Effects of Releases of Sediment from Reservoirs on Stream Biota

by

James V. Ward

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EFFECTS OF RELEASES OF SEDIMENT FROM RESERVOIRS ON STREAM BIOTA

By

Lawrence J. Gray and James V. Ward

Department of Zoology and Entomology

Colorado State University

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COLORADO WATER RESOURCES RESEARCH INSTITUTE
Colorado State University
Fort Collins, Colorado

Norman A. Evans, Director

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ABSTRACT

EFFECTS OF RELEASES OF SEDIMENT FROM RESERVOIRS ON STREAM BIOTA

Evaluation of temporal and longitudinal effects of sediment releases from reservoirs on downstream physical-chemical conditions, benthic algae, and macroinvertebrates was conducted at Dry Creek-Reservoir No. 15, Colorado, and North Platte River-Guernsey Reservoir, Wyoming. In Dry Creek, flushing of sediments using stored water increased discharge, scoured stream substrates, and altered concentrations of dissolved constituents. Suspended solids increased downstream as substrates were brought into suspension. Algal abundance decreased 90%+ and did not recover for 9 months. Invertebrates were decimated with the initial release. Although full recovery was possible in 2-3 months, further releases reduced recovery rates and shifted community composition to small, fast-growing species. Upstream flows were used to flush sediments from Guernsey Reservoir. Suspended solids concentrations in the North Platte River increased from < 20 mg 1^{-1} to > 300 mg 1^{-1}. Because fine particulates remained in suspension, mean particle size of stream substrates was unaltered. Other water quality parameters were unaffected, except for an increase in phosphorus. Higher phosphorus levels contributed to an increase in benthic algae despite reduced light penetration. Densities of chironomids decreases 90%+ during sediment release from increased drift rates and mortality, but densities recovered to initial levels in 3 weeks after the release ended. Densities of mayflies and oligochaetes increased. Changes in benthic populations were highly correlated with changes in suspended solids concentrations.

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CHAPTER I

INTRODUCTION

1.1 Statement of the Problem

Accumulation of sediment in reservoirs and subsequent reductions in storage capacity has become an increasingly serious problem in the United States (Brown 1975). One method of extending the useful life of reservoirs is through periodic flushing of sediments (Simons 1979). Although the effects of reservoirs on many aspects of stream ecosystem dynamics have been documented (Ward and Stanford 1979), effect of sediment flushing has received little attention.

Profundal sediments behind dams and stream sediments below dams differ greatly in particle size composition, organic matter content, and chemical constituents. In the Rocky Mountains, profundal sediments are typically composed of fine silts and clays with relatively high organic matter content (2-35% by weight; Bergersen 1977, Buscemi 1961, Edmonds and Ward 1979, Wunderlich 1974). Reservoirs act as "sinks" for many inorganic and organic substances (Hannan 1979) and may contain relatively high levels of nutrients (nitrogen and phosphorus) and toxins such as trace metals (Brown 1975). In contrast, downstream substrates are composed of cobble "armor" (Simons 1979) that are low in organic content (< 1% by weight). Whereas streams are well-aerated where surface releases prevail, profundal sediments may be anaerobic, particularly during stratification (Buscemi 1961, Edmonds and Ward 1979).

Sediment flushing may thus greatly alter downstream habitats through increased bedload and suspended fine particulates, increased organic loading, enrichment with nutrients, and contribution of potential toxins. In Nisbet's

(1961) account, reservoir sediments turned the Rhone River into a mud flow and affected downstream habitats for many kilometers.

1.2 Effects of Suspended and Settled Solids on Macroinvertebrates and Primary Producers

Substrate type is a major determinent of macroinvertebrate distribution and abundance (Cummins and Lauff 1969, Cummins et al. 1966, Hynes 1970). Each species shows distinct preferences for particular substrates as a consequence of respiration requirements, food-gathering mechanisms, case-building behavior, and other life history characteristics (Cummins 1973, Cummins et al. 1966, Mackay 1977, Rabeni and Minshall 1977).

Flushing of reservoir sediments is expected to adversely affect macroinvertebrate communities through increased discharge, potential toxic effects, alteration of substrates from deposition of fine particulates, and increased suspended solids. If water stored in a reservoir is used for flushing sediments, the rapid increase in discharge downstream will scour substrates and decimate benthic organisms similar to flash floods (Siegfried and Knight 1977). Short-term toxic effects are likely if releases are low in dissolved oxygen and high in hydrogen sulfide. Effects of settled sediment vary in proportion with amounts deposited. Slight deposition has little or no effect (Bjornn et al. 1977, Rabeni and Minshall 1977), while heavy amounts greatly decrease populations (Chutter 1968, Cordone and Kelly 1961, Gammon 1970, Nuttall and Bielby 1973). In artificial stream channels, Bjornn et al. (1977) found that total numbers of species that prefer cobble substrates significantly decreased when cobbles were > two-thirds buried in silt. If settled solids persist, faunal composition is altered from a community dominated by erosional zone organisms (e.g., stoneflies, simuliids, heptageniid mayflies) to one composed of depositional forms (e.g., tubificid oligochaetes, burrowing mayflies) (Gammon 1970, Nuttall and Bielby 1973). Increases in suspended solids without deposition can affect benthic populations. Gammon (1970) found that increases in suspended material of 20-80 mg/1 above normal levels caused a 45-70% reduction in total numbers and affected all taxa (i.e., effects were non-selective). Reductions occurred as organisms drifted downstream, and drift rates were linearly related to suspended solids concentration up to 160 mg/1. Rosenberg and Wiens (1975) also found elevated drift rates when bank sediments were experimentally introduced into a stream channel. In contrast to Gammon (1970), overall drift rates were independent of suspended solids concentration, and chironomids drifted more than other organisms. Hamilton (1961) found no change in numbers of benthic organisms subjected to high concentrations of sand and silt in suspension.

Periphytic algae downstream from reservoirs typically exhibit enhanced productivity compared to unregulated streams as a consequence of elevated concentrations of nutrients (Lowe 1979). Flushed sediments are expected to greatly decrease standing crops <u>via</u> increased scouring, reductions in light intensity, smothering by settled solids, and possible toxic effects (Friant et al. 1980, Sorensen et al. 1977).

For algae and macroinvertebrates, inputs of profundal sediments could lead to higher densities and biomass than pre-release levels as a consequence of higher nutrient levels, thus resembling changes noted below sewage outfalls (see Hynes 1960). However, communities would be subject to greater fluctuation because of substrate instability (Tebo 1955).

Fluctuations would be enhanced by a return to surface releases low in suspended solids (i.e., water with greater competence to transport settled solids) (Simons 1979).

1.3 Longitudinal and Seasonal Aspects of Sediment Releases

The effects of sediment release are expected to decrease in proportion to distance downstream from deposition of materials in suspension and inflow from tributaries (Nisbet 1961). The actual distances involved will depend on amounts of sediment released, particle size composition, discharge, and channel hydraulics. Seasonal variations in effects and recovery patterns are likely to occur from differences in base flows prior to release and life history characteristics of species present. In the Rocky Mountains, dam releases are typically low in winter as reservoirs fill, high during spring runoff, and variable in late summer when releases depend on irrigation demand. The greatest impact and slowest recovery of organisms after sediment release are expected to occur in winter because of lower discharge (hence more sedimentation), lower stream temperatures, and the importance of winter as the major growth period for many macroinvertebrates. Flushing during spring runoff and summer will have lesser effects as a result of high flows (less sedimentation), warm temperatures, emergence of winter species (Stanford 1971), and subsequent recruitment of warm-water species with rapid life cycles.

1.4 Objectives

Objectives of this investigation were to:

1. Assess the short-term effects of sediment releases on physical and chemical parameters in the receiving stream.

2. Determine longitudinal and temporal recovery patterns of physical and chemical parameters following perturbation.

3. Determine associated effects on aquatic biota from modification of physical-chemical parameters due to sediment release.

4. Assess spatial and temporal recovery patterns of biotic parameters (biomass, community structure, and density) following perturbation.

5. Determine the short-term effects and the recovery potential of biotic and abiotic parameters as a function of the season when sediment is released.

CHAPTER II

DESCRIPTION OF STUDY AREAS

2.1 Dry Creek-Reservoir No. 15

Reservoir No. 15 is located on Dry Creek in the North Poudre River Basin, Larimer County, Colorado (Fig. 2.1). No releases from the reservoir had occurred from October 1980 to February 1981. On 17 February, releases were started to provide storage for spring runoff and clear downstream channels of debris.

Two sampling sites were selected for study of release effects on downstream biota. Site 1 is located immediately downstream from the dam outlet in a tributary channel that joins with Dry Creek. The channel is 2-3 m wide and incised 2-3 m into surface soils and rocks. During non-release periods, flow consists entirely of seepage. Site 2 is located 200 m downstream from the confluence of Dry Creek and the tributary channel. About 80% of non-release flow is derived from upstream springs. Stream substrates primarily consist of cobble at Site 1 and gravel at Site 2. The sparse riparian vegetation is composed of willows (Salix sp.) and grasses. The area is used for livestock grazing.

2.2 North Platte River-Guernsey Reservoir

Guernsey Reservoir, one of the first reservoirs in the North Platte Reclamation Project, is located in southeastern Wyoming near Guernsey (Fig. 2.2). From 1927 until 1957, when Glendo Reservoir was built 16 km upstream, 29,000 acre-feet of sediments accumulated in Guernsey Reservoir and reduced its capacity to 44,800 acre-feet. Since 1936, an annual "silt run" has been conducted during July and August to sluice sediments from Guernsey. The main

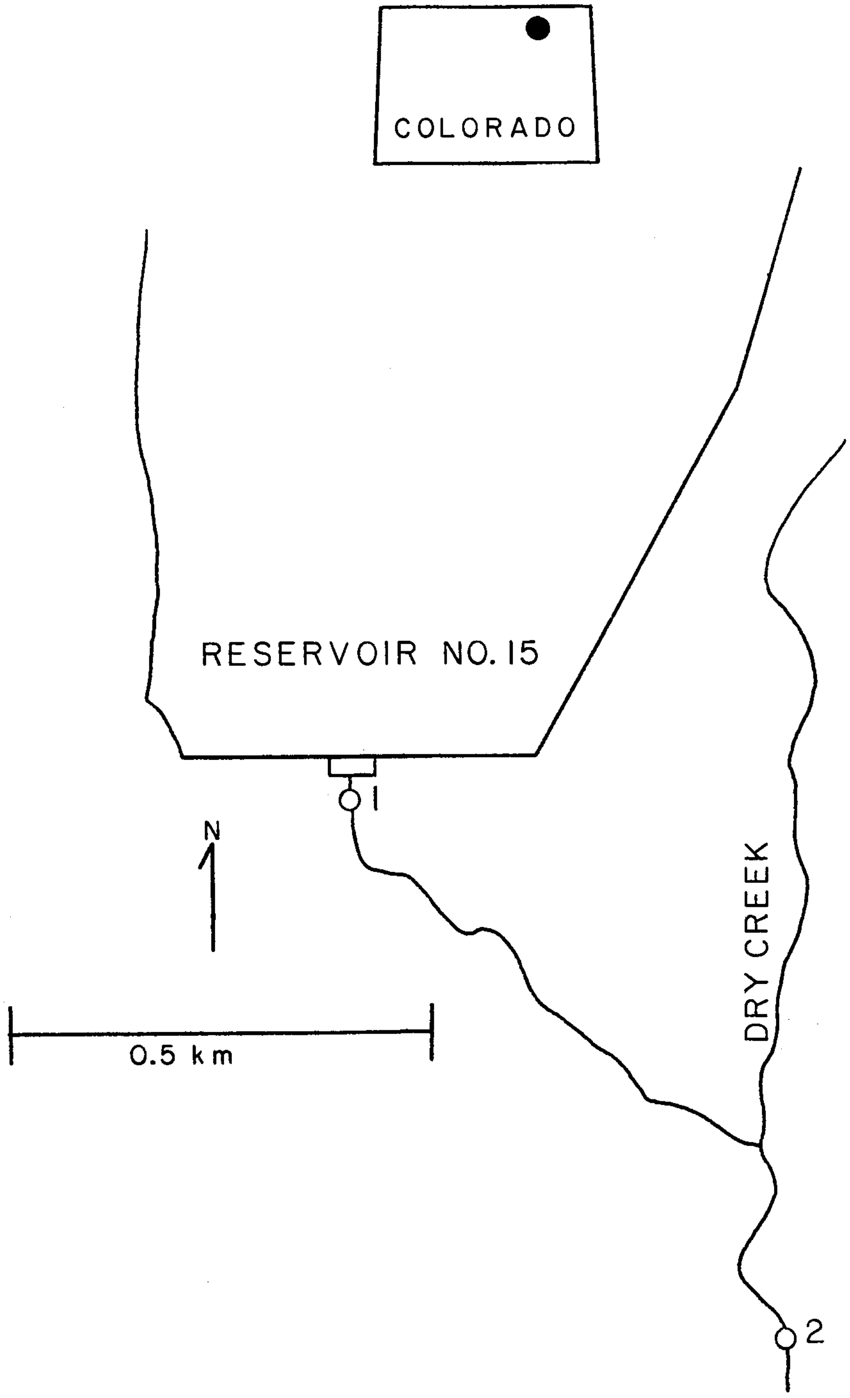
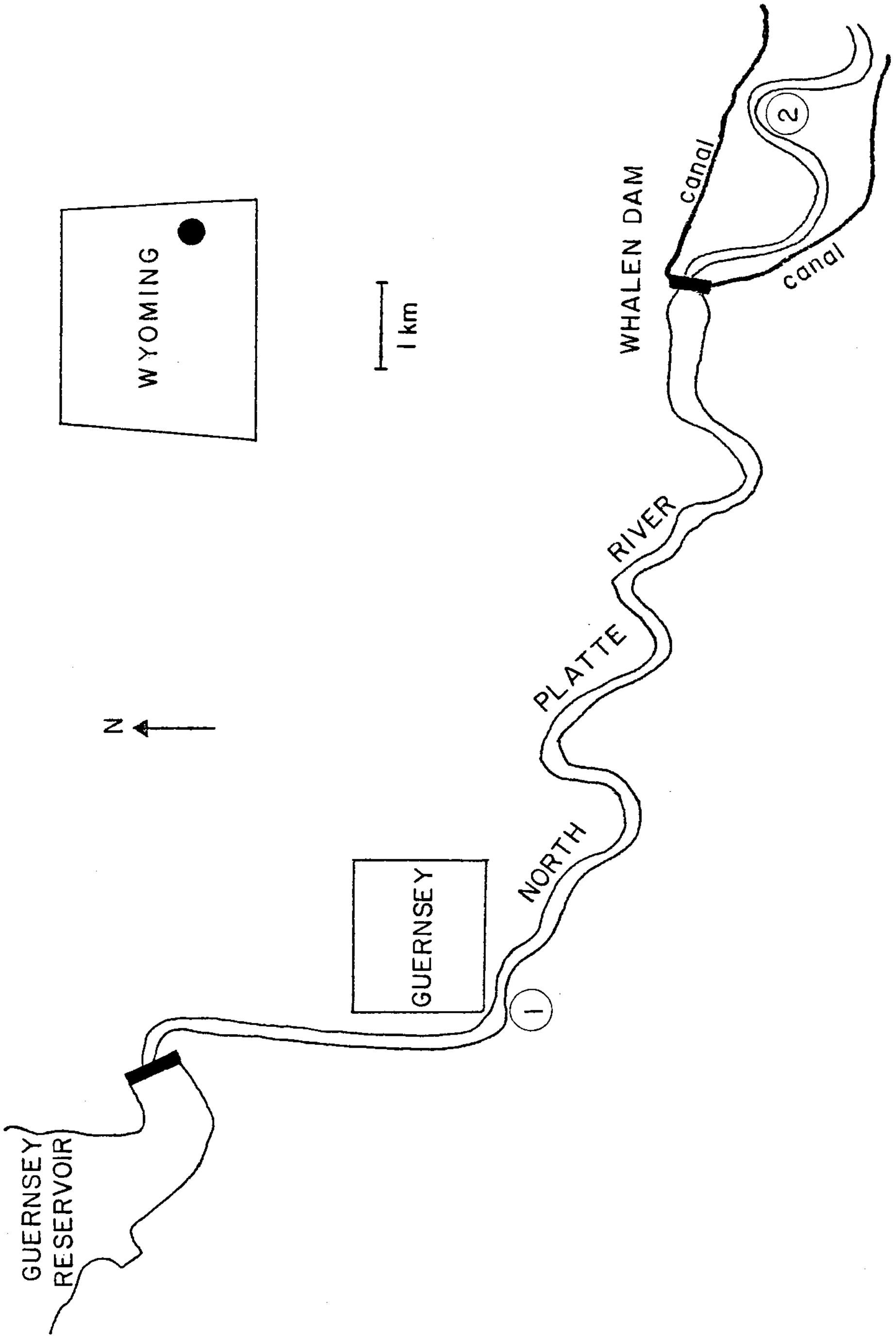


Figure 2.1. Study area at Dry Creek-Reservoir No. 15, Larimer County, Colorado.

Figure 2.2. Study area at North Platte River-Guernsey Reservoir, Platte-Goshen Counties, Wyoming. Circled numbers show locations of sampling sites.



purpose of the silt run is to reduce bank erosion and seepage losses from downstream irrigation canals.

A silt run is conducted by reducing upstream flows from Glendo Reservoir, lowering Guernsey Reservoir to less than 10% of total capacity, and using subsequent releases from Glendo to sluice sediments. Consequently, discharge and most water quality parameters (except suspended solids) show little change from pre-silt run levels. Mean concentrations of total suspended solids (TSS) are maintained at 300-400 mg/l, and this material consists of 75% silt-sized particles and 25% clay (U.S. Geological Survey 1980, U.S. Bureau of Reclamation personal communication).

Downstream from Guernsey Reservoir, the North Platte River is 50-100 m wide and 0.2-3.0 m deep during peak flows. Average stream gradient is 1 m km^{-1}. Basin vegetation consists of pinyon-juniper woodlands on bluffs with mature gallery forests and cultivated crops in valleys. Flow regimes are determined by irrigation demand, thus the river is virtually dry from October to May.

