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The River Kelvin at Garscubie Estate showing the macrophytic cover (August 1982)

STUDIES ON THE ALGAE OF THE POLLUTED
RIVER KELVIN

A thesis submitted to the
University of Glasgow
for the degree of
Doctor of Philosophy
in the Faculty of Science

by

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FEBRUARY 1984

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DEDICATION

To My Parents

To My Homeland

To whom I belong

DECLARATION

I hereby declare that this thesis is composed of work carried out by myself unless otherwise cited or acknowledged and that the thesis is of my own composition. The research was carried out within the period October 1979 - September 1982. This dissertation has not in whole or in any part been previously presented for any other degree.

Signed

 REZAN M.S. ANBER

DATE: 19/2/1984

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CONTENTS

	<i>Page No.</i>
Dedication	i
Declaration	ii
Acknowledgements	iii
Contents	iv
Summary	xiv
CHAPTER 1. INTRODUCTION	1
CHAPTER 2. THE RIVER KELVIN	29
CHAPTER 3. MATERIALS AND METHODS	44
3.1 Sampling Methods	44
3.2 Cleaning of Glassware	44
3.3 Chemical Analysis	45
3.3.1 Determination of reactive phosphorus	45
3.3.2 Determination of reactive silicate	48
3.3.3 Determination of nitrate	51
3.3.4 Determination of nitrite	57
3.3.5 Determination of ammonia	59
3.3.6 pH determination	63
3.3.7 Dissolved oxygen and BOD ₅	64
3.3.8 Determination of heavy metals, Zn, Pb and Cd	68
3.4 Pigment Analysis	68
Determination of chlorophyll <u>a</u> and phaeopigments	68
3.5 Enumeration of phytoplankton	72

	Page No.
3.6 Epiphytic Algae	74
3.6.1 Macrophyte population	74
3.6.2 Methods for the assessment of epiphytic algae growing on the macrophytic angiosperms	76
3.6.3 Methods of using artificial and natural substrata	78
3.7 Carbon Fixation Studies	82
3.7.1 ^{14}C light and dark bottle technique for measuring productivity of the phytoplankton	82
3.7.2 ^{14}C light and dark bottle technique for measuring productivity of the epiphytes	84
3.8 Algal Bioassays	85
3.8.1 Algal assay procedure	85
3.8.2 Measurement of chlorophyll	87
3.8.3 Measurement of photosynthesis	88
 CHAPTER 4. RESULTS	
4.1 Physico-Chemical Analysis	93
4.1.1 Current flow	93
4.1.2 Water temperature	95
4.1.3 pH	99
4.1.4 Dissolved oxygen	101
4.1.5 Biological oxygen demand (BOD_5)	104
4.1.6 Reactive phosphate	108
4.1.7 Nitrate + nitrite	111
4.1.8 Ammonia	115
4.1.9 Reactive silicate	118

	Page No.
4.2 Biological Data	
4.2.1 The phytoplankton	122
4.2.2 Chlorophyll <u>a</u> and the phaeopigments	160
4.2.3 Carbon fixation by the phytoplankton	165
4.2.4 Filamentous algae	168
4.2.5 Flowering plants	173
4.2.6 The epiphytic algal flora	177
4.2.7 Species diversity of the epiphytic population	180
4.2.8 Use of natural substrata	199
4.2.9 Carbon fixation by the epiphytic diatoms using ^{14}C	201
4.3 Bioassay Procedure to Assess Growth Potential of the River Water	202
4.3.1 Measuring cell number	203
4.3.2 Cell appearance in river water	215
4.3.3 Measurements of chlorophyll <u>a</u> contents	216
4.3.4 Pigment measurements	218
4.3.5 Rates of photosynthesis	221
CHAPTER 5. GENERAL DISCUSSION	224
REFERENCES	266

INDEX OF FIGURES

Figure	Page No.
2.01 The River Kelvin showing sampling and gauging stations	30
2.02 Diagram showing sources of effluents found in the River Kelvin	31
3.01 Cadmium-copper column used for reducing nitrate to nitrite	53
3.02 A calibration standard curve relating the NH_4Cl concentration with optical density	62
3.03 Diagram of grapnel used for sampling macrophyte populations	75
3.04 The layout of the apparatus used in the oxygen electrode experiment	90
3.05 A diagram of the oxygen electrode	91
4.01 Mean monthly flows in the River Kelvin	94
4.02 Mean pH and temperature values for the river throughout sampling period	97
4.03 pH and temperature profiles for the River Kelvin	98
4.04 Dissolved oxygen and BOD_5 in the River Kelvin	103
4.05 Oxygen and BOD_5 profiles for the River Kelvin	107
4.06 Reactive phosphorus concentrations observed in the River Kelvin	110
4.07 The total nitrate + nitrite levels observed in the River Kelvin	114
4.08 Concentrations of ammonia observed in the River Kelvin	117

Figure	Page No.
4.09 Reactive silicate concentration observed in the River Kelvin	121
4.10 Profiles for the total nitrate + nitrite and ammonia for the River Kelvin	123
4.11 Reactive phosphorus and reactive silicate profiles for the River Kelvin	124
4.12a The total number of the suspended diatoms recorded in the River Kelvin during April 1980-September 1981	137
4.12b The total number of the suspended diatoms recorded in the River Kelvin during October 1981-September 1982	138
4.13 The seasonal variations recorded for the suspended diatoms in the River Kelvin	141
4.14 Chlorophyll <u>a</u> contents in the River Kelvin	162
4.15 The phaeopigment contents in the River Kelvin	164
4.16 Percentages of the different species of the epiphytic diatoms at Station 2 during 1980	182
4.17 Percentages of the different species of the epiphytic diatoms at Station 4 during 1980	183
4.18 Percentages of the different species of the epiphytic diatoms at Station 6 during 1980	184
4.19 Percentages of the different species of the epiphytic diatoms at Station 8 during 1980	185
4.20 Percentages of the different species of the epiphytic diatoms at Station 9 during 1980	186

Figure	Page No.
4.21 Percentages of the different species of the epiphytic diatoms at Station 2 during 1981 and 1982	188
4.22 Percentages of the different species of the epiphytic diatoms at Station 3 during 1981-1982	189
4.23 Percentages of the different species of the epiphytic diatoms at Station 4 during 1981 and 1982	191
4.24 Percentages of the different species of the epiphytic diatoms at Station 5 during 1980-1982	192
4.25 Percentages of the different species of the epiphytic diatoms at Station 6 during 1981 and 1982	193
4.26 Percentages of the different species of the epiphytic diatoms at Station 7 during 1981 and 1982	195
4.27 Percentages of the different species of the epiphytic diatoms at Station 8 during 1981 and 1982	196
4.28 Percentages of the different species of the epiphytic diatoms at Station 9 during 1981 and 1982	198
4.29 <i>Ankistrodesmus falcatus</i> cell numbers plotted as $\log n_t$ against t during December 1980	206
4.30 <i>Scenedesmus quadricauda</i> cell numbers plotted as $\log n_t$ against t during December 1980	208
4.31 <i>Scenedesmus quadricauda</i> cell numbers plotted as $\log n_t$ against t during January 1981	210
4.32 <i>Scenedesmus quadricauda</i> cell numbers plotted as $\log n_t$ against t during February 1981	212

<i>Figure</i>	<i>Page No.</i>
4.33 <i>Scenedesmus quadricauda</i> cell numbers plotted as $\log n_t$ against t during March 1981	213
4.34 <i>Scenedesmus quadricauda</i> cell numbers plotted as $\log n_t$ against t during April 1981	214
4.35 Percentages of the abnormal cells formation in the coenobia of <i>Scenedesmus quadricauda</i>	217
4.36 Chlorophyll <u>a</u> measurements in the <i>Scenedesmus quadricauda</i> cultures after one week and 2 weeks cultivation extracted in hot methanol	219
4.37 Chlorophyll <u>a</u> and phaeopigments in the cultures of <i>Scenedesmus quadricauda</i> using 90% acetone for extraction	220
4.38 The photosynthesis rate expressed as the percentage of control in the <i>Scenedesmus quadricauda</i> cultures	222

INDEX OF PLATES

Plate		Page No.
2.03	Station 1 - River Kelvin near its source	33
2.04	Station 2 - Inchbelly Bridge	33
2.05	Station 3 - B757 Road Bridge	35
2.06	Station 4 - Luggie Water	35
2.07	Station 5 - Spring Farm Bridge	36
2.08	Station 6 - Bardawi Bridge	36
2.09	Station 7 - Allander Water	38
2.10	Station 8 - Balmuildy Bridge	38
2.11	Station 9 - Lady Campbell Bridge	39
2.12	Station 10 - Kelvin Bridge	39
4.01)		134
)		
4.02)	The diatoms recorded in the River Kelvin	135
)		
4.03)		136
4.04	The filamentous algae recorded in the River Kelvin	169
4.05	<i>Cladophora</i> spp. forming a green carpet on the river bed at Station 4	171
4.06	<i>Cladophora fracta</i> a single tuft of about 5 metres length from Station 4	171
4.07	<i>Potamogeton filiformis</i>	174
4.08	<i>Potamogeton natans</i>	174
4.09	The unicellular green algae used in the algal bioassay experiments - normal and abnormal cells	204

INDEX OF TABLES

Table	Page No.
4.01 Water temperature recorded for the River Kelvin	96
4.02 pH values recorded in the River Kelvin	100
4.03 Mean data for the dissolved oxygen concentration recorded in the River Kelvin	102
4.04 Mean data for the BOD_5 values recorded in the River Kelvin	105
4.05 Mean data for the phosphate P recorded in the River Kelvin	109
4.06 Mean data for the total nitrate + nitrite concentration recorded in the River Kelvin	112
4.07 Mean data for the ammonia concentrations recorded in the River Kelvin	116
4.08 Mean data for the reactive silicate recorded in the River Kelvin	119
4.09 The systematic list of the diatoms recorded in the River Kelvin	126
4.10 Chlorophyll a measurements recorded in the River Kelvin	161
4.11 Phaeopigment measurements in the River Kelvin	163
4.12 ^{14}C fixation by the phytoplankton	166
4.13 The distribution of the macrophytic angiosperms observed in the River Kelvin	176
4.14 The total number of the epiphytic diatoms and the percentages of their distribution on the different regions of <i>Potamogeton natans</i>	178

	Page No.
Table	
4.15 The total number of the epiphytic diatoms and the percentages of their distribution on the different regions of <i>Potamogeton filiformis</i>	179
4.16 ^{14}C fixation by the epiphytic diatoms	201
4.17 The cell numbers in the 2-day-old cultures of <i>Scenedesmus quadricauda</i> and their growth maximum compared with the control	209

SUMMARY

The macroscopic and microscopic vegetation of the River Kelvin was studied in relation to pollution during the period October 1979 - September 1982.

The main sources of pollution in the river were from treated domestic sewage and industrial effluents. Water samples from 10 stations on the river were analysed chemically for the amounts of nutrients based on monthly sample collections. The parameters used for estimating the pollution were phosphate P, nitrate + nitrite N, ammonia, dissolved oxygen and biological oxygen demand. The results showed high but variable nutrient levels at the different stations, being almost always highest at Luggie Water, one of the tributaries of the Kelvin, and the lowest were always nearer the source. No signs of deoxygenation were observed in the river, with average oxygen levels not less than 6 mg l^{-1} . The average BOD_5 values were not very high, being always $< 5 \text{ mg l}^{-1}$. During summer the nutrient levels were higher than in autumn and winter. Analysis of river samples for heavy metals indicated that levels were not significant, and flow rates were analysed from gauging station data supplied by the Clyde River Purification Board. Dissolved silica was also measured in the river as it is an essential element for diatom growth. Unlike the other nutrients, the dissolved silica was found in high levels near the source.

Microscopic examination of the samples was carried out as well as determination of chlorophyll a, phaeopigments and carbon fixation measurements. The phytoplankton (= tychoplankton) in the River Kelvin

was composed mainly of diatoms with the largest populations at stations with highest nutrient levels. The phytoplankton was present in maximum numbers during April-June, followed by a decrease during July and another peak during August. Thirty eight different species of diatoms were recorded in the River Kelvin, dominated by *Cyclotella meneghiniana*, *Gomphonema parvulum*, *Navicula avenacea*, *Nitzschia thermalis* and *Synedra ulna*. The carbon fixation results coincided more with chlorophyll a peaks than with the numbers of diatoms.

The phytoplankton population reflected the epiphytic flora; thus they were mainly attached species on the river macrophytes and other substrata dislodged by water movement. The epiphytic flora of the macrophytic angiosperms also consisted mainly of diatoms with the above species being the most numerous. The numbers of the epiphytic diatoms were determined for three different regions of *Potamogeton* spp., e.g. top, middle and basal regions, and the carbon fixation of the epiphytic diatoms from the three regions was also measured. Whilst the diatom numbers were often more on the basal and middle regions, carbon fixation by diatoms from the apical region was always of a higher value. An attempt was also made to estimate algal colonisation on sterilised natural substrata.

The filamentous algae formed the larger populations during the mid summer periods, being dominated by *Cladophora glomerata*, *Cladophora fracta*, *Oedogonium* sp. and *Vaucheria* sp. *Cladophora* spp. were observed forming a green blanket at some stretches of the river. Four species of macrophytic angiosperms were recorded in the River Kelvin, *Potamogeton filiformis*, *Potamogeton natans* and *Sparganium emersum*. Of these,

Potamogeton natans was the most abundant macrophyte. These were prominent in spring and summer with rapid defoliation at the onset of autumn.

Algal bioassay experiments were carried out to assess the water quality of the river using the coenobial green alga *Scenedesmus quadricauda* as the test organism. Its growth in the water samples from the river was compared with the growth in a balanced medium and estimated by measuring the cell numbers per unit volume. Chlorophyll a and the phaeopigments were also measured as well as the cell appearance in the cultures and measurements of photosynthesis rates. The bioassay results showed higher growth rates always in the water samples from the station with the highest nutrient levels and the lowest in the sample from near the source of the river. Abnormal cells in the coenobia of *Scenedesmus quadricauda* were noticed in the river cultures, being of higher percentages during winter than in summer. Chlorophyll a measurements were high during spring and summer, particularly during August when maximum values were recorded, then a dramatic fall during September. The phaeopigments were recorded in high proportions in the cultures during winter months. The photosynthesis rates were also at maximum during August with a fall during September in all the river cultures.

In general, the River Kelvin, whilst containing large quantities of treated domestic and industrial effluents, by virtue of its relatively high flow rates at most times of the year, can be described as having a "physiologically rich" lotic system which supports an abundant and diverse algal flora. The diatoms and filamentous green algal species

found at the stations are typical of the "recovery zones" of polluted rivers and it would seem that the effluent loading of the river is well contained by its natural recovery processes.

CHAPTER ONE

INTRODUCTION

Water plants have received much less attention from botanists than have land plants and the plants of rivers and streams least of all. Streams are complex habitats ranging from mountain torrents to quiet, almost still, lowland waters and may be deep or shallow, large or small. These physical differences will clearly affect the vegetation both macroscopic and microscopic (Wang and Evans, 1969). The present state of studies on river ecology has been ably described by Whittton (1975), viz. "studies on lotic algae are in many ways less advanced than studies on the phytoplankton of standing waters. The water industry has been on the whole less interested in the algae of rivers and streams than those of lakes and reservoirs, and so fewer people have worked on the former. Changes in a lotic environment are often both more rapid and less predictable than in a lentic one. There are factors such as floods and desiccation which frequently play a major role in the former, but are only seldom encountered in the latter".

In Britain, relatively little is known about the ecology of phytoplankton in rivers compared with lakes and reservoirs. Many of the studies have been qualitative and few quantitative. Thus Fritsch (1902; 1903) described the principal phytoplankton species for the River Thames and found a well marked living phytoplankton all the year round with diatoms forming a very large percentage of the population. This study was continued by Rice (1938) for the period 1928-1932, sampling with a 0.1 mm mesh tow-net, and he also examined the phytoplankton of a number of backwaters. Fritsch (1905) studied the phytoplankton of the River Trent at Nottingham

and Cam at Cambridge during one month (August) and found that both rivers compared in some degree to the situation in the Thames. Butcher (1924) made preliminary studies on the River Wharfe and found that the plankton consisted of diatoms which steadily increased to a maximum from the end of April to May. Quantitative studies for the streams Rheidol and Melindwr in Cardiganshire, Wales, were carried out by Reese (1937). Diatoms were the only frequent species during October until December and the period from April to October was characterized by an abundant growth of numbers of Chlorophyceae, and certain diatoms such as species of *Gomphonema*, *Achnanthes* and *Nitzschia*. Studies on the Rheidol were continued by Jones (1949a) who made an ecological survey of the river after recovery from severe lead pollution. It was found that phanerogamic vegetation was present only in the lower reaches of the river, whilst in the upper reaches the bryophyte vegetation was well developed. The algal flora was rich with *Lemanea* and *Stigeoclonium* being the dominant types. Butcher (1946a) gave estimations for the algal growth on the river bed of three highly calcareous streams, the Itchen at Alresford, and the Test and Avon in Hampshire. He showed that there was generally a single annual cycle with a maximum in midsummer and minimum in winter. There was a single algal community throughout the year dominated by *Cocconeis placentula* Ehr., *Gomphonema* spp. obviously varied in quantities but their maxima was not at the same time every year and the average community increased downstream due to the progressive eutrophication of the water which contained great amounts of calcium, other mineral salts and certain amounts of organic plant nutrients. Jones (1949b)

described the flora of two calcareous rivers Swadde and Clydach in south Wales. Here the dominant green alga was *Ulothrix* with a rich diatom flora consisting of *Diatoma vulgare* Bory, *Ceratoneis arcus* Kütz, *Achnanthes* spp., *Cocconeis placentula* and *Navicula viridula* Kütz. Jones (1951) made an ecological investigation for the River Towy in central Wales and he found that during the dry and sunny summer the river had a rich flora consisting of 130 species of algae. The dominant green algae were *Oedogonium*, *Spirogyra* and *Scenedesmus* while in a cool wet summer the flora was dominated by *Stigeoclonium*, *Ulothrix* and various diatoms. Williams (1950; 1954) described some occasional records of nanophytoplanktons from the River Dee at Chester. Phytoplankton of the River Lee in Hertfordshire were studied by Swale and Belchar (1959) and Swale (1964). They described species comprising the benthic diatom population in relation to their frequent appearance in the plankton. *Stephanodiscus hantzschii* Grün. was the dominant alga of the plankton in the years with low discharge. Round (1960a) examined the diatom flora of some springs in Malham Tarn region of Yorkshire. The flora was related to the alkaline status of the water and consisted mainly of species characteristic of running water such as *Meridion circulare* Grav. *Navicula cryptocephala* Kütz and *Amphora ovalis* Grun. Whitton and Dalpra (1968) compared the floristic changes in the River Tees and Skerne in England with former studies by Butcher (1932b; 1933) and Butcher, Longwell and Pentelow (1937). They found that the dominant diatoms of the reaches studied were unchanged within the time, (i.e. *Cyclotella meneghiniana* Kütz, *Cocconeis placentula*, *Navicula* sp.

and *Rhoicosphenia curvata* Kütz. Only one marked change had occurred since 1933, that was the intensive growth of *Enteromorpha intestinalis* (L.) Grev. which was not on Butcher's list. Swale (1969) reported on a more intensive study of phytoplankton for two rivers, the Severn at Preston and the Stour at Essex. Both showed small centric diatoms of the genera *Cyclotella* and *Stephanodiscus* to be the dominant algae in spring and summer. During the period April 1967 - April 1968 Kowalczewski and Lack (1971) studied the phytoplankton production and respiration of the River Thames and the Kennet at Reading. They found spring, summer and autumn chlorophyll peaks in the Thames (max. 219 mg m^{-3}) but there was very little variation in the Kennet (max. 38.2 mg m^{-3}). In both rivers, lowest concentrations were found during winter. Respiration rates fluctuated in both rivers. Lack (1971, 1973) and Berrie (1972) carried out investigations on the former rivers. In both rivers there were spring and autumn blooms of centric diatoms; Chlorophyceae were most abundant in summer. In the Thames, the population size was related to the discharge. The highest numbers were always in the periods of low discharge. In the Kennet increases in the cell numbers were with the discharge increase, due to the influx of benthic forms. Moore (1976) described the seasonal succession of algae from the Avon, a large slow-flowing river in southern England. He found that the flooding and water velocity affected the standing crop of most communities. Nutrients did not limit the growth in the river except the summer competition of diatoms for silicon. Moss (1977) carried out a survey for the turbidity and phytoplankton in the Norfolk Broads and rivers of East

Anglia. The nutrient loading and flushing coefficient of the waterway caused the difference in species composition and distribution of phytoplankton. Aykulu (1978) studied the seasonal changes in composition and cell densities of phytoplankton for the River Avon and he found that the phytoplankton of the river consisted of three algal groups; centric diatoms which increased in spring and autumn and Chlorophyceae and Cryptophyceae in summer. Phytoplankton increased downstream totally. Furet (1979) investigated the phytoplankton of the River Wye system. He found that centric diatoms only contributed conspicuously to the phytoplankton population in the river. However, throughout the whole river the greatest growth of phytoplankton also occurred in the warmest months. Recently Holmes and Whitton (1981a) studied the phytobenthos of the River Tees and its tributaries. *Cladophora glomerata* (L.) Kütz. was dominant at most downstream sites and spring diatom outbursts occurred in the upstream site one month before occurring in the downstream one. Pennate diatoms appeared to be dominant in both upstream and downstream sites. Holmes and Whitton (1981b) also described the plant communities in four fast flowing rivers (Tyne, Wear, Tees and Swale) in England. Centric and pennate diatoms formed the majority of cells with green algae second. Centric diatoms were relatively more abundant at downstream sites.

For the aquatic macrophytes a number of bryophytes are only

found in association with running water, (Watson 1919). In Britain several species of *Ranunculus* are confined to fast rivers and *Potamogeton* is frequently found as the dominant in polluted rivers (Haslam, 1978). Butcher (1927, 1933) divided the vegetation of each chemical water type into communities each with characteristic bottom deposits. He found that deposition of silt was the most important environmental condition affecting the flora of the river. The current is the chief factor which affects macrophyte distribution and quantities are continuously reduced in rivers with large sudden floods. Other authors have described similar correlations e.g. (Hynes, 1970; Whitton and Buckmaster, 1970 and Haslam, 1971). Edwards and Owens (1960, 1962) and Owen and Edwards (1961, 1962) investigated the freshweights of aquatic macrophytes and productivity for four streams in southern England (Test, Ivel, Yare and Chess). They recorded summer maxima for the crops between June and September. Conclusions were drawn concerning the effect of sewage effluent discharge, and the progressive change in the dominance of plant species found suggested that pollution and lack of nutrients were not limiting the growth of the macrophytes, but the amount of solar radiation available did affect their growth. A net primary productivity of 1.48, 1.80 and 2.30 gm organic carbon m^{-2} day $^{-1}$ was recorded. The average increase in dissolved oxygen concentration brought about by plant growth was equivalent to about 1:65 ppm per mile. In larger and deeper rivers (River Frome) average spring biomasses of 240 and 127 gm m^{-2} were estimated in consecutive years (Westlake, 1968). Whitton and Buckmaster

(1970) in a study of River Wear macrophytes found that the flow rate reduction and associated changes in the substratum changes the macrophyte flora on passing down the River Wear. Holmes, Lloyd, Potts and Whitton (1972) stated that "any further changes in management such as the mixing of different river water, may be expected to bring about floristic changes". Edwards, Benson-Evans, Learner, Williams and Williams (1972) made a biological survey of the River Taff in South Wales. They found that the river bed at the headwaters was full of an extensive growth of *Ranunculus* sp., but in downstream situations *Mimulus* sp. replaced it. *Cladophora glomerata* and *Ulothrix* sp. formed extensive growths at all sites accompanied by pennate diatoms such as *Diatoma*, *Nitzschia*, *Navicula* and *Cocconeis*. Lack (1973) studied the macrophytes of the Thames and the Kennet at Reading. The dominant macrophytes were *Acorus calamus* L. and *Nuphar lutea* (L.) Sm. Investigations on River Tees macrophytes were carried out by Holmes and Whitton (1977a). *Ranunculus penicillatus* Dumort. was invading the river. The results of this study were compared with the vegetation of the River Swale in Yorkshire (Holmes and Whitton, 1977b) and it was found that angiosperms such as *Myriophyllum alterniflorum* D.C. and *Potamogeton natans* L. which were present in the Tees above the projected abstraction point were absent from the Swale above the inflow point. Holmes and Whitton (1981c) gave an account of a partial survey of the macrophytes of the River Tyne and compared it with their previous study, Holmes, Lloyd, Potts and Whitton (1972). They found

similar species distribution with few further species recorded such as *Cladophora aegagropila* Kütz. and *Potamogeton perfoliatus* L. They also investigated the River Tees and its tributaries (Holmes and Whitton 1981a). Bryophytes showed a greater cover throughout the year and the blue-green alga *Phormidium* occurred in autumn. The ecology of the River Lambourn, England, has been investigated by Ham, Wright and Berrie (1981); Ham, Cooling, Hiley, McLeish, Scorgie and Berrie (1982) and Wright, Cameron, Hiley and Berrie (1982). Of the macrophytes, *Ranunculus penicillatus* was the dominant in the unshaded parts. It grew rapidly in spring and summer and its growth was related with the mean discharge. Shading, which is caused by water turbidity due to the aggregation of epiphytic algae on the surface of the plant, restricted its growth. On the other hand the macrophyte *Bernula erecta* Huds. Coville was dominant on the shaded sections and its maximum growth was during autumn.

Although there are a considerable number of studies on the ecology of attached algae in rivers and streams little is known of the relative numbers of algae present or their distribution over the stream bed. However studies dealing with water quality are of great relevance. As stated by Whitton (1975) "It is difficult to relate together much of the studies carried out on other rivers because of the great diversity of the sampling techniques and presentation of results. The difficulties of sampling heterogenic vegetation which reflect the discontinuities of the rivarian environment in both space and time and the absence of a theoretical framework to bring together data collected, are two fundamental

reasons for the adverse nature of studies of benthic communities of the lotic environment". As natural populations are sometimes difficult to sample, studies on benthic algae have more frequently been carried out on the populations which developed on artificial substrata exposed in the water for a known period of time. The use of artificial substrata has been reviewed by Sladekova (1962) and Schwoerbel (1970).

Fritsch (1929) studied the encrusting algal communities of a number of rapidly flowing streams between Lynton and Ilfracombe on the north coast of Devonshire. The entire flora of such rapid streams consists of such attached forms. He distinguished three types of algal communities belonging to the Cyanophyceae and only one diatom, *Cocconeis placentula*, which was present in considerable numbers. In course of investigations of the biology of the River Lark, Butcher, Pentelow and Woodley (1931a & b) found that there was a very extensive growth of algae on stones and other submerged objects on the river bed for the most part of the year. They used submerged glass slides for their growth studies. Spring growths on the slides consisted of diatoms and the summer growths were chiefly green algae of the Chaetophoraceae and blue green algae of the Chamaesiphonaceae. Butcher (1932a) added further knowledge to the former study by describing few species such as *Sporotetras pyriformis* sp. et.gen.nov., *Ulvella frequens* Snow, *Stigeoclonium farctum* Kütz, *Gongrosira incrassans* Reinsch and *Chaetopeltis megalocystis* Schmidle. Also Butcher (1932b) used a rigid metal photographic printing frame laid on the bed of the

stream which held five glass slides in the Rivers Tees and Lark. From these he recorded the growth of diatoms such as *Synedra ulna* (Nitz) Ehr. and *Gomphonema olivaceum* (Lyngb.) Kütz and the green filamentous algae *Ulothrix zonata* (Weber and Mohr) Kütz in spring. The summer period was dominated by *Cocconeis placentula* and *Stigeoclonium farctum*. In the winter period there was very little growth of diatoms. Butcher (1940) investigated the algae that grew on submerged slides in the River Hull, Yorkshire, over a period of 2½ years. The dominant algae were *Cocconeis placentula* and *Ulvella frequens*. This community has been found in rivers having a good amount of nitrate and other mineral salts. Benthic algae are known to be washed from their microhabitats into suspension, into streams and rivers of all sizes (Blum, 1956), but little is known of the seasonal pattern of drifting algae. Round (1957) described the diatom flora of some springs and streams in Buckinghamshire. The sediment and epiphytic communities yielded a large number of diatoms of the genera *Achnanthes*, *Nitzschia*, *Cocconeis* and *Synedra*. The epilithic samples were poor in species with generally a single dominant (e.g. *Gomphonema*). Douglas (1958) gave methods for estimating populations of attached algae, particularly diatoms, in a small stony stream in Lancashire. Diatoms were a predominant group in the flora. The main diatom communities were *Gomphonema*, *Synedra*, *Eunotia* and *Cocconeis*. She related the poor growth in the upper reaches to a peaty deposit on the substratum, and an increase downstream was due to the washing of cells in the upper reaches and their deposition lower down. Round (1960b) examined the diatom flora which was growing on sediments and plants in streams around Malham Tarn in

Yorkshire. He related the flora to position rate of water flow and the nature of the supporting plants and sediments. He recorded dominant epiphytic communities such as *Cocconeis placentula*, *Cymbella ventricosa* Kütz. and *Synedra ulna* and among the epiphytic communities there were species of *Frustulia vulgaris* Thwaites., *Cymbella ventricosa* and *Amphora ovalis*. Bell (1976) studied the ecology of epiphytic microalgae on submerged macrophytes in an eutrophic canal between Wigan and Liverpool. He investigated the attached algal populations on both *Elodia canadensis* Michx. , which was the dominant macrophyte, and on introduced glass microscopic slides by means of direct counts and Chlorophyll *a* extraction. Epiphytic densities showed only a single peak in spring while the total population was greatest in summer but also high in spring. Diatoms (mainly *Navicula* spp., *Nitzschia*, *Gomphonema* and *Cocconeis*) were most important in early and late summer, while the green *Stigeoclonium* spp., *Spirogyra* spp. and "Coccoid green" algae tended to dominate the population in mid summer and winter. The contribution of the blue-green algae *Oscillatoria limosa* (Roth) C.A. Agardh and *Lyngbya* spp. to the community was mainly confined to the summer months. Marker and Gunn (1977) investigated two eutrophic streams, Bere stream and the River Frome and two soft water streams, Ober and Dockens Water in southern England. They measured the chlorophyll which was largely related to pennate diatoms which had detached from the benthic flora. Maximum chlorophyll concentration occurred regularly in April for the eutrophic streams and Ober Water, but in Dockens Water the seasonal variation in the chlorophyll concentration

was different. There was a slight indication of lower chlorophyll concentrations in July and August, corresponding to a rise in the percentage degradation. Moore (1977a) studied the seasonal succession of attached algae for a tributary of the River Wylde in southern England. He correlated high rates of disappearance of algae from the substrate with flooding. The algae showed maximum development during the summer due to the high water temperature. The greatest diversity of the epiphytic algae were found in association with *Cladophora glomerata* and once again it was dominated by diatoms and members of the Chlorophyceae, Cyanophyceae and Euglenophyceae. Moore (1977b) also studied the standing crop of the phytoplankton and epiphytic algae for the Kennet and Avon Canal in southern England. He recorded low standing crops (not exceeding 5×10^4 cells ℓ^{-1}), during 1973 and 1974, due to competition with higher plants for some nutrient substances since phosphorus, nitrogen and silicon occurred abundantly in the water. *Achnanthes minutissima* Kütz was always predominant in the epiphytic assemblages on *Cladophora glomerata*. During a study of the seasonal changes in the diatom of the River Wye, Surrey, Moss (1977) found that *Rhoicosphenia curvata*, which is an epiphytic diatom, predominated in the river, and multiplied vigorously during the early spring and even in late winter to the extent that it was the source of turbidity in the water.

Mitchell (1980) studied 166 sites in South Wales rivers for factors affecting the distribution of epiphytic diatoms associated with riverine bryophytes and *Cladophora glomerata*. Diatom population on bryophytes showed a single summer peak, the dominant diatom species being *Eunotia*

pectinalis (Kutz.) Rbh. *Diatoma hiemale* (Lyngb.) Heib..

Achnanthes lanceolata (Breb.) Grun. and *Fragilaria* spp. changed little throughout the year. He showed that quality and level of the water were the most important factors affecting the distribution of diatoms and pH and organic enrichment were the most important factors influencing diatom and macrophyte distribution. Pentecost (1982) studied two limestone streams in northern England. He divided the stream into two sections. The region above the main waterfall was covered with a layer of tubaceous "aufwuchs" and the flora consisted of algae of two distinct groups, those attached to rock, stones and bryophytes and those which were heavily encrusted forming stony concretions and tufa. *Cratoneuron commutatum* Hedw. and *Eurhynchium riparioides* Hedw. were the dominant bryophytes. The lower reaches were dominated by calcareous "aufwuchs" and colonial algae.

There are relatively very few published studies on the lotic environment epilithic algae, probably because reliable quantitative estimates are difficult to obtain. Marker (1976a and b) used extracted chlorophyll *a* as a means of estimating the seasonal changes of the epipellic algae in a small hard water stream, Bere, and two soft water streams, Dockens and Ober. He recorded an April maximum in Bere which was dominated by pennate diatoms, but during the summer, a lime encrusted community developed which was overlayed by growths of *Vaucheria* and *Cladophora* in the late summer. In the soft water streams densities were lower and there were no clear spring maximum. Jones (1978) investigated the epilithic algae in Wifin Beck, a small stony stream in the English Lake District using a

direct counting technique for counting the algae directly on the surface of the stone (Jones, 1974). *Cocconeis placentula* was the dominant diatom with a large number of encrusting green, blue-green and red algae in the middle reach. *Cocconeis* standing crops were largely related to the rate of flow and its highest growth was in autumn, whereas the green, blue-green and red algae's highest growth was in spring.

The epipellic flora of rivers has been grossly neglected, yet very slow flowing rivers have algal communities living on the sediments. Moore (1977c) studied the epipellic, epilithic and planktonic algae in a small woodland stream Highland Water in southern England. He correlated shifts in the density of the three communities to low light intensities brought about by the canopy of tree leaves and not to changes in temperature or the concentration of phosphorus, nitrate and silicon. Recently Aykulu (1982) investigated the epipellic algal flora of the River Avon. Samples were collected by drawing a glass tube across the surface of the sediment. The habitat was dominated by diatoms, *Nitzschia* (particularly the needle-like diatom *Nitzschia acicularis* W.S.M.) and *Amphora* spp. which exhibited conspicuous growth phases at all stations. Spring peaks occurred in April and May. The non-diatom flora has shown less obvious seasonal development, the most striking feature was the great euglenoid growth in the summer and spring.

River pollution is not a recent problem. Indeed some of the British rivers are in a far better state at the present time than they have been for a hundred years. "The pollution of rivers

really began at the time of the industrial revolution, in the early nineteenth century, which resulted in a rapid deterioration in the quality of many rivers" (Best and Ross 1977). Rivers were the more usual victims of pollution where there was direct discharge of sewage and industrial wastes.

Sewage is a complex mixture of organic and inorganic natural substances as well as man-made materials. Butcher (1955) defined pollution by the actual changes brought about in the biological balance of the stream. Wisdom (1956) defined pollution as "The addition of something to water which changes its natural qualities so that the riparian owner does not get the natural water of the stream transmitted to him". Hawkes (1957) described pollution as any discharge of a substance into a stream which alters appreciably the composition or distribution of plant or animal communities by changing any of the following inter-related factors; (i) physical and chemical nature of the water, (ii) nature of the stream bed and (iii) current. He mentioned that the direct discharge of toxic material reduces both the number of individuals and the number of species. Any substance which, although inert and non-toxic, so changes the physical nature of the stream bed as to alter the community must be considered a pollutant. He also mentioned that if the physical nature of a discharge such as colour or temperature affects the physical nature of the river water which may also affect the communities then this must also be considered as a form of pollution.

The availability of food and oxygen and the chemical nature of the water are among the important factors determining the different

communities in stream life. The discharge of decomposable complex organic matter such as sewage to a stream changes the nature of the plant communities to an extent depending upon the degree of pollution. A self-purification process follows and different communities become successively established. Hawkes (1957) outlined and revised Kilkwitz and Marsson's scheme (1908, 1909) of ecological systems of saprobes in which they distinguished three zones of existence. The work has also been revised by Kolkwitz (1950) and Liebmann (1951) in the light of later work.

Polysaprobic: characterized chemically by a high concentration of complex decomposable organic matter derived from sewage discharges and some industrial effluents, oxygen being absent or present in traces only. H_2S is produced so considerable odour is present. Characterized biologically by the restriction of the community to a few groups, bacteria and protozoa being the most common. Culture counts of $>10^6 \text{ ml}^{-1}$ can be obtained.

Messaprobic: chemically defined by well-established oxidation process. Subdivided into two zones; α -Mesaprobiic (α -M) zone contains a high content of amino acids arising from the breakdown of complex compounds. The oxygen content may be considerable and because of the development of chlorophyll containing organisms especially algae, the oxygen content increases by day and declines by night. Biologically it still contains high numbers of bacteria; culture counts normally being $100,000 \text{ ml}^{-1}$. β -Mesaprobiic (β -M) zone is chemically distinguished by the continuing oxidation or mineralization and is the region of ammoniacal compounds and of the

fatty acids. The oxygen content is fairly high being never less than 50% of saturation. Biologically characterized by a fall in the bacteria counts to always $<100,000 \text{ ml}^{-1}$. However in the $\frac{S}{H}$ -mesaprofic zones there is a great diversity of plants and animals.

Oligosaprofic: this is the zone of completed oxidation or mineralization, organic substances having broken down. The water is clear and rich in oxygen, except on occasions when water blooms (growths of unicellular algae) develop and cause turbidity. Biologically characterized by a further fall in bacterial counts to $<100 \text{ ml}^{-1}$, and a wide range of species of plants and animals, including fish, is to be found.

There are many indices for classification of rivers based on biological data. Thorpe and William (1980) recommended few indices for the use of algologists and suggested the use of algae for the biological surveillance of rivers. These indices were:

1. The saprobian system including the Pantle and Buck index (1955).
2. The Palmer index (1969) or its reversed form as recommended by Benson-Evans et al. (1975) as it is self stabilizing in the original form.
3. Shannon index (Shannon and Weaver 1963) as a species diversity index.
4. The sequential comparison index (Cairns et al. 1968; Archibald, 1972) which is a semi-quantitative method.

Indices suggested by Hellawell (1978) have not been included either because of being more cumbersome to apply or they do not reflect community changes, e.g. Margalef's Index (1951).

Numerous studies have been carried out on the biological changes in rivers due to effluent discharge. Carpenter (1924) studied the flora of certain streams in the Aberystwyth district and she related "the relative poverty of the flora and fauna to pollution by lead mining and lead washing operations which affected the river by the discharge of galena particles, and also by the formation of lead salts in diffusible form through chemical interaction with the natural water. The cessation of the last of the lead mining and washing operations affecting the Rivers Rhiedol and Ystwyth was followed by a marked and rapid increase in the flora and fauna". In a survey of the tidal and non-tidal reaches of the River Tees Alexander, Southgate and Bassindale (1935) and Butcher, Longwell and Petelow (1937), described the effect of sewage effluent on the flora of the river. The upper reaches of the river supported little rooted vegetation. Although the effluent came from a small sewage works, the quantity was small and the dilution great with the result that the pollutational effects were only local. A change in the sessile microflora was observed immediately below the discharges. In the non-polluted stretch the algal community was dominated by the diatom *Achnanthes* spp. with the green algae *Chaetopeltis* in the summer. Below the effluent this community was replaced by one in which *Cocconeis* and *Chaemosiphonopsis* were dominant. The entry of the polluted River Skerne produced some marked changes in the flora, and resulted in an increase in the rooted vegetation (e.g. *Potamogeton*

interruptus). The abundance of *Cladophora glomerata* and *Sphaerotilus* was also shown to be due to pollution. Newton, Reese and Davies (1937) and Newton (1944) investigated the Rivers Rhiedol and Melindwr in Cardiganshire, mid-Wales. The presence of high levels of Zn and Pb caused serious pollution. The paucity of vegetation in the rivers were caused by the toxicity of zinc. The lack of colonization by plants was partly due to the blown material from old mine pits which accumulated around the vegetation during and after winds. The presence of detritus from the mines thickly covering the river beds mechanically deterred the establishment of macrophytes and reduced the microflora. Pentelow, Butcher and Grindley (1938) studied the effect of milk wastes on the flora of the Bristol Avon during 1935 and 1936. Immediately below the discharge there was a reduction in the algal growth but some distance below there was a marked increase; during the second year when the pollution was less the algal peak was nearer to the discharge. Below the effluent similar species were found to those above including *Navicula viridula* and *Surirella ovata* Kütz. in the spring and *Cocconeis*, *Chamaesiphon*, *Ulvella* and *Stigeoclonium tenuissimum* (C.A.Ag) Kütz. in the summer; *Cladophora glomerata* was present below the effluent. Jones (1940) investigated the zinc polluted River Ystwyth in Wales, and related the degree of pollution with the flood. This increases when the level of the river falls below normal. The river was silted up with broken rock and gravel carried out down from the mine workings; the main stream was completely devoid of phanerogamic vegetation and submerged grass and the bryophytic vegetation was scanty. This

study was continued by Jones (1958) who found that the river continued to be polluted. The flora above the mines was rich. *Stigeoclonium* was the dominant green alga mixed with a little *Zygnema* and *Spirogyra*; *Tabellaria flocculosa* (Roth.) Kütz. and *Ceratoneis arcus* were the dominant diatoms, but they disappeared below the source of pollution and the river flora seemed poor in quantity and variety. Butcher (1946b) studied the effect of three large effluents on the River Churnet, namely Leek Sewage Works, Leekbrook Dye Works and Copper Works. Below the first effluent discharge into the river sparse growth of algae increased to a high figure; *Gomphonema parvulum* Kütz., *Nitzschia palea* (Kütz.) W. Sm. and *Stigeoclonium tenue* were common, *Cocconeis* and *Chamaesiphon* were very rare. Below the dye works ammonia considerably increased with a slight reduction in oxygen content of the water. *Cocconeis* and *Chamaesiphon* were among the dominant diatoms. After the copper works' discharge, algae were exceedingly rare. After about 15 miles down river from the discharge point the algae had increased very greatly, the principal species being green algae and some diatoms. Butcher (1955) investigated the polluted condition of the Trent. He related types of pollution closely to the biological changes they produce, *Sphaerotilus natans* and often *Stigeoclonium tenue* were found in the polysaprobic zone, *Nitzschia palea* and *Gomphonema parvulum* in the mesosaprobic zone and *Cocconeis placentula*, *Synedra ulna* and *Chamaesiphon incrassatus* Grunow in Rabenhorst in the oligotrophic zone. Jolly and Chapman (1966) made a qualitative study of the effect of pollution on Farmer's Creek and Cox's River in Wales, based on monthly collections of plants

for a period of a year. Extensive growths of *Stigeoclonium* and diatoms immediately below the outfall was rapidly ~~suppressed~~ ^{PROMOTED} by the effluent. However it was found that the nutrients produced by mineralization of the sewage had induced a more extensive growth of algae in Cox's River. Casey (1969) recorded observations made on the chemical composition of some English chalk streams, River Frome, Doddings farm chalk spring and Bere Stream. He presented information on the rainfall, discharge, alkalinity, calcium, potassium, soluble phosphate, soluble silicate, nitrate and diurnal oxygen variation. The results showed that the chalk spring components remained fairly constant throughout the year. There were few fluctuations in the small chalk stream, and the large chalk stream showed the largest fluctuations. Benson-Evans and Williams (1970) have given a review of the biological studies carried out on the River Usk in Wales. The tributary (River Clydach) was described as very subject to spates causing scouring of species, especially the upper reaches together with high suspended solids from the mountain sides and old iron workings, but the most concern was an effluent which introduced domestic sewage and industrial waste containing anionic detergents. The river above the effluent supported a good diatom flora with peaks in spring and autumn. Downstream of the effluent these populations were reduced, but after the river recovery due to the gradual dilution of the effluent, a slight increase in the diatoms had occurred. Results showed a decrease in the diatom population caused by the detergents entrance into the river whilst it was stimulating the growth of perennial algae such as *Cladophora* and *Lemanea*. So the inorganic

nutrients are capable of supporting the growth of algae which may appear in large numbers during the summer months. Egglislow and Shakley (1971) sampled the suspended microscopic particles from a small stream, Shelligan Burn, and the River Almond in Scotland. The commonest material was non-cellular detritus with living diatoms. Some of the quantities of diatom species such as *Achnanthes minutissima*, *Cymbella ventricosa* and *Cocconeis placentula* varied considerably over the course of the stream. Filamentous green algae were fairly evenly distributed along the whole stretch of the river whilst zoospores and Chlorophyceae were much in higher concentration in one stretch of the stream.

Transparency of the water in some rivers has a great affect on the distribution of the vegetation above and below the pollution source. Tom, Hinge and White (1971) investigated the effect of Rye Meads sewage effluent upon the River Lee. In the upper reaches vegetation was visible and covered the whole bed of the river, submerged plants were present but below the discharge vegetation was hardly visible. Submerged plants cannot develop in the absence of light penetration. Water lilies were the only abundant vegetation below the discharge with *Myriophyllum* sp. and *Ceratophyllum* sp. occurring in few places in shallow marginal areas. Mann, Britton, Kwalczewski, Lack, Mathews and McDonald (1972) measured the productivity and energy flow at all trophic levels in the River Thames which carried considerable quantities of treated sewage effluent. They found that the photosynthesis in the water column in the main river was much higher than the photosynthesis in

a heavily polluted tributary in which respiration exceeded photosynthesis. Abdullah and Royle (1972) investigated the heavy metal content of some rivers in Wales. They found that the Welsh rivers are the primary source of heavy metals, Zn, Pb and Cu in Cardigan Bay. Casey and Newton (1973) surveyed the chemistry of the River Frome to prepare a model of the chemical environment of the river and to investigate the factors influencing variation in chemical composition. Their project was based on weekly measurements at 16 sampling points on the river. Flooding decreased the alkalinity and calcium concentration. The highest values for nitrates were at peak flows, phosphorus and potassium varied most and both showed a reduction between February and May. McLean and Benson-Evans (1974, 1977) studied some south Wales rivers for the distribution and the degree of branching of *Stigeoclonium tenue* in relation to organic pollution. It colonized organically polluted areas. The greatest tolerance for a number of environmental factors was expressed mainly in the spring months when the organisms became widely distributed in diverse habitats ranging from organically polluted to fairly clean conditions. They related the degree of branching of the filaments with physical and chemical parameters of the environment. Water velocity and depth did not affect the degree of branching whereas the high nutrient condition of the water decreased the degree of branching. McLean and Jones (1975) studied the Rivers Ystwyth and Clarach for the flora tolerance to heavy metals. They found that *Hormidium* sp. was the most tolerant filamentous green alga present, the composition of the diatom floras *Diatoma hiemale* and *Fragilaria capucina* Desm. was different in polluted sites as compared to that in the cleaner areas.

