

AN ABSTRACT OF THE THESIS OF

James L. Clayton for the degree of Doctor of Philosophy in Soil Science presented on January 14, 1985.

Title: Weathering Rate, Nutrient Supply, and Denudation in Forested Watersheds, Southwestern Idaho Batholith

Redacted for Privacy

Abstract approved: _____

John E. Baham

Biogeochemical data from four small watersheds in the Silver Creek study area, southwestern Idaho, provide the basis for: 1. An estimate of chemical and erosional denudation rates (CD and ED), 2. an estimate of plagioclase weathering rate, 3. a rational basis for estimating the rate of nutrient supply to soil from mineral weathering, and 4. an evaluation of the impact of clearcutting on the nutrient budget in a coniferous forest. The watersheds are located in the Idaho batholith, and soils are formed on coarse-textured quartz monzonite parent material, are shallow, and exhibit minimal profile development. Multi-aged stands of ponderosa pine and Douglas-fir comprise the overstory. Contemporary denudation rates for four watersheds average $235 \text{ kg ha}^{-1} \text{ y}^{-1}$, equivalent to 8.9 mm per 10^3 years. Modal CD rates slightly exceed ED rates, although mean ED exceeded CD over the 11-year period of study. Percent rock outcropping in the watersheds is positively correlated and volume of soil accumulated is negatively correlated with the probability that ED exceeds CD in any given year. Chemical weathering of plagioclase averages $267 \text{ kg ha}^{-1} \text{ y}^{-1}$. Preferential weathering of anorthite results in its selective depletion relative to albite based on net export of Na and Ca in streams. This is supported by data indicating relative enrichment of albite in the soil compared to fresh rock. Rates of cations released during chemical weathering exceed amounts retained in the ecosystem. The amounts retained represent a realistic estimate of potential supply rate from weathering. Plant uptake and biomass increment provides the largest sink for nutrients

released from weathering, thus time-variant growth dynamics strongly influence nutrient supply rates. Cations retained on new exchange sites formed during clay formation from weathering provide a sink for less than 2 percent of the cations retained. Under present stand conditions in Silver Creek, approximately half the K, Ca, and Mg released from weathering is retained in the ecosystem. Clearcut logging and slash burning caused small, but statistically significant increases in dissolved N losses of approximately ten times natural rates over a 4-year period. Dissolved transport of other elements was not increased. Small increases in sediment transport of K, Ca, Mg, N, S, and P occurred 3 years following harvest. Large losses resulting from bole removal ranged from 4 percent of total ecosystem N to 21 percent of available K. Rates of replacement from precipitation, primary mineral weathering, and N_2 fixation are sufficient to restore the ecosystem to the nutrient status prior to harvest in 50 years.

Weathering Rate, Nutrient Supply, and Denudation
in Forested Watersheds, Southwestern Idaho Batholith

by

James L. Clayton

A THESIS

submitted to

Oregon State University

in partial fulfillment of
the requirements for the
degree of

Doctor of Philosophy

Completed January 14, 1985

Commencement June 1985

APPROVED:

Redacted for Privacy

Assistant Professor of Soil Science in charge of major

Redacted for Privacy

Head of department of Soil Science

Redacted for Privacy

Dean of Graduate School

Date thesis is presented January 14, 1985

Typed by Gwladys Deiss for James L. Clayton

ACKNOWLEDGMENTS

I am grateful to E. Kennedy-Ketcheson, G. Ketcheson, W. Megahan and T. Mosko of the Boise Forestry Science Laboratory for assistance with streamflow, precipitation and sediment production data; L. Peterson of the University of Oregon who provided XRF data on rock samples; C. White, Boise State University who provided refractive indices of plagioclase grains; and D. Perry, Oregon State University for providing biomass and incremental growth data. I am grateful to D. Kennedy, Boise Forestry Sciences Laboratory, for chemical analyses and field assistance in all phases of this work. This thesis was substantially improved by manuscript reviews and helpful discussions with: P. Adams, J. Baham, R. Beschta, P. Birkeland, S. Colman, D. Dethier, J. Hornbeck, E. Jenne, W. Minshall, D. Perry, E. Stone, F. Swanson, C. Wells, M. Wilson, S. Wood and C. Youngberg. I thank John Baham and Chet Youngberg, Department of Soil Science, for their timely advice and direction in all phases of my graduate work.

This research was financed entirely by the U.S. Department of Agriculture, Forest Service.

TABLE OF CONTENTS

INTRODUCTION	1
EROSIONAL AND CHEMICAL DENUDATION IN THE SOUTHWESTERN IDAHO BATHOLITH	5
AN ESTIMATE OF PLAGIOCLASE WEATHERING RATE IN THE IDAHO BATHOLITH BASED UPON GEOCHEMICAL TRANSPORT RATES	30
A RATIONAL BASIS FOR ESTIMATING ELEMENTAL SUPPLY RATE FROM WEATHERING	48
THE NUTRIENT BUDGET IN A MIXED CONIFER FOREST IN THE IDAHO BATHOLITH - HARVEST IMPLICATIONS	67
BIBLIOGRAPHY	93

LIST OF FIGURES

<u>Figure</u>	<u>Page</u>
1.1. Location map of the study area.	9
1.2. Cumulative probability distributions of annual water yield.	15
1.3. Annual range in chemical and erosional denudation.	17
1.4. Chemical denudation as a function of water yield.	18
1.5. Erosional denudation as a function of water yield.	19
1.6. Graphical relation of CD, ED, and annual water yield showing threshold values.	20
1.7. Volume of soil and percent rock outcrop plotted vs. the probability that ED exceeds CD.	22
1.8. Frequency distribution for annual ED and CD, SC-5.	24
1.9. Frequency distribution for the ratio ED/CD, SC-5.	25
2.1. Location map of the study area.	33
2.2. Predominance diagrams for the systems $K-Al-Si-H_2O$ and $Na-Al-Si-H_2O$ at 25°C and 100 kPa.	39
2.3. Plots of annual Na and Ca efflux vs. annual water yield.	41
2.4. Preferential weathering of the calcic core of a plagioclase grain to kaolinite.	45
4.1. Map of the two study watersheds.	73
4.2. Plot of NO_3^- -N concentration vs. time before and after logging.	86
4.3. Regressions of springtime NO_3^- -N flux, SC-6 vs. SC-5 before and after logging.	87

LIST OF TABLES

<u>Table</u>	<u>Page</u>
1.1. Descriptive data for the study watersheds.	10
1.2. Mean chemical and erosional denudation rates.	16
1.3. Average composition of the chemical denudation component.	28
2.1. Descriptive data for the study watersheds.	34
2.2. Total Na and Ca analyses on A horizon, C horizon, and rock samples.	47
3.1. Average concentration of elements in rain and snow.	57
3.2. Annual precipitation influx and stream efflux of elements.	58
3.3. Standing crop, annual increment and mortality of timber stands.	59
3.4. Estimated feldspar weathering and kaolinite formation rates in watershed SC-5.	62
3.5. Estimated cation distribution adsorbed on kaolinite.	64
3.6. Weathering release of Na, K, Ca, Mg, and amount retained in the ecosystem.	65
4.1. Standing crop, annual net increment and mortality of timber stands.	81
4.2. A comparison of nutrient concentrations in green needles to recently abscised needles.	82
4.3. Nutrient budgets for six elements, Silver Creek, Idaho.	84
4.4. Elemental redistribution from slash additions and elemental losses from bole removal.	89

WEATHERING RATE, NUTRIENT SUPPLY, AND DENUDATION IN
FORESTED WATERSHEDS, SOUTHWESTERN IDAHO BATHOLITH

INTRODUCTION

Much of central Idaho is underlain by a large intrusion of granitic rock, the Idaho batholith, emplaced in the early and middle Cretaceous (Larsen and Schmidt, 1958). The Idaho batholith is part of a chain of large intrusive plutons extending along the western border of North America. Its surface outcroppings cover an area of approximately 41,000 km². Most of this is forested land with shallow, coarse-textured cohesionless soils that have proven to be extremely erodible. There are limited areas of moderate relief, but most of the topography consists of steep slopes separated by narrow ridges and valleys.

The Idaho batholith is also an area characterized by tremendously valuable resources. Rich in minerals, timber, and grazing, it undergirds much of the non-farm economy of Idaho and its neighboring states. Over one half million animal-months of sheep grazing and nearly one quarter million animal-months of cattle grazing take place within the batholith each year. Total timber volume is estimated at 60 billion board feet, and 90 sawmills are drawing timber from batholith lands. Streams in the Idaho batholith provide 55% of the spawning and rearing areas for salmon and steelhead passing McNary Dam on the Columbia River. Watershed values in the batholith are also very high. Annual water yield for the batholith is nearly 50 area cm. ($2.25 \times 10^{10} \text{ m}^3$) which is about 50 percent of the total water yield for the state.

Management of these high resource value lands is greatly complicated by the inherent erodibility of batholith soils and the potential for subsequent sediment damage to streams and fish. Concern over management policy peaked in the 1960's, and is best exemplified by a 13-year moratorium that was placed on all logging

and road construction activities in the 3370-km² South Fork of the Salmon River drainage after severe erosion and sedimentation problems in 1964 and 1965 decimated salmon spawning beds in this river. Management decisions have also been questioned because of a lack of knowledge concerning the potential for loss of on-site productivity due to soil loss accompanying erosion. Soils in the Idaho batholith are low in water holding capacity per unit depth of soil, have a relatively low cation exchange capacity because of low clay content, and are low in exchangeable bases. Surface erosion removes those horizons of the soil that are richest in the attributes that enhance productivity. Because logging activities and associated road construction have been shown to accelerate erosion in the Idaho batholith (Megahan and Kidd, 1972), the USDA Forest Service became concerned about logging activity and the potential for productivity loss. Central among these concerns were questions about accelerated nutrient loss to streams due to timber harvest activities, and the long-term consequences to ecosystem nutrient status from repeated entries.

Processes of weathering, denudation, and nutrient supply and loss are intimately linked, yet the available literature in these fields shows a distinct lack of synthesis. This is particularly the case if one desires information on nutrient supply rates from weathering. The geological/mineralogical and soils literature commonly contains recent papers on weathering processes and rates, and the geomorphological literature contains numerous papers on chemical denudation (Saunders and Young, 1983, summarized several dozen published in the last decade). Yet the estimates of nutrient supply from weathering of primary minerals are uncommon and often disputed. There are no currently accepted techniques for estimating supply from weathering, and separate techniques applied to the same area have resulted in published results differing by a factor of two (Likens et al., 1977).

Research needed to answer these land management concerns would require information on rates of nutrient transfer in natural and disturbed forest ecosystems, amounts of nutrients removed with timber harvest, and estimates of nutrient re-supply rates to harvested sites. Research was already underway in the Silver Creek Study Area, located in the southwestern Idaho batholith, to investigate changes in dissolved stream chemistry due to logging. It was logical to extend this study to include phytomass dynamics in order to provide the data needed for nutrient budgeting.

Several small watersheds were in calibration phase in the mid-1970s in Silver Creek in preparation for paired watershed studies to evaluate timber harvest and road construction effects. Four of these watersheds (SC-1, SC-2, SC-5, and SC-6) were selected to study natural erosional and chemical denudation rates; two were selected to study mineral weathering rates (SC-1 and SC-2); one was selected (SC-5) to construct a detailed nutrient budget emphasizing elemental supply from weathering; and one watershed (SC-6) was selected to evaluate timber harvest effects.

These particular studies were selected because they (1) fit in with the overall objectives of the Silver Creek Research Program; (2) provide information on the potential for productivity loss from timber harvest; and (3) the lithologically uniform and mineralogically simple parent material in this portion of the Idaho batholith is ideal for studying nutrient supply rates from weathering.

The need for reliable data on rates of nutrient supply to forest soils has intensified as a result of the trend toward more complete utilization of the tree crop, which greatly increases removal of nutrient capital from the site. Some studies have indicated that K and Ca would be required to be added through fertilization to sustain site productivity under whole tree harvest management. On low to medium productivity sites, fertilization of public lands is not economically feasible. With knowledge about

natural rates of supply, land managers can suit rotation lengths and harvest intensities to their sites with greater confidence.

The first of the following four papers presents chemical and erosional denudation rates in four watersheds in the Idaho batholith. These rates are correlated with annual water yields, and frequency distributions developed on the long-term hydrologic record in the watersheds. Denudation rates are discussed in terms of soil and geomorphic characteristics of the watersheds. The second paper uses a mass balance approach on Na and Ca in two watersheds to estimate the weathering rate of plagioclase. Preferential weathering of anorthite relative to albite is indicated by net export rates of Ca and Na in streams. Soil analyses support this by indicating relative enrichment of albite in C horizons. The third paper describes a logical basis for estimating the actual rate of elemental supply to a forest ecosystem based on weathering rate, and stand growth dynamics. A rate of supply is calculated for a stand in one watershed. The fourth paper evaluates a timber harvest treatment in one watershed in light of nutrient loss accompanying harvest on the basis of time required for the forest ecosystem to reach its pre-harvest nutrient level.

EROSIONAL AND CHEMICAL DENUDATION IN THE
SOUTHWESTERN IDAHO BATHOLITH

James L. Clayton and Walter F. Megahan

ABSTRACT

Annual erosional denudation (ED) and chemical denudation (CD) rates are determined on four forested watersheds located in the Idaho batholith, Idaho, U.S.A. Bedrock in the study area is a coarse-grained quartz monzonite. Annual sediment yields are computed from sediment collected in settling ponds at the watershed mouths and corrected for trapping efficiency. Solution loads for denudation components (Na, K, Ca, Mg, SiO_2 , Al_2O_3 and FeO) are computed using equations relating concentration to flow, and by integration of mean daily flow values. Eleven years of water chemistry data and 19 to 23 years of sediment yield data are used in the denudation estimates. Strong relationships exist between CD, ED and annual water yield, so probability distributions of annual water yield are used in Monte Carlo simulations to establish frequency distributions of annual CD and ED rates. Mean annual total denudation for the four watersheds ($\text{CD} + \text{ED}$) equals $235 \text{ kg ha}^{-1} \text{ y}^{-1}$ or 8.9 Bubnoffs. The 11-year mean ED rates exceed CD rates for three of four watersheds, although it is more probable in any given year that CD will exceed ED for three of the watersheds. ED is temporally more variable than CD for all watersheds. Simulations of denudation over 1,000-year periods indicate that the frequency of occurrence of ED is skewed toward higher denudation values, and the modal ED rate is substantially lower than the mean value. CD rates are less skewed, with mean and mode in the same class. Ninety-seven percent of CD is comprised of Na, Ca and SiO_2 fluxes. Episodic high ED rates are related to high peak flows from snowmelt runoff associated

with winters of large snow accumulation. Perturbations such as fire or road construction would be expected to accelerate ED more than CD.

KEY WORDS Denudation Dissolved load Sediment load
 Forested basins Granitic soil

INTRODUCTION

The Idaho batholith covers an area of 41,000 km² in central Idaho and western Montana. It is a member of a chain of large intrusive bodies extending inland along the western border of North America. Large areas of rather uniform lithologies predominate in the batholith, although there is a systematic progression from older, more mafic rocks in the west to progressively younger, more felsic rocks in the east. Most of the region is forested, and the lands are administered by the USDA Forest Service. Management of these lands is complicated by the high erosion potential and inherent low nutrient status of the coarse textured, cohesionless soils.

Small watersheds provide a useful field laboratory for evaluating the effects of various management activities on natural biogeochemical processes. Our interests deal primarily with changes in water yield, peak flows, sediment yields, and geochemical cycling caused by logging and associated activities such as road construction, slash disposal and site preparation for tree regeneration. This research is directed at predicting the effects of these activities on both onsite productivity and downstream resources. In order to make these predictions it is also important to study the processes of weathering, erosion and slope and channel storage under undisturbed conditions. For these reasons we are interested in estimates of contemporary erosional and chemical denudation in this area.

In a recent review of denudation rate studies, Saunders and Young (1983) pointed out that most studies include field measurements of 1-3 years duration. Because climatic factors, and therefore watershed responses, vary significantly over time, it is necessary to have longer periods both for calibration prior to treatment and for evaluation of treatment effects. Our data set for precipitation, streamflow and sediment yield ranges from 19 to 23

years on four watersheds; our stream and precipitation chemistry data covers 11 years. In addition, we have omitted from this study watersheds that we suspected to have water inputs from sources outside their drainage divides or losses to deep seepage upstream from sampling stations at the watershed mouth. This assumption of "basin tightness" is essential to a denudation computation (Clayton, in press). The objective of this paper is to quantify the magnitude of erosional and chemical denudation rates in the southwestern Idaho batholith, and to relate temporal variation in denudation to annual water yields.

DESCRIPTION OF THE STUDY AREA

The Silver Creek study area is located in the southwestern portion of the Idaho batholith, 44°25' N latitude and 115°45' W longitude (Figure 1.1). Four watersheds were selected for this study on the basis of having similar hydrologic and topographic characteristics (Table 1.1). These watersheds are forested, the principal overstory species being ponderosa pine (Pinus ponderosa Laws.) and Douglas-fir (Pseudotsuga menziesii Mirb. Franco). Soils on slopes are generally shallow, coarse textured (gravelly loamy coarse sands or coarse sandy loams), and weakly developed, exhibiting only A and C horizons. Some soils at lower elevations on alluvial terraces are greater than 1 m in depth. These more stable soils commonly have weak cambic horizons. Slopes on the watersheds are steep, with gradients ranging from 50 to 75 percent. The area has a mediterranean climate with dry summers and cool, moist winters. Annual precipitation averages 100 cm, with about 65 percent falling as snow in the winter (Megahan et. al, 1983). Peak runoff coincides with the spring snowmelt period. High intensity convective rainstorms are common during the summer months.

Bedrock in the area is uniform, light colored quartz monzonite (Clayton, in press), typical of a large portion of the main inner facies of the batholith (Ross, 1963). This rock contains

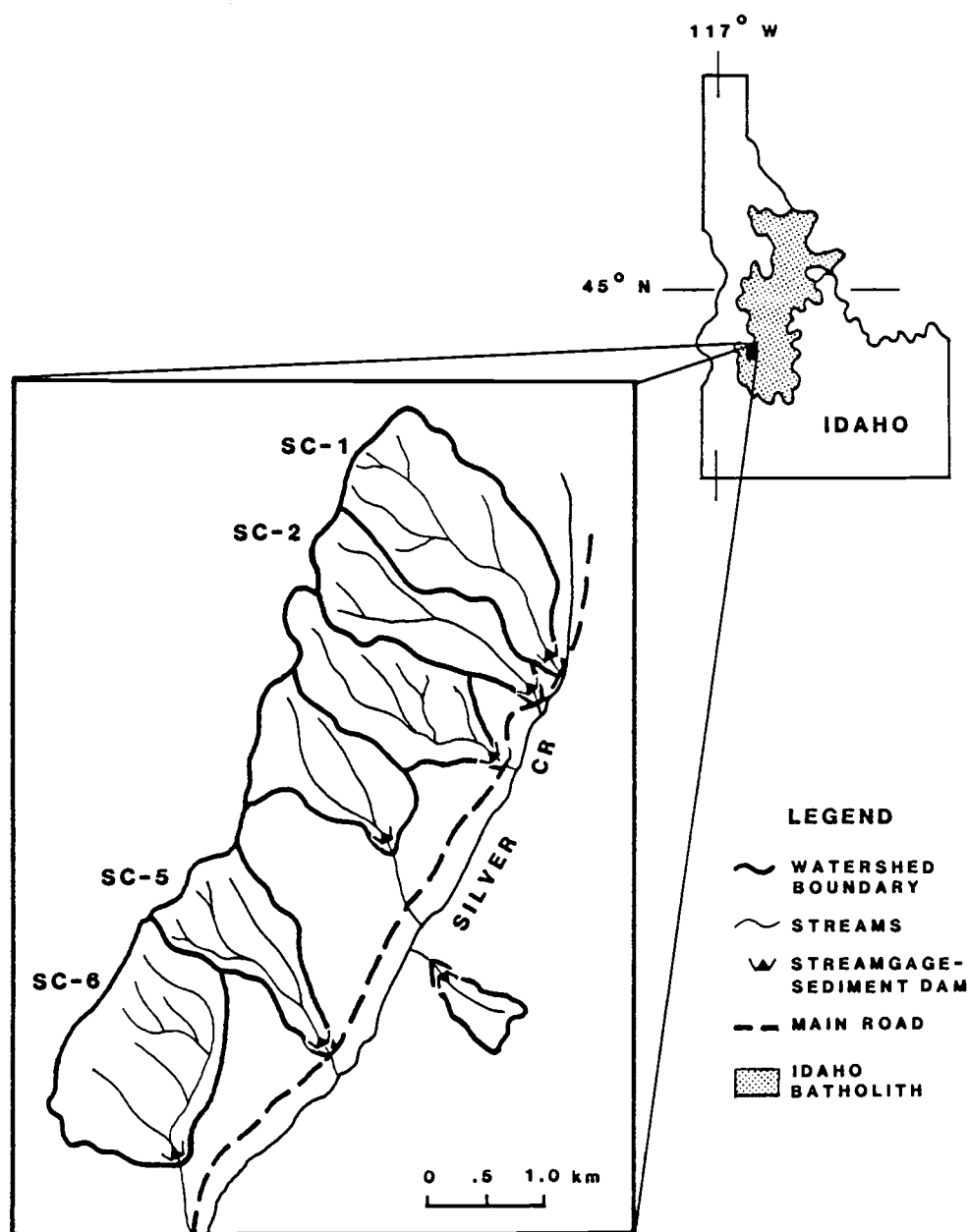


Figure 1.1. Location map of the study area in the Idaho batholith.

Table 1.1.--Descriptive data for the study watersheds.

Parameter	Watershed			
	SC-1	SC-2	SC-5	SC-6
Area, km ²	1.86	1.18	1.09	1.62
Mean annual precipitation, cm [†]	112	109	98	99
Annual water yield, cm [‡]	43.7	46.7	34.2	30.4
Elevation, highest point, m	2079	2066	1775	1775
Elevation at mouth, m	1490	1480	1395	1425
Relief ratio	0.248	0.251	0.203	0.189
Drainage density, km ^{-1§}	6.51	7.63	6.27	8.27
Mean annual instantaneous [‡] peak flow, m ³ sec ⁻¹ km ⁻²	0.17	0.16	0.10	0.10

† Based on an isohyetal map constructed from 17 years of records.

‡ Based on 17 years of records.

§ From 1:12,000 base map and estimated June 1 surface water.

approximately equal amounts of quartz, orthoclase and An₁₉ plagioclase, with minor amounts of biotite. Prior research has shown that the dominant weathering reaction on a mass per unit time basis is hydrolysis of plagioclase (Clayton, 1985). Further, preferential weathering of anorthite results in its selective removal relative to albite based on analysis of net export rates of dissolved Ca and Na in streamwater. Supporting data indicate albite enrichment of the soil relative to albite content of the fresh rock.

Watersheds SC-1 and SC-2 were undisturbed through the period of record of this study. Watershed SC-5 had 2,995 m of road construction in 1980. This construction disturbed 3.7 ha of ground surface area, but none of the eroded material reached the mouth of the

watershed during WY 1981 or 1982. Roads were constructed in SC-5 incorporating extraordinary design features to reduce erosion and sedimentation, and all erosion has been mapped in an accompanying study. Watershed SC-6 was partially cutover in WY 1977. Timber was harvested in twenty-three percent of the area and all yarding was done with a helicopter. There are no roads in SC-6. Clayton and Kennedy (1985) have evaluated the effects of this helicopter logging activity on stream chemistry and sediment production; they found no increases in denudation components resulting from the logging.

METHODS

Denudation components considered in our analysis include: 1. mineral sediment, 2. inorganic minerals transported with particulate organic matter, and 3. dissolved elements derived from mineral weathering (Na, K, Ca, Mg, Al, Si and Fe). Annual inputs of these elements by precipitation are subtracted from effluxes measured at the watershed mouths. Methods for making denudation computations are described in detail below.

