WASTEWATER EFFECTS ON EPILITHON, PARTICULARLY SEWAGE FUNGUS, AND WATER QUALITY IN THE MANAWATU RIVER, NEW ZEALAND

A thesis presented in partial fulfilment of the requirements of the degree
Doctor of Philosophy in Biotechnology
at Massey University

John Martin Quinn

1985

ABSTRACT

Epilithon development, in relation to the discharge of domestic sewage, dairy factory and meatworks wastewaters, and its effects on water quality were studied in laboratory channels and in the Manawatu River. During the three year period of the study the organic material inputs to this river were progressively reduced to meet the requirements of water rights designed to limit the in-river BOD_5 to 5 $g.m^{-3}$ at the end of a defined mixing zone with the objective of maintaining adequate oxygen levels and controlling sewage fungus growth.

Laboratory channel studies demonstrated that, for a given BOD_5 addition, untreated dairy factory wastewater increased the heterotrophic growth 2-3 times more than primary treated meatworks wastewater. Similar observations were made in the Manawatu River. These varied growth responses could be accounted for by the different relative contributions of dissolved and low molecular weight (< 1000 daltons) organic compounds in the different wastewaters. The dissolved or low molecular weight (determined after sample ultrafiltration) BOD_5 therefore provide more reliable general sewage fungus control parameters than BOD_5.

Current velocity and spates had marked influences on the development of benthic communities. Maximum sewage fungus biomasses on the natural bed were observed at current velocities of 0.2 to 0.45 $m.s^{-1}$. Short heterotrophic fronds occurred at the maximum current velocity investigated of 1.16 $m.s^{-1}$. Small spates of up to 50 to 70 $m^3.s^{-1}$ caused preferential sloughing of heterotrophs over epilithic phototrophs which had developed on concrete plates at river flows of approximately 25 $m^3.s^{-1}$. Flows in excess of approximately 150 $m^3.s^{-1}$ removed growths of *Cladophora glomerata* which had developed at sites where the pre-spate current velocity was 0.3 to 0.4 $m.s^{-1}$. Much higher flows, in excess of 400 $m^3.s^{-1}$, were required to remove

the dense growths of the macrophyte *Potamogeton crispus*.

Observations of sewage fungus biomass at various depths in the Manawatu River and growth rates on both upper, sunlight-exposed, and lower, shaded, surfaces of concrete plates suspended in the water column indicated that solar radiation inhibition of heterotrophic growth is not important in the Manawatu River.

These heterotrophic growths in the river were replaced by heavy phototroph-dominated epilithon as organic concentrations were reduced. Both communities had significant impacts on the suspended biomass and dissolved oxygen levels in the river.

A computer model simulating summer low flow conditions in the Manawatu River predicted that the river can sustain average respiration rates of 20 and 24 g O_2 m^{-3} d^{-1} at mean river temperatures of 21°C to 12°C respectively without breaching the statutory minimum permissable dissolved oxygen concentration of 5 $g.m^{-3}$. A multiple regression model of the factors influencing epilithon respiration was developed from *in situ* chamber studies of a range of epilithic community types. This gave adequate predictions when tested against measurements over reaches below the discharges and predicted that the benthic biomass resulting in the maximum permissible respiration rates decreased from approximately 143 g AFDW m^{-2} at 12°C to 34 g AFDW m^{-2} at 21°C.

A management strategy limiting the organic, but not the nutrient, inputs to the Manawatu River was shown to be unlikely to ensure consistent maintenance of the statutory minimum dissolved oxygen concentration.

The implications for management of the river are discussed.

ACKNOWLEDGEMENTS

- Dr Paul McFarlane for setting up this project and his able and enthusiastic supervision.

- Professor R L Earle and Dr Graham Manderson for providing facilities and encouragement.

- Mr John Alger and Mr Paul Shore for their efficient help with equipment construction and maintenance and their good humour.

- Dr Paul Austin for assistance with mathematics, Dr Richard Archer and Dr Andrew Cleland for help with the simulation model and Mr Donald Cleland for computing advice.

- Dr Mike Freeman, for sending me the research proposal, constructing the respiratory chambers and many valuable discussions.

- Dr John Boyle, University of Exeter, for the gift of one of his respirometers.

- Mr Barry Gilliland, of MRWB, for discussions, the loan of equipment and general cooperation throughout this study.

- Mr Jeff Watson, of MRWB, for provision of river flow data.

- Scientists at Water Quality Centre Hamilton for advice and cooperation especially Mr Chris Hickey, Dr Bryce Cooper and Dr Bob Wilcock.

- Mr Chris Hall for technical assistance with the nutrient analyses.

- Technical staff of the dischargers for help in the unromantic task of wastewater collection and provision of discharge data especially Mr Jim Anderson (PNCC), Mr Tim Meredith and Mrs Robyn Cotton (MCDC) and Mr Alan Hines (BCWS).

- The farmers, Mr Robert Wattes, Mr Jim Reichenbach, Mr David Penny and Mr Duncan Campbell, for providing access to the river sites and their help and interest.

- National Water and Soil Conservation Organisation for funding this research.

- State Services Commission for the provision of my Study Award.

- Staff of the Soil Conservation Centre, Aokautere, for cooperation in the production of the final thesis.

- Nicola Hunter for boundless patience in typing the thesis.

- Shardell for lots of TLC.

- Jacob for being an easy first child and especially for sleeping at night (usually).

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CHAPTER 1: INTRODUCTION

The development of epilithon in rivers has marked effects on the quality of the overlying water and river aesthetics. The Manawatu River, near Palmerston North (latitude 40°21'S), New Zealand (Fig 1.1), has had a history of nuisance epilithon growths. Heterotroph dominated epilithon, commonly referred to as sewage fungus, has frequently been reported below the wastewater discharges downstream of Palmerston North (Fig 1.2) (MOW, 1957; Hirsch, 1958; Currie, 1977, 1978; Freeman, 1983) and, during summer, filamentous algae often proliferate upstream of the discharges (Currie, 1977; Freeman, 1983). The former growths in particular cause a marked deterioration in the river aesthetics and have been implicated as an important factor causing low oxygen conditions leading to a fish kill downstream from the wastewater discharges in 1978 (Currie, 1980; Currie and Rutherford, 1982).

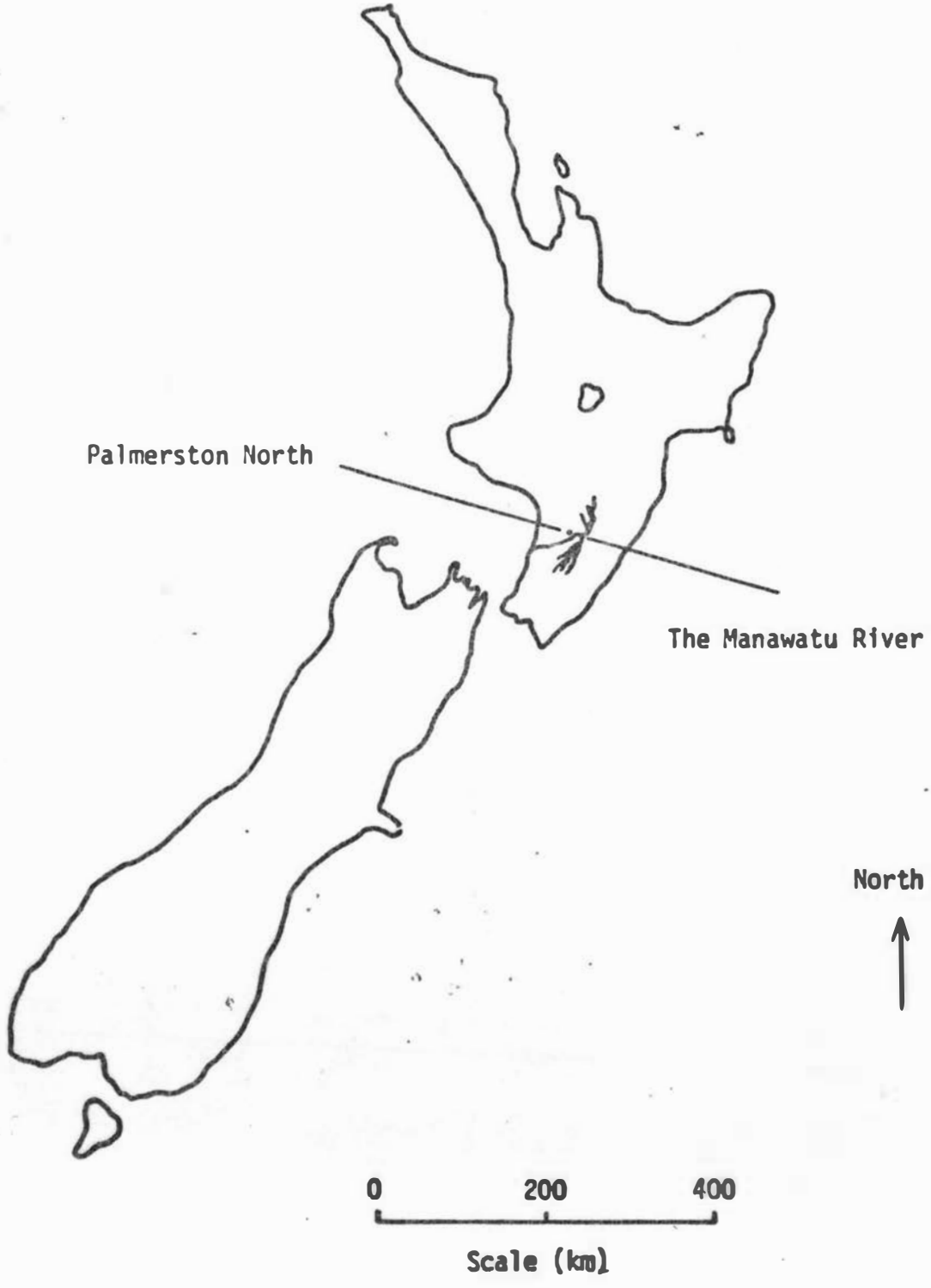
The Manawatu Regional Water Board (MRWB) has attempted to control the sewage fungus growth and dissolved oxygen depletion to acceptable levels by issuing water rights which restrict the organic material inputs by the wastewater dischargers to a level calculated to limit the in-river five day biochemical oxygen demand (BOD_5) to less than 5 $g.m^{-3}$ at the end of a defined waste mixing zone (i.e., Site D, Fig 1.2).

The changes in wastewater inputs to the river during the period from 1982 to 1985 due to the requirements of these water rights provided an opportunity to study the relationships between organic wastewater concentrations and the development of epilithon, especially sewage fungus, and the effects of these factors on river aesthetics and water quality, with respect to the concentration of dissolved oxygen, organic material, and suspended biomass.

The following chapters describe investigations of these relationships made in the Manawatu River and complementary studies in a laboratory channel system.

The studies were oriented towards providing data and models for use in resolving the complex management problems presented by the situation in the Manawatu River.

Figure 1.1 Map of New Zealand



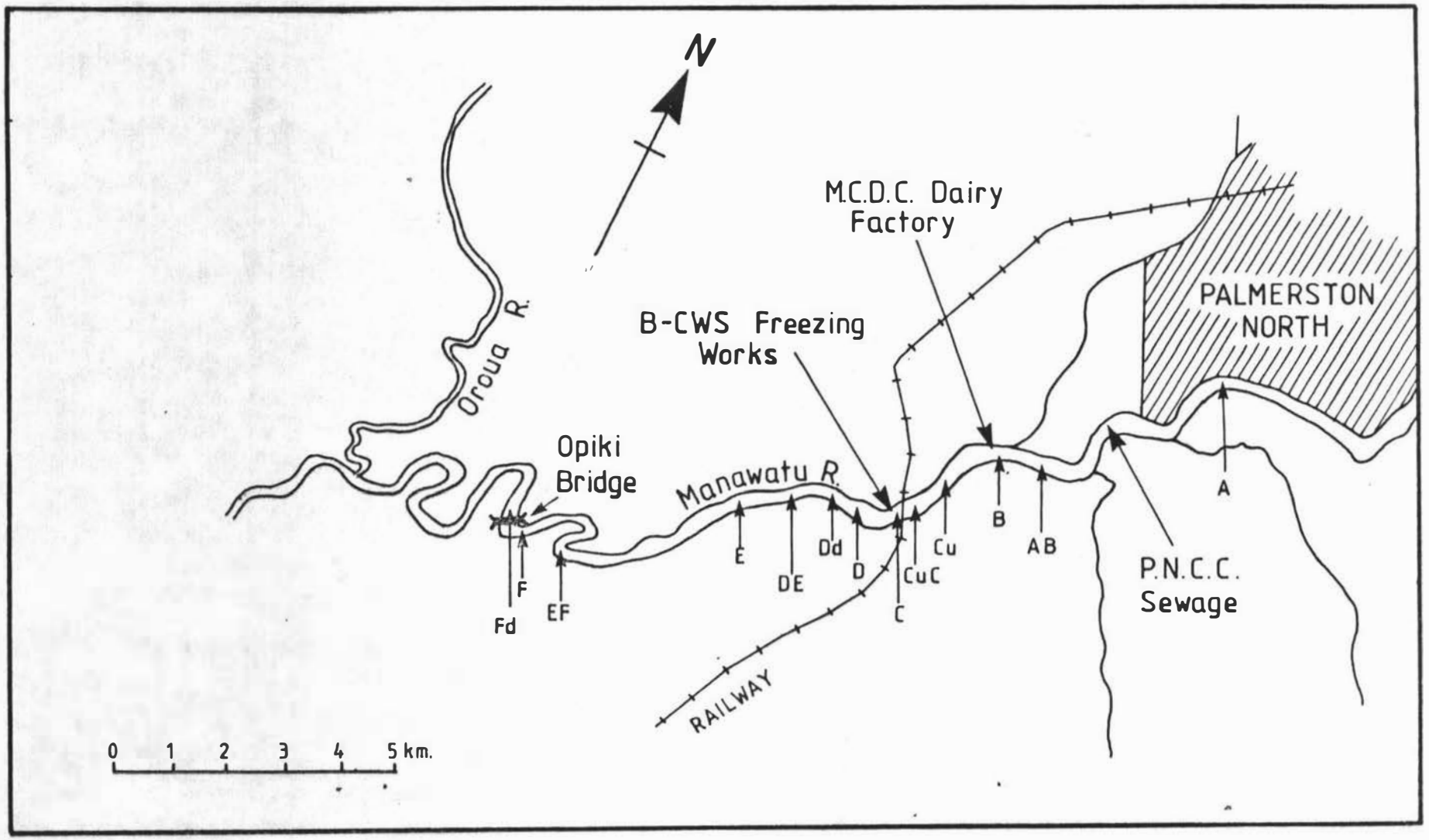


Fig. 1·2: Manawatu River Location map, showing study sites (A - Fd) and wastewater discharges

CHAPTER 2. LITERATURE REVIEW.

2.1 INTRODUCTION

The first reported observations of sewage fungus in the Lower Manawatu River were in March 1957 (MOW, 1957; Hirsch, 1978) when it was reported as being common at site C (Fig. 1.2) and abundant at sites D and E. Filamentous green algae were also reported as occurring upstream of Palmerston North and below the city as far as site EF (Fig. 1.2). A similar distribution of algae and sewage fungus existed during the summers of 1971 to 1972, when sewage fungus covered 80% of the bed at Site D (Anderson, 1972), and 1975 to 1977 (Currie, 1977) when lumps of sewage fungus were observed suspended in the water column at distances of up to 20 to 30 kilometers below Site D. These made the river "particularly unattractive". The attached sewage fungus growths were also visually offensive, caused unpleasant odours as the river receded during low flows and excluded or smothered aquatic insects (Currie, 1977).

The minimum in-river dissolved oxygen level of 5 $g.m^{-3}$, required by the D classification of the river below the waste discharges under the Water and Soil Conservation Act 1967, was met during the occasional surveys conducted at this time (Currie, 1977). However in January 1978 severe night-time deoxygenation occurred resulting in a major fish kill for several kilometres below Site C (Currie, 1978). Sewage fungus was prolific from below Site B to Site E at the time of the fish kill.

Later, more detailed studies indicated that sewage fungus played an important role in the very rapid removal of oxygen and organic material from the river observed below the discharges (Currie and Rutherford, 1982; Hickey and Rutherford, 1983).

Thus an understanding of the factors affecting the growth and activity of the sewage fungus community is essential for the management of the lower Manawatu River and many other rivers

effected by sewage fungus in New Zealand and overseas (Section 2.3.1).

2.2 CHARACTERISTICS OF THE MANAWATU RIVER AND WASTES RECEIVED

2.2.1 MANAWATU RIVER

2.2.1.1 Physical Characteristics

The Manawatu River is approximately 230 km long and has a catchment of 5800 km^2 (Fig. 1.1). Its many tributaries arise in the Tararua and Ruahine Ranges and drain predominantly agricultural land. Over the study reach (Fig. 1.2), where the river is sixth order, it receives three minor tributaries which typically contribute 0.02 to 0.10 $m^3.s^{-1}$ of water during summer flows (Currie, 1977).

The gradient in the river is approximately 1 $m.km^{-1}$ between Site A and Site EF decreasing to approximately 0.5 $m.km^{-1}$ at Site F and 0.1 $m.km^{-1}$ four kilometres below this site (Currie, 1977). Bottom substrate size generally decreases with decreasing slope (Hynes, 1970) and below Site F the bed changes from gravel to silt (Currie, 1980).

The river's median and annual mean flows at Site A (Fig. 1.2) are 60.1 $m^3.s^{-1}$ (Fig. 2.1) and 106.4 $m^3.s^{-1}$ respectively (Watson, 1984). The monthly average flow varies considerably during the year (Fig. 2.2) with the highest values occurring during the winter and the lowest during late summer. The daily flow data for the period from October 1983 to May 1984 (Fig. 2.3) show that the flow pattern is characterised by sudden freshes and relatively short periods of stable conditions.

Travel times over the study reach have been calculated at flows of 15.6 $m^3.s^{-1}$, 20.0 $m^3.s^{-1}$ and 26.3 $m^3.s^{-1}$ using the results of dye studies (Currie and Rutherford, 1982; Wilcock, 1984(a)). At 26.3 $m^3.s^{-1}$ the average velocities, based on the dye peak travel times between the study sites (Fig 1.2), varied from 0.48 $m.s^{-1}$ to 0.54 $m.s^{-1}$ (Currie and Rutherford,

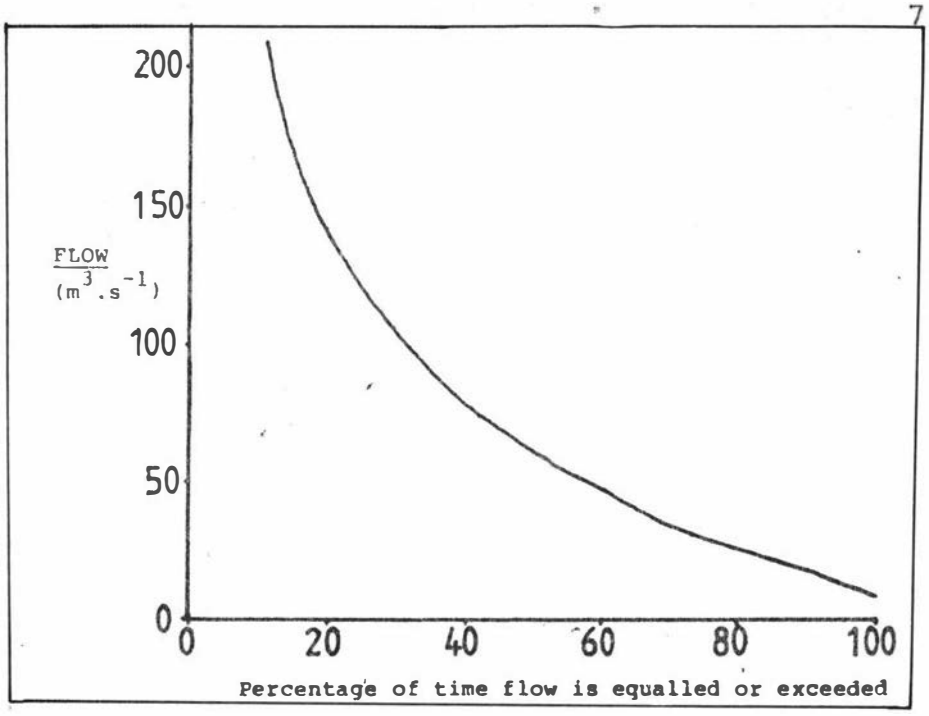


Figure 2.1: Manawatu River Flow Duration Curve (site A, 1972-1982).

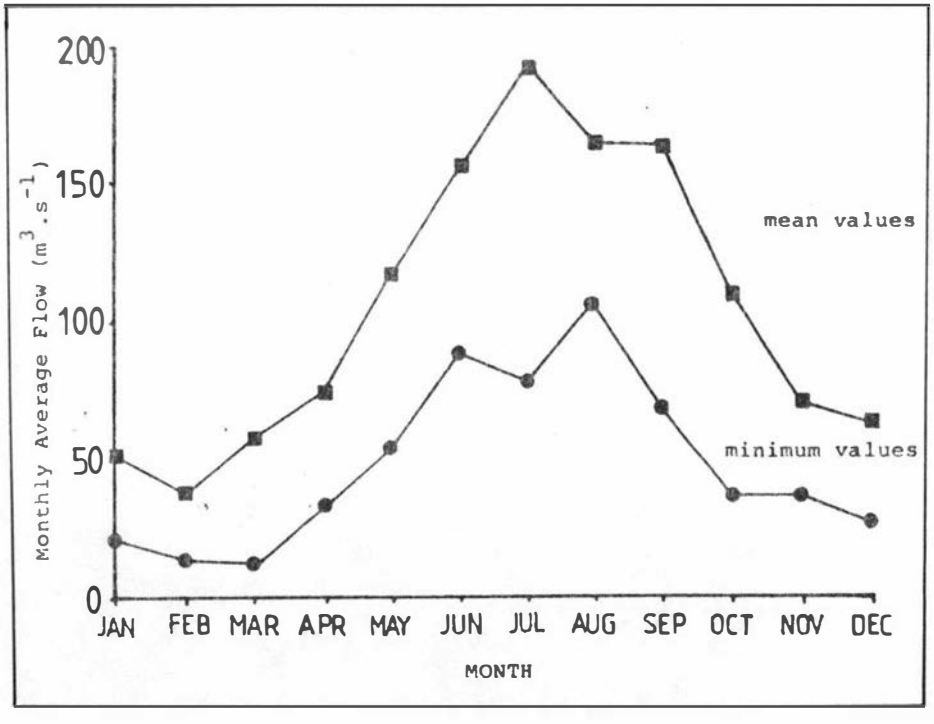


Figure 2.2: Manawatu River Monthly Mean Flows; 1972-1982.

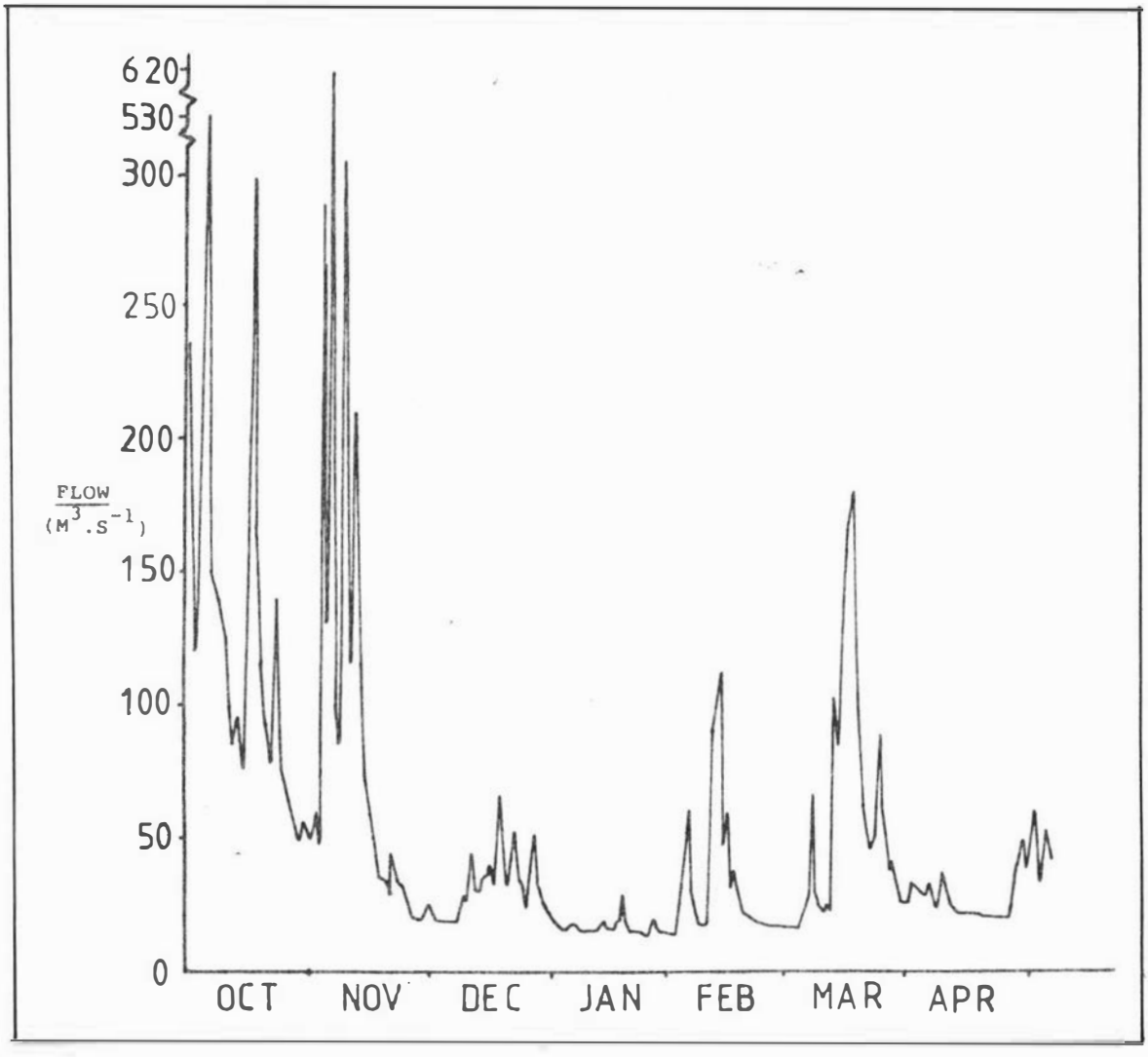


Figure 2.3: Manawatu River Flow, October 1983 to May 1984, at Site A.

1982). At 15.6 $m^3.s^{-1}$ the mean velocity for the reach between Sites C and F was 0.31 $m.s^{-1}$. However within this reach the mean velocities for shorter, 1.4 to 4 km long, river sections varied from 0.18 to 0.67 $m.s^{-1}$ (Wilcock, 1984 a). This range of current velocities for short river stretches reflects the riffle-pool-run nature of the river which becomes more pronounced as flows decline.

Assuming that the river slope and bed friction forces are independent of flow, average velocities at various flows can be estimated using measured velocity and flow data in equation (2.1) (Rutherford and Currie, 1979), derived from the Manning Equation:

$$u_b = u_A \left(\frac{Q_B}{Q_A} \right)^{0.4} \qquad (2.1)$$

where u_A, u_B = known and unknown average velocities ($m.s^{-1}$) at flows Q_A, and Q_B ($m^3.s^{-1}$) respectively.

Using the current velocity data developed from the dye tracer studies at 26.3 $m^3.s^{-1}$ and 15.6 $m^3.s^{-1}$, equation (2.1) predicts that the average current velocities at the 50% and 96% low flows of 60.1 $m^3.s^{-1}$ and 13 $m^3.s^{-1}$ will be 0.73 $m.s^{-1}$ and 0.29 $m.s^{-1}$ respectively. Since the optimal current velocities for sewage fungus are within the approximate range of 0.18 to 0.8 $m.s^{-1}$ (Section 2.3.4.5), flows of less than the 96% low flow to the median flow will provide average current velocities suitable for sewage fungus development.

The river's natural reaeration was studied using methyl chloride as a tracer over three 2.4 to 5.8 km long sections of the river between Sites C and F (Fig. 1.2) at a flow of 15.6 $m^3.s^{-1}$ on 2 March 1983 (Wilcock, 1984(a)). The resultant reaeration coefficient (k_2) values were approximately 1.4 times greater than those calculated from hydraulic radius data using the equation of O'Connor and Dobbins (1958). This conforms to the pattern observed in a number of other New Zealand rivers (Wilcock, 1984(b)).

Longitudinal dispersion has been studied over the study reach where it was found to have a negligible effect on the removal

of organic material (Rutherford, Gilliland and McBride, 1982). This finding is in agreement with that of Dobbins (1964) who concluded that longitudinal dispersion has a negligible effect on BOD_5 and oxygen profiles in most streams and implies that models of self-purification in the Manawatu River need not include this factor.

2.2.1.2 Biological Characteristics and Water Quality

Water quality investigations over the study reach, utilising macro-benthic invertebrate community structure indicate a progressive deterioration in water quality below the successive discharges during the summer (Currie, 1977; Suckling, 1980). At Site E the community had returned to a "clean water" fauna in the February 1977 study (Currie, 1977). However during the winter and spring months, when the river flows were high and variable (Fig. 2.2) and the effluent discharges lower (Sections 2.2.4 and 2.2.5), there was little difference in the invertebrate numbers or composition throughout the study reach (Suckling, 1980).

Night-time oxygen depletion has been observed in the Manawatu River below the waste discharges in a number of studies (Johannesson, 1958; Currie, 1977; Currie, 1978; Freeman, 1983; Hickey and Rutherford, 1983). These were all conducted during the summer when algae occurred above the discharges and sewage fungus below them. At river flows greater than 20 $m^3.s^{-1}$ this oxygen depletion did not breach the requirement of the river's D classification that the dissolved oxygen concentration exceed 5 $g.m^{-3}$ (Johannesson, 1958; Currie, 1977; Freeman, 1983). However during two studies when the flows were 11.5 $m^3.s^{-1}$ (Currie, 1978, Fig. 2.4) and 12.5 $m^3.s^{-1}$ (Freeman, 1983) this requirement was not met at Sites EF and F (Fig. 1.2) respectively for 12 hours during the night and morning. The data presented in Figure 2.4 were obtained three days after the beginning of a major fish kill below Site C on 28 January 1978. This event was attributed to the low dissolved oxygen concentrations (Currie, 1978) and the respiration of the sewage fungus, which occurred from below the MCDC outfall (Fig. 1.2) at the time of the kill, was prosposed as an important factor in causing the oxygen depletion (Currie, 1980). Indirect

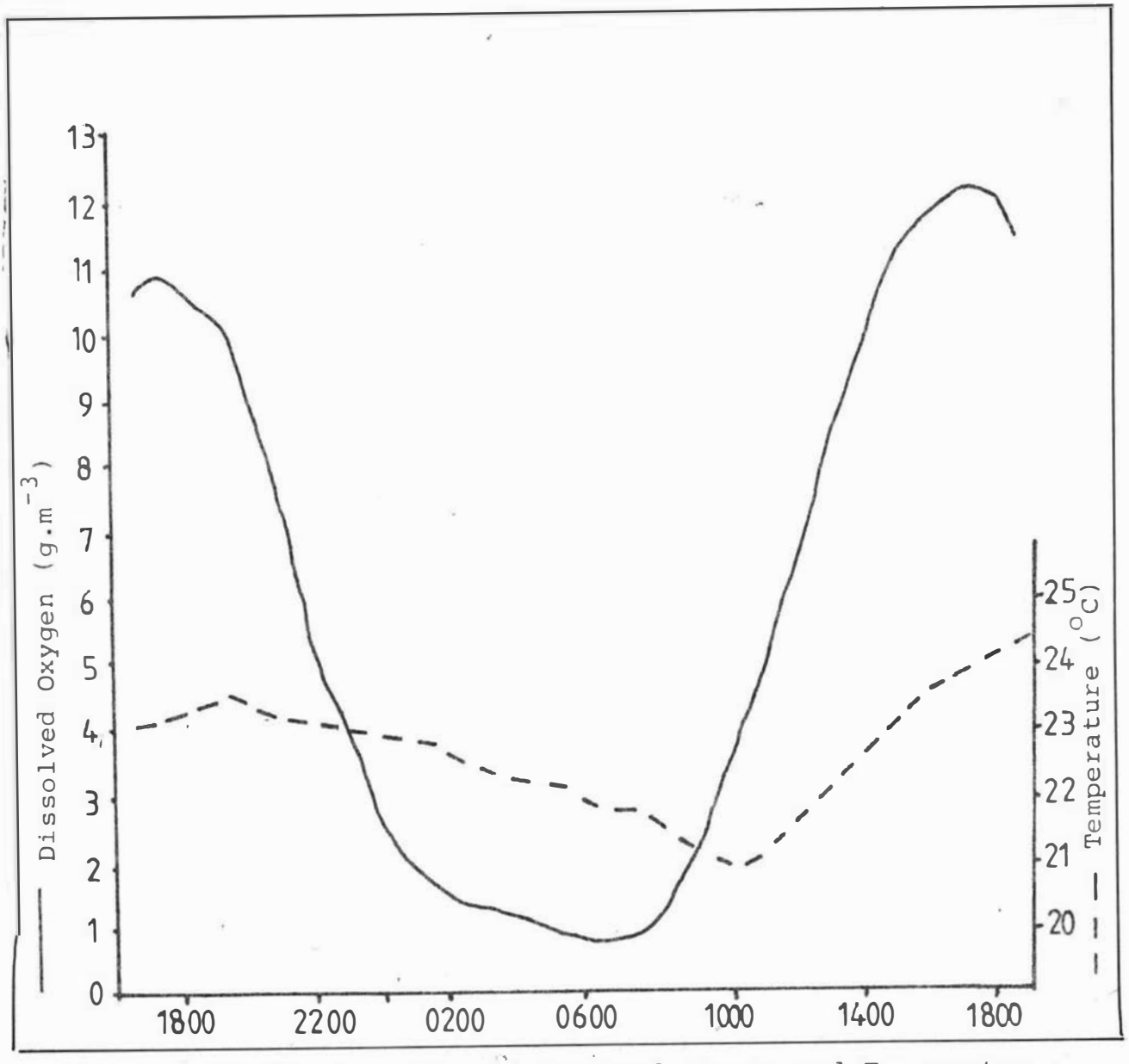


Figure 2.4: Manawatu River Dissolved Oxygen and Temperature Profiles; 31 January to 1 February 1978, site EF (Currie,1978).

evidence for this was provided by the observations of very high BOD_5 decay rates over the river reaches containing prolific sewage fungus growths (Section 2.3.6) (Currie, 1977; Currie and Rutherford, 1982) which implies that its activity is important in dissolved oxygen removal from the river.

This deoxygenation effect of sewage fungus and its degrading effect on river aesthetics were the main reasons for the development of a growth control strategy (Currie, 1980).

However investigations of algal proliferations above the waste discharge area have shown that these have significant effects on the river dissolved oxygen levels causing these to drop by as much as 3.3 $g.m^{-3}$ below saturation at night (Freeman, 1983). This reduces the reserves of dissolved oxygen available to meet the demands of the sewage fungus communities below the discharges and also indicates that the phototrophic component of the communities below the outfalls may contribute significantly to the problems of night-time oxygen depletion. Further evidence for this was provided by study of oxygen dynamics between sites D and E, when a mixed phototrophic and sewage fungus community occurred. This showed that the photosynthetic oxygen production almost matched the reach community respiration during the early afternoon (Hickey and Rutherford, 1983) and indicated that the phototrophic component of the benthic community below the outfalls was significant.

While the regional water board has developed a sewage fungus control strategy, no attempt has been made to control the growth of the phototrophic community below the outfalls by limiting the discharge of nitrogen or phosphorus in the new water rights (Section 2.2.2). Above the discharges the biomass of the filamentous green alga *Cladophora glomerata* (L) Kutz often reaches 100 g drywt.m^{-2} during extended summer flows (Freeman, 1983). Its growth rate is apparently unaffected by the river concentrations of dissolved inorganic nitrogen (DIN), which usually exceed 100 $mg.m^{-3}$, but the growth of algae at the moderate current velocities of 0.4 to 0.7 $m.s^{-1}$ studied becomes limited by dissolved reactive phosphorus (DRP) when levels drop

to approximately 4 $mg.m^{-3}$ (Freeman, 1983). However below the discharges the levels of these nutrients are elevated considerably (Currie, 1977; Freeman, 1983) and it can be calculated from the data of Cooke *et al.*(1980) that the additions of the primary-treated PNCC and BCWS effluents to the river at the 96% low flow of 13 $m^3.s^{-1}$ increase the concentration of DIN by 320 and 95 $mg.m^{-3}$ respectively and the concentration of DRP by 40 and 15 $mg.m^{-3}$ respectively. Further unknown nutrient addition also occurs due to the MCDC discharge.

The effects of these additions on phototrophic growth have not been studied but studies upstream of the discharges (Freeman, 1983) indicated that the background nutrient levels are sufficient to sustain algal growth at moderate current velocities most of the time. Almost complete removal of phosphorus from the effluents would therefore be required to prevent the algal growth limitation by nutrients, which occasionally occurs upstream of the outfalls during summer, from being alleviated below the outfalls. These results suggest that there is little point in removing the nutrients added by the wastewater discharges since nutrient limitation only occurs occasionally upstream of these.

The upstream algal communities have also been shown to induce pH fluctuations in the river water (Freeman and McFarlane, 1982; Freeman, 1983), by removing CO_2 during photosynthesis during the day resulting in the removal of H^+ ions and a consequent rise in pH (Stumm and Morgan, 1981). When the algal biomass was minimal the pH above the discharges was relatively stable at approximately 7.5 $\pm$ 0.4. However at high algal biomass levels diurnal pH fluctuations as large as from 7.3 to 8.8 were recorded (Freeman, 1983). Below the discharges the increased respiration of the more heterotrophic communities reduces the pH fluctuation to a degree (Freeman and McFarlane, 1982). However fluctuations from 7.1 to 8.25 were observed at Site D on 9 February 1982 (Freeman, 1983). These data indicate that the pH conditions in the Manawatu River favour bacterial rather than fungal development in response to organic addition (Section 2.3.4.8).

The pH fluctuations also interact with ammonia added by the dischargers resulting in levels of unionised ammonia exceeding the recommended maximum levels of the United States Environmental Protection Agency at Site D during the late afternoon on occasion (Freeman and McFarlane, 1982).

2.2.2 WATER RIGHTS

Prior to January 1983 the discharge of the effluent to the Manawatu River below Palmerston North was authorised by water rights, and Pollution Advisory Council permits, under the Water and Soil Conservation Act 1967. However the results of a number of water quality surveys, especially those of Currie (1977), showed that the conditions were far from satisfactory (Section 2.2.1) and in December 1977 the Manawatu Regional Water Board (MRWB) established the Lower Manawatu River Technical Committee (LMRTC). This committee comprised members of the MRWB, the local county council, the dischargers, and the Wellington Acclimatisation Society and was asked to report on the following (LMRTC, 1978):

1. The definition of acceptable water quality in the lower Manawatu River.
2. Evaluation of parameters to achieve acceptable water quality.
3. Evaluation of the waste load allocation between the dischargers.

Six meetings were held between December 1977 and January 1980 and studies of river travel times, mixing and organic material removal rates (Rutherford and Currie, 1979; Rutherford *et al.*, 1982) were initiated.

The committee adopted the use of an in-river 5 day Biochemical Oxygen Demand (BOD_5) limit of 5 $g.m^{-3}$ at the end of a defined waste mixing zone at Site D (Fig. 1.2) as the primary method of maintaining water quality in the river. This value was not to be exceeded on average during the day nor at all during the night with the dual aims of preventing deoxygenation to below the dissolved oxygen concentration of 5 $g.m^{-3}$ required by the D

classification of the river and controlling sewage fungus growth. This in-river BOD_5 level was assumed to be suitable for controlling sewage fungus to within the waste mixing zone based on the findings of Curtis *et al.* (1971) and observations in the Manawatu River during the February 1977 survey (Currie, 1977). It was assumed that this BOD_5 limit would also prevent deoxygenation based on observations during the 24 hour surveys in December 1975 and February 1977. No attempt was made to model the effect of BOD_5 on dissolved oxygen because it was believed that the large number of factors involved in controlling oxygen levels (i.e., BOD_5 decay, algal and sewage fungus respiration, algal production and reaeration) made this impossible (Currie, 1977).

Once this in-river BOD_5 limit was established the BOD_5 loading of each discharger was decided based on the raw BOD_5 received and the anticipated BOD_5 decay between each outfall and between the BCWS outfall and Site D (the end of the "mixing zone"). It was anticipated that reducing the BOD_5 loading would reduce the heterotrophic biomass within the mixing zones and thus reduce the BOD_5 decay rates below the levels observed in studies on 22 March 1979 (Currie and Rutherford, 1982). However no information on this effect was available and so the BOD_5 decay rates (k_1 values, See Section 2.3.6) used in the water right allowable waste loading equations were derived by multiplying the observed reach k_1 values on 22 March 1979 by 0.66 (Rutherford and Currie, 1979; LMRTC, 1980).

The adoption of this waste loading control strategy allows the dischargers to vary their waste loading according to the river flow (Fig. 2.5). The allowable loads are not variable below the 96% low flow of 13 $m^3.s^{-1}$ which defines the maximum treatment required. This approach to water quality management is unique in New Zealand and gives the dischargers greater flexibility in their waste treatment and also allows full use of the assumed river assimilative capacity for BOD_5.

Limits were also placed on the allowable effluent levels of pH, temperature, suspended solids, grease and oil. Ammonia

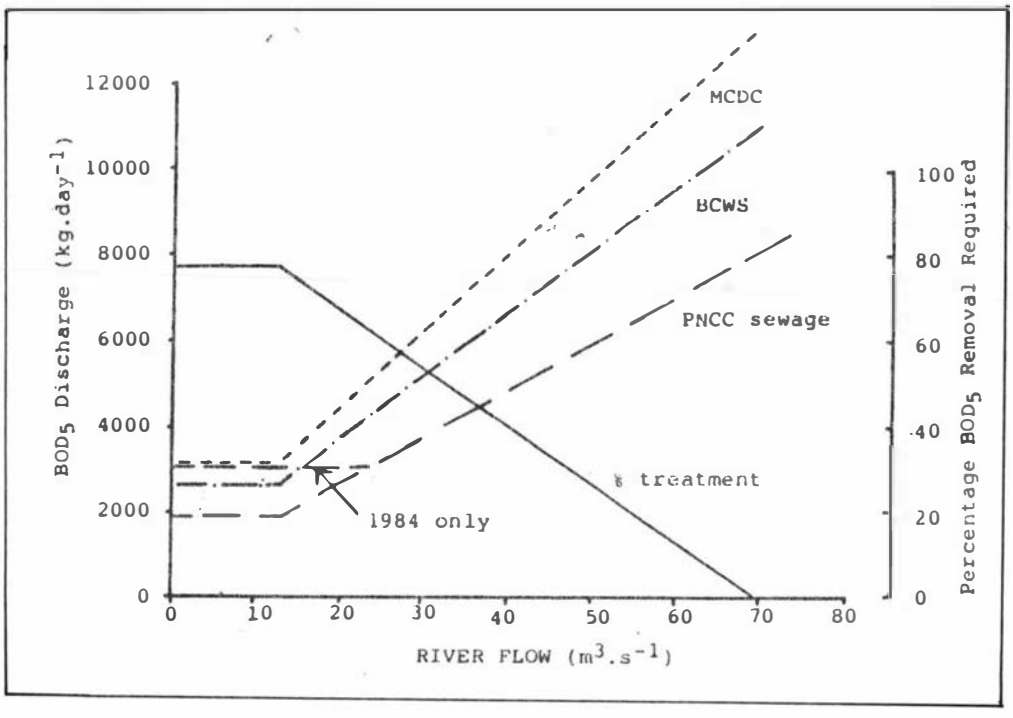


Figure 2.5: Permissible BOD_5 Loads to Manawatu River and Required Wastewater Treatment Under New Water Rights (adapted from LMTRC, 1980).

discharge was not limited due to doubts about the suitability of the recommended guidelines but it was acknowledged that ammonia limits might need to be implemented at a later date if problems arose. Nutrients were not limited since prolific algal growth was observed upstream of the discharges and the class D classification standards did not mention nutrients (Currie, 1977; LMRTC, 1978).

New Zealand Pharmaceuticals and the Linton Military Camp were not required to reduce their inputs since these were negligible compared to the three main discharges (LMRTC, 1978) (Section 2.2.6).

The dischargers were given three to four years to design and install the required treatment facilities to meet their new water rights which came into effect on 1 January 1983 (BCWS and MCDC) and 1 January 1984 (PNCC).

The salient features of the individual rights are discussed further in Sections 2.2.3 to 2.2.5.

2.2.3 PALMERSTON NORTH CITY CORPORATION

2.2.3.1 Introduction

The Palmerston North City Corporation (PNCC) receives the sewage flow from the city's population of 61,000 and some of the associated industries. The details of the effluent characteristics and present water right requirements are given in Tables 2.1 and 2.2 and Figure 2.5. The point of discharge to the river is shown in Figure 1.2.

Until January 1984 the PNCC operated a primary sedimentation treatment plant but during 1984 alum flocculation was also utilised to reduce the organic loading during low flows. From 5 March 1985 two aerated lagoons, with a total hydraulic residence time of approximately four days (Anderson 1984), were used to further reduce the organic load to the level required by the new water right.

TABLE 2.1: Summary of PNCC Effluent Characteristics

Parameters	Values	References
Dry weather Daily flow (m^3)	21,000	Anderson (1984)
Primary Effluent Characteristics:		
(a) Main Components	Carbohydrate, organic acids, protein and some fats.	Painter (1971)
(b) Expected BOD_5 : N: P ratio	100:20:3	Cooke *et al.* (1980)
(c) Average BOD_5 ($g.m^{-3}$)	285	Anderson (1983)
Discharge Pattern		
(a) Daily	diurnal pattern	Anderson (1984), Cooke *et al.* (1980)
(b) Weekly	± constant 7 days	Anderson (1983)
BOD_5 loading to river ($k.g.d^{-1}$)		
(a) Pre-1984	6000	Anderson (1983)
(b) 1984 maximum allowable at river Q < 25 $m^3.s^{-1}$	3238	Gilliland (1984)
(c) Post 1984 maximum allowable at 13 $m^3.s^{-1}$	1954	Gilliland (1984)
Effluent Treatment		
(a) Pre-1984	Primary sedimentation	Cooke *et al.* (1980)
(b) 1984	Primary sedimentation plus alum flocculation as required	Anderson (1984)
(c) Post 1984	Primary sedimentation plus aerated lagoons	Anderson (1984)

Table 2.2

Main Industrial Inputs to PNCC Sewage (Campbell, 1984)

Industry	Approximate Volume ($m^3.d^{-1}$)	Approximate BOD_5 loading ($Kg.d^{-1}$)	% of Total BOD_5 loading to sewers
Fermentation Industries Ltd	500	1300-3300	20-40
Lion Breweries Ltd	150	225	3
Milk Processing PN Ltd	300	300-600	4-8
Glaxo NZ Ltd	300-600	100-250	1.5-3.5
	1400	2000-4000	30-50

Primary sedimentation is also used as the treatment method by Hamilton City, the only other inland New Zealand city (population > 20,000) discharging to a river. However the Hamilton effluent differs in that it is also chlorinated prior to discharge. Most inland New Zealand towns discharging to rivers treat their waste in oxidation ponds, with or without prior primary treatment (Ferrier *et al.*, 1982).

2.2.3.2 General Characteristics of Organic Material in Domestic Sewage

The organic material composition of the PNCC effluent has not been studied and is expected to be affected by the industrial inputs (Section 2.2.3.3). However consideration of the literature relating to normal domestic sewage, without industrial inputs, does give some insight into the types of organic compounds expected in sewage. The main components are:

a) *Carbohydrates*: Raw sewage contains about 70 $g.m^{-3}$ of sugars in solution with glucose accounting for 50% of this amount. Sucrose and lactose are also important with smaller proportions of galactose, fructose, xylose and arabinose (Painter and Viney, 1959). Most of these sugars would remain after primary sedimentation treatment but some of the 30-40 $g.m^{-3}$ of carbohydrates in suspension (mainly high molecular weight polymers e.g., cellulose and starch) would be removed.

b) *Fats and Grease*: These usually total 40-100 $g.m^{-3}$ in raw sewage (Painter, 1971) but the data from the Manakau sewage treatment system indicates that primary sedimentation should produce a reduction of approximately 60% (ARA, 1983).

c) *Protein and Amino Acids*: In general free amino acids are under 5 g $N.m^{-3}$ (i.e. approximately 30 g protein.m^{-3}) whilst bound amino acids (peptide and protein) are 4-15 g $N.m^{-3}$ (approximately 25-95 $g.m^{-3}$ protein) in raw sewage (Painter, 1971). Primary sedimentation of the Manakau City sewage reduced the organic nitrogen content by 26% (ARA, 1983).

d) *Soluble acids*: Acetic acid is always the main volatile fatty acid in raw domestic sewage (6-37 $g.m^{-3}$) followed by propionic acid, butyric, valeric, formic and caproic acids. Together these account for 90% of the total volatile acid content (Painter, 1971).

Non-volatile acids account for about half the total organic acid content (Painter, 1971).

2.2.3.3 PNCC Wastewater Characteristics and Industrial Inputs

The normal pattern of BOD_5 loading for a domestic waste is one where low loadings occur between midnight and early morning, rising quickly to a peak in mid-morning, then declining from late afternoon to the low nighttime level (Painter, 1971). For the PNCC effluent (Fig 2.6) this pattern is affected by the three hour delay due to reticulation and treatment (Cooke *et al.*, 1980) and the discharge of the effluent from Fermentation Industries Ltd, a factory producing yeast (Anderson, 1984). This factory contributes approximately 500 $m^3.d^{-1}$ of effluent with a BOD_5 load of 1300 to 3300 $kg.d^{-1}$ (Table 2.2). This load consists largely of dissolved carbohydrates and is discharged over a period of a few hours daily except on Sundays, when no discharge occurs (Anderson, 1984). During the period when this discharge passes through the primary sedimentation tanks the effluent BOD_5 increases from the normal day-time level, of 200 to 300 $g.m^{-3}$, to 400 to 650 $g.m^{-3}$ (Anderson, 1983).

The other major industrial inputs to the PNCC sewage are listed in Table 2.2. The combined industrial discharges contribute approximately 30 to 50% of the total BOD_5 loading to the sewage treatment plant and their relatively high concentrations compared with raw domestic sewage result in the effluent from the primary sedimentation tanks having a higher average BOD_5 concentration (approximately 285 $g.BOD_5.m^{-3}$) than expected for New Zealand, primary treated, domestic sewage without any industrial input (110 to 130 $g.BOD_5.m^{-3}$) (ARA, 1983).

2.2.4 MANAWATU COOPERATIVE DAIRY COMPANY (MCDC) DISCHARGE

2.2.4.1 Introduction

This company produces butter, milk powder and casein and discharges its waste to the Manawatu River just below Site B (Fig. 1.2).

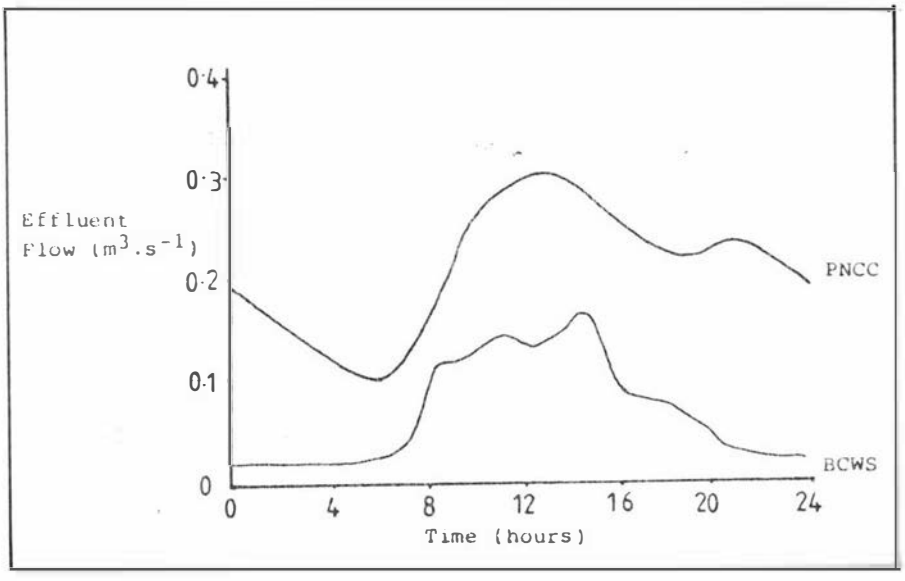


Figure 2.6: Typical PNCC and BCWS Effluent Hydrographs (after Cooke et al, 1980).

The factory was one of 81 dairy factories operating in NZ in 1980 (Galpin, 1981). Of these 22 discharged wastes directly to rivers and streams making dairy factories one of the most important sources of industrial waste pollution. Of the remaining factories six discharged to municipal sewers, seven discharged to the sea, two to estuaries and 45 discharged onto the land by spray irrigation (Galpin, 1981).

2.2.4.2 Wastewater Characteristics

The characteristics of the MCDC effluent are summarised in Table 2.3. The main organic constituent of dairy waste is lactose with casein and fats also being present (Painter, 1971). However the concentrations vary widely depending upon the type of products and technology employed (Jones, 1974; Barnett *et al.*, 1982). In a large multi-product factory, such as MCDC, waste composition would also be expected to vary as milk is diverted to different products. However all dairy products contain large amounts of readily biodegradable organic substances and in the BOD test the initial oxygen demand is high with about 50% of the total oxygen demand exerted in the first 24 hours (Barnett *et al.*, 1982).

The effluent's nutrient composition has not been studied but the available information (Marshall, 1976) suggests that a factory such as MCDC should produce an effluent with adequate total phosphorus and total nitrogen to allow bacterial growth utilising the available carbon although the BOD_5:N of 100:4.3 is slightly below the optimum of 100:5 (Section 2.3.4.4). However the location of the outfall 3.8 km below the PNCC effluent discharge, which contains nutrient in excess to that required for its BOD_5 oxidation, reduces the likelihood of nutrient limitation of sewage fungus growth in response to the addition of the MCDC effluent.

The volume and quality of the MCDC effluent varies seasonally with peak effluent production of 5000 to 6000 $m^3.day^{-1}$ for January through March. Very little discharge occurs between May and June as the cows prepare for calving in late July to early September (Meredith, 1982).

Table 2.3

Summary of MCDC Effluent Characteristics

Parameters	Values	References
Daily flow (m^3)	up to 6,000	Meredith (1982)
Effluent Characteristics:		
(a) Main components	lactose, casein and some fats	Painter (1971)
(b) Expected ratio BOD_5:N:P	100:4.3: ND	Marshall (1976)
(c) Average BOD_5 (i) 1982/83 season	($g.m^{-3}$) 2500	Meredith (1983)
(ii) 1983/84 season	($g.m^{-3}$) 1400	Meredith (1984)
Discharge pattern:		
(a) daily	± constant	Meredith (1982)
(b) weekly	± constant	Meredith (1982)
BOD_5 loading to river ($kg.d^{-1}$):		
(a) Pre-1983	up to 35,000	Meredith (1982)
(b) Post-January 1983 permitted at river Q < $13m^3.s^{-1}$	3300	LMRTC (1980)
Effluent Treatment:		
(a) Pre-1983	none	Meredith (1983)
(b) Post-1983	irrigation	Meredith (1983)

The effluent flow has increased in recent years with the opening of a casein factory in 1981 and an increase in milk intake. In 1977 the maximum discharge was 1610 $m^3.d^{-1}$ (Meredith, 1982). The casein factory usually contributes about 85% of the BOD_5 load and over 90% of the effluent volume. The milk powder and butter factories each contribute approximately 7% of the total BOD_5 load and approximately 2-4% of the effluent volume. Effluent production is quite variable throughout the day and does not follow any clear daily pattern (Meredith, 1982).

In order to meet the requirements of the new water rights (Figure 2.5, Table 2.3) the company has installed a spray irrigation system for disposal of waste surplus to that which may be **discharged** to the river. As stated earlier, this form of disposal is utilised by about half of the dairy factories in New Zealand. Long-term studies have shown that spray irrigation of dairy wastes creates a more favourable biological and chemical environment for pasture production (McAuliffe, 1978; Barnett *et al*, 1982). This form of disposal is particularly appropriate here because the company's water right allows high discharge to the river when high river flows occur (Fig. 2.5). Such conditions usually coincide with wet periods when conditions for land disposal are least favourable. However this method of meeting the water right conditions is not expected to cause large changes in the organic composition of the effluent discharged to the river as is expected with the biological treatment processes adopted by the other discharges (Sections 2.2.3 and 2.2.5). Nevertheless, in the summer of 1983/84 the MCDC reduced the effluent's BOD_5 concentration by approximately 45% on average compared with the previous summer by irrigating the most concentrated waste lines (Meredith, 1984).

2.2.5 BORTHWICKS CWS (BCWS) DISCHARGE

2.2.5.1 Introduction

New Zealand's export slaughter houses process

approximately 40 million animals a year. Eleven of these slaughter houses discharge to rivers or estuaries (Cooper, 1982) and these represent a large potential source of pollution.

The BCWS meatworks is of medium size by comparison with other New Zealand export meat processing works (Cooper *et al.*, 1979), having four sheep slaughter chains. In addition to the slaughter-house waste, the BCWS effluent contains fellmongery effluent and the wastewater from the adjacent Kiwi Bacon Company (Hinde, 1982).

2.2.5.2 Wastewater Characteristics

The characteristics of the BCWS effluent are summarised in Table 2.4. Slaughter-house wastewaters are largely organic in nature and contain high levels of protein and fat. Typical values for a primary treated effluent are 70 $g.m^{-3}$ and 500 $g.m^{-3}$ respectively. Other organic compounds, such as carbohydrates, are present in comparatively low concentrations (< 100 $g.m^{-3}$) (Cooper *et al.*, 1979). About 40-50% of the BOD_5 is exerted in the first 24 hours of the test indicating that a significant portion of the waste is readily biodegradable under aerobic conditions (Cooper, 1982).

Studies of the BCWS effluent's nutrient composition (Cooke *et al.*, 1980) show that the BOD_5:N:P ratio of 100:6:0.8 is approximately equal to the optimum for *Sphaerotilus* growth or waste water treatment of 100:5:1 (Section 2.3.4.4).

Although the fellmongery discharge represents only a few percent of the total flow it would be expected to contribute approximately 20% of the total works effluent Chemical Oxygen Demand (COD) (Cooper, 1982) and contributes lime and sodium sulphide to the wastewater on an intermittent basis (Cooke *et al.*, 1980).

No information is available on the effluent from the Kiwi Bacon Factory but this would be expected to be similar to the meat-works waste, comprising largely slaughter board wastes.

Table 2.4

Summary of BCWS Effluent Characteristics

Parameters	Values	References
Daily flow (m^3)	up to 12000	Hinde (1984)
Primary effluent characteristics:		
(a) Main Components	Protein and fats with some carbohydrate	Cooper et al. (1979
(b) Expected ratio BOD_5:N:P	100:6:0.8	Cooke et al. (1980)
(c) Average BOD_5 ($g.m^{-3}$)	1000	Hinde (1983)
Discharge pattern:		
(a) daily	diurnal pattern	Cooke et al. (1980)
(b) weekly	up to 5 days/week	Francis (1985)
BOD_5 loading to river ($kg.d^{-1}$)		
(a) Pre-1983 :	up to 12000	Hinde (1983)
(b) Post-January 1983 permitted at river Q < $13m^3.5^{-1}$	2722	LMRTC (1980)
Effluent treatment:		
(a) Pre-April 1984	primary	Hinde (1984)
(b) Post-April 1984	primary plus anaerobic lagoon treatment	Hinde (1984)

Between about 300 and 470 pigs are slaughtered on Mondays, Wednesdays and Fridays from August to November (Hinde, 1982). This is calculated to contribute about 400-600 kg BOD_5 per killing day (derived from Cooke *et al.*, 1980).

Despite the water right's requirement that the BOD_5 loading to the river be reduced from 1 January 1983 (LMRTC, 1980), the BCWS effluent received treatment by primary sedimentation alone until April 1984 (Gilliland, 1984). Seven other New Zealand meatworks discharging to rivers or estuaries utilise primary sedimentation as the sole treatment method (Cooper *et al.*, 1979). Some further unspecified degree of treatment is achieved in the two kilometre long, open, anaerobic drain which conveys the effluent from the sedimentation tanks to the river discharge site just below Site C on the Manawatu River (Fig. 1.2). However for much of the period from January 1983 to April 1984 the company was in violation of its water right conditions.

As shown in Figure 2.5, the present water right requires that the peak BOD_5 loading of 12000 $kg.day^{-1}$ be reduced to a maximum of 2722 $kg.day^{-1}$ at river flows of 13 $m^{-3}.s^{-1}$ or less with lesser reductions at flows between 13 $m^3.s^{-1}$ and 70 $m^3.s^{-1}$ (LMRTC, 1980). Since April 1984 anaerobic lagoons have been installed to provide secondary treatment to meet the 2722 BOD_5 $kd.day^{-1}$ requirement continuously. These ponds are also employed by one of the other thirteen NZ meatworks employing secondary treatment. Other processes employed are irrigation, aerobic treatment, physico-chemical treatment, anaerobic-aerobic lagoons in series, aerobic plastic media towers (Cooper, 1982) and anaerobic upflow sludge blanket systems (Anonymous, 1983). Anaerobic lagoons have been shown to be capable of reducing meatworks effluent BOD_5 by 83% but the anaerobic degradation of organic nitrogen compounds also results in high effluent ammonia levels (Cooper, 1982). An increase in the average ammonia concentration from 24 to 58 g NH_3-$N.m^{-3}$ was recorded in one three-month study of an anaerobic pond system (Cooper, 1982). This study also showed that anaerobic treatment produced no significant removal of total nitrogen. The pre-treatment phosphorus levels were not

determined in this study but comparison of the total phosphorus levels of the anaerobic pond effluent and the BCWS primary treated effluent (Cooke *et al.*, 1980) indicates that no significant removal of total phosphorus had occurred.

To overcome the problem of high effluent ammonia levels anaerobic pond effluents are usually treated further in aerobic lagoons or irrigated to land (Cooper, 1982).

Anaerobic treatment of meatworks wastewaters which include fellmongery effluents, such as the BCWS effluent, inevitably results in odour problems due to evolution of hydrogen sulphide (Cooper, 1982). To avoid this problem the fellmongery waste will be separated and will bypass the lagoons. Where fellmongery waste treatment is required anaerobic lagoons, utilising red photosynthetic bacteria which oxidise sulphide to elemental sulphur and sulphate, are frequently used. These ponds can produce 90% removal of sulphide and 70% removal of COD (Cooper, 1982).

2.2.5.3 Diurnal and Seasonal Flow Variations

The typical diurnal flow pattern for effluent leaving the primary sedimentation tanks is given in Figure 2.6. Where the effluent is discharged without treatment in the anaerobic lagoons, this results in a peak discharge to the river during the afternoon (Currie, 1977) due to the delay in passage through the drain. Anaerobic treatment of the waste will reduce the variation in the discharge pattern.

For much of the killing season (November to August) waste discharge occurs on five days per week but from April the number of slaughter chains operating and the days on which killing occurs progressively decreases resulting in lower effluent discharges.

During the off-season, which normally lasts from August or September to November, the Kiwi Bacon Company provides the only significant effluent flow on the three days per week on which pig slaughter occurs (Hinde, 1983).

2.2.5.4 Summary

The BCWS discharge has historically contributed a considerable amount of organic materials and nutrients to the Manawatu River, especially during summer. The anaerobic treatment is expected to be effective in reducing the effluent's organic concentrations but will have little effect on the nutrient load.

2.2.6 MINOR DISCHARGES

Two further minor waste discharges occur within the study reach. The Linton Military Camp discharges approximately 360 $m^3.d^{-1}$ of trickling filter treated domestic sewage contributing 5 kg $BOD_5.d^{-1}$ at a point between Sites A and B (Fig. 1.2). NZ Pharmaceuticals Limited, which manufactures steroid acids from animal bile, discharges effluent containing up to 136 kg $BOD_5.d^{-1}$ to the river about 400 m upstream of Site C.

These discharges are negligible by comparison to the three main discharges discussed above.

2.3 SEWAGE FUNGUS

2.3.1 COMPOSITION AND DESCRIPTION

The term sewage fungus (Butcher, 1932) is commonly used to describe the benthic macroscopic slime communities dominated by heterotrophic microorganisms, which occur in response to organic waste discharges. The community varies in structure from thin films and plumose fronds in rapid flowing water to long filamentous growths at more moderate velocities. Its colour may vary widely from white to brown, pink or orange (Curtis, 1969; Gray, 1982).

Synonomous terms for sewage fungus found in the literature are "abswasser-pliz" (Tiegs, 1939), slime infestation (Harrison and Heukelekian, 1958) and "heterotrophic biocoenosis" (Wuhrmann, 1954). As the community grows attached to stones it can be classed as epilithon and where it grows attached to macrophytes or algae it can be classed as epiphyton. Sewage fungus

communities have also been referred to as periphyton (Capblancq and Cassan, 1979), a term usually used to describe algal communities attached to all types of bottom supports in streams (Hynes, 1970).

Of 30 sewage fungus outbreaks examined from New Zealand streams 22 were dominated by the filamentous bacteria *Sphaerotilus natans* Kutzing, 1933 , five by zoogloeal bacteria, two by both these organisms and one by *Beggiatoa* sp. (Cooper, 1983). The data presented in Table 2.5 show that these were also the most frequent dominant organisms in sewage fungus growths at 178 sites surveyed in the United Kingdom (Curtis and Harrington, 1971). True fungi, stalked protozoans and algae were also occasionally dominant or co-dominant in sewage fungus in the UK survey. *Leptomitus lacteus* was rarely dominant in this survey but appears to have been an important component of sewage fungus in the larger European rivers (Butcher, 1932; Carpenter, 1970) especially at acid pH values (Liebmann, 1951; Vallin, 1958).

This shows that a number of different bacteria, fungi, protozoa and even algae can be dominant or co-dominant in sewage fungus communities. This variable nature makes sewage fungus a very difficult entity to define. However studies in New Zealand and the United Kingdom have shown the most important matrix forming sewage fungus organisms to be the bacteria *Sphaerotilus natans* and *Zoogloea* sp. (Curtis and Harrington, 1971; Cooper, 1983).

The structural strength of *S. natans* is provided by both the sheath or trichome, which surrounds the chains of cells and attaches to the bottom substrate by a hold fast (Phaup, 1968), and the capsule or slime layer surrounding the sheath. Acid hydrolysis of the sheath yielded reducing sugars (36% of dry weight), amino sugars (11% of dry weight), protein (27% of dry weight) and lipid (5.2% of dry weight) (Romano and Peloquin, 1963). The slime layer is polysaccharide in nature containing fucose, glucose, galactose, and glucuronic acid (Gaudy and Wolfe, 1962). Similar bacterial polysaccharide matrices (termed the glycocalyx) have been shown to provide the means of

Table 2.5

Occurrence of the Most Common Constituents of Sewage Fungus in a U.K. Survey

(Adapted from Curtis and Harrington, 1971)

(The data are expressed as a percentage of the 178 sites samples)

		Dominant or Co-dominant*	Abundant	Infrequent	Present at % of sites
Bacteria:	*Zoogloea* spp.	58.5	30.2	3.8	92.5
	Sphaerotilus natans	52.1	19.5	17.6	89.2
	Flavobacterium spp.	3.1	11.3	25.8	40.2
	Beggiatoa alba	6.3	6.9	14.5	27.7
Algae:	Diatoms	4.4	15.1	20.7	40.2
	Stigeoclonium tenue	3.1	5.7	1.9	10.7
Protozoa:	*Carchesium polypinum*	6.3	5.7	4.4	16.4
Fungi:	*Geotrichum candidum*	4.4	2.5	0.6	7.5
	Leptomitus lacteus	3.1	-	0.6	3.7
	Fusarium aquaeductum	1.9	-	-	1.9

* This column totals more than 100% because of co-dominance.

attachment of most bacteria to solid substrates (Costerton *et al.*, 1978).

The other predominant organism in sewage fungus, *Zoogloea* sp., also produces a polysaccharide extra-cellular slime in which the single cells are embedded.

The dominant matrix-forming organisms give cohesion to a diverse community of epiphytic, non-filamentous bacteria, protozoa and diatoms. Higher organisms such as rotifers, nematodes, insect larvae and annelid worms are also found associated with the slime.

As the river flows downstream from a discharge the activity of the heterotrophs progressively reduce the organic content of the water. Thus the conditions become increasingly less favourable for the heterotrophs and a transition to an autotrophic community occurs (Fig. 2.7) (Sections 2.3.4.2, 2.3.4.3).

This shows that a continuum exists between phototrophic and heterotrophic dominated communities. Sewage fungus is therefore not a clear cut, defineable entity but rather a mixed benthic community towards the heterotrophic end of this continuum.

The position of benthic communities on this continuum may be determined both qualitatively and quantitatively. Qualitative determinations focus on the macroscopic appearance of the assemblage (Wuhrmann, 1974, Section 3.3.2) while the quantitative assessments use the ratio of total biomass to photosynthetic pigment. Examples are given in Table 2.6.

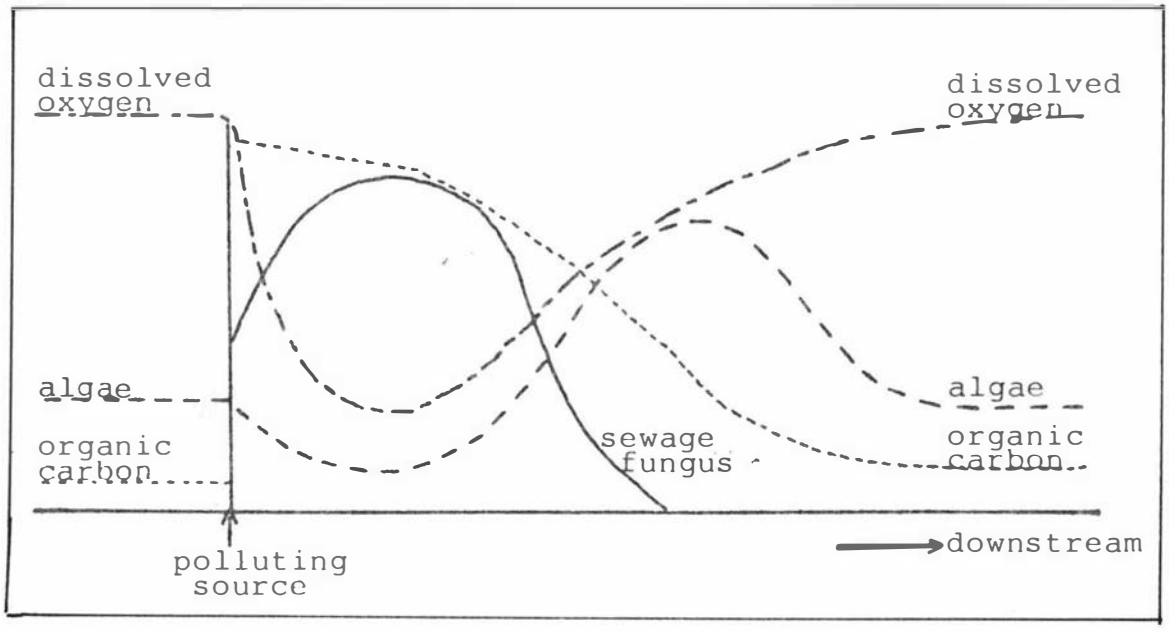


Figure 2.7: Qualitative Trends in Organic Pollution and Recovery in a River (from Hynes, 1960).

TABLE 2.6 Quantitative Indices Describing a Community's Position on the Heterotrophic-Autotrophic Continuum.

Index	Calculation	Reference
Autotrophic Index (AI)	$\frac{\text{Ash-free dry weight}}{\text{Chlorophyll a}}$	Weber (1973)
P/H ratio (P/H)	$\frac{\text{Absorbance at 663 nm of methanol extract}}{\text{Ash-free dry weight}}$	Wuhrmann (1974)
Trophic Index	$\frac{\text{Chlorophyll a-derived organic carbon}}{\text{Ash-free dry weight-derived organic carbon}}$	Clark *et al.* (1979)
Heterotrophic Index (HI)	$\frac{\text{ATP-derived organic carbon}}{\text{chlorophyll a}}$	Horner and Welch (1981)

The AI value is preferable to the P/H ratio because it distinguishes between chlorophyll a and its degradation products and unlike the Trophic Index does not involve assumptions regarding the ratios of organic carbon to chlorophyll a and ash-free dry weight. AI values for pure algal cultures vary from 50 to 100 (Cooper and Wilhm, 1975) whereas for sewage fungus much higher values occur. For example, *Sphaerotilus natans* dominated slimes developing in the polluted river Agout, France, had AI values ranging from 300 to 25000 during the course of one year of study (Capblancq and Cassan, 1979a).

However high ash free dry weight values can also result from samples containing significant amounts of non-viable settled detritus (Clark *et al.*, 1979) or sediments with a high clay content (Dankers and **Laane**, 1983). Thus AI values obtained from such samples must be viewed critically. These problems may be overcome by using the HI value of ATP-derived organic carbon to chlorophyll a if the equipment and time required for ATP analysis are available.

The ratios of photosynthetic oxygen production to respiratory oxygen consumption (P/R values) also provide a potential means of quantification of a community's position on the heterotrophic-autotrophic continuum (Traaen, 1975). By definition heterotrophic communities such as sewage fungus have P/R values less than 1 whereas autotrophic communities have values greater than 1 (Odum, 1956).

However there are difficulties with this definition because photosynthetic oxygen production is strongly influenced by incident light. Thus a given community may have a P:R value of more than one on a sunny day and less than one on a cloudy or turbid day.

This shows that the description of a benthic community as sewage fungus is imprecise and can be misleading. A clearer understanding of the position of the community on the heterotrophic-autotrophic continuum is provided by the description of the overall macroscopic abundance of the heterotrophic and autotrophic components of the community and microscopic examination of the growths. Quantitative description using the Autotrophic Index and the measurement of functional characteristics such as P/R values also aid description. However reference to environmental conditions, especially light, is important for interpretation of the functional parameters.

2.3.2 ORGANIC SUBSTRATES UTILISED BY SEWAGE FUNGUS

The sewage fungus matrix forming organisms require carbon for growth and as an energy source. *S. natans* has been shown to be capable of growth on a variety of low molecular weight carbohydrates, alcohols and organic acids in batch culture (Table 2.7). The observation of no growth response to lactose reported by Stokes (1954) is anomalous in that it contradicts the findings of Linde (1913) and Scheurning and Hohnl (1956) and the frequent observation that dairy factory discharges, in which lactose is the predominant carbohydrate (Painter, 1971), cause sewage fungus growths in rivers in New Zealand (Cooper, 1983) and Eire (Gray and Clarke 1984).

TABLE 2.7 Carbon Sources for *Sphaerotilus natans* (Batch Experiments)

Key: + growth on energy source
- no growth on energy source

Carbon Source	Growth Response	Reference
A) Carbohydrates:		
Glucose, galactose, maltose, sucrose	+	(Lackey and Wattie 1940; Stokes 1954; Linde 1913; Sheuring and Hohnl 1956)
fructose	+	(Scheuring and Hohnl 1956; Mulder and van Veen 1963)
lactose	+	(Scheuring and Hohnl 1956; Linde 1913)
lactose	-	(Stokes 1954)
xylose,	-	(Stokes 1954)
xylose, ribose	+	(Scheuring and Hohnl 1956)
arabinose, rhamnose	-	(Scheuring and Hohnl 1956; Stokes 1954)
cellobiose	-	(Scheuring and Hohnl 1956; (Lackey and Wattie 1940)
starch, inulin	-	(Scheuring and Hohnl 1956; Linde 1913
dextrin, cellulose, gum arabic	-	(Linde 1913)
maltodextrin	-	(Roberts 1978)
mannose	+	(Phaup 1968)
B) Alcohols:		
ethanol, butanol, glycerol	+	(Stokes 1954; Hohnl 1955; Scheuring and Hohnl 1956)
mannitol, sorbitol	+	(Stokes 1954; Hohnl 1955; Scheuring and Hohnl 1956)
methanol	+	(Hohnl 1955, Scheuring and Hohnl 1956)
C) Organic acids:		
Succinic, fumaric, butyric	+	(Stokes 1954)
lactic, pyruvic, acetic	+	(Stokes 1954, Scheuring and Hohnl 1956)
malic, propionic, malonic	+	(Scheuring and Hohnl 1956)
tartaric, oxalate	+	(Scheuring and Hohnl 1956)
B hydroxy butyric	+	(Mulder and van Veen 1963)
citric	-	(Mulder and van Veen 1963)
aspartic, glutamic	+	(Mulder and van Veen 1963; Scheuring and Hohnl 1956)
asparagine	+	(Scheuring and Hohnl 1956)
threonine, tyrosine	-	(Scheuring and Hohnl 1956)
glycine, cystine	-	(Scheruing and Hohnl 1956)
methionine	+	(Phaup 1968)

Wuhrmann (1974) compared the specific uptake rates for individual sugars and amino acids from a mixture fed to bore water in outdoor channels containing heavy *S. natans* dominated sewage fungus. Of the sugars added sucrose, glucose and fructose were taken up at similar rates whereas the uptake of galactose was much slower. Of the amino acids aspartate, glutamate, leucine and glycine were removed more readily than lysine and histidine when added at similar concentrations. Cystine was not removed at all.

Continuous culture experiments have shown that at a glucose concentration of 10 $g.m^{-3}$ the growth rate of *S. natans* in pure culture is half its maximal value for unlimited substrate (**Lau** *et al.*, 1984).

The carbohydrates and alcohols shown to be utilised by *Zoogloea ramigera* are listed in Table 2.8. This organism is able to denitrify (Williams and Unz, 1983) and can use tyrosine as its sole source of carbon and nitrogen if vitamin B_{12} is provided (Crabtree and McCoy, 1974).

TABLE 2.8 Carbohydrates and Alcohols utilised by *Zoogloea ramigera* (Crabtree and McCoy, 1974)

Carbohydrates:

xylose	mannose	rhamnose	lactose
fructose	arabinose	fucose	trehalose
glucose	galactose	maltose	cellobiose
sorbose	sucrose	salicin	

Alcohols:

ethanol	sorbitol	mannitol	inositol
glycerol			

Leptomitus lacteus can utilise acetate and most low molecular weight fatty acids but not sugars (Schade, 1940; Burnett, 1968). Butcher (1932) states that *L. lacteus* tends to grow at lower concentrations of organic effluent than *S. natans* and for

this reason tends to be more frequent in the larger European rivers where greater dilutions are possible than in rivers in the UK.

Beggiatoa alba Vaucher, 1803; Trevisan, 1845 is a filamentous bacterium but unlike *S. natans* has no sheath and is motile, exhibiting bending and flexing and gliding motility (Leadbetter, 1974). Its metabolism is mixotrophic with both hydrogen sulphide and organic nutrients being oxidised. Hence *Beggiatoa* is typically found in rivers receiving wastes containing sulphide (Liebmann, 1951). The following organic compounds can act as sole carbon source for metabolism and growth: ethanol and the organic acids acetate, fumarate, lactate, malate, pyruvate, and succinate (Mezzino *et al*, 1984). Growth was not obtained using glucose, glycerol, asparagine, aspartate, glutamate, citrate, formate or glycoxylate whilst only one strain grew weakly on isocitrate as sole carbon source. None of the strains tested by these authors could hydrolyse gelatin, starch or casein but all strains were able to grow using nitrate, nitrite, ammonia or casamino acids as sole nitrogen source.

2.3.3 EXTRACELLULAR POLYSACCHARIDE MATRIX

In addition to its role in substrate attachment and providing cohesion to the sewage fungus community (Costerton *et al*, 1978; Geesey, 1984), the extracellular polysaccharide matrices produced by *S. natans* and *Zoogloea ramigera* appear to have the following nutritional functions:

(i) *Food Reserve*: The observation that the thickness of the matrix of *S. natans* varies with the organic nutrient supply (Phaup, 1968) indicates that it may act as an extracellular food reserve.

(ii) *Adsorptive surface*: The matrix of *Zoogloea ramigera* has been shown to adsorb dissolved organic matter, such as amino acids,(Dugan *et al.*, 1971) and metallic ions and can accumulate significant concentrations of heavy metals (Gray & Clarke, 1984).

(iii) *Extracellular Enzyme Attachment Site*: It has been suggested that bacterial extracellular slime layers aid the uptake of organic compounds by conserving and concentrating extracellular enzymes (Costerton *et al.*, 1978; Tonn and Gander, 1979). These enzymes cleave large organic compounds producing compounds small enough to be taken up into the bacterial cells.

Through these mechanisms the extracellular polysaccharide matrices produced by the two most common sewage fungus organisms appear to greatly enhance the ability of these organisms to grow in habitats where organic compounds are present in low concentrations and are often present only intermittently.

2.3.4 FACTORS CONTROLLING GROWTH

2.3.4.1 Introduction

For sewage fungus to develop a metabolisable organic substrate must be present in a sufficient concentration for the growth rate of one of the matrix forming organisms to exceed its loss rate.

Losses occur due to predation and sloughing due to current abrasion acting on its own and in concert with the death of the cells which maintain the filaments attachment to the substrate and the degradation of the extracellular slime.

This implies that there must be a threshold amount of organic material required for sewage fungus biomass to be macroscopically observable. However a number of other factors are involved in determining this threshold. These are reviewed in the following section.

2.3.4.2 Degradable Organic Materials

Most effluents that promote sewage fungus growth are comprised of a wide range of organic constituents (see Sections 2.2.3 to 2.2.5) so that the measurement of the concentration of organic material is usually carried out using comprehensive

measures rather than analyses for specific components. The most common measurement parameters used in studies of sewage fungus growth are the 5 day Biochemical Oxygen Demand (BOD_5), the Chemical Oxygen Demand (COD) or the Total or Dissolved Organic Carbon (TOC or DOC) tests.

An intensive survey of 178 sites in the UK, below domestic, industrial and agricultural discharges, indicated that heavy sewage fungus growth was associated with waters having BOD_5 concentrations of 5 to 30 $g.m^{-3}$ and DOC (GFC filtered) concentrations of 6 to 30 $g.m^{-3}$ (Curtis and Harrington, 1971).

Growths were not usual at less than 5 $g.m^{-3}$ BOD_5 or less than 6 $g.m^{-3}$ DOC and where growths occurred they tended to be light or moderate in extent. The concentration of total soluble carbohydrate, measured as glucose equivalents, ranged from less than 1.1 to 30 $g.m^{-3}$ at sites where sewage fungus occurred. However at the majority of sites the water contained 1.1-8.0 $g.m^{-3}$ of soluble carbohydrates, with a large proportion in the 1.1-3.0 $g.m^{-3}$ range.

Observations of sewage fungus growth at the upstream end of outdoor channels fed settled domestic sewage showed a similar pattern of response to BOD_5 (Fig. 2.8) (from Eichenberger, 1975).

These results are in agreement with those of Zimmerman (1961) who obtained growth of *S. natans* dominated sewage fungus in the same channel system when settled domestic sewage was diluted to give a BOD_5 of 5.5-6.5 $g.m^{-3}$ at a current velocity of 0.2 $m.s^{-1}$. Growth did not occur at this loading when the current velocity was reduced to 0.05 $m.s^{-1}$. In contrast, increasing the current velocity to 0.8 $m.s^{-1}$ allowed the growth of a *S. natans* dominated community at 3.3-3.5 $g.m^{-3}$ BOD_5 but not at 1.6-1.8 $g.m^{-3}$ BOD_5.

These data show that the threshold organic substrate concentration for the development of sewage fungus varies with current velocity (Section 2.3.4.5).

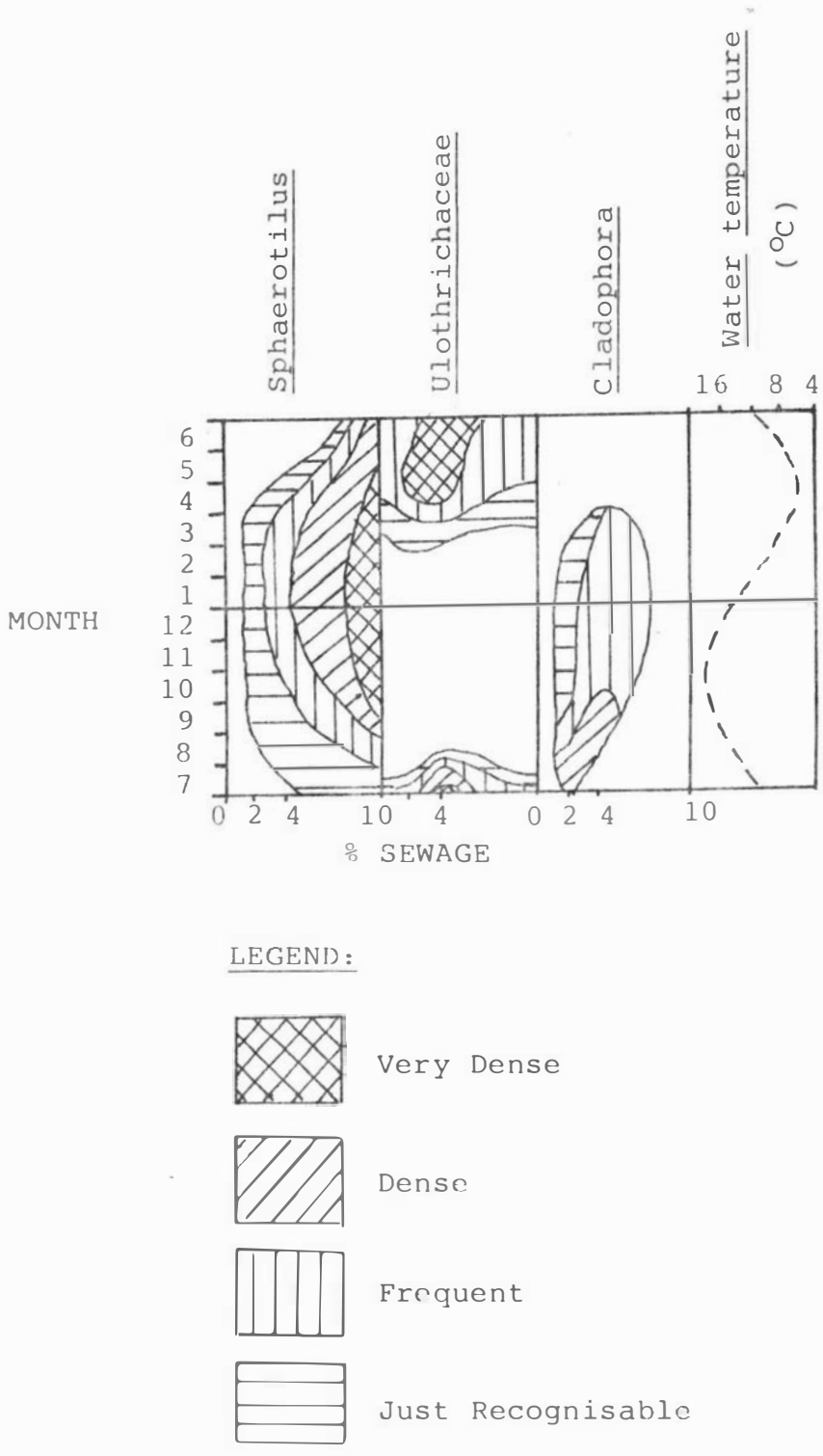


Figure 2.8: Growth of Micro-organisms in Outdoor Channels fed 0, 2, 4 & 10% Settled Sewage with a BOD_5 of 90 to 150 $g.m^{-3}$ (after Eichenberger, 1975).

However different effluents contain varying proportions of the easily assimilated low molecular weight compounds and the comprehensive BOD_5, COD and DOC tests are unable to distinguish such effects. Although the BOD_5 test does distinguish between degradable and recalcitrant organics it does not distinguish between readily and moderately degradable organics. Thus equivalent BOD_5 concentrations, achieved by dilution of primary treated and biologically treated sewage, did not produce comparable associations of microorganisms when fed to outdoor channels. With primary treated sewage increasing the BOD_5 from 2-2.5 $g.m^{-3}$ to between 4 and 6 $g.m^{-3}$ caused a shift in the community structure from phototrophic domination to a heterotrophic dominated *Sphaerotilus* community (Wuhrmann, 1954). By contrast when the domestic sewage received biological treatment, by the low rate activated sludge process, BOD_5 levels as high as 13 $g.m^{-3}$ did not result in *Sphaerotilus* dominance (Wuhrmann, 1954). This suggested that the biological treatment produced qualitative as well as quantitative changes in the composition of the sewage and that comprehensive measures such as BOD_5 and Total Organic Carbon (TOC) could not distinguish these.

On the other hand some industrial wastes contain higher proportions of easily assimilated organics than sewage and produce sewage fungus growth at much lower BOD_5 levels. Pulp and paper mills frequently produce effluents with significant amounts of low molecular weight carbohydrates, particularly glucose (Roberts, 1978; Gillespie and McKenzie, 1981). An increase in the in-river BOD_5 of only 0.7 $g.m^{-3}$ from one paper mill produced heavy sewage fungus growth (Hughes, 1969) and similarly sewage fungus occurred in the lower Columbia River at locations where pulp and paper mill effluent increased the BOD_5 by 0.4 $g.m^{-3}$ on average (Cormack and Amberg, 1959).

Studies of paper mill effluents by Roberts (1978) indicate that the low molecular weight components of this waste water are the most important in promoting sewage fungus growth. He found that glucose addition in the range of 0.5 to 10 $g.m^{-3}$ (i.e., approximately 0.5-10 $g.m^{-3}$ BOD_5) produced similar biomass of *S.*

natans dominated sewage fungus growth at the beginning of a 5.4 m long once-through laboratory channels after two weeks. However these results were obtained in the absence of light (Webb, 1982) and therefore do not relate directly to the stream situation (Section 2.3.4.3).

The addition of sufficient sucrose to borewater to produce concentrations of 3.0 and 1.5 g.sucrose.m^{-3} (i.e., BOD_5 approximately 3 and 1.5 g.m^{-3} respectively) produced *S. natans* growths in the first 30 m of outdoor channels (Edelmann and Wuhrmann, 1978). In the channel receiving 1.5 g.sucrose m^{-3} filamentous phototrophs also developed and after 8 to 18 days the ratio of daily photosynthetic oxygen production to total respiration (P/R) was 1.0 compared to 0.5 in the more highly loaded channel.

No detailed studies have been published relating specifically to the effects of slaughter-house or dairy factory wastes on sewage fungus growth.

This review suggests that sewage fungus control measures based on comprehensive organic material parameter such as BOD_5, COD or TOC are unlikely to be successful unless used with reference to a specific type of wastewater. Several studies have indicated that the low molecular weight organic compounds are primarily responsible for promoting sewage fungus growth. These can be partitioned by ultrafiltration using membranes with appropriate molecular weight cut-offs (Wilander, 1972; Addie *et al.*, 1973; Ogura, 1974; Gloor *et al.*, 1981). The use of this technique in combination with the measurement of organic material concentration by one of the comprehensive tests may provide a more reliable generalised method for controlling sewage growth. However it is clear that factors other than the concentration of organic materials also affect sewage fungus growth.

2.3.4.3 Solar Radiation

The ratio of phototrophs to heterotrophs in the benthic communities of outdoor channels fed domestic sewage has

been shown to vary according to the ratio of chemical energy input (E_s, kcal) to solar energy input (E_L,kcal) (Wuhrmann, 1974). The heterotrophs (mainly *S. natans*) rapidly overgrew the phototrophs at E_S/E_L ratios of 25-30, at temperatures 12 to 16°C, or 10, at temperatures of less than 10°C. At E_S/E_L values of less than 5 the heterotrophs were not macroscopically visually observable. This relationship is also expected to be affected by the nature of chemical energy input in terms of its availability to the heterotrophs (Wuhrmann, 1974, Sections 2.3.2 and 2.3.4.2).

Solar radiation effects on sewage fungus could be both direct, by inhibiting the growth of heterotrophs, and indirect, by increasing the water temperature (Section 2.3.4.7) and providing the energy for the growth of phototrophic species. Since filamentous algae provide an excellent support for *S. natans* growth (Edelmann and Wuhrmann, 1978), competition between sewage fungus and these phototrophs for benthic attachment sites is not likely to limit sewage fungus growth. However when the competing phototrophs are unicellular species their occupation of available benthic attachment sites may limit the heterotroph development. An increase in the phototrophic component of a sewage fungus community will alter its effects on water quality, for example by affecting the community's daytime oxygen consumption rate (Section 2.3.5).

The ozone layer, present in the Earth's atmosphere provides an effective barrier to high energy, ultra-violet (UV) radiation with wavelengths less than about 300 nm (Jagger, 1967). However, near-ultraviolet radiation (300-400 nm) reaches the Earth's surface and is capable of killing bacterial cells by a variety of actions including damage to enzymes involved in DNA repair, effects on membrane transport and metabolic systems, induction of growth delay and damage to cell membranes (Kelland *et al.*, 1983). Violet and blue light (380-480 nm) has also shown to inhibit some bacteria (Muller-Neugluck and Engel, 1961), and benthic algae (Antoine and Benson-Evans, 1983), but is generally of lesser importance than the higher energy UV wavelengths.

Light inhibition is more pronounced on non-pigmented than pigmented bacteria (Rheinheimer, 1980). This probably results from pigments providing protection by absorbing harmful wavelengths before they cause cellular damage.

Solar radiation varies seasonally at the temperate latitudes. For example near Palmerston North (latitude 40°21'S) the average daytime total radiation varied from 175 $W.m^{-2}$ in July 1983 to 400 $W.m^{-2}$ in December 1983 (Basher, 1983; McNaughton, 1984) and near-ultraviolet (UV) wavelengths (of 300-380 nm) comprise 5 to 6% of this radiation on average in New Zealand (Basher, 1983). However the shortest wavelengths in the near ultraviolet range apparently show an even greater annual variation. UV radiation of less than 313.2 nm ranged from 0.3 $W.m^{-2}$ at midday in clear weather in winter to 1.8 $W.m^{-2}$ in mid-summer in Washington DC (latitude 38°56.5'N) (Coblentz and Stair, 1944). Thus solar radiation inhibition could be expected to result in seasonal changes in sewage fungus growth and different growth rates in shaded and exposed areas.

S. natans-dominated sewage fungus has been observed to be restricted to shorter river reaches below organic discharges during summer than during winter in a number of field studies (Butcher, 1932; Cawley, 1958; Amberg and Cormack, 1959; McKeown, 1963, Phaup and Gannon, 1967) and outdoor channel studies (Eichenberger and Wuhrmann, 1966). These results indicate that during summer partial inhibition of growth due to solar radiation may result in a higher organic substrate concentration being required for *S. natans* growth. However the following mechanisms may also be involved in producing the seasonal variations in the reach length occupied by *S. natans*:

(i) seasonal variations in invertebrate grazing (Ormerod *et al.*, 1966)

(ii) more rapid removal of organics from the water column at higher river temperatures restricting the area with suitable organic concentrations for sewage fungus development (Cawley, 1958).

More convincing evidence of solar radiation reducing *S. natans* growth rate was provided by further observations in 0.15 to 0.2

m deep outdoor channels. *S. natans* biomass at the beginning of the channels fed diluted sewage was less dense during summer than during winter (Fig 2.6) (Eichenberger and Wuhrmann, 1966; Eichenberger, 1975) and was more dense in shaded than exposed areas, although the latter effect was not quantified (Eichenberger, 1975).

The heterotrophic growth rate during the first 14 to 20 days incubation at the beginning of these channels fed 12% sewage at different seasons was 5.6 $gAFDW.m^{-2}.d^{-1}$ at 65 to 150 $cal.cm^{-2}.d^{-1}$ total surface radiation (65-150 $W.m^{-2}$ average for 12 hours sunlight) but decreased to approximately $3gAFDW.m^{-2}.d^{-1}$ at 320 to 470 $cal.cm^{-2}.d^{-1}$ total surface radiation (310 to 455 $W.m^{-2}$ average for 12 hours sunlight) (Eichenberger, 1975).

The results of a laboratory study provide further evidence that solar radiation can inhibit *S. natans*. This showed that the growth and respiration of *S. natans* were inhibited by 120 $W.m^{-2}$ of artificial radiation, provided by a high pressure mercury lamp (750W) (Favre, 1975). This inhibition was entirely reversible, providing the time of irradiation did not exceed 2½-3 hours and the cells were in a medium permitting protein synthesis. The range of inhibiting wavelengths extended, with declining efficiency, from about 305 nm to 540 nm with partial inhibition disappearing at about 14 $W.m^{-2}$.

The nutrient status of the growth medium also influences the effects of light (Favre, 1975). In a medium containing inorganic nitrogen, light of wavelengths from 305 nm to 540 nm inhibited *S. natans* respiration whereas in an organic nitrogen media the inhibitory band only extended from 305 nm to 420 nm. This was attributed to the organic nitrogen medium enabling more efficient repair of damage caused by light and more rapid cellular replication.

Difficulties in applying Favre's results in the field arise due to the discontinuous spectrum of high pressure mercury lamps such as that used in his work (Fig. 2.9) compared with the

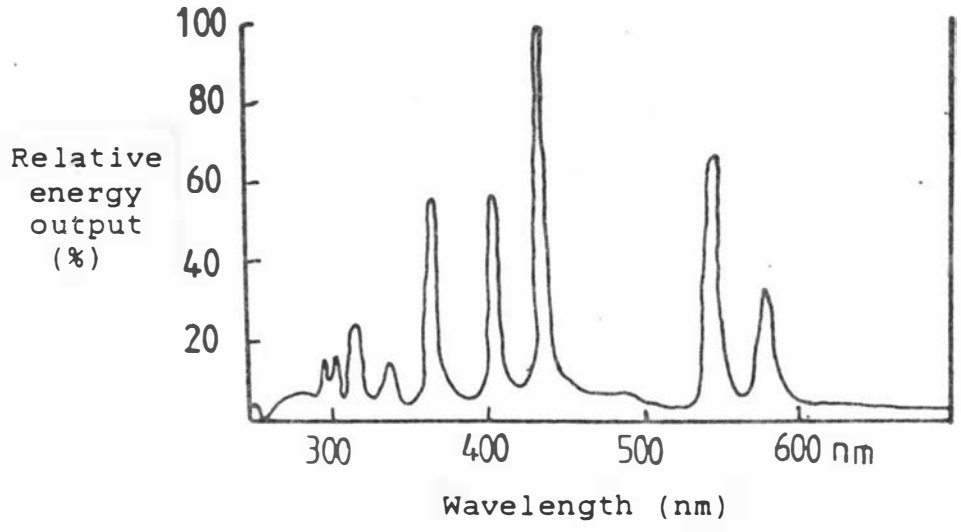


Figure 2.9: Spectral Output of 500 W High Pressure Mercury Vapour Lamp (after Claesson, 1964).

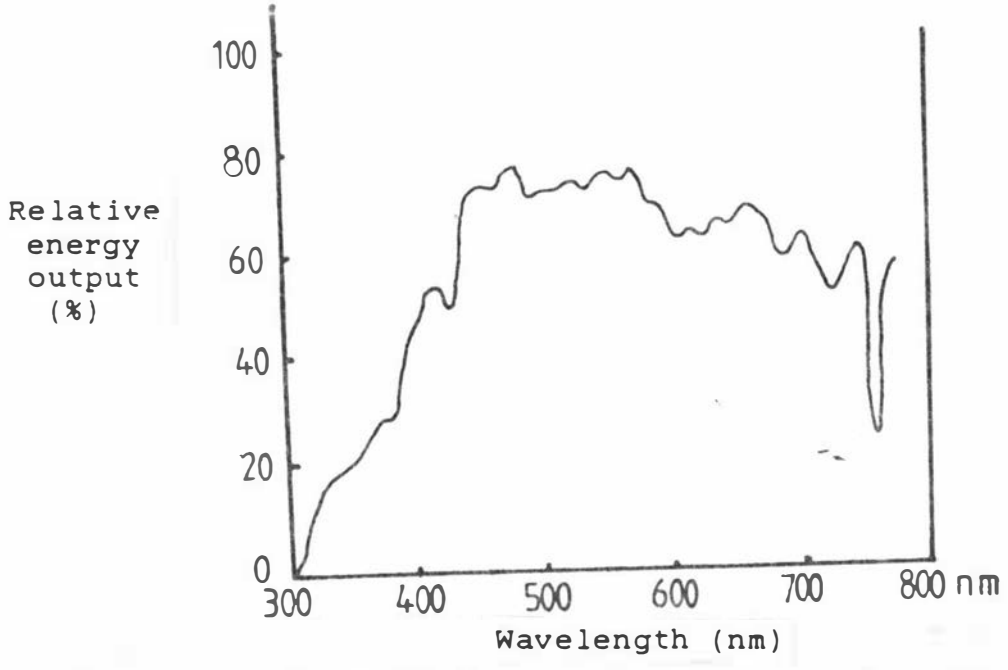


Figure 2.10: Spectral Energy Distribution of Daylight (mean of 16 skies with and without sun) (after Henderson & Hodgkiss, 1963).

continuous solar spectrum in nature (Fig. 2.10). Some photo-reactivation mechanisms, (e.g., the enzymes cleaving thymine dimers produced in DNA by UV radiation) are induced by visible light radiation (400-700 nm) (Pelczar *et al*, 1977). It is possible that the high presure mercury lamp's spectrum has relatively higher levels of the harmful wavelengths than sunlight but contains lower levels of the wavelengths required for the reactivation systems.

The comparison of the level of artificial radiation, at which Favre (1975) observed irreversible inhibition of *S. natans* (120 $W.m^{-2}$) with the solar radiation levels, at which growth occurred in the outdoor channels (455 $W.m^{-2}$) (Eichenberger, 1975) suggests that the radiation produced by the lamps is more harmful per unit of total radiation than natural solar radiation.

The absorption of the most damaging wavelengths by dissolved and suspended materials in natural streams may reduce light effects in nature. Although short wavelength radiation (<550 nm) penetrates 1 m of distilled water with nearly 100% transmission (Ruttner, 1963), in natural water dissolved and suspended organic materials absorb radiation such that the peak transmission generally occurs in the green region (480-550 nm) and transparency decreases rapidly towards the longer and shorter wavelengths (Fig. 2.11) (Hutchinson, 1957). This suggests that if light inhibition is an important factor in the Manawatu River growth of *S. natans* should be more luxuriant with increasing depth at a given river cross-section, where all other relevant variables are constant, because light decreases with depth.

