# Periphyton functional and structural response, within semi-natural surrogate streams, to artificially induced water quality perturbations

by

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to the required standard

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## Abstract

The concept, design and operation of a standardized, flow-through, split-stream surrogate mesocosm (stream trough) is described within which, *in-situ*, experimental research was conducted on the contained, biotic component of natural periphyton communities. The surrogate streams enable i) sensitive, sub-lethal and long-term monitoring for detecting environmental impact in ambient waters, and ii) development of standardized protocols for biological toxicity tests and nutrient loading. Siphoned water from a large reservoir supply negates the need for electrically powered pumps; the facility is relatively inexpensive, easily fabricated from Poly-Vinyl-Chloride (PVC) and minimal repair and maintenance is required for year-round operation. Four basic structural components comprise the system including siphon apparatus and header box, manifold supplying water and treatment additives, surrogate streams, and a drainage system.

High quality drinking water drawn from Victoria, British Columbia's Sooke and Humpback Reservoirs, was used in experimental protocols exploring periphyton biomass accrual changes resulting from manipulating phosphorus, nitrogen, light intensity, pH and velocity. How substrata affected accrual was assessed using borosilicate microscope slides, natural river rock and poly-styrofoam sheets. These initial studies sought to elucidate how various stressors affect properties of benthic microalgal communities *in-situ*, since trough mesocosms permit this complex community to be manipulated using a standardized sampling and processing methodology. The development of a standard method within a conceptual framework was directed toward elucidating the integrating role of periphyton, in shallow lakes and streams, with other ecosystem processes.

Following development of the surrogate streams, a four year field study (1988-1991) quantified the nutrient contribution of sewage effluent (with particular reference to phosphorus), from the resort municipality of Whistler, to the Cheakamus River, British Columbia, Canada. The stream trough study revealed the river's limited phosphorus assimilative capacity, with

downstream aesthetics and fisheries habitat adversely affected at periphyton biomass values exceeding 2500 $\mu g/cm^2$ (dry weight). Based upon these studies, a Liquid Waste Management Plan (LWMP) recommended effluent discharge directly into the adjacent Squamish River.

A surrogate stream study, on the latter river, in the summer of 1991, was conducted to assess how the effluent diversion would affect water quality, especially with respect to the development of undesirable algal growth. The latter could adversely affect fisheries spawning and rearing habitat downstream of the discharge point. Following a year-long public review of an expanded range of possible LWMP options, enhanced orthophosphorus reduction, using biological tertiary treatment, was adopted. The absence of an adequate database, upon which to model processes effecting biomass accrual, in the Lower Cheakamus River, signals further study of nutrient loading dynamics in both the river and the reservoir, together with an expanded investigation of top-down processes linking herbivory and algal production.

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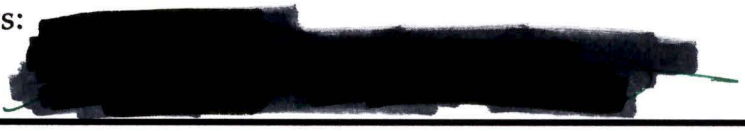

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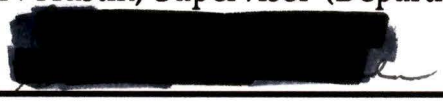

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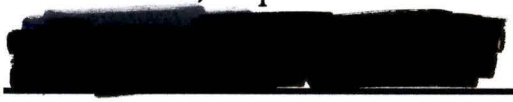

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## Acknowledgements

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He knew, as did Socrates, that we know nothing.

Max Born, speaking of Albert Einstein after his death. (Clark, 1971)

## Chapter I. General Introduction

During the last two decades numerous investigations have confirmed detrimental alterations to the habitat of aquatic organisms, primarily resulting from inappropriate application of agricultural, aquacultural and industrial technologies (Almer, 1974) and the deforming technology of civilization. These studies have generally been concerned with aquatic resources in Western (Almer, 1974; Blanck, 1985; Flower & Battarbee, 1983; Fjerdingstad and Nilssen, 1983; Bellot *et al.*, 1989) and Eastern Europe (Banasak, 1989; Ciesielska *et al.*, 1989), Eastern Canada (Schindler, 1987), the United States (Galloway and Crosby, 1979) and Japan. Initially, changes in aquatic community structure following toxic deposition were examined. Largely, because of cost, measurement of deformation of aquatic ecosystems by pollution has been carried out by addressing indirect physical and chemical parameters (Lucey *et al.*, 1986). Increasing confidence, however, is being placed upon the assessment and monitoring of direct pollution effects by employing the affected organisms themselves, measuring responses ranging from their presence or absence, to alterations in their physiological or behavioral patterns (Dixit *et al.*, 1992).

Recent studies integrating physico/chemical and biological responses to pollution have shown that general or regional comparisons are of little value due to area-specific patterns of rainfall, runoff, topography and the variable capacity of aquatic systems to buffer specific, deleterious, compounds. Dixit *et al.*, (1992), describing the use of algae (diatoms) as powerful indicators of environmental change, promotes studying the biotic components of the specific ecosystems being perturbed, since the latter integrate information on changing environmental dynamics. Current perceptions, resulting from field-based ecological research, indicate that further resolution of the complex interactions effecting ecosystem periodicity and fluctuations require long-term studies (Remmert, 1985; Odum, 1985).

Schindler (1987) cautions that in the absence of such long-term studies our spatial and temporal understanding and prediction of the nature and extent of eco-system response to biotic and abiotic disruptions will remain

fragmentary. Although a substantial literature exists documenting environmentally induced, acute, sub-lethal and fatal toxicity criteria for larger, economically valuable, species including fish, shellfish, Crustacea, insects and their larvae (De Pauw and Vanhooren, 1983), considerably less emphasis has been paid to the more difficult to work with organisms such as microbiota or *microwildlife* (Austin 1969). This is surprising given the acknowledged importance of the proportional metabolic contribution of microbiota (Bell *et al.*, 1979; Ladd *et al.*, 1979; Welch, 1980) and their basic position in trophic systems.

Aquatic microwildlife include two major communities, the plankton (pelagic) and periphyton (benthic and attached). A pluralistic terminology concerning the latter exists in the literature, in which Round (1981), Lakatos (1976) and Aloi (1990) have discussed inadequacies. For the purposes of this work, however, the term periphyton (or biofilm), as defined by Cooke (1956) will be used:

> *those organisms (both plant and animal) attached or clinging to stems, leaves or rooted plants or other surfaces projecting above the bottom.*

Weitzel (1979) distinguishes between periphyton "all aquatic organisms (microflora) growing on submerged substrates", and *aufwuchs* "those organisms that are firmly attached to a substrate and including those free-living forms within the mat but not penetrating into it". Others (Sladeckova, 1962; Odum and Hoskin, 1957) have also described the *aufwuchs* community as that assemblage of organisms growing attached to, or clinging upon, free surfaces of submerged objects such as stems, roots and leaves of living plants, rock, wood and animals. Recognizing that several terms have become inter-changeable, Newman and McIntosh (1989) have cautioned that use of the periphyton community, in monitoring responses to contamination, requires proper characterization and "procedurally-defining" both the biotic and abiotic components. In this study the algal component of the total periphyton sampled (on both natural and artificial substrata) will be reported upon and, thus, includes in addition to the above the viruses, bacteria, decomposers and other microbiota (and sediment and detritus), important in

modifying nutrient flow through the community between the microbial (see reviews by Stockner, 1987, 1988, 1991) and algal constituents.

In addition to microalgal quantity (biomass), the biochemical expression (Steinman *et al.*, 1987b) of algae (proximate composition and nutritional profile) can be influenced by nutrients (Thompson *et al.*, 1990; Chelf, 1990) and herbivory (Steinman *et al.*, 1987a). It is surprising that so few studies exist using natural periphyton communities to test the linkage between fluctuations in ambient nutrient concentrations and their effects on structural change (i.e. biochemical alteration) in both the short-term (physiological) and long-term (species composition)(Austin *et al.*, unpub.). Since the periphyton community is an important primary energy producer and subsequent energy transducer between trophic levels in lotic ecosystems (Sinsabaugh *et al.*, 1991; Wetzel, 1983), alterations in periphyton structure and function may have important implications for higher trophic levels, including invertebrate herbivore, larval and fish production. Thus, one rationale for understanding aquatic biotic response to environmental stresses result from the need to protect ecosystems from undesirable human–non-human interactions.

Knowledge of how pollutants (heavy metals, biocides, nutrients etc.) adversely stress aquatic communities is important to future freshwater management (Odum, 1985; Schindler, 1987; Barret, 1985; Risser, 1985). A central concept of such management must, therefore, be the understanding of basic wildlife responses to human induced perturbation through an examination of species-specific and community tolerance limits; this information can only be obtained whilst the sensitive organisms and habitats remain relatively unperturbed.

There is increasing recognition of the sensitivity of, and diminishing capacity of, freshwater systems to assimilate the complex array of both natural (nutrients) and non-natural (i.e. chlorine-based) compounds discharged into them. However, the effects of environmental disturbances, especially toxic substances, on most freshwater species are basically unknown, particularly at lower trophic levels. Even less well understood are sub-lethal or chronic effects of non-toxic substances, such as nutrients, which can deleteriously affect water and habitat quality. One aspect of managing freshwater habitat

has been the need to predict the assimilative capacity of a particular water body to nutrient enrichment. The difficulty in making such predictions is revealed in the literature in which a wide range of nutrient concentrations (for phosphorus) are known to have caused undesirable algal (both phytoplankton and periphyton) blooms, under both laboratory and field conditions:

| | | |
|---|---|---|
| Lab culture | < 5 µg/L | (Wong & Clark, 1978; Horner *et al.*,1983) |
| Field | > 20 µg/L | (Tilman & Kilham, 1976; Tilman, 1977; Bothwell, 1985) |
| Lake (Gt. Central Lk.) | > 10-30 µg/L | (Stockner & Shortreed, 1978; Perrin *et al..*,1987) |
| River (Thompson R.) | < 3 µg/L | (Bothwell, 1985; 1988) |

Thus, it would not appear likely that simple estimates of how a specific environment will respond to nutrient enrichment can be made in developing management protocols. Extrapolation from laboratory bench models to real time, natural conditions should, at best, be used to design the requisite field-based studies on which management models can be made.

Cullen (1990), commenting on the turbulent conflicts between water science and water management, outlines a series of prerequisites for improving management of aquatic resources. Cullen argues that, in particular, prediction is an essential aim in limnological research and the establishment of criteria by which results should be judged. Description and explanations are important preconditions but are not themselves sufficient endpoints. The increasing public concern about our deteriorating environment has resulted in a decline in public trust of traditional risk management practices (Cullen, 1990; Morgan, 1993). Since risk management is a process for evaluating values for use in policy, thus inadequate prioritizing of values can result in inappropriate policy. Hopefully, the maturing sub-discipline of risk management, together with increased public participation in the decision-making process, will reduce the confrontational tenor of past management processes.

The research reported here illustrates experiment-based understanding how stressors affect functional and structural properties of the microalgal component of periphyton communities. The benefit of understanding how to manipulate this complex community (Newman & McIntosh, 1989) cannot be revealed without adequate standardized sampling and processing methods (Aloi, 1990). A surrogate stream or semi-natural stream trough system (flow-through surrogate stream troughs or flumes; Bothwell, 1983; 1985) was developed to provide a standardized method using surrogate stream mesocosms, to experiment on their contained periphyton, under controlled *in-situ* conditions. It was hoped that the surrogate streams would provide for i) *long-term sensitive monitoring for detecting effects in ambient waters,* and, ii) *standardized protocols for biological toxicity tests and nutrient loading* (Day *et al.*, 1988).

One of the most pervasive aquatic pollutants is that of nutrient loading which threatens to exceed the assimilative capacity of many natural aquatic systems (Bothwell and Stockner, 1980; Steinman and McIntire, 1990; Stockner and Shortreed, 1985; Schindler, 1974, 1977, 1987). Hill *et al.*, (1992) and Rosemond *et al.*, (1993) indicate that a major focus for freshwater ecologists should be the elucidation of the relative importance of *bottom-up* (i.e. nutrient loading) vs *top-down* (herbivory, predation) processes in structuring aquatic communities. It is with just such concerns that our development of standard field-based methods, within a conceptual framework, have been directed toward understanding the integrating role of lentic (flowing waters) periphyton with other ecosystem processes (Sand-Jensen and Borum, 1991).

Margalef (1981) noted that "Stress is something that puts into action the mechanism of homeostasis", which Odum (1985) and Schindler (1987) have applied in a postulate for aquatic communities which when subjected to acute, low-level stressors induce structural (species composition and architecture) disequilibrium but retain, or conserve, functional (biomass, biochemistry and productivity) homeostasis. This appears to contradict Stockner & Shortreed (1978) and Perrin *et al..*, (1987) and Meyer and Likens (1979) observations. These workers determined that adding substantial concentrations of orthophosphorus, in *in situ* phosphorus cycling

experiments, was required to cause detectable community alterations. In contrast, Riber and Wetzel (1987) demonstrated that recycling of phosphorus within the periphyton community accounted for the majority of phosphorus turnover. They predicted, therefore, that only small amounts of external phosphorus would be required for increased biomass accrual. Using contained periphyton in the surrogate streams, I proposed to experimentally test Odum's hypothesis that, at near, or below, minimum chemically detectable concentrations (<1.0 μg/L)(i.e. low level), chronic additions of orthophosphorus should not increase biomass accrual; that is, functional equilibrium is retained. Should our results confirm Odum's hypothesis, they would refute the findings of Riber and Wetzel. Understanding how stress alters periphyton structure (species composition and architecture) and function (biomass and biochemistry) may have relevance for the health and survival of juvenile salmonids (Hyatt and Stockner, 1985; Perrin *et al.*, 1987; Mundie *et al.*, 1991), important in our bio-region, and for aesthetic considerations (Nordin, 1984). Understanding functional and structural responses of periphyton to nutrient enrichment also has important implications for determining management protocols when determining maximum permissible discharge of nutrient enriched effluent into freshwater receiving environments. Thus, the information on periphyton accrual development patterns could be used to determine water quality criteria.

## Historical perspective

### Periphyton as a Bioassay

Stokes (1984) and others (Steinman *et al.*, 1987; McIntire and Colby, 1978) have demonstrated that ecological modeling based upon experimental microcosms can provide realistic predictions for field situations. The use of algal communities (for a review see Shubert, 1984), contained within such enclosures, are advocated because, in part, these organisms have short life-cycles, are easily dispersed and usually survive as resting stages under unfavorable conditions, thereby providing a source of 'reinvading' species when conditions once again become favorable. Since the majority of freshwater algae are microscopic, the high surface-to-volume ratios are such that physiological response times are short. At higher trophic levels, greater complexity often renders the same approach more complex and less useful. Periphyton communities are useful for analyzing processes of post-disturbance resilience, resistance and recovery because they are ubiquitous, relatively easy to sample and measure, contain species with short generation times, permitting monitoring of multiple generations through several successional seres (Steinman and McIntire, 1990); a number of characteristics are listed in Table 1 which identify why the algal component of periphyton make useful bioassay organisms. Since this community is attached and more easily manipulated than plankton, studies of environmental factors which interact with nutrients to affect the communities successional trajectories are made possible. Environmental factors which affect seasonal trajectories and can be studied using periphyton include: irradiance (Steinman and McIntire, 1987), current velocity (Horner *et al.*, 1990; Steinman and McIntire, 1987), substratum (Goldsborough and Hickman, 1990) and endogenous nutrient re-cycling (Dodds, 1991a,b; McCormick and Stevenson, 1991). Surprisingly few studies, however, report on nutrient flux as a function of velocity.

In an effort to further our assessment and understanding of biotic responses to perturbed conditions affecting water quality, studies have generally been approached as either field-based observations or laboratory-based manipulations within artificial microcosms. A fragmented literature exists

describing methods for studying the periphyton of both lentic and lotic systems (Wetzel, 1983a; Cairns, 1982; Austin *et al.*, 1981; Weitzel, 1979a,b; Beak *et al.*, 1973; Gale and Thompson, 1974; Bothwell, 1983); the majority of this research has traditionally consisted of qualitative observations (Bothwell, 1983; Lock *et al.*, 1990. ).

Many authors have advocated ecosystem studies (Stockner, 1991; Wetzel, 1991), however, to precisely measure which parameters structure aquatic ecosystems requires development of field-based protocols, in which the environmental parameters themselves are manipulated. An early technique of introducing limited control, whilst retaining the realism of the natural environment, was to introduce artificial substrata into lotic habitats. The periphyton community, colonizing the substrata, could then be disturbed, transplanted from one environ to another and sampled and examined at selected junctures (Cairns, 1982). Preliminary studies sought to identify factors limiting or modifying the usefulness of biofilm development (Newman and McIntosh, 1989) under two general categories - methods and environmental. Methodological studies included artificial vs natural substrata composition (Tippet, 1970; Gale *et al.*, 1979; Blinn *et al.*, 1980), loss and removal of organisms upon substrata retrieval (Lucey *et al.*, 1987; Gale, 1975; Ertl, 1971), and comparisons of sampling methods (Bott *et al.*, 1978). Studies of environmental variables affecting periphyton community development included light, nutrient species and concentration, velocity, effects of grazer type and density, and biofilm community structure on artificial and natural substrata (Gale *et al.*, 1979; Fontaine and Nigh, 1983; Cox 1988; D'Angelo *et al.*, 1992; Elwood *et al.*, 1981). In a caution to periphyton investigators, however, Weitzel (1979b), employing replicated side-by-side experiments, observed that periphyton productivity could be as significantly affected by field site location and sampler design as by ambient water quality. Another problem common to the early, field-based studies noted above was they did not permit control of the aquatic environment itself, or those biotic and abiotic terrestrial features modifying them. Surrogate streams, however, as a type of mesocosm (Odum and Hoskin 1957), do provide control over both the environmental factors and study methods.

## Experimental Environments

Laboratory conducted simulations of ecosystems, designed to mimic or model natural ecosystem dynamics, typically employ artificial environments in which water is static (bottle bioassay; Wong and Beaver, 1980; 1981; Wong *et al.*, (1978)) or recirculated (artificial streams) under rigidly controlled conditions (McIntire, 1966a,b; 1968; 1973; McIntire *et al.*, 1964; Cooke, 1971; Honig and Buikema, 1980; Steinman *et al.*, 1987a). A number of laboratory systems use a flow-through design at or adjacent to the environment actually being tested (Table 2). Whilst such studies often minimally describe qualitatively and quantitatively how aquatic wildlife might respond under natural conditions (Cairns, 1980, 1986) to the transport, fate and effects of episodic or continuing disturbances (Wevers *et al.*, 1988), there is increasing interest in using *in-situ* microscopic wildlife as subtle biological indicators of changing water quality. Since natural systems usually are not accessible for experimental manipulation (Mills and Schindler, 1989), an evolving option is to use semi-natural systems such as surrogate streams established in the wild. These streams are subject to the natural climatic (seasonal) and weather (daily) characteristics which affect both the wild and contained aquatic communities under investigation (Stockner and Shortreed, 1978; Bothwell and Stockner, 1980; Bothwell and Jasper, 1983; Bothwell, 1983; 1989).

A review of the literature reveals that in attempting to extrapolate data, to natural aquatic environments (Dudzik *et al.*, 1979), obtained from laboratory-based artificial micro- or mesocosms, certain conceptual constraints of this method must be addressed. Artificial microcosms, however, established within natural systems (i.e. semi-natural), combine some of the characteristics of both. Semi-natural microcosms offer enhanced control of physical (light, temperature, velocity) and chemical (pH, nutrient profiles, toxin concentrations) parameters, multi-variable manipulations (nutrient flux), recruitment, and the role of substrata in establishing the contained community architecture. These systems also ensure that the natural dynamic nature of environmental parameters remain relatively undisturbed within the controlled environment. A brief review of the literature describing a variety of stream (or flume) systems is presented in Table 2, together with the

degree of environmental realism, scale and authors research objectives. The surrogate streams described here were developed to provide a standardized methodology (of flow-through stream troughs; Bothwell, 1983; 1985), for experimenting on contained periphyton, under controlled *in-situ* conditions; in the mild winters of our local year-round operations are possible.

The system described here has a number of unique attributes in its favour, especially employment of in-stream hydraulic energy, obviating any need for electrical power. Within B.C. there are more than 2000 storage and diversion works creating impounded lakes or reservoirs (B.C. Annual Report, 1989/90) exclusive of waterfalls or steep stream slope gradients which also provide numerous possible study sites. Siting the original surrogate streams at one of the Greater Victoria Water District (G.V.W.D.) water-supply reservoirs (Humpback)(Figures 1 and 2), supplying potable water to the Capital Regional District on Southern Vancouver Island, was based on a number of characteristics considered ideal for research including:

i) reservoir of high quality oligotrophic water, with a relatively high flushing rate; the non-stratified nature of the impoundment is appropriate for the system requirements;

ii) energy potential of site situated below impoundment, permitting use of siphoned water to continuously fill and flush the streams, *thus avoiding costly and unpredictable pumps* and their electrical requirements;

iii) a suitable drainage system to receive waste water overflow from experimental operations;

iv) suitable nearby protected creeks, as well as more distant problem localities, in which to evaluate portability potential of the system;

v) recent and ongoing limnological assessment of this in-line, surge reservoir and its major source of relatively unperturbed supply (Sooke Reservoir);

vi) close proximity to University laboratory facilities to minimize travel and servicing costs;

vii) restricted public access and opportunity for vandalism; site routinely patrolled by water board employees;

viii) proximity of residence and chlorination facility provides shelter, storage and electrical outlets for use of instrumentation, microscopes, etc.

Using the surrogate stream mesocosm complex to develop an integrative biological data base, together with physical and chemical water quality criteria, would enable environmental managers, and regulatory agencies, to potentially:

i) respond rapidly to queries regarding possible perturbations or stressors;

ii) facilitate the prediction of further consequences of such perturbations along the food chain to culturally important organisms, including fish;

iii) predict downstream effects originating in upstream drainage basins receiving effluents;

iv) enhance recovery or rehabilitation of damaged habitats by using appropriate biological replacement procedures and seed communities;

v) make predictive correlations between physical and chemical measurements and biological consequences;

vi) establish relative indices of aquatic assimilative or tolerance capacities.

Concept, design and system installation of the first microcosm complex (reported on under the acronym INFLEX - INField Laboratory EXperiment) was begun in 1982 and has been in operation since 1984 (Austin and Lucey, 1984). Similar systems were subsequently installed:

i) at a trout hatchery, using pumps to create a recirculating system to effect water renovation through periphyton-based nutrient removal,

ii) on the Cheakamus and Squamish Rivers (the latter also required the use of pumped water to effect a flow-through system) at sites downstream from a sewage treatment plant to experimentally determine periphyton sensitivity to phosphorus loading (Lucey *et al.*, 1993),

iii) within both forested and harvested sites, on primary streams supplying the principal reservoir (Sooke) of the G.V.W.D., to measure land-use induced alterations in below chemically-detectable nutrient concentrations within the streams.

In the subsequent chapter the original site location and basic system design is described, followed by a chapter on the initial nutrient loading experiments conducted to determine the systems bioassay capability. The fourth chapter describes our use of the stream troughs to determine the assimilative capacity

of the Cheakamus and Squamish Rivers, and their sensitivity to orthophosphorus (especially at concentrations below 1 μg/L). The latter study also sought to determine the water quality and habitat implications that enhanced discharge of orthophosphorus could have on downstream periphyton accrual.

Table. 1. Characterization of periphyton, especially the algal component, as a community-level bioassay.

| Characteristic | Author |
|---|---|
| contain simple, predominantly single-celled, microscopic organisms; high surface area to volume ratio (note: this author refers to diatoms) | Dixit *et al.*, 1992 |
| quantify rate of degradation/recovery in water quality; high production rates | Dixit *et al.*, 1992 |
| organisms cosmopolitan | Dixit *et al.*, 1992 |
| furnish data on background (reference) water quality & natural variablity | Dixit *et al.*, 1992 |
| supply high-quality food for higher trophic levels | Dixit *et al.*, 1992 |
| communities are ubiquitous, relatively easy to sample and measure | Stokes 1984; Steinman & McIntire 1990 |
| contain species whose generation times are short (replicate rapidly); physiological or genetic changes to toxic exposure relatively rapid; stratified age structure | Cairns 1982 |
| predominantly asexual or vegetative modes of reproduction; flexible life history strategies | Stokes 1984; Steinman & McInite 1990 |
| permit monitoring of multiple generations through alternate successional seres | Stokes 1984; Steinman & McIntire 1990 |
| algae are easily dispersed; migrate rapidly | Stokes 1984; Dixit *et al.*, 1992 |

| | |
|---|---|
| usually survive as resting stages under unfavourable conditions | Stokes 1984 |
| provide a source of 'reinvading' species when conditions once again become favourable | Stokes 1984 |
| post-disturbance recovery of higher trophic levels may depend on periphyton's high-quality proximate composition | Steinman & McIntire 1990; Ridley-thomas 1989; Austin *et al.*, 1991 |
| relatively easy to determine species end-points (death; growth inhibition and/or stimulation); monitor changes in physiology (respiration, photosynthetic rate), biochemical changes (proximate composition), ecological changes (community structure, dominance), species growth rate changes (abundance), histology; | Walsh and Merrill 1984 |
| Relative ease of sampling this aggregated community permits preliminary estimates of population density variance, to determine degree of replication | Morin 1985 |
| Meets requirement of two or more trophic levels of organisms with more than one species at each level;<br><br>community is capable of supporting populations at their natural levels through at least one life cycle of the highest trophic level organism contained;<br><br>capable of replicating manipulated and control communities;<br><br>capable of removing organisms without disturbing the populations | Cairns 71, 74, 77, 1980; Lucey *et al.*, 1986; Lucey *et al.*, 1987; Deniseger *et al.*, 1986 |
| rare species and common species equally likely to exhibit opportunistic and conservative strategies - permits assessment of competition amongst species (alteration of available resources or via direct effect) | Lewis 1977; Walsh and Merrill 1984 |

| | |
|---|---|
| differentiate between species toxicological sensitivity to single or coctail mixtures | Walsh and Merrill 1984 |
| biota continuously integrate environmental variables, thus requiring less frequent sampling to assess ecological alterations than physical or chemical parameters | Dixit *et al.*, 1992 |
| analysis requires small samples; many samples can be relatively easily collected, transported processed and archived | Dixit *et al.*, 1992 |
| relative ease of obtaining large numbers of individuals, of numerous species, permits robust statistical and multivariate procedures for assemblage analysis | Dixit *et al.*, 1992 |
| community will develop readily on artificial sampling substrata, especially members of the Bacillariophyceae (diatoms); community contains species common to other aquatic communities (planktonic) | Snoeijs 1991; Dixit *et al.*, 1992; Lucey *et al.*, 1986 |
| energy flow; define chemical-physical pathways and partitioning (abiotic and biotic nutrient cycling); boundary/ sediment layer flux | Cairns 1986; Raven 1992 |
| permits assessment of functioning ecosystem with functional cybernetic or negative feedback loops | Cairns 1986 |
| occupies position between physical and chemical milieu and higher trophic levels | Stoermer 1978 |
| containable community for use as biomonitor; community contains elements potentially amenable to electronic automation | Cairns *et al.*, 1977 |

Table 2. Chronological sequence of literature referencing artificial stream system research. Streams range from small completely artificial systems (in lab, small scale enclosed) to those approximating natural streams (in field, large scale, under natural conditions). IL = in lab; IF = in field; RC = recirculating system; FT = flow through. Scale = stream morphology as length, width, depth, diameter.

| Author | Yr | Location | IL | IF | RC | FT | Scale | Research Objectives - Comments |
|---|---|---|---|---|---|---|---|---|
| Whitford *et al* | 60 | Raleigh | X | | X | | 80 cm(l) | current and growth |
| Whitford *et al* | 64 | Raleigh | X | | X | | 80 cm(l) | irradiance, current and growth |
| McIntire *et al* | 64 | Corvallis | X | | X | X | 3 m(l); 25 cm(w); 20 cm(d) | stream water; 2.0 l/sec; 24 cm/sec; parallel wooden troughs; primary production |
| Schumacher & Whitford | 65 | | X | | X | | 12 cm dia. | Modified Barcroft respirometer; 18 cm/sec; round flasks; respiration & P uptake |
| Phinney & McIntire | 65 | Corvallis | X | | | | 3 m(l); 25 cm(w); 20 cm(d) | |
| McIntire | 66 | Corvallis | X | | X | X | 3 m(l); 25 cm(w); 20 cm(d) | current velocity effects on periphytic biomass, AFDM; 90% replacement every 2 hrs (@ 2.0 l/min); 9 cm/sec & 38 cm/sec |
| Hemens & Mason | 68 | Pretoria, S.Africa | | X | | X | 3140 m(l); 10.2 cm(w); 12.7 cm(d) | rhomboid shaped (10.2 base, 48.3 cm top); 68 l/min, 24 hr retention; sewage nutrient removal by algae |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Rose & McIntire | 70 | Oregon | X | | X | X | 3 m(l); 30.5 cm(w); 16 cm(d) | current velocity, light intensity, species composition on accumulation of dieldrin residues; 28 cm/sec; stream water partially filtered; 4 hr residence time |
| Kehde & Wilhm | 72 | Stillwater, Oklahoma | X | | X | | 2.4 m(l); 0.15 m(w); 0.07 m(d) | stream water inoculum; 8 cm/sec; N+P enriched; effects of snail grazing on periphyton community structure |
| Benoit & Puglisi | 73 | EPA, Duluth | X | | | | | simple flow-splitting chamber and siphon for proportional diluters |
| Ruthven & Cairns | 73 | Blacksburg | | | | X | 24.5 cm(l); 2.5 cm(w); 1.5 cm(d) | tap water charcoal filtered; protozoan response to metals; 100 ml/min |
| Cushing *et al* | 84 | Columbia River | | | | | | mineral cycling by periphyton |
| Maki & Johnson | 76 | E. Lansing | X | | | X | 8 m(l); 0.6 m(w); 25.4 cm(d) | gravity feed; creek water; stream vol of 550 l; 100 l/min; replacement time of 5.5 min; periphyton metabolism (net primary production and respiration) affected by lampricide TFM |
| Sigmon *et al* | 77 | Savannah River | | X | | X | 91.5 m(l); 0.61 m(w); 0.31 m(d) | 0.8-0.9 hl/min; pumped groundwater; periphytic response to low levels of mercuric ions |
| Gerhart *et al* | 77 | Duluth | X | | X | X | 2.3 m(l); 10.0 cm(w); | 10-30 cm/sec; 1 l/min replacement; 57 min retention period; pumped lake water; coal leachates on periphyton community structure |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Eichenberger & Schlatter | 78 | | | X | | X | 75 m(l); 0.2 m(w); 0.2 m(d) | ground water; 1 l/sec; herbivorous insects grazing on periphyton production |
| Stockner & Shortreed | 78 | Carnation Ck. | | X | | X | 4.25 m(l); 9.0 cm(w); 2.0 cm(d) | nutrient enrichment and autotrophic production |
| Eichenberger | 79 | Zurich | | X | | X | 65 m(l); 20 cm(w); 20 cm(d) | ground water; 1.5 l/sec; 15 cm/sec; 9 parrellel channels; asbestos-cement; nutrients, essential metals |
| Honig & Buikema | 80 | Blacksburg | X | | | X | 91.4 cm(l); 20.3 cm(w); 15.2 cm(d) | de-chlorinated tap water; structural attributes (AFDM, dry weight, Chl *a*, ATP); artificial refinery mixture |
| Clark *et al* | 80 | Blacksburg | | X | | X | 4 m(l); 39 cm(w); 35 cm(d) | pumped river water; 30 l/sec; 9 cm/sec; 3-5 cm deep water; 53% replacement @ min; 6 stream complex; reviewed use of artificial streams |
| Hemmer | 80 | Florida | X | | | | | pressurized proportional diluter; electric air pump |
| Garton | 80 | EPA, Corvallis | X | | | | | electric pump driven toxicants to header box for delivery to test tanks |
| Ahlf & Weber | 81 | Elbe, Hamburg, | | X | | X | 10.5 cm(l); 19 cm(dia) | in-site bioassay; heavy metal toxicity |
| Sumner & McIntire | 82 | Corvallis | X | | X | X | 3 m(l); 8 cm(w); 8 cm(d) | stream water; 10 cm/sec; 0.75 l/min exchange rate; grazer-periphyton interactions |
| Welton *et al* | 82 | Dorest, Eng | X | | X | | | sampler comparison; benthic invertebrates; trophic relationships |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Krewer & Holm | 82 | Athens, Georgia | X | | | X | 2.4 m(l); 46 cm(w); 51 cm(d) | 8 linked, contiguous tanks; well water; 500 l/day; 12 h/tank retention time; total dissolved phosphorus/ chl *a* relationship |
| Wilde | 82 | Aiken | X | | | | | 81 sec retention time; periphytic response to thermal pollution |
| Bothwell | 83 | Thompson River | X | X | | X | 2.0 m(l); 19 cm(w); 5 cm(d) | pumped river water; 47 l/sec; 50 cm/sec; 0.8 cm depth; 60 sec system retention; 4 sec trough residence time; entire system enclosed; troughs covered in transparent glass, one with plywood (dark); net biomass accrual; settlement rates |
| Meier & Dilks | 84 | Huron River | | X | | X | 110 m(l); 18.3 cm(w); 15.2 cm(d) | pumped river water; periphytic oxygen production |
| Kersting | 84 | Leersum | X | | X | | | 160 l vol; pond water, artificial media; autotrophs, herbivors, decomposers; algal toxicity; herbicides; |
| Pringle | 87 | Carp Creek, Michigan | | X | | X | 50 cm(l); 14.5 cm (dia) | 18-20 cm/s; nutrient dosing-substrata interactions; |
| D'Angelo *et al* | 91 | Coweeta, North Carolina | | X | | X | 20 m(l); 20 cm (w) | 0.25 L/s; gravel bottom; regulated headbox feed; covered with shade cloth (20% hv); leaf decomposition; toxic input; nutrient dosing |

# Chapter II. Periphyton Functional and Structural Response, within Surrogate Streams, to Artificially Induced Water Quality Perturbations

## Introduction

The studies reported here were designed as part of a larger exploration of how environmental parameters affect water quality and aquatic habitat and are reflected in changes in the functional and structural characteristics of the periphytic community. The surrogate streams were designed to experimentally test a number of hypotheses in a field-based setting. Whilst the findings reported here concern the role of nutrient enrichment induced alterations to the community, two other aspects of the broader study have previously been reported. Biochemical profiles of nutrient induced changes to the periphyton have been described (Ridley-Thomas (1989); Ridley-Thomas *et al.*, (1989a,b); Ridley-Thomas *et al.*, (1990)), together with ecotoxicological alterations (heavy metals)(Ros, 1989) and the broadly used biocide Glyphosate® (Austin *et al.*, 1990).

To determine whether the periphyton community contained in the streams could respond to subtle alterations in physical and chemical parameters, the preliminary study, conducted at the Humpback Reservoir site, consisted of a series of experiments (1) at different water velocities, and 2) under phosphorus and nitrogen enrichment. These experiments were repeated at different seasons and the nutrient enrichment was also conducted under reduced irradiance. The study program is outlined in Table 4 (and in the methods section), which also summarizes weather data (mean maximum and minimum temperatures, total precipitation and mean daily sunshine). The first four experiments were conducted between the autumn of 1987 and summer of 1988. The fourth experiment was abruptly terminated following the application, by the G.V.W.D., of copper sulphate to an algal bloom in Sooke Reservoir; the algicide killed the mesocosm contained periphyton.

Additional experiments were conducted during the summer and autumn of 1988 to examine periphyton accrual on different artificial substrata. The last experiment was conducted during the autumn and winter of 1989 and

explored the toxicity of the herbicide Glyphosate® to periphyton (Austin *et al.*, 1991). It was determined that at low concentrations this compound can enhance biomass accrual, through increased phosphorus loading, an observation not previously reported in the literature.

The next suite of experiments reported here (Lucey *et al.*, 1993) explored the use of surrogate streams to determine whether, in the Cheakamus and Squamish Rivers, orthophosphorus was the nutrient limiting periphyton accrual. Also tested was whether a correlation existed between increased periphyton biomass accrual and orthophosphorus. This study was initiated at the request of the Ministry of Environment, during the fall of 1987, and was conducted concomitantly with the studies at Humpback. These simultaneous studies enabled us to explore whether the contained periphyton in the widely separated streams would respond in a similar manner. The four year field study (1988-1991), on the Cheakamus River, used the surrogate streams as a supplement to a whole-river experiment and water quality sampling regimen. The program of study sought to quantify the nutrient contribution of sewage effluent (with particular reference to phosphorus), from the resort municipality of Whistler's sewage treatment plant (STP), into the adjacent receiving environment (Cheakamus River) and to determine whether enhanced algal accrual (16km) downstream was a result of the STP discharge. This study is described in Chapter three.

## Methods

### Stream Trough Design, Construction and Operation

#### System Components - historical overview

Laboratory and semi-natural surrogate streams described in the literature (Table 2) commonly consist of four structural components - i) a water delivery system (header box), ii) a manifold supplying water and additives (mixing boxes) to, iii) a series of "riverine-containment" devices (stream troughs) discharging into iv) an exhaust or water collection (drainage) system (Bothwell, 1983; Honig and Buikema, 1980; Stockner and Shortreed, 1978; Garton, 1980). The majority of the reported systems, however, use electrical pumps to supply water for the troughs. The system described here uses water siphoned from an impoundment to provide a continuous supply of water thus requiring no electrical power for its operation. The following sections outline in detail the four components (header box and mixing boxes, troughs and drainage system) and modifications designed, constructed and field-tested during nine years of continuous operation. The system uses PVC as the surrogate stream bed throughout; this material is easily machined and fabricated, readily available in standard configurations, light weight and easily transported, relatively inexpensive, cleans quickly and can be easily repaired in the field.

#### Site and Water Supply

#### Greater Victoria Water District Watershed

The prototype streams were sited at the downstream end of the Greater Victoria Water District (G.V.W.D.) water supply catchment basins; the primary reservoir for which is Sooke, which discharges its water into a tunnel, whose flow is diverted into two surge reservoirs - Japan Gulch and Humpback (Figure 1). The surrogate stream facility is located adjacent to Humpback dam (Figure 2), which, periodically, also receives water from reservoirs other than Sooke. The G.V.W.D. catchment basins lie within the Insular and coastal Mountain Limnological region, on south-eastern Vancouver Island. The region is subjected to the moderating influence of the

Japanese current (which warms coastal air masses), producing year round mild temperatures, which permitted all-weather operation (see Bothwell, 1983 - design for harsh weather conditions). Regional mountain ranges protect the watersheds from the prevailing winds with the resulting light precipitation occurring in winter.

The principal sub-catchment basins supplying the Sooke Reservoir lie within the Very Dry Maritime Coastal Western Hemlock Biogeoclimatic subzone (CWHxm)(Meidinger and Pojar, 1991), within which can be found numerous ecosystem types (McKean and Munteanu, 1981). These include salal-lichen complexes on exposed rock; well-drained areas dominated by Douglas Fir, Garry Oak, Arbutus and Red cedars; and, low lying wetlands comprising willow, skunk cabbage, *Spirea* spp., sedges and grasses. Whilst a detailed description of the forest ecosystem containing G.V.W.D. catchment basins can be found in Meidinger and Pojar (1991), the following is a brief précis of the Krajina Biogeoclimatic classification of this region:

> Climate, csb, Cfb; day degrees over 6 °C, 2500 to 3500; frost free days, 150 to 250; mean annual temperature, 9 to 11 °C; January mean temperature, 1 to 4 °C; July mean temperature, 16 to 19 °C; 5 to 7 months above 10 °C; 0 months below 0 °C; snowfall to 25 to 107 cm is 3 to 10% of total precipitation; driest month precipitation, 1.5 to 4.8 cm; wettest month precipitation, 12.7 to 26.4 cm; clouds common in winter, rare in summer; elevation: S.E. Vancouver Island, 0 to 450 meters, latitude, 48° to 50°20"N (in the rain shadow of Vancouver Island mountains).

The Carboniferous and Devonian bedrock is principally comprised of "Metchosin volcanic" materials which include basalt flows, tuffs and agglomerates with intrusive diabase dykes. Subsequent intrusions of batholitic diorite, "Saanich" granodiorite and quartz diorite, and Sicker porphyrites invasively deformed these early parental materials. Pre-Pleistocene erosion processes reduced this maturing surface to a coastal lowland plain. The unmetamorphosed sedimentary Meso–Lower Cenozoic age formations typical of the northern edge of the G.V.W.D. watersheds, have been provisionally termed the Cowichan group whose lithological characters consist of conglomerates, sandstones and shales. Distinctly calcareous rocks are rare. Interspersed within these areas are volcanic bedrocks associated with

coarse textured glacial till (gravely, sandy loam) and occasional shallow, well decomposed, organic deposits. Adjacent to some of the lakes, in the complex of watersheds, can be found low lying areas having a moderate susceptibility to ponding and flooding, in which marshes, bogs and fens have developed. Meidinger and Pojar (1991) have briefly described the soil profile found in this subzone, which typically consist of frequently acidic Mors, which contain a limited pool of nutrients; the acidic parent materials produce a clay mineral and nutrient poor soil.

The overall effect of this biogeography on water quality is to yield a year-round water supply which is slightly acidic (pH 6.0 to 7.0) and nutrient poor; these waters have a hardness value ($CaCO_3$) which ranges between 14 and 25 (mg/L), resulting in vulnerability to metallic toxicity (Clark and Morrison, 1982). With the exception of mercury and cobalt, concentrations of organic and non-organic compounds, ions and metals were found to be within the 'Objective Concentrations' established for Canadian Drinking Water Quality (Ont. Min. Environ., 1978). Mercury levels in Humpback Reservoir, although occasionally above objective concentrations, were below maximum acceptable concentrations. (A more detailed assessment of the water quality of the principal reservoir - Sooke - can be found in Lucey *et al.*, 1984 and Lang and Austin, 1984.) Since the project began, considerable emphasis has been placed on describing the water quality of the numerous reservoirs, lakes and streams comprising the G.V.W.D. catchment basins (Lucey and Austin, unpubl. data; Hetherington *et al.*, 1993).

Odour or obnoxious taste causing organisms (eg, *Peridinium* sp., *Ceratium hirundinella*, *Trachelomonas* sp., *Cryptomonas erosa*) are not typically observed, nor are potentially toxic organisms (eg, *Microcystis* sp.) in other than insignificant quantities (Lucey and Austin, unpubl. data). The single occurrence, in recent times, of the presence of large numbers of taste and odour causing organisms occurred during the summer of 1988 and 1989 following an inter-basin water transfer from Deception into Sooke Reservoir. Species sensitive to low concentrations of heavy metals such as copper, zinc and lead (*Dinobryon divergens* Imhof and *Asterionella formosa* Hassall) are frequently observed in abundance. The water quality implications of the

above indicate the physico/chemical and biological characteristics of the source water supplying the surrogate streams (principally Sooke Reservoir) are typical of oligotrophic waters within the Insular and Coastal mountain limnological region, not previously perturbed by industrial, commercial or urban activities.

### Sooke Reservoir

Sooke Reservoir is (1988) 7.26 kilometers long with a maximum width of 1.47 kilometers and a main axis running NNE. It is located 32km northwest of the Capital Regional District (Figure 1), British Columbia, Canada (123°42'W, 48°33'N) and having an area of 593 ha, the reservoir is one of the larger water bodies on Vancouver Island. The reservoir has very low turbidity and apparent colour values (Lang and Austin, 1984; Lucey and Austin, unpub. data) and Secchi Disk depths (6 - 10 m) typical of oligotrophic waters. The reservoir exhibits a spring (end of April to early May) thermocline, which generally stabilizes at approximately 12-20m during August and September. Hypolimnetic and epilimnetic waters temperatures range from 6-8 and 16-20 °C, respectively. These water masses become well mixed during November and December, characteristic of warm, monomictic lakes. As a result of the mild climate, minimal ice formation occurs during the winter.

The reservoir is comprised of two major basins of undetermined interaction. The maximum depth is 68m and depth decreases from north to south. The major inflow is Rithet creek, with lesser input from Whiskey and Judge Creeks and Horton, Deer and Begbie Lakes. Table 3 lists selected morphometric parameters of the Sooke Reservoir (after Lang and Austin, 1984; Lucey *et al.*, 1984; Brown and Austin, 1971).

### Humpback Reservoir

Humpback reservoir lies 25 Km. northwest of Victoria, British Columbia, Canada (123°42'W, 48°33'N). Humpback reservoir's quality water reflects that of its source (Sooke Reservoir), exhibiting high clarity and Secchi Disk values (6 m mean annual depth), low total hardness, total alkalinity, macronutrients (including total phosphate, nitrate, nitrogen and silica) and a pH range between 6.3 and 7.5 (Lucey and Austin, 1984)(Table 4 provides a brief selection

of chemical parameters measured during the experimental period). A more complete assessment of the water quality can be found in Lang and Austin (1984).

### Surrogate Streams

The surrogate stream trough design, together with standard operation procedures, is described in detail in APPENDIX A. The following is a brief description of the system. Water is siphoned over the dam face from three depths (0.5, 1.0 and 1.5 m) into a large header box. A constant head of pressure within the box is maintained by ensuring inflow exceeds discharge to troughs, with the excess flow discharged through an overflow standpipe (Figure 3). Water to the troughs is metered through a valve which discharges into a manifold distribution system. Each manifold supplies two, paired troughs. The 12 individual troughs were supported on three plywood tables (two pairs per table). The troughs consist of PVC rectangular boxes, whose sloped sides minimize shading. Artificial sample substrata are placed at intervals along the trough bottom. Effluent discharge, via a common drainage system, empties into an adjacent swamp, which flows into a small creek. This basic trough design has been used in all the studies reported in this thesis.

## Water Sampling

### Physical/Chemical Sampling

The following sections outline the materials and methods used throughout the studies except where a specific procedure was used, which is described in the respective chapter immediately prior to the discussion of results.

Chemical analyses of the water supply were limited by the expense of nutrient analyses by an outside laboratory. However, commercial laboratory analysis was required given the extremely low concentrations of N and P in the oligotrophic supply reservoirs (Table 5). During the study, reservoir water samples were obtained from the header box using 1.0 or 2.0 L polyethylene bottles supplied by the G.V.W.D. or the laboratory conducting the analysis (M.B. Research, Sidney, B.C.). Samples for analysis in our laboratory at the University of Victoria (pH and specific conductivity) were also collected in polyethylene bottles obtained from the same source; bottles were used only

once. The water samples were taken at 0.5 m depth in the header box, placed in a cooler with ice packs and taken to the laboratories and analysed the same day.

During the study period, additional physical and chemical analyses were made of water within each trough during each experimental trial. Parameters measured included flow rates, irradiance (surface and sub-surface), water temperature, dissolved oxygen, pH, conductivity, and oxidation-reduction-potential (ORP). These parameters were measured to assess possible intra- and inter-trough differences.

### Water Chemistry, Light and Temperature

Water chemistry samples were analyzed using the techniques outlined in Table 5. Specific conductivity and Oxydation Reduction Potential (ORP) were measured on room temperature water samples (20-22 °C) using a standard conductivity cell (HYDROLAB System 8000 Water Quality Instrument, with Hydrolab 8100 Series Water Quality Data Transmitter (Hydrolab Corp., P.O. Box 9406, Austin, Texas); pH was measured using an Ionalizer Model 407A Specific Ion Meter (Orion Research Inc., Cambridge, Mass.), with a Baxter Canlab probe. Standards used to calibrate the meters included pH: pH 4.01 (25 °C) potassium hydrogen phthalate (10.21 g/L), 7.00 (25 °C) sodium phosphate, dibasic (5.775 g/L) and 10.00 potassium phosphate, monobasic (3.538 g/L)(Fisher Scientific, Nepean, Ontario). ORP and Specific Conductance employed standards outlined in the Hydrolab manual. Sensitivity ranges of the chemical constituents measured are described in Table 5 (A.P.H.A. 1985).

Measurements taken within the surrogate streams included flow rates which were determined through the timed collection of ten replicate, paired stream discharges. Flow rates were remeasured each time the nutrient stock solutions were replenished (a minimum of once a week).

Irradiance values were obtained using a Biospherical Instruments Inc., Quantum Light Meter, Model QSL-100 (San Diego, California; Quanta $sec^{-1}$ $cm^{-2}$), to obtain a reading above the water and at the center of the sample substrata. Irradiance values were obtained at the inflow, middle and outflow sections of each stream. The values were consistently similar, thus fewer

irradiance values were taken during latter stages of the study. The location of a hill to the south-west of the facility frequently resulted in the abrupt formation of clouds which caused fluctuations in irradiance intensity at the site. The data were plotted as percentage differences between air and wet values, in an attempt to compensate for fluctuations in solar radiation.

Temperatures were obtained using the HYDROLAB, D.O. Meter and/or a glass encased thermometer. No temperature calibration was made, other than to place the probe in an ice-water bath, which yielded a value which ranged from 0.0 to 0.5 °C. Dissolved oxygen values were obtained using a YSI Model 57 Oxygen/Temperature meter (Serial number 1122, equipped with a YSI model 5739 probe and YSI membrane kit (Yellow Springs Instrument Company, Yellow Springs, Ohio); calibration used the air calibration procedure outlined in the operations manual. The instrument was calibrated before and after each measurement series; if a measurement series required > 30 minutes a mid-point calibration was conducted. Although the manufacturer claims a higher degree of accuracy, the range of error appeared to be in the order of $\pm$ 0.5 mg/L.

## Biological Sampling

### Microbial

Since preconditioning of substrata may affect periphyton colonization, an assessment of the microbiological community, attached to sample substrata, was undertaken. Borosilica slides were introduced into the streams on August 19, 1987 and removed on the 22nd; a second set was introduced (in the absence of nutrient enrichment) on December 8, 1987 and retrieved after 8 days. The two sample periods were conducted during the summer (high light and temperature) and winter (low light and temperature). Slides were taken randomly from the middle rack (mid-stream), placed in new glass jars containing water from the streams and transported to the lab on ice. Each sample was analyzed for both mucilage cover (percent adsorption on slides) and microbial identification (MB Research and Development Ltd., Sidney, B.C.). Sample analysis occurred December 16, 1987, within 8 hours of retrieval. In winter sampling the slides were taken from odd-numbered streams; velocities were mid: 1 and 3, low: 5 and 7 and high: 9 and 11. Where

possible, bacteria were identified to species and basic information concerning the organisms ecology and habitat described (Appendix D).

### Surrogate Stream Studies

A suite of experiments were conducted at the Humpback site and are summarized in Table 4. The following outlines each experiment including time of initiation, study duration, experimental setup. A complete detailed description is to be found in Ridley-Thomas (1989). The relatively large number of streams (for which physical, chemical and biological data was collected), together with laboratory preparation time in working up the data, precluded replication. It was recognized that it would be preferable to have at least three replicates for each treatment, however, this necessitated considerable additional cost for stream construction, field and lab sampling and preparation time; the required funds were not available. An examination of the literature revealed that this is a common problem with field-based programs. The lack of replication does not seriously impair the usefulness of conducting such studies (Pringle 1987)(Table 2).

### Velocity Experiment

Water velocity has been identified as an important environmental factor in defining periphyton community structure. Two water velocity studies were conducted in 1) the autumn, beginning on August 19, 1987), lasting 63 days, and 2) the winter, beginning on December 8, 1987, lasting 64 days. In each study water velocities of 7 (low), 14 (medium) and 28 (high) cm/second were used. Two stream pairs (four streams in total) were randomly selected for treatment at each of the three velocities. The autumn velocities were 1-4 (high), 5-8 (medium) and 9-12 (low) cm/s, whilst during the winter study they were 1-4 (low), 5-8 (medium) and 9-12 (high) cm/s. Sample substrata were harvested at weekly intervals and the periphyton analyzed for biomass, ash content and proximate composition. Species composition was determined for the last sample date only.

### Nutrient Enrichment Experiments

Nutrient enrichment studies were conducted (Table 4) during the summer and autumn of 1988. The first study, which began on May 17, 1988 (and ran

for 45 days), examined both nutrient enrichment and nutrient enrichment under reduced irradiance (33% of ambient). Two stream pairs were used as controls (one each for the two light regimens), one pair for phosphorus enrichment (under each light condition), one pair for phosphorus and nitrogen enrichment (under both light conditions) and the last pair for nitrogen enrichment under ambient light. Velocities were maintained at the intermediate (approximately 15 cm/sec) rate. Sample material was pooled from each stream pair.

### Artificial Sample Substrata Experiments

During the summer (beginning on July 4, 1988) and autumn (beginning on August 19, 1988) (Table 4) two experiments were conducted in which three different sample substrata were used, as with the above enrichment study. Natural river rock, glass slides and styrofoam were placed in the bottom of the streams and subsequently sampled for biomass accrual.

### Periphyton Sampling

Periphyton samples, for biomass accrual and species identification, were obtained from each stream by removing the substrata units (slide, etc.) to a 250 mL glass jar, taken to the laboratory, where the biomass was harvested by scraping with new, single edge razor blades and resuspended in 250 mL of distilled water. Aliquots were removed for each analytical procedure (gravimetric analysis, chlorophyll *a* and species composition). Ashing followed the method of Ridley-Thomas *et al.*, (1989). Chlorophyll *a* analysis followed the procedures outlined in Standard methods. The volume and surface area of the sample substrata and containment jars were recorded to permit estimation of biomass accumulation as a function of surface area.

Quantitative (numerical abundance (N.A.) and biovolume (Biov.)) and qualitative (species composition) estimates of the periphyton were determined using samples preserved (in the field) in a neutral-buffered (di-sodium tetraborate) formalin solution (approximately 2%). Qualitative algal systematics used a Carl Zeiss model 65317 Research Photoscope equipped with phase-contrast and apochromatic optics (maximum magnification of 1250x). Species identifications were determined wherever possible; those organisms

which defied accurate taxonomic identification were photographed and/or drawn, described and a unique identification code assigned. Additionally, Hyrax mounts were produced to facilitate diatom identification. In addition to taxonomic journal references, the following source works were utilized in identification: Hustedt (1930), Prescott (1962), Cleve-Euler (1968), Bourelly (1966, 1968, 1970), Patrick & Reimer (1966, 1975), Schoeman and Archibald (1976), Starmach (Tom 2, 4, 5, 6, 7, 10, 11, 14; 1964-1977), Ramanathan (1964), Randhawa (1959) and Foged (1981).

### Algal Numerical Abundance

Determinations of algal abundance were made on a Carl Zeiss inverted microscope using Utermöhl sedimentation chambers (Lund *et al.*, 1957). Simultaneously, a series of analyses of the phytoplankton communities of the G.V.W.D. lakes and reservoirs was conducted. The large number of repetitive analyses of oligotrophic waters with low population densities required development of a more efficient procedure for using 100 ml Utermöhl chambers (Appendix B).

Each water sample was gently shaken (36 standard fore-arm shakes)(Lucey *et al.*, 1986), the resuspended sample sub-sampled (25 mL) and stained (Lugol's), and allowed to settle for 24 hours. Organisms were enumerated using randomly selected fields (Sandgren & Robinson, 1984) to ensure accurate and verifiable relative abundance estimates. Twenty to forty microscope fields (10 fields replicated 2 or 4 times, using different aliquots obtained from the single field sample)(at a magnification of 200X) were examined and individual cell and colony numbers recorded. Relative numerical abundance's were converted to (i.e. standardized as) organisms per millilitre. The conversion formula is described in the subsequent chapter.

### Algal Biovolume

Biovolume estimates for each dominant organism (accounting for >10% of the biovolume) were determined. Biovolume estimates were calculated for each species using one of five "standard" shapes: square, rectangular box, cylinder, sphere or obvoid.

Although more than one approximation of some shapes are possible (i.e. obvoid), calculations were made on the basis of the following formula:

| | | |
|---|---|---|
| 1 | Cube | H x L x W |
| 2. | Cylinder | Pi x $R^2$ X H |
| 3. | Sphere | 4/3 X $\pi$ X $R^2$ |
| 4. | Ellipsoid | 4/3 X $\pi$ X [(L/2)(W/2)]2 |
| 5. | Rectangular box | H X L X W |
| 6. | Oblate | 4/3 X $\pi$ X [(L/2)(W/2)]2 |

where L = length, W = width, H = height, R = radius; it is recognized that there are other formulae for obvoid-like shapes. The ellipsoid is a prolate spheroid having rotation about a major axis (i.e. blimp shape); the oblate is an oblate spheroid having rotation about a minor axis (i.e. Smartie® candy). The formula used to calculate biovolume for each species used one of the above shapes which best approximated the shape of the organism (Wetzel and Likens, 1991).

Dimensions of individuals for each species were determined and based upon the mid-range value of dimensions provided in the literature used for taxonomic identification. Biovolume computations were expressed as cubic micrometers per millilitre ($\mu m^3$ /mL).

Biovolume data were compiled in a spreadsheet (Microsoft EXCEL) and the dominant organisms by biovolume plotted (>10%/mL), together with numerical abundance and biovolume estimates of the major taxonomic groupings by season.

### Cheakamus River/ Daisy Reservoir Experiments

Nutrient dosed streams (Bothwell and Jasper, 1983; McIntire *et al.*, 1964) were used to measure algal accrual at increasing nutrient concentrations. All streams received siphoned reservoir water (obtained from a depth of approximately 5-7 meters), using a flow-through velocity of 1.0 L/sec. This velocity was determined as being the flow most typical of shallow side-channels of the Lower Cheakamus River, in which the heaviest algal growth was observed. The gravity-feed nutrient-dosing apparatus included 20 L carboys (Pringle 1987) containing the stock solutions of nitrogen ($NaNO_3$),

phosphorus ($Na_2HPO_4$), or both, in conjunction with a drip feed system (Flexiflow® gravity gavage set (Ross Co., Ohio)). Stock N and P concentrations were added to de-ionized water in the carboys. Nutrients from each carboy were added to the streams at a constant flow (1.5 ml/min) and replenished weekly. Drip-feed rates were measured and maintained to ensure desired dosing concentrations. Reservoir nutrient volumes were measured to determine the quantity of nutrient discharged into each stream on a weekly basis. The 1988 nutrient-dosing regimen consisted of 3.1, 1.8, 1.04, 0.7 and 0.5 µg/L/sec orthophosphorus; 3.75:0.5 µg/L/sec N+P; and 4.25% N; nutrient enhancement was designed to bracket background concentrations (1.0 and 25 P and N, respectively). Although the drip feed mechanism could not provide precisely measured dosing concentrations, nutrient-dose rates were relatively consistent across time.

The 1989 nutrient-dosing study resulted in concentrations of 0.40, 0.42, 0.25, 0.12, 0.08 and 0.02 µg/L P and of 4.0% N:P; background concentrations were 1.0 and 25 µg/l, P and N, respectively. The 1991 study replicated the concentration of 0.05 µg/L/sec, mid-way between the previous years two lowest test concentrations. As before, control streams received only Daisy reservoir water; water was siphoned from the same depth as in the two previous years.

Sample substrata used included river rock (previously collected from the adjacent river and cleaned, dried and placed in the streams), styrofoam and glass slide surfaces. For river sites periphyton was collected from river rock surfaces by harvesting the algae by scraping the area within a measured, circular disk (Ertl, 1971). Effort was made to standardize substrata depth, solar angle, and water velocity.

Periphyton from ten circular disks (total of 166 $cm^2$) were obtained from each river site and pooled. Rocks were selected which visually appeared to have the greatest biomass accumulation, since one objective of the study was to determine the maximum (not mean) accrual at each site. At all river sites and within each pair of streams, rocks were selected and ten disks scraped and pooled together for biomass determinations. The large number of sample sites and water chemistry measurements rendered it impossible, within the

projects limited budget, to analyze each disk separately, which would have provided the replication necessary for statistical assessments.

All samples were split providing equal aliquots for measuring biomass and chlorophyll *a*. Additional samples (obtained in the same manner) were obtained for community analysis. Samples for species identifications were placed in small glass jars and preserved with formalin, as noted earlier.

Water chemistry and periphyton biomass assessments were conducted by the British Columbia Environmental Laboratory and Zenon Environmental Inc. Samples for low-level (<5.0 µg/L) nutrient determinations were collected in laboratory-prepared amber glass bottles, field-filtered (apparatus pre-washed in 10% hydrochloric acid), and analyzed in a dedicated sample stream. Field blanks were conducted in the same manner as above to monitor field techniques. Method detection levels (MDL's) of inorganic N and orthophosphorus were 5.0 and 1.0 µg/L, respectively.

## Results

### Humpback System

#### Physical and Chemical

The principal source of water for Humpback Reservoir is Sooke Reservoir, although water can be diverted from the Goldstream or Waugh Creek watersheds (Figure 1). Sooke water enters Humpback via the Kapoor Tunnel which discharges into Japan Gulch Reservoir and thence through a short flume into Humpback. Water from Sooke enters Humpback in the northern corner and flows out through the intake tower, located mid-way along the dam (Figure 3). As noted in the materials and methods, water for the surrogate streams was siphoned over the dam immediately adjacent to the intake tower and screening complex. At nominal flows of 1.0 L $s^{-1}$ the yearly through-put would be approximately $7.6 \times 10^8$ L $yr^{-1}$. The siphoned water (Figure 3) has been drawn from the reservoir throughout all periods of the year, including the coldest months (December through February), without any interruption in flows. Although ice has formed on the edges of the surrogate streams, it has not resulted in any disruption in flow through the header-box, supply manifolds or discharge pipes. Throughout the sampling period the quality of water withdrawn from Humpback was characterized as oligotrophic (Table 6); whilst the source water was predominantly phosphorus limited there were periods in late summer when the N:P ratio resulted in nitrogen limitation.

#### Light

Irradiance values in the streams (Figure 4; summer trial (Table 4)) were highest at the surface (air), and reduced below the surface; values were lower between the slides. The differences between the inlet and outlet were minimal, whilst there was an increased light attenuation between slides as biomass accrual increased. Differences between streams, in the absence of significant accrual, were represented by the normal fluctuations observed as clouds pass overhead. Within-stream fluctuations were often measured during partially cloudy periods, resulting in occasional differences between measurements made at inlet and outlet ends of the troughs (eg. Figure 4,

stream 9). The measurements taken between the substrata yielded irradiance values lower than those in the open water column, as a result of the filamentous canopy adhering to the substrata.

Mid-summer (July, 1988) irradiance patterns (Figure 5) were similar to those of late spring (May; Figure 4), in which values were highest at the surface and lowest between substrata; values in open water lay between the latter. Values obtained above the rock substrata were similar to those observed between the rocks. The values obtained above the white styrofoam were higher than those for the other substrata, but less than open water. Between-stream differences were less than variation in surface values. This variability was similar to that observed in May (Figure 4).

Although lower in absolute terms, winter (February, 1988) irradiance patterns within the streams were similar to those of the summer with surface values highest and sub-surface values similar at the inflow and outflow locations (Figure 6). Values obtained between the substrata were similar to those of the open water column. As during the summer, variability in surface irradiance values occurred during the measurement period, as a result of cloud cover. Irradiance values dropped from the beginning of the study period (Figure 6; far left, December 12) to the cessation of the experiment (far right, December 20), with a 400 to 500% reduction in surface irradiance. Absolute differences between surface and subsurface values were highest at onset of the study (July) and least at the end (February).

Spring values were mid-way between winter and summer irradiances, with similar patterns between surface and subsurface irradiance values (Figure 7). Increased variation existed between cloud cover and cloud-free days during this period.

### Temperatures

Temperatures measured in the streams (during the winter, summer or autumn) exhibited minimal differences either within streams (inflow, mid-stream and outflow) or between streams (Figure 8). Estimating the mean of the stream temperatures across time indicated a summer maximum in July, 1988 (>22°C) and a winter minimum in December, 1988 (<5°C).

