

Logging of mature Douglas-fir in western Oregon has little effect on nutrient output budgets

C. WAYNE MARTIN

*Northeastern Forest Experiment Station, Forestry Sciences Laboratory, USDA Forest Service,
Durham, NH 03824, U.S.A.*

AND

R. DENNIS HARR

*Pacific Northwest Research Station, Forestry Sciences Laboratory, USDA Forest Service,
Corvallis, OR 97331, U.S.A.*

Received May 18, 1988

Accepted September 9, 1988

MARTIN, C. W., and HARR, R. D. 1989. Logging of mature Douglas-fir in western Oregon has little effect on nutrient output budgets. *Can. J. For. Res.* **19**: 35-43.

Precipitation and streamwater quantity have been sampled continuously on three adjacent watersheds since 1964 at the H. J. Andrews Experimental Forest on the western slopes of the Cascade Mountains of Oregon. The chemistry of streamwater has been sampled since 1972, and the chemistry of precipitation since 1973. These watersheds were covered primarily by a mature 130-year-old forest of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). In 1974, one of the watersheds was clear-cut, the logs being removed primarily by a high-lead cable system. At the same time, 60% of the basal area was removed in a shelterwood cut from the adjoining watershed. The third watershed remained uncut for reference. Both harvested watersheds were broadcast burned in 1975 and planted to Douglas-fir in 1976. Streamwater chemistry data and input-output budgets for the three watersheds are presented for 10 years after cutting. Nitrate nitrogen was the only measured solute affected by logging. Despite a 30-fold increase, nitrate nitrogen concentrations were so low that more than twice as much nitrate nitrogen was added to the clearcut from precipitation as was lost in streamwater.

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La quantité de précipitations et d'eau d'écoulement a été mesurée de façon continue dans trois bassins hydrographiques adjacents depuis 1964 à la Forêt de démonstration H. J. Andrews située sur le versant occidental des Monts Cascade en Orégon. Les propriétés chimiques de l'eau d'écoulement ont été mesurées depuis 1972 et celles de la précipitation depuis 1973. Ces bassins étaient couverts principalement par une vieille forêt âgée de 130 ans de Sapin de Douglas (*Pseudotsuga menziesii* (Mirb. Franco). En 1974, un des bassins fut coupé à blanc, les billes de bois ayant été récupérées principalement au moyen d'un système de câbles aériens. Au même moment, on a prélevé 60% de la surface terrière lors d'une coupe progressive dans le bassin voisin. Le troisième bassin est demeuré intact comme témoin. En 1975, les deux bassins ayant subi des coupes ont été brûlés à la volée, et ils furent par la suite plantés avec du Sapin de Douglas en 1976. Les données chimiques concernant l'eau d'écoulement et les budgets intrant-extrant pour les trois bassins sont présentés pour les 10 années suivant la coupe. L'azote sous forme de nitrate a constitué le seul soluté affecté par la récolte des arbres. En dépit d'une hausse de 30 fois, les concentrations de l'azote sous forme de nitrate étaient si faibles que plus de deux fois plus de l'azote sous forme de nitrate furent ajoutés à la coupe par les précipitations que ce qui fut perdu par l'eau d'écoulement.

[Traduit par la revue]

Introduction

Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests west of the Cascade Mountains of Oregon and Washington are among the most productive forests of the world. These forests cover nearly 7 million hectares. Even-aged management initiated by clear-cutting or shelterwood harvesting followed by broadcast burning and planting is the recommended silvicultural system for the forest type (Williamson and Twombly 1983).

In the late 1960s and early 1970s, research, particularly in the northeastern United States, indicated that deforestation resulted in substantial losses of plant nutrients through leaching to streamwater (Likens *et al.* 1969, 1970). Shortly thereafter, studies in Douglas-fir and red alder (*Alnus rubra* Bong.) forests of the Oregon Coast Range indicated that clear-cutting followed by slash burning increased nutrient losses from these watersheds (Brown *et al.* 1973). Information from the State of Washington indicated that growth of young Douglas-fir plantations was often limited by inadequate soil nitrogen (Heilman and Gessel 1963). As a result, several studies were initiated in the Pacific Northwest to

assess the effect of clear-cutting and slash burning on the cycling of essential plant nutrients (Cromack *et al.* 1979; Fredriksen *et al.* 1975; Gessel and Cole 1965; Kimmins 1977; Sollins *et al.* 1981; Sollins and McCorison 1981).

The present study was initiated in 1972 on three adjacent calibrated watersheds on the H. J. Andrews Experimental Forest. The purpose was to assess changes in stream chemistry and in the input-output budgets of plant nutrients caused by clear-cutting and shelterwood harvesting of mature Douglas-fir forests.

The study area

Watershed characteristics

The study area, located in the H. J. Andrews Experimental Forest, 72 km east of Eugene, Oregon, consists of three small watersheds: watershed 6 (WS-6) is 13.0 ha; watershed 7 (WS-7) is 15.4 ha; and watershed 8 (WS-8) is 21.4 ha. The watersheds are located between 863 and 1190 m elevation (Table 1). Sideslope gradients generally range from 20 to 40%.

