IN-STREAM DISTRIBUTION AND HYDRAULIC AND ECOLOGICAL EFFECTS OF VOLCANIC ASH FROM MOUNT ST. HELENS ERUPTION

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ABSTRACT

The catastrophic eruption of Mt. St. Helens in southwestern Washington on May 18, 1980 deposited a plume of volcanic ash from Washington to western Montana. Ash deposits in northern Idaho were significant with depths varying from 2 to 3 mm in the fringes to as much as 50 mm in the central area of the plume as it crossed the Idaho panhandle. Varying amounts of the ash were deposited directly in streams located in the plume area. This study is an evaluation of the changes in the in-stream ecological conditions resulting from volcanic ash deposition and transport. This interdisciplinary evaluation of the in-stream effects of the new volcanic ash deposition on the physical and biotic components of streams in the ashfall plume area was studied by recreating, on a simulated basis, conditions prior to, during, and after the ash deposition, and by field verification measurements taken in the Palouse River to support the simulation studies.

The results obtained from the simulation and field studies clearly indicate catastrophic influences of volcanic ash deposition and transport on attached benthic algae biomass and on the dislocation of invertebrates and interruption of food-chain relationships. While the short-term impact was moderate to severe on selected populations, all of the principal species tested demonstrated considerable persistence and resilience such that long-term effects of the ash on the in-stream ecosystem are judged to be minimal.

Chapter 1. Introduction

The May 18, 1980 catastrophic eruption of a southwestern Washington volcano, deposited a plume of volcanic ash from Washington to western Montana (Fig. 1). Ash deposits in the northern Idaho plume area varied from 2 or 3 mm at the fringes to as much as 50 mm in the Plummer-St. Maries area. Considerable variations in deposition were observed and were likely the result of wind pattern and topography.

The importance of detritus to aquatic trophic structures has been widely documented (Teal, 1957; Nelson and Scott, 1962; Minshall, 1967; Fisher and Likens, 1972; Berrie, 1975; Anderson and Sedell, 1979). Detritus is especially important to the community structure of lower order streams since small headwater streams typically derive a greater percentage of their energy from allochthonous sources than do higher order streams (Anderson and Sedell, 1979; Naiman and Sedell, 1979). A stream like Mosquito Creek, Idaho, that is covered by a heavy canopy may derive from 66-99% of its energy from allochthonous organic material (Teal, 1957; Nelson and Scott, 1962; Fisher and Likens, 1972). Leaf material is usually considered to be the major source of allochthonous input (Anderson and Sedell, 1979), so leaf pack processing has received a great deal of attention (Nykvist, 1961; Mathews and Kowalczewski, 1969; Kaushik and Hynes, 1971; Petersen and Cummins, 1974; Reice, 1974; Suberkropp, et al., 1976; Triska and Sedell, 1976; Benfield, et al., 1977; and Herbst, 1982). Tree species (Nykvist, 1961; Petersen and Cummins, 1974; and Herbst, 1982), leaf age (Edwards and Heath, 1975), nitrate manipulation (Triska and Sedell, 1976), sediment types (Reice, 1974), and many other factors have been related to

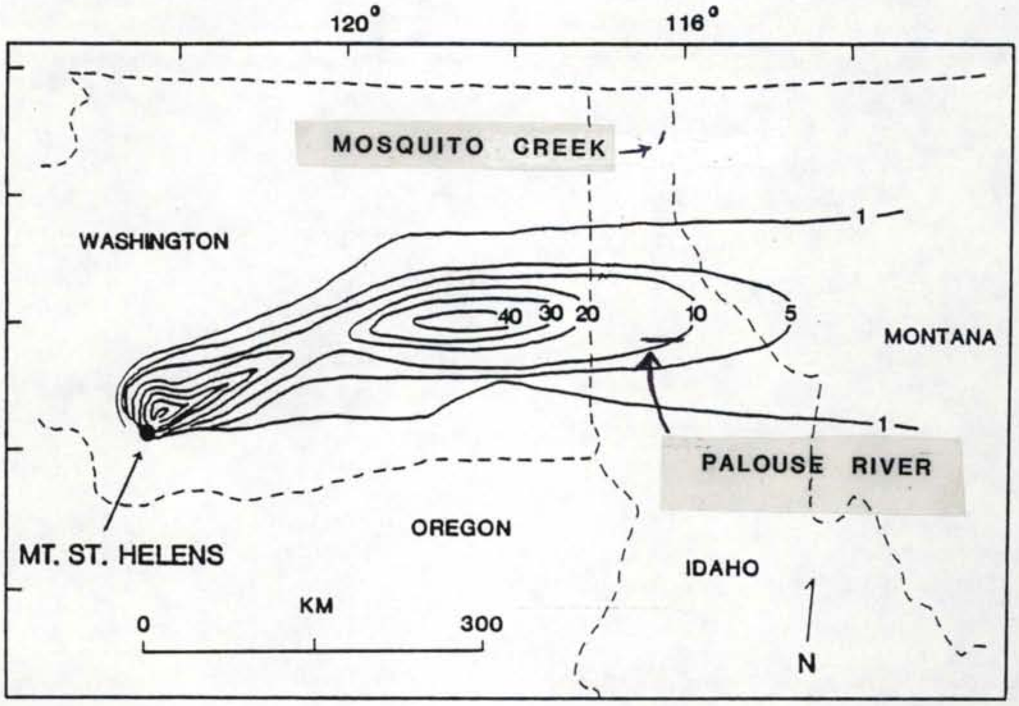


Figure 1. Map of the Pacific Northwest, showing ashfall from the 1980 Mount St. Helens eruption (isopleths=mm). Adapted from Sarna-Wojcicki et al, 1981.

decomposition rate. This section of the study was conducted to determine the effect of a coating of volcanic ash on detrital leaching decomposition as measured by weight loss.

Varying amounts of fine-grained volcanic ash were deposited directly in streams located in the plume area. Early observations in some of these streams indicated a variety of effects relating to in-stream deposition and transport, microhabitat characteristics and stream biota. These early indications inferred the possibility that some in-stream ecological conditons might be altered significantly by the volcanic ash transport and deposition in northern Idaho streams. The lack of empirically derived information on the cause-effect relationships of Mount St. Helens tephra on stream ecosystems fostered this interdisciplinary study to recreate on a simulated basis, conditions prior to, during and after a volcanic eruption.

The intent of this study was to determine the specific cause, extent, and nature of the in-stream effects of the new volcanic ash deposition on the physical and biotic components of streams in the ashfall plume area in a way that the resultant responses could be more predictable, and therefore, more manageable. The study efforts were focused on an interdisciplinary assessment of ash-related problems in a stream system representative of streams in the ashfall plume area. These efforts were augmented by simulations conducted in the laboratory and in artificial channels where ecological conditions could be closely controlled and monitored.

The specific research objectives were: 1) To assess microhabitat characteristics affected by volcanic ash in a representative stream system; 2) To determine the extent and movement of ash through a stream system; and 3) To assess the effects of stream-transported and stream-deposited volcanic ash on primary and secondary production in natural, but controlled-flow, stream channels.

Chapter 2. Study Area Descriptions

2.1 Graves Meadow Site Description

The Graves Meadow site is located at approximately river mile 160 (257 km) on the Palouse River of Northern Idaho, approximately 10 miles (16 km) north and east of the community of Harvard, Idaho. (Figure 2).

2.1.1 Geology: The watershed is predominately underlain by Precambrian sedimentary strata of the Belt Supergroup that has been altered by contact metamorphism due to the intrusion of the Cretaceous Idaho Batholoth (Rember and Bennett, 1979). Diopside gneiss, biotite gneiss and biotite quartzite interbedded with schist of the Wallace Formation underlie 80- 90% of the watershed. There are, also, small outcrops of vitreous quartzite of the Striped Peak Formation with some feldspathic quartzite. The southeast margin of the watershed is underlain by material of the Idaho Batholith. The Graves Meadow site itself is overlain by Quaternary alluvium. During the May 18, 1980 Mt. St. Helens eruption the watershed received approximately an inch (25 mm) of air-fall ash. Due to this geologic setting the streambed sediments of the Palouse River in the Graves Meadow area are similar to those found in the Idaho Batholith areas of Central Idaho.

2.1.2 History: There has been heavy logging in the watershed over the past 80 years. Logging operations are currently harvesting second growth timber in the watershed. Prior to the 1930's, extensive gold dredging occurred in the upper reaches of the Palouse River and the North Fork of the Palouse River one mile (2 km) below the Graves Meadow site. Some minor diversions are being made from the Palouse River to support the logging operations - mostly for dust control along

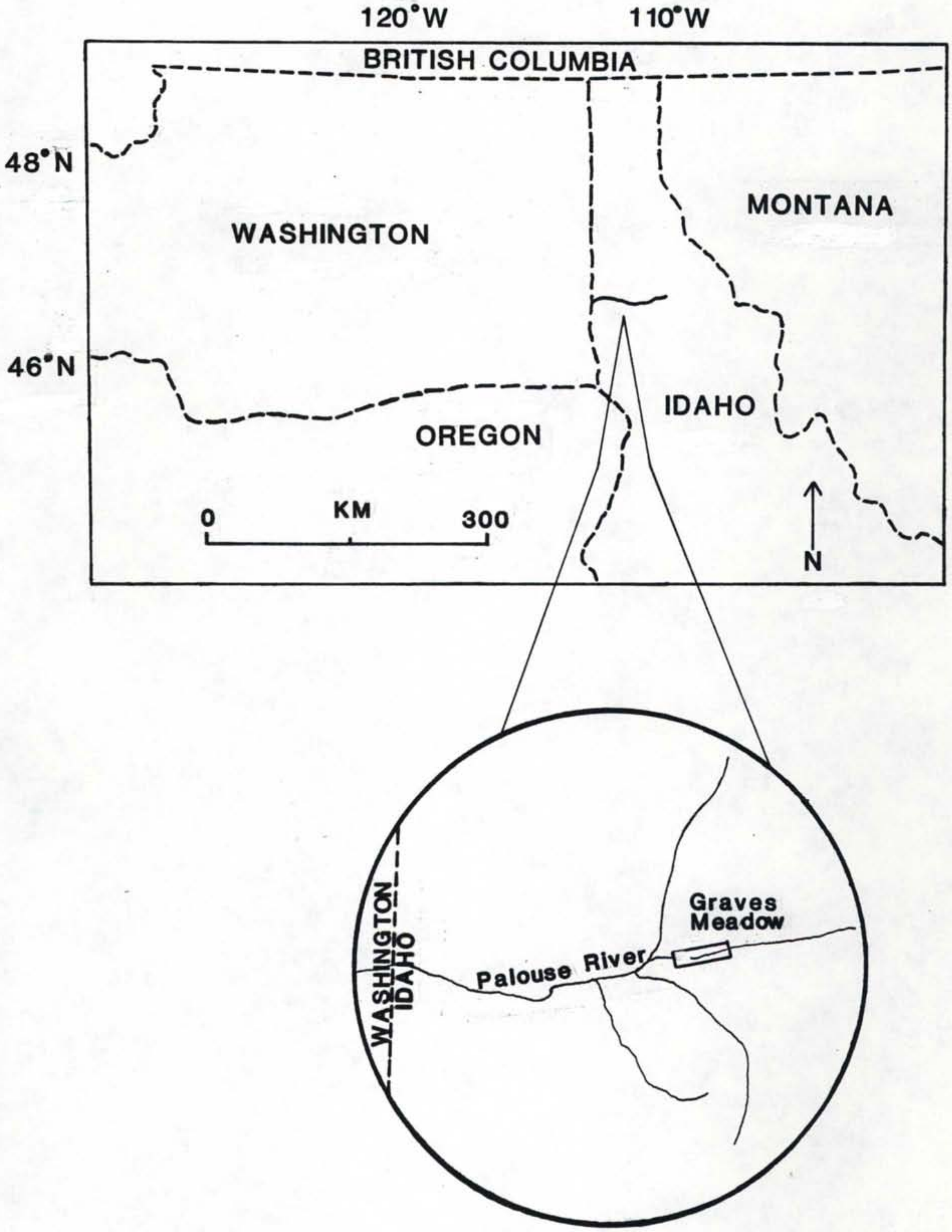


Figure 2. Map of the Palouse River and the Graves Meadow intensive site.

the roads. Cattle grazing is apparent in the meadows along the Palouse River both below and upstream of the site.

2.1.3 Hydrology: The study reach of the Palouse River is located in a third order stream draining a heavily forested watershed of 14.4 sq. mi. (37 km^2). The average slope of the Palouse River upstream of the site is 0.011 ft/ft (m/m). The late August 1981 discharge less than 10 cfs (0.28 cms) after an atypical runoff period in which high runoff was in February rather than the usual April high runoff period (Figures 3 and 4). Figure 4 shows that the May 18 ashfall occurred after the peak runoff during 1981, but while the flows were still high.

2.1.4 Stream Environment: The Graves Meadow reach of the Palouse River supports a large diversity of primary producers (Figure 1). Pool sediments were frequently covered by mats of displaced *Vaucheria* and other algae. Flat round mats of *Vaucheria* were found on the larger rocks in riffles and runs along with *Monostroma*. Globular colonies of *Nostoc* and 10 cm tufts of *Ulothrix* were common on flat stones in riffles. Diatoms were common throughout the reach but particularly in association with *Ulothrix*. Dominant diatoms were *Gomphonema* and *Cymbella*. In addition to algae cover, the mosses *Fontinalis* and *Scorpidium* and the liverwort *Ricciocarpus* were found in riffles and runs or in the splash zone. *Callitriche* (a vascular macrophyte) was also occasionally found in pool shallows. Pool substrates varied from fine particulate organic matter to unimbedded small cobbles. Riffle substrates were predominantly unimbedded small cobbles with some large cobbles. Run substrates were intermediate-sized between riffles and pools.

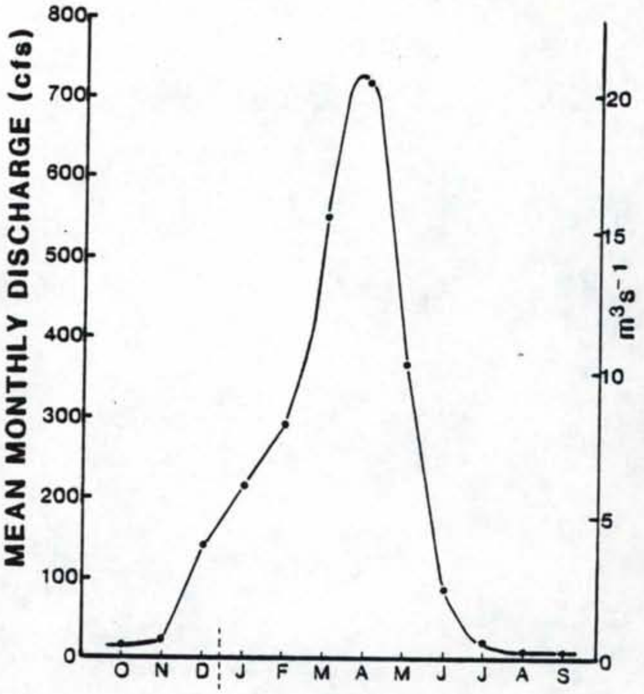


Figure 3. Palouse River discharge near Potlatch at RM 132 means for 1976, 1977, 1979 and 1980.

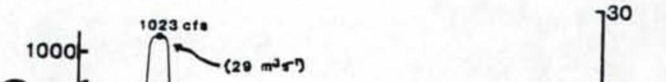


Figure 4. Palouse River discharge near Potlatch at RM 132, monthly discharge in 1980, showing May 18 ashfall occurred after peak discharge but while flows were still high.

2.2 Mosquito Creek Site Description

The Mosquito Creek site is located 1.3 miles (2.0 km) ENE of Clark Fork, Idaho on the former Clark Fork Ranger Station in Bonner County, Idaho north of the shores of Lake Pend Oreille.

2.21 Geology: The Mosquito Creek watershed is underlain by sedimentary strata of the Belt Supergroup. The upper reaches of Mosquito reaches of Mosquito Creek are underlain by quartzite with some grayish shales of the Prichard formation. The lower reaches are underlain by the Striped Peak formation which is chiefly greenish and reddish shales, sandstones and quartzites. The areas of the Striped Peak formation that Mosquito Creek traverses are generally overlain by Pleistocene glacial till and terraces of stratified outwash. (Anderson, 1947)

2.2.2 History: For the first half of the 20th century there was extensive logging and mining in the watershed. The timber in the watershed is predominately second growth. Most of the original growth had been logged off or burned over (Anderson, 1947) before the mid 1940's. Currently there are some small logging operations in the watershed, but these are limited to the recovery of downed timber for firewood. For the period 1923 through 1950 there was extensive mining of lead and silver in the watershed. Diversions were made from Mosquito Creek to supply power for an ore concentrator operation at the Lawrence Mine located a few hundred yards to the southeast of the current research station. At present, some minor diversions are made from Mosquito Creek for residential use and stock watering. Road construction and maintenance in the watershed are responsible for periodic influxes of silt-laden runoff which increases the turbidity of the creek at the research station to 200 NTUs.

2.2.3 Hydrology: Mosquito Creek is a third order stream draining a steep (2160 ft - 7000 ft; 658 m - 2130 m) heavily forested lenticular shaped watershed of 5 sq. mi. (13 km^2) extending north and east of the site. Fifty percent of the watershed's area lies above 3250 feet (922 m). The average slope of the main branch of Mosquito Creek is 0.082 ft/ft (m/m). Discharges ranged from 3.0 - 10.6 cfs (.08 - 0.3 cms) measured over the course of the investigation.

Chapter 3. Materials and Methods

The research approach included laboratory investigations, testing in a small, controlled stream in artificial channels, and verification based on investigations in a reach of the Palouse River impacted by the May 18 ashfall. Each of the separate phases of the investigation is described in this chapter. Field investigations in the Graves Meadow reach of the Palouse River were conducted in the early stages of the project. These had the purpose of providing information which would verify (or not verify) the results of simulations conducted in the artificial channels. The laboratory investigations were conducted to establish basic principles of ash behavior and impacts which would then be tested in the artificial channel simulations. The laboratory studies were conducted during the winter between the first field season on the Palouse River and the second field season on the artificial channels.

3.1 Palouse River Field Investigation

A survey was run on a 1330 ft (404 m) reach of the Palouse River at the Graves Meadow site to determine the long-term effects of the volcanic ashfall on the stream environment. The reach was divided into 44 stream habitats (15 runs, 12 riffles, 8 pools, and 9 intermediate). A detailed topographic map, bottom and water surface profiles, and 69 detailed stream cross sections were made to characterize the reach. Sediment was sampled in riffle and run habitats by a three-probe freeze core technique.

3.1.1 Cross Section: Sixty nine detailed stream cross sections at the Graves Meadow site were made (Figure 5). For every section a

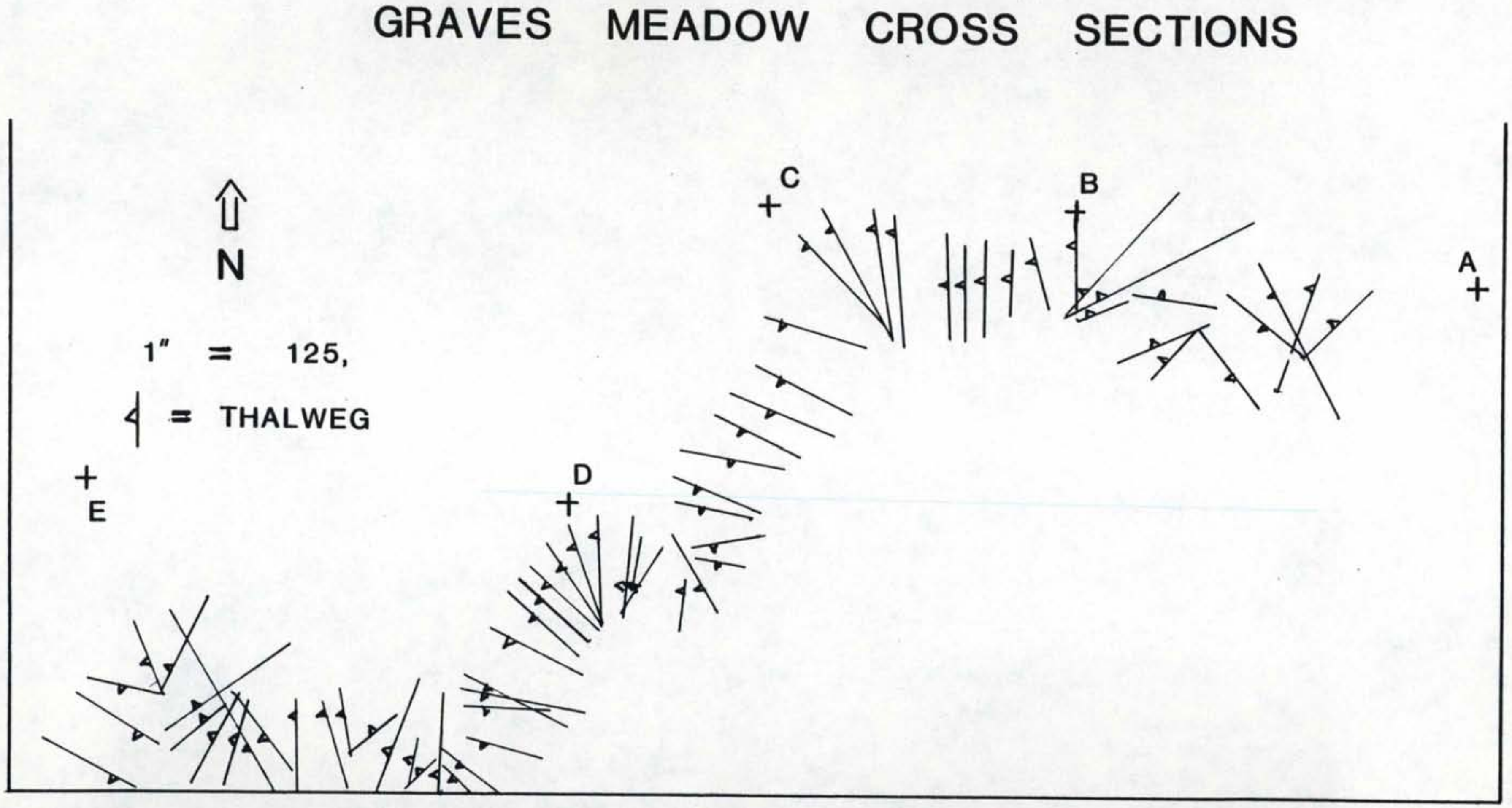


Figure 5. Graves Meadow cross sections and location map, showing detail of hydraulic geometry for this site.

fiberglass surveying tape was stretched between stakes on each bank. A surveying level and level rod were used to read elevations for each break in slope along the cross section with the distances being referenced to the stakes. The stakes were later surveyed and located on the topographic map. An HP-85 microcomputer was used to reduce the data for each section. A true scale and five to one vertical exaggeration scale cross sections were then drawn for each section. (See Appendix A)

3.1.2 Freeze Core Sampling: Fifteen freeze core samples were taken from riffle and run habitats at the Graves Meadow site. Specially designed triple-tube freeze cord samplers were driven 10 to 18 inches (254-457 mm) into the substrate. Liquid carbon dioxide was circulated through the tubes to freeze the water in the substrate for 3-4 inches (76-102 mm) around the tubes. Once frozen, a relatively undisturbed sediment sample of 20 to 80 lbs (9-36 kg) was pulled. The core was then photographed, divided into stratigraphic units and analyzed for ash composition and particle size distribution in the laboratory.

The stratigraphic units for each freeze core sample were sieved into eleven fractions (U.S. Std. Sieve sizes; #50, #60, #70, #80, #100, #120, #140, #170, #200, #230, #270) and petrographically analysed for volcanic glass and pumice. Approximately 200 grains of each fraction were examined through a petrographic microscope using reflective indices on the oil immersion mounted grains.

3.1.3 Hydraulic Characteristics: Average values for flow velocity, Darcy-Weisbach friction factor, shear velocity, and tractive force were estimated for riffle and run habitats. At selected riffles and runs in the Graves Meadow reach, measurements were made of water surface width perpendicular to the flow and of flow depth. Actual current measurements were made for each qualitative flow class. We assumed that Mannings n value for the measured sections was 0.035. Darcy-Weisbach friction factor, shear velocity, and tractive force were then calculated for each section using the formulas:

$$f = (116n^2)/(R^{1.3}) \qquad \text{(Eq. 1)}$$

$$U^* = V \quad f/B \qquad \text{(Eq. 2)}$$

$$T = \rho(U^*)^2 \qquad \text{(Eq. 3)}$$

where f = Darcy-Weisbach friction factor; n = Mannings friction factor ($1/L^6$); R = hydraulic radius for the assumed cross section shape (L); U^* = shear velocity (L/T); V = flow velocity (L/T); T = tractive force for assumed geometry (F/L^2); ρ the fluid density (M/L^3). A summary of the hydraulic parameters can be found in Table 1.

3.1.4 Attached Benthic Algae: Plant communities of the Graves Meadow reach of the Palouse River were quantified by measurement of apparent attached benthic algae (ABA) biomass and chlorophyll "a" on natural rock surfaces incubated in the river. Our goal was to utilize natural rock substrate yet circumvent the usual problem of quantitative sampling from rocks where irregular rock surfaces render relation to surface area difficult. Our approach was to place numbered, air-dried and cleaned natural rocks in the river at random locations on benthic invertebrate and sediment composition transects.

After a 28 day incubation, the rocks were removed and the area of ABA coverage circumscribed with a marking crayon. Depth, off-bottom velocity, and exact location were noted for each rock. Each rock was brushed and rinsed to remove the ABA from the entire algae-covered area. The rocks were saved for later determination of area of algal

Table 1. Flow Parameters for Graves Meadow,

95% Confidence Interval Standard Error Means

GRAVES MEADOW

	Riffles n = 4		Run n = 9		Mosquito Creek Channels
	Conf. Int.	Mean	Conf. Int.	Mean	
Velocity ft/sec (m/s)	0.4 (0.12)	1.8 (0.54)	0.2 (0.07)	0.5 (0.15)	0.5 (0.15)
Friction factor f	0.186	0.279	0.140	0.187	0.217
Shear velocity U* ft/sec (m/s)	0.060 (0.018)	0.317 (0.097)	0.028 (0.009)	0.076 (0.023)	0.0754 (0.023)
Shear stress T lbf/ft^2 (N/m^2)	0.002 (0.077)	0.156 (7.47)	0.002 (0.086)	0.006 (0.297)	0.0110 (0.527)

coverage. This was achieved by coating the top surface of each rock with a quick-dry theatrical latex coating brushed on to a cheesecloth mat. After drying (1 hour), the latex cheesecloth coating was peeled off and its surface area determined by planimetry of a photocopied image. The cheesecloth mat prevented distortion by stretching. Miscellaneous cuts in the mat were made so it could lie flat for photocopying. In this way, accurate surface area measurements were obtained of natural, irregularly-shaped rock substrates.

Each sample was split for determination of ash-free dry weight ("organic wt") and trichromatic chlorophyll "a" (APHA 1981). The autotrophic index (Organic wt/chorophyll wt) was used to measure the primary production efficiency of the community based on individual rock samples.

3.1.5 Benthic Insects: Standardized benthic insect samples were collected with a $0.025 m^2$ modified Hess sampler fitted with a 200 micron mesh net. Design and operation of the Hess sampler is described by Usinger (1956). Forty-eight randomly selected samples were collected from each riffle, run and pool habitat located in the study reach. Riffles were visually determined as those areas having moderate to steep gradient and a substrate of cobbles causing surface turbulence; runs were reaches where increased depth, decreased gradient and/or smooth substrate creating a chiefly laminar surface flow; and pools were areas having quiet water with little or no observable current.

The physical variables measured at each sample location included water depth, current velocity and substrate. Current readings were taken at 0.6 the distance from the water surface to the bottom with a Marsh-McBirney #201 magnetic current meter. The bottom substrate was described using the substrate classification described in Bjornn et al. (1977). This substrate classification records three major characteristics of the surface substrate: the size range of the dominant substrate; the degree of embeddedness of cobble-sized rocks or larger; and the size of the particles surrounding or embedding cobble-size rocks. Other bottom features recorded were algae cover, presence of moss (*Fontinalis*) and deposition of particulate organic matter (POM).

The samples were preserved in 70% alcohol and brought back to the laboratory where the invertebrates were sorted from the debris with the aid of a 2X illuminator-magnifier. All insect groups except some Diptera were identified to species or morphospecies, counted and numerically expressed as density/m^2.

3.2 Laboratory Simulations

3.2.1 *Deposition Characteristics of Ash*: Using functions developed by Camp and Stein (1943) for open channel settling basin and paddle flocculation equipment, a series of jar tests were run to detemine percent ash removel from the flow regime as a function of the rootmean-square velocity gradient (Gm). In application, Gm values can then be computed for a given reach of stream. With the Gm value, the amount of material removed from the flow regime and incorporated into the bedload can be estimated from the empirical curves developed in this report.

Laboratory: In the test, 4000 mg of ash from the May 18th eruption were mixed with 1000 ml of University of Idaho tap water and stirred at a given rotational speed in a Coffman Industries Model 73 Magnadrive Jar Testing Machine. There were six replications at each of ten different rotational speeds. After an hour at a given rotational speed, 25 ml of the mixture was withdrawn with a volumetric pipette from 3/4 inches (19 mm) below the surface. Each sample was vacuum filtered through a weighed 2.1 cm Whatman Glass Fiber Paper. The filter was oven dried at 217 F° (103 C°) and reweighed. The mean, standard deviation, standard error of the mean and 95% confidence interval for the standard error of the mean were computed for the percent removal of material at a given energy level. The 95% confidence interval for percent removal versus the root-mean-square velocity gradient (Gm) was plotted (Figure 6).

Models were developed to predict the scour transport and deposition of volcanic tephras in stream environments. A series of laboratory and flume studies were conducted to determine the quantities and distribution of silt-size volcanic ash that may be incorporated into a stream reach. These models were then tested in the Mosquito Creek channels.

Deposition: In the depositional tests, 4000 mg of ash was stirred at a given rotational speed in 1000 ml of University of Idaho tap water by a Coffman Industries Model 73 Magna-drive Jar Testing Machine. After an hour at a given rotational speed the supernatant was siphoned off with a vacuum pump. A pipette method was used to determine the grainsize distribution of the material in the supernatant. The test was repeated at four different energy levels. The post-depositional grainsize distribution was compared to the distribution of the initial material to determine the percent of each grainsize removed. The critical grainsize for that rotational speed was the grainsize where 50% of that size was removed. For each energy level a fluid velocity (U), shear velocity (U*), and tractive force (T_C) were computed. These parameters were combined with the critical grainsize to calculate

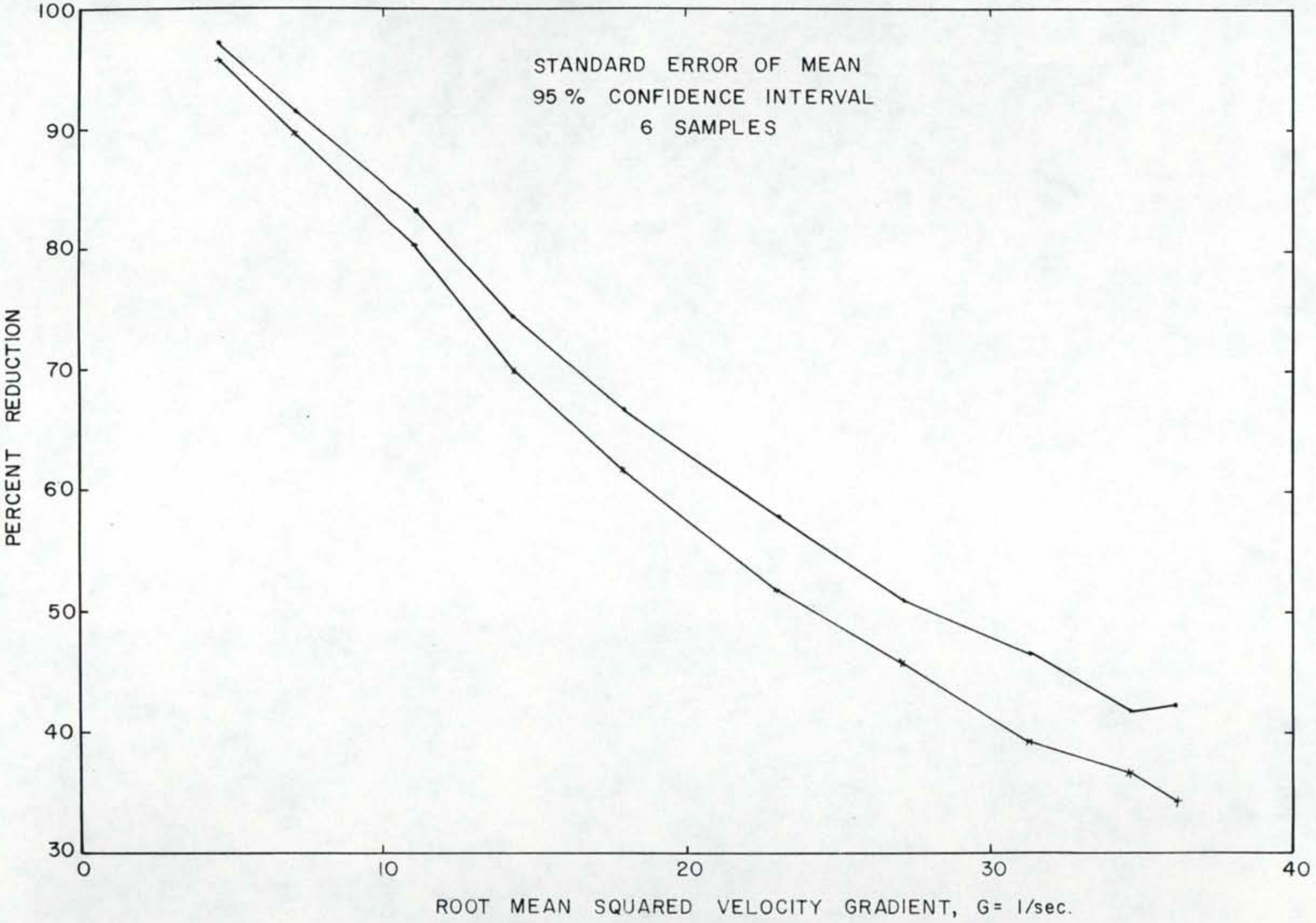


Figure 6. Turbulent Ash Settling Behavior.

a particle Reynolds number (R_d) and the Shields Parameter (T_d) for that energy level:

$$R_d = (U)D_c \,/\, \nu \qquad \text{(Eq. 4)}$$

$$T_d = T_c/((G_s-G_f)*D_c) \qquad \text{(Eq. 5)}$$

where R_d - Reynolds number for deposition; U - fluid velocity; D_c - diameter of the grainsize that is 50% removed; - kinematic viscosity of the fluid; T_d - Shields parameter for Deposition; T_c - the critical tractive force for the condition; G_s, G_f specific weight of the sediment and fluid respectively. A pseudo-Shields diagram for deposition was developed by plotting the Reynolds number (R_d) versus Shields Parameter (T_d) on log-log paper (Figure 7). Gessler's (1965) procedure was then followed to produce a percent removal versus critical tractive force ratio curve (Inverso 1982).