### pH

Hydrogen ion concentrations measured in the streams (Figure 9) indicated minimal variability between streams, within any single sampling period. Variation across time was also minimal, with pH values generally remaining between 6.5 and 7.0. The single exception of a significant difference in pH occurred in stream 10 in September, 1988; this sample period also evidenced the greatest variability between streams.

### Conductivity

A trend was observed of increasing conductivity values from a summer (July & September) low to a winter (December) high (Figure 10). Values ranged from 0.04 to >0.08 $\mu S/cm^2$. Inter- and intra-stream variability was, in general, less than 10%.

### Dissolved Oxygen

Dissolved oxygen values exhibited minimal (<5%) inter-stream differences (Figure 11), during any sampling period. The highest dissolved oxygen values where measured during the winter (December and January; >14 mg/L) and lowest during the summer period (July and September; between 8 and 9 mg/L). Minimal differences (<5%) in dissolved oxygen concentrations were observed between stream inflows and outflows.

### Oxidation-Reduction-Potential

Oxidation-Reduction-Potential values indicated very little difference (<10%) between streams (Figure 12). The lowest values (<0.15) were measured during the winter (December), whilst the highest values (>0.2) were observed during the summer (September).

## Biological

### Microbial Fauna

Mucilage cover on the introduced substrata was evident after only two days. Mucilage consisted of an organic matrix, adsorbed onto the substrata, comprising a jelly-like secretion of complex polysaccharides, proteins and cellulose. Eleven dominant bacterial species were identified on the slides

placed in the stream troughs August 19, 1987 and retrieved August 22. A second set of substrata placed in the troughs (December 8, 1987, out eight days later) also resulted in extensive mucilage cover but a different bacterial community, exhibiting sixteen different organisms. Only one species, *Brevibacterium acetylicum*, was common to both sample periods.

### Periphyton

The percentage abundance of the major taxonomic groups contained within the streams (at three different water velocities - 28, 14 and 7 cm/sec), during the autumn (871021) are shown in Figure 13 (Table 4). Bacillariophyta were present and co-dominant in all cases (Figure 14) (as an understory community), ranging from 44 to 87 percent abundance. With the exception of streams 1 and 3, Chlorophytes (Figure 13) were also co-dominants, forming a floating canopy (Figure 15). Chlorophyte abundance proportions ranged from 7 to 40 percent (Figure 14). Streams 1 and 3 maintained a Cyanophyte community (of 29 and 47 percent, respectively), a group which was not abundant in any other streams.

An analysis of variance ( one-way Anova), comparing the total cell counts at the three different velocities, yielded a P of 0.6242, with an F-test value of 0.1426. The relatively large P indicates that there were no between-group effects, whilst the F-test suggests that there were some differences in between-groups variances. However, analysis of the raw data demonstrated that there was one stream which had an abnormally large number of Cyanophytes, so the Anova was rerun with this data removed. This produced a P of 0.1935 and an F test score of 0.7732, indicating a substantial between-groups effect, with little within-group variance. Analysis (with the outlier removed) of the group means and group standard deviation yielded values at the three velocities of: fast, 1813.0 and 839.2; medium, 2825.5 and 530.3; and slow, 2043.0 and 778.3, respectively. A comparison of the groups means indicated no significant pairwise differences (T of 2.262, at a rejection level of 0.05) among the means with using all the data. Removal of the Cyanophyte data did not alter this finding (critical T value of 2.306 at a rejection level of 0.05); standard errors and critical values of differences varied between comparisons given the unequal sample sizes ( 12 vs 11 cases).

During the autumn and winter velocity studies, analysis of the community structure indicated that biodiversity (expressed as major taxonomic groups) was greater in the autumn than in the winter (Figure 16). The autumn community consisted of a diatom understory (both stalked and non-stalked forms), with a Chlorophyte canopy (principally of filamentous forms). Cyanophytes were also present in reduced numbers, compared with the other major groups. The composition of the autumn community indicates that as velocity increased there was a corresponding increase in both filamentous Chlorophytes and diatoms, with a decrease in non-chain forming diatoms. The largest difference occurred between 14 and 28 cm/sec. Each velocity was twice the next lower value; the difference in absolute terms was 7 cm/sec between the lowest and middle velocity and 14 cm/sec between the middle and highest velocity.

At the end of the autumn velocity experiment the dominant species (in order of decreasing numerical abundance) consisted of *Achnanthes microcephala*, *Synedra* sp B., *Tabellaria fenestrata*, *Oscillatoria* spp., *Mougeotia* sp A., *Synedra* sp. A (Figure 17). The dominant species patterns, expressed as relative percentages of the top six species (Figure 18), indicated that *A. microcephala* comprised the diatom understory of the 7 and 14 cm/sec streams, but was a sub-dominant at the high velocity. *Synedra* sp. B was present principally at the highest and lowest velocities, being slightly less common at 14 cm/sec. *Tabellaria fenestrata* was most frequently observed at the highest velocity, as was *Oscillatoria*. *Mougeotia* dominated at the highest velocity, being a sub-dominant at 14 and 7 cm/sec. *Synedra* sp. A exhibited a somewhat variable distribution being highest at 28 cm/sec and at approximately the same numerical abundance at 7 and 14 cm/sec. Table 7 outlines the numerical abundance, in decreasing rank order, of the dominant species enumerated on the final day of the autumn velocity experiment. When all species enumerated throughout the experiment are compared (Table 8), of the 122 species encountered 116 comprised only 20% of the numerical abundance, with the majority encountered less than fifty times. The top 85 percent of individual organisms counted were accounted for by seven species (or 5.7% of the species encountered). Community structural differences between velocities during the winter was insignificant, with all

winter communities consisting principally of non-chainforming diatoms (99 %).

An analysis of variance using the mean cell counts of the most abundant algae yielded a P of 0.0038, with an F test score of 0.0068, indicating a substantial between-groups effect, with considerable within-group variance. This suggests that at the individual species level there is considerable difference between replicate streams. Mean and group standard deviation for the three velocities were fast, 95.0 and 2.1602; medium, 84.8 and 2.0616; and slow, 71.5 and 11.9, respectively. Pairwise comparisons indicated that 2 groups (fast and medium) had means which were not significantly different from one another. Thus, statistical analysis of the periphyton community both by total cell count and by the dominant species yielded different results of between-group effects.

Periphyton biomass accrual values during the study period (Figure 19) indicated that the maximum values (60 $g/m^2$) were obtained at the highest velocity (28 cm/sec) in the autumn (871021)(Ridley-Thomas, 1989). A positive correlation between biomass accrual and velocity was observed. The winter biomass values were significantly lower than those obtained in the autumn, with the same positive correlation between velocity and accrual. Maximum winter biomass (21 $g/m^2$; 28 cm/sec) was moderately greater than the maximum autumn value obtained at the lowest velocity (14 $g/m^2$ at 7 cm/sec).

The percentage ash (dry mass) of the periphyton, obtained at the end of the velocity experiment (Figure 20), indicated that no significant difference existed between velocities, for either autumn or winter communities. There was, however, a difference between the winter and autumn communities ash content. The winter communities (67%) contained approximately 5% more ash than did the autumn community (61%).

The relative abundance of the major taxonomic groups contained within the streams subjected to N+P enrichment and reduced irradiance, during the summer period (May 17, 1988), are shown in Figure 21. The control evidenced an increase in the relative proportion of non-chainforming

diatoms between the middle and end of the experiment. There was a concomitant decrease in chainforming diatoms and filamentous Chlorophytes during the same period. Under conditions of N and P and N+P enrichment a similar pattern of community structure dominance was observed, with variability apparent in the relative proportions of each group. Whereas under reduced irradiance, no significant difference in community architecture was observed from either the control or enrichment treatments, the largest difference was observed under light limited, nutrient enrichment. The latter community was dominated by diatoms (chain and non-chainforming) and filamentous Chlorophytes.

An analysis of variance of day 28 % protein (dry weight), for each of the treatments, gave a P of 0.0736 with an F test score of 0.7092 indicating a significant between-groups effect, but with small within-group variance. The pairwise comparison of means indicated that there were two groups in which the means were not significantly different from each other, with the nitrogen treatment showing the most difference. The mature community (day 46), however, yielded an Anova score of P equal to 0.0001 and F test score of 0.7924. Again this showed little within-group variance but with significant between-group differences. The pairwise means comparison indicated that there was a light effect under both light treatments (with and without NP addition). There was no statistical evidence of either an N or a P effect.

Comparisons of variance between the % protein of younger communities (expressed as a function of organic matter), under different treatments, produced a P of 0.0808 and an F test score of 0.6565; this indicated significant between group effects with little within-group variance. Pairwise mean comparisons showed a significant difference between the NP under reduced irradiance and N addition from the rest of the groups. The same assessment of mature periphyton yielded a P of 0.0709, with an F test score of 0.6485, again demonstrating between-group effects. The means comparisons indicated two groups in which the means were not significantly different from one another. There was no light or nutrient effect observed. Thus, there was no significant difference in the % protein under reduced light if expressed as a function of the communities organic content but was different when it's inorganic

component is included. There was no evidence of either an N or a P effect. This difference confirms the caution urged by Newman and McIntosh (1989) who recommend procedurally defining the periphyton community under investigation, with respect to it's use as a bioassay. The absence of any measure or estimate of power precluded assessing whether the experimental design was adequate to test the hypothesis concerning treatment effect.

Biomass accrual ($g/m^2$) during the enrichment and light limitation experiment is shown in Figure 22 (Table 4). The control demonstrated a steady increase in biomass accrual, with a small loss in biomass at the end of the third week. The control, under light limitation, exhibited an accrual of similar magnitude, but without any interim loss in biomass. All other treatments had an accrual similar to that of the controls, during the first two weeks. Accrual, however, was increased after this initial period. The addition of orthophosphorus resulted in maximum accrual after 3 weeks, with a subsequent sloughing of the filamentous canopy. A similar pattern was observed with the addition of nitrogen, however, the peak biomass was less and the sloughing occurred 20 days after the biomass peak observed with orthophosphorus addition. Enhancement of N and P resulted in a pattern similar to that of P alone, although the sloughing ended earlier and was followed by a second period of accrual. Under nutrient enrichment and light limitation, biomass continued to increase, and had not entered a period of sloughing by the cessation of the 45 day experiment.

The community exhibited lower ash content at 28 days than at 46 days (Figure 23). The lowest ash content in the 28 day community was found in the control, light limited and P enriched treatments, whilst the maximum ash was found in the N and P enriched, but light limited, condition. At day 46 the lowest ash content was found in the N and P (light limited) treatment, with the N and P enriched community having the highest.

The relative abundances of major taxonomic groups contained in the streams during the four major seasons are shown in Figure 24. Non-chainforming diatoms dominated the community structure in all four seasons, especially during the winter. Chainforming diatoms were a significant sub-dominant only during the spring. Filamentous Chlorophytes increased in dominance,

from their initial appearance in the spring, becoming a major biomass contributor during the late summer and early autumn. Non-filamentous Chlorophytes, Cyanophytes and Chrysophytes comprised a minor component of the biomass during the summer and autumn, being essentially absent during the winter and spring.

Seasonal biomass accrual exhibited an autumn maximum (35.0 $g/m^2$) and a winter minimum (11.3 $g/m^2$)(Figure 24); spring accrual (23.3 $g/m^2$) was mid-way between the latter, with summer accrual (13.3 $g/m^2$) slightly above that of the winter. Percentage ash (as dry mass) increased from its lowest value during spring (42%) through summer (50%) and autumn (61%) to its highest concentration during winter (67%) (Figure 24).

During two experiments to determine the potential toxicity of Glyphosate to periphyton (Table 4), biomass accrual developed under similar increase-decrease cycles, to those observed in the previously described experiments. Biomass in controls was less than the treatment streams, with the exception of the initial growth cycle. In nutrient enriched troughs, accrual development patterns were similar to those of the control, albeit with higher accrual (Figure 25). The experimental design required that Glyphosate treatments receive nutrient enrichment for a period of between 44 and 50 days. During this period all streams exhibited the same patterns of accrual, effectively providing eight replicates (Austin *et al.*, 1991). Algal species composition indicated that all numerically dominant species were present in all streams and treatments, although some patchiness was noted between streams and treatments. Numerically diatoms dominated the community, with *A. minutissima* eventually dominating all streams as a dense understory. Adnate diatoms (eg. *A. minutissima* )were the early colonizers with stalked and chainforming, vertically oriented, diatoms creating an initial canopy. Mature communities consisted of a diatomaceous understory and filamentous canopy. Although Chlorophytes (4 of the top 6 species) dominated algal biovolume, the understory consisted principally of the diatoms *Synedra, Fragilaria, Gomphonema* and *Tabellaria* spp.

Chlorophyll *a* accrual during the July Glyphosate experiment (developed on glass sampling substrata) again demonstrated the above described increase:

decrease accrual peaks (Figure 26), with variability in accrual between streams and differences also existed as to when peak accrual occurred. Minimum chlorophyll values were observed in streams 8, 11 and 12 (<0.1 $\mu g/cm^2$). Maximum accrual occurred in streams 3 and 4, and 9 and 10, in which peak chlorophyll values reached in excess of 1.0 $\mu g/cm^2$); accrual began earlier in these streams and sloughing had not commenced, to any significant extent, upon cessation of the trial. Phaeophytin values closely followed those of chlorophyll *a* in all streams, reflected both in magnitude and timing (Figure 26).

The general trend of autumn chlorophyll accrual (Table 4), cultivated on glass substrata, was similar to that of the previous summer experiment (i.e. an increase: decrease: increase accrual cycle)(Figure 27). Minimal accrual occurred in streams 11 and 12 (<0.75 $\mu g/cm^2$), whilst the maximum was in streams 3, 7, 8, and 10 (>1.9 $\mu g/cm^2$). As noted above, phaeophytin accrual patterns (Figure 27) closely paralleled those of chlorophyll *a*.

The general trend in chlorophyll *a* accrual, on styrofoam sheets, reflected an earlier onset of accrual, without evidencing an increase:decrease:increase cycle (Figure 28). Streams 3, and 7 through 10 exhibited maximum biomass of >13 $\mu g/cm^2$. Streams 11 and 12 had the lowest accrual at <6 $\mu g/cm^2$. Phaeophytin patterns (Figure 28), whilst showing a similar trend to chlorophyll *a*, were more variable in magnitude than those measured on glass substrata (Figure 28).

Maximum chlorophyll *a* values, measured on river rock, were noted in streams 3 and 4 (>25 $\mu g/cm^2$) with the lowest in streams 11 and 12 (<5 $\mu g/cm)^2$; the value in stream 2 was less than 1 $\mu g/cm^2$ by the end of the experiment, but had achieved an earlier peak of almost 5 $\mu g/cm^2$ (Figure 29). The general trend, for both chorophyll and phaeophytin (Figure 29), was as that noted above - a persistent increase with the maximum occurring at the cessation of the experiment (except as noted for stream 2).

### Discussion

Stream periphyton communities are theoretically one of the most productive of aquatic systems (Wetzel, 1983a) and contribute significantly to total primary

aquatic productivity and accrual (Weitzel, 1979). Investigations of periphyton in both laboratory and natural settings have yielded conceptual models of how this community develops under a wide range of environmental conditions. Studies have included how environmental factors affect species composition and distribution (McIntire, 1966a,b; Clark *et al.*, 1980; Bothwell and Jasper, 1983; Steinman and McIntire, 1990), productivity and biomass accrual (Bothwell, 1989) successional processes (Steinman and McIntire, 1990) and architectural transformations (Hudon *et al.*, 1987; McIntire, 1973; Steinman and McIntire, 1986). Only recently have investigations of periphyton proximate analysis (protein, lipid, carbohydrate) been undertaken (Steinman *et al.*, 1976).

The periphyton community is an important energy producer and subsequent energy transducer between trophic levels in lotic ecosystems (Sinsabaugh *et al.*, 1991; Wetzel, 1983). Steinman and McIntire (1990) note that for studying post-disturbance resilience, resistance and recovery processes in aquatic environments periphyton communities are useful because they are ubiquitous, relatively easy to sample and measure, typically contain species whose generation times are short, and thus permit monitoring of multiple generations through alternate successional seres. In assessing how function affects community structure, Rosemond *et al.*, (1993) and Weber and Lodge (1990) urge that a major focus for freshwater ecologists should be to evaluate the relative importance of *bottom-up* and *top-down* processes. A consistent research focus in understanding how freshwater communities are structured (Weber and Lodge, 1990; Wetzel, 1983) has been the influence of nutrients on pelagic and benthic autotrophic assemblages. The traditional approach to evaluate such abiotic and biotic interactions has been quantitative. Linkages between nutrient enhancement and subsequent increases in algal standing crop (Schindler, 1974, 1977, 1987; Cattaneo, 1978), and alterations in species composition and architecture, food-web structures and nutrient cycling (Sand-Jensen and Borum 1991) have been documented. Also studied have been environmental factors which interact with nutrients to affect periphyton community successional trajectories, including irradiance (Steinman *et al.*, 1989), current velocity (Horner *et al.*, 1990; Reiter and Carlson, 1986) and

endogenous nutrient re-cycling (Dodds, 1991a; McCormick and Stevenson, 1991).

Cairns (1993) has urged adoption of a public policy of ecological restoration of aquatic ecosystems. In earlier works also defining the need for development of new information bases, with which to test hazard evaluations, Cairns (1986) and others (Day *et al.*, 1988) have recommended "*research to develop standardized and validated methods of conducting multispecies toxicity tests.*" A number of surrogate stream systems have been developed throughout the past three decades (Table 2). Examination of these designs, however, demonstrates that field systems which are simple, reliable and which do not require the use of pumps, and other electrical apparatus, are relatively few. The system described here is easily constructed, at relatively low cost, and requires minimal maintenance; it has been operated at a number of locations, under a wide variety of seasonal conditions, since 1984, without any major problems. Although the proto-type system described here has been operational within a protected watershed, two other systems have been constructed in areas frequented by the public and not subject to regular surveillance (see section on Cheakamus and Squamish Rivers, Chapter III). To date we have not experienced problems with vandalism, which may in part be due to our policy of permitting unrestricted access to the facilities. Whilst the systems have typically been constructed within fenced enclosures for protection from interference by non-humans, the doors to these enclosures have never been locked. Instead, a sign describes the purpose of the study and cautions potential intruders to look but not disturb any of the equipment - to date we have had many visitors but none have damaged or disturbed any study.

Measurements of temperature, light, dissolved oxygen and chemical parameters indicated that, between surrogate streams, differences in these water quality parameters were minimal. Figure 28 indicates that biomass accrual between streams under the same water quality conditions was remarkably similar. On one occasion, however, we did experience significant differences between one stream and all others, although no obvious reason could be found for this difference. The significant differences in seasonal

water quality parameters were usual for the climatic region in which the system is located.

Mundie *et al.*, (1991), conducting mesocosm stream studies in Carnation Creek, Vancouver Island, noted that nutrients (N + P) were limiting autotrophic accrual; Stockner and Shortreed (1978) also observed that in these nutrient-poor waters increased standing stocks of periphyton occurred following enhancement of these two nutrients. Bothwell (1988) found that maximum periphyton production occurred below growth saturation concentrations of 10 $\mu g\ P\ L^{-1}$ and 100 $\mu g\ N\ L^{-1}$. In all studies conducted within our system, periphyton biomass accrual increased subsequent to phosphorus enrichment, whilst marginal increases occurred when nitrogen alone was increased.

In all experiments conducted within our surrogate streams there was a general accrual development pattern of increase:decrease:increase, reflecting accrual, sloughing and regrowth. There were, however, some streams in which biomass sloughing had not occurred by the end of the study. Using laboratory streams to study the effects of current velocity and light on the structure of periphyton, Steinman and McIntire (1986) and Steinman *et al.*, (1989) observed a similar biomass accrual pattern. This pattern has been attributed to sloughing of the mature community (Steinman *et al.*, 1989). One study which supports this hypothesis is Lamberti *et al.*, (1989) who measured increased downstream export of organic detritus from mature communities.

Steinman and McIntire (1986) have described alterations in periphyton community architecture as a *predictable series of seral stages.* These workers, on the basis of laboratory experiments, have proposed an hypothesis of periphyton succession in which the following sequence of events occur: i) an organic matrix (and associated polysaccharide mucilage), with bacteria; ii) small adnate diatoms, with a low vertical profile; iii) short vertically-oriented diatoms (frequently stalked); and iv) long, filamentous algae. This laboratory based development sequence was confirmed during our studies of periphyton seral development within our field-based, semi-natural surrogate streams, under a variety of environmental stresses. Examination of the initial colonization process, to confirm the presence of bacteria and/or organic

matrix formation (Appendix B), indicated a diverse bacterial community on the introduced glass substrata after only 3 days. The presence of the bacteria, as stage one (Steinman and McIntire, 1986) colonists, reconfirmed the observations of other workers (Hudon and Bourget, 1981; Korte and Blinn, 1983).

The three dimensional community structure on our sampling substrata confirms the laboratory observations of Steinman and McIntire (1986). The small adnate diatoms, such as *A. minutissima*, were found to form a thin "felt", above which the rosette forming *Synedra* spp. and chain-forming *Tabellaria fenestrata* formed an understory canopy. The filamentous Chlorophytes, in the mature community, formed a dense overstory canopy. The latter was especially evident in streams subjected to nutrient enrichment, particularly orthophosphorus. As noted by Steinman *et al.*, (1989), who hypothesized Chlorophyte domination under high irradiances, the addition of orthophosphorus markedly enhanced this effect.

In our experiments the community structures developed during the spring, summer and autumn all conformed to the generalized architecture discussed above, with the principal difference being the degree of Chlorophyte canopy development. The development in our surrogate streams of a Chlorophyte canopy in mature periphyton confirmed that of other studies (McIntire, 1968; Shortreed and Stockner, 1983). As with these other studies, we postulate that exposure of the streams to high light intensities (i.e. direct sunlight) throughout the studies duration resulted in a Chlorophyte dominated community. As the mature canopy developed, and irradiance values diminished in the understory, diatom numbers increased. The increase in inorganic material (ash) reflected both the increased proportion of diatoms, as well as the trapping of suspended solids by the filamentous canopy. The increased presence of diatoms in the understory may be due to the tolerance of prostrate diatoms for low light intensities; such understory habitats would result in selection for such shade tolerant species (Hudon *et al.*, 1983). We observed an increase in the relative proportion of single-celled diatoms, in the understory, after 48 days, further evidence that light inhibition may have been affecting community structure.

The development of Chlorophyte dominated assemblages, in our streams, may indicate a competitive advantage, to filamentous greens, over diatoms, under high photon fluxes. This confirmed McIntire (1968) and Steinman and McIntire's (1986) findings, in laboratory streams, which suggest that competitive interactions (and not facilitative) as suggested by Connell and Slatyer (1977) are primarily responsible for structuring the observed successional stages, and changes, in our streams. Similarly, the analogy of vertical stratification in higher plant communities and lotic periphyton, as pointed out by Steinman and McIntire (1986), was confirmed in our studies, using surrogate stream streams under outdoor conditions. Since our streams are not subject to the episodic stresses (i.e. flow/ velocity fluctuations) frequently observed in natural streams there was likely a higher than normal degree of homogeneity in community structure. This precluded determining whether inhibition affected the trajectory of successional development in our stream-contained periphyton communities. In subsequent field studies on mountain streams (Chapter III) a comparison between stream contained periphyton communities and periphyton on natural substrata was possible.

In contrast, the winter community, developing under low photon fluxes, consisted essentially of small, adnate diatoms. Although one of the dominant species, *T. fenestrata*, is typically a chainformer, during the winter period it was almost exclusively observed as single or paired cells (Ridley-Thomas, 1989). Winter communities had the highest relative proportion of inorganic matter (ash), reflecting the community dominance by diatoms (i.e. silicon frustules) and lower growth rates and accrual, resulting in part from the higher percentage of dead materials. Higher suspended solids, originating from increased winter runoff may have also contributed to enhanced inorganic material being sedimented onto the sampling surfaces. This would suggest that during the winter, when nutrients are not limiting, the reduced irradiance may be affecting community structure by limiting primary productivity to a rate not capable of sustaining growth and in which herbivory, whilst minimal, equals or exceeds accrual.

In control surrogate streams the community structure followed a trajectory similar to that of manipulated streams but occurring at a different rate. This

may reflect that i) light was not limiting, and ii) that, as the biomass increased, internal nutrient cycling may have been a more important factor than in those communities receiving external nutrient supplements (Riber and Wetzel, 1987). Thus, although the control communities successional trajectory may have been the same, it responded at a slower rate than did streams receiving additional nutrients. These findings confirm the prediction of Riber and Wetzel (1987) that addition of small amounts of the limiting nutrient would permit enhanced biomass accrual. The statistical analysis supports the subjective, visual observations of increased biomass accrual under ortho-P enrichment. These findings would, thus, refute the hypothesis suggested by Odum (1985), that substantial concentrations of phosphorus are required to induce enhanced periphyton biomass accrual. This study also confirmed Schindler's (1987) recommendation that in-field studies, conducted at the ecosystem level, are required to determine how environments will respond following stress-induced perturbations.

The theory that within any given section of a stream all periphyton patches are on the same successional trajectory has been discussed by Steinman and McIntire (1990). The periphyton communities contained in our streams, for equivalent periods of time, under the same per-treatment conditions, typically developed along a similar seral path. However, since on one occasion when the pre-treatment community in one stream developed a community structure dominated by Cyanophytes and did not achieve significant biomass accrual, there exists some potential for non-parallel successional trajectories, perhaps based upon differential initial recruitment. Post-treatment community structures indicated that, under nutrient loading and high irradiance, our periphyton communities repeatedly exhibited a successional trajectory characterized by a diatomaceous understory and Chlorophyte canopy. Measurements of light intensity during community development indicated that the diatomaceous understory was subjected to reduced irradiance intensity as the Chlorophyte canopy thickened. Riber and Wetzel (1987), examining the micro-environmental conditions within Chlorophyte canopies observed that restricted water velocities likely resulted in diminished nutrient concentrations as boundary layer nutrient flux rates declined. Correspondingly, it would seem likely that optimal nutrient ratios

would also be less likely encountered by understory biota as the limiting nutrient became further exhausted.

Our studies have repeatedly confirmed that increased biomass accrual occurred when periphyton was subjected to nutrient enrichment, as shown by Stockner and Shortreed (1978), Horner *et al.*, (1990), Mundie *et al.*, (1991) and Bothwell and Jasper (1983). Whilst this alteration has been the subject of numerous investigations, few studies have attempted to separate the effect of nutrient concentration from that of velocity. The importance of nutrient flux has remained a relatively unexplored area. Future studies conducted in our streams will include preliminary experiments on nutrient flux.

Nutritional composition of periphyton communities likely have important implications for higher trophic levels. The environmental variables which may affect periphyton proximate composition have been partially studied, using our surrogate streams (Austin *et al.*, unpubl. data.; Ridley-Thomas, 1989). One study presently underway is how nutrient flux and ratios can affect accrual and/or periphyton proximate composition, which could have important implications for higher trophic levels *vis-a-vis* biotic health, fecundity and egg maturation.

Newman and McIntosh (1989) have described the need for carefully procedurally-defining and documenting periphyton communities used in monitoring the bioaccumulation of toxins. The community developed within our stream facility did not contain the large amounts of inorganic matter often associated with periphyton communities grown in water containing high concentrations of inorganic salts or suspended solids which precipitate, markedly enhancing the ash percentage. However, we did confirm their caution that characterizing the biochemical profile depended on firstly defining the community (i.e. protein as a percentage of organic vs inorganic biomass).

In conclusion, the capacity of the periphyton community component of our surrogate streams to respond to alterations in a variety of physical and chemical parameters has been demonstrated. The periphyton community also exhibited differences in rates of, and end-points of, succession, when

subjected to nutrient ratios and concentrations at, or below, chemically detectable thresholds. In other words, in the absence of detectable water quality differences at the chemically detectable level there were manifest differences at the biological level. These characteristics suggest that the stream complex reported here could provide important, repeatable information, using a standardized sampling protocol, for modelling periphyton community response to subtle or threshold environmental perturbations. The subsequent section describes how surrogate streams were used to measure impact, on a pristine mountain river, of treated domestic waste waters from a remotely located, recreationally accelerated, development in a wilderness area in B.C. (Resort Municipality of Whistler)(RMOW).

### Summary

In this work the algal component of lotic periphyton communities exhibited sensitivity to specific nutrient (N and P) ratios and concentrations. The results reported here have implications for water quality control and management of aquatic resources through advancing our understanding of how variability in periphyton primary production and nutritional composition affects trophic energy partitioning in shallow-water benthos. The latter is increasingly important as regulatory agencies require whole stream assessments prior to the introduction of such pollutants as biocides and nutrients into the environment (Day *et al*, 1988).

The hypothesis that nutrients influence the biochemical (nutritional) composition of lotic primary producers was demonstrated, although subsequent trophic implications have yet to be thoroughly field tested. That these processes are so poorly understood is surprising given the recognized importance of periphyton production in lotic systems. Periphyton community structure (species composition and architecture) and function (biomass and biochemistry) have important implications for higher trophic levels, including invertebrate herbivore, larval and fish production. The biochemical profile of nutrient stressed communities may be useful as an index of "post-disturbance" recovery. Future work will explore the practicality of generating a periphyton community, of known structure and

nutritional content, for use in investigations of energy transfer to higher trophic levels, especially at the community level.

Research on "bottom-up" effects is important in providing additional criteria for assessing direct linkages (Power *et al.*, 1985; Weber and Lodge, 1990) between algal community structure and food quality (Steinman *et al.*, 1987), and herbivore fitness and phenology (Lamberti *et al.*, 1992) or, indirectly, with carnivorous predators. Two principal mechanisms have been postulated to account for indirect effects - trophic and behaviour linkages (Rosemond *et al.*, 1993; Hill *et al.*, 1990, 1992). Given the importance of these linkages (Paine, 1971), there has been surprisingly little research conducted on how nutrient ratio and concentration fluctuations (Feller, 1989; Biggs, 1990a,b; Horner *et al.*, 1990) alter lotic microalgal protein:lipid ratios and accumulation (Chelf, 1990; Roessler, 1990) - i.e. the cascade (three trophic-level) effect of algal food quality on the direct/indirect structuring of higher trophic levels, such as herbivores and carnivores (Perrin *et al.*, 1987). Additional research objectives, testable using the facility described here, are outlined in the final chapter.

Table 3. Morphometric characteristics of Sooke Reservoir.

| Parameter | |
|---|---|
| Elevation (m) | 180.7 |
| Length (km) | 6855 |
| Max. width (m) | 1463 |
| Max. depth (m) | 68 |
| Surface area (ha) | 593 |
| Volume ($10xm^3$) | $1.22 \times 10^8$ |
| Shoreline perimeter (m) | $2.4 \times 10^4$ |
| Mean depth (m) | 20.6 |
| Volume development | 0.91 |
| Ratio mean Z:max Z | 0.30 |
| Area of littoral (<7m) (ha) | 165 |
| Watershed area (ha) | $7.25 \times 10^4$ |

Z = depth.

Table 4. Summary of time course and weather data (from the Victoria Federal Meterological Centre) during the four seasons. NA = not available/collected.

| Parameter | Autumn | Winter | Spring | Summer * | Summer | Autumn | Autumn | Fall/Winter |
|---|---|---|---|---|---|---|---|---|
| Mean Maximum Temperature (C) | 20.8 | 6.5 | 11.8 | 18.1 | | | 17.8 | 17.8 |
| Mean Minimum Temperature (C) | 7.7 | -0.3 | 3.1 | 7.7 | | | 5.9 | 5.9 |
| Total Precipitation (mm) | 2 | 202 | 181 | 47 | | | NA | NA |
| Mean Daily Sunshine (h) | 9.2 | 2.2 | 5.4 | 8.3 | | | NA | NA |
| Dates of Experiments - Begin | 870819 | 871208 | 880221 | 880517 | 880704 | 880819 | 890826 | 890925 |
| - End | 871021 | 880210 | 880426 | 880701 | 880725 | 880929 | 891121 | 891221 |
| Experiments Duration (d) | 63 | 64 | 67 | 45* | 21 | 41 | 88 | 87 |
| Experimental treatments | velocity (7, 14 & 28 cm/sec) | velocity (7, 14 & 28 cm/sec) | | nutrient (N,P; N+P) enrich-ment; light intensity | substrata = rock, glass slides, styro-foam | substrata = rock, glass slides, styro-foam | glypho-sate addition; N+P addition; 18 l/min | glypho-sate addition; N+P addition; 18 l/min |

* Experiment interrupted due to water quality constraints (copper sulphate addition to source (Sooke Reservoir) water).

Table 5. Methods used to analyze water quality (from McQuaker (1976) and Clark and Morrison (1982)). Abreviations of analytical techniques listed below. 1 - mg/l $CaCO_3$; 2 - mg/l N; 3 - Relative units; 4 - mg/l P; 5 - mg/l $SiO_2$; 6 - µS/cm (microsiemens/cm). Dissolved values filtered through 0.45 µ filter.

| Parameter | Analytical Technique | Instrument Range | Units |
|---|---|---|---|
| Alkalinity: total: pH 4.5 | Potentiometric | 0.5-500.0 | 1 |
| Nitrogen Ammonia Dissolved | Froz. Aut. Berth.[a] | 0.005-0.500 | 2 |
| Nitrogen Ammonia Dissolved | Direct Nessler | 0.0-2.0 | 2 |
| Nit: Nitrate + Nitrite | Froz. Aut. Berth.[a] | 0.02-2.00 | 2 |
| Nitrogen: Organic | Calcul. Result[b] | 0.01-10.0 | 2 |
| Nitrogen: Total Kjeldahl | Frz. Dig. Aut. Col[c] | 0.01-10.00 | 2 |
| Nitrogen: Total | Calcul. Result[d] | 0.02-10.0 | 2 |
| pH | Electrometric[e] | 0.0-14.0 | 3 |
| Phos: Dissolved orthophosphate | Froz. Aut. Ascorb[f] | 0.003-0.500 | 4 |
| Phosphorus: Total Dissolved | Frz. Dig. Aut. Asc[g] | 0.003-0.500 | 4 |
| Phosphorus: Total | Frz. Dig. Aut. Asc[g] | 0.003-0.500 | 4 |
| Silica: Reactive | Auto. Moly-Blue[h] | 0.5-30.0 | 5 |
| Specific Conductance | Cond. Meter | 1-60,000 | 6 |

a Froz. Aut. Berth. = Automated* Berthelot, frozen method.
b Calculate result = difference of Kjeldahl Nitrogen minus ammonia nitrogen (TKN - Amm.N = Organic N).
c Frz. Dig. Aut. Colour = Digestion Automated* Indophenol.
d Total Nitrogen = Calculated result = sum of Kjeldahl Nitrogen plus Nitrite and Nitrate Nitrogen concentrations (TKN + ($NO_2$+$NO_3$) N = Total N).
e Electrometric = pH meter, glass electrode reference electrode OR automated microprocessor-controlled titrator
f Froz. Aut. Ascorbic = Automated* ascorbic acid reduction
g Frz. Dig. Aut. Asco = Persulfate digestion, automated* ascorbic acid reduction.
h Auto. Moly-Blue Col. = Automated* ascorbic acid reduction, colorometric.
* Automated - Technicon Automated II.

Table 6. Water chemistry values measured during the winter months at Humpback Reservoir; water chemistry samples taken from header box. See Table 5 for analysis methodology. pH - Relative units; Spec. Cond - µS/cm (microsiemens/cm); Alk - mg/l $CaCO_3$; Si - mg/l $SiO_2$ ; N -- mg/l N; P - mg/l P.

| Parameters | Sample Dates | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | 87-12-16 | 87-12-22 | 87-12-29 | 88-01-06 | 88-01-19 | 88-02-10 | 88-02-24 | 88-02-29 |
| pH | | | | | | | 6.8 | 6.6 |
| Spec. Cond. | | | | | | | 57 | 54 |
| Alkalinity | | | | | | | 13.4 | 13.4 |
| Si: React. | 1.6 | 1.5 | 2.0 | 1.8 | 3.3 | 2.3 | 5.8 | 5.1 |
| N: Org-Total | 0.06 | 0.10 | 0.11 | 0.09 | 0.04 | 0.03 | 0.10 | 0.08 |
| N: Kjel. Total | 0.08 | 0.11 | 0.13 | 0.10 | 0.05 | 0.04 | 0.10 | 0.09 |
| N: Total | 0.29 | 0.27 | 0.24 | 0.22 | 0.20 | 0.14 | 0.24 | 0.22 |
| N: Ammonia | 0.019 | 0.010 | 0.017 | 0.013 | 0.010 | 0.012 | <0.005 | 0.009 |
| N: $No_3$+$NO_2$ | 0.21 | 0.16 | 0.11 | 0.12 | 0.15 | 0.10 | 0.14 | 0.13 |
| P: Ortho. Diss.-P | <0.003 | <0.003 | <0.003 | <0.003 | <0.003 | <0.003 | <0.003 | <0.003 |
| P: Total | 0.008 | 0.005 | 0.006 | 0.005 | 0.004 | 0.004 | 0.005 | 0.003 |
| P: Tot. Dissolved | 0.004 | 0.004 | 0.003 | 0.003 | 0.003 | 0.003 | 0.003 | <0.003 |

Table 7. Mean cell counts for the six most abundant algae enumerated at the end of the first velocity experiment (October 21, 1987). See Table 4 for details of the experimental design. Experiment 1 = 28 cm/sec; experiment 2 = 14 cm/sec; experiment 3 = 7 cm/sec.

| Relative Proportions (Percent) for the Six Most Abundant Taxa for Trough Experiments as October 21, 1987 | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Experiment 1 | | | | Experiment 2 | | | | Experiment 3 | | | |
| | TR-1 | TR-2 | TR-3 | TR-4 | TR-5 | TR-6 | TR-7 | TR-8 | TR-9 | TR-10 | TR-11 | TR-12 |
| Achnanthes microcephala | 7 | 6 | 2 | 6 | 12 | 15 | 15 | 18 | 20 | 46 | 26 | 23 |
| Synedra sp. B | 7 | 10 | 4 | 5 | 1 | 2 | 3 | 5 | 12 | 6 | 7 | 3 |
| Tabellaria fenestrata | 5 | 9 | 3 | 12 | 2 | 1 | 1 | 1 | 6 | 2 | 4 | 1 |
| Oscillatoria spp. | 26 | 25 | 0 | 1 | 0 | 0 | 5 | 0 | 0 | 0 | 1 | 0 |
| Mougeotia sp. A | 5 | 15 | 1 | 6 | 0 | 0 | 0 | 1 | 5 | 2 | 2 | 1 |
| Synedra sp. A | 5 | 3 | 2 | 1 | 1 | 1 | 1 | 2 | 5 | 2 | 2 | 1 |
| Other Taxa | 45 | 33 | 88 | 69 | 83 | 81 | 75 | 73 | 52 | 42 | 58 | 72 |
| Total | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |

| Relative Proportions (Percent) for the Six Most Abundant Taxa for Trough Experiments as October 21, 1987 | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | TR-1 | TR-2 | TR-3 | TR-4 | TR-5 | TR-6 | TR-7 | TR-8 | TR-9 | TR-10 | TR-11 | TR-12 |
| Achnanthes microcephala | 13 | 9 | 16 | 19 | 74 | 78 | 60 | 67 | 41 | 80 | 62 | 82 |
| Synedra sp. B | 12 | 15 | 29 | 16 | 7 | 10 | 11 | 19 | 25 | 10 | 17 | 10 |
| Tabellaria fenestrata | 9 | 13 | 28 | 37 | 10 | 5 | 5 | 4 | 13 | 3 | 10 | 3 |
| Oscillatoria spp. | 48 | 37 | 3 | 4 | 0 | 0 | 18 | 1 | 0 | 0 | 2 | 0 |
| Mougeotia sp. A | 10 | 22 | 11 | 21 | 2 | 1 | 1 | 2 | 11 | 4 | 5 | 2 |
| Synedra sp. A | 9 | 4 | 13 | 3 | 6 | 6 | 6 | 7 | 11 | 3 | 4 | 3 |

Table 8. List of all algal species enumerated at the end of the velocity experiment (October 21, 1987). See Table 4 for additional details of experimental design.

| Unique ID Code | Genus/species | Count |
|---|---|---|
| aa | **Bacillariophyta** | |
| Achmic | Achnanthes microcephala (Kuetz.) Grun. | 23519 |
| Astfor | Asterionella formosa Hassall | 197 |
| Astspp | Asterionella spp. Hassall | 48 |
| Cocdim | Cocconeis diminuata Pant. | 2 |
| Cocpla | Cocconeis placentula Ehrbg. | 1 |
| CocspA | Cocconeis sp. A Ehrenberg. | 1 |
| Cocspp | Cocconeis spp. Ehrenberg. | 9 |
| Cycbod | Cyclotella bodanica Eulen. | 47 |
| Cyccom | Cyclotella compta (Ehrbg.) Kuetz. | 28 |
| CycspA | Cyclotella sp. A Kuetzing | 18 |
| Cycspp | Cyclotella spp. Kuetzing | 4 |
| Cymcvp | Cymbella cymbiformis var. punetetu (C. Ag.?/Kuetz.?) VanHeurck | 1 |
| Cymcym | Cymbella cymbiformis (C. Ag.?/Kuetz.?) VanHeurck | 56 |
| CymspA | Cymbella sp. A C.A. Agardh | 170 |
| CymspC | Cymbella sp. C Agardh | 1 |
| Cymspp | Cymbella spp. Agardh | 16 |
| Diavul | Diatoma vulgare Bory | 2368 |
| Diavvl | Diatoma vulgare var. linearis | 63 |
| Dipell | Diploneis elliptica (Kuetz.) Cleve. | 1 |
| Epispp | Epithemia spp. Brebission | 1 |
| Eu?Me? | Eunotia spp. (Ehr.) or Melosira spp. (Agardh) | 2 |
| Eun1sp | Eunotia spp. Ehrenberg. | 14 |
| Euncur | Eunotia curvata (Kuetz.) Lagerst. | 6 |
| Eunrob | Eunotia robusta Ralfs. | 47 |
| Fra2sA | Fragilaria sp. A Lyngbye | 224 |
| Fra2sB | Fragilaria sp. B Lyngbye | 167 |
| Fra2sC | Fragilaria sp. C Lyngbye | 13 |
| Fra2sD | Fragilaria sp. D Lyngbye | 111 |
| Fra2sp | Fragilaria spp. Lyngbye | 12 |

Table 8. Continued. List of all algal species enumerated at the end of the velocity experiment (October 21, 1987). See Table 4 for additional details of experimental design.

| | | |
|---|---|---|
| Fracro | Fragilaria crotonensis Kitton | 794 |
| Fravir | Fragilaria virescens Ralfs | 1116 |
| Frurho | Frustulia rhomboides (Ehrbg.) DeToni | 2 |
| Gomacu | Gomphonema acuminatum Ehrenberg | 5 |
| Gomavc | Gomphonema acuminatum var. coronatum (Ehrbg.) W. Smith | 34 |
| Gomave | Gomphonema acuminatum var. elongatum (W.Sm.) Carr | 1 |
| Gomcon | Gomphonema constrictum Ehrenberg | 8 |
| Gomcvc | Gomphonema constrictum va. capitatum (Ehrbg.) Grun. | 1 |
| Gomoli | Gomphonema olivaceum (Lyngb.) Kuetz. | 124 |
| Gomols | Gomphonema olivacium (stalked) (Lyngb.) Kuetz. | 11 |
| GomsAs | Gomphonema sp. A (stalked) Ehrenberg | 31 |
| GomspA | Gomphonema sp. A Ehrenberg | 1 |
| GomspB | Gomphonema sp. B Ehrenberg | 2 |
| Gomspp | Gomphonema spp. Ehrenberg | 10 |
| Gomsps | Gomphonema spp. (stalked) Ehrenberg | 14 |
| Gyr1sp | Gyrosigma spp. Hassall | 4 |
| MelspA | Melosira sp. A C.A. Agardh | 411 |
| MelspB | Melosira sp. B C.A. Agardh | 15 |
| Melspp | Melosira spp. C.A. Agardh | 539 |
| Nav1sA | Navicula sp. A Bory | 42 |
| Nav1sB | Navicula sp. B Bory | 33 |
| Nav1sC | Navicula sp. C Bory | 1 |
| Nav1sD | Navicula sp. D Bory | 1 |
| Nav1sp | Navicula spp. Bory | 80 |
| Nitlin | Nitzschia linearis W. Smith | 1 |
| Nitsig | Nitzschia sigmoidea (Ehrbg.) W. Smith | 52 |
| Nitviv | Nitzschia vivax W. Smith | 4 |
| Pin1sA | Pinnularia sp. A Ehrenberg | 2 |
| Pin1sX | Pinnularia sp. X Ehrenberg | 1 |
| Pinspp | Pinnularia spp. Ehrenberg | 5 |
| Pinstr | Pinnularia streptoraphe Cleve. | 1 |

Table 8. Continued. List of all algal species enumerated at the end of the velocity experiment (October 21, 1987). See Table 4 for additional details of experimental design.

| | | |
|---|---|---|
| Rhieri | Urosolenia eriensis (Smith) | 39 |
| Rhogib | Rhopalodia gibba (Ehrbg.) O. Mueller | 24 |
| Rhospp | Rhopalodia spp. O. Mueller | 2 |
| Sta2sp | Stauroneis spp. Ehrenberg | 1 |
| Stapho | Stauroneis phoenocentron (Nitz.) Ehrbg. | 1 |
| Stespp | Stephanodiscus spp. Ehrenberg. | 4 |
| Surbis | Surirella biserata Breb. | 1 |
| SurspA | Surirella sp. A Turpin | 1 |
| Surspp | Surirella spp. Turpin | 2 |
| Synfas | Synedra fasciculata (Ag.) Kutz. | 11 |
| Synfil | Synedra filiformis Grun. | 326.5 |
| SynspA | Synedra sp. A Ehrenberg | 4489 |
| SynspB | Synedra sp. B Ehrenberg | 9561 |
| SynspC | Synedra sp. C Ehrenberg | 889 |
| SynspD | Synedra sp. D Ehrenberg | 4 |
| Synspp | Synedra spp. Ehrenberg | 1 |
| Synuvl | Synedra ulna var. longissima (W.Sm.) Brun. | 193 |
| Tabfen | Tabellaria fenestrata (Lyngb.) Kuetz. | 12323 |
| Tabflo | Tabellaria flocculosa (Roth.) Kuetz. | 929 |
| | | |
| aa | **Chlorophyta** | |
| Ankfal | Ankistrodesmus falcatus (Corda) Ralfs | 9 |
| AnkspB | Ankistrodesmus sp. B Corda | 2 |
| Ankspi | Ankistrodesmus spiralis (Turner) Lemm. | 31 |
| Ankspp | Ankistrodesmus spp. Corda | 4 |
| Aphspp | Aphanocheate spp. A. Braun | 4 |
| Artcon | Arthrodesmus convergens Ehrenberg | 1 |
| Bulspp | Bulbochaete spp. C.A. Agardh | 10 |
| Claspp | Cladophorales spp. | 1 |
| Clocyn | Closterium cynthia DeNot. | 1 |
| Clopar | Closterium parvulum Naeg. | 1 |

Table 8. Continued. List of all algal species enumerated at the end of the velocity experiment (October 21, 1987). See Table 4 for additional details of experimental design.

| | | |
|---|---|---|
| ClospA | Closterium sp. A Nitzsch ex Ralfs | 5 |
| Clospp | Closterium spp. Nitzsch ex Ralfs | 4 |
| Cosabb | Cosmarium abbreviatum Racib. | 7 |
| Cosavp | Cosmarium abbreviatum var. planctonicum West & West | 1 |
| Cosbin | Cosmarium binum Nordst. | 10 |
| Cosbio | Cosmarium bioculatum Breb. | 5 |
| Cosbly | Cosmarium blyttii Wille | 55 |
| Cosd?o | Cosmarium depressum (Naeg.) or C. orbiculatum? Ralfs | 1 |
| Cosdep | Cosmarium depressum (Naeg.) Lund | 2 |
| Cosgra | Cosmarium granatum Breb. | 2 |
| Cosmon | Cosmarium moniliforme (Turpin) Ralfs | 3 |
| Cospac | Cosmarium pachydermum Lund | 1 |
| Cospun | Cosmarium punctulatum Breb. | 34 |
| CosspA | Cosmarium sp. A Corda ex Ralfs | 5 |
| Cosspp | Cosmarium spp. Corda ex Ralfs | 1 |
| Cossub | Cosmarium subcrenatum Hantz. | 20 |
| Crurec | Crucigenia rectangularis (A. Br.) Gay | 9 |
| Dicpul | Dictyosphaerium pulchellum Wood | 1 |
| Dicspp | Dictyosphaerium spp. Naegeli | 6 |
| Euaden | Euastrum denticulatum (Kirchn.) Gay | 5 |
| Eudspp | Eudorina spp. Ehrenberg. | 10 |
| Gemord | Geminella ordinata (West & West) Heering | 1 |
| GemspA | Geminella sp. A Turpin | 679 |
| Gemspp | Geminella spp. Turpin | 871 |
| Gonmon | Gonatozygon monotaenium DeBary | 7 |
| LGballs | Little Green Balls | 21 |
| Miapin | Micrasterias pinnatifida (Kuetz.) Ralfs | 2 |
| Micsol | Micrasterias sol (Ehrbg.) Kuetz. | 5 |
| Micspp | Microspora spp. Thuret | 8 |
| Mou1sA | Mougeotia sp. A C.A. Agardh | 2666 |
| Mou1sB | Mougeotia sp. B C.A. Agardh | 2033 |

Table 8. Continued. List of all algal species enumerated at the end of the velocity experiment (October 21, 1987). See Table 4 for additional details of experimental design.

| | | |
|---|---|---|
| Mou1sC | Mougeotia sp. C C.A. Agardh | 386 |
| Mou1sD | Mougeotia sp. D C.A. Agardh | 115 |
| Mou1sp | Mougeotia spp. C.A. Agardh | 196 |
| OedspA | Oedogonium sp. A Link in Nees | 26 |
| Oedspp | Oedogonium spp. Link in Nees | 51 |
| Oo?Ch? | Oocystis A. Braun or Chrysospora spp. | 1 |
| Ooceli | Oocystis ellipticaW. West | 11 |
| Oocevm | Oocystis elliptica var. minor W. West | 25 |
| OocspA | Oocystis sp. A A. Braun | 15 |
| Oocspp | Oocystis spp. A. Braun | 9 |
| Panspp | Pandorina spp. Bory | 1 |
| Ped1sp | Pediastrum spp. Meyen | 35 |
| Pedtet | Pediastrum tetras (Ehrbg.) Ralfs. | 4 |
| Scearc | Scenedesmus arcuatus Lemm. | 5 |
| Scearm | Scenedesmus armatus (Chodat) G. Smith | 1 |
| Scebij | Scenedesmus bijuga (Turpin) Lagerh. | 16 |
| Scequa | Scenedesmus quadricauda (Turpin) Breb. | 9 |
| Sceqvq | Scenedesmus quadricauda (Turpin) Breb. var. quadricauda | 1 |
| Scespp | Scenedesmus spp. Meyen | 17 |
| Spasch | Sphaerocystis Schroeteri Chodat | 1 |
| Sphavo | Sphaerozosma aubertianum var. order West | 1 |
| Spispp | Spirogyra spp. Link in C.G. Nees | 22 |
| Spopla | Spondylosium planum (Wolle) West & West | 179 |
| Spopvm | Spondylosium pygmaeum (Cooke) W. West var. monile | 39 |
| Sta3cu | Staurodesmus cuspidatus (Breb.) Teil. | 1 |
| Sta3de | Staurodesmus dejectus (Breb.) Teil. | 8 |
| Staana | Staurastrum anatinum Cooke & Wills | 17 |
| Staarc | Staurastrum arctiscon (Ehrbg.) Lund | 4 |
| Stafur | Staurastrum furcigerum Breb. | 1 |
| Stagra | Staurastrum granulosum (Ehrbg.) Ralfs | 1 |
| Stainf | Staurastrum inflexum Breb. | 1 |

Table 8. Continued. List of all algal species enumerated at the end of the velocity experiment (October 21, 1987). See Table 4 for additional details of experimental design.

| | | |
|---|---|---|
| Stalun | Staurastrum lunatum Ralfs | 2 |
| Stamin | Staurastrum minnesotense Wolle | 7 |
| Staoph | Staurastrum ophiura Lund | 1 |
| Stapol | Staurastrum polymorphum Breb. | 1 |
| StaspA | Staurastrum sp. A Meyen ex Ralfs | 2 |
| StaspC | Staurastrum sp. C Meyen ex Ralfs | 1 |
| Staspp | Staurastrum spp. Meyen ex Ralfs | 2 |
| Stibac | Stichiococcus bacillaris Naeg. | 23 |
| Stispp | Stigioclonium spp. Kuetzing | 184 |
| Ulo1sA | Ulothrix sp. A Kuetzing | 353 |
| Ulo1sp | Ulothrix spp. Kuetzing | 880 |
| Xanant | Xanthidium antilopeum (Breb.) Kuetz. | 3 |
| Xancri | Xanthidium cristatum Breb. | 1 |
| Xansub | Xanthidium subhastiferum W. West | 2 |
| ZygspA | Zygnema sp. A C.A. Agardh | 489 |
| Zygspp | Zygnema spp. C.A. Agardh | 341 |
| | | |
| aa | **Chrysophyta (Golden Browns)** | |
| Chr2lo | Chrysophaerella longispina Lauter. | 1 |
| Dinbav | Dinobyron bavaricum Imhof | 1364 |
| Dindiv | Dinobyron divergens Imhof | 407 |
| Malspp | Mallomonas spp. Perty | 1 |
| | | |
| | | |
| aa | **Cyanophyta (Blue-Greens)** | |
| An?No? | Anabaena spp. (Bory) Bornet & Flahault or Nostoc spp. (Vaucher) | 9 |
| Ana1sA | Anabaena sp. A (Bory) Bornet & Flahault | 50 |
| Ana1sp | Anabaena spp. (Bory) Bornet & Flahault | 4 |
| ChrspA | Chroococcus sp. A Naegeli | 40 |
| Gom2la | Gomphosphaeria lacustris Chodat | 215 |
| Gom2sp | Gomphosphaeria spp. Kuetzing | 61 |

Table 8. Continued. List of all algal species enumerated at the end of the velocity experiment (October 21, 1987). See Table 4 for additional details of experimental design.

| | | |
|---|---|---|
| Lynspp | Lyngbya spp. C.A. Agardh ex Gomont | 1 |
| Mer1sA | Merismopedia sp. A Meyen in Wiegmann | 38 |
| Mer1sp | Merismopedia spp. Meyen in Wiegmann | 603 |
| Nosspp | Nostoc spp. Vaucher | 6 |
| Osc1sp | Oscillatoria spp. Vaucher ex Gomont | 5129 |
| OscspA | Oscillatoria sp. A Vaucher ex Gomont | 62 |
| | | |
| aa | **Protozoan** | |
| Cilpsp | Ciliated protozoan spp. | 27 |
| EucspA | Eucillia sp. A ?? | 5 |
| Eucspp | Eucillia spp. ?? | 8 |
| Vorspp | Vorticella spp. Ehrenberg | 22 |
| | | |
| aa | **Pyrrhophyta** | |
| Cerhir | Ceratium hirundinella (O.F.M.) Bergh. | 4 |
| Gle1sA | Glenodinium sp. A ?? | 1 |
| Glespp | Glenodinium spp. (Ehrenb.) Stein | 1 |
| Percin | Peridinium cinctum (O.F.M.) Ehrbg. | 17 |
| Perpus | Peridinium pusillum (Penard) Lemm. | 15 |
| PerspA | Peridinium sp. A Ehrenberg | 70 |
| PerspB | Peridinium sp. B Ehrenberg | 4 |
| Perspp | Peridinium spp. Ehrenberg | 1 |

Table 9. Listing of algal species in decreasing numerical abundance enumerated at the end of the velocity experiment (October 21, 1987). See Table 4 for details of the experimental design.

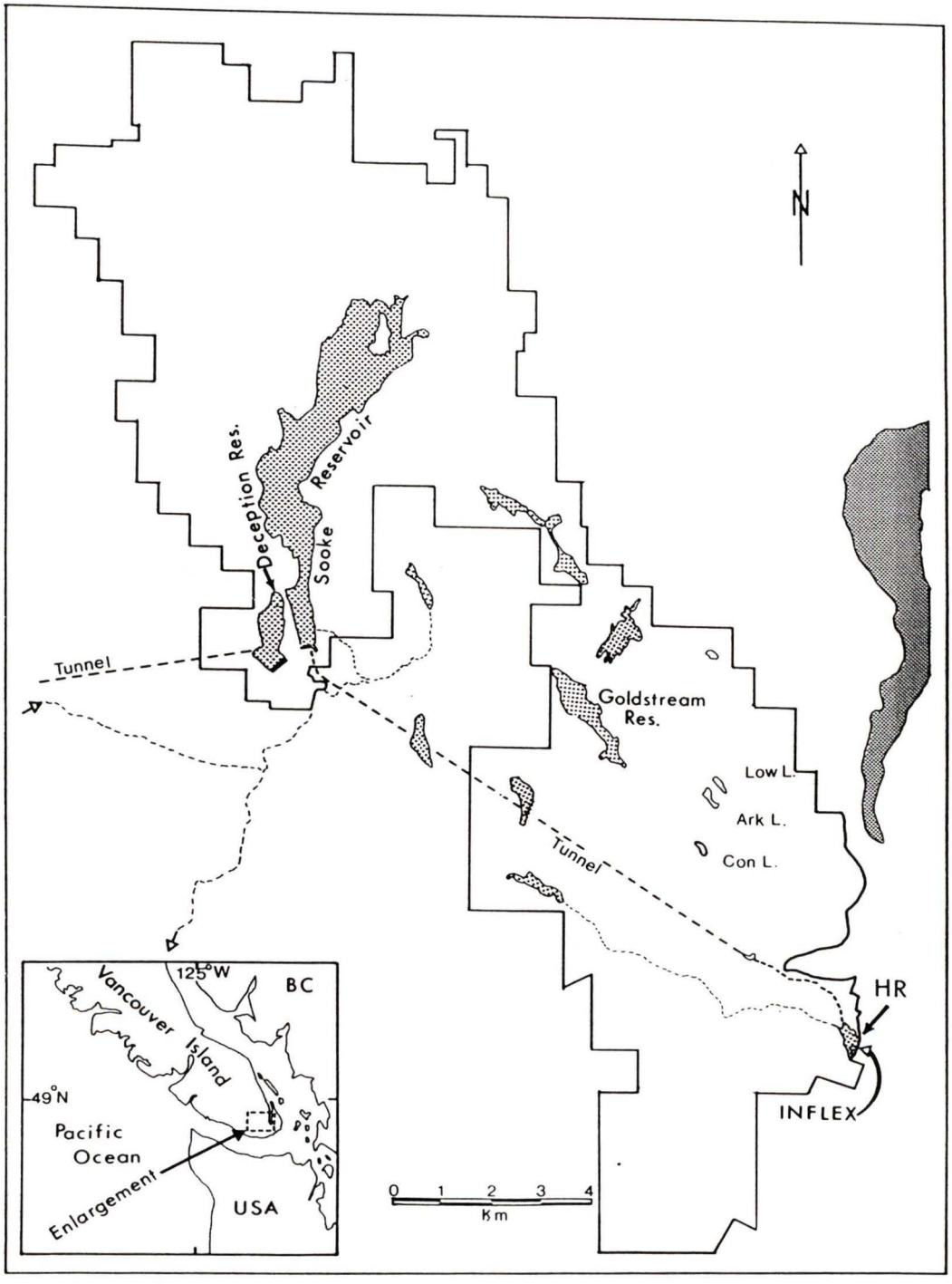
| | | |
|---|---|---|
| Bac | Achmic | 13890 |
| Bac | SynspB | 3834 |
| Bac | Tabfen | 3322 |
| Cya | Osc1sp | 3065 |
| Chl | Mou1sA | 1934 |
| Bac | SynspA | 1568 |
| Chl | Mou1sB | 931 |
| Chl | Ulo1sp | 822 |
| Chl | Gemspp | 695 |
| Chl | GemspA | 504 |
| Cya | Mer1sp | 452 |
| Bac | Melspp | 369 |
| Bac | Fracro | 305 |
| Bac | Tabflo | 294 |
| Chl | Ulo1sA | 271 |
| Chl | Stispp | 184 |
| Bac | Fravir | 183 |
| Bac | SynspC | 169 |
| Chl | Spopla | 165 |
| Chl | Zygspp | 142 |
| Chl | Mou1sC | 111 |
| Chl | ZygspA | 77 |
| Bac | Gomoli | 64 |
| Chl | Mou1sD | 60 |
| Bac | Synfil | 59 |
| Bac | Diavul | 56 |
| Cya | Ana1sA | 50 |
| Bac | MelspA | 50 |
| Bac | CymspA | 48 |
| Chl | Mou1sp | 41 |
| Bac | Synulv | 39 |
| Bac | Fra2sB | 38 |
| Chr | Dindiv | 36 |
| Chl | Ped1sp | 35 |
| Bac | Nav1sp | 33 |
| Bac | GomsAs | 31 |
| Chl | Gemspp | 30 |
| Bac | Cymcym | 28 |
| Chr | Dinbav | 26 |
| Chl | OedspA | 26 |
| Chl | Stibac | 23 |
| Chl | Cosbly | 21 |
| Chl | LGballs | 21 |
| Bac | Eunrob | 18 |
| Bac | Gomavc | 17 |
| Chl | OocspA | 15 |
| Bac | Gomsps | 14 |
| Chl | Ankspi | 12 |
| Bac | Fra2sA | 12 |
| Pyr | PerspA | 12 |

| | | |
|---|---|---|
| Bac | Astfor | 11 |
| Chl | Cospun | 11 |
| Bac | Gomols | 11 |
| Pyr | Perpus | 10 |
| Chl | Crurec | 9 |
| Chl | Eudspp | 9 |
| Chl | Micspp | 8 |
| Bac | Eun1sp | 7 |
| Chl | Oocspp | 7 |
| Bac | Rhogib | 7 |
| Chl | Scespp | 7 |
| Chl | Staana | 7 |
| Chl | Bulspp | 6 |
| Bac | Cymspp | 6 |
| Chl | Dicspp | 6 |
| Bac | Fravir | 6 |
| Chl | SpispA | 6 |
| Pro | Vorspp | 6 |
| Chl | Gonmon | 5 |
| Chl | Stamin | 5 |
| Cya | Ana1sp | 4 |
| Chl | Ankspp | 4 |
| Chl | Aphspp | 4 |
| Chl | Clospp | 4 |
| Bac | Cocspp | 4 |
| Chl | Cossub | 4 |
| Chl | Ankfal | 3 |
| Chl | Cosbio | 3 |
| Bac | Cycbod | 3 |
| ??? | GBF | 3 |
| Bac | Nav1sA | 3 |
| Chl | Oedspp | 3 |
| Bac | Coodim | 2 |
| Chl | Cosbin | 2 |
| Chl | Cosmon | 2 |
| Chl | Euaden | 2 |
| Bac | Gomacu | 2 |
| Bac | Gomcon | 2 |
| Bac | GomspB | 2 |
| Bac | Gomspp | 2 |
| Cya | Nosspp | 2 |
| Bac | Pinspp | 2 |
| Chl | Spispp | 2 |
| Chl | Sta3de | 2 |
| Chl | Stalum | 2 |
| Chl | StaspA | 2 |
| Chl | Xanant | 2 |
| Chl | Artcon | 1 |
| Chl | Clocyn | 1 |
| Chl | ClospA | 1 |

Table 9. Continued. Listing of algal species in decreasing numerical abundance enumerated at the end of the velocity experiment (October 21, 1987). See Table 4 for details of the experimental design.

| | | |
|---|---|---|
| Chl | Cosabb | 1 |
| Chl | Cosgra | 1 |
| Bac | Cyccom | 1 |
| Chl | Dicpul | 1 |
| Bac | Dipell | 1 |
| Bac | Frurho | 1 |
| Bac | Gomave | 1 |
| Bac | Gyr1sp | 1 |
| Chl | Miapin | 1 |
| Bac | Nitlin | 1 |
| Bac | Pinstr | 1 |
| Bac | Rhieri | 1 |
| Chl | Scearm | 1 |
| Chl | Scequa | 1 |
| Chr | Sphavo | 1 |
| Bac | Sta2sp | 1 |
| Chl | Sta3cu | 1 |
| Chl | Staarc | 1 |
| Chl | Staoph | 1 |
| Chl | StaspC | 1 |
| Bac | SurspA | 1 |
| Bac | Surspp | 1 |
| Chl | Xansub | 1 |

Figure 1. The Greater Victoria Water District's two principal watersheds of the Goldstream and Sooke rivers and their reservoirs (Goldstream, Deception and Sooke). The latter supplies, via the Kapoor Tunnel, Humpback Reservoir (HR), from which water was siphoned to supply the surrogate stream (IN Field Laboratory EXperiment – INFLEX) system. Inset: shows location of watersheds relative to Southern Vancouver Island, British Columbia, Canada.


