RECOVERY AFTER LOGGING IN STREAMS WITH AND WITHOUT BUFFER STRIPS IN NORTHERN CALIFORNIA

by

DON C. ERMAN and DONALD MAHONEY
Department of Forestry and Resource Management
University of California, Berkeley

CALIFORNIA WATER RESOURCES CENTER
University of California Davis, California 95616

The research leading to this report was supported by the University of California, Water Resources Center, as part of Water Resources Center Project UCAL-WRC-W-586 and the McIntire-Stennis Program of the Agricultural Experiment Station of the University of California.

June 1983

Contribution No. 186
ISSN 0575-4941

ABSTRACT

Streams with narrow buffers (less than 30 m) and without buffers in northern California were sampled 6-10 years after logging and 5 or 6 years after an initial post-logging study in order to evaluate recovery rates. Six narrow-buffered streams and six unbuffered streams were grouped in blocks with suitable unlogged (control) streams.

Unbuffered streams showed considerable but incomplete recovery based on a diversity index of macroinvertebrates. Compared to the mean of control streams the mean diversity of logged streams was 9.1% lower in 1980-81 compared to 25.2% lower in 1975. Narrow-buffered streams, by contrast, have changed little since the last survey. The mean diversity was 12.5% lower than controls in 1980 compared to 12.4% in 1975, and the six streams showed a positive association between buffer width and diversity index.

By employing a measure of transportable sediment stored in the stream bed we determined that the logged and narrow-buffered streams still contain significantly more fine sediment than comparable control streams.

Narrow buffers were not effective in promoting a more complete or rapid rate of recovery than streams without buffers.

Acknowledgments

Nancy Erman was in charge of quality control in processing, taxonomic consistency, and laboratory procedures and she keyed the Trichoptera. Marilyn Meyers keyed the Ephemeroptera and Plecoptera. Jim Anderson, Brian Plant, Deborah Raines, and Craig Smith did the tedious and vital invertebrate processing. Field assistance was provided by Brian Plant, Rob Leidy and Terry Seefield. U.S. Forest Service personnel kindly provided information on access and recent activities in National Forest watersheds.

TABLE OF CONTENTS

Page

INTRODUCTION 1

STUDY AREAS 1

METHODS 5

General Procedures 5

Invertebrate Samples 5

Sample Collection and Processing 5

Data Analysis 6

Periphyton Pigments 6

Light 6

Transportable Sediment and Detritus 7

RESULTS 10

Logged Streams Without Buffers 10

Invertebrate Populations 10

Transportable Sediment and Detritus 10

Periphyton Pigments and Incident Radiation 10

Narrow-Buffered Streams 16

Invertebrate Populations 16

Transportable Sediment and Detritus 16

Periphyton Pigments and Incident Radiation 16

Stream Recovery 23

Streams Without Buffers 23

Streams With Narrow Buffers 23

Environmental and Invertebrate Community Correlations 26

DISCUSSION 30

    Stream Recovery 30

    Effects of Logging on Invertebrate Communities 33

    Effects of Logging on the Stream Environment 33

        Transportable Sediment and Detritus 33

Periphyton Pigments 34

Page

CONCLUSIONS 35

LITERATURE CITED 36

APPENDICES

A. Testing Macroinvertebrate Sampling 40

B. Testing the Method of Transportable Sediment Estimation 44

C. Environmental Characteristics of Logged (L), Narrow-Buffered (N), Recently Disturbed (RD), and Control (C) Streams in Northern California 47

D. Transportable Sediment and Detritus 49

E. Periphyton Pigments for 29 Streams in Northern California 50

INTRODUCTION

One of the major environmental concerns of logging is its impact on aquatic resources. Reduction in nearstream vegetation and disturbance of the land surface frequently lead to elevated sediment loads, increased water temperature, disruption of aquatic food webs, and decreased habitat diversity (Karr and Schlosser 1978). Changes in biomass and species composition of stream biota from improper logging have been shown repeatedly (Cordone 1956, Chapman 1962, Calhoun and Seeley 1963, Hall and Lantz 1969, Moring and Lantz 1975, Newbold et al. 1980). In some cases logging has resulted in greater density and biomass of young salmonids (Burns 1972, Moring and Lantz 1974, Hall et al. 1978). Factors which might ameliorate undesirable effects of logging are unclear but an increase in primary production within the stream is one mechanism that has been postulated to explain increased standing crops of benthos and fish (Murphy et al. 1981).

Bufferstrips of vegetation along streams are effective in reducing changes in stream communities caused by timber harvesting and are a recommended management practice to ameliorate adverse effects of logging (FWPCA 1970, Hall and Lantz 1969, California Fish and Game, no date, Lantz 1970, 1971, Moring and Lantz 1974, Sadler 1970, Karr and Schlosser 1978). Not all bufferstrips are totally effective, however. Extensive post-logging analysis of over 50 small northern California streams consistently found lower invertebrate diversity indices in streams with no buffers and in some with less than 30 m buffers (Erman et al. 1977). In streams with buffers greater than 30 m, diversities were not different from unlogged streams. This study also found evidence of lower diversity in streams logged more than a decade earlier.

A clearer understanding of how logging activities affect stream communities, especially in relationship to streamside bufferstrips, would be invaluable to resource managers. Our study was undertaken to increase our understanding of those processes through an analysis of invertebrate communities and environmental factors affecting those communities. In particular we wanted to determine the rate of recovery of invertebrate communities in streams that had lowered diversities in the years immediately following logging. We returned to approximately half of an original group of stations sampled in 1975 -- essentially all the logged and narrow-buffered streams studied by Erman et al. 1977). Although streams with narrow buffers generally were inadequately protected in the short term, we hypothesized that streams with narrow buffers might exhibit more rapid or complete recovery than logged streams without buffers. Data on macroinvertebrate composition, algal pigment standing crop, fine sediment abundance, and other physical-conditions of the streams were collected to test this hypothesis.

Several study designs can be used to evaluate environmental impacts on natural resources (long-term before and after analysis, extensive before and after analysis, intensive post-treatment analysis, extensive post-treatment analysis). The extensive post-treatment analysis was used in this study since it provides a wide perspective and a more general interpretation of results (Erman et al. 1977, Hall et al. 1978). In addition, a large number of environmental parameters have been sampled at each station so that relationships between biological and physical factors can be more intensively examined both in the treated streams and in the control streams taken as separate groups.

STUDY AREAS

Six logged sites, 6 narrow-buffered sites, and 17 control sites were sampled from 15 streams grouped into nine blocks in northern California (Fig. 1). All but three streams were located in watersheds dominated by Douglas-fir *(Pseudotsuga menziesii)*, true fir *(Abies* spp.), or mixed conifer (Pinus, *Abies,* and others) and hardwoods *(Quercus* spp. and Acer *macrophyllum).* The remaining streams were located in watersheds dominated by redwood *(Sequoia sempervirens)* and grand fir *(Abies grandis).* Sample sites were relocated as closely as possible to the original stations (Erman et al. 1977). Three new streams were chosen to replace control streams that could not be used because of human disturbance since the previous study. In the first study (Erman et al. 1977), sites were chosen so that each station within a logging area was matched with two control stations to constitute a block. That design was repeated with the exception of 1) the Copper block, which had only one treatment and one control (North Fork Copper was not flowing in its former bed), 2) the Two Bit block, and 3) the Redwood block. Original controls were recently disturbed in the two latter blocks. A description of watershed characteristics for streams lacking buffers (logged streams) and their respective controls is in Table 1, and those of streams with narrow buffers (buffered streams) and their

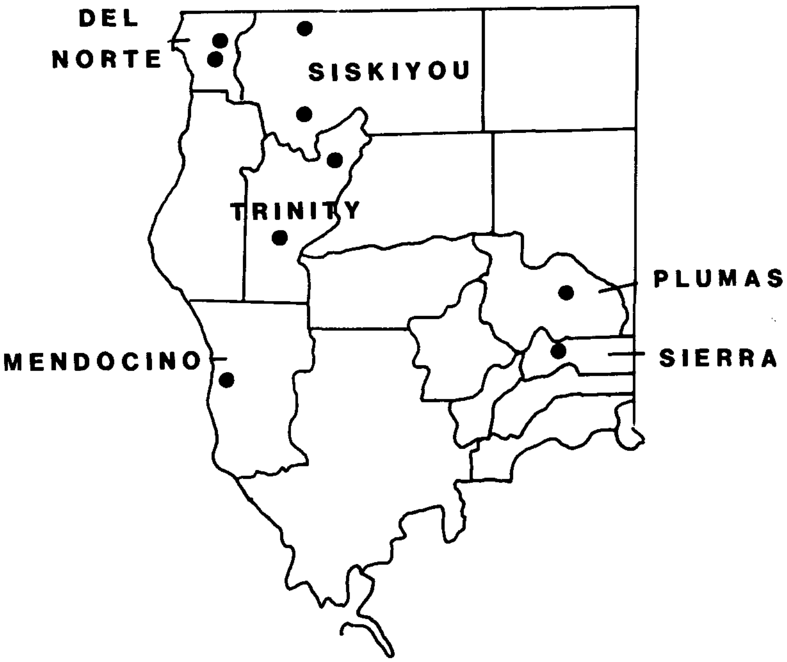

FIGURE 1. General location of study sites in northern California.

respective controls is in Table 2.

Control streams are generally similar to disturbed streams within a block in size, topography, and vegetation type. Little North Fork (unblocked in the original study) is a larger stream which for comparison has been included in the adjacent Garden Block. All other streams are typically near headwaters to reduce upstream influences as much as possible. Logging occurred between 6 and 10 years prior to the study (Tables 1, 2) using clear-cut cable, clear-cut tractor, and selective tractor logging methods. Buffer strips varied in average width from 3 to 25 meters (slope distance). The width of a stream buffer often varied considerably and some buffers graded into a cut rather than ended abruptly.

In the original study, logging roads were far from the streams except for crossings. In the 5 years since that study, construction or upgrading of roads near streams has occurred to some extent in 3 (Redwood, Philpot and Two Bit) of the 9 block areas, minimal new logging occurred in Philpot, Copper, Two Bit, and Taylor blocks, and substantial new logging occurred in the Redwood block several kilometers above the sample site on Hare Creek. Three streams that were included in these blocks in 1975 were obviously affected by the recent disturbances (South Fork Noyo had considerable new slash in the stream and a hydrogen sulfide smell, Naufus Creek had fresh sediment deposits from newly constructed roads, and Indian Creek had new skid trail crossings and skid trails along its banks). These streams were sampled in 1980 as comparisons with 1975 and to present a complete picture of as many of the original streams as possible. These streams are called recently disturbed (RD) and are treated separately in the presentation of results.

Most watersheds had extensive regrowth of brush or mixed conifer and hardwood species, although two of the clearcut watersheds (Two Bit and Four Bit) still had large areas of bare ground. Riparian shrubs *(Salix, Alnus,* and others) formed thick patches in places over the streams in the logged areas. In the bufferstrip blocks, no windfall of bufferstrip trees had fallen into the stream and nearstream vegetation appeared unchanged except along one stream (Garden) which had a 3 m bufferstrip. A road crossing above the

TABLE 1.
WATERSHED CHARACTERISTICS OF CONTROL (C, UNLOGGED), LOGGED (L), AND RECENTLY DISTURBED (RD) STREAMS IN NORTHERN CALIFORNIA

Block: Stream	Status		Logging Conditions[1]				Location
	Former	Present	Year	Type[2]	Area (ha)	% of Watershed	
Copper:							
Copper A	L	L	1973	ST	59	100	Shasta-Trinity National Forest
Copper B	C	C	-	-	-	-	
Two Bit:							
Two Bit	L	L	1972	CC	14	11	Klamath National Forest
Lower Four Bit	L	L	1972	CC	7	9	
Upper Four Bit	C	C	-	-	-	-	
Indian	C	RD	-	-	-	-	
Taylor:							
Lower Taylor	L	L	1972	ST	60	10	Plumas National Forest
Upper Taylor	C	C	-	-	-	-	
E. Br. Lights	C	C	-	-	-	-	
New York:							
Mid New York	L	L	1974	ST	33	7	Tahoe National Forest[3]
New York Trib	L	L	1974	ST	11	11	
Upper New York	C	C	-	-	-	-	
Empire	-	C					

[1] Data from Erman et al. (1977)

[2] ST - Selective Tractor
CC - Cable Clearcut

[3] Cut watersheds are on private land within Tahoe National Forest.

TABLE 2
WATERSHED CHARACTERISTICS OF CONTROL (C, UNLOGGED), NARROW-BUFFERED (N), AND RECENTLY DISTURBED (RD) STREAMS IN NORTHERN CALIFORNIA

Block: Stream	Status		Logging Conditlons[1]					Location
	Former	Present	Year	Type[2]	Area (ha)	% Watershed	Buffer Width(m)	
Redwood:								
N. Fk. S. Fk. Noyo	C	C	-	-	-	-	-	Jackson State Forest
S. Fk. Noyo	C	RD	-	-	-	-	-	
Hare	N	N	1974	CC	400	5	7	
Philpot:								
Philpot	C	C	-	-	-	-	-	Shasta-Trinity National Forest
Philpot Trib.	C	C	-	-	-	-	-	
Naufus	N	N, RD	1973	CT	11	6	5	
Packsaddle:								
N. Packsaddle	C	C	-	-	-	-	-	Six Rivers National Forest
U. Packsaddle	C	C	-	-	-	-	-	
Lower Packsaddle	N	N	1974	CC	24	4	20	
Fall:								
N. Fall	-	C	-	-	-	-	-	Six Rivers National Forest
S. Fall	C	C	-	-	-	-	-	
Knopki	N	N	1974	CC	12	15	25	
Garden:								
Whites	-	C	-	-	-	-	-	Klamath National Forest
Murphy	C	C	-	-	-	-	-	
Garden	N	N	1970	CT	60	16	3	
LNF	N	N	1970	CC	15	0.3	22	

[1] Datafrom Erman et al. (1977)

[2] CC -Cable Clearcut
CT - Tractor Clearcut

sampling station on Garden Creek had failed and a large debris pile completely covered the stream for several hundred meters as noted in the original study. Sampling was done below this disturbance.

METHODS

General Procedures

Twenty-five stations were sampled from June 1980 to August 1980 and 4 stations were sampled in June 1981. Each station was sampled once. Water samples were analyzed immediately for alkalinity (Woods 1975), pH, and Ca-Mg hardness (portable Hach tests). Also, measurements were taken of the following characteristics: temperature, discharge using a velocity-area technique and a current meter (John 1978), vegetation type, bankfull discharge width (mean of five measurements taken at 10 m intervals), and a visual estimate of overhead percent cover. Elevation, latitude, longitude, drainage area, and relief ratio are from Erman et al. 1977 or from appropriate maps (Appendix C). The magnitude of the average annual flood relative to the size of the drainage, area was estimated from the relationship:

$$\text{Bankfull width} = (\text{constant})(\text{drainage area})$$

where a value of 0.38 was used for the exponent b (after Dunne and Leopold 1978). The value of the constant for each stream in units of feet per square mile is used to estimate the intensity of the average annual spring runoff peak.

A modified rating of stream channel stability (Pfankuch 1975) was determined for each station. Numerical scores are given for 15 characteristics of the streambed and bank. The percentage composition of bottom materials was visually estimated by using 8 size classes. Six of the size classes (everything under 3 ft diameter) were used to graphically estimate the geometric mean diameter (dp) and its standard deviation (og) as presented by Ir.nman (1952). Since these values are Based on visual estimates they are approximations.

Invertebrate Samples

Sample Collection and Processing

At each station 16 benthic samples were taken with a 0.1 m2 modified Surber sampler (Mundie 1971). This sampler has a coarse inner mesh net (2.5 mm) and a fine outer net (0.366 mm). Large rocks enclosed by the sampler were hand cleaned of all organisms and the stream bottom within the area of the sampler was disturbed to a depth of about 10 cm. The sampler is described in detail in Erman et al. (1977). Organisms collected in the large mesh net were picked out by hand and saved along with contents of the fine mesh net. In order to reduce laboratory processing and identification we used a subsampling procedure. The contents of four Surber samples were grouped together as they were collected (a cluster). In streams with large variation in water current each group came from sites of similar current. The cluster was thoroughly homogenized in a container by hand with care taken not to damage any organisms. A tight fitting grid of four compartments divided the container into four equal parts. One part was removed by mouth suction into a small bottle and saved for analysis in 70% alcohol, and the other three parts (containing live organisms) were returned downstream of the sample section.

Each subsample obtained in this way, while equal in size to a single Surber sample, actually estimated the mean of four samples. At each station for each stream, four clusters of four samples (i.e., 16 Surber samples) were collected. The precision of a single subsample estimating the cluster mean was examined beforehand by counting a number of subsamples (Appendix A). Some subsample counts did not follow a Poisson distribution (e.g., some animals became enmeshed together), thus values were log transformed to normalize the distribution before statistical testing.

At three stations (South Fork Noyo, N. Packsaddle, Murphy), the total number of individuals in a cluster was too small to subsample. In these cases the probable number of rare species that would have beenmissedby subsampling (i.e., species with fewer than four individuals) were excluded from computations of diversity in order to make the values comparable to stations where subsampling was employed. At two stations (New York Tributary, Upper Taylor), only eight Surber samples could be taken because of the small size of the stream and limited habitat at those stations. The value obtained after pooling eight samples (two clusters) at these two stations was used because asymptotic values were reached after two clusters at all stations (Appendix A).

