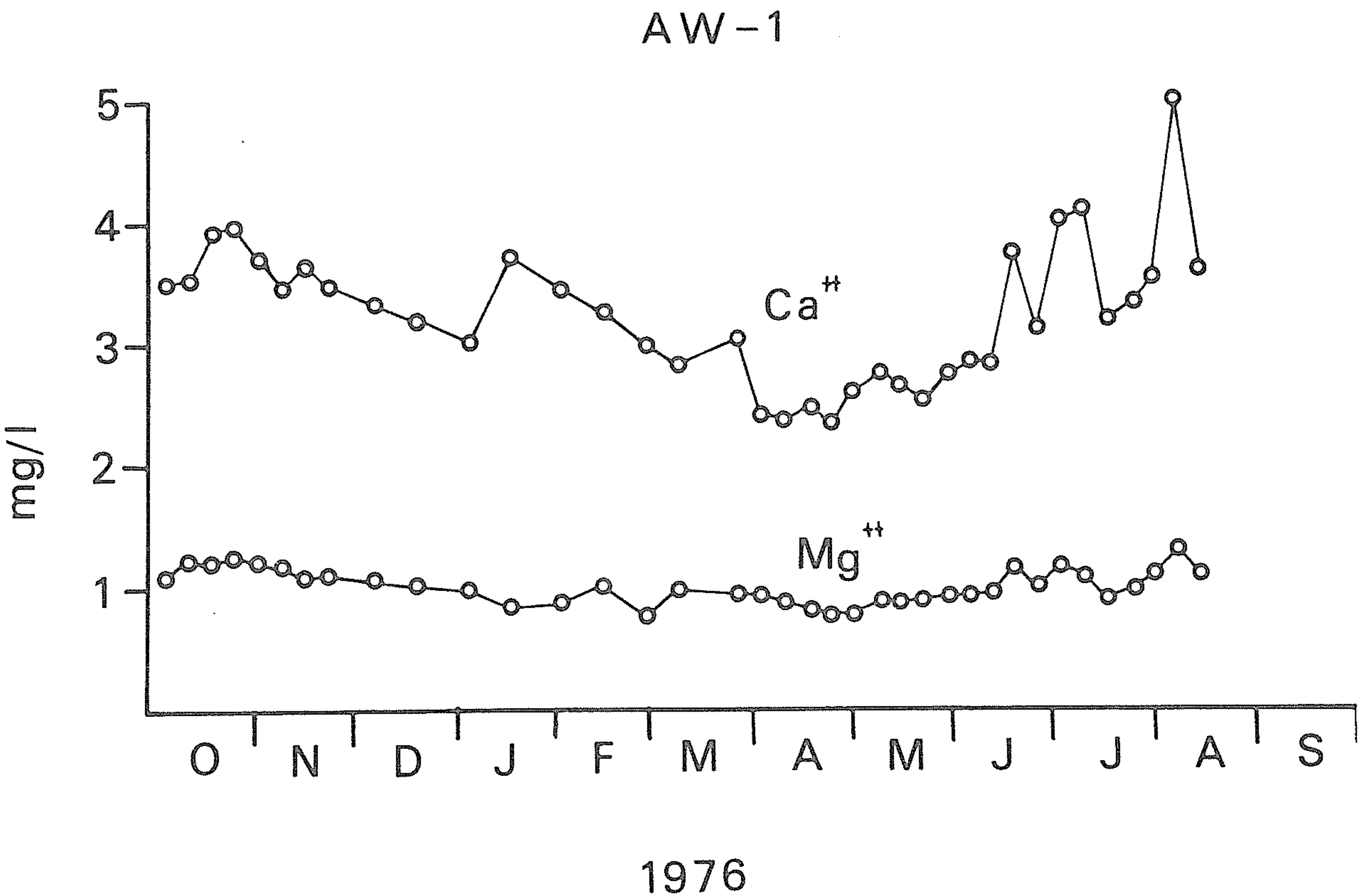


Figure 17. Sodium and K^+ concentrations of the stream water from an aspen watershed (AW-2) before and after a 25% thinning operation.