N
Deception Res.
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Sooke
Tunnel
Goldstream
Res.
Low L.
Ark L.
Con L.
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BC
Vancouver Island
49°N
Pacific
Ocean
Enlargement
USA

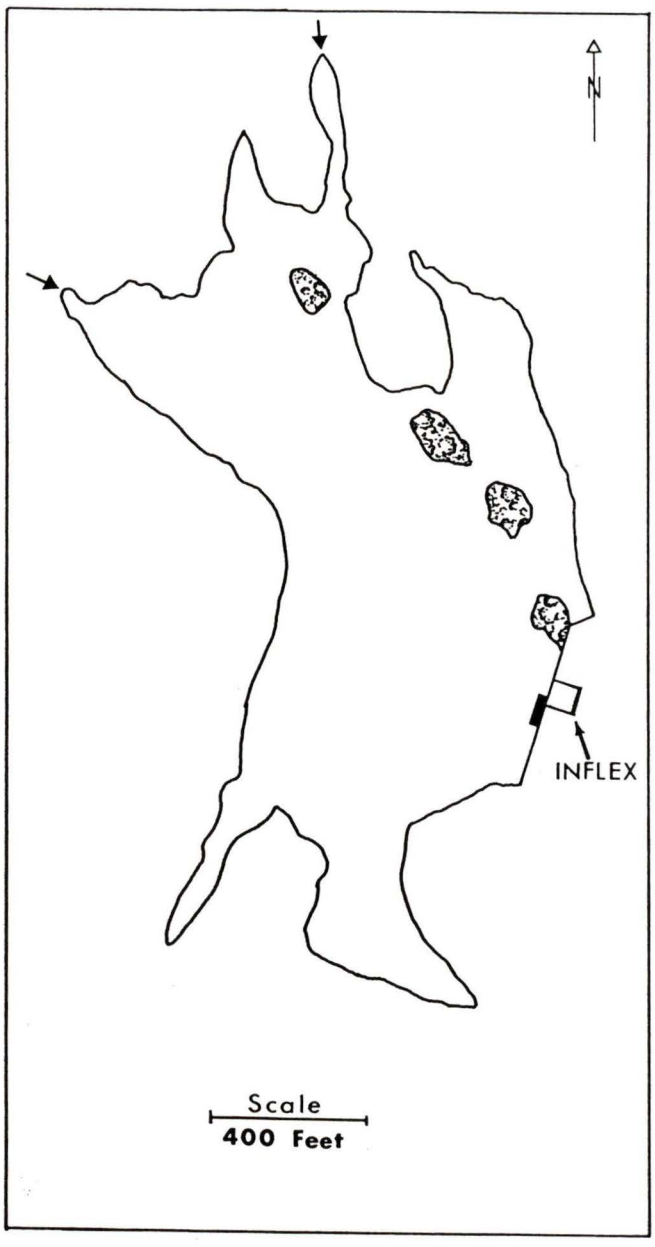
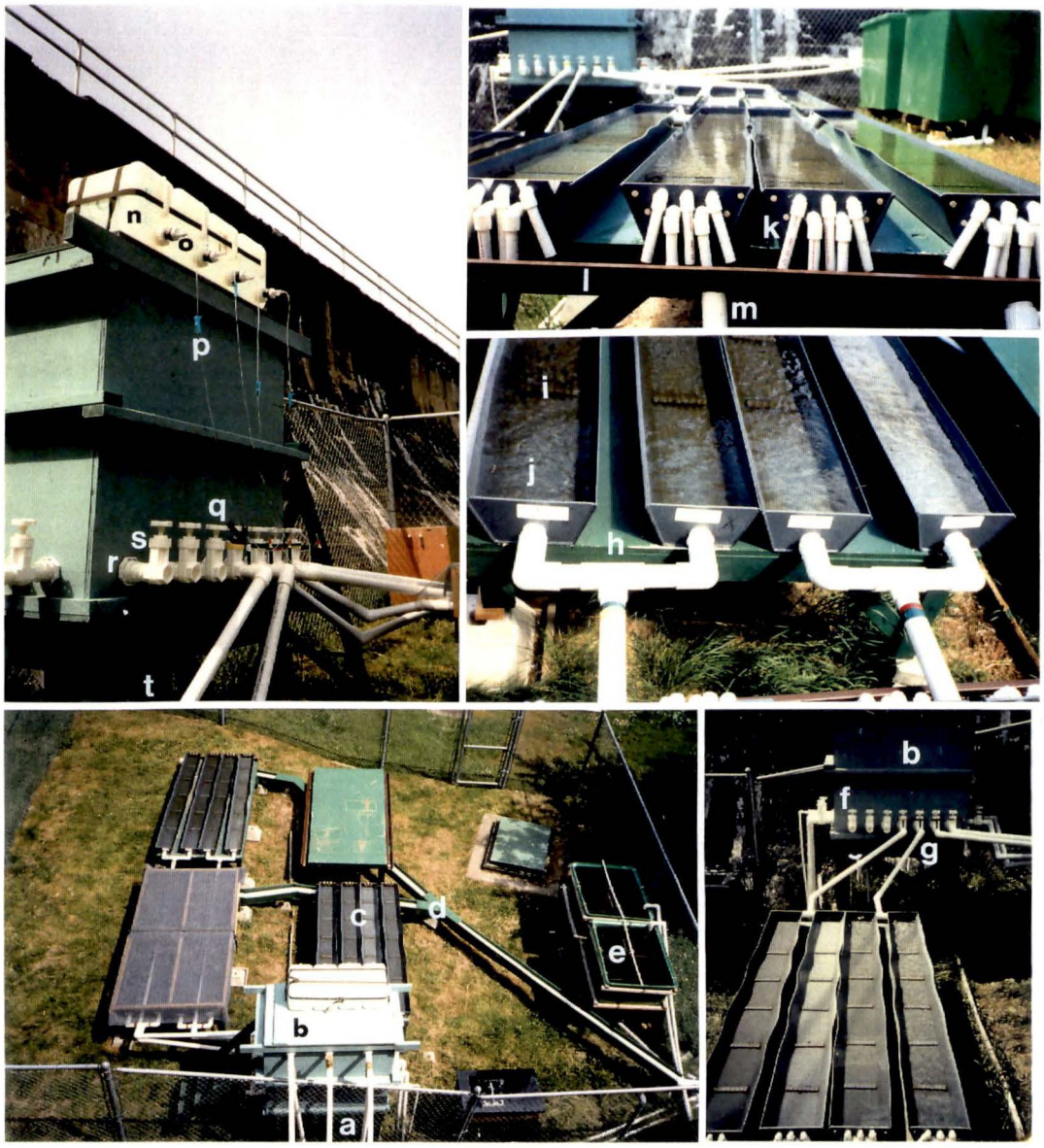



Figure 2. Humpback reservoir showing location of the INFLEX facility adjacent to the dam; also shown are a number of small islands and the two principal inlet sources of water (arrows). The domestic water supply flows from the reservoir immediately south of the INFLEX site; the solid black rectangle adjacent to the dam houses the potable water supply intake screens.

Figure 3. The surrogate stream (INFLEX) system. Composite photos designated as L = lower and left; M = middle; R = right; U = upper. The water is siphoned over the dam (a; LL) into the header box (b), from which it is metered (f; LR) through supply manifolds (g; LR), to the paired (h; MR) streams (c; LL). Paired streams receive water from a single manifold (g) split into two inlet pipes (h) immediately prior to discharge into the streams. Nutrient dosing of each stream consists of the nutrient reservoir containing concentrated nutrient (n; UL), the "gavage" drip-feed metering system (o), delivery line (p) and an injector port (q). Header box discharge is metered (f) via gate valves (s; UL) affixed via sealed through-hull fittings (r). The over flow from the header box (t) is discharged into the common drain (d; LL). Artificial substrata are attached to the stream (i; MR) at fixed intervals; note the clear area immediately downstream of the inlet region (j). The streams discharge into an exhaust-pipe complex (k; UR), which draws water from a cross-section of each stream. Water then discharges into a stream (l) linking all troughs on each support table; each exhaust stream drains (m) into a linked common drain (d; LL); the latter discharges into a common conduit which receives backflush water from a cleaning device which filters the G.V.W.D. water supply. The water from this common drain then flows through a small marsh prior to entering a first order stream which ultimately drains into the Bilston/Metchosin Creek. Two additional tanks (e; LL) containing large artificial substrata were used for cultivating periphyton biomass.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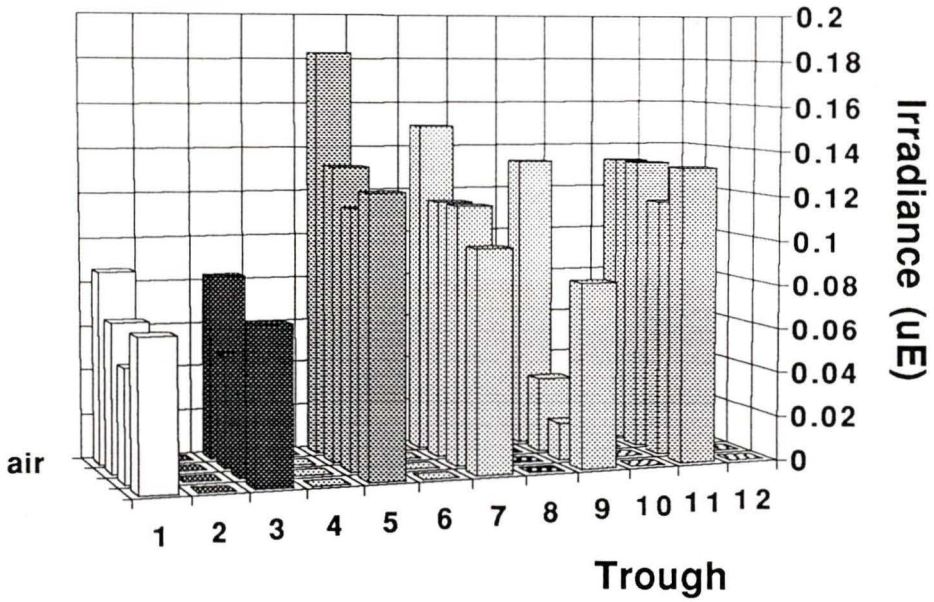



Figure 4. Summer (May, 1988) irradiance values within streams (numbered 1 - 12). Left hand axis (reading from front to back) indicates location at which irradiance measurements were obtained: at outflow, between slides, inflow and in the air.

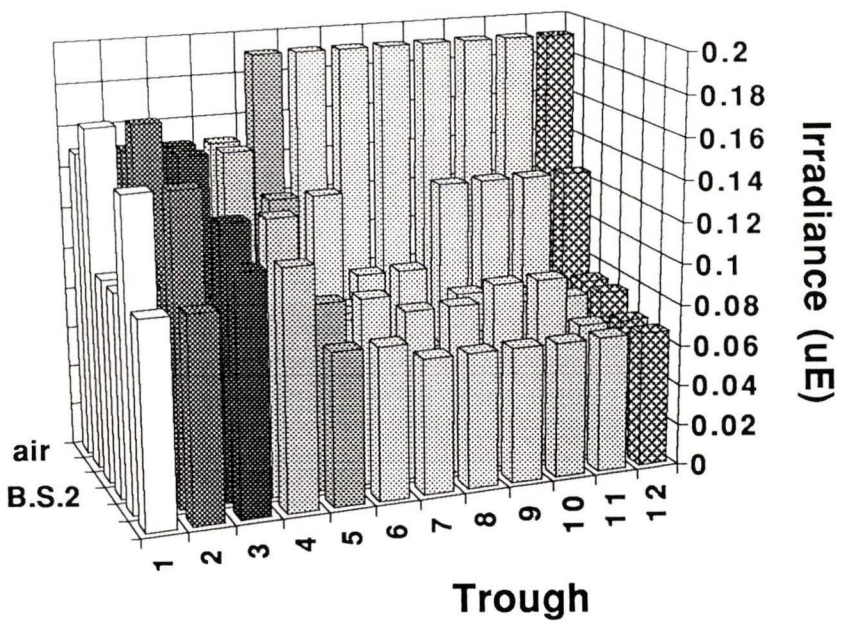



Figure 5. Summer (July, 1988) irradiance values within streams (numbered 1 - 12). Left hand axis (reading from front to back) indicates location at which irradiance measurements were obtained: rocks (closest), styrofoam, between slides (B.S. 1 & 2), inflow and in the air.

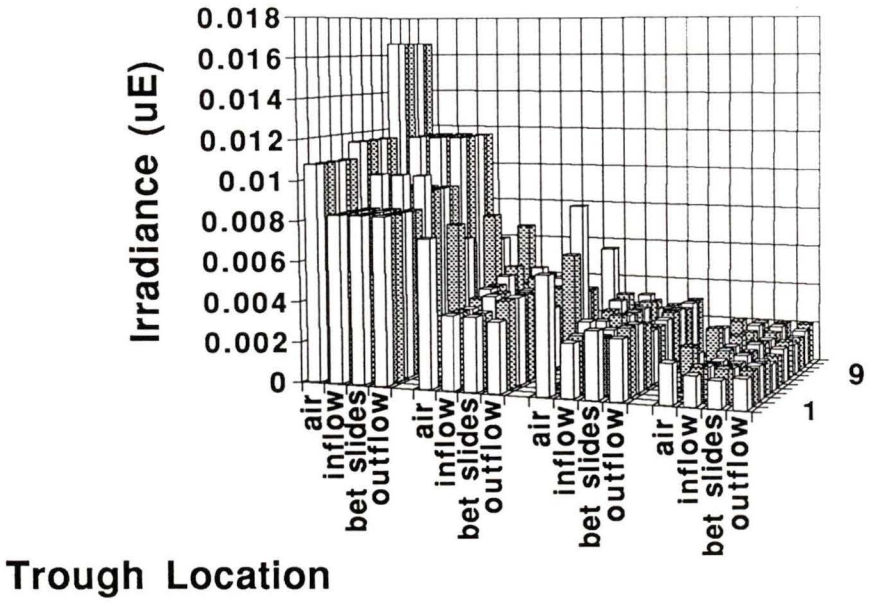



Figure 6. Winter (December, 1987) irradiance values within streams (right hand axis numbered from front to back 1 - 12, respectively). The X axis indicates the location at which the measurements were obtained. Measurements were obtained (from left to right) on December 12, 16, 22 and 29.

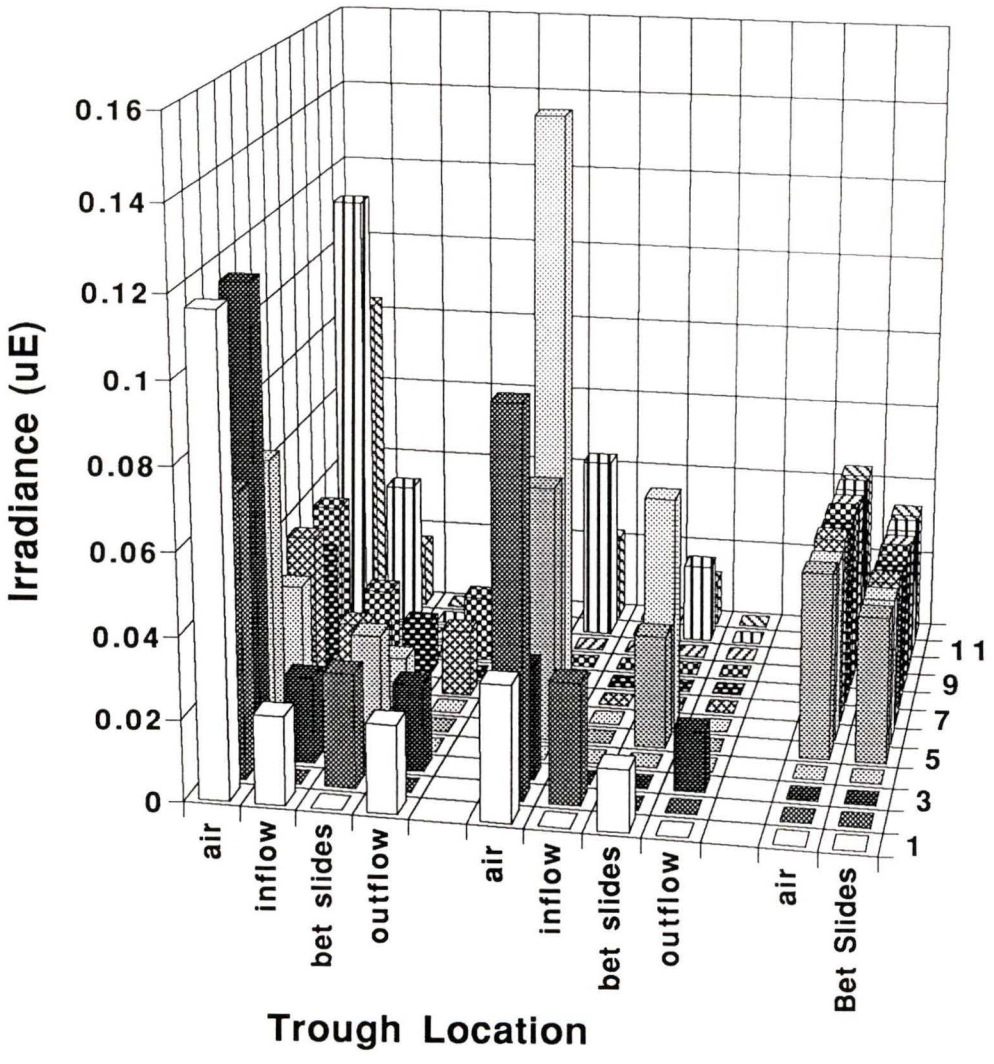



Figure 7. Spring (January) irradiance values within streams (right hand axis numbered from front to back 1 - 12, respectively). The X axis indicates the location at which the measurements were obtained. Three sets of measurements (from left to right) were obtained on January 6 and 20 and February 8, 1988.

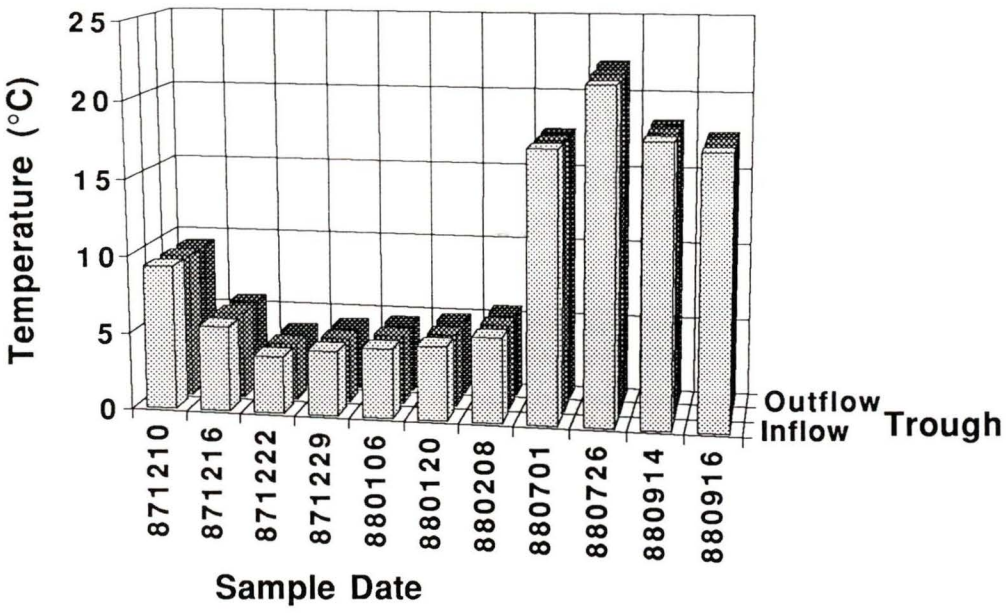



Figure 8. Temperature measurements within streams measured at three locations (right hand axis; front to back): inflow, mid-stream and outflow. Sample dates listed as year, month and day.

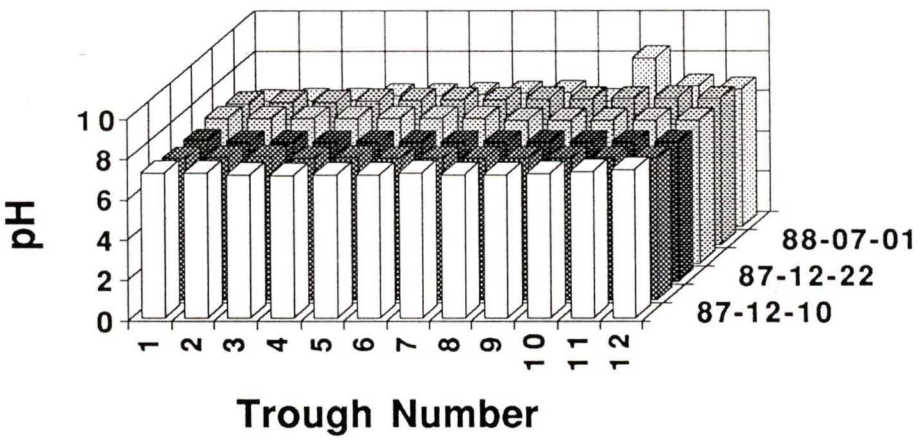



Figure 9. Seasonal hydrogen ion concentrations (pH) measured within the streams (x axis; numbered 1 - 12); sample dates were (right hand axis; front to back) 871210, 871216, 871222, 871229, 880701, 880726, 880916. Sample dates listed as year, month and day.

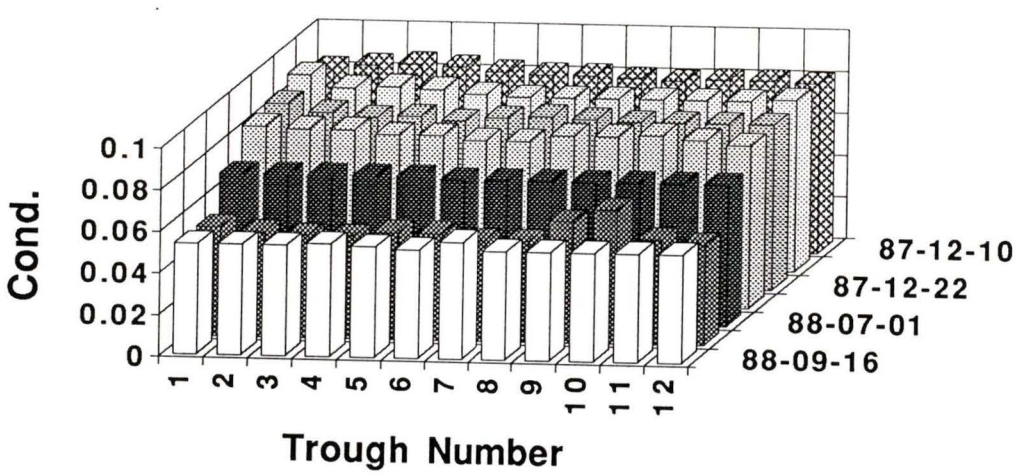



Figure 10. Conductivity (μS $cm^{2-1}$) values measured within the streams (x axis; numbered 1 - 12); sample dates were (right hand axis; back to front) 871210, 871216, 871222, 871229, 880701, 880726, 880916. Sample dates listed as year, month and day.

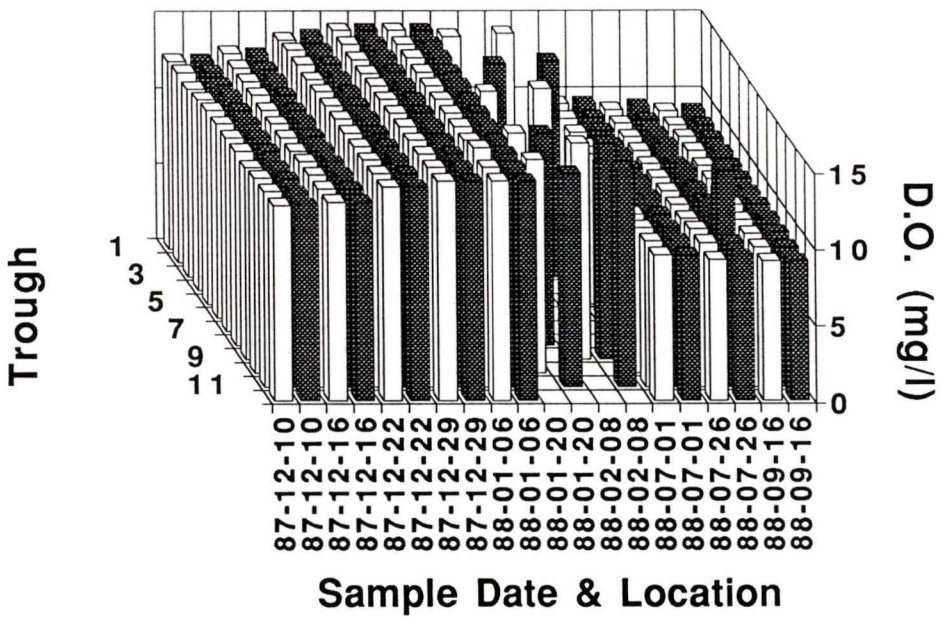



Figure 11. Dissolved oxygen values within the streams (left hand axis; numbered 1 - 12 back to front); for each sample date there were paired measurements taken at the inflow and outflow (left and right, respectively) for each stream. Sample dates listed as year, month and day.

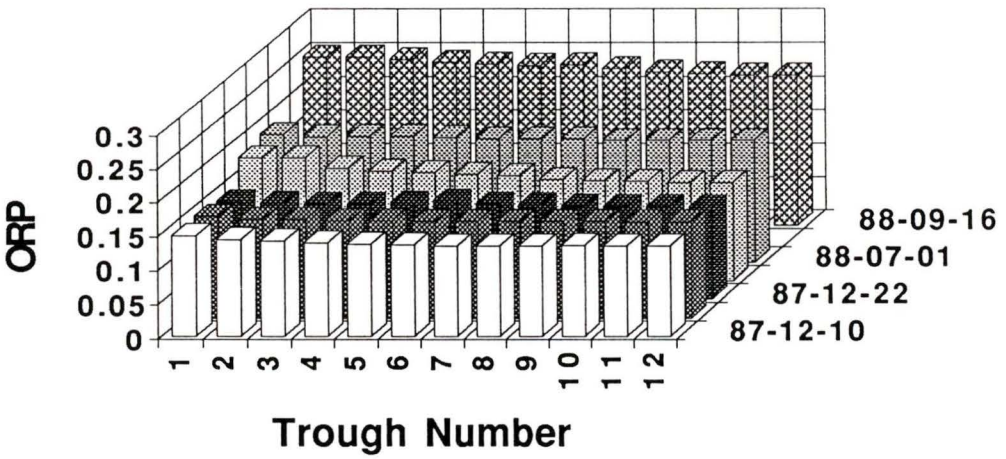



Figure 12. Oxidation-reduction potential (ORP) measured within the streams (at the outflow); sample dates were (right hand axis; front to back) 871210, 871216, 871222, 871229, 880701, 880916. Sample dates listed as year, month and day.

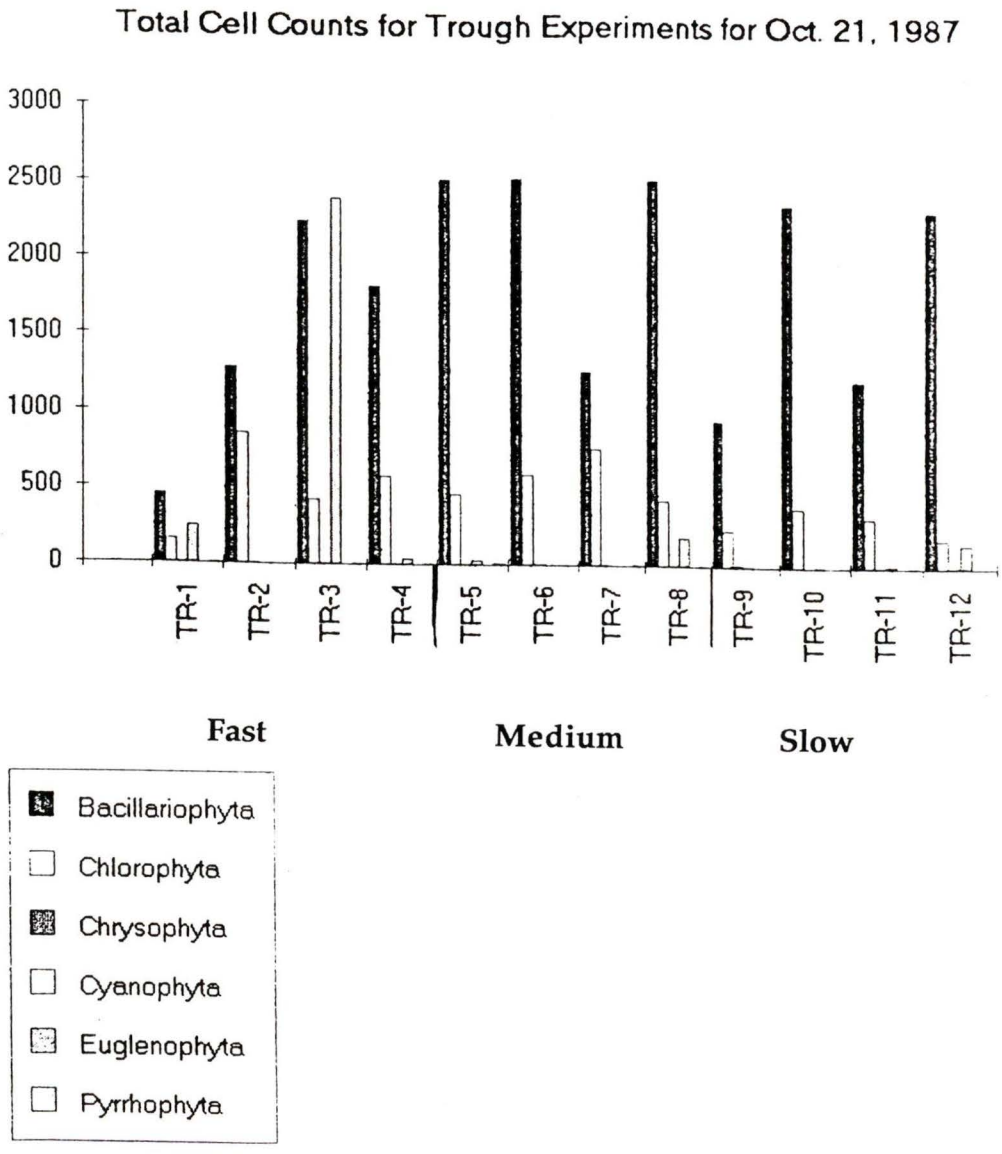



Figure 13. Total cell counts, for major taxonomic groups, at the conclusion of the autumn velocity experiment (October 21, 1987). Stream flow rates were: troughs 1-4 – 28 cm/sec; troughs 5-8 – 14 cm/sec; and troughs 9-12 – 7 cm/sec.

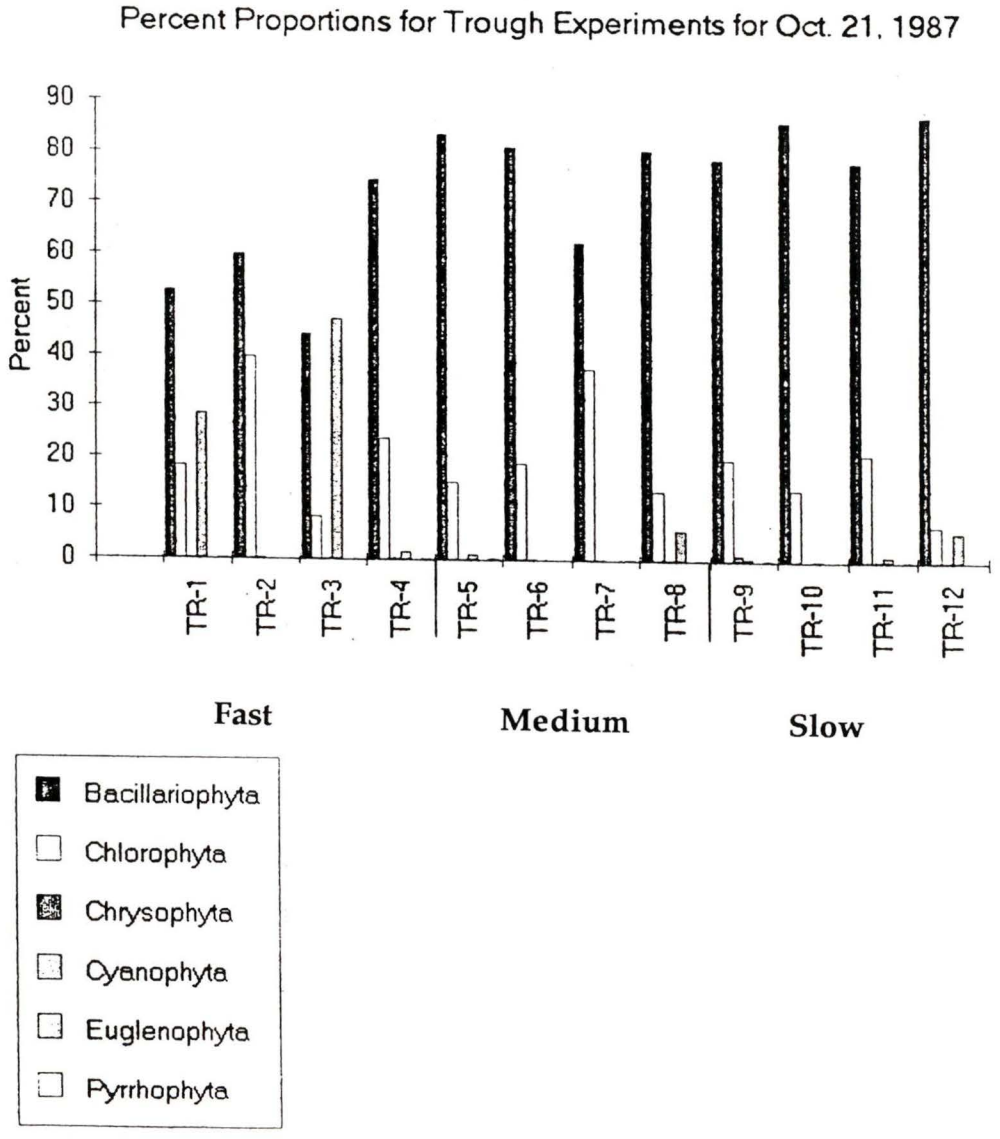
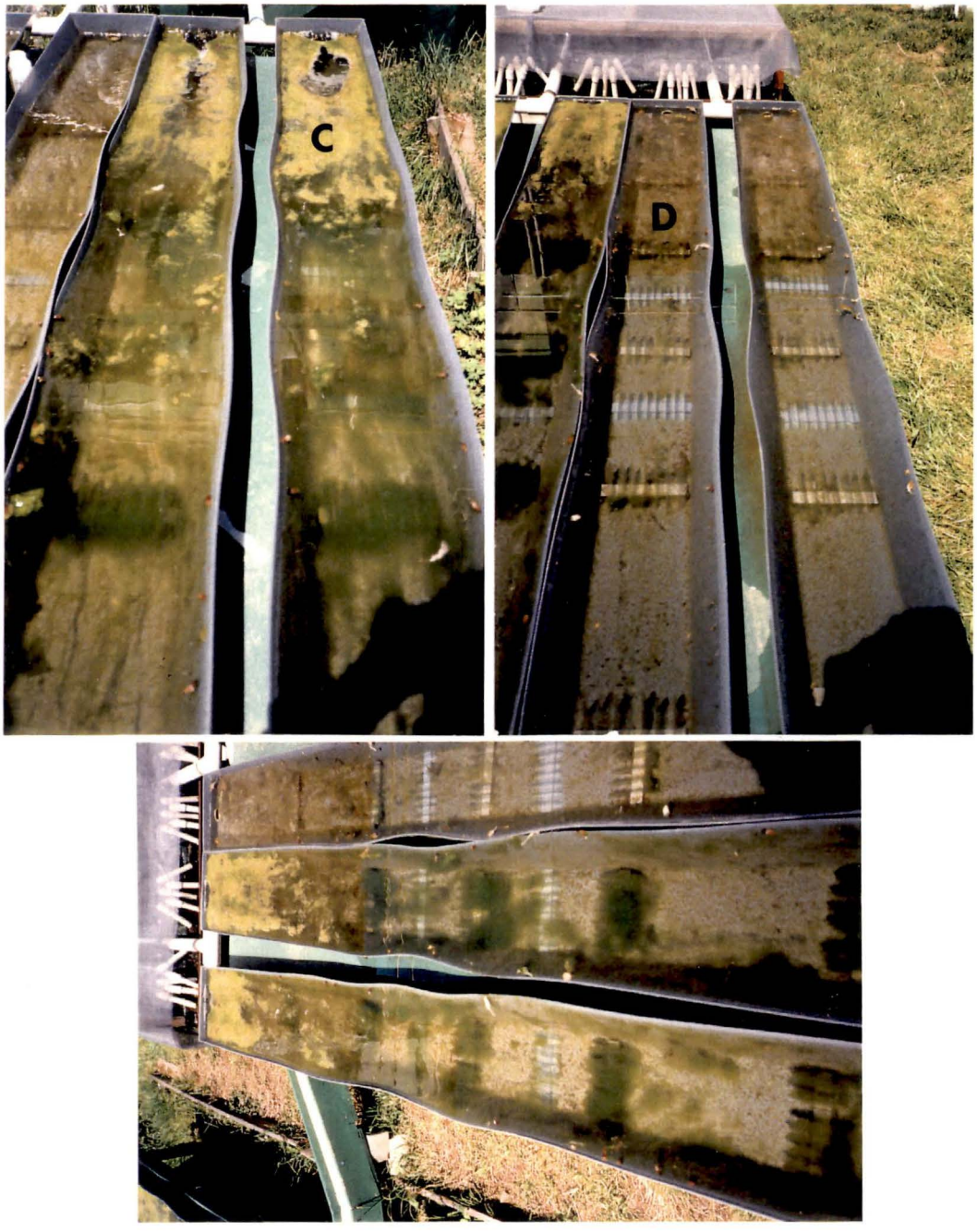



Figure 14. Percentages and numerical abundance values of major taxonomic groups at the end of the autumn velocity experiment (October 21, 1987). Stream flow rates were: troughs 1-4 – 28 cm/sec; troughs 5-8 – 14 cm/sec; and troughs 9-12 – 7 cm/sec.

Figure 15. Photographs showing the Chlorophyte canopy (C) and diatomaceous (D) understory. Note the heavier accrual below the inlet section of the trough. Maximum accrual occurred on the slides.


C
D

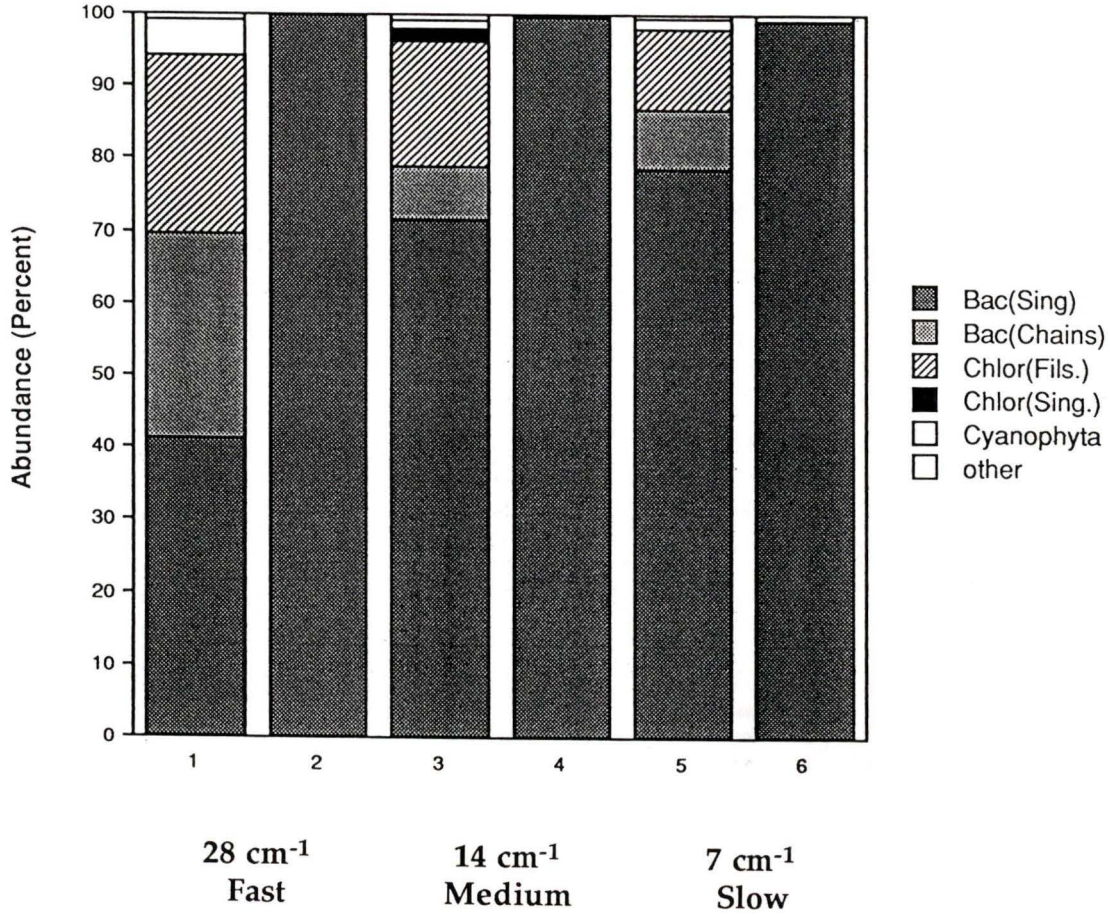



Figure 16. Percentage abundance values of major taxonomic groups contained within the streams upon conclusion of the velocity experiments. Each stream # represents the data from the four streams run at each velocity, where:

| Velocity (cm/sec) | Stream # | |
|---|---|---|
| | Autumn | Winter |
| 28 | 1 | 2 |
| 14 | 3 | 4 |
| 7 | 5 | 6 |

Autumn (October 21, 1987); Winter (February 9, 1988). (After Ridley-Thomas, 1989).

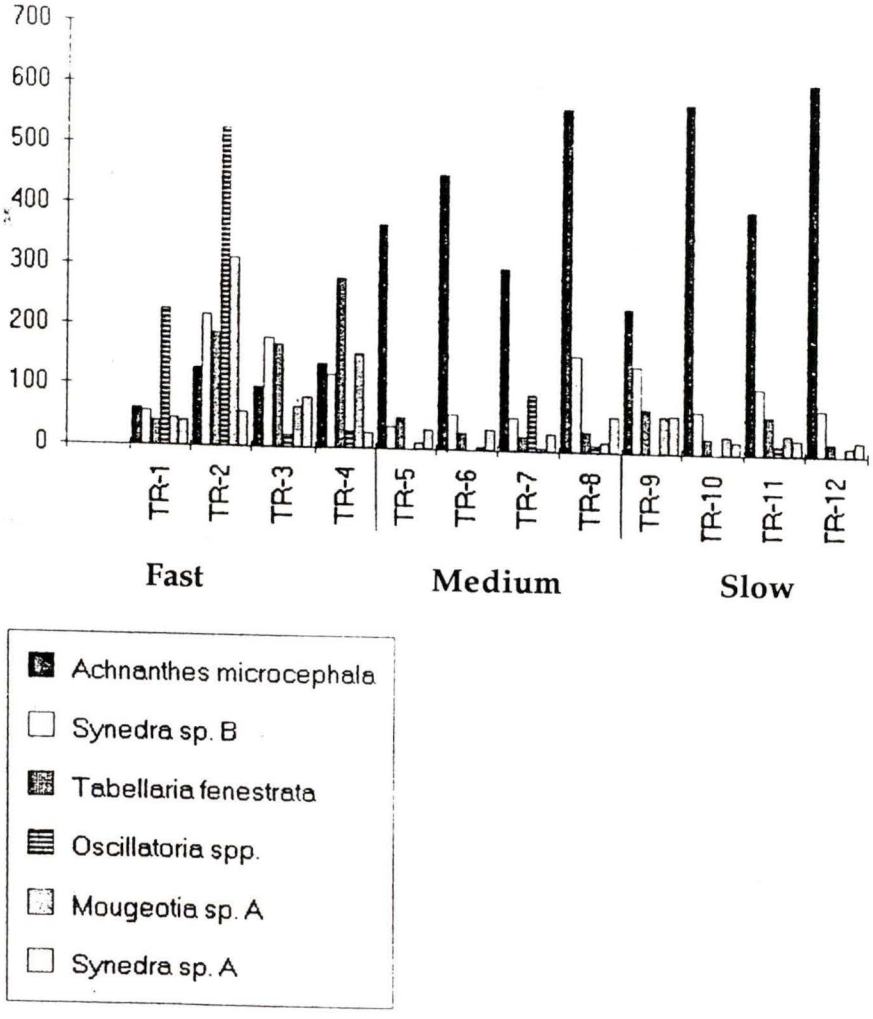



Figure 17. Cell counts for the six most abundant taxa, upon completion of velocity experiment (October 21, 1987).

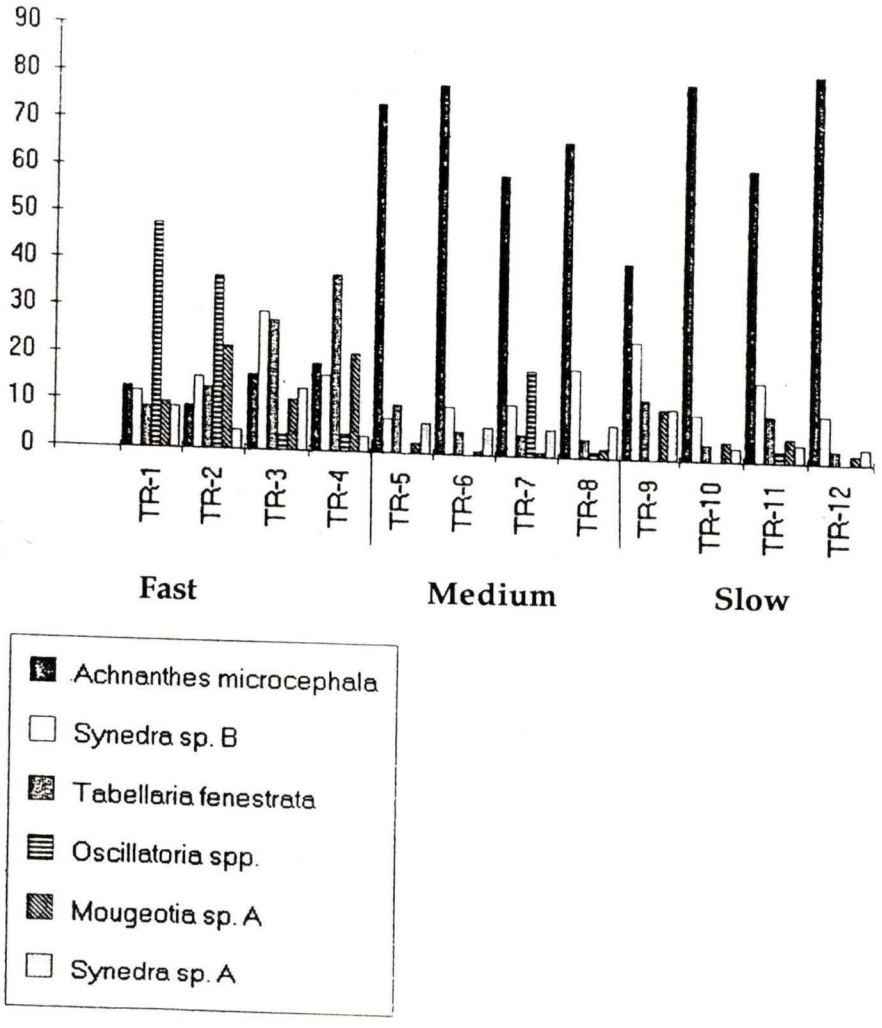



Figure 18. Relative proportions, expressed as percentage, of the six most abundant taxa, upon completion of the velocity experiment (October 21, 1987).

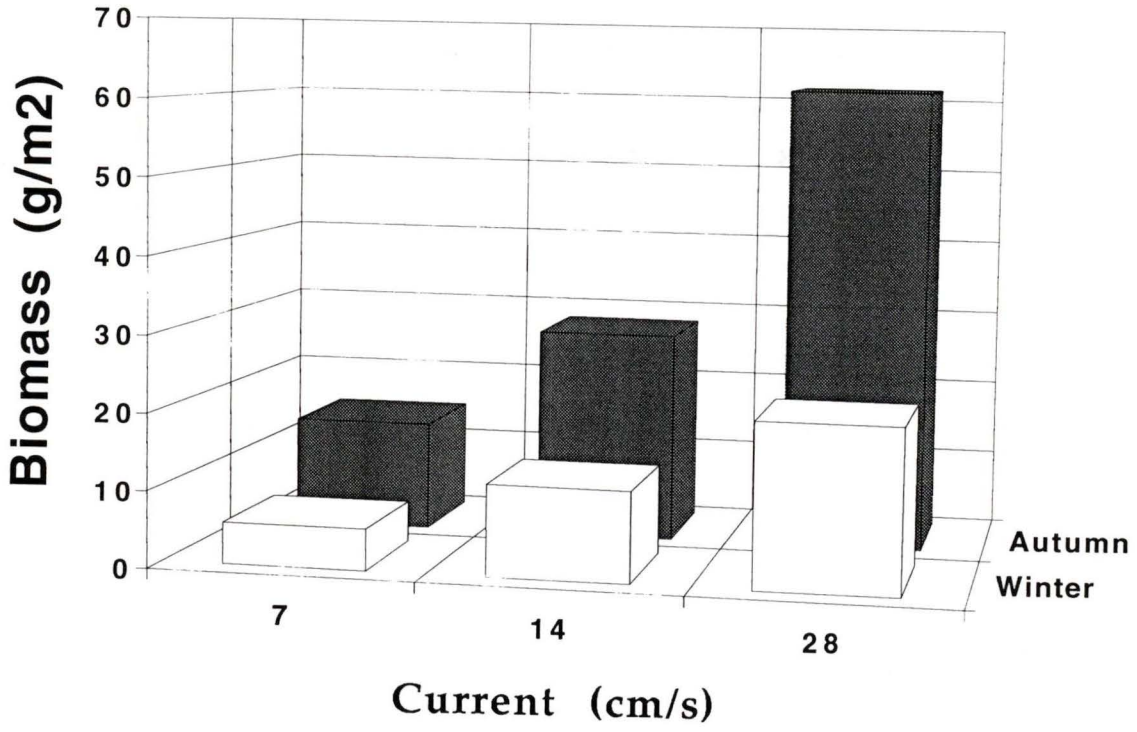



Figure 19. Autumn (October 21, 1987) and winter (February 9, 1988) biomass accrual at three velocities. (After Ridley-Thomas, 1989).

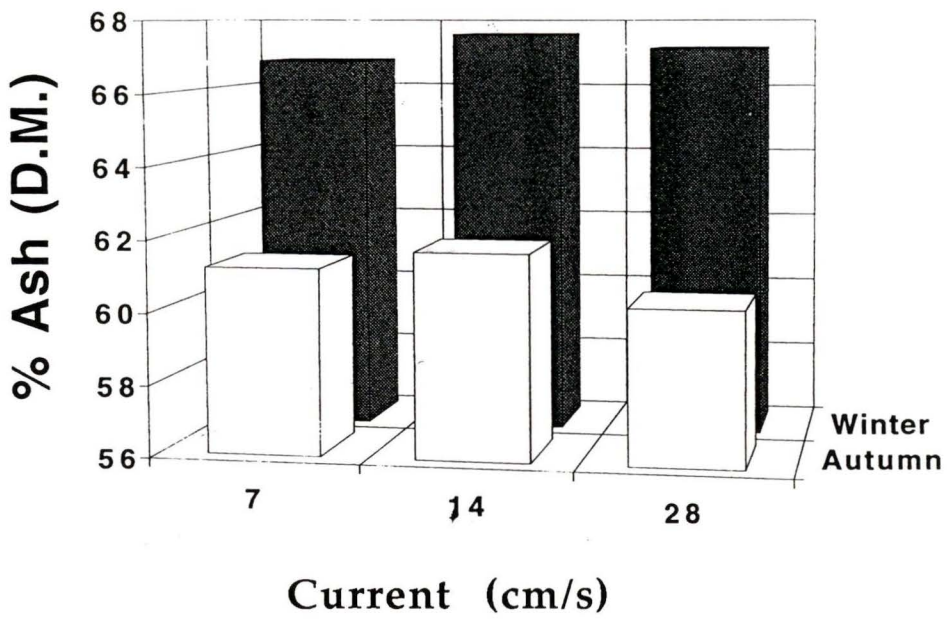



Figure 20. Percentage ash, expressed as dry mass, within mature periphyton communities, cultivated at three different velocities, during the autumn (October 21, 1987) and winter (February 9, 1988). (After Ridley-Thomas, 1989).

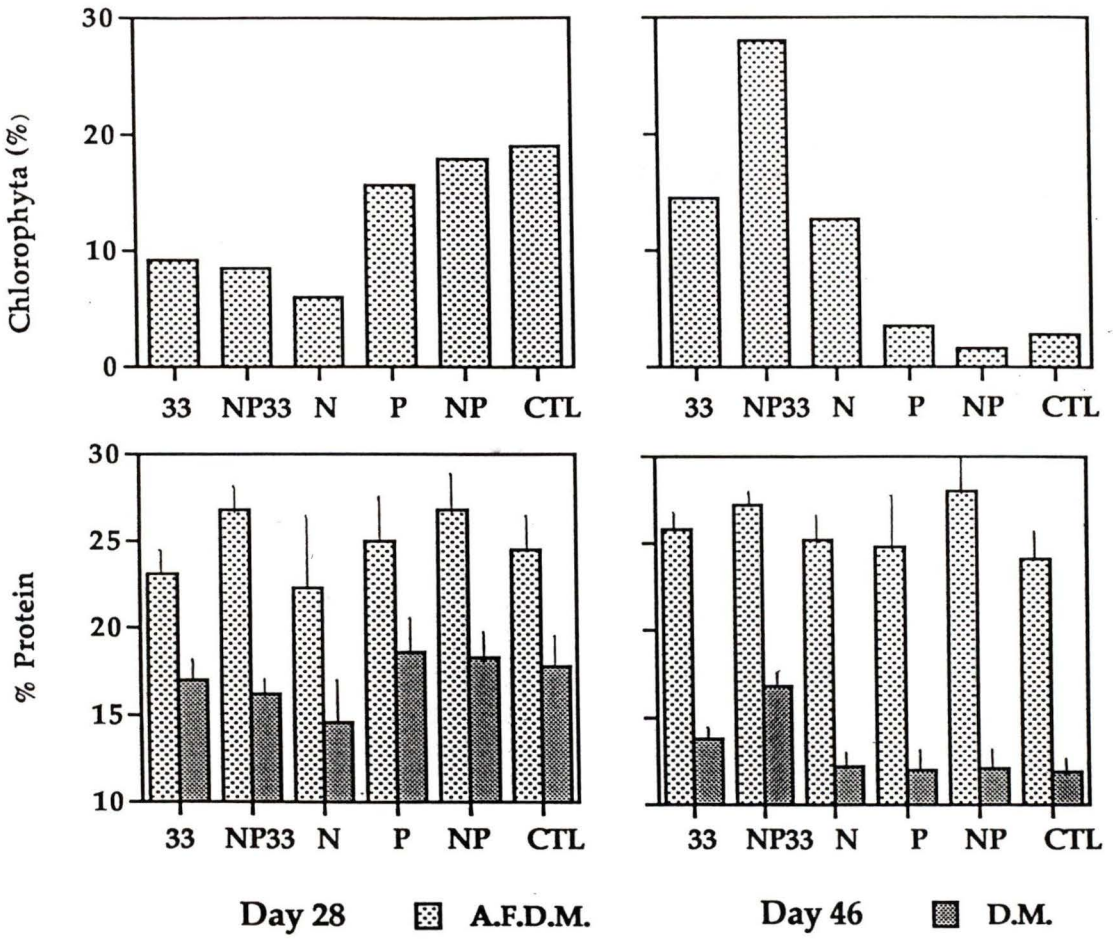
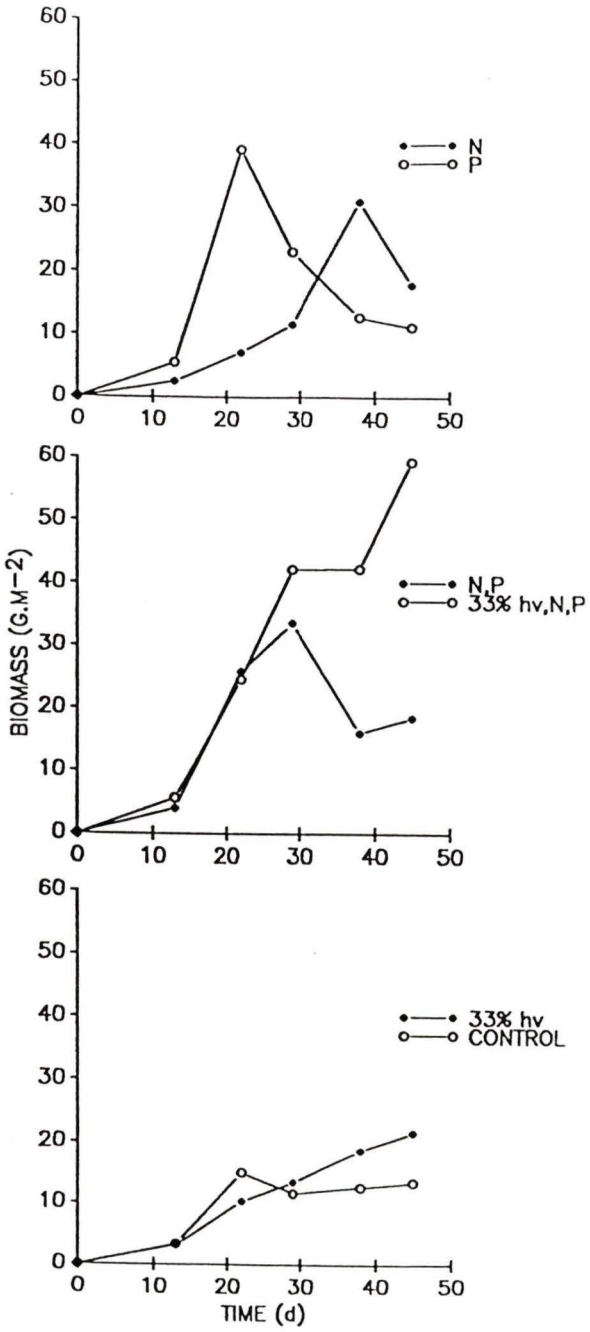



Figure 21. Percentage abundance of Chlorophyta (upper) and related protein (lower) as Ash Free Dry Matter (A.F.D.M.) and as Dry Mass (D.M.) under regimens of increased N and P, N+P, control, 33% hv and N+P at 33% hv within the streams. (After Ridley-Thomas, 1989).

Figure 22. Biomass accrual under regimens of increased N and P (upper), N+P and N+P at 33% hv (middle) and at background concentrations (control) under normal and reduced irradiance values (lower). (After Ridley-Thomas, 1989).


BIOMASS (G.M-2)
TIME (d)
N
P
N,P
33% hv,N,P
33% hv
CONTROL

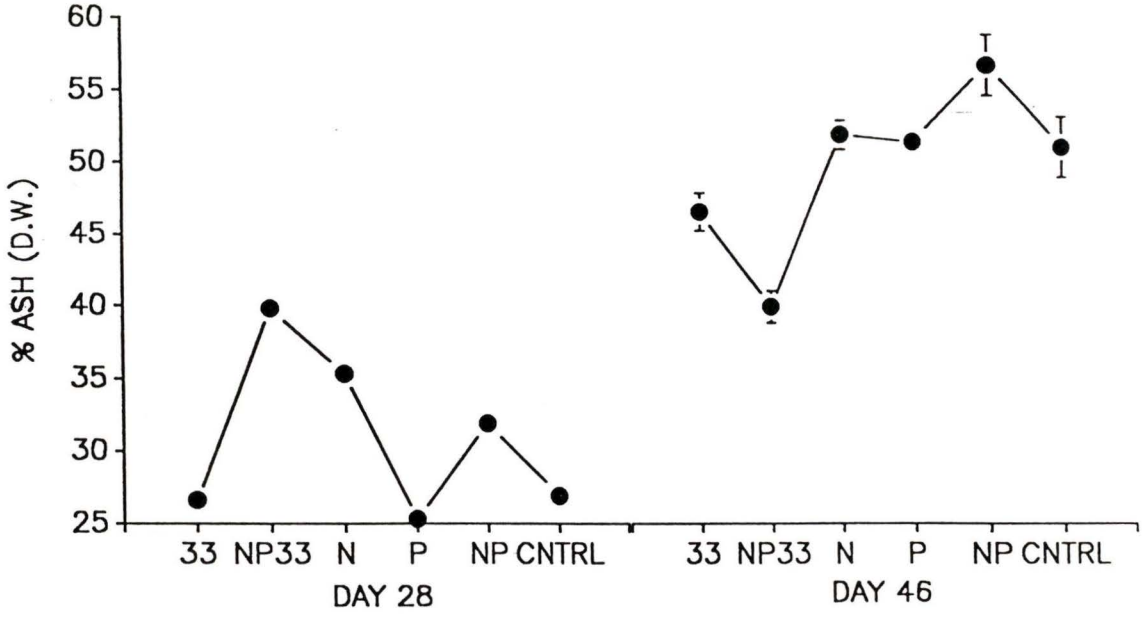



Figure 23. Percentage ash (dry mass) in periphyton for each treatment measured at day 28 and experiments end (day 45). (After Ridley-Thomas, 1989).