In the laboratory, samples were stained with rose bengal solution and were handpicked from detritus under a microscope (7x-30x). Most of the Ephemeroptera, Plecoptera, and Trichoptera were identified to genus. The Coleoptera and Diptera were usually identified only to family. Keys of Usinger (1956), Edmondson (1959), and Wiggins (1977) were used. A composite taxa list for each location was based on the four subsamples.

Data Analysis

The number of individuals, number of taxa, evenness component of diversity and the Shannon diversity index were used as dependent variables in statistical analysis. The diversity index was computed from

$$H' = - \Sigma p_i \ln p_i$$

and the evenness component from

$$e = \frac{H'}{\log g_e S}$$

where $p_i = x_i / \Sigma x_i$ is the sample proportion of the i^{th} taxon, and S is the number of taxa.

The diversity index from four pooled subsamples was used in regression analysis. The mean of the four subsamples was used in analysis of variance. Analysis of variance procedures followed those for nested or hierarchical sampling designs (Bennett and Franklin 1954, LeRoux and Reimer 1959). With this technique, some information about the within-cluster variation is lost because not all 16 Surber samples at a station were retained. However, within-cluster variation is sometimes considered insignificant compared to the variation within and between streams. The null hypothesis was that streams within a block do not have differences in diversity, evenness, density or number of taxa, thus the ratio of within to between stream variation fluctuates about the value of 1. A studentized range test (Q test) was used to test which differences between streams within blocks were significant (Chiang and Selvin 1978).

Periphyton Pigments

Algae were sampled for pigment content at sixteen sample points in riffles with 6-cm diameter or smaller rocks. At each point all surface rocks were collected from an area of 220 cm^. Samples were combined in groups of four and were systematically subsampled as for invertebrates. Four subsamples from each station were placed in a 1:1 solution of 90% acetone and dimethyl sulfoxide for 24 hours for extraction of pigments. This solvent mixture was used because the spectrophotometric extinction coefficients for acetone apply to this mixture (Shoaf and Lium 1976), and because preliminary laboratory tests showed that the superior solvent properties of dimethyl sulfoxide gave better extraction results from algae securely attached to rocks than did acetone by itself. Absorbance of the extracts was measured on a Beckman spectrophotoir.eter (Model SD, 4 nm band width) for wavelengths from 400 to 700 nm. Amounts or chlorophyll a and of phaeophytin a were estimated from equations in Vollenweider (1974).

Total pigment absorption potential was estimated by measuring with a planimeter the area under the curve of optical density vs. wavelength. That part of the spectrograph due to chlorophyll a and phaeophytin a was subtracted from the total area to give an index of the absorption potential of other pigments.

Light

Light was measured with a Li-Cor quantum radiometer. Ten instantaneous readings were taken directly over the center of the streambed along the stream at 5 m intervals. Readings were taken as closely as possible to 1230 hours (Pacific Daylight time), the brightest time of day. All streams but three in the redwoods where fog was present were sampled on clear days. Graphs were developed, using the radiometer, of the distribution of energy flux over time for a full clear day and were used to determine total amounts of radiation. Readings not taken at 1230 hrs were adjusted accordingly. Readings for the redwood stream where fog was present were adjusted by using graphs developed for those foggy conditions. Values are presented as approximate total daily radiation.

Transportable Sediment and Detritus

An index of stored fine sediments, called transportable sediment, was determined by disturbing a known area of stream bottom and catching the resultant sediment drift in downstream cylindrical traps. In this study fine sediments were those that passed through a 0.3 mm sieve. These size classes, categorized as sand and silts (Todd 1970), have been shown to have biological effects (Cordone and Kelley 1961). Because these size classes are flushed through stream systems rapidly (Beschta 1979), they may reflect the present condition of the watershed and stream channel more adequately than would larger sizes which may take a number of years to move through the system, if they move out of the watershed at all (Trimble 1981).

Each trap (82 mm in diameter, 42 mm depth) was partially filled with glass marbles (12 mm diameter) to aid in retaining deposited sediment. A trap was closed with a tight lid after filling with water and buried to just above the lip of the trap. Several rows of traps were placed at measured distances (up to 4 m) below a sample point. A stream riffle was chosen with as uniform a current and depth as possible.

When all measurements of depth and current at a trap were taken, the lids of the sediment traps were carefully removed, working from upstream first, taking care that no sediment entered them at this time. Then a sampling site immediately upstream was disturbed to a given depth (10 cm) with a hand trowel for a set time. In streams with moderate to heavy amounts of fine sediments, it was impossible to deplete the store of fines by disturbing the substrate. In those cases disturbance was usually limited to two minutes. The same time limit was used for particular stream blocks so that comparisons among streams could be made.

The area and depth of bottom disturbed was noted, as was the average width of the sediment plume over the traps. As soon as the stream cleared, the trap lids were replaced, beginning downstream, and removed from the stream. In the laboratory traps were emptied into a vacuum flask, the marbles and trap rinsed, and the contents filtered (0.45 μm-mesh) and oven-dried.

In general, unless the sampling stretch was irregular, the amount of sediment deposited decreased downstream in an exponential fashion as predicted by Einstein (1968), thus giving the sediment drift a longitudinal profile resembling a hyperbola. Occasionally differences in current and depth between adjacent microsites led to greater deposition at a site upstream than one downstream. In those cases data were smoothed to fit a probable curve.

We extrapolated from the trap the rate of sediment drift above and below the traps to calculate the total amount of sediment moved. We assumed that sediment was not resuspended from the containers. The area under a curve of sediment deposition per cm^2 vs. distance was measured by planimeter and equalled total deposition for a one cm strip downstream. This value multiplied by the mean width of the plume and divided by the volume disturbed, was the sediment (mg/cm^3) originally present at the site.

Sediment sizes varied with the distance from the origin where traps were placed and also with the water speed, because fast currents carried larger particles into the first trap. To standardize the effect of current differences among sample sites, we sieved out the larger size classes after burning off organic matter and based all estimations only on those size classes deposited at least in the first two rows of traps. Sediments deposited in those size classes theoretically should be independent of water speed (Einstein 1968). Current affects the distance over which a given size class is deposited but will not affect the amount deposited because the method assumes that all the disturbed sediment is deposited regardless of current speed. We tested this method by using known amounts and sizes of sediment and found an average overestimate of 18% for size classes typical of field situations (see Appendix B for details of test).

Small sized detritus was also collected by this technique. After weighing the dried filter with inorganic sediment and detritus, the sample was burned at 550 C to estimate the percentage of organic matter. Laboratory tests showed 8.5% of organic detritus remained as ash.

Sampling with a modified Surber sampler in Hare Creek

TABLE 3

MEAN VALUES OF POOLED SUBSAMPLE CLUSTERS FOR INVERTEBRATE COMMUNITY MEASURES AND DENSITY OF SELECTED TAXA FROM LOGGED (L) AND CONTROL (C) STREAMS. STREAMS ARE GROUPED IN BLOCKS. INDIAN CREEK, A FORMER CONTROL, WAS RECENTLY DISTURBED (RD).

Stream	Density (No./m^2)	Total Taxa	Shannon Diversity	Evenness	*Baetis* (No./m^2)	Chironomidae (No./m^2)
Copper A (L)	3480	52	2.75	.696	698	573
Copper B (C)	2230	38	2.28	.627	278	873
Two Bit (L)	5280	53	2.78	.699	805	653
L. Four Bit (L)	5710	49	2.44	.627	460	1863
U. Four Bit (C)	4680	53	2.91	.733	548	555
Indian (RD)	5340	50	2.20	.562	145	413
L. Taylor (L)	3920	39	2.27	.620	485	1665
U. Taylor (C)	1400	36	2.33	.650	208	550
E. Br. Lights (C)	5380	52	3.00	.758	265	928
Mid New York (L)	7530	62	2.57	.624	623	1825
New York Trib (L)	16600	43	1.59	.422	325	2465
U. New York (C)	5040	52	2.74	.694	295	1018
Empire (C)	9800	63	2.59	.624	1620	3533

RESULTS

Logged Streams Without Buffers

Invertebrate Populations

Within most blocks control streams had greater diversity and higher evenness than the logged streams when calculations were based on pooled cluster subsamples (Table 3). The differences between control and logged stream diversity and evenness were significant (P = 0.05, Wilcoxon signed rank test). There was no significant difference between logged and control stations for density or number of taxa. All blocks had significant differences in diversity among stations (tested on cluster means by analysis of variance), whereas other aspects of the invertebrate community were significantly different in only two or three of the blocks (Table 4).

Individual differences between stations within blocks were tested by a studentized multiple range test (Q test). Copper A now significantly exceeded its control in diversity and number of taxa, whereas New York Trib had lower diversity and evenness and higher density than controls (Table 5). Recently disturbed Indian Creek also had lower diversity and evenness than its control. Lower Four Bit had lower diversity than its control while Upper Taylor had lower diversity and number of taxa than one control (East Branch Lights) but not the other (Upper Taylor). Mid New York was not different from its two controls by any community measure.

Baetis and Chironomidae, two of the more abundant taxa in the 1975 study, were still the two most abundant taxa in general for all stations (Table 3). Chironomidae made up 24% of the density in logged stations and 26% in controls.

Transportable Sediment and Detritus

The quantity of transportable sediment within each block was higher for the mean of logged stations than their control mean (Table 6). No data were obtained for the Copper block. Statistical tests were limited by the few degrees of freedom included from three logged blocks; however, these data are included later with narrow-buffered streams to test for differences. When individual logged stations were compared to the mean of block controls only Mid New York had less sediment than its controls while two stations (Two Bit, New York Trib) exceeded control means by over 400% (Table 6, Fig. 3).

The range of values for individual stations was from 64.9 mg/cm^ (Empire Creek) to 242.1 mg/cm (Upper New York) for control stations and from 130.2 mg/cm^ (Mid New York) to a high of 883.1 (New York Trib) for the logged stations (Appendix D).

Transportable fine detritus was collected at the same time as sediment in the traps (Appendix D). Quantities varied from 21.2 mg/cm to 68.0 ing/cm^ for control means while means of logged stations ranged from 23.7 mg/cm^ to 126 mg/cm^ (Table 6). Three of the five logged stations had more detritus than the mean of their block controls (Fig. 2). New York Trib, which had the greatest difference in sediment compared to controls, also had the greatest difference in transportable detritus (+212%, Table 6, Fig. 2).

Periphyton Pigments and Incident Radiation

Logged streams had higher chlorophyll a concentrations (12% - 500%) than control . streams except in the Two Bit block (Table 7, Fig. 3). Two Bit and Lower Four Bit had more phaeophytin ***a*** (360% and 237%, respectively) than the control mean, while Copper A, New York Trib and'Mid New York had less than controls (Fig. 3, Table 7). Only Lower Taylor had more chlorophyll ***a*** and phaeophytin a than the mean of its controls (Fig. 3). When the logged stations were arranged in order of increasing total pigment complex an interesting pattern with respect to chlorophyll and phaeophytin was revealed (Fig. 3). With low quantities of total pigment, chlorophyll a was less than the controls while phaeophytin a exceeded controls. At higher levels of total pigment this pattern was reversed.

Incident radiation was greater in logged than control streams in all but Copper A (Table 6, Appendix C), which had regrown a dense understory of riparian vegetation. In addition, Copper B (control) was partially opened earlier by clearing during construction of a road crossing.

TABLE 4
SUM OF SQUARES AND SIGNIFICANCE (NESTED ANOVA BY BLOCK) BASED ON SUBSAMPLE ESTIMATES OF CLUSTER MEANS OF INVERTEBRATE DIVERSITY (H'), EVENNESS, DENSITY AND NUMBER OF TAXA

BLOCK	Value of $F_{.05}$	Diversity			Evenness			Log Density			Number of taxa		
		sum of squares between	sum of squares within	F_{obs}	sum of squares[1] between	sum of squares[1] within	F_{obs}	sum of squares between	sum of squares within	F_{obs}	sum of squares between	sum of squares within	F_{obs}
Logged													
Two Bit	3.49	.323	.125	10.40	1.68	.900	1.46	.003	.061	0.17	28.8	76.60	1.50
Copper	5.99	.114	.090	7.69	0.22	.948	12.10	.018	.020	6.15	47.5	3.53	13.50
Taylor	4.26	.240	.112	9.67	0.71	.666	4.78	.043	.054	2.92	89.5	23.20	17.40
New York	3.49	.812	.079	41.30	5.16	.576	35.90	.145	.054	10.60	15.7	6.10	2.34
Narrow Buffer													
Redwood	4.26	.318	.054	29.00	2.20	.792	12.50	.052	.128	1.83	33.2	20.20	7.37
Philpot	4.26	.298	.056	24.10	1.12	.477	10.70	.049	.029	7.48	118.5	59.30	18.10
Garden	3.49	.849	.091	37.20	2.87	.996	6.76	.155	.054	11.40	257.3	42.70	24.10
Packsaddle	4.26	.003	.125	0.11	0.23	.783	1.30	.094	.056	7.49	61.1	30.10	9.13
Fall	4.26	.101	.046	9.80	0.27	.477	2.56	.025	.091	1.24	15.4	12.90	5.37

---

[1] Actual values are 10^{-2} lower.

TABLE 5
CLUSTER MEAN VALUES AND STANDARD DEVIATION FOR LOG DENSITY, EVENNESS, NUMBER OF TAXA, SHANNON DIVERSITY AND STUDENTIZED RANGE TESTS (Q)[1] (p = 0.05) FOR EACH BLOCK

Stream	Log Density			Evenness			Number of taxa			Diversity		
	X	S.D.	Q	X	S.D.	Q	X	S.D.	Q	X	S.D.	Q
Two-Bit (L)	2.72	0.08	.20	.738	.058	.121	37.0	4.0	10.6	2.66	0.21	.43
L. Four-Bit (L)	2.71	0.26		.685	.090		34.3	4.3		2.41	0.29	
U. Four-Bit (C)	2.65	0.15		.785	.038		37.5	4.7		2.84	0.09	
Indian (RD)	2.71	0.12		.610	.004		30.8	6.8		2.08	0.17	
Copper A (L)	2.53	0.12	.20	.754	.041	.134	34.0	3.0	6.5	2.65	0.06	.42
Copper B (C)	2.34	0.11		.683	.101		24.3	3.2		2.17	0.33	
Lights (C)	2.71	0.17	.34	.781	.013	.108	36.8	3.9	6.3	2.82	0.13	.44
Up. Taylor (C)	2.42	0.19		.699	.037		23.5	3.4		2.20	0.15	
L. Taylor (L)	2.58	0.11		.665	.086		28.5	1.7		2.23	0.33	
Mid New York (L)	2.87	0.07	.23	.658	.043	.092	42.8	2.9	12.0	2.47	0.17	.24
New York Trib (L)	3.22	0.06		.433	.017		36.0	3.5		1.55	0.01	
Up. New York (C)	2.69	0.15		.735	.072		40.0	1.6		2.71	0.27	
Empire (C)	2.95	0.20		.663	.027		45.3	2.6		2.53	0.08	
Hare (N)	2.49	0.09	.47	.687	.051	.120	18.3	1.7	5.9	2.20	0.17	.29
N.Fk.S.Fk.Noyo (C)	2.18	0.22		.700	.029		18.3	1.7		2.03	0.11	
S.Fk. Noyo (RD)	2.28	0.34		.512	.087		17.3	3.9		1.44	0.15	
Naufus (N, RD)	2.20	0.11	.23	.651	.056	.091	18.3	4.0	7.2	1.87	0.16	.31
Philpot (C)	2.23	0.14		.801	.046		22.8	3.3		2.50	0.18	
Philpot Trib (C)	2.49	0.09		.736	.037		33.3	3.5		2.58	0.07	
Garden (N)	2.15	0.12	.28	.660	.108	.118	20.3	4.4	7.9	1.97	0.32	.36
LNF (N)	2.65	0.09		.642	.025		35.3	1.0		2.29	0.11	
Whites (C)	2.56	0.16		.779	.021		38.0	3.9		2.83	0.07	
Murphy (C)	2.31	0.08		.846	.009		41.8	2.8		3.16	0.09	
L. Packsaddle (N)	2.53	0.16	.31	.706	.058	.116	34.8	5.7	7.2	2.50	0.26	.47
U. Packsaddle (C)	2.38	0.07		.764	.007		26.5	1.3		2.50	0.02	
N. Packsaddle (C)	2.10	0.21		.765	.086		24.3	3.8		2.43	0.38	
Knopki (N)	2.16	0.25	.39	.776	.067	.091	28.8	2.2	4.7	2.61	0.21	.29
N. Fall (C)	2.36	0.18		.756	.033		28.3	2.8		2.44	0.10	
S. Fall (C)	2.35	0.11		.824	.034		33.3	2.2		2.89	0.07	

[1] Q values are the smallest differences between two means which are significant at the 0.05 level by the Studentized Range Test.

TABLE 6

TRANSPORTABLE SEDIMENT AND DETRITUS FOR INDIVIDUAL TREATMENT STATIONS AND CONTROL MEANS.

Data are for material passing through 0.30 mm sieve. Values in parentheses are for mean of treatments in blocks with more than one treatment station.