Two study sites were selected after preliminary sampling in June 1981 (Fig. 2.2). Site 1 is located 4 km downstream from Guernsey Dam. Substrates consist of cobbles one-third to one-half buried in a clay "armor." Site 2 is 2 km downstream from Whalen Dam, a low-head structure that diverts river flows into two large irrigation canals. Substrates are cobble-gravel with little armor.

CHAPTER III

MATERIALS AND METHODS

3.1 Physical-Chemical Parameters

Water samples for chemical analysis were collected in acid-washed, polyethylene bottles, transported on ice, and filtered immediately upon return to the laboratory (Whatman® GF/F filters, pore size 0.7 µm). Analyses were performed on refrigerated samples within 48 h.

Nitrate was determined after reduction with cadmium to nitrite by a dianotization technique (Golterman et al. 1978). This method measures nitrite + nitrate. Soluble reactive phosphorus (SRP) was measured colorimetrically by the method of Murphy and Riley (1962). Conductivity was measured with a Hach® Model 2200 conductivity meter, pH by electrode, and alkalinity by titration with H_2SO_4. Sulfate was measured turbidimetrically with $BaCl_2$. Calcium and magnesium were determined titrimetrically with EDTA. Dissolved organic carbon was determined by dichromate oxidation (Maciolek 1962).

Dissolved oxygen and total sulfide samples were collected in 300 ml BOD bottles. Dissolved oxygen was determined by the Winkler method (azide modification). Total sulfide was measured idometrically after precipitation with $CdCl_2$ (Golterman et al. 1978).

TSS samples were collected with a US DH-48 sampler (U.S. Inter-Agency Committee on Water Resources 1965), filtered on Whatman® GF/F filters, dried at 80°C, and weighed to the nearest 0.1 mg on an analytical balance. Stream substrates were collected with a metal cylinder (900 cm^3) attached to a support rod. Samples were air-dried, placed in a series of standard sieves, and shaken on a mechanical shaker (Ingram 1971). Silt and clay fractions were determined by the pipette method (Galehouse 1971). Mean phi (Ø) values were determined by the pipette method (Galehouse 1971). Mean phi (Ø) values were calculated from moment statistics (McBride 1971). Discharge in Dry Creek was estimated by stream cross-sectional area and velocity.

3.2 Macroinvertebrates-Benthos

Quantitative benthic samples in Dry Creek were taken with a core sampler that enclosed an area of 325 cm^2. Sediments within the sampler were removed to a depth of 10 cm and elutriated through 270 μm mesh. Samples from the North Platte River were taken with a modified Surber sampler that enclosed an area of 320 cm^2 (net mesh = 500 μm). An attached rod and 3 cm bottom bolts helped anchor the sampler in place. Substrate materials were either kicked or placed by hand into the net and returned to shore for elutriation.

Organisms were preserved in Kahle's fluid and later transferred to ethanol for enumeration. Biomass (wet weight) of Dry Creek organisms was determined by blotting excess moisture and weighing to the nearest 2 mg on an analytical balance. Size classes of aquatic insects from the North Platte River were based on measurements of total length (excluding cerci) at 10X using an ocular micrometer.

3.3 Macroinvertebrates-Drift

Macroinvertebrate drift (= organisms transported downsteam in the water column) was sampled at Site 1 using drift nets (46 cm wide X 31 cm high X 96 cm long, 250 μm mesh) staked in the streambed. Four samples were collected at 4 h intervals during 3 diel periods: 7-8 July (1 week before silt run), 16-17 July (start of silt run), and 26-27 July 1982 (peak of silt run).

Concurrent measurements included TSS, net discharge, temperature, and dissolved oxygen. Benthic densities were determined from 8 random samples taken on the first day with the modified Surber sampler.

3.4 Macroinvertebrates-Laboratory Studies

Laboratory studies were conducted during summer 1982 to determine the response of chironomids (Limnochironomus fumidus Johannsen) from the North Platte River to varying concentrations of suspended solids. Experimental chambers consisted of glass bowls (250 ml capacity) placed on magnetic stirrers. Stirbars placed in small crucibles within the bowls provided current speeds of 4-6 cm s^{-1} (Craig 1977). Diatomaceous earth was used as the source of suspended sediment. This material was washed and filtered to obtain a mean particle diameter of 6-12 μm (pipette analysis), a mean size equivalent to the sediments released during the Guernsey silt runs (U.S. Geological Survey 1980).

In each test run, 5-10 chironomid larvae (7-9 mm total length) were placed in an experimental chamber and allowed to acclimate to chamber conditions for 24 h. Diatomaceous earth, weighed to the nearest 0.1 mg, was then added to obtain the desired concentration of suspended solids. Because this material eventually settled out or became trapped on chironomid tubes, two more additions of sediment were made during each short-term (24 h) test. During long-term (several days) tests, sediment was added once daily. No sediment was added to concurrent control chambers that contained < 10 mg 1^{-1} TSS due to bits of debris and food particles.

The water used in all test runs was obtained from the North Platte River at the time the chironomids were collected. The water was filtered before use (Whatman® GF/F filters). Physical-chemical parameters during the test runs were: temperature = 25 ± 2°C, photoperiod = 15L:9D (natural), pH = 7.7 ± 0.2, and alkalinity = 4.2 ± 1.0 meq l^{-1}.

3.5 Benthic Algae

Quantitative algal samples were taken by scraping 8-50 cm^2 from the upper surfaces of large rocks (5-20 cm diameter). Scrapings were filtered to remove excess water (Whatman® GF/A glass fiber filters) and frozen for 48 h. Biomass as chlorophyll _a_ was determined by the methanol extraction technique of Tett et al. (1977). Light intensity in the North Platte River was measured with a Beckman Enviroeye® meter.

CHAPTER IV

EXPERIMENTAL DESIGN

4.1 Dry Creek

Sampling in Dry Creek was directed towards a general description of the magnitude of changes in physical-chemical and biotic parameters following multiple episodes of releases from Reservoir No. 15. Samples for water chemistry (dissolved constituents and TSS) were collected in duplicate, whereas 3-5 samples were taken for macroinvertebrates and algae at each study site. Sampling was conducted daily at the time of initial releases (February 1981) and later reduced to biweekly-monthly intervals. Sampling was terminated in January 1982. Substrate samples (approx. 1 kg at each study site) were collected on 16 February (a day before the initial release), 20 February (Day 3 after initial release), 26 August (Day 190), and 19 December 1981 (Day 305, 3 months after the final release).

4.2 North Platte River-1981 Silt Run

The predictability and short duration of the Guernsey silt run allowed development of a more complex experimental design than was possible in Dry Creek. Goals of the 1981 study were to determine: 1) changes in densities of macroinvertebrates during the silt run, and to compare these changes with *a priori* hypotheses for individual taxa, 2) changes in algal abundance during the silt run, 3) the relationship between changes in the biota and physical-chemical conditions, and 4) the relative contribution of physical-chemical changes to the overall biotic change.

Five taxa of macroinvertebrates were chosen for intensive study: *Baetis insignificans* McDunnough (Ephemeroptera), *Tricorythodes minutus* Traver

(Ephemeroptera), Orthocladiinae spp. (Diptera:Chironomidae, primarily *Orthocladius* sp.), Chironomini spp. (Diptera:Chironomidae, mostly *Limnochironomus fumidus*), and Oligochaeta spp. These taxa comprised 95% of total numbers of macroinvertebrates in the North Platte River at the time of the silt run.

Baetis insignificans is a small, streamlined mayfly that inhabits upper surfaces of coarse substrates in moderate currents. Previous studies indicate that baetid mayflies tolerate high concentrations of TSS and moderate amounts of sediment deposition (Bjornn et al. 1977, Gammon 1970, Hamilton 1961, Nuttall and Bielby 1973). Therefore, *B. insignificans* was expected to be unaffected by the silt run.

Tricorythodes minutus is a sprawling mayfly on fine sediments. It is a characteristic mayfly of warm, turbid streams in the western U.S. (Edmunds and Musser 1960). Gammon (1970) found that populations of *Tricorythodes* sp. increased with high turbidity and sediment build-up, thus numbers of *T. minutus* were expected to increase during the silt run.

Chironomids have exhibited variable responses to suspended and settled solids. High turbidities alone may cause either a decrease, increase, or no change in population densities (Bjornn et al. 1977, Gammon 1970, Rosenberg and Wiens 1975). Heavy sedimentation with high concentrations of TSS may cause either decreased or increased numbers (Gammon 1970, Nuttall and Bielby 1973). In the North Platte River, chironomids should decrease in numbers during the silt run. Although deposition of sediments might eventually create favorable conditions for some taxa, we expected that chironomids colonizing before the silt run to be more adapted to the clear flows and relatively silt-free substrates that existed in May and June.

Populations of oligochaetes consistently increase in turbid, silted habitats (Hamilton 1961, Nuttall and Bielby 1973). During the silt run, these organisms were expected in increase in number from downstream drift of benthos from Guernsey Reservoir and increases in silted substrates.

The set of environmental variables included concentrations of TSS and substrate mean ϕ. Phi values are expected to change from deposition of fine particulates during the silt run. In addition, they describe natural variation in distribution of macroinvertebrates.

Fifteen samples were taken at each site on each date. A sample consisted of measurements on concentration of TSS (collected as close to the stream bottom as possible), substrates (collected adjacent to Surber samples), numbers of each macroinvertebrate, and algal biomass. Three samples were taken in each of 5 locations at each site. A location was *ca*. 200 m^2 of stream bottom from bank to thalweg in depths > 20 cm. Locations were randomly selected on 6 July and thereafter sampled each time as a basis for comparison. Principal sampling dates were 6 July (pre-silt run); 9, 16, and 23 July (silt run); and 6 and 20 August 1981 (post-silt run). The 1981 silt run was conducted from 9 July to 29 July.

Multivariate analysis of variance was used to test the null hypothesis of no overall change in each variable set for the period 6 July to 6 August ($N = 75$ per site). If the null hypothesis was rejected, then univariate tests were performed. Densities of macroinvertebrates may fluctuate from life history phenomena alone (e.g., synchronous emergence and recruitment). For example, the mayflies chosen for intensive study are known to have rapid life cycles. *T. minutus* completes larval development in 34d at 18° (Newell 1976), and *B. insignificans* has several generations during summer in northwestern Colorado (Gray and Ward 1978). Thus acceptance or rejection of the null hypothesis was determined with additional informaton provided by size class distributions. Canonical correlations between rates of change in each set of variables were determined to further substantiate changes in biota caused by the silt run. Rates of change were calculated as the first derivative of the slope of a polynomial regression computed for each variable at each location for the period 6 July to 6 August (N = 40). This procedure eliminates within-location variation (mostly "noise") and implies percentage rates of change that are linearly and additively related (Green 1979). All variables except substrate phi were log-transformed before analysis. Statistics were calculated with SPSS programs (Nie et al. 1975).

4.3 North Platte River-1982 Silt Run

The primary goal of the drift study conducted during the 1982 silt run was to determine if downstream drift was responsible for the increase in oligochaete densities in the North Platte River observed during 1981 (i.e., oligochaetes were derived from Guernsey Reservoir) and the decrease in chironomid numbers (see Sec. 5.2). Alternative mechanisms for changes in population size are *in situ* reproduction for oligochaetes and either emergence or mortality for chironomids.

An increase in oligochaete numbers due to drift from Guernsey Reservoir would be indicated by a large increase in drift rates, particularly at the start of the silt run when surficial sediments are scoured. If no significant increase in drift occurred, then enhanced reproduction within the river would be the most probable mechanism. Oligochaetes and other sediment-dwelling organisms do not exhibit behavioral drift, or active movement into the water column (Waters 1972). Drift rates of these organisms are typically low and constant, depending on occasional individuals disturbed from the stream bottom.

If downstream drift was responsible for the losses of chironomids during 1981, then changes in drift rates should reflect changes in TSS. This hypothesis was tested by regressing rates of change in drift rates and TSS for each chironomid taxon. Rates of change were computed from polynomial equations for each 4-h time interval over the period of study (see Sec. 4.2). A linear relationship between rates of change in drift and TSS would indicate that drift was the main mechanism causing reductions in benthic densities. No relationship would indicate mortality was more important, if emergence and recruitment were continuous.

4.4 Laboratory Studies

Larvae of _L. fumidus_, like many other Chironominae, construct tube-like retreats composed of debris particles cemented together by a salivary secretion. The tubes are open at both ends to allow water currents to pass through. The larvae rarely leave their retreats except to protrude a short distance while feeding. Pupation also occurs in the tubes.

The response of _L. fumidus_ larvae and pupae to suspended sediment additions in laboratory chambers was evaluated over short (24 h) and long 5-14 days) intervals. During preliminary observations, we found that larvae increased the length of their tubes when sediment was added. As older segments of the tubes became coated with sediment particles, new segments were added. Thus the principal goal of the short-term tests was to determine the relationship between tube length and concentration of added sediment.

Long-term tests were conducted to examine potential mortality of larvae and pupae at concentrations of TSS encountered during the Guernsey silt run. These tests were allowed to continue until all test larvae had either died or emerged as adults.

CHAPTER V

RESULTS AND DISCUSSION

5.1 Dry Creek

5.1.1 Physical-Chemical Parameters

Before the initial release of water from Reservoir No. 15, Dry Creek was characterized by high pH, high buffering capacity, high nitrate-nitrogen, low SRP, high total dissolved solids, and low suspended solids (Appen. A). The main difference between sampling sites was a greater concentration of nitrate-nitrogen at Site 2 caused by inflows of groundwater. Stream water temperatures varied with air temperature, and the diel range was 9°C. Reservoir No. 15 was not stratified before the initial release. Concentrations of dissolved constituents were lower than Dry Creek.

The initial release of water occurred at 0850 on 17 February. Discharge in Dry Creek instantaneously increased from less than 0.1 to 1.4 m^3s^{-1}. The outflow was highly turbid for an hour as sediments near the outlet were brought into suspension (Appen. A). According to local irrigation managers, the duration of turbid water from the reservoir is typical of most reservoirs in the area. The initial release also produced instantaneous changes in temperature and concentrations of dissolved materials that reflected differences in composition between reservoir and stream water. An exception was SRP which increased from a pre-release concentration of less than 0.01 mg l^{-1} to 0.26 mg l^{-1} at Site 1. Parameters that showed little change were pH and dissolved organic carbon.

The initial flood wave arrived at Site 2 in 17 minutes. Along its path, the flood wave brought substrates into suspension and dried salts on gravel bars into solution. Consequently, suspended solids increased from less than

3 mg l^{-1} to more than 7,900 mg l^{-1} and conductivity increased from 930 to 1,380 μS cm^{-1}. Unlike Site 1, high levels of TSS continued for a day. Scouring of highly organic muds in Dry Creek decreased dissolved oxygen from near 100% to 50% saturation. However, no hydrogen sulfide was detected.

Subsequent water quality changes in Dry Creek depended on reservoir discharges. During non-release periods, dissolved constituents returned to pre-release ranges, and water temperatures exhibited diel cycles that were eliminated during releases. Although concentrations of suspended materials were greater during release periods, planktonic organisms comprised most of the suspended material.

Substrates in Dry Creek consisted of pebbles at Site 1 and gravels at Site 2 (Table 5.1). The initial reservoir release doubled mean particle size at both sites, indicating that scour exceeded deposition. At Site 1, mean particle size changed little during the irrigation season, whereas mean size at Site 2 decreased as a result of extensive bank erosion. Three months after reservoir releases ended, mean size in Site 1 was equivalent to pre-release substrates. This decrease was apparently caused by decomposition of friable shales. At Site 2, mean size increased from removal of clays and silts by clear flows from upstream springs.

5.1.2 Macroinvertebrates

Principal macroinvertebrate taxa in Dry Creek before the initial release were chironomids (Orthocladiinae), oligochaetes, hydropsychid caddisflies, and small mayflies (Appen. B). Total density and biomass were 33,000 organisms m^{-2} and 33 g m^{-2}, respectively. Initial reservoir releases decimated the entire fauna from high current velocities and substrate scour.

Table 5.1. Substrate particle size (mean Ø and corresponding mm) in Dry Creek, Larimer County, Colorado during 1981.

Date	Day From Initial Release	Site 1		Site 2	
		Ø	mm	Ø	mm
16 II	-1	-4.5	22.6	-1.7	3.2
20 II	3	-5.2	36.8	-2.6	6.1
26 VIII	190	-5.1	34.3	-1.7	3.2
19 XII	305	-4.6	24.3	-2.4	5.3

Calculations based on suspended sediment samples at Site 2 show that greater than 4 X 10^6 organisms were swept downstream during the first minute of the initial release.

Recovery of the fauna did not occur during release periods. The inverse relationship between abundance of macroinvertebrates and discharge indicates that high flows deterred colonization (Fig. 5.1). Recovery was possible only during non-release periods, including 2-4 week intervals between releases during the irrigation season and an extended period in the fall and winter after releases were terminated.

The intermittent nature of reservoir releases allowed an opportunity to examine recovery rates and changes in species composition after repeated disruption. Recovery rates of individual taxa varied with increasing frequency of disturbance. Some taxa, such as tipulid larvae, stoneflies, amphipods, and snails were eliminated with the initial reservoir release and did not recover until several months after the final release. Others, like hydropsychid caddisflies, chironomids, and oligochaetes, had high recovery rates after the initial release, but reduced rates after multiple releases (Table 5.2). Only a few of the taxa present, particularly *T. minutus*, *Baetis* spp., and simuliids, were able to maintain a high rate of recovery regardless of disturbance frequency. These species have rapid life cycles (Sec. 4.2) compared to the univoltine life cycles of tipulids and stoneflies.