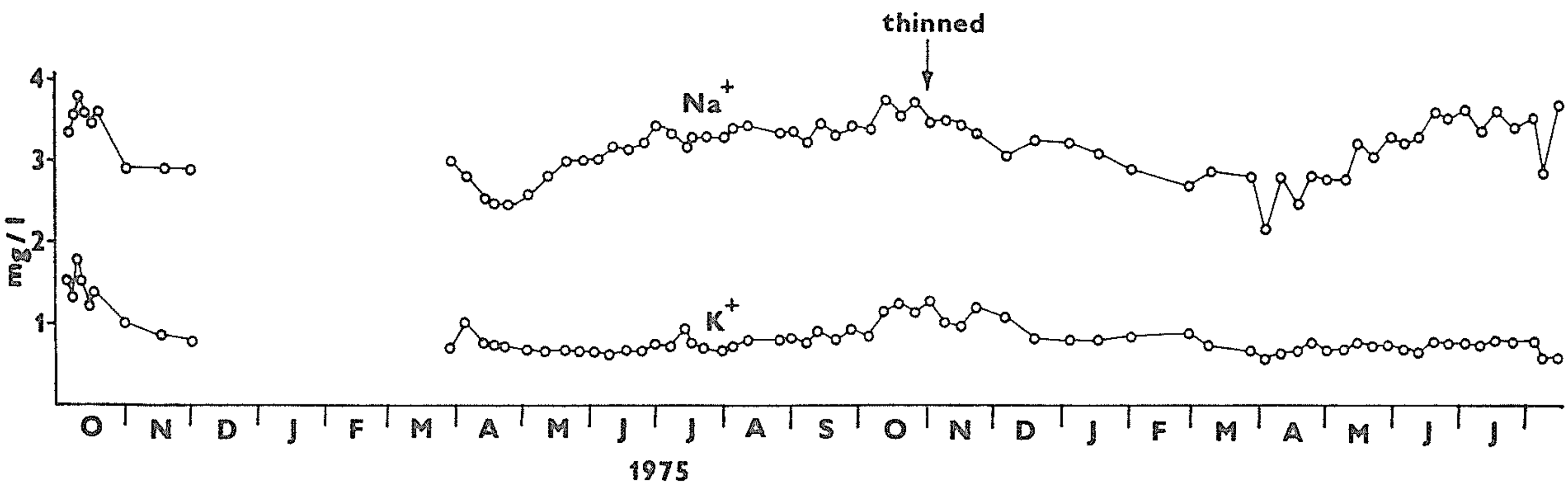


Figure 18. Sodium and K^+ concentrations of the stream water from the control aspen watershed (AW-1)