In application one first computes the fluid parameters (G_f, ν), obtains the grainsize distribution for the material being introduced (G_s, D_c), the flow velocity (U), the tractive force (T) for the reach, and for each grainsize, the depositional Reynolds number (Eq 4). From Figure 7 one obtains the corresponding Shields parameter (T_d) and back-calculates the critical tractive force (T_c) for that grainsize. The ratio of critical tractive force to the tractive force in the reach (T_c/T) is calculated and entered into the Depositional Gessler's Diagram (Figure 8) to find the percent of that grainsize that is removed from the flow regime and incorporated into the bed. Knowing the initial mass of material at each grainsize interval and the percent of that material incorporated into the bed, one can compute the grainsize distribution and the amount of material removed from the flow.

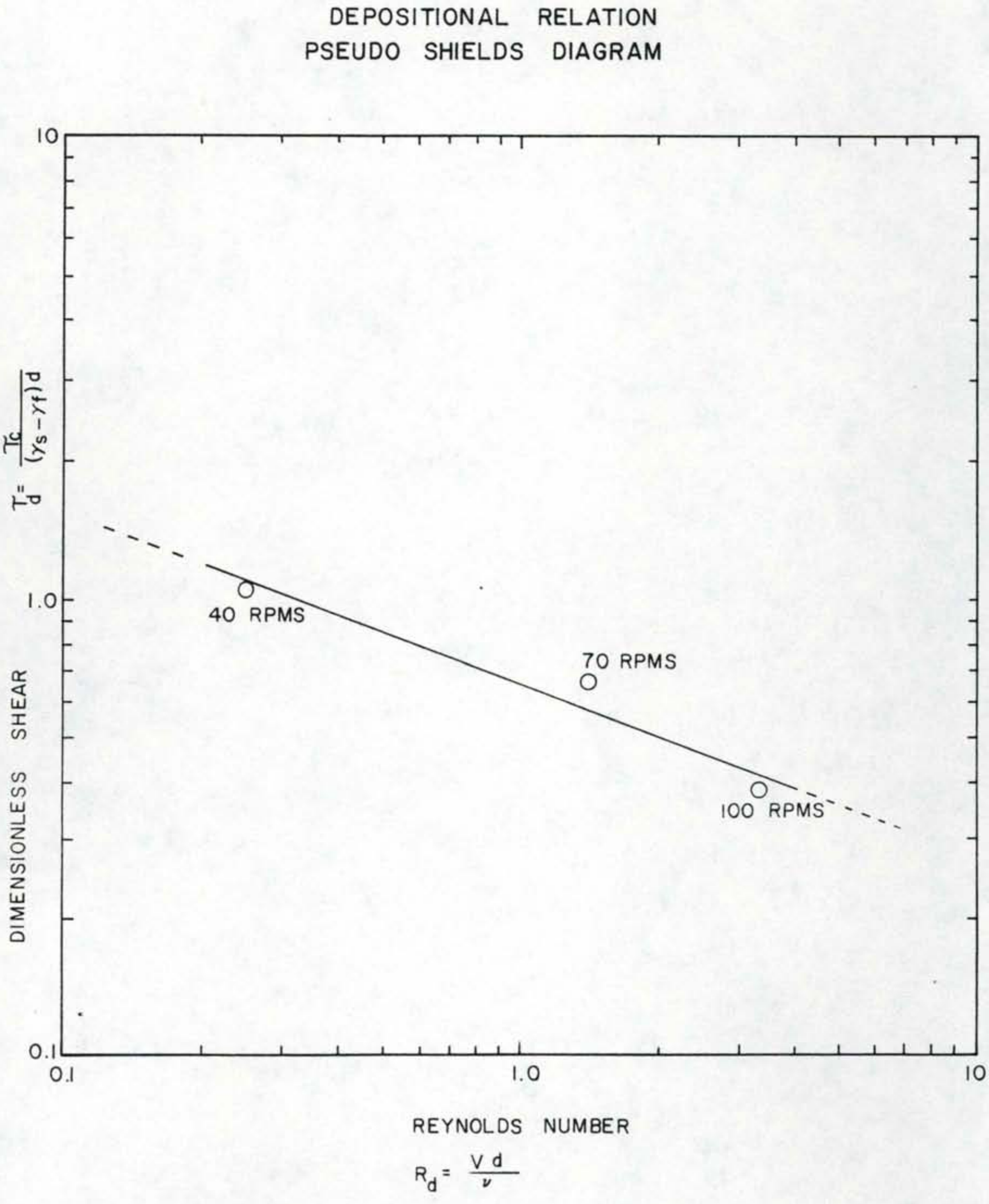


Figure 7. Depositional Relation Pseudo Shields Diagram

Scour: In the scour study, five seperate beds of ash were aqueously deposited in the University of Idaho's 26 foot (7.9 m) tilting flume and were scoured under different flow conditions. For each test the pre- and post- scour grainsize distributions were determined by a hydrometer technique (ASTM D422-63) as outlined by Bowles (1978). The flow velocity, tractive force, shear velocity and percent of bed removal were also determined. Following the same procedure as in the depositional investigation, the percent removal of each grainsize was computed along with the critical grainsize (50% removal). For the scour tests, the inital bed depth was 3/4 inch. In each test the flow was increased by increments every hour. The flow at which the entire bed was in motion as ripples of 3/4 inch (19 mm) amplitude was considered the critical flow, and the corresponding critical tractive force was computed. An extension of the traditional Shields diagram (Figure 9) was developed by plotting the particle Reynolds Number (R_S) against the Shields Parameter (T_S) for each condition:

$$R_S = (U^*)D_C / \nu \qquad \text{(Eq. 6)}$$

$$T_S = T_C/((G_S-G_f)D_C) \qquad \text{(Eq. 7)}$$

where R_S - Reynolds number for scour; U^* - shear velocity for bed; D_C - diameter of the grainsize that is 50% removed; ν - kinematic viscosity of the fluid; T_S - Shields parameter for scour; T_C - the critical tractive force for the condition; G_S, G_f specific weight of the sediment and fluid respectively. A Gessler diagram was then developed for scour (Figure 10).

The technique to apply the scour procedure is simular to that for deposition. First R_S (Eq. 6) is computed for each grainsize. Then,

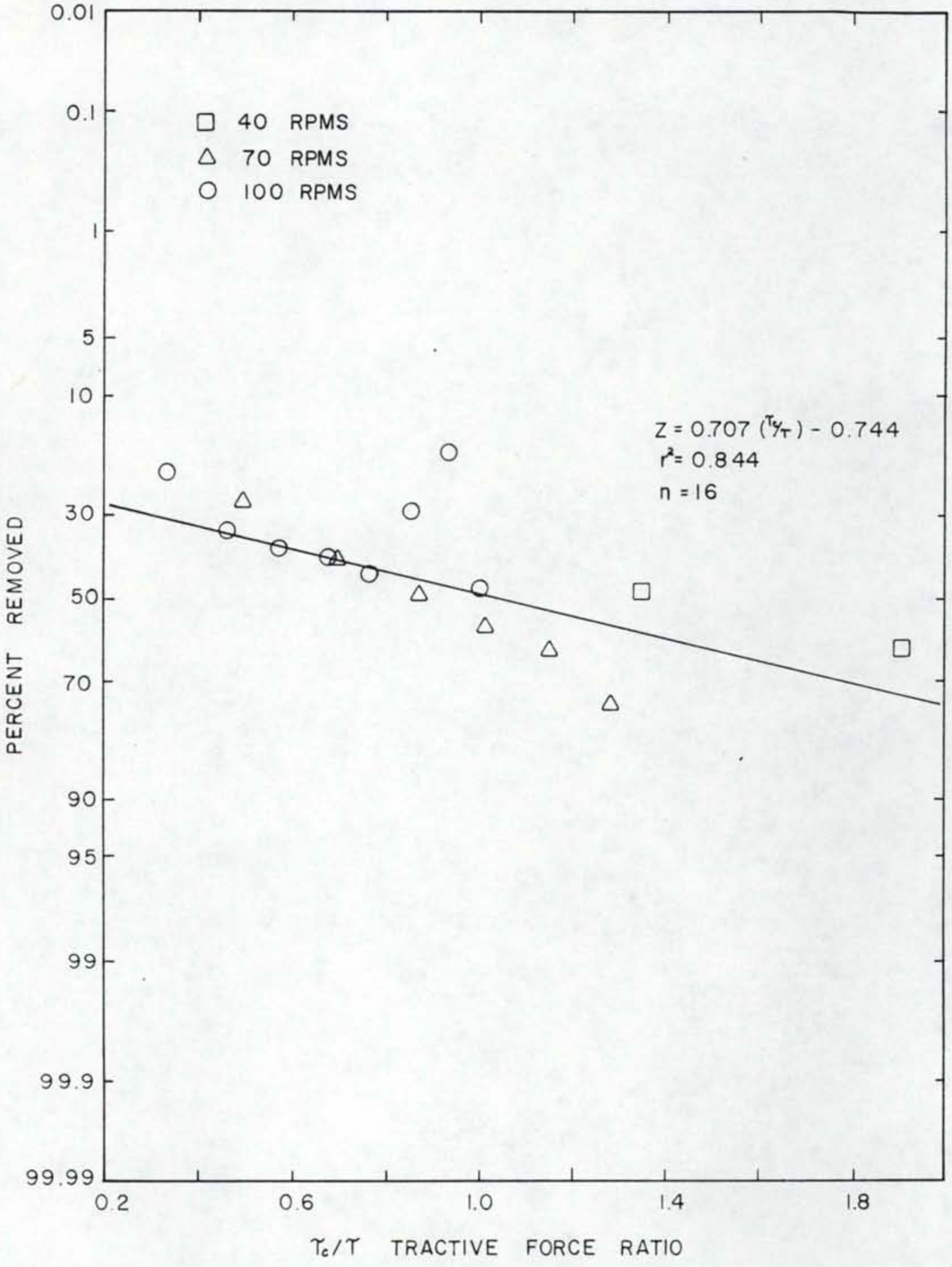


Figure 8. Depositional Gessler Diagram

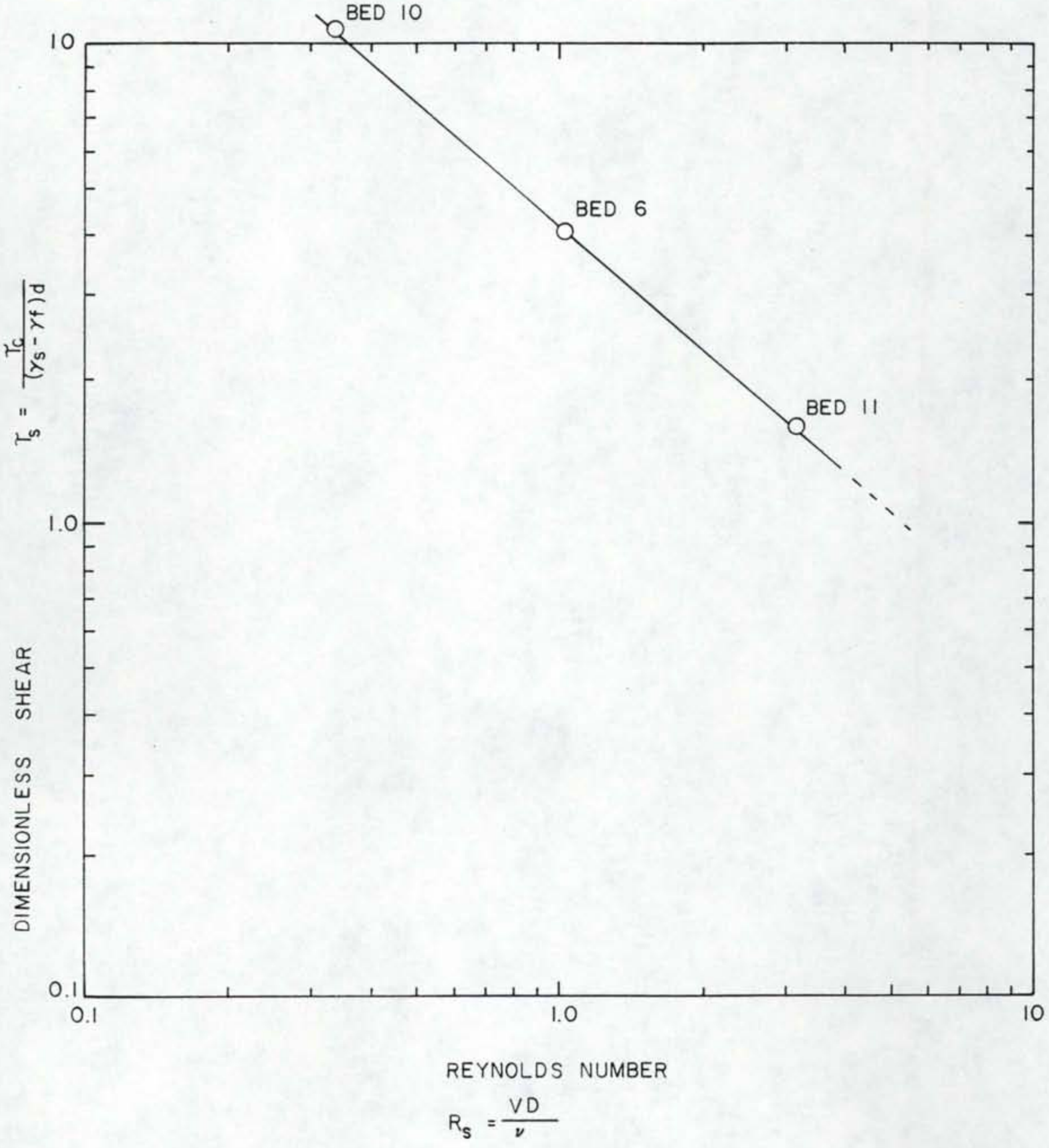


Figure 9. Scour Relation
Extended Shields Diagram

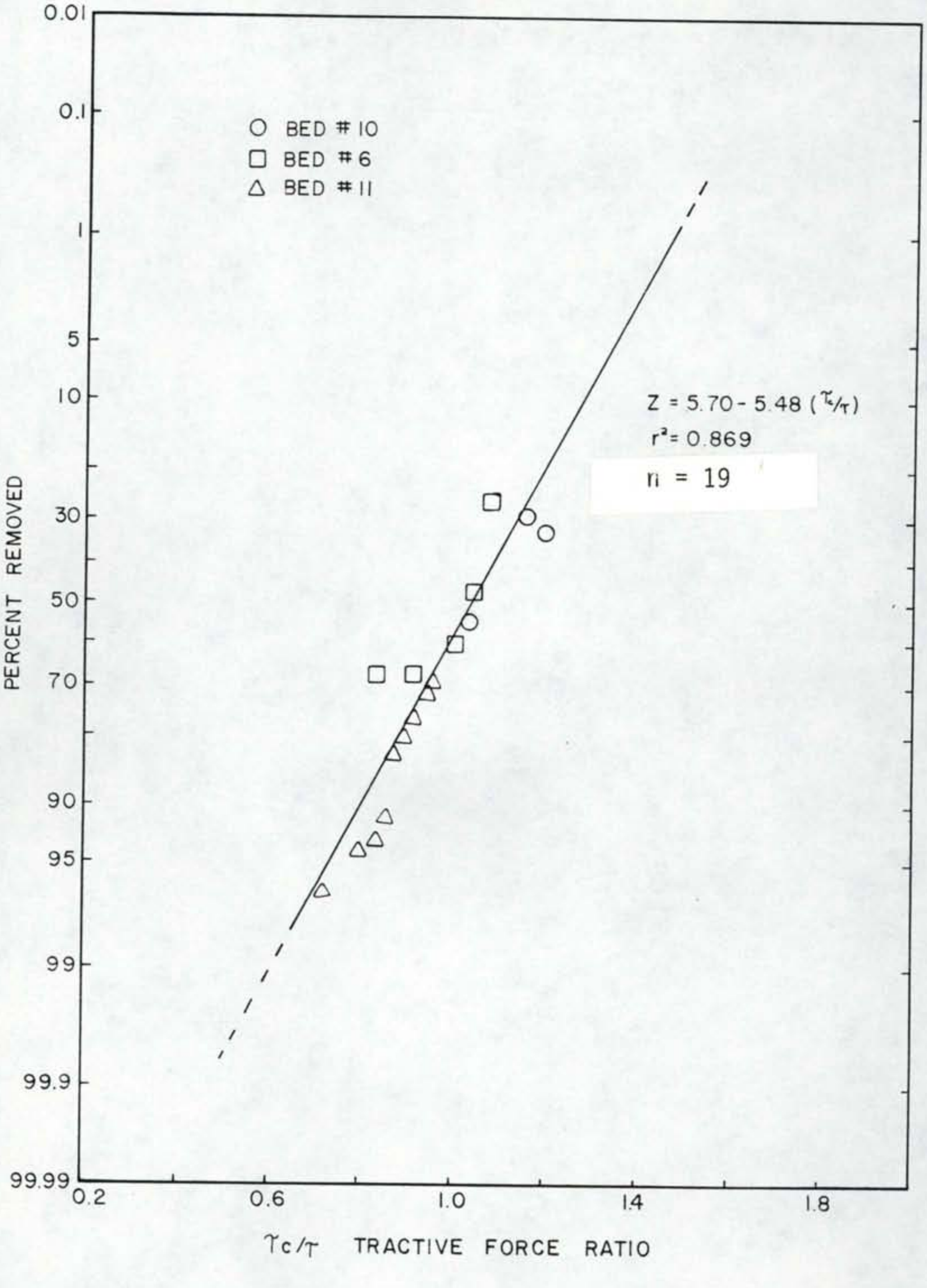


Figure 10. Scour Gessler Diagram

the corresponding T_S value is obtained from the Gessler diagram, the appropriate T_C value is back-calculated, and the T_C/T ratio is computed. Using the Scour Gessler Diagram, (Figure 10) the percent of material removed from the bed can be calculated.

3.2.2 Laboratory Flume Studies - Insects: The experiments used to test suspended ash effects on *Hesperophylax occidentalis* were designed to: maintain suspended ash concentrations at desired levels, maintain high ash concentrations (>2000 mg/l) over extended periods of time, and permit randomly-assigned replications of control and test conditions.

Laboratory experiments were conducted in plexiglass, flow-through streams similar to those described by Brusven (1973). Unchlorinated tap water was recycled through the stream and a 141-liter plastic holding reservoir. The passover system was equipped with a thermostatically-controlled refrigerator unit connected to a stainless steel cooling coil.

The effects of volcanic ash on feeding rates of the stream detritivore (shredder) *H. occidentalis* were examined. The test material was Mount St. Helens volcanic ash collected near the University of Idaho campus within one week of the May, 1980 ashfall. The ash was sieved through a 125 micron mesh screen, mixed with water, and added to the test chambers as a slurry.

The test and control chambers consisted of 2-liter glass beakers. The beakers were placed in a plexiglass stream maintained at 5°C (± 1°C). The water in each beaker was aerated with an airstone. The plexiglass stream was placed on top of magnetic stirrers, which were positioned beneath each beaker. A coarse-mesh screen (1 mm mesh) was used to divide the beakers into an upper and lower chamber to protect the insects from the stirring rods.

Two tests were conducted at different suspended ash concentrations (1000-1500 mg/l and 2000-3000 mg/l). Volcanic ash treatments were randomly allocated to five of the beakers, with the remaining five beakers serving as control replicates. Two, 14-day trials were conducted with a different set of 20 insects (2/beaker) in each trial.

Test insects were initially individually weighed to the nearest 0.1 mg then paired in a manner to obtain similar combined wet weights for all pairs. The ten resulting insect pairs were randomly allocated to the ten experimental beakers. At the end of each 14-day trial, wet weights of these insects were obtained together with their ash-free dry weight (AFDW). The end-of-trial ratio of wet weights to AFDW was used to estimate the initial AFDW from the initial wet weights. The initial to final AFDW difference was calculated as weight change for each specimen.

For each replicate, Alder (*Alnus* sp.) leaf disks (21 mm diameter) were placed in each beaker. Leaf breakdown due to insect feeding was estimated for each beaker as the differences in ash-free dry weight (AFDW) between disks kept in insect-free containers and those having insects. Because the weights of the insects used differed among trials, the Consumptive Index (mg AFDW Leaf breakdowns/mg animal AFDW/day) of Walsbauer (1968) was calculated. Water samples from each container were dried and filtered to determine suspended ash concentrations.

3.3 Mosquito Creek Simulation

The May 18th, 1980 Mt. St. Helens ashfall was simulated in a set of artificial flumes at the Mosquito Creek site. Plume ash was injected over a 24-hour period to simulate the natural rise and fall in suspended sediment and turbidity that may have occurred in streams in the plume area. During the simulation, suspended sediment, turbidity and insect drift measurements were taken. Benthic samples were also taken to determine the quantities and distribution of ash particles incorporated into the channel and the effect of the ash loading event on attached benthic algae and benthic insect populations.

3.3.1 Flume Configuration: The Mosquito Creek flumes consisted of three parallel fiberglassed plywood channels (2 ft w x 40 ft l; 0.6 m w x 12.2 m l) situated parallel to the creek (Figure 11). Water was diverted from Mosquito Creek into a riprapped basin, then over V-notched weirs into each channel. The flow over the weirs plunged 1.5 ft (0.5 m) into a riprapped 4 ft (1.2 m) long head-box before it entered the channels. Flash boards were placed at the tail end of each channel to control water depth in that channel. Water flow into the channels was controlled by a diversion dam in Mosquito Creek. The flow rates (equal in each channel) could be regulated over the range of 0.0 to 2.5 cfs. Each channel could also be fitted with a 1/8 inch plexiglass cover below the water surface to eliminate atmospheric contact for algae community respiration experiments.

Six (6) cubic yards (4.6 m^3) of substrate materials were obtained from a braided stream deposit of reworked Pleistocene glacial till in the Lightning Creek drainage 1.5 miles (2.4 km) west of the Mosquito Creek site. Boulders and cobbles larger than 6 inches (150 mm) (about 1/3 the volume) were hand-culled from the deposit. The remaining material was screened through 1/2 inch (13 mm) hardware cloth and recombined in the proportions of two parts of material coarser than 1/2 inch (13 mm) with one part material finer than 1/2 inch (13 mm). This

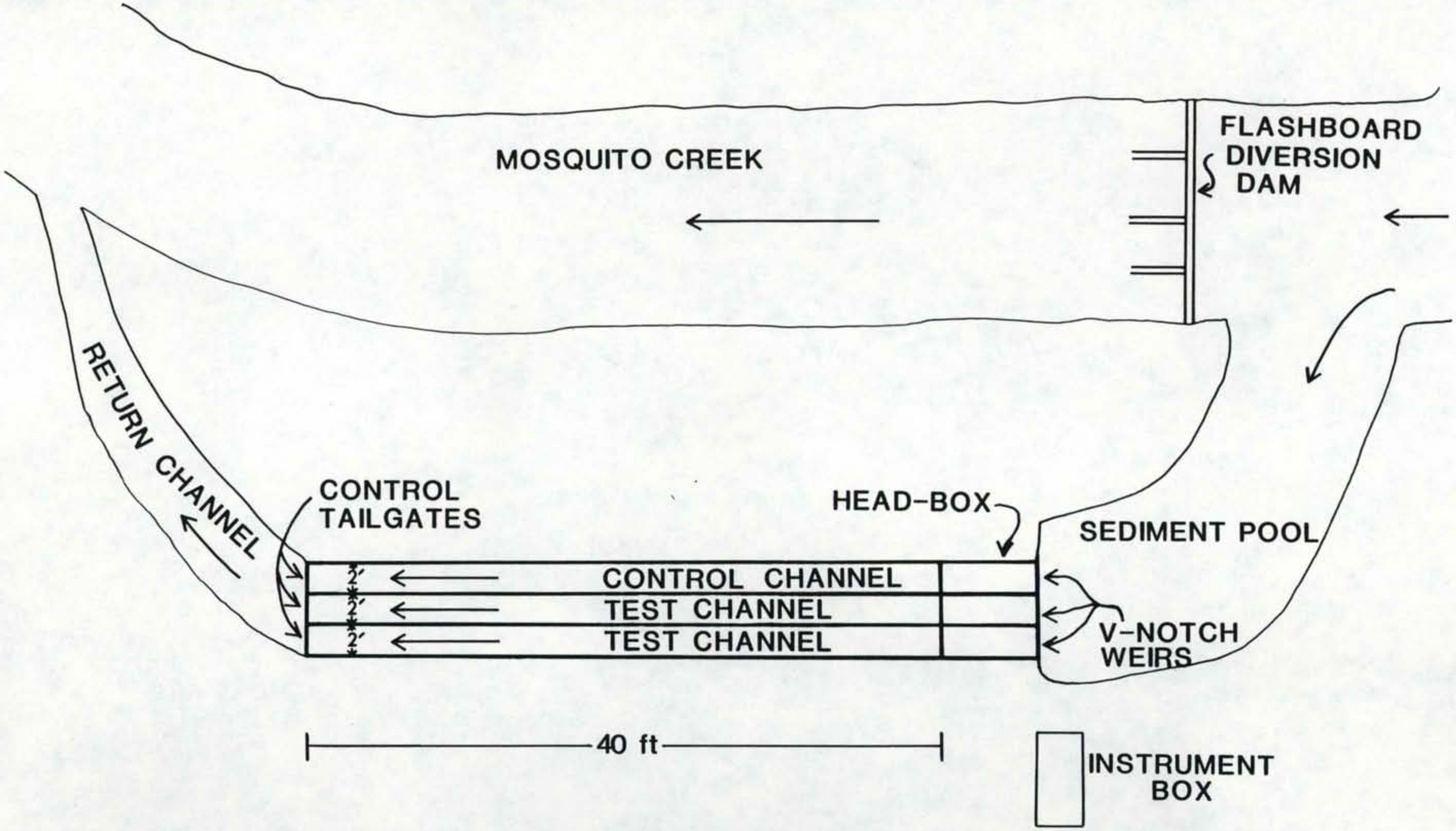


Figure 11. Sketch Showing Layout of Mosquito Creek Flumes at the UI Clark Fork Station

reworking was done so the bed materials in the flumes would more closely reflect the distribution of materials in Mosquito Creek and to ensure uniformity of bed material between channels.

After all the canister samplers (described in Section 3.3.4) had been placed, the remainder of the bed was added flush with the top of the canister. A 5 inch (127 mm) deep bed of alluvial materials were placed in each channel. Once the substrate had been deposited, the water flow of 2.5 cfs (0.071 cms) was started and maintained for three hours to clean and align the bed material.

An attempt was made to measure the bedload movement out of the channels. The lowest flash board in each channel was replaced by a 1 x 24 inch (25 x 610 mm) frame made of 3/8 inch (10 mm) diameter stainless steel rod. To this frame was attached a pyramidal shape Naugahide sleeve that tapered to fit a 2 inch (51 mm) diameter aluminum pipe. The pipe was then fitted with nets to collect the insect drift. Suspended sediment and turbidity measurements were made on the water flowing from these pipes.

3.3.2 <u>Hydraulic Instrumentation</u>: Measurements of flow rate, turbidity, temperature, conductivity, and dissolved oxygen were made electronically with the aid of a Hewlett-Packard 3497A/HP85 data acquisition system.

Flow rates were determined at the V-notched weirs at the front end of each flume. The depth of water in the pool in front of the weirs was measured in a stilling well by a float connected to a multi-wound potentiometer. The HP3497/HP85 system read the resistance in the potentiometer and then calculated the head on the weirs before applying an empirical curve to determine the flow rate for the channels. The flow in each channel was the same.

Turbidity was measured with a Hach Ratio Turbidimeter with a flow-through cell attachment. Plastic tubes 1/4 inch (6.3 mm) in diameter were placed in the bedload sampler. Water was drawn through the tube and the flow-through turbidity cell by a small chemical pump attached to the exit side of the cell. Dilution water, when needed, was drawn through the same system from the control channel. The turbidity meter was read by the data acquisition system and the micro-computer applied the appropriate adjustment for dilution.

Temperature and electrical conductivity of the water in the pool at the diversion dam in Mosquito Creek were measured with a Marsh-McBirney electromagnetic velocity probe. These readings were monitored by the data acquisition system.

Dissolved oxygen was measured in pool at the tail end of the flume using an Extech dissolved oxygen meter with electronic output signal to the data aquisition system. Parts per million of dissolved oxygen were read and recorded directly by the data acquisition system. The micro-computer, using water temperature readings, applied an empirical curve adjusted for the elevation at the Mosquito Creek site, then computed the percent oxygen saturation.

Prior to the simulation, the Darcy-Weisbach friction factor (f) was determined for each channel. The elevations of the bed and water surface were measured using a surveying level for a given flow. The depth of flow, slope of the energy line, and hydraulic radius were then calculated. From these parameters the Darcy-Weisbach f was computed. During the test this value of f was used to calculate the shear velocity (U*) and tractive force on the bed (T).

$$U^* = \quad f/B \; V \qquad \text{(Eq. 8)}$$

$$T = \rho \, (U^*)^2$$

where U^* is shear velocity (L/T); f is the Darcy-Weisbach friction factor; V is the flow velocity in the channel (L/T); T is tractive force (F/L^2); ρ - density of the fluid (M/L^3).

3.3.3 Ash Injection Routine: During the simulation, 1430 lbs (650 kg) of plume ash from the May 18, 1980 Mt. St. Helens eruption was introduced at the head end of the test flume over a twenty-four hour period following a triangular input scheme (Figure 13) with the peak occurring at twelve hours into the test. The grainsize distribution of the ash used in the test (Figure 12) was determined as an average of six samples.

To simulate the hypothetical rise and fall of suspended sediment in a stream, a series of assumptions were made: 1) the ash fell at a constant rate for a twelve hour period; 2) a uniform reach of the stream with twenty-four hours of retention time was contained in the plume area; 3) a fixed percentage of ash settled out for the given hydrologic conditions; and, 4) the stream acted as a continuous conveyor belt one foot wide (305 mm), moving at the same rate top to bottom, with the monitored section of the belt at the end of the plume area. With these assumptions, a plot of the ash flux per unit of time that passed the monitored section would form a triangle with a base twenty-four hours times the rate surface flow, and a peak equal to the total mass of ash per square foot. The limitations of machinery used and water use permit issued for the research station dictated that the that the input of ash could not exceed 150 lbs/hr (68 kg/hr), thus, the ash was injected as outlined in Figure 13.