The varying recovery capabilities of the taxa in Dry Creek resulted in distinct shifts in community composition after multiple reservoir releases (Fig. 5.2). The cluster diagram indicates the relative similarity in community composition during each recovery period between reservoir releases

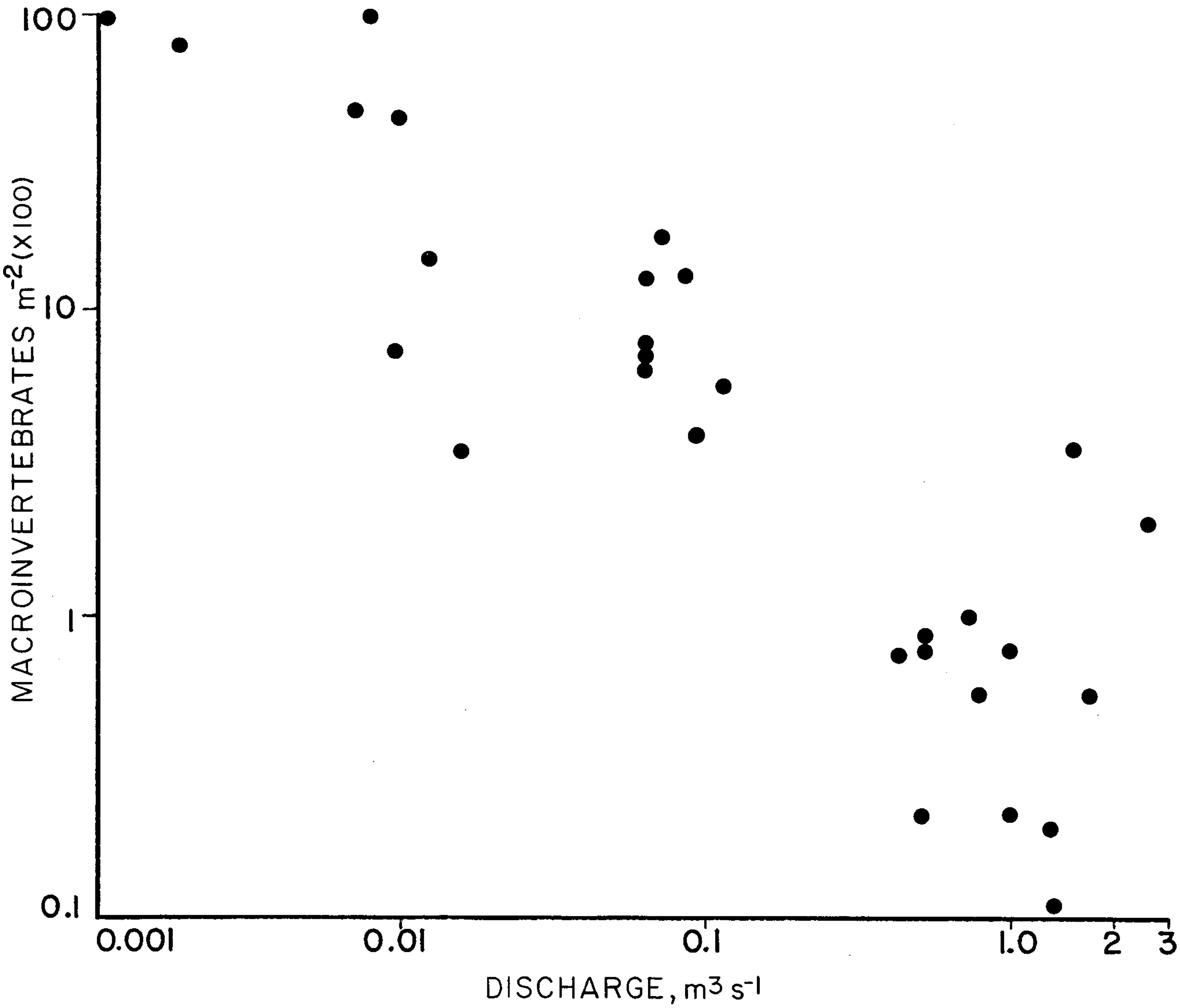


Figure 5.1. Relationship between macroinvertebrate density and discharge in Dry Creek. Samples collected after 1-2 weeks of exposure to a given discharge. Note logarithmic scales.

Table 5.2 Recovery rates of selected macroinvertebrates and total taxa in Dry Creek, Larimer County, Colorado, with increasing frequency of disturbance. Rates are the number of days required to reach mean sample density on day -1 on recovery days 47 (1 release), 97 (2 releases), 190 (3 releases), and 245 (4 releases). Rates were computed from the slopes of regression equations fitted to $\log_e$- transformed sample numbers during each recovery period.

Taxon	Number of Reservoir Releases 1	2	3	4
Tricorythodes minutus	30	51	44	30
Baetis spp.	7	10	14	11
Simulium arcticum	4	1	2	2
Chironomidae spp.	43	98	109	160
Hydropsychidae spp.	43	72	61	175
Oligochaeta spp.	24	46	83	112
Total taxa	44	82	97	110

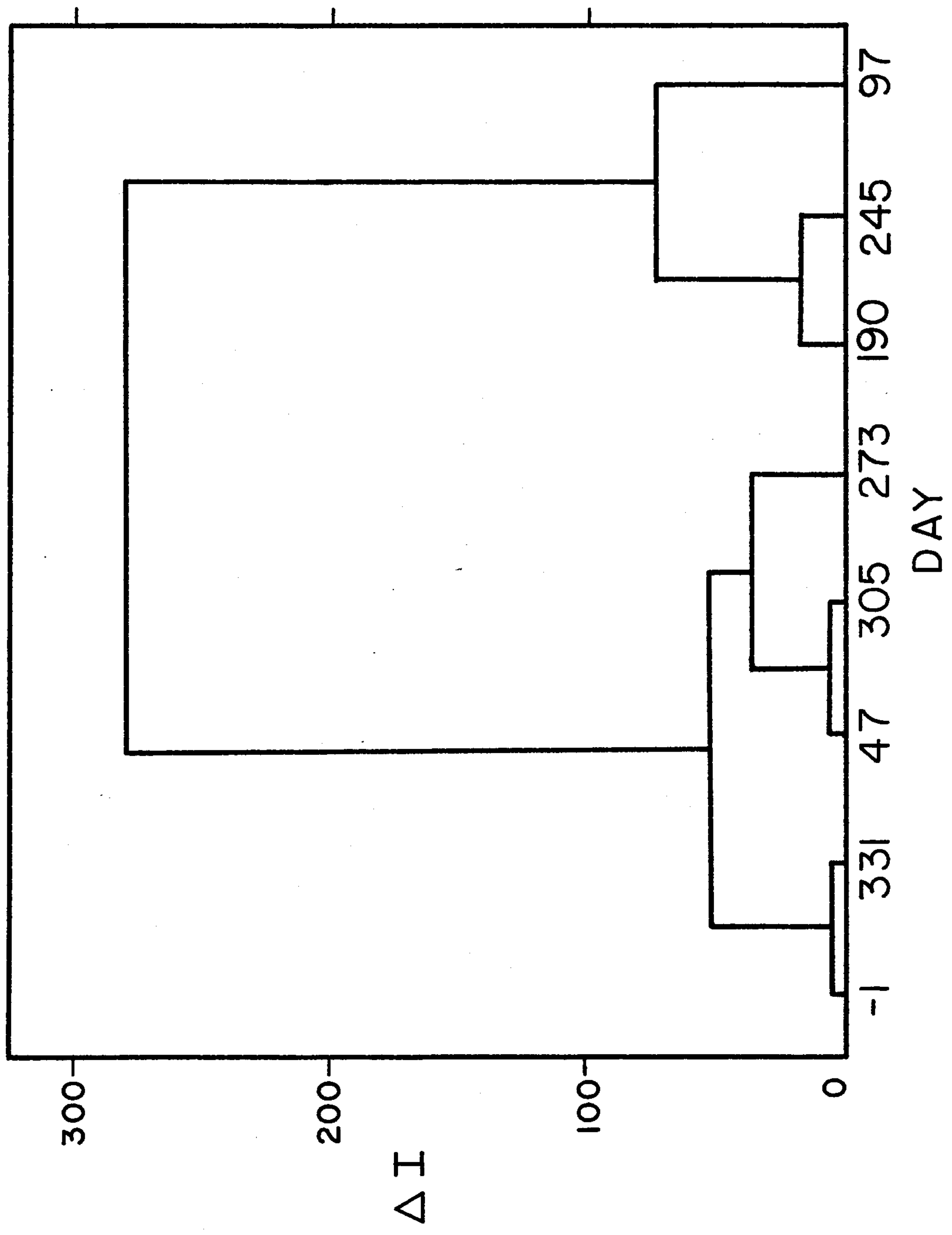


Figure 5.2. Cluster diagram showing similarity in species composition during recovery periods in Dry Creek. See text.

(combined data from Sites 1 and 2). The clustering algorithm is Orloci's (1975) measure of mutual information; if delta-I is zero, then the two communities have the same composition. Recovery dates are the number of days after the initial reservoir release. Each recovery date, except for Days 245 to 331, was separated from the next by a release period.

The community on the first recovery date (Day 47) was similar to that present before the initial release, refecting the high rates of recovery of many dominant taxa (e.g., orthoclad chironomids, hydropsychids, and oligochaetes). After multiple releases, the community became increasingly dominated by small, rapid-growing mayflies and simuliids. Simuliids were dominant on Day 97 and mayflies were more abundant on Days 190 and 245. During the extended recovery period after the final release (Days 245 to 331), community composition again returned to that present before the first release.

Community structure in Dry Creek thus reflected the frequency of disturbance through a selection of a subset of the species present. This subset contained species pre-adapted by their rapid life cycles to quickly recolonize and emerge before the next reservoir release.

We estimate that 44 days would be required for total densities to recover to pre-release levels after the initial reservoir release. After 4 releases, about 110 days would be needed for full recovery.

5.1.3 Benthic Algae

The dominant algae in Dry Creek before the initial release from Reservoir No. 15 were filamentous greens (especially *Cladophora glomerata*), diatoms, and crustose blue-greens. Abundance as chl. *a* was 206 mg m^{-2}

(Fig. 5.3), combined data from Sites 1 and 2). Substrate scour during the initial release reduced abundance by more than 90%, and crustose blue-greens were the only algae remaining. Unlike macroinvertebrates, no significant recovery occurred until releases were terminated in the fall.

5.2 North Platte River-1981 Silt Run

5.2.1 Physical-Chemical Parameters

The 1981 silt run had little effects on most water quality parameters other than suspended solids (Appen. C). The only chemical parameter to show significant change was SRP which increased from less than 0.01 mg l^{-1} to 0.06 mg l^{-1} at Site 1 (0.02 mg l^{-1} at Site 2). Nitrate-nitrogen was below detection limits (= 0.05 mg l^{-1}). Dissolved oxygen remained at or near saturation throughout the silt run, and hydrogen sulfide was not detected. Midday water temperatures were 19-23°C during the study period.

Discharge from Guernsey Reservoir during the silt run varied from 109 to 150 m^3s^{-1} with a mean of 136 m^3s^{-1} (Fig. 5.4). Flow variations in this range did not affect sampling locations.

Concentrations of TSS increased 20-fold during the silt run to peak values of 339 mg l^{-1} at Site 1 and 422 mg l^{-1} at Site 2 (Fig. 5.5). Lack of a downstream decline in TSS demonstrates that flushed silts and clays remained in suspension for long distances. Only slight deposition of suspended materials occurred on the streambed. The silt-clay fraction comprised less than 1% of substrate materials, and mean particle size was unaltered at both sites (Fig. 5.5). Heavier deposits (up to 3% silt-clay) were present only along submerged vegetation near the shoreline.

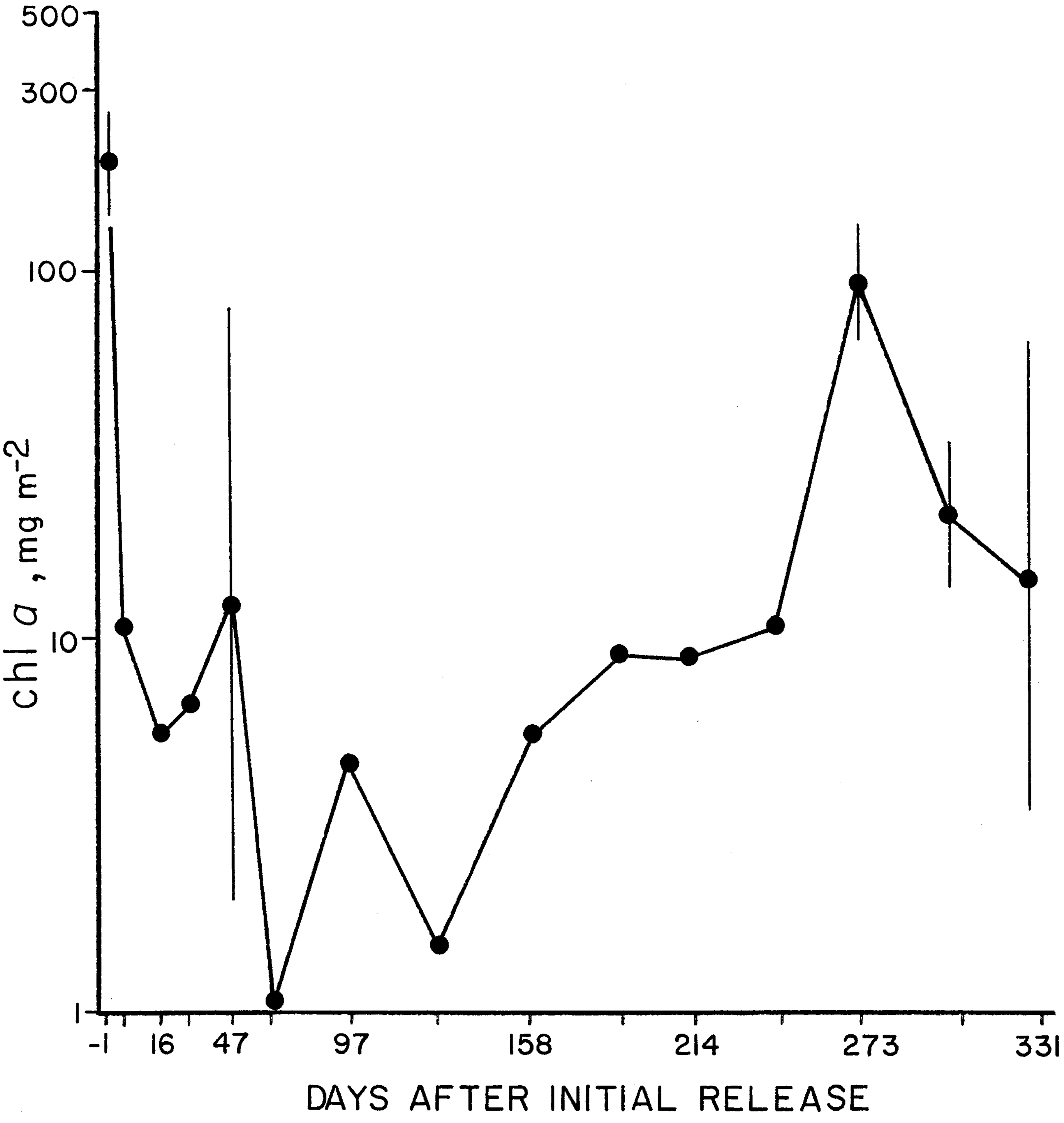


Figure 5.3. Abundance of benthic algae in Dry Creek for the period 16 Feb. 1981 (Day -1) to 14 Jan. 1982 (Day 331). Vertical bars indicate 95% confidence limits.

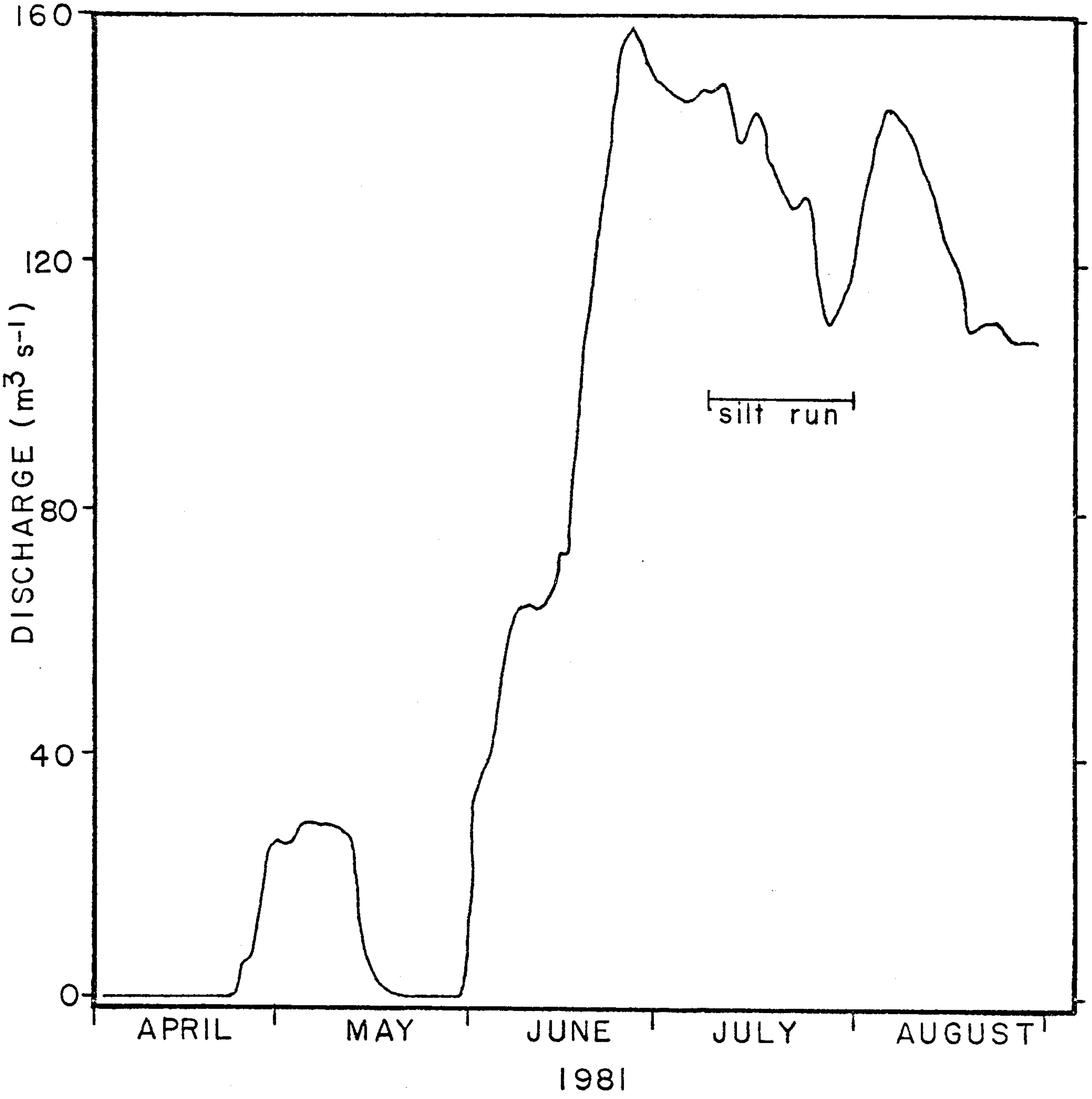


Figure 5.4. Discharge in the North Platte River for the period 1 April-31 August 1981. Data provided by Mr. David Wilde, U.S. Bureau of Reclamation, Casper, Wyoming.