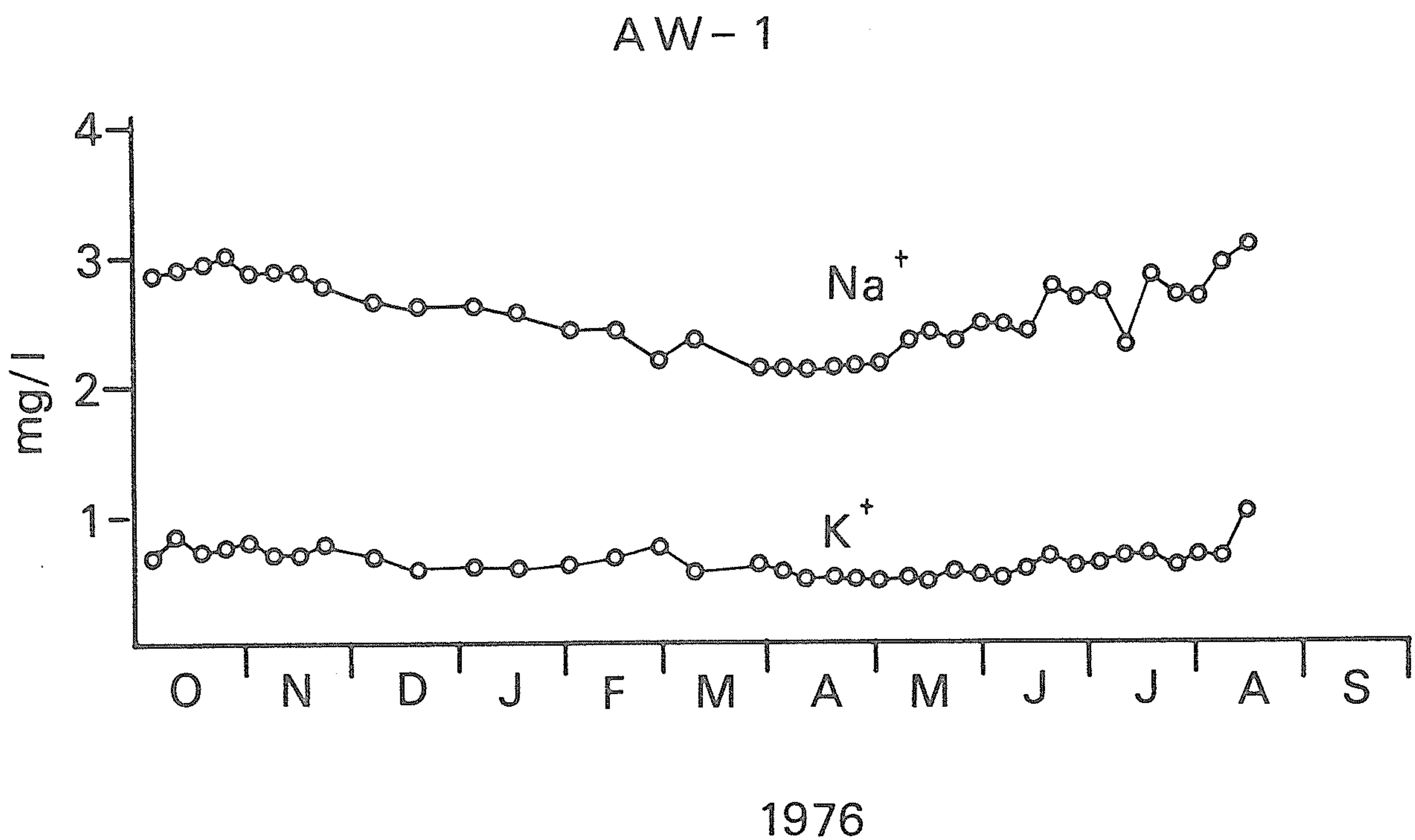


Table 10. NO_3-N and NH_4-N budgets for the thinned (AW-2) and control (AW-1) aspen watersheds. Thinning occurred in October, 1975. Data are in kg/ha

Watershed	AW-2		AW-1	
	NO_3-N	NH_4-N	NO_3-N	NH_4-N
1974-75				
input	2.8	1.7	2.8	1.7
output	0.12	0.38	0.13	0.28
net change	+2.7	+1.3	+2.7	+1.4
1975-76				
input	1.4	0.9	1.4	0.9
output	0.14	0.07	0.04	0.10
net change	+1.3	+0.8	+1.4	+0.8

thinning operation did not kill any root systems and any nitrifier inhibition by the root systems should not have been affected. Nor is it known whether plant uptake was affected. Since evapotranspiration was not measureably changed, the same uptake may have occurred but distributed among fewer individuals. Ongoing studies of productivity and nutrient uptake in the thinned watershed should answer the latter question. Since this thinning practice did not measureably affect N loss from the watershed, additional cutting experiments are being planned. In October of 1977 a 25% strip clear cutting operation will be performed. This practice will definitely affect root systems, sprout production and plant uptake of N.

The nitrogen budgets for a range of communities over an elevational gradient support the hypothesis that growing, undisturbed, terrestrial communities have the capacity to accumulate N and reduce N outputs to streams. The cationic nature of NH_4-N is an important property which explains an ecosystem's ability to retain this form of N. NO_3-N is normally considered easily leached and to avoid N loss, most hypotheses suggest that the nitrification process is inhibited. There is a great deal of information which suggests that this is true, especially in the acidic coniferous forests or acidic deciduous forests of the Northeast. One of the more interesting aspects of this study is the fact that a community (aspen) which has a relatively high nitrification potential, and in fact, demonstrates large variations in NO_3-N concentrations in the forest floor, is also good at minimizing N loss to streams. Such a community must have processes capable of changing NO_3-N to other forms to be able to minimize loss. In this paper I suggest that the dissimilatory denitrification pathway, electron sink pathway, and plant uptake are the most logical methods to reduce the loss of NO_3-N which is formed and accumulates in soil. It is important in future research that not only the effect of a disturbance on nitrification is considered but also the effect on processes which reduce the NO_3-N which is formed. The studies of ski area development and timber cutting in aspen did not demonstrate increased N losses. Either the nitrification process was not increased by these activities, or if it was, the NO_3-N transformation capacity was not exceeded. Additional research has now been initiated in the form of trenched plot experiments which will evaluate the role of living vegetation on the nitrification process as well as on plant uptake. Results from these studies will help to evaluate the role of other processes as well as to develop experiments to quantify that role.

LITERATURE CITED

Alexander, M. 1971. Microbial ecology. John Wiley and Sons, Inc. 511 pp.

American Public Health Assoc. 1965. Standard methods for the examination of water and wastewater including bottom sediments and sludges. Amer. Public Health Assoc., Inc. 769 p.