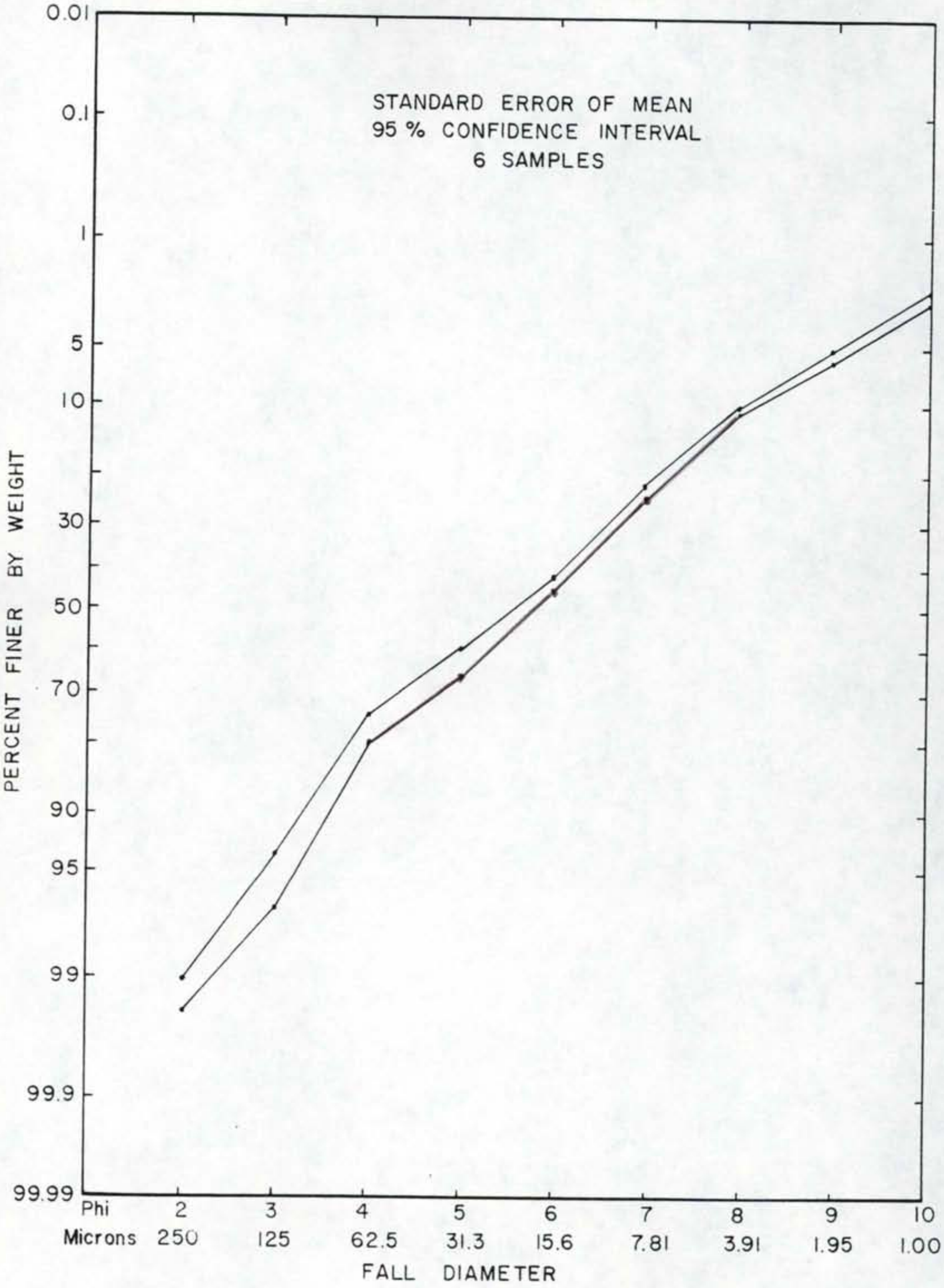


Figure 12. Ash Grain Size Distribution

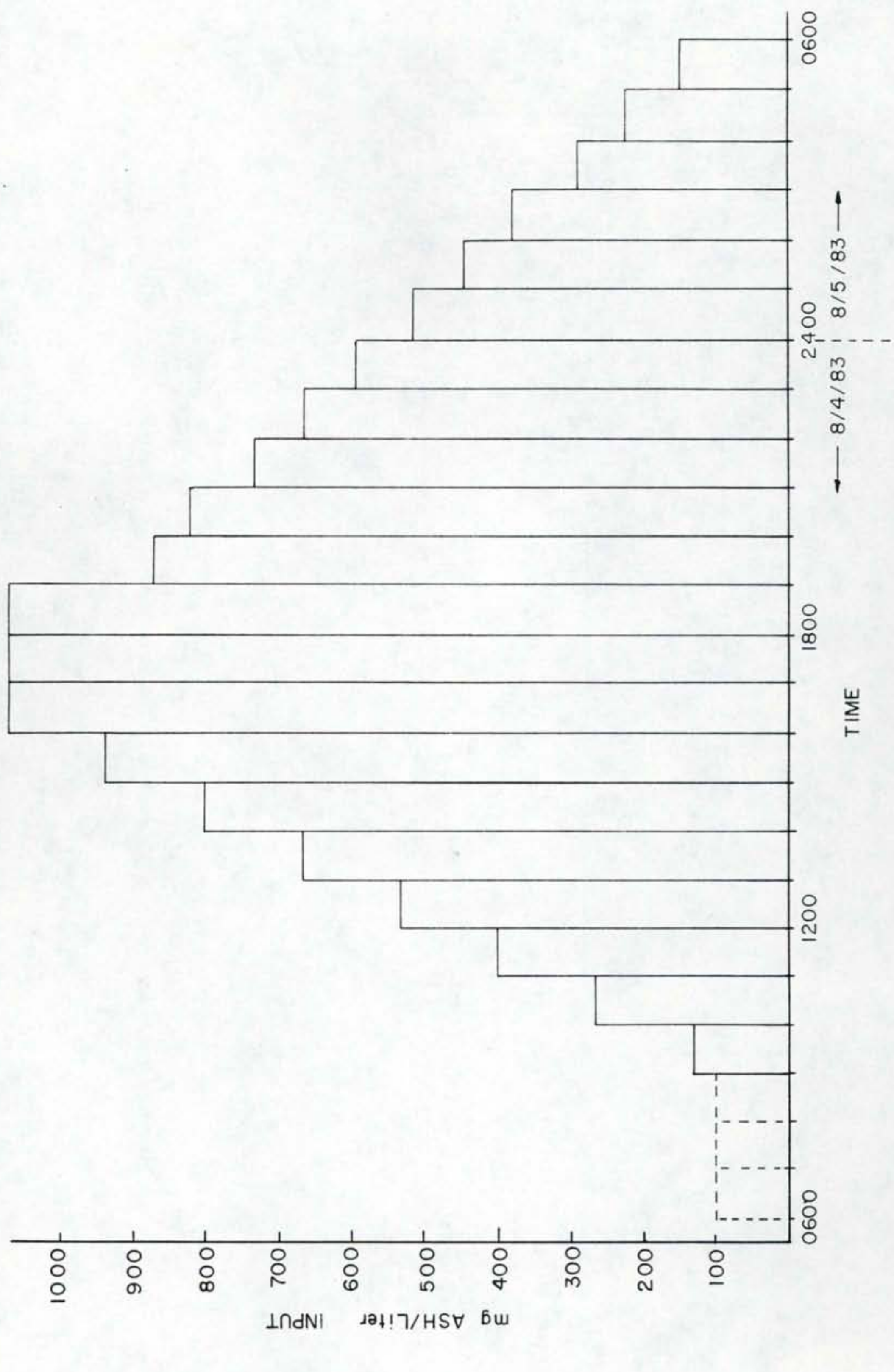


Figure 13. Ash Injection Schedule

A modified Scott Precision Flow Fertilizer Spreader was used to inject the ash: the Spreader was fitted with a variable speed electric motor and was mounted over the head end of the test flume, so that the ash could fall directly on the nappe of the flow as it came over the weir. The flow, plunging into the head box, provided ample mixing energy to uniformly distribute ash across the test channel inlet. Since the ash clumped when moist, heat lamps had to be provided inside the spreader to keep the ash dry for uniform distribution.

3.3.4 Composite Sampling Procedure: Sampling of insect drift, attached benthic algae, benthic insect populations, and bed materials was done over a period of seven weeks (three prior to the simulation, four following) to determine the immediate impact on and the short-term recovery of benthic ABA and BMI communities.

A large number (126) of specially designed benthic samplers were placed in the channels to simultaneously collect attached benthic algae, insects, and bed materials over the course of the investigation. Samples were collected in seven sets of six samples from each channel (Table 2). Three sets of samples were collected over a three-week period prior to the simulation. Four additional sets of benthic samples were taken over the four weeks immediately following the simulation to determine the impact on and recovery rates of the benthic environment.

The canister samplers consisted of a 5 inch (127 mm) diameter by 5 inch (127 mm) high cylinder made from 1/4 inch (6.3 mm) hardware cloth. Thus the volume of each canister was 0.057 ft^3 (0.0016 m^3). Each cylinder was fitted inside a 6x3x15 inch (162x76x381 mm) 3 mil plastic sample sack which was seated around the basket base, thereby permitting free flow of water, fine sediments, and organisms through each basket.

Table 2. Time schedule of Sampling in the Mosquito Creek experimental stream channels, 1982.

Date	Task
30 June	Water into channels
13 July	1st sample period
20 July	2nd sample period
2 Aug	3rd sample period
4 Aug	0600 Initiate large (catastrophic) ash dump in left channel
	1800 Peak of large ash dump
5 Aug	0600 End of large ash dump (Total dump = 17.5 lb/ft^2)
5 Aug	4th sample period
6-10 Aug	1000-1300 hrs Slow ash dump in center channel (0.175 $lb/ft^2/day$)
11 Aug	5th sample period
20 Aug	6th sample period
21-22 Aug	1st stream capping experiment
25 Aug	Sample 1st set of colonization rocks
27 Aug	7th sample period
4 Sept	Sample 2nd set of colonization rocks
11-12 Sept	2nd stream capping experiment

Attached to the sample sack was a 6 inch (152 mm) diameter 16 gage wire ring that had two 12 inch (25 mm) 20 gage wire handles. When the samplers were charged, the cylinders were placed inside the sample sacks and the sacks were packed down around the outside of the cylinder on the bottom. Substrate was placed in and around the sampler with the handles protruding from the top of the bed.

Each channel was divided into upstream and downstream reaches (20 ft long; 6.1 m). Each reach was then subdivided into 40 cells with two columns of twenty cells centered 6 inches (152 mm) from the channel walls. Twenty-one basket samplers were placed in randomly selected cells. The same pattern of placement was used both upstream and downstream for all three channels (126 samplers in all). The twenty-one locations were then randomly grouped into seven sets of three replicates each. The samplers for those three locations in both the upstream and downstream reaches in the three channel (18 samplers in each set) were then pulled as a set during each of the next seven weeks.

The canisters were loaded in rows from right to left across the channels working from upstream to downstream in the channels. The substrate was taken from a blended batch of gravel (cobbles that did not readily fit the can were discarded) and the material dumped into the canisters and lightly packed.

The canister samplers were pulled from their predetermined location starting at the downstream end of the flume and proceeding toward the head end. A single rock was first removed from the bed surface within the canister for ABA sampling. After a nylon organdy drift net was placed downstream of the canister location, the rock for attached benthic algae was removed. The sample itself was taken by pulling the ring and sack through the substrate to the surface of the water, while holding the cylinder in place. Thus the whole sample substrate, including cylinder and water column, was pulled. The sample, involving the first two inches (50 mm) of the substrate, was vigorously stirred by hand in the sample sack to suspend the insects. The supernatant was immediately poured through a nylon organdy net to capture the insects. Material that remained in the net was saved for later entomological analysis. The rest of the sample was combined with the supernatant in an air-tight container for later grainsize analyses. The hole made in the bed by the removal of the sampler was filled with sand, gravel and cobbles of material similar to that in the bed.

Each hour during the Mosquito Creek simulation, suspended sediment and turbidity samples were taken before the next level of ash introduction was started, to determine the amount of sediment moving near the surface and near the bed for a given ash input. Suspended sediment samples were taken in 300 ml BOD bottles from the lower end of the test flume and from the pipe at the end of the bedload and insect sampler. The two suspended sediment samples were then vacuum-drawn through preweighed Millipore filters. Later, the filters were oven dried, reweighed and the differences in filter weight used to calculate the amount of suspended sediment moving near the surface and near the bed for the ash concentration in question.

Sieve and hydrometer analyses were used to characterize the grainsize distributions for the 126 benthic samples and ash used in the Mosquito Creek simulation. The samples were oven dried at 230° F (100° C) for at least twenty-four hours. The samples were then sieved into 18 fractions (3", 2", 1 1/2", 1", 3/4", 5/8", 1/2", 3/8", #4, #8, #10, #16, #30,, #40, #50,, #100, #200, and minus #200). A hydrometer test (ASTM D421-58 and ASTM D422-63) as outlined by Bowles (1978) was run on the minus #200 fraction. For each sample a computer-generated synthetic grainsize distribution was computed for full phi intervals (phi = -log2(d_i/1000) where d_i is the grain diameter in microns) from -6 phi (2.5 in; 64 mm) to 10 phi (0.00004 in; 1 μm). These synthetic grainsize distributions were used for the rest of the calculations involving grainsize.

The amount of material incorporated into the bed was determined statistically from the increase in the material finer than 0 phi (0.039 inch; 1 mm) immediately following the simulation. The mean, standard deviation, standard error of the mean and 95% confidence interval of the standard error of the mean were computed for the percent finer by weight for the 17 fractions ranging from -6 phi to 10 phi (2.5 to 0.00004 inch; 64 mm to 1 μm). Since all the ash material added during the simulation was finer than 0 phi (0.039 inch; 1 mm), a difference of independent means t-test was run on the fraction finer than 0 phi. The mean percent finer than 0 phi of the post-simulation samples (6 samples) was subtracted from the mean of the pre-simulation samples (16 samples) for the test channel to quantify the ash incorporated into the streambed materials.

3.3.5 <u>Algal Colonization and Respiration</u>: As with Palouse River samples, organic weight and chlorophyll "a" were determined on rocks removed from the experimental stream channels. Rocks were taken from the surface rubble of the buried canisters. Samples were either dried for organic weights or processed for chlorophyll "a" immediately. We sampled according to the schedule in Table 2 and processed ABA samples according to APHA (1981).

3.3.6 Benthic Insects: Total density (numbers/canister and per m^2) and dry-weight biomass (mg/canister and per m^2) of the insects collected in canister samples (described previously) were used to estimate invertebrate standing crops in the channels. Because of the large differences in algal growth between the upstream and downstream sections, separate invertebrate standing crop estimates were calculated for each section.

Samples collected two, three and five weeks (13 July, 20 July, and 2 August) after initiation of channel watering were used to determine insect colonization rates prior to the ash impact. Samples collected on 2 August represented the density in the channels just prior to ash injection (4-5 August). Immediate effects of volcanic ash on the aquatic insects were assessed by comparing between-channel differences in densities prior to the ash introduction (2 August), with between channel differences occurring following the impact (5 August). This comparison is predicated on the assumption that any between-channel differences in the fauna was due solely to ash.

Recovery of the test channel during the two weeks following the ash impact was assessed by comparing invertebrate densities in the test and control channels on 11 August and 20 August. The canister samples collected from the center control channel (C_1) after 5 August and from all channels on 27 August were not included in these analyses because of channel alterations caused by primary production tests.

Insect drift sampling in the Mosquito Creek experimental channels was divided into three phases: 1) pre-impact; 2) ash impact, and

3) recovery/low-level ash impact. As the channels colonized, insect drift was sampled two weeks (12-13 July) and three weeks (19-20 July) after the channels had been watered. Half-hour drift samples were collected during six key times of the day over a 24-h period. After five weeks of colonization, the ash impact phase was initiated. For 24 h on 3-4 August, immediately prior to the simulated ash fall, and for 24 h during the ash introduction (4-5 August), insect drift was sampled at one-hour continuous intervals. Insect drift was also sampled one week (10-11 August) and two weeks later (19-20 August) during the recovery/-low-level ash introduction phase. Scheduling and duration of sampling was similar to the colonization phase, except that some additional half-hour drift samples were collected just prior to the six key times.

Drift net placement was similar during the colonization and recovery/low-level ash introduction phases, but differed on the days prior to, during, and immediately following the major ash fall. Insect drift into and from the channels was sampled simultaneously during colonization and recovery by placing nets at both the head and tail ends of each channel. Also, estimates of immigration or emigration at selected times necessitated the placement of nets at the downstream ends of the channels for a half hour before sampling both upstream and downstream drift. Samples collected during the three days associated with the simulated ash-fall were taken only at the downstream ends of the channels because the device used to introduce ash prevented placement of nets at the upstream ends.

The insect drift nets were designed to sample the entire width and water column of the channel. The nets consisted of a 60.96 x 30.48 x

80 cm nylon organdy net (pore size 275 microns) secured to a metal frame. During all tests, water discharge was held constant at $0.02\ m^3s^{-1}$.

Drift samples were preserved in 70% ethyl alcohol at the Mosquito Creek site and transported to the University of Idaho for laboratory sorting and identification. Insects were sorted from debris under 2x magnification. Samples were re-examined for quality control of sorted material and adjustments made in insect counts when appropriate. Since water discharge was identical during all sampling periods, drift sampling results could be expressed as insects per unit time.

3.3.7 Detrital Decomposition and Leaching: Norway maple *Acer Platanoides* L. leaves were collected on 4 July 1982 from three trees in a stand in Moscow Idaho, and cut into 3 cm diameter disks. The disks were dried at 40° C, weighed in approximately 0.75 g lots to the nearest 0.0001 g, and placed in 2 oz jars. The first part of the experiment consisted of one hundred jars treated in four equal groups and randomly placed 2 cm apart in a wooden rack. The entire rack was covered with nylon mesh (350 micron) to exclude most adult macroinvertebrates. The four treatments were: 1) a control, 2) 3.64 g/jar volcanic ash (comparable to the amount that fell in Moscow, Idaho (Hooper, et al., 1980)), 3) 7.28 g/jar volcanic ash, and 4) 3.64 g/jar soil, ashed and sieved through a 125 sieve. Ash and soil particle size distributions were given in Table 3. The second part of the experiment consisted of another 80 jars in 4 treatments of 20 jars each, placed randomly in a second rack. Forty of the jars were individually covered with 350 micron nylon mesh to exclude macroinvertebrates and the other 40 with 8000 micron nylon mesh to allow invertebrate colonization.

Half of the jars covered by each size mesh were treated with 3.64 g/jar ash, the other half were left as controls. Both racks of jars were filled with distilled water, the ash and soil allowed to settle 24 hrs prior to submerging the racks in a completely canopied pool area of Mosquito Creek on 11 July 1982. Five jars from each treatment were sampled at 15, 29, 57, and 85 days; the last 5 jars from each treatment in the 100-jar rack were sampled at 113 days. After sampling, the treatments covered with 8000 μ mesh were hand sorted under a disecting scope for macroinvertebrates. Macroinvertebrates were counted and identified. The contents of each jar was rinsed in clean water and the leaf disks dried at 40° C and weighed. Temperature, flow, dissolved oxygen, pH, hardness, alkalinity, Kjeldahl nitrogen, and conductivity were monitored during the study, though not continuously. Water exchange rate in the jars was determined from dye tests using spectrophotometer measurements of dry concentrations.

An additional 84 jars were used to determine leaching rates. The four treatments of 21 jars each were as follows: 1) a control, 2) 3.64 g/jar ash, 3) 7.28 g/jar ash, and 4) 3.64 g/jar soil. Jars were filled with distilled water treated with 1:10,000 HgCl to prevent microbial growth, capped, and placed in the stream. Three jars from each treatment were sampled daily for 7 days. After sampling, the contents of each jar was rinsed in clean water, dried at 40° C and weighed.

Decay rates for both leaching and decomposition experiments were determined using the model $X_t/X_o = e^{-kt}$ where X_t is the dry weight at time t, X_o is the initial dry weight, and k is the decay coefficient (Olson, 1963; Wieder and Lang, 1982). A simple linear regression was performed on log-transformed values of X_t/X_o with the intercept set to

Table 3. Particle size distribution of volcanic ash and Moscow, Idaho soil ashed and sieved through 125 micron - mesh sieve (Phi = $\log_2$ diameter (mm)).

		Total Sample	
Microns	Phi	Ash % Finer	Soil % Finer
1000.0	0.0	100.0	100.0
500.0	1.0	99.5	100.0
250.0	2.0	98.7	100.0
125.6	3.0	93.7	100.0
62.5	4.0	77.7	96.1
31.3	5.0	55.3	48.5
15.6	6.0	38.9	19.6
7.8	7.0	16.5	8.1
3.9	8.0	5.4	3.8
1.9	9.0	0.0	1.9
1.0	9.9	0.0	0.9

zero. Analysis of variance and a Duncan's multiple range test were performed on the data to determine whether or not there were differences in mean percent loss due to treatment. Log X_t/X_0 was analyzed as a function of treatment, number of days decomposed, and the crossed effects of treatment by number of days decomposed. The Statistical Analysis System (1979) was used for all statistical computations.

Chapter 4. Results

4.1 Palouse River

4.1.1 *Channel Hydraulic Geometry*: Cross-section locations of the Palouse River at the Graves Meadow site were shown in Figure 5. Flow parameters for riffle and run environments were reported in the previous chapter in Table 1.

Petrographic analyses of the core samples taken of streambed sediments at Graves Meadow showed that 2 to 4 percent volcanic glass became permanently incorporated into the silt-size fraction of the sedimentary units that were active at the time of the May 18th eruption. Only traces of Mt. St. Helens volcanic glass were found in the lower sedimentary units.

4.1.2 *Attached Benthic Algae*: Figure 14 presents a diagrammatic representation of the ABA community at the Graves Meadow site on the Palouse River. Riffles were dominated by the diatoms *Navicula*, *Gomphonema*, and *Cocconeis*; desmids (*Cosmaruim* and *Closterium*); the green *Ulothrix*; and jelly-like balls of the blue green *Nostoc* on rock surfaces. Rock surfaces in very high velocity areas such as the crests of small wateralls, were often covered with thick growths of *Monostroma*, and *Vaucheria*. The bottom areas of runs were dominated by clumps of *Vaucheria* on rocks or *Callitriche*. Rock and log surfaces in pool areas commonly were covered with the mosses *Fontinalis* and *Scorpiduim* and the algae *Riccicarpus*. The bottom areas of runs were dominated by clumps of *Vaucheria* on rocks or *Callitriche*. Pool substrates varied from fine particulate organic matter to unimbedded small cobbles. Riffle

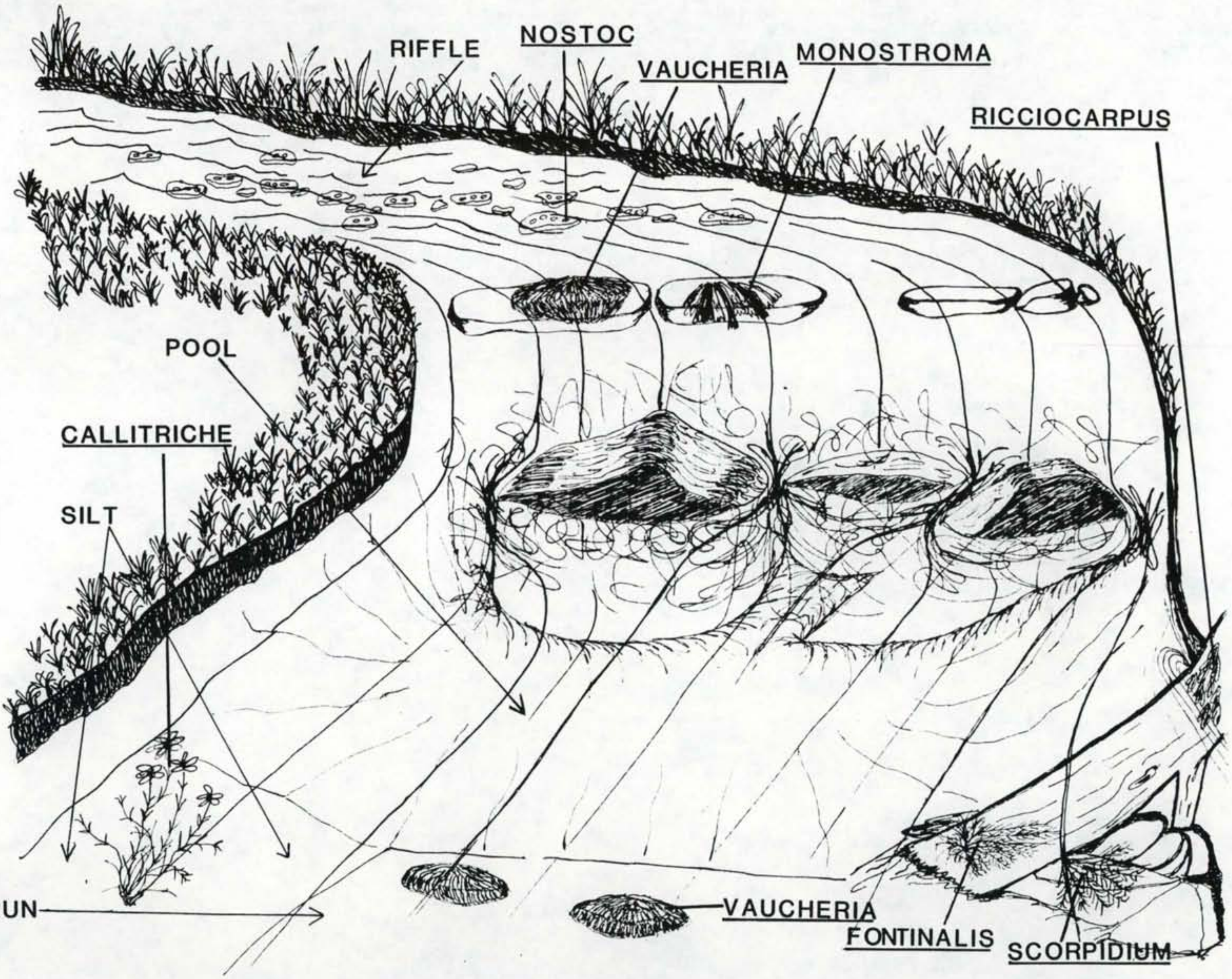


Figure 14. Overview of Attached Benthic Algae.

substrates were predominantly mixed cobbles. Run substrates were intermediate between riffles and pools.

Organic weight, chlorophyll "a", and autotrophic indices used to quantify the ABA communities were pooled by riffle, run, or pool locations (Table 4). Organic weights were highest in pool habitats but were significantly different ($P \geq 0.1$) from riffles or runs only in October samples. Mean pool organic weights were 4473 and 7428 mg m^{-2} in August and October, 1981; 3750 and 2065 mg m^{-2} in riffles, and 3462 and 5974 mg m^{-2} in runs.

Chlorophyll "a" was not significantly different between pools, riffles, and runs. Pooled August and October samples were 10.3, 5.6, and 5.3 mg m^{-2} for pools, riffles, and runs, respectively.

Autotrophic indicies (AI) were not significantly different between pools, riffles and runs, but pool means were usually higher than riffle and run means. In August, the pool means AI was 1066 compared to 574 and 595 for the riffle and run samples. In October, pool mean AI was 817 compared to 383 and 838 for riffles and runs. Pooled values were 932, 502 and 686 for pools, riffles, and runs. The higher AI in pools and runs reflects greater deposition of detrital organic matter in the deeper more quiescent waters. Despite the lack of statistically significant separation of the ranges, we believe that at least later in the season, riffle AI's were, in fact, lower than pool and run AI values. This was due to higher organic weights in these latter areas, not higher chlorophyll weight in the riffles.

4.1.3 Benthic Insects: The macroinvertebrates collected during the summer, 1981, from the Upper Palouse River could not be empirically related to Mt. St. Helens volcanic ash fallout since baseline data on

Table 4. Organic weights, chlorophyll "a", and autotrophic indices from Graves Meadow, North Fork of the Palouse River, 1981.

	mg Org. Wt./m^2			mg Chl"a"/m^2			Autotrophic Index		
	Pool	Riffle	Run	Pool	Riffle	Run	Pool	Riffle	Run
August 1981									
n	6	5	10	8	6	10	6	5	10
$\bar{x}$	4472.6	3749.7	3461.5	3.35	6.17	5.83	1066.1	574	595.1
s	4351.8	3115.1	3020.6	3.23	7.02	5.21	955.8	375.9	313.5
C.I. 90%	893-8053	780-6720	1711-5212	1.2-5.5	0.4-11.9	2.8-8.9	280-1852	216-932	413-777
October 1981									
n	10	7	10	7	3	6	7	3	6
$\bar{x}$	7428	2065	5974	18.3	4.5	4.3	817	383	838
s	4949	1966	5517	17	3.3	3.3	649	283	624
C.I. 90%	4559-10,297	621-3509	2776-9172	5.8-30.8	0-10.1	1.6-7	340-1294	0-860	325-1351
Aug.+Oct. Pooled									
n	16	12	20	15	9	16	13	8	16
$\bar{x}$	7120	2767	4718	10.3	5.6	5.3	932	502	686
s	5049	2528	4517	13.7	5.9	4.5	780	337	451
C.I. 90%	4907-9333	1456-4078	2972-6464	0-21.1	1.9-9.3	3.3-7.3	130-1735	276-728	488-884

the proportion of fine sediments in the streambed substrates and benthic invertebrate composition were not available prior to ash fallout. The reported results, therefore, relate to the composition and distribution of the invertebrate fauna with inferences directed at assessing general health and habitat quality one year subsequent to the ashfall. The 144 samples collected randomly from the riffles, runs and pools at Graves Meadow yielded a diverse macroinvertebrate fauna of 67 insect species excluding Chironomidae, 24 genera, genera-groupings, or tribes of Chironomidae, and three phyla of non-insect invertebrates. The benthic invertebrate standing crop at the time of sampling averaged 3010/m^2 and 2.8 g/m^2 dry weight biomass.

Distinct species-habitat relationships were evident among the insects in the study area (Table 5). Chironomidae (Diptera) were most abundant in the runs and pools -- Ephemeroptera and Trichoptera were most abundant in the riffles. However, two mayflies (*Centroptilum* sp. and *Paraleptophlebia bicornuta*) and one midge (*Eukieffierella* sp. 3) showed an opposite trend. A particularly dramatic difference in habitat preference occurred between the baetid mayflies, *Baetis tricaudatus* and *Centroptilum* sp. Of the 327 *B. tricaudatus* collected, 324 were from the riffles. By contrast, of the 113 *Centroptilum* found among 51 sites, only one specimen was collected from a riffle site.

Microhabitat affinities within major habitats were also noted. Riffles with cobbles had relatively fewer elmid riffle beetles (327/m^2 vs. 642/m^2) and more net-spinning caddisflies, *Wormaldia* sp. (449/m^2 vs. 642/m^2) than the cobble-free sites. The blackfly, *Simulium* sp. and the mayfly *Serratella teresa* were virtually restricted to riffles having macrophytes. Among the common Chironomidae inhabiting the run

Table 5. Relationships between habitat, distribution and density of common (10/m^2) Palouse River macroinvertebrates July, 1981. Total number of observations are 47 riffles, 48 runs and 48 pools.

	Density (no./mo^2)			Number of Observation		
Faunal-Habitat Affinities	Riffles	Runs	Pools	Riffles	Runs	Pools
Riffle Fauna						
Simumlium sp.	77	0	1	19	0	1
Eukieffierella sp. 3	29	0	0	10	0	0
Epeorus albertae	32	3	0	20	2	0
Baetis tricaudatus	270	2	0	39	2	0
Serratella teresa	62	3	0	18	1	0
Serratella tibialis	57	0	0	25	0	0
Wormaldia sp.	375	2	2	38	1	2
Rhyacophila vaccua	42	2	2	24	2	2
Glossosoma sp.	81	1	0	28	1	0
Riffle-Run Fauna						
1° Riffle						
Skwala sp.	123	31	2	40	19	2
Baetis hageni	82	7	0	26	8	0
Non-specific						
Heptagenia criddlei	32	16	1	17	11	1
Drunella grandis	17	12	1	13	11	1
Riffle-Run-Pool Fauna						
1° Riffle						
Chloroperlidae	51	11	14	26	9	5
1/ Riffle-Run-Pool gradiant						
Optioservus seriatus	342	117	15	43	35	10
Zaitzevia parvula	116	22	7	33	12	4
1° Riffle and run						
Eukieffierella sp. 1	46	42	14	17	14	4
Diamesinae	42	42	8	22	22	7
Ameletus sparsatus	32	21	5	16	15	3

1/ Importance of habitat declines from riffle to run and from run to pool.

Table 5. (continued)

Faunal-Habitat Affinities	Density (no./mo^2)			Number of Observation		
	Riffles	Runs	Pools	Riffles	Runs	Pools
Non-Specific						
Oreodytes angustior	21	30	17	14	14	13
Trichocladius sp. 2	12	10	12	8	10	6
Cricotopus spp.	131	184	162	30	36	26
1° Run and Pool						
Hexatoma sp	23	107	64	16	37	29
Tanypodinae	39	152	136	26	37	28
Tanytarsini	211	1185	1530	37	42	43
Paralebtophlebia bicornuta	20	52	57	11	18	18
1° Pool						
Palpomyia sp	10	8	36	9	9	19
Oligochaeta	17	22	64	13	16	20
Run-Pool Fauna						
Non-Specific						
Cricotopus sp. 1	7	265	305	5	22	29
Paratendipes sp	1	20	14	1	12	8
Tribelos sp	4	127	125	4	15	26
Psectrocladius sp. 1	0	130	231	0	26	32
Phaenopsectra sp.	0	40	79	0	19	21
Centroptilum sp.	1	35	58	1	21	29
1° Pool						
Heterotrissocladius sp. 1	1	130	670	1	26	39
Sigara sp	1	7	52	1	2	8
Sphaeriidae	2	9	31	2	6	14

and pool sites, the densities of Tanypodinae, Cricotopus spp., and Psectrocladius sp. at sites with algal mats, were more than 3 times greater than of sites without an abundance of algae.

