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Changes in Nitrogen Cycling at an Old-Growth Douglas-fir Site After Disturbance

P. SOLLINS[2], K. CROMACK, JR.[2], F. M. MC CORISON[2], R. H. WARING[2], AND R. D. HARR[3]

ABSTRACT

Effects of disturbance on the N cycle in a 450-year-old Douglas-fir stand (*Pseudotsuga menziesii*) were studied in an experiment in which herbicides were used to kill all vegetation while minimally disturbing the litter layer and soil. Nitrogen concentration in falling foliage was greater on the treated area than on a control area, as was soil moisture. Nitrate and Kjeldahl N concentrations in the soil solution were greater on the treated than on the untreated area, but only at or below the bottom of the rooting zone (≥ 1-m depth). On the untreated area, nitrate was present in solution in significant amounts only at the 2-m depth.

Additional Index Words: **nitrate, nitrification, Kjeldahl N, soil water potential, C/N ratio, herbicides.**

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Export of dissolved N from terrestrial ecosystems is important because it can affect N availability, therefore productivity, and because export directly affects stream processes and water quality. Rates of N export can increase dramatically after disturbance. For example, Bormann et al. (1968) reported an increased nitrate loss from a forested watershed at Hubbard Brook after clearcutting and applying herbicide. Subsequent studies showed that nitrate loss increased after clearcutting much less in other parts of the United States than at Hubbard Brook (see Vitousek and Melillo, 1979), and that nitrate levels were higher at Hubbard Brook than at most other sites before disturbance as well as after. In western Oregon, Fredriksen et al. (1975)[4] found that nitrate levels were very low in stream water draining undisturbed Douglas-fir forests and that, though they rose substantially after cutting, they were still much lower than they had been at Hubbard Brook. An extensive study by Vitousek et al. (1979) identified three phenomena that can be important in preventing or delaying nitrate loss after disturbance: (i) accumulation of ammonium in soil solution and on cation exchange sites can be inhibited or delayed, (ii) conversion of ammonium to nitrate can be inhibited or delayed, and (iii) nitrate can accumulate in the soil but not leach to stream water or ground water.

At Watershed 10 (WS-10) at the H. J. Andrews (H.J.A.) Experimental Forest, a study of the internal element cycles of an old-growth forest (Sollins et al., 1980) showed that the first of the mechanisms of Vitousek et al. (1979) was probably responsible for the low levels of nitrate loss from the undisturbed system. Specifically, the C/N ratio of the forest floor and vegetation was very high (> 100), which suggested that ammonification rates were low and that lack of oxidizable N, relative to amounts of oxidizable C, prevented nitrifiers from competing effectively with the heterotrophic microflora.

In 1975, WS-10 was clearcut as part of the US/IBP Coniferous Forest Biome study of ecosystem structure and functioning. The substantial nitrate increase that occurred after clearcutting is the subject of another paper. The subject of this paper is an earlier experiment conducted in an 1,800-m² portion of the same watershed, which also resulted in increased nitrate production. Herbicide was sprayed on the understory vegetation and injected directly into the old-growth trees. The object was to kill the vegetation without disturbing the litter and soil. We expected a decrease in uptake of N and water as the vegetation died and an increase in mobilization of N from fallen foliage and dead roots. We, therefore, expected increases in soil moisture and N concentrations in the soil solution. In this paper, we present results of the experiment and discuss mechanisms that may explain the results.

SITE DESCRIPTION

The study was conducted on the 10.24-ha watershed designated WS-10 on the western boundary of the H.J.A. Experimental Forest in the west-central Cascade Mountains of Oregon. WS-10 has been the site of intensive research by the U.S. Forest Service and the Coniferous Forest Biome (US/IBP).

Elevations range from 430 m at the stream gauging station to about 670 m at the southeastern ridgeline. The average gradient along the main stream is about 25°C, but side slopes are much steeper, ranging from 25 to 50°C. Annual precipitation, most falling as rain between October and May, averaged 240 cm from May 1972 to May 1976 (Waring et al., 1978). Snow accumulations up to 30 cm are not uncommon in winter, but seldom persist more than 2 weeks. The mean air temperature was 7.9°C over the 1972–1976 period. Daily averages varied from −20 to +39°C.

