Water Qual. Res. J. Canada, 2000
Volume 35, No. 1, 73-94

Slow-Release Fertilizer for Rehabilitating Oligotrophic Streams: a Physical Characterization

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A combination of habitat degradation and overharvesting of anadromous salmonids in many of B.C.'s oligotrophic watersheds has prompted the addition of inorganic nutrients to streams, which increases autotrophic production and aids in the restoration of salmonid production. A new slow-release fertilizer (7-40-0, N-P_2O_5-K_2O, percent by weight) was examined to determine its phosphate (PO_4^{3-}) release rates using laboratory and field trough experiments. A series of indoor trough experiments indicated that the fertilizer pellet dissolution rate (0.393 g · days$^{-0.401}$) was independent of the experimental range of water velocity (0.03–0.30 m · s^{-1}), fertilizer pellet size (2–9 g) and water temperature (8–14.5°C). Resulting phosphate additions (0.5–5 µg P · L^{-1}) in outdoor trough experiments increased periphyton biomass and altered the dominance pattern of periphytic diatoms. An optimal phosphate concentration for periphyton biomass was achieved with calculated 3.0 µg P · L^{-1} phosphate additions from May to June. In June to July, periphyton biomass increased proportionately to fertilizer additions. Saturation of the relative specific growth rate of the diatom community occurred with 1.0 µg P · L^{-1} phosphate additions. Nitrogen analysis was not conducted since inorganic nitrogen is typically available in non-limiting concentrations (i.e., >50 µg · L^{-1} dissolved inorganic nitrogen) in the majority of B.C.'s oligotrophic salmonid streams. These studies indicated that slow-release fertilizer may be effective in stimulating autotrophic production and restoring salmonid production in nutrient deficient streams.

Key words: **stream rehabilitation, anadromous salmonids, slow-release fertilizer, nitrogen, phosphorus, nutrients, periphyton**

Introduction

Human activities during the past century, particularly intensive forest harvesting practices, fishing, urbanization, industrialization and impoundment construction, have negatively impacted the health of British Columbia's numerous wild salmonid stocks (Slaney et al. 1996). Overharvesting and alteration of salmonid habitat reduces the return of spawning adults which naturally fertilize streams for their progeny

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(Larkin and Slaney 1996). Spawning adults transport marine-derived nutrients to freshwater habitats in the form of excretion, gametes and carcass decomposition which are effectively recycled to maintain stream productivity (Bilby et al. 1996).

Oligotrophic stream conditions, both human induced (i.e., cultural oligotrophication [Ney 1996, Stockner and MacIssac 1996]) and naturally occurring in granitic coastal systems, can become adequately fertile with low-level addition of inorganic nutrients. Nutrient addition increases primary production (algal growth) at the base of the food chain, stimulates aquatic insect growth, provides food for juvenile salmonids and subsequently increases salmonid growth (Ward and Slaney 1988), survival and return to their spawning grounds (Johnston et al. 1990).

A number of studies have demonstrated that low-level additions of nitrogen and phosphorus to nutrient-deficient streams result in increased autotrophic production, mainly diatoms, which are an excellent food source for several species of stream invertebrates (Stockner and Shortreed 1978; Peterson et al. 1985; Perrin et al. 1987; Johnston et al. 1990). Fertilization effects higher in the lotic food chain were examined by Mundie et al. (1991), Gregory (1983) and Quamme (1994), who determined that the amount of periphyton available influenced the numbers of aquatic herbivores. Lastly, several studies concluded that fish density and growth correlate with nutrient status and food supply in a stream (Slaney and Northcote 1974; Mason 1976; Wilzbach 1985; Johnston et al. 1990; Deegan and Peterson 1992; Slaney and Ward 1993).

Bothwell (1988, 1989) observed that the concentration of phosphorus reported to produce dense periphyton accumulations in rivers varies in the literature (Stockner and Shortreed 1978; Horner and Welch 1981; Peterson et al. 1983, 1985; Perrin et al. 1987; Johnston et al. 1990; Slaney et al. 1994). Bothwell noted this discrepancy was partly due to equating aquatic phosphate concentrations required to saturate cellular growth rates with concentrations required to maximize growth rates of an entire benthic community. In other words, denser periphyton communities require higher phosphate concentrations to saturate and maintain growth of the whole community (Bothwell 1989) and cannot be compared to requirements at the cellular level. To clarify the issue, Bothwell (1988, 1989) studied periphytic diatom communities in the South Thompson River (B.C.) to determine areal relative specific growth rates as a function of soluble reactive phosphorus concentration. Relative specific growth rates are expressed as a proportion of the maximum growth rate in order to factor out physical variations, primarily temperature (Bothwell 1988). Bothwell (1989) and Quamme (1994) observed that periphyton growth rates were nearly constant throughout the year with 0.3 to 2.5 $\mu g\ P \cdot L^{-1}$ of orthophosphate maximizing the specific cellular growth rates of periphyton. Areal periphyton biomass was further stimulated by higher phosphorus concentrations (Bothwell 1989), but sloughing occurred when phosphate concentrations were $>10\ \mu g \cdot L^{-1}$.

Addition of limiting nutrients causes increases in periphytic algal

abundance as well as possible changes in species composition, depending on the amount of nutrient added. Reported algal community changes include increased dominance by certain species, reduced species diversity, loss of rare species and higher biomass (Miller et al. 1992). Although the effects of nutrient additions are reversible, streams undergoing long-term fertilization treatment and habitat rehabilitation are expected to become self-sustaining by natural fertilization from riparian vegetation and anadromous salmonid escapement. Establishing an optimal nutrient addition regime requires a knowledge of stream chemistry, concentrations of limiting nutrients and a fertilizer product which releases nutrients at a predictable rate. Characteristics of fertilizer pellet dissolution play a vital role in establishing the nutrient status of a stream and time of fertilizer application. A logical fertilizer application frequency would be once yearly in springtime, with nutrients releasing continually over a period of 3 to 4 months, depending on stream-specific nutrient requirements.

This is the second of two studies that assessed the performance of a new slow-release pellet fertilizer for increasing nitrogen and phosphorus concentrations of streams. The first study (Sterling and Hall, in prep.) examined phosphorus chemistry in waters of varying pH, alkalinity, iron and humic material. Physical characteristics of the pellet fertilizer, specifically factors affecting fertilizer dissolution and algal growth rates and biomass, are examined in this paper. Three factors that are hypothesized to influence fertilizer pellet dissolution rates are temperature, stream velocity and pellet size (surface area). The resulting phosphorus concentrations and their effect on algal growth rates, biomass and community structure are also examined.

Methodology

Fertilizer Characteristics

The slow-release fertilizer pellets used in these experiments were developed in 1994–95 by IMC Vigoro Inc. (Winter Haven, FL) and contained 14% magnesium, 7% nitrogen and 40% P_2O_5 by weight existing as the $MgNH_4PO_4 \cdot H_2O$ compound. The fertilizer was compressed into ~9 g pellets with an unpolymerized Saran™-like binder called Daratak® XB-3631 (a vinylidene chloride-acrylic acid-2-ethylhexyl acrylate polymer) to form the pellet. This provided a slow release of nutrients while dissolving in aqueous conditions. A constant nitrogen and phosphorus composition of the fertilizer pellet during dissolution was verified by Mouldey and Ashley (1996).