4.2 Laboratory Simulation of Sediment Transport

4.2.1 Laboratory Flume Studies: Turbidity was correlated with suspended sediment concentrations (r^2=0.715; n=23). Linear regression produced the formula:

$$C = 3.3N + 83$$

where C is the concentration of ash in mg/L; and N is the turbidity reading in NTUs. The formula may be restricted to the range of 20 - 100 NTUs, the range of values tested.

The critical tractive force for an aqueously deposited bed of May 18, 1980 Mt. St. Helens plume ash was found to be 0.03 lbf/ft^2 (1.4 Pascals). This value agrees quite closely with value for lean clays in a loosely packed bed found in Russian studies as reported by Chow (1959).

4.2.2 Bulk Removal of Suspended Sediment From Flow: There was no statistically signficant difference between concentration of suspended sediment near the surface and near the bed in the test flume (t=-0.094; d.f.=19; paired comparison t-test). This finding is not surprising due to the large variability in the samples and the fact that the differences in sample depth was only four inches (102 mm).

The concentration of suspended sediment at the end of the test flume was reduced 43% to 62% (95% confident interval standard error of the mean) over the concentration of sediment at the head end of the flume. This agrees quite well with the values predicted from the laboratory test for the bulk removal of ash given the flow conditions (50% to 56%).

4.2.3 Insects: A reduction of leaf processing, as expressed by consumption and Consumptive Index, in the test vessels by Hesperophylax occidentalis compared to the control vessels was highly significant ($p<0.01$) for the two experiments involving different ash-concentration levels (Table 6, Fig. 15). Average percent reduction in CPOM conversion was not as great in the first test (40% to 56%) where the suspended ash concentrations were approximately half that of the second test, indicating that inhibition of feeding became more pronounced as ash concentration increased. The weights of the insects remained fairly constant or showed slight average losses during the course of the trials with no appreciable weight-loss differences between the test and control insects in either trial (Fig. 16).

4.3 Laboratory Simulations - Mosquito Creek

4.3.1 Petrographic Analyses of Deposited Tephras: Petrographic analyses were run to determine the composition of the volcanic tephras that became incorporated into the silt-size sediment fraction. The sediment fraction finer than 200 mesh (0.003 inch; 74 microns) for the samples following the Mosquito Creek simulation were optically analyzed to determine the amount of Mt. St. Helens volcanic glass that had been added to the reach. Since particles of Mazama ash existed in the Lighting Creek drainage (source of the substrate) the pre-simulation Mosquito Creek sediment samples were also petrographically analyzed to determine the percent of Mazama ash in the fine sediment fraction. These data were combined with the petrographic data from the Graves Meadow site.

It was found that there was a 5.3 to 7.6 percent increase in the material finer than 0 phi (0.039 inch; 1 mm) immediately after the simulation (95% confidence interval of difference of independent means;

Table 6. Consumption and consumptive index for *Hesperophylax occidentalis* maintained for 14-d in control chambers with no ash and test chambers with suspended ash (2 specimens/replicate; C=control chambers, T=test chambers).

	Experiment 1 - Test levels: 1-1.5 g/l Ash				Experiment 2 - Test levels: 2-3 g/l Ash			
	Consumption mg AFDW/ Animal/day		Consumptive Index mg AFDW/consumption/ mg animal AFDW/day		Consumption mg AFDW/ Animal/day		Consumptive Index mg AFDW/consumption/ mg animal AFDW/day	
	C	T	C	T	C	T	C	T
	1.5	0.8	0.10	0.05	1.1	0.4	0.08	0.02
	1.5	0.9	0.07	0.04	0.9	0.5	0.08	0.04
Individual	1.1	1.0	0.10	0.06	1.5	0.5	0.11	0.03
Replicates	1.7	1.1	0.09	0.07	1.2	0.7	0.10	0.05
	2.1	1.2	0.10	0.07	1.4	0.7	0.10	0.06
$\overline{X}$	1.55	0.94	0.094	0.056	1.20	0.54	0.096	0.042
$(\overline{X}_c-\overline{X}_t) \pm$ 95% C.I.	0.61 ± 0.39		0.038 ± 0.017		0.66 ± 0.29		0.054 ± 0.024	
t	3.66		5.39		5.29		5.21	
p	.006		.0007		.0007		.0008	

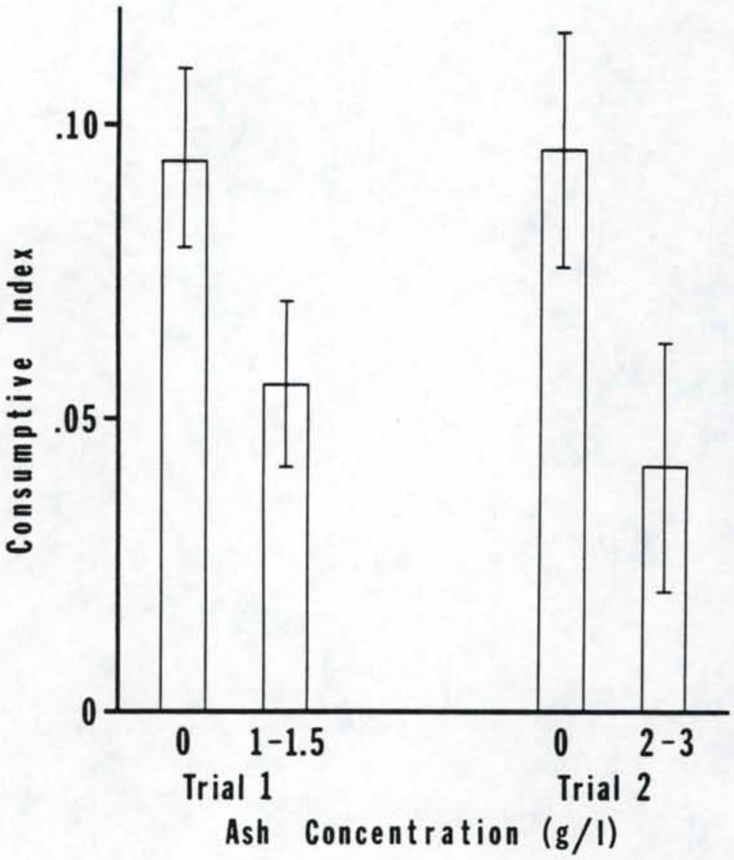


Figure 15. Means and 95% C.I. (vertical lines) of consumptive index (mg AFDW consumption/mb animal AFDW/day) by Hesperophylax occidentalis placed in control chambers with no ash and test chambers with ash concentrations of 1-1.5 g/l and 2-3 g/l (2 specimens per chamber and 5 chambers per concentration for each trial).

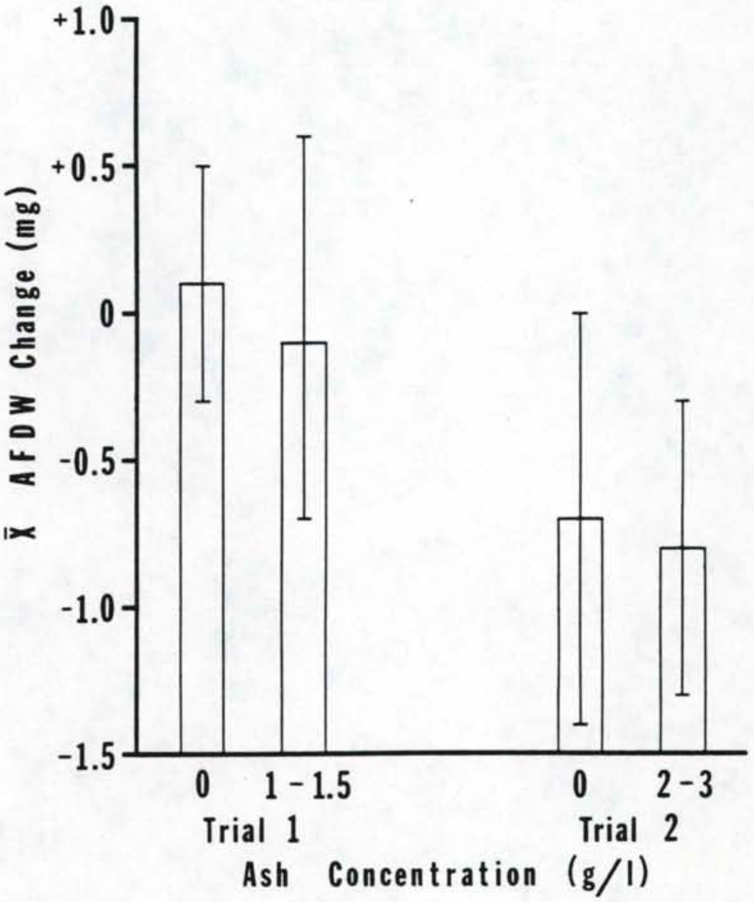


Figure 16. Means and 95% C.I. (vertical lines) of change in AFDW of Hesperophylax occidentalis maintained for 14 days in control chambers with no ash and test chambers with ash concentrations of 1-1.5 g/l and 2-3 g/l (2 specimens per chamber and 5 chambers per concentration for each trial).

M=6.4; Sd=0.56; d.f.=20; t=2.086). The total increase of sediment in the channel was estimated to be between 225 to 326 lbm (102 to 148 kg). The 95% confidence intervals for the standard error of the mean for the percent finer by weight were plotted for the pre- and post simulation grain-size distributions (Figure 17).

4.3.2 Sediment Transport Model Verification: The laboratory model developed to predict the amount and distribution of ash material incorporated into a stream reach was applied to the Mosquito Creek simulation. The model predicted 449 lbm (204 kg) of ash would be incorporated into the channel with the distribution shown in (Figure 17). The model over predicted the amount of material incorporated into the bed. The predicted grain-size distribution deviated outside the observed distribution for value between 3 phi and 8 phi (.0025 to .00016 inch; 63 um to 4 um). The maximum difference between the predicted percent finer values and values outside the range of the observed values was 1.0%.

4.4 Algal Growth in Stream Channels and Mosquito Creek

Consistently high water quality of Mosquito Creek rendered it an ideal water source for experimental work. Through June-September of 1982, temperature ranged from 6.0-8.1° C, oxygen from 10.0-11.5 mg l^{-1}, and flow volume 5.8-10.0 cfs. Chemical water quality as expressed by pH (7.7-7.9), conductivity (112-128 mho cm^{-2}) and total hardness (50-82 mg l^{-1}), and total alkalinity (60-99 mg l^{-1}) reflected the geochemically diverse watershed, even in the small Mosquito Creek drainage. Nutrient concentrations were low, with NO_3 N concentrations of ≤ 0.05 mg l^{-1} total kjeldahl N 0.18-1.23 mg l^{-1}, and total phosphorus < 0.01 mg l^{-1}.

Community development of attached benthic algae in the three channels was markedly different over the July-August growth period. An

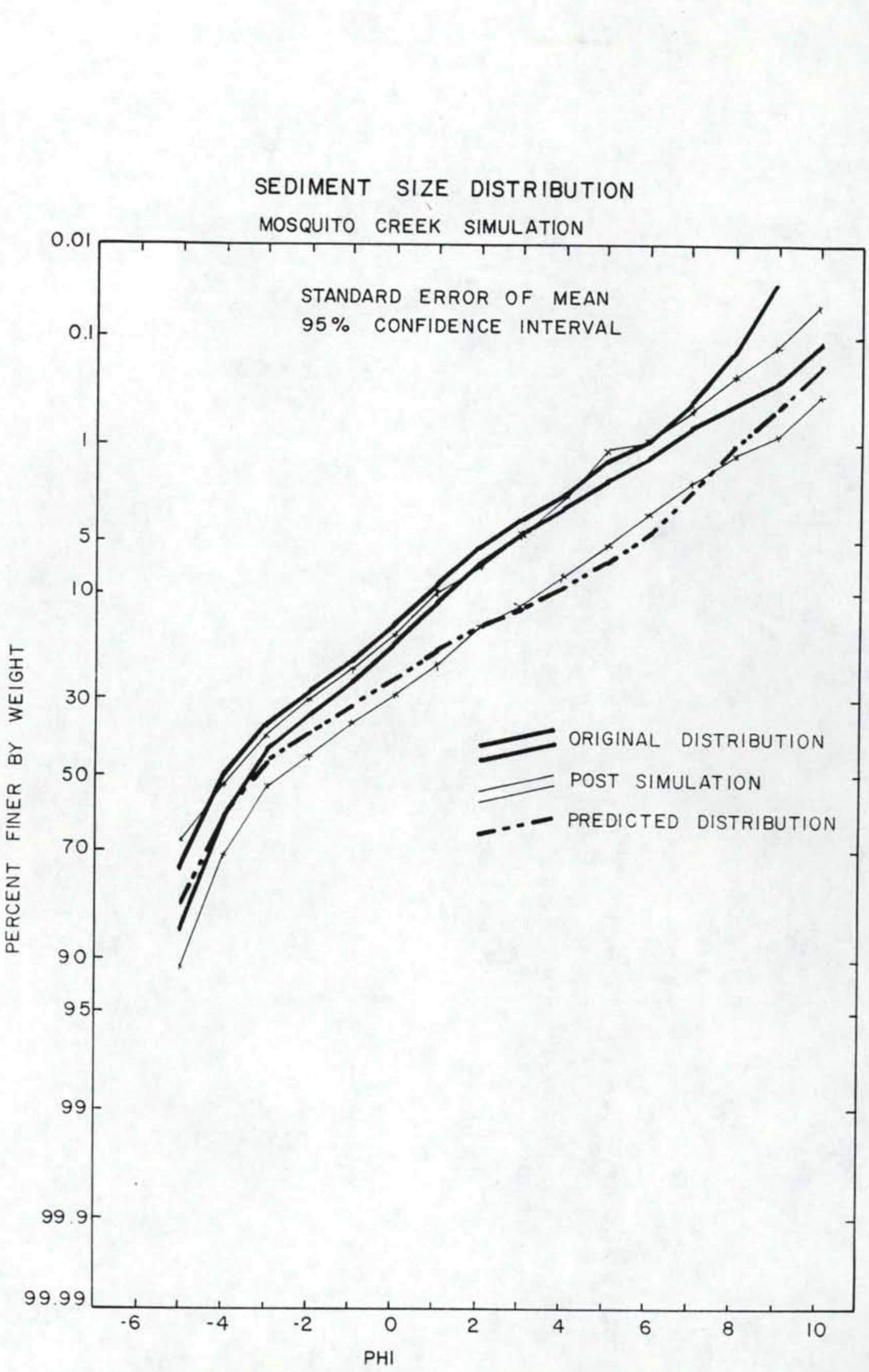


Figure 17. Sediment size distribution in Mosquito Creek simulation channels.

unexpected development was the difference between upper and lower sections of each channel (Table 7). The upper/lower difference was especially evident in the treatment channels where ash deposition provided good growing conditions for the algal dominant Hydrurus. This filamentous clumped form dominated lower channel ABA and was, in fact, the prinicpal source of variability between replicates. It was especially heavy in the lower sections of the low-level loading channel, peaking at 82.1 mg m^{-2} chlorophyll "a" on 11 August (Tables 7 and 8). At that time, the lower/upper ratio was 5.78. This phenomenon of increased growth in the lower channel sections was also seen in the control channel where the summer-long average of all lower/upper ratios was 2.17. Summer-long lower/upper ratios in the catastrophic and low-level loading channels were 1.74 and 2.29 respectively (Table 8).

Biomass as indicated by chlorophyll "a" peaked in all channels by the second week of August, then declined before a slight increase in late August. The decline was due to sloughing of these heavy growths after a population peak in mid-August.

Even though both monochromatic and trichromatic chlorophyll "a" were measured throughout the study (Figures 19 and 20), only trichromatic results will be discussed. The lower variation of trichromatic results better describe ABA responses.

Algal growth as indicated by chlorophyll accumulation was consistently higher in the experimental channels than in adjacent Mosquito Creek (Figure 20). Channel water velocity was slower, but we believe the biggest factor in the approximately 400% increased algal growth was increased light in the control channels from a smoother water surface and vegetation clearing around the channels. Control channel algae

Table 7. Composition of attached benthic algae in the Mosquito Creek experimental stream channels in July - August, 1982. Up to five dominant genera listed in decreasing order of abundance.

Treatment	Channel Section	DATE					
		13 July	20 July	2 Aug	5 Aug	11 Aug	20 Aug
Control	Upper		*Fragilaria* *Navicula* *Melosira* *Ceratoneis* *Cymbella*		*Hydrurus* *Melosira* *Frugilaria* *Ceratoneis* *Cymbella*	*Melosira* Unk. Fil. Green *Ceratoneis* *Navicula* *Fragilaria*	
	Lower	*Ulothrix* *Ceratoneis* *Fragilaria* *Synedra* *Navicula*		*Ceratoneis* *Fragilaria* *Melosira*			*Navicula* *Fragilaria* *Cymbella* *Ceratoneis* *Melosira*
Catastrophic Loading	Upper				*Ceratoneis* *Fragilaria* *Navicula* *Melosira* *Cymbella*		*Navicula* *Ceratoneis* *Cymbella* *Melosira* *Ulothrix*
	Lower	*Melosira* Unk. Fil. Green *Fragilaria* *Navicula* *Ceratoneis*	*Ceratoneis* *Fragilaria* *Navicula* *Melosira* *Synedra*	*Ceratoneis* *Fragilaria* Unk. Fil. Green *Melosira* *Ulothrix*		*Hydrurus* *Melosira* *Fragilaria* *Asterionella* *Ceratoneis*	
Low Level Loading	Upper	*Ulothrix* Unk. Fil. Green *Ceratoneis* *Navicula*				*Hydrurus* *Ceratoneis* *Fragilaria* *Navicula* *Cymbella*	
	Lower		*Melosira* *Fragilaria* *Ceratoneis* *Navicula* Unk. Fil. Green	*Hydrurus* *Ceratoneis* *Melosira*	*Hydrurus* *Ceratoneis* *Ulothrix* *Fragilaria* *Melosira*		*Hydrurus* *Ceratoneis* *Melosira* *Navicula* *Ulothrix*

Table 8. Trichromatic chlorophyll "a" (mg m^{-2}) in the Mosquito Creek experimental stream channel in July - August, 1982. Channel values split into upper and lower channel sections. Values are means of five replicates.

Treatment	Channel Section	DATE							
		13 July	20 July	2 Aug	5 Aug	11 Aug	20 Aug	27 Aug	Mean
Control	Upper	1.8	5.7	20.4	13.1	19.3	18.2	10.3	12.7
	Lower	1.9	5.0	49.7	50.5	53.6	13.7	35.3	30.0
	Lower/Upper	1.06	0.88	2.44	3.85	2.78	0.75	3.43	2.17
Catastrophic Loading	Upper	1.9	4.8	13.9	18.3	17.3	4.8	6.7	9.7
	Lower	2.0	5.3	47.8	23.2	32.1	12.3	5.8	18.4
	Lower/Upper	1.05	1.10	3.44	1.27	1.86	2.56	0.87	1.74
Low Level Loading	Upper	1.7	6.5	18.6	14.4	14.2	4.6	12.6	10.4
	Lower	2.2	5.8	35.3	35.0	82.1	9.5	21.2	27.3
	Lower/Upper	1.29	0.89	1.90	2.43	5.78	2.07	1.68	2.29

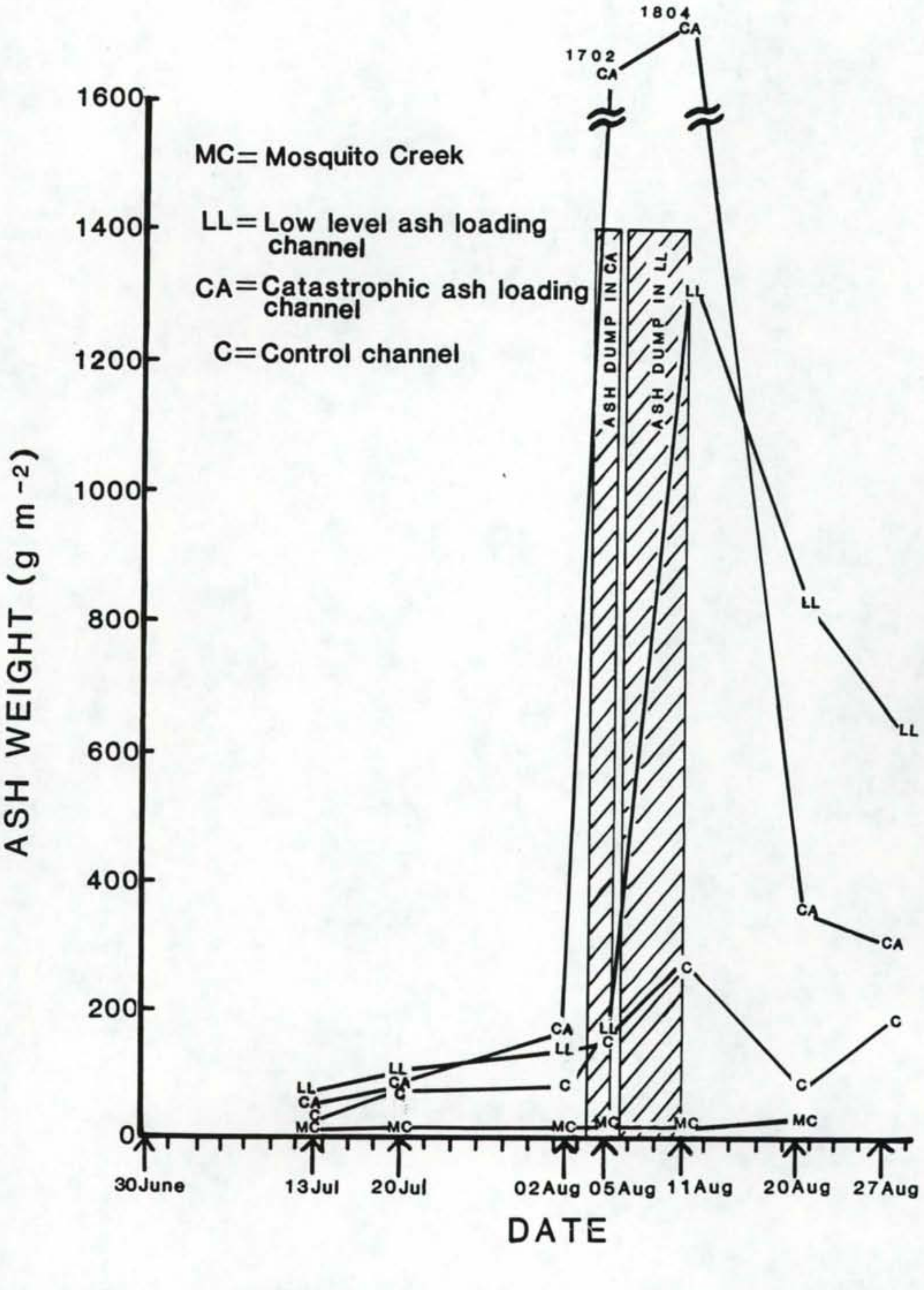


Figure 18. Ashed weight of attached benthic algae in the Mosquito Creek stream channels and in Mosquito Creek, in relation to a catastrophic volcanic ash dump and a chronic volcanic ash dump. June - August, 1982

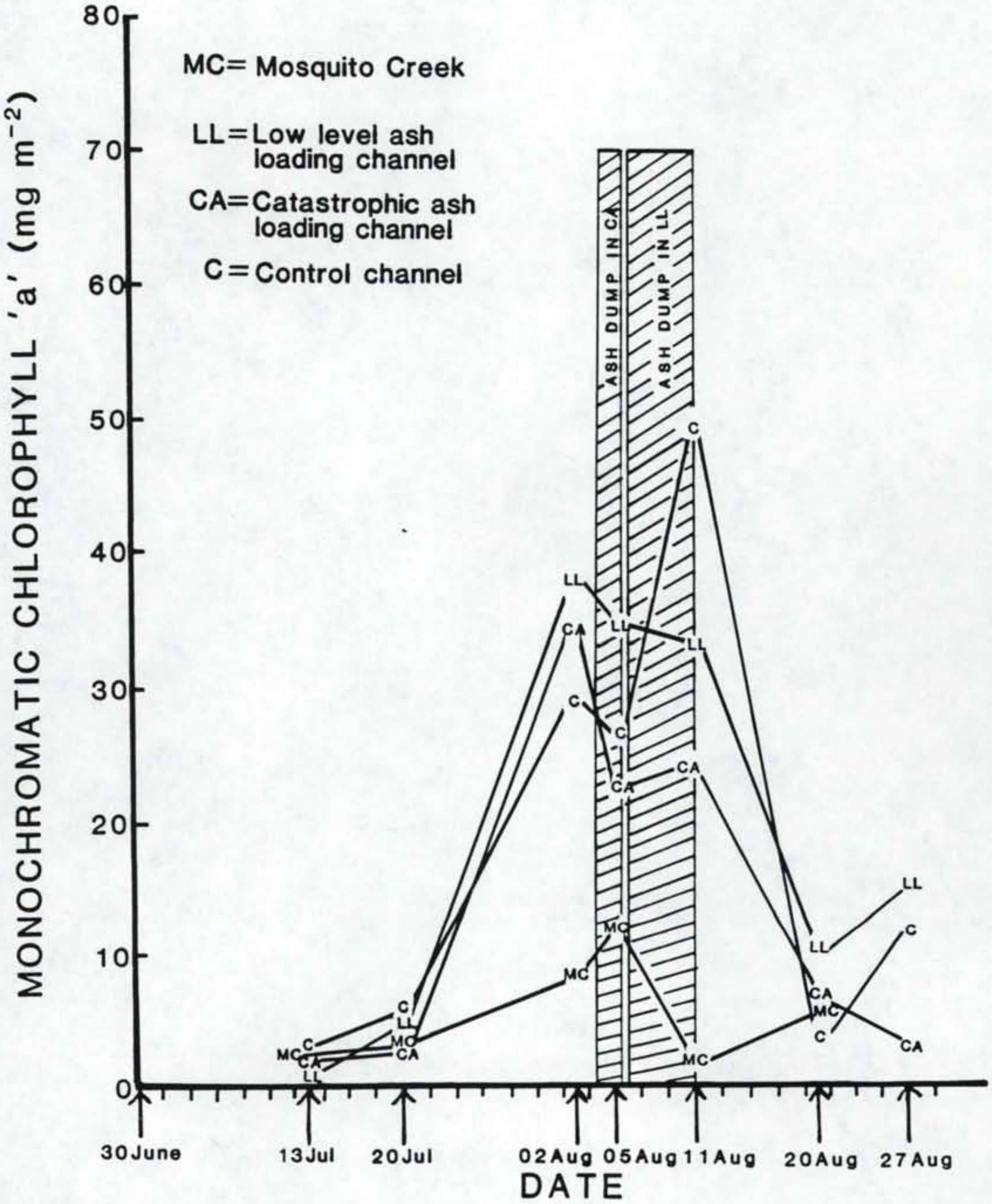


Figure 19. Monochromatic chlorophyll "a" (mg m^{-2}) of attached benthic algae in the Mosquito Creek stream channels in relation to a catastrophic ash dump and a chronic ash dump.
June - August, 1982

averaged 18.6 mg m^{-2} trichromatic chlorophyll "a" while Mosquito Creek averaged 4.7 mg m^{-2} through the summer.

High variability was encountered between replicates. We took five replicates per treatment, but two factors increased sample variance. Our use of natural rock substrates gave more applicability to natural streams but did contribute to high variance. Secondly, the very thick growths of long clumps of Hydrurus in the lower channels contributed to high variance, and the situation was worsened by a sloughing period in midsummer.

4.4.1 Catastrophic Ash Loading Effects on ABA: The ash loading regime in Table 2 produced a catastrophic ash loading event which continued over a 24 hour period. Resulting ash content of the ABA increased in the CA channel from 185 g m^{-2} before to 1702 g m^{-2} immediately after the loading event. Control channels showed no increase. By 15 and 22 days after the event, ash content had dropped back to 400 g m^{-2} in the CA channel (Figure 21).

Response of the ABA community is shown by chlorophyll and ash-free weight plots of Figures 19, 20, and 21. After the catastrohic ash loading in the CA channel, chlorophyll in that channel dropped within 24 hours to a point 25% lower than the two control channels at that time (channel L1 had not yet received ash loading). After 6 more days, the impacted channel recovered slightly but was now 32% below the much increased control channel. After a total of 15 days, both channels were similar. Only the 6 day negative impact of the dump was significant at the 90% level.

Pooled before-after data show a fairly clear negative impact on ABA chlorophyll "a" by the catastrophic ash loading (Figure 22). Despite

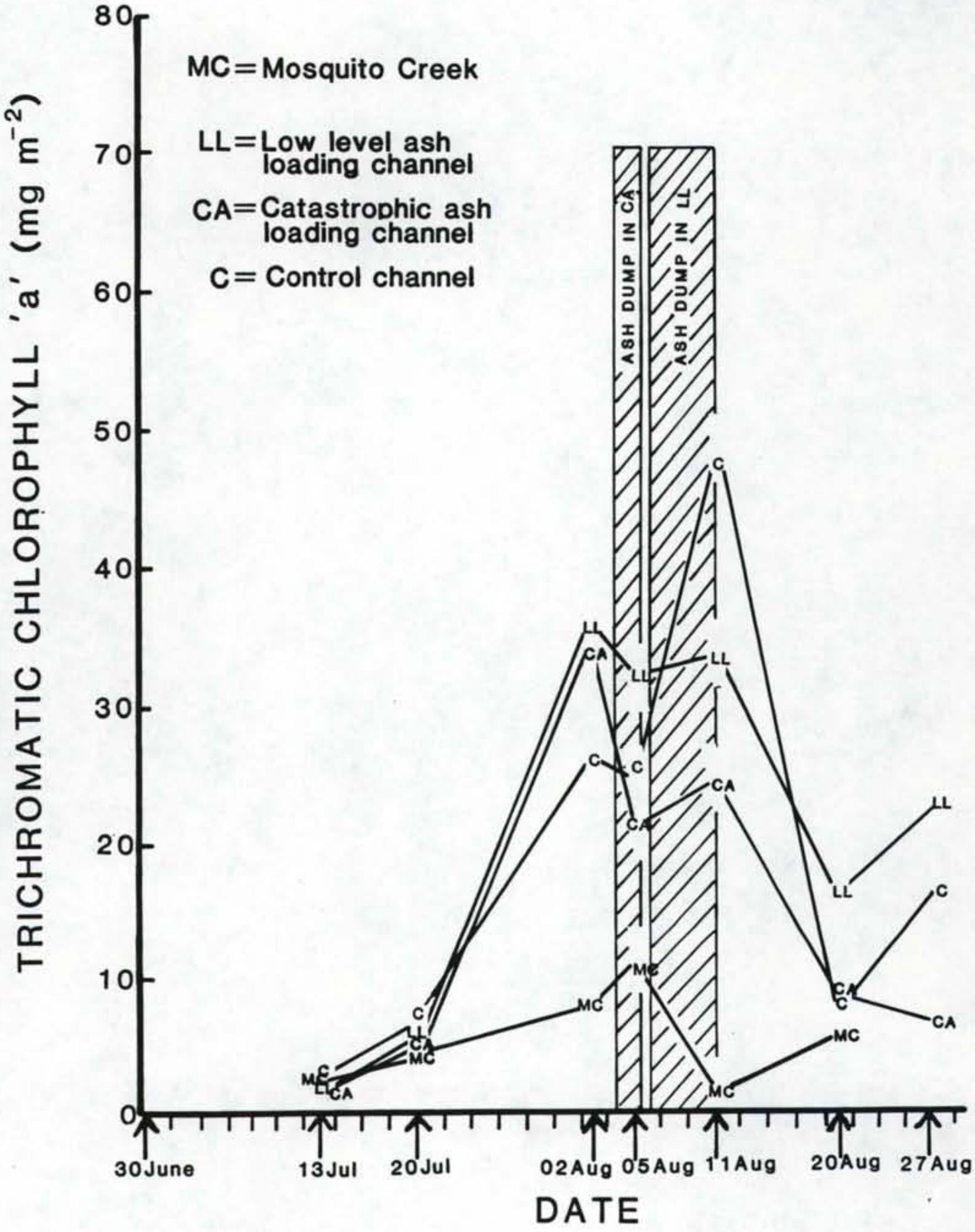


Figure 20. Trichromatic chlorophyll "a" (mg m^{-2}) in the Mosquito Creek stream channels in relation to a catastrophic ash dump and a chronic ash dump. June - August, 1982

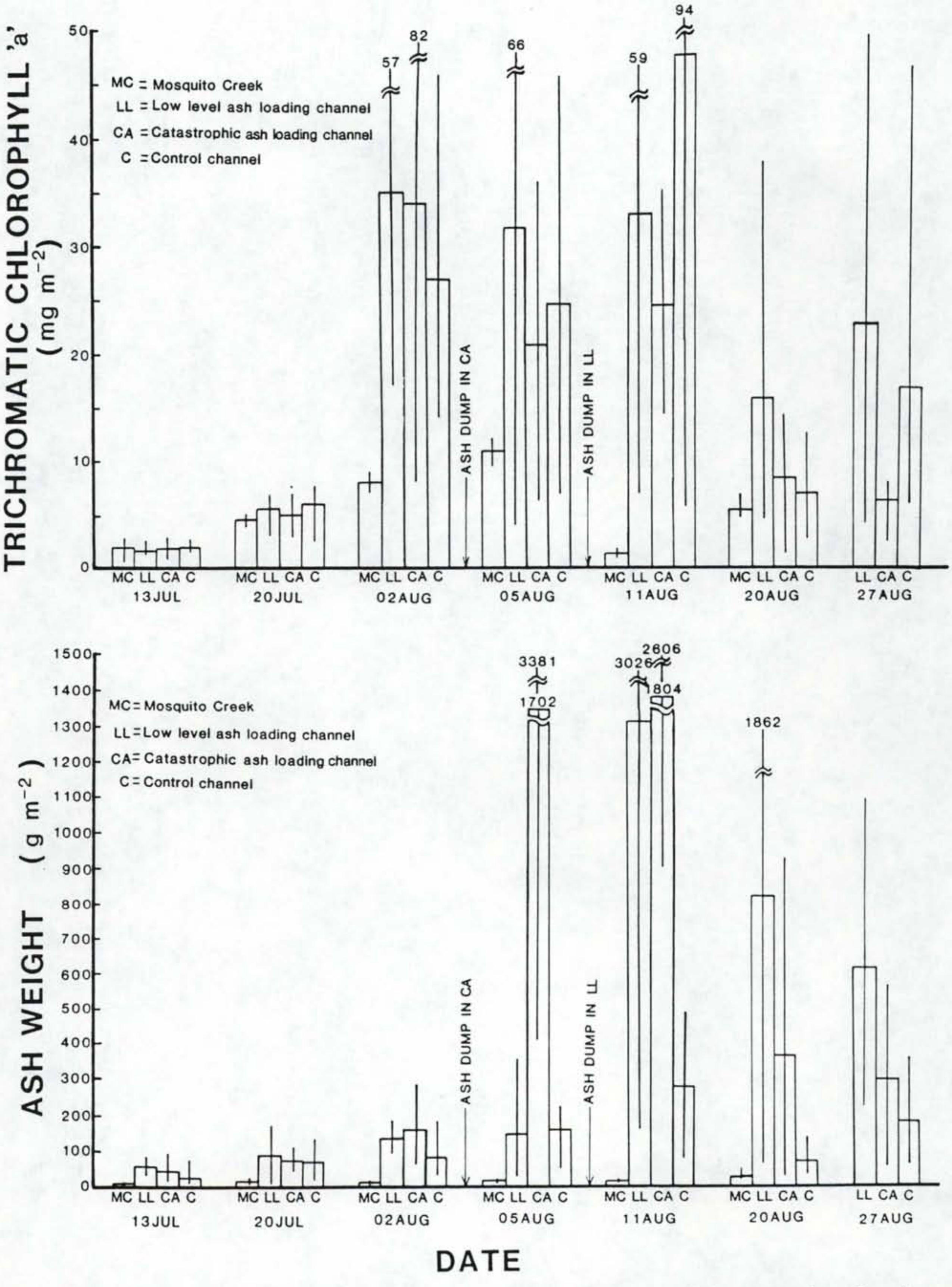


Figure 21. Response of attached benthic algae in the Mosquito Creek channels to catastrophic and low level ash loading, July-August, 1982.