Chemical Denudation

Annual precipitation on each watershed was computed by adjusting the 17-year average precipitation input from an isohyetal map based on eleven gages by the annual deviation from the 17-year mean for all gages. Precipitation chemistry has been sampled at four locations in the Silver Creek area using bulk (dryfall plus wetfall) collectors since 1974. During the period April through October, precipitation samples were collected for analysis whenever a sufficient volume had accumulated. During the winter, fresh snow samples were collected monthly for analysis. Samples were analyzed for Na and K by flame emission spectrophotometry, and Ca and Mg by atomic absorption spectrophotometry. Silica was determined by reducing molybdosilicic acid to the heteropoly blue complex

according to the method of Greenberg et. al. (1980). Fe and Al were not found in sufficient concentrations in our precipitation to warrant their analysis.

Mean concentrations of elements are lower in snow than rain in the study area. Elemental inputs with precipitation were computed by multiplying annual precipitation volumes by long term mean elemental concentrations in rain or snow, and using the assumption that 65 percent of annual precipitation falls as snow.

Annual water yields were computed from continuously recorded streamflow records using water-level recorders and Parshall flumes. Water samples for chemical analysis were collected monthly during winter, biweekly during the summer, and at 2- to 4-day intervals during peak flows which correspond with spring snowmelt. Samples were analyzed for Na, K, Ca, Mg and SiO_2 using the same techniques described for precipitation. Fe analyses of stream water were done colorimetrically with o-phenanthroline, and Al was determined colorimetrically by the Eriochrome cyanine R technique. Methods for Fe and Al analyses are described by Greenberg et. al. (1980). Dissolved losses at each watershed mouth were computed using equations that correlate elemental concentration and instantaneous stream discharge. Utilizing these regression equations, daily fluxes were generated from mean daily flow, and these were summed over a period of a year to compute fluxes in $\text{kg ha}^{-1} \text{y}^{-1}$. This technique averages the prediction error of the regression over 365 observations, and thus provides a highly reliable estimate of efflux for an unbiased regression model. Al concentration is not correlated with discharge, so we allowed Al concentration to equal a constant in the regression equation. This leads to a negligible error because dissolved Al species never exceed 0.01 percent of chemical denudation in our watersheds.

Chemical denudation was computed by subtracting annual precipitation inputs of Na, K, Ca, Mg and SiO_2 from annual dissolved

effluxes of these elements, then summing these values plus dissolved effluxes of FeO and Al_2O_3 .

Erosional Denudation

Sediment volumes retained behind debris dams at the watershed mouths were surveyed each year after snowmelt and at the end of the summer. During each survey, several representative core samples were taken for organic matter analysis. The number of core samples taken has ranged from four to 25, depending on the volume of sediment retained behind the dams. Sediment bulk densities were computed using a regression equation of density on percent organic matter developed by Megahan (1972). The trapping efficiency of the dams on each stream was checked by sampling for total sediment load passing the dam spillways over a variety of flow events. Efficiencies were found to correlate well with the total volume of sediment retained during a given sample interval. During low runoff years, low sediment volumes accumulate, and a relatively high percentage of sediment is organic. Because of the low density of organic sediment, the trapping efficiency of the dams is relatively low. During high runoff periods, a large percentage (>95%) of sediment transported to the dams is coarse granitic material, and the dams are quite efficient at trapping this material. Trap efficiencies for lithic plus organic sediment ranged from 23 to 97 percent over the life of the study. We used the relationships developed between total dam catch (volume basis) and trap efficiency to predict actual volume of sediment yield for each sample period, and adjusted this volume by the organic matter bulk density regression to compute mass of lithic sediment yield for each water year.

We consider the entire annual lithic sediment yield from each watershed composed of primary and secondary minerals, to comprise the annual erosional denudation component. Organic sediment samples from WY 1975 and 1976 were digested in a mixture of HClO_4 and HNO_3 , then analyzed for Na, K, Ca and Mg by previously described methods.

Transport of these elements in organic sediment is part of the chemical denudation loss from each watershed. Total transport of these elements in organic sediment ranged from 0.17 to 0.24 $\text{kg ha}^{-1} \text{ y}^{-1}$ and never comprised more than 0.3 percent of chemical denudation from a watershed in any year, so we have omitted this component from further computations.

Flow Frequency Analysis

The 23 years of streamflow records for watershed SC-5 provide adequate record for developing a frequency analysis of total annual water yield. Annual water yields for the western U.S. have been shown to conform most commonly to either a two parameter Gamma or two parameter log-normal frequency distribution (Markovic, 1965). The data set was applied to both distributions and the two parameter log-normal provided the best fit based on a chi-square test. The other study watersheds have shorter records (20 years for SC-6 and 19 years for SC-1 and SC-2). Based on the proximity of all four watersheds, the log normal distribution for annual water yields was assumed to be appropriate in all cases and the data were fitted accordingly. Cumulative probability distributions for the four study watersheds are plotted on Figure 1.2.

RESULTS

Denudation Based on Sample Data

The mean chemical and erosional denudation data for each watershed over the 11-year study period along with other statistics are presented in Table 1.2. For three of the four watersheds, mean erosional denudation rates exceed chemical denudation rates. On the fourth watershed mean chemical and erosional rates are approximately

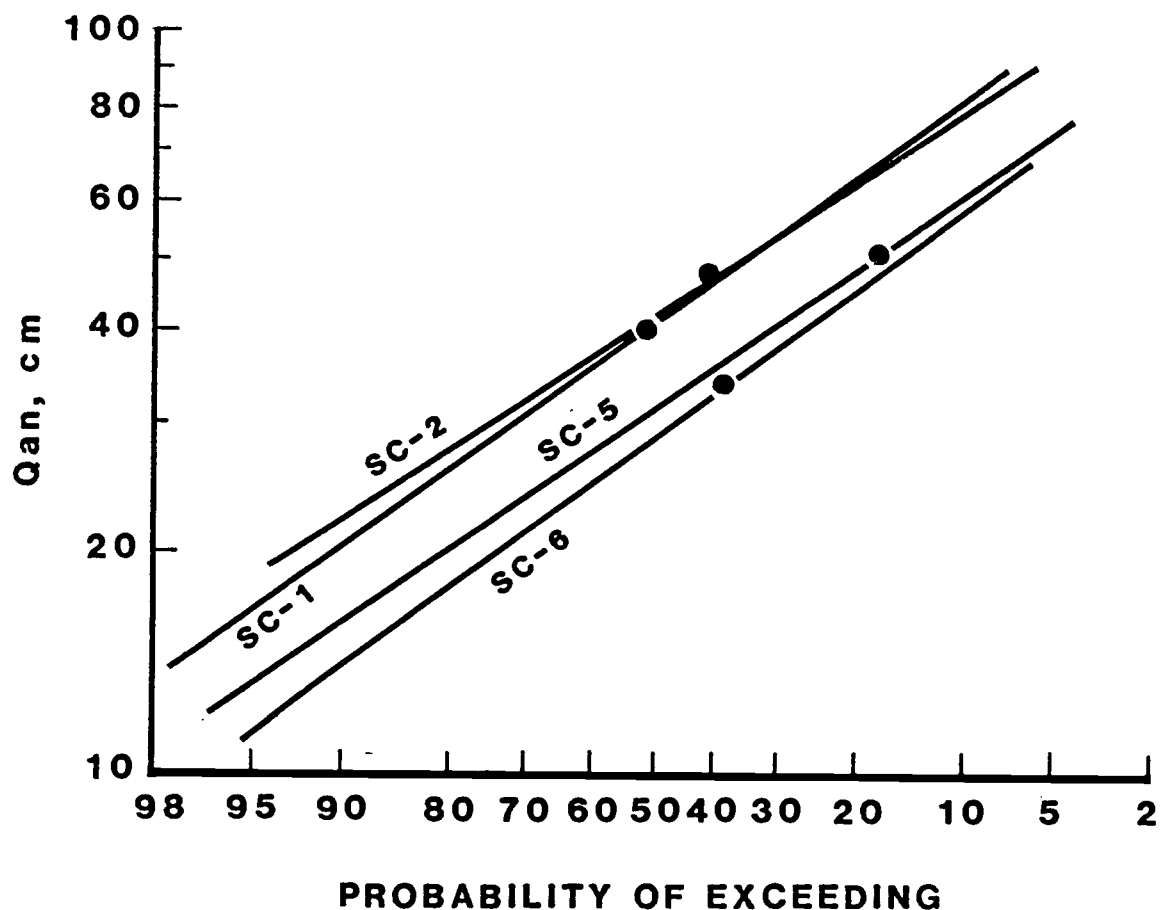


Figure 1.2. Cumulative probability distributions of annual water yield, Q_{an} , for the four study watersheds. Threshold values described in the text are indicated by ●.

equal. There is considerably higher variability in annual erosional denudation than chemical denudation both within and between years (Figure 1.3).

In order to explain some of the temporal variation in denudation we explored the relationship between annual water yield and erosional and chemical denudation. Chemical denudation is highly correlated with annual water yield, and increases linearly

Table 1.2.--Mean chemical and erosional denudation ($\text{kg ha}^{-1} \text{ y}^{-1}$), standard deviation (s), standard error of the mean (\bar{S}_y), and 95 percent upper confidence limit (UCL) and lower confidence limit (LCL) about the mean for four watersheds in the Idaho batholith, based on WY 1972-1982 inclusive.

<u>Watershed</u>	<u>SC-1</u>		<u>SC-2</u>		<u>SC-5</u>		<u>SC-6</u>	
Parameter	CD	ED	CD	ED	CD	ED	CD	ED
Mean	105	141	115	154	112	111	95	122
s	38	110	42	160	42	104	40	119
\bar{S}_y	12	33	13	48	13	31	12	36
UCL	130	215	143	261	140	181	122	202
LCL	79	67	86	46	85	41	68	42

with water yield increases (Figure 1.4). This relationship is expected because transport of dissolved elements is a function of the total volume of solution as long as the supply of solute is not limited. There is a logical upper bound to the linear relationship between chemical denudation and water yield when solute supply becomes limiting, but this limit is apparently beyond the range of our data. The regression coefficients for the four watersheds are statistically significant at $\alpha = .05$ (Figure 1.4).

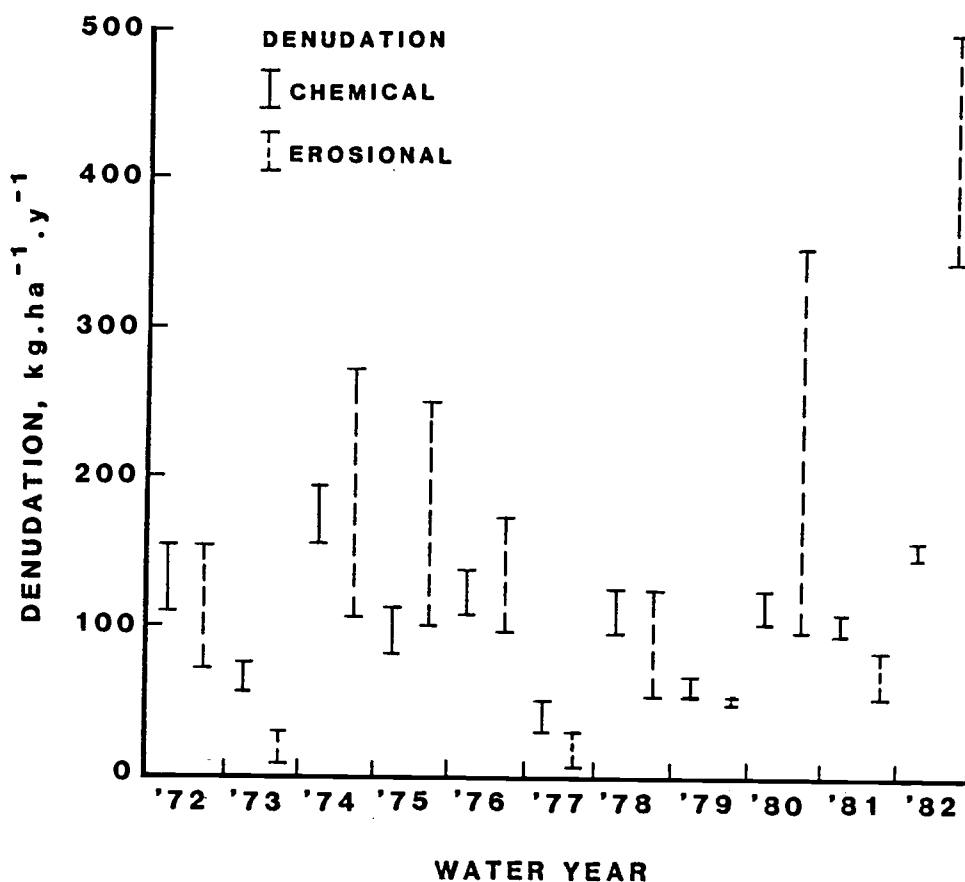


Figure 1.3. Annual range in chemical and erosional denudation for four study watersheds in the Idaho batholith.

Erosional denudation rates are related to increasing water yield by a power function (Figure 1.5). The correlation coefficients are not as high as for the chemical denudation rate correlations, but annual water yield still explains 73 to 80 percent of the variance in erosional denudation. These regression coefficients also are significant at $\alpha = 0.5$.

Logically, erosional denudation should be related to stream power, and therefore peak flows. In our snowmelt dominated systems, high peakflows are correlated with high snow accumulation years, and

hence, high annual water yields, and apparently this is reflected in sediment transport rates. Erosional denudation rates are less than chemical denudation until annual water yields exceed a threshold value for each watershed (Figure 1.6). An iterative solution to estimate the point of equality for regression equations of chemical and erosional denudation as a function of annual water yield for each watershed yields threshold values of 40.9, 46.7, 50.7 and 34.5 cm of annual flow for watersheds SC-1, SC-2, SC-5 and SC-6, respectively. By entering these threshold values in Figure 1.2, it is possible to estimate the probability of occurrence of the threshold

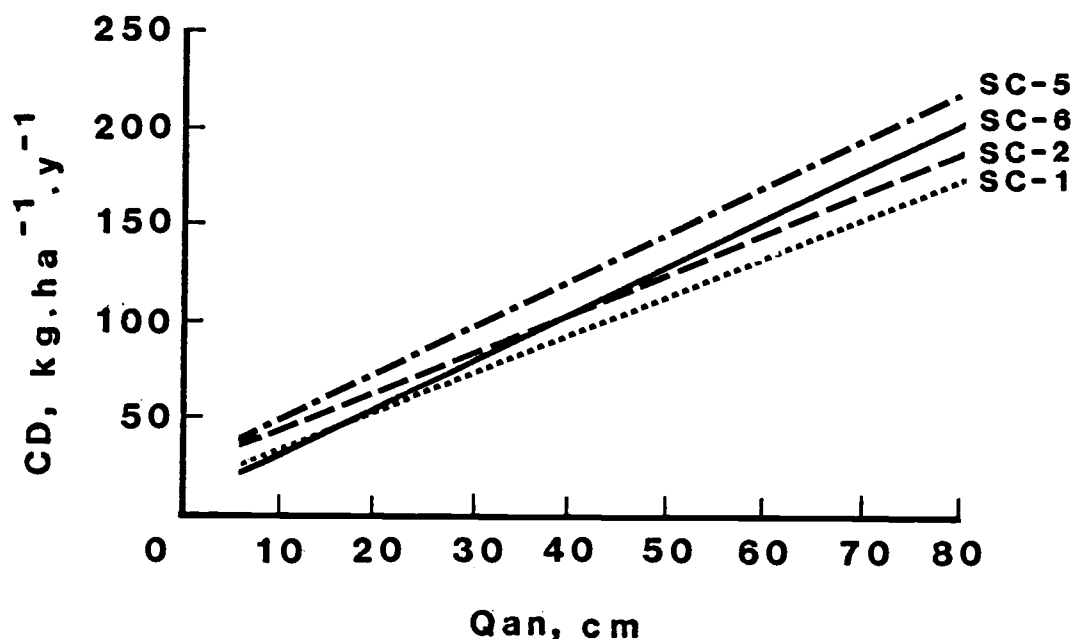


Figure 1.4. Chemical denudation rates plotted as a function of annual water yield for the four study watersheds.

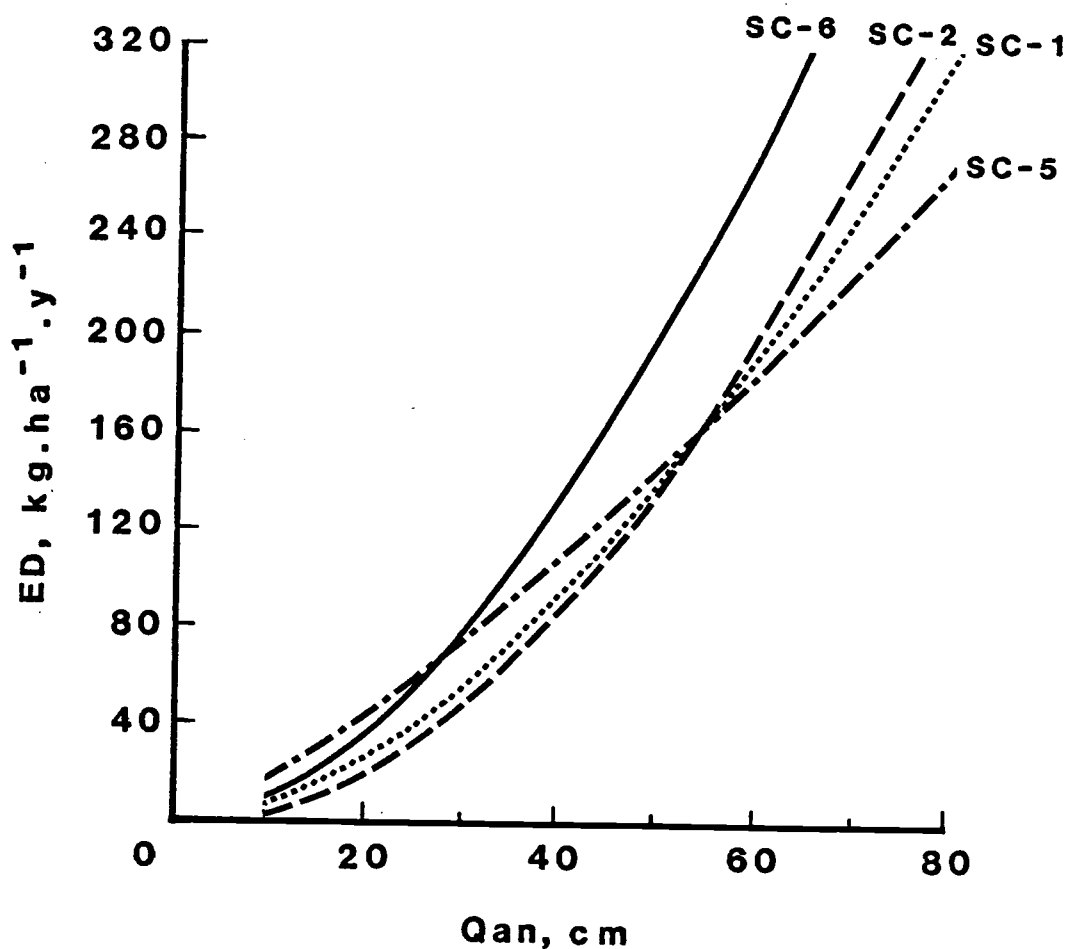


Figure 1.5 Erosional denudation rates plotted as a function of annual water yield for the four study watersheds.

for each watershed. The respective probabilities of exceeding the threshold values for watersheds SC-1, SC-2, SC-5 and SC-6 are 0.51, 0.40, 0.17 and 0.38. Another way to state this is that annual rates of erosional denudation can be expected to exceed chemical denudation 51, 40, 17 and 38 percent of the time, respectively, for these undisturbed, forested watersheds.

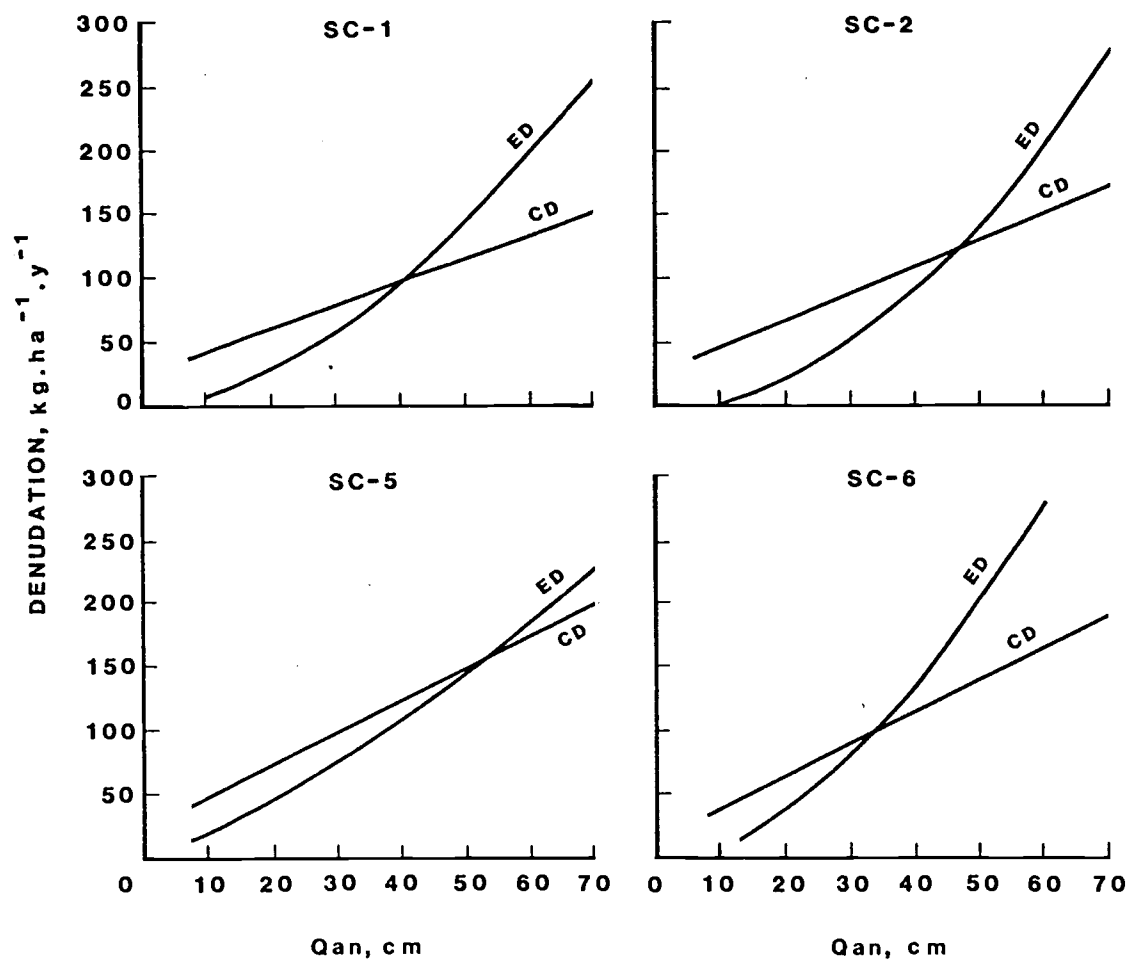


Figure 1.6. The graphical relationship between chemical denudation, CD, and erosional denudation, ED, as a function of annual water yield. Points of intersection (threshold values), indicate the lower limit of water yield where $ED > CD$.