The experimental watersheds are underlain by two units of the Sardine Formation; one contains welded and nonwelded ash flows, and the other, basalt and andesite lava flows (Swanson and James

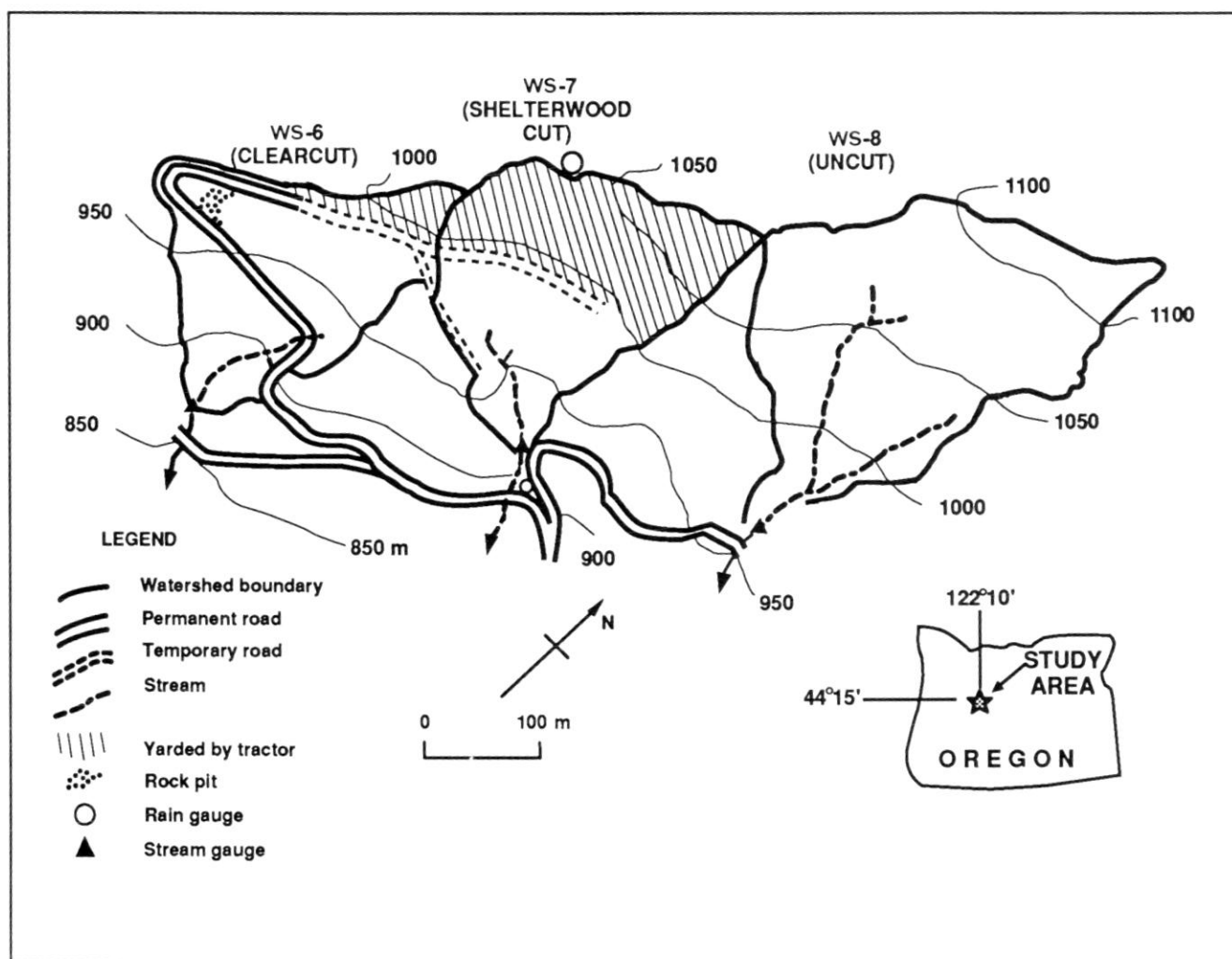


FIG. 1. Map of watersheds 6, 7, and 8 at the H. J. Andrews Experimental Forest, Oregon.

TABLE 1. Summary of characteristics and harvesting activities, watersheds 6, 7, and 8

	WS-6	WS-7	WS-8
Area, ha	13.0	15.4	21.4
Elevation, m	863-1013	908-1097	955-1190
Aspect	S	SSE	SSE
Roads, ha	1.2	0	0
Type of cut	Clear-cut	Shelterwood	Uncut
Area logged, ha	13.0	15.4 ^a	0
% logged	100	100 ^a	0
By tractor	10	60	0
By high lead	90	40	0
% burned	100 ^b	50 ^b	Unburned

NOTE: Watersheds 6 and 7 were logged during the summer of 1974.

^aAbout 44 m²·ha⁻¹, or 60% of total basal area, was removed in a shelterwood cut throughout the watershed.

^bLogging residue was burned in the spring of 1975.

1975). The soils on 86% of the area are Typic Haplorthods, while the soil of the remaining 14% is an Andic Cryumbrept. The surface soils are sandy or silty loams 90-110 cm deep, while the subsoils are loams or clay loams from 1-3 m deep.

Forest stands in WS-6 and WS-7 were composed mainly of 30-year-old Douglas-fir with scattered old-growth about 450 years old (Franklin and Dyrness 1973). WS-8 contained a higher proportion of old-growth Douglas-fir than either WS-6 or WS-7 before logging. Primary understory vegetation included vine maple (*Acer*

circinatum Pursh), rhododendron (*Rhododendron macrophyllum* D. Don ex G. Don), and salal (*Gaultheria shallon* Pursh). Ten years after clear-cutting, planted Douglas-fir trees were beginning to overtop the 2 m tall snowbrush (*Ceanothus velutinus* Dougl. ex Hook.) shrubs that became established after the broadcast burning of logging residue.

Climate and streamflow

The maritime climate of the study area is influenced greatly by the Pacific Ocean 160 km to the west. Mean January and July temperatures were 2 and 11°C, respectively. Annual precipitation averages 219 cm, and ranged from 125 cm in 1977 to 290 cm in 1972 (Harr *et al.* 1982). About 80% of annual precipitation occurs in the October-March period, during low- to moderate-intensity frontal storms of relatively long duration (12-72 h). Most precipitation falls as rain, but snow is common. A snowpack generally develops in early December. In some years, pack depth may reach more than 225 cm and contain more than 85 cm of water equivalent.