Indoor Trough Studies

Indoor channels located at the B.C. Ministry of Environment's Fraser Valley Trout Hatchery were used to determine the release rates of fertilizer pellets (~2, 6 and 9 g sizes) under various water temperatures (8, 10 and 14.5°C), velocities (0.03, 0.15 and 0.30 $m \cdot s^{-1}$) and flow rates (0.011 to 0.11

$m^3 \cdot s^{-1}$). Water velocities were measured with a Marsh McBirney, Inc. Model 2000 portable flowmeter. Custom-made Plexiglass troughs (Fig. 1), modeled after ones used by Bothwell (1983, 1988), were equipped with various dams at the beginning and/or end of the troughs to reduce turbulence. There were a total of 10 troughs used for the experiment, including a replicate using 10°C and 0.15 $m \cdot s^{-1}$ conditions. Each trough contained 13 to 14 galvanized wires strung with 9 fertilizer pellets (3 of each size) per wire.

Three water temperatures, 6.5, 9.5 and 17.5°C, were available, using the hatchery heating and cooling capabilities, and had a maximum flow limit, depending on water demand at the hatchery. Appropriate water

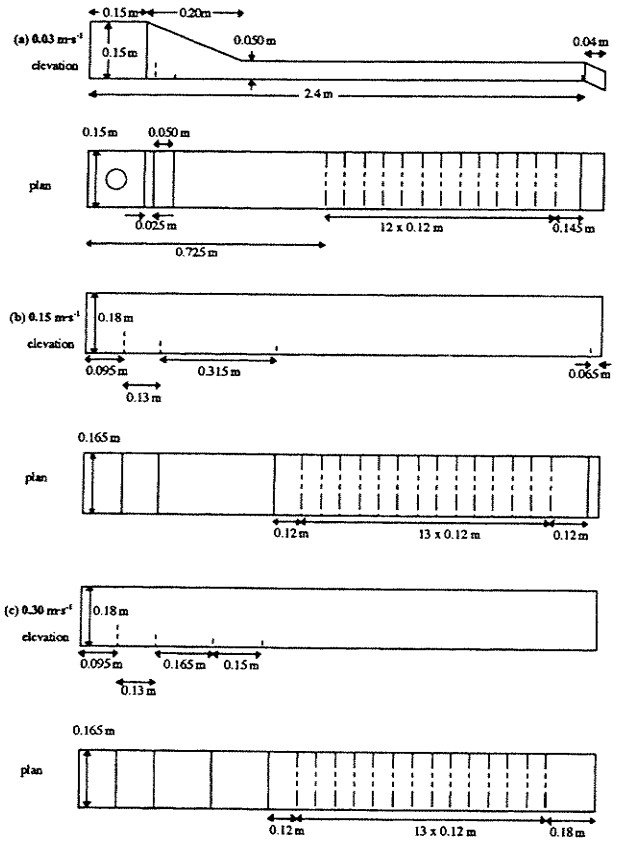

Fig. 1. Dimensions of Plexiglass troughs for the three velocities: (a) 0.03 $m \cdot s^{-1}$, (b) 0.15 $m \cdot s^{-1}$ and (c) 0.30 $m \cdot s^{-1}$. The vertical dotted lines in the elevation views represent dams of heights (left to right) to give non-turbulent flow through the troughs: (a) 0.025 m, 0.005 m and 0.015 m; (b) 0.06 m, 0.045 m, 0.015 m and 0.015 m; (c) 0.06 m, 0.045 m, 0.015 m and 0.015 m. There were no dams at the end of troughs for 0.30 $m \cdot s^{-1}$ velocities. Uneven dashed lines in the plan views represent the rows of fertilizer pellets. Water flows left to right.

temperatures and velocities that simulated B.C. stream conditions were obtained by mixing water of various temperatures in head tank sources as well as adjusting the trough slopes. Temperature gradients occasionally occurred across the troughs and were eliminated by positioning the water hoses. Selected water chemistry measurements for the experimental water supply are shown in Table 1.

Table 1. Selected water chemistry parameters of Fraser Valley Trout Hatchery well No. 4 and head tank

Parameter	Average value[a]
pH	8.0
Hardness	160 mg · L^{-1} as $CaCO_3$
Alkalinity at pH 4.5	88 mg · L^{-1} as $CaCO_3$
Dissolved ammonia	7 μg N · L^{-1}
Total phosphorus	<40 μg · L^{-1}
Dissolved phosphorus	<40 μg · L^{-1}
Total and dissolved calcium	49 mg · L^{-1}
Total iron	<0.05 mg · L^{-1}
Dissolved iron	<0.003 mg · L^{-1}
Total and dissolved magnesium	9 mg · L^{-1}

[a] Symbol "<" indicates the measurement was below the analytical detection limit.

The pellets were cut into ~2, 6 and 9 g sizes, and then drilled, rinsed, oven dried, desiccated, individually weighed and strung, separated by 5 mm diameter plastic beads, onto 18-gauge galvanized wire. Two to three beads were strung on each end of the wire to distance the pellets from the slower moving water at the trough edges. The wires were numbered and fastened to the trough walls. Each sampling day, the pellets furthest downstream were removed, oven dried (18 hours at 70°C), desiccated and weighed to determine dissolution rate. Sampling took place over a period of 3 months and occurred 2, 4, 7, 11, 14, 18, 26, 32, 47, 60, 74 and 88 days after the pellets were introduced into the troughs.

A ratio of average weight lost to average initial weight was used as the dependent measure in the statistical analysis to simplify calculations. A stepwise linear regression model determined entry order of the physical variables and indicated their relative importance as predictors of fertilizer pellet weight loss. A regression equation was derived to express average weight lost/average initial weight for an 88 day experimental period for pellets of 2 to 9 g in water of temperatures 8-14.5°C and velocities 0.03 to 0.30 m · s^{-1}.

Outdoor Periphyton Growth Studies

Troughs

Periphyton accrual growth experiments were conducted in two outdoor troughs (9.5 m × 0.30 m × 0.19 m [Fig. 2]) lined with heavy plastic at Cultus Lake's Department of Fisheries and Oceans' Salmon Research Facility. Water was pumped from 4.5 m below the surface of Cultus Lake into 20 L buckets at the trough heads. Rocks contained in the bottom of perforated buckets minimized turbulence caused by inflowing water. An Optic StowAway temperature monitor (Onset WTA08, Hoskin Scientific) was attached to one of the buckets and recorded hourly water temperatures throughout the experiments. Water temperature ranged from 14 to 19°C from May 23 to June 27 during the first experiment, and from 18 to 23°C from June 27 to August 1, 1995, over the course of the second experiment — velocities remained constant at 0.15 m · s^{-1}. Silt accumulated on the bottom of the troughs over the course of both experiments, but none was observed on the raised periphyton blocks.

As in the previous experiment, fertilizer pellets were strung, separated by beads, on galvanized wire and attached to these experimental troughs at various depths in order to evenly distribute nutrients over a short distance. Periphyton blocks were placed close to the north side of

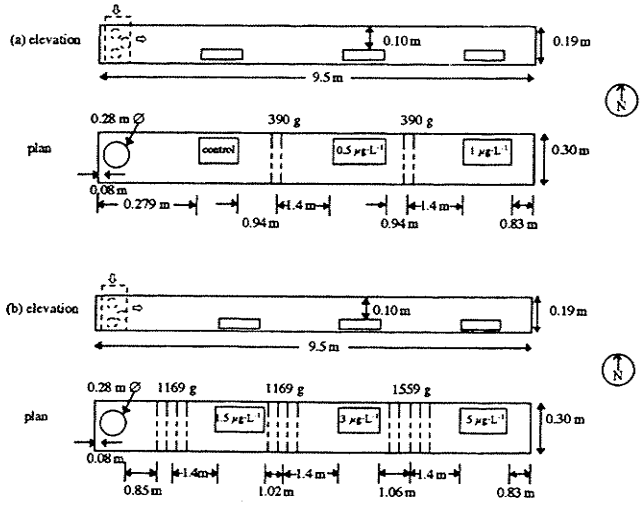

Fig. 2. Outdoor trough dimensions with water flowing from the west at 0.15 m · s^{-1}. Buckets (0.28 m diameter) to capture inflowing water are represented by circles in the plan view. Trough (a) contained periphyton blocks for 0, 0.5 and 1.0 µg P · L^{-1} from fertilizer pellets and trough (b) contained blocks for 1.5, 3.0 and 5.0 µg P · L^{-1} fertilizer addition. Groups of pellets represented by dashed lines and weights were ~8 cm apart. The 1.4 m distances were measured from the centre of the groups of pellets to the blocks.