Variations in the probability of erosional denudation exceeding chemical denudation among the watersheds suggest fundamental differences in watershed properties. For example, soil accumulation should be inversely proportional to the probability of exceedance of erosional denudation. A detailed soil survey is available for the study watersheds. Average soil depths for each soil mapping unit were weighted by mapping unit area within each watershed to provide an estimate of the total volume of soil present in each watershed. These data were then plotted against the percentage of occurrence of erosional denudation on Figure 1.7a. The correlation was not tested for statistical significance, but the expected inverse relation between soil accumulation and probability of erosional denudation is apparent.

The complement to the relationship in Figure 1.7a would also be expected. That is, areas of outcropping bedrock should be directly proportional to the probability of exceedance of erosional denudation. Such data are also available from the soil survey and are plotted as a percentage of the total area for each watershed against the probability that erosional denudation exceeds chemical denudation on Figure 1.7b. Again, the expected trend is apparent.

Monte Carlo Simulation

The denudation data in Table 1.2 are based on 11 years of record. However, utilizing the probability distribution of annual streamflow and the denudation versus flow relationships in Figure 1.6, it is possible to simulate long-term records of denudation using Monte Carlo procedures. This was done for the SC-5 watershed where streamflow records were longest.

Following standard data generation techniques for estimating flow probabilities from random numbers, annual flow values for SC-5 were generated to conform to parameters of the 2-parameter log-normal sample frequency distribution. Generated flows were then

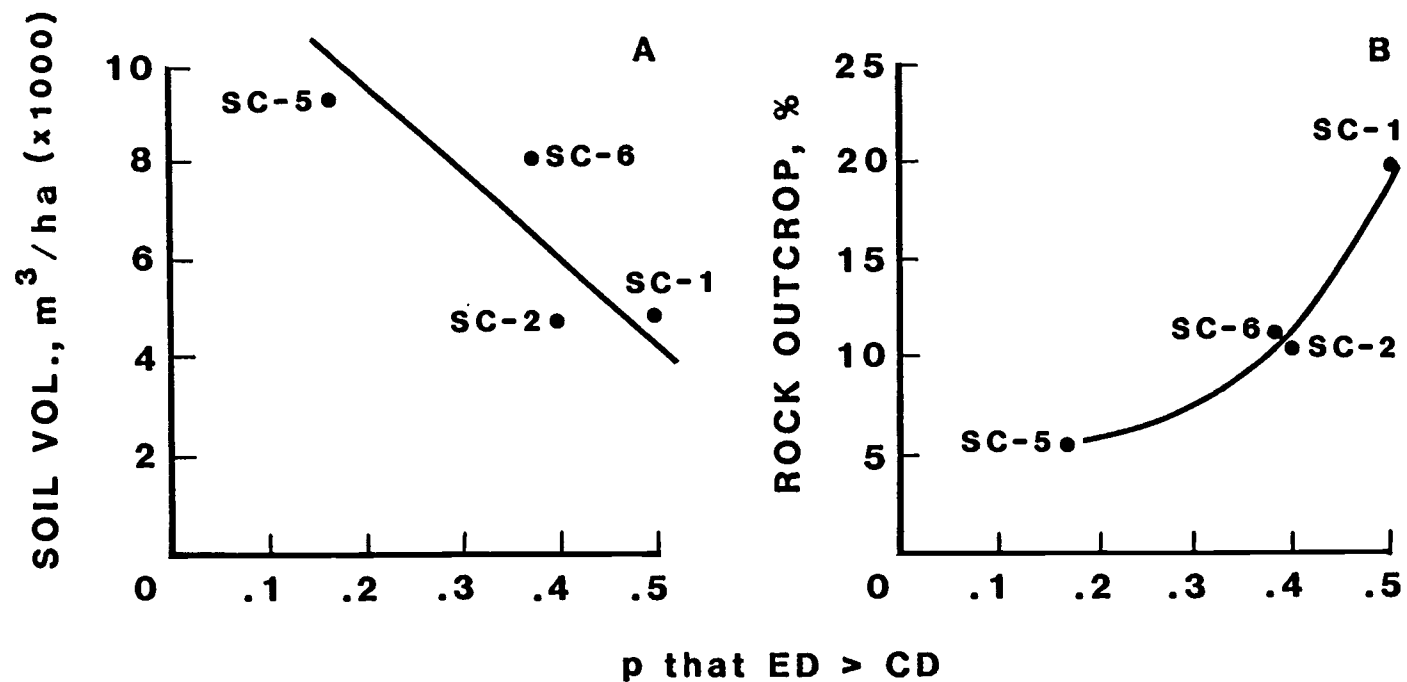


Figure 1.7. Volume of soil in each watershed (a) and percent rock outcrop in each watershed (b) plotted vs. the probability that erosional denudation, ED, exceeds chemical denudation, CD.

used to predict annual erosional and chemical denudation based on the regression equations for SC-5 relating denudation to annual streamflow.

A series of twenty 1,000-year sequences of erosional and chemical denudation were generated using this procedure and frequency distributions of the values were developed for each series. In addition, the frequency distribution of the ratio of erosional to chemical denudation was developed for each 1,000-year series. The generated frequency distributions represent the long-term probability of denudation processes assuming undisturbed forest vegetation and long-term climatic stability.

A representative data set was selected to illustrate the results of the analysis (Figure 1.8). Note that the peak of the frequency distribution for erosional denudation occurs in the 60 $\text{kg ha}^{-1}\text{yr}^{-1}$ class compared to the 100 $\text{kg ha}^{-1}\text{yr}^{-1}$ class for chemical denudation. In addition, the peak erosional denudation occurs less frequently than the peak chemical denudation rate. However, the distribution of erosional denudation is more skewed so that extreme values are about double those for chemical denudation. The average annual erosional denudation rate for the simulated 1,000-year-period was 99 $\text{kg ha}^{-1}\text{yr}^{-1}$ compared to 111 $\text{kg ha}^{-1}\text{yr}^{-1}$ for the 10-year sample period. In contrast, average simulated and sampled chemical denudation rates were equal amounting to 112 $\text{kg ha}^{-1}\text{yr}^{-1}$.

The frequency distribution of the ratios of simulated erosional and chemical denudation is much more symmetrical than the individual denudation rates with a peak occurring between 0.7 and 0.8 (Figure 1.9). Ratio values reach 1.7 during high flow years. It is apparent that the relative importance of erosional versus chemical denudation processes is a function of annual runoff with erosional processes dominating during high runoff years. Climatic trends that increase runoff would increase the relative importance of erosional denudation. Progressive runoff increases ranging from 10 to 80

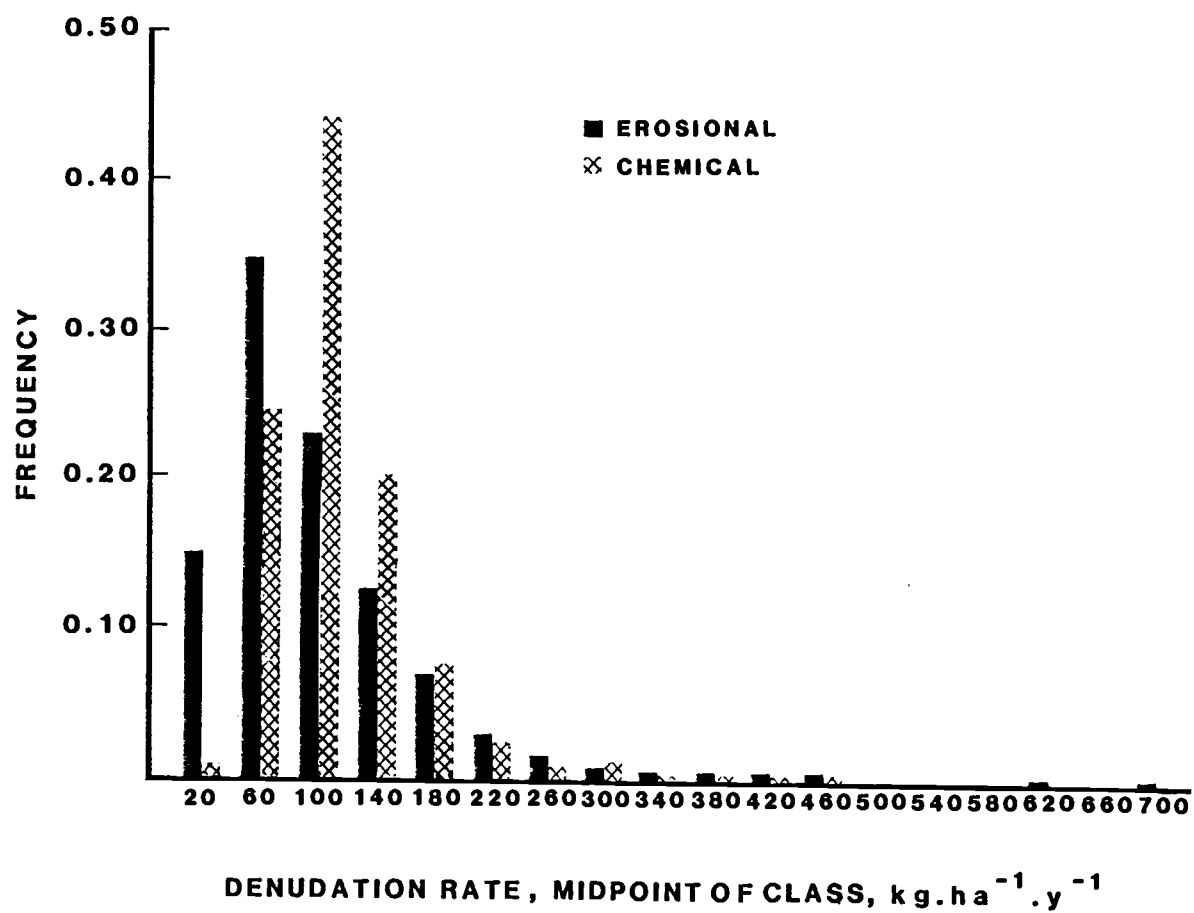


Figure 1.8. Frequency distribution for annual erosional and chemical denudation in SC-5.

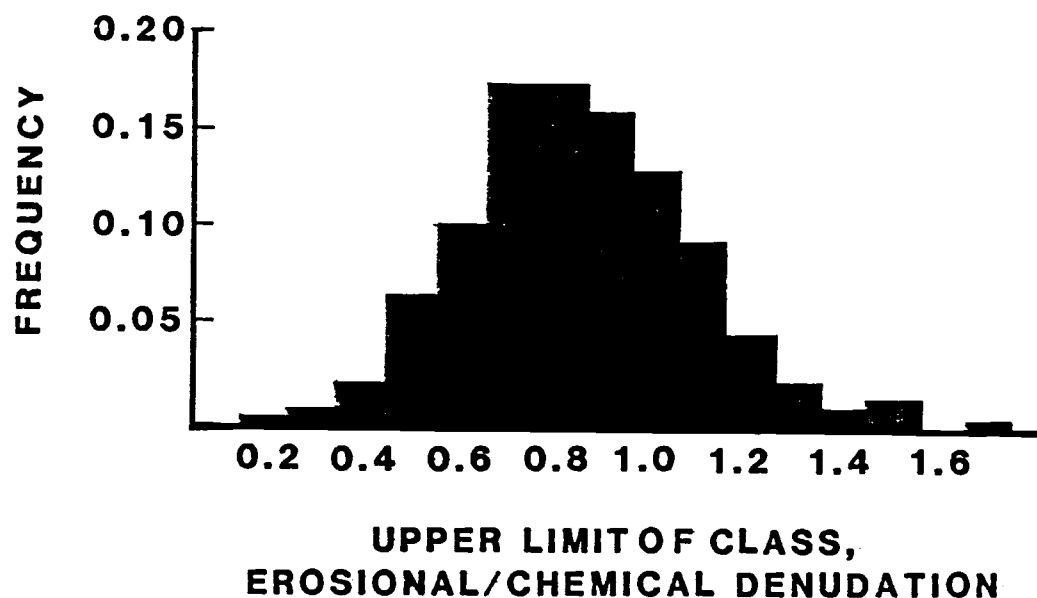


Figure 1.9. Frequency distribution for the ratio of annual erosional/chemical denudation, SC-5.

percent were introduced into the denudation simulation model to evaluate the breakeven point between erosional and chemical denudation processes. The point of equality occurred at runoff increases of about 50 percent. At higher runoff values, erosional denudation increasingly exceeded chemical denudation.

DISCUSSION

In computing chemical denudation rates, certain assumptions were made. We assumed no losses of meteoric water to deep seepage

and, therefore, no associated losses of dissolved elements. Further, we assumed that all water and dissolved elements leaving the watershed originated from precipitation and weathering, and there were no water sources originating from outside the catchment divide. We tested this assumption by computing a Cl^- budget in the four watersheds. This test relied on the fact that there is a negligible source of Cl^- from rock weathering (Cl^- concentration in rock = $0.07 \pm 0.01\%$); that the only other source of Cl^- is from precipitation; and biological cycling of Cl^- is in a steady state (reasonable in view of the low biological demand and high mobility of Cl^-). Over the 5-year period 1978-1982, stream effluxes of Cl^- equalled 89 percent, 93 percent, 98 percent and 104 percent of Cl^- inputs in precipitation for the four watersheds. Because of the sampling and analytical errors inherent in computing these budgets, we choose to accept the assumption of no substantial gains or losses of water due to springs or deep seepage.

We also assumed that biological uptake and release of denudation components within the forest stand is in a steady state. Plant demand for nutrients is cyclic during the life of a stand. During periods of rapid biomass incremental growth, plant demand for Ca and K may lower the chemical denudation estimate, and conversely, following a major disturbance to a stand (fire, disease) decay and nutrient release may cause an overestimate of long term chemical denudation. All four watersheds are forested with multi-canopied stands. There has been no logging activity other than the previously described harvest in SC-6, and we have established that there are no effects to chemical denudation from that activity. There have been no extensive fires in the area since 1941. On that basis, we assume that biological controls on chemical denudation are typical of the long term influence due to a forested cover type on the watersheds.

Erosion rates on the study watershed approximate a minimum because of the well developed forest cover. However, the

probability that such conditions will last for 1,000 years is very small. Aside from man-caused accelerated erosion, natural disasters such as extreme storms, wind storms, insect attacks and wildfire are likely to cause accelerated erosion. For example, based on a 75-year history of fire occurrence in the vicinity,^{1/} we can expect to burn off the entire study basin an average of at least three times in a 1,000-year-period. Both surface and mass erosion rates increase drastically as a result of fire (Megahan and Molitor, 1972; Gray and Megahan, 1981). Other natural disasters, especially major storm events, further accelerate erosion. Detailed simulation of these impacts on the long-term denudation rates is beyond the scope of this report. However, it is very likely that the additional effect of accelerated erosion over a 1,000-year-period would cause long term dominance of erosional denudation over chemical denudation.

Nearly 97 percent of total chemical denudation is made up of dissolved Na, Ca and SiO_2 transport (Table 1.3). The source of the Na and Ca, and the majority of SiO_2 is from plagioclase weathering. Clayton (1985) estimated a 17-year average plagioclase weathering rate of $236 \text{ kg ha}^{-1} \text{ y}^{-1}$. The modal An content of plagioclase in the study watersheds is An_{19} , however preferential weathering of anorthite is indicated by the high Ca fluxes in stream water. Clayton estimates that albite weathering averages $138 \text{ kg ha}^{-1} \text{ y}^{-1}$. Soil mineral analyses indicate a 7 to 11 percent increase in the weight ratio Na:Ca over the same ratio in fresh rock samples, suggesting albite enrichment in the soil. This is further evidence

^{1/}Unpublished data. Max Muffley, Fire Control Officer, Emmett RD, Boise National Forest.

Table 1.3.--Average composition of the chemical denudation component from four watersheds in the Idaho batholith based on WY 1972-1982 inclusive. Denudation is expressed in $\text{kg ha}^{-1} \text{ y}^{-1}$.

Component	Denudation		Percent of Total Chemical Denudation
	Mean	Std. Dev.	
Na	12.3	4.0	11.5
K	1.63	0.75	1.5
Ca	13.6	5.7	12.7
Mg	1.43	0.54	1.3
SiO_2	77.4	28.8	72.5
FeO	0.33	0.24	0.3
Al_2O_3	0.061	0.032	<0.1

of the preferential weathering rate of anorthite in the soil. Erosional denudation of weathered residuum is likely to set an upper limit on albite enrichment in the soil, and assures a supply of unweathered plagioclase to the soil, sustaining the preferential weathering of anorthite.

The 11-year mean value on total denudation (erosional plus chemical denudation) for the four watersheds is $235 \text{ kg ha}^{-1} \text{ y}^{-1}$, which equals $8.9 \text{ B}^{2/}$. This figure falls within the low range for montane watersheds according to a recent summary paper by Saunders and Young (1983). Their results indicate that lower rates are

$^{2/}\text{B}$ = Bubnoffs, the standard units for denudation expressed in $\text{mm}/10^3 \text{ y}$. This represents a rate of lowering of the earth's surface, assuming rock specific gravity = 2.65.

expected on siliceous rocks and on watersheds that are forested. This value is approximately three times the denudation rates calculated for Hubbard Brook (watershed 6), New Hampshire (Likens et al., 1977) and for Pond Branch, Maryland (Cleaves et al., 1970). Both these study sites in the eastern United States are small watersheds with a forest cover. Chemical denudation rates exceed erosional denudation rates by 5:1 in both Hubbard Brook and Pond Branch. Lower erosional denudation at these sites compared to the Silver Creek watersheds might be explained, in part, by gentler topography and by more even distribution of precipitation over the year in the Eastern United States.

Marchand (1971) estimated a denudation rate of 1.4 to 1.9 B in the White Mountains, California, on a forested area underlain by quartz monzonite. Although his study area is lithologically similar to ours, climatic conditions in the White Mountains are significantly drier and colder. Zeman and Slaymaker (1978) reported a denudation rate of 12.5 B in a perhumid, forested mountainous basin in British Columbia, underlain by quartz diorite. Mean ratio of chemical to erosional denudation in this study was 0.64 to 1, in contrast to our ratio of 0.80 to 1. Dissolved SiO_2 contributed 47 percent of chemical denudation in British Columbia in contrast with our figure of 72.5 percent.

AN ESTIMATE OF PLAGIOCLASE WEATHERING RATE IN THE IDAHO BATHOLITH
BASED UPON GEOCHEMICAL TRANSPORT RATES

James L. Clayton

INTRODUCTION

Geochemical transport rates through spatially well defined systems (usually drainage basins) have provided insight into the kinds and magnitudes of geomorphic processes on the earth's surface. For example, comparisons of annual solution load transport and sediment load transport from basins have been interpreted in light of erosion rates, bedrock susceptibility to physical and chemical weathering, local climate, vegetation cover, and landform (Leopold et al., 1964; Carson and Kirkby, 1972; Gregory and Walling, 1973; Meybeck, 1976). In small basins underlain by bedrock with a uniform mineralogy, more detailed investigations can provide estimates of contemporary chemical weathering rates. These studies require mass balances of precipitation and stream chemistry, and either an assumption of steady state intrasystem elemental cycling or quantification of intrasystem cycling rates.

Various studies along these lines present non-carbonate rock weathering rates (cf. Cleaves et al., 1970, 1974; Marchand, 1974; Reynolds and Johnson, 1972; Likens et al., 1977; Verstraten, 1977). With the exception of Cleaves et al. (1974) who studied serpentinite, rocks in the other studies contained such mixed lithologies that it was difficult to assign weathering rates to individual minerals, and only general rates of weathering for the rock as a whole were presented.

The coarse-grained quartz monzonite facies (Ross, 1963) of the Idaho batholith underlies most of the southwestern Idaho montane watersheds. These watersheds provide an ideal laboratory for studying plagioclase weathering rates. The rock is relatively free of accessory minerals containing Na or Ca, and large plutons of uniform lithology are common to the area. Inputs of Na and Ca from precipitation were calculated from 17 years of precipitation intensity data and 7 years of bulk rain and snow chemistry data. Efflux rates of Na and Ca were generated from mean daily flow data over 17

years using a reliable relationship between concentration of the two elements and instantaneous stream discharge for the study watersheds. The difference between stream efflux and inputs from precipitation was considered to be the input into the surface water system by weathering. Calculation by this method shows that plagioclase weathering does not deliver Na and Ca to the surface waters in proportion to the molar ratios in the mineral. Ca is preferentially released relative to Na in the surface waters. Various biotic and physical sinks for Na within the watersheds were explored and rejected. Molar Na:Ca ratios in soils when compared to fresh rock suggest that differential weathering rates of anorthite and albite in the normally zoned plagioclase explain the relative enrichment of Ca in stream water.

DESCRIPTION OF THE RESEARCH AREA

The Idaho batholith covers an area of 41,000 km² in central Idaho and western Montana. It is a member of a chain of large intrusive bodies extending inland along the western border of North America. Much of the Idaho batholith is composed of an inner zone of uniform, light colored rock of granodiorite and quartz monzonite composition (Ross, 1963). A border zone of more calcic rock may have originally constituted an envelope that enclosed much of the main facies. The batholith is zoned spatially and chemically; older, more mafic rocks are found at the western edge, becoming progressively more felsic and younger in an easterly direction (Schmidt, 1964).

The Silver Creek study area is located along Silver Creek, tributary to the Middle Fork of the Payette River in the southwestern batholith (44°25'N latitude and 115°45'W longitude, Figure 2.1). Bedrock is a coarse-grained quartz monzonite, typical of the main inner facies. Two watersheds were selected for study on the basis of their hydrologic characteristics and the lack of deep subsurface springwater feeding into the streams. Descriptive data

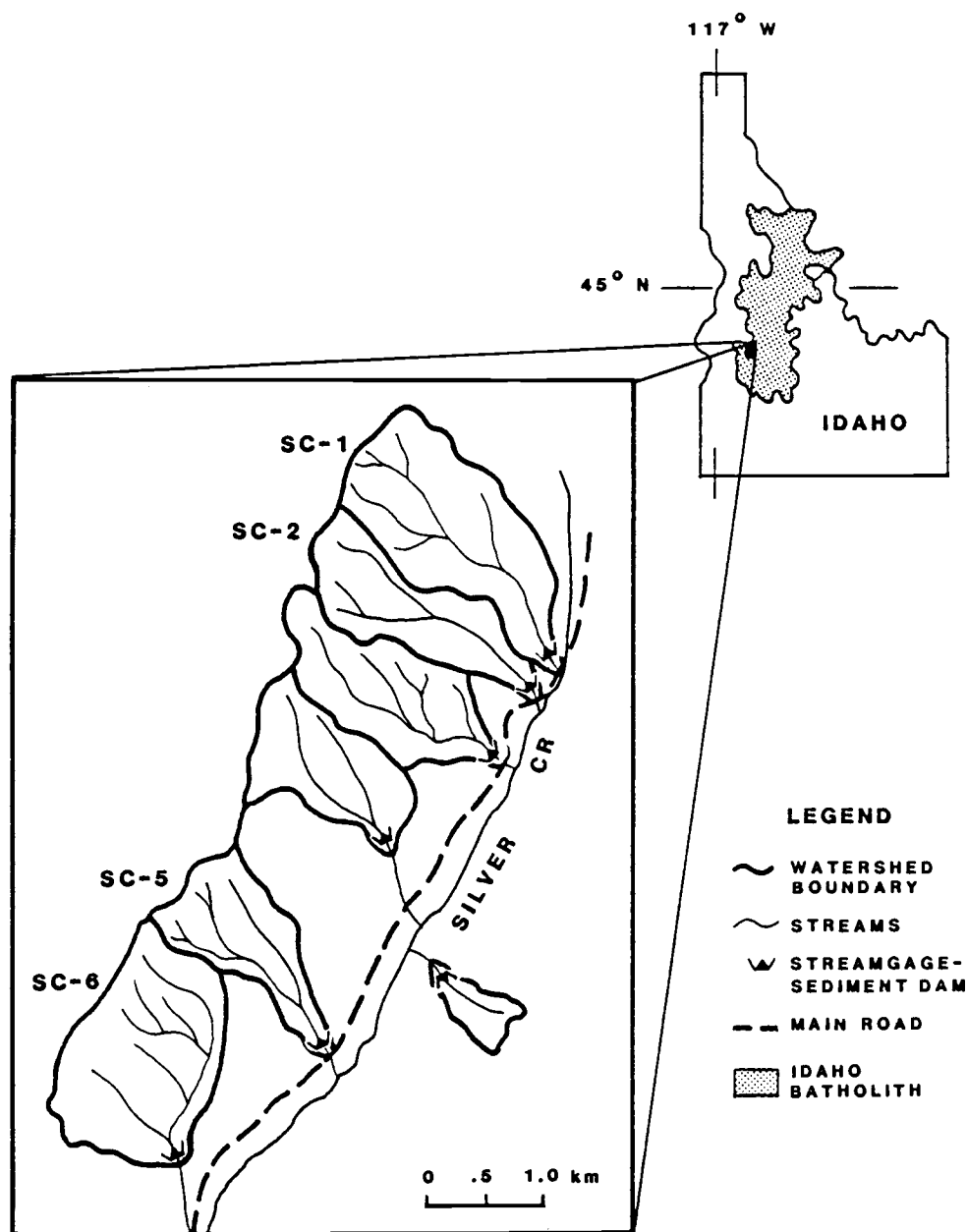


Figure 2.1. Location map of the study area in the Idaho batholith.

on the watersheds are presented in Table 2.1. Streamflow and precipitation have been monitored continuously on these watersheds since 1964, and stream and precipitation chemistry has been monitored since 1973.

