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A Method to Estimate the Production Rate of a Stream Bottom Invertebrate[1]

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ABSTRACT

An approach toward the development of a method to estimate the production rate of primary consumers in a stream was as follows: For a given unit area of stream bottom, the production rate (B) is equal to the algebraic sum of the rate of change in population density (P) and the difference between the rates of removal (R) and accrual (A):

$$B = P + (R - A) .$$

Since removal and accrual both appear, for certain species, to be principally in the form of downstream drift, minimal production rates could be estimated with data on population density and total drift into and off of the unit areas of study. Population density data were obtained from periodic bottom samples, and drift nets were employed in a sampling procedure to estimate total drift rates at the various stream stations.

Production rates of a mayfly, *Baetis vagans* McDunnough, were estimated for two 24-hour periods in two riffles and two pools. The mean production rate on the riffles was 0.28 gram per square meter per day. In the pools, B was negative, indicating consumption; consumption rate in the pools was 0.45 gram per square meter per day.

INTRODUCTION

The estimation of energy flow rates within an aquatic ecosystem must include the production rates of usually three trophic levels: The primary producers (plants), primary consumers (benthic invertebrates, zooplankton), and secondary consumers (fish). Of these three groups, the development of methods for the estimation of production rates has been least advanced for the primary consumers. In the present study an approach was made toward the development of a method by which the production rate of the primary consumers of a stream could be estimated. For some species, downstream drift appears to be the major agent of removal. The proposed method was to obtain a measurement of the net rate of removal from an area of stream bottom, which is equal to a minimal estimate of production rate.

Production rates (biomass produced per unit time regardless of whether it survives to the end of that time (Clarke, 1946)) for benthic invertebrates have been computed previously from growth rate and population density. Growth rates have been determined by periodic measurement or weighing of the individuals in a recognizable age-group (Borutsky, 1939; Anderson and Hooper, 1956; Teal, 1957) or by observations of caged organisms (Odum, 1957). The estimation of population density can usually be made by bottom sampling, but the determination of growth rate *in situ* of those species where the age of an individual cannot be recognized is a major obstacle. In a stream the high rate of downstream drift of certain species (Waters, 1962) complicates the measurement of growth rate because the downstream movement results in an apparent continuous dis-

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persal of populations through a variety of stream bottom types and other environmental conditions.

A different approach to the estimation of production rate of stream invertebrates was suggested by the apparent correlation between drift rate and productive capacity of streams (Waters, 1961). Furthermore, Needham (1938) has discussed the riffles as the areas of fish-food production, and the pools, which harbor fish and receive drifting foods from upstream, as the areas of consumption. The net contribution of a riffle to an adjoining pool downstream by means of drift should therefore be a measure of the productive capacity of that riffle. The approach was based on utilizing a given area of stream bottom, such as a riffle, as the basic unit of study rather than the organism's population. For such an area, the production rate (B) is equal to the algebraic sum of the rate of change in population density (P), and difference between the rate of removal (R) and the rate of accrual (A).

$$B = P + (R - A) \qquad (1)$$

The rate of change in population density is either positive or negative depending on whether it is increasing or decreasing. Removal from an area of stream bottom can be attributed to downstream drift, predation by aquatic carnivores, death and decomposition or scavenging on the area, and emergence of insects. Accrual is primarily in the form of drift from upstream.

Valley Creek, a small Minnesota stream, was appropriate for employing the above approach for several reasons: (1) Clearly distinguished riffles and pools were characteristic of the stream and available as study units. (2) Drift rates were high and easily measured. (3) Predation was low on the riffles because few fish and no carnivorous invertebrates inhabited the riffles. (4) The swift water of the riffles would probably cause dead, dying, or emerging organisms to be included in the drift.

METHODS

The study was conducted during July 1960 in Valley Creek, Washington County, Minnesota, a small trout brook which, compared to other trout streams in the state, is highly productive (Waters, 1961). The stream is characterized by clear, alkaline waters with 220 p.p.m. total alkalinity and a maximum summer temperature of about 18° C. Rubble and gravel riffles are the predominant bottom types, and a few pools occur among them. The dominant vegetation is watercress, *Nasturtium officinale*, and a benthic moss, *Fontinalis* sp. The principal invertebrates are the amphipod, *Gammarus limnaeus*; the mayfly, *Baetis vagans*; a snail, *Physa* sp.; and several caddisflies including *Glossosoma intermedium*, *Hydropsyche* sp., and several species of *Limnephilus*. Only the brook trout, *Salvelinus fontinalis*, and a sculpin, *Cottus* sp., comprise the fish species present.

Only the data for the mayfly, *Baetis vagans* McDunnough, are included in this report, since it was the only riffle species which was sufficiently abundant in the drift at this time of year.

Production rates were computed as equal to the change in population density, plus the difference between the drift off the unit area (riffle or pool) and the drift into the area, according to formula (1) modified as follows:

$$B = P + (D_0 - D_1) \qquad (2)$$

where D_0 = drift rate off the area, the principal form of removal, and
D_1 = drift rate into the area, or accrual.

Population densities were not obtained from the riffles on which the drift was measured, but, rather, from a similar riffle about 150 meters downstream. The estimates of production rate were minimal because removal due to predation, decomposition, and emergence was not measured. These losses were probably small. The drift sampling included the measurement, for two 24-hour periods, of drift rates at points above, between, and below two pools and two riffles. Negative values of B occurring in pools were termed "consumption rates." These too were minimal because the extent of production in the pools was unknown; however, it is probably safe to assume on the basis of this insect's known preference for swift-water riffles that production in the pools was negligible.

Two pools and two riffles were selected where the two types of habitat were contiguous and clearly distinct from each other (Fig-

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The relationship between primary production and production of profundal bottom invertebrates in a Danish eutrophic lake

PÉTUR M. JÓNASSON (Hillerød, Denmark)

With 3 figures and 1 table in the text

Introduction

The purpose of this lecture is to outline the ecological relationship between primary production (production of organic matter by the phytoplankton) and production of organic matter by the profundal bottom-fauna community and to provide a causal analysis of its timing relative to the general rhythm of the lake.

Lake Esrom, the second largest lake in Denmark, has proved admirably suited for this purpose since it is large, 17.3 sq. km, and consisting of a single regular basin without submerged banks or depressions of which 9 sq. km, or 52% of the total lake area, exceeds 15 m depth.

Large tributaries are lacking and the outflow never exceeds 1 m^3 per sec. This means that the body of water remains in the lake basin for a long time, 7.5 years on an average which, of course, is an advantage for the present purpose, since it greatly stabilizes the primary production.

Physiographical information of interest for primary and secondary production in Lake Esrom is mentioned in BERG (1938), JÓNASSON and MATHIESEN (1959) and JÓNASSON (1961).

Primary production

The production of organic matter by the phytoplankton was determined by means of the carbon-14 technique. The particular method used in Lake Esrom is described by JÓNASSON and MATHIESEN (1959). The rates of gross production per unit surface were calculated from the surface to the depth where compensation occurs on the basis of the curves of vertical distribution.

The determinations were carried out weekly, bimonthly or monthly during 1955—1962 and are all given in Fig. 1 together with a few preliminary measurements from 1954.

The figure indicates that the primary production during winter — especially in December and January — is quite negligible. Thus for December the maximum amounts to 0.16 g C/m^2 (8. XII. 1955) and the minimum, the lowest value found in the lake so far, to 0.03 g C/m^2 (11. XII. 1956). The low light-intensities during

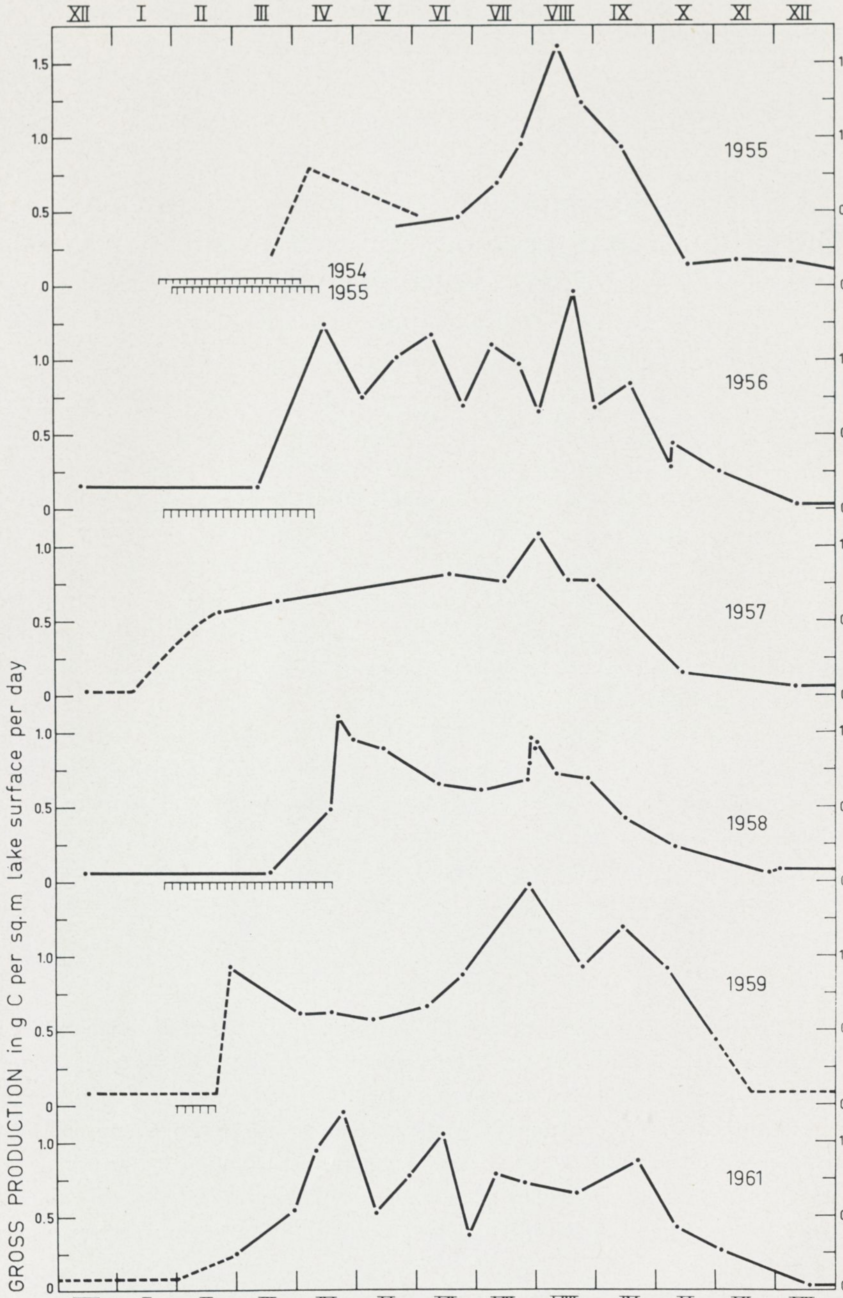

Fig. 1. Primary production of the phytoplankton during the period 1954—1962 expressed as g C per sq. m lake surface per day. Each dot represents one observation which again consists of 4—8 *in situ* experiments made at various depths on the same day (cf. Jónasson and Mathiesen 1959, Fig. 6).

this period are often further reduced by ice and snow. The lake may be ice-covered from January to April. During a winter with ice the amount of production has already considerably increased before the ice disappears. Immediately after the ice breaks up an additional increase in production leads to the spring maximum. Ice periods are indicated in the scheme by solid lines at the abscissa axis each year (Fig. 1).

The measurements in the spring of 1958 illustrate well this enormous increase:

1957	December 10	0.06 g C/m² per day	
1958	March 18	0.06 g C/m² per day	
1958	April 18	0.48 g C/m² per day	(last day of the ice-period)
1958	April 22	1.11 g C/m² per day	
1958	April 29	0.95 g C/m² per day	

A corresponding rapid increase in the spring maximum was observed in 1956 (1959) and 1961.

An entirely different type of seasonal succession occurs when ice is absent, which happened during winter 1956—57. Under such circumstances the production increases gradually:

1956	December 11	0.03 g C/m² per day
1957	February 23	0.56 g C/m² per day
1957	March 22	0.63 g C/m² per day

The production seems to continue without any spring maximum.

In 1959 the lake was ice-bound only during the short period 31. I.—20. II. A spring maximum is recorded on 28 February, while the succeeding experiments give values which are in good agreement with those of the spring 1957:

1958	December 12	0.08 g C/m² per day
1959	February 28	0.93 g C/m² per day
1959	April 2	0.61 g C/m² per day
1959	April 18	0.62 g C/m² per day
1959	May 9	0.57 g C/m² per day
1959	June 6	0.66 g C/m² per day

The primary production during the summer period varies considerably between years (Fig. 1). However, it is obvious that reduced production succeeds the spring maximum. Also the figure indicates that there is a definite summer maximum every year within the period 30 July to 20 August. The absolute size of this maximum ranges from 0.96 to 1.61 g C/m^2 in 1958 and 1955 respectively. An important feature of the curves is the rapid decline in production immediately following the summer maximum and continuing during autumn until the minimum is reached in December.

An estimation of seasonal succession shows the striking fact of irregular curves in 1956 and 1961 as compared with the curves for the years 1955, 1957, 1958 and 1959. A satisfactory and more detailed interpretation of the variation would require measurement of several environmental factors affecting primary production. However, it is reasonable to conclude that the constant weather conditions and insolation have caused the regular shape of the curve and the marked summer maximum during 1955 and 1959. A corresponding regularity in seasonal trend was found in 1958, only with a much reduced summer maximum. It is reasonable to suppose that the fluctuations in the summers of 1956 and 1961 reflect the stormy weather during these summers which caused horizontal displacements of plankton algae by wind pressure. These years no stratification was present in the lake in late August, while usually the thermocline persists until late October or November.

The measurements in Lake Esrom seem to give support to the following general conclusions as regards the seasonal succession: When the lake has been ice-covered there is a well-defined maximum in April—May. If the lake is ice-free during winter the primary production in spring extends over a longer period without a distinct maximum, light being the limiting factor during the early part of the period. The production is low in June prior to the July increase. The increase continues until the maximum is reached in August. Then a rapid decline commences, lasting until the minimum is reached in winter.

In Lake Esrom the gross production per year was found to be:

In 1955:	180 g C per sq. m lake surface
In 1956:	205 g C per sq. m lake surface
In 1957:	180 g C per sq. m lake surface
In 1958:	140 g C per sq. m lake surface
In 1959:	225 g C per sq. m lake surface
In 1961:	175 g C per sq. m lake surface

The mean annual gross production of Lake Esrom is estimated to be 185 g C/m^2. This corresponds to about 450 g organic matter. The net production, estimated at 75% of the gross production, is, therefore, 135 g C/m^2 corresponding to about 340 g organic matter.