Changes in discharge and chemical composition were related to biological and physical conditions in South Winterborne, England, (Casey and Ladle, 1976). They found that Winterborne has a rich flora of algae and higher plants of which the annual sequences were influenced by the flow regime and the effects of human interference by cutting and removal of plants and addition of nutrients. Davies and Hawkes (1981) studied the effect of sewage effluent discharge into the River Cole by regular collections of water samples from 6 comparable stations. The effluent reduced the dissolved oxygen concentration and increased the ammonia, phosphorus and BOD_5 . The effects were more severe during the summer. Say and Whitton (1981) gave an account of the chemistry and floristic composition of stream sites in Gillgill Burn in northern England. The flora in the unpolluted reaches had 61 species, the polluted sites had a maximum of 41 species and the sites with the highest zinc (25.6 mg l^{-1}) had 25 species and it was comparatively rich. They related the different floristic changes to the changes in zinc levels. Recently Foster (1982a & b) described the algal flora of the Rivers Hayle and Gannel which drain copper and lead mining regions in Cornwall. Over the year of study the dominant filamentous algae remained unchanged at most sites. A *Microspora* community was a characteristic feature of all the mine sites, whilst *Zygnemales* community of *Spirogyra* and *Mougeotia* species were growing in low metal pollution. However polluted sites downstream of the mines had an intermediate flora of *Zygnemales*, *Microsporales* and *Ulothrichales*.

In summary pollution of rivers by effluents containing organic,

inorganic and toxic substances is a problem especially in countries which are highly populated and industrialized. Included in the source of pollution are sewage effluent, industrial effluents, farm and agricultural wastes and run-off from land.

When an effluent containing a lot of different materials enters the river, conditions become unsuitable for its organisms, either because of direct toxicity and the high nutrient levels or to the dissolved oxygen depletion which is due to either the growth of heterotrophic organisms or to increasing the water turbidity, by heavy loads of suspended matters, and reducing the rivers transparency. This reduces the light penetration and leads to the lack of photosynthesis and an increase in respiration, then an increase in the BOD_5 .

All these factors adversely affect the environment and changes the habitat by disappearance of some communities and extensive growth of others, and sometimes affecting the physical nature of the organisms. The effect of pollution is more severe at summer months due to low flow and lack of dilution. Fortunately rivers have the ability of self-purification and there are appreciable flows of clean diluting water available in the receiving river for recovery.

Rivers are strict individuals each of which varies in its own way so as to make nonsense of anything but a very broad general classification. There are few systems for the classification of rivers into different zones. This subject is discussed by Hawkes in Whitton (1975) under the heading "River zonation and Classification".

Carpenter (1928) was the first British worker to attempt classification of river zones. Her ideas were influenced by German investigators Steinmann (1907) and Thienemann (1912). She divided rivers into four fish reaches, the head stream, the trout beck, the minnow reach and the lowland reach. Butcher (1933) outlined a classification of river reaches on the basis of the larger plants that are to be found in them. He divided rivers into 5 zones.

1. The free floating communities occur in still waters.
2. Rooted plants with floating leaves occur in clear water and in backwaters of streams.
3. The communities of rooted plants entirely submerged occur in fairly deep waters on substrata of soft mud.
4. The reed swamps occur in very slowly flowing and still water.
5. The hemi-halophytic communities occur where there is a considerable proportion of salts in the soil.

He thought that this system is perhaps more useful than one based on fishes because fishes are an unsatisfactory basis for classification, as they often move into areas where they cannot maintain themselves, but plants are there to be seen and they do not move around and mislead one. Ohle (1937) used calcium concentration, distinguishing three classes having respectively 0.1 mg l^{-1} Ca, $10-25.5 \text{ mg l}^{-1}$ and $>25.5 \text{ mg l}^{-1}$ Ca. Calcium concentration is one of the greatest variables in freshwater and it is the only ion which has been found to correlate with faunistic and floristic groupings.

Tansley (1939) used the microvegetation as basis for classifying rivers. He also drew up a table indicating the relationship between current velocity and the nature of substratum, including the type of vegetation. Illies (1961) gave a generalized scheme of river zone classification for worldwide application. He made his primary division into two zones, the upstream, Rhithron, and downstream, Potamon.

1. Rhithron is defined as that part of the stream from its source down to the lowermost point where the annual range of monthly mean temperatures does not exceed 20°C. The current velocities are high and the flow volume is small. The substratum may be composed of fixed rock, stones or gravel and fine sand. Only in pools and sheltered areas is mud deposited.

2. Potamon is the remaining downstream stretch of river where the annual range of monthly mean temperatures exceeds 20°C, or in tropical latitudes with a summer maximum of the monthly mean exceeding 25°C the current velocity over the river bed is low and tends to be laminar. The river bed is mainly of sand or mud, although gravel may also be present. In the deeper pools oxygen may be depleted, light penetration is limited as mud is deposited.

Pennak (1971) gave a worldwide system for classifying brooks, streams and small rivers using different criteria, width, flow, current speed, substrata, summer temperature, winter temperature, turbidity, total dissolved organic matter, total dissolved inorganic matter, water hardness, dissolved oxygen, rooted aquatic plants

and streamside vegetation. Some of these criteria have much greater biological implications than others. He concluded that the lotic habitats which displayed similarity in the above features have ecologically similar biotas and the same kinds of species. Nevertheless, there are many kinds of polluted and unpolluted lotic habitats that cannot be classified with respect to these criteria.

This general review of algal ecology in rivers in Britain has indicated widely varying interpretation of both the feature and governing algal distribution in relation to the pollution. The key feature which emerges, as stated on p.25, is that rivers are individual ecosystems. For this reason the background studies to the present work on the River Kelvin are represented in the next chapters.

CHAPTER TWO

2. The River Kelvin:

Since each river is a highly individual ecosystem, a description of the River Kelvin will follow with relevant data on pollution and previous work on biological data.

The River Kelvin (Figure 2.01) is about 35 km in length and runs from its source in the Campsie Fells at Kelvinhead through farmland, villages and small Burghs and part of the City of Glasgow before it joins the River Clyde.

The major polluting loads discharged into the Kelvin and its tributaries are from local authority sewage treatment works (Figure 2.02). Although most of these works produce good quality effluents, there is very little dilution available in the stream and in few cases almost all the flow in the stream consists of sewage effluent e.g. as at Bishopbriggs Burn (613723). In fact, about 60% of the flow in the river consists of sewage effluent.

Farming is another potential source of pollution by two processes. Silage effluent, which is released from the crop during fermentation and storage, contains carbohydrates which cause growth and replication of bacteria in water and this leads to deoxygenation even in fast flowing streams. Slurry, which is kept inside during winter time, has similar properties to crude domestic sewage and would cause effects similar to silage effluent if allowed to escape to water courses.

For this study, 10 different sampling stations were chosen along the length of the river. These stations are also those used by the Clyde River Purification Board (CRPB)^{*} for their routine monitoring

*The assessments of water quality given are taken from the Annual Reports of the Clyde River Purification Board.

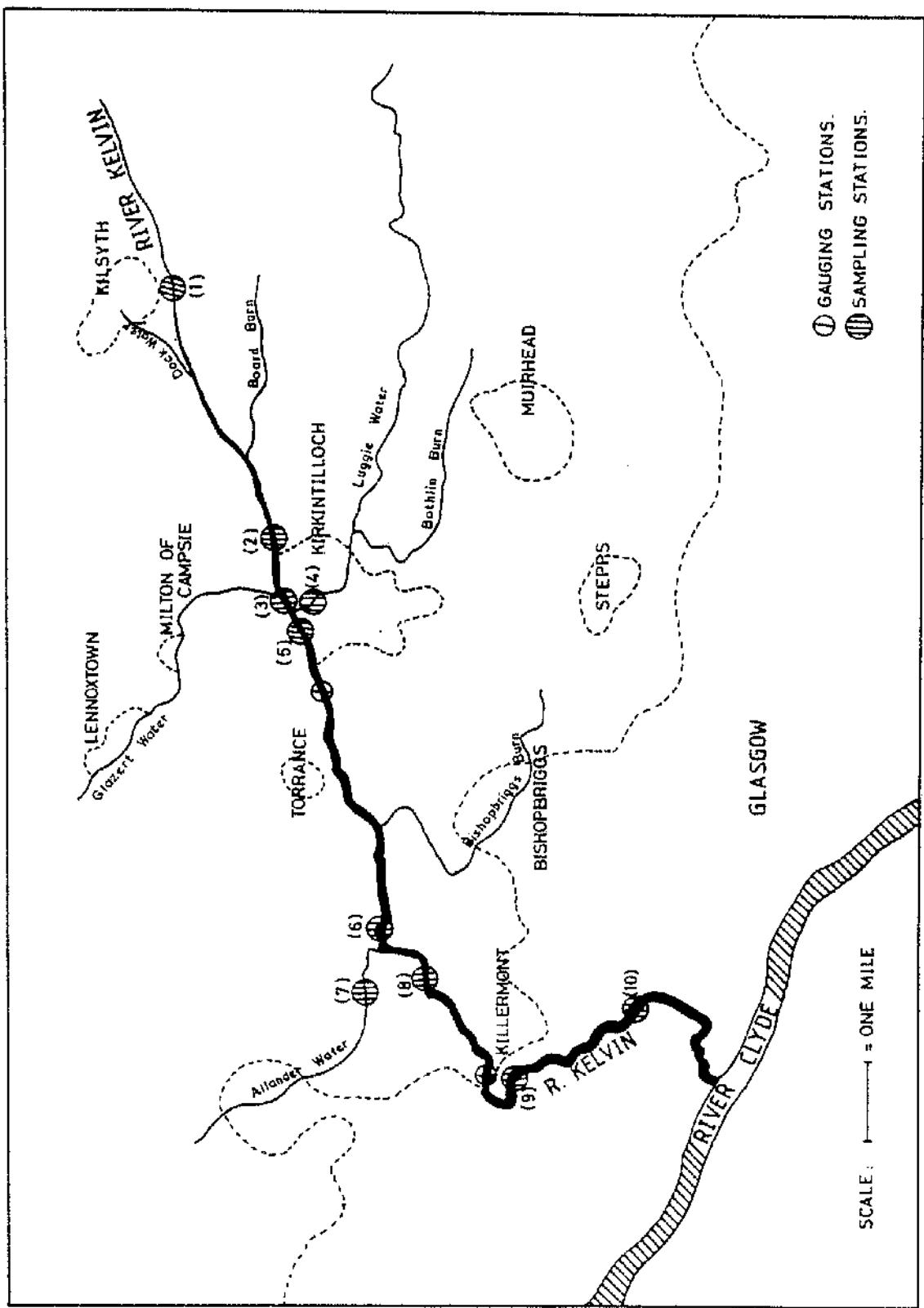


Figure 2.01. The River Kelvin showing sampling and gauging stations.

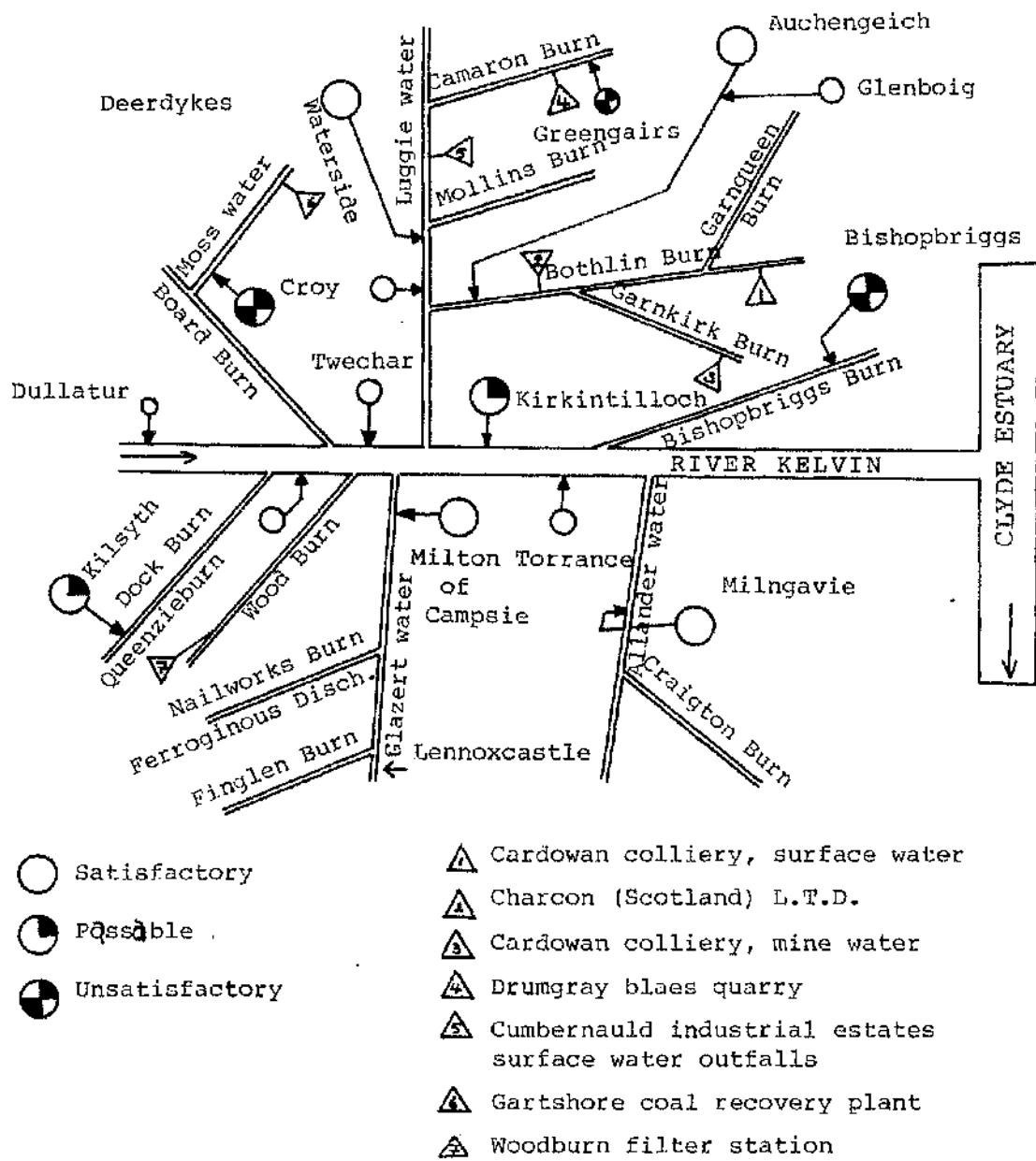


Figure 2.02: Diagram showing sources of effluents found in the River Kelvin. (Provided by CRPB).

and they were of variable water quality, and included all the burns, (Luggie Water, Glazert Water, Allander Water, Dock Water, Board Burn and Bishopbriggs Burn) which run into the Kelvin carrying sewage effluents.

Station 1 (Grid Reference 757783; River Kelvin near its source; (plate 2.03) lies upstream from Kilsyth. The river is about 4-6 feet wide and is stagnant with mats of vegetation on the sides. It flows in a south westerly direction through farmland before it reaches Auchinstarry Bridge (719770), where it receives the sewage effluent from the village of Dullatur (746773). This effluent causes a little pollution. At about 5.5 km from its source, the Kelvin is joined by the Dock Water (700764) which is seriously polluted by the effluent from Kilsyth sewage treatment works and drainage from old mine workings together with contaminated drainage from the former coke ovens site at Dumbreck. The Dock Water affects the River Kelvin downstream of the confluence. Just downstream of this point two tributaries enter the river within one mile of each other. The first is the Queenzie Burn (695762) which carries a satisfactory effluent from the village of Queenzie Burn. The second is the Board Burn (687753), a reasonably clean stream carrying a high quality effluent from Croy sewage works. The effluent from Twechar sewage works enters the river after the confluence of the Board Burn; this has no apparent pollutional effect on the river. Beyond Inchbelly Bridge (station 2; 668749; plate 2.04) the river becomes slow moving and has a canal-like appearance for approximately 4.5 km.

At Kirkintilloch the good quality water from the Glazert Water



Plate (2.03) Station 1 River Kelvin near its source



Plate (2.04) Station 2 Inchbelly Bridge

(656748) joins the Kelvin. Glazert Water is a fairly clean river which carries sewage and industrial effluents from the Universal Pulp Containers Works at Milton of Campsie and the sewage effluents originate from Lennoxtown, Lennox Castle Hospital and Milton of Campsie. This effluent has, in the past, been a major cause for concern, but a new comprehensive drainage system has solved this problem. The input from Glazert Water brings an improvement in the River Kelvin.

At the B757 road bridge (station 3; 654745; plate 2.05) which is about 100 yards from the junction of the Glazert Water with the Kelvin, zonation in the river is noticeable where the turbid water of the Kelvin has not mixed with the clean Glazert Water. Within a short distance the river is joined by the polluted Luggie Water (station 4; 653745; plate 2.06). Luggie Water originates from the town of Cumbernauld and receives effluents of borderline quality from the sewage works at Condorrat and Waterside and it has three branches. The first, Cameron Burn, carries unsatisfactory sewage effluent from the Greengairs (785705). The second, the Mollins Burn, carries ferruginous mine-water from the Bedlay Colliery (721706). The third is Bothlin Burn. At times of low rainfall more than half of the flow of the Luggie Water results from sewage effluent.

The Kelvin at Spring Farm Bridge (station 5; 650743; plate 2.07) looks turbid and flows at a fairly normal speed in a widening river. Approximately 100 yards downstream of the CRPB's gauging station at Dryfield, the river receives sewage effluent from Kirkintilloch

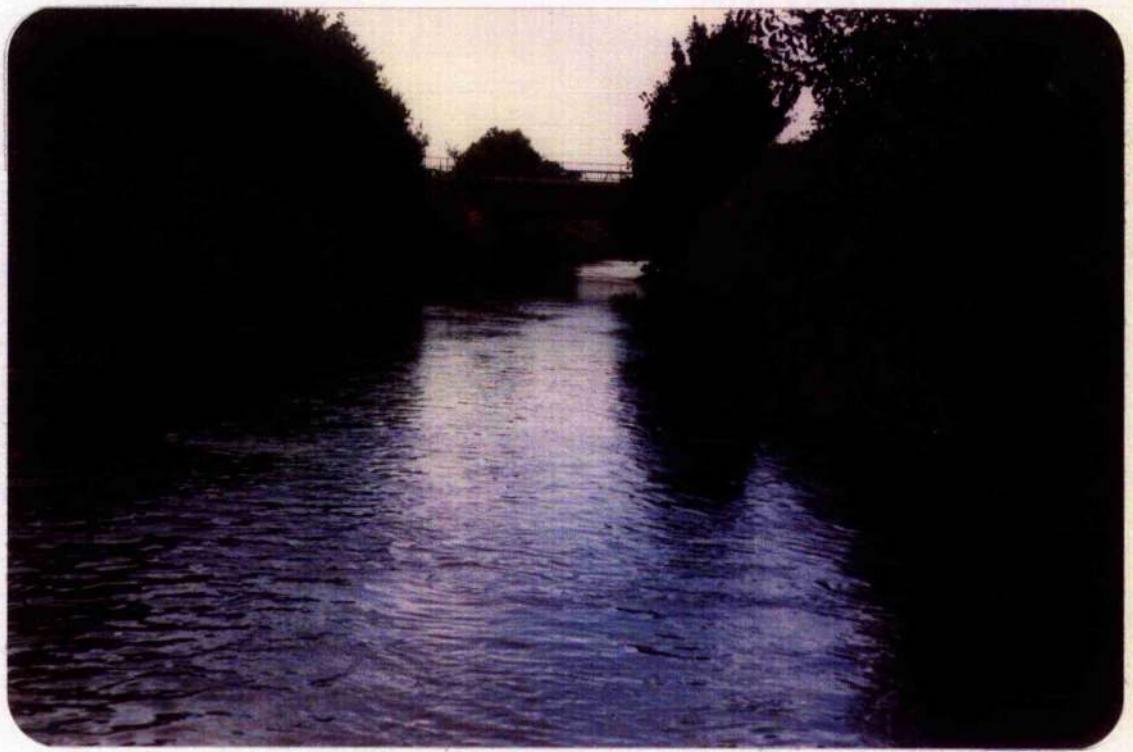


Plate (2.05) Station 3 B757 Road Bridge



Plate (2.06) Station 4 Luggie Water



Plate (2.07) Station 5 Spring Farm Bridge

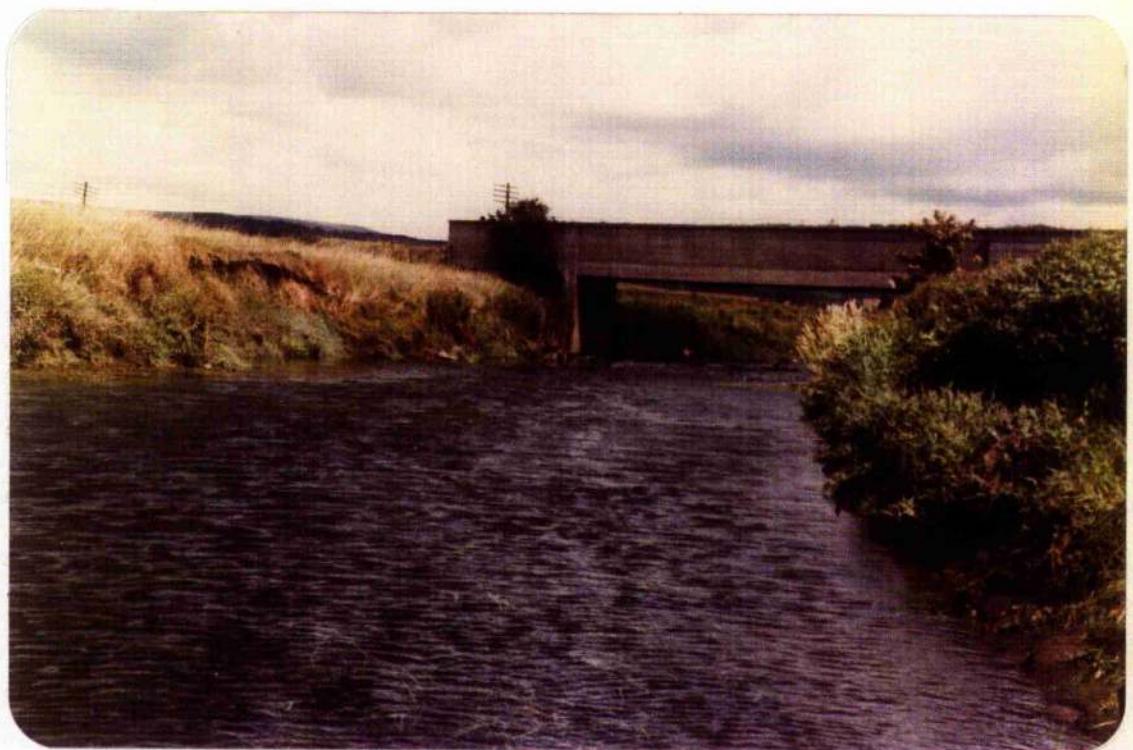


Plate (2.08) Station 6 Bardawi Bridge

sewage works. This is generally of marginal quality as it is volumetrically overloaded. Two and a half kilometers further downstream the sewage works at Torrance discharges its satisfactory effluent into the river.

At about 16 km from the source (one and a half kilometers from Torrance) the Kelvin is joined by the polluted Bishopbriggs Burn (613723), which receives the unsatisfactory and seriously polluted insufficiently diluted effluent from the Bishopbriggs sewage works. The river at Bardawie Bridge (station 6; 588729; plate 2.08) becomes slower moving and runs through farmlands. At about 18.5 km the Allander Water (station 7; 575729; plate 2.09) joins the main river. Allander Water is a fairly clean stream. In its lower reaches Milngavie sewage works discharges good quality sewage effluent. Downstream of Allander the river reaches Balmuildy Bridge (station 8; 579718; plate 2.10) with a little improvement.

From Balmuildy Bridge the Kelvin flows sluggishly through the outskirts of Glasgow to Killermont Bridge (554706) and the Garscube Estate. This estate owned by the University of Glasgow is used in part as an experimental farm. The sampling station here is at Lady Campbell Bridge (station 9; 552704; plate 2.11). Once the Kelvin leaves Garscube Estate it is faster flowing and it reaches the Kelvin Bridge (station 10; plate 2.12) at the west end of the City of Glasgow a short distance downstream from the Botanic Gardens. There are a number of small waterfalls which assist in the oxidation of the

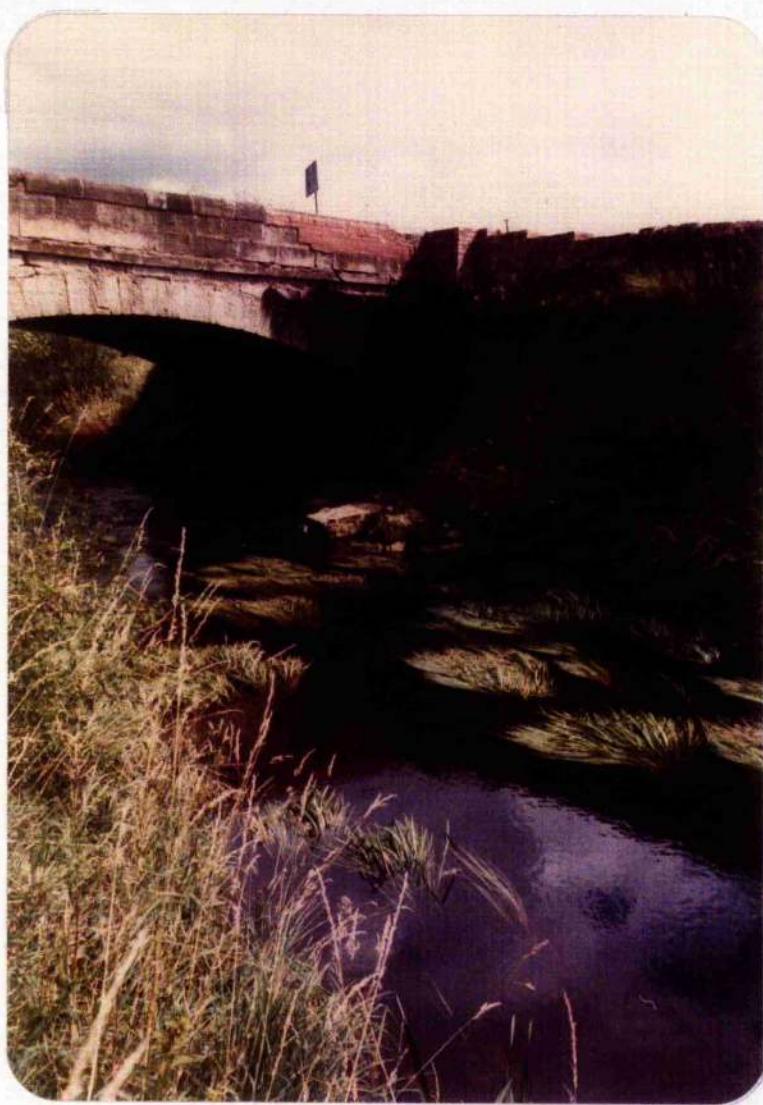


Plate (2.09) Station 7 Allander Water

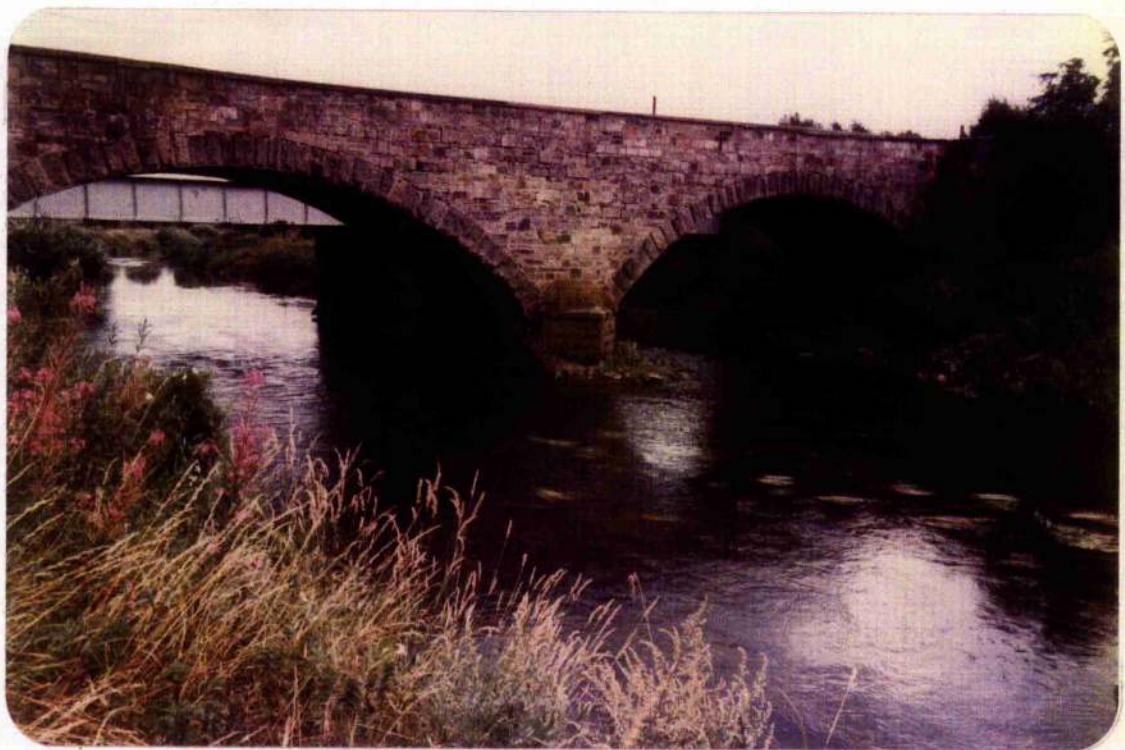


Plate (2.10) Station 8 Balmuildy Bridge



Plate (2.11) Station 9 Lady Campbell Bridge (Garscube Estate)

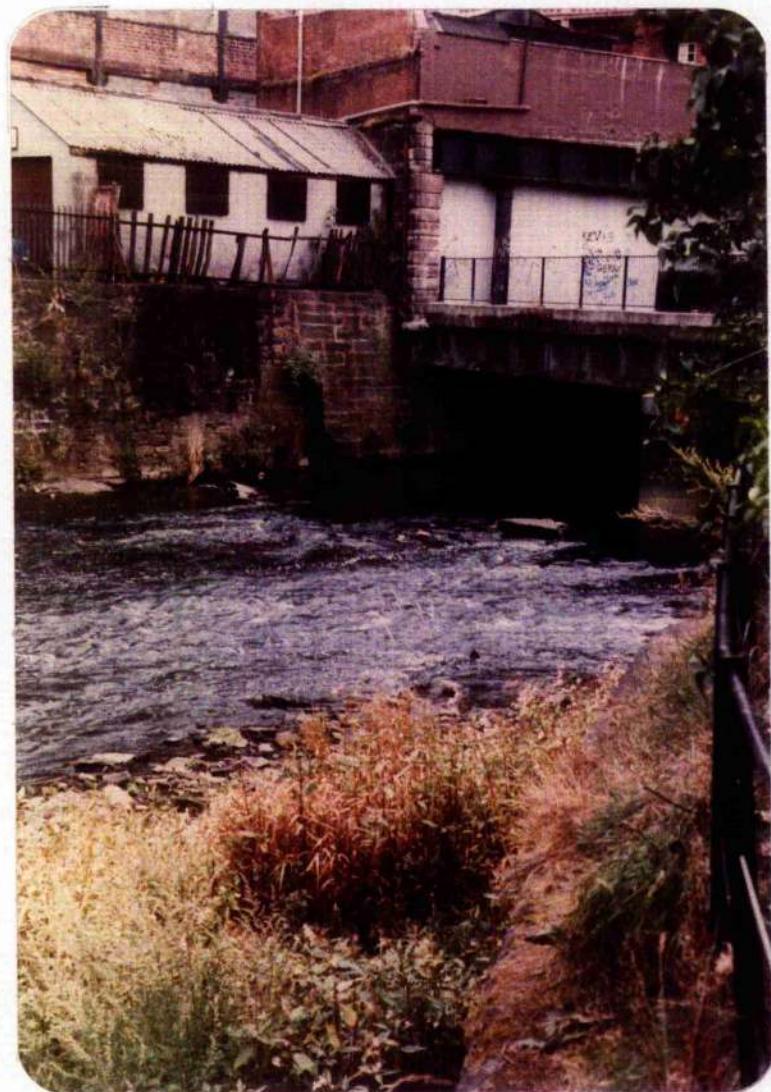


Plate (2.12) Station 10 Kelvin Bridge

the organic pollutants due to the aeration, so the quality of the water improves and the river widens considerably at the lower reaches before it joins the River Clyde.

The River Kelvin carried a large amount of sewage effluent and a moderate amount of industrial effluent and these contribute largely to the unsightly condition of the river. The river also carried a large percentage of undissolved solids possibly due in part to the boulder clay and calciferous sandstone deposits on its bed (Mulla-Ali, 1981). There is a lot of rubbish on the river bed in some sites and throughout the city (e.g. prams, bicycles, car parts, barrels, gardening implements, etc.).

During 1980 (according to the CRPB Annual Report for 1980) the poor quality of the Dock Water continued to affect a large stretch of the River Kelvin. There were no other changes affecting the water quality of the Kelvin, with the exception of improvements to the drainage system at the Dawsholm Incinerator plant. The Luggie Water showed no real change in quality although it continued to be affected frequently by the poor quality effluent from Deerdykes sewage treatment works. The Bothlin Burn showed some slight improvement in the latter part of the year as a result of a reduction in the ammoniacal load from Auchengeich sewage treatment works. The biotic indices of the river were clean (trout abundant) upstream of Kilsyth, becoming moderately polluted (trout and coarse fish) at about 8 km downstream, returning back again to clean after further 3 km until 28 km downstream and becoming moderately polluted at 30 km downstream.

In (1981) the general water quality of the River Kelvin was

unchanged from the previous year (CRPB Annual Report for 1981). Some improvements were carried out to prevent flooding of the Kilsyth sewage treatment works by the Dock Burn. Two major fish kills occurred in the Glazert Water during the year immediately downstream of the outfall from Lennox Castle sewage treatment works. Both kills were thought to have been caused by toxic constituents present in the works final effluent. However the general conditions of the river remained excellent. The biotic indices of the river were considered as clean from the source until about 18 km downstream, and for the rest of the river it was moderately polluted.

In terms of Carpenter's (1928) classification, River Kelvin falls into the "Trout beck" and "Minnow reaches" categories. The river supports a trout population for a long distance downstream. Other reaches of the river support stickleback populations. These observations are supported by the investigation by Pipe and Hunter (1976). They studied the fish population and the bioaccumulation of metals in fish muscles in the River Kelvin and Luggie Water. They recorded abundant trout populations in the river as far downstream as Torrance Bridge. Only one trout was collected in Garscube Estate. There was a downstream limit of trout population. Dock Burn was fishless due to the heavy pollution. The other polluted reach studied was the river at Twechar downstream of Dock Burn confluence. Only sticklebacks tolerated the degree of pollution and stoneloaches were absent. Similar fish distribution in the river was recorded by Soulsby in 1970.

The trout distribution in Luggie was different. Trout were

absent from the upstream sites (Sauciehall farm and waterside area), whilst stickleback populations were present. There were plentiful trout at the lowest site about 100 metres from the Luggie-Kelvin junction. It was possible that trout had moved into the Luggie from the Kelvin. Heavy metal concentrations Zn, Cu, Cr were determined in trout muscles. It was found that the metal concentrations were dependent on the weight of the fish rather than the area from which they were collected. The reason for this was unknown.

Earlier records, mostly of chemical data on the River Kelvin system, have been kept by the Clyde River Purification Board (CRPB) and were given in their annual reports up to date. Apart from records on fish, the data collected were purely physical and chemical. These reports all indicate that the River Kelvin, on the whole, was considered to be clean at its source to moderately polluted and polluted downstream. These observations are described in detail in the next section of this thesis.

There has not been any detailed vegetational study of the river. Mulla-Ali (1981) used river water isolates of bacteria for the assessment of water pollution by a short time (3-5 minutes) oxygen polarographic method. He tested the river water samples and sewage samples from different stages of the treatment process. He found that higher levels of oxygen uptake occurred higher BOD_5 values. His results showed a significant relationship between the results of the rapid microbiological test and BOD_5 determination for river water samples, and a highly significant relationship for the results

of the sewage samples and the standard substrate solution.

To assess water quality, the algal association in the water may be used as long as the indicative value is known of the association identified. Brinley (1942); Patrick (1963) and Fjerdningstad (1964) have expressed views on the indicative value of algal communities. They erected saprobic systems on the basis of algal and bacterial communities by placing the organisms into different groups according to their capability of subsisting and reproducing in waters having different contents of nutrients. Fjerdningstad (1964) assessed the relation of the species to pollution on the basis of the value of the saprobic valency of the organisms. Several studies, Al-Saadi *et al.* (1979), considered that the degree of water pollution was reflected by the algal flora.

In the present investigation a qualitative and quantitative study of the seasonal variation of both the phytoplankton and the epiphytic algal population in the River Kelvin was made from near its source to near its junction with the River Clyde from February 1980 to September 1982. The effect of pollution on the algal population of the main river and its tributaries was studied by investigating both the physico-chemical and biological aspects of the river.

CHAPTER THREE

3. MATERIALS AND METHODS

3.1 Sampling Methods:

Sampling at various stations in the River Kelvin was carried out once a month for chemical analysis and phytoplankton enumeration.

Samples were collected using a 10 litre bucket which had been cleaned previously by washing it with 10-15% Decon 90 and rinsed several times with distilled water. The bucket was lowered down, by means of a long rope, to the middle region of the river from the bridges at each station. The river water was allowed to pass through it for a while. This operation was used instead of rinsing the bucket with river water; then it was raised up filled with the water sample. Water temperature had been taken immediately by placing a clean thermometer inside the bucket, which was rinsed with distilled water after every use. A 5 litre clean polythene container was rinsed with the water sample then filled and stoppered. Dissolved oxygen and BOD_5 sampling bottles (250 ml) (clear glass for oxygen and amber for BOD with narrow-neck and ground glass stoppers), were numbered on two different places for sample identification. The bottles were placed in the bottom of the bucket, which contained the sample, and were left to fill with the sample making sure that all bubbles were eliminated. The glass stoppers were dropped into the necks of the bottles under the water surface so that excess water was displaced without air bubbles formation.

3.2 Cleaning of Glassware

All the glassware and the bottles used in the analytical processes was cleaned properly after every analysis by soaking

overnight in 5-10% of Decon 90, then rinsed several times and filled with distilled water until the next analysis. Spectrophotometer cells were treated in the same way and they were left in ethanol inside a beaker. Glassware used for ammonia determination was given extra treatment, being rinsed twice and filled with ammonia free distilled water.

3.3 Chemical Analysis

3.3.1 Determination of reactive phosphorus:

All methods for the estimation of dissolved phosphorus are colorimetric and rely on the formation of a phosphomolybdate complex and its subsequent reduction to a coloured blue compound.

The procedure used was that of Murphy and Riley (1962) which is a rapid and easy analysis as described by Strickland and Parsons (1972). All reagents used were of analytical quality.

Samples were analysed within the first 2 hours of completion of sampling and they were not preserved in any way. The following special reagents were required for the analysis and were prepared as follows:

Ammonium molybdate solution:

15.0 gm of finely crystalline, analytical reagent quality ammonium paramolybdate (NH_4)₆ Mo_7O_{24} .4H₂O were dissolved in 500 ml of distilled water. This solution was made up freshly one night before the analysis.

Sulphuric acid solution:

140 ml of concentrated sulphuric acid (sp.gr. 1.82) was added cautiously to 900 ml of distilled water. After

cooling this solution was stored in a glass bottle and it was stable indefinitely.

Ascorbic acid solution:

27 gm of good quality ascorbic acid was dissolved in 500 ml distilled water. This solution was prepared in small quantities freshly before each analysis.

Potassium antimonyl tartarate solution:

0.34 gm of good quality potassium antimonyl tartarate were dissolved in 250 ml of distilled water. This solution was stored in a polythene bottle and was renewed every three months.

Mixed reagent:

500 ml of mixed reagent was prepared by mixing 100 ml ammonium molybdate, 250 ml sulphuric acid, 100 ml ascorbic acid and 50 ml potassium antimonyl tartarate solutions. This reagent was always prepared for immediate use and the excess was discarded.

Fifty ml of each sample were measured into 50 ml measuring cylinder (2 replicates per sample) which had been rinsed with the sample to be analysed. These measuring cylinders were transferred to a water bath maintained between 15-25°C for 20 minutes. Five ml of freshly-prepared mixed reagent was pipetted to each 50 ml sample and mixed thoroughly. The extinction of the solution was measured after 10-15 minutes in a Unicam 4 cm glass cell at a wavelength of 855 °A using red filter. A Unicam SP 600 spectrophotometer was

always used. Cell to cell blanks were determined by filling all 4 cm cells with distilled water, the deviation from the reference cell was noted. The reagent blank was estimated for each set of determinations using 50 ml of distilled water.

The procedure was calibrated using a standard phosphate solution of potassium dihydrogen phosphate. 0.816 gm of A.R. anhydrous potassium dihydrogen phosphate (KH_2PO_4) was dissolved in 1,000 ml of distilled water; 1 ml of this solution contained 6.0 μg at P.L.⁻¹. This standard was stored in a dark bottle with 1 ml of chloroform as preservative and for surety it was renewed every three months. Five ml of this concentrated solution was diluted to 500 ml with distilled water in a volumetric flask and used immediately. Four standards were prepared consisting of 5 ml of this dilute phosphate solution, which is equivalent to 3 μg at P.L.⁻¹, made to a volume of exactly 100 ml with distilled water in a 100 ml graduated measuring cylinder. This standard solution was carried through the phosphate determination with the samples and blanks.

The calibration factor (F) was calculated from the following expression:

$$F = \frac{3.0}{E_{\text{standard}} - E_{\text{blank}}}$$

where E_{standard} was the mean extinction of 4 standards and E_{blank} was the mean extinction of the blank.

The phosphate phosphorus of the sample in μg at P.L.⁻¹ were then determined by multiplying the factor F by the corrected extinction in the two aliquots from each sample. This factor is fairly constant, around 12 for a 4 cm cell.

3.3.2 Determination of reactive silicate:

Soluble silica (dissolved silica) or silicate is almost always determined by a colorimetric method depending on the production of a silicomolybdate complex. The yellow colour of this complex may be measured directly, but this is less sensitive and satisfactory than the method which reduces the yellow colour to a heteropoly intense blue colour.

The procedure used in this investigation is a modification of the method of Mullin and Riley (1955) as described by Strickland and Parsons (1972).

Samples for the analysis were returned back to the laboratory in polythene containers within 2-2.5 hours of the completion of sampling.

The following special reagents were required for the analysis:

Molybdate reagent:

Four gm of analytical reagent quality ammonium molybdate $(\text{NH}_4)_6\text{Mo}_7\text{O}_{24} \cdot 4\text{H}_2\text{O}$ (finely crystalline) was dissolved in 300 ml distilled water. 12 ml of concentrated hydrochloric acid (sp.gr. 1.18) was added with mixing then the solution was made up to 500 ml with distilled water. This reagent was freshly prepared every month.

Metol-sulphite solution:

Three gm of AR anhydrous sodium sulphite (Na_2SO_3) was dissolved in 250 ml distilled water followed by the addition of 5 gm of metol (P-methylaminophenol sulphate) when the metol was dissolved the solution was then

filtered through No. 1 Whatman filter paper and kept in a clean glass bottle which was tightly stoppered. This solution tended to deteriorate rapidly and was freshly prepared before every analysis.

Oxalic acid solution:

50 gm of AR oxalic acid dihydrate, $(COOH)_2 \cdot 2H_2O$, were shaken with 500 ml of distilled water. This produced a saturated solution which was decanted from the crystals before use. This solution was stable indefinitely and was prepared every three months.

Sulphuric acid solution:

A 50% v/v solution was prepared by adding 250 ml concentrated (sp.gr. 1.82) AR sulphuric acid cautiously to 250 ml of distilled water. This was cooled to room temperature and the volume made up to 500 ml with distilled water. This solution was prepared in quantities of 1 litre.

Reducing reagent:

150 ml of metol-sulphite solution were mixed with 90 ml of oxalic acid solution, 90 ml of 50% sulphuric acid were then added slowly with mixing and the volume was made up to 950 ml with distilled water. The reagent was used immediately after preparation.

Twenty-five ml aliquots of each sample were dispensed into 2 replicate narrow-necked polythene bottles, which had been previously washed with the sample to be analysed. The bottles were then placed

in a water bath maintained between 18-25°C for 30 minutes to reach a constant temperature.

Ten ml of molybdate solution was then added to each sample and mixed thoroughly. The mixture was allowed to stand for 10 minutes. Fifteen ml of the reducing agent was then added rapidly and the solution was mixed immediately. The solution was allowed to stand for 3-4 hours to allow complete reduction of the silico-molybdate complex. The extinction was measured in a 1 cm Unicam glass cell against distilled water. A Unicam SP 600 spectrophotometer was used with a red filter at a wavelength of 810 Å. Cell to cell blanks were determined by filling all 1 cm cells with distilled water and measuring the deviation from zero. All extinctions were corrected for cell to cell blanks and reagent blanks.

Reagent blanks accompanied each set of determinations two 25 ml aliquots of distilled water, which was kept in polythene containers, being carried through the analysis procedure. Calibration standards were also included with every set of analysis.

Standard silicate solution: dried AR sodium silicofluoride Na_2SiF_6 forms a convenient standard, Strickland and Parsons (1972). 0.960 gm of fine powder were dissolved in 100 ml distilled water in a plastic beaker. The solution was then transferred to a 1,000 ml volumetric flask and made up to the mark with distilled water; this solution was rapidly transferred to a polythene bottle for storage, as it rapidly picks up silica from glass. The solution was stable indefinitely, which was an advantage over most standards consisting of sodium silicate.

One ml of this solution contained 5 µg at Si; 5 ml of this concentrated solution was diluted to 250 ml with distilled water to give the final concentration of 4 µg at Si. L^{-1} . This diluted solution was prepared for immediate use. Three 25 ml aliquots of this standard silicate solution were carried through the experimental procedure with the samples and blanks.

The calibration factor, F was calculated using the following expression

$$F = \frac{100}{E_{\text{standard}} - E_{\text{blank}}}$$

where E_{standard} was the mean extinction of 3 standards and E_{blank} was the mean extinction of 2 blanks. The value should be close to 100.

The silicate silicon concentration in µg at Si. L^{-1} of any sample can then be determined by multiplying the factor, F by the corrected mean extinction of the two aliquots from each sample.

3.3.3 Determination of nitrate:

A colorimetric method was used in the present investigation for nitrate determination based on the reduction of nitrate almost quantitatively to nitrite by passing the sample through a column containing cadmium filings coated with metallic copper, based on the investigations published by Wood, Armstrong and Richards (1967), and described by Strickland and Parsons (1972). The nitrite produced was determined by diazotizing with sulphanilamide and coupling with N-(1-naphthyl)-ethelenediamine to form a highly coloured azo dye.

Apparatus:

The reducing column Fig. (3.01) was prepared by joining three pieces of glass tubing end-to-end. A 10 cm length of tubing, 5 mm inside diameter, was joined to a length of 30 cm tubing, 10 mm inside diameter (which contained the metal filings) which in turn was joined to a 35 cm tube, 2 mm in diameter. The last tube was bent just below this join into a U-shape so that it ran parallel to the 10 mm diameter tube and then its end was bent over to form an inverted U-siphon.