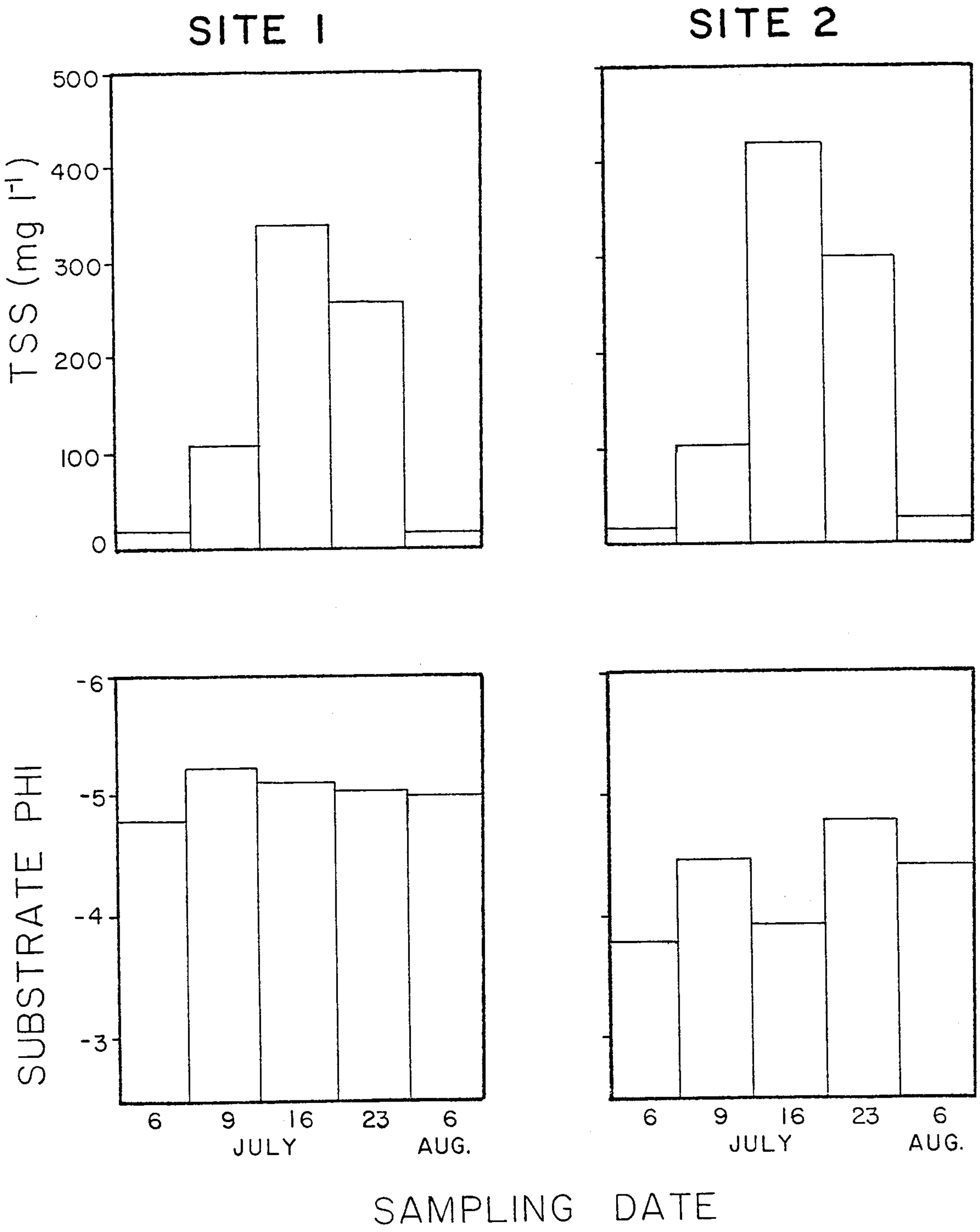


Figure 5.5. Concentrations of total suspended solids (TSS) and substrate particle size (mean phi units) in the North Platte River during the 1981 silt run. Overall differences in TSS and phi were significant at both study sites ($P<0.01$). Differences in TSS alone were significant at both sites ($P<0.01$), but phi values were not significantly different at either Site 1 ($P = 0.49$) or Site 2 ($P = 0.10$).

5.2.2 Macroinvertebrates

Overall densities of macroinvertebrates significantly changed during the silt run at both study sites, and individual taxa exhibited trends consistent with *a priori* predictions (Fig. 5.6). Life history phenomena alone did not account for density changes with the possible exception of mayflies at Site 1.

The abundance of chironomids decreased more than 90% within two weeks after the beginning of the silt run. Although declines were similar for both taxa, the silt run affected different life stages. Reproduction was continuous for *Orthocladius* sp. but not for *L. fumidus* (Fig. 5.7). For both species, pupae were absent from 16-23 July (peak of silt run), whereas 5-10 pupae sample^{-1} were collected before and after the silt run. The mechanisms influencing these selective effects on various life stages are discussed in Sec. 5.3 and 5.4. Despite the high losses in July, chironomid densities recovered to pre-silt run levels within 3 weeks after releases ended.

Density trends for mayflies at Site 2 followed initial predictions (Fig. 5.6). *T. minutus* increased from 14 individuals sample^{-1} on 6 July to 53 sample^{-1} on 6 August. Density of *B. insignificans* did not significantly change. Size class distributions for both species were stable with continuous recruitment and emergence. At Site 1, densities of both species increased during the silt run, although these increases may have resulted from synchronous emergence on 9 July and subsequent recruitment during the next 2 weeks (Fig. 5.7).

Densities of oligochaetes increased at both study sites. Data collected during the 1982 silt run suggest these increases were the result of *in situ* reproduction rather than drift of individuals from Guernsey Reservoir (see

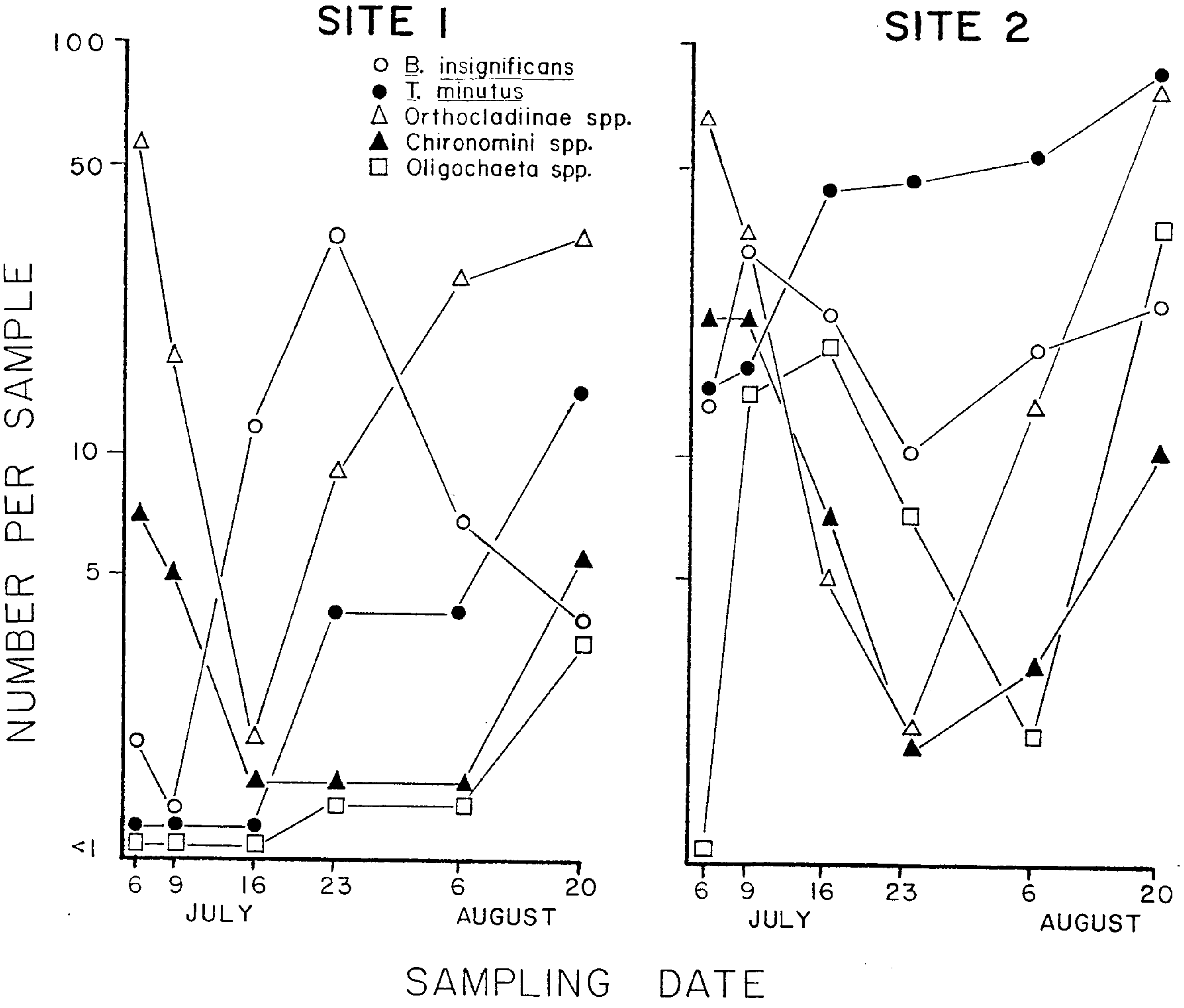


Figure 5.6. Mean densities of macroinvertebrates in the North Platte River during the 1981 silt run. Overall differences in densities between 6 July and 6 August were significant at Site 1 (P<0.01) and Site 2 (P<0.01). Differences in densities during this period were significant for *B. insignificans* (P<0.01, Site 1), *T. minutus* (P<0.01, Site 1; P<0.01, Site 2), Orthocladiinae spp. (P<0.01, Site 1; P<0.01, Site 2), Chironomini spp. (P<0.01, Site 1, P<0.01, Site 2), and Oligochaeta spp. (P = 0.03, Site 1; P<0.01, Site 2). Differences were not significant for *B. insignificans* at Site 2 (P = 0.12).

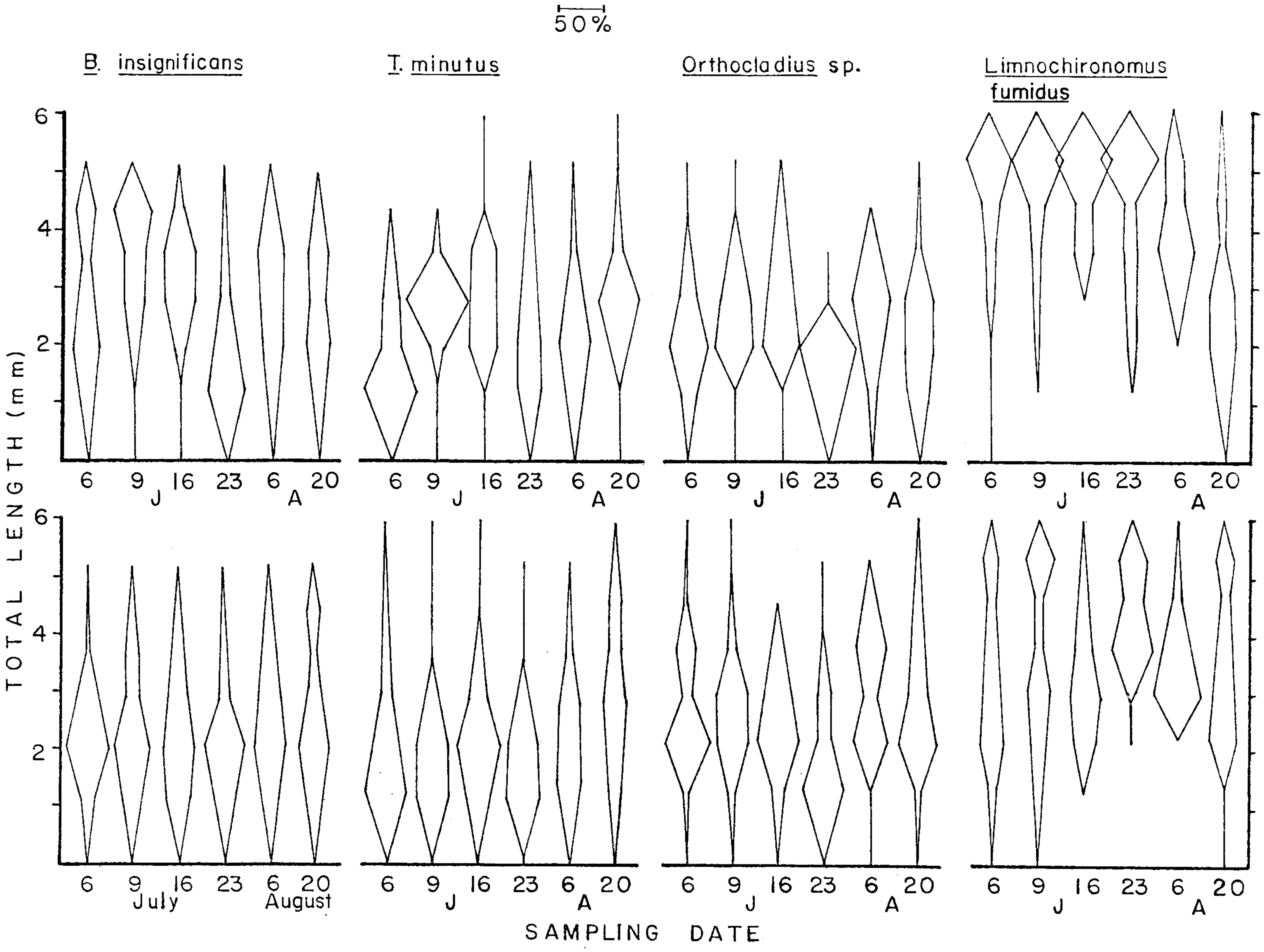


Figure 5.7. Size class distributions of aquatic insects in the North Platte River during the 1981 silt run. Top row = Site 1, bottom row = Site 2.

Sec. 5.3). Very high densities after the silt run (up to 100 organisms sample^{-1}) were found in heavily-silted, shoreline habitats.

The silt run contributed to an increase in benthic algae, especially *Cladophora* (Sec. 5.2.3). Algal mats trapped some sediment, thereby providing microhabitats and food materials for many macroinvertebrates (Williams and Winget 1979). Among such organisms were *Hydroptila* sp., a microcaddisfly that feeds exclusively on *Cladophora* as a late instar (Cummins 1973), and *Dactylobaetis* sp., a small mayfly adapted to warm, silted streams (Edmunds et al. 1976) (Appen. D).

Overall, there was a strong correlation between rates of change in macroinvertebrate populations and environmental parameters (Fig. 5.8), thus providing further evidence that the silt run was primarily responsible for effects on benthic organisms. Because substrates were unaltered by the silt run, nearly all of the biotic change can be attributed to the increase in suspended solids. Correlations between rates of change in benthic densities of individual taxa and TSS were significant (all $P < 0.01$): *B. insignificans*, $r = 0.48$; *T. minutus*, $r = -0.89$; Orthocladiinae spp., $r = -0.96$; Chironomini spp., $r = -0.89$; and Oligochaeta spp., $r = 0.62$ (Appen. D). Correlations between individual taxa and substrate phi were not significant for *B. insignificans* ($r = -0.14$, $P > 0.10$), *T. minutus* ($r = -0.23$, $P > 0.10$), and Chironomini spp. ($r = 0.29$, $P > 0.10$). Changes in substrates were important for orthoclad chironomids ($r = 0.41$, $P < 0.01$) and oligochaetes ($r = -0.36$, $P = 0.02$).