Bormann, F. H. and G. E. Likens. 1967. Nutrient cycling. Science 155(3761): 424-429.

Bormann, F. H., G. E. Likens, T. G. Siccama, R. S. Pierce, and J. S. Eaton. 1974. The export of nutrients and recovery of stable conditions following deforestation of Hubbard Brook. Ecol. Monogr. 44:255-277.

Brady, N. C. 1974. The nature and properties of soils. 8th ed., Macmillan Publ. Co., Inc., 639 pp.

Bremmer, J. M. 1965. Inorganic forms of nitrogen, p. 93-149. In: W. Bartholemew and F. Clark (eds.). Soil Nitrogen. Amer. Soc. Agron., Madison, Wisconsin.

Edmondson, W. T. 1970. Phosphorus, nitrogen, and algae in Lake Washington after diversion of sewage. Science 169 No. 3946.

Fredrikson, R. L. 1971. Comparative chemical water quality--natural and disturbed streams following logging and slash burning. In: A symposium - Forest Land Uses and Stream Environment. Oregon State University.

Brook Watershed Ecosystem. Ecol. Monogr. 40:23-47.

Likens, G. E., F. H. Bormann, N. M. Johnson, D. W. Fisher, and R. S. Pierce. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook Watershed-Ecosystem. Ecol. Monogr. 40:23-47.

Likens, G. E. and F. H. Bormann. 1972. Nutrient cycling in ecosystems. p. 25-67. In: Weins, J. (ed.), Ecosystem structure and function. Oregon State Univ. Press, Corvallis, Oregon.

Rice, E. L. and S. K. Pancholy. 1972. Inhibition of nitrification by climax ecosystems. Am. J. Botany. 59:1033-1040.

Rice, E. L. and S. K. Pancholy. 1973. Inhibition of nitrification by climax ecosystems. II. Additional evidence and possible role of tannins. Am. J. Botany. 60:691-702.

Ryther, J. H., and W. M. Dunstan. 1971. Nitrogen, phosphorus and euthrophication in the coastal marine environment. Science 171 No. 3975.

Stanford, G., J. O. Legg, Stanislaw Dzienia, and E. C. Simpson, Jr. 1975. Denitrification and associated nitrogen transformations in soils. Soil Sci. 120:147-152.

Taylor, A. W. 1971. Nutrients in streams draining woodland and farmland near Coshocton, Ohio Water Resources Res. 7:81-89.

Timmons, D. R., R. F. Holt, and J. J. Latterell. 1970. Leaching of crop residues as a source of nutrients in surface runoff water. Water Resources Res. 6:1367-1375.

Gessel, S. P. and D. W. Cole. 1965. Movement of elements through a forest soil as influenced by tree removal and fertilizer additions p. 95-105. Youngberg, C. T. (ed.) In: Forest Soil Relationships in North America, Second N. Amer. Forest Soils Conf.

Gosz, J. R., G. E. Likens, F. H. Bormann. 1973. Organic matter and nutrient dynamics of the forest floor in the Hubbard Brook Forest. Symposium on the Belowground Ecosystem. Sept. 5-7, Colo. State University, Fort Collins.

Gosz, J. R. 1975 . Nutrient budgets for undisturbed ecosystems along an elevational gradient in New Mexico. p. 780-799. In: Howell, F. G., J. B. Gentry, and M. H. Smith (eds.) Mineral cycling in Southeastern ecosystems. ERDA Symposium Series (CONF-740513), Technical Information Center, Office of Public Affairs, 898 pp.

Gosz, J. R. 1977a. The influence of reduced stream flows on water quality. Symposium on energy development on the Colorado River, Resources for the Future, (in press).

Gosz, J. R. 1977b. Effects of ski area development and use on stream water quality of the Santa Fe Basin, New Mexico. Submitted to Forest Science.

Holt, R. F., D. R. Timmons, and J. J. Latterell. 1970. Accumulation of phosphates in water. Agric. and Food Chem. 18:781.

Kurtz, L. T. 1970. The fate of applied nutrients in soils. Agric. and Food Chem. 18:773-780.

Likens, G. E. and F. H. Bormann. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard

Wang, Wun-Cheng and R. L. Evans. 1970. Nutrients and quality in impounded water. J. Am. Water Works Assoc. 62:510-541.

Vitousek, P. 1977. The regulation of element concentrations in mountain streams in the Northeastern United States. Ecol. Monogr. (in press).