Preparation of the reducing column:

- a) Coarse cadmium filings were produced from a stick of cadmium and sifted to obtain the fraction which passed through a 2 mm screen but were retained by a 0.5 mm screen.
- b) Sixty gm of filings were stirred with 500 ml of 2% w/v solution of copper sulphate pentahydrate, $\text{CuSO}_4 \cdot 5\text{H}_2\text{O}$, until the blue colour had left the solution and the semicolloidal copper particles began to enter the supernatant liquid.
- c) A small plug of glass wool was placed at the bottom of the column which was then filled with the supernatant liquor from the preparation of the cadmium-copper above.
- d) A sufficient amount of the cadmium-copper mixture was poured in to produce a column about 25-30 cm in length, tapping the column firmly was necessary to settle the filings.
- e) The column was washed thoroughly with dilute ammonium chloride solution. The flow rate was fixed so that 100 ml

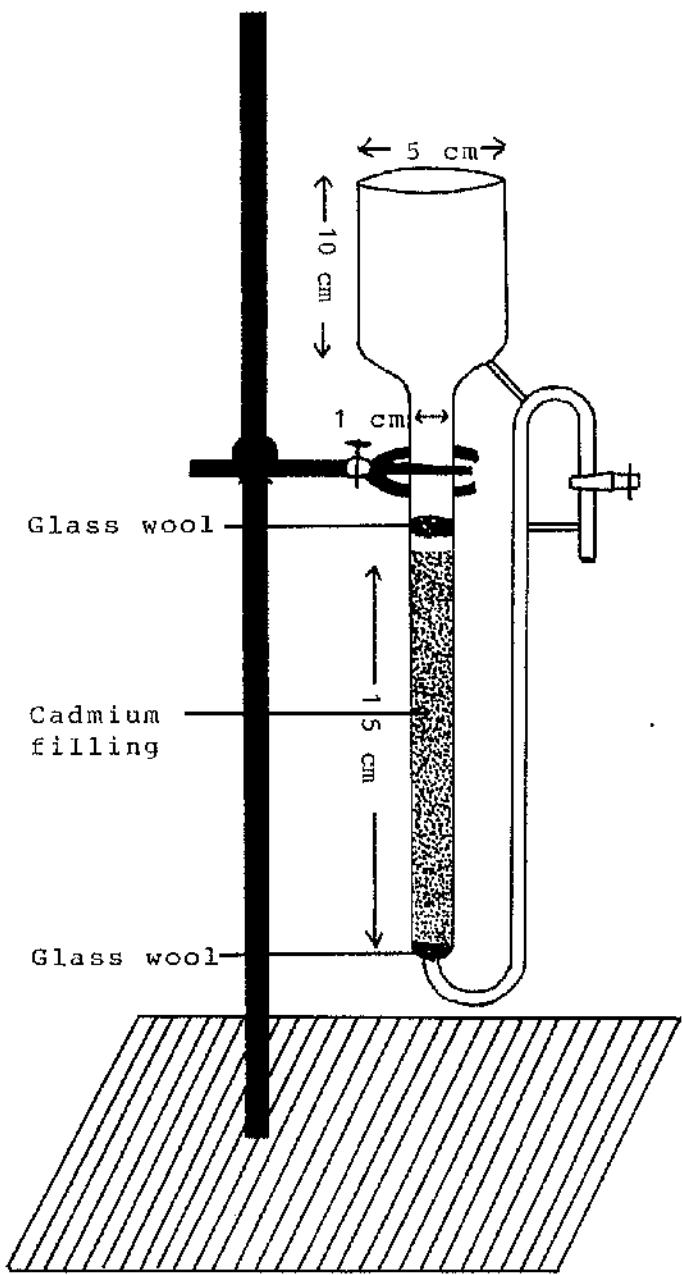


Figure 3.01 Cadmium-copper column used for reducing nitrate to nitrite.

of the solution needed between 8-12 minutes to flow completely through the column. This was controlled by loosening or packing the glass wool at the bottom of the column.

- f) A small plug of glass wool was placed on the top of the column to prevent cadmium filings being washed into the top chamber when solutions were added to the column.
- g) The column was covered with dilute ammonium chloride solution when not in use.
- h) When the efficiency of the column was suspect (usually after passing about 100 samples through it) the column was repacked by emptying its contents into a beaker. The filings were washed vigorously twice with 500 ml of 5% v/v hydrochloric acid solution. The acid was decanted and the metal was washed with 300-500 ml of distilled water until the water was no longer acid ($\text{pH} > 5$). The liquid was decanted and the metal was left as dry as possible. It was then treated with the copper-sulphate solution as described above.

Special reagents:

- a) Concentrated ammonium chloride solution: 125 gm of analytical reagent quality of ammonium chloride were dissolved in 500 ml of distilled water and stored in a glass bottle.
- b) Dilute ammonium chloride solution: 25 ml of the

concentrated ammonium chloride solution was diluted to 1,000 ml with distilled water. This solution was stored in a glass bottle.

c) Sulphanilamide solution: 5 gm of sulphanilamide was dissolved in a mixture of 50 ml of concentrated hydrochloric acid (sp.gr. 1.18) and about 300 ml of distilled water. The volume was made up to 500 ml with water. This solution was stable and it was renewed every 3 months.

d) N-(1-Naphthyl)-Ethylenediamine dihydrochloride solution: 0.5 gm of the dihydrochloride was dissolved in 500 ml of distilled water. The solution was stored in a dark bottle and renewed every month.

Procedure:

Fifty ml of aliquot sample was placed in a graduated measuring cylinder, two replicates of each sample were taken. Two ml of concentrated ammonium chloride solution were added to each sample and shaken. Any excess sample or column wash was removed from the column reservoir (using a short plastic tubing which was fixed on the top of a disposable syringe) to about 0.5 cm above the cadmium and about 10 ml portion of sample was added to rinse out the previous sample which was in the column. The remainder of the sample was then added and the graduated cylinder was placed under the discharge tap. Fifteen ml of the sample was allowed to pass through the column and was used for rinsing the cylinder. The next

15 ml of reduced sample was collected and diluted to 30 ml with distilled water. The sample was then treated as a nitrite sample. For a full description of the method see nitrite determination.

Extinctions were determined within half an hour using 1 cm Unicam glass cell at a wavelength of 543 A° using blue filter. A Unicam SP 600 spectrophotometer was used throughout the investigation. When the red colour of the final solution was very high, in other words when the nitrate concentration was high and the extinction was out of the spectrophotometer's range the solution was diluted 10 times by pipetting 5 ml of it into a 50 ml volumetric flask and making it up to the mark with distilled water and mixed thoroughly. The resultant extinction was multiplied by 10.

Each series of determinations was accompanied by three standards and 2 blanks of distilled water. A nitrate standard was prepared by dissolving 1.02 gm of analytical reagent quality of potassium nitrate, KNO₃, in 1,000 ml of distilled water (1 ml = 10 µg at N.). This solution was stable indefinitely in the absence of evaporation. Two ml of this concentrated solution was diluted to 1,000 ml with distilled water in a volumetric flask. This was prepared freshly before use. The concentration was 20 µg at N.l⁻¹. Three 50 mls of this diluted solution were put in measuring cylinders and carried through the complete procedure. The calibration factor, F was determined by the calculation of the expression.

$$F = \frac{20}{E_{\text{standard}} - E_{\text{blank}}}$$

The concentration of nitrate-nitrogen and nitrite-nitrogen in any sample in $\mu\text{g at N.l}^{-1}$ can be determined by multiplying the corrected mean extinction of the sample by the calibration factor F. The value of F was near to 25.

3.3.4 Determination of nitrite:

All nitrite estimations are based on a diazotization process. Under acid conditions the nitrite ions react with an aromatic amine (R-NH_2) to form a diazo compound which is coupled with a second aromatic amine (Ar-NH_2) to form a red azo dye, Martin and Goff (1972). The intensity of the final colour was proportional to the amount of nitrite present.

Samples for nitrite were determined within 3 hours of completion of sampling. The following special reagents were required for the analysis:

Sulphanilamide solution:

See nitrate determination.

N-(1-Naphthyl)-ethylenediamine dihydrochloride solution:

See nitrate determination.

Procedure:

Fifty ml aliquot of each sample was measured into a 100 ml conical flask which had been rinsed with the sample. 1.0 ml of sulphanilamide was added, using a disposable 1 ml syringe, followed by mixing. After 2-3 minutes 1.0 ml of naphthylene ethylene diamine solution was added and mixed immediately.

After 20-30 minutes the extinction of the solution was measured in a 4 cm Unicam glass cell again with distilled

water at a wavelength of 543 Å°. Samples with very high concentrations were diluted 10 times before reading the extinction. It was then corrected by subtracting the mean distilled water reagent blank (E_{blank}) which was run in duplicate.

The concentration of nitrite-nitrogen in µg at N.λ⁻¹ was calculated by multiplying the calibration factor, F, by the corrected extinction for each sample.

The standard nitrite solution was prepared by dissolving 0.345 gm of anhydrous analytical reagent quality sodium nitrite, NaNO₂, which was dried at 110°C for one hour in 1 litre of distilled water. This solution was stored in a dark bottle with 1 ml of chloroform as a preservative; it was renewed every 2 months (1 ml of this solution contained 5 µg at N.). Five ml of this concentrated solution was diluted to 500 ml with distilled water for immediate use. Four standards were prepared by pipetting 2 ml of the dilute solution into a 50 ml measuring cylinder. The volume was made up to 50 and the standards were transferred to 4 dry Erlenmyer flasks and 2 more flasks were prepared with 50 ml of distilled water in each to act as blanks. These were carried through the analysis procedure, and the mean extinction ($E_{standard}$) was found. The calibration factor for the determination was calculated from the expression:

$$F = \frac{2.0}{E_{standard} - E_{blank}}$$

the value for F was very close to 2.0.

3.3.5 Determination of ammonia:

The method used in the present study was the one used by Chaney and Marbach (1962) as it was described by Mackereth, Heron and Talling (1978). It was based on reacting ammonia ($\text{NH}_3 + \text{NH}_4^+ - \text{N}$) with phenol and hypochlorite in an alkaline solution to form indophenol blue; the reaction was catalysed by nitroprusside. The ammonia present in the sample was proportional to the resulting absorbance.

All the reagents were made up in ammonia free distilled water which was prepared by passing the water through a cation exchange column which was prepared as follows:

A glass tube of about 2.5 cm in diameter and 30 cm in length of which one end was drawn down to a diameter of about 1 cm and fitted with a rubber tubing closed with a screw clip. A small plug of glass wool was inserted into the drawn-out section of the tube, which was supported with the rubber tube at the bottom.

A mixture of exchange resin (Amberlite IR-120) and distilled water was poured in at the top of the tube; the resin settled to form a compact column until 5 cm from the top was left empty. Care was taken for the resin to be always covered with water.

The top of the tube was closed with a rubber bung through which a short glass tube was passed and connected by a short rubber tubing to a 5 litre polythene container.

Four litres of 2 N Hydrochloric acid were placed in

the container and was allowed to pass through the column at a flow rate of 20 ml/minute. This process ensured that the resin was converted completely to the hydrogen form.

The column was washed by allowing 4-5 litres of distilled water and care was taken to ensure that the column did not run dry at any time.

Special reagents:

a) Phenol-nitroprusside reagent: 15.0 gm of phenol and 0.015 gm of sodium nitroprusside (added as 1 ml of 1.5% w/v aqueous solution, freshly prepared) was dissolved in 500 ml of water. This solution was kept in a dark bottle in the refrigerator and renewed every 3 months.

b) Alkaline hypochlorite solution: 10 gm of sodium hydroxide was dissolved in 400 ml water and the solution cooled. 2.5 ml of 3.0 N hypochlorite solution was added and the mixture was made up to 500 ml. This solution was kept in the refrigerator and renewed every 3 months.

The normality of the hypochlorite solution was checked each time before preparing the reagent. An acidified dilution containing iodine (e.g. 25 ml water + 2 gm kI, dissolved, + 10 ml glacial acetic acid + 5 ml hypochlorite solution diluted x 10) was titrated with 0.1 N sodium thiosulphate solution, using starch as indicator (see dissolved oxygen determination).

c) Standard ammonium chloride solution: 3.821 gm of

NH_4Cl was dissolved in distilled water and made up to 1 litre. One ml of this standard solution contained 1 mg $\text{NH}_4^+ \text{-N}$.

Procedure:

As a result of the high concentration of the ammonia which was expected to be in the samples, all water samples were diluted 10 times before the determination; i.e. simply by pipetting 10 ml of the sample to a clean 100 ml volumetric flask and made up to the mark with ammonia free water and mixed thoroughly. Twenty ml of this diluted sample was pipetted into a 50 ml volumetric flask which had been rinsed twice with the sample to be analysed.

Duplicates of each sample were analysed. Two ml of phenol nitroprusside was added using a pipette and mixed followed by the addition of 2 ml of alkaline hypochlorite reagent.

The volume was made up to 50 with ammonia free distilled water. The flasks were stoppered and transferred to a water bath which maintained a 25°C temperature and were protected from direct strong light by covering the water bath with aluminium foil. The samples were allowed to stand for one hour to allow complete colour formation.

The absorbance of the solution was measured at 635 A° using 1 cm Unicam glass cells again with ammonia free distilled water blank which was similarly prepared.

A calibration standard curve (Fig. 3.02) was prepared with every set of analysis to determine the mean factor relating absorbance to

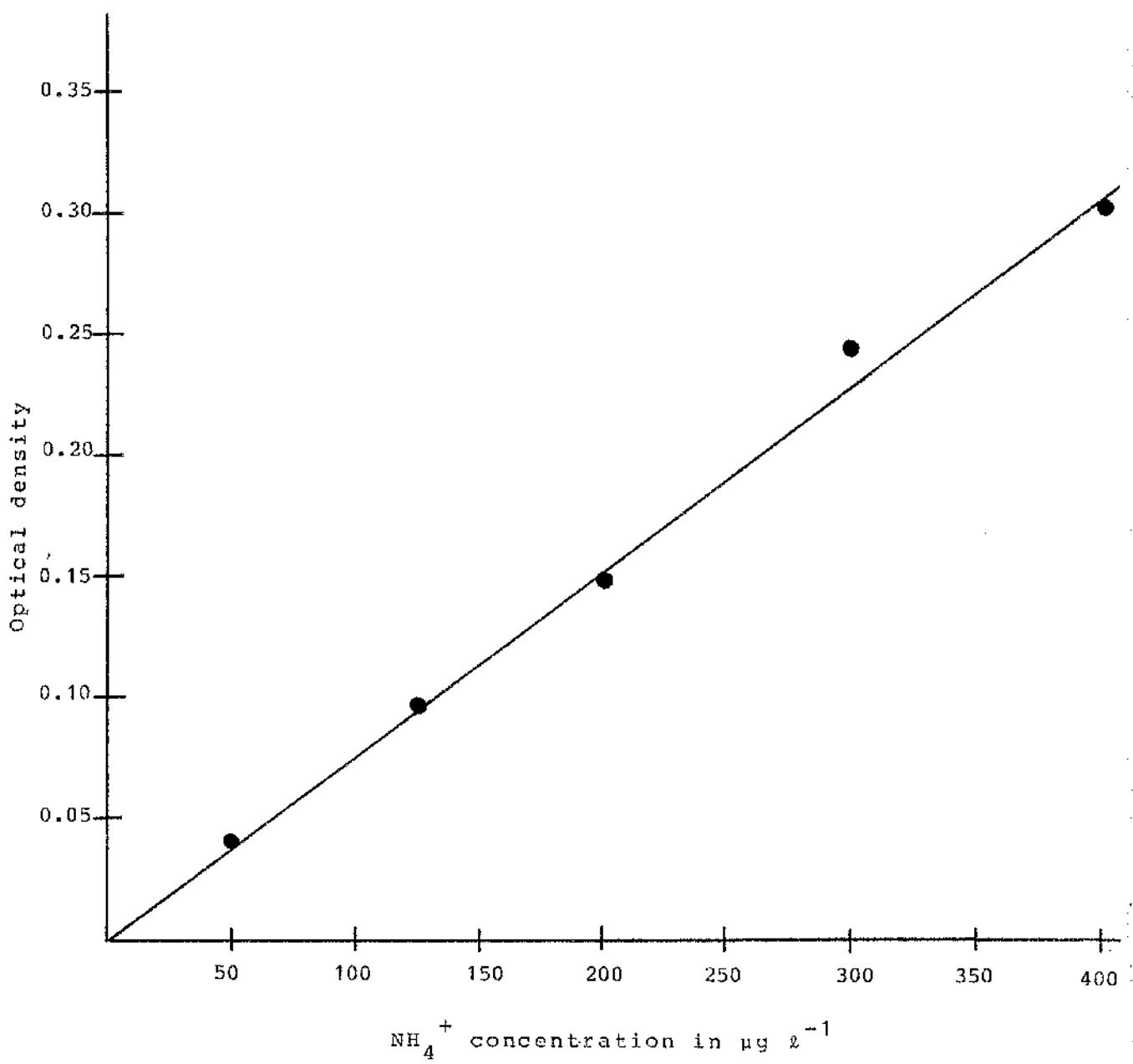


Figure 3.02: A calibration standard curve relating the NH_4^+ concentration with optical density.

concentration. The graph was prepared by reading the absorbance for a set of different concentrations, standard solutions which were made up from the standard ammonium chloride solution. The concentrations used were 50, 100, 200, 300 and 400 g $\text{NH}_4^+ \text{-N}^{-1}$.

The ammonia in the samples was determined by reading the concentration against the absorbance on the graph and multiplying the result by 20 (during the procedure, samples were diluted 20 times).

3.3.6 pH determination:

This was checked immediately after the completion of sampling. About 200 ml of each sample was placed in Erlenmeyer flasks which were rinsed with the sample. The flask mouth was covered with a piece of foil and allowed to stand in the room temperature to warm up. The pH was then determined using a pH meter of the Electronic Instruments Ltd., model 7020.

The electrode was standardized using commercially available pH buffer tablets having pH values of 4, 7, and 9.4.

The electrode was washed with distilled water and dried with a tissue before immersion in any sample. Two-three minutes were allowed for equilibration once the electrode was immersed in the sample. The pH was then read directly.

3.3.7 Dissolved oxygen and Biological Oxygen Demand (BOD₅) :

The principle of Winkler's method (1888) is that when concentrated solution of divalent manganese and alkaline potassium iodide are added to the water sample, white manganous hydroxide is first formed and then oxidised to manganic hydroxide by the molecularly dissolved oxygen. The brown manganic hydroxide settles to the bottom of the bottle. Sulphuric acid is then added and this dissolves the manganic hydroxide and iodine is liberated. The liberated iodine takes up the excess iodide and forms I₃. This solution is then titrated against thiosulphate and the end-point of the titration is indicated by a starch solution. This method had been used for purer clean waters which contains no iron, nitrites or organic matter. Waters with high concentrations of the substances mentioned gave significant errors. A Rideal-Stewart modification of Winkler's method (Rideal and Stewart, 1901) was advised to be used as a regular procedure for all kinds of water. The only disadvantage of the method was the use of additional steps in the analysis.

The method applied in this investigation was according to the approved methods for physical and chemical examination of water (Institution of Water Engineers, 1960).

Reagents:

- a) Manganous sulphate: 250 gm of MnSO₄.4H₂O A.R. was dissolved in distilled water and made up to 500 ml. This solution was kept in a stoppered glass bottle.
- b) Alkaline Iodide: 350 gm of potassium hydroxide A.R. (KOH) and 75 gm of potassium iodide A.R. (KI) were

dissolved in distilled water and made up to 500 ml. This solution was cooled and stored in a glass bottle.

c) Concentrated sulphuric acid (sp.gr. 1.82).

d) Sodium thiosulphate: 3.1025 gm of $\text{Na}_2\text{S}_2\text{O}_3$ was dissolved in freshly boiled distilled water and made up to 1 litre in a volumetric flask (N/80) and stored in an amber bottle and renewed every month.

e) Sodium oxalate solution: 2.0 gm of $(\text{COONa})_2$ A.R. was dissolved in 100 ml of distilled water and stored in a glass bottle. The solution was made up frequently due to its quick deterioration.

f) Potassium permanganate solution: 1.9755 gm of KMnO_4 A.R. was dissolved in 50 ml distilled water and the volume was made up to 100 (N/8). This was stored in a glass bottle.

g) Potassium fluoride solution: this solution was prepared by dissolving 10 gm of KF in 100 ml distilled water and stored in a glass bottle.

h) Starch solution (Indicator): for preparing this solution 2 gm potato starch was suspended in a mixture of 400 ml of distilled water and 30 ml of 20% potassium hydroxide. The solution was stirred until almost clear, and allowed to stand for an hour. Hydrochloric acid was used for neutralizing the solution using litmus paper. One ml of glacial acetic acid was added for preservation and stored in a glass bottle.

Procedure:

Samples were returned to the laboratory in the white and amber bottles. The oxygen in the white bottles was fixed

immediately for determination of the initial oxygen concentration, while the amber bottles were incubated, at room temperature and complete darkness, for five complete days then they were analysed for the amount of oxygen left in them (final oxygen).

Long narrow volumetric pipettes were used for adding the reagents.

- a) The stopper was removed from the bottle and exactly 0.7 ml of concentrated H_2SO_4 was added just below the surface of the water. The stopper was placed on again and the bottle inverted a few times for mixing.
- b) Two ml of potassium fluoride was added and mixed thoroughly.
- c) Two-three drops of potassium permanganate was added and mixed by repeated inverting of the bottle. A violet colour was formed and the bottles were kept for 5 minutes until the colour had either partly gone or disappeared completely.
- d) One ml of potassium oxalate was added. The bottles were kept for 10 minutes after mixing, until the excess violet colour had completely disappeared.
- e) One ml of manganous sulphate and 4 ml of alkaline iodide was added to the bottle. The samples were shaken vigorously for 0.5-1 minute and left to allow the precipitate to settle down.
- f) 1.5 ml of concentrated sulphuric acid was added and

mixed well by inverting the bottle several times until all the precipitate was dissolved and the solution was clear.

g) Using a clean measuring cylinder 100 ml of the sample, in which the oxygen was fixed, was transferred to a clean and dry Erlenmeyer flask. This was titrated rapidly against sodium thiosulphate from a 50 ml volumetric burette. 0.5 ml of the starch solution was added to each sample to act as an indicator.

The results were calculated directly by the volume of sodium thiosulphate used in the titration. The number of mls were numerically equal to the dissolved oxygen content in parts per million (or mg l^{-1}) and no additional calculation was necessary.

For BOD_5 , the samples were analysed after 5 days incubation to determine the final concentration of oxygen. Subtracting the final concentration from the initial concentration of oxygen gave the value for BOD_5 .

The BOD_5 is the amount of oxygen which is consumed by the micro-organisms in a water sample within a limited period which is usually five days. This test has been widely used by limnologists for monitoring water pollution. It relates the number of organisms, which is proportional to the amount of nutrients in a water sample, with the amount of oxygen which is consumed by them; certainly greater numbers of organisms consume higher amounts of oxygen.

3.3.8 Determination of the heavy metals Zinc (Zn), Lead (Pb) and Cadmium (Cd)

The common method for determining heavy metals in water samples is using the atomic absorption spectrophotometer which is based on the direct aspiration of the sample into an air-acetylene flame, APHA (1976). This method has been used in the present study.

Samples were brought back to the laboratory in the polythene containers. Two hundred and fifty ml of each sample was filtered through 0.45 μ HA Millipore membrane filter and put into clean and dry polythene bottles of 250 ml capacity. Samples were acidified by adding 1 ml of concentrated Nitric acid (HNO_3) to minimize absorption of the metals on the container walls.

Samples were stored in -20°C for 2-3 days, then they were thawed and read directly on the atomic absorption spectrophotometer (A.A), (Perkin Elmer Model 306). Spectrosol standard solutions from BDH Chemicals for the metals were used for setting the instrument to a known concentration.

The figure which was shown on the screen was read as ppm of "dissolved" metal in the sample.

Finally the bottles were washed thoroughly and rinsed with 1 + 1 nitric acid then with distilled water before re-use.

3.4 Pigment Analysis

Determination of chlorophyll α and the phaeopigments:

A spectrophotometric method for chlorophyll α estimation was given by Richard and Thompson (1952) as modified by Parsons and Strickland (1963) and described in Strickland and Parsons (1972).

The procedure which was used in this investigation was for measuring the total quantity of chlorophyll α and phaeophytin α . However measuring the amount of non-active chlorophyll α in terms of the quantity and phaeopigments was often enough for a routine observation. Moss (1967) and Lorenzen (1967) have described two similar procedures for this determination. The same were used in this study. The equations employed were the ones which were given by Lorenzen (1967). The method was based on the extraction in 90% acetone and the extinction of the extract was measured before and after treatment with dilute acid. The amount of the phaeopigments in the sample was measured by the change which follows the acidification.

Samples were brought back to the laboratory in polythene containers and placed in the shade away from direct light; the whole procedure was carried out in subdued light.

Apparatus:

- a) Millipore filtration apparatus designed to hold 47 mm diameter filters with a 1 litre volume reservoir.
- b) Stoppered graduated glass centrifuge tubes of 15 ml capacity.
- c) A small size mortar.
- d) Small screw top glass vials.

Special reagents:

- a) 90% acetone: 100 ml of distilled water was pipetted into 1 litre volumetric flask and made up to the mark with analytical reagent grade acetone or redistilled acetone.
- b) Magnesium carbonate suspension: one gram of finely

powdered (lightweight) A.R. $MgCO_3$ was added to 100 ml of distilled water and shaken. This solution was dispensed from a plastic washing bottle.

c) Hydrochloric acid: 50 ml of concentrated hydrochloric acid was diluted to 100 ml with distilled water.

Procedure:

Five hundred ml of the sample was measured by means of a measuring cylinder and placed into the reservoir of the filtration apparatus which was fitted with a 4.5 cm Whatman GF/C glass fibre filter. One ml of magnesium carbonate suspension was then added to the sample. More than one filter was used so often due to the rapid blockage with the high percentage of the suspended matter in the sample. The filter was then removed from the apparatus and the excess glass fibre paper was trimmed with a clean pair of scissors. Filters were then stored in small screw top vials and stored in -20°C. Within 2-3 days the filters were thawed and crushed with 3-4 mls of 90% acetone using the pestle and mortar. The dispersed glass fibres were transferred quantitatively into a 15 ml centrifuge tube. Ninety per cent acetone was used to rinse the mortar. Overall the volume of the acetone used should not exceed 10 ml. The tubes were then covered with a double layer of aluminium foil, for light prevention, and placed in the refrigerator at 2°C in complete darkness for 20 hours to allow the pigments to extract.

The tubes were then removed and allowed to warm up nearly to room temperature. Ninety per cent acetone was added to make the extracts exactly 12 ml. The contents were centrifuged for 10-15 minutes between 3,000 and 4,000 r.p.m. then the clear supernatant liquid was decanted into a Unicam glass 4 cm spectrophotometer cell, the extinction of which was measured against a cell containing 90% acetone at 665 and 750 ^{nm}. Two drops of 50% HCl were added to each sample, mixed, and allowed to stand 4-5 minutes; the extinctions were measured again at the same wavelengths. All readings were corrected for cell to cell blank. The concentration of chlorophyll α and the phaeopigments in the water was calculated from the expression:

$$\text{chlorophyll } \alpha (\text{mg m}^{-3}) = \frac{26.7 (665_o - 665_a) \times v}{v \times l}$$

$$\text{phaeopigments } (\text{mg m}^{-3}) = \frac{26.7 (1.7 665_a - 665_o) v}{v \times l}$$

where 665_o was the extinction at 665 ^{nm} before acidification, 665_a the extinction at 665 ^{nm} after acidification, v the volume of acetone for extraction (12 ml), V the volume of water filtered in litres (0.5l), and l the path length of the cuvette (4 cm).

3.5 Enumeration of phytoplankton:

The enumeration technique used in this investigation was a modification of the membrane filtration technique of McNabb (1960). On return to the laboratory the samples were thoroughly shaken and a suitable amount was taken for filtration. It was not possible to filter more than 50 ml of the sample due to the high suspended matter in the water which blocked the filter and made the filtration difficult. This formed a brown film on the filter covering the phytoplankton and making the counting and identification very difficult especially during the periods of high flow rate. To overcome part of this problem this method was modified during the last one-and-a-half years of the study. One hundred ml of the sample was placed in a graduated measuring cylinder and a few drops of Lugol's iodine was added. The sample was left to stand for sedimentation for 4-5 days (APHA, 1976; Furet and Benson-Evans, 1982). The excess water was siphoned out until 20 ml of the sample was left then this 20 ml was filtered. The advantages of this method was needing less time for filtration and colouring the phytoplankton with iodine which made it easier for enumeration.

The aliquot was measured into the reservoir of a Millipore filtration unit which had been previously cleaned with two washes of distilled water. Twenty-five mm diameter Millipore HA filters with 0.45μ pore size were always used. The filter flask (Buchner flask) was connected to a vacuum pump and the filtration was continued until the whole water had disappeared. The vacuum pump was then disconnected leaving the planktonic organisms as a film on the filter.

The membrane filter was removed carefully from the filtration unit and the edges were trimmed using a clean pair of scissors and placed, filtering-surface up, on a clean microscope slide. Three drops of Univert immersion oil were added to the filter and the slide was stored at room temperature. The clear filters were then covered with a clean cover glass. The excess oil was removed with a tissue and the mount was sealed with clear nail varnish. Slides which were prepared in this manner have shown no sign of deterioration after 24 months when kept in the dark.

The phytoplankton was counted using a Leitz-made Ortholux microscope Model 769484. Twenty-five to thirty random fields were viewed. The total number of the phytoplankton in every field was counted. This method was suitable for the samples because the phytoplanktons were mainly diatoms.

Sedimentation and counting with a haemocytometer was tried as a subsequent method but it was not successful because of having relatively small numbers of diatoms except on few occasions.

The number of algae per litre could be found for any species knowing the total count in 30 fields, the area of the filter, the area of the field and the number of litres filtered from the following equation:

$$\text{Number of algae } l^{-1} = \frac{\text{Area of filter} \times \text{total counts in 30 fields}}{\text{Total area of 30 fields} \times \text{Volume of sample in litre}}$$

3.6 Epiphytic Algae

3.6.1 Macrophyte population: sampling method:

Samples were analysed qualitatively only because of rupturing and cutting the macrophytes during the sampling. The banks of the river are high and they are covered with dense bushes on the sides, so that direct collecting from the river was not possible. An iron grapnel, Fig. 3.03, was used for this purpose. It was made from an iron collar which held 4 wire prongs at one end. The other end was tightened to a long strong rope which was held very tightly during the sampling process. Samples were taken from the bridges on each sampling station; the grapnel was thrown into the river against the water current and was left to sit on the river bed, then it was drawn over the river bed for a distance and pulled out. The wire prongs were acting as hooks catching the filamentous algae and macrophytic vegetation on its way.

The algae were put in plastic bags with few mls of formalin for preservation and marked, while the macrophytic angiosperms (about 1 meter long) were cut down by using a clean pair of scissors into 3 parts, top, middle and bottom. Four-five cm portions from every part were taken at random and placed separately in clean glass vials and marked.

In the laboratory the algae were transferred to a large petri-dish; using fine forceps a very small amount of the algae were placed over a clean microscope slide with a drop of water and with the aid of the forceps the filaments of algae were spread over the slide. A cover slip was placed on it and examined for the algal identification

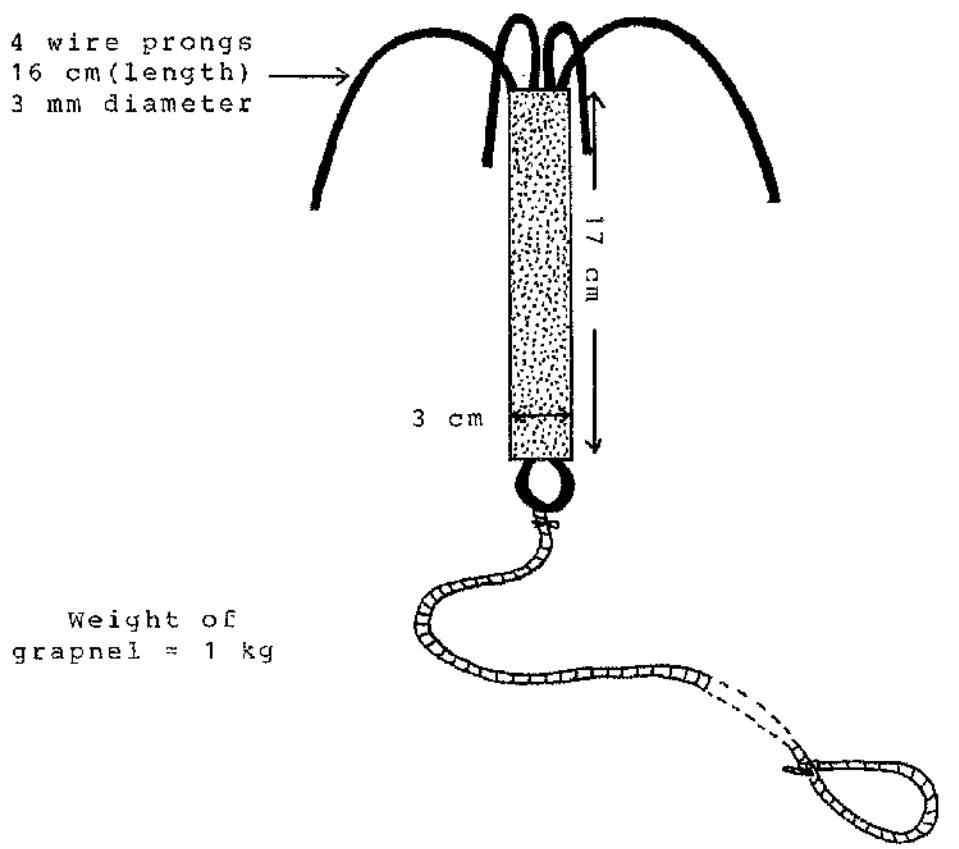


Figure 3.03: Diagram of grapnel used for sampling macrophyte population.

and their accompanied epiphytic diatoms. Ten slides per sample were tested with randomly taken samples. The macrophytes were similarly treated for the assessment of the epiphytic microalgae growing on them. This will be described in detail in the next section.

3.6.2 Methods for the assessment of epiphytic algae growing on the macrophytic angiosperms

There are few references on the quantitative studies of attached algal communities. This caused problems in the development of quantitative sampling techniques for the community. Authors used different techniques for this purpose (e.g. vigorous shaking, washing with jets of water, brushing and scraping, grinding the material or boiling in acid). Some investigators used combinations of these methods. However two different methods were used in the present investigation. The first one involved boiling in concentrated nitric acid (Main and McIntire, 1974; Sullivan and Rimer, 1975). One gram (fresh weight) of the macrophyte material was cut into small pieces and placed in a Pyrex beaker of 100 ml capacity. An amount of concentrated nitric acid was added sufficient to cover the plant material. The beaker was marked and put over a hot plate and boiled for 10 minutes and the pieces of the plant stirred using a glass rod. The whole process was carried out in the fume cupboard. The beaker was removed from the hotplate and allowed to cool. The acid was emptied into a measuring cylinder leaving the plant pieces in the beaker which were then rinsed 2-3 times with distilled water added to the acid. The samples were allowed to stand for sedimentation

and the supernatant was discarded by siphoning. The dislodged epiphytic algae were washed four times with distilled water (by sedimentation) until the acid remaining in the solution was washed off. Finally the diatoms were concentrated to 5 ml and they were placed in small vials. This method was used during summer (1980). The disadvantage of this method was that it had destroyed other kinds of algae, if there were any, rather than the diatoms.

Moss (1981) introduced a method for the assessment of the epiphytes, and this procedure was applied in the summers of 1981 and 1982.

In the laboratory, distilled water was added to the macrophyte portions in the vials. The plants were shaken by hand vigorously for 3 minutes then the water was poured off into a measuring cylinder. More water was added to the vials and re-shaken. This was repeated 4-5 times until the water remained clear after shaking. The total volume of washing was measured and mixed thoroughly. Four-five drops of Lugol's iodine solution (Lund, Kipling and Le Cren, 1958) was added to each sample and allowed to stand for sedimentation for 4-5 days. The epiphytes were then concentrated to 10 ml in distilled water and transferred to small vials.

The clean macrophyte portions which were left in the vials were then placed in a marked petri-dish and oven dried at 70-80°C for 24 hours.

A Weber B.S. 748 haemocytometer counting chamber was used for counting the algae. The counts were estimated by the number of algae

per gram of dry or fresh weight according to the procedure applied.

For mounting the diatoms, samples which were prepared by the second procedure needed cleaning. They were treated with hydrogen peroxide solution (ca. 27.5%), (Koivo, 1976).

Three to five ml of hydrogen peroxide was added to each sample and boiled for 5 minutes then they were cooled. Using the sedimentation they were washed 3 times with distilled water and finally they were concentrated to a small volume (the diatom frustules from the first procedure were also concentrated to a small volume).

Two to three drops of the frustules suspension were put on a clean microscope slide and dried gently on a hot plate. One to two drops of Pleurax mounting medium (Hanna, 1949) were added. Finally a coverslip was placed carefully and the slide was removed from the hot plate. The coverslip was pressed very gently to remove the excess media and any air bubbles.

3.6.3 Methods of using artificial and natural substrata:

Submerged glass slides are the common artificial substrata used widely in attached algal studies (Sladecak and Sladeckova, 1963, 1964).

In this investigation, Station 9 (Garscube Estate) was chosen to be a convenient site for this experiment due to its location in the grounds of Glasgow University.

As a trial method, 4 slides were fixed horizontally and vertically at the sides of a large cork under which a small piece of iron was fixed to keep the cork under water. These slides were suspended

as sets of 6-8 at one time, by means of strong fishing lines from the bridge. Two days after, all the slides were washed away by the current; the same event happened when this method was repeated. It was then decided to use suspended pebbles instead as they offered rough natural substrata better than the smooth surfaced slides.

For this purpose, pebbles were collected from Glazert Water (the only pebbly tributary of the river) and washed under the tap after scraping the surface. They were then washed and left for 24 hours in 50% hydrochloric acid and rinsed thoroughly with water. A small hole, enough for passing a fishing line through, was made in the centre of the pebble then they were put into an autoclave and sterilized for half-an-hour. This treatment made sure that no organisms were left on the surface of the pebble.

In the field, the sterilized pebbles were suspended from the bridge into the river in such a way as to be just above the sediment, by means of a strong fishing line. Sets of 10-12 pebbles were suspended at the one time and it was planned to collect 2-3 pebbles at weekly intervals. This experiment was not successful. Few pebbles were collected in many trials as the sudden fast flows flushed many of them away and pedestrians and children who used the bridge as a pathway often cut the lines.

The pebbles collected were placed into a wide mouth jar which contained Bold's Basal media (Bischoff and Bold, 1963) and transferred into a growing cabinet at 15°C and light/dark regime of 16/8h.

At weekly intervals small areas of the pebble surface were scraped by a scalpel, the material placed on a microscope slide with a drop of water, covered with a coverslip and examined for the

organisms attached and grown on them.

Attempt were made to sample the epipellic algal flora of the river by drawing a glass tube across the surface of the sediment (Aykulu 1982). This was not successful due to the presence of rubbish and metal parts on the sediment and the fast current washing away the mud on the bed.

3.7 Carbon Fixation Studies

3.7.1 ^{14}C light and dark bottles technique for measuring the productivity of phytoplankton

Vollenweider (1971) described the ^{14}C technique for estimating the primary productivity of the phytoplankton by measuring the rate of sodium [^{14}C] bicarbonate uptake during photosynthesis over a known period of time. His method has been widely used *in situ*.

In this investigation, an *in situ* was not possible due to the fast water currents and the inconvenience of the stations which were located separately outside the city. Measurements were made of phytoplankton from five stations only; Station (1), the river head, a non-polluted site, Station (4), Luggie Water which was one of the most polluted tributaries of the river, Station (6) Bardowie Bridge, one of the most polluted sites of the river, Station (9), Garscube Estate, which was a very convenient polluted site and Station (10) Kelvin Bridge, the last station of the river.

The following equipment and reagents were used:

- a) Light glass bottles with ground glass stoppers and dark bottles of 126 ml capacity which were prepared by covering the glass surface by a double layer of black scotch tape.

- b) Millipore filtration apparatus which holds filters of 4.5 cm and Millipore filters, type PH, of 0.3 μm pore size.
- c) Liquid scintillation counter (LKB Wallac model 1211 minibeta).
- d) Plastic (6 ml) minivials.

Reagents:

- a) Sodium [^{14}C] bicarbonate aqueous solution, CFA 431 Batch 40, from the Radiochemical Centre at Amersham.
- b) Triton scintillant which was prepared by mixing 5 gm PPO (2,5-diphenyloxazole) + 0.3 gm dimethyl-popop-1,4-Di-[2-(4-methyl-5-phenyloxazolyl)]-Benzene + 100-200 ml Triton. The volume was made up to 1 litre with toluene.
The ^{14}C bicarbonate solution was prepared before the inoculation. Using a microsyringe, an amount of ^{14}C bicarbonate was diluted with alkaline distilled water to a concentration of 1 $\mu\text{ci ml}^{-1}$. Alkaline distilled water was prepared by adding a few drops of a sodium hydroxide solution and a volume of distilled water until pH 8.5.

Procedure:

- a) In the field, two light and one dark bottles were filled with the water from the river sample, stoppered tightly and placed inside a dark wooden box.
- b) Water temperature was checked and more water sample was put in a 500 ml bottle for testing pH and alkalinity.
- c) In the laboratory, the samples were inoculated

rapidly by adding 1 ml of the working ^{14}C solution, using a disposable syringe and mixed.

- d) Immediately the bottles were transferred to a growth room and placed under fluorescent strip lights at $4\text{-}6 \text{ Wm}^{-2}$ at $15 \pm 1^\circ\text{C}$.
- e) Six hours later, the bottles were placed inside the dark wooden box and filtered rapidly through the Millipore filters. The filter and the filter holder were washed with 15-20 ml of distilled water. This operation was carried out in the absence of direct light.
- f) Using clean forceps the Millipore membrane filter was removed and put into a minivial with 5 ml of the scintillant and stoppered.
- g) One ml of the working solution and 5 ml of the scintillant were mixed in a vial for measuring the strength of the solution.
- h) Finally the vials were transferred into the liquid scintillation counter and the activities were expressed as counts per minute (cpm).
- i) Sample pH was measured (see pH determination). The alkalinity measured using the Mackereth (1963) procedure by titrating 100 ml of the sample against 0.01N HCl using 5 drops of 4.5 BDH indicator.
- j) A quenching curve for the efficiency of the scintillation counter (Neame and Homewood, 1974) was prepared as follows:

Twenty μl of ^{14}C hexadecane (Batch B37) was added to 5 ml of the scintillant in a minivial; the activity was measured. Few drops of a concentrated river sample were added at a time and the activities were measured after every addition. The (cpm) for 20 μl ^{14}C hexadecane is constant = 1.806×10^3 . Thus the efficiency per cent was calculated by

$$\frac{\text{Resultant activities}}{\text{Estimated activities}} \times 100 .$$

The quench curve was prepared by plotting the efficiency against the ratio which was given by the counter.

The ^{14}C assimilated in $\text{mg.C.l}^{-1} 3\text{h}^{-1}$ or $\text{mg.C.m}^{-3} 3\text{h}^{-1}$ was calculated from the expression given by Wetzel and Likens (1979).

$$(X) (\text{C}) = (\text{a}) (\text{b}) (\text{d})$$

$$(X) = ^{12}\text{C assimilated}$$

$$(\text{C}) = ^{14}\text{C activity added} =$$

$$\frac{(\mu\text{ci } ^{14}\text{C added}) \times (\text{dps of } ^{14}\text{C} = 3.7 \times 10^4)}{\text{efficiency factor of the counter}}$$

$$= \frac{1 \times 222000 \text{ dpm}}{1/\text{efficiency\%}/100} \quad \text{The resultant activities were}$$

expressed as (cpm) in accordance to the counts given by the liquid scintillation counter.

$$(\text{a}) = ^{12}\text{C available}$$

$$= \text{Total alkalinity} \times \text{pH factor}^*$$

$$(\text{b}) = ^{14}\text{C assimilated}$$

$$= (\text{filter counts from light bottle/min} \times K_1) - (\text{counts from dark bottle / min} \times K_2) / 1.06$$

* pH factor was taken from the table given by Saunder, Trama and Buchanan (1962) for converting total alkalinity to milligrams of carbon per litre.

K_1 = Volume correction factor for aliquots filtered
and sample bottles.

$$= \frac{\text{Volume of sample bottle} - \text{Volume of } ^{14}\text{C added}}{\text{Volume of aliquot filtered}} = \frac{125 - 1}{V}$$

$$(d) = \text{Dimensional factor to convert mg l}^{-1} \text{ to mg m}^{-3} = 1000$$

All the glassware used for this experiment was left in 20-25% Decon 90 for 3-4 days, after every use, then they were rinsed thoroughly with tap and distilled water and dried.

3.7.2 ^{14}C light and dark bottle technique for measuring productivity of the epiphytes:

Attempts had been made by many investigators to measure the productivity of the epiphytes using the ^{14}C uptake technique, (Wetzel, 1964; Vollenweider and Samaan, 1972; Hickman and Klarer, 1973, 1975).

In this investigation a modified technique was applied i.e. measuring the productivity of the epiphytes from different regions of the supporting plants. Two stations (6 and 9) were chosen as a field for this experiment as they were two polluted stations and the same macrophytic angiosperm species existed in both.

The angiosperms were sampled, cut and placed into vials using the same technique which was used in the assessment of the epiphytes (see 3.6.1). Duplicate samples were taken. In the laboratory the epiphytes were dislodged, by shaking the plant in sterilized Bold's basal media (Sec. 3.6.3.), using the Moss (1981) procedure. The epiphytes in the media were poured off into a stoppered 100 ml measuring cylinder; the volume was made up to 80 ml by adding more media. The cylinder was shaken until a homogenous solution was gained.

Three 20 ml samples of this homogenous solution were pipetted into two light and one dark glass bottles (125 ml). The volume was made up to 125 ml by adding sterilized Bold's media. Finally the bottles were placed under the same conditions for ^{14}C fixation as used with the phytoplankton. The incubation was for 6 hours under the same conditions of temperature and light intensity. The alkalinity and pH of the media were also measured.

The last 20 ml of the epiphytes suspension was used for counting the cell numbers per gram dry weight of the macrophytic supporting plants.

3.8 Algal Bioassays

3.8.1 Algal assay procedure:

Several unicellular algae have been used in algal bioassays. Those species show sensitivity to nutrient concentrations and to contaminants in bioassays (Scherfig, Dixon, Justice, 1978, 1979; Scherfig, Dixon, Justice and Carrilla, 1981). Ihotsky (1979) had reviewed the methodology of algal assay and represented a good selected bibliography on the topic.

The principles of this test were based on the "algal bioassay procedure" which was originally termed the "bottle test", EPA (1971). The single principle of this procedure was that if the nutrient data in a water sample was not giving enough information for the amount of algal growth in it, the water sample could be assayed by measuring the growth of a standard species of algae. This growth could be used as a basic comparison between different water samples

and comparing these with the growth in a suitable medium. This idea was applied in the present investigation.