In summary, outdoor channel and laboratory experiments indicate that solar radiation may be an important factor affecting the growth of at least one important "sewage fungus" species. However the differences between the radiation spectrum of the lights used in the laboratory experiments and natural light casts doubt upon the relevance of the latter results in nature.

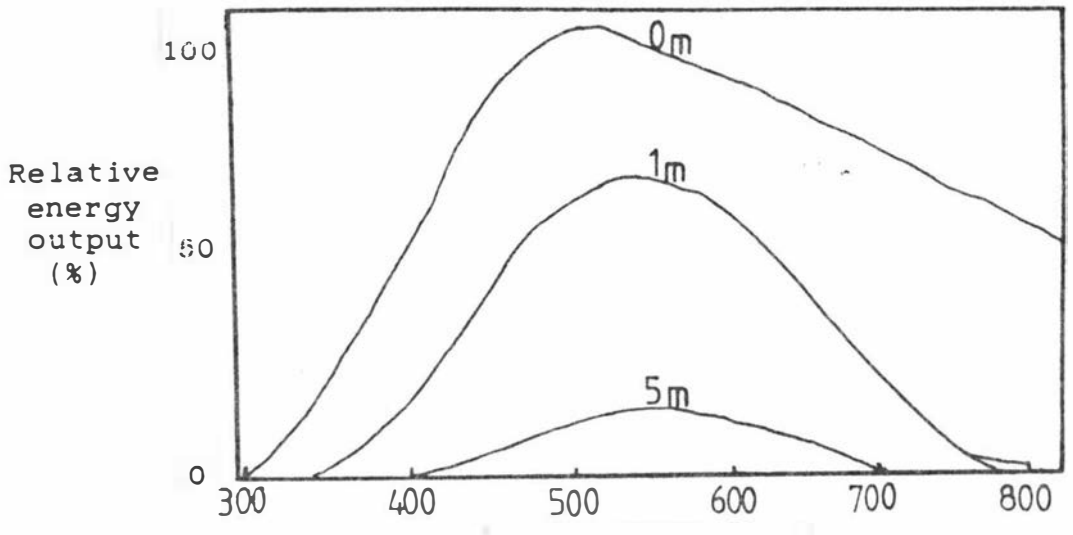


Figure 2.11:

Variation of spectral composition of light with depth. Om depth gives an approximate spectral energy distribution for sunlight. 1m and 5m give energy distribution at 1m and 5m depth in Lunzer Untersee, Austria on a clear day (from Sauberer,1939,in Hutchinson,1957).

Technical difficulties in measuring the harmful radiation wavelengths and difficulties in distinguishing between the effects of increased growth of phototrophs and radiation inhibition on sewage fungus community structure suggest that it may be impracticable to conclusively demonstrate radiation inhibition in the field. On the other hand, it should be possible to measure the integrated effects of solar radiation by observing benthic community structure at different depths at a given river cross-section. Also, where near-surface growth of sewage fungus occurs, it should be possible to quantify radiation levels at which total inhibition does not occur in nature.

2.3.4.4 Nutrients

In addition to a source of carbon, sewage fungus organisms require nitrogen, phosphorus and a range of micro-nutrients for growth. Pure culture studies with *S. natans* have shown a basal salt requirement for Na^+, K^+, Ca^{++}, Mg^{++}, SO_4^- and Cl^- (Lackey and Wattie, 1940) but adequate supplies of these for growth would be expected to occur in natural water (Hynes, 1970). Most amino acids can serve as the sole nitrogen source for *S. natans* (Mulder and van Veen, 1963), *Zoogloea* (Crabtree & McCoy, 1974), *Beggiatoa alba* (Mezzino *et al.*, 1984) and *Leptomitus lacteus* (Schade, 1940; Burnett, 1968). The latter species cannot utilize inorganic nitrogen sources (Schade, 1940; Burnett 1968) but ammonia and nitrate can serve as nitrogen sources for *Beggiatoa alba* (Mezzino *et al.*, 1984) and for *S. natans*, provided methionine is present (Stokes, 1954; Mulder & van Veen, 1963; Okrend and Dondero, 1964; Dias *et al.*, 1968). *Zoogloea* can utilize ammonia provided vitamin B_{12} and biotin are present (Crabtree & McCoy, 1974), but no data on its use of nitrate have been sighted.

Studies of *Sphaerotilus* growth in indoor channels fed dechlorinated tap water containing glucose, ammoniacal nitrogen and phosphate phosphorus showed that optimum growth occurred at a BOD_5:N:P ratio of 100:5:1 (Phillips, 1960). This is in agreement with the ratio recommended for biological wastewater treatment (Benefield & Randall, 1980). Thus if the BOD_5 :N

ratio is greater than 20:1 or the BOD_5:P ratio is greater than 100:1 then nutrient limitation of the rate of growth would be expected. However there may be sufficient nutrient to allow some growth.

No studies showing nitrogen limitation of sewage fungus growth have been published. No evidence of nitrogen limitation was observed in the U.K. sewage fungus survey and although sewage fungus was usually associated with high ammoniacal nitrogen concentrations (> 0.5 $g.m^{-3}$), it was concluded that the nitrogen supply in unpolluted waters was generally adequate to support growth (Curtis & Harrington, 1971).

Phosphorus has only rarely been found to limit growth of sewage fungus. In the U.K. survey growth occurred over a range of phosphate concentrations from less than 20 $mgP.m^{-3}$ to greater than 5 $gP.m^{-3}$ (Curtis & Harrington, 1971). Phosphorus limitation of sewage fungus growth in outdoor channels was observed however when 10 $g.m^{-3}$ of sucrose was added to water containing 50-200 $mgN.m^{-3}$ and < 1-3 $mgPO_4P.m^{-3}$ (Ormerod *et al.*, 1966). Here the addition of up to 0.5 $mgNH_4$-$N.m^{-3}$ did not change the growth rate but adding as little as 2 $mgPO_4$-$P.m^{-3}$ increased the growth and changed the composition of the community from one dominated by *S. natans* and *Zoogloea* to one dominated by the fungi *Fusarium aquaeductuum*.

It has been suggested that phosphorus limitation may also have limited *S. natans* growth in the lower Columbia River during summer (Amberg and Cormack, 1959). This river receives pulp mill effluent, which is deficient in both N and P, and during the summer, when the phosphate concentration ranged from 0 to 30 $mg.m^{-3}P$, *S. natans* was replaced by filamentous green algae. However the BOD_5 concentration was also lower during this period and high light and temperature levels may also have aided algal growth (section 2.3.4.3). Recent studies comparing the abilities of lake algal and bacterial populations to remove phosphorus (Currie and Kalff, 1984) suggest that it is unlikely that the algae could out-compete the bacteria for phosphorus if adequate organic material was present.

2.3.4.5. Current Velocity

Current aids sewage fungus growth by replenishing the oxygen and nutrient supply in the overlying water and supporting the attached flocs in the water column, thereby increasing the exposed surface area. However at high current velocities frictional forces cause sewage fungus to be dislodged and restricts growth to a thin biofilm.

The range of optimum current velocities reported in the literature for *S. natans*-dominated communities is presented in Table 2.9.

TABLE 2.9: Optimum Current Velocities for Sewage Fungus growth.

Velocity Range ($m.s^{-1}$)	Situation	Predominant Organisms	References
0.18-0.45	Channel Studies	*S. natans*	Phaup & Gannon (1967)
0.12-0.60	River observations	*S. natans*	Amberg & Cormack (1959)
0.16-0.60	River observations, 176 U.K. sites	*S. natans* and *Zoogloea*	Curtis & Harrington (1971)
0.2-0.8	Channel Studies	*S. natans*	Zimmerman, (1961)

At low velocities the attached filamentous sewage fungus organisms are not supported in the water column and loose their competitive advantage over single-celled species resulting in a community shift from filamentous to single-celled forms. This shift was observed to occur below 0.12 $m.s^{-1}$ by Amberg & Cormack (1959). Thus zoogloeal bacteria tend to be prevalent in sewage fungus at lower velocities than *S. natans* (Curtis, 1969). *Leptomitus lacteus* shows a similar response to current velocity to *S. natans* (Wuhrmann, 1964).

At each velocity within the optimal range, provided stable growth conditions occur, an equilibrium biomass concentration is eventually established at which biomass gains by growth are equalled by losses due to sloughing and predation. An abrupt increase in current velocity often results in excessive sloughing followed by the establishment of a new growth

equilibrium at the new higher velocity (Amberg & Cormack, 1959).

Current velocity is also important in determining the amount of growth in response to a given concentration of organic material. Equal growth occurred in channels fed 1.25, 2.5 and 3.75 $gBOD_5.m^{-3}$ of spent sulphite solids liquor (pulp mill effluent) if the current was varied so that all channels received the same organic loading i.e., $gBOD_5.s^{-1}$ (Amberg and Cormack, 1959). Zimmerman's observations (1961) (section 2.3.4.2) also show that current velocity can compensate for the effect of reduced organic concentration on *Sphaerotilus* growth to a certain degree. However adding 5.5-6.5 $gBOD_5.m^{-3}$ of domestic sewage at 0.2 $m.s^{-1}$ current velocity (loading = 1.2 $gBOD_5.s^{-1}$) produced *Sphaerotilus* dominated growth whereas a similar loading obtained by adding 1.6-1.8 $gBOD_5.m^{-3}$ at 0.8 $m.s^{-1}$ (loading = 1.4 $gBOD_5.s^{-1}$) did not. This indicates that the interaction between current velocity, organic concentration and growth is more complex than Amberg & Cormack's (1959) results suggest.

Current velocity may also interact with predation in affecting sewage fungus growth in some situations. Increasing the current velocity in outdoor channels fed diluted paper mill effluent from 0.05 $m.s^{-1}$ to 0.2 $m.s^{-1}$ reportedly increased the biomass of *Sphaerotilus*-dominated sewage fungus (Ormerod *et al.*, 1966). These higher velocities were thought to have dislodged grazing crustaceans which, along with chironomid larvae, were believed to have decimated the growth at 0.05 $m.s^{-1}$.

2.3.4.6 Predation

The activity of grazing invertebrates effects the standing crop of sewage fungus by removing ingested biomass and by enhancing detachment and loss as drift. Grazers associated with sewage fungus range from minute ciliate protozoans to rotifers, nematodes, annelids and macro-invertebrates such as insect larvae and molluscs (Curtis and Curds, 1971; Gray, 1982). Little is known regarding the species utilisation of these grazers but consideration of the size of the organisms

involved suggests that the macro-invertebrates are the main grazers of the ensheathed filamentous bacteria and fungi whereas the smaller organisms feed on the extracellular polysaccharide matrix, non-filamentous bacteria and associated organisms.

No studies on the effects of predation on sewage fungus biomass have been reported but studies of phototrophic periphyton communities in experimental channels have shown that loss of biomass as drift is increased by molluscs (Sumner and McIntire, 1982) and orthoclad chironomid larvae (Eichenberger and Schlatter, 1977). Channels including molluscs had up to 30% less biomass than those without (Sumner and McIntire, 1982) and Orthoclad larvae densities of more than about 3 cm^{-2} produced 5-7 fold increases in drift compared to channels with densities of 1-3 cm^{-2} (Eichenberger and Schlatter, 1977).

These effects may be less severe in natural situations where the invertebrate populations may be controlled by higher organisms but the results demonstrate the potential for predation to affect benthic community development.

2.3.4.7 Temperature

Temperature affects sewage fungus biomass directly by altering the growth and metabolic rates of the matrix organisms and indirectly by altering those of the organisms grazing on the community and the autotrophic species which may compete for bottom space on the stream bed. Seasonal temperature variations also coincide with variations in other important ecological factors such as light and grazers life cycles making the elucidation of temperature effects from field data difficult.

Pure culture studies have shown that *S. natans* grows from about 5°C to 35°C, with optimum growth occurring between 20 and 25°C (Mulder and van Veen, 1974; Phaup, 1968). Most strains of *Zoogloea ramigera* grow in pure culture between 9 and 37°C with an optimum at 28°C, although some strains may grow at lower temperatures (Unz and Dondero, 1967). The optimum temperature

reported for *Leptomitus lacteus* is much lower at 8°C (Zehender and Boek, 1964).

A number of channel studies have shown that sewage fungus growth rate increases with temperature over the range 4°C to 24°C (MOT, 1970; Amberg and Cormack, 1959; Ormerod *et al.*, 1966), with a doubling of growth rate reported for a 10°C rise in temperature (Ormerod *et al.*, 1966). Nevertheless the ultimate biomass is often the same or greater at lower temperatures if the studies were continued for long enough (MOT, 1970). Similar observations have been made in studies of slime development on trickling filters (Shephard and Hawkes, 1976).

The net effect of temperature on the metabolic and growth rates on sewage fungus apparently manifests itself in growths occupying longer stretches of river in winter than in summer (Butcher, 1932) although solar radiation inhibition may also be involved in producing this effect (Section 2.3.4.3). Addition of approximately 4 $g.m^{-3}$ BOD_5 of pulp mill effluent to the Altamaha River, Georgia, containing a background BOD_5 of approximately 1 $g.m^{-3}$ BOD_5, resulted in *Sphaerotilus* growth for about 15 miles below the outfall during winter, when the water temperature was about 10°C. During summer, when the temperature approaches 30°C, the added BOD_5 was removed very rapidly and growth receded to within 200 m of the outfall (Cawley, 1958). The effects of temperature on metabolism of sewage fungus is discussed further in Section 2.3.5.

2.3.4.8 pH

In pure culture *S. natans* and *Zoogloea ramigera* grow best near pH 7 (Lackey and Wattie, 1940; Unz and Dondero, 1967) but *S. natans* growth has been observed from pH 6 (Lackey and Wattie, 1940) to 10 (Stokes, 1954).

Fungi generally prefer a more acidic pH (Gaudy and Gaudy, 1980) and *L. lacteus* grows at pH 4.3 to 7.5 with the optimum range between 5.4 and 6.0 (Schade, 1940; Zehender and Boek, 1964).

This suggests that, if an adequate supply of organics is present and other environmental conditions favour sewage fungus development, mildly acid pH conditions will favour fungal growth whilst neutral to mildly alkaline conditions will favour bacterial growth. This pattern has been observed in the field (Ormerod *et al.*, 1966) and in laboratory continuous culture experiments (Dias *et al.*, 1968).

2.3.4.9 Oxygen

Pure culture studies have shown that *S. natans*, *Z. ramigera*, *B. alba* and *L. lacteus* are all strict aerobes (Lackey and Wattie, 1940; Schade, 1940; Unz and Dondero, 1967; Mezzino *et al.*, 1984). *S. natans* growth rate was reportedly not affected by the dissolved oxygen concentration at levels as low as 0.1 $g.m^{-3}$ when grown in suspension (Lau *et al.*, 1984). However, laboratory studies of fixed biological film systems have shown that oxygen diffusion into thick slime growth can limit the depth within the slime to which aerobic metabolism occurs (Williamson and McCarty, 1976). Whether growth is limited by oxygen (electron acceptor) or the organic substrate (electron donor) depends upon their relative concentrations in the overlying water, their rates of diffusion into the slime layer and the ratio in which they are required for growth. Theoretical considerations, made assuming two thirds of the glucose is incorporated in new cells and one third oxidised, indicate that oxygen does not limit biofilm growth using glucose unless the concentration ratio of glucose to oxygen is greater than 9 (Williamson and McCarty, 1976). This implies that dissolved oxygen will limit sewage fungus growth at water column concentrations less than approximately 1 $g.m^{-3}$ where the organic substrate concentration is equivalent to 9 g glucose.m^{-3}.

When insufficient oxygen is available for complete oxidation of substrates to water and CO_2, soluble organic intermediates are formed (Hoehn and Ray, 1973). These may diffuse into the more aerobic layers of the slime and be oxidised or diffuse out of the slime. However this is likely to result in a reduction in the growth rate in response to a given substrate concentration.

The available evidence suggests, however, that the dissolved oxygen concentration is unlikely to limit sewage fungus growth in rivers except at unusually low dissolved oxygen and high organic substrate concentrations.

2.3.5 EFFECTS OF SEWAGE FUNGUS ON OXYGEN DYNAMICS

Dissolved oxygen is of primary importance in the maintenance of a healthy stream community. It is generally accepted that the minimum acceptable concentration is 5 g $O_2.m^{-3}$ (Train, 1979) and the D classification of the Manawatu River, under the Water and Soil Conservation Act 1967, requires that this level be maintained.

The dissolved oxygen content of a stream results from the balance of oxygen input and removal processes and atmospheric reaeration (Figure 2.12).

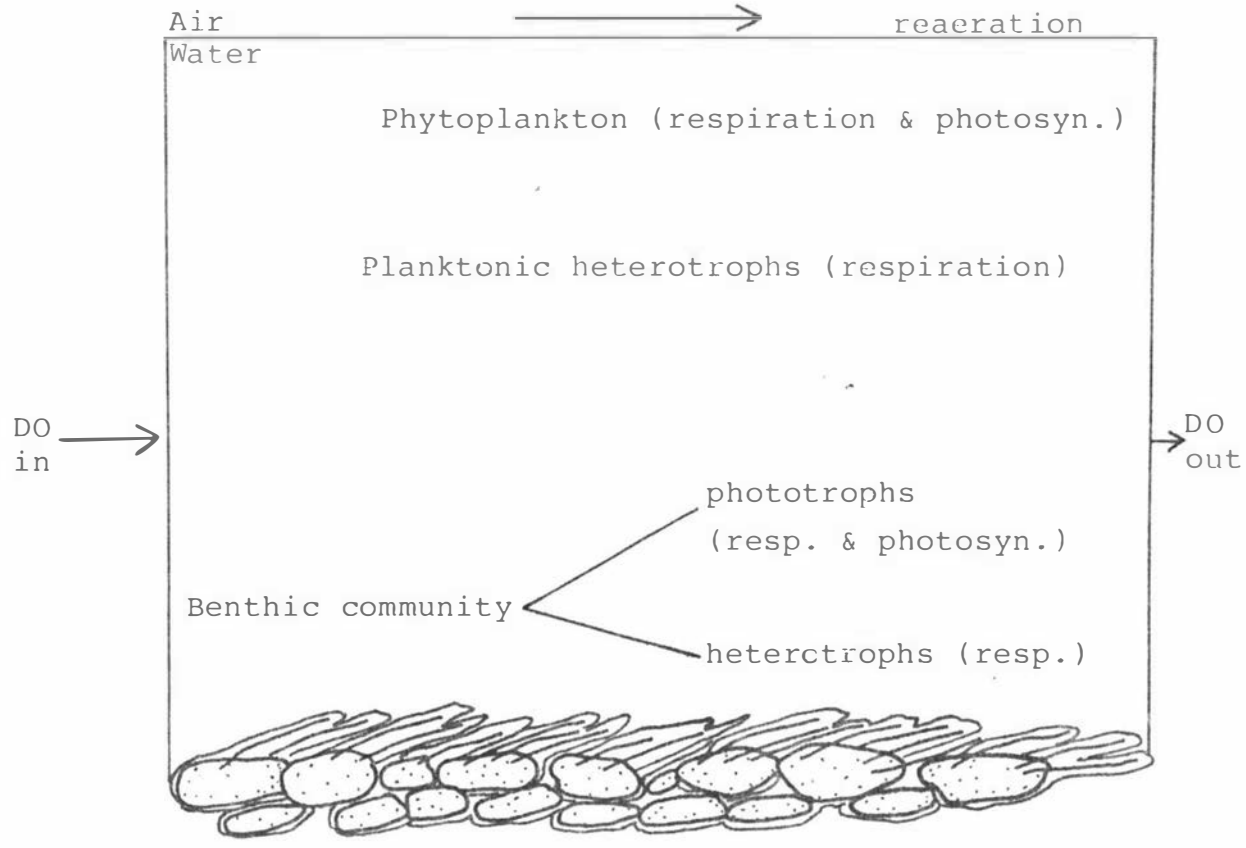
Oxygen input occurs by photosynthetic oxygen production by phototrophs suspended in the water column and attached to the bed. In simple terms this process can be summarised by the equation:

$$CO_2 + H_2O \xrightarrow{\text{Solar energy}} (CH_2O) + O_2 + H_2O \qquad (2.2)$$

Oxygen removal results from the respiration of aquatic organisms (phototrophs and heterotrophs) and the chemical oxidation of reduced substrates. Respiration is essentially the reverse of equation (2.2).

Depending on the difference between the dissolved oxygen concentration in the water and the oxygen saturation concentration at the river temperature, oxygen is lost or gained at varying rates from the atmosphere by reaeration. The rate of reaeration (k_2) can be determined using gas tracer techniques or calculated using formulae based on the average depth and velocity of the river (Wilcock, 1982).

Figure 2.12: A Representation of the Major Sources and Sinks of Oxygen in the Manawatu River below the Effluent Discharges (after Freeman,1983).



$$DO_{out} = DO_{in} + \text{Gross Photosynthetic } O_2 \text{ Production} - \text{Respiration} \pm \text{Atmospheric Reaeration}$$

DO Sources: Phytoplankton
Benthic phototrophs
Reaeration

DO Sinks: Phytoplankton
Planktonic heterotrophs
Benthic phototrophs
Benthic heterotrophs
Reaeration

Thus the processes affecting in-river oxygen can be summarised in the equation:

$$\frac{\Delta O_2}{\Delta t} = P - R + k_2(C_s - C) \qquad (2.3)$$

P = phototrophic production (g $O_2.m^{-3}.hr^{-1}$)
R = community respiration (g $O_2.m^{-3}.hr^{-1}$)
C_s = saturation oxygen concentration at river temperature (g $O_2.m^{-3}$)
C = oxygen concentration (g $O_2.m^{-3}$)
k_2 = atmospheric reaeration rate (hr^{-1})
t = time interval (hr)

Where sewage fungus occurs the respiration of the primary heterotrophs (bacteria, fungi) and associated higher organisms predominates causing rapid oxygen removal. This effect is particularly marked at night when the photosynthetic oxygen production by any phototrophs associated with the sewage fungus and upstream of the polluted area ceases whilst the phototrophic respiration continues.

In the Manawatu River very rapid night-time oxygen depletion has been observed over reaches below the waste discharges containing sewage fungus (Currie, 1977, 1978, 1980; Freeman, 1983). This problem is particularly acute during the summer when low flows and high temperatures coincide causing increased activity of benthic organisms. During the day the activity of prolific algal growths in the river offsets the sewage fungus respiration but at night very low dissolved oxygen conditions have some times developed (Fig. 2.4) causing fish kills (Currie, 1978) (Section 2.2.1.2).

The community respiration rates (based on measurements made in the dark) of some sewage fungus and phototrophic benthic communities are listed in Table 2.10. No quantitative comparisons of the biomass specific respiration rates of natural heterotrophic communities and phototrophic communities under similar conditions have been sighted in the literature. However the specific respiration rates of sewage fungus in a

TABLE 2.10 Benthic Community Respiration Rates

Community	Respiration Rate g $O_2 . m^{-2} . d^{-1}$	Comments	References
Sewage fungus	40-60	below dairy outfall Waitoa R., NZ.	Hickey (1982)
Sewage fungus and macrophytes	58	effluent channel	Curtis (1972)
Sewage fungus P/R = .01	35-53	R. Lark, England.	Butcher *et al.* (1930)
Sewage fungus P/R = .008	29	White R., Indiana	Denham (1938)
Extensive *Cladophora*	20	R. Ivel	Edwards and Owens (1962)
Extensive algal periphyton	10-20	above dairy outfall Waitoa R., NZ	Hickey (1982)
Algal periphyton	5-17	above discharges Manawatu R.	Freeman (1983)
Algae/Macrophytes	6.7-15.4	R. Ivel	Edwards and Owens (1962)
Algal periphyton	av 7.3	Truckee R.	O'Connell and Thomas (1965)
Sparse Macrophytes	2-10	Fort River, Massachusetts	Fisher and Carpenter (1976)
Algal periphyton	1-4	Laboratory Stream	McIntire and Phinney (1965)

river averaged 38.6 mg O_2.g dry weight^{-1}.h^{-1} at a mean dissolved oxygen concentration of 7 g.m^{-3} and temperatures of 21.7 to 24.0°C (MOT, 1966) whereas the specific respiration rates for macrophytes measured in the laboratory at 20°C ranged from 1 to 2.2 mgO_2.g dry weight^{-1}.h^{-1} (Owens and Maris, 1964). These data and those in Table 2.10 indicate that the respiration rates of natural heterotrophic communities are considerably higher than those of even extensive phototrophic periphyton and macrophyte communities.

Since the sewage fungus organisms require dissolved organic materials for growth and respiration the concentration of these would be expected to affect the respiration rate. However short-term variations (in terms of hours) in organic concentrations are observed to have little effect on sewage fungus respiration rates (Capblancq and Cassan, 1979b; Hickey, 1982).

Temperature is also expected to affect sewage fungus respiration markedly. Microbial reaction rates typically double for each 10°C rise (Q_{10} = 2) (Mandelstam and McQuillen, 1973). However the two available studies report a wide range of Q_{10} values. Analysis of weight specific respiration rates for sewage fungus from the River Agout incubated at 10°C, 15°C and 20°C gave Q_{10} values of 1.35 and 1.7 for sewage fungus after 2 and 4 weeks development respectively (Capblancq and Cassan, 1979b). By contrast, seasonal variations in respiration rates of sewage fungus in the River Culm suggest a Q_{10} of 2.0 (Boyle and Scott, 1984) as was observed for sewage fungus growth (Ormerod *et al.*, 1966) (see Section 2.3.4.7). However seasonal differences in biomass and dissolved organic concentrations were not allowed for in Boyle & Scott's (1984) study.

The results of laboratory biofilm studies suggest that sewage fungus growth will only be limited by dissolved oxygen at low concentration and high organic substrate concentrations (section 2.3.4.9). No account has been taken of direct effect of oxygen concentration on the rate of oxygen depletion in the river dissolved oxygen models sighted in the literature (e.g.,

Streeter and Phelps, 1925; Dobbins, 1964; Damaskos & Papadopoulos, 1983).

However *in situ* measurements of the respiration rates of dense sewage fungus communities, made using a respiratory chamber, indicate that the dissolved oxygen concentration is important in controlling these (Hickey, 1982). During an observation over a one hour period the respiration rate of growths below a dairy factory outfall declined from 50 $gO2.m^{-2}.d^{-1}$ (at DO 8.8 to 7.5 $g.m^{-3}$) to 27 $gO_2.m^{-2}.d^{-1}$ (at DO = 6 to 5 $g.m^{-3}$) and finally to 12 $gO_2.m^{-2}.d^{-1}$ (at DO = 4 $gO_2.m^{-2}.d^{-1}$). Similarly, increasing the chamber current velocity (and thus the turbulence) produced a large increase in the sewage fungus respiration rate. These changes were apparently due to the changes in oxygen concentration in the media and the rate of diffusion into the sewage fungus since addition of sufficient glucose to increase the chamber concentration by 20 g glucose.m^{-3} had no effect on the respiration rate except in one investigation of a fresh growth on an artificial substrate.

The respiration rates of phototrophic species are known to be affected by the dissolved oxygen concentrations. The rates of four aquatic macrophytes were found to decrease logarithmically with decreasing dissolved oxygen within the experimental limits of 17 to 1.2 $g.m^{-3}$ dissolved oxygen (Owens & Maris, 1964), whilst algal periphyton communities showed a curvilinear reduction in response to declining dissolved oxygen (McIntire, 1966). This suggests that the response of sewage fungus respiration to the dissolved oxygen concentration may vary with current velocity and the relative contribution and nature of phototrophic species to the total biomass.

In conclusion, sewage fungus has been shown to have a large effect on oxygen removal in streams but the relationship between the biomass and oxygen removal and the affects of physical and water quality factors and community composition on this require further research before reliable models for oxygen depletion of rivers dominated by sewage fungus can be formulated.

2.3.6 EFFECTS OF SEWAGE FUNGUS ON ORGANIC SELF-PURIFICATION

Stream organic self-purification can be defined as the removal of dissolved or particulate organic material from flowing water (Wuhrmann, 1964). Eventually self-purification reduces the concentration of dissolved organics to a level where growth of heterotrophs is no longer favoured (see fig. 2.7 and section 2.3.4.2). Thus an understanding of this process is essential for prediction of the extent of sewage fungus development in response to a given discharge.

Self-purification occurs as a result of the following processes: biological oxidation, bioadsorption, physical adsorption, sedimentation and volatilization (Kittrell & Kochtitsky, 1947; Velz, 1970; Wuhrmann, 1974; Bhargava, 1983). The rates of these processes are determined by the presence of attached and suspended micro-organisms, temperature, channel morphology, current velocity, turbulence, and the nature and concentration of the organic material present in the water column (Wuhrmann, 1974; Wright & McDonnell, 1979; Trulear & Characklis, 1982).

Where sewage fungus occurs the dense concentration of heterotrophic microorganisms with their associated extracellular polysaccharide matrix enhances the biological oxidation and bioadsorption processes and very rapid removal of dissolved organic pollutants is observed (Velz, 1970; Wuhrmann, 1974). However this removal of dissolved organics also results in the accumulation of dense growths of biomass on the river bed which then contribute considerable amounts of particulate organic material to the water column due to shearing and sloughing of the biomass. Typical aerobic heterotroph yields are in the range 0.4 to 0.6 on a caloric or organic carbon basis, depending on the carbon and nitrogen sources (Wuhrmann, 1974; Bryers & Characklis, 1982). Thus for each gram of organic carbon removed from the water approximately 0.5 g of organic carbon is produced as new biomass. Sloughing losses of this biomass produces microscopic

and macroscopic drift resulting in aesthetic degradation of the water and problems associated with the clogging of fishing nets and water intakes. The resuspended organic material tends to settle out in low current velocity zones where its decay creates further oxygen depletion.

Since the classic studies of Streeter and Phelps (1925) organic self-purification in rivers has usually been described using a first order decay constant k_1 defined as:

$$k_1 = \frac{1}{t} \ln \frac{Lo}{L} \qquad (2.4)$$

where k_1 = reach first order decay constant
t = reach travel time
Lo,L = organic material concentrations at upper and lower extremes of the reach respectively

This assumes that the rate of removal of BOD_5 is proportional to its concentration.

Values of k_1 for BOD_5 removal from streams containing sewage fungus growths (Table 2.11) lie at the top of or above the range of 0.02 to 0.21.hr^{-1} reported in a general survey of data for streams (Harremoes, 1982). Values for k_1 as high as 0.44 hr^{-1} have been recorded in the Manawatu River over short reaches with heavy sewage fungus growth (Currie & Rutherford, 1982).

TABLE 2.11: Self-purification Rates of Rivers Containing Sewage Fungus.

Organic Source	First Order BOD_5 Decay Constant ($k_1.hr^{-1}$)	Location	Reference
Sewage & Dairy	0.44	Manawatu R	Currie & Rutherford (1982
Dairy	0.37	Waitoa R, N	Hickey (1982)
Meatworks & Dairy & Sewage	0.17	Manawatu R.	Currie & Rutherford (1982
Paper Mill	0.16	R Culm, UK	Boyle & Scott (1984)
Sewage & Industrial	0.10	Stream, USA	Kittrell & Kochtitzky (1947)

However the measurement of organic self-purification using the removal of organics from the water column without reference to the river's hydraulic radius characteristics can be misleading and this complicates comparison between different rivers. For the same rate of benthic organic uptake per unit surface area, more rapid depletion of organics in the water column (k_1) will be observed in shallow rivers than deeper ones.

Wuhrmann (1964) described three alternative measures of self-purification:

(1) Amount of self-purification = mass of organic substrate removed.$time^{-1}$.

(2) Rate of self-purification = mass of organic substrate removed.unit $volume^{-1}$.$time^{-1}$. (or mass removed.unit surface area.$^{-1}$.$time^{-1}$)

(3) Specific elimination rate = mass of organic substrate removed.unit $biomass^{-1}$.$time^{-1}$.

His outdoor channel studies (Wuhrmann 1964, 1974; Wuhrmann *et al*, 1966) have clearly shown the importance of heterotrophic microorganisms and the concentration and nature of the organic material on the specific elimination rate of organic substrates (Fig. 2.13, Section 2.3.2).

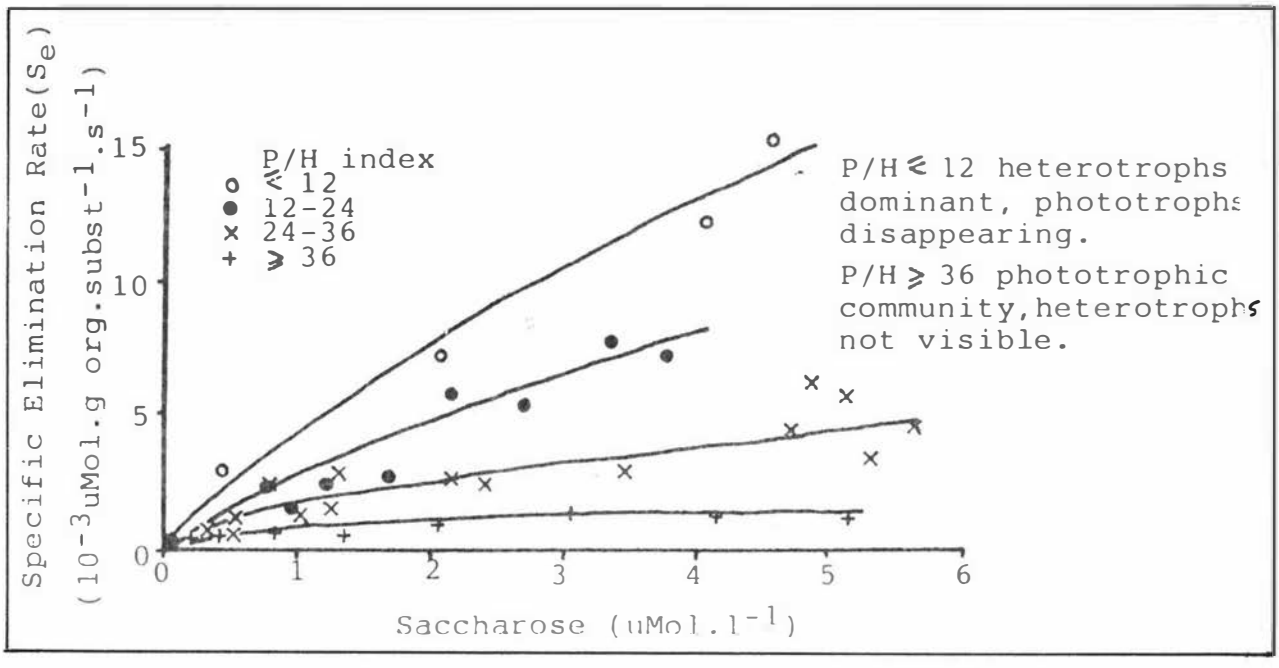


Figure 2.13: Correlation of sucrose concentration in river water with the specific elimination rate S_e of the biomass at various levels of heterotrophy(P/H index). Results of individual channel tests at u=0.15-0.2m.s^{-1}, season: June, t= approx. 10^{0}C (From Wuhrmann et al. 1966).

The data presented in Table 2.12 also indicate that the organic substrate concentration influences the organic removal rate. This probably results from the higher organic substrate concentrations producing more heterotrophic communities (Edelmann and Wuhrmann, 1978) and greater penetration of the organic substrate into the sewage fungus growths, increasing the biomass specific uptake.

Table 2.12: Sewage Fungus Sucrose Removal Rates: Outdoor Channel Studies

Initial Sucrose Concentration ($g.m^{-3}$)	Removal Rate ($g\ C.m^{-2}.d^{-1}$)	Biomass Concentration ($g\ AFDW.m^{-2}$)	Reference
5.0	2.55	17	adapted from Traaen *et al.* (1972)
5.0	1.75	5	adapted from Traaen *et al.* (1972)
3.0	0.94 & 1.15	ND	adapted from Edelmann and Wuhrmann (1978)
1.5	0.67 & 0.58	ND	adapted from Edelmann and Wuhrmann (1978)

Laboratory studies of biological film systems (Sanders, 1966; La Motta, 1976) demonstrated that the areal organic removal rate was directly proportional to the film thickness up to a critical value corresponding to the depth of penetration of the substrate into the film. Beyond this thickness no increase in uptake occurred with increasing biomass development. These critical film thickness values (as measured using a staged microscope) varied from 10.2 to 16.0 um, at influent concentrations of 2.2 to 5.2 g glucose.m^{-3}, to 65.1 um at 200 glucose.m^{-3}.

Sewage fungus communities which had developed on artificial substrates in a polluted river had much higher specific glucose uptake rates during the early stages of growth (Capblancq and Cassan, 1979b). When these were incubated for three hours in water containing 20 g glucose.m^{-3}, the seven day growths (biomass = 3 g AFDW.m^{-2}) had a specific glucose uptake rate of 1.9 mg.glucose g $AFDW^{-1}.hr^{-1}$ whereas the rate for the 14 to 28 day growths (biomass = 14 to 20 g AFDW.m^{-2} respectively) was approximately 0.5 mg glucose.g $AFDW^{-1}.d^{-1}$.

However sewage fungus communities on natural substrates often develop as fronds or long streamers with a larger exposed

surface area for contact with the over-flowing water than a similar biomass growing as a thick film or mat. This would be expected to increase the mass transfer into the slime and suggests that the biomass at which the maximum organic uptake occurs may be significantly higher for such sewage fungus growths.

This review shows that the effect of sewage fungus or organic self-purification is complex and that there is a need for further research before reliable models can be developed to predict the distance below a discharge within which the organic substrate concentrations will be suitable for sewage fungus growth.

2.4 GENERAL CONCLUSIONS OF THE LITERATURE REVIEW

2.4.1 SUMMARY OF THE MAIN POINTS OF THE REVIEW

The salient conclusions of the review can be noted as follows:

(i) Sewage fungus growth is widespread in rivers receiving organic wastes throughout the world. Its growth depends upon a large number of interacting factors affecting the development of the matrix organisms.

(ii) *S. natans* usually dominates the growth in New Zealand streams but a range of other bacteria and fungi may be present or, occasionally, predominant.

(ii) The physical characteristics of the Manawatu River make it suitable for sewage fungus development provided that an adequate amount of suitable organic material is present.

(iv) Of the three main types of discharge to the Manawatu River only one, primary-treated, domestic sewage, has been studied for its effects on sewage fungus development. The other two main discharge types are important sources of organic

pollution in New Zealand rivers. The three discharges have quite different compositions and are likely to have different effects on sewage fungus growth for equivalent organic loadings as measured by comprehensive parameters such as BOD_5, COD and TOC.

(v) Sewage fungus has significant impacts on water quality in the Manawatu River affecting dissolved oxygen, organic concentrations and aesthetic values. However detailed studies on the relationship between sewage fungus biomass and the water quality effects are lacking, especially field studies.

(vi) Algal growth in the Manawatu River upstream of the waste discharges has been studied in detail but little is known of the effects of the discharges on algal growth or the contribution to the water quality problems of algal growth below the discharges.

(vii) Because of the sequential manner in which the effluents are discharged to the Manawatu River only the effects of the PNCC discharge can be studied in isolation in the river. However the sequential addition of the waste waters to the river and self-purification processes provide a range of water quality conditions at different sites within the study reach at which benthic community development can be investigated.

(viii) Most detailed studies of sewage fungus have been conducted in indoor or outdoor experimental channels where environmental variables can be controlled.

2.4.2 TOPICS WARRANTING INVESTIGATION

It was concluded from the literature review that the following topics relating to sewage fungus and water quality warranted investigation in this study:

(i) The usefulness of limiting the in-river BOD_5 to 5 $g.m^{-3}$ as a sewage fungus control measure and the relationship between generalised organic concentration parameters and sewage fungus growth.

(ii) The affects of the three main discharges on sewage fungus growth before and after treatment to meet their new water right conditions.

(iii) The effects of sewage fungus on self-purification and oxygen dynamics and the influence of temperature, dissolved oxygen and organic substrate concentrations on these effects.

(iv) The effects of phototrophic populations on self-purification and oxygen dynamics.

(v) The effects of light on sewage fungus growth in nature.

(vi) The effects of current velocity on sewage fungus growth in the Manawatu River.

(vii) The relationship between benthic and suspended sewage fungus biomass.

CHAPTER 3.

MATERIALS AND METHODS

3.1 INTRODUCTION

The topics warranting investigation (Section 2.4.2) were studied in the Manawatu River study reach (Fig. 1.2) and in laboratory channels. As far as possible the sampling and analytical techniques used in both types of study were standardised to aid comparison of the results obtained in the two systems.

3.2 BENTHIC COMMUNITY ANALYSIS AND SAMPLING STATEGIES

3.2.1 QUALITATIVE ANALYSES AND SAMPLING STRATEGIES

3.2.1.1 Microscopic Examination

Epilithon samples collected from natural or artificial substrates (Section 3.2.2) were stored on ice in bottles containing river water. When the delay between collection and examination exceeded one day the samples were preserved by addition of sufficient formalin to give a final concentration of 10% (V/V).

The samples were examined at 600x magnification using a phase contrast microscope (Leitz Ortholux, Ernst Leitz, Wetzlar, Germany). The main heterotrophic components were identified to species level where possible using the keys and manuals of Cooke (1963), Eikelboom (1975), Eikelboom and van Buijsen (1981) and Gray (1982). The dominant meiofauna were identified to genus level using the manuals of Martin (1968) and Eikelboom and van Buijsen (1981) when these were abundant. Phototrophs were identified to genus level using the manuals of Foged (1979), Palmer (1980) and Pridmore and Hewitt (1982).

The occurrences and relative abundances of the various species or genera were recorded on a standard community analysis sheet (Appendix A).

3.2.1.2 Macroscopic Examination

The macroscopic appearance of the benthic communities was described using a phototrophic/heterotrophic dominance scale adapted from Wuhrmann (1954) and heterotrophic and phototrophic abundance scales (Appendix B). For the purposes of this thesis bacteria and fungi were referred to as heterotrophs.

3.2.2 BIOMASS SAMPLING

3.2.2.1 Benthic Biomass

The fragile nature of sewage fungus growth makes quantitative sampling of growths adhering to the natural substrate difficult. Even if an area of a bed is isolated from current effects within a sampling chamber, significant losses can result from the flocs disintegrating as the stones are uplifted. These problems become increasingly acute as the growths age.

To overcome this problem the following technique, which exploits the growths fragile nature, was developed.

i) A 0.4 m long cylindrical sampling chamber, made from galvanised iron, was used to isolate a 0.049 m^2 area of riverbed.

ii) The volume within the chamber (V_c = 10 to 17 l) was calculated from measurements of the water depth within the chamber.

iii) The stones and chamber contents were stirred vigorously for approximately one minute with a stick. This dislodged nearly all the attached growth and disturbed it homogeneously within the water in the chamber.

iv) A two litre sub-sample (Vs) of the chamber's contents was collected and allowed to settle for 15 minutes after which time the supernatant was poured off and the settled material

(SA) was collected and stored in one or two 120 ml polyethylene containers on ice for biomass measurement.

v) Immediately after removal of the 2 l sub-sample the chamber was removed allowing the remaining suspended material to be carried downstream. The stones within the sampled area were examined and, if significant amounts of biomass remained attached, these were collected and their biomass (SB) removed using rubber gloves and a stiff scrubbing brush. The biomass (SB) was measured as ash-free dry weight (AFDW) (Section 3.2.3.1).

The total biomass was then calculated using the formula:

$$\text{Total biomass } (gAFDW.m^{-2}) = \frac{((SA \times \frac{Vc}{Vs}) + SB)}{0.049} \qquad (3.1)$$

A number of measurements under a range of conditions showed that prior to stirring the bed the suspended biomass concentration was negligible compared to that after stirring. Thus the background suspended biomass was neglected in equation (3.1).

Where the biomass consisted of filamentous algae or macrophytes the sampling chamber was simply used to isolate a defined area of river bed from which all the attached biomass was removed by uplifting the rocks and dislodging their epilithon or by uprooting the macrophytes.

These techniques allowed the quantitative measurement of the benthic biomass but difficulties arose in comparing the areal benthic biomass growth rates at different sites within the study reach due to the variation in surface area provided by the bottom substrate. At sites A to D (Fig. 1.2) this predominantly consisted of 4 to 8 cm maximum length pebble and cobble but at site F pebbles of 1 to 4 cm maximum length were predominant (Section 4.2).

To overcome this problem and reduce the time required for biomass sampling, the routine quantitative measurement of benthic biomass growth rates at sites with different water quality characteristics was carried out using artificial substrates (Section 3.2.2.2). By contrast, the technique described in this section was used to measure the benthic biomass in the following situations:

i) At points within reaches over which oxygen dynamics and self-purification were studied using two station techniques (Sections 3.6.5 and 3.6.6)

ii) At sites where chamber studies were undertaken (Sections 3.6.2 and 3.6.3)

iii) At a range of biomass concentrations where qualitative macroscopic biomass assessment (Section 3.2.1.2) had been undertaken. These data were used to correlate the qualitative and quantitative biomass measurement techniques.

3.2.2.2 Artificial Substrates

The use of artificial substrates for measuring biomass development allows rapid sampling with minimal sloughing losses and simplifies the comparison of growth rates between sites with different natural substrates.

Artificial substrates used in previous studies of sewage fungus include: glass slides (Butcher, 1932; Bott and Brock, 1970), lengths of yarn (McKeown, 1963; Phaup and Gannon, 1967), wire mesh (Roberts, 1978), PVC plates (Curtis *et al.*, 1971), PVC plates on floating cylinders (Capblancq and Cassan, 1979(a)), glazed tile (Heukelekian and Crosby, 1956), unglazed tile (Wilson *et al.*, 1960; Ormerod *et al.*, 1966; Curtis *et al.*, 1971) and concrete and asbestos-cement (Heukelekian and Crosby, 1956).

Substrates with rough surfaces give better initial growth of *Sphaerotilus natans* but surface texture does not affect fungal attachment (Curtis *et al.*, 1971).

Initial trials were conducted comparing lengths of braided nylon rope, flat plates of PVC, perspex and asbestos-cement pegged into the river bed and flat concrete plates placed onto the bed. These showed that the pegged substrates, which protruded above the level of the river bed, tended to accumulate floating materials confusing the growth measurements whereas the concrete plates did not. The loss rate of the pegged substrates was also greater due to the pegs working loose. Need for attachment to the bed also made the use of these substrates more laborious than the use of the concrete plates.

For these reasons the concrete plates were chosen for use in the growth experiments. Comparison of sewage fungus development on the upper surfaces of these and flat stones after 4 to 5 days showed the two surfaces to be very similar in two trials (Section 4.3). However it was necessary to place the newly-made concrete plates in running tapwater for approximately one week prior to use to leach surface lime and chemicals from the substrates before use. Between experiments the plates were thoroughly scrubbed and soaked in dilute hypochlorite solution (Ormerod *et al*, 1966).

3.2.2.3 Suspended Biomass

(a) *Total Suspended Biomass*: This was measured by filtering 1 or 2 two litre samples of river water through pre-combusted, 70 mm diameter, GF/C filters (Whatman Corporation, Springfield Mill, Kent, England) and measuring the biomass as AFDW (Section 3.2.3.1). These filters collect all particles of greater than about 1 um diameter (Sheldon, 1972).

(b) *Particulate BOD_5*: This was determined from the total 5 day biochemical oxygen demand (BOD_5) and the GF/C filtrable BOD_5 analyses (Section 3.3.3.2) using the equation:

$$\text{Particulate } BOD_5 \ (g.m^{-3}) = BOD_5 - fBOD_5 \qquad (3.2)$$

(c) *Coarse Suspended Organic Material*: This was collected by holding a 0.15 m diameter mesh net with 1.7 mm apertures in the

current for a sufficient time to filter approximately 1 m^3 of water. The volume filtered was estimated by noting the filtration period and measuring the current velocity at the net height using a pigmy current meter (OTT Kepton meter). The biomass collected on the net was measured using both AFDW and settled volume.

3.2.3 BIOMASS MEASUREMENT

3.2.3.1 Total Biomass

The total biomass was measured as ash free dry weight (AFDW) (APHA, 1980) and settled volume. The latter measurement was made in the graduated sample containers used for collection or after 10 minutes settling in an appropriate measuring cylinder.

3.2.3.2 Photosynthetic Pigments

The photosynthetic pigment content of the biomass samples was analysed by one of the following two methods.

(i) E_{663}: During the 1982-83 season river experiments the photosynthetic pigment content was occasionally assessed using the relatively crude method of Wuhrmann (1974). The pigment was extracted from about 1 g of wet weight of biomass in 50 ml of 100% methanol, in the dark, at 20°C, for 24 hours. After centrifugation at 500 g for 20 minutes the absorbance of the supernatant was measured at wavelengths of 663 and 750 nm in an Hitachi Model 101 Spectrophotometer. The latter measurement was subtracted from the former to correct for turbidity (APHA, 1980) and the resultant pigment concentration expressed as E_{663} per unit surface area sampled.

(ii) *Chlorophyll a and Phaeophytin a*: The plate biomass samples collected during the laboratory channel experiments (Section 3.7.9), the river experiments in the 1983/84 field season and benthic biomass samples collected from the river chamber studies (Section 3.6.3) were analysed for chlorophyll a and phaeophytin a using the spectrophotometric method of APHA

(1980), based on the method of Lorenzen (1967). Pigment extraction in 90% (V/V) acetone was aided by disrupting the cells by freezing the samples before analysis and, where significant amounts of filamentous algae were present, by maceration in a tissue grinder prior to extraction. The extract absorbances were measured in a Pye Unicam Spectrophotometer (Model SP6-550).

3.2.3.3 Autotrophic and P/H Indices

The values of total biomass and photosynthetic pigment content of the samples were used to calculate the autotrophic index (AI) or P/H index values (Section 2.3.1).

3.3 RIVER WATER AND EFFLUENT SAMPLING STRATEGIES AND ANALYSES

3.3.1 SAMPLING STRATEGIES

3.3.1.1 River Water Sampling

Samples of river water were collected to investigate the effects of water quality on benthic community development (Section 3.5), oxygen dynamics and self-purification (Section 3.6). The sampling strategy was complicated by the following factors:

(i) The diurnal discharge pattern of the PNCC and BCWS effluents
(ii) The weekly discharge pattern of the BCWS effluent.
(iii) The variability of the MCDC discharge both during and between days.

The collection of 24 hour composite samples below the discharges was therefore required to accurately assess the water quality. Below the BCWS discharge variation between the weekdays and weekends would also need investigation if experiments lasted longer than the working week.

Only one automatic sampler (Manning S4040, Scotts Valley, California) was available during the 1982/83 season, while two were used during the 1983/84 season (Manning S4040 and Manning S4400-A2). These were usually employed at sites D, E or E/F.

The samplers were fitted with 25 m long hoses to enable the collection of water from the main river current. In order to prevent the hose inlet being smothered by debris collecting on its supporting stake this was located downstream of a deflector (fig. 3.1). Samples were collected at 15 minute intervals and stored on ice in bottles within the sampler base.

During the early part of the 1983/84 season a number of samples collected with the automatic samplers were found to have an unexpectedly high concentration of organic material. This was eventually found to have been caused by leaching of organics from the "Nylex" hoses (model OV002, Nylex N.Z. Ltd, Auckland) used with the samplers. The contamination was initially removed by rinsing with a dilute $H_2SO_4/K_2Cr_2O_7$ solution but contamination recurred one week later. These hoses were then replaced with non-toxic PVC hoses (Feltex Rubber Company, Catalogue Number 3412013, Auckland) and hoses supplied by Manning Environmental Corporation which were found to be consistently free from leachable organics.

Where the automatic samplers were not used, grab samples were collected in acid-cleaned two litre, plastic sample bottles and stored on ice until returned to the laboratory (within 5 hours of collection).

The sampling sites were all located at points where the effluents were known to be fully mixed with the river water (Gilliland, 1982) and coincided with the sites used in the studies of benthic biomass development.

3.3.1.2 Effluent Sampling

Effluent samples were collected for analysis for organic material and/or nutrient content as outlined below:
(i) *PNCC Effluent*: The samples were collected at the overflow from the primary treatment system, located approximately 0.5 kilometres from the river outfall. Either grab samples were

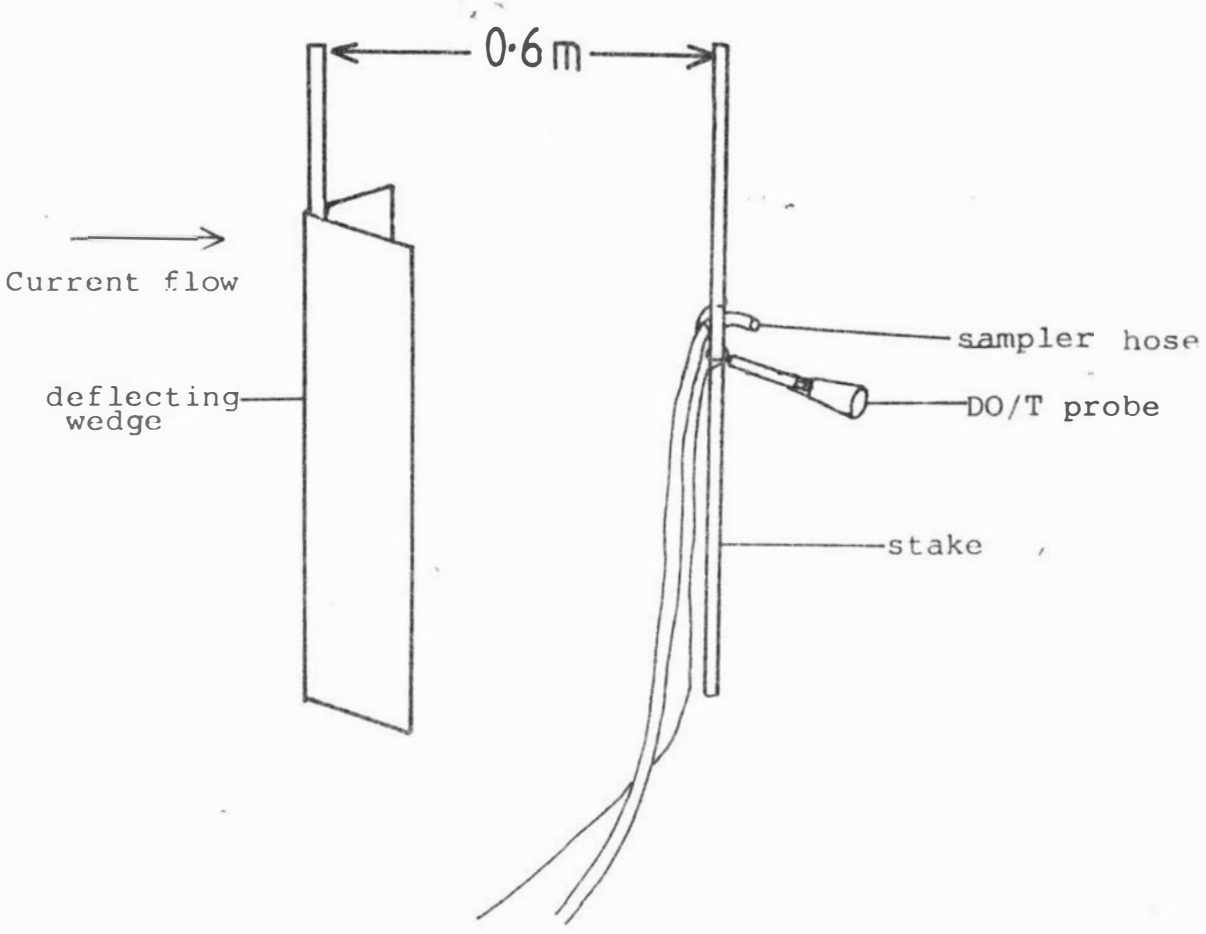


Figure 3.1: Drawing of Debris Deflecting Wedge at River Sampling Station.

collected or a Manning automatic sampler (Model S4400 or S4040-A3) was used to collect samples at fifteen minute intervals to make hourly composites. Effluent flow data (Anderson, 1983)were used to make a flow-related, twenty-four hour, composite sample from the discrete hourly samples.

(ii) *MCDC Effluent*: The effluent was sampled at the company's effluent sampling station at the river outfall by MCDC staff. Flow-related, twenty-four hour, composite samples were collected using a Masterflex pump-head (No. 7015-20 Cole-Palmer International, Chicago, USA) set to sample after every ten cubic metres of effluent was discharged as recorded on a Manning ultrasonic flow measuring device (Manning Model UTX 2100, Scotts Valley, California). The samples were stored during collection in a refrigerator at 4°C and then transferred to 120 ml polyethylene containers and frozen until analysis. The samples for lactose analysis were stored for up to four months and the samples for nitrogen and phosphorus analysis were stored for up to six months.

(iii) *BCWS Effluent*: The only effluent samples analysed were those used in the laboratory channel experiments (Section 3.7.7 and 5.5). These were mixed samples collected over intervals of thirty to fifty minutes between 1045 and 1150 AM from the outflow channel of the primary sedimentation tanks. These tanks were located approximately two kilometres from the river outfall. The samples were either analysed fresh or frozen in twenty litre plastic containers and stored up to one month prior to analysis.

3.3.2 ORGANIC MATERIAL ANALYSIS

3.3.2.1 Introduction

The water samples collected during the growth experiments in the 1982/83 season were analysed by a variety of methods in order to assess the methods' suitability as sewage fungus control parameters. In the self-purification studies the BOD_5 was used as the organic measurement parameter to enable comparison with the results of previous studies in the

Manawatu River and most other polluted water self-purification studies.

3.3.2.2 5 day Biochemical Oxygen Demand (BOD_5)

(a) Introduction:

This measure of organic material concentration is widely used by water management authorities for analysis of waters and wastewaters. The test is a bioassay measuring the oxygen consumed in the oxidation of the readily degradable organics in a water sample to carbon dioxide plus water by seed micro-organisms within an air-tight 200-300 ml bottle containing ample nutrients and held in the dark at 20°C for 5 days. Where the sample contains significant amounts of protein or reduced inorganic nitrogen, nitrification may also exert an oxygen demand. Nitrification can be suppressed by addition of inhibitors, in which case the result is termed the carbonaceous BOD_5 ($cBOD_5$) (APHA, 1980).

The BOD_5 can be further divided into the suspended and dissolved fractions by filtration and addition of a seed of known BOD_5 to the filtered sample prior to incubation.

The test has a number of inherent drawbacks due to the natural variations in the microbial seed present, the large dilutions required for analysis of strong wastewaters, causing low precision, and the 5 day delay involved.

The variation in seed can be overcome to an extent by the use of the same seed in all samples tested on a given day and by testing the seed with a glucose/glutamate standard (APHA, 1980).

(b) Analytical Procedure:

BOD_5 was usually analysed according to the method of APHA (1975) without addition of a nitrification inhibitor. Where nitrification inhibition was carried out this was achieved by addition of 10 mg 2-chloro-6-(tricholoromethyl) pyridine (TCMP) per litre of dilution water and the results reported as $cBOD_5$ (APHA, 1980). All samples were analysed in duplicate or

triplicate. The initial and final dissolved oxygen concentrations were analysed using stirring dissolved oxygen (DO) probes (model 5720, Yellow Springs International (YSI) Instrument Company, Yellow Springs, Ohio). These were calibrated in water saturated air. At the end of the incubation the probe was calibrated in air and checked against distilled water containing 0.2 ml.100 ml^{-1} of 2% $HgCl_2$ incubated with the samples. It was assumed that the oxygen content of this sample should be unchanged before and after incubation. To eliminate the risk of the loss of the bottles' water seals due to evaporation, the BOD bottles tops were covered with "parafilm" (Smith *et al.*, 1982) and to, avoid nutrient limitation, all river water samples were diluted by at least 25% with standard dilution water.

The filtrable BOD_5 ($fBOD_5$) was determined by analysing the test sample after filtration through a pre-washed, 70 mm diameter GF/C filter paper. These filter papers have a nominal pore size of 1 µm (Sheldon, 1972), whereas dissolved nutrients are normally determined as those passing through a 0.45 um pore size filter (Goldberg *et al.*, 1952). However trials showed that there was no significant difference ($P = 0.29$) between the BOD_5 of laboratory channel water, with a total BOD_5 of 3.0 ± 0.5 $g.m^{-3}$, when replicate samples were analysed after filtration by each technique. The BOD_5 of five GF/C filtered channel effluent samples was 1.72 ± 0.09 gm^{-3} ($\bar{x} \pm s$) compared with 1.55 ± 0.29 $g.m^{-3}$ ($\bar{x} \pm s$) for five, 0.45 µm pore-size membrane, filtered samples of the same water.

The use of the GF/C filters had the added advantage of allowing the collection of sufficient sample for triplicate analysis using a single filter and allowing more rapid filtration than the cellulose-acetate, 0.45 µm filters tested (Millipore Corporation, Bedford, Massachusetts, USA).

In the case of river or channel water samples 0.7 to 1.0 litres of water were filtered under a low vacuum pressure (< 200 mm Hg) provided by a hand pump. In the case of wastewater samples approximately 0.2 litres were filtered under a higher vacuum of 500 mm Hg provided by an electric pump. The filtered samples

were diluted to an appropriate concentration with dilution water containing sufficient seed to exert a BOD_5 of 0.3 to 0.8 gm^{-3}. The BOD_5 and $fBOD_5$ of the seeded samples were calculated according to APHA (1975).

A variety of seed materials were used during the study including river water, laboratory, channel effluent and settled domestic sewage. All performed satisfactorily when tested with the glucose/glutamate standard (APHA, 1980).

3.3.2.3 Chemical Oxygen Demand (COD)

In the COD test the organic material present is oxidised in a boiling mixture of potassium dichromate and sulphuric acid containing silver sulphate as catalyst. The COD is determined either by titration of the excess dichromate with ferrous ammonium sulphate (APHA, 1980) or by measuring the absorbance of the digested samples due to Chromium III at 600 nm (Jirka and Carter 1975).

The test conditions promote the oxidation of nearly all organic compounds (Moore *et al*, 1951) and has the advantage that the total analysis can be completed within about 3 hours.

In this study wastewater samples were analysed for COD and/or filtrable COD (fCOD = GF/C filtrable) and/or dissolved COD (dCOD = 0.45μm pore size membrane filtrable) using the method of Jirka and Carter (1974). Three to eight replicate, 2 ml, wastewater samples were analysed, depending on the anticipated sample variation.

3.3.2.4 Lactose

The lactose content of MCDC effluent samples was analysed using a YSI Model 27 sugar analyser (YSI, 1978).

3.3.2.5 Ultrafiltration

The concentration of low molecular weight organic compounds in the three main wastewaters discharging to the Manawatu River study reach was investigated by measuring the BOD_5 and/or COD of the samples before and following

ultrafiltration using "diaflo" membranes (Amicon Corporation, Lexington, Mass., USA). After pre-filtration through pre-washed, 0.45 um membrane filters, the samples were filtered through a 76 mm diameter YM2 membrane within a stirred cell (Amicon Model 402) at a positive filtration pressure of 310 $kN.m^{-2}$ using compressed air. These membranes have a nominal molecular weight cut-off of 1000 daltons so that the organics passing through them can be considered low molecular weight. Most of the organic compounds known to stimulate the growth of the sewage fungus matrix organisms (Section 2.3.4.2) fall into this category.

The ultrafiltration membranes were prepared by soaking in distilled water for at least 1 hour with several water changes and by filtering approximately 300 ml of MilliQ quality water (Millipore Corporation) prior to the addition of the first wastewater sample. Between samples the filters were rinsed with MilliQ quality water and the first 10 ml of new sample filtrate collected was discarded in order to prevent contamination of the new filtrate with water from the proceeding one remaining in the porous filter support (Wilander, 1972).

3.3.3 NUTRIENT ANALYSIS

3.3.3.1 Introduction

Nutrient analyses were undertaken primarily to check that heterotrophic growth rates were not being limited by low nitrogen or phosphorus concentrations in relation to the available carbon (Section 2.3.4.4). Since the predominant matrix organisms can utilise both inorganic and organic nutrients (Section 2.3.4.4), the water samples collected during various channel and river experiments were analysed for their total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP) contents. In addition some samples were also analysed for dissolved reactive phosphorus (DRP), total nitrogen (TN) and total phosphorus (TP).

3.3.3.2 Total Nitrogen and Total Phosphorus

Samples for nutrient analysis were filtered through

prewashed filters, as required, and frozen in polyethylene containers (Golterman *et al*, 1978) for up to eight months prior to analysis.

The total nitrogen and total phosphorus determinations were carried out using the method of Valderrama (1981) until the beginning of the 1983/84 field season when the similar, but more thoroughly tested, method of Ebina *et al.* (1983) was adopted. Both procedures involve simultaneous, high temperature oxidation of nitrogen and phosphorus compounds to nitrate and phosphate respectively by potassium persulphate. The oxidation occurs in two stages:

(i) Nitrogen compounds are oxidised to nitrate in the initially alkaline medium. As the digestion proceeds the resulting bisulphate ions counteract the effect of the sodium hydroxide in the oxidising reagent and lower the pH of the medium allowing:

(ii) Digestion of phosphorus containing compounds to phosphate. Subsamples were neutralised and analysed for DRP, using the method of Smith *et al.* (1982), based on that of Murphy and Riley (1962), and nitrate, using two cadmium/copper reduction columns (Smith *et al.*, 1982). Wastewater samples were diluted with distilled, deionised water to ensure that the sample COD utilised less than 10% of the added $K_2S_2O_8$ (Nydahl, 1978).

The digestion yields were checked by analysis of orthophosphoserine ($NH_2.CH(COOH).C_2O.PO_3H_2$) standards with each run. The efficiences of the nitrate reduction columns and reagents were checked regularly with nitrate and phosphate standards. The results of the digestion yield checks are presented in Table 3.1.

TABLE 3.1: Nitrogen and Phosphorus Digestion Yields

Nutrient	% Yield ($x \pm s$)	Concentration ($mg.m^{-3}$)	n
Phosphorus	100.6 ± 7.5	1000	11
Phosphorus	101 ± 2	100	2
Nitrogen	96 ± 13	452	14

3.3.3.3 Dissolved Reactive Phosphorus (DRP)

This analysis was carried out using the method of Smith *et al.* (1982). The hexanol extraction modification was used in the analysis of the samples collected at site A during the 1983/84 season.

3.4 MEASUREMENT OF PHYSICAL CHARACTERISTICS OF THE RIVER

3.4.1 REACH CHARACTERISTICS

The physical characteristics of the various reaches were important for calculation of the river respiration and photosynthetic rates by single and two-station analyses (Sections 3.6.4 and 3.6.5) and for the calculation of in-river self-purification between sites (section 3.6.6). The characteristics were measured as follows:

(a) *River Width.* This was initially measured at representative locations at sites A, D and DE with a line. Stakes were then positioned to determine changes with changing flows.

(b) *River Flow.* This was measured continuously at a permanent gauging site 2 kilometres upstream of Site A (Fig. 1.2) by the MRWB. The gauge rating curve (i.e., the relationship between the gauge height and river flow) was checked weekly.

(c) *Reach Current Velocity.* Using the flow data obtained from (b), the reach average velocity (u) was calculated using one of two methods:

(i) Linear interpolation between u value recorded in dye studies conducted at flows of 15.6 $m^3.s^{-1}$, 20.0 $m^3.s^{-1}$ (Wilcock, 1984(a)) and 26.3 $m^3.s^{-1}$ (Rutherford and Currie, 1979)

(ii) Where the flow was outside the ranges for which $\bar{u}$ could be calculated by interpolation, the value was calculated using the current velocity data for the flow closest to the flow in question and equation (2.1) (section 2.2.1.1) relating u to discharge, derived from the Manning Equation.

The reach travel time was then calculated by dividing the reach length by the reach mean velocity.

Comparison of the average velocities predicted at a flow of 15.6 $m^3.s^{-1}$ using the data obtained at a 26.3 $m^3.s^{-1}$ flow (Currie and Rutherford, 1982) in equation (2.1) with those observed at 15.6 $m^3.s^{-1}$ showed that the predicted velocities were on average 17% greater than those observed (Wilcock, 1984(c)). However the predicted values were just within the experimental uncertainty of the measured values ($\pm$ 17%) suggesting that equation (3.1) provides a reasonable method for calculating average velocities in the Manawatu River (Wilcock, 1984(c)).

(d) *Average Depth*: The average depth (h) was calculated using data from (a) - (c) using the following equation (Hynes, 1970):

$$h = \frac{Q}{w.u} \qquad (3.6)$$

where Q = average flow ($m^3.s^{-1}$)
w = average width (m)
u = average velocity ($m.s^{-1}$)

(e) *Reach Reaeration*: Reach reaeration coefficients ($k_{2(20)}$ were determined under calm wind conditions, by the methyl chloride gas tracer technique, for reaches between site C and F (Fig. 1.2) at a flow of 15.6 $m^3.s^{-1}$ (Wilcock, 1984(a)). These values were used in two station respiration and productivity analyses

for these reaches for flows from 15 to 20 $m^3.s^{-1}$. The Manawatu River k_2 values determined, by the gas tracer technique, were on average 1.4 times those calculated from the hydraulic radius data using the O'Connor and Dobbins (1958) equation (Section 2.2.1.1). Thus the $k_{2(20)}$ values (i.e., k_2 values at 20°C) for reaches upstream of site C and below this site at flows in excess of 20 $m^3.s^{-1}$ were calculated by multiplying, by 1.4, the values predicted by the values predicted by the equation of O'Connor and Dobbins (1958):

$$k_{2(20)} = 3.734\ u^{0.5}\ h^{-1.5}\ (\text{base e/day}^{-1}) \qquad (3.7)$$

where u and h were calculated as outlined in (c) and (d) above.

These $k_{2(20)}$ values were corrected for the average temperature for the interval over which oxygen changes were measured using the equation of Elmore and West (1961):

$$k_{2(T)} = k_{2(20)}\ (1.0241)^{T-20} \qquad (3.8)$$

where T = average temperature for the interval (°C).

Wind can markedly increase the reaeration rate (Elder and Gloyna, 1969). Thus the measurement of respiration and productivity by diurnal curve analysis was limited to occasions when there was little or no wind or wind from directions which did not result in surface waves on the river (i.e., northwest, east or south-east winds).

3.4.2 INTERSITE SUBSTRATE COMPARISON

The distribution of stone sizes over the study reach was investigated. Five 0.09 m^2 areas were randomly selected as sites A, D, E, EF, F and Fd (Fig. 1.2). All stones greater than 1 cm maximum length found lying on the surface within each sample area were measured for their maximum length and counted into a specific size class. This provided a basis for comparing the size distribution of stones at the sites.

3.4.3 RIVER BED ROUGHNESS

The effect of bed roughness on the available surface area per unit flat area of bed was investigated at sites C and D using a modification of the method of Puncochar (1977). Five areas of

0.021 m^2 were randomly chosen by throwing a quadrat onto the river bed at the water's edge. All the exposed surfaces within each quadrat area were covered with aluminium foil. The exposed surface area was then estimated from the weight of the foil.

3.4.4 SOLAR RADIATION

Solar radiation data were obtained from various sources during the course of this study.

(a) *Daily Integrals (Watt hours.m^2 or Megajoules (mJ .m^{-2})*: The total surface radiation was measured continuously by the Ministry of Agriculture and Fisheries (MAF) at Tiritea, approximately 3 kilometres west of the Manawatu River study area, or by NZ Meteorological Service at Ohakea, approximately 25 kilometres north of the study area.

(b) *Daily Integrals (Underwater PAR)*: The underwater photosynthetically available radiation (PAR, Einsteins.m^{-2}.d^{-1}) was measured continuously at a depth of 0.3 m in the Manawatu River at site A (Fig. 1.2) on 39 days during the summer of 1982/83 (Freeman, 1983).

(c) Instantaneous measurements of the total surface radiation (W.m^{-2}) were made using a LI-COR meter (Model LI 185B, LI-COR, Inc. Lincoln, Nebraska, USA).

(d) *Instantaneous PAR*: Surface and under water instantaneous measurements of PAR were made using a LI-COR meter (LI-185B). The quantum sensor was adapted for underwater use by sealing in clear plastic tubing. This reduced the sensitivity by 5% and was allowed for in the results. This equipment was kindly loaned by the Agronomy Dept., Massey University.

The vertical extinction coefficient (k_e) of the river water was determined from measurements of PAR at 10 cm depth intervals through the water column. These data were plotted on log-normal graph paper and the k_e value determined from the line of

best fit through the data using equation (3.9) below:

$$k_e(m^{-1}) = \frac{\ln I_1 - \ln I_2}{z_2 - z_1} \qquad (3.9)$$

where I_1 and I_2 are the PAR at depths z_1 and z_2 respectively.

These values were calculated over a range of observed turbidity and cloud cover conditions. The PAR at different depths can be calculated knowing the surface PAR or subsurface PAR at a different depth using equation (3.10) below:

$$I_{z_2} = I_{z_1} e^{-kz_2} \qquad (3.10)$$

where I_{z_1}, I_{z_2} = PAR at depths Z_1, and Z_2 respectively.

3.4.5 DISSOLVED OXYGEN AND TEMPERATURE

Dissolved oxygen and temperature were measured using Yellow Springs International (YSI) probes (Models 5720, 5739) attached to either hand held meters (YSI Model 57) or recording monitors (YSI Model 56) after initial trials showed satisfactory agreement with Winkler dissolved oxygen measurement (APHA, 1980) and temperature readings made with a previously calibrated mercury-in-glass thermometer.

Calibration of the monitors was conducted in moist air. In the two station studies, the probes at each station were also checked against each other in water in the laboratory before and after each run.

Where instrument drift occurred this was allowed for by assuming a linear increase in drift over the interval between calibration checks and adjusting the observed values accordingly.

The accumulation of floating debris around the probe was prevented by placing a deflecting wedge upstream of the stake to which the probe was attached (Fig. 3.1). Care was taken to ensure that non-shaking oxygen probes were placed far enough

behind the deflector so that the current velocity over the oxygen membrane was at least 0.4 $m.s^{-1}$.

Below the waste discharges shaking oxygen probes were used to prevent microbial overgrowth of the oxygen membranes which was found to cause artifically low values.

3.5 INVESTIGATIONS OF THE EFFECTS OF WATER QUALITY AND SOME PHYSICAL FACTORS ON BENTHIC COMMUNITIES

3.5.1 WATER QUALITY EFFECTS

The effects of water quality on sewage fungus growth were studied by examining the benthic biomass, using the microscopic and macroscopic visual techniques (Sections 3.2.1), and by measuring the rates of growth on concrete substrates (section 3.2.2.2) at sites of differing water quality. This was assessed from analysis of grab and/or composite water samples (section 3.3.1.1), and the waste discharge data provided by the MRWB and dischargers.

During the growth experiments 25 plates with a surface area of 150 cm^2 were placed at each of up to six sites at locations with the most similar current and depth characteristics available. The current velocity was measured 50 mm above the surface of the plates using an Ott pygmy current meter. Five plates were removed at each site at intervals throughout the biomass development usually commencing at day 3 or 4. The biomass attached to the upper horizontal surface was scraped off with a rubber glove and, if necessary, a scrubbing brush. On some occasions the biomass attached to the undersides of the plates was also sampled. The samples were stored on ice in the dark until analyses were commenced, within 8 hours of collection. Subsamples for chlorophyll analysis were frozen in the dark and analysed within 1 month of collection.

Usually the experiment was terminated when river flow conditions changed to the extent that unusual biomass losses

occurred and/or water quality conditions were significantly altered.

3.5.2 SOLAR RADIATION EFFECTS

(i) Suspended Plate Growth Experiment

Artificial substrates were used to investigate the effects of solar radiation on sewage fungus growth in an experiment at site D between 29/11/83 and 2/12/83. Five concrete plates were clamped horizontally in a rack (Fig. 3.2) which held them 0.3 m off the river bed and 0.45 m below the water surface so that the current velocity at the exposed upper and shaded lower surfaces was the same (0.3 $m.s^{-1}$).

After 3 days incubation the biomass on the upper and lower surfaces of the plates was sampled and analysed for total biomass and chlorophyll a content (section 3.2.3.2). It was assumed that all other important variables affecting sewage fungus development were the same on both sides of the plates so that any differences observed between the upper and lower surface growth resulted from the different solar radiation regimes.

(ii) Biomass Observations

The effect of solar radiation on sewage fungus growth was also investigated by assessing the near-surface growth at site C, using the visual assessment techniques (section 3.2.1.2), during periods of stable flow and settled weather during January 1984.

3.5.3 CURRENT VELOCITY EFFECTS ON SEWAGE FUNGUS

(i) Plate Growth Experiment

The effect of current velocity on growth rate was investigated by measuring the biomass on the upper surfaces of 4 to 5 plates incubated at site C for 5 days to 28/2/84 at similar depths at each of the following current velocity ranges (measured 50 mm above the plates):

0.17 - 0.15 $m.s^{-1}$

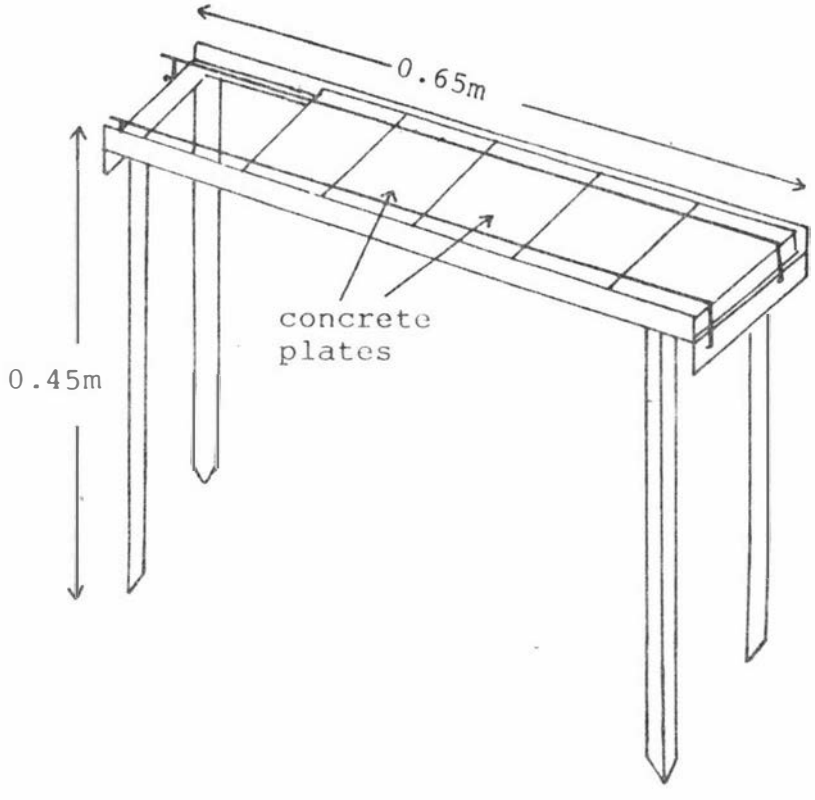


Figure 3.2: Rack used to Support Concrete Plates Horizontally in the Water Column.

0.27 - 0.19 $m.s^{-1}$
0.46 - 0.44 $m.s^{-1}$
0.57 - 0.36 $m.s^{-1}$

(ii) Bed Biomass Observations

Variations in the natural biomass were observed at different current velocities at one site after periods of stable flows. The biomass was assessed using the macroscopic visual techniques (section 3.2.1.2) and by microscopic examination (section 3.2.1.1) at current velocities, measured 50 mm off the bed, ranging from 0.12 to 0.62 $m.s^{-1}$, at site C on 11/1/84, and from 0.22 to 1.16 $m.s^{-1}$, at site Cu in the MCDC mixing zone on 1/3/84.

3.6 RIVER OXYGEN DYNAMICS AND SELF-PURIFICATION

3.6.1 INTRODUCTION

The effects of sewage fungus and other benthic communities were investigated in the Manawatu River using three techniques.
(i) Chamber studies.
(ii) Single station oxygen curve analysis at site A.
(iii) Two station oxygen curve analysis below the waste discharges.
The methods used were based on those of O'Connell and Thomas (1965).

Two types of chamber were used to study the effects of sewage fungus and other benthic communities on oxygen production and removal and the interactions between these processes and other environmental variables.

3.6.2 BOYLE AND SCOTT CHAMBER

During the 1982/83 season an opaque cylindrical chamber kindly given by Dr J D Boyle of Exeter University, UK, was used (Fig. 3.3). This was modified by substituting the original 2 volt single speed motor (Johnson No. 170) driving the impeller with a 6 volt, 6 speed, geared motor (Marx Schaltgetriebe Richard)

operated at 180 rpm. This modification was made after it was found that the original motor drained its batteries rapidly causing the impeller stirring speed to decline after about 3 hours use.

The chamber was submerged in the river and connected to either a sealed base in which stones were placed (Fig. 3.3[b]) or an open cylindrical base which had previously been dug into the riverbed (Fig. 3.3[a]), and left to develop a site-typical biomass.

The dissolved oxygen concentration and temperature were continuously monitored for a period of at least 45 minutes by a YSI probe (Model 5739) connected to a YSI Model 56 recording meter set at a chart speed of 10 $cm.hr^{-1}$. The initial total respiration rate in the chamber was calculated from the slope of the asymptote to the initial dissolved oxygen vs. time curve. The respiration rate occurring in the water column (WCR) was calculated by either:

(i) Measuring the rate of change in dissolved oxygen in the chamber with the sealed base attached but without stones included.

(ii) Measuring the rate of change of dissolved oxygen of unstirred river water in 300 ml BOD bottles incubated in the dark at the river temperature for 1 to 2 hours. Dissolved oxygen measurements were made at the beginning and end of the incubation using a stirred BOD probe (YSI, Model 5720) and a YSI Model 57 meter.

These studies showed that the water column respiration was negligible compared to the benthic respiration (section 5.2) and its effects on oxygen depletion in the chambers could be ignored when calculating the benthic respiration rate (BR). This was calculated using the formula:

$$BR = \frac{(Vc - Vs)\ (C_1 - C_2)}{t} \times \frac{10000}{SA} \quad (mgO_2.m^{-2}.hr^{-1}) \quad (3.11)$$

where Vc = chamber volume (2.66 l)

Vs = volume of stones where sealed base containing

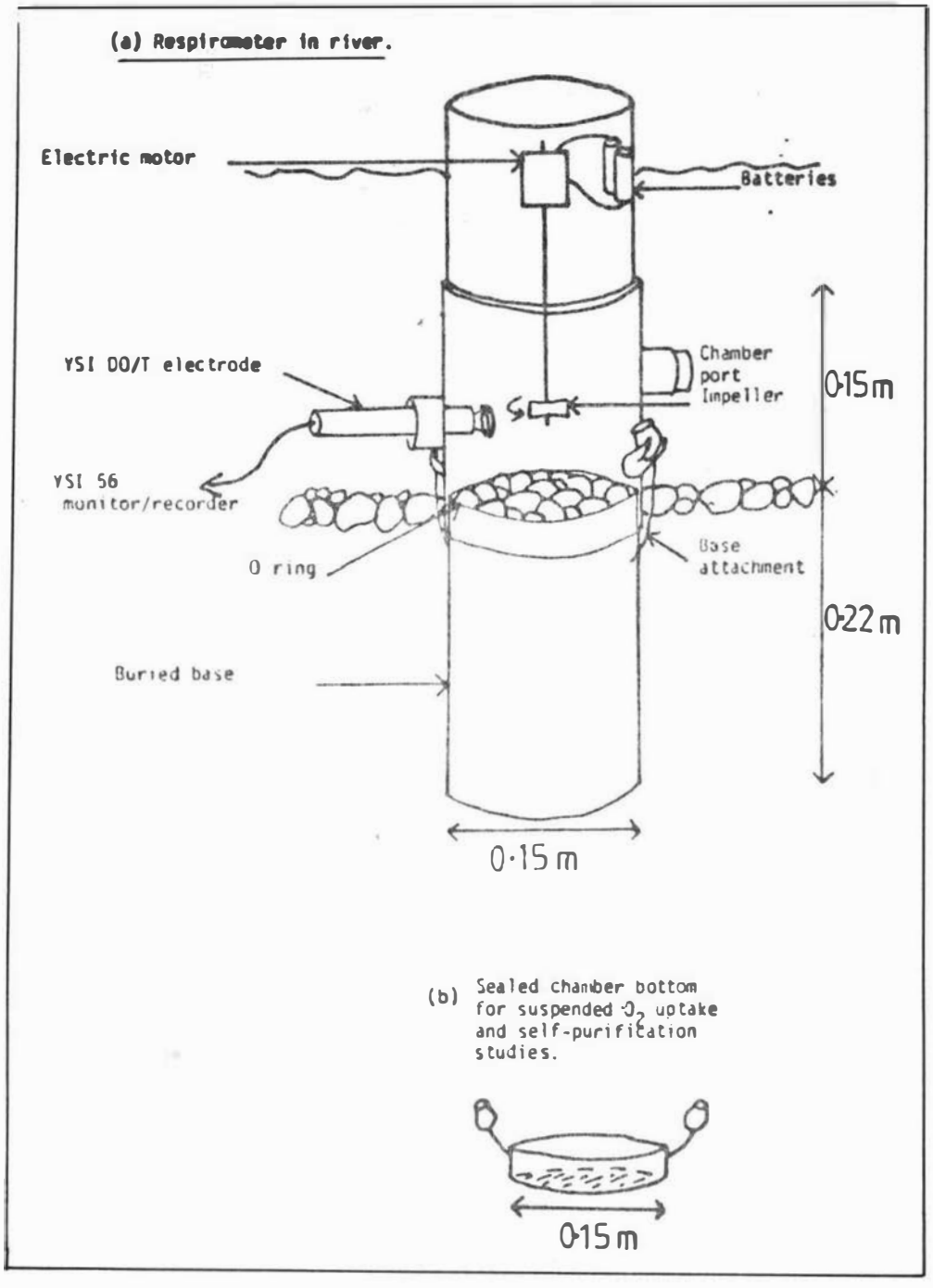


Figure 3.3: Boyle and Scott Respirometer.

stones used (Vs = 0 where buried base used) (1)

C_1, C_2 = initial and final DO for period studied ($mg.l^{-1}$)

SA = cross-sectional area of the chamber base (cm^2)

The daily BR was calculated by multiplying the hourly rate by 24.

The benthic biomass attached to stones enclosed within the sealed chamber base was removed by scraping using rubber gloves and a scrubbing brush. The benthic biomass within the buried base area was estimated by sampling adjacent areas with similar biomass using the chamber sampling technique (section 3.2.3.1). These measurements were used to calculate the biomass specific dissolved oxygen removal rates. The benthic biomass was also assessed using the techniques for qualitative analysis of community composition (Section 3.2.1).

The effect of organic material on the respiration rates was investigated by analysing, for BOD_5 and $fBOD_5$, water samples collected when the chamber was first attached to the base (section 3.3.2.2). Organic self-purification was determined by also analysing the BOD_5 and $fBOD_5$ of the chamber water at the end of the experiment.

3.6.3 FREEMAN CHAMBERS

3.6.3.1 Introduction

During the 1983/84 season two clear, perspex chambers employing unidirectional, laminar flow were used (Fig. 3.4, plate 3.1). These were constructed by Dr M C Freeman and Mr P Shaw of the Biotechnology Dept., Massey University. Comparison of this chamber with the Boyle chamber showed that, at least for the relatively low biomass levels studied, the two chambers gave similar results with respect to respiration and self-purification rates (section 5).

The flow within the Freeman chambers was provided by a submersible, centrifugal bilge pump (Bilge Captain, 12 V, TM

BP-40, Taiwan Magnetics Co. Ltd) powered by a heavy duty motor vehicle battery, adapted so that a potential of 6 or 8 volts could be used. This gave a level of turbulence above the substrate (as assessed by epilithon movements) similar to that occurring near the bed at the epilithon collection sites, where the current velocity, 50 mm above the bed, varied from 0.3 to 0.5 $m.s^{-1}$.

3.6.3.2 Experimental Procedure

The following procedures were used in all experiments:

(i) The chamber was filled with water from the main river flow and immersed to a depth of 0.20 to 0.25 m.

(ii) A 2 l water sample was also collected from the river when the chamber was filled. This was stored on ice in the dark until returned to the laboratory up to four hours later.

(iii) Stones with attached epilithon were carefully placed in the chamber. When macrophytes were studied these were carefully uprooted and held in the chamber by clips attached to the plant's roots and a weight.

(iv) With the YSI dissolved oxygen and temperature (DO/T) probe in place in the chamber's recycle arm and with the lid open, the recycling flow was initiated and all air bubbles removed.

(v) After the lid was attached and screwed down tightly, the recycle pump was turned on.

(vi) The chamber was covered with a black butyl rubber sheet and the temperature and the rate of change of oxygen in the darkened chamber recorded on a YSI 56 recorder at a chart speed of 10 $cm.hr^{-1}$.

(vii) After at least 30 minutes or the time taken for a 1 gm^{-3} drop in DO to occur, the cover was removed and the rate of change of oxygen concentration measured in the light for approximately 30 minutes.

(viii) The recycle pump was switched off and the chamber and contents removed from the river.

(ix) On most occasions the water within the chamber was sampled and stored on ice in the dark until returned to the laboratory, up to four hours later (Section 3.6.3.5)

(x) The stones were removed individually and their epilithon

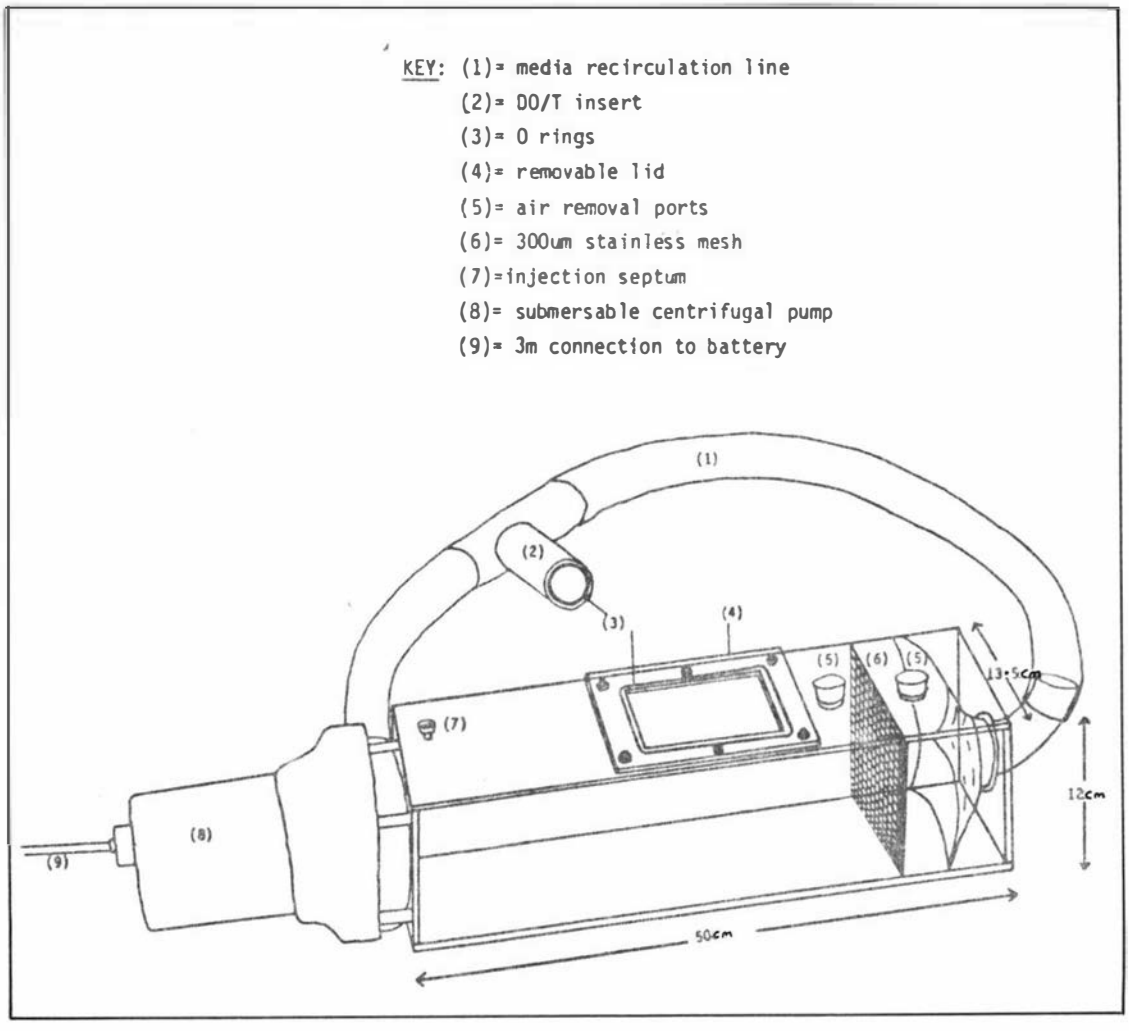


Figure 3.4: Perspex Respiratory Chamber Constructed by M.C.Freeman.

Plate 3.1: Two Freeman Chambers in Use Measuring Epilithon Photosynthesis and Respiration (covered chamber) at Site C, 12/1/84.

removed by scraping with rubber gloves and a scrubbing brush. Biomass samples were stored on ice for analysis as AFDW, settled volume and phaeophytin-corrected chlorophyll a analysis (section 3.2.3). The chamber biomass was also assessed visually (section 3.2.1.2) and a sample was examined microscopically for community composition (section 3.2.1.1). (xi) The volume of the cleaned stones was determined by displacement.

These chambers were also used to investigate the respiration and oxygen production rates of epilithon developing on artificial substrates (section 3.2.2.2) during two experiments at site 0 (Fig. 1.2) over the intervals 15/11/83 - 9/12/83 and 21/11/83 - 16/12/83 (section 5.3.2).

On each sampling occasion two plates were placed in a respirometer on top of small pebbles which raised the plates about 15 mm off the floor of the chamber allowing a flow of water to the undersides of the plates similar to that occurring in the river. The respiration and oxygen production rates were then measured using the procedures outlined above.

3.6.3.3 Respiration Calculation

The chamber respiration rate (CR) was calculated from the asymptote to the initial section of the oxygen versus time curve using equation (3.12) below:

$$CR = \frac{(Vc - Vs)(C_1 - C_2)}{t} \; mgO_2.hr^{-1} \qquad (3.12)$$

where Vc = chamber volume (5.55 l)
Vs = volume of stones (l)
C_1 = initial DO ($mg.l^{-1}$)
C_2 = final DO ($mg.l^{-1}$)
t = time (hr^{-1})

As outlined earlier (section 3.6.2) the water column respiration rate was very low and was assumed to be zero when calculating Rk. Thus Rp was calculated using equation (3.13):

$$BR = CR \times \frac{10000}{SA} (mgO_2.m^{-2}.hr^{-1}) \qquad (3.13)$$

where SA = the cross-sectional area of the chamber base (cm^2).

The daily BR was calculated by multiplying the hourly BR by 24. Dividing by the biomass values yielded the biomass specific respiration rates.

3.6.3.4 Calculation of Photosynthetic Oxygen Production Rates

The chamber net oxygen production rate (CNP) was calculated for each light condition investigated using equation (3.14):

$$CNP = \frac{(Vc - Vs)(C_2 - C_1)}{t} (mgO_2.hr^{-1}) \qquad (3.14)$$

(Symbols as defined for equation (3.12)).

Water column production has been shown to be negligible in the Manawatu River (Hickey and Rutherford, 1983) and was assumed to be zero in these analyses.

The benthic gross oxygen production rate (BGPR) was then calculated using equation (3.15):

$$BGPR = (CR_1 + CNP) \times \frac{10000}{SA} (mgO_2.m^{-2}.hr^{-1}) \qquad (3.15)$$

where CR_1 = CR immediately prior to removal of the chamber cover, calculated from equation (3.12).

These instantaneous BGPR values were converted to daily rates by multiplying by 12 hours in order to allow easier comparison of the data with those obtained from the whole river studies (sections 3.6.4 and 3.6.5) and laboratory channel studies (section 3.7.10).

The biomass specific gross oxygen production rates were obtained by dividing the daily BGPR by the biomass values and the ratio of photosynthesis to respiration (P/R values) were calculated by dividing the daily BGPR by the daily BR values.

3.6.3.5 Self-purification

The effect of the epilithon on self-purification was investigated during most experiments by analysing the water samples collected from the river as the chamber was filled and from within the chamber at the end of the experiment for BOD_5, $fBOD_5$, TDN and TDP (sections 3.3.2 and 3.3.3). The BOD_5 and $fBOD_5$ analyses were commenced within four hours of collection and the samples for nutrient analysis were filtered and frozen for later analysis (up to 8 months after collection).

These initial and final concentrations were used to calculate the area specific self-purification rates, assuming that the water column removal was negligible, using equation (3.16):

$$NUR = \frac{(Vc - Vs)(C_1 - C_2)}{t} \times \frac{10000}{SA} \quad (g.m^{-2}.hr^{-1}) \quad (3.16)$$

where NUR = nutrient uptake rate or organic material uptake rate

C_1 = initial concentration

C_2 = final concentration

Vc, Vs, t, SA as defined in equation (3.12).

Dividing these values by the biomass concentration values yielded the biomass specific removal rates.

The effects of organic material and nutrient addition on BR, BGPR and self-purification were investigated on six occasions by injecting sufficient MCDC wastewater, to raise the initial COD of one of two chambers run concurrently with similar biomasses by approximately 7.7 $g.m^{-3}$ (=5 $g.m^{-3}$ BOD_5).

The initial organic and nutrient concentrations in the chamber after waste water injection were determined by analysis of the 2 litre river water sample, collected as the chamber was filled, to which sufficient MCDC effluent was added to give the same initial concentration as in the chamber. The final concentrations were determined by analysis of the chamber water at the end of the experiment.

3.6.3.6 Light Effects

The effect of light on photosynthesis was investigated by measuring the photosynthetically available radiation (PAR) (section 3.4.4) at the depth of the biomass under clear sky conditions, under cloudy conditions or after covering the chamber with one to four layers of wind break material (black, 3 mm mesh, Donaghy Industries, Textile Division, Christchurch, NZ). These reduced the PAR at the depth of the biomass by 62%, 83%, 94% and 98% for one, two, three and four thicknesses respectively.

3.6.3.7 Temperature Effects

Two methods were used to investigate the effect of temperature on BR, BGPR and self-purification:

(i) The rates obtained from similar communities under different temperature conditions on different occasions were compared.

(ii) Rates were compared for similar biomasses incubated concurrently in a chamber at river temperature and in a chamber within a 60 litre tank containing river water chilled by 5°C using frozen glycol-filled plastic "slikka" pads.

3.6.4 SINGLE STATION DO DIEL ANALYSIS

The analysis of the continuous oxygen and temperature profiles recorded (section 3.4.5) at site A, upstream of the waste discharges (Fig. 1.2), were used to calculate the night-time community respiration rates on five occasions during 1984 and both night-time community respiration and daily gross primary production rates on three occasions during 1984. Single station analysis can be validly used in this circumstance since no point source discharges occur near site A and no longitudinal gradients in phototrophic growth are known. The finite difference method employed was based on those of Odum (1956) and O'Connell and Thomas (1965).

The rate of change of dissolved oxygen concentration at site A, where no lateral flow the catchment occurs, is described by the following equation (3.17):

$$\frac{dO_2}{dt} = k_2 (C_s - C) + GP - R \ (gO_2.m^{-3}.hr^{-1}) \qquad (3.17)$$

where $k_{2(T)}$ = reaeration coefficient (base e) at the average temperature during t (hr^{-1}) (section 3.4.1).

C_s = saturation concentration of oxygen at the ambient river temperature during t (from Golterman *et al* (1978)) (gm^{-3})

C = average DO during t ($g.m^{-3}$)

GP = gross primary productivity ($gO_2.m^{-3}.hr^{-1}$)

R = community respiration ($gO_2.m^{-3}.hr^{-1}$)

The community respiration was calculated for each night hour (when GP = 0) using equation (3.17) and the recorded hourly changes in oxygen and temperature. The daily respiration was calculated by multiplying the mean night time respiration rate by 24.

The gross primary production was calculated for each hour using equation (3.17) assuming that the average night time respiration also occurred throughout the day. This assumption is a simplification of the true situation since the respiration of aquatic plants and algae increases with temperature and dissolved oxygen (Owens and Maris, 1964; McIntire, 1966). However studies by McDonnell (1982) have shown that the estimates of total daily respiration, net primary production and gross primary production were similar when either a constant or variable uptake rate was assigned to the plant community. Estimates of the maximum net primary production were reduced however when variable respiration was assumed but this is of minor importance in this study.

The daily gross production was calculated by multiplying the mean of the hourly gross production values by the number of daylight hours. Multiplying these R and GP values by the average depth in metres yielded the areal community respiration and gross primary production rates ($gO_2.m^{-2}.d^{-1}$).

3.6.5 TWO STATION DISSOLVED OXYGEN CURVE ANALYSIS

Below the waste discharges gradients occurred in organic material concentration and benthic communities so that two station analysis was required for the measurement of the in-river respiration and oxygen production rates over given reaches. The methods employed were similar to those used in the single station analysis (section 3.6.4), except that the changes in oxygen measured were those occurring between two sites.

The factors governing the oxygen concentration over a reach of interest where no significant accrual occurs from the catchment are depicted in the following mass-balance:

$$\frac{-(C_1-C_2)}{t} = GP - R + k_{2(T)} \left(\frac{(C_s - C_1)}{2} + \frac{(C_s - C_2)}{2}\right) (gO_2 m^{-3} t^{-1}) \quad (3.18)$$

where C_1 and C_2 = DO concentrations (gm^{-3}) at the upstream and downstream sites respectively (section 3.4.5)

t = the travel time between the two sites (section 3.4.1)

$k_{2(T)}$, C_s = as defined in equation (3.17)

R = reach respiration ($gO_2.m^{-3}$)

GP = reach gross photosynthetic oxygen production ($gO_2.m^{-3}$)

This equation was solved for GP-R at hourly intervals. The mean reach respiration (r) was determined by calculating the mean of the GP-R values over the reach for the hours of darkness, when GP is zero, and converted to a daily rate (RR = $gO_2m.^{-3}.d^{-1}$) by multiplying by 24/t. The reach gross oxygen production (GP) was calculated at hourly intervals from the daytime data by substituting r for R in equation (3.18). When fewer than five hours GP data were missing from the twenty-four hour period the missing values were estimated by consideration of the trend observed over the rest of the interval. The daily gross oxygen production rate (GPR = $gO_2.m^{-3}.d^{-1}$) was calculated by multiplying the mean of the

hourly calculated reach GP values by the number of daylight hours divided by the reach travel time (t).

The RR and GPR values were used to calculate the reach P/R value and were converted to areal values ($gO_2.m^{-2}.d^{-1}$) by multiplying by the average depth.