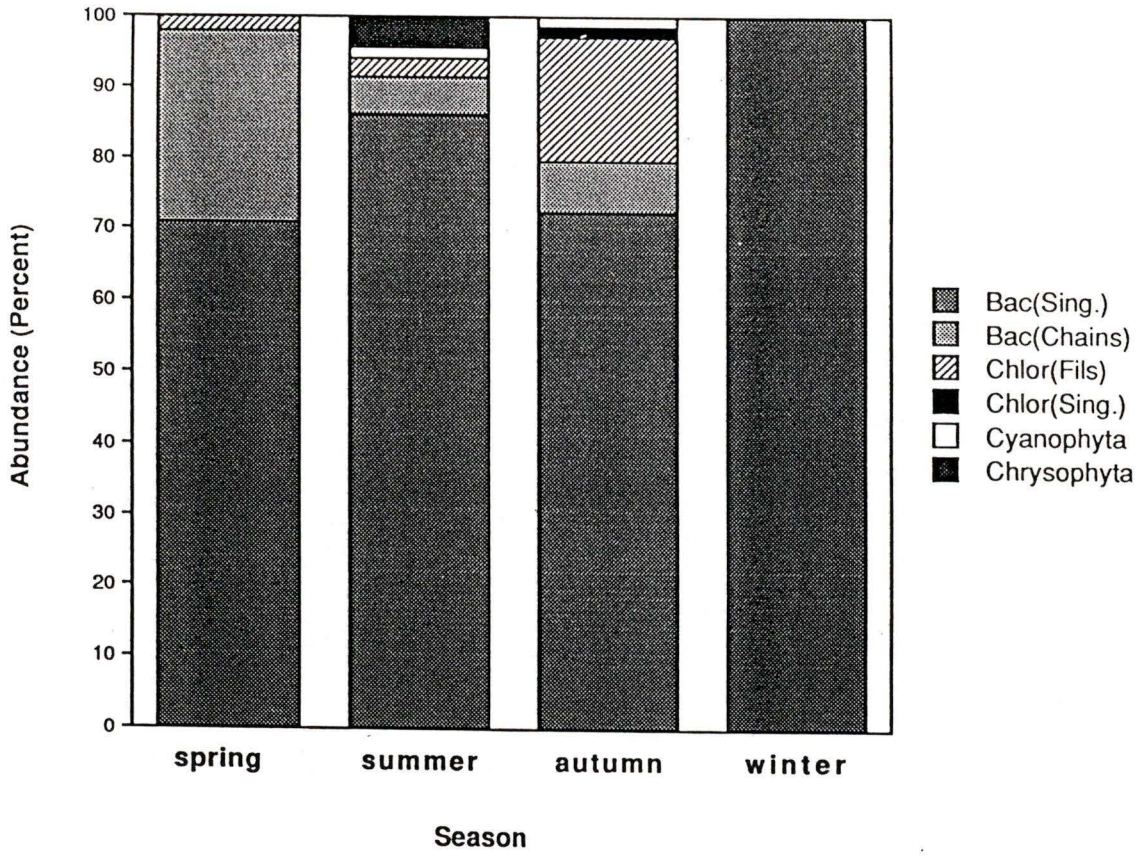



Figure 24. Percentage abundance values of major taxonomic groups contained within the surrogate streams at the end of the experimental runs, conducted in the four seasons. (After Ridley-Thomas, 1989).

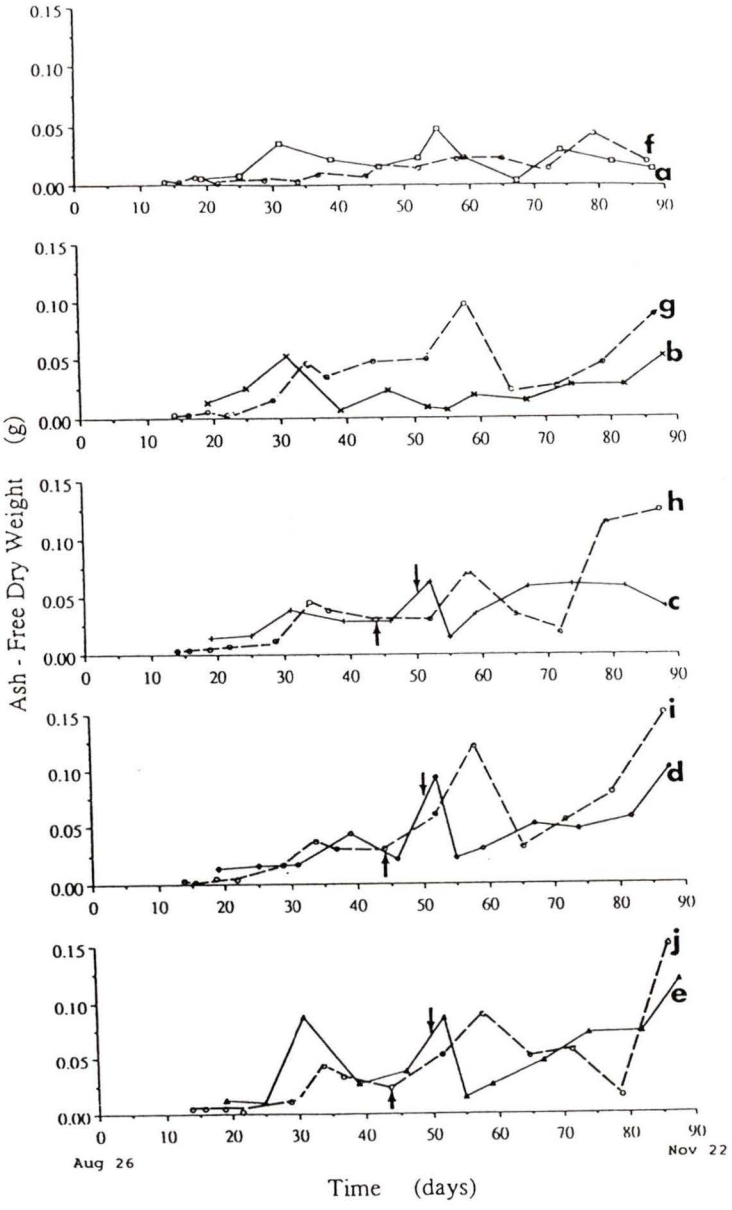



Figure 25. Accrued biomass, developed at the Humpback Reservoir INFLEX system, where a,f are controls (lake water only) and g,b are nutrient enriched. Two overlapping trials were conducted in which Glyphosate was added, to water flowing through the streams, at varying concentrations where: Experiment 1 = August 26, 1989 and Experiment 2 = September 25, 1989. Treatments were - lake water (a,f); nutrients (b,g); nutrients + Glyphosate (mg/L)(0.0019-c, 0.1347-d, 0.2874-e); (0.0011-h, 0.0896-i, 0.0977-j). Arrows indicate addition of Glyphosate. (see Austin *et al* 1991 for details.)

Figure 26. **Upper** Summer biomass estimates (measured as chlorophyll *a* ($\mu g/cm^2$)) of stream periphyton cultivated on borosilicate (glass) slides; streams numbered 1 - 12 (x axis); sample dates right hand axis (front to back) 880706, 880711, 880714, 880718, 880722, 880725. Sample dates listed as year, month and day.

**Lower** Phaeophytin ($\mu g/cm^2$) obtained from stream cultivated periphyton during the period 880706, 880711, 880714, 880718, 880722, 880725. Axis parameters as above.

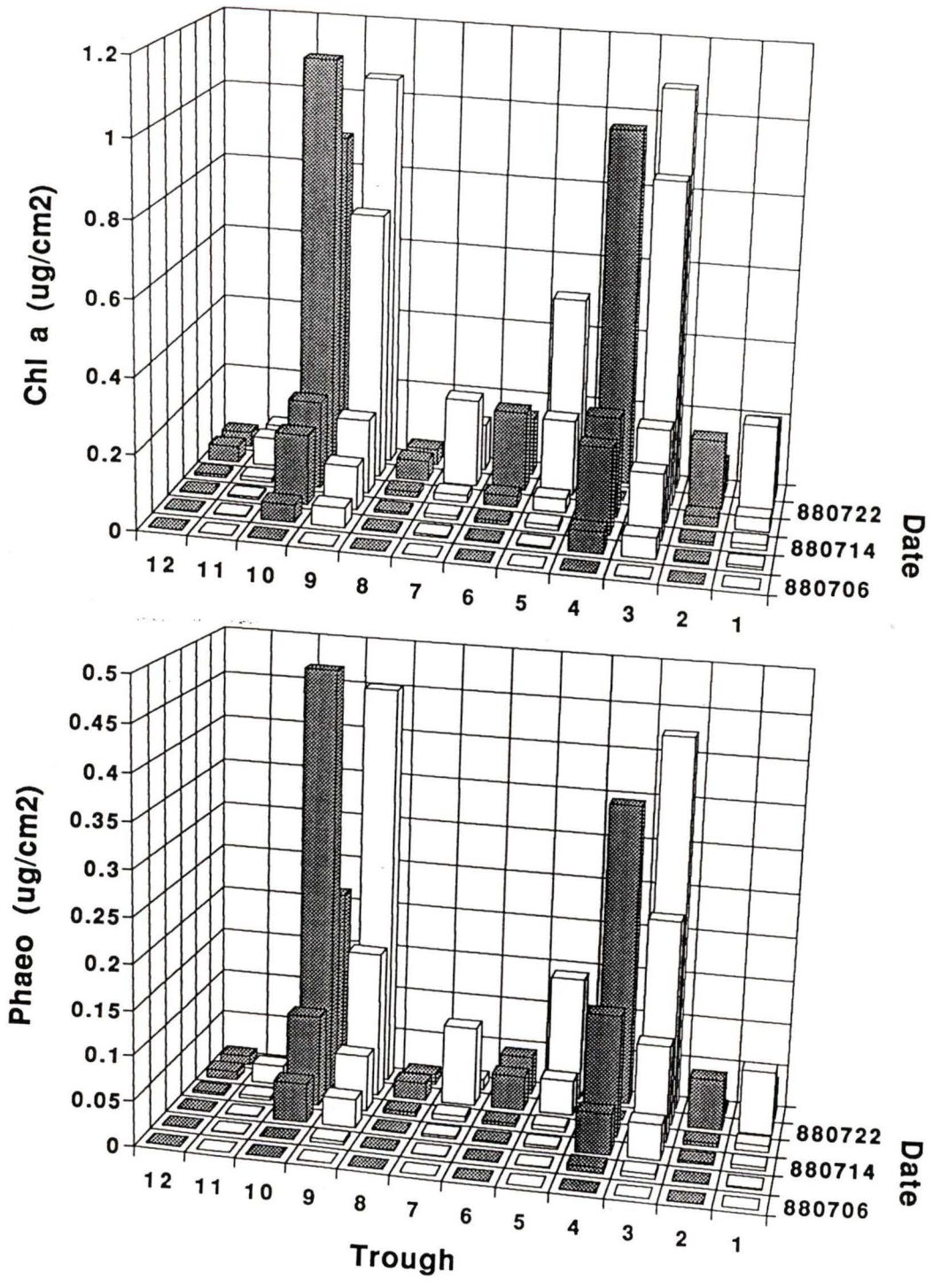

Chl a (ug/cm2)
1.2
1
0.8
0.6
0.4
0.2
0
12 11 10 9 8 7 6 5 4 3 2 1
880722
880714
880706
Date
Phaeo (ug/cm2)
0.5
0.45
0.4
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
12 11 10 9 8 7 6 5 4 3 2 1
Trough
880722
880714
880706
Date

Figure 27. **Upper** Autumn biomass estimates (measured as chlorophyll *a* ($\mu g/cm^2$)) obtained from stream cultivated periphyton, grown on borosilicate slides, during the period - 880822, 880818, 880829, 880902, 880908, 880914, 880929. Axis parameters as in Figure 26.

**Lower** Phaeophytin ($\mu g/cm^2$) obtained from stream cultivated periphyton, grown on borosilicate slides, during the period - 880822, 880818, 880829, 880902, 880908, 880914, 880929. Axis parameters as in Figure 26.

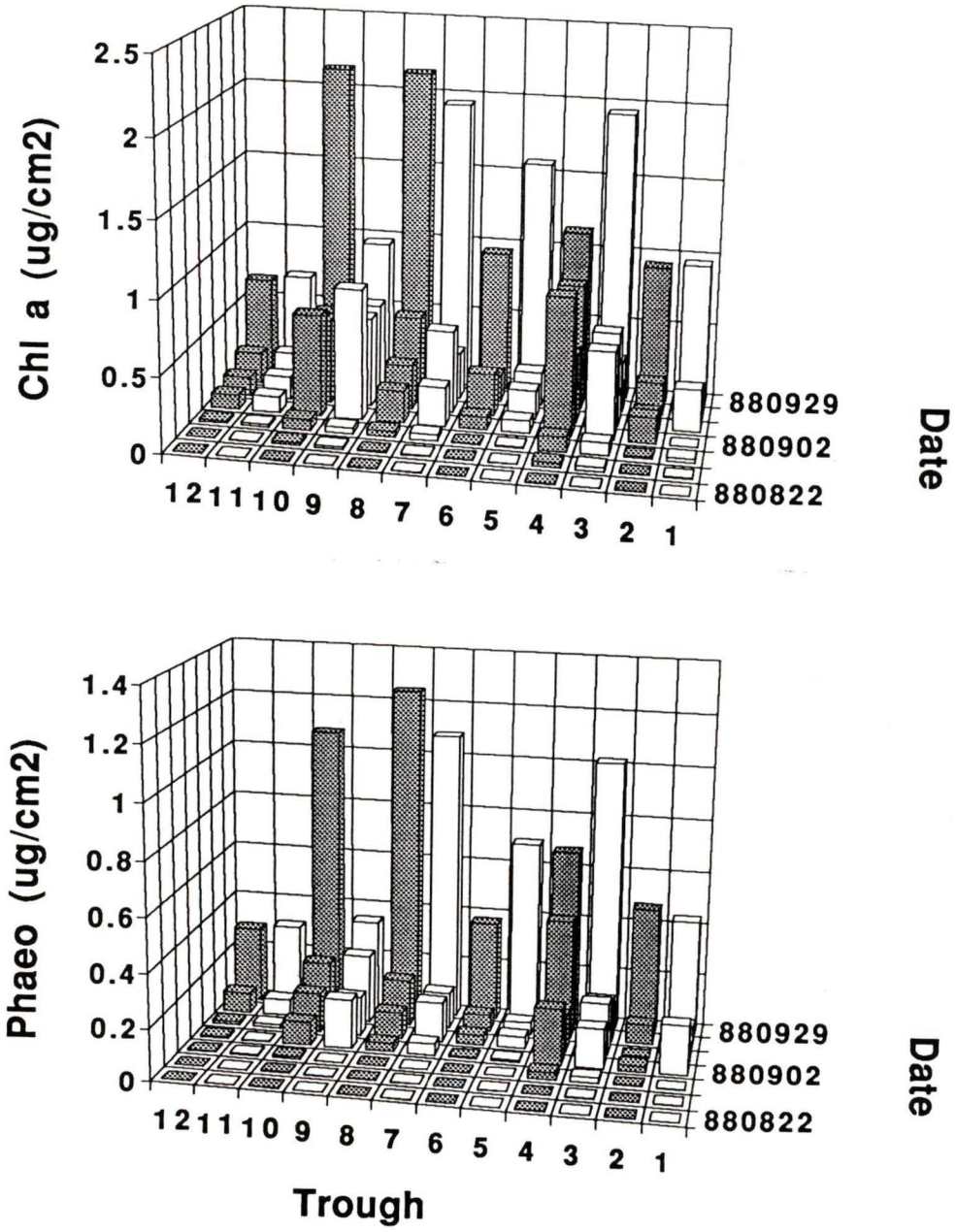

Chl a (ug/cm2)
2.5
2
1.5
1
0.5
0
12 11 10 9 8 7 6 5 4 3 2 1
880929
880902
880822
Date
Phaeo (ug/cm2)
1.4
1.2
1
0.8
0.6
0.4
0.2
0
12 11 10 9 8 7 6 5 4 3 2 1
Trough
880929
880902
880822
Date

Figure 28. **Upper** Chlorophyll *a* ($\mu g/cm^2$) obtained from stream cultivated periphyton, grown on styrofoam sheets, during the period - 880822, 880818, 880829, 880902, 880908, 880914, 880929. Axis parameters as in Figure 26.

**Lower** Phaeophytin ($\mu g/cm^2$) obtained from stream cultivated periphyton, grown on styrofoam sheets, during the period - 880822, 880818, 880829, 880902, 880908, 880914, 880929. Axis parameters as in Figure 26.

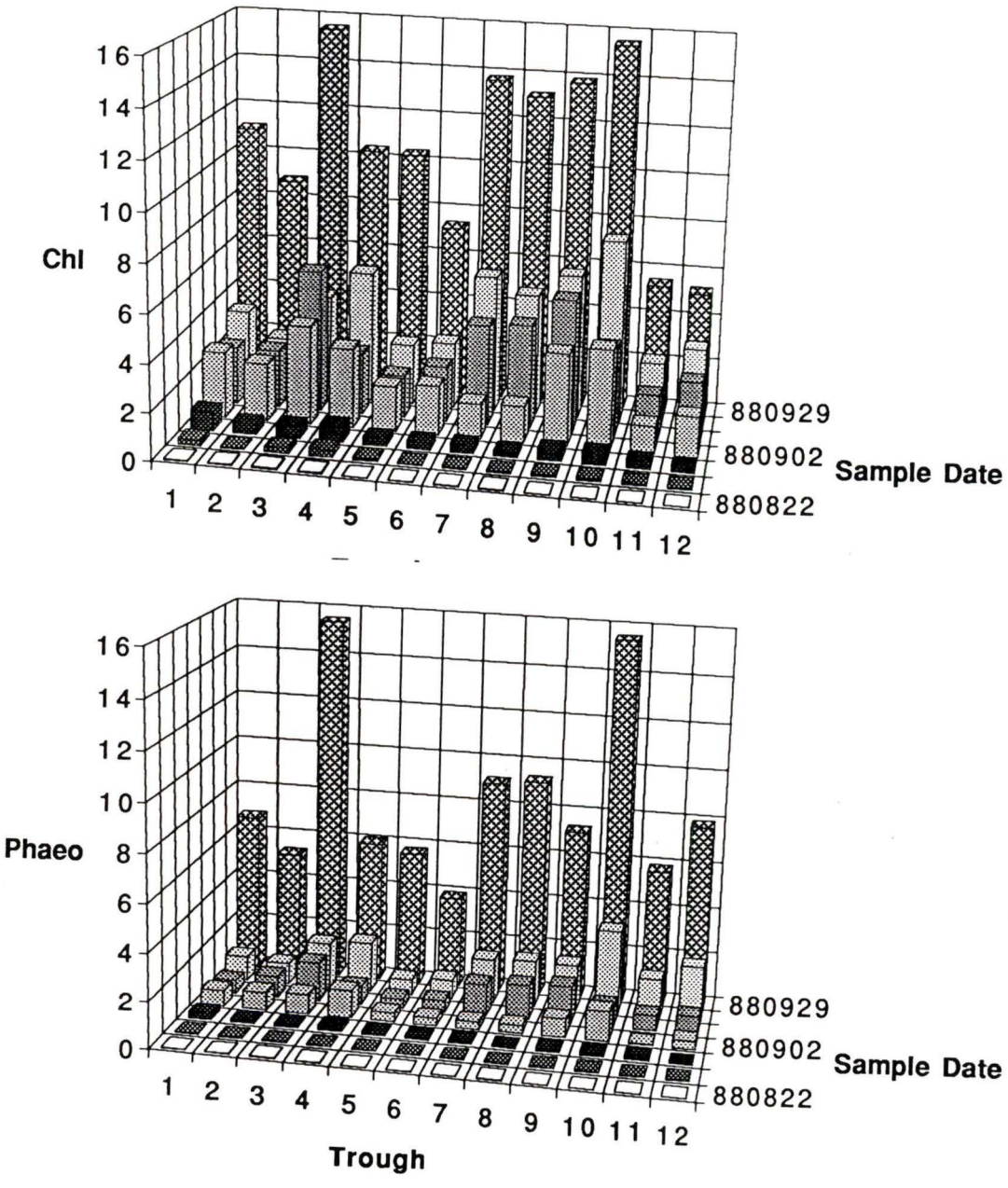
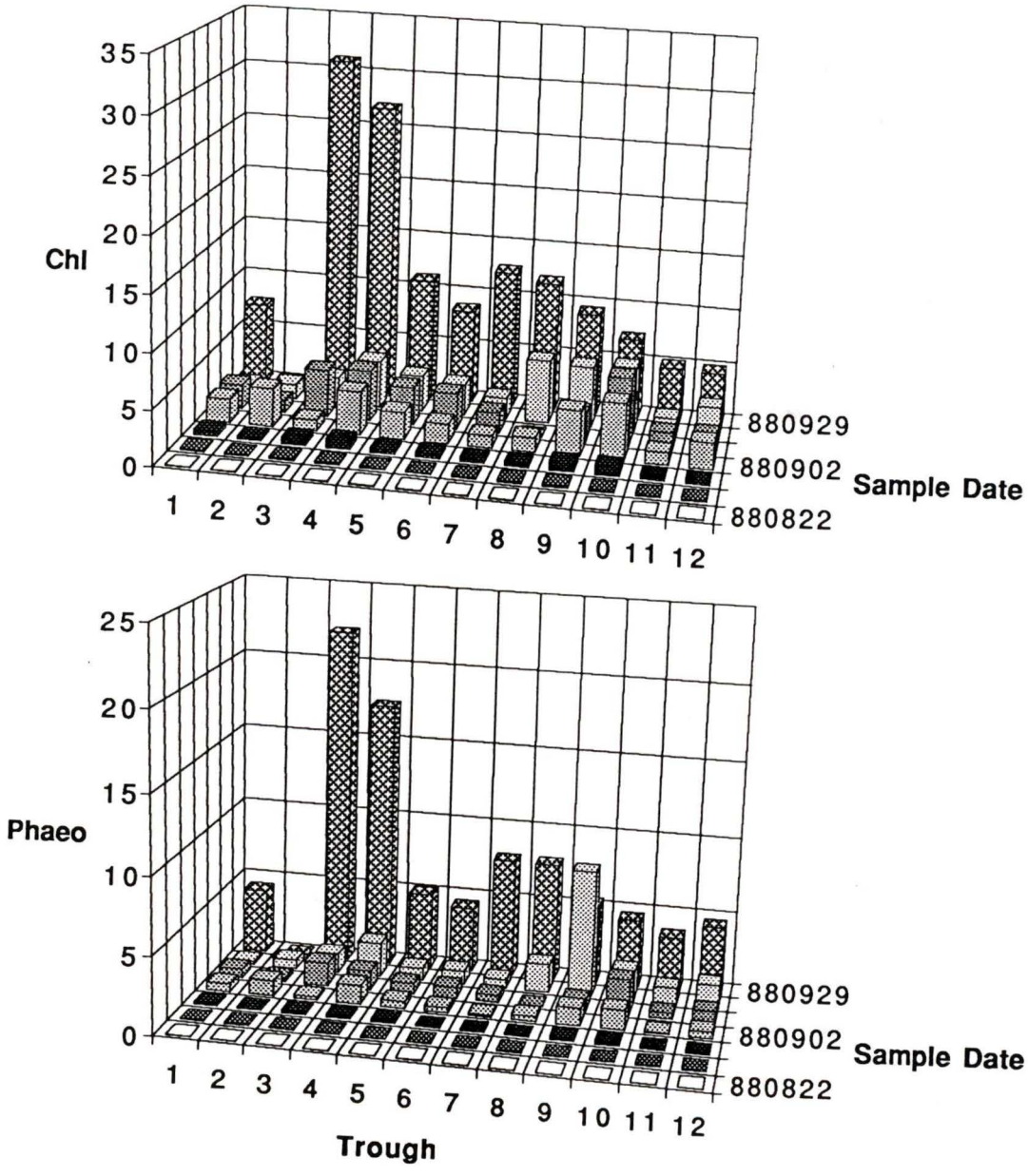

Chl
16
14
12
10
8
6
4
2
0
1 2 3 4 5 6 7 8 9 10 11 12
880929
880902
880822
Sample Date
Phaeo
16
14
12
10
8
6
4
2
0
1 2 3 4 5 6 7 8 9 10 11 12
Trough
880929
880902
880822
Sample Date

Figure 29. **Upper** Chlorophyll *a* ($\mu g/cm^2$) obtained from stream cultivated periphyton, grown on river rock, during the period - 880822, 880818, 880829, 880902, 880908, 880914, 880929. Axis parameters as in Figure 26.

**Lower** Phaeophytin ($\mu g/cm^2$) obtained from stream cultivated periphyton, grown on river rock, during the period - 880822, 880818, 880829, 880902, 880908, 880914, 880929. Axis parameters as in Figure 26.


Chl
35
30
25
20
15
10
5
0
1 2 3 4 5 6 7 8 9 10 11 12
880929
880902
880822
Sample Date
Phaeo
25
20
15
10
5
0
1 2 3 4 5 6 7 8 9 10 11 12
Trough
880929
880902
880822
Sample Date

# Chapter III. Impact of Domestic Waste Waters on the Water and Habitat Quality of a Pristine Mountain River (Cheakamus)

## Introduction

Although society's public health concerns with respect to domestic wastewater discharge have traditionally focused on pathogen transmission, the term 'health' is being expanded to include ecosystem processes. For example, treated sewage effluent discharge typically results in downstream nutrient enrichment, with associated reductions in the health of receiving water quality and habitat. In the Pacific Northwest, and in inland waters elsewhere (Chessman *et al.*, 1992; Steinman *et al.*, 1991), the discharge of nutrient enriched effluents into freshwater frequently results in increased algal accrual and production, as background, growth limiting nutrient (typically phosphorus) concentrations are exceeded(Welch *et al.*, 1992). Impairment of aquatic habitat as a result of algal biomass accrual can include loss or degradation of aesthetics, fisheries (spawning and rearing habitat), recreational activities (kayaking and fishing), water supply (taste and odours), and the capacity for further nutrient assimilation. Thus, the long-term preservation and protection of our aquatic resources requires that accurate estimates of their sensitivity to, and assimilative capacity for, nutrient enrichment, be developed.

Historically, the assessment of pollution has been largely through physical and chemical measurements, which have been used indirectly to inexpensively estimate the biological consequences. Much of the biological assessment of pollution effects has been centered on laboratory procedures, traditionally using single-species bioassays. Literature reviews of such methods (Cairns, 1974; 1986) recommend that future land-use and environmental regulation be based upon both laboratory and in-field observation and experiment (e.g. using micro- or meso-cosms, split lakes or surrogate streams). Similarly, Day *et al.*, (1988) recommended that "*biological testing and monitoring must be integrated with chemistry in a multi-disciplinary manner when applied in hazard assessment and regulatory control.*" Recognition of these concerns is reflected in regulatory decisions

increasingly requiring collaboration of heretofore separate types of studies (Dennis and Patil, 1977; La Point and Perry, 1989).

As discussed earlier, periphyton communities can be an important reflection of the health of surface waters by integrating such environmental parameters as nutrients (Hill *et al.*, 1992; Mulholland *et al.*, 1991a,b), grazing (DeNicola *et al.*, 1990; Hart and Robinson 1990; Hill *et al.*, 1992), velocity and turbidity (Hill and Harvey 1990; Horner *et al.*, 1990). Following public anxiety concerning discharge of the Resort Municipality of Whistler's treated sewage effluent into the receiving waters of the Cheakamus River (Figure 30), we used our surrogate streams to investigate the hypothesis that phosphorus was this river's limiting nutrient, to model the river's sensitivity to phosphorus enrichment, and to determine whether the river's assimilative capacity for this nutrient could be determined. The project was undertaken as a result of increased algal biomass accrual in the Lower Cheakamus River (L.C.R.) and the perception that the degradation of water and habitat quality was threatening the river's in-stream fishery and aesthetic value. The Cheakamus, located a little more than one hour from Vancouver (British Columbia), flows through a wilderness region internationally renowned for its outdoor recreation and highly regarded for the stream fishing of coho, spring salmon and steelhead.

### History & Site Description

The Resort Municipality of Whistler (RMOW), 125 kilometers north of Vancouver, and approximately 50 kilometers north of Squamish, straddles the catchment basin divide of the Cheakamus (flowing south) and Green Rivers (flowing northeast) (Figure 30). As the RMOW experienced rapid growth throughout the 1970's, a Sewage Treatment Plant (STP), consisting of secondary treatment with chemical precipitation of phosphorus, was constructed adjacent to the upper Cheakamus River with treated effluent discharged to the river. The river then flows 10 km southward into Daisy Reservoir, a B.C. Hydro facility designed to divert river flow (approximately 80%)(compare Figures 38 and 39) laterally, through a 12 km long tunnel/penstock complex, for hydro-electric generation on the Squamish River. The discharge from the turbines flows through a tailrace before

mixing with the Squamish River (Figure 30) 40 km upstream of the latter's estuary. The headwaters of the Lower Cheakamus River (that portion of the river from Daisy dam downstream to the Cheakamus-Squamish confluence) consist of Daisy Reservoir water that is required (Department of Fisheries, Federal Government of Canada) to spill through a cone valve (at the base of the dam), with a minimum (winter) discharge of 203,000 $m^3$ $d^{-1}$ (83 cubic feet second)(Figure 32). The Cheakamus and Green River catchment basins consist of steep, glaciated peaks, with V-shaped valleys characteristic of Coastal Mountains in which the weathering of the extensive volcanic intrusions, and often phosphorus rich parent material, releases phosphorus (Squamish Plan, 1981).

The post-1970's increased algal growth in the Lower Cheakamus River resulted in a report (Nordin, 1984) which recommended a field study to assess water quality characteristics including acute toxicity, bacterial pathogenicity and eutrophication. This report also drew attention to possible water quality alterations originating as a result of the Daisy impoundment.

The resulting study identified a 100 m zone of acute toxicity and aesthetic deterioration immediately below the STP discharge. Also revealed was how impoundments such as Daisy Reservoir (constructed in the mid-1950's) could stabilize water depth and velocity flow regimens. The impoundment was also thought to reduce turbidity and, as a physical barrier, to reduce density and diversity of herbivorous macroinvertebrates, secondary consumers known to reduce algal standing crop through grazing (Steinman *et al.*, 1987). In the late 1980's algal standing crop in the Lower Cheakamus River was marginally below that established to maintain optimal water quality. Throughout this period biomass accrual in the Upper Cheakamus River (U.C.R.) was considerably lower. Of principal concern to regulatory agencies was whether projected increased discharges, from the RMOW's STP, of dissolved orthophosphorus and inorganic nitrogen would increase Lower Cheakamus River algal accrual. Measurements showed that the STP's discharged effluent contributed only approximately 10% of riverine N & P. As a result of the impoundment it was hypothesized that marginal orthophosphorus increases to the Lower Cheakamus River could markedly

enhance algal accrual (Riber and Wetzel, 1987). The literature suggested that increased orthophosphorus concentrations in the 1.0 - 3.0 µg/L range had enhanced eutrophication elsewhere and could do so within the Lower Cheakamus River.

### Study Outline

The protocols of our long term study are outlined in Figure 34 in which the research objectives for each year are identified, the findings characterized and the subsequent years study objectives defined. Limnological sampling sites were located along both Cheakamus and Squamish Rivers, and on Daisy Reservoir, whilst the surrogate stream mesocosms were sited below the Daisy Reservoir dam (Figure 30). Additional samples were obtained from the effluent discharge stream prior to its confluence with the Cheakamus River. Whether orthophosphorus concentrations would increase algal growth downstream of the discharge was assessed in 1987. At that time a whole-river experiment was conducted, in which *pickling liquor*, normally added to post-secondary effluent as a phosphorus-precipitator, was omitted during the summer test period. This resulted in a fourteen-fold increase in orthophosphorus concentrations entering the river. Measurements of orthophosphorus concentrations and algal biomass accrual were made in the Cheakamus; periphyton biomass accrual exceeded historical values.

Surrogate split streams were established on the Lower Cheakamus River, to which additions of N, P and N+P were made to correlate enhanced nutrient concentrations and periphyton accrual. A second system was installed adjacent to the Squamish River in 1991; sedimentation chambers were required (Mueller 1970a,b). Nutrient dosing trials were conducted in 1988, 1989 and 1991; elements of the 1988, 1989 and 1991 results are discussed here, together with the findings of the 1992 and 1993 public participation process designed to develop a Liquid Waste management Plan for the RMOW.

### Fisheries Resource

The fishery resource in the Squamish River includes important spawning runs of coho (*Oncorhynchus kisutch*), chinook (*O. tshawytscha*), chum (*O.*

*keta*) and pink (*O. gorbuscha*) salmon. Resident and anadromous rainbow/steelhead (*Salmo gairdneri*) and cutthroat (*S. clarkii*) trout, together with dolly varden char (*Salvelinus malma*) are important sport fish in this river. Both spawning and rearing habitat are included in the major river reaches of concern located in and below the power-house tailrace.

Because of an impassable falls the Lower Cheakamus River contains anadromous species only in the lower 17 km of the river. Rainbow, cutthroat trout, dolly varden and mountain whitefish (*Prosopium willamsoni*) are found throughout the lower river system. Anadromous species include chum, pink, coho, chinook and sockeye (*O. nerka*) salmon (in descending order of importance), together with steelhead trout. The late-winter steelhead runs are rated second in importance in the province, in terms of angling success, with the mainstream of the Squamish River being eighth.

Daisy Reservoir sportfish populations include rainbow trout (stocked), kokanee salmon (*O. nerka*) and dolly varden char. Spawning of these species occurs in the three streams on the west side of the reservoir. The Upper Cheakamus River supports resident rainbow trout with the primary spawning and rearing habitat extending from the falls, above the reservoir, upstream to the Millar Creek falls which is 600 m downstream of the STP. The upstream reach, above the STP, supports considerably fewer fish than lower reaches. This is thought to be due to a greater river gradient (2%) upstream than down (1.4%) and less suitable spawning substrate (fewer areas of small gravel).

### Results

The sample site locations on the Cheakamus and Squamish Rivers are shown in Figure 30. The 1989 proposed pipeline (discussed below), planned adjacent to highway 99 (marked with asterisks) beginning at the STP and ending in the B.C. Hydro water diversion tunnel (Shadow Lake).

Periphyton biomass, expressed as chlorophyll *a* ($\mu g/cm^2$), during the period mid-July through mid-August (1987) for the Cheakamus is shown in Figure 36. In the upper section of the river (4 km downstream of the STP) biomass

exhibited a significant increase from less than 0.2 to 5.5 $\mu g/cm^2$ in the first two weeks (early June) of sampling, before decreasing and then increasing again. The pattern of accrual in the Lower Cheakamus was similar but of significantly less magnitude, with a maximum of 1.5 $\mu g/cm^2$ after two weeks. The following year, in 1988, following cessation of pickling liquour (to effect phosphorus removal), algal biomass in the Lower Cheakamus increased earlier with a maximum of 3 $\mu g/cm^2$ by mid-July. Biomass accrual in the Upper Cheakamus slowly increased until mid-July, then it increased rapidly, reaching a maximum of approximately 3.75 $\mu g/cm^2$ by the end of July. Accrual at the end of July was not significantly different in these two river sections.

During the summer of 1987, biomass accrual at four sites along the Cheakamus was lower in the upper Cheakamus than below the dam (Figure 37). Typically, accrual upstream of the STP (2000 $\mu g/cm^2$ maximum) exceeded that downstream of the plant (i.e. 1988, 1989 300 $\mu g/cm^2$). Biomass downstream of the plant, in the summer of 1987, was approximately 5 times that normally recorded at this site. Immediately downstream of the dam maximum accrual was 3000 $\mu g/cm^2$, whilst further downstream values reached 2500 $\mu g/cm^2$. During the 1988 summer, biomass accrual patterns in the upper Cheakamus were similar to the previous year, although magnitudes were less (upstream of STP 1300 $\mu g/cm^2$, downstream 300 $\mu g/cm^2$). Lower Cheakamus 1988 biomass accrual patterns were similar to the previous year, however, the magnitude had increased substantially (below the dam 5200 and further downstream 6000 $\mu g/cm^2$). Biomass accrual in the same locations in 1989 again demonstrated the same patterns, at the same locations, with accrual in the same range as that of 1988. In the Upper Cheakamus values were less than 500 $\mu g/cm^2$ (maximum) and 6000 $\mu g/cm^2$ at the two sites below Daisy dam. In the lowest section of the Cheakamus, below the canyon, values of the same magnitude as those observed below the dam were measured.

During the summer of 1987 noticeable increases in orthophosphorus concentrations and biomass accrual were measured in the Upper Cheakamus River, downstream of the STP discharge. When compared with 1988 data

(n=10 for both years), the 1987 orthophosphorus values 4.0 km downstream of the discharge were at least 2 times those of 1988 (geometric mean of 3.6 µg/L in 1987 and 1.7 µg/L in 1988). Upstream values averaged approximately 1.2 µg/L in both years. Although chlorophyll *a* and biomass values were highest in the Lower Cheakamus, measurements obtained from sampling substrata in 1987 revealed greater accrual in the Upper Cheakamus (Figure 38). In 1988 accrual was highest in the Lower Cheakamus (Figure 39), indicating 1987 Upper Cheakamus River values were largely due to the increased orthophosphorus concentrations.

Upper Cheakamus River algal biomass levels were significantly elevated downstream of the discharge in 1987 but this trend was not apparent in the Lower Cheakamus River. While Upper Cheakamus River orthophosphorus values were doubled (112%), Lower Cheakamus River orthophosphorus increases were barely detectable (3%), immediately downstream of the dam. This increase is within the ranges of measurement error and field sampling error. Yearly fluctuations below the dam (velocity, flow patterns, orthophosphorus concentrations) could mask biomass increases resulting from elevated upstream orthophosphorus concentrations. It was anticipated that, given the fourteen-fold increase in phosphorus upstream, there would have been a significant biomass increase downstream of the dam.

Tributary flows into the Lower Cheakamus River during 1988 indicate that the river itself, together with Callaghan Creek account for approximately 56 percent of river flow, with Brandywine Creek contributing an additional 9 percent; the remainder consisted of 6 percent from Millar Creek and the 29 percent from all remaining tributaries (Table 10).

Partitioning of the orthophosphorus concentrations entering the Lower Cheakamus River for the years 1986/87, 1988 and 1990 (Table 10) indicate the main stem itself contributed between 22 and 28% of this nutrient. The Callaghan Creek contribution varied between 30 and 34%, whilst the remaining tributaries conveyed between 18% (1990) and 25% (1986/87). The only discharge into the Cheakamus which evidenced a significant increase during this period was that from the STP, being 9% (1986/87) and 30% (1990).

Monthly mean effluent discharge flows ($m^3$/day) during 1987 (Figure 40) showed maximum throughput from the STP occurred during December through April, the prime skiing months. Minimal flows occur during October and November, the period of lowest recreational population. The discharge flows parallel the number of non-resident visits to the resort (C. Jennings, pers. comm. 1991).

Water velocities in the Squamish and Lower Cheakamus Rivers (Figure 41) indicate the Cheakamus generally evidencing a significantly lower velocity throughout most of the period (July through October, 1991). Although velocities generally decreased throughout the summer a maximum velocity of 100 cm/second in the Lower Cheakamus occurred during the storm event of August 30/31, 1991; average velocities were typically below 45 cm/second.

Dilution ratios, calculated for minimum Cheakamus River flows, indicated that the period of maximum effluent dilution occurred during May through August, when dilution ratios exceed 100:1 (Figure 42). This period coincides with the onset of the summer freshet (Figures 38 and 39).

### Silica

Dissolved reactive silica concentrations in the upper Cheakamus were similar to those in the lower Cheakamus in the first week of July (Figure 43), after which they decreased from approximately 4 to 3 mg/L. The opposite trend was observed in the Lower Cheakamus where concentrations increased from 4 to 8 mg/L by late July, after which they remained relatively constant, reflecting an approximate doubling of the upstream concentration.

### Macroinvertebrates

Macroinvertebrate diversity indices calculated, in 1987, for the Upper Cheakamus, downstream of the STP, exhibited considerable variability (Figure 44), with a sudden decrease in index value in the second week of August, which then returned to late July values. Conversely, the index for the lower Cheakamus evidenced a relatively constant value throughout the sample period.

Total dry weight of macroinvertebrates (Figure 45) in the upper Cheakamus, downstream of the STP, showed a pattern of increasing from approximately 150 mg/0.45m$^2$/5cm depth, in late July, to more than 650 mg/0.45m$^2$/5cm depth in late August, after which they returned to previous values (Figure 45). Total numbers of macroinvertebrates showed a similar numerical abundance pattern, increasing from 500 to 3000 and decreasing to 1000 organisms. Similarly, in the Lower Cheakamus macroinvertebrate dry weight (total) was relatively stable throughout this sampling period, never exceeding 300 mg. Numerical abundance patterns evidenced a maximum in mid-September of 1500 organisms.

### Periphyton

Periphyton biomass accrual in the stream streams, in 1988, are shown in Figure 46a. The minimum value occurred under nitrogen enrichment, which yielded a final accrual of approximately 2000 μg/cm$^2$, not significantly different from the control. Enrichment with orthophosphorus, at all concentrations, resulted in enhanced biomass. Biomass maxima was approximately 7500 μg/cm$^2$, at an orthophosphorus concentration of 3.1 μg/L/sec. Periphyton accrual in 1989 resulted in a maximum control biomass of 200 μg/cm$^2$; under nitrogen enrichment accrual did not exceed 400 μg/cm$^2$. At an orthophosphorus enrichment of 0.42 μg/L/sec biomass accrual achieved a maximum of 1800 μg/cm$^2$ (Figure 46b). At an enrichment concentration of 0.02 μg/L/sec no difference with that of the control was observed; however, at 0.08 μg/L/sec a maximum biomass of 600 μg/cm$^2$ was measured.

Periphyton accrual was plotted for two sites on the Cheakamus River, during the summer period, for the years 1987, 1988 and 1989 (Figure 47). Although considerable variability was evident, the general trend saw increased accrual across years and a within-year pattern of maximum accrual in mid to late August, followed by a loss of biomass as the autumn progressed. Maximum-minimum accrual ranged from less than 400 μg/cm$^2$ to a maximum of 3200 in 1987 and 6000 μg/cm$^2$ in 1988 and 1989. Biomass values exceeded the maximum permissible criterion of 2500 μg/cm$^2$, for a significant portion of the sampling period, in both 1988 and 1989.

The correlation between orthophosphorus enrichment (as a percentage of the 1989 background concentration (1.0 μg/L)) and increased biomass (also as a percentage of background) is shown in Figure 48. The correlation at the lowest orthophosphorus concentrations showed that a 0.5 percent increase in orthophosphorus (above background concentration) resulted in a 13 percent increase in algal biomass (over background).

During the 1988 surrogate stream experiments biomass accrual at a proposed enrichment treatment of 0.5 μg/L is plotted in Figure 49. Also plotted is the actual concentration of orthophosphorus added (as a function of the background concentration of 1.0 μg/L). During the third week of August, when orthophosphorus was inadvertently added at a rate significantly higher than normal, there was an almost instantaneous increase in biomass accrual, which subsided when the orthophosphorus dose returned to normal. This pattern of biomass increase was different than the norm since the orthophosphorus was added to an existing periphyton community, compared with that added at the beginning of an experiment, in which recruitment and community development was initiated *de novo*.

Periphyton biomass accrual in the lower Cheakamus, downstream of Daisy dam, between 1987 and 1991, from July through mid-September, showed considerable variability (Figure 50). The general trend, however, saw increased growth with each subsequent year, for any given weekly sampling period. The apparent exception was the period after the second week in August, 1991, in which accrual was minimal, following a major rain storm and after the 1st of September, following a second major rain storm. The extremely high discharge through the dam caused extensive scouring of the bed rock in the lower Cheakamus, eliminating all periphyton biomass. The community had begun to re-establish itself by the second week of September. The plot shows the proposed aesthetics objective of 2500 $\mu g/cm^2$ and the maximum permissible fisheries objective of 5000 $\mu g/cm^2$.

Turbidity values measured in the upper and lower Cheakamus Rivers, in 1987 and 1988 (Figure 51), indicated that in both years, during the June to September period the upper river experienced significantly higher light attenuation. In general, the same turbidity pattern was observed in both

sections of the river, with a trend of decreasing turbidity as summer proceeds. During a mid-August rain storm in 1988, however, turbidity increased by a factor of at least 300 percent, whilst that of the Lower Cheakamus did not alter appreciably, due to sediment trapping within Daisy Reservoir.

Turbidity values obtained within the Squamish River, both upstream and downstream of the tailrace, and within the tailrace, indicate a positive correlation with river stage (not shown here) and light attenuation (Figure 52). The maximum value recorded during the last week of August and first week of September were obtained during, and after, a significant rainstorm; this was estimated to be a fifty or one-hundred year storm event (B. Moore, pers. comm. 1992). The turbidity in the tailrace results from water discharged from Daisy Reservoir, being indicative of Upper Cheakamus turbidity. In general, Cheakamus turbidity was significantly lower than either the Upper or Lower Squamish (Figure 53).

Annual solids loading in the Squamish River (Figure 54) indicated a maxima non-filterable residue (@ 105°C) occurred during July, paralleling closely the hydraulic staging profile of this river (Figure 35).

The relationship between turbidity and suspended solids (non-filterable residue) is characterized in Figure 55. There appears to be a reasonable correlation between increased turbidity and non-filterable residue, particularly in the range (40 - 110 mg/L) evidenced during maximum flood on the Squamish River.

A comparison of treated sewage-effluent discharges, from the STP, into the Cheakamus River, during the summer months of July and August, from 1983 through 1991 is shown in Figure 56. The reduced discharge in 1990, coincided with a significant downturn in the Canadian economy, resulting in fewer recreational visits to the RMOW. Also plotted are the allowable effluent discharge flows, at different orthophosphorus concentrations of effluent, which if exceeded would effect a maximum permissible algal biomass (2500 $\mu g/cm^2$) in the Lower Cheakamus River (Figure 50). Since the early 1980's the quantity of treated effluent discharged during the months of July and August has increased approximately 400% (Figure 56), paralleling the increased

nutrient loading to the Upper Cheakamus River. Maximum permissible effluent discharges from the STP, based on the increased orthophosphorus predicted in the lower Cheakamus River are shown in Figure 57. At the 1989 orthophosphorus concentration a maximum discharge of approximately 6200 $m^3$/day would be permitted.

The temporal separation between steelhead (spring) and salmon (late summer and autumn) adult migration and spawning patterns, in the Squamish River, are shown in Figure 58a. The juveniles of all species exhibit downstream migration in the spring. Figure 58b indicates the onset of biomass accrual in the Upper and Lower Cheakamus and Squamish Rivers (for 1991), which coincides with the reduced hydraulic stage of the Cheakamus River (from Figure 32). Also seen is the relation between maximum biomass accrual in the shallow side-channels and the timing of salmon spawning.

Periphyton biomass accrual developed in the surrogate streams, at the Daisy dam site during the summer of 1991 (Figure 59), attained a maximum value of 2500 $\mu g/cm^2$ for the control and 4900 $\mu g/cm^2$ in streams subjected to an orthophosphorus enrichment of 0.05 $\mu g/L$. Maximum biomass accrual occurred in the last week of September, in 1991. Periphyton development in the streams adjacent to the Squamish River was minimal, with maximum accrual not exceeding approximately 100 $\mu g/cm^2$.

An *aesthetics standard* was established (Figure 58) at a maximum biomass of 2500 $\mu g/cm^2$, for the Lower Cheakamus River. By plotting the weekly biomass data from 1987 to 1989 at two stations in the Lower Cheakamus River to assess the average biomass level during this period, the standard was determined as that biomass which generated public complaints to Government agencies. The 1989 average summer biomass was 2365 $\mu g/cm^2$ (or 5.7% less than the 2500 $\mu g/cm^2$ maximum).

Data summaries from the 1988 and 1989 surrogate stream studies are shown in Figure 46a and b, respectively. The 1988 data indicates that additional nitrogen induced minimal increase in biomass accrual. When orthophosphorus was added at any concentration there was a significant

increase in accrual; N and P combinations also increased accrual. The overall trend in the controls was for biomass to increase slowly for four weeks until a more rapid increase was observed in the last ten days of the experiment. Addition of phosphorus produced the same trend, however, with each higher orthophosphorus addition the slope of the accrual curve steepened, indicating an earlier onset of enhanced accrual. In 1989 a significant increase in growth (relative to that of the control) occurred at an orthophosphorus concentration of 8% above background. Late August biomass values were lower in 1989 than in 1988.

The surrogate stream data showed a 13% biomass increase could result from a 0.5% increase in background orthophosphorus concentration. Based upon water chemistry data, it was known that Upper Cheakamus River orthophosphorus concentrations (i.e. above Daisy Lake) were approximately 400% higher than that measured at the dam. (The remaining 75% was thought to be retained within the sediment load which settles out within the reservoir.) The Upper Cheakamus River orthophosphorus values, at a position in the river closest to Daisy Lake, are approximately 42% of the values measured immediately below the STP discharge. Based upon this downstream orthophosphorus gradient, a 0.25% orthophosphorus increase at the dam site would be equivalent to a 2.35% orthophosphorus increase immediately downstream of the STP discharge. The average summer (89) effluent discharge of 5339 $m^3$ $d^{-1}$ received an average 670:1 dilution with river water, based upon an average summer flow calculated on river flows from the 1924-48 and 1982-88 period (Water Survey). The average orthophosphorus concentration of the 1989 effluent was 200 $\mu g$ $L^{-1}$, while the average background orthophosphorus concentration was 1.0 $\mu g$ $L^{-1}$. Therefore, for the latter effluent quality to effect an increase in riverine orthophosphorus by 2.35%, effluent discharge would need to be increased by approximately 10.2%. The linkage between the permissible biomass maximum in the Lower Cheakamus (as a result of increased P upstream) and discharge of treated effluent from the STP is characterized in Figure 57. Thus, should effluent discharge (at an orthophosphorus concentration of 200 $\mu g$ $L^{-1}$) exceed 5884 $m^3$ $d^{-1}$, the downstream assimilative capacity would be surpassed. The latter was reached in 1991 (Figure 56).

## Discussion

The sensitivity of many coastal British Columbia lotic and lentic habitats to phosphorus-based eutrophication has been well documented (Stockner and Shortreed, 1978; Shortreed *et al.*, 1984; Perrin *et al.*, 1984; 1987; Mundie *et al.*, 1991). Bothwell *et al.*, (1989) has demonstrated the sensitivity of the Thompson River periphyton community to phosphorus mediated eutrophication. The 1988-1989 data reported here confirm these authors' findings, of lotic periphyton sensitivity to orthophosphorus, for the Cheakamus River.

Cascadia, the fastest growing urban region in North America, comprises a region of the Pacific-northwest, whose southern terminus is Portland, Oregon and northern limit is Vancouver, British Columbia. Within this region two major rivers discharge into the Pacific ocean - the Columbia (Portland) and the Fraser (Vancouver), together with the Squamish/Cheakamus Rivers which, flowing into Howe Sound, form the northern edge of the Vancouver city limits. Since the early 1970's the Cheakamus River valley has been the site of extensive wilderness recreational development and future projections indicate this trend will continue (Chantler, 1993). At present, the principal impediment to future residential development is nutrient enrichment of aquatic habitat, resulting from the discharge of treated sewage effluent (B. Moore, 1993; pers. comm.).

Means of long term flows for both the Squamish and Cheakamus Rivers indicate basal minima occur during the winter (January through late March or early April), with the maximum flows occurring in the summer (June through August), following the spring freshet (Figure 35). The flows in the Squamish River are at least an order of magnitude greater than the Cheakamus, with the Upper Cheakamus having the least flows.

Mean monthly inflows (cfs) to Daisy Reservoir (based on 1961-1990 average basin inflows) can be divided into the three constituent components - base flow, snowmelt and rainfall (Figure 31). Maximum base flows occur in June and continue into November, with minima in March through May.

Maximum rainfall contributions occur during the September to December period, when the freezing level is not yet low enough to result in precipitation retention through snow accumulation. Snowmelt produces the largest single input into the reservoir during the period May through August. Although not shown in this graph, maximum inflows of 35,000 cfs are not uncommon. Cheakamus River mean monthly flows are plotted in Figure 32, demonstrating a similar pattern as that of the Squamish, except that maximum flows occur in June and July, with May contributing significantly more to river volume discharged than in the Squamish (Figure 35). Other flow regimens (L.M. Bell, M.Sc. thesis, U.B.C.) demonstrate similar discharge patterns for the Squamish (1922 - 1973) and Cheakamus (1967 - 1973) Rivers, as those we calculated using more recent data.

The 1988 study found that although biomass increased as a result of eutrophication (Figure 46a and b), this experiments range of concentrations were too high to characterize the threshold below which no additional biomass would accrue. The data demonstrated that the Lower Cheakamus River was considerably more sensitive to phosphorus at the lowest dosage concentration applied in the surrogate streams (0.5 µg/L). The data indicated that should background concentrations in the Lower Cheakamus River (1.0 µg/L) increase by the 0.5 µg/L treatment concentration there would be an unacceptable increase in biomass above the maximum permissible value (Figure 46a, d) established for the maintenance of water and habitat quality. Since the Cheakamus River valley has been subject to persistent development pressures, such an increase in orthophosphorus concentrations could occur if further development, which could effect enhanced nutrient loading, were to take place. The minimum chemically-detectable concentration of orthophosphorus was 1.0 µg/L, higher than the lowest concentration added to the streams. Our data had shown that, even at this low level, there was a significant enhancement of periphyton accrual. It was believed that maximum acceptable increases were best discussed in terms of percent addition over background, rather than as absolute increase in algal accrual, since additional algal growth could partially mask orthophosphorus concentration elevations by increased orthophosphorus uptake. A second series of nutrient dosing experiments were then designed to assess algal

accrual at concentrations considerably below chemically detectable orthophosphorus concentrations, to determine how much additional phosphorus the Lower Cheakamus River could assimilate before exceeding the maximum permissible biomass.

Figure 60 (lower left photograph) represented a typical side-channel in the Lower Cheakamus River, showing increased irradiance, evidenced by the sun bleached river rock following riparian canopy removal, and heavy algal biomass accrual in the shallows (sample site M; Figure 30).

The 1989 study again showed periphyton biomass accrual (both accrual and timing) at below chemically detectable orthophosphorus concentrations. Given the consistent correlation between increases in orthophosphorus and biomass, we were able to estimate proposed nutrient loading and assimilation rates, based upon biomass accrual patterns, in both the streams and at river sample sites (Lucey *et al.*, 1989; 1992). Provincial Fisheries staff had indicated in 1989 that the Lower Cheakamus River algal accrual was approaching a value which would begin to have detrimental affects on fry habitat in the shallow river margins and side-channels; it was also determined that the same biomass would be aesthetically unacceptable to the public. The significant biomass accumulation in side-channels was shown to have at least two adverse affects on fisheries. Firstly, biomass would clog the interstices between substrata in the shallow side channels, forcing fry into the central channel where they are subjected to higher water velocities. This would require more energy expenditures to remain within a given reach of the stream or river. The reduced volume of suitable side-channels would increase stocking densities, although the implications of this on fish health are not known. Secondly, as the biomass becomes senescent in the fall and sinks to the bottom in the shallow side channels, the decaying vegetation can affect spawning habitat, especially if it strips oxygen from the gravel beds. The senescent biomass typically accumulates in the side channels in which there are both low flows and velocities. No studies on these aspects have been conducted in the Lower Cheakamus River to date.

As a result of the demonstrated sensitivity of the Lower Cheakamus River to eutrophication (Figures 56a-d) and the increased discharge of effluent from

the RMOW's STP public concern resulted in a M.O.E.L.&P. required Liquid Waste Management Plan (LWMP) for the municipality. The plan was designed to achieve minimal discharge of orthophosphorus, by the STP, into the Upper Cheakamus River, *which should then ameliorate the periphyton biomass accumulating in the Lower Cheakamus River.* Although a number of options were explored, the final option selected was construction of an effluent pipeline, to be buried adjacent to, and lying parallel with, Highway 99. The pipeline would originate at the STP and end at the small lake immediately upstream of the B.C. Hydro penstocks (Figure 30). This effluent, discharged at the STP into the pipeline, would be diverted into the Squamish River (via the tunnel/ penstock/ generating plant complex), thereby avoiding STP-based nutrient loading of the Cheakamus River and Daisy Reservoir. It was proposed by the consultant that given the high flows (Figure 35) and turbidity (solids loading; Figure 52) in the Squamish River, the additional nutrients entering the Squamish, via the proposed pipeline, would not likely create an algal accrual problem. Since nothing was known, however, of the functional and structural attributes of the periphyton community of the Squamish River, nor how the proposed nutrient loading could alter them, it was recommended that a preliminary study, similar to that being conducted on the Cheakamus, be undertaken.

A study was conducted during the summer of 1991, whilst additional surrogate streams were initiated at the Daisy Reservoir site (Figure 60, 61). Preliminary findings of the 1991 study (Lucey *et al.*, 1992) indicated that the summer of 1991 was non-typical both climatically and hydrologically (Figure 62 and 63). In particular, it was shown that, in the Squamish River, the necessary coincidence between a *climatic window* and a *hydrologic window,* within which environmental conditions would be favorable for periphyton accrual, did not occur. Although prior to the major storm event, the 1991 river and stream data for the Cheakamus had similar growth patterns to previous years (1987-1990), growth patterns in the Squamish were not thought to be typical of those which would likely have occurred. In the absence of understanding of the nutrient dynamics in the Squamish River, the 1992 report recommended postponement of the proposed pipeline option,

until further water and habitat quality studies in the Squamish River could be undertaken.

Although not presented here, our data have shown that downstream impoundment-modified environmental parameters effected increased algal accrual through i) post-freshet, nutrient-rich deep-reservoir discharges to the LCR (Marsden, 1989), ii) clear-water discharge whose reduced turbidity resulted in higher Lower Cheakamus River irradiance values (Horner *et al.*, 1990), and, iii) discontinuous downstream drift of macroinvertebrates (and insect diversity), thereby reducing grazer stress on the periphyton biomass accrual (Steinman *et al.*, 1987; Winterbourn 1990; Lamberti *et al.*, 1992). The presence of the impoundment is thought to prevent downstream invertebrate drift since surface water, containing most insects, is retained and diverted to the penstocks, whilst downstream discharge to the Lower Cheakamus River is via the deep-cone valve at the base of the dam. The Lower Cheakamus River's extensive shallow side-channel zones (Biggs and Close, 1989), with minimal riparian canopy (i.e. enhanced irradiance), provide optimal habitat for maximum periphyton accrual. The typically low flows during this period limit the degree to which the Lower Cheakamus River community is subjected to scouring stresses, unlike the communities in the UC and Squamish Rivers.

Differences also exist with respect to benthic substrata, in particular, the Squamish main-stem has extensive expanses of sand laden benthic habitat. These shallows thus minimize both velocity (Antoine and Benson-Evans, 1982) and flow (Whitford, 1960) and maximize irradiance, important considerations in periphyton community development (Hill and Harvey, 1990). Riber and Wetzel (1987), studying boundary-layer and internal diffusion effects on phosphorus fluxes in periphyton communities, suggest that, while nutrient-cycling provides for community maintenance, net accumulation of biomass requires episodic orthophosphorus additions in the bulk flow across the community. Downing and McCauley (1992) have shown that in addition to phosphorus sensitivity (i.e. as the limiting nutrient) community accrual is also affected by N:P ratios (Levine and Schindler, 1992; Stockner and Shortreed, 1978; 1985).

The unusual summer hydraulic regimen prevented any assessment of the normal range of periphyton development within the Squamish River. The heavy sediment smothering and scouring following the heavy rainstorm of August 31, 1991 (Figure 64) removed all periphyton in the river. This storm resulted in a significant challenge to the stream design we had implemented since, as seen in Figure 65, the river actually flowed through the fenced, experimental stream compound, although without doing any significant damage. It required one day to make the system operational, following subsidence of the flooding. Further, the storm demonstrated the need to establish the test system in the lee of the bridge abutment, which deflected most of the river's energy.

The literature documents (Spence and Hynes, 1971; Holmes and Whitton, 1981) the presence and pattern of algal growth downstream of impoundments, which my study also reports. Whilst the works cited describe, in a general context, the situation of impounded rivers they all differ significantly in one aspect with respect to the Daisy Reservoir complex. The cited works all describe a deep hypolimnetic discharge in which all the water upstream is discharged downstream of the impoundment, with the discharged water being considerably colder than the inflows. The latter condition has been shown to alter the downstream invertebrate community dynamics. The dam and reservoir structure at Daisy differs in two respects to those of the cited works. Firstly, the majority of water is diverted laterally through pen stocks to the Squamish River. The downstream discharge consists of hypolimnetic water which as warm as that upstream and thus it is not clear whether the ecological processes acting in the cited impoundments are similar to those we investigated. The shallow depths and low residence times of Daisy likely minimize temperature differences between inflow and outflows. Since the cited works strongly identify reduced temperature as an important environmental attribute further study will be required to assess its effect on Cheakamus downstream invertebrate communities. The hydraulic regimen of Daisy Reservoir would also appear to differ considerably from those in the cited works, further cautioning the extrapolation of published inferences to the system we have been studying.

The author recognizes the limitation inherent in using three data points at the lower limits of resolution to predict the degree of sensitivity of the lower Cheakamus River to phosphorus. An alternative interpretation of the 1988/89 findings (expressed as a percentage of the 1988 data) would produce a phosphorus/biomass accrual relationship considerably different than that proposed here. The phosphorus concentrations used in 1988 and 1989 were sufficiently different to make it inadvisable to replot the relationship between phosphorus and biomass because the 1988 data did not accurately reflect the relationship between phosphorus concentration and biomass at the very low end of phosphorus concentrations. I believe that the appropriate interpretation of the stream data, with respect to phosphorus/algal biomass, lies in the accuracy of prediction afforded by the two different interpretations. The use of the 1988 data to express phosphorus/biomass relationships would argue that the algal community is less sensitive to phosphorus loading than that predicted by the 1989 data. Further, using the 1988 data predicts that at very low, non-chemically detectable phosphorus concentrations there should be little or no increase in biomass accrual. I believe that the 1989 data more accurately reflects this relationship and, therefore, the limited data used does represent the phosphorus/biomass accrual patterns observed in the downstream Cheakamus River. The literature, unfortunately, does not contain published references to algal accrual at such low nutrient concentrations. Therefore, the optimum mechanism for resolving how best to interpret the 1988/89 data would be to assess stream contained periphyton accrual at a phosphorus concentration between the two lowest concentrations cited for the 1989 study. This experiment was conducted in the summer of 1991 (Figure 59).

The 1991 stream data again demonstrated that the Cheakamus River algal community was particularly sensitive to phosphorus loading, and that, furthermore, the sensitivity was within the range predicted by my interpretation of the 1988/1989 data. The 1991 results must therefore be interpreted either as reflecting the authors assessment of this river's sensitivity to phosphorus or showing the rivers considerable between-year variation, with the 1991 findings, although predicted, representing random variation. The experimental evidence of this study suggests that non-

chemically detectable concentrations of orthophosphorus can exert substantial influence on biomass accrual. Thus, the variation in biomass accrual between years is not, in my opinion, necessary to understanding nutrient/biomass relationships. The experimental stream data demonstrates that although algal accrual is sensitive to phosphorus concentration the relationship is neither a simple one nor is it strictly linear. There appear to be thresholds beyond which increased addition of phosphorus does not yield biomass at the same rate as at lower concentrations; the system may have become limited by nitrogen or other factors.

The author recognizes that, although few data exist concerning the nutrient status and cycling in the UCR, and within Daisy Reservoir, the nutrient dynamics of the latter water body, separating the Upper and Lower Cheakamus River, likely plays an important role in the downstream cycling of nutrients from the Upper Cheakamus River to the Lower Cheakamus River (Stockner, pers. commun. 1992; Lucey *et al.*, 1992). The nutrient cycling within the rivers themselves is also likely influenced by their temperature and water velocity (D'Angelo *et al.*, 1991), biological communities and by groundwater intrusions (Hynes, 1983). These factors are likely to exert a greater affect on biomass accrual in the Lower Cheakamus River, than either the Upper Cheakamus or Squamish Rivers, given the hydrologically modified environment in the Lower Cheakamus River. The impoundment is also believed to have an indirect effect on the grazing pressure exerted on Lower Cheakamus periphyton community (Steinman, 1991). The process of nutrient spiraling in the Upper Cheakamus would likely result in minimal sequestering within the periphyton community, given the high velocities (>1.0 m/sec), high river volume to surface ratios, episodic scouring, high turbidity and low biomass values typically recorded (Mulholland, 1993; pers. comm.).