Secondary production

Primary production of phytoplankton of the above-mentioned order causes oxygen lack in the hypolimnion during summer. In Jónasson 1961, Fig. 2, was shown a transverse section of the western part of the lake and the influence of oxygen and temperature stratification on life-conditions in the lake basin.

Low temperature implies low metabolism and slow growth and development. The influence of decreasing oxygen content of the water on the respiration of profundal lake species has been described in a previous paper (Berg, Jónasson and Ockelmann 1962).

A fundamental metabolic adaptation in the profundal fauna of Lake Esrom is that a relatively high respiratory rate is maintained even at a very low oxygen content of the water. This is particularly so in *Chironomus anthracinus* Zett., the quantitatively most important species. The table, which comprises all the species concerned, illustrates this well.

Table 1. The oxygen consumption of profundal lake species measured in μl/hr/g at 11°.

Species	Oxygen content of water (% of gas mixture) 1%	19%	Remarks
Tubifex barbatus	1.9	23.1	sublittoral
Ilyodrilus hammoniensis	3.3	13.1	profundal
Corethra flavicans		27.3	profundal
Chironomus anthracinus	18.0	29.9	profundal
Chironomus anthracinus		51.6	sublittoral
Procladius pectinatus	7.4	52.7	profundal
Pisidium casertanum	1.8	16.8	profundal

The highest respiration at 19% is found in *Tubifex barbatus* (sublittoral), *C. anthracinus* (both profundal and sublittoral) and *Procladius pectinatus*. At 1% *C. anthracinus* is the only one to maintain a markedly high rate of respiration, viz. 18.0 µl/hr/g.

How does this best-adapted species *Chironomus anthracinus*, withstand the life-conditions during summer stagnation period?

C. anthracinus, which feeds on phytoplankton, requires 1 or 2 years for its development (Jónasson 1961, Fig. 6). The eggs hatch in May—June. Fig. 2 shows how the 2nd instar increases in wet weight during the summer stagnation period. The same is the case in wet weight for the 3rd instar. On the contrary dry weight is constant in both instars during the summer stagnation period. Increase in water content simply reflects starvation i. e. in spite of the relatively high rate of respiration at 1% oxygen the larvae are not able to utilize the phytoplankton summer maximum as food.

After the autumn overturn on 29 September temperature and oxygen content increases. Within 10 days the 2nd and 3rd instar larvae have metamorphosed and 80% of the population now consists of 4th instar larvae.

Wet weight increases are 255 and 750% in 2nd and 3rd instars respectively
dry weight increases are 210 and 605% in 2nd and 3rd instars respectively

During the period 10—25 October the larvae increase their
wet weight by 75—160% and their
dry weight by 220—285%

After this outburst growth stops on account of lack of food since the production of phytoplankton declines rapidly in October. This situation continues during winter but by 1 March an increase in weight can be demonstrated due to the early phytoplankton-production in an icefree lake. Final growth takes place in late April and early May at the same time as the spring maximum of phytoplankton. The larvae increase their weight to 13.14 mg and emerge. The larvae not emerging weigh on an average 10.36 mg. These larvae increase their weight during June—July to 13.10 mg. During September—November the larvae lose weight. At the same time their water content increases as in the previous year. In April—May (1962) the larvae again increase their weight to 13 mg.

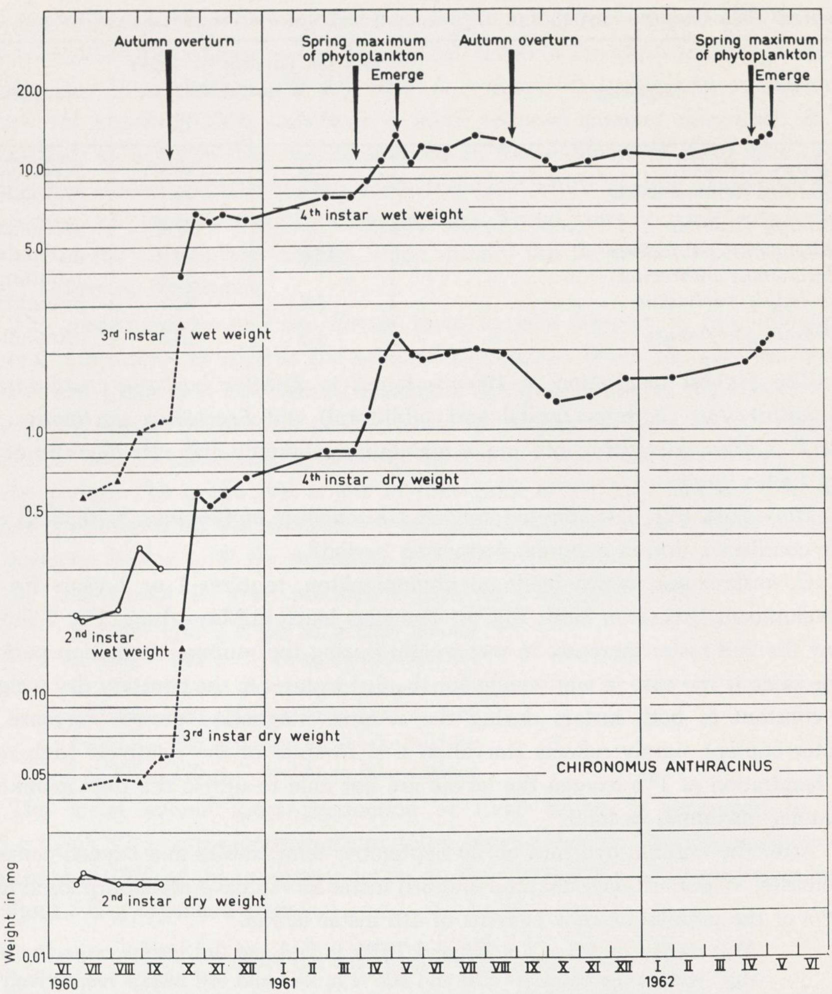

Fig. 2. Variation in wet and dry weight in the larvae of *C. anthracinus* (syn. *C. bathophilus*) through a two-year period in relation to temperature and oxygen (summer stagnation period and autumn overturn) and primary production of the phytoplankton (spring and summer maximum, autumn decline).

C. anthracinus has 4 larval instars, and the weights of the 2nd, 3rd and 4th instars are shown in the figure. Many larvae have a two-year life cycle (23 months) but part of the population has a one-year life cycle (11 months, cf. Jónasson 1961, Fig. 6). Growth takes place in limited periods mainly just after autumn overturn of the first year and during spring maximum of phytoplankton prior to the emergence in May. The second-year-larvae increase their weight in June, lose weight after fall overturn, and regain their weight during spring maximum.

The diagram is based on weighings of 8000 larvae.

The growth of *Chironomus anthracinus* in the profundal zone is thus limited to very short periods. Growth takes place in June, in October after autumn overturn, and during the spring maximum of phytoplankton. It is therefore clear that the ecological importance of primary production for the bottom fauna community lies not only in the size of the annual gross production but also very much in the seasonal distribution of annual gross production.

Acknowledgement

The dry- and wet-weight determinations underlying Fig. 2 were carried out in the research department of Niels Steensens Hospital, Copenhagen. All assistance received from K. Brunfeldt, M. Sc., and his staff is gratefully acknowledged.

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Discussion

Henson, E. B.: Do you attribute the decrease in weight of the individual larvae to lack of oxygen slowing metabolism? In Cayuga Lake, which has adequate oxygen, I have also noted a decrease in larval weight preceding pupation.

Jónasson: The constant dry weight in 2nd and 3rd larval instar during summer stagnation period 1960 (cf. Fig. 2) shows, obviously, that the larvae are not able to increase their dry weight under these life conditions. This observation was made several times in previous years. During the second year the larval weight is again constant through the summer stagnation period in July—August, but after the autumn circulation this second year the weight decreases markedly — on an average from 1.93 mg to 1.37 mg dry weight (12—13 mg to approximately 10 mg wet weight) — a loss of about 30%. A likely explanation is that just after autumn overturn there is a high metabolism on account of high temperature and water rich in oxygen, and the production of fresh food decreases rapidly (cf. Fig. 1).

Prior to pupation I find increase in larval weight. The average weight of larvae just before pupation is higher than the average weight of pupae.

Larkin: What is the average proportion of *Chironomus anthracinus* which emerge at the end of the first year?

Jónasson: The average proportion emerging at the end of the first year varies between years, but for the years 1955, 1957, 1959, and 1961 the average percentage is 23—38. Between sampling stations in the lake within the same year there is also a difference ranging from 9—47%.

Mann, K. H.: Can the profundal animals maintain a constant level of metabolism at oxygen concentrations intermediate to those given in the paper?

Jónasson: In *Tubifex barbatus* and *Ilyodrilus hammoniensis* the oxygen consumption decreases a little with decreasing oxygen content of the water until the critical point, at 4 and 2% O_2, respectively, is reached (Fig. 3). In both cases the oxygen consumption decreases by only 25% when we move from oxygen saturated water to the critical point. Below the critical point the oxygen con-

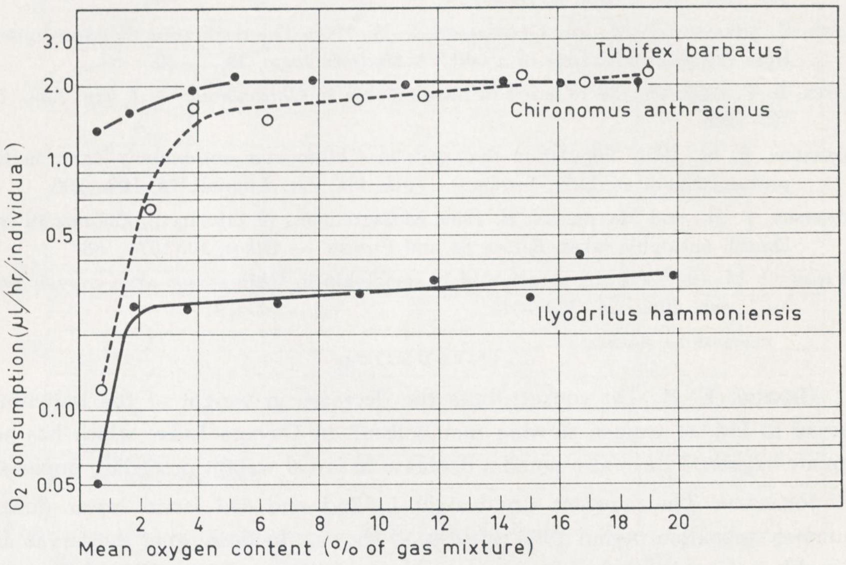

Fig. 3. The influence of decreasing oxygen content of the water on the respiration of three profundal lake-species.

sumption drops steeply. In *C. anthracinus* the respiration can be kept constant throughout the interval from air-saturation to 5% oxygen. The critical point of *C. anthracinus* is not very distinct because the oxygen consumption is still considerable below the critical point, about 70% of the rate at air-saturation.

Marlier: Do you know how many weeks or months *Chironomus anthracinus* can stand oxygen lack?

Jónasson: The longest period I have observed the larvae to survive in Lake Esrom is 3½ months. During this period the oxygen content in the water layers near the bottom was always less than 1 ml/l, i. e. the oxygen content at the mud surface must be zero or very close to zero (oxygen saturation from 14 to 0).

MORGAN: The ability of *C. anthracinus* to survive at a very low oxygen concentration can enable it to survive during exceptional conditions in the sublittoral. In a Scottish lake which I studied the *Myriophyllum spicatum*, which normally remains green, decayed right down during one winter. The following spring *C. anthracinus* emerged in exceptionally high numbers, but none of the other species emerging increased in numbers. This suggests that *C. anthracinus* was the only species able to take advantage of the decomposing vegetable matter at the low oxygen concentration which probably existed.

JÓNASSON: This is a very interesting observation on the habitat of *C. anthracinus* as a contrast to the habitat in Lake Esrom. Here the highest numbers of larvae are found in the finest bottom deposits in mid lake, while the coarser deposits in the sublittoral near the shore are inhabited by fewer individuals.

Experimental studies with *C. plumosus* (EWER 1942, WALSH 1947) show that these larvae are able to survive even after inhibition of the respiratory pigment by carbon monoxide for 16 hours. *C. anthracinus* is less well-adapted.

REYNOLDSON: Mr. JÓNASSON has shown that *Chironomus anthracinus* feeds well and grows only at short periods over the year despite its respiratory adaptation. Are the other less well-adapted organisms similarly restricted?

JÓNASSON: Quantitatively, *C. anthracinus* is the most important and, therefore, the best-investigated species. The other species e. g. *Corethra (Chaoborus) flavicans* do not occur in the profundal during the summer stagnation period. The same applies to *Procladius pectinatus*. The growth of *Corethra* takes place within two limited periods: the first during the pelagic stage, in August—September, the second in June, prior to pupation, in their mud-dwelling and pelagic stage. *Procladius* occurs as small larvae in the profundal after autumn overturn, and they emerge in spring or early summer.

Two constant mud-dwelling and quantitatively-important species, *Ilyodrilus hammoniensis* and *Pisidium casertanum*, are likely to show dependence on the same external factors as *C. anthracinus*. However, it is difficult to demonstrate this convincingly owing to methodological and biological problems.

[1970]

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FEDERAL AID IN FISH AND WILDLIFE RESTORATION

ANNUAL AND JOB COMPLETION REPORT

Project F-53-R-5

LAKE AND RESERVOIR INVESTIGATIONS

Job 6. Introduction of Opossum Shrimp into Idaho Lakes

Period Covered: March 1, 1969 to February 28, 1970

(Covers Job 6 of F-53-R-5 and Final Completion Report for Mysis Shrimp Transplants 1965 - 1969)

By

John T. Heimer
Fishery Research Biologist
January, 1970

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ANNUAL AND JOB COMPLETION REPORT

RESEARCH PROJECT SEGMENT

State of Idaho

Project No. F-53-R-5

Job No. 6

Name: LAKE AND RESERVOIR INVESTIGATIONS

Title: Introduction of Opossum Shrimp into Idaho Lakes

Period Covered: March 1, 1969 to February 28, 1970 (Final Report)

ABSTRACT:

Since 1965 approximately 4,000,000 opossum shrimp Mysis relicta have been collected at Waterton Lake, Alberta, or Kootenay Lake, British Columbia and stocked in fourteen Idaho lakes or reservoirs. The shrimp were shipped in lots of 10,000 each in plastic bags containing water and oxygen. The iced bags in styrofoam coolers were transported by truck or airplane.