No additions of tributaries or waste effluents occurred between sites D and F (Fig. 1.2) but between each of the adjacent sites upstream of site D discharges and/or small tributary inflows occurred. If significant accrual occurs between the upstream and downstream sites this can effect the C_2 value and needs to be accounted for in calculating the respiration and gross oxygen production values. However, the effects of the small tributary inflows between sites A and B were assumed to be negligible because of their low summer flows (section 2.2.1.1) and the similar dissolved oxygen regimes to the Manawatu River anticipated in these.

The direct river oxygen dilution effect (DE) of an effluent discharge between the 2 sites was allowed for in calculating reach respiration and gross production, where this effect was calculated to be 0.1 gm^{-3} or more. The dilution effect was calculated using equation (3.19):

$$DE = (C_1 - Cd) \frac{Qd}{Qr + Qd} \quad (gO_2.m^{-3}) \qquad (3.19)$$

where C_1, Cd = oxygen concentration at the upstream river site and in the effluent respectively ($g.m^{-3}$)

Qd, Qr = effluent flow and river flow at the upstream site respectively ($m^3.d^{-1}$)

The DE values were then added to the C_2 values to allow calculation of the reach respiration and gross production rates.

On several occasions the contribution of BOD_5 removal to the total respiration within a reach was investigated by analysing the BOD_5 content of composite samples collected, over periods of up to twenty-four hours, at each end of the reach

concurrently with the oxygen and temperature measurements.

3.6.6 TWO-STATION SELF-PURIFICATION STUDIES

The organic self-purification over various river reaches was successfully studied on nine occasions by analysis of composite samples collected at two stations at the upper and lower extremes of the reach. Several additional attempts to use this technique were thwarted by equipment malfunction and sample contamination resulting from the use of unsuitable sampling hoses (section 3.3.1.1).

Automatic samplers (section 3.3.1.1) were used to collect samples at 15 minute intervals and make hourly composites for periods of up to 24 hours. On one occasion each hourly composite sample was analysed for organic content but on all other occasions the samples at each station that corresponded to the same parcel of water, as determined from travel time calculations (section 3.4.1), were composited before analysis.

Two studies of BOD_5 removal rates were conducted over river reaches receiving waste discharges (A to B and B to C, Fig. 1.2). In these cases the discharger recorded the effluent flow rates (Qe) and collected flow-related composite samples for the time intervals corresponding to the downstream river sampling periods. The analysis of these effluent samples allowed the calculation of the average organic materials loading rates (LR). Samples were collected in the river above each outfall and at the end of each reach. In order to calculate the reach self-purification in this situation it was necessary to assume that the effluent and river water were fully mixed immediately below the outfall. This assumption causes a slight underestimation of the self-purification rates. Thus the initial BOD_5 just below the outfall (i.e., at the beginning of the reach) was calculated using the equation (3.20)

$$S_1(gm^{-3}) = Su + \frac{LR}{Q + Qe} \qquad (3.20)$$

where S_1, Su = BOD_5 just below and just above the outfall respectively ($g.m^{-3}$)

LR = effluent BOD_5 loading rate ($g.s^{-1}$)

Q, Qe = river and effluent flow rates ($m^3.s^{-1}$)

The self-purification rate over the reach was calculated using two methods:

(i) The first order exponential decay rate (k_1)

$$k_1 = \frac{1}{t} \ln \frac{S_1}{S_2} \qquad (3.21)$$

where S_1, S_2 = organic material concentrations at the beginning and end of the reach

t = reach travel time (hrs) (section 3.4.1)

(ii) The linear decay self-purification: (SP)

$$SP = \frac{S_1 - S_2}{t} \ (g.m^{-3}.hr^{-1}) \qquad (3.22)$$

and

$$SP = \frac{S_1 - S_2}{t} \times h \ (g.m^{-2}.hr^{-1}) \qquad (3.23)$$

where S_1, S_2, t are as defined in equation (3.21)

h = average depth over the reach (m)

3.7 LABORATORY CHANNEL EXPERIMENTS

3.7.1 INTRODUCTION

During the winter periods laboratory channel experiments were undertaken to investigate the effect of organic waste concentration on the following under controlled conditions:

(i) epilithon development, especially sewage fungus

(ii) benthic dissolved oxygen consumption and photosynthetic oxygen production rates

(iii) benthic organic removal rates

By conducting these experiments in a controlled, but relatively realistic, situation it was intended that quantitative data could be obtained that would be applicable to the field situation. In practice variations in waste loading rates and environmental factors in the field make such data difficult to obtain.

A continuous flow, recirculating channel system, similar to that of Curtis *et al.* (1971), was chosen for use. This type of system has advantages over continuously mixed reactor systems used by some authors in sewage fungus studies (Dias and Heukelekian 1967; Dias *et al.*, 1968; Yoshikawa and Takiguchi, 1979) in that realistic current velocity and lighting conditions can be provided and readily quantified. The recirculatory channels have much lower waste feed and water requirements than once-through continuous flow systems with similar cross-sectional areas (Traaen et al., 1972) and allow the simulation of the benthic growth conditions at a wide range of distances below an outfall by altering the channel HRT (section 7.3.5). Difficulties exist in applying the results of these continuously mixed systems to the river where the flow regime is essential plug-flow (Rutherford *et al.*, 1982). However this problem can be largely overcome (section 7.3.5).

Of the three main discharges to the Manawatu River study area, the MCDC and BCWS effluents were chosen for study in the laboratory channel systems for the following reasons:
(i) Unlike the PNCC effluent, neither of these discharges could be studied for their effects on epilithic development in isolation in the Manawatu River.
(ii) There was less known about the effects of these effluents than about those of primary treated domestic sewage (section 2.3.4.2).
(iii) Field observations in this study and previous studies indicated that these effluents were of greater importance in promoting sewage fungus growth in the Manawatu River than the PNCC effluent.
(iv) The higher concentration of organic material in these effluents than in the PNCC effluent reduced the volumes of these wastewaters required to be fed to the channels.

3.7.2 CHANNEL CONSTRUCTION

Two recirculating channel systems were constructed of 2.78 m lengths of "Stormcloud" P.V.C. guttering (AHI Plastic Extensions, Auckland) joined at each end to give a total length

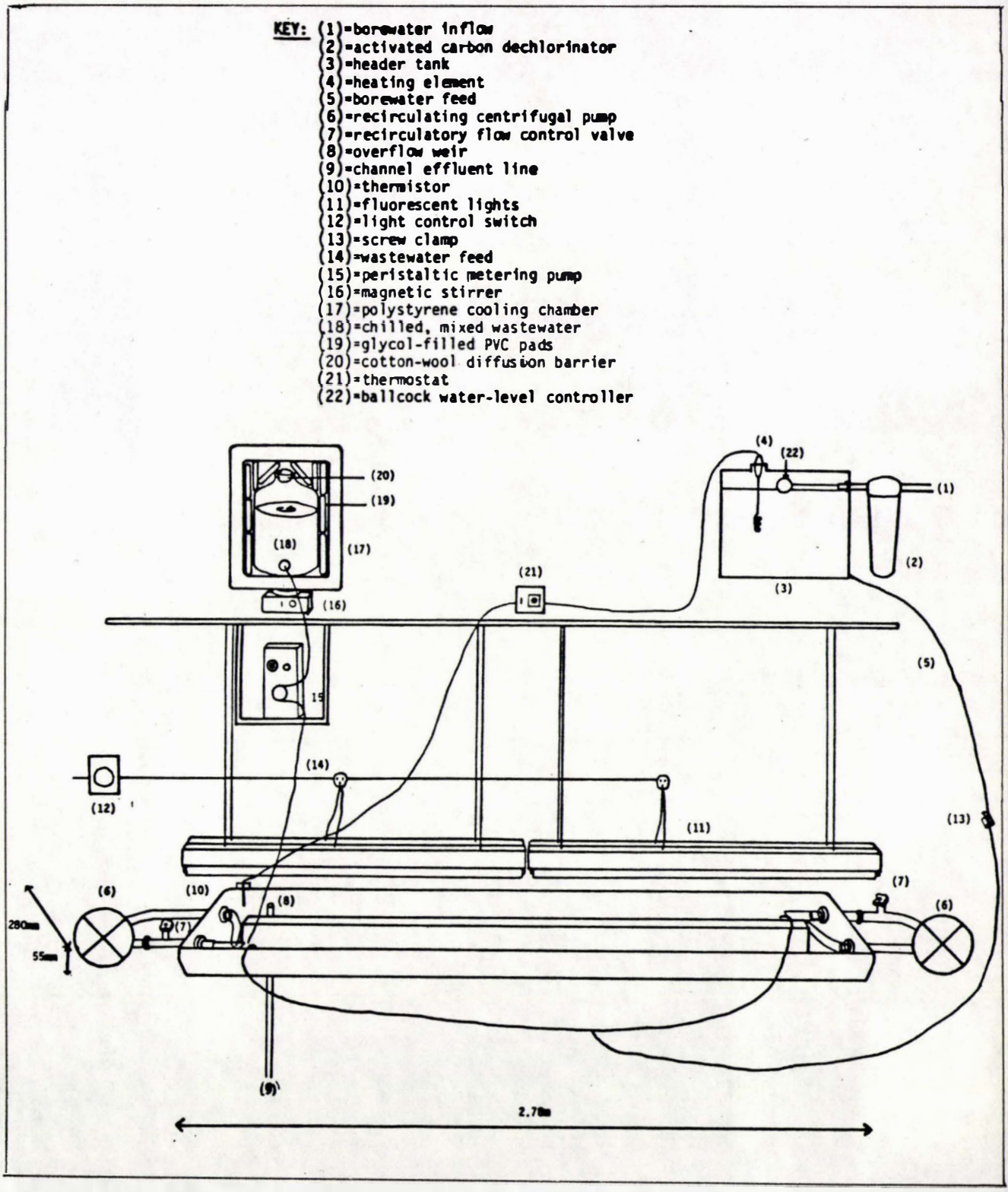


Figure 3.5: Schematic Diagram of Laboratory Recirculatory Channel.

Plate 3.2: Laboratory Channels in Operation.

of 5.56 m (Fig. 3.5, Plate 3.2). These were lined with removable concrete plates of about 100 cm^2 surface area which, along with the channel walls, provided the surface for epilithon growth. The water height above the plates was set at approximately 37 mm by the adjustable outlet. The width at the surface was 105 mm and the wetted perimeter 165 mm. At this water depth the channel held approximately 18 litres of water and the total surface area for growth, including that in the tubing, was 0.942 m^2 (Wetted Surface Area : Volume ratio = 52 $m^2.m^{-3}$).

Recirculatory water flow was provided by a centrifugal pump (H E Shacklock Ltd, Washing Machine Pump part No. 388344) at each end of the channel. These were each capable of delivering 40 litres per minute of water and provided a current velocity of 0.2 to 0.25 $m.s^{-1}$ in the channel beyond the turbulent zone within 0.4 m of the recirculatory pump inflow. At this velocity an element of water would circuit the channel approximately 2.5 times a minute. Recirculating liquid flow was controlled manually by a valve located at the delivery side of the pump. Current velocity was determined by measuring the time of travel along the channel length of suspended material.

3.7.3 LIGHTING

Light was provided by two 45 or 60 watt cool-white fluorescent tubes hung in series above the middle of each channel length. The spectral energy distribution of these lights' output (Fig. 3.6) differs somewhat from that of sunlight but energy is provided over a range of wavelengths within the photosynthetically active region and output of near-ultraviolet energy is low.

The lights were adjusted to give approximately 65 $uE.m^{-2}.s^{-1}$ of photosynthetically available radiation (PAR) measured underwater 20 mm above the concrete substrates. The photoperiod of 12 hours/day was controlled by a timer switch. Thus the daily input of PAR was 2.81 $E.m^{-2}.d^{-1}$. Using data for continuously measured PAR at 0.3 m depth in the Manawatu River at site A (Fig. 1.2) on 39 days during the summer of 1982-1983

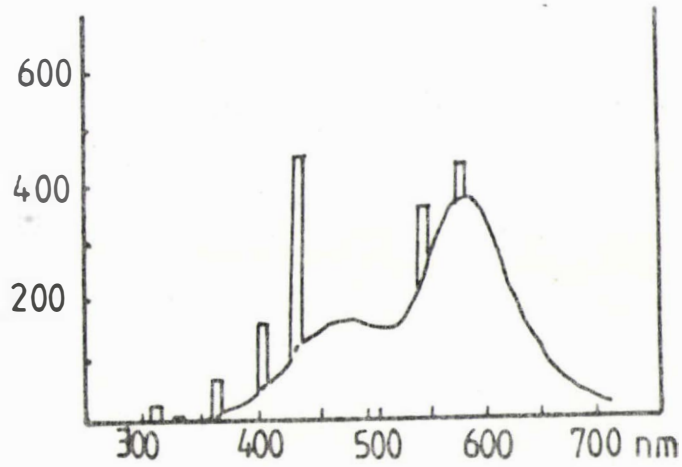


Figure 3.6: Spectral Output of a Cool White Fluorescent Lamp (based on 40 W T_{12} lamp) (after Jagger, 1967).

(Freeman, 1983) and applying a typical clear water vertical extinction coefficient of 1.2, the channel light level is equivalent to that occurring at 1.0 m depth on an overcast summer's day. This method gives an average daily PAR input at 1.0 m during this period of 6 $E.m^{-2}.d^{-1}$ (range 2.9-12.2$E.m^{-2}.d^{-1}$). Applying the extinction coefficient of 1.2 to PAR data collected at the surface on a clear day in winter (21/7/83), the channel daily PAR input is equivalent to that occurring at 0.9 m in the river under these conditions.

3.7.4 OXYGEN

Due to the high reaeration occurring in the channels ($k_{2(20)}$ = 5.6 hr^{-1}), resulting from their shallowness, high current velocity and surface disturbance caused by the recirculating pumps, significant oxygen depletion did not occur. Continuous recording of dissolved oxygen during experiments when 3.8 $g.m^{-3}$ and 6.5 $g.m^{-3}$ COD MCDC effluent were added showed the dissolved oxygen was always between 6 and 8.6 $g.m^{-3}$.

3.7.5 TEMPERATURE

The water temperature was controlled at 20 $\pm$ 1°C in the channels by a thermistor in one channel connected via a temperature controller (Phillips Witromat) to either a submersible heating element in the borewater header tank or to the channel room's refrigeration and heating units.

3.7.6 BOREWATER DECHLORINATION AND FLOW-RATE CONTROL

The borewater supply to the channels was dechlorinated by passing through one of two activated carbon filter systems:
(i) Aquapore Model AP117, 5 micron (MCDC experiments, 1983)
(ii) A filter of length 0.8 m and diameter 0.15 m packed with 20 x 40 mesh "Filtrasorb" carbon (BCWS experiments, 1984). This filter was based on one of the systems tested for dechlorination by Seegert and Brooks (1978).

Method (i) reduced the total available chlorine, measured by the diethyl-p-phenylene diamine (DPD) method (APHA, 1980), from

around 0.7 $g.m^{-3}$ to from below the limit of detection (0.01 $g.m^{-3}$) to 0.08 $g.m^{-3}$. Method (ii) reduced the bore water total available chlorine of 0.2 $g.m^{-3}$ to below the limit of detection to 0.04 $g.m^{-3}$.

After passing through the filter the borewater collected in a header tank from which make-up water was fed to the channels via butyl rubber inlet hoses controlled manually by screw clamps. The volume of water in the channel was controlled at about 18 litres by adjusting the height of the overflow weir to the appropriate level and the hydraulic residence time (HRT), or dilution rate, of 11.2 minutes was controlled by varying the borewater flow rate.

3.7.7 WASTEWATER COLLECTION, STORAGE AND CHANNEL FEED CONTROL

Wastewater used for MCDC channel experiments was collected in bulk at the MCDC outfall over 1.5 hour intervals on two occasions during periods of normal plant operation (Meredith, 1983) and frozen at -20°C until required.

Wastewater used in the BCWS experiments was collected in bulk (300-400 l) at the outflow from the primary sedimentation tanks on six occasions between 1045 and 1100 hr during normal periods of plant operation (Hinde, 1984). On return to the laboratory the samples collected over this period were composited and thoroughly mixed. The 20 to 30 litre sample containers were then refilled with the sample and frozen until required. The COD of a sample of this mixed effluent was determined (section 3.3.2.3) and used to calculate the channel loading rates. On some occasions the mixed sample was also analysed for TDN, TDP BOD_5, $fBOD_5$ and ultrafiltered before analysis for BOD_5 and COD (Sections 3.3.2.2 and 3.3.2.3).

During experiments sufficient wastewater for up to 5 days was thawed, analysed for COD (section 3.3.2.3) and stored at 4°C. One to two days channel feed were stored in 10 litre containers housed in polystyrene boxes ("chillibins") and maintained between 4 and 10°C by frozen glycol-filled PVC pads ("slikka

pads"). The feed reservoirs were stirred so that any particulate material was homogeneously suspended. The feeds were metered to the channels using peristaltic pumps (ISCO Model 1612, or Masterflex Model W21R051).

The volume remaining in the feed container was measured when each new feed was put on. These data were used to calculate the mean waste loading rate during each experiment.

At about weekly intervals the BOD_5 and $fBOD_5$ concentrations (sections 3.3.2.2) within the channels and their influents were determined by analysing 2 litre samples of the channels' effluents and influents. The samples were also filtered (0.45µm Millipore) and frozen in polyethylene containers for later analysis for TDN and TDP as required (section 3.3.3.2).

3.7.8 SEEDING

The channels were seeded with 6 to 8 small river stones (3 cm diameter), with mixed epilithic populations attached, collected in the Manawatu River at sites below the discharges. These were placed at the ends of the channels at the beginning of the experiments and were removed after 36 hours.

3.7.9 EPILITHON SAMPLING AND ANALYSIS

Epilithon growth occurred on the sides of the channels and on the 100 cm^2 concrete plates which lined them. At three to nine day intervals five plates were uplifted and the attached growths removed. The plates were then soaked in dilute hypochorite solution, scubbed clean, and rinsed in tap water before being returned to the channel. The biomass removed from each plate was analysed for settled volume, ash free dry weight (AFDW), phaeophytin a - corrected chlorophyll a. (section 3.3.2) and species composition (section 3.2.1.1).

3.7.10 EPILITHIC OXYGEN PRODUCTION AND RENEWAL STUDIES

Initial plans to use oxygen and temperature diurnal curve analytical techniques (Odum, 1956) to study epilithon oxygen

production and removal rates were abandoned after the channels were found to have very high reaeration (k_2) rates. At a current velocity of 0.27 $m.s^{-1}$, a k_2 value of 5.6. hr^{-1} was determined and this would be expected to vary considerably as biomass developed altering surface flow characteristics.

As an alternative, the perspex respiratory chambers, used in the field studies were used in these investigations (Fig. 3.4). The respirometer was immersed in channel effluent in a large vessel. Depending on the amount of attached biomass, one or two plates from the channel were placed in the chamber which was then closed and the recycle pump turned on. The epilithic oxygen uptake rate was measured in the dark for 30 minutes by a YSI dissolved oxygen and temperature probe located within the recycle loop of the chamber. The epilithic oxygen production rate was then measured over a 30 minute period under fluorescent lights which provided 45 $uE.m^{-2}.s^{-1}$ PAR at the depth of the epilithon Gross oxygen productivity for a 12 hour day/night cycle was calculated using the observed net productivity and respiration rates. In general this procedure was repeated for a second plate or pair of plates. Subsequent to these measurements the plates were removed from the chamber and analysed for the biomass parameters (section 3.2.3). This allowed the calculation of biomass-specific respiration and oxygen production rates.

3.8 COMPUTER MODELLING STUDIES

The assimilative capacity of the Manawatu river was investigated by computer modelling studies using a Prime 1 computer. Specifically, the respiration rate that could be sustained under a range of different conditions while maintaining the dissolved oxygen concentration above 5 g $O_2.m^{-3}$, as required by the rivers D classification, was studied.

The whole river oxygen dynamics studies (Section 5.2.3) showed that dissolved oxygen depletion was primarily a night-time phenomenon. Thus the studies focussed on the oxygen content of a slug of water flowing downstream from the PNCC outfall through the waste discharge zone (Fig. 1.2) from sunset to

sunrise. Since the two major fish kills occurred at the end of January, the ten hour night interval of this period was used in the model. The dye tracer studies (Wilcock, 1984(a)) showed that at a river flow of 20 $m^3.s^{-1}$, the slug would travel 11.3m below the PNCC outfall during the night arriving at a point approximately 2 kilometres upstream of site EF at sunrise. Thus at low flows a slug entering the waste discharge zone at sunset would remain within the river reach affected by heavy benthic biomass development at dawn.

The FORTRAN computer program used in these studies (see below) was developed with the assistance of Dr R.H. Archer and Dr A.C. Cleland of the Biotechnology Department, Massey University. This calculated the change in oxygen at one minute intervals throughout the ten hour period of darkness for given values of respiration, reaeration, temperature, and initial dissolved oxygen concentration.

TABLE 3.2: Fortran Program Used in Computer Oxygen Modelling Studies

```
CHARACTER*40 F1,F2
DIMENSION O(601)
Write(1,'(///)')
CALL CONA ('DATA FILE :-')
READ (1,'(A40)') F1
CALL CONA ('OUTPUT FILE :-')
READ (1,'(A40)') F2
OPEN (5,FILE=F1)
OPEN (6,FILE=F2)
Read (5,*) A1,A2,A3,A4,A5
O(1)=A1
N=0
DO 100 I=2,601
T=A2-A3*REAL(I)
C=13.55-.225*T
X2T=A5*(1.0241**(T-20))
O(I)=O(I-1)-A4+X2T*(C-O(I-1))
N=N+1
IF (N .GE. 60) THEN
```

```
      CALL COUA ('.')
      WRITE (6,200) O(I),T
      N=O
  200 FORMAT(2F10.3)
      ENDIF
  100 CONTINUE
      CLOSE(5)
      CLOSE(6)
      stop
      END
```

where

T = temperature (°C)

c = saturation dissolved oxygen concentration ($g.m^{-3}$)

x_2T = k_2 at the river temperature (min^{-1})

O = dissolved concentration ($g.m^{-3}$)

A_1 = initial dissolved oxygen at sunset ($g.m^{-3}$)

A_2 = initial temperature (°C)

A_3 = temperature decay factor (0.0033 gives 2°C decline in 10 hours)

A_4 = respiration rate ($g\ O_2.m^{-3}.min^{-1}$)

A_5 = reaeration $k_{2(20)}$ value (min^{-1})

3.9 DATA ANALYSIS

Statistical analyses such as regression, correlation, and t-tests were performed by the Minitab Statistical package (Minitab, 1982) using a Prime 1 Computer.

CHAPTER 4.
SOME PHYSICAL FACTORS AFFECTING SEWAGE FUNGUS GROWTH

4.1 INTRODUCTION

The research described in this chapter assessed the effects of various physical factors on the growth of sewage fungus. Initially, the nature of the substrate to which growths were attached was investigated and growth rates on natural and artificial substrates (concrete plates) were compared to assess the applicability of artificial substrates to river studies. Subsequently, the effects of current velocity and light intensity on sewage fungus growths in the Manawatu River were studied.

4.2 NATURAL SUBSTRATES COLONISED BY SEWAGE FUNGUS

Sewage fungus was observed to grow attached to any stable substrate present in the river. These included natural stones (plates 4.1 and 4.4), logs (plate 4.2), algal filaments (plates 4.3, 4.5 and 4.6) and the macrophyte *Potamogeton crispus*. However, qualitative observations indicated that filaments of the algae *Cladophora glomerata* provide a better attachment surface than the leaves of *P. crispus*.

The results of the intersite substrate comparison, presented in Figure 4.1, show that the size ranges of the stones making up the river bed surface were similar at sites A to E (Fig 1.2). Between Sites E and Fd (located 0.5 km downstream of site F (Fig 1.2))there was a progressive reduction in the number of stones in the larger size classes. At site Fd the river bed substrates consisted of coarse sand amongst which a small amount of fine gravel occurred. This unstable bed material was not observed to allow sewage fungus development beyond strands of 5 to 10 mm length.

The observed reduction in the size of the bed material below Site E where the bed slope declines (Section 2.2.1.1) is in agreement with the general rule that substrate size decreases with bed slope (Hynes, 1970).

Plate 4.1: Sewage Fungus Covering Bed at Site C on 31/3/82.(heterotroph abundance level=6; phototroph abundance level=C)

Plate 4.2:Sewage Fungus Growing on a Submerged Log at Site C, 31/3/82.

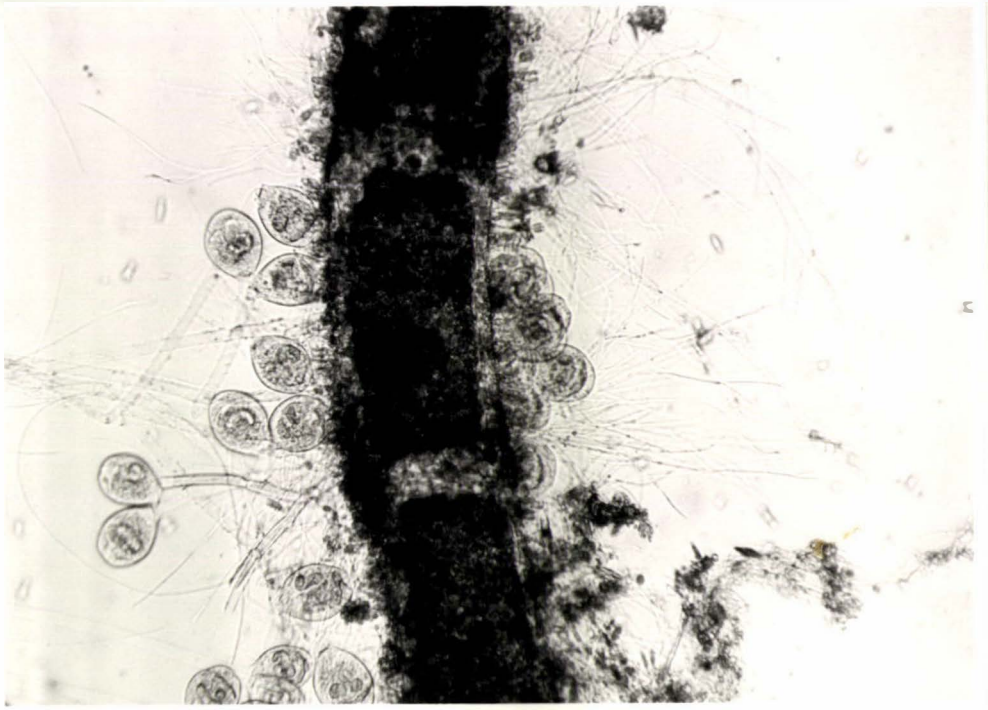


Plate 4.3: Micro-photograph of *C.glomerata* strand Showing Attached Growth of *S.natans* and the Peritrichous Protozoan *Opercularia* spp.(bright-field; mag = 150x).

Plate 4.4:Sewage Fungus Growth on the Natural Bed and on Concrete Plates After Seven Days Incubation at Site F, 21/10/82 (heterotroph abundance=5; phototroph abundance=B).

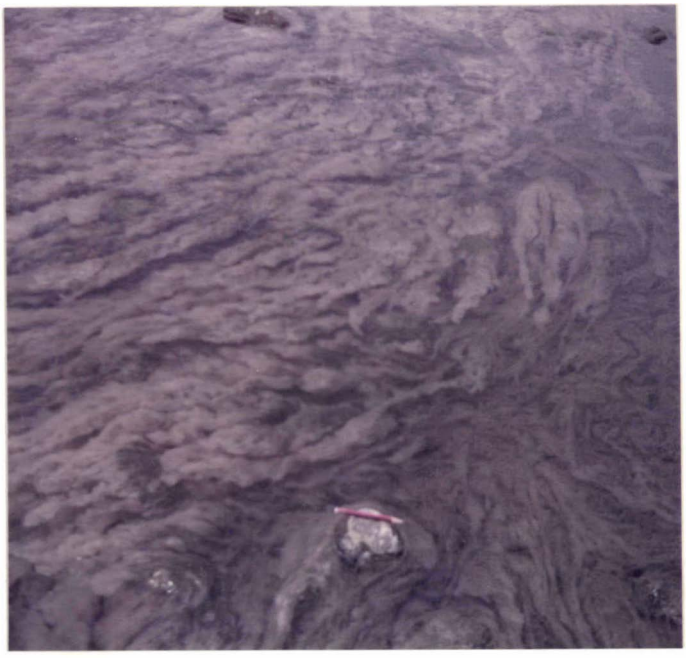


Plate 4.5: Sewage Fungus Growth at a Current Velocity of 0.25 $m.s^{-1}$ at Site C, 11/1/84 (heterotroph abundance level=6; phototroph abundance level=E).

Plate 4.6: Prolific *S.natans*-dominated Sewage Fungus Over-growing *Cladophora* Filaments Near the Water Surface at Site C, 11/1/84.

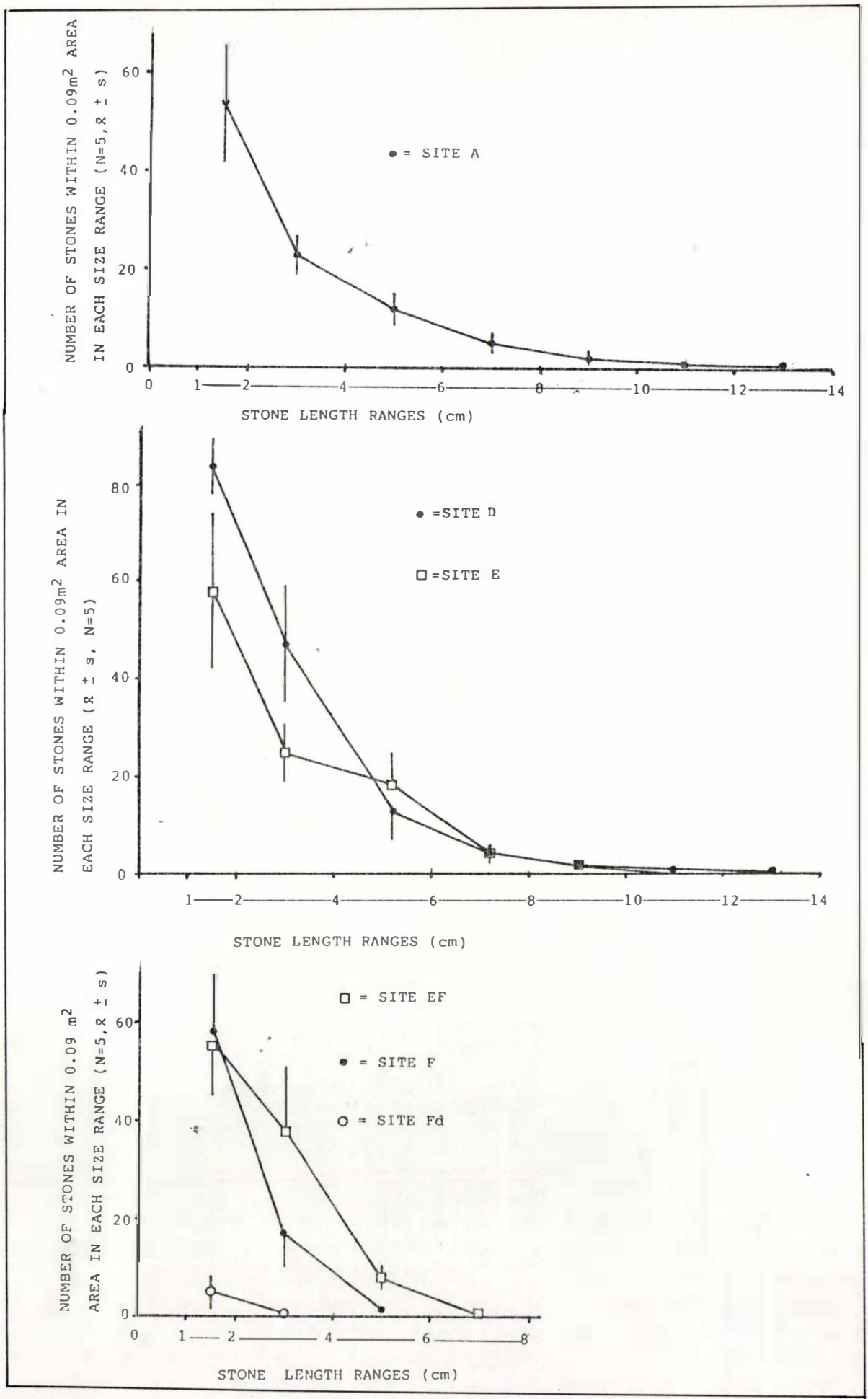


Figure 4.1: Comparison of Stone Size Ranges at Selected Sites on the Manawatu River

These changes in substrate size composition below Site E are expected to result in differences in the amount of exposed surface area for colonisation by epilithon per square metre of river bed. This complicates the comparison of areal biomass data from different sites and was part of the impetus for the use of artificial substrates in investigations of the effects of water quality and physical factors on sewage fungus growth rates (Section 3.2.2.2).

4.3 SEWAGE FUNGUS GROWTH RATES ON NATURAL AND ARTIFICIAL SUBSTRATES

The rates of biomass development on the upper surfaces of concrete plates and natural river stones were compared on two occasions. The conditions during the experiments are presented in Table 4.1 and the results are presented in Table 4.2.

In both experiments microscopic examination of the tan/pink coloured biomass showed the heterotrophs to be totally dominant. *S. natans* was very abundant and *Flavobacterium* sp common on both the natural and artificial substrates and no qualitative differences were observed in the communities from each substrate type.

There was no statistically significant difference (t-tests, $P < .05$) between the growth on the natural stones and concrete plates at Site C but at Site D the growth on the concrete plates was significantly greater (t-test, $P = 0.08$) than on the stones. However the comparison indicates that the upper surfaces of the plates provide at least as good a surface for sewage fungus growth as those of natural river stones and therefore concrete substrates, treated as outlined in Section (3.2.2.2), were frequently used to measure the growth of sewage fungus in both the Manawatu River and in laboratory channels.

The growth of sewage fungus on the bed and on concrete plates after seven days incubation at Site F (Fig. 1.2) is shown in plate 4.4.

Table 4.1 Conditions During Comparisons of Natural and Artificial Substrate

Parameters	Values	
	14-19 October 1982 Site C	6-10 December 1982 Site D
River Flow ($m^3.s^{-1}$; $\bar{x} \pm s$)	42 ± 5	40 ± 8
Current Velocity ($m.s^{-1}$)	0.4	0.35
Depth (m)	0.4	0.4
Temperature (°C)	10-12	17-19
Dissolved Oxygen ($g.m^{-3}$)	10-11.5	7.5-9
Total Surface Radiation:*		
Daily Input ($MJ.m^{-2}.d^{-1}$)($\bar{x} \pm s$)	18.5 ± 4.3	25.7 ± 6.5
Daytime Average ($W.m^{-2}$)	417	485
BOD_5 ($g.m^{-3}$)	11.8**	9.0***
$fBOD_5$ ($g.m^{-3}$)	ND	6.0***

* at Ohakea 27 km NW of Study Area
** grab sample 0930 hr 15 October 1982
*** 24 hour composite sample 7-8 December 1982

TABLE 4.2 Growth on Natural and Artificial Substrates in Manawatu River (Figure 1.2)

Site	Sampling Period (1982)	Number of Samples (n)	Average Biomass ($gAFDW.m^{-2}$) ($x \pm S$)	
			Stones	Concrete Plates
C	14-19 October	4	5.8 ± 2.2	5.85 ± 1.9
D	6-10 December	5	5.6 ± 0.9	6.5 ± 0.2

4.4 CURRENT VELOCITY EFFECTS ON SEWAGE FUNGUS

4.4.1 BENTHIC BIOMASS OBSERVATIONS

The effect of current velocity on sewage fungus biomass was investigated by visual assessment of the biomass in the Manawatu River after periods of relatively stable conditions at Sites C and Cu (Fig. 1.2) on 11 January 1984 and 1 March 1984 respectively. Since the conditions prior to the two investigations were very similar (Table 4.3), the results are presented together in Figure 4.2. On each occasion the growth consisted of *S. natans* dominated sewage fungus growing over strands of the filamentous green algae *Cladophora glomerata*. The strong algal strands provided an excellent support for the *S. natans* filaments enabling 30 to 50 cm long streamers to develop over the optimal current velocity range of 0.2 to 0.45 $m.s^{-1}$ (Plates 4.5 and 4.6), as measured 50 mm above the bed. At velocities of less than 0.15 $m.s^{-1}$ sewage fungus did not develop. Above 0.45 $m.s^{-1}$ the sewage fungus was progressively reduced to shorter growths and the community became algal dominated. At 0.63 $m.s^{-1}$ sewage fungus growth was limited to films and short plumose fronds of length 1-3 cm and at 1.08 and 1.16 $m.s^{-1}$ to growths of length 1-2 cm (Plate 4.7). The longest fronds occurred on the downstream sides of the stones where zones of greatly reduced current velocity (dead zones) occur (Hynes, 1970).

4.4.2 SEWAGE FUNGUS GROWTH RATE EXPERIMENT

The effect of current velocity on sewage fungus growth was studied using artificial substrates at Site C in the Manawatu River (Fig. 1.2) between 23 and 28 February 1984. The conditions at this site during the interval are summarised in Table 4.4.

The results, presented in Figure 4.3, relate to the growth on the upper surfaces of five concrete plates placed at each of three different current velocities at locations within 50 metres of each other. Unfortunately, localised changes in the

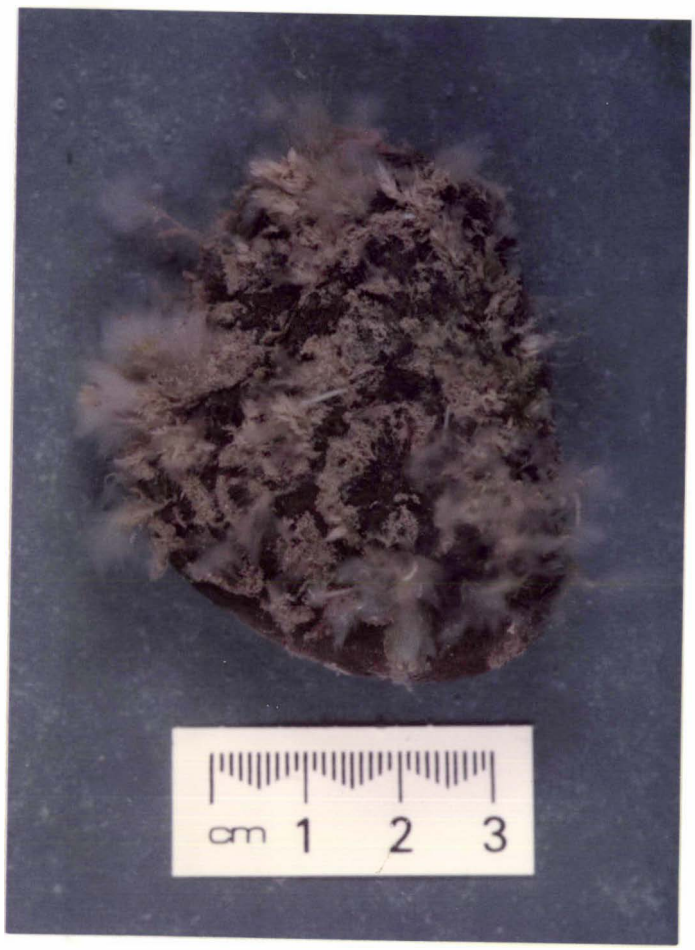


Plate 4.7: Short Sewage Fungus Fronds Attached to a Stone Collected at a Current Velocity of 1.16m.s^{-1} at Site Cu on 1/3/84.

TABLE 4.3 Conditions Prior to Observation of the Effects of Current Velocity on Biomass

Parameters	Values	
	Observation (1) on 11/1/84	Observation (2) on 1/3/84
Site	C	Cu
Riverflow ($m^3.s^{-1}$)	15.1	16.8
Interval since last spate (days)	14	16
Interval since river flow exceeded 20 $m^3.s^{-1}$ (days)	8	8
Temperature (°C)	16-20	19-22
BOD_5 during previous 8* days ($g.m^{-3}$)	5.5	5.6
Total Surface Radiation during previous 8 days:**		
Daily Input ($MJ.m^{-2}.d^{-1}$) ($x \pm S$)	24.7 ± 4.9	23.1 ± 2.6
Daytime Average ($W.m^{-2}$)	458	486

* average BOD_5 at site calculated from waste discharge and riverflow data (Gilliland, 1984) using the first order BOD_5 decay equation (Section 2.3.5):

$$C = Co \cdot e^{-kt} \qquad (4.1)$$

where C = BOD_5 in river after time t
Co = initial BOD_5 at outfall assuming immediate mixing
k_1 = BOD_5 decay rate
assuming
i) k_1 (PNCC-MCDC) = $0.07.hr^{-1}$ (average for 29/2/84-1/3/84, (Section 5.3.2.1)
ii) k_1 (MCDC-Cu) = $0.157.hr^{-1}$ (average for 29/2/84-1/3/84, (Section 5.3.2.1)
iii) k_1 (MCDC-C) = $0.157.hr^{-1}$ (average for 29/2/84-1/3/84, (Section 5.3.2.1)
iv) BOD_5 above PNCC = 0.7 $g.m^{-3}$ and 1.4 $g.m^{-3}$ prior to 11/1/84 and 1/3/84 respectively (measured 7/1/84 and 29/2/84 respectively).

** measured at Tiritea 8 km east of study area.

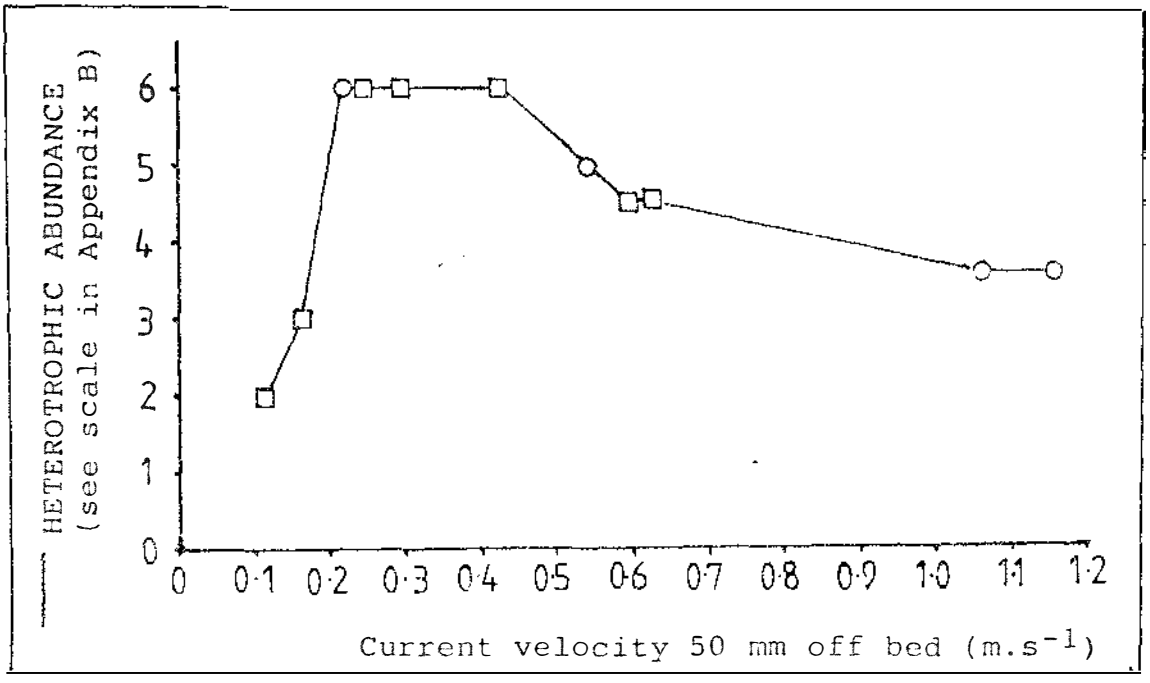


FIG.4.2: Current velocity vs heterotrophic biomass at two sites in the Manawatu River (□ = site C, 11/1/84, O = site Cu).

TABLE 4.4 Conditions at Site C, 23-28/2/84

Parameters	Values
River Flow (m^{-3}) ($\bar{x} \pm S$)	18 ± 1
Calculated average BOD_5 ($g.m^{-3}$)*	3.9
Current velocity ranges ($m.s^{-1}$)	0.17-0.15; 0.27-0.19; 0.46-0.44; 0.57-0.36
Depth at plates (m)	0.25-0.30
Water temperature 29/2-1/3/84 (°C)	20-22
DO 29/-1/3/84 ($g.m^{-3}$)	3.5-7.9
Total Surface Radiation**	
Input ($MJ.m^{-2}.d^{-1}$) ($\bar{x} \pm s$)	23.7 ± 6
Daytime average ($W.m^{-2}$)	499

* calculated from waste discharge and river flow data using the first order decay equation (4.1) (Section 4.3.1) and assuming:

i) k_1 (PNCC-MCDC) = 0.07 hr^{-1} (av for 29/2/84-31/3/84, Section 5.3.2.1)

ii) k_1 (MCDC-D) = 0.157.hr^{-1} (av for 29/2/84-31/3/84, Section 5.3.2.1)

iii) BOD_5 above PNCC = 1.4 $g.m^{-3}$ (av for 29/2/84-31/3/84, Section 5.3.2.1)

** Measured at Tiritea 3 km west of study area.

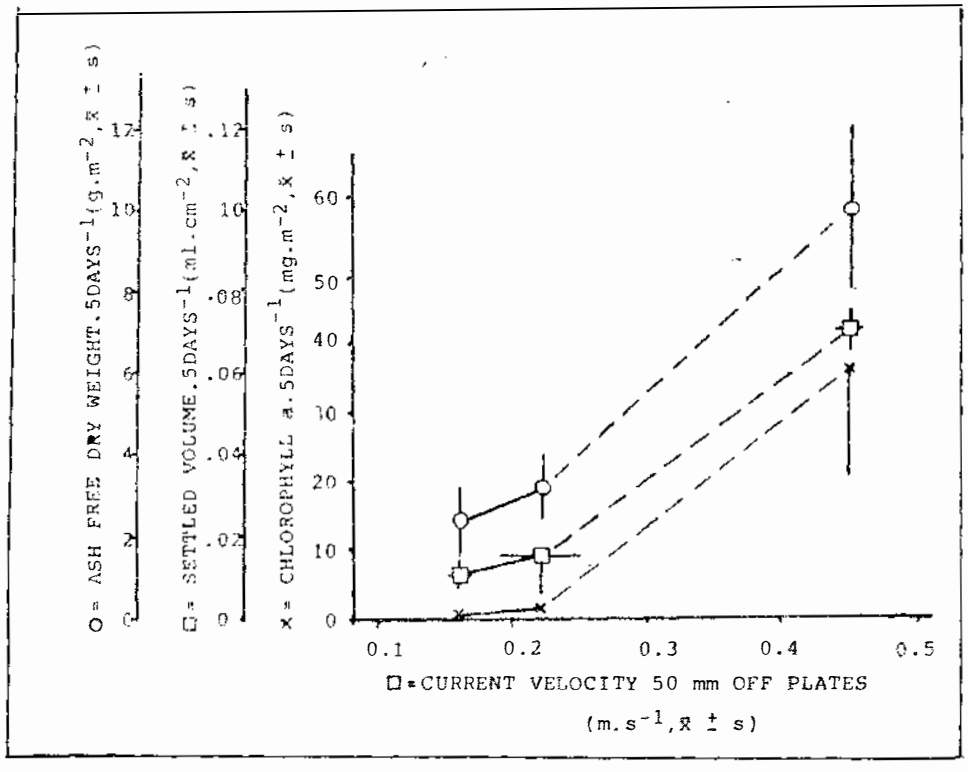


Figure 4.3: Biomass Development on Upper Surfaces of Concrete Plates, Site C, 23 to 28/2/84.

flow pattern at the location where the velocity was initially 0.57 $m.s^{-1}$ caused the velocity to decline to 0.36 $m.s^{-1}$ during the 5 day growth interval. Due to this large range of velocities the biomass data from these plates are not presented in Figure 4.3.

A large increase in all biomass parameters occurred with the increase in average velocity from 0.22 $m.s^{-1}$ to 0.45 $m.s^{-1}$ (Figure 4.3). Sewage fungus was only visually apparent at the two higher velocities. The plates subjected to velocities from 0.57 to 0.36 $m.s^{-1}$ during the experiment developed sewage fungus similar to that on the plates at 0.45 $m.s^{-1}$. At 0.16 and 0.22 $m.s^{-1}$ the biomass consisted largely of detritus and snails. *Physa acuta* was by far the most abundant snail but *Gyraulus corinna* and *Potamopyrgus antipodarum* also occurred. The former two species belong to the families Planorbidae and Physidae respectively, which contain the snails most resistant to organic pollution in North America (Harman, 1974). *P. antipodarum* is a general scavenger and detritivore (Towns, 1979) and is capable of maintaining a "normal" respiration rate in water with a dissolved oxygen concentration as low as 3 $g.m^{-3}$ (Hudson, 1975).

At the two lower average velocities (0.16 and 0.22 $m.s^{-1}$) these snails accounted for 40% of the settled volume and approximately 60 to 80% of the AFDW and numbered on average 13 per 150 cm^{-2} plate. At the higher average velocity (0.45 $m.s^{-1}$) 0 to 5 snails occurred per plate and they contributed only 0 to 5% of the total AFDW.

Grazing by snails has been shown to increase epilithic algal growth rates at low snail densities (120 snails.m^{-2}), this effect being attributed to the snails releasing bound nutrients (Kedhe and Wilhm, 1972). However it is likely that at the snail densities observed at the two lower current velocities (870 snails.m^{-2}) in this experiment their grazing may have contributed to the reduction in epilithon development. An ingestion rate of 3.1 ug.mg dry $weight^{-1}.h^{-1}$ was recorded for *P. antipodarum* fed on thin, predominantly heterotrophic epilithon at 10 $\pm$ 2°C (Rounick and Winterbourn, 1983). From

this the snails on the plates at 0.22 $m.s^{-1}$ (biomass = 2.8 g $AFDW.m^{-2}$) can be calculated to consume 2.1 g dry weight.m^{-2} of epilithon during the 5 day growth interval under the river conditions (assuming that the grazing rate Q_{10} = 2 and 1 g AFDW = 1 g snail dry weight).' This suggests that the decrease in snail density with increasing current velocity from 0.22 to 0.45 $m.s^{-1}$ could account for approximately 20% of the increase in slime biomass observed.

These results and previous findings (Section 2.3.4.5) suggest that increasing the current velocity above the average of 0.22 $m.s^{-1}$ may have resulted in increased growth of sewage fungus via three mechanisms:

(i) increasing the mass transport above the growth surface of organic material, nutrients and oxygen (Zimmerman, 1961)

(ii) increasing the turbulence at the growth surface and thus improving the mass transfer of organic material, nutrients and oxygen into the epilithon and suspending the growths in the water column, increasing the surface area for mass transfer (Castaldi and Malina, 1981; Rickey, 1982).

(iii) reducing the grazing rate by controlling the density of grazing snails to a lower level than at the lower velocities (Ormerod *et al.*, 1966).

In the river it is not possible to study the importance of the snails' grazing in limiting the epilithon growth. In preventing the snails from colonising the plates the current flow over the surface is disturbed. However this could be studied in duplicate laboratory channel systems by comparing the epilithon growth rates in the presence and absence of introduced grazing macro-invertebrates.

4.4.3 EFFECTS OF SPATES

Data on the effects of a spate on sewage fungus biomass data was collected from Site C (Fig. 1.2) on 20 September 1982. The

biomass adhering to plates incubated for 13 days under relatively uniform conditions (Table 4.5) was measured immediately before, and thirty minutes after, the onset of a spate (Table 4.6).

Within this thirty minute period the river flow increased from 48 to 142 $m^3.s^{-1}$ (Watson, 1984), the water level increased by 0.5 m and the average current velocity of the plate incubation site increased from approximately 0.3 $m.s^{-1}$ to 0.7 $m.s^{-1}$. The suspended biomass, caught in a 1.7 mm mesh net was also measured thirty minutes after the onset of the spate and compared with the value recorded at Site C on 4 October 1982 under conditions very similar to those on 20 September 1982 prior to the spate.

Subsequent to the second biomass sampling the river level rose by a further 0.4 m during the next thirty minute period. This prevented the collection of further samples. The river flow eventually peaked at 310 $m^3.s^{-1}$, 3.25 hours after the first abrupt rise in flow (Watson, 1984).

The conditions during the 13 day incubation period allowed the development of a mixed phototrophic/heterotrophic epilithon with significant biomass of *S. natans* overgrowing a diverse diatom community (Plate 4.8). This is reflected in the intermediate biomass P/H index values recorded.

The reduction in attached biomass and the increase in suspended biomass thirty minutes after the arrival of the fresh show that an abrupt rise in current velocity causes large, immediate sloughing losses.

The results of a growth experiment at Site C from 2 December 1983 to 6 January 1984 indicate the heterotrophs are dislodged more readily by increases in current than the phototrophs. Figure 7.10 (Section 7.2.6.2) shows that following a number of small spates the average AI value of the biomass on substrates at Site C dropped from 490 on 7 December 1983 to 227 on 22 December 1983. After a period of stable flows the average AI

TABLE 4.5 Conditions at Site C, 7-20/9/82

Parameter	Values
River discharge ($m^3.s^{-1}$)	59 ± 16
Temperature (°C)	12 ± 1.5
Surface Total radiation (Ohakea):	
i) ($MJ.m^{-2}.d^{-1}$)	12 ± 3.3
ii) Daytime Average ($W.m^{-2}$)	290 ± 76
Increase in river BOD_5 due to discharge ($g.m^{-3}$):	
i) PNCC (5.6 km upstream)	1.2 ± 0.2
ii) MCDC (1.8 km upstream)	2.2 ± 0.6

TABLE 4.6 Effects of a Spate on Benthic and Suspended Biomass

Parameter	Before Spate	30 minutes into Spate
Plate Biomass		
AFDW ($g.m^{-2}$) (x ± S)	21.33 ± 5.3 (n=14)	14 ± 2.6 (n=5)
Settled volume ($ml.cm^{-2}$) (x ± s)	0.226 ± 0.055 (n=14)	0.079 ± 0.013 (n=5)
P/H index (x ± S)	15.6 ± 2.8 (n=8)	ND
Suspended Biomass ($mg.m^{-3}$)	35*	1006

* value recorded site C 4/10/82 under conditions very similar to those prior to fresh on 20/9/82.

Plate 4.8: Epilithon on Artificial Substrates at Site C on 20/9/82 after Thirteen Days Incubation (heterotroph abundance level=4; phototroph abundance level=D).

value increased to 477 on 6 January 1984. Similar trends occurred at Site B during the same interval (Figure 7.10).

The effects of a spate on epilithon respiration are discussed in Section 5.2.2.2. The effect of river flow on *Potamogeton crispus* development is discussed in section 7.2.5.

4.4.4 DISCUSSION

The results of the bed biomass observations and the plate growth experiment suggest that a critical current velocity exists below which sewage fungus does not develop. The plate growth experiment indicated that for development on bare surfaces, under experimental conditions, this velocity was between 0.22 $m.s^{-1}$ and 0.44 $m.s^{-1}$ (measured 50 mm above the plates). By contrast, the river bed biomass observations indicate that where filamentous algae have been able to become established (possibly during periods of higher current velocity), providing a very large surface for heterotroph attachment in the water column, the critical velocity is between 0.18 and 0.22 $m.s^{-1}$ and the optimal range is from about 0.2 to 0.45 ms^{-1} for similar water quality conditions to those during the biomass observations.

The lower critical velocity where the algae occur probably results from these providing an attachment site above the "boundary layer" low velocity region that exists 1 to 3 mm above the substrate surface (Hynes, 1970). Above the boundary layer, which is reduced in thickness with increasing current velocity, the current velocity increases rapidly and effectively increases the nutrient supply and the shear stress on the grazing snails, possibly limiting their abundance. Clearly, if the grazing rate is reduced the rate of biomass development will be increased.

These critical values for sewage fungus development are somewhat higher than those reported by previous authors (Section 2.3.4.5) and may reflect the low organic levels of the water at the study sites and/or the grazing pressure due to the snails in the plate experiments in this study.

The observation of sewage fungus growth at a bed current velocity of 1.16 $m.s^{-1}$ extends the range over which growths have been observed above the previous maximum of 0.9 $m.s^{-1}$ reported by Curtis and Harrington (1971). However the presence of the filamentous algae *C. glomerata* amongst the assemblage may have facilitated the sewage fungus growth at the highest velocities. Although the aesthetic degradation due to sewage fungus was reduced at current velocities above 0.6 $m.s^{-1}$, the organic material and oxygen uptake rate by these lower biomasses may still be significant since thin films have been shown to have high weight specific activities (Section 2.3.6; Capblancq and Cassan, 1979[b]).

4.5 GROWTH INHIBITION BY SOLAR RADIATION

4.5.1 BED BIOMASS OBSERVATIONS

Benthic biomass was observed at depths from a few centimetres to 0.5 m at Site C (Fig. 1.2) on 11 January 1984. Luxuriant growths of *S. natans* dominated sewage fungus were observed on both stones and strands of *Cladophora* within a few centimetres of the water surface (Plate 4.5) and no qualitative, visual changes in sewage fungus biomass were observed with increasing depth.

The conditions at the observation site over the week prior to 11 January 1984 are summarised in Table 4.7.

The growths just below the water surface would be subjected to solar radiation levels very close to the surface radiation values given in Table 4.7. Under these conditions *S. natans* is not prevented from forming heavy growths. This was despite the average surface total radiation during the 15 day-light hours being 3 to 4 times higher than the levels, produced by high pressure mercury lamps, which inhibited *S. natans* growth and respiration irreversibly after three hours exposure in the laboratory (Favre, 1975) (Section 2.3.4.3). However these observations do not eliminate the possibility that these levels of solar radiation reduced the heterotrophic growth rate to

TABLE 4.7 Conditions at Site C 4/11/84 - 11/1/84

Parameters	Values $x \pm s$	range
River flow ($m^3.s^{-1}$)	16.1 ± 0.9	15.1-17.3
Total Surface Radiation*:		
i) Input ($MJ.m^{-2}.d^{-1}$)	24.8 ± 4.9	18.7-32.8
ii) Daytime average ($W.m^{-2}$)	458 ± 90	350-610
iii) Calculated daytime average near UV radiation** (300-380 nm) ($W.m^{-2}$)	25	19-34
iv) Calculated average daytime PAR*** ($\mu E.m^{-2}.s^{-1}$)	930	710-1240
Dissolved oxygen ($g.m^{-3}$)	N.D.	5-10
Temperature (°C)	N.D.	17-20
Calculated average BOD_5 **** ($g.m^{-3}$)	4.8	N.D.

* Measured at Tiritea 8 km east of Site C

** Calculated assuming that near UV radiation = 5.5% of total radiation (Basher, 1984)

*** Calculated assuming uE.m-2.s-1 PAR = 2.03 $W.m^{-2}$ TR, from numerous surface observations

**** Calculated from average river flow and waste discharge data using the first order decay equation (4.1) (Section 4.3.1) and assuming:

i) k_1 (PNCC-B) = $0.07.hr^{-1}$ (average for reach PNCC-B 29/2-1/3/84, Sectior 5.2.2.1)

ii) k_1 (MCDC-D) = 0.157.hr-1 (average for reach MCDC-Cu, 29/2/83, Section 5.3.2.1)

iii) BOD_5 above PNCC = 0.7 g.m-3 (measured grab sample on 7/1/84)

TABLE 4.8 Conditions During Plate Growth Experiment, Site D, 29/11/83-2/12/83

Parameters	Values
River flow ($m^3.s^{-1}$)	18.4-25.8
Total Depth at Site (m)	0.75-0.9
Depth of plate level (m)	0.45-0.6
Current velocity at plate level ($m.s^{-1}$)	0.26-0.30
Temperature 29/11/83-30/11/83 (°C)	14.2-16.2
D.O. 29/11/83-30/11/83 ($g.m^{-3}$)	8.35-11.7
Calculated average BOD_5* ($g.m^{-3}$)	7.8
Measured BOD_5 1230-0230 hr 30/11/83 ($g.m^{-3}$)	5.3
Total Surface Radiation**	
i) Daily input ($MJ.m^{-2}.d^{-1}$, $x \pm S$)	22.9 ± 4.4
ii) Daytime average ($W.m^{-2}$)	451
iii) Calculated average daytime PAR*** ($uE.m^{-2}.s^{-1}$)	915
iv) Calculated average daytime PAR**** at plate depth (0.52 m) ($uE.m^{-2}.s^{-1}$)	490

Upstream discharges operating BCWS, MCDC and PNCC

* Calculated from waste discharge and river flow data using the first order decay operation (4.1) (Section 4.3.2) and assuming:

i) k_1 (reach PNCC-B) = $0.07.hr^{-1}$ (average for 29.2.84- 1/3/84, Section 5.3.2.1)

ii) k_1 (reach MCDC-D) = $0.157.hr^{-1}$ (average for MCDC-Cu on 29/2/84, Section 5.3.2.1)

iii) BOD_5 above PNCC = 0.2 $g.m^{-3}$ (grab sample 30/11/83)

** Measured at Tiritea, 8 km east of study site.

*** Assuming PAR ($uE.m^{-2}.s^{-1}$) = 2.03 TR ($W.m^{-2}$), derived empirically from numerous observations made under a range of cloud conditions.

**** Assuming the vertical extinction coefficient for PAR (k_e) = 1.2 (value measured at Site D on 21/11/83).

some extent. To investigate sub-lethal effects a growth experiment on concrete plates was conducted.

4.5.2 PLATE GROWTH EXPERIMENT

In order to investigate the effects of natural solar radiation on sewage fungus growth rate in a river, biomass development was compared on the illuminated upper surfaces and shaded lower surfaces of concrete plates held horizontally in the middle of the water column in a supporting rack. The experiment was conducted over three days from 29 November 1983 to 2 December 1983 at Site D (Fig. 1.2). The conditions during the experiment are summarised in Table 4.8 and the results of the epilithon analyses are presented in Table 4.9.

TABLE 4.9: Sewage Fungus Growth, Site D, 29/11/83-2/12/83

Parameters	Upper Surfaces ($x \pm s$)	(n)	Lower Surfaces ($x \pm s$)	(n)	t-test Significance level (P)
AFDW ($g.m^{-2}$)	5.27 ± 1.61	5	2.85 ± 0.46	5	0.032
Chlorophyll a ($mg.m^{-2}$)	9.7 ± 1.1	2	1.4 ± 0.8	2	0.073
AI values	666 ± 5	2	2490 ± 1040	2	n.s.
Settled volume ($ml.cm^{-2}$)	0.045 ± 0.016	5	0.049 ± 0.015	5	n.s.
Relative heterotrophs* abundance	S > F = Z		S > F > Z		

* = where S = *Sphaerotilus natans*; F = *Flavobacterium* spp; Z = *Zoogloea* spp.

The data in Table 4.9 show that there was no significant difference between the biomass settled volume values on the exposed and shaded sides of the plates but that AFDW and chlorophyll a values were significantly higher on the upper surface. The heterotrophic community structure was similar on both the illuminated and shaded surfaces. Phototrophs

(diatoms) were more abundant on the upper surface resulting in lower AI values.

The higher biomass, measured as AFDW, on the upper surfaces cannot be entirely accounted for by the increased algal biomass present. Assuming that the chlorophyll a represents 1 to 2% of algal biomass (Cooper and Wilhm, 1975), the algal biomass only accounts for 0.4 to 0.8 g AFDW.m^{-2} of the average of 2.4 g.AFDW.m^{-2} of additional biomass noted on the upper than the lower surfaces. Visual and microscopic examination showed more detritus to be present amongst the upper surface biomass and this may have contributed to the increased AFDW. This suggests that a better approach may have been to have compared the growth on the upper surfaces of two sets of plates, one of which was beneath a shade above the water surface. This was considered but decided against due to likely problems with vandalism of such obvious apparatus left in the field and the fairly conclusive results obtained in this experiment with respect to settled volume values, which are much less effected by settled detritus than are AFDW values.

These results do not suggest any significant inhibition of the growth of sewage fungus under the light conditions experienced in this experiment. No reduction in the total biomass (as AFDW or settled volume) was observed under conditions of near maximum summer illumination at a depth of 0.45 to 0.6 m when compared with the biomass which developed simultaneously at almost zero light. Since the average daytime surface total radiation during the experiment was about 10% greater than the average recorded during December 1983, when radiation values were maximal, and since the plates were held at a height 0.2-0.5 m above the average river bed level during flows between 13 and 30 $m^3.s^{-1}$, the results suggest that solar radiation inhibition of benthic heterotrophic growth is not important in the Manawatu river at any time during the year.

4.5.3 DISCUSSION

The biomass observations and plate growth experiments do not

support the hypothesis that solar radiation has an important inhibitory effect on *S. natans* development in nature (Eichenberger, 1975; Favre, 1975). One possible explanation for Favre's observations of total inhibition of *S. natans* at radiation levels of around 120 $W.m^{-2}$, well below the maximum surface radiation in the Manawatu River, is that this resulted from an experimental artifact due to the discontinuous spectrum of the high pressure mercury lamps used in his study (Fig. 2.9). This spectrum may have contained levels of damaging near-ultraviolet radiation but been lacking in some of the wavelengths present in the solar spectrum required for activation of the cells' photo-repair mechanisms.

The results of the plate growth experiment also conflict with observations of the effects of seasonal variations in surface total radiation on *S. natans* growth in 0.15 to 0.2 m deep outdoor channels fed domestic sewage (Eichenberger, 1975; Section 2.3.4.3).

Experiments in which insecticide was added to these channel systems showed that the grazing of Orthoclad Chironomids has a very marked effect on the attached biomass in the channels fed 2% domestic sewage (Wuhrmann, 1974). Thus seasonal variations in the grazer numbers and activity could result in seasonal variations in the heterotrophic abundance in the channels receiving more dilute sewage. However, since the Orthoclads did not occur in the channel receiving 12% sewage (Wuhrmann, 1974), this cannot explain the observation of reduced growth rates at the beginning of the channel receiving this inflow during periods when the daily surface total radiation input was 14 to 41% lower than the average value recorded during the plate growth experiment (Eichenberger, 1975; Section 2.3.4.3). This suggests that, unless some unknown seasonal effect was involved in the channel study results, the contrasting results of the effects of similar surface total radiation levels on heterotroph growth rates may have resulted from qualitative and/or quantitative differences in the radiation reaching the growth surfaces in the two studies.

Nevertheless, the results of the biomass observations show that in the Manawatu River solar radiation does not inhibit the development of luxuriant sewage fungus near the water surface during summer. Furthermore, the results of the plate growth experiments indicate that summer solar radiation levels do not cause any reduction in sewage fungus growth rate on substrates suspended 0.2 to 0.5 m above the average river-bed level at typical summer flows of between 13 and 30 $m^3.s^{-1}$.

CHAPTER 5.

EFFECTS OF BENTHIC COMMUNITIES ON STREAM OXYGEN DYNAMICS

5.1 INTRODUCTION

Benthic communities have been shown to have large effects on stream oxygen dynamics in many rivers including the Manawatu (sections 2.2.1.2 and 2.3.5), where they have been implicated as an important factor producing low dissolved oxygen conditions leading to fish kills (Currie, 1980). Thus a better understanding of the relationships between the factors involved would be most useful for the management of the river to maintain the statutory dissolved oxygen requirement (Section 2.1) and avoid fish kills.

This chapter presents the results of investigations of the factors affecting the respiration and photosynthetic oxygen production rates of benthic communities in the Manawatu River and in laboratory channels.

The aims of the investigations were:

(i) To quantify the relationships between the benthic respiration rate (BR) and gross photosynthetic production rate (BGPR) and the environmental variables namely: biomass, community composition, temperature, BOD_5 concentration, dissolved oxygen concentration and light.

(ii) To use these data to produce models of the factors affecting these processes that would be useful for management purposes.

(iii) To assess the relative importance of benthic and suspended biomass respiration in the Manawatu River.

(iv) To define the nuisance level of benthic biomass which would result in depletion of the river dissolved oxygen to less than the 5 $g.m^{-3}$ minimum concentration permitted by the D classification (Section 2.1).

In-river studies were undertaken using BOD bottles, *in situ* respiratory chambers and two station techniques (Sections 3.6.2 to 3.6.5). The oxygen removal and production of epilithon developing in laboratory artificial channels were studied using respiratory chambers (Section 3.7.10).

5.2 SUSPENDED BIOMASS EFFECTS ON OXYGEN REMOVAL

The respiration rate of the biomass suspended in the water column was investigated over periods of 1.2 to 4 hours in experiments using a stirred Boyle respirometer or quiescent BOD bottles (Section 3.6.2). In these experiments the suspended biomass respiration averaged only 0.06 g $O_2.m^{-3}.h^{-1}$ (= 1.4 g $O_2.m^{-3}.d^{-1}$) but ranged from 0 to 0.18 g $O_2.m^{-3}.h^{-1}$ (Table 5.1).

The average initial respiration rate, calculated from BOD_5 progression studies, of samples collected between sites D and E (Fig 1.2) was just within the range of values observed in this study at 0.17 g $O_2.m^{-3}.h^{-1}$. (Hickey and Rutherford, 1983).

Comparison of these data with the results of the *in situ* chamber benthic respiration studies (Section 5.3) and whole river studies (Section 5.4) indicates that at moderate to high benthic biomass levels the suspended biomass respiration has only a very minor affect on total oxygen removal.

5.3 *IN SITU* CHAMBER STUDIES IN THE MANAWATU RIVER

5.3.1 INTRODUCTION

Submersible respiratory chambers (Sections 3.6.2 and 3.6.3) were used to study *in situ* in the Manawatu River the effects on river dissolved oxygen dynamics of benthic communities and the interactions of benthic processes with physical and water quality factors. The biomass within the chambers and the physical and water quality conditions during the experiments could be controlled and quantified (Sections 3.6.2 and 3.6.3). Thus these studies allowed collection of data suitable for construction of mathematical models for prediction of benthic respiration and gross photosynthetic oxygen production rates.

TABLE 5.1 Water Column Respiration Rates, Manawatu River

Date	Site (Fig 1.2)	Respiration Rate ($gO_2.m^{-3}.hr^{-1}$)	Specific Respiration Rate ($gO_2.gAFDW^{-1}.hr^{-1}$)	Apparatus
07/12/82	D	0.18	0.063	Boyle Respirometer
16/12/82	D	0.07	0.016	Boyle Respirometer
06/01/83	D	0.09	0.016	Boyle Respirometer
18/02/83	E	0	0	Boyle Respirometer
02/03/83	A	0	0	BOD Bottles
04/03/83	B	0	0	BOD Bottles
04/03/83	C	0	0	BOD Bottles
07/03/83	D	0.08	ND	BOD Bottles
07/03/83	EF	0.17	ND	BOD Bottles
11/04/83	D	0.05	ND	BOD Bottles
12/04/83	C	0.07	ND	BOD Bottles
23/11/83	D	0.03	0.012	BOD Bottles
16/12/83	D	0.03	0.036	BOD Bottles
$\bar{x} \pm s$		$0.06 \pm .06$	$0.016 \pm .016$	
$24 \times \bar{x}$		$1.44\ gO_2.m^{-3}.d^{-1}$	$0.384\ gO_2.gAFDW^{-1}.d^{-1}$	

The following experiments were undertaken:

(i) study of the activity of epilithon developing on concrete artificial substrates (Section 3.2.2.2) at regular intervals during incubation at site D in the Manawatu River (Fig 1.2) at the beginning of the 1983/84 summer period.

(ii) Concurrent studies in two Freeman chambers (Section 3.6.3) of the activity of epilithon attached to natural stones under river conditions of temperature, initial water quality and light and after alteration of one of these variables (Sections 3.6.3.5 to 3.6.3.7).

(iii) Studies of the respiration of epilithon on natural substrates using the Boyle chamber (Section 3.6.2).

The data collected in these experiments is summarised in Appendices C, D and E.

In the river studies the results were obtained from single measurements except on 11/4/83, when duplicate respiration measurements were made using the Boyle chamber. On this occasion the standard deviation of the mean was 5.5%. Similarly during the early stages of the laboratory channel experiments, when the biomass was similar throughout the channels, the standard deviations were generally less than 10% of their respective mean areal respiration and gross photosynthetic rates (Section 5.5.2 and 5.5.3), determined from duplicate measurements using the Freeman chambers. This suggests that the chamber measurements gave reasonably precise measurements of the respiration and gross photosynthetic rates.

The respiration and gross photosynthesis of biomass suspended in the water column were not allowed for in calculating the benthic rates from the chamber measurements (Sections 3.6.2 and 3.6.3) since the suspended biomass activity had a negligible effect on the rates measured due to the high bed surface area to water column volume ratio of the chambers (Section 6.2.2). At the maximum observed suspended biomass respiration rate of 4.3 g $O_2.m^{-3}.d^{-1}$ (Section 5.2), this only represents 0.6 to 7.2% of the total chamber respiration rate over the range of measured values. This suggests that neglecting this portion of the

total chamber respiration could have resulted in over-estimation of the benthic respiration by up to 7.2% for the lowest benthic respiration rate measured if the maximum suspended biomass respiration rate occurred. However in most cases the over-estimation would be less than 2%.

Suspended biomass gross photosynthesis was assumed to have a similar negligible effect. River water incubated in the light at site D for four hours on 22/2/83 showed no change in dissolved oxygen suggesting that aquatic production was small (Hickey and Rutherford, 1983).

For the purposes of data analysis, those communities in which the heterotrophic abundance scale value (Appendix B) was 4, 5 or 6 were classed as sewage fungus whereas all the other bed communities were classed as phototrophic. However, most of the sewage fungus communities contained a significant algal component, as is reflected in their AI values (Appendices C and E).

The phototrophic communities studied were algal except on 12 March 1984 when the macrophyte *Potomogeton crispus* was compared with the algae *Cladophora glomerata* for its effects on oxygen dynamics at site EF.

The interactions of the various physical and water quality parameters with the benthic communities effects on oxygen dynamics was analysed by the following techniques:

(i) Linear regression analysis of the interactions between variables and the respiration and gross photosynthetic rates obtained from studies in which the physical water quality characteristics of the river water in the chamber were not altered experimentally. This technique produced empirical mathematical models with potential usefulness in water management.

(ii) Comparison of the benthic communities effects on oxygen dynamics with and without experimental alteration of

light and/or temperature and/or water quality conditions.

5.3.2 PLATE BIOMASS STUDIES AT SITE D

The respiration and gross photosynthetic oxygen production of epilithon developing on concrete plates was investigated using the *in situ* chambers (Section 3.6.3.3) during two studies at site D at the beginning of the summer growth period in November and December 1983. The aims of the experiments were to investigate:

(i) the relationships between epilithon biomass and its effects on stream oxygen dynamics

(ii) the changes in epilithic community type during growth below the wastewater discharges.

The effects of organic waste addition on epilithon development during the latter experiment are discussed in Section 7.2.6.2.

The results of the two experiments are presented in Figures 5.1 and 5.2 and Appendix C.

Epilithon developed and increased in activity rapidly under the conditions existing during the incubation (Figs 5.1 and 5.2) but there was a general decline in both the weight specific benthic respiration rate (WSBR) and the chlorophyll a specific benthic gross production rate (chla.SBGPR) with increasing biomass. However two exceptions to this pattern occurred. A low WSBR value was recorded on 18/11/83 (Fig 5.1) and the decrease in biomass between 10 and 16/12/83, resulting from a small spate on 11/12/83, did not result in increased WSBR or chla.SBGPR values (Fig 5.2). The former anomaly resulted from the large amount of detritus and silt which settled on the plate surfaces between 15 and 18/11/83 causing elevated AFDW values. The second anomaly probably resulted from the combined effects of increased current velocity causing preferential sloughing of the heterotrophs (Sections 4.3.3 and 7.2.6.2) and the reduction of the algal respiration in response to lower average light conditions (Boardman, 1977) in the turbid conditions during and following the spate.

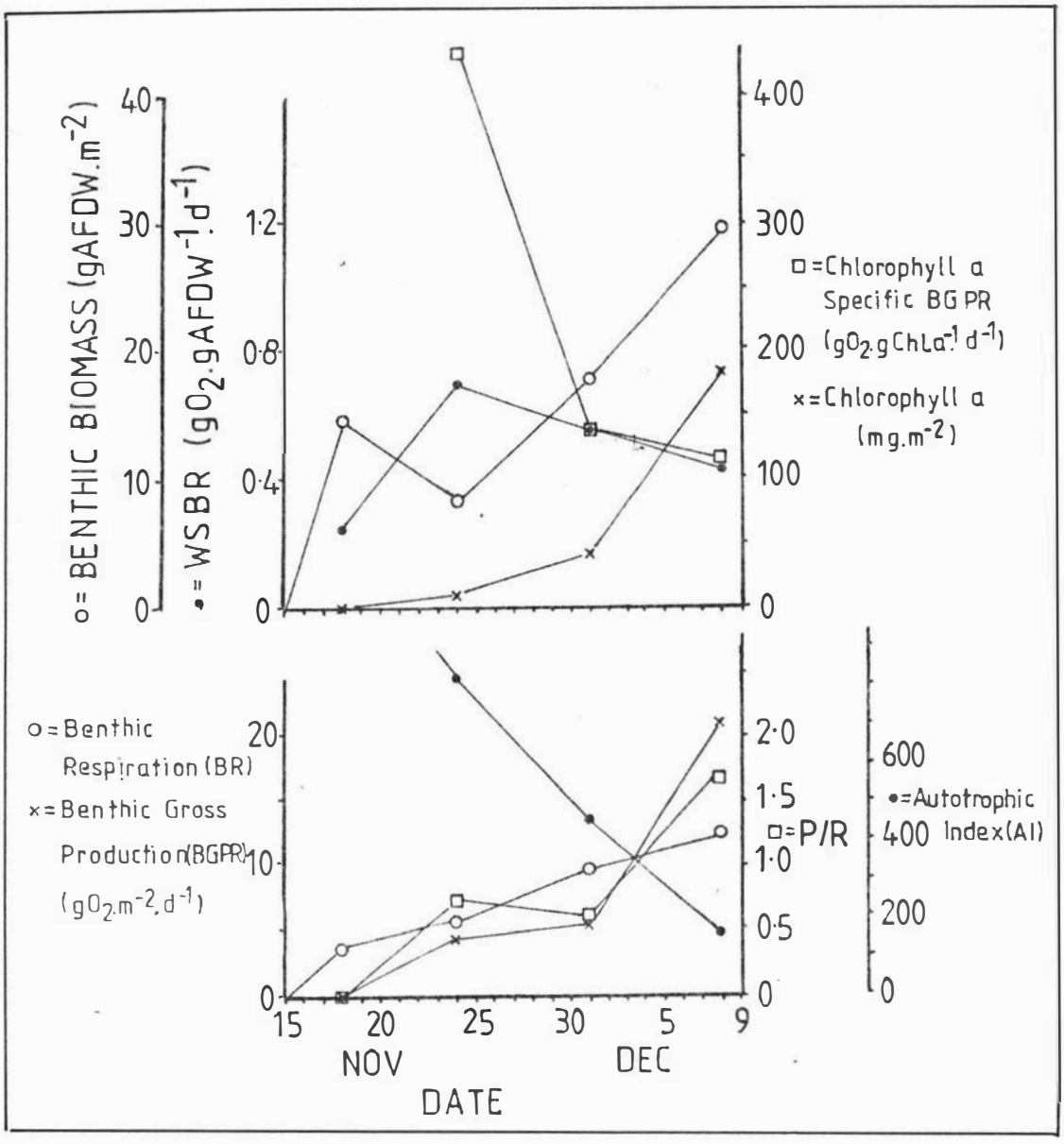


Figure 5.1: Results of Measurements of Epilithon Growth and Activity on Concrete Plates at Site D, 15/11/83 to 8/12/83.

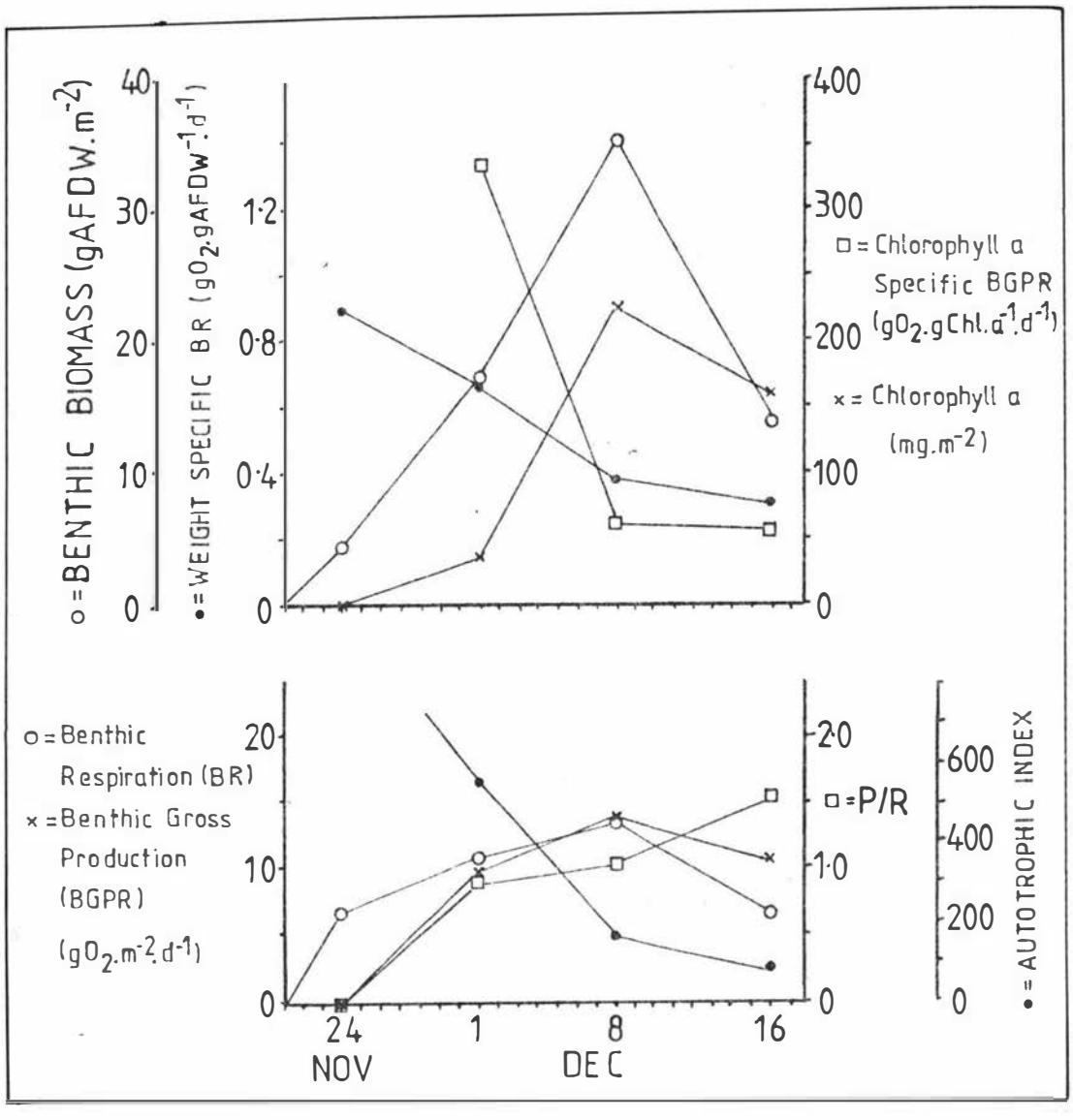


Figure 5.2: Results of Measurements of Epilithon Growth and Activity at Site D, 21/11/83 to 16/12/83.

Analysis of the combined data from the two experiments indicates strong relationships between the WSBR and chlorophyll a specific BGPR and natural logarithm of the benthic biomass (as AFDW) and algal biomass (as chlorophyll a) respectively (Figs 5.3 and 5.4). This is discussed in Section (5.5.4).

The multiple regression analyses of the factors affecting the BR and BGPR were prevented by the significant correlations between the temperature and both benthic biomass (AFDW) ($r = 0.840$; $P < .02$) and algal biomass (chlorophyll a) ($r = 0.896$; $P < .01$). However these data were used for multiple regression analysis in combination with natural bed biomass in Section (5.3.5).

In both experiments there was a steady decline in the autotrophic index values and a general increase in the P/R values during the incubation interval (Figs 5.1 and 5.2). This implies that under the river conditions, the substrates were initially colonised by heterotrophs but the relative abundance of phototrophs increased with time.

5.3.3 NATURAL BED SEWAGE FUNGUS RESPIRATION

The benthic respiration rates (BR) of natural bed sewage fungus communities, defined as those communities in which the macroscopic heterotroph abundance scale values were 4, 5 or 6 (Appendix B), were measured using the *in situ* chambers over a range of biomass levels and conditions (Appendices D and E). The results of the studies in which the neither temperature nor waste-water concentration were altered experimentally indicate that the benthic biomass has a large influence on BR. This is confirmed by the parsimonious multiple regression equation (5.3) (Table 5.2) which shows that biomass and temperature account for 86.9% of the variation in benthic respiration. Although the initial BOD_5 was positively correlated with BR at the 90% confidence level it did not have a significant effect on the multiple regression model at the 95% level (t value = 0.85). This suggests that the initial BOD_5 has only a minor effect on BR over the range of

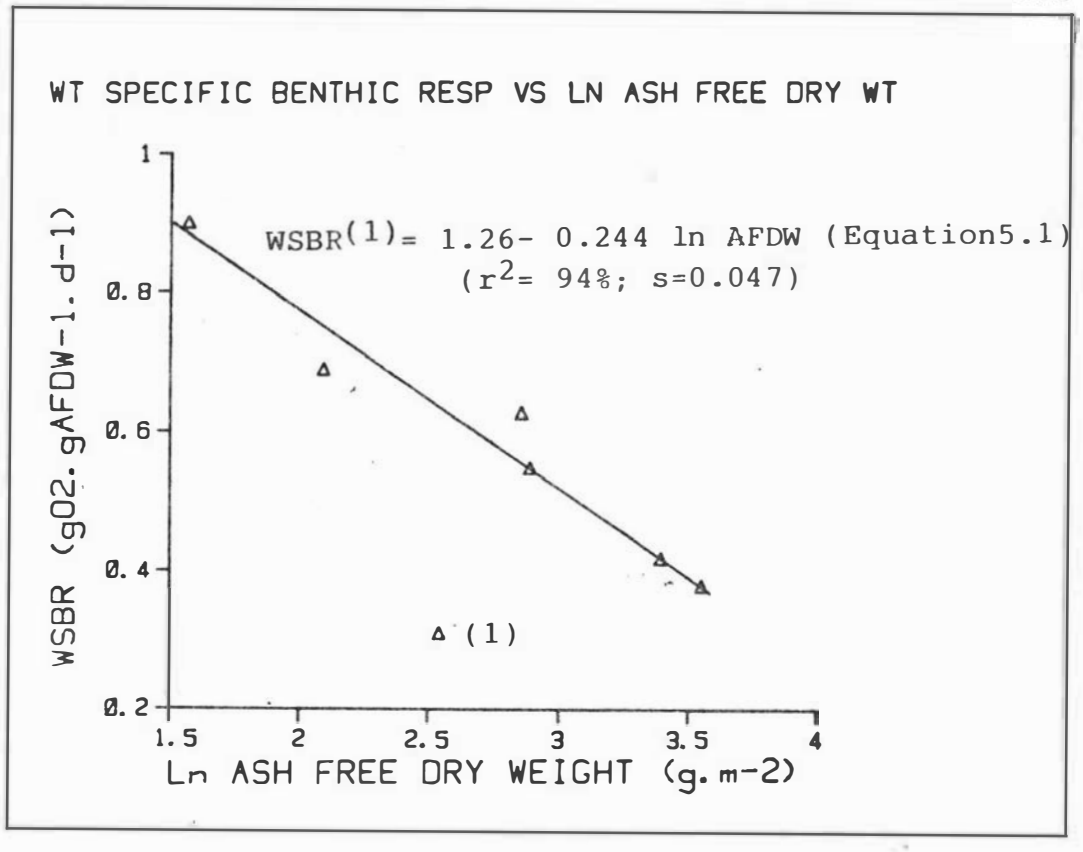


Figure 5.3: Graph of Weight Specific Benthic Respiration Rate Versus ln Ash Free Dry Weight of Plate Biomass at Site D (fig.1.2), 18 November to 16 December 1983.
(1)= datum point collected on 16/12/83 after a fresh had occurred causing sloughing & low light; excluded from mode

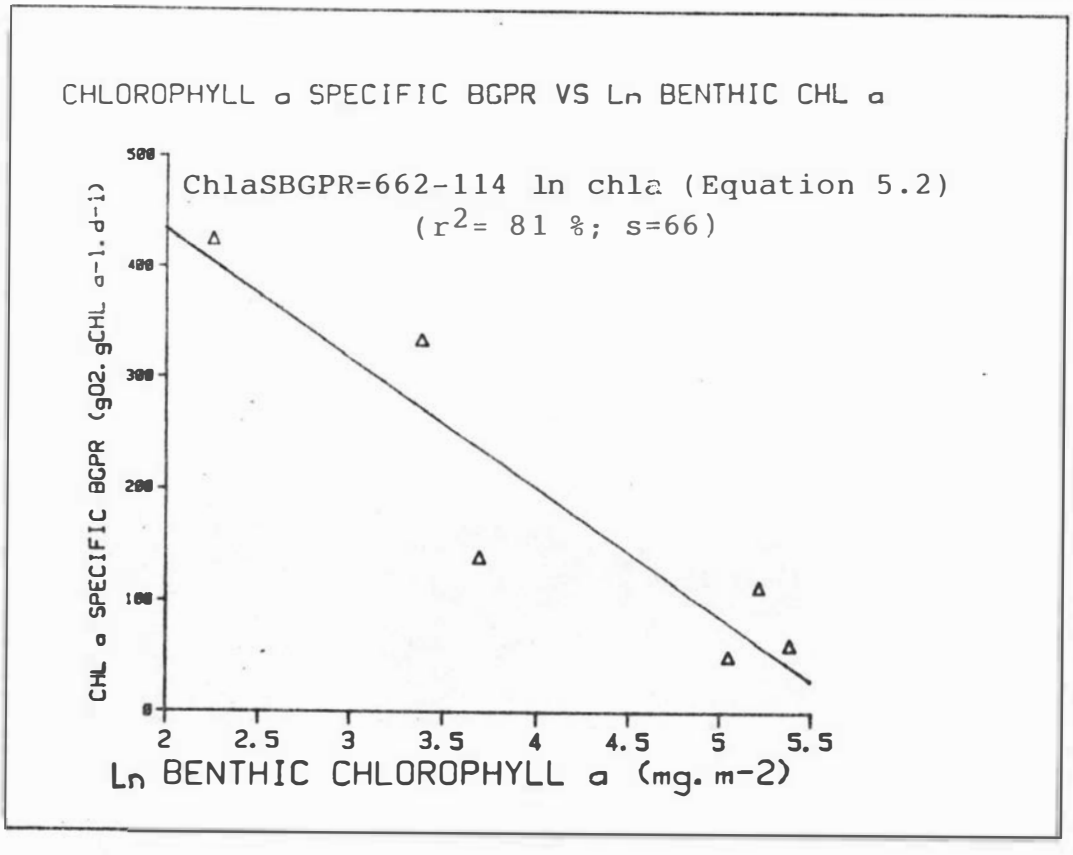


Figure 5.4: Graph of Chlorophyll a Specific Benthic Gross Production Rate Versus ln Chlorophyll a Concentration of Plate Biomass at Site D, 18 November to 16 December 1983.

TABLE 5.2 Parsimonious Regression Equation Describing the Effects of Statistically Significant Predictors (at the 95% level) on the Benthic Respiration Rate of Natural Bed Sewage Fungus Communities

$BR = -13.3 + 0.151\ BM + 1.07\ T$ (Equation 5.3)

Units = $gO_2.m^{-2}.d^{-1}$

r^2 = 86.9% (adjusted for 9 degrees of freedom); standard deviation of data about regression equation (s) = 2.28

Importance of Predictors

Predictor	Coefficient	St dev of Coefficient	t ratio = Coeff/S.D.	Percentage Contribution to SS reduction
Constant	-13.3	4.98	-2.77	
BM	0.15	0.02	6.69	65.5%
T	1.07	0.26	4.08	24.1%

Application Range of Regression Equation

BM = 8.5 - 106 g AFDW.m^{-2}

T = 11.0 - 20.7°C

Range of Other Variables During Experiments

Initial BOD_5 = 0.7 - 7.3 g.m^{-3}

Initial DO = 5.3 - 11 g.m^{-3}

concentrations and biomass values in these experiments (Table 5.2). However the apparent, unexpectedly, low influence of initial BOD_5 on sewage fungus BR may be partly due to the positive, though statistically insignificant (at P = 0.1), correlation between the initial BOD_5 and the benthic biomass (r = 0.512). To confirm the statistically insignificant effects of initial BOD_5 observed using multiple regression, simultaneous chamber studies were made of similar natural river communities with one chamber being filled with river water and another with river water supplemented with MCDC wastewater (Section 3.6.3.5). These showed that the average weight specific benthic respiration rate (WSBR), measured over 0.5h intervals, was not affected by addition of 6.7 g $BOD_5.m^{-3}$ at a high background level (7.0 $gBOD_5.m^{-3}$) nor by a minor increase of 1.8 $gBOD_5.m^{-3}$ at a background level of 4.0 $g.m^{-3}$ (Table 5.3). However a 36% increase in average WSBR was observed in one experiment when the particularly low background river BOD_5 (1.2 $g.m^{-3}$) was increased to 7.1 $g.m^{-3}$. These limited results suggest that short-term variations in organic concentration do not result in increased sewage fungus respiration unless the background organic material concentration is sufficiently low to cause the communities' reserves to become depleted. The effect of more prolonged BOD_5 increases due to waste-water discharge variations on sewage fungus respiration rates measured by the whole river, two station technique is discussed in Section (5.4.3).

TABLE 5.3 Effects of BOD_5 Addition on Sewage Fungus Respiration Rate at Site C

Date	Temperature (°C)	Benthic Biomass ($gAFDW.m^{-2}$)	Initial BOD_5 ($g.m^{-3}$)		Average Weight Specific BUR over 0.5 hr ($gO_2.gAFDW.d^{-1}$)	
			Back-ground	After Addition	@ River BOD_5	@ Elevated BOD_5
12/1/84	20.2	50-55	7.3	14.0	0.36	0.36
23/3/84	17.5	22-24	1.2	7.1	0.316	0.43
26/4/84	15	93-104	4.0	5.8	0.188	0.189

Two experiments were conducted to investigate the independent

effect of temperature on the respiration and gross photosynthesis of communities with a significant heterotrophic component. The average weight specific rates of similar communities were compared concurrently in river water at the ambient river temperature and in river water which had been chilled by 5°C using glycol-filled PVC pads (Section 3.6.3.7). These experiments gave conflicting results (Table 5.4). The anticipated reduction in average WSBR at the lower temperature was observed on 23/3/84 but not on 22/2/84. The anomalous result on 22/2/84 cannot be explained by different organic concentrations in the two chambers since these were found to be the same ($fBOD_5$ = 4.3 $g.m^{-3}$) and, since the initial dissolved oxygen concentration was not observed to have a significant effect on the respiration rate in the other chamber experiments, the 1.2 ppm higher initial dissolved oxygen concentration at the lower temperature is not likely to have countered the expected reduction in metabolic rate due to the reduced temperature. Thus the anomaly remains unexplained.

TABLE 5.4: Effects on Sewage Fungus Respiration and Gross Photosynthesis of Artificial Lowering of Water Temperature; Site C

Date	River Temp' (°C)	Weight Specific BR* ($gO_2.gAFDW^{-1}.d^{-1}$)		Weight Specific BGPR* ($gO_2.gAFDW^{-1}.d^{-1}$)	
		@ River Temp	@ River Temp -5°C	@ River Temp	@ River Temp -5°C
22/2/84	18.0	0.18	0.18	0.27	0.22
23/3/84	17.5	0.316	0.213	ND	ND

* Average values recorded over 0.5 hr, not the initial rates

5.3.4 NATURAL BED PHOTOTROPHIC COMMUNITY EFFECTS

Six *in situ* chamber studies of bed phototrophic communities were undertaken to compare their effects on benthic respiration

(BR) with those of the heterotroph dominated sewage fungus communities. In this analysis the phototrophic dominated communities were defined as those in which the phototrophic abundance scale value was E or F and the heterotroph abundance level less than 3 (Appendix B). These communities were algal, dominated by *C. glomerata*, except in one instance, on 12/3/84 at Site EF, when the activity of the macrophyte *Potamogeton crispus* was compared with that of *C. glomerata*.

Due to the small number of observations of bed algal communities' gross photosynthetic oxygen production rates (BGPR) (four) the combined data from all the epilithic communities was used to investigate the relationships between BGPR and algal biomass and other factors (Section 5.3.5).

Multiple regression analysis of the data from the algal phototrophic dominated communities in Appendices D and E showed that temperature had a significant effect on the benthic respiration rate (BR) at the 95% level (Fig 5.5) whereas the biomass (BM) and initial BOD_5 did not. The lack of any effect due to biomass results from the narrow range of benthic biomass values of the algal communities studied ($\bar{x} \pm s = 48 \pm 8$ $gAFDW.m^{-2}$).

The temperature coefficient of 1.84 in the algal communities' respiration equation (5.4) is considerably higher than the coefficient of 1.07 observed in the parsimonious equation (5.1) for the sewage fungus communities (Section 5.3.3). Although the data sets used to generate these equations are small they indicate that temperature has a stronger effect on algal than sewage fungus communities. This could be explained by the higher average temperatures resulting largely from higher average radiation inputs. Thus, at high temperatures the algal activity is increased due to the stimulatory effect of higher average light levels (Darley, 1982) in addition to the effect of temperature on cellular reaction rates (Section 2.3.4.7). By contrast the heterotrophs' activity is only enhanced by latter effect of increased temperature.

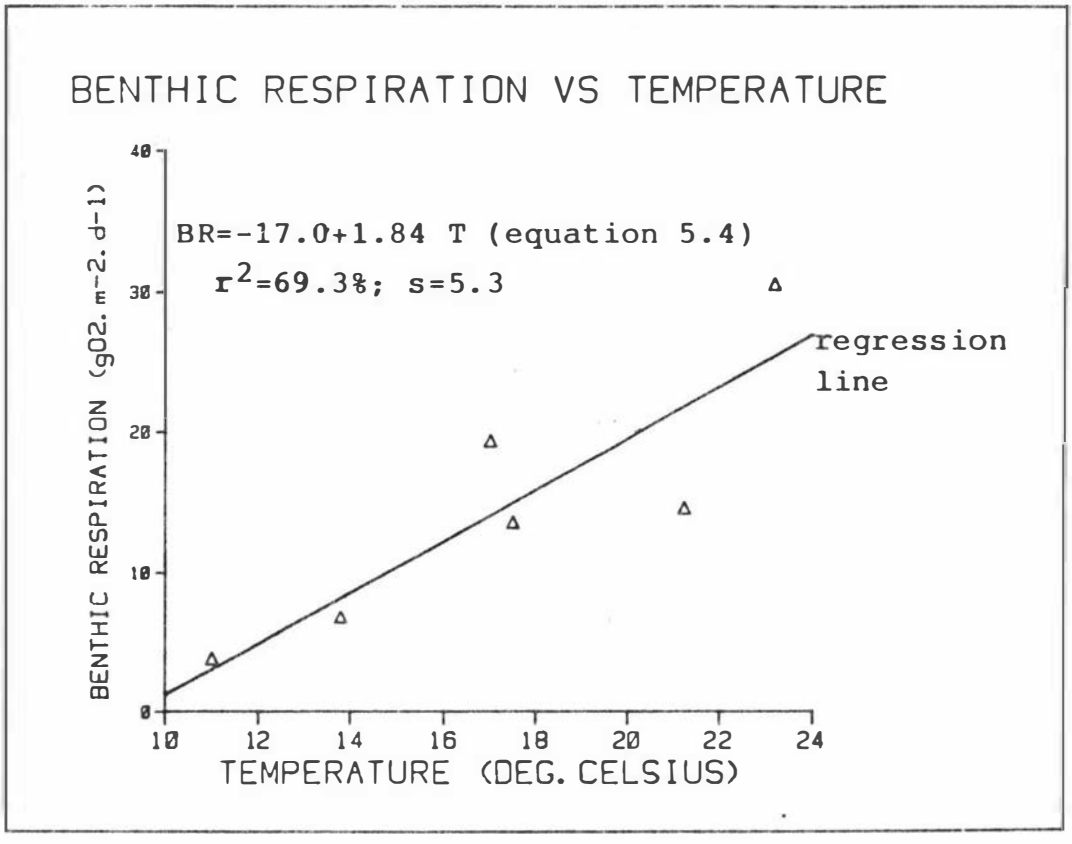


Figure 5.5; Benthic Respiration versus Temperature; Bed Algal Communities.

The independent short-term effects of wastewater addition on algal activity were investigated in two chamber experiments. In each experiment one Freeman chamber was filled with river water and another, containing a similar benthic community, was filled with river water supplemented with MCDC wastewater (Section 3.6.3.5). These investigations showed that under the study conditions, where the background concentrations of $fBOD_5$ and nutrients were relatively high, the wastewater additions had no significant effect on the average weight specific benthic respiration rates measured over 0.5 hour intervals or the weight specific benthic gross production rate measured over 0.4 hour intervals after the initial, 0.5 hour, dark incubation period (Table 5.5).

The respiration and gross photosynthetic oxygen production rates of natural bed communities dominated by the algae *Cladophora glomerata* and the macrophyte *Potamogeton crispus* were compared in chamber experiments conducted at site EF (Fig 1.2) on 12/3/84. The conditions during the experiments are summarised in Table 5.6 and the results are presented in Table 5.7.

The WSBR of *P. crispus* is much lower than that of *C. glomerata* (Table 5.7). Similarly at full sunlight the WSBGPR and chlorophyll a specific BGPR of *P. crispus* were respectively only 0.44 and 0.27 times that of *C. glomerata*. However, the data in Figure 5.7 show that at low light levels the two species specific BGPR's converge.

It is possible that the results of the macrophyte investigation could have been effected by storage and re-utilisation of oxygen in their internal lacunae causing a delay in the change in the dissolved oxygen concentration of the chamber water (Coffey, 1981). However the immediate response of the chamber oxygen concentration observed when the light conditions were altered suggests that this effect was not important.