Our study confirmed that of Lohman *et al.*, (1992) who have shown that periphytic chlorophyll *a* was significantly reduced following major storm events. We also confirmed their finding that the most rapid increase in post-storm chlorophyll *a* accrual occurred at nutrient enriched sites. Interestingly, the community structures of the Upper and Lower Cheakamus River's differ

with the former dominated by a diatomaceous community and the latter by a Chlorophyte canopy and diatom understory (Figure 66). The Chlorophytes are a sub-dominant community group in the Upper Cheakamus River. It should be noted, however, that any understanding of the periphyton community structures will require that both bottom-up and top-down processes be further explored (Rosemond *et al.*, 1993). Power and her students (1991) have demonstrated that the number of trophic linkages within a food web (especially non-algal elements) can have a significant affect on periphyton community architecture and accrual (Power 1992). Knowledge of the number of trophic levels at the sample sites on the Cheakamus River should reveal whether they are different and if such differences are affecting periphyton structure (Vaughn *et al.*, 1993).

The principal criticism of this study has been the linking of STP-based orthophosphorus and the excessive Lower Cheakamus River algal accrual. The major concern expressed has been that there exists no statistically based, scientific evidence that the phosphorus leaving the plant could reach the river below the dam; further, that the reservoir must be trapping the phosphorus in the sediment, thereby rendering it biologically unavailable downstream. A second distinction is made between scientific conclusions, deduced from an objective analysis of the facts and reasoned interpretation of environmental parameters as a basis for establishing regulatory permitted discharges. The implication is that this distinction is real and that the former is in some manner not also based on objectively derived ecological fact. Thus, the issue is not whether, for the ecosystem in question, there exists a body of knowledge which irrefutably proves the causal link between effluent discharge and loss of water quality but whether there exists enough knowledge to recommend a cautious permitting approach until the causal links are established. One aspect of this approach is the increasing recognition that the use of single nutrient criteria for protecting aquatic habitat and quality is less likely to be as effective as are pollution abatement permits based upon ecosystem functions.

## Summary

This study's findings have confirmed that the Lower Cheakamus River is very sensitive to phosphorus enrichment, at a level of at least an order of magnitude lower than previously reported in the literature for similar streams and rivers. As a result of this work and inasmuch as management of the aquatic natural resources of the Cheakamus River have been the focus of considerable public attention, the original LWMP has been subject to a reevaluation (Table 11). An element of public concern which has recently been receiving increased attention is the recognition that as the rapidly increasing population density of the Cascadia region increases, there will be concomitant pressure on the Cheakamus/ Squamish River (and likely Green River) as primary recreational corridors. The establishment of a public advisory working committee, to significantly increase the extent of public participation in the development of the LWMP, for the RMOW, resulted in a wide range of options being explored. Recently reduced to three, these options will be subject to open public review at public workshops in the communities of Squamish and Whistler, in the latter portion of March, 1993.

Given the extensive resource management literature recommending adoption of non-point source regulation of environmentally perturbing activities (i.e. no end-of-pipe permits), the author has recommended (Lucey *et al.*, 1992) that a watershed management plan for the Squamish, Cheakamus and Green Rivers be developed. Regulation of resource use within these valleys, which could generate higher than natural background nutrient loadings, will dictate future water and habitat quality. In addition, since little is known of the ecology of these watersheds, especially that of the aquatic habitat, it is recommended that water quality studies be undertaken to determine the relationships between nutrient loading and the health of aquatic wildlife.

## Recommendations

Throughout the autumn of 1992 and the winter/spring of 1993, a Public Participation Committee examined a re-evaluation of Liquid Waste Management Plan options (Table 11). The options originally defined (11) were reduced to three. At present the option which has been identified by the RMOW is for an expansion of the existing plant, which would receive all liquid wastes from the community, together with biological tertiary treatment to further reduce phosphate loadings. It is not clear from the studies conducted thus far that the installation of such additional treatment will significantly reduce the periphyton biomass accrual below the Daisy Dam. It is the recommendation of this author that until such time as the installation of such a facility can be demonstrated to reduce the phosphate loading, and the resultant algal accrual, in the Lower Cheakamus River, such an expense of public funds is contrary to the overall best interests in maintaining the ecological health of the region. Since the capital and operating costs of such enhanced treatment are a significant expenditure of public funds likely to be spent on environmental mitigative processes in this valley, such an expenditure may not be the most effective in preserving the wilderness nature of the Cheakamus and Squamish River upon which much of the valley communities economics are dependent.

Additional studies have been proposed, and at present are being initiated, to further our understanding of the nutrient spiraling on the Cheakamus River and within Daisy reservoir. Additional surrogate stream studies are being conducted to determine algal response to nutrients in the surface and deeper waters of the reservoir. Two separate systems have been established, one draws water from the surface and the other from twenty feet below the surface. Orthophosphorus is being added to two of the four streams, of each system, to determine periphyton accrual response to below chemically detectable increases in phosphorus. Table 11 outlines the studies being undertaken during the summer of 1993.

The alternatives outlined in Table 11 have centered around the primary option of expanding the STP's existing treatment capacity, together with an

additional mode of treatment to further reduce the orthophosphorus load in the discharged effluent; effluent organic carbon and nitrogen have not received much attention. The principal aim of this process has been the need to reduce the phosphorus loading in the Lower Cheakamus River (below Daisy Dam), the accrual of which is energized by increasing phosphorus loading.

The limited data on loading and sources of phosphorus, in the Upper Cheakamus River and within Daisy Reservoir, make understanding the downstream nutrient spiraling difficult. Further, the data here suggest that the reduction in phosphorus loading from the treatment plant will not significantly reduce the phosphorus concentrations, and hence the algal biomass accrual, below the dam. The absence of data on the environmental fate and cycling of phosphorus within the reservoir, and from phosphorus rich volcanic parent materials being weathered and mobilized within the catchment basin, necessitate additional studies to begin our understanding of where and how nutrients are being transported through this system. It would thus seem premature, in the absence of this information, to require the additional biological-based stripping of orthophosphorus from the RMOW's STP. It should be noted, however, that it is inappropriate to implement, in the LWMP, the biological-based stripping process without first conducting additional investigations designed to elucidate how nutrients cycle through this river system. If the biological-based nutrient stripping is not mandated at this time then studies must be conducted to determine what other management options will effect an improvement in the downstream aquatic habitat and water quality. For example, it may be that a modified protocol for management of the Daisy Reservoir will be required. A modest proposal on how to financially administer the needed study funds is presented as the final section of Chapter V (general conclusions).

Table 10. Percentage of dissolved orthophosphorus and flow contributions from tributaries, and the STP, to the Cheakamus River.

| Source | 1986/87 | 1988 | 1990 | Flows |
|---|---|---|---|---|
| Cheakamus R. | 27.6 | 24.9 | 22.0 | 32.9 |
| Callaghan Ck. | 37.7 | 33.8 | 29.9 | 23.5 |
| Brandywine Ck. | 11.2 | 9.0 | 8.0 | 8.9 |
| Millar Ck. | 0.5 | 0.4 | 0.3 | 5.9 |
| Rubble Ck. | 13.8 | 11.3 | 10.0 | - |
| Smaller Cks. | - | - | - | 28.8 |
| Effluent (STP) | 9.2 | 20.6 | 29.8 | - |
| Total | 100.0 | 100.0 | 100.0 | 100.0 |

Table 11. The original expanded liquid waste management plan options developed through a public consultation process and the reduced sub-set of options from which the final option #3 was selected. * Up-grade treatment plant to an advanced, chemical treatment process.

| **Nov. 5 1992** | # Treat-ment plants 1 | 2 | All reduce comp-onents | Cheak-amus R. | Squa-mish R. | Howe Sound | **Discharge to** Green R. | Ground | **Disposal to** Forest Land | Wet-lands | Cheak-amus River **Research** | Summer irrigation & winter snow |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| A | 1 | | | + | | | | | | | | |
| B | 1 | | + | + | | | | | | | | |
| C | 1 | | + | portion | | | | | | | | portion |
| D | 1 | | + | | | | | + | | | | |
| E | 1 | | + | winter | | | | summer | | | | |
| F | 1 | | + | | + | | | | | | | |
| G | 1 | | + | | | + | | | | | | |
| H | | 2 | + | + | | | + | | | | | |
| I | | 2 | + | | | | | + | | | | |
| J | 1 | | + | | | | | | + | | | |
| K | | 2 | + | | | | | | | + | | |
| | | | | | | | | | | | | |
| **Feb. 4 1993** | | | | | | | | | | | | |
| 1 | * | | + | + | | | | | | | + | |
| 2 | * | | + | storage, dischrge | | | | | | | + | |
| 3 | | * | + | | | | | Emerald Estates | | | + | |

Figure 30. Cheakamus and Squamish River sample site and stream-trough locations; Site 1 at the base of Daisy Dam. The B.C. Hydro diversion tunnel, through which the majority of Cheakamus River water flows, discharges into a 0.25 Km. side-channel, the confluence of which lies 0.5 Km. above the Site 2 stream-trough location.

Cheakamus River sample sites (A-M), principal tributaries, Whistler Village (RMOW), STP, B.C. Hydro tunnel, stream-trough facility (adjacent to K); small box on insert = Cheakamus R. watershed. Numeric codes = M.O.E. unique sample site designation; D/S = downstream; arrow = flow direction; dashed line = watershed boundary between Cheakamus and Green Rivers.

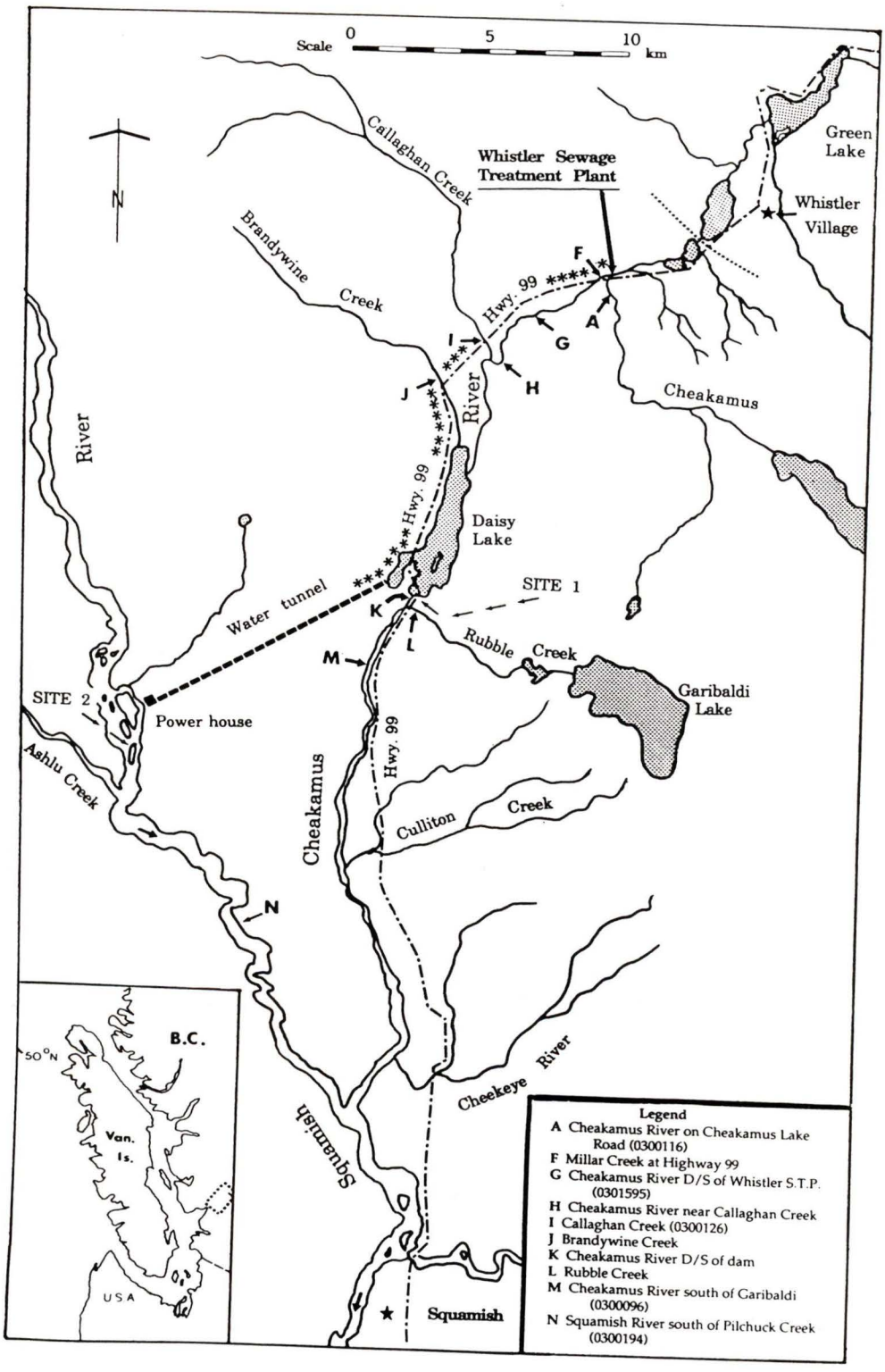
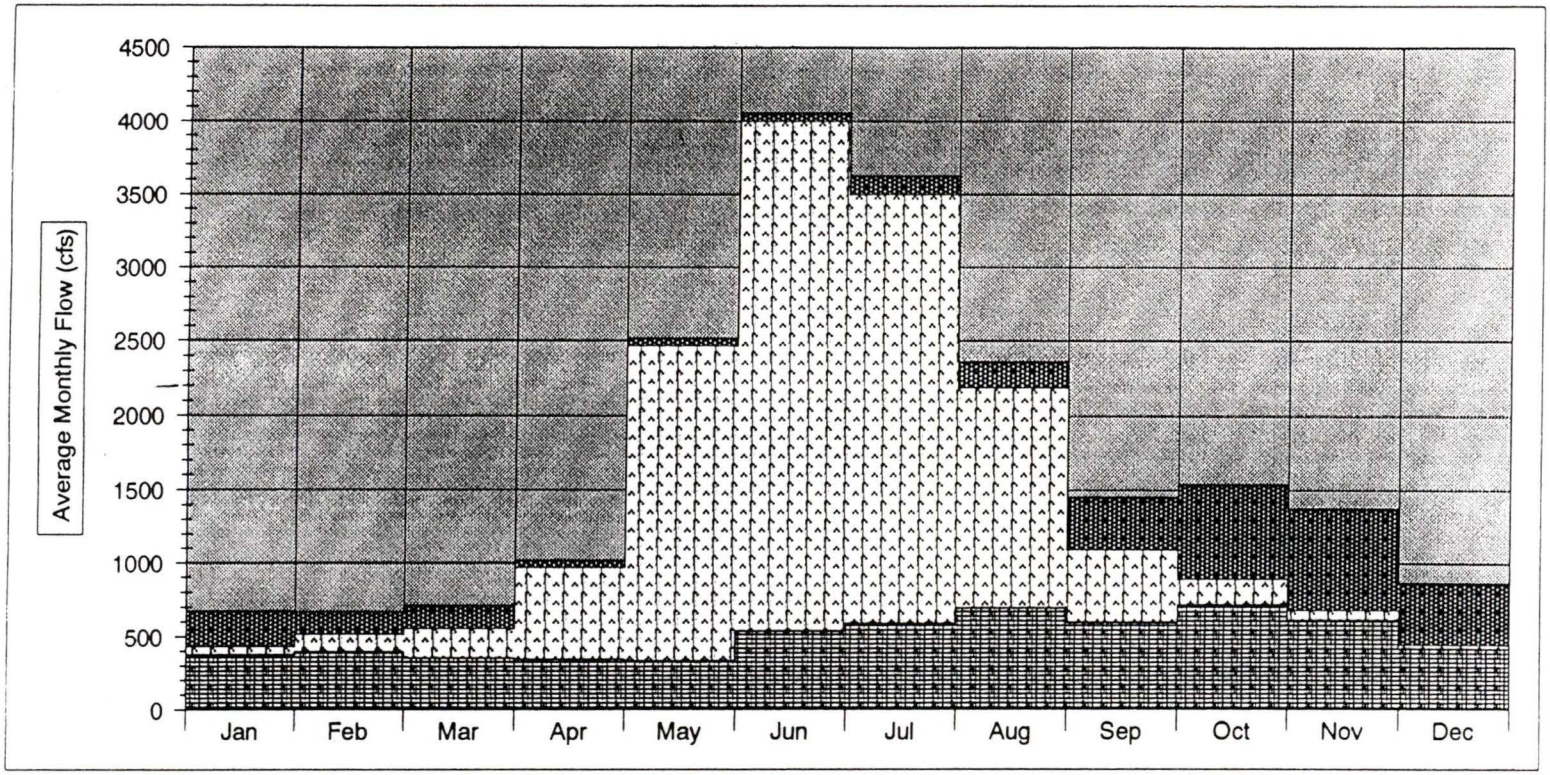

Scale 0 5 10 km
N
Callaghan Creek
Whistler Sewage Treatment Plant
Green Lake
Whistler Village
Brandywine Creek
Hwy. 99
F
A
G
I
H
J
River
Cheakamus
River
Hwy. 99
Daisy Lake
SITE 1
Water tunnel
K
L
Rubble Creek
M
Garibaldi Lake
SITE 2
Power house
Ashlu Creek
Hwy. 99
Culliton Creek
Cheakamus
N
B.C.
50°N
Van. Is.
USA
Squamish River
Cheekeye River
Squamish
Legend
A Cheakamus River on Cheakamus Lake Road (0300116)
F Millar Creek at Highway 99
G Cheakamus River D/S of Whistler S.T.P. (0301595)
H Cheakamus River near Callaghan Creek
I Callaghan Creek (0300126)
J Brandywine Creek
K Cheakamus River D/S of dam
L Rubble Creek
M Cheakamus River south of Garibaldi (0300096)
N Squamish River south of Pilchuck Creek (0300194)

Figure 31. Mean monthly Cheakamus River inflows to Daisy Reservoir for the period 1961 to 1990, consisting of base flows, snowmelt and rainfall components. (From B.C. Hydro).

# Daisy Monthly Inflows

components

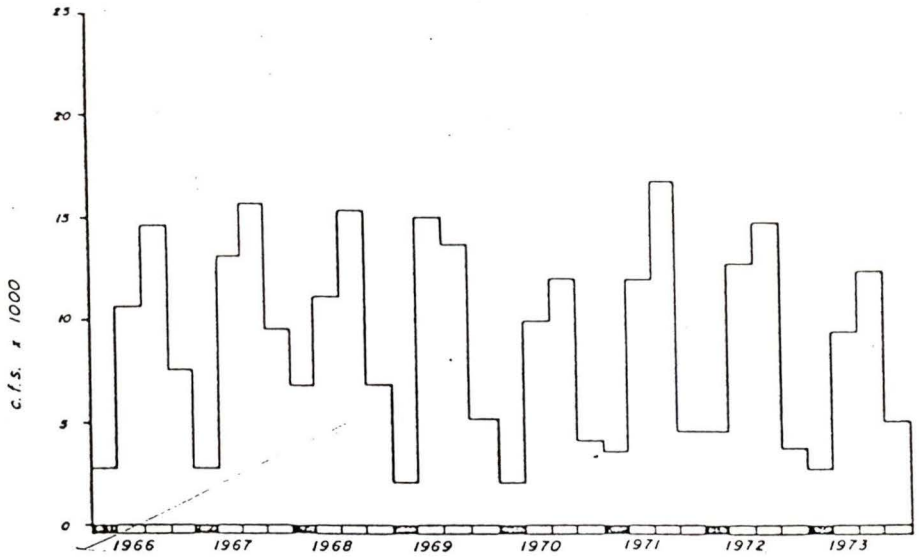


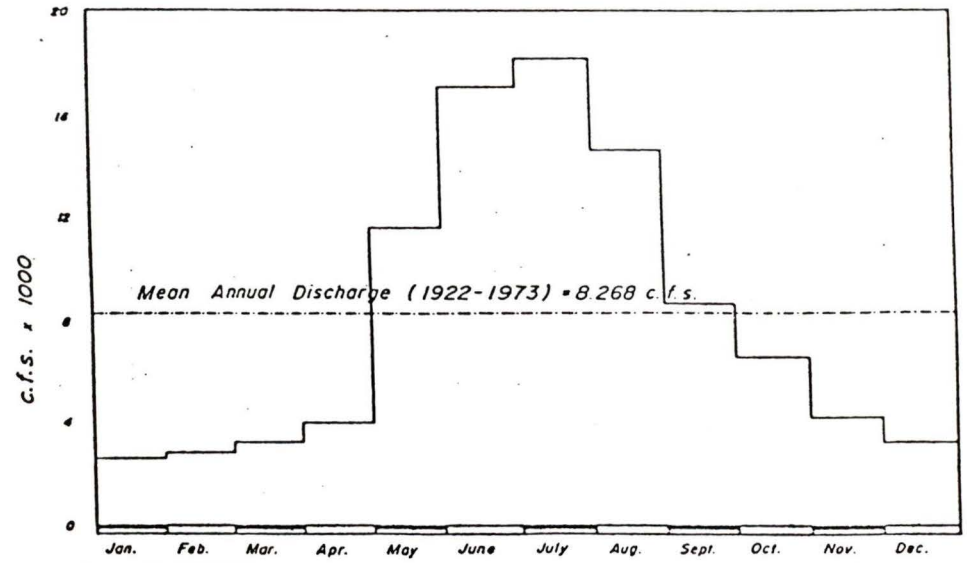

1961 -90 average basin inflow prorated to compositionfactors provided by Operations Hydrology in 1977

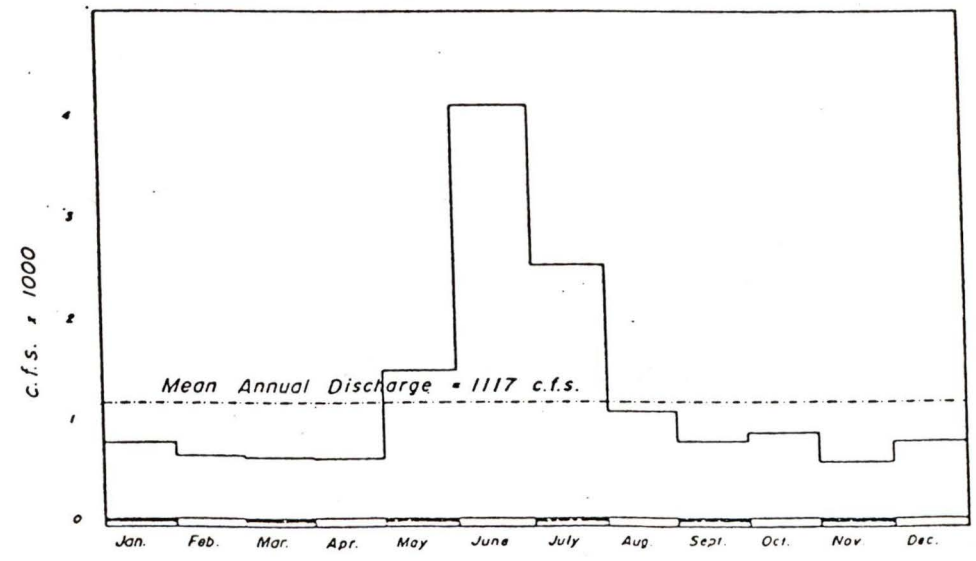
BASE SNOWMELT RAIN

By; WGJ
28-Apr-92

Figure 32. Upper: Squamish River mean annual discharges (1966-1973); Middle: Squamish River mean monthly discharges (1922-1973); Lower: Cheakamus River mean monthly discharges into the Squamish River for the period 1967 to 1973. (from L.M. Bell, M.A.Sc. thesis, U.B.C.)


c.f.s. x 1000
25
20
15
10
5
0
1966
1967
1968
1969
1970
1971
1972
1973


c.f.s. x 1000
20
16
12
8
4
0
Mean Annual Discharge (1922-1973) = 8,268 c.f.s.
Jan.
Feb.
Mar.
Apr.
May
June
July
Aug.
Sept.
Oct.
Nov.
Dec.


c.f.s. x 1000
4
3
2
1
0
Mean Annual Discharge = 1117 c.f.s.
Jan.
Feb.
Mar.
Apr.
May
June
July
Aug.
Sept.
Oct.
Nov.
Dec.

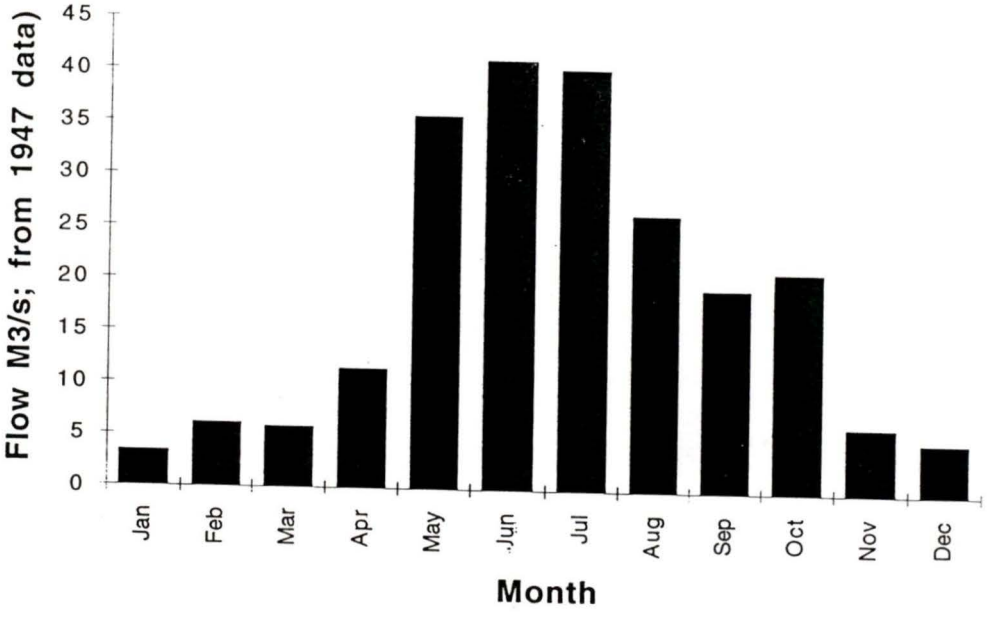
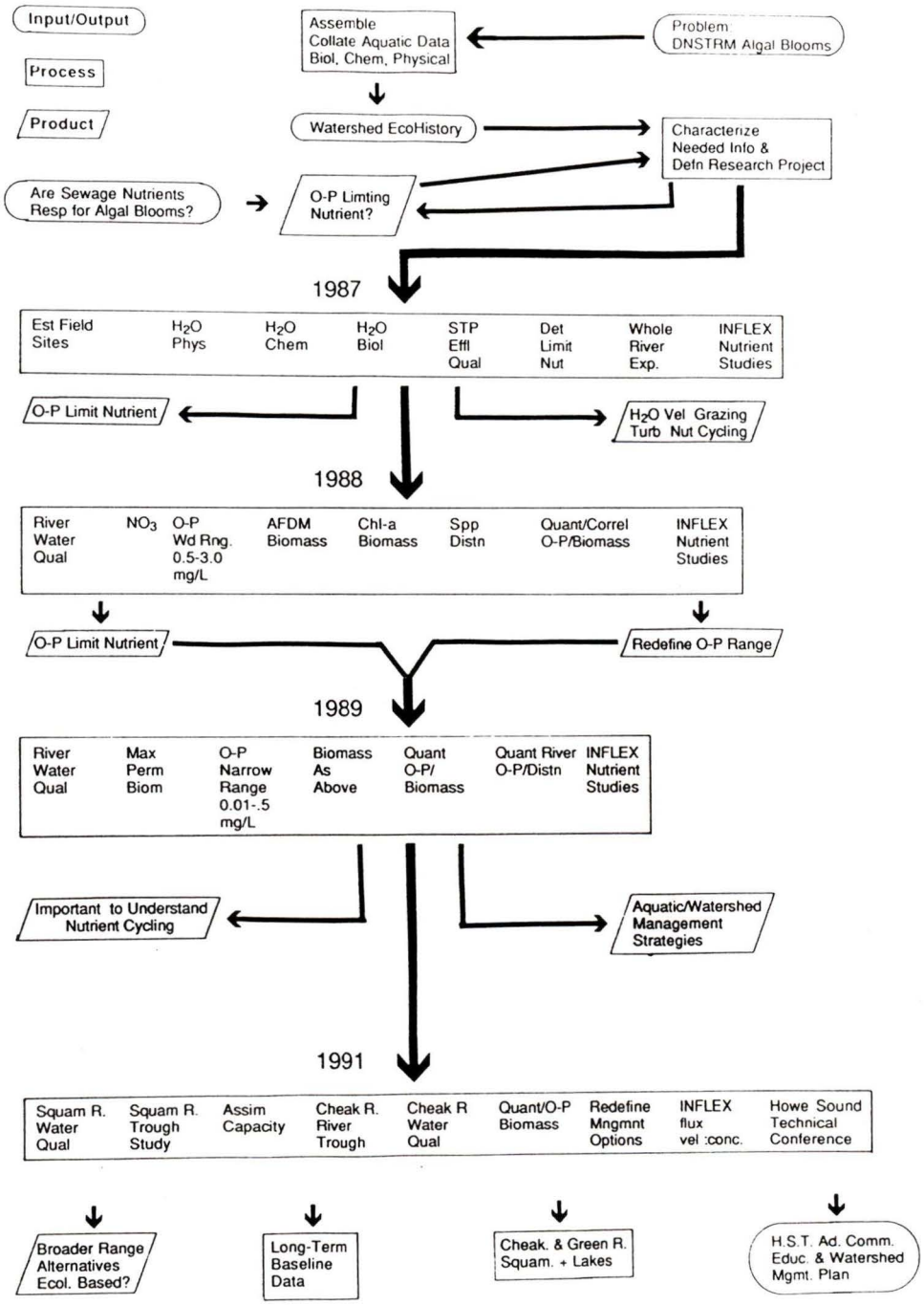



Figure 33. Mean monthly Cheakamus River flows ($m^3\ s^{-1}$) for the period 1947 to 1975.

Figure 34. Schematic flow chart of the research protocol developed to address the concern raised by increased algal biomass in the lower Cheakamus River. Linked functional attributes of the protocol are identified by three distinct shapes, noted in the upper left-hand corner: input/output variables, process or activities, and products or understanding. The public advisory committee was designed and implemented in 1992/1993.

Abbreviations:

DNSTRM - Downstream
O-P - orthophosphate
Vel - velocity
Turb - turbidity
Nut - nutrient
Wd Rng - wide range
AFDM - Ash Free Dry Mass
Quant/Correl - quantify & correlate
Max Perm Biom - Maximum permissible biomass
H.S.T. Ad. Comm. - Howe Sound Technical Advisory Committee
Educ. - Education
Mgmt. - Management


Input/Output
Process
Product
Assemble
Collate Aquatic Data
Biol, Chem, Physical
Problem:
DNSTRM Algal Blooms
Watershed EcoHistory
Characterize
Needed Info &
Defn Research Project
Are Sewage Nutrients
Resp for Algal Blooms?
O-P Limting
Nutrient?
1987
Est Field Sites
H2O Phys
H2O Chem
H2O Biol
STP Effl Qual
Det Limit Nut
Whole River Exp.
INFLEX Nutrient Studies
O-P Limit Nutrient
H2O Vel Grazing Turb Nut Cycling
1988
River Water Qual
NO3
O-P Wd Rng. 0.5-3.0 mg/L
AFDM Biomass
Chl-a Biomass
Spp Distn
Quant/Correl O-P/Biomass
INFLEX Nutrient Studies
O-P Limit Nutrient
Redefine O-P Range
1989
River Water Qual
Max Perm Biom
O-P Narrow Range 0.01-.5 mg/L
Biomass As Above
Quant O-P/ Biomass
Quant River O-P/Distn
INFLEX Nutrient Studies
Important to Understand Nutrient Cycling
Aquatic/Watershed Management Strategies
1991
Squam R. Water Qual
Squam R. Trough Study
Assim Capacity
Cheak R. River Trough
Cheak R Water Qual
Quant/O-P Biomass
Redefine Mngmnt Options
INFLEX flux vel :conc.
Howe Sound Technical Conference
Broader Range Alternatives Ecol. Based?
Long-Term Baseline Data
Cheak. & Green R. Squam. + Lakes
H.S.T. Ad. Comm. Educ. & Watershed Mgmt. Plan

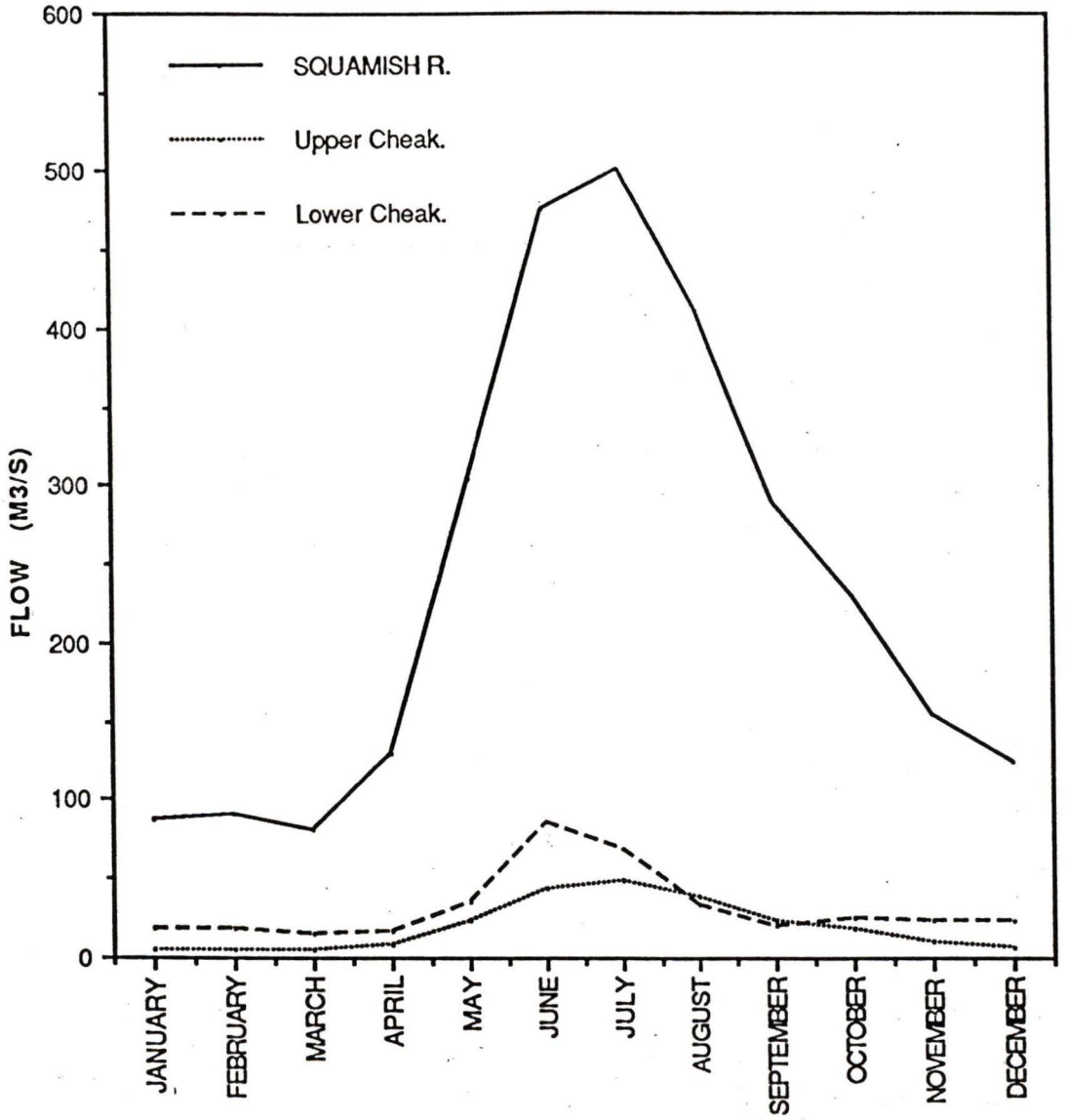
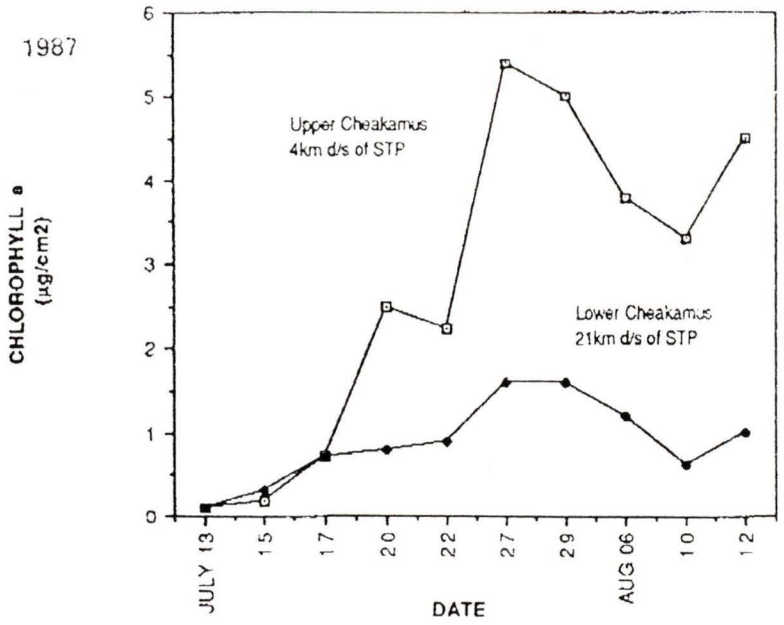



Figure 35. Long term, monthly flow averages in both the Squamish and Upper and Lower Cheakamus Rivers.

Figure 36. Estimates of Cheakamus River algal biomass (measured as chlorophyll *a*) accrual patterns at sample sites 4 and 21 Km below the STP.


1987
CHLOROPHYLL a
(µg/cm2)
Upper Cheakamus
4km d/s of STP
Lower Cheakamus
21km d/s of STP
0
1
2
3
4
5
6
JULY 13
15
17
20
22
27
29
AUG 06
10
12
DATE

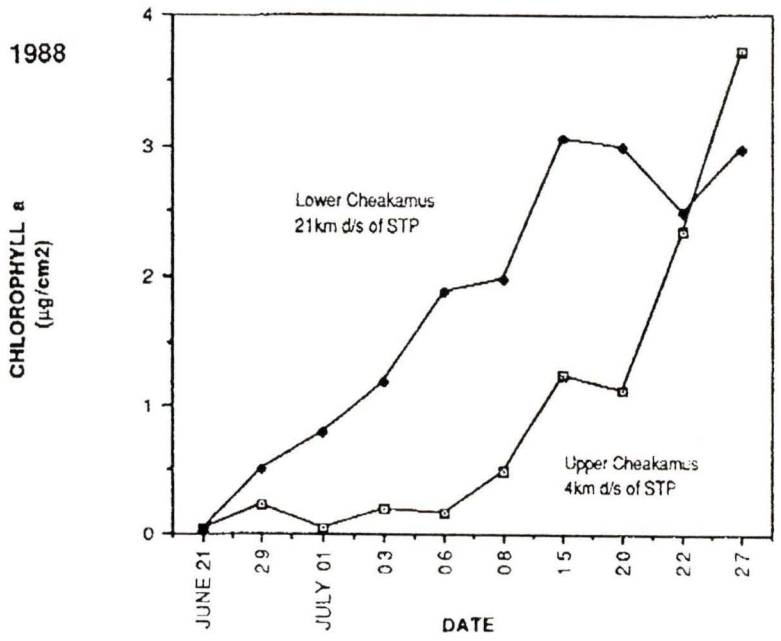
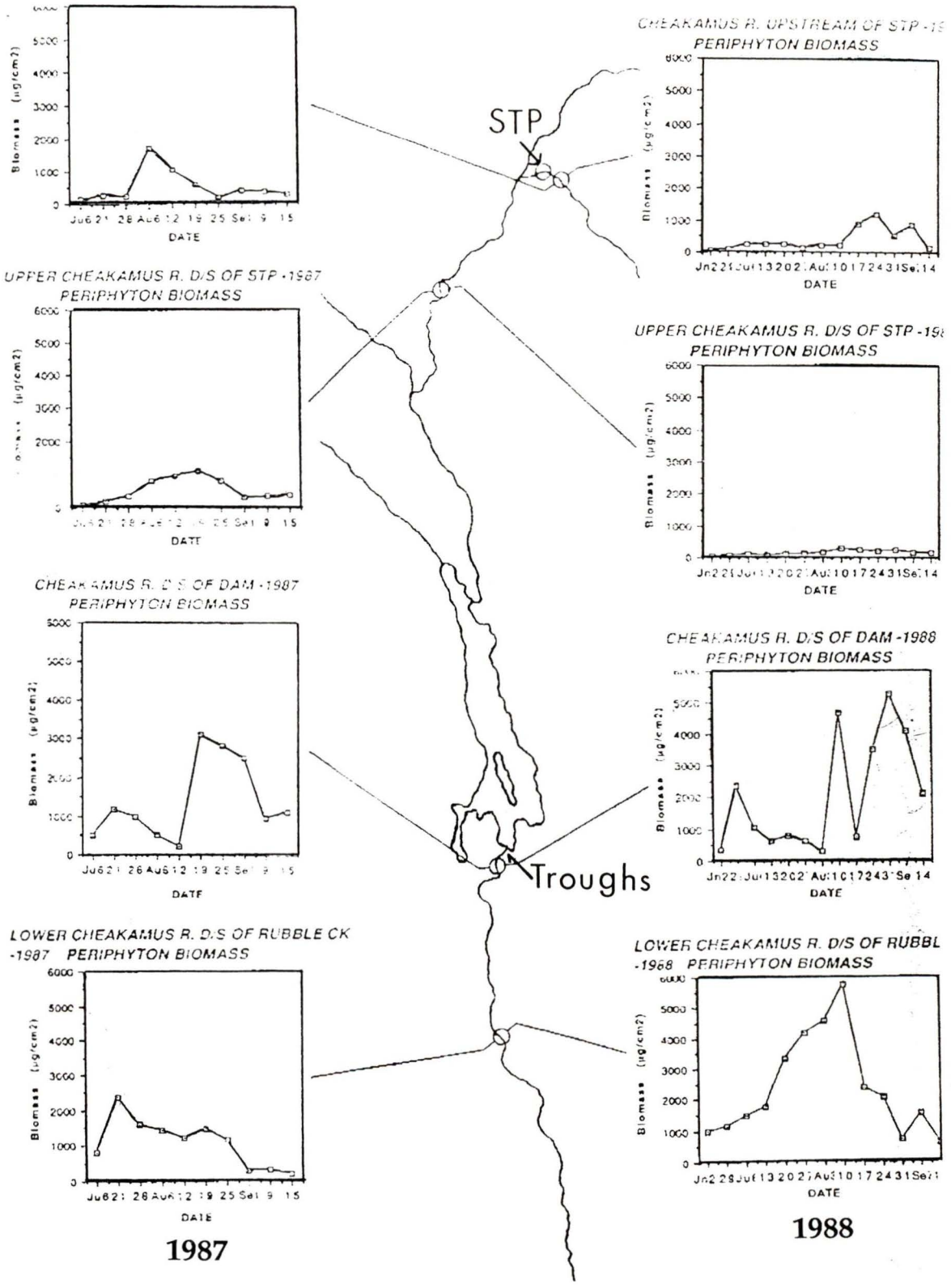

1988
CHLOROPHYLL a
(µg/cm2)
Lower Cheakamus
21km d/s of STP
Upper Cheakamus
4km d/s of STP
0
1
2
3
4
JUNE 21
29
JULY 01
03
06
08
15
20
22
27
DATE

Figure 37. Algal biomass ($\mu g/cm^2$) measured at four Cheakamus River stations, during the summer periods of 1987 through 1989.


CHEAKAMUS R. UPSTREAM OF STP -19
PERIPHYTON BIOMASS
STP
UPPER CHEAKAMUS R. D/S OF STP -1987
PERIPHYTON BIOMASS
UPPER CHEAKAMUS R. D/S OF STP -19
PERIPHYTON BIOMASS
CHEAKAMUS R. D/S OF DAM -1987
PERIPHYTON BIOMASS
CHEAKAMUS R. D/S OF DAM -1988
PERIPHYTON BIOMASS
Troughs
LOWER CHEAKAMUS R. D/S OF RUBBLE CK
-1987 PERIPHYTON BIOMASS
LOWER CHEAKAMUS R. D/S OF RUBBL
-1988 PERIPHYTON BIOMASS
Biomass (µg/cm2)
DATE
1987
1988

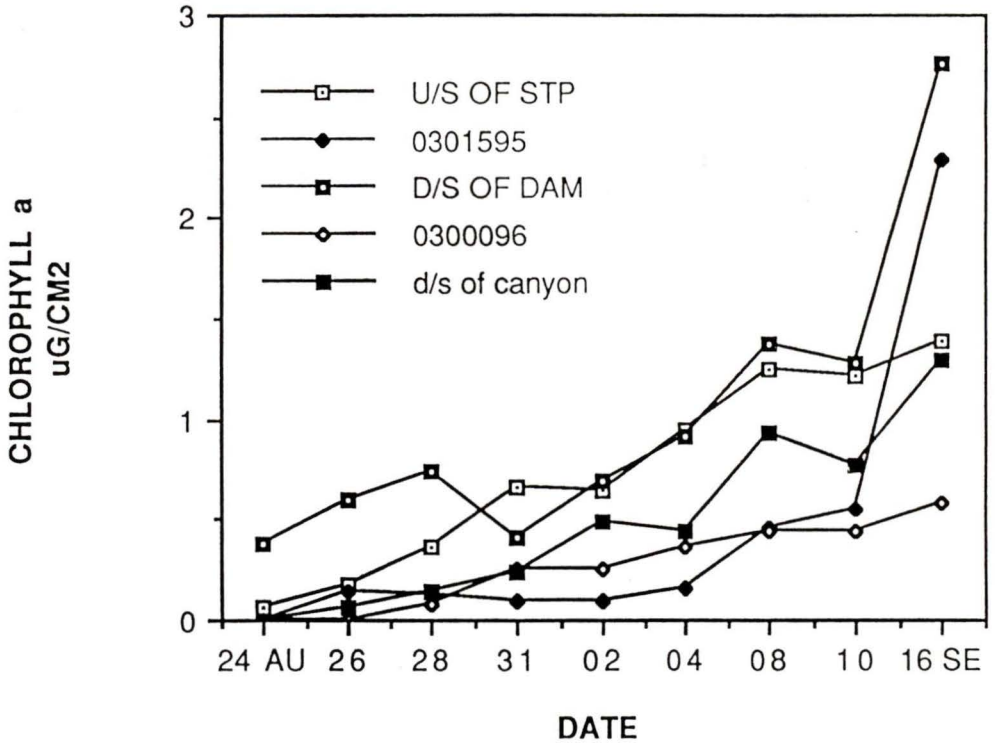
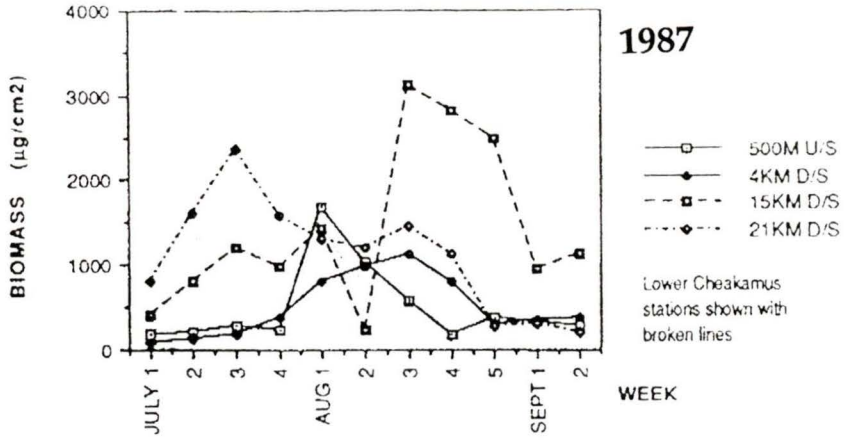
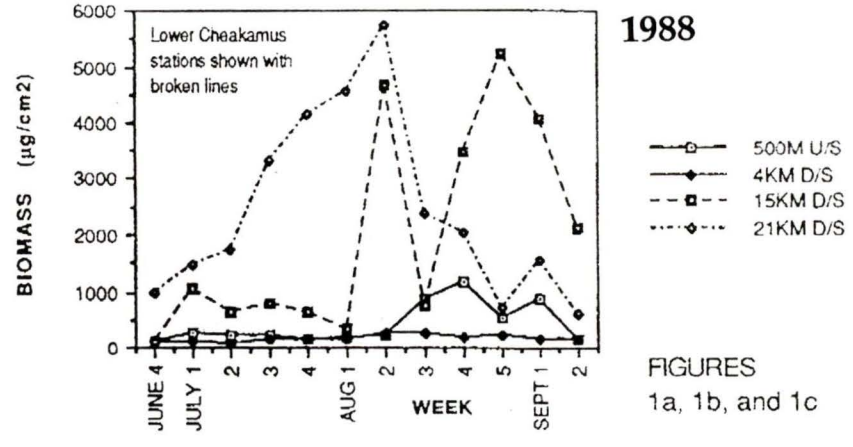


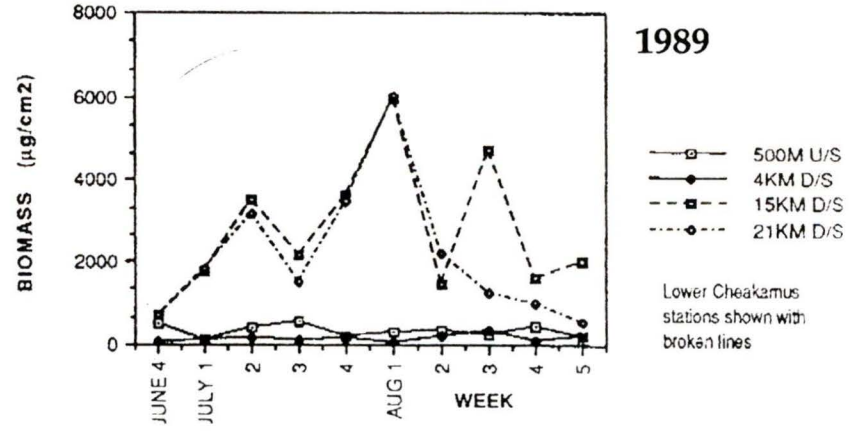

Figure 38. Cheakamus River chlorophyll *a* accrual during the 1987 season.

Figure 39. Biomass accrual measurements obtained at four Cheakamus River sites comparing 1987, 1988 and 1989 data.




FIGURES
1a, 1b, and 1c



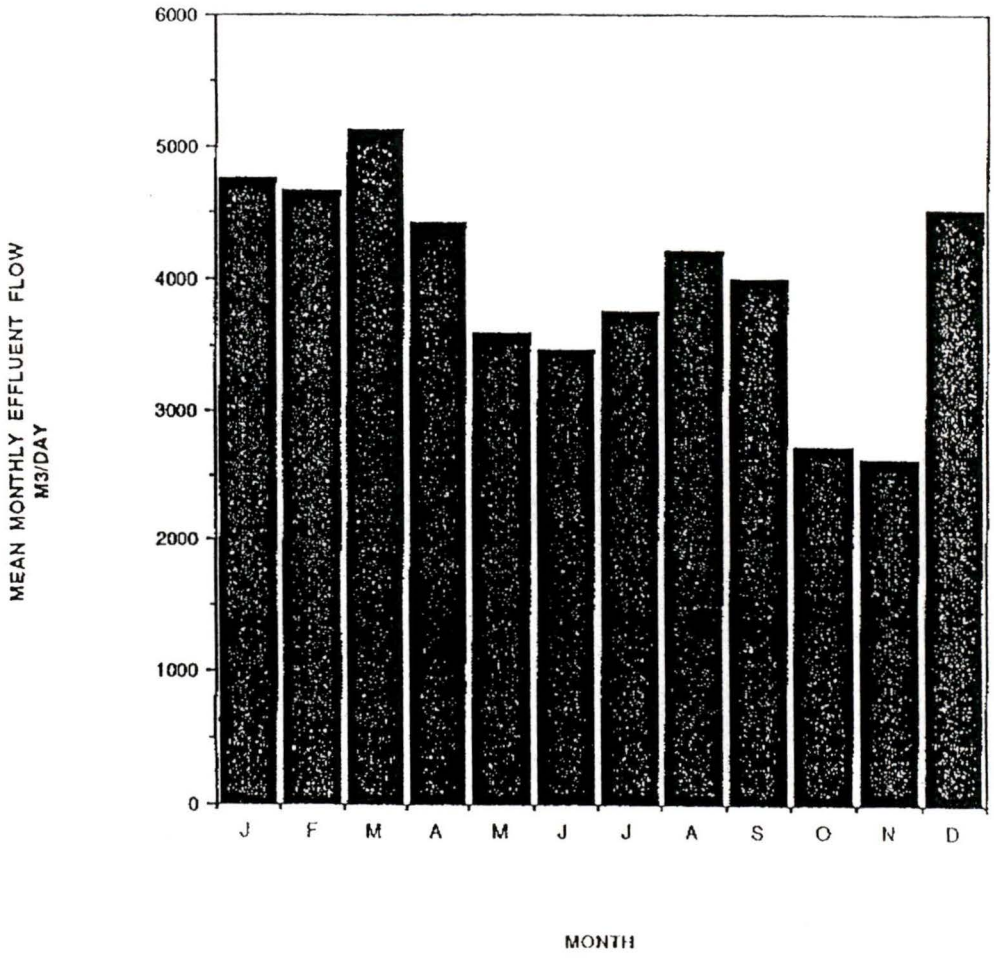
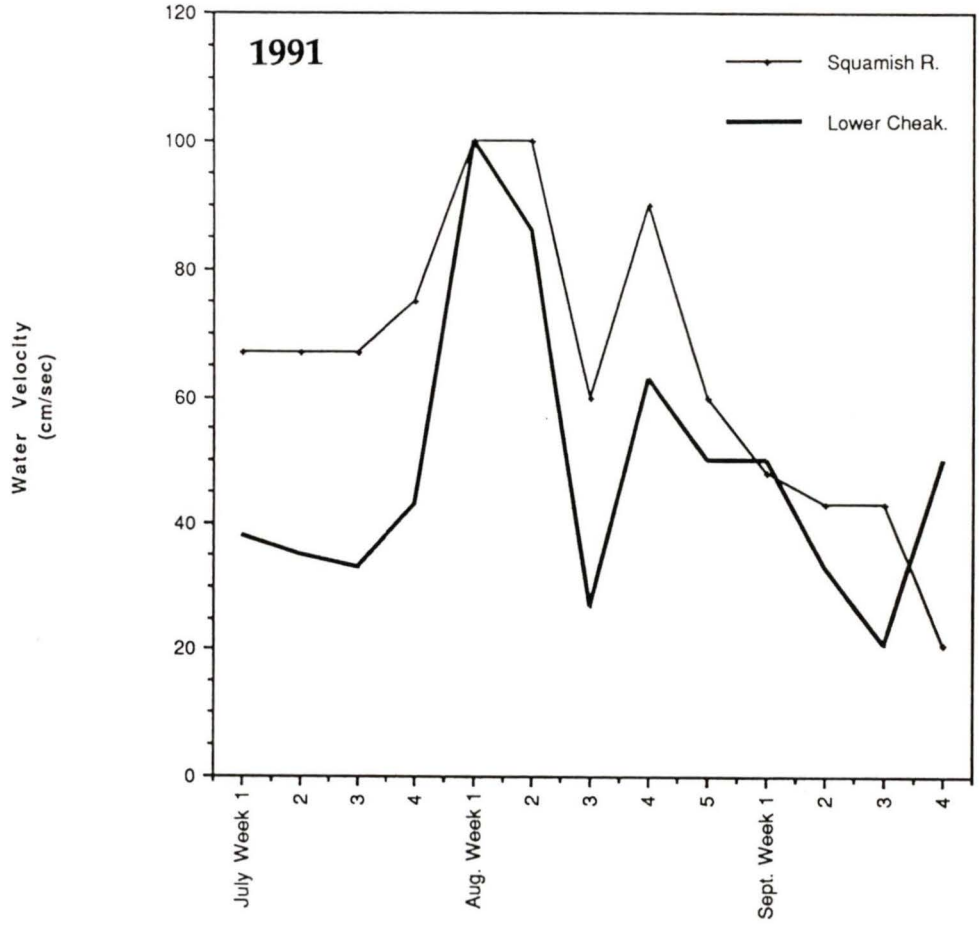



Figure 40. Monthly Whistler sewage treatment plant discharge flows during 1987.

Figure 41. Cheakamus and Squamish River velocities measured during 1991.


1991
Squamish R.
Lower Cheak.
Water Velocity (cm/sec)
120
100
80
60
40
20
0
July Week 1
2
3
4
Aug. Week 1
2
3
4
5
Sept. Week 1
2
3
4

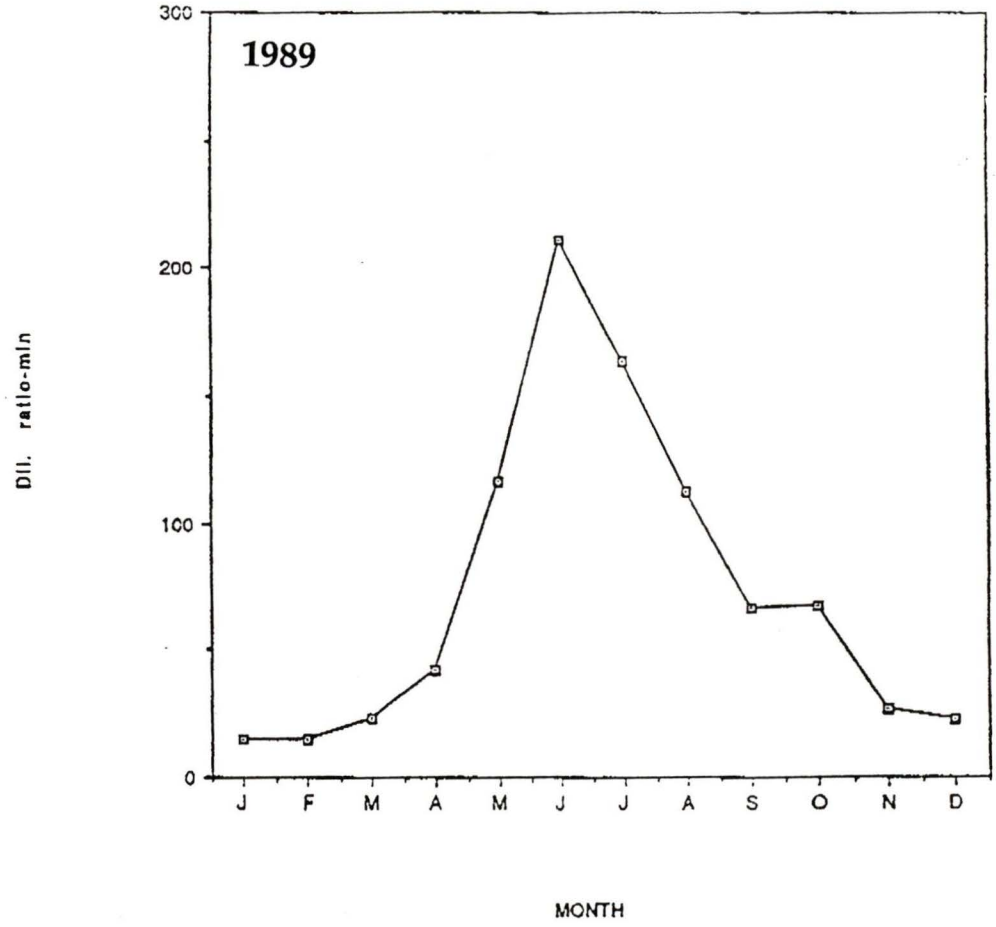



Figure 42. Effluent dilution ratio during minimum Cheakamus River flows.

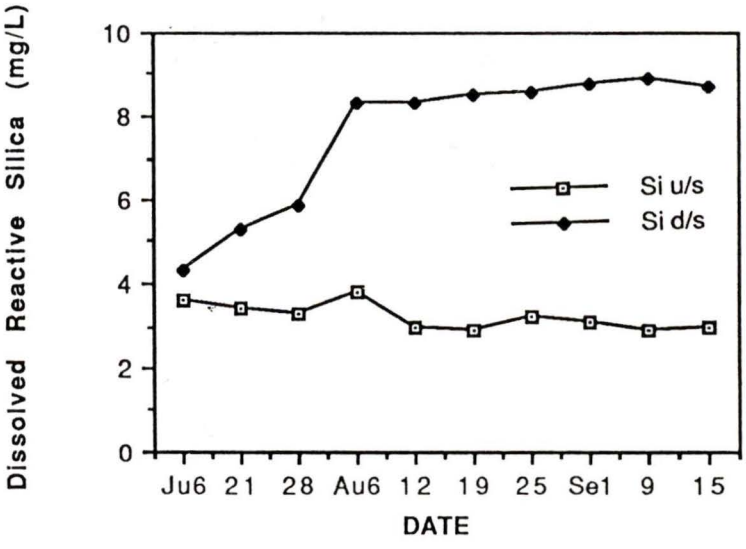
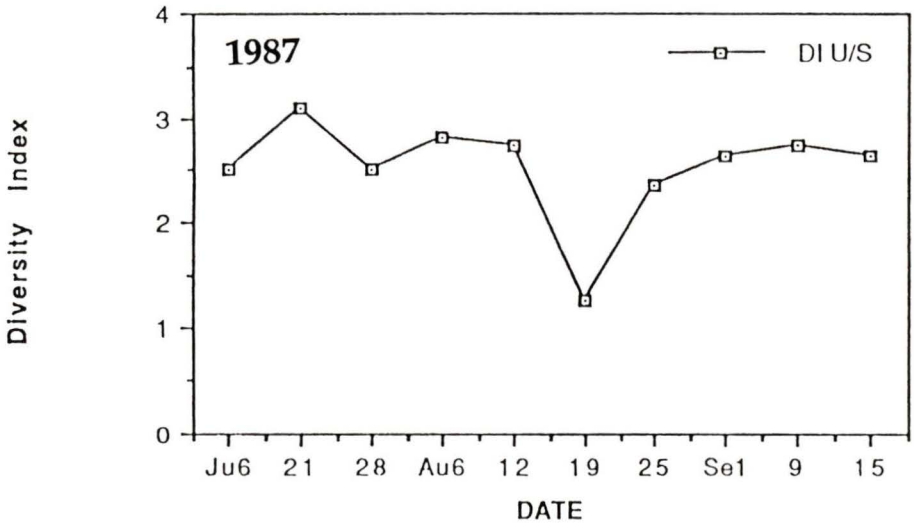



Figure 43. Dissolved reactive silica values in the upper and lower Cheakamus River for 1987.

Figure 44. Macroinvertebrate diversity indices at sample sites 4 and 21 Km below the STP.