[1] Contribution from Oregon State Univ. and the U.S. Forest Service, Pacific Northwest For. and Range Exp. Stn., Corvallis, Oreg. The mention of trade names in this paper does not constitute endorsement by Oregon State Univ. or the U.S. Forest Service. Received 17 March 1980.

[2] Research Associate, Assistant Professor, Research Assistant, and Professor, respectively, Dep. of Forest Science, School of Forestry, Oregon State Univ. F. M. McCorison is now at Tongas National Forest, Petersburg, Alaska.

[3] Research Hydrologist, Forestry Sciences Lab., USDA For. Serv., Pacific Northwest For. and Range Exp. Stn., Corvallis, OR 97331.

[4] R. L. Fredriksen. 1975. Nitrogen, phosphorus, and particulate matter budgets of five coniferous forest ecosystems in the western Cascade Range, Oregon. Ph.D. Thesis. Oregon State Univ., Corvallis. 127 p.

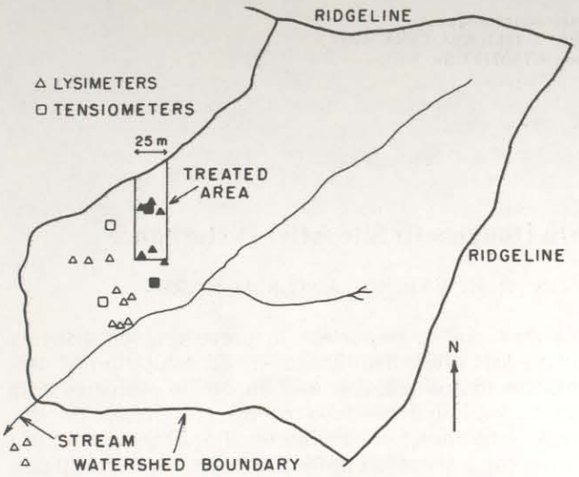


Fig. 1—Map of Watershed 10 showing location of lysimeters and tensiometers in the treated area (black symbols) and control area.

WS-10 is located in an area underlain by andesitic tuffs and breccias (James, 1977).[3] Soils, formed either in residual parent material or in colluvium originating from these deposits, are classified as Typic Dystrochrepts (Soil Survey Staff, 1960). They consist of a weakly developed A1 horizon 20 cm thick overlying a weakly developed B1-B2-B3 sequence 50–80 cm thick. Beneath the B horizon is partially weathered soil parent material ranging in thickness from 100 to 800 cm. Most roots occur in the upper 30 cm of the soil profile; few roots are found below 100 cm (Santantonio et al., 1977). Soil textures range from gravelly, silty clay loam to very gravelly clay loam.

Vegetation on WS-10 was dominated by a seral, 450-year-old, site class III (Dilworth, 1974) Douglas-fir forest. The understory was western hemlock (*Tsuga heterophylla*), Douglas-fir (*Pseudotsuga menziesii*), and other conifers and broad-leaved species (Dyrness et al., 1974; Zobel et al., 1976). Biomass and production data for WS-10 have been published by Grier and Logan (1977) and internal nutrient cycling budgets by Sollins et al. (1980).

At the time of the study, the forest floor on WS-10 (O1 + O2 horizons) averaged about 50 metric tons/ha dry weight, excluding logs (Grier and Logan, 1977), and was classified a duff-mull according to the system of Hoover and Lunt (1952). Fallen logs averaged 190 metric tons/ha dry wt (Grier and Logan, 1977) and occupied about 20% of the land surface (R. Fogel and M. Ogawa, unpublished data).

The treated area was the upper portion (approximately 1,800 m²) of the north slope of WS-10 (Fig. 1). The dense understory in that portion consisted primarily of the sclerophyllous evergreen hardwood species *Castanopsis chrysophylla* (Dougl.) A. DC., *Rhododendron macrophyllum* G. Don, and *Gaultheria shallon* Pursh.