In September, 1969, between 200 and 300 shrimp were captured in Cavanaugh Bay at Priest Lake. It appears that shrimp have become established in this lake as they had not been planted in this area since 1967 and were not found during seine hauls in previous years.

In September, 1969 one shrimp was recovered from Pend Oreille Lake (Idlewild Bay). It was impossible to determine whether this shrimp was from recent transplants in the area or reproduction in the lake. Shrimp have not been found in other lakes to date.

Submitted by:

John T. Heimer
Fishery Research Biologist

RECOMMENDATIONS:

Collect between 500,000 and 1,000,000 opossum shrimp *Mysis relicta* annually and introduce them into selected Idaho waters. Individual lakes should be stocked three years in succession.

Investigate handling and shipping techniques which influence the survival and propogation of transplanted *Mysis*.

Make test hauls in Idaho lakes using techniques that are successful in Canada.

TECHNIQUES USED:

In 1965 and 1966 opossum shrimp were collected at Waterton Lake, Alberta, Canada. Collections were made during the day with an outboard powered boat rigged with a wooden platform and a chain saw winch to facilitate retrieval of the trawling net from deep water. The trawling net was constructed of nylon bobbinetting with an opening of 22 by 58 inches. A sled apparatus attached to the net frame kept the net slightly above the lake bottom to reduce the amount of debris collected in the net. The net was towed for 10 to 15 minute intervals. Captured shrimp were placed in milk cans in the boat and periodically transferred to a local fish hatchery until enough had been collected for shipment.

From 1967 through 1969, *Mysis* were collected at Kootenay Lake, British Columbia, Canada. Collections were made at night by towing a trawl net at mid-water depths. The collected shrimp were kept in 30-gallon plastic cans in the boat from one to two hours (depending upon quantities collected) and then transferred to portable hatchery troughs with running water. Some mortality was noted while holding the shrimp before transit. This mortality may have been due to handling and/or the method of capture used.

Shrimp were transported in 10-quart plastic milk sacks containing about five quarts of water and 10,000 (7.5 ounces of shrimp). The remainder of the sack was inflated with oxygen. Two sacks and approximately three pounds of ice were placed in each styrofoam cooler transported by airplane to south Idaho lakes. Ice often was omitted from coolers trucked to north Idaho lakes.

FINDINGS:

From 1965 to 1969 shrimp were released in twelve natural lakes and two reservoirs in Idaho (Table I). Most of the shrimp were planted in areas where the water was 100 feet deep or more. Time in transit varied from 3 to 25 hours. Temperature change of water in the plastic sacks during transit varied from a 5°F. increase to a 10°F. decrease (where ice was added). Generally survival during transit was good. However, greater mortalities were noted in some sacks than others. On occasions when mortality was high, it is possible that a large percentage of dead or injured shrimp were loaded into the sacks as identification of dead shrimp is difficult during loading.

Overnight live box tests at various lakes indicated survival of fifty percent or greater. However, some of the shrimp used for the live box test may have been dead or injured.

Since 1965 annual test hauls for Mysis shrimp have been made at Priest Lake. In September, 1969, shrimp were captured at Priest Lake (Cavanaugh Bay), the first indication of shrimp survival and population establishment in Idaho. Between 200 and 300 shrimp were captured in about 30 minutes trawl time. Two distinct sizes of shrimp were captured, indicating two age classes.

In September, 1969, one shrimp was recovered from Pend Oreille Lake (Idlewild Bay). It was impossible to determine if this shrimp was from recent transplants in the area or reproduction in the lake.

Several years may elapse before transplanted shrimp become established in any given lake; however, the netting of significant numbers at Priest Lake only five years after the initial stocking is encouraging.

Inasmuch as the initial experimental period is over and methods have been fairly well perfected, we will continue the transplantations with State funding. Lakes stocked with shrimp will be monitored periodically with nets and by checking fish stomachs to note shrimp survival and density.

Table I. Number of opossum shrimp *Mysis relicta* planted in Idaho lakes and reservoirs, 1965 - 1969.

Water	Number of shrimp planted and year				
	1965	1966	1967	1968	1969
Priest Lake	295,000	400,000	300,000		
Upper Priest Lake				100,000	100,000
Coeur d'Alene Lake				200,000	200,000
Pend Oreille Lake		50,000	210,000	300,000	300,000
Payette Lake		100,000	100,000	94,500	
Anderson Ranch Reservoir	30,000	100,000	100,000		
Redfish Lake		50,000	60,000	52,500	
Alturas Lake		50,000	60,000	42,000	
Stanley Lake		50,000	40,000	31,500	
Pettit Lake			40,000	31,500	
Palisades Reservoir				115,500	210,000
Palisades Lake					40,000
Warm Lake					75,000
Upper Payette Lake					75,000
Totals	325,000	800,000	910,000	967,500	1,000,000

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Effects of a Pool on Stream Invertebrate Drift

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Abstract: Invertebrate drift entering and leaving a pool in Club Stream, Otsego Co., Michigan, was studied between June and September 1971. Drift rates out of the pool were greater than drift rates into the pool for most taxa on all dates. Increase in drift rate between the upstream and downstream ends of the pool (expressed as percent of number entering the pool) averaged 86.4% (range 3.3 - 707.7%) and decreases in drift rate averaged 24.4% (range 1.2 - 72.1%). The source of drifting organisms was a rheophilic fauna living on logs and twigs along the margin of the pool. Patterns of drift periodicity of most taxa were similar at the two sampling sites, but nocturnal peaks of drift rate of Ephemeroptera and *Gammarus* occurred earlier in the evening at the downstream site. Estimates of deposition rate suggested that the total number of organisms deposited in the pool per day was greater than the numbers drifting into or out of the pool. Fish predation on drift probably was negligible, because the number of trout in the pool was small and the fish present were large.

Introduction

High invertebrate drift rates reported for many rivers (reviewed by Waters, 1972) have raised questions regarding the ability of benthos to withstand sustained high rates of attrition (Waters, 1965) and the effect of drift on benthos dynamics (Ulfstrand, 1968). Interpretation of the impact of drift on benthos dynamics depends on knowledge of the fate of drift (Ulfstrand, 1968). Although pools have been suggested as major sites of drift consumption (Waters, 1962), little is known about the fate of drift in pools. Waters (1962) reported large decreases in drift rate in pools and attributed these losses to fish predation and to deposition with subsequent mortality and decomposition. Elliott (1967), however, reported large numbers of organisms drifting through pools, and his attempts to collect depositing organisms on trays were unsuccessful.

The objectives of this study were to: (1) determine if pools act as sites of drift consumption; (2) estimate deposition rates in pools, and (3) assess fish predation on drift. This study was carried out between June and September 1971.

Description of Study Area

Club Stream originates ca. 1 km SE of Vanderbilt, Otsego Co., Mich., and flows NE approximately 17 km to its confluence with the Sturgeon River. The pool studied (Fig. 1) was 6 km upstream from the mouth of Club Stream and was located in a sharp bend which is characteristic of pools in this stream. The pool was bounded upstream and downstream by riffles with modal water depths at base flow of 26

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were emptied at 2-hr intervals, thereafter at 8-hr intervals. A plank was suspended between the banks and iron pipes driven into the bottom at midstream so that it was possible to service the upstream drift net without disturbing the stream bottom (Fig. 1).

The volume of water filtered by drift nets differed between sampling sites and among dates. Drift catches were converted to drift densities (Waters, 1969a), and the product of drift density and total daily discharge was used to estimate total daily drift entering and leaving the pool. Drift densities as numbers/5000 m^3 of water were used to compare periodicity of drift entering and leaving the pool.

Deposited organisms were collected in metal trays (15 x 15 x 8 cm), filled to a depth of 3 cm with 1-2 cm diam gravel collected from the stream and rinsed with boiling water to kill the macrofauna. Ten trays were placed on the pool bottom for 24 hr on sampling dates. Five trays were equipped with covers made of 3/8 inch (0.95 cm) hardware cloth to prevent predation by fish or large invertebrates. A sponge-lined metal lid was pressed and held over trays to minimize loss of organisms during recovery.

Settling of drifting organisms between the upstream and downstream ends of the pool was studied by suspending nets from the top and bottom of iron rods driven into the bottom at Points B and C, Figure 1. Conical drift nets (471 μ mesh), 17 cm diam at the opening and 1 m long, were used. Top nets were positioned 5 cm below the surface; bottom nets were positioned 5 cm above the bottom. Samples were collected between 2100-2400 hr EST.

Fish populations were sampled biweekly between 15 June and 1 September using a portable electro-fishing unit. Three passes were made through the pool and the species and standard lengths recorded.

RESULTS

Drift rates out of the pool were greater than drift rates into the pool for most taxa on all dates (Table 2). Only *Ephemerella* and *Simulium* exhibited higher drift rates downstream on all dates. Other taxa exhibited decreased drift rates downstream on at least one date. On a taxon-by-taxon, date-by-date basis, increases in drift rate between the upstream and downstream ends of the pool averaged 85.4% (range 3.3-707.7%) and decreases in drift rate averaged 24.4% (range 1.2-72.1%). There was no relationship between the magnitude of change in drift rate and discharge or water temperature. Decreases in drift rate of three taxa and total fauna on 7-8 August, however, were coincident with the lowest discharge recorded during the study.

Qualitative observations indicated a substantial fauna of rheophilic insects, predominantly *Baetis, Ephemerella, Hydropsyche* and *Simulium,* present on logs and twigs along the pool margin. Insect abundance on these substrates increased between May and mid-June, decreased in July and increased again in September and October. Unfortunately, quantitative analysis of the pool fauna was not attempted.

and 32 cm, respectively. Pool surface area was 136.5 m^2, maximum depth was 122 cm and modal depth was 68 cm. Pool substrates were primarily fine gravel, sand and silt. Distribution and abundance of substrate types were variable between dates owing to discharge instability. Discharge rates and maximum-minimum water temperatures on each sampling date are given in Table 1.

METHODS

Drift was sampled upstream and downstream from the pool at points where changes in bottom slope marked the transition between riffle and pool (Points A and D, Fig. 1). Drift nets (471 μ mesh) 30 cm sq at the opening and 1.2 m long were constructed with straight sides and a boxed end. This design allowed debris to collect at the back of the net, thereby permitting sampling for periods up to 10 hr before clogging and backwashing occurred. In the stream, drift nets were attached to iron rods driven into the bottom. Since water depth at the upstream sampling site was always less than 30 cm, the drift net rested on bottom; at the downstream sampling site where minimum water depth was 35 cm, the net was kept 2 cm below the water surface and never rested on bottom. In the first two drift collections, nets

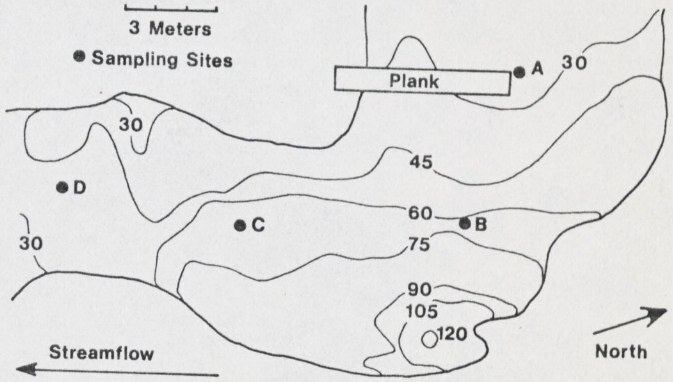

Fig. 1.—Contour map of the pool showing location of sampling sites. Numbers indicate depth in cm

TABLE 1.—Discharge and water temperature in Club Stream on the six sampling dates

Date	Discharge x10^3 m^3/day	Maximum-minimum water temperature (C)
June 17-18	71.2	14-17
July 7-8	71.1	19-21
July 19-20	71.6	20-23
July 30-31	69.7	20-22
August 7-8	65.9	16-20
August 23-24	67.0	12-15

highest when there were large (percent) increases in drift rate downstream. The number deposited per unit area was small; but if the area of the pool bottom is assumed to be the same as the area of the pool surface (136.5 m^2), the number deposited per day was substantial. In all cases these estimates of total deposition rate greatly exceeded the numbers entering or leaving the pool.

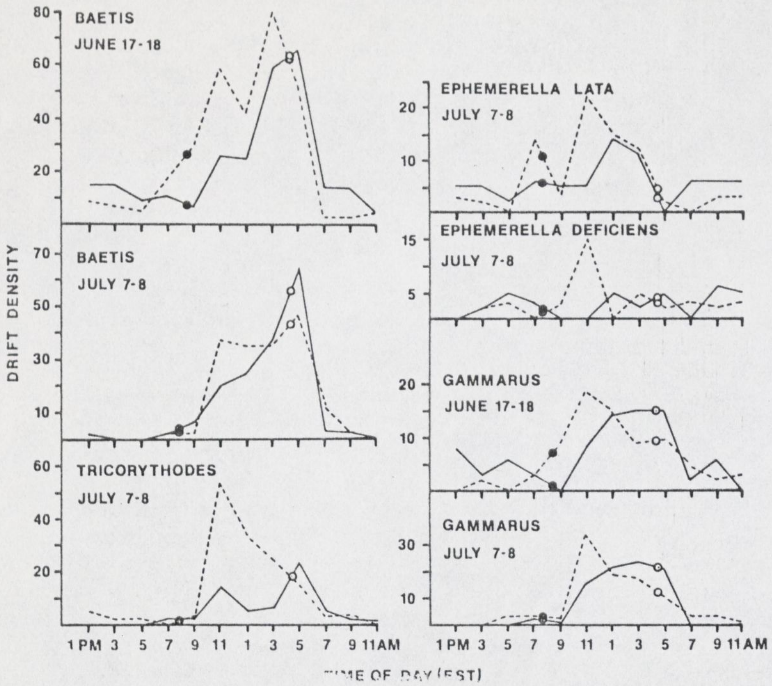

Fig. 2.—Comparison of the diel periodicity of drift of Ephemeroptera and *Gammarus* entering (——) and leaving (........) the pool (● denotes sunset, o denotes sunrise)

TABLE 3.—Estimated numbers of organisms deposited on the pool bottom per 24 hr. A comparison of numbers deposited in covered and uncovered trays (*see* text) on 7-8 July is also given

Taxon	Density (No./m²)					
	July 7-8			July 19-20	July 30-31	August 7-8
	Covered	Uncovered	Mean			
Baetis	53	62	58	44	75	31
Tricorythodes	53	44	49	22	120	31
Ephemerella	18	9	14	62	18	9
Lepidostoma	36	27	32	71	4	9
Simulium	62	116	89	142	13	13
Gammarus	44	98	71	13	173	67
Total Fauna	338	418	378	408	506	204

The diel periodicity of drift entering and leaving the pool was compared using data collected 17-18 June and 7-8 July. For most taxa, the periodicity of drift entering and leaving the pool was similar. For *Gammarus* and taxa of Ephemeroptera, however, nocturnal peaks of drift density occurred earlier in the evening at the downstream site (Fig. 2).