1987
DI U/S
Diversity Index
4
3
2
1
0
Ju6 21 28 Au6 12 19 25 Se1 9 15
DATE

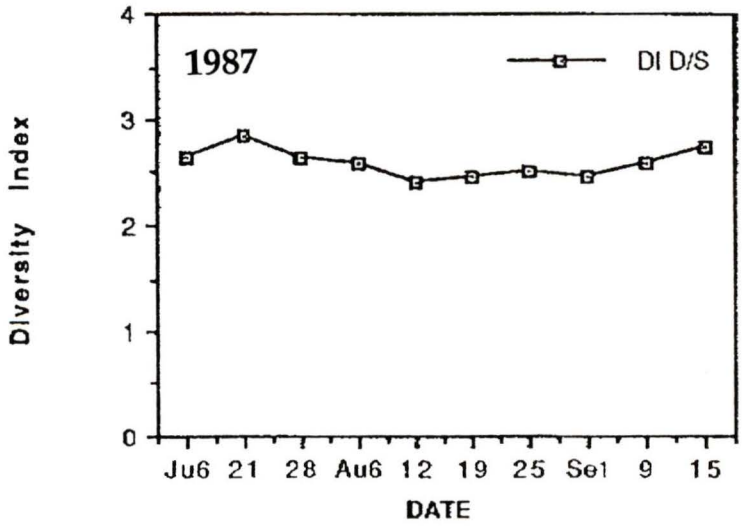
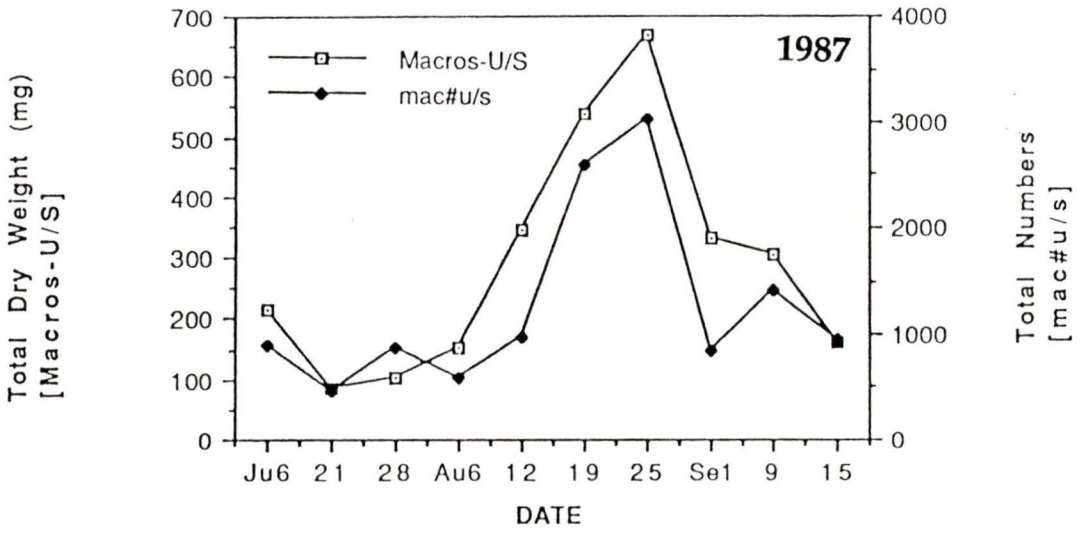

1987
DI D/S
Diversity Index
4
3
2
1
0
Ju6 21 28 Au6 12 19 25 Se1 9 15
DATE

Figure 45. Total dry weight and numbers of macroinvertebrates at sample sites 4 and 21 Km below the STP.


Total Dry Weight (mg)
[Macros-U/S]
Macros-U/S
mac#u/s
1987
Total Numbers
[mac#u/s]
700
600
500
400
300
200
100
0
4000
3000
2000
1000
0
Ju6 21 28 Au6 12 19 25 Se1 9 15
DATE

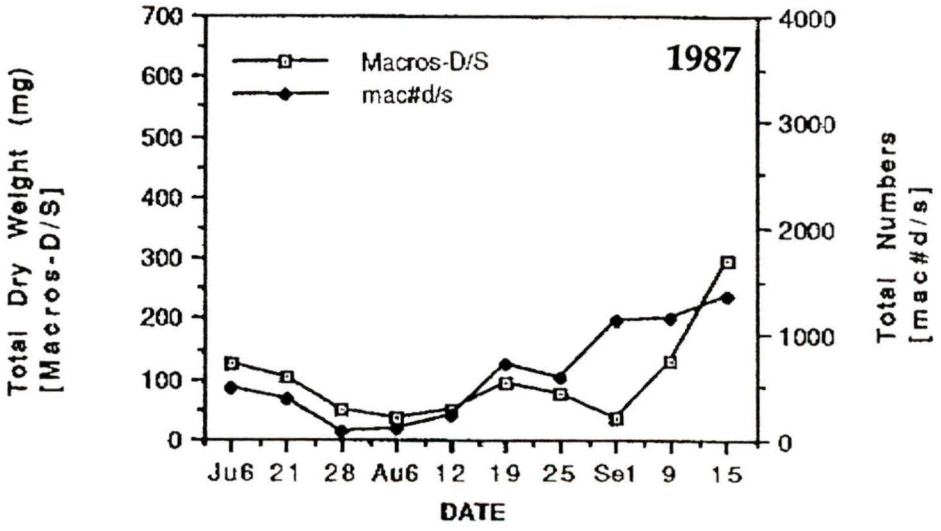
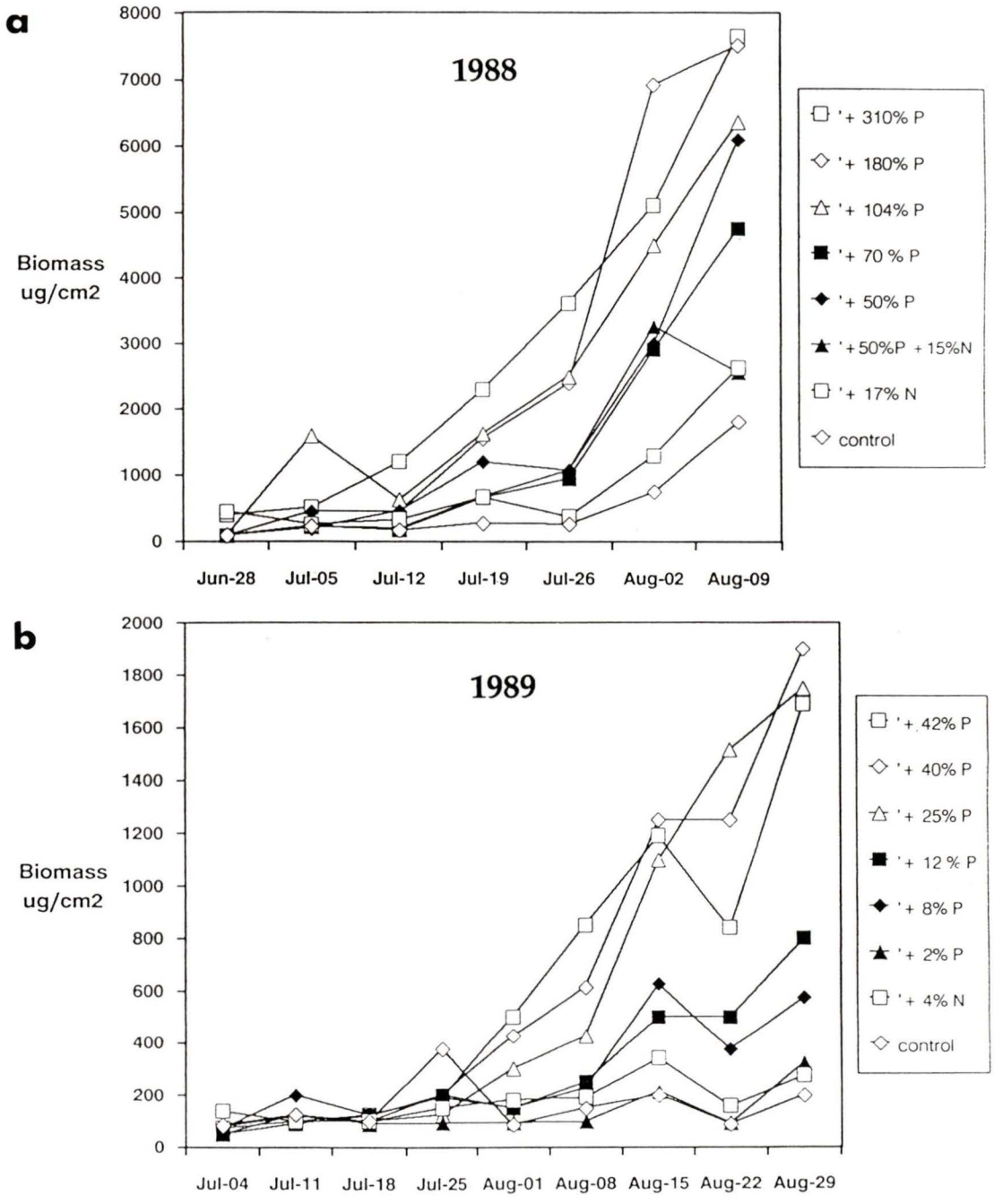
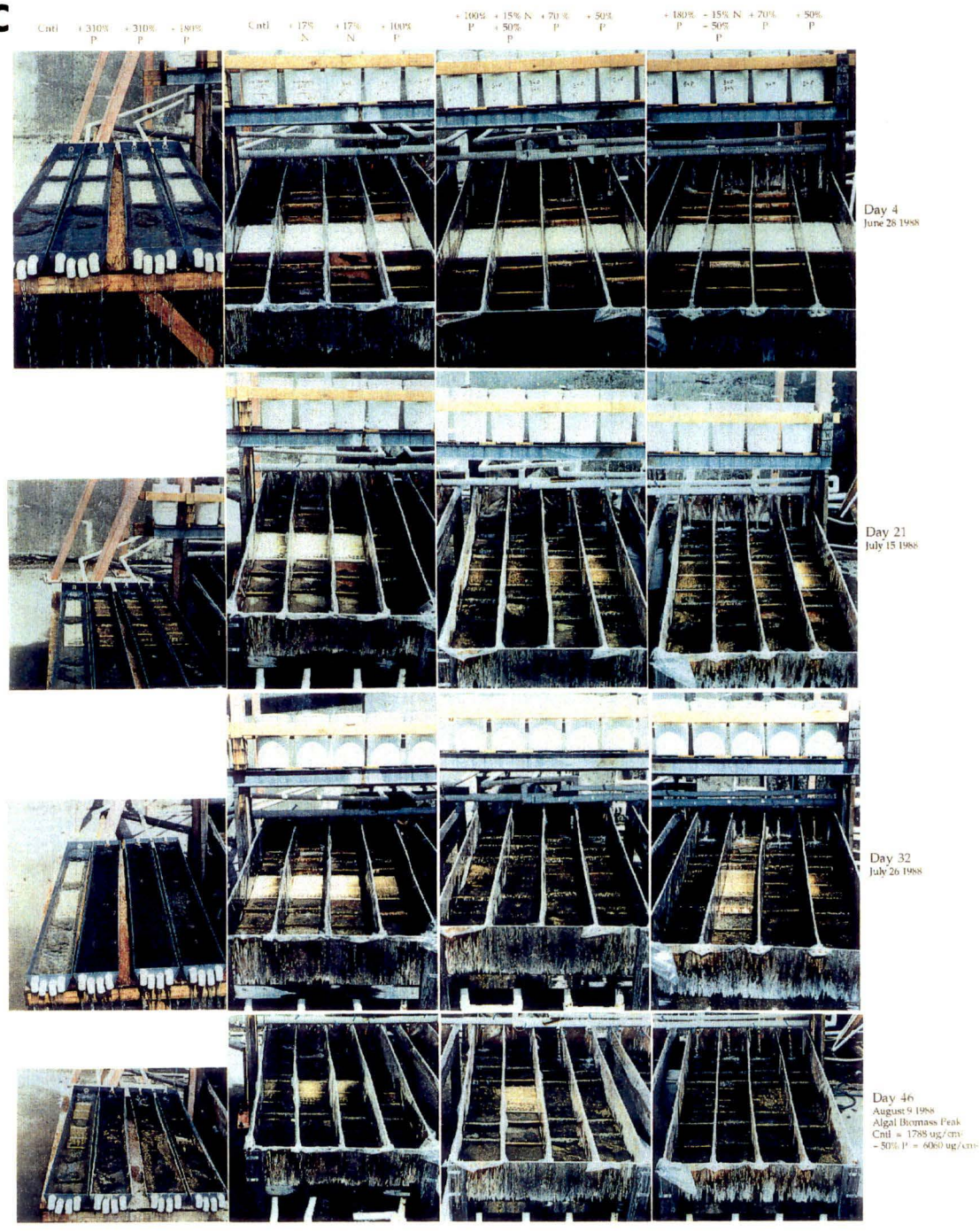

Total Dry Weight (mg)
[Macros-D/S]
Macros-D/S
mac#d/s
1987
Total Numbers
[mac#d/s]
700
600
500
400
300
200
100
0
4000
3000
2000
1000
0
Ju6 21 28 Au6 12 19 25 Se1 9 15
DATE

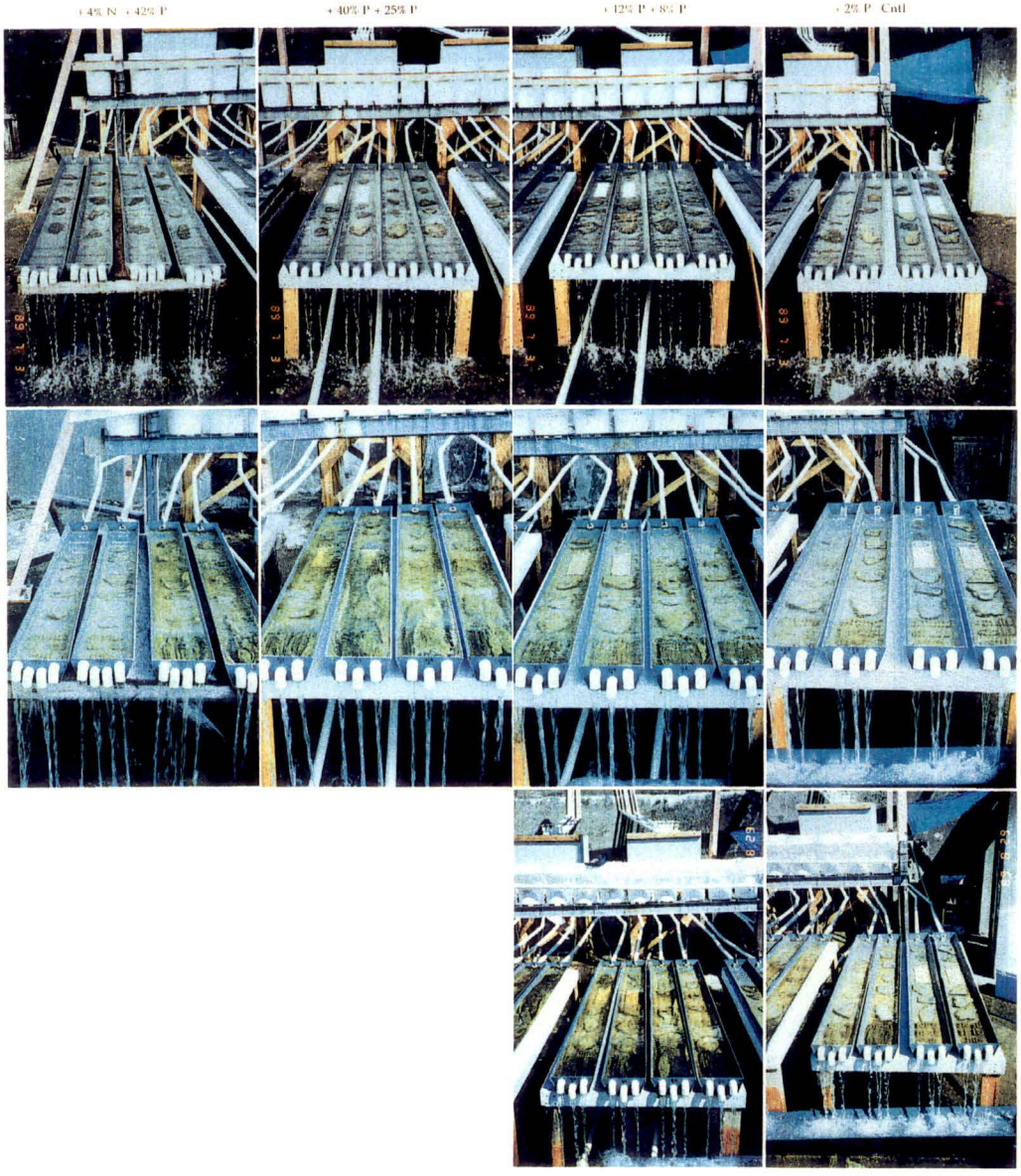
Figure 46a/b. Biomass increases, obtained for 1988 and 1989, at different ortho-P and inorganic N concentrations within paired stream-troughs. Percentage is calculated as a percentage of background were P = 1.0 and N = 25 $\mu g\ L^{-1}$ ; flux rates of nutrient addition were $\mu g\ L^{-1}$ per second.
c. Biomass accrual during the 1988 trough study.
d. Biomass accrual during the 1989 study. Note PVC replacement troughs.
e. Close-up of biomass in troughs after day 53. Note styrofoam and rock substrata.
f. Close-up of biomass on glass slides.
g. Plot of biomass as a function of enhanced orthophosphorus concentration.


a
1988
Biomass ug/cm2
8000
7000
6000
5000
4000
3000
2000
1000
0
Jun-28
Jul-05
Jul-12
Jul-19
Jul-26
Aug-02
Aug-09
'+ 310% P
'+ 180% P
'+ 104% P
'+ 70 % P
'+ 50% P
'+ 50%P + 15%N
'+ 17% N
control
b
1989
Biomass ug/cm2
2000
1800
1600
1400
1200
1000
800
600
400
200
0
Jul-04
Jul-11
Jul-18
Jul-25
Aug-01
Aug-08
Aug-15
Aug-22
Aug-29
'+ 42% P
'+ 40% P
'+ 25% P
'+ 12 % P
'+ 8% P
'+ 2% P
'+ 4% N
control

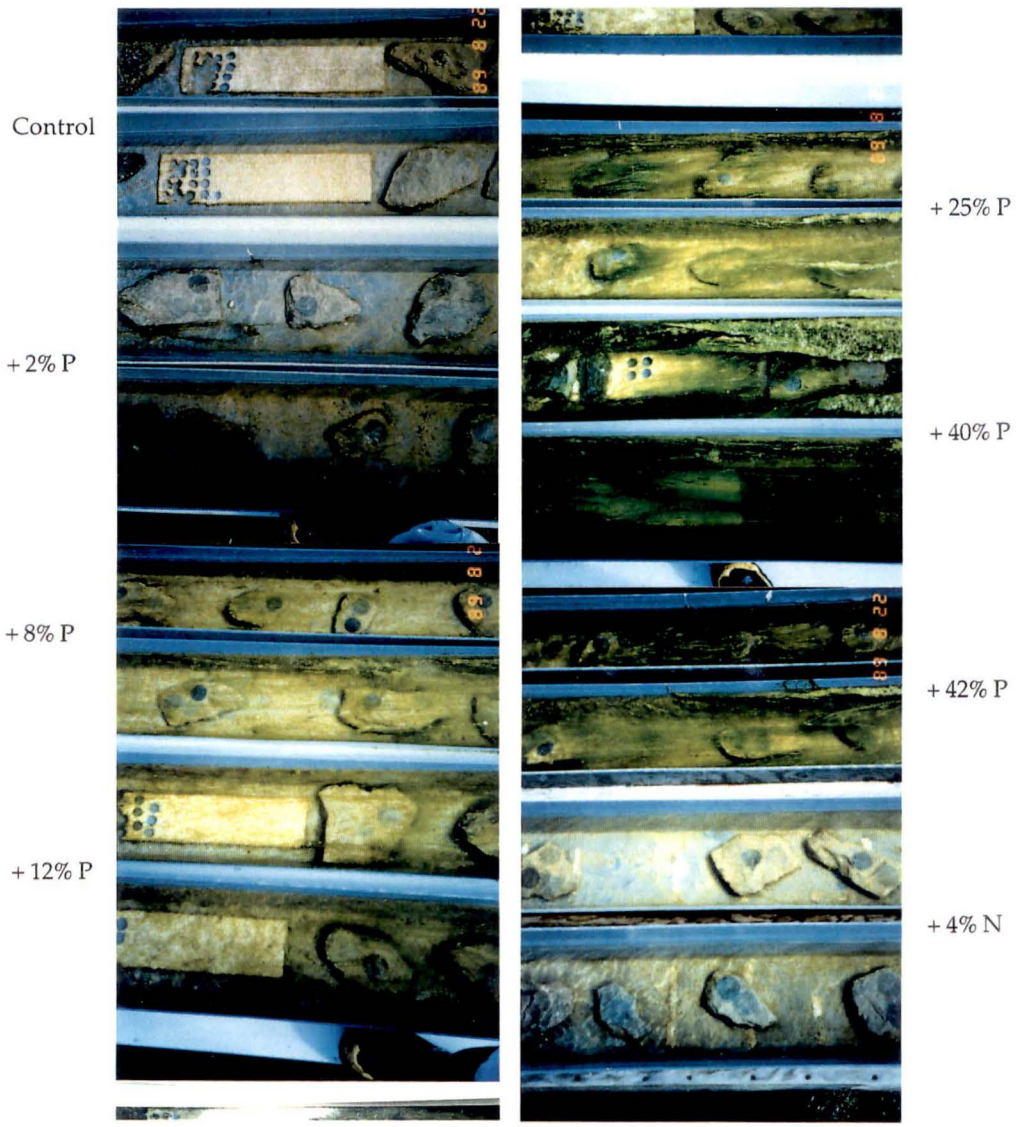
C


Cntl
+310% P
+310% P
+180% P
Cntl
+17% N
+17% N
+100% P
+100% P
+15% N + 50% P
+70% P
+50% P
+180% P
+15% N - 50% P
+70% P
+50% P
Day 4
June 28 1988
Day 21
July 15 1988
Day 32
July 26 1988
Day 46
August 9 1988
Algal Biomass Peak
Cntl = 1788 ug/cm²
+50% P = 6060 ug/cm²

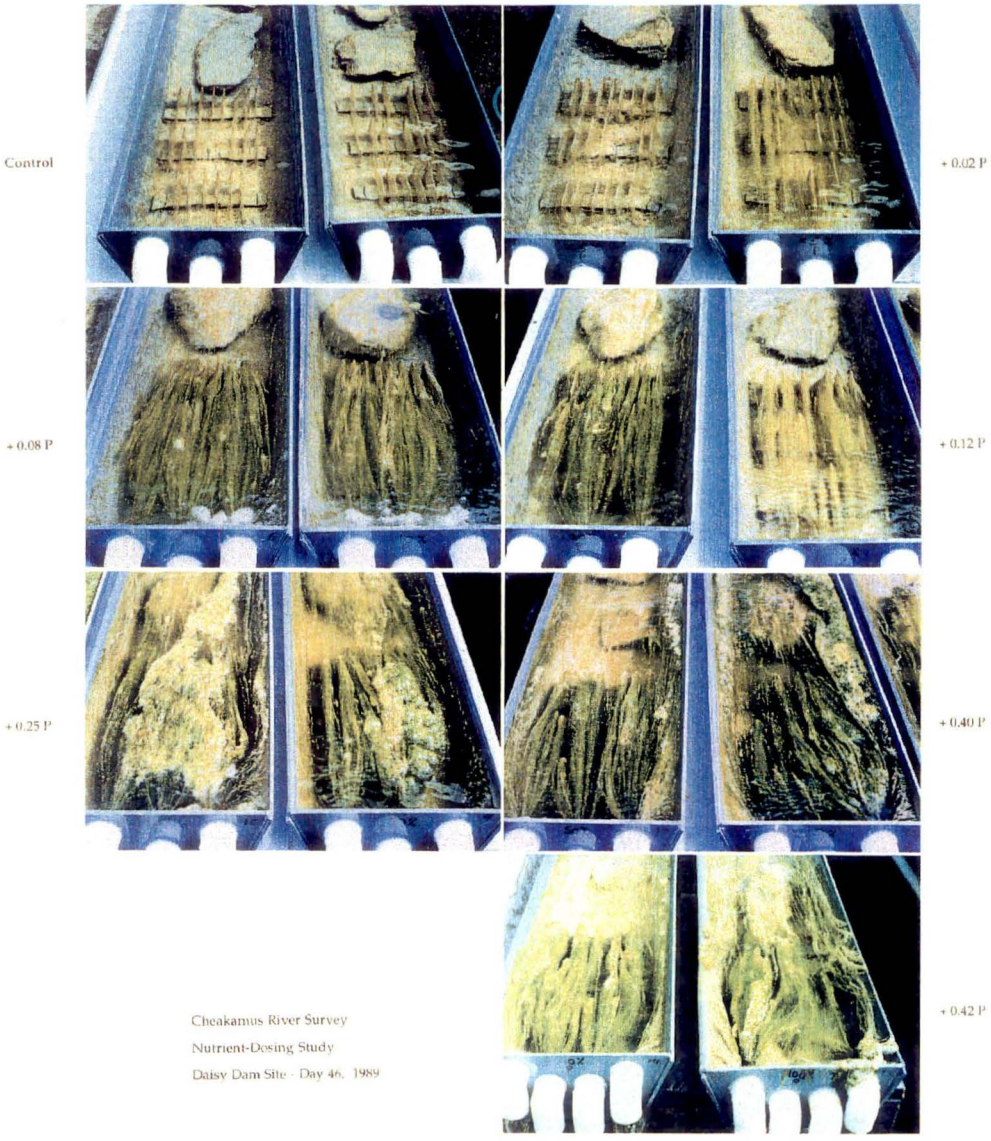
**d**


+ 4% N + 42% P
+ 40% P + 25% P
+ 12% P + 8% P
+ 2% P Cntl
Day 4
July 4 1989
Day 46
August 15 1989
Day 60
August 29 1989

**e**



**f**




Cheakamus River Survey
Nutrient-Dosing Study
Daisy Dam Site - Day 46, 1989

**g**

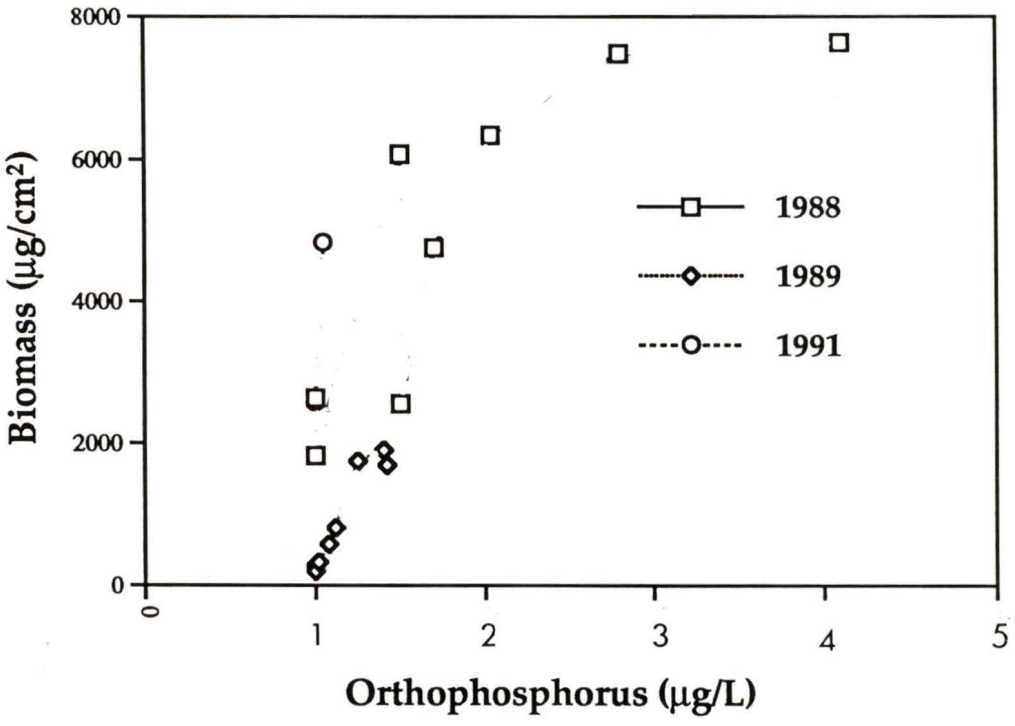

8000
6000
4000
2000
0
Biomass (µg/cm²)
0
1
2
3
4
5
Orthophosphorus (µg/L)
1988
1989
1991

Figure 47. Weekly algal biomass accrual for the months July (Jl), August (A) and September (S), 1987 to 1989, at two sample sites on the Lower Cheakamus River. 15 km = site K; 21 km = site M (Figure 1). In 1987 and 1988 accelerated accrual occurred earlier at site M, whereas in 1989 site K accrual was higher throughout the summer.

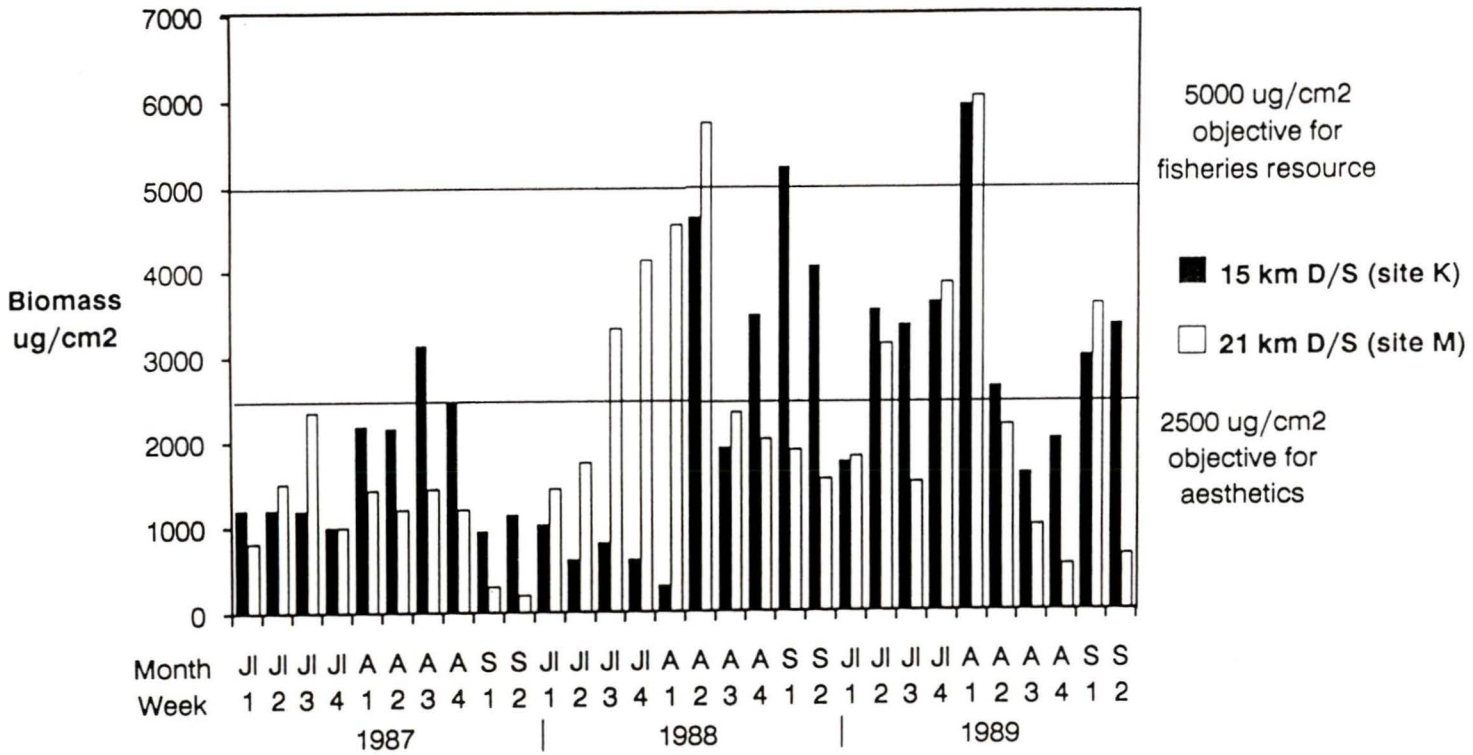
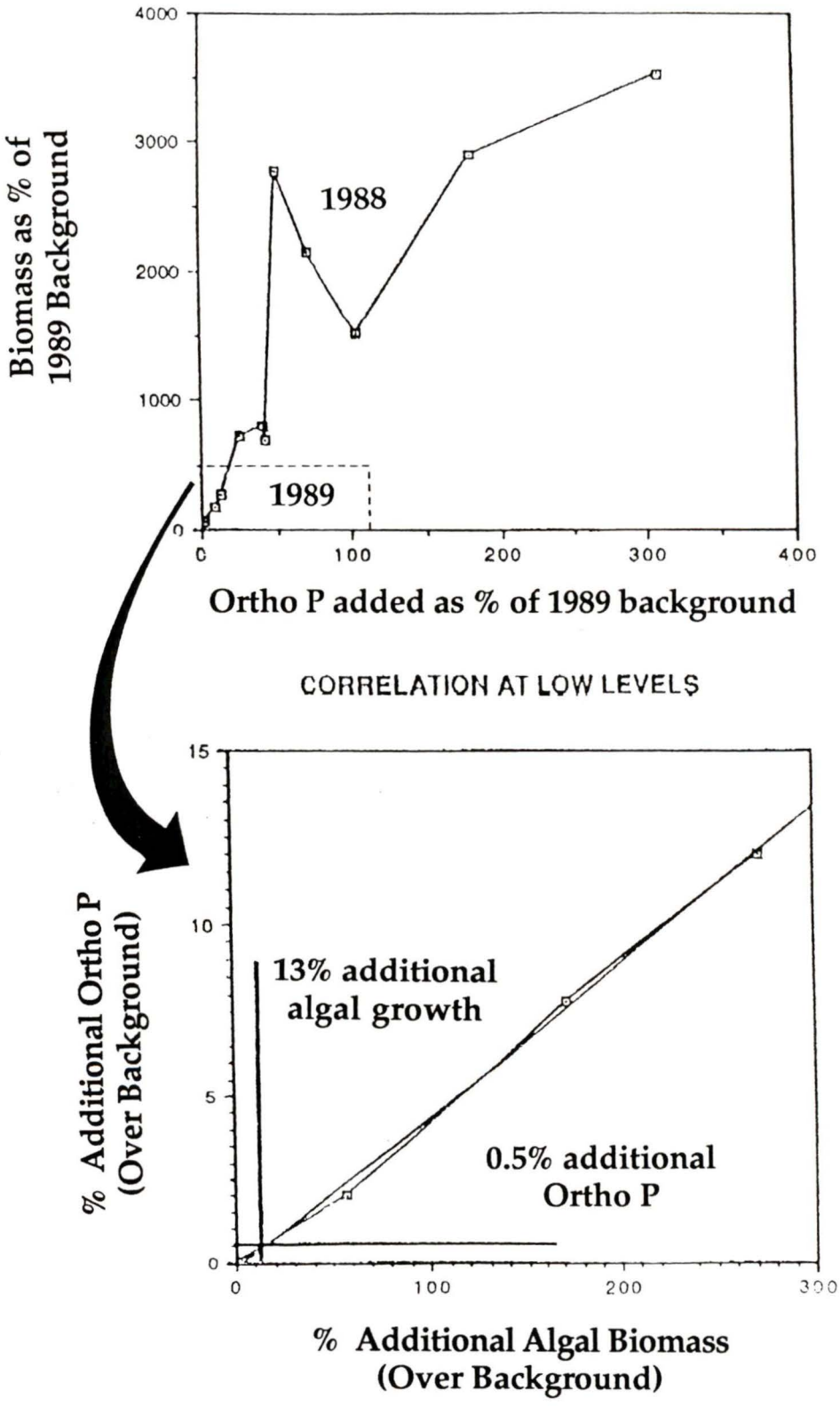

7000
6000
5000
4000
3000
2000
1000
0
Biomass
ug/cm2
Month Jl Jl Jl Jl A A A A S S Jl Jl Jl Jl A A A A S S Jl Jl Jl Jl A A A A S S
Week 1 2 3 4 1 2 3 4 1 2 1 2 3 4 1 2 3 4 1 2 1 2 3 4 1 2 3 4 1 2
1987
1988
1989
5000 ug/cm2
objective for
fisheries resource
15 km D/S (site K)
21 km D/S (site M)
2500 ug/cm2
objective for
aesthetics

Figure 48. Relationship between increased orthophosphate and periphytic biomass, using 1988 and 1989 data at all concentrations (upper) and at very low values (lower).


Biomass as % of 1989 Background
4000
3000
2000
1000
0
1988
1989
0
100
200
300
400
Ortho P added as % of 1989 background
CORRELATION AT LOW LEVELS
% Additional Ortho P (Over Background)
15
10
5
0
13% additional algal growth
0.5% additional Ortho P
0
100
200
300
% Additional Algal Biomass (Over Background)

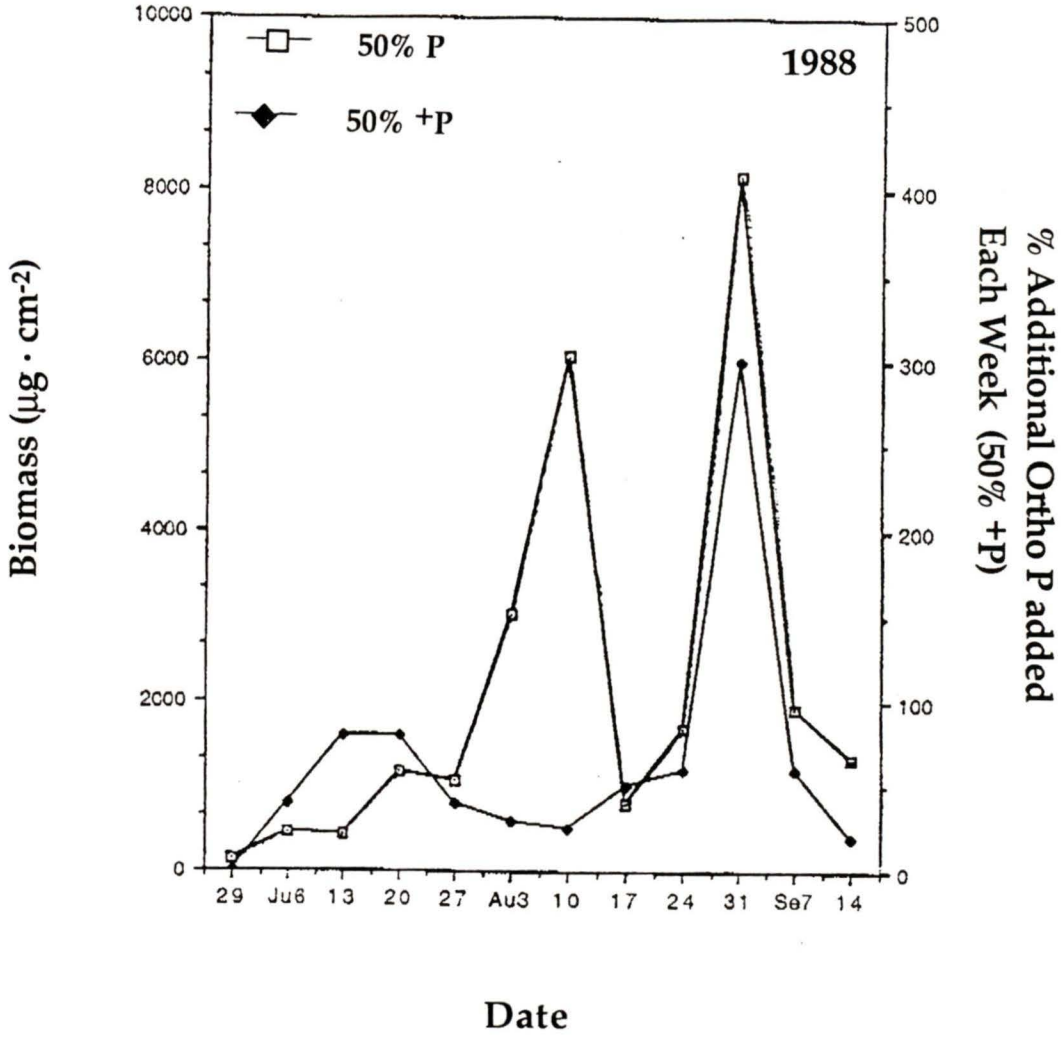
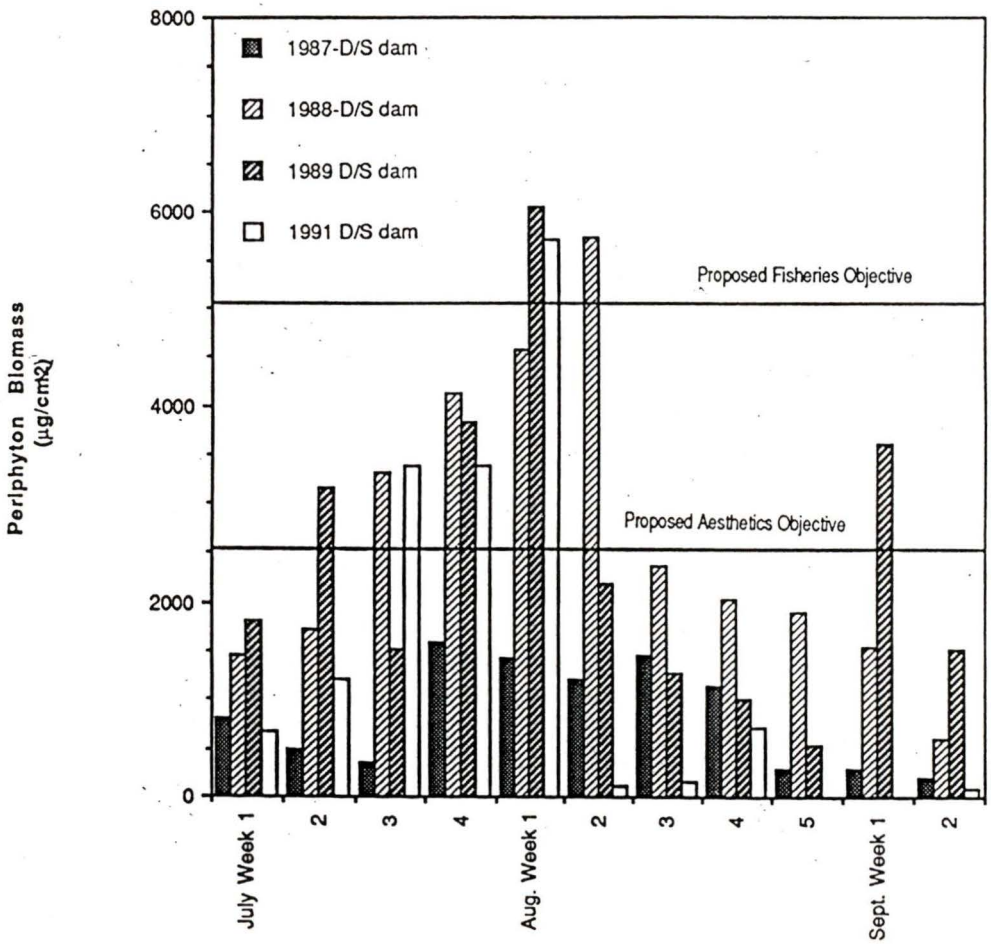



Figure 49. Biomass accrual under a proposed 0.5 µg/L ortho-P enrichment protocol; the actual ortho-P concentrations are plotted. The sudden spike beginning August 24th occurred when a damaged drip-feed discharged phosphorus at a significantly higher rate.

Figure 50. Periphyton biomass values (,from natural river rock substrata) sampled in the Lower Cheakamus River, downstream of the dam, between 1987 and 1991.


8000
6000
4000
2000
0
1987-D/S dam
1988-D/S dam
1989 D/S dam
1991 D/S dam
Proposed Fisheries Objective
Proposed Aesthetics Objective
Periphyton Biomass
(µg/cm2)
July Week 1
2
3
4
Aug. Week 1
2
3
4
5
Sept. Week 1
2

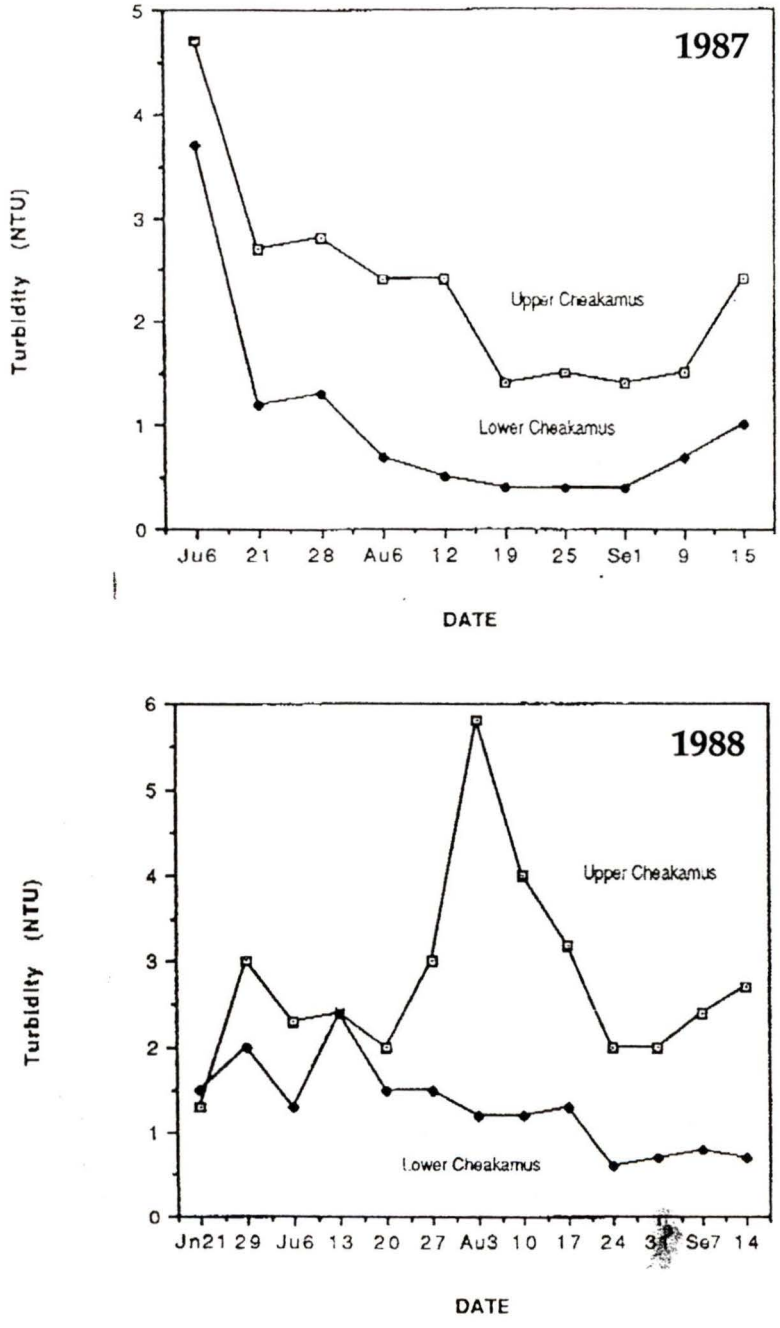



Figure 51. Turbidity values in the upper and lower Cheakamus River during 1987 (upper) and 1988 (lower).

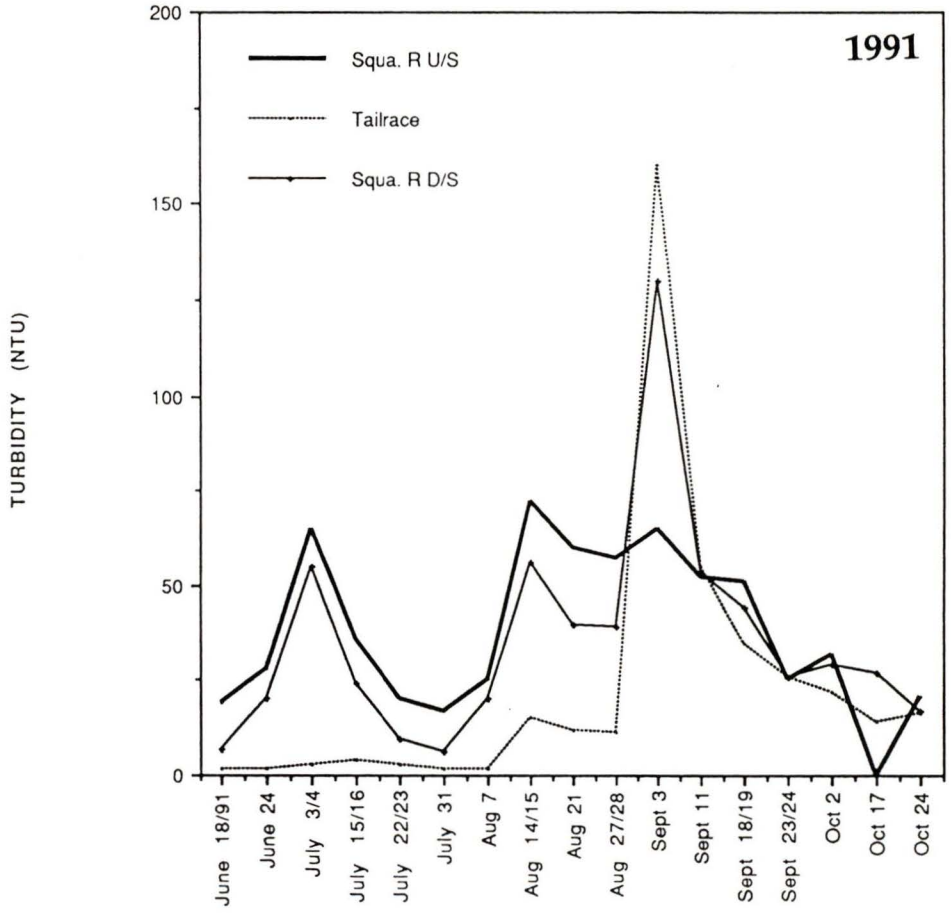



Figure 52. Turbidity values obtained from the Squamish River during 1991. Note the low values during the summer in the tailrace.

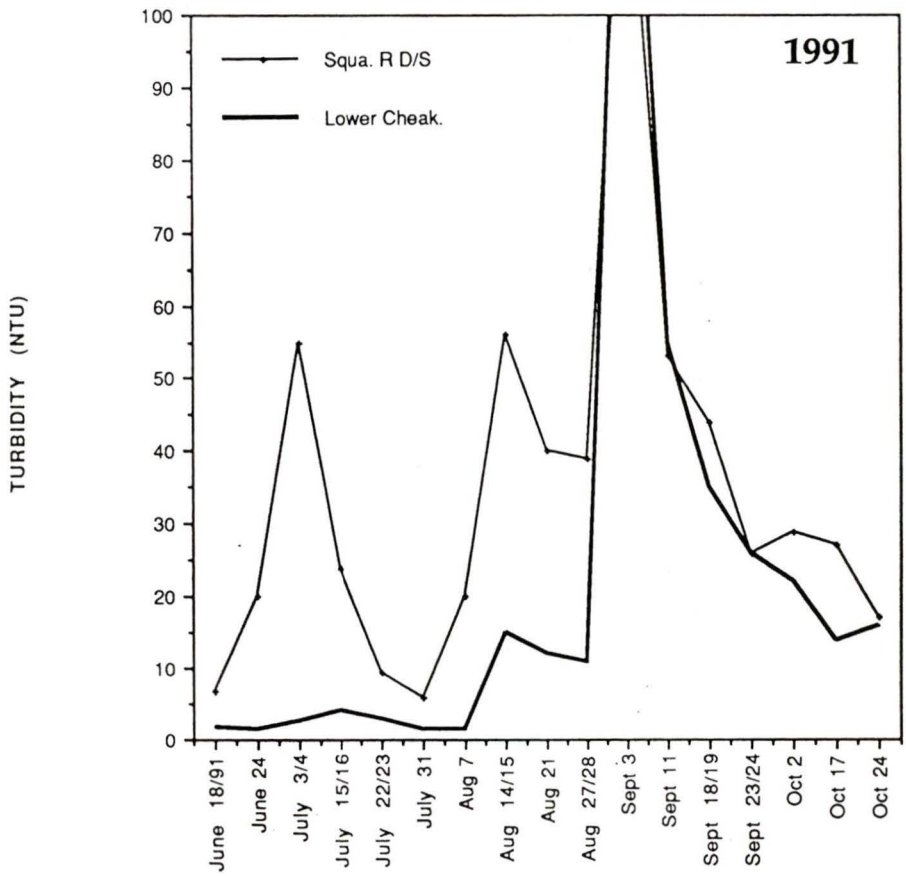



Figure 53. Turbidity observed in the Squamish and Cheakamus Rivers during the summer months of 1991. Note the extreme values during the two hundred year storm event of August.

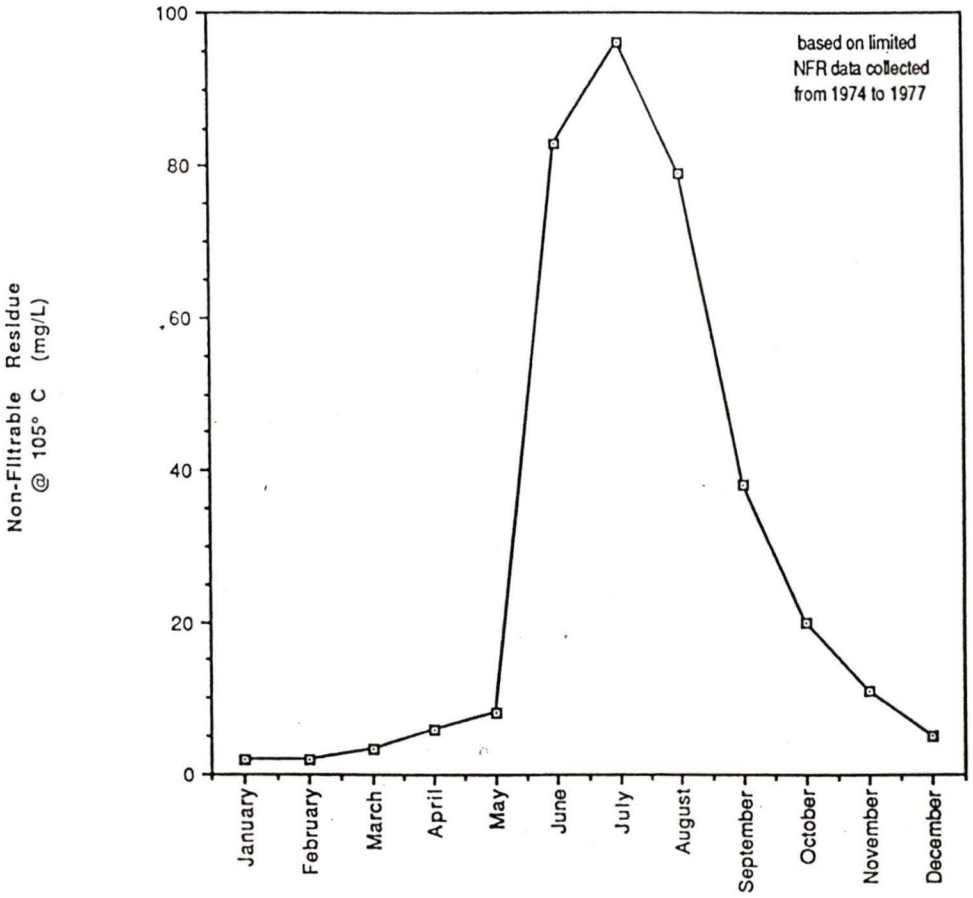



Figure 54. Annual solids loadings in the Squamish River based on limited Non-Filterable Residue data collected from 1974 to 1977.

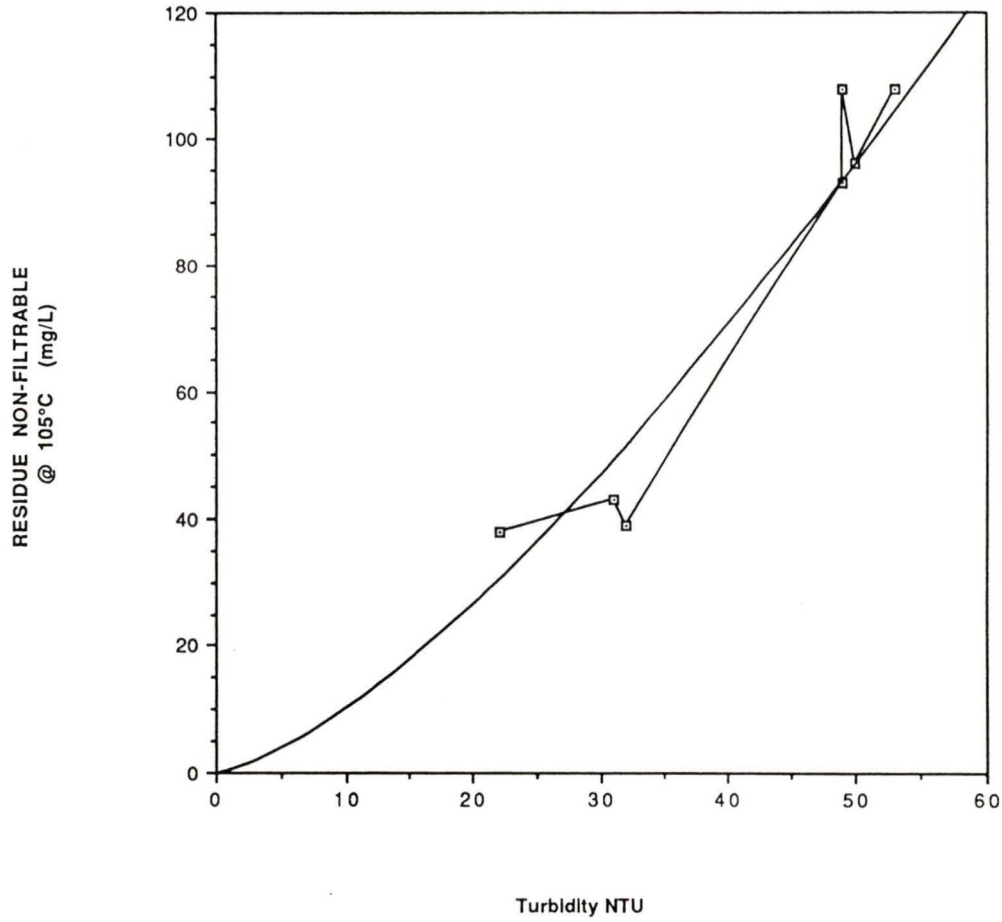



Figure 55. Relationship between turbidity and suspended solids.

Figure 56. Comparison of i) treated sewage-effluent discharges, from the RMOW STP, into the Cheakamus River during the summer months of July and August, 1983 to 1991 (left-hand axis) with ii) allowable effluent flows (at different ortho-P concentrations), which would effect a maximum permissible biomass (2500 $\mu g\ cm^{-2}$) in the Lower Cheakamus River.

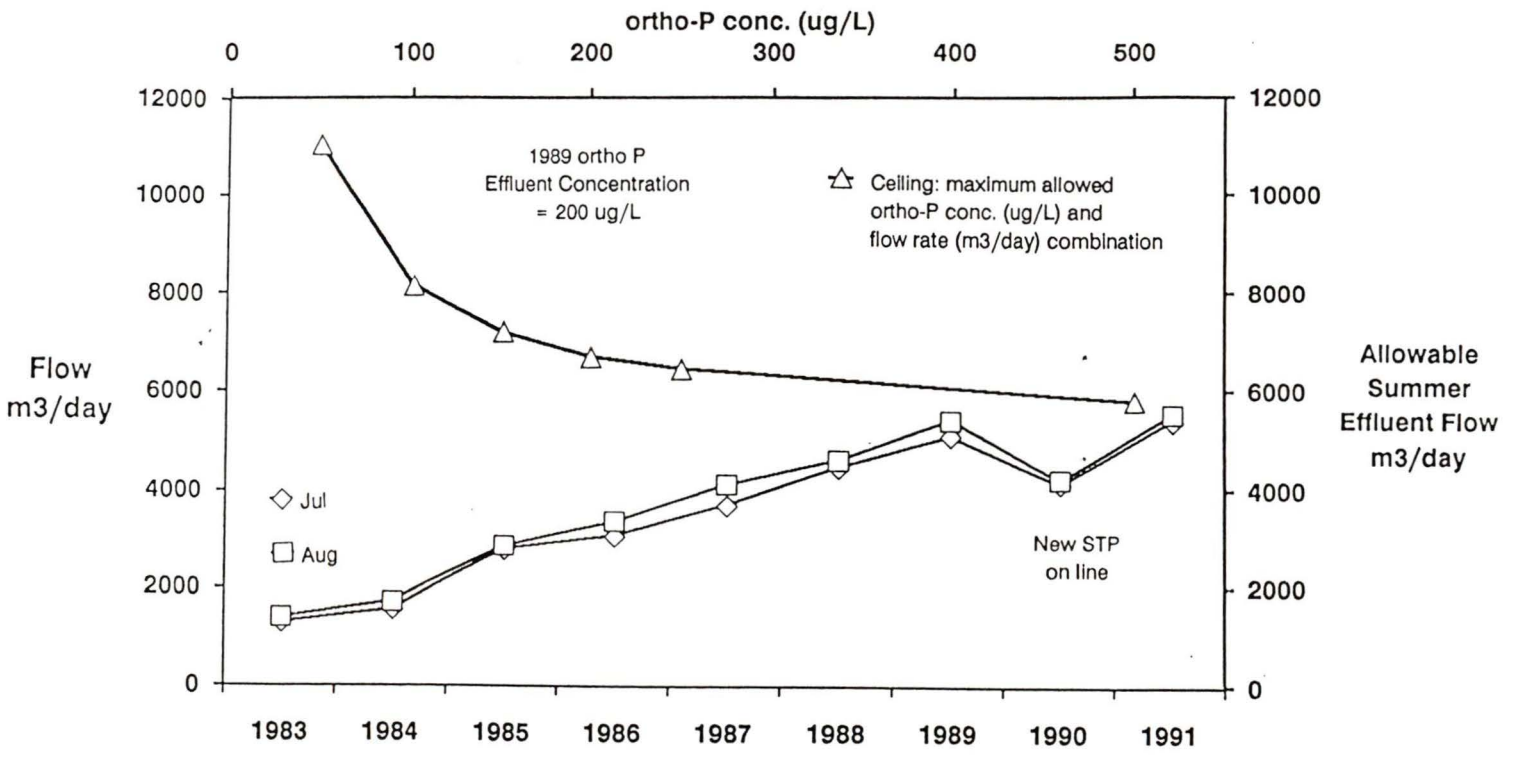

ortho-P conc. (ug/L)
0
100
200
300
400
500
12000
10000
8000
6000
4000
2000
0
Flow
m3/day
1989 ortho P
Effluent Concentration
= 200 ug/L
Ceiling: maximum allowed
ortho-P conc. (ug/L) and
flow rate (m3/day) combination
Jul
Aug
New STP
on line
Allowable
Summer
Effluent Flow
m3/day
1983
1984
1985
1986
1987
1988
1989
1990
1991

Figure 57. Maximum permissible effluent discharges from the Squamish Sewage Treatment plant calculated on the basis of ortho-P concentration.