METHODS

On 15 April 1974, concentrated monosodium methanearsonate at 0.72 kg/liter was injected directly into the conducting tissue of trees after thick bark was removed with an axe. Solution was injected with a "hypohatchet" at 10- to 20-cm intervals around the stem at a 1.3-m height. Understory shrubs were hand sprayed with the herbicide Roundup (N-phosphonomethylglycine at 0.36 kg active material/liter) diluted 1:100 in water containing 0.25% wetting agent.

Bourdon tensiometers, calibrated to be read directly in pressure units, were installed in June 1974 at 0.3- and 0.9-m depths in the soil or soil parent material (Harr, 1977). A pair was placed at each soil depth at each of four locations (Fig. 1). Because soil in the lower portion of the treated area was too rocky for installation, the four tensiometers for that portion were placed just downslope from the lower boundary.

Porous-cup tension lysimeters were used to sample soil solution at

[3] M. E. James. 1977. Rock weathering in the central western Cascades. M.S. Thesis. Univ. of Oregon, Eugene. 119 p.

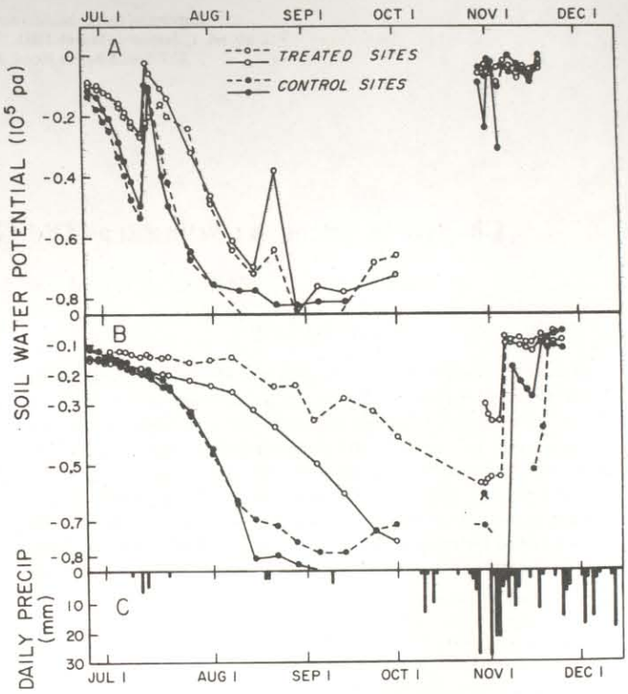


Fig. 2—Comparison of water potential at two sites in the control area and two sites in the treated area. Points are means of two values at each depth and location: (*A*) 0.3-m depth, (*B*) 0.9-m depth, and (*C*) daily precipitation.

two locations in the treated area and three locations in the adjacent untreated area. An additional set was established just outside the watershed boundary (Fig. 1) to serve as a control for gauging effects of later clearcutting and as a control for this study. Three depths, 0.3 m, 1.0 m, and 2.0 m, were chosen to correspond with the bottom of the A horizon, the bottom of the rooting zone, and a point well below the rooting zone. At each location, nine cups (three replicates at each of three depths) were installed by auguring holes, sealing the bottom with bentonite slurry, backfilling with 200-mesh quartz sand, and sealing the top with bentonite to prevent water from percolating down the installation shaft (Parizek and Lane, 1970). Cups were emptied and evacuated to their bubble pressure (−80 kPa) three times weekly. Vacuum decreased to at most −50 kPa between evacuations, depending on the sample volume that had accumulated.

Cups installed in the control area during winter 1974 began functioning dependably in March. Cups were installed at the treated area in early summer 1974, but soils were by then too dry to permit sample extraction. October rainfall (Fig. 2) was insufficient to wet the soil, and the first soil solution samples were obtained in early November. Sampling ended in May 1975 when logging began, except at the control site outside the watershed. Water samples from each cup were accumulated over 3-week intervals in a collection vessel on site. Because the number of samples was still large (54 cup samples and about 150 samples of throughfall, litter leachate, and stream water gathered routinely every 3 weeks as part of other studies at WS-10), they had to be stored frozen until analyzed. Storage time varied from 2 to 8 weeks, but was identical for all samples from a given date. After filtration, samples were analyzed for nitrate (plus nitrite) colorimetrically by diazotization after Cd reduction (Am. Public Health Assoc., 1971) and for Kjeldahl N (organic plus NH_4^+) (Jackson, 1958).