Block: Stream	Sediment (mg/cm^2)			Detritus (mg/cm^2)		
	Control (Mean)	Treatment	Change (%)	Control (Mean)	Treatment	Change (%)
Two Bit: [1]	90.6	(333.2)	(+268)	24.0	(23.7)	(-1)
Two Bit (L)		491.2	+442		29.7	+24
L. Four Bit (L)		175.1	+ 94		17.7	-26
Taylor:	150.4			21.2		
L. Taylor (L)		208.1	+ 38		38.9	+83
New York:	153.5	(506.6)	(+230)	68.0	(126.6)	(+86)
Mid New York (L)		130.2	- 15		41.2	-39
New York Trib (L)		883.1	+475		211.9	+212
Redwood:	51.4			16.3		
Hare (N)		58.2	+ 13		11.1	- 32
Philpot:	63.1	25.0				
Naufus (N, RD)		136.7	+117		14.6	- 42
Packsaddle:[1]	6.4			1.9		
L. Packsaddle (N)		6.6	+ 3		2.6	+ 27
Fall:[1]	36.2			13.0		
Knopki (N)		16.3	- 55		13.6	+ 5
Garden:	147.5			19.4		
Garden (N)		481.6	+227		49.0	+152

[1] Includes only one control stream.

TABLE 7

SUMMARY DATA FOR CHLOROPHYLL A, TOTAL PIGMENT MATRIX, INCIDENT RADIATION, AND PERCENT PHAEOPHYTIN A. DATA ARE FOR CONTROL MEANS, INDIVIDUAL TREATMENT STATIONS, AND PERCENT DIFFERENCES OF TREATMENT FROM CONTROL. VALUES IN PARENTHESES ARE FOR MEAN OF TREATMENTS IN BLOCKS WITH MORE THAN ONE TREATMENT STATION.

Block:	Ch 1 *a* (mg/m^2)			Pigment Matrix (moles 10^{-7} $/cm^2$)		
Stream	Control Mean	Treatment	Difference %	Control Mean	Treatment	Difference %
Two Bit[1]:	8.7	(2.8)	(- 68)	16.4	(8.8)	(- 47)
Two Bit (.L)		2.4	- 72		7.8	- 52
L. Four Bit (L)		3.1	- 64		9.7	- 41
Copper[1]:	2.5			6.5		
Copper A (L)		15.0	+500		26.4	+306
Taylor:	10.5			18.8		
L. Taylor (L)		12.9	+.23		23.8	+ 27
New York:	16.5	(20.3)	(+ 23)	38.4	(40.1)	(+ 4)
Mid New York (L)		22.2	+ 34		43.5	+ 13
New York Trib (.L)		18.3	+ 12		36.6	- 5
Redwood[1]:	6.3			14.4		
Hare (N)		3.8	- 40		8.0	- 42
Philpot:	5.9			12.2		
Naufus (N)		3.1	- 47		11.1	- 9
Garden:	14.2	(9.7)	(- 32)	25.6	(18.3)	(- 29)
LNF (N)		7.5	- 47		16.1	- 37
Garden (N)		11.8	- 17		20.4	- 20
Packsaddle:	3.4			7.2		
L. Packsaddle (N)		2.7	- 21		7.7	+ 7
Fall:	11.7			19.3		
Knopki (N)		9.8	- 16		16.1	- 17

---

[1] Includes only one control stream.

TABLE 7 (cont.)

Block:	Incident Radiation/(Ein/m^2-day)			% Phaeophytin		
Stream	Control Mean	Treatment	Difference %	Control Mean	Treatment	Difference %
Two Bit[1]:	3.8	(37.3)	(+882)	6.3	(25.2)	(+299)
Two Bit (.L)		36.9	+871		29.0	+360
L. Four Bit (L)		37.6	+893		21.3	+238
Copper[1]:	64.9			40.5		
Copper A (L)		4.2	- 94		19.5	- 52
Taylor:	32.1			13.1		
L. Taylor (L)		50.0	+ 56		17.0	+ 30
New York:	10.9	(23.2)	(+113)	38.5	(28.4)	(- 27)
Mid New York (L)		24.0	+121		22.3	- 42
New York Trib (.L)		22.3	+105		34.4	- 11
Redwood[1]:	5.5			22.0		
Hare (N)		16.0	+209		28.8	+ 31
Philpot:	50.0			17.9	28.8	+ 31
Naufus (N)		49.0	- 2		57.0	+218
Garden:	9.7	(35.1)	(+262)	17.8	(16.6)	(- 7)
LNF (N)		60.0	+519		17.8	0
Garden (N)		10.1	+ 4		15.3	- 14
Packsaddle:	10.5			19.7		
L. Packsaddle (N)		30. 1	+187		33.0	+ 68
Fall:	1.8			19.4		
Knopki (N)		1.8	0		9.5	- 51

[1] Includes only one control stream.

Narrow-Buffered Streams

Invertebrate Populations

Pooled cluster subsample values for community measures and density of *Baetis* and Chironomidae are given in Table 8. Diversity and evenness were significantly lower in narrow-buffered than controls ($P = 0.05$ and 0.01, respectively, Wilcoxon signed rank test), but density and number of taxa were not different between treatment and control streams.

Within blocks, analysis of variance showed four of five narrow-buffered blocks had significant differences in diversity (.Table 4). Naufus, Garden and Little North Fork had significantly lower diversity than their controls while Hare Creek, Knopki, and Lower Packsaddle were not different from controls (Q test, Table 5). Results from the Fall block were difficult to interpret because diversity of Knopki was not different from either control, but the controls were significantly different from each other.

There was a significant correlation between buffer width and diversity for all narrow-buffered streams ($P < 0.02$, $r_S = 0.90$, Spearman's rank correlation) showing an increase in diversity with an increase in buffer width (Fig. 4). The relationship was also significant when the recently disturbed narrow-buffered station (Naufus) was removed from the calculations.

Chironomidae and Bast-is made up the largest percentage of total density (Table 8), together accounting for about 50% and 30% of density in treatments and controls, respectively.

Transportable Sediment and *Detritus*

There was considerable variation in the quantity of transportable sediment and detritus among the blocks of narrow-buffered streams. The mean amount of transportable sediment in controls ranged from 6.4 mg/cm^2 in the Packsaddle block to 147.5 mg/cm^2 in the Garden block (Table 6). Individual narrow-buffered stations ranged from a low of 6.6 mg/cm^2 in Lower Packsaddle to a high of 481.6 mg/cm^2 (Garden). In all but the Fall block, narrow-buffered streams had more transportable sediment than the mean of comparable controls. The amount of sediment in narrow-buffered streams as a percentage difference from controls is shown in Fig. 2. Naufus and Garden (the smallest buffers) showed large percentage differences from controls and, along with Little North Fork (for which no sediment data were collected), were also the streams with lower diversity than controls. The inverse relationship between diversity and the amount of sediment was marginally not significant ($P < 0.07$, Spearman's rank test) with only five narrow-buffered streams.

A trend was not as clear for transportable detritus (Table 6, Fig. 2). In three blocks (Garden, Packsaddle, Fall) buffered streams exceeded the mean of controls. The range of control means was from 1.9 mg/cm^2 (Packsaddle block) to a high of 25 mg/cm^2 (Philpot block). The range for narrow buffers was from 2.6 mg/cm^2 (Lower Packsaddle) to a high of 49.0 mg/cm^2 (Garden). Data for individual streams are given in Appendix D.

When taken as a group, narrow-buffered and logged stations had significantly more transportable sediment than block control means ($p = 0.05$, Wilcoxon signed rank test). This difference in treatments from controls was substantial, averaging 133%.

Periphyton Pigments and Incident Radiation

In all narrow-buffered blocks, chlorophyll a concentrations were significantly lower ($p = 0.05$, Wilcoxon signed rank test) in the buffered streams than in respective controls (Table 7). The differences from controls ranged from 16% (Knopki) to 47% (Naufus and Little North Fork). Streams with the narrowest (Garden, 3 m) and widest (Knopki, 25 m) buffers had similar quantities of chlorophyll a and similar differences from the controls (Table 7). Total pigment concentration was also significantly lower in buffered streams (excluding Naufus) than their controls ($P < 0.02$).

Just as in the logged streams without buffers, there was a trend among the narrow buffers that as total pigment increased, there was less phaeophytin a relative to their respective controls (Fig. 3). This inverse relationship between total pigment and percent phaeophytin was significant (Spearman's rank test, $P < 0.01$).

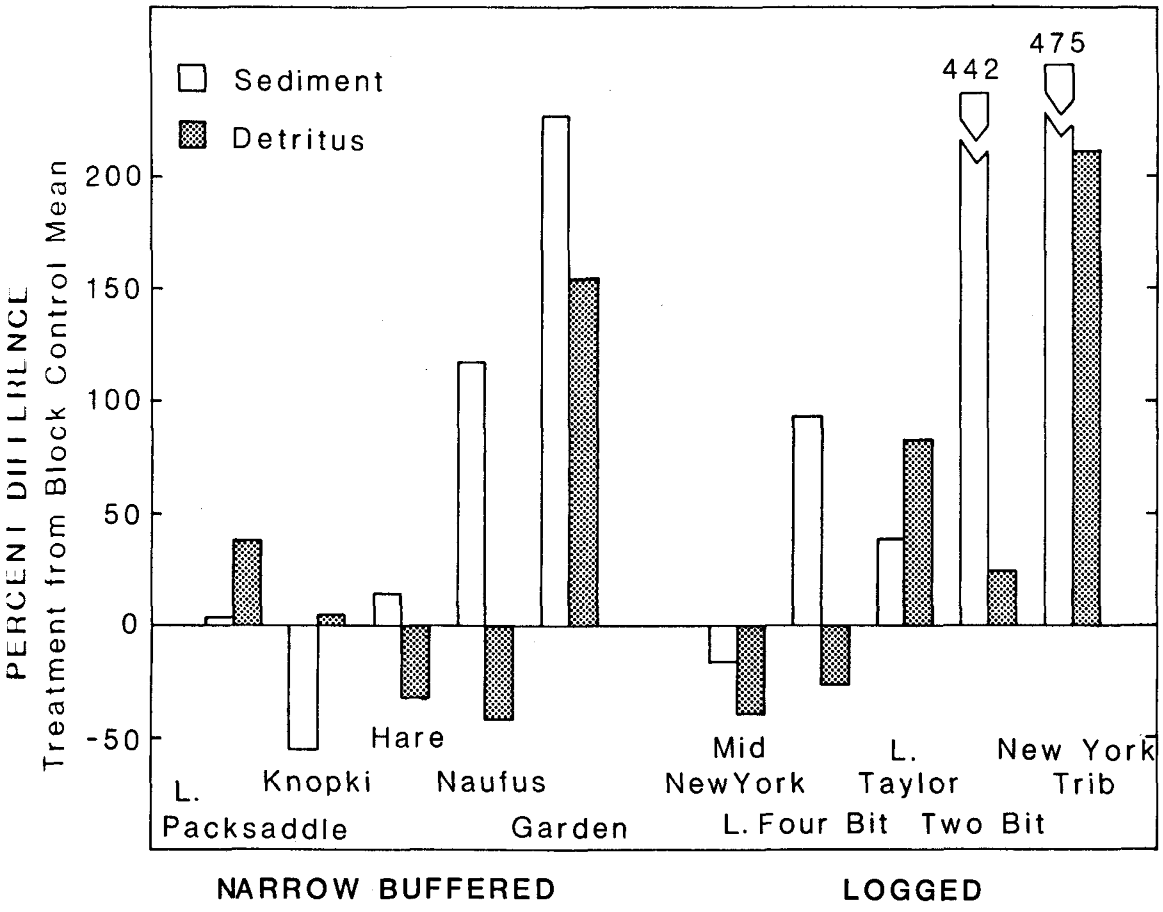

Figure 2. Percentage difference in transportable sediment and detritus.

Unbuffered Two Bit Creek in 1980 eight years after logging.

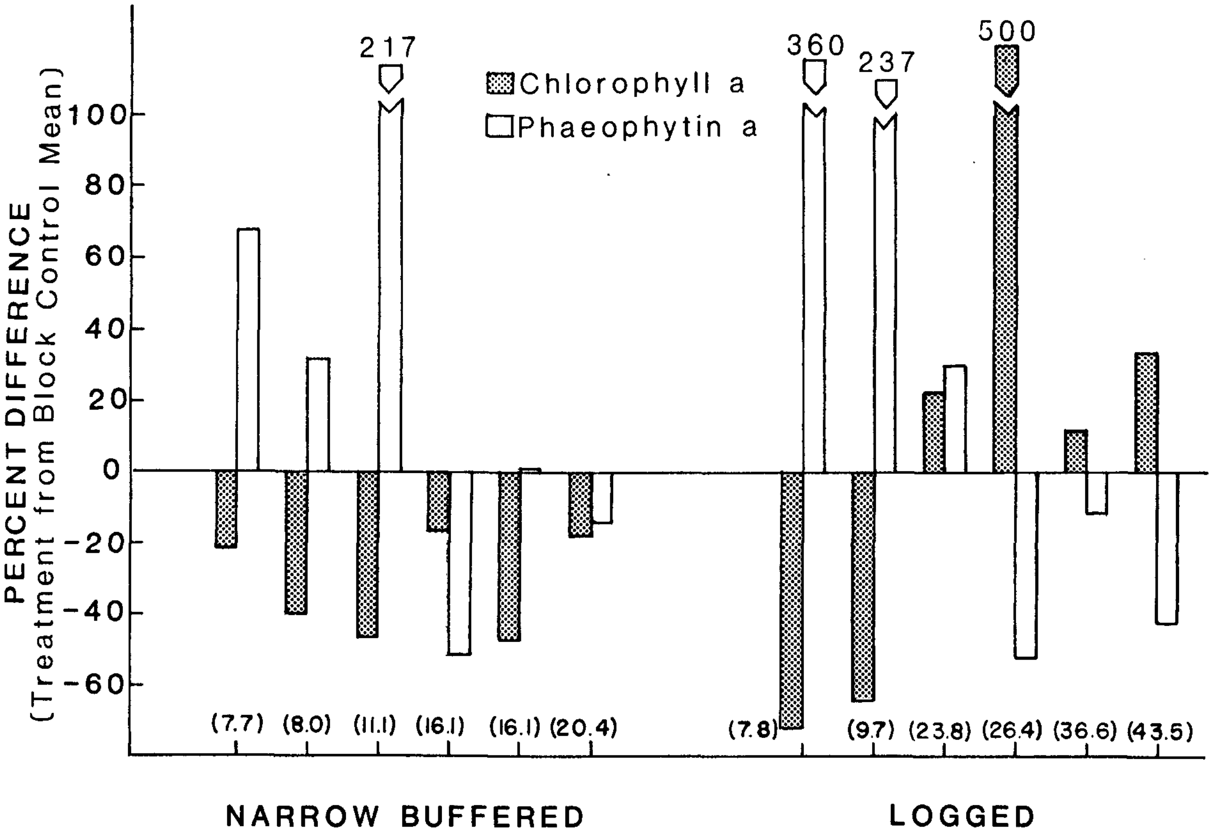

Figure 3. Percentage difference in chlorophyll *a.* and phaeophytin *a* compared to mean of controls for each block. Values in parentheses are the total pigment complex.

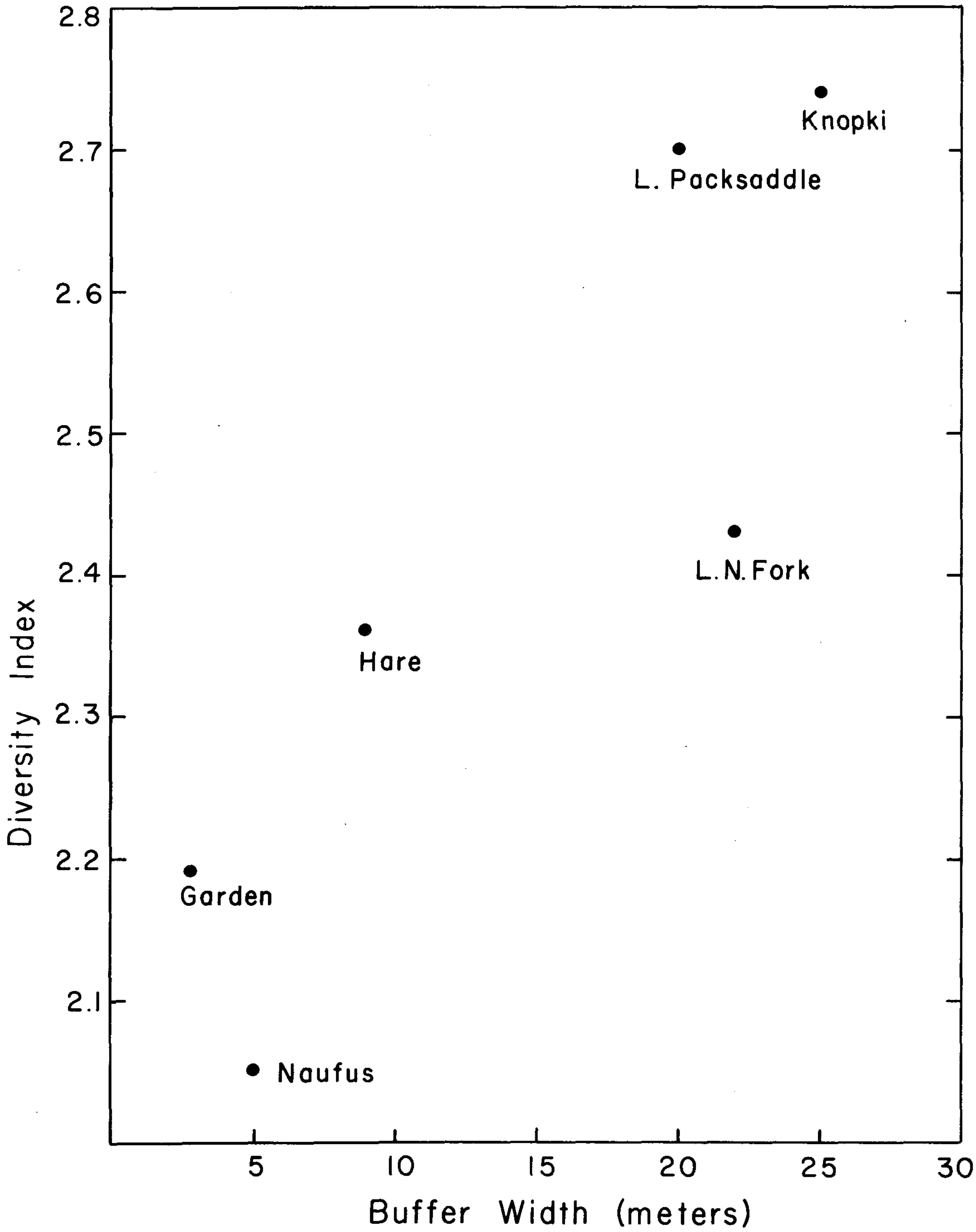

Figure 4. The relationship between buffer width and invertebrate diversity for narrow-buffered streams.