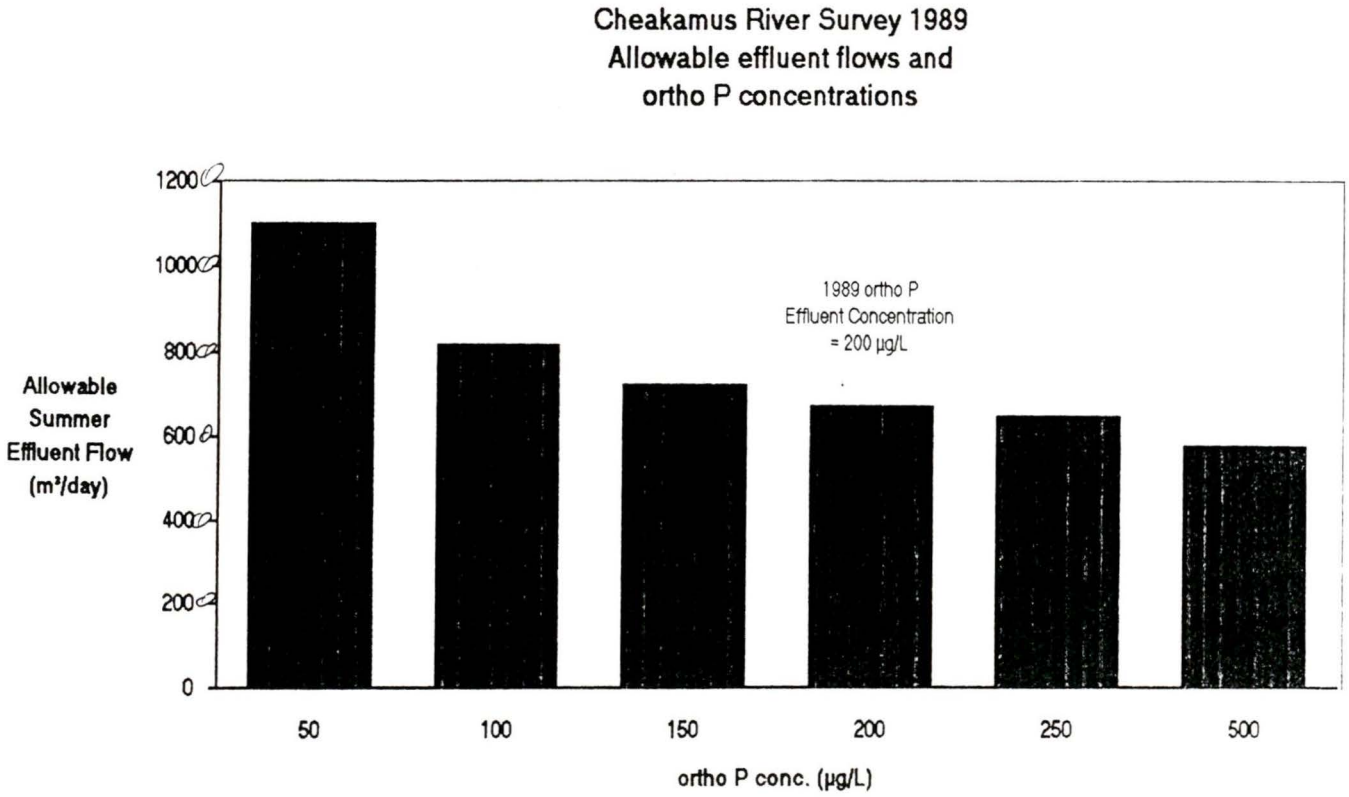
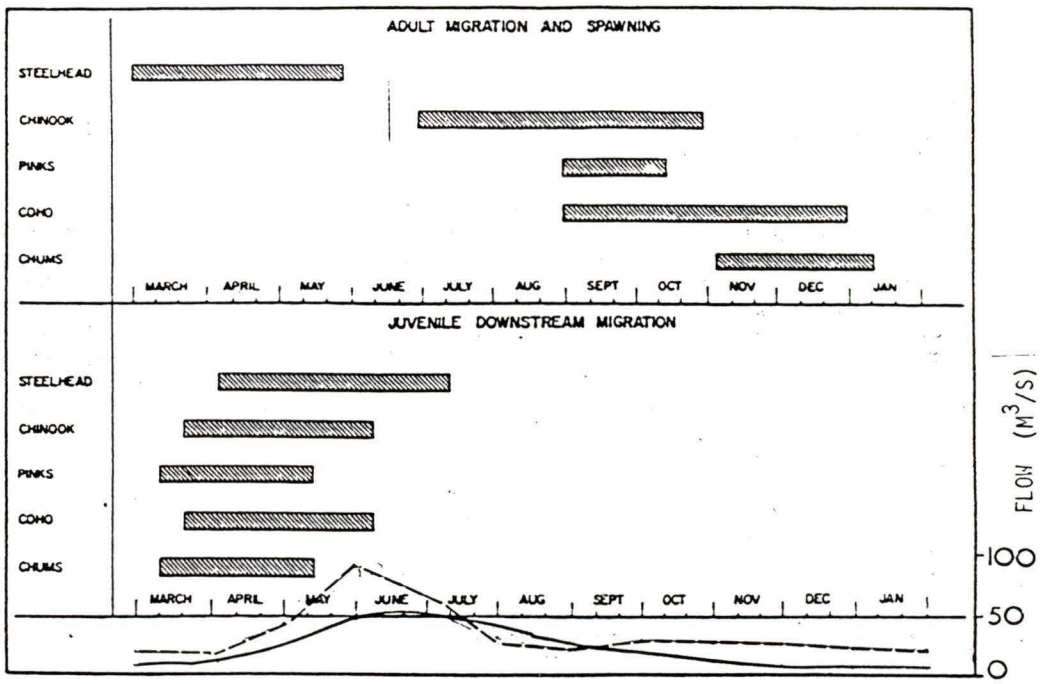

Cheakamus River Survey 1989
Allowable effluent flows and
ortho P concentrations
Allowable
Summer
Effluent Flow
(m³/day)
1200
1000
800
600
400
200
0
1989 ortho P
Effluent Concentration
= 200 µg/L
50
100
150
200
250
500
ortho P conc. (µg/L)

Figure 58. Comparison of periphyton biomass in the Squamish, and Upper and Lower Cheakamus Rivers (lower graph) with long term flow averages (upper figure, right hand axis; taken from Figure 27) in the Upper (solid line) and Lower (dashed line) Cheakamus River and with fish migratory behaviour.


ADULT MIGRATION AND SPAWNING
STEELHEAD
CHINOOK
PINKS
COHO
CHUMS
MARCH
APRIL
MAY
JUNE
JULY
AUG
SEPT
OCT
NOV
DEC
JAN
JUVENILE DOWNSTREAM MIGRATION
STEELHEAD
CHINOOK
PINKS
COHO
CHUMS
MARCH
APRIL
MAY
JUNE
JULY
AUG
SEPT
OCT
NOV
DEC
JAN
FLOW (M3/S)
100
50
0

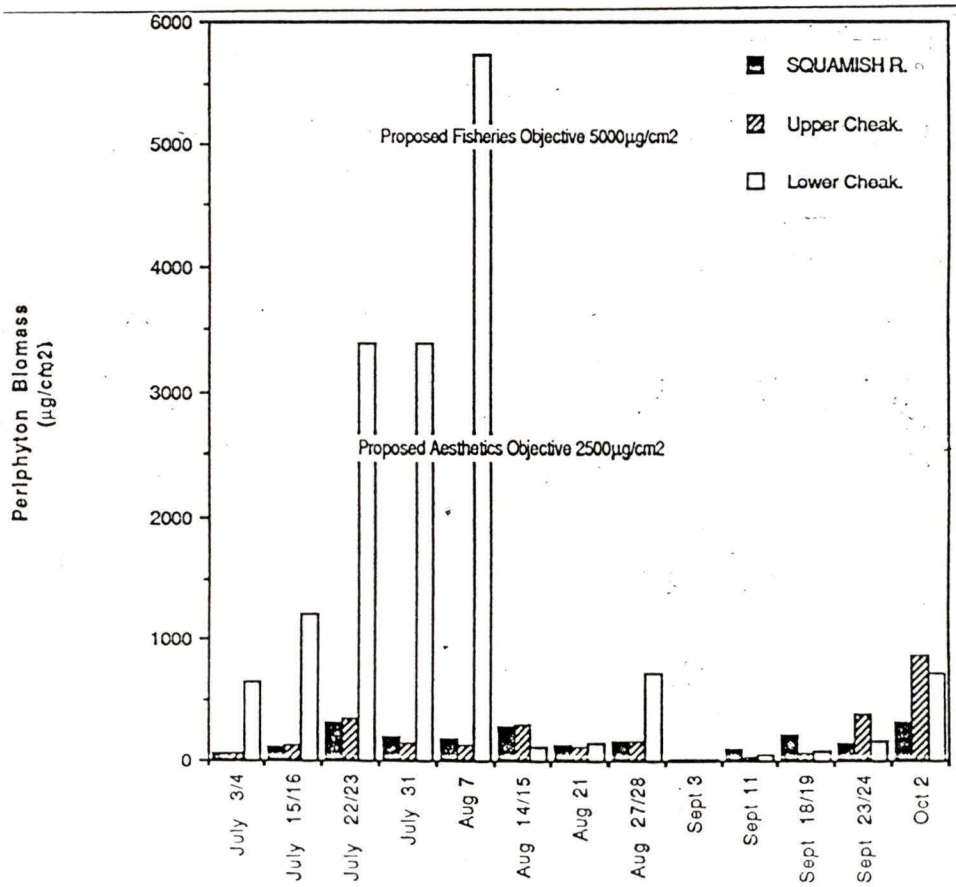

Periphyton Biomass (µg/cm2)
6000
5000
4000
3000
2000
1000
0
SQUAMISH R.
Upper Cheak.
Lower Cheak.
Proposed Fisheries Objective 5000µg/cm2
Proposed Aesthetics Objective 2500µg/cm2
July 3/4
July 15/16
July 22/23
July 31
Aug 7
Aug 14/15
Aug 21
Aug 27/28
Sept 3
Sept 11
Sept 18/19
Sept 23/24
Oct 2

Figure 59. Periphyton biomass obtained in the nutrient-dosing, semi-natural stream troughs, at both the Squamish and Daisy Dam sites. Background concentrations as in Figure 46a/b.

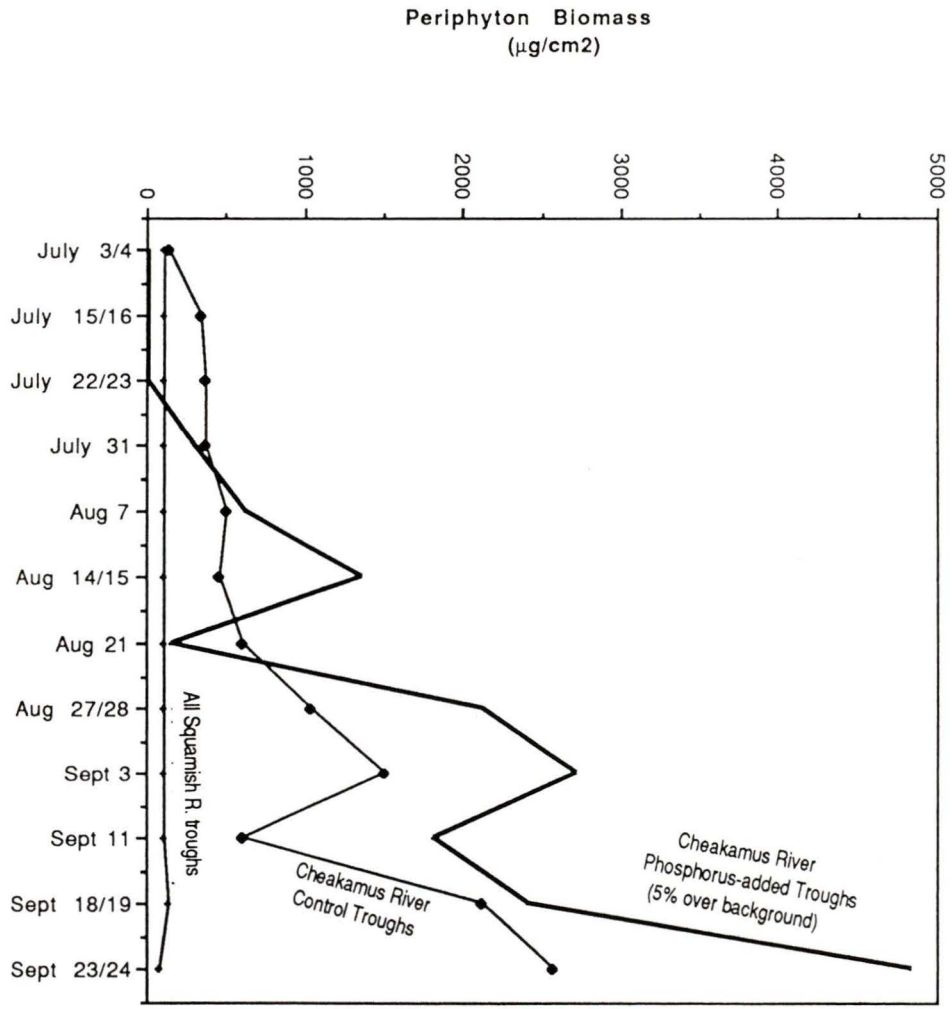
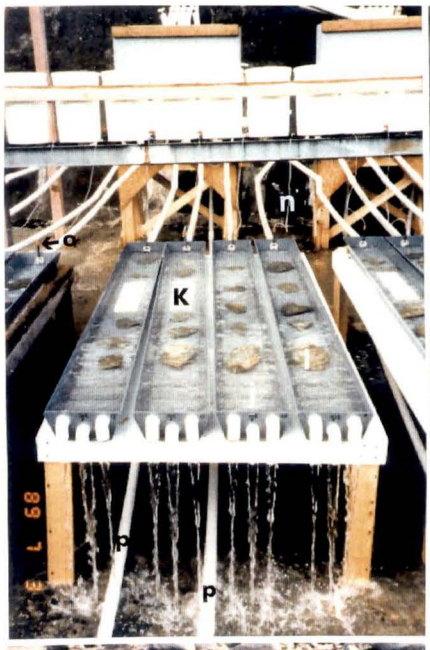
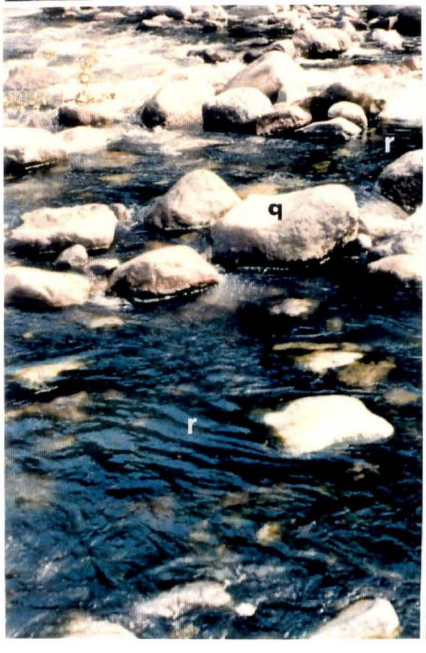

Periphyton Biomass
(µg/cm2)
0
1000
2000
3000
4000
5000
July 3/4
July 15/16
July 22/23
July 31
Aug 7
Aug 14/15
Aug 21
Aug 27/28
Sept 3
Sept 11
Sept 18/19
Sept 23/24
All Squamish R. troughs
Cheakamus River
Control Troughs
Cheakamus River
Phosphorus-added Troughs
(5% over background)

Figure 60. The Cheakamus River surrogate stream complex, located adjacent to, and below, the Daisy Dam (a; UR). The prototype stream system, assembled in 1988, used at this site is shown in the upper right and consists of a multiple siphon array (b), header box (c), distribution manifolds (d), wooden stream troughs (e) and PVC replacement troughs (f), and discharge plumbing (g); the nutrient dosing system was elevated above the stream troughs (h). The replacement system (MR), erected in 1989, required multiple siphon lines (b) and dual header boxes (c); all streams were constructed of PVC. The lower right photograph depicts additional system components including distribution manifolds (i), substrata both artificial (j-glass slides; k-styrofoam) and natural (l-river rock), and nutrient reservoir shielding (m). The upper left photograph provides a close-up of the artificial and natural substrata; also visible are the nutrient dosing delivery lines (n) and injector ports (o), together with the header box overflow discharge lines (p). Subsequent drainage was collected in troughs (visible in LR) which were discharged via PVC pipes to concrete aprons edge. The lower left photograph shows a typical side-channel in the Lower Cheakamus River, showing enhanced irradiance (q), evidenced by the sun drenched river rock following riparian canopy removal, and heavy algal biomass accrual (r) in the shallows; photo taken at sample site M (Figure 30).


n
o
K
p
p


a
c
h
d
e
g


b
c
c


r
q
r


j
l
k
m

Figure 61. Photograph in upper right shows the Squamish River surrogate stream compound, with the bridge (a; along which the electric power was strung) connecting the Island with the mainland, the river water supply (b; large diameter white PVC pipe), feeding the dual header boxes (c), supplying the distribution manifolds (d), which fed the stream troughs (e), which drained into a paired common discharge line (f). Also shown are the nutrient dosing reservoirs (g). The periphyton sampling technique (LR; modified after Ertl, 1960?) showing river rock which has been scrapped (h), together with the neophrem sealed circular disk (i), scraper (j), suction bulb (k) and labelled sample container (l). The upper left photograph shows the header boxes receiving the 2700 liter/minute pumped flow; note the "T" joint (m) connecting the discharge elbow (n) which allows the elbow to travel in a vertical arc which is used to regulate the discharge flow into the header box, thereby obviating the need for a large gate or ball valve. The middle left photograph shows the pump (o), with its protective housing (p), suspended from the bridge. The pump is retrieved from the river through the use of a chain hoist (q). The primary discharge pipe was constructed from flexible PVC (r), joined the solid PVC pipe (s) via a coupling (t); a small platform (u) provided an operating surface from which to service the pump. Electrical power for the pump was obtained from the 600 volt transmission lines (v), which was then transferred to the hydro pole in the upper right hand corner (w). A ground cable was strung under the bridge to a position immediately in front of the two individuals on the working platform. An in-line, dual chamber, sedimentation chamber (lower left) was installed immediately upstream of the INFLEX site to effect a reduction in the sediment load being conveyed to the stream troughs. Two bottom feed lines (x) conveyed the sediment back to the river. Note the sand below the clarifier and that trapped in the corrugation of the sign (below the word "Clarifier"), 3 meters above normal; river width is 100 meters.

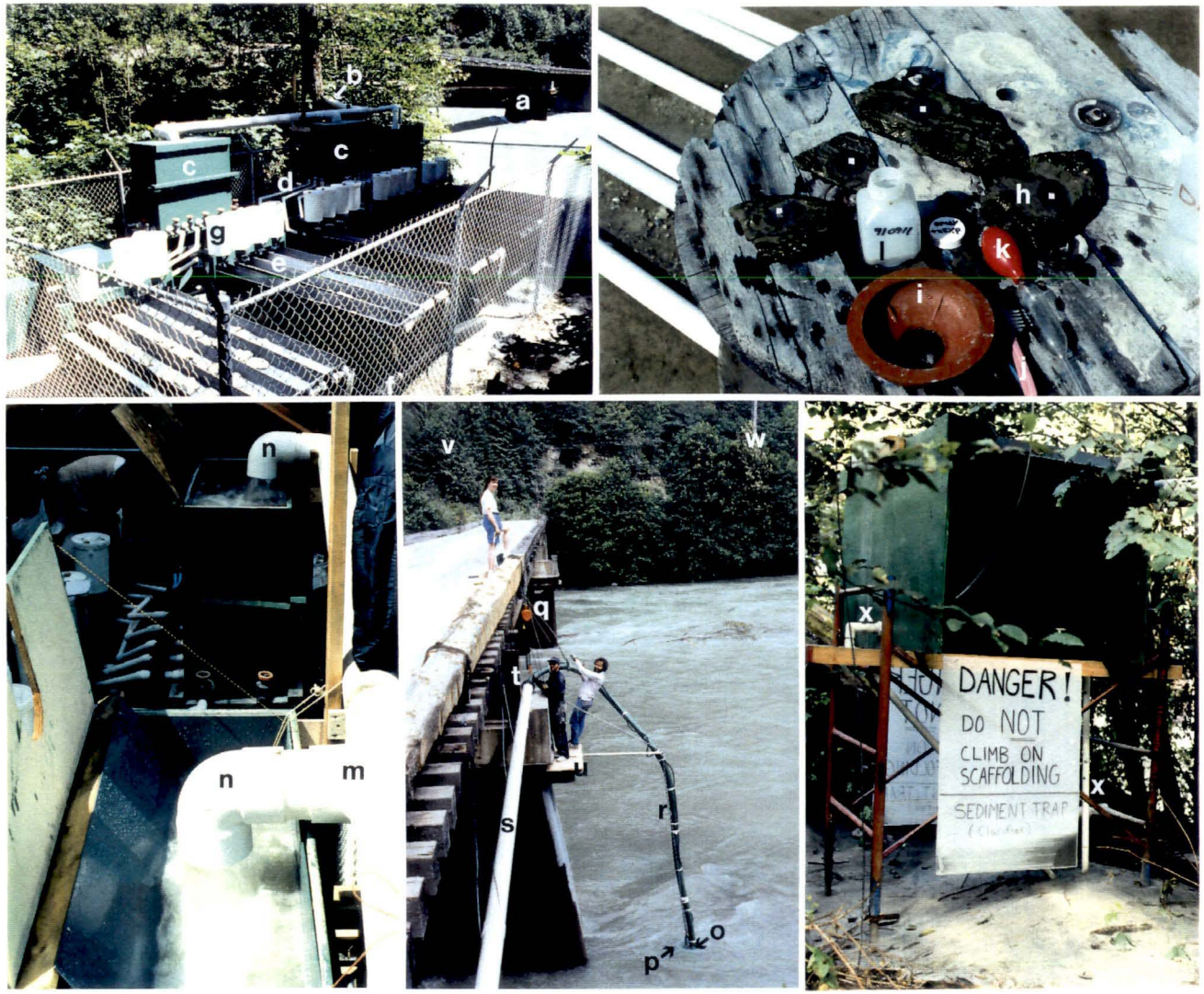

a
b
c
c
d
e
g
h
i
k
m
n
n
n
q
r
s
t
u
v
w
x
x
DANGER!
DO NOT
CLIMB ON
SCAFFOLDING
SEDIMENT TRAP

Figure 62. The two hundred year storm event of August 31, 1991 resulted in the extraordinary summer discharge from Daisy Reservoir (LL), which resulted in a significant increase in the stage of the Lower Cheakamus (approx. 40,000 cfs); the river flow had returned to its normal flow by September 11th (UL; approx. 100 cfs); the estimated difference in river stage was 7 meters. The photograph in the upper right shows a typical shallow side-channel in the Lower Cheakamus, at station M; note the narrow central channel (a) and extensive side channel development (b). Samples of natural river rock were removed from a section of the side channel, exhibiting heavy growth, and ten replicate circles of periphyton scrapped (white squares; MR) to remove as much of the community as possible. Note the heavy sediment (c; MR), smothering the periphyton community, which occurred in the upper Cheakamus following subsidence of the August 31 storm event.

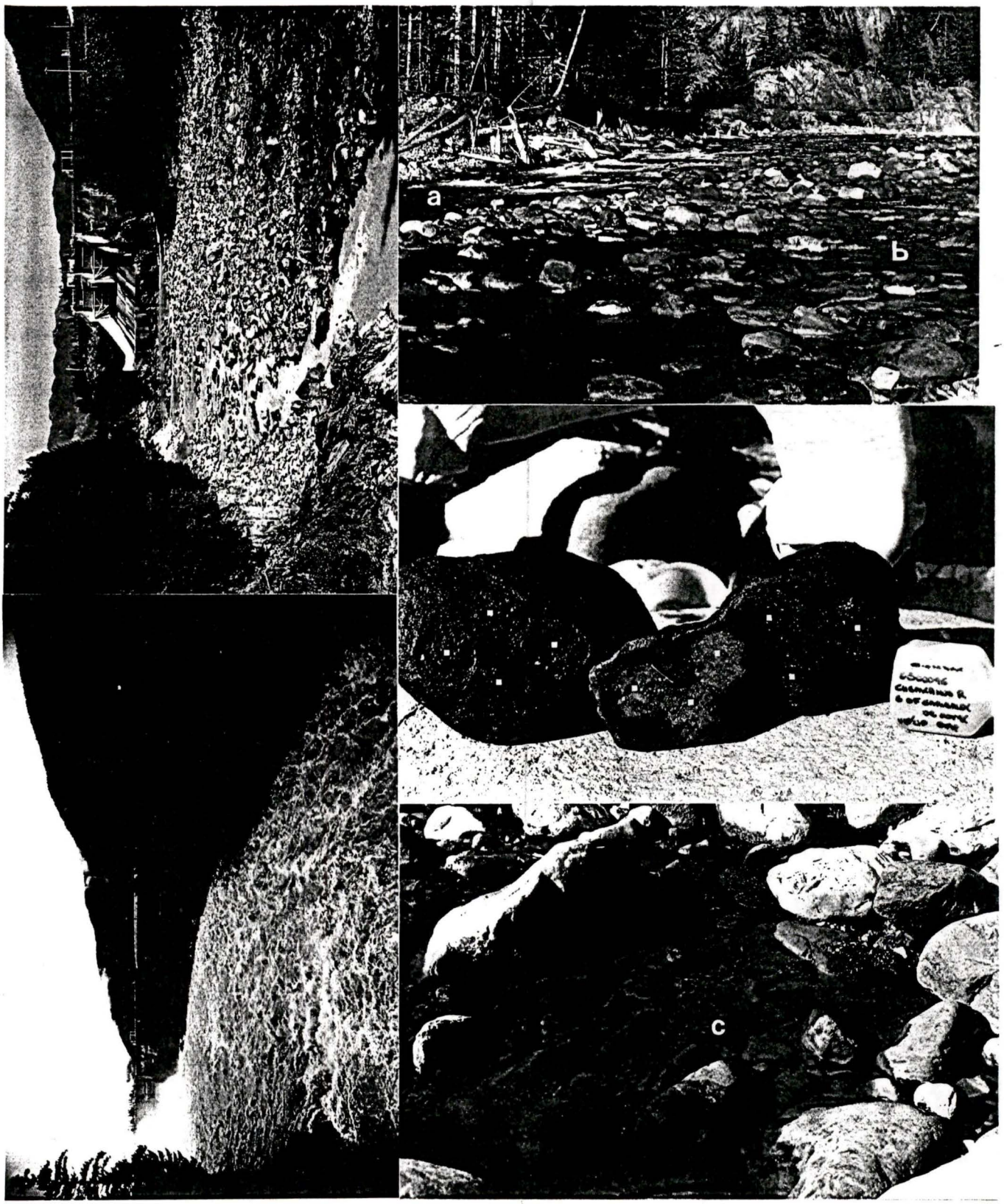

a
b
c

Figure 63. Aftermath of a major storm event at the Squamish River INFLEX site. Upper right photograph shows the degree to which a typical prolonged, heavy precipitation event can raise the river's stage. This photograph was taken on August 9, 1991, whilst that of the lower right was taken October 2, 1991. The river level has dropped approximately 3 meters. The photograph in the upper left was taken July 13, 1991 and shows the compound in its normal operational mode. The photograph lower left shows the extent to which the swollen river mobilizes the fine sandy sediment; note the depth to which the discharge pies have been buried, relative to their previous degree of exposure. Note also the debris covering the chain link fence (a) and the massive log deposited against the fence (b); the single pipe (c) is discharging sediment from the base of the clarifier. In the lower right photograph the electrical conduit, along side the bridge (d) is visible, as is the float (e) within which was suspended the pump.


d
e
a
b
c

Figure 64. Low water levels in the Daisy Reservoir (UL) showing the siphon float (a) and subsurface intake lines (b; 7 meters in length). Extensive benthic sediments are exposed (UR)(e) during this drawdown period; the central channel of the reservoir (c) leads to the spillways, whilst it is through the side channel (d) that water is diverted to the tunnel under Cloud-Burst Mountain, prior to being discharged through the hydro generating station adjacent to the Squamish River. The photograph in the lower left shows the shallow side channel development downstream of the Squamish River INFLEX site, which was typically covered in a fine sediment (f). The periphyton community at this site was observed to have been a filamentous Chlorophyte (g; UR) growing on the exposed muds on the banks of the river channel (h; LL); virtually no visible periphyton was observed on the cobble substrata. All photographs taken October 2, 1991.


a
b
c
d
e
f
g
g
h

Figure 65. These photographs demonstrate the variability of periphytic growth which was observed on August 21, 1991, prior to the massive flood of August 31, 1991. Upper left - upstream of the STP; this control site evidences the minimal growth typical of this site. Upper right - the moderate periphytic growth normally found at site G (2.5 km downstream of the STP); note the golden brown colour of the community and absence of Chlorophytes. Lower left - downstream of the Daisy Dam, the periphyton community is significantly heavier and dominated by Chlorophytes. Lower right - taken the same day as the above photographs, the Squamish River INFLEX system, throughout this study, was routinely inundated with sediment-laden river water which rapidly smothered the substrata, precluding all but the smallest periphyton development.

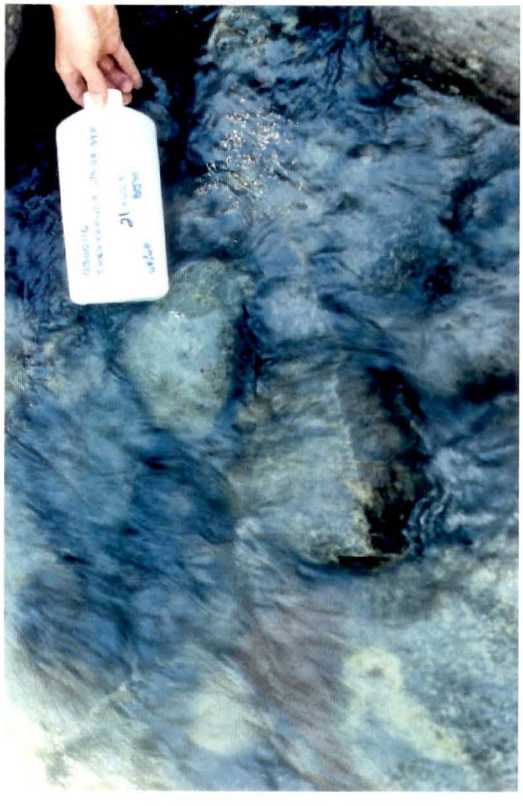


Figure 66. Periphyton biomass accrual within the INFLEX troughs located below Daisy Dam during the summer of 1991. The chronological order of the photographs is A:July 3; B:August 10; C:August 21; and, D:August 31. All troughs were allowed to colonize under equivalent environmental conditions, prior to initiation of treatment; a diatomaceous community (A-a) had developed in all troughs by July 3rd. Upon initiation of the experimental treatment, the lateral troughs received an ortho-phosphorus addition of 0.05 µg/L (B-b); the two central control troughs received only lake water. Note the significantly advanced periphyton community development under nutrient enrichment (B-c), whilst the control troughs community evidence a markedly lower periphyton biomass accrual, together with a fine sediment covering the substrata (d). The community has begun to exhibit filamentous canopy sloughing (C-e) by August 21; note also the Chlorophyte development in the control troughs. Note the almost complete loss of the Chlorophyte canopy in all troughs when exposed to extreme turbidity (D-f), following the two hundred year storm event. The photograph in the upper right portrays the extent of the damage within the Squamish INFLEX compound (September 2, 1919)(compare this with the lower left photograph in Figure 12). Note the scoured excavation, and debris, along both sides of the riverside fence which is not present along the left hand side of the fence indicating the river flow was from right to left. The large green box at the upper right hand edge was a header box which was completely destroyed by the large log in the foreground; the chain link fence acted as a protective barrier minimizing damage inside the compound.


C
e
A
a
B
b
b
c
d
D
f

# Chapter IV. Additional Applications of Surrogate Stream Mesocosms

## Preamble

To further illustrate the utility of the surrogate stream mesocosms to assess the usefulness of periphyton as a bioassay community, experiments conducted at two additional sites is reported below. The first was the use of surrogate streams to determine the effect of logging activity on natural stream water quality and secondly use of the streams in a recirculating mode to effect nutrient stripping from intensive fish culture tanks, described below under A and B.

## A - Response of Periphyton Grown in Surrogate Streams in Greater Victoria Water District Watersheds to Moniter Logging & Soil Erosion

### Introduction

The conservative logging practices of the G.V.W.D. were believed by the staff of their forestry division to minimize adverse effects on the water quality of their main supply reservoirs (Sooke and others). However, no independent scientific investigations have been conducted to verify this. Recent (1988-89) water quality problems in the southern basin of Sooke Reservoir are thought to have resulted from inadvertent nutrient enrichment originating in a water diversion from the adjacent Deception Reservoir. This resulted in phytoplankton blooms, which imparted undesirable tastes and odours to the Sooke Reservoir water. Prevention of future taste and odour problems requires an understanding of watershed nutrient cycles. Since a major source of lake nutrients originate in the forest, understanding lake nutrient cycles (loading) must begin by measuring nutrients moving through watershed forests into streams, and then into lakes. Stream nutrient enrichment typically results from soil erosion following such land-use activities as logging or road construction.

The major inflow into Sooke Reservoir is via Rithet Creek (Figure 67). The watershed of this creek has been subjected to considerable logging activity. In

order to determine whether the removal of the forest was resulting in nutrient loading of the creek, and thus the reservoir, a study was undertaken to measure nutrient movement from the hillside and down the creek into the reservoir. The study sought to further our understanding on how physical, chemical and biological processes affect nutrient cycles in forest and aquatic environments. This study focused on forested and logged areas, small feeder streams and the larger streams flowing into Sooke Reservoir. Since G.V.W.D. aquatic ecosystems are known to be nutrient deficient, excessive nutrient enrichment of streams, and thus reservoirs, may result in undesirable changes in water quality of short or long term significance.

The study objectives were to:

i) quantify watershed nutrient loads entering streams and partition loading by source - from forested areas and soil erosion from logged areas.

ii) measure qualitative nutrient differences between forested and disturbed lands which enter streams and then Sooke Reservoir.

iii) determine environmental factors normally regulating nutrient loading within the watershed.

iv) correlate stream nutrient loading, algal growth and water quality.

v) determine if the knowledge obtained from the Rithet watershed is applicable to other G.V.W.D. watersheds.

**Methods**

This investigation used standard physical and chemical parameters to measure water quality at discrete time intervals. Surrogate streams were placed adjacent to the natural streams from which water was diverted into the mesocosms (Figure 68). Periphyton was developed within the surrogate streams in response different nutrient loadings producing a measurable nutrient-loading-biomass accrual relationship. The presence of periphytic species known to be sensitive to water quality changes provided an important

list of local water quality indicator species. These indicator species are essential in reflecting subtle long term, predictable changes in water quality.

The study consisted of the following elements:

1. identify study site - two adjacent hillsides, a control (forested) and an experimental (logged, or soon to be logged cut-block).

2. establish surrogate streams above and below a cut-block to permit assessment of nutrient levels entering the stream as a result of land disturbance.

3. establish a control set of streams along a forested stream.

4. to one set of streams at each site nutrients were added at selected concentrations to correlate biofilm species changes with nutrient enrichment. Nutrient enrichment should increase biofilm growth by a measurable amount.

5. stream nutrient loading was measured using both physical and chemical analysis, and by assessing changes in the biofilm community grown in the surrogate streams.

6. physical, chemical and periphyton parameters were examined at selected sites along Rithet Creek.

7. a nutrient dosing experiment was conducted within Rithet Creek to measure periphyton sensitivity to nutrient loading and to determine what the limiting nutrient is.

**Study Implications**

The information generated by this study furthered our understanding of i) how nutrient loading of streams occurs, ii) what affect nutrients have on stream water quality, iii) how stream nutrients affect the nutrient loading of Sooke Lake, and, iv) the potential for undesirable taste and odour algal blooms.

This information has also been helpful in modelling nutrient loading in other G.V.W.D. watershed streams and lakes. The data was used to determine:

1. the optimal size and location of cut-blocks on hillsides; this will minimize soil erosion, and stream and lake nutrient loading.

2. the mass movements of nutrients from the forest into lakes, necessary to model where and how nutrient loading of Sooke Reservoir is occurring. Undesirable algal blooms may be generated in nutrient sensitive Sooke Reservoir if certain critical nutrient thresholds are exceeded.

3. the quantity and quality of nutrients entering the reservoirs which is important for modelling plankton (algal) species and biomass, which largely determine water quality characteristics.

4. strategies for raising dams and lake levels which requires knowledge of long-term changes in the plankton community. Since raising Sooke Lake water levels may increase nutrient loading, additional loading from streams must be minimized to prevent undesirable plankton growth.

**B - Recirculating Systems to Effect Nutrient Uptake in Aquaculture Systems and a Portable, Small-scale Flow-through System**

Intensive cultural processes in the aquaculture industry can result in nutrient loading far exceeding background concentrations. The sources of nutrients include undigested foodstuffs and metabolic wastes (faeces and urine) discharged by the cultured organisms. In open flow-through systems the nutrient load is discharged into downstream receiving waters, whereas, in closed recirculating systems the nutrient loads increase unless additional water is added to effect dilution. A convenient recirculating system, in which both trout and carp were being reared, located in close proximity to the University, was available with a local aquaculture producer, Mountain Trout Sales. Since the nutrient concentrations within the ponds often exceeded values considered detrimental to the health of the contained fish, it was decided that periphyton, contained in surrogate streams, might be used to uptake the nutrients. The system was established (Figure 69) and a variety of artificial substrata were employed on which a periphyton community was developed. The experiment was designed to determine which substrata configuration most efficiently removed nutrients (especially ammonia).

A small scale periphyton generating system has been designed to effect a portable bioassay sampler (Figure 70), using clear acrylic tubes with internally mounted substrata. The advantage of such a system lies in its ease of transportation over rough terrain. A similar system was developed and used by Petersen *et al.*, (1983) (and modified by Pringle 1987) to measure periphyton sensitivity to phosphorus loading in an Alaskan stream. The difference between the two designs rests in that the latter was an in-stream system, whereas, that described here was placed on the stream bank, receiving its water supply via gravity feed. Another surrogate stream system modification has recently been devised which used the PVC streams but uses a header box constructed of a flexible material, forming a soft-sided bag. The bag-like header-box would be held within an aluminium frame. The intent would be to produce a system which could be disassembled into components easily carried by one person, for deployment at locations not accessible by vehicle. The stream support tables should also be aluminium, with supply manifolds consisting of snap-on fittings and thru-hull fittings for ease of field assembly.

Figure 67. Topographical map of the Rithet Creek (RC) watershed, the primary catchment basin for Sooke Reservoir (Figure 1). The surrogate streams were located on two western primary order streams (FS - Forested Site; HS - Harvested Site). The seven sample site locations on Rithet Creek are also labelleled (1 – 7). Inset shows Sooke Reservoir catchment basin, with location of Rithet Creek in north-western corner.

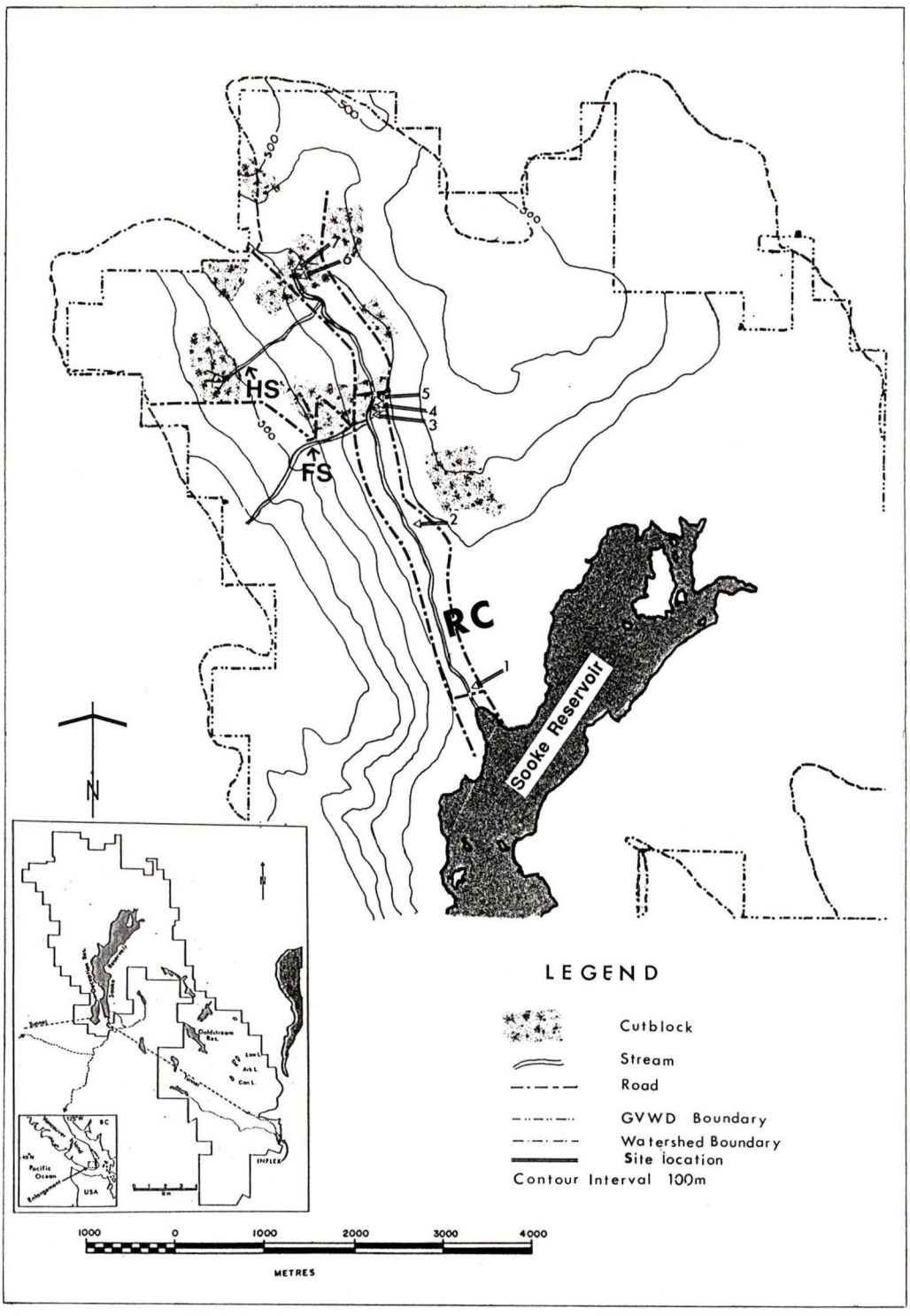

500
300
HS
FS
RC
1
2
3
4
5
6
7
Sooke Reservoir
N
LEGEND
Cutblock
Stream
Road
GVWD Boundary
Watershed Boundary
Site location
Contour Interval 100m
1000 0 1000 2000 3000 4000
METRES

Figure 68. Location of surrogate stream compounds adjacent to first-order streams flowing through both a forested (A; UL) and harvested (B; UR) section of the Sooke Reservoir watershed; the gravity feed supply lines (a) consisted of multiple PVC pipes. These two streams flow into Rithet Creek which discharges into Sooke Reservoir (Figures 1, 17). The upper middle photograph is of the periphyton community typical of the stream in B; the UM photograph was taken in the immediate foreground of photograph B. The stream facilities are shown in the lower photographs; C - forested, D - harvested. The following elements of the systems are discernible - a:feed lines; b:header box; c:supply manifolds; d:streams; e:neutral density screening; f:exhaust system; g:nutrient dosing reservoirs.


A
a
B
a
C
a
b
c
g
c
d
e
f
D
a
b
c
g
d
e
f

Figure 69. Recirculating stream system using two tanks within which fish were retained. The system comprised an electric pump (a; U), a clarifier (b), streams (c) and the reservoir fish-tanks (d). The clarifier contained two compartments - a downdraft plate clarifier (e; L) and an updraft chamber (f) from which the clarified water flowed into the white PVC distribution manifolds . The streams contained four different substrata configurations including vertically arrayed flat-surface plates (g), river cobble (h), vertically arrayed sinusoidal plates (i) and sinusoidal plates arranged perpendicularly to the axis of water flow (not shown). Each reservoir was supplied by four streams, consisting of one-each of the above four sample substrata configurations.

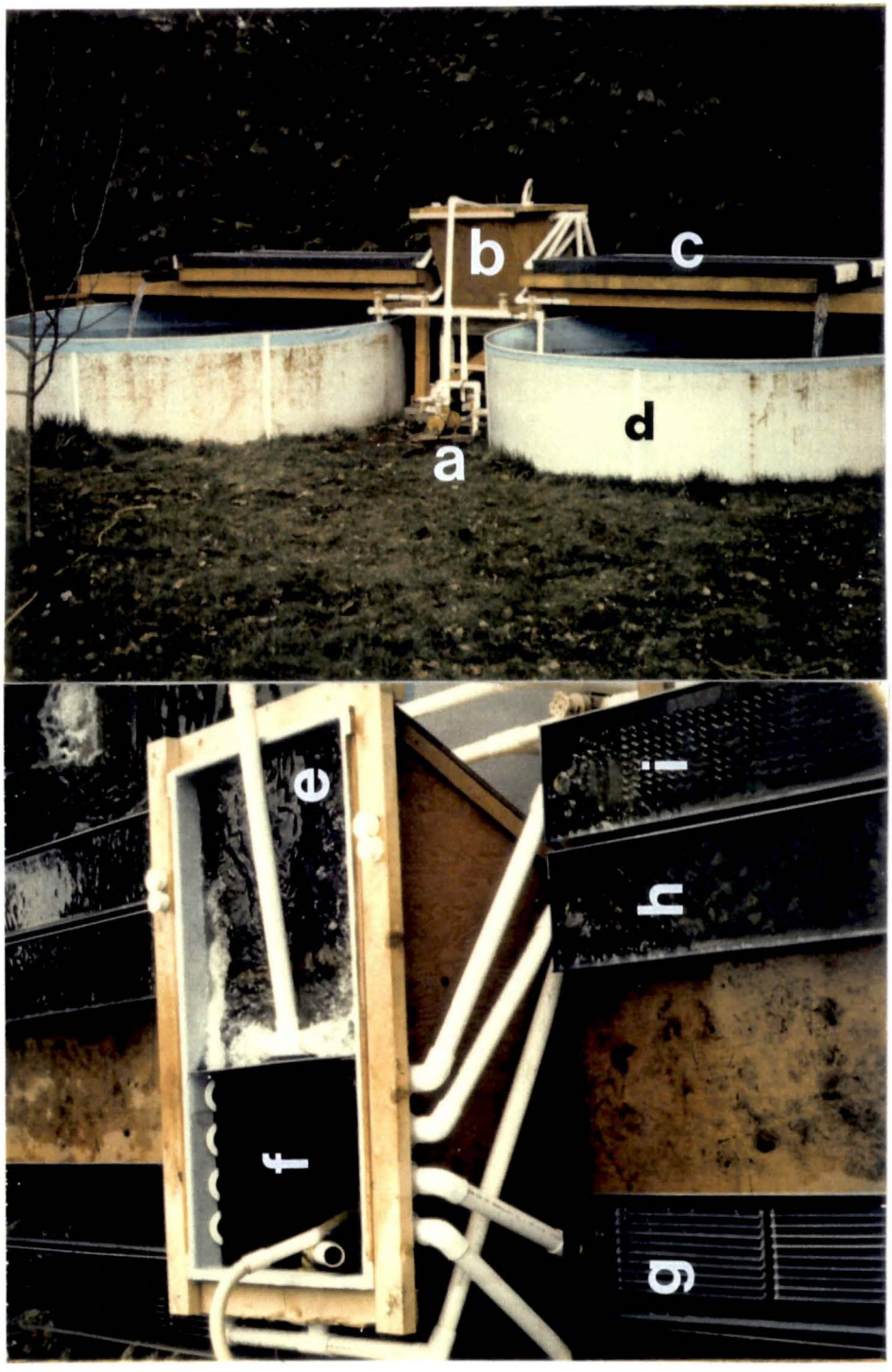
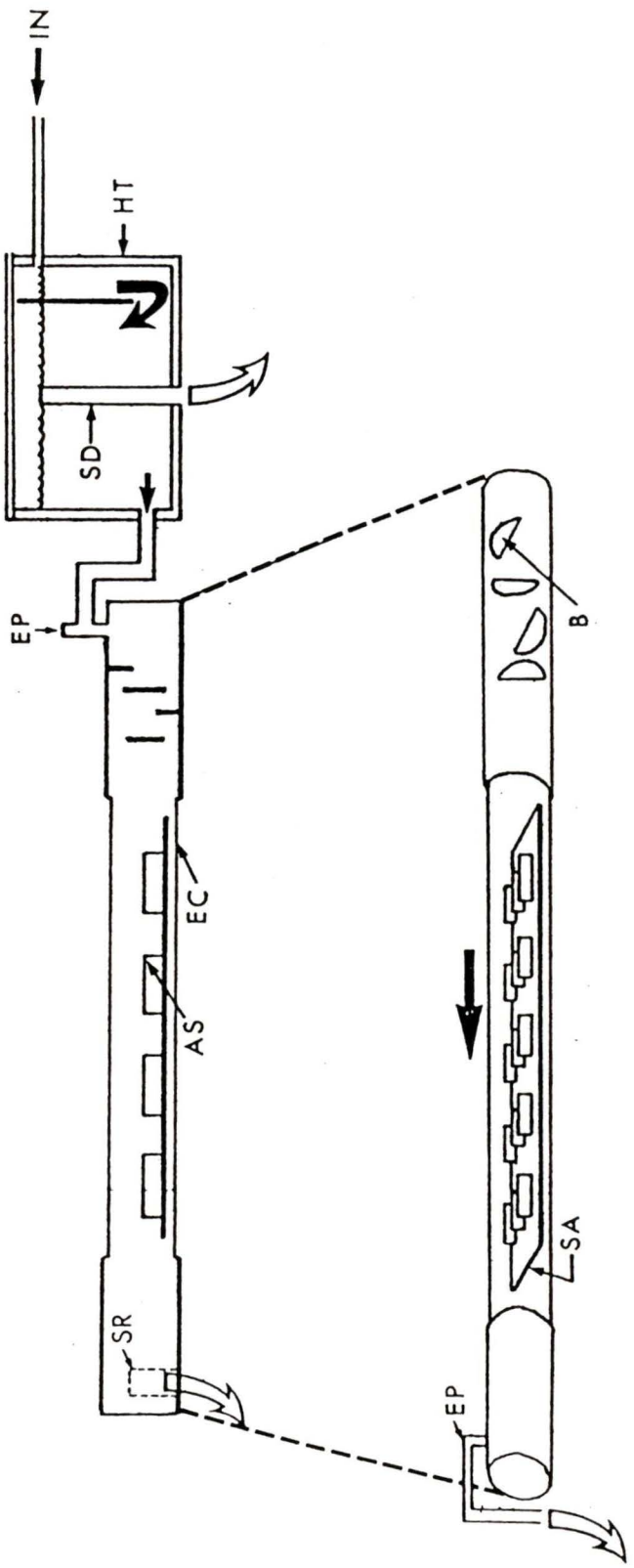

a
b
c
d
e
f
g
h
i

Figure 70. Cross sectional diagram of portable bioassay field sampler showing clear acrylic tubes with internally bottom-mounted vertical substrata (slides). Abbreviations – IN, inlet stream; HT, header tank; SD, stand-pipe drain; EP, exhaust port over-flow; EC, clear acrylic tube; AS, artificial substrata; SA, slide mounting apparatus; SR, slide removal handle.


IN
HT
SD
EP
B
EC
AS
SA
SR
EP

## Chapter V. General Conclusions

Atmospheric and land pollutants which adversely stress aquatic habitat have resulted in an increasing number of studies using periphyton to monitor pollution sources (Chessman, 1985) and nutrient enhancement (Bothwell, 1989; Lowe *et al.*, 1986). It is important to understand the mechanisms effecting alterations in the primary producer segment before attempting to predict the affects of environmental stresses on higher trophic levels. Periphyton communities have been used to test ecological hypotheses and to further our understanding of such important areas as whole-system productivity (Cushing and Wolf, 1984), nutrient spiraling (D'Angelo *et al.*, 1991), food web interactions and trophic status (Stevenson *et al.*, 1985) and terrestrial-aquatic energy inter-actions (Power *et al.*, 1985).

The findings presented here confirm those of Stokes (1984) and others (Steinman and McIntire, 1986; Schindler *et al.*, 1973) that experimental mesocosms and ecological modeling can provide realistic predictions of events in field situations. The results reported here confirmed the use of the algal component of periphyton as a sensitive bioassay to model community dynamics as advocated by Wetzel (1983a,b), Cairns (1982), Weitzel (1979a,b) and Bothwell (1989).

In a review of freshwater periphyton field techniques, Aloi (1990) laments the lack of standardization of methods used by periphyton researchers, especially for studies assessing community structure, biomass, and primary production (Rosemarin and Gelin, 1978). Sand-Jenson and Bohr (1983) call for establishing efficient standard methods for sampling periphyton on non-uniform substrata, whilst Sladeckova (1962) recommended development of a methodological manual. One such methodology which has received increasing attention is the use of unidirectional, flow-through streams (Bothwell, 1989), in which experimental manipulation of the contained periphyton is conducted under natural or semi-natural conditions (Warren and Davis, 1971). The data presented here confirmed the use of surrogate streams, using the indigenous organisms resident in the stressed lotic habitat, to measure subtle changes in nutrient concentration. This finding is

important in establishing how best to measure and monitor biological perturbations in natural systems which might not be detectable physically or chemically.

Recently, a symposium (Day *et al.*, 1988), on the use of community-level bioassays, has resulted in the call for development of reliable, field-based techniques with which to investigate and monitor the effects of anthropogenic induced stresses on aquatic communities. After designing and field-testing surrogate streams for nine years, in several investigations and locations these systems have proven to be valuable for use as a standardized method for such studies. The system reported here has generated data elucidating the effects of low level stressors including heavy metals, acid downshock, a herbicide (Glyphosate®), ammonia effects on aquacultured trout, and the eutrophication of secondarily treated sewage effluent, in addition to the influence of irradiance, current and nutrient enrichment on contained periphyton communities.

Surrogate stream studies avoid some of the complexities of island theory by permitting the researcher to deal practically with recruitment and permits more than one parameter to be simultaneously manipulated relatively easily (i.e. light, nutrient enrichment, velocity). Warren and Davis (1971) and others (Schindler, 1989) question whether much of the ecological research conducted in laboratories can be extrapolated to field conditions to further our understanding of natural wildlife population and community responses. Similarly, Roos (1983) suggests that to understand processes underlying periphyton community responses to environmental stresses the autecology of periphyton must be developed under semi-natural, controlled conditions. This should permit a more thorough understanding of the dynamic relationships linking solar irradiance and productivity, light inhibition (i.e. excessive intensities and recent concerns about ultraviolet wavelengths), density dependence and inhibition/competition, and herbivory, all of which are known to structure periphyton communities,.

Only recently have the effects of environmental stress on biochemical composition, and thus nutritional quality of phytoplankton, been studied

(Harrison *et al.*, 1990). Increased appreciation of the productive capability and biochemical (proximate) composition of periphyton communities as an alternative nutritional source or supplement will require a thorough understanding of how such stresses as nutrient and light limitation and current affect the biochemistry of the algal component and, subsequently, herbivores which feed upon them.

Ecosystem models are increasingly important in developing new sources of aquaculture feedstocks (Ridley-Thomas, 1989), fisheries resource management and habitat restoration (see extensive reviews by Stockner), water renovation and understanding how an increasing array of pollutants and land-use policies affect aquatic trophic structure and water quality. Prior to the development of such models, however, must be the development of standardized periphyton research methodologies.

The applicability of Hairston *et al's.*, (1990) hypothesis that *herbivore density is controlled by predation and parasitism and not by limitations in available plant biomass* in lotic environments should be amenable for study using surrogate split stream mesocosms. Surrogate streams could be used to manipulate periphyton herbivores and predators (*sensu* Steinman *et al.*, 1987a,b; Steinman, 1991), their parasites and changes in biomass accrual and its biochemical composition. The latter is important since Hairston's thesis addresses only food quantity and does not examine the trophic consequences of food quality (biochemical composition). Understanding how specific alterations of environmental parameters results in communities of definable structure, function and thus nutritional value will further understanding of the trophic role of periphyton and, ultimately, its terrestrial linkages (Power *et al.*, 1985). Whilst some fundamental contributions to our knowledge of periphyton dynamics have been made (Phinney and McIntire, 1965; McIntire, 1966a,b; 1968; 1973; Connell and Slater, 1977; Goldsborough and Robinson, 1986) inconclusive evidence exists to describe successional patterns in natural communities, especially relating community function and structure interactions, and their resultant effects on productivity. Specific energy inputs and stresses may quantitatively and qualitatively characterize both

algal nutritional energy available to higher trophic levels (Steinman and McIntire, (1986) and its palatability (Goldsborough and Robinson, 1986).

Since alterations in energy inputs (nutrients, irradiance, current) induce lotic community structure and qualitative functional changes, the work by Steinman and McIntire (1989) and Goldsborough and Robinson (1986) contradicts that of Odum (1985) who contends that at low stress levels quantitative functionality (biomass and productivity) are unaltered. Our work in surrogate streams has demonstrated that low level nutrient inputs alter biomass. However, such alterations can only be understood by examining a suite of other environmental parameters which also affect biomass accrual. Changes in biochemical composition can be related to specific intervention in energy inputs and are an index of the nutritional role of periphyton. Factors effecting such dynamic community processes as colonization, succession and stability are still largely poorly understood; whilst some hypothesis testing to elucidate these processes has been conducted in laboratory streams, little work has been undertaken on wild periphyton communities or those cultured in surrogate split stream mesocosms, located in the field. Preliminary work in our streams has shown (Ridley-Thomas, 1989) that biochemical composition of periphyton communities can be altered by reducing P or releasing the community from phosphorus limitation.

Given both the recognized importance of autochthonous production in lotic systems (Weitzel, 1979) and that periphyton primary producers occupy a basic energetic role in lotic trophic systems, understanding their role in transferring energy within the trophic hierarchy is important to modeling interactions between producers and consumers. Pringle's (1987) work on nutrient supply suggests early colonizers are capable of using luxury consumption to exploit initial pulses of phosphorus. This hypothesis could be tested by comparing the early community successional development (structural) of different contained communities receiving the same phosphorus enrichment (background concentration as control) but delivered either at a constant low concentration or at higher pulsed concentrations. A

third community would receive both the above (continuous plus episodic pulsed phosphorus).

Subsequent investigations should examine periphyton community response (structural, functional, biochemical) to multiple parameter manipulations, to increase model realism. The generation of a defined periphyton community, with known structure and nutritional content, would permit exploration of energy transfer mechanisms from primary producers to secondary and higher trophic levels.

**Funding for Ecological Research**

Finally, throughout this study one overriding aspect of my research has challenged me both ethically and intellectually - *how do we as scientists communicate, to the non-scientists, our concerns about our species seemingly relentless destruction of our planet.* In particular, how do we acquire the necessary financial resources to conduct that research which is essential to understanding how to halt or reverse the destruction but which may not have the appropriate political flavour or which has undesirable industrial or commercial implications. It was with these, and similar, questions in mind that I propose the following funding program.

The central concept is that ecological research funding should be long-term, devoid of political interference or agenda, subject to peer review (at all levels), fiscally effective, whilst containing a strong element of training and public education (Wetzel, 1991). Fiscally it would be healthy if the monies were of sufficient capital that only the *real* interest would be used to fund programs, ensuring no depreciation of the original capital. It was with these cautions in mind that the following objectives have been developed (assisted by Mr. Robert V. Wickett of Russel & DuMoulin):

1. Creation of an organization capable of managing and dispensing research funds for projects undertaken within the Squamish, Cheakamus and Green River catchment basins.

2. The organization will be a non-profit society incorporated pursuant to the provisions of the *Societies Act* of British Columbia.

3. The directors of the organization may directly or indirectly receive grants from the organization to conduct research.

4. All research would be conducted by graduate students in existing laboratories, in association with an accredited university graduate program. Supervisors and graduate committee members could act as directors of the society.

5. All research proposals would be subject to a peer review process, administered by the directors using external reviewers. In the case of a directors student application, to avoid potential conflict of interest, the director in question would abstain from discussion or voting on the proposal. All members of the board of directors must disclose any interest, direct or indirect, in any proposal before the society.

6. The organization will be controlled by scientists appointed by the society Chair in the first instance and by consensus thereafter; in the case of the board not achieving consensus, the Chair shall appoint replacement directors. Various levels of government may be represented on the board of directors but government must not have majority control.

7. The organization will control and manage research money dispensed from the interest earned on invested capital, provided by government, industry and private citizens.

8. Financial accountability for each research project would be provided by the appropriately accredited authorities at the grantees university. The duration of the grantees funding will be at the discretion of the directors, however, a minimum of three and four years funding would be made available for Masters and Doctoral students, respectively.

9. The society will be governed by nine (or any odd number) of directors. A minority of these directors are to be appointed by various levels of government and the term of the directors could be fixed or indefinite and government will retain the discretion to replace their appointed directors at will, subject to their acceptance by the remaining directors.

10. The society will have a trust account wherein its research monies are held and invested pursuant to investment rules specified in the articles of incorporation. Funds would only be dispersed upon the signature of a majority of the board of directors.

The above constitutes the essence of a proposal which has been vetted by a number of scientists, politicians and senior bureaucrats, who have endorsed the concept. In addition, a standing committee originating from a public advisory group over-seeing the development of a Liquid Waste Management Plan for the RMOW, consisting of industry, Crown Corporation, the four levels of government and public interest group representatives, have also endorsed the concept.

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# APPENDIX A

Surrogate Stream Design, Construction and Operation

**A Surrogate mesocosm, split-stream design and components and construction (described in the direction of flow)**

**Siphon**

The in-field system, protected by a fenced (chain-link) compound, lies at the base of an impounded surge reservoir (Figure 71, upper photograph). The semi-natural stream water was siphoned across the dam (Figure 71, middle photograph). Siphon intakes were suspended from a float (Figure 71, lower photograph) consisting of a welded foam-filled, plastic box, anchored to the dam by tubular pipes. This arrangement maintained the float at a pre-determined distance from the dam, minimizing entrainment of surface debris, and permited the float to rise and fall as water levels change. The latter was accomplished (Figure 72) by having a 'T' fitting, at both ends of the pipe, ride in a pair of 'U' clamps, attached to a railing running along the dam face. Suspended from the float were three, 3.81 cm (ID) PVC pipes (class 160 SDR26 pressure pipe), which siphoned water from three different depths (0.5, 1.0 and 1.5 meters). The intake design ensured that siphoned water recruited the biological community (and chemical constituents) from a vertical profile of the reservoir, minimizing the potential problem of stratified water and community distribution. The submerged pipe ended with a one-way valve, such that when siphon flow was interrupted, the water in the pipe reversed its direction of flow, closing the diaphragm, sealing the pipe. The retention of water within the pipe facilitated the resumption of flow (described in more detail under Siphon Initiation below). The float structure was designed to be robust enough to withstand considerable wind displacement, which could (and initially did) damage it.

The rigid siphon pipe leading from the lake (Figure 71 and 72) was interrupted at the lake-side edge of the dam by a short length (1.0 m) of flexible PVC hose. This flexible, pivoting section, constructed of wrapped, steel wired pipe, eliminated the stress otherwise placed on rigid tube as fluctuating water levels or wave action raised or lowered the float assembly. As the siphon pipe crossed the highest point on the dam-top a vertical 'T' joint with a gate valve (1.27 cm) was installed, to bleed trapped air from the

system (see section detailing siphon initiation and operation). Just upstream from the header box a second 'T' joint split the flow into a by-pass, consisting of two ball-valves separated by a length of PVC rigid pipe. At the lower end of this pipe and above the second valve was a perpendicular 'T' joint, to which was attached a second ball valve and a garden-hose coupling. This bypass was used in siphon initiation and when the header box required isolation without interrupting the siphon. All fittings were cemented to prevent siphon generated pressures from bursting connections; pressures within the siphon increase proportionately to the length of the downstream siphon-arm.