Deposition rates of the most abundant taxa are given in Table 3, as is a comparison of numbers deposited in covered and uncovered trays on 7-8 July. There were no significant differences between numbers in covered and uncovered trays (t-test), suggesting that predation on deposited organisms by either fish or invertebrates was not important. Deposition rates did not appear to be related to absolute drift rate, discharge or water temperature. Deposition rates were

TABLE 2.—Total daily drift rates of the most common taxa and total fauna into and out of the pool. Net change in drift rate expressed as percentage of the number entering the pool given in parentheses

Taxon	June 17-18		July 7-8		July 19-20	
	In	Out	In	Out	In	Out
Baetis	3745	4471	2661	2503	2334	2305
	(19.4)		(10.7)		(−1.2)	
Tricorythodes	....	....	1024	2090	1675	2406
			(104.1)		(43.6)	
Ephemerella	2036	2905	853	1223	544	1031
	(42.7)		(43.4)		(89.5)	
Lepidostoma	1809	3218	2133	2233	2076	3251
	(77.9)		(4.7)		(56.6)	
Hydropsyche	1225	983	498	484	358	687
	(−19.8)		(−2.8)		(91.9)	
Simulium	1282	1324	4835	6172	3136	13074
	(3.3)		(27.7)		(316.9)	
Gammarus	1139	1040	995	1294	1446	1332
	(−8.7)		(30.1)		(−7.9)	
Total Fauna	18938	23439	14519	23818	13805	26349
	(23.8)		(64.1)		(90.9)	

TABLE 2.—(continued)

Taxon	July 30-31		August 7-8		August 23-24	
	In	Out	In	Out	In	Out
Baetis	990	2133	870	461	17098	19698
	(115.5)		(−47.0)		(15.2)	
Tricorythodes	6859	11988	1476	1107	2358	3243
	(74.8)		(−25.0)		(37.5)	
Ephemerella	265	558	132	264	121	214
	(110.6)		(100.0)		(76.9)	
Lepidostoma	404	237	13	105	1152	938
	(−41.3)		(707.7)		(−18.6)	
Hydropsyche	4210	5353	896	1213	2278	3094
	(27.1)		(35.4)		(35.8)	
Simulium	1129	2286	1186	1358	509	844
	(102.5)		(14.5)		(65.8)	
Gammarus	1478	3053	659	184	831	1581
	(106.6)		(−72.1)		(90.3)	
Total Fauna	17313	29427	6827	5786	26063	32173
	(70.0)		(−15.2)		(23.4)	

Attempts to study the settling of drifting organisms in the pool were unsuccessful because of the small numbers of naturally drifting organisms collected. In general, there were no large changes in the vertical distribution of drifting organisms between the sampling sites. In a few cases drift density at the surface increased between the sampling sites, and in only one case (*Simulium,* 9 August) did drift density at the bottom increase between the sampling sites.

A maximum of two brown trout (*Salmo trutta* L.) were collected in the pool on any date between 15 June and 1 September. Catch of fish did not increase when the pool was blocked upstream and downstream with wire mesh prior to sampling. Mean standard length of all trout collected was 41.2 cm (range 37.7-47.4). Small sample sizes precluded gut analysis. A very small number of sculpins (*Cottus* sp.) and dace (*Rhinichthys* sp.) were also collected.

DISCUSSION

The pool studied did not act as a site of drift consumption, but rather it served as a source of drifting organisms. It is usually assumed that few rheophilic organisms inhabit pools (Hynes, 1970). Although rheophilic insects may not inhabit fine sediments on pool bottoms, a substantial fauna was present on logs and twigs along pool margins. Current velocities along the outside margin of the pool often exceeded 50 cm/sec, making this habitat comparable to riffle habitat. Little is known about stream faunas living on woody substrates (*see* Marlier, 1954; Nilson and Larimore, 1973).

Percent increases in drift rate between the upstream and downstream ends of the pool were greater than corresponding changes in drift rate between the ends of riffles and runs. Increases in drift rate between the ends of riffles reported by Waters (1962) averaged 28.8% (range 11.4-53.8%), and the increase in drift rate between the ends of a run reported by Elliott (1967) was 15.3%.

One of the most important problems for rheophilic insects inhabiting pools is obtaining an adequate oxygen supply. Most aquatic insects depend on passive diffusion of oxygen through gills or the body surface to satisfy oxygen requirements (Hynes, 1970). Oxygen supply depends both on the oxygen content of the water and on the rate of water renewal at respiratory surfaces. The rate of renewal is a function of current velocity and/or turbulence. At low water temperatures when oxygen consumption rates of ectothermic insects are low and oxygen solubility is high, the slow currents in pools may be adequate to meet oxygen demands. As water temperature increases, oxygen consumption rates increase but oxygen solubility decreases. At this time the slow currents in pools may not provide an adequate renewal of oxygen supply resulting in respiratory stress.

It is hypothesized that respiratory stress accounts for changes in faunal abundance in the pool. The increase in faunal abundance on logs and twigs in June may have been due to their movements from the pool bottom to compensate for decreasing oxygen concentration

and increasing oxygen demand with increasing water temperature. Current velocities at the bottom of the pool rarely exceeded 20 cm/sec, whereas current velocities along the pool margin often exceeded 50 cm/sec. Williams and Hynes (1973) reported *Chimarra aterrima* migrated from the streambed in riffles to roots along the stream margin as water temperatures approached 28 C.

Further increases in water temperature in July and early August presumably caused respiratory stress which resulted in drift and subsequent diminution of the fauna living on logs and twigs. Photonegative organisms like Ephemeroptera normally spend daylight hours in cryptic microhabitats (Waters, 1972) where renewal of oxygen supplies largely depend on current velocity in adjacent areas. Madsen (1968) reported that oxygen supply to cryptic microhabitats was about 6% of that to current-exposed microhabitats. Oxygen supplies in cryptic microhabitats, therefore, may be inadequate to meet the oxygen requirements of the insects. At night oxygen supplies may become limiting because of increased oxygen consumption rates controlled by endogenous physiological rhythms (Zoladek and Kapoor, 1971; Ulanoski and McDiffett, 1972; P. Klotman, pers. comm.). Inadequate oxygen supplies result in respiratory stress which could cause insects to move to current-exposed microhabitats where there is more oxygen but where the probability of erosion and drifting is also greater.

Considerable effort has been spent characterizing and classifying patterns of drift periodicity (*e.g.,* Müller, 1966; Elliott, 1969; Waters, 1969b). Most invertebrates exhibit diel periodicities of drift with a nocturnal maximum (Waters, 1972), but there are seasonal, geographic and specific differences in the number and timing of nocturnal peaks. Results of this study suggest that some of this variability may be related to the current velocity in the habitat where drifters originate.

It is hypothesized that differences in timing of peak drift rates also are related to respiratory stress. In pools, where oxygen supply to cryptic microhabitats is low, increases in oxygen consumption rate after sunset should cause insects to move immediately to current-exposed habitats resulting in a peak of drift just after sunset. In riffles where oxygen supply to cryptic microhabitats is greater, oxygen may not become limiting until late in the night after an extended period of high oxygen consumption rates by the insects and of microbial community respiration. The result would be a delay of peak drift until late in the night. Not all taxa exhibited differences in drift periodicity at the two sites and this may reflect differences in respiratory physiology between taxa (Dodds and Hisaw, 1924).

Although deposition occurred in the pool, it apparently had little relationship to changes in drift rate. Highest deposition rates were not associated with decreases in drift rate but with the largest (percent) increases in drift rate. Furthermore, the estimated numbers deposited per day in the pool greatly exceeded the numbers drifting into or out of the pool. This latter point may be misleading because trays used to measure deposition rates were not distributed randomly on the pool bottom but were concentrated near the deepest part. If deposition rates were greater in this area, then estimates of the total numbers deposited in the pool each day are overestimates. Further, sediments used in the deposition trays were coarser than sediments on the pool bottom. Consequently, there may have been a higher rate of accrual of organisms in the trays than on the pool bottom.

Deposition rates may be more a measure of the activity of the fauna living in the pool than a measure of numbers deposited from upstream areas. To account for the large numbers deposited in the pool each day, it is suggested there was daily redistribution of the pool fauna between logs and twigs along the pool margin and the pool bottom. Insects presumably drifted from the logs and twigs and were deposited on the bottom at night and then subsequently returned to the pool margin. Drift densities at the top and bottom of the water column were comparable suggesting that organisms were being added to the drift continuously. Analysis of water movement in the pool using fluorescein dye suggested mixing did not account for similarity of drift densities at the surface and bottom because mixing was important only near the downstream end of the pool.

Predation by brown trout probably had little effect on drift rates, because the numbers in the pool were small and the fish present were large. The low fish population density was probably due to territorial behavior (Kalleberg, 1958) which kept other fish, particularly smaller ones, out of the pool. Metzlaar (1929) reported that large brown trout (> 38 cm) feed little on insects and small invertebrates. T. F. Waters (pers. comm.) indicated small brook trout [*Salvelinus fontinalis* (Mitchell)] were abundant in pools where he observed marked decreases in drift rate. Clemens (1928) and Ricker (1930) reported that small brook trout (< 30 cm) feed predominantly on larvae and nymphs of aquatic insects and small crustaceans.

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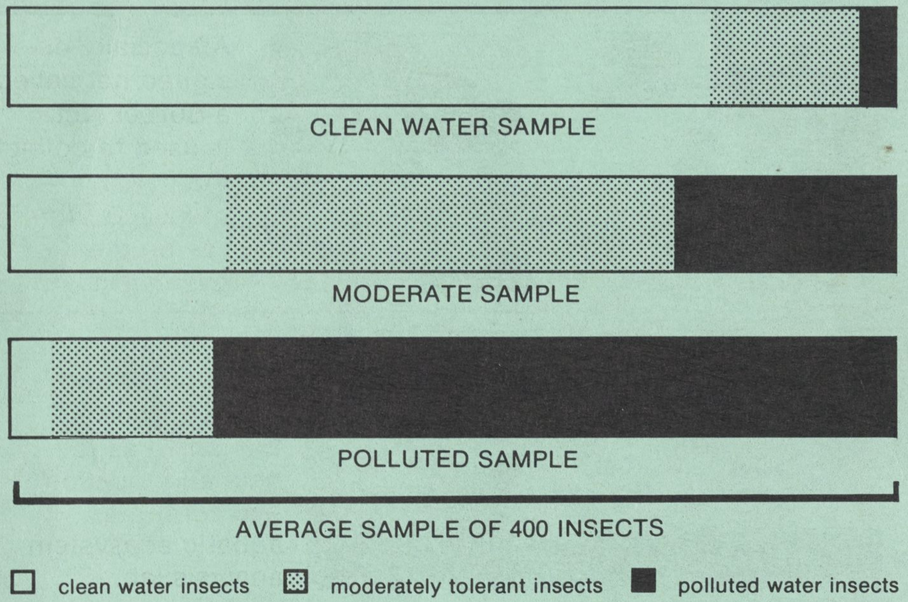
How it Works

An average sample will consist of about 400 insects, representing a cross-section of about 20 different varieties.

400 macroinvertebrates may seem like a healthy number of insects at first glance, but closer observation tells the real story.

THE KEY IS HOW MANY OF EACH KIND



From the results of the analysis, land managers can often tell what type of land use activities are occurring on a stream they have never seen.

For example, the count can show if abandoned mining operations could be releasing traces of heavy metals into the water. Managers can also tell if livestock are grazing along the stream banks and to what extent. The samples can predict whether or not seepage from recreation areas or campgrounds could present a problem to the watershed.

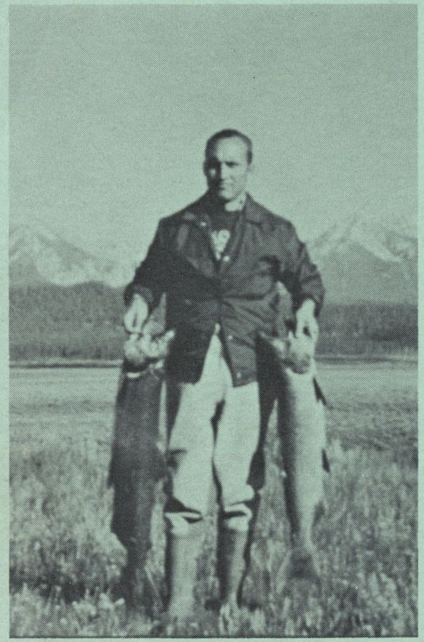
[1979]

Why Bother?

The macroinvertebrate analysis programs began in 1973. Before then, chemical and physical testing were the major sources of water quality information. Although accurate for many situations, chemical and physical analysis alone didn't give enough biological information to predict long-range effects or to monitor change over time.

Results of macroinvertebrate analysis can provide technical data to land managers that can be helpful in formulating management techniques and alternatives.

Some Uses:



Public Law now requires government agencies to know the quality of waters on lands they manage.

Data collected can be used to evaluate the effects of present and future forest uses.

The types of macroinvertebrates present can be helpful in evaluating a stream's fishery potential.

Data can be used to evaluate various management techniques such as those used to control the effects of cattle grazing upon the stream habitat.

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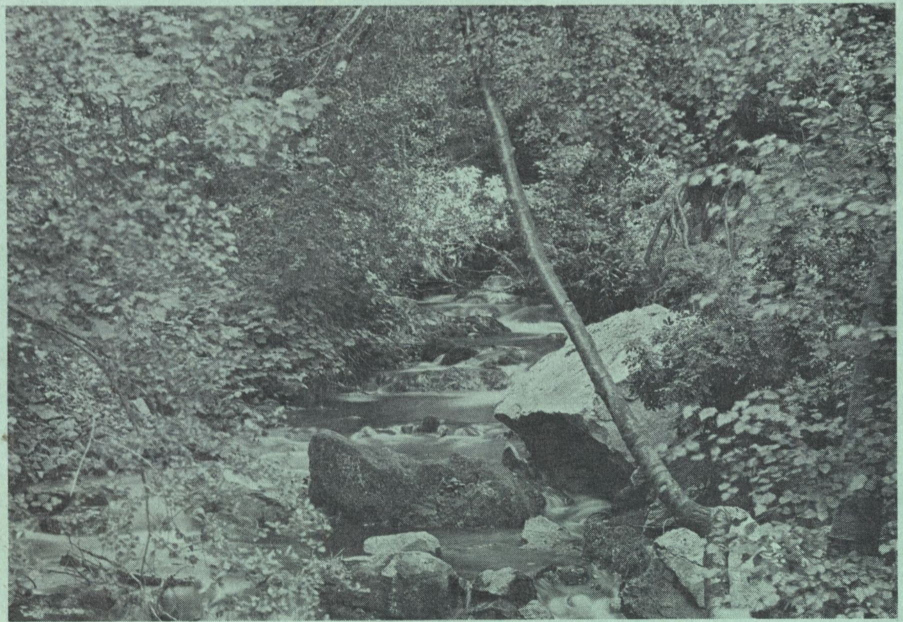
Macro What?

