Ecology (1977) **58**: pp. 16–30

FIRE AND NUTRIENT CYCLING IN A DOUGLAS-FIR/LARCH FOREST[1]

NELLIE M. STARK
School of Forestry, University of Montana, Missoula, Montana 59812 USA

Abstract. Twenty control burns performed with a wide range of fuel loadings and moisture conditions were used to study the effectiveness of old fuel reduction under standing Douglas-fir/larch forest. This paper reports the influence of burning on nutrient retention and loss from the soil. Sixty % of the fires were successful in reducing residual fuels with no accelerated loss of nutrients below the root zone. Net losses of Ca^{+2} and Mg^{+2} occurred below the root zone when soil surface temperatures exceeded 300°C, but were insignificant when soil surface temperatures remained below 200–300°C. No other elements were lost (net) from the soil as a result of burning. Precipitation on control soils delivers as much Ca^{+2} as is normally lost below the root zone in the absence of fire. Iron concentration in the soil water is a good indicator of the intensity of burn. The hotter the fire, the less iron in the soil water as a result of the alkaline pH. Ash shows a definite pattern of nutrient release under the influence of precipitation. Homogeneous subsamples of litter showed predictable nutrient losses when ignited at different temperatures. Overland flow and surface erosion are of little significance on this soil type. Decomposition of Douglas-fir litter was only slightly more rapid on hot burned substrates than on control (unburned) substrates. When the biological life concept was applied to this soil, it showed that this soil is young and capable of withstanding many years of cyclic intensive burns.

Key words: Ash; biological life concept; decomposition; Douglas-fir/larch; fire; nutrients; Montana.

INTRODUCTION

Seventy-five yr of fire protection in western Montana have resulted in sizeable build ups of old fuels on the forest floor. Data are not available on the rate of decay of stumps and trunk wood in this environment, but the persistence of heavy logging and thinning slash suggest that woody decay is slow. Fuel accumulations are producing a serious wild fire hazard in some forests (Brackebusch 1970).

Controlled burning is a feasible means of reducing the heavy loading of residual fuels. Much of the residual material is not sound or usable by industry. Burning of unmarketable forest products has come under criticism because of the air pollution problems. Conservationists are concerned about possible aesthetic damage caused by burning, about nutrient losses into streams and possible eutrophication. Foresters have expressed concern about insect infestations resulting from burning under standing timber. The threat of escape fires is always a problem.

The ecological effects of fire are of great concern (Ahlgren and Ahlgren 1960, Behan 1970, Habeck 1970, Janda 1971, Johnson et al. 1969, and Hembree and Raintree 1971). A series of studies by Cleaves et al. (1970) attempted to quantify weathering. A number of recent papers describe the effects of burning on water quality (Fredriksen 1971, Grier and Cole 1971). Others have looked at soil microbiology and fire (Jorgensen and Hodges 1970, Neal et al. 1965, Wright and Tarrant 1957). Knight (1966) reported sizeable losses of nitrogen from the forest floor as a result of burning. Lewis (1974) has shown the effects of fire on nutrient movement in a South Carolina pine forest, and Ralston and Hatchell (1971) reported on the physical alterations in North Carolina soils following prescribed burning. Smith (1970) reported significant differences in the concentrations of soil nutrients before and after burning. Other studies in the northern Rocky Mountains by Striffler and Mogren (1971) reported on erosion following a severe burn.

None of these studies has produced a means of evaluating the seriousness of soil nutrient losses associated with burning. In watershed studies, the amount of eutrophication is the recognized measure of the damage from burning or other land treatment. To date, no method has been devised to quantify the seriousness of elemental losses from the soil itself.

The study reported here attempts to quantify nutrient losses below the root zone in terms of the impact of burns of different intensities on the "biological life" of the soil. The biological life is a concept which evolved from research in the Temperate Zone and in the tropics on depleted white sands. The theory combines accepted soil genesis and succession theories, and adds a series of decline stages describing the ultimate wearing out of highly weathered and leached soils on a peneplain.

Little attention has been paid to the loss of nu-

---

[1] Manuscript received 10 March 1976; accepted 11 July 1976.

trients and damage to the fertility of the soil resulting from controlled burns. The watershed approach has been most frequently used in studying fire and nutrient cycling because it shows immediate impacts on water quality and is a means of synthesizing the influence of the treatment over an entire system (Johnson et al. 1968). It is more difficult to measure the impact of any treatment on the soil itself, but Grier and Cole (1971) have studied the influence of slash burning on ion transport in Washington soils. The lysimeter approach is clearly better suited to soils lying over permeable bedrocks and for studies on the effect of treatment on the immediate soil fertility, while the watershed approach must be restricted to areas with impermeable bedrock and abundant streams.

Since the Montana study area has neither impermeable bedrock nor frequent streams, the only feasible approach was to study by lysimetry the changes in water quality below the root zone.

The fuels and fire portion of this study were conducted by Rodney A. Norum of the Northern Forest Fire Laboratory and the smoke volume measurements were made by Robert W. Steele of the School of Forestry of the University of Montana (Norum 1976, Steele 1976).

The cooperative study conducted 20 control burns under standing timber in a north slope, larch/Douglas-fir (*Larix occidentalis* Nutt., *Pseudotsuga menziesii* [Mirb.] Franco) forest with a wide range of fuel loadings, fuel moisture, and meteorological conditions in order to define those conditions which would most nearly meet the following objectives:

1) safely reduce residual fuels with little or no damage to standing timber (or possible light thinning),
2) provide limited seedbed preparation and stimulation of browse species,
3) produce minimal air pollution and smoke problems,
4) provide added available nutrients to stimulate growth at some point in the ecosystem (ideally for trees), and
5) not result in massive nutrient losses below the root zone.

Only the nutrient cycling portion is reported here.

METHODS

The study area

The studies were conducted at 1464 m elevation on a north slope in the larch/Douglas-fir type on the Lubrecht Experimental Forest 52 km east of Missoula, Montana. The quartzitic argillites are derived from the Bonner Formation and are sufficiently abundant to constitute a talus slope with a 1 m mantle of soil. The soils are sandy loams of the Holloway Series. The climate has warm (to 36°C), dry summers and cold (to −35°C), wet winters with 1 to 3 m of snow. Annual precipitation averages 466 mm (Steele 1975).

The forest is dominated by western larch and Douglas-fir $\approx$ 70 yr old with frequent ponderosa pine (*Pinus ponderosa* Laws.) and occasional lodgepole pines (*Pinus contorta* var. *murrayana* [Grev. & Balf.] Engelm.). The shrub layer has fire-adapted species such as bear-berry (*Arctostaphylos uva-ursi* [L.] Spreng.), mock azalea (*Menziesia ferruginea* Smith), several species of huckleberry (*Vaccinium* spp. L.) and serviceberry (*Amelanchier alnifolia* Nutt.). The most common herbs are beargrass (*Xerophyllum tenax* [Pursh] Nutt.), and a variety of grasses.

Field methods

The plots.—Areas of $\approx$ one-third ha were ringed with fire lines and a twenty-five point sampling grid was set with 5-m intervals between the points and within the one-third hectare. Buffer strips of at least 5 m were left on all sides of the grid. This grid was used for sampling fuels using Brown's (1974) fuel inventory system, fuel moisture (gravimetric), burn temperatures using thermal points, soil and soil water quality, vegetation, insect damage, and many other parameters. In the study as a whole, $>$ 100 parameters were measured on each of 20 burn plots and 19 control plots.

The fires.—The fires were lit under standing timber on each plot between 1300 and 1600 h using drip torches with a mixture of diesel fuel and gasoline. The lighting began at the top of the slope and progressed downslope in bands 5 m wide. Each strip was allowed to heat out before the next was ignited. Radiosonde and other on-site meteorological data were collected during and after the burns (Norum 1976). The fires were allowed to burn themselves out within the fire lines.

Field sampling.—Samples of all green fuels (shrubs, herbs) were collected from 40 plots, each 1-m^2 for total elemental analysis, and 40 litter samples were collected from 0.1 m^2 plots. These were analyzed with wood and bark over a size range to show the elemental content of the fuels.

Soils profile descriptions, and gravimetric soil moisture were taken in the field. The depth of the root zone, percentage roots by weight, and the percent slope were measured for the study areas. Air temperature, wind, and precipitation quantity (one site) and quality (four sites) were measured before, during, and after the burns.