the troughs to eliminate shading from the south walls. Shading would cause uneven periphyton growth on the Styrofoam surface of the periphyton blocks (Fig. 2). The inflowing water was adjusted to maintain a depth of approximately 0.10 m above the periphyton blocks at a velocity of 0.15 m · s^{-1}. Algal growth on the fertilizer pellets, which would inhibit nutrient release, was discouraged by placing plywood sheets above the pellets to provide shade. In natural streams this shade would be provided by riparian vegetation. Water temperature, velocity and light exposure were consistent for all periphyton blocks and troughs; however, temperature and light levels underwent natural daily and seasonal fluctuations.

Water characteristics

Control and fertilized water samples were analyzed within 24 hours of sampling for nutrient chemistry. Soluble reactive phosphate (SRP) concentrations in ambient and fertilized water were very close to the detection limit based on the limited sensitivity of the SRP analytical method; additionally, the quality control of the laboratory was not adequate. Thus, accurate verification was not possible; however background concentration is estimated at 2 μg P · L^{-1} phosphate. Six sets of samples were taken from May to July 1995. Chemistry of the 4.5 m deep Cultus Lake water source is shown in Table 2.

Periphyton substrates

Artificial periphyton substrates (0.2 m × 0.1 m × 0.4 m) consisted of a 6 mm open-cell Styrofoam sheet strapped to Plexiglass and bolted to a concrete block. Six artificial substrates were placed in the outdoor troughs to monitor each fertilizer addition (0, 0.5, 1.0, 1.5, 3.0 and 5.0 μg P · L^{-1}) to the trough water. Four days after the periphyton blocks were put in the troughs, the Styrofoam sheets were softly brushed horizontally and vertically to avoid patchy colonization by removing excess filamentous algae

Table 2. Cultus Lake water chemistry background levels, May to July 1995

Parameter	μg · L^{-1} [a]
Total organic nitrogen	< 40[b]
Total nitrogen	< 50
NH_3	5
NO_2	< 1
$NO_3 + NO_2$	< 5
SRP	3 ± 3
TP	7 ± 7

[a] Forms of nitrogen expressed as μg · N L^{-1} and forms of phosphorus as μg · P L^{-1}.

[b] Symbol "<" indicates the measurement was below the analytical detection limit.

and evenly distributing the algal cells. Experimental nutrient treatments were applied in two 4-week studies: May 23 to June 23 and June 26 to July 31, 1995. The experiments ended when sloughing of algae from the periphyton substrates was observed.

Fertilizer

The amount of fertilizer pellets required was determined using results from the previous indoor dissolution experiments based on 7.4 $L \cdot s^{-1}$ flowing through each trough. The 14°C and 0.15 $m \cdot s^{-1}$ data were used for the calculations: over the course of 6 weeks (42 days), approximately 20% of the 9 g pellet dissolved, releasing 1.72 g fertilizer and 0.3 g phosphorus. A linear dissolution rate, additive releases and a constant flow (for uniform release of calculated phosphate concentrations) were assumed to exist over the experimental period. In the first trough, upstream of any fertilizer pellets, was a control periphyton block (0 $\mu g\ P \cdot L^{-1}$), followed by two fertilizer sources (390 g of pellets) located upstream of their respective periphyton blocks to provide 0.5 $\mu g\ P \cdot L^{-1}$ and additively 1 $\mu g\ P \cdot L^{-1}$; the second trough required two 1,169 g pellet sources to provide 1.5 $\mu g\ P \cdot L^{-1}$ and additively 3 $\mu g\ P \cdot L^{-1}$ as well as an additional 1,559 g to provide 5 $\mu g\ P \cdot L^{-1}$ (Fig. 2). The total amount of fertilizer used was 4,677 g (~519 pellets).

Periphyton sampling

Periphyton growth was measured as chlorophyll *a* content per unit area of Styrofoam substrate. Periphyton was sampled at 2-week intervals by removing quadruplicate Styrofoam cores from each periphyton block using a 6.2 cm^2 pill bottle as a punch; growth was determined by assessing chlorophyll *a* concentrations in each sample core. After each sampling, the strands of fertilizer pellets were lightly brushed to remove excess algae, and water velocity (0.15 $m \cdot s^{-1}$) was monitored with the Marsh McBirney flowmeter. Periphyton samples were taken from four of six periphyton blocks (0.0, 0.5, 1.5 and 3.0 $\mu g\ P \cdot L^{-1}$) on three occasions (June 19, July 10 and July 17, 1995), preserved in Lugol's solution and analyzed for algal species identification by Dr. Frances Pick (Department of Biology, University of Ottawa).

Sample and data analyses

Chlorophyll *a* ($\mu g \cdot m^{-2}$) was analyzed according to the method outlined in Standard Methods for the Examination of Water and Wastewater (APHA et al. 1992). Extracts were read on a calibrated 10-AU fluorometer (Turner Designs, Sunnyvale, California). Periphyton samples on 6.2 cm^2 Styrofoam cores were frozen in black plastic bags until later (1 to 2 months) fluorometric analysis. Chlorophyll *a* was extracted in 10 to 15 mL of 90% acetone and 10% $MgCO_3$ by sonicating for 15 minutes in cold water, incubating at 4°C overnight, sonicating again for 10 to 15 minutes and then centrifuging for 5 minutes at 2000 rpm in a refrigerated centrifuge.

Cell enumerations were made using the Utermöhl method on a Wild M40 inverted microscope, and cell counts and dimensions were recorded

on a computerized counter. Regression analysis of periphyton accrual rates were computed using chlorophyll *a* concentrations ($\mu g \cdot m^{-2}$) in terms of time, and calculated phosphate (PO_4^{3-}) concentration ($\mu g\ P \cdot L^{-1}$) added to the water.

Results and Discussion

Indoor Trough Fertilizer Solubility Studies

Factors affecting fertilizer dissolution

A stepwise linear regression model was utilized to determine the factors that influenced fertilizer pellet weight loss. The order of variable entry revealed that exposure time accounted for most of the pellet dissolution ($n = 1107$; $P < 0.01$; $r^2 = 0.73$) followed by water velocity, pellet size and water temperature. Although these factors entered the model, they accounted for little of the pellet weight loss ($n = 1107$; $P < 0.01$; $r^2 = 0.79$). No significant interactions were found between any of the variables as indicated by a correlation matrix of regression coefficients. Rehberg (1997 pers. comm., IMC Vigoro Inc., Winter Haven, FL) supported the observation that temperature did not influence dissolution rates of the pellet fertilizer.

The finding that all other variables (temperature, velocity, size) have little influence on fertilizer dissolution simplifies the stream loading calculations — nutrient release is primarily a function of water exposure. This simplification is advantageous for spring and summer fertilization when stream temperature and velocity conditions change rapidly. Therefore, the only effect pellet size (surface area) has on fertilizer dissolution is duration of application, with larger pellets having a longer residence time in the water. Theoretically, size is expected to influence the pellet dissolution since surface area is related to dissolution rate; however, the small range of sizes tested and available for practical application in streams results in a minimal effect of pellet size on dissolution rate.