Subsequent tests with frozen soil-solution and stream water samples showed that nitrate levels decreased slowly during storage. Samples that originally contained 40–350 ppb nitrate lost from 3 to 5 ppb after 4 weeks and from 7 to 15 ppb after 12 weeks. Samples that originally contained from 5 to 12 ppb nitrate lost 4 ppb after 4 weeks and from 4 to 9 ppb after 12 weeks. Thus, all our nitrate values are probably underestimated by 3–15 ppb, but because samples from a given date

Table 1—Composition of foliage from herbicide-treated and control *P. menziesii*, *C. chrysophylla*, and *R. macrophyllum*.

Foliage type	N	Lignin†	Cellulose	C/N‡	Sclerophyll index§
	% dry wt				
P. menziesii					
Untreated					
Green	0.95	11.64	16.56	53	474
Senescent	0.48	18.37	14.90	104	1102
Herbicide treated	0.88	11.32	17.06	57	513
C. chrysophylla					
Untreated					
Green	0.87	14.99	20.28	57	648
Senescent	0.31	17.46	19.73	161	1917
Herbicide treated	0.97	14.85	19.06	52	559
R. macrophyllum					
Untreated					
Green	0.91	9.46	19.18	55	503
Senescent	0.26	10.92	20.09	192	1914
Herbicide treated	0.92	10.04	17.24	54	477

† Acid insoluble only, may slightly underestimate total.
‡ Approximate based on C = 50% (Bollen, 1969).
§ Ratio of lignin + cellulose to protein (calculated as N × 6.25). Modified from Loveless (1961).

were treated identically, our conclusions regarding differences between treatment and control and differences among the three depths should not be affected.

Foliage samples were collected twice in 1974. In July, green foliage was collected from untreated *Pseudotsuga menziesii* (Mirb.) Franco, *Castanopsis*, and *Rhododendron*. Attached dead foliage was collected from herbicide-treated plants in late August, at which time normally senescent foliage was also collected from untreated plants in the adjacent area. Composite samples representing at least five individuals of each species were analyzed for N by micro-Kjeldahl (Jackson, 1958) and for lignin and cellulose (Van Soest, 1963).

RESULTS

Foliage and Soil Moisture

Overstory foliage began turning yellow about 3 months after the herbicide was injected. The one-sided leaf-area index declined from 9.6 (Gholz et al., 1976; Grier and Logan, 1977) to visually estimated values of about 5.0 by mid-July and 2.0 by mid-November 1974. By May 1975 when logging began on the watershed, the remaining foliage on the herbicide-treated trees was brown. We do not know whether living tissue still existed in the trees at the time they were felled.

The herbicide treatment affected soil water potential (Fig. 2). At the 0.3-m depth, soil in the treated area remained more moist during the summer than did soil in the adjacent untreated area (Fig. 2A). Effects at the 0.9-m depth were even more pronounced (Fig. 2B).

Herbicide treatment also substantially changed the quality of litterfall, causing a large decrease in the C/N ratio. Nitrogen concentrations in dead foliage on herbicide-treated trees were 2–3 times greater than those in senescent foliage from untreated trees (Table 1), which shows that N was not withdrawn from the foliage before death. Sclerophyll index, inversely related to decomposition rate (Cromack and Monk, 1975), was substantially lower in the foliage from treated trees (Table 1).

Solution Chemistry

Kjeldahl N concentrations in soil solutions averaged from 2 to 100 times the nitrate concentrations (Table 2). Averaged over all depths, Kjeldahl N was significantly greater ($p < 0.01$) in soil solutions from the defoliated area (190 ppb) than in solutions from the untreated area (150 ppb). Differences were statistically significant at the 2.0-m depth ($p < 0.01$), but not at the 0.3- or 1.0-m depth.