The assay was carried out in the laboratory where the test algae were exposed directly to the water samples under constant temperature and light condition.

Ankistrodesmus falcatus (Corda) Ralfs. Strain No. 202/5A and *Scenedesmus quadricauda* (Turp.) de Brébisson strain No. 276/4B were the two unicellular green algae (Chlorophyceae) used for this assay. Komarek and Lhotsky (1979) reviewed references for the use of these two organisms, more frequently the second one, in assays as they are species recommended for testing the trophic potential of water. They were obtained from the Culture Centre of Algae and Protozoa, Cambridge, England.

The algae arrived from the culture centre growing on agar slopes. To keep them growing they were transferred to Bold's basal solid growth media in petri dishes. These plates were placed in a growth cabinet at 15°C under light intensities of 3.2 Wm^{-2} light/dark regime of 18/6 hours.

An amount of algae from the agar plates was transferred into clean (100 ml) Erlenmeyer flasks containing 50 ml of Bold's basal liquid media. The flasks were plugged with cotton wool and set up in a growth room of 15°C and constant fluorescent illumination of 5 Wm^{-2} ; under the above conditions the two species of algae grew satisfactorily.

River samples from all the stations were filtered through glass microfibre (G/CF) filter paper then through Millipore filters 4.5 cm

in diameter and 0.45 μm pore size. Fifty ml of the filtered sample were put into a clean dry (100 ml) Erlenmeyer flask triplicate for each sample made. One ml of one-week-old culture was added to each flask and mixed. Another two flasks were prepared by adding 1 ml of the cultures into 50 ml of the media to act as a control, then they were plugged with cotton wool, transferred to the culture growth room under the same conditions. Separate sets were made for the two different genera.

The cultures were shaken daily and the amount of the organism produced was determined every 2-4 days by counting cell numbers using a counting chamber (Weber, B.S. 748, England). The counts obtained from the control were compared with the counts from the river samples.

3.8.2 Measurements of chlorophyll:

Chlorophyll content was measured after one week and two weeks cultivation by extracting the pigments in hot methanol (Holden, 1965). Twenty ml of the algal cultures were filtered through a Millipore filter membrane 0.45 μm pore size, 2.5 cm in diameter; the algae was resuspended in redistilled methanol. The chlorophyll solution was covered with aluminium foil and placed in a hot water bath (45°C) for 10 minutes. The solution was then passed through No. 1 Whatman filter paper and the filtrate collected in an aluminium foil-covered test-tube. The optical density of the resulting chlorophyll solution was measured in a 600 SP spectrophotometer against a redistilled methanol blank at 650 and 665 nm. The equation used to determine the chlorophyll content was:-

$$\text{Chlorophyll (mg ml}^{-1}\text{)} = 25.5 \times \text{O.D.}_{650} + 4 \times \text{O.D.}_{665} \times \frac{20}{1000}$$

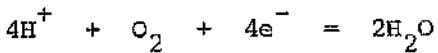
The last term is a dilution factor.

Chlorophyll α and the phaeopigments were estimated in the cultures 2 weeks after the cultivation. Ninety per cent acetone was used for the extraction (see 3.4). Attempts had been made to use methanol for this extraction, Marker (1972), but large errors were obtained due to the turbidity which occurred in the final extracted solution.

3.8.3 Measurements of photosynthesis:

Photosynthesis in the cultures were measured by measuring the oxygen evolution which allows the photosynthetic process as a whole to be assessed. This was measured in a modified Clark oxygen electrode, (Rank Bros., Bottisham, Cambridge, England) of the type described by Delieu and Walker (1972).

The apparatus consisted of a platinum wire sealed in plastic ~~circular~~ as the cathode, and an anode of ~~circular~~ silver wire bathed in a saturated KCl solution. The electrodes were separated from the reaction mixture by an oxygen permeable teflon membrane. The reaction mixture in the plastic container was stirred constantly with a small magnetic stirring rod. When a voltage was applied across the two electrodes using the polarizing meter, the platinum electrode became negative with respect to the reference electrode and the oxygen in the solution is thought to undergo electrolytic reduction at the cathode:-



the flow of current in the circuit when the polarising voltage was

set between 0.5 and 0.8V varied as a linear relationship to the partial pressure of the oxygen in solution. The instrument was usually operated at a polarising voltage of about 0.65V. The current flowing was measured by connecting the electrode to a sensitive potentiometric chart recorder.

The reaction chamber was kept at a constant temperature by circulating water from a temperature-controlled water bath.

Where applicable a light source was used to illuminate the reaction chamber. The intensity of this light was 140 Wm^{-2} as measured by a U.D.T. Model 40 x Opto-meter. A diagram of the layout of the apparatus is shown in Figure 3.04 and a diagram of the electrode is shown in Figure 3.05.

Calibration of the oxygen electrode was carried out by determining the zero oxygen level (by addition of sodium dithionite) and the concentration of oxygen in air-saturated distilled water.

The oxygen electrode requires uniform chlorophyll contents in the cultures of 0.03 mg/3 ml, thus large quantities of the river samples were used to culture the test algae. The chlorophyll was determined using hot methanol (see 3.8.2). An amount of each culture was filtered through Millipore filters of 0.45 μm pore size and 4.5 cm in diameter to give the above mentioned chlorophyll content and resuspended immediately in 3 ml of distilled water. This suspension was poured into the oxygen electrodes reaction chamber and the oxygen evolution was measured for 5 minutes. This was recorded on a chart recorder which was set on a high sensitivity of 90 and 95 using the instrument sensitivity control.

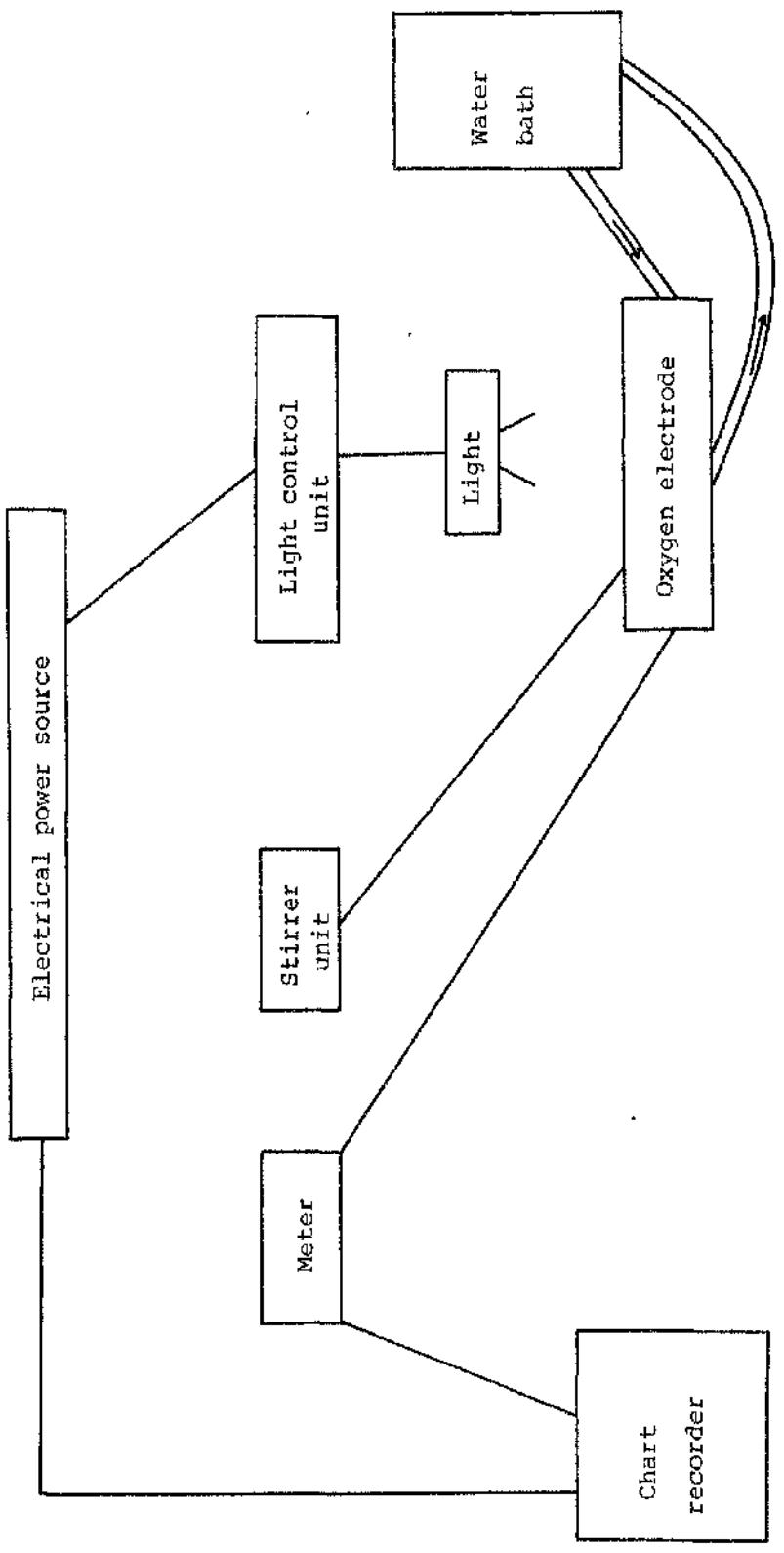


Figure (3.04) The layout of the apparatus used in the oxygen electrode experiment.

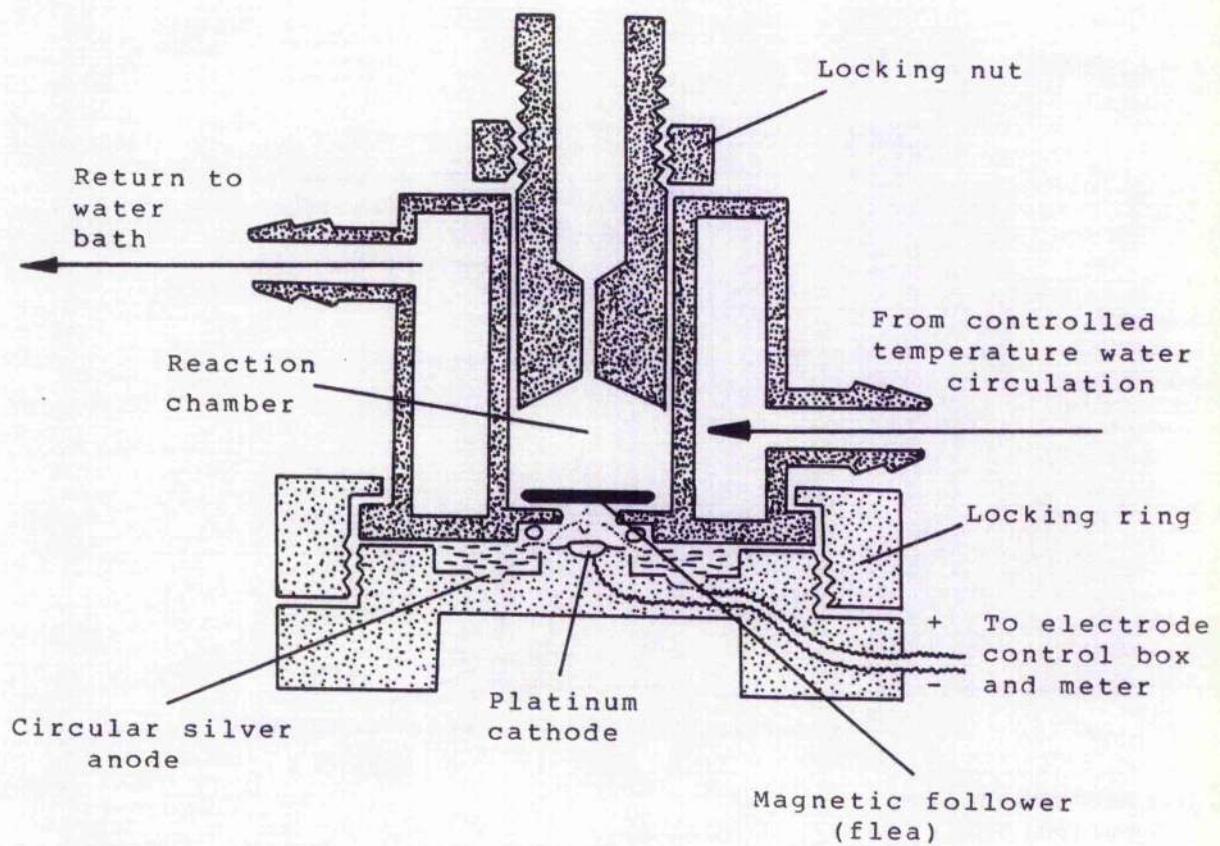


Figure (3.05) A diagram of the oxygen electrode.

The equation below gave the accepted way to express rates of oxygen evolution in terms of $\mu\text{moles } \text{O}_2 \text{ mg chl}^{-1} \text{ hour}^{-1}$.

$$\text{Rate } (\mu\text{mole } \text{O}_2 \text{ mg Chl}^{-1} \text{ hour}^{-1}) = \frac{\text{O}_2 \text{ content of 3 ml of air-saturated distilled water (in } \mu\text{moles) at } 25^\circ\text{C}}{\text{No. of chart recorder units between zero } \text{O}_2 \text{ and air-saturated distilled water points}}$$

$$\frac{\text{No. of chart recorder Units } \text{O}_2 \text{ changed}}{\text{Time (minutes)}} \times \frac{60}{\text{mg Chl. present in 3 ml algal suspension}}$$

The value for O_2 content for 3 ml of air-saturated distilled water at 25°C = 0.260.

CHAPTER FOUR

4. RESULTS

4.1 Physico-Chemical Analysis

4.1.1 Current flow

Data for current flow in the River Kelvin were obtained from the Clyde River Purification Board.* Mean daily flows were measured at their two gauging stations, the first one at Killermont one mile downstream of Station 5, and the second at Dryfield which is one mile upstream of Station 9. (Fig. 2.01).

Figure 4.01 shows the maximum, minimum and the mean monthly flows for the two stations. Maximum discharge rates of 57.61 - 61.48 cumecs occurred during autumn while the lowest of 1.14 - 1.5 cumecs were observed during spring and summer periods.

During 1980, maximum flows of 43.29 and 50.77 cumecs occurred in November while the minimum of 1.14 and 2.44 cumecs was during May at Dryfield and Killermont respectively. Highest flows for the river in 1981 were observed during October (57.61 and 61.48 cumecs), while the minimum of 1.48 cumecs at Dryfield and 1.68 cumecs at Killermont was during April. Finally in 1982 maximum flows of 47.04 and 55.14 cumecs were during January and the minimum of 1.3 and 1.67 cumecs were at April.

During 1980, the river's flow rate was continuously high during the autumn - winter months (September - March) while during 1981 two peaks for the high flows were observed, September - December and January - February. A low flow rate was recorded during December 1981 whilst very low temperature and some ice formation was observed.

Comparing the flow rates in the seasons of plant growth (April - October), in 1980 the low discharge period which occurred between

* = C.R.P.B. in the subsequent pages.

Figure 4.01: Mean monthly flows in River Kelvin
at the two gauging stations (Killermont
and Dryfield). Data provided by the
CRPB.

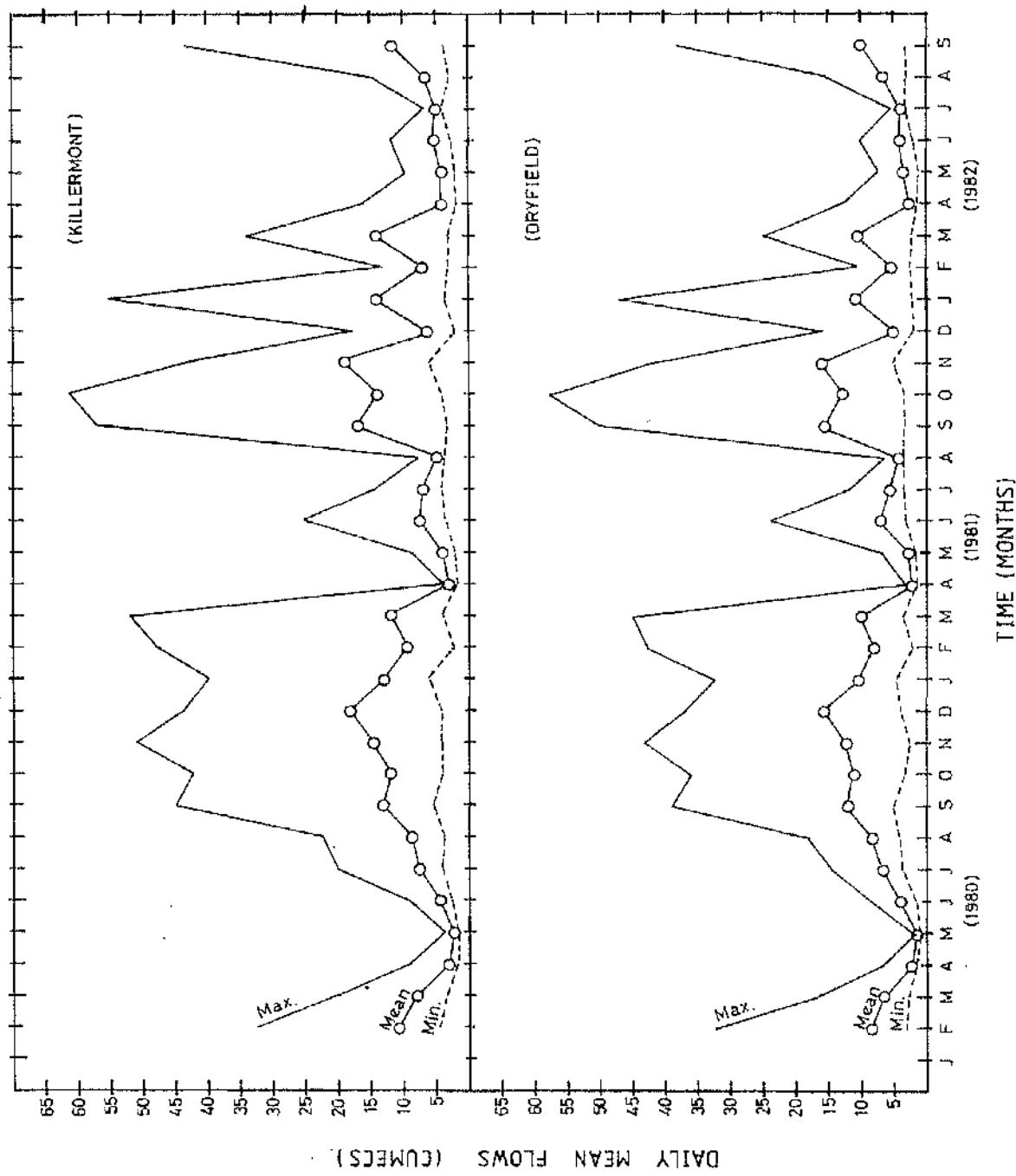


Figure 4.01

April - July was relatively shorter than in the years 1981 and 1982.

During the last two years, the flow was continuously low between

April - August with a little increase in June 1981.

Generally the values at Killermont were higher than the ones at Dryfield, suggesting higher flow rates in the lower stations compared with the upper ones.

Visual observations at the individual stations showed that at 1 and 6 flow rates were slow. Stations 7 and 10 had fast flows and the latter had small waterfalls just before the bridge. The rest of the stations were moderately slow to fairly fast currents.

4.1.2 Water temperature

Temperature in the River Kelvin varied seasonally with a range from 1.0°C in winter to 18°C in summer (Table 4.01). During December 1981 and January 1982 there was some ice formation in the river with temperatures down to -1.5°C as recorded by the C.R.P.B. This phenomenon was not seen on collection days.

Temperatures did not vary significantly with the stations (Table 4.01). Figure 4.02 shows the monthly mean temperature for all the stations during the period of this study. In 1980 the maximum of 16°C was recorded in July while the minimum of 3.8°C was in March. In 1981 the lowest temperature of 1.0°C was observed in December, whilst the highest, 14.5°C, was during August. In 1982 temperatures were between 2.8 - 14.0°C in January and July respectively. The range of temperature change was similar with the seasons over the years.

Table (4.01) Water temperature ($^{\circ}\text{C}$) recorded for River Kelvin at the 10 stations with mean temperature for all the stations and the \pm standard deviation during the period February 1980-September 1982.

Time	STATIONS										Mean	$S.D. \pm$
	1	2	3	4	5	6	7	8	9	10		
<u>1980</u>												
Feb.	6.00	5.50	5.50	6.00	5.50	5.50	5.50	5.50	5.50	6.00	5.65	0.24
March	5.40	4.80	4.20	5.50	2.80	3.00	3.00	3.10	3.00	3.00	3.78	1.09
April	11.0	11.0	10.8	12.5	11.5	11.5	11.2	11.5	11.6	11.5	11.4	0.55
May	11.0	12.0	12.0	12.5	12.0	14.0	12.0	13.5	13.5	13.0	12.6	0.93
June	12.5	15.0	14.0	14.5	14.0	15.5	13.8	15.5	15.5	14.6	14.6	1.00
July	15.0	16.5	15.5	16.0	15.8	16.5	15.5	16.0	16.5	16.8	16.0	0.57
Aug.	10.9	12.5	12.0	12.5	12.0	14.3	12.5	13.3	13.5	13.0	12.7	0.94
Oct.	7.10	6.00	6.00	7.00	6.50	6.50	7.40	6.40	5.80	5.50	6.40	0.61
Nov.	9.00	8.60	8.20	8.60	7.90	8.80	8.50	8.50	8.50	8.50	8.40	0.28
Dec.	3.80	3.80	3.80	5.00	4.20	4.80	4.00	4.00	4.50	4.50	4.20	0.44
<u>1981</u>												
Jan.	6.60	6.50	6.60	7.20	6.90	7.00	6.00	6.60	6.80	7.00	6.70	0.34
Feb.	2.00	2.00	2.30	2.50	2.00	2.50	1.50	2.20	1.80	1.80	2.00	0.31
March	7.00	7.00	7.00	7.80	7.20	7.50	7.00	7.30	7.50	7.80	7.30	0.32
April	9.00	8.40	7.80	8.30	7.80	9.20	8.60	9.00	9.60	9.00	8.50	0.62
May	9.60	10.8	10.8	11.8	11.4	12.9	12.0	13.0	13.5	12.8	11.9	1.23
June	10.6	12.7	12.3	12.7	12.7	14.0	12.7	13.3	14.3	13.5	12.9	1.03
July	11.5	15.2	13.5	14.0	14.0	16.0	13.5	15.8	16.5	16.0	14.6	1.57
Aug.	12.8	14.5	14.5	14.5	14.2	15.5	14.5	15.5	16.4	15.8	14.8	1.01
Sept.	11.2	11.0	11.0	11.2	11.0	11.5	11.5	11.5	11.5	11.5	11.3	0.23
Oct.	5.00	5.00	5.50	6.00	5.50	5.50	5.50	6.00	6.50	6.00	5.60	0.39
Nov.	5.40	5.60	5.70	5.90	5.90	5.50	5.50	5.50	5.50	5.60	5.60	0.17
Dec.	1.00	1.30	1.30	1.50	1.50	1.50	1.00	1.20	1.00	1.00	1.23	0.22
<u>1982</u>												
Jan.	2.40	2.70	2.70	3.00	3.00	3.00	3.00	3.00	3.00	3.00	2.88	0.21
Feb.	5.00	5.00	5.00	5.50	5.00	5.50	4.80	5.00	5.00	5.00	5.00	0.23
March	4.00	4.70	4.20	5.80	5.00	6.00	4.50	5.80	6.20	6.00	5.16	0.89
April	7.50	8.00	8.00	9.00	8.50	9.50	9.00	9.50	10.0	10.0	8.90	0.88
May	10.0	11.0	11.0	11.8	11.8	13.0	12.0	12.0	13.5	13.4	12.0	1.12
June	11.0	12.5	12.3	12.5	12.5	13.5	13.5	13.8	14.0	13.9	13.0	0.95
July	13.5	16.4	16.0	16.0	16.5	18.0	16.0	16.0	16.0	16.0	16.0	1.09
Aug.	11.5	11.5	11.0	12.0	11.0	12.0	11.0	12.0	12.0	12.0	11.6	0.50
Sept.	8.40	8.70	8.80	9.00	9.00	9.00	8.90	8.90	8.90	8.80	8.84	0.18

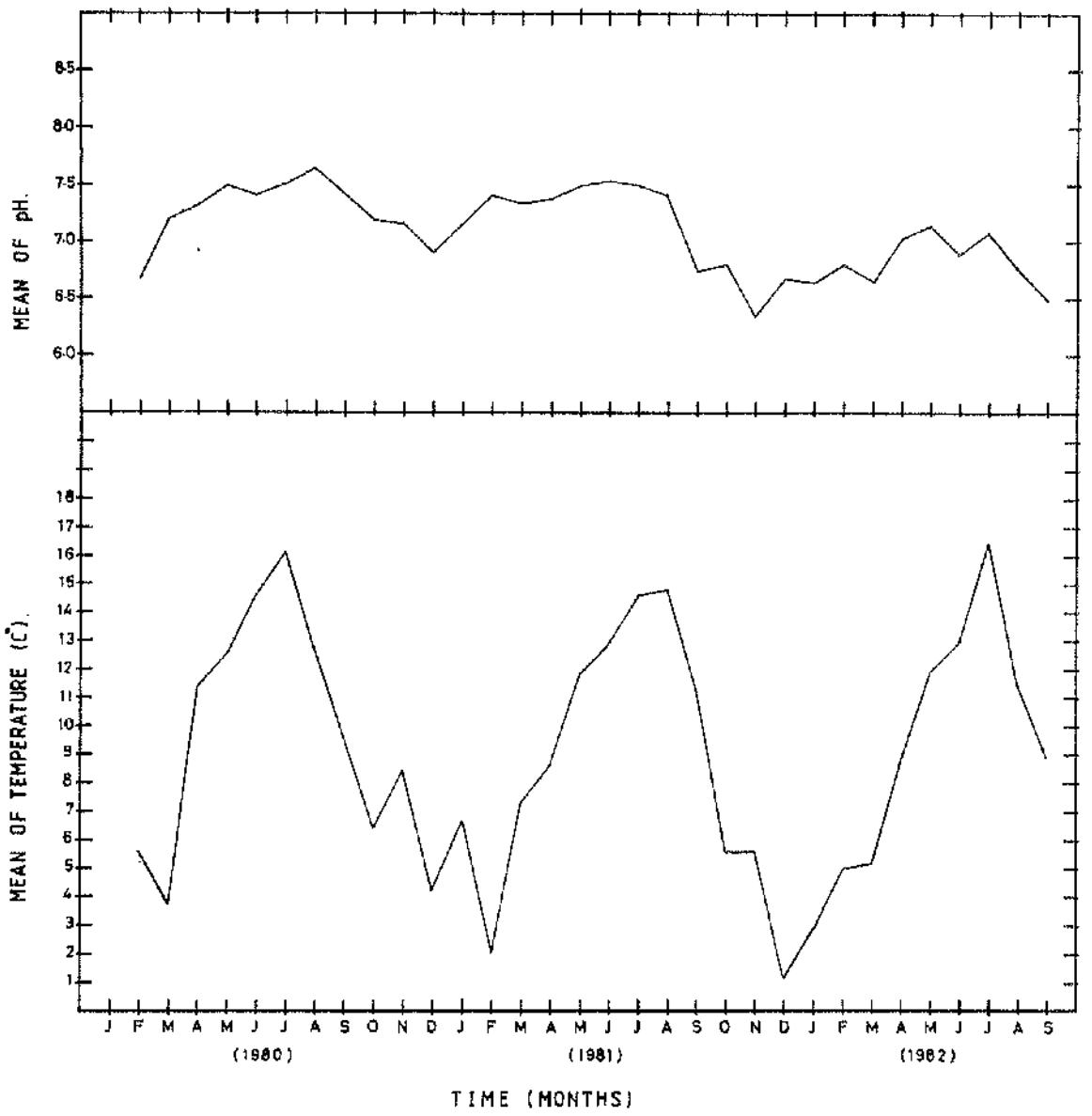


Figure 4.02: Mean pH and temperature values for the river throughout sampling period. The mean obtained from all the stations at each sampling time.

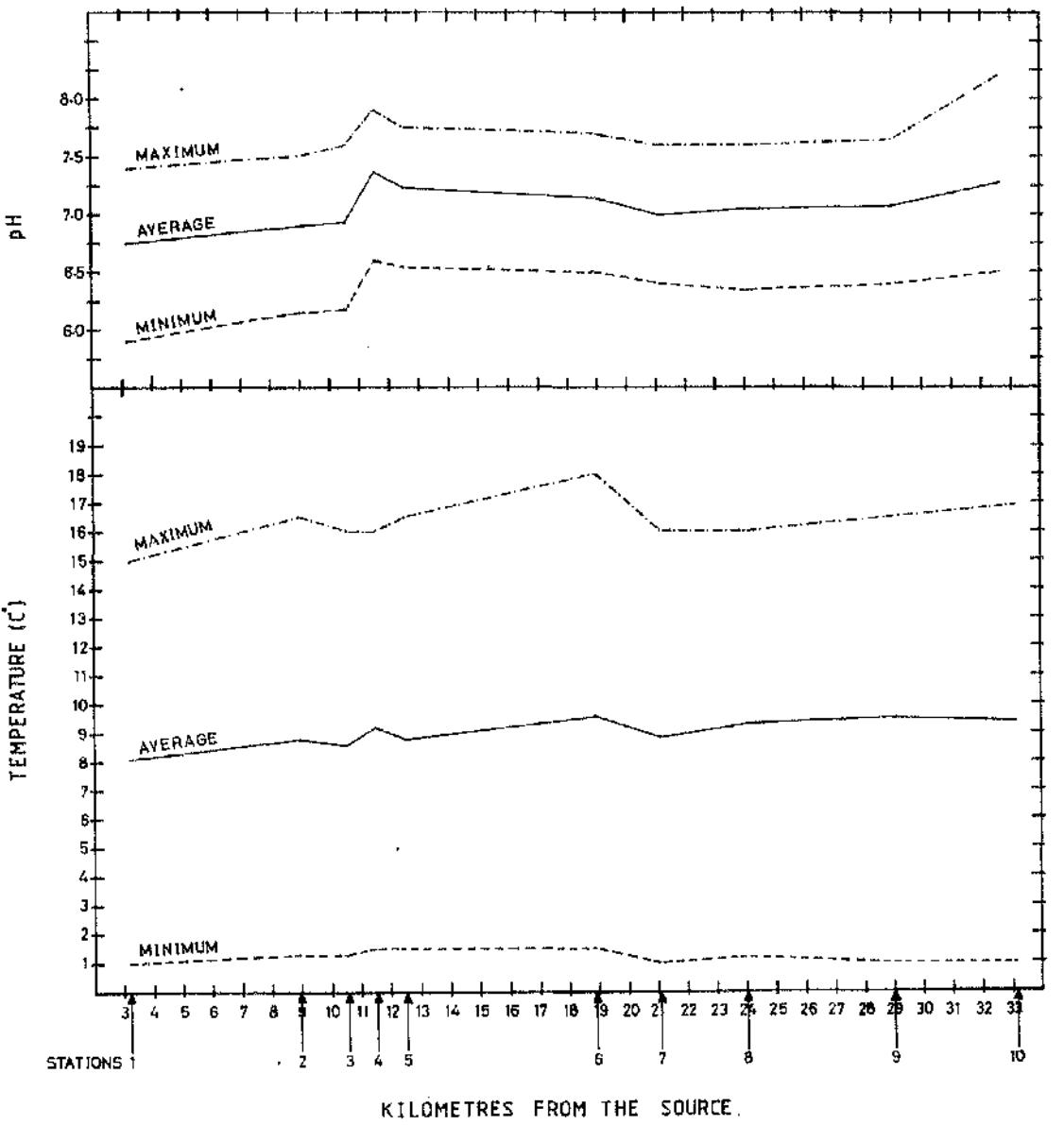


Figure 4.03: pH and temperature profiles for the River Kelvin. The means obtained from the data for the entire sampling period for each station.

Figure 4.03 shows the average temperature profile recorded in the River Kelvin during the triennial period 1980-1982. In general terms, the profile indicates relatively constant patterns at maximum, mean and minimum values. Temperature differences were minimal along the length of the river. There was an increase by 0.5 - 1.5°C on going downstream. Maximum values of 18°C were recorded at Station 6 whilst the minimum of 1.0°C was observed at Station 1.

4.1.3 pH

The annual range for the pH was 6-8 (Table 4.02). pH did not vary significantly within the stations (Table 4.02). In general, as the river passed downstream there was a slight pH increase.

Figure 4.02 shows the monthly mean pH values for the stations during the triennial period 1980-1982. Generally the pH values recorded during the period February - August were greater than the ones observed in the period September - January during which the flow rate was maximum. pH values showed similar seasonal patterns with a decrease during the final year, 1982, particularly during June.

Figure 4.03 displays river profiles for pH recorded during the period of this study, and indicates relatively constant patterns at maximum, mean and minimum values. Station 1 had the lowest mean of 6.75. Passing downstream to Stations 2 and 3 there was a slight pH increase, averages of 6.9 and 6.92 were recorded respectively. The polluted Luggie Water, Station 4, with its mean pH value of 7.4 had affected the Kelvin by increasing the levels in the main river to 7.2 at Station 5. The range stayed almost constant for Stations 6, 8 and 9. A slight decrease was observed at Station 7 due to the clean and

Table (4.02) pH values recorded at the 10 stations of the River Kelvin with the mean values for all the stations and \pm standard deviation during the period February 1980 - September 1982.

Time	1	2	3	4	5	STATIONS					Mean	S.D. \pm
						6	7	8	9	10		
1980												
Feb.	7.2	6.2	7.1	7.6	7.0	6.5	6.8	6.7	6.5	7.0	6.7	0.34
March	6.8	7.2	7.1	7.4	7.3	7.3	7.1	7.2	7.3	7.2	7.2	0.20
April	7.2	7.1	7.4	7.5	7.45	7.5	6.9	7.1	8.2	7.3	7.3	0.36
May	7.4	7.5	7.4	7.5	7.6	7.5	7.2	7.4	7.5	7.6	7.5	0.12
June	7.3	7.4	7.5	7.6	7.5	7.3	7.3	7.4	7.4	7.4	7.4	0.16
July	7.3	7.3	7.4	7.4	7.8	7.6	7.4	7.5	7.5	7.8	7.5	0.18
Aug.	7.3	7.5	7.6	7.9	7.8	7.7	7.5	7.6	7.65	8.0	7.6	0.21
Oct.	6.8	7.0	7.1	7.6	7.4	7.2	7.1	7.2	7.15	7.4	7.2	0.22
Nov.	7.0	7.1	7.0	7.3	7.3	7.2	7.1	7.2	7.1	7.3	7.2	0.11
Dec.	6.5	6.8	6.6	7.2	7.0	7.1	6.9	7.0	7.0	7.1	6.9	0.21
1981												
Jan.	6.8	7.0	7.0	7.6	7.3	7.0	7.0	7.2	7.15	7.3	7.2	0.22
Feb.	7.0	7.25	7.3	7.9	7.6	7.5	7.3	7.4	7.5	7.6	7.4	0.23
March	6.8	7.1	7.2	7.7	7.5	7.4	7.4	7.5	7.3	7.5	7.3	0.25
April	7.1	7.2	7.3	7.6	7.4	7.4	7.5	7.4	7.4	7.5	7.4	0.15
May	7.2	7.4	7.5	7.7	7.7	7.5	7.6	7.4	7.5	7.6	7.5	0.14
June	7.2	7.4	7.5	7.8	7.6	7.6	7.6	7.4	7.6	7.7	7.5	0.17
July	7.3	7.2	7.3	7.8	7.6	7.6	7.6	7.5	7.4	7.5	7.5	0.21
Aug.	7.4	7.4	7.4	7.6	7.5	7.5	7.4	7.4	7.3	7.4	7.4	0.08
Sept.	6.3	6.5	6.5	7.3	6.9	6.9	6.9	6.8	6.8	6.9	6.7	0.29
Oct.	6.5	6.7	6.6	7.3	7.0	6.9	6.7	6.8	6.8	7.0	6.8	0.23
Nov.	5.9	6.2	6.2	6.7	6.6	6.6	6.4	6.35	6.4	6.5	6.4	0.23
Dec.	6.0	6.4	6.5	7.0	6.9	6.9	6.7	6.7	6.8	6.9	6.7	0.30
1982												
Jan.	5.9	6.3	6.4	6.9	6.9	6.9	6.8	6.8	6.8	6.9	6.6	0.34
Feb.	6.1	6.6	6.6	7.3	7.1	7.0	6.9	6.8	6.9	7.0	6.8	0.33
March	6.2	6.5	6.5	7.1	6.9	6.8	6.5	6.7	6.7	6.9	6.7	0.27
April	6.7	6.95	7.0	7.3	7.3	7.1	6.9	7.0	7.1	7.2	7.0	0.18
May	6.5	6.9	6.9	7.3	7.1	6.9	6.8	6.8	6.9	7.0	6.9	0.20
June	6.5	6.7	6.7	7.1	6.9	6.8	6.8	6.8	6.9	7.0	6.8	0.17
July	7.0	6.8	6.9	7.5	7.3	7.2	6.7	7.1	7.2	7.5	7.1	0.28
Aug.	6.3	6.6	6.7	7.2	6.9	6.9	6.7	6.8	6.8	7.0	6.8	0.25
Sept.	6.2	6.3	6.4	6.9	6.7	6.6	6.5	6.5	6.5	6.6	6.5	0.20

satisfactory condition of the water at this station, Allander Water, with a mean pH value of 7.0. An increase was observed at Station 10 in which the average of 7.3 was recorded.

4.1.4 Dissolved Oxygen

Dissolved oxygen levels recorded for the river were generally higher during winter than in summer. The annual values observed ranged between $1.9 - 11.8 \text{ mg l}^{-1}$ (Table 4.03).

There were significant differences between the stations and between seasons (Table 4.03) and they all followed a uniform annual pattern (Figure 4.04). During the winter period, October - March, dissolved oxygen was at its highest value when the flow rate was maximum and the water temperature was minimum. During winter 1981 - 1982, the period of the lowest water temperature and ice formation, a slightly higher oxygen content was recorded for all the stations.

Oxygen depletion was observed during the spring - summer periods, April - September, when the flow rates were minimal and temperatures were high. Stations 1 and 2 had almost the same values and they were lower than the rest of the stations with Station 1 having the least of all. Their values were noticeably lower than the rest during the summers of 1980 and 1981. After Glazert Water joined the river, oxygen values increased in the main river at Station 3. Although Luggie Water (Station 4) was the most polluted station on the river, it had good oxygen values. The dissolved oxygen stayed the same for the rest of the river with noticeable increases at Station 7 which is the clean Allander Water and at Station 10 which was the station having the highest oxygen content and small waterfalls.

Table (4.03) Mean data for the dissolved oxygen concentrations (mg l^{-1}) recorded in the River Kelvin during the period February 1980-September 1982 and the two way ANOVA for the data.

Figure 4.04: Dissolved oxygen and BOD_5 in mg l^{-1}
obtained from River Kelvin during the period
February 1980 - September 1982.

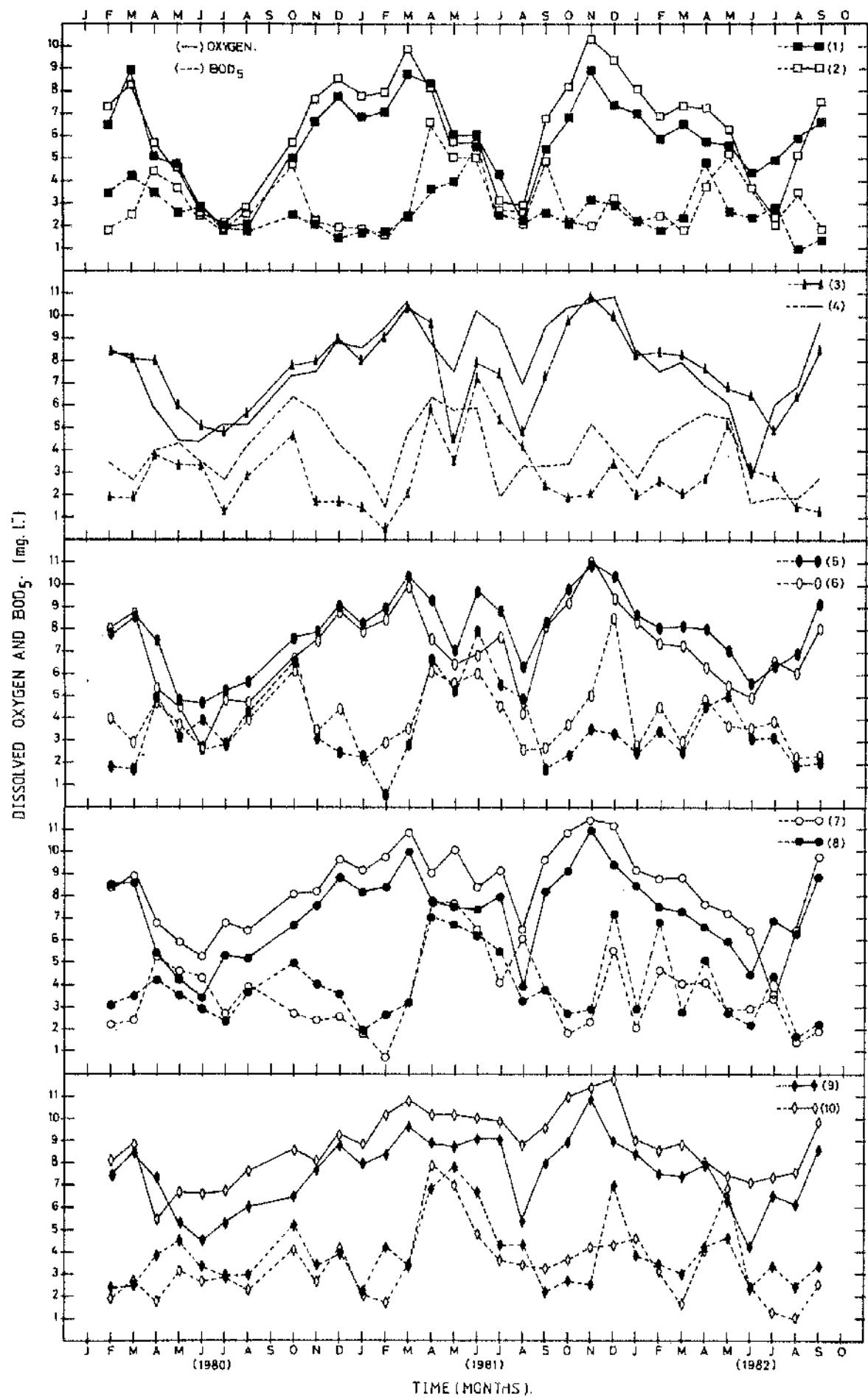


Figure 4.04

Figure 4.05 shows the annual profile for the dissolved oxygen in the River Kelvin. The profile indicates clearly that the oxygen content increased as the river passes downstream. Generally for the whole profile Station 1 had the lowest values and Station 10 the highest. Maximum values showed a fairly constant picture with a slight increase as the river reached Station 10. The average values for Stations 1 and 2 were the same but there was an increase in the averages as Glazert Water joined the Kelvin at Station 3. The average values for the dissolved oxygen stayed almost constant for the rest of the stations with increases at Station 7, which is the clean Allander, and Station 10 in which the highest average of 8.8 mg l^{-1} was observed. The minimum values showed fluctuations, Stations 1 and 2 had the same values but the values increased as the Kelvin mixed with the clean Glazert Water. Although the minimum values for the dissolved oxygen at the polluted Luggie Water, Station 4, were lower than Station 3, the values in the main river stayed the same at Station 5. The polluted Bishopbriggs Burn decreased the minimum values again in the main river at Station 6 and the values increased slightly as the clean Allander mixed with the Kelvin. Minimum oxygen values increased as the Kelvin reached Stations 9 and 10.

4.1.5 Biological Oxygen Demand (BOD_5)

The actual values for BOD_5 observed in the Kelvin ranged between a maximum of 8.5 mg l^{-1} and a minimum of 0.5 mg l^{-1} (Table 4.04). Significant differences were recorded both with the stations and seasonally (Table 4.04).

Table (4.04) Mean data for the BOD_5 values ($mg\ l^{-1}$) recorded in the River Kelvin during the period February 1980 - September 1982 and the two way ANOVA for the data.

Time	S T A T I O N S									
	1	2	3	4	5	6	7	8	9	10
1980										
Feb.	3.45	1.80	1.90	3.45	1.80	4.00	2.20	3.10	2.40	1.80
March	4.25	2.50	1.90	2.70	1.70	2.90	2.45	3.50	2.45	2.65
April	3.50	4.45	3.95	3.95	4.90	4.70	5.30	4.20	3.80	1.75
May	2.60	3.70	4.45	3.35	3.15	3.70	4.60	3.55	4.50	3.15
June	2.81	2.50	3.40	3.50	3.90	2.70	4.30	2.90	3.37	2.65
July	1.80	1.90	1.25	2.70	2.85	2.87	2.65	2.35	2.90	2.93
Aug.	1.75	2.55	2.95	4.25	4.30	3.95	3.95	3.65	2.90	2.25
Oct.	2.50	4.75	4.65	6.40	6.60	6.15	2.70	4.95	5.25	4.10
Nov.	2.10	2.20	1.75	5.80	3.15	3.45	2.40	4.00	3.40	2.70
Dec.	1.45	1.90	1.70	4.30	2.45	4.40	2.60	3.60	3.90	4.15
1981										
Jan.	1.70	1.87	1.45	3.42	2.27	2.27	1.87	1.93	2.20	2.05
Feb.	1.70	1.60	0.50	1.70	0.57	2.90	0.70	2.65	4.23	1.70
March	2.40	2.40	2.10	4.83	2.80	3.55	3.20	3.20	3.40	3.40
April	3.63	6.63	5.93	6.43	6.68	6.15	7.83	7.08	6.85	7.95
May	3.98	5.33	3.55	5.80	5.28	5.60	7.73	6.70	7.85	7.03
June	5.60	5.00	7.30	5.98	7.90	6.10	6.50	6.20	6.70	4.80
July	2.45	2.70	5.40	1.95	5.55	4.63	4.08	5.53	4.33	3.58
Aug.	2.13	2.65	4.20	3.30	4.90	2.60	6.10	3.30	4.35	3.40
Sept.	2.61	4.93	2.45	3.35	1.73	2.70	3.75	3.83	2.18	3.25
Oct.	2.13	2.15	1.93	3.48	2.35	3.70	1.83	2.70	2.73	3.65
Nov.	3.18	2.05	2.15	5.33	3.53	5.05	2.35	2.90	2.48	4.20
Dec.	2.95	3.28	3.40	4.10	3.30	8.50	5.55	7.23	7.00	4.35
1982										
Jan.	2.26	2.24	2.00	2.83	2.45	2.75	2.08	2.90	3.85	4.65
Feb.	1.80	2.45	2.68	4.48	4.43	4.58	4.70	6.80	3.43	4.18
March	2.33	1.83	2.10	5.18	2.48	2.98	4.08	2.83	3.03	1.65
April	4.88	3.73	2.78	5.70	4.53	4.88	4.10	5.13	4.28	4.15
May	2.65	5.20	5.25	5.40	5.08	3.65	2.80	2.75	4.68	6.90
June	2.35	3.18	3.20	1.68	3.05	3.58	2.93	2.23	2.35	2.35
July	2.80	2.13	2.85	1.90	3.23	3.80	3.43	4.38	3.35	1.25
Aug.	0.95	3.50	1.50	1.88	1.88	2.28	1.38	1.65	2.40	1.05
Sept.	1.38	1.83	1.25	2.78	2.00	2.30	1.98	2.28	3.35	2.58

Source	D.F.	S.S.	M.S.	F.factor	Error	Total
Rep.	1	0.16	0.16	0.9	309	619
A Time	30	877	29	2177**	4.15	1504
B Stations	9	121	13	1001**	0.13	
A*B	270	501	10	138**		

** Significant differences at 1% level.

Figure 4.04 shows clearly that the levels were generally higher during summer than the winter with few occasions (i.e. December 1981) when high levels were recorded in Stations 4-9 with low discharges of maximum 16 cumecs (Figure 4.01). Station 1 had the lowest BOD values of all the stations. The values increased slightly as the river reached Station 2 especially during October 1980 and summer 1981. The values increased further downstream. The highest values were recorded at Stations 4, the Luggie Water, and 6 which was affected by the polluted Bishopbriggs Burn. Very low values for the Stations 3, 4, 5 and 7 were observed during February 1981. However, there was not a uniform seasonal pattern for all the stations. There were many fluctuations. Occasionally the BOD values were almost equal to the values of the dissolved oxygen e.g. during August 1981 at Stations 2, 3, 7, 8 and 9.