Fertilizer loading calculations

As exposure time is the primary determiner of pellet weight loss, the 9 g pellet size and 9.7°C water temperature were chosen to determine rate of pellet weight loss (Fig. 3) based on practicality of pellet size and typical spring water temperatures.

A linear regression of log average weight loss/average initial weight versus log time for velocities 0.03, 0.15 and 0.30 $m \cdot s^{-1}$ gave r^2 values of 0.964, 0.965 and 0.983, respectively. A linear regression of average weight ratios for the three velocities resulted in the equation ($r^2 = 0.983$):

$$\log(\text{avg wt lost/avg initial wt}) = 0.401 \cdot \log[\text{time (days)}] - 1.37 \quad (1)$$

Transformation of the equation to determine the pellet weight required for a given length of fertilization is:

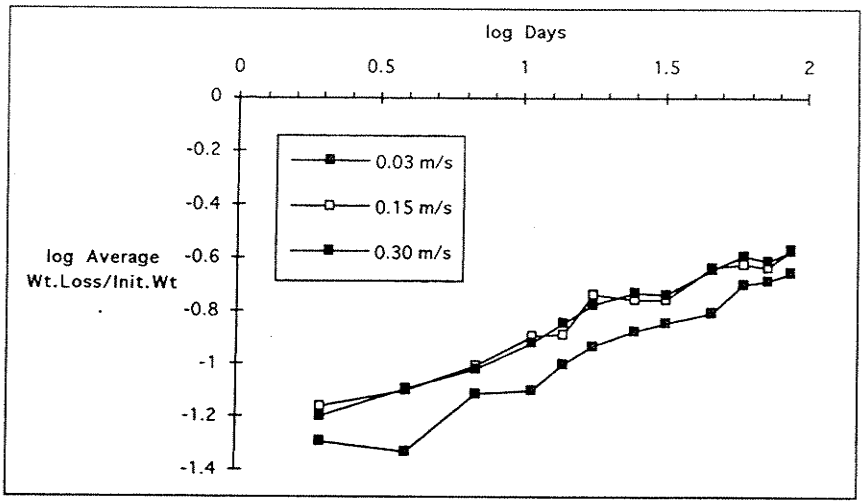

Fig. 3. Log plot of average weight loss over average initial weight (g) over an 88-day period for 9-g pellets exposed to three velocities in 9.7°C water.

$$\text{pellet weight lost} = 0.393 \cdot (\text{days})^{0.401} \quad (2)$$

whereby a weight of 9.21 g (N=355) was substituted as the average initial weight for 9 g pellets. Thus, these pellets dissolve at a rate of 0.393 g · days$^{-0.401}$ when completely exposed, irrespective of water temperature, velocity and pellet size. However, further experiments with a greater range of pellet sizes should be conducted to confirm their lack of influence on dissolution rate.

Extrapolation to field conditions

Conditions in the natural environment may cause slower dissolution rates rather than the maximum weight losses observed in these experiments. For example, pellets often become wedged between rocks and substrate in streams, which limits water exposure, or they are moved to slower flowing regions by high-water velocities. Depending on sunlight exposure and water temperature, some pellets may develop algae and microbial growths on their surface, as observed in this study. These field conditions can prolong the release of nutrients from the fertilizer pellets and need to be considered when calculating fertilizer amounts for optimal stream fertilization. Stream-specific fertilization can be achieved by manufacturing pellets of various release rates and sizes. Further experiments should include a wider temperature and velocity range to represent typical B.C. streams during spring and summer months (6 to 25°C; 0.1 to 2 m · s^{-1}).

Outdoor Trough Periphyton Growth Studies

The previous experiment determined which factors influenced fertil-

izer dissolution rates. These findings were applied to this experiment where periphyton growth as a result of fertilizer addition was considered. Two 4-week studies demonstrated the potential for increasing periphyton growth by introducing various fertilizer concentrations to outdoor channels. Linear and semi-log trends of chlorophyll *a* accrual in response to fertilization are illustrated in Fig. 4a and b and 5a and b, respectively. In both experiments, the periphyton accumulation, measured as chlorophyll *a*, increased exponentially after the first week for all fertilizer treatments and followed with a period of slower growth.

Periphyton growth behavior can be described by Monod's cellular growth rate kinetics: an initial cellular immigration or colonization lag period is followed by exponential growth and finally by a plateau signi-

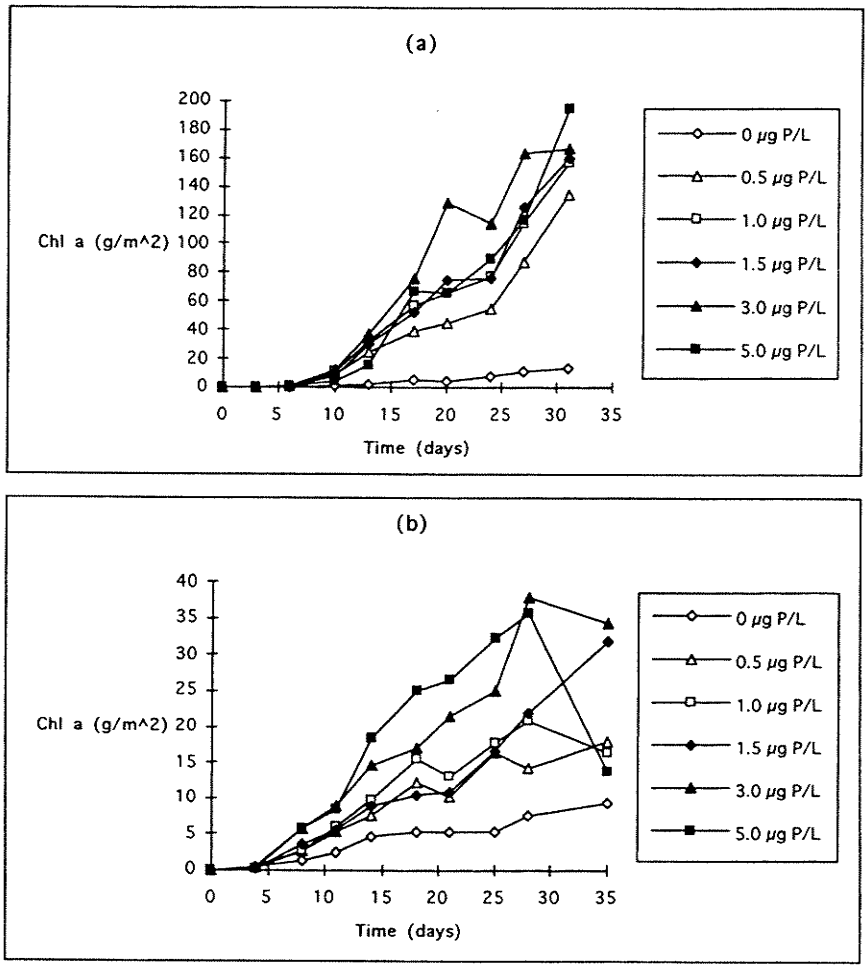

Fig. 4. Chlorophyll *a* accumulation over time for various phosphorus concentrations in experiments (a) May 19–June 23, 1995, and (b) June 26–July 31, 1995.