Streamflow has been measured continuously with H-flumes since October 1963. During the 1964-1974 prelogging period, annual streamflow averaged 143 cm at WS-6, 121 cm at WS-7, and 135 cm at WS-8, the unlogged watershed.

Watershed treatments

A permanent all-weather road occupying 1.2 ha of the watershed area was constructed through WS-6 in 1964. Clear-cutting began in May 1974 and was completed in August 1974. Logs in 90% of the watershed were yarded uphill by a high-lead cable system; logs in the 10% at the ridge top were yarded downhill by

TABLE 2. Mean annual specific conductance, pH, alkalinity, and concentrations of dissolved silica and suspended sediment in streamwater from three watersheds at the H. J. Andrews Experimental Forest (n = years of observation)

	Reference		Clearcut		Shelterwood	
	Mean	SE	Mean	SE	Mean	SE
Specific conductance, (μ S)						
Preharvest ($n=0$)	—	—	—	—	—	—
Postharvest ($n=7$)	32	0.8	35	1.0	36	1.2
pH						
Preharvest ($n=2$)	7.3	0.1	7.3	0.1	7.3	0.1
Postharvest ($n=9$)	7.3	0.1	7.3	0.1	7.3	0.1
Alkalinity (HCO_3^-), $\text{mg}\cdot\text{L}^{-1}$						
Preharvest ($n=3$)	18	1.1	23	1.1	20	1.3
Postharvest ($n=10$)	20	0.8	22	0.6	23	0.8
Dissolved silica (SiO_2), $\text{mg}\cdot\text{L}^{-1}$						
Preharvest ($n=3$)	9.03	1.46	10.61	1.20	9.80	1.07
Postharvest ($n=10$)	12.78	0.98	13.89	0.86	15.15	1.11
Suspended sediment, $\text{mg}\cdot\text{L}^{-1}$						
Preharvest ($n=3$)	7.94	4.56	4.19	1.65	2.13	0.50
Postharvest ($n=10$)	8.20	1.68	4.24	1.37	2.14	0.46

tractor (Fig. 1). Logging residue was broadcast burned in the spring of 1975, and the watershed was planted with Douglas-fir seedlings in the spring of 1976.

In WS-7, 60% of the total basal area was removed in a shelterwood cut during the summer of 1974. Logs in the upper 60% of the watershed were yarded by tractor. Logs in the lower 40% were yarded partially suspended by a skyline cable system. Logging residue was broadcast burned only on the lower half of the watershed in the spring of 1975. WS-8 remains undisturbed.

Methods

Samples of precipitation for chemical analyses were collected in large openings in the forest. Stainless-steel funnels caught the precipitation, which passed through plastic tubing to a polyethylene carboy washed with 0.5 M HCl and thoroughly rinsed with deionized water. The carboy was in an underground chamber to keep the sample cool. Precipitation samples were 3-week composite samples of bulk precipitation.

Streamwater samples were collected in clean, acid-washed polyethylene carboys by a proportional water sampler located at the stream gage (Fredriksen 1969). The discrete samples were composited into one sample which was collected every 3 weeks and transported to the laboratory.

The pH was measured potentiometrically at room temperature, converted to H^+ for all data analyses, and subsequently reconverted to pH. Bicarbonate (HCO_3^-) was determined at room temperature by potentiometric titration to pH 4.5 with 0.01 M H_2SO_4 and reported as HCO_3^- or $\text{HCO}_3\text{-C}$. Specific conductance was measured on a Wheatstone bridge at room temperature, corrected to 25°C, and reported in microsiemens.

Nitrate nitrogen ($\text{NO}_3\text{-N}$) was measured by the cadmium reduction method, manually from 1972 through 1978, with color measured by spectrophotometer, and automatically since 1978 on a Technicon Autoanalyzer¹ (Armstrong *et al.* 1967; Franson 1980).

Dissolved organic nitrogen plus ammonium nitrogen ($\text{NH}_4\text{-N}$) was determined by Nesslerization following macro-Kjeldal digestion of filtered samples. Particulate nitrogen was estimated by the

difference between the macro-Kjeldahl digestion and Nesslerization of filtered and unfiltered samples (Franson 1980). Filtered samples were passed through glass-fibre Whatman GF/C filters through 1983 and through GF/F filters since then.

Phosphate phosphorus ($\text{PO}_4\text{-P}$) was determined by the ascorbic acid method. Dissolved organic phosphorus was determined by the ascorbic acid method after persulfate digestion of unfiltered samples (Franson 1980). Particulate phosphorus was estimated by the difference between the digestion of unfiltered and filtered samples.

Calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^+), and potassium (K^+) were determined by atomic absorption spectroscopy. A Jerrel-Ash atomic absorption spectrophotometer was used from 1972 to 1975. From 1975 to 1980, a Varian 1200 model was used, and from 1980 through 1985, a Perkin-Elmer 5000 model.

Dissolved silica (SiO_2) was measured manually by the molybdo-silicate or molybdate yellow method through 1978; since 1978, the automated molybdate blue method has been used (Franson 1980). Suspended sediment was measured by filtration through washed and weighed GF/C or GF/F filter papers. The filters were then dried at 70°C for 48 h, cooled in a desiccator, and weighed.