The BOD profile for the Kelvin for the period 1980-1982 (Figure 4.05) shows clearly that the maximum showed some fluctuations. Station 1 had the lowest value with an increase downstream to Stations 2 and 3. There was an unexpected decrease in the maximum value at Station 4 (Luggie Water). The values increased further as the river gets to Stations 5 and 6, the latter had the highest of all. The BOD decreased slightly as the clean Allander joined the Kelvin at Station 8 and stayed constant for the rest of the river. The average values were almost constant along the stretch of the River Kelvin, Station 1 had the lowest contents. There was an increase at Station 4, Luggie Water, but the values stayed almost the same for the rest of the stations. Minimum values also showed fluctuations. Station 3 had

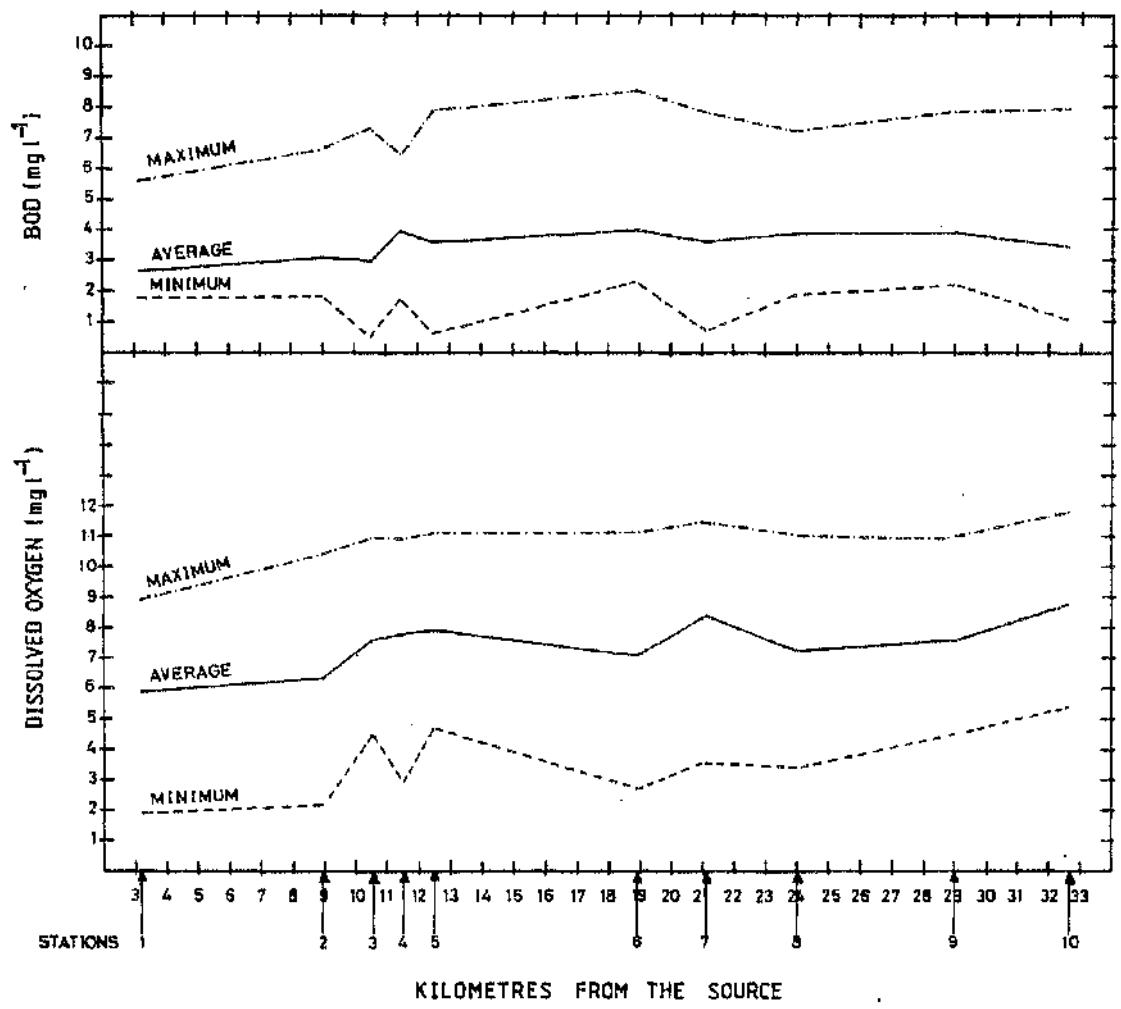


Figure 4.05: Oxygen and BOD profiles for the River Kelvin.
The means obtained from the data for the entire
sampling period for each station.

the least values which were affected by the clean Glazert Water.

There were other stations with very low values i.e. Stations 5, 7 and 10. The rest of the stations had almost constant values with Station 6 the highest.

4.1.6 Reactive phosphate

Phosphate phosphorus levels in the River Kelvin were high along the whole stretch of the river. Values ranged between 0.26 - 34 μg at l^{-1} (Table 4.05). The minimum value of 0.26 μg at l^{-1} was recorded at Station 1 during November 1981 while the maximum of 34 μg at l^{-1} observed at Station 6 during June 1981.

The phosphorus levels for the stations varied significantly from one another and also seasonally (Table 4.05) but they are followed by a uniform seasonal pattern (excluding Station 1). Figure 4.06 shows clearly the seasonal pattern for reactive phosphorus at all the stations. Generally levels were higher during the spring-summer period (April-September) and relatively low in autumn-winter (October-March). The phosphorus levels during summer 1981 were higher than the ones observed during 1980 and 1982.

Station 1 had the least phosphorus content of all the stations. Its values stayed almost constant for the whole period of this study with levels being slightly higher during summer. There was only one noticeable increase in the phosphorus content during August 1980 when values of 4.6 μg at l^{-1} were recorded. The polluted Dock Water and Broad Burn caused a rapid increase in phosphorus levels in the Kelvin at Station 2, with ranges of 0.81 - 24.1 μg at l^{-1} . As the clean

Table (4.05) Mean data for the phosphate P concentration ($\mu\text{g at } \text{L}^{-1}$) recorded in the River Kelvin during the period February 1980-September 1982 and the two way ANOVA for the data.

Time	S T A T I O N S									
	1	2	3	4	5	6	7	8	9	10
<u>1980</u>										
Feb.	2.28	4.70	5.31	9.77	7.06	11.8	3.28	9.34	9.20	9.70
March	0.68	1.73	1.28	2.92	1.82	3.11	2.34	3.09	3.06	3.53
April	1.50	13.5	13.5	15.2	14.1	15.2	13.5	14.9	14.1	13.5
May	1.47	17.2	11.0	24.6	19.1	25.9	8.49	24.6	24.0	24.6
June	1.30	15.2	9.00	23.1	10.5	20.8	7.90	22.5	22.0	20.0
July	1.29	7.50	5.10	22.5	9.40	13.8	3.24	9.40	9.58	8.46
Aug.	4.60	7.30	5.70	26.5	14.0	21.1	8.20	19.0	15.7	12.6
Oct.	0.97	9.29	6.01	23.4	12.2	20.1	4.60	17.8	17.6	16.2
Nov.	0.26	0.81	1.67	3.25	1.43	3.00	1.00	2.17	2.13	2.68
Dec.	0.84	2.26	1.93	5.98	3.32	3.45	1.43	4.13	4.26	5.02
<u>1981</u>										
Jan.	0.45	1.46	1.48	6.00	3.55	5.05	1.34	4.12	4.03	4.40
Feb.	1.99	6.39	5.12	21.5	12.1	21.0	9.18	18.8	19.2	15.6
Mar.	0.92	2.40	2.24	6.27	3.79	5.60	3.38	5.17	5.20	5.66
April	1.70	15.4	11.4	23.7	15.8	24.5	18.2	24.6	25.5	18.4
May	1.53	24.1	12.8	30.8	19.8	27.1	16.1	26.4	24.5	16.2
June	1.60	12.0	9.40	26.9	22.9	33.9	17.2	31.5	22.8	22.6
July	0.92	7.10	6.39	26.5	15.3	26.5	6.96	21.2	17.1	16.3
Aug.	0.80	17.9	17.3	24.9	23.9	26.2	15.7	26.2	24.3	24.9
Sept.	0.64	1.97	1.92	3.25	2.40	3.20	2.56	2.56	3.10	3.20
Oct.	0.80	4.02	3.94	12.8	6.37	10.3	5.66	8.80	9.11	8.18
Nov.	1.12	2.55	2.03	4.69	2.93	3.94	2.23	3.35	3.25	3.45
Dec.	1.84	5.17	6.70	17.0	12.3	14.9	11.8	14.6	17.1	13.3
<u>1982</u>										
Jan.	0.63	2.98	2.01	5.85	4.01	6.22	1.84	5.31	6.14	6.30
Feb.	1.51	8.14	5.40	16.4	10.1	14.6	4.79	11.4	11.8	10.4
March	1.44	8.18	7.80	21.8	13.8	18.3	11.4	16.5	16.3	13.4
April	2.11	17.3	11.2	26.8	20.9	22.1	16.2	21.1	21.5	21.0
May	2.05	18.2	12.2	19.3	14.5	22.4	16.6	19.0	21.3	17.6
June	1.11	8.35	7.73	20.8	12.2	17.5	12.2	13.7	12.9	10.1
July	1.67	22.3	20.0	30.9	24.4	29.1	22.3	27.3	25.2	23.7
Aug.	0.87	4.33	3.78	16.8	6.67	8.57	3.77	7.07	7.38	7.11
Sept.	0.79	2.37	1.85	3.12	2.31	3.65	1.71	3.48	3.28	3.47
Source	D.F.	S.S.	M.S.	F.factor	Error	Total
Res	1	0.01	0.01	N/A	N/A	N/A

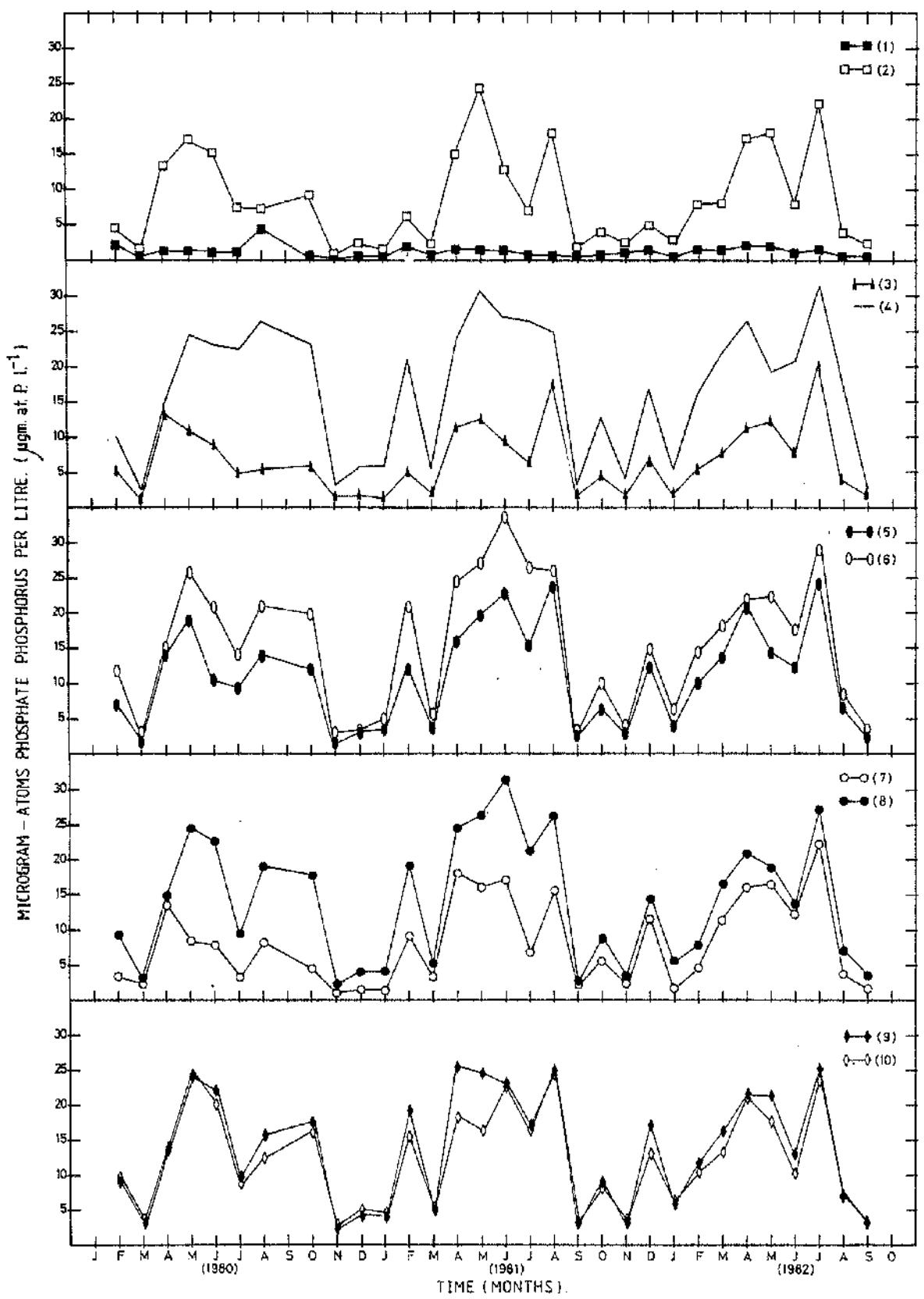


Figure 4.06: Reactive phosphorus concentration in $\mu\text{g at P l}^{-1}$ observed in the River Kelvin during the period February 1980 – September 1982.

Glazert Water joined the Kelvin, phosphorus levels decreased slightly in the main river and ranges of $1.28 - 20.0 \mu\text{g at l}^{-1}$ were observed at Station 3. Very high phosphorus levels were observed at Station 4, due to the influence of sewage treatment works. The values ranged between $3 \mu\text{g at l}^{-1}$ during March 1980 and $31 \mu\text{g at l}^{-1}$ during July 1982. This high content had affected the main river by increasing its levels to the range of $1.43 - 24.4 \mu\text{g at l}^{-1}$ at Station 5. The highest phosphorus levels were recorded at Station 6 due to the polluted Bishopbriggs Burn. The values ranged between $3 \mu\text{g at l}^{-1}$ during November 1980 and $34 \mu\text{g at l}^{-1}$ during June 1981. The clean Allander Water, Station 7, with its relatively low contents which ranged between $1 - 22 \mu\text{g at l}^{-1}$ affected the levels slightly in the main river and values of $2.3 - 24.4 \mu\text{g at l}^{-1}$ were observed at Station 8. The phosphorus content in the River Kelvin had improved slightly as the river passed to Station 9 in which values ranged between $2 - 25.5 \mu\text{g at l}^{-1}$ were recorded. Values stayed almost constant at Station 10 ($2.7 - 25 \mu\text{g at l}^{-1}$).

4.1.7 Nitrate + Nitrite

The total nitrate + nitrite Nitrogen concentrations (Table 4.06) which were recorded along the stretch of the River Kelvin were high with significant variations seasonally and with the stations (Table 4.06). The values observed ranged between the minimum of $2.9 \mu\text{g at l}^{-1}$ at Station 10 during September 1981, when minimum levels for all the stations were recorded, and the maximum of $256.2 \mu\text{g at l}^{-1}$ observed at Station 4 during March 1981.

Table (4.06) Mean data for the total Nitrate + Nitrite concentration ($\mu\text{g at } \text{l}^{-1}$) recorded in the River Kelvin during the period February 1980–September 1982 and the two way ANOVA for the data.

Time	S T A T I O N S										Source	D.F.	S.S.	M.S.	F.factor	Error	Total
	1	2	3	4	5	6	7	8	9	10							
1980																	
Feb.	74.7	94.3	107	143	120	119	68.3	127	111	123							
March	80.3	82.7	72.7	111	87.0	78.9	63.2	94.5	96.9	96.8							
April	40.1	50.4	53.4	53.7	53.1	45.1	55.4	56.0	56.0	55.7							
May	38.7	56.2	84.3	90.0	93.0	88.6	31.1	90.8	86.5	86.5							
June	31.1	54.9	65.0	153	148	152	38.2	127	171	155							
July	40.6	67.3	70.1	124	102	115	49.3	75.6	80.0	83.3							
Aug.	15.6	29.6	33.9	208	184	127	19.2	189	195	183							
Oct.	26.7	38.7	31.2	206	124	220	27.4	177	173	187							
Nov.	58.4	48.2	38.3	93.3	70.3	61.8	39.3	75.6	85.2	88.7							
Dec.	66.3	46.9	47.7	80.0	57.8	70.5	42.7	64.3	67.0	88.2							
1981																	
Jan.	57.8	52.5	60.7	110	79.6	90.1	40.0	79.0	73.4	79.2							
Feb.	38.5	65.2	76.9	113	80.5	106	38.6	86.2	116	97.6							
March	105	120	121	256	183	194	109	178	181	188							
April	38.9	64.4	63.2	206	122	152	15.3	30.0	152	167							
May	29.0	42.6	35.1	150	88.1	87.0	138	87.7	90.3	88.4							
June	33.1	62.2	75.1	236	164	176	33.1	160	127	118							
July	14.3	46.1	43.0	82.4	55.1	74.4	9.38	58.7	59.4	52.6							
Aug.	17.9	77.8	72.0	215	153	158	24.3	131	136	126							
Sept.	12.1	11.9	15.0	22.3	18.8	20.8	15.1	19.9	20.6	2.90							
Oct.	23.9	32.4	39.5	53.9	46.6	59.1	21.1	63.8	54.3	60.1							
Nov.	31.6	29.9	23.4	42.1	31.8	37.4	21.2	34.2	30.3	37.2							
Dec.	36.0	45.2	55.3	100	74.0	70.6	26.7	66.2	63.4	74.8							
1982																	
Jan.	26.0	28.1	30.0	44.4	35.9	39.8	27.2	38.4	41.7	42.6							
Feb.	40.4	64.4	65.7	140	92.0	90.1	32.1	85.3	98.5	87.6							
March	36.0	46.3	52.2	83.8	71.7	78.7	28.7	88.6	86.8	85.9							
April	31.9	44.7	42.0	105	68.5	113	17.4	109	101	119							
May	29.3	56.5	54.4	202	122	136	14.2	121	131	131							
June	17.0	48.6	48.4	137	81.1	93.5	23.0	74.3	92.3	85.8							
July	23.6	114	113	125	111	113	36.4	131	120	25.6							
Aug.	24.6	46.1	41.6	113	56.6	77.3	37.6	64.7	53.9	58.8							
Sept.	42.6	52.2	61.2	105	65.2	78.8	56.7	74.8	77.7	81.4							

Rep. 1
1

0.023 0.21

0.023 0.23

619

Figure 4.07 shows the seasonal variations in the total nitrate + nitrite N content in the River Kelvin at the 10 different stations. Generally levels were higher in the spring-summer period (March-October) than the autumn-winter (November-February) period with a lot of fluctuation in both of the seasons. Levels were lower during the year 1982 than the years 1980 and 1981. Stations 1, 2, 3 and 7 followed more or less the same seasonal pattern having mostly similar levels which were relatively low compared with the rest of the stations.

Station 1 had the lowest nitrate + nitrite N content. Its levels did not vary very much seasonally. There was a noticeable increase in the content during March 1981 when the value of $105 \mu\text{g at } \ell^{-1}$ was observed. The minimum value recorded at Station 1 was $12 \mu\text{g at } \ell^{-1}$ during September 1981. Levels increased slightly as the Kelvin joined the polluted Dock Water and Broad Burn. The values recorded ranged between $12 - 120.4 \mu\text{g at } \ell^{-1}$. Values stayed mostly constant at Station 3. It seems that the Glazert Water did not have any effect on reducing the levels. Values ranging between a minimum of $15 \mu\text{g at } \ell^{-1}$ and a maximum of $120.5 \mu\text{g at } \ell^{-1}$ were observed. The polluted Luggie Water, Station 4, had the highest nitrate + nitrite N levels among all the stations. Its content ranged between $22.3 \mu\text{g at } \ell^{-1}$ and $256.2 \mu\text{g at } \ell^{-1}$.

These high nitrate + nitrite N contents caused a dramatic increase in the main river. Levels of $19 - 220 \mu\text{g at } \ell^{-1}$ were recorded at Station 5. The polluted Bishopbriggs Burn did not have any effect on the nitrate + nitrite N content in the Kelvin. The values stayed almost the same as the river passed to Station 6 in which a range of $21 - 220 \mu\text{g at } \ell^{-1}$ were observed. The clean Allander Water, Station 7,

Figure 4.07: The total nitrate + nitrite levels in $\mu\text{g at l}^{-1}$
observed in the River Kelvin during the period
February 1980 - September 1982.

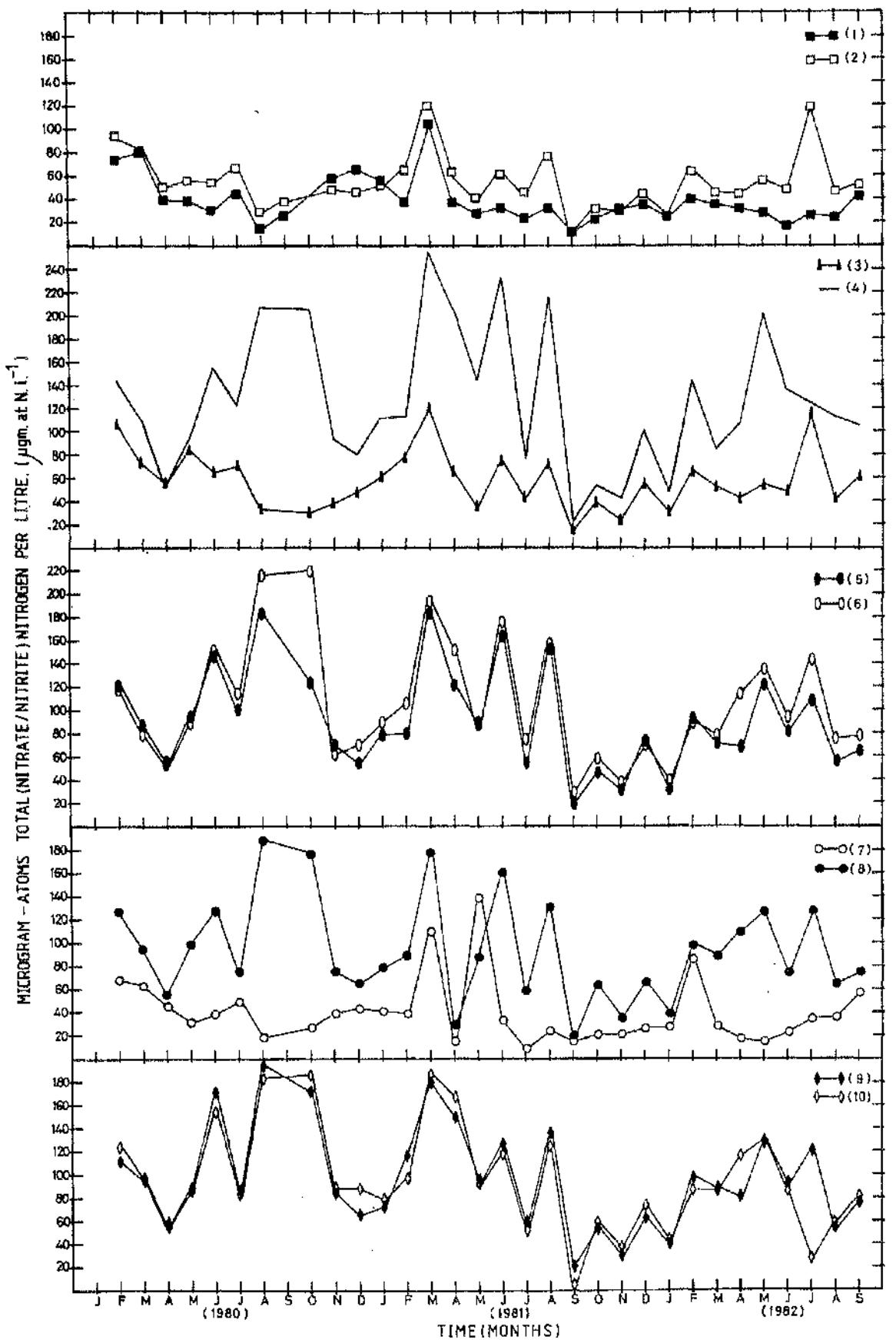


Figure 4.07

with its range of 9.4 - 138 μg at l^{-1} also did not have an effect on the nitrate + nitrite N levels in the main river. Values stayed almost constant for the rest of the river. There was only one noticeable reduction in the levels at Stations 7 and 8 and that was during April 1981.

Taking the nitrite levels into consideration, all the stations had followed the nitrate seasonal pattern with lower levels which ranged between 0.41 μg at l^{-1} recorded during March 1980 at Station 1 and 99.7 μg at l^{-1} observed at Station 4 during April 1981.

4.1.8 Ammonia

High levels for ammonia were recorded in the River Kelvin with variable concentrations ranging between a maximum of 6,660 $\mu\text{g l}^{-1}$ and a minimum of 7 $\mu\text{g l}^{-1}$ (Table 4.07). Highly significant variances were observed seasonally and locally with the stations (Table 4.07).

Figure 4.08 shows the seasonal variations of ammonia for all the stations. Obviously it indicates that all the stations, excluding Station 1, had followed a uniform annual pattern but with different levels. Values varied during the 3 years of the study. Generally ammonia levels were at maximum during spring (March-June) whilst relatively lower values were observed during the summer, autumn and winter (July-February).

During 1980 the concentrations were relatively low compared with the years 1981 and 1982. The highest values for all the stations, excluding Station 1, were recorded during spring 1981. There was another peak at Stations 4, 5, 6, 9 and 10. The levels went down very rapidly during July, August and September.

Table (4.07) Mean data for the Ammonia concentration ($\mu\text{g l}^{-1}$) recorded in the River Kelvin during the period February 1980-September 1982 and the two way ANOVA for the data.

Time	S T A T I O N S									
	1	2	3	4	5	6	7	8	9	10
<u>1980</u>										
Feb.	105	85	155	222	195	227	95	285	201	220
March	13	55	31	78	51	53	28	48	54	52
April	370	1240	864	1520	1480	1120	1242	1160	1170	660
May	100	715	567	662	522	570	392	585	435	210
June	110	1180	250	1164	562	184	570	370	200	140
July	172	470	220	90	174	214	40	154	164	50
Aug.	30	764	424	200	402	304	434	210	290	96
Oct.	464	964	610	1132	684	600	264	574	769	644
Nov.	140	280	100	460	264	300	120	252	252	248
Dec.	108	328	212	532	344	384	108	386	404	464
<u>1981</u>										
Jan.	120	384	308	324	292	384	92	312	308	322
Feb.	404	678	572	4440	4540	4540	256	682	4200	4680
March	80	300	180	380	240	400	170	320	340	240
April	540	1880	1160	2840	1800	900	940	220	720	420
May	180	2120	1400	2820	1760	1000	1260	1020	1020	400
June	360	2320	5400	3400	6480	6660	1700	6040	4940	1760
July	200	760	19	90	180	380	460	370	280	80
Aug.	247	508	476	34	282	105	392	148	25	7
Sept.	170	264	244	204	212	184	192	184	200	164
Oct.	164	750	530	1000	B02	842	490	740	770	748
Nov.	84	270	154	414	274	340	112	244	220	240
Dec.	630	948	908	1148	1028	1188	948	1080	1124	1260
<u>1982</u>										
Jan.	274	334	460	880	674	794	250	716	756	770
Feb.	299	625	430	687	633	706	362	655	683	615
March	123	429	299	485	496	453	734	460	413	513
April	360	2620	1560	5050	4080	1570	1270	1800	1800	840
May	420	1960	1170	1530	1430	780	1490	970	780	560
June	260	1180	810	1110	820	510	970	480	470	230
July	670	4426	1470	260	790	350	1640	570	370	260
Aug.	107	428	265	228	231	306	177	248	228	159
Sept.	58	289	158	107	195	186	90	187	135	136
Source	D.F.	S.S.	M.S.	F.Factor	Error	Total				
A Rep.	1	0.75	0.75		0.39					
A Time	30	398756019	13291867		8637617**					
					309					619
					475					475

Figure 4.08: Concentrations of ammonia in $\mu\text{g l}^{-1}$ observed
in the River Kelvin during the period
February 1980 – September 1982.

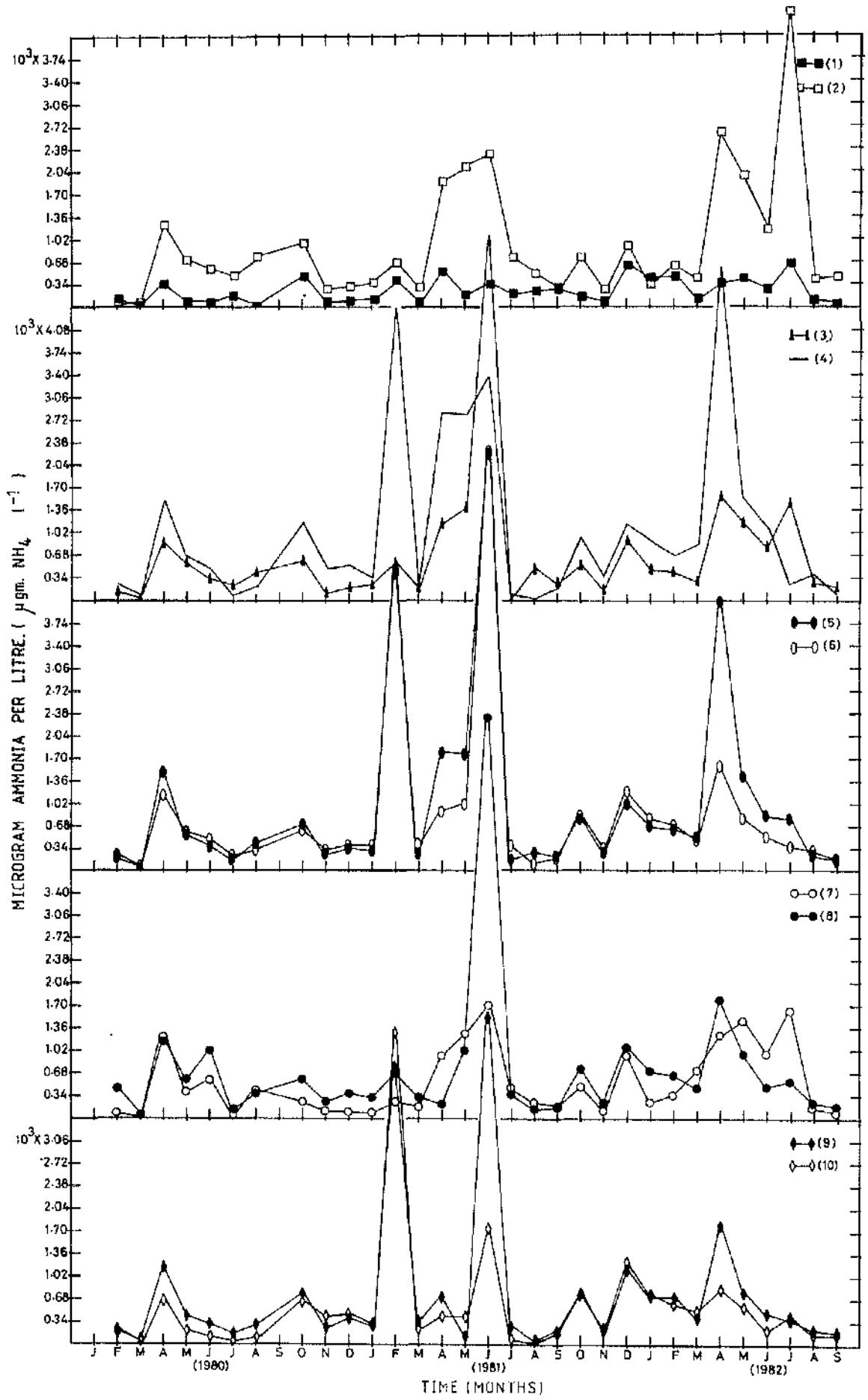


Figure 4.08

Station 1 with its lowest levels of all the stations did not show any real seasonal variation. Values stayed low, minimal all the time, ranging from $13 - 670 \mu\text{g l}^{-1}$. Levels increased as the river passed to Station 2 and joined the polluted Dock Water and Broad Burn, with $55 - 4,426 \mu\text{g l}^{-1}$ recorded. The ammonia content in the Kelvin decreased slightly as it mixed with the clean Glazert Water. A range of $19 - 5,400 \mu\text{g l}^{-1}$ was recorded at Station 3. The highest levels for all the stations were recorded at Station 4, Luggie Water, in which a range of $34 - 5,050 \mu\text{g l}^{-1}$ was observed. This high ammonia content increased the levels in the main river and values of $51 - 6,480 \mu\text{g l}^{-1}$ were recorded at Station 5. In spite of mixing with the polluted Bishopbriggs Burn, the Kelvin values stayed almost constant at Station 6 ($53 - 6,660 \mu\text{g l}^{-1}$).

Station 7, Allander Water, with its relatively low ammonia content of $28 - 1,700 \mu\text{g l}^{-1}$ did not have a real effect on the main river. The levels stayed almost the same for the rest of the river with a very slight improvement at Station 10 in which a range of $7 - 4,680 \mu\text{g l}^{-1}$ was observed.

4.1.9 Reactive silicate (dissolved silica)

Many fluctuations were observed in the silicate concentration along the River Kelvin. Values ranged between a minimum of $5.4 \mu\text{g l}^{-1}$ and a maximum of $281 \mu\text{g l}^{-1}$ (Table 4.08) at Stations 1 and 10 respectively. Significant variances were observed seasonally and locally with the stations (Table 4.08).

Table (4.08) Mean data for the reactive silicate ($\mu\text{g at } \text{L}^{-1}$) recorded in the River Kelvin during the period February 1980-September 1982 and the two way ANOVA for the data.

Time	S T A T I O N S									
	1	2	3	4	5	6	7	8	9	10
<u>1980</u>										
Feb.	155.4	157.5	155.4	153.4	155.4	155.4	125.4	150.3	154.4	154.4
March	124.0	124.0	105.9	131.8	113.7	124.0	116.3	121.5	124.0	124.0
April	122.0	105.0	93.20	65.40	67.30	67.70	66.00	65.20	57.20	42.80
May	203.2	163.3	129.2	137.9	129.2	122.6	80.50	115.40	110.3	105.2
June	180.5	149.8	130.5	140.0	135.4	139.8	98.7	123.2	120.5	124.9
July	168.8	141.3	137.6	162.4	147.7	152.9	105.5	139.2	136.6	135.0
Aug.	158.5	142.0	131.6	101.8	119.3	122.6	105.7	129.6	129.6	123.9
Oct.	164.7	158.1	141.5	132.1	141.0	155.3	124.9	141.5	160.5	146.3
Nov.	118.7	113.4	105.6	132.3	112.7	127.6	111.5	122.9	123.5	126.2
Dec.	116.2	103.2	98.70	111.3	104.1	111.0	101.9	113.4	117.9	119.8
<u>1981</u>										
Jan.	130.5	119.9	120.5	131.8	123.5	128.1	100.2	121.1	118.5	120.6
Feb.	263.7	257.4	162.9	149.0	157.7	161.3	116.1	142.1	156.5	150.1
Mar.	112.7	102.0	105.0	110.9	108.2	114.9	102.9	112.5	111.4	110.6
April	125.5	94.29	67.70	15.25	45.90	56.12	47.11	57.77	49.58	25.39
May	135.5	82.70	54.25	33.00	46.23	55.70	77.29	46.36	54.36	44.93
June	182.9	153.6	144.4	108.8	120.4	130.0	105.0	122.3	121.8	122.8
July	167.3	108.9	109.4	72.76	93.55	104.9	60.88	94.70	99.50	97.90
Aug.	142.8	114.0	107.1	32.70	63.90	62.20	62.30	58.20	42.10	22.60
Sept.	114.2	111.3	115.6	141.1	135.5	134.0	120.3	133.1	128.8	135.0
Oct.	108.3	107.8	107.0	105.0	106.2	107.2	84.79	106.8	106.8	106.0
Nov.	111.6	94.80	80.98	100.1	89.96	85.85	82.88	86.64	86.29	94.43
Dec.	225.0	194.0	189.9	171.8	150.2	183.5	148.5	280.8	189.4	186.4
<u>1982</u>										
Jan.	125.8	113.4	116.6	124.8	122.1	128.0	111.4	123.9	125.3	124.4
Feb.	154.7	144.1	118.6	106.8	115.3	123.7	97.46	114.0	121.6	117.0
March	155.5	141.8	130.9	104.6	121.8	131.4	103.4	127.3	129.1	170.9
April	173.0	101.1	81.24	35.12	57.39	52.19	62.31	50.34	39.91	14.55
May	139.7	89.95	70.05	36.65	54.45	45.95	69.90	49.18	43.69	48.09
June	145.2	140.5	131.9	138.1	135.2	135.7	94.79	127.6	128.6	121.9
July	180.0	118.8	95.30	162.5	54.72	20.12	62.15	14.09	5.35	5.54
Aug.	134.7	120.1	118.8	130.0	119.9	122.6	103.2	118.5	115.2	112.5
Sept.	161.9	138.1	142.6	154.7	145.3	150.3	127.1	141.4	138.7	138.1
Source	D.F.	S.S.	M.S.	F.factor	Error	Total				
Rep.	1	0.98	0.98	0.26	309	619				
A Time	30	6223883	20796	5594.6**	1148.6	994052				

The seasonal pattern (Fig. 4.09) did not show a fixed picture for the whole period of this study. Stations showed a uniform annual pattern with many fluctuations. Unlike the other nutrients, silicate values were higher during autumn-winter months when flow rates were recorded, and low during the spring-summer period which was the season of diatom growth and low discharges.

Generally during summer 1980 the silicate content in the River Kelvin was higher compared with the levels observed during summer 1981 and 1982. There was a noticeable silicate depletion in almost all of the stations during April and another reduction was observed at Station 10 only during December.

During 1981 there were two peaks for all the stations during February and December when the highest values for the whole period of this study were recorded. Reductions of the silicate content were observed during 1982 for Stations 3-10 during April and May and for Stations 5-10 during July.

However, unlike the other dissolved inorganic chemicals in the river, the highest reactive silicate levels were observed at Station 1. Levels decreased slightly as the Kelvin passed to Station 2. Joining the Glazert Water with the Kelvin, levels decreased further and stayed almost constant for the rest of the stations on the River Kelvin with a slight decrease at Station 7.

Figure 4.09: Reactive silicate concentration observed in the River Kelvin during the period February 1980 - September 1982.

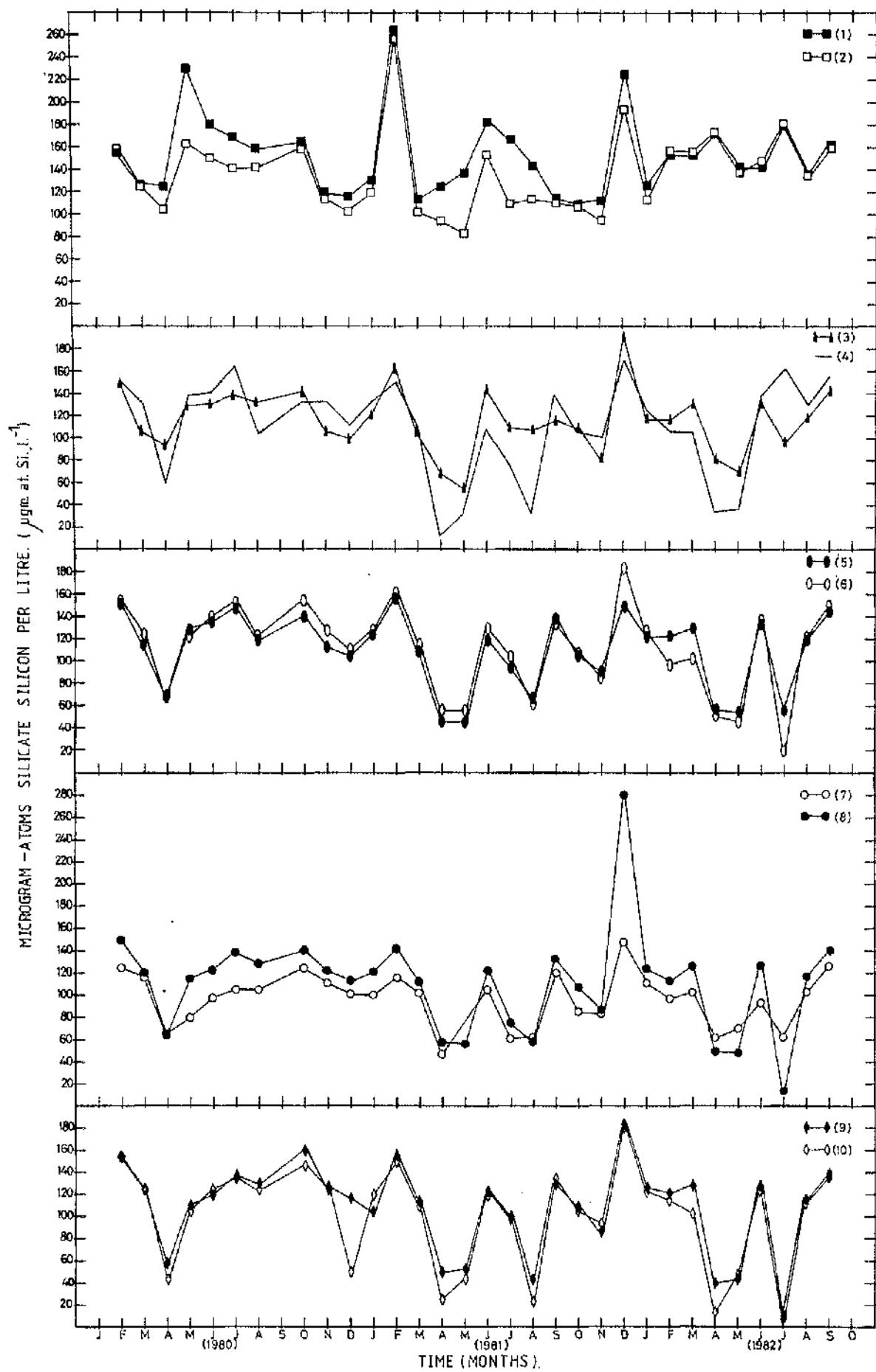


Figure 4.09

Taking the profiles for all the nutrients into consideration, Figures 4.10 and 4.11 displayed clearly the annual averages, maximum and minimum for the reactive phosphate, nitrate + nitrate N, ammonia and reactive silicate. For the first three nutrients it clearly indicates that Station 1 had the least contents. Levels increased as the river passed downstream. Luggie Water, Station 4, showed maximum values at all times, except for silicate, and Stations 3 and 7 displayed reductions.

The reactive phosphate profile showed an almost constant pattern for the average maximum and minimum values. The total nitrate + nitrite and ammonia profiles showed almost constant values at their minimum levels for all the stations but the maximum values fluctuated widely.

The average values within stations showed higher differences in the nitrate + nitrite profile compared with the ammonia profile, but the variance in the maximum values was higher in the latter.

4.2 Biological Data

4.2.1 The Phytoplankton

The term "phytoplankton" is used here to describe the algal cells suspending in the river. As will be seen, very few of the species observed can be regarded as typically planktonic, and the populations observed are mainly of benthic diatoms detached from substrata and then suspended in the river. Thus they would better be described as "tychoplanktonic".

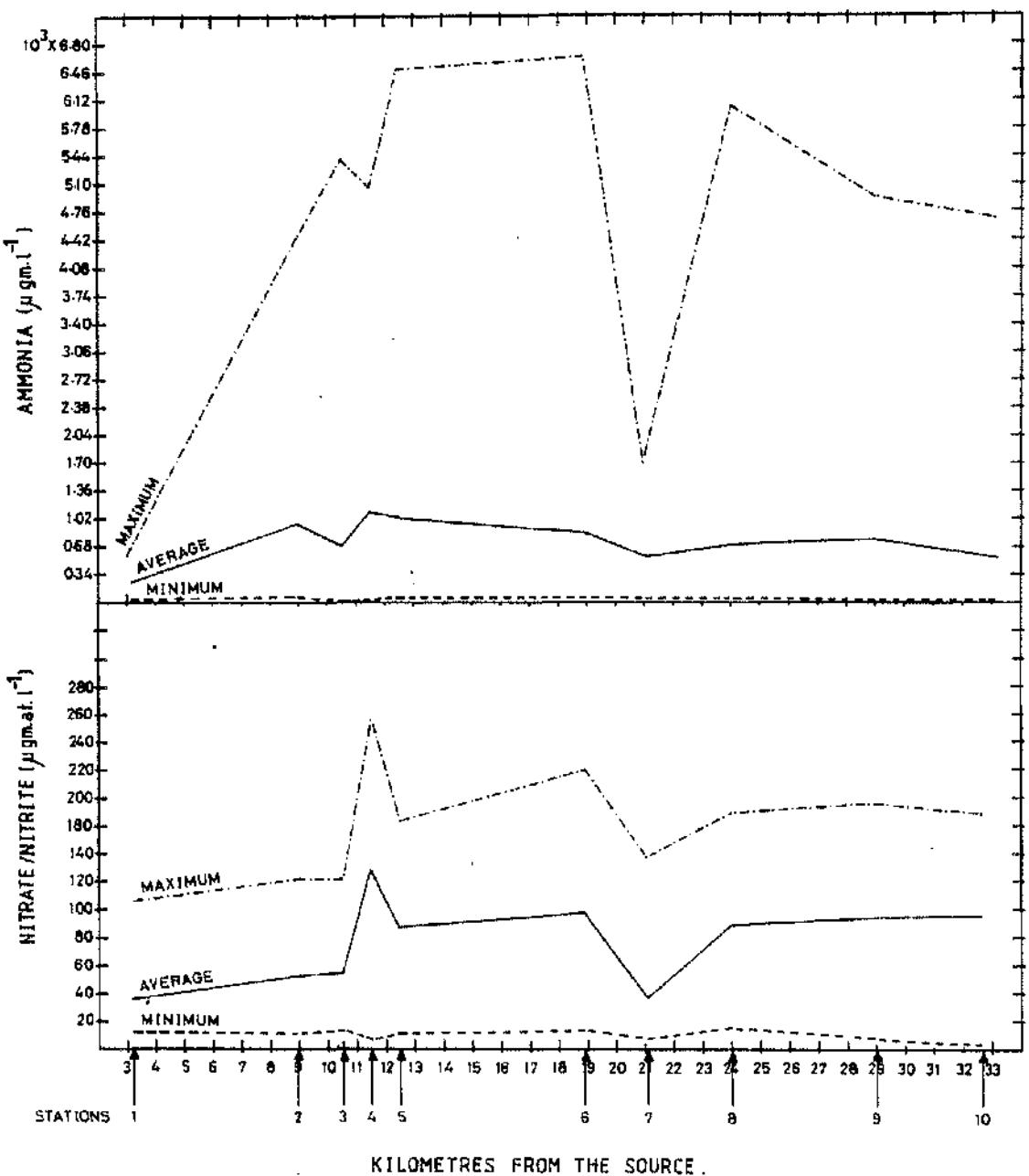


Figure 4.10: Profiles for the total nitrate + nitrite N in $\mu\text{g}\cdot\text{at. l}^{-1}$ and ammonia in $\mu\text{g}\cdot\text{l}^{-1}$ for the River Kelvin. The means obtained from the data for the entire sampling period for each specific station.