After the burns, 72 soil water tubes were carefully installed to avoid ash contamination above the tree root zone (0–10 cm), within the tree root zone (20–25 cm), and below the tree root zone (50–55 cm) in 19 of the burned plots and in 5 control plots. These were pumped to obtain soil water for water quality studies whenever there was mobile water in the soil. The quantity of water which was lost below the root zone each year was measured using a simple 0-tension lysimeter modified from the plan by Kline and Jordan (1970). Lysimeter data were converted to gravitational water loss at 50 cm depth for a 1 m^2 surface area, and actual nutrient losses for the same surface area were calculated from 50–55 cm soil water data corrected for control nutrient loss and precipitation input. Since free water can be collected from this soil only one time per year, lysimeter samples were inadequate to show what changes were occurring in soil water chemistry. Soil water extracted in the field showed the same trends in nutrient content as did 0.002N H_2SO_4 soil extracts.

The quantity and quality of water moving overland was measured using land wiers on eight burn and two control sites with 300 m^2 microwatersheds. The land wiers were made of 50 cm wide aluminum sheets laid in a V-pattern on the slopes. The upper edge of the V was angled and anchored to penetrate the litter by one centimeter. Each V was 8 m across and fed into a funnel and a reservoir at the point of the V. Wooden guards at the lower margin of the V retained solids and channeled water into the funnels. Each land wier had a roof to keep direct precipitation from reaching the aluminum pad.

Decomposition on burned and unburned substrates was studied for 2 yr using fine-mesh Saran® bags and 5.0000 g samples of Douglas-fir needles. The samples were left in the field during the cold, wet season (November to May) and during the drier, warmer season (May to November). The needles were dried at 100°C for 20 h and weighed from a desiccator.

Laboratory methods

Soil bulk density and particle density were measured in the usual way. Total elemental content of soil and rock used HF-$HCLO_4$ digestion (Hesse 1971).

All plant samples were dried at 65°C for 48 h, ground to pass a 1 mm sieve, and 1.0000 g subsamples were dry ashed at 525°C ± 5°C for 2 h. The ash was taken up in heated 6N H_2SO_4, brought to 50 ml volume, and analyzed for Ca^{+2}, Cu^{+2}, Fe^{+3}, K^+, Mg^{+2}, Mn^{+3}, Na^+ and Zn^{+2} using an atomic absorption spectrophotometer. One % lanthanum was added to the samples for analysis of Ca^{+2} and Mg^{+2}. Phosphate was measured on 5-ml aliquots of the solution using the phospho-molybdophosphoric acid technique (Farber 1960). Total nitrogen was determined by a modified micro-Kjeldehl method using 1.0 g samples and salicyclic acid.

All water samples were either analyzed fresh or frozen for not over 1 wk. The pH, fluoride and nitrate levels were determined using Orion® specific ion meters. The elements Ca^{+2}, Cu^{+2}, Fe^{+3}, K^+, Mg^{+2}, Mn^{+3}, Na^+ and Zn^{+2} were measured on the atomic absorption spectrophotometer. Phosphate was determined by the most sensitive method described by Farber (1960).

To study the rate of release of biologically important elements from ash, 5.0000 g ash samples were carefully placed on filter paper in funnels. Each ash sample (freshly collected within 24 h of each burn) was leached with a series of 100 ml aliquots of either precipitation (melted snow) or distilled H_2O. Ash samples were leached 10 times with 100 ml each time. The water from the ash was analyzed in the same manner as other water samples. Some ash samples were allowed to freeze or partially dry between leachings to study the influence of these two forces on nutrient release.

To determine how much of each element was potentially available in the ash, 1.0000 g samples were taken up in 6N HCL, heated, brought to 50 ml volume at room temperature, and analyzed in the same manner as plant samples. The elements lost during leaching were studied in light of the total of each element present in the ash.

The effects of burning temperature on nutrient losses from plant material has not received much attention. To determine how surface burn temperatures might influence nutrient loss from litter, homogeneous 1.0000 g dried (100°C) samples of <0.1 mm ground litter were placed in crucibles which had previously been heated in a muffle furnace to 1000°C for 2 h to oxidize any solid residues on the containers. The crucibles were cooled in desiccators before and after ashing. Five samples of highly uniform finely ground litter were ashed at each of the following temperatures: 100°, 200°, 300°, 400°, 500°, 600°, 700°, 800°, 900° and 1000°C with these temperatures maintained for 2 h. This treatment approaches field conditions. The samples were then analyzed in the same manner as other plant material including gravimetric ash determinations. Similar subsamples of finely ground litter were digested wet using an H_2SO_4-$HCLO_4$-HNO_3 process and 4-h digestion time. The amount of each element lost at each temperature was related to the total amount of that element present based on the wet digestion data. Nitrogen losses at each temperature were determined by taking the ash from a 1.0000 g sample up in 5 ml of concentrated H_2SO_4 instead of 10 ml of 6N H_2SO_4. The sulfuric acid and ash mixture was then processed by the usual micro-Kjeldahl procedure for organic nitrogen (Black 1965). Nitrogen losses resulting from ashing were expressed in relation to total organic nitrogen which was measured on 1.0000 g subsamples of the same lot of finely ground litter.

RESULTS

Soils

The soils of all study plots were similar. Their average bulk density was 1.23 g/cm^3 with a particle density of 2.6 g/cm^3 averaging all depths to 55 cm. The soil fraction < 2 mm averaged 11.9% clay, 42.9% silt, and 45.2% sand. The soil was 55% rock and stones. Pore space formed 28% of the volume of the soil with large cavities and tunnels which resulted in an unusually high infiltration rate (297 l · m^{-2} · min^{-1} with 38% surface moisture). The cation exchange capacity ranged from 2.4 to 9 meq/100g with an average of 3.08 meq/100g. Roots formed 1.8% of the soil weight. The surface soil averaged 7.4% organic matter with 2.8% at 20–25 cm, and 1.6% at 50–55 cm depth.

In terms of the total elemental content of the soil (<2 mm) plus rock (>2 mm), this system is extremely limited in the amount of calcium (4.8% of the biologically essential cations measured), and in phosphate. Iron and potassium each comprise 42% of all cations measured. Magnesium represents 4.8% of the biologically essential cations. Assuming an annual range of 30–85% base saturation and a cation exchange capacity (C.E.C.) of 3.08 meq/100g, a 1 m^2 surface area to the depth of the feeder roots by calculation can hold from 3,856 to 17,993 meq cations/0.6m^3. If averages of the percent base saturation (48%) and C.E.C. (3.08 meq/100g) measurements are used, this soil can store 13,367 meq cations/0.6 m^3 at any point in time. Calculations follow the formula: (volume to 60 cm × bulk density) – % roots – [% rock × av. C.E.C. (<2 mm) × 20% (est. C.E.C. > 2 mm), × 100] – % Base saturation = active storage capacity of root zone.

The average storage capacity of 13,367 meq/0.6m^3 would not hold all of the June cations (22,951 meq/0.6m^3) and would result in a net estimated loss of 9,584 meq/0.6m^3 to the vegetation. June cation data are used because this is the period of maximum input of cations by snow melt. Measured cation losses in soil water below the root zone ranged from 86 to 180 meq · 0.6m^{-3} · yr. If the plant uptake data (9,584 meq/0.6m^3) are reduced by 180 meq of lost cations, then the plants would have taken up 9,404 meq/0.6m^3.

The next question is whether the plants growing on a meter square could tie up 9,400 meq of cations in a year's time. If we use the average percent ash for herbs, shrubs, trees encompassing their roots, bark, wood, leaves and needles, then ≈ 10% of the vegetation is ash (cations + anions). The anions storage capacity of the soil has not been considered here because it is technically almost impossible to measure and characterize. If 9,400 meq of cations were incorporated into the above ground vegetation and roots, this would mean, on the basis of 10% ash, on the average in all compartments of the ecosystem, ≈ 2.1 kg · m^{-2} · yr^{-1} of organic production. This figure was derived from the average elemental content of all fuels combined and weighted by the percent constituted by each fuel of the total fuel weights. It does not disagree with productivity figures for temperate forests listed by Spurr and Barnes (1973).

When only hot burns are considered, the extractable cations comprised 25,246 meq/0.6m^3 (June 1974), which would result in a loss factor of about 180 meq/0.6m^3, and a high plant uptake figure of 11,699 meq/0.6m^3. Analysis of the fuels (green herbs, shrubs, litter, dead sound wood, dead rotten wood in a variety of size classes showed a range of 1,658 to 6,951 meq of cations per m^2 of surface area which would be likely to be released in a 90 to 100% burn of relief fuels and other surface organic matter (excluding living and dead standing trees).