The WSBR observed for *P. crispus* in these experiments was 2.5 to 4.3 times greater than the rates observed (per g dry

TABLE 5.5 Effect of MCDC Wastewater Addition on Respiration and Gross Production Rates of Algal Communities at Site DE

Parameter		
Date	18/1/84	2/2/84
Community Macroscopic Appearance	IIIE	2IF
Temperature (°C)	17.5	23.5
Initial $fBOD_5$ ($g.m^{-3}$; $\bar{x} \pm s$)		
(a) background	3.7 ± 0.3	3.1 ± 0
(b) after addition	6.0 ± 0.7	10.4 ± 0.15
Initial TDN ($mg.m^{-3}$; $\bar{x} \pm s$)		
(a) background	ND	442 ± 11
(b) after addition	720 ± 150	1020 ± 11
Initial TDP ($mg.m^{-3}$; $\bar{x} \pm s$)		
(a) background	65 ± 7	172 ± 11
(b) after addition	102 ± 8	556 ± 19
Average Benthic Respiration Rate over 0.5 h ($gO_2.gAFDW^{-1}.d^{-1}$)		
(a) background	0.27	0.50
(b) after addition	0.27	0.47
Benthic Gross Production Rate after 0.5 h incubation ($gO_2.gAFDW^{-1}.d^{-1}$ (12 hrD:N))		
(a) background	0.50	0.70
(b) after addition	0.47	0.74

TABLE 5.6 Experimental Conditions During Comparison of Benthic Algal and Macrophyte Communities, Site EF, 12 March 1984

Parameter	*P. crispus*	*C. glomerata*
Total Biomass ($gAFDW.m^{-2}$)	31	55.3
Algal Biomass (mg chlorophyll $a.m^{-2}$)	456	508
Temperature (°C)	20.2	21.2
Initial BOD_5 ($g.m^{-3}$) ($x \pm s$)	2.0 ± 0.1	1.7 ± 0.4
Initial DO ($g.m^{-3}$)	7.8	7.0
Initial TDN ($mg.m^{-3}$) ($x \pm s$)	505 ± 15	642 ± 3
Initial TDP ($mg.m^{-3}$) ($x \pm s$)	105 ± 0	81 ± 0

Table 5.7 Comparison of the Respiration and Gross Photosynthetic Oxygen Production Rates of the Algae *C. glomerata* and the Macrophyte *P. crispus*, Site EF, 12 March 1984

Parameter	*P. crispus*	*C. glomerata*	Ratio of Rates
BR ($gO_2.m^{-2}.d^{-1}$)	4.8	14.6	.33
WSBR ($gO_2AFDW^{-1}.d^{-1}$)	0.15	0.26	.58
BGPR ($gO_2.m^{-2}.d^{-1}$) (full sunlight)*	7.6	31.3	.24
WSBGPR ($gO_2.gAFDW^{-1}.d^{-1}$) (full sunlight)*	0.25	0.57	.44
chlaSBGPR ($gO_2.gchla^{-1}.d^{-1}$) (full sunlight)*	16.7	62	.27

* calculated for a 12 hour day, night cycle assuming constant BGPR at measured rate.

weight) for three macrophytes at 20°C in the laboratory (Owens and Maris, 1964). This probably reflects the different experimental conditons.

Although these results show clearly than *C. glomerata* has a greater biomass specific effect on river oxygen dynamics than *P. crispus*, the macrophyte can form much denser growths. Dense swards of *P. crispus* can grow to reach the surface at depths of 0.3 to 0.4 m in the Manawatu River. The biomass in one such sward at site EF on 12/3/84 was 420 $gAFDW.m^{-2}$ whereas the maximum biomass density for *C. glomerata* was approximately 70 $gAFDW.m^{-2}$ and the maximum observed during this study was 144 $gAFDW.m^{-2}$ at Site DE on 2/2/84.

Thus *P. crispus* swards could produce similar or greater effects than *C. glomerata* on river oxygen dynamics due to the higher biomass densities developed. However the biomass specific BR and BGPR of *P. crispus* in these dense swards may be lower than observed at the biomass concentrations used in the chamber experiment due to the self-shading and reduction in current velocity within the dense swards. Larger respiratory chambers (Hickey, 1982) or respiratory tunnels (James, 1974) would be required to study the respiration and photosynthetic oxygen production rates of the swards. Nevertheless, the much lower BGPR light saturation level observed for *P. crispus* than for *C. glomerata* (fig 5.7) suggests that the former sample, which was collected from a sward, was acclimated to lower incident light levels than the latter sample although both were collected from locations of similar depth and current velocity within 10 m of one another.

An alternative possible explanation is that *P. crispus* has a low light saturation threshold as an adaptation to allow exploitation of low light habitats. These alternative hypotheses could be tested by chamber studies on *P. crispus* samples from dense swards, where the average light conditions are reduced due to self-shading, with samples from more sparsely growing aggregations, where self-shading is minimal.

The weight specific respiration rates of these predominantly phototrophic communities are compared with those obtained from sewage fungus communities under similar biomass and temperature conditions in Table 5.8.

This comparison shows that WSBR values of the *S. natans* dominated sewage fungus communities were approximately 2.6 and 1.5 times greater than those of the predominantly phototrophic macrophyte and algal communities respectively. However the highest WSBR recorded in this biomass range (0.58 to 0.67 $g.O_2.gAFDW^{-1}.d^{-1}$ @ 46 to 49 $gAFDW.m^{-2}$) was obtained from a *C. glomerata* dominated community with a thin film of *S. natans* attached at site DE on 2/2/84 when the temperature was 23°C. Unfortunately no observations of the WSBR of sewage fungus communities were made at this temperature. Nevertheless these results suggest that similar biomasses of sewage fungus and algae with a thin heterotrophic film attached might be expected to have broadly similar night-time respiration rates during summer lowflow conditions in the Manawatu River.

5.3.5 ANALYSIS OF COMBINED CHAMBER DATA

The previous section (5.3.4) indicated that under summer low-flow conditions in the Manawatu River similar biomasses of sewage fungus and of algae with thin heterotrophic films attached have broadly similar respiration rates. Thus in this section the data collected in the investigations of algal and sewage fungus dominated communities on natural and artificial substrates were analysed together in an attempt to produce a generally applicable model of the factors affecting benthic respiration rates (BR) from the larger combined data base. Again only the studies in which the conditions were not altered experimentally were used in the statistical analysis. The data collected from the macrophyte *P. crispus* was also excluded from this analysis since its affects on BR and benthic gross photosynthetic oxygen production (BGPR) had been shown to be quite different from those of the epilithic communities (Section 5.3.4).

TABLE 5.8 Comparison of Weight Specific Respiration Rates of Macrophyte, Algal and Sewage Fungus Communities over a Temperature Range of 20.2 to 21.2°C

Community Type Dominant Organism	Macrophyte *P. crispus*	Algae *C. glomerata*	Sewage fungus *S. natans*	Sewage fungus *S. natans*
Site	EF	EF	D	C
Date	12/3/84	12/3/84	8/12/83	12/1/84
Biomass ($AFDW.m^{-2}$)	31	55.3	36.2	54.6
AI	68	100	ND	664
Visual Classification (Appendix B)	0IE	0IE	5ID	6IC
Initial BOD_5 ($g.m^{-3}$)	2.0	1.7	0.7	7.3
Initial DO ($g.m^{-3}$)	7.8	7.0	11.0	5.3
Temperature (°C)	20.2	21.2	20.7	20.2
WSBR ($gO_2.gAFDW^{-1}.d^{-1}$)	0.15	0.26	0.38	0.393

The combined algal and sewage fungus data were also used to investigate the relationships between BGPR, algal biomass (ABM, measured as chlorophyll a) and other factors.

The parsimonious multiple regression equation for BR (Table 5.9) shows that the benthic biomass (BM) and temperature (T) have statistically significant influences on BR and together account for 73% of the variation in BR observed. The individual effects of these factors are shown in Figure 5.6. None of the other factors tested (initial BOD_5, Autotrophic Index (AI), and initial dissolved oxygen (DO)) were significantly correlated with BR (Table 5.10), nor did any have statistically significant effects on the regression model of BR, at the 95% confidence level.

Comparison of the respiration rates of the benthic communities (Fig 5.6) with those of the suspended biomass (Section 5.2) shows that benthic respiration always has the dominant effect on oxygen dynamics in the Manawatu River.

The construction of multiple linear regression models for the factors affecting the benthic gross production rate using the results of the experiments in which the light conditions were not altered experimentally was expected to be complicated due to the anticipated correlation between the predictors light and temperature. These were found to be significantly correlated ($r = 0.59$, $P < 0.05$) in this data set making it impossible to discern their individual effects. However experiments in which the PAR at the depth of the biomass was varied independently, by placing shades over the chambers (Section 3.6.3.6), showed that the incident PAR has a strong influence on BGPR, particularly at low light levels (Figs 5.7-5.9). The results of these experiments suggest that an approximately linear increase in BGPR occurs with increasing PAR up to approximately 200 to 300 $uE.m^{-2}.s^{-1}$. At light levels greater than these the rate of increase in BGPR with increasing light was reduced.

An experiment during mid-autumn, when the temperature and radiation input were low (Fig 5.9), indicated that the BGPR

TABLE 5.9 Parsimonious Regression Equation Describing the Effects of Statistically Significant Predictors (at the 95% level) on Benthic Respiration Rate of Epilithon on Natural and Artificial Substrates

$$BR = -18.9 + 1.41\ T + 0.158\ BM \qquad \text{(Equation 5.5)}$$

Units = $gO_2.m^{-2}.d^{-1}$

r^2 = 73.0% (adjusted for 21 degrees of freedom); standard deviation of data about the regression equation (s) = 3.63

Importance of Predictors

Predictor	Coefficient	St dev of Coefficient	t ratio = Coeff/S.D.	Percentage of Sum of Square reduction accounted for
Constant	-18.84	4.40	-4.30	
BM	1.41	0.24	5.93	37.4%
T	0.158	.028	5.55	37.8%

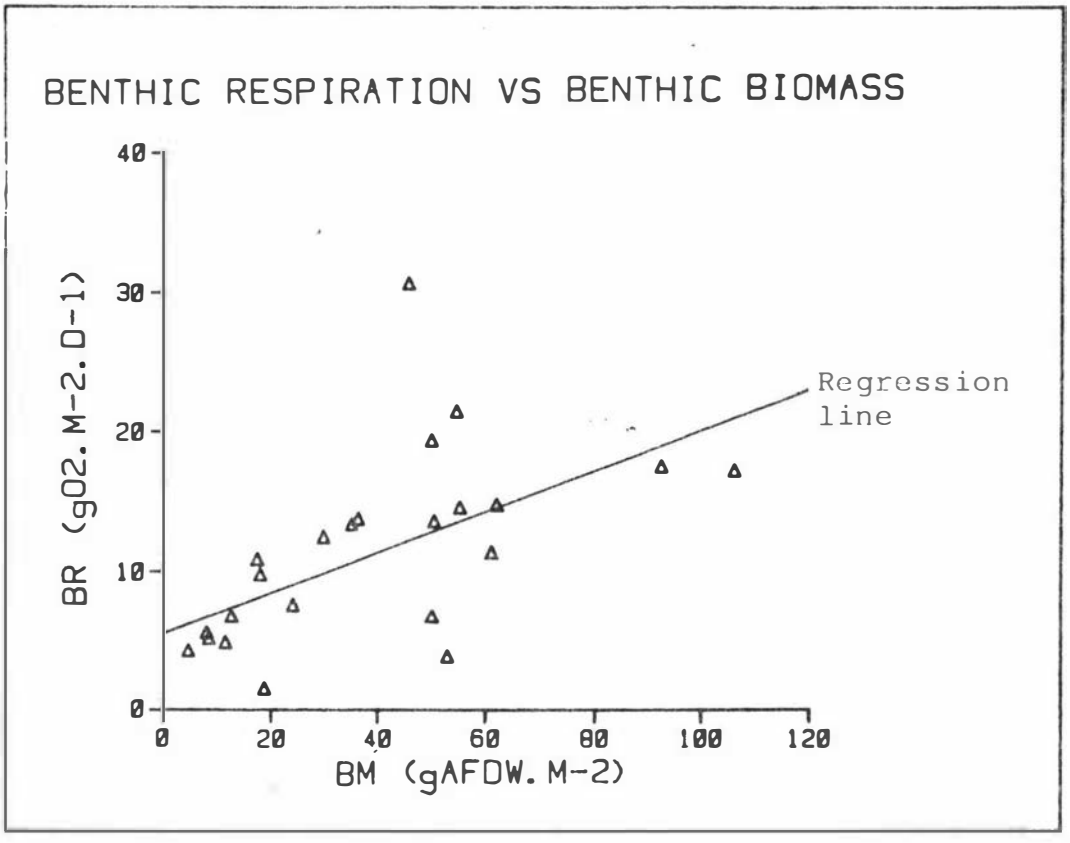
Application Range of Regression Equation

T = 11.0 - 23.2°C

BM = 4.8 - 106 $gAFDW.m^{-2}$

Range of Other Variables During Experiments

Initial BOD_5 = 0.7 - 7.0 $g.m^{-3}$
Initial DO = 5.3 - 12.7 $g.m^{-3}$
AI = 85 - 3637



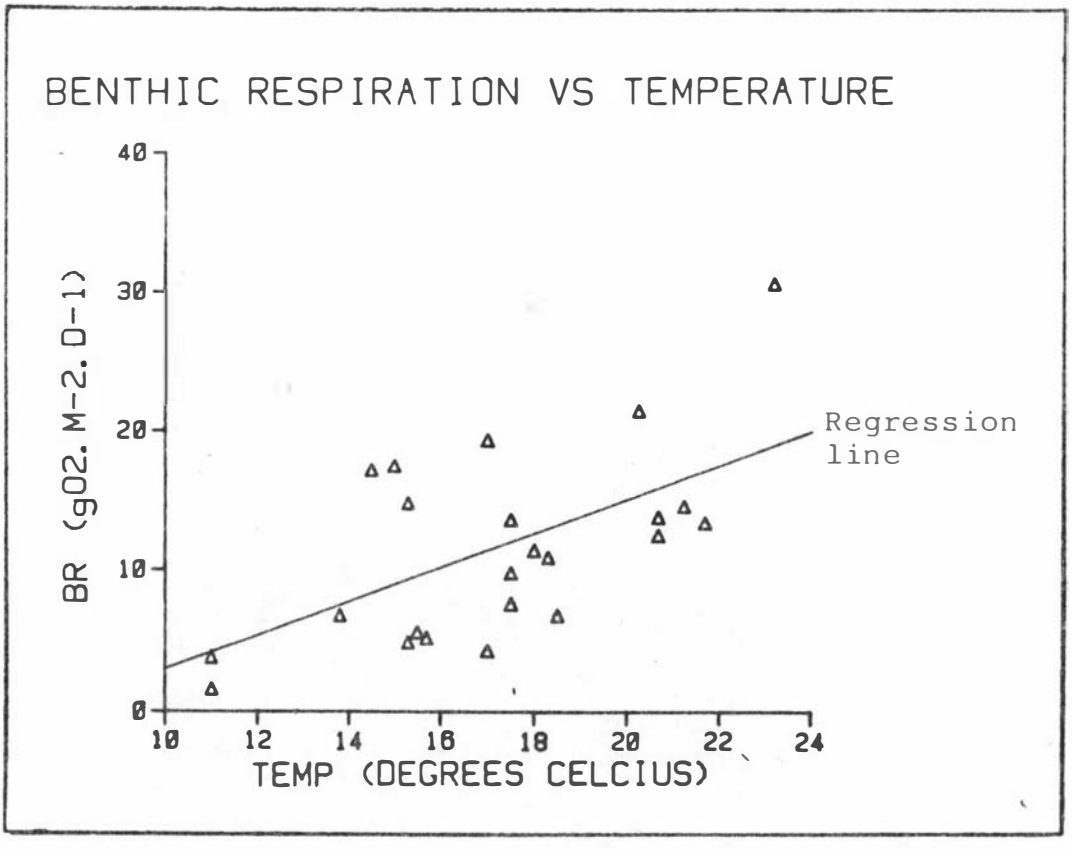


Figure 5.6: Graphs Showing the Relationships Between the Benthic Respiration Rate and Statistically Significant Predictors.

TABLE 5.10 Correlation Coefficient Matrix of Benthic Respiration and Benthic Gross Production Rates and some Environmental Factors; combined natural and artifical substrate data excluding those obtained under experimentally altered conditions.

	BR	T	BM	Initial BOD_5	AI	WSBR	BGPR	Chla.sBGPR	PAR	ABM
T	0.612									
BM	0.568	-0.075								
Initial BOD_5	0.362	0.033	0.278							
AI	-0.285	-0.033	-0.425	0.349						
WSBR	0.054	0.416	-0.605	0.301	0.751					
BGPR	0.732	0.730	0.402	0.145	-0.436	0.160				
chla.sBGPR	-0.107	0.145	-0.522	-0.139	0.934	0.869	-0.220			
PAR	0.266	0.586	-0.247	0.036	0.404	0.667	0.410	0.675		
ABM	0.358	0.051	0.794	0.057	-0.644	-0.571	0.681	-0.608	-0.167	
Initial DO	-0.367	-0.317	-0.250	-0.335	-0.042	0.070	-0.316	-0.084	-0.115	-0.222

Key:

BR = Benthic Respiration Rate ($gO_2.m^{-2}.d^{-1}$)
T = Temperature (°C)
BM = Benthic biomass ($gAFDW.m^{-2}$)
Initial BOD_5 = BOD_5 at the beginning of the experiment ($g.m^{-3}$)
AI = Autotrophic Index
WSBR = Weight specific benthic respiration rate ($gO_2.gAFDW^{-1}.d^{-1}$)
BGPR = Benthic gross photosynthetic oxygen production rate ($gO_2.m^{-2}.d^{-1}$) (calculated for 12 h light at level during measurement. d^{-1})
chla.sBGPR = BGPR/chlorophyll a concentration ($gO_2.gchla^{-1}.d^{-1}$)
PAR = Photosynthetically available radiation at the depth of the biomass
ABM = Algal biomass (mg chlorophyll a.m-2)
Initial DO = Initial dissolved oxygen concentration ($g.m-3$)

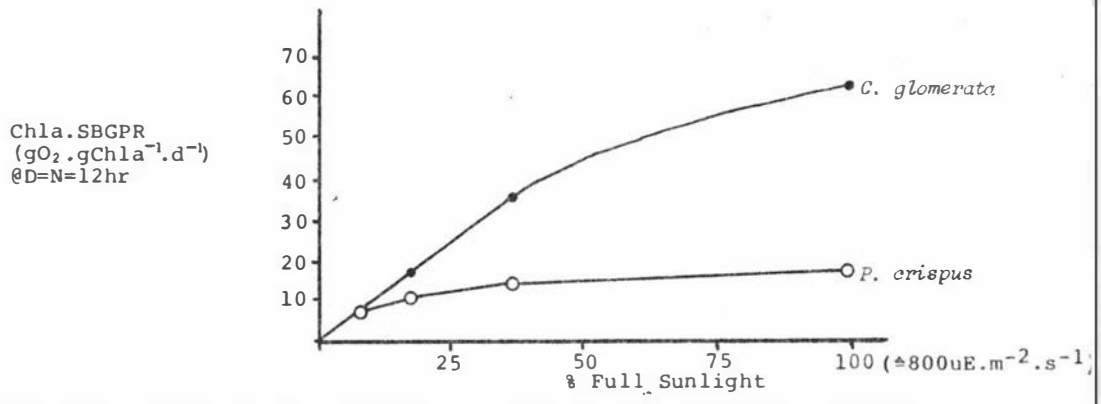


Fig 5.7: Chlorophylla Specific BGPR versus % Full Sunlight, Site EF, 12/3/84 (T=20-21.5°C, Total Radiation during previous 4 days ($x \pm s$) = 15.9±1.6 mJ.m^{-2}.d^{-1}).
Biomass and water quality given in Table 5.11.

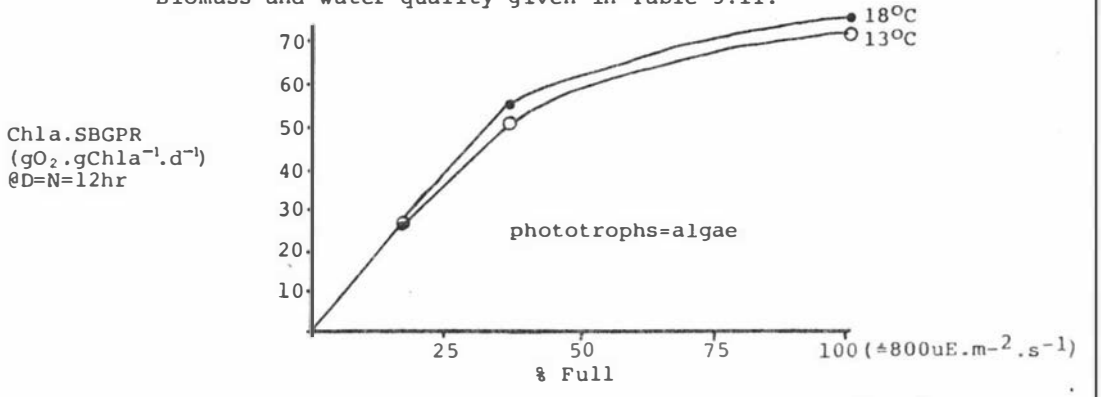


Fig 5.8: Chlorophylla Specific BGPR versus % Full Sunlight, Site C, 22/2/84 (Total radiation during previous four days ($x \pm s$) = 20.6±3.4 mJ.m^{-2}.d^{-1}).
Biomass = 219 mg.Chla.m^{-2}.

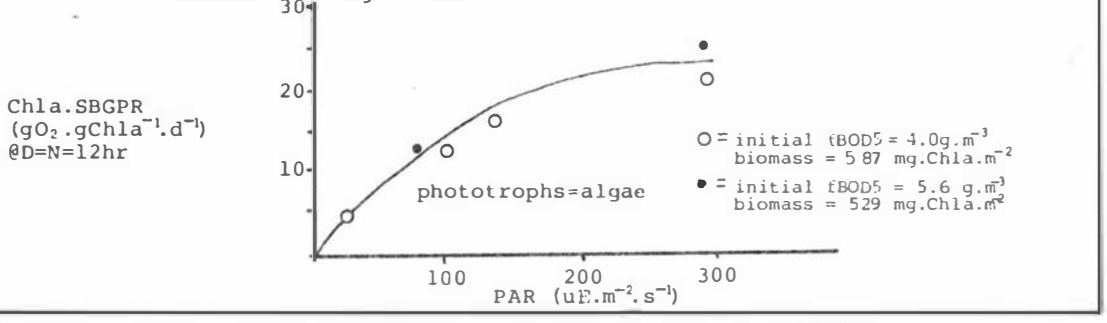


Fig 5.9: Chlorophylla Specific BGPR versus PAR, Site C, 26/4/84 (T=15°C, Total radiation during previous four days ($x \pm s$) = 11.5±1.6 mJ.m^{-2}.d^{-1}).
Biomass = 529 and 587 mg.Chla.m^{-2}.

approached a saturation level, at approximately 300 $uE.m^{-2}.s^{-1}$ PAR, beyond which further increases in light did not increase the BGPR. At the higher levels of temperature and radiation input during summer (Figs 5.7 and 5.8) the BGPR of the communities, other than *P. crispus* (see Section 5.3.4), appeared to increase with increasing light to full sunlight levels. However, due to the small number of light levels tested, it was not possible to determine whether light saturation of BGPR was reached in any of these studies. Nevertheless the results suggest that BGPR light saturation level of the algae is affected by the temperature and light conditions to which the community has been recently exposed. Both these factors have previously been shown to have independent effects on the BGPR saturation level in laboratory stream studies (McIntire and Phinney, 1965; McIntire, 1975).

When the results of the shading experiments (Figs 5.7-5.9) were also included in BGPR multiple regression equation data set the correlation between light and temperature was no longer significant (at $P = 0.05$) and their individual effects could be discerned (Table 5.11).

The results of the shading experiment (Figs 5.7-5.9) indicated that a square root, rather than a linear function, would provide a better model of the effects of light on BGPR and the plate growth studies results (Fig 5.4) indicated that BGPR is related to the natural logarithm of the benthic algal biomass concentration (ABM). Including these functions in the regression model of the effects of ABM, light (as PAR) and temperature on the BGPR improved the relationship between the observed and predicted values and yielded the parsimonious equation in Table 5.11. This shows that light and ABM account for most of the variation in BGPR observed with temperature variations having a minor influence (not significant at the 95% level).

Although the data set which generated this model included a number of estimated light values (Appendix E), the model should be useful for predicting the BGPR of epilithon of

TABLE 5.11 Parsimonious Regression Equation Describing the Effect of Light, Temperature and Algal Biomass on the Benthic Gross Photosynthetic Oxygen Production Rate (BGPR)

$$BGPR = -2.71 + 0.063 \sqrt{PAR} + 0.022\ T + 0.376\ \ln ABM \quad \text{(Equation 5.6)}$$

Units = $gO_2.m^{-2}.h^{-1}$

r^2 = 78.2% (adjusted for 24 degrees of freedom); standard deviation of data about the equation (s) = 3.45

Importance of Predictors

Predictor	Coefficient	St dev of Coefficient	t ratio = Coeff/S.D.	Percentage of Sum of Square reduction accounted for
Constant	- 2.71	0.46	-5.92	
$\sqrt{PAR}$	0.063	0.008	7.76	39.9%
T	0.022	0.022	1.02	9.3%
ln ABM	0.376	0.064	5.90	31.6%

Application Range of Regression Equation

T = 11.0 - 23.2°C

PAR = 0 - 900 $\mu E.m^{-2}.s^{-1}$ (at the bed)

ABM = 9.5 - 690 mg chlorophyll $a.m^{-2}$

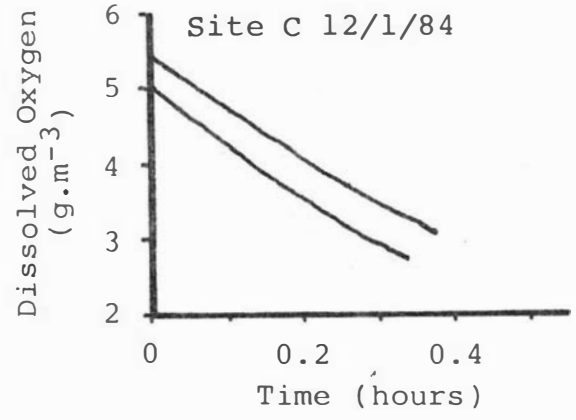
streams where the values of the predictors lie within the fairly broad bounds of the model data set (Table 5.11).

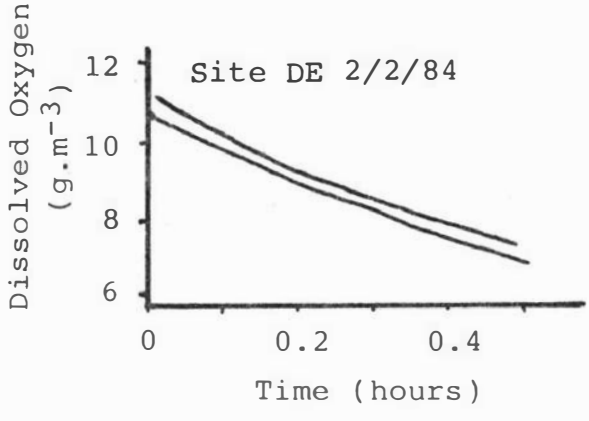
The chlorophyll a specific BGPR values of the unshaded bed biomass samples of 1.3 to 10.5 $gO_2.gchla^{-1}\ h^{-1}$ (Appendix E) were greater than those observed in a similar *in situ* chamber study of algae in Polish rivers (0.6 to 1.8 $gO_2.g\ chla^{-1}.h^{-1}$) (Bombowna, 1972) but generally similar to those observed in an English chalk stream (2.7 to 5.65 $gO_2.gchla^{-1}.h^{-1}$) (Marker, 1976). However the values recorded for thin epilithon growths attached to the artificial substrates (35.4 to 4.2 $gO_2.gchla^{-1}.h^{-1}$ for biomasses of 9.5 to 156 $mg.chla^{-1}.m^{-2}$ respectively (Appendix C)) were often considerably higher than the previous observations. The high values of the artificial substrate growth may be partly due to the flat upper growth surfaces of the substrates providing unshaded light conditions to a greater proportion of the total algal biomass than would occur for similar populations on a rough cobble bed.

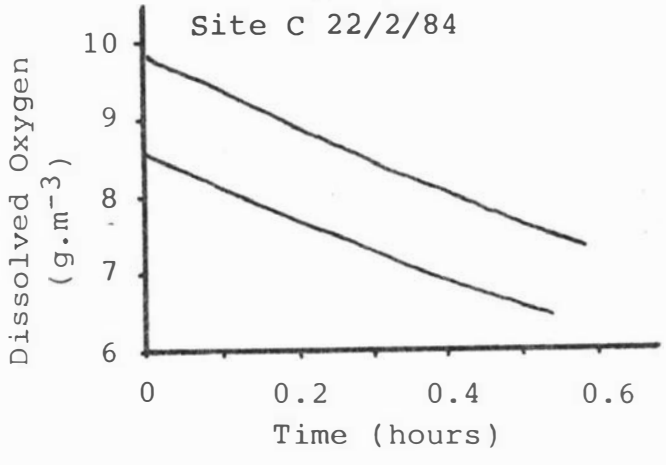
5.3.6 RESPIRATION RATE VARIATIONS DURING IN SITU CHAMBER EXPERIMENTS

The dissolved oxygen versus time plots obtained during the *in situ* chamber experiments were linear for intervals during which the change in dissolved oxygen concentration was up to approximately 1.5 $g.m^{-3}$. However, when the respiration rate was measured (in the dark) over a sufficient interval for the dissolved oxygen to decline by more than 1.5 $g.m^{-3}$, these plots were observed to be curved (Fig 5.10). Thus the respiration rates measured from the plots decreased with time. The conditions during the experiments in which respiration was measured for sufficient time for this to be observed are presented in Table 5.12. The change in respiration rate during the interval was calculated from the tangents to the dissolved oxygen versus time plots.

Changes in dissolved oxygen or dissolved organic material concentrations ($fBOD_5$) could cause the observed reductions in







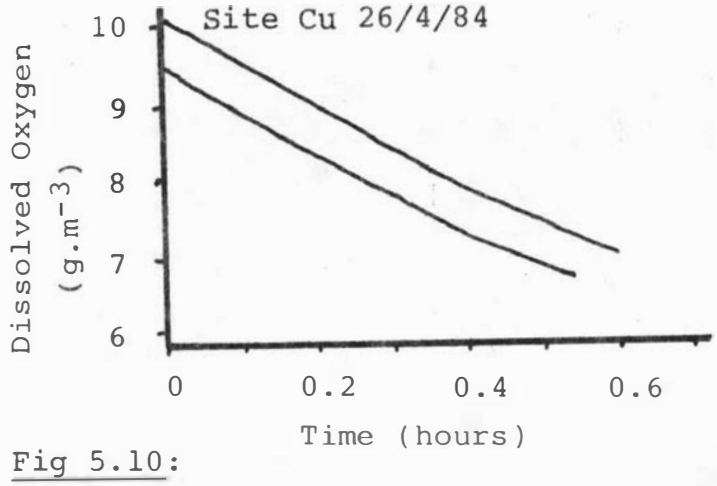


Fig 5.10:

Dissolved Oxygen versus Time plots during prolonged *In situ* Chamber studies.

respiration rates (Section 2.3.5). Since both declined with time during most of these experiments it is difficult to distinguish between their individual effects on the respiration rate.

At site C on 12/1/84 the respiration rate was reduced by 16 to 18% of the initial rate when the dissolved oxygen dropped by approximately 2.5 $g.m^{-3}$ from 5.0 and 5.4 $g.m^{-3}$ respectively. Increasing the initial $fBOD_5$ concentration, by addition of MCDC wastewater, had no appreciable effect on this decline. On the other occasions, when the dissolved oxygen concentrations were higher, a decrease in respiration rate of approximately 22 to 32% occurred for a 2.5 $g.m^{-3}$ drop in dissolved oxygen concentration (Table 5.12). However these percent reductions in the respiration rates are less than those observed over 0.4 hours in *in situ* chamber studies of a senescent sewage fungus community in the Waitoa River (Hickey, 1983) where a 46% decrease occurred for a 2.5 $g.m^{-3}$ drop in dissolved oxygen.

The activity of the algal dominated community at Site DE on 2/2/84 did not appreciably alter the $fBOD_5$ concentration of the water in the chamber with the lower initial $fBOD_5$ concentration (Table 5.12). Thus, in this case, dissolved oxygen limitation appears to be responsible for the reductions in respiration rate observed (22% and 44% of the initial rate for 2.5 and 4.5 $g.m^{-3}$ decreases in dissolved oxygen respectively). Increasing the initial $fBOD_5$ by 7.4 $g.m^{-3}$, by addition of MCDC wastewater, did not alter the reduction in respiration rate (23% of the initial rate) after a 2.5 $g.m^{-3}$ drop in dissolved oxygen but a smaller reduction in respiration rate (32%) was observed for a 4.3 $g.m^{-3}$ drop in dissolved oxygen.

Despite this latter result, these data support the hypothesis that the dissolved oxygen concentration of the overlying water can limit the respiration of algal dominated communities. However the variations in respiration rate of sewage fungus communities during the prolonged chamber experiments do not allow the assessment of the effects of dissolved oxygen since these could not be distinguished from the effects of reduction in the $fBOD_5$ concentration during these experiments.

TABLE 5.12 Conditions during prolonged chamber studies

Site (Fig 1.2)	C	C	DE	DE	C	C	Cu	Cu
Date	12/01/84	12/01/84	02/02/84	02/02/84	22/02/84	22/02/84	26/04/84	26/04/84
Biomass ($gAFDW.m^{-2}$)	54.6	49.8	48.7	45.8	70	61.1	92.6	104.4
AI	664	664	ND	ND	323	282	158	197
Macroscopic Community Description	6IE	6IE	2IF	2IF	5/6IE	5/6IE	6IE	6IE
Temperature (°C)	20	20.2	24	24	13.5	18	15	15.2
Total duration of experiment (hr)	1.17	1.38	1.03	1.2	1.6	1.53	1.58	1.72
Initial $fBOD_5$ ($g.m^{-3}$)	$5.4 \pm .5$	$11.2 \pm .4$	$10.5 \pm .1$	3.1 ± 0	4.3 ± 1.1	$4.3 \pm .1$	$4.0 \pm .3$	$5.6 \pm .2$
$fBOD_5$ end experiment ($g.m^{-3}$)	$2.1 \pm .4$	$3.0 \pm .2$	$7.5 \pm .1$	$3.4 \pm .2$	3.9 ± 1.0	3.6 ± 1.2	$2.0 \pm .4$	$1.7 \pm .4$
Initial DO ($g.m^{-3}$)	5.4	5.0	10.5	11	9.6	8.7	9.3	10.3
DO end dark phase of experiment ($gO_2.m^{-3}$)	3.0	2.5	6.2	6.5	7.6	6.3	6.6	6.6
Change in DO ($g.m^{-3}$)	2.4	2.5	4.3	4.5	2.0	2.4	2.7	3.7
% reduction in respiration rate during dark phase	18	16	32	44	28	32	28	26
Treatments	None	$fBOD_5$	$fBOD_5$	None	T	None	None	$fBOD_5$

5.3.7 CONCLUSIONS

In situ chamber experiments have shown that benthic communities may have large effects on stream oxygen dynamics and that suspended biomass has a comparatively minor influence. The effects of benthic communities on benthic respiration (BR) and benthic gross photosynthetic oxygen production (BGPR) were shown to be complex being influenced by a number of factors. However 73% of the variation in BR observed in the experiments could be accounted for by variations in the benthic biomass (as AFDW) and temperature.

A reduction in the BR of algal and sewage fungus communities was observed during prolonged chamber experiments, when BOD_5 and DO declined, and addition of BOD_5 did increase the BR of heterotrophic dominated communities at low background levels. However neither the initial DO nor the initial BOD_5 concentration had a statistically significant effect (at the 95% level) on the regression model developed for BR.

Comparison of the WSBR values of different types of benthic community, at similar biomass levels during summer low flow conditions, indicated that the values for sewage fungus exceeded those of *C. glomerata* dominated algal communities which in turn were greater than those of the macrophyte *P. crispus*. However the WSBR values of algal dominated communities with thin heterotrophic films attached were similar to those expected for sewage fungus communities under comparable conditions during summer low flows.

The BR of algal dominated communities was more strongly influenced by temperature than that of the sewage fungus dominated communities. This probably results from high temperatures being due to high radiation input which enhances the metabolic activity of the algae but not the heterotrophs.

The multiple regression equations describing the BR of sewage fungus communities (Equation 5.1) and epilithon generally

(Equation 5.5) should provide useful models of the factors affecting BR of shallow streams with gravel beds under similar environmental conditions to those encountered in this study.

The BGPR values of the benthic communities were influenced by the algal biomass, light and, to a lesser extent, the temperature. Using these parameters, 78.2% of the variation in the BGPR data could be accounted for in the regression model (Equation 5.6). This model should be useful in water management for predicting the BGPR of algal-dominated streams where the conditions fall within the relatively broad range covered in the *in situ* chamber studies.

5.4 WHOLE RIVER OXYGEN DYNAMICS STUDIES

5.4.1 INTRODUCTION

The effects of benthic communities and environmental factors on river dissolved oxygen levels were also studied by analysis of continuous in-river recordings of dissolved oxygen and temperature over night or full day periods.

During the 1982/83 season two, two-station studies were undertaken downstream of the waste discharges. During the 1983/84 season two-station studies were carried out downstream of the waste discharge zone at approximately weekly intervals from the beginning of the stable flows until one week after a major fish kill occurred. Studies were also undertaken within the waste discharge zone and above the waste discharges.

The results obtained from below the waste discharge zone were used to test the linear regression models of benthic respiration rate generated from the *in situ* chamber studies (Section 5.3)

A computer model, designed to simulate conditions in the Manawatu River during summer low-flows, was used to investigate the effects on night-time oxygen depletion of variations in the respiration rate, the reaeration rate, the temperature and the dissolved oxygen concentration at sunset.

5.4.2 OXYGEN DYNAMICS ABOVE THE WASTE DISCHARGE ZONE

The results of five, single station, dissolved oxygen and temperature studies undertaken during 1984 are presented in Table 5.13. These respiration and gross photosynthesis rates (Section 3.6.4) are low compared with those observed under similar temperature and flow conditions during the previous two summers when respiration and gross photosynthetic rates of approximately 5 to 17 $gO_2.m^{-2}.d^{-1}$ and 5 to 13 $g.m^{-2}.d^{-1}$ respectively were recorded (Freeman, 1983). The lower values during the 1983/84 season apparently result from the lower algal biomass present (Section 7.2.4).

During January and February 1984 phototrophic growth at site A consisted of a light diatom film on the stones with a chlorophyll a concentration of approximately 10 $mg.m^{-2}$. A total benthic biomass (Section 3.2.2.1) of 28 ± 12 $gAFDW.m^{-2}$ was measured on 16/1/84 (Section 7.2.4) but this was largely due to settled detritus and silt. By contrast, at the beginning of February 1983, when the maximum respiration rates were observed, growths of *Cladophora glomerata* covered most of the riverbed. Algal biomasses of around 40-45 g dry weight.m^{-2} were recorded (Freeman, 1983) and the total bed biomass was approximately 50 $gAFDW.m^{-2}$.

5.4.3 OXYGEN DYNAMICS WITHIN THE WASTE MIXING ZONES

The respiration rates recorded for reaches within the waste mixing zone (Table 5.14) were approximately four to ten times greater than those observed above the discharge zone at Site A (Table 5.13, Section 5.4.2). Both the phototroph-dominated communities, between the PNCC discharge and Site B, and the sewage fungus communities, between the MCDC discharge and sites CuC and D, had high respiration rates. These resulted in rapid dissolved oxygen depletion at night (Fig 5.11).

The P/R value decreased as the heterotrophs became more dominant in the benthic communities (Table 5.14). The negative

TABLE 5.13 Respiration and Gross Oxygen Production Rates Upstream of Manawatu River Discharges, 1984

Date	16-17/1	24-25/1	31/1-1/2	8-9/2	29/2-1/3
Respiration ($gO_2.m^{-2}.d^{-1}$)	3.9	5.3	4.2	4.7	1.9
Gross Oxygen Production ($gO_2.m^{-2}.d^{-1}$)	ND	4.0	ND	4.0	2.3
P/R	ND	0.75	ND	0.84	1.2
Physical Conditions:					
River Flow ($m^3.s^{-1}$)	16.3	14.8	14.2	19.4	16.8
Average River Temperature (°C)	17	20.3	18.4	18.5	21
Total Surface Radiation ($MJ.m^{-2}.d^{-1}$)*	21.4	26.8	30.2	28.0	19.9
Chemical Conditions:					
BOD_5 ($g.m^{-3}$) ($x \pm s$) **	0.7 ± 0.1	0.4 ± 0.3	0.6 ± 0.1	0.6 ± 0.1	1.4 ± 0.5
TDN ($mg.m^{-3}$) ($x \pm s$) **	281 ± 14	291 ± 17	229 ± 1	ND	279 ± 15
DRP ($mg.m^{-3}$) ($x \pm s$) **	16 ± 10	2.7 ± 0.3	0.5 ± 0.2	12 ± 10	4.2 ± 0.9
Biomass Conditions:					
Heterotroph Abundance ***	O	O	O	O	O
Phototroph Abundance ***	C	C	C	C	B-C

* Measured by MAF at Tiritea 5 km east of study site
** Results of grab samples
*** Abundance scales given in Appendix B.

TABLE 5.14 Results of Whole River Oxygen Dynamics Studies Within the Manawatu River Waste Discharge Zone, 1984

River Reach (Fig 1.2)	PNCC-B	PNCC-B	MCDC-CuC	MCDC-D	MCDC-D
Date	16-17/1	29/2-1/3	29/2-1/3	15-16/1	16-17/1
Respiration ($gO_2.m^{-3}.d^{-1}$)	18.8	23.9	35.2	22.9	24.5
Respiration ($gO_2.m^{-2}.d^{-1}$)	20.7	26.1	39.8	19.3	20.6
Gross Oxygen Production ($gO_2.m^{-2}.d^{-1}$)	ND	15.1	-4.3	7.75	ND
P/R	ND	0.58	-0.14	0.39	ND
BOD_5 Removal Rate ($gBOD_5.m^{-2}.d^{-1}$)	ND	6.4	36.9	ND	ND
Minimum Dissolved Oxygen ($g.m^{-3}$)	6.7	5.1	3.4	5.0	5.15
River Conditions:					
Average Flow ($m^3.s^{-1}$)	18	16.8	16.8	16.9	18
Average Temperature (°C)	17.2	21	20.7	17.5	17.0
Total Surface Radiation ($MJ.m^{-2}.d^{-1}$)*	21.4	21.1	21.1	20.9	21.4
Average Initial BOD_5 ($g.m^{-3}$)**	2.4	3.7	6.8	4.1	3.5 + 3.7 ****
Biomass Conditions:					
(a) AFDW ($g.m^{-2}$)	83	100*****	150*****	120	120
(b) Average Heterotroph Abundance ***	1	1	6	5.5	5.5
(c) Average Phototroph Abundance ***	F	F	D	D-E	D-E

* Measured at Tiritea, 5 to 8 km east of river reaches

** Calculated from discharge data and measured values above outfalls

*** Abundance scales given in Appendix B.

**** The BCWS discharge resulted in an average calculated increase in the BOD5 of 3.7 g.m-3 just below the midpoint of the reach (Fig 1.2)

***** Values estimated from visual abundance observations

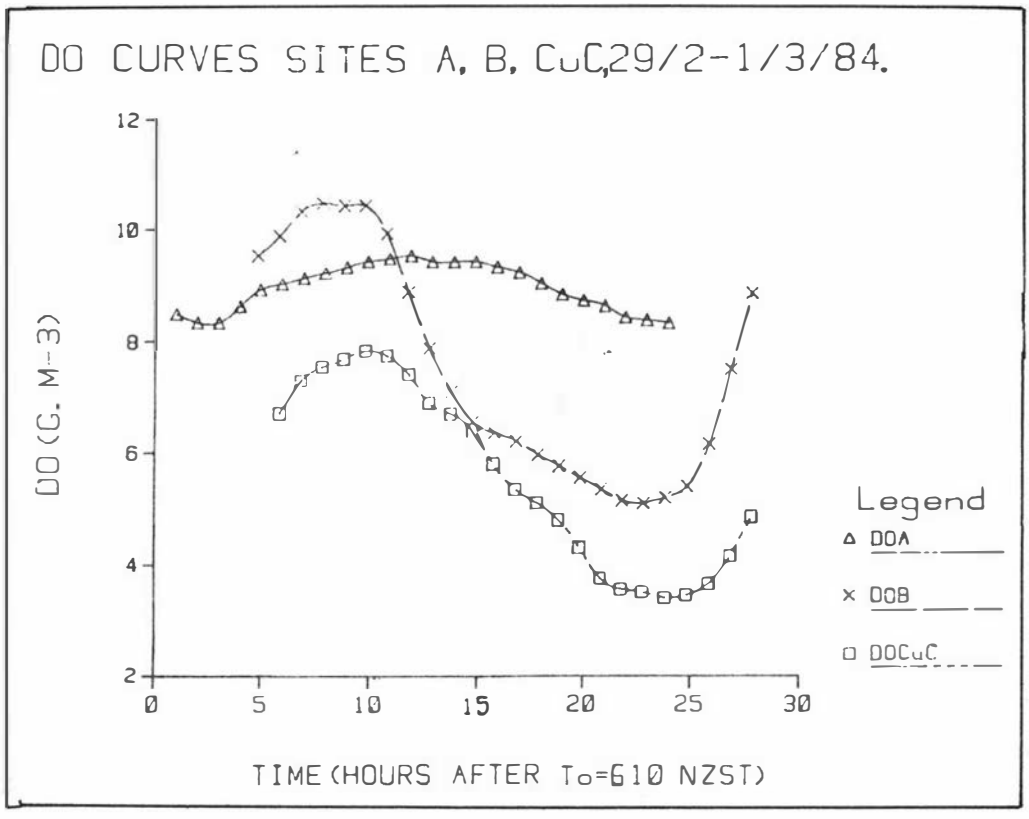


Figure 5.11: Diurnal Variations in Dissolved Oxygen Concentration at Sites A, B and CuC in the Manawatu River (fig.1.2), 29/2-1/3/84 (temperature= 19.8-22.2°C).

gross photosynthetic oxygen production (GPR) and P/R values recorded for the reach MCDC to CuC indicated that during the daylight hours heterotrophic respiration increased to an extent that offset any simultaneous photosynthetic oxygen production. The average dissolved oxygen removal over the reach MCDC-CuC for the whole day on 29/2-1/3/84 was 41.9 $gO_2.m^{-2}.d^{-1}$ compared to 39.8 $gO_2.m^{-2}.d^{-1}$ calculated from the night hours (Table 5.14). The lower average night-time oxygen removal rate results from lower respiration during the first five hours after sunset (Table 5.15). This apparently resulted from the lower BOD_5 loading from MCDC during this interval when the average BOD_5 addition was sufficient to increase the in-river BOD_5 by 1.7 ± 0.6 $g.m^{-3}$ ($\bar{x} \pm s$). This value is low compared to that during the rest of the day on 29/2-1/3/84 (Table 5.15) and the average value of 3.4 ± 1.0 $gBOD_5.m^{-3}$ ($\bar{x} \pm s$) calculated for the previous five days. Addition of sufficient MCDC effluent to raise the average river BOD_5 increase to 3.8 ± 2.1 $g.m^{-3}$ ($\bar{x} \pm s$) (Table 5.15) resulted in a 56% increase in the reach respiration during the last 6 hours of darkness (Table 5.15). However the reach respiration rate during the last six hours of darkness was relatively constant despite quite large variations in the average BOD_5 increase due to the MCDC addition over this period as shown by the high standard deviation of the mean BOD_5 increase (Table 5.15). Thus the addition of BOD_5 , as MCDC effluent, had a large influence on the respiration of sewage fungus communities previously exposed to lower than usual BOD_5 levels. However short-term variations (in terms of hours) have little effect on the respiration of communities which have recently been exposed to relatively high BOD_5 conditions (Table 5.15, fig 5.11). These results support the findings of *in situ* chamber studies on the effects of waste addition on sewage fungus respiration rate (Section 5.3.3).

The reach MCDC-CuC, which contained heavy sewage fungus growth, was highly heterotrophic and the dissolved oxygen content of water passing through it was reduced throughout the night and day whereas the dissolved oxygen concentration of the water passing through the phototroph dominated reach PNCC-B increased during the daylight hours. (Table 5.14, Fig 5.11). The

TABLE 5.15 Variations in MCDC BOD_5 Addition and River Respiration Rate for Reach MCDC-CuC, 29/2-1/3/84

Time (hours, NZST)	Calculated **** Background BOD_5 above MCDC outfall ($g.m^{-3}$)	Calculated increase in river BOD_5 due to MCDC discharge ($g.m^{-3}$) ($x \pm s$)	Reach MCDC-CuC Respiration Rate ($gO_2.m^{-2}.d^{-1}$; $x \pm s$)
1930 - 2430	3.0	1.7 ± 0.6 *	30.4 ± 4.3
2430 - 530	3.4	3.8 ± 2.3 *	47.6 ± 2.9
1930 - 530	3.2	3.0 ± 2.1 *	39.8 ± 9.6
1030 - 930	2.9	3.1 **	41.9 ± 7.8 ***

* Calculated from results of flow related hourly composite samples analyses (Meredith, 1984)

** Calculated from results of 23 hour flow related composite sample analysis (Meredith, 1984)

*** Nett respiration during 24 hours; daytime photosynthesis not allowed for.

**** Calculated from hourly waste discharge data and river flow data using the first order decay equation (4.1) (Section 4.4.1) given that
(i) background BOD_5 above PNCC = $1.38\ g.m^{-3}$ (measured)
(ii) k_1 (PNCC-MCDC) = $0.07.h^{-1}$ (measured)

similarity between the respiration rate and the BOD_5 removal rate (Table 5.14) suggests that within the reach most of the respiration is due to BOD_5 oxidation. By contrast, the oxidation of the BOD_5 removed in the reach PNCC to B on 29/2-1/3/84 only accounted for 24% of the total respiration. This indicates that phototroph respiration is much more important in the latter reach. Similar observations were made over reaches below the waste mixing zone (i.e., below site D, Fig 1.2) (Section 5.4.4).

The minimum dissolved oxygen concentration of 5 $g.m^{-3}$, required by the river classification (Section 2.1), was barely maintained at site D on the morning of 16/1/84 (Table 5.14), prior to the recommencement of the BCWS discharge. However this requirement was breached on 1/3/84 at site CuC (0.5 km upstream of the BCWS outfall) (Fig 5.11) and during the early morning (0630 to 0700 hours NZST) on 3/2/84, when the dissolved oxygen concentrations at sites B, CuC and C were 4.7, 2.8 and 1.9 $g.m^{-3}$ respectively at the river temperature of 20°C (Gilliland, 1984). Numerous dead fish were observed downstream of site C on 2/2/84 (Section 5.4.4). Dissolved oxygen concentrations below the minimum statutory requirement were also measured at sites below the effluent mixing zone on several other occasions during the 1983/84 summer (Fig. 5.12) and it is likely that the minimum requirement was breached on other unmeasured occasions.

The effect of the benthic communities within and below the waste discharges on dissolved oxygen removal is discussed further in Section 5.4.5.

5.4.4 WHOLE RIVER STUDIES OF DISSOLVED OXYGEN DYNAMICS BELOW THE WASTE DISCHARGE ZONE

The results of the two station studies over the reaches between sites D and E and sites Dd and E (Fig 1.2) are presented in Table 5.16. These show that similar respiration rates occurred to those observed within reaches receiving waste discharges.

TABLE 5.16 Results of Whole River Oxygen Dynamics Studies Between Sites D and E

River Reach (Fig 1.2)	D-E**	D-E	Dd-E	D-E
Date	22-24/2/83	6-7/12/83	15-16/12/83	6-7/1/84
Respiration ($gO_2.m^{-3}.d^{-1}$)	31.5	33.6	23.4	32.0
Respiration ($gO_2.m^{-2}.d^{-1}$)	32.8	33.1	21.7	27.8
Gross Oxygen Production Rate ($gO_2.m^{-2}.d^{-1}$)	13.2	34.2	ND	28.4
P/R	0.40	1.03	ND	1.02
River Conditions:				
Average Flow ($m^3.s^{-1}$)	20	19	30	16.8
Average Temperature (°C)	19	20.3	18.4	18.2
Total Surface Radiation ($MJ.m^{-2}.d^{-1}$)	24.5	31.1	23.8	23.0
Average Initial BOD_5 ($g.m^{-3}$)	8.9	8.15	6.5	5.9
Average Initial TDN ($g.m^{-3}$)	1.35	ND	0.969	ND
Average Initial TDP ($g.m^{-3}$)	0.211	ND	0.085	ND
Benthic Biomass:				
(a) AFDW ($g.m^{-2}$)	40-80	51-83	44-80	120
(b) Heterotroph Abundance*	4	5	5	3
(c) Phototroph Abundance*	C-D	D	D	F

* Scales in Appendix B

** Data collected with assistance of Mr C.W.Hickey

Different organic loadings prior to the measurements resulted in large differences in the established bed biomass within the reach on 22-24/2/83, to that on 6-7/1/84. Prior to the 22-24/2/83 measurements, the MCDC and BCWS discharges increased the river BOD_5 by approximately twice the amounts, of 2.3 to 2.7 $g.m^{-3}$ and 1.9 to 2.2 $g.m^{-3}$ respectively, permitted by their new water rights (Section 2.2.4 and 2.2.5) (Gilliland, 1984), whereas the BCWS discharge had been halted for three weeks prior to 6-7/1/84, due to an industrial dispute, and during this period the MCDC BOD_5 loadings exceeded those permitted by 17% on average (Gilliland, 1984).

On the former occasion the riverbed community consisted of a moderate sewage fungus biomass overgrowing an algal film whereas on the latter occasion *C. glomerata* filaments, up to one metre in length, dominated and sewage fungus was only present as isolated fronds (Table 5.16).

Despite the total biomass (as AFDW) on 6-7/1/84 being approximately twice that on 22-24/2/83, similar respiration rates were recorded. However the phototroph dominated community was much more photosynthetically active and, for similar temperature and light conditions to those on 22-24/2/83, had a much higher P/R value (Table 5.16).

On 6/12/83 *S. natans* dominated sewage fungus growths covered most of the river bed. However short (5 to 10 cm long) filamentous algal growths also occurred and the sewage fungus contained a large number of diatoms giving a dark tan colouration to the growths. Under the river conditions of relatively high values of BOD_5, light and temperature, this community had high GPR and RR values similar to those of the phototrophic dominated community on 6-7/1/84.

A major fish kill occurred in the Manawatu River downstream from site C on 1/2/84. The biomass, respiration and dissolved oxygen data collected at river sites below site D during the interval from the beginning of relatively stable flow conditions until ten days after the first dead fish were found

have been summarised (Fig 5.12). These data are also presented in Appendix F along with the temperature, light, BOD_5 and nutrient data collected concurrently.

The data show that the bed biomass and community respiration rate increase rapidly during the periods of declining or relatively stable flows (Fig 5.12). Early in the interval sewage fungus growths, dominated by *S. natans* and *Flavobacterium* spp., were common or covered most of the bed in the reaches from sites D to E and D to EF (Fig 1.2). By mid-January macroscopic bacterial growths were restricted to reach from the MCDC outfall to site D, although phototrophic growths also occurred within this zone (Sections 7.2.2 and 7.2.4). Below the mixing zone (i.e., below site D) a heavy phototrophic biomass, dominated by *C. glomerata* and, to a lesser extent, *P. crispus*, occurred.

The activity of the communities within and above the reach resulted in large diurnal dissolved oxygen fluctuations, which confirm that phototrophic activity has important effects on oxygen dynamics both within and below the discharge zone. The minimum dissolved oxygen concentration of 5 $g.m^{-3}$, required by the river's D classification, was not met in the reaches below the mixing zone during the night on several occasions (Fig 5.12). By contrast, at site A above the discharges (Fig 1.2), the average minimum night-time dissolved oxygen concentration recorded during January/February 1984 was 8.2 $g.m^{-3}$. On 31/1/84-1/2/84 the dissolved oxygen at site Dd was less than 5 $g.m^{-3}$ from midnight until 920 AM with the minimum concentration of 2.1 $g.m^{-3}$ recorded at 6 AM (Fig 5.13). Eels (*Anguilla* sp.) and trout (*Salmo trutta*) recovered from fish traps (Fyke nets) at site Dd by commercial fishermen on the morning on 1/2/84 were found to be dead and many more dead fish, including bullies (*Gobiomorphus cotidianus*) and blackflounder (*Rhombosolea retiaria*), were found on 2/2/84.

On 9-10/2/84 the diurnal variations in dissolved oxygen at sites Dd and EF were less than those observed on 31/1/84 to 1/2/84 (Fig 5.12). This implies reduced respiration and gross photosynthetic rates both above site Dd and between sites Dd

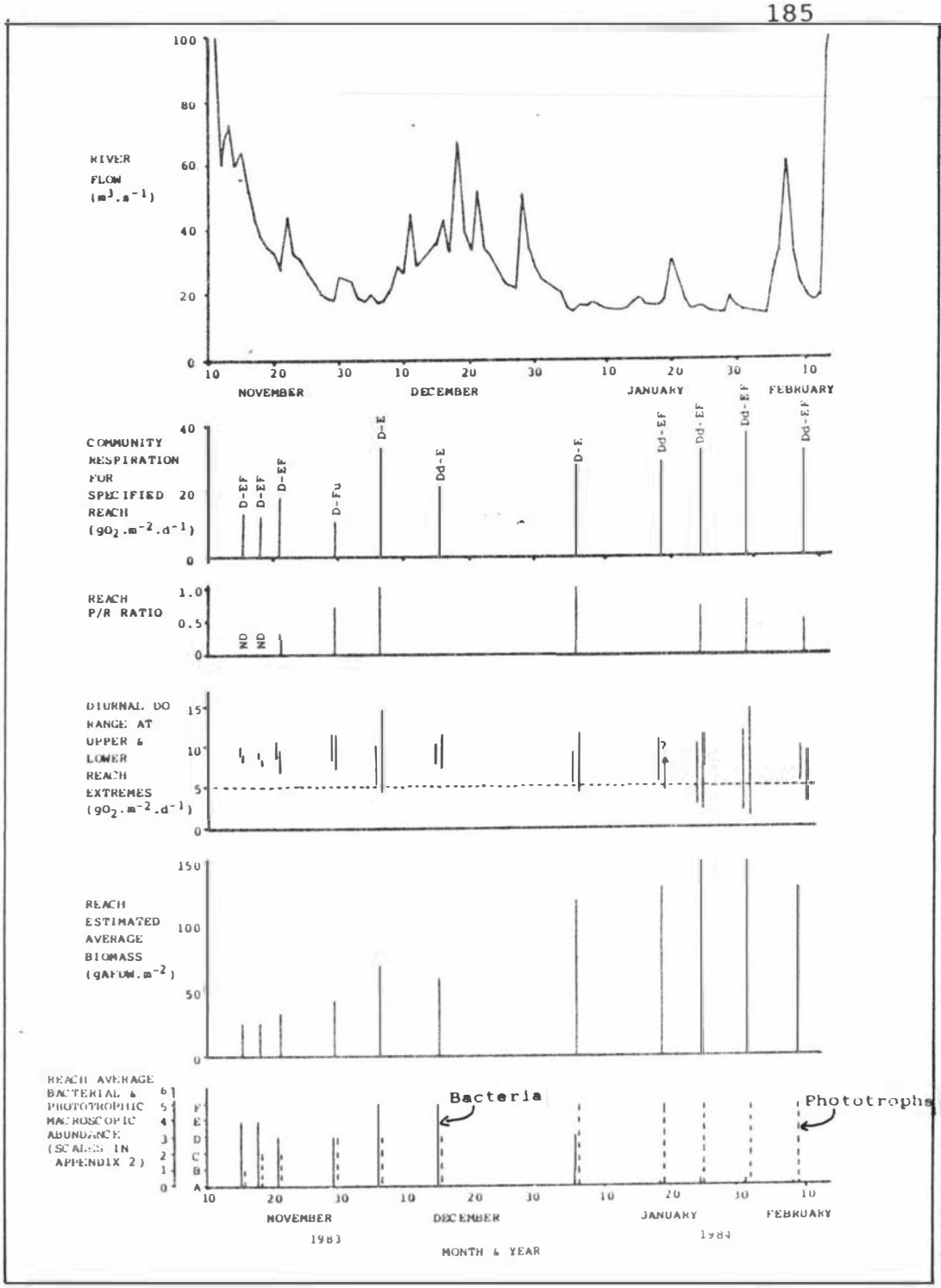


Figure 5.12: Variations in River Flow, Dissolved Oxygen Community Respiration and Benthic Biomass and Community Structure Over Reaches Below the Mixing Zone in the Manawatu River, 10 November 1983 to 10 February 1984.

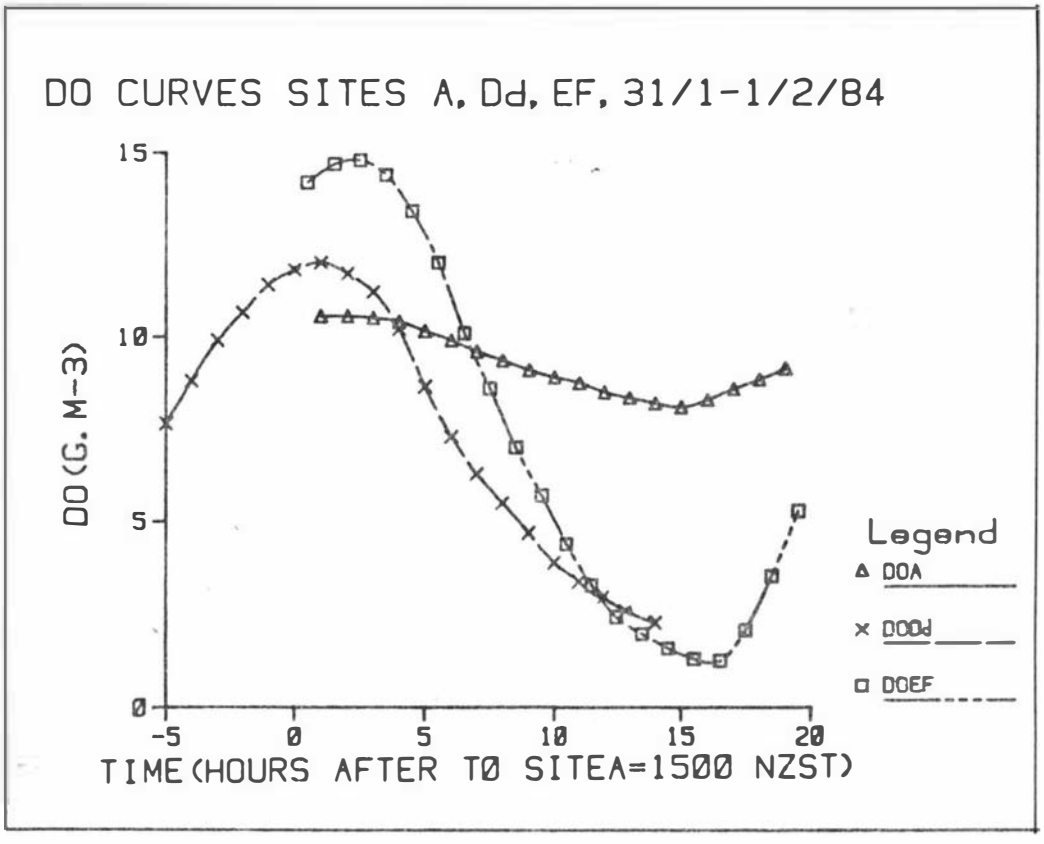


Figure 5.13: Dissolved Oxygen (DO) Variations at Sites A, Dd and EF on 31/1-1/2/84 ; temperature = 17-20.5^{o}C.

and EF. These reductions probably reflect the combined affects of the lower BOD_5 at site Dd, due to the land application of the part of the BCWS effluent from 6/2/84 (Gilliland, 1984), and reduced bed biomass and increased turbidity (causing reduced light at the river bed) due to the small spate which occurred from 5 to 7 February (Fig 5.12).

At the time of the fish kill, BCWS had not installed the additional waste treatment facilities required to meet the new water right conditions and the company's discharge data (Hinde, 1983) shows that it was in breach of these conditions throughout most of the period when it discharged during the summer. Because of this and the fact that dead fish were found from approximately the BCWS outfall downstream, initially that company alone received the blame for the low dissolved oxygen conditions and the fish kill (Ford, 1984; Renton, 1984). However the data collected above the BCWS outfall (Fig 5.11) and at site D below the outfall prior to the commencement of effluent discharge for 1984 on 16 January (Table 5.14) and on 2/2/84 (Section 5.4.3) show that even without the addition of the BCWS discharge the river did not meet its classification dissolved oxygen requirement. This implies that the BCWS BOD_5 loading to the river was only one of a number of factors which caused the low dissolved oxygen conditions.

The effects of various factors on reach respiration and gross photosynthetic rates are considered along with the data obtained within the waste discharge zone in Section 5.4.5 .

5.4.5 FACTORS AFFECTING RIVER RESPIRATION BELOW THE DISCHARGES

The data collected in the whole river oxygen dynamics studies below the effluent discharges (Appendix F) were analysed for interactions between the river respiration rate and environmental factors.

The linear regression equation (5.7) (Table 5.17) show that the benthic biomass, temperature and daily average initial

TABLE 5.17 Parsimonious Regression Equation Describing the Effects on the River Respiration Rate of Statistically Significant Factors (at the 95% level

$$RR = -21.9 + 0.0937\ BM + 1.58\ T + 1.90\ \text{Initial } BOD_5 \quad \text{(Equation 5.7)}$$

Units = $gO_2.m^{-3}.d^{-1}$

r^2 = 88.2% (adjusted for 16 degrees of freedom); standard deviation of data about the equation (s) = 2.81

Importance of Predictors

Predictor	Coefficient	St dev of Coefficient	t ratio = Coeff/S.D.	Percentage of Sum of Square reduction accounted for
Constant	- 21.9	7.4	-2.94	
BM	0.0937	0.0173	5.42	55.2%
T	1.58	0.43	3.67	12.3%
Initial BOD_5	1.90	0.34	5.55	22.7%

Application Range of Regression Equation

BM (benthic biomass) = 26-150 g AFDW.m^{-2}
T (temperature) = 15.2-21.0°C
Initial BOD_5 = 2.0-9.0 g.m^{-3}

Range of Other Environmental Variables

River flow = 14.8-58 $m^3.s^{-1}$

BOD_5 each have statistically significant effects (at the 95% level) on the river respiration rate. The regression equation (5.7) can account for 88.2% of the variation in the data using these three predictors. This is in general agreement with the results of the *in situ* chamber studies (Section 5.3) except that the initial chamber BOD_5 was not a statistically significant predictor of the benthic respiration rate in the former studies. The use of average daily initial BOD_5 values in the whole river studies, rather than the instantaneous values used in the chamber studies, may have contributed to the observed statistical significance of initial BOD_5 on the river respiration rate. The contrasting results suggest that the daily average initial BOD_5 does influence the river respiration rate although short-term variations may be relatively unimportant, possibly due to the metabolism of stored material by the heterotrophs.

The effect of the daily average initial BOD_5 on the river respiration rate is more pronounced when the data obtained from river reaches dominated by heterotrophs are modelled separately. As for the combined data set, the initial BOD_5, temperature and biomass all have statistically significant effects on the respiration rate but the initial BOD_5 accounts for most of the reduction in the sums of squares observed (Table 5.18). The parsimonious equation derived from this data set accounts for 95.4% of the variation in the river respiration observed and should be useful for predicting oxygen depletion in heterotroph dominated river reaches for the Manawatu River.

However the biomass data used to generate equations (5.7) and (5.8) were only semi-quantitative, being obtained from sampling (Section 3.2.2.1) at only up to three representative sites per reach and/or comparison of visual observations (Section 3.2.1.2) with calibrated biomass observations made on other occasions. Thus, in contrast to the data used to generate the regression models based on the *in situ* chamber measurements (Section 5.3), there is considerable uncertainty in the benthic biomass data used. This uncertainty could not be quantified

TABLE 5.18 Parsimonious Regression Equation Describing the Effects of Initial BOD_5, Temperature and Benthic Biomass on the Respiration Rate of Heterotroph Dominated River Reaches

$$RR = -25.3 + 2.05 \text{ Initial } BOD_5 + 1.78\ T + 0.0694\ BM \quad \text{(Equation 5.8)}$$

Units = $gO_2.m^{-3}.d^{-1}$

r^2 = 95.4% (adjusted for 9 degrees of freedom); standard deviation of data about the equation (s) = 1.9

Importance of Predictors

Predictor	Coefficient	St dev of Coefficient	t ratio = Coeff/S.D.	Percentage of Sum of Square reduction accounted for
Constant	- 25.3	7.5	-3.38	
Initial BOD_5	2.05	0.36	5.73	70.7%
T	1.78	0.49	3.63	16.8%
BM	0.0694	0.0164	4.25	9.2%

Application Range of Regression Equation

BM (benthic biomass) = 26-150 g AFDW.m^{-2}
T (temperature) = 15.2-21.7°C
Initial BOD_5 = 2.0-8.9 g.m^{-3}

Range of Other Environmental Variables

River Flow = 16.8-58 $m^3.s^{-1}$
Total Surface Radiation = 20.1-31.1 MJ.$m^{-2}.d^{-1}$

without intensive sampling of the entire reach. Nevertheless, the models derived from the whole river studies have the advantage that they were obtained from the unaltered river system for which predictions are required whereas the measurements in the *in situ* chambers required some unavoidable disturbance of the communities and alteration of the flow conditions, although these were minimised as much as possible (Section 3.6.3.1). This suggests that the results of the models derived from both situations should be considered when making predictions regarding the effects of management options on the river respiration rate.

5.4.6 VERIFICATION OF CHAMBER STUDY REGRESSION MODELS

The regression models of benthic respiration generated from the *in situ* chamber studies of heterotrophic dominated communities (Equation 5.3, Section 5.3.3) and the combined results of the chamber studies of these and the algal dominated communities (Equation 5.5, Section 5.3.5) were assessed by comparing the benthic respiration rates predicted for the biomass, temperature and initial BOD_5 data in the whole river studies below the discharges, with the measured river respiration rates.

In addition the following equation derived by Busch and Fisher (1981) from *in situ* chamber studies of *C. glomerata*, diatom and bluegreen algal assemblages was also assessed:

Busch and Fisher Equation: $R = 5.74\ B^{0.26}\ (1.10)^{T}$ where:

R = community respiration ($mg.O_2.m^{-2}.h^{-1}$)

B = benthic biomass ($gAFDW.m^{-2}$)

T = temperature (°C).

This was the only model for predicting benthic respiration rates of epilithon of relevance to this study sighted in the literature.

Since the chamber studies relate largely to the benthic respiration rate (Section 5.3.1), whereas the river rates

include the respiration of both the benthic community and that occurring in the water column, of approximately 1 metre depth, the values predicted using the equations derived from the chamber studies were expected to be lower than those observed in the river two station studies by an amount equal to the suspended biomass respiration (Section 5.2). Comparison of the values predicted by equations 5.5 and 5.3 with the measured rates in the two station studies (Table 5.19) shows that the predicted values were respectively 4.1 and 5.3 $gO_2.m^{-2}.d^{-1}$, or 15.9 and 20.2%, lower on average. This suggests that to adjust the chamber benthic respiration rate equations to suitable predictive equations for use in modelling the Manawatu River respiration rate a suspended biomass respiration rate of approximately 4 $gO_2.m^{-3}.d^{-1}$ in the 1 metre deep water column (= 4 $gO_2.m^{-2}.d^{-1}$) should be included in equations. This value is equal to the maximum suspended biomass respiration observed in this study (Section 5.2). Including this factor in equations (5.5) and (5.3) generates the following equations for the river respiration rate (RR; $gO_2.m^{-2}.d^{-1}$)

(5.5) $RR = -14.9 + 1.41\ T + 0.158\ BM$ (Equation 5.9)

(5.3) $RR = -9.3 + 1.07\ T + 0.151\ BM$ (Equation 5.10)

These adapted models generally predict rates very close to those observed in the river two station studies (Table 5.20) indicating that they would be useful predictive tools for use in water management. Since equation (5.9) was derived from the combined results of sewage fungus and algal dominated communities it should give more generally applicable results than equation (5.10), which was derived from the sewage fungus communities alone.

By contrast, the rates predicted by the Busch and Fisher model compare very poorly with those observed in the river two station studies (Table 5.19). The likely reasons for this poor performance are that the data from which the model was constructed were obtained from quiescent chambers and that the rates measured were calculated from measurements at the beginning and end of a one hour incubation period (Busch and Fisher, 1981). The respiration rates of benthic communities

TABLE 5.19 Comparison of Observed River Respiration Rates and Benthic Respiration Rates Predicted by Regression Models developed from Chamber Data

River Reach	Date	Measured River Respiration $gO_2.m^{-2}.d^{-1}$	Model Predicted Benthic Respiration ($gO_2.m^{-2}.d^{-1}$)			Measured-Predicted Respiration ($gO_2.m^{-2}.d^{-1}$)			Percentage Underestimation of measured rate (%)		
			(5.5)	(5.3)	(B & F)	(5.5)	(5.3)	(B & F)	(5.5)	(5.3)	(B & F)
D-EF	15/11/83	13.7	7.8	7.7	1.5	5.9	5.9	12.2	43.3	43.5	89.2
D-EF	17/11/83	12.5	10.6	9.9	1.8	1.9	2.6	10.7	15.3	20.9	85.7
D-EF	21/11/83	18.2	10.2	9.8	1.7	8.0	8.4	16.5	44.2	46.1	90.6
D-Fu	29/11/83	10.8	9.5	9.6	1.6	1.3	1.2	9.2	12.2	11.0	85.5
Dd-EF	18/01/84	28.8	23.5	22.9	2.1	5.3	5.9	26.7	18.4	20.4	92.5
Dd-EF	24/01/84	32.5	32.6	30.4	3.3	-0.1	2.1	29.2	-0.2	6.4	89.8
Dd-EF	31/01/84	37.6	30.5	28.8	2.9	7.1	8.8	34.7	19.0	23.3	92.3
Dd-EF	09/02/83	32.4	29.4	27.4	3.2	3.0	5.0	29.2	9.2	15.4	90.1
D-E	22/02/84	32.8	17.4	16.1	2.4	15.4	16.7	30.3	47.0	50.9	92.5
Dd-E	06/12/84	33.1	20.8	19.0	2.9	12.3	14.1	30.2	37.2	42.6	91.3
Dd-E	15/12/84	21.7	16.5	15.4	2.3	5.2	6.2	19.4	23.8	28.8	89.4
PNCC-B	16/01/84	20.7	18.5	17.6	2.2	2.2	3.1	18.5	10.8	14.8	89.2
PNCC-B	29/02/84	26.7	26.5	24.3	3.4	0.2	2.4	23.3	0.7	9.1	87.4
B-CuC	29/02/84	39.8	34.0	31.5	3.6	5.8	8.3	36.1	14.6	20.8	90.8
B-D	15/01/84	19.3	24.7	23.5	2.5	-5.4	-4.2	16.8	-28.2	-22.0	86.9
B-D	16/01/84	20.6	24.0	23.0	2.4	-3.4	-2.4	18.2	-16.6	-11.7	88.3
B-E	06/01/84	27.8	22.6	21.3	2.6	5.2	6.5	25.2	18.8	23.5	90.7
Mean $\bar{x}$						4.1	5.3	22.7	15.9	20.2	89.5
Std Dev						$\pm$ 5.2	$\pm$ 5.2	$\pm$ 8.3	$\pm$ 20.4	$\pm$ 19.4	$\pm$ 2.2

TABLE 5.20 Comparison of Measured River Respiration Rates and the Rates Predicted by Regression Models Developed from Chamber Data and Adapted to Allow for Suspended Biomass Respiration

River Reach	Date	Measured Respiration Rate ($gO_2.m^{-2}.d^{-1}$)	Model Predicted Respiration Rate (5.9)	(5.10)	Measured-Predicted Values ($gO_2.m^{-2}.d^{-1}$) (5.9)	(5.10)	Percentage underestimation of measured rate (%) (5.9)	(5.10)
D-EF	15/11/83	13.7	11.8	11.7	1.9	1.9	14.0	14.3
D-EF	17/11/83	12.5	14.6	13.9	-2.1	-1.4	-16.7	-11.1
D-EF	21/11/83	18.2	14.2	13.8	4.0	4.4	22.2	24.1
D-Fu	29/11/83	10.8	13.5	13.6	-2.7	-2.8	-24.8	-26.0
Dd-EF	18/01/84	28.8	27.5	26.9	1.3	1.9	4.5	6.5
Dd-EF	24/01/84	32.5	36.6	34.4	-4.1	-1.9	-12.5	-5.9
Dd-EF	31/01/84	37.6	34.5	32.8	3.1	4.8	8.3	12.7
Dd-EF	09/02/84	32.4	33.4	31.4	-1.0	1.0	-3.1	3.1
D-E	22/02/84	32.8	21.4	20.1	11.4	12.7	34.8	38.7
Dd-E	06/12/83	33.1	24.8	23.0	8.3	10.1	25.1	30.5
Dd-E	15/12/84	21.7	20.5	19.4	1.2	2.2	5.4	10.4
PNCC-B	16/01/84	20.7	22.5	21.6	-1.8	-0.9	-8.5	-4.5
PNCC-B	29/02/84	26.7	30.5	28.3	-3.8	-1.6	-14.3	-5.9
B-CuC	29/02/84	39.8	38.0	35.5	1.8	4.3	4.5	10.8
B-D	15/01/84	19.3	28.7	27.5	-9.4	-8.2	-48.9	-42.7
B-D	16/01/84	20.6	28.0	27.0	-7.4	-6.4	-36.1	-31.1
D-E	06/01/84	27.8	26.6	25.3	1.2	2.5	4.4	9.9
Mean $\bar{x}$					0.1	1.3	2.4%	1.9
S.D.					$\pm$ 5.2	$\pm$ 5.2	$\pm$ 22%	$\pm$ 21.4%

have been shown to increase with turbulence (Hickey, 1982) and to decrease during prolonged incubation periods (Section 5.3.6). Both these effects would tend to reduce the epilithon respiration rates measured using the methods of Busch and Fisher (1981). The failure of the resultant regression equation to give reasonable predictions emphasises the importance of attempting to match the turbulence within the respiratory chamber with that at the natural bed and the use of the initial respiration rate for model construction, rather than the average rate observed over a prolonged incubation.

Unfortunately, the lack of quantitative phototrophic biomass measurements in the whole river oxygen dynamics studies prevented the use of these data to test the linear regression equation models for benthic gross photosynthetic oxygen production derived from the *in situ* chamber studies.

5.4.7 COMPUTER MODELLING STUDIES

A computer model (Section 3.8) was used to investigate the assimilative capacity of the Manawatu River with respect to the respiration rate sustainable without the reduction of the dissolved oxygen concentration to less than 5 $g.m^{-3}$ during the night hours.

The effects of the reaeration rate, the sunset dissolved oxygen concentration and the temperature regime on the night-time dissolved oxygen depletion predicted by the model for a given respiration rate are summarised in Table 5.21. The predicted effects of various respiration rates for typical summer low-flow reaeration, temperature and initial dissolved oxygen conditions are shown in Figure 5.14.

Run 2 (Table 5.21) and Figure 5.14 show that under summer low-flow conditions an average respiration rate of 20 $gO_2.m^{-3}.d^{-1}$ over the reach below the PNCC outfall traversed at night allows maintenance of the required dissolved oxygen conditions. Reducing the k_2 value by 40% results in lower dawn dissolved oxygen values (Run 1) and reduces the maximum

TABLE 5.21 Results of Computer Modelling Studies : Effects of Reaeration Rate, Temperature and Initial Dissolved Oxygen Concentration on Dissolved Oxygen Depletion During a Night of Ten Hours Duration

Model Run	Respiration Rate ($gO_2.m^{-3}.d^{-1}$)	Temperature (°C)	Reaeration $k_{2(20)}$ (day^{-1})	Dissolved Oxygen ($g.m^{-3}$) Sunset	Predicted at Sunrise
1	20	22 - 20	2.8	9.8	4.3
2	20	22 - 20***	3.93*	9.8**	5.0
3	20	22 - 20	4.4	9.8	5.2
4	20	22 - 20	3.93	9.2	4.8
5	20	22 - 20	3.93	10.2	5.0
6	20	22 - 20	3.93	6.4	4.3
7	20	20 - 18	3.93	9.8	5.3
8	20	15 - 13	3.93	9.8	6.2
9	23.7	15 - 13	3.93	9.8	5.0
10	17	22 - 20	2.8	9.8	5.0
11	16.5	22 - 20	3.93	6.4	5.0

* = Average $k_{2(20)}$ measured in gas tracer studies at a river flow of 15.5 $m^3.s$ in the Manawatu River on 2/3/83 (Wilcock, 1984(a))

** = Typical sunset dissolved oxygen concentration

*** = Temperature variation during night observed on 29/2/84-1/3/84

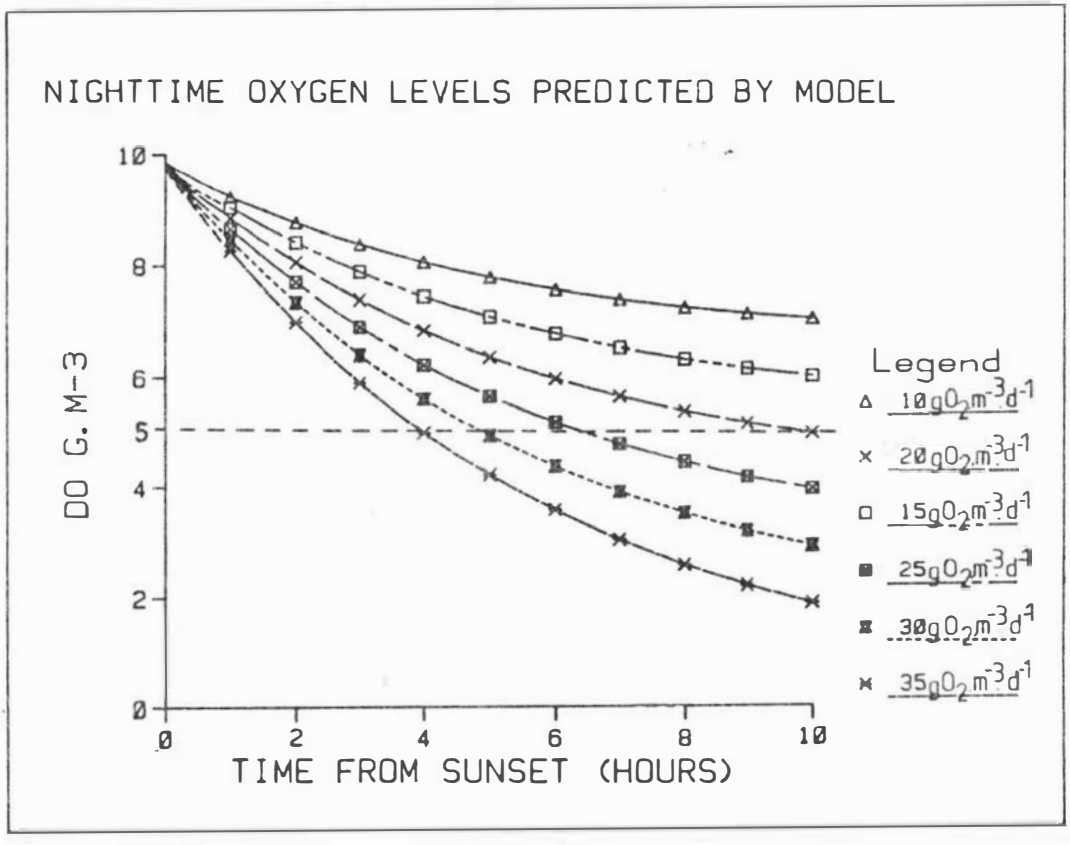


Figure 5.14: Night-time Dissolved Oxygen Levels Predicted at River Respiration Rates of 10-35 $gO_2.m^{-3}.d^{-1}$ by a Computer Model Simulating Manawatu River, Summer, Low-flow Conditions.

acceptable respiration rate from 20 to 17 $gO_2.m^{-3}.d^{-1}$ (Run 10). However variations in the sunset dissolved oxygen concentration over the approximate range of observed upstream values (9.2 to 10.2 $g.m^{-3}$) had a minor effect on the dawn dissolved oxygen concentration (Runs 3 to 5).

The sunset dissolved oxygen concentration in Run 6 was equal to that observed at site CuC below the reach from the MCDC outfall containing heavy sewage fungus, on 29/2/84 (Section 5.4.3) Although the sunset dissolved oxygen value was 3.4 $g.m^{-3}$ lower than that at sunset in Run 2, the dawn concentration was only 0.7 $g.m^{-3}$ lower due to the greater oxygen input in the latter case resulting from increased atmospheric reaeration at the lower average dissolved oxygen concentration. The model predicts a dawn dissolved oxygen concentration of 5 $g.m^{-3}$ under the conditions in Run 6 if the respiration rate is reduced to 16.5 $g.O_2.m^{-3}.d^{-1}$ (Run 11). A short sewage fungus zone, such as occurred between the MCDC outfall and site D during the 1983/84 summer, can therefore aggravate dissolved oxygen problems by:

(i) causing significant oxygen depletion through the reach

(ii) reducing the dissolved oxygen reserves available to meet the demands of downstream communities.

The substantial reduction in sewage fungus growth in this reach is therefore an important step towards resolving the dissolved oxygen depletion problems in the Manawatu River. This is discussed further in Section (7.2.2). However both phototroph dominated and sewage fungus dominated reaches below the waste discharges were observed to often have respiration rates in excess of 20 $gO_2.m^{-3}.d^{-1}$ during summer low flow conditions (Sections 5.4.3 and 5.4.4). The elimination or restriction of sewage fungus but not phototrophic growths is therefore unlikely to eliminate the dissolved oxygen depletion problems.

Comparison of Runs 2, 7 and 8 shows that temperature variations over the range observed during the spring to autumn growth season also have a significant effect on oxygen depletion for a given set of respiration, reaeration and sunset dissolved oxygen conditions. The effect of temperature on the maximum allowable

river respiration rate in order to maintain the dissolved oxygen concentration of an element of water traversing the river reach below the PNCC outfall to site EF at night above 5 $g.m^{-3}$ is shown in Figure 5.15. The maximum allowable rate increases from 20 to 24.5 $gO_2.m^{-3}.d^{-1}$ as the average night-time temperature decreases from 21 to 12°C, under constant, summer low-flow, conditions of reaeration, sunset dissolved oxygen concentration and night length (Fig 5.15).

Equations (5.9) and (5.10), adapted from the *in situ* chamber studies for use in predicting the Manawatu River respiration rate (Section 5.4.6), and equation (5.7), developed from the two station oxygen studies data (Section 5.4.5), were used to investigate the maximum benthic biomass levels allowable without causing dissolved oxygen concentrations of less than 5.0 $g.m^{-3}$.

The adapted chamber models predict that the nuisance level of benthic biomass, with respect to its effect on dissolved oxygen depletion, is inversely related to temperature (Fig 5.16). The more generally applicable model (Equation 5.9), which was developed from data obtained from a broad spectrum of epilithic communities, predicts that the nuisance biomass concentration decreases from 143 to 34 $gAFDW.m^{-2}$ with increasing temperature from 12 to 21°C (Fig 5.16).

This implies that during the winter and early spring, when the temperature is generally less than 14°C, heavy benthic biomasses, close to the maximum values observed during the summer low flows, would not cause the classification dissolved oxygen requirement to be breached, although these would result in nuisance levels of suspended biomass (Section 6.3). However during the critical summer low flow period the biomass should not exceed approximately 34 $gAFDW.m^{-2}$. The lower respiration rate observed for almost purely algal epilithon than for communities containing varying amounts of heterotrophic growth (Section 5.3.5) indicates that if the organic material input to the river was reduced to a level where purely algal communities occurred below the outfalls the maximum allowable benthic

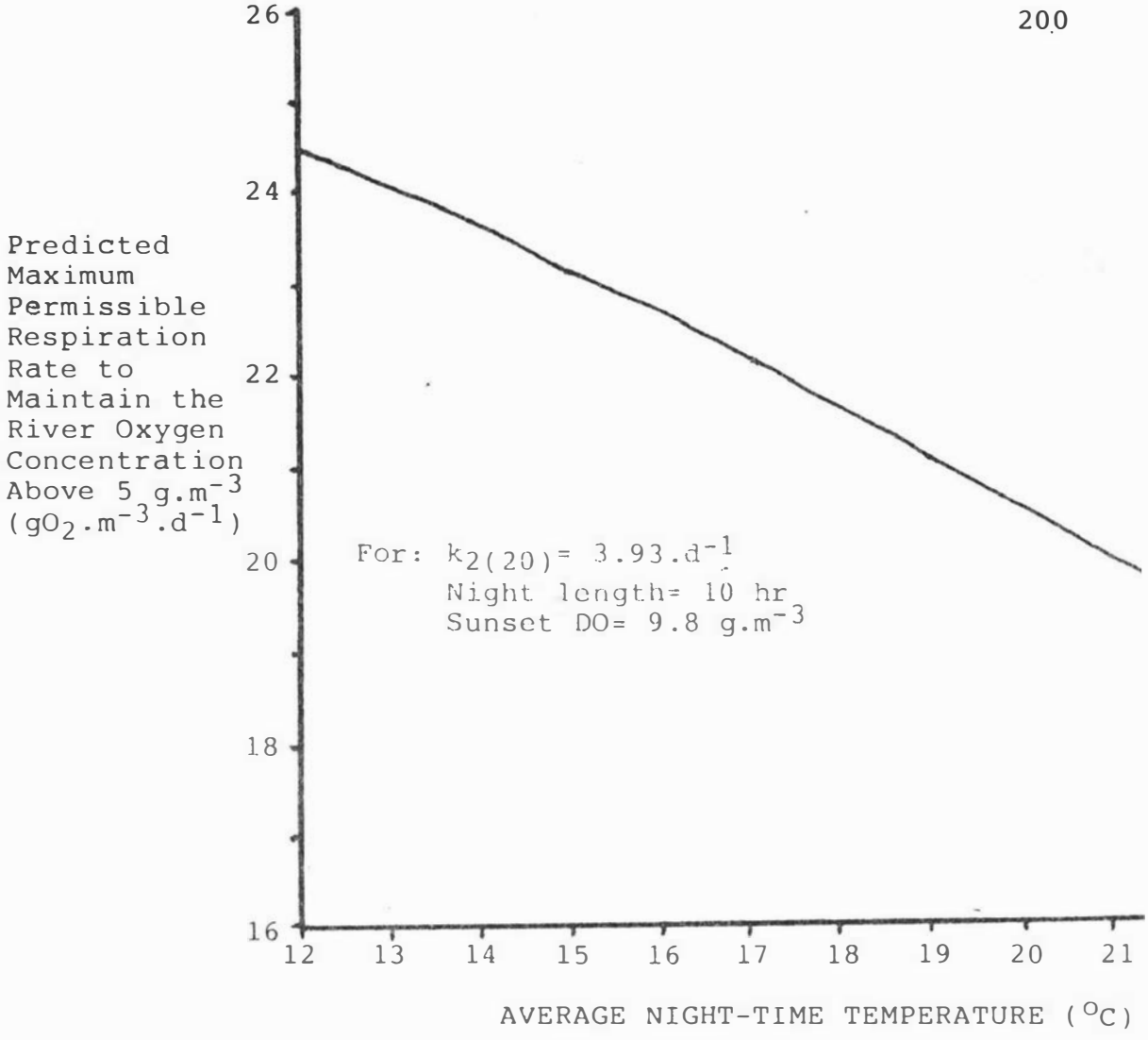


Figure 5.15: Graph of the Maximum Permissible Respiration Rate Predicted by the Computer Model Simulating Manawatu River Summer Low-flow Conditions versus Average Night-time Temperature.

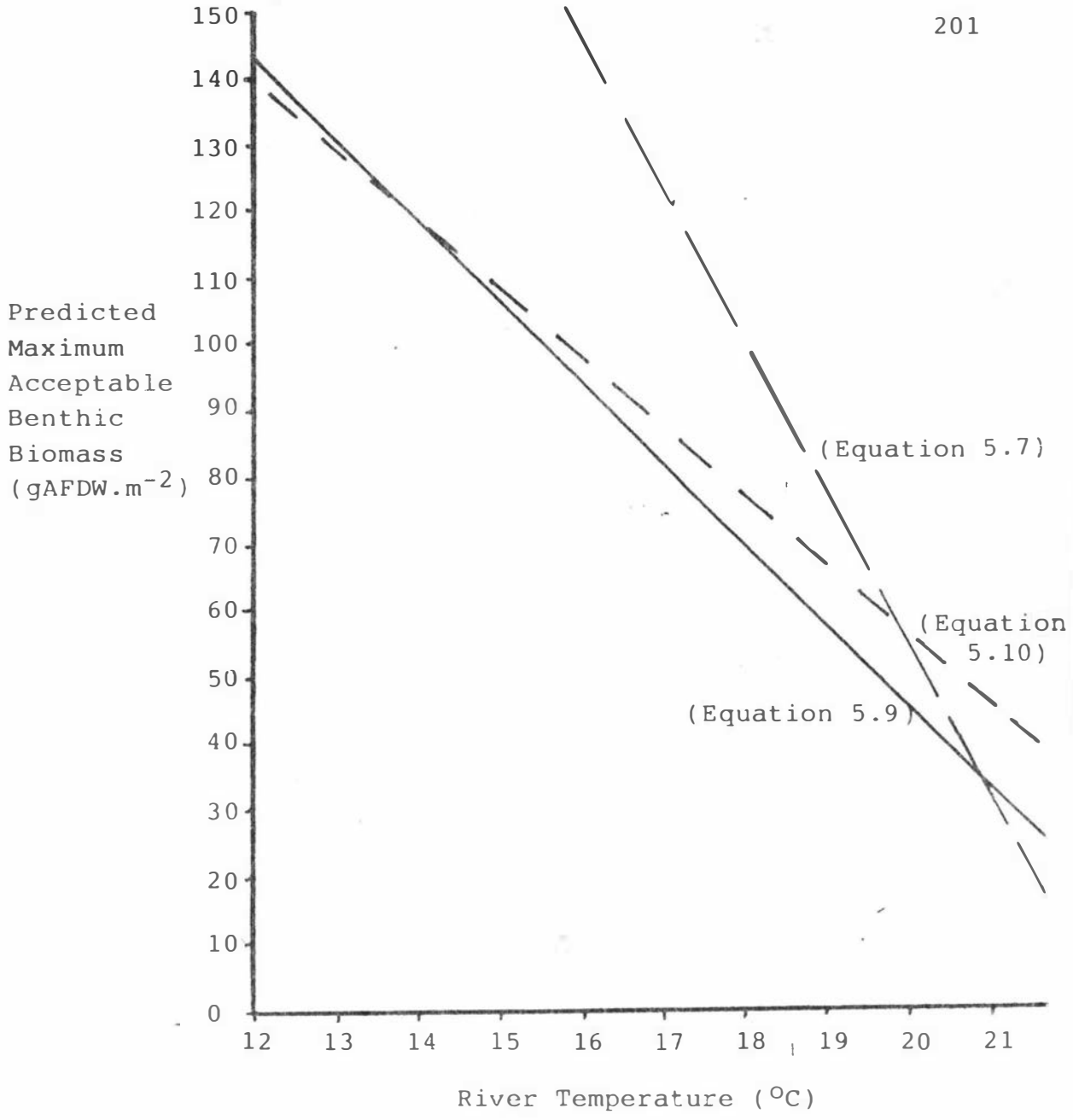


Figure 5.16: Maximum Acceptable Benthic Biomass Concentrations Predicted by the Whole River Respiration Model Solved for an Initial BOD_5 of 3g.m^{-3} (Equation 5.7) and the Adapted *In situ* Chamber Models (Equations 5.9 & 5.10) Versus River Temperature.

biomasses may be greater than predicted by the adapted regression models which were biased towards communities with some heterotrophic content (Appendices C to E).

The occurrence of an initial BOD_5 term, as an index of the reach BOD_5 conditions, as a statistically significant predictor in the multiple regression models derived from the two station studies (Section 5.4.5) complicates their use for prediction of the maximum allowable benthic biomass over the longer reach below the PNCC outfall traversed at night under summer low flow conditions (9 to 10.5 km at flows of 20 to 13 $m^3.s^{-1}$ respectively). These models were derived from observations over shorter, 1.5 to 6 km long, reaches without effluent discharges or with a single known discharge directly below the upstream site (except on 16-17/1/84). By contrast the reach traversed below the PNCC discharge at night receives the MCDC and BCWS discharges 3.8 and 5.6 km below the PNCC discharge respectively. This makes it difficult to estimate an appropriate initial BOD_5 term to use in the modelling exercise to predict the maximum acceptable benthic biomass. If it is assumed that the average BOD_5 concentration over the reach below PNCC traversed at night corresponds to that in the two station study reaches when the initial BOD_5 was 3 $g.m^{-3}$ then the more generally applicable model (Equation 5.7), developed from the combined results of all the studies over reaches below the discharges, predicts the maximum allowable biomass levels given in Figure (5.16). Although this model predicted similar maximum acceptable biomass levels in the 20-21°C region to those predicted by the models developed from the *in situ* chamber data (Equations 5.9 and 5.10), much lower acceptable biomass values were predicted at temperatures above 21°C (fig 5.16). This probably results from the relatively narrow range of temperatures and the lack of situations of simultaneous high temperature and low or moderate biomass concentration in the data base from which equation (5.7) was derived (Appendix F). This indicates that the maximum acceptable biomass values predicted by this model are less reliable than those predicted by the models derived from the *in situ* chamber data.

5.4.8 CONCLUSIONS

Both sewage fungus and phototroph dominated benthic communities which developed in the Manawatu River during summer lowflows had high night-time respiration rates. This often resulted in night-time dissolved oxygen depletion to levels below 5 $g.m^{-3}$, required by the river's D classification.

A computer model was constructed to simulate the effect of various respiration rates on the dissolved oxygen concentration of an element of water passing through the waste discharge zone at night. This model predicted a maximum allowable respiration rate of 20 $gO_2.m^{-3}.d^{-1}$ under summer low flow conditions. The permissable respiration rate increased with decreasing temperature.

A sewage fungus dominated reach below the MCDC outfall caused marked reduction in the dissolved oxygen concentration of water passing through it during the day and night. This significantly reduced the allowable night-time respiration rate further downstream. The substantial reduction of the heterotrophic biomass over this reach is an important step towards maintaining the river classification dissolved oxygen requirements.

The night-time respiration rates observed over the reaches below the discharges dominated by phototrophs often exceeded 20 $gO_2.m^{-3}.d^{-1}$. Eliminating or restricting sewage fungus growth but not phototroph growth is therefore unlikely to remedy the dissolved oxygen depletion problems occurring in the Manawatu River.

Multiple regression models of the river respiration rate developed from the results of the two station dissolved oxygen studies of reaches below the discharges showed that initial BOD_5, temperature and benthic biomass all had significant effects on the river respiration rate. These factors accounted for 88.2% of the variation in the river respiration rate

observed in the complete data set and 95.5% of that observed in the heterotrophic dominated reaches. The models generated should be useful for predicting the respiration rates of various reaches below waste discharges in the Manawatu River.

The respiration rates predicted by the empirical models derived from the *in situ* chamber studies compare reasonably well with the rates measured in the river below the waste discharges, especially after being adjusted to include a suspended biomass respiration rate. This suggests that these models can be usefully employed to predict the benthic respiration rate for a given set of temperature and biomass conditions. These models show that the nuisance level of benthic biomass, capable of reducing the river dissolved oxygen content to less than 5 $g.m^{-3}$ at night, is strongly influenced by temperature. At 12°C and 21°C the models predict that benthic biomasses of 139 and 143 $gAFDW.m^{-2}$ and 34 to 45 $gAFDW.m^{-2}$ respectively constitute a nuisance.

5.5 LABORATORY CHANNEL STUDIES

5.5.1 INTRODUCTION

Laboratory channels (Section 3.7) were used to study the effect of benthic communities on oxygen dynamics under a range of waste loading conditions. Different experiments were conducted by adding a range of concentrations of untreated MCDC effluent or primary treated BCWS effluent (Table 5.22) to borewater with a background BOD_5 of 0.11 to $0.59 g.m^{-3}$. The effects of the wastewater loadings on epilithon development and self-purification are discussed in section (7.3) and section (6.2.3) respectively.

TABLE 5.22: Summary of Laboratory Channel Experiments Undertaken

Experiment	Dates	Wastewater	Influent Wastewater Addition ($g.m^{-3}$)	
			COD	$=BOD_5$*
A (control)	07/06/84-20/07/84	None	0	0
B	04/10/83-02/11/83	MCDC	1.9 ± 0.1	1.2 ± 0.6
C	08/08/83-22/09/83	MCDC	3.8 ± 0.6	2.5 ± 0.4
D	08/08/83-22/09/83	MCDC	6.5 ± 1.0	4.2 ± 0.6
E	10/04/84-21/05/84	BCWS	1.9 ± 0.25	1.14 ± 0.15
F	10/04/84-31/05/84	BCWS	4.2 ± 0.4	2.5 ± 0.2
G	05/06/84-10/07/84	BCWS	7.3 ± 0.9	4.4 ± 0.5
H	12/07/84-27/07/84	BCWS	14.4 ± 0.7	8.6 ± 0.4

* Calculated using COD:BOD_5 conversion factors of 0.65 for the MCDC effluent (Meredith, 1982) and 0.6 for the BCWS effluent (from 5 samples tested; $x \pm s = 0.6 \pm 0.05$).

Since the channels were recirculatory with a constant inflow and outflow and a hydraulic residence time of approximately 11.2 minutes (Section 3.7) the organic conditions in the channels do not correspond to those directly below a river discharge causing a similar BOD_5 increase to that in the channel influent. Rather they correspond to conditions some distance downstream of the outfall. The theory outlined in Section (7.3.5) predicts that an outfall to the Manawatu River giving a similar effluent dilution to the channel influent would produce similar organic concentrations in the water column to those in the channel at sites 6.1 km and 3.3 km downstream for river flows of 28 and 13 $m^3.s^{-1}$ respectively.