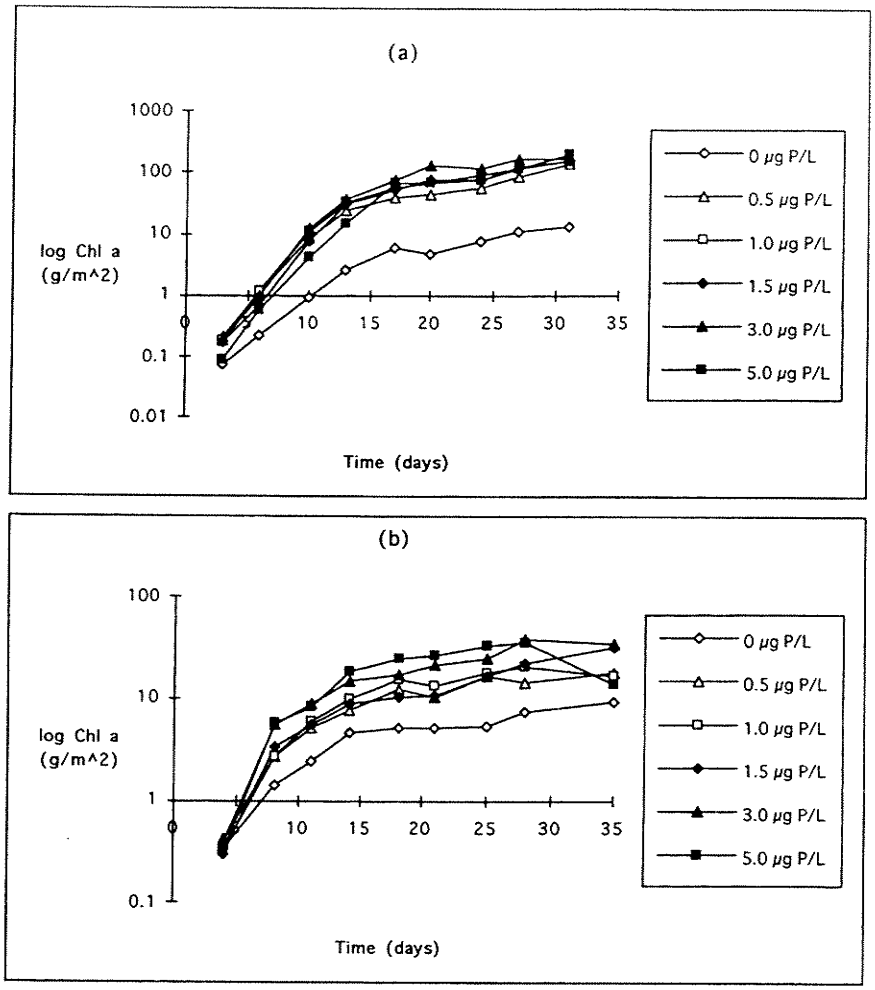

Fig. 5. Semi-log plot of chlorophyll *a* accumulation over time for various phosphorus concentrations in experiments (a) May 19–June 23, 1995, and (b) June 26–July 31, 1995.

fying physical (i.e., light) and/or chemical (i.e., nutrient) limitations (Bothwell 1989; Perrin et al. 1995). However, a biomass plateau and decline, as observed by Bothwell (1989), were not present here due to the short time frame of this experiment (1 versus 2 months).

Periphyton colonization

Colonization data (occurring at <10 days, before periphyton reached an exponential, community controlled growth phase [Bothwell 1988, 1989, 1997 pers. comm., Department of Fisheries and Oceans, Nanaimo, B.C.]) were used to determine the phosphate concentration which maxi-

mizes or saturates algal *cellular* growth rates. These concentrations were calculated by expressing growth rate, k, as a proportion of the maximum rate, k_{max}, from each experiment to give the relative specific growth rate (k/k_{max}). This ratio accounted for variations in temperature, light and other physical factors (Bothwell 1985). A plot of k/k_{max} versus added phosphate concentrations (Fig. 6) shows periphytic algal growth saturated with ~1 μg · P L^{-1} additions for the early summer experiment (May 26 to June 5). The concentrations of optimal and saturated phosphate addition could not be determined for the mid-summer (June 26 to July 31) experiment because of rapid colonization, but the concentrations were expected to be similar for diatom communities throughout the summer season.

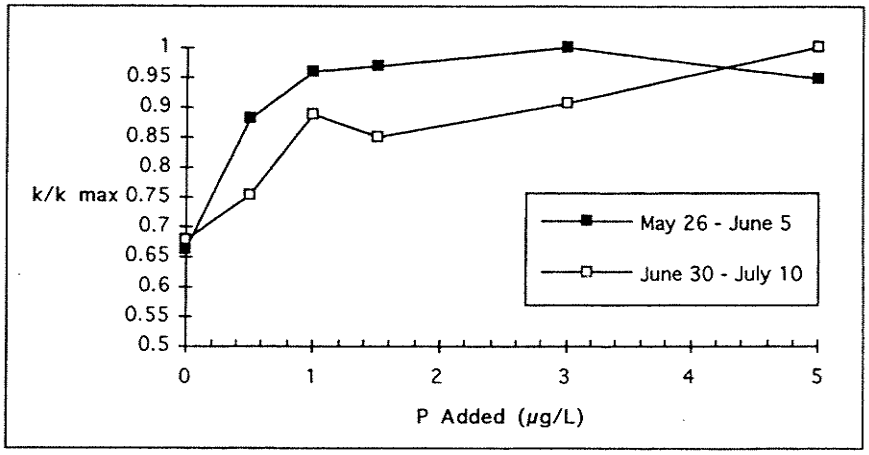

Fig. 6. Normalized periphyton growth rates in response to phosphate addition for the first 10 days of each experiment.

The phosphate saturation concentration for relative specific growth rates of the diatom community, obtained with the calculated addition of 1 μg P · L^{-1}, corresponds to saturation levels with ~0.2 to 0.5 μg P · L^{-1} additions found in the Thompson River, B.C. (Bothwell 1988). Perrin et al. (1995) reported similar saturation concentrations in the Athabasca River, Alberta, with additions between 0.2 and 1.0 μg P · L^{-1}.

Small differences between experiments could be attributed to variations in algal species and/or variations in ambient SRP concentrations. For example, ambient SRP concentrations averaged 3 μg P · L^{-1} from the end of May to the end of July for this experiment, averaged 2.3 μg P · L^{-1} in the South Thompson River, B.C. over the same time frame (Bothwell 1988), and averaged 0.9 μg P · L^{-1} in Athabasca River for the September to October experiment in northern Alberta (Perrin et al. 1995).

Exponential periphyton growth

Rates of periphyton accrual after colonization (~10 days) as a result of nutrient addition were computed by a least squares (log-linear regression) fit of chlorophyll *a* levels over time:

$$y = a \cdot 10^{kt} \quad (3)$$

where y is the Chl *a* concentration ($\mu g \cdot m^{-2}$) at day t, a is the initial chlorophyll *a* concentration, and k is the specific net growth rate (μg Chl *a* $\cdot m^{-2} \cdot day^{-1}$) (Bothwell 1988). Growth rates (k) of the periphyton colony (versus cellular growth rates) ranged from 0.0506 to 0.0783 μg Chl *a* $\cdot m^{-2} \cdot day^{-1}$ ($r^2 = 0.997$ to 0.999) for May 23 to June 23 and 0.0207 to 0.0324 μg Chl *a* $\cdot m^{-2} \cdot day^{-1}$ ($r^2 = 0.999$) for June 26 to July 31. However, these rates were only valid over an approximate 10- to 28-day time frame while the algae were in exponential growth phase. These growth rates, determined from a trough study, represent an upper limit for extrapolation to diatom-dominated, phosphate-limited streams because of low grazing pressure (Bothwell 1989), stability of the periphyton substrate (Aloi 1990), maximum nutrient release, which was achieved by exposing the entire pellet surface area to flowing water, and obstructed sunlight exposure, which slowed algal growth on pellet surfaces. Further experiments should be performed over a longer time frame to consider Monod's plateau phase of cellular growth kinetics and determine the long-term periphyton crop resulting from nutrient addition.