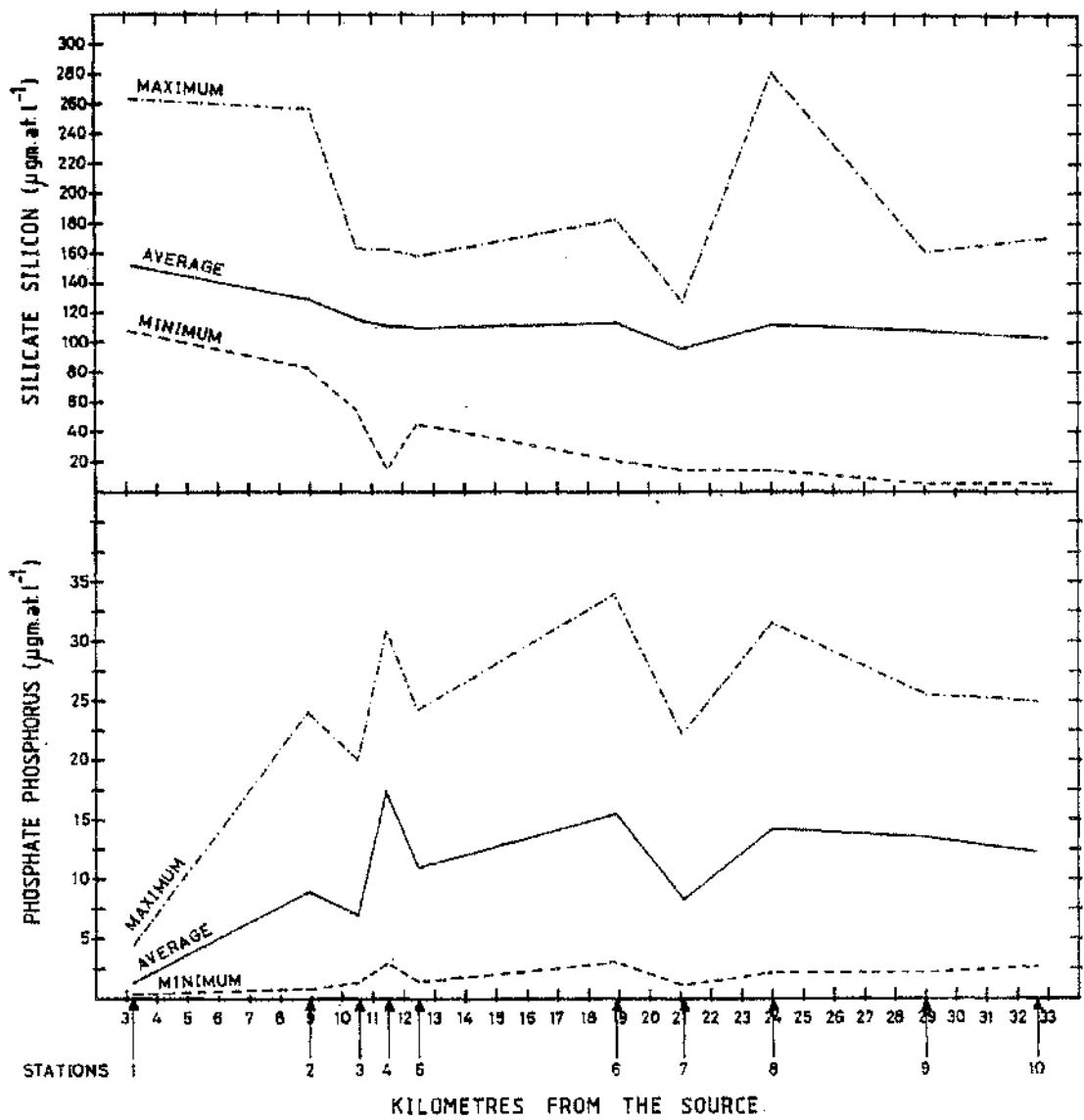


Figure 4.11: Reactive phosphorus and reactive silicate profiles in $\mu\text{g at } l^{-1}$ for the River Kelvin. The means obtained from the data for the entire sampling period for each specific station.

From the early qualitative examinations of the phytoplankton present at the 10 stations along the stretch of the river it was found that there was a marked diatom periodicity accompanied occasionally by a few unicellular green algae. The latter occurred on few occasions and in small numbers. These algae were - *Ankistrodesmus falcatus*, *Chlamydomonas* sp., *Buglena* sp., *Peroniella planktonica* G.M. Smith and *Scenedesmus quadricauda*. They were observed during summer at all the stations excluding Stations 1 and 7.

Thirty-eight diatom species were identified in the river samples. Similar species were found suspended in the water and epiphytic on the macrophytes. Table (4.09) gives a list of the diatoms recorded in the Kelvin during the triennial period of this study. Only one diatom species from the family Naviculaceae could not be identified (Plate 4.03, Fig. 19). Plates 4.01, 4.02 and 4.03 display the photographs of the common diatom after cleaning the frustules with concentrated hydrochloric acid. The quantitative data for the diatoms recorded in the river are shown in Figures 4.12a and b (cells l^{-1}) for the 10 stations; they also indicate the seasonal variations of the total diatom population suspended in the river. Generally low counts were observed during the whole period but with a spring-summer "outburst" at the time of reduced flow rates and relatively higher water temperature and increasing day length. This was during March-August with the major pulses between April-July. During the autumn-winter period, at the time of increased flow rates and low temperatures (September-February), very few diatoms were observed. Occasionally due to the dense suspended matter covering

Table 4.09 The systematic list of the diatoms recorded in the River Kelvin during the period April 1980 - September 1982

Achnanthes lanceolata (Breb) Grun. (Plate 4.02, Figure 11)

Cleve - Euler (1953), p. 241, Fig. 527

Patrick and Reimer (1966), p. 269, Pl. 18, Fig. 1-10

Bourrelly (1968), p. 306, Pl. 70, Fig. 14-18

Achnanthes subinflata Östr (Plate 4.02, Figure 12)

Cleve - Euler (1953), p. 241, Fig. 526

Asterionella formosa Hassal.

Smith (1856), Vol. 2, p. 81

Cleve - Euler (1953), p. 155, Fig. 401

Patrick and Reimer (1966), p. 159, Pl. 9, Fig. 1-3

Bourrelly (1968), p. 294, Pl. 63, Fig. 3-4

Ceratoneis (=Hannaea) arcus (Ehr.) Kütz.

Smith (1853), Vol. 1, p. 16, Pl. II, Fig. 15

Cleve - Euler (1953), p. 154, Fig. 373

Patrick and Reimer (1966), p. 132, Pl. 4, Fig. 20

Bourrelly (1968), p. 290, Pl. 64, Fig. 8

Cocconeis pediculus Ehr

Smith (1853), Vol. 1, Pl. III, Fig. 31

Cleve - Euler (1953), p. 240, Fig. 494

Patrick and Reimer (1966), p. 240, Pl. 15, Fig. 3-4

contd.

Coccoeis placentula Ehr. (Plate 4.03, Figure 23a & b)

Smith (1853), p. 21, Pl. III, Fig. 32

Cleve - Euler (1953), p. 240, Fig. 492

Patrick and Reimer (1966), p. 240, Pl. 15, Fig. 7

Bourrelly (1968), p. 304, Pl. 66, Fig. 2-3

Cyclotella meneghiniana Kütz. (Plate 4.02, Figure 14)

Cleve - Euler (1951), p. 154, Fig. 63

Bourrelly (1968), p. 264, Pl. 54, Fig. 10

Cymbella ventricosa Kütz. (Plate 4.02, Figure 10)

Cleve - Euler (1955), p. 227, Fig. 1177

Bourrelly (1968), p. 354, Pl. 95, Fig. 5

Diatoma elongatum Ag.

Smith (1856), Vol. 2, p. 40, Pl. XL, Fig. 311

Cleve - Euler (1953), p. 151, Fig. 330

Diatoma vulgare Bory. (Plate 4.02, Figure 15)

Smith (1856), Vol. 2, p. 39, Pl. XL, Fig. 309

Cleve - Euler (1953), p. 151, Fig. 329

Patrick and Reimer (1966), p. 109, Pl. 2, Fig. 9

Bourrelly (1968), p. 286, Pl. 59, Fig. 5

Eunotia curvata (Kütz.) Lagerst. (Plate 4.02, Figure 7)

Patrick and Reimer (1966), p. 189, Pl. 10, Fig. 4

contd.

Fragilaria capucina Desm. (Plate 4.01, Figure 5)

Smith (1856), Vol. 2, p. 22, Pl. XXXV, Fig. 296

Cleve - Euler (1953), p. 153, Fig. 357

Patrick and Reimer (1966), p. 118, Pl. 3, Fig. 5

Bourrelly (1968), p. 292, Pl. 63, Fig. 9

Fragilaria virescens Ralfs. (Plate 4.01, Figure 4)

Smith (1856), Vol. 2, p. 22, Pl. XXXV, Fig. 297

Cleve - Euler (1953), p. 153, Fig. 361

Patrick and Reimer (1966), p. 119, Pl. 3, Fig. 7-9

Bourrelly (1968), p. 292, Pl. 64, Fig. 9-10

Gomphonema constrictum Ehr. (Plate 4.03, Figure 24)

Smith (1853), Vol. 1, p. 78, Pl. XXVIII, Fig. 236

Cleve - Euler (1955), p. 230, Fig. 1261

Bourrelly (1968), p. 364, Pl. 96, Fig. 1

Gomphonema olivaceum (Lyngb.) Kutz. (Plate 4.03, Figures 25 & 26)

Smith (1853), p. 80, Pl. XXIX, Fig. 244

Cleve - Euler (1955), p. 231, Fig. 1291

Bourrelly (1968), p. 364, Pl. 96, Fig. 7

Gomphonema parvulum Kutz. (Plate 4.03, Figure 27)

Cleve - Euler (1955), p. 230, Fig. 1269

Dawson (1972)

contd.

Hantzschia amphioxys (E.) Grun. erw

Cleve - Euler (1952), p. 147, Fig. 1419

Bourrelly (1968), p. 376, Pl. 103, Fig. 2

Melosira granulata (Ehr.) Ralfs. (Plate 4.02, Figure 9)

Cleve - Euler (1951), p. 152, Fig. 15

Bourrelly (1968), p. 261, Pl. 55, Fig. 5

Melosira herzogii Lemm.

Cleve - Euler (1951), p. 152, Fig. 19

Bourrelly (1968), p. 261, Pl. 55, Fig. 1

Melosira varians Ag. (Plate 4.02, Figure 8)

Smith (1856), Vol. 2, p. 57, Pl. 11, Fig. 332

Cleve - Euler (1951), p. 152, Fig. 20

Crawford (1971)

Meridion circulare (Grev.) Ag.

Smith (1856), Vol. 2, p. 6, Pl. XXXII, Fig. 277

Cleve - Euler (1953), p. 151, Fig. 312

Patrick and Reimer (1966), p. 113, Pl. 2, Fig. 15

Bourrelly (1968), p. 288, Pl. 60, Fig. 5

Navicula avenacea Breb. (Plate 4.03, Figure 16)

Cleve - Euler (1953), p. 250, Fig. 807

contd.

Navicula contraria Patr. (Plate 4.03, Figure 17)

Patrick and Reimer (1966), p. 530, Pl. 50, Fig. 16

Navicula sp. (Plate 4.03, Figure 19)

Navicula cuspidata Kütz. (Plate 4.03, Figure 20)

Patrick and Reimer (1966), p. 464, Pl. 43, Fig. 9-10

Bourrelly (1968), p. 336, Pl. 84, Fig. 9

Navicula rhyncocephala Kütz. (Plate 4.03, Figure 18)

Smith (1853), Vol. I, p. 47, Pl. XVI, Fig. 132

Cleve - Euler (1952), p. 251, Fig. 817

Patrick and Reimer (1966), p. 505, Pl. 48, Fig. 6-8

Nitzschia acicularis W. Sm.

Smith (1853), p. 43, Pl. XV, Fig. 122

Cleve - Euler (1952), p. 150, Fig. 1509

Bourrelly (1968), p. 382, Pl. 103, Fig. 14

Nitzschia dissipata (Kütz.) Grün.

Cleve - Euler (1952), p. 149, Fig. 1463

Bourrelly (1968), p. 382, Pl. 103, Fig. 12-13

Nitzschia linearis W. Sm. (Plate 4.03, Figure 22)

Cleve - Euler (1952), p. 149, Fig. 1480

Bourrelly (1968), p. 378, Pl. 105, Fig. 1 & 2

contd.

Nitzschia sigma W.Sm.

Smith (1853), Vol. 1, p. 39, Pl. XIII, Fig. 108

Cleve - Euler (1952), p. 149, Fig. 1470

Bourrelly (1968), p. 378, Pl. 105, Fig. 5-7

Nitzschia thermalis (Kütz.) Grün. (Plate 4.03, Figure 21)

Cleve - Euler (1952), p. 148, Fig. 1445

Pinnularia divergens W. Smith

Smith (1853), Vol. 1, p. 57, Pl. XVIII, Fig. 177

Cleve - Euler (1955), p. 222, Fig. 1071

Patrick and Reimer (1966), p. 603, Pl. 56, Fig. 1-4

Bourrelly (1968), p. 344, Pl. 89, Fig. 8

Rhoicosphenia curvata (Kütz.) Grün. erw.

Cleve - Euler (1953), p. 244, Fig. 601

Patrick and Reimer (1966), p. 282, Pl. 20, Fig. 1-5

Bourrelly (1968), p. 306, Pl. 66, Fig. 4-6

Stauroneis phoenicenteron Ehr.

Smith (1853), Vol. I, p. 59, Pl. XIX, Fig. 185

Cleve - Euler (1953), p. 255, Fig. 944

Patrick and Reimer (1966), p. 359, Pl. 29, Fig. 3-4

Bourrelly (1968), p. 332, Pl. 82, Fig. 3

Surirella striatula Turpin. (Plate 4.02, Figure 13)

Smith (1853), Vol. 1, p. 32, Pl. IX, Fig. 64

Cleve - Euler (1952), p. 153, Fig. 1569

Synedra acus Kütz. (Plate 4.01, Figure 3)

Smith (1853), Vol. 1, p. 71, Pl. XII, Fig. 89

Cleve - Euler (1953), p. 154, Fig. 385

Patrick and Reimer (1966), p. 135, Pl. 5, Fig. 1

Synedra pulchella (Ralfs) Kütz. (Plate 4.01, Figure 1)

Smith (1853), p. 70, Pl. XI, Fig. 84

Cleve - Euler (1953), p. 154, Fig. 387

Patrick and Reimer (1966), p. 146, Pl. 6, Fig. 10

Bourrelly (1968), p. 294, p. 64, Fig. 3

Synedra ulna (Nitz.) Ehr. (Plate 4.01, Figure 2a, b & c)

Smith (1853), p. 71, Pl. XI, Fig. 90

Cleve - Euler (1953), p. 154, Fig. 382

Patrick and Reimer (1966), p. 148, Pl. 7, Fig. 1-2

Bourrelly (1968), p. 294, p. 64, Fig. 5

Tabellaria fenestrata (Lyngb.) Kütz. (Plate 4.02, Figure 6)

Smith (1856), Vol. 2, p. 46, Pl. XLIII, Fig. 317

Cleve - Euler (1953), p. 150, Fig. 303

Patrick and Reimer (1966), p. 103, Pl. 1, Fig. 1-2

Bourrelly (1968), p. 295, Pl. 62, Fig. 8-10

contd.

Tabellaria flocculosa (Roth.) Kütz.

Smith (1856), Vol. 2, p. 45, Pl. XLIII, Fig. 316

Cleve - Euler (1953), p. 150, Fig. 302

Patrick and Reimer (1966), p. 104, Pl. 1, Fig. 4-5

Bourrelly (1968), p. 295, Pl. 62, Fig 10-11

Plates 4.01, 4.02, 4.03

The diatoms recorded in the River Kelvin during the period
April 1980 - September 1982 (see Table 4.09).

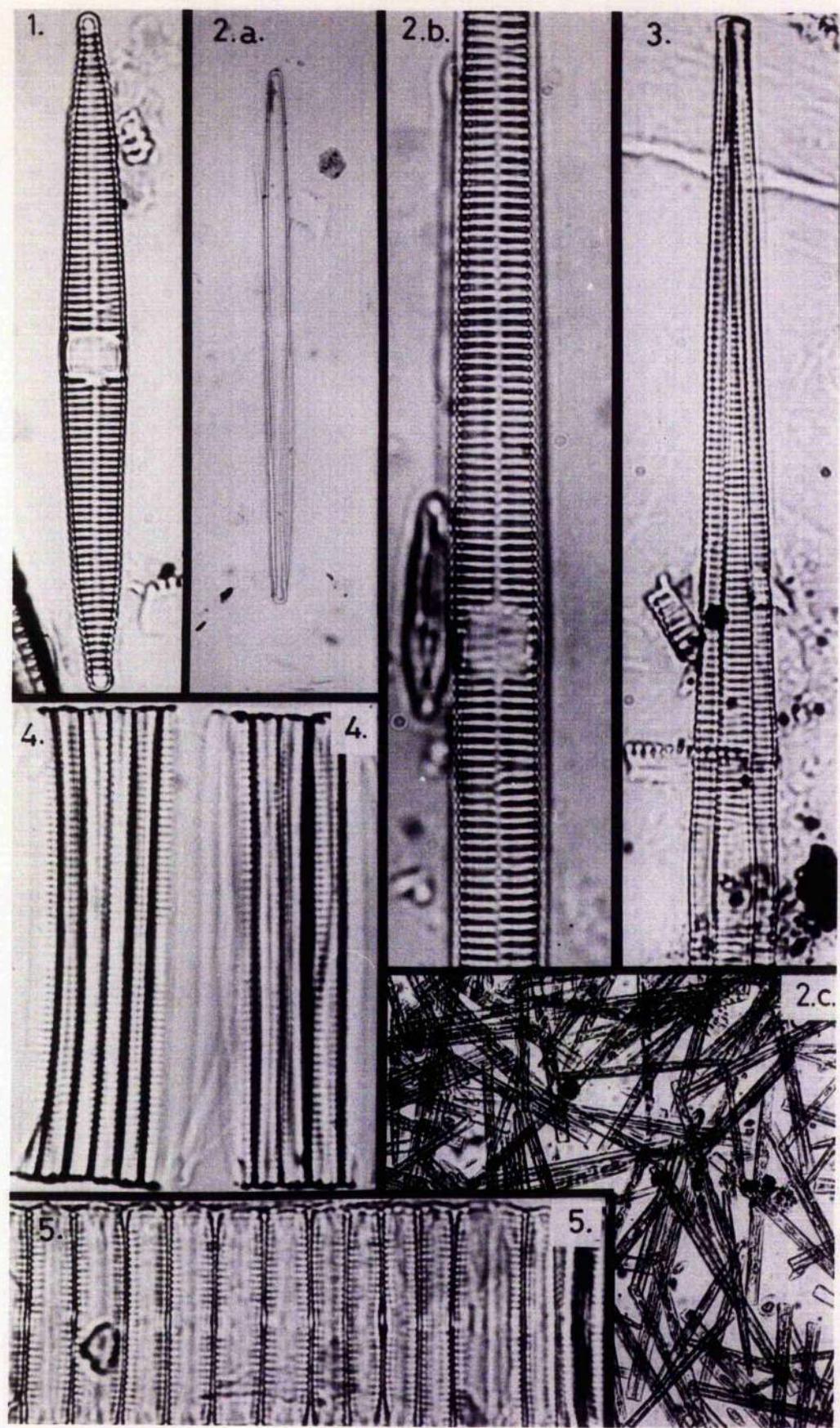


Plate (4.01)

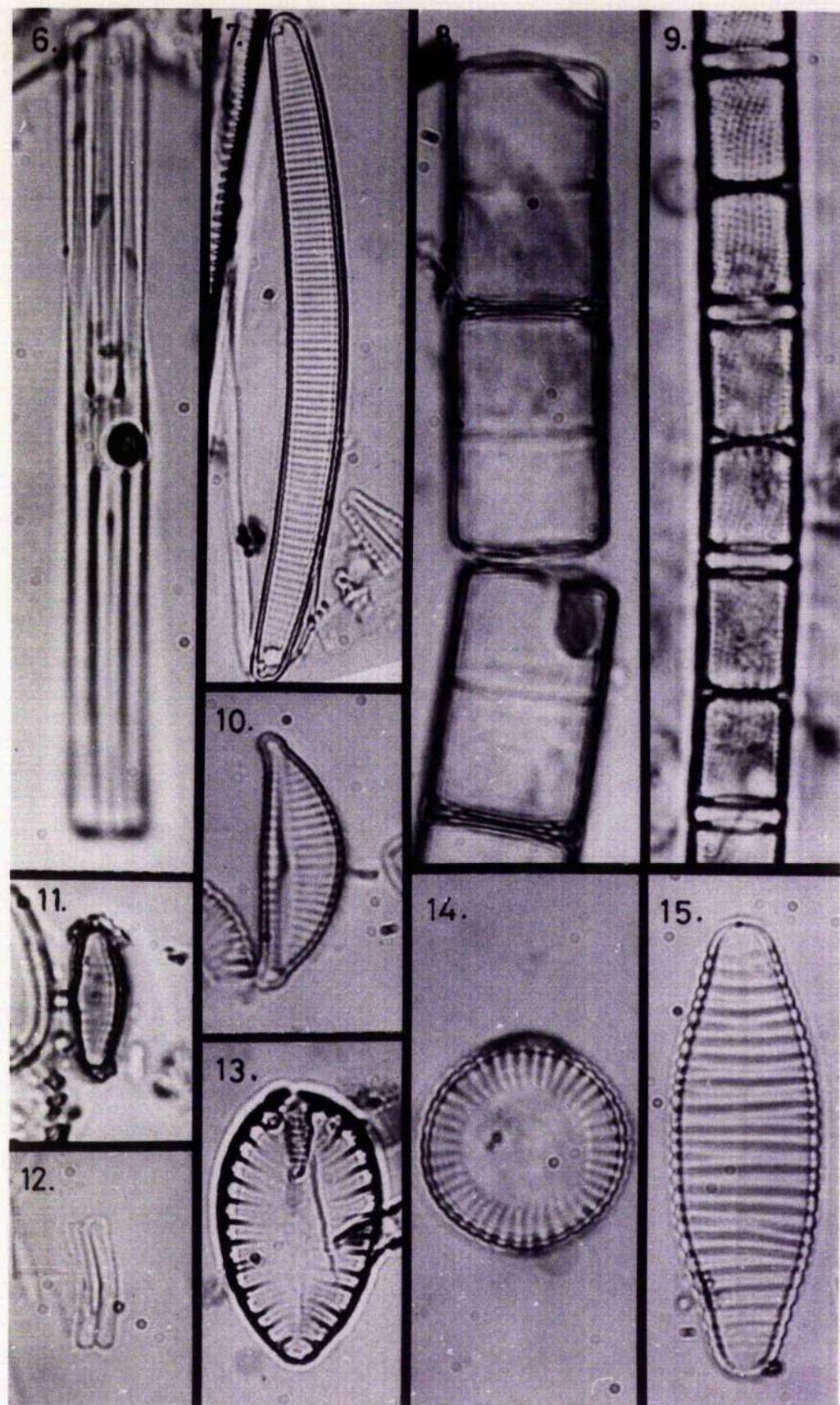


Plate (4.02)

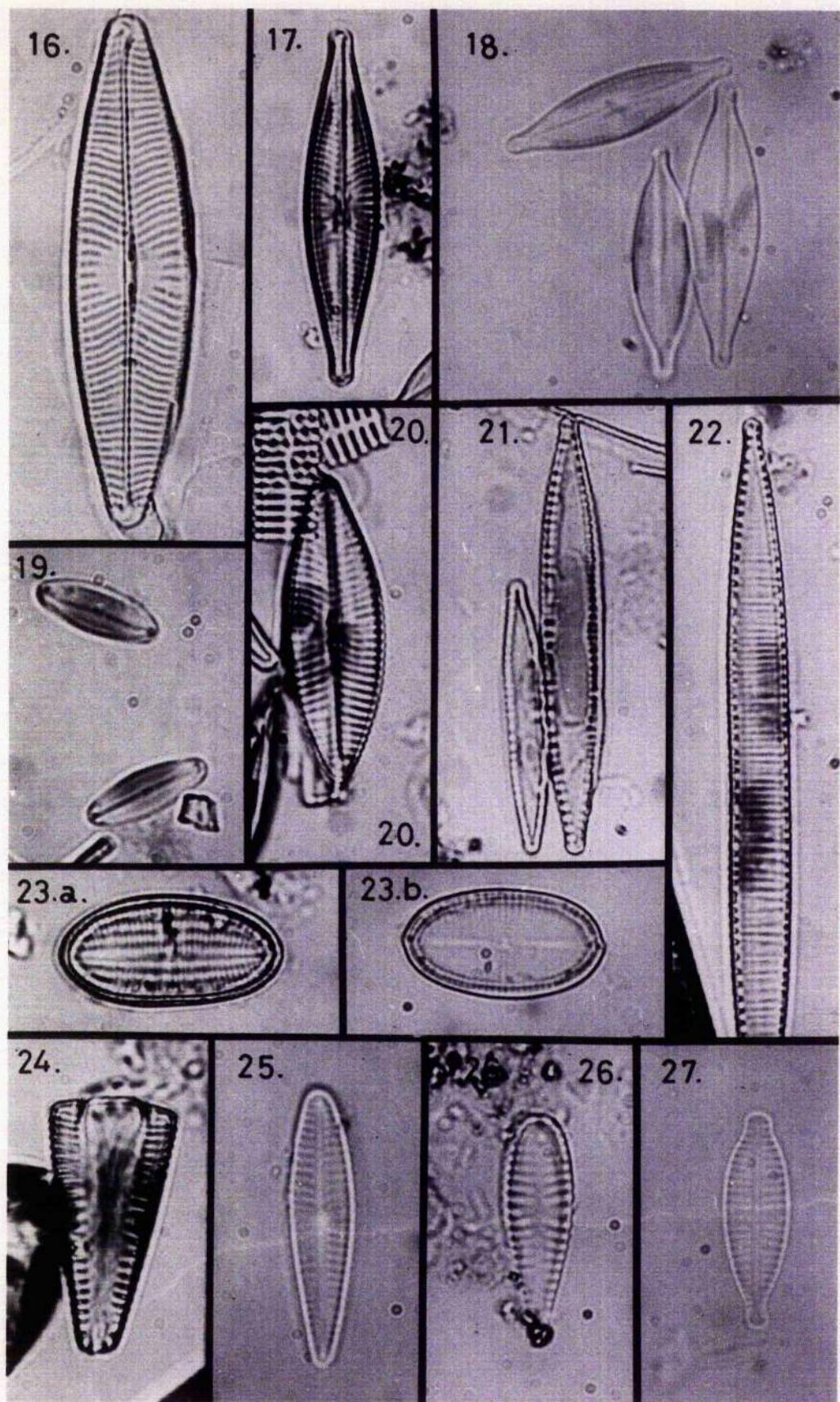


Plate 4.03)

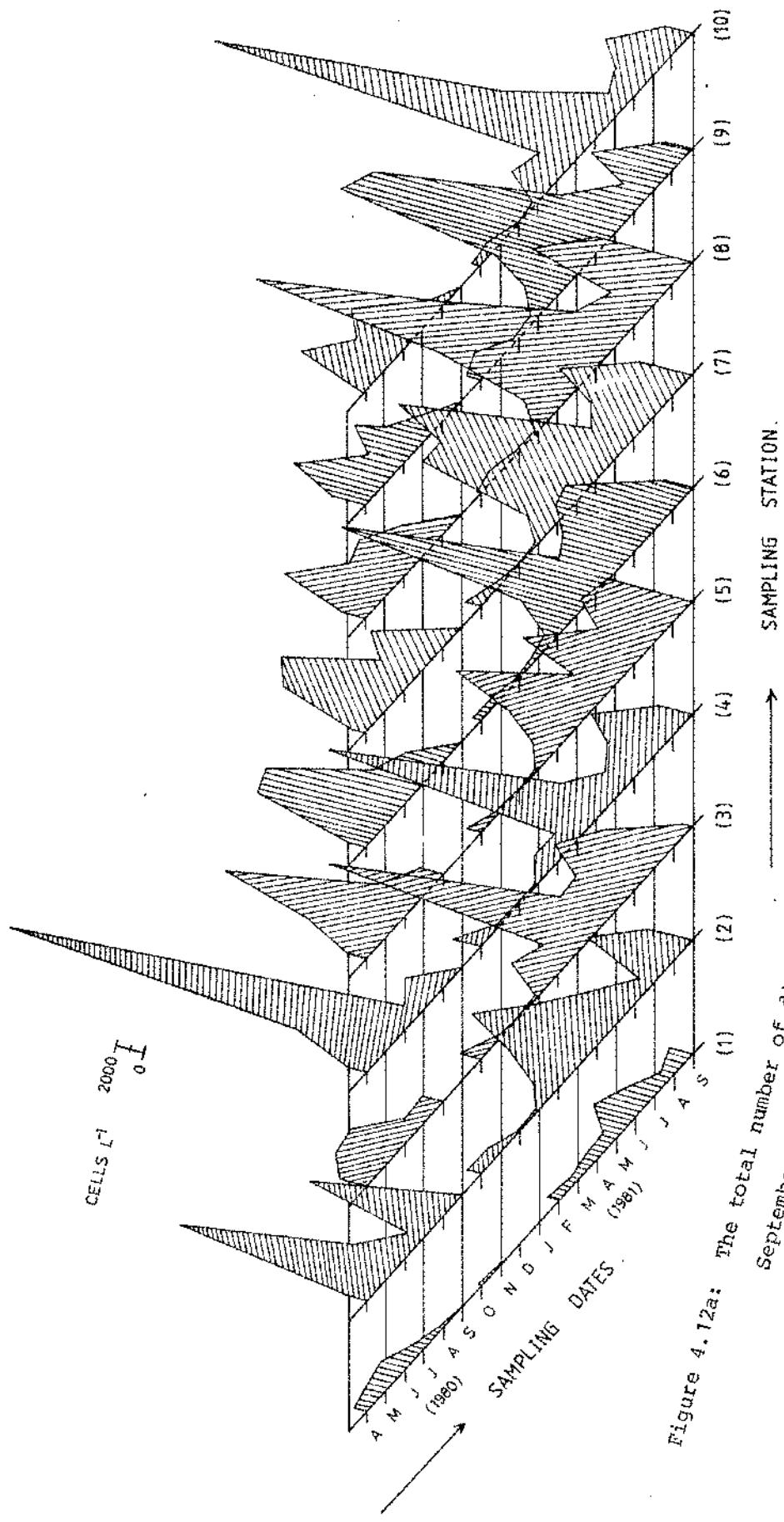


Figure 4.12a: The total number of diatoms recorded in the River Kelvin during April 1980 - September 1981 at its 10 sampling stations.

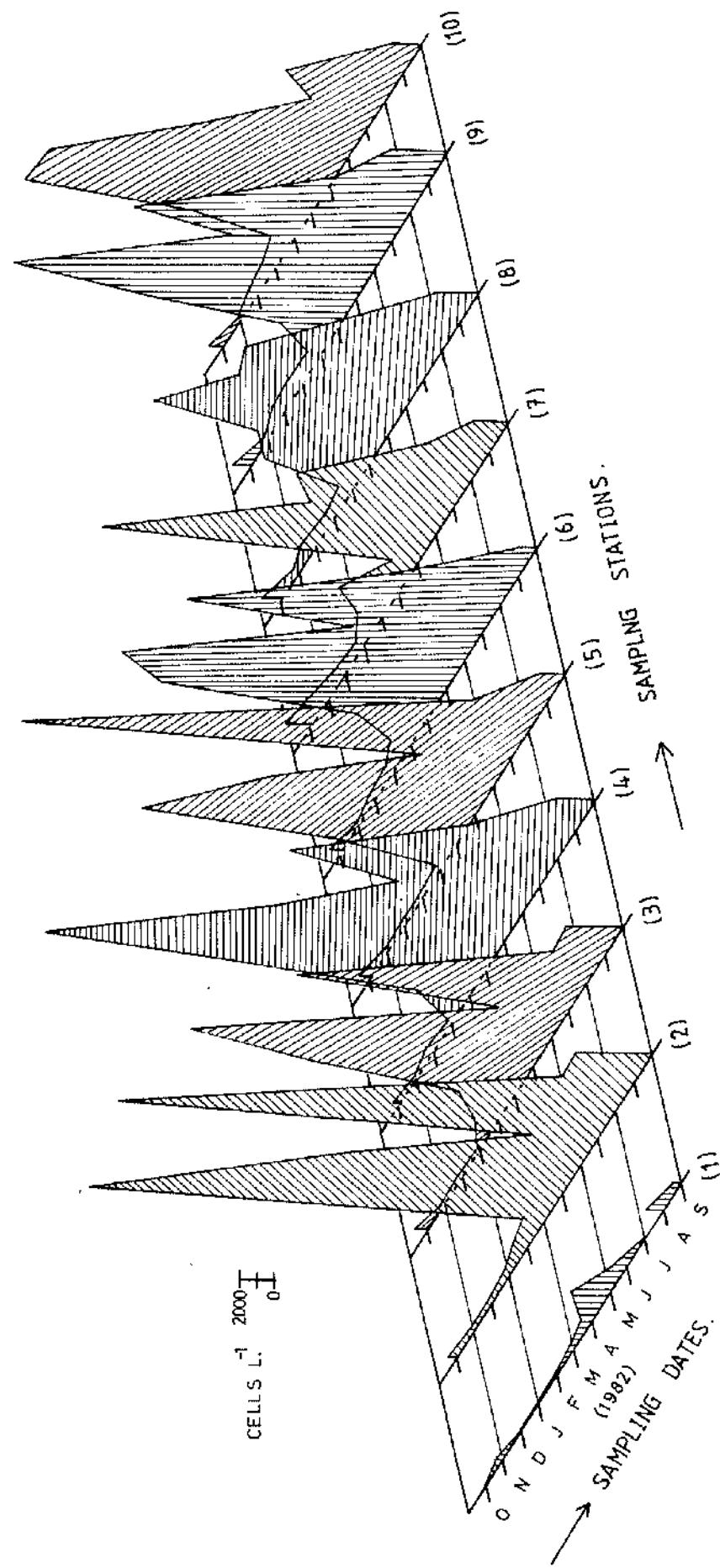


Figure 4.12b: The total number of diatoms in $cells\ l^{-1}$ recorded in the River Kelvin during October 1981 - September 1982 at its 10 sampling stations.

the membrane filter it was very difficult to count the diatoms, particularly at Station 1 where counts were often zero.

The total number of diatoms were relatively low during the year 1980 with one peak during May which continued into June for the river as a whole. The highest peak of 28,133 cells L^{-1} was recorded at Station 4 during June. The numbers increased slightly during 1981 with a pulse during April being the maximum at Station 10 (27,000 cells L^{-1}). Another peak occurred during August at all the stations, excluding Station 1 when the maximum of 10,000 cells L^{-1} was recorded at Station 8. During 1982 the diatom numbers for all the stations were higher than in the former two years. There were two main "pulses", the first one during May in which a maximum of 27,051 cells L^{-1} was observed at Station 4, and the second one during July with the highest numbers at Station 5 (34,488 cells L^{-1}). The peak for Station 1 (2,386 cells L^{-1}) was observed during May. There was a noticeable reduction for all the stations during June, being more severe at Stations 2-7. Station 1 had the smallest number of all the stations with numbers ranging from (0-8,300 cells L^{-1}) during January and April 1981 respectively. The total number of the diatoms increased downstream where the river was joined by its tributaries. The rest of the stations had more or less the same total diatom numbers. The high peaks occurred conspicuously at different stations but cell numbers were especially high at Station 4. Station 10 appeared different from the rest of the stations, when its high peak during April continued until May. The cell numbers fell during July but another pulse followed in August.

Two features were striking. The first was the general spring and summer growth peaks with autumn and winter reductions and the second was the individual characteristics of each station. It must be remembered that a river is a dynamic system. Any samples represent the suspended organisms borne along at a point in time and coming from sources upstream. Hence any analysis of the river phytoplankton has to bear in mind the course of events upstream and at any one station we are recording the suspended diatoms carried in by the various inflows.

A few species of diatoms were almost always recorded along the stretch of the river during the main annual growth period. These were - *Cyclotella meneghiniana*, *Gomphonema parvulum*, *Navicula avenacea*, *Nitzschia thermalis* and *Synedra ulna*. They appeared either as a single dominant species or a few of them together in different quantities and accompanied by other species occurring occasionally at various stations. Figure 4.13 displays the number and the seasonal pattern of the diatoms in the river at the 10 stations during the period April 1980 - September 1982. Station 1 was generally characterised by low numbers of diatoms through the years of this survey. Exceptional periods were during May and June 1980 when *Synedra ulna* ($720 \text{ cells } l^{-1}$) and *Gomphonema parvulum* ($2,457 \text{ cells } l^{-1}$) respectively dominates the river at this station. These two were part of a spring outburst and were typical for the river as a whole during 1980. Both were observed at Station 2 during May (*Gomphonema* $1,555 \text{ cells } l^{-1}$; *Synedra* $11,000 \text{ cells } l^{-1}$) and June ($1,307 \text{ cells } l^{-1}$ for the former; $560 \text{ cells } l^{-1}$ for the latter). This station was the

Figure (4.13) The seasonal variations in the suspended diatoms recorded for the River Kelvin at the 10 stations during the period April 1980-September 1982. The relative widths of the bars represent the number of cells cm^{-3} . The scale is as follows:-

10 9 8 7 6 5 4 3 2 1

1	<	500
2	>	500 < 1000
3	>	1000 < 2000
4	>	2000 < 3000
5	>	3000 < 4000
6	>	4000 < 5000
7	>	5000 < 7000
8	>	7000 < 10000
9	>	10000 < 15000
10	>	>15000

STATION (1)

LIST OF THE DIATOMS	1980					1981					1982					
	A	M	J	J	A	O	N	D	J	F	M	A	M	J	J	A
<i>A. lanceolata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>A. formosa</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. arcus</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. placentula</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. meneghiniana</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. ventricosa</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>D. elongatum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>D. vulgare</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>E. curvata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>F. virgescens</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>G. constrictum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>G. olivaceum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>G. parvulum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>H. amphioxys</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>M. harzagii</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>M. varians</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>M. circulare</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. avenacea</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. contraria</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>Navicula sp.</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. cuspidata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. rhyncocephala</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. acicularis</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. dissipata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. linearis</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>P. divergens</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>R. curvata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>S. phoenicenteron</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>S. striatula</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>S. ulna</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>T. fenestrata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>T. flocculosa</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

contd.

STATION (2)

LIST OF THE DIATOMS	1980					1981					1982					
	A	M	J	J	A	O	N	D	J	F	M	A	M	J	J	A
<i>A. lanceolata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>A. formosa</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. arcus</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. placentula</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. meneghiniana</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. ventricosa</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>D. elongatum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>D. vulgare</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>E. curvata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>F. virescens</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>G. constrictum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>G. olivaceum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>B. parvulum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>M. herzogii</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>M. varians</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>M. circulare</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. avenacea</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. contraria</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>Navicula sp.</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. cuspidata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. rhyncocephala</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. acicularis</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. dissipata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. linearis</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. thermalis</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. sigma</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>P. divergens</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>R. curvata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>S. phoenicenteron</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>S. striatula</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>S. ulna</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>T. fenestrata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
<i>T. flocculosa</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

contd.

STATION (3)

LIST OF THE DIATOMS	1980			1981			1982					
	A	M	J	A	O	N	D	J	F	M	A	S
<i>A. lanceolata</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>A. formosa</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. arcus</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. placentula</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. meneghiniana</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. ventricosa</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>D. vulgare</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>E. curvata</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>F. capucina</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>F. virescens</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>G. constrictum</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>G. olivaceum</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>G. parvulum</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>H. amphioxys</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>M. varians</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>M. circulare</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. avenacea</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. contraria</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Navicula sp.</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. cuspidata</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. rhyncocephala</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. dissipata</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. linearis</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>N. thermalis</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>P. divergens</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>R. curvata</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>S. phaeocenteron</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>S. striatula</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>S. ulna</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>T. flocculosa</i>	—	—	—	—	—	—	—	—	—	—	—	—

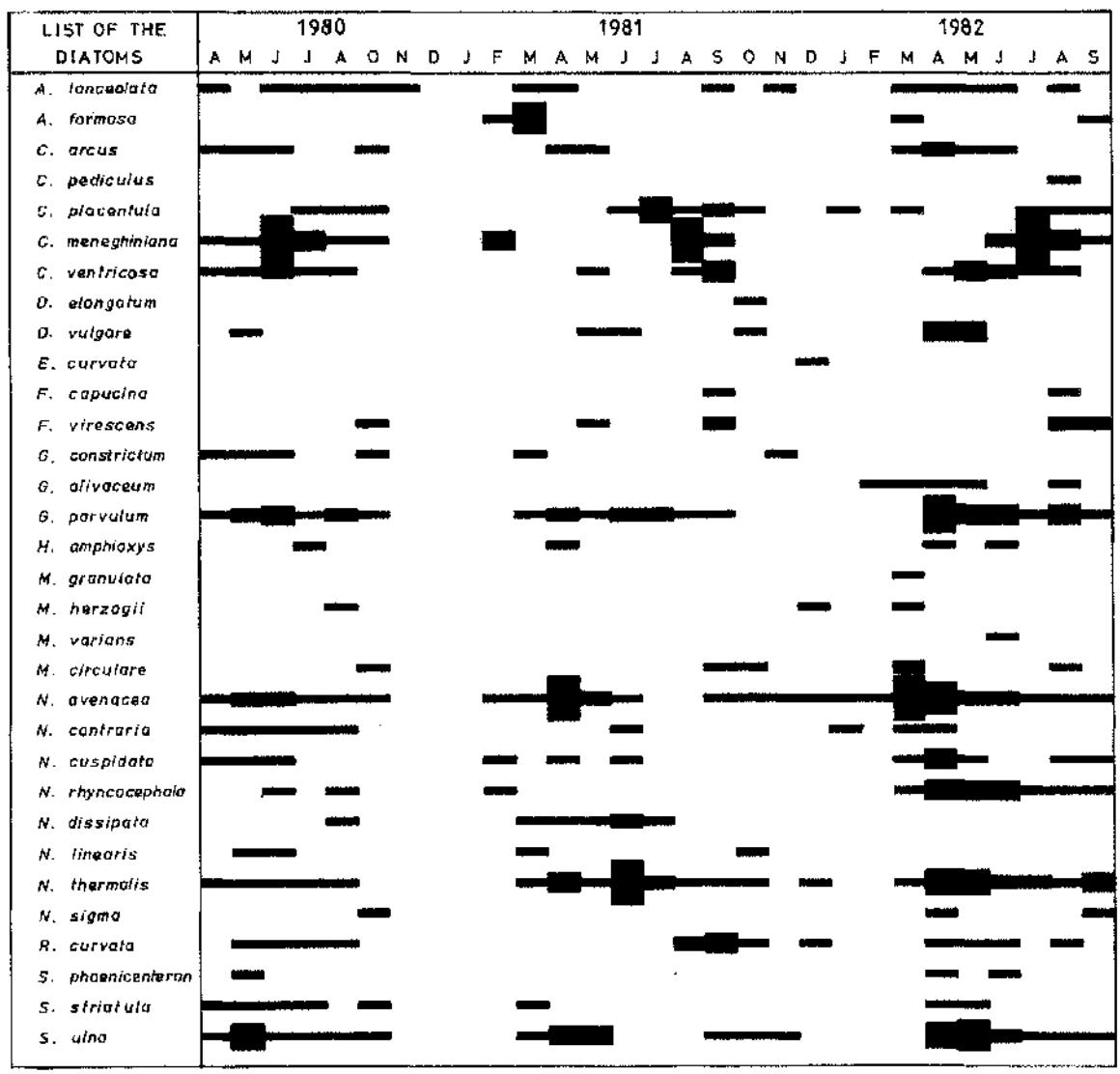
contd.

STATION (4)

LIST OF THE DIATOMS	1980						1981						1982					
	A	M	J	J	A	O	N	D	J	F	M	A	M	J	J	A	S	
<i>A. lanceolata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>A. formosa</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>C. pediculus</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>C. placentula</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>C. meneghiniana</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>C. ventricosa</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>D. elongatum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>D. vulgare</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>E. curvata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>F. virescens</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>G. constrictum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>G. olivaceum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>G. parvulum</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>H. amphioxys</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>M. varians</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>M. circulaire</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>N. avanacea</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>N. contraria</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>N. cuspidata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>N. rhyncocophala</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>N. dissipata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>N. linearis</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>N. thermalis</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>N. sigma</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>P. divergens</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>R. curvata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>S. striatula</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>S. ulna</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	
<i>T. fenestrata</i>	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	

contd.