Nitrate concentrations were affected strongly by the herbicide treatment (Table 2). Nitrate-N on the treated area averaged 66.1 ppb, about 12 times the value for the control area ($p < 0.01$). (One presumably spurious value of 1,150 ppb was dropped from the control data.) Concentrations were significantly greater on the treated area than on the control area at the 1.0- and 2.0-m depth ($p < 0.01$). The mean concentration was slightly lower at the 0.3-m depth, but the difference was not significant.

Nitrate concentrations increased with soil depth in the treated area, averaging < 1 ppb, 11.9 ppb, and 170.5 at the 0.3-, 1.0-, and 2.0-m depths. All differences were significant at $p < 0.01$ except between the 0.3- and 1.0-m averages ($p < 0.05$).

Nitrate concentrations also increased with depth in the control area, though not as dramatically as in the treated area (Table 2). Averages at 0.3 m and 1.0 m were not significantly different. However, the average at 2.0 m was significantly greater ($p < 0.05$) than averages at both 1.0 m and 0.3 m.

Seasonal changes in concentrations were evident from the lysimetry data. At the 2.0-m depth, Kjeldahl N concentrations on the untreated part of the watershed were highest in 1974–1975 in early fall (corresponding to the onset of the rainy season) and decreased to a steady, lower level by the end of January (Fig. 3). The same pattern was observed also at the 0.3- and 1.0-m depths and again in 1975–1976 in samples from all depths of the control area just off of WS-10. Nitrate concentrations in samples from the treated area increased steadily until logging terminated the sampling, whereas those in

Table 2—Kjeldahl and nitrate N concentrations in soil solution taken at various depths from herbicide-treated and control areas. Mean of samples accumulated over 3-week intervals during November 1974–May 1975.

Substance	0.3-m depth		1.0-m depth		2.0-m depth		All depths	
	Control	Treated	Control	Treated	Control	Treated	Control	Treated
	ppb							
Kjeldahl N	220	190	130	220	100	180†	150	190†
NO_3^--N‡	2.1	<1.0	1.0	11.9§	11.9	170.5†	5.3	66.1§

† Significantly different from control ($p < 0.01$) based on paired two-tailed t-test.
‡ May include small amounts of nitrite.
§ Significantly different from control ($p < 0.01$) based on rank-sum test.

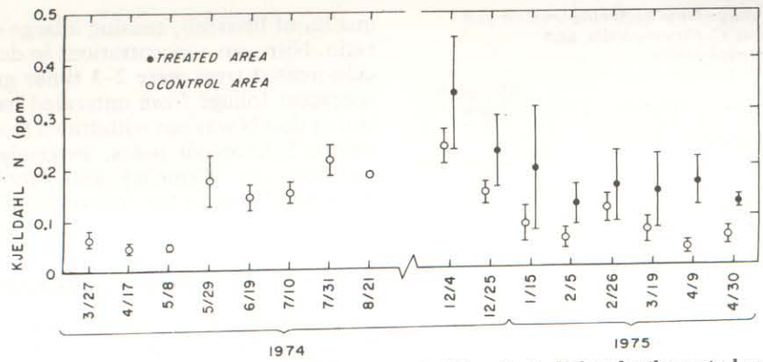


Fig. 3—Comparison of dissolved Kjeldahl N in control and treated areas at the 2.0-m depth. Values for the control area are means of 4–12 lysimeter cups. Values for the treated area are means of 2–6 cups. The vertical bars are standard error estimates.

the control samples showed no obvious pattern with time. Because of this temporal variation, as well as effects of variable storage time, all averages over time have large standard errors.

DISCUSSION

We expected dissolved N concentrations to be higher on the herbicide-treated plots because of reduced uptake by the vegetation and because of increased N mobilization during decay of foliage, fine roots, and mycorrhizae. However, we would have expected the difference to be greatest in the zone most intensively rooted, whereas in fact differences were greatest at or below the 1.0-m depth. At the 0.3-m depth neither Kjeldahl N nor nitrate averages differed significantly between control and treated areas. A substantial influx of Kjeldahl N into solution seemed to occur on the treated area between the 0.3- and 1.0-m depths. Kjeldahl N decreased between the 0.3- and 1.0-m depths by 90 ppb on the control area, but increased by 30 ppb on the treated area (difference significant at $p < 0.05$).