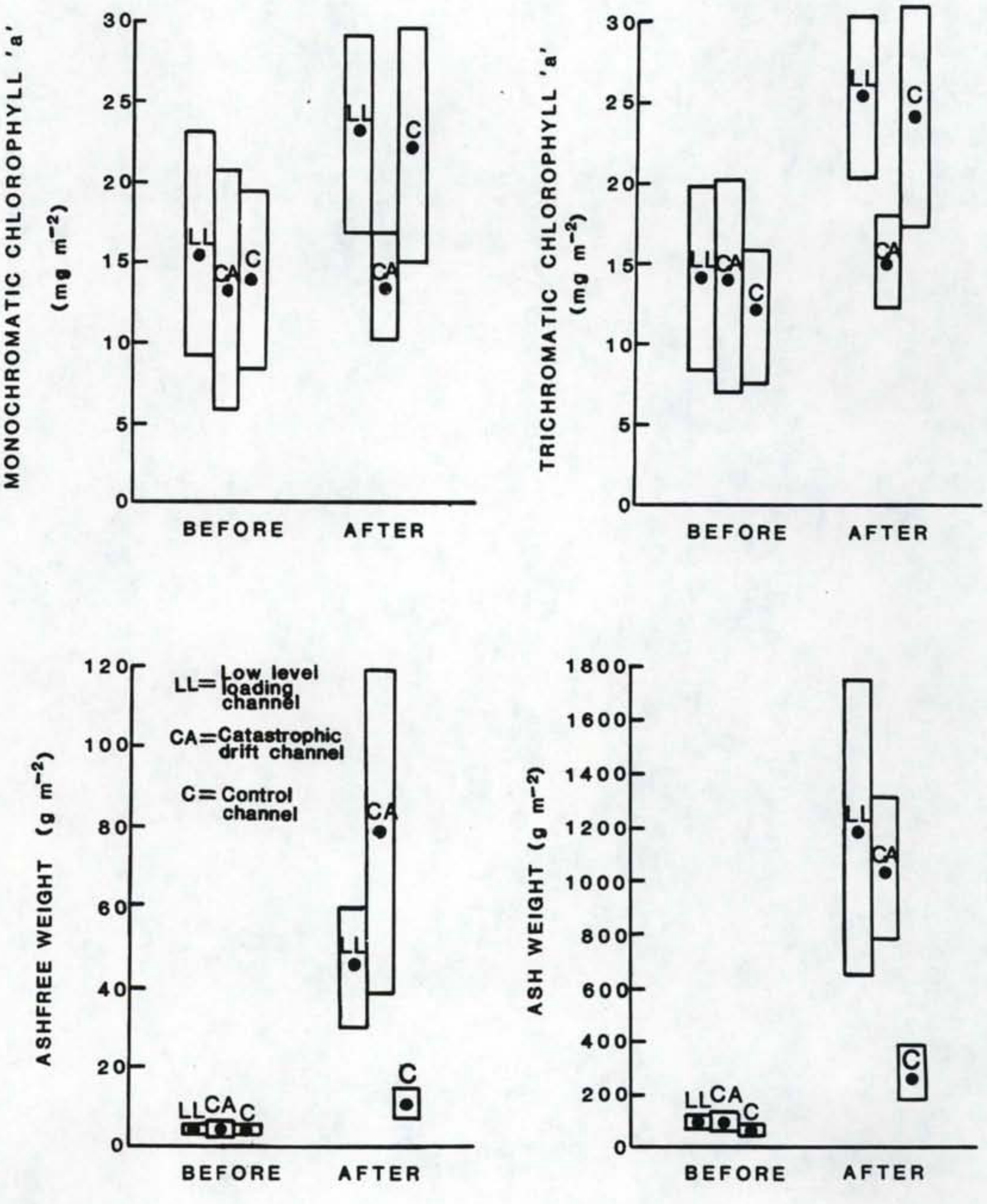


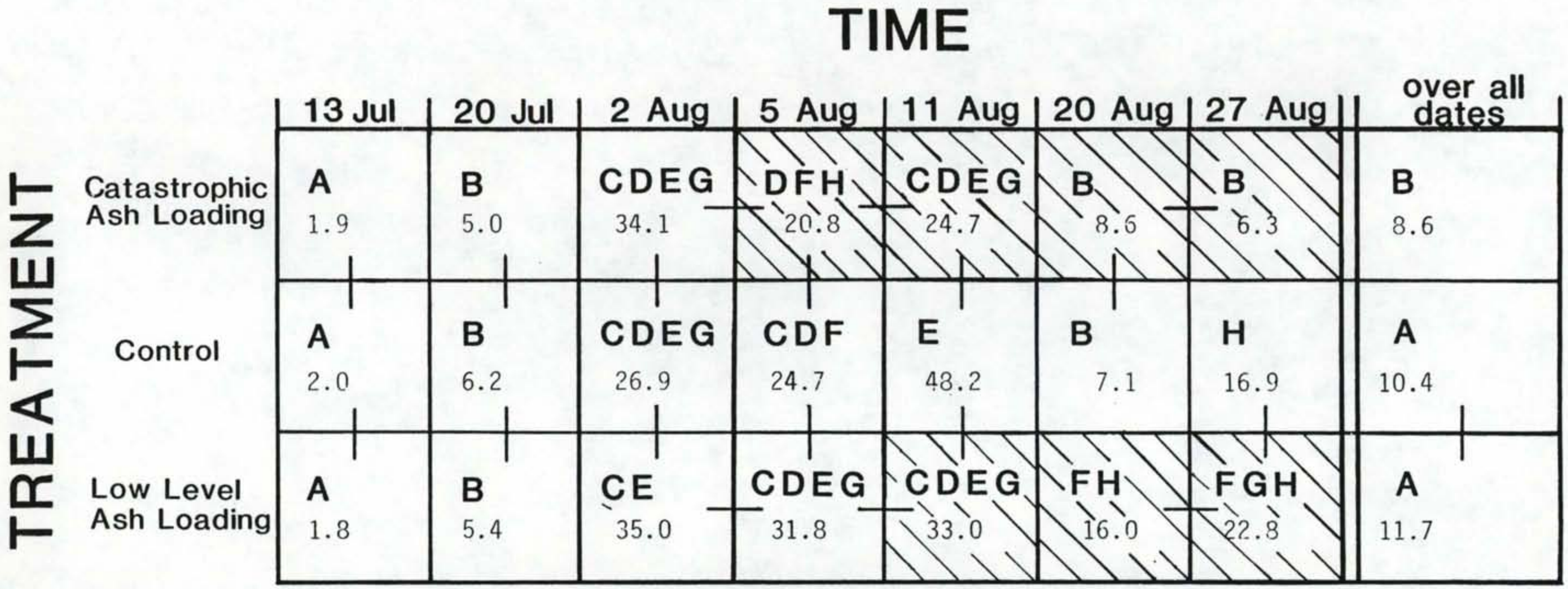
Figure 22. Measures of attached benthic algae and ash weight in the Mosquito Creek channels. All dates and replicates pooled into "before" and "after" ash loading periods.

the 38% lower chlorophyll of the pooled catastrophic ash loaded channel, the very slight overlap of data ranges caused statistical nonsignificance. Nevertheless, we feel that the apparent differences between means are probably real.

4.4.2 Low Level Ash Loading Affects on ABA: Low level ash loading resulted in an increase of ABA ash content from 170 g m^{-2} to 1300 g m^{-2} after five days (Figure 18). By 9 and 16 days after cessation of loading, ash content of the ABA dropped back to 820 and 690 g m^{-2}, respectively. Control ash content increased to 290 g m^{-2} immediately after the low level loading and declined after cessation of loading. The slight increase may have been due to leakage from the adjacent test channel. Mosquito Creek ash content showed no change during the entire pre- or post-test period.

Chlorophyll content of the ABA increased nearly 100% in the control channel during the loading period as the stream approached its annual high productivity. The low level loading channel meanwhile increased only 6%. This increase was significant at the 90% level (Figure 23). By 15 and 22 days after cessation of loading, the low level loading channel had stabilized at chlorophyll levels 50-100% higher than the control channel (but no significance at 90%).

Control ABA chlorophyll "a" ranged from 3 mg m^{-2} early in the summer to 50 mg m^{-2} by mid summer, down to 8, and finally back up to 18 mg m^{-2} by late summer. The biomass decline in mid-simmer was apparently caused by sloughing of large clumps of attached algae which accumulated in the lower ends of the channels. These clumps were the principle cause of our high within-treatment variability despite five replicates.



TREATMENT	13 Jul	20 Jul	2 Aug	5 Aug	11 Aug	20 Aug	27 Aug	over all dates
Catastrophic Ash Loading	A 1.9	B 5.0	CDEG 34.1	▨ DFH 20.8	▨ CDEG 24.7	▨ B 8.6	▨ B 6.3	B 8.6
Control	A 2.0	B 6.2	CDEG 26.9	CDF 24.7	E 48.2	B 7.1	H 16.9	A 10.4
Low Level Ash Loading	A 1.8	B 5.4	CE 35.0	CDEG 31.8	▨ CDEG 33.0	▨ FH 16.0	▨ FGH 22.8	A 11.7

Figure 23. Schematic representation of trichromatic chlorophyll "a" (mg m^{-2}) in the three Mosquito Creek test channels. Blocks with the same letters are not statistically different at the 90% level. Black line between blocks indicates blocks connected are not different. Shading indicates ash present.

Table 9. Organic content of samples from various sources in Mosquito Creek channels.

Source	Detailed Description	Percent Organic Matter		
		$\bar{x}$	n	80% C.I.
Benthic Algae Filters	All channels before dump	14.7%	34	4.6 -24.9%
	Control channel after dump	6.1	18	5.0 - 7.1
	Treatment channels after dump	5.7	28	4.1 - 7.2
Fines from Canisters (<200 mesh)	All channels before dump	4.1	19	3.7 - 4.5
	Control channel after dump	4.7	13	4.4 - 5.0
	Treatment channel (CA) after dump	3.6	21	3.4 - 3.9
Non-Insect Residue From Insects Samples	All channels before dump	18.7	31	16.7 -20.7
	Control channel after dump	15.1	23	13.7 -16.6
	Treatment channels after dump	18.5	29	16.5 -20.5
Drift	Control channels	68.1	21	N.S.
	Treatment channels	68.3	11	N.S.

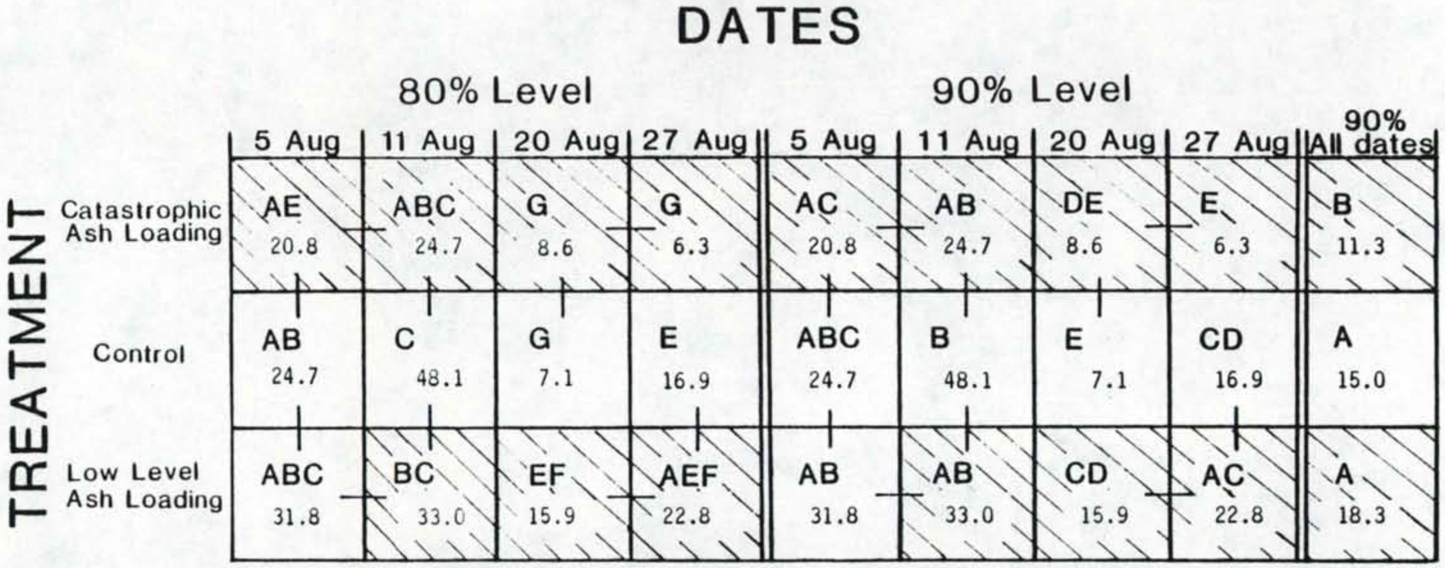
Ash-free (organic) content of the ABA increased significantly ($P \leq 20\%$) for two weeks after cessation of both ash dumps. The increase may have been caused by ash particles trapping detritus. Within three weeks, mean ash-free content had fallen to pre-dump levels.

Organic content of various detrital fractions in treatent channels was significantly different from control channels (Table 10). In cannister sediments, organic content of the fines fraction averaged 3.6% ... significantly lower than the 4.7% of the control channel. In non-insect residue from insect samples, treatment channels contained 18.5% organic content, significantly higher than the 15.1% of control channels. Benthic algal filters and drift samples showed no difference between treatment and control channels.

4.5 Benthic Insects

4.5.1 Benthic Insects - Standing Crop: Insect colonization in newly watered stream channels was slow during the first three weeks with the upstream sections colonizing more rapidly (13 June and 20 June; Figures 25-29). Towards late July the colonization rate increased rapidly, and by the time of the 4-5 August ash impact, aquatic insect densities averages $20 \times 10^3/m^2$ and were relatively similar in all channels and channel sections. The majority of the insects (80%) were midges (Chironomidae), with the mayfly, *Baetis bicaudatus*, comprising an additional 11% of the 2 August benthos.

The close similarity of densities among the channels two days before the ash impact was no longer evidenced in the samples collected at the conclusion of the 24-hour ash injection (Figure 25). The 5 August samples from the upstream section of the test channel averaged about one-half the chironomid density and one-tenth the *Baetis* density



	80% Level				90% Level				
	5 Aug	11 Aug	20 Aug	27 Aug	5 Aug	11 Aug	20 Aug	27 Aug	90% All dates
Catastrophic Ash Loading	AE 20.8	ABC 24.7	G 8.6	G 6.3	AC 20.8	AB 24.7	DE 8.6	E 6.3	B 11.3
Control	AB 24.7	C 48.1	G 7.1	E 16.9	ABC 24.7	B 48.1	E 7.1	CD 16.9	A 15.0
Low Level Ash Loading	ABC 31.8	BC 33.0	EF 15.9	AEF 22.8	AB 31.8	AB 33.0	CD 15.9	AC 22.8	A 18.3

Figure 24. Schematic representation of trichromatic chlorophyll "a" (mg m^{-2}) in the three Mosquito Creek test channels. Blocks with the same letter are not statistically different at the 80% or 90% levels. Black lines between blocks indicates blocks connected are not different. Shading indicates ash present.

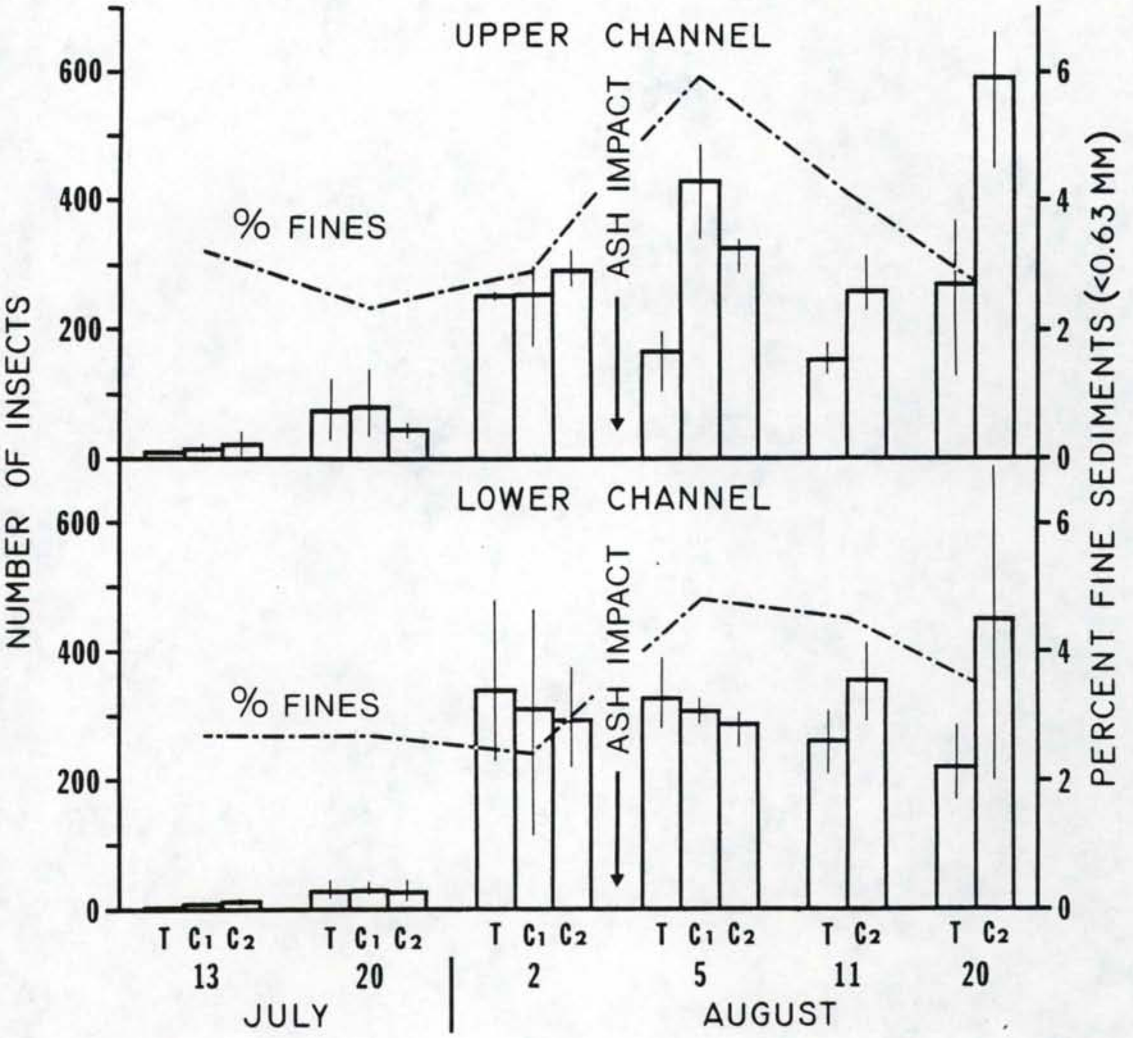


Figure 25. Mean numbers and ranges (vertical lines) of total insects per canister (N=3) and mean percent fine sediments (<0.062 mm) in test channel canisters (T=test channel, C_1 and C_2=control channels). 1982.

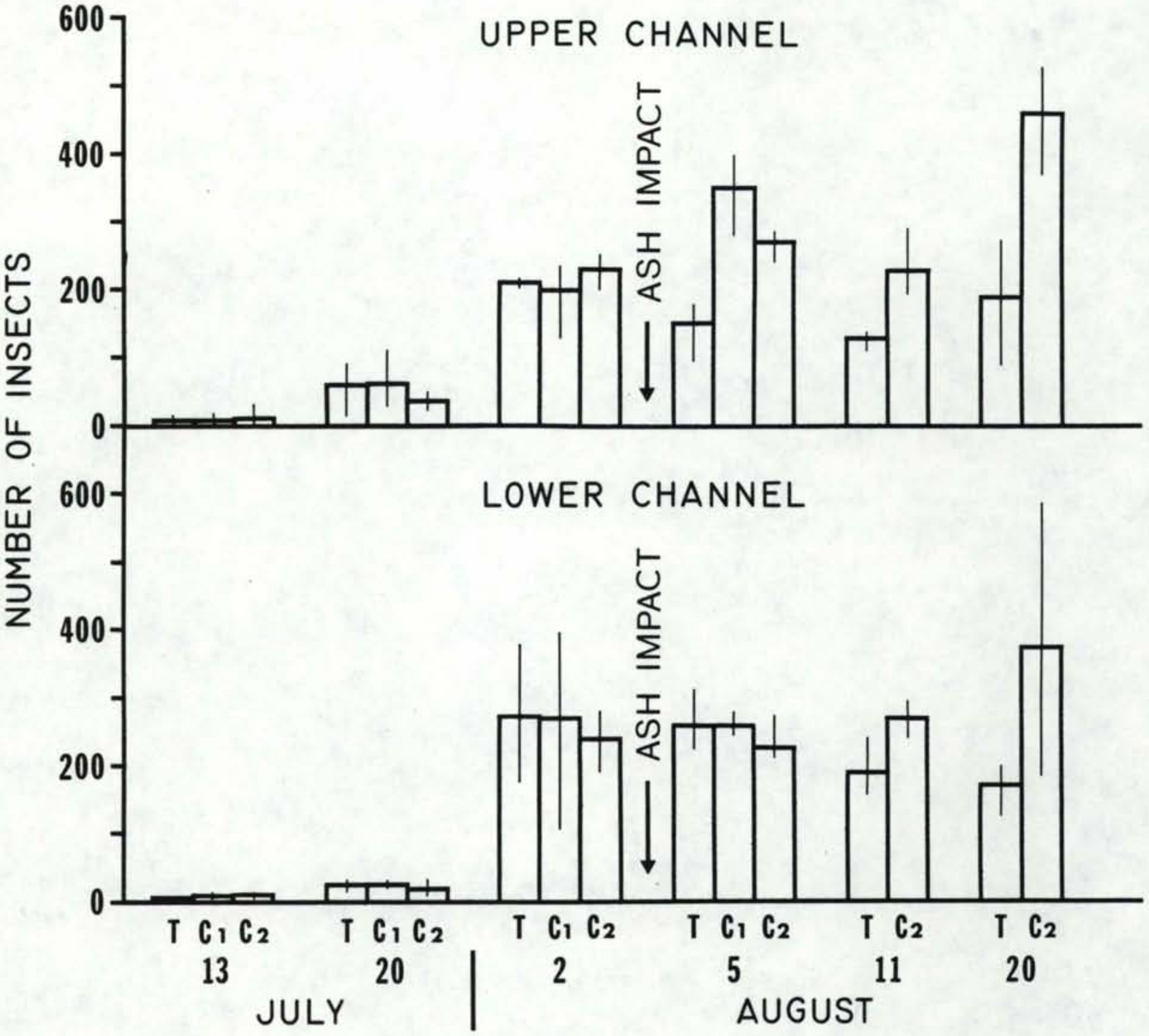


Figure 26. Mean numbers and ranges (vertical lines) of chironomids per canister (N=3; T=test channel, C_1 and C_2=control channels) 1982.

compared to the upstream half of the control channels (Figures 26, 27). The downstream (lower) sections of the three channels, however, continued to yield comparable densities of the midges and mayflies immediately following the ash impact.

Densities of <u>Baetis</u> mayflies in the test channel returned to levels similar to control channel densities during the recovery period (11 August and 20 August, Figure 27). However, during the same time, chironomid densities in the test channel remained depressed in relation to the control channel densities which increased markedly (Figure 26).

The stonefly <u>Yoraperla</u> <u>brevis</u> had low densities and high sampling variability by comparison to Chironomidae and <u>Baetis</u> in both test and control channels (Figure 28), but contributed appreciably (19%) to total biomass in the channels. Variability in total insect biomass between test and control channels was also apparent throughout the study as was biomass variability among samples within a channel (Figure 29).

4.5.2 <u>Benthic Insects - Drift colonization</u>: Insect drift into and out of the experimental channels increased steadily during the four week colonization phase (Figure 30). Approximately three times as many insects drifted into the channels on the last day of colonization as entered on the first day drift was sampled. There was also a marked increase in the percentage of insects settling-out of the incoming drift (Figure 31 and Table 10). By the time ash was introduced, we estimate that at least 90% of the insects entering the channels were settling in them. Also, after one month of colonization, there was a two-fold increase in insects drifting out of the channels (Figure 30) due to both increased standing crop in the channels as a source of drift and insects drifting through the channels but not settling in them.

Table 10. Insect recruitment into Mosquito Creek Channels during two key times (mid-day and night) prior to, one week after, and two weeks after ash impact. 1982.

Date	Time	North Control	Channel Center Control	Test
7/19-20	1230	72	22	24
	0100	56	96	212
	Total	128	118	236
8/10-11	1230	126	92	130
	0100	368	246	170
	Total	494	338	300
8/19-20	1230	50	42	110
	0100	418	818	304
	Total	468	860	414

Table 11. Drift of *Baetis bicaudatus*, Chironomidae, and *Yoraperla brevis* as a percent of total drift in the Mosquito Creek Experimental Channels, Clark Fork, Idaho 7/12/82 - 8/20/82.

	IN			OUT		
	Center Control	North Control	Test	Center Control	North Control	Test
7/12-13	.70	.71	.59	.77	.81	.84
7/19-20	.73	.62	.66	.90	.86	.86
8/3-4	.84	.86	.70	.86	.86	.88
8/4-5	.88	.84	.70	.88	.84	.91
8/10-11	.88	.94	.99	.92	.89	.83
8/19-20	.88	.89	.87	.96	.96	.93
All	.86	.85	.90	.90	.90	.89

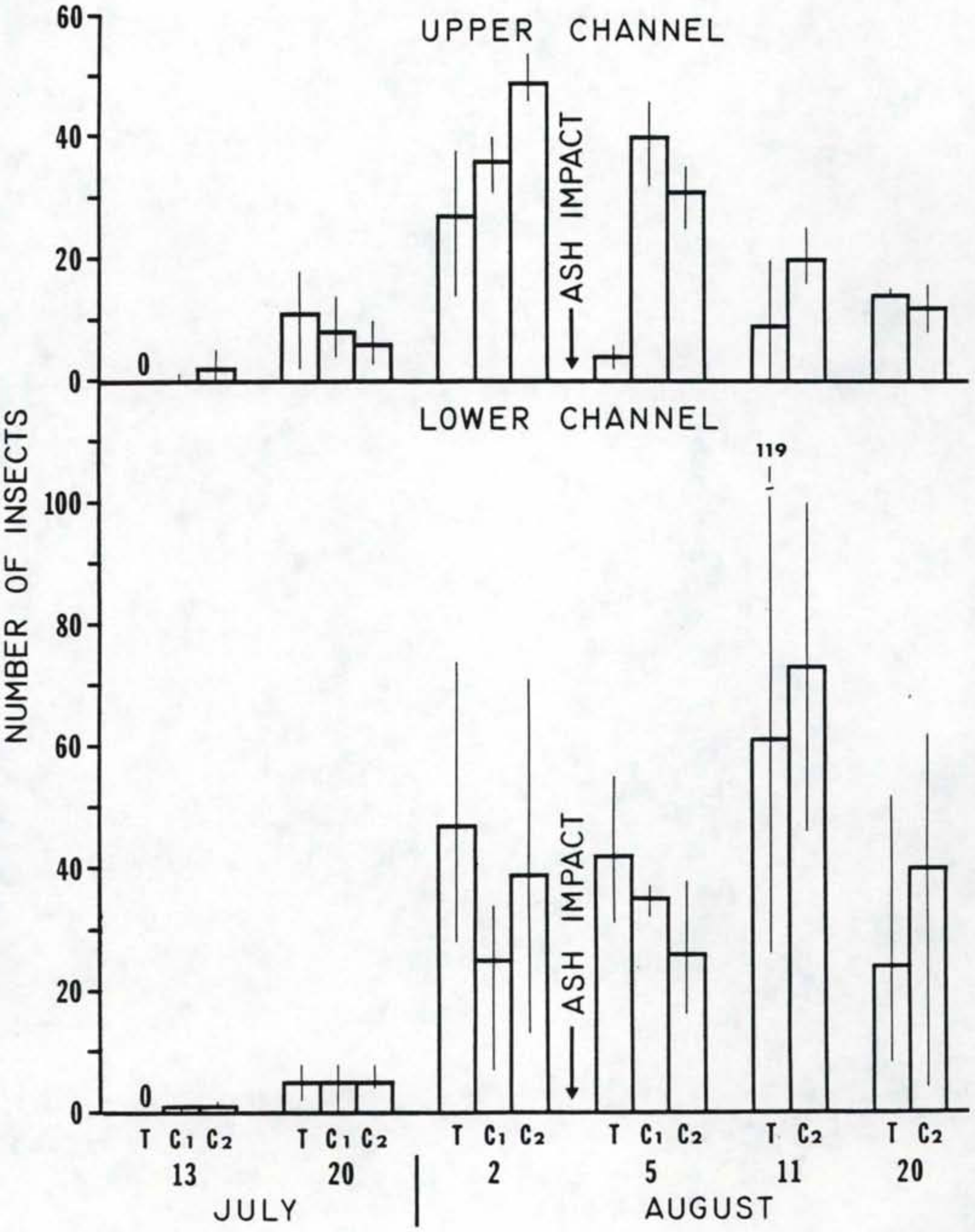


Figure 27. Mean numbers and ranges (vertical lines) of <u>Baetis bicaudatus</u> per canister (N=3; T=test channel, C_1 and C_2=control channels). 1982.