Concentration data have been volume weighted by multiplying the concentrations of individual samples by the volume of precipitation or streamflow represented by the sample, and then dividing by the total volume of precipitation or streamflow for the water year. Input-output budget data were calculated by multiplying the mean annual concentration by the volume of precipitation or streamwater for the water year. The water year was from October 1 to September 30.

Results

Specific conductance

The specific conductance of precipitation at the H. J. Andrews Experimental Forest averaged only 5 μ S. After precipitation had passed through the canopy and soils, mixed with soil moisture and groundwater, and become streamflow, its chemistry had changed considerably. The specific conductance of the stream in the reference watershed averaged 32 μ S over the 7 years of record (Table 2). Clear-cutting and shelterwood harvesting, with subsequent slash burning, did not cause major changes in streamwater conductance.

¹The use of trade, company, or corporation names in this paper is for the information and convenience of the reader. Such use does not constitute an official endorsement or approval by the U.S. Department of Agriculture or the Forest Service of any product or service to the exclusion of others that may be suitable.

TABLE 3. Annual inputs ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$) of nutrients dissolved in precipitation at the H. J. Andrews Experimental Forest

	No. of years of observation	Input	
		Mean	SE
Bicarbonate carbon	12	8.5	0.70
Nitrate nitrogen	12	0.7	0.07
Particulate nitrogen	6	0.4	0.03
Dissolved organic N plus $\text{NH}_4\text{-N}$	12	1.0	0.05
Orthophosphate phosphorus	9	0.1	0.01
Particulate phosphorus	9	0.1	0.01
Dissolved organic phosphorus	9	0.1	0.01
Calcium	10	3.7	0.68
Magnesium	12	0.9	0.10
Sodium	12	5.1	0.56
Potassium	12	2.0	0.31
Dissolved silica	12	5.4	1.43
Suspended sediment	12	32.2	3.33

pH

Mean annual pH of precipitation at the three watersheds from 1973 through 1984 averaged 5.6, with a range of 4.7–6.0. Mean annual streamwater pH at all three watersheds was 7.3 ± 0.1 throughout the study. This neutralization of precipitation acidity was due to cation exchange as the water passed through the forest canopy and the soil on its way to becoming streamflow. Clear-cutting, shelterwood harvesting, and slash burning did not affect the mean annual pH of the streams (Table 2).

Alkalinity

Bicarbonate was by far the predominant anion in both precipitation and streamwater. The mean annual HCO_3^- concentration in precipitation was $2.0 \text{ mg} \cdot \text{L}^{-1}$ over the 12-year period, with little variation from year to year. These concentrations resulted in average annual additions of $8.5 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ of $\text{HCO}_3\text{-C}$ to the watersheds (Table 3).

Although the three watersheds were contiguous, there were some differences in chemistry between watersheds both before and after cutting. Bicarbonate concentration was higher in the streams from the treated watersheds than from the reference watersheds, both before and after harvesting (Table 2). Clear-cutting, shelterwood harvest, and slash burning did not affect HCO_3^- concentrations in streams (Table 2).

Streamflow was consistently higher from the clear-cut watershed, both before and after cutting (Harr *et al.* 1982). As a result, HCO_3^- loss was 1.56 times greater from the clear-cut watershed than from the reference watershed before harvesting; this ratio did not change after cutting (Table 4).

Nitrogen

Mean annual $\text{NO}_3\text{-N}$ concentrations in precipitation averaged $0.030 \pm 0.002 \text{ mg} \cdot \text{L}^{-1}$ over the period of record. Nitrate was the only ion in streamwater that responded to harvesting (Table 5). For the 3 years before harvest, $\text{NO}_3\text{-N}$ averaged $0.001 \text{ mg} \cdot \text{L}^{-1}$ at all three streams. For the first 3 years after harvest, the reference stream averaged $0.004 \text{ mg} \cdot \text{L}^{-1}$, and the clearcut averaged $0.036 \pm 0.006 \text{ mg} \cdot \text{L}^{-1}$. By the 6th year after clear-cutting, $\text{NO}_3\text{-N}$ concentrations in the WS-6 stream were back to reference

levels. The $\text{NO}_3\text{-N}$ concentrations in the stream from the shelterwood cutting increased to an average of $0.051 \text{ mg} \cdot \text{L}^{-1}$ the 1st year after cutting and dropped to $0.009 \text{ mg} \cdot \text{L}^{-1}$ the 2nd year. While $\text{NO}_3\text{-N}$ concentrations increased an order of magnitude after cutting on both watersheds, the absolute values were so low (Table 5) that even with the 30% increase in flow, the $\text{NO}_3\text{-N}$ discharge averaged $0.3 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ over the 10 years after cutting (Table 4). Input of $\text{NO}_3\text{-N}$ by precipitation averaged $0.7 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ for the same period (Table 3) for a net gain to the system after clear-cutting.

Harvesting and burning had little effect on particulate N or on dissolved organic N plus $\text{NH}_4\text{-N}$. Loss of fine particulate N averaged $\leq 0.2 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ for the 6 postharvest years from all three watersheds (Table 4). Discharge of dissolved organic N plus $\text{NH}_4\text{-N}$ averaged $0.4\text{--}0.7 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ over the 3 years before harvesting, and averaged $0.3\text{--}0.6 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ after harvest (Table 4). These differences were well within the standard errors of estimate. Dissolved organic N plus $\text{NH}_4\text{-N}$ discharge from the clear-cut watershed was consistently lower than from the reference before harvest, and consistently higher after harvest because of increased streamflow after harvest.