Table 2.1. Descriptive data for the study watersheds.

Parameter	Watershed	
	SC-1	SC-2
Area, km ²	1.86	1.18
Mean annual precipitation, cm ^{1/}	112	109
Annual water yield, cm ^{1/}	43.7	46.7
Elevation, highest point, m	2079	2066
Elevation at mouth, m	1490	1480
Relief ratio	0.248	0.251
Drainage density, km ⁻¹ ^{2/}	6.51	7.63
Mean annual peak flow, m ³ .sec ⁻¹ .km ⁻² ^{1/}	0.17	0.16

^{1/}Based upon 17 years of record.

^{2/}From 1:12,000 base map and estimated June 1 surface water.

The watersheds are forested with mature stands of timber. The principal overstory species are ponderosa pine (Pinus ponderosa Laws.) and Douglas-fir (Pseudotsuga menziesii Mirb. Franco). Soils are generally shallow, coarse textured (gravelly loamy coarse sands or coarse sandy loams), and weakly developed, exhibiting only A and C horizons. Precipitation averages approximately 110 cm per year and falls mainly as snow.

Mean annual soil temperatures at 50 cm depth are less than 8°C on north-facing slopes, and between 8° and 15°C on south-facing

slopes (Nelson, 1976). Soils are generally very wet during snowmelt and dry out rapidly during the summer, reaching lowest water contents in September. In 1977, the year with most complete soil water data, average soil water tension in the upper 1.2 m of soil ranged from <0.01 MPa in early May to 0.4 MPa in early September. For the same period, water tension in the upper 0.3 m of soil ranged from <0.01 MPa to >1.5 MPa.

DATA COLLECTION AND CHEMICAL ANALYSIS

Precipitation chemistry was sampled at four locations in the Silver Creek study area over the elevation range 1420 m to 2070 m. Bulk precipitation samples (wet plus dry fallout) were collected in 250 mm orifice diameter plastic containers during the rain season from May through October. These samples were collected for chemical analysis whenever sufficient precipitation (approximately 100 ml) was present. Clean snow samples, when available, were collected on a monthly basis during winter (November through April). Precipitation samples were returned to the laboratory, filtered through $0.45\ \mu\text{m}$ membrane filters and refrigerated until analysis. Stream water was collected for chemical analysis at biweekly intervals from June through October and monthly from November through March. During spring snowmelt, stream samples were collected more frequently to adequately sample the rising and receding limbs of the melt-generated hydrograph. Stream samples were filtered and sub-samples preserved with acid and refrigerated at 2°C .

Stream and precipitation samples were analyzed for a variety of elements and compounds including the following: Na^{+} , Ca^{++} , SiO_2 , Al^{+3} , and Cl^{-} . Total alkalinity and pH were measured on unfiltered and unpreserved samples within a few hours of collection. Sodium was determined by flame emission spectroscopy and Ca^{++} by atomic absorption spectroscopy. Silica was determined colorimetrically following formation of the silicomolybdate complex and reduction to

the heteropoly blue form with sulfite. Aluminum was determined colorimetrically by the Eriochrome cyanine R dye technique, and Cl^- was determined by titration with $\text{Hg}(\text{NO}_3)_2$, using diphenylthiocarbazone to detect the presence of free Hg^{++} . Total alkalinities were determined by titrating with 0.01 M H_2SO_4 to pH 4.3.

Stream discharge was monitored continuously through Parshall flumes using Stevens Model A-35 water level recorders. A network of 12 recording raingages and six snow storage gages located in SC-1, SC-2 and 4 adjoining watersheds measured precipitation. An isohyetal map based upon 17 years of precipitation data was used to arrive at the average precipitation figures presented in Table 2.1.

ANALYSIS OF DATA

A. Assumption of Basin Tightness

The mass balance technique requires an assumption that all the water leaving the watershed originates as precipitation that has recently fallen within the topographic bounds of the watershed. Surface water emanating from springs whose origin is from interbasin transfer or from deep sources will have a chemistry dictated by geochemical processes other than weathering near the earth's surface, and thus will confound the weathering rate estimate. In addition, water lost to deep seepage in the watershed will cause an underestimate of weathering because fluxes of solutes will be lost to deep seepage. Springs are common in the southwestern Idaho batholith, and the presence of two on watersheds in the Silver Creek area precluded their inclusion in this study. Losses to deep seepage are difficult to detect but were of concern in the Idaho batholith because of the deep fracture and joint patterns that exist.

To test if the two selected basins have springs or deep seepage losses, I used a Cl^- conservation balance similar to the method of Cleaves et al. (1970). This test is designed to check if Cl^- inputs from precipitation approximately equal Cl^- outputs in streamflow measured at the flume at the bottom of the watershed. The assumptions: (1) that there is a negligible mineral source of Cl^- from weathering (a reasonable assumption for the study area; $[\text{Cl}^-]$ in rock = $0.07 \pm 0.01\%$); (2) that the only other source of Cl^- is from meteoric inputs of precipitation. Further, I assumed that intra-system cycling of Cl^- is in a steady state, which is likely because of the low biologic demand and high mobility of Cl^- .

The Cl^- budget was tested for the 3 years from October 1, 1978 through September 30, 1981. Precipitation inputs were estimated from 57 samples of rain and seven samples of snow collected over this period. During the three years, Cl^- inputs were 4.7 and 4.6 kg ha^{-1} for SC-1 and SC-2 respectively, and effluxes were 4.9 and 4.8 kg ha^{-1} respectively. Because of the sampling and analytical errors associated with computing Cl^- inputs and losses, there is no reason to suspect these differences are real. From this I conclude there are no substantial spring water inputs or deep seepage losses.

B. Assumption of Thermodynamic Equilibrium

In using streamwater transport of solutes to estimate weathering rates, an implicit assumption is made that the thermodynamic activities of selected aqueous species of the solutes of interest and of water in the stream are, "within acceptable limits," similar to their activity in the zone of most active weathering. What qualifies as "acceptable limits" is difficult to define. In addition, there are well known problems with equating water extracted or freely drained from soils with water held at high energy levels (high matric potentials) in soil pores. Mattigod and Kittrick (1980) point out that the activity of water in soil is highly variable in time and space, and that it can shift mineral

equilibria. Sposito (1981) demonstrated this effect on the Al-K-Si-H₂O system, and showed a marked shift in the stability fields favoring kaolinite or muscovite at the expense of gibbsite as H₄SiO₄ activity decreases, when the activity of water is lowered to commonly found soil values (e.g., [H₂O] = 0.5).

To define an "acceptable limit" I compared the range in activities in streamwater of dissolved species on predominance diagrams to the suspected equilibrium weathering products (Figure 2.2). These diagrams show the favored solid phases in equilibrium with solute activities. Prior research in the Idaho batholith indicated that the common stable weathering product in freely drained soils is kaolinite (Clayton, 1974; Clayton et al., 1979). Predominance diagrams for the systems Na-Al-Si-H₂O and K-Al-Si-H₂O were constructed using thermodynamic data of Robie et al. (1979) for all solid phases except Na-smectite. The thermodynamic data of Feth et al. (1964) were used for reactions involving Na-smectite with other phases. The predominance diagrams were constructed using the assumption that Al is conserved in the system.

To estimate range of activities in stream water (Figure 2.2), I used annual maximum and minimum concentrations of SiO₂, Na, and K, which correspond to sample collections during baseflow and peakflow discharges respectively for water years 1978 to 1980. The estimated activities were not corrected for ionic strength (specific conductance ranged from 12 to 83 μ S) or temperature (range from 277 to 293 K).

For both the Na and K systems, streamwater chemistry appears to be in equilibrium with kaolinite as the solid phase. This suggests that the assumption that chemistry of the streamwater is similar to the chemistry of water in the weathering environment is reasonable.

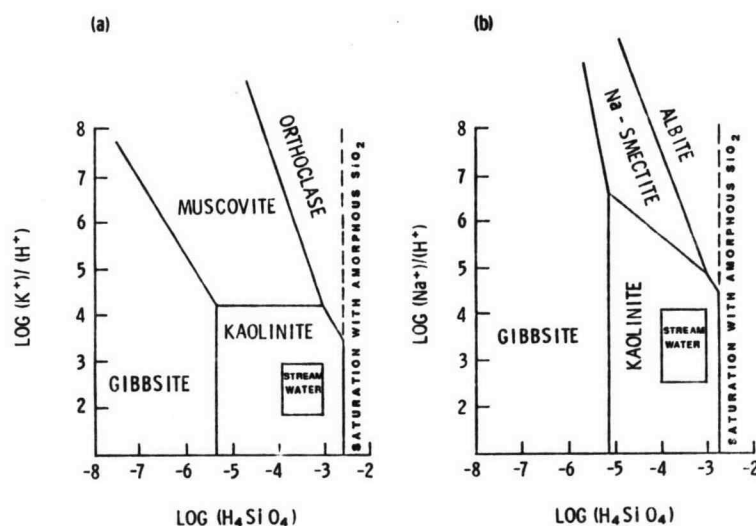


Figure 2.2. Predominance diagrams for the systems K-Al-Si-H₂O (a) and Na-Al-Si-H₂O (b) at equilibrium at 25° C and 100 kPa pressure. Domains of stream chemistry at high and low flows for the four watersheds are plotted on the diagrams. Thermochemical data after Robie et al. (1978) and Feth et al. (1964).

C. Computation of Na and Ca Released by Weathering

Annual precipitation inputs of Na and Ca were computed for the watersheds using an average figure of 110 cm of precipitation per year, and the assumption that 65 percent of this precipitation fell as snow. The 65 percent assumption was based upon data from Megahan et al. (1983) for an erosion study conducted in the Silver Creek

area. From 53 station samples of rain and 11 station samples of snow, Na and Ca concentrations in precipitation, mg/liter were as follows:

	<u>Na, mg/liter</u>		<u>Ca, mg/liter</u>	
	<u>Mean</u>	<u>Std. dev.</u>	<u>Mean</u>	<u>Std. dev.</u>
Rain	0.18	0.15	1.37	0.44
Snow	0.06	0.033	0.22	0.06

Total annual inputs of Na equal $1.12 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{y}^{-1}$ and Ca equal $6.86 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{y}^{-1}$ based upon the concentration and mean annual precipitation data.

The computation of annual effluxes of Na^+ and Ca^{++} dissolved in streamwater requires integration of point sample concentrations and discharge values over time. Regression equations were developed relating instantaneous concentration, C_i and instantaneous discharge, q_i . For Na^+ and Ca^{++} , these equations are of the form:

$$C_i = a \cdot q_i^{-b} \quad [1]$$

Concentrations are negatively correlated with discharge, because dissolution rates do not keep pace with increasing flow volumes during high run-off events. Utilizing the regression equations, daily fluxes were generated from mean daily flows, and these daily fluxes were then summed over a period of a year to compute fluxes in $\text{kg} \cdot \text{ha}^{-1} \cdot \text{y}^{-1}$. This technique averages the prediction error of the regression over 365 or 366 observations of flow, and thus provides a highly reliable estimate of efflux for an unbiased regression model.

Annual efflux data were regressed on annual water yield in centimeters from both watersheds (Figure 2.3). Regressions were positive, linear, and highly correlated ($r^2 > .96$). The long-term average efflux of Na and Ca from the watersheds was computed from these regression equations using the 17-year mean annual water yield

figure in Table 2.1. There were no significant differences in either intercept or slope for the regression of annual flux on annual water yield for the two watersheds, therefore, the data were pooled to compute an average efflux for the 2 watersheds.

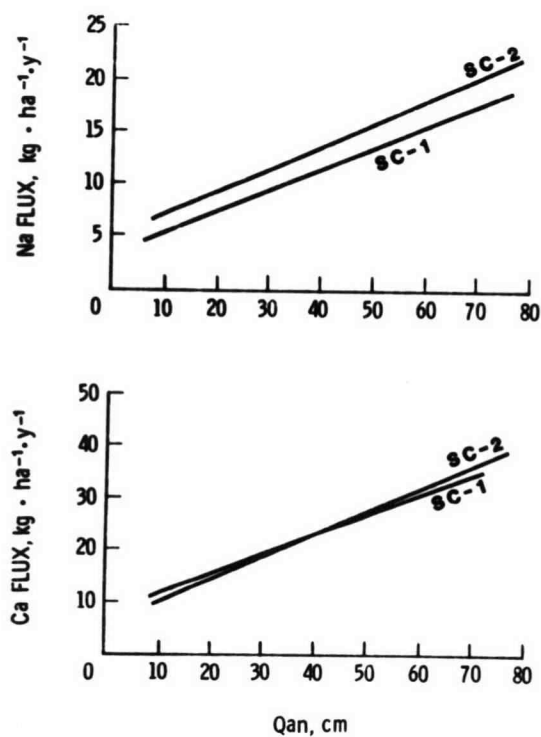


Figure 2.3. Plots of annual efflux of Na and Ca, $\text{kg} \cdot \text{ha}^{-1} \cdot \text{y}^{-1}$ vs. annual water yield in centimeters. Numbers on the regression lines correspond to the watersheds. Regressions are based on 8 years of water chemistry data. Annual water yields ranged from 9 to 73 cm over the 8 years.

Based upon these calculations, the long-term Na efflux averaged $13.6 \text{ kg.ha}^{-1}.\text{y}^{-1}$, and Ca efflux averaged $24.8 \text{ kg.ha}^{-1}.\text{y}^{-1}$. Subtracting meteoric inputs from precipitation resulted in a weathering release figure of $12.5 \text{ kg.ha}^{-1}.\text{y}^{-1}$ for Na^+ and $18.0 \text{ kg.ha}^{-1}.\text{y}^{-1}$ for Ca^{++} .

There are few comparative studies that provide calcium and sodium weathering release data on similar parent materials in western North America. Marchand (1971) did a similar study in the White Mountains in California on a small watershed underlain by quartz monzonite (An_{25-30}). The study site is drier (approximately half the mean annual precipitation of Silver Creek) and cooler than the study reported in this paper. Weathering release of Na^+ was $8 \text{ kg.ha}^{-1}.\text{y}^{-1}$ and Ca^{++} was $17 \text{ kg.ha}^{-1}.\text{y}^{-1}$ in the White Mountains. Zeman and Slaymaker (1978) conducted a similar study in the Coast Range of British Columbia. Their watershed is underlain by a hornblende bearing quartz diorite, so I suspect plagioclase is more anorthitic than in Silver Creek or the White Mountains. In addition, their study area is much wetter, receiving approximately 450 cm of precipitation compared to 110 in Silver Creek and 40 in the White Mountains. Na^+ release by weathering in the Coast Range of British Columbia averaged $12.4 \text{ kg.ha}^{-1}.\text{y}^{-1}$ and Ca^{++} release averaged $34.4 \text{ kg.ha}^{-1}.\text{y}^{-1}$.

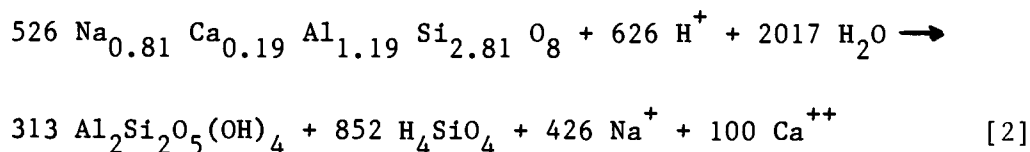
WEATHERING RATE COMPUTATION

Based on the net Na and Ca efflux data, albite ($\text{NaAlSi}_3\text{O}_8$) weathering proceeded at $142 \text{ kg.ha}^{-1}.\text{y}^{-1}$ and anorthite ($\text{CaAl}_2\text{Si}_2\text{O}_8$) weathering proceeded at $125 \text{ kg.ha}^{-1}.\text{y}^{-1}$. This is a best estimate for weathering rate under present climatic conditions and is equivalent to a total mean value for plagioclase weathering of $267 \text{ kg.ha}^{-1}.\text{y}^{-1}$. Annual values for the 17 years of streamflow records ranged from a low of $73 \text{ kg.ha}^{-1}.\text{y}^{-1}$ in water year 1977 to a high of $504 \text{ kg.ha}^{-1}.\text{y}^{-1}$ in water year 1974. The molar Na:Ca ratio computed from the above weathering rates is 1.2, equivalent to incongruent

dissolution of an An_{45} plagioclase. Such a plagioclase is considerably more calcic than previously reported for this area of the batholith (An_{25} , Schmidt, 1964; An_{17-26} , Larsen and Schmidt, 1958).

It is difficult to find outcroppings that are fresh and contain sufficient unaltered plagioclases to characterize the albite-anorthite content in the Silver Creek study area. Furthermore, fresh outcrops commonly have atypical mineralogy, or atypically low fracture density, making them suspect for characterizing the modal plagioclase mineral in the area. Recently excavated sites such as roadcuts provide the only opportunity for sampling fresh rock other than core drilling. Two roads in watersheds between SC-2 and SC-5 run centrally through the study area. These roads have cuts sufficiently deep to expose fresh rock in several locations. Fourteen rocks were sampled along these roads, returned to the lab, and ground in an impact mill in preparation for various analyses. The index of refraction of three to five plagioclase grains lying on the 001 cleavage plane was determined for each specimen. Using the method of Phillips and Griffen (1981), the modal anorthite content of these rocks is An_{16-21} .

A balanced hydrolysis reaction for the weathering of An_{19} plagioclase to kaolinite is:



For this reaction, the sodium:calcium ratio, Na:Ca (molar), released on hydrolysis is 4.3, considerably larger than the Na:Ca of approximately 1.2 found in stream efflux. Thus, a distinct discrepancy exists between the molar ratios of Na:Ca transported in the stream and that predicted from dissolution of An_{19} plagioclase.

This discrepancy could be due to a differential weathering rate for albite and anorthite, or a differential transport rate for Ca^{++} and Na^+ released. A differential transport rate would require a sink for Na^+ somewhere on the watersheds, either in the soil as a secondary weathering product or in organic matter.

The existence of a sink for Na released on weathering is not likely. Some Na^+ is undoubtedly adsorbed on new exchange sites arising from the formation of kaolinite. If we were to assume an exchange capacity of 0.15 moles of charge per kg of kaolinite formed (a reasonable maximum figure for kaolinite), this is only sufficient to retain 1.93 percent of the cation charge arising from Na^+ and Ca^{++} released during hydrolysis. Further, Ca^{++} is preferentially adsorbed (neutral, 1M NH_4OAc extraction) in Silver Creek soils by between one and two orders of magnitude.^{3/} Other sinks such as Na in the lattice of secondary minerals are too small to consider.

Preferential uptake and immobilization of Na relative to Ca by biota is an equally unlikely explanation. Most of the live phytomass in Silver Creek is in old growth timber, and I assume net annual uptake is near zero. One young stand located in SC-5 that is accreting biomass contains nearly five times as much Ca as Na, and annual net uptake of Ca is over six times annual net uptake of Na in this stand. This would result in a larger Na:Ca ratio in stream water relative to that released from weathering. No other large biotic sinks for Na exist.

Cleaves et al. (1970) indexed the annual mass of oligoclase weathering to kaolinite to Na transport in a stream on the basis that Na is highly mobile and minimally affected by a biomass sink. In this study, conducted in the Piedmont of Maryland, plagioclase was the only mineral containing Na. Ca and soluble SiO_2 weathering

^{3/}Data on file, Forestry Sciences Laboratory, Boise, Idaho.

rates were adjusted to conform to Na weathering, which the authors assumed was equal to Na transport. Indexing weathering of An_{19} plagioclase in the Silver Creek area to Na transport would lead to a large underestimate of anorthite and total plagioclase weathering.

A differential weathering rate for albite and anorthite provides a more logical explanation for the discrepancy in the molar Na:Ca ratio of the modal mineral (4.3) and mean annual stream efflux (1.2). Clayton et al. (1979) observed that plagioclase feldspars in the Idaho batholith are commonly zoned and the more calcic core is often preferentially weathered (Figure 2.4). Such preferential weathering of anorthite might initially account for an enrichment in Ca transport. However, this process would also result in a relative

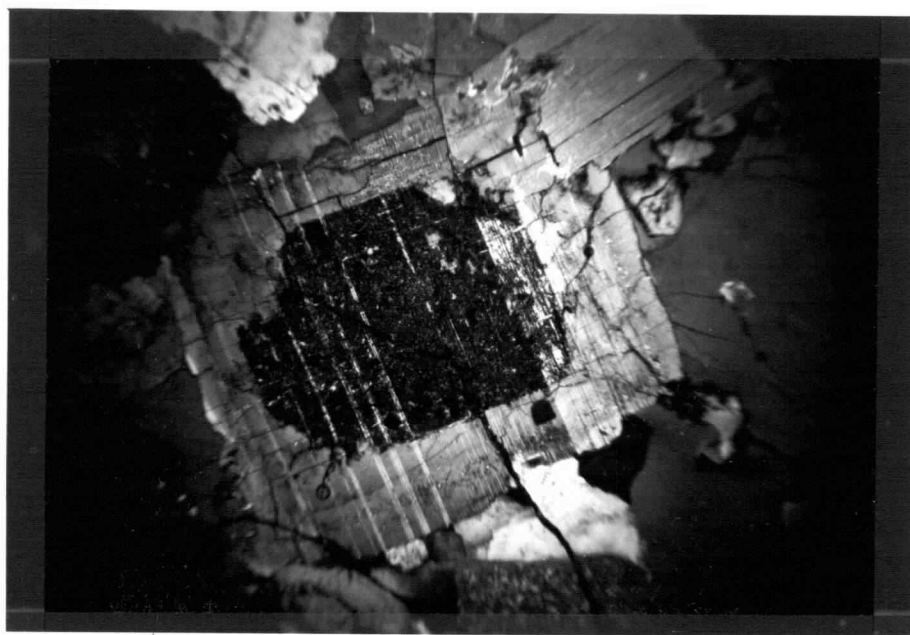


Figure 2.4. Preferential weathering of the more calcic core of a plagioclase to kaolinite. Magnification = 50x.

enrichment of albite in the soil and thus would ultimately be self-limiting. But preferential chemical weathering coupled with moderate year to year erosion rates and episodic high erosion rates might sustain a supply of An_{19} plagioclase to the weathering environment. Clayton (1981) estimated that erosional denudation (sediment transport) in SC-2 (referred to as D Creek op. cit) approximately equaled chemical denudation over 6 years, 1975 to 1980 inclusive. During years with high climatic stresses, erosional denudation greatly exceeds chemical denudation, so it is reasonable to assume erosional processes may remove albite accumulating in a weathering residuum or soil. It is also reasonable to assume that soils might retain the evidence of preferential anorthite weathering and exhibit Na:Ca ratios higher than fresh rock.