At three mature Douglas-fir stands on or near the H.J.A. Experimental Forest, 75% of the fine roots (<2-mm diam) occurred in the top 0.3 m of the mineral soil (Santantonio, 1979). Consequently, it is unlikely that much of the apparent influx of N on the treated area between 0.3 and 1.0 m was a result of root decomposition or decreased uptake. Temperature differences between treated and control areas are not implicated because any effect of temperature would presumably be more pronounced near the soil surface. At this point, the source of the increased N at the 1.0-m depth is still unclear. A more careful experiment, designed specifically to look at concentration patterns with depth, is currently being conducted.

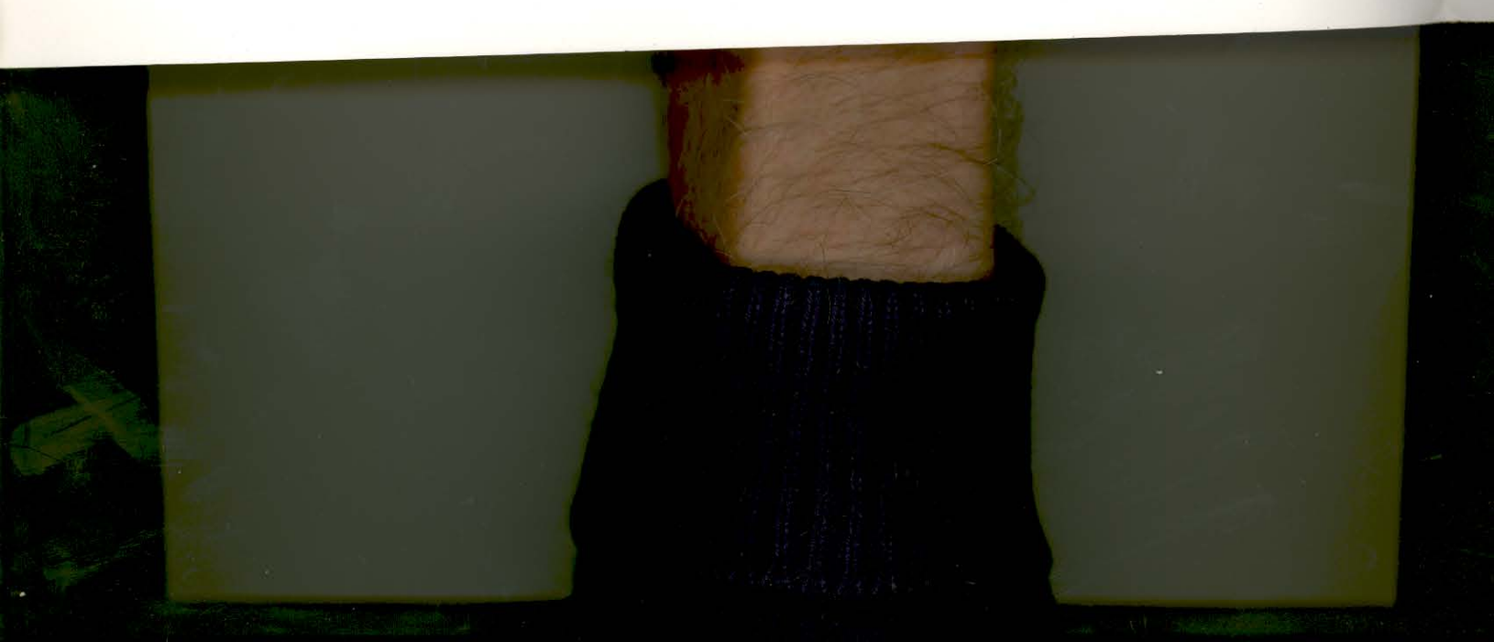
The greater amounts of nitrate on the treated plot may be due to greater amounts of oxidizable (Kjeldahl) N present there. This does not, however, explain why nitrate levels increased with depth on the control plot, while Kjeldahl N levels decreased.