Additions of nitrogen, as particulate N, dissolved organic N, $\text{NH}_4\text{-N}$, and $\text{NO}_3\text{-N}$, to these watersheds by precipitation averaged $2.1 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ (Table 3). In the postharvest period, N losses from the reference watershed averaged $0.7 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$, the clearcut averaged $1.1 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$, and the shelterwood averaged $0.5 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ (Table 4). The highest loss occurred from the clear-cut watershed the 2nd year after cutting; that year the N budget was balanced at $1.5 \text{ kg} \cdot \text{ha}^{-1}$.

Phosphorus

In precipitation, $\text{PO}_4\text{-P}$ averaged $0.004 \text{ mg} \cdot \text{L}^{-1}$, with a range of near 0 to $0.013 \text{ mg} \cdot \text{L}^{-1}$. Deposition of $\text{PO}_4\text{-P}$ over the 9-year period of record averaged $0.1 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ (Table 3). In streamwater, $\text{PO}_4\text{-P}$ from the reference watershed averaged $0.022 \text{ mg} \cdot \text{L}^{-1}$ over the period of the study, with no apparent seasonal change and no apparent trend over time (Table 5). This resulted in an average discharge of $0.27 \pm 0.03 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ of $\text{PO}_4\text{-P}$ from the reference watershed for the 10 years (Table 4). Concentrations of $\text{PO}_4\text{-P}$ in the stream from the shelterwood cut were similar to those from the reference watershed before and after cutting (Table 5). Concentrations in the stream from the clearcut were 40% less than from the reference watershed before and after harvest. While there were consistent differences between streams, harvesting did not produce changes larger than the standard errors (Tables 4 and 5).

The average annual deposition of phosphorus as the sum of particulate and dissolved organic phosphorus and $\text{PO}_4\text{-P}$ was $0.3 \text{ kg} \cdot \text{ha}^{-1}$ (Table 3). The average annual loss from the reference watershed was $0.6 \text{ kg} \cdot \text{ha}^{-1}$ for a net loss of $0.3 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$. For the same postharvest years, the net loss from the clear-cut watershed was also $0.3 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$, while the shelterwood lost $0.2 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ (Table 4).

Cations

In precipitation, Ca^{2+} concentrations averaged $0.18 \text{ mg} \cdot \text{L}^{-1}$, with little year to year variation. Deposition of Ca^{2+}

TABLE 4. Mean annual outputs ($\text{kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$) of carbon, nitrogen, and phosphorus dissolved in streamwater from three watersheds at the H. J. Andrews Experimental Forest (n = years of observation)

	Reference		Clearcut		Shelterwood	
	Mean	SE	Mean	SE	Mean	SE
Bicarbonate carbon						
Preharvest ($n=3$)	55.0	16.3	85.9	28.33	49.6	13.20
Postharvest ($n=10$)	47.1	3.7	75.6	5.20	49.3	3.30
Nitrate nitrogen						
Preharvest ($n=3$)	0.01	0.0	0.01	0.00	0.01	0.00
Postharvest ($n=9$)	0.04	0.0	0.31	0.08	0.08	0.02
Particulate nitrogen						
Preharvest ($n=0$)	—	—	—	—	—	—
Postharvest ($n=6$)	0.2	0.04	0.2	0.04	0.1	0.01
Dissolved organic N plus $\text{NH}_4\text{-N}$						
Preharvest ($n=3$)	0.7	0.3	0.5	0.18	0.4	0.10
Postharvest ($n=10$)	0.5	0.1	0.6	0.07	0.3	0.05
Orthophosphate phosphorus						
Preharvest ($n=3$)	0.33	0.10	0.33	0.13	0.28	0.08
Postharvest ($n=7$)	0.24	0.03	0.23	0.03	0.23	0.03
Particulate phosphorus						
Preharvest ($n=0$)	—	—	—	—	—	—
Postharvest ($n=9$)	0.1	0.01	0.11	0.02	0.08	0.01
Dissolved organic phosphorus						
Preharvest ($n=3$)	0.32	0.07	0.36	0.10	0.32	0.06
Postharvest ($n=7$)	0.18	0.03	0.26	0.04	0.18	0.03

TABLE 5. Mean annual nitrogen and phosphorus concentrations ($\text{mg}\cdot\text{L}^{-1}$) in streamwater from three watersheds at the H. J. Andrews Experimental Forest (n = years of observation)

	Reference		Clearcut		Shelterwood	
	Mean	SE	Mean	SE	Mean	SE
Nitrate nitrogen						
Preharvest ($n=3$)	0.001	0.0003	0.001	0.001	0.001	0.0006
Postharvest ($n=9$)	0.003	0.0007	0.020	0.005	0.006	0.002
Particulate nitrogen						
Preharvest ($n=0$)	—	—	—	—	—	—
Postharvest ($n=7$)	0.003	0.003	0.005	0.002	0.002	0.002
Dissolved organic N plus $\text{NH}_4\text{-N}$						
Preharvest ($n=3$)	0.046	0.009	0.026	0.004	0.029	0.004
Postharvest ($n=10$)	0.039	0.005	0.034	0.003	0.029	0.003
Orthophosphate						
Preharvest ($n=2$)	0.021	0.002	0.016	0.003	0.022	0.002
Postharvest ($n=7$)	0.022	0.001	0.014	0.001	0.022	0.001
Particulate phosphorus						
Preharvest ($n=0$)	—	—	—	—	—	—
Postharvest ($n=10$)	0.010	0.002	0.009	0.003	0.009	0.003
Dissolved organic phosphorus						
Preharvest ($n=3$)	0.022	0.005	0.018	0.000	0.026	0.006
Postharvest ($n=10$)	0.017	0.001	0.016	0.002	0.017	0.002