Samples of A horizon, C horizon, and bedrock were collected at two sites along the road where the previously described rocks were sampled. Samples of C horizon and rock from both sites were sent to the University of Oregon Department of Geology for x-ray fluorescence (XRF) analysis. Duplicate samples plus the two A horizon samples were digested in our laboratory using a mixture (1:1 vol.) of HF and $HClO_4$. Results of both analyses showed enrichment of Na relative to Ca in the C horizons of the soil compared to the rock samples.

Results of the A horizon analyses showed a 9 percent enrichment in Ca relative to Na. Most likely this results from relative enrichment of Ca in the litter that is continually added to the surface soil. Table 2.2 summarizes the results of the rock and soil analyses.

SUMMARY

Chemical weathering of plagioclase to kaolinite averaged $267 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{y}^{-1}$ and ranged from 73 to $504 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{y}^{-1}$ over a 17 year study in the southwestern Idaho batholith. Preferential weathering

of anorthite resulted in its selective removal relative to albite based upon net export rates of dissolved Ca and Na in streamwater. Supporting data indicated relative enrichment of albite in the soil. Based on two different analysis techniques, the Na:Ca ratio in soil C horizon ranged from 7 to 11 percent greater than the ratio in fresh rock. Erosional denudation of weathered residuum is likely to set an upper limit on albite enrichment in the soil and assures a supply of unweathered plagioclase in the weathering environment, thus sustaining the observed preferential weathering of anorthite.

Table 2.2.--Total Na and Ca analyses on A horizon, C horizon, and rock samples by XRF or HF-HClO₄ digestion and atomic absorption - flame emission spectroscopy.

<u>Analysis technique</u>	<u>Sample</u>	<u>Molar</u>		<u>% Na enrichment in soil</u>
		<u>Na:Ca</u>	<u>(Precision)</u>	
XRF	C	3.39	(±0.032)	7%
XRF	Rock	3.18	(±0.03)	
HF Digest	A	2.77	(±0.14)	(9% depletion)
HF Digest	C	3.26	(±0.16)	11%
HF Digest	Rock	2.94	(±0.15)	

A RATIONAL BASIS FOR ESTIMATING ELEMENTAL SUPPLY RATE
FROM WEATHERING

James L. Clayton

ABSTRACT

Based on a nutrient supply study conducted in the Idaho batholith, I have estimated the amounts of cations released by chemical weathering that are retained in a mixed Douglas-fir ponderosa pine forest ecosystem. The estimate includes cations adsorbed by new secondary minerals formed during weathering plus net nutrient uptake by biomass. This provides a more realistic estimate of potential nutrient supply to soil by weathering than previous published weathering rates that include a large component that is leached from the ecosystem. Maximum rates of nutrient retention occur during maximum foliar increment, a situation approximated by the stand conditions in the study watershed. Net annual uptake of cations in biomass greatly exceeds annual increase in soil retention. Approximately half the K^+ , Ca^{++} , and Mg^{++} released by primary mineral weathering are retained in the ecosystem.

INTRODUCTION

Land managers considering intensive tree harvesting, short rotation management, or even conventional harvesting on depauperate soils are often concerned with the implications of harvest to the nutrient capital of their site. Nutrient loss to the system must be counterbalanced by resupply over a reasonable time, or fertilization is required. Estimates of nutrient supply to ecosystems by primary mineral weathering are uncommon in the literature, although nutrient budget studies frequently include an estimate of weathering release based upon dissolved cation efflux rates from watersheds. The implication is that this weathering rate defines the upper bound of potential elemental supply to the ecosystem. This upper bound greatly exceeds the maximum actual retention of cations released by weathering in temperate forest ecosystems. Actual nutrient supply rates to the system are time variant, and may be considered equal to the net accrual to the soil and biotic pools.

Nutrient budgets in forested ecosystems have been studied intensively over the past 20 years. Seminal papers by Duvigneaud and Denaeyer-DeSmet (1964), Ovington (1962, 1965), and Tamm (1964) pointed to the concerns regarding perpetuation of a supply of nutrients to forested lands in Europe that have been under management (widescale clearing and natural regeneration) for well over 1,000 years. They suggest that the nutrient capital of soils is gradually depleted by successive croppings, and the ultimate and necessary result is a decline in timber production.

In the past decade, study results have generally indicated that tree bole harvest removes nutrients and causes accelerated leaching losses at rates compatible with natural nutrient inputs (Wells and Jorgensen 1979). However, concern has shifted from normal, sawlog harvesting to more complete or total utilization of the tree crop, which greatly increases removal of nutrient capital from the site (Boyle et al. 1973; Weetman and Webber 1972; Malkonen 1973; Kimmins

1977; Alban et al. 1978; Leaf 1979). Weaver and Forcella (1977) estimated that whole tree harvest of climax Rocky Mountain forests would result in increased nutrient drains of 3, 6, 4, and 3 times bole removal rates for N, P, K, and Ca, respectively. Johnson et al. (1982) reported increased export of N, P, K, and Ca from whole tree harvest ranging from 2.6 to 3.3 times conventional bole removal rates from a mixed oak forest at Oak Ridge National Laboratory. They concluded that soil amendments might be necessary to sustain Ca supplies to this ecosystem under whole tree harvest management.

Much of the merchantable timber in the Western United States is located on steep, mountainous landscapes with high natural erosion rates. Soils on these sites are often shallow and weakly developed. Following logging, accelerated erosion can occur, depleting the soil nutrient pools in these ecosystems. Fertilization to replace nutrients in forest ecosystems that have lost productivity can only be considered a practical solution in highly productive stands under intensive management. Kimmins (1977) correctly pointed out that nutrient management would be important in the future.

Nutrient budgeting has been used extensively to predict the long-term consequences of particular harvest strategies. However, nutrient budgeting is still an inexact science, prone to numerous errors. Leaf (1979) expressed the concern that in our haste to provide data on nutrient cycles to land managers, our sampling techniques may have been so crude that we have provided a measure of disservice rather than of service. This may be most apparent for estimates of nutrient supply rates to the ecosystem from primary mineral weathering. Clayton (1979) reviewed a variety of techniques for arriving at estimates of nutrient supply from weathering, and concluded that nutrient budgeting techniques probably provide the current most accurate numbers.

The principal shortcoming of previously published nutrient supply rates is that elemental release rates from weathering, are

considered equivalent to maximum potential supply rates. These supply rates are based on elemental loss from ecosystems by soil solution transport below lysimeters or as dissolved transport in streams, and thus include a component not available for tree nutrition. In temperate forest ecosystems of the world, rates of nutrient release from weathering generally exceed nutrient retention, principally because of: (1) lack of an efficient sink mechanism in the soil for immobilizing elements released from weathering, and (2) lack of an effective exploitation of the soil volume by roots. Obviously, these forest ecosystems have evolved within the nutrient retention limitations of the existing soil sink and root exploitation strategy. The land manager's concern and need for realistic estimates of nutrient replenishment rates is directed to managed forest ecosystems, where large nutrient removals result from harvesting.

A more rational upper bound for potential nutrient supply rate from weathering can be computed from annual net elemental accrual in vegetation and litter, plus annual increases in cation adsorption sites in the soil arising from secondary mineral formation and changes in soil organic matter. Vitousek and Reiners (1975) presented a hypothetical relationship between the degree of maturity of an ecosystem and its ability to retain nutrients. They suggested that the difference between net input and output of a nutrient element is proportional to the rate of accrual of that element into the biomass increment.

The research reported here presents a concept for estimating nutrient gains to a forested ecosystem in southwestern Idaho in which litter layer and soil organic matter content are assumed to be in a steady state. These assumptions are reasonable in view of the age and profile development of the soils, and in view of the forest stand maturity and lack of recent fire activity. A mass balance approach to arrive at a geochemical budget for a watershed is used to investigate the potential and actual supply rates of five

essential elements to a forested ecosystem. The macronutrients Ca, K, Mg, S, and P are considered in this study. Cycling of the nonessential element Na is also considered because of its magnitude and similarity of transport to other cations in the biogeochemical cycle. Results indicate that weathering release of S is considerably lower (not detectable) than meteoric inputs to the Idaho batholith. This result was not unexpected (Strahler and Strahler 1973). Therefore, weathering is not an important pathway to consider in the S budget of this ecosystem. Weathering release rates of P from primary minerals are difficult to determine because of the lack of mobility of P. The rate of supply of P from weathering is discussed in light of this problem.

SITE CHARACTERISTICS, SAMPLING, AND LABORATORY ANALYSIS

This research was conducted on watershed SC-5, one of the Silver Creek research watersheds located in the southwestern Idaho batholith (44°25'N latitude and 115°45'W longitude). SC-5 is a 1.09-km² watershed draining in a southeast direction. Slopes on the watershed range from 20 to 65 percent, and commonly have south or east facing aspects. Elevation at the mouth of the watershed is 1395 m and the highest elevation is 1775 m. Bedrock in this area of the batholith is a coarse-grained quartz monzonite, typical of the main inner facies of the southern Idaho batholith (Ross 1963). The modal rock in SC-5 contains quartz, plagioclase (An₁₆₋₂₁), orthoclase, and minor amounts of biotite (Clayton, in press). Rock near the surface is moderately well-weathered to well-weathered according to the classification scheme of Clayton et al. (1979). These classes of weathering indicate rocks of low mechanical strength, and quartz grains are the only minerals to appear fresh to the unaided eye. Feldspars and biotite show considerable alteration to clay minerals in thin section.

Soils in the area were mapped on Forest Service Resource Photography base photos (1:15,840) at the family level. Soils are

generally shallow, coarse textured, and weakly developed, exhibiting only A and C horizons. Typic and Lithic Xerorthents predominate on southerly slopes. Cryorthents, Cryumbrepts, and Cryopsamments are common on other aspects. Textural classes are sandy or sandy-skeletal. Single samples of major horizons from the four soils most common to SC-5 were sampled from 28 soil pits for laboratory analysis. Bulk densities were determined in the field by coring. Laboratory analyses included:

1. pH - saturation paste
2. Cation exchange capacity by Na saturation (Chapman 1965)
3. Exchangeable bases, neutral, 1 M NH_4OAc extract
4. Exchange acidity, BaClH_2 - TEA method (Peech 1965)
5. Percent organic matter, Walkley - Black method (Allison 1965)

No samples for nutrient analysis were taken below C horizons, although many of the soil contacts with bedrock are classified as paralithic. Both weathering and tree roots are commonly observed in bedrock in the batholith, often to depths greater than 10 m. Roots are generally confined to fractures in rock, but presumably they derive both water and nutrients from this zone during some periods of the year. Primary mineral weathering below the soil is slow compared to rates in the soil because of the abundance of primary minerals in young batholith soils, higher specific surface of grains in the soil, presence of organic acids, more frequent leaching and greater temperature fluctuations (Clayton et al., 1979).

The watershed is forested, and the principal overstory species are ponderosa pine (Pinus ponderosa Laws.) and Rocky Mountain Douglas-fir (Pseudotsuga menziesii Mirb. Franco). The timber stand is uneven aged, multicanopied, and averages about 24 MBF/acre (Scribner) in trees >27 cm diameter at breast height (d.b.h.). The oldest trees are approximately 400 yr in age, and all ponderosa

pine. Intermediate Douglas-fir and ponderosa pine trees are in a 180 to 200 yr age class, and there is a discontinuous pole-sized class of Douglas-fir, 80 yr old. There have not been any fires larger than 0.4 ha in SC-5 in the last 60 yr, and no evidence of major fires for over 80 yr. Basal area averages $25.2 \text{ m}^2 \text{ ha}^{-1}$. Two habitat types (Steele et al. 1981) predominate on the watershed: Douglas-fir/ninebark (Physocarpus malvaceus (Greene) Kuntze), ponderosa pine phase; and Douglas-fir/white spirea (Spiraea betulifolia Pall.), ponderosa pine phase.

Incremental phytomass growth and standing crop estimates for SC-5 were determined in the following way:

Two hundred forty-five 0.008-ha plots were established to provide a 3 percent cruise of SC-5. On all trees, d.b.h. and height by species were recorded; increment cores were taken on a one out of five random subsample to provide the heartwood-sapwood transition and last 5 years' diameter increment. Using the current diameter, diameter increment, and height, various allometric equations predicted incremental growth by plant part. The foliage and branch estimates for ponderosa pine and Douglas-fir are predicted using equations from Brown (1978). Boles of pine <39 cm d.b.h. are predicted using unpublished equations provided by Patrick Cochran, Bend Silviculture Laboratory, Bend, OR. Pine boles >39 cm d.b.h. and Douglas-fir boles and roots are predicted from equations of Gholz et al. (1979). No equations for ponderosa pine roots were deemed suitable for the research area, so they were estimated using the assumption that pine has the same branch:root ratio predicted for Douglas-fir.^{4/} Understory plants other than ponderosa pine and

^{4/}The author is indebted to David Perry, Dept. of Forest Science, Oregon State University for providing the standing crop and incremental growth data.

Douglas-fir constitute <5 percent of the total vascular plant phytomass in the watershed.^{5/}

Tissue chemistry data for ponderosa pine and Douglas-fir from Clayton and Kennedy (1980) are coupled with the standing crop and incremental growth data to provide annual uptake, net uptake, and standing crop of elements in the stand. Half the samples collected by Clayton and Kennedy are from SC-5; the other half were from an adjacent watershed in the Silver Creek study area.

Streamflow and precipitation in SC-5 have been monitored continuously since 1960. Water chemistry studies were begun in 1973 and precipitation chemistry studies in 1974. Stream and precipitation chemistry sampling and techniques for computing annual elemental fluxes into and out of the watershed by these pathways are described by Clayton (in press). Precipitation influxes are computed for each element assuming 98 cm mean annual precipitation, which is arrived at from an isohyetal map of SC-5 based on 18 years of record from three rain gages. Chemical composition of snow is different from rain in Silver Creek, and the volume ratio 65:35 snow:rain is used in computing chemical content (Megahan et al. 1983). Finally, chemical concentrations from 93 station samples of rain and 11 station samples of snow are used to compute elemental inputs in precipitation.

Dissolved effluxes in streamwater are computed using equations that correlate concentration in mg/liter and instantaneous stream discharge in m^3/sec . These equations predict a mean daily flux from mean daily flow. Annual efflux in $kg\ ha^{-1}\ yr^{-1}$ is computed by

^{5/} Estimate made by Russell A. Ryker, Research Forester, Intermountain Forest and Range Experiment Station, Boise, Idaho.

summing mean daily flux over the number of days in the year. Long-term average effluxes are not computed by averaging the 10 years of available record. Instead, I have used a strong relationship that exists between annual efflux and annual water yield in cm, and the 21-year mean annual water yield figure of 33.5 cm for SC-5 to predict long-term average annual efflux. I describe this procedure in greater detail elsewhere (Clayton, 1985).

In order to test if the watershed is tight, i.e., if there are any losses of water and dissolved nutrients to deep seepage below the flume, and to test if there are water sources from springs, the Cl^- budget for the watershed was examined from 1978 through 1981. The sole source of Cl^- input is from precipitation, and there are no large biotic or abiotic sinks for Cl^- in the system. Over the 3-year period, there was a 4 percent excess of Cl^- outflux over influx in precipitation (Clayton, 1985). Because of the errors associated with sampling and computing the fluxes, the significance of the 4 percent difference is unknown.

ANALYSIS OF DATA

Clayton (1979) suggested that a mass balance equation of the form $I_{\text{rw}} = (I_{\text{out}} - I_{\text{in}} [\text{ppt}]) + \Delta I_{\text{p+s}}$ can be used in a watershed study to estimate elemental release from weathering. In this example I_{rw} = annual release of element I from rock weathering, I_{out} = annual dissolved efflux of I in the stream, $I_{\text{in}} (\text{ppt})$ = annual input in precipitation, and $\Delta I_{\text{p+s}}$ = the annual change in storage of element I in the plant and soil compartments of the ecosystem. The following section develops the data base required to solve this equation.

Precipitation and Stream Chemistry

The concentration of nutrients in snow and rain are presented in Table 3.1, with the number of samples and coefficient of variation.

Table 3.1.--Average concentration of various elements collected in rain and snow samples, Silver Creek study area.

CV = coefficient of variation; n = number of samples.

Element	Concentration					
	Snow	CV	n	Rain	CV	n
	mg/liter			mg/liter		
Na	0.06	55	11	0.18	72	93
K	0.09	17	11	0.26	46	93
Ca	0.22	27	11	1.37	32	93
Mg	0.014	26	11	0.064	57	93
S	0.28	29	11	0.68	49	53
P	0.001	54	11	0.009	66	53

Using the 65:35 assumption of snow:rain ratio and using 98 cm as the long-term average annual precipitation, chemical influxes in precipitation are presented in Table 3.2. In addition, stream effluxes and efflux minus influx data are presented in Table 3.2. The difference between efflux and influx is the magnitude of elemental release from weathering that is lost from the ecosystem by leaching. This provides data for the first half of the mass balance equation.

The mobility of elements in the soil-water system or, more generally, the mobility of elements when not tied up by plants greatly influences the flux data presented in Table 3.2. This is particularly the case for phosphorus, which is immobilized in the

Table 3.2.--Annual precipitation influx and stream efflux entering and leaving watershed SC-5.

Element	Stream efflux	Precipitation influx	Difference
	kg ha ⁻¹ yr ⁻¹	kg ha ⁻¹ yr ⁻¹	kg ha ⁻¹ yr ⁻¹
Na	12.6	1.00	11.6
K	3.41	1.53	1.88
Ca	16.1	6.10	10.0
Mg	1.55	0.31	1.24
S	3.22	3.6	-0.38
P	0.049	0.033	0.016

soil in both organic compounds and inorganic complexes with Ca, Fe, Al, and clays (Jackson, 1964). More information about the reaction sequence: primary mineral P → non-labile P → labile P is required to describe P supply rate from weathering. A mass balance of elemental fluxes within the watershed does not supply the detail necessary to estimate primary mineral P release, and the reaction rate of non-labile P → labile P may be more important in controlling available supply rate to tree roots.

Incremental Growth and Net Nutrient Uptake

Table 3.3 presents an estimate of standing crop, average yearly net increment, and yearly mortality for ponderosa pine and Douglas-fir in SC-5.

The annual tissue shedding for foliage is 712 kg ha⁻¹ for ponderosa pine and 549 kg ha⁻¹ for Douglas-fir. New foliage production is 752 and 658 kg ha⁻¹ yr⁻¹, respectively, leading to the net incremental growth figures for foliage presented in the table.

Table 3.3.--Standing crop, annual net increment and annual mortality of uneven aged stands of ponderosa pine and Douglas-fir in SC-5. All values are in $\text{kg ha}^{-1} \pm$ standard error.

	Ponderosa pine			Douglas-fir		
	Standing crop	Annual net increment	Mortality ^{a/}	Standing crop	Annual increment	Mortality ^{a/}
	kg ha^{-1}	kg ha^{-1}	kg ha^{-1}	kg ha^{-1}	kg ha^{-1}	kg ha^{-1}
Foliage	3,844 \pm 403	40.6 \pm 3.6	11.8	3,506 \pm 289	108 \pm 8.0	18.6
Branches:						
<0.64 cm	127 \pm 12	2.1 \pm 0.5	5.0	1,631 \pm 131	48.7 \pm 3.6	10.5
0.64 - 2.5 cm	4,504 \pm 469	37.3 \pm 3.4	23.4	2,694 \pm 227	86.3 \pm 6.5	49.6
2.5 - 7.6 cm	5,446 \pm 678	57.1 \pm 5.6	44.3	1,869 \pm 268	85.8 \pm 10.5	19.6
>7.6 cm	4,502 \pm 787	59.0 \pm 8.8	0	82 \pm 19	4.5 \pm 1.0	0
Sapwood	42,381 \pm 4937	514.4 \pm 48.1	365	42,556 \pm 5,085	1,768 \pm 176	518
Heartwood	9,878 \pm 1,269	130.1 \pm 14.7	40.7	9,164 \pm 1,421	436 \pm 56	112
Bark	7,873 \pm 821	83.5 \pm 7.3	64.2	7872 \pm 928	324 \pm 32	123
Roots	33,049 ^{b/}	425	158	14,227 \pm 1,848	618 \pm 67	173
Total	111,604	1,349	711	83,601	3,479	1,024

^{a/} Mortality figures refer to death only, not annual tissue shedding.

^{b/} Pine roots are estimated using the same root:top ratio as Douglas-fir.

Other incremental growth figures are predicted directly by the allometric equations previously discussed.

Plant chemistry data of Clayton and Kennedy (1980) were used to convert incremental growth and mortality data into net changes in standing crop of nutrients. The budgeting procedure is was tested. The assumption is good for some elements (Ca, Mg) and poor for others (K, P, S). However, those elements that are redistributed through the plant in phloem tissue prior to abscission satisfy a requirement that otherwise would be met through the pathway litter-fall, decay and uptake from the soil. The mass balance equation is not pathway dependent for net uptake, and so the requirement of the plant is the same in either case. The same reasoning is used with regard to root shedding. Sollins et al. (1980) assumed 30 percent of fine roots (≤ 5 mm) died back annually in the western Cascades of Oregon. Presumably, the elemental content of these roots may differ during the year, particularly when comparing the time of active growth to senescence. I chose to take the mean elemental concentration of fine roots and apply it to the net incremental growth to compute net uptake.

Net annual uptake in $\text{kg ha}^{-1} \text{ yr}^{-1}$ based upon the above discussion is as follows: Na = 1.94, K = 2.42, Ca = 10.1, Mg = 1.18, S = 0.59, P = 0.61.

Change in Soil Storage

Elements released from weathering are retained in the soil as: (1) soluble salts present in pore water, (2) adsorbed as ions on exchange sites, (3) incorporated into the lattice of secondary minerals, or (4) incorporated in living or dead organic matter. As pointed out previously, the great majority of P in soils is incorporated in organic compounds or highly insoluble complexes with Al, Ca, and Fe. Most of the S in well-drained soils of humid regions is incorporated in organic matter. Under the assumption that soil

organic matter is in a steady state in SC-5, it appears that S is in a tight economy in Silver Creek. Precipitation inputs ($3.64 \text{ kg ha}^{-1} \text{ yr}^{-1}$) slightly exceed solution losses ($3.22 \text{ kg ha}^{-1} \text{ yr}^{-1}$), and the difference approximates the net annual vegetation requirement ($0.59 \text{ kg ha}^{-1} \text{ yr}^{-1}$). Bedrock in the Idaho batholith is low in S content (Larsen and Schmidt 1958), the modal content ranging from a trace to 0.03 percent. Clayton (1981) estimated denudation in a nearby watershed at $6.3 \text{ mm}/1000 \text{ yr}$. Assuming S release rates parallel denudation rates, weathering would provide less than $0.05 \text{ kg ha}^{-1} \text{ yr}^{-1}$ S using the 0.03 percent figure and a bulk density of 2.6 g/cm^3 for the rock.

The four cations are retained principally on cation exchange sites in the soil. Cations incorporated in organic matter (nonadsorbed) and the lattice of secondary minerals, plus free salts of the cations in pore water, constitute a smaller pool in the soil. Under the assumption of organic matter steady state, this smaller pool represents no net gain to the soil. Clayton (1974) and Clayton et al. (1979) found that kaolinite is the secondary mineral commonly formed in batholith soils from feldspar weathering. Plagioclase weathers directly to form kaolinite. Orthoclase frequently forms a fine-grained dioctahedral mica intermediate, sericite, which subsequently weathers to kaolinite. This intermediary holds some K^+ in unavailable interlayer positions; however the fact that K^+ is leached from the watersheds indicates that it is released in sufficient amounts to occupy new exchange sites and satisfy plant growth requirements.