Soil water

Hot burn indicates soil surface temperatures > 300°C with all litter burned away, medium burn indicates surface temperatures of 180–300°C with up to half of the litter gone, and light burn indicates surface temperatures < 180°C, with the litter surface scorched.

Table 1 shows higher anions in soil water at all depths in the soil from burn sites as compared to control soil water from all depths. Cation levels in the burn soil water were 2 to 3 × higher than in the control soil water, for all three depths. The anions moved into the burn soil water more slowly than did the cations. Total anions were 5.9 mg/l in the surface soil water (A) in 1973, decreasing to 5.2 mg/l in 1974 and 1975. The soil water from 20–25 cm depth (B) in burn sites increased in anions from 2.6/l in 1973 to 3.0 mg/l in 1974 and 5.3 mg/l in 1975 showing a wave of slow downward movement for the anions. At 50–55 cm depth (C), the burn soil water had an average of 1.7 mg/l in 1973, increasing to 2.6 mg/l at that depth in 1974, and further increasing to 6.6 mg/l in 1975 (Table 1).

The cations for all burn intensities combined moved most readily into the soil solution during the 1st year after burn (Table 1). Surface (A) total cations in soil water from burns measured 62.5 mg/l in 1973, 35.4 mg/l in 1974, 28.5 mg/l in 1975, and 51.9 mg/l for 1976. At 20–25 cm depth (B), soil water showed an average of 59.2 mg of total cations/l in 1973, 30.6 mg/l in 1974, 26.1 mg/l in 1975 and

TABLE 1. Mean annual anion and cation load (mg/l) in soil water by depth and year, Lubrecht Fire Study

Depth (cm)		Year	Anions[a]	Cations
Burns				
0–10	A	1973	5.9	62.5
20–25	B	1973	2.6	59.2
50–55	C	1973	1.7	42.7
0–10	A	1974	5.2	35.4
20–25	B	1974	3.0	30.6
50–55	C	1974	2.6	25.1
0–10	A	1975	5.2	28.5
20–25	B	1975	5.3	26.1
50–55	C	1975	6.6	22.2
0–10	A	1976	17.3	51.9
20–25	B	1976	8.1	29.6
50–55	C	1976	6.3	17.9
Controls				
0–10	A	1973	2.0	27.5
20–25	B	1973	1.9	21.5
50–55	C	1973	—	—
0–10	A	1974	3.0	12.2
20–25	B	1974	2.4	13.1
50–55	C	1974	1.8	12.2
0–10	A	1975	2.3	12.1
20–25	B	1975	3.2	12.1
50–55	C	1975	2.0	14.0
0–10	A	1976	4.8	10.3
20–25	B	1976	5.3	24.3
50–55	C	1976	4.2	9.8

[a] Anions = F^-, NO_3^-, PO_4^-, Cations = Ca^{+2}, Cu^{+2}, Fe^{+3}, K^+, Mg^{+2}, Mn^{+3}, Na^+, Zn^{+2}.

TABLE 2. Mean total anions and cations by season of burn and burn intensity, for each soil depth and year

	Depth (cm)	Total anions (mg/l)	Total cations (mg/l)
1973–1974			
Hot spring burns	A 0–10	14.2	55.2
	B 20–25	4.2	43.9
	C 50–55	5.9	41.1
Medium spring burns	A 0–10	1.9	22.6
	B 20–25	2.8	25.6
	C 50–55	1.8	16.6
Light spring burns	A 0–10	1.2	17.4
	B 20–25	1.3	12.9
	C 50–55	2.7	15.0
Hot fall burns	A 0–10	3.1	45.9
	B 20–25	3.5	49.7
	C 50–55	2.8	34.8
Medium fall burns	A 0–10	5.5	73.4
	B 20–25	3.2	53.3
	C 50–55	1.7	21.4
Light fall burns	A 0–10	1.8	20.9
	B 20–25	0.9	17.9
	C 50–55	1.7	12.7

	Depth (cm)	Total anions mg/l	Total cations mg/l	Total cations me/l
1974–1975				
Hot spring burns	A 0–10	5.4	37.2	1.7
	B 20–25	8.1	37.8	1.6
	C 50–55	11.3	39.3	1.8
Medium spring burns	A 0–10	2.1	25.7	1.0
	B 20–25	2.0	21.5	0.9
	C 50–55	0.9	11.2	0.5
Light spring burns	A 0–10	2.4	12.4	0.6
	B 20–25	2.7	15.9	0.8
	C 50–55	2.2	11.7	0.5
Hot fall burns	A 0–10	12.2	27.4	1.2
	B 20–25	10.5	30.2	1.4
	C 50–55	26.8	30.6	1.4
Medium fall burns	A 0–10	6.5	46.3	2.0
	B 20–25	2.7	31.5	1.3
	C 50–55	4.0	24.3	1.1
Light fall burns	A 0–10	1.3	14.7	0.7
	B 20–25	1.1	12.6	0.6
	C 50–55	1.2	10.9	0.5

29.6 mg/l in 1976. Some cations reached the C depth (50–55 cm) in 1973 (42.7 mg/l compared to ≈ 18 mg/l for the control soil water). Total cations in the soil water from burn plots decreased to 25.1 mg/l in 1974 at 50–55 cm (C), 22.2 mg/l in 1975 and 17.9 mg/l in 1976. The high surface cation and anion concentrations in 1976 (3rd-year post-burn) reflect closely the twice normal precipitation in that year (749 mm) producing greater solubility of ions in the remaining ash from hot burns. The control sites showed a cation and anion build up at 20–25 cm in 1976.

Fluctuations by years in the amount of cations by depth were greatest between 1973 and 1974 with little difference between 1974 and 1975. Each plot studied had ≈ 12 measurements for any one depth and element for a year. The number of collections each year varied from 8 to 16 depending on whether water was available at each attempted collection. Anion and cation concentrations in the soil water correlated well with burn intensity for each sampling site. Some sites always showed the heaviest cation loading in the surface layers while others showed a more rapid downward movement with heaviest cations accumulating in the soil water from 20–25 cm depth, and occasionally at 50–55 cm depth.

Table 2 shows total anions as mg/l in soil water, and average total cations as mg/l and meq/l. Conversion of meq/l was essential in describing the likelihood of cation retention in the soil since C.E.C. is measured as meq/100g. Table 2 further breaks the burns down by intensity, year, and season of burn, and the average total cation and anion load by depth. In 1973–74, the 1st post-burn year, soil water from the hot spring burns had more anions than all other seasons and intensities. Fall medium burns produced

TABLE 3. Mean levels of elements as mg/l in soil water averaging all depths, 3 yr post-burn Lubrecht Fire Study

Controls	Ca^{+2}	Cu^{+2}	Fe^{+3}	F^-	K^+	Mg^{+2}	Mn^{+3}	NO_3^-	Na^+	PO_4^-	Zn^{+2}
All depths	5.4	.04	.2	.04	4.0	1.4	.02	4.5	.4	.7	.03
Hot burns All depths	18.5	.03	.04	.04	6.5	4.7	.15	5.8	2.7	.35	.02
Medium burns All depths	10.5	.02	.07	.05	5.1	2.4	.13	4.8	.85	.25	.02
Light burns All depths	5.0	.05	.2	.03	3.8	1.3	.12	2.8	.60	.07	.10

soil water with the highest cation load (meq/l) in 1974–75, and both spring and fall light burns had significantly low cation loads in the soil water at all depths (Table 2). During the 2nd year, (1974–75), fall hot burns showed the highest anion levels at all depths in the soil leachate, with hot spring burns second in position. The cations were all lower in concentration in the soil water during the second year, with no clear differences between the two hot treatments but a distinctly low level for the lightest burns from both seasons.

Individual cations behaved quite differently (Table 3). Calcium, K^+, Mg^{+2}, and Na^+ were most concentrated (mg/l basis) in soil water from the hottest burns through the 3rd year following burning. Iron, and to a lesser extent Zn^{+2}, were lowest in the hottest burns even after 3 yr of leaching. Lower levels of cations with depth occurred on most burn soils, and was especially evident on the hottest burns. Lightly burned plots showed high levels of cations at 20–25 cm depth sooner than did those which underwent hot burns.

Table 4 shows the standard deviation for hot burns and controls by depth and element. Three yr of natural seasonal variability produced high standard deviations. These data have had seasonal effects removed. Within intensity groupings and seasonal data for soil water from any depth, the standard deviation was normally low.