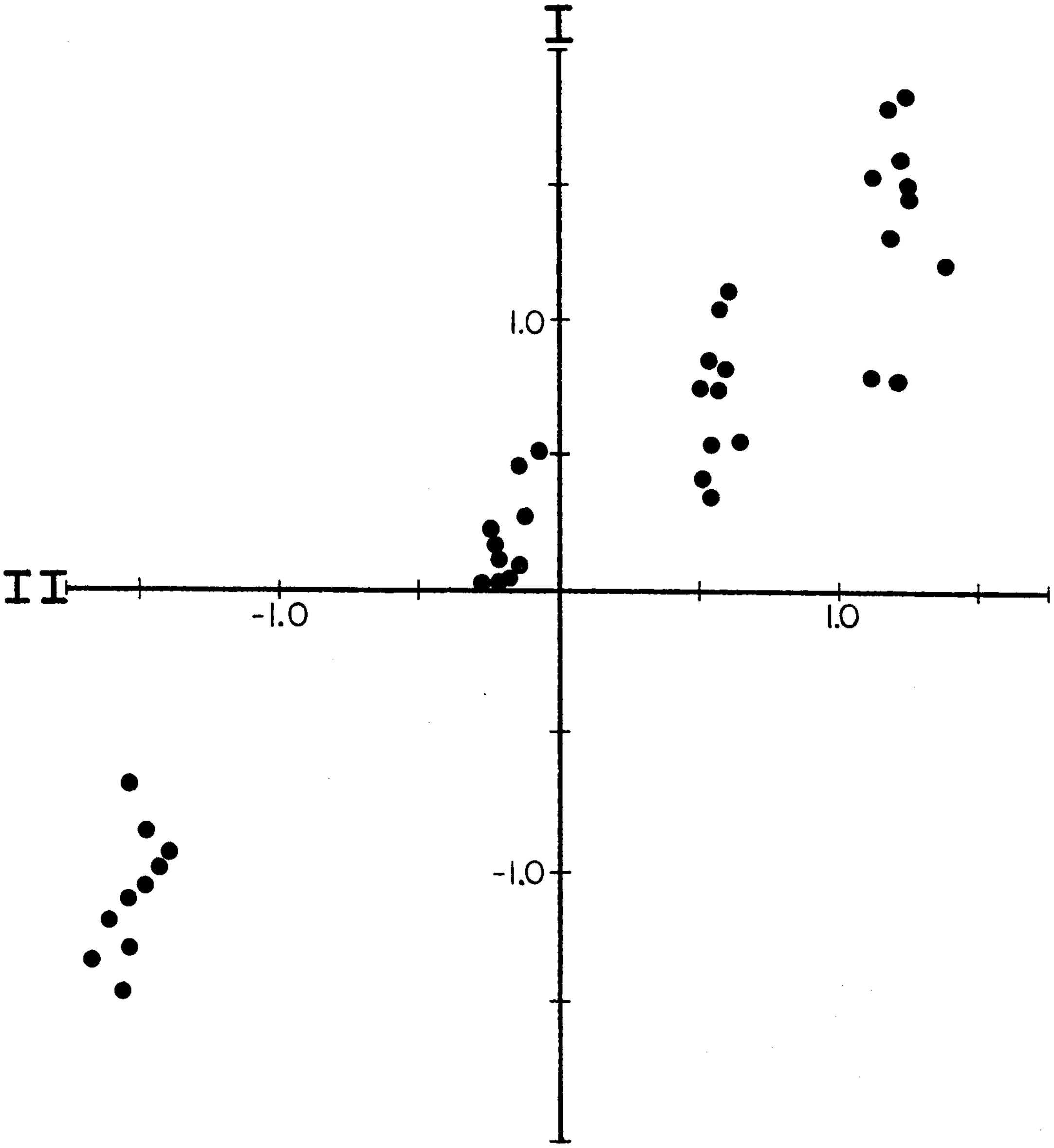


Figure 5.8. Overall relationship between rates of change in densities of macroinvertebrates and environmental parameters during the 1981 silt run. Values are canonical variate scores calculated from the first correlation equation ($P<0.01$): 0.16 Δ log B. insignificans + 0.06 Δ log T. minutus - 0.70 Δ log Orthocladiinae spp. - 0.15 Δ log Chironomini spp. + 0.10 Δ log Oligochaeta spp. = 1.01 Δ log TSS + 0.03 $\emptyset$. Other possible correlation equations were not significant ($P = 0.94$), indicating that biotic changes mainly resulted from silt run effects. Axis I = scores for biological variables, Axis II = scores for environmental variables.

5.2.3 Benthic Algae

Algal assemblages in the North Platte River were dominated by *Cladophora glomerata*. Other algae included epiphytic diatoms and crustose blue-greens. Before the 1981 silt run, algal abundance was 34 mg chl. *a* m^{-2} at Site 1 and 17 mg m^{-2} at Site 2 (Fig. 5.9). At the peak of the silt run (23 July), abundance had significantly increased to 79 mg m^{-2} and 56 mg m^{-2} at Sites 1 and 2, respectively. Increases were largely caused by growth of *Cladophora*. Filaments less than 2 cm in length on 6 July increased to several decimeters in length by 6 August. Abundance measurements after 16 July are undoubtedly underestimates, because they do not include filaments that detached from substrates. Drifting algal mats were frequently observed after late July, and these mats caused considerable clogging of irrigation gates.

The rapid increase in algal abundance during the silt run resulted from several factors. First, light intensities were not reduced to severely limiting levels for photosynthesis. At depths up to 1 m, 15-40% of available light reached the streambed. Second, *Cladophora* was probably limited by phosphorus before the silt run, and subsequent increases in SRP favored high growth rates (Herbst 1969, Wong and Clark 1976). In addition to dissolved forms of phosphorus, algae can extract phosphorus adsorbed on fine sediment particles (Perry and Stanford 1982). Third, optimal temperatures for *Cladophora* growth are in the range of 20-25° (Herbst 1969, Gerloff and Fitzgerald 1976). Midday water temperatures at both study sites were within this range (Sec. 5.2.1). Finally, other physical-chemical parameters, such as pH, hardness (Appen. C), and current velocities (50-100 cm s^{-1}) favored rapid growth (Whitton 1970).

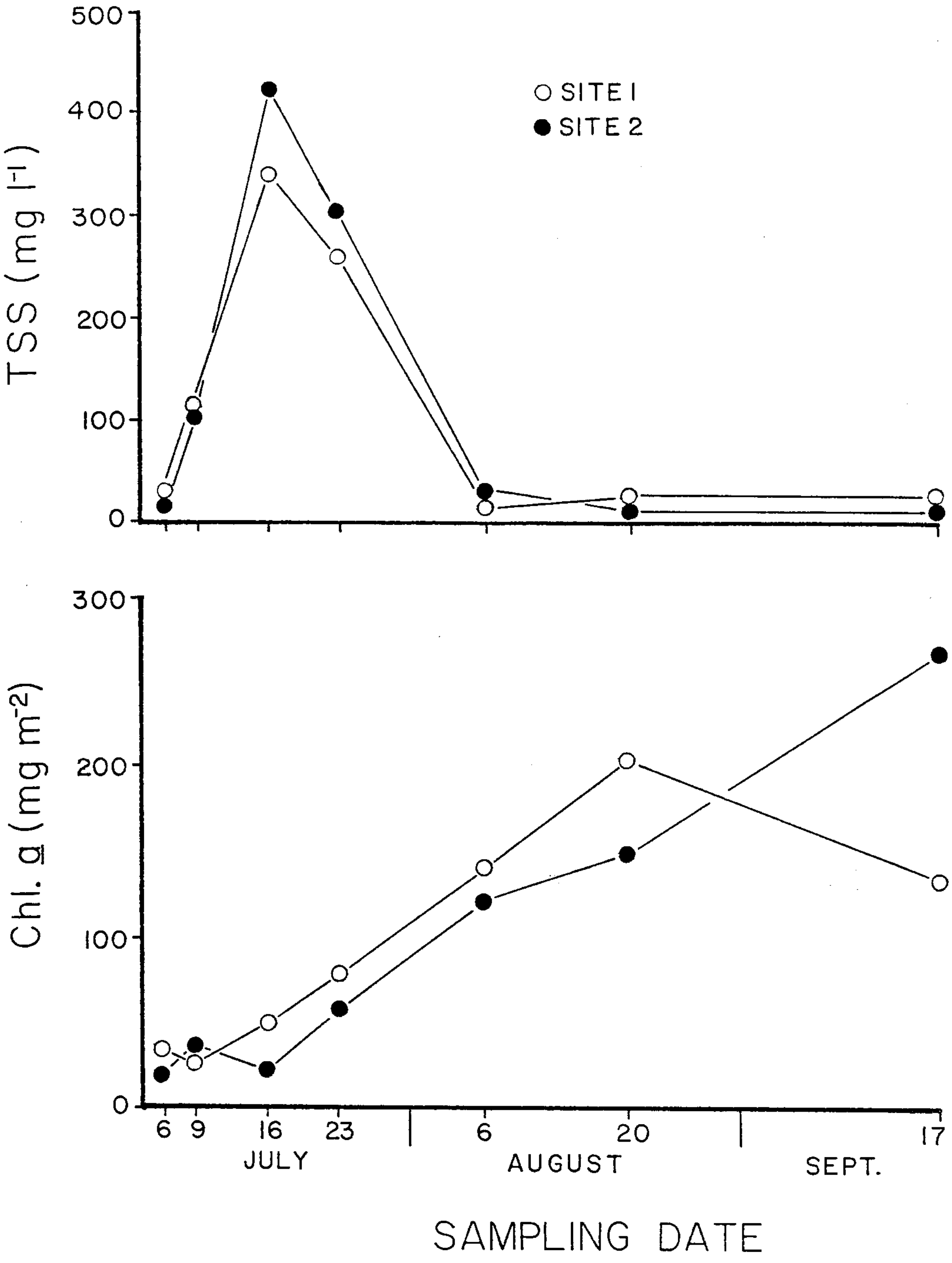


Figure 5.9. Abundance of algae and concentrations of TSS in the North Platte River during the 1981 silt run. Increases in chl. a during the period 6 July-23 July were significant at Site 1 ($P\bar{<}0.01$) and Site 2 ($P<0.01$).

5.3 North Platte River-1982 Silt Run

The macroinvertebrate fauna at Site 1 during July 1982 was dominated by chironomids, simuliids, and oligochaetes (Appen. E). Many taxa, particularly mayflies and caddisflies, were rare or absent compared to the same period in 1981. The low densities and diversity of the fauna in July resulted from the lack of reservoir releases until late June. Consequently, most of the streambed had been inundated for only 2 weeks before the first samples were taken. By the start of the silt run, benthic densities had tripled and more taxa were present.

Like the 1981 silt run, physical-chemical parameters changed little except for TSS. Dissolved oxygen remained high (8-9 mg/1), and the diel range of water temperatures on all sampling dates was 18-23°C. Mean concentrations of TSS were 18 mg/1 one week before the silt run, 43 mg/1 on the first day of the silt run, and 374 mg/1 at the silt run peak (Fig. 5.10).

Benthic densities and drift rates of oligochaetes showed no significant changes during the silt run (Appen. E). Overall drift rates were low (1-8 individuals/$10^4 m^3$), and no diel periodicity was evident. Thus increases in oligochaete populations observed during latter stages of the 1981 silt run more likely resulted from *in situ* reproduction rather than drift of benthos from Guernsey Reservoir.

Chironomids comprised 80%+ of drifting organisms during the study period. In contrast to many previous studies (see Waters 1972), chironomid drift followed a diel pattern with nighttime rates typcally 3-10X greater than daytime rates (Fig. 5.10). No significant differences were detected between 4-h intervals within each time period.

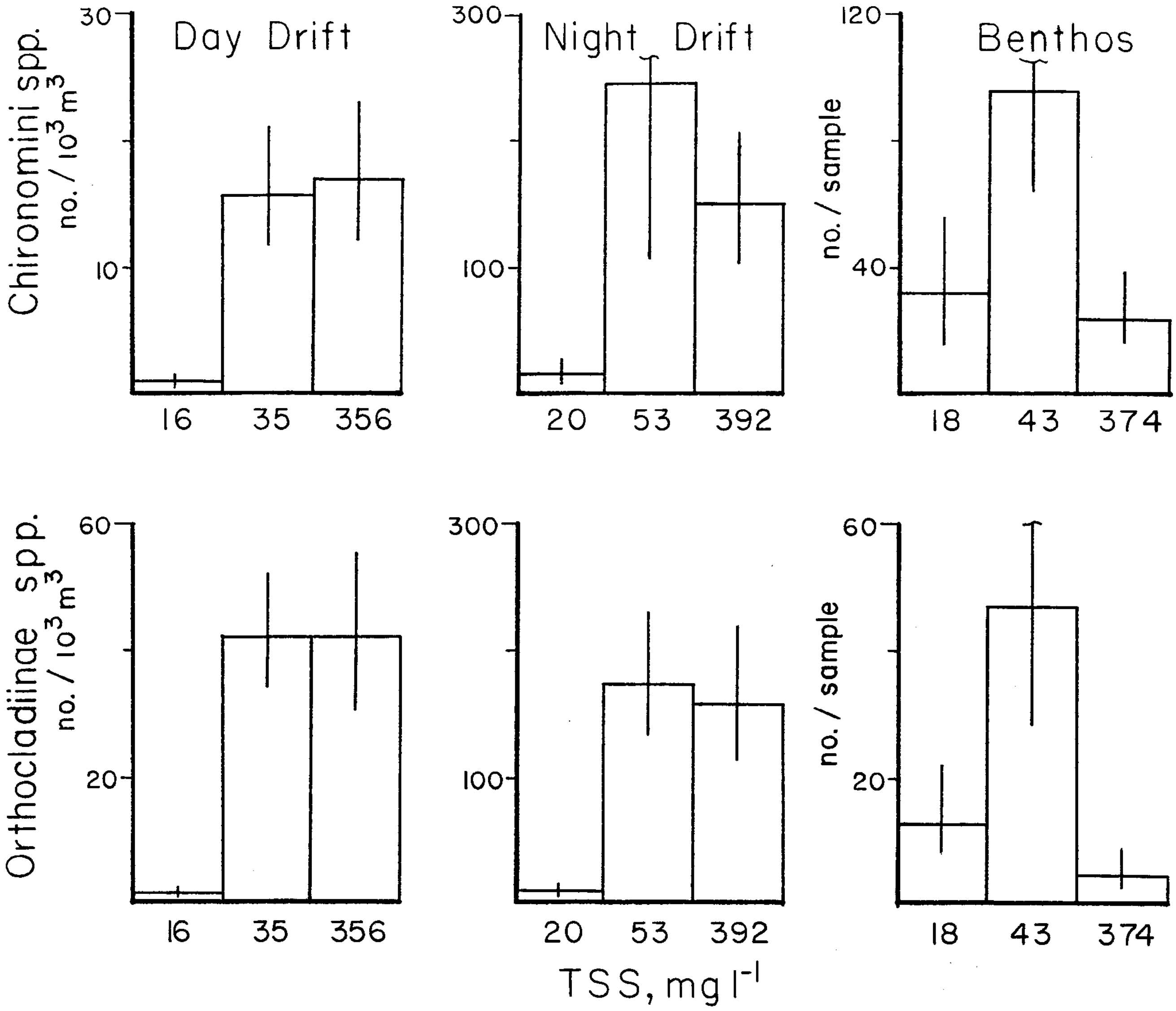


Figure 5.10. Drift rates and benthic densities of chironomids during the 1982 silt run. Vertical bars indicate 95% confidence limits.

Drift rates of chironomid larvae and pupae greatly increased with the start of the silt run. After adjusting for increases in benthic densities, drift rates were 4-6X greater than pre-silt run rates. High rates for larvae were maintained during the silt run despite significant benthic losses (Fig. 5.10). For pupae, benthic density and drift rate declined as the silt run progressed.

The shift in size distribution noted for _L. fumidus_ larvae during the 1981 silt run (Sec. 5.1.2) also occurred during 1982. Large larvae nearing pupation (greater than 5mm total length) exhibited less tendency to drift than earlier instars. During the sampling period, the proportion of large larvae in the benthos increased from 0.3% to 45.8%, whereas their proportion of total drift increased from 0.1% to 20.0%. Thus the shift in size distribution was caused in part by lower drift rates of later instars. Other factors influencing this shift are discussed in Sec. 5.4.

The high drift rates for chironomids at the start of the silt run show that downstream movements could account for the initial, rapid declines in benthic numbers. However, as concentrations of TSS increased, drift rates did not correspondingly increase (Fig. 5.11). In view of results from laboratory studies (Sec. 5.4), we conclude that mortality contributed to benthic losses as the silt run progressed. Thus the linear relationship between rates of change in benthic populations and TSS (Sec. 5.1.2) resulted from high drift rates at relatively low, but increasing concentrations of TSS at the beginning of the silt run and increased mortality with extended exposure to high concentrations.

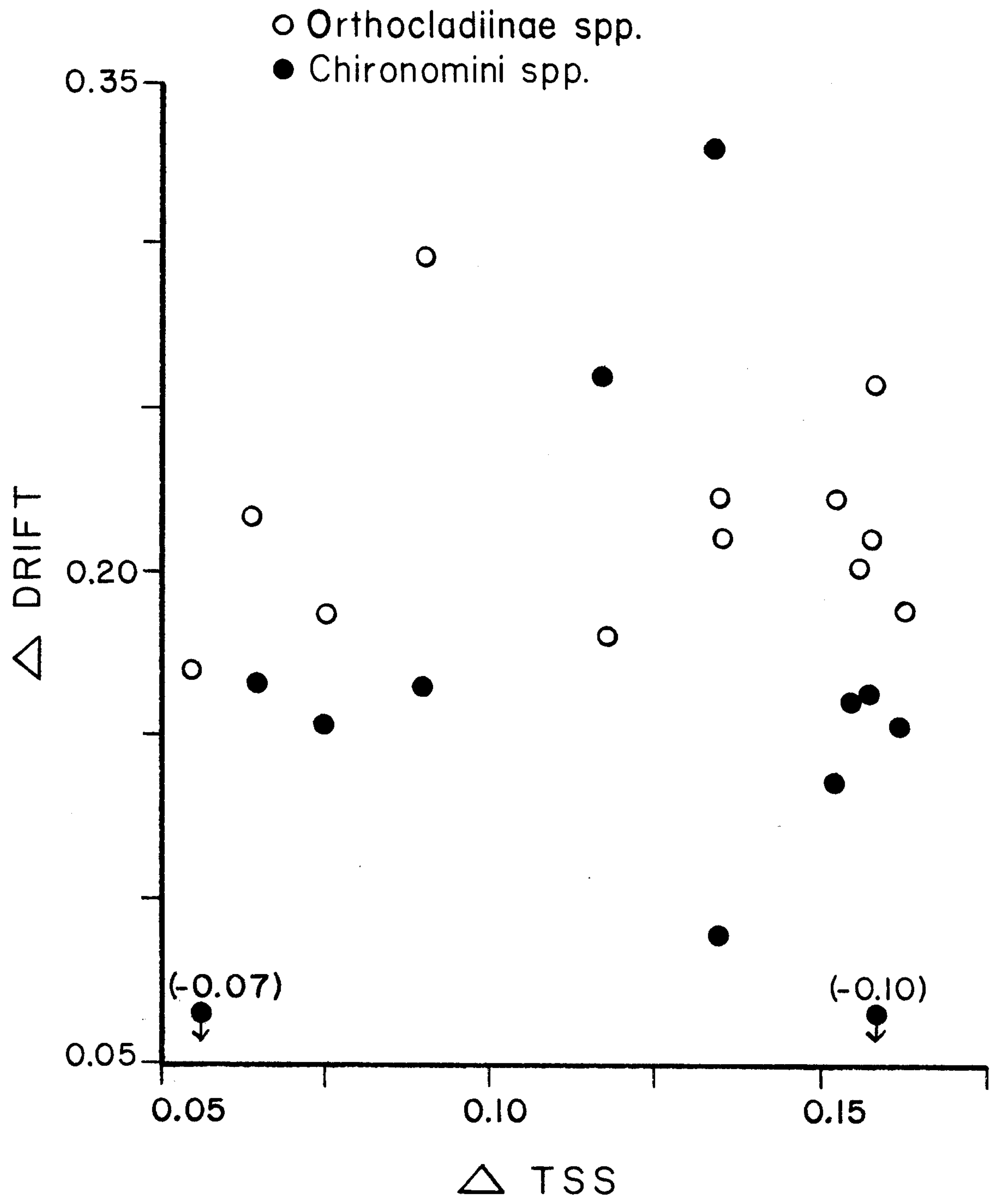


Figure 5.11. Relationship between rates of change in drift of chironomids and TSS during the 1982 silt run. Regression equations were not signficant for either Chironomini spp. (r = 0.10, N = 12) or Orthocladiinae (r = 0.09, N = 12).