To estimate the annual change in cation storage in the soil one needs information on: (1) the amount of kaolinite formed annually, (2) the exchange capacity of the kaolinite, and (3) what ratios cations are arrayed on the exchange sites. The amounts of feldspar weathered annually can be estimated from the mass balances on cations from the rock weathering equation, ignoring the change in soil storage for a first approximation. For example, sodium

released annually from weathering of albite is equal to stream losses minus precipitation inputs plus net vegetation uptake. In equation form:

$$\begin{aligned} \text{Na}_{\text{rw}} &= (\text{solution loss} - \text{precipitation input}) + \Delta v \\ &= (12.6 \text{ kg ha}^{-1} \text{ yr}^{-1} - 1.00 \text{ kg ha}^{-1} \text{ yr}^{-1}) + 1.94 \text{ kg ha}^{-1} \text{ yr}^{-1} \\ &= 13.5 \text{ kg ha}^{-1} \text{ yr}^{-1} \end{aligned}$$

Adjusting the mass of Na weathered by the ratio: formula weight of albite/atomic weight of Na results in $149 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of albite weathered. Similar calculations for orthoclase and anorthite give 31 and $138 \text{ kg ha}^{-1} \text{ yr}^{-1}$ weathered respectively. Hydrolysis of 1 mole of albite or orthoclase forms 0.5 mole of kaolinite; 1 mole of anorthite hydrolyzes to form 1 mole of kaolinite. Table 3.4 summarizes the annual feldspar weathering and kaolinite formation rates.

Table 3.4.--Estimated feldspar weathering and kaolinite formation in SC-5 based on annual release of Na^+ , K^+ , and Ca^{++} .

Mineral	Primary mineral weathered		Kaolinite formed	
	$\text{kg ha}^{-1} \text{ yr}^{-1}$	$\text{mols ha}^{-1} \text{ yr}^{-1}$	$\text{mols ha}^{-1} \text{ yr}^{-1}$	$\text{kg ha}^{-1} \text{ yr}^{-1}$
Orthoclase	31	112	56	14
Albite	149	569	284	73
Anorthite	<u>138</u>	<u>495</u>	<u>495</u>	<u>128</u>
Total	318	1,176	835	215

The exchange capacity of kaolinite was estimated as follows. Cation exchange capacity (CEC) of A and C horizons from the four common soils in SC-5 were plotted versus percent organic matter. CEC was considered the sum of the exchangeable bases plus exchange acidity. A linear least squares fit ($r^2 = 0.84$) gave an intercept at 0 percent organic matter of 0.009 moles of charge per kg of soil. Finally, the CEC was corrected for percent clay in the soil. Clay

content ranges from 5 to 10 percent; modal clay content is 8.5 percent. CEC of the clay fraction was computed to be $100/8.5 \times 0.009$ or 10.5 cmoles of charge per kg of clay. This rather gross technique for computing the CEC of kaolinite provides a reasonable value, near the midpoint of the accepted range of CEC values for kaolinite presented by Van der Marel (1958). Assuming $215 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of kaolinite formed, this yields 22.6 moles of charge per ha annually.

I arrayed cations on the new exchange sites to correspond with the mean saturation percentage of each cation from the lab analyses of SC-5 soils. These means are weighted by areal occurrence of soils mapped in the watershed. These percentages were determined by setting the sum of the exchangeable bases plus exchange acidity equal to 100 percent. Table 3.5 presents the data on annual cation accretion retained on new exchange sites on kaolinite created from feldspar weathering.

I also considered cyclic changes in cations stored in the soil pool associated with periods of rapid or slow stand growth. One might expect depletion of soil Ca or K during periods of rapid nutrient accrual to the stand, and subsequent replenishment of the soil pool following stand maturity. I was unable to document this by comparing the same soils in an adjacent watershed under a mature forest stand to soils in SC-5. Apparently, cation release from primary mineral weathering is capable of sustaining sufficient cation activity to maintain similar saturation percentages of adsorbed cations in spite of vegetation demand. Alban (1982) found a similar situation when comparing soil K pools under Populus and Pinus stands in which Populus had three times the demand for K. Similarly, they suggested that release of K from nonexchangeable mineral forms had kept pace with uptake.

Table 3.5.--Estimated cation distribution adsorbed on kaolinite formed from feldspar weathering assuming 22.6 moles per ha of new exchange sites created annually.

Cation	Percent saturation	Moles of cation retained	Mass retained kg ha ⁻¹ yr ⁻¹
Na+	0.3	0.068	0.0016
K+	3.0	0.68	0.027
Ca++	25.9	2.93	0.117
Mg++	5.7	0.64	0.016
H+ (acidity)	65.1	--	--

DISCUSSION

Table 3.6 compares masses of the four cations released from weathering to the masses retained in the ecosystem as annual accrual to the soil and vegetation pool. The percentages of cations released by weathering that are retained in the ecosystem, either in incremental biomass growth or on new cation exchange sites, are presented as "Percent efficiency" in Table 3.6. It is readily apparent that under the present stand structure, the plant pool is accreting these cations in much larger amounts than is the soil. Further, the data in Table 3.6 suggest that following a disturbance such as fire or clearcutting, the successional dynamics of plant reestablishment will control the rate of cation accretion to the ecosystem. The rates of nutrient supply presented in this paper should be evaluated in light of the current stand growth dynamics, and what we know about successional dynamics.

Table 3.6.--Total weathering release of Na, K, Ca and Mg, and amount retained in the soil and vegetation as annual incremental gains to these two pools. Percent efficiency refers to the percent of cation released that is retained in the ecosystem.

Element	Released by weathering	Retained by vegetation	Retained by soil	Efficiency
	kg ha ⁻¹ yr ⁻¹	kg ha ⁻¹ yr ⁻¹	kg ha ⁻¹ yr ⁻¹	percent
Na	13.5	1.94	0.002	14
K	4.3	2.42	0.03	57
Ca	20.1	10.1	0.12	51
Mg	2.4	1.18	0.02	50

The direction of succession in Douglas-fir/ninebark habitat types in the Idaho batholith has been studied in some detail. From this one can speculate about succession in the Douglas-fir/spirea habitat type, and thus, have a good picture of succession in SC-5. This report provides data on species coverage changes with time following disturbance by fire or logging. In SC-5 we might expect greater than 90 percent coverage within 10 years of a disturbance. Species present would depend on whether or not fire played a role in the disturbance. For example, following burning, Ceanothus velutinus might cover 60 to 80 percent of the area; without burning (but with considerable ground disturbance) Ceanothus velutinus might be present on only 5 percent of the area with a concomittant increase in Ribes viscosissimum and Ribes cereum. There were slight increases in average shrub canopy volume 5 years following clear-cutting and burning in a watershed adjacent to SC-5. Restocking of conifers is slow and coniferous biomass increment is less than that of shrubs for at least 20 years.

As conifers slowly replace shrubby vegetation, incremental growth rates are maintained at a high level or increase. The structure of the stand is important at this point in determining nutrient increment. Stands with large crown ratios require abundant nutrients, so well-stocked stands with good spacing between trees probably present a situation of high nutrient increment. Watershed SC-5 is carrying somewhat low volumes due to understocking in lower diameter classes. Density related mortality is occurring on only 13 percent of the area, although all plots with mortality were in the larger diameter classes. Timber stand improvement might increase nutrient increment in SC-5. However, the data presented here may be a reasonable maximum for a natural stand in the Silver Creek area.

THE NUTRIENT BUDGET IN A MIXED CONIFER FOREST IN THE
IDAHO BATHOLITH - HARVEST IMPLICATIONS

James L. Clayton and Debora A. Kennedy

ABSTRACT

Nutrient budgets for a mature ponderosa pine/Douglas-fir (Pinus ponderosa Dougl./Pseudotsuga menziesii (Mirbel) Franco var. glauca (Beissn.) Franco) forest in the southwestern Idaho batholith are used to evaluate the potential for accelerated nutrient losses associated with timber harvesting. Soils formed from granitic rock in the Idaho batholith are commonly shallow, coarse textured, and highly erodible following disturbance. Cycling and annual budgets for K, Ca, Mg, S, P, and N are described for an undisturbed control watershed, and accelerated losses associated with clearcutting, helicopter yarding, and broadcast burning of slash are evaluated in an adjacent treated watershed. There were small but statistically significant increases in dissolved N losses of approximately 10 times pre-harvest rates over a 4-year period following treatment. Dissolved transport of other elements was not increased. Small but significant increases in sediment nutrient transport of all elements occurred 3 years following harvest. The largest losses in nutrients were due to bole removal and ranged from 4 percent of total ecosystem N to 21 percent of ecosystem K. Rates of replacement from precipitation, primary mineral weathering, and N_2 fixation would restore the ecosystem to the nutrient status prior to harvest in 50 years. Based on these findings, logging systems that minimize erosion should not cause unacceptable nutrient loss over a normal timber stand rotation.

Additional Index Words: nutrient cycling, granitic soil, accelerated nutrient losses.

Timber harvesting, slash disposal, and site preparation activities reduce the nutrient capital of forested ecosystems. Nutrient loss is of greatest concern where intensive harvesting systems and site preparation are used (Leaf, 1979). At an extreme, Freedman et al. (1981) estimated as much as 500 percent of soil available N and 300 percent of soil available Ca were removed in a whole-tree clearcutting operation in a red spruce balsam fir stand in central Nova Scotia. The results of several studies have indicated that normal tree bole harvesting removes nutrients and accelerates leaching losses at rates compatible with natural nutrient inputs (Wells and Jorgensen, 1979). In the steep, mountainous western United States, however, logging often accelerates erosion, and there is a potential for unacceptable productivity loss caused by nutrient removal from harvest, coupled with accelerated transport of nutrients in solution and sediment.

Timber harvesting often results in accelerated loss in solution of one or more nutrients for a period of several days to more than a year (Gessel and Cole, 1965; Fredriksen, 1971; Pierce et al., 1972; Brown et al., 1973; Hornbeck et al., 1973; Likens et al., 1977; Swank and Waide, 1979; Feller and Kimmins, 1984). Several elements have been shown to increase in concentration in streamwater following disturbance, and the magnitude, duration and species increasing are often quite site specific.

Dissolved NO_3^- losses almost universally increase following disturbance, although the levels and duration of effect are highly variable. Vitousek et al. (1979) and Vitousek and Melillo (1979) suggest several processes that could reduce NO_3^- loss from disturbed forests, (increased vegetative regrowth to immobilize NO_3^- , nitrification inhibitors, lack of water for transport); however, they suggest that more fertile sites have a high potential for NO_3^- loss following disturbance in any case. Accelerated solution loss of other elements has been documented following normal timber harvest practices. Gessel and Cole (1965) documented excessive leaching of

potassium and calcium in the Cedar River Watershed, Washington, measured by tension lysimeters below the root zone. Fredriksen (1971), working with old growth Douglas-fir in the H.J. Andrews Forest, Oregon, showed significant increases in solution loss of $\text{NH}_3\text{-N}$ and Mn following timber harvest. Brown et al. (1973) showed slight, significant increases in K loss in the Coast Range of Oregon following clearcutting Douglas-fir and burning the slash. Aubertin and Patric (1974) reported slight increases in soluble o-PO_4^{-3} following clearcutting in the Fernow Experimental Forest, West Virginia.

Accelerated nutrient loss associated with increased erosion and sedimentation following timber harvest has received only minimal study compared to nutrient loss in solution. Removal of protective cover inevitably increases surface erosion of forest soils (Megahan and Kidd, 1972; Mersereau and Dyrness, 1972). Most studies indicate that yarding, not tree felling, causes soil disturbance and erosion (Brown and Krygier, 1971; Fredriksen, 1970). Dyrness (1972) and Clayton (1981) have shown that aerial yarding systems cause considerably less soil disturbance than ground skidding.

Timber cutting can increase the potential for mass erosion on steep slopes. Harvesting reduces interception and transpiration of water, resulting in increased piezometric levels on a slope (Gray and Megahan, 1981). Harvest also lowers apparent soil cohesion by reducing root tensile strength through decay (Burroughs and Thomas, 1977; Swanston and Walkottem, 1970; Waldron et al., 1983). Megahan et al. (1978) indicate that in the Idaho batholith landslides are most common 4 to 10 years following logging, ostensibly due to the lag time required for root decay.

Data on nutrient loss associated with accelerated erosion, and transport of nutrients bound to sediment are not widely available. Fredriksen (1972) addressed the question, but reported minimal solid phase nutrient loss values associated with sediment. He attributed

this in part to a sampling problem associated with the episodic nature of major sediment-producing events. DeByle and Packer (1972) reported adsorbed cation losses ranging from 0.5 percent to 2 percent of the total amount available in the surface 30 cm of soil. Their study was conducted in a mixed stand of old-growth Douglas-fir, Engelmann spruce and western larch in Montana, on moderate slopes averaging 24 percent. We may suspect that on steeper slopes and with more erosive soils, potential nutrient loss accompanying erosion is of greater consequence; however, there are few data to support this contention.

Nutrient loss accompanying slash burning by volatilization and conversion can be considerable. Few studies provide quantitative data on atmospheric losses from burning, but a recent paper by Feller and Kimmins (1984) from southwestern British Columbia suggests that losses of cations, P, and particularly, N, may be substantial. Atmospheric loss of N accounted for 75 percent of total ecosystem N loss including log export.

Central Idaho is largely occupied by the Idaho batholith, an area of steep slopes overlain with highly erodible decomposed granitic rock. Although the batholith holds considerable volumes of commercial timber, logging road construction has triggered such massive landslides that timber harvesting on steep slopes is often done with the aid of large helicopters. The landslide hazard attenuated, the question remains: are nutrients being seriously depleted through removal of logs, the burning of slash, and through accelerated surface erosion that accompanies even helicopter logging? Although results of other studies suggest that on stable, well-vegetated mountainous terrain, timber harvesting poses no serious threat to nutrient budgets, few studies were conducted in areas similar to the batholith ecosystem. To answer these questions, the USDA Forest Service initiated a study addressing these concerns. Nutrient losses from bole removal, and accelerated dissolved and sediment losses are evaluated in light of nutrient

replacement rates from precipitation, N_2 fixation, weathering and nutrient capital remaining on-site following treatment.

MATERIALS AND METHODS

Study Area

The research reported here was conducted in the Silver Creek Study Area ($44^{\circ} 25' N$ latitude; $115^{\circ} 45' W$ longitude), Boise National Forest in southwestern Idaho. Several paired-watershed and site specific studies on the environmental effects of timber harvesting and roading are currently underway in the Silver Creek Study Area (Figure 4.1). Watershed SC-6, the treated watershed, is 163 ha in area. The adjacent watershed, SC-5, is 110 ha in area and served as an untreated control for this study. The vegetation, soils, aspect and hydrologic response to storms and snowmelt are similar for the two watersheds based on soil, vegetation and habitat type surveys, and hydrologic data for the two watersheds (unpublished). Bedrock in the area is a coarse-grained quartz monzonite, and is typical of a large portion of the central and southern portions of the Idaho batholith (Ross, 1963).

Soils of the Idaho batholith are coarse textured, lack cohesion, and exhibit a high potential for accelerated erosion following disturbance. Four families of soils make up a mosaic in the watersheds and are distributed primarily on the basis of slope and aspect. On southerly slopes Sandy-Skeletal, Mixed Typic Xerorthents predominate. On other aspects we find Sandy-Skeletal, Mixed Typic Cryorthents, Sandy-Skeletal Mixed Typic Cryoborolls and Mixed Alfic Cryopsamments. All soils have one or more A horizons, typically with a total thickness ranging from 100 to 250 mm overlying C horizons. Surface and subsoil horizons are commonly glos or gcos1 textures. O horizon depths range from <10 mm to >120 mm, and averaged 23 mm on SC-6 prior to treatment (Clayton, 1979).

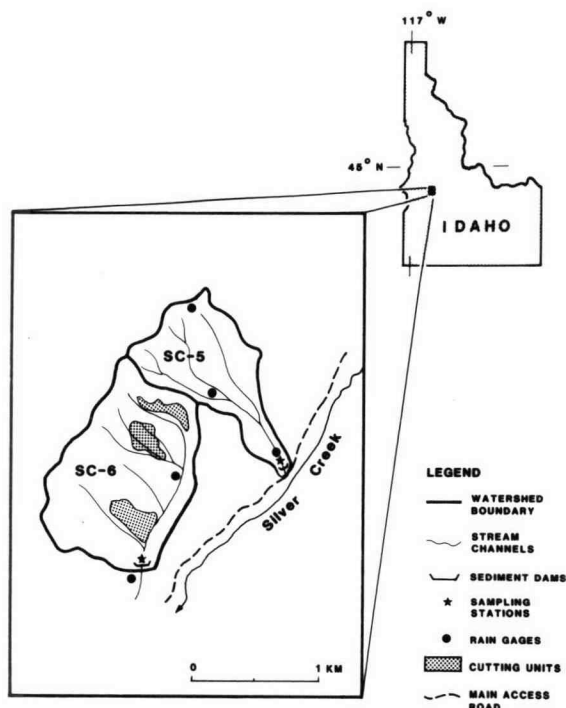


Figure 4.1. Map showing the location of the two study watersheds, cutting units, and precipitation, streamwater, and sediment sampling locations.

The majority of O horizon material is slightly decomposed needles and twigs. Bedrock contacts are generally shallower than 1 m.

Two habitat types (Steele et al., 1981) predominate on the watersheds: Douglas-fir/ninebark, ponderosa pine phase, and Douglas-fir/white spirea, ponderosa pine phase. The stands contained similar volumes of pine and Douglas-fir prior to harvest.

Treatment

South slopes on the watershed were clearcut and the logs were yarded by helicopter to minimize site damage. This is a common yarding practice on steep (>60 percent) slopes in the Idaho batholith. Logging began in September and was completed in November 1976. The logging prescription was a 250-mm diameter limit cut, which essentially resulted in a clearcut because the stand was composed primarily of overmature ponderosa pine and Douglas-fir. Stand basal area on the cutting units was $26.2 \text{ m}^2/\text{ha}$ prior to cutting; residual stand basal area following harvest was $3.3 \text{ m}^2/\text{ha}$. Total area of the watershed is 163 ha of which 38 ha or 23 percent was cut. All cutting units were on southerly aspects. Buffer strips bordered the cutting units, and averaged 15 m adjacent to first- or second-order stream channels, and 30 m adjacent to the third order (main) stream channel. Volume of timber removed was $263 \text{ m}^3/\text{ha}$, corresponding to a merchantable volume of 23.4 M bd. ft/acre. Slash was lopped, scattered, then broadcast burned. The burn was somewhat incomplete and resulted in 22 percent of the soil surface being covered by unburned or partially burned logs, limbs and needles. Average depth of unburned slash was 100 mm (Clayton, 1981).

Sampling and Laboratory Procedures

Sediment

Sedimentation rates have been determined at the mouths of both watersheds since 1962 by surveying the volume of sediment retained behind dams. Surveying was done in the spring following snowmelt and again in the fall. At the time of each survey, core samples were taken to provide estimates of bulk density and organic matter percent determined by wet combustion (Allison, 1965). The number of core samples varies with the volume of sediment retained, and has ranged from five to 30. The trapping efficiency of the dams on both

streams were checked by sampling for total sediment load passing the dam spillways over a variety of flow events. Efficiencies were found to correlate well with the total volume of sediment retained during a given sample interval. During low runoff years, low sediment volumes accumulate, and a relatively high percentage of sediment is organic. Because of the low density of organic sediment, the trapping efficiency of the dams is relatively low. During high runoff periods, a large percentage (>95 percent) of sediment transported to the dams is coarse granitic material, and the dams are quite efficient at trapping this material. Trap efficiencies for lithic plus organic sediment ranged from 23 to 97 percent over the life of the study. We used the relationships developed between total dam catch (volume basis) and trap efficiency to predict actual volume of sediment yield for each dam. We consider nutrients transported with sediment to include cations adsorbed on lithic and organic exchange complexes, plus all nutrient elements contained in organic sediment. This definition precludes elements in the lattice of primary or secondary minerals. Exchangeable cations were determined on the <2 mm fraction of lithic sediment by extraction with neutral, 1 M NH_4OAc , followed by flame emission or absorption spectrophotometry. Organic sediment was separated from lithic sediment using a flowing air flotation chamber. The organic fraction was digested in $\text{HClO}_4\text{-HNO}_3$ according to the technique of Johnson and Ulrich (1959), and the digest analyzed for cations as above, and total P by the molybdate blue-ascorbic acid method (Greenberg et al., 1980). Total S was analyzed using a Leco induction furnace by the technique of Tiedemann and Anderson (1971), and N by Kjeldahl digestion with salicylate followed by distillation and acid titration (Bremner, 1965).

Soil and Forest Floor

Soils on both watersheds were mapped at the family level on Forest Service resource photography base photos (1:15,840). Clayton (1984) describes the sampling procedures and laboratory analyses

conducted on the soil and litter samples. Soil data used in this paper include: neutral 1 M NH_4OAc extractable bases, total S, Kjeldahl N, and bicarbonate extractable P. Quantity of litter under undisturbed conditions and litter chemistry were studied in SC-5. Pre-harvest and post-treatment litter depths only are available for the treated portions of SC-6 (Clayton, 1981). Litter present in SC-5 was estimated from 280 plot samples established on a systematic grid. A 0.2-m^2 rectangular frame was centered over each grid point, and all O horizon material was cut inside the frame and weighed, and subsamples were taken for moisture determination. The mean mass of the 280 plot samples was used as the average litter present on watershed SC-5. Twenty-four randomly selected subsamples of litter from the 280 plots were analyzed for nutrients. Subsamples were oven-dried at 70°C , ground in a Wiley Mill to pass a 40-mesh screen, then digested in a mixture of $\text{HClO}_4\text{-HNO}_3$ according to the technique of Johnson and Ulrich (1959). Digests were analyzed for K (flame emission spectroscopy), Ca and Mg (flame atomic absorption spectroscopy), and total P (molybdate blue - ascorbic acid method). Dried and ground litter samples were analyzed for sulfur and Kjeldahl N using the same techniques as for sediment.

Precipitation

Precipitation chemistry has been sampled at four locations in the Silver Creek Study Area using bulk (dryfall plus wetfall) collectors since 1974. During the period April through October, precipitation samples were collected for analysis whenever a sufficient volume had accumulated. During the winter months, fresh snow samples were collected monthly from snow course sites (1420 and 1680 m elevations) for analysis. Precipitation samples were analyzed for total Kjeldahl N, $\text{SO}_4^{2-}\text{-S}$ (turbidimetric), o-PO_4^{3-} , Ca^{++} , Mg^{++} , and K^+ . Volumes of precipitation input were computed from an isohyetal map of the two watersheds based on 18 years of record for three rainages.

Stream

Stream samples were collected at biweekly intervals from the end of snowmelt until early November (first nonmelting snowfall), and at monthly intervals during the winter snow accumulation period. During the active snowmelt runoff period, streams were sampled more frequently to provide a good sample of the melt hydrograph. The interval between these samples ranges from daily to every 3 days. Samples were either preserved with a mixture of HOAc and formaldehyde, chloroform, or left unpreserved, depending on the subsequent chemical analyses. Preservation techniques are according to standard methods of analysis (Greenberg et al., 1980). Stream samples were analyzed for dissolved concentrations of the three cations as above; Kjeldahl N, $\text{NH}_3\text{-N}$ by the phenate method (Greenberg et al., 1980), NO_3^- plus NO_2^- -N by cadmium reduction (Wood et al., 1967), SO_4^{2-} -S and o-PO_4^{3-} as above. The NO_3^- plus NO_2^- are reported as NO_3^- -N hereafter because NO_2^- is rarely detectable in our watersheds. Annual water yields were computed from continuously recorded stream-flow records using water-level recorders and Parshall flumes.