AQUATIC ECOSYSTEM ANALYSIS PROGRAM
MACROINVERTEBRATE PHASE

US DEPARTMENT OF AGRICULTURE
FOREST SERVICE
INTERMOUNTAIN REGION

NATIONAL FOREST SYSTEM · COOPERATIVE FORESTRY · FORESTRY RESEARCH · FOREST SERVICE · US · DEPARTMENT OF AGRICULTURE

Clean Water



As the use of our natural resources increases, the impact of activities such as:

- Road construction and maintenance
- Hunting, fishing, camping
- Timber harvesting
- Grazing needs
- Expanded energy explorations

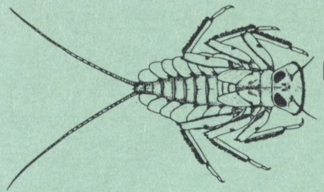
have placed an added interest on the water quality of mountain lakes and streams.

In order to measure the effect of these activities on waters leaving National Forest boundaries, a relatively new and reliable tool has been developed.

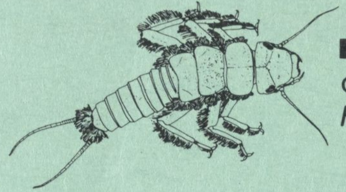
Aquatic Macroinvertebrate Analysis, or in everyday terms, the study of water insects, has shed new light on techniques for monitoring stream conditions.

Insects

THERE ARE THOSE INSECTS:



■ who thrive best in clean water, such as the Mayfly, *Epeorus sp.*



■ who flourish with a moderate amount of pollution present, such as the Stonefly, *Hesperoperla Sp.*

■ who live mainly in highly contaminated waters, such as the Blackfly, *Simuliidae.*

The majority of macroinvertebrates studied make their home in the rocks of stream beds. Preliminary studies have shown researchers that one of three types of insect will generally dominate a given community.

By taking controlled samples of these insects from their natural habitat, the Forest Service has been able to develop a clearer picture of a stream's general "Health" in terms of habitability and strength of the aquatic food chain.

Samples are collected spring, summer, and fall over a period of 3 to 5 years.

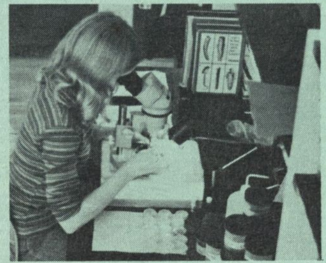
Nearly 200 sample sites are now located in Utah, Idaho, Nevada, Wyoming, and Montana.

How it's Done

Sample sites are selected from representative streams throughout the mountain regions.

A specially designed net called a Surber Net is used to collect the colony of insects who live on the river rocks.

The insect samples are strained and bottled. They are labled as to date and location and sent to an aquatic ecosystem analysis lab.



At the lab the samples are examined under a dissecting microscope to determine the number of each insect species. The samples are dried and weighed.

Careful records of counts and weights for each sample at each location are compiled.

Author Unknown
[ca. 1981]

THE AFFECTED AQUATIC ENVIRONMENT

The community of organisms native to the upper Colorado River mainstem is adapted to survive under a harsh and highly fluctuating regime of flows, turbidity, and temperature. Within the primary potential impact area between De Beque and the Colorado-Utah state line, the fauna has historically experienced annual temperature ranges from near freezing to in excess of 80°F (27°C). Since approximately 70% of the total annual flow in the Colorado occurs during spring snowmelt, peak flows are usually well over an order of magnitude greater than base flows. Turbidity has likewise varied greatly as a function of dilution.

Because of the harsh nature of this environment, the plant community in this section of river is somewhat limited. As in most riverine systems, bacteria, periphyton, and filamentous algae represent the start of the food base (Hynes, 1970). Here, this primary productivity appears to be limited by turbidity, the shifting nature of the substrate, and the lack of extensive shallow areas rather than nutrient availability.

Rooted aquatic plants tend to be relatively intolerant of harsh and unstable conditions. As a result, few are represented in this section of river. Those that are tend to be confined to shallow, quiet backwaters during the summer. A few emergent macrophytes such as cattails (<u>Typha</u> sp.) are also present sporadically along the shoreline.

External (allochthonous) inputs of organic matter appear to be minimal relative to the river volume in this region. Terrestrial vegetation is sparse and riparian trees, such as cottonwood, willow, and salt cedar, exist in only a narrow strip along the bank. The canyon nature of much of this section further shields the river from organic inputs. Terrestrial insects, however, have been implied to be a significant part of the diet of several stream fishes (CDOW, 1981).

Few data exist concerning the invertebrate fauna in the Colorado River between De Beque and the Colorado-Utah state line. The best available detailed information is in Burkhard and Lytle (1978). As in most riverine systems, invertebrate distribution tends to be more a function of substrate (microhabitat) rather than the generalized environment (Ames, 1977; Pennak, 1978). However, certain generalizations can be made. Most species present tend to be adapted to withstand high summer temperatures and high silt loads. For this reason, the stoneflies (Plecoptera) are poorly represented relative to sites further upstream. Ephemeroptera and Diptera are the most common macroinvertebrates present. Trichoptera and Coleoptera are also common. Megaloptera, Hemiptera, Odonata, and Lepidoptera are also represented as minor percentages of the total insect biomass (W. D. Fronk, personal communication). Crayfish (probably introduced) are common in this section of the river.

Because of the harsh environment posed by the Colorado River mainstem and the evolutionary isolation of the river, many of the native fishes are endemic to this system -- that is, found nowhere else, and with a long evolutionary history of adaptation to tolerate its wide extremes. Thus, the adult forms of most of the native fish do not appear to have been adversely affected by man induced flow reductions or increased turbidity and salinity (Behnke and Benson, 1980). Diversion of river water has resulted in a loss of approximately 25% of virgin flow which reduces spring peak flows and summer base flows (USGS, 1976). Grazing has increased turbidity and irrigation return flows have resulted in the doubling of salinity over historic levels (USGS, 1976).

The native fish fauna of the upper Colorado River basin is poor in species due to long isolation from surrounding drainages. Thirteen species are native (Table 1) to the upper basin. Of these, five are species specialized for life in the main river channels -- squawfish, humpback and bonytail chubs, razorback sucker, and flannelmouth sucker. Only the flannelmouth sucker continues to maintain its abundance. The other four species have drastically declined due to a changing environment and the impact of non-native fishes. For example, a reduction in flows during spring spawning reduces the quantity and quality of flooded back-waters important as nursery areas for many native young-of-the-year (Joseph and Sinning, 1977; CDOW, 1981). Temperature is often a factor controlling time of spawning, so modification of temper-ature regimes could result in egg deposition or hatching at inopportune times.

Non-native fishes have been introduced into the upper Colorado River for at least the last 100 years. Habitat alterations and a paucity of fish species have allowed many preadapted species to successfully become established. For a bibliography listing most important publications concerning historic and present population levels see USFWS Resource Publication 135 (Wydoski, et. al. 1980).

Examination of the specific habitats from which particular species were sampled (CDOW, 1981) reveals that channel catfish and large carp now dominate the deepwater, low velocity areas once solely utilized by squawfish and humpback chub. Sunfish, catfish, and medium size carp now dominate the larger backwaters and ponded areas important for young of the rare native fishes. Red and sand shiners are extremely abundant over shallow sand bottoms and may have resulted in the decline of the speckled dace by direct competition. The bluehead and to a lesser extent the flannelmouth suckers appear to occupy a very broad range of habitat types and have remained common despite introductions. Both the razorback sucker and the bonytail chub are believed to prefer canyon areas so some of their decline may be due to the reduction in this habitat type as well as heavy egg and juvenile predation by non-natives. Within the next two years, information will become available from both Fish and Wildlife Service and Colorado Division of Wildlife studies. These data will be permanently stored in a computer to allow correlation of species distribution, species association and several physical habitat parameters.

At present, no important recreational or commercial fishery exists on this portion of the river. Minor angling for channel catfish occurs, but relatively low concentrations do not attract large numbers of anglers to this inaccessible area.

TABLE 1

RESIDENT FISHES OF THE COLORADO RIVER
(De Beque to the Colorado/Utah State Line)

COMMON NAME	SCIENTIFIC NAME	STATUS	SOURCE	CURRENT ABUNDANCE	HISTORIC ABUNDANCE
CATOSTOMIDAE -					
White Sucker	Catostomus commersoni	-	I	common	-
Bluehead Sucker	Catostomus discobolus	-	N	common	common
Flannelmouth Sucker	Catostomus latipinnis	-	N*	common	common
Razorback Sucker	Xyrauchen texanus	E-C	N*	rare	common
CENTRARCHIDAE -					
Green Sunfish	Lepomis cyanellus	-	I	uncommon	-
Bluegill Sunfish	Lepomis macrochirus	-	I	uncommon	-
Largemouth Bass	Micropterus salmoides	-	I	uncommon	-
Black Crappie	Pomoxis nigromaculatus	-	I	uncommon	-
COTTIDAE -					
Mottled Sculpin	Cottus bairdi	-	N	common	common
CYPRINIDAE -					
Carp	Cyprinus carpio	-	I	common	-
Humpback Chub	Gila cypha	E-F,C	N*	rare	locally abundant
Bonytail Chub	Gila elegans	E-F,C	N*	extinct	common
Roundtail Chub	Gila robusta	-	N*	common	common
Red Shiner	Notropis lutrensis	-	I	common	-
Sand Shiner	Notropis stramineus	-	I	common	-
Fathead Minnow	Pimephales promelas	-	I	common	-
Colorado Squawfish	Ptychocheilus lucius	E-F,C	N*	rare	common
Speckled Dace	Rhinichthys osculus	-	N	rare	common
CYPRINODONTIDAE -					
Plains Killifish	Fundulus zebrinus	-	I	uncommon	-

TABLE 1 (Cont.)

COMMON NAME	SCIENTIFIC NAME	STATUS	SOURCE	CURRENT ABUNDANCE	HISTORIC ABUNDANCE
ESOCIDAE -					
Northern Pike	Esox lucius	-	I	sporadic	-
ICTALURIDAE -					
Black Bullhead	Ictalurus melas	-	I	common	-
Channel Catfish	Ictalurus punctatus	-	I	common	-
POECILIIDAE -					
Mosquito Fish	Gambusia affinis	-	I	uncommon	-
SALMONIDAE -					
Brown Trout	Salmo trutta	-	I	sporadic	-

primary source: Endangered Wildlife Investigations Job Progress Report SE-3-3, Sept. 1981.

* = endemic
E = endangered
C = Colorado list
F = Federal list
I = introduced
N = native

ENDANGERED AND THREATENED FISHES

Colorado squawfish, *Ptychocheilus lucius*. Federal and state endangered species.

Major adult habitat sites listed include the deep water, large volume areas of Ruby Canyon and Westwater Canyon (Utah). Especially during high flows, squawfish will make considerable movements and individuals can occasionally be found up to the diversion dam at Palisades, about 15 miles above Grand Junction. Also, squawfish will frequently move into large backwater areas, such as the Walter Walker pond at Grand Junction. Spawning sites of squawfish have not been identified in the Colorado River, but it can be assumed that spawning habitat is similar to that documented in the Yampa River -- large gravel and cobble bars in the main river channel. Young squawfish seek quiet, protected areas such as side channels, lagoons and embayments soon after hatching. Young-of-the-year squawfish were found in Ruby Canyon (at Black Rocks) in 1979 and in a side channel a few miles below Loma in 1980. Squawfish spawn at a water temperature of about 70° - 72°F (20° - 21° C). The flow regime of July and August, in relation to maintaining the habitat of the rearing areas and their connections to the main channel, may be of critical importance for establishing the success of a year-class. Further detailed data relating to the factors governing reproductive success should be available from the USFWS endangered species study and the Colorado Division of

Wildlife's monitoring program.

A comprehensive review of information on the Colorado squawfish is given in Behnke and Benson (1980).

Humpback chub, <u>Gila cypha</u>. Federal and state listed endangered species.

The Black Rocks area of Ruby Canyon has the greatest concentration of humpback chub known from the upper Colorado River basin. The humpback chub has rather specific habitat preferences -- deep water canyon areas. Thus, this species was always restricted in its distribution, but became rare after much of its prime canyon habitat was lost during the creation of Lake Powell and Flaming Gorge Reservoir. Adult humpback chub evidently may move some distance during high water. Occasional specimens have been taken above Grand Junction in the Colorado River, but the maintenance of a population evidently is dependent upon an area of deep water, canyon habitat such as in Ruby Canyon and in Westwater Canyon.

Humpback chub spawn at water temperatures of about 65° F (18° C). Spawning has not been observed. Young-of-the-year and juvenile humpback chubs, as young squawfish, are found in quiet, low velocity waters such as backwaters and embayments off of the main river channel, but in Ruby Canyon, they have been found along the shoreline of the main river channel. More comprehensive predictive data on factors influencing successful reproduction of humpback chub can be expected from the USFWS and CDOW studies, but preliminary analysis suggests that humpback chub year-class strength may not be as dependent on backwater and side channel habitat as is the case with the squawfish.

In 1980 the USFWS study team captured "strange" specimens of chub from below De Beque in the Colorado River. These specimens were tentatively identified as Gila cypha, but a detailed analysis of taxonomic characters obtained by Richard Valdez (USFWS, Grand Junction) identifies the De Beque chub as the roundtail chub, Gila robusta. Mr. Valdez presented his data at the annual meeting of the American Fisheries Society (Albuquerque, N.M., September, 1981). Dr. R. R. Miller, University of Michigan, and the leading authority on the taxonomy of the genus Gila, examined the data from the De Beque chub specimens and told me they are definitely not G. cypha and should be recognized as G. robusta.