5.4 Laboratory Studies

The short-term response of L. fumidus larvae to increased concentrations of TSS was an increase in tube length (Fig. 5.12). Significant increases in tube length compared to controls first occurred at 50 mg/1, thus the minimum concentration affecting larval behavior is between 30-50 mg/1. This concentration is only 10-30 mg/1 above non-silt run levels for the North Platte River (Sec. 5.1.1). At concentrations of 160 mg/1 and above, the amount of increase in tube length declined compared to increases at lower concentrations. Thus high concentrations exceeded the ability of larvae to construct new tube material.

The increase in larval tube length with increasing levels of TSS represents a potentially severe energetic strain on fast-growing larvae. Although the energy involved in construction of retreats has not been evaluated for chironomids, it is a significant part of energy expenditures in other aquatic insects. Larvae of the caddisfly Sericostoma personatum, for example, use 58-66% of the total energy available for growth in secretions for case construction (Iversen 1974). If these values are similar for chironomids, then L. fumidus larvae could be expending all available energy for tube construction at peak concentrations of TSS during the silt run.

Short-term exposure of larvae and pupae to elevated levels of TSS did not increase mortality compared to controls (Table 5.3). During long-term exposure to 160 mg/1, higher mortality was noted amont pupae but not larvae. All larvae successfully pupated after 5 days, and about 2/3 of these successfully emerged. Pupal mortality was apparently caused by suffocation, because pupae were unable to extend their tubes as the suspended material adhered to outside surfaces.

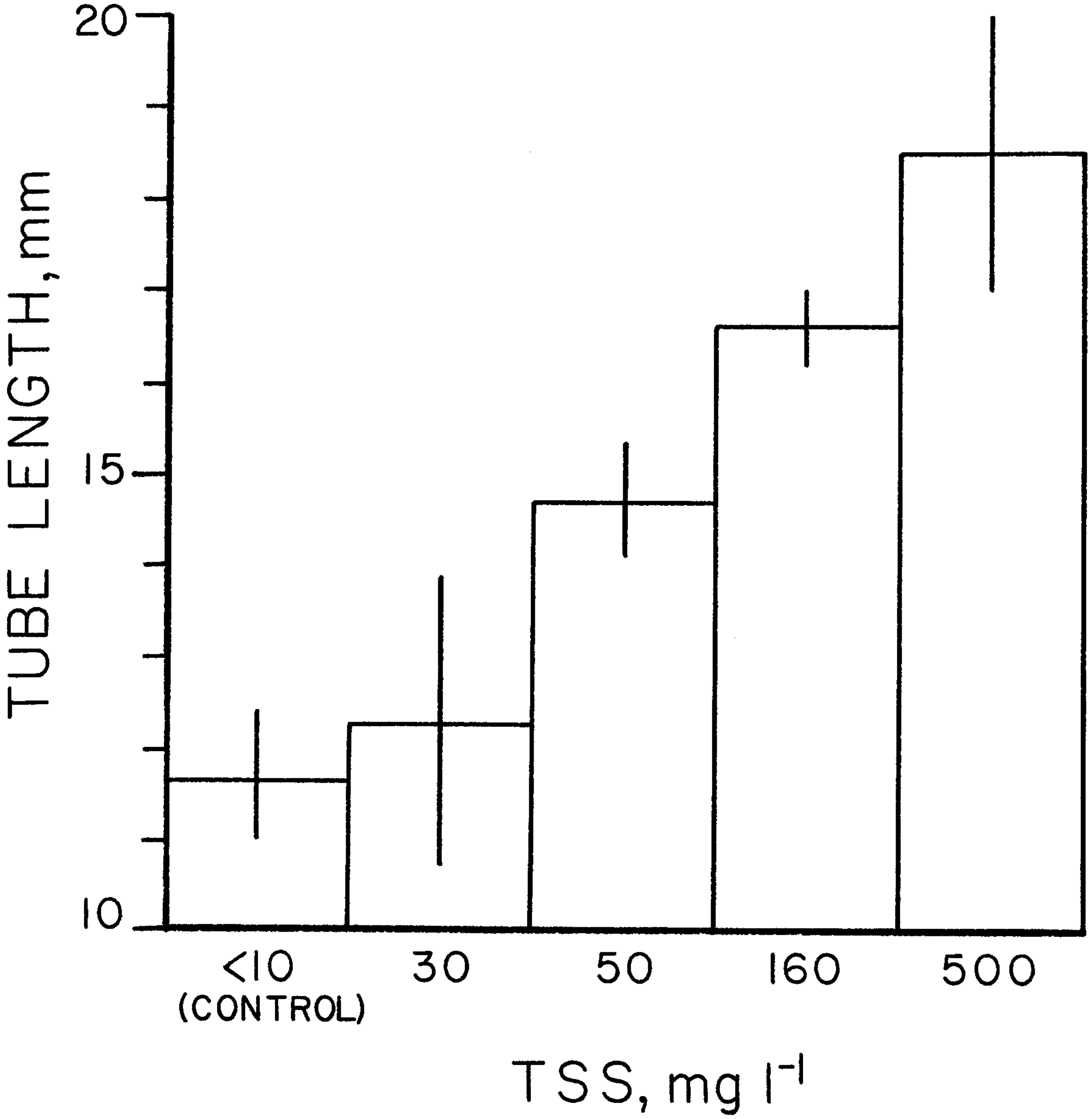


Figure 5.12. Tube lengths of Limnochironomus fumidus larvae in response to additions of suspended sediment. Vertical bars indicate 1 SD. Tube length was related to [TSS] by the equation: tube length (mm) = 1.02 $\log_e$ [TSS] + 11.2 ($r = 0.60$, $N = 55$).

Table 5.3 Mortality of *Limnochironomus fumidus* larvae and pupae during laboratory tests of suspended solids effects.

	Larvae		Pupae	
Treatment Group	Total in test	% mortality	Total in test	% mortality
All controls	48	12.5	30	0
Short-term tests (combined data)	43	11.6	12	8.5
Long-term tests				
at 160 mg/l TSS	11	0	11	36.4
at 400 mg/l TSS	17	88.2	2	50.0

At 400 mg/l, high mortality occurred among larvae with extended exposure. After 2 weeks, only 2 larvae (12% of total) had pupated, and only one of these emerged as an adult. Most larval mortality occurred after 4 days, or the period when pupation normally took place. Thus the high energy costs of tube construction at high concentrations delayed pupation and eventually resulted in death.

The loss of chironomids during the Guernesey Reservoir silt run is related to the extended period (several weeks) of high suspended solids. Under natural conditions that cause high TSS, such as floods, the period of stress is relatively short. Behavioral mechanisms to survive such periods, such as increasing tube length, involve relatively small amounts of energy over the total developmental period. The long period of silt run demands much greater amounts of energy, thus disrupting normal development and increasing mortality.

CHAPTER VI

SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

6.1 Summary and Conclusions

The release of sediment from reservoirs had variable effects on downstream physical-chemical parameters, depending on the source of water used for flushing. In Dry Creek, sediments were flushed from Reservoir No. 15 by using stored water. The rapid, large increase in discharge scoured downstream substrated and immediately altered concentrations of dissolved constituents and water temperatures as a consequence of differences between reservoir and stream water. Suspended solids increased downstream as stream substrates were brought into suspension. Few parameters other than suspended solids were changed in the North Platte River during sediment flushing from Guernsey Reservoir as a result of the use of upstream flows from Glendo Reservoir as the source of water. However, the flushed silts and clays remained in suspension, thus affecting habitats for many kilometers downstream. At both study areas, sediment releases increased downstream concentrations of dissolved phosphorus for periods of a few hours to several weeks. Increases probably occurred in particulate phosphorus bound to sediments.

The scouring of stream substrates in Dry Creek during reservoir releases decimated benthic algae. Algal abundance declined more than 90% after the first release and did not fully recover until several months after the final release. Community composition shifted from a dominance by green algae and diatoms before releases began to crustose cyanobacteria during the release period.

In the North Platte River, nutrients flushed with Guernsey sediments contributed to an increase in algal abundance despite reduced light intensities. The ensuing algal "bloom," composed of *Cladophora glomerata* and epiphytes, produced drifting mats that clogged downstream irrigation structures for the remainder of the release period.

The initial release from Reservoir No. 15 caused extensive losses of benthic invertebrates. All taxa experienced significant declines in density, thus effects were non-selective. Although a few, rare taxa did not recover for a year, most taxa exhibited high recovery rates. Further reservoir releases, however, lowered recovery rates and shifted community composition to small, fast-growing organisms, such as mayflies and simuliids. Recovery rates depended more on the frequency of reservoir releases than on the season of release. Full recovery of the community to pre-release levels of density and biomass was possible if releases did not occur for 2-3 months.

Flushing of sediments from Guernsey Reservoir had selective effects on the benthic fauna. Mayflies and oligochaetes generally increased, whereas chironomids greatly decreased. These effects were caused primarily by the increase in suspended solids, because stream substrates were not significantly altered. The loss of chironomids was related to higher rates of downstream drift at relatively low, but increasing concentrations of suspended solids, and high mortality at peak concentrations (*ca*. 400 mg/l). Despite these losses, populations of chironomids recovered to initial densities within 3 weeks after the flushing process ended. Increased densities of other taxa resulted from the greater abundance of algae. In addition to their value as food, algae trapped some sediment, thereby providing more favorable microhabitats.

The rapid response of macroinvertebrates in the North Platte River to environmental changes caused by the flushing of reservoir sediments reflects the highly perturbed conditions created by the reservoir throughout the year. Temporal contraction of river discharge to brief periods in spring and summer combined with extreme fluctuations in releases select for species that are able to colonize and complete larval development between major disturbances (Henricson and Müller 1979). Warm stream temperatures, continuous recruitment, and rapid life cycles allow high population rates, thus an additional disturbance, such as the release of sediments, has only short-term effects on extant populations.

6.2 Recommendations

If the flushing of reservoir sediments must occur, then results of this study indicate that certain procedures will reduce adverse effects on the abundance and composition of downstream biotic communities. These procedures relate to the rate, frequency, and season of flushing.

Sediments should be released as gradually as possible to minimize substrate scour and rates of increase in suspended solids. In addition, profundal sediments may be anaerobic and contain potential toxins, such as hydrogen sulfide. Slow rates of release will allow time for reaeration and dilution. Sediments may also contain high concentrations of limiting plant nutrients, particularly phosphorus. Slow release rates would reduce peak concentrations, thus the likelihood of nuisance algal "blooms." Finally, gradual releases with lower peak concentrations of suspended solids would reduce mortality among some invertebrates (e.g., chironomids) and settling of sediments on stream substrates. Our data indicate that peak concentrations of TSS should not exceed 200 mg/l in streams with typically low levels of TSS during normal flows.

Results from the Dry Creek study indicate that sediments should be released during one distinct period, or several closely-spaced periods, rather than many, widely-spaced intervals. Many invertebrates had high recovery rates after a single disturbance, whereas recovey times greatly lengthened after multiple disruptions. Benthic algae could only recover during an extended period without disruption.

Although the season of release was not an important factor in determining the extent of adverse effects of sediment release at the sites studied, it does become a factor in streams with perennial flows, e.g., reaches with minimum flow requirements. In this situation, a greater proportion of the biota is composed of organisms with long life cycles and more distinct periods of emergence and recruitment (Stanford and Ward 1981). Because faunal composition and life cycles may vary from stream to stream, pre-release studies would be required to determine major periods of recruitment and growth.

LITERATURE CITED

Bergersen, E. P. 1976. Aging and evaluation of Twin Lakes sediments. pp. 29-29 In: LaBounty, J.F. (ed.). Studies of the benthic environment of Twin Lakes Colorado. U.S. Dept. of Interior, Bureau of Reclamation, REC-ERC-76-12, Denver, Colorado. 47 p.

Bjornn, T.C., M.A. Brusven, M.P. Molnau, J.H. Milligan, R.A. Klamt, E. Chacho and C. Schaye. 1977. Transport of granitic sediment in streams and its effects on insects and fish. Bull. No. 17; Forest, Wildlife and Range Exp. Sta., U. of Idaho, Moscow. 43 p.

Brown, R.J. 1975. Reservoir and lake sedimentation. Nat. Tech. Inform. Serv., Springfield, Va. 110 p.

Buscemi, P.A. 1961. Ecology of the bottom fauna of Parvin Lake, Colorado. Trans. Am. Microscop. Soc. 80: 266-307.

Chutter, F.M. 1969. The effect of silt and sand on the invertebrate fauna of streams and rivers. Hydrobiologia 34: 57-76.

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

Craig, D.A. 1977. A reliable chilled water stream for rheophilic insects. Mosquito News 37: 773-774.

Cummins, K.W. 1973. Trophic relations of aquatic insects. Ann. Rev. Entomol. 18: 183-206.

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

Cummins, K.W., C.A. Tryon, Jr., and R.T. Hartman (eds.). 1966. Organism-substrate relationships in streams. Spec. Publ. No. 4, Pymatuning Lab. Ecol., U. of Pittsburg, Penn. 145 p.

Edmonds, J.S. and J.V. Ward. 1979. Profundal benthos of a multibasin foothills reservoir in Colorado, USA. Hydrobiologia 63: 199-208.

Edmunds, G.F., Jr., S.L. Jensen, and L. Berner. 1976. The mayflies of North and Central America. U. Minn. Press, Minneapolis. 330 p.

Edmunds, G.F. and G.G. Musser. 1960. The mayfly fauna of Green River in the Flaming Gorge Reservoir Basin, Wyoming and Utah. pp. 112-123 In: Ecological Studies of the flora and fauna of Flaming Gorge Reservoir Basin, Wyoming and Utah. U. Utah. Anthrop. Pap. No. 48.

Friant, S.L., R. Patrick and L Lyons. 1980. Effects of nonsettleable biosolids on stream organisms. JWPCF 52: 351-363.

Galehouse, J.S. 1971. Sedimentation analysis. pp. 69-94 In: Carver, R.E. (ed.). Procedures in sedimentary petrology. Wiley-InterScience, John Wiley & Sons, New York. 653 p.

Gammon, J.R. 1970. The effect of inorganic sediment on stream biota. U.S. Environ. Protect. Agency, 18050 DWC, Washington, D.C. 141 p.

Gerloff, G.C. and G.P. Fitzgerald. 1976. The nutrition of Great Lakes *Cladophora*. U.S. Environ. Protect. Agency, EPA-600/3-76-044, Duluth, Minn. 111 p.

Golterman, H.L., R.S. Clymo, and M.A.M. Ohnstad. 1978. Methods for physical and chemical analysis of fresh waters. IBP Handbook No. 8, 2nd Ed. Blackwell Scientific, Great Britain. 214 p.

Gray, L.J. and J.V. Ward. 1978. Environmental effects of oil shale mining and processing, Part II: the aquatic macroinvertebrates of the Piceance Basin, Colorado, prior to oil shale processing. U.S. Environ. Protect. Agency, EPA-600/3-78-097, Duluth, Minn. 40 p.

Green, R.H. 1979. Sampling design and statistical analysis for environmental biologists. John Wiley & Sons, New York. 257 p.

Hamilton, J.D. 1961. The effect of sand-pit washings on a stream fauna. Verh. Int. Verein. Limnol. 14: 435-439.

Hannan, H.H. 1979. Chemical modifications in reservoir-regulated streams. pp. 75-94 In: Ward, J.V. and J.A. Stanford (eds.). The ecology of regulated streams. Plenum Press, New York. 398 p.

Henricson, J. and K. Müller. 1979. Stream regulation in Sweden with some examples from Central Europe. pp. 183-199 In: Ward, J.V. and J.A. Stanford (eds.). The ecology of regulated streams. Plenum Press, New York. 398 p.

Herbst, R.P. 1969. Ecological factors and the distribution of *Cladophora glomerata* in the Great Lakes. Am. Midl. Nat. 82: 90-98.

Hynes, H.B.N. 1960. The biology of polluted waters. U. of Toronto Press, Ontario, Canada. 202 p.

Hynes, H.B.N. 1970. The ecology of running waters. U. of Toronto Press, Ontario, Canada. 555 p.

Ingram, R.L. 1971. Sieve analysis. pp. 49-67 In: Carver, R.E. (ed.). Procedures in sedimentary petrology. Wiley-InterScience, John Wiley & Sons, New York. 398 p.

Iversen, T.M. 1974. Ingestion and growth in *Sericostoma personatum* (Trichoptera) in relation to the nitrogen content of ingested leaves. Oikos 25: 278-282.

Lowe, R.L. 1979. Phytobenthic ecology and regulated streams. pp. 25-34 In: Ward, J.V. and J.A. Stanford (eds.). The ecology of regulated streams. Plenum P ress, New York. 398 p.

Mackay, R.J. 1977. Behavior of Pycnopsyche (Trichoptera: Limnephilidae) on mineral substrates in laboratory streams. Ecology 58: 191-195.

Maciolek, J.A. 1962. Limnological organic analysis by quantitative dichromate oxidation. U.S. Fish and Wildlife Service Res. Rep. No. 60, Washington, D.C. 61p.

McBride, E.F. 1971. Mathematical treatment of size distribution data. pp. 109-127 In: Carver, R.E. (ed.). Procedures in sedimentary petrology. Wiley-InterScience, John Wiley & Sons, New York. 653 p.

Murphy, J. and J. P. Riley. 1962. Determination of phosphate in natural waters. Analyt. Chim. Acta 27: 31-36.