Table 5 shows the amount of nutrient increase or decrease in soil water from the three depths as meq/l for light and hot burns for 1 yr of post-burn data, compared to each other and to the controls for each element studied. Light burns were always lower than the controls in Fe^{+3}, F^-, Mn^{+3} and Zn^{+2}; and lower at two out of three depths in Ca^{+2}, K^+, NO_3-N, Na^+, and PO_4^--P. Hot burns were lower in only Fe^{+3} from all three depths compared to the soil water from control soils. Hot burns were consistently higher than controls in the levels of Ca^{+2}, K^+, Mg^{+2}, Mn^{+3}, and NO_3^--N in soil water from all three depths studied, and higher in two out of three depths for Cu^{+2}, Na^+, and Zn^{+2} (Table 5). Soil water from hot burns was higher than the same from light burns at all depths in Ca^{+2}, K^+, Mg^{+2}, Mn^{+3}, nitrate, and phosphate, and higher at two of the three depths in Cu^{+2}, F^-, Na^+ and Zn^{+2}. These patterns of elemental concentrations are related to altered solubility of the ions as a result of the quality of ash produced (i.e., burn intensity).

Table 6 compares the soil water from the two hottest single burns to the two lightest burns studied with data expressed as meq/l for cations and mg/l

TABLE 4. Standard deviation and means for elements in soil water from Lubrecht soils by depth over 3 yr (mg/l basis)

	Depth (cm)	Ca^{+2}	Cu^{+2}	Fe^{+3}	F^-	K^+	Mg^{+2}	Mn^{+3}	NO_3^{-N}	Na^+	PO_4^{-P}	Zn^{+2}
Standard deviations												
Hot burns	0–10	9.0	.01	.005	.09	6.1	0.4	.001	1.9	0.4	1.0	.01
	20–25	6.1	.01	.008	.005	3.9	0.3	.001	1.4	0.9	1.0	.01
	50–55	2.0	.02	.009	.004	1.0	0.3	.001	0.8	1.2	.5	.02
Controls	0–10	2.3	.001	.002	.001	1.0	0.4	.001	1.2	.03	.69	.01
	20–25	2.0	.01	.02	.002	1.0	0.8	.001	1.8	.06	0.8	.01
	50–55	0.9	.002	.02	.002	0.8	.001	.001	0.9	.03	0.7	.01
Means												
Hot burns	0–10	21.8	.05	.03	.27	23.0	3.0	.01	9.2	1.1	1.3	.02
	20–25	14.8	.05	.05	.04	17.6	2.4	.02	7.3	2.5	1.4	.04
	50–55	10.8	.07	.07	.03	6.8	3.3	.03	5.6	3.7	.8	.07
Controls	0–10	7.5	.02	.08	.02	4.2	1.1	.005	6.9	.29	.87	.04
	20–25	6.4	.08	.19	.06	3.9	2.1	.02	8.4	.32	1.2	.06
	50–55	3.2	.05	.15	.05	3.4	0.5	.02	2.8	.19	1.0	.04

TABLE 5. Comparison of soil water from the hottest and lightest burns and controls after one year, meq/l* basis, Lubrecht Fire Study. Nutrient increase +; decrease –

	Depth	Ca^{+2}	Cu^{+2}	Fe^{+3}	F^-	K^+	Mg^{+2}	Mn^{+3}	NO_3^{-N}	Na^+	PO_4^{-P}	Zn^{+2}
Light burn to control	A	– .108	+.0063	–.0072	–.045	–.010	+.007	–.007	– .445	+.150	–.966	–.0001
	B	– .162	+.0001	–.0148	–.070	–.034	–.043	–.001	– .402	–.081	–.741	0
	C	+ .051	–.0004	–.0032	–.022	+.012	+.057	–.004	+ .024	–.004	+.200	–.0003
Hot burn to control	A	+1.033	–.0015	–.0077	+.024	–.971	+.572	+.007	+1.377	–.021	–.823	–.0005
	B	+ .890	+.0012	–.0046	–.061	+.200	+.510	+.016	+ .475	+.061	–.258	+.0016
	C	+1.154	+.0003	–.0062	–.032	+.439	+.751	+.029	+1.198	+.145	–.335	+.0003
Hot burn to light burn	A	+1.141	–.0060	–.0005	+.069	+.981	+.565	+.014	+1.822	–.027	+.144	–.0004
	B	+1.051	+.0011	+.0102	+.009	+.235	+.553	+.017	+ .878	+.142	+.490	+.0018
	C	+1.103	+.0007	–.0030	–.010	+.427	+.694	+.033	+1.174	+.149	+.535	+.0006

* Data not rounded off to allow further testing.

for anions. Significant differences occurred between soil water from hot and light burns for Fe^{+3} which was nearly always lowest in the hottest burns. Soil water from the hot burns was generally higher in all other elements except Fe^{+3}. During the 2nd year the levels of most cations in soil water from hot and light burns were lower than in the first year. This was especially true of K^+ and Ca^{+2}.

Statistical analysis

Calcium, Mg^{+2}, Mn^{+3}, Na^+ and K^+ were always significantly higher (5% level) in soil water from burned sites than from control sites during the first year. During the second year, Ca^{+2}, Mg^{+2}, Mn^{+3}, Na^+ and PO_4-P increased in concentration in the soil water from burned sites when compared to control sites. For soil water, the levels of Ca^{+2} correlated closely with the levels of Mg^{+2} and K^+, especially at 0–10 cm depth. Copper correlated positively with H^+ concentration, but Ca^{+2} was inversely related to the H^+ populations.

At most depths the levels of Ca^{+2}, Cu^{+2}, Fe^{+3}, K^+, Mg^{+2}, Mn^{+3}, NO_3^--N and PO_4-P were significantly different (5% level) between soil water from plots burned with low, and high temperatures. Season of the year produced significant differences in the levels of Ca^{+2}, Cu^{+2}, F^-, Mn^{+3}, and Na^+ for most control depths.

For fall burns, the soil water was significantly different from spring burns in H^+, Ca^{+2}, Fe^{+3}, F^-, K^+, Mg^{+2}, Mn^{+3}, NO_3^--N, Na^+, PO_4^--P and Zn^{+2} for all three intensities studied, mainly at the surface

TABLE 6. Average elemental levels in soil water by depth from the two hottest and lightest burns, Lubrecht Fire Study, after 1 and 2 yr, as meq/l and mg/l*

	Depth	H^+ ($\times 10^{-4}$)	Meq/l							Cations (meq/l)	Anions (mg/l)
			Ca^{++}	Cu^{++}	Fe^{+++}	K^+	Mg^{++}	Mn^{+++}	Na^+	Zn^{++}		
1973–1974											
Hot	0–10	.003	1.489	.002	.007	0.571	0.688	0.098	0.209	0.002	3.0	6.4
	20–25	.004	1.295	.002	.011	0.383	0.517	0.103	0.219	0.001	3.3	2.5
	50–55	.003	0.914	.001	.009	0.245	0.415	0.016	0.205	0.001	1.7	0.9
Light	0–10	.003	0.274	.001	.023	0.183	0.164	0.013	0.132	0.001	1.4	0.8
	20–25	.006	0.320	.001	.028	0.191	0.166	0.023	0.119	0.001	2.7	1.8
	50–55	.003	0.370	.001	.021	0.148	0.179	0.013	0.113	0.002	1.6	0.9
1974–1975											
Hot	0–10	.0024	0.85	.001	.004	0.147	0.165	0.004	0.100	0.0013	1.5	15.4
	20–25	.0015	0.92	.003	.012	0.162	0.376	0.003	0.144	0.0022	1.7	3.9
	50–55	.0022	0.67	.005	.010	0.442	0.553	0.0016	0.094	0.0020	1.8	2.6
Light	0–10	.0097	0.29	.002	.022	0.130	0.082	0.001	0.029	0.0027	0.6	1.1
	20–25	.0054	0.20	.009	.009	0.076	0.071	0.0005	0.027	0.0015	0.4	0.9
	50–55	.0034	0.23	.001	.022	0.100	0.060	0.0018	0.017	0.0019	0.4	0.9

* Data not rounded off to allow further testing.