Hybridization among Gila chubs is common. I believe the population of chubs near De Beque represents G. robusta that is slightly introgressed with G. cypha. That is, a small proportion (perhaps 10%) of G. cypha genes are incorporated into this population of roundtail chub.

More complete information on humpback chub is found in Behnke and Benson (1980).

Bonytail chub, *Gila elegans*. Federal and state listed endangered species.

No documented record of bonytail chub exists for the Colorado River in Colorado, but it can be assumed that the native range of this species was similar to that of the squawfish -- upstream to about Rifle or Glenwood Springs (Behnke and Benson, 1980).

Intensive sampling in the Colorado River (Lake Powell to De Beque) in recent years has failed to find a bonytail chub. In 1980 a few specimens indicative of bonytail X humpback chub hybrids were taken in the Green River of Desolation Canyon. Dr. R. R. Miller, however, has identified these specimens as *G. cypha* (personal communication). It appears unlikely that a population of *G. elegans* persists in the upper Colorado River basin. The only known bonytail chubs are found in Lake Mohave and Lake Havasu in the lower Colorado basin. Offspring from these fish are currently being raised at the Willow Beach, Arizona, National Fish Hatchery.

Razorback sucker, Xyrauchen texanus. Endangered Colorado, not on federal list.

Little is known about the razorback sucker. Although undoubtedly rare, this fish is difficult to capture and records of young razorback suckers in the wild are virtually unknown. Behnke and Benson (1980) and McAda and Wydoski (1980) reviewed what is known about the species. The results of recent studies in Senator Wash Reservoir, California, were presented at a Colorado River water symposium (Las Vegas, November, 1981) and at the Desert Fishes Council meeting (Death Valley, November, 1981).

As with other rare native fishes, the decline of the razorback sucker appears to be related to poor reproductive success. In Senator Wash Reservoir, razorback suckers were observed to spawn over inundated rocky shore areas. Egg predation by non-native fishes, however, appeared to completely eliminate all eggs before hatching. Razorback suckers have been reported to spawn at water temperatures from 54 F (13 C) to 68 F (20 C). McAda and Wydoski (1980) described spawning groups of razorback suckers congregated over gravel bars in the Colorado River and in the lower Yampa River. The appearance of razorback suckers every spring in off channel ponds created from gravel excavation suggests that "lagoon" type of habitat is preferred for spawning and as rearing habitat for the young. This type of habitat is also preferred by non-native centrarchid species and by carp, which, evidently effectively suppress reproduction by egg predation.

Of all the rare, native fishes, the razorback sucker is the only species known to presently occur upstream from the diversion dam at Palisades, Colorado. Adult razorbacks have been observed in the river near De Beque, particularly in April, evidently congregating for spawning. Spawning has not been observed here, nor have any young razorbacks been found.

Colorado River cutthroat trout, _Salmo_ _clarki_ _pleuriticus_. Threatened Colorado, not on federal list.

The decline of the Colorado River cutthroat trout is the result of replacement by brook and brown trouts and of hybridization with rainbow trout. Two populations are known to exist in small tributaries in the Parachute Creek drainage -- Northwater Creek and JQS Gulch. Other, cutthroat-like trout occur in other sections of the Parachute Creek drainage, but these trout have an obvious hybrid influence from rainbow trout.

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[1985]

TECHNIQUES FOR COLLECTING

AND ANALYZING

BENTHIC MACROINVERTEBRATE SAMPLES.

May 8, 1985

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INTRODUCTION:

The concept of using biological parameters as indicators of water quality and the relative health of aquatic ecosystems has been accepted by biologists for some time. Although physical and chemical measurements are important, they are,by themselves,inadequate for the assessment of water quality. Such measurements may fail to discern low levels of contamination or intermittant perturbations which nonetheless have affects on aquatic communities. Whereas a chemist may need to make numerous tests over extended periods of time in order to obtain average values, relatively few samples of aquatic biota will be representative of a community which is a product of all the conditions which have occurred in the recent past. Biological communities will show the effects of toxins or other perturbations of their enviroment long after the actual causes are no longer in evidence(Goodnight, 1973)

Benthic macroinvertebrates have been frequently cited as the most suitable biological indicators of water quality(Gaufin,1973; Weber,1973). They are large enough to be easily collected and, if necessary, identified under field conditions. (Gaufin, 1973) Their life cycles are long enough that they reflect past, as well as present, conditions at a site(Keup et al.,1966; Gaufin,1973). Because their mobility is restricted, compared to groups such as fish, they can not easily move in an out of an area, and so they are directly affected by all enviromental fluctuations(Hynes,1965; Wilhm, 1970). The utility of macroinvertebrates as indicators of enviromental change in aquatic ecosystems is further enhanced by the variable and characteristic response of the diverse taxa to different types of perturbations and modifications(Goodnight,1973).

Hellawell(1978) gives the following reasons for the adoption of macro-invertebrates as the principle group for the biological monitoring of water quality:

(1) popularity--there is a clear advantage in using a group acceptable to those who are to undertake surveillance;

(2) methodology--sampling procedures are well developed and can usually be operated by a single worker;

(3) identification--keys are available for most groups and are in preparation for the difficult groups such as the larvae of chironomids and Trichoptera;

(4) high 'hysteresis' value--their sedentary or relatively stationary habit allows meaningful spatial analyses of survey results and their relatively long life cycles facilitate temporal analyses;

(5) heterogeneity--several phyla are represented making it likely that at least some groups would respond to a given enviromental change;

(6) abundance under favorable conditions facilitates quntitative analysis.

This paper has the following objectives:

(1) To review the techniques and devices that have been used in the sampling of macroinvertebrate benthos in lotic waters.

(2) To likewise review method of data analysis and interpretation which have been applied to samples of aquatic macroinvertebrates.

(3) To determine what sampling techniques and data analysis would best be applied to a study of the effects of Taylor Draw Dam on the benthic macroinvertebrate in the White River in Northwestern Colorado.

DESIGN OF SAMPLING:

Macroinvertebrate sampling can be either qualitative or quantitative (although many sampling techniques probably lie somewhere in between). Qualitative sampling is relatively easy to conduct, but results may be less than satisfactory for the purposes of some studies. Qualitative sampling is usually restricted to general assessments of taxa of aquatic macroinvertebrates present, although observations of relative abundance can be made. In quantitative sampling the objective is to estimate the number of organisms present with sufficient precision so that the results may be subjected to statistical analysis.

If sampling is to be quantitative, the following points should be taken into consideration:

Sampling Location:

When studing the effects of any physical, chemical or biological modification of a river, which can be identified as beginning at a specific point, at least one station should be located below, and one above, this point. Actually, having two stations above the source of the perturbation is preferable; in this way it will be possible to establish the amount of normal variation which occurs between these control stations. If it is important to determine the downstream extent of the effects of the perturbation additional stations should also be establish below its source.

Once sampling stations have been chosen it is necessary to determine the exact locations where sampling replicates will be taken. There at least three commonly used approaches for doing this. Random sampling is the method where the sampling station is divided up into sampling unit sized spaces, each of which has an equal chance of being sampled. Random sampling allows the application of certain statistical methods. It often proves inefficient in rivers as a result of the variable amounts of different habitat types. Stratified random sampling deals with this problem by randomly locating sample replicates within specific habitat types. If habitat types are of different sizes, the number of replicates to be located in each should be adjusted accordingly. In systematic sampling the initial replicate is located at random, while subsequent replicates are located at fixed or regular intervals from it. When this last approach is used, samples cannot be used to determine the statistical error of the mean. There is also the risk that the regular intervals chosen may correspond to some natural variation in the habitat type, and thus the community.

(Merrit and Cummins,1984).

Sample Size:

This should be related to the area or volume of habitat in which the organisms being studied normally move about in, and upon the portion of the bottom fauna that is being investigated(Cummins,1970). Because stream benthic macroinvertebrates frequently exhibit contagious or clumped distribution, a greater number of small sampling units is more efficient than fewer sampling units with greater area(Cummins,1970). Ideally, in quantitative sampling, the sampling unit should be small enough to include just one organism (Hellawell,1978). This is rarely possible in practice. It is also necessary that the sampling unit be no smaller than the largest sediment particle.

Number of Samples:

The number of samples which need to be taken is a function of three things: (1) The size of the mean; (2) the degree of aggregation exhibited by the population; (3) the desired precision of the mean estimate. If degree of aggregation and desired precision are the same, higher density communities will require fewer samples. Most aquatic macroinvertebrate have been found to have the type of distribution referred to as clumped or patchy. How this is interpreted will be a function of the size of the size of the sampling unit and the number collected. All else being equal, communities with non-random(clumped) distribution require a greater number of sampling units for a given degree of precision. Any time greater precision is required, it will be necessary to increase the number of sampling units(Merrit and Cummins,1984).

Sampling Frequency:

This is largely dependant upon the type of study being conducted. Production studies and studies of possible shift in timing of life cycles require the most frequent sampling(Hellawell, 1978). According to Cummins(1970), sampling should be more frequent during the early life stages of the organisms being studied to discern the exponential growth and survivorship curves typical of aquatic macroinvertebrate species. Early life stages of most temperate zone species of aquatic macroinvertebrates are in the late fall or spring. Complete faunal studies of any type, but particularly those looking at species diversity, require at least seasonal frequency. Sampling every 100-300 degree-days has been reccommended as a useful approach by Merrit and Cummins(1984). This is because the life cycles of many aquatic insects, which generally compose a high percentage of the benthic macrofauna in rivers, are under thermal control. It is important that samples which are to be compared on a temporal or spatial basis, be collected in the same season. In a study where time and/or manpower is limited, sampling might be conducted once in the spring and once in the late summer or early fall.

SAMPLING TECHNIQUES:

The number of techniques used to sample lotic macrobenthos is very nearly proportional to the number of investigations conducted. Hellawell(1978) gives one of the most comprehensive and recent reviews. He divides techniques into the broad categories of "active" and "passive" methods.

Most of the conventional methods for collecting benthic macroinvertebrates are considered "active". Samples are available quickly relative to "passive" methods. They are usually intended to take a representative sample of the the macrobenthic community. This is dependant upon the particular sampler, the habitat in which it is used, and the skill of the individual operating it (Hellawell,1978).

In contrast to "active" methods, "passive" sampling collects organisms by relying on their normal behavior patterns. "Passive" are generally believed to depend less upon the ability of the worker. A disadvantage is that they take more time to collect samples-sometimes as long as six to ten weeks.

What follows is a list of the sampling techniques reviewed and described by Hellawell(1978):

I. Active Methods

A. Benthos netting methods

1. In running waters macrobenthos may be detached from stones or by a kicking technique. They are then washed downstream and into the net by the current. Attempts to quantify this method by disturbing the benthos for a fixed time, or over a fixed area have been made.(Macan, 1958; Hynes, 1961)

2. The Surber sampler is a combined net and quadrat. Organisms washed off rocks within the quadrat are washed downstream into the net by the current. Enables quantitative estimates of faunal density to be made; restricted to flowing water less than a meter deep and with relatively small substrae particles. (Surber,1937; Welch, 1948; Macan, 1958)

B. Shovels and dredges

1. Generally employed in shallow waters, shovels are pushed through the substrate. They are basically nets, mounted on a robust frame with a cutting edge in front. They usually have a hood and wings to prvent loss of organisms. (Macan,1958)

2. Dredges are usually used in deeper water sampling, and are pulled rather than pushed. Generally they are constructed for heavier use than shovels. Some models are fitted with teeth or hydraulic jets to aid in disturbing the substrate as they are dragged through it. Some also have screens to prevent them from picking up larger stones.(Holme,1971; Needham and Usinger, 1956; Castagna, 1967)

C. Cylinder and Box Samplers

--open ended cylinders or boxes which are pushed down on or into the river bed to isolate a portion of it. The lower end of the sampler may be provided with serrations to aid in penetration, a rubber gasket may be used as a seal.

--Methods for collecting benthos withing the cylinder:

--Hand or dip-net(Hynes,1971)

--A cylinder which fits inside the sampler and has a rotary valve on the bottom(Wilding, 1940; Welch, 1948)

--In running waters the cylinder may have a mesh covered opening upstream and an opening leading into a net on the dowstream side. The bottom is stirred up and benthic materials are washed into the net(Hess,1941).

--Neill(1938) developed a method similar to Hess, but used movable doors over the upstream and net openings.

--Hughs(1975) used Neill's sampler in conjunction with an electric current to induce organisms to release their hold on the substrate.

D. Grabs

--designed to remove a portion of the substrate by a biting action; organisms are removed from this in the laboratory; whereas a dredge is dragged over the substrate, a grab enters it vertically; usually the jaws are closed by a strong spring or the action of a lever and pulley mechanism.

--in shallow habitats it may be possible to operate grabs using a a rod as opposed to the cables used in deeper waters; penetration is increased by pusing on the rod.

--efficiency of grabs depends upon: (1) penetration depth; (2) closure angle; (3) degree of jaw closure; (4) shock-wave formation; (5) stability.

E. Core samplers.

--essentially tubes which are pushed vertically into the substrate and retain substrate materials when they are removed

--corers generally take smaller samples than grabs, their diameters are usually less than 15 cm.

-- Corers are usually of similar design; early models were constructed entirely of brass, more recently plastic tubes have been used to facilitate viewing upon recovery,(Kajak,et al., 1965; Brinkhurst et al.,1969); multiple corers designed to expedite sampling(Hamilton et al.,1970;Hakala,1971); a self closing valve, at the top of the tube, is necessary to allow free flow of water through the tube on its descent; piston core samplers may be used to penetrate resistant substrate(Kajak,1971)

--shock-waves are a problem for corers as well as grabs.

--adding weights to corers can increase penetration, but will increase instability(the corer may tip over and not enter the streambed vertically(Brinkhurst et al., 1969).

--simple shallow water core samplers have been designed by Dendy(1944) and Mordukhai-Boltovlkoi(1958).

F. Air-lift samplers.

--air-jets scour the substrate and the rising air lifts water, lighter substrate, and fauna up a pipe to the surface.

--Pearson et al.(1973) have designed an air-lift sampler which can be operated from the surface in water up to 4 meters deep.

--an air-lift sampler similar to those used in marine archeology was designed by Mackey(1972); the sampler has the tendency to work like a corer in soft substrates and is capable of lifting unionid mussels and stones up to 40 mm. in diameter; this sampler is thought to be less quantitative than Pearson's, as it may have a tendency to draw materials from places other than immediately below the sampler.

G. Samplers for macrophyte dwelling fauna

1. Wisconsin trap--a large bag attache to a hinged fram(36 x 37 cm.) The bag is pulled over the top of a plant and the hinged frame is closed, cutting off the plant at the base and trapping it and its associated fauna inside the bag(Welch, 1948).