It should be noted that the respiration and gross photosynthetic rates are derived from the growth on top of flat surfaces of the channel plates whereas measurements (Section 3.4.3) showed that the roughness of the Manawatu River cobble bed (Section 4.2) causes the light exposed area per unit flat surface area to be increased by 1.63 times on average at sites C and D within the waste mixing zone (Fig 1.2). Limited biomass development may also occur on the under surfaces of the river stones, although this probably becomes relatively

unimportant once the biomass develops to the point where the interstitial spaces are filled and mass transfer of oxygen and organic substrate is restricted due to the growth on the exposed surfaces (Section 7.2.6.2). The effect of the bed roughness is to raise the heterotrophic respiration rates per unit area considerably. For example the increase in light exposed surface area alone raises the heterotrophic respiration by 1.63 times.

5.5.2 EFFECTS OF ORGANIC WASTE ADDITION ON EPILITHON PHOTOSYNTHESIS

The addition of the MCDC and BCWS wastewaters had comparatively minor effects on the channel epilithon gross photosynthetic oxygen production rates (Figs 5.17 and 5.18).

The rates measured for the epilithon in the control channel were broadly similar to those in experiments B and C (MCDC effluent) and experiments E, G and H (BCWS effluent). The rates were also similar to those observed for periphyton in laboratory streams receiving streamwater under similar light conditions (McIntire, 1975). The limited data from experiments D and F show that in some instances higher gross photosynthetic rates did occur in the channels receiving the organic wastes. However the differences between the maximum gross photosynthetic rates in the control and treatment channels were much less than the differences in the maximum respiration rates (Section 5.5.3).

The high background nutrient levels in the borewater (Table 5.23) offers an explanation for the comparatively minor effect of the waste additions on the epilithon phtosynthetic rates.

These background levels of nitrogen (TDN = 717 to 942 $mg.m^{-3}$) and phosphorus (TDP = 74 to 87 $mg.m^{-3}$) would be expected to prevent nutrient limitation of algal growth (Wuhrmann and Eichenberger, 1975; Freeman, 1983; Horner *et al*, 1983). Thus the added nutrients from the wastewaters had little effect.

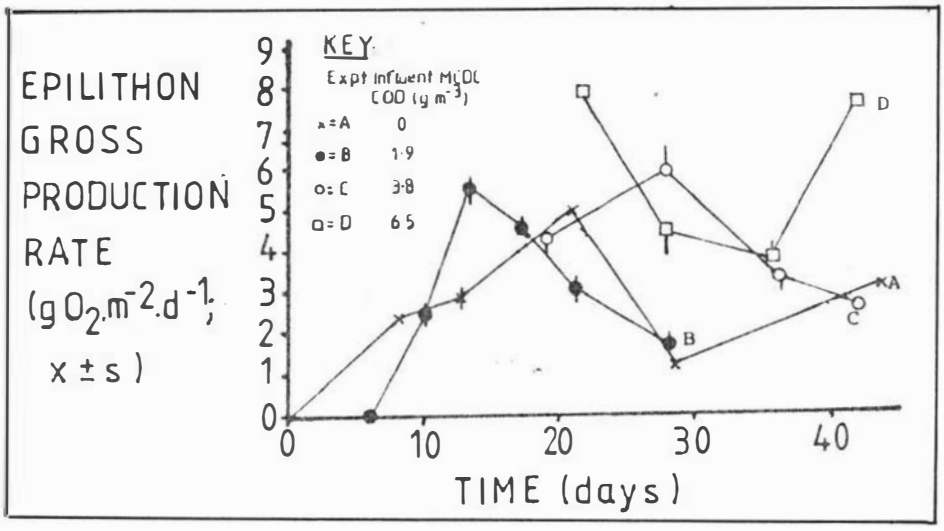


Figure 5.17: Gross Photosynthetic Oxygen Production of Channel Epilithon Under Different MCDC Wastewater Loadings (calculated for 12 hours light/day at 45 uE.m^{-2}.s^{-1}PAR).

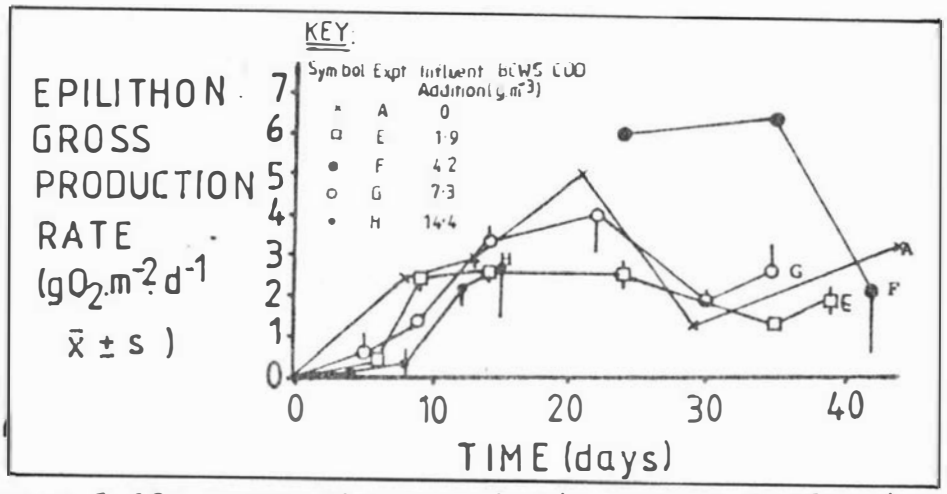


Figure 5.18: Gross Photosynthetic Oxygen Production of Channel Epilithon Under Different BCWS Wastewater Loadings(calculated for 12 hours light/day at 45 uE.m^{-2}.s^{-1}PAR).

TABLE 5.23 Results of Channel Experiment Nutrient Analyses

Experiment	Wastewater	Day	TDN ($mg.m^{-3}$; $x \pm s$)		TDP ($mg.m^{-3}$; $x \pm s$)	
			Influent	Effluent	Influent	Effluent
A	None	8	717 ± 0	669 ± 11	74 ± 1	67 ± 0
A	None	22	942 ± 32	882 ± 0	76 ± 2	90 ± 4
A	None	30	898 ± 0	798 ± 23	87 ± 4	76 ± 11
D	MCDC	11	1360 ± 70	1170 ± 70	259 ± 3	252 ± 52
D	MCDC	32	1600	1600 ± 50	233 ± 33	208 ± 9
G	BCWS	15	883 ± 30	775 ± 5	119 ± 1	116 ± 1
G	BCWS	31	ND	1200 ± 10	126 ± 7	126 ± 2

The increases in epilithon gross photosynthesis that were observed in experiments C and D are likely to have resulted from organic inputs producing plumose fronds and streamers of filamentous microorganisms which became densely colonised by algae. The lighting conditions and oxygen and nutrient mass transfer within these growths may have been more favourable than those within the cohesive epilithic mat which developed in the control channel. The effects on oxygen dynamics of cohesive epilithic mats and communities composed of freely moving filamentous algae are compared in Section (5.5.5).

5.5.3 EFFECTS OF ORGANIC WASTE ADDITIONS ON EPILITHON RESPIRATION

The addition of 1.9 to 6.5 g COD.m^{-3} of MCDC wastewater resulted in marked increases in the respiration of the channel epilithon compared with that occurring in the control experiment (Fig 5.19). Although the results from experiments C and D were quite variable and did not cover the entire period of the experiments, the data indicate a general increase in epilithon respiration with increasing organic loading.

Similar COD loadings of BCWS meatworks effluent to those in the MCDC studies produced lower epilithon respiration rates (Fig 5.20) but a similar trend of increased respiration with increased organic loading was observed. However addition of 1.9 gCOD.m^{-3} did not cause any obvious respiration rate increase above that of the control channel epilithon.

The epilithon in experiment H began to slough off between days 12 and 15 (Section 7.3.4) resulting in the large variation in respiration rates on day 15 (Fig 5.20). Because of the sloughing, the experiment was terminated on day 15 due to the lack of plates with growth from day 0 attached. The respiration rate of the epilithon samples which had not suffered sloughing losses on day 15 was 14.8 $\pm$ 2.0 $gO_2.m^{-2}.d^{-1}$. This represents a similar maximum increase in respiration rate to those observed in response to MCDC wastewater loadings of 3.8 and 6.5 g COD.m^{-3} (Fig 5.19). The very early occurrence

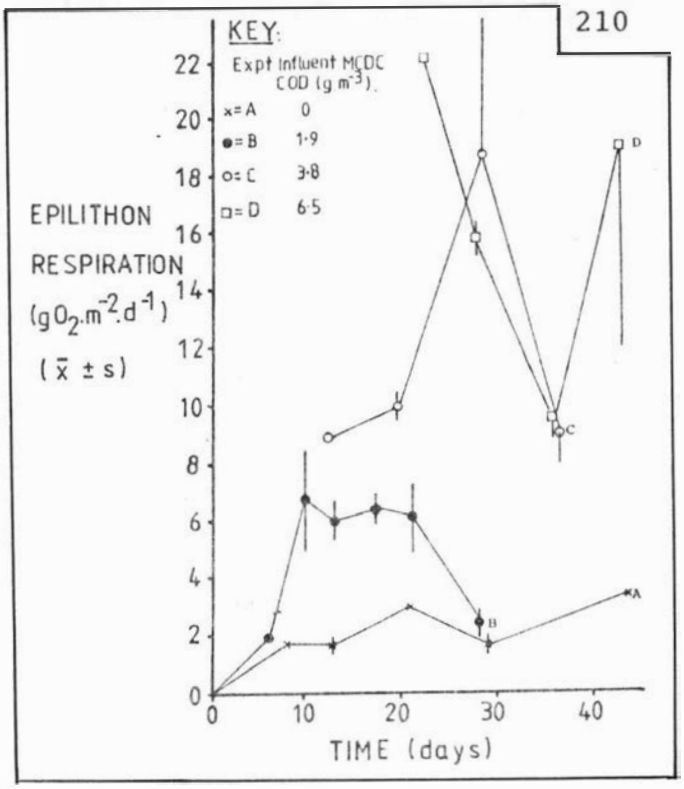


Figure 5.19: Respiration of Channel Epilithon Under Various MCDC Wastewater Loadings.

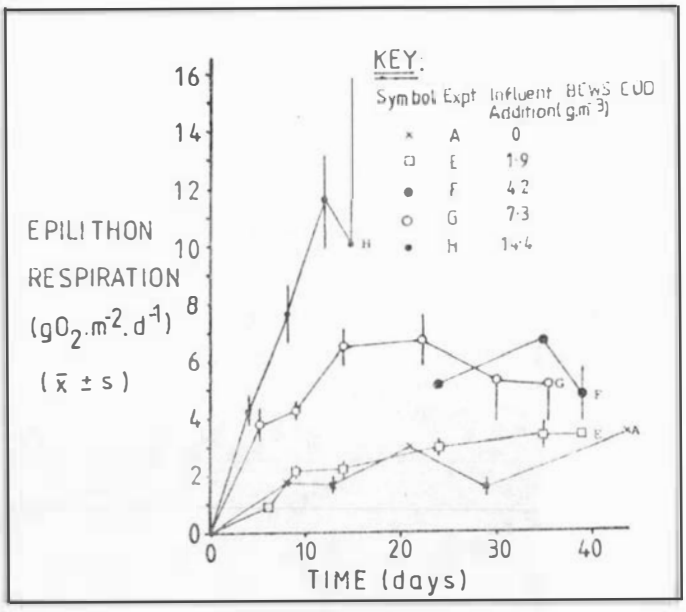


Figure 5.20: Respiration of Channel Epilithon Under Various BCWS Wastewater Loadings.

of sloughing in experiment H may have been promoted by a twelve hour period of elevated temperatures (30°C) due to the accidental alteration of the temperature control system overnight on day 8 to 9.

BCWS wastewater loadings of 4.2 $gCOD.m^{-3}$ and 7.3 $gCOD.m^{-3}$ only resulted in increases in the epilithon respiration rate of 3 to 4 $gO_2.m^{-2}.d^{-1}$, similar to those resulting from a MCDC wastewater loading of 1.9 $g.m^{-3}$.

The observation that the channel epilithon gross photosynthetic rates were not markedly affected by the wastewater additions (Section 5.3.4) implies the algal respiration in the treatment channels was also similar to that in the control and that the increase in respiration in response to wastewater additions was largely due to increased heterotrophic activity. Thus the results of the respiration studies suggest that two to three times as much BCWS wastewater COD addition is required to produce the same increase in heterotrophic activity as a given COD of MCDC wastewater.

The results of filtration-fractionation studies of the two wastewaters offers an explanation for this observation. The data in Table 5.24 show that the MCDC wastewater contains much greater proportions of dissolved and low molecular weight organic material than that of BCWS. Furthermore, analyses of nine, twenty-four hour composite MCDC wastewater samples, collected on randomly selected days between 19/12/82 and 9/2/83, showed that lactose (MW = 342 daltons) contributed 36 $\pm$ 11.5% ($\bar{x} \pm s$) to the total COD (Appendix G). The percentage contribution of lactose to the total COD of the MCDC wastewater used in experiments C and D was towards the top of the range of values for the 24 hour composite samples at 44%. Since the dissolved, and particularly the low molecular weight, organics are known to promote the growth of the heterotrophic slime forming organisms (Section 2.3.4.2), the greater concentration per unit COD of these compounds in the MCDC than in the BCWS wastewater could explain the greater increase observed in heterotrophic activity per unit MCDC COD addition.

TABLE 5.24 Results of Filtration/Ultrafiltration Fractionation Studies of Wastewater Organics

Effluent	Date Collected	Treatment Received	Sample Type	Dissolved Fraction	YM2 Ultrafiltrable Fraction
PNCC	7/9/84*	Primary	Grab, 1020 AM	30% of COD	15% of COD
PNCC	29/2-1/3/84	Primary + Alum flocculation	24 h composite	76% of BOD_5	ND
PNCC	15-16/1/85	Primary	24 h composite	44% of COD	21% of COD
MCDC	29/2/84	None	9 h composite	82% of BOD_5	ND
MCDC	6-7/9/84	None	24 h composite	82% of COD	43% of COD
BCWS	1/6/84	Primary	1 h composite (channel feed)	35% of COD	11.3% of COD
BCWS	16/6/84	Primary	1 h composite (channel feed)	36% of COD	15.6% of COD

* one of the two sedimentation tanks was inoperative when the sample was collected.

However the contribution of lactose to the total COD of the MCDC wastewater discharged to the Manawatu River is expected to be reduced in the future as the effluent lines with the highest lactose concentration are diverted to land disposal (Meredith, 1984). This would be expected to reduce the heterotrophic growth in response to the effluent per unit COD or BOD_5 discharged.

The application of the channel results directly to the river situation is complicated by a number of factors (Section 7.3.5). However if the following assumptions are made some further tentative conclusions can be drawn. Assumptions:

(i) that the roughness of the Manawatu River bed results in an increase of 1.63 times in the benthic respiration rate over those observed for the flat substrates used in the laboratory channels (Section 5.5.1).

(ii) that the ratio of phototroph respiration to gross photosynthetic rates (as $gO_2.m^{-2}.d^{-1}$) in the channels receiving wastewater discharges equals the average ratio of 0.77 measured for the control channel biomass.

Given these assumptions it can be calculated from the data in figures 5.17 to 5.20 that the maximum increases in heterotroph respiration at the "channel-river equivalent site" (Section 7.3.5) are approximately 3, 4.5, 6.5 and 21 $gO_2.m^{-2}.d^{-1}$ in response to sufficient discharge of BCWS wastewater to river water with a background BOD_5 of 0.11 to 0.59$g.m^{-3}$ (Section 7.3.2) to increase the COD by 1.9, 4.2, 7.3 and 14.4 $g.m^{-3}$ respectively. Similarly maximum heterotroph respiration increases of the "channel-river equivalent site" of approximately 8, 23 and 26 $gO_2.m^{-2}.d^{-1}$ are predicted in response to addition of sufficient MCDC wastewater to increase the river COD by 1.9, 3.8 and 6.5 g $COD.m^{-3}$ respectively. The anticipated increase in phototroph activity in response to these discharges in the river situation (Section 5.4.2) would be expected to further augment the benthic respiration rate by an unknown amount. However the results show that to limit the

increase in benthic heterotroph respiration at the "channel-river equivalent site" to an arbitrarily adopted, maximum acceptable level of 8 $gO_2.m^{-2}.d^{-1}$ the addition to an unpolluted stream of a meatworks effluent such as the BCWS effluent tested should be restricted to about 7.3 $gCOD.m^{-3}$ (= 4.4 $gBOD_5.m^{-3}$). Similarly the discharge of untreated dairy factory effluent, similar to the MCDC samples tested, should be restricted to 1.9 $gCOD.m^{-3}$ (= 1.2 $gBOD_5.m^{-3}$).

These results demonstrate the usefulness of recirculating laboratory channel systems for comparing the influence of individual wastewaters on epilithon development and consequent water quality impacts and identifying the level of dilution that results in a shift to a heterotrophic dominated community.

5.5.4 VARIATIONS IN BIOMASS SPECIFIC RESPIRATION AND PHOTOSYNTHETIC RATES DURING THE CHANNEL EXPERIMENTS

The variations in the channel epilithon biomass and the biomass specific rates of respiration (WSBR) and chlorophyll a specific gross photosynthetic oxygen production (chla.SBGPR) during experiments A, B, E and G are shown in Figures 5.21 to 5.24.

These show a consistent pattern of inverse relationships between WSBR and total biomass and also chl a.SBGPR and chlorophyll a. The WSBR and chlorophyll a SBGPR declined most rapidly with biomass increases up to approximately 10 $gAFDW.m^{-2}$ and 100mg $chla.m^{-2}$ respectively. This is consistent with the trend observed in the Manawatu River growth experiments at site D (e.g., Figs 5.1 to 5.4). The relationships between WSBR and biomass were very similar to that observed by Capblancq and Cassan (1979(b)) for periphyton developing on artificial substrates in a polluted river. A similar relationship between specific BGPR and algal biomass (*Cladophora* sp.) was also observed by Pfeifer and McDiffett (1975).

The following factors are likely to have been important in producing the reduction in WSBR of the heterotrophs with increasing biomass:

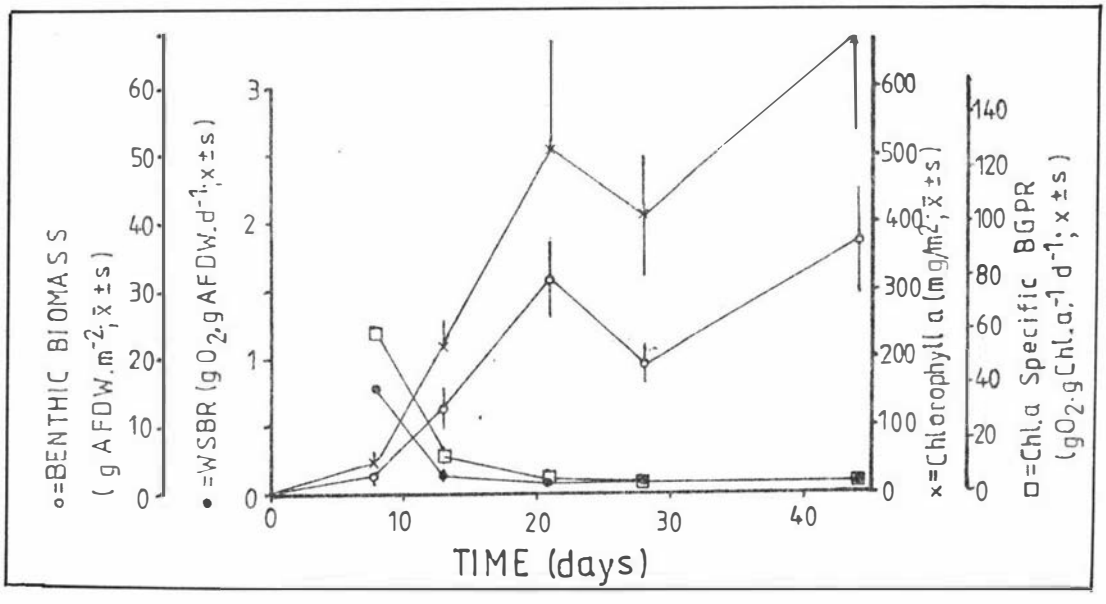


Figure 5.21:Variations in Biomass, Weight Specific Respiration and Chlorophyll a Specific Gross Production During Channel Experiment A (control= no wastewater addition to influent).

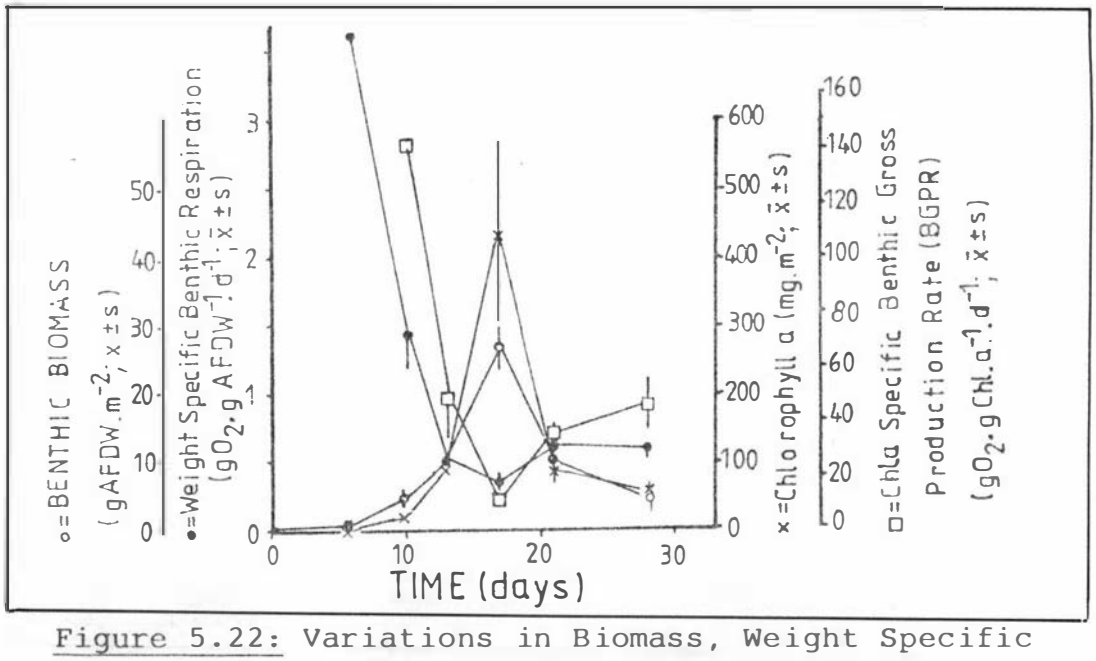


Figure 5.22: Variations in Biomass, Weight Specific Respiration and Chlorophyll a Specific Gross Production During Channel Experiment B (1.9 gCOD.m^{-3} MCDC waste-water in influent).

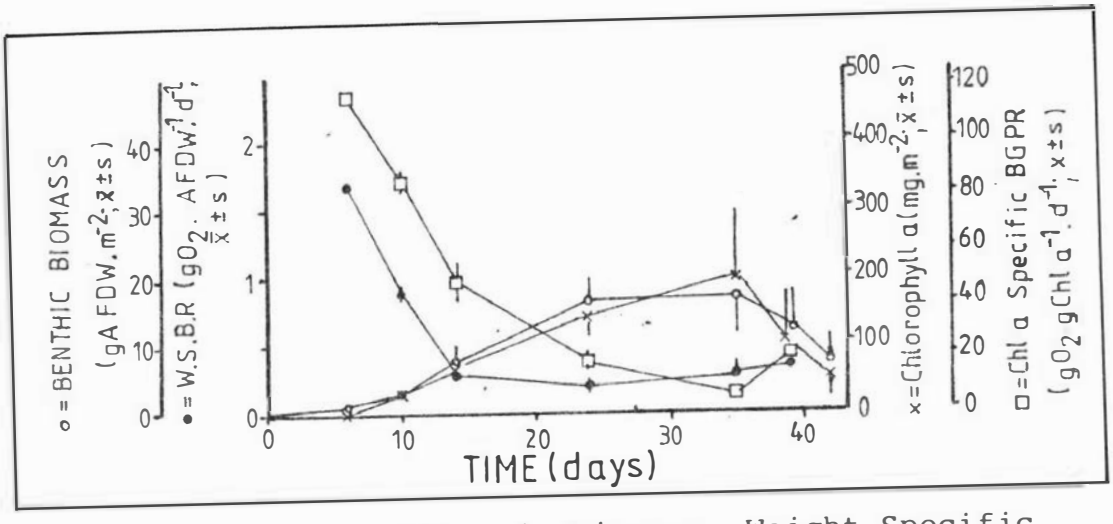


Figure 5.23: Variations in Biomass, Weight Specific Respiration and Chlorophyll a Specific Gross Photosynthesis During Channel Experiment E (1.9gCOD.m^{-3} BCWS waste-water in influent).

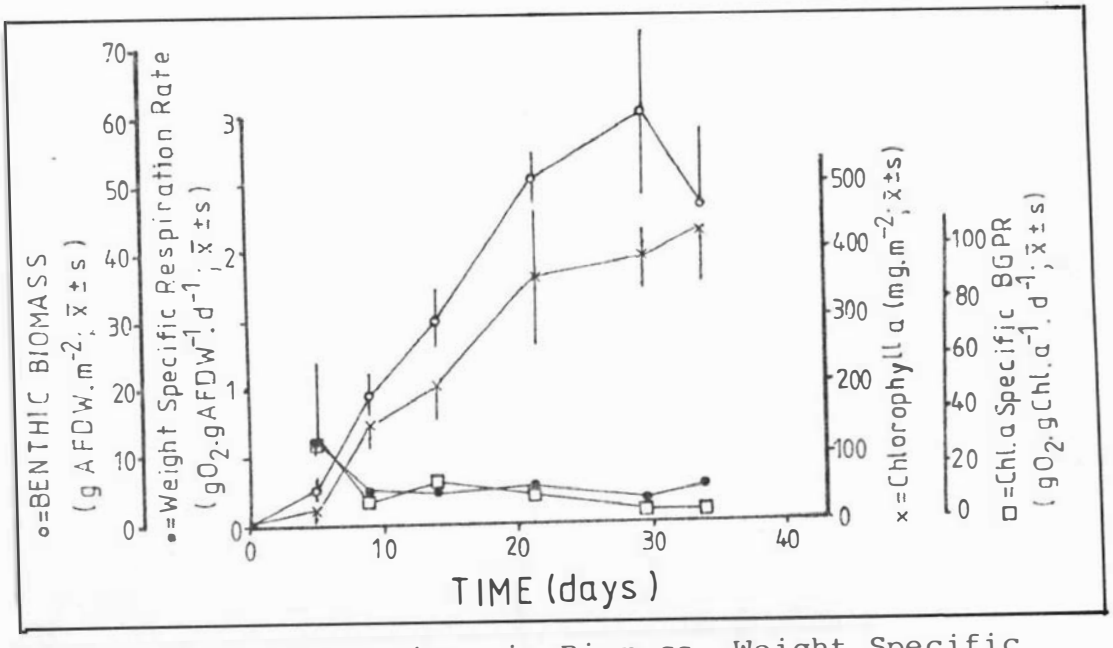


Figure 5.24: Variations in Biomass, Weight Specific Respiration and Chlorophyll a Specific Gross Photosynthesis During Channel Experiment G (7.3gCOD.m^{-3} BCWS waste-water in influent).

(i) reduced organic substrate availability per unit biomass as the channel biomass increased (Section 7.3).

(ii) reduced availability of oxygen to the lower cells in the epilithon due to the metabolism of overgrowing cells and greater distances from the bulk media to the lower cells due to the overgrowth.

(iii) reduced availability of organic substrate to the lower cells of the epilithon due to the effects outlined in (ii).

For the phototrophs reduced light availability to the lower cells of the epilithon due to absorbance by pigmented cells in the upper layers (Losee and Wetzel, 1983) is the most likely cause of the reduction in chlorophyll a SBGPR with increasing algal biomass. The high nutrient levels and low nutrient removal observed in the channels at all biomass levels (Table 5.23) indicate that nutrient limitation is not important in this situation. By contrast, light has been shown to limit BGPR in both river experiments (Section 5.3.5) and laboratory studies (Fig 5.25). Since the respiration of shaded phototrophic cells is generally lower than non-shaded cells (Boardman, 1977; Darley, 1982) due to lower enzyme levels (Losee and Wetzel, 1983), this shading effect would also result in a reduction in average phototroph respiration with increased phototrophic biomass.

Comparison of the biomass specific photosynthetic rates of the channel communities (Figs 5.21 to 5.24) and the river communities (Figs 5.4, 5.7-5.9) shows that the channel rates are considerably lower for similar temperature and biomass conditions. However the results of the river epilithon shading studies (Figs 5.7 to 5.9) indicate that under the light conditions during the channel measurements (45 $\mu E.m^{-2}.s^{-1}$), the river epilithon would have similar chlorophyll a specific BGPR's to the channel epilithon at corresponding biomass levels.

The algal dominated channel epilithon also had considerably lower respiration rates than the algal dominated river

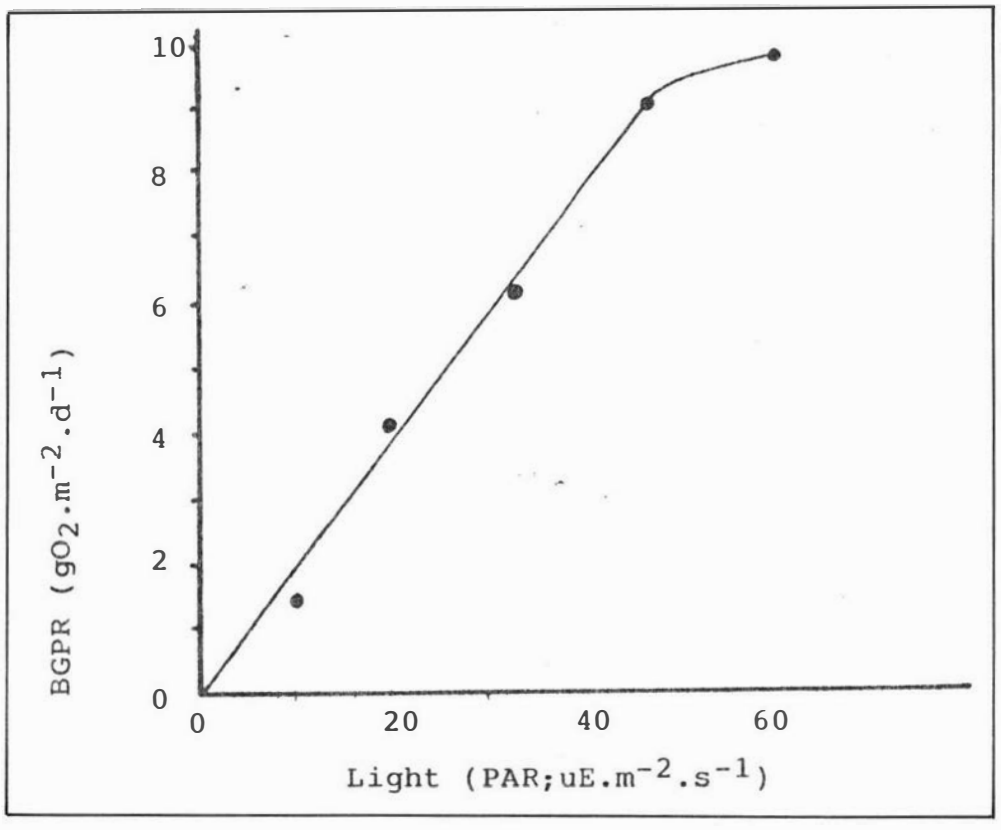


Figure 5.25: Benthic Gross Photosynthetic Oxygen Production Rate Versus Light; Channel Biomass, day 42 of Experiment D.

communities under similar temperature and biomass conditions. For example the *Cladophora glomerata* dominated growths at site EF on 12/3/84 had a WSBR of 0.26 O_2gAFDW^{-1}.d^{-1} (Biomass = 508 mg chla.m^{-2}; T = 21.2°C) compared to values of approximately 0.08 gO_2.gAFDW^{-1}.d^{-1} for the biomass in the control channel (Experiment A, Fig 5.21) at similar biomass levels. This probably results from the lower light input to the channel epilithon causing lower respiration rate as discussed earlier. The daily input of PAR to the channel epilithon was 2.8 E.m^{-2}.d^{-1} whereas the estimated average input over the week prior to 12/3/84 at the depth of sample collection at site EF (0.4 m) was 18 E.m^{-2}.d^{-1} (calculated from total radiation data collected at Tiritea, 13km west of Site EF(assuming that PAR (uE.m^{-2}.s^{-1}) = 2.08 x TR (W.m^{-2}) and that the vertical extinction coefficient k_e = 1.2 m^{-1}).

These results emphasise the importance of light input in determining the effect of epilithon on stream oxygen dynamics. The results also imply that the channel respiration and gross photosynthetic oxygen production data for the phototrophic communities cannot be used to predict the effects of the epilithon on oxygen dynamics in the Manawatu River, although the data may be applicable to shaded stream systems. Nevertheless, assuming that the increased epilithon respiration in response to wastewater addition results from increased heterotrophic activity (Section 5.5.3), the channel results can be used to predict the increase in heterotrophic respiration at the river equivalent site in response to discharges.

5.5.5 INVESTIGATIONS OF THE EFFECTS OF TURBULENCE ON PHOTOTROPHIC CHANNEL COMMUNITIES

The epilithon in the control channel fed borewater (Expt A) consisted of a cohesive mat dominated by the cyanobacterium *Oscillatoria* sp. , diatoms and the filamentous green algae *Stigeoclonium* sp. However in the turbulent zones, within 0.30 m of the recycle pump inflows, a filamentous algal community dominated by *Stigeoclonium* sp. occurred. Blue green algae and diatoms were less frequent here and the algal filaments were able to move independently in the current.

The respiration and gross photosynthetic rates of these communities were measured on day 44 of experiment A and are compared in Table 5.25.

The results indicate that for similar total biomasses the filamentous algal community from the turbulent zone had a greater weight specific respiration rate than the cohesive mat community when measured in the Freeman chamber under identical conditions (Section 3.7.10). The former community also has a greater weight specific gross photosynthesis rate despite its lower chlorophyll a concentration and its chlorophyll a specific gross photosynthetic rate was 2.55 times that of the mat community.

The increased activity of the filamentous community could be explained by the following factors:

(i) differences in the metabolic activity of the species in the two communities as adaptations to allow exploitation of different habitats.

(ii) better diffusion of oxygen and nutrients to the discrete algal filaments than to the cells in the cohesive mat.

(iii) lateral movements of the upper filaments of the filamentous algal community allowing more light to penetrate to the lower cells of the assemblage than in the cohesive mat community thus increasing the efficiency of utilisation of the incident light energy.

5.5.6 CONCLUSIONS

These experiments show that MCDC untreated dairy factory wastewater produces approximately two to three times the increase in heterotrophic respiration in the epilithon for a given COD or BOD_5 addition to that observed for BCWS primary-treated meatworks wastewater. This is apparently due to the MCDC wastewater containing about twice as much dissolved organic material and about three times as much low molecular

TABLE 5.25 Comparison of Respiration and Gross Photosynthesis of Epilithon from the Laminar and Turbulent Flow Zones of the Channels (Experiment A, day 44)

Parameters	Laminar Flow Zone	Turbulent Flow Zone
Growth Type	Cohesive mat	Filamentous algae
Current Velocity ($m.s^{-1}$)	0.2	.27-.47
Biomass		
(i) gAFDW.m^{-2}	38.5	41.8
(ii) Settled Volume ($ml.cm^{-2}$)	0.19	0.24
(iii) chlorophyll a ($mg.m^{-2}$)	849	533
(iv) phaeophytin a ($mg.m^{-2}$)	451	445
(v) AI	45.3	78.4
Weight Specific Respiration ($gO_2.gAFDW^{-1}.d^{-1}$)	.089	.113
Weight Specific BGPR ($gO_2.gAFDW^{-1}.d^{-1}$)*	.084	.125
Chlorophyll a Specific BGPR ($gO_2.gchla^{-1}.d^{-1}$)	3.83	9.8

* calculated for 12 hours daylight.day^{-1} at 45 $uE.m^{-2}.s^{-1}$ PAR.

weight organic material as the primary treated BCWS wastewater for a given COD.

In addition to these increases in heterotrophic respiration the nutrients in the wastewaters may have further effects on the river dissolved oxygen dynamics by increasing the phototrophic communities addition of oxygen during the day and removal at night. Unfortunately the high background nutrient levels in the borewater and low light conditions in the channel experiments compared to the river situation prevent the use of the results for prediction of the effects of the discharges on the development of the phototrophic epilithon in the Manawatu River and consequent effects of this component of the community on dissolved oxygen dynamics. However the data may be useful for predicting the effects of algae on oxygen dynamics in shaded stream systems.

Differences in the overall morphology of phototrophic epilithon communities due to differences in turbulence were shown to have marked effects of their photosynthetic respiratory activity. Communities composed of discrete algal filaments from the turbulent zones of the control channel (fed borewater only) were more metabolically active than cohesive mat communities composed of blue green algae, filamentous green algae, and diatoms from the laminar flow region of the channel.

CHAPTER 6.
EFFECTS OF BENTHIC COMMUNITIES ON STREAM SELF-PURIFICATION AND SUSPENDED BIOMASS PRODUCTION

6.1 INTRODUCTION

In the previous chapter it was shown that both heterotrophic and phototrophic benthic communities can reduce river water quality by depleting the oxygen concentration. The growth of these communities also purifies the river water, by removing dissolved organics and nutrients (Section 2.3.6), and produces suspended biomass when the benthic growths slough.

Self-purification is generally viewed positively since it reduces the potential of the water to support further problematic benthic communities downstream but the production of suspended biomass reduces the value of the river to the public in general and fishermen in particular by:

(i) reducing the aesthetic value due to the presence of "unnatural" suspended material.
(ii) reducing the water clarity.
(iii) clogging irrigation and water supply intakes.
(iv) clogging anglers' lines and fishing nets, especially those used to catch whitebait.
(v) producing a secondary oxygen demand when the biomass settles in areas of low current velocity and is degraded.

This chapter presents the results of studies on the removal rates of organic material (BOD_5, $fBOD_5$) and dissolved nutrients (TDN and TDP) measured during the oxygen dynamics experiments using *in situ* chamber and two station techniques (Sections 5.3 and 5.4 respectively) and during the laboratory channel experiments (Sections 5.5 and 7.3).

The effect of benthic communities on the levels of suspended coarse particulate organic material (CPOM) was studied by filtering approximately 1 m^3 of river water through a coarse mesh net (Section 3.2.2.3) at sites with different benthic communities under a range of conditions.

6.2 EFFECTS OF BENTHIC COMMUNITIES ON STREAM SELF-PURIFICATION

6.2.1 IN SITU CHAMBER STUDIES IN THE MANAWATU RIVER

6.2.1.1 Introduction

The rates of removal of $fBOD_5$, TDN and TDP were measured during a number of studies of the effects of benthic communities on oxygen dynamics using the Freeman *in situ* chambers (Section 5.3) . A range of communities from phototrophic to heterotrophic dominated were studied. In most of the experiments the nutrient removal rates were compared concurrently using two identical chambers, one containing untreated river water and the other river water with sufficient MCDC effluent added to increase the BOD_5 by approximately 5 $g.m^{-3}$ (Section 3.6.3.5).

6.2.1.2 Dissolved Organic Material Removal Rates

The $fBOD_5$ removal rates ($fBOD_5R$) calculated (Section 3.6.3.5) from the results of the *in situ* chamber experiments are presented in Table 6.1. These show that the highest rates were recorded for heterotrophic sewage fungus communities at site C at high organic concentrations. Figures 6.1 and 6.2 confirm that the $fBOD_5R$ is influenced strongly by the initial $fBOD_5$ and the AI values.

At similar initial $fBOD_5$ values, there is a general reduction in $fBOD_5$ removal rate with decreasing AI values (i.e., decreasing heterotrophic content of the communities) (Fig 6.1). Also communities with similar AI values have greater $fBOD_5$ removal rates at elevated initial $fBOD_5$ concentrations (Table 6.1 and Fig 6.2). However a high $fBOD_5$ removal rate was observed for a *C. glomerata* dominated community with only a very light growth of filamentous bacteria attached (Site DE, 2/2/84) under conditions of high temperature and initial $fBOD_5$ (Table 6.1).

Statistical analysis of the data showed that the $fBOD_5$ removal rate was significantly correlated with the initial $fBOD_5$

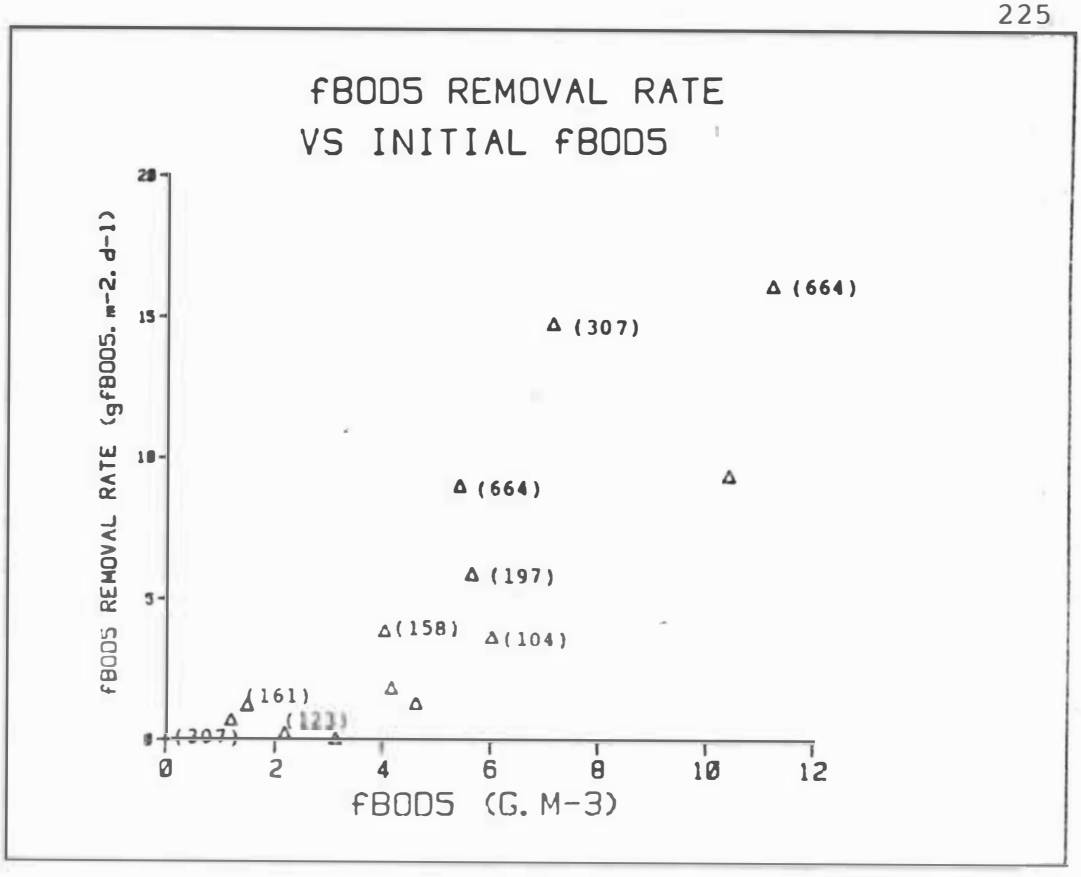


Figure 6.1: Graph of $fBOD_5$ Removal Rate Versus Initial $fBOD_5$.(bracketed values= chamber epilithon AI)

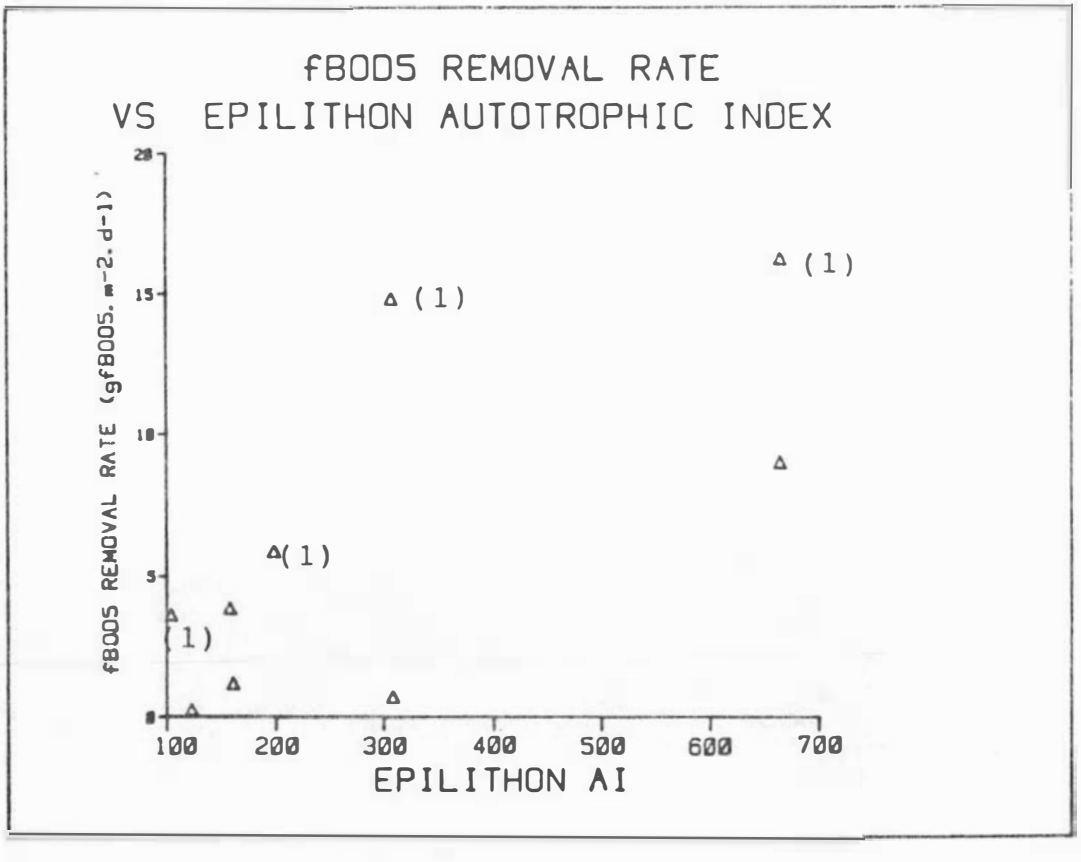


Figure 6.2: Graph of $fBOD_5$ Removal Rate Versus Chamber Epilithon AI ((1)= samples to which extra MCDC effluent added).

TABLE 6.1: Results of *in situ* Chamber Studies of Epilithon Effects on $fBOD_5$ Removal Rates

Date	Site	Temp (°C)	Biomass ($gAFDW.m^{-2}$)	AI	Macroscopic Assessment	$fBOD_5$ ($g.m^{-3}$) Initial ($\bar{x} \pm s$)	$fBOD_5$ ($g.m^{-3}$) Final ($\bar{x} \pm s$)	$fBOD_5$ Removal ($g.m^{-2}.d^{-1}$)	Average respiration ($gO_2.m^{-2}.d^{-1}$)	Specific $fBOD_5$ Removal ($g.gAFDW^{-1}.d^{-1}$)	Calculated Chamber k_1 (hr^{-1})	Predicted Manawatu R* k_1 (hr^{-1})
12/01/84	C	20.8	49.8	664	6IC	11.2± 0.4	2.0 ±0.25	16.17	16.8	0.325	0.990	0.124
12/01/84	C	20.2	54.6	664	6IC	5.4 ± 0.5	2.1 ± 0.4	9.0	20.1	0.165	0.810	0.101
23/03/84	C	17.5	24.0	307	5ID	7.12±0.22	2.25±0.34	14.8	10.3	0.615	1.19	0.149
23/03/84	C	17.4	24.0	307	5ID	1.2 ± 0.2	1.5 ±0.21	0.71	7.6	- 0.03	-0.16	- 0.02
26/04/84	C	15.0	92.6	158	4ID	4.02±0.26	2.04±0.45	3.88	17.0	0.042	0.43	0.054
26/04/84	C	15.8	04.4	197	4II	5.63±0.21	1.7 ±0.36	5.88	17.5	0.056	0.654	0.082
08/12/83	D	21.0	29.6	161	4IC	1.46±0.07	1.05±0.16	1.21	12.5	0.041	0.227	0.028
24/11/83	D	16.0	8.5	ND	3IIC	4.6 ±0.35	3.9 ± 0.1	1.31	5.2	0.154	0.097	0.012
14/11/83	D	15.0	11.6	ND	3IIC	4.15±0.07	3.3 ± 0.4	1.83	4.9	0.159	0.129	0.016
18/01/84	DE	18.0	50.0	104	1IIF	6.0 ±0.75	5.17±0.32	3.64	13.4	0.07	0.33	0.041
18/01/84	DE	17.7	50.6	123	1IIF	2.18	2.1 ±0.08	0.25	13.6	0.005	0.038	0.005
02/02/84	DE	24.0	48.7	ND	2IIF	10.4±0.13	7.53±0.14	9.44	22.8	0.194	0.310	0.039
02/02/84	DE	23.2	45.8	ND	2OOF	3.13±0.03	3.4 ± 0.2	0.06	23.1	- 0.001	-0.07	- 0.001

*= where Predicted Manawatu $k_1 = \frac{\text{Calculated chamber } k_1}{8}$

($r = 0.869$; $P < 0.001$) and AI ($r = 0.719$; $P < 0.05$). Similarly the exponential $fBOD_5$ removal rate (Section 2.3.6 (k_1) was also significantly correlated with the initial $fBOD_5$ ($r = 0.636$; $P < 0.05$) and AI values ($r = 0.606$; $P < 0.01$). However since the initial $fBOD_5$ and AI values were themselves correlated ($r = 0.589$; $P < 0.05$), multiple regression could not be used to model the effects of these two predictors on $fBOD_5$ removal rate or k_1.

The initial dissolved oxygen concentration had no statistically significant effect on the $fBOD_5$ removal rate ($r = -0.469$; $P > 0.05$) and, as might be expected for a data set containing such a range of community types, no significant correlation existed between the $fBOD_5$ removal rate and the total biomass (AFDW) ($r = 0.014$; $P > 0.1$) or temperature ($r = 0.126$; $P > 0.1$).

Comparison of the $fBOD_5$ removal rates, measured for the duration of the runs, and the average dark respiration rates (BR), measured during the first half of the runs, showed that, with one exception, the $fBOD_5$ removal rate was less than the BR (Table 6.1). This implies that the oxidation of organic material previously uptaken and/or produced by photosynthesis are important in the respiration of these communities. The difference between the two rates was greatest under low $fBOD_5$ conditions. The one observation of a $fBOD_5$ removal rate exceeding the respiration rate was made at site C on 23/3/84 when extra $fBOD_5$ was added to a sewage fungus community experiencing low $fBOD_5$ conditions (background $fBOD_5 = 1.2$ $g.m^{-3}$). This was also the one experiment in which adding organic material was observed to increase the respiration rate over that observed under ambient water quality conditions (Section 5.3). This suggests that under the conditions prior to extra organic waste addition the organic substrates available to the heterotrophic organisms were depleted to such an extent that the respiration rate was reduced. Once the organic supply was replenished, uptake and metabolism increased rapidly.

The different bed surface area to water column volume ratios in the chambers (8 $m^2.m^{-3}$) and in the river (1 $m^2.m^{-3}$) creates difficulties in comparing data obtained from the *in situ* chambers and two station studies since at higher bed surface area to volume ratios the volumetric removal rate is more rapid for a given benthic removal rate (as g $fBOD_5.m^{-2}.d^{-1}$). Since the results show that the $fBOD_5$ removal rate is influenced by the $fBOD_5$ concentration in the water this would result in lower rates being measured in chambers when relatively long incubation intervals, such as those used in these studies (1 to 1.8 hours), are employed. This effect would be most pronounced in experiments in which large changes in $fBOD_5$ concentration occurred, such as the experiments in which extra $fBOD_5$ was added to sewage fungus communities.

This implies that the *in situ* chamber results for linear $fBOD_5$ removal rates should not be used directly to formulate predictive models based on linear regression methods and that chamber studies of self-purification should utilise incubation periods of the minimum length required to obtain a significant drop in organic levels. The data in Table 6.1 imply that for phototroph dominated communities incubation periods such as those used in this study (1 to 1.8 hours) are appropriate for chambers of similar bed surface area to water column volume ratios. However incubation periods of 0.25 to 0.5 hours would be more appropriate for studies of heterotrophic communities at moderate to high organic concentrations.

Since the k_1 calculation assumes that organic removal is dependent upon the organic concentration in the water column, the effect of the more rapid organic depletion on these values is expected to be less pronounced than for the $fBOD_5$ removal rates. However the difference between the chamber and river surface area to volume ratios results in much greater k_1 values being recorded in the chambers than in the river for similar conditions (Section 6.2.2.1). This latter effect can be circumvented to allow comparison of the chamber and river k_1 values by multiplying the chamber values by the quotient of the river surface area to volume ratio divided by the chamber ratio.

6.2.1.3 Dissolved Nitrogen and Phosphorus Removal

The rates of removal of total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP) calculated (Section 3.6.3.5) from the results of the *in situ* chamber studies are presented in Table 6.2 and 6.3 respectively.

The data indicate that the removal of TDN is related to the initial TDN concentration in the water (Fig 6.3) ($r = 0.862$; $P < .01$). High TDN removal rates were recorded for both algal and sewage fungus dominated communities under conditions of high initial TDN (Table 6.2). The small size of the data set prevents further detailed analysis of the factors affecting TDN removal rates. However comparison of the TDN removal rates of *Cladophora glomerata* and *Potamogeton crispus* at site EF on 12/3/84 (Table 6.2) indicates that the more metabolically active algal species (*C. glomerata*) (Section 5.3.4) also has a higher TDN removal rate.

The TDP removal rates varied from 0 to 0.123 g $TDP.m^{-2}.d^{-1}$ except on one occasion (2/2/84, site DE) when a rate of 0.66 g $TDP.m^{-2}.d^{-1}$ was observed (Table 6.3). Although this rate is substantially greater than those observed on other occasions, it was recorded under conditions of high light (800 $uE.m^{-2}.s^{-1}$PAR), temperature and initial TDP, when very high respiration and gross photosynthesis rates were recorded (28.2 and 36.1 $gO_2.m^{-2}.d^{-1}$ respectively). Since high nutrient removal rates would be expected under these conditions the value recorded is probably a true result rather than a sampling or analytical artifact.

Comparison of the biomass specific TDP removal of *C. glomerata* and *P. crispus* at site EF on 12/3/84 shows that the rates were similar despite the greater metabolic activity and TDN removal of the algae, *C. glomerata*.

TABLE 6.2: Results of *in situ* Chamber Studies of Nitrogen Removal in the Manawatu River

Date	Site	Temperature (°C)	Biomass ($gAFDW.m^{-2}$)	AI	Initial TDN ($g.m^{-3}$)	TDN removal rate ($gTDN.m^{-2}.d^{-1}$)	Specific TDN removal rate ($gTDN.gAFDW^{-1}.d^{-1}$)	Macroscopic Community Assessment
12/01/84	C	20.8	49.8	664	0.601	0.4910	0.00900	6IC, sewage fungus
12/01/84	C	20.2	54.6	664	0.321	0.2660	0.00534	6IC, sewage fungus
18/01/84	DE	18.0	50.0	100	0.720	ND	ND	1IIF, *C. glomerata*
02/02/84	DE	24.0	48.7	*	1.020	1.0520	0.02160	2IF, *C. glomerata*
23/03/84	C	17.5	24.0	307	0.793	1.0370	0.04170	5ID, sewage fungus
23/03/84	C	17.4	24.0	307	0.412	0.0970	0.00400	5ID, sewage fungus
26/04/84	C	15.0	92.6	158	0.438	-0.0960	-0.00100	4ID, sewage fungus
12/03/84	EF	20.0	31.0	68	0.505	0.0878	0.00283	0IE, *P. crispus*
12/03/84	EF	21.0	55.0	109	0.642	0.2610	0.00470	0IE, *C. glomerata*

TABLE 6.3: Results of *in situ* Chamber Studies of Phosphorus Removal in the Manawatu River

Date	Site	Temperature (°C)	Biomass ($gAFDW.m^{-2}$)	AI	Initial TDP ($g.m^{-3}$)	TDP removal rate ($gTDP.m^{-2}.d^{-1}$)	Specific TDP removal rate ($gTDP\ gAFDW^{-1}.d^{-1}$)	Macroscopic Community Assessment
12/01/84	C	20.8	49.8	664	0.379	0.0955	0.00175	6IC, sewage fungus
12/01/84	C	20.2	54.6	664	0.195	0.0	0.0	6IC, sewage fungus
18/01/84	DE	18.0	50.0	100	0.194	0.0	0.0	1IIF, *C. glomerata*
18/01/84	DE	17.7	50.6	100	0.065	0.0	0.0	1IIF, *C. glomerata*
02/02/84	DE	24.0	48.7	ND	0.556	0.6600	0.01350	2IF, *C. glomerata*
23/03/84	C	17.5	24.0	307	0.227	0.0864	0.00360	5ID, sewage fungus
26/04/84	C	15.0	92.0	158	0.062	0.0060	0.00006	4ID, sewage fungus
26/04/84	C	15.8	104.4	197	0.158	0.0622	0.00059	4ID, sewage fungus
12/03/84	EF	20.0	31.0	68	0.105	0.0650	0.00210	0IE, *P. crispus*
12/03/84	EF	21.0	55.0	109	0.081	0.1230	0.00220	0IE, *C. glomerata*

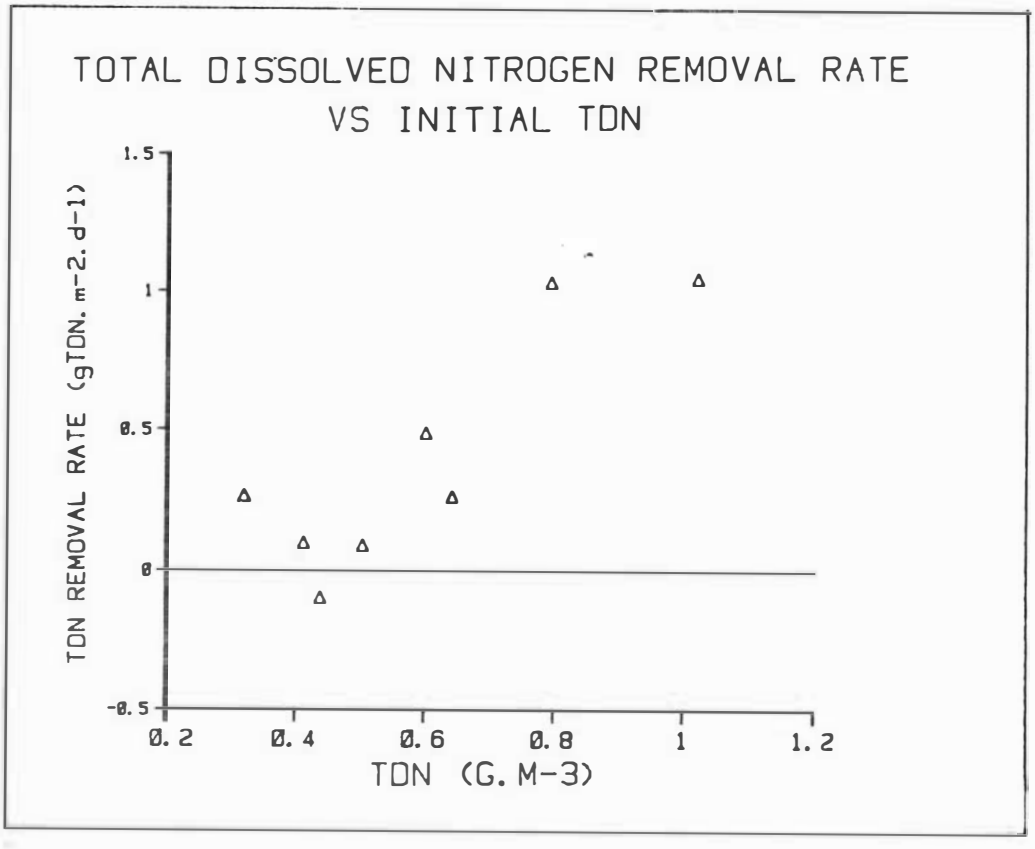


FIGURE 6.3: Graph of Total Dissolved Nitrogen Removal Rate versus Initial Total Dissolved Nitrogen Concentration.

6.2.2 TWO STATION SELF-PURIFICATION STUDIES IN THE MANAWATU RIVER

6.2.2.1 Organic Self-purification

The results of the two station studies of organic self-purification in the Manawatu River, presented in Table 6.4 and 6.5, show similar trends to those observed *in situ* chamber studies (Section 6.2.1.2). The organic removal rates were greatest over the reaches containing heavy sewage fungus growths and in which the initial organic material concentrations were high. Under these conditions $fBOD_5$ removal rates of 35.3 to 44.3 $g.m^{-2}.d^{-1}$ were recorded and oxidation of the BOD_5 removed almost matched the respiration rate over the reach from the MCDC outfall to site CuC. By contrast, at lower initial organic concentrations and over reaches where the macroscopic abundance of heterotrophs was low, the organic removal rates were lower and the oxidation of the BOD_5 removed accounted for less than 25% of the reach respiration. Similar low values were recorded for reaches with light heterotrophic growth at low temperature (D-E 22/4/82). Intermediate organic removal rates were observed for the phototroph dominated reach Dd-EF on 31/1-1/2/84 for a moderate initial organic concentration.

Oxidation of the BOD_5 removed in the reach from the PNCC outfall to site B only accounted for 25% of the reach respiration (Table 6.4). The aerated lagoon secondary treatment system, which commenced operation on 5/3/85, is expected to reduce the PNCC effluent BOD_5 load to the river by 95% (Section 2.2.3.4). This removal is expected to reduce the reach respiration rate by approximately 23% under similar summer low flow conditions in future. No reduction in phototroph respiration is expected in response to the secondary treatment since the phosphorus removal, of up to 30% (Section 2.2.3.4) will not reduce the in-river phosphorus concentration to the limiting level (Sections 2.2.1.2 and 2.2.3.4). Nevertheless the reduction in heterotroph respiration should reduce the reach respiration under summer low flow conditions

TABLE 6.4: Results of Two Station Studies of BOD_5 Removal in the Manawatu River

Parameters	Values							
Reach	D-E	MCDC-CuC	D-E	Dd-EF	Dd-EF	Dd-EF	PNCC-B	
Biomass:								
(i) Macroscopic Assessment	6IC	6ID	31D	0-1IF	1-2IF	102IF	1-2IF	
(ii) $gAFDW.m^{-2}$	ND	150	ND	130	150	130	100	
Study Dates	25/3/82	29/2-1/3/84	22/4/82	24-25/1/84	31/1-1/2/84	9-10/2/84	29/2-1/3/84	
Average River Flow ($m^{-3}.s^{-1}$)	17.7	16.8	19.1	14.8	14.8	18.3	16.8	
Average River Temperature (°C)	18	20.7	13.2	19.7	18.2	19.7	21	
BOD_5 ($g.m^{-3}$)								
(i) Initial ($x \pm s$)	7.47*	$9.84 \pm .88$	5.37**	$5.23 \pm .13$	$6.4 \pm .46$	$2.84 \pm .13$	4.04***	3.73****
(ii) Final ($x \pm s$)	4.95*	$7.7 \pm .06$	4.89**	$4.28 \pm .06$	$3.7 \pm .01$	$2.18 \pm .12$	$3.07 \pm .52$	$2.86 \pm .12$
BOD_5 removal								
(i) k_1 (hr^{-1})	0.194	0.157	0.05	0.036	0.10	0.054	0.071	.070
(ii) $g.m^{-2}.d^{-1}$	25.6	36.9	5.8	4.2	11.9	3.4	6.74	6.04
Reach respiration ($gO_2.m^{-2}.d^{-1}$)	ND	39.8	ND	32.5	37.6	32.4	26.1	
BOD_5 removal as % of reach respiration	ND	93	ND	13	31.6	10.5	25	

* Mean of 4 hourly composite samples for 900-1300 hrs (D) and 11-1500 hrs (E)

** Mean of 3 grab samples for 1312-1244 hrs (D) and 1503-1535 hrs (E)

*** BOD_5 at A + BOD_5 added by PNCC 730-2130 hr 29/2/84 (Anderson, 1984)

**** BOD_5 at A + BOD_5 added by PNCC 2030-430 hr 29/2-11/3/84 (Anderson, 1984)

TABLE 6.5: Results of Two Station Studies of $fBOD_5$ Removal in the Manawatu River

Parameters	Values				
Reach	D-E	MCDC-CuC	Dd-EF	Dd-EF	Dd-EF
Biomass:					
(i) Macroscopic Assessment	6IC	6ID	0-1IF	1-2IF	1-2IF
(ii) $gAFDW.m^{-2}$	ND	150	130	150	130
Study Dates	1-2/2/83	29/2-1/3/84	24-25/1/84	31/1-1/2/84	9-10/2/84
Average River Flow ($m^{-3}.s^{-1}$)	27.3	16.8	14.8	14.8	18.3
Average River Temperature (°C)	18	20.7	19.7	18.2	19.7
$fBOD_5$					
(i) Initial ($x \pm s$)	$7.28 \pm .39$	$7.77 \pm .31$	2.53 ± 0.11	$4.26 \pm .08$	$1.87 \pm .13$
(ii) Final ($x \pm s$)	$4.53 \pm .04$	$5.8 \pm .14$	1.65 ± 0.56	$2.66 \pm .10$	$1.3 \pm .10$
$fBOD_5$ removal					
(i) k_1 (hr^{-1})	0.393	0.195	0.078	0.086	0.075
(ii) $g^{fBOD_5}.m^{-2}.d^{-1}$	44.3	35.3	3.9	7.7	3.0
Reach respiration ($gO_2.m^{-2}.d^{-1}$)	ND	39.8	32.5	37.6	32.4

to approximately the 20 $gO_2.m^{-3}.d^{-1}$ level, which the modelling studies predict to be equal to the maximum acceptable summer low-flow respiration rate (Section 5.4.7).

By contrast removal of the oxygen demand due to oxidation of the BOD_5 removed in the phototroph dominated reach Dd to EF would only reduce the respiration rate to 26 to 29 $gO_2.m^{-3}.d^{-1}$. This implies that unless the phototrophic activity below the discharges is limited, unacceptable oxygen depletion is likely to occur in the river during summer low flow conditions after the water rights limiting BOD_5 discharge have been met.

Due to the high biomass levels in the river two station studies, few data points overlap with those in the *in situ* chamber studies (Table 6.1) to allow comparison of the two data sets. However the two station $fBOD_5$ removal rates and k_1 values (Table 6.5) were approximately twice those recorded in the *in situ* chambers (Table 6.1) for similar community types and initial $fBOD_5$ concentrations. This probably results from the combined effects of the higher benthic biomass in the two station studies and the effect of the different bed surface area to water column volume ratios on the kinetics of organic material removal (Section 6.2.1.2).

The BOD_5 k_1 values recorded for the reaches containing heavy sewage fungus growth were in a similar range to those recorded in previous studies over such reaches in the Manawatu River by Currie (1977) (k_1 = 0.20.hr^{-1}, reach D-E) and Currie and Rutherford (1981) (k_1 = 0.43.hr^{-1}, reach C-D). The BOD_5 k_1 values recorded for reach Dd-EF during early 1984, when phototrophs were dominant (Table 6.4), were similar to the values obtained for the phototroph dominated reach from E to F by Currie (1977) (k_1 = 0.06.hr^{-1}, 16/12/75; k_1 = 0.04.hr^{-1}, 12/2/77).

6.2.2.2 Dissolved Nitrogen and Phosphorus Removal

The removal of total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP) was investigated during three studies over the reaches Dd to EF and D to E. The TDN removal

rates (Table 6.6) for the reach Dd to EF were within the range of values recorded in the *in situ* chamber experiments (Section 6.2.1.3) but a higher value was recorded for the reach D to E on 2/2/83, when a heavy sewage fungus biomass was present. High removal rates for $fBOD_5$ (Section 6.2.2.1) and TDP (Table 6.6) were recorded simultaneously over this reach indicating that the community was growing rapidly.

An anomalous increase in TDP was recorded for the reach Dd to EF on 24-25/1/84 and no nett TDP removal occurred on 31/1-1/2/84 (Table 6.6). These results could be explained by the low oxygen conditions during the night on these occasions (Table 6.6). These apparently resulted in anaerobic conditions within the river bed gravels since black sulphide deposits were observed on the undersides of stones and gas bubbles (probably methane) were observed rising from the bed at site Dd at 8 am on 1/2/84. Under anaerobic conditions the release into the water column of phosphorus, previously bound in sediments, would be expected (Wetzel, 1975; Swyngedouw *et al.*, 1984).

6.2.3 LABORATORY CHANNEL STUDIES

6.2.3.1 Introduction

The analysis of samples of the channels' influent and effluent flows collected at about weekly intervals provided information on the effects on self-purification of epilithon developing under a range of organic loading conditions as well as providing a check on the channel water quality conditions. However, since the biomass and activity measurements focussed on the biomass adhering to plates incubated from the beginning of the experiment (t_o) rather than the total channel biomass, the biomass specific self-purification data obtained is semi-quantitative except on the first biomass and water sampling occasion. On the later occasions the biomass data obtained from the plates incubated from t_o overestimated the total channel biomass since approximately 5% of the total channel biomass was removed on each sampling day.

Nevertheless the organic self-purification data obtained during the experiments provide useful information and insights into

TABLE 6.6: Results of Nutrient Removal Studies in the Manawatu River (Fig 1.2)

Parameters	Values		
Study Date	24-25/1/84	31/1-1/2/84	1-2/2/83
Reach	Dd-EF	Dd-EF	D-E
River flow ($m^3.s^{-1}$)	14.8	14.8	27.3
Average Temperature (°C)	19.7	18.2	18
Dissolved Oxygen Range ($g.m^{-3}$)	1.25-14.8	2.1-12	5-9*
Biomass			
(i) gAFDW.m^{-2}	130	150	ND
(ii) Macroscopic Assessment	0-1IIF	1-2IIF	6IE
Initial TDN ($g.m^{-3}$) ($x \pm s$)	0.631 ± .001	0.884 ± .039	0.511 ± .019
Final TDN ($g.m^{-3}$) ($x \pm s$)	0.380 ± .007	0.723 ± .005	0.370 ± .021
TDN Removal Rates			
($gTDN.m^{-3}.d^{-1}$)	1.097	0.704	2.707
($gTDN.m^{-2}.d^{-1}$)	1.108	0.711	2.274
Initial TDP ($g.m^{-3}$) ($x \pm s$)	0.102 ± .001	0.142 ± .002	0.160 ± .001
Final TDP ($g.m^{-3}$) ($x \pm s$)	0.131 ± .001	0.145 ± .011	0.143 ± .006
TDP Removal Rates			
($gTDP.m^{-3}.d^{-1}$)	-0.127	Not significant	0.326
($gTDP.m^{-2}.d^{-1}$)	-0.128	Not significant	0.274

* Measured at Site E on 24-25/1/83

the interactions between organic waste addition, epilithon development and organic self-purification.

The removal of total dissolved nitrogen (TDN) and phosphorus (TDP) were also investigated on eight occasions.

6.2.3.2 Organic Material Removal During the Laboratory Channel Experiments

A generally similar pattern of removal of organic material was observed during each of the seven experiments in which wastewater addition occurred. Regardless of the channel influent $fBOD_5$ concentration, in the range 1.5 to 6.9 $g.m^{-3}$, the effluent $fBOD_5$ was between 0.5 and 2.0 $g.m^{-3}$ for most of the time during each experiment (Figs 7.21 to 7.27). Since the $fBOD_5$ of the effluent control channel (fed borewater) was 0.11 to 0.59 $g.m^{-3}$, this implies that most of the added dissolved organic material was removed by the channel biomass in each experiment.

The organic material removal during experiments C and G are presented in Figures 7.22 and 7.26 respectively. These show the typical pattern of organic removal during the experiments in which the channels were fed MCDC and BCWS wastewaters respectively (Figs 7.21-7.27).

In experiment C (fig 7.22) significant removal of $fBOD_5$ occurred on day 1 (3.5 g $fBOD_5$ $m^{-2}.d^{-1}$) when a thin heterotrophic film dominated by the filamentous bacteria *S. natans* and *Flavobacterium* sp. was present (biomass = 1.6 g AFDW.m^{-2}; settled volume = 0.035 ml.cm^{-2}; AI = 5400). The $fBOD_5$ removal rate was then fairly constant at approximately 7 g $fBOD_5$ $m^{-2}.d^{-1}$, from day 8 (biomass = 12 $\pm$ 2 g AFDW.m^{-2}; settled volume = 0.100 ml.cm^{-2}; AI = 266 $\pm$ 67) but declined at the time of biomass sloughing (after day 27).

A similar variation of $fBOD_5$ removal was observed in experiment G (fig 7.26) although no significant removal was observed on day 2 in this instance. The removal rate was similar from day 9 to 31 when the biomass on the plates incubated from day 0 varied from 19 to 60 g AFDW.m^{-2}.

The particulate BOD_5 in the channel effluent (i.e., the difference between the BOD_5 and $fBOD_5$) represents the sum of the particulate organic material entering the channel in the influent which does not settle out in the channel plus the biomass produced in the channel which is lost to the water column. These values were low at the beginning of both experiments C and G when biomass sloughing losses would be expected to be low. This indicates that most of the influent particulate BOD_5 settled out in the channels. However as the experiments proceeded the particulate BOD_5 in the channels' effluent increased indicating sloughing of biomass produced. The values were maximal at the time of maximum sloughing.

This shows that although the heterotrophic epilithic organisms reduce the concentration of dissolved organic material in water column they also add increasing amounts of particulate material as the biomass develops.

The maximum $fBOD_5$ removal rates observed in each experiment are presented in Table 6.7. These values are similar to those recorded in the *in situ* chamber studies in the Manawatu River at similar initial $fBOD_5$ concentrations (Table 6.1). The results show that the maximum removal rate increases with increasing organic loading to the channels. However the highest removal rates were frequently observed at low biomass concentrations especially at the lower influent loading rates.

6.2.3.3 Nutrient Removal Rates During the Laboratory Channel Experiments

The results of the nutrient analyses undertaken during the channel experiments are presented in Table 6.8 along with the nutrient removal rates calculated from these data.

These show that the high background nutrient levels in the borewater were further increased by the addition of the organic wastewaters. The nutrient removal rates were generally at the lower end of the range of values observed in the *in situ* chamber experiments in the Manawatu River (Section 6.2.1.3). No significant relationships between nutrient removal rates and other factors were observed.

TABLE 6.7: Summary of Operating Conditions for the Laboratory Channel Experiments and the Maximum $fBOD_5$ Removal Rates Observed.

Experiment	B	C	D	E	F	G	H
Wastewater	MCDC	MCDC	MCDC	BCWS	BCWS	BCWS	BCWS
Average influent Wastewater Concentration ($gCOD.m^{-3}$)	1.9 ± 0.1	3.8 ± 0.6	6.5 ± 1.0	1.9 ± 0.25	4.2 ± 0.4	7.3 ± 0.9	14.4 ± 0.7
Maximum $fBOD_5$ Removal Rate ($g\ fBOD_5.m^{-2}.d^{-1}$)	2.0	7.0	14.2	2.8	2.8	3.9	5.2
Day of Experiment (d)	10	13	8	6	6	23	14
Influent $fBOD_5$ ($g.m^{-3}$) ($x \pm s$)	1.50 ± 0.31	3.76 ± 0.76	6.2 ± 0.96	1.55 ± 0.2	2.22 ± 0	2.39 ± 0.05	4.0 ± 0.22
Effluent $fBOD_5$ ($g.m^{-3}$) ($x \pm s$)	0.70 ± 0.20	0.93 ± 0.15	0.45 ± 0.03	0.4 ± 0.32	1.07 ± 0.32	0.81 ± 0.11	1.88 ± 0.15
Biomass ($gAFDW.m^{-3}$)	4.6	20	13	1	5	50**	35
Settled Volume ($ml.cm^{-2}$)	0.034	0.10	0.13	0.01	0.023	0.2	0.2
AI	225	300	500	245	ND	150	500

* Near maximal rates were recorded at 11 $gAFDW.m^{-2}$ (Fig 6.4)

** Near maximal rates were recorded at 19 $gAFDW.m^{-2}$ (Fig 6.5)

TABLE 6.8: Results of Nutrient Removal Studies During Laboratory Channel Experiments

Experiment	Day	TDN			TDP			Biomass ($gAFDW.m^{-2}$)	AI
		Influent ($x \pm s$)	Effluent ($x \pm s$)	Removal ($g.m^{-2}.d^{-1}$)	Influent ($g.m^{-3}$)	Effluent ($g.m^{-3}$)	Removal ($g.m^{-2}.d^{-1}$)		
A	8	.717 ± 0	.669 ± .001	.117	.074 ± .001	.067 ± 0	.017	2.8	54
A	22	.942 ± .032	.882 ± 0	.128	.076 ± .002	.090 ± .004	-0.03	31.8	67
A	31	.898 ± 0	.798 ± .023	0.244	.087 ± .004	.076 ± .011	NS	19	51
D	8	1.36 ± .070	1.17 ± .07	0.432	.259 ± .003	.252 ± .052	NS	11	270
D	29	1.60 ± .050	1.66	NS	.233 ± .033	.208 ± .009	NS	50	150
G	15	.883 ± .030	.775 ± .005	.142	.119 ± .001	.116 ± .001	.007	32	151
G	32	ND	ND	ND	.126 ± .007	.126 ± .002	NS	50	150
H	14	1.27 ± .02	1.25 ± .01	NS	.169 ± .005	.150 ± 0	0.046	20	250

The apparent increase in TDP during passage through the channel on day 22 of experiment A is anomalous. Short-term variations in the influent nutrient concentrations could have produced this result but contamination of the effluent sample is a more likely cause.

6.2.4 CONCLUSIONS

The dissolved organic material (as $fBOD_5$) removal rate in the Manawatu River is most strongly influenced by the organic concentration of the water and the heterotrophic content of the benthic community. Total benthic biomass had no statistically significant effect on the removal rate over the range of biomass and organic concentrations investigated in the *in situ* chamber studies.

The total biomass required to achieve the maximum removal rate increased with increasing organic loading but comparatively light benthic growths produced significant organic removal rates at high $fBOD_5$ concentrations in the laboratory channels.

The oxidation of BOD_5 had a minor effect on the respiration rate for the reaches from PNCC outfall to site B and below the waste mixing zone (i.e., reach Dd-EF). The removal of the organic loading to the river would reduce the respiration rate of the reach PNCC-B to below the maximum acceptable level of 20 $gO_2.m^{-3}.d^{-1}$ derived from the modelling studies (Section 5.4.7) but the respiration of the reach Dd-EF would remain above this level during summer lowflows.

The nutrient removal studies using the *in situ* chambers in the river indicate that the dissolved nitrogen removal rate is dependent upon the initial concentration. The phosphorus removal rate was quite variable but a high rate was observed for a metabolically active mixed epilithon under conditions of high temperature and high initial phosphorus concentration. The two station studies in the Manawatu River showed substantial nitrogen removal occurred below the waste discharges during summer low flow conditions during low night-

time oxygen conditions. By contrast the daily average phosphorus levels either increased downstream or remained the same over the reach suggesting that the night-time anoxic conditions in the benthos caused release of phosphorus previously bound in the sediments into the water column.

6.3 EFFECTS OF BENTHIC COMMUNITIES ON SUSPENDED BIOMASS

6.3.1 INTRODUCTION

This section presents the results of studies on the effect on the production of suspended coarse particulate organic material (CPOM) of benthic growths of sewage fungus and other communities within the Manawatu River study reach (Fig 1.2). These involved filtering a measured volume of river water (Section 3.2.2.3), of approximately 1 m^3, through a mesh net with 1.7 mm apertures (i.e., the netting used in whitebait fishing nets).

The nuisance level of suspended CPOM production is difficult to define and would differ for the different impacts. However collection of the samples using a whitebait mesh net allows direct measurements of the nuisance effect on this activity.

Field experiments conducted during the initial phase of a spate had shown that sudden increases in current velocity promote massive sloughing of benthic biomass (Section 4.4.3) and laboratory channel studies had shown that the sloughing losses can increase at the end of the epilithon growth phase (Section 6.2.3.2).

6.3.2 RESULTS OF SAMPLING RUNS

The suspended CPOM concentrations and qualitative benthic biomass data recorded on sampling runs during 1982 to 1983 and 1983 to 1984 are presented in Figures 6.4 and 6.5 respectively.

At sites A and B, where phototrophs dominated the benthic community, the suspended CPOM concentrations were low (less

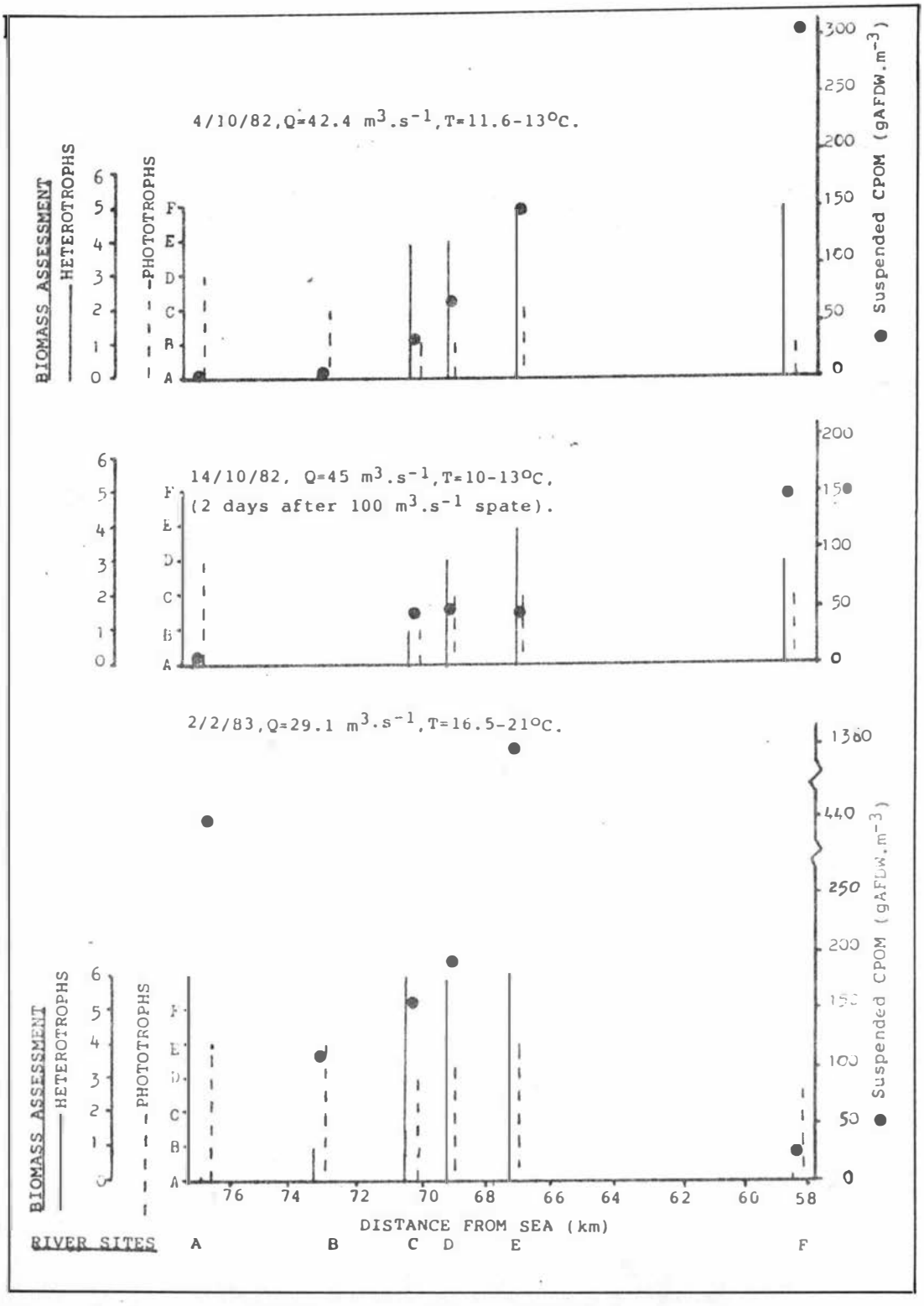


Figure 6.4 : Benthic Biomass and Suspended Coarse Particulate Organic Material (CPOM) in the Manawatu River, October 1982 to February 1983.

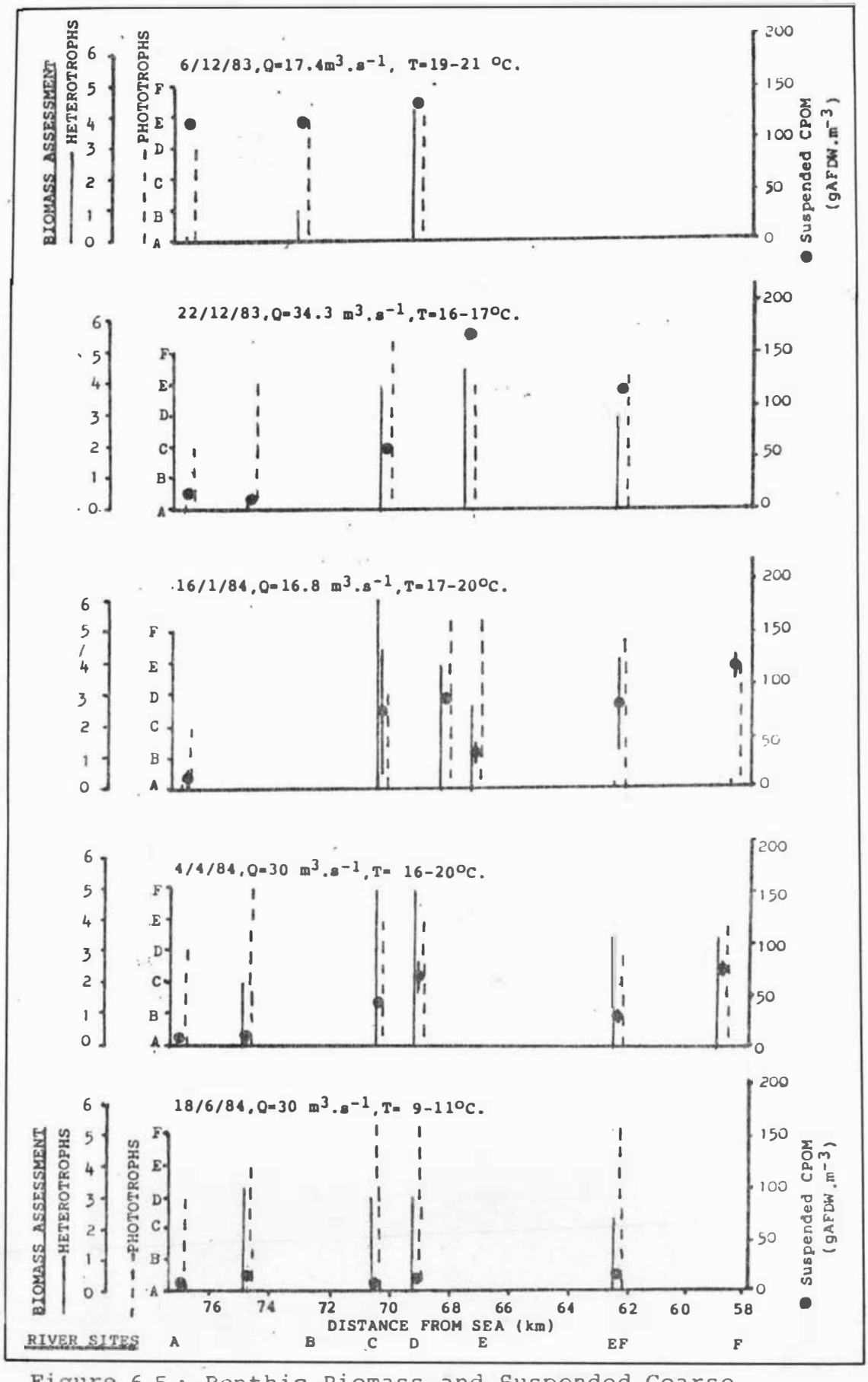


Figure 6.5 : Benthic Biomass and Suspended Coarse Particulate Organic Material (CPOM) in the Manawatu River, December 1983 to June 1984.

than 15 mg AFDW.m^{-3}) during the spring, winter and autumn sampling runs. This probably resulted from the low benthic biomass conditions and, when moderate to high biomass occurred (4/4/84 and 18/6/84), low growth rates expected under the autumn and winter conditions (section 5.3.5). However during summer the suspended CPOM values were similar to or greater than those in the reach containing sewage fungus growths. On 2 February 1983 (Fig 6.4) the high value at site A (440 mg AFDW.m^{-3} CPOM) resulted from the sloughing of a diatom mat, dominated by *Gomphonema* sp ., which had developed over the previous three weeks. At site B this community was replaced by one dominated by the filamentous green algae *Stigeoclonium* sp . which resulted in lower suspended CPOM values. Elevated values were also recorded at site A on 6/12/83 (Fig 6.5) when *C. glomerata* was developing and at site F on 16/1/84 (Fig 6.5) when a diverse phototrophic community occurred.

Suspended CPOM levels within the reaches containing macroscopic sewage fungus growths were quite variable. However when benthic sewage fungus growths were common or covered most or all of the bed at velocities of 0.25 to 0.5 m.s^{-1} (abundance levels 4 to 6), the values were always greater than 35 mg AFDW.m^{-3} and often exceeded 100 mg AFDW.m^{-3}. The highest values at the respective sites were recorded when the sewage fungus covered the river bed.

At a suspended CPOM concentration of 100 g AFDW.m^{-3} the whitebait net mesh catch net became clogged after approximately three minutes at a current velocity of 0.5 m.s^{-1}. This would almost certainly be considered a nuisance level by whitebait fishermen.

The higher suspended CPOM values at site F under similar benthic biomass levels to those at sites C to E on 4/10/82 and 14/10/82 (Fig 6.4) probably resulted from the comparatively long 3 kilometre run above the site. Suspended biomass would tend to remain suspended through this reach due to the lack of quiescent areas (pools) which enhance biomass settling.

6.3.3 CONCLUSIONS

Both phototrophic and sewage fungus dominated benthic communities can produce large increases in suspended CPOM concentrations causing a potential nuisance to river users. Elevated levels were recorded in reaches containing sewage fungus and in phototroph dominated reaches during biomass sloughing and periods when high growth rates would be expected. Low levels were recorded over reaches with low benthic heterotrophic biomass and reaches with moderate to high phototrophic biomass under conditions when low growth rates were expected.

CHAPTER 7.

EFFECTS OF ORGANIC WASTEWATER DISCHARGES AND SOME PHYSICAL FACTORS ON BENTHIC COMMUNITY DEVELOPMENT

7.1 INTRODUCTION

In the previous two chapters it was shown that both photo-trophic and heterotrophic benthic communities can create a nuisance by impinging on human uses of the river and inhibiting the maintenance of a healthy aquatic ecosystem. Sewage fungus development is considered to be particularly troublesome due to the greater detrimental impact of this "unnatural-looking" material on river aesthetics, in addition to its effects on suspended biomass production, and oxygen depletion. Modelling studies have indicated that the benthic biomass concentration resulting in unacceptable night-time deoxygenation decreases from approximately 150 $gAFDW.m^{-2}$ at 12°C to approximately 34 to 45 $gAFDW.m^{-2}$ at 21°C (Section 5.4.7). An acceptable benthic community might therefore be defined as one in which the biomass concentration does not exceed approximately 34 to 45 $gAFDW.m^{-2}$ during summer lowflows and sewage fungus fronds are not macroscopically visible on the riverbed (heterotrophic biomass less than abundance level 3). The results presented in the previous two chapters show that these conditions were not consistently met in the Manawatu River during the study period.

In the studies presented in this chapter the relationships between organic waste concentration and the development of epilithon, in general, and sewage fungus, in particular, were studied by experiments and observations in the Manawatu River study reach and in two identical laboratory channels. These studies involved the following:

(i) Qualitative assessment of the composition and macroscopic abundance of the natural benthic communities at river sites with different organic waste concentrations.

(ii) Measurement of epilithon growth rates on concrete artificial substrates incubated under a range of organic

concentrations. This allowed the estimation of the time taken for "nuisance levels" of epilithon to develop under different wastewater loading conditions.

(iii) Measurement of the growth, respiratory and photosynthetic rates of epilithon developing in the laboratory channels when different concentrations of MCDC or BCWS wastewaters were added under standard conditions of light, temperature and current velocity.

In the river experiments the variable nature of the wastewater discharges (Sections 2.3.2 to 2.2.5) and river flow, and the difficulties encountered with the river water sampling (Section 3.3.1.1), necessitated the use of estimated average organic concentrations calculated from the effluent discharge data and results of self-purification studies (Section 6).

In addition observations were made on the trends in macrophyte development throughout the study reach.

7.2 MANAWATU RIVER STUDIES

7.2.1 SITE DAILY MEAN BOD_5 CALCULATIONS

The Manawatu River study reach (Fig 1.2) provided a range of organic wastewater concentrations due to the following factors:

(i) the sequential addition to the river, at 3.8 and 1.8 kilometre intervals, of the three different types of wastewater discharge (Section 2.2).

(ii) the changes in organic material concentration due to the requirements of the new water rights (Sections 2.2.2 to 2.2.5) and seasonal and industrial stoppages.

(iii) the presence of a 10 km long reach, with relatively unaltered physical conditions, between the end of the waste mixing zone (Site D) and Opiki Bridge (Site F (Fig 1.2) which receives no further discharges or tributary inflows and within

which self-purification processes result in a range of water quality conditions (Section 6.2.2.1).

This reach was therefore suitable for studying benthic communities under a range of different conditions. However a number of problems were encountered when dealing with the river system. For example the variable nature of the effluent discharges (Sections 2.2.3 to 2.2.5) and river flow (Section 2.1.1.1) complicated the interpretation of organic waste concentration data obtained from both grab and twenty-four hour composite river water sampling. Because of this the mean daily BOD_5 values at the different sites were calculated using effluent loading data and river flow data and the organic removal rates inferred after consideration of the rates recorded in this study (Section 6) and previous studies of the Manawatu River (Currie, 1977; Rutherford and Currie, 1979; Currie and Rutherford, 1982).

The mean BOD_5 concentrations at each river site were calculated, by computer, for each day from 1/2/82 to 18/6/84 (Appendix I). Within the waste discharge zone the following first order decay equation was used:

$$BOD_5(B) = (BOD_5(A)+(WW/(Q \cdot 86.4))).e^{(-k_{1(T)}.(t_{20}.((\frac{20}{Q})^{0.4})))}$$

(Equation 7.1)

where:

$BOD_5(B)$ = BOD_5 at the downstream site ($g.m^{-3}$)

$BOD_5(A)$ = BOD_5 directly above the upstream discharge ($g.m^{-3}$)

WW = daily wastewater BOD_5 loading ($kg.d^{-1}$) (see Table 7.1)

Q = Manawatu River flow ($m^3.s^{-1}$) (Watson, 1984)

$t_{(20)}$ = Reach travel time at Q = 20 $m^3.s^{-1}$ (Wilcock, 1984(a))

BOD_5 at site A = 0.6 $g.m^{-3}$ (the average value recorded during this study) (e.g., Table 7.5)

$k_{1(T)}$ = River first order decay rate calculated from Manawatu River values (Table 7.2) adjusted for

TABLE 7.1 Wastewater BOD_5 Loading Rates Used in Manawatu River BOD_5 Calculations

Wastewater Discharge	Wastewater Treatment Practised Over Interval	Interval	BOD_5 Loading Rate used ($kg.BOD_5.d^{-1}$)	Comments
PNCC	Primary (Section 2.2.3)	01/02/82-31/12/83, 13/03/84-31/03/84 +) 27/04/84-16/06/84	6377	Mean of seven measurements during August and November 1983 (Anderson, 1983)
PNCC	Primary plus alum flocculation (Section 2.2.3)	01/01/84-12/03/84 01/04/84-26/04/84	3400	Mean of seven samplings during alum dosing of primary effluent January and February 1984 (Anderson, 1984)
MCDC	None (Section 2.2.4)	01/02/82-18/06/84	$x \pm s = 7860 \pm 7221$ maximum = 35397 miminum = 0	Measured BOD_5 loads calculated from flow related composite sample analyses and continuous flow measurements (Meredith, 1984)
BCWS	Primary (Section 2.2.5)	01/02/82-30/04/84	$x \pm s = 2860 \pm 3810$ maximum = 10800 minimum = 0	Calculated from BCWS kill figures (Francis, 1985) using the conversion factor of 0.86 kg BOD_5/lamb unit calculated from sixteen days data January to April 1983 (Gilliland, 1983)
BCWS	Primary plus anaerobic (Section 2.2.5)	01/05/84-18/06-84	$x \pm s = 2115 \pm 1790$ maximum = 4175	Calculated from BCWS kill figures (Francis, 1985) using conversion factor of 0.35 kg BOD_5/lamb unit calculated from May and June 1984 (Gilliland, 1984).

TABLE 7.2 BOD_5 Decay Rate Values (k_1) Used in Site BOD_5 Calculations.

Reach	Intervals	$k_{1(20)}$ (hr^{-1})	Comments
PNCC	01/02/82-31/12/83, 13/03/84-31/03/84 +) 27/04/84-16/06/84	0.15	Measured value at higher BOD (Currie and Rutherford, 1982
PNCC	01/01/84-12/03/84 01/04/84-26/04/84	0.063	Measured value at lower BOD_5 loading on 29/2-1/3/84 (Section 6.2.2.1)
MCDC-C	01/02/82-18/06/84	0.20	From summer measurements (Section 6.2.2.1)
BCWS-D and D-E	01/02/82-18/06/84	0.20	From summer measurements in this study (Section 6.2.2.1) and Currie (1977).
E-E/F	01/02/82-30/04/83	0.15	From results of this study (Section 6.2.2.1).
	01/05/83-18/06-84	0.08	Average measured rate over reach D-EF January-February 1984 (Section 6.2.2.1).
E/EF	01/02/82-18/06/84	0.06	Average measured value, Currie (1977).

the average temperature using the van't Hoff-Arrhenius relationship:

$$k_{1(T)} = k_{1(20)} \, \Theta^{(T - 20°C)}$$ (Metcalf and Eddy, 1979)

where the Θ value of 1.112 was obtained by solving the van't Holf Arrhenius equation for Θ after substituting the $k_{1(T)}$ and $k_{1(20)}$ values for the respiration rates predicted by equation 5.3 (Table 5.2) for a constant typical biomass of 50 $gAFDW.m^{-2}$ at temperatures of 11 to 20°C (i.e., the range of average river temperatures during the periods prior to biomass observations (Figs 7.1 and 7.2)). This approach is based on the assumption that a stoichiometric relationship exists between the heterotrophic respiration rate and the BOD_5 removal rate. This is expected to be valid over the time period of days of interest here.

The Θ value of 1.112 obtained from the heterotroph respiration rate relationship in the Manawatu River is similar to the value of 1.135 reported for BOD_5 removal in the Mississippi River over the temperature range between 4 and 20°C (Schroepfer *et al*, 1964).

At sites below the waste discharge zone (i.e., below site D) the daily mean BOD_5 concentrations were calculated using the following similar equation.

$$BOD_5(B) = BOD_5(A).e^{(-k_{1(T)} \cdot (t_{20} \cdot (\frac{20}{Q})^{0.4}))}$$ (Equation 7.2)

where: $BOD_5(A)$ = BOD_5 at the adjacent upstream site ($g.m^{-3}$) and $k_{1(T)}$, t_{20}, Q are as defined in Equation (7.1).

It was considered that these calculated BOD_5 values gave a better indication of the average BOD_5 conditions at the river sites than the results of analyses of grab or even 24 hour composite river water samples collected on one occasion at the time of biomass observations or during a growth experiment.

The calculated values were used in the investigations of the relationships between wastewater concentrations and heterotroph

and phototroph macroscopic abundance (Sections 7.2.2 and 7.2.4) and epilithon growth rates (Sections 7.2.5 and 7.2.6).

7.2.2 EFFECTS OF WASTEWATER CONCENTRATION ON MACROSCOPIC ABUNDANCE OF HETEROTROPHS (BACTERIA AND FUNGI)

The results of the macroscopic abundance surveys of heterotrophic micro-organisms (bacteria and fungi) in the Manawatu River between March 1982 and July 1984 and other relevant environmental data are summarised in Figures 7.1 and 7.2. The straight lines and cubic spline curves joining the data points are included to aid their identification rather than for use in interpolating values between observations, although this could be justified for some of the variables (e.g., radiation and temperature).

The level of macroscopic abundance of heterotrophic micro-organisms (bacteria and fungi) was observed to vary through the year and at different river sites at any instance (Figs 7.1 and 7.2). However at site A, above the discharges, the heterotroph abundance was always zero (i.e., no obvious growth on hand held stones).