TABLE 8

VALUES OF POOLED SUBSAMPLE CLUSTERS FOR COMMUNITY MEASURES AND MEAN DENSITY OF SELECTED TAXA FOR NARROW-BUFFERED (N), CONTROL (C), AND RECENTLY DISTURBED (RD) STREAMS

Stream	Density (No./m^2)	Shannon Diversity	Evenness	Baetis (No./m^2)	Chironomidae (No./m^2)	No. of Taxa
Hare (N)	3130	2.36	0.648	603	1025	38
So. Fork Noyo (RD)	2320	1.54	0.436	115	1493	34
No. Fork, So. Fork Noyo (C)	1620	2.26	0.651	270	45	32
Naufus (N, RD)	1640	2.05	0.582	103	723	34
Philpot (C)	1760	2.80	0.777	263	210	37
Philpot Trib (C)	3080	2.73	0.698	513	608	50
L. Packsaddle (N)	3550	2.70	0.675	938	508	55
U. Packsaddle (C)	2410	2.64	0.698	633	165	44
N. Packsaddle (C)	1400	2.58	0.699	88	518	40
Knopki (N)	1630	.2.74	0.713	173	470	46
N. Fall (C)	2450	2.58	0.699	155	530	40
S. Fall (C)	2290	3.09	0.760	158	310	58
Garden (N)	1440	2.19	0.670	648	185	36
Little North Fork (N)	4700	2.43	0.611	1613	958	53
Whites (C)	3760	2.94	0.737	763	453	54
Murphy (C)	2070	3.26	0.795	193	223	60

Narrow-buffered Lower Packsaddle in 1980 six years after logging.

There was no relationship between buffer width and amount of incident radiation. There was also no simple relationship between the amount of incident radiation and either chlorophyll a or phaeophytin a (Table 7, Appendix E). As noted above all the narrow-buffered streams had less chlorophyll a and almost all had less total pigment than controls even though no similar pattern existed for incident radiation levels. Three of the narrow buffers had more light (187 to 519% of control means) than the mean for block controls, while three had virtually no difference (-2.0 to 4.0%) from controls (Table 7). We combined logged and narrow-buffered streams as treatments for a statistical test (Wilcoxon signed rank test) of differences in amount of incident radiation from block controls. This test showed that treatment streams received significantly ($p = 0.05$) more light than their controls.

Stream Recovery *Streams*

Without Buffers

Invertebrate diversity was generally higher in the present analysis when compared to the same stations sampled in 1975 (Table 9). However, there was no correlation in the ranks of diversity for the two studies (Spearman's rank correlation, $P > 0.35$). Percentage difference in diversity of treatment from control is shown in Fig. 5 for both studies. Control stations averaged 0.26 diversity units higher than the same stations in 1975, whereas logged stations averaged 0.62 units higher than the same stations in 1975. In 1975 logged stations had averaged 0.60 diversity units lower than their block controls while in 1980-81 the same stations were only 0.24 less than their block controls. Thus there was a 0.36 increase in mean invertebrate diversity in the logged blocks compared to controls from 1975 to 1980-81. In terms of percent difference, diversity averaged 25.2% lower in 1975 at logged stations relative to controls while in 1980-81 this figure was only 9.1%.

Comparison of another measure of recovery was made by testing ranks of Pfankuch ratings in 1975 vs. 1980-81. Ratings of treatment streams in 1980-81 were significantly lower (thus improved) than in 1975 ($P < 0.001$, Wilcoxon signed rank test). The mean Pfankuch rating was 88.3 (1980-81) vs. 103.5 (1975). Comparison of all control stations showed no difference in ratings between study periods (67.1 in 1975, 72.8 in 1980-81).

Some of the changes in diversity between studies were a result in 1980-81 of more taxa and less concentrated density in a few taxa than in 1975. For example, mean number of taxa in logged streams was 49.7 in 1980-81 compared to 34.8 in 1975 and the mean percentage of density made up of Chironomidae was 24% in logged streams in 1980-81 compared to 51% in 1975.

Streams *With Narrow Buffers*

As in the logged blocks, diversity of narrow-buffered stations was generally higher than in 1975 (Table 9). Comparison of ranks of diversity for both studies showed they were highly correlated ($P < 0.05$), Spearman's rank). Control stations in narrow-buffered blocks averaged 0.34 units of diversity higher than the same stations in 1975, whereas, the mean of narrow-buffered stations in 1980 was 0.30 higher than the same stations in 1975. In 1975 the narrow-buffered stations had averaged 0.27 lower than their block controls while in 1980 this figure was 0.31. This value represents little change (-0.04) in diversity units in narrow-buffered streams compared to controls from 1975 to 1980. In terms of percent variation, diversity averaged 12.4% lower in narrow-buffered streams in 1975 relative to controls while in 1980 diversity averaged 12.5% lower. Naufus Creek (RD) was excluded from these calculations, but it is included for comparison for Fig. 6 to show percentage difference in diversity between treatment and control for both studies.

Knopki, the widest of the narrow-buffered stations, had higher diversity than controls in 1975 but was slightly less than controls in 1980 (Fig. 6). The relationship between buffer width and diversity was still significantly correlated (as noted earlier) and slopes of simple linear regression for the relationship were nearly identical in both studies; 0.023 in 1975, 0.025 in 1980. Pfankuch ratings of stream stability were also not significantly different, 101.8 in 1975 and 95.8 in 1980 (Wilcoxon signed rank test).

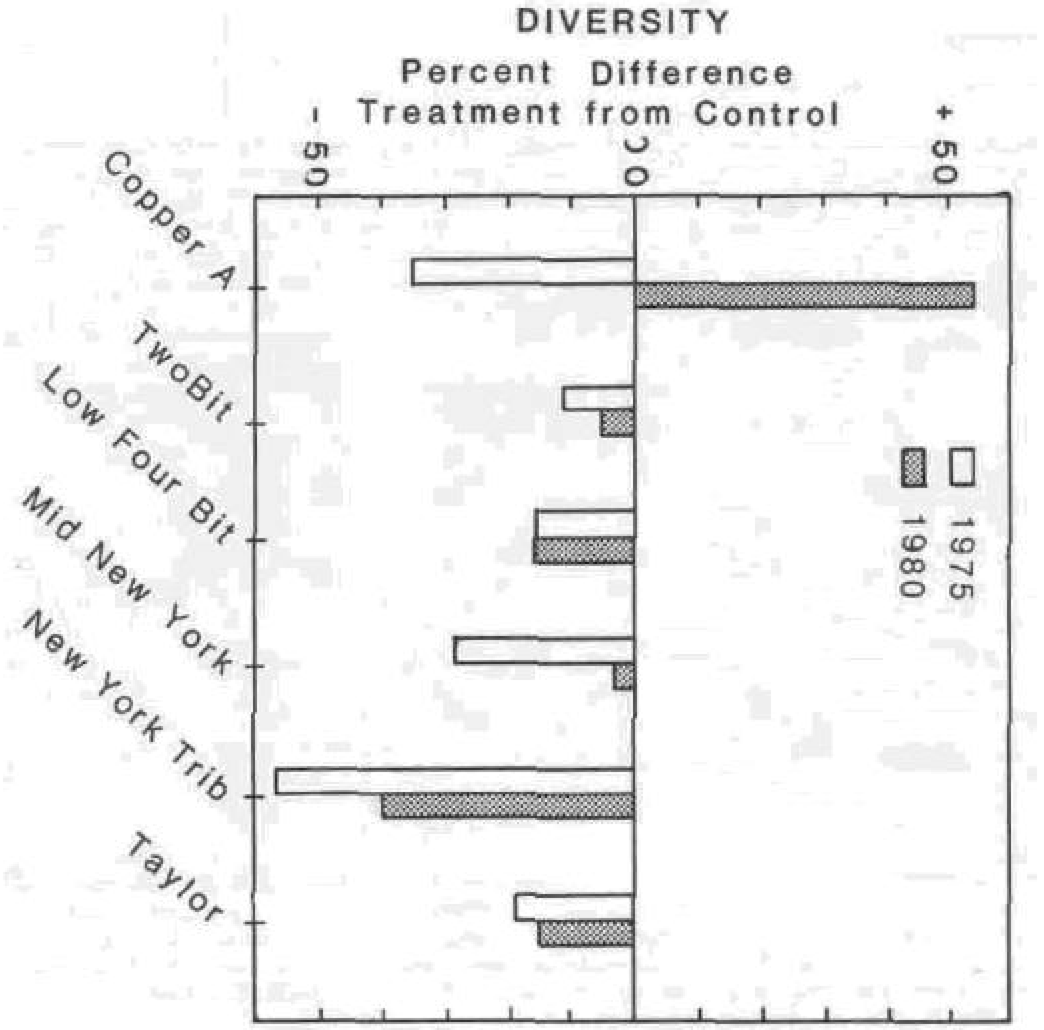

Figure 5. Percentage difference in diversity for 1975 study and in 1980-81.

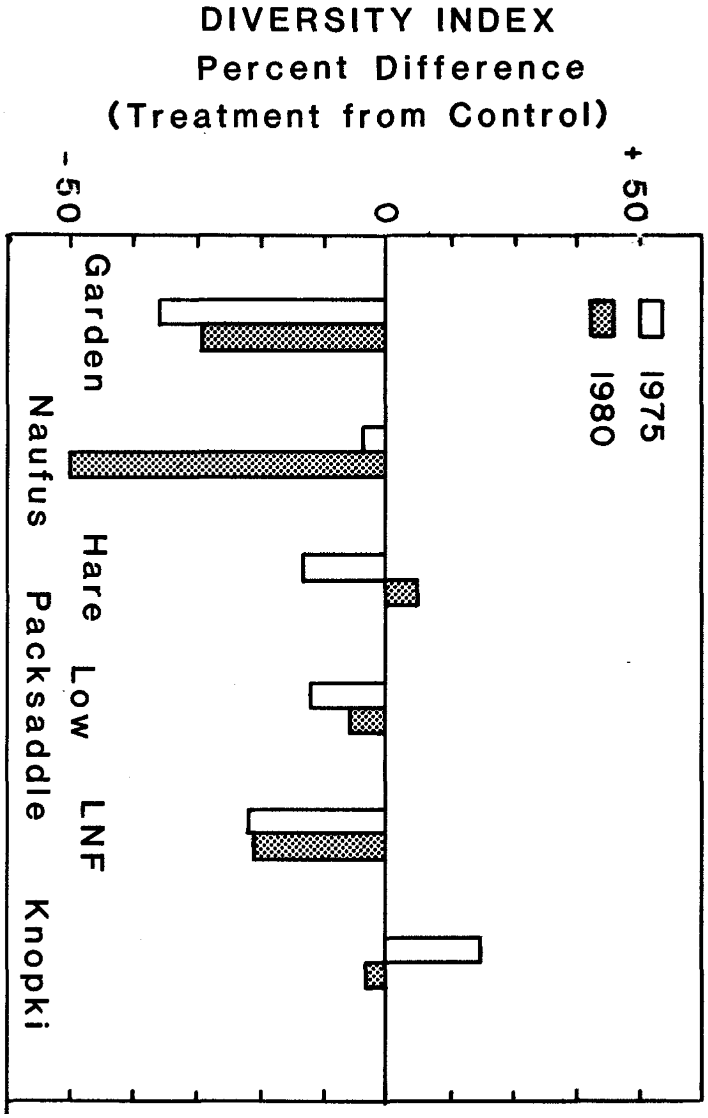

Figure 6. Percentage difference in diversity for narrow-buffered streams in 1975 and 1980. Streams are arranged in increasing buffer width. Naufus was recently disturbed.

TABLE 9

COMPARISON OF DIVERSITIES FOR LOGGED, NARROW-BUFFERED, AND CONTROL STREAMS FOR 1975 (Erman et al. 1977), AND 1980-81

	DIVERSITY (H')				
	1975		1980-81		CHANGE
	x	S.D.	x	S.D.	1975 to 1980-81
Logged Blocks					
Treatments (n)	1.78 (6)	0.55	2.40 (6)	0.44	+0.62
Controls (n)	2.38 (7)	0.35	2.64 (6)	0.32	+0.26
Difference	-0.60		-0.24		+0.36
(%)	(-25.2)		(-9.1)		
Narrow-buffered Blocks					
Treatments (n)	2.18 (5)	0.37	2.48 (5)	0.23	+0.30
Controls (n)	2.45 (9)	0.25	2.79 (9)	0.48	+0.34
Difference	-0.27		-0.31		-0.04
(%)	(-12.40)		(-12.50)		

Environmental and Invertebrate Community Correlations

Several significant correlations between invertebrate community parameters were found (Table 10). Although correlations do not necessarily imply a causal mechanism, the interesting feature of this analysis was that different patterns emerged within the treatment and control groups. In control streams the general pattern was for density, number of taxa, and diversity to increase together. Thus in controls increased diversity was associated with increased density ($p < 0.05$), and increases in taxa were associated with higher overall density ($p < 0.05$). By contrast, in treatment streams increased density of some taxa lowered the overall diversity, and the evenness component of diversity was inversely correlated with total density ($p < 0.01$).

A number of environmental characteristics also were significantly related to measurements of the invertebrate community, and these relationships also differed between treatment and control streams. In controls diversity was associated with stream hydrological characteristics — fewer taxa and lower diversity occurred in streams with higher discharge levels ($p < 0.05$). In treatment streams diversity was associated most with substrate characteristics. Higher amounts of both transportable sediment and detritus were associated with higher invertebrate density, lower evenness, and lower diversity.

Relationships between periphyton pigments and invertebrate community parameters also varied between treatment and control streams. Number of taxa increased in control streams with an increase in chlorophyll a and greater density was associated with higher total pigments as well as degraded pigments (Table 10). In treatment streams, invertebrate density increased with an increase in degraded pigments ($p < 0.01$) but with no concomitant increase in number of taxa. There was a negative correlation of degraded pigments with evenness ($p < 0.01$) and the diversity index ($p < 0.05$).