Large differences in periphyton accrual (i.e., $mg \cdot L^{-1}$ accumulation per unit time) at the end of the exponential growth phase in fertilized ($N:P_2O_5 = 7:40$) water flowing at 0.15 $m \cdot s^{-1}$ were apparent between the two seasons: maximum values reached 195 $g \cdot m^{-2}$ for early summer (May 23 to June 23) and 38 $g \cdot m^{-2}$ for the mid-summer studies (June 26 to July 31). These areal chlorophyll *a* concentrations are much higher than field observations reported in the literature. Horner and Welch (1981) observed chlorophyll *a* concentrations in excess of 420 $mg \cdot m^{-2}$ in less than 3 weeks. However, the Seattle, Washington, area creek had an average velocity of 0.60 $m \cdot s^{-1}$ and nutrient concentration of 38 $\mu g\ P \cdot L^{-1}$ during the summer experiment. Fertilization of an oligotrophic stream in the winter (January 29 to April 2, 1985) by Bothwell (1989) resulted in diatom communities on Styrofoam substrates reaching 7.5 to 200 $mg \cdot m^{-2}$ chlorophyll *a* with similar 0.1 to 5 $\mu g\ P \cdot L^{-1}$ additions, but also at a higher water velocity (0.50 $m \cdot s^{-1}$). Stockner and Shortreed (1978) obtained comparable 80.0 $mg \cdot m^{-2}$ chlorophyll *a* for 0.27 $\mu g\ P \cdot L^{-1}$ concentrations (N:P = 13) and 133.6 $mg \cdot m^{-2}$ for 0.29 $\mu g\ P \cdot L^{-1}$ (N:P = 39) at 0.40 $m \cdot s^{-1}$ water velocity. Average ambient SRP concentration for Stockner and Shortreed's June 25 to August 20 experiment was 0.11 $\mu g\ P \cdot L^{-1}$. Even lower biomass was observed by Slaney and Ward (1993) on artificial substrata (2.9 to 37.4 $mg \cdot m^{-2}$ chlorophyll *a* accrual for <3 $\mu g\ P \cdot L^{-1}$ concentrations in the Salmon River, Vancouver Island, from June 9 to August 5, 1992. Major factors creating a wide range of results between field and these controlled trough experi-

ments include season of fertilizer application, light levels and reduction of periphyton biomass by grazing insects. Our observations and those of our colleagues show that nutrient addition (0.5 to 5 $\mu g \cdot P \cdot L^{-1}$) clearly increased periphyton biomass. Over 27 days, the early summer experiment (May 23 to June 23) resulted in a 7.8 to 14.5 times increase in chlorophyll *a* biomass over the control, and 1.9 to 5.1 times increase over 28 days in a mid-summer experiment (June 26 to July 31). Mundie et al. (1991) observed a similar increase (3.5 times) of chlorophyll *a* biomass from 10 $\mu g \cdot L^{-1}$ phosphorus treatments (≥eight times the control concentration) at Carnation Creek, B.C. In the upper Nechako River, B.C., low-level whole-river fertilization was effective; enrichment for 2 months with 4.5 to 7 $\mu g \cdot L^{-1}$ P and 18 to 27 $\mu g \cdot L^{-1}$ N resulted in 10-fold increases in peak chlorophyll *a* in the first month (Slaney et al. 1994).

In this trough study, maximum biomass levels were reached with 3 $\mu g \cdot P \cdot L^{-1}$ phosphate additions, due to significant sloughing at the 5 $\mu g \cdot P \cdot L^{-1}$ level; the experiment was terminated once sloughing commenced. Sloughing of large quantities of accumulated biomass can be attributed to space limitation and self-shading phenomena, whereby cells closest to the substrate are weakened from poor light and intolerable chemical conditions (Stockner and Shortreed 1978; Jasper and Bothwell 1986), or nutrient limitation (Perrin and Richardson 1997). Biomass variations between the early and mid-summer experiments could be explained by several environmental factors such as light, temperature and nutrient levels. Although seasonal variations in light levels did not appear to influence algal growth rates (Bothwell 1988), phototoxicity by the ultraviolet spectrum is hypothesized to reduce periphyton growth in unshaded sites during very low flow (Perrin and Johnston 1986).

Comparisons between controlled field studies and river applications of periphyton biomass in response to phosphate concentrations can be misleading (Perrin et al. 1995). In situ periphyton accrual and growth rates can be affected by invertebrate grazers, water velocity, substratum stability, turbidity, light levels, nutrient concentration, nutrient ratios and immigration of algal species from upstream (Perrin et al. 1995; Miller et al. 1992; Aloi 1990). The extent of these interactions need to be determined before extrapolating nutrient requirements from trough to river systems. Further experiments are needed for lower fertilizer additions in potential stream sites over a longer time frame (months to years). For example, a 2-year lag in the control of epilithic algae by grazers in the Kuparuk River, Alaska, was observed by Peterson et al. (1993). They attributed it to the long life cycle (1 to 3 years) of dominant insects. Nutrient calculations should aim for maximum periphyton growth in mid- to late-summer months when most enrichment is needed, but should not be high enough to cause excess algal growth or sloughing. In British Columbia, the Ministry of Environment standards for maximum periphytic algal biomass are 50 $mg \cdot m^{-2}$ chlorophyll *a* for aesthetic and recreation considerations, and 100 $mg \cdot m^{-2}$ chlorophyll *a* for aquatic life in streams (Nordin 1985).

Table 3. Periphyton species abundance for three sampling periods[1]

Algal species	June 19/95				July 10/95				July 17/95			
	P+0 µg P · L^{-1}	P+0.5 µg P · L^{-1}	P+1.5 µg P · L^{-1}	P+3.0 µg P · L^{-1}	P+0 µg P · L^{-1}	P+0.5 µg P · L^{-1}	P+1.5 µg P · L^{-1}	P+3.0 µg P · L^{-1}	P+0 µg P · L^{-1}	P+0.5 µg P · L^{-1}	P+1.5 µg P · L^{-1}	P+3.0 µg P · L^{-1}
Achnanthes minutissima	R	C	R	—	R	C	R	—	R	R	R	—
Amphora ovalis	R	R	R	—	R	R	—	—	—	R	—	R
Cyclotella bodanica	C	A	C	C	C	C	R	C	A	C	R	C
Cymbella sp.	R	R	R	R	R	R	R	R	R	R	R	R
Diatoma elongatum	C	D	D	A	—	D	D	A	R	A	A	C
Diatoma vulgare	R	A	D	R	—	C	C	—	R	R	C	—
Eunotia exigua	R	C	R	R	C	C	R	R	R	R	R	R
Fragilaria capucina	—	—	—	D	—	—	—	—	—	—	D	—
Fragilaria construens	C	C	C	C	—	R	R	R	—	—	R	R
Fragilaria construens var.venter	R	C	C	C	—	R	R	R	R	R	R	R
Fragilaria crotonensis	—	—	C	C	R	R	A	C	R	C	C	C
Fragilaria virescens	A	A	C	R	D	A	C	C	D	C	C	R
Gomphoneis sp.	—	A	A	A	—	—	R	A	—	—	C	D
Gomphonema accuminatum	R	—	R	R	—	R	R	R	R	—	—	R

(continued)