Vegetation

Two hundred forty-five 0.008-ha plots were established to provide a 3 percent cruise of SC-5 to determine incremental phyto-mass growth and standing crop for trees. On all trees diameter at breast height (1.3 m) and height by species were recorded; on a one-out-of-five random subsample, increment cores were taken to provide the heartwood-sapwood transition and previous 5 years' diameter increment. Volume of wood (boles) exported from SC-6 during harvest was computed from variable plot samples taken prior to and immediately following harvest. One hundred twenty-four plots were sampled prior to harvest, and 65 following harvest.

The field data on tree diameter increment and current height and diameter were used in a variety of allometric equations to

predict incremental growth by plant part as follows. The foliage and branch estimates for ponderosa pine and Douglas-fir were predicted using equations from Brown (1978). Boles of pine <390 mm d.b.h. were predicted using unpublished equations provided by Patrick Cochran, Bend Silviculture Laboratory, Bend, Oregon. Pine boles >390 mm d.b.h. and Douglas-fir boles and roots were predicted from equations of Gholz et al. (1979). No equations for ponderosa pine roots were available, so they were estimated using the assumption that pine has the same branch:root ratio predicted for Douglas-fir. Mortality was estimated using the Reineke relation between mean diameter and stocking density (Reineke, 1933). Understory plants other than ponderosa pine and Douglas-fir constitute <5 percent of the total vascular plant phytomass in the watershed and were ignored (R. Ryker and R. Steele, personal communication).

Tissue chemistry data for ponderosa pine and Douglas-fir from Clayton and Kennedy (1980), and previously unpublished root chemistry data from Silver Creek were coupled with the standing crop and incremental growth data to provide annual uptake, net uptake, and standing crop of elements in the stand.

Data Analysis

Streamflow and sediment

Dissolved elemental fluxes from SC-5 and SC-6 are computed using equations that correlate elemental concentration in mg liter^{-1} and instantaneous stream discharge in $\text{m}^3 \text{sec}^{-1}$. Utilizing these regression equations, daily fluxes were generated from mean daily flow, and these are summed over a period of a year to compute annual fluxes in $\text{kg ha}^{-1} \text{y}^{-1}$. This technique averages the prediction error of the regression over 365 observations, and thus provides a highly reliable estimate of efflux for an unbiased regression model. Average annual fluxes are computed using 21-year-average water yields by using the highly correlated linear relationship between

single year flux and water yield in cm. This allows us to use the 21 year streamflow record for both watersheds, which we believe is more representative of long-term trends than the 10 years of water chemistry data that are available.

Precipitation

Nutrient inputs in precipitation are computed for each element using the weighted mean annual precipitation, value of 100 cm for SC-5. Chemical composition of snow is different from rain in Silver Creek, and the volume ratio 65:35 snow:rain is used in computing chemical content, based on data from Megahan et al. (1983). Chemical concentrations from 93 samples of rain and 11 samples of snow are used to compute elemental inputs in precipitation.

The solution loss and sediment loss figures are measured at the watershed mouth, but onsite leaching and erosion data are not incorporated in the budgets. This implies that the sediment source is from erosion of the soil onsite and therefore represents a loss of available nutrients from the soil pool. Some of the sediment is from riparian zone bank sloughing, and much of it is from very localized disturbances. Prorating this loss over the entire watershed (or to the cutting units in the case of accelerated erosion) is a sacrifice in accuracy done for the sake of simplicity of presentation. We checked the similarity in solution chemistry, measured with porous ceramic tension lysimeters at 0.9 m depth, to stream chemistry during snowmelt for 2 years, and we found the correspondence generally satisfactory. Swank and Caskey (1982) found differences in dissolved NO_3^- -N concentration with distance transported in a stream due to denitrification. We are currently conducting experiments on this in the Silver Creek Study Area.

RESULTS AND DISCUSSION

Undisturbed Forest Nutrient Budgets

The nutrient budgets for SC-5 are based on input estimates from precipitation, primary mineral weathering, N_2 fixation; loss estimates from stream solution and sediment transport; and intrasystem cycling in vegetation. Inputs from precipitation and stream losses are from direct measurements as discussed previously. Weathering inputs of K, Ca and Mg are computed by difference using a mass balance equation in which the element released is equal to solution loss minus precipitation input plus change in storage in vegetation and soil. Clayton (1984) showed that annual storage changes in soil for cations released by weathering can be ignored in SC-5 because new exchange sites constitute a very small sink (22.6 moles of charge (+) per hectare annually, which will retain slightly over $100 \text{ g ha}^{-1} \text{ y}^{-1}$ for the sum of all cations). In contrast annual vegetation increment is very important to the weathering release calculation. Weathering release of N and S is considered zero because both elements are in extremely low concentrations in coarse grained quartz monzonite rocks in the Idaho batholith (Larsen and Schmidt, 1958). Weathering inputs of P using the mass balance technique were attempted but are considered unreasonably high as discussed below.

Intrasystem cycling of nutrients in vegetation and return to the forest floor is computed using the biomass increment data in Table 4.1. The incremental growth models indicate annual additions of foliage for ponderosa pine and Douglas-fir of 752 and 658 $\text{kg ha}^{-1} \text{ y}^{-1}$, respectively, and annual foliage shedding of 712 and 549 $\text{kg ha}^{-1} \text{ y}^{-1}$, respectively based on an average needle retention

Table 4.1.--Standing crop, annual net increment and annual mortality of uneven-aged stands of ponderosa pine and Douglas-fir in SC-5. All values are in $\text{kg ha}^{-1} \pm \text{standard error}$.

	Ponderosa Pine			Douglas-fir		
	Standing crop	Annual increment	Mortality [†]	Standing crop	Annual increment	Mortality [†]
 kg ha^{-1}					
Foliage	3,844 \pm 403	40.6 \pm 3.6	11.8	3,506 \pm 289	108 \pm 8.0	18.6
Branches:						
<6.4 mm	127 \pm 12	2.1 \pm 0.5	5.0	1,631 \pm 131	48.7 \pm 3.6	10.5
6.4 - 25 mm	4,504 \pm 469	37.3 \pm 3.4	23.4	2,694 \pm 227	86.3 \pm 6.5	49.6
25 - 76 mm	5,446 \pm 678	57.1 \pm 5.6	44.3	1,869 \pm 268	85.8 \pm 10.5	19.6
>76 mm	4,502 \pm 787	59.0 \pm 8.8	0	82 \pm 19	4.5 \pm 1.0	0
Sapwood	42,381 \pm 4,937	514.4 \pm 48.1	365	42,556 \pm 5,085	1,768 \pm 176	518
Heartwood	9,878 \pm 1,269	130.1 \pm 14.7	40.7	9,164 \pm 1,421	436 \pm 56	112
Bark	7,873 \pm 821	83.5 \pm 7.3	64.2	7,872 \pm 928	324 \pm 32	123
Roots	33,049 ^{††}	425	158	14,227 \pm 1,848	618 \pm 67	173
Total	111,604	1,349	711	83,601	3,479	1,024

[†] Mortality figures refer to death only, not annual tissue shedding.

^{††} Pine roots are estimated using the same root:top ratio as Douglas-fir.

time of 5 years (D. A. Perry, 1982, personal communication). For tissue other than foliage, concentrations of elements (Clayton and Kennedy, 1980) multiplied by annual increment are considered uptake. This is the case for roots also because we have no annual root shedding data. Our phytomass data don't provide foliage mass by needle age class, although we have needle chemistry data for three age classes plus recently abscised needles (shaken from trees). We chose to use the mean nutrient concentration of three age classes of foliage multiplied by the annual additions of foliage as the gross annual foliar requirement, and these same concentrations multiplied by foliage shedding as the foliar contribution to litterfall. This procedure ignores internal recycling prior to abscission and thus overestimates both nutrient return in litterfall and uptake. This is a pathway dependent error, and does not affect the overall budget. The magnitude of this error can be approximated by comparing the mean foliar nutrient concentrations for the three age classes of green needles with abscised needles (Table 4.2).

Table 4.2.--A comparison of nutrient concentrations in green needles (mean of three age classes) to recently abscised needles collected in SC-5.

Element	Ponderosa pine		Douglas-fir	
	green	abscised	green	abscised
.....percent by weight.....				
Na	0.037	0.085	0.050	0.089
K	0.739	0.300	0.655	0.166
Ca	0.176	0.380	0.786	1.04
Mg	0.107	0.126	0.139	0.086
P	0.159	0.092	0.123	0.068
N	1.39	0.69	1.17	0.55
S	0.108	0.063	0.087	0.054

Increased concentrations of Na and Ca in abscised needles probably reflect mass loss from carbohydrate and nutrient translocation plus water loss. These concentrations are greater in pine than Douglas-fir. If the mass loss causes an approximate doubling of Na and Ca concentrations, then we might estimate that greater than 75 percent of N, K and S is translocated prior to abscission, and greater than 50 percent of the Mg and P content is translocated. Because of the magnitude of this internal cycling process and our inability to separate it from uptake, we have presented the gross plant requirement as uptake plus translocation (+), and consequently the annual foliage shedding as litterfall plus translocation (-). These relationships are presented as a budget in Table 4.3.

The difference between inputs and losses is accounted for by annual vegetation increment for the elements K, Ca and Mg. This is an obvious result of the fact that we computed primary mineral weathering by difference. The major pathway for supply of K, Ca and Mg is by primary mineral weathering. Relatively large amounts of Ca are supplied by weathering of anorthite in An_{19} plagioclase (Clayton, 1985). The main source of K is from weathering of orthoclase. The orthoclase content of the modal rock in Silver Creek is approximately equal to plagioclase; however, orthoclase weathering rates are nearly an order of magnitude slower than anorthite rates. The principal source of Mg is from biotite weathering. The Mg content of fresh rock is 0.33 percent by weight, compared to 1.64 percent Ca and 1.65 percent K based on X-ray fluorescence data.

For N the difference in system inputs and losses approximates annual increment using any of the reasonable range of published published values for N_2 fixation from the same habitat types in the

Table 4.3.--Nutrient budgets for six elements in a mixed ponderosa pine Douglas-fir forest, Silver Creek, Idaho.

INPUTS	Element					
	K	Ca	Mg	S	P	N
	kg ha ⁻¹	kg ha ⁻¹	kg ha ⁻¹	kg ha ⁻¹	kg ha ⁻¹	kg ha ⁻¹
1. Precipitation	1.5	6.1	0.31	3.6	0.033 [†]	1.5
2. Primary mineral weathering	4.3	20	2.4	0	0.07 [†]	0 ^{††}
3. N ₂ fixation	-	-	-	-	-	3-6 ^{††}
SUM	5.8	26.1	2.7	3.6	0.10	5(est.)
LOSSES						
1. Dissolved stream transport	3.4	16.1	1.6	2.2	0.05	0.04
2. Sediment transport	0.002	0.05	0.01	0.005	0.0003	0.02
SUM	3.4	16.1	1.6	2.2	0.05 [†]	0.06
DIFFERENCE	2.4	10	1.1	1.4	0.05 [†]	5(est.)
VEGETATION						
1. Uptake and translocation (+)	12.6	21	3.3	2.2	2.7	22
2. Litterfall and translocation (-)	9.0	5.8	1.5	1.3	1.8	14
3. Aboveground mortality	1.0	3.5	0.3	0.2	0.2	2
4. Root mortality	0.3	1.8	0.3	0.1	0.1	0.7
ANNUAL INCREMENT	2.3	10.1	1.2	0.6	0.6	5.3
POOLS						
1. Standing crop of trees	189	589	84	39	38	346
2. Litter	34	347	34	14	19	248
3. Soil	184	3191	111	390	175	3160
SYSTEM TOTAL	407	4127	229	443	232	3754

[†] P release from weathering was not calculated by difference in a mass balance equation because it leads to an unacceptably high value. The value of 0.07 kg ha⁻¹ y⁻¹ is derived from a long-term denudation estimate of 235 kg ha⁻¹ y⁻¹ (data on file, Boise Forestry Sciences Laboratory and 0.03 percent P in rock).

^{††} This range of N₂ fixation is estimated from asymbiotic fixation data of Jurgensen et al. (1979) for the same habitat types, and the low range of symbiotic fixation associated with Ceanothus velutinus after Youngberg and Wollum (1976).

Northern Rocky Mountains (Jurgensen et al., 1979) or central Oregon ponderosa pine forests (Youngberg and Wollum, 1976).

The vegetation requirement for P exceeds the difference in inputs minus losses. We could balance this by increasing P weathering inputs, however this requires unacceptably large weathering release values for P. Instead, we have proposed an upper value of $0.07 \text{ kg ha}^{-1} \text{ y}^{-1}$ based on the P concentration in rock and a denudation estimate. P cycling in SC-5 appears to be in a tight economy, with inputs to the system only slightly exceeding losses. Vegetation requirements depend upon mobilization of the large litter and soil reserves, and the P supply should be considered particularly vulnerable to accelerated erosion in light of its low re-supply rates from weathering and precipitation.

Sulfur inputs from precipitation exceed system losses plus annual vegetation increment by $0.8 \text{ kg ha}^{-1} \text{ y}^{-1}$. The sink(s) for this excess input are unknown. Feller (1977) observed decreases in SO_4^{-2} concentration as water travelled down through mineral soil to groundwater, and suggested immobilization by anion adsorption.

Accelerated Nutrient Loss with Harvest

Solution Chemistry

Instantaneous concentrations and annual fluxes of dissolved species in streamwater were tested for treatment effects. Preliminary scanning of time trends in concentration and ratios of annual flux (treated: control) indicated that NO_3^- -N was the only dissolved species showing effects of treatment. In addition, it was apparent that the treatment effect was most pronounced during winter storms or spring snowmelt (Figure 4.2). This effect was tested for statistical difference and the magnitude of the effect was quantified using regression analysis. The concentration and flow

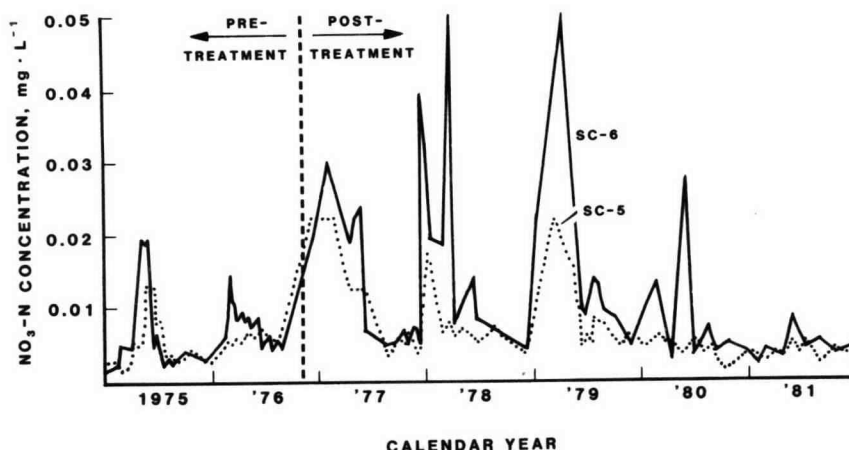


Figure 4.2. Plot of NO_3^- -N concentration, $\text{mg} \cdot \text{liter}^{-1}$ vs. time for watershed SC-5 (untreated) and SC-6 (clearcut and broadcast burned). Logging was done in September and October 1976.

data for each year were separated into periods of accelerated NO_3^- -N flux corresponding to spring snowmelt. Snowmelt period and peak NO_3^- -N concentrations coincide for both watersheds. Mean daily NO_3^- -N fluxes were generated for each sample date during the melt period and weighted by the number of days in the sample period. These values, (mean daily flux times number of days) for SC-6 were regressed on values for the same sample periods for SC-5. The slope of the linear regression of SC-6 NO_3^- -N flux on SC-5 NO_3^- -N flux prior to treatment is 3.8. Following treatment, the slope of the regression equation is 9.02 and is significantly different at $\alpha = 0.05$ (Figure 4.3).

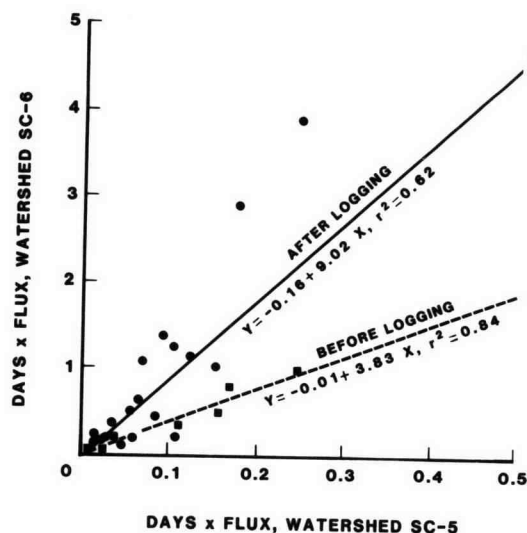


Figure 4.3. Regressions of springtime NO_3^- -N flux, SC-6, vs. SC-5, before and after logging. The slopes of the two regressions are significantly different at $\alpha = 0.05$.

These results suggest that clearcutting and burning increased NO_3^- -N loss from the cutting units during periods of spring snowmelt or high winter runoff. These accelerated losses averaged two and one-half times normal NO_3^- -N transport over the 4-year post-harvest period. During this period NO_3^- -N concentration never exceeded $0.05 \text{ mg liter}^{-1}$. If we ignore instream processes of N immobilization or loss through denitrification, we would estimate the accelerated NO_3^- -N transport averaged 10.4 times normal from the cutting units or $0.11 \text{ kg ha}^{-1} \text{ y}^{-1}$. Although this figure should be considered a minimum because of the lack of information on instream changes, it

is unlikely that accelerated solution loss of N exceeds $1 \text{ kg ha}^{-1} \text{ y}^{-1}$ from the cutting units.

Erosion and Sediment Loss

Logging and burning also accelerated erosion rates for a few years following treatment (Clayton, 1981). The accelerated erosion over the first winter averaged 1.8 Mg ha^{-1} , increased to 13 Mg ha^{-1} the first summer following harvest, and decreased to 4 Mg ha^{-1} over the next winter and summer periods. These increases in erosion have not been reflected in increased masses of sediment transport measured at the watershed mouth. This is in part due to considerable on-slope and in-channel storage. SC-6 normally stores 12 times its average annual sediment yield behind natural obstructions within the channel system (Megahan 1982). Sediment bedforms within the channel plus on-slope storage of sediment add to this figure. In water year 1980, however, the sediment yield on a volume basis in SC-6 was $8.8 \text{ m}^3 \text{ km}^{-2}$, which exceeds the 95 percent confidence interval about the regression of SC-6 sediment yield on SC-5 sediment yield. Because of a lower bulk density for sediment in SC-6 in 1980, the mass was not significantly greater than would be predicted without harvest at the 95 percent level. The lower density reflects a higher proportion of organic matter in the sediment. The associated increase in total nutrient transport is small but significant ($\alpha = 0.05$), and ranges from 0.01 kg ha^{-1} of K to 0.17 kg ha^{-1} of N lost from the cutting units. Accelerated sediment transport of Mg, Ca, S and P fall between the K and N transport rates.

Bole Transport

The accelerated losses of nutrients transported as solution load or sediment load are insignificant when compared to the large losses from bole removal. Table 4.4 presents the elemental losses and the percentages of the total ecosystem pool removed through harvesting. These data are based on an estimate of 11, 45, and 17

Mg ha⁻¹ of ponderosa pine heartwood, sapwood, and bark, respectively, and similarly 8, 45, and 17 Mg ha⁻¹ for Douglas-fir. These biomass removals were accompanied by large transfers of living

Table 4.4.--Elemental redistribution on the site from slash additions and elemental losses from bole removal during timber harvesting expressed as mass per unit area. Losses are also reported as a percentage of the total ecosystem pool (standing crop plus litter plus soil) as presented in Table 4.2.

Element	Slash added prior to burn	Bole loss	
		Mass	Percent of pool
	kg ha ⁻¹	kg ha ⁻¹	Percent
K	125	85	21
Ca	414	352	9
Mg	68	29	13
S	25	22	5
P	35	12	5
N	248†	136	4

†Much of the N addition with slash was subsequently lost by volatilization during burning.

biomass to slash on the site. We estimate that 8 Mg ha⁻¹ of foliage, 22 Mg ha⁻¹ of branches and tops, and nearly 50 Mg ha⁻¹ of roots were left on the site following harvest. This potentially results in a large mobilization of elements following burning and decay (Table 4.4). For example, we estimate that 125 kg ha⁻¹ of K, 414 kg ha⁻¹ of Ca, and 248 kg ha⁻¹ of N were contained in slash. Retention mechanisms on the forest floor and in the soil, developing vegetation and buffer strips were apparently effective since none of

these "mobilized" elements were detected at the watershed mouth, with the exception of small amounts of NO_3^- -N.

Much of the N returned to the site with litter may have been subsequently lost as a volatile product of burning, probably as N_2 gas (De Bell and Ralston, 1970). Feller and Kimmins (1984) estimated N losses from slash burning in British Columbia of nearly 1 Mg/ha which was more than three times the log export of N. Our data only indicate N content in slash to be approximately twice log export values, so our volatile loss as a percent of total loss would be less. Clayton (1981) estimated litter reductions on cutting units in SC-6 due to burning of 7 percent. This could result in an additional 17 kg ha⁻¹ of N loss. Feller and Kimmins (1984) also reported losses of all other nutrients from convective fly-ash removal due to the slash burn. The percentages of ecosystem total removed were smaller, and the ultimate fate of this material is unknown.

CONCLUSIONS

Clearcutting and helicopter yarding of logs in the Idaho batholith minimally accelerated nutrients lost by stream transport, either in solution or as sediment load. Although onsite estimates of erosion indicated a short-term increase of approximately ten times normal rates, slope storage and buffer strips kept most sediment out of the stream. Small increases in all elements studied accompanied accelerated organic sediment transport the third year following treatment. Nitrate -N showed the only statistically significant increase in solution transport over the 4-year post-harvest study. We estimate that NO_3^- -N flux averaged approximately ten times normal flux from the cutting units for 4 years; however, this only amounts to 0.11 kg ha⁻¹ y⁻¹ over the postharvest period.

Watershed SC-5 carried somewhat lower volumes of timber than SC-6 did prior to harvest due to understocking in lower diameter

classes. For example, mass of heartwood plus sapwood in SC-5 averages 103 Mg ha^{-1} based on the 3 percent cruise and allometric equations. In SC-6, the cutting units yielded an average timber volume of $33.8 \text{ M bd. ft. acre}^{-1}$ which equals 109 Mg ha^{-1} assuming an average piece size of 0.38 m (15 inches). For this reason, the intrasystem cycling rates for nutrients in SC-5 should not be directly applied to cutting units in SC-6. Nevertheless, the nutrient requirements of the trees in SC-5 provide a benchmark for evaluating the environmental consequences of accelerated nutrient loss with harvest. The largest nutrient export from the site occurred with bole harvest. Four percent of total ecosystem N was removed with harvest (not including volatile losses); 21 percent of ecosystem K (not including primary mineral or other unavailable mineral K) was removed.

It is not readily apparent what these accelerated nutrient losses mean to long-term productivity of the site. Kimmins (1977) suggested that one might evaluate the recovery process on the basis of ecological rotation, which he defined as "a tree rotation which permits the return of the site to the ecological condition that existed prior to rotation". We might extend this to include the nutrient capital of the site, and thus have a basis for evaluating the effects of accelerated nutrient loss. Nutrient demands of a forest differ with stand age and degree of development. Johnson (in press) points out that the rate of nutrient accumulation in coniferous forests is greater prior to crown closure than after, however, demand for some elements (notably K) is very high in the first 10 years of stand development and declines thereafter. Demand for other macronutrients remains high for possibly twice this length of time.