**Header-box**

Water siphoned across the dam falls through a 3.81 cm PVC pipe (G) and was temporarily retained in a PVC-lined header box (HB)(M) (dimensions 1.0 x 1.0 x 1.5m - 1.5$m^3$)(Figure 72). By maintaining a constant volume within the box, water was supplied to the streams at a constant pressure. The box was mounted on a robust wooden platform (designed to withstand a mass of 3500 Kg), such that the base of the header box was above the streams. Since the HB liner consisted of welded PVC, small fractures in the welds resulted in leaks, which could not be repaired without interrupting flows to the streams. These leaks could cause catastrophic failure to the HB by undermining the ground on which the structure sits and should not be ignored. Wherever possible a concrete base pad should be constructed, with proper drainage away from the header box and streams.

The header box consisted of a tightly fitting PVC liner (N) inside a plywood box; the top of the tank was sealed with a hinged lid (O), allowing ready access, whilst eliminating primary productivity within the box by excluding light, and preventing entry of air-borne debris. Although a few invertebrate species had attached themselves to the inner walls they never become a problem. The lid also permited above-box location of the carboys housing a variety of effluents which could be gravity fed into distribution manifolds. The water height within the box was maintained by siphoning a volume of water in excess of that required to operate all surrogate streams (flow rates of up to 1125 L per minute). The excess was discharged via a stand-pipe drain (P) into a

nearby creek. Water flow into the tank was regulated by a gate-valve (Q) and water was distributed into the manifold supply system via a second gate-valve (Qa). All perforations through the PVC liner use rubber and cork sealed, through-hull (bulk-head) fittings to maintain the integrity of the PVC liner. Whilst the liner was welded, all other fittings were press fitted - no glue was used beyond the header box. This facilitates rapid in-field disassembly for cleaning and repair or design modifications. Water residence time within the system was less than one minute, with water flow through the streams, depending on velocity and flow rates, typically less than 10 seconds. These high hydraulic flushing rates prevented water stagnation and also prevented ice formation during the coldest months (for a description of cold weather streams see Bothwell, 1983).

A number of header boxes, of the design described above, have been successfully used since 1983. However, simpler alternatives may provide greater ease of installation and reduced cost. One such design could use pre-fabricated, round tanks of approximately 2000 L capacities. They are lighter than the header boxes described above and should be easier to install. The major difficulty was thought to be the sealing of thru-hull fittings to the tanks curved wall; possibly a modified seal could be fashioned or molded to provide the necessary rigidity to ensure an adequate bulkhead seal. Another design would use a flexible bag, suspended in an aluminium frame. The thru-hull fittings would consist of snap-on, quick-disconnect fire-hose couplings, similar to fire-fighting water supply containers suspended beneath helicopters. Our evolving system will test such designs.

**Supply manifold**

Water flowed from the base of the header box through a gate-valve (Gv-c), regulating water volumes discharged to streams, into PVC manifold pipes. Each manifold fed one set of paired streams (T)(Figure 72B/D). Immediately distal to the gate-valve (Gv-c) was an injection port (IP), through which nutrient additions can be added to the water supply, prior to its discharge into the streams. Placing the injector ports close to the header-box ensures thorough mixing before discharge into the streams. The additives were

contained in 20 L carboys (C), stored above the header box and accessed by stairs adjacent to the header box.

**Nutrient Dosing Apparatus**

Nutrients, stored in carboys (C), were supplied to the streams using a FLEXFLO (Ross Laboratories, Columbus, Ohio 43216) gravity gavage set (Figure 72C). This apparatus permited nutrient additive concentrate to be gravity fed at variable rates into the distribution manifold. Final additive concentrations were established by adjusting the flow regulator (FR) to a predetermined drip rate. The drip rate was determined by selecting a hypodermic needle, of given bore diameter, which discharged the treatment solution at a fixed rate. The drip rates were established using a carboy, set-up in the laboratory, and empirically measuring the discharge rate through needles of various bore diameters.

The gravity-feed nutrient-dosing apparatus consisted of carboys (20 L) containing stock solutions of nitrogen ($NaNO_3$), phosphorus ($Na_2HPO_4$), or both, in conjunction with a drip feed system (Flexiflow gravity gavage set (Ross Co., Ohio)). Consistent molar concentrations of stock N and P solutions were used and added to de-ionized water in the carboys; nutrient from the carboys dripped at a constant flow (1.5 mL $min^{-1}$) to achieve the desired, thoroughly mixed, nutrient concentration within each stream. Carboys were replenished weekly and drip feed rates monitored and altered as necessary.

The system described here has been used to deliver a variety of nutrients and toxins including sulphuric acid (Lucey *et al.*, 1986); di-Sodium hydrogen orthophosphate, Sodium nitrate, Ridley-Thomas (1989); heavy metals (Ros, 1989); and treated sewage effluent (Lucey *et al.*, 1993). Nutrient additive concentrates were prepared using distilled or deionized water, so that no undissolved solids could block the flow through the gavage tubing or needles. At the Cheakamus and Squamish River experimental site, lidded 20 L buckets were placed up-stream with bottom mounted gavage gravity apparatus. Concentrates were chosen which would result in continuous delivery for five to seven days. At these drip rates, time spent in the field maintaining the system was minimized.

**Siphon Initiation**

Initially the siphon assembly did not contain a siphon by-pass. The siphon flow was established after sealing the end of the submerged (lake-side) tube-end, closing the valve Q and filling the siphon assembly at port I. This was done by drawing water from the lake in buckets. Whilst effective, this procedure was cumbersome and required two individuals. The first person had to seal the submerged tube-end and quickly unseal it while it was held submerged, to prevent interruption of the siphon and to reposition the tube-end at the proper depth (E), by reattaching it to the perpendicular float-arm. (The procedure required a boat, with safety considerations dictating two people.) The second person was required to fill the siphon assembly and then, simultaneously with the unsealing of the submerged tube-end, quickly open the valve Q, to initiate the siphon. This procedure has worked well, however it has certain inherent limitations and a second, superior siphon initiation procedure has been designed and successfully field-tested for the past three years at both the Humpback and Whistler sites.

A modified siphon has been designed to be operated by one person without requiring the use of a boat and permits the header box and streams to be isolated from siphon flows without interrupting siphon-flows. This option allows the system to be repaired, maintained, or cleaned, or new experiments to be initiated without re-starting the siphon.

Siphon initiation (with a dry system) begins with closing valves Q and Kb and opening I, Ka and L. If a pressurized water supply is available a garden hose is attached to the distal side of valve L. The hose is then used to rapidly fill the system; the pressure of the water flowing into the submerged end acts with gravity to close the one way valve (Aa) attached to the submerged tube-end, eliminating the need for additional individuals and a boat, and permiting the submerged pipe to be permanently attached to the perpendicular float-arm. Entrained air within the pipe was displaced by the water and blown off through the valve I. The valves L, and then I, were closed and valve Kb opened to initiate the siphon; the extra weight of water in the by-pass further facilitates starting the siphon.

Once the siphon was operational, water can be directed into the header-box by opening valve Q and shutting valve Ka, which isolates the siphon by-pass; if used, the garden hose can then be disconnected. Conversely, with the by-pass operational and the header-box isolated, the garden hose can be used to generate a pressurized water supply to facilitate cleaning the system or for other purposes within the confines of the system. If a pressurized water supply was not available, the siphon assembly filling procedure was modified by unscrewing valve I and filling with manually drawn water; when the piping was full, the valve threads were resealed with Teflon tape and the closed valve was screwed back into the T-joint; the primed siphon was initiated as outlined above for a system with a pressurized water supply. To interrupt the siphon one simply opens valve I.

### Streams

The semi-natural streams (Figure 72D) were constructed of 4.5 mm PVC sheet, heat bent to produce an open rhomboid the dimensions of which were base (18.0 cm), open-top (26.0 cm) sides (11.0 cm), height (10 cm) and length (245 cm). The angle of base sides (114°) was designed to minimize shading of contained periphyton communities. The PVC was obtained in 122 x 244 cm (4' x 8') sheets, cut into the appropriate widths and a wooden jig was used to obtain the proper bend angle. It was essential that each stream blank be molded at precisely the correct angle to facilitate the welding of the end-plates - imprecise angle formation can result in gaps between the stream wall and end-plate, which may lead to weld induced structural deformation.

Each PVC welded end-plate was drilled to accept either a single supply port (SP) into which a threaded slip/male adapter was screwed, or five exhaust ports (EP), consisting of two staggered rows of holes (2.5 cm dia.) into which were pressure-fit 90° elbows (E). These direct stream waste water into a gutter system (GS), which discharges into a PVC (15.0 cm dia.) network linking all streams, which eventually discharge into a swamp and thence into a small stream. Water height within the streams was regulated by partially blocking water flow in lower ports; it was important not to completely occlude these lower ports, otherwise deeper water becomes stagnant, with only surface

water entrainment occurring. Light intensity (quantity) (Figure 3) was controlled using neutral density PVC mesh screens which were suspended over the streams on thin wooden frameworks. Light quality can be controlled using coloured plastic sheeting, cited in descriptions of laboratory based investigations, but no work has been conducted in our facilities on affects of light quality on periphyton.

**Artificial Substrata**

A variety of substrata have used within the streams on which to develop a contained periphyton community. These include glass microscope slides (GS), PVC slides, styrofoam sheets (SS), corrugated plastic sheet, river rock (RR) and plexiglass; the PVC surfaces of the stream have also provided a surface on which communities have become established and subsequently removed and analyzed. Artificial substrata provide replicatable, easily handled sampling surfaces, whilst allowing some measure of control of the community variability on natural substrata (Patrick 1971; Sladeckova, 1962; Beak *et al.*, 1973; Cairns, 1982).

**Exhaust/Discharge manifolds**

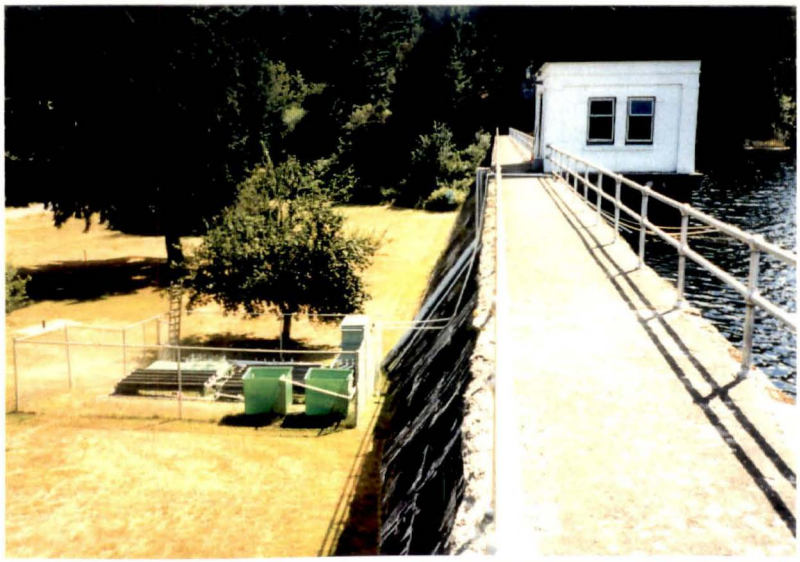
Water which has flowed through the streams and been discharged via the 90° elbows into the gutter stream (Figure 72D), was subsequently collected by means of an inter-connected common drainage system (Figure 3 - LL). This common drainage system then discharges into an adjacent wetland, prior to flowing into a local creek. The excess water from the header box also was discharged into the common drainage, and provides significant dilution of the water emanating from the streams. Additional flushing of the marsh was provided through episodic discharge from the G.V.W.D. screen back-flushing procedure, housed in the building adjacent to the surrogate stream facility (Figure 3, upper photograph).

**Summary**

The stream design described here has been used continuously for nine years with no significant design alterations. The improved siphon design has

reduced the need for operating from boats and has allowed rapid siphon initiation. The header box design currently used has proven to be somewhat cumbersome and expensive to move to new sample sites. At the site of the Cheakamus/ Daisy Dam an overhead crane was needed to lower the header box to the dam-base. The new design modifications noted above should markedly reduce set-up time and permit one individual to transport the system, which would increase the number of potential sites at which the potentially portable system could be used. In sites requiring the use of electrical pumps, the increased expense of transformers, electrical cable, consumption meters, ground-fault interrupters, etc has been compared with the siphon system. The need to obtain appropriate operational permits also requires additional lead-time in system installation. Overall the siphon system has been demonstrated to be a simple, easily operated facility which was relatively inexpensive to construct and implement.

Figure 71. Siphon apparatus supplying water to the header box (U). Water siphoned across the dam passed through a flexible joint (a; M), flowing past a "T" joint (b) prior to discharge into the header box. The "threaded T" section, with removable valve, at the highest point of the intake lines permited the system to be primed, prior to initiating the siphon supply; the submerged end of each intake line had a one-way valve attached (f; L), to assist in maintaining the filling of the intake line. The flexible anchoring tubes (c) ensured the float (d) did not drift laterally from its fixed position. The flexible connector at the dams edge (a) permited the intake pipes to scribe an arc in the vertical axis as the reservoir surface level changed during drawdown and recharge periods. The three siphon lines (e) drew water from 0.5, 1.0 and 1.5 meters below the surface.




b
a
a

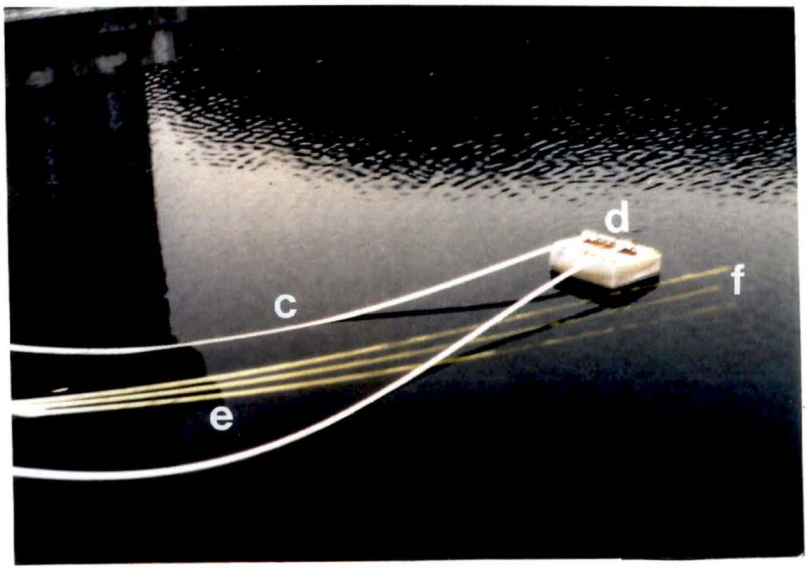
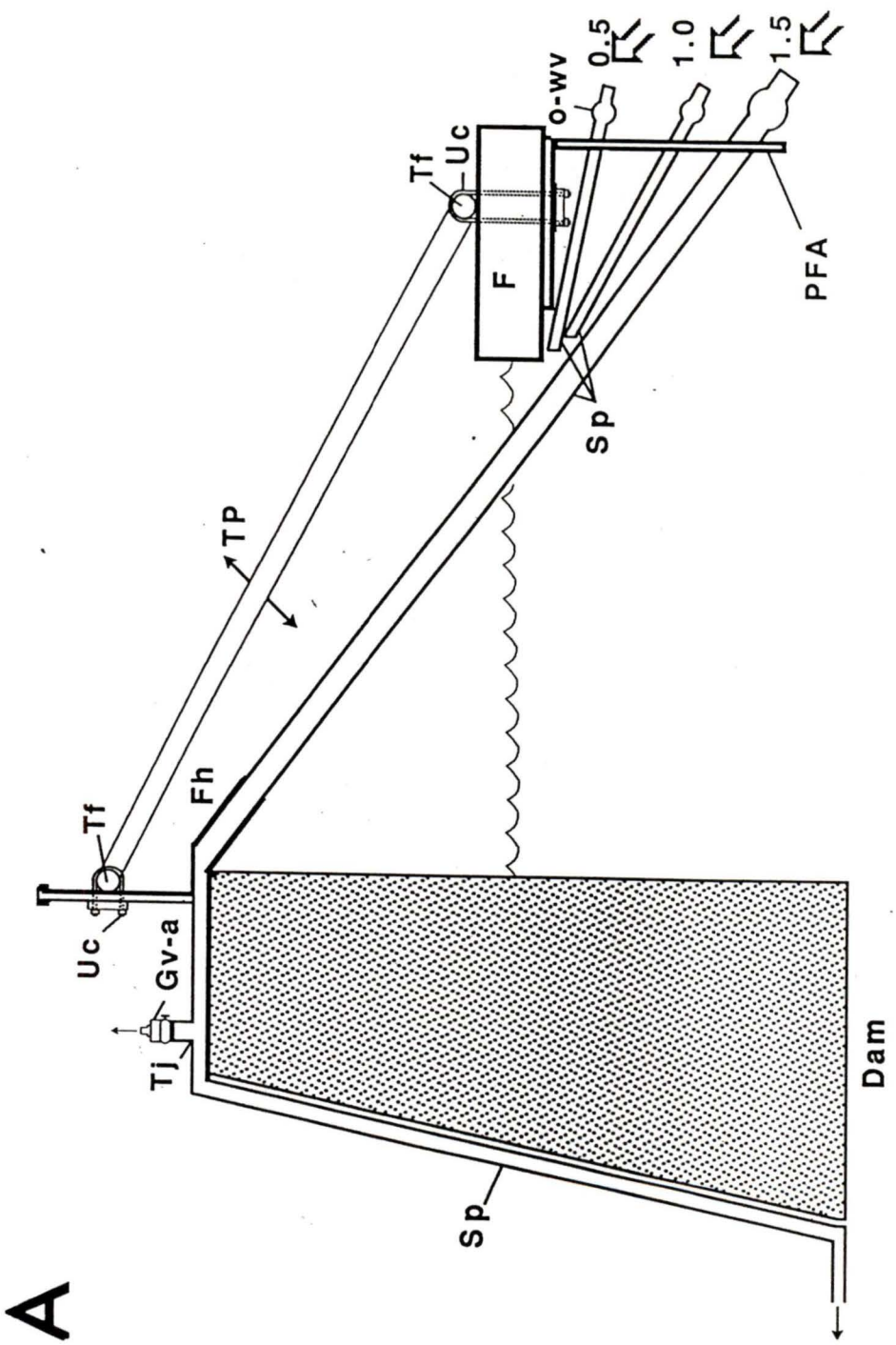

d
c
f
e

Figure 72. Schematic diagrams of the surrogate stream system showing its essential components. A) siphon supply; B) header box and bi-pass/siphon charging unit; C) gavage nutrient dosing assembly; and, D) stream with sampling substrata. Symbols for each diagram as follows:

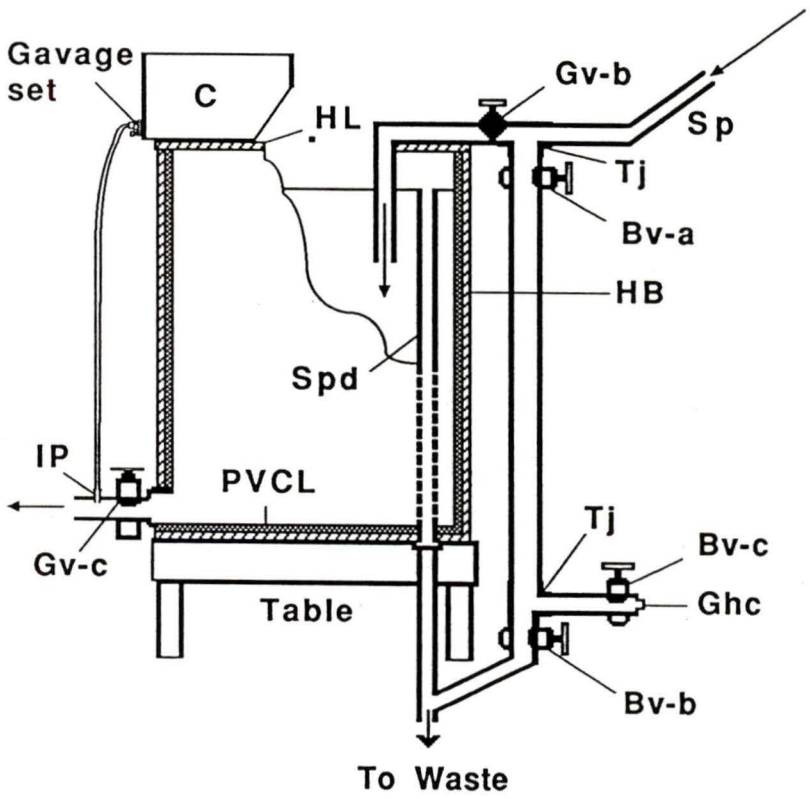
A F - float; Fh - flexible hose; Gv-a - Gate valve; o-wv - one-way valve; PFA - ; Sp - Supply pipes; Tj - T joints; Tf - T flex-joint; TP - ; Uc - U clamp.

B Bv-a, b, c - Ball valve; C - Chemical reservoir; Ghc - hose connector; Gv-c - Gate valve; HB - Header Box; HL - Header Box Lid; IP - Injection Port; PVCL - PVC Liner; Sp - Supply pipe; Spd - Stand pipe drain; Tj - T joint.

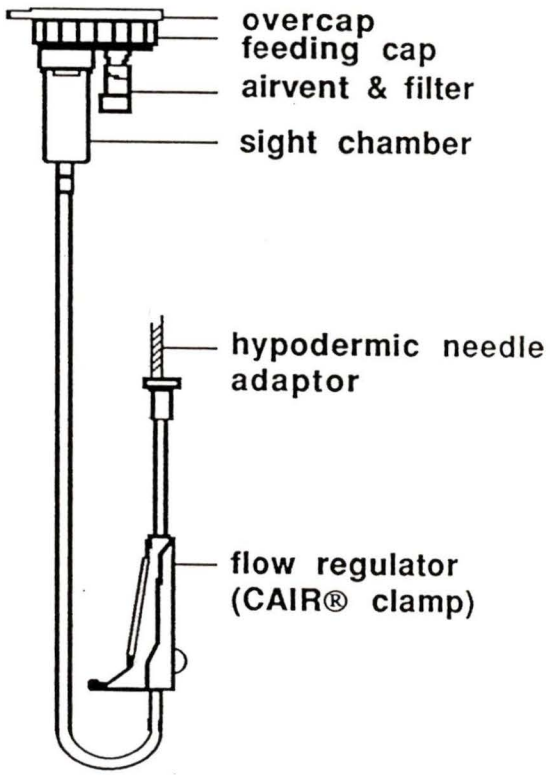
D E - Exhaust tube; EP - End Plate; GS - Glass Slide; RR - River Rock; SP - Supply Plate; SS - Scanning (Electron Microscope) Stubs.


A
Uc
Tf
Gv-a
Tj
Fh
TP
Tf
Uc
F
o-wv
0.5
1.0
1.5
PFA
Sp
Sp
Dam

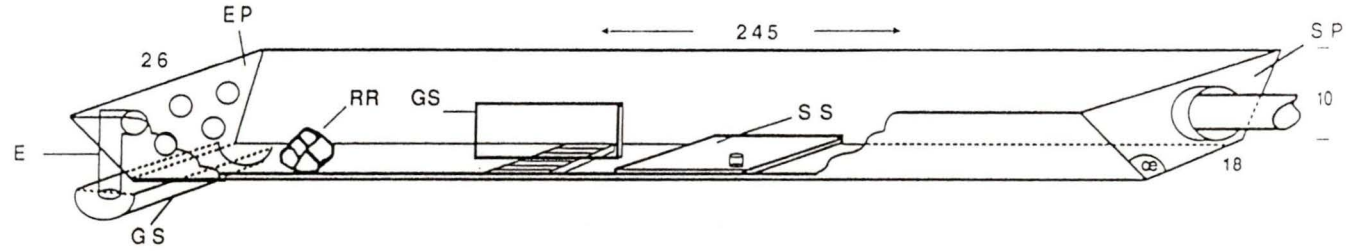
# B


Gavage
set
C
HL
Gv-b
Sp
Tj
Bv-a
HB
Spd
IP
PVCL
Tj
Bv-c
Gv-c
Table
Ghc
Bv-b
To Waste

C


overcap
feeding cap
airvent & filter
sight chamber
hypodermic needle
adaptor
flow regulator
(CAIR® clamp)

# D



# APPENDIX B

A Modified Support Base to Increase Efficiency When Preparing Multiple, Simultaneous Utermöhl Sedimentation Chambers

## INTRODUCTION

The growing recognition of the need to directly measure biological responses to aquatic perturbations (Cairns 1974; Odum 1985; Day *et al.*, 1988) has resulted in an increased use of microscopic organisms (or "micro-wildlife" (Austin 1969)), especially the sedentary microhaptobenthos or periphyton community (Weitzel 1979; Cairns 1982; Wetzel 1983; Aloi 1990). Methods to enhance microscopic enumeration of microwildlife include concentration of biota through water displacement (Dodson and Thomas 1964), onto filters (McNabb 1960; Moore 1963), settling onto permanent slides (or coverslips) (Coulon and Alexander 1972; Crumpton and Wetzel 1981), and examination within chambers. Types of chambers used include haematocytometers, and those developed by Sedgewick-Rafter, Naumann, Kolkwitz and Utermöhl (Lund *et al.*, 1957). Modifications of the original Utermöhl chamber (Taylor *et al.*, 1986) have permitted reliable, quantitative assessment of much of the planktonic microscopic wildlife as bioassay organisms; unfortunately, given their extremely small size, the microbial community (Stockner 1991) requires other methods. Whilst Aloi (1990), amongst others, laments the lack of attention paid to developing improved, efficient standard equipment and methods for sampling microwildlife, especially the periphyton, arguably the Utermöhl sedimentation technique has achieved the distinction of being the most widely used laboratory procedure for estimating the abundance of microwildlife populations (Taylor *et al.*, 1986).

One limitation of counting microorganisms on the bottom of a sedimentation chamber is the time required to make the counts. We have used Utermöhl chambers extensively during examination of long-term changes in the plankton and periphyton of oligotrophic lakes (Austin 1983; Lucey *et al.*, 1986; Deniseger *et al.*, 1986b) and streams, on both natural (Deniseger *et al.*, 1986a) and artificial (Austin *et al.*, 1990) substrata. Since algal densities in our oligotrophic waters were generally very low, we typically needed to assess sample volumes of 100 mL, which necessitated long sedimentation times (usually 24 hours). This increases chamber turn-around time, reducing enumeration efficiency unless a large number of chambers are available. A second major time-consuming component of using the large-

volume sedimentation tubes is the post sedimentation disassembly and cleaning of chambers; this problem is exacerbated when there is need to assemble numerous large chambers. The removal of silicone, used to seal the 100 mL cylinders to their base plates, also requires considerable time. Contamination of the basal cover-slip with silicone necessitates chamber disassembly, cleaning and reassembly. We have developed two simple modifications of the Utermöhl process (Wetzel and Likens 1991) to reduce the time normally required for preparation and use of multiple 50 and 100 mL chambers.

**Holding Rack Modification**

Utermöhl sedimentation apparatus (Figure 73) consists of a base-plate containing the glass coverslip, a sedimentation tube and a glass or plastic top-plate for sealing the vertical tube, once it is filled. Traditionally the underside of the tube's flat rectangular base is coated with a silicone sealant and then firmly placed on the horizontal base plate housing the coverslip, upon which the particulates settle. Following complete sedimentation, the tube and supernatant must be removed without disturbing the particulates on the coverslip. This is effected, without disturbing the base plate, by sliding the vertical tube laterally, placing it over the discharge hole at the end of the base-plate and removing the top-plate (to break the seal) allowing the supernatant to be drained from the vertical tube which is then removed from the base plate. Another coverglass is then placed above the sedimentation well of the base plate to exclude dust, prevent evaporation and to minimize optical aberrations induced by air bubbles. Unfortunately, when the vertical tube is slid across the sedimentation well, small quantities of sealant often remain forming a film on the water in the well, which can end up on the base-plate coverslip. The silicone adhering to the coverslip is difficult to remove using normal cleaning procedures. It is then necessary to disassemble the base plate and remove and clean the coverslip with a cleansing solution. This increases the likelihood of breaking the costly coverslips.

Our modification of the Utermöhl procedure using 100 mL tubes resides in sealing a plexi-glass square (Figure 73) to the upper end of the sedimentation

tube (a; b; G), instead of sealing the tube to the base plate (A; G). The tube is then inverted, and the staining solution (Lugol's) and sample liquids added. The base-plate is inverted, centered over the sedimentation tube's rectangular base, pressed firmly together, and the whole rotated quickly 180°. The assembly is then carefully placed on the PVC table and left for 24 hours. The vertical chamber is then slid laterally until it is over the evacuation hole, as in the standard method. The next step is normally to remove the glass disk coverslip, at the top of the sedimentation tube, allowing evacuation of the contained sample. Since in our modification the disk is sealed to the tube an alternative evacuation procedure is required. This is effected by placing a Pasteur pipette tip-end under the evacuation hole (Figure 73; PP); a quick compression of the bulb forces air into the vertical tube, causing the vacuum within the vertical tube to be broken and the contained sample to flow into a shallow vessel (DC) placed below the evacuation hole. The absence of any sealant between the vertical tube and base-plate eliminates the problem of cover-slip contamination, thereby also reducing the risk of coverslip breakage and, thus, enumeration costs. The vertical tube can be cleaned with a detergent, rinsed and air dried. When using the technique described here it should be recognized that a sample volume of slightly less than 100 mL will result from the tube and sedimentation base plate not being completely filled. This requires measuring the volume of discharged water to calculate estimates of phytoplankton population density.

**Modified Support Base**

The need to perform numerous, simultaneous counts required development of a table to hold the chambers. The table we have designed requires the Utermöhl apparatus to be handled only once, through provision of a stable platform on which the vertical tubes can be slid away from the chamber well. The stable platform is important for random sedimentation, an assumption in the statistical assessment of this procedure (Lund *et al.*, 1957). Experience with this design has demonstrated that a table housing ten chambers is the maximum that can be handled easily. PVC construction is based upon low cost, ease of procurement, machining and cleaning, simplicity of repair and maintenance and durability.

The table has dimensions of 74.0 cm (length), 16.5 cm (breadth), 1.0 cm (thickness). The wells within which the sedimentation chamber sits were 4.5 cm in diameter and offset from the longitudinal axis. The evacuation holes were laterally offset from the wells. Three pairs of legs support the table; the legs were made of PVC doweling snugly pressed into holes drilled two-thirds of the thickness of the PVC plate. Any discontinuity in height can be removed by sanding the legs. The distance between the evacuation hole and the table edge must be less than the radius of the Utermöhl sedimentation tube, such that when the tube is slid laterally over the evacuation hole a small portion of the tube protrudes beyond the base plate. This protrusion is necessary to permit the introduction of air into the tube, to initiate draining of the supernatant (Figure 73). Care must be taken that all sharp edges have been beveled when machining of the table has been completed.

**Statistical Assessment**

Consideration that the new procedure might adversely affect the sedimentation of cells within the tube was evaluated. An assessment was made, by sampling a mono-culture of *Scenedesmus,* with numerical abundances estimated using both the traditional and modified sedimentation processes. Table 12 shows that no significant differences existed between samples analyzed using the two techniques.

Since our work with large numbers of chambers requires different microscopes for enumerating algal cells, each of which has slightly different optical configurations, a mathematical formula was derived to yield the number of cells per unit area of periphytic colonized substrata (or volume of

lake water). A conversion factor, for any given microscope, can be determined using the following formulae:

$$\frac{100}{\text{(\# fields counted) (X)}} \times \frac{1}{\text{(\# mls in chamber)}} = \text{\# organisms ml}^{-1}$$

where: $X = \frac{\text{(area) field}}{\text{(area) chamber}} \times 100$

and, for illustration (Hetherington *et al.*, 1993), actual values used were

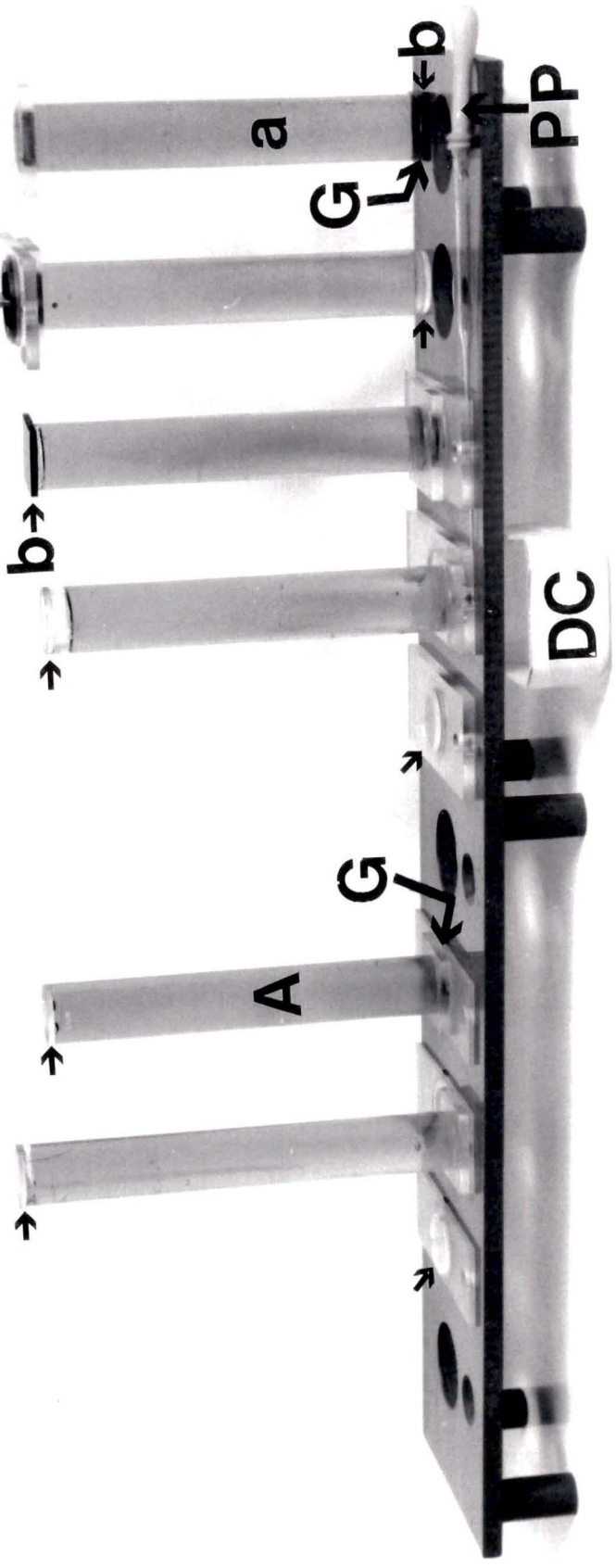
| | | |
|---|---|---|
| diameter of counting chamber | 2.60 | cm |
| diameter of field | 0.84 | mm |
| area of counting chamber | 5.31 | $cm^2$ |
| area of field | 0.55 | $mm^2$ |
| area of 40 fields | 0.22 | $cm^2$ |
| conversion factor | 5.31 | $cm^2 = 0.96\ ml^{-1}$ |
| for 40 fields per ml | 0.22 | $cm^2$ |
| volume of chamber | 25.0 | ml |

where 0.96 was approximated as 1.0. On the basis of our test results and hundreds of analyzed samples, we have not observed any significant differences between these two procedures. The modified Utermöhl sedimentation technique described here has thus been shown to yield the same population estimates as the traditional method, however, it is significantly more effective in terms of ease, speed of use and cost of preparing algal samples for microscopic enumeration.

Table 12. Statistical assessment of four-way comparision of old and new procedure, where the value of each cell represents the total number of *Scenedesmus* cells counted per chamber, with 40 fields per chamber counted. Old/New style = assembly procedure; Old/New base = sedimentation chamber style. Right-hand columns show mean (upper) and std. dev. (lower) of three trials, at left.

| | Old Style | New Style | Old Style | New Style |
|---|---|---|---|---|
| Old Base | 3435 | 3867 | 3801.67 | 3800.00 |
| | 4142 | 3694 | 355.37 | 109.10 |
| | 3861 | 3896 | | |
| New Base | 3941 | 3462 | 4003.67 | 3584.67 |
| | 4102 | 3722 | 80.21 | 129.51 |
| | 4013 | 3591 | | |

Figure 73. Modified Utermöhl settling chambers (A) and the multiple chamber holding rack. The photograph outlines the essential steps in the traditional process (Left of center) and the modification (Right of center) described in the text. The letters correspond to the following elements: Utermöhl chamber (A,a); glass disk (small arrows); plexiglass square (b); stopcock grease (G); discharge container (DC); Pasteur pipette (PP).


a
b
G
PP
b
DC
G
A

# APPENDIX C

Utermöhl Raw Data and Statistical Assessment

| | 1 | 2 | 3 | 4 | 5 | 6 |
|---|---|---|---|---|---|---|
| new style | 65 | 60 | 68 | 67 | 68 | 71 |
| old base plate | 80 | 79 | 52 | 87 | 78 | 78 |
| | 53 | 49 | 73 | 88 | 88 | 94 |
| | | | | | | |
| new style | 118 | 105 | 87 | 88 | 92 | 76 |
| new base plate | 93 | 74 | 95 | 100 | 92 | 79 |
| | 79 | 78 | 65 | 98 | 123 | 106 |
| | | | | | | |
| old style | 92 | 97 | 102 | 83 | 89 | 86 |
| old base plate | 104 | 84 | 90 | 119 | 138 | 76 |
| | 100 | 103 | 108 | 94 | 68 | 114 |
| | | | | | | |
| old style | 82 | 109 | 109 | 90 | 111 | 90 |
| new base plate | 90 | 113 | 102 | 98 | 98 | 100 |
| | 115 | 97 | 105 | 91 | 85 | 102 |
| | | | | | | |
| new style | 66 | 92 | 115 | 120 | 79 | 80 |
| ols base plate | 122 | 102 | 112 | 100 | 96 | 82 |
| | 84 | 118 | 96 | 110 | 78 | 90 |
| | | | | | | |
| | *Row 2* | | | *Row 3* | | |
| | | | | | | |
| | Mean | 68.75 | | Mean | 74.325 | |
| | Standard Error | 1.946314729 | | Standard Error | 2.236179061 | |
| | Median | 68 | | Median | 73.5 | |
| | Mode | 68 | | Mode | 80 | |
| | Standard Deviatio | 12.30957518 | | Standard Deviatio | 14.14283818 | |
| | Variance | 151.525641 | | Variance | 200.0198718 | |
| | Kurtosis | -0.793523699 | | Kurtosis | 0.681805889 | |
| | Skewness | -0.095060125 | | Skewness | 0.126304907 | |
| | Range | 45 | | Range | 69 | |
| | Minimum | 45 | | Minimum | 42 | |
| | Maximum | 90 | | Maximum | 111 | |
| | Sum | 2750 | | Sum | 2973 | |
| | Count | 40 | | Count | 40 | |

| 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 |
|---|---|---|---|---|---|---|---|
| 71 | 68 | 87 | 79 | 69 | 62 | 86 | 90 |
| 68 | 93 | 74 | 84 | 70 | 66 | 80 | 64 |
| 84 | 94 | 98 | 73 | 75 | 93 | 65 | 96 |
| | | | | | | | |
| 63 | 97 | 66 | 60 | 86 | 88 | 80 | 116 |
| 90 | 106 | 111 | 93 | 96 | 95 | 94 | 90 |
| 106 | 112 | 104 | 100 | 120 | 95 | 88 | 92 |
| | | | | | | | |
| 100 | 74 | 92 | 80 | 82 | 104 | 110 | 104 |
| 116 | 101 | 83 | 79 | 130 | 109 | 86 | 118 |
| 92 | 118 | 104 | 94 | 72 | 92 | 98 | 78 |
| | | | | | | | |
| 117 | 87 | 95 | 101 | 120 | 109 | 85 | 90 |
| 106 | 118 | 89 | 90 | 90 | 96 | 94 | 112 |
| 95 | 106 | 115 | 79 | 120 | 100 | 116 | 106 |
| | | | | | | | |
| 90 | 90 | 108 | 117 | 86 | 102 | 80 | 110 |
| 146 | 83 | 47 | 58 | 57 | 44 | 108 | 84 |
| 86 | 94 | 120 | 79 | 134 | 85 | 94 | 104 |
| | | | | | | | |

| *Row 4* | | | *Row 6* | | | *Row 7* | |
|---|---|---|---|---|---|---|---|
| | | | | | | | |
| Mean | 79.475 | | Mean | 93.9 | | Mean | 92.875 |
| Standard Error | 2.227159712 | | Standard Error | 8.193477706 | | Standard Error | 1.773084918 |
| Median | 76 | | Median | 86 | | Median | 94.5 |
| Mode | 73 | | Mode | 80 | | Mode | 95 |
| Standard Deviatio | 14.08579481 | | Standard Deviatio | 51.82010302 | | Standard Deviatio | 11.21397365 |
| Variance | 198.4096154 | | Variance | 2685.323077 | | Variance | 125.7532051 |
| Kurtosis | -0.449190709 | | Kurtosis | 35.62406172 | | Kurtosis | 0.196751066 |
| Skewness | 0.104341304 | | Skewness | 5.81613256 | | Skewness | -0.239240998 |
| Range | 59 | | Range | 345 | | Range | 49 |
| Minimum | 49 | | Minimum | 60 | | Minimum | 68 |
| Maximum | 108 | | Maximum | 405 | | Maximum | 117 |
| Sum | 3179 | | Sum | 3756 | | Sum | 3715 |
| Count | 40 | | Count | 40 | | Count | 40 |

| 15 | 16 | 17 | 18 | 19 | 20 | 21 |
|---|---|---|---|---|---|---|
| 68 | 80 | 51 | 73 | 88 | 48 | 67 |
| 86 | 79 | 73 | 80 | 70 | 64 | 60 |
| 73 | 89 | 71 | 68 | 59 | 68 | 76 |
| | | | | | | |
| 88 | 84 | 86 | 87 | 92 | 85 | 78 |
| 102 | 91 | 95 | 95 | 68 | 95 | 77 |
| 90 | 70 | 95 | 60 | 95 | 95 | 98 |
| | | | | | | |
| 74 | 83 | 90 | 72 | 64 | 53 | 78 |
| 101 | 134 | 124 | 86 | 104 | 110 | 108 |
| 98 | 106 | 68 | 92 | 120 | 75 | 94 |
| | | | | | | |
| 95 | 86 | 74 | 102 | 98 | 96 | 112 |
| 104 | 99 | 105 | 110 | 100 | 96 | 114 |
| 86 | 92 | 86 | 90 | 106 | 138 | 92 |
| | | | | | | |
| 94 | 86 | 108 | 113 | 105 | 98 | 90 |
| 116 | 93 | 104 | 98 | 78 | 101 | 120 |
| 89 | 86 | 109 | 82 | 110 | 94 | 72 |

| | *Row 8* | | | *Row 10* | |
|---|---|---|---|---|---|
| | Mean | 90.15789474 | Mean | 85.625 | |
| | Standard Error | 2.469306983 | Standard Error | 2.0282263 | |
| | Median | 90 | Median | 85 | |
| | Mode | 95 | Mode | 93 | |
| | Standard Deviatio | 15.22183055 | Standard Deviatio | 12.82762944 | |
| | Variance | 231.7041252 | Variance | 164.5480769 | |
| | Kurtosis | -0.364279745 | Kurtosis | -0.11378204 | |
| | Skewness | 0.052519368 | Skewness | -0.323838083 | |
| | Range | 63 | Range | 57 | |
| | Minimum | 60 | Minimum | 53 | |
| | Maximum | 123 | Maximum | 110 | |
| | Sum | 3426 | Sum | 3425 | |
| | Count | 38 | Count | 40 | |

| 22 | 23 | 24 | 25 | 26 | 27 | 28 | 29 |
|---|---|---|---|---|---|---|---|
| 60 | 67 | 45 | 63 | 48 | 87 | 68 | 78 |
| 69 | 45 | 84 | 67 | 82 | 66 | 42 | 71 |
| 68 | 76 | 94 | 106 | 73 | 104 | 108 | 63 |
| | | | | | | | |
| 80 | 100 | 84 | 80 | 79 | 94 | 96 | 90 |
| 80 | 100 | 89 | 97 | 93 | 106 | 96 | 86 |
| 77 | 75 | 88 | 69 | 77 | 83 | 88 | 101 |
| | | | | | | | |
| 84 | 61 | 94 | 95 | 76 | 77 | 81 | 105 |
| 96 | 116 | 85 | 91 | 116 | 120 | 111 | 104 |
| 80 | 94 | 106 | 84 | 91 | 104 | 106 | 104 |
| | | | | | | | |
| 100 | 105 | 96 | 95 | 90 | 90 | 110 | 106 |
| 102 | 118 | 102 | 98 | 128 | 94 | 111 | 108 |
| 122 | 110 | 82 | 106 | 68 | 98 | 108 | 99 |
| | | | | | | | |
| 86 | 91 | 72 | 95 | 104 | 92 | 102 | 99 |
| 96 | 89 | 90 | 100 | 75 | 65 | 66 | 94 |
| 82 | 96 | 103 | 111 | 89 | 133 | 89 | 84 |

| *Row 11* | | | *Row 12* | | | *Row 14* | |
|---|---|---|---|---|---|---|---|
| | | | | | | | |
| Mean | 103.275 | | Mean | 96.225 | | Mean | 98.175 |
| Standard Error | 2.794451347 | | Standard Error | 2.189862278 | | Standard Error | 1.717141147 |
| Median | 104 | | Median | 96 | | Median | 97.5 |
| Mode | 104 | | Mode | 94 | | Mode | 90 |
| Standard Deviatio | 17.67366214 | | Standard Deviatio | 13.84990512 | | Standard Deviatio | 10.86015418 |
| Variance | 312.3583333 | | Variance | 191.8198718 | | Variance | 117.9429487 |
| Kurtosis | -0.654656685 | | Kurtosis | -0.462738019 | | Kurtosis | -0.626304424 |
| Skewness | -0.122687796 | | Skewness | -0.242497735 | | Skewness | 0.053183462 |
| Range | 71 | | Range | 52 | | Range | 46 |
| Minimum | 67 | | Minimum | 68 | | Minimum | 74 |
| Maximum | 138 | | Maximum | 120 | | Maximum | 120 |
| Sum | 4131 | | Sum | 3849 | | Sum | 3927 |
| Count | 40 | | Count | 40 | | Count | 40 |

| 30 | 31 | 32 | 33 | 34 | 35 | 36 |
|---|---|---|---|---|---|---|
| 78 | 58 | 80 | 80 | 68 | 58 | 49 |
| 64 | 90 | 91 | 104 | 111 | 56 | 88 |
| 78 | 91 | 84 | 87 | 69 | 69 | 69 |
| | | | | | | |
| 89 | 405 | 81 | 70 | 82 | 90 | 95 |
| 95 | 98 | 112 | 85 | 79 | 110 | 100 |
| 90 | 70 | 82 | 112 | 83 | 105 | 66 |
| | | | | | | |
| 93 | 93 | 70 | 90 | 100 | 78 | 93 |
| 105 | 96 | 114 | 67 | 104 | 120 | 92 |
| 94 | 116 | 120 | 84 | 96 | 86 | 80 |
| | | | | | | |
| 89 | 97 | 89 | 84 | 104 | 100 | 117 |
| 90 | 112 | 100 | 106 | 106 | 107 | 100 |
| 104 | 82 | 108 | 116 | 113 | 87 | 89 |
| | | | | | | |
| 100 | 103 | 86 | 85 | 116 | 92 | 105 |
| 110 | 82 | 95 | 100 | 106 | 80 | 78 |
| 85 | 94 | 120 | 90 | 108 | 88 | 100 |
| | | | | | | |
| | *Row 15* | | | *Row 16* | | |
| | | | | | | |
| | Mean | 102.175 | | Mean | 99.925 | |
| | Standard Error | 1.491380147 | | Standard Error | 2.292794594 | |
| | Median | 101 | | Median | 103 | |
| | Mode | 90 | | Mode | 106 | |
| | Standard Deviatio | 9.432316245 | | Standard Deviatio | 14.50090625 | |
| | Variance | 88.96858974 | | Variance | 210.2762821 | |
| | Kurtosis | 0.077913442 | | Kurtosis | 0.288557825 | |
| | Skewness | 0.439126563 | | Skewness | -0.001379171 | |
| | Range | 43 | | Range | 70 | |
| | Minimum | 85 | | Minimum | 68 | |
| | Maximum | 128 | | Maximum | 138 | |
| | Sum | 4087 | | Sum | 3997 | |
| | Count | 40 | | Count | 40 | |

| 37 | 38 | 39 | 40 | | | |
|---|---|---|---|---|---|---|
| 85 | 82 | 56 | 54 | | | |
| 68 | 68 | 79 | 63 | | | |
| 71 | 77 | 89 | 76 | | | |
| | | | | | | |
| 86 | 72 | 81 | 80 | | | |
| 117 | 82 | 91 | 68 | | | |
| 94 | 90 | 80 | 84 | | | |
| | | | | | | |
| 93 | 78 | 83 | 72 | | | |
| 128 | 112 | 70 | 84 | | | |
| 96 | 96 | 112 | 112 | | | |
| | | | | | | |
| 108 | 84 | 106 | 99 | | | |
| 90 | 96 | 85 | 110 | | | |
| 105 | 70 | 104 | 104 | | | |
| | | | | | | |
| 74 | 112 | 120 | 86 | | | |
| 96 | 92 | 88 | 102 | | | |
| 80 | 112 | 105 | 93 | | | |
| | | | | | | |
| *Row 18* | | | *Row 19* | | *Row 20* | |
| | | | | | | |
| Mean | 96.225 | | Mean | 91.375 | Mean | 96.9 |
| Standard Error | 2.166909503 | | Standard Error | 3.238923432 | Standard Error | 2.359867445 |
| Median | 94.5 | | Median | 94.5 | Median | 94 |
| Mode | 86 | | Mode | 100 | Mode | 94 |
| Standard Deviatio | 13.70473903 | | Standard Deviatio | 20.48475043 | Standard Deviatio | 14.92511221 |
| Variance | 187.8198718 | | Variance | 419.625 | Variance | 222.7589744 |
| Kurtosis | -0.647183513 | | Kurtosis | 0.822900877 | Kurtosis | 0.068112677 |
| Skewness | -0.073461208 | | Skewness | -0.183937501 | Skewness | 0.767690696 |
| Range | 54 | | Range | 102 | Range | 62 |
| Minimum | 66 | | Minimum | 44 | Minimum | 72 |
| Maximum | 120 | | Maximum | 146 | Maximum | 134 |
| Sum | 3849 | | Sum | 3655 | Sum | 3876 |
| Count | 40 | | Count | 40 | Count | 40 |

| os | ns | | *B3* | | *C3* | |
|---|---|---|---|---|---|---|
| 3425 | 3849 | | | | | |
| 4131 | 3675 | | Mean | 3801.66667 | Mean | 3800 |
| 3849 | 3876 | | Standard Error | 205.174181 | Standard Error | 62.984125 |
| 3927 | 3456 | | Median | 3849 | Median | 3849 |
| 4087 | 3715 | | Mode | #N/A | Mode | #N/A |
| 3997 | 3583 | | Standard Devia | 355.372105 | Standard Devia | 109.091705 |
| | | | Variance | 126289.333 | Variance | 11901 |
| | | | Kurtosis | #DIV/0! | Kurtosis | #DIV/0! |
| | | | Skewness | -0.58873855 | Skewness | -1.61345559 |
| | | | Range | 706 | Range | 201 |
| | | | Minimum | 3425 | Minimum | 3675 |
| Mean | 3801.66667 | | Maximum | 4131 | Maximum | 3876 |
| Standard Devia | 355.372105 | | Sum | 11405 | Sum | 11400 |
| Mean | 3800 | | Count | 3 | Count | 3 |
| Standard Devia | 109.091705 | | *B6* | | *C6* | |
| Mean | 4003.66667 | | | | | |
| Standard Devia | 80.2080628 | | Mean | 4003.66667 | Mean | 3584.66667 |
| Mean | 3584.66667 | | Standard Error | 46.3081466 | Standard Error | 74.7715038 |
| Standard Devia | 129.508044 | | Median | 3997 | Median | 3583 |
| | | | Mode | #N/A | Mode | #N/A |
| | | | Standard Devia | 80.2080628 | Standard Devia | 129.508044 |
| | | | Variance | 6433.33333 | Variance | 16772.3333 |
| | | | Kurtosis | #DIV/0! | Kurtosis | #DIV/0! |
| | | | Skewness | 0.37144328 | Skewness | 0.05790187 |
| | | | Range | 160 | Range | 259 |
| | | | Minimum | 3927 | Minimum | 3456 |
| | | | Maximum | 4087 | Maximum | 3715 |
| | | | Sum | 12011 | Sum | 10754 |
| | | | Count | 3 | Count | 3 |

# APPENDIX D

Microbial Identification and Mucilage Cover

Patrick Lucy/University of Victoria
P.O. #35446 - Sample #W0342

| Sample | CFU/15.24 $cm^2$ | Organisms Found |
|---|---|---|
| Slide #1 | 1,280,000 | Brevibacterium acetylicum -habitat not known, likely soil/water |
| | 480,000 | Vibrio fluvialis II -water |
| | 50,000 | Vibrio ml migripulchritudo -water |
| | 400,000 | Staphylococcus sp -skin |
| | 10,000 | Coryneform -background |
| | 20,000 | Alternaria -fungi |
| | 10,000 | Cylindrocarpon -fungi |
| Slide #2 | 200,000 | Brevibacterium acetylicum -habitat not known, likely soil/water |
| | 90,000 | Staphylococcus warneri -skin |
| | 290,000 | Staphylococcus saprophyticus -skin |
| | 10,000 | Brochothrix sp -soil |
| | 120,000 | Vibrio fluvialis II -water |
| | 200,000 | Staphylococcus cohnii -human skin |
| | 20,000 | Pseudomonas sp -soil, water |
| | 40,000 | Coryneform -background |
| | 20,000 | Staphylococcus intermedius -nasal membranes |
| | 10,000 | Alternaria -fungi |
| Slide #0 | 310,000 | Brevibacterium acetylicum -habitat not known, likely soil/water |
| | 40,000 | Staphylococcus warneri -skin |
| | 10,000 | Brochothrix sp -soil |
| | 20,000 | Vibrio fluvialis II -water |
| | 10,000 | Staphylococcus cohnii -human skin |
| | 30,000 | Brevibacterium rufescens -soil, water |
| | 20,000 | Pseudomonas sp -soil, water |
| | 10,000 | Coryneform -background |
| H20 | 430,000 | Brevibacterium acetylicum -habitat not known, likely soil/water |
| | 96,000 | Vibrio fluvialis II -water |
| | 14,000 | Enterobacter cloacae -environment, food |
| | 10,000 | Pseudomonas sp -soil, water |
| | 2,000 | Staphylococcus intermedius -nasal membranes |

[redacted]
Microbiologist

Patrick Lucy/University of Victoria
P.O. #35446 - Sample #WO342

100 units = 0.39mm

| | Units | #/Field | |
|---|---|---|---|
| Slide #1 | 20 | 3 | |
| | 1 | 16 | |
| | 20 | 5 | |
| | (2 x 25) | 7 | |
| | 1 | 4 | |
| | 20 | 8 | |
| | 1 | 3 | |
| | 20 | 4 | |
| | 1 | 22 | |
| | 20 | 3 | |
| | (1.5 x 230) | 1 | 10% covered |
| | (20 x 1) | 1 | |
| | 20 | 13 | |
| | 1 | 20 | 80% cover of musalage |
| | .5 | 50 | |
| Slide #2 | 20 | 2 | |
| | 1 | 56 | |
| | 1 | 490 | |
| | 20 | 3 | |
| | 1 | 24 | |
| | 20 | 8 | |
| | 1 | 15 | |
| | 20 | 1 | |
| | 1 | 154 | |
| | 20 | 2 | |
| | 1 | 47 | |
| | 20 | 7 | |
| | (30 x 1) | 1 | |
| | (8 x 42) | 1 | |

...3

Patrick Lucy/University of Victoria
P.O. #35446 - Sample #W0342

| | | |
|---|---|---|
| Slide #3 | (.5 x .1) | 302 |
| | 1 | 4 |
| | (.5-1 x 20) | 2 |
| | (22 x 8) | 1 |
| | (.5 x .1) | 8 |
| | 1 | 13 |
| | (.3) | 100 |
| | 20 | 1 |
| | (5 x 6) | 1 |
| | (15 x 4) | 3 |
| | 18 x 15 | 1 |
| | 20 | 2 |
| | 1 | 10 |

Client: Patick Lucy/University of Victoria
Sample No. WQ506
P.O. #39342

| Sample | CFU/100cc | Organisms Identified |
|---|---|---|
| A | 32,000 | Brevibacterium ml acetylicum -habitat not known |
| | 8,000 | Cellulomonas gelida -soil |
| | 4,000 | Listeria sp -water/soil |
| | 2,000 | Penicillium sp -fungus |
| B | 2,000 | Rhizopus sp -fungus |
| | 1,200 | Cellulomonas gelida -soil |
| | 54,000 | Listeria sp -water/soil |
| | 16 | Microbacterium ml roseus -water |
| | 4,800 | Bacillus sp -background, soil |
| C | 4,000 | Penicillium sp -fungus |
| | 8,000 | Cellulomonas gelida -soil |
| | 16,000 | Listeria sp -water/soil |
| | 4,000 | Bacillus sp -background |
| | 2,000 | Kurthia gibsonii -soil/surface water |
| | 32 | Sphaerotilus sp -slow running FW |
| D | 2,000 | Penicillium sp -fungus |
| | 26,000 | Cellulomonas gelida -soil |
| | 32,000 | Listeria sp -water/soil |
| | 10,000 | Bacillus sp -background |
| | 2,000 | Kurthia gibsonii -soil/surface water |
| E | 20,000 | Trichoderma viride -fungus |
| | 10,000 | Cellulomonas gelida -soil |
| | 20,000 | Listeria sp -water/soil |
| | 2,000 | Bacillus sp -background |
| F | 24,000 | Trichoderma viride -fungus |
| | 20,000 | Cellulomonas gelida -soil |
| | 50,000 | Listeria sp -water/soil |
| | 2,000 | Derxia gummosa -soil |

...2

100 units = 0.39mm
slides in 325mL H20
slides = 50.8x76.2x1mm

| | Units | #/0.25cc | | Units | #/0.25cc |
|---|---|---|---|---|---|
| Slide A | (3x10) | 4 | Slide B | 14x11 | 1 |
| | 3x15 | 1 | | 5x8 | 1 |
| | 4x18 | 2 | | 4x7 | 1 |
| | 5x18 | 4 | | 10x8 | 1 |
| | 4x10 | 2 | | 5x25 | 1 |
| | 4x15 | 15 | | 5x5 | 2 |
| | 4x20 | 3 | | 3x3 | 2 |
| | 5x20 | 1 | | 7x7 | 1 |
| | 8x15 | 3 | | 3x110 | 1 |
| | 8x18 | 2 | | 10x7 | 77 |
| | 5x22 | 1 | | 2x8 | 1 |
| | 5x56 | 1 | | 2x18 | 1 |
| | 4x26 | 1 | | 2x7 | 1 |
| | 2x34 | 2 | | 1x12 | 1 |
| | 3x40 | 1 | | 2x10 | 1 |
| | 2x15 | 3 | | 2x11 | 1 |
| | 2x10 | 5 | | 2x12 | 1 |
| | 3x10 | 1 | | 2x20 | 1 |
| | 2x2 | 5 | | 3x8 | 3 |
| | 4x8 | 1 | | 2x10 | 1 |
| | 4x15 | 2 | | 4x8 | 1 |
| | 5x5 | 1 | | 2x4 | 2 |
| | 4x6 | 1 | | 12x4 | 17 |
| | 2x35 | 1 | | 12x6 | 6 |
| | 2x2 | 8 | | 4x62 | 1 |

| | Units | #/0.25cc | | Units | #/0.25cc |
|---|---|---|---|---|---|
| Slide C | 2x26 | 1 | Slide D | 4x11 | 1 |
| | 2x30 | 1 | | 3x5 | 2 |
| | 2x20 | 1 | | 4x6 | 1 |
| | 2x22 | 1 | | 2x2 | 5 |
| | 7x10 | 20 | | 20x2 | 1 |
| | 4x15 | 10 | | 10x2 | 2 |
| | 4x12 | 6 | | 10x3 | 1 |
| | 8x12 | 7 | | 8x2 | 1 |
| | 2x6 | 2 | | 12x3 | 1 |
| | 2x8 | 2 | | 5x20 | 2 |
| | 4x8 | 1 | | 8x12 | 2 |
| | 8x8 | 1 | | 4x15 | 3 |
| | 6x76 | 1 | | 2x4 | 1 |
| | 5x9 | 1 | | 1x68 | 1 |
| | 11x5 | 1 | | 1x22 | 1 |
| | 6x6 | 1 | | | |
| | 3x3 | 1 | | | |
| | 4x3 | 1 | | | |
| | 2x2 | 3 | | | |
| | 8x24 | 1 | | | |
| | 8x304 | 1 (Nematode) | | | |

100 units = 0.39mm
slides in 325mL H20
slides = 50.8x76.2x1mm

| | Units | #/0.25cc | | Units | #/0.25cc |
|---|---|---|---|---|---|
| Slide E | 7x10 | 50 | Slide F | 7x10 | 48 |
| | 4x15 | 1 | | 1x10 | 5 |
| | 8x15 | 1 | | 1x6 | 3 |
| | 6x15 | 13 | | 3x9 | 10 |
| | 4x14 | 19 | | 1x16 | 2 |
| | 3x3 | 2 | | 2x31 | 1 |
| | 5x5 | 1 | | 2x24 | 2 |
| | 4x3 | 1 | | 1.5x20 | 1 |
| | 2x10 | 6 | | 2x46 | 1 |
| | 2x9 | 1 | | 4x84 | 1 |
| | 5x10 | 2 | | 4x18 | 14 |
| | 3x56 | 1 | | 6x18 | 2 |
| | 4x8 | 1 | | 4x8 | 7 |
| | 2x16 | 1 | | 3x8 | 17 |
| | 3x16 | 1 | | 6x8 | 4 |
| | 5x26 | 1 | | 2x1 | 3 |
| | 1x18 | 1 | | 6x4 | 1 |
| | 1x12 | 1 | | 6x8 | 1 |
| | 4x112 | 1 (Nematode) | | 2x36 | 1 |

[redacted]
Microbiologist

# VITA

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Given Names: William Patrick
Place of Birth: Vancouver, B.C.
Date of Birth: June 11 1950

Educational Institutions Attended:

| | |
|---|---|
| Langara College | 1972 to 1978 |
| University of Victoria | 1978 to 1993 |
| Simon Fraser University | 1979 |

Degrees Awarded:

| | | |
|---|---|---|
| B.Sc. | University of Victoria | 1981 |
| B.A. | University of Victoria | 1990 |

Honours and Awards:

| | |
|---|---|
| The Howard English Bursary | 1980 |
| British Columbia Government JobTrac Supplement | 1987 |
| The Amelia Leith Memorial Award | 1987 |
| British Columbia Post-Secondary Scholarship | 1987 |
| Graduate Entrance Scholarship, M.B.A Program, Dalhousie University | 1988 |
| British Columbia Government JobTrac Supplement | 1988 |
| Natural Sciences and Engineering Research Council of Canada Postgraduate Scholarship | 1989 |
| Presidents Research Scholarship | 1989 |
| Graduate Teaching Fellowship | 1991/92 |

Publications:
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Title of Thesis: Periphyton functional and structural response, within semi-natural surrogate streams, to artificially induced water quality perturbations

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William Patrick Lucey

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