STATION (5)



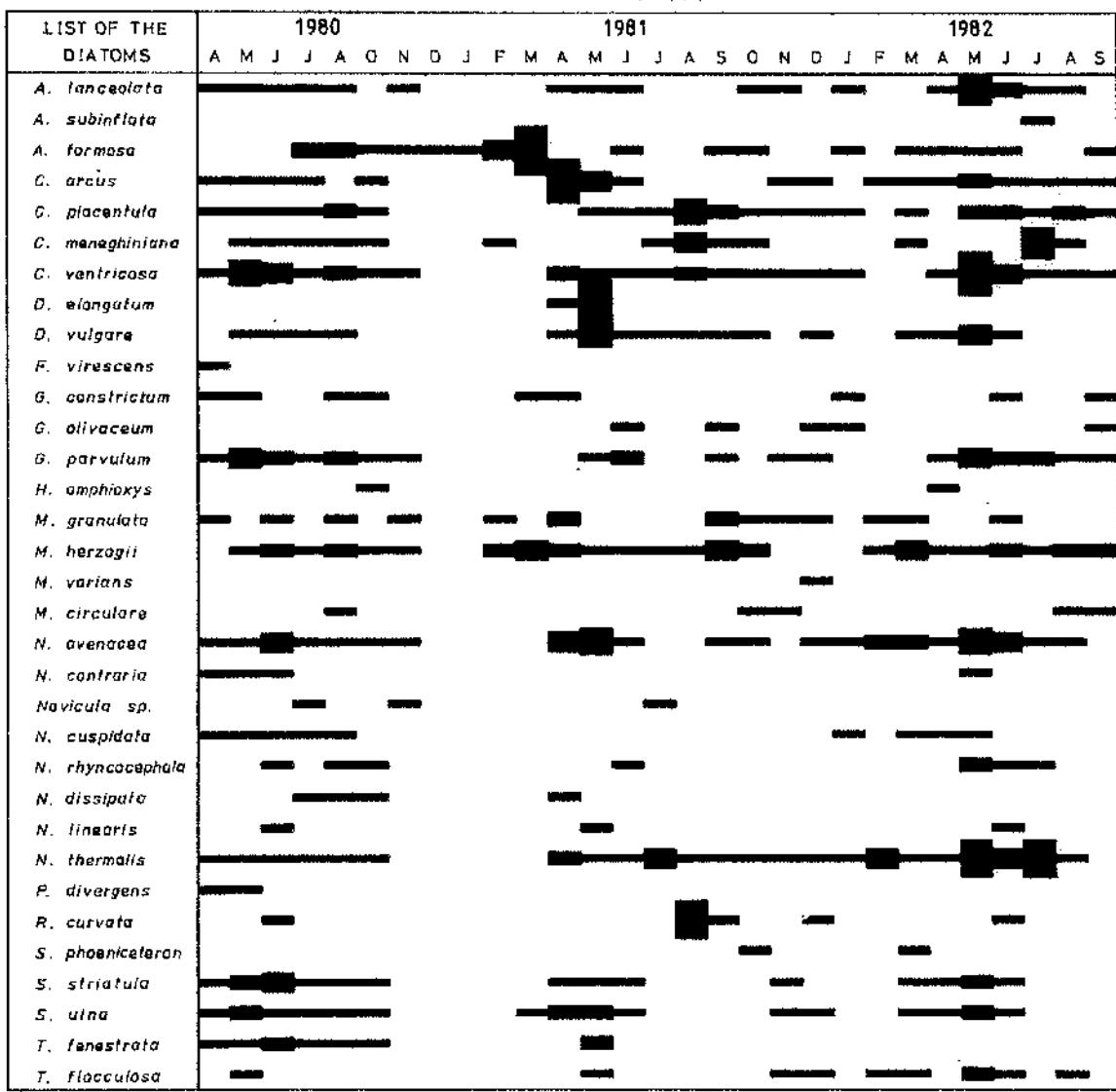
contd.

STATION (6)

LIST OF THE DIATOMS	1980					1981					1982				
	A	M	J	A	O	N	D	J	F	M	A	M	J	A	S
<i>A. lanceolata</i>															
<i>A. formosa</i>															
<i>C. arcus</i>															
<i>C. placentula</i>															
<i>C. meneghiniana</i>															
<i>C. ventricosa</i>															
<i>D. elongatum</i>															
<i>D. vulgare</i>															
<i>F. virescens</i>															
<i>G. constrictum</i>															
<i>G. olivaceum</i>															
<i>G. parvulum</i>															
<i>H. amphioxys</i>															
<i>M. granulata</i>															
<i>M. herzogii</i>															
<i>M. varians</i>															
<i>M. circulare</i>															
<i>N. avenacea</i>															
<i>N. contraria</i>															
<i>Navicula sp.</i>															
<i>N. cuspidata</i>															
<i>N. rhyncocephala</i>															
<i>N. aciculata</i>															
<i>N. dissipata</i>															
<i>N. linearis</i>															
<i>N. thermalis</i>															
<i>N. sigma</i>															
<i>R. curvata</i>															
<i>S. phoenicenteron</i>															
<i>S. striatula</i>															
<i>S. ulna</i>															
<i>T. fenestrata</i>															

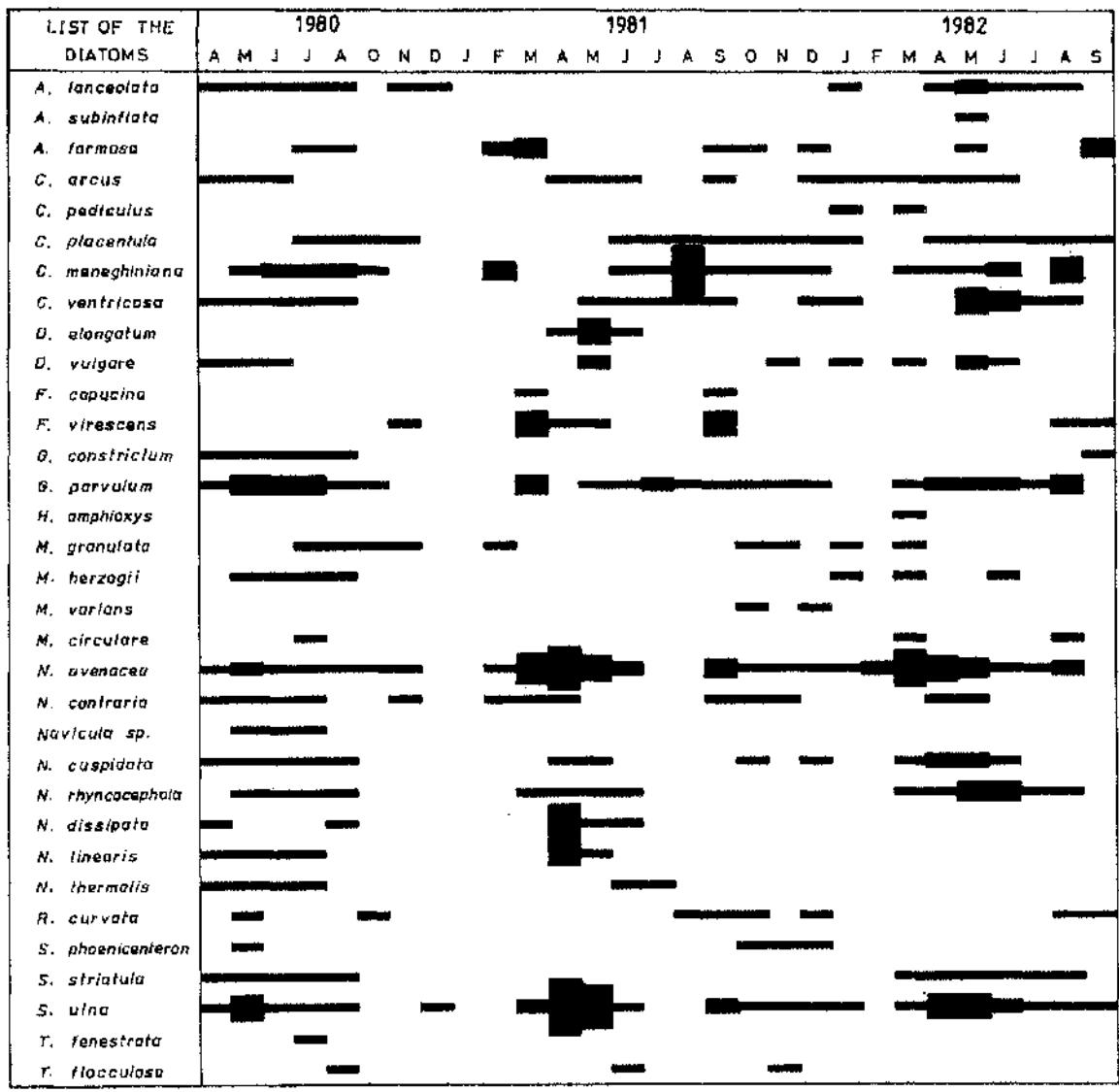
contd.

STATION (7)



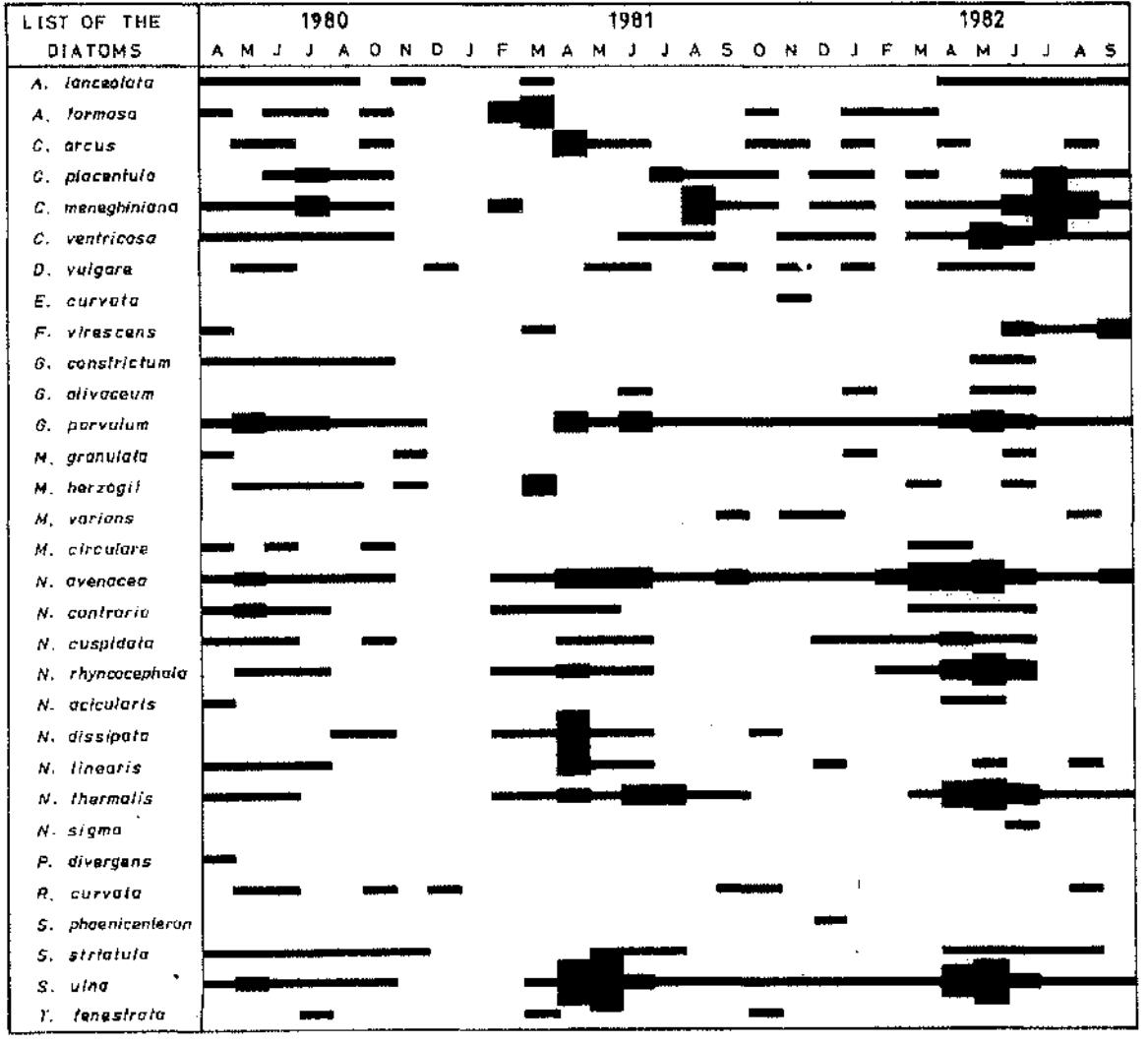
contd.

STATION (B)



contd.

STATION (9)



contd.

STATION (10)

only one having *Synedra ulna* in amounts >500 cells L^{-1} during June 1980. The two species were also observed at Station 3 during the same time (*Gomphonema parvulum* 720 cells L^{-1} and *Synedra ulna* 1,040 cells L^{-1}). The latter was replaced by *Cymbella ventricosa* (1,166 cells L^{-1}) during June; Glazert Water joined the main river above this station.

Luggie Water (Station 4) differed from the other stations (excluding Station 10) by showing the spring outburst for *Synedra* (1,010 cells L^{-1}) earlier during April 1980 and continuing into May (3,833 cells L^{-1}). *Gomphonema parvulum* was also recorded during May and June (650 cells L^{-1}). *Cymbella ventricosa* (1,040 cells L^{-1}) were recorded accompanying them during May while it was replaced by *Cyclotella meneghiniana* during June when maximum numbers of 26,400 cells L^{-1} were obtained. The high peak at this station would be referable to this species.

At Station 5 *Synedra ulna* (2,460 cells L^{-1}) and *Gomphonema parvulum* (935 cells L^{-1}) were recorded during May. The latter were recorded during June also in amounts of 1,267 cells L^{-1} whilst the former was replaced by *Cyclotella meneghiniana* (8,000 cells L^{-1}). *Cymbella ventricosa* and *Navicula avenacea* were also observed at this station as it was a continuous flow from Station 3.

Station 6 showed similarity with the previous station with *Synedra ulna* (4,270 cells L^{-1}) again the dominant during May, accompanied by *Gomphonema parvulum* (1,900 cells L^{-1}). During June the latter stayed almost the same while the former had been replaced by *Cyclotella meneghiniana* (4,270 cells L^{-1}). In addition to the

other two species which were recorded at the previous station

Nitzschia thermalis was also observed at this station. All were in amounts between 500-1,000 cells L^{-1} .

Allander Water, Station 7, also principally supported *Synedra ulna* (500 cells L^{-1}) and *Gomphonema parvulum* (1,500 cells L^{-1}) during May but the river was dominated by *Cymbella ventricosa* (2,370 cells L^{-1}) at this station. During June these showed reductions in numbers being replaced by *Navicula avenacea* (1,425 cells L^{-1}) and *Surirella striatula* (1,335 cells L^{-1}). *Melosira herzogii* was found in small quantities.

At Station 8, *Synedra ulna* (2,533 cells L^{-1}) was dominant accompanied by *Gomphonema parvulum* (1,470 cells L^{-1}). The latter was observed in similar quantities during June and showed dominancy at this station and also at Station 9 during May (1,360 cells L^{-1}) accompanied by *Synedra ulna*; *Navicula avenacea* and *Navicula contraria* were also recorded. During June the only dominant species found was *Gomphonema parvulum* (840 cells L^{-1}) at this station.

As with Station 4, *Synedra ulna* (685 cells L^{-1}) showed dominancy during April at Station 10 in being present with *Navicula avenacea* (700 cells L^{-1}). They increased in number to 2,700 cells L^{-1} respectively during May accompanied by *Navicula cuspidata* (800 cells L^{-1}). *Gomphonema parvulum* (710 cells L^{-1}) was found with *Cyclotella meneghiniana* (625 cells L^{-1}) replacing the rest during June.

The spring outburst in 1981 started during April at all the stations. During this period more variety of species were observed compared with the previous year. Principally *Synedra ulna* again dominated the river at almost all the stations but *Gomphonema parvulum*

was replaced by *Navicula avenacea* at all the stations excluding Station 1. These two were accompanied by other species of diatoms and the main river was mostly dominated by *Nitzschia thermalis* during June.

During April *Gomphonema parvulum* ($4,980$ cells L^{-1}) was the dominant diatom at Station 1 with *Synedra ulna* ($2,490$ cells L^{-1}). *Achnanthes lanceolata*, *Gomphonema olivaceum*, *Navicula avenacea* and *Nitzschia dissipata* were also found. All these showed a fall in numbers during May but they increased slightly again during June when *Nitzschia thermalis*, *Navicula avenacea* and *Gomphonema parvulum* were obtained. At Station 2 *Synedra ulna* ($3,444$ cells L^{-1}) dominated the river during April accompanied by *Navicula avenacea* ($2,000$ cells L^{-1}). The former increased in quantity to $5,083$ cells L^{-1} during May and declined with the latter during June when *Nitzschia thermalis* ($1,136$ cells L^{-1}) was the dominant; *Gomphonema parvulum* was also found at this time. *Synedra* dominated the river at Station 3 during April accompanied by *Navicula avenacea* ($2,445$ cells L^{-1}). Meanwhile *Ceratoneis arcus* was obtained in high amounts of $4,100$ cells L^{-1} at this station. This species and *Hantzschia amphioxys* were characteristic features of this station where *Nitzschia dissipata* ($1,086$ cells L^{-1}), *Nitzschia thermalis*, *Diatoma vulgare* and *Gomphonema parvulum* were also found. During May the diatoms showed a loss in numbers with *Synedra ulna* still the dominant ($2,877$ cells L^{-1}) but during June it was replaced by *Nitzschia thermalis* ($2,660$ cells L^{-1}). At Station 4 *Navicula avenacea* ($13,575$ cells L^{-1}) dominated the river during April with *Synedra ulna* ($2,510$ cells L^{-1}); *Nitzschia dissipata*

(1,045 cells l^{-1}) and *Diatoma elongatum* were also recorded. During May all these diatoms showed a fall in quantity but *Navicula avenacea* was still the dominant (1,700 cells l^{-1}) accompanied by *Rhoicosphenia curvata* (1,300 cells l^{-1}). This was a new species recorded at this station. All the diatoms declined and were replaced by *Nitzschia thermalis* (1,120 cells l^{-1}) during June, and also at the same time dominated the population at Station 5 (5,830 cells l^{-1}). At this station *Navicula avenacea* (5,000 cells l^{-1}) showed dominancy during April accompanied by *Synedra ulna* (1,670 cells l^{-1}) and *Nitzschia thermalis* (1,250 cells l^{-1}). The first two continued their growth until May when *Synedra ulna* (1,840 cells l^{-1}) dominated the river at this station being present with *Navicula avenacea*. At Station 6 *Synedra ulna* (4,935 cells l^{-1}) was dominant during April accompanied by *Navicula avenacea* (1,733 cells l^{-1}) and three species of *Nitzschia*. These were *Nitzschia dissipata* (3,600 cells l^{-1}), *Nitzschia thermalis* (2,535 cells l^{-1}) and *Nitzschia linearis* (2,000 cells l^{-1}). The first one and the last were observed in the main river in high quantities starting from this station i.e. below the junction with the Bishopbriggs Burn. *Ceratoneis arcus* (1,735 cells l^{-1}) was also found as it entered the main river at Station 3, but all the populations showed a rapid fall in quantity during May with *Synedra ulna* (2,750 cells l^{-1}) still the dominant. During June *Nitzschia thermalis* (2,611 cells l^{-1}) was the dominant at this station; *Gomphonema parvulum*, *Navicula avenacea* and *Synedra ulna* were also observed. As with Station 3 *Ceratoneis arcus* (5,135 cells l^{-1}) was found in high quantities at Station 7 and was the dominant during April. It was accompanied by *Melosira Herzogii*

(1,500 cells L^{-1}) and *Navicula avenacea* (1,100 cells L^{-1}). These differed during May when *Diatoma elongatum* (7,940 cells L^{-1}) was the dominant with *Diatoma vulgare* (2,760 cells L^{-1}), *Navicula avenacea* (2,520 cells L^{-1}) and *Ceratoneis arcus* which was reduced to 1,080 cells L^{-1} . All these declined during June when *Gomphonema parvulum* (875 cells L^{-1}) dominated at this station. *Synedra ulna* gave the first sign of the spring outburst at Station 8 during April (11,080 cells L^{-1}) accompanied by *Navicula avenacea* (5,540 cells L^{-1}), *Nitzschia dissipata* (4,155 cells L^{-1}) and *Nitzschia linearis* (2308 cells L^{-1}). *Synedra* stayed during May also (6,820 cells L^{-1}) accompanied by *Navicula avenacea* (2,220 cells L^{-1}) and *Diatoma elongatum* (2,378 cells L^{-1}); similar to Station 7). During June a loss in quantities for all the diatoms was observed, being replaced by *Nitzschia thermalis* (2,270 cells L^{-1}). At Station 9 the river was dominated again by *Synedra ulna* (6,355 cells L^{-1}) during April accompanied by *Nitzschia dissipata* (5,300 cells L^{-1}), *Ceratoneis arcus* (2,120 cells L^{-1}), *Gomphonema parvulum*, *Navicula avenacea*, *Nitzschia thermalis* (the last three all about 1,100 cells L^{-1}). *Synedra ulna* (14,370 cells L^{-1}) showed dominancy during May also being present with *Navicula avenacea* (1,440 cells L^{-1}). During June few species together shared dominancy and replaced *Synedra* which declined rapidly. These were *Gomphonema parvulum* (1,100 cells L^{-1}), *Navicula avenacea* and *Nitzschia thermalis* (both at about 1,620 cells L^{-1}). At Station 10 the two diatoms were observed indicating the spring outburst during April (*Synedra ulna* 21,090 cells L^{-1} and *Navicula avenacea* 2,125 cells L^{-1}). These were recorded during May also (2,710 and 2070 cells L^{-1}) respectively; *Fragilaria virescens*

(2,585 cells L^{-1}) was also found. These showed a fall in numbers during June *Navicula avenacea* (785 cells L^{-1}) the dominant.

During the main annual growth period of 1982 *Synedra ulna* was again observed along the stretch of the river but in smaller numbers and the dominancy was taken by other species of diatoms at the different stations. These were mainly *Gomphonema parvulum*, *Navicula avenacea* and *Nitzschia thermalis*; *Cyclotella meneghiniana* and *Cymbella ventricosa* were also recorded in high quantities occasionally.

Station 1 was dominated only once by *Nitzschia thermalis* during the whole growth period of 1982 during May. More species were obtained on passing downstream to Station 2 where *Navicula avenacea* (9,067 cells L^{-1}) was the dominant during April. It accompanied *Nitzschia thermalis* (7,980 cells L^{-1}), *Diatoma vulgare* and *Navicula rhyncocephala* (both at about 4,110 cells L^{-1}). *Suzirella striatula* (1088 cells L^{-1}) was also found. *Synedra ulna* (2,027 cells L^{-1}) was recorded during May when *Fragilaria virescens* (4,250 cells L^{-1}) was the dominant at this station accompanied by *Navicula avenacea* (1,308 cells L^{-1}) and *Navicula rhyncocephala* (1,675 cells L^{-1}). The diatoms showed a loss in numbers during June when *Gomphonema parvulum* and *Navicula avenacea* were observed. The latter was the dominant at Station 3 during April (4,170 cells L^{-1}) accompanied by *Ceratoneis arcus* (1,961 cells L^{-1}), *Navicula rhyncocephala* (1,438 cells L^{-1}), *Nitzschia thermalis* (1,050 cells L^{-1}) and *Synedra ulna* (1,896 cells L^{-1}). The same species were found in the river at this station during May but *Synedra ulna* was the dominant. During June the diatoms declined with a few species obtained in amounts

500 cells L^{-1} . These were *Gomphonema parvulum*, *Navicula avenacea* and *Navicula rhyncocephala*. During April *Gomphonema parvulum* (10,220 cells L^{-1}) had formed the main components of the population at Station 4. It was accompanied by *Navicula avenacea* (4,580 cells L^{-1}), *Navicula rhyncocephala* (3,760 cells L^{-1}), *Nitzschia thermalis* (2,860 cells L^{-1}). *Diatoma elongatum* and *Synedra ulna* were also observed. Similar to the previous station, the same species were recorded during May and June but showing a reduction in quantities during the latter months. This population decline was replaced by *Cyclotella meneghiniana* (4,450 cells L^{-1}). Stations 5 and 6 were similar to Station 4 in supporting the same diatoms during the growth period. *Gomphonema parvulum* (4,470 cells L^{-1}) was the dominant at Station 5 during April while it was recorded in quantities of 1,860 cells L^{-1} at Station 6 where *Navicula avenacea* (3,680 cells L^{-1}) and *Synedra ulna* (3,465 cells L^{-1}) showed dominancy. The last two were also found at Station 5 (3,695 and 2,385 cells L^{-1}) respectively with *Nitzschia thermalis* (2,745 cells L^{-1}) at Station 5; 2,092 cells L^{-1} at Station 6), *Navicula rhyncocephala* and *Navicula cuspidata* (both at about 1,500 cells L^{-1} at the two stations) were also obtained. During May the same species were again observed (*Synedra ulna* 3,460 cells L^{-1} , the dominant at Station 5 with *Nitzschia thermalis* and *Navicula avenacea*, both at about 3,650 cells L^{-1} , found dominating at Station 6). In addition, *Cymbella ventricosa* was recorded (1,708 cells L^{-1} at Station 5; 2,070 cells L^{-1} at Station 6). The same species were also observed during June but showing a reduction in numbers. The spring outburst population was

accompanied by *Cymbella ventricosa* ($6,215$ cells L^{-1}) at Station 7 during May. It was accompanied by few species, *Nitzschia thermalis* ($4,686$ cells L^{-1}), *Achnanthes lanceolata* ($3,575$ cells L^{-1}), *Navicula avenacea* ($2,450$ cells L^{-1}), *Gomphonema parvulum* and *Diatoma vulgare*. The loss in quantities during June also observed at this station as shown in Figure 4.12b. Station 8 was dominated by *Synedra ulna* and *Navicula avenacea* (both at about $2,500$ cells L^{-1}). They were observed during May also when the population was dominated by *Nitzschia thermalis* ($3,546$ cells L^{-1}) accompanied by other species (*Cymbella ventricosa* $2,600$ cells L^{-1} and *Navicula rhyncocephala* $1,077$ cells L^{-1}). *Nitzschia thermalis* stayed dominant during June also ($2,225$ cells L^{-1}) accompanied again by the former two. *Synedra ulna* dominated the river at Station 9 during April ($3,600$ cells L^{-1}) being with *Navicula avenacea* and *Nitzschia thermalis* (both at about $2,060$ cells L^{-1}); *Navicula rhyncocephala* was also found. During May *Synedra ulna* was still the dominant species ($5,760$ cells L^{-1}) being present with the species recorded during April (all at about $3,670$ cells L^{-1}) in addition to *Cymbella ventricosa* and *Gomphonema parvulum*. During June *Cyclotella Meneghiniana* was found with the rest of the diatoms obtained during the previous month. Finally at the last station, Station 10, *Navicula avenacea* ($7,195$ cells L^{-1}) was the dominant during April accompanied by *Synedra ulna* and *Navicula cuspidata* (both at $2,685$ cells L^{-1}). *Navicula rhyncocephala* and *Diatoma vulgare* were also found. The same species were found during May *Navicula avenacea* and *Navicula rhyncocephala* (both at $3,750$ cells L^{-1}) with *Gomphonema parvulum*, *Nitzschia thermalis* (both at about

2,165 cells L^{-1}) and *Synedra ulna* (2,095 cells L^{-1}). During June *Navicula rhyncocephala* was the dominant and the rest observed in smaller numbers; *Cyclotella meneghiniana* (2,095 cells L^{-1}) was also recorded.

These data have described the components of the main spring pulses which occurred at all the stations during the years 1980, 1981 and 1982. There were other occasional pulses recorded either during summer or late winter and early spring. The planktonic diatom *Asterionella formosa* formed the main components during the latter. It appeared during March and it was not observed again until September. This feature was recorded at Stations 2, 3 and 8 when it was found during 1981 and September 1982 (3,760 cells L^{-1} at 2, 2,250 cells L^{-1} at 3 and 1,750 cells L^{-1} at 8). It was found at Stations 5, 6 and 7 (3,650, 1,740 and 8,950 cells L^{-1} respectively) during March 1981. It was accompanied by *Melosira herzogii* (1,695 cells L^{-1} at Station 7) which was recorded during March 1982 also. *Asterionella formosa* was obtained at Station 9 during February and March 1981 (1,200 and 3,170 cells L^{-1} respectively) during the same period *Melosira herzogii* was recorded dominating the river at Station 10 (2,400 cells L^{-1} and 1,450 cells L^{-1} respectively) and it was found during March 1982 also (1,050 cells L^{-1}). *Navicula avenacea* showed some growth during March 1981 and 1982 at most of the stations. It was found during March 1982 at Stations 4, 5, 6, 9 and 10 showing the highest numbers at Station 4 (12,750, 5,430, 2,975, 2,060 and 4,520 cells L^{-1} respectively). It was found at Station 4 during February 1982 also (3,010 cells L^{-1}). At Station 8 it was recorded during March 1981 (3,575 cells L^{-1}) and 1982 (4,710 cells L^{-1}).

The summer peaks were mostly observed during July 1982 and partly during August at all the stations, excluding Station 1. These were mainly due to the high quantities of the planktonic diatom *Cyclotella meneghiniana*. During July it was found in large amounts at Station 2 ($22,447 \text{ cells } \ell^{-1}$) accompanied by *Nitzschia thermalis* ($3,378 \text{ cells } \ell^{-1}$) *Gomphonema parvulum* ($1,090 \text{ cells } \ell^{-1}$) and *Synedra ulna* ($1,852 \text{ cells } \ell^{-1}$). At Station 3 it was obtained in amounts of $10,570 \text{ cells } \ell^{-1}$ again accompanied by the diatoms mentioned at the previous stations. At the rest of the stations it was recorded as follows: $13,145 \text{ cells } \ell^{-1}$ at 4, $32,365 \text{ cells } \ell^{-1}$ at 5, $18,830 \text{ cells } \ell^{-1}$ at 6, $3,860 \text{ cells } \ell^{-1}$ at 7 accompanied by *Nitzschia thermalis* ($4,320 \text{ cells } \ell^{-1}$), $11,770 \text{ cells } \ell^{-1}$ at 8, $15,300 \text{ cells } \ell^{-1}$ at 9 and $2,690 \text{ cells } \ell^{-1}$ at 10. It was found during August also but in lesser quantities.

For the whole period of this survey the species which were found in smaller amounts are not mentioned. Their occurrences can be clearly seen in Figure 4.13.

4.2.2 Chlorophyll a and the phaeopigments

Measurements of chlorophyll a were used as an indirect method for algal biomass estimations as it reflects the sum of the living suspended photosynthetic organisms. The phaeopigments were measured as a correction for the former as representing the dead and decomposing plant cells.

Figure (4.14) and Table (4.10) displays the amount of chlorophyll a and Figure (4.15), Table (4.11) shows the phaeophytin quantities in the river during the period February 1980-September 1982.

Table (4.10) Chlorophyll a measurements (mg m^{-3}) at the 10 stations of the River Kelvin during the period February 1980 - September 1982.

	1	2	3	4	5	6	7	8	9	10
<u>1980</u>										
Feb.	0.32	1.28	0.96	1.12	2.08	0.96	0.48	0.16	0.48	0.32
March	0.00	0.96	0.64	1.12	1.44	1.28	6.41	0.80	4.65	1.28
April	1.50	1.44	2.50	2.90	2.40	3.20	2.00	2.70	3.80	7.20
May	1.60	10.3	4.33	3.52	4.65	4.33	3.52	3.20	2.24	6.41
June	2.59	3.20	3.52	9.29	8.01	10.7	6.89	5.61	3.64	3.68
July	0.32	2.80	4.00	5.45	3.20	1.28	1.92	2.32	4.80	2.16
Aug.	0.64	0.32	6.30	4.80	8.01	2.08	2.24	1.60	0.64	0.96
Oct.	0.00	1.04	0.16	6.08	5.13	17.3	4.48	1.28	0.50	1.92
Nov.	0.08	0.32	0.32	0.32	0.03	0.08	0.16	0.48	0.16	0.32
Dec.	0.16	0.08	0.32	0.16	0.16	0.16	0.08	0.16	0.16	0.16
<u>1981</u>										
Jan.	0.64	0.48	0.16	0.03	0.16	1.12	0.64	7.40	0.80	0.72
Feb.	0.48	0.32	0.80	0.64	0.64	0.48	0.16	0.16	0.80	0.32
March	0.35	0.60	0.65	1.63	1.85	2.40	0.50	0.96	2.60	3.10
April	0.16	0.80	1.60	2.10	2.10	3.50	0.16	2.40	3.80	5.10
May	2.00	16.0	4.76	2.80	3.90	3.50	2.90	2.90	1.95	4.80
June	1.12	0.96	1.60	1.60	1.20	0.48	0.30	0.48	0.64	0.49
July	0.32	0.32	0.48	0.96	0.48	3.20	0.96	0.48	0.48	0.96
Aug.	1.10	17.9	8.17	0.48	6.41	6.09	1.12	2.08	5.77	0.16
Sept.	2.40	0.16	0.20	0.88	2.24	1.50	2.64	0.32	6.25	0.50
Oct.	0.50	0.64	0.00	1.12	0.48	1.12	0.32	0.50	1.50	1.76
Nov.	0.64	0.10	0.32	0.48	0.30	0.80	1.40	1.28	0.32	1.28
Dec.	1.44	0.34	1.12	0.80	0.00	0.16	1.12	0.00	2.24	0.80
<u>1982</u>										
Jan.	2.08	1.44	0.90	1.28	0.00	0.96	1.44	0.80	0.64	0.00
Feb.	0.80	0.20	0.96	0.16	2.56	0.01	1.44	2.24	0.16	0.00
March	0.64	0.32	0.96	1.76	0.96	2.72	0.00	0.16	1.60	2.24
April	2.56	2.72	6.09	1.44	6.41	5.61	2.56	6.57	9.96	8.81
May	3.52	2.73	4.33	3.20	3.84	3.04	3.04	3.20	1.12	3.20
June	0.32	7.05	3.36	2.08	4.17	1.76	2.08	2.56	1.12	0.32
July	5.61	9.61	4.17	3.68	3.36	18.1	3.04	15.0	4.65	16.2
Aug.	0.80	4.17	0.32	0.80	2.72	1.12	1.60	2.56	0.48	3.84
Sept.	0.00	0.48	0.00	0.00	0.16	1.28	0.16	0.80	0.48	0.00

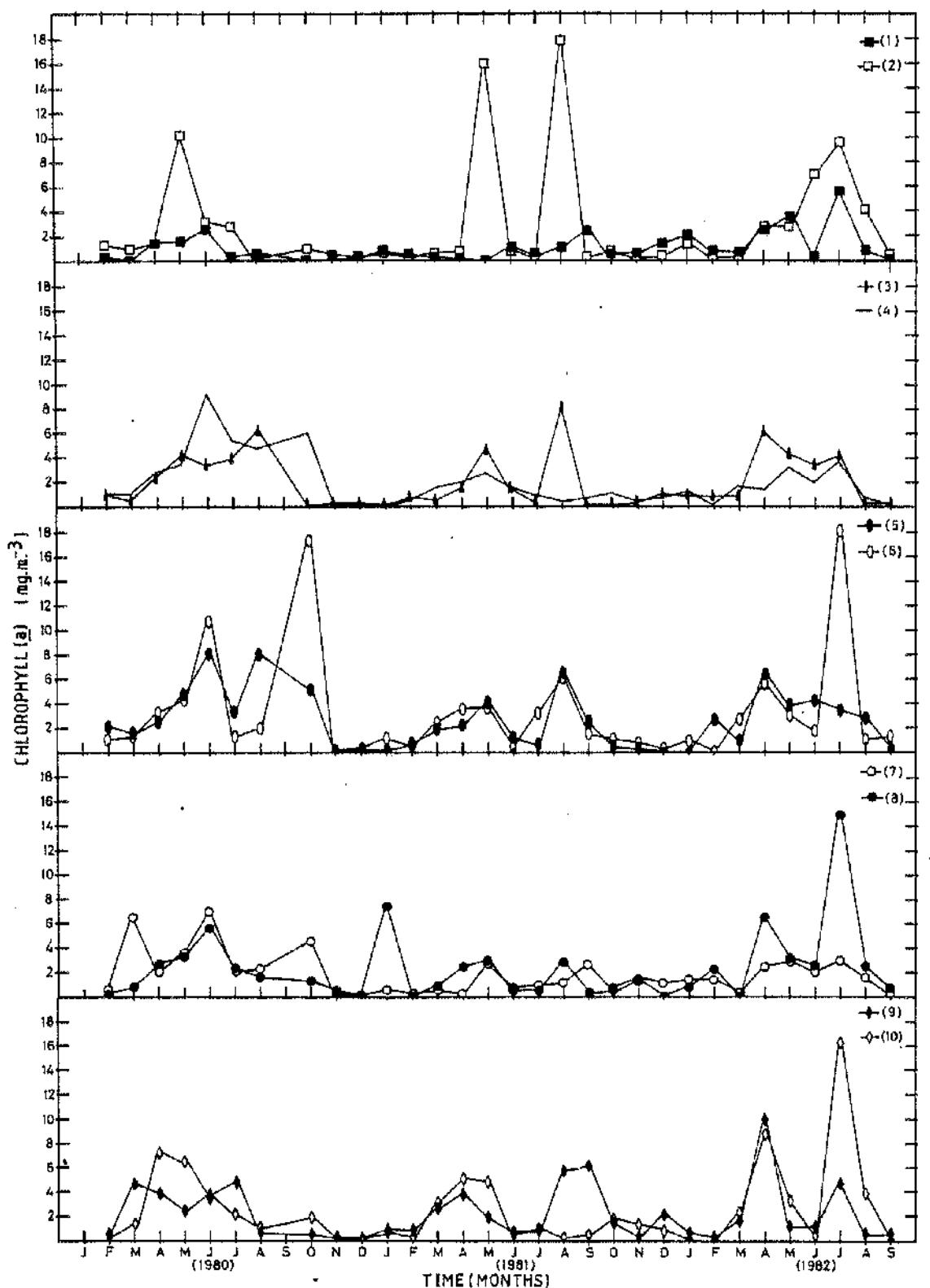


Figure 4.14: Chlorophyll a content in mg m^{-3} in the River Kelvin at 10 different stations during the period February 1980 - September 1982.

Table (4.11) Phaeopigment measurements (mg m^{-3}) in the River Kelvin at the 10 stations during the period February 1980 - September 1982.

Time	S T A T I O N S									
	1	2	3	4	5	6	7	8	9	10
1980										
Feb.	1.75	3.86	3.76	1.23	0.16	1.39	2.72	2.50	1.76	1.03
March	1.89	2.07	2.27	2.24	1.25	1.19	1.49	1.44	2.00	2.42
April	3.45	7.19	9.80	11.4	7.57	10.9	1.80	5.10	4.30	4.30
May	5.00	12.1	5.77	4.02	12.2	7.56	4.89	6.66	5.16	3.68
June	1.09	2.29	3.88	1.36	9.61	3.40	5.45	2.69	1.01	2.26
July	2.00	4.00	8.90	14.7	8.00	5.00	4.30	3.40	4.30	3.22
Aug.	1.99	3.60	5.90	4.23	1.67	1.69	2.46	1.42	3.22	1.39
Oct.	3.18	2.21	4.42	7.75	1.70	3.61	1.00	2.49	2.24	2.17
Nov.	0.34	1.24	0.13	1.13	1.48	0.82	1.40	0.52	0.62	0.16
Dec.	0.62	1.90	2.03	5.40	4.10	4.88	2.27	4.00	3.98	4.20
1981										
Jan.	0.03	1.09	0.74	1.74	2.98	5.38	2.05	1.15	0.99	0.96
Feb.	0.75	1.80	7.68	1.60	1.15	0.97	1.50	1.89	1.10	1.90
March	0.95	1.07	2.30	1.10	1.80	1.20	1.30	1.44	0.95	2.20
April	0.62	0.00	0.00	0.00	0.00	1.50	1.75	0.50	2.50	3.50
May	0.34	6.80	9.10	8.30	6.10	7.00	10.3	1.90	8.75	7.10
June	3.30	1.80	1.87	3.90	2.80	3.10	3.00	3.30	2.60	0.66
July	2.60	2.90	2.66	1.17	3.30	2.60	2.60	1.90	2.30	1.70
Aug.	4.09	8.65	1.46	10.5	12.0	16.6	7.06	9.58	7.35	9.60
Sept.	1.17	3.20	3.33	1.36	0.45	1.28	0.59	3.94	3.00	7.48
Oct.	0.88	2.50	1.80	0.80	0.64	1.57	1.36	2.44	3.33	1.04
Nov.	1.27	2.50	3.72	4.90	4.04	3.57	0.58	0.96	3.72	2.31
Dec.	1.03	1.65	0.90	5.14	4.82	2.42	0.56	1.62	1.91	1.30
1982										
Jan.	4.68	0.24	0.43	2.50	2.68	0.61	0.13	0.10	0.70	1.51
Feb.	1.50	2.39	0.72	2.39	2.48	0.00	7.64	1.35	2.69	1.94
March	0.37	2.15	1.84	2.27	2.77	0.19	2.85	2.31	0.98	0.90
April	4.61	6.25	4.45	7.98	3.35	4.71	3.04	6.89	6.54	7.90
May	1.30	5.13	5.77	3.75	4.90	7.27	3.91	6.66	6.84	6.22
June	2.03	5.17	3.81	2.85	1.11	3.51	1.17	1.47	3.48	7.50
July	5.38	5.53	2.56	6.97	6.06	7.27	3.24	7.64	7.47	0.24
Aug.	1.44	0.24	3.83	4.02	0.03	2.80	0.98	1.59	5.08	11.6
Sept.	3.84	3.33	6.36	7.85	4.33	3.09	2.53	3.01	3.56	10.6

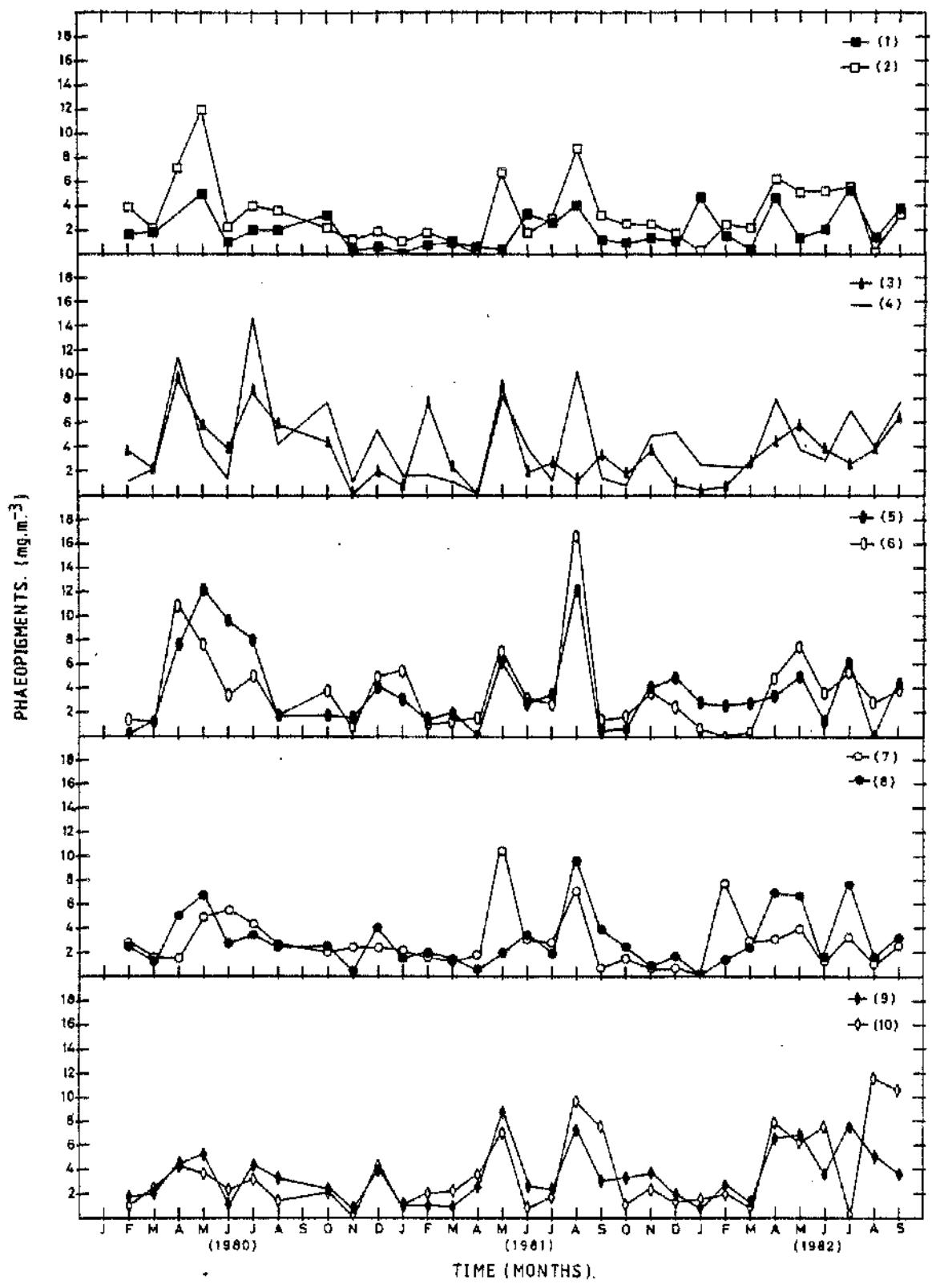


Figure 4.15: The phaeopigment contents in mg m^{-3} in the River Kelvin at the 10 stations during the period February 1980 - September 1982.

The chlorophyll a quantities in fact out ~~surpassed~~ ^{of phase with} the numbers of diatoms, and these results indicate that these high values are probably due to the suspended plant fragments and debris from various sources, including the diatoms, but the diatom quantities are masked. Even at Station 4, Luggie Water, with its maximum diatom peaks, the chlorophyll values were relatively low compared with the other stations.

This idea is borne out also by the phaeophytin data for which on occasions higher values than the chlorophyll a was recorded. This is probably due either to technical errors in extraction or that the bulk of the chlorophyll a data is really phaeophytin, e.g. breakdown products, and the living diatom quantities are still hidden.

4.2.3 Carbon fixation by the phytoplankton

This technique was applied in the survey to estimate the carbon fixing potential of the suspended diatoms. It was carried out on the water samples from 5 stations: Station 1 (the river's head receiving no pollution); Stations 4 and 6, being the stations having the highest nutrient loads and high diatom populations; Station 9 for its location in the university lands and at the entrance of the City of Glasgow, also because it showed some similarities to other stations, e.g. 2 and 8; Station 10, being the last and the station with the highest dissolved oxygen content.

Table (4.12) shows the carbon fixed by the phytoplankton in the river samples during the period April-September 1982. The fixation recorded varied significantly at the stations and also seasonally. The values observed ranged between $0 - 67.5 \text{ mg C m}^{-3} \text{ h}^{-1}$ recorded for

Table (4.13) ^{14}C ($\text{mg C m}^{-3} \text{ 3h}^{-1}$) fixed by the phytoplankton of the River Kelvin during the period April-September 1982.