Clayton (1984) provided estimates of cation release by weathering in Silver Creek watershed SC-5, and retention efficiencies in soil and vegetation. Based on these rates of supply, nutrient status of the ecosystem could return to prior-to-harvest

condition in 24 years for Mg and 35 years for both K and Ca. This assumes, however, that there is no lag in vegetative reestablishment following clearcutting. Clayton (1984) suggests adding 10 years to those figures because data suggest it may take that long to achieve greater than 90 percent coverage by shrubs, forbs and grasses (Steele, R. and K. Geier-Hayes, 1983. The Douglas-fir/ninebark habitat type in Central Idaho - succession and management. Draft Report, USDA Forest Service, Intermountain Forest and Range Experiment Station and Intermountain Region, 76 p.).

Rates of N replacement are highly dependent on plant succession, and particularly, whether or not fire played a role in the disturbance. In SC-6 where broadcast burning was used to dispose of slash, re-invasion by Ceanothus velutinus Dougl. var. velutinus was common. Without burning, C. velutinus is less common and N_2 fixation rates are far lower. Assuming average figures for N_2 fixation by C. velutinus (Youngberg and Wollum, 1976), total site N might be replaced in approximately 20 years. Replacement of S by precipitation would take less than 10 years. These estimates of replacement time all assume that large clearcuts such as in SC-6 will be successfully reforested by planting. Delaying regeneration will cut down on success in both re-stocking the stand and returning nutrients to the site.

BIBLIOGRAPHY

1. Alban, D. H. 1982. Effects of nutrient accumulation by aspen, spruce, and pine on soil properties. *Soil Sci. Soc. Amer. J.* 46:853-861.
2. Alban, D. H., D. A. Perala, B. E. Schlaegel. 1978. Biomass and nutrient distribution in aspen, pine and spruce stands on the same soil type in Minnesota. *Can. J. For. Res.* 8:290-299.
3. Allison, L. E. 1965. Organic carbon. p. 1367-1378. In *Methods of soil analysis*, C. A. Black et al. (ed.) Amer. Soc. of Agron., Madison, Wis.
4. Aubertin, G. M. and J. H. Patric. 1974. Water quality after clearcutting a small watershed in West Virginia. *J. Environ. Qual.* 3:243-249.
5. Bethlahmy, N. 1962. First year effects of timber removal on soil moisture. *Intl. Assoc. Sci. Hydrol. Bull.* 7(2):34-38.
6. Bishop, D. M. and M. E. Stevens. 1964. Landslides on logged areas in southeast Alaska. USDA For. Serv. Res. Paper NOR-1, 18 p.
7. Boyle, J. R., J. J. Phillips and A. R. Ek. 1973. "Whole tree" harvesting: nutrient budget evaluation. *J. For.* 71:760-762.
8. Bremner, J. M. 1965. Total Nitrogen. p. 1149-1178. In *Methods of Soil Analysis*, part 2, C.A. Black (ed. in chief), Amer. Soc. Agron., Madison, Wis.

9. Brown, G. W., A. R. Gahler, and R. B. Marston. 1973. Nutrient losses after clear-cut logging and slash burning in the Oregon Coast Range. *Water Resources Res.* 9(5):1450-1453.
10. Brown, G. W. and J. T. Krygier. 1971. Clearcut logging and sediment production in the Oregon Coast Range. *Water Resources Res.* 7(5):1189-1199.
11. Brown, J. K. 1978. Weight and density of crowns of Rocky Mountain conifers. *USDA Forest Service Res. Paper INT-197*, Intermountain Forest and Range Expt. Sta., Ogden, Utah. 56 p.
12. Burroughs, E. R. and B. R. Thomas. 1977. Declining root strength in Douglas-fir after felling as a factor in slope stability. *USDA For. Serv. Res. Paper INT-190*, 27 p. Intermountain Forest and Range Expt. Sta., Ogden, Utah.
13. Carson, M. A., and M. J. Kirkby. 1972. "Hillslope Form and Process." Cambridge University Press.
14. Chapman, H. D. 1965. Cation exchange capacity. p. 891-901. In *Methods of soil analysis*, C. A. Black (ed.) Amer. Soc. of Agron., Madison, Wis.
15. Clayton, J. L. 1974. Clay mineralogy of soils in the Idaho batholith. *Geol. Soc. Amer. Bull.* 85: 229-232.
16. Clayton, J. L. 1979. Nutrient supply to soil by rock weathering. In *Proceedings: Impact of Intensive Harvesting on Forest Nutrient Cycling*. p. 75-96. State Univ. of New York, Syracuse, N.Y.

17. Clayton, J. L. 1981a. Soil disturbance caused by clearcutting and helicopter yarding in the Idaho batholith. USDA Forest Serv. Res. Note INT-305, 7 p. Intermountain Forest and Range Expt. Sta., Ogden, Utah.
18. Clayton, J. L. 1981b. Magnitude and implications of dissolved and sediment elemental transport from small watersheds. p. 83-98, In D. M. Baumgartner (Ed.) Proc. Interior West Watershed Management Symposium, Wash. St. Univ. Coop. Extens., Pullman, Washington.
19. Clayton, J. L. 1984. A rational estimate of elemental supply rate by weathering. p. 405-419. In Proceedings, 6th North American Forest Soils Conference, E. L. Stone (ed.).
20. Clayton, J. L. 1985. An estimate of plagioclase weathering rate in the Idaho batholith based upon geochemical transport rates. In Rates of Weathering of Rocks and Minerals, S. M. Colman (ed.), Academic Press, New York, (in press).
21. Clayton, J. L. and D. A. Kennedy. 1980. A comparison of the nutrient content of Rocky Mountain Douglas-fir and ponderosa pine trees. USDA For. Serv. Res. Note INT-281, 13 p. Intermountain Forest and Range Expt. Sta., Ogden, Utah.
22. Clayton, J. L. and D. A. Kennedy. 1985. The nutrient budget of a mixed conifer forest in the Idaho batholith - harvest implications. Soil Sci. Soc. Amer. J., (in press).
23. Clayton, J. L., W. F. Megahan, and D. Hampton. 1979. Soil and bedrock properties: weathering and alteration products and processes in the Idaho batholith. USDA Forest Service Research Paper INT-237. 35 p. Intermountain Forest and Range Expt. Sta., Ogden, Utah.

24. Cleaves, E. T., A. E. Godfrey, and O. P. Bricker. 1970. Geochemical balance of a small watershed and its geomorphic implications. *Geol. Soc. Amer. Bull.* 81: 3015-3032.
25. Cleaves, E. T., D. W. Fisher, and O. P. Bricker. 1974. Chemical weathering of serpentinite in the eastern Piedmont of Maryland. *Geol. Soc. Amer. Bull.* 85: 437-444.
26. DeBell, S. S. and C. W. Ralston. 1970. Release of nitrogen by burning light forest fuels. *Soil Sci. Soc. Amer., Proc.* 34(6):936-938.
27. DeByle, N. V. and P. E. Packer. 1972. Plant nutrient and soil losses in overland flow from burned forest clearcuts. p. 296-307, In S. C. Csallany, T. G. McLaughlin and W. D. Striffler (Eds.), *Proceedings of a Symposium on Watersheds in Transition*. Amer. Water Resources Assoc. and Colorado St. Univ., Fort Collins.
28. Duvigneaud, P., and S. Denaeyer-De Smet. 1964. Le cycle des éléments biogènes dans l'écosystème forêts (forêts tempérées caducifoliées). *Lejeunia* 28:1-148.
29. Dyrness, C. T. 1965. Soil surface conditions following tractor and high-lead logging in the Oregon Cascades - J. For. 63:272-275.
30. Dyrness, C. T. 1967. Soil surface conditions following skyline logging. USDA Forest Serv. Res. Note PNW-55, 8 pp. Pacific NW Forest and Range Expt. Sta., Portland, Oregon.
31. Dyrness, C. T. 1972. Soil surface conditions following balloon logging. USDA Forest Serv. Res. Note PNW-182, 7 p. Pacific NW Forest and Range Expt. Sta., Portland, Oregon.

32. Feller, M. C., and J. P. Kimmins. 1984. Effect of clearcutting and slash burning on streamwater chemistry and watershed nutrient budgets in southwestern British Columbia. *Water Resour. Res.* 29(1):29-40.
33. Feth, J. H., C. E. Roberson, and W. L. Polzer. 1964. Sources of mineral constituents in water from granitic rocks, Sierra Nevada, California, and Nevada. U.S. Geol. Survey Water-Supply Paper 1535-I, U.S. Government Printing Office, Washington, D. C., 70 p.
34. Fredriksen, R. L. 1970. Erosion and sedimentation following road construction and timber harvest on unstable soils in three small western Oregon watersheds. USDA Forest Serv. Res. Paper PNW-104, 15 p. Pacific NW Forest and Range Expt. Sta., Portland, Oregon.
35. Fredriksen, R. L. 1971. Comparative chemical quality-natural and disturbed streams following logging and slash burning. p. 126-137. In J. T. Krygier and J. D. Hall (ed.) Symposium on Forest Land Use and the Stream Environment, Oregon St. Univ., Corvallis.
36. Fredriksen, R. L. 1972. Nutrient budget of a Douglas-fir forest on an experimental watershed in Oregon. P. 115-131. In Proceedings, Research on coniferous forest ecosystems symposium, Bellingham, Washington.
37. Freedman, B., R. Morash, and A. J. Hanson. 1981. Biomass and nutrient removals by conventional and whole-tree clear-cutting of a red-spruce-balsam fir stand in central Nova Scotia. *Can. J. For. Res.* 11:249-257.

38. Gessel, S. P. and D. W. Cole. 1965. Influence of removal of forest cover on movement of water and associated elements through soil. J. Amer. Water Works Assoc. 57(10):1301-1310.
39. Gholz, H. L., C. C. Grier, A. G. Campbell, and A. T. Brown. 1979. Equations for estimating biomass and leaf area of plants in the pacific northwest. Res. Paper 41, Oregon St. Univ., Forestry Res. Lab., Corvallis, OR., 39 p.
40. Gray, D. H. and R. P. Brenner. 1970. The hydrology and stability of cutover slopes. p. 295-396, In Proc., Interdisciplinary aspects watershed management, Amer. Soc. Civil Engineer.
41. Gray, D. H. and W. F. Megahan. 1981. Forest vegetation removal and slope stability in the Idaho batholith. USDA Forest Serv. Res. Paper INT-271, 23 p. Intermountain Forest and Range Expt. Sta., Ogden, Utah.
42. Greenberg, A. E., J. J. Connors, and D. I. Jenkins. 1980. Standard methods for the examination of water and wastewater. (15th edit.). Amer. Publ. Health Assoc., Amer. Water Works Assoc., and Water Pollution Control Federation, 1134 p.
43. Gregory, K. J., and D. E. Walling. 1973. Drainage Basin Form and Process: a Geomorphological Approach. John Wiley and Sons, New York, 456 p.
44. Hornbeck, J. W., G. E. Likens, R. S. Pierce, and F. H. Bormann. 1973. Stripcutting as a means of protecting site and streamflow quality when clearcutting Northern hardwoods. p. 209-225. In Forest Soils and Forest Land Management, Proc. 4th North Amer. Forest Soils Conf., Laval Univ. Press, Quebec, Canada.

45. Jackson, M. L. 1964. Chemical composition of soils. In Chemistry of the Soil, Firman E. Bear (ed.). Reinhold Publ. Corp., New York. 515 p.
46. Johnson, C. M. and A. Ulrich. 1959. Analytical methods for use in plant analysis. Univ. of Calif. Agric. Expt. Sta. Bull. 766, 78 p.
47. Johnson, D. W. (in press). The effects of harvesting schedules on nutrient depletion in forests. Proceedings of the IUFRO Productivity Symposium, University of Washington, published as a General Technical Report, Pacific Northwest Forest and Range Expt. Sta.
48. Johnson, D. W., D. C. West, D. E. Todd, and L. K. Mann. 1982. Effects of sawlog vs. whole-tree harvesting on the nitrogen, phosphorus, potassium, and calcium budgets of an upland mixed oak forest. Soil Sci. Soc. Am. J. 46:1304-1309.
49. Jurgensen, M. F., S. F. Arno, A. E. Harvey, M. J. Larsen, and R. D. Pfister. 1979. Symbiotic and non-symbiotic nitrogen fixation in northern Rocky Mountain ecosystems. p. 294-308, In J. C. Gordon, C. T. Wheeler, and D. A. Perry (Eds.) Proc. Symbiotic Nitrogen Fixation in the Management of Temperate Forests Workshop, Oregon St. Univ., Corvallis, Oregon.
50. Jurgensen, M. F., A. E. Harvey, and M. J. Larsen. 1981. Effects of prescribed fire and soil nitrogen levels in a cutover Douglas-fir/Western larch forest. USDA For. Serv. Res. Paper INT-275, 6 p., Intermountain Forest and Range Expt. Sta., Ogden, Utah.

51. Kimmins, J. P. 1977. Evaluation of the consequences for future tree productivity of the loss of nutrients in whole-tree harvesting. *For. Ecol. Manage.* 1:169-183.
52. Larsen, E. S. and R. G. Schmidt. 1958. A reconnaissance of the Idaho batholith and comparison with the southern California batholith. *U.S. Geol. Survey Bull.* 1070-A, U.S. Government Printing Office, Washington, D.C., 32 p.
53. Leaf, A. L. 1979. Preface to the Proceedings: Impact of Intensive Harvesting on Forest Nutrient Cycling. State Univ. of New York, Syracuse, N.Y. p. ii.
54. Leopold, L. B., M. G. Wolman, and J. P. Miller. 1964. "Fluvial Processes in Geomorphology." W. H. Freeman and Co., San Francisco and London.
55. Likens, G. E., F. H. Bormann, R. S. Pierce, J. S. Eaton, and N. M. Johnson. (1977). *Biogeochemistry of a Forested Ecosystem.* Springer Verlag, New York, Heidelberg, Berlin, 146 p.
56. Malkonen, E. 1973. Effect of complete tree utilization on the nutrient reserves of forest soils, p. 375-386. In *IUFRO Biomass Studies*, Univ. of Maine, Orono, Me.
57. Marchand, D. E. 1971. Rates and modes of denudation in the White Mountains, Eastern California, *Am. J. of Sci.* 270:109-135
58. Marchand, D. E. 1974. Chemical weathering, soil development, and geochemical fractionation in a part of the White Mountains, Mono and Inyo Counties, California. *U. S. Geol. Survey Prof. Paper* 352-J, U. S. Government Printing Office, Washington, D. C., p. 379-424.

59. Marcovic, R. D. 1965. Probability functions of best fit to distributions of annual precipitation and runoff. Hydrology Papers, Colorado State University, Ft. Collins, Colorado, No. 8, 33 p.
60. Mattigod, S. V., and J. A. Kittrick. 1980. Temperature and water activity as variables in soil mineral activity diagrams. Soil Sci. Soc. Amer. J. 44: 149-154.
61. Megahan, W. F. 1972. Volume weight of reservoir sediment in forested areas. J. of the Hydraulics Division, Proc. of the Amer. Soc. of Civil Engineers, pp. 1335-1342.
62. Megahan, W. F. 1982. Channel sediment storage behind obstructions in forested watersheds draining the granitic bedrock of the Idaho batholith. In Proc. Workshop, Sediment Budgets and Routing in Forested Drainage Basins, 1979 May 30-June 1; Corvallis, Oregon. Tech. Paper PNW 141, USDA Forest Serv., Pacific Northwest Forest and Range Expt. Sta., p. 114-121.
63. Megahan, W. F., N. F. Day, and T. M. Bliss. 1978. Landslide occurrence in the western and central Northern Rocky Mountain physiographic province in Idaho. p. 116-139, In C. T. Youngberg (Ed.) Forest Soils and Land Use, Proc. 5th North Amer. For. Soils Conf., Colorado St. Univ., Ft. Collins, Colorado.
64. Megahan, W. F. and W. J. Kidd. 1972. Effect of logging roads on sediment production rates in the Idaho batholith. USDA Forest Serv. Res. Paper INT-123, 14 p. Intermountain Forest and Range Expt. Sta., Ogden, Utah.

65. Megahan, W. F. and D. C. Molitor. 1975. Erosion effects on wildfire and logging in Idaho. Watershed Management Symposium, Amer. Soc. of Civ. Eng., Irrigation and Drainage Div., Logan, UT, p. 423-444.
66. Megahan, W. F., K. A. Seyedbagheri, and P. C. Dodson. 1983. Long-term erosion on granitic roadcuts based on exposed tree roots. *Earth Surface Processes and Landforms* 8: 19-28.
67. Mersereau, R. C. and C. T. Dyrness. 1972. Accelerated mass wasting after logging and slash burning in western Oregon. *J. Soil and Water Conserv.* 27(3):112-114.
68. Meybeck, M. 1976. Dissolved and suspended matter carried by rivers: composition, time and space variations, and world balance. In *Interactions Between Sediments and Fresh Water* (H. L. Goltermen, ed.), pp 25-32. Amsterdam, The Netherlands.
69. Nelson, D. O. 1976. Soil Survey of the Middle Fork Payette River Area, Idaho. National Cooperative Soil Survey, U. S. Dept. of Agriculture, Washington, D. C., 55 p.
70. Olsen, J. S. 1963. Energy storage and the balance of producers and decomposers in ecological systems. *Ecol.* 44:322-331.
71. Ovington, J. D. 1962. Quantitative ecology and the woodland ecosystem concept. *Adv. Ecol. Res.* 1:103-192.
72. Ovington, J. D. 1965. Nutrient cycling in woodlands. In *Experimental Pedology*, Hallsworth, E. G., and D. V. Crawford (eds.) p. 208-215. Butterworths, London.

73. Peech, M. 1965. Exchange acidity. p. 905-913. In Methods of soil analysis, C. A. Black, (ed.) Amer. Soc. of Agron., Madison, Wis.
74. Phillips, W. R., and D. T. Griffen. 1981. Optical Mineralogy; the Nonopaque Minerals. W. H. Freeman and Co., San Francisco, 469 p.
75. Pierce, R. C., C. W. Martin, C. C. Reeves, G. E. Likens, and F. H. Bormann. 1972. Nutrient loss from clearcuttings in New Hampshire. In Proceedings: National Symposium on Watersheds in Transition, p. 285-295, Colorado St. Univ., Fort Collins.
76. Reinhart, K. G. 1967. Watershed calibration methods. p. 715-723. In International Symposium on Forest Hydrology, Pergamon Press, New York.
77. Reynolds, R. C., and N. M. Johnson. 1972. Chemical weathering in the temperate glacial environment of the northern Cascade Mountains. *Geochimica et Cosmochimica Acta* 36: 537-554.
78. Robie, R. A., B. S. Hemingway, and J. R. Fisher. 1979. Thermodynamic properties of minerals and related substances at 298.15 K and 1 bar (10^5 Pascals) pressure, and at higher temperatures. U.S. Geol. Survey Bull. 1452, U. S. Government Printing Office, Washington, D. C. 456 p.
79. Ross, C. P. 1963. Modal composition of the Idaho batholith. U. S. Geol. Surv. Prof. Paper 475-C, p. C86-C90.
80. Saunders, I. and A. Young. 1983. Rates of surface processes on slopes, slope retreat and denudation. *Earth Surface Processes and Landforms* 8:473-501.

81. Schmidt, D. L. 1964. Reconnaissance petrographic cross section of the Idaho batholith in Adams and Valley Counties, Idaho. U.S. Geol. Survey Bull. 1181-G, U. S. Government Printing Office, Washington, D. C. 50 p.
82. Sollins, P., C. C. Grier, F. M. McCorison, K. Cromack, Jr., R. Fogel, and R. L. Fredriksen. 1980. The internal element cycles of an old-growth Douglas-fir ecosystem in western Oregon. Ecol. Monog. 50(3):261-285.
83. Sposito, G. 1981. The Thermodynamics of Soil Solutions. Oxford Clarendon Press. 223 p.
84. Steele, R., R. D. Pfister, R. A. Ryker, and J. A. Kittams. 1981. Forest habitat types of Central Idaho. USDA For. Serv. Gen. Tech. Rprt. INT-114, 138 p. Intermountain Forest and Range Expt. Sta., Ogden, Utah.
85. Strahler, A. N., and A. H. Strahler. 1973. Environmental geoscience: interaction between natural systems and man. John Wiley and Sons, Inc., New York. 511 p.
86. Swank, W. T. and W. H. Caskey. 1982. Nitrate depletion in a second-order mountain stream. J. Environ. Qual. 11(4):581-584.
87. Swank, W. T. and J. B. Waide. 1980. Interpretation of elemental cycling research in a management context: evaluating potential effects of alternative management strategies on site productivity. p. 137-158. In R. H. Waring and J. Franklin (Eds.), Forest Ecosystems: Fresh Perspectives from Ecosystem Analysis. Oregon St. Univ. Press, Corvallis, Oregon.

88. Swanston, D. N. and W. J. Walkotten. 1970. The effectiveness of rooting as a factor in shear strength of karta soils. USDA Prog. Reprt. Study No. FS-PNW 1604:26, Portland, Oregon.
89. Tamm, C. O. 1964. Determination of nutrient requirements of forest stands. *Int. Rev. Forestry Res.* 1:115-170.
90. Tamm, C. O. 1979. Nutrient cycling and productivity of forest ecosystems. p. 2-21. *In* Proc. Impact of Intensive Harvesting on Forest Nutrient Cycling. State Univ. of New York, Syracuse, New York.
91. Tiedemann, A. R., and T. D. Anderson. 1971. Rapid analysis of total sulfur in soils and plant material. *Plant and Soil* 35:197-200.
92. Van der Marel, H. W. 1958. Quantitative analysis of kaolinite. *J. Int. Etud. Argiles* 1:1-19.
93. Verstraten, J. M. 1977. Chemical erosion in a forested watershed in the Oesling, Luxembourg. *Earth Surface Processes* 2: 175-184.
94. Vitousek, P. M., J. R. Gosz, C. C. Grier, J. M. Melillo, W. A. Reiners, and R. C. Todd. 1979. Nitrate losses from disturbed ecosystems. *Science* 204:469-474.
95. Vitousek, P. M. and J. M. Melillo. 1979. Nitrate losses from disturbed forests: patterns and mechanisms. *For. Science* 25(4):605-619.
96. Vitousek, P. M., and W. A. Reiners. 1975. Ecosystem succession and nutrient retention: a hypothesis. *Bioscience* 25:376-381.

97. Waldron, L. J., S. Dakessian, and J. A. Nemson. 1983. Shear resistance enhancement of 1.22 - meter diameter soil cross sections by pine and alfalfa roots. Soil Sci. Soc. Am. J. 47: 9-14.
98. Weaver, T., and F. Forcella. 1977. Biomass of fifty conifer forests and nutrient exports associated with their harvest. Great Basin Naturalist 37:395-401.
99. Weaver, C. E., and L. D. Pollard. 1973. The chemistry of clay minerals. Elsevier Scientific Publishing Co., Amsterdam, London, New York. 213 p.
100. Weetman, G. F. and B. Webber. 1972. The influence of wood harvesting on the nutrient status of two spruce stands. Can. J. For. Res. 2:351-369.
101. Wells, C. G., and J. R. Jorgensen. 1979. Effect of intensive harvesting on nutrient supply and sustained productivity. In Proceedings: Impact of Intensive Harvesting on Forest Nutrient Cycling. p. 212-230. State Univ. of New York, Syracuse, N.Y.
102. Wood, E. D., F. A. J. Armstrong and F. A. Richards. 1967. Determination of nitrate in sea water by cadmium - copper reduction to nitrite. J. Marine Biol. Assoc. U.K. 47:23-31.
103. Youngberg, C. T. and A. G. Wollum II. 1976. Nitrogen accretion in developing Ceanothus velutinus stands. Soil Sci. Soc. Amer. J. 40:109-112.

104. Zeman, L. J. and O. Slaymaker. 1978. Mass balance model for calculation of ionic input loads in atmospheric fallout and discharge from a mountainous basin. Hydrological Sciences Bull. 23(1):103-117.