Table 3. *(concluded)*

Algal species	June 19/95				July 10/95				July 17/95			
	P+0 µg P · L^{-1}	P+0.5 µg P · L^{-1}	P+1.5 µg P · L^{-1}	P+3.0 µg P · L^{-1}	P+0 µg P · L^{-1}	P+0.5 µg P · L^{-1}	P+1.5 µg P · L^{-1}	P+3.0 µg P · L^{-1}	P+0 µg P · L^{-1}	P+0.5 µg P · L^{-1}	P+1.5 µg P · L^{-1}	P+3.0 µg P · L^{-1}
Melosira varians	R	—	R	D	—	—	R	A	—	—	C	—
Navicula cryptocephala	—	A	C	R	R	C	R	R	A	C	R	R
Oscillatoria sp.	—	—	R	C	—	—	R	—	—	—	R	—
Rhopalodia gibba	—	—	R	R	—	R	R	R	R	R	—	—
Surirella robusta	C	C	C	—	C	A	C	—	A	A	C	—
Synedra acus	R	R	—	—	R	R	—	—	R	R	—	C
Synedra nana	C	R	C	R	R	D	C	—	R	D	C	R
Synedra tabulata	C	R	R	—	R	—	—	—	—	R	—	—
Synedra ulna	—	A	D	A	A	A	A	D	A	D	D	A
Tabellaria fenestrata	D	A	R	—	—	—	—	R	—	A	R	—
Tabellaria flocculosa	A	R	—	D	R	—	D	D	—	—	D	A

[1] R = rare (<3 cells per transect); C = common (3–10 cells); A = abundant (10–30 cells); D = dominant (>30 cells); and — = no observation.

Periphyton Community Composition

Periphyton was qualitatively sampled on three occasions over the course of the two experiments for species composition: June 19, water temperature = 17.5°C; July 10, T = 20.5°C; and July 17, 1995, T = 21°C. The controlled increase in dissolved nutrients (nitrogen and phosphorus) confirmed an alteration of algal species present in the stream community and denser growth by the more dominant species (Table 3). These changes were also observed by Peterson et al. (1985), Keithan et al. (1988) and Miller et al. (1992). Abundant species included chrysophyta and cyanophyta divisions, with the majority of communities comprised of diatoms. Nutrient addition did not alter species succession from diatoms toward blue-green *Oscillatoria*, which remained at low numbers for all nutrient treatments. This is advantageous as *Oscillatoria* is not consumed by invertebrate grazers and would subsequently inhibit fertilization efforts. An oligotrophic indicator species of periphytic diatoms, *Tabellaria flocculosa*, exhibited a reduction in species abundance in the June 19 and July 10 data at low P concentrations, and corresponds to observations in a tundra river by Peterson et al. (1985). However, chlorophyll *a* concentrations significantly increased for added phosphate concentrations of 1.5 and 3.0 $\mu g\ P \cdot L^{-1}$. This is attributed to a reduction in numbers of other competitive species such as *Diatoma sp.* and *Synedra sp.* (Peterson et al. 1985).

A dominance of diatoms was observed in the periphyton samples (Table 3) as well as in other experiments using both natural and artificial substrates (Keithan et al. 1988; Johnston et al. 1990). However, in other stream nutrient enrichment experiments, the percentage of diatoms in the algal community changed minimally (Stockner and Shortreed 1978; Mundie et al. 1991; Miller et al. 1992; Peterson et al. 1993). Species that reacted favorably to nutrient addition in Cultus Lake troughs included *Diatoma sp., Fragilaria capucina, Fragilaria crotonensis, Gomphoneis sp., Melosira varians, Oscillatoria sp., Synedra nana* (at low concentrations), *Synedra ulna,* and *Tabellaria flocculosa*. Keithan et al. (1988) also observed *Melosira varians* to be phosphorus limited in their Pennsylvania study site.

Despite these observations, Miller et al. (1992) determined that the largest variation in diatom community structure was between years, while less variation was associated with river fertilization. In order to accurately determine the effects of nutrient additions on periphytic algal communities, it is apparent that observations must be made over longer time frames with more frequent sampling than was done in this experiment.

Conclusions

Studies of the slow-release pellet fertilizer demonstrate it is a simple and effective method for increasing nutrient concentrations in streams experiencing cultural oligotrophication. Dissolution of fertilizer pellets (~2 to 9 g) under a range of water temperature and flow conditions, chosen to simulate B.C.'s streams, was influenced predominately by length of

water exposure. Other variables (pellet size, water temperature and flow) had a much smaller influence on pellet dissolution. The pellet's uncomplicated dissolution characteristics allow for nutrient addition to be determined by pellet size, which correlates to length of application. The pellets dissolved at a rate of 0.393 $g \cdot days^{-0.401}$ when fully exposed to the flowing water. In field conditions only about half a pellet is exposed to flowing water when placed on the river substrate; thus dissolution rates in field conditions would be approximately half the experimental rate, or 0.2 $g \cdot days^{-0.401}$. Application of the fertilizer pellets to outdoor troughs resulted in relative specific growth rate (i.e., cellular response) saturation of diatom communities with phosphate additions of ~1 $\mu g \cdot L^{-1}$; Bothwell (1988) and Perrin et al. (1995) noted similar values. Although saturation of diatom cellular growth rates occurs at low SRP concentrations, Bothwell (1989) observed that biomass continues to increase with additions up to 50 $\mu g\ P \cdot L^{-1}$; he explains this phenomena by changes in cellular to community controlled growth rates, whereby higher phosphorus concentrations are needed to saturate and maintain growth of the denser algal community. Four weeks of periphyton analyses determined fertilizer additions above 3 $\mu g\ P \cdot L^{-1}$ produced enough periphyton biomass to result in sloughing, although grazing effects from invertebrates were not present. Long term studies over several months to years, are needed to examine more closely the algal community changes, equilibrium growth, and the range of annual variations. Field testing of this product are described in Mouldey Ewing and Ashley (1996, 1998) and Mouldey Ewing et al. (1998).

Acknowledgments

Financial assistance for this project was provided by the Columbia Basin Fish and Wildlife Compensation Program, Forest Renewal B.C.'s Watershed Restoration Program, and an NSERC operating grant to Ken Hall, Department of Civil Engineering, University of British Columbia. We are grateful to the Fraser Valley Trout Hatchery, Abbotsford, and the Cultus Lake Salmon Research Laboratory, British Columbia, for use of their troughs and technical assistance. Thanks also to Bob Land and Brian Toth (Abbotsford Fisheries Research and Development Section) for their technical assistance, and to Frances Pick, University of Ottawa, for algal species identification. Bob Rehberg from IMC Vigoro provided information on the slow-release pellet fertilizer characteristics. Guy Kirsch, Engineering Workshop, U.B.C., is acknowledged for his diligent sawing of the pellets into the smaller sizes, and Sarah Mouldey Ewing provided valuable assistance with information from B.C. Environment. Special thanks for the guidance from Max Bothwell regarding the periphyton studies. We are extremely grateful for the review provided by Hugh Dobson.

References

Aloi JE. 1990. A critical review of recent freshwater periphyton field methods. Can. J. Fish. Aquat. Sci. **47**:656–670.

American Public Health Association (APHA), American Water Works Association (AWWA), and Water Environment Federation (WEF). 1992. Standard methods for the examination of water and wastewater, 18th ed. American Public Health Association, Washington, DC.

Bilby RE, Fransen BR, Bisson PA. 1996. Incorporation of nitrogen and carbon from spawning coho salmon into the trophic system of small streams: evidence from stable isotopes. Can. J. Fish. Aquat. Sci. **53**:164–173.

Bothwell ML. 1983. All-weather troughs for periphyton studies. Water Res. **17**:1735–1741.