TABLE 7. Elemental content of plants from burned and unburned areas, Lubrecht Fire Study

		μg/g								
Species	Conditions	Ca^{+2}	Cu^{+2}	Fe^{+3}	K^{+}	Mg^{+2}	Mn^{+3}	Na^{+}	PO$_4$-P	Zn^{+2}
Boletus sp.	Burn	75	77.5	1,425	290,000	7,500	250	1,900	187,000	495
	Control	75	140.0	460	340,000	11,050	125	1,775	325,000	940
Vaccinium membranaceum	Burn	7,263	13.6	154.0	15,750	1,588	1,196	94	6,813	24.9
	Control	8,850	10.4	101.8	3,700	1,350	1,150	109	5,975	12.9
Epilobium angustifolium L.	Burn	10,375	14.1	255	8,750	2,575	515	210	19,625	49.7
	Control	13,050	9.6	230	3,600	765	295	95	11,000	34.2

depth. In time, the number of significantly different ions became less with the highly soluble Na^{+} the first element to return to control levels. The element input from winter decay under a heavy snowpack was considerable during the second year. In general, the greatest differences in ionic concentrations in soil water occurred between hot and control and hot and light burns.

Precipitation and thrufall

During the first year following the burns, an average of 560 l/m^2 of bulk precipitation and thrufall reached the forest floor. The Lubrecht area receives an average of 466 mm/yr in precipitation with a range from 309 to 749 mm/yr. The average storm intensity was 0.07 cm/h with a maximum of 2.45 cm/h. Bulk precipitation added 125 meq $\cdot$ m^{-2} $\cdot$ yr^{-1} of total cations measured in this study, or 1,250,000 meq $\cdot$ ha^{-1} $\cdot$ yr^{-1}. Large amounts of calcium (423 mg $\cdot$ m^{-2} $\cdot$ yr^{-1}), potassium (1406 mg $\cdot$ m^{-2} $\cdot$ yr^{-1}), nitrate (751 mg $\cdot$ m^{-2} $\cdot$ yr^{-1}), and phosphate (243 mg $\cdot$ m^{-2} $\cdot$ yr^{-1}) are brought into the system by bulk precipitation. On the study area, $\approx$ 25% of the surface received bulk precipitation and 75% received thrufall, so that data for elemental input were weighted to show this balance. Stemflow measurements were not within the scope of this study which emphasized outside inputs.

Overland flow

No truly satisfactory means of measuring overland flow has yet been devised, but the system of land wiers used here did work reasonably well. Overland flow on control microwatersheds (300 m^2) averaged 0.17 g/m^2 of sediments moved annually while burned microwatersheds averaged 1.9 g/m^2 of sediments moved. This means that burning temporarily accelerated erosion by a factor of 9 × on 35–42% slopes. At this rate, it would require $\approx$ 50,500 yr to erode away the entire root zone to 60 cm, assuming a 50-yr burn frequency and hot burns ($>$ 300°C). The 2nd year sediment load was 2.4 × greater on the burned slopes compared to unburned slopes. Only a small amount of cations (0.018 meq $\cdot$ m^{-2} $\cdot$ yr^{-1}) were moved in solution in overland flow on control microwatersheds, and only slightly more (0.043 meq $\cdot$ m^{-2} $\cdot$ yr^{-1}) on burned microwatersheds during the 1st year. These figures are only approximations because on two occasions the collection containers overflowed.

Elemental content of plants

Nutrient levels in all fuels, and in some of the small successional species, were measured to determine if and where elemental sinks occurred in this ecosystem. The data are too lengthy to include here. The shrubs and herbs had an average of 3.9% ash and the litter (leaf, fermentation, humus combined) showed 14.6% ash. If completely burned, litter could release 1,230 to 5,609 meq/m^2, while the shrubs and herbs combined from clip plots could release 1,658 to 6,957 meq cations/m^2.

A few plants showed unusual concentrations of biologically important elements (Table 7). *Boletus* sp., a Basidiomycete, concentrated heavily on hot burn sites the first year and showed in the fruiting bodies (cap and stipe) 1,425 μgFe^{+3}/g; 290,000–340,000 μgK^{+}/g dry weight; 7,500–11,050 μgMg^{+2}/g; 1,775–1,900 μgNa^{+}/g; 187,000–325,000 μgPO$_4$-P/g; and 495–940 μgZn^{+2}/g of tissue. Calcium in this fungal fruiting body was low (75 μg/g, which constitutes only 0.03% of all of the cations) and is to be expected in this type of plant. Potassium formed 96% of the total cations measured in *Boletus*. Little differences in ion level occurred in fungal tissue from burn and control soils except for Fe^{+3}.

Vaccinium membranaceum Dougl. (huckleberry) concentrated significantly high levels of iron, potassium, and zinc (Table 7). Another concentrator was *Epilobium angustifolium* L. or fireweed where tissue from burned substrates showed significantly high levels of K^{+}, Mg^{+2}, Mn^{+3}, Na^{+}, PO$_4$-P, and Zn^{+2}.

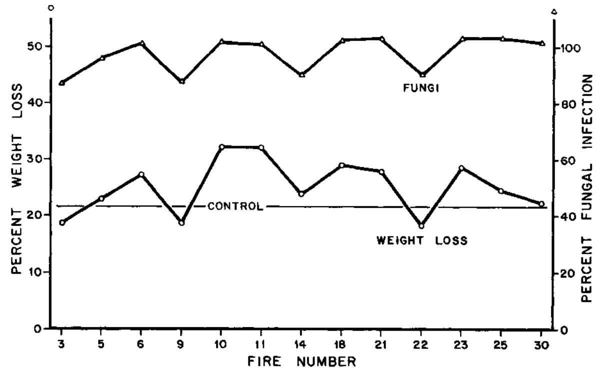

FIG. 1. The influence of burns of different intensities on the decomposition of Douglas-fir needles, Lubrecht Fire Study, as percent needle weight loss and percent fungal infection from gauze counts. Fire 22 was hot (300°C+), Fire 18 was cool (180°C).

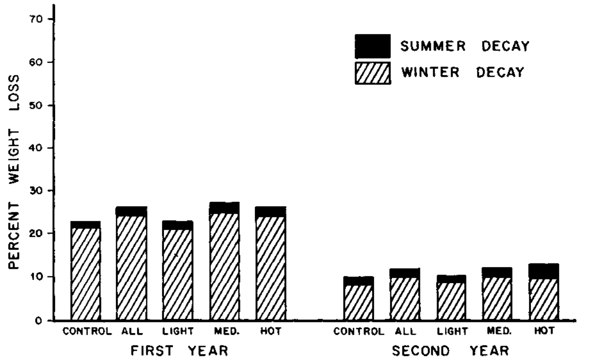

FIG. 2. Mean seasonal percent weight loss of Douglas-fir needles on burned and unburned substrates for winter and summer by burn intensity for 2 yr, Lubrecht Fire Study.

Litter decomposition

Figure 1 shows a close correlation with gauze counts of fungi (performed directly on the Saran® mesh containers) and the percent weight loss of Douglas-fir needles during the first year of study. This relation did not hold during the second year, partly because it was not possible to separate dead and living fungi under a dissecting scope. All but three plots produced conditions after the burn which stimulated decomposition above that of control levels (Fig. 1), although the difference was usually not significant.

Weight loss of the needles during the first year exceeded those during the second year, especially during the winter months. No significant differences in percent weight loss were measured in either year between substrates which had received different intensities of burn. The needles lost from 22 to 27% of their dry weight during the 1st year, and from 8 to 9% during the second winter. Summer weight losses ranged from 1.5% to 3.4% in the first year, and from 1.1 to 2.3% in the second summer. Winter weight losses were 21 to 25% in the first winter (Fig. 2). Weight losses during the second winter averaged 1.87% of the original weight, and treatment differences had largely disappeared.

Ash

Ash was leached in different ways to study the release rate for biologically essential elements. Figure 3 shows a gradual increase in the hydrogen ions released with time over 10 successive 100-ml leachings of 5 g ash samples with natural precipitation. The ash from hot burns increased in hydrogen ions more slowly than did ash from medium and light burns. The drastic increase in hydrogen ion concentration at the fourth leaching is related to a short period of drying.

Total cations and anions decreased rapidly in the first and second leachings, and then leveled out to a steady release rate with the anions slightly higher than the cations thru the tenth leaching (Fig. 4). Note that, in nature, the reverse is true, suggesting complexing of the anions in soil. Freezing of moist ash increased the pH of the leachate from 7.6 to 9.4, Ca^{+2} from 0.57 to 2.6 mg/l, Cu^{+2} from 0.0001 to 0.002, Fe^{+3} from 0.0005 to 0.0068, K^{+} from 0.14 to 9.64, Mg^{+2} from 0.1 to 0.56, Mn^{+3} from 0.0012 to 0.0063, and Na^{+} from 0.007 to 0.22 mg/l. Drying and rehydration produced a similar increase in nutrient release rate.