At Site B (Fig 1.2) the discharge of sufficient primary treated or primary plus chemically treated, domestic sewage (Table 7.1) to increase the background BOD_5 of approximately 0.6 $g.m^{-3}$ to between 1.2 and 3.3 $g.m^{-3}$ did not result in the growth of heterotrophic fronds on the river bed except during June 1984 (Fig 7.2). On this occasion isolated fronds of the true fungus *Leptomitus lacteus* were present. This species has an optimum temperature for growth of 8°C (Zehender and Boek, 1964) and was also occasionally present as isolated fronds at other sites below the discharges during winter but was not observed at river temperatures in excess of 14°C. The growth of *S. natans* dominated bacterial slimes at Site B was always limited to barely visible strands attached to algae and thicker films on the undersides of stones and artificial substrates (Section 7.2.6.2). However on 5/12/84 *S. natans* dominated fronds were common 300 m below the PNCC outfall where the primary treated

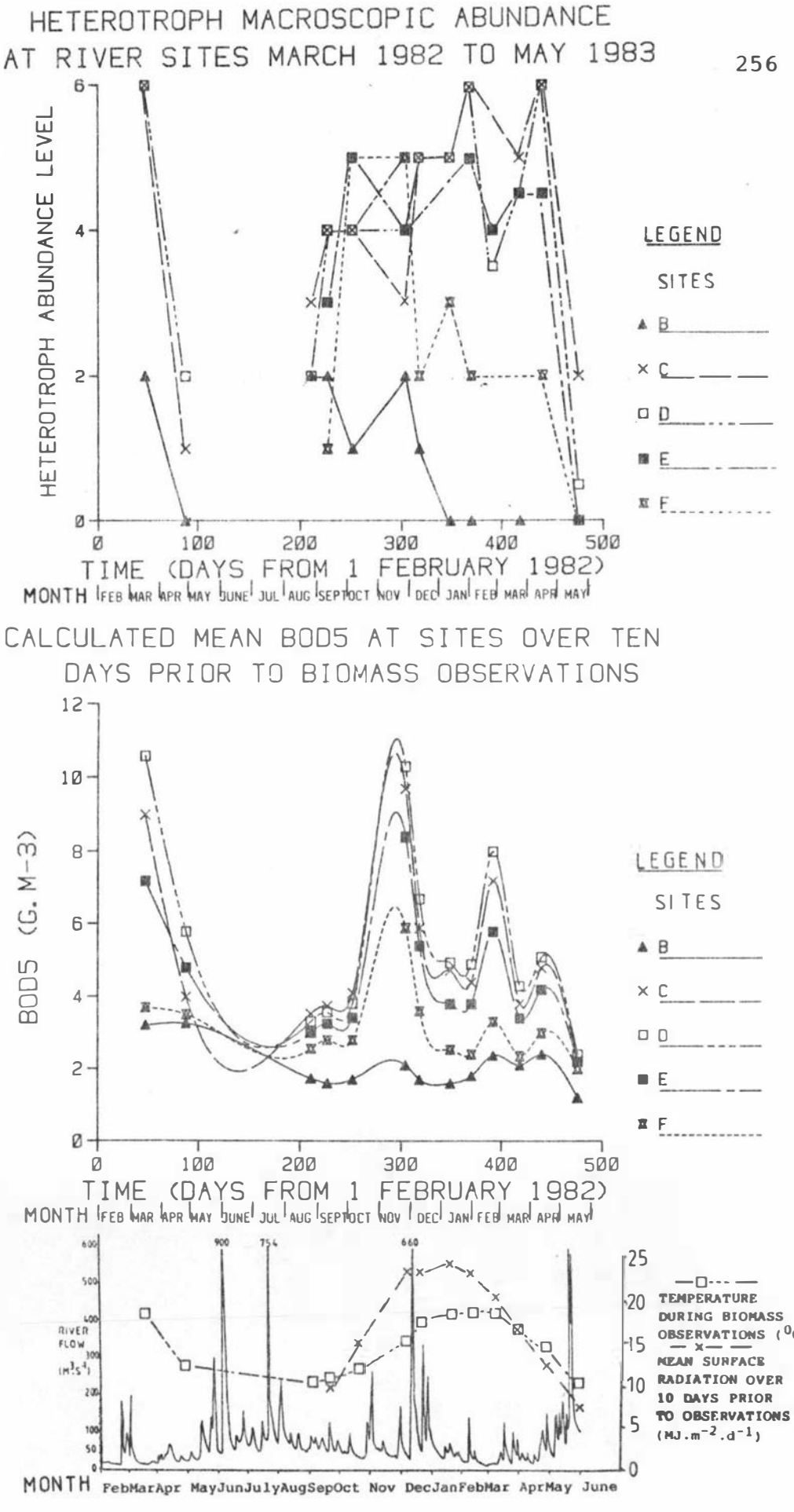


Figure 7.1: Heterotroph Macroscopic Abundance and Related Environmental Data at Sites in the Manawatu River, 1/2/82 to 1/6/83.

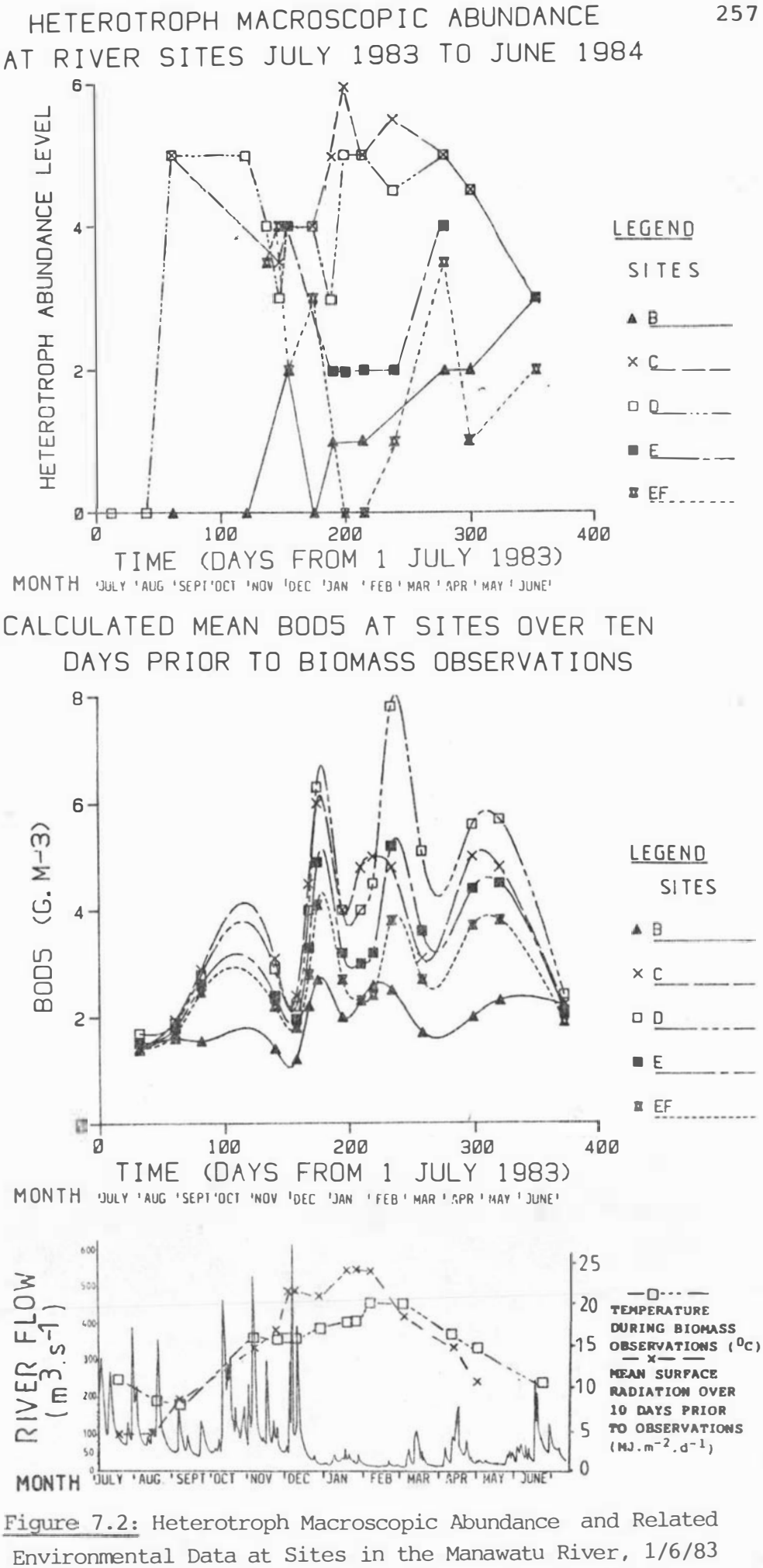


Figure 7.2: Heterotroph Macroscopic Abundance and Related Environmental Data at Sites in the Manawatu River, 1/6/83 to 18/6/84.

sewage effluent was not fully mixed with the river water. These observations are consistent with those made in outdoor channel studies on the effects of primary treated sewage on benthic community development (Wuhrmann, 1954; Zimmermann, 1961; Eichenberger, 1975) (Section 2.3.4.2).

At site C, 1.8 km downstream from the MCDC discharge (Fig 1.2), macroscopically visible heterotrophic growths were observed in the surveys except during the late autumn and winter periods of minimal MCDC discharge (April to late-August 1982; May to mid-August 1983; May to mid-August 1984) (Figs 7.1 and 7.2).

At current velocities of 0.3 to 0.5 $m.s^{-1}$, heterotroph fronds were common (level 4), covered many surfaces (level 5) or covered the whole river bed (level 6) at calculated mean BOD_5 values of 2.9 to 9.0 $g.m^{-3}$ (Fig 7.3). Thus comparitively minor increases in BOD_5 due to the MCDC discharge often resulted in a large increase in heterotroph abundance at site C. For example at the end of August 1983 the heterotroph abundance level increased from 0 at site B, where the calculated mean BOD_5 was 1.55 $g.m^{-3}$, to 5 at site C in response to the discharge of sufficient MCDC effluent to increase the river BOD_5 by 1.4 $g.m^{-3}$ and raise the calculated mean BOD_5 at site C to 2.9 $g.m^{-3}$ (Fig 7.2). These results show that an in-river BOD_5 limit of 5 $g.m^{-3}$ is not effective in preventing macroscopic heterotrophic growth when most of the organic material is untreated dairy factory effluent.

One anomalous observation was made on 26/11/82 when heterotrophs were only present as isolated fronds, dominated by *S. natans* and *Flavobacterium*, despite very high MCDC discharge levels and a calculated mean BOD_5 concentration at site C of 9.7 $g.m^{-3}$ (Figs 7.1 and 7.3). Low epilithon growth rates, similar to those at site A, were also recorded at site C on 22-26/11/82 (Section 7.2.6.3). At site D the heterotrophs were more abundant (level 5) and the epilithon growth rates were an order of magnitude higher than those at site C (Fig 7.13, Section 7.2.6.3) despite a very minor BOD_5 increase of 0.6 $g.m^{-3}$ due to the BCWS discharge of primary treated

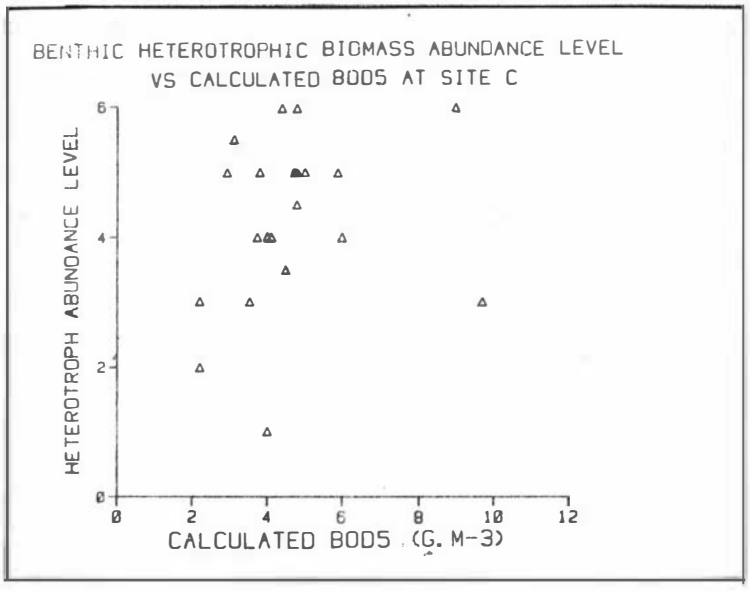


Figure 7.3: Graph of Benthic Heterotrophic Biomass Abundance Level (Appendix B) at Site C Versus Calculated Mean BOD_5 Over the Ten Days Prior to the Biomass Observations.

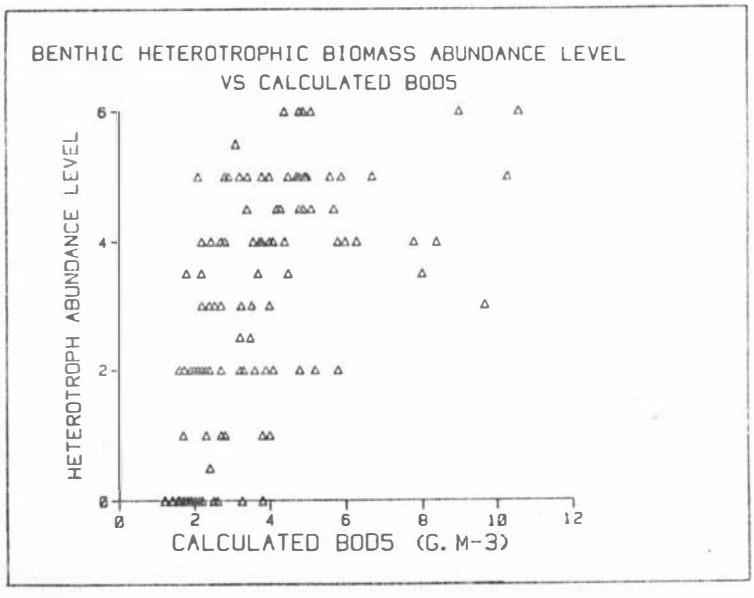


Figure 7.4: Graph of Benthic Heterotrophic Macroscopic Abundance Level (Appendix B) at all River Sites Versus Calculated Mean BOD_5 Over the Ten Days Prior To the Biomass Observations.

meatworks effluent. Further downstream at sites E and F the heterotroph abundance was similar to that at site D (Fig 7.1) but growth rates at sites EF and F were higher than at D (Fig 7.13) despite lower BOD_5 concentrations (Fig 7.1). The results of the laboratory channel studies (Section 5.5.3 and 7.3) and river observations at the end of April 1982 (see below) indicate that it is unlikely that the increased heterotroph abundance and epilithon growth rates below site C resulted directly from the BCWS organic loading. One possible explanation for this anomaly is that components of the MCDC wastewater produced growth inhibition at the high concentrations at site C but this effect was removed at the downstream sites where the concentration of the inhibiting components would be expected to be reduced by self-purification processes.

Nutrient limitation of heterotrophic growth is not a likely explanation since both the PNCC and MCDC discharges contribute significant nutrient loads upstream of site C (Sections 2.2.3, 2.2.4 and Table 7.5). Inhibition of growth by solar radiation is also unlikely since heterotrophs were abundant at the other sites further downstream on 26/11/82 and at site C on other occasions when similar radiation inputs occurred (Figs 7.1 and 7.2, Section 4.4.1). Also growth rates studies at site D did not show solar radiation inhibition of growth under higher average daily total surface radiation levels (24.6 $MJ.m^{-2}.d^{-1}$) (Section 4.4.2).

The observations on 29/4/82 (Fig 7.1) indicate that the PNCC wastewater has a lesser effect on heterotroph abundance per unit BOD_5 than the MCDC wastewater. A mean calculated BOD_5 of 4 $g.m^{-3}$ over the ten days prior to this occasion did not result in macroscopic heterotrophic growth at site C. This was apparently due to the low relative contribution of the MCDC wastewater to the BOD_5. The MCDC discharge was only sufficient to cause an average increase in the river BOD_5 by 0.8 $g.m^{-3}$ whereas the PNCC discharge was sufficient to increase the river BOD_5 by 3.3 $g.m^{-3}$ upon mixing.

During the summer months (i.e., December to March inclusive) of 1982/83 the MCDC BOD_5 discharge exceeded the authorised level by 65 $\pm$ 101% ($x \pm s$) whereas during the corresponding period of 1983/84 the discharge exceeded that permitted by only 8 $\pm$ 52% ($x \pm s$) (calculated from MCDC discharge and river flow data (Gilliland, 1984) using the LMRTC (1980) allowable discharge formula). The calculated mean BOD_5 values during these intervals at site C over the ten day periods prior to the biomass observations declined from between 3.8 and 7.2 $g.m^{-3}$ during 1982/83 to between 3.1 and 5.0 $g.m^{-3}$ during 1983/84. This only resulted in a minor decrease in heterotroph abundance at site C (Figs 7.1 and 7.2). This suggests that the allowable increase in the river BOD_5 due to the MCDC discharge would need to be reduced to less than that permitted by the current water right (2.9 to 2.4 $BOD_5.m^{-3}$ for river flows of 13 to 30 $m^3.s^{-1}$) in order to substantially reduce the sewage fungus growth in the reach from the outfall to site C. However the expected further reduction in the PNCC BOD_5 loading upstream (Fig 1.2) due to the aerated lagoon treatment from 5/3/84 (Section 2.2.3.4) may reduce the effect of the MCDC addition to a limited degree. These results and those from 29/4/82, discussed earlier, indicate that the maximum allowable increase in river BOD_5 due to the MCDC discharge without producing macroscopic heterotrophic fronds at site C under summer low flow conditions lies between 0.8 and approximately 2.5 $g.m^{-3}$.

The heterotrophic biomass at site D, 1.4 km below the BCWS discharge, results from the combined effects of all the upstream discharges operating. However the results suggest that the MCDC discharge has a large influence. Heterotrophic fronds were uncommon (level 3) or absent (levels 0-2) at site D during the seasonal shut-down of the MCDC discharge (from April to late-August 1982, May to mid-August 1983 and May to mid-August 1984) but were common (level 4) or covered most or all of the river bed (level 5 and 6 respectively) during the periods of significant MCDC discharge except on 6/1/84 (Figs 7.1 and 7.2). During the late winter and spring months of nil BCWS discharge pinkish-tan heterotrophic fronds were common or

covered most of the bed at site D at calculated mean BOD_5 levels, due mainly to the MCDC discharge, of 2.8 to 3.4 $g.m^{-3}$ (Figs 7.1 and 7.2). By contrast at the end of April 1982 a mean BOD_5 of 5.8 $g.m^{-3}$ did not produce the growth of heterotrophic fronds at site D (Fig 7.1). On this occasion only a small fraction of the BOD_5 at site D was due to the MCDC discharge, which increased the mean river BOD_5 by only 0.8 $g.m^{-3}$ upon mixing, 3.2 km upstream of site D. Most of the BOD_5 at D was due to the BCWS discharge, which increased the river BOD_5 by 3 $g.m^{-3}$, 1.4 km upstream of D, but some would also have been due to the primary treated PNCC discharge, which increased the river BOD_5 by 3.3 $g.m^{-3}$, 7 km upstream of site D.

These results suggest that the BCWS wastewater has a lesser effect on heterotrophic growth per unit BOD_5 than the MCDC wastewater. The April 1982 results indicate that an average in-river BOD_5 limit of 5.0 $g.m^{-3}$ may be appropriate for eliminating macroscopic heterotrophic growth in response to primary treated meat-works and domestic sewage effluents but it is clear from the observations at sites C and D that a lower limit is required to control growth in response to untreated dairy factory wastewaters. This corroborates the results of the laboratory channel comparison of the effects of the MCDC and BCWS wastewaters on heterotrophic respiration rates (Section 5.4) and epilithon growth rates (section 7.3).

At the sites downstream of the waste discharge zone (i.e., downstream of site D) (Fig 1.2) the BOD_5 values were progressively reduced by self-purification processes and a general pattern of declining heterotrophic abundance with distance was usually observed (Figs 7.1 and 7.2).

The relationship between calculated mean BOD_5 over the ten days prior to biomass observations and heterotroph abundance level for all the observations at sites below the discharges (i.e., sites B to F) are presented in Figure 7.4. Although there was a highly significant correlation between the heterotrophic abundance level and BOD_5 ($r = 0.55$, $P < .001$), there was considerable variation in the levels at which macroscopic

heterotrophic fronds were common (i.e., level 4). On some occasions BOD_5 values of 2.2 $g.m^{-3}$ resulted in this heterotroph abundance level whereas on others macroscopic fronds did not occur at BOD_5 levels up to 5.8 $g.m^{-3}$. As discussed previously this variation in the effect of BOD_5 on heterotrophic abundance appears to result from the growth inhibition at high MCDC wastewater concentrations and the different compositions of the BOD_5 discharged by the three discharges.

Unfortunately it was not possible to study the natural benthic communities in the river under summer low-flow conditions where all the dischargers were operating within the requirements of their new water rights during the course of the main study period due to delays by the BCWS and PNCC in meeting their rights and frequent breaches of the allowable discharge by MCDC. Nevertheless the conditions during early and mid-January 1984, after the BCWS discharge had been halted for three to four weeks due to an industrial stoppage, and during February 1985 give some indication of the situation expected when the rights are complied with.

During the ten day period prior to the survey on 6/1/84 the MCDC BOD_5 discharge averaged 4% less than the allowable level whilst the PNCC discharge was 85% greater than the level permitted by the new PNCC water right which applied from 1/1/85. Under these conditions the sewage fungus zone (where heterotrophic abundance was between levels 4 and 6) was restricted to a 2.5 km long reach from the MCDC outfall to midway between sites C and D (Fig 1.2).

Over the ten days prior to 16/1/84 the MCDC discharge exceeded that permitted by 30% on average and the zone of macroscopic sewage fungus growth extended approximately 3.7 km below the MCDC outfall. This suggests that the permitted MCDC discharge will need to be further restricted if the zone of macroscopic heterotrophic growths (sewage fungus) below the outfall is to be eliminated during summer. The reduction of sewage fungus growths in this region has been shown to be an important step in overcoming the oxygen depletion problems in the river (Section 5.4).

The results of a benthic biomass abundance survey on 15/2/85, after a three week period of declining flows, indicate that reduction of the present permitted MCDC BOD_5 discharge by 35% should eliminate the zone of macroscopic sewage fungus growth during the summer lowflows (Table 7.3). On this occasion heterotrophic fronds were scarce at site C (Table 7.3) after a period when the PNCC discharge, of primary plus alum flocculation treated, domestic sewage, was very close to the permitted BOD_5 loading and the MCDC BOD_5 discharge averaged 35% less than the maximum loading permitted, giving a calculated BOD_5 addition upon mixing of 1.8 ± 0.5 $g.m^{-3}$ ($x \pm s$) for the ten days prior to the observations. However the results of the winter surveys discussed earlier suggest that a lower limit may be appropriate at other times of the year.

The lack of any obvious increase in heterotrophic abundance at site D on 15/2/85 (Table 7.3), where the discharge of anaerobically treated meatworks effluent plus fellmongery waste 1.45 km upstream was sufficient to increase BOD_5 to 4.85 ± 1.4 $g.m^{-3}$, indicates that this waste water has a minor influence on the heterotrophic growth per unit BOD_5 addition compared with the MCDC, untreated, dairy factory wastewater.

These results show that the threshold BOD_5 concentration for the development of abundant heterotrophic growths (sewage fungus) varies for different types of wastewater. In the Manawatu River it was necessary to limit the increase in the river BOD_5 upon mixing due to the MCDC effluent discharge to 1.8 $g.m^{-3}$ to prevent the common occurrence of macroscopic heterotrophic fronds at site C, 1.8 km below the outfall, during summer. However addition of 1.4 $g.m^{-3}$ BOD_5 as MCDC effluent during winter did not prevent the common occurrence of the heterotrophic growths at this site. Several factors may contribute to this apparent seasonal difference in heterotrophic abundance (Sections 2.3.4.3) namely temperature effects on organic removal rates (Section 7.2.1) and ingestion rates of grazers, seasonal variations in grazer density, and partial inhibition due to solar radiation during summer. However the results of plate growth experiment (Section 4.4.2)

TABLE 7.3 Macroscopic Abundance of Heterotrophs and Phototrophs at Manawatu River Sites, 15/02/85 (Riverflow = 13 $m^3.s^{-1}$)

Site	Calculated BOD_5 over Previous Ten Days* ($g.m^{-3}$)	Percent exceedence of Permitted Daily BOD_5 Discharge ($\bar{x} \pm s$)	Phototroph Abundance Level	Heterotroph Abundance Level
A	0.6		D-E	0
B	1.9 ± 0.2	4 ± 10%	F	2-3
C	3.05 ± 0.5	-35 ± 20%	F	2-3
D	4.85 ± 1.4	21 ± 10%	F	2-3

* Calculated as outlined in Section 7.2.1 but

Assuming:

(i) Background BOD_5 site A = 0.6 $g.m^{-3}$
(ii) PNCC loading = 2500 kg $BOD_5.d^{-1}$ (Anderson, 1985)
(iii) MCDC daily measured values (Meredith, 1985)
(iv) BCWS loading = 4300 $kg.d^{-1}$ (weekdays) (Gilliland, 1985)
(v) k_1 PNCC-B = 0.07.d^{-1} (measured 29/2-1/3/84)
(vi) k_1 MCDC-C = 0.10.d^{-1} (assumed value at low initial BOD_5)
(vii) k_1 BCWS-D = 0.15.d^{-1} (assumed value)

indicate that this latter effect is not significant in the Manawatu River.

7.2.3 SPECIES COMPOSITION OF THE HETEROTROPHIC GROWTHS IN THE MANAWATU RIVER

The heterotrophic growths were dominated by bacterial species (Table 7.4) as was expected under the neutral to alkaline conditions in the Manawatu River (Sections 2.2.1.2 and 2.3.4.8). Nevertheless isolated fronds of the fungus *Leptomitus lacteus* occurred at several sites during winter (Section 7.2.2).

Sphaerotilus natans was always dominant or co-dominant (Table 7.4). This species was also dominant or co-dominant in twenty-four of the thirty New Zealand outbreaks examined (Cooper, 1983), twelve of the fifteen outbreaks examined in Eire (Gray and Clarke, 1984) and 93 of the 178 outbreaks examined in the United Kingdom (Curtis and Harrington, 1971). The filamentous bacterium *Flavobacterium* sp. was co-dominant or abundant during spring, giving the growths a pink colouration, but was uncommon or not detected during the rest of the year (Table 7.4), when the heterotrophic fronds were white, tan or brownish-green, depending upon their algal content. This species was not reported in any of the thirty New Zealand outbreaks examined by Cooper (1983) but was abundant in 20 of the 178 samples examined in the UK survey (Curtis and Harrington, 1971). *Zoogloea* sp., which was dominant or co-dominant in seven of the thirty New Zealand outbreaks examined by Cooper (1983) and 104 of the outbreaks in the UK survey (Curtis and Harrington, 1971), sometimes occurred amongst the heterotrophic fronds but was never dominant in the Manawatu River samples (Table 7.4).

A small amount of *Beggiatoa* sp growth was observed at site D on one occasion. This genus was dominant in one outbreak (below a refuse tip leachate inflow) in the New Zealand sewage fungus survey (Cooper, 1983).

These results show that the dominant heterotroph in the Manawatu River (*Sphaerotilus natans*) was also the most

TABLE 7.4 Seasonal Variations in Bacterial and Fungal Species Abundance within Heterotrophic Fronds at sites C and D, August 1983 to June 1984.

Date	Bacterial and Fungal Species Present in Significant Amounts and Relative Abundance	
	Site D	Site C
09/08/83	S > Z	ND
30/08/83	S	S
29/10/83	S = F	ND
24/11/83	S > F	S = F > > Z
02/12/83	S > F > Z = B	S > > F
11/01/84	ND	S > > F
16/01/84	S	ND
24/01/84	S	S
31/01/84	S	S
09/02/84	S	S
22/02/84	S	S
28/02/84	S > > F = Z	ND
23/03/84	ND	S > > Z
04/04/84	S	ND
26/04/84	ND	S
18/06/84	S > L = Z > F	S > Z > L

Key

S = *Sphaerotilus natans*
F = *Flavobacterium* sp.
Z = *Zoogloea* sp.
L = *Leptomitus lacteus*
ND = Not determined
B = *Beggiatoa* sp.

frequently reported dominant species in sewage fungus outbreaks in New Zealand and Eire and the second most frequent dominant or co-dominant species observed in an extensive survey in the United Kingdom. The other commonly dominant species in the previous surveys were also present in the Manawatu River growths. The results of the studies on the effects of the wastewater discharges on heterotrophic growth in the Manawatu River should therefore be relevant to the situation in many New Zealand and overseas rivers with broadly similar physical characteristics.

7.2.4 EFFECTS OF WATER QUALITY ON THE ABUNDANCE OF PHOTOTROPHS

The effects of the wastewater discharges on the macroscopic abundance of phototrophs, at the medium current velocities investigated (0.3 to 0.6 $m.s^{-1}$, measured 50 mm off the bed), varied seasonally and differed during the summers of 1982/83 and 1983/84 (Figs 7.5 and 7.6). The effects were relatively minor, involving alteration of the species composition and minor increases in abundance, during the winter and spring periods of high unstable flows and during the summer and autumn of 1982/83, when the background nutrient levels were generally sufficient to sustain algal growth above the discharges (Freeman, 1983). Under these conditions the standing crop of phototrophs was apparently controlled largely by the physical conditions of surface radiation, turbidity, temperature and river flow.

The large spates during May and June 1982 removed the algal biomass which had developed during the previous summer and autumn and low light and temperature conditions during the winter months limited the phototroph biomass to a patchy dark brown diatom turf (Fig 7.5), dominated by *Navicula* sp. The high flows during November and December 1982 resulted in very low phototrophic biomass at all sites and it was not until the more stable flow conditions of January 1983 that significant growths were able to develop.

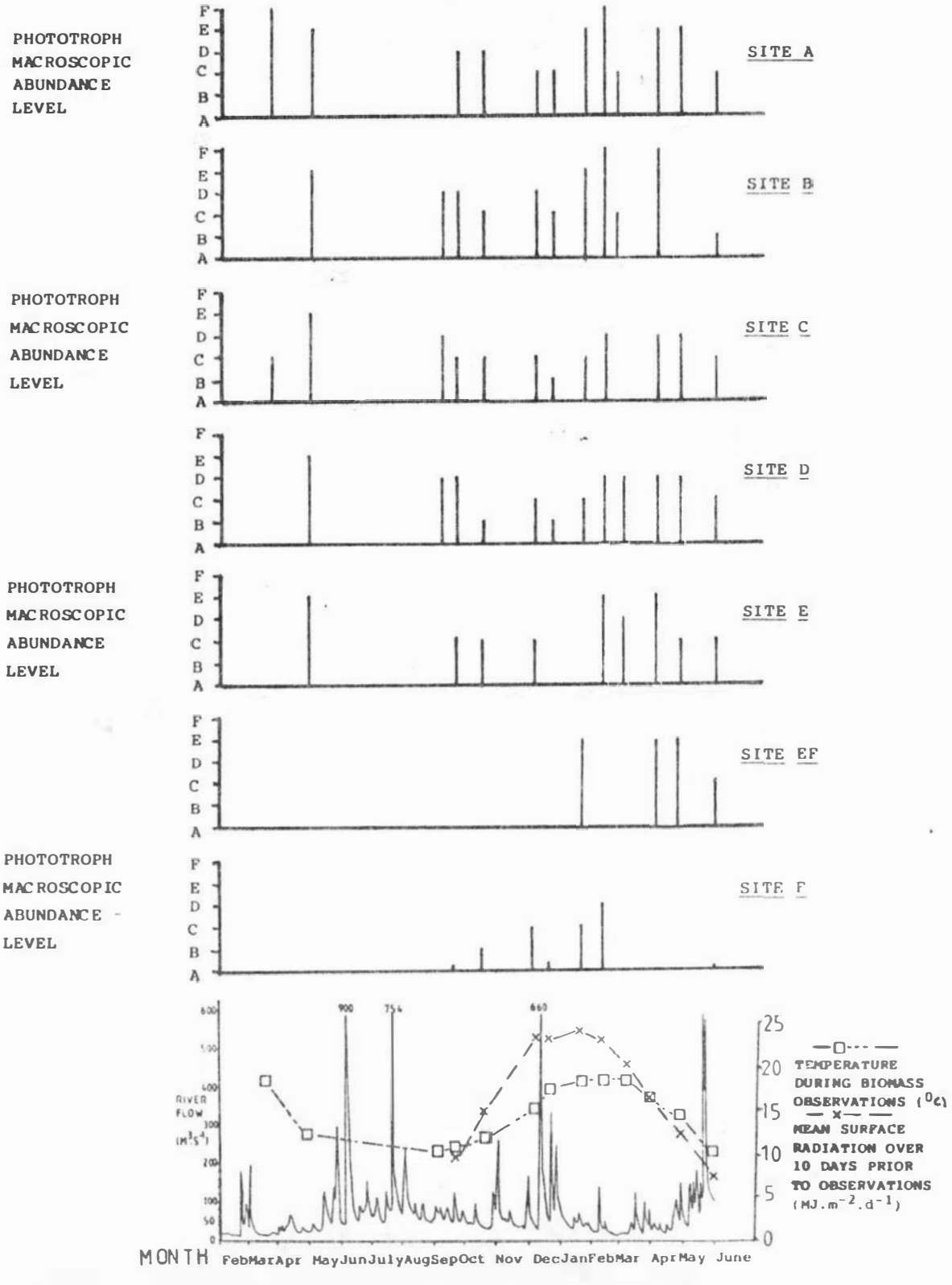


Figure 7.5: Variations in Macroscopic Abundance of Phototrophs at Manawatu River Sites and Environmental Factors, February 1982 to June 1983.

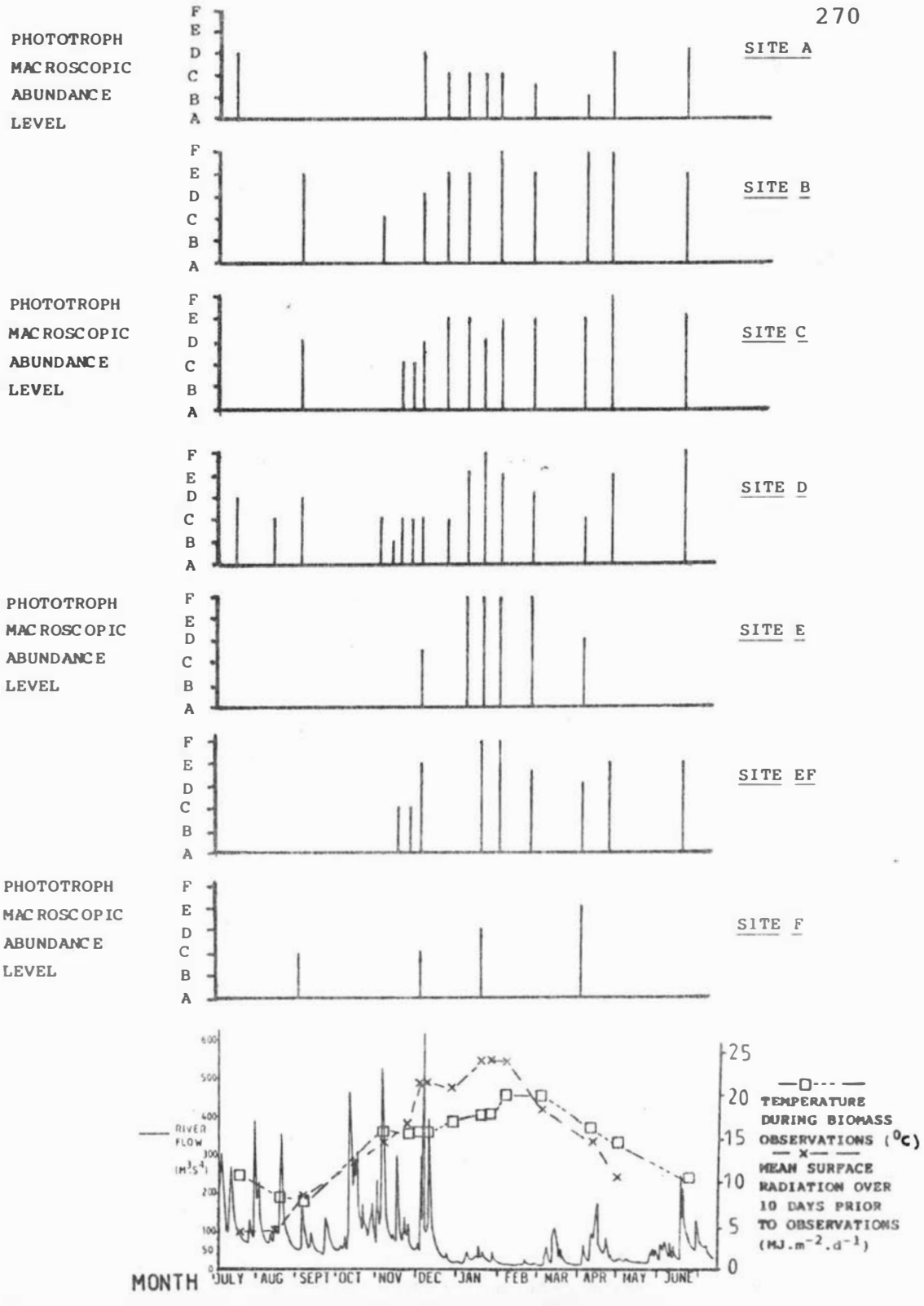


Figure 7.6: Variation in Macroscopic Abundance of Phototrophs at Manawatu River Sites and Some Environmental Factors, July 1983 to June 1984.

Nutrient availability tests have shown that phosphorus concentrations below approximately 4 $mg.m^{-3}$, as dissolved reactive phosphorus (DRP), limit the growth of the predominant filamentous green algae *Cladophora glomerata* growing during summer at moderate current velocities of 0.4 to 0.7 $m.s^{-1}$ (Freeman, 1983). During January 1985 the DRP concentration above the discharges was between 3.5 and 5 $mg.m^{-3}$ except at the beginning of the month when a concentration of 14 $mg.m^{-3}$ was recorded (Freeman, 1983) and a bloom of the stalked diatom *Gomphonema* sp. covered the bed at site A (Fig 7.5). This species and the filamentous green algae *Stigeoclonium tenue* also covered the bed at site B below the PNCC discharge (Fig 1.2) and a diverse community of diatoms and unicellular algae developed amongst the heterotrophic fronds further downstream at sites C, D and E (Fig 7.5).

The *Gomphonema* sp. dominated community at site A began to slough off the bed at the end of January 1983 (Section 6.3.2) and the phototroph biomass was reduced drastically at sites A and B after a 145 $m^3.s^{-1}$ spate on 9 February 1983 (Fig 7.4; Freeman 1983). Subsequently the DRP levels at site A increased to between 6.6 and 9.7 $mg.m^{-3}$ during February and then declined to between 3.5 and 7 $mg.m^{-3}$ during March (Freeman, 1983). *Cladophora* growths developed during this interval (Freeman, 1983) and by 23 March 1983 covered all the bed at site B, covered most of the bed at sites A, E and EF and were common at sites C and D (Fig 7.5), where the growths were completely overgrown by heterotrophic fronds. Similar phototroph abundances were recorded at most sites on 12 April 1983 but between this date and 17 May 1983 frequent spates reduced the phototrophic community to an algal film and isolated strands of *Cladophora*.

The phototrophic abundance was not assessed at site A between May and December 1983, but at the downstream sites the winter phototrophic biomass was again dominated by a patchy dark brown diatom turf dominated by *Navicula* sp., as during the previous year, although isolated strands of *Cladophora glomerata* and clumps of the macrophyte *Potamogeton crispus* also occurred.

During the summer of 1983/84 the background DRP concentration was consistently low (less than 10 $mg.m^{-3}$) and usually below the critical limiting level for *C. glomerata* growth in the Manawatu River of 4 $mg.m^{-3}$ DRP (Freeman, 1983) (Table 7.5). Under these conditions phototrophic colonies were only just visible on the bed as an algal film at site A. However their abundance increased dramatically at sites below the discharges where the nutrient levels rose sharply (Fig 7.6) (Section 2.2.1.2). For example on 29/2-1/3/84 the total dissolved phosphorus concentration (TDP) at site B was 47 $\pm$ 3 $mg.m^{-3}$. Further downstream at sites D and EF the daily average levels of TDP ranged from 100 to 140 $mg.m^{-3}$ during late January 1984 (Section 6.2.2.2).

The results of a survey of the total benthic biomass (Section 3.2.2.1) at locations of moderate current velocities (0.3 to 0.4 $m.s^{-1}$, 50 mm off the bed) and depth (0.3 to 0.35 m) at sites throughout the study reach on 16 January 1984 provide both quantitative and qualitative data on the effects of the discharges on the natural bed phototrophic biomass under summer conditions when the background phosphorus concentration at site A limited their growth (Fig 7.7).

At site A, where the total benthic biomass was 28 $\pm$ 12 $gAFDW.m^{-2}$, much biomass was contributed by settled detritus and snails rather than epilithon. At site B heavy growths dominated by *C. glomerata* gave a total biomass of 83 $\pm$ 26 $gAFDW.m^{-2}$. The phototrophic biomass was somewhat reduced at site C but the total biomass increased due to heavy heterotrophic growth.

Downstream of site C the heterotrophs became progressively less abundant as the BOD_5 concentration was reduced by self-purification but increased phototrophic growth maintained high total benthic biomass levels. At sites EF and F an almost purely phototrophic *C. glomerata* dominated community occurred.

The progressive reduction in the total benthic biomass at these sites was apparently due to the progressive decline in stone

TABLE 7.5 Nutrient and Organic Concentrations at Site A During Summer of 1983/84 (Grab samples)

Date	BOD_5 ($g.m^{-3}$) ($x \pm s$)	Total Dissolved Nitrogen (TDN, $mg.m^{-3}$) ($x \pm s$)	Dissolved Reactive Phosphorus (DRP, $mg.m^{-3}$) ($x \pm s$)
30/11/83	0.1 ± 0	392 ± 9	3.2 ± 0.7
07/12/83	0.2 ± 0	ND	0.8 ± 0.3
07/01/84	0.7 ± 0	317 ± 3	3.7 ± 0.1
19/01/84	0.7 ± 0.3	281 ± 14	8.8*
25/01/84	0.4 ± 0.3	291 ± 17	2.7 ± 0.3
01/02/84	0.6 ± 0.1	ND	ND
10/02/84	0.6 ± 0.1	ND	ND
29/1-1/3/84	1.4 ± 0.5	279 ± 15	4.2 ± 0.9

* = contamination of the duplicate sample, with a measured DRP of 23.4 $mg.m^{-3}$, suspected.

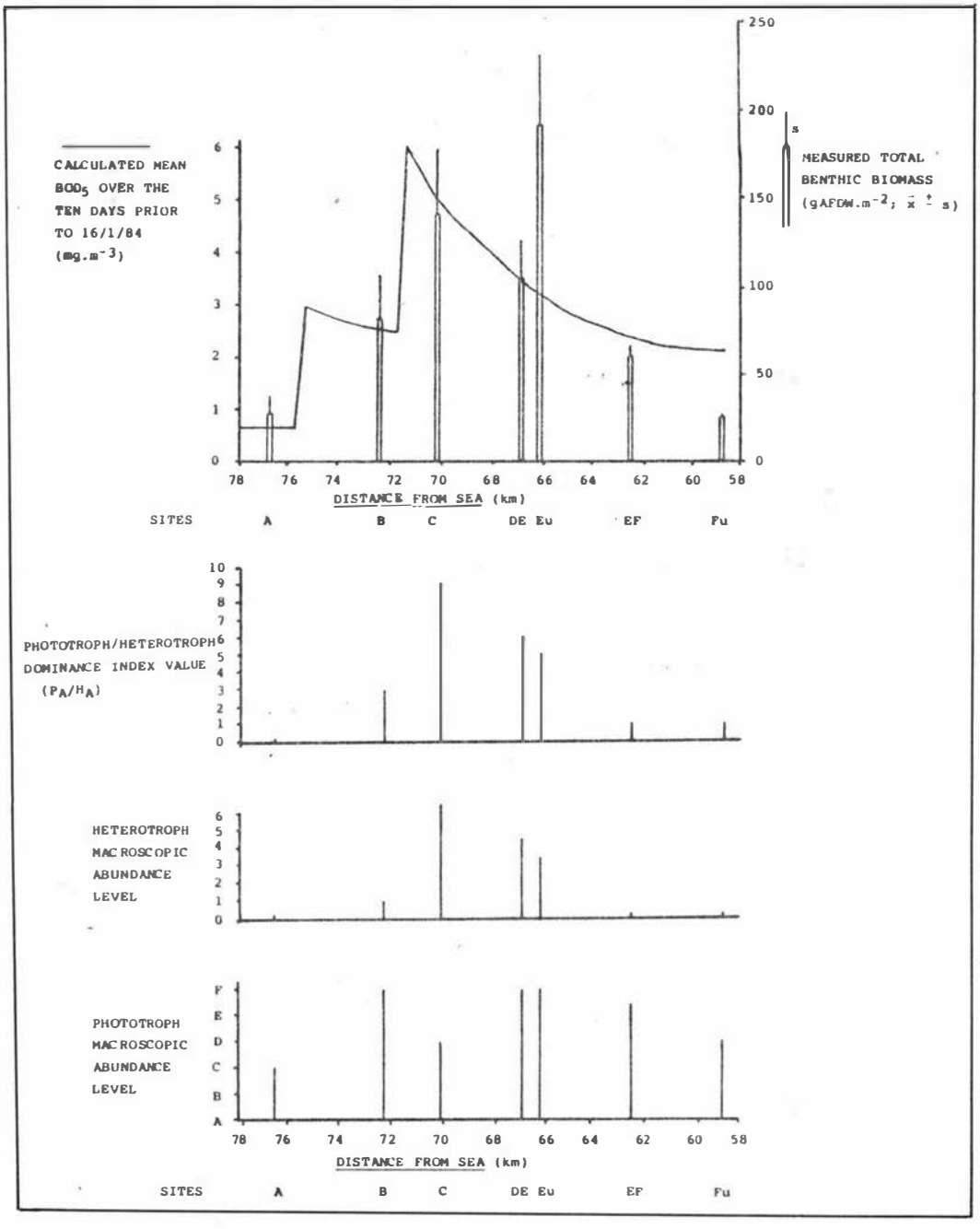


Figure 7.7: Benthic Biomass and Community Composition on 16/1/84 and Mean Calculated BOD_5 Over the Ten Previous Days at Sites in the Manawatu River.

size downstream of site E (Section 4.2). This results in the attached growths being more readily abraided since lighter stones roll more easily under the frictional force due to the current (Schmidt 1961).

The variations in phototroph abundance around the waste discharges in this and other surveys (Figs 7.5 and 7.7) with reduced abundance in areas of high organic concentration and increasing abundance further downstream have been observed in outdoor experimental channel studies (Eichenberger, 1972) and river surveys (Hynes, 1960). This is generally attributed to the combined effects of inhibition of phototroph growth by wastewater components and their exclusion by the rapid growth of heterotrophs preventing their colonisation on the growth surfaces (Hynes, 1960; Wuhrmann, 1974). Downstream of the outfalls the heterotrophic activity progressively reduces the dissolved organic material concentration and the ratio of light to chemical energy input becomes more favourable for phototrophs which grow rapidly at the elevated nutrient levels.

The low levels of phosphorus and algal biomass at site A during the summer of 1983/84 compared to the previous summer probably resulted from the low average catchment runoff (as assessed from the average river flow) during this period. Previous studies have shown that the phosphorus concentration at site A above the discharges is related to the river flow. However the effect of spates on the river phosphorus varied on different occasions indicating that the condition of the catchment, with respect to its ability to retain phosphorus, and location and intensity of the rainfall affect the phosphorus input during spates (Freeman, 1983). The low average river flow of 27 $m^3.s^{-1}$ during the 1983/84 summer growth season months, of December to February inclusive, would therefore be expected to have resulted in lower nutrient input to the river from catchment runoff than during this period fo 1982/83 when the average river flow was 70 $m^3.s^{-1}$. This consideration should also relate to the summer months of the period 1970 to 1982 when the mean river flow for these periods was 54 $m^3.s^{-1}$ (Watson, 1984). However, examination of the available

data on algal growth above the waste discharges shows that algal growths were prolific near site A during the summers of 1976/77 (Currie, 1977), 1977/78 (Gilliland, 1978) and 1981/82 (Freeman, 1983) when the mean river flows for the months December to February inclusive were 59.5 $m^3.s^{-1}$, 28.6 $m^3.s^{-1}$ and 45 $m^3.s^{-1}$ respectively (Watson, 1984).

Changes in inorganic fertiliser application rates between 1982/83 and 1983/84 do not explain the lower phosphorus in the latter season since the quantity of fertiliser sold to buyers in the catchment area by the major supplier increased by 2.5 percent in 1983/84 (Cooper, 1984).

The nutrient input to the river from dairy sheds and piggeries, which number approximately 800 in the catchment area upstream of site A (Gilliland, 1985), was progressively reduced in the interval from 1975 to 1980 by the use of land disposal and anaerobic/facultative pond treatment systems (Gilliland, 1981). Although the latter treatment system is not expected to reduce the phosphorus content of the wastewater, it has been found that less than 10% of the pond systems have any direct outflow to a receiving water during summer (Gilliland, 1981). However the reduction of the nutrient input to the river from this source does not explain the reduced phosphorus levels and algal growth during the summer of 1983/84 as compared with that of 1982/83.

In conclusion, the low phosphorus levels at site A during the summer months of 1983/84 probably resulted from the low river flows. However the 1977/78 observations of prolific algal growth above the discharges under similar average summer river flow conditions indicates that low average river flows are not a reliable indicator of algal-growth limiting phosphorus conditions at site A. The likely reasons for this are the effects of the condition of the catchment soils and the location and intensity of the rainfall on the phosphorus loading of the runoff (Freeman, 1983).

The results of the biomass survey on 15 February 1985 (Table 7.3) show that on this occasion phototrophic abundance was

similarly high below each of the discharges. The dominant species observed below the discharges (*Cladophora glomerata* and *Potamogeton crispus*) also occurred upstream at site A but their growth was patchy and less luxuriant. The *Cladophora glomerata* filaments were a noticably lighter green colour than at the downstream of the discharges. Similar variations in the colour of *Cladophora* above and below organic wastewater discharges were made by Butcher *et al* (1937). The darker colour of the algal growths below the discharges suggest that these had greater chlorophyll a concentrations and were generally in better physiological condition than those upstream where nutrients were less abundant.

In summary the macroscopic visual observations of phototroph abundance during the study period show that the addition of the PNCC discharge usually resulted in a significant increase in phototroph abundance during the summer lowflows. This effect was particularly marked during the summer of 1983/84 when the upstream DRP concentration was generally less than the critical limiting value for *Cladophora glomerata* growth.

At sites C, D and E where heterotrophs were often abundant, phototroph abundance was less than at site B during the summer of 1983/84 but similar growths occurred at B, C and D during February 1985 when the heterotroph abundance was significantly lower than during previous years (Section 7.2.2). At site EF the phototroph abundance was usually similar to that at site B during the summer low flows but at site F epilithic phototroph development was restricted by the small size of the bed material.

7.2.5 EFFECT OF RIVER FLOW ON POTAMOGETON CRISPUS DEVELOPMENT

The river flow conditions during 1983 and 1984 were characterised by lack of high winter flows (Table 7.6). This appears to have resulted in the substantial increase in the abundance of the macrophyte *Potamogeton crispus*.

TABLE 7.6 Frequency of Large Spates in the Manawatu River, 1972 to 1984 (Watson, 1985)

Year	Number of Spates Exceeding Given Flow at Palmerston North		
	>500 $m^3.s^{-1}$	>800 $m^3.s^{-1}$	>1000 $m^3.s^{-1}$
1972	8	3	2
1973	5	1	0
1974	14	5	3
1975	9	4	2
1976	15	4	3
1977	11	3	2
1978	9	4	2
1979	5	3	2
1980	5	4	2
1981	6	2	2
1982	3	1	0
1983	3	0	0
1984	0	0	0
$\bar{x} \pm s$, 1972-1981	8.7 ± 3.7	3.3 ± 1.1	2.0 ± 0.8

P. crispus growths were not reported in any of the previous biological surveys of the Manawatu River. Isolated clumps were first observed in November 1982 at sites C and D (Fig 1.2) but their abundance remained low during the summer of 1982/83. During 1983 the growths became increasingly common below the discharges and developed as dense swards throughout the summer of 1983/84, when small isolated clumps were also observed at site A. The growths were most common in areas where the current velocity was less than 0.3 $m.s^{-1}$ at river flows of less than 20 $m^3.s^{-1}$ but swards were also observed at locations at sites DE and EF where the current velocity was 0.4 to 0.5 $m.s^{-1}$ during low flows. At many sites the plants grew to near the water surface at depths of 0.4 m and biomass densities of 170 to 420 $gAFDW.m^{-2}$ were measured within swards at site EF during February and March 1984. By November 1984 *P. crispus* growths had developed into dense growths extending from the water's edge out 5 metres or more into the river at low current velocity sites throughout the study reach.

Preliminary studies of the effects of these communities on stream oxygen dynamics indicated that, due to the high biomass densities developed, they have the potential to cause large diurnal dissolved oxygen fluctuations (Section 5.3.4).

Qualitative visual observations of the abundance of *P. crispus* at locations at sites DE and EF where the current velocity was 0.3 to 0.4 $m.s^{-1}$ at flows of 20 $m^3.s^{-1}$ showed no obvious change in biomass after flows of up to 180 $m^3.s^{-1}$ during March 1984. By contrast the growths of *C. glomerata* present at these sites on 22 February 1984 were substantially reduced by 4 April 1984. Previous studies also showed that river flows of 200 $m^3.s^{-1}$ caused substantial reduction of *C. glomerata* at sites upstream of the discharges (Freeman, 1983). This suggests that the densely growing macrophyte, with its strong stems and roots within the river bed, has more resistance to increasing current velocity than the algae.

The river peak flow data (Table 7.6) provide an explanation for this unusual development of *P. crispus* in the Manawatu River over 1983 and 1984. The data show that the number of spates

when the flow exceeded 500, 800 and 1000 $m^3.s^{-1}$ were substantially lower in the period 1982 to 1984 than over the rest of the period for which flow records were available. In 1983 no flows in excess of 800 $m^3.s^{-1}$ were recorded and during 1984 the maximum flow was 440 $m^3.s^{-1}$ (Watson, 1985).

This lack of periodic very high flows appears to have resulted in *P. crispus* growth being able to continue unchecked throughout 1983 and 1984 whereas in previous years any biomass developing was removed by the periodic large flushes.

These observations suggest that the occurrence of *P. crispus* in the study reach is a temporary phenomenon. However the relationship between the peak river flows, the development of *P. crispus*, and its effects on the ecology of the river system should be investigated further before any developments that would reduce the peak flows are considered in the future (e.g., water harvesting for irrigation or hydroelectric developments).

7.2.6 EFFECTS OF WATER QUALITY AND PHYSICAL FACTORS ON EPILITHON GROWTH RATES

7.2.6.1 Introduction

The surveys of the abundance and composition of benthic communities in the Manawatu River (Sections 7.2.2 and 7.2.4) showed that the wastewater discharges have significant impacts on heterotroph abundance and, during extended low flows, on phototroph abundance. However the qualitative nature of these observations prevents their use for determining epilithon growth rates under various sets of water quality and physical conditions. Such data are required for the prediction of the time interval within which problematic epilithic biomass concentrations develop under different conditions.

In this section the results of field investigations of the relationships between epilithon growth rates on concrete plates, organic material concentrations and physical conditions are presented. The upper surfaces of these plates have been shown to provide at least as good a growth surface for

heterotrophic epilithic organisms as the upper surfaces of natural stones in the Manawatu River (Section 4.3).

The aim of the studies was to quantify in the field the epilithon growth rates and community structures under a range of water quality, temperature and surface radiation conditions. The results obtained were intended to aid river management authorities by allowing the prediction of the time required for the development of epilithic biomass concentrations capable of causing unacceptable dissolved oxygen depletion (Section 5.4.7).

Most experiments were commenced with the intention of measuring the epilithon growth rates during periods of relatively steady flow conditions in order to minimise interpretational difficulties arising from variations in biomass sloughing due to current velocity changes (Section 4.3). However the frequent small spates which occurred in the river (Figs 7.1 and 7.2) caused many of the experiments to be terminated after only one sampling after four or five days incubation. Nevertheless a number of more prolonged studies of twenty-three to thirty-eight days duration were also conducted (Section 7.2.6.2).

The general procedures followed in each experiment are outlined in Section 3.5.1. During two experiments at site D the respiration and gross photosynthetic rates of the epilithon developing on the concrete plates were also measured (Section 5.3.2).

7.2.6.2 Prolonged Epilithon Growth Studies

The results of the growth studies, in which the plates were incubated for periods exceeding twenty days, are presented in Figures 7.8 to 7.12. The plates were incubated at locations at each site with similar depths and similar current velocities (Table 7.7). Efforts were made to place the plates at locations with current velocities within the optimum range for sewage fungus development on flat clean surfaces (Section 4.4) but localised changes in current flow resulted in sub-optimal current velocities during part of experiment PR_4.

TABLE 7.7 Summary of Temperature, Depth and Current Velocity Conditions at Substrate Incubation Sites during Growth Experiments

Experiment	Interval	Sites	Typical Current Velocity 50 mm above plates (range, $m.s^{-1}$)*	Typical Depth at plates (range, m)*	Average Temperature (°C)
PR_1	07/09/82-30/09/82	C	0.3-0.4	0.3-0.5	11.5
PR_2	21/12/83-15/12/83	D	0.3-0.45	0.35-0.55	18
PR_3	30/11/83-06/01/84	A	0.28-0.35	0.3-0.45	18
PR_3	30/11/83-06/01/84	B	0.28-0.38	0.3-0.45	18
PR_3	02/12/83-06/01/84	C	0.3-0.40	0.3-0.45	15
PR_4	16/01/84-23/02/84	A	0.28-0.35	0.3-0.5	19
PR_4	16/01/84-21/02/84	C	0.21-0.4	0.3-0.5	19
PR_4	16/01/84-21/02/84	DE	0.3-0.5	0.3-0.5	19
PR_4	16/01/84-21/02/84	EF	0.3-0.5	0.25-0.5	19
PR_5	23/02/84-04/04/84	A	0.34-0.3	0.35-0.5	17
PR_5	23/02/84-04/04/84	B	0.44-0.6	0.35-0.5	17
PR_5	23/02/84-04/04/84	C	0.36-0.64	0.35-0.5	17

* = values measured during sampling when flow conditions relatively low. Higher values occurred during spates.

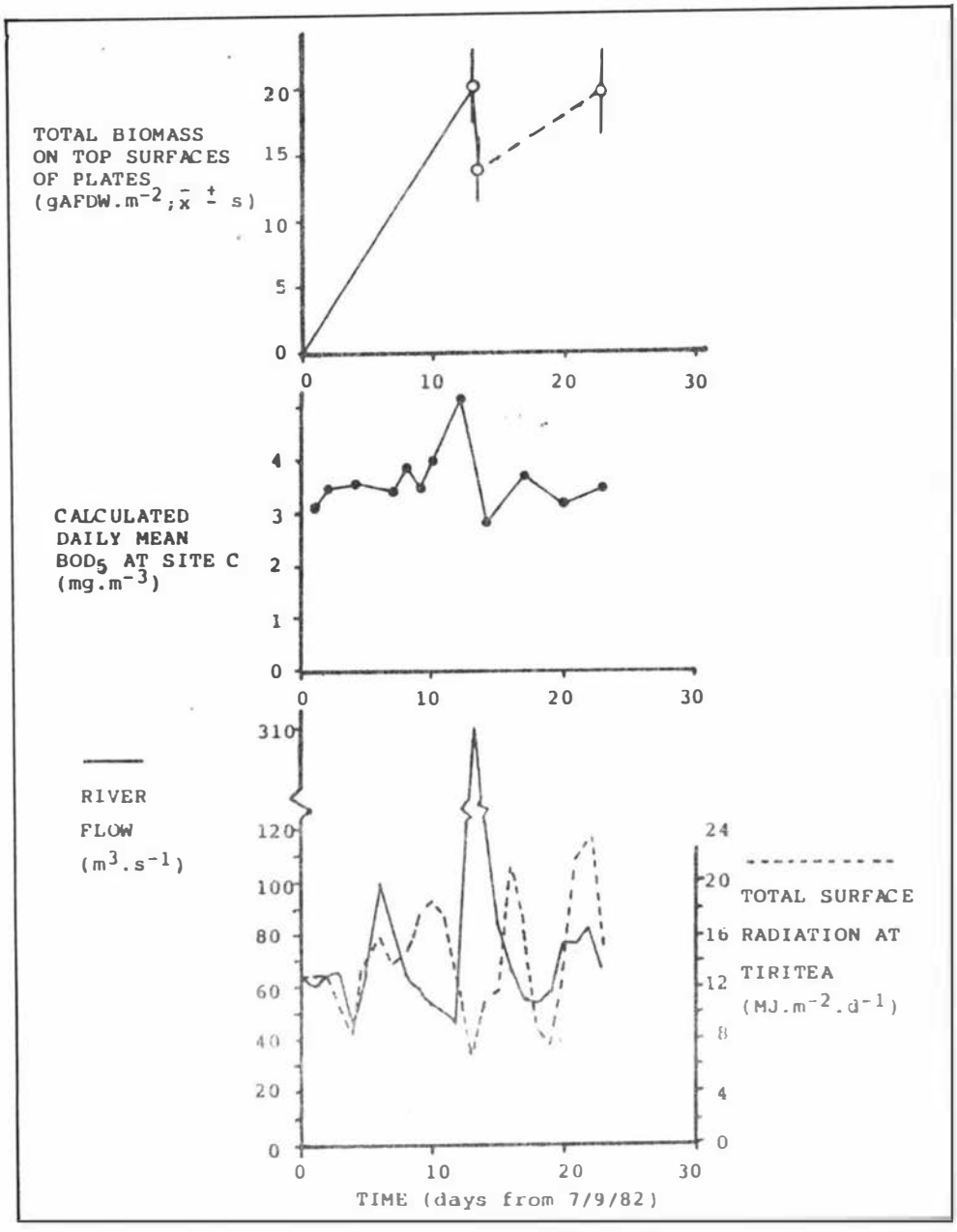


Figure 7.8: Epilithon Growth and Related Environmental Data During Experiment PR1, Site C, 7-30/9/82.

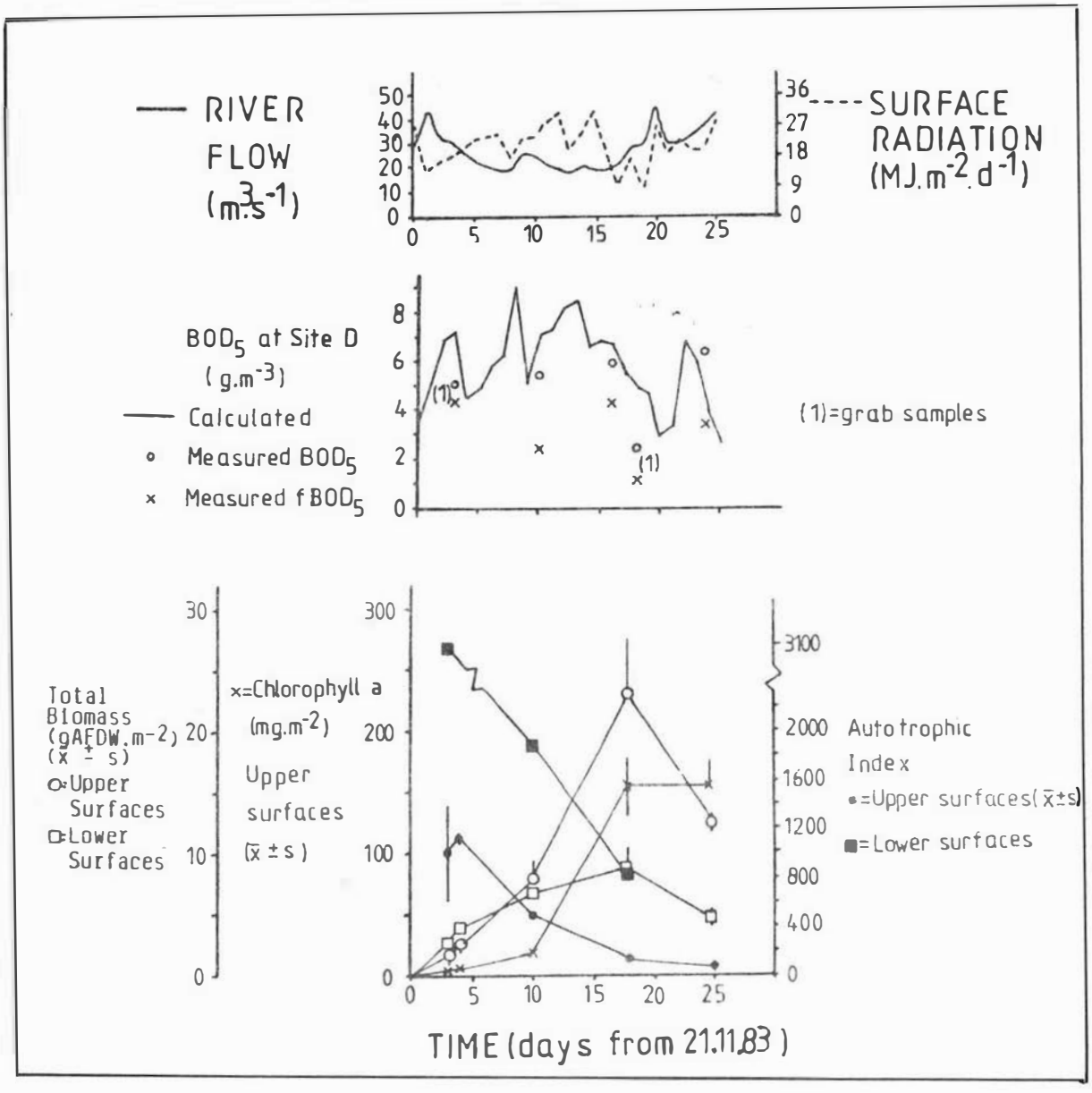


Figure 7.9: Epilithon Growth and Related Environmental Data During Experiment PR2.

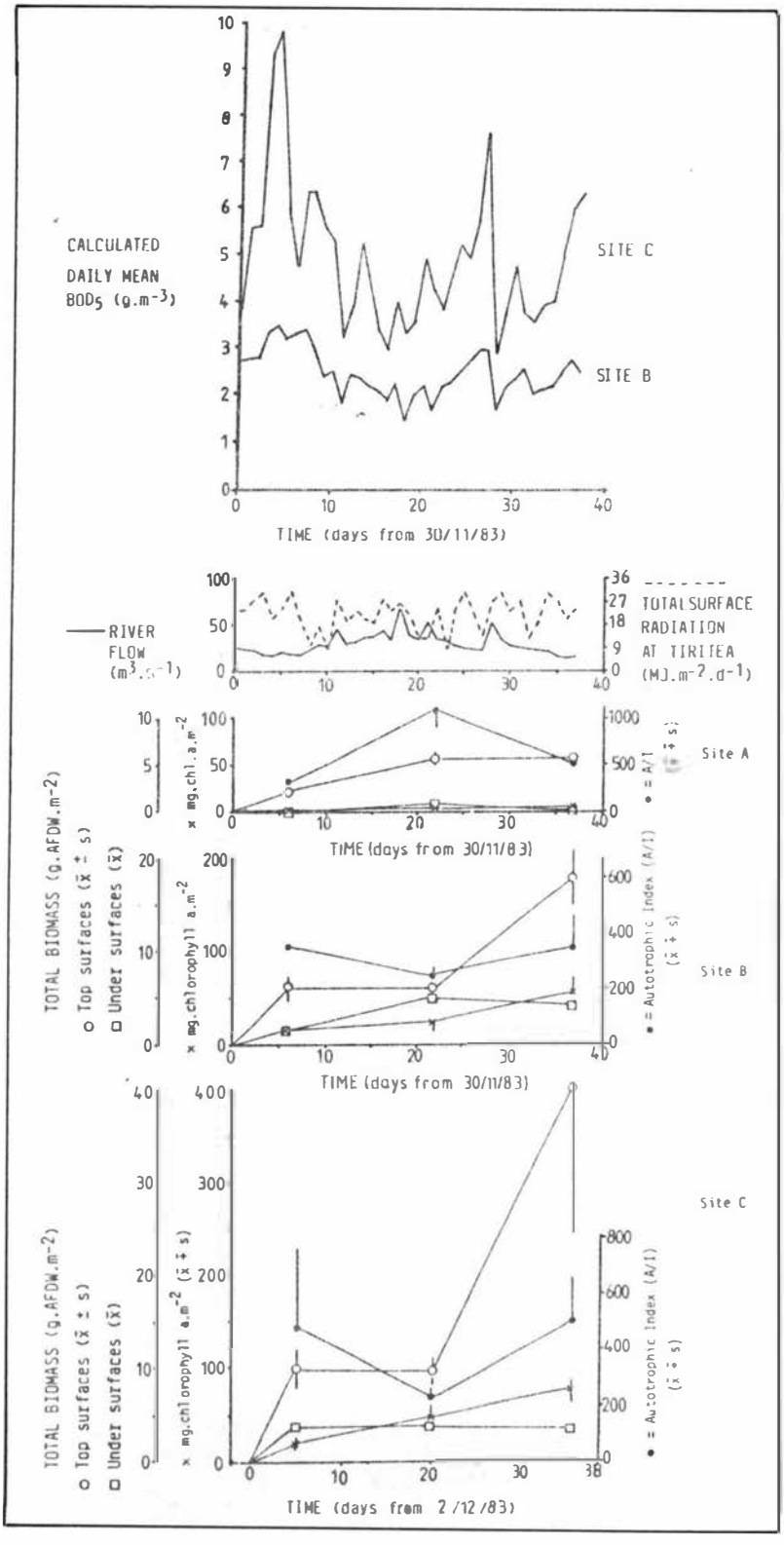


Figure 7.10: Epilithon Growth and Related Environmental Data During Experiment PR3.

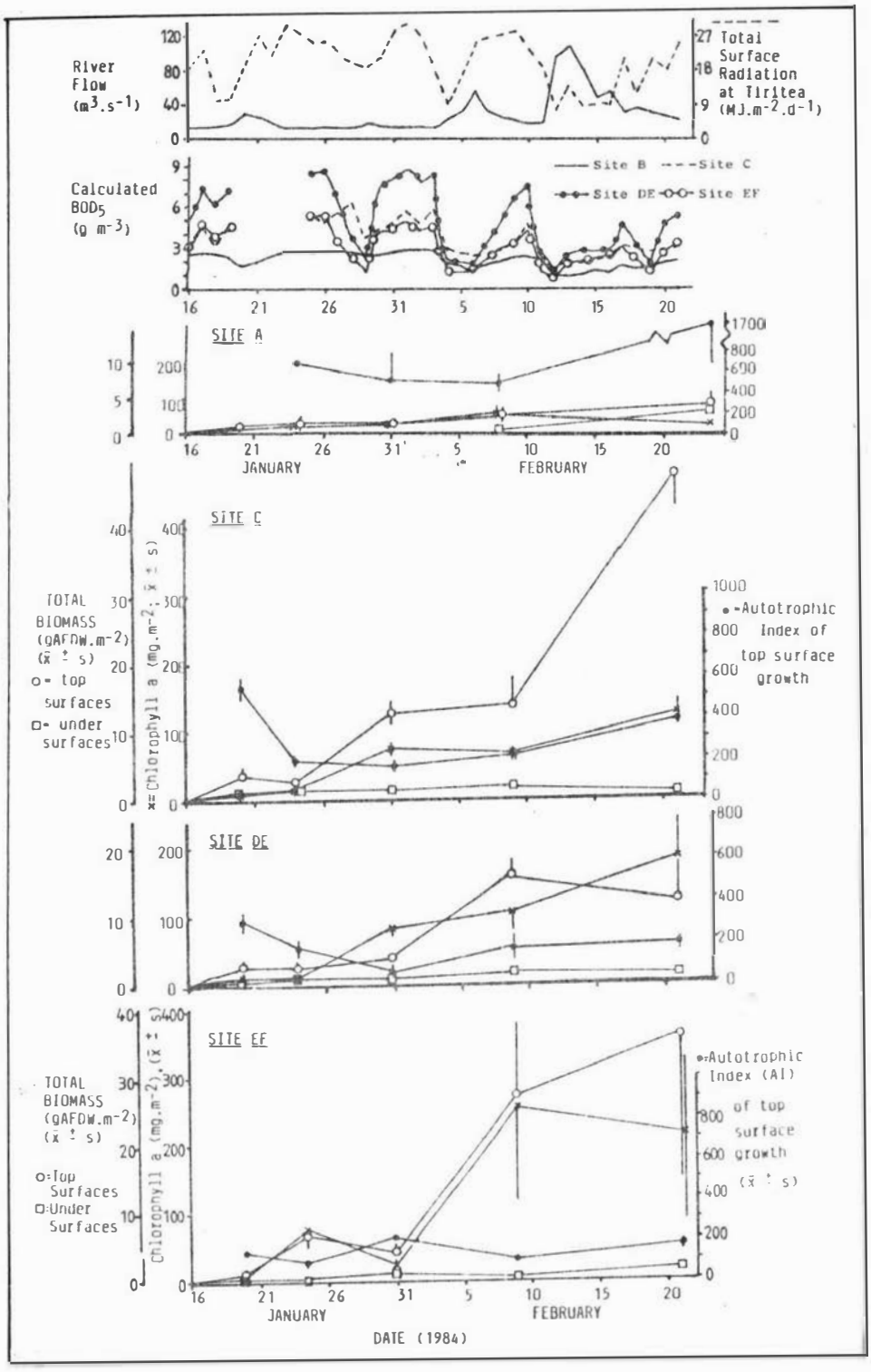


Figure 7.11: Epilithon Growth and Related Environmental Data During Experiment PR4.

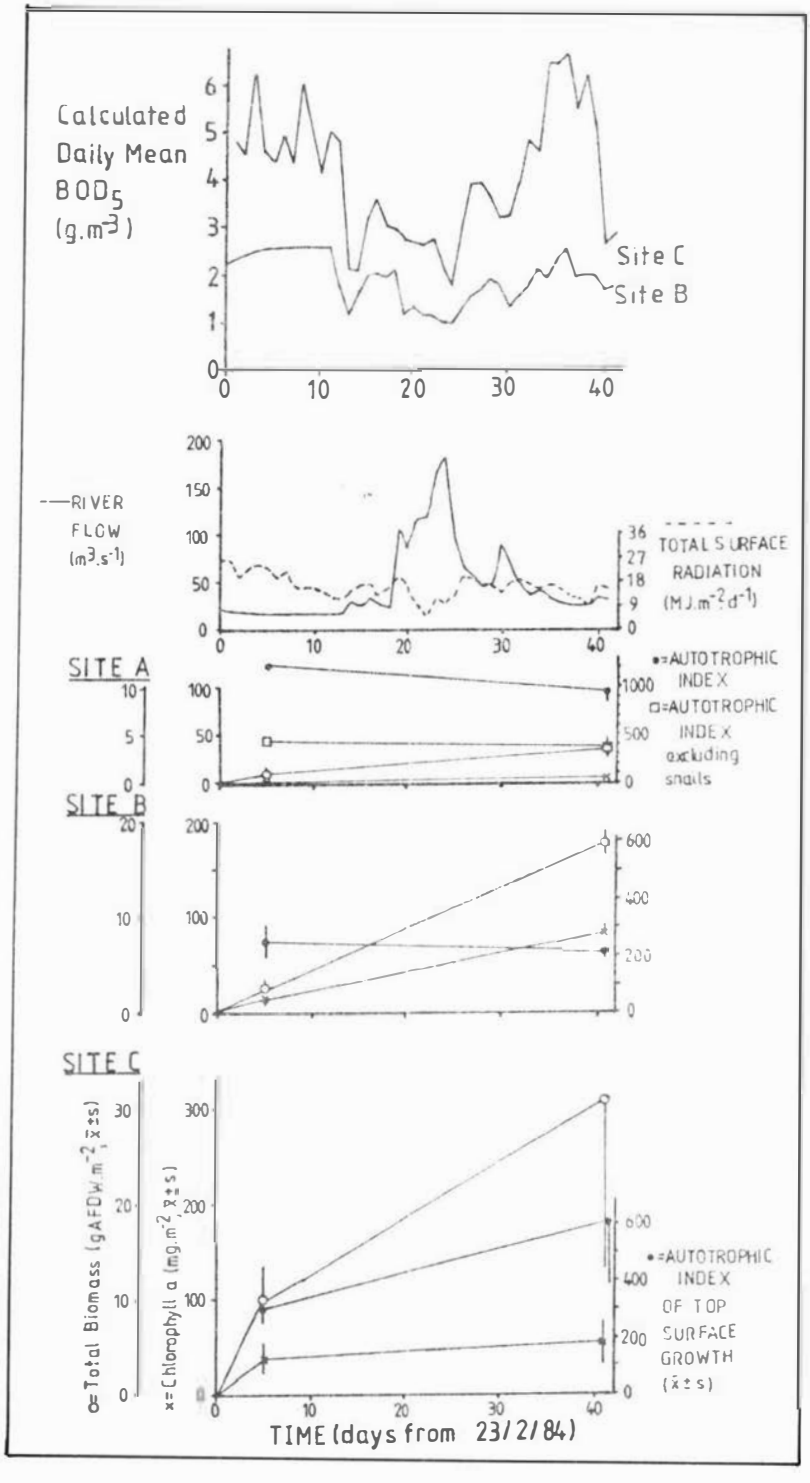


Figure 7.12: Epilithon Growth and Related Environmental Data During Experinent PR5.

Significantly higher rates (t tests; $P < .05$) of phototrophic growth (measured as chlorophyll a) were observed on plates at the sites below the discharges than at site A in each experiment during the 1983/84 summer period (Figs 7.10 to 7.12). Nutrient and BOD_5 analyses of water samples collected above site A at this time indicated low phosphorus and BOD_5 conditions (Table 7.5, Section 7.2.3) and the low biomass on the top surfaces of the plates at site A consisted of very light algal growth amongst settled detritus and snails (mostly *Potamopyrgus antipodarum*). The snails contributed significantly to the total biomass on the upper surfaces of the plates at site A, comprising 64% and 61% of the total biomass (as AFDW) on average on days 5 and 41 respectively of experiment PR_5 (Fig. 7.12). This high relative contribution of snails and detritus to the biomass at site A resulted in anomalously high autotrophic index (AI) values of 400 to 1700 (figs. 7.8 to 7.10) for a site without any obvious heterotrophic growth (Section 2.3.1). Removal of the snails reduced the AI value from approximately 1100 to 400 on one occasion (fig. 7.10) but it was not possible to further separate the epilithic biomass produced on the plate surface from settled detritus.

These results indicate that the epilithon AI value on its own is of little value as an indicator of the position of the community on the heterotrophic-phototrophic continuum (Section 7.3.1) in situations where settled detritus and invertebrates contribute much of the total biomass as AFDW. In such situations the heterotrophic index (HI) (Horner and Welch, 1981), which measures the ratio of ATP-derived organic carbon to chlorophyll a-derived organic carbon (Section 2.3.1), would be a more useful parameter.

At sites below the discharges increases in BOD_5 and nutrient levels produced rapid increases in both the total and algal biomass over periods of relatively stable or declining river flows (Figs 7.8 - 7.12, Table 7.8). In experiment PR_3 (Fig 7.10) the addition of sufficient primary treated domestic sewage to raise the BOD_5 from the background level at site A of

TABLE 7.8 Calculated Increases in River Nutrient Levels due to Wastewater Discharges .

Discharge	Nutrient	Calculated Mean Increase in River Nutrient Concentration upon mixing during Experiments* ($mg.m^{-3}$)				
		PR_1	PR_2	PR_3	PR_4	PR_5
PNCC	Phosphorus	35	80	76	25	48
PNCC	Nitrogen	232	538	507	136	320
MCDC	Phosphorus	38	62	43	33	39
MCDC	Nitrogen	105	169	118	58	107
BCWS	Phosphorus	0	7	8	19	11
BCWS	Nitrogen	0	49	27	140	84

* Assuming Wastewater BOD_5 loadings given in Table 7.1 and that:

(i) PNCC BOD_5 :N:P ratio
  (a) @ BOD_5 loading 6377 $kg.d^{-1}$ (Table 7.1) = 100:20:3 (Cooke *et al.*,1980)
  (b) @ BOD_5 loading = 3400 $kg.d^{-1}$ (Table 7.1) = 100:3:1.9 (measured) 29 /2-1/3/84, Appendix H)

(ii) MCDC BOD_5 :N:P ratio = 100:4.9:1.8 (mean ratio of three twenty-four hour composite samples collected 9-25/1/84, Appendix H)

(iii) BCWS BOD_5 :N:P ratio = 100:6:0.8 (Cooke *et al*,1980).

approximately 0.6 $g.m^{-3}$ (Table 7.5) to 3.2 $\pm$ 0.2 $g.m^{-3}$ ($\bar{x} \pm s$) at site B produced four and five-fold increases in the algal and total biomass developed after six days incubation respectively. The further addition of sufficient MCDC effluent to raise the average BOD_5 to 7.1 $\pm$ 2.3 ($\bar{x} \pm s$) at site C over the first five days incubation produced further significant increases (t tests; $P < .05$) in the plate upper surface biomass development rate with a heterotroph dominated, sewage fungus, community developing. Growth rates of 1.94 $gAFDW.m^{-2}.d^{-1}$ and 3.78 mg $Chla.m^{-2}.d^{-1}$ were calculated for this interval at site C compared with 1.0 g $AFDW.m^{-2}.d^{-1}$ and 2.5 mg $Chla.m^{-2}.d^{-1}$ over the first six days incubation at Site B. Between the last two sampling occasions in experiment PR_3 (Fig. 7.10) the calculated mean BOD_5 concentrations at site B and C, of 2.4 $\pm$ 0.3 $g.m^{-3}$ and 4.8 $\pm$ 1.3 $g.m^{-3}$ ($\bar{x} \pm s$) respectively, were lower than at the beginning of the experiment and lower total biomass growth rates, of 0.78 $gAFDW.m^{-2}.d$ and 1.64 $gAFDW.m^{-2}.d^{-1}$ respectively, were calculated. However higher sloughing losses expected due to the spate during the latter interval probably contributed to the lower rates observed.

Changes in current velocity were shown to have resulted in rapid sloughing of epilithic biomass (Section 4.4.3) in experiment PR_1 (fig 7.8). Epilithon biomass reduction following a smaller spate was also observed at the end of experiment PR_2 (fig. 7.9) and between the second and third sampling of experiment PR_3, when a number of small spates occurred, no net increase in total biomass was recorded at sites B or C (fig. 7.10). The results of these latter experiments indicate that spates cause greater sloughing of the fragile heterotrophs than the algae. In each case the epilithon remaining shortly after the spates had lower AI values and the algal biomass either remained the same or increased slightly (Figs 7.9 and 7.10).

A similar increase in epilithon growth at sites B and C to that observed in experiment PR_3 occurred in experiment PR_5. Over the first five days of this experiment the addition of sufficient domestic sewage, treated by primary sedimentation

with alum flocculation (Section 2.2.3.4), to give a calculated BOD_5 at site B of 2.3 ± 0.1 $g.m^{-3}$ ($\bar{x}$ ± s) gave an epilithon growth rate of 0.53 $gAFDW.m^{-2}.d^{-1}$ and at site C the further addition of sufficient MCDC waste water to raise the BOD_5 to 5.0 ± 0.8 $g.m^{-3}$ ($\bar{x}$ ± s) gave a significantly higher growth rate (t tests, $P < .05$) of 2.0 $gAFDW.m^{-2}.d^{-1}$ for this interval (Fig 7.12). However the epilithon at site C in this instance had lower AI values than that grown over five days in experiment PR_3 at the higher BOD_5 of 7.1 ± 2.3 ($\bar{x}$ ± s) (i.e. 300 ± 56 c.f. 543 ± 56 ($\bar{x}$ ± s) during experiment PR_3). This indicates that the lower heterotrophic growth in response to lower average BOD_5 conditions at site C in experiment PR_5 was compensated for by increased phototroph growth.

Over the first four days of experiment PR_4 low MCDC effluent discharge rates produced an only minor increase in BOD_5 at site C compared with site B (Fig 7.11) and the average BOD_5 of 4.0 ± 0.7 $g.m^{-3}$ ($\bar{x}$ ± s) gave a growth rate of 0.97 $gAFDW.m^{-2}.d^{-1}$ at site C. This rate was similar to that at site B, and half that at site C, at the beginning of experiment PR_3 (fig. 7.10).

Localised changes in the flow at the plates at site C in experiment PR_4 due to the development of the adjacent benthic communities, resulted in sub-optimal current velocities (section 4.4.2) 50 mm above the plate surfaces on 31/1/84 (0.21 $m.s^{-1}$) and 9/2/84 (0.24 $m.s^{-1}$) and heterotrophic growth was relatively low (fig. 7.10) compared with that observed under similar BOD_5 conditions during experiments PR_3 and PR_5 (figs. 7.10 and 7.12). By contrast between 9/2/84 and 21/2/84, when higher river flows resulted in current velocities in excess of the value of 0.3 $m.s^{-1}$, measured 50 mm above the plates on 21/2/84, growth was rapid (2.8 $gAFDW.m^{-2}.d^{-1}$) at BOD_5 concentrations similar to or less than those observed earlier in the experiment (i.e. 2.8 ± 0.9 $g.m^{-3}$, ($\bar{x}$ ± s)). The development of *Cladophora glomerata* filaments on the plates over this interval also contributed to the increase in biomass observed and would have further aided heterotroph development by providing a growth support above the benthic boundary layer (Section 4.4.2).

These results suggest that, at similar current velocities in the optimum range for heterotroph development, the epilithon growth rate at site C is related to the BOD_5. However the results of experiment PR_4 (fig 7.11) do not suggest any general relationship between BOD_5 due to different waste waters and heterotrophic growth rate. Lower epilithon growth occurred over the first four days of experiment PR_4 at site DE than at site C despite higher calculated mean BOD_5 values at the former site (fig. 7.11). There was also a general decline in the epilithon Autotrophic Index values downstream from site C on 20/1/84 and 21/2/84 after periods of optimal velocities for heterotrophic growth at all sites. This indicates less favourable conditions for heterotrophic growth at the sites downstream of site C despite similar or greater calculated BOD_5 concentrations.

The results of the wastewater ultrafiltration-fractionation studies (Table 5.24, section 5.5.3) provide an explanation for these observations. These results showed that the primary treated BCWS wastewater, which results in the increase in calculated BOD_5 over the values at site C at sites DF and EF, has a smaller proportion of dissolved and low molecular weight organic material than the MCDC waste water, which is discharged 1.8 km upstream of site C (Fig 1.2). Since the dissolved and, in particular, the low molecular weight components stimulate the heterotrophic growth (section 2.3.2), this explains why the comparatively high BOD_5 values at sites DE and EF due mainly to the BCWS discharge (fig. 7.10) have relatively minor effects on heterotrophic growth.

Throughout experiment PR_4 heterotrophs only occurred in trace amounts of site EF (section 7.2.2) indicating that the concentration of organic material suitable for their growth was low. This was supported by the $fBOD_5$ values of 1.3 to 2.7 $g.m^{-3}$ of daily composite water samples collected at site EF during this interval (Table 6.5). However the nitrogen and phosphorus levels were high (Table 6.6). Thus the water quality conditions at site EF resemble those that would be expected at site D if the wastewaters were treated biologically

to reduce their BOD_5 but not their nitrogen and phosphorus concentrations. This is the expected outcome of the PNCC and BCWS treatment systems (Sections 2.2.3.4 and 2.2.5.2).

Throughout experiment PR_4 phototrophs dominated the epilithon at site EF and the AI values were between 96 and 216 (fig. 7.9). However the species dominating the community varied considerably during the incubation period. On the first two sampling occasions a number of different diatom and unicellular green algal species were abundant but much of this community sloughed off the plates prior to 31/1/84. The interval 31/1/84 to 9/2/84 saw the rapid growth (2.62 $gAFDW.m^{-2}.d^{-1}$) of a community dominated by two filamentous species, the green algae *Stigeoclonium tenue* and the blue green bacteria *Oscillatoria* sp. amongst which a variety of unicellular green algae and diatoms were abundant. Between 9/2/84 and 21/2/84 this community was replaced by one dominated by *Cladophora glomerata*. This pattern of phototroph succession was similar to that observed at sites A and B during the summer of 1982/83 (Section 7.2.4) and further complicates the prediction of the time taken for the biomass of phototrophic dominated communities growing on clean substrates to reach nuisance levels.

Nevertheless the results at site EF during experiment PR_4 show that, under summer flow conditions, low organic concentrations and high nutrient levels produce rapid algal growth in the Manawatu River. Nuisance biomass concentrations of 34-45 $gAFDW.m^{-2}$ predicted for temperatures of 21°C by the oxygen modelling studies (Section 5.4.7), developed on the flat upper surfaces of the plates after 35 days incubation. However the use of the growth rates measured on the upper surface of the substrates to predict those on the natural bed of the river is complicated by the following factors:

(i) The roughness of the natural bed providing more surface area for attachment that is exposed directly to the sunlight and current per unit flat surface area than on the flat upper surfaces of the plates.

(ii) The development of biomass in some instances on the under

sides of surface stones and on subsurface gravels exposed to interstitial water flow.

Measurements (Section 3.4.3) showed that at sites C and D the exposed surface area to flat bed area ratios were 1.64 $\pm$ 0.15 and 1.62 $\pm$ 0.13 ($\bar{x} \pm s$) respectively and similar ratios would be expected at sites A to E where the stone size distributions are alike (section 4.2). This suggests that to convert the plate surface growth rates to natural bed growth rates at sites A to E in the Manawatu River study area (fig. 1.2) the former rates should be multiplied by a conversion factor of 1.63.

The growth on the undersides of the stones and amongst the subsurface gravels presents a more difficult problem.

In a number of experiments the slime layer which developed on the undersides of the plates at sites below the discharges was measured. This was always more heterotrophic (i.e. had higher AI values) than the upper plate surface growths (e.g. fig. 7.9) but accumulated some algal biomass, presumably by planktonic cells becoming embedded in the slime layer. The undersurface community was dominated by the dominant bacterial species of the upper surface growths (i.e. *S. natans* and *Flavobacterium* sp. (Section 7.2.3)) except at site EF during experiment PR_4, when the sessile ciliate *Opercularia* sp. was dominant, and at site A where any biomass present was due to snails (*Potamopyrgus antipodarum*) and insect larvae.

The under surface biomass reached a maximum at 9 gAFDW.m^{-2} in PR_2 (Fig 7.9) 4 to 5 gAFDW.m^{-2} in PR_3 (Fig 7.10) and 1.8 gAFDW.m^{-2} in PR_4 (Fig 7.11). Variations in this biomass would be expected to result from both changes in river water organic content and the permeability of the riverbed gravels. Thus the progressive clogging of the interstitial spaces expected during the extended periods of lower flows from November 1983 to February 1984, due to the accumulation of detritus and biomass, could explain the progressive reduction in maximum undersurface biomass observed in experiments PR_2 to PR_4 when the river water quality was fairly similar.

These results suggest that the undersides of surface stones and subsurface gravels exposed to interstitial flow provide suitable attachment sites for limited heterotrophic biomass development early during periods of stable flow conditions but become progressively less important with time. In view of the comparatively minor and variable importance of these attachment surfaces they were omitted when predicting the natural bed biomass under different conditions from the plate upper surface biomass data.

Multiplying the plate upper surface growth values by 1.63 gives bed equivalent biomass concentrations at site EF during experiment PR_4 (fig. 7.11) of 45 gAFDW.m^{-2} on 9/2/84 (day 24) and 58 gAFDW.m^{-2} on 21/2/84 (day 36). Thus, under the conditions at site EF during experiment PR_4 nuisance bed biomass concentrations of 34 to 45 gAFDW.m^{-2} are predicted 21 to 24 days on the natural bed.

Similarly the growth rates observed on the upper surfaces of plates incubated at site D in experiment PR_2 (fig. 7.9) indicate that this nuisance bed biomass concentration of 34 to 45 gAFDW.m^{-2} (= plate biomass of 21 to 28 gAFDW.m^{-2}) would be reached under study conditions in approximately 16 to 19 days. Under the conditions during the stable or declining flow periods of experiment PR_3 (fig. 7.10) the plate upper surface growth rates indicate that these biomass concentrations would develop at sites B and C in approximately 23 to 35 days and 10 to 13 days respectively. Periods of stable or declining low-flow conditions of over 25 days duration frequently occurred in the Manawatu River during the study period (Figures 7.1 and 7.2).

These results illustrate that epilithon develops very rapidly to concentrations that constitute a potential nuisance with respect to the river oxygen levels at the maximum observed summer temperature of 21°C. This nuisance level was reached most rapidly at BOD_5 levels of 5 to 7 g.m^{-3} at site C below the MCDC outfall where a heterotroph dominated, sewage fungus, community occurred on the upper surfaces of the plates.

Further downstream at site EF, where the concentration of organic material suitable for heterotrophic growth was low but nutrient levels high, phototroph dominated communities also reached this nuisance density during the growth studies, although their initial development was slow and a succession of community types occurred.

These results and the benthic biomass observation (Sections 7.2.1 and 7.2.7) show that the substantial reduction of the wastewater organic loadings without a simultaneous substantial reduction in the nutrient loadings has not prevented biomass concentrations developing which constitute a nuisance at summer temperatures of 21°C. Rather this has resulted in the replacement a heterotroph dominated biomass by a slower growing but still problematic phototrophic dominated biomass.

These latter communities may be more persistent during erratic flow conditions since the phototrophs are less readily dislodged from the bed by small spates than the heterotrophs. This is particularly true for the macrophytes whose biomass was not substantially affected by river flows of several hundred cubic metres per second.

Reduction of the wastewater nutrient concentrations to a level that would limit phototrophic development to below the summer temperature nuisance levels presents significant technical problems. Since the background phosphorus levels are often above those which limit growth during summer (Freeman, 1983, Table 7.4) and phosphorus uptake rates are relatively low compared with those of BOD_5 (Sections 6.2.1.3 and 6.2.2.2), almost complete removal of phosphorus from the wastewaters would be required. Nitrogen and phosphorus removals from domestic sewage of 70 to 80% are achievable using modified activated sludge treatment processes (e.g., Bardenpho and Phosphostrip processes) (McFarlane, 1982). However only minor phosphorus removal (less than 30%) is expected in the current PNCC aerated lagoon secondary treatment (Stall et al., 1976) and no significant phosphorus removal is expected in the BCWS anaerobic treatment ponds (Section 2.2.5.2).

This suggests that it is unlikely that epilithon biomass can be limited to levels below those causing the river classification oxygen concentration to be breached during summer low flows at temperatures of 21°C unless land disposal of the wastewaters is adopted. Further assessment of this by studies on the relationship between phototrophic epilithon growth and nutrient concentrations achievable by the best available technology would be useful.

These results are discussed further along with those of the short-term growth studies in Section (7.2.6.3).

7.2.6.3 Results of Short-term Epilithon Growth Studies

The highly erratic river flow conditions between October 1982 and March 1983 (fig. 7.1) resulted in the epilithon growth experiments conducted during this interval being terminated after three to eight days. Nevertheless the results of these experiments (SR_1 to SR_7) presented in Figures 7.13 to 7.19, provide quantitative information on the effects of the wastewater discharges on epilithon growth.

In order to simplify the comparison of the results from experiments of three to eight days duration, the data are presented as growth rates calculated assuming a linear increase in biomass with time. The data from the prolonged growth experiment PR_2 (fig. 7.9) indicate that this assumption is reasonable for heterotroph dominated epilithon.

As was observed in the prolonged growth experiments (Section 7.2.6.2), the growth rates at site A, as $gAFDW.m^{-2}d.^{-1}$, were always significantly lower (t tests, $P < .01$) than those at sites below the discharges. However during experiment SR_6 the settled volume development rate at site A was significantly greater (t tests, $P < .05$) than at sites B and C (fig. 7.18). On this occasion, when a bloom of the stalked diatom *Gomphonema* sp. occurred at site A, the growth rate of 0.81 ± 0.11 $gAFDW.m^{-2}.d^{-1}$ ($x \pm s$) was much higher than the average rate of all the other experiments ($x \pm s = 0.15 \pm 0.16$

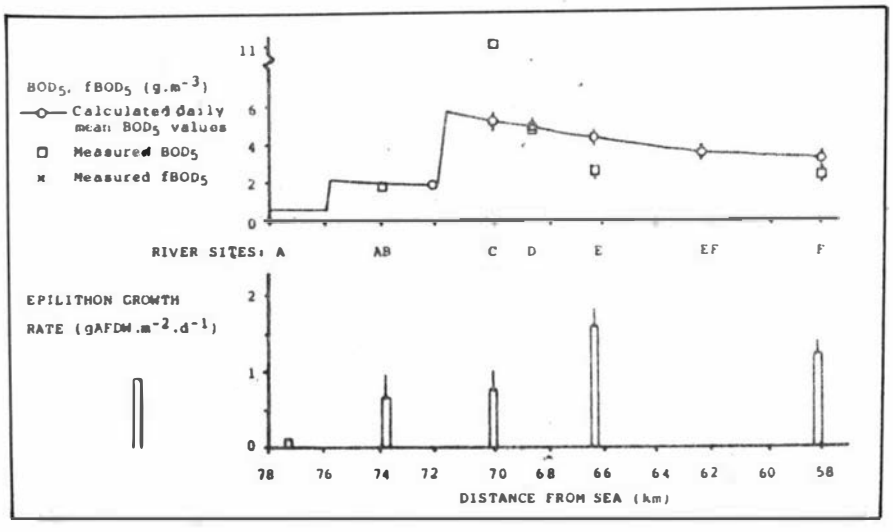


FIGURE 7.13:Results of Epilithon Growth Experiment SR1: 4-8/10/82; average temperature = 12°C; flow = 48.1 ± 7.1 $m^3.s^{-1}$; total surface radiation= 19.0 ± 6.0 $MJ.m^{-2}.d^{-1}$.

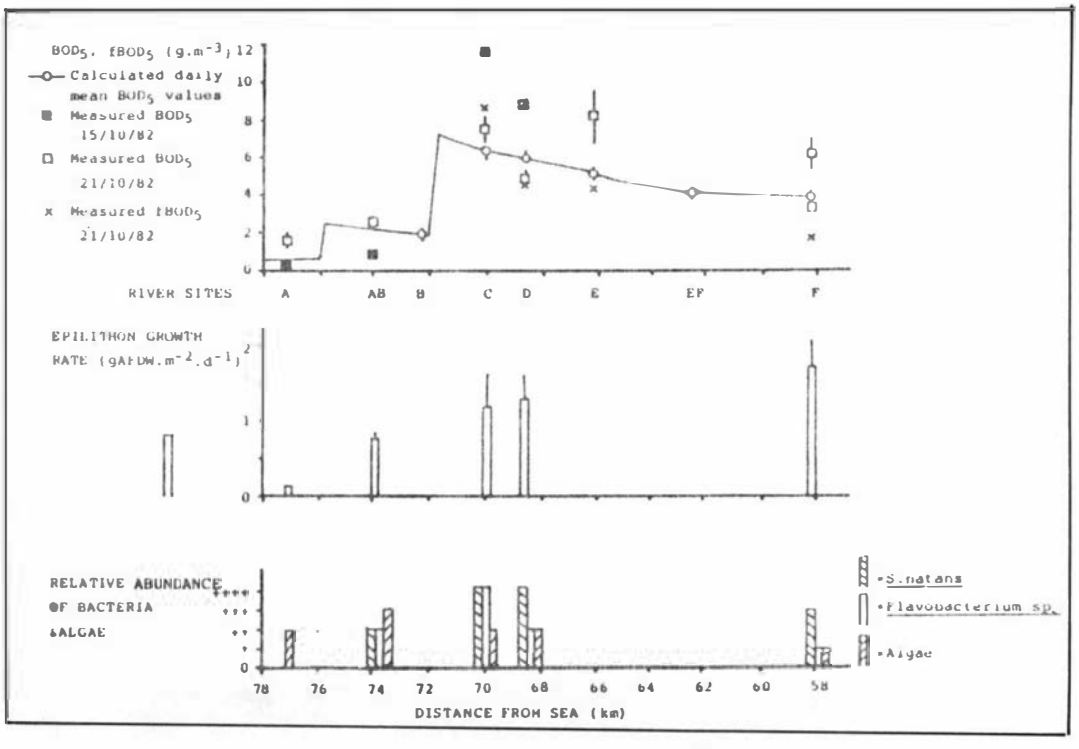


FIGURE 7.14: Results of Epilithon Growth Experiment SR2: 14-19/10/82; average temperature=12°C; flow=41.9 ± 5.1 $m^3.s^{-1}$; total surface radiation = 18.5 ± 4.3 $MJ.m^{-2}.d^{-1}$.

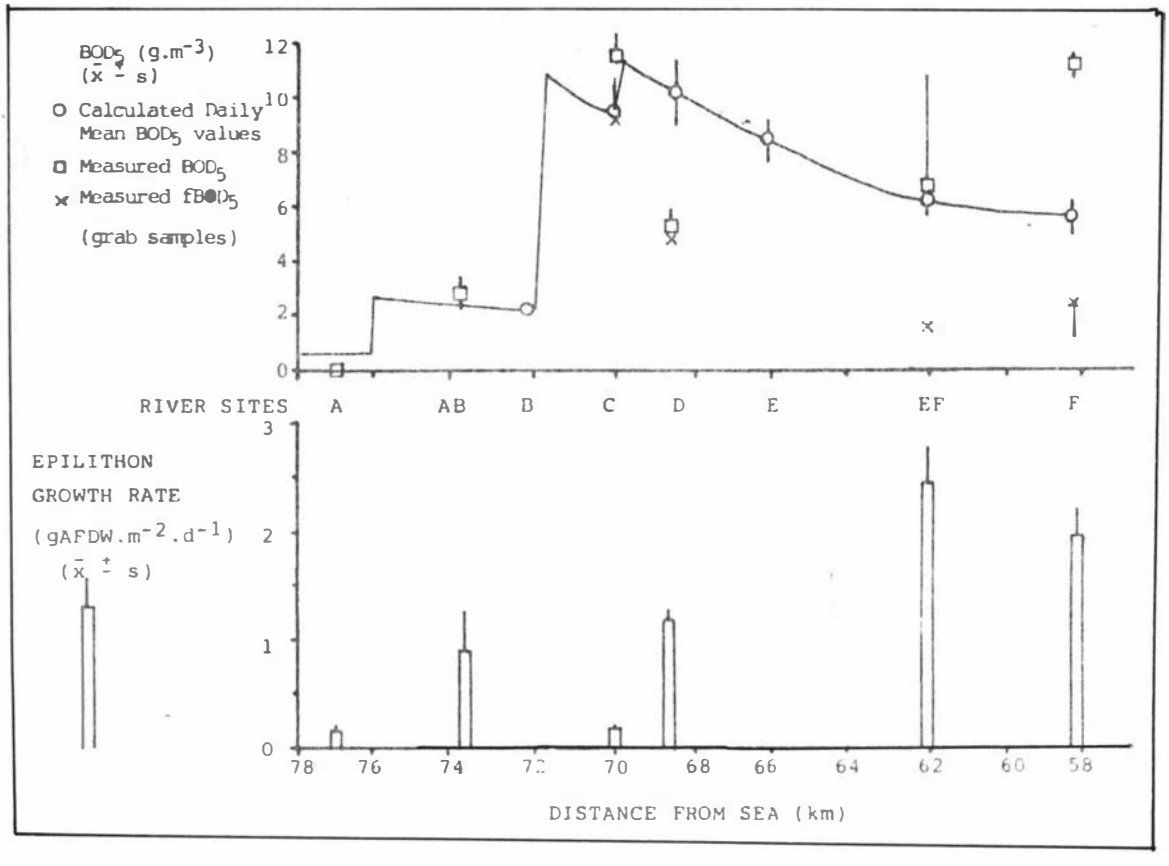


FIGURE 7.15: Results of Epilithon Growth Experiment SR3: 22-26/11/82; average temperature= 16°C; flow= 34.8 ± 5.2 $m^3.s^{-1}$; total surface radiation = 22.9 ± 6.7 $MJ.m^{-2}.d^{-1}$.