The high nitrate levels at and below 1.0 m could be the result of a decrease in amounts of readily oxidizable C compounds below the rooting zone. Such a decrease is important because bacteria that oxidize N (nitrifiers) grow slowly and can compete effectively for reduced N with bacteria that oxidize C compounds (heterotrophs) only when organic C is present in limiting amounts (Jansson, 1958). The phenomenon is illustrated well in a recent experiment in which nitrate concentrations in the soil solution of a stand of girdled *Liriodendron tulipifera* decreased markedly after application of sucrose to the forest floor (Edwards and Ross-Todd, 1979; Johnson and Edwards, 1979).

Decreased availability of readily oxidizable C compounds below the rooting zone is certainly to be expected. Photosynthesis is the only significant source of reduced C in most ecosystems. Massive amounts are introduced each year into the upper soil horizons by root and mycorrhizal turnover (Harris et al., 1977; Fogel and Hunt, 1979; Santantonio, 1979) and by root exudates (Smith, 1976). However, most of this C is oxidized to CO_2 in the rooting zone, or adsorbed on mineral surfaces (Greenland, 1971; Sollins et al., 1980); comparatively small amounts of oxidizable C penetrate downward. More data on concentrations of oxidizable C (both particulate and dissolved) are needed before this explanation can be pursued; and it is essential that concentrations be measured at a series of depths. Had we sampled only within the rooting zone, we would not have detected any difference in nitrate levels.

The general pattern of response to disturbance at WS-10 fits well with that described and explained by Vitousek et al. (1979) and Vitousek and Melillo (1979). However, their studies were based for the most part on measurements made at a single depth. When soil depth is added as a dimension, new patterns appear and await explanation. Kimmins and Feller (1976) measured N-nitrate concentrations at a series of depths before and after clearcutting and slashburning in a young coastal hemlock forest in British Columbia. They, like us, found that nitrate concentrations increased more in the ground water than in the litter leachate or the rooting-zone solution.

Herbicide treatment prevented redistribution of N before abcission, thus decreasing the C/N ratio of leaf fall. This effect could have been predicted by anyone familiar with the processes of foliage kill by herbicides and of nitrogen redistribution. However, the phenomenon is often ignored in discussions of the role of herbicides in forestry. It is not mentioned in an as-



sessment of the impact of banning 2,4,5-T on forestry in the United States (USDA, 1979), nor does Kimmins (1975) discuss it among ecosystem-level effects of herbicides. Newton (M. Newton, Oregon State Univ., personal communication) and Norris (L. Norris, U.S. Forest Service, personal communication) also indicate that the effect is not widely appreciated. That it can be significant is shown by Gottschalk and Shure (1979) in a recent study of herbicide effects on leaf litter decomposition in an oak-hickory forest; there, treatment substantially increased the decomposition rate of foliage litter.

Kjeldahl N rather than nitrate accounted for most N in the soil solution. Studies of stream water exiting WS-10 and many other coniferous watersheds (Fredriksen, 1975; Gosz, 1978; Tiedemann et al., 1978; Lewis and Grant, 1979) have produced the same conclusion and have also shown that most of the Kjeldahl N is organic N, not ammonium. Unfortunately, studies have been conducted in coniferous forests in which only nitrate (and sometimes ammonium) levels were measured, but in which total reduced N was not assayed (e.g., Coats et al., 1976; Feller and Kimmins, 1979). To the extent that these studies attempt to draw conclusions about fluxes of total N, they could be seriously in error.

Leonard et al. (1979) measured organic N in stream water, but did not report their results because they felt that analytical methods used by them and others were not entirely adequate. We expect this means that we and others have, if anything, underestimated the importance of dissolved organic N. Assessment of the impact of forest management practices or other disturbances on N budgets and water quality must include measurement of total N transfers, not just transfers of nitrate N.

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