TABLE 10

SIGNIFICANT PEARSON CORRELATION COEFFICIENTS BETWEEN ENVIRONMENTAL AND INVERTEBRATE COMMUNITY PARAMETERS FOR CONTROL (n = 12) AND TREATMENT (n = 10) STREAMS

Parameters		Control	Treatment
Diversity with	Number of taxa	+ .698*	+ .594
	Sediment	+ .164	- .642*
	Detritus	+ .046	- .767**
	Degraded pigments	- .070	- .652*
	Discharge	- .772**	- .023
	Current velocity	- .651*	- .001
Density with	Sediment	+ .083	+ .721*
	Detritus	+ .186	+ .896**
	Total pigments	+ .678*	+ .617
	Degraded pigments	+ .631*	+ .902**
Evenness with	Density	- .344	- .823**
	Sediment	+ .089	- .642*
	Detritus	+ .039	
	Degraded pigments	- .338	- .861**
			- .849**
Number of taxa with	Density	+ .624*	+ .256
	Chlorophyll a	+ .639*	+ .202
	Discharge	- .665*	- .228
Sediment with	Bankfull index	- .592*	- .473
Detritus with	Current velocity	- .615*	+ .069
Bankfull index with	Avg. annual precip.	+ .561	+ .686*
Avg. current velocity with	Discharge	+ .819*	+ .574
	Drainage area	+ .334	+ .749*
Chlorophyll a with	Sediment	+ .662*	+ .370
	Percent degraded	+ .625*	- .226
Percent Chlorophyll a in pigments with	Total radiation	- .182	+ .673*
	Chlorophyll a	- .073	+ .659*
Total pigments with	Detritus	+ .629*	+ .560
	Chlorophyll a	+ .984**	+ .950**
Degraded pigments with	Total pigments	+ .879**	+ .774**
	Detritus	+ .729**	+ .883**
	Sediment	+ .515	+ .668*
	Chlorophyll a	+ .730**	+ .698*
Other pigments with	Total pigments	+ .158	+ .858**
	Chlorophyll a	+ .376	+ .880**
Pfankuch rating with	Mean grain size	- .237	- .765**

*$p < 0.05$

**$p < 0.01$

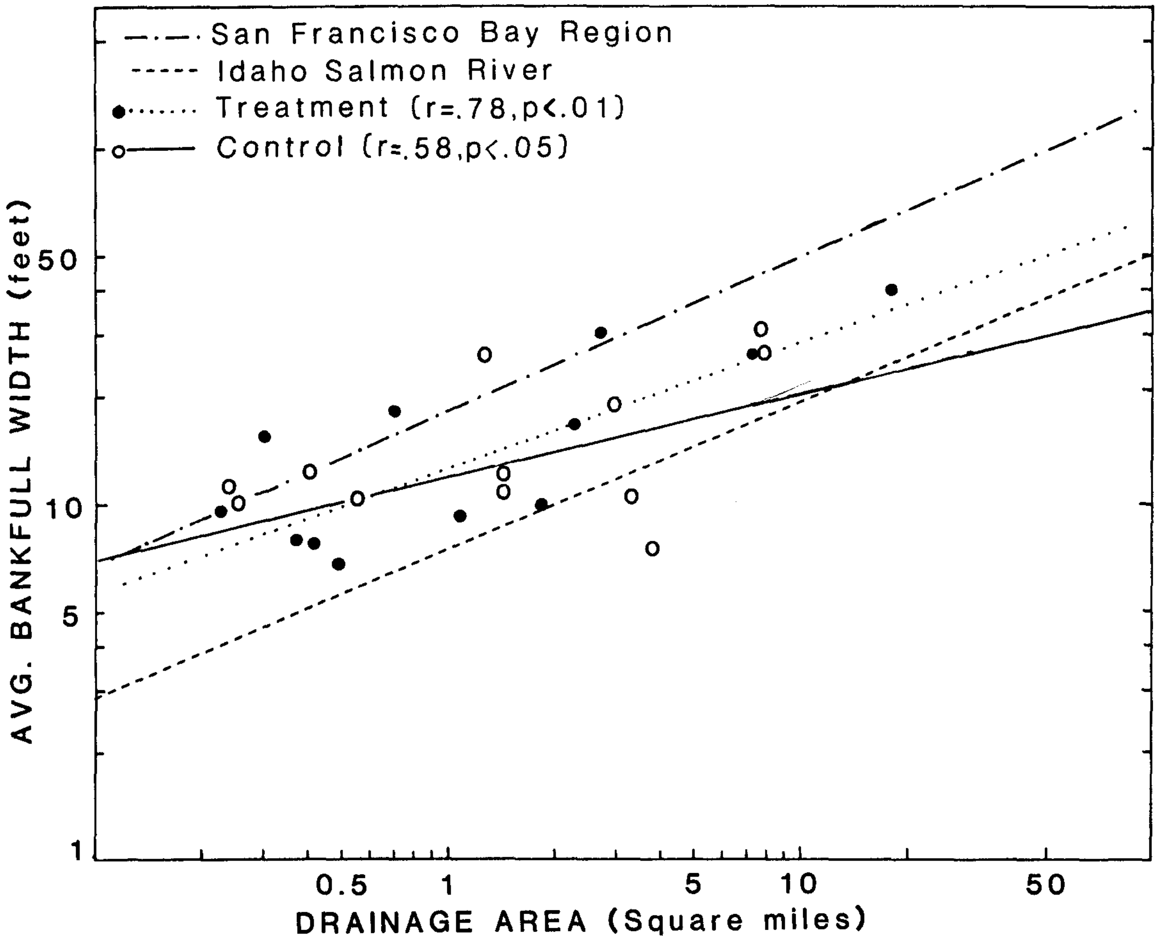

Figure 7. Relationship between bankfull width and drainage area for treatment (n = 12) and control (n = 12) streams. Data for other regions from Dunne and Leopold (1978).

TABLE 11

MULTIPLE STEPWISE REGRESSION OF ENVIRONMENTAL CHARACTERISTICS AND INVERTEBRATE COMMUNITY PARAMETERS

	Dependent	Independent	Sign	F	Explained Variation (%)	Multiple R	R^2
Treatments	Shannon Diversity	Degraded pigments	-	17.8**	43	.883	.779
		Avg. annual precip.	+	9.8**	15		
		Bankfull index	-	5.6*	21		
	Density	Sediment	+	52.5**	48	.976	.953
		Detritus:sediment ratio	+	46.0**	25		
		Total radiation	+	22.5**	13		
		Mean grain size	-	9.1**	9		
	Evenness	Degraded pigments	-	64.0**	72	.958	.918
		Avg. annual precip.	+	14.0**	5		
		Bankfull index	-	10.5**	14		
Controls							
	Shannon Diversity	Discharge	-	29.1**	60	.943	.889
		Bankfull index	-	0.6	10		
		Degraded pigments	-	11.8**	5		
		Chlorophyll a	+	8.8**	14		
	Density	Total pigments	+	13.7**	46	.779	.606
		Sediment	-	3.4	15		
	Evenness	Current velocity	-	52.7**	29	.970	.940
		Degraded pigments	-	52.0**	31		
		Chlorophyll a	+	24.2**	16		
		Mean grain size	+	24.2**	18		

*$p < 0.05$

**$p < 0.01$

The bulk of the pigment matrix in both treatment and control streams was composed of degraded pigments and chlorophyll a and so these three parameters increased simultaneously (Table 10). Fine detritus levels paralleled changes in total pigments and degraded pigments in control streams and transportable sediment was positively correlated with chlorophyll a. In treatment streams fine detritus was correlated only with degraded pigments, and although chlorophyll a concentrations were unrelated to physical parameters, the percentage of chlorophyll a in the pigments increased with decreasing levels of sunlight.

The Pfankuch ratings of all streams ranged only from good to fair (48 to 113) (Appendix C). Two narrow-buffered streams and 9 control streams ranked good while 4 narrow-buffered, all 6 logged and 5 control streams ranked fair. The Pfankuch rating was inversely correlated with mean grain size diameter in treatments indicating a possible destabilizing effect of fine sediments on the stream channel.

The log of individual drainage areas was correlated with the log of bankfull widths ($p = 0.05$ controls, 0.01 treatments. Fig. 7). Bankfull width was found to rise more rapidly with respect to drainage area in treatments than in controls.

An overall correlation of community measures with environmental variables was analyzed by a series of stepwise multiple regression. Only the factors which accounted for most of the variation (R^2) are reported although all 13 variables were added (Table 11).

About 78% of the variation in diversity of treatment streams was accounted for by three variables — degraded pigments (43%), bankfull index (21%), and average annual precipitation (15%). The same general pattern applied to the evenness component with the three variables accounting for somewhat more variation. Four different variables accounted for most of the variation in density of treatment streams.

Two to four variables accounted for most of the variation (60.6 - 94.0%) in control stream community measures and only bankfull index, fine sediment and degraded pigments were important to both treatment and control stations.

DISCUSSION Stream Recovery

We have defined recovery as the extent to which the diversity index of macroinvertebrate communities in logged or narrow-buffered streams resembles unlogged streams now as compared to the past. In an earlier study (Erman et al. 1977, Roby et al. 1977, Newbold et al. 1980) two measures of community similarity (Euclidian distance and chord distance) and two measures of diversity (Shannon and the reciprocal Simpson index) were used to evaluate the effects of logging on invertebrate communities in these streams. The Shannon diversity index and Euclidean distance gave equivalent results and clearly showed the differences between treatment and control stations. We used the Shannon index in the current study because we had fewer stations and had lost some controls compared to the earlier work. The Shannon index is sensitive to numerous particulars of sampling technique as well as taxonomic completeness (Hughes 1978), thus we used, as nearly as possible, the same stations, the same sampler, and the same sample periods as used in 1975. We departed from the previous study by using different people, and by taking subsamples of 16 samples. Taxonomic categories (mixed levels from species to order) were kept similar and reference collections were used to maintain consistency but differences inevitably occur, especially in the identity of Diptera.

We systematically found higher densities and diversity in 1980-81 than in 1975 primarily because of differences in techniques. Hence, comparison of absolute values of diversity between years is limited. Nevertheless, within a study diversity differences of treatment (logged and buffered) from controls are valid (Hughes 1978). In addition, because the current samples have higher diversity in general compared to 1975, a substantially lower diversity index at a particular station now than in 1975 was interpreted as evidence of increased disturbance.

Recovery of benthic communities has occurred in streams without buffers when compared to their condition shortly after logging. However, recovery is incomplete and some logged streams still have significantly lower diversity than controls. The primary basis for concluding stream recovery comes from the much narrower difference in diversity in 1980-81 between logged and control streams than in 1975. Streams without buffers averaged 9% lower diversity than controls in 1980-81 compared to *25%* lower in 1975.

Although most streams showed recovery, the variation among streams in their rate of recovery was large and reflects differences in the degree of disturbance, differences among regions, and natural variation among watersheds within a region. For example, in 1975 Copper A (2 years post logging) was the most disturbed station of all streams studied and had high density of chironomids (about 7000/m^2) that resulted in a very low diversity index (Erman et al. 1977). Seven years after logging, chironomids were about one-tenth as abundant as in 1975 and the number of taxa had increased dramatically (from 20 to 52). Vegetation had regrown around the logged stream and on the surrounding watershed. This watershed receives the most precipitation of all the logged watersheds we studied and invertebrate diversity is the most recovered both with respect to the previous study and to the control block (Copper B) which it now exceeds in diversity. Roadwork bordering the control, while not affecting sedimentation into the stream, did expose a section of the stream to full sunlight. The greater diversity in Copper A relative to the control is the only case where a treatment stream exceeded a control.

In the New York block abundant vegetation had also regrown on this watershed. No canopy remained over New York Trib and only 33% remained over Mid New York in 1975 (see photos), whereas in 1981, both stations were virtually covered by riparian regrowth. One station (Mid New York) appears recovered relative to the previous study and to controls; however, New York Trib still has a very low diversity. A limnephilid caddisfly larva *(Farula),* present at all stations in the block, was extremely abundant (4237/m^2) in New York Trib and lowered the diversity index as a result. This station also had the highest sediment and detritus of all stations sampled. Road failure was thought a major factor in causing the New York block to be the second most disturbed logged block in the earlier study (Erman et al. 1977), but the road and crossing over New York Trib appeared stable in 1981. Apparently, residual sediment stored in the channel and banks from the previous disturbance has continued to influence invertebrate communities. Roads are a major source of sediment in logged watersheds (Burns 1970, Brown and Kriegier 1971) and high sediment levels have been associated with changes in invertebrate density (Tebo 1955, Burns 1972, Barton 1977, Lenat et al. 1981) and community structure.

Streams with narrow buffer strips, judged less adequate than wide (30 m) buffer strips earlier (Newbold et al. 1980), have recovered little from their condition in 1975. In 1980, 7-10 years after logging, these streams showed effects comparable to streams without protective buffers. Five of the six narrow-buffered streams had no additional logging activities since the last study yet virtually the same relationship persists between increasing diversity with increasing buffer width (up to 25 (%) for this group). The mean difference in diversity between narrow-buffered streams and controls has changed only 0.1% since 1975 and averaged 12.5% lower in treatments than controls; a greater difference than we observed for unbuffered streams.

The observed relationship of diversity to buffer width in 1975 was a major reason for concluding that buffer width was an important variable for protecting stream communities from the impact of logging. Although the group of streams with less than 30 (%) buffers showed increasing impacts with decreasing buffer width, as evidenced by the diversity index, some protection was nevertheless provided compared to the logged streams without suffers (Newbold et al. 1980). If these narrow-buffered streams also recovered more rapidly or completely compared to unbuffered streams then their value in a longer term management system would be further enhanced. Possible compromises between short-term disturbance but rapid recovery with narrow buffers versus full protection with wide buffers could become part of the planning by forest managers. Our results do not support such a compromise. Narrow-buffered streams which initially showed logging impacts have not recovered more completely or more rapidly than the unbuffered streams and continue to exhibit the same relationship of decreasing invertebrate diversity with decreasing buffer width. Measures of stream channel stability (PfanRuch ratings) and transportable fine sediment also support these trends in invertebrate diversity. Unbuffered streams have improved Pfankuch ratings compared to 1975 while no change was evident in narrow-buffered streams. In general treatment streams have more fine sediment than controls even 6-10 years after logging.

These results show that persistent, long-term impacts are present in many of the treatment streams and that there is a difference in the rate of recovery between unbuffered and buffered streams. Other workers also have noted that benthic invertebrate communities take a long time to recover after disturbance from logging and some have concluded that the process is dependent on recovery of terrestrial vegetation

New York Tributary as it looked in 1981 (above) and in 1975 (below) 1 year after logging.

(Gurtz 1981, Haefner and Wallace 1981, Murphy et al. 1981). We have some evidence that streams showing most recovery are located in watersheds with higher average annual precipitation, and we assume that high precipitation leads to rapid vegetation regrowth. Average annual precipitation accounted for a significant amount of variation in treatment stream diversity. Copper and New York blocks had the greatest recovery since 1975 and they have higher average annual precipitation than the other logged blocks. Likewise, Hare, Lower Packsaddle and Knopki have higher precipitation than other narrow-buffered streams, and they have the least difference from controls in 1980.

Generally, the logged stations had recovered more relative to controls and had changed more since 1975 than the narrow-buffered stations. These differences appeared not only in the average change in diversity since 1975 but also in changes of a few abundant taxa. In all the logged stations one of the most obvious differences since the last study was the reduced dominance of Chironomidae. They made up about 24% of the density in 1980-81 while in 1975 they accounted for over 50%. A similar trend in ***Baetis*** and ***Nemoura*** has occurred and although these same taxa were often the most abundant at a station, they made up a smaller percentage of the total density of logged stations than in 1975. A similar reduction in relative abundance of these taxa did not occur in narrow-buffered streams. Together Chironomidae and ***Baetis*** made up about 50% of total density in both studies. Chironomids seem especially clear indicators of logging-based disturbance as shown by field studies and manipulations of artificial stream channels (Burns 1972, Newbold 1977, Triska et al. 1983).

This difference in recovery rate between unbuffered and narrow-buffered streams may result from some inhibition of the process by partial buffers or, more likely, may be a function of the rate at which the process changes over time. If stream benthic communities are perturbation independent systems, then recovery after logging disturbance is slow and a plot of diversity over time is S-shaped (Vogel 1980). Thus greatly disturbed systems could show rapid early recovery (unbuffered streams), but thereafter slowly approach the pre-disturbed (control) condition. In this way, less disturbed streams (narrow-buffered) would start recovery in the slow phase, thus appearing to show less change than greatly disturbed streams. A factor which confounds our analysis is the method of logging used in the two treatments. Four of the six logged stations had selective logging while all of the narrow-buffered stations were in clear-cuts.

Effects of Logging on Invertebrate Communities

There are few other studies that have looked at invertebrate diversity in relation to logging. In one study in New Zealand Graynoth (1979) compared logged, wide-buffered, and control streams for one year after logging and found significantly lower diversities in one clearcut but unbuffered stream relative to a control stream and found no difference between a wide-buffered and control stream. In another clearcut and unbuffered stream he found lower diversities than in the control but was not sure if this was due solely to logging or to intrinsic differences between watersheds. Murphy and Hall (1981), on the other hand, found increased species richness among predatory insects after clearcutting of nine sites in the Cascade Mountains of Oregon relative to paired controls.

Several studies have shown invertebrate density and biomass increase in clearcut sites (Murphy and Hall 1981, Murphy et al. 1981, Graynoth 1979, Gurtz 1981, Haefner and Wallace 1981). In general increased density occurs in a small number of taxa thus leading to a less even distribution of individuals among all taxa. Our results Indicated that increased density often occurred along with decreased evenness and thus diversity since the evenness component strongly influences the Shannon diversity index. There were three notable exceptions to this pattern in our study. Naufus Creek (N, RD), Garden (N), and South Fork Noyo (RD) all had lowered density, lowered number of taxa and diversities compared to controls. These differences are more typical of effects from local disturbance than from overall timber harvesting (Erman et al. 1977) and suggest these streams receive the most severe impacts of all stations.

Effects of Logging on the Stream Environment

Transportable Sediment and Detritus

Higher levels of transportable sediment than in controls remain in most treatment streams in this study and this may be an effect of movement of pulses of sediment down the stream channel from erosion from previous years, the scouring and eroding of stream banks by high flows, or continued erosion from roads or bare slopes. The amount of transportable sediment was negatively correlated with invertebrate diversity in our study primarily because of higher density in streams with more sediment. Other authors have found variable effects of sediment on benthos (Gammon 1970, Murphy and Hall 1981, Lenat et al. 1981). The transportable sediment consisted of size classes less than 0.3 mm. Increases in fine sediment in the top layer of the substrate have been reported by many (Burns 1970, Scrivener and Brownlee 1980, Paustian and Bestcha 1979). Murphy and Hall (1981) found increased fines 5 to 10 years after logging depending on stream gradient, with low gradient streams flowing through clearcuts having more fine sediments than old growth controls and with high gradient clearcut streams having less fine sediment than controls. They felt that a reduction in large debris in the high gradient clearcut streams allowed greater flushing of fines than occurred in controls. But in another study of a clearcut stream in Oregon flushing of fines was insufficient for returning permeability of gravel beds to pre-logging levels, even after seven years (Moring 1982). Permeability levels in a control and wide-buffered streams were unaffected, and Moring (1982) stated that "during disruptive land use stream flow may have a more significant role in depositing excess sediment than flushing it from the system."