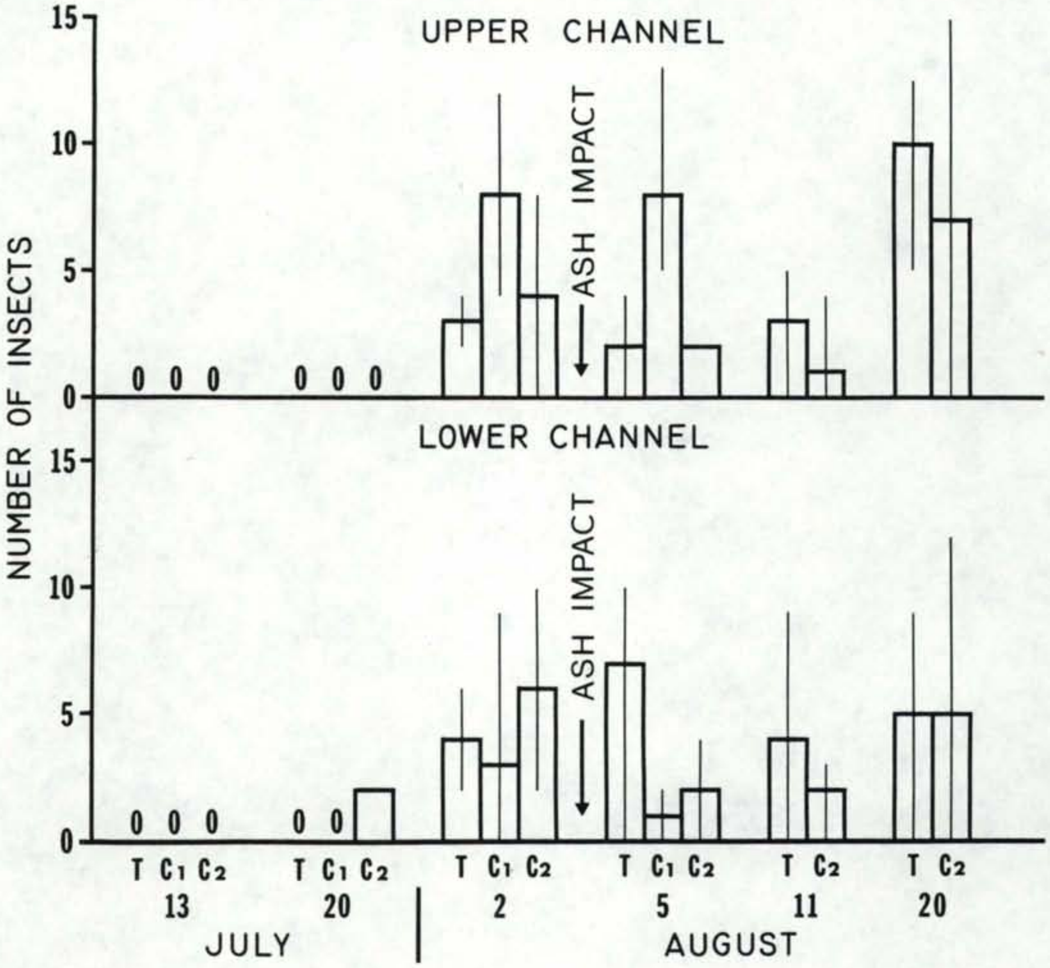


Figure 28. Mean numbers and ranges (vertical lines) of Yoraperla brevis per canister (N=3; T=test channel, C_1 and C_2=control channels). 1982.

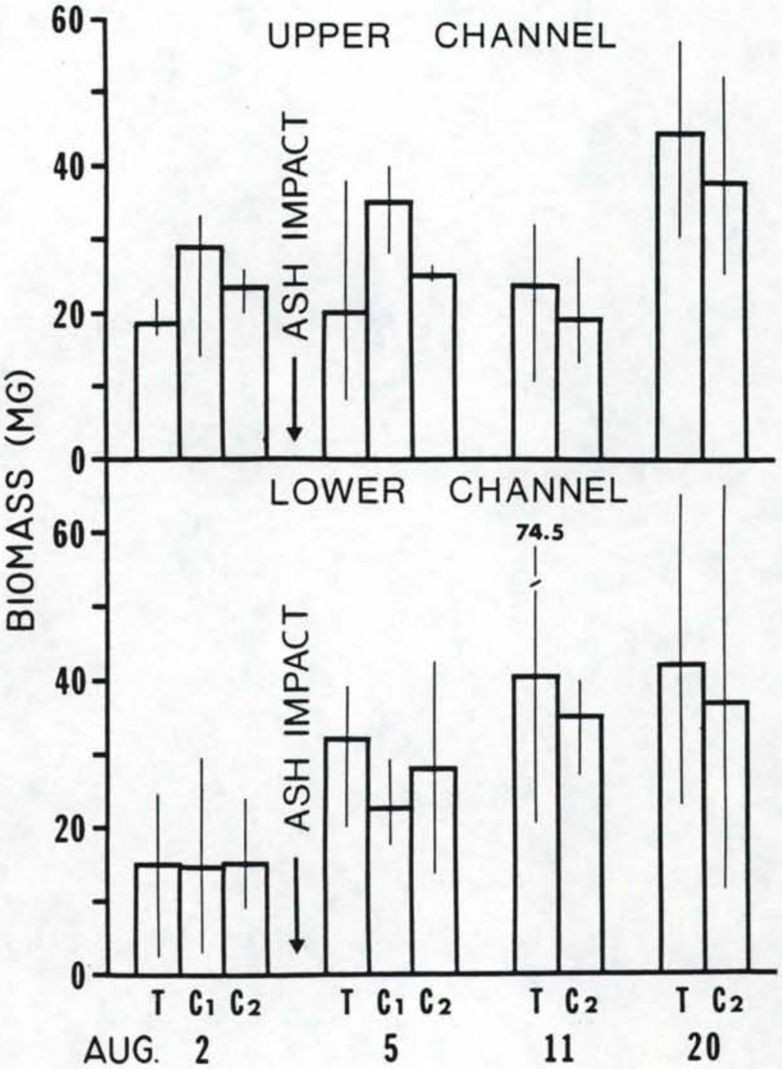


Figure 29. Mean biomass (mg) and ranges (vertical lines) of insects collected per canister (N=3; T=test channel; C_1 and C_2= control channels). 1982.

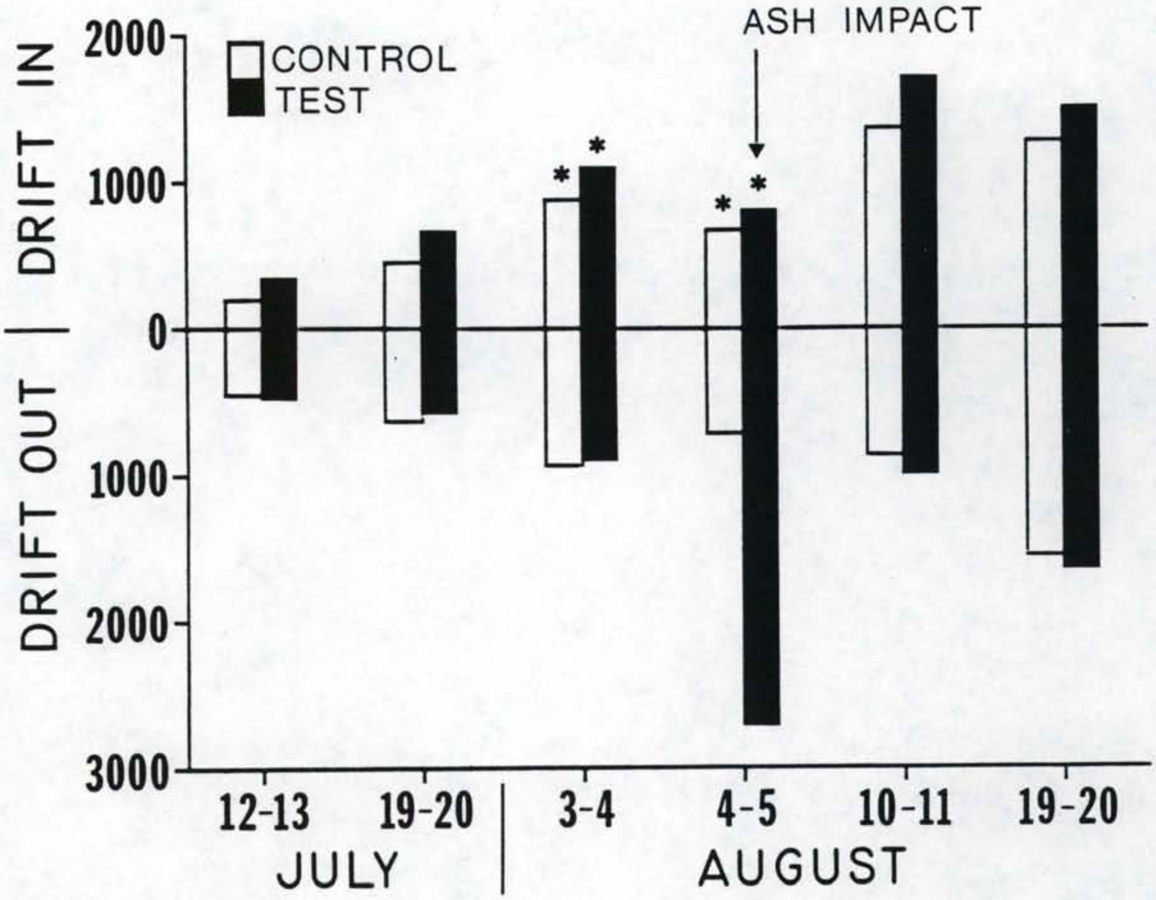


Figure 30. Total insect drift into and out of the Mosquito Creek experimental channels, July-August, 1982. Control drift is the mean of the two control channels. *Drift is estimated by calculating ratio of drift in/drift out from the four other sampling dates.

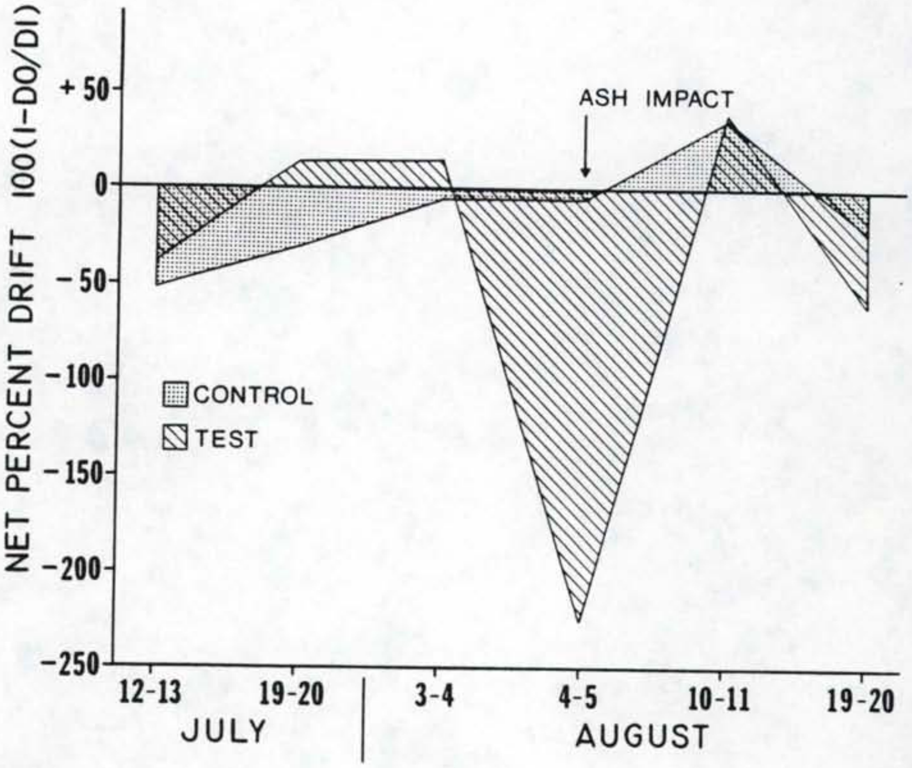


Figure 31. Net percent total insect drift at the Mosquito Creek experimental channels, July-August, 1982. Control drift is the mean of the two control channels. DO=Drift Out, DI=Drift In.

Three taxa accounted for the majority of aquatic insects drifting into and out of the Mosquito Creek Channels (Table 11). These were the mayfly Baetis bicaudatus, the midge complex, (principally the orthoclad Thienemanniella sp.) and the stonefly Yoraperla brevis.

Among the three dominant drift organisms there were substantial differences in their drift between the initial and final phases of the colonization period (Figures 32-33; Table 12). Initially, B. bicaudatus comprised 32-39% of the total organisms drifting into the three channels and 58-72% of the insects drifting out. By the end of the colonization period, drift of B. bicaudatus into the channels had decreased slightly (24-36% of total) but accounted for 30-36% of the insects emigrating. Conversely, during colonization the proportion of Chironomidae in drift increased in both the drift entering and leaving the channels (Figure 33; Table 12). On 12-13 July, midges accounted for 18-27% of the organisms moving into the channels and 2-17% drifting out. By 3-4 August midges comprised 45-52% of the insects captured in upstream nets and 47-56% at the downstream nets. Yoraperla brevis drift displayed yet a different pattern than either B. bicaudatus or Chironomidae. Few Y. brevis drifted during the colonization period; also, the drift of these small stoneflies into the channels was little different than the numbers departing.

The diel drift patterns of the dominant drift organisms displayed some similarities and several notable differences. Although B. bicaudatus nymphs were present throughout the day, they clearly displayed a nightime drift pattern. Initially, maximum drift occurred at 0100 h, but, as sampling continued, peak drift of B. bicaudatus shifted to 2100 h. Chironomidae showed a constant drift pattern during the first few

Table 12. Total drift of *Baetis bicaudatus*, *Chironomidae* and *Yoraperla brevis* into and out of the Mosquito Creek Experimental Channels, Clark Fork, Idaho 7/12/82 - 8/20/82.

	IN			OUT		
	Center Control	North Control	Test	Center Control	North Control	Test
Baetis bicaudatus						
7/12-13	.38	.39	.32	.67	.58	.72
7/19-20	.39	.32	.42	.72	.70	.66
8/3-4	.30	.36	.24	.30	.36	.30
8/4-5	.20	.21	.17	.20	.21	.22
8/10-11	.38	.51	.40	.46	.44	.36
8/19-20	.48	.58	.54	.57	.57	.53
All	.37	.44	.46	.49	.48	.39
Chiromonidae						
7/12-13	.27	.26	.18	.09	.17	.02
7/19-20	.29	.26	.17	.14	.11	.12
8/3-4	.52	.47	.45	.52	.47	.56
8/4-5	.66	.61	.52	.66	.62	.49
8/10-11	.44	.40	.56	.43	.43	.44
8/19-20	.34	.29	.31	.37	.36	.36
All	.44	.38	.41	.39	.39	.40
Yoraperla brevis						
7/12-13	.05	.06	.09	.01	.06	.10
7/19-20	.05	.04	.07	.04	.05	.08
8/3-4	.02	.03	.01	.02	.03	.02
8/4-5	.02	.01	.01	.02	.01	.20
8/10-11	.08	.03	.04	.03	.02	.03
8/19-20	.06	.02	.02	.02	.03	.04
All	.05	.03	.03	.02	.03	.10

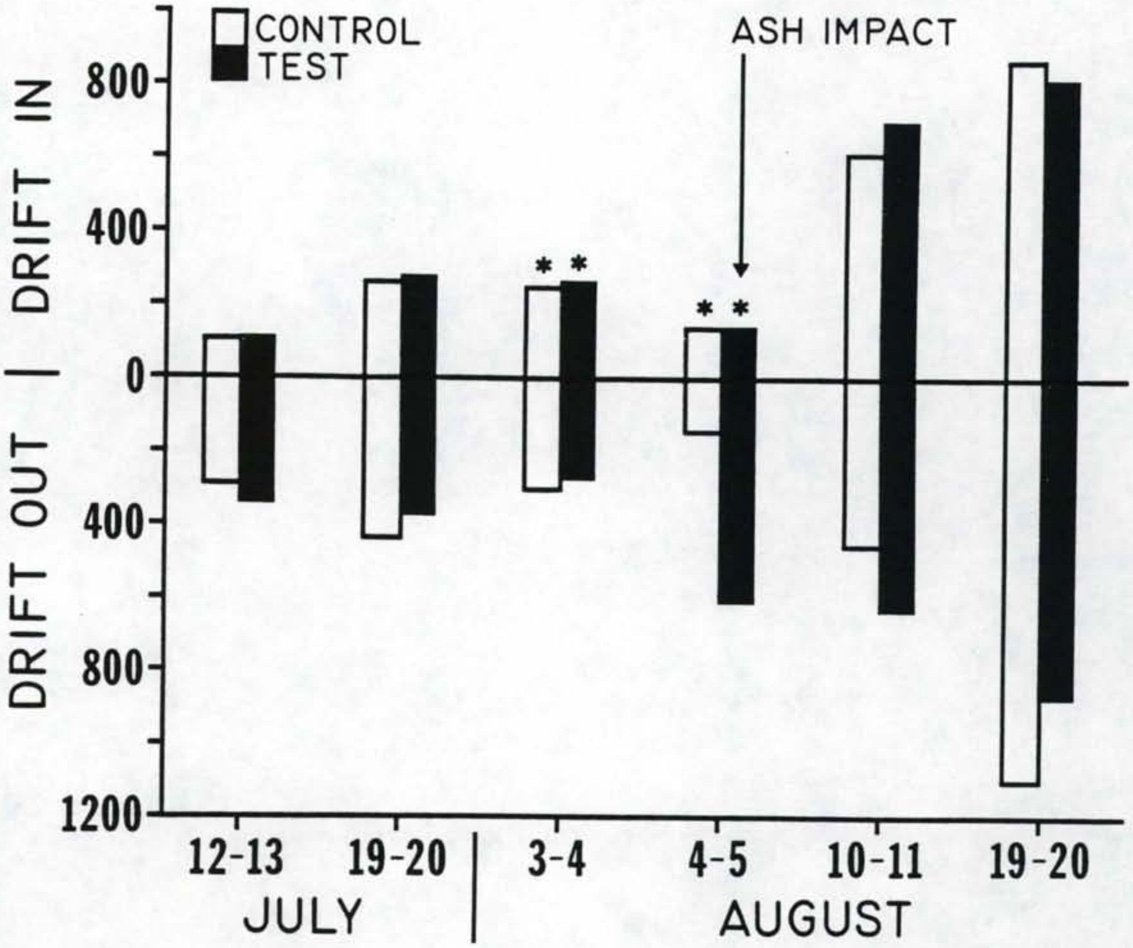


Figure 32. Drift of Baetis bicaudatus into and out of the Mosquito Creek channels, July-August, 1982. Control Drift is the mean of the two control channels. *Drift is estimated by calculating ratio of Drift in/Drift out on the four other sampling dates.

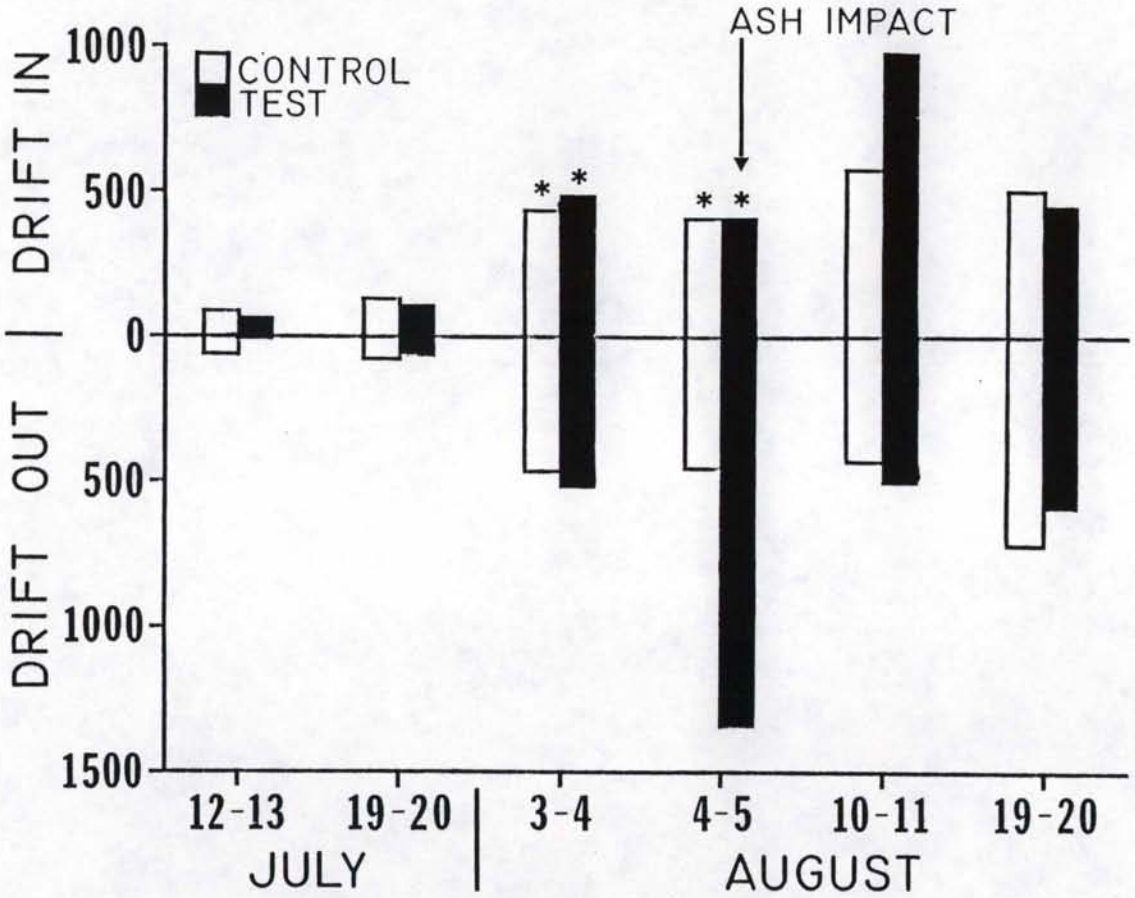


Figure 33. Drift of chironomidae into and out of the Mosquito Creek channels, July-August, 1982. Control of Drift is the mean of the two control channels. Drift is estimated by calculating ratio of Drift in/Drift out on the four other sampling dates.

weeks of colonization, but by 3-4 August there was evidence of a daytime pulse in midge drift activity. Few Y. brevis drifted during the day time; they showed a propensity for crepuscular or night-time drifting.

4.5.3 Drift--Ash Impact: Volcanic ash caused catastrophic insect drift in the Mosquito Creek experimental channels (Figures 34-37). Drift samples collected from the test channel during our six key times contained three times as many insects as those collected on the day prior to the simulated ash fall. Drift out of the test channel was four times as great as that out of the control channels during the six key times (Figure 30). Since we sampled continuously prior to and during the ash impact, we also have an absolute count of the insects drifting out of the three channels. On the day before the simulated ash fall, 24, one-hour samples from the two control channels and the test flume contained 3574, 3574 and 3488 insects respectively. Numbers of insects drifting during the 24 hours of ash impact were 2701 and 2712 in the two control channels and 9069 in the test channel.

Baetis bicaudatus, Chironomidae and Y. brevis accounted for 91% of the total drift organisms in the test channel while ash was added to the water column (Table 4). Numbers of drifting B. bicaudatus and Chironomidae (Midges) increased dramatically during the ash test (Figures 35 and 36). Four times as many B. bicaudatus and three times as many midges drifted out of the test channel compared to the control streams. The change in Y. brevis drift was most pronounced. In the test channel the drift of these stoneflies increased thirty-six fold on the day of ash input compared to the day before ash fall (Figure 37). Also, drift

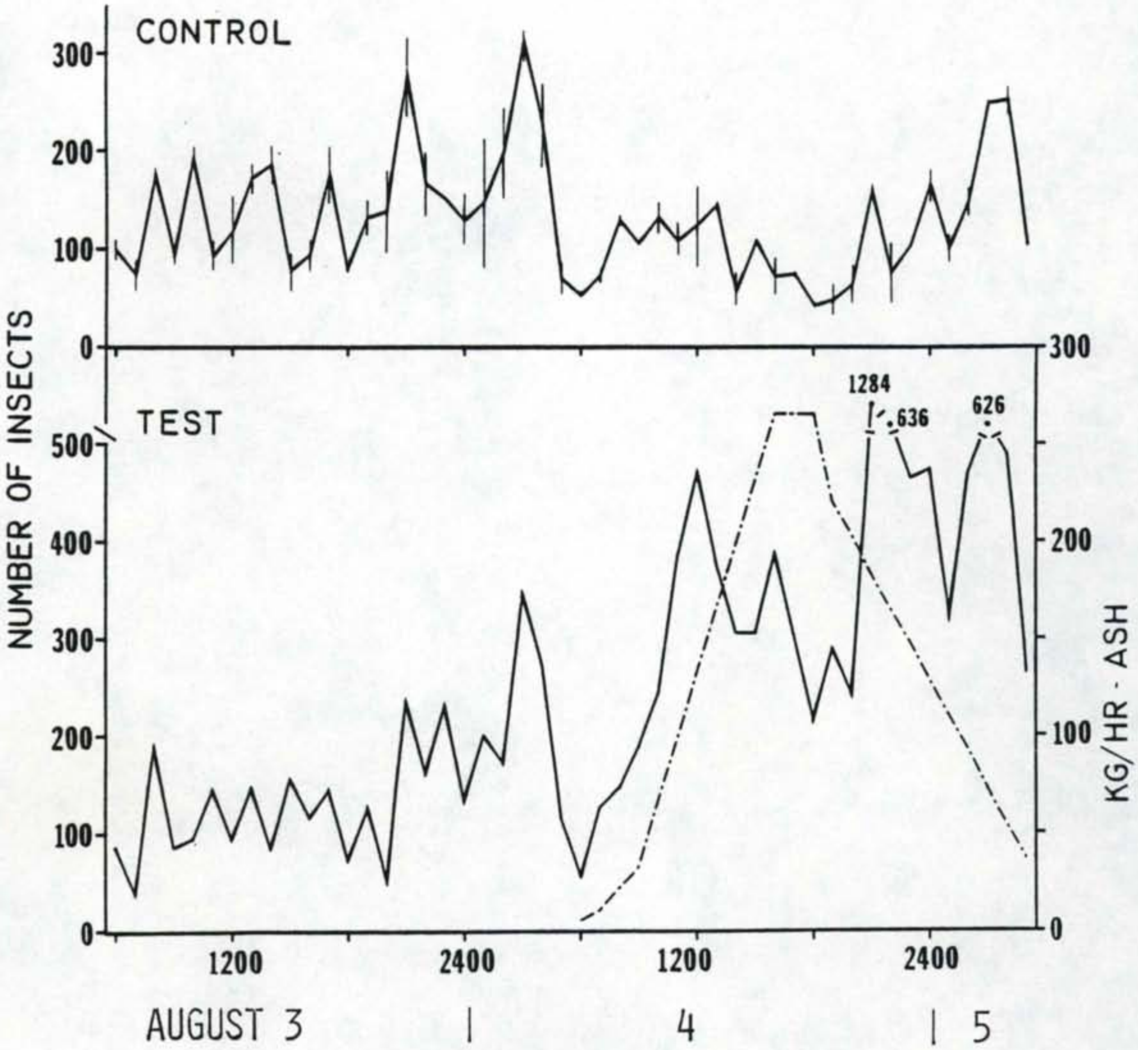


Figure 34. Total insect drift (solid line) in the Mosquito Creek experimental channels before and during ash impact, August 3-5, 1982. Dotted line is kg per hour of ash.

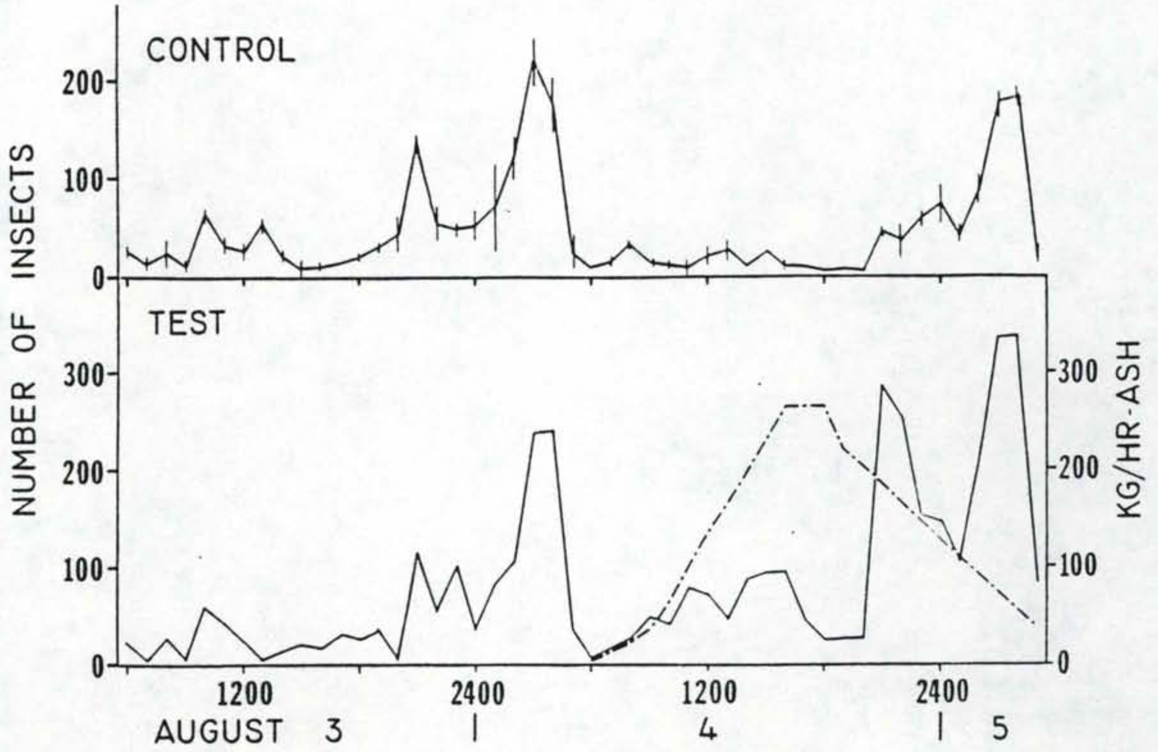


Figure 35. Drift of Baetis bicaudatus (solid line) in the Mosquito Creek experimental channels before and during ash impact, August 3-5, 1982. Dotted line is kg per hr. of ash.

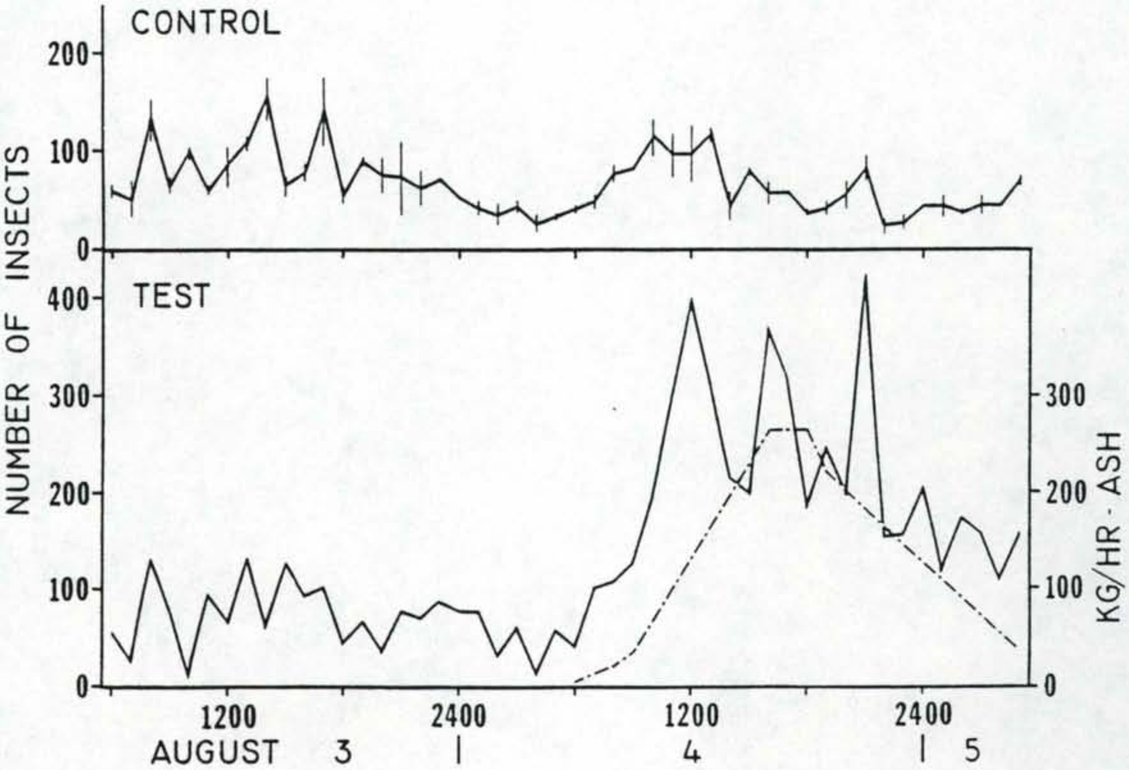


Figure 36. Drift of Chironomidae (solid line) in the Mosquito Creek experimental channels before and during ash impact, August 3-5, 1982. Dotted line is kg per hr. of ash.