Newell, R.L. 1976. The life history of a multivoltine mayfly Tricorythodes minutus (Traver) with special reference to the effects of temperature on growth and development. Ph.D. thesis, Idaho State U., Pocatello. 129 p.

Nie, N.H., C.H. Hull, J.G. Jenkins, K. Steinbrenner, and D.H. Bent. 1975. Statistical package for the social sciences, 2nd ed. McGraw-Hill, New York. 675 p.

Nisbet, M. 1961. Un example de pollution de rivière par vidage d'une retenue hydroelectrique. Verh. Int. Verein. Limnol. 14: 678-680.

Nuttall, P.M. and G.H. Bielby. 1973. The effect of china-clay wastes on stream invertebrates. Environ. Pollut. 5: 77-86.

Orlóci, L. 1975. Multivariate analysis in vegetation research. Dr. W. Junk, The Hague.

Pennak, R.W. 1953. Fresh-water invertebrates of the United States. Ronald Press, New York. 769 p.

Perry, W.B. and J.A. Stanford. 1982. Algal growth stimulus by phosphorus in Flathead Lake, Montana, sediments. Northwest Sci. 56: 48-52.

Rabeni, C.F. and G.W. Minshall. 1977. Factors affecting microdistribution of stream benthic insects. Oikos 29: 33-43.

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

Siegfried, C.A. and A.W. Knight. 1977. The effects of washout in a Sierra foothill stream. Am. Midl. Nat. 98: 200-207.

Simons, D.B. 1979. Effects of stream regulation on channel morphology. pp. 95-111 In: Ward, J.V. and J.A. Stanford (eds.). The ecology of regulated streams. Plenum Press, New York. 398 p.

Sorensen, D.L., M.M. McCarthy, E.J. Middlebrooks, and D.B. Porcella. 1977. Suspended and dissolved solids effects on freshwater biota: a review. U.S. Environ. Protect. Agency, EPA-600/3-77-042, Washington, D.C. 64 p.

Stanford, J.A. 1971. Quantitative sampling of macroinvertebrates in the Cache la Poudre River, Colorado. M.S. Thesis, Colorado State U., Fort Collins. 63 p.

Stanford, J.A. and J.V. Ward. 1981. Preliminary interpretations of the distribution of Hydropsychidae in a regulated river. pp. 323-328 In: G.P. Moretti (ed.). Proc. of the 3rd Int. Symp. on Trichoptera, Series Entomologica, Vol. 20, Dr. W. Junk, The Hague.

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

Tett, P., M.G. Kelly, and G.M. Hornberger. 1977. Estimation of chlorophyll _a_ and phaeophytin _a_ in methanol. Limnol. Oceanogr. 22: 579-580.

United States Geological Survey. 1980. Water resources data for Wyoming, Water Year 1979, Vol. 1: Missouri River Basin. Nat. Tech. Inform. Serv., Springfield, Va. 642 p.

United States Inter-Agency Committee on Water Resources. 1965. A study of methods used in measurement and analysis of sediment loads in streams. Rep. J, St. Anthony's Fall Hydraulic Lab., Minneapolis, Minn. 5 p.

Ward, J.V. and J.A. Stanford (eds.). 1979. The ecology of regulated streams. Plenum Press, New York. 398 p.

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

Whitton, B.A. 1970. Biology of _Cladophora_ in freshwaters. Water Research 4: 457- 476.

Williams, R.D. and R.N. Winget. 1979. Macroinvertebrate response to flow manipulation in the Strawberry River, Utah (U.S.A.). pp. 365-376 In: Ward, J.V. and J.A. Stanford (eds.). The ecology of regulated streams. Plenum press, New York. 398 p.

Wong, S.L. and B. Clark. 1976. Field determination of the critical nutrient concentrations for _Cladophora_ in streams. J. Fish. Res. Bd. Can. 33: 85-92.

Wunderlich, R.C. 1974. Evaluation of lake pollution by analysis of sediments. M.S. thesis, Colorado State Univ., Fort Collins. 77 p.

Appendix A. Physical-chemical parameters in Dry Creek, Larimer County, Colorado, for the period 16 February 1981 to 14 January 1982.

Table A.1. Site 1.
Table A.2. Site 2.
Table A.3. Dry Creek upstream from Site 2 and Reservoir No. 15.

Table A.1.

Day	Date	Time	°C	pH	Conductance µS cm	Ca mg/l	Mg mg/l	Total Alkalinity meg/l	SO_4-S mg/l	NO_3-N mg/l	SRP mg/l
-1	16 Feb.	1015	9.0	8.10	1,050	187	56	4.0	268	0.21	<0.01
0	17 Feb.	0845	8.0	8.00	1,180	213	55	5.0	291	0.06	<0.01
		0850	5.5	8.15	680	70	67	4.7	80	0.60	0.26
		0940	5.5	8.32	680	47	40	4.5	98	0.65	<0.01
1	18 Feb.	1110	5.3	8.35	520	56	44	3.2	51	0.13	0.07
3	20 Feb.	0913	6.0	8.50	530	53	42	3.6	51	0.20	0.34
7	24 Feb.	0953	4.0	8.60	490	46	22	3.5	40	0.05	0.23
13	2 Mar.	1025	5.0	8.15	750	107	27	7.6	168	0.05	<0.01
16	5 Mar.	0910	5.0	8.40	450	42	19	2.6	48	0.11	0.02
22	11 Mar.	1115	6.0	8.40	410	40	17	2.5	50	0.08	0.01
30	19 Mar.	1030	5.0	8.30	410	43	17	2.3	48	0.07	<0.01
47	5 Apr.	1338	12.0	8.10	1,070	173	31	3.4	250	0.19	<0.01
59	17 Apr.	1050	11.0	8.20	450	40	17	2.1	33	<0.05	<0.01
77	5 May	0910	14.0	8.45	390	37	12	2.1	28	0.05	<0.01
97	25 May	1343	19.0	8.30	860	195	35	3.2	261	<0.05	<0.01
126	23 June	1100	18.0	8.60	360	30	11	1.8	24	<0.05	0.07
158	25 June	1045	21.0	8.10	360	42	4	1.8	28	<0.05	<0.01
190	26 Aug.	1400	21.6	8.25	1,110	146	27	3.0	198	<0.05	0.02
214	19 Sept.	1200	17.0	---	330	34	9	3.0	20	<0.05	0.12
245	20 Oct.	1115	14.3	8.05	1,130	181	40	3.2	244	<0.05	<0.01
273	17 Nov.	1135	12.5	8.15	1,190	205	29	3.4	249	<0.05	<0.01
305	19 Dec.	1050	6.3	8.00	1,350	240	34	3.7	291	<0.05	<0.01
331	14 Jan.	1100	6.6	8.00	1,100	181	50	3.5	201	0.05	0.06

Table A.1. (continued)

Date	Time	Dissolved Organic Carbon, mg/1	Dissolved Oxygen, mg/1	H_2S-S mg/1	Total Suspended Solids, mg/1	Discharge m^3s^{-1}
16 Feb.	1015	13.2	9.79	0	15.1	<0.1
17 Feb.	0845	12.7	-	-	16.3	<0.1
	0850	12.1	8.98	0	536.3	1.4
	0940	11.8	10.15	0	21.7	1.4
18 Feb.	1110	10.4	10.65	0	14.0	1.4
20 Feb.	0913	7.3	10.78	0	9.6	0.9
24 Feb.	0953	7.8	10.85	0	14.0	1.4
2 March	1025	9.1	9.85	0	12.1	<0.1
5 March	0910	8.7	10.25	-	12.7	1.0
11 March	1115	8.4	10.55	-	16.8	0.9
19 March	1030	7.7	10.05	-	9.4	0.4
5 Apr.	1338	7.8	9.52	-	11.6	<0.1
17 Apr.	1050	6.5	8.85	-	22.0	2.2
5 May	0910	6.3	8.42	-	13.9	3.0
25 May	1343	7.8	9.33	-	2.4	<0.1
23 June	1100	6.7	7.50	-	48.0	0.5
25 July	1045	9.0	6.95	-	38.2	0.7
26 Aug.	1400	9.2	9.15	-	15.1	<0.1
19 Sept.	1200	---	7.80	-	35.6	1.6
20 Oct.	1115	8.7	9.56	-	13.3	<0.1
17 Nov.	1135	7.2	8.75	-	3.2	<0.1
19 Dec.	1050	---	9.48	0	3.6	<0.1
14 Jan.	1100	9.7	10.65	-	8.7	<0.1

Table A.2

Day	Date	Time	C	pH	Conductance µS cm-1	Ca mg/l	Mg mgl	Total Alkalinity meg/l	SO_4-S mg /l	NO_3-N mg/l	SRP mg/l
-1	16 Feb.	1225	11.0	8.40	980	104	61	5.8	205	3.15	<0.01
0	17 Feb.	0722	4.7	8.20	930	103	58	5.4	191	3.51	<0.01
		0907	6.0	8.10	1,380	197	123	5.3	379	1.67	<0.01
		1005	5.5	8.33	770	72	44	5.0	129	0.76	<0.01
1	18 Feb.	1135	5.5	8.50	550	56	28	3.7	68	0.17	<0.01
3	20 Feb.	0818	5.0	8.50	550	53	45	4.0	54	0.21	0.07
7	24 Feb.	1015	5.0	8.50	510	48	23	3.6	44	0.07	0.06
13	2 March	1054	4.5	8.40	570	61	25	3.7	77	1.00	<0.01
16	5 March	0955	5.0	8.40	490	46	25	3.1	55	0.28	<0.01
22	11 March	1146	6.5	8.40	430	43	19	2.6	51	0.12	0.01
30	19 March	1110	5.0	8.30	450	46	23	2.6	58	0.19	<0.01
47	5 April	1415	16.0	8.40	1,090	103	51	4.6	191	2.39	<0.01
59	17 April	1120	12.0	8.45	450	40	18	2.6	40	<0.05	<0.01
77	5 May	0935	14.0	8.45	400	37	12	2.1	35	<0.05	<0.01
97	25 May	1415	20.0	8.35	770	111	59	4.5	164	3.58	<0.01
126	23 June	1155	19.0	8.60	410	37	10	1.9	36	0.13	0.03
158	25 July	1125	22.0	8.20	430	42	17	1.9	40	0.16	<0.01
190	26 Aug.	1500	24.5	8.25	1,090	88	49	4.5	150	2.04	<0.01
214	19 Sept.	1233	17.0	---	350	34	11	1.8	24	<0.05	<0.01
245	20 Oct.	1145	13.1	8.40	1,060	111	54	4.6	182	2.68	<0.01
273	17 Nov.	1145	11.8	8.35	1,050	107	52	4.3	163	3.38	0.54
305	19 Dec.	1130	4.6	8.38	1,100	144	34	4.6	163	3.26	<0.01
331	14 Jan.	1125	4.5	8.20	1,050	106	43	4.6	128	3.08	<0.01

Table A.2. (continued)

Date	Time	Dissolved Organic Carbon, mg/l	Dissolved Oxygen, mg/l	H_2S-S mg/l	Total Suspended Solids, mg/l	Discharge m^3s^{-1}
16 Feb.	1225	8.4	9.60	0	27.4	0.1
17 Feb.	0722	9.5	9.55	-	2.3	0.1
	0907	14.0	6.70	0	7,913.6	1.4
	1005	10.0	-	-	1,092.4	1.4
18 Feb.	1135	9.9	10.53	0	77.1	1.4
20 Feb.	0818	7.9	10.40	0	25.5	0.9
24 Feb.	1015	7.5	10.70	0	56.7	1.4
2 Mar.	1054	8.3	10.20	0	48.9	0.1
5 Mar.	0955	8.5	10.50	0	57.6	1.0
11 Mar.	1146	7.8	10.40	-	20.6	0.9
19 Mar.	1110	8.0	10.09	-	18.5	0.5
5 April	1415	6.3	8.20	-	11.5	<0.1
17 April	1120	7.6	8.85	-	48.9	2.3
5 May	0935	6.3	8.46	-	20.3	3.0
25 May	1415	6.3	8.78	-	4.5	0.1
23 June	1155	5.1	7.48	-	56.7	0.5
25 July	1125	13.1	7.43	-	55.3	0.7
26 Aug.	1500	7.3	7.65	-	7.2	0.1
19 Sept.	1233	---	7.93	-	37.0	1.7
20 Oct.	1145	5.0	9.93	-	10.3	0.1
17 Nov.	1145	7.6	9.60	-	11.9	<0.1
19 Dec.	1130	---	10.43	0	24.8	<0.1
14 Jan.	1125	7.4	10.45	-	18.0	<0.1

Table A.3.

Parameter	Reservoir 16 Feb.	Dry Creek 16 Feb.	 20 Feb.	 11 March
Temperature, °C	4.5	10.5	7.0	10.5
pH	8.30	8.40	8.28	8.30
Conductance, µS cm^{-1}	480.	830.	910.	830.
Ca++, mg/1	53.	93.	88.	88.
Mg++, mg/1	23.	46.	76.	47.
Total Alkalinity, meq/1	3.0	5.4	6.0	4.6
SO_4-S, mg/1	57.	138.	146.	146.
NO_3-N, mg/1	0.06	3.48	1.49	2.29
SRP, mg/1	0.01	<0.01	<0.01	0.01
Dissolved Organic Carbon, mg/1	11.0	7.1	5.0	6.8
Dissolved Oxygen, mg/1	12.42	9.18	-	-
H_2S-S, mg/1	0	-	-	-
Total Suspended Solids, mg/1	-	14.5	-	16.1
Discharge, m^3 s^{-1}	-	<0.1	<0.1	<0.1

Appendix B. Macroinvertebrate data for Dry Creek, Larimer County, Colorado, for the period 16 February 1981 to 14 January 1982.

Table B.1. Densities (mean number per 325-cm ^{2}sample) at Site 1. * = 1.

Table B.2. Same for Site 2.

Table B.3. Total density (number per m^2) and biomass (grams wet weight per m^2) at Sites 1 and 2; mean values are given with factors for deriving 95% confidence intervals (in parentheses): mean $\overset{\times}{\div}$ factor = 95% conf. limits.

Table B.1

	Sampling Date									
Taxon	16 II	20 II	5 III	19 III	5 IV	17 IV	25 V	23 VI	25 VII	26 VIII
Ephemeroptera										
Tricorythodes minutus	13	0	*	0	7	*	2	0	1	17
Baetis spp.	0	0	*	*	0	*	0	0	*	2
Caenis simulans	0	0	0	0	4	0	1	0	0	1
Trichoptera										
Hydropsyche sp.	5	0	0	1	21	2	13	*	0	0
Cheumatopsyche sp.	16	*	1	1	12	2	14	*	2	56
Hydroptila sp.	0	0	0	0	*	0	0	0	0	0
Diptera										
Tipula sp.	*	0	0	0	0	0	0	0	0	0
Hexatoma sp.	*	0	0	0	0	0	0	0	0	0
Simulium arcticum	2	0	0	*	*	0	*	18	*	5
Limnophora sp.	0	*	0	0	3	0	0	*	*	0
Ceratopogonidae sp.	0	0	*	0	0	1	0	1	*	*
Empididae sp.	0	0	0	0	0	0	*	*	0	0
Orthocladiinae spp.	933	1	6	20	292	9	21	6	10	74
Chironominae spp.	23	*	0	0	35	7	25	*	2	11
Tanypodinae spp.	0	0	0	0	2	*	0	0	0	0

Table B.1 (Continued)

Taxon	Sampling Date									
	16 II	20 II	5 III	19 III	5 IV	17 IV	25 V	23 VI	25 VII	26 VIII
Lepidoptera										
Petrophyla sp.	0	0	0	0	0	0	*	0	0	*
Plecoptera										
Isoperla quinquepunctata	0	0	0	0	0	0	0	0	0	0
Amphipoda										
Hyallela azteca	0	0	0	0	0	0	0	0	0	0
Gastropoda										
Physa sp.	0	0	0	0	1	0	0	0	0	2
Oligochaeta spp.	321	1	0	*	97	42	23	1	1	41

Table B.1 (continued)

Taxon	Sampling Date				
	19 IX	20 X	17 XI	19 XII	14 I
Ephemeroptera					
Tricorythodes minutus	1	44	17	173	14
Baetis spp.	1	1	16	3	1
Caenis simulans	0	0	0	0	0
Trichoptera					
Hydropsyche sp.	0	2	23	8	19
Cheumatopsyche sp.	23	42	136	45	24
Hydroptila sp.	*	*	1	3	0
Diptera					
Tipula sp.	0	0	0	0	2
Hexatoma sp.	0	0	0	*	*
Simulium arcticum	12	4	254	12	6
Limnophora sp.	0	0	0	0	0
Ceratopogonidae sp.	*	0	0	0	*
Empididae sp.	*	0	0	3	1
Orthocladiinae spp.	51	103	1,719	2,142	1,540
Chironominae spp.	1	1	9	71	155
Tanypodinae spp.	0	0	0	2	0
Stratiomyiidae sp.	0	0	0	*	*
Lepidoptera					
Petrophyla sp.	*	0	3	4	1
Plecoptera					
Isoperla quinquepunctata	0	0	0	0	0
Amphipoda					
Hyallela azteca	0	0	0	0	0

Table B.1 (continued)

Taxon	Sampling Date				
	19 IX	20 X	17 XI	19 XII	14 I
Gastropoda					
Physa sp.	1	4	3	6	6
Oligochaeta spp.	15	780	803	644	449

Table B.2

Taxon	Sampling Date								
	16 II	20 II	5 III	19 III	5 IV	25 V	23 VI	25 VII	26 VIII
Ephemeroptera									
Tricorythodes minutus	6	0	2	1	1	4	0	4	17
Baetis spp.	9	0	2	1	16	20	0	3	45
Caenis simulans	0	0	0	0	0	0	0	0	0
Trichoptera									
Hydropsyche sp.	8	*	4	4	3	4	0	1	1
Cheumatopsyche sp.	6	*	*	2	1	3	0	4	8
Hydroptila sp.	0	0	0	0	0	0	0	0	0
Diptera									
Tipula sp.	1	0	0	0	2	0	0	*	0
Hexatoma sp.	3	0	*	*	0	0	0	*	0
Simulium arcticum	0	0	2	*	2	184	1	0	1
Limnophora sp.	0	0	0	0	0	0	0	0	0
Ceratopogonidae sp.	0	*	*	0	*	3	3	*	0
Empididae sp.	0	0	1	1	0	0	0	0	0
Orthocladiinae spp.	477	4	10	12	75	7	*	*	5
Chironominae spp.	11	0	3	3	77	97	1	14	72
Tanypodinae spp.	*	0	0	0	2	0	0	*	2
Lepidoptera									
Petrophyla sp.	0	0	0	*	0	0	0	0	0
Plecoptera									
Isoperla quinquepunctata	2	0	0	0	*	*	0	0	0
Amphipoda									
Hyallela azteca	1	0	0	0	0	0	0	0	0