DISCUSSION

Soils

The calculations of the cation storage capacity of the soil show that 9,400 meq of cations would be lost from the soil based on the maximum cation loading applied from the ash. Measurements show that this forest ecosystem above and below ground aver-

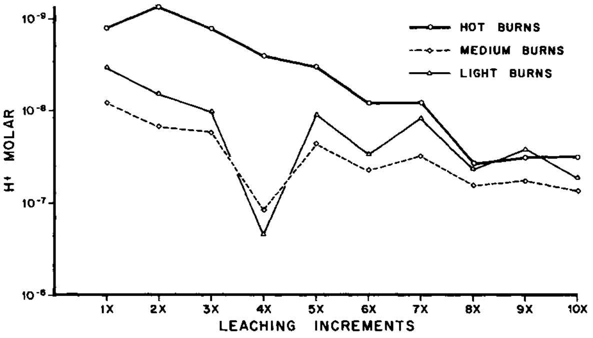

FIG. 3. Changes in the H+ concentration of ash leachate from 5 g ash samples leached with ten 100-ml increments of precipitation, as mg/l at room temperature.

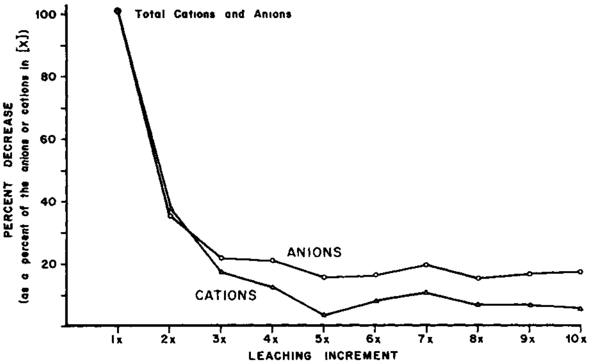

FIG. 4. Percentage decrease in the release of total cations over the original level (1×) for all ash samples leached with ten 100-ml increments of precipitation, all plots as me/l, and anions as mg/l.

ages ≈ 10% ash. On this basis, the calculated annual productivity per square metre will be 2.1 kg (dry weight). This seems a bit low although it is not intended as an estimate of net annual productivity, but only as a means of determining if 9,400 meq of cations/m^2 could be accommodated by the growing vegetation in a year's time. Obviously, other nutrient changes would occur from the exchange sites and from freshly weathered minerals which released cations into the soil solution. At this point, it does appear that the vegetation could take up substantial amounts of cations resulting from a burn. The regrowth of shrubs, herbs, and grasses during the first year following burning was quite substantial. There are no measurements of shrub biomass after burning, but pre-burn biomass data show 3 to 4 kg/m^2, so that 1 or 2 kg dry weight/m^2 could be easily produced by rapidly resprouting vegetation. Unfortunately, there are no productivity data for Douglas-fir ecosystems from this area to use for comparison.

The shrub and herb data show a maximum of 6,951 meq of cations/m^2 of surface area which could be released from ground dead and live fuels by burning. This figure is probably low because the random sampling system used did not include large clumps of willow and stumps. These larger shrubs rarely burned so that they are not a realistic part of the fuel for burning under standing timber.

Soil water

The anions appear to move into the soil more slowly than do the cations. The anion storage capacity of soil is an elusive and difficult thing to measure. The abundant negative charges on the phosphate ion may slow its movement, but this argument would not hold for nitrate. Fluoride and phosphate are insoluble in pure water, but somewhat soluble in acid soil water. The alkaline quality of surface soil water during the 1st year following burning may reduce phosphate solubility and slow downward movement. If this is true, the ecosystem is benefited since both nitrate and phosphate are ions which are often scarce in the soil and likely to be limiting to growth in western Montana. Natural mechanisms which favor the retention of phosphate in available form following burning are beneficial to recovery of the vegetation. Rapid recovery of the shrubs by resprouting and herbs is important to nutrient retention since their roots serve as a sink for freshly released elements. If the shrub roots are killed by fire, which rarely occurs, larger nutrient losses can be expected.

The levels of cations entering the soil after a burn decrease in 2 yr resulting in a higher hydrogen ion concentration which influences the solubility of the cations and anions in the soil. Movement of the cations into the soil solution, onto the exchange sites, and into the roots is benefited by gradual downward movement of water and leached ions. Additional studies on other soil types suggest that any soil can be crudely rated in terms of its nutrient retention powers based on soil texture, organic content, clay content, porosity, and the annual precipitation. The slower the movement of cations, the greater is the chance of uptake, provided the pH of the soil solution is not so high that elemental solubilities are greatly altered. Reduction of the infiltration rate as a result of a hot burn increases the chances of overland flow and surface nutrient loss, while reducing elemental input from ash may be most detrimental to only local streams, but on an infertile soil, reduced nutrient input from surface runoff can damage not only local streams, but the fertility of the soil itself. Vogl and Ryder (1969) showed poorer growth on burned sites under slash piles as a result of reduced infiltration and flooding of the exchange sites with one ion. The role of some of the carbon particles left after burning in retaining ions is not understood.

Medium burns are hard to define, but soil water is clearly different chemically between hot and lightly burned sites.

Calcium, which was low in concentration (and in percent composition) in the soil and parent material was not lost in significant amounts below the root zone in control and lightly burned sites (< 300°C). When heavy fuel loadings were burned (> 8 kg/m^2) and high surface soil temperatures were reached (>300°C), losses of calcium did occur. The levels of calcium added each year in precipitation equalled or exceeded the calcium loss below the root zone on control and light to medium burn sites.

Magnesium did show a slight loss below the root zone in control sites, and precipitation additions of Mg^{+2} were not sufficient to offset Mg^{+2} losses. The hottest burns caused accelerated losses of Mg^{+2} below the root zone. Magnesium also represents a low percent composition in the soil-rock system, so that treatments which accelerate magnesium losses must be considered as detrimental to the biological life of the soil. No other biologically essential elements were lost in extreme amounts below the root zone. This means that on both control sites and overall burn intensities studied, precipitation input was adequate to cover leaching losses. A change in annual precipitation could alter this balance. Nutrient losses through smoke and future harvest have not been measured in this study, but data are available from adjacent areas.

The influence of bases added to the soil in reducing the hydrogen ion concentration and elevating soil pH is most noticeable in the changed solubility of acid soluble elements such as iron. The iron content of the soil extracts and soil water reflected the intensity of burn quite closely. In general, the hotter the burn, the more ash produced, the more bases released, the fewer hydrogen ions, and the less soluble iron in the soil solution. Statistically, the relationship held up for all plots regardless of season of burn. It appears that the levels of iron in soil water following burning can be used as a rough indicator of the burn intensity.

The nitrate levels in soil water from hot burns were nearly always significantly higher than the nitrate in control soil water. This agrees with results from Wright and Tarrant (1957) who have shown higher levels of nitrogen following burning. Others (Knight 1966) have shown nitrogen losses from the soil resulting from burning, but the discrepancy appears to be related to time and depth of sampling, burn temperatures, and analytical procedures. The fact that fresh ash leached with precipitation showed considerable quantities of nitrate confirms the idea that not all nitrogen is lost during burning.

The phosphate levels in soil water from hot burns were elevated to levels not significantly higher than for control soil water which suggests minimal phosphate losses during burns of this range of intensities. This phosphate addition is almost certainly critical to the rapid recovery of the understory vegetation since phosphate is not normally abundant in this soil solution.

The higher levels of calcium, potassium, magnesium, manganese, and in some cases sodium in soil water from hot burns when compared to light burns are not surprising considering the concentration of these elements in the vegetation and dead fuels. Medium burn intensities are hard to characterize and show fewer significant differences in soil water quality than do hot and light intensities. For the best nutrient release with minimal nutrient losses, medium intensity fires are probably most like natural light ground fires and are probably least damaging to the soil's productive capabilities.

Fall burns had greater impact on soil water chemistry compared to spring burns because fall burns were generally hotter, the fuels were drier, and a greater range of fuels burned. Few shrubs or herbs burned in spring, but many of these were completely burned in the fall. Logs which had been wet all winter burned poorly in spring, but quite well after a summer of drying. Fall burns showed, in general, a greater increase in soil nutrients in the spring of the 2nd year because this was the first major leaching of these soils, while spring burns had undergone some leaching by fall rains.