Our measure of transportable detritus includes not only small debris from leaf fall and other terrestrial sources but also any dead or dying algae present at the sample site. Transportable sediment and detritus are highly intercorrelated (Gurtz et al. 1980, Murphy et al. 1981), thus possibly the ratio of sediment to detritus may be as important as either factor alone (Barton 1977, Gurtz et al. 1980). Among treatment stations the range of detritus levels was much greater than in control stations. Increases in algal production and the presence of algal blooms may have contributed to this variability since detritus and degraded pigments were highly correlated. Gurtz (1981) also found greater variability in the detrital component of the seston than the inorganic component after clearcutting and attributed this response to changes in retention capabilities of the clear-cut stream. Murphy et al. (1981) found that organic matter in the streambed corresponded to channel gradient and not to successional stage of the surrounding vegetation, and that "dead algae probably enriched the organic detritus in the streambed."

Periphyton Pigments

Changes in invertebrate communities after logging have been attributed partially to changes in food pathways brought about by alteration of the primary producer biomass (Erman et al. 1977, Vannote et al. 1980, Murphy et al. 1981). Our analysis used several aspects of periphyton pigment standing crop; chlorophyll *a,* phaeyphytin a, other pigments, and total pigment matrix.

One would expect that logging would increase light following opening of the canopy, and increased periphyton in the attached algal community would follow. Algal blooms were reported in Oregon after clearcutting (Hansmann and Phinney 1973) and by Likens et al. (1979) in conjunction with increased nutrient inputs. Lyford and Gregory (1975) observed a change in another Oregon stream from a diatom to a filamentous green algal community after clearcutting and increased solar radiation. Periphyton biomass increased fivefold in stream channels with little shade and some nutrient additions in a northern California stream (Triska et al. 1983). In general, we found most logged and narrow-buffered streams still received significantly more light than controls, and mean daily radiation was similar for both treatment types (29.2 Ein/m^2-day in logged, 27.8 Ein/m^2-day for narrow-buffered). But the amount of chlorophyll a or total pigments was not consistently different from controls. Chlorophyll a of logged streams exceeded mean amounts in most controls while buffered streams had less chlorophyll a than controls. In fact, treatment streams as a group had a negative correlation between radiation and the percent chlorophyll a in total pigments. Thus as radiation levels increased (generally the treatment streams) other pigments and phaeophytins became relatively more abundant than chlorophyll a. A similar relationship between algal biomass and chlorophyll has been observed; algae growing in bright light have less chlorophyll a (Brown and Richardson 1968) and a higher biomass to chlorophyll ratio (Mclntire and Phinney 1965, Lyford and Gregory 1975) than algae growing in shade. This ratio also may increase with increasing age of algae (Kureyshevich 1980), or may decrease under grazing pressure (Kehde and Wilhm 1972). Another factor which makes interpretation of chlorophyll a levels difficult is the complex pigment system of algal populations. In many species of blue-green algae, phycobiliproteins capture more light energy than do chlorophylls (Meyers et al. 1978) as they may in red algae, dominant in several of our streams.

These results illustrate the difficulty of relying on chlorophyll a as a clear indicator of disturbance or recovery following logging. The continued impact of logging on the streams we studied showed in higher radiation levels (compared to controls) and a greater percentage of non-chlorophyll a pigments compared to controls. Also when the total pigment is low in treatment streams, the degraded pigments in particular are much higher than in controls.

Such a relationship between total pigment and phaeophytins might occur under conditions of cyclic periphyton blooms. Algae would build up then slough off in a decay phase resulting in our observation that several treatment streams had less chlorophyll but more phaeophytins than controls. At three stations (Naufus, South Fork Noyo, Copper B) algal blooms recently died as evidenced by masses of decaying filamentous mats caught in rocks and deposited in pools.

While cyclic blooms and senescence of algae may lead to greater annual primary production in a disturbed stream than in a comparable control, this production would be less sustained and predictable than in an undisturbed stream, especially because blooms occur in summer when terrestrial detritus inputs are at a minimum. Two of the three streams in our study, where blooms had just decayed (Naufus, South Fork Noyo), had decreased density and diversity of benthic invertebrates. In streams where enhanced algal blooms do occur, primary producers may provide as much energy for collector-gatherer functional groups as they do for grazers, either directly through the detrital pathways as senescent cells or through enrichment of the organic matter of the streambed (Murphy et al. 1981).

CONCLUSIONS

1. Small streams without buffers and with narrow buffers (< 30 m) still exhibit significantly lower macroinvertebrate diversity 6-10 years after initial logging activity than undisturbed controls.

2. Streams without buffers show substantial but incomplete recovery 7-8 years post-logging compared to their immediate post-logging conditions.

3. Streams with narrow buffers on the average show little recovery 6-10 years post-logging and still have a correlation of increasing diversity with increasing buffer width. This result supports the conclusions of an earlier study that unless logging is carefully executed, buffers less than 30 (m) are inadequate to protect the stream. Furthermore, narrow buffers do not enhance recovery rate compared to unbuffered streams.

4. Both logged and narrow-buffered streams 6-10 years after logging have significantly more fine sediment in the top substrate layers than comparable control streams. A few taxa have higher density in stations with more fine sediment and as a result the diversity index is lower in those stations.

LITERATURE CITED

Barton, B. A. 1977. Short-term effects of highway construction on the limnology of a small stream in southern Ontario. Freshwat. Biol. 7:99-108.

Bennett, C. A., and N. L. Franklin. 1954. Statistical analysis in chemistry and the chemical industry. Wiley and Sons, New York.

Beschta, R. L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. Wat. Resour. Res. 14:1011-1016.

Beschta, R. L. 1979. Debris removal and its effects on sedimentation in an Oregon Coast Range stream. Northwest. Sci. 53(1):71-77.

Beschta, R. L., and W. L. Jackson. 1979. The intrusion of fine sediments into a stable gravel bed. J. Fish. Res. Board Can. 36:204-210.

Brown, G. W., and J. T. Krygier. 1971. Clear-cut logging and sediment production in the Oregon Coast Range. Wat. Resour. Res. 7:1189-1198.

Brown, T. E., .and F. L. Richardson. 1968. The effect of growth environment on the physiology of algae: light intensity. J. Phycol. 4:38-54.

Bums, J. W. 1970. Spawning bed sedimentation studies in northern California streams. Cal. Fish Game 56(4):253-270.

Burns, J. W. 1972. Some effects of logging and associated road construction on northern California streams. Trans. Amer. Fish. Soc. 101(1):1-17.

California Fish and Game. No date. Illustrations of logging operations and practices which protect North Coast salmon and spawning streams -- and those that don't; Fish News for timber operators and fishermen. Bull. No. 4.

Calhoun, A., and C. Seeley. 1963. Logging damage to California streams — 1962. Calif. Dept. Fish and Game, Inland Fish. Admin. Kept. 63-2. 15 pp.

Cassie, R. M. 1971. Sampling and statistics. Pp. 174-209, In Edmondson, W. T., and Winberg, G. G. (eds.). Secondary Productiviyy in Freshwaters. IBP, Handbook 17, Blackwell Scientific Publ., London.

Chapman, D. W. 1962. Effects of logging upon fish resources of the west coast. J. Forest. 60:533-537.

Chiang, C. L., and S. Selvin. 1978. Biology and public health statistics. Manual for Bio-Environmental Statistics. Univ. Calif., Berkeley.

Cordone, A. J. 1956. Effects of logging on fish production. Calif. Dept. Fish and Game, Inland Fish Admin. Rept. 56-7. 98 pp.

Cordone, A. J., and D. W. Kelley. 1961. The influences of inorganic sediment on the aquatic life of streams. Cal. Fish Game 47:189-228.

Dunne, T., and L. B. Leopold. 1978. Water in Environmental Planning. W. H. Freeman and Co., San Francisco. 818 pp.

Edmondson, W. T. (ed.). 1959. Freshwater Biology. Wiley and Sons, New York. 1248 pp.

Einstein, H. A. 1968. Deposition of suspended particles in a gravel bed. J. Hydraulics Div., ASCE 5:1197-1205.

Elliott, J. M. 1971. Some methods for the statistical analysis of samples of benthic invertebrates. ' Freshwat. Biol. Assoc. Scient. Pub. No. 25.

Erman, D. C., J. D. Newbold, and K. B. Roby. 1977. Evaluation of streamside bufferstrips for protecting aquatic organisms. Contribution No. 165. California Water Resources Center, Davis, California. 48 pp.

Federal Water Pollution Control Administration. 1970. Industrial waste guide on logging practices. U.S. Dept. Int., FWPCA, Portland, Oregon.

Gammon, J. R. 1970. The effect of inorganic sediment on stream biota. Environmental Protection Agency. Water Quality Office, Washington, D.C. 18050 DWC 12/70. 141 pp.

Graynoth, E. 1979. Effects of logging on stream environments and faunas in Nelson. New Zealand J. Mar. Fresh. Res. 13:79-109.

Gurtz, M. E. 1981. Ecology of stream invertebrates in a forested and a commercially clear-cut watershed. Ph.D. Thesis. University of Georgia. 146 pp.

Gurtz, M. E., J. R. Webster, and J. B. Wallace. 1980. Seston dynamics in southern Appalachian streams: effects of clear-cutting. Can. J. Fish. Aquat. Sci. 37:624-631.

Haefner, J. D., and J. B. Wallace. 1981. Shifts in aquatic insect populations in a first-order southern Appalachian stream following a decade of old field succession. Can. J. Fish. Aquat. Sci. 38:353-359.

Hall, J. D., and R. L. Lantz. 1969. Effects of logging on the habitat of coho salmon and cutthroat trout in coastal streams. Pp. 355-375. ln_ Northcote, T. G. (ed.), Symposium on salmon and trout in streams. Univ. British Columbia, Vancouver. 388 pp.

Hall, J. D., M. L. Murphy, and R. S. Aho. 1978. An improved design for assessing impacts of watershed practices on small streams. Verh. Inter. Verein Limnol. 20:1359-1365.

Hansmann, E. W., and H. K. Phinney. 1973. Logging and periphyton communities. Ecology 54:194-199.

Hawkins, C. P., and J. R. Sedell. 1981. Longitudinal and seasonal changes in functional organization of macroinvertebrate communities in four Oregon streams. Ecology 62(2): 387-397.

Hughes, B. D. 1978. The influence of factors other than pollution on the value of Shannon's diversity index for benthic macroinvertebrates in streams. Water Res. 12:539-364.

Hynes, H. B. N. 1961. The invertebrate fauna of a Welsh mountain stream. Arch. Hydrobiol. 57:344-388.

Innman, D. L. 1952. Measures for describing the size distribution of sediments. J. Sed. Petrol. 22:125-145.

John, P. H. 1978. Discharge measurements in lower order streams. Int. Reveu ges. Hydrobiol. 63(6):731-755.

Karr, J. R., and I. J. Schlosser. 1978. Water resources and the land-water interface. Science 201:229-234.

Kehde, P. E., and J. L. Wilhm. 1972. The effects of grazing by snails on community structure of periphyton in laboratory streams. Amer. Mid. Nat. 87(1):8-24.

Kureyshevich, A. V. 1980. Chlorophyll a content in blue-green algal cultures differing in cell volume. Hydrobiol. J. 15(5):67-74.

Kutkuhn, J. H. 1961. Notes on the precision of numerical and volumetric plankton estimates from small-sample concentrates. Limnol. Oceanogr. 3:69-81.

Lantz, R. L. 1970. Immediate effects of logging on the freshwater environment of salmonids. Annual Prog. Rept., Anadromous Fish Proj., Oregon State Game Comm., Portland. 20 pp.

Lantz, R. L. 1971. Guidelines for stream protection in logging operations. Rep. of Res. Div., Oregon State Game Comm., Portland. 29 pp.

Lenat, D. R., D. L. Penrose, and K. W. Eagleson. 1981. Variable effects of sediment addition on stream benthos. Hydrobiologia 79:187-194.

Le Roux, E. J., and C. Reimer. 1959. Variation between samples of immature stages, and of mortalities from some factors, of the eye-spotted bud moth *Spilonota ocellana* (D&S) (Lepidoptera: Olethreutidae), and the pistol casebearer. *Colcophora serratella* (1.) (Lepidoptera: Coleophoridae) on apple in Quebec. Can. Ent. 91:428-499.

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

Lund, J. W. G., C. Kipling, and E. D. LeCren. 1958. The inverted microscope method of investigating algal numbers and the statistical basis of estimations by counting. Hydrobiologia 10:143-171.

Lyford, J. H., Jr., and S. V. Gregory. 1975. The dynamics and structure of periphyton communities in three Cascade Mountain streams. Verh. Inter. Verein. Limnol. 19: 1610-1616.

Mclntire, C. D., and H. K. Phinney. 1965. Laboratory studies of periphyton production and community metabolism in lotic environments. Ecol. Monogr. 35:237-258.

McNeil, W. J., and W. H. Ahnell. 1964. Success of pink salmon spawning relative to size of spawning bed materials. U.S. Fish and Wildlife Ser., Spec. Sci Rep. Fish. No. 469. 15 pp.

Meyers, J., J. Graham, and R. T. Wang. 1978. On spectral control of pigmentation in *Anacystis nidulans* (Cyanophyceae). J. Phycol. 14:513-518.

Moring, J. R. 1982. Decrease in stream gravel permeability after clear-cut logging: an indication of intragravel conditions for developing salmonid eggs and alevins. Hydrobiologia 88:295-298.

Moring, J. R., and R. L. Lantz. 1974. Immediate effects of logging on the freshwater environment of salmonids. Final Rep. AFS 58. Oregon Wildlife Comm., Portland.

Moring, J. R., and R. L. Lantz. 1975. The Alsea watershed study: effects of logging on the aquatic resources of three headwater streams of the Alsea River, Oregon. Part I - Biological Studies. Fishery Res. Rep. No. 9, Oregon Dept. Fish and Wildlife, Corvallis.

Mundie, J. H. 1971. Sampling benthos and substrate materials, down to 50 microns in size, in shallow streams. J. Fish. Res. Board Can. 28:840-860.

Murphy, M. L., C. P. Hawkins, and N. H. Anderson. 1981. Effects of canopy modification and accumula.ted sediment on stream communities. Trans. Amer. Fish. Soc. 110:469-478.

Murphy, M. L., and J. D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. Can. J. Fish. Aquat. Sci. 38:137-145.

Newbold, J. D. 1977. The use of macroinvertebrates as indicators of logging impact on streams with an evaluation of bufferstrip effectiveness. Ph.D. Disser., Univ. Calif., Berkeley. 103 pp.

Newbold, J. D., D. C. Erman, and K. B. Roby. 1980. Effects of logging on macroinvertebrates in streams with and without bufferstrips. Can. J. Fish. Aquat. Sci. 37:1076-1085.

Paustian, S. J., and R. L. Beschta. 1979. The suspended sediment regime of an Oregon coast range stream. Wat. Res. Bull. 15:144-154.

Pfankuch, D. 1975. Stream reach inventory and channel stability evaluation. USDA Forest Service, Northern Region. 26 pp.

Pielou, E. C. 1975. Ecological Diversity. Wiley, New York. 165 pp.

Roby, K. B., D. C. Erman, and J. D. Newbold. 1977. Biological assessment of timber management activity impacts and bufferstrip effectiveness on National Forest streams of northern California. Earth Resources Mono. i?l. USDA Forest Service, Reg. 5, San Francisco, Calif.

Rosenberg, D. M., and A. P. Weins. 1978. Effects of sediment addition on macroinvertebrates in a northern Canadian river. Water Res. 12:753-763.

Rounick, J. S., and S. V. Gregory. 1981. Temporal changes in periphyton standing crop during an unusually dry winter in streams of the western Cascades, Oregon. Hydrobiologia 83:197-205.

Sadler, R. R. 1970. Buffer strips — a possible application of decision theory. Bur. Land Manage. Tech. Note. Eugene, Oregon. 11 pp.

Scrivener, J. C., and M. J. Brownlee. 1980. A preliminary analysis of Carnation creek gravel quality data 1973-1980. In: Proceedings from the Conference "Salmon-spawning Gravel: A Renewable Resource in the Pacific Northwest?" Seattle, Washington. Oct. 6-7, 1980.

Shoaf, W. T., and B. W. Lium. 1976. Improved extraction of chlorophyll a and b from algae using dimethyl sulfoxide. Limnol. Oceanogr. 21.: 926-928.

Tebo, L. B., Jr. 1955. Effects of siltation, resulting from improper logging, in the bottom fauna of a small trout stream in the southern Appalachians. Prog. Fish Cult. 17:64-70.

Todd, D. (ed.). The Water Encyclopedia: a compendium of useful information on water resources. Port Washington, N.Y. Water Information Center. 559 pp.

Trimble, S. W. 1981. Changes in sediment storage in Coon Creek Basin, Driftless Area, Wisconsin, 1853 to 1975. Science 214:181-183.