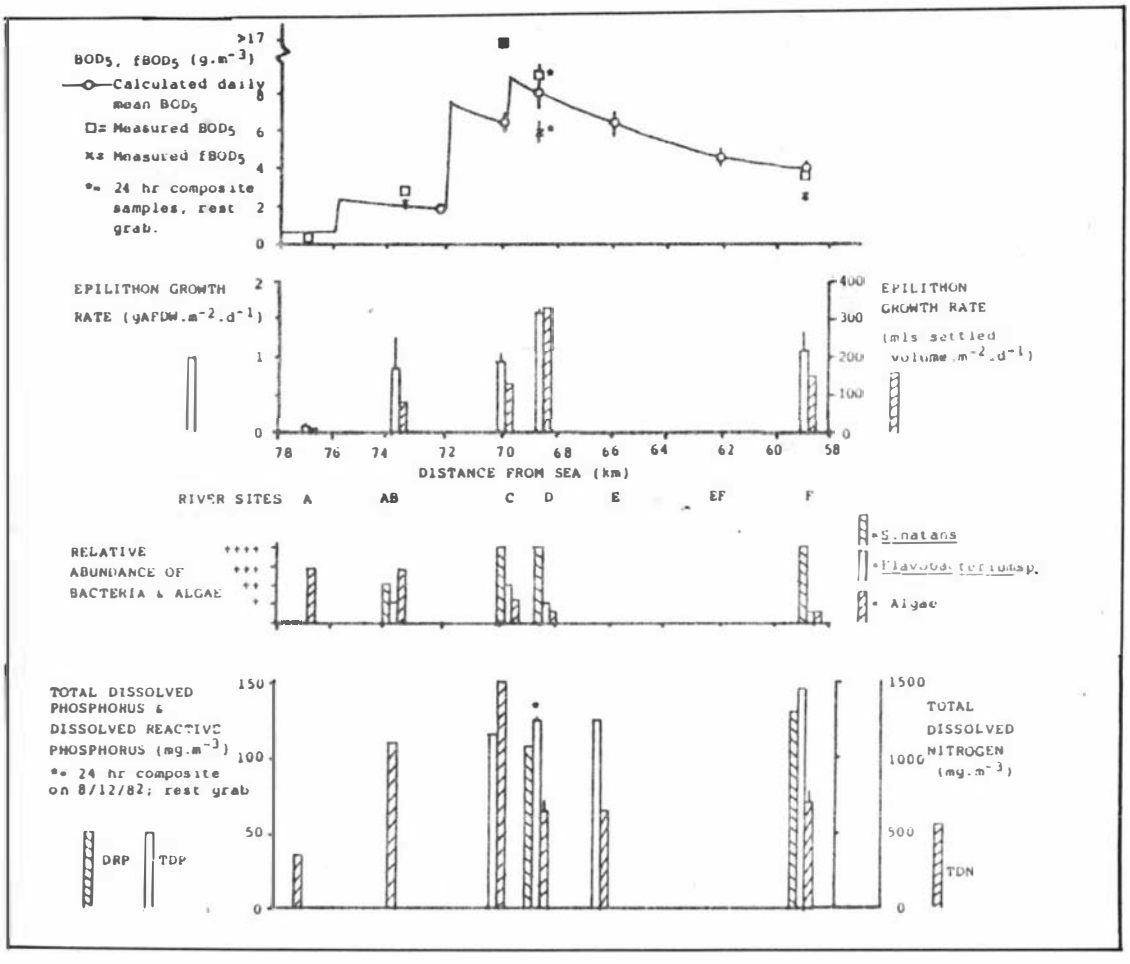


FIGURE 7.16: Results of Epilithon Growth Experiment SR4: 6-10/12/82; average temperature = 18°C ;flow= 42.5 ± 7.0 $m^3.s^{-1}$;. total surface radiation = 28.3 ± 3.4 $MJ.m^{-2}.d^{-1}$.

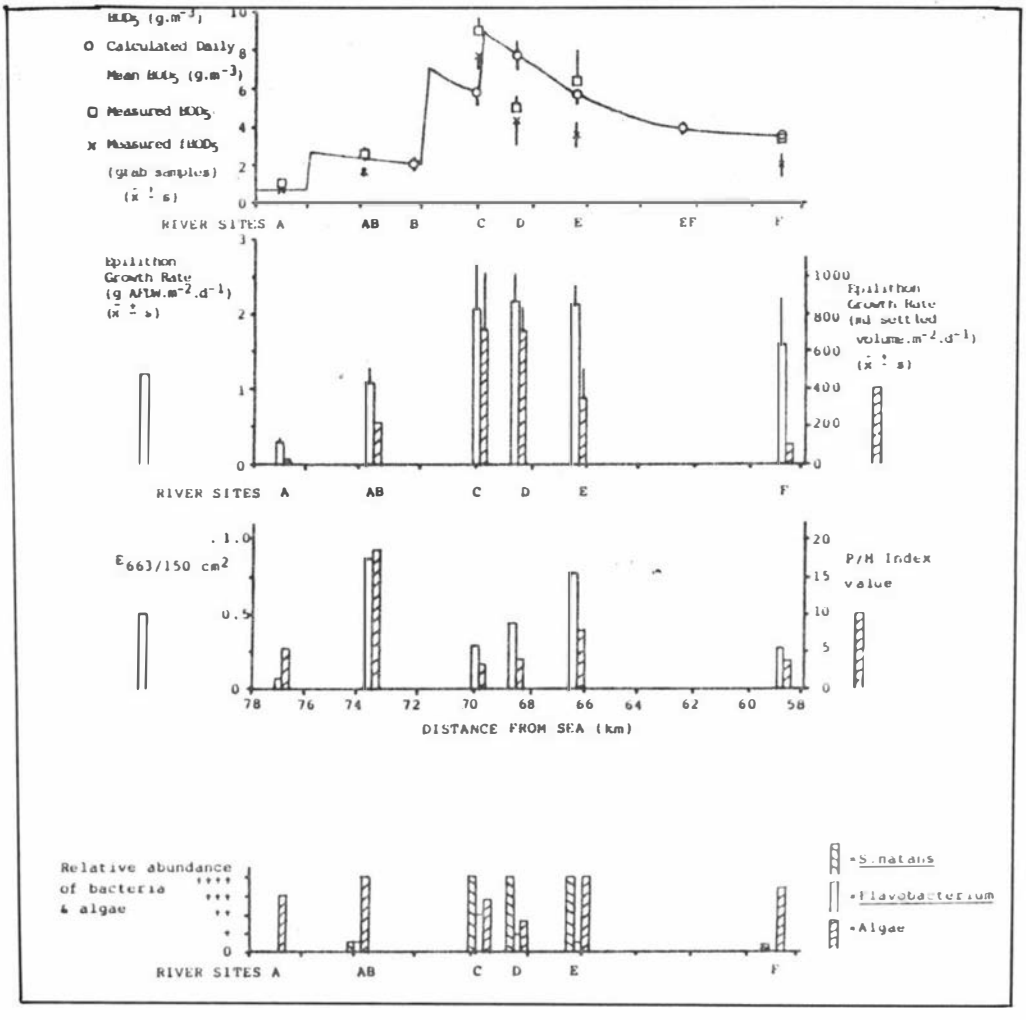


FIGURE 7.17: Results of Epilthon Growth Experiment SR5: 10-13/1/83; average temperature= 21°C; flow= 35.3 ± 3.5 $m^3.s^{-1}$; total surface radiation= 21.9 ± 6.1 $MJ.m^{-2}.d^{-1}$. Background nutrient levels at site A: DRP=12.2 $mg.m^{-3}$, dissolved inorganic nitrogen = 357 $mg.m^{-3}$ (Freeman,1983).

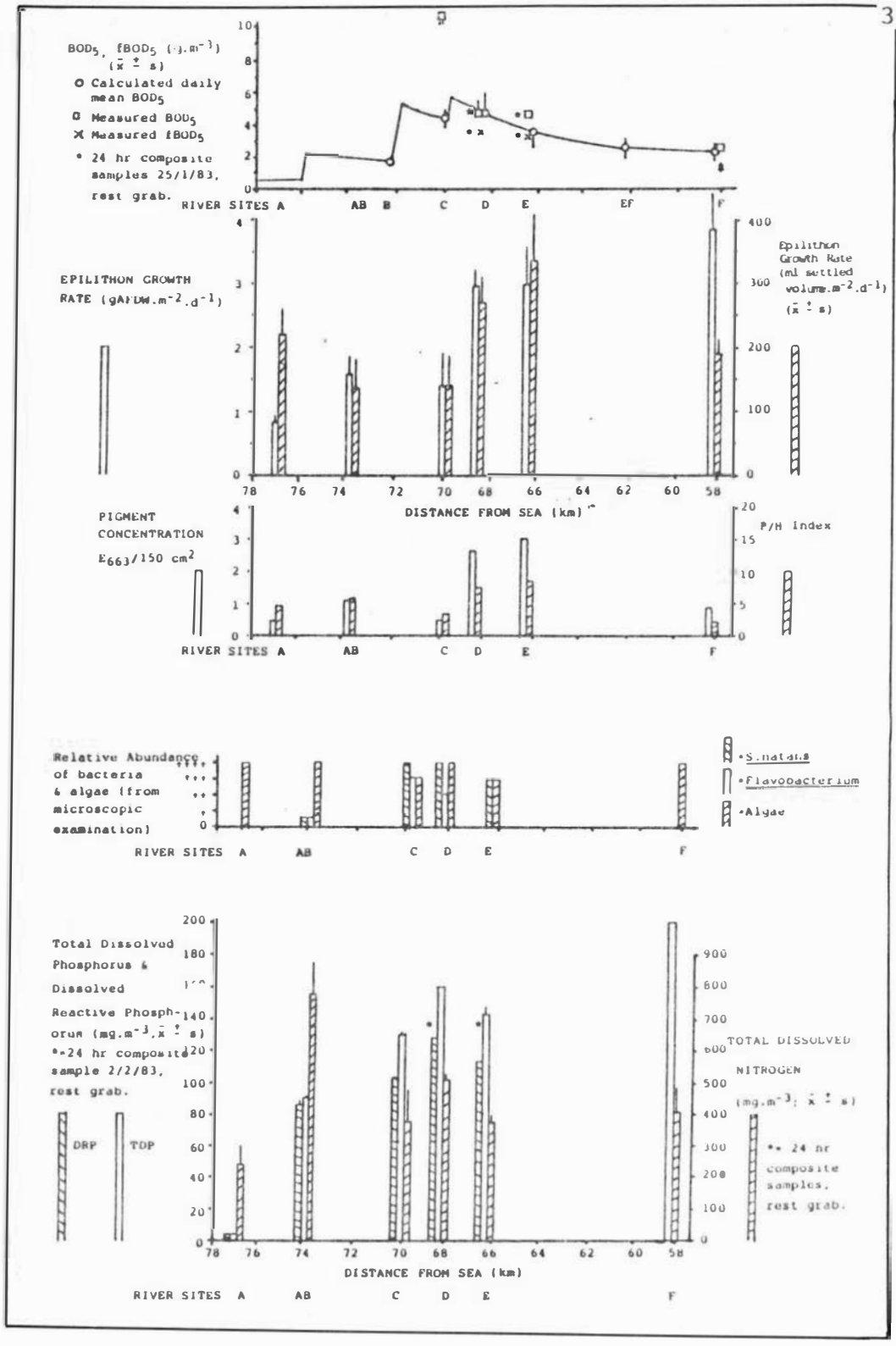


FIGURE 7.18: Results of Epilithon Growth Experiment SR6; 23-31/1/83, average temperature= 19°C, flow= 45.9 ± 7.6 $m^3.s^{-1}$, total surface radiation= 24.1 ± 3.9 $MJ.m^{-2}.d^{-1}$.

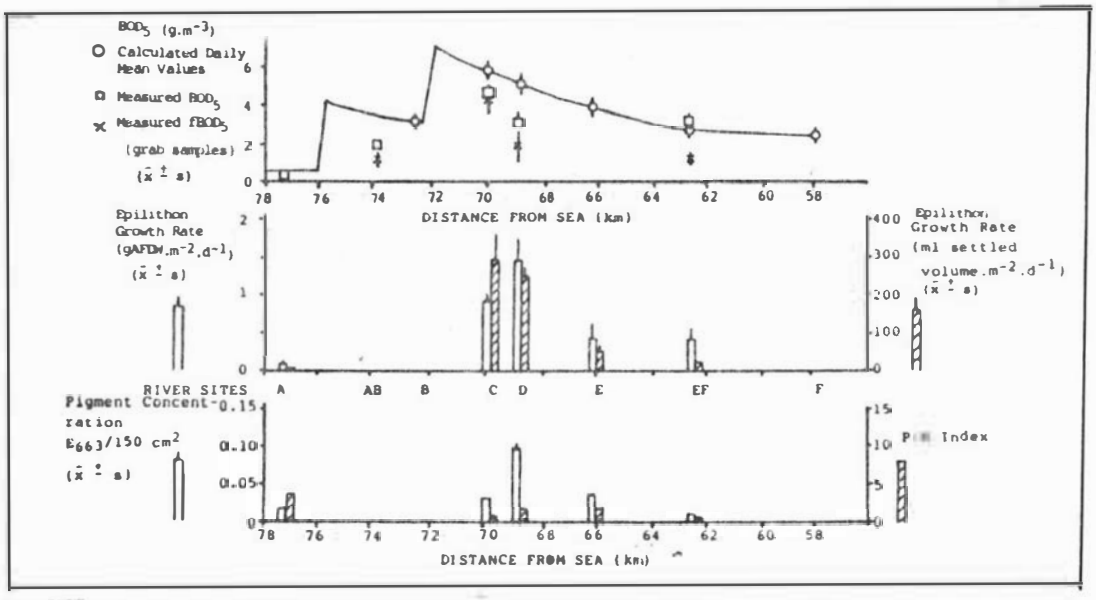


FIGURE 7.19: Results of Epilithon Growth Experiment SR7; 21-24/3/83; Temperature = 16-17°C; Flow= 19.9 ± 2.0 $m^3.s^{-1}$; Total Surface Radiation= 15.5 ± 5.2 $MJ.m^{-2}.d^{-1}$; no BCWS discharge.

gAFDW.m^{-2}.d^{-1}). The low density of the plate biomass at site A compared with that of the downstream sites resulted from the larger apparent contribution of *Gomphonema* sp., which is attached to the bed by a low density stalk, to the total biomass at this site. At the sites below the discharges *Gomphonema* sp. was less dominant and a variety of stalk-less algae, as well as *S. natans* and *Flavobacterium* sp. , were abundant. The relatively low P/H index values (Section 2.3.1) recorded at site A in experiment SR_6 (Fig 7.18) despite phototroph dominance probably also results from the relatively large contribution of the non-pigmented *Gomphonema* stalks to the total biomass.

The physical and water quality conditions during experiment SR_6 do not provide any explanation for the unusual, prolific growth of *Gomphonema* sp. at site A. The DRP concentration was low (Fig 7.18) and similar light and temperature conditions occurred on several other occasions (Table 7.9). However the combination of physical, water quality and biological factors clearly favoured *Gomphonema* sp. development.

The addition of sufficient primary treated domestic sewage to increase the river BOD_5 by approximately 1.5 to 2.1 g.m^{-3} upon mixing did not result in heterotrophic domination of the plate upper surface epilithon at site AB, 2.2 km below the PNCC discharge, in any of the experiments. Filamentous bacterial species were present amongst the epilithon that developed but phototrophs were dominant (e.g. figs. 7.14, 7.16 - 7.18). These observations are consistent with the results of outdoor channel studies on the effects of domestic sewage on benthic community development at current velocities similar to those above the plates (Wuhrmunn, 1954; Eichenberger, 1975).

The BOD_5 and nutrients (Table 7.10) added by the PNCC discharge resulted in significantly higher (t tests, $P < .05$) epilithon growth rates, measured as gAFDW.m^{-2}.d^{-1}, than at site A during each experiment. The growth rate at site B increased from 0.67 $\pm$ 0.28 gAFDW.m^{-2}.d^{-1} ($\bar{x} \pm s$) during early spring (experiment SR_1, fig. 7.13) to 1.57 $\pm$ 0.30 gAFDW.m^{-2}.d^{-1} ($\bar{x} \pm s$) in mid-

TABLE 7.9 Physical Conditions During Short-term Epilithon Growth Studies

Experiment	Dates	Riverflow ($\bar{x} \pm s$) ($m^3.s^{-1}$)	Total Surface Radiation at Ohakea ($MJ.m^{-2}.d^{-1}$) ($\bar{x} \pm s$)	Average River Temperature (°C)
SR_1	04-08-10/82	48.1 ± 7.1	19.0 ± 6.0	12
SR_2	14-19/10/82	41.9 ± 5.1	18.5 ± 4.3	12
SR_3	22-26/11/82	34.8 ± 5.2	22.9 ± 6.7	16
SR_4	06/10/12/82	42.5 ± 7.0	28.3 ± 3.4	18
SR_5	10-13/01/83	35.3 ± 3.5	21.9 ± 6.1	21
SR_6	23-31/01/83	45.9 ± 7.6	24.1 ± 3.9	19
SR_7	21-24/03/84	19.9 ± 2.0	15.5 ± 5.2	17.5

TABLE 7.10 Calculated Increases in River Nutrient Levels due to Wastewater Discharges

		Mean Increases During Experiments ($mg.m^{-3}$)						
Discharge	Nutrient	SR_1	SR_2	SR_3	SR_4	SR_5	SR_6	SR_7
PNCC	Phosphorus	46	52	63	52	63	48	111
	Nitrogen	305	350	424	347	418	322	741
MCDC	Phosphorus	81	114	191	123	110	75	77
	Nitrogen	104	145	244	157	140	95	98
BCWS	Phosphorus	0	0	14	20	26	8	0
	Nitrogen	0	0	102	153	195	58	0

Assuming: Wastewater BOD_5 loadings given in Table 7.1 and that:

(i) PNCC BOD_5:N:P ratio = 100:20:3 (Cooke *et al.*, 1980)
(ii) MCDC BOD_5:N:P ratio = 100:2.8:2.2 (mean ratio of seven 24 hour composite samples collected 31/12/82-8/5/83, Appendix H)
(iii) BCWS BOD_5:N:P ratio = 100:6:0.8 (Cooke *et al.*, 1980).

summer (experiment SR_6, fig. 7.18) under similar water discharge and flow conditions. Since epilithon gross primary production increased with light and temperature (Section 5.3.5), the increases in these parameters (Table 7.1) probably account for the seasonal variation in the growth rate.

The effect of the MCDC dairy factory wastewater discharge on epilithon development was complex, resulting in growth enhancement on some occasions and inhibition on others.

In experiments SR_2 (Fig 7.14) and SR_5 (Fig 7.17) the further addition of sufficient MCDC effluent to increase the mean river BOD_5 concentration after mixing by 5.2 and 5.0 $g.m^{-3}$ respectively resulted in significantly higher epilithon growth rates at site C than at site A/B above the MCDC outfall (t tests, $P < 0.1$ and $P < .002$, respectively). By contrast in experiments SR_1 and SR_4 MCDC BOD_5 additions sufficient to increase the mean river BOD_5 by 3.7 and 5.6 $g.m^{-3}$ respectively on mixing did not significantly increase the epilithon growth at site C over that at site A/B (t tests, $P > 0.1$). However on these occasions the growth rate at sites E and F further downstream were significantly greater than at site B and C (t tests; $P < .05$) despite the absence of further significant discharges in experiment SR_1 (fig. 7.13) and the comparatively minor increase in BOD_5 due to the BCWS discharge in experiment SR_4 (fig. 7.16).

In experiment SR_3 (fig. 7.15) addition of sufficent MCDC effluent to increase the mean river BOD_5 after mixing by 8.7 ± 1.5 $g.m^{-3}$ ($\bar{x} \pm s$) resulted in a significant decrease in epilithon growth rate compared with that at site A/B (t tests, $P < .002$) with rates similar to those at site A recorded (fig. 7.15). The bed biomass was also very low at site C at this time (section 7.2.2). Epilithon growth rates increased progressively at the downstream sites D and EF and was also high at site F. As discussed previously (section 7.2.2), the results of the laboratory channel studies (section 5.5.3 and 7.3) and river biomass observations (section 7.2.2) indicate that the addition, just below site C, of sufficient BCWS

primary meatworks effluent to increase the river BOD_5 by 1.7 $g.m^{-3}$ on average made only a minor contribution to the substantial increase in epilithon growth observed. Rather these results and those of experiments SR_1 and SR_4 suggest that epilithon growth was sometimes inhibited by components of the MCDC waste water but epilithon grew well further downstream where the inhibitory compounds would be expected to be reduced by self-purification processes. The high BOD_5 of grab samples collected during the morning at site C compared with the mean levels calculated from the daily BOD_5 discharge data (figs. 7.13 - 7.18) and the diurnal effluent discharge data (e.g. Table 5.14) show that, due to the highly variable nature of the MCDC discharge, the wastewater concentrations at site C were often two to three times the mean values.

It was not possible to discern the effects of the BCWS discharge on epilithon observed during the short term growth studies due to the marked effect of the MCDC wastewater discharged 1.8 km upstream of the BCWS outfall. The individual effects of these two wastewaters were compared in laboratory channel experiments (section 7.3). The comparison of the results of experiments SR_1, SR_6 and SR_7 demonstrates the effects of seasonal factors on the length of the river reach over which rapid heterotrophic growth occurred for similar wastewater loadings. In experiment SR_1 (fig. 7.13) during early spring (temperature (t) = 12°C, total solar radation = 19.0 ± 6.0 $MJ.m^{-2}.d^{-1}$ ($\bar{x} \pm s$) Table 7.9)) heterotroph dominated growth occurred at sites C to F (Fig 1.2). By contrast during experiment SR_6 (T = 19°C, TSR = 24.1 ± 3.9 $MJ.m^{-2}.d^{-1}$ ($\bar{x} \pm s$)) the addition at a similar river flow, of similar amounts of BOD_5 by the PNCC and MCDC plus an additional loading from the BCWS did not result in filamentous bacterial growth at site F (fig. 7.18). The low epilithon P/H values at site F were evidently due to the large amount of settled detritus and silt which accumulated on the plates causing high AFDW values. This also resulted in the higher density (i.e. AFDW : settled volume ratio) of the biomass at site F compared with that of the upstream sites (Fig 7.18). Comparison of the calculated mean BOD_5 data for the two experiments (Figs 7.13 and 7.18) indicates that the more rapid BOD_5 removal rate at the higher

temperature in SR_6 was important in causing the contraction of the zone of prolific heterotroph in growth observed in this experiment.

In addition to the reduction in the length of heterotroph dominated river reach, an increase in the epilithon growth rates was observed at site B, C and E during the summer experiment (SR_6).

In experiment SR_7 (Fig 7.19) the calculated BOD_5 increase due to the PNCC discharge was higher than during SR_1 whilst the BOD_5 increase due to the MCDC discharge was similar. The temperature during SR_7 (17°C) was higher than that during SR_1 (12°C) but the total surface radiation was 3.5 $MJ.m^{-2}.d^{-1}$ lower (Table 7.9). These conditions produced more rapid heterotrophic dominated growth at sites C and D than in experiment SR_1 but comparatively slow growth at sites E and F. In this instance the reduction of the length of the river reach affected by rapid heterotrophic growth (i.e. 5 km c.f. > 13 km during experiment SR_1) was partly due to the reduced river flow, increasing the travel times between the sites thus allowing more time for organic material removal. Equation 2.1 (Section 2.2.1.1) predicts that the travel times increase by 30% for a reduction in river flow from the mean value during experiment SR_1 (48.1 $m.^3s^{-1}$) to that in experiment SR_7 (19.9 $m^3.s^{-1}$). Allowing for this, other seasonal factors reduced the length of the river reach affected by rapid heterotrophic epilithon growth in experiments SR_7 compared with experiment SR_1 by at least 4 km (i.e. (13 x 0.7) - 5 km). The effects of temperature on the BOD_5 removal rates would be expected to be important but temperature effects on grazing activity, and seasonal variations in grazing numbers may have also been involved. Solar radiation inhibition (Sections 2.3.4.3 and 4.5) was not likely to have been important in producing the observed differences between these experiments since the average total surface radiation values were lower during experiment SR_7 than during SR_1 (Table 7.9). Seasonal variations in the extent of the river reach containing microscopic heterotroph dominated growths (sewage fungus) were

also observed in the Manawatu River (Section 7.2.2), a number of other rivers (Cawley, 1958; Amberg and Cormack, 1959; Phaup and Gannon, 1967; McKeown, 1983), and outdoor channel studies (Eichenberger and Wuhrmann, 1966).

In several experiments the measured BOD_5 values were much greater than the $fBOD_5$ values indicating that much of the total organic material degradable in the BOD_5 test was particulate rather than dissolved (figs. 7.14 - 7.19) and therefore not available to the sewage fungus matrix forming heterotrophs. This demonstrates that the simultaneous measurement of the river water BOD_5 and $fBOD_5$ concentrations gives a much clearer understanding of the nature of the organic material present and its likely effects on benthic community development than either parameter alone.

7.3 LABORATORY CHANNEL STUDIES

7.3.1 INTRODUCTION

The location of the MCDC and BCWS discharges at short distances from one another, downstream of the PNCC discharge (Fig 1.2), prevented the investigation of their individual effects on epilithon development in the Manawatu River, since the quality of the water to which they discharged was always modified to some degree by the PNCC discharge or PNCC and MCDC discharges. In order to compare the effects of the two downstream discharges (MCDC and BCWS) the epilithon development was studied in two identical laboratory channel systems (Section 3.7) operated under a range of influent wastewater concentrations (Table 5.22). The respiration and gross photosynthetic oxygen production rates of the resulting epilithon communities were discussed in Section (5.5) and the organic material and nutrient removal rates were discussed in Section (6.2.3). This section considers the biomass development rates and community types observed in these experiments.

7.3.2 EPILITHON GROWTH IN THE ABSENCE OF ORGANIC WASTE ADDITION

In the control channel (Fig 7.20) the conditions of low influent BOD_5 concentrations (0.11 to 0.59 $g.m^{-3}$), high nutrient levels (717 to 898 $mgTDN.m^{-3}$ and 74-87 $mgTDP.m^{-3}$, Table 6.8, Section 6.2.3.3) and low, but regular, light input (Section 3.7.3) promoted the development of a cohesive, phototrophic, epilithon mat. The low nutrient removal rates observed (Section 6.2.3.3) indicate that growth was not nutrient limited. The community was composed of diatoms (predominantly *Synedra* sp , *Fragillaria* sp. and *Navicula* sp.), unicellular and colonial green algae (predominantly *Ankistrodesmus* sp. and *Scenedesmus* sp.), the cyanobacteria *Oscillatoria* sp. and the filamentous green algae *Stigeoclonium tenue*. *S. tenue* was most abundant in the turbulent zones at the recycle pump in flows (Fig 3.5) (Section 5.5.5). The community reached a maximum thickness of 4 mm on the channel side walls.

The phototrophic nature of the growths is reflected in the low mean AI values of 47 to 67 (Fig 7.20) and low respiration rates compared with the epilithon of the channels receiving organic waste addition (Section 5.5.3).

After an initial colonisation phase of approximately eight days, epilithon growth was rapid until approximately day 21. During this growth phase and on day 43 the epilithon caused a net increase in the BOD_5 of the water passing through this channel. The growth rate (2.2 $gAFDW.m^{-2}.d^{-1}$) during this rapid growth phase between days 8 and 21 was comparable to that of a phototroph community of similar species composition growing under comparable high nutrient and temperature conditions at site EF in the Manawatu River between 31/1/84 and 9/2/84 during experiment PR_4 (i.e., 2.6 $gAFDW.m^{-2}.d^{-1}$) (Section 7.2.6.2 and Fig 7.9). This similarity of growth rate was somewhat surprising given that the average light input to the channel biomass (2.8 $E.m^{-2}.d^{-1}$ PAR (Section 3.7.3)) was only about one tenth of the calculated input at the depth of the biomass at site EF during the interval 31/1/84 to 9/2/84 (calculated from

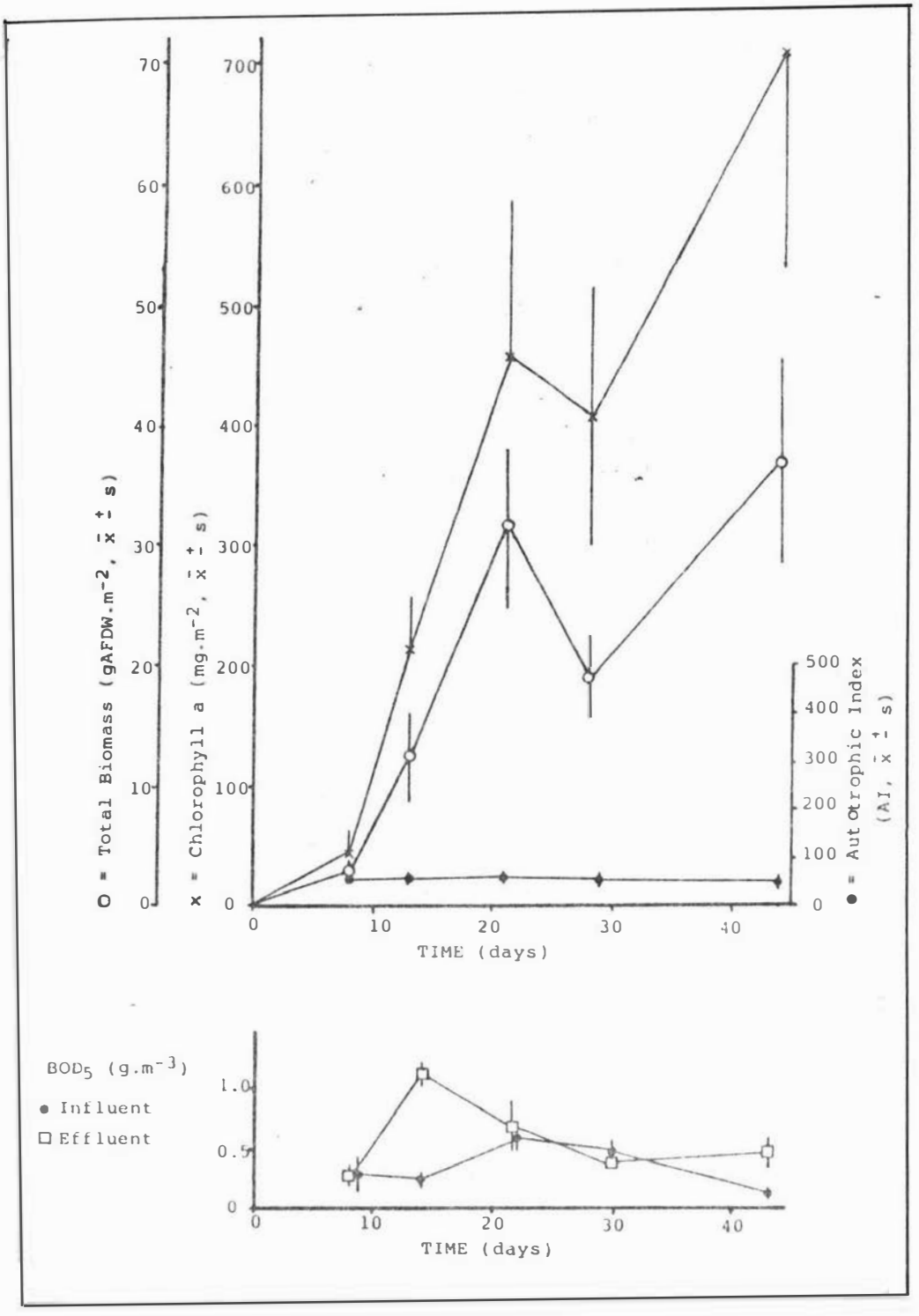


FIGURE 7.20: Results of Laboratory Channel Experiment A (control, no wastewater addition to influent)

the mean total surface radiation (TSR) data assuming 1 $W.m^{-2}$ TSR = 2.03 $uE.m^{-2}.s^{-1}$ PAR and a vertical extinction coefficient of 1.5 m^{-1}).

The comparable growth rates observed under the two situations are partly explained by the higher chlorophyll a content and lower respiration rates of the channel epilithon. This epilithon had mean AI values of 54 to 67 during the rapid growth phase (Fig 7.20) compared with mean values of 107 to 216 for the biomass at site EF (Fig 7.11). The respiration rates of the plate biomass at EF over the growth interval was not measured but *C. glomerata* dominated, river epilithon at site EF on 12/3/84 had a weight specific respiration rate at 21.2°C 2.3 times greater than comparable *Stigeoclonium tenue* growths from the turbulent zone of the control channel (Tables 5.6 , 5.7 and 5.25). Thus the channel epilithon probably had a lower respiration rate than that at site EF during the plate biomass growth phase. High chlorophyll a concentrations and low respiration rates are characteristic of algal communities adapted to growth under low light regimes (Darley, 1982).

Higher sloughing losses in the river due to the small spate which occurred between 31/1/84 and 9/2/84 (Fig 7.11) and invertebrate grazing, which was not significant in the channels, probably also limited the growth rate at site EF to some extent.

These results show that epilithon development in the control channel was comparable to that occurring at high nutrient concentrations and relatively low organic material concentrations in the Manawatu River. The growth rates in the control channel were much higher than those at site A above the wastewater discharges (e.g., Figs 7.11-7.12) where phototrophic growth was limited by low phosphorus concentrations (Section 7.2.6.2). Thus the effects of the wastewater discharges on epilithon growth in the laboratory channels were expected to be different to those in the river.

Since the background nutrient levels in the borewater did not limit phototrophic growth, the nutrients added by the

wastewater discharges were not expected to increase phototrophic growth. However the wastewater organics were expected to promote the development of heterotrophic micro-organisms and this alteration of the biological conditions was expected to affect phototroph development.

The effects of the addition of the MCDC and BCWS wastewaters to the channels on epilithon growth are discussed in the following sections (Sections 7.3.3 and 7.3.4).

7.3.3 EFFECTS OF MCDC WASTEWATER ADDITION ON EPILITHON DEVELOPMENT IN LABORATORY CHANNELS

The addition of sufficient MCDC wastewater to increase the channel influent COD by 1.9 $\pm$ 0.1 $g.m^{-3}$ (Experiment B), 3.8 $\pm$ 0.6 $g.m^{-3}$ (Experiment C) and 6.5 $\pm$ 1.0 $g.m^{-3}$ (Experiment D) resulted in changes in the channel epilithon community structure and/or maximum biomass (Figs 7.21 to 7.23) compared with that in the control channel (Fig 7.20).

The epilithon development followed a similar pattern in each experiment involving MCDC wastewater addition. Initial growth was almost entirely due to the filamentous bacteria *Sphaerotilus natans* and to a lesser extent *Flavobacterium* sp. and *Zoogloea* sp. These were also the predominant heterotrophs in the Manawatu River (Section 7.2.3). The heterotrophic nature of the initial growths was reflected in very high initial autotrophic index (AI) values (Fig 7.21-7.23). After about eight days unicellular green algae and diatoms appeared in significant numbers and grew rapidly amongst the matrix provided by the filamentous bacterial growth. Neither filamentous green nor cyanobacteria, which were both abundant in the control channel periphyton, developed in the channels fed MCDC wastewater. This probably resulted from unavoidable differences in the channel seed material collected from sites below the wastewater discharges immediately prior to the beginning of each experiment.

This epilithon developed until a maximum biomass concentration was reached and subsequently sloughing occurred. The timing of

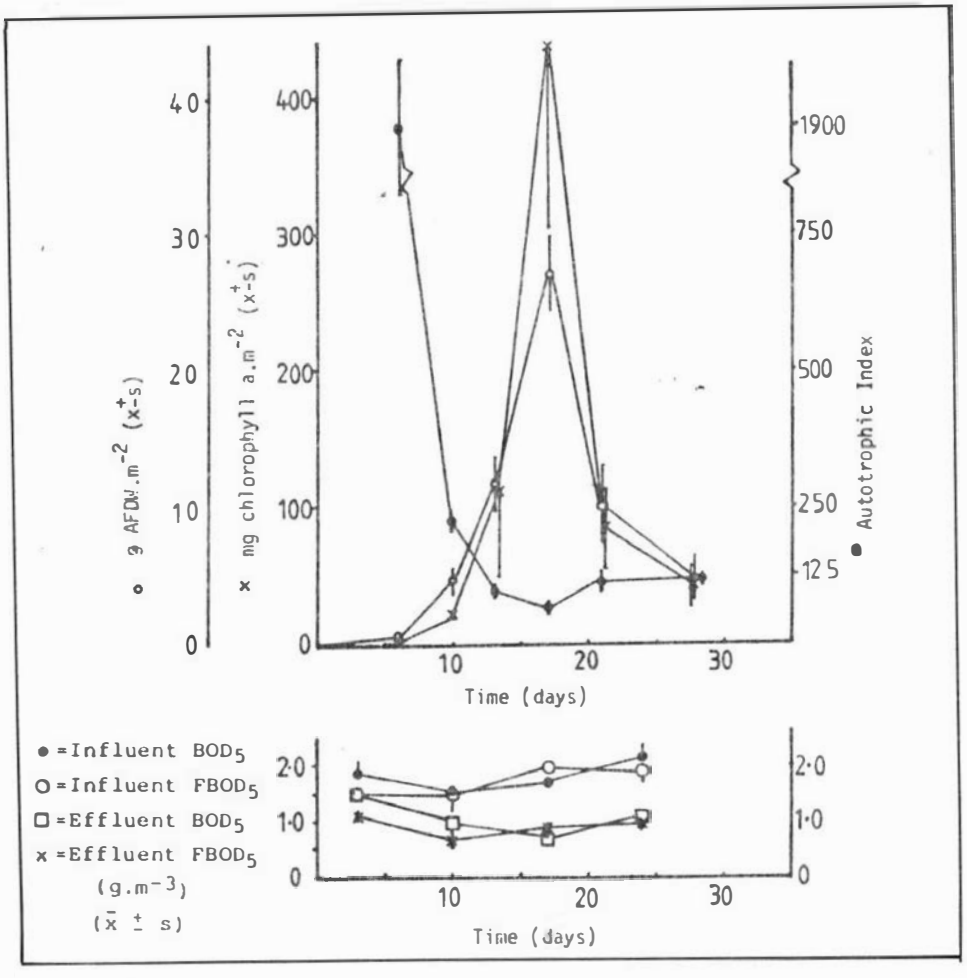


Figure 7.21: Epilithon Growth and Water Quality During Channel Experiment B (1.9 gCOD.m^{-3} MCDC waste-water in influent).

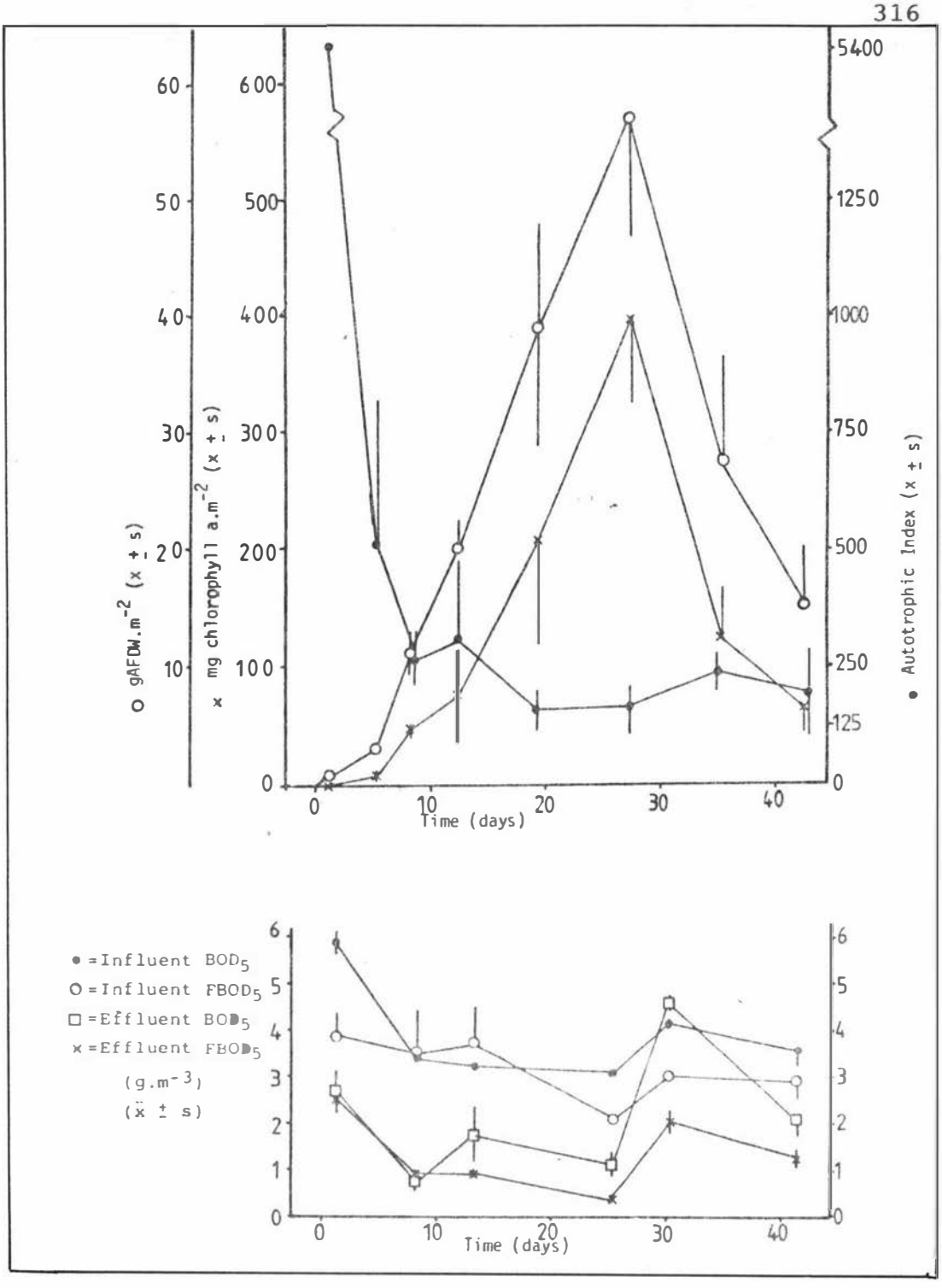


Figure 7.22: Epilithon Growth and Water Quality During Channel Experiment C (3.8 gCOD.m^{-3} MCDC waste-water in influent).

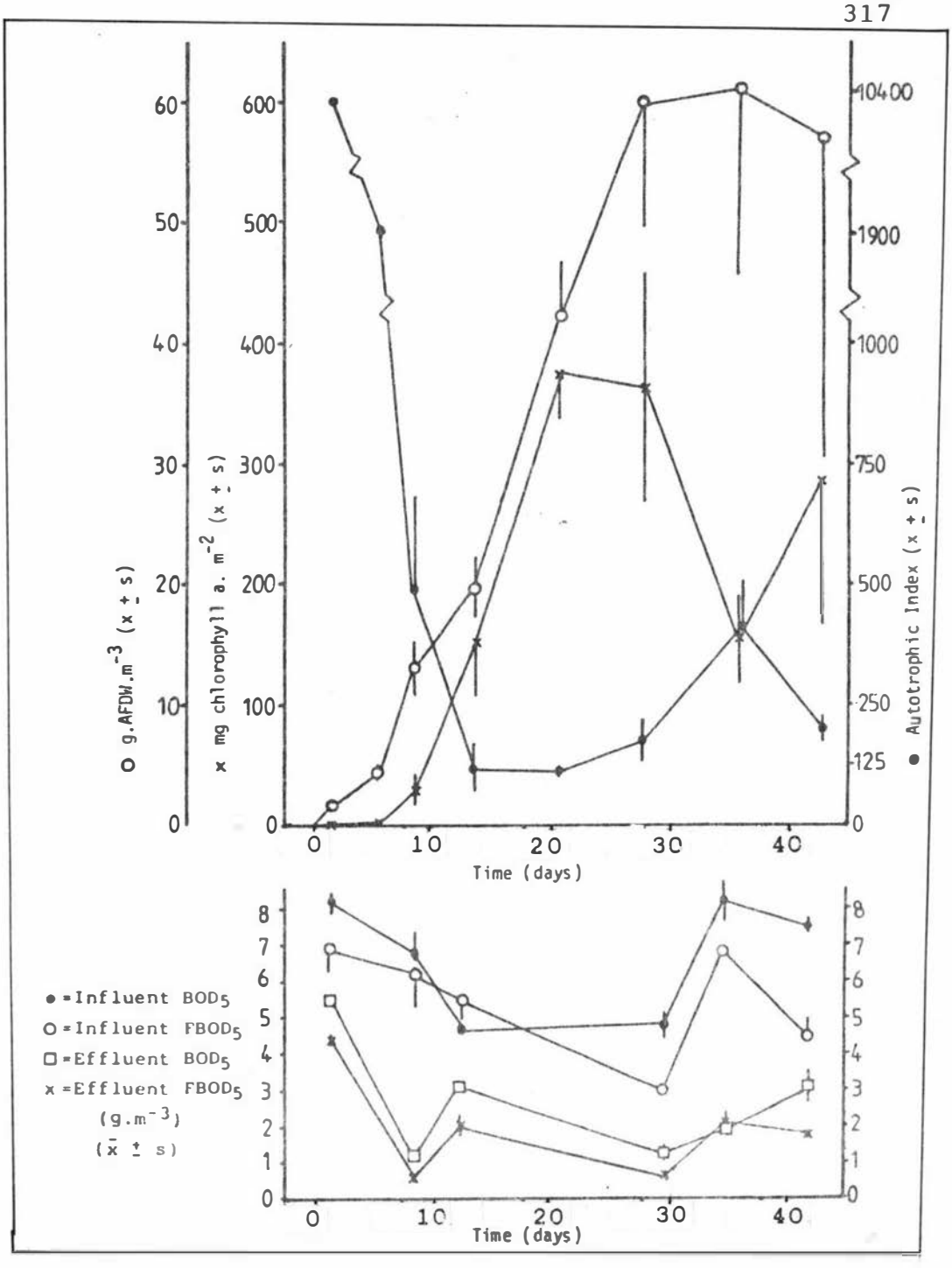


Figure 7.23: Epilithon Growth and Water Quality During Channel Experiment D (6.5 gCOD.m^{-3} MCDC waste-water in influent).

these events was dependent upon the wastewater loading, with sloughing occurring sooner at the lower loading rates.

At the time of biomass sloughing the epilithon had lost its slimy texture and microscopic examination showed that many of the *S. natans* trichomes, which provided much of the matrix for community development, were devoid of cells. This suggests that the biomass had developed to a stage where the available organic substrate supply could no longer sustain the lower matrix organisms adjacent to the support. After these died their trichomes and surrounding extracellular slime layer ceased to be maintained and the epilithon matrix broke away from the bed under the force of the water current.

After sloughing occurred in the 1.9 $g.m^{-3}$ and 3.8 $g.m^{-3}$ COD channels (Experiments B and C) the filamentous bacteria did not re-establish, except in the turbulent zones at the recirculating pump inflows where heterotrophic flocs occurred. Instead an epilithic mat composed of unicellular green and brown algae and diatoms developed with the heterotrophic forms uncommon. This occurred despite in-channel $fBOD_5$ concentrations which had allowed rapid growth of these organisms early in the experiment (Figs 7.21 and 7.22). This phenomenon may be explained by competition between the heterotrophs and phototrophs for attachment sites. At the beginning of the experiment the bare substrate provided a large area for colonisation and growth of the heterotrophs in the mixed seed and they grew rapidly. However when sloughing occurred the phototrophs were much more abundant that initially. This may have enabled them to move onto the bare areas and thus prevent recolonisation by the filamentous bacteria. Without the matrix provided by the *Sphaerotilus* trichomes the unicellular phototrophs present did not reach the former biomass levels (see chlorophyll a data, Figs 7.21 and 7.22) and only formed a thin film over the substrate. The occurrence of heterotrophic flocs in the turbulent zones indicates that these areas were not suitable for colonisation by the phototrophic species present thus allowing the heterotrophs to become established.

A progressive increase occurred in the maximum biomass and the time interval prior to sloughing with increasing influent wastewater loading. This suggests that the higher organic concentrations allow a thicker epilithon biofilm to develop before the supply of organic material to the basal cells, providing epilithon attachment, becomes limiting leading to their death and sloughing of the epilithon. Limitation of the epilithon basal cell's metabolism by organic material rather than oxygen is supported by consideration of the concentration ratio of oxygen to $fBOD_5$ in the channel water and their relative diffusivities. Since the channel dissolved oxygen concentration always exceeded 6 $g.m^{-3}$ (Section 3.7.4), the former ratio was always greater than 1 (Figs 7.21 to 7.23) whilst the relative diffusivity of the oxygen to the channel $fBOD_5$ in water and biomass would be greater than the value of 3:1 recorded for oxygen and glucose (Howell and Atkinson, 1976). This suggests that the channel $fBOD_5$ concentration would have to be at least three times the dissolved oxygen concentration before respiration would be limited by the oxygen supply.

The results in figures 7.21 to 7.23 show that increasing the influent MCDC wastewater concentration above 1.9 $gCOD.m^{-3}$ caused a marked increase in both the maximum biomass concentration (as $gAFDW.m^{-2}$) and the initial epilithon development rate over the first eight days, when the growth was mainly due to heterotrophic development. In experiment B (Fig 7.21) the addition of 1.9 $gCOD.m^{-3}$ to the influent produced a similar maximum total biomass and a similar total biomass at day eight to that in the control channel (Fig 7.20). However comparison of the AI values on day eight shows that the biomass in experiment B was much more heterotrophic. At 3.8 and 6.5 $gCOD.m^{-3}$ MCDC wastewater addition to the channel influents (Figs 7.22 and 7.23) the maximum biomass concentrations were approximately twice that in the control experiment and the mean biomasses after eight days were 4.3 and 4.6 times that in the control channel on day eight respectively.

At some time during the first eight days of the experiments the channel (= effluent) BOD_5 and $fBOD_5$ concentrations were reduced

to similar low levels in experiments B, C and D (Figs 7.21 to 7.23). Between days five and twenty-eight the rapid growth of algae and heterotrophs resulted in growth rates in both experiments C and D similar to those during the growth phase of the control experiment (i.e., 2.4 and 2.5 $gAFDW.m^{-2}.d^{-1}$ in experiments C and D respectively). In experiment B the growth rate between days ten and eighteen was more rapid (3.2 $gAFDW.m^{-2}.d^{-1}$) than in the control experiment and those in which MCDC wastewater loadings were higher (Fig 7.21) but after eighteen days the biomass sloughed off the plates. The steady decline in AI values during the rapid growth phase of experiment B (Fig 7.21) showed that the growth was largely due to algae whereas in experiments C and D the epilithon AI values, which were two to three times the control values (Figs 7.20, 7.22, 7.23), and macroscopic and microscopic examination of the epilithon, showed it to be composed of both heterotrophs and phototrophs. The contribution of heterotrophs to the epilithon was also reflected in the high respiration rates of the epilithon in experiments C and D (Section 5.5.3).

In conclusion these experimental results predict that the addition of sufficient MCDC wastewater to cause a calculated increase in the river COD of 3.8 to 6.5 $g.m^{-3}$ upon complete mixing will have significant effects upon the heterotroph development at the river equivalent site downstream of the outfall (Section 7.3.5). Addition of sufficient MCDC wastewater to cause a calculated increase of 1.9 $gCOD.m^{-3}$ (1.2 $gBOD_5.m^{-3}$) in the river upon mixing will increase the initial growth of heterotrophs at the river equivalent site but will have a much smaller effect upon the epilithon activity (Section 5.5.3) and the maximum epilithon biomass at the river equivalent site provided that the phototroph growth in the river is not nutrient limited. If this is the case this effluent addition may alter the total biomass and activity but the channel experiments results cannot be used to predict these effects.

7.3.4 EFFECTS OF BCWS PRIMARY TREATED MEATWORKS WASTEWATER ADDITION ON EPILITHON DEVELOPMENT IN LABORATORY CHANNELS

The results of experiments E to H (Table 5.22) (Figs 7.24-7.27) show a general pattern of increased initial epilithon growth rate and maximum total biomass prior to sloughing with increased influent BCWS COD from 1.9 $gCOD.m^{-3}$ to 14.4 $g.COD.m^{-3}$ (= 1.1 to 8.6 $gBOD_5.m^{-3}$). However interpretation of these experiments results and their comparison with those of experiments A to D are complicated by two factors:

i) A 12 hour period of temperatures up to 30°C overnight on day nine of experiment H (Fig 7.27), due to the accidental alteration of the temperature control system. This probably led to the unexpected early sloughing on day fifteen of this experiment.

ii) The relatively high contribution of settled particulate organic material from the BCWS wastewater to the plate AFDW values.

Comparison of the average ratio of influent $fBOD_5$ to BOD_5 in the MCDC addition experiments (B to D) ($\bar{x} \pm s = 0.81 \pm 0.15$) with that in the BCWS addition experiments (E to H) ($\bar{x} \pm s = 0.51 \pm 0.16$) and the results of the filtration-fractionation studies of the two wastewaters (Table 5.24, Section 5.5.3) show that the BCWS wastewater added considerably more particulate organic material per unit BOD_5 than the MCDC wastewater. The MCDC particulate organic material did not accumulate amongst the biomass, indicating that it was relatively easily degraded to assimilable organics and/or that it passed through the channel. By contrast, the solids in the BCWS wastewater did accumulate on the plates and resulted in higher AI and AFDW values than would thave occurred due to the growth of epilithic micro-organisms alone. This implies that the results of the respiration and photosynthetic rate measurements discussed earlier (Section 5.5) provide a better means of comparing the effects of the two wastewaters on epilithon development than the biomass measurements.

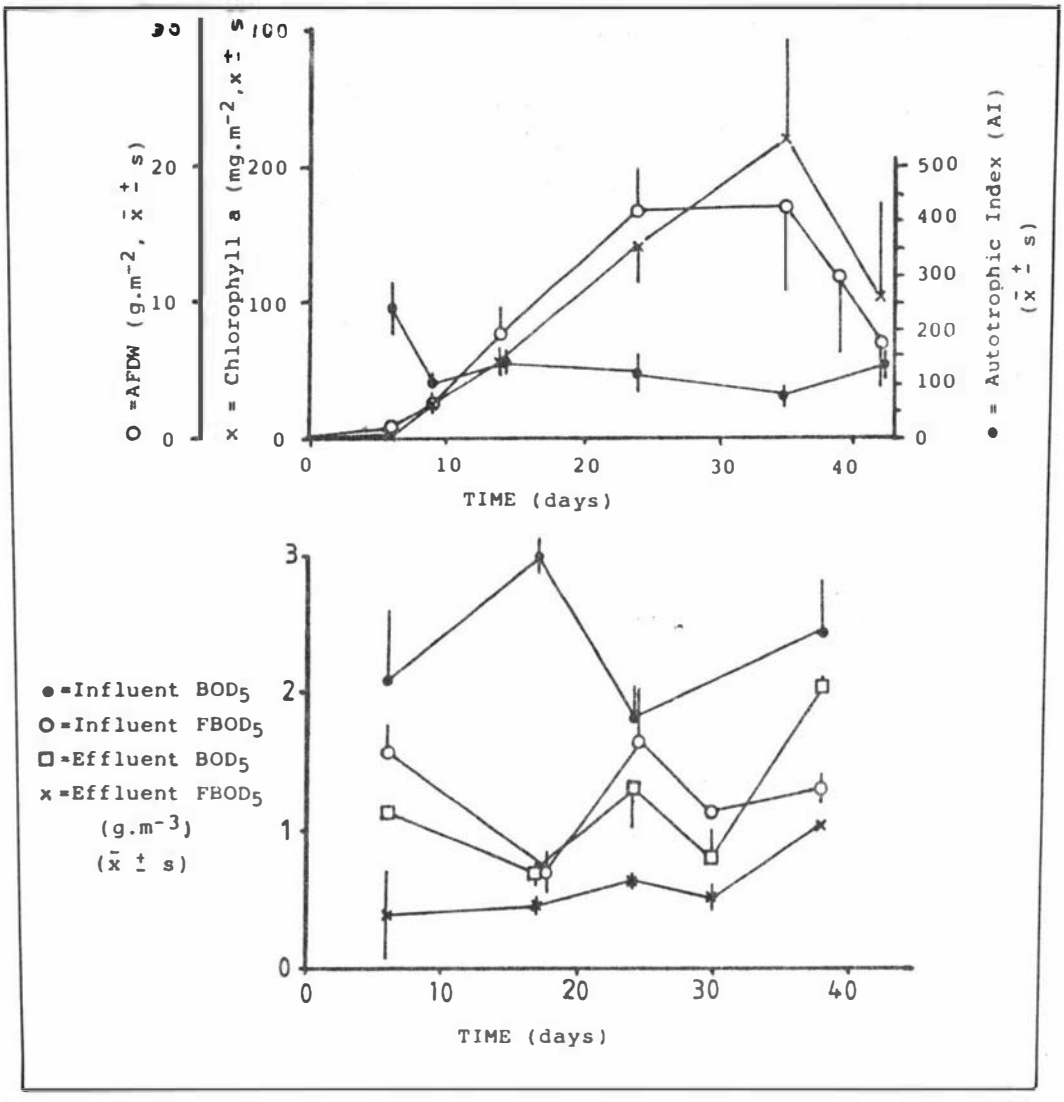


Figure 7.24: Epilithon Growth and Water Quality During Channel Experiment E(1.9 g COD.m^{-3} BCWS waste-water in influent).

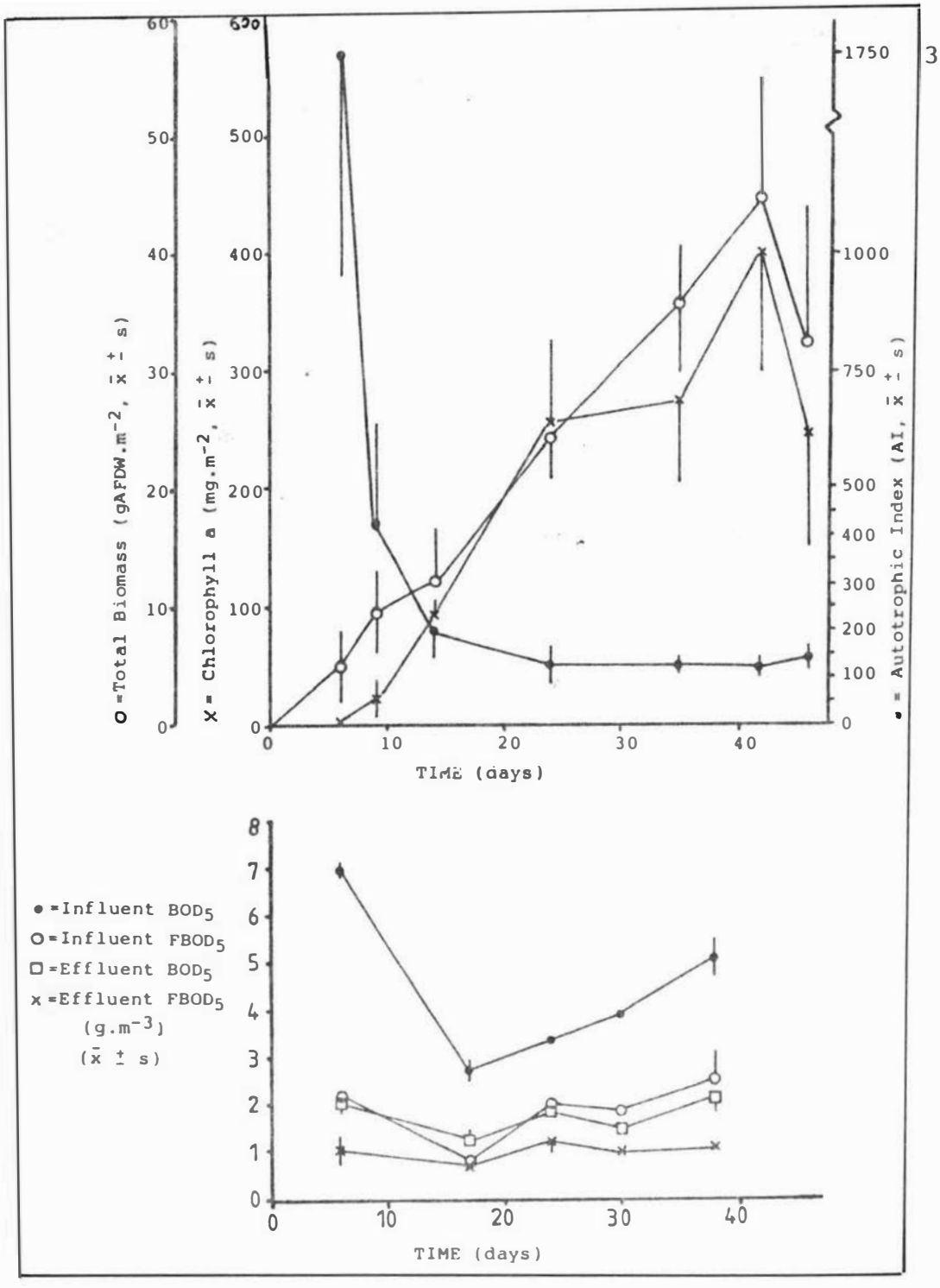


Figure 7.25: Epilithon Growth and Water Quality During Channel Experiment F (4.2 gCOD.m^{-3} BCWS waste-water in influent).

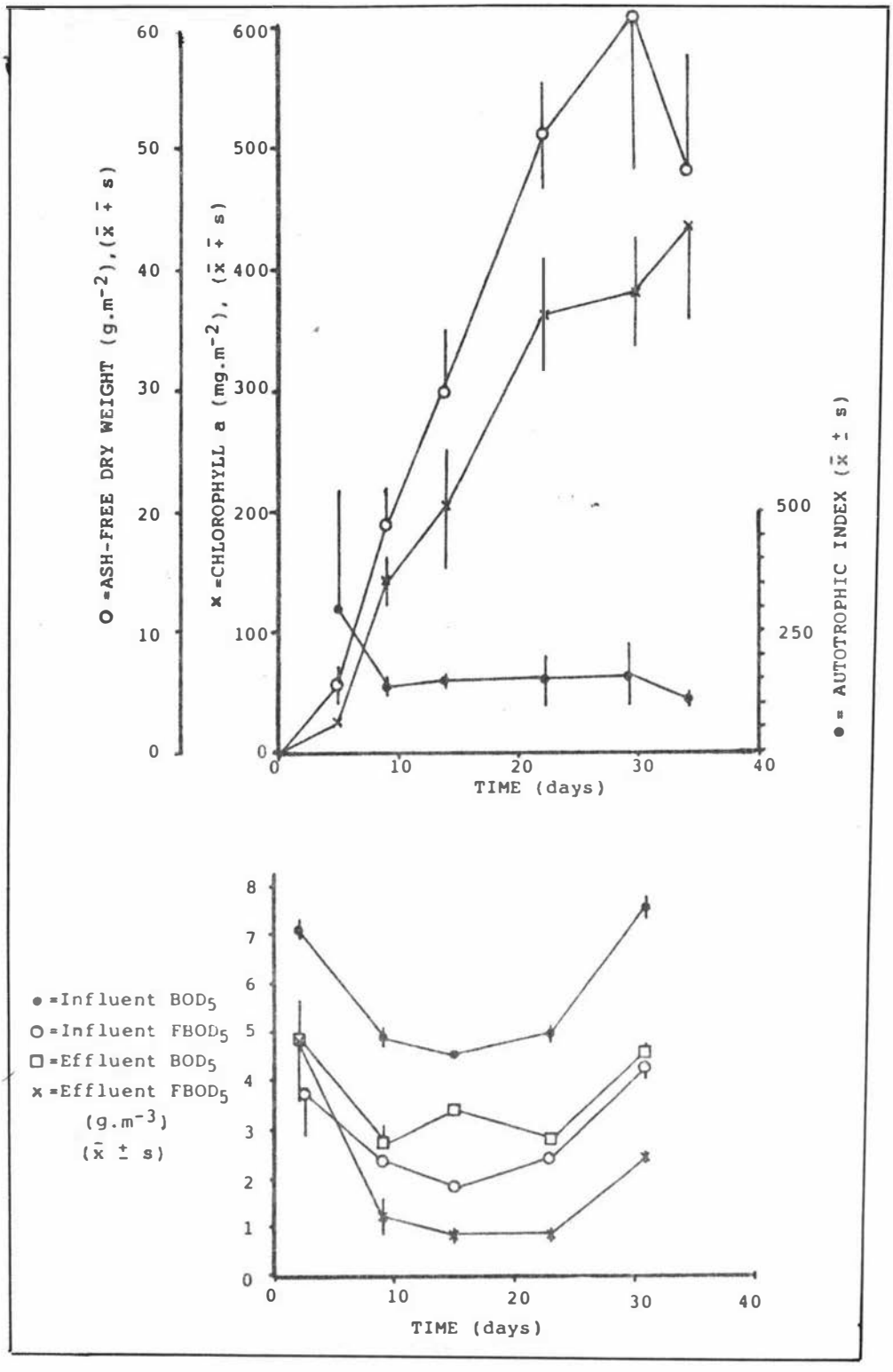


Figure 7.26: Epilithon Growth and Water Quality During Channel Experiment G (7.3 gCOD.m^{-3} BCWS waste-water in influent).

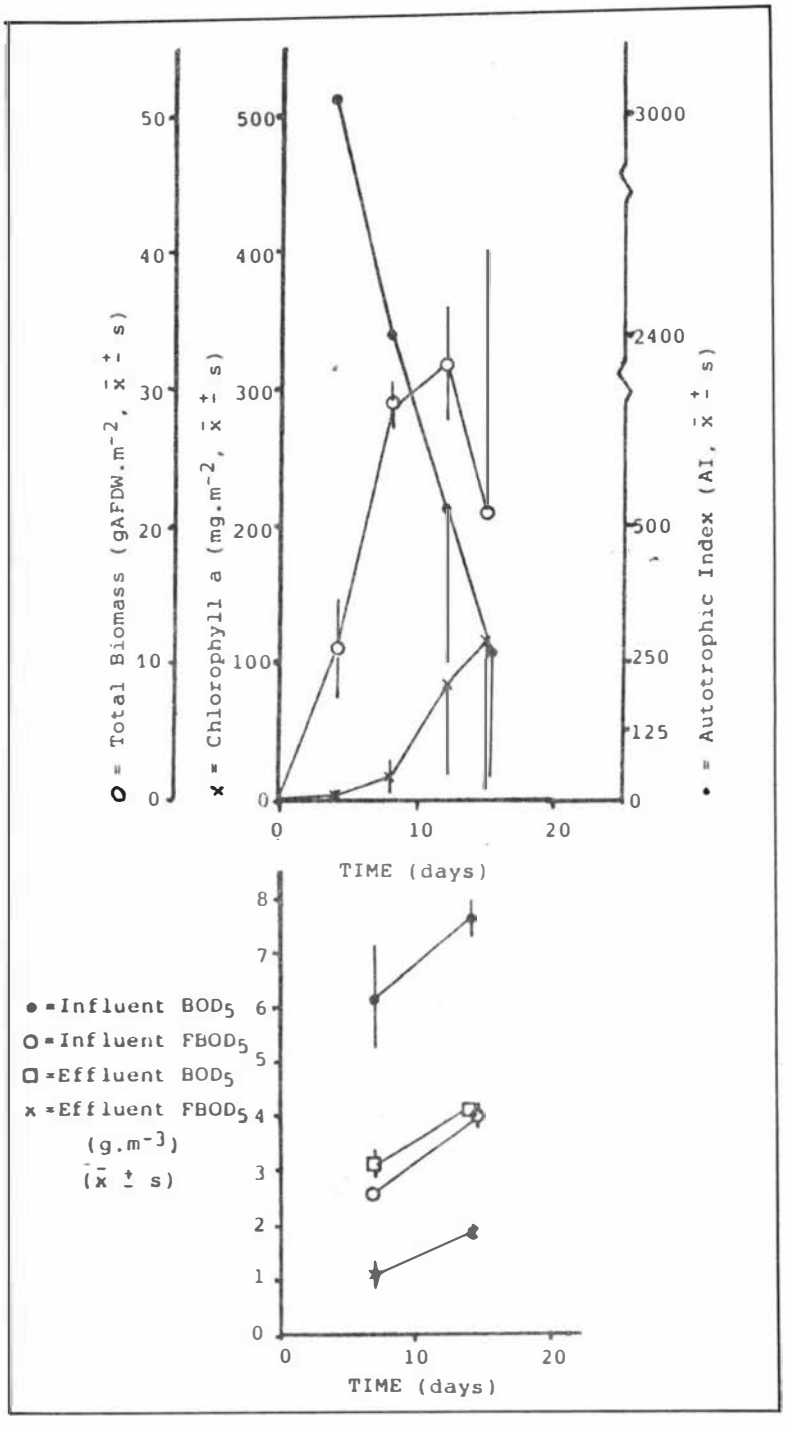


FIGURE 7.27: Results of Laboratory Channel Experiment H (BCWS wastewater concentration in influent= 14.4 $gCOD.m^{-3}$ (= 8.8 g $BOD_5.m^{-3}$)).

This effect of the particulate material on the plate biomass values was most noticeable during the first eight days of experiment H when a BCWS wastewater sample with an unusually high particulate BOD_5 content was used as the channel feed, giving an influent $fBOD_5$:BOD_5 ratio of 0.31 on day six (Fig 7.25). This apparently resulted in the unexpectedly high plate biomass AI values at the beginning of this experiment.

The epilithon that developed during experiments E and F, when the mean influent BCWS COD additions were 1.9 and 4.2 $g.m^{-3}$ respectively (= 1.1 and 2.5 $gBOD_5.m^{-3}$ respectively), was dominated by phototrophs. Microscopic examination of the growths showed that heterotrophs (mostly *S. natans*) were common initially but these became uncommon to scarce latter in the experiments. *S. tenue* was abundant in both experiments along with a diverse assemblage of diatoms (predominantly *Navicula* sp., *Fragillaria* sp., *Synedra* sp. and *Eunotia* sp.) and the colonial and unicellular green algae *Scenedesmus* sp. and *Chlorella* sp..

S. natans was very abundant at the beginning of experiment G (Fig 7.26) (mean influent BCWS COD loading = 7.3 $g.m^{-3}$ (=4.4 $gBOD_5.m^{-3}$) but subsequently its relative abundance declined progressively and it was uncommon by day twenty-nine. From day nine phototrophs were dominant and a diverse assemblage of diatoms, green algae (including *S. tenue*) and the blue green algae (*Oscillatoria* sp.) occurred.

Increasing the mean influent BCWS COD loading further to 14.4 $g.m^{-3}$ (= 8.6 $gBOD_5.m^{-3}$) caused a marked change in the channel epilithon (Fig 7.27). The **chlorophyll** a data show that the initial phototrophic growth was much slower than in experiment G and the filamentous species *S. tenue* and *Oscillatoria* sp., which developed in experiment G, were not present. By contrast, the heterotrophs developed rapidly and dominated the epilithon. *S. natans* was most abundant but *Zoogloea* sp. and *Beggiatoa alba* were also common. The abrupt decline in the growth rate (as AFDW) between days eight and twelve suggests that the 12 hour period of 30°C temperatures on day 9 had a

detrimental effect on the epilithon leading to the early sloughing on day 15. Nevertheless these results and those of the photosynthetic and respiration rate measurements (Sections 5.5.2 and 5.5.3) show that increasing the mean BCWS COD in the channel influent from 7.3 to 14.4 $g.m^{-3}$ (= 4.4 to 8.3 $gBOD_5.m^{-3}$) caused a marked increase in the heterotroph abundance amongst the channel epilithon. By contrast a much lower mean channel influent wastewater concentraton, of 3.8 $gCOD.m^{-3}$ (= $2.5gBOD_5.m^{-3}$), as MCDC wastewater in experiment C (Fig 7.2) resulted in the abundant growth of heterotrophs and much higher epilithon respiration rates indicating increased heterotrophic activity (Section 5.5.3). The different heterotrophic growth enhancement effects of the two wastewaters, per unit COD or BOD_5, is apparently related to their different compositions with respect to dissolved and low molecular weight organic material (Section 5.5.3).

7.3.5 RELATING THE LABORATORY CHANNEL RESULTS TO THE RIVER SITUATION

The laboratory channel experiments provide useful information on the comparative effects of similar concentrations of MCDC and BCWS wastewaters on epilithon development (Sections 7.3.3 and 7.3.4) and metabolic rates (Sections 5.5.2 and 5.5.3). However the use of the channel results for prediction of the effects of the discharges on epilithic growth in the river requires the estimation of the distance below the outfall at which the growth conditions are similar to those in the channel.

Because the channel epilithon respond to the in-channel rather than influent water quality, the channel observations relate to growth at a point some distance downstream of a waste discharge rather than directly below it. This distance may be estimated by considering the ratios of wetted surface area (WSA) to volume (V) in the river and in the channel, the channel's hydraulic residence time (τ), differences in the two systems' flow regimes and organic material removal occurring in the water column in the river.

To obtain a first estimate of the distance down the river from the discharge relating to the channel situation it was assumed that only the epilithic organisms contribute to the removal of organic material and that the different flow regimes have no effect. Under these conditions the distance as river travel time is related to the different ratios of wetted surface area to volume in the 2 systems by the following formula:

$$t(\text{mins}) = \text{channel}\,\tau\,(\text{mins}) \quad \frac{\frac{WSA}{V}\ \text{channel}\left(\frac{m^{-2}}{m^{-3}}\right)}{\frac{WSA}{V}\ \text{river}\left(\frac{m^{-2}}{m^{-3}}\right)}$$

(Equation 7.2)

This equation yields a river travel time of 357 minutes (5.95 h) when solved for the following conditions:

i) τ = 11.2 minutes (Section 3.7.6)

ii) channel $\frac{WSA}{Vol}$ ratio = 52 (Section 3.7.2).

iii) river $\frac{WSA}{Vol}$ ratio = $\frac{1}{\text{mean depth}}$ x bed roughness factor (Section 7.2.2)

= 1 x 1.63

= 1.63

Multiplying by the average current velocity (u) of 0.33 $m.s^{-1}$ over the study reach at 20 $m^3.s^{-1}$ (Wilcock, 1984(a)) predicts that the channel river equivalent site is located 7.3 km downstream of the outfall. However, in contrast to the channel, in the river significant water column removal of organic material may be anticipated. This will effectively shorten the distance from the outfall to the point relating to the channel situation. The water column removal of organic matter can be estimated from BOD_5 decay data for dairy effluents. Using the k_1 value of 0.51 d^{-1} (Barnett *et al.*, 1982) and the equation for first order decay in a plug flow system (Appendix J, Equation 22) gives a reduction in BOD_5 of 11.9% over the 5.95 hour travel time.

This procedure overestimates the watercolumn removal somewhat since the activity of the benthic organisms also lowers the organic substrate concentrations and this would reduce the

uptake by the suspended organisms. However neglecting this effect, an 11.6% reduction of the distance predicted by equation 7.1 gives a distance of 6.4 km below the outfall as the location of the channel-river equivalent site. This distance would be further reduced due to the channel being a continuously mixed system whereas dye studies have shown that the Manawatu river is essentially a plug-flow, once through system with longitudinal dispersion having only a minor effect on organic removal rates (Rutherford, *et al.*; 1982).

Theoretical considerations and experimental comparisons of waste treatment systems show that organic material removal rates are greater in plug flow systems than in continuously mixed systems (e.g., Levenspiel, 1972, Rittmann, 1982). Thus in a plug flow system the initial substrate concentration, Co, is reduced to the level C, attained in an equivalent continuously mixed system (w.r.t. flow, Co, WSA: Vol ration) operated at an hydraulic residence time (τ) in a time less than τ.

It can readily be shown (Appendix J) from the mass balance equations for these two types of systems that the time (t_1) taken in a plug flow system for the initial substrate concentration (Co) to be reduced to the concentration C_1 obtained in a continuously mixed system at specified τ and k_1 values can be calculated from the equation

$$t_1 = \frac{\ln(1+k_1\tau)}{k_1} \qquad \text{(Equation 7.3)}$$

For the channel system data the value of k_1 can be determined from the equation for a continuously mixed system.

Solving equation (7.2) for the fBOD$_5$ k_1 values derived from the growth and plateau phases of channel experiments B to H, using the data in figures (7.21) to (7.27), gives an average t_1 value of 6.6 $\pm$ 1.7 ($\bar{x} \pm s$) (n = 27) minutes. Using the data from the higher loaded MCDC wastewater experiments (C and D) alone gives a t_1 value of 5.1 $\pm$ 1.4 ($\bar{x} \pm s$; n = 9) minutes. These imply

that in a plug flow system $fBOD_5$ removal would occur in approximately half the time of residence of water in the channel (11.2 minutes).

Since the heterotrophic community growth rate is dependent on the substrate concentration in the overflowing water, this implies that the heterotrophic growth observed in the channels relates to a site about 0.59 (= 6.6/11.2) times the distance of 6.4 km below the outfall predicted from considerations of the WSA:volume ratios in the channel and river and water column organic removal rates i.e., approximately 3.8 km downstream of the outfall at a river flow of 20 $m^3.s^{-1}$. Similarly the t_1 value predicted from experiments C and D implies that the heterotrophic growth in these channels relates to that expected at a site about 2.9 km below the outfall at a river flow of 20 $m^3.s^{-1}$ (i.e., 5.1/11.2 x 6.4 km).

Longitudinal dispersion increases the distance below the outfall required to reach the channel organic material concentration beyond that predicted above for a true plug flow system. This effect can be estimated from the Wehner and Wilhelm equation accounting for dispersion effects in systems showing first order kinetics presented in graphical form by Levenspiel (1972).

Using the dispersion and velocity data for the reaches between sites B, C, D and E below the MCDC outfall in the Manawatu River (Fig 1.2) at a flow of 26.3 $m^3.s^{-1}$ (Rutherford *et al* 1982), this equation predicts that the river dispersion effects over a distance of 2.4 to 3.8 km increase the distance below the discharge to the site relating to channel conditions by about 10%. Thus, at a river flow of 20 $m^3.s^{-1}$, the distances below an outfall of the river equivalent sites derived using mean t_1 values calculated from experiments B to H and from experiments C and D become 4.2 and 3.2 km respectively.

Following the same procedure for river flows of 13 $m^3.s^{-1}$ (= 96% low flow) (assuming u = 0.29 $m.s^{-1}$, h = 0.9 m, w = 50 m) and 28 $m^3.s^{-1}$ (assuming u = 0.5 $m.s^{-1}$, h = 1.0 m, w = 60 m) predicts distances below the outfall to the river equivalent

site of 3.3 km and 6.1 km respectively for the average t_1 value generated from experiment B to H, and 2.6 and 4.7 km respectively, for the average t_1 value generated from experiments C and D.

This shows that as the river flow declines the increased WSA/volume ratio and reduced average velocity (u) result in a shortening of the distance to the river equivalent site.

The distances between the outfall and the theoretical channel-river equivalent site at low flows are sufficiently short for the results of the laboratory channel experiments to be useful for management of the Manawatu River. Epilithon growth at sites closer to the outfalls could be investigated using the same system by reducing the hydraulic residence time of the channel water.

This methodology for predicting the distance downstream of an outfall of the river equivalent site was assessed by comparison of the upper plate surface epilithon growth in experiment D (Fig 7.23) (MCDC COD addition to influent = 6.5 $g.m^{-3}$ (=4.2 $gBOD_5.m^{-3}$)) with that in the Manawatu River at site D, 3.2 km downstream of the MCDC outfall during experiment PR_2 (Fig 7.9). At the beginning of this experiment, on 21/11/83, the river benthic biomass was just becoming established after a period of high flows (Fig 5.12) so that the biomass conditions were similar to those in the channel at the beginning of the growth experiments. The average MCDC wastewater discharge during experiment PR_2 was sufficient to increase the river COD by 5.7 $\pm$ 2.9 ($\bar{x} \pm s$) (= 3.7 $\pm$ 1.9 $gBOD_5.m^{-3}$) and the theory outlined above predicts that at the mean river flow of 27.5 $\pm$ 8 $m^3.s^{-1}$ ($\bar{x} \pm s$) the river equivalent site to which the channel results relate would be 4.6 km below the MCDC outfall i.e., 1.4 km below site D.

These two experiments provide closest match available of a channel experiment and river growth near the predicted river equivalent site under similar wastewater loading conditions. However the comparison is further complicated by the following factors:

i) the average calculated background BOD_5 (Section 7.2.1) above the MCDC outfall of 2.7 $g.m^{-3}$ due to the background river BOD_5 and that added by the PNCC but not removed at the MCDC outfall.

ii) the further average increase in the river BOD_5 of 2.2 $\pm$ 0.6 $g.m^{-3}$ ($\bar{x} \pm s$) due to the BCWS discharge 1.4 km upstream of site D on days 8 to 12, 15 to 17 and 23 to 25 of experiment PR_2.

iii) the higher light conditions at the plate depth in the river than in the channel (Section 3.7.3).

iv) the higher seed concentration in the river than in the channel.

v) the different daily discharge patterns in the two systems i.e., continuous discharge of the channel c.f. diurnal pattern of discharge to the river (Sections 2.2.3 to 2.2.5).

vi) the absence of grazing macro-invertebrates such as snails and insect larvae in the channel system.

vii) the lower, variable temperature (i.e., 15 to 20°C) and variable flow conditions in the river.

Because of these differences in the river and channel conditions and the location of site D 1.4 km upstream of the predicted river equivalent site, the comparison does not provide an unequivocal test of the river equivalent site prediction methodology. Nevertheless the similarity of the total biomass and algal biomass development rates in channel experiment D (Fig 7.21) and river experiment PR_2 (Fig 7.9) up to day 18, when a spate caused sloughing of the river biomass, does indicate that the approach is reasonable. The usefulness of the channels for predicting the heterotrophic growth response to wastewater addition is also shown by the similarity of the maximum allowable increase in MCDC BOD_5 in the river, upon complete mixing of 1.8 $g.m^{-3}$, without causing a marked increase in heterotrophic growth at site C in summer predicted by the river observations (Section 7.2.2) with the critical infuent BOD_5 concentration causing a marked increase in channel heterotrophic growth of between 1.2 and 2.5 $g.m^{-3}$ predicted from the channel studies (Section 7.3.3).

These results show that recirculatory, flow-through laboratory channels have potential for use for comparison of the effects of different wastewaters on epilithon development and for predicting the effects of wastewater discharges on epilithon growth at a specific location in a river. However further comparative studies of the epilithon growth in the channels and at river-channel equivalent sites are required before the latter use can be adopted with confidence.

7.4 CONCLUSIONS

The relationships between heterotroph development and river water COD or BOD_5 concentration differed considerably for different types of wastewater. Macroscopic benthic biomass observations in the Manawatu River and epilithon growth rate studies in the river and laboratory channels showed that the MCDC wastewater has a much greater stimulatory effect on heterotroph development for a given BOD_5 concentration than either the PNCC or BCWS wastewaters. This corroborates the findings of the laboratory channel epilithon respiration rates studies (Section 5.4). However at particularly high MCDC wastewater concentrations (giving a mean calculated BOD_5 of 9.8 $g.m^{-3}$ at site C) epilithon growth was apparently inhibited.

Calculated average in-river BOD_5 concentrations of up to 3.3 $g.m^{-3}$, due mainly to PNCC wastewater and up to 5.8 $g.m^{-3}$, due mainly to the BCWS and PNCC wastewaters, did not result in the macroscopic growth of heterotrophic fronds (i.e., sewage fungus development). These results, and previous studies on the effects of primary treated domestic sewage, suggest that where the BOD_5 results from primary treated domestic sewage or meatworks wastewaters an in-river BOD_5 limit of 5 $g.m^{-3}$ should be effective in preventing the development of macroscopic heterotrophic fronds.

By contrast, heterotrophic fronds were common or abundant at calculated mean BOD_5 values as low as 2.9 $g.m^{-3}$ when most of the organic material was due to MCDC wastewater. This effluent had a major impact on sewage fungus growth in the Manawatu

River. Reducing the calculated increase in the river BOD_5 upon mixing of this wastewater to the maximum level permitted by the current water rights (i.e., 2.4 to 2.9 $g.m^{-3}$ for river flows of 30 to 13 $m^3.s^{-1}$ respectively) did not prevent abundant sewage fungus growth from developing at site C, 1.8 km downstream of the MCDC outfall during the summer of 1983/84. However sewage fungus fronds were scarce at site C during February 1985 when the calculated increase in the river BOD_5 due to the MCDC discharge averaged 35% less than the maximum level permitted i.e., (1.8 ± 0.5 $gBOD_5.m^{-3}$; $\bar{x} \pm s$) and the calculated mean BOD_5 at site C was 3.0 $g.m^{-3}$. This suggests that limiting the MCDC discharge to this level will provide an appropriate means of controlling the development of macroscopic heterotrophic growths to an acceptable level during summer. During the winter the allowable BOD_5 increase would need to be lower since a calculated increase of 1.4 $gBOD_5.m^{-3}$ due to the MCDC wastewater during winter resulted in macroscopic growth at site C.

The reduction of the wastewater BOD_5 concentrations in the Manawatu River during the period from 1983 to 1985, due to the reduced BOD_5 inputs required by the water rights, resulted in the progressive reduction of the extent of the sewage fungus zone in the river. However the nutrient levels below the discharges remained well in excess of the levels known to limit algal growth in the river and high phototrophic community biomasses were recorded within and below the sewage fungus zone. The growth rates of the phototrophs on bare surfaces were significantly greater below than above the wastewater discharges and the results of the growth studies indicate that nuisance bed biomass concentrations would be reached on bare surfaces in 21 to 35 days at the phototroph dominated sites B and EF during summer. This time interval for the development of nuisance bed biomass concentrations is more than twice that of 10 to 13 days predicted for heterotrophic biomass at BOD_5 levels of 5 to 7 $g.m^{-3}$, mainly comprising MCDC wastewater. However the algal epilithon sloughed less readily than the heterotrophs when subjected to small spates of up to approximately 70 $m^3.s^{-1}$ so that once established the nuisance

phototrophic biomasses were less readily removed by small flow increases.

A reduction in the size and frequency of peak river flows, due to natural climatic factors, was also shown to have had a large effect on the ecology of the river by allowing dense macrophyte communities to develop throughout the study reach. Any developments which would cause similar changes in the natural river flow regime, such as the construction of impoundments for hydroelectric or water harvesting purposes, would result in increased prevalence of densely growing macrophytes in the lower Manawatu River.

CHAPTER 8. GENERAL CONCLUSIONS

River studies, involving visual observations of the benthic biomass and the measurement of growth rates on artificial substrates showed that limiting the river BOD_5 concentration to 5 $g.m^{-3}$ does not provide a reliable method for preventing the development of macroscopic heterotrophic fronds (sewage fungus). Whilst this control measure is suitable for controlling sewage fungus growth in response to discharges of primary treated domestic sewage and meatworks wastewaters, a much lower BOD_5 concentration was necessary to limit heterotrophic growth following the discharge of untreated MCDC dairy factory wastewater. In this instance limiting the river BOD_5 increase upon complete mixing to 1.8 $g.m^{-3}$ provided an effective means of preventing sewage fungus growth at site C, 1.8 km downstream of the MCDC outfall, during summer lowflows. However a lower allowable BOD_5 increase of less than 1.4 $g.m^{-3}$ was required to prevent the common occurrence of sewage fungus fronds at this site in winter.

Differences in the proportion of the total BOD_5 that was available to the matrix forming organisms of the sewage fungus provided an explanation for the different relationships observed between wastewater BOD_5 and sewage fungus growth. Wastewater ultra-filtration fractionation studies indicated that the filtrable BOD_5 or the low molecular weight BOD_5 or COD (as measured by ultra-filtration techniques) were likely to provide more reliable generalised parameters for use in controlling sewage fungus. However, further comparative studies of the heterotrophic growth response to a greater range of wastewaters is required before a specific concentration of one of these parameters can be recommended with confidence as a sewage fungus control measure.

The reduction of the BOD_5 concentration in the Manawatu River during the study period due to the introduction of new water rights resulted in the shortening of the river reach containing sewage fungus. During a survey on 15 February 1985, after a

period when the MCDC BOD_5 input was 35% lower than the maximum level permitted, only isolated sewage fungus fronds were observed. However the reaches formerly dominated by the heterotrophs developed heavy algal dominated growths. Algal growth rates were always significantly higher at the sites below than above the wastewater discharges. Although these algal dominated reaches often added very large amounts of oxygen to the river during the day, their night-time respiration rates were frequently similar to those of reaches containing moderate or heavy sewage fungus growths. *In situ* chamber comparisons of the weight specific respiration rates of almost purely algal (*Cladophora glomerata* dominated) growths with those of sewage fungus under similar conditions showed that the algae's rates were 0.67 times those of the sewage fungus. However when the *Cladophora* growths had a thin heterotrophic film attached the community's weight specific respiration rate was similar to that of Manawatu River sewage fungus growth.

A computer model was developed to simulate the night-time dissolved oxygen levels under summer low flow conditions in the Manawatu River. This model predicted that the maximum night-time respiration rate allowable so that an element of water passing downstream from the PNCC outfall at night maintained its dissolved oxygen concentration above the 5 $g.m^{-3}$ statutory minimum varied from 24.5 to 20 $g.O_2.m^{-3}.d^{-1}$ for average night-time water temperatures of 12 to 21°C respectively.

A regression model (Equation 5.9) predicted that the benthic biomass concentrations producing these maximum allowable respiration rates declined progressively from 143 $gAFDW.m^{-2}$ at 12°C to 34 $gAFDW.m^{-2}$ at 21°C, the maximum average night-time temperature observed in this study.

Studies of epilithon growth on concrete plates predicted that during summer the interval of relatively stable flows required for a bed biomass of 34 $gAFDW.m^{-2}$ to be reached was ten days at BOD_5 levels of 5 to 7 $g.m^{-3}$ at site C below the MCDC outfall. By contrast, under conditions of low organic materials

concentrations but high nutrient levels, similar to those expected near the discharges when the waste treatment systems are fully operational, the studies predicted that the nuisance biomass concentration of 34 gAFDW.m^{-2} would develop on bare surfaces after a twenty-one day long interval of relatively stable flows.

Current velocity and the occurrence of spates where shown to be important factors regulating benthic community development. Benthic biomass observations indicated that the optimum range of current velocities, measured 50 mm above the bed, for sewage fungus growth was 0.2 to 0.45 $m.s^{-1}$ but light growths occurred at the maximum velocity investigated of 1.16 $m.s^{-1}$. Growth on bare surfaces increased markedly at velocities above approximately 0.3 $m.s^{-1}$.

Observations of sewage fungus biomass at various depths and growth rates on both upper, sun-light exposed, and lower, shaded, surfaces of concrete plates suspended in the water column of the Manawatu River indicated that solar radiation inhibition of heterotrophic growth is not important in this river.

During summer flow conditions small spates, in which the river flow increased from around 20 $m^3.s^{-1}$ to 50 to 70 $m^3.s^{-1}$, caused preferential sloughing of heterotrophs over algae attached to concrete plates. Flows in excess of approximately 150 $m^3.s^{-1}$ substantially reduced the *C. glomerata* biomass at sites where the current velocity had previously been 0.3 to 0.4 $m.s^{-1}$ at river flows of approximately 20 $m^3.s^{-1}$ but had little apparent effect on the biomass of the macrophyte *Potamogeton crispus* at the same location. The marked reduction in the frequency of peak flush events (i.e., flows in excess of 500 to 1000 $m^3.s^{-1}$) during the study period compared with previous years apparently resulted in the proliferation of this latter species throughout the study reach. Any developments in the upper river catchment that reduced the frequency or magnitude of the peak flush events (e.g., impoundments for hydroelectric power generation or water harvesting) would probably result in macrophytes becoming abundant in the lower Manawatu River.

The results show that it is unlikely that the oxygen requirement of Manawatu River's D classification will be maintained by a river management strategy which limits organic material but not nutrient concentrations. However it is doubtful whether it would be practicable given current wastewater treatment to reduce the wastewater nutrient inputs to a sufficient extent to cause a marked decrease in the algal growth enhancement observed below the discharges. This area is clearly worthy of further investigation.

If this hypothesis is substantiated the river's dissolved oxygen standard will only be consistently maintained if part or all of the wastewaters are diverted to land disposal during the summer lowflow period.

The two objectives of the maintenance of an adequate level of water quality to ensure a healthy and diverse river ecosystem and the use of the river for disposal of large volumes of wastewater produced near Palmerston North may not be compatible in the Manawatu River where conditions are suitable for phototroph proliferation given adequate nutrient levels, high summer river temperature and extended stable flow conditions.

List of Abbreviations

ABM	=	Algal biomass
AFDW	=	Ash free dry weight
AI	=	Autotrophic index
BCWS	=	Borthwick CWS Limited
BGPR	=	Benthic gross photosynthetic oxygen production rate
BM	=	Benthic biomass
BOD_5	=	5 day biochemical oxygen demand
BOD_5UR	=	BOD_5 uptake rate
BR	=	Benthic respiration rate
Chla	=	Chlorophyll a
Chla.SBGPR	=	Chlorophyll a specific benthic gross photosynthetic oxygen production rate
COD	=	Chemical oxygen demand
CPOM	=	Coarse particulate organic material
DIN	=	Dissolved inorganic nitrogen
DO	=	Dissolved oxygen
DRP	=	Dissolved reactive phosphorus
$fBOD_5$	=	BOD_5 of GF/C filtered, seeded sample
k_1	=	First order BOD_5 decay rate
k_2	=	Reaeration rate
k_e	=	Vertical extinction coefficient of light in water
LMRTC	=	Lower Manawatu River Technical Committee
MCDC	=	Manawatu Cooperative Dairy Company
MRWB	=	Manawatu Regional Water Board
PAR	=	Photosynthetically available radiation
PNCC	=	Palmerston North City Corporation
P/R	=	Ratio of gross photosynthesis to respiration rate
Q	=	River flow
RR	=	River respiration rate
sp	=	Species
TN, TDN	=	Total nitrogen, total dissolved nitrogen
TP, TDP	=	Total phosphorus, total dissolved phosphorus
TDNUR	=	Total dissolved nitrogen uptake rate
TDPUR	=	Total dissolved phosphorus uptake rate
WSBGPR	=	Weight specific benthic gross photosynthetic oxygen production rate
WSBR	=	Weight specific benthic respiration rate.

APPENDIX A: MICROSCOPIC EXAMINATION CHECKLIST.

Date = Site = Temp. = C.V. = Depth =

Biomass = Floc Abundance =

Relative Abundance Scale : Very abundant ++++, Abundant +++, Common ++, Uncommon +, Trace -

Dom' Matrix Organisms	Rel. Abund.	Dom. Rotifers	Rel. Abund.	Dom. Macro-inverts	Rel. Abund.	Dom. Protozoans	Rel. Abund.	Dom' Associated Algae	Rel. Abund.
S. natans		Euchlanis		Chiron, SpA		Vorticella		Navicula	
L. lacteus		Philodina		Chiron SpB		Opercularia		Fragilaria	
Zoogloea		Proales				Epistylis		Synedra	
Beggiatoa		Epiphanes		Olinga		Amphileptidae		Cymbella	
Oscillatoria		Rotaria		Deleatidium		C. uncinata		Tabellaria	
Cladophora		Encentrum		P. antipodarum		C. cucullulus		Achnanthes	
Ulothrix				G. corinna		Paramecium		Chlorella	
Lyngbya				P. acuta		Colpidium		Diatoma	
Stigeoclonium				Hydora sp.		Bodo		Staurastrum	
Flavobacterium				H. parumbripennis		Stentor		Nitischia	
				H. umbripennis		Aspidisca		Asterionella	
				P. bidens		Euplotes		Gomphonema	
				N. confusum		Glaucoma		Meriden	
				A. colonica		Uronema		Ankistrodesmus	
				O. albiceps		Tachysoma		Staurastrum	
				Pycnocentrodes spp.				Scenedesmus	
				Hirudinea				Dynobryon	
				Tubificidae				Diatomella	
				Naididae				Rhoicosphenia	
				Nematoda				Selenastrum	
				Polychaeta				Oocystis	
								Chlorococcum	
								Hantschia	

APPENDIX B: KEYS TO EPILITHON MACROSCOPIC ABUNDANCE SCALES

A. Phototroph/Heterotroph Dominance Scale, P_A/H_A (After Wuhrmann, 1954)

Classification	Observation
0	Phototrophs only.
3	Phototrophs dominant, but small colonies of sessile ciliates clearly visible on stone surfaces and/or strands of filamentous heterotrophic microorganisms on undersides of stones.
5	Phototrophs dominant, *S. natans* or other filamentous heterotrophic microorganisms visible in small colonies or sessile celiates in large colonies.
7	Proportion in relative abundance of phototrophic and heterotrophic microorganisms around 1:1.
9	Heterotrophic microorganisms dominant, autotrophs still persisting in clearly visible groups.
11	Heterotrophic microorganisms only, forming mass developments. Thick anaerobic sludge below surface growth.
13	Heterotrophic microorganisms only. Anaerobic associations eve in the surface layer of growth.

B. Heterotrophic Microorganisms

(i) Filamentous/Zooloeal forms (Primary heterotrophs)

Classification	Observation
0	Not visible on hand-held boulders.
1	Visible on hand-held boulders but not visible on bed.
2	Strands visible on stones, no floc units (= 1 cm lengths of floc).
3	Isolated flocs visible <5/periscope field (= 177 cm^2).
4	Flocs common.
5	Flocs abundant covering most surfaces.
6.	Flocs covering all surfaces.

(ii) Sessile ciliates (Secondary heterotrophs)

I	Not visible on hand-held boulders.
II	Isolated colonies on undersides of boulders.
III	Isolated colonies on surfaces of boulders.
IV	Abundant colonies on surfaces of boulders.

C. Phototrophic Microorganisms

Classification	Observation
A	Not visible on hand-held boulders.
B	Visible on hand-held boulders but not visible on bed.
C	Present as clearly visible colonies on bed.
D	Covering many surfaces.
E	Covering most surfaces.
F	Covering bed.

APPENDIX C Data from *in situ* Freeman Chamber Studies on Plate Biomass at Site D

Date	Plate Incubation Period (days)	Temperature (°C)	BR ($gO2.m^{-2}.d^{-1}$)	Initial BOD_5 ($g.m^{-3}$)	Initial DO ($g.m^{-3}$)	Biomass ($gAFDW.m^{-2}$)	AI	WSBR ($gO_2.gAFDW^{-1}.d^{-1}$)	BGPR* ($gO_2.m^{-2}.d^{-1}$)	P/R*	ChlaSBGPR* ($gO_2.gChla^{-1}.d^{-1}$)	PAR ($uE.m^{-2}.s^{-1}$)	ABM ($mgChla.m^{-2}$)
16/12/83	26	18.5	6.8	ND	11.0	12.7	83	0.31	10.60	1.56	51	400	156.0
08/12/83	17	21.7	13.4	1.6	11.1	34.9	160	0.38	13.60	1.01	62	400	218.0
08/12/83	23	20.7	12.5	1.5	10.1	29.6	161	0.42	21.00	1.68	114	800	184.0
01/12/83	10	18.3	10.9	ND	10.5	17.3	585	0.63	9.80	0.90	334	800	29.5
01/12/83	16	17.5	9.8	ND	10.6	17.7	444	0.55	5.65	0.32	140	600	40.0
24/11/83	3	17.0	4.3	5.0	10.6	4.8	3637	0.90	ND	ND	ND	ND	*
24/11/83	9	15.5	5.6	ND	11.0	8.1	855	0.69	4.00	0.71	425	900	9.5
18/11/83**	3	18.5	3.5	ND	8.9	14.5	ND	0.24	ND	ND	ND	ND	*

* calculated assuming constant photosynthesis for 12 hours daylight/day.
** data not included in linear regression analyses due to settled silt and detritus.