Bothwell ML. 1985. Phosphorus limitation of lotic periphyton growth rates: an intersite comparison using continuous-flow troughs (Thompson River system, British Columbia). Limnol. Oceanogr. **30**:527–542.

Bothwell ML. 1988. Growth rate responses of lotic periphytic diatoms to experimental phosphorus enrichment: the influence of temperature and light. Can. J. Fish. Aquat. Sci. **45**:261–270.

Bothwell ML. 1989. Phosphorus-limited growth dynamics of lotic periphytic diatom communities: areal biomass and cellular growth rate responses. Can. J. Fish. Aquat. Sci. **46**:1293–1301.

Deegan LA, Peterson BJ. 1992. Whole-river fertilization stimulates fish production in an Arctic tundra river. Can. J. Fish. Aquat. Sci. **49**:1890–1901.

Gregory SV. 1983. Plant-herbivore interactions in stream systems, p. 157–189. *In* Barnes JR, Minshall JW (ed.), Stream ecology: application and testing of general ecological theory. Plenum Press, New York.

Horner RR, Welch EB. 1981. Stream periphyton development in relation to current velocity and nutrients. Can. J. Fish. Aquat. Sci. **38**:449–457.

Jasper S, Bothwell ML. 1986. Photosynthetic characteristics of lotic periphyton. Can. J. Fish. Aquat. Sci. **43**:1960–1969.

Johnston NT, Perrin CJ, Slaney PA, Ward BR. 1990. Increased juvenile salmonid growth by whole-river fertilization. Can. J. Fish. Aquat. Sci. **47**:862–872.

Keithan ED, Lowe RL, DeYoe HR. 1988. Benthic diatom distribution in a Pennsylvania stream: role of pH and nutrients. J. Phycol. **24**:581–585.

Larkin GL, Slaney PA. 1996. Trends in marine derived nutrient sources to south coastal British Columbia streams: impending implications to salmonid production. Watershed Restoration Management Report No. 3, B.C. Ministry of the Environment, Lands and Parks.

Mason JC. 1976. Response of underyearling coho salmon to supplemental feeding in a natural stream. J. Wild. Manage. **40**:775–788.

Miller MC, DeOliveria P, Gibeau GG. 1992. Epilithic diatom community response to years of PO_4 fertilization: Kuparuk River, Alaska (68 N Lat.). Hydrobiologia **240**:103–119.

Mouldey Ewing SE, Ashley KI. 1996. Development and testing of slow release fertilizer for restoring salmonid habitat, 1995 progress report. Watershed Restoration Management Report No. 3, B.C. Ministry of the Environment, Lands and Parks.

Mouldey Ewing SE, Ashley KI. 1998. Development and testing of slow release fertilizer for restoring salmonid habitat, 1996 progress report. Watershed Restoration Project Report No. 9, B.C. Ministry of the Environment, Lands and Parks.

Mouldey Ewing SE, Ashley KI, Wilson G. 1998. Development and testing of slow release fertilizer for restoring salmonid habitat, 1997 progress report. Watershed Restoration Project Report No. 10, B.C. Ministry of the

Environment, Lands and Parks.

Mundie JH, Simpson KS, Perrin CJ. 1991. Responses of stream periphyton and benthic insects to increases in dissolved inorganic phosphorus in a mesocosm. Can. J. Fish. Aquat. Sci. **48**:2061–2072.

Ney JJ. 1996. Oligotrophication and its discontents: Effects of reduced nutrient loading on reservoir fisheries. Am. Fish. Soc. Symp. **16**:285–295.

Nordin RN. 1985. Water quality criteria for nutrients and algae. Water Quality Unit, Resource Quality Section, Water Management Branch, B.C. Ministry of the Environment, Lands and Parks.

Perrin CJ, Johnston NT. 1986. A continued evaluation of stream fertilization as a salmonid enhancement technique. Limnotek Research and Development Inc.

Perrin CJ, Bothwell ML, Slaney PA. 1987. Experimental enrichment of a coastal stream in British Columbia: effects of organic and inorganic additions on autotrophic periphyton production. Can. J. Fish. Aquat. Sci. **44**:1247–1256.

Perrin CJ, Chambers PA, Bothwell ML. 1995. Growth rate and biomass responses of periphytic algae to nutrient enrichment of stable and unstable substrata, Athabasca River. Northern River Basins Study Project Report No. 46, Edmonton.

Perrin CJ, Richardson JS. 1997. N and P limitation of benthos abundance in the Nechako River, British Columbia. Can. J. Fish. Aquat. Sci. **54**:2574–2583.

Peterson BJ, Hobbie JE, Corliss TL, Kriet K. 1983. A continuous-flow periphyton bioassay: tests of nutrient limitation in a tundra stream. Limnol. Oceanogr. **28**:583–591.

Peterson BJ, Hobbie JE, Hershey AE, Lock MA, Ford TE, Vestal JR, McKinley VL, Hullar MAJ, Miller MC, Ventullo RM, Volk GS. 1985. Transformation of a tundra stream from heterotrophy to autotrophy by addition of phosphorus. Science. **229**:1383–1386.

Peterson BJ, Deegan L, Helfrich J, Hobbie JE, Hullar M, Moller B, Ford TE, Hershey A, Hiltner A, Kipphut G, Lock MA, Fiebig DM, McKinley V, Miller MC, Vestal JR, Ventullo R, Volk G. 1993. Biological responses of a tundra stream to fertilization. Ecology. **74**:653–672.

Quamme DL. 1994. Phosphorus limited community dynamics of stream benthic algae and insects. Master's thesis, Department of Zoology, University of British Columbia, Vancouver.

Slaney PA, Northcote TG. 1974. Effects of prey abundance on density and territorial behavior of young rainbow trout (*Salmo gairdneri*) in laboratory stream channels. J. Fish. Res. Board Canada. **31**:1201–1209.

Slaney PA, Ward BR. 1993. Experimental fertilization of nutrient deficient streams in British Columbia, p. 128–141. *In* Shooner G, Asselin S (ed.). Le développement du saumon atlantique au Québec: connaître les règles du jeu pour réussir. Colloque international de la Fédération québécoise pour le saumon atlantique. Québec, décembre 1992. Collection *Salmo salar* n°1.

Slaney PA, Rublee WO, Perrin CJ, Goldberg H. 1994. Debris structure placements and whole-river fertilization for salmonids in a large regulated stream in British Columbia. Bull. Mar. Sci. **55**:1160–1180.

Slaney TL, Hyatt KD, Northcote TG, Fielden RJ. 1996. Status of anadromous salmon and trout in British Columbia and Yukon. Fisheries. **21**:20–35.

Sterling MS, Hall KJ. In preparation. Slow-release fertilizer for rehabilitating oligotrophic streams: a chemical characterization. For submission to Water Qual. Res. J. Canada.

Stockner JG, MacIsaac EA. 1996. British Columbia lake enrichment programme:

two decades of habitat enhancement for sockeye salmon. Regul. Rivers Res. Manag. **12**:547–561.

Stockner JG, Shortreed KRS. 1978. Enhancement of autotrophic production by nutrient addition in a coastal rainforest stream on Vancouver Island. J. Fish. Res. Board Canada. **35**:28–34.

Ward BR, Slaney PA. 1988. Life history and smolt-to-adult survival of Keogh River steelhead trout (*Salmo gairdneri*) and the relationship to smolt size. Can. J. Fish. Aquat. Sci. **45**:1110–1122.

Wilzbach MA. 1985. Relative roles of food abundance and cover in determining habitat distribution of stream-dwelling cutthroat trout (*Salmo clarki*). Can. J. Fish. Aquat. Sci. **42**:1668–1672.