Triska, F. J., F. C. Kennedy, R. J. Avanzino, and B. N. Reilly. 1983. Effect of simulated canopy cover on regulation of nitrate uptake and primary production by natural periphyton assemblages, p. 129-159. In Fontalne, T. D., Ill, and Bartell, S. M. (eds.). Dynamics of lotic ecosystems. Ann Arbor Science, Michigan. 494 pp.

Usinger, R. L. (ed.). 1956. Aquatic insects of California. Univ. Calif. Press, Berkeley. 508 pp.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Gushing. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37:130-137.

Vogel, R. J. 1980. The ecological factors that produce perturbation-dependent ecosystems, p. 63-94. In: Cairns, J., Jr. (ed.). The recovery process in damaged ecosystems. Ann Arbor Science, Michigan. 167 pp.

Vollenweider, J. A. 1974. A manual on methods for measuring primary production in aquatic environments. Blackwell Scientific Publications, London.

Walkotten, W. J. 1976. An improved technique for sampling streambed gravel. U.S. For. Serv. Res. Note PNW 281, 11 pp.

Waters, T. F. 1969. Subsampler for dividing samples of stream invertebrate drift. Limnol. Oceanogr. 14:813-815.

Wiggins, G. B. 1977. Larvae of the North American Caddisfly Larvae (Tricoptera). University of Toronto Press, Toronto. 401 pp.

Williams, D. D. 1980. Some relationships between stream benthos and substrate heterogeneity. Limnol. Oceanogr. 25(1):166-172.

Wilhm, J. L. 1970. Effect of sample size on Shannon's formula. Southwest. Nat. 14:441-445.

Woods, R. D. 1975. Hydrobotanical Methods. University Park Press, Baltimore. 173 pp.

APPENDIX A
TESTING MACROINVERTEBRATE SAMPLING

Subsampling

The cost of sample processing is one of the major limitations to extensive surveys of invertebrate populations. Because aquatic organisms usually have clumped distributions, precise estimates of the mean require many samples. To reduce the number of samples retained for processing, several methods of subsampling have been employed (Lund et al. 1958, Hynes 1961, Kutkuhn 1961, Waters 1969, Mundie 1971, Williams 1980). To examine the effectiveness of our field subsampling we used three sets of clusters (two from Two Bit Creek, one from Four Bit Creek). Each cluster was made up of four unmixed Surber samples. The samples were then sorted and identified as usual except all debris was retained. The four samples (debris and organisms) were then combined, homogenized and divided as in the field situation. All four subsamples were resorted and identified.

Results of the three clusters before and after mixing the samples are given in Table Al. After mixing, each subsample gave a close estimate of the mean, as predicted by theory. Coefficients of variation (CV) for density ranged from 8.3% - 117, after mixing compared to 40 - 56% before mixing. For taxa represented by more than 40 individuals per cluster, the mean coefficient of variation was 21% after vs. 71% before mixing. Subsampling had little effect on the number of taxa.

Slight differences in number of individuals (less than 3% between mixed and unmixed) before and after mixing were a result of losses in reprocessing. Taxa with numerous small individuals (primarily Chironomidae) were sometimes difficult to completely disperse because they stuck to debris and filamentous algae. This factor accounts in part for the higher CV for taxa with many individuals than for samples as.a whole.

The results of the subsampling procedure on Shannon diversity is given in Table A2. Again, the amount of variation was much lower after mixing (0.9 - 3%) than before mixing (3 - 13%). Thus, a single subsample from a cluster is also a representative value of the mean diversity of the cluster.

Diversity Index and Sample Size

Ordinarily, adding successive samples increases a diversity index until an asymptote is reached that is characteristic of the habitat (Pielou 1975). In the unmixed four samples from the test locations (Table A2) diversity increased sharply as each additional sample was pooled (Fig. Al). After mixing, little change in diversity (0.1 units per sample) occurred with each additional subsample.

The effect of sample sizes (and subsampling) used in our field situations on the diversity index is shown for the first six streams sampled (Fig. A2). In addition Indian Creek diversity values of 16 successively pooled samples without clustering and subsampling are plotted for comparison (Fig. A2). In all cases the asymptote appears reached by the time eight samples (two clusters) were pooled. Wilhm (1970) reported that invertebrate diversity reached an asymptote before 10 samples were pooled in each of 13 different stream habitats. Similarly, Hughes (1978) found that the asymptote was reached after four O.lm^2 samples were pooled and that no further increase was noted, even after pooling 50 samples.

TABLE Al

DATA FOR LABORATORY SUBSAMPLE TRIALS

(Numbers per 0.1 m^2 sample)

Subsample	Cluster 1 [a]				Cluster 2 [a]				Cluster 3 [b]			
	Before mixing		After mixing		Before mixing		After mixing		Before mixing		After mixing	
	Density	Taxa	Density	Taxa	Density	Taxa	Density	Taxa	Density	Taxa	Density	Taxa
1	569	19	1440	21	1615	23	1402	21	1959	26	1630	21
2	2327	21	1190	24	521	18	1661	26	1166	22	1691	25
3	1859	22	1306	24	1503	25	1315	25	2545	24	1829	26
4	619	20	1438	22	2541	24	1616	24	1119	24	1496	22
Total	5374	28	5374	27	6176	27	5999	27	6789	30	6646	29
mean	1344	20.5	1344	22.8	1545	22.5	1500[e]	24	1697	24	1662[e]	23.5
C.V. [c]	56	6.3	9.3	6.6	54	13.8	11	9.0	40	6.8	8.3	10.1
Overdispersion (C') [d]	0.44		0.02		0.29		0.01		0.16		0.01	

[a] Two Bit Creek, logged.

[b] Four Bit Creek, logged.

[c] coefficient of variation (standard deviation as a percentage of the mean).

[d] Index of overdispersion (C') = $(s^2 - x)/x^2$ (Cassie 1971).

[e] Reduced numbers after mixing due to loss during processing.

TABLE A2

DATA FOR LABORATORY SUB SAMPLE TRIALS

Diversity (H')

Subsample	Cluster 1[a]		Cluster 2 [a]		Cluster 3 [b]	
	Before mix	After mix	Before mix	After mix	Before mix	After mix
1	1.58	1.89	2.13	2.17	2.09	2.10
2	1.83	1.86	1.85	2.14	1.96	2.05
3	1.81	1.94	2.05	2.12	2.02	2.17
4	1.94	1.81	2.08	2.14	2.05	2.07
mean	1.79	1.87	2.03	2.14	2.03	2.10
C.V. [c] (%)	13	3	6	0.9	3	2.4
pool total [d]	1.86	1.89	2.10	2.15	2.07	2.11

[a] Two Bit Creek

[b] Four Bit Creek

[c] Coefficient of Variation

[d] "Before" and "After" values are different due to losses during processing.

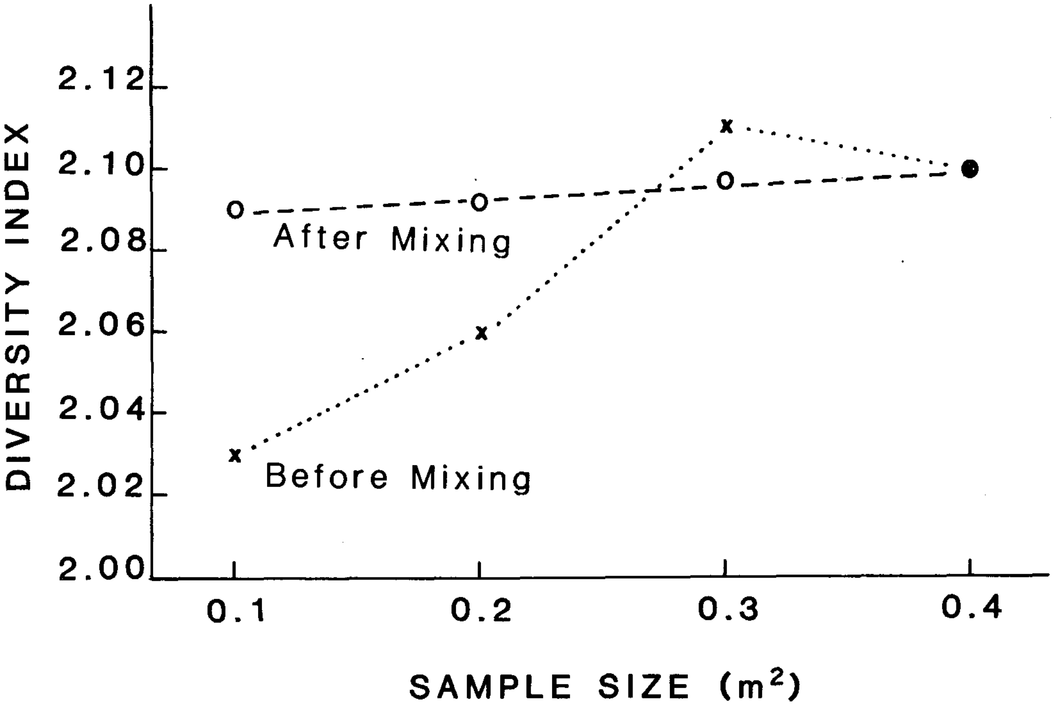

Figure Al. Mean diversity index for successively pooled samples before and after cluster mixing. Values for 0.2m and 0.3m are means of all possible combinations.

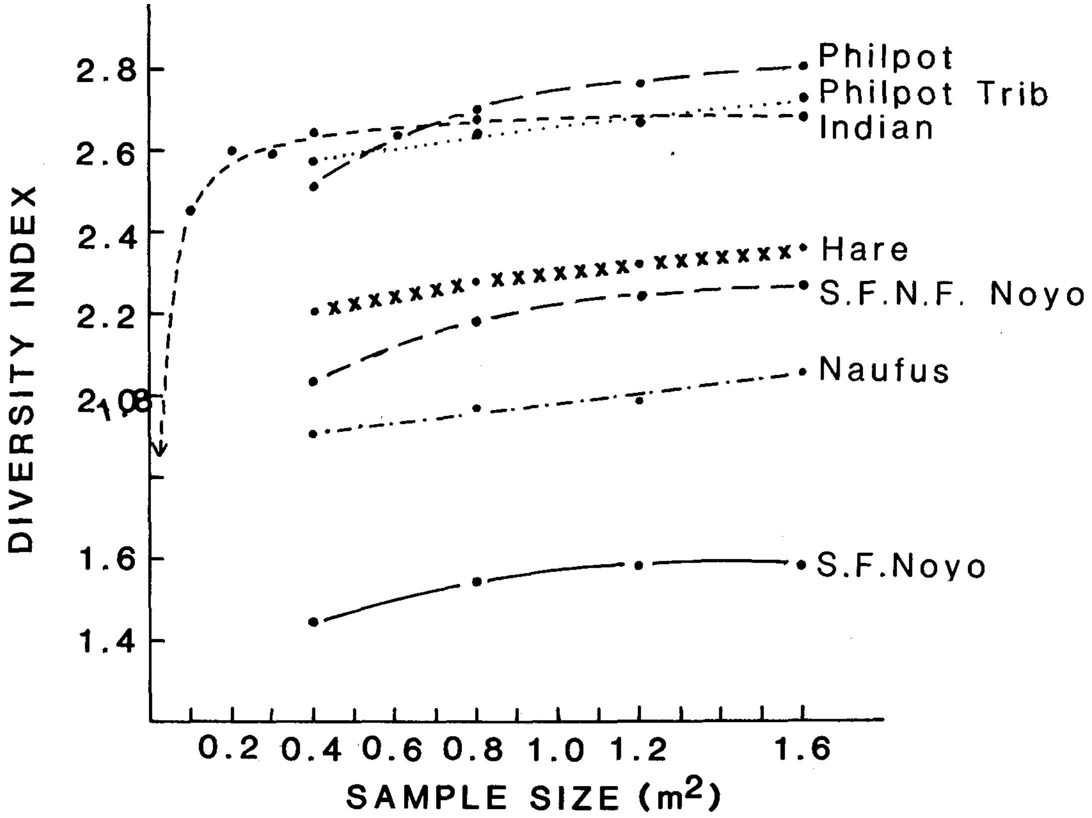

Figure A2. Mean diversity index for successively pooled invertebrate samples for seven streams. Each point represents the mean of all possible sequences except Indian Creek which is a single random order sequence.

APPENDIX B TESTING THE METHOD OF TRANSPORTABLE SEDIMENT ESTIMATION

The quantity of sediments stored in stream bottoms has been measured in a variety of ways (McNeil and Ahnell 1964, Mundie 1971, Walkotten 1976). Most of these methods are expensive in time or money. To adequately quantify stored sediment numerous samples are necessary. Bums (1970), for example, took 20 core samples per station. Placing of cores may dislodge small size classes (Beschta and Jackson 1979), or large rocks may make placement of coring devices impossible (Graynoth 1979). An alternative approach for measuring fine sediment was used here. Fine sediments were disturbed in a known area of stream bottom and the resultant sediment drift was deposited in downstream traps. A similar procedure was used by Barton (1977) to collect sediment below an ongoing stream disturbance.

To test the accuracy of this method of estimating transportable sediment, we conducted experimental trials in Strawberry Creek on the University of California, Berkeley campus. Known amounts of predetermined size classes (< 0.3 mm, 0.3 mm to 0.125 mm, < 0.21 mm, < 0.125 mm) were enclosed in a circular tray 31 cm in diameter. Larger rocks were added to the tray to mimic a natural substrate. This tray was then embedded into a sample reach, and when opened served as the sample site (Fig. Bl). All other steps were identical to field situations. After the release of sediment the tray was sealed and unreleased sediments were measured in the laboratory. A typical curve of sediment deposition per row is shown in Fig. B2. The theoretical effect of different stream currents on deposition is Illustrated in Fig. B3.

Results from six trials are given in Table Bl. The mean error of 14 estimates from actual released sediment totals was 18%; 12 were overestimated. The mean error for the size class below 0.125 mm was 23%. The largest known source of error in the testing procedure was release of sediments from beneath and around the tray during the disturbance of the sediments in the tray. Up to 10% of the sediment caught in traps was in size classes that were not added to the tray. Stream sources probably contributed also to the size classes which were in the tray because the bed of Strawberry Creek contained substantial silts and clays of different color from the test sediments. The range of deviations of the estimates from the actual totals might well be less if this extra source of sediment were taken into account. We cannot confirm this greater precision, however, because the extraneous sediment could not be separated from the test material.

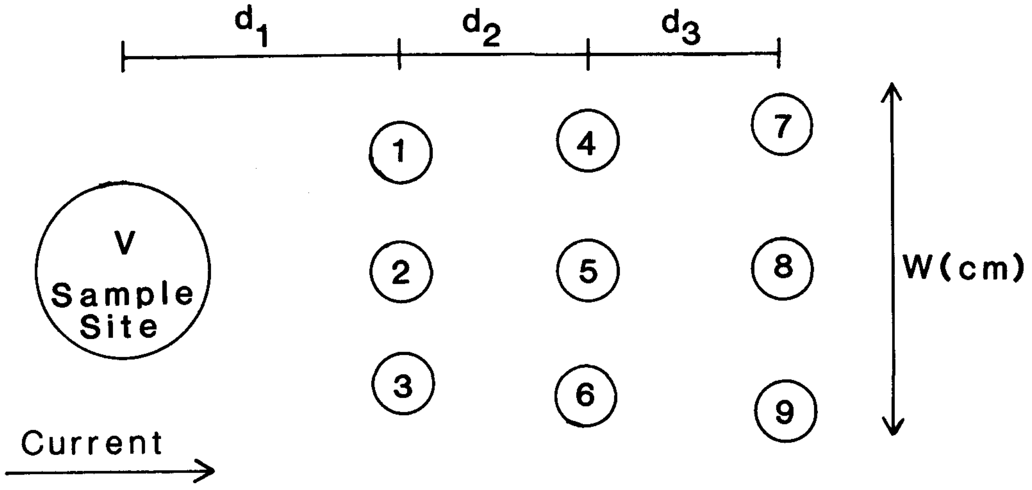

Figure Bl. Diagram of sampling array for transportable sediment, and detritus.

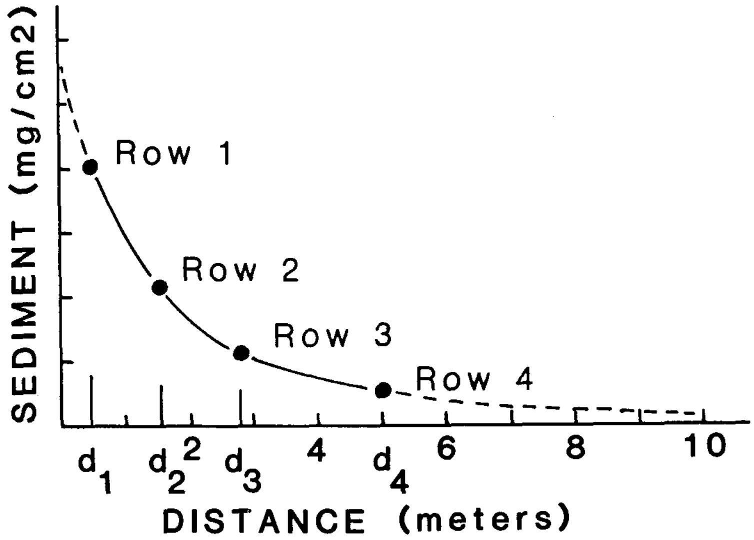

Figure B2. Hypothetical change in sediment deposition over distance downstream from sample point.