averaged $3.7 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$ (Table 3) over the 10 years of record, with a range of $1.2\text{--}7.3 \text{ kg}\cdot\text{ha}^{-1}$. Streamwater from the shelterwood and clear-cut units was $0.3\text{--}0.6 \text{ mg}\cdot\text{L}^{-1}$ richer in Ca^{2+} than that from the reference watershed before cutting, and $0.5 \text{ mg}\cdot\text{L}^{-1}$ richer after cutting (Table 6). Losses of Ca^{2+} from the reference watershed averaged $36 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$, with a range of $13.8\text{--}59.0 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$ (Table 7). This represents a net loss of $32.3 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$ of Ca^{2+} from the ecosystem, or 10 times the input from precipitation. While

the three watersheds were contiguous, Ca^{2+} losses in streamwater varied considerably. Losses of Ca^{2+} from the shelterwood cut were similar to those from the reference watershed both before and after cutting; however, Ca^{2+} losses from the clearcut were 64% greater than losses from the reference watershed before cutting and 61% greater after cutting. In both harvested watersheds, there were no increases in Ca^{2+} losses following cutting.

Concentrations of Mg^{2+} in precipitation averaged $0.04 \text{ mg}\cdot\text{L}^{-1}$ over the course of the study. As a result,

TABLE 6. Mean annual cation concentrations ($\text{mg}\cdot\text{L}^{-1}$) in streamwater from three watersheds at the H. J. Andrews Experimental Forest (n = years of observation)

	Reference		Clearcut		Shelterwood	
	Mean	SE	Mean	SE	Mean	SE
Calcium						
Preharvest ($n=3$)	2.40	0.21	3.05	0.44	2.70	0.13
Postharvest ($n=8$)	3.04	0.14	3.54	0.14	3.58	0.19
Magnesium						
Preharvest ($n=3$)	0.64	0.08	0.86	0.10	0.80	0.01
Postharvest ($n=10$)	0.57	0.02	0.89	0.01	0.93	0.02
Sodium						
Preharvest ($n=3$)	2.30	0.05	1.80	0.16	1.76	0.05
Postharvest ($n=10$)	2.51	0.06	1.90	0.04	2.01	0.03
Potassium						
Preharvest ($n=3$)	0.36	0.16	0.29	0.15	0.41	0.15
Postharvest ($n=10$)	0.43	0.02	0.40	0.01	0.56	0.02

TABLE 7. Mean annual outputs ($\text{kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$) of cations, silica, and suspended sediment in streamwater from three watersheds at the H. J. Andrews Experimental Forest (n = years of observation)

	Reference		Clearcut		Shelterwood	
	Mean	SE	Mean	SE	Mean	SE
Calcium						
Preharvest ($n=3$)	39.7	13.4	62.2	24.3	35.9	11.2
Postharvest ($n=8$)	34.7	3.7	58.7	5.8	37.3	3.2
Magnesium						
Preharvest ($n=3$)	9.4	2.0	17.3	6.5	10.3	3.0
Postharvest ($n=10$)	6.9	0.7	15.7	1.3	10.3	0.8
Sodium						
Preharvest ($n=3$)	36.3	10.7	35.6	12.7	23.2	7.0
Postharvest ($n=10$)	30.5	2.7	33.4	2.9	22.4	1.9
Potassium						
Preharvest ($n=3$)	6.4	4.1	6.3	4.5	5.5	3.1
Postharvest ($n=10$)	5.2	0.4	6.9	0.5	6.1	0.4
Dissolved silica						
Preharvest ($n=3$)	151.9	54.4	211.2	75.8	134.0	45.4
Postharvest ($n=10$)	157.9	19.5	244.8	25.0	167.3	16.9
Suspended sediment						
Preharvest ($n=3$)	139	92	73	42	26	10
Postharvest ($n=10$)	111	25	79	28	26	6

Mg^{2+} deposition averaged $0.9 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$, with a range of $0.51\text{--}1.75 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$ (Table 3). Mean annual concentrations of Mg^{2+} from the reference stream averaged $0.6 \text{ mg}\cdot\text{L}^{-1}$ for the period of record. Mean annual volume-weighted output from the reference watershed averaged $7.7 \pm 0.7 \text{ kg}\cdot\text{ha}^{-1}$ for an average net loss of $6.8 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$ (Table 7). As with Ca^{2+} , there were differences in Mg^{2+} concentrations between watersheds before and after harvest. In the stream from the clearcut, Mg^{2+} averaged $0.9 \text{ mg}\cdot\text{L}^{-1}$ before and after cutting; Mg^{2+} in the stream from the shelterwood increased from 0.8 to $0.9 \text{ mg}\cdot\text{L}^{-1}$ with cutting. Strong seasonal cycles in Mg^{2+} concentration were also apparent in these streams. Before harvesting, Mg^{2+} losses were 186% higher from the clearcut than from the reference watershed (Table 7). Losses from the shelterwood cut before harvest were similar to those from the reference watershed. Harvesting had virtually no effect on Mg^{2+} loss from either watershed.

In precipitation, Na^+ concentrations averaged $0.24 \text{ mg}\cdot\text{L}^{-1}$ over the period of record, with a range of 0.11--

$0.43 \text{ mg}\cdot\text{L}^{-1}$. Deposition of Na^+ averaged $5.1 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$ (Table 3). Sodium was the second most abundant cation, with an average concentration of $2.5 \text{ mg}\cdot\text{L}^{-1}$ in the reference stream for the 13 years of record (Table 6). Annual means increased by 0.1 and $0.2 \text{ mg}\cdot\text{L}^{-1}$ after cutting at the clearcut and shelterwood cut, respectively; the reference watershed increased by $0.2 \text{ mg}\cdot\text{L}^{-1}$ for the same period. Output of Na^+ from the reference watershed averaged $32.7 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$, with a range of $11.9\text{--}49.1 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$ (Table 7). Mean annual Na^+ losses from the clear-cut watershed were similar to those from the reference watershed, and essentially did not change with harvest. Loss of Na^+ in WS-7 before the shelterwood cut was only 64% of the reference value, but was identical before and after cutting.