APPENDIX D Data from *in situ* Boyle Chamber Studies on the Effects of Natural Bed Biomass on River Oxygen Depletion

Site	Date	Temperature (°C)	BR ($gO_2.m^{-2}.d^{-1}$)	Initial BOD_5 ($g.m^{-3}$)	Biomass ($gAFDW.m^{-2}$)	Heterotroph Abundance level	Phototroph Abundance level	Weight Specific BR ($gO_2.gAFDW^{-1}.d^{-1}$)	Initial DO ($g.m^{-3}$)
D	08/12/83	20.7	13.8	6.7	36.2	5	D	0.38	11.0
D	24/11/83	15.3	4.9	4.1	11.6	3	C	0.42	10.8
A	21/04/83	17.0	19.4	*	50.0	0	F	0.39	12.7
D	15/04/84	15.3	14.8	4.6	62.0	*	*	0.24	9.5
A	12/04/83	13.8	6.8	*	50.2	0	E	0.13	10.2
D	11/04/83	14.5	17.2	*	106.0	4.5	*	0.16	10.5

* no data

APPENDIX E: Data from Freeman *in situ* Chamber Studies on the Effects of Natural Bed Biomass on River Oxygen Dynamics

SITE	DATE	TEMPERATURE (^{0}C)	BR ($gO_2.m^{-2}.d^{-1}$)	INITIAL BOD_5 ($g.m^{-3}$)	BIOMASS ($gAFDW.m^{-2}$)	AI	HETEROTROPH ABUNDANCE LEVEL	PHOTOTROPH ABUNDANCE LEVEL	WSBR ($gO_2.gAFDW^{-1}$-d)	BGPR** ($gO_2.m^{-2}.d^{-1}$)	PR**	W.S.BGRP** ($gO_2.gAFDW^{-1}.d^{-1}$)	Chla.SBGPR** ($gO_2.gCHla^{-1}.d^{-1}$)	PAR ($\mu E.m^{-2}.s^{-1}$)	Chl.a ($mg.m^{-2}$)	INITIAL DO ($g.m^{-3}$)	RUNS USED IN REGRESSION ANALYSES
D	19/06/84	11.00	3.80	1.4	53.0	137	2.0	F	0.072	6.00	1.58	0.115	15.7	100	383	11.6	✓
D	19/06/84	11.00	1.50	1.4	18.6	239	4.0	C	0.082	2.30	0.94	0.120	29.5	100	78.0	10.5	✓
C	26/04/84	15.00	19.60	7.0	104.4	197	4.0	D	0.188	13.70	0.95	0.132	26.0	210	529	10.2	
C	26/04/84	15.00	17.50	5.5	92.6	158	4.0	D	0.189	12.50	0.99	0.135	21.3	300	587	9.3	✓
C	23/03/84	17.50	7.60	1.2	24.0	307	ND	ND	0.316	5.30	0.70	0.220	67.9	800	99	9.2	✓
C	23/03/84	12.75	4.70	1.2	22	346	ND	ND	0.213	ND	ND	ND	ND	ND	90	9.2	
C	23/03/84	15.00	7.00	6.0	22	346	ND	ND	0.320	ND	ND	ND	ND	ND	90	8.5	
C	23/03/84	17.40	10.30	7.1	24.0	307	ND	ND	0.430	6.50	0.63	0.270	83.0	400	99	9.4	
EF	12/03/84	21.25	14.60	1.7	55.3	100	0.0	E	0.260	31.30	2.14	0.570	ND	800*	690	7.0	✓
EF	12/03/84	20.25	4.80	2.0	31.0	68	0.0	E	0.150	6.00	1.60	0.250	16.7	800*	456	7.8	✓
C	22/02/84	13.00	15.20	5.5	70.0	323	5.5	E	0.217	15.40	1.23	0.220	71.0	800*	275	9.5	✓
C	22/02/84	18.00	11.40	5.5	61.1	282	5.5	E	0.186	16.40	1.50	0.270	75.5	800*	307	8.5	✓
DE	02/02/84	23.20	30.70	4.6	45.8	ND	2.0	F	0.67	32.20	1.39	0.700	ND	800*	ND	10.5	✓
DE	02/02/84	24.00	28.20	13.8	48.7	ND	2.0	F	0.58	36.10	1.58	0.740	ND	800*	ND	10.5	✓
DE	18/01/84	17.50	13.61	3.7	50.6	100	1.0	E	0.270	14.73	1.08	0.29	45.0	400*	412	11.8	✓
DE	18/01/84	17.50	13.40	8.0	50.0	100	1.0	E	0.270	16.46	1.23	0.330	34.0	400*	601	11.5	✓
C	12/01/84	20.25	21.50	7.3	54.6	664	6.0	C	0.393	11.00	0.62	0.200	ND	400*	84	5.3	✓
C	12/01/84	20.25	18.30	14.0	49.8	664	6.0	C	0.370	9.48	0.62	0.190	126.0	400*	75	5.0	✓
D	24/11/83	15.7	5.2	5.0	8.5	ND	4	C	0.613	ND	ND	ND	ND	ND	ND	10.6	✓

* estimated values for clear (800 $uE.m^{-2}.s^{-1}$) and overcast conditions (400 $uE.m^{-2}.s^{-1}$)

** calculated assuming constant gross photosynthesis at measured rate for 12 hours/day

APPENDIX F Results of Whole River Dissolved Oxygen Dynamics Studies Below the Effluent Discharges

Reach	Date	Q	Mean Night time Respiration ($g.m^{-3}.d$)	Mean Night time Respiration ($g.m^{-2}.d^{-1}$)	Gross Oxygen Production ($g.m^{-3}.d^{-1}$)	Gross Oxygen Production ($g.m^{-2}.d^{-1}$)	P/R	T (°C)	AFDW ($g.m^{-2}$)	Macroscopic Abundance Hetero-trophs	Macroscopic Abundance Photo-trophs	Average surface raditation (Watt $hour.d^{-1}$)	Initial BOD_5 ($g.m^{-3}$)
D-Fu	24/11/82	31.0	25.90	23.7	*	*	*	15.6	*	4.5	C	8280	9.00
D-E/F	15/11/83	58.0	11.40	13.7	*	*	*	16.0	26	4.0	B	6628	2.50
D-E/F	17/11/83	41.0	11.60	12.5	*	*	*	18.0	26	4.0	C	7227	2.00
D-E/F	21/11/83	34.0	19.80	18.2	6.20	6.80	0.31	16.8	34	2.5	C	6023	5.00
D-Fu	29/11/83	22.0	13.30	10.8	9.80	7.90	0.74	15.2	44	3.0	D	5597	5.30
Dd-EF	18/01/84	17.3	27.90	28.8	*	*	*	15.5	130	0.0	F	2817	8.00
Dd-EF	24/01/84	14.8	31.90	32.5	22.80	22.40	0.69	19.7	150	0.5	F	7452	5.20
Dd-EF	31/01/84	14.8	36.90	37.6	31.10	31.70	0.84	18.2	150	1.5	F	8382	6.40
Dd-EF	09/02/84	18.3	30.30	32.4	15.90	17.00	0.52	19.7	130	1.5	F	7793	2.80
D-E	22/02/83	20.0	31.50	32.8	12.70	13.20	0.40	19.0	60	4.0	C-D	6802	8.90
D-E	06/12/84	19.0	33.60	33.1	34.70	34.20	1.03	20.3	70	5.0	D	4004	8.15
Dd-E	15/12/84	30.0	23.40	21.7	*	*	*	18.4	60	5.0	D	6611	6.46
PNCC-B	16/01/84	18.0	18.80	20.7	*	*	*	17.2	83	1.0	D	5945	2.40
PNCC-B	29/02/84	16.8	23.90	26.7	13.50	15.10	0.58	21.0	100	1.0	F	5866	3.70
MCDC-Cu	29/02/84	16.8	35.20	39.8	-4.35	-4.90	-0.12	20.7	150	6.0	D	5866	6.80
B-D	15/01/84	16.9	22.90	19.3	9.20	7.75	0.39	17.5	120	5.5	D-E	5802	4.10
B-D	16/01/84	18.0	24.50	20.6	*	*	*	17.0	120	5.5	D-E	5945	4.70
D-E	06/01/84	16.8	31.95	27.8	32.67	28.40	1.02	18.2	100	3.0	F	6391	5.90

* No data

APPENDIX G: Results of Lactose Analyses on Twenty-four hour Flow-related Composite MCDC Wastewater Samples

Date	COD* ($g.m^{-3}$)	Lactose ($g.m^{-3}$)	Lactose COD as % total COD (%)
19/12/82	3862	980	28.5
21/12/82	ND	990	ND
31/12/82	4703	1310	31
05/01/83	3856	1020	30
20/01/83	3851	690	20
22/01/83	1892	790	47
23/01/83	2037	1010	56
24/01/83	2379	800	38
01/02/83	4542	1120	28
09/02/83	3121	1280	46

* results of analyses by MCDC staff (Meredith, 1983)

APPENDIX H Results of Wastewater Nutrient Analyses

(a) Results of Nutrient Analyses of twenty-four hour, flow-related, composite MCDC Wastewater Samples.

Date	BOD_5* ($g.m^{-3}$)	TN ($g.m^{-3}$) ($\bar{x} \pm s$)	TP ($g.m^{-3}$) ($\bar{x} \pm s$)	Ratio BOD_5:N:P
31/12/82	3057	76.4 ± 0.35	65.5 ± 2.7	47:1.2:1
20/11/83	3850	ND	56.2	68:ND:1
22/01/83	1892	103.4 ± 1.5	60	31:1.7:1
23/01/82	2038	.92 ± 13	ND	
09/02/83	2035	ND	33.6 ± 1.2	60:ND:1
29/03/83	1474	64.0 ± 5.6	43.9 ± 2.1	34:15:1
14/03/83	1085	66.1 ± 9.9	49.7	22:1.3:1
12/04/83	1767	61.7 ± 7.1	57.9 ± 4.5	29:1.1:1
08/05/83	2099	ND	32.2	65:ND:1
09/01/84	794	39.8 ± 0.8	17.7 ± 0.1	43:2.2:1
10/01/84	2136	75.5 ± 1.9	44.6 ± 2.0	48:1.7:1
25/01/84	772	44.8 ± 3.8	10.8 ± 1.3	71:4.1:1

* MCDC Analyses Results (Meredith, 1984)

(b) Results of Nutrient Analysis of a twenty-four hour flow-related composite sample of PNCC wastewater after Treatment by Primary Sedimentation with Alum flocculation.

BOD_5 ($g.m^{-3}$)*	Total Nitrogen ($g.m^{-3}$)	Total Phosphorus ($g.m^{-3}$)
190	19.6	3.6

* PNCC Analysis Result (Anderson, 1984)

APPENDIX I: River Environmental Data, Wastewater Discharge Data, and Calculated BOD_5 Values at River Sites.

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
1	10282	25.5	12535	9078	27.2	2.5	6.8	8.7	6.3	3.9	3.6
2	20282	23.2	12963	8089	27.8	2.6	7.4	9.1	6.5	4.0	3.7
3	30282	10.5	8620	8480	11.4	4.3	10.7	14.7	9.2	4.7	4.2
4	40282	14.7	9766	7015	7.7	3.5	8.9	11.0	7.3	4.1	3.7
5	50282	13.8	11609	8798	25.5	3.6	10.4	13.5	8.9	4.9	4.4
6	60282	13.5	11287	0	23.7	3.7	10.4	8.4	5.5	3.0	2.7
7	70282	14.6	9118	0	1.5	3.5	8.6	6.9	4.6	2.6	2.3
8	80282	14.9	13031	10280	ND	3.5	10.6	14.1	9.4	5.3	4.8
9	90282	14.6	ND	7736	ND	3.5	ND	ND	ND	ND	ND
10	100282	13.1	ND	10299	ND	3.8	ND	ND	ND	ND	ND
11	110282	12.7	ND	9810	ND	3.8	ND	ND	ND	ND	ND
12	120282	12.6	12222	9712	ND	3.9	11.6	15.4	10.0	5.4	4.8
13	130282	12.4	17026	0	ND	3.9	14.8	11.8	7.6	4.1	3.7
14	140282	12.4	16114	0	ND	3.9	14.2	11.3	7.3	3.9	3.5
15	150282	12.2	ND	9717	ND	3.9	ND	ND	ND	ND	ND
16	160282	12.0	ND	10321	ND	4.0	ND	ND	ND	ND	ND
17	170282	12.0	10240	8983	ND	4.0	10.7	14.4	9.3	4.9	4.4
18	180282	11.9	11102	0	23.4	4.0	11.4	9.0	5.8	3.1	2.8
19	190282	172.0	11241	0	24.7	0.9	1.6	1.5	1.3	1.0	1.0
20	200282	142.0	8329	0	23.4	1.0	1.6	1.5	1.3	1.0	0.9
21	210282	43.1	15894	0	9.9	1.8	5.2	4.5	3.5	2.4	2.2
22	220282	26.7	11196	0	24.5	2.4	6.1	5.1	3.7	2.4	2.2
23	230282	37.9	9650	0	12.8	1.9	4.3	3.7	2.8	1.9	1.7
24	240282	80.5	ND	6	15.3	1.3	ND	ND	ND	ND	ND
25	250282	97.0	ND	0	21.2	1.2	ND	ND	ND	ND	ND
26	260282	105.8	9146	0	17.2	1.1	2.0	1.8	1.5	1.2	1.1
27	270282	63.4	13922	0	23.7	1.5	3.5	3.2	2.5	1.8	1.7
28	280282	41.2	10604	0	8.2	1.8	4.2	3.7	2.8	1.9	1.8
29	10382	30.0	11050	0	8.1	2.2	5.5	4.7	3.5	2.2	2.1
30	20382	31.0	12472	0	19.6	2.2	5.8	4.9	3.6	2.4	2.2
31	30382	70.3	9037	0	ND	1.4	2.6	2.3	1.9	1.4	1.3
32	40382	186.5	9737	8740	ND	0.9	1.5	1.8	1.5	1.2	1.2
33	50382	87.0	10532	9562	ND	1.2	2.4	3.2	2.6	1.9	1.9
34	60382	50.6	10686	0	ND	1.6	3.6	3.2	2.5	1.7	1.6
35	70382	35.6	9972	0	ND	2.0	4.5	3.9	2.9	2.0	1.8
36	80382	27.5	8774	10044	ND	2.4	5.2	7.5	5.4	3.4	3.2
37	90382	21.1	9101	10546	ND	2.8	6.5	9.5	6.7	4.0	3.7
38	100382	19.2	14485	10094	ND	2.9	9.3	12.0	8.3	4.9	4.5
39	110382	17.7	8113	9218	14.2	3.1	6.9	9.9	6.8	4.0	3.6
40	120382	15.2	12296	6202	16.7	3.4	10.1	11.4	7.7	4.3	3.9
41	130382	14.8	6207	0	11.8	3.5	6.9	5.6	3.7	2.1	1.9
42	140382	13.9	11134	0	9.9	3.6	10.1	8.1	5.4	2.9	2.7
43	150382	14.4	11528	9057	13.9	3.5	10.1	13.2	8.7	4.8	4.4
44	160382	12.8	11414	8331	ND	3.8	11.0	13.9	9.0	4.9	4.4
45	170382	16.0	ND	3576	ND	3.3	ND	ND	ND	ND	ND
46	180382	16.8	ND	8896	ND	3.2	ND	ND	ND	ND	ND
47	190382	20.1	16981	7859	ND	2.9	10.0	11.5	8.1	4.8	4.4
48	200382	18.2	9282	0	ND	3.0	7.3	6.0	4.1	2.4	2.2
49	210382	16.0	14390	0	12.5	3.3	10.7	8.7	5.9	3.3	3.0
50	220382	15.1	ND	8050	18.3	3.4	ND	ND	ND	ND	ND
51	230382	14.8	11910	9111	5.2	3.9	11.2	14.8	11.0	7.2	6.7
52	240382	16.4	21538	9279	ND	3.7	15.6	18.3	13.8	9.2	8.6
53	250382	19.1	11440	8859	ND	3.3	8.8	11.7	8.9	6.1	5.7
54	260382	21.1	15252	8718	ND	3.1	9.8	12.2	9.4	6.5	6.1
55	270382	21.0	14430	0	ND	3.1	9.5	8.3	6.4	4.4	4.2
56	280382	19.2	16128	0	ND	3.3	11.0	9.6	7.3	5.0	4.7
57	290382	15.3	11260	0	ND	3.9	10.5	9.0	6.7	4.4	4.1
58	300382	14.8	11965	7751	ND	3.9	11.2	14.1	10.5	6.9	6.4
59	310382	14.4	17556	10265	ND	4.0	14.9	18.8	14.0	9.1	8.5
60	10482	19.4	17025	9071	ND	3.3	11.3	13.9	10.7	7.3	6.8
61	20482	38.7	17856	8330	ND	2.1	6.5	7.8	6.4	4.8	4.6
62	30482	30.5	15280	0	ND	2.4	7.2	6.4	5.1	3.7	3.5
63	40482	23.7	8278	0	ND	2.8	6.1	5.4	4.2	3.0	2.8
64	50482	38.6	13389	8335	ND	2.1	5.4	6.8	5.6	4.2	4.0
65	60482	24.1	14261	6271	ND	2.8	8.4	9.6	7.6	5.3	5.0

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
66	70482	29.6	11679	4348	ND	2.4	6.2	6.8	5.5	4.0	3.7
67	80482	43.1	15555	0	ND	1.9	5.5	4.9	4.1	3.1	2.9
68	90482	40.5	12187	0	ND	2.0	4.9	4.5	3.7	2.8	2.6
69	100482	68.9	17939	0	ND	1.5	4.1	3.8	3.2	2.6	2.5
70	110482	61.4	ND	0	ND	1.6	ND	ND	ND	ND	ND
71	120482	54.5	ND	0	ND	1.7	ND	ND	ND	ND	ND
72	130482	44.0	13622	6618	ND	2.0	5.2	6.2	5.4	4.4	4.3
73	140482	36.6	3339	9120	ND	2.2	3.2	5.2	4.5	3.6	3.5
74	150482	31.7	4523	0	ND	2.5	3.9	3.6	3.1	2.5	2.4
75	160482	31.0	8042	8713	ND	2.5	5.1	7.3	6.2	4.9	4.8
76	170482	33.8	2836	0	ND	2.4	3.2	3.0	2.5	2.0	2.0
77	180482	28.5	2843	0	ND	2.7	3.7	3.4	2.9	2.3	2.2
78	190482	22.6	2015	2161	ND	3.2	4.0	4.5	3.8	2.9	2.8
79	200482	20.7	2444	7739	ND	3.4	4.5	7.5	6.2	4.7	4.5
80	210482	18.7	ND	4462	ND	3.6	ND	ND	ND	ND	ND
81	220482	17.5	1501	5731	ND	3.8	4.7	7.1	5.8	4.4	4.2
82	230482	19.6	1222	6288	ND	3.5	4.1	6.6	5.5	4.1	4.0
83	240482	31.0	1771	0	ND	2.5	3.1	2.8	2.4	1.9	1.9
84	250482	28.0	1650	0	ND	2.7	3.3	3.0	2.6	2.0	1.9
85	260482	22.2	1424	8317	ND	3.2	3.8	6.9	5.7	4.4	4.2
86	270482	21.1	2309	8524	ND	3.3	4.4	7.6	6.3	4.9	4.6
87	280482	ND	616	8422	ND	ND	ND	ND	ND	ND	ND
88	290482	ND	ND	8088	ND	ND	ND	ND	ND	ND	ND
89	300482	ND	1849	7756	ND	ND	ND	ND	ND	ND	ND
90	10582	21.4	0	0	ND	3.3	3.3	3.0	2.5	1.9	1.8
91	20582	23.3	0	0	ND	3.1	3.1	2.8	2.4	1.8	1.7
92	30582	41.8	0	7639	ND	2.1	2.1	3.6	3.1	2.5	2.5
93	40582	40.7	0	8338	ND	2.1	2.1	3.8	3.3	2.7	2.6
94	50582	32.6	0	4619	ND	2.4	2.4	3.5	3.0	2.4	2.3
95	60582	27.2	0	6362	ND	2.8	2.8	4.7	3.9	3.1	3.0
96	70582	23.8	0	7284	ND	3.0	3.0	5.5	4.6	3.6	3.4
97	80582	22.6	0	0	ND	3.2	3.2	2.9	2.4	1.8	1.8
98	90582	22.5	0	0	ND	3.2	3.2	2.9	2.4	1.9	1.8
99	100582	23.5	0	7442	ND	3.1	3.1	5.7	4.7	3.7	3.5
100	110582	27.8	0	8427	ND	2.7	2.7	5.3	4.4	3.5	3.4
101	120582	33.4	0	7864	ND	2.5	2.5	4.6	4.1	3.5	3.4
102	130582	64.3	0	8283	ND	1.6	1.6	2.8	2.5	2.2	2.2
103	140582	134.6	0	6762	ND	1.1	1.1	1.5	1.5	1.3	1.3
104	150582	111.8	0	0	ND	1.2	1.2	1.2	1.1	1.0	1.0
105	160582	90.5	0	0	ND	1.3	1.3	1.3	1.2	1.1	1.0
106	170582	63.1	0	7285	ND	1.6	1.6	2.7	2.4	2.2	2.1
107	180582	49.3	0	6065	ND	1.9	1.9	3.0	2.7	2.4	2.3
108	190582	42.0	0	6787	ND	2.1	2.1	3.5	3.2	2.8	2.7
109	200582	40.3	0	0	ND	2.2	2.2	2.1	1.9	1.6	1.6
110	210582	134.0	0	5650	ND	1.1	1.1	1.5	1.4	1.3	1.2
111	220582	111.2	0	0	ND	1.2	1.2	1.2	1.1	1.0	1.0
112	230582	134.7	0	0	ND	1.1	1.1	1.1	1.0	0.9	0.9
113	240582	254.2	0	6871	ND	0.9	0.9	1.1	1.1	1.0	1.0
114	250582	212.6	0	4770	ND	0.9	0.9	1.1	1.1	1.0	1.0
115	260582	161.6	0	7294	ND	1.0	1.0	1.4	1.3	1.2	1.2
116	270582	114.0	0	7648	ND	1.2	1.2	1.8	1.7	1.5	1.5
117	280582	118.5	0	6740	ND	1.2	1.2	1.7	1.6	1.4	1.4
118	290582	76.5	0	0	ND	1.5	1.5	1.4	1.3	1.2	1.1
119	300582	57.0	0	0	ND	1.7	1.7	1.7	1.5	1.3	1.3
120	310582	55.5	0	5133	ND	1.8	1.8	2.6	2.3	2.1	2.0
121	10682	47.5	0	6584	ND	2.0	2.0	3.2	2.9	2.5	2.4
122	20682	41.5	0	8003	ND	2.1	2.1	3.9	3.5	3.0	2.9
123	30682	36.5	0	7933	ND	2.3	2.3	4.3	3.8	3.3	3.2
124	40682	34.0	0	0	ND	2.5	2.5	2.3	2.1	1.8	1.7
125	50682	ND	0	0	ND	ND	ND	ND	ND	ND	ND
126	60682	ND	0	0	ND	ND	ND	ND	ND	ND	ND
127	70682	ND	0	0	ND	ND	ND	ND	ND	ND	ND
128	80682	215.0	0	4848	ND	0.9	0.9	1.1	1.1	1.0	1.0
129	90682	153.0	0	4547	ND	1.0	1.0	1.3	1.2	1.1	1.1
130	100682	116.0	0	5606	ND	1.2	1.2	1.6	1.5	1.4	1.3
131	110682	96.0	0	4216	ND	1.3	1.3	1.7	1.5	1.4	1.4
132	120682	ND	0	0	ND	ND	ND	ND	ND	ND	ND
133	130682	ND	0	0	ND	ND	ND	ND	ND	ND	ND
134	140682	72.5	0	0	ND	1.5	1.5	1.4	1.3	1.2	1.2
135	150682	66.0	0	4420	ND	1.6	1.6	2.2	2.0	1.8	1.7

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
136	160682	60.5	0	5083	ND	1.7	1.7	2.4	2.2	1.9	1.9
137	176682	56.5	0	4132	ND	1.8	1.8	2.4	2.2	1.9	1.9
138	180682	60.0	0	0	ND	1.7	1.7	1.6	1.5	1.3	1.3
139	190682	78.0	0	0	ND	1.4	1.4	1.4	1.3	1.1	1.1
140	200682	90.0	0	0	ND	1.3	1.3	1.3	1.2	1.1	1.1
141	210682	90.0	0	0	ND	1.3	1.3	1.3	1.2	1.1	1.1
142	220682	74.0	0	4739	ND	1.5	1.5	2.0	1.9	1.7	1.6
143	230682	72.5	0	4448	ND	1.5	1.5	2.0	1.9	1.7	1.6
144	240682	97.0	0	4222	ND	1.3	1.3	1.7	1.5	1.4	1.4
145	250682	77.0	0	0	ND	1.5	1.5	1.4	1.3	1.1	1.1
146	260682	101.0	0	0	ND	1.3	1.3	1.2	1.1	1.0	1.0
147	270682	145.0	0	0	ND	1.1	1.1	1.0	1.0	0.9	0.9
148	280682	114.0	0	0	ND	1.2	1.2	1.1	1.1	1.0	1.0
149	290682	84.0	0	4004	ND	1.4	1.4	1.8	1.7	1.5	1.5
150	300682	70.5	0	3741	ND	1.5	1.5	2.0	1.8	1.6	1.6
151	10782	62.5	0	5379	ND	1.6	1.6	2.4	2.1	1.9	1.8
152	20782	56.5	0	4484	ND	1.7	1.7	2.4	2.2	1.9	1.8
153	30782	55.5	0	0	ND	1.8	1.8	1.7	1.5	1.3	1.3
154	40782	89.0	0	0	ND	1.3	1.3	1.3	1.2	1.0	1.0
155	50782	136.5	0	2265	ND	1.1	1.1	1.2	1.1	1.0	1.0
156	60782	82.5	0	4099	ND	1.4	1.4	1.8	1.7	1.5	1.4
157	70782	66.5	0	4230	ND	1.6	1.6	2.1	1.9	1.7	1.6
158	80782	58.0	0	4299	ND	1.7	1.7	2.3	2.1	1.8	1.8
159	90782	51.0	0	4219	ND	1.9	1.9	2.5	2.3	2.0	1.9
160	100782	45.2	0	0	ND	2.0	2.0	1.9	1.7	1.4	1.4
161	110782	42.5	0	0	ND	2.1	2.1	2.0	1.8	1.5	1.5
162	120782	41.0	0	0	ND	2.1	2.1	2.0	1.8	1.5	1.5
163	130782	45.7	0	2159	ND	2.0	2.0	2.3	2.1	1.8	1.7
164	140782	98.1	0	4101	ND	1.3	1.3	1.6	1.5	1.3	1.3
165	150782	120.6	0	3213	ND	1.2	1.2	1.4	1.3	1.1	1.1
166	160782	98.2	0	0	ND	1.3	1.3	1.2	1.1	1.0	1.0
167	170782	73.9	0	0	ND	1.5	1.5	1.4	1.3	1.1	1.1
168	180782	70.8	0	0	ND	1.5	1.5	1.4	1.3	1.2	1.1
169	190782	98.0	0	1364	ND	1.3	1.3	1.4	1.2	1.1	1.1
170	200782	81.0	0	1629	ND	1.4	1.4	1.5	1.4	1.2	1.2
171	210782	980.0	0	1657	ND	0.7	0.7	0.7	0.7	0.6	0.6
172	220782	400.0	0	1656	ND	0.8	0.8	0.8	0.8	0.7	0.7
173	230782	211.0	0	1473	ND	0.9	0.9	1.0	0.9	0.8	0.8
174	240782	144.0	0	0	ND	1.1	1.1	1.0	1.0	0.9	0.9
175	250782	114.0	0	0	ND	1.2	1.2	1.1	1.1	0.9	0.9
176	260782	102.0	0	1120	ND	1.2	1.2	1.3	1.2	1.1	1.1
177	270782	89.0	0	1543	ND	1.3	1.3	1.4	1.3	1.2	1.2
178	280782	78.0	0	1610	ND	1.4	1.4	1.6	1.4	1.3	1.2
179	290782	74.0	0	1535	ND	1.5	1.5	1.6	1.5	1.3	1.3
180	300782	188.0	0	1398	ND	1.0	1.0	1.0	0.9	0.9	0.8
181	310782	117.0	0	0	ND	1.2	1.2	1.1	1.0	0.9	0.9
182	10882	102.0	0	0	ND	1.2	1.2	1.2	1.1	1.0	1.0
183	20882	117.0	0	0	ND	1.2	1.2	1.1	1.0	0.9	0.9
184	30882	263.0	0	0	ND	0.9	0.9	0.8	0.8	0.7	0.7
185	40882	149.0	0	0	ND	1.1	1.1	1.0	0.9	0.9	0.8
186	50882	115.0	0	0	ND	1.2	1.2	1.1	1.0	0.9	0.9
187	60882	97.5	0	0	ND	1.3	1.3	1.2	1.1	1.0	1.0
188	70882	81.7	0	0	ND	1.4	1.4	1.3	1.2	1.1	1.1
189	80882	80.0	0	0	ND	1.4	1.4	1.4	1.2	1.1	1.1
190	90882	79.5	0	0	ND	1.4	1.4	1.4	1.2	1.1	1.1
191	100882	67.0	0	0	ND	1.6	1.6	1.5	1.4	1.2	1.2
192	110882	71.0	0	0	ND	1.5	1.5	1.4	1.3	1.2	1.1
193	120882	118.0	0	0	ND	1.2	1.2	1.1	1.0	0.9	0.9
194	130882	77.5	0	0	ND	1.4	1.4	1.4	1.3	1.1	1.1
195	140882	65.0	0	0	ND	1.6	1.6	1.5	1.4	1.2	1.2
196	150882	62.7	0	0	ND	1.6	1.6	1.6	1.4	1.2	1.2
197	160882	63.5	0	0	ND	1.6	1.6	1.5	1.4	1.2	1.2
198	170882	58.0	0	0	ND	1.7	1.7	1.6	1.5	1.3	1.2
199	180882	58.0	10000	0	ND	1.7	3.5	3.4	3.0	2.6	2.6
200	190882	102.0	10000	0	ND	1.2	2.3	2.2	2.0	1.8	1.8
201	200882	77.4	10000	0	ND	1.4	2.8	2.7	2.5	2.2	2.1
202	210882	66.7	10000	0	ND	1.6	3.2	3.0	2.7	2.4	2.3
203	220882	59.4	10000	0	ND	1.7	3.5	3.3	3.0	2.6	2.5
204	230882	58.0	10000	0	ND	1.7	3.5	3.4	3.0	2.6	2.6
205	240882	53.0	10000	0	ND	1.8	3.8	3.6	3.2	2.8	2.7

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
206	250882	48.5	10000	0	ND	1.9	4.1	3.9	3.5	3.0	2.9
207	260882	53.5	10000	0	ND	1.8	3.8	3.6	3.2	2.8	2.7
208	270882	60.0	10000	0	ND	1.7	3.4	3.3	3.0	2.6	2.5
209	280882	45.5	ND	0	ND	2.0	ND	ND	ND	ND	ND
210	290882	53.1	ND	0	ND	1.8	ND	ND	ND	ND	ND
211	300882	62.3	ND	0	ND	1.6	ND	ND	ND	ND	ND
212	310882	49.0	ND	0	ND	1.9	ND	ND	ND	ND	ND
213	10982	65.5	16005	0	7.0	1.6	4.2	4.0	3.6	3.2	3.1
214	20982	91.5	21347	0	14.7	1.3	3.8	3.7	3.4	3.0	2.9
215	280882	45.5	ND	0	ND	2.0	ND	ND	ND	ND	ND
216	290882	53.1	ND	0	ND	1.8	ND	ND	ND	ND	ND
217	300882	62.3	ND	0	ND	1.6	ND	ND	ND	ND	ND
218	310882	49.0	ND	0	ND	1.9	ND	ND	ND	ND	ND
219	10982	65.5	16005	0	7.0	1.6	4.2	4.0	3.6	3.2	3.1
220	20982	91.5	21347	0	14.7	1.3	3.8	3.7	3.4	3.0	2.9
221	30982	62.3	14572	0	13.0	1.6	4.1	3.9	3.6	3.1	3.0
222	40982	ND	16512	0	5.3	ND	ND	ND	ND	ND	ND
223	50982	ND	9081	0	10.8	ND	ND	ND	ND	ND	ND
224	60982	92.6	ND	0	14.8	1.3	ND	ND	ND	ND	ND
225	70982	62.3	ND	0	12.8	1.6	ND	ND	ND	ND	ND
226	80982	60.6	8150	0	2.3	1.7	3.1	2.9	2.7	2.3	2.3
227	90982	64.0	11129	0	12.9	1.6	3.5	3.3	3.0	2.6	2.5
228	100982	64.4	ND	0	0.0	1.6	ND	ND	ND	ND	ND
229	110987	46.5	6846	0	8.3	2.0	3.5	3.3	3.0	2.5	2.5
230	120982	65.9	ND	0	13.6	1.6	ND	ND	ND	ND	ND
231	130982	99.6	ND	0	15.7	1.3	ND	ND	ND	ND	ND
232	140982	73.0	13205	0	13.6	1.5	3.4	3.2	2.9	2.5	2.4
233	150982	64.0	13766	0	14.8	1.6	3.8	3.6	3.2	2.8	2.7
234	160982	59.5	10125	0	17.4	1.7	3.4	3.2	2.9	2.4	2.4
235	170982	52.7	11138	0	18.4	1.8	4.0	3.7	3.3	2.8	2.7
236	180982	50.5	ND	0	17.2	1.8	ND	ND	ND	ND	ND
237	190982	45.2	14055	0	11.4	2.0	5.2	4.8	4.3	3.5	3.4
238	200982	46.8	ND	0	7.0	1.9	ND	ND	ND	ND	ND
239	210982	123.3	17266	0	10.9	1.1	2.6	2.5	2.3	2.0	2.0
240	220982	84.7	ND	0	11.7	1.4	ND	ND	ND	ND	ND
241	230982	67.0	ND	0	21.1	1.5	ND	ND	ND	ND	ND
242	240982	55.8	10730	0	16.5	1.7	3.7	3.5	3.1	2.6	2.5
243	250982	54.8	ND	0	7.6	1.7	ND	ND	ND	ND	ND
244	260982	57.8	ND	0	6.6	1.7	ND	ND	ND	ND	ND
245	270982	77.0	12950	0	13.0	1.4	3.2	3.0	2.7	2.4	2.3
246	280982	76.1	ND	0	21.8	1.4	ND	ND	ND	ND	ND
247	290982	82.4	ND	0	23.2	1.4	ND	ND	ND	ND	ND
248	300982	67.4	12400	0	15.1	1.5	3.5	3.3	2.9	2.5	2.4
249	11082	54.0	12677	0	14.2	1.8	4.2	3.9	3.5	2.9	2.9
250	21082	46.3	14539	0	8.1	1.9	5.2	4.9	4.3	3.5	3.4
251	31082	43.3	ND	0	22.8	2.0	ND	ND	ND	ND	ND
252	41082	42.4	11231	0	22.1	2.1	4.8	4.5	3.9	3.2	3.1
253	51082	46.3	18955	0	8.5	1.9	6.2	5.8	5.1	4.2	4.1
254	61082	59.3	19316	0	22.6	1.7	5.1	4.8	4.2	3.6	3.5
255	71082	50.5	14655	0	22.5	1.8	4.9	4.5	4.0	3.4	3.3
256	81082	42.0	13386	0	19.1	2.1	5.4	5.0	4.4	3.6	3.5
257	91082	29.0	18663	0	14.4	2.7	9.2	8.5	7.2	5.8	5.6
258	101082	29.0	19017	0	10.5	2.7	9.3	8.6	7.3	5.9	5.6
259	111082	105.0	13864	0	21.1	1.2	2.6	2.5	2.3	2.0	2.0
260	131082	53.0	18768	0	19.5	1.8	5.5	5.1	4.5	3.8	3.7
261	141082	45.0	18559	0	19.6	1.9	6.1	5.6	4.8	3.9	3.7
262	151082	40.6	ND	0	14.8	2.1	ND	ND	ND	ND	ND
263	161082	40.0	18794	0	22.9	2.1	6.8	6.3	5.3	4.2	4.1
264	171082	50.5	ND	0	22.7	1.8	ND	ND	ND	ND	ND
265	181082	39.7	ND	777	12.1	2.1	ND	ND	ND	ND	ND
266	191082	35.8	ND	1490	18.7	2.2	ND	ND	ND	ND	ND
267	201082	33.5	21119	1551	23.4	2.3	8.6	8.3	7.0	5.5	5.2
268	211082	36.5	16252	1576	23.7	2.2	6.7	6.5	5.5	4.3	4.2
269	221082	35.4	11697	1574	23.8	2.3	5.6	5.5	4.7	3.7	3.5
270	231082	41.5	14244	0	17.9	2.0	5.5	5.1	4.3	3.4	3.3
271	241082	70.7	14098	0	21.6	1.5	3.5	3.3	2.9	2.4	2.4
272	251082	106.6	18318	0	16.4	1.2	3.0	2.8	2.6	2.2	2.1
273	261082	136.8	ND	2205	13.6	1.1	ND	ND	ND	ND	ND
274	271082	109.5	20496	2797	19.3	1.2	3.2	3.2	2.9	2.5	2.4
275	281082	117.0	ND	2992	25.5	1.1	ND	ND	ND	ND	ND

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
276	291082	89.0	10403	2786	6.3	1.3	2.5	2.7	2.4	2.0	2.0
277	301082	33.0	11420	0	24.3	2.4	5.8	5.3	4.5	3.5	3.3
278	311082	100.0	19009	0	18.6	1.2	3.2	3.1	2.7	2.3	2.3
279	11182	78.3	19192	2758	12.2	1.4	4.0	4.0	3.6	3.0	2.9
280	21182	185.0	17055	2654	8.6	1.0	1.9	2.0	1.8	1.6	1.6
281	31182	252.0	14333	2776	26.6	0.9	1.5	1.5	1.4	1.2	1.2
282	41182	128.0	9735	2914	12.5	1.1	1.9	2.0	1.7	1.5	1.4
283	51182	89.0	13255	3019	24.4	1.3	2.8	2.9	2.5	2.1	2.0
284	61182	74.0	15648	0	21.0	1.4	3.6	3.3	2.8	2.2	2.2
285	71182	61.0	ND	0	22.5	1.6	ND	ND	ND	ND	ND
286	81182	58.0	16480	3999	27.4	1.6	4.4	4.7	4.0	3.1	3.0
287	71182	48.0	16363	2969	23.7	1.8	5.2	5.2	4.4	3.3	3.2
288	101182	56.0	22023	5990	25.0	1.6	5.6	6.1	5.1	4.0	3.8
289	111182	48.0	ND	5866	28.6	1.8	ND	ND	ND	ND	ND
290	121182	43.0	ND	5908	9.9	1.9	ND	ND	ND	ND	ND
291	131182	82.0	ND	0	26.7	1.3	ND	ND	ND	ND	ND
292	141182	82.0	ND	0	20.2	1.3	ND	ND	ND	ND	ND
293	151182	59.0	ND	6068	28.6	1.6	ND	ND	ND	ND	ND
294	161182	50.0	ND	5664	22.0	1.8	ND	ND	ND	ND	ND
295	171182	42.0	25403	5642	24.6	2.0	7.9	8.3	6.8	5.2	4.9
296	181182	37.0	ND	7359	23.2	2.1	ND	ND	ND	ND	ND
297	191182	34.0	35397	6651	25.2	2.2	12.3	12.7	10.3	7.6	7.2
298	201182	43.0	ND	0	22.6	1.9	ND	ND	ND	ND	ND
299	211182	34.0	ND	0	26.8	2.2	ND	ND	ND	ND	ND
300	221182	33.0	ND	4660	30.2	2.3	ND	ND	ND	ND	ND
301	231182	33.0	25581	6880	12.7	2.3	9.7	10.6	8.5	6.3	5.9
302	241182	31.0	26015	3292	27.4	2.4	10.4	10.2	8.2	6.0	5.7
303	251182	44.0	24635	9050	23.0	1.9	7.4	8.5	7.1	5.4	5.1
304	261182	33.0	28133	8470	21.3	2.3	10.5	11.7	9.4	6.9	6.6
305	271182	78.0	ND	0	10.4	1.4	ND	ND	ND	ND	ND
306	281182	149.0	21147	0	7.7	1.0	2.5	2.3	2.0	1.6	1.5
307	291182	208.0	26335	8667	21.1	0.9	2.2	2.5	2.2	1.8	1.7
308	301182	118.0	23417	8838	19.9	1.1	3.1	3.5	3.0	2.4	2.3
309	11282	79.0	20577	8116	28.0	1.3	3.9	4.5	3.7	2.8	2.7
310	21282	60.0	16755	11306	24.9	1.5	4.2	5.5	4.4	3.3	3.1
311	31282	50.0	18709	11320	8.5	1.7	5.3	6.7	5.3	3.9	3.7
312	41282	76.0	18340	0	31.7	1.4	3.7	3.4	2.8	2.1	2.0
313	51282	55.0	30204	0	18.2	1.6	6.9	6.1	5.0	3.6	3.4
314	61282	51.0	22968	10592	25.8	1.7	6.0	7.2	5.7	4.2	3.9
315	71282	45.0	ND	9970	30.6	1.8	ND	ND	ND	ND	ND
316	81282	39.0	19753	11535	31.8	2.0	6.7	8.5	6.6	4.6	4.4
317	91282	35.0	17746	11458	25.1	2.1	6.8	8.8	6.8	4.7	4.4
318	101282	31.0	33381	11074	15.4	2.3	12.1	13.6	10.4	7.0	6.6
319	111282	246.0	35098	0	8.2	0.9	2.3	2.2	2.0	1.7	1.6
320	121282	613.0	29483	0	27.9	0.7	1.2	1.2	1.1	1.0	0.9
321	131282	247.0	20601	10620	27.4	0.9	1.7	2.0	1.8	1.5	1.5
322	141282	148.0	14400	10926	31.4	1.0	2.0	2.5	2.2	1.8	1.7
323	151282	88.0	25767	9408	25.0	1.3	4.2	4.8	4.0	3.1	2.9
324	161282	66.0	17520	12132	26.4	1.5	4.0	5.3	4.3	3.2	3.1
325	171282	54.0	ND	11063	32.9	1.6	ND	ND	ND	ND	ND
326	181282	55.0	ND	0	25.5	1.6	ND	ND	ND	ND	ND
327	191282	68.0	12210	0	28.0	1.4	3.2	2.9	2.4	1.8	1.7
328	201282	73.0	16583	9976	6.8	1.4	3.6	4.5	3.7	2.8	2.7
329	211282	227.0	ND	12006	9.8	0.9	ND	ND	ND	ND	ND
330	221282	332.0	ND	10350	33.3	0.8	ND	ND	ND	ND	ND
331	231282	109.0	21354	4062	21.7	1.1	3.1	3.1	2.6	2.0	1.9
332	241282	79.0	ND	0	18.9	1.3	ND	ND	ND	ND	ND
333	251282	ND	ND	0	22.9	ND	ND	ND	ND	ND	ND
334	261282	ND	ND	0	12.3	ND	ND	ND	ND	ND	ND
335	271282	256.0	17482	0	30.0	0.8	1.5	1.4	1.3	1.1	1.0
336	281282	151.0	25874	0	30.7	1.0	2.7	2.5	2.1	1.7	1.6
337	291282	111.0	ND	7515	20.8	1.1	ND	ND	ND	ND	ND
338	301282	97.0	ND	4490	17.8	1.2	ND	ND	ND	ND	ND
339	311282	93.0	16780	0	24.6	1.2	3.0	2.7	2.2	1.7	1.6
340	10183	84.0	20252	0	30.8	1.3	3.6	3.2	2.7	2.0	1.9
341	20183	ND	21619	0	32.9	4.0	ND	ND	ND	ND	ND
342	30183	62.0	17421	0	30.9	1.5	4.1	3.7	2.9	2.1	2.0
343	40183	51.0	16754	0	23.3	1.6	4.7	4.1	3.2	2.3	2.1
344	50183	42.0	20738	9988	28.5	1.8	6.4	7.6	5.8	4.0	3.7
345	60183	52.0	14752	11672	19.2	1.6	4.3	5.7	4.5	3.1	3.0

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
346	70183	42.0	20070	11144	23.5	1.8	6.2	7.7	5.9	4.0	3.8
347	80183	48.0	18228	0	16.9	1.7	5.2	4.6	3.6	2.5	2.3
348	90183	42.0	14743	0	27.9	1.8	5.1	4.4	3.4	2.3	2.1
349	100183	39.0	15012	11220	14.9	1.9	5.4	7.2	5.4	3.7	3.4
350	110183	35.0	ND	12402	26.4	2.0	ND	ND	ND	ND	ND
351	120183	32.0	15378	10915	24.4	2.1	6.4	8.4	6.2	4.1	3.8
352	130183	30.0	21183	7565	14.4	2.2	8.5	9.4	6.9	4.5	4.1
353	140183	64.0	13080	0	32.2	1.5	3.4	3.0	2.4	1.7	1.6
354	150183	50.0	16657	0	28.1	1.7	4.8	4.2	3.3	2.3	2.1
355	160183	34.0	16754	0	25.0	2.1	6.5	5.6	4.2	2.7	2.6
356	170183	48.0	14946	11247	28.2	1.7	4.6	6.1	4.7	3.3	3.1
357	180183	38.0	14344	10760	29.0	1.9	5.4	7.1	5.4	3.6	3.4
358	190183	35.0	13318	10439	26.5	2.0	5.5	7.3	5.4	3.6	3.4
359	200183	75.0	21533	7113	24.7	1.3	4.1	4.5	3.7	2.7	2.6
360	210183	55.0	11135	9769	32.4	1.6	3.5	4.6	3.6	2.6	2.4
361	220183	43.0	9724	0	17.4	1.8	3.9	3.4	2.6	1.8	1.7
362	230183	47.0	11291	0	26.2	1.7	3.9	3.4	2.7	1.9	1.7
363	240183	37.0	11548	0	26.7	2.0	4.8	4.1	3.1	2.1	1.9
364	250183	38.0	15519	7831	26.1	1.9	5.7	6.7	5.0	3.4	3.2
365	260183	54.0	14647	10528	20.4	1.6	4.1	5.4	4.2	3.0	2.8
366	270183	58.0	13274	11472	29.8	1.5	3.7	5.0	4.0	2.8	2.7
367	280183	40.0	14675	10632	23.1	1.9	5.2	6.8	5.2	3.5	3.3
368	290183	49.0	13172	0	17.6	1.7	4.2	3.7	2.8	2.0	1.9
369	300183	44.0	11380	0	23.4	1.8	4.2	3.6	2.8	1.9	1.8
370	310183	31.0	11355	9656	28.1	2.2	5.5	7.3	5.4	3.5	3.2
371	10283	29.0	14484	9899	29.8	2.3	6.7	8.6	6.3	4.0	3.7
372	20283	25.0	8177	11127	29.4	2.5	5.3	8.2	5.9	3.7	3.4
373	30283	24.0	15976	11623	14.2	2.6	8.3	11.0	7.9	4.9	4.5
374	40283	22.5	9919	10283	20.9	2.7	6.5	9.2	6.5	4.0	3.7
375	50283	23.1	15248	0	24.2	2.6	8.3	7.0	5.0	3.0	2.8
376	60283	22.5	11492	0	19.4	2.7	7.1	5.9	4.2	2.6	2.3
377	70283	24.5	13379	11209	8.4	2.5	7.3	9.9	7.1	4.4	4.1
378	80283	32.2	9551	12006	7.8	2.1	4.8	7.3	5.4	3.5	3.3
379	90283	145.0	11930	11409	20.7	1.0	1.8	2.4	2.0	1.6	1.6
380	100283	77.0	9412	10657	15.3	1.3	2.5	3.5	2.8	2.1	2.0
381	110283	47.0	13700	8764	15.4	1.7	4.4	5.5	4.3	2.9	2.8
382	120283	38.0	13700	0	16.6	1.9	5.2	4.5	3.4	2.3	2.1
383	130283	28.4	12684	0	7.4	2.3	6.3	5.3	3.9	2.5	2.3
384	140283	29.0	16770	11055	17.4	2.3	7.4	9.5	7.0	4.5	4.1
385	150283	55.0	15059	11963	27.0	1.6	4.1	5.6	4.4	3.1	2.9
386	160283	37.0	7687	10196	27.7	2.0	3.9	5.7	4.3	2.9	2.7
387	170283	30.0	19970	11113	26.3	2.2	8.2	10.1	7.4	4.8	4.4
388	180283	26.0	13420	9235	23.4	2.4	7.0	8.9	6.4	4.0	3.7
389	190283	23.5	13989	0	21.0	2.6	7.7	6.5	4.6	2.9	2.6
390	200283	18.7	21547	0	21.0	3.0	12.7	10.5	7.2	4.3	3.9
391	210283	20.0	12419	11071	9.8	2.9	8.1	11.3	7.9	4.7	4.3
392	220283	19.4	13051	6913	25.9	2.9	8.6	10.0	7.0	4.1	3.8
393	230283	19.4	5298	11775	24.7	2.9	5.2	9.3	6.5	3.8	3.5
394	240283	18.3	8877	8859	24.6	3.0	7.1	9.8	6.8	4.0	3.6
395	250283	18.3	6994	10689	24.5	3.0	6.2	9.9	6.8	4.0	3.6
396	260283	15.1	6803	0	14.3	3.4	7.1	5.8	3.9	2.2	2.0
397	270283	11.0	7405	0	24.1	4.2	9.5	7.6	4.7	2.5	2.2
398	280283	16.5	9697	8619	18.5	3.2	8.1	10.9	7.4	4.2	3.8
399	10383	15.6	12305	9808	14.5	3.4	9.9	13.1	8.8	5.0	4.5
400	20383	16.3	9169	8782	24.0	3.3	7.9	10.8	7.3	4.2	3.8
401	30383	22.0	6672	0	16.8	2.9	5.7	4.9	3.7	2.5	2.3
402	40383	20.0	6340	0	20.3	3.1	5.9	5.1	3.8	2.5	2.3
403	50383	25.5	6192	0	23.7	2.6	4.9	4.2	3.3	2.2	2.1
404	60383	23.5	4483	0	23.3	2.8	4.5	3.9	3.0	2.0	1.9
405	70383	19.2	5386	0	9.7	3.2	5.7	4.9	3.6	2.4	2.2
406	80383	17.7	5386	0	10.1	3.4	6.1	5.2	3.8	2.5	2.3
407	90383	25.1	7640	9165	6.9	2.7	5.5	7.9	6.1	4.1	3.9
408	100383	34.3	6549	12041	17.2	2.2	4.0	6.6	5.2	3.7	3.5
409	110383	48.5	2705	11773	13.6	1.8	2.3	4.2	3.5	2.6	2.4
410	120383	40.5	4267	0	17.6	2.0	3.0	2.6	2.1	1.5	1.5
411	130383	29.5	6404	0	12.8	2.4	4.4	3.9	3.0	2.1	2.0
412	140383	23.7	3576	0	16.3	2.8	4.1	3.6	2.7	1.9	1.7
413	150383	33.9	4835	12347	20.3	2.2	3.5	6.3	5.0	3.6	3.4
414	160383	36.4	8233	11508	5.9	2.1	4.2	6.5	5.2	3.7	3.5
415	170383	149.0	16367	10237	21.9	1.0	2.2	2.6	2.3	1.9	1.9

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
416	180383	52.6	15588	0	21.6	1.7	4.6	4.1	3.4	2.5	2.4
417	190383	ND	7657	0	21.0	ND	ND	ND	ND	ND	ND
418	200383	ND	8912	0	15.7	ND	ND	ND	ND	ND	ND
419	210383	22.0	6962	0	16.8	2.9	5.8	5.0	3.8	2.5	2.4
420	220383	19.6	5347	0	20.0	3.1	5.6	4.8	3.6	2.3	2.2
421	230383	18.0	5583	0	9.8	3.5	6.5	5.7	4.4	3.1	2.9
422	240383	23.5	7828	0	9.7	2.9	6.1	5.5	4.4	3.2	3.0
423	250383	90.0	10638	0	10.0	1.3	2.5	2.3	2.1	1.7	1.7
424	260383	103.0	7476	0	11.7	1.2	2.0	1.8	1.6	1.4	1.3
425	270383	51.0	7146	0	14.8	1.8	3.2	2.9	2.5	2.0	1.9
426	280383	38.5	6420	0	15.1	2.1	3.8	3.4	2.8	2.2	2.1
427	290383	44.4	8646	0	17.6	1.9	3.9	3.5	3.0	2.3	2.2
428	300383	84.2	8962	0	15.0	1.3	2.4	2.3	2.0	1.6	1.6
429	310383	42.0	8169	0	14.0	2.0	3.9	3.6	3.0	2.3	2.2
430	10483	31.5	4169	0	12.0	2.4	3.7	3.3	2.7	2.1	2.0
431	20483	26.5	ND	0	14.3	2.7	ND	ND	ND	ND	ND
432	30483	43.0	10694	0	13.3	2.0	4.4	4.1	3.4	2.7	2.5
433	40483	48.0	7600	0	10.0	1.8	3.4	3.1	2.7	2.1	2.0
434	50483	32.3	8994	0	17.0	2.4	5.1	4.6	3.8	2.9	2.7
435	60483	32.5	6721	0	12.0	2.3	4.4	4.0	3.3	2.5	2.3
436	70483	29.4	8138	0	4.9	2.5	5.2	4.7	3.8	2.9	2.7
437	80483	26.0	5647	8993	11.1	2.7	4.8	7.4	6.0	4.4	4.2
438	90483	48.1	ND	0	16.4	1.8	ND	ND	ND	ND	ND
439	100483	31.6	ND	0	13.5	2.4	ND	ND	ND	ND	ND
440	110483	26.4	8189	0	16.4	2.7	5.7	5.1	4.1	3.1	2.9
441	120483	25.4	6423	9112	16.2	2.8	5.3	8.0	6.6	5.0	4.8
442	130483	21.5	4411	9342	13.6	3.2	5.2	8.5	7.0	5.2	4.9
443	140483	21.0	6315	9285	10.6	3.3	6.2	9.5	7.7	5.7	5.4
444	150483	19.3	10151	0	3.5	3.5	8.5	7.6	6.1	4.5	4.3
445	160483	38.5	8524	0	9.2	2.1	4.4	4.0	3.4	2.7	2.6
446	170483	45.7	7269	0	15.2	1.9	3.5	3.3	2.8	2.3	2.2
447	180483	33.4	7952	9095	15.8	2.4	4.7	6.8	5.7	4.5	4.3
448	190483	27.8	6674	9354	4.4	2.7	5.0	7.6	6.3	4.8	4.6
449	200483	27.0	11974	9274	7.9	2.7	7.1	9.5	7.9	6.0	5.8
450	210483	42.2	11100	9311	12.2	2.0	4.7	6.3	5.4	4.3	4.2
451	220483	73.7	10121	0	7.7	1.5	2.9	2.7	2.4	2.1	2.0
452	230483	88.8	12832	0	11.9	1.3	2.8	2.7	2.4	2.1	2.0
453	240483	116.5	13369	0	8.0	1.2	2.4	2.3	2.1	1.8	1.8
454	250483	71.0	7514	0	5.9	1.5	2.6	2.4	2.2	1.9	1.8
455	260483	53.0	7636	9481	6.6	1.8	3.3	4.7	4.1	3.4	3.3
456	270483	48.0	ND	5720	9.1	1.9	ND	ND	ND	ND	ND
457	280483	158.0	9143	9212	8.5	1.0	1.6	2.1	2.0	1.7	1.7
458	290483	88.0	8056	0	7.9	1.3	2.3	2.2	2.0	1.7	1.6
459	300483	69.0	9930	0	5.6	1.5	3.0	2.8	2.5	2.1	2.1
460	10583	67.0	ND	0	12.5	1.5	ND	ND	ND	ND	ND
461	20583	55.0	3377	0	4.7	1.8	2.4	2.3	2.0	1.8	1.8
462	30583	44.0	ND	9318	11.1	2.0	ND	ND	ND	ND	ND
463	40583	40.0	ND	8258	6.8	2.1	ND	ND	ND	ND	ND
464	50583	34.0	ND	9065	5.4	2.4	ND	ND	ND	ND	ND
465	60583	96.0	ND	0	8.5	1.3	ND	ND	ND	ND	ND
466	70583	143.0	6012	0	8.2	1.1	1.5	1.5	1.3	1.3	1.2
467	80583	154.0	6017	0	11.5	1.0	1.5	1.4	1.3	1.2	1.2
468	90583	120.0	9011	0	8.1	1.1	2.0	1.9	1.7	1.6	1.6
469	100583	85.0	ND	9236	5.0	1.4	ND	ND	ND	ND	ND
470	110583	159.0	ND	8996	6.4	1.0	ND	ND	ND	ND	ND
471	120583	105.0	9321	8815	4.4	1.2	2.2	2.9	2.7	2.5	2.5
472	130583	186.0	ND	0	11.9	1.0	ND	ND	ND	ND	ND
473	140583	105.0	ND	0	11.1	1.2	ND	ND	ND	ND	ND
474	150583	93.0	ND	0	9.2	1.3	ND	ND	ND	ND	ND
475	160583	85.0	10486	0	3.9	1.4	2.7	2.6	2.4	2.2	2.2
476	170583	72.0	12316	9065	3.0	1.5	3.3	4.4	4.0	3.7	3.6
477	180583	85.0	12720	8982	7.6	1.4	3.0	3.9	3.5	3.3	3.2
478	190583	157.0	4619	7618	5.9	1.0	1.4	1.8	1.7	1.6	1.5
479	200583	119.0	8615	0	1.3	1.2	1.9	1.9	1.7	1.6	1.6
480	210583	610.0	8615	0	5.8	0.7	0.9	0.9	0.8	0.8	0.8
481	220583	400.0	8615	0	2.7	0.8	1.0	1.0	0.9	0.9	0.9
482	230583	585.0	3517	0	10.0	0.7	0.8	0.8	0.7	0.7	0.7
483	240583	353.0	16388	9245	10.4	0.8	1.3	1.5	1.5	1.4	1.4
484	250583	214.0	ND	9241	0.9	0.9	ND	ND	ND	ND	ND
485	260583	152.0	3775	9124	4.7	1.0	1.3	1.8	1.7	1.6	1.6

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
486	270583	134.0	3651	0	8.2	1.1	1.4	1.3	1.3	1.2	1.2
487	280583	134.0	3651	0	7.9	1.1	1.4	1.3	1.3	1.2	1.2
488	290583	129.0	3651	0	3.6	1.1	1.4	1.4	1.3	1.2	1.2
489	300583	104.0	1063	0	4.8	1.2	1.3	1.3	1.2	1.1	1.1
490	310583	107.0	ND	9014	6.8	1.2	ND	ND	ND	ND	ND
491	10683	191.0	2480	9290	7.3	1.0	1.1	1.5	1.4	1.4	1.4
492	20683	204.0	2125	7880	4.2	0.9	1.0	1.4	1.3	1.2	1.2
493	30683	306.0	300	0	9.9	0.8	0.8	0.8	0.8	0.7	0.7
494	40683	219.0	300	0	6.5	0.9	0.9	0.9	0.8	0.8	0.8
495	50683	158.0	300	0	9.9	1.0	1.0	1.0	0.9	0.9	0.9
496	60683	114.0	300	0	10.7	1.2	1.2	1.2	1.1	1.0	1.0
497	70683	96.0	300	0	6.5	1.3	1.3	1.3	1.2	1.1	1.1
498	80683	111.0	300	9247	6.1	1.2	1.2	2.0	1.8	1.7	1.7
499	90683	160.0	300	9030	5.6	1.0	1.0	1.5	1.4	1.4	1.4
500	100683	270.0	300	9088	5.0	0.9	0.9	1.2	1.1	1.1	1.0
501	110683	237.0	300	0	4.4	0.9	0.9	0.9	0.8	0.8	0.8
502	120683	191.0	286	0	9.7	1.0	1.0	0.9	0.9	0.8	0.8
503	130683	141.0	286	8841	11.4	1.1	1.1	1.7	1.5	1.5	1.4
504	140683	110.0	286	8576	10.1	1.2	1.2	1.9	1.8	1.7	1.6
505	150683	93.0	286	8922	9.1	1.3	1.3	2.2	2.0	1.9	1.9
506	160683	83.0	286	7999	11.0	1.4	1.4	2.3	2.1	2.0	1.9
507	170683	90.0	286	0	11.0	1.3	1.4	1.3	1.2	1.1	1.1
508	180683	92.0	286	0	8.9	1.3	1.4	1.3	1.2	1.1	1.1
509	190683	92.0	286	0	6.8	1.3	1.4	1.3	1.2	1.1	1.1
510	200683	80.0	286	7423	5.6	1.4	1.5	2.3	2.1	1.9	1.9
511	210683	81.0	286	8323	7.9	1.4	1.4	2.4	2.2	2.0	2.0
512	220683	72.0	286	8609	5.0	1.5	1.5	2.6	2.4	2.2	2.2
513	230683	70.0	286	8818	3.9	1.5	1.6	2.7	2.5	2.3	2.2
514	240683	133.0	200	0	6.0	1.1	1.1	1.1	1.0	0.9	0.9
515	250683	124.0	176	0	8.9	1.1	1.2	1.1	1.0	1.0	1.0
516	260683	95.0	176	0	7.4	1.3	1.3	1.3	1.2	1.1	1.1
517	270683	84.0	176	0	3.4	1.4	1.5	1.5	1.2	1.2	1.1
518	280683	387.0	211	6315	4.3	0.8	0.8	0.9	0.9	0.9	0.8
519	290683	353.0	212	5920	5.5	0.8	0.8	0.9	0.9	0.9	0.9
520	300683	197.0	545	4179	2.4	0.9	1.0	1.1	1.1	1.0	1.0
521	10683	232.0	545	0	3.1	0.9	0.9	0.9	0.8	0.8	0.8
522	70783	146.0	545	0	6.3	1.1	1.1	1.1	1.0	0.9	0.9
523	30783	98.0	540	0	6.7	1.3	1.3	1.3	1.2	1.1	1.1
524	40783	98.0	540	0	6.7	1.3	1.3	1.3	1.2	1.1	1.1
525	50783	88.0	540	9187	8.0	1.3	1.4	2.3	2.2	2.0	2.0
526	60783	80.0	540	8673	2.7	1.4	1.5	2.5	2.2	2.1	2.1
527	70783	72.0	425	6081	3.8	1.5	1.6	2.3	2.1	2.0	1.9
528	80783	66.0	425	0	1.2	1.6	1.7	1.6	1.4	1.3	1.3
529	70783	70.0	425	0	1.2	1.5	1.6	1.5	1.4	1.3	1.3
530	100783	69.0	10	0	4.3	1.5	1.5	1.5	1.3	1.2	1.2
531	110783	97.0	10	3017	5.5	1.3	1.3	1.5	1.4	1.3	1.3
532	120783	86.0	10	1581	6.5	1.4	1.4	1.5	1.4	1.3	1.2
533	130783	77.0	200	3174	4.7	1.4	1.5	1.8	1.6	1.5	1.5
534	140783	112.0	200	3150	8.5	1.2	1.2	1.4	1.3	1.3	1.2
535	150783	90.0	200	3121	3.7	1.3	1.4	1.6	1.5	1.4	1.4
536	160783	87.0	200	0	4.2	1.4	1.4	1.3	1.2	1.1	1.1
537	170783	147.0	400	0	5.5	1.1	1.1	1.0	1.0	0.9	0.9
538	180783	168.0	400	3039	3.2	1.0	1.0	1.2	1.1	1.0	1.0
539	190783	362.0	400	2943	6.2	0.8	0.8	0.9	0.8	0.8	0.8
540	200783	210.0	300	3100	8.9	0.9	0.9	1.1	1.0	0.9	0.9
541	210783	144.0	300	3163	8.6	1.1	1.1	1.3	1.2	1.2	1.2
542	220783	113.0	300	1498	4.8	1.2	1.2	1.3	1.3	1.2	1.2
543	230783	100.0	300	0	6.5	1.3	1.3	1.3	1.2	1.1	1.1
544	240783	95.0	200	0	7.3	1.3	1.3	1.3	1.2	1.2	1.2
545	250783	81.0	200	957	9.5	1.4	1.5	1.5	1.4	1.4	1.3
546	260783	79.0	200	1695	10.1	1.5	1.5	1.6	1.5	1.5	1.4
547	270783	70.0	183	1671	9.9	1.6	1.6	1.8	1.6	1.6	1.5
548	280783	64.0	183	1664	7.6	1.6	1.7	1.9	1.7	1.7	1.6
549	290783	60.0	183	919	9.9	1.7	1.7	1.8	1.7	1.6	1.6
550	300783	55.0	394	0	6.6	1.8	1.9	1.8	1.7	1.6	1.6
551	310783	51.0	394	0	7.3	1.9	2.0	1.9	1.8	1.7	1.6
552	10883	50.0	394	0	7.2	1.9	2.0	1.9	1.8	1.7	1.7
553	20883	59.0	1081	0	7.1	1.7	1.9	1.9	1.7	1.6	1.6
554	30883	51.0	1081	0	4.1	1.9	2.1	2.0	1.9	1.8	1.8
555	40883	170.0	841	0	4.3	1.0	1.1	1.0	1.0	1.0	0.9

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
556	50883	105.0	841	0	5.2	1.3	1.3	1.3	1.2	1.2	1.2
557	60883	94.0	7106	0	5.1	1.3	2.2	2.1	2.0	1.9	1.8
558	70883	77.0	4929	0	8.5	1.5	2.2	2.1	2.0	1.9	1.8
559	80883	58.0	5813	0	4.9	1.8	2.8	2.7	2.5	2.4	2.4
560	90883	61.0	1700	0	4.9	1.7	2.0	1.9	1.8	1.7	1.7
561	100883	99.0	6133	0	4.9	1.3	2.0	1.9	1.8	1.7	1.7
562	110883	79.0	3866	0	5.3	1.5	2.0	1.9	1.8	1.7	1.7
563	120883	59.0	8310	0	8.8	1.7	3.3	3.1	2.9	2.8	2.7
564	130883	52.0	6240	0	10.6	1.9	3.2	3.1	2.8	2.7	2.6
565	140883	50.0	4420	0	5.3	1.9	2.9	2.8	2.6	2.4	2.4
566	150883	64.0	6804	0	6.6	1.6	2.8	2.7	2.5	2.4	2.3
567	160883	62.0	7051	0	8.7	1.7	2.9	2.8	2.6	2.5	2.4
568	170883	54.0	9656	0	8.2	1.8	3.8	3.6	3.4	3.2	3.1
569	180883	49.0	10002	0	12.7	2.0	4.2	4.0	3.7	3.5	3.4
570	190883	45.0	3045	0	8.9	2.1	2.8	2.7	2.5	2.3	2.3
571	200883	41.0	4742	0	10.2	2.2	3.5	3.3	3.0	2.8	2.8
572	210883	51.0	11889	0	4.2	1.9	4.4	4.3	3.9	3.7	3.6
573	220883	142.0	15599	0	4.2	1.1	2.3	2.2	2.1	2.0	2.0
574	230883	120.0	10863	0	9.9	1.2	2.2	2.1	2.0	1.9	1.9
575	240883	87.0	7206	0	8.4	1.4	2.3	2.2	2.1	2.0	2.0
576	250883	106.0	7196	0	6.2	1.2	2.0	1.9	1.8	1.7	1.7
577	260883	85.0	7373	0	13.9	1.4	2.3	2.3	2.1	2.0	2.0
578	270883	72.0	11312	0	10.7	1.5	3.3	3.1	2.9	2.8	2.7
579	280883	66.0	7082	0	12.0	1.6	2.8	2.7	2.5	2.4	2.3
580	290883	60.0	8631	0	12.0	1.7	3.3	3.2	2.9	2.8	2.7
581	300883	55.0	8992	0	9.6	1.8	3.6	3.5	3.2	3.0	3.0
582	310885	55.0	6896	0	11.0	1.8	3.1	2.9	2.6	2.4	2.4
583	10983	70.0	ND	0	6.9	1.5	ND	ND	ND	ND	ND
584	20983	49.0	9117	0	7.9	1.9	3.9	3.7	3.3	3.0	2.9
585	30983	45.0	5164	0	12.0	2.0	3.2	3.0	2.7	2.5	2.4
586	40983	62.0	25148	0	11.3	1.6	6.0	5.7	5.1	4.8	4.6
587	50983	48.0	11469	0	15.1	1.9	4.4	4.2	3.8	3.5	3.4
588	60983	41.0	ND	0	7.6	2.1	ND	ND	ND	ND	ND
589	70983	87.0	ND	0	4.6	1.4	ND	ND	ND	ND	ND
590	80983	57.0	10316	0	13.7	1.7	3.6	3.5	3.1	2.9	2.8
591	90983	100.0	ND	0	11.1	1.3	ND	ND	ND	ND	ND
592	100983	142.0	14938	0	11.6	1.1	2.2	2.1	2.0	1.9	1.9
593	110983	462.0	14332	0	8.7	0.8	1.1	1.1	1.0	1.0	1.0
594	120983	357.0	16990	0	10.6	0.8	1.3	1.3	1.2	1.2	1.2
595	130983	219.0	13165	0	15.0	0.9	1.6	1.5	1.4	1.4	1.4
596	140983	285.0	13330	0	12.7	0.8	1.4	1.3	1.3	1.2	1.2
597	150983	257.0	ND	0	13.9	0.9	ND	ND	ND	ND	ND
598	160983	ND	24331	0	7.8	ND	ND	ND	ND	ND	ND
599	170983	201.0	11223	0	17.2	0.9	1.6	1.5	1.4	1.3	1.3
600	180983	144.0	14146	0	15.5	1.1	2.1	2.1	1.9	1.8	1.8
601	190983	110.0	14287	0	13.3	1.2	2.6	2.4	2.2	2.0	1.9
602	200983	172.0	12874	0	10.2	1.0	1.8	1.7	1.6	1.5	1.4
603	210983	123.0	24053	0	12.6	1.1	3.2	3.0	2.7	2.5	2.5
604	220983	116.0	27897	0	7.3	1.2	3.7	3.5	3.2	2.9	2.8
605	230983	118.0	12228	0	15.9	1.1	2.2	2.1	1.9	1.8	1.7
606	240983	89.0	16758	0	13.3	1.3	3.3	3.1	2.7	2.5	2.4
607	250983	106.0	28251	0	8.8	1.2	4.0	3.8	3.4	3.1	3.0
608	260983	140.0	12773	0	13.2	1.1	2.0	1.9	1.8	1.6	1.6
609	270983	179.0	13082	0	11.3	1.0	1.8	1.7	1.5	1.4	1.4
610	280983	134.0	ND	0	11.5	1.1	ND	ND	ND	ND	ND
611	290983	97.0	ND	0	14.0	1.3	ND	ND	ND	ND	ND
612	300983	79.0	ND	0	14.9	1.4	ND	ND	ND	ND	ND
613	11083	74.0	12867	0	4.2	1.4	3.2	3.0	2.7	2.4	2.4
614	21083	237.0	13134	0	6.1	0.9	1.5	1.4	1.3	1.2	1.2
615	31083	120.0	ND	0	11.6	1.1	ND	ND	ND	ND	ND
616	41083	156.0	15498	0	10.5	1.0	2.1	2.0	1.8	1.7	1.6
617	51083	530.0	30362	0	7.5	0.7	1.4	1.3	1.2	1.2	1.2
618	61083	352.0	21323	0	12.4	0.8	1.5	1.4	1.3	1.2	1.2
619	71083	260.0	17633	0	8.1	0.9	1.6	1.5	1.4	1.3	1.3
620	81083	150.0	19061	0	13.7	1.0	2.4	2.3	2.1	1.9	1.9
621	91083	138.0	19446	0	8.0	1.1	2.5	2.4	2.1	1.9	1.9
622	101083	125.0	17954	0	15.9	1.1	2.6	2.4	2.1	1.9	1.9
623	111083	101.0	12463	0	12.3	1.2	2.5	2.3	2.0	1.8	1.8
624	121083	85.0	13154	0	17.8	1.3	2.9	2.7	2.3	2.1	2.0
625	131083	96.0	20853	0	17.7	1.2	3.5	3.2	2.8	2.5	2.4

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
626	141083	77.0	11442	0	14.5	1.4	2.9	2.7	2.3	2.0	2.0
627	151083	77.0	24110	0	14.2	1.4	4.6	4.2	3.6	3.2	3.1
628	161083	300.0	11819	0	16.6	0.8	1.2	1.2	1.1	1.0	1.0
629	171083	167.0	15175	0	21.0	1.0	1.9	1.8	1.6	1.5	1.5
630	881083	116.0	12976	0	13.2	1.1	2.3	2.1	1.9	1.7	1.7
631	191083	93.0	7700	0	16.2	1.3	2.1	2.0	1.7	1.5	1.5
632	201083	79.0	12680	0	8.7	1.4	3.0	2.8	2.4	2.1	2.0
633	211083	142.0	9985	0	10.1	1.0	1.8	1.7	1.5	1.4	1.3
634	221083	82.0	11872	0	13.3	1.3	2.8	2.6	2.2	2.0	1.9
635	231083	89.0	18572	0	14.3	1.3	3.4	3.2	2.7	2.5	2.4
636	241083	104.0	11627	0	12.7	1.2	2.3	2.2	1.9	1.7	1.7
637	251083	80.0	13628	0	15.9	1.4	3.1	2.9	2.5	2.2	2.1
638	261083	71.0	11431	0	20.3	1.4	3.1	2.8	2.4	2.1	2.1
639	271083	60.0	19966	0	19.0	1.6	4.9	4.5	3.8	3.3	3.2
640	281083	53.0	12436	0	17.5	1.7	4.0	3.7	3.1	2.7	2.6
641	291083	49.0	9636	0	17.8	1.8	3.7	3.4	2.8	2.4	2.3
642	301083	55.0	7890	0	22.2	1.7	3.1	2.8	2.4	2.1	2.0
643	311083	52.0	9875	0	9.4	1.7	3.6	3.3	2.7	2.4	2.3
644	11183	51.0	7433	0	15.1	1.7	3.2	2.9	2.4	2.1	2.0
645	21183	66.0	ND	0	15.7	1.5	ND	ND	ND	ND	ND
646	31183	48.0	ND	ND	ND	ND	ND	ND	ND	ND	ND
647	41183	294.0	33031	0	5.8	0.8	2.0	1.9	1.8	1.7	1.6
648	51183	129.0	22979	0	11.4	1.1	2.9	2.8	2.4	2.2	2.1
649	61183	620.0	17047	0	11.4	0.7	1.0	1.0	0.9	0.9	0.9
650	71183	101.0	ND	ND	ND	ND	ND	ND	ND	ND	ND
651	81183	84.0	11244	0	9.2	1.3	2.7	2.5	2.1	1.9	1.8
652	91183	390.0	30933	0	12.2	0.8	1.6	1.6	1.4	1.4	1.3
653	101183	216.0	18569	0	8.4	0.9	1.8	1.7	1.6	1.4	1.4
654	111183	114.0	15389	0	23.3	1.1	2.5	2.4	2.1	1.9	1.8
655	121183	116.0	9606	0	14.9	1.1	2.0	1.9	1.6	1.5	1.4
656	131183	72.0	9925	0	29.2	1.4	2.8	2.6	2.2	2.0	1.9
657	141183	60.0	11362	0	23.9	1.6	3.5	3.2	2.7	2.4	2.3
658	151183	64.0	11663	0	19.4	1.5	3.4	3.1	2.6	2.3	2.2
659	161183	52.0	5977	0	28.3	1.7	2.9	2.6	2.2	1.9	1.8
660	171183	44.0	3786	0	28.2	1.9	2.7	2.5	2.0	1.8	1.7
661	181183	38.0	4481	0	24.2	2.1	3.2	2.9	2.4	2.0	1.9
662	191183	35.0	4484	0	10.6	2.2	3.4	3.1	2.5	2.1	2.0
663	201183	33.0	4324	0	28.0	2.3	3.5	3.2	2.6	2.2	2.1
664	211183	28.0	4353	0	29.7	2.5	4.0	3.6	2.8	2.4	2.2
665	221183	44.0	16658	0	13.7	1.9	5.6	5.1	4.2	3.6	3.5
666	231183	33.0	18928	0	16.4	2.3	7.8	7.0	5.6	4.8	4.5
667	241183	31.0	18159	0	17.9	2.4	8.0	7.1	5.7	4.8	4.6
668	251183	27.0	7134	0	19.8	2.6	5.1	4.5	3.6	3.0	2.8
669	261183	23.0	6747	0	22.5	2.9	5.6	5.0	3.9	3.2	3.0
670	271183	20.7	7733	0	23.2	3.1	6.6	5.7	4.4	3.6	3.4
671	281183	18.8	4455	3097	24.9	3.3	5.5	6.2	4.8	3.9	3.6
672	291183	18.4	8331	5272	17.8	3.4	7.5	9.0	6.9	5.6	5.2
673	301183	25.8	3106	5102	22.6	2.7	3.8	5.1	4.0	3.4	3.2
674	11283	25.0	7478	6274	23.1	2.7	5.6	7.1	5.6	4.7	4.4
675	21283	23.2	6730	6302	28.0	2.9	5.6	7.3	5.7	4.7	4.4
676	31283	18.9	12371	0	30.6	3.3	9.3	8.1	6.2	5.1	4.7
677	41283	17.9	12532	0	21.2	3.5	9.9	8.5	6.5	5.3	4.9
678	51283	20.2	6135	3422	24.7	3.2	6.0	6.7	5.2	4.2	4.0
679	61283	19.4	3045	6124	31.1	3.3	4.7	6.8	5.2	4.3	4.0
680	71283	18.6	6082	2668	18.6	3.4	6.4	6.8	5.2	4.2	3.9
681	81283	21.6	8157	0	10.2	2.9	6.4	5.5	4.1	3.3	3.1
682	91283	29.1	9706	0	17.4	2.4	5.5	4.8	3.8	3.1	2.9
683	101283	27.0	8005	0	8.7	2.5	5.3	4.6	3.6	2.9	2.7
684	111283	45.3	6652	0	27.5	1.8	3.2	2.9	2.4	2.0	1.9
685	121283	29.0	4671	0	19.1	2.4	3.9	3.4	2.7	2.2	2.1
686	131283	30.7	9822	7957	22.7	2.3	5.3	7.0	5.4	4.5	4.2
687	141283	34.0	7961	8078	19.6	2.2	4.4	6.0	4.7	3.9	3.7
688	151283	36.8	4251	4964	19.0	2.1	3.2	4.0	3.2	2.7	2.5
689	161283	42.3	4482	0	28.5	1.9	2.9	2.6	2.1	1.8	1.7
690	171283	33.0	6255	0	23.3	2.2	4.0	3.5	2.8	2.3	2.2
691	181283	69.0	13077	0	25.9	1.4	3.3	3.0	2.5	2.2	2.1
692	191283	39.0	6431	0	21.6	2.0	3.6	3.2	2.5	2.1	2.0
693	201283	34.0	10034	0	12.2	2.2	5.0	4.4	3.5	2.9	2.7
694	211283	52.0	13651	0	12.7	1.7	4.2	3.8	3.1	2.7	2.6
695	221283	34.0	6226	0	24.6	2.2	3.9	3.5	2.7	2.3	2.2

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
696	231283	33.0	5806	0	9.2	2.2	3.9	3.4	2.7	2.2	2.1
697	241283	28.0	8296	0	23.3	2.5	5.2	4.6	3.5	2.9	2.7
698	251283	24.3	6043	0	29.7	2.7	5.0	4.3	3.3	2.7	2.5
699	261283	22.3	7032	0	23.0	2.9	5.7	5.0	3.8	3.0	2.8
700	271283	21.6	11108	0	13.7	2.9	7.6	6.6	4.9	4.0	3.7
701	281283	50.7	6333	0	27.3	1.7	2.9	2.6	2.0	1.7	1.6
702	291283	33.9	ND	0	29.3	2.1	ND	ND	ND	ND	ND
703	301283	28.3	7370	0	23.0	2.4	4.7	4.1	3.1	2.5	2.3
704	311283	25.0	3130	0	27.2	2.6	3.7	3.2	2.4	1.9	1.7
705	10184	23.3	4025	0	12.1	2.0	3.6	3.0	2.2	1.8	1.6
706	20184	21.8	4509	0	19.0	2.1	3.9	3.3	2.4	1.9	1.8
707	30184	20.3	4165	0	30.5	2.2	4.0	3.4	2.5	1.9	1.8
708	40184	16.8	4741	0	27.9	2.5	4.9	4.1	2.9	2.2	2.1
709	50184	14.9	5844	0	19.0	2.7	6.1	5.0	3.5	2.6	2.4
710	60184	16.8	7382	0	24.0	2.5	6.3	5.2	3.7	2.8	2.6
711	70184	16.6	4882	0	22.0	2.5	5.1	4.2	3.0	2.3	2.1
712	80184	17.3	2952	0	33.0	2.5	3.9	3.3	2.3	1.8	1.7
713	90184	16.1	2975	0	28.5	2.6	4.2	3.5	2.4	1.9	1.7
714	100184	15.4	7863	0	18.7	2.7	7.0	5.8	4.1	3.1	2.8
715	110184	15.1	ND	0	25.3	2.7	ND	ND	ND	ND	ND
716	120184	15.1	4920	0	17.7	2.7	5.5	4.5	3.2	2.4	2.2
717	130184	15.4	4628	0	21.2	2.7	5.2	4.3	3.0	2.3	2.1
718	140184	17.5	7081	0	26.8	2.5	5.9	5.0	3.5	2.7	2.5
719	150184	18.8	3093	0	22.5	2.3	3.8	3.2	2.3	1.8	1.6
720	160184	16.9	1859	6338	19.3	2.5	3.5	6.0	4.2	3.3	3.0
721	170184	16.3	3732	9595	23.5	2.6	4.5	8.6	6.1	4.7	4.3
722	180184	16.4	1176	9005	9.9	2.6	3.2	7.2	5.1	3.9	3.6
723	190184	18.2	4775	9953	10.4	2.4	4.7	8.5	6.1	4.7	4.3
724	200184	30.8	ND	0	19.2	1.7	ND	ND	ND	ND	ND
725	210184	26.3	ND	0	27.2	1.9	ND	ND	ND	ND	ND
726	220184	18.6	ND	0	21.6	2.4	ND	ND	ND	ND	ND
727	230184	15.6	ND	9716	30.2	2.7	ND	ND	ND	ND	ND
728	240184	15.8	ND	10143	27.8	2.6	ND	ND	ND	ND	ND
729	250184	15.8	4881	10845	25.8	2.6	5.3	10.0	7.0	5.4	4.9
730	260184	14.7	3484	11073	25.9	2.8	4.8	10.1	7.0	5.3	4.8
731	270184	14.2	ND	11053	20.9	2.7	ND	ND	ND	ND	ND
732	280184	14.1	6319	0	19.6	2.8	6.2	4.9	3.1	2.2	1.9
733	290184	19.0	3314	0	18.5	2.2	3.7	3.0	2.0	1.4	1.3
734	300184	15.9	3718	11123	21.1	2.5	4.4	9.0	5.8	4.2	3.7
735	310184	14.8	3559	11369	30.1	2.7	4.5	9.6	6.1	4.3	3.9
736	10284	14.2	5301	11730	30.3	2.7	5.6	10.9	6.9	4.8	4.3
737	20284	13.9	3465	10582	27.1	2.8	4.7	9.6	6.1	4.3	3.8
738	30284	13.8	5423	9931	16.4	2.8	5.8	10.2	6.4	4.5	4.0
739	40284	26.4	3433	0	8.8	1.8	2.9	2.4	1.7	1.3	1.2
740	50284	32.8	3522	0	15.9	1.6	2.5	2.1	1.5	1.2	1.1
741	60284	61.0	7767	0	25.7	1.2	2.3	2.0	1.6	1.3	1.2
742	70284	32.5	ND	9032	26.8	1.6	ND	ND	ND	ND	ND
743	80284	23.5	3487	8937	27.7	2.0	3.2	5.7	3.9	3.0	2.7
744	90284	19.4	3356	11925	28.4	2.2	3.6	7.9	5.3	3.9	3.5
745	100284	17.2	4977	11411	23.4	2.4	4.7	9.1	5.9	4.3	3.9
746	110284	19.3	2033	0	19.4	2.2	3.1	2.5	1.7	1.2	1.1
747	120284	91.5	5237	0	7.1	1.0	1.5	1.4	1.1	0.9	0.9
748	130284	113.9	13549	10823	14.1	0.9	2.1	2.7	2.2	1.9	1.8
749	140284	83.0	8405	12452	8.3	1.0	2.0	3.1	2.5	2.1	2.0
750	150284	47.8	ND	11342	9.8	1.3	ND	ND	ND	ND	ND
751	160284	59.1	7290	11953	9.8	1.2	2.3	3.8	2.9	2.4	2.2
752	170284	31.1	6973	9212	21.2	1.6	3.6	5.5	3.9	3.0	2.8
753	180284	37.9	ND	0	12.5	1.5	ND	ND	ND	ND	ND
754	190284	33.1	4008	0	20.6	1.6	2.6	2.2	1.6	1.3	1.2
755	200284	26.5	3029	10239	17.2	1.8	2.8	5.5	3.8	2.9	2.7
756	210284	22.4	2705	10289	25.2	2.0	3.0	6.2	4.2	3.2	2.9
757	220284	20.7	4402	6852	19.2	2.1	3.9	5.8	3.9	2.9	2.6
758	230284	19.6	ND	12539	26.4	2.2	ND	ND	ND	ND	ND
759	240284	18.8	5719	11049	25.6	2.3	4.7	8.5	5.7	4.1	3.7
760	250284	18.0	4804	0	19.3	2.3	4.5	3.6	2.4	1.7	1.6
761	260284	17.7	7578	0	23.2	2.4	6.1	5.1	3.7	2.8	2.6
762	270284	17.2	4184	5952	25.0	2.5	4.6	6.7	4.8	3.7	3.4
763	280284	17.0	3635	11602	22.9	2.5	4.3	9.3	6.6	5.0	4.6
764	290284	16.9	4579	12114	19.9	2.5	4.8	10.0	7.1	5.4	5.0
765	10384	16.8	3557	12142	22.4	2.5	4.3	9.6	6.8	5.2	4.8

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
766	20384	16.6	6668	11345	15.4	2.5	6.0	10.7	7.5	5.8	5.3
767	30384	ND	6693	0	16.2	ND	ND	ND	ND	ND	ND
768	40384	16.7	3041	0	15.9	2.5	4.1	3.4	2.4	1.8	1.7
769	50384	16.8	4820	11297	14.5	2.5	5.0	9.7	6.9	5.3	4.8
770	60384	28.6	9315	12128	12.5	1.8	4.7	7.7	5.8	4.7	4.4
771	70384	66.0	6162	17199	12.6	1.1	2.0	4.2	3.4	2.9	2.8
772	80384	33.4	1496	11421	14.9	1.6	2.0	4.7	3.6	3.0	2.8
773	90384	24.4	3249	11394	17.3	2.0	3.2	6.7	4.9	3.9	3.7
774	100384	23.5	3896	0	17.2	2.0	3.5	3.0	2.2	1.7	1.6
775	110384	25.5	3017	0	13.9	1.9	3.0	2.5	1.9	1.5	1.4
776	120384	22.7	2227	11853	15.1	2.1	2.9	6.9	5.1	4.0	3.7
777	130384	103.0	15472	12138	19.7	1.8	2.7	3.5	3.0	2.6	2.5
778	140384	85.0	ND	4052	18.1	1.3	ND	ND	ND	ND	ND
779	150384	116.0	17086	11791	8.5	1.1	2.6	3.3	2.8	2.5	2.4
780	160384	119.0	18568	10698	5.7	1.1	2.7	3.3	2.8	2.5	2.4
781	170384	163.7	17817	0	11.0	1.0	2.1	2.0	1.8	1.6	1.6
782	180384	180.1	13719	0	10.5	1.0	1.8	1.6	1.5	1.3	1.3
783	190384	97.5	16359	9185	13.1	1.2	2.9	3.6	3.1	2.8	2.7
784	200384	63.2	14815	11504	19.8	1.5	3.9	5.2	4.3	3.8	3.6
785	210384	56.0	12702	11793	18.2	1.6	3.9	5.4	4.5	3.9	3.8
786	220384	44.5	7955	11606	17.2	1.9	3.6	5.6	4.6	3.9	3.7
787	230384	48.1	6709	8108	16.5	1.8	3.1	4.4	3.6	3.1	2.9
788	240384	89.6	17178	0	14.0	1.3	3.2	3.0	2.6	2.3	2.2
789	250384	63.2	14826	0	17.8	1.5	3.9	3.5	3.0	2.6	2.5
790	260384	48.9	14934	0	18.4	1.8	4.8	4.3	3.5	3.1	2.9
791	270384	36.6	9474	7242	16.9	2.1	4.6	5.9	4.7	4.0	3.8
792	280384	41.3	19076	10772	15.1	2.0	6.4	8.1	6.6	5.6	5.3
793	290384	31.0	13264	6435	17.2	2.3	6.4	7.5	5.9	5.0	4.7
794	300384	27.3	11783	0	15.7	2.5	6.6	5.8	4.6	3.8	3.6
795	310384	25.3	9455	0	12.2	1.9	5.4	4.8	3.7	3.1	2.9
796	10484	24.5	10891	0	11.1	2.0	6.1	5.4	4.2	3.4	3.2
797	20484	24.4	8213	9040	16.9	2.0	5.1	7.7	6.0	4.9	4.6
798	30484	32.4	3246	0	15.1	1.7	2.6	2.3	1.9	1.6	1.5
799	40484	30.0	3419	12153	12.2	1.7	2.8	6.1	4.8	4.0	3.8
800	50484	26.0	4738	12055	13.0	1.9	3.6	7.2	5.6	4.7	4.4
801	60484	24.7	7900	11113	9.8	2.0	5.1	8.5	6.9	5.8	5.5
802	70484	23.2	7733	0	7.3	2.1	5.3	4.7	3.8	3.2	3.0
803	80484	23.3	10262	0	6.6	2.1	6.3	5.6	4.5	3.8	3.6
804	90484	32.0	14677	11342	15.5	1.7	6.2	8.8	7.2	6.2	5.9
805	100484	34.8	8678	10891	12.4	1.6	4.1	6.5	5.4	4.7	4.5
806	110484	27.7	ND	10651	13.4	1.9	ND	ND	ND	ND	ND
807	120484	24.8	ND	9304	10.0	2.0	ND	ND	ND	ND	ND
808	130484	23.3	15104	9403	15.4	2.1	8.3	11.0	8.8	7.4	7.0
809	140484	22.4	11010	0	13.2	2.1	6.8	6.1	4.9	4.1	3.9
810	150484	21.7	10451	0	8.3	2.2	6.8	6.0	4.8	4.0	3.8
811	160484	21.4	8013	9391	12.9	2.2	5.8	9.0	7.2	6.0	5.7
812	170484	21.2	7018	9474	9.0	2.2	5.4	8.7	6.9	5.8	5.5
813	180484	21.1	10451	5093	9.0	2.2	6.9	8.3	6.6	5.5	5.2
814	190484	21.2	8013	0	11.4	2.2	5.8	5.2	4.1	3.4	3.2
815	200484	20.7	7018	0	12.7	2.3	5.5	4.9	3.8	3.2	3.0
816	210484	20.5	ND	0	10.7	2.3	ND	ND	ND	ND	ND
817	220484	20.4	3804	0	10.5	2.3	4.1	3.6	2.8	2.4	2.2
818	230484	20.5	3350	0	12.2	2.3	3.8	3.4	2.7	2.2	2.1
819	240484	20.1	3056	0	13.5	2.3	3.8	3.3	2.6	2.2	2.1
820	250484	20.5	1967	0	10.7	2.3	3.2	2.8	2.2	1.9	1.8
821	260484	20.0	582	10658	9.8	3.3	3.6	7.8	6.2	5.2	4.9
822	270484	41.3	4200	10519	3.2	2.0	3.0	5.1	4.2	3.7	3.6
823	280484	50.1	1203	0	7.7	1.8	2.0	1.9	1.6	1.4	1.3
824	290484	39.8	974	0	10.5	2.1	2.3	2.1	1.8	1.5	1.5
825	300484	60.9	8882	11130	4.0	1.6	3.1	4.5	3.9	3.5	3.3
826	10584	38.4	2368	836	ND	2.1	2.7	2.5	2.1	1.9	1.8
827	20584	53.6	0	11573	ND	1.7	1.7	2.4	2.0	1.8	1.7
828	30584	41.7	0	10600	ND	2.0	2.0	2.8	2.3	2.0	1.9
829	40584	ND	0	9679	ND	ND	ND	ND	ND	ND	ND
830	50584	29.3	0	0	ND	2.5	2.5	2.3	1.9	1.6	1.5
831	60584	25.8	0	0	ND	2.9	2.9	2.6	2.2	1.9	1.9
832	70584	71.4	0	10351	ND	1.5	1.5	2.0	1.7	1.6	1.6
833	80584	67.7	0	10380	ND	1.5	1.5	2.0	1.8	1.6	1.6
834	90584	55.2	0	10496	ND	1.7	1.7	2.3	2.1	1.9	1.8
835	100584	48.0	0	9562	ND	1.9	1.9	2.5	2.2	2.0	1.9

ROW	DATE	FLOW	MCDC	BCWS	TSR	CALCULATED BOD_5 AT SITES B	C	D	E	EF	F
836	110584	80.6	0	9496	ND	1.4	1.4	1.8	1.6	1.5	1.4
837	120584	56.7	0	0	ND	1.7	1.7	1.6	1.4	1.3	1.2
838	130584	43.1	0	0	ND	2.0	2.0	1.9	1.6	1.5	1.4
839	140584	38.9	0	10598	ND	2.2	2.2	3.0	2.6	2.3	2.3
840	150584	37.6	0	10809	ND	2.2	2.2	3.1	2.7	2.4	2.3
841	160584	78.8	0	10525	ND	1.4	1.4	1.8	1.6	1.5	1.5
842	170584	34.8	0	0	ND	2.3	2.3	2.1	1.8	1.6	1.6
843	180584	39.8	0	4843	ND	2.1	2.1	2.4	2.1	1.9	1.8
844	190584	32.5	0	0	ND	2.4	2.4	2.2	1.9	1.7	1.6
845	200584	64.0	0	0	ND	1.6	1.6	1.5	1.3	1.2	1.2
846	210584	43.3	0	10691	ND	2.0	2.0	2.8	2.4	2.2	2.1
847	220584	35.1	0	11621	ND	2.3	2.3	3.4	2.9	2.6	2.5
848	230584	31.3	0	11740	ND	2.5	2.5	3.7	3.2	2.8	2.7
849	240584	28.8	0	10920	ND	2.7	2.7	3.8	3.3	2.9	2.8
850	250584	25.5	0	10513	ND	2.9	2.9	4.2	3.5	3.1	2.9
851	260584	181.0	0	0	ND	1.0	1.0	0.9	0.9	0.8	0.8
852	270584	218.0	0	0	ND	0.9	0.9	0.9	0.8	0.8	0.8
853	280584	118.0	0	10864	ND	1.2	1.2	1.5	1.3	1.2	1.2
854	290584	74.0	0	11945	ND	1.5	1.5	2.0	1.8	1.6	1.6
855	300584	75.0	0	10757	ND	1.4	1.4	1.9	1.7	1.6	1.5
856	310584	69.0	0	10644	ND	1.5	1.5	2.0	1.8	1.6	1.6
857	10684	63.5	0	9617	ND	1.6	1.6	2.1	1.8	1.7	1.6
858	20684	57.5	0	0	ND	1.7	1.7	1.6	1.4	1.3	1.2
859	30684	56.2	0	0	ND	1.7	1.7	1.6	1.4	1.3	1.2
860	40684	53.4	0	0	ND	1.8	1.8	1.6	1.4	1.3	1.3
861	50684	131.2	0	7030	ND	1.1	1.1	1.3	1.2	1.1	1.1
862	60684	112.8	0	6830	ND	1.2	1.2	1.4	1.3	1.2	1.2
863	70684	75.8	0	11102	ND	1.5	1.5	2.0	1.8	1.7	1.6
864	80684	60.0	0	10366	ND	1.7	1.7	2.3	2.0	1.9	1.8
865	90684	50.4	0	0	ND	1.9	1.9	1.8	1.6	1.5	1.4
866	100684	43.4	0	0	ND	2.1	2.1	1.9	1.7	1.6	1.5
867	110684	65.2	0	0	ND	1.6	1.6	1.5	1.4	1.3	1.3
868	120684	44.0	0	7509	ND	2.0	2.0	2.6	2.3	2.1	2.1
869	130684	38.5	0	11584	ND	2.2	2.2	3.2	2.9	2.6	2.6
870	140684	40.0	0	7548	ND	2.2	2.2	2.8	2.5	2.3	2.2
871	150684	33.7	0	0	ND	2.5	2.5	2.3	2.0	1.8	1.8
872	160684	31.0	0	0	ND	2.6	2.6	2.4	2.1	1.9	1.9
873	170684	27.6	0	0	ND	2.8	2.8	2.6	2.3	2.1	2.0
874	180684	27.5	0	0	ND	2.8	2.8	2.6	2.3	2.1	2.0

Key: FLOW = Manawatu River Flow at Site A ($m^3.s^{-1}$)

MCDC = Manawatu Cooperative Dairy Company BOD_5 addition to the river ($kgBOD_5.d^{-1}$)

BCWS = Borthwick CWS daily kill figures (lamb units).

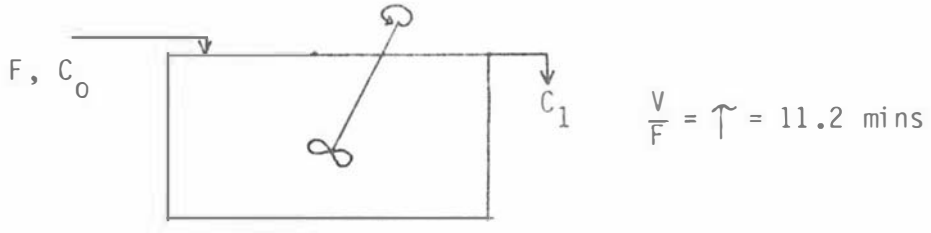
TSR = Total Surface Radiation ($MJ.m^{-2}.d^{-1}$).

BOD_5 = g,m^{-3}.

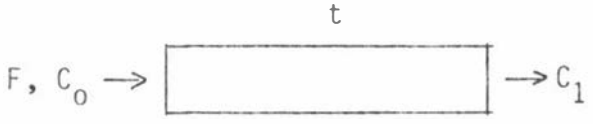
APPENDIX J: Relating a continuously mixed to a plug flow system (developed with the assistance of Dr P.C.Austin).

Consider the continuously mixed and plug flow systems given in Figure J.1 below.

Fig J.1: Equivalent Continuously Mixed and Plug Flow Systems.



Continuously mixed system (= channel system).



Plug flow system. If F, C_o, C_1, WSA : vol ratio are the same as in the continuously mixed system what is the travel time t.?

From continuously mixed system mass balance

$$\text{i.e.,} \quad F.C_o - F.C_1 = k_1 C_1 V \qquad (1) \div F \ (= \frac{V}{\tau})$$

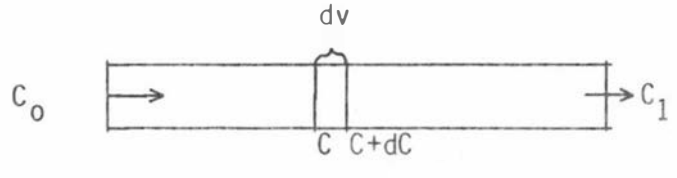
$$\frac{C_o}{C_1} - 1 = k_1 \tau \qquad (2) \div C_1$$

$$\frac{C_o}{C_1} = 1 + k_1 \tau \qquad (3)$$

$$C_1 = \frac{1}{1+k_1\tau} C_o \qquad (4)$$

$$k_1 = \frac{\frac{C_o}{C_1} - 1}{\tau} \qquad (5)$$

For the plug flow system consider the mass balance occurring over a small section d v.



Mass balance for this small section : Mass of substrate in = mass of substrate out + substrate used in reaction

$$\text{i.e., } F\,C_o - F\,(C + dC) - r\,dv = o \qquad (6)$$

$$\text{where reaction rate } r = \frac{dC}{dt} = -k_1 C \qquad (7)$$

from (6)
$$F\,dc = r\,dv \qquad (8)$$

$$\int_{Co}^{C_1} \frac{dc}{r} = \int_{o}^{v} \frac{1}{F}\,dv \qquad (9)$$

RHS
$$\int_{o}^{v} \frac{1}{F}\,dv = \frac{1}{F}\int_{o}^{v} dv \qquad (10)\ (F = \text{constant})$$

$$= \frac{1}{F}\left[v \,\middle|\, {}^{v}_{o}\right] \qquad (11)$$

$$= \frac{1}{F}\left[v - o\right] \qquad (12)$$

$$= \frac{v}{F} \qquad (13)$$

LHS
$$\int_{C_o}^{C_1} \frac{dC}{r} = \int_{C_o}^{C_1} \frac{dC}{k_1 C} \quad \left(r = \frac{dC}{dt}\ (7)\right) \qquad (14)$$

$$= \frac{-1}{k_1}\int_{C_o}^{C_1} \frac{dC}{C} \quad (k_1 \text{ constant for constant T}) \qquad (15)$$

$$= \frac{-1}{k_1} \ln C \,\Big|\, {}^{C_1}_{C_o} \qquad (16)$$

$$= \frac{-1}{k_1}\left(\ln C_1 - \ln C_o\right) \qquad (17)$$

$$= \frac{-1}{k_1}\left(\ln \frac{C_1}{C_o}\right) \qquad (18)$$

Combining (13) + (18)

$$-\frac{1}{k_1} \ln \frac{C_1}{C_o} = \frac{V}{F} \tag{19}$$

$$\ln \frac{C_1}{C_o} = \frac{-k_1 V}{F} = -k_1 t \quad (\frac{V}{F} = t) \tag{20}$$

$$\frac{C_1}{C_o} = e^{-k_1 t} \tag{21}$$

$$C_1 = C_o e^{-k_1 t} \tag{22}$$

Problem

What is the time t at which the substrate concentration in the plug flow system C_1 equals that in the continuous flow system C_1 at hydraulic residence time τ ?

Solution:

Combining (4) + (22)

$$\frac{1}{1 + k_1 \tau} C_o = C_o e^{-k_1 t} \tag{23}$$

$$\times \ (1 + k_1 t) \rightarrow \quad 1 = (1+k_1 \tau) e^{-k_1 t} \tag{24}$$

$$\text{take ln} \rightarrow \quad 0 = \ln (1+k_1 \tau) + \ln e^{-k_1 t} \tag{25}$$

$$0 = \ln (1+k_1 \tau) - k_1 t \tag{26}$$

$$\ln (1+k_1 \tau) = k_1 t \tag{27}$$

$$t = \frac{\ln (1+k_1 \tau)}{k_1} = \frac{V}{F} \tag{28}$$

where k_1 for channel can be found from the formula for a continuously mixed system i.e. $k_1 = \frac{\frac{C_o}{C_1} - 1}{\tau}$ (5)

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List of Abbreviations

ABM	= Algal biomass
AFDW	= Ash free dry weight
AI	= Autotrophic index
BCWS	= Borthwick CWS Limited
BGPR	= Benthic gross photosynthetic oxygen production rate
BM	= Benthic biomass
BOD_5	= 5 day biochemical oxygen demand
BOD_5UR	= BOD_5 uptake rate
BR	= Benthic respiration rate
Chla	= Chlorophyll a
Chla-SBGPR	= Chlorophyll a specific benthic gross photosynthetic oxygen production rate
COD	= Chemical oxygen demand
CPOM	= Coarse particulate organic material
CR	= Chamber respiration rate
dCOD	= Dissolved COD
DIN	= Dissolved inorganic nitrogen
DO	= Dissolved oxygen
DOM	= Dissolved organic matter
DRP	= Dissolved reactive phosphorus
$fBOD_5$	= BOD_5 of GF/C filtered, seeded sample
g	= Grams
GPR	= Gross production rate
h	= Hours
k_1	= First order BOD_5 decay rate
k_2	= Reaeration rate
k_e	= Vertical extinction coefficient of light in water
LMRTC	= Lower Manawatu River Technical Committee
MCDC	= Manawatu Cooperative Dairy Company
MRWB	= Manawatu Regional Water Board
NUR	= Nitrogen uptake rate
PAR	= Photosynthetically available radiation
PNCC	= Palmerston North City Corporation
P/R	= Ratio of gross photosynthesis to respiration rate
Q	= River flow
RR	= River respiration rate
sp	= Species
TN, TDN	= Total nitrogen, total dissolved nitrogen
TP, TDP	= Total phosphorus, total dissolved phosphorus
TDNUR	= Total dissolved nitrogen uptake rate
TDPUR	= Total dissolved phosphorus uptake rate
u	= Current velocity
uE	= Micro Einsteins
uv	= Ultra-violet
W	= Width
WSA	= Wetted surface area
WSBGPR	= Weight specific benthic gross photosynthetic oxygen production rate
WSBR	= Weight specific benthic respiration rate
WW	= Wastewater daily BOD_5 loading