The larger number of elements which was significantly different (by intensity) in soil water from 50–55 cm depth during the second year indicates the gradual downward movement of these elements, and shows that the accelerated nutrient losses following burning were still present, although somewhat reduced over 1st year losses. The elemental content of soil water was most significantly influenced by burn intensity, and by date of collection (relative to snowmelt). Sodium moved rapidly into the soil because it is highly soluble over a wide pH range and is monovalent.

The high levels of ionic additions to control soil water during the second year were biased by one of the five sites which had deeper than average litter (21 cm). The second winter had a good snow pack (2.1 m) which lasted from November through late May. The result was rapid litter decomposition with significant amounts of ions released.

Precipitation and thrufall

The amount of precipitation and the maximum and average storm intensities were never great enough in themselves to cause overland flow or erosion. The elements added to the soil surface from bulk precipitation and recycled from thrufall were significant in offsetting most losses of elements overland or below the root zone.

The condition of the crown (living, lightly scorched, or heavily scorched), influenced the amount of nutrients collected in thrufall. Thus, it is necessary to collect from a large number of sites to obtain results which are truly representative of the entire stand. Sodium losses were considerable from both living and dead crowns.

Overland flow

The storm intensity range and infiltration characteristics of this soil would indicate no possible overland flow. Overland flow was measured during periods immediately following burning when the soil surface permeability was limited by ash which forms a greasy cover, and by frozen surface soil during snowmelt: In general, neither the quality or quantity of water moving over the surface was great. Neither solid nor liquid nutrient losses were significant on these slopes (to 42%) because of the rapid regrowth of understory vegetation and uneven freezing of the soil. Water moved only a few metres before it encountered unfrozen surface soil where infiltration could occur.

Elemental content of plants

The elemental content of the fuels is generally low because a large percentage of the fuel was rotten wood > 7.6 cm in diameter. This type of wood has much iron, calcium, and potassium, but much of the nitrogen and phosphate have been removed during the earlier stages of decay.

Plants like *Vaccinium membranaceum* which are adapted to sprout after fire also appear to be quite efficient in storing elements released from burning. The amount of living roots of shrubs and herbs in the upper 30 cm of soil following a burn is critical to the nutrient retention capabilities of the soil. Certain fungi and particularly rhizomorph tissues serve as elemental sinks in the soil storing large amounts of elements which might otherwise be lost from the system (Stark 1972).

Litter decomposition

Litter decomposition rates were increased on bare mineral soil with ash over control levels, but the differences are marginally significant. The trend continued for both years with a decrease in the significance between decay rates on heavily burned and unburned substrates in the second year. The greatest weight losses occurred during the first winter under the snow. The influence of burn intensity on subsequent litter decomposition rates is not clear.

Ash

Both freezing and thawing, and alternate wetting and drying influence the rate of elemental release from ash. These differences in release rate are probably the result of physical and some chemical effects on the ash particles themselves. To some extent, the number of storms per year, temperature extremes as well as the length, intensity, and amount of precipitation will influence the amount of ions released from ash. The first leaching always releases the most ions if the ash has not developed low wettability. Hydrophobic properties of some ash will delay nutrient release.

The Biological life of the soil and fire

The biological life of soil is defined as the years that a particular soil is capable chemically of supporting trees. All soils form initially from weathering or abrasion of solid rock. The elemental content of the soil formed from rock is dependent on the initial concentration of biologically essential ions in the minerals of the rock and on ionic additions from bulk precipitation and other sources. The biological life of any soil is dependent on how rapidly the total weatherable nutrient reserves of that soil are removed or leached away. Removal can be through dust, smoke, harvest, erosion, upheaval, overland flow, or leaching over very long periods of time. In theory, excessive losses of any biologically essential element which is low in percent of total cations, or low in available or total concentration in the soil-rock system will accelerate the decline of the biological life of that soil.

When elemental limitation occurs, those species with high nutrient demands begin to drop out and those species with well-developed direct nutrient cycling predominate (Stark 1969). Leaching losses become more significant to the vegetation because the limiting element(s) prevent the development of larger and deeper root systems capable of penetrating to zones with freshly weathered minerals. The trees become progressively more dependent on recycling nutrients from their own organic litter and less dependent on nutrient uptake from the soil. Long periods of leaching remove more of the limiting nutrient(s) resulting in smaller, less diverse forests. Precipitation input of nutrients becomes even more important to the vegetation. Ultimately, a stage of system nutrient depletion is reached which renders the site incapable chemically of supporting trees. Ideally, we want to maintain soils at maximum productivity as long as possible.

The biological life of most soils covers thousands of years, from the point in time when the soil is deep enough and sufficiently weathered to provide adequate available nutrients to grow a tree, to the time when the soil minerals are so depleted that they can no longer maintain (with the aid of precipitation) a level of elements in available form adequate to support a forest. It is possible to quantitatively describe the stage of any soil in terms of the theoretical years left to its biological life. It is also possible, using this approach, to study the effects of accelerated nutrient loss resulting from various land use treatments on the length of the biological life of the soil.

The data taken from this fire study have been integrated to demonstrate the approach which is based on nutrient loss in soil water below the root zone (60 cm). These measurements are applied to the following model:

$$A = (B - C - P + E + O + H + S - D - W)^{-1} \times R \times T,$$

where: A = biological life remaining (years),
B = burn or treatment elemental loss measured per rotation as $\text{meq} \cdot \text{m}^{-2} \cdot \text{rotation}$ below the root zone,
C = control loss per rotation as meq/m^2 below the root zone (× years with accelerated loss),
S = nutrient losses in smoke, dust as $\text{meq} \cdot \text{m}^{-2} \cdot \text{rotation}$,
P = precipitation input per rotation as $\text{meq} \cdot \text{m}^{-2}$,
E = erosional loss as $\text{meq/m}^2\text{/rotation}$,
O = overland flow loss per rotation as $\text{meq} \cdot \text{m}^{-2}$,
H = harvest losses per rotation as meq/m^2,
T = total elemental content of rock + soil to the limit of the root zone (60 cm) as meq/volume of root zone,
D = deep soil nutrient pumping by deep roots (beyond 60 cm),
W = weathering as meq/m^2 to depth of the root zone/yr,
R = rotation time of treatment in years.

This approach does not guarantee that all of any element will be released by weathering during the calculated biological life. Available weathering data do not allow more than a guess.

In the formula above, S, D, and H were not measured because they were not within the scope of the study. The elemental input of calcium for control soils and burn soils from precipitation (P) is 21.2 meq/m^2 surface area. Or:

$$A = \frac{61{,}967}{\begin{pmatrix}97.8\text{meq/m}^2 - 20.8\text{m}^3 - 21.2\text{meq}\\ \quad + E + O + H + S - D - W\end{pmatrix}} \times R\,(50)$$

$$= \frac{61{,}967}{55.8} \times 50 = 55{,}500 \text{ years until all Ca}^{+2}\text{ is in theory lost from hot burns on a 50-year rotation.}$$

The biological life of this soil based on magnesium losses is ≈ 11,000 yrs, assuming a 50-yr burn rotation and weathering adequate to release all magnesium in the upper 60 cm during that time. The loss of calcium below the feeder root zone (from soil water quality and quantity at 50–55 cm) on control sites was 20.8 meq/m^2 (C) or no net loss. This means that the control soil, in the absence of fire and harvesting, and with no change in precipitation or climate, could go on for a long time and not be depleted of total calcium. The actual loss of calcium for burn soils was 45 meq/m^2 (B/year [corrected for precipitation input]). The data can be projected on a 3-yr period of accelerated nutrient loss, with a decline of calcium loss of at least 50% for each of the 3 yr, and over a 50-yr fire rotation. This assumes no harvesting, and no significant erosion or overland flow losses. The Ca^{+2} lost in 1 m (to 60 cm) of surface area could be divided by the total calcium actually and potentially available in the soil and rock (T) through weathering. This gives the number of 50-yr fire rotations which can occur before all calcium is lost from the system. The approach assumes that weathering will be adequate to release all of the calcium bound in the minerals to the depth of 60 cm in that time. The number of 50-yr fire rotations (R) possible can be multiplied by 50 to give the actual years that the soil will contain calcium (assuming minimal or no erosion). This figure should be corrected for the minimum amount of calcium needed to support a forest. The final figure of 55,000 yr means that, in theory, if there were no geologic, glacial, vegetation or climatic changes (which there will be), this soil could support trees for another 55,000 yrs. This figure is of no absolute value, but is useful for comparison. If a fire of intensities > 300°C could occur every 10 yr instead of every 50 yr, then the biological life would be reduced to 11,000 yr. In this forest type, fuel accumulations are not sufficient to support a hot fire every 10 yr. Fire frequency is used here only for comparison.