Concentrations of K^+ in precipitation averaged $0.09 \text{ mg}\cdot\text{L}^{-1}$, with a range of $0.04 \text{ mg}\cdot\text{L}^{-1}$ in 1975 to $0.23 \text{ mg}\cdot\text{L}^{-1}$ in 1979. Deposition of K^+ averaged $2.0 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$ (Table 3). Concentrations of K^+ in the reference stream averaged $0.4 \text{ mg}\cdot\text{L}^{-1}$, with little year to year

variation and little seasonal variation (Table 6). Output of K^+ from the reference watershed averaged $5.6 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$, with a range of $1.7\text{--}14.4 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$, for an average net loss of $3.6 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$. Loss of K^+ from the harvested watersheds was similar to that from the reference watershed (Table 7).

Dissolved silica

Concentrations of dissolved silica (SiO_2) in precipitation averaged $0.26 \text{ mg} \cdot \text{L}^{-1}$ over the period of record; the range of annual values was $0.06\text{--}0.88 \text{ mg} \cdot \text{L}^{-1}$. Deposition averaged $5.4 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ (Table 3).

Concentrations of SiO_2 in the reference stream averaged $12.0 \text{ mg} \cdot \text{L}^{-1}$ over the period of record (Table 2). There was, however, an increase in silica over time, from a low of $6.6 \text{ mg} \cdot \text{L}^{-1}$ in 1973 to $16.3 \text{ mg} \cdot \text{L}^{-1}$ in 1984. There was great variability among samples ($2.8\text{--}22.7 \text{ mg} \cdot \text{L}^{-1}$), but seasonal trends were not apparent.

Concentrations of SiO_2 did not increase because of harvesting (Table 2). Concentrations were higher after cutting on both treated watersheds, but this was a result of the continual increase in SiO_2 also reported for the reference watershed.

Losses of SiO_2 from the reference watershed averaged $156.2 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ (Table 7), with a range of $42.4\text{--}263.7 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$. There were differences in net loss between the three watersheds but the differences before and after cutting were within the standard errors and were influenced by the increase in concentration over time.

Particulates

Concentrations of dust suspended in precipitation averaged $1.6 \text{ mg} \cdot \text{L}^{-1}$, with a range of $0.9\text{--}2.9 \text{ mg} \cdot \text{L}^{-1}$. Annual deposition was $32.2 \pm 3.3 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ (Table 3).

Concentrations of suspended sediment in the reference stream averaged $8.1 \text{ mg} \cdot \text{L}^{-1}$ (Table 2), with a range of annual means of $0.9\text{--}16.9 \text{ mg} \cdot \text{L}^{-1}$ and individual values as high as $164.5 \text{ mg} \cdot \text{L}^{-1}$. Concentrations from the clear-cut watershed averaged $4.2 \text{ mg} \cdot \text{L}^{-1}$ before and after harvest. Average annual concentrations from the shelterwood cut were $2.1 \text{ mg} \cdot \text{L}^{-1}$ before and after logging.

Nearly 4 times as much suspended sediment left the reference watershed in streamflow as was deposited by precipitation. Only twice as much was lost from the clearcut as was deposited, and there was a net gain on the shelterwood watershed (Table 7).

Discussion

Logging, especially clear-cutting, sets in motion major changes in the forest environment. Removal of the forest canopy allows sunlight to reach the forest floor, which increases soil temperatures. Removing the trees reduces transpiration which, in turn, increases soil moisture and streamflow. Warmer, wetter soil conditions promote microbial decomposition of soil organic matter, particularly the forest floor. Faster decomposition may lead to increased rates of nitrogen mineralization to NH_4^+ , followed by increased nitrification, producing NO_3^- . Excess NO_3^- may be lost from the site by leaching through the soil to below the root zone and ultimately to the streams. Nitrification releases H^+ ions to cation exchange sites, which in turn accelerates leaching of other ions. These processes have been described as the reorganization phase of forest succession

following major natural or anthropogenic disturbances (Bormann and Likens 1979).

Elevated NO_3^- concentrations in soil solution and in streams following clear-cutting have been reported for many timber types throughout North America (Vitousek and Melillo 1979). The magnitude of nutrient losses following clear-cutting ranges from high concentrations and high net losses of NO_3^- and cations with acidification of streams in the northern hardwood forest of New Hampshire (Hornbeck *et al.* 1986; Martin *et al.* 1986) to modest increases in the coniferous forest of the Pacific Northwest (Feller and Kimmins 1984; Sollins and McCorison 1981).

Our data indicate that clear-cutting and shelterwood cutting of a mature 130-year-old Douglas-fir forest at the H. J. Andrews Experimental Forest resulted in only minor changes in stream chemistry. Concentrations of NO_3^- increased substantially over preharvest levels. Annual NO_3^- losses (in kilograms per hectare) in streamwater from the clearcut increased by an order of magnitude after cutting. But the absolute values were miniscule and remained less than precipitation input. The pH of the streams remained at 7.3. The concentrations and net losses of $\text{PO}_4\text{-P}$ and cations to streamwater were not increased by cutting.

In a study of nitrogen and carbon changes in soil solutions following cutting of an old-growth Douglas-fir forest, also at the H. J. Andrews Experimental Forest, Sollins and McCorison (1981) reported similar results. Concentrations of NO_3^- during the first 2 years after cutting increased as much as 100-fold, but the total NO_3^- leached to streamflow remained $\leq 1.2 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$. The H^+ contributed to the weathering of the cations, but the contribution was miniscule compared with normal weathering. Any increase was not detectable, given the normal level of variability.