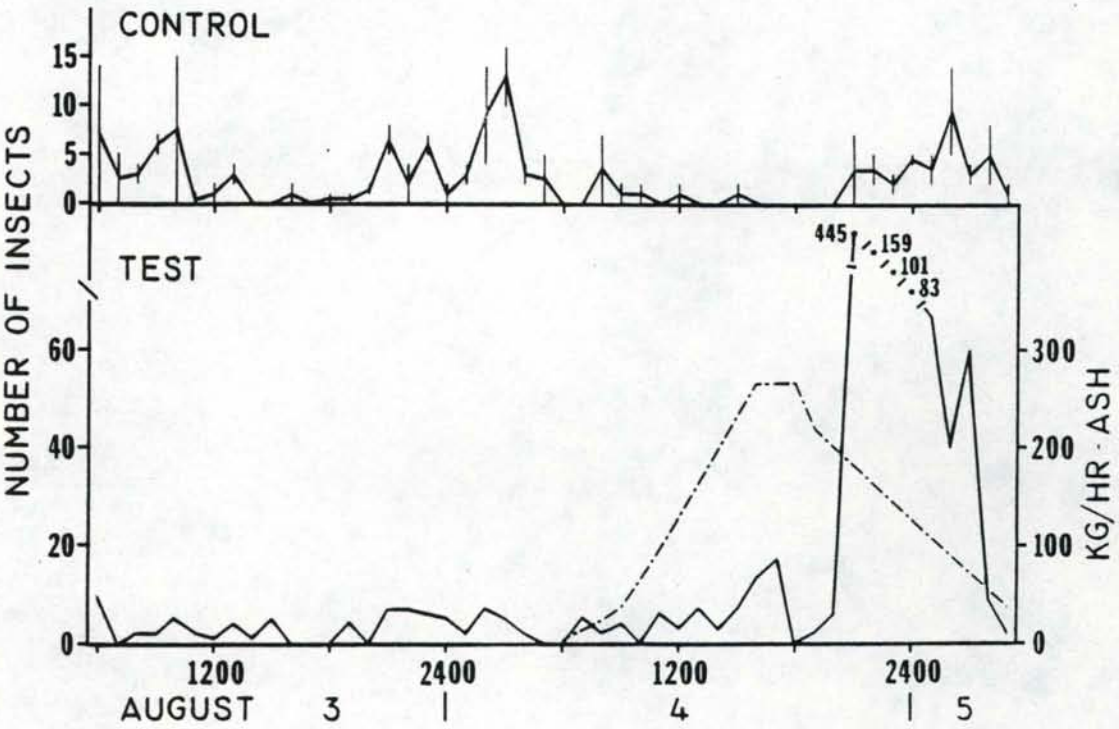


Figure 37. Drift of Yoraperla brevis (solid line) in the Mosquito Creek experimental channels before and during ash impact, August 3-5, 1982. Dotted line is kg per hr. of ash.

samples from the test channel contained 60 times as many Y. brevis as those from the two control channels.

Catastrophic drift by the three dominant insects was influenced by the entrained diel behavioral drift patterns of those insects. Although B. bicaudatus initially drifted in increased numbers as ash loading was increased, the drift of these mayflies declined during the afternoon despite greater ash inputs (Figure 35). A sudden pulse in Baetis drift occurred at dusk, drift declined after midnight, then peaked before dawn. Drift of midges increased dramatically throughout the day as ash loading increased (Figure 36) and unlike B. bicaudatus, remained high in the early morning. The diel pattern of midge drift in the test channel was similar to the control streams but the peaks were more pronounced. Increased drift of Y. brevis occurred during the daylight hours as ash was added (Figure 37), but the greatest response of this organism to ash input was delayed until shortly after dark (2100 h) when 445 Y. brevis were collected in a single hour. Their drift remained high throughout the evening and early morning, then dropped suddenly at dawn.

4.5.4 Drift-Recovery/Low-Level Ash Impact: Insect drift during this phase of the study reflected both the short-term recovery of the test channel and the effect of low-level ash input. One week after ash impact, greater numbers of insects entered the channels than had entered prior to the simulated ash fall (Figure 30); however, fewer insects settled in the test channel than in the two remaining channels (Table 10). During six key times, both the center and north channels had similar numbers of insects drifting into them, but more insects drifted out of the center channel (the channel that received one week of low-level ash input). Nine days later (19-20 August), greater numbers of insects continued to drift out of the channels than entered. However, insect drift from the center channel was now virtually identical to that from the north channel (1948 and 1942 insects, repectively). Baetis bicaudatus, Chironomidae, and Y. brevis continued to dominate the drift (Table 11). During the two sampling periods after ash injection, they accounted for 83-99% of the aquatic insects drifting into or out of the flumes. Drift of B. bicaudatus in and out of the channels increased throughout the recovery phase (Figure 32). Compared to our first sampling date, there was a seven fold increase in the drift of this mayfly into the channels and approximately three times as many emigrating. One week after the simulated ash fall, greater numbers of Chrionomidae drifted into the channels, but by August 19-20 the numbers of midges entering the channels declined (Figure 10). Midge drift out of the channels, although ten times greater than during the initial colonization samples, was little different from that of the day before ash injection. Nine days later, the number of midges emigrating increased in all three channels. However, the relative abundance of Chironomidae in both the drift entering and leaving the channels declined during the last two August sampling dates. Yoraperla brevis drifted in greater numbers by the end of our sampling period than they had at anytime except on the day of simulated ash fall in the test channel. Among all three channels, drift of Y. brevis into the channels exceeded that leaving.

Changes in the diel drift patterns of the three most numerous insects were observed during the recovery, low-level ash impact phase. On August 10-11 peak numbers of B. bicaudatus drifted during darkness. Nine days later an additional peak was apparent at 0500 h and at 0900 h. Drift of Chironomidae into the channels displayed a relatively constant pattern during our last two sampling dates. However, drift periodicity of midges was observed in samples collected at the downstream ends of the channels during those times. As in the previous sampling periods, maximum drift of Y. brevis was observed in the evening or early morning.

4.6 Community Oxygen Response

Control and low level ash loading channels were capped with a 1/8-inch thick plexiglass cover to exclude gas exchange with the atmosphere yet permit photosynthesis so that differential community photosynthesis and respiration responses could be measured. In August, the low-level ash channel showed a 50% greater oxygen response by mid-day during a capped run (Figure 38). This higher photosynthesis reflected the increased algal biomass in the ash-impacted channel.

Repetition of this capped experiment in mid-September did not produce a comparable response (Figure 39). Photosynthesis was low and unable to compensate for community respiration in the capped channels. Both control and low-level loaded channels showed oxygen loss in the water moss with passage through the capped channels. Differential responses were not evident. Two factors caused the lack of response in September: 1) Algal biomass had greatly declined by mid-September with decreasing light and temperature and, 2) the ash-impacted channel had largely recovered from effects of the ash dump by then.

4.7 Detrital Decomposition and Leaching

4.7.1 Instream Jar Tests: The physical and chemical parameters of Mosquito Creek remained stable for the first two months of the study (only temperature) was recorded after the first two months). The day-time temperature ranged from 9.0 to 5.1°C from July 12 to October 31. Diel changes were typically less than 2°C.

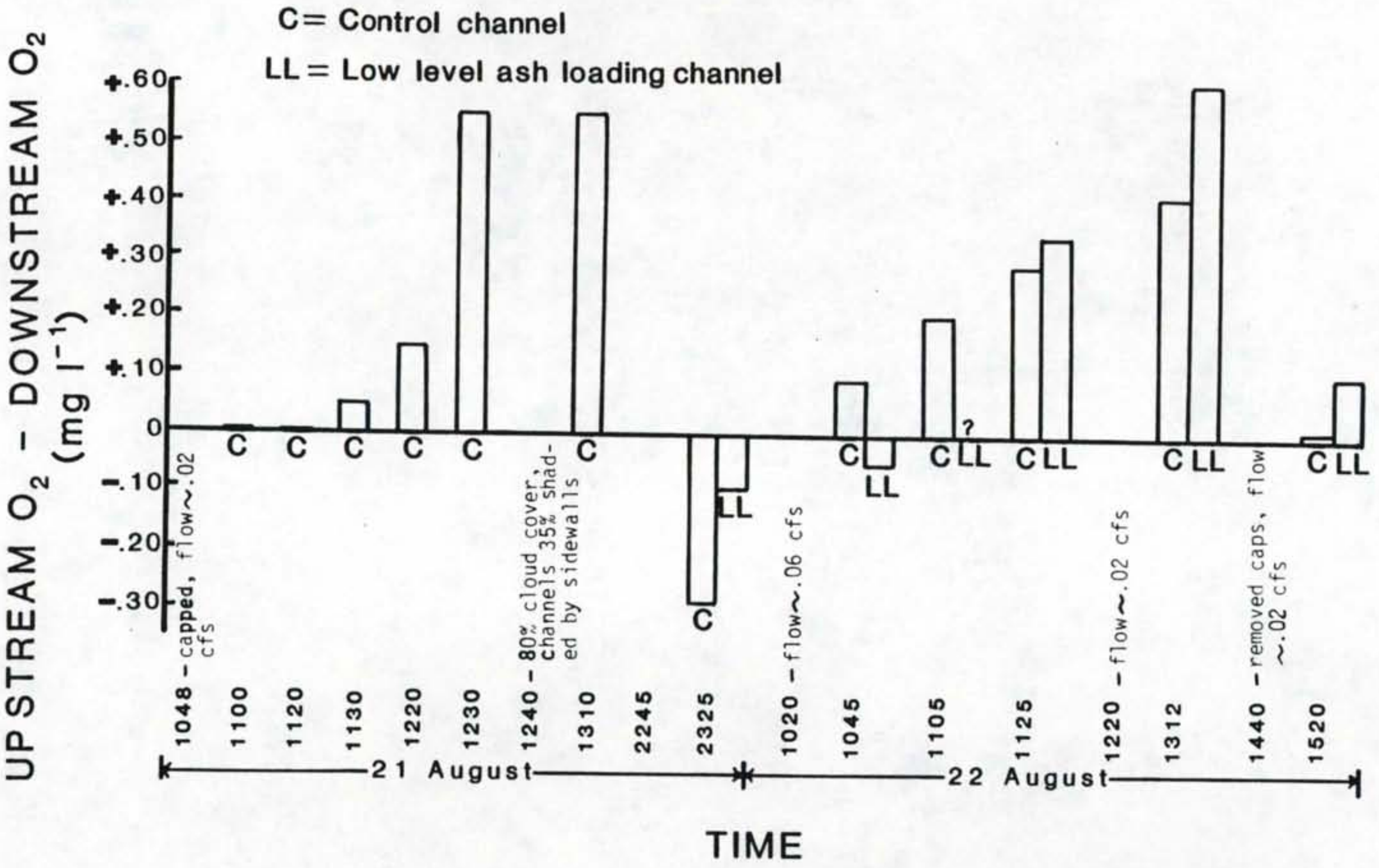


Figure 38. Comparative oxygen response in the low level ash impacted channel versus control channel. Both channels isolated from the atmosphere.

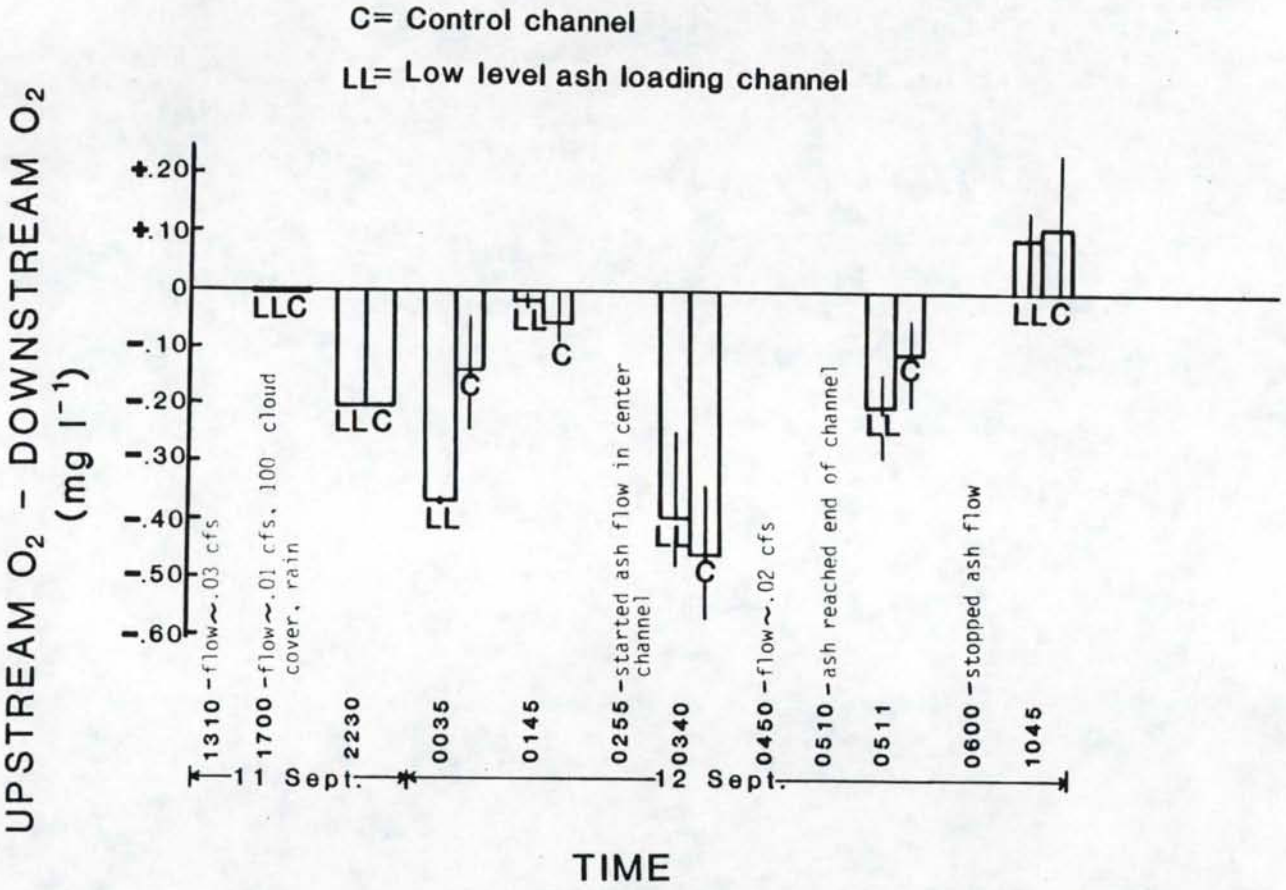


Figure 39. Comparative oxygen response in the low level ash impacted channel versus control channel. Both channels isolated from the atmosphere.

The partial replacement time for water in the jars was 44 minutes for 99% replacement for the jars covered by the small mesh (350µ) and 15 minutes for 99% replacement for jars covered by the large mesh (8 mm). These replacement times are an average for the whole jar; the upper was recorded after the first two months). The daytime temperature ranged from 9.0 to 5.1°C from July 12 to October 31. Diel changes were typically less than 2°C.

The partial replacement time for water in the jars was 44 minutes for 99% replacement for the jars covered by the small mesh (350µ) and 15 minutes for 99% replacement for jars covered by the large mesh (8 mm). These replacement times are an average for the whole jar; the upper portion of the jar has a much shorter replacement time while the bottom cm of the jar may never exceed 99% replacment, particularly in those jars treated with ash or soil.

Most of the ash and soil settled to the bottom of the jars leaving a light coating of ash on all but the bottom disks with occasional thicker patches on surface leaf disks. Leaf disks buried in ash or sediment near the bottom of the jar changed little from their initial state, while surface leaves darkened in color and became frayed at the edges.

4.7.2 <u>Leaching</u>: The soluble components of leaves leached quickly once placed in the water, often within 24 hrs (Nykvist, 1963; Petersen and Cummins, 1974). In this experiment, a log-normal plot of percent dry weight remaining showed a decrease in the steepness of the slope at 3 days, so calculations of leaching rate were based on the first three days of leaching data (Table 13), (Figure 40).

A comparison of decay coefficients showed no significant difference (alpha=0.1) between any of the four treatments, nor were the decay coefficients of the three sediment treatments grouped together relative to the control (Table 13).

Table 13. Leaching decay rates over a three day period for norway maple leaves in four treatments.

Treatment	Decay Coefficient ($K x 10^{-3}$)	90% Confidence Limits ($x10^{-3}$)	r^2 of Regression	% Loss Per Day
Control	52.9	44.8-61.0	.95	5.2
3.64g ash	48.0	40.2-55.8	.94	4.7
7.28g ash	65.8	51.5-80.1	.90	6.4
3.64g soil	60.6	47.5-73.6	.90	5.9

The analysis of variance and Duncan's multiple range test indicated no difference in mean percent remaining between the control and 3.64 g ash/jar and between the 7.28 g ash/jar and the 3.64 g soil/jar treatments. The control and lower ash treatments did have a significantly lower mean percent loss than did the other two treatments, however (Table 14).

Table 14. Results of analysis of variance and Duncan's test (alpha=0.1) for norway maple leaves in four treatments (leaching effect). The dependent variable is Log X_t/X_0 where X_t is the dry dry weight at time t and X_0 the initial dry weight. Groups with the same letter are not significantly different.

Analysis of Variance			Duncan's Test		
Source variable	F	Prob>F	Treatment	Group	Mean (X_t/X_0)
Model	6.9	.0001	3.64g ash	A	.90
treatment	12.37	.0001	control	A	.89
number of days	17.24	.0001	7.28g ash	B	.87
(treatment)(days)	0.72	.6371	3.64g soil	B	.86

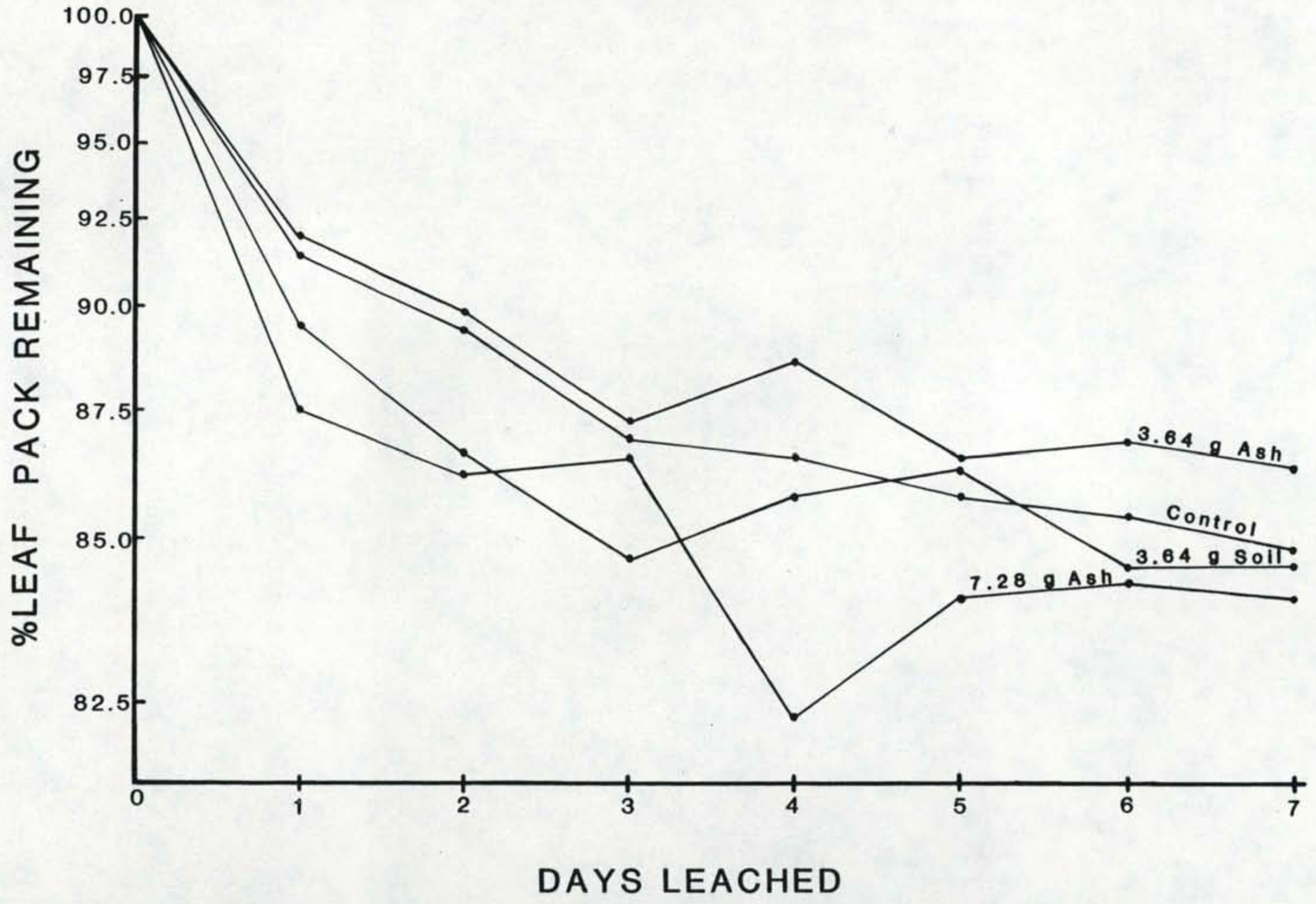


Figure 40. Leaching rates of maple leaf packs in Mosquito Creek as a function of applied volcanic ash or soil over a 7 day period.

4.7.3 Leaf Pack Experiments Without Invertebrates: Decay coefficients with 90% confidence limits indicated no significant difference between the control and other treatments (Table 15), although the decomposition rate was significantly faster in the treatment of 3.64 g ash/jar than in either the 3.64 g soil/jar or the 7.28 g ash/jar treatments. The model fit the data extremely well.

Table 15. Decay coefficients for normay maple leaves in four treatments (without invertebrates).

Treatment	Decay Coefficient ($K \times 10^{-3}$)	90% Confidence Limits ($\times 10^{-3}$)	r^2 of Regression	% Remaining after 1 yr ($\%R_{365}$)
control	5.7	5.2-6.1	.96	13
3.64g ash	6.2	5.8-6.7	.96	10
7.28g ash	5.0	4.6-5.4	.94	16
3.64g soil	5.3	4.8-5.6	.96	14

The analysis of variance and Duncan's multiple range test indicated a significant difference (alpha=0.1) between the control and both the 7.28 g ash/jar and the 3.64 g soil/jar treatments in mean percent remaining (Table 16). The significance of crossed terms will not allow us to make conclusions about decay rates, however (Weider and Lang, 1982). The mean percent remaining in the control treatment was less than either the higher ash treatment or the soil treatment.

Table 16. Results of analysis of variance and Duncan's test (alpha=0.1) for norway maple leaves in four treatments (without invertebrates). The dependent variable is Log X_t/X_o where X_t is the dry weight at time t and X_0 the initial dry weight. Groups with the same letter are not significantly different.

Analysis of Variance			Duncan's Test		
Source variable	F	Prob>F	Treatment	Group	Mean (X_t/X_o)
Model	54.5	.0001	3.64g soil	A	.72
treatment	8.4	.0001	7.28g ash	A	.71
number of days	243.0	.0001	Control	B	.69
(treatment)(days)	3.2	.0010	3.64g ash	B	.68

4.7.4 With and Without Invertebrates: For the without-macroinvertebrates treatment set the decay coefficient was significantly smaller (alpha=0.1) for the treatment receiving ash than it was for the corresponding treatment without ash (Table 17). For the with-macroinvertebrates treatment set, although a comparison of confidence intervals between ash and non-ash treatments failed to show significant difference, paired comparison techniques did show a smaller decay coefficient for the ash treatment at the alpha=0.1 level. The decay coefficient for the treatment without invertebrates and with ash was significantly lower than all others and the decay coefficient for the treatment with invertebrates and without ash was higher than all others but the difference was not significant when comparing 90% confidence limits.

Table 17. Decay coefficients for norway maple leaves in four treatments (with and without invertebrates).

Treatment	Decay Coefficient ($K \times 10^{-3}$)	90% Confidence Limits ($\times 10^{-3}$)	r of Regression	% Remaining after 1 yr (%R_{365})
Without invertebrates				
with ash	5.8	5.2-6.4	.93	12
without ash	7.3	6.8-7.8	.97	7
With invertebrates				
with ash	7.2	6.7-7.7	.97	7
without ash	8.2	7.7-8.8	.98	5

Statistical differences between treatment means using an analysis of variance and a Duncan's multiple range test matched the differences found between decay coefficients of treatments. The treatments with ash for both treatment sets (with and without insects) had a significantly higher (alpha=0.1) mean percent remaining than the corresponding without ash treatment (Table 18). The treatment set which allowed macroinvertebrate colonization had a lower mean percent remaining than treatments without invertebrates.

Table 18. Results of analysis of variance and Duncan's test (alpha=0.1) for norway maple leaves in four treatments (with and without invertebrates). The dependent variable is Log X_t/X_o where X_t. is the dry weight at time t and X_0 is the initial dry weight. Groups with the same letter are not significantly different.

Analysis of Variance			Duncan's Test		
Source variable	F	Prob>F	Treatment	Group	Mean (X_t/X_o)
Model	43.2	.0001	no invert., ash	A	.74
treatment	13.6	.0001	invert. and ash	B	.70
number of days	191.2	.0001	no invert., no ash	B	.69
(treatment)(days)	3.8	.0007	invert., no ash	C	.66

The treatments allowing macroinvertebrate colonization contained roughly the same invertebrate species and total numbers of invertebrates (Table 19). Chironomids predominated in both with- and without-ash treatments (85% and 79% respectively).

Table 19. Macroinvertebrates found in treatments with and without ash. Functional groups after Merrit and Cummins (1978): S=shredder, C=collector, G=grazer, and P=predator. Numbers are totals found for all four sample dates.

Invertebrate	Functional Group	Without Ash	With Ash
Hydracarina			
Lebertia sp.	P	4	5
Sperchon sp.	P	1	5
Plecoptera			
Yoraperla brevis	S	2	0
Ephemeroptera			
Ephemerella infrequence	C	10	12
Ameletus sp.	C	27	9
Trichoptera			
Rhyacophila sp.	P	1	2
Neothremma sp.	C	0	2
Diptera			
Tipulidae			
Tipula sp.	S/C/G	0	1
Psycodidae			
Pericoma sp.	C	15	5
Chironomidae			
Corynoneura sp.	C	40	46
Cricotopus sp. 1	S/C	87	97
Cricotopus sp. 2	S/C	7	3
Phaenopsectra sp.	C	21	22
Zavrelia sp.	C	53	64
Pentaneura sp.	C/P	11	8
Empididae sp.	P	1	0
TOTAL		280	281
Shannon-Weaver Diversity Index			
(based on genus)		3.35	2.93

Chapter 5. Discussion

5.1 Physical Assessments

Similar amounts of volcanic glass were found to have been incorporated in the silt-size fraction from the three localities. The percentages of volcanic glass in the sediment fraction finer than 200 mesh (0.003 inch; 74 µm) were found to be 2 to 6% (Mt. St. Helens) for the Mosquito Creek simulation; 2 to 4% (Mt. St. Helens) for the Graves Meadow site; and 2 to 4% (Mazama) for the Lighting Creek sediments. These similarities may indicate that the same mechanism for the incorporation of volcanic glass into the silt-size sediment fraction was dominant at all three localities, despite the time interval between depositions.

The bulk increases of fine grain sediment in a stream reach during an ashfall event appears to be from the non-volcanic glass component of the tephras. During the Mosquito Creek simulation 75-80% of the ash injected was volcanic glass. The amount of silt-size material (4 phi = 0.002 inch = 63 µm) in the test reach doubles (2.7% pre-impact to 5.3 post-impact). However only 4% of the bed material finer than 4 phi was volcanic glass. This implies that only 8% of the total deposited material was volcanic glass. The remainder had to be made up of a disproportionate amount of the non-glass component of the tephra.

Different densities of tephra components can explain the differential settling behavior. The depositional model assumed that volcanic ash was a mixture of grainsizes of homogeneous densities. Actual differences in density would be reflected in the distribution of apparent fall diameters. In reality the ash is a heterogeneous mixture of minerals with widely varying densities. Volcanic glass is vesicular, tending to trap air and become less dense. The denser ferromagnesium and plagioclase components settle more rapidly and thus form a larger percentage of the material ending up in the bed. However, after deposition only the volcanic glass can be positively identified as foreign.

Similar proportions of Mt. St. Helens volcanic glass were observed in the strata at the Graves Meadow site as were found in the bed after the Mosquito Creek simulation. Since the Mosquito Creek channels have some of the same hydraulic characteristic as the riffle and run habitats of the Graves Meadow site, it can be inferred that similar quantities of non-volcanic glass sediments were incorporated into the upper sedimentary units of the Graves Meadow reach of the Palouse River.

5.2 Algae Response

5.2.1 Attached Benthic Algae

The greatly increased growths of ABA in both control and experimental stream channels over growths in the adjacent Mosquito Creek were due to a combination of factors. Passage through the channels added ≤ 0.5 C to the stream water ... a factor contributing to greater growth, but probably not significantly so. Water depth was a uniform 3 inches in the channels, about 1/3 the average depth in Mosquito Creek. This increased light availability due to shallower depth was probably the greatest factor in the increased production of the channels. Since the channel depth was uniform throughout, and differential upstream-downstream shading was not evident, light variations due to these sources would not have accounted for the up-to sixfold increase in ABA in downstream channel reaches. We observed that the upper channel sections were more turbulent than lower sections and believe that these turbulence differences, with accompanying differences in light penetration, accounted for greater downstream biomass.

Both ash treatments initially (3-5 days) resulted in depressed ABA chlorophyll levels. The catastrophically-loaded channel biomass remained depressed through the study, but the low-level loaded channel responded within two weeks and following ABA growth exceeded control ABA concentrations. Gray and Ward (1982) also observed increased growths of ABA in response to sediments, but attributed the increased Cladophora growths in the North Platte River to coincidental increases in nutrients. No such nutrient increases occurred in our studies according to ash nutrient analyses of Skille et al. (1983). In the Mosquito Creek channels, the physical effect of interstice filling permitted greater ABA growth. It appears that in the even-flow environment of the channels, these clay-sized fines were stable enough to permit and even to enhance ABA growth.

5.3 Secondary Productivity -- Insects

No acute mortality occurred among test specimens (Hesperophylax occidentalis) during the laboratory experiments. Brusven and Hornig (in press) also found a lack of acute mortality among several species of aquatic insects maintained in volcanic ash concentrations of 2000 mg/l for 14 days. These findings agree with previous studies that found: 1) suspended sediment is not intrinsically toxic to aquatic life (Wilbur, 1969); and 2) volcanic ash is not chemically toxic (Fruchter et al. 1980). However, sublethal effects of volcanic ash have been reported by Gersich and Brusven (1982).

A possible sublethal effect of suspended volcanic ash on aquatic insects may be reduced feeding activity, as was indicated by our laboratory experiments. However, the immediate effect would have lasted for only the few hours or days while ash remained in the water column. Consequently, any reduction in insect productivity due to decreased feeding activity was probably negligible.

Despite the inferred non-acute toxic effects of volcanic ash, there was an insect standing crop decline of greater than 50% in the upper section of the Mosquito Creek test channel immediately following the 24-hr ash impact. The reduction in insect density was clearly the result of emigration from the channel in the form of catastrophic drift. Three types of invertebrate drift have been described from streams (Waters, 1972): 1) constant drift - a continuous, random drift throughout the day; 2) behavioral drift - drift which occurs with a diel periodicity and is largely genetically entrained; and 3) catastrophic drift - increased drift due to extreme fluctuations in a physical variable such as stream discharge or in response to pollution inputs.

The drift of Baetis bicaudatus, Chironomidae, and Yoraperla brevis, the major insects in the Mosquito Creek Channel collectively, or individually displayed constant, behavioral and catastrophic drift or a combination of the three drift types. Drift of the three taxa increased during the course of our study due to colonization of the stream channels themselves and increased recruitment from Mosquito Creek. High drift propensity has been associated with different life stage (Waters, 1972; Lehmkuhl and Anderson, 1972) and increased production (Waters, 1961; Hall, Waters and Cook, 1980). Several investigations (Rosenberg and Wiens, 1975; White and Gammon, 1976) have found a catatostrophic drift response by aquatic insects to sediment loading. Drift patterns were relatively similar among all three Mosquito Creek channels until volcanic ash was introduced and resulted in "catastrophic" drift.

A bimodal drift response was evidenced by B. bicaudatus, i.e. a diel drift response having two identifiable peaks. Low numbers of B. bicaudatus drifted during the day, followed by two peaks during darkness --the largest one just after dark and another prior to sunrise. Baetis bicaudatus drift in the control channels and in the test channel on the day before ash input displayed the "alteruns pattern" described by Waters, (1972) which had a low pulse after sunset and a large peak before sunrise. The effect of ash on B. bicaudatus drift was an immediate response during daylight hours followed by a shift in the peak period of behavioral drift.

Like B. bicaudatus, volcanic ash had a pronounced impact on the drift of Chironomidae. Catastrophic drift occurred during the daytime when there was normally a relatively high level of constant drift. Catastrophic drift continued throughout the evening, then declined somewhat by dawn. The midge drift pattern in the test channel closely followed the ash loading curve. We believe volcanic ash affected midges immediately and in a sustained way because of the tendency for midges to drift continuously irrespective of a light-triggering mechanism associated with many other stream insects (Waters, 1972).

The stonefly Y. brevis showed a catastrophic drift response to ash loading. The response occurred during a period of normally high behavioral drift. Despite the high inputs of volcanic ash during the day, catastrophic drift of Y. brevis was mostly delayed until dark. This delayed drift response may be due to a combination of diel activity rhythms and ash loading. Yoraperla brevis is secretive during the day, hiding in the substrate and becoming active after sunset. The massive exodus of Yoraperla was disproportionately large when compared to its density in the substrate and densities of the other major taxa. We propose that feeding inhibition caused by ash coatings in leaf litter and attached benthic algae is a possible explanation for the delayed drift response to ash inputs. We collected drift samples at 2100 h, the time of maximum Y. brevis drift on the major impact day, and one and two days after the simulated ash fall. Few Y. brevis drifted during the latter two days, therefore suggesting the population had been severely depleted.