2. Macan's sampler--Works like a hand operated grab. It is lowered over the plant or plants and the jaws are closed by means of long handles, again, cutting off the plant at the base(Macan,1949).

H. Diver-operated samplers

--divers can assist in the collection of conventional samples in deeper water or operate equipment themselves; this may help in the correct placement and satisfactory performance of the sampling gear.

II. Passive Methods

A. Drift samplers

--usually nets, placed in the current to collect animals that are moving actively or passively with it.

--as some species are more inclined to drift than others, this is a selective sampling technique.

--drift sampling is a qualitative method; it is difficult relate to animals in the drift with benthic communities.

--catch is influenced by the position of the net in the water column-may catch benthic dweller if too low, aerial species if too close to the surface.

--many net patterns are susceptible to clogging and backflow; this can be prevented by using a net with a small upstream opening and a relatively large net area.

--most common mesh sizes 0.3 to 0.5 mm.

--the results obtained from drift sampling depend on pattern and position of the sampler, the volume sampled, mesh size and time of sampling.

B. Artificial-substrate samplers

--the design of artificial-substrate samplers is the result of attempts to decrease the variability inherent in conventional methods.

--objective of artificial substrate sampling is to provide the fauna with a colonizable substrate which may subsequently be retreived by the investigator.

--artificial substrate samplers may be designed to mimic natural substrates; even so they are generally a good deal more uniform.

1. Plane-surface samplers--Examples include concrete blocks (Britt, 1955), metal plates(Mundie, 1965), metal cones (Wolfe and Peterson, 1958), strips of polyethelene sheet (Williams and Obeng,1962), andvertical polyethelene plates (Besch _et al_.,1962). Plane-surface samplers are often selective.

2. Multiple-plate samplers--The original was made from eight plates of 8 cm. sq., 3 mm, thick, tempered hardboard(Hester and Dendy,1962). The plates were assembled on a long bolt with 2.5 cm. spacers. Samplers are usually suspended in mid-water by means of a wire and left for colonization. Variations on the original sampler include 14-plates(Fullner, 1971), round plates to deal with the alignment problem (Mason _et al_., 1973) and sand blasted PVC plates(Hellawell and Jellings, unpublished).

3. Basket samplers--Earliest models were rock-filled wire-mesh baskets of Wene and Wickliff(1940) and Scott's(1958) 'brush-boxes'. More recent models include barbecue-baskets filled with limestone or porcelain spheres(Mason _et al_.,1967; Anderson and Mason,1968; Fullner,1971; Mason _et al_.,1973) Many other natural and artificial materials have been used to fill barbecue and other types of baskets(Jacobi,1971; Dickson _et al_., 1971; Hellawell and Jellings, unpublished) When natural materials such as stones are used, the amount of interstitial space available is unknown and probably not constant. Some researchers have suspended their baskets from floats(Mason _et al_., 1967; 1973); others have let them rest on the bottom. Special net and sampler design(Hilsenhoff, 1969) and the far simpler method of attaching a fine foldable mesh bag to the sampler itself(Bull,1968) have been used to avoid the loss of animals when recovering samplers.

4. Synthetic mesh/artificial weeds.-- "Conservation webbing" has been tested by Simmons and Winfield(1971) and Prins and Black (1971). The former used it rolled in baskets, the latter attached to the undersides of styrofoam floats. Macan and Kitching(1972) constructed artificial _Littorella_ and _Carex_ from polypropylene rope and shrimp net. Finished 'mats' were weighted with stones in pockets of mesh and used to collect benthic fauna in a lake.

5. Bricks--Bricks and tiles with grooves in the undersides(Britt, 1955; Albrecht,1953; Macan,1958) were judged to be of poor value in sampling due to the small amount of volume they actually offered for colonization. 'Air-bricks', used by Hellawell and Jellings(unpublished), provide interstitial,as well as external surface area, for colonization.

6. Log samplers--Used in a study of colonization in rivers by Nilsen and Larimore(1973). Smooth, solid logs, 6 cm. in diameter and 23 cm. long, were colonized by a fauna not quite as diverse as that found on natural logs. Logs were anchored by means of a wire attached to an eyebolt drilled into one end of each log. Colonization was not complete when the samplers were collected after periods of four to six weeks.

7. Embedded samplers--Most of these are variations on the trays filled with rocks and placed in excavations in the substrate by Moon(1935). Some of the more unique varieties include:
   --A sampler by Coleman and Hynes(1970) which used nested cylinders(2), the inner one which was perforated to allow for interstitial water flow. When it was retreived, a mesh bag was rolled up around the inner cylinder from the bottom to prevent the loss of animals.
   --Radford and Hartland-Rowe(1970) used perforated cans with good results.
   --Ford(1962) used a box, embedded in the bottom, with open ends facing upstream and downstream. After the sampler was allowed to sit out for five to six weeks, plexiglass plates were slid into grooves to seal off the open ends, and the sampler was retreived.

Although the literature abounds with accounts of sampling devices of all varieties, there are few studies which compare the performances of more than two of these at a time. Studies which attempt a critical look at anything like the full range of samplers are rarer still. No attempts have been made to compare and evaluate the performances of all(or even most) of the different sampling techniques under similar or controlled conditions.

Cummins(1962) describes five basic types of benthic sampling methods, but offers no preference regarding specific techniques. Cummins(1975), states that techniques which completely remove sediments are frequently the most efficient. He also notes that "artificial substrata answer the question, 'What diversity and density would be present if the substrate was like that of the sampler?'"

Hynes(1971) makes the following reccomendations with regard to methods for sampling benthic macroinvertebrates in streams:

1. For sand and silt substrates, an Ekman grab, operated by hand in shallow water, and with a pole in deeper water.

2. For stony substrates, cylindrical box sampler, preferably with a foam rubber skirt to give it a better seal against the bottom.

3. For rooted aquatic plants, a Wisconsin net.

4. For algae, or plants attached to rocks, scraping of a known area upstream of a hand held net, or hand collection of a known volume.

Southwood(1978) and Merritt and Cummins(1984) give comprehensive listings of sampling devices and techniques, but make no attempt to rate their performance outside of giving the types of habitat they are suitable for. Welch(1948) and Welch(1980) give brief reviews of techniques for stream bottom sampling, but offer no advice on which techniques might be preferable under which conditions.

Based primarily on a review of the literature, Hellawell(1978) gives the following recommendations for sampling benthic macroinvertebrates in deep, medium and shallow lotic habitats:

Deep Water(greater than 3 meters):

Slow:

(1) Multiple corer(especially for soft sediments)
(2) Diver operated air-lift
(3) Ponar grab in hard substrates; Ekman grab for soft substrates.

Fast:

--Fast water could prevent corers and grabs from entering the substrate perpendicularly; divers would require extra weight to avoid being swept downstream and the return pipe for an air-lift sampler could be extremely difficult to control. In the fastest waters, artificial substrate samplers, attached by a chain or cable, might be the only feasible method.

Water of moderate depth(1 meter to 3 meters):

--In general methods favored for deep waters are suitable for waters of moderate depth. The performance of corers and grabs might be improved by operating them with poles as opposed to cables. Comparisons between grabs and air lift samplers indicated their abilities to take different species differed, but that the results they gave were basically similar(Pearson *et al*., 1973). Again, anchored artificial substrates are probably the only method which will work in the swiftest waters.

Shallow Water(less than 1 meter):

Swift, substrate of boulders or large stones:

Qualitative:

1. Lifting stones upstream of a hand-net.
2. Artificial substrates.

Quantitative:

--Surber sampler where it can be located correctly(substrate size does not exceed quadrat area).

Moderate velocity, substrate of gravel:

Qualitative:

--Kicking upstream of a hand-net.

Quantitative:

1. Surber sampler.
2. Air-lift sampler if depth is adequate.

Slow, substrate of gravel to sand:

Qualitative:

--Shovel sampler.

Quantitative:

1. Cylinder samplers.
2. Air-lift sampler if depth is adequate.

Very slow to static, substrate of sand or silt:

Qualitative:

--Shovel sampler.

Quantitative:

1. Core sampler.
2. Air-lift sampler if depth is adequate.

INDICES AND OTHER METHODS OF ANALYZING DATA:

Hellawell(1978) also gives the most complete, recent review of methods for the analysis of benthic macroinvertebrate samples from streams. He recognizes two broad approaches: Pollution indices are those which were designed to measure the response of communities to pollution and particularly organic enrichment; diversity indices are based on theoretical concepts of the structure of communities. Along with diversity and pollution indices Hellawell looks at comparitive indices or coefficients of association, ranking methods and some techniques which are intrinsically visual. As he points out the purpose of these methods is primarily to summarize large quantities of data. In the process some information is bound to be lost or obscured. The best indices and analysis methods are those which lose the least information and may even illuminate things which were not clear from the raw data. What follows is is an outline of the methods reviewed by Hellawell:

I. Pollution Indices

A. Numerical developments of the Saprobien System.

1. Relative Purity(or Pollution) Index(Knopp,1954; 1955)
2. Saprobity Index of Pantle and Buck(Buck, 1955a; 1955b; Pantle, 1956; 1960)
3. Saprobity Index of Zelinka and Marvan(Marvan,1961)
4. Saprobity Index of Dittmar(1959)

B. Biotic Indices.

1. Trent Biotic Index(Woodiwiss, 1964)
2. Lothians Index(Graham,1964; 1965)
3. Biotic Score(Chandler,1970)
4. Empirical Biotic Index(Chutter,1972)
5. Palmer's Index(Palmer, 1969)
6. Becks's Index(Beck, 1954)

7. Taxonomic Ratios

--wet weight of insects to tubificids(King and Ball, 1964; Goodnight and Whitley(1960)

--tubificids and Limnodrilus hoffmeisteri to all other insects (Brinkhurst, 1960a; 1960b;)

--Gammarus to Asellus(Hawkes and Davis,1971)

II. Diversity Indices

A. Models of community structure.

1. Logarithmic Series Model(Fischer *et al.*, 1943)

2. Lognormal Distribution Model(Preston,1948)

3. Ordered Random Interval or 'Broken Stick' Model(MacArthur,1957)

B. Indices of Community Diversity.

1. Mehinick's Index(Mehinick,1964)

2. Margalef's Index(Margalef,1951)

3. Simpson's Index(Simpson,1949)

4. Information Theory Indices(Shannon,1948; Pielou,1969)

5. McIntosh's Index(McIntosh,1967)

6. Sequential Comparison Index(Cairns *et al.*,1968)

III. Comparative Indices-Coefficients of Similarity of Association

A. Qualitative

1. Kothe's 'Arterfehlbetrag' or Species Defecit(Kothe,1962)

2. Coefficients of similarity

a. Jaccard(1912)

b. Kulezynski(1928)

c. Sorenson(1948)

d. Mountford(1962)

3. Coefficient of Association T(Looman and Campbell,1960;Orth,1973)

B. Quantitative

1. Raabe's Coefficient(Raabe,1952)

2. Czekanowski's Coefficient(Czekanowski,1913)

C. Measures of Distance(Sokal,1961)

IV. Ranking Methods.

A. Spearman's Rank Correlation Coefficient(Spearman,1913)

B. Kendall's Rank Correlation Coefficient(Kendal, 1962)

V. Graphic Methods.

A. Recurrent Groups(Fager,1957)

B. Cluster Analysis(Sokal and Sneath,1963)

1. Single Linkage Cluster Analysis

2. Average Linkage Cluster Analysis

3. Dendograms

C. Minimum Spanning Trees(Gowen and Ross,1969)

D. Control Charts(Kendall and Stuart, 1968)

Although indices are used frequently there is little information available which compares them or evaluates their performances for specific types of studies.

Galat(1974) looked at Beck's Biotic Index, Trophic Condition Index, Shannon Diversity Index, the eveness coponent of Shannon's Index, Equitability as calculated from MacArthur(1957) by Lloyd and Ghelardi(1964), and the Sequential Comparison Index. He evaluated the performance of each with respect to background variability at a single station, variability between several'homogeneous' stations, and between several stations known to have different water qualities. None of these indices showed a significant correlation to the number of individuals taken; Beck's and Shannon's Indices were affected by the numbers of taxa present. Beck's Biotic , Shannon Diversity and the Sequential Comparison Index, all proved insensitive to slight organic enrichment; however, Shannon's Index and the Sequential Comparison Index were both sensitive to moderate differences in water quality. Scores calculated for the Sequential Comparison Index depended upon who they were calculated by. Trophic Condition, Eveness and Equitability indices, were all sensitive measures of organic enrichment and background in the stream macroinvertebrate community.

None of these indices was entirely satisfactory under all of the circumstances tested. When Trophic Condition Index and Eqitability were simultaneously expressed on a graph using Bivariate Statistical Procedures they gave what Galat concluded was a "meaningful and sensitive expression of invertebrate community structure."

Dejong(1975) calculated species diversity on sets of real and artificial data using three different indices. The three indices used were Shannon's, Simpson's and McIntosh's. Each was analyzed to determine its relationship to the richness and eveness coponents of diversity. Simpson's and McIntosh's indices were found to be more clearly representative of diverstiy when expressed in probits or graphed on probibility paper. Simpson's and McIntosh's indices were also found to be more influenced by eveness and less by richness than Shannon's. All three indices show a linear relationship to richness and eveness converted by taking the log to the base 2.

Hellawell(1978) applied numerous indices and other techniques for analysis of benthic macroinvertebrate samples to data collected on the River Cynon(Learner et al., 1971) and the River Derwent(Hynes,1961). Data on the River Cynon was collected at nine stations. The upper four stations were relatively clean, the remaining five were subject to varying degrees of organic and industrial pollution. The River Derwent was a relatively unpolluted river. Data was collected at a single station for a period of nine years. In 1961 sampling revealed a tremendous increase in mayflies of the genus Baetis and a corresponding decrease in numbers of several other species. No explanation was found for this event.

In addition to the five broad types of data analysis listed from Hellawell above, two ways of using the basic data, biomass and number of taxa were looked at. The performances of the various indices and other methods were judged by comparing direction and relative magnitude of changes with those observed in the initial consideration of the basic data, and with the overall pattern as discerned from conclusions based on all of the analyses. A summary of Hellawell's comments follows:

Basic Data(number of taxa and biomass)

Although simple, this approach worked well in light of the criteria given above. Although they might prove unmanageable in larger or more extensive surveys, species lists, relative abundances and biomass are probably adequate for preliminary work.

Pollution Indices

Shortcomings of systems developed specifically for the detection of pollution were illustrated by the failure of the Saprobien Indices of Pantle and Buck, and Zelinka and Marvan. on the River Derwent.