Why did clear-cutting followed by slash burning of the mature Douglas-fir forest at the H. J. Andrews Experimental Forest, events that dramatically changed the forest and soil environment, result in so little change in the input-output budgets? The high C:N ratio of the forest floor organic matter could cause NH_4^+ released during decomposition to be immobilized by the decomposing organisms, and thus it would not be available for nitrification (Vitousek *et al.* 1979). Ratios of C:N must be below 25–35 for N to be released into solution (Carlyle 1986). In the top 15 cm of soil, C:N ratios below the litter layer in five old-growth stands at the H. J. Andrews Forest averaged 18.6–25.4 (McNabb *et al.* 1986). However, C:N ratios as high as 100 have been reported in the forest floor of these Douglas-fir forests (Sollins *et al.* 1980). Large quantities of decomposing wood, such as logging slash, stumps, roots, and old logs, can become N sinks because of their C:N ratios, which may be as high as 500 (Sollins *et al.* 1980; Vitousek and Melillo 1979). Despite the slash burning, our watershed contained large amounts of woody debris during the recovery phase, possibly as much as $200 \text{ Mg} \cdot \text{ha}^{-1}$ in fallen logs alone (Sollins *et al.* 1980).

Ammonium can be fixed in clay lattices or may be adsorbed on cation-exchange sites of clay-rich soils (Vitousek and Melillo 1979). As the clay content of the soils in WS-6 and WS-7 is about 5% (Brown 1975), fixation and adsorption are not likely to be much of a factor in the immobilization of NH_4^+ .

In basic, neutral, or slightly acidic soils, NH_3 may be volatilized (Vitousek *et al.* 1982). Soils in WS-6 and WS-7 have a pH of 5.2–5.7 (Brown 1975), which would seem to preclude significant volatilization of NH_3 . However, volatilization may be a factor in the riparian zone because streamwater has a pH of 7.3.

Both NH_4^+ and NO_3^- may be taken up by regrowing vegetation on the clear-cut sites (Vitousek *et al.* 1979). This is certainly a factor 3–5 years after cutting, but probably of little importance initially because of the slow rate of revegetation of these Douglas-fir sites (Harr *et al.* 1982). Another possibility is that mineralization and nitrification do occur, but because of the long summer drought common in this region, there is insufficient precipitation to leach the NO_3^- from the deep soils to the streams (Vitousek and Melillo 1979).

These Douglas-fir sites of the western Cascades of Oregon seem to resist substantial losses of plant nutrients following clear-cutting, because of the high C:N ratios of the forest floor, the annual summer droughts which delay leaching, and the deep soils which provide an opportunity for substantial storage of nutrients, especially on cation-exchange sites. These conclusions are confirmed by the results of a nearby study of an old-growth Douglas-fir site, also at the H. J. Andrews Experimental Forest (Sollins and McCorison 1981). Data from their lysimeter nests indicated substantial NO_3^- concentrations only at the 2.0-m depth, where the C:N ratio of soil organic matter was low. They also reported the highest concentrations in the autumn, at the end of the summer drought.

Few comparative data exist for the Pacific Northwest. One comprehensive study in British Columbia (Feller and Kimmins 1984) compared the impacts of clear-cutting with those of clear-cutting followed by slash burning on input-output budgets. Their study sites were low-elevation watersheds forested with 70- to 90-year-old western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), western red cedar (*Thuja plicata* Donn), and Douglas-fir growing on shallow Typic Haplorthods derived from igneous rocks. Reported changes in streamwater chemistry resulting from clear-cutting and slash burning were similar to ours. Concentrations of NO_3^- increased by an order of magnitude the 1st year after treatment, but dropped below pretreatment levels by the 3rd year. They also reported a significant increase in Cl^- , and modest increases in Na^+ , Mg^{2+} , K^+ , H^+ , and conductivity for the 1st year after treatment. They reported net losses (in kilograms per hectare) as a result of clear-cutting only for NO_3^- , K^+ , and Cl^- , and only for the first 3 years after clear-cutting. Slash burning after cutting did not cause any greater losses than clear-cutting alone.

Conclusions

Clear-cutting and shelterwood harvesting followed by broadcast burning of slash in mature Douglas-fir forests of western Oregon did not increase net losses of nutrients from the sites of streamwater. Nitrate nitrogen was the only constituent to increase after harvesting. Despite a 30-fold increase in concentration, losses in streamwater were less than half the input from precipitation. These results are comparable with previously published results from the clear-cutting of an old-growth Douglas-fir site, also at the H. J. Andrews Experimental Forest. They also parallel results from a study of clear-cutting and burning a mixed

coniferous forest growing on shallow soils derived from igneous rocks in British Columbia. Douglas-fir forests growing on deep, fertile soils derived from volcanic rocks respond to clear-cutting very differently from deciduous forests of the eastern United States growing on shallow, acidic, infertile soils.

Acknowledgments

This study was initiated by R. L. Fredriksen. We gratefully acknowledge fieldwork by A. Levno, R. Mersereau, G. Lienkaemper, and C. Creel, and data management by D. Henshaw and T. Larrabee. Water chemistry analyses were performed by E. Holcombe and C. Jones under Supplement Nos. PNW-82-321 and PNW-83-341 to the Master Memorandum of Understanding between the USDA Forest Service and Oregon State University. Financial support for field facilities and data collection, management, and analysis was also provided through grants from the National Science Foundation (DEB 7611978, DEB 7925939, DEB 8012162, and BSR 8300370) and Oregon State University.

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