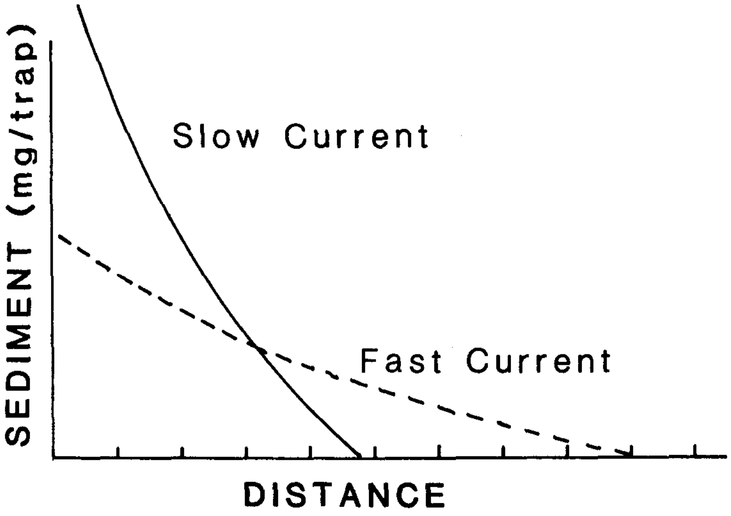

Figure B3. Relative effect of stream current on sediment deposition versus distance. Area under each curve is theoretically equal.

TABLE Bl
COMPARISON OF ESTIMATED AND ACTUAL

SEDIMENT RELEASES

Test	Size Class (mm)	Mean Depth (cm)	Mean Current (cm/sec)	Sediment			Error (%)
				Amount Available (g)	Actual Loss (g)	Estimated[1] Loss (g)	
1	< .125	7.0	38.5	300.6	226.8	275.0	+21.3
2	< .125	7.7	38.5	234.7	213.3	222.0	+ 4.1
3a	< .125	14.3	39.5	67.2	52.5	74.4	+41.7
b	.300-.125	14.3	39.5	239.5	162.1	180.7	+11.5
c	< .300	14.3	39.5	306.7	214.6	216.7	+ 1.0
4a	< .125	6.5	42.0	54.5	43.2	52.0	+20.3
b	.300-.125	6.5	42.0	266.3	151.1	191.6	+26.8
c	< .300	6.5	42.0	320.8	194.6	236.4	+21.5
5a	< .125	7.9	49.0	145.1	138.0	124.3	+10.0
b	.210-.125	7.9	49.0	182.3	153.1	178.3	+16.5
c	< .210	7.9	49.0	327.4	290.5	260.0	-10.2
6a	< .125	10.4	44.4	74.9	70.8	99.6	+40.7
b	.210-.125	10.4	44.4	106.9	89.5	97.8	+ 9.3
c	< .125	10.4	44.4	181.7	160.3	188.6	+17.7

[1] Amount collected in traps includes amount lost from tray and some disturbed from stream bottom in the test size classes.

The method has advantages and limitations when compared with other techniques. One advantage is its ability to quantify fines in stream substrates that are too rocky for the placement of coring devices. Another advantage of this method is its size. Once the technique is mastered, one person can sample several streams in a day in conjunction with other sampling programs. The amount of equipment and the amount of material saved are small enough that remote locations can be sampled. This method allows practical measurement of fine sediments in studies which cover a larger number of streams where an exhaustive analysis of the sediments would consume a major proportion of the study resources. The method samples a large area (greater than 1000 cm2) compared to a coring device and may give a more reliable estimate when the sample area is a significant proportion of the riffle area, as is the case in many small first and second order streams.

A limitation of the method is the practical difficulty of disturbing a section of stream bottom vigorously enough to wash the site clean of buried fine sediment. In undisturbed streams one can accomplish complete cleaning but in streams with heavy deposits it may be impossible. The alternative, disturbing a site for a standard time, will lead to underestimates of sediment densities when compared to other methods, but these estimates are still useful for comparisons among streams. The final numerical value is calculated using a visual estimate of the width of the sediment plume, and thus the result is limited by visual acuity of the observer. Other limitations are the size classes which can be estimated. In most streams (except for those with swift currents) sediments larger than 0.3 mm will be deposited within 0.5 m of the sample point. Obtaining exact measurements of the area disturbed is another limitation. The area has a general color change to aid the observer, but estimating the depth disturbed again depends on observer judgment. These limitations are less pronounced in smaller streams, and so the method may be best used there.

APPENDIX C

ENVIRONMENTAL CHARACTERISTICS OF LOGGED (L), NARROW-BUFFERED (N), RECENTLY DISTURBED (RD), AND CONTROL (C) STREAMS IN NORTHERN CALIFORNIA

Stream	Status	Sample Day (1980)	Elevation^1 (m)	Drainage area^1 (m)	Relief ratio^1	Geological^{1,3} Type	Alkalinity (ppm)	pH	Ca + Mg Hardness (ppm)	Canopy density (%)	Total radiation (Ein/m^2/day)	Bankfull width (m)	Bankfull index (ft/mi^2)
N.Fk.S.Fk. Noyo	C	174	61	1885	.05	kr	1	7.4	100	50	5.5	9.4	13.5
S.Fk. Noyo	RD	175	49	2675	.06	kr	36	8.0	79	70	2.8	4.1	5.1
Hare	N	176	55	1933	.05	kr	19	-	110	30	16.0	8.2	11.5
Philpot	C	177	807	1994	.08	m	131	8.5	170	55	65.0	8.2	11.3
Philpot Trib	C	178	807	362	.13	m,ub	95	8.5	230	70	35.0	3.4	9.4
Naufus	N	178	1158	183	.14	m	38	7.4	90	5	49.0	5.3	19.6
N. Packsaddle	C	197	707	65	.45	ub,gr	13	8.0	32	85	4.8	3.0	17.0
U. Packsaddle	C	198	707	554	.36	ub,gr	-	-	-	80	6.2	6.8	16.5
L. Packsaddle	N	199	576	660	.30	ub,gr	25	8.0	43	30	30.1	9.0	20.1
N. Fall	C	199	512	307	.34	gr	21	7.8	72	100	2.0	7.8	23.5
S. Fall	C	200	512	246	.34	gr	22	7.8	60	100	1.6	3.1	13.5
Knopki	N	200	899	80	.39	ub,ju	14	8.0	60	100	1.8	4.6	25.0
Whites	C	233	942	570	.21	m	18	8.2	82	75	12.4	3.9	9.3
Murphy	C	232	746	880	.22	m	50	8.3	102	80	7.0	2.2	4.3
Little N. Fork	N	231	963	4467	.12	m,ms,mv	17	8.0	110	0	60.0	11.9	12.3
Garden	N	232	683	388	.36	m	23	8.0	115	50	10.1	2.8	9.0
Copper B	C	179	1024	108	.35	ub,bi	47	8.5	135	65	64.9	3.6	16.4
Copper A	L	179	950	59	.32	ub,bi	42	8.2	80	8	4.2	2.9	16.6
U. Four Bit	C	218	1219	65	.20	m	20	8.0	78	40	3.8	3.3	19.3
Indian	RD	219	1182	119	.27	m	43	8.2	82	15	58.0	3.4	14.5
L. Four Bit	L	217	1146	110	.19	m	21	8.0	72	30	37.6	2.4	11.3
Two Bit	L	217	1182	124	.24	m	29	8.0	80	55	36.9	2.1	9.2
E. Br. Lights	C	241	1701	837	.12	gr	15	7.8	82	50	37.5	3.2	6.6
U. Taylor	C	241	1926	248	.60	gr	5	6.8	160	50	26.7	2.0	6.6
L. Taylor	L	242	1778	565	.43	gr	-	-	-	0	50.0	6.7	11.6
U. New York	C	182^2	1060	360	.24	lp	12	7.6	64	25	9.8	3.6	10.6
Empire	C	183^2	1331	736	.11	-	16	8.0	100	75	12.0	5.7	11.6
Mid New York	L	182^2	1030	472	.24	lp	12	7.7	76	25	24.0	2.9	7.5
New York Trib	L	182^2	1085	96	.37	lp	13	7.8	64	45	22.3	2.4	11.7

---

[1] Data from Erman et al. (1977).

[2] Sample year is 1981.

[3] kr = Undivided Cretaceous marine
m = Pre-Cenozoic metamorphic
ub = Mesozoic ultrabasic intrusives
gr = Mesozoic granitics
ju = Upper Jurassic marine
ms = Pre-Cretaceous meta sedimentaries
mv = Pre-Cretaceous meta volcanic
bi = Mesozoic basic intrusives
lp = Paleozoic marine

Stream	Discharge (m^3/sec)	Avg. current velocity (cm/sec)	Avg. annual precipitation (in.)	Temp. (C^o)	Pfankuch	Sediment Size		Longitude[1]	Latitude[1]
						d_g	σ_g	(decimal	degrees)
N.Fk.S.Fk. Noyo	.115	40	50	13.0	93	3.0	-	123.67	39.40
S.Fk. Noyo	.036	28	50	13.5	64	1.0	34.4	123.67	39.38
Hare	.073	49	50	13.0	108	3.7	8.3	123.73	39.39
Philpot	.080	36	40	14.5	59	21.1	7.0	123.18	40.47
Philpot Trib	.019	32	40	12.5	104	9.8	6.6	123.18	40.47
Naufus	.004	19	44	14.0	73	9.2	7.0	123.33	40.46
N. Packsaddle	.018	26	85	13.0	62	10.5	10.5	123.72	42.91
U. Packsaddle	.067	33	85	13.0	56	8.6	9.0	123.72	42.90
L. Packsaddle	.083	31	85	14.0	64	21.7	7.7	123.73	42.91
N. Fall	.019	29	85	14.0	70	7.1	11.8	123.75	42.89
S. Fall	.015	28	85	14.0	63	9.8	19.7	123.72	42.88
Knopki	.005	27	85	13.0	113	3.0	37.0	123.73	42.93
Whites	.054	33	40	14.0	74	8.0	2.3	123.04	41.16
Murphy	.012	28	40	14.0	65	8.6	8.6	123.17	41.23
Little N. Fork	.835	40	40	13.0	90	11.0	10.0	123.18	41.38
Garden	.082	36	40	16.5	104	1.6	32.1	123.88	41.33
Copper B	.012	31	60	14.0	95	10.0	-	122.67	41.07
Copper A	.016	37	60	10.0	97	13.1	4.7	122.67	41.07
U. Four Bit	.004	33	40	10.5	87	4.2	18.5	123.50	41.99
Indian	.035	41	40	9.5	104	1.5	5.8	123.52	41.99
L. Four Bit	.002	33	40	10.5	95	1.5	5.1	123.50	41.99
Two Bit	.010	33	40	10.5	99	0.6	12.6	123.48	41.99
E. Br. Lights	.065	31	24	10.0	54	5.6	6.3	120.72	40.21
U. Taylor	.135	43	24	12.5	73	4.3	15.8	120.72	40.16
L. Taylor	.145	40	24	-	84	13.0	12.4	120.70	40.17
U. New York	.015	20	56	10.6	62	2.6	26.0	120.78	39.57
Empire	.017	41	56	10.0	48	8.0	20.1	120.47	39.38
Mid New York	.035	35	56	10.6	77	6.5	10.9	120.78	39.56
New York Trib	.086	34	56	12.2	78	5.6	9.3	120.78	39.56

Data from Erman et al. (1977).

APPENDIX D

TRANSPORTABLE SEDIMENT AND DETRITUS

Current velocity and depths are means over all sediment traps at a station.

Block: Stream	Status	Sediment (mg/cm^2)	Detritus (mg/cm^2)	Current Velocity (cm/sec)	Depth (cm)
Two Bit:					
U. Four Bit	C	90.6	24.0	5.2	4.2
Indian	RD	217.2	21.2	33.3	-
L. Four Bit	L	175.1	17.7	30.1	7.0
Two Bit	L	491.2	29.7	12.1	-
Taylor:					
E. Br. Lights U.	C	132.7	27.6	14.0	13.0
Taylor	C	168.0	14.7	37.0	26.0
L. Taylor	L	208.1	38.9	25.0	26.0
New York:					
U. New York	C	242.1	120.6	16.9	5.3
Empire	C	64.9	15.4	16.6	12.3
Mid New York	L	130.2	41.2	25.7	8.6
New York Trib	L	883.1	211.9	29.3	4.9
Redwood					
N.Fo.S.Fo. Noyo	C	51.4	16.3	10.0	14.0
S.Fo. Noyo	RD	50.8	29.3	17.4	4.0
Hare	N	58.2	11.1	24.0	9.5
Philpot:					
Philpot	C	60.4	41.0	6.0	9.5
Philpot Trib	C	65.7	8.9	15.0	8.0
Naufus	N, RD	136.7	14.6	16.6	7.0
Packsaddle:					
U. Packsaddle	C	6.4	1.9	10.1	21.0
L. Packsaddle	N	6.6	2.6	19.7	14.0
Fall:					
S. Fall	C	36.2	13.0	9.1	13.0
Knopki	N	16.3	13.6	4.5	9.0
Garden:					
Whites	C	115.5	19.7	4.5	15.0
Murphy	C	179.0	19.0	7.6	8.0
Garden	N	481.6	49.0	36.3	13.0

APPENDIX E

PERIPHYTON PIGMENTS FOR 29 STREAMS IN.NORTHERN CALIFORNIA

Stream	Status	Chlorophyll a		Phaeophytin a			Pigment Matrix			
		X (mg/m^2)	S.D.	A[1]	B[2] (percent)	C[3]	Total[4]	% Chl a[5]	Degraded[6]	Other[7]
N.Fk.S.Fk. Noyo	C	6.33	0.94	22.0	16.5	16.1	14.4	30.0	2.3	7.8
S.Fk. Noyo	RD	8.94	1.86	22.3	24.3	20.5	22.1	27.8	4.5	11.4
Hare	N	3.84	0.81	28.8	14.0	30.0	8.0	33.1	2.4	3.0
Philpot	C	7.69	2.59	17.0	15.5	10.6	16.3	32.0	1.7	9.4
Philpot Trib	C	4.04	1.42	18.8	9.0	16.9	8.0	33.6	1.4	4.0
Naufus	N,RD	3.05	0.67	57.0	29.5	32.2	11.1	18.9	3.6	5.4
N. Packsaddle	C	1.28	0.81	29.4	35.5	23.0	3.9	23.2	0.9	2.1
U. Packsaddle	C	5.42	0.74	10.0	9.3	2.9	10.5	35.4	0.3	6.5
L. Packsaddle	N	2.66	1.13	33.0	16.0	51.1	7.7	25.6	3.9	1.8
N. Fall	C	13.21	4.92	26.0	10.0	25.3	23.2	39.1	6.1	8.3
S. Fall	C	10.05	1.38	12.8	4.8	6.5	16.1	43.1	1.0	8.1
Knopki	N	9.78	1.43	9.5	8.8	3.0	16.1	41.7	0.5	8.9
Whites	C	11.04	1.67	14.8	12.5	8.7	21.0	36.0	1.8	11.4
Murphy	RD	17.25	5.74	20.8	12.0	15.7	30.2	39.1	4.7	13.7
LNF	N	7.49	3.07	17.8	12.8	10.4	16.1	31.8	1.7	9.3
Garden	N	11.83	3.96	15.3	12.3	7.3	20.4	39.8	3.5	10.7
U. Four Bit	C	8.67	4.54	6.3	16.8	1.0	16.4	36.3	0.1	10.3
Indian	RD	6.51	2.00	27.5	21.8	28.3	11.5	32.7	3.3	4.5
L. Four Bit	L	3.06	0.87	21.3	32.8	19.6	9.7	21.8	1.9	5.7
Two Bit	L	2.37	0.91	29.0	35.3	30.2	7.8	20.9	2.4	3.6
Copper B	C	2.46	0.75	40.5	14.5	56.7	6.5	26.0	3.7	1.2
Copper A	L	14.98	5.03	19.5	10.5	13.9	26.4	38.9	4.7	12.5
E. Lights	C	11.64	2.61	15.5	8.3	9.2	22.1	36.0	2.0	12.1
U. Taylor	C	9.33	1.86	10.7	7.0	4.5	15.4	41.6	0.7	8.3
L. Taylor	L	12.91	2.74	17.0	11.0	11.3	23.8	37.2	2.7	12.3
U. New York	C	16.60	3.90	36.5	17.5	47.7	37.6	30.3	17.9	8.3
Empire	C	16.40	4.20	40.5	13.4	60.1	39.1	28.8	23.5	4.4
Mid New York	L	22.19	10.00	22.3	16.5	21.8	43.5	35.0	9.5	19.2
New York Trib	L	18.25	6.60	34.4	10.5	43.5	36.6	34.1	15.9	8.2

[1] Percent by weight calculated from before and after acidification, spectrophotometric readings at 660 nm.

[2] Percent by weight calculated by changes at 410 and 430 nm after acidification.

[3] Total pigment matrix after acidification as phaeophytin a relative to total as chlorophyll *a.*

[4] Total area under spectrograph in units of absorbance times nanometers or moles 10'/cm- corrected for sample size.

[5] Area under spectrograph due to chlorophyll a as percent of total area.

[6] Percent from 3 above multiplied by 4 above.

[7] 4 above minus 6 and minus 5 multiplied by 4.