Catastrophic drift of B. bicaudatus and Chironomidae during the 24 hours after the initial ash introduction substantially reduced their densities in the test channel. The large density reduction of Y. brevis was not similarly substantiated from canister sample analysis, but only conjectured, because of the overall paucity of this species in our density estimate both before and after ash introduction. The drift samples, which, in effect, sampled the entire width and depth of the channel, were clearly more effective in detecting benthic insect responses to ash than were standing crop estimates. The latter did provide, however, supporting evidence on ash impact, especially for the most abundant species. Benthic insect densities differed markedly between channel sections (i.e. upstream vs. downstream sections). This variability was likely due to a combination of three factors: algal mat development, differential colonization from upstream areas, and the upstream sections were more impacted by ash. Luxuriant algal mats covered much of the downstream sections, but were largely absent from the upper sections. High numbers of Chironomidae have been associated with dense growths of filamentous algae (Mundie, 1971). These downstream algal mats may have provided better refuge for insects when subjected to ash and served as collection points for drifting insects. Prior to ash injections, upstream and downstream canisters contained similar numbers of Chironomidae. However, after impact there were substantially more midges in the lower reaches.

The distance insects drift during a 24-h period is a point differing views among researchers. Some investigators reported daily drift migration of up to 100 m by aquatic insects (Waters, 1965; McClay, 1970), others (Elliot, 1971; Townsend and Hildrew, 1976) concluded that the majority of drifting insects travelled less than 2 m during a 24-h day. Although not empirically determined in this study, we speculate, based upon previous investigations, that numerous insects would traverse the entire length of the channel in a single day. Short intermediate or long stops in downstream drift would be fostered by the presence of algal mats which extend into the water column and provide refuge for drifting insects (Maurer and Brusven, 1983).

Substrate impaction by sand and silt has been shown to severely reduce colonization and standing crop of benthic insects (Cordone and Kelly, 1961; Cummins and Lauff, 1969; Brusven and Prather, 1974; Luedtke and Brusven, 1976; Bjornn et al. 1977). The surface appearance of the ash-coated substrates in our experimental channels would suggest that the microhabitat was severely altered. The bottom density of the benthic insect community was appreciably reduced as expected.

Recovery of benthic insect populations in the Mosquito Creek test channel proceeded slowly after ash inputs were halted. We found a substantial reduction in drifting insects in the test channel one week after ash injection. Sediment analysis indicated that appreciable volcanic ash remained in the channel on 11 August, one week after the principal ash introduction. The reduced settle-out probably reflected decreased habitat quality due to the heavily embedded condition of the substrate. By 20 August, ash on the substrate in the test channel had been appreciably displaced; also, drift cleansing of the substrate combined with increased benthic insect recruitment from drift suggests that the insect standing crop was beginning to recover two weeks after ash impact.

The July 1981, Palouse River data cannot empirically assess recovery of benthic invertebrates in terms of standing crop due to the lack of pre-impact density and biomass data. However, the close habitat affinities found among the Palouse River insects and insects from the Mosquito Creek Channel indicate that recovery from the 1980 Mt. St. Helens eruption had likely occurred during the two years following the eruption. The present fauna reflect a diversity and abundance, similar to insect communities in other northern Idaho streams unaffected by the 1980 eruption.

5.4 Algal Leaching

5.4.1 Leaching

Leaf leachates contain both an organic and an inorganic fraction. The organic fraction consists primarily of sugars, organic acids, and polyphenols. Soluble organic compounds are 5 to 30 percent of the dry weight of the organic portion of plants Alexander (1977) and Nykvist (1961) reported that 33 percent of the inorganic fraction of birch (Betula verrucosa) was water soluble. For green norway maple leaves, about 15% of the original dry weight was leached in three days. This is similar to the mean leaching loss for several species determined by Petersen and Cummins (1974). The leached portion of detritus is decomposed more readily than the non-soluble fraction. Nykvist (1961) reported a biological half life for birch litter leachates in an artificial system of less than 21 days). Soluble portions are therefore more readily available as a nutrient source to plants, insects, and eventually fish.

Both ash and soil seemed to enhance the amount of material leached during the first three days; the enhancement by soil was more pronounced than that by the ash. The enhancement was significant only with a Duncan's multiple range test and not with a comparison of decay coefficients, however. This effect, if valid, is surprising since even a small amount of sediment should reduce circulation across the leaf surface. In this experiment the importance of circulation may have been minimized by capping the jars. The abrasiveness of the sediments may have caused a higher material leaching in those treatments.

5.4.2 Leaf Pack Experiments

Organic matter is used by the microflora as both an energy source and a carbon source. The most readily available constituent is the water-soluble fraction discussed above. Other fractions, in order of increasing resistance to microbial breakdown, are hemicellulose, cellulose (usually the most abundant fraction), lignin, ether and alcohol soluble constituents (fats, waxes, resins, polyphenols), proteins, and inorganics (Alexander, 1977). The overall rate of decay is usually modelled by a single exponential equation which assumes that the weight loss in a given time period is proportional to the weight at the beginning of that time period. Weider and Lang (1982) discuss this model and several others. The single exponential model fit our data very well for all treatments.

Green leaves were used for the experiment rather than newly abscissed leaves due to time constraints. This probably resulted in a faster decomposition rate than would normally be experienced since during scenescence the percentage of the water-soluble fraction decreases (Alexander, 1977). Furthermore, the amount of polyphenolic materials (which inhibit microbial growth, decrease macroconsumer palatability, and may form recalcitrant complexes) increases (Edwards and Heath, 1975; Suberkropp, et al., 1976). We are assuming, however, that the relative differences between treatments are unaffected by leaf age.

Both leaf pack experiments support the conclusion that an ash coating decreased the overall amount of leaf pack decomposition. Other investigators have had similar results with other fine sediments. Ou and Alexander (1974) found that a 0.38-cm layer of glass microbeads drastically inhibited chitin degradation, and Reice (1975) found that a silt substrate caused more of a decrease in the breakdown of leaf litter than did larger sediments. The faster decay rate for the ash than for the soil treatments may be accounted for by the slight particle size differences between ash and soil or the soil used may have been contaminated by pesticides or herbicides; there is no evidence to suggest that ash affects decay rates by a different mechanism than other fine sediments. Herbst (1980) offered five explanations for decreases in detrital processing rates due to complete burial, two of which might apply here: 1) anaerobic conditions, and 2) decreased colonization by microorganisms and invertebrates.

Anaerobic conditions result in a lower energy yield per unit organic carbon decomposed, fewer organisms, and slower decay rates (Alexander 1974). Reddy and Patrick (1975) found CO_2 evolution from soil and straw in a flooded anerobic chamber to be half that of aerobic conditions with very little C loss as CH_4. Assuming the decay rate of anaerobic leaf disks is half the aerobic rate, then if anaerobiosis alone were limiting decay rates in our experiment, the average effective amount of each sediment treatment jar that would have to be anaerobic is 33 percent (calculated from the average decay coefficient for treatments with sediment compared to the average decay coefficient for treatments without sediment). Although this figure is only a rough estimate, typically less than 10 percent of the leaf disks in sediment treatments were completely buried and hence presumably anaerobic; an additional 15 percent may have been affected to a lesser degree by ash deposits (based on observations). We can therefore conclude that at least one other factor than oxygen availability is limiting the decay rates in sediment treatments.

The ash and fine soil may act as a barrier to microorganism colonization, even when leaf disks are not completely buried. Ou and Alexander (1974) found that a layer of 29 micron microbeads probably serves as a mechanical barrier to microbial movement. A thin layer of ash or soil, which contains particles smaller than 29 microns, would be sufficient to interfere with microbial use of a leaf surface. Total microbial exclusion from 16 percent of the sediment treatment leaf disks would explain our results. The green color of the completely buried leaves suggests that this may be a factor; without microbial exclusion, one would expect the typical black color of anaerobic decomposition. The actual differences found in the amount of decomposition between treatments is most likely the result of a combination of anaerobic conditions and mechanical blockage of microbial movement.

The reported importance of detritus as a direct food source for macroinvertebrates is variable. Mathews and Kowalczewski (1969) reported that disappearance of leaf litter in the River Thames was unaffected by the presence or absence of a normal fauna. Triska's (1970) decomposition rates with invertebrates excluded are similar to rates reported with invertebrates present. Triska's data were recalculated by Petersen and Cummins (1974); conversly, Short, et al. (1980) attributed their fast decay rates to a high biomass of shredders per gram of leaf pack. Petersen and Cummins (1974) found that the presence of detritivores had a marked effect on the rate of hickory leaf processing. In Mosquito Creek, macroinvertebrates were probably not important to the detrital decomposition process since the total number of shredders was so small. The difference in decay rates between with- and without macroinvertebrate treatments is more likely due to differences in water exchange rate than to consumption rates of insects. The detritus is directly important to both the shredders and collectors, however, since the fine particulate organic matter (FPOM) derived from microbially conditioned leaves is higher in food value than natural stream FPOM (Ward and Cummins, 1979). Invertebrates were found mostly in the upper portion of the leaf pack in both control and ash treatments. Since the upper leaf disks appeared to be only marginally disturbed by ash, it is not surprising that there was little difference in insect numbers of species, and only a slight difference in diversity between ash treatments and controls.

Although a coating of ash decreases the decomposition rate of leaf packs, this does not mean that the nutrients stored in the buried leaves are lost to the system. Burial of detritus by sediment can be viewed as a storage mechanism; the leaf material may be released to the system later. Herbst (1980) likens detrital burial to a damping effect on the seasonal oscillations of allochthonous energy and nutrient input. The amount of buried detritus that would be permanently lost to the system is unknown, but in Mosquito Creek that loss is likely to be small.

Summary

This study employed an interdisciplinary approach to assess the effects of Mt St. Helens volcanic ash on natural and artificial stream ecosystems. Physical and biological scientists in the fields of Engineering, Limnology, Aquatic Entomology and Stream Ecology were included in the study team. The objectives of the study addressed: 1) the assessment of microhabitat characteristics affected by volcanic ash in a representative stream system; 2) the determination of the extent and movement of ash through a stream system; and 3) the evaluation of the effects of suspended and deposited volcanic ash on primary and secondary production in natural and artificial stream channels.

We studied a natural stream (Palouse River-northern Idaho), and artificial channels in the laboratory and field (Mosquito Creek-northern Idaho). The morphometric characteristics of the naturally ash-impacted stream were measured by conventional survey techniques to assess sinuosity, cross-section configurations and general habitat characterization of riffles, runs and pools. The streambed in selected habitats was evaluated structurally by freeze core removal of samples. Particle size distribution was determined by sieve analysis; recognition of volcanic ash was by spectrophometric analysis. Macrobenthic invertebrates were sampled with a modified Hess Sampler from common data-generating locations selected by the interdisciplinary team.

In the Mosquito Creek experimental channel 0.0016 m^2 canisters were randomly imbedded and sampled to evaluate the impacts of mechanically applied volcanic ash on the physical and biotic characteristics in the channel. Specially designed drift nets encompassing the entire widths of the channels, were used to measure sediment transport and invertebrate drift.

The long-term, natural effects of volcanic tephras on a stream ecosystem were investigated at the Graves Meadow ranch of the Palouse River one year after the May 18, 1980 Mt. Saint Helens volcanic eruption. The algae and aquatic insect populations were found to be diverse and abundant and distributed in a manner indicative of streams in the region unimpacted by the May 18 eruption. We conclude that one year after the catastrophic eruption, the stream environment in the reach studied had nearly fully recovered from the event.

Nevertheless, in controlled laboratory experiments we assessed short-term impacts of catastrophic and low-level ashfall events. Both catastrophic and chronic low-level ash treatments initially suppressed attached benthic algae standing crops and chlorophyll "a" levels. The catastrophically-loaded channel ABA remained depressed through the eight-week study, but the low-level loaded channel responded with recovery after two weeks with growth exceeding ABA concentrations in the control channel.

Subjecting leaf-shredding insects, e.g. the stonefly *Pteronarcys californica* and the caddisfly *Hesperophylax occidentalis*, to ash suspension levels similar to those occurring during the May 18th eruption for periods of 14 days reduced food consumption by half. Simulated ash injection into the Mosquito Creek experimental channels at levels comparable to the May 18 natural volcanic deposition in northern Idaho caused a catastrophic increase (4X) in insect drift and a moderate to large decrease in standing crop at or near the point of impact. A delayed density response occurred in the lower reach of the impacted channel. The mean density of the principal insect species during the three weeks of the impact test was reduced to approximately half that in the control channel. Within one week following the simulation test, overall numbers of drifting insects were similar in the test and control channels. However, the settle-out rate from drift in the impacted channels was substantially reduced (1/3) compared to the control channel. After three weeks, the settle-out rate in the impacted channel had returned to normal.

The simulated ash deposition effects on the ecology of streams provided an opportunity to recreate with considerable detail and precision the phenomena associated with volcanic tephra. The results obtained clearly indicate catastrophic influences of volcanic ash on attached benthic algae biomass and the dislocation of invertebrates and interruption or inhibition of food chain relationships. While the extreme presence of suspended ash was only apparent for a few weeks in natural streams the depositional presence was apparent much longer. The latter was diminished by spates and natural hydrological events.

While the short-term impact was moderate to severe on selected populations, all of the prinicpal species tested demonstrated considerable resilience and persistence, assuring perpetration into future. One can infer that single, catastrophic events as experienced over northern Idaho from Mt. Saint Helens eruption did not cause the demise of primary and secondary production elements normally characteristic of healthy community structure, but produced a pronounced, albeit short-term oscillation over time. Repeated catastrophic events at levels comparable to or greater than those experienced during 18 May, 1980 may have profoundly different effects, however.

Watersheds are the recipients of ash deposition--streams and rivers are the conduits that transport ash, at least in part, from the system. Vegetation, climate, topography and time are the principle variables that tend to influence the disposition of volcanic ash occurring as a natural, but rare phenomenon of nature in North America.

Literature Cited

Alexander, M. 1977. Introduction to soil microbiology. Wiley, New York, New York, USA.

American Public Health Association. 1981. Standard methods for the examination of water and wastewater. In conjunction with the American Water Works Association and the Water Pollution Control Federation. Washington, D.C. 1134 p.

Anderson, N.H. and J.R. Sedell. 1979. Detritus processsing by macroinvertebrates in stream ecosystems. Annual Review of Entomology 24:351-77.

Austin, A., S. Lang, and M. Pomeroy. 1981. Simple methods for sampling periphyton with observations on sampler design criteria. Hydrobiologia 85(1):33-48.

Benfield, E.F., D.S. Jones, and M.F. Patterson. 1977. Leaf pack processing in a pastureland stream. Oikos 29:99-103.

Berrie, A.D. 1976. Detritus, micro-organisms and animals in fresh water. Pages 323-338 in J.A. Anderson, and A. Macfadyen, editors. The role of terrestrial and aquatic organisms in decomposition processes. The 17th Symposium of the British Ecological Society 15-18 April 1975. Blackwell Scientific Publications, Oxford, England.

Bisson, P.A. 1978. Diel food selection by two sizes of rainbow trout Salmo gairdneri in an experimental stream. Journal of Fisheries Research Board of Canada 35:971-975.

Bjornn, T.C., M.A. Brusven, M.P. Molnau, J.H. Milligan, R.A. Klamt, E. Chacho and C. Schaye. 1977. Transport of granitic sand and its effects on insects and fish. For. Wildlife Range Experiment Station Bulletin 17, University of Idaho, Moscow, 43 pp.

Bohle, H.W. 1978. Relation between food supply, drift and microdistribution of larvae of Baetis rhodani: investigation in a stream model. Archiv fur Hydrobiologie 84(4):500-524.

Boling, R.H., Jr., E.D. Goodman, J.A. Van Sickle, J.O. Zimmer, K.W. Cummins, R.C. Peterson, and S.R. Reice. 1975. Toward a model of detritus processing in a woodland stream. Ecology 56:141-151.

Bothwell, M.L., and J.G. Stockner. 1980. Influence of secondarily treated kraft mill effluent on the accumulation rate of attached algae in experimental continuous-flow troughs. Canadian Journal of Fisheries & Aquatic Sciences 37:248-254.

Bott, T.L., J.T. Brock, C.E. Cushing, S.V. Gregory, D. King, and R.C. Peterson. 1978. A comparison of methods for measuring primary productivity and community respiration in streams. Hydrobiologia 60:3-12.

Brusven, M.A. 1973. A closed system plexiglass stream for studying insect-fish-substrate relationships. Prog. Fish Culturist 35:87-89.

Brusven, M.A. and C.E. Hornig. In Press. Effects of volcanic ash on acute toxicity and substrate preferences of stream insects. J. Kansas Entomol. Soc.

Brusven, M.A. and K.V. Prather. 1974. Influence of stream sediments on distribution of macrobenthos. J. Entomol. Soc. Brit. Columbia 71:25-32.

Busch, D.E. and C.D. McIntire. 1976. Plant production in experimental streams: A cooperative research project with Weyerhaeuser Company. Water Resources Research Institute, Oregon State University, Corvallis, Oregon, USA.

Cairns, J. Jr., R.L. Dickson, J. Smrcher, P.H. King, C.W. Randall, J.H. Crowe, D.R. Huber. 1973. Systems simulation of the effects of tertiary treatment for carbon, nitrogen, and phosphorus removal upon primary productivity, standing crop and community structure of autotrophic and hetertrophic communities in receiving model streams. Virginia Polytechnic Institute and State University, Office of Water Resources Research, PB-231-759, Springfield, VA, USA.

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

Cummins, K.W. and G.H. Lauff. 1969. The influence of substrate particle size on the microdistribution of stream macrobenthos. Hydrobiologia 34:145-181.

Edwards, C.A. and G.W. Heath. 1975. Studies in leaf litter breakdown III. The influence of leaf age. Pedobiologia 115:348-354.

Elliot, J.M. 1971. The distances travelled by drifting invertebrates in a Lake District Stream. Oecologia 6:360-379.

Eloranta, P. and S. Kunnas. 1979. The growth and species communities of the attached algae in a river system in central Finland. Archiv fur Hydrobiologie 86(1):27-44.

Fisher, S.G. and G.E. Likens, 1972. Stream ecosystem: organic energy budget. Bio Science 22:33-35.

Flanagan, P.W. and F.L. Bunnell. 1976. Decomposition models based on climatic variables, substrate variables, microbial respiration and production. Pages 437-457 in J.M. Anderson and A. Macfadyen, editors. The role terrestrial and aquatic organisms in decomposition processes. The 17th Symposium of the British Ecological Society 15-18 April 1975. Blackwell Scientific Publications, Oxford, England.

Frenzel, S.A. 1983. Effects of volcanic ash on the benthic environment of a mountain stream, northern Idaho. U.S. Geological Society Water-Resource Investigation 82-4106. Boise, ID 32 pp.

Fruchter, J.S., D.E. Robertson, J.C. Evans, K.B. Olsen, E.A. Lepel, J.C. Laul, K.H. Abel, R.W. Sanders, P.O. Jackson, N.S. Wogman, R.W. Perkins, H.H. Van Tuyl, R.H. Beauchamp, J.W. Shade, J.L. Daniel, R.L. Erickson, F.A. Sehmel, R.N. Lee, A.V. Robinson, O.R. Moss, J.K. Briant, W.C. Cannon. 1980. Mount St. Helens ash from the 18 May, 1980 eruption: chemical, physical, mineralogical, and biological properties. Science 209:1116-1125.

Furuta, Y.K. Tanaka, and T. Ito. 1977. Periphyton biomass of attached algae and biological estimation of water pollution level using attached diatoms in the Chijuma River and its tributary, Nagano Prefecture. Bulletin of Freshwater Fisheries Research Laboratory (Tokyo) 27(2):63-74.

Gardner, R.H., R.V. O'Neal, J.B. Mankin, and J.H. Carney. 1981. A comparison of sensitivity analysis and error analysis based on a stream ecosystem model. Ecological Modelling 12:173-190.

Geesey, G.G., R. Mutch, J.W. Costerton, and R.B. Green. 1978. Sessile bacteria: An important component of the microbial population in small mountain streams. Limnology and Oceanography 23(6):1214-1223.

Gerhart, D.Z., S.M. Anderson and J. Richter. 1977. Toxicity bioassays with periphyton communities: design of experimental streams. Water Research 2:567-570.

Gerisch, F.M. and M.A. Brusven. 1982. Volcanic ash accumulation and ash-voiding mechanisms of aquatic insects. Journal of the Kansas Entomological Soceity 55(2):290-296.

Gill, J.L. 1981. Evolution of statistical design and analysis of experiments. Journal of Dairy Science 64(6):1494-1519.

Greene, E.A. 1982. Characterization and movement of Mount St. Helens tephra on a high mountain lake watershed in northern Idaho. Unpublished Masters Thesis, University of Idaho, Moscow, ID, USA.

Hall, Ronald T., Thomas F. Waters and Edwin F. Cook. 1980. The role of drift dispersal in production ecology of a stream mayfly. Ecology 61(1):37-43.

Hamala, J.A., S.W. Duncan, and D.W. Blinn. 1981. A portable pump sampler for lotic periphyton. Hydrobiologia 80:189-191.

Harvey, R.S. 1975. Baseline studies of Upper Three Runs Creek and the experimental streams of the artificial stream laboratory. United States Energy Research and Development Administration, Contract No. AT(07-2)-1.

Herbst, G.N. 1980. Effects of burial on food value and consumption of leaf detritus by aquatic invertebrates in a lowland forest stream. Oikos 35:411-424.

Herbst, G.N. 1982. Effects of leaf type on the consumption rates of aquatic detritivores. Hydrobiologia 89:77-87.

Hooper, P.R., I.W. Herrick, E.R. Laskowski and C.R. Knowles. 1980. Composition of the Mount St. Helens ashfall in the Moscow-Pullman area on 18 May 1980. Science 290:1125-1126.

Hornick, L.E., J.R. Webster, and E.F. Benfield. 1981. Periphyton production in an Appalachian mountain trout stream. American Midland Naturalist 106(1):22-36.

Huntsberger, D.V. and P. Billingsley. 1981. Elements of statistical inference, fifth edition. Allyn and Bacon, Inc., Boston, USA.

Hynes, H.B.N. 1970. The ecology of running waters. Liverpool University Press, Liverpool, England.

Kaufman, L.H. 1980. Stream aufwuchs accumulation processes: effects of ecosystems depopulation. Hydrobiologia 70:75-81.

Kelly, M.G., G.M. Hornberger, and B.J. Cosby. 1974. Continuous automated measurement of rates of photosynthesis and respiration in an undisturbed river community. Limnology and Oceanography 19(2): 305-312.

Kureknov, I.I. 1966. The influence of volcanic ashfall on biological processes in a lake. Limnology and Oceanography 11(3):426-429.

Ladle, M., J.S. Welton, and J.A.B. Bass. 1980. Invertebrate colonization of the gravel substratum of an experimental recirculating channel. Holarctic Ecology 3:116-123.

Lehmkuhl, D.M. and N.H. Anderson. 1972. Microdistribution and density as factors affecting the downstream drift of mayflies. Ecology, 53(4):661-667.

Lemly, A.D. 1982. Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. Hydrobiologia 87:229-245.

Luedtke, R.J. and M.A. Brusven. 1976. Effects of sand sedimentation on colonization of stream insects. Journal of Fisheries Research Board of Canada 33(9):1881-1886.

McLay, C. 1970. A theory concerning the distance travelled by animals entering the drift of a stream. Journal of Fisheries Research Board of Canada 27(2):359-370.

Madoni, P. and P.F. Ghetti. 1980. Etude de la dynamique des populations de cilies d'un torrent experimental pendant deux annees. Hydrobiologia 74:273-282.

Marcus, J.H., D.W. Sutcliffe, and L.G. Willoughby. 1978. Feeding and growth of *Asellus aquaticus* (Isopoda) on food items from the littoral of Windermere including green leaves of *Elodea canadensis*. Freshwater Biology 8:505-519.

Matthews, C.P. and A. Kowalczewski. 1969. The disappearance of leaf litter and its contribution to production in the River Thames. Journal of Ecology 57:543-552.

McIntire, C.D. and J.A. Colby. 1978. A hierarchial model of lotic ecosystem. Ecological Monographs 48:167-190.

McKnight, D.M., G.L. Feder and E.A. Stiles. 1980. Toxicity of Mount St. Helens ash leachate to a blue-green alga. Geological Survey Circular 850-F.

Merritt, R.W. and K.W. Cummins, editors. 1978. An introduction to the aquatic insects of North America. Kendall and Hunt, Dubuque, Iowa, USA.

Minshall, G.W. 1967. Role of allochthonous detritus in the trophic structure of a woodland springbrook community. Ecology 48(1):139-149.

Moore, J.W. 1977. Some factors effecting algal densities in a eutrophic farmland stream. Oecologia 29:257-267.

Mundie, J.H. 1971. The diel drift of Chironomidae in an artificial stream and its relation to the diet of coho salmon fry. *Oncorhynchus kisutch* (Walbaum) Can. Entomol. 103:289-297.

Naiman, R.J. and J.R. Sedell. 1979. Benthic organic matter as a function of stream order in Oregon. Archiv fur Hydrobiologie 87(4): 404-422.

Nelson, D.J. and D.C. Scott. 1962. Role of detritus in the productivity of a rock outcrop community in a piedmont stream. Limnology and Oceanography 7:396-413.

Neveu, A. 1980. Influence d'une fine sedimentation dans un canal experimental sur la densite du macrobenthos sa composition et sa composition et sa consommation par des salmonides. Bulletin Francais de Pisciculture 276:104-122.

Nykvist, N. 1961. Leaching and decomposition of litter III. Experiments of lean litter of *Betula verrucosa*. Oikos 12(2):249-263.

Olson, J.S. 1963. Energy storage and the balance of producers and decomposers in ecological systems. Ecology 44(2):322-331.

Ou, L. and M. Alexander. 1974. Effects of glass microbeads on the microbial degradation of chitin. Soil Science 118(3):164-167.

Paerl, H.W. 1973. Detritus in Lake Tahoe: structural modification by attached microflora. Science 180:496-498.

Peterson, R.C. and K.W. Cummins. 1974. Leaf processing in a woodland stream. Freshwater Biology 4:343-368.

Reddy, K.R. and W.H. Patrick, Jr. 1975. Effect of alternate aerobic and anaerobic conditions on redox potential, organic matter decomposition, and nitrogen loss in flooded soil. Soil Biology and Biochemistry 7:87-94.

Reice, S.R. 1974. Environmental patchiness and the breakdown of leaf litter in a woodland stream. Ecology 55:1271-1282.

Rember, W.C. and E.H. Bennett. 1979. Geologic Map of the Pullman Quadrangle, Idaho, Idaho Department of Lands, Bureau of Mines and Geology.

Rodgers, J.H., Jr., K.L. Dickson and J. Cairns, Jr. 1979. A review and analysis of some methods used to measure functional aspects of periphyton. Pages 142-167 in R.L. Weitzel, editor, Methods and measurements of periphyton communities: a review, American Society for testing and materials. Espey-Houston and Assoc., Inc., Austin, Texas, USA.

Rosenberg, D.M. and A.P. Wiens. 1975. Experimental sediment addition studies on the Harris River, N.W.T. Canada: The effect of macroinvertebrate drift. Verh. International, Verein. Limnology 19: 1568-1574.

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

Sarna-Wojcicki, A.M., J.S. Shipley, R.B. Waitt, Jr., D. Dzurisin, and S.H. Wood. 1981. Area distribution, thickness, mass, volume, and grainsize of air-fall ash from the six major eruptions of 1980. p. 577-600. In: Lipmons, P.W. and D.R. Mullineaux, editors. The 1980 eruptions of Mount St. Helens, Washington. USGS Professional Paper 1250.

Saunders, G.W. 1976. Decomposition in fresh water. Pages 341-373 in J.W. Anderson and A. Macfadyen, editors. The role of terrestrial and aquatic organisms in decomposition processes. The 17th Symposium of the British Ecological Society 15-18 April 1975. Blackwell Scientific Publications, Oxford, England.

Shanks, R.E. and J.S. Olson. 1961. First year breakdown of leaf litter in southern appalachian forests. Science 134:194-195.

Shelly, T.E. 1979. The effect of rock size upon the distribution of species of Orthocladiinae (Chironomidae: Diptera) and *Baetis intercalaris* McDunnough (Baetidae: Ephemeroptera). Ecological Entomology 4:95-100.

Short, R.A., S.P. Canton and J.V. Ward. 1980. Detrital processing and associated macroinvertebrates in a Colorado mountain stream. Ecology 61(4):727-732.

Skille, J.M., C.M. Falter and W. Kendra. 1983. Effects of volcanic tephra from the Mount St. Helens eruption on plankton and benthic communities of the Coeur d'Alene Lake deltas. Completion Report to OWRT, Washington, D.C. June 1983.

Smith, M.E. and J.L. Kaster. 1981. Improved stream sampler for periphyton. Progressive Fish Culturist 43(3):124-125.

Smith, O.L. 1979. An analytical model of the decomposition of soil organic matter. Soil Biology and Biochemistry 11:585-606.

Statistical Analysis System (SAS). 1979. User's guide. SAS Institute, Sparks Press, Raleigh, North Carolina, USA

Suberkropp, K.F., G.L. Godshalk and M.J. Klug. 1976. Changes in the chemical composition of leaves during processing in a woodland stream. Ecology 57:720-727.

Suberkropp, K.F. and M.J. Klug. 1974. Decomposition of deciduous leaf litter in a woodland stream I. A scanning electron microscope study. Microbial Ecology 1:96-103.

Suberkropp, K.F. and M.J. Klug. 1976. Fungi and bacteria associated with leaves during processing in a woodland stream. Ecology 57: 707-719.

Teal, J.M. 1957. Community metabolism in a temperate cold spring. Ecological Monographs 27:283-302.

Tett, P., C. Gallegos, M.C. Kelly, G.M. Hornberger, and B.J. Cosby. 1978. Relationships among substrate, flow, and benthic microalgal pigment density in the Mechums River, Virginia. Limnology and Oceanography 23(4):785-797.

Townsend, C.R. and A.G. Hildrew. 1976. Field experiments on the drifting, colonization and continuous redistribution of stream benthos. Journal Animal Ecology 45:759-772.

Triska, F.J. 1970. Seasonal distribution of aquatic hyphomycetes in relation to the disappearance of leaf litter from a woodland stream. Doctoral Dissertation. University of Pittsburgh, Pittsburgh, Pennsylvania, USA.

Triska, F.J. and J.R. Sedell. 1976. Decomposition of four species of leaf litter in response to nitrate manipulation. Ecology 57:783-792.

Trotter, D.M. and A.C. Hendricks. 1979. Attached, filamentus algal communities. Pages 58-69 in R.L. Weitzel, editor. Methods and measurements of periphyton communities: a review, American Society for Testing and Materials. Espey-Houston and Assoc., Inc., Austin, Texas, USA.

Usinger, R.L., (ed.). 1956. Aquatic Insects of California. University of California Press, Berkeley. 508 pp.

Visco, S., R. Coler and O.T. Zajicek. 1979. An evaluation of benthic nutrient regeneration in a projected flood control impoundment. Journal of Environmental Science Health A14(5):399-414.

Waldbauer, G. 1968. The consumption and utilization of food by insects. Adv. Insect Physiol. 5:229-288.

Ward, G.M. and K.W. Cummins. 1979. Effects of food quality on growth rate and life history of a stream collector (Paratendipes albimanus (Meigen)). Ecology 60:57-64.

Waters, T.F. 1961. Standing crop and drift of stream bottom organisms. Ecology 42:532-537.

Waters, T.F. 1965. Interpretation of invertebrate drift in streams. Ecology 46:327-334.

Waters, T.F. 1972. The drift of stream insects. Ann. Rev. Entomol. 17:253-272.

Weber, T.W. II. 1980. Response of phytoplankton to acidification in experimental streams. United States Environmental Protection Agency, National Technical Information Service. EPA-600/3-81-042.

Weber, C.I., editor. 1973. Biological field and laboratory methods for measuring the quality of surface waters and effluents. United States Environmental Protection Agency, EPA-670/4-73-001, Cincinnati, Ohio, USA.

Weider, R.K. and G.E. Lang. 1982. A critique of the analytical methods used in examining decomposition data obtained from litter bags. Ecology 63(6):1636-1642.

Welton, J.S. 1980. Dynamics of sediment and organic detritus in a small chalk stream. Archiv fur Hydrobiologie 90(2):162-181.

Wetzel, R.G. 1979. The role of the littoral zone and detritus in lake metabolism. Archiv fur Hydroibologie Beihefte Ergebinisse der limnologie 13:145-161.

White, D.S. and J.R. Gammon. 1977. The effect of suspended solids on macroinvertebrate drift in an Indiana creek. Pro. Indiana Acad. Sci. 86:182-188.

Whitman, R.P., T.P. Quinn, and E.L. Brannon. 1982. Influence of suspended ash on homing behavior of adult chinook salmon. Transactions of American Fisheries Society 111:63-69.

Wissmar, R.C., A.H. Devol, A.E. Nevissi, and J.R. Sedell. 1982. Chemical changes of lakes within the Mount St. Helens blast zone. Science 216:175-178.

Wissmar, R.C., A.H. Devol, J.T. Staley, and J.R. Sedell. 1982. Biological responses of lakes in the Mount St. Helens blast zone. Science 216:178-181.

Witkamp, M. and J.S. Olson. 1963. Breakdown of confined and nonconfined oak litter. Oikos 14(2):138-147.