Lothian 's Index was found to be insensitive on both rivers. Evidently its scale was too large for the magnitude of changes observed.

Although it was more sensitive than Lothian's Index, Trent's Index did not show a clear pattern with either data set.

Chandler's Biotic Score proved valuble with data from the River Cynon and the River Derwent. It should be kept in mind that low scores are not necessarily indications of pollution or perturbation.

Diversity Indices

Most of these were effective in detecting all changes in community structure, including those resulting from pollution.

Menhinick's Index did not perform well for the time series data from the River Derwent.

Shannon's Index was apparently unaffected by using higher level taxonomic groups such as familie and orders.

McIntosh's Index worked extremely with all of the data

The lognormal model of Preston was judged to be a poor on for defining the structure of benthic macroinvertebrate communities.

Comparative Indices:

The Species Deficit method of Kothe worked well for the River Cynon but was not useful in looking at data from the Derwent.

None of the four coefficients of similarity performed well in comparing consecutive stations; Jaccard's illustrated the 1961 change in the community in the Derwent when each years data was compared with the average for all six years. In matrix form(all station pairs compared) Jaccard's Coefficient clearly depicted discontinuities in the River Cynon stations.

Both Mountford's Index and Coefficient of Association T worked well in matrix form when applied to the Cynon data; they were less than satisfactory for data from the River Derwent.

Raabe's and Czekanowski's coefficients were both effective in comparing years or pairs of stations. They worked better still in matrix form. Sokal's distanc measure worked well for the Derwent. Because it is the easiest to calculate, Czekanowski's coefficient is preferred.

Ranking Methods :

Spearman's and Kendall's gave similar results. Spearman's is preferred for ease of calculation, and because its limits are not effected by ties.

Recurrent Grouping and Cluster Analysis

Fager's Recurrent Grouping Method an effective, objective way of looking at the affinities of samples.

Average-linkage clustering method is also useful.

Both methods are tedious to calculate.

DISCUSSION:

The preceding review was done with the intention of determining which methods of sampling and data analysis would be the most appropriate to apply to a study of the effects of regualtion on macroinvertebrate communities in the White River in Northwestern Colorado. In this section of the paper, I will attempt attempt to draw some conclusions regarding this.

The White River flows over 200 miles from its source in the Flattop Mountains of Colorado to its confluence with the Green River in Utah. Over this distance it flows through a variety of terrestrial habitats, including pine, spruce and aspen forests, semi-desert shrubs, lands under cultivation, cottonwood bottoms, and sandstone canyons. Above the town of Meeker, the river is a prime trout and whitefish fishery; below Meeker it undergoes a transition, becoming steadily more turbid, and below the confluence of Piceance Creek it contains primarily species which are considered "rough-fish". Flannel-mouth suckers dominate in terms of biomass and numbers. Other native species include Blue-head suckers(Catostomus discobolus), round-tail chubs(Gila robusta) and the endangered Colorado River squawfish(Ptychocheilus lucius).

In August of 1982, construction was begun on Taylor Draw Dam at river mile 104.5 from the confluence with the Green River. The dam was scheduled for completion the following fall, but delays of construction were caused by exceptionally high runoff in the spring. As a result, the dam was not closed until September of 1984.

Fieldwork for this study was begun in September of 1983 and continued in the summer of 1984. It will be completed in the Fall of 1985 after a final season of data collection.

Design of Sampling:

To date, no attempt has been made to randomize macroinvertebrate sampling in this study. Random and stratified random sampling on the White River are complicated because large portions of the river bottom can not be sampled due to depth and current velocity, and those portions which can be sampled are irregular and variable in size and shape. Because of the limits of time and manpower, a random sampling scheme for all the currently existing stations(five below, and one above, the dam). Random,or stratified random,sampling may be attempted at stations above, and immediately below, the dam.

Samples will be collected at least once a month from April to October, except when this is prevented by runoff.

Using a kick-screen and T-sampler(Mackie and Baily, 1981), techniques described by Cummings(1975), Hellawell(1978) and Merrit and Cummings(1984) will be used to determine optimum sample size and number.

Sampling Methods:

Sampling methods used to date include a Surber square-foot sampler, a one meter square kick-screen and barbecue baskets filled with native river rocks and imbedded in the substratum. None of these techniques has proved satisfactory. All are restricted to relatively shallow water, and so a considerable portion of the river bottom cannot be sampled. The Surber sampler generally took very few organisms and could be operated only in shallowest of water. The kick-screen could be operated in deeper water and took far greater numbers of organisms, but is considered to be only semi-quantitative. Basket samplers collected large numbers of organisms, but I feel that by snagging large quantities of extremely course organic matter they created an artificially improved habitat for organisms depending on such materials as a food source.

None of the techniques found in the literature appear to hold much promise for sampling deep water habitat on the White River. In such areas the swift current would foil the use of grabs, corers or air-lift samplers.

Because of the large average substrate size and the armoring which is predominant on the river bottom grabs and corers would not even be able to collect samples. Cost is another factor in discounting air-lift samplers. Artificial substrate samplers are not appropriate for the purposes of this study; some of the changes which may occurr in macroinvertebrate communities below the dam, would be the result of physical changes in the substrate. Such changes might not affect organisms dwelling in artificial substrates.

It appears the only alternative is to restrict sampling to shallow water habitats. Kick-screens are probably adequate for comparison between stations, and within stations for samples taken at different times. Comparison with more quantitative methods would be useful in determining just how effective this technique is. Construction of a relatively cheap modification of the Hess-type sampler is described by Mackie and Baily(1981). Although there is no other information available regarding this sampler, these authors compared it with a Surber sampler and found it to be more efficient in terms of the numbers of organisms taken in most taxa. Cylinder and box samplers of this general type are considered by most authors to be more quantitative methods than bottom-net type samplers such as the Surber sampler and kick-screen. Current plans are to construct such a sampler for comparison to kick-screen samples in 1985.

Data Analysis:

Diversity and comparative indices are probably the most applicable to this study. It should be possible to use comparative indices for comparisons between different station pairs, or between two samples collected at different times at a single station. Czekanowski's Coeffecient, Mountford's Index and Coefficient of Association-T are likely candidates for this purpose. Of the diversity indices, McIntosh's was found by Hellawell(1978) to be the most sensitive. Having been found by DeJong(1975) to weight the eveness component of diversity more heavily than species richness, this index may be comparable to Equitability --one of the measures favored by Galat(1974). Equitability and McIntosh's Index will probably be compared for White River macroinvertebrate data. Other techniques to be used will be decided upon when data analysis has proceeded further.

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FISHES, MACROINVERTEBRATES, AND AQUATIC HABITATS OF THE PURGATOIRE RIVER IN PIÑON CANYON, COLORADO

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ABSTRACT—We describe physical habitat and aquatic biota of a relatively undisturbed canyon reach of the Purgatoire River and its tributaries in southeastern Colorado. Flow regimes are highly variable due to unpredictable, brief, intense summer floods. River habitat consists of long, deep, silty pools with few large boulders separated by short cobble riffles, whereas tributaries contain isolated pools maintained by groundwater. Water chemistry and temperatures were within ranges tolerable by plains stream fishes at all sites. Benthic macroinvertebrate assemblages included representatives of four regional faunas. At lotic sites, a few taxa of two detritus-collecting functional groups predominated. The fish fauna is depauperate, consisting of only 11 native species which we divided into river, perennial stream, and generalist faunal associations. However, most fishes have generalized habitat, trophic, and reproductive requirements, which seem to adapt them to survive the harsh environmental conditions in this plains stream. Although the four species that made up 95% of individuals generally persisted at river sites over the 5- to 7-year period sampled, the abundance of red shiners declined markedly from 1983 to 1987 despite favorable flow regimes.

Little is known of the ecology and physical characteristics of streams of the western Great Plains (Matthews, 1988). However, fish and macroinvertebrate assemblages in such systems can be expected to differ from those in more mesic regions due to extremes in physical characteristics such as flow, turbidity, and temperature. Relatively pristine habitats in streams of the western plains are scarce due to a profusion of human activities, such as channelization, damming, dewatering, mining, overgrazing, use of agricultural chemicals, release of municipal and industrial wastes, and the introduction of nonnative species (Cross and Moss, 1987). Although the Purgatoire River is not immune to these perturbations, the reach we discuss here is one of few comparatively undisturbed remnants in the region and, therefore, offers an opportunity to study habitat and aquatic biota in a relatively natural stream of the western plains.

The purpose of this paper is to describe the flow regime, physical habitat, and water chemistry of the Purgatoire River and its tributaries in Piñon Canyon and to document the species composition, distribution, and ecological characteristics of assemblages of benthic macroinvertebrates and fishes of these waters. Few reports of aquatic biota exist for this general area (Cope and Yarrow, 1875; Jordan and Evermann, 1891; D. Vana-Miller, in litt.), and none for the specific reach. We show that this reach harbors depauperate assemblages of entirely native aquatic biota which have generalized ecological requirements that adapt them for persistence in plains streams.

MATERIALS AND METHODS—*Study Area*—The Purgatoire River was a seventh-order (sensu Strahler, 1957) tributary of the Arkansas River in southeastern Colorado (Fig. 1). Our study area included a sixth-order reach of the river and 10 ephemeral, intermittent, or perennial tributaries adjacent to and on the Piñon Canyon Maneuver Site (PCMS), a 988 km^2 United States Army facility in Las Animas Co. used for tracked vehicle training since summer 1985. Watershed area at the United States Geological Survey gage (07126485) near the downstream end of the reach was 6,825 km^2.

Although headwaters of the Purgatoire River were in the Rocky Mountains, the basin below Trinidad, Colorado (about 30 km upstream of the study area), was in the Southwestern Tablelands ecoregion (Omernik and Gallant, 1987). The climate there was semiarid with mean annual precipitation of about 30 cm,

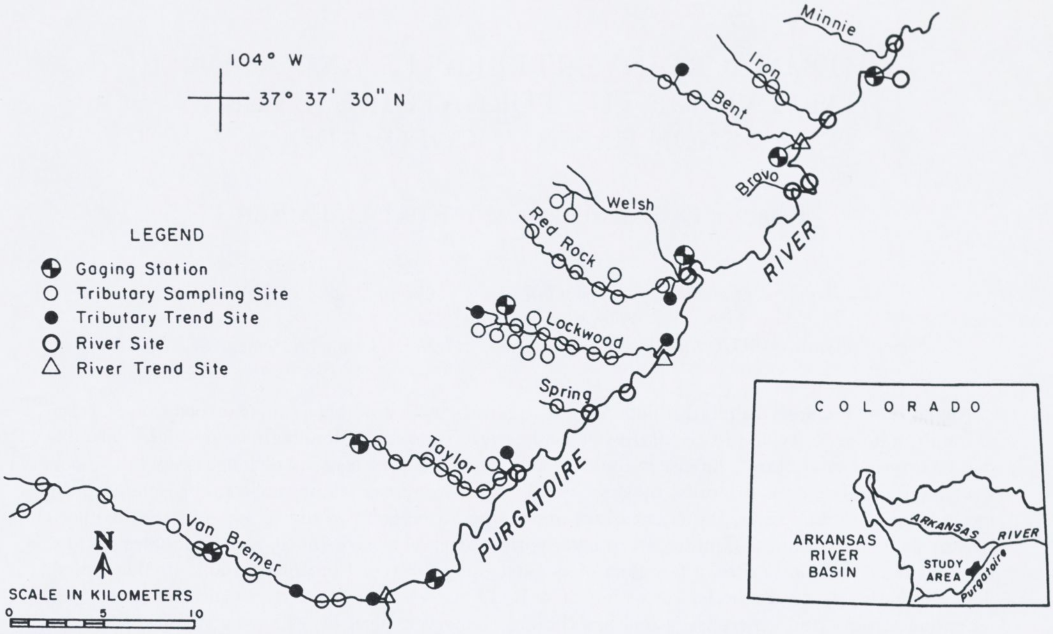

FIG. 1—Map showing location of the Purgatoire River and 10 tributaries in Piñon Canyon. Sites shown are those sampled during 1987 through 1989. The three river trend sites and seven tributary trend sites were sampled repeatedly (see text).

most of which fell from localized, intense thunderstorms (von Guerard et al., 1987). Mean monthly air temperatures ranged from −1°C in January to 23°C in July (Gese et al., 1988). Vegetation varied from piñon-juniper (*Pinus edulis* and *Juniperus monosperma*) in the uplands, to shortgrass prairie (primarily blue grama, *Bouteloua gracilis*) on the plains (Shaw et al., 1989). Riparian vegetation was primarily willow (*Salix exigua*) and cottonwood (*Populus deltoides*; few reached large size) but was often invaded by tamarix (*Tamarix pentandra*).

The Purgatoire River flowed through a Dakota sandstone canyon 60 to 1,800 m wide and up to 150 m deep. Since January 1975, discharge from the mountainous headwaters has been regulated by Trinidad Dam, a hypolimnial-release reservoir (2.1×10^8 m^3 capacity) 32 km upstream of the study reach. The 10 tributaries we studied originated on upland slopes northwest of the river and downcut through sandstone canyons as they approached Piñon Canyon (Fig. 1). Four tributaries (Taylor Arroyo, Lockwood Canyon, Red Rock Arroyo, and Bent Canyon) were intermittent with permanent, isolated lentic pools where their channels intersect the water table. Van Bremer Arroyo was perennial, but discharge was supplemented by irrigation water returning to the river during May through October (von Guerard et al., 1987). The other five tributaries were ephemeral or intermittent but contained no aquatic habitats that supported fish (Fausch and Bramblett, 1991).

Biological Sampling—Fish communities were sampled at 15 river and 51 tributary sites during 1983 and 1987 through 1989. All river sites and four tributary sites were sampled during August and September 1983, prior to Army training. Twelve of 15 river sites were sampled during September and October 1987, nine of which were located just downstream from tributary confluences (two tributaries joined before reaching the river) and three located about midway between confluences (Fig. 1). Sites midway between the confluences of Van Bremer and Taylor, Taylor and Spring, and Red Rock and Bravo were sampled only in 1983. Three of the confluence sites, located near the upstream (Van Bremer), middle (Lockwood), and downstream (Bent) ends of the reach, were also sampled during March 1988, September 1988, and March 1989. Two riffles and two pools were sampled at each river site.

From October 1987 through June 1988, 51 tributary sites were sampled during a broad spatial survey (Fig. 1). Thirty-five sites were permanent pools in the four intermittent tributaries where fish were found, although one pool that held fish dried up soon after sampling and two others appeared permanent but had no fish (the most upstream site in Taylor Arroyo was above a small waterfall, the only migration barrier found). Six pools in three other tributaries appeared