Another value of the method is in comparing the nutrient losses from different treatments. Treatment can be from logging, burning, fertilizing, or any other land practice. If treatment A reduces the biological life of a soil by 10-fold over treatment B, then the more damaging treatment must be put in perspective. If the biological life of the soil in question is 50 yr with treatment A and 500 yr with treatment B (which is realistic for some tropical spodosols), then cost differences between the treatments can hardly be considered if the management objective is to maintain soils capable of growing trees.

In the case of fires on Holloway Series soils in western Montana, the most severe burning conditions possible under standing timber reduced the biological life from infinity (theoretical) to 55,000 yr. Catastrophic, climatic, erosional, and glacial cycles are all likely to interfere with natural processes over such a long time span. Here we can say that 55,000 yr tells us that we can burn on this soil type for a very long time with no problems of lost soil fertility, if the soil is not chemically fragile. Chemically fragile soils are so young that weathering has supplied low levels of available nutrients which can be depleted by too frequent or too intense treatment. The Holloway soils are somewhat fragile in respect to Mg^{+2} and Ca^{+2}. Fire or other disruptive land treatments applied too often or too severely during the early developmental stages can produce "nutrient shock" or a condition of temporary nutrient depletion where one or more available nutrients is limiting to tree growth until the elements can be restored by weathering. Sixty percent of the burns were < 300°C and did not result in any accelerated nutrient losses, but did reduce up to 60% of old fuel.

When the biological life of a soil extends beyond the predicted time for major geologic, or climatic changes, then economics and silviculture should be the first concern. A 10-fold productivity reduction is not significant if it involves millions of years. We should try to use the least damaging practices on long-lived soils only if economics will allow it, and the practice is silviculturally and ecologically sound. In nature, occasional hot wild fires must certainly result in massive nutrient losses. Undoubtedly, these fires have good effects which have not been fully studied.

The damage land use does to our wildlife and vegetation can best be evaluated through the positive damage to the soil, whose productivity (with climate) governs the amount and type of vegetation possible, and the wildlife that can survive on it. Although this approach leaves something to be desired, it is an attempt at quantifying the reduction in soil fertility associated with land uses.

ACKNOWLEDGMENTS

This research was supported by the U.S. Forest Service (Northern Forest Fire Laboratory), the Office of Water Research and Technology, and McIntyre-Stennis Funds. The author thanks Robert W. Steele for devoted service to this project.

LITERATURE CITED

Ahlgren, I. F., and C. E. Ahlgren. 1960. Ecological effects of forest fires. Bot. Rev. **26**:483–533.

Behan, M. J. 1970. The cycle of minerals in forest ecosystems, p. 11–29. *In* The Role of Fire in the Intermountain West—a Symposium. Intermountain Fire Res. Council. Univ. of Montana, School of Forestry, Missoula, Montana.

Brackebusch, A. P. 1970. Cause for Concern, p. 119–126. *In* The Role of Fire in the Intermountain West —a Symposium. Intermountain Fire Res. Council. Univ. of Montana School of Forestry, Missoula, Montana.

Bremner, J. M. 1965. Total nitrogen, p. 1164–1166. *In* C. A. Black [ed.] Methods of Soil Analysis, Part 2. Am. Soc. Agron., Inc., Madison, Wisconsin.

Brown, J. K. 1974. Handbook for inventorying downed woody material. U.S. Dep. Agric. Forest Service General Technical Report Int-16.

Cleaves, E. T., A. E. Godfrey, and D. P. Bricker. 1970. Geochemical balance of a small watershed and its geomorphic implications. Geol. Soc. Am. Bull. **81**: 3015–3032.

Farber, L. [ed.]. 1976. P. 466–483. *In* Standard methods for the examination of water and waste water. American Public Health Association and American Water Works Assoc., New York, N.Y.

Fredriksen, R. L. 1971. Comparative chemical water quality on natural and disturbed streams following logging and slash burning. Proc. Symp. Forest Land Uses and Stream Environment. Oregon State University.

Grier, C., and D. W. Cole. 1971. Influence of slash burning on ion transport in a forest soil. Northwest Sci. **45**:100–106.

Habeck, J. R. 1970. Fire ecology investigations in Glacier National Park. Botany Dep., Univ. of Montana, Missoula, Montana. 80 p.

Hembree, C. H., and F. H. Raintree. 1971. Chemical degradation on opposite flanks of the Wind River Range, Wyoming. U.S. Geological Survey Water Supply Paper **153** S-E. 9 p.

Hesse, P. R. 1971. A Textbook of Soil Chemical Analysis. Chemical Publishing Co., Inc., New York. 520 p.

Janda, R. J. 1971. An evaluation of procedures used in computing chemical denudation rates. Geol. Soc. Am. Bull. **82**:67–80.

Johnson, N. M., G. E. Likens, F. W. Bormann, and R. S. Pierce. 1968. Rate of chemical weathering of silicate minerals in New Hampshire. Geochim. Cosmochim. Acta **32**:531–545.

Johnson, N., G. Likens, F. H. Bormann, D. W. Fisher, and R. S. Pierce. 1969. A working model for the variation in stream water chemistry at the Hubbard Brook Experimental Forest, New Hampshire. Water Resources Research **5**:1353–1363.

Jorgensen, J. R., and C. S. Hodges, Jr. 1970. Microbial characteristics of a forest soil after twenty years of prescribed burning. Mycologia **62**:721–726.

Kline, J., and C. F. Jordan. 1970. Tritium movement in soils of a tropical rain forest, p. 129–131. *In* H. T. Odum and R. Pigeon [eds.] A tropical rain forest. U.S. Atomic Energy Commission.

Knight, H. 1966. Loss of nitrogen from the forest floor by burning. For. Chron. **42**:149–152.

Lewis, W. M. 1974. Effects of fire on nutrient movement on a South Carolina pine forest. Ecology **55**: 1120–1127.

Neal, J. L., E. Wright, and W. B. Bollen. 1965. Burning Douglas-fir slash: physical, chemical, and microbial effects in the soil. For. Res. Lab., Oregon State Univ., Corvallis, Res. Pap. **1**.

Norum, R. A. 1976. *In press.* Fuel reduction relationships associated with understory burning in Larch/Douglas-fir stands. Tall Timbers No. 14.

Ralston, C. W., and G. E. Hatchell. 1971. Effects of prescribed burning on the physical properties of soil, p. 68–85. *In*: Prescribed burning symposium. U.S. Dep. Agric. For. Serv. Southeastern For. Expt. Stn., Ashville, North Carolina, USA.

Smith, D. W. 1970. Concentrations of soil nutrients before and after fire. Canadian J. Soil Sci. **50**:18–28.

Spurr, S. H., and B. V. Barnes. 1973. Forest ecology. Ronald Press, New York. 571 p.

Stark, N. 1969. Nutrient cycling in the Amazon Basin, p. 172–177. *In* J. M. Idrobo [ed.] II Simposio y foro de Biologia Tropical Amazonica. Colombia, South America.

———. 1972. Nutrient cycling pathways and litter fungi. BioScience **22**:355–360.

Steele, R. W. 1975. Weather data summary, 1956–1973, for Lubrecht Experimental Forest. Montana For. Conserv. Exp. Stn. Misc. Pap. No. **7**.

Steele, R. W. 1976. *In press.* Smoke movement associated with understory burning in a Larch/Douglas-fir stand. Tall Timbers No. **14**.

Striffler, W. D., and E. W. Mogren. 1971. Erosion, soil properties and revegetation following a severe burn in the Colorado Rockies, p. 25–36. *In* C. W. Slaughter, R. J. Barney, and G. M. Hausen [eds.] Fire in the Northern Environment—A Symposium. Pacific Northwest Forest and Experimental Station. U.S. For. Serv., Portland, Oregon.

Vogl, R. J., and C. Ryder. 1969. Effects of slash burning on conifer reproduction in Montana's Mission Range. Northwest Sci. **43**:135–147.

Wright, E., and R. F. Tarrant. 1957. Microbiological soil properties after logging and slash burning. U.S. Forest Service. P.N.W.F. and R.E.S. Res. Note **157**.