Table B.2

Taxon	Sampling Date								
	16 II	20 II	5 III	19 III	5 IV	25 V	23 VI	25 VII	26 VIII
Gastropoda									
Physa sp.	3	0	0	0	0	0	0	0	0
Oligochaeta spp.	49	3	3	2	40	57	3	4	9

Table B.2 (continued)

Taxon	Sampling Date				
	19 IX	20 X	17 XI	19 XII	14 I
Ephemeroptera					
Tricorythodes minutus	1	5	2	18	6
Baetis spp.	3	25	2	17	7
Caenis simulans	0	0	0	0	0
Trichoptera					
Hydropsyche sp.	0	1	1	5	3
Cheumatopsyche sp.	2	4	1	3	0
Hydroptila sp.	0	0	0	0	0
Diptera					
Tipula sp.	0	0	0	*	0
Hexatoma sp.	1	2	5	1	5
Simulium arcticum	0	3	19	4	4
Limnophora sp.	0	0	0	0	0
Ceratopogonidae sp.	2	*	0	0	1
Empididae sp.	0	0	0	0	0
Orthocladiinae spp.	2	31	175	259	142
Chironominae spp.	6	11	0	21	16
Tanypodinae spp.	*	2	*	*	*
Stratiomyiidae sp.	*	0	0	0	*
Lepidoptera					
Petrophyla sp.	0	0	0	0	0
Plecoptera					
Isoperla quinquepunctata	0	0	0	*	0
Amphipoda					
Hyallela azteca	0	0	0	0	*

Table B.2 (continued)

Taxon	Sampling Date				
	19 IX	20 X	17 XI	19 XII	14 I
Gastropoda					
Physa sp.	1	0	0	*	0
Oligochaeta spp.	3	21	28	54	8

Table B.3

Day after initial release	Date	Site 1		Site 2	
		No. m^{-2}(X10^3)	g m^{-2}	No. m^{-2}(X10^3)	g m^{-2}
-1	16 II	40.3(2.5)	42.2(3.0)	17.7(2.0)	36.3(1.7)
3	20 II	0.1(3.2)	0.1(14.0)	0.2(12.6)	0.6(8.0)
16	5 III	0.2(4.0)	0.2(6.0)	0.8(5.8)	1.2(9.9)
30	19 III	0.8(1.6)	0.8(4.0)	0.8(4.1)	2.8(2.4)
47	5 IV	14.6(1.4)	12.3(6.3)	6.8(2.0)	32.9(6.0)
59	17 IV	1.9(1.1)	1.2(3.8)	---	---
97	25 V	3.1(4.6)	5.7(6.3)	11.7(2.8)	12.1(5.3)
126	23 VI	0.8(4.4)	0.4(5.3)	0.2(2.0)	0.4(7.0)
158	25 VII	0.6(2.8)	0.4(6.4)	0.9(2.6)	1.9(8.5)
190	26 VIII	6.4(2.1)	7.7(2.7)	4.9(1.6)	2.8(3.3)
214	19 IX	3.2(5.1)	3.5(4.0)	0.6(2.1)	0.2(3.2)
245	20 X	30.1(2.0)	10.4(9.2)	3.2(1.8)	1.3(4.6)
273	17 XI	91.8(1.3)	48.3(3.3)	7.2(2.1)	4.2(1.7)
305	19 XII	95.9(1.5)	51.9(1.6)	11.7(3.3)	5.2(8.5)
331	14 I	68.2(1.5)	44.5(2.8)	5.9(2.4)	2.7(2.5)

Appendix C. Physical-chemical parameters in the North Platte River, Platte-Goshen Counties, Wyoming, for the period 11 June to 20 August 1981.

Table C.1. Site 1. * = < 1.

Table C.2. Site 2.

Table C.1

Parameter	Sampling date 11 June	2 July	6 July	9 July	16 July	23 July	6 Aug.	20 Aug.
pH	8.50	8.25	8.33	8.30	8.13	8.28	8.30	8.15
Conductance, μS^{-1}	750	690	820	690	780	750	700	530
Ca^{++}, mg/1	61	63	62	57	58	61	56	55
Mg^{++}, mg/1	24	21	20	24	25	22	22	19
Total Alkalinity, meq/1	3.0	2.9	2.9	3.0	2.9	2.9	2.8	2.6
SO_4-S, mg/1	76	78	100	84	93	90	78	60
SRP, mg/1	<0.01	0.01	<0.01	0.03	0.06	0.02	0.01	0.01
NO_3-N, mg/1	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Organic Carbon, mg/1	11.4	12.7	9.1	10.7	10.5	11.4	9.3	7.9
Dissolved Oxygen, mg/1	8.17	9.00	8.00	8.46	8.00	7.90	8.87	10.3
H_2S-S, mg/1	0	0	0	0	0	0	0	0
Midday Temperature,°C	20.1	18.0	20.4	19.6	21.7	22.1	23.0	23.7
Total Suspended Solids, mg/1; mean	7.6	19.2	18.2	107.3	339.3	259.3	16.1	9.8
95% confidence limits	5.8 to 10.1	15.0 to 24.6	16.4 to 20.0	104.3 to 110.5	333.6 to 344.8	254.7 to 264.0	15.3 to 16.9	7.9 to 12.1

Table C.2

Parameter	Sampling date 11 June	2 July	6 July	9 July	16 July	23 July	6 Aug.	20 Aug.
pH	8.50	8.25	8.39	8.30	8.20	8.29	8.31	8.19
Conductance, μS^{-1}	750	670	820	720	750	750	670	530
Ca^{++}, mg/l	62	64	61	57	62	60	58	56
Mg^{++}, mg/l	24	21	21	23	21	21	20	18
Total Alkalinity, meq/l	3.0	3.1	3.0	3.0	3.0	3.0	2.8	2.6
SO_4-S, mg/l	76	72	101	91	83	85	80	67
SRP, mg/l	<0.01	0.01	<0.01	<0.01	0.02	0.02	<0.01	0.01
NO_3-N, mg/l	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Organic Carbon, mg/l	10.2	9.7	10.5	10.1	10.8	14.3	8.3	7.3
Dissolved Oxygen, mg/l	8.25	9.05	7.90	8.10	7.62	7.90	8.80	10.36
H_2S-S, mg/l	0	0	0	0	0	0	0	0
Midday Temperature,°C	23.6	18.6	22.8	19.3	21.0	22.1	22.9	23.2
Total Suspended Solids, mg/l; mean	15.7	22.7	15.0	103.0	422.4	302.8	29.0	7.8
95% confidence limits	8.2 to 30.0	16.9 to 30.4	12.0 to 18.6	100.5 to 105.6	410.3 to 434.8	295.0 to 311.1	27.1 to 30.9	6.3 to 9.6

Appendix D. Macroinvertebrate data for the North Platte River, Platte-Goshen counties, Wyoming, for the period 11 June to 20 August 1981.

Table D.1. Mean densities at Site 1.

Table D.2. Same for Site 2.

Table D.3. Rates of change in biological and environmental variables at sampling locations in the North Platte River, 1981 silt run. Rates computed as the first derivative of the slopes of polynomial regressions for the period 6 July-6 August 1981. All variables ll except substrate phi were $\log_e$-transformed before analysis.

Table D.1.

Taxon	Sampling Date						
	11 VI	6 VII	9 VII	16 VII	23 VII	6 VIII	20 VIII
Ephemeroptera							
Baetis insignificans	*	2[a] (1-3)	1 (0-2)	12 (8-18)	35 (26-47)	7 (5-11)	4 (2-8)
Tricorythodes minutus	*	0.6 (.4-1)	0.5 (.4-.7)	0.8 (.6-1)	4 (2-8)	4 (3-5)	14 (10-19)
Heptagenia simplicoides	0	0	*	*	*	*	*
Ephoron album	0	0	0	*	0	0	0
Ephemerella inermis	0	0	0	0	*	0	0
Ameletus sp.	0	0	*	0	*	*	0
Dactylobaetis sp.	0	0	0	0	0	0	1
Trichoptera							
Hydropsyche sp.	*	1	*	1	1	1	2
Hydroptila sp.	0	*	0	*	1	2	6
Polycentropus sp.	0	0	0	0	0	*	0
Limnephilus sp.	0	0	0	0	0	0	0
Diptera							
Orthocladiinae spp.	5	56 (38-83)	17 (12-25)	2 (1-3)	9 (4-19)	27 (21-35)	33 (22-51)
Chironomini spp.	*	7 (5-10)	5 (3-8)	1.4 (1.0-1.9)	1.4 (1.0-2.1)	1.4 (1.1-1.8)	6 (4-9)
Simulium sp.	*	1	1	1	*	*	0
Tanytarsini spp.	*	1	1	*	*	*	1
Tanypodinae spp.	0	0	*	*	*	0	*
Ephydridae spp.	0	*	*	*	0	0	0

Table D.1. (Continued)

Taxon	Sampling Date						
	11 VI	6 VII	9 VII	16 VII	23 VII	6 VIII	20 VIII
Amphipoda							
Gammarus sp.	0	*	0	0	*	*	*
Oligochaeta spp.	3	0.8 (0.6-1.1)	0.6 (0.4-0.9)	0.2 (0.1-0.3)	1.2 (0.9-1.5)	1.2 (0.9-1.7)	4 (2-6)

---

[a] Values in parentheses are 95% confidence limits.

Table D.2.

Taxon	Sampling Date						
	11 VI	6 VII	9 VII	16 VII	23 VII	6 VIII	20 VIII
Ephemeroptera							
Baetis insignificans	5	13[a/] (6-28)	31 (19-49)	22 (10-47)	10 (4-25)	18 (11-29)	23 (12-45)
Tricorythodes minutus	85	14 (8-27)	16 (9-30)	44 (32-61)	46 (32-67)	53 (31-91)	87 (69-110)
Heptagenia simplicoides	11	1	1	1	3	2	1
Ephoron album	15	1	*	*	*	*	0
Ephemerella inermis	*	0	0	0	0	0	0
Ameletus sp.	1	0	0	0	0	*	0
Dactylobaetis sp.	0	0	0	0	0	1	3
Trichoptera							
Hydropsyche sp.	1	*	1	1	1	1	8
Hydroptila sp.	1	0	*	*	*	1	10
Polycentropus sp.	0	0	*	*	*	*	0
Limnephilus sp.	0	0	0	0	0	0	*
Diptera							
Orthocladiinae spp.	148	68 (38-119)	34 (18-64)	5 (2-12)	2 (1-4)	13 (7-23)	86 (52-142)
Chironomini spp.	31	21 (11-40)	21 (10-42)	7 (3-18)	2 (1-4)	3 (2-6)	10 (6-19)
Simulium sp.	3	2	2	1	*	0	*
Tanytarsini spp.	*	1	2	2	0	0	1
Tanypodinae spp.	0	*	*	*	*	0	*
Ephydridae spp.	0	0	0	0	0	0	0

Table D.2. (Continued)

Taxon	Sampling Date 11 VI	6 VII	9 VII	16 VII	23 VII	6 VIII	20 VIII
Amphipoda							
Gammarus sp.	0	0	0	0	0	0	0
Oligochaeta spp.	26	0.1 (0-0.2)	14 (7-30)	18 (9-35)	7 (3-13)	2 (1-6)	35 (24-52)

[a] Values in parentheses are 95% confidence limits.

Table D.3.

			Rate of Change Per Day						
Time Period	Site	Location	Baetis insignificans	Tricorythodes minutus	Orthocladiinae spp	Chironomini spp	Oligochaeta	TSS	Substrate phi
6 VII to 9 VII	1	1	+0.28	-0.02	-0.16	-0.05	-0.02	+0.28	-0.07
		2	+0.25	+0.08	-0.21	-0.17	-0.02	+0.28	+0.04
		3	+0.16	+0.07	-0.12	-0.08	+0.04	+0.29	-0.06
		4	+0.21	+0.08	-0.33	-0.13	-0.01	+0.30	-0.02
		5	+0.25	+0.06	-0.30	-0.12	-0.01	+0.28	+0.01
	2	1	-0.11	+0.09	-0.37	-0.15	+0.28	+0.30	-0.11
		2	-0.14	0.00	-0.32	-0.18	+0.23	+0.29	+0.04
		3	+0.08	+0.17	-0.24	-0.16	+0.11	+0.31	-0.06
		4	+0.03	+0.12	-0.21	-0.13	+0.14	+0.33	-0.02
		5	+0.47	+0.08	-0.27	-0.14	+0.15	+0.31	-0.07
9 VII to 16 VII	1	1	+0.16	+0.01	-0.07	-0.04	-0.01	+0.11	-0.03
		2	+0.13	+0.06	-0.09	-0.10	-0.01	+0.11	+0.02
		3	+0.09	+0.05	-0.05	-0.05	+0.03	+0.12	-0.04
		4	+0.13	+0.07	-0.16	-0.09	0.00	+0.12	0.00
		5	+0.14	+0.06	-0.15	-0.06	0.00	+0.11	+0.01
	2	1	-0.05	+0.06	-0.22	-0.12	+0.12	+0.13	-0.07
		2	-0.07	+0.02	-0.16	-0.11	+0.14	+0.12	+0.01
		3	+0.03	+0.11	-0.14	-0.11	+0.05	+0.14	-0.05
		4	0.00	+0.08	-0.10	-0.08	+0.05	+0.15	-0.03
		5	+0.40	+0.07	-0.15	-0.07	+0.08	+0.14	-0.03
16 VII to 23 VII	1	1	+0.03	+0.03	+0.03	-0.03	-0.01	-0.06	+0.01
		2	+0.01	+0.04	+0.02	-0.03	0.00	-0.05	0.00
		3	+0.02	+0.02	+0.01	-0.03	+0.01	-0.05	-0.01
		4	+0.04	+0.07	+0.02	-0.06	+0.01	-0.06	+0.01
		5	+0.03	+0.06	+0.01	-0.01	+0.02	-0.06	+0.01
	2	1	+0.02	+0.03	-0.07	-0.09	-0.03	-0.03	-0.02
		2	0.00	+0.05	0.00	-0.05	+0.04	-0.04	-0.01
		3	-0.01	+0.04	-0.04	-0.05	-0.02	-0.03	-0.04
		4	-0.02	+0.04	0.00	-0.04	-0.05	-0.03	-0.03
		5	+0.33	+0.05	-0.03	0.00	0.00	-0.03	+0.01

Table D.3. (continued)

Time Period	Site	Location	Rate of Change Per Day Baetis insignificans	Tricorythodes minutus	Orthocladiinae spp	Chironomini spp	Oligochaeta	TSS	Substrate phi
23 VII to 6 VIII	1	1	-0.22	+0.09	+0.21	0.00	-0.01	-0.39	+0.10
		2	-0.24	-0.01	+0.25	+0.11	+0.03	-0.38	-0.04
		3	-0.13	-0.02	+0.14	+0.03	-0.01	-0.40	+0.03
		4	-0.13	+0.05	+0.36	+0.01	+0.03	-0.41	+0.04
		5	-0.19	+0.05	+0.33	+0.11	+0.05	-0.40	+0.02
	2	1	+0.15	-0.04	+0.33	-0.03	-0.34	-0.36	+0.07
		2	+0.15	+0.11	+0.32	+0.07	-0.15	-0.38	-0.06
		3	-0.11	-0.09	+0.17	+0.06	-0.16	-0.37	-0.02
		4	-0.08	-0.04	+0.21	+0.06	-0.23	-0.40	-0.04
		5	+0.19	+0.01	+0.22	+0.14	-0.16	-0.37	+0.10

Appendix E. Drift rates and benthic densities of macroinvertebrates in the North Platte River, Site 1, for the period 7 - 27 July 1982. Mean values are given with factors for 95% confidence (in parentheses): mean $\overset{\times}{\div}$ factor = 95% conf. limits.

Appendix E.

Date	Taxon	Drift Rate No./10^3 m^3 Day	Drift Rate No./10^3 m^3 Night	Benthos No./Sample
7-8 VII				
	Orthocladiinae spp.	1.4 (1.5)	8.5 (1.3)	13 (1.7)
	Chironomini spp.	1.2 (1.5)	17.5 (1.7)	31 (1.8)
	Chironomidae pupae	0.2 (1.6)	0.6 (4.0)	0
	Oligochaeta sp.	0.1 (2.0)	0.4 (2.0)	7 (2.6)
	Simulium sp.	0.2 (1.7)	1.7 (2.0)	3 (4.0)
	All taxa	3.3 (1.4)	32.2 (1.4)	59 (1.9)
16-17 VII				
	Orthocladiinae spp.	41.6 (1.2)	172 (1.3)	47 (1.7)
	Chironomini spp.	15.6 (1.3)	245 (2.3)	96 (1.5)
	Chironomidae pupae	3.8 (1.7)	25.5 (2.1)	3 (1.8)
	Oligochaeta sp.	0.4 (3.0)	0.5 (3.0)	3 (2.2)
	Simulium sp.	0.8 (2.0)	7.5 (2.2)	9 (2.3)
	All taxa	65.5 (1.2)	525 (1.7)	170 (1.5)
26-27 VII				
	Orthocladiinae spp.	40.6 (1.3)	157 (1.4)	5 (1.8)
	Chironomini spp.	17.0 (1.4)	147 (1.4)	25 (1.6)
	Chironomidae pupae	0.7 (2.5)	1.6 (2.9)	1 (2.5)
	Oligochaeta sp.	0.8 (3.2)	0.2 (3.1)	1 (4.0)
	Simulium sp.	1.3 (2.2)	0	0
	All taxa	65.4 (1.3)	337 (1.3)	40 (1.5)