Stations	$^{14}\text{C mg. C. m}^{-3}. \text{ 3h}^{-1}$					
	April	May	June	July	August	September
1	1.54	2.54	0.0	4.75	3.79	0.88
	0.05	2.20	0.0	3.42	1.53	0.60
4	4.36	1.52	2.15	0.31	1.01	2.24
	3.61	1.39	1.85	0.19	0.72	2.86
6	13.3	9.0	4.49	69.4	0.88	1.80
	12.5	9.45	4.25	65.5	0.86	1.78
9	24.2	2.76	2.81	2.55	0.70	1.82
	28.4	2.82	2.89	2.24	1.03	1.77
10	18.0	3.79	2.47	1.34	0.74	2.07
	16.8	3.86	1.39	0.90	0.75	2.09

Two way ANOVA for the above table

Source	D.F.	S.S.	M.S.	F.factor
Replicate	1	1.45	1.45	1.89
A Time	5	1804.87	360.97	473.06 **
B Stations	4	1669.75	417.44	547.05 **
A*B	20	6160.93	5.0	403.70 **

** Significant differences at 1% level.

the Stations 1 and 6 respectively. Maximum fixation was found at the latter station at most times of applying these measurements.

The carbon fixation data did not show good correlation with the diatom cell numbers observed during the same period but tended to follow more closely the chlorophyll a values. Although high pulses in diatom cell numbers occurred mostly at Station 4 (Luggie Water), the fixations obtained were relatively low. Higher carbon fixation was measured at almost all the stations during April compared with the other months, being the highest at Station 9, coinciding with a noticeable pulse in chlorophyll a levels at the same station, whilst the diatom cell numbers were higher at Station 4. During May, June and July the maximum fixations were recorded at Station 6 for all the three months whilst the highest cell number for the diatoms was found in decreasing order at Stations 9, 10 and 6 respectively. This also correlated with the chlorophyll a amounts especially during July when the highest fixation for the whole period was recorded at Station 6. At the same time the highest fixation for Station 1 also coincided with its highest chlorophyll pulse. For the latter station the diatom cell numbers could not be estimated due to the very dense organic matter, whilst for the former the highest cell numbers were observed. During August very low fixations and low chlorophyll a amounts were found at all the stations with the maximum fixation at Station 1, whilst the highest diatom cell numbers were recorded at Station 6. Finally during September the carbon fixations values showed a slight improvement compared with the previous month although a fall in the diatom numbers and also the chlorophyll a

were found at all the stations. The maximum fixation was at Station 4 even though the quantities of diatoms at Station 9 were higher. The ^{14}C data fits the chlorophyll a however which may thus in part reflect the diatom activity.

4.2.4 Filamentous algae

Nine species of filamentous algae were recorded at the different stations, members of green and blue-green algae. The following is a list of the species observed:

Cladophora fracta (Dillw.) Kütz (Plate 4.04 Fig. 5)

Cladophora glomerata (L.) Kütz (Plate 4.04 Fig. 3)

Lyngbya aerugineo-caerulea (Kütz) Gomont

Microspora sp.

Oscillatoria sp.

Oedogonium sp. (Plate 4.04 Fig. 1)

Spirogyra sp.

Ulothrix sp. (Plate 4.04 Fig. 4)

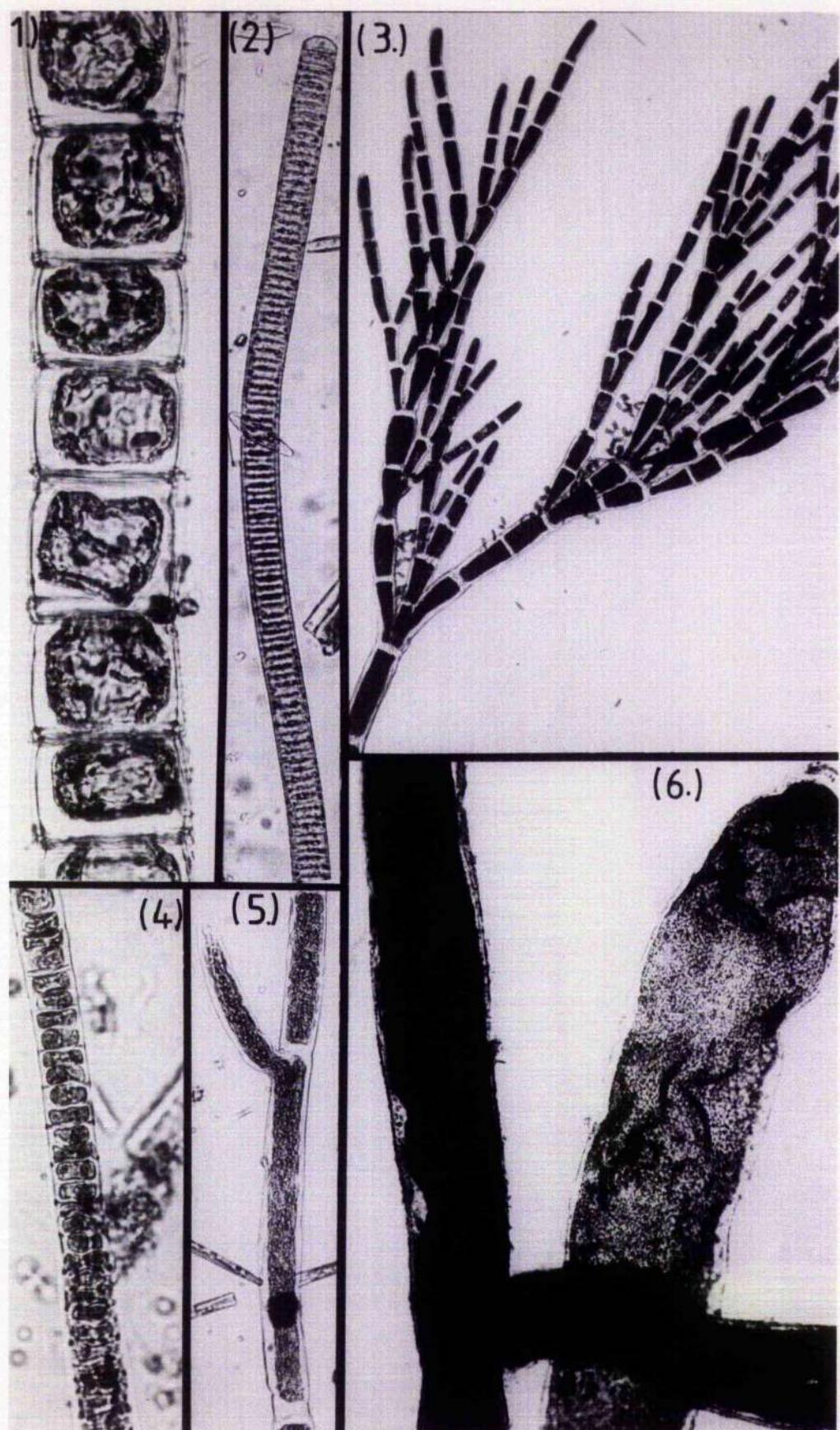
Vaucheria sp. (Plate 4.04 Fig. 6)

Oedogonium and *Vaucheria* could not be identified at species level because of the lack of the reproductive stages. Some of these algae are shown in Plate 4.04.

Their growth period was limited to the summer. They started to appear in mid-April and they were abundant from May until August. Their decline started during September and they had disappeared completely by the autumn-winter period.

They were found in very high quantities at some stations and attached to any substance in the river. This attachment was loose

Plate (4.04) The filamentous algae recorded in the River Kelvin
during the period February 1980-September 1982
(see Sec. 4.2.4).



Plate(4.04)

and was affected by current flows. For example, the relatively high flow rate recorded during June 1981 (Fig. 4.01) washed away all the filamentous algae which dominated the river at that time, but by July and August they again dominated the river.

Unlike the diatoms, no seasonal variations were observed with the different species. Some were found over the whole period at every station.

The stations varied in their contents of these filamentous algae. None were found at Station 1 during the whole period of this study. At Station 2 all the 9 species were recorded, with *Cladophora fracta*, *Cladophora glomerata* and *Oedogonium* present in large quantities while the rest were in small amounts scattered among the larger filaments. At Station 3 very small amounts of few species of the algae were observed. These were *Cladophora fracta* and *Oedogonium* sp. with lengths never exceeding 50 cm. The polluted Luggie Water (Station 4) was the area supporting the highest amount of filamentous algae in the river with the highest quantities during June-August at the time of low flow. The algae appeared as a green carpet covering the sediment (Plate 4.05). This cover consisted mainly of *Cladophora fracta*, *Cladophora glomerata* and *Vaucheria* sp. *Oedogonium* sp. and *Ulothrix* sp. were also observed in small quantities. At this station the length of *Cladophora* exceeded 4 metres (Plate 4.06). At Station 5 fairly large amounts of the filamentous algae were recorded. These were again dominated by *Cladophora* spp., *Oedogonium* sp., *Vaucheria* sp. and *Microspora* sp. and the length of *Cladophora* was less than 1 m. The records at this station were only for the first half of this study.



Plate 4.05 *Cladophora* spp. forming a green carpet on the river bed
at Station 4 (Luggie Water) during July 1982.

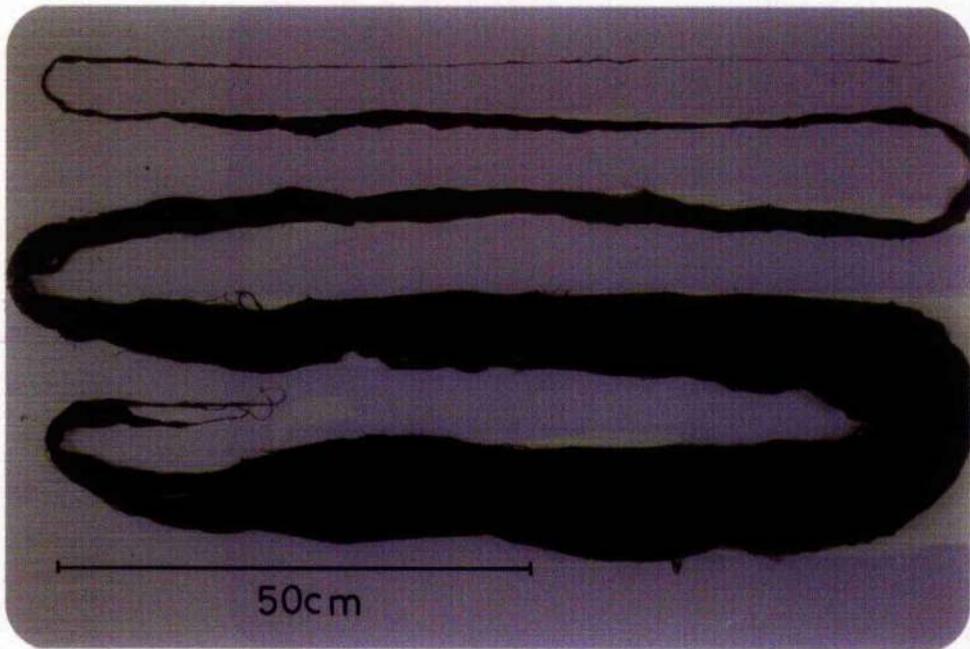


Plate 4.06 *Cladophora fracta* - a single tuft of about 5m length
collected from the same station as in 4.05 at the same
time.

The algae started to disappear with the building of a new bridge over the river. By the time the bridge was completed they disappeared completely and records were not obtained during summer 1982. At Station 6, where the river becomes very slow and sluggish, masses of the algae were found attached mostly to the macrophytic angiosperms which were growing heavily at this station. The same species were found as were observed at Luggie Water with the addition of *Microspora*. No *Cladophora* was observed at Station 7; the dominant filamentous algae observed was *Oedogonium* sp. This was attached to the river bed and was accompanied by *Ulothrix* sp., *Spirogyra* sp., *Oscillatoria* sp., *Lyngbya* sp., and *Vaucheria* sp. All the 9 species of filamentous algae were found at Stations 8 and 9 with higher quantities at Station 9, Garscube Estate, where the *Cladophora* reached 3 m in length and sometimes more. *Cladophora fracta*, *Cladophora glomerata*, *Oedogonium* and *Vaucheria* were the dominant species. The river was dominated completely by *Cladophora glomerata* at Station 10, attached to rocks and rubbish in the river. Its length never exceeded 20-30 cm and sometimes very small amounts of *Vaucheria* was found accompanying it.

At all the different stations along the stretch of the river where these filamentous algae were observed, they were accompanied by masses of epiphytic algae. These were mainly diatoms of the same species that were recorded epiphytic on the macrophytic angiosperms and this aggregation of the diatoms was observed at Station 9 during May 1980 when *Synedra ulna* formed a brownish layer on *Cladophora* in the river (Plate 4.01, Fig. 2C).

Only one desmid, a *Closterium* sp., was observed in the river growing among the filaments of the algae at mostly all the stations excluding Station 7.

Due to the difficulties of sample collection mentioned in section 3.6.1 the filamentous algae could not be measured quantitatively.

4.2.5 Flowering plants

The river supported 4 species of aquatic angiosperms.

These were:

Elodea canadensis Michx.

Potamogeton filiformis Pers. (Plate 4.07)

Potamogeton natans L. (Plate 4.08)

Sparganium emersum Rehm.

These were identified according to Haslam, Sinker and Wolseley (1975).

These plants were present during the summer period at the same level as the filamentous algae. They first become noticeable at the end of April and beginning of May, being very short, pale in colour and leafless. They were larger in size and beginning to dominate the river by the end of May, reaching their maximum during June, July and August. These rooted angiosperms resisted the high flow rates during their period of growth but by mid-September they were decaying and disappearing. Their time of disappearance depended on the strong flow rates during that period. Some angiosperms were collected during October and winter when a low flow rate was recorded e.g. during December 1981 a small plant of *Potamogeton natans* was collected at Station 9 but it was not in a healthy condition. *Potamogeton natans* was very abundant

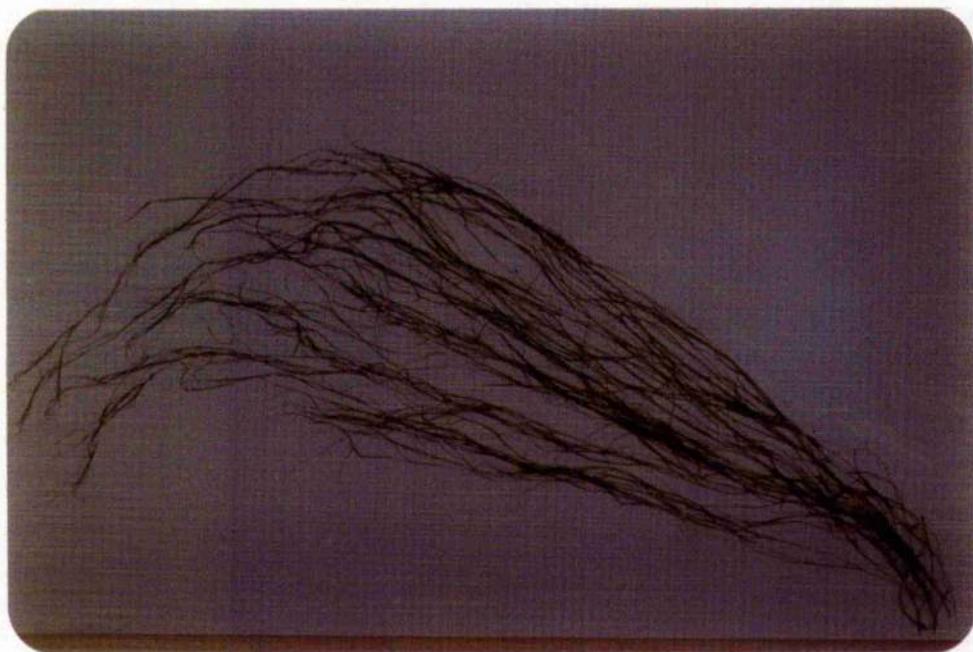


Plate 4.07 *Potamogeton filiformis* pers.

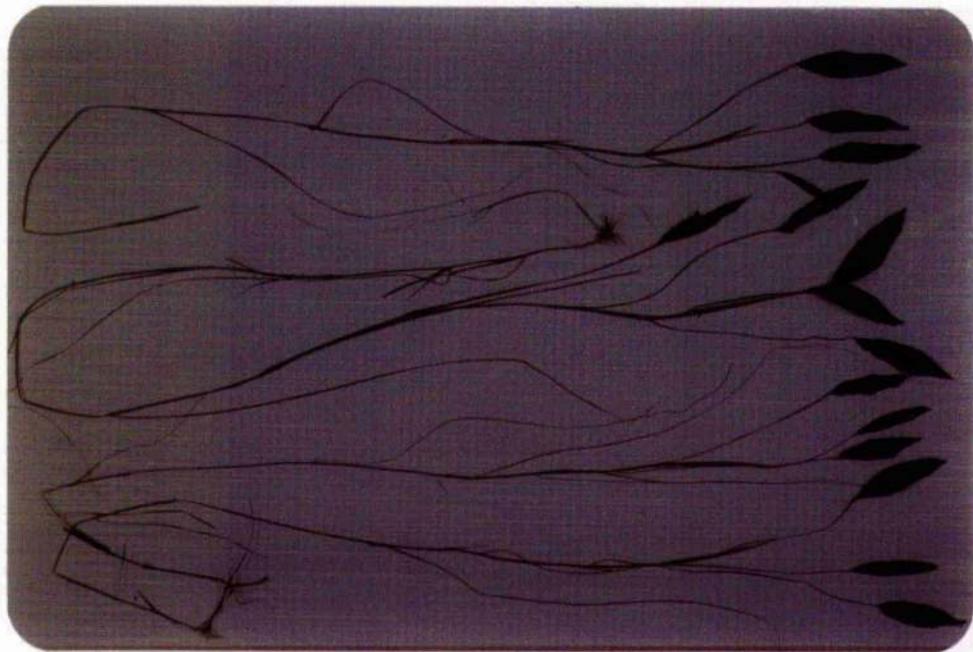


Plate 4.08 *Potamogeton natans* L.

and dominated the river with *Sparganium emersum* at most stations. Its length was 1.5m with floating leaves on the water surface of a brownish green colour. *Sparganium emersum* looked like scattered grass in the river of a green colour, and lengths of about 1m. These two were observed covering the river bed at most stations during the peak growth periods. *Potamogeton filiformis* was found at only 3 stations and was of a bright green colour and a length of about 1m. *Elodea canadensis* was a dwarf angiosperm found in patches at only one station (7) in the river.

Table 4.13 shows the distribution of these different macrophytic angiosperms along the stretch of the river at the 10 stations. At Station 1 there were a few plants of *Sparganium emersum*. At Station 2 *Potamogeton natans* became abundant with the length of more than 1m covering the river bed accompanied by *Sparganium emersum*. At Station 3 only a few plants of *Potamogeton natans* were recorded, scattered in the river, and of lengths not exceeding 60-70 cm. These species were replaced by *Potamogeton filiformis* at Station 4 (Luggie Water) and this plant was common in the river at this station. It had disappeared completely at Station 5, where few scattered *Potamogeton natans* plants of length 60-70 cm only were obtained. This population showed a reduction in numbers during 1981 and 1982 i.e. when the new bridge was finished. A very heavy growth of *Potamogeton natans* reaching a maximum length of about 1.5m was observed at Station 6. This was accompanied by small numbers of *Sparganium emersum*. The latter was found as the common macrophyte in the river at Station 7 which was the only station supporting the growth of *Elodea canadensis*. A few

Table (4.13) The distribution of the macrophytic angiosperms observed in the River Kelvin at its 10 different stations during the period 1980-1982.

(++) abundant, (++) common, (+) few and (-) absent.

Stations	<i>E. canadensis</i>	<i>P. filiformis</i>	<i>P. natans</i>	<i>S. emersum</i>
1	-	-	-	+
2	-	-	+++	++
3	-	-	+	-
4	-	++	-	-
5	-	-	+	-
6	-	-	+++	+
7	+	-	+	++
8	-	+	++	++
9	-	+	+++	+++
10	-	-	-	-

scattered plants of *Potamogeton natans* of lengths 60-70 cm were also obtained from this station. *Potamogeton natans* and *Sparganium emersum* were the common macrophytes growing in the river at Station 8 with a few *Potamogeton filiformis* plants.

The same species were found at Station 9, Garscube Estate, but with a much heavier growth (see Plate 2.11). At the last station, Station 10, no macrophytic angiosperms were observed during the whole period of this study.

4.2.6 The epiphytic algal flora

Most other methods of measuring epiphytic algae have tended to concentrate on the apical regions of the supporting plants (Moss 1981). Preliminary examination of *Potamogeton natans* and *Potamogeton filiformis* showed that the epiphytic diatoms were present over all the plant. Thus three different regions of the supporting plant (top, middle and basal) were examined for (i) the numbers of the epiphytes and (ii) the species diversity.

From the preliminary examinations carried out during 1980, it was found that the epiphytic flora was represented mainly by diatoms. During the growth periods of the years 1981 and 1982 these diatoms were measured quantitatively and the cell numbers were estimated per gram dry weight of the three different regions (top, middle and basal) of the supporting plant *Potamogeton natans* at most of the stations, except Station 4 where *Potamogeton filiformis* was the macrophyte available. These analyses were carried out during the period May - September, the period when the macrophytes are most prominent, for the years 1981 and 1982.

The diatom population differed according to time, station and location on the different regions of the supporting plant. They varied much on the apical, middle and basal portions of the same plant. Table (4.14) displays the total number of the epiphytic diatoms and also the percentage of their distribution along *Potamogeton natans* at Stations 2, 3, 5, 6, 7, 8 and 9 during 1981 and 1982. Table (4.15) shows the epiphytic diatom population along *Potamogeton filiformis* at Station 4 during the same period.

Table (4.14) The total number of the epiphytic diatoms per gram dry weight and the percentage of their distribution on the different regions of *Potamogeton natans* in the River Kelvin during 1981 and 1982 (Total number of cells $\times 10^5$)

Year	Month	cell gm ⁻¹ dry wt.	Distribution	S T A T I O N S						9
				1	2	3	5	6	7	
May		T M B	35 31 34	-	8 40 52	11 48 41	-	-	38 8 55	42 27 31
Total			46.10	-	107.8	178.0	-	-	62.40	20.10
June		T M B	57 21 22	-	-	54 42 4	9 36 58	11 61 28	49 39 12	
Total			52.20	-	-	96.90	18.90	-	5.40	16.80
July		T M B	78 15 72	23 34 43	29 40 32	40 23 37	27 34 39	24 32 44	17 67 16	
Total			105.6	53.0	25.30	37.50	97.0	2.50		22.30
Aug		T M B	43 20 37	40 12 47	50 32 18	29 54 18	39 38 24	78 13 9	51 30 19	
Total			74.70	130.7	262.0	131.6	12.30	64.60		62.80
Oct.		T M B	-	-	-	-	-	-	-	16
Total			-	-	-	-	-	-	-	54
May		T M B	57 24 19	-	-	18 21 61	-	-	19 18 63	47 11 43
Total			39.50	-	-	124.5	-	-	71.90	15.90
June		T M B	63 15 23	-	-	90 4 6	-	-	58 29 13	25 60 15
Total			4.00	-	-	23.00	-	-	4.30	4.02
July		T M B	18 60 21	18 28 54	44 36 20	10 44 46	57 25 17	17 50 33	2 47 51	
Total			38.10	8.30	36.50	103.1	8.70	33.50	13.20	

Table (4.16) The total number of the epiphytic diatoms per gram dry weight and the percentage of their distribution on the different regions of *Potamogeton filiformis* in River Kelvin at Station 4 during 1981 and 1982. (Total number of cells $\times 10^5$)

Year	Month	cell gm ⁻¹ dry wt.	% of the distribution	Station 4
1981	May		T M B	5 20 74
		Total		3.8
	June		T M B	14 64 21
		Total		7.6
	July		T M B	30 25 45
		Total		104
	Aug.		T M B	28 28 44
		Total		300.4
	June		T M B	13 53 34
		Total		4.7
1982	July		T M B	18 41 41
		Total		22.5
	Aug.		T M B	27 25 49
		Total		10.1

Generally the populations recorded during 1981 were larger than in 1982 with the greater numbers always at Station 6. During the whole period an even distribution was observed only at Station 2 during May 1981. Higher populations on the apical region were found almost always at the former station whilst at Stations 3 and 4 the maximum was on the basal portions. At the other stations there was not a fixed distribution for the epiphytic diatoms and they were changeable with the time. The population on the middle region often tended to be the highest again without being in a particular time or location. Although the distribution of the epiphytic diatom population varied on the different regions of the supporting plant, they never formed 65% of the whole population on any area at any station except at Station 6 when the apical region supported 90% of the population during June and August 1982.

During October 1981, due to the low current flow on the collection day, a relatively small *Potamogeton* was collected at Station 9 but it carried a very low population of the epiphytic diatoms.

The data was missing on a few occasions, due either to the non existence of the plant at that time or to the difficulty of getting a sample when the flow rate was relatively high, together with the water turbidity causing very low visibility.

4.2.7 Species diversity of the epiphytic populations

As a result of the two different methods used for dislodging the epiphytic algae from the macrophytic angiosperms, variable results were obtained. During 1980, when boiling in concentrated nitric acid was used, the population was composed mainly of two diatom species.

This low species diversity was also observed in the suspended diatoms at the same time for which low population numbers were also recorded compared with the subsequent years. This technique destroyed the mucilaginous attachments of the epiphytes by hydrolysis and also killed all the algae cells (if there were any) before they became detached. During 1981 and 1982 when Moss's method was applied, greater species diversity were observed and the cells dislodged without being destroyed. Small young filaments of *Cladophora* were also found occasionally attached to the supporting plant during late summer months. The diversity of the epiphytic diatom populations reflected the suspended diatoms during the same period.

During 1980, *Coccconeis placentula* and *Gomphonema parvulum* were the two most numerous epiphytic algae at all the stations (Figures 4.16, 4.17, 4.18, 4.19, 4.20, 4.22 and 4.24). Generally they showed a well defined, recurring seasonal pattern of abundance. *Coccconeis placentula* formed the greater part of the epiphytic population at almost all the stations, particularly on the basal regions. It was also found on the apical and middle regions, except during August when *Gomphonema parvulum* tended to appear in greater numbers but it was still accompanied by the former. Station 8 differed from the other stations in that *Gomphonema parvulum* predominated during the whole period on the whole supporting plant except on the basal region during August and October.

During 1981 and 1982 the former two diatom species were again the most numerous epiphytic species but they failed to occur in a well defined seasonal pattern. They appeared with few other species

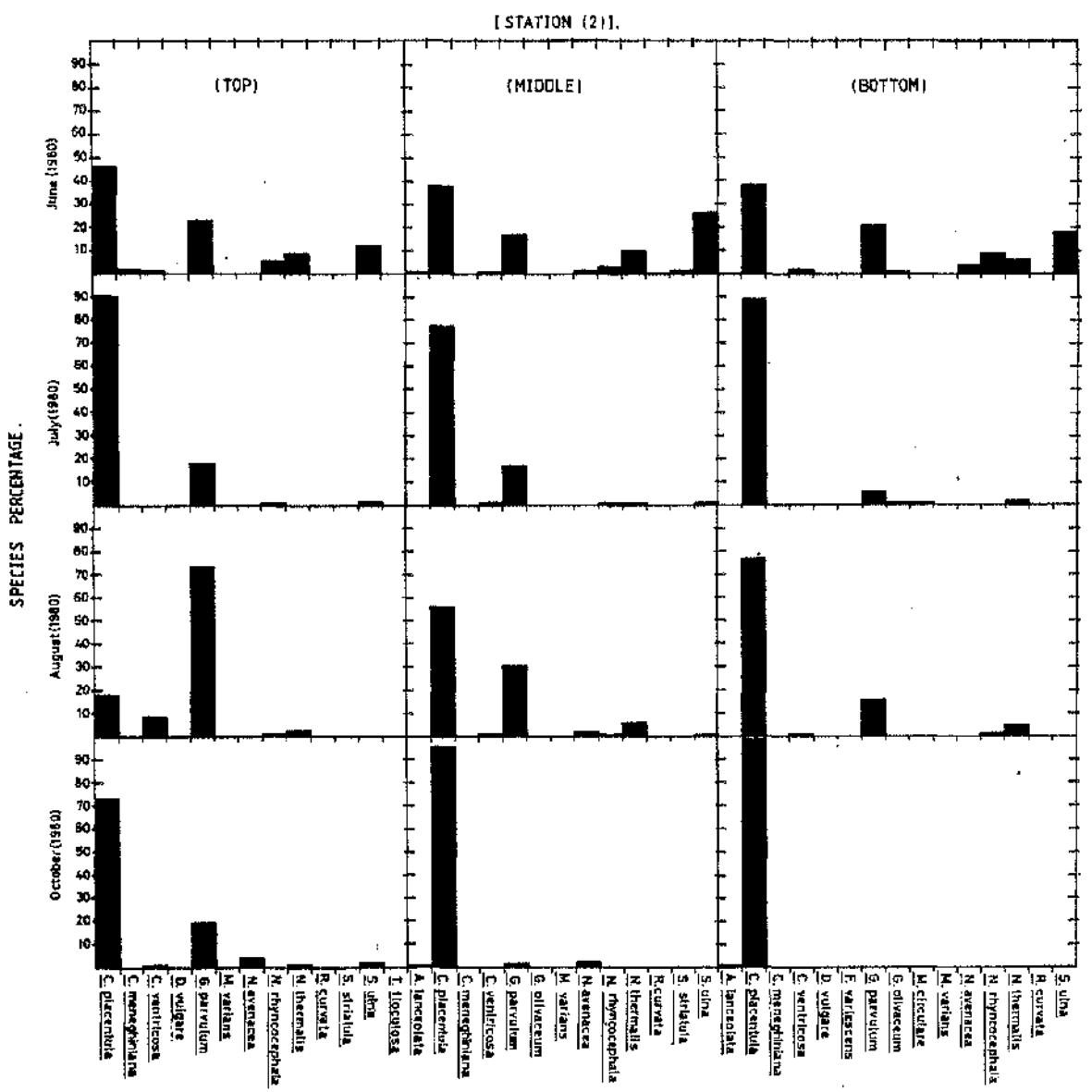


Figure (4.16) Percentage of the different species of the epiphytic diatoms recorded on the different regions, Top, Middle and Bottom, of *Potamogeton natans* in the River Kelvin at Station 2 during 1980.

Figure (4.17) Percentage of the different species of the epiphytic diatoms recorded on *Potamogeton filiformis* in the River Kelvin at Station 4 during 1980.

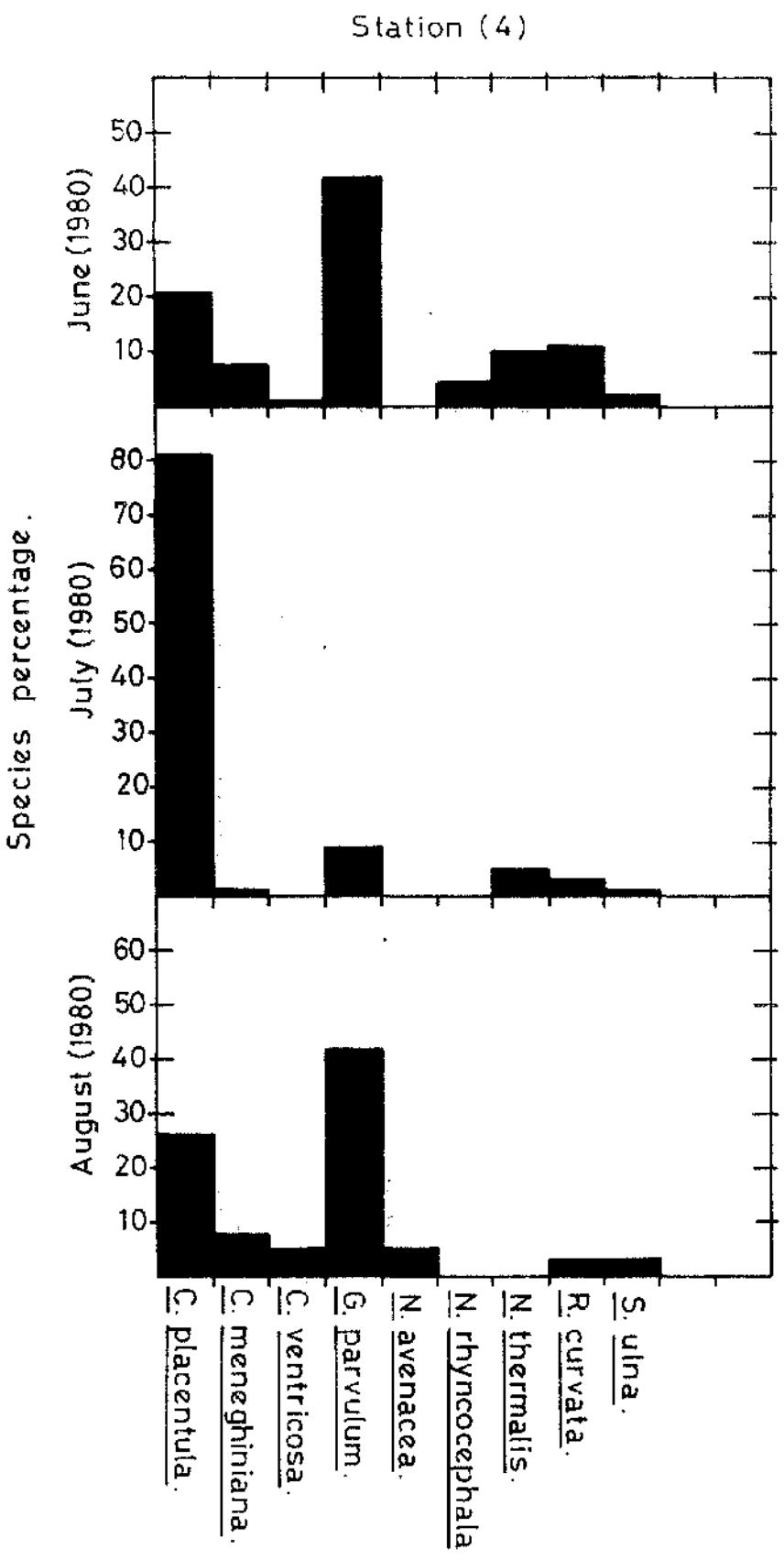


Figure (4.17)

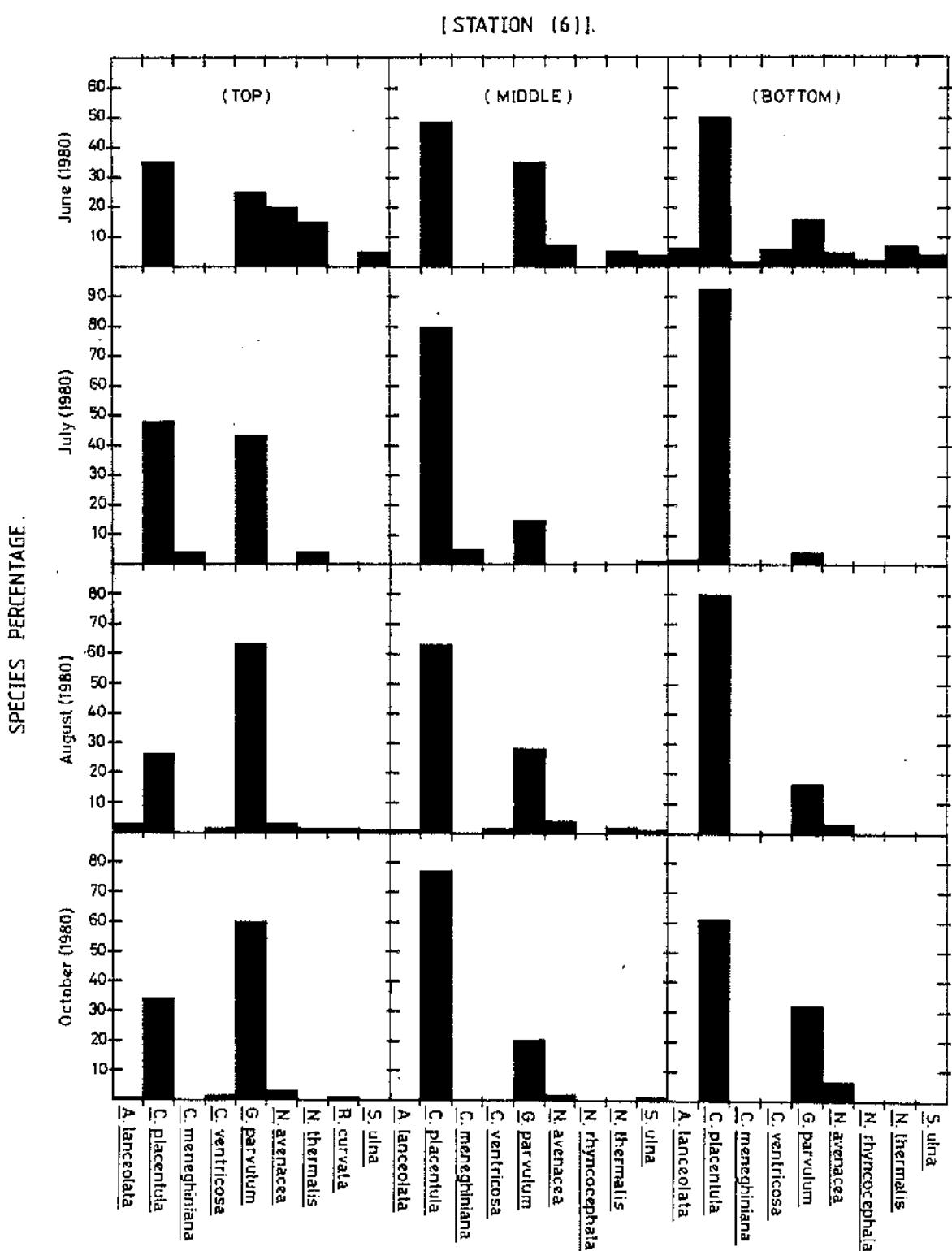


Figure (4.18) Percentage of the different species of the epiphytic diatoms recorded on the three regions, Top, Mid and Bottom, of *Potamogeton natans* in the River Kelvin at Station 6 during 1980.

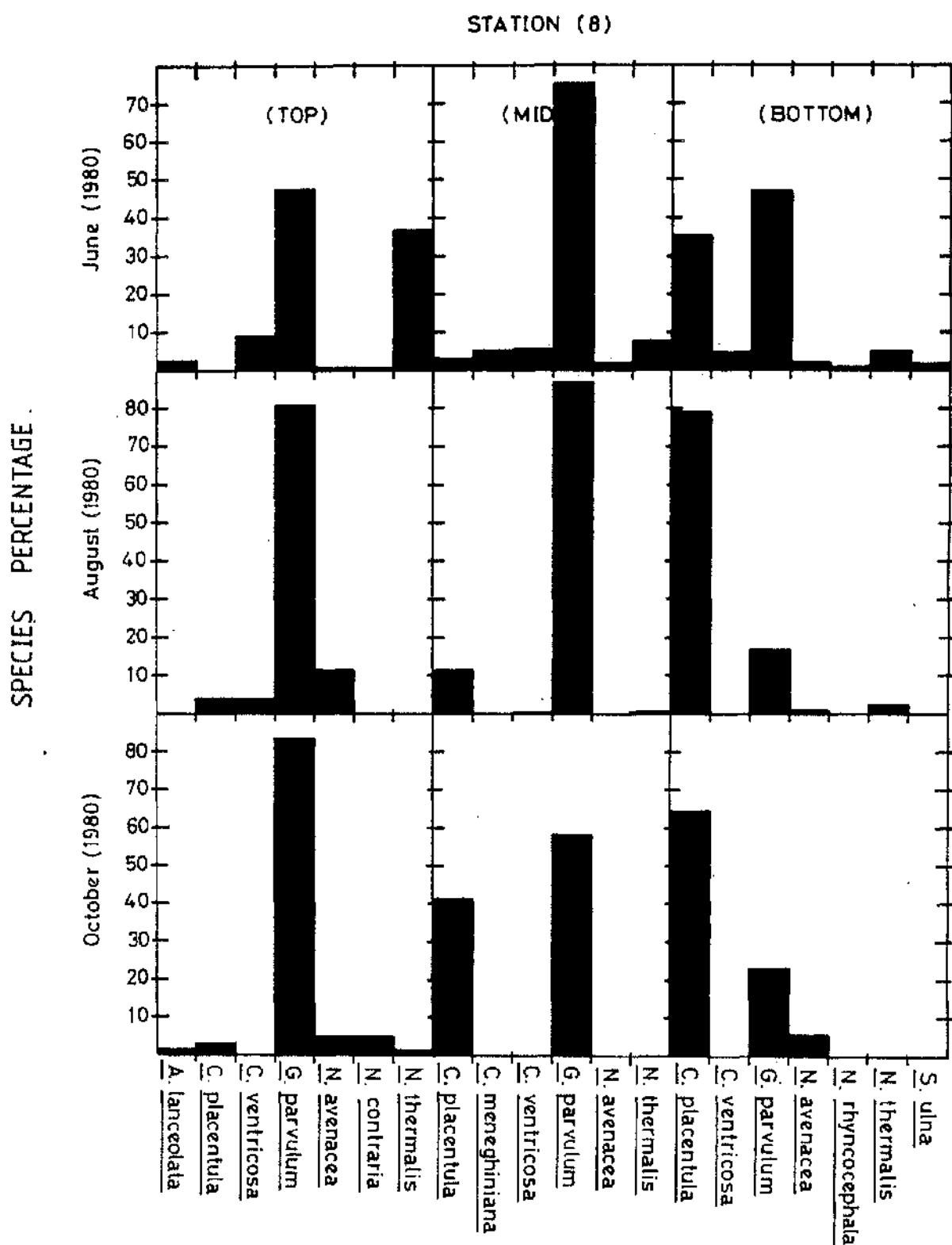


Figure (4.19) Percentage of the different species of the epiphytic diatoms recorded on the three regions, Top, Mid and Bottom, of *Potamogeton natans* in the River Kelvin at Station 8 during 1980.

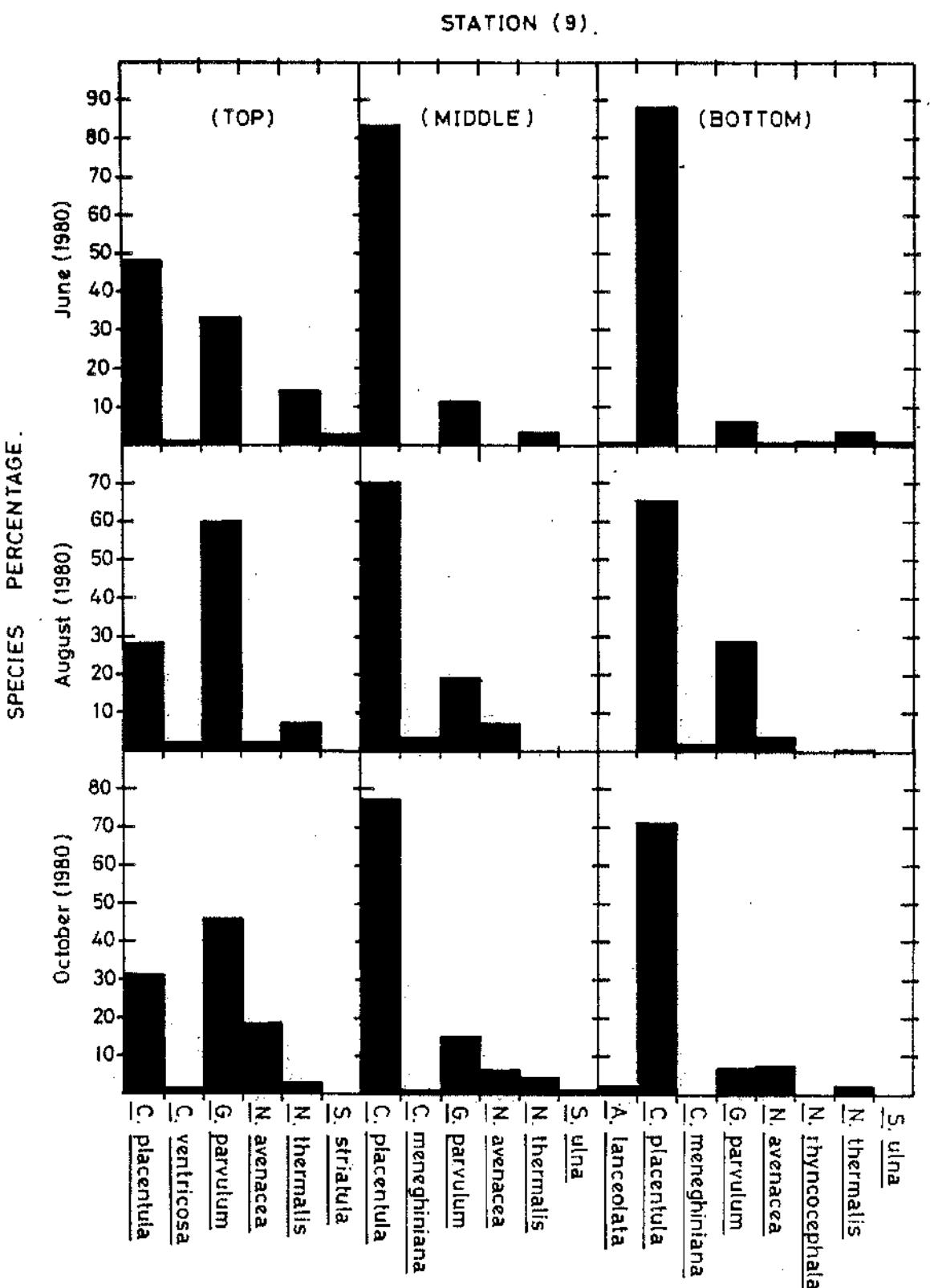


Figure (4.20) Percentage of the different species of the epiphytic diatoms recorded on the three regions, Top, Mid and Bottom, of *Potamogeton natans* in the River Kelvin at Station 9 during 1980.

of diatoms, mainly *Cyclotella meneghiniana*, *Navicula avenacea*, *Navicula rhyncocephala*, *Nitzschia thermalis* and *Synedra ulna*.

During 1981 *Comphonema parvulum* and *Nitzschia thermalis* were found in high numbers at Station 2 (Fig. 4.21) in May and June and on the whole supporting plant. During the former month they accompanied *Synedra ulna* but in the latter *Coccconeis placentula* was observed instead, being greater in numbers on top and middle portions. *Nitzschia thermalis* increased in numbers during July on the whole plant with *Coccconeis placentula* on the middle region. The latter was observed in large numbers on the distal part during August, whilst on the middle and base, *Melosira varians* was observed (this station supported the highest population of *Melosira varians*). *Coccconeis* had shifted its population down to the middle and basal regions during October whilst *Nitzschia thermalis* was found on the top. During May 1982 the same species as in the previous May were recorded at these stations with some changes. These were the greater numbers of *Navicula rhyncocephala* on all the regions during May and June, the delay in the appearance of *Synedra ulna* until June and July and the replacement of *Coccconeis placentula* by *Cyclotella meneghiniana* on all the parts during the latter month.

At Station 3 (Fig. 4.22) *Nitzschia thermalis* was recorded in great numbers during July and August 1981 on the whole supporting plant except the mid region during the latter month where *Cyclotella meneghiniana* was found instead. It was also recorded on the other two regions accompanied by *Comphonema parvulum* on the top and *Melosira varians* on the base. During July 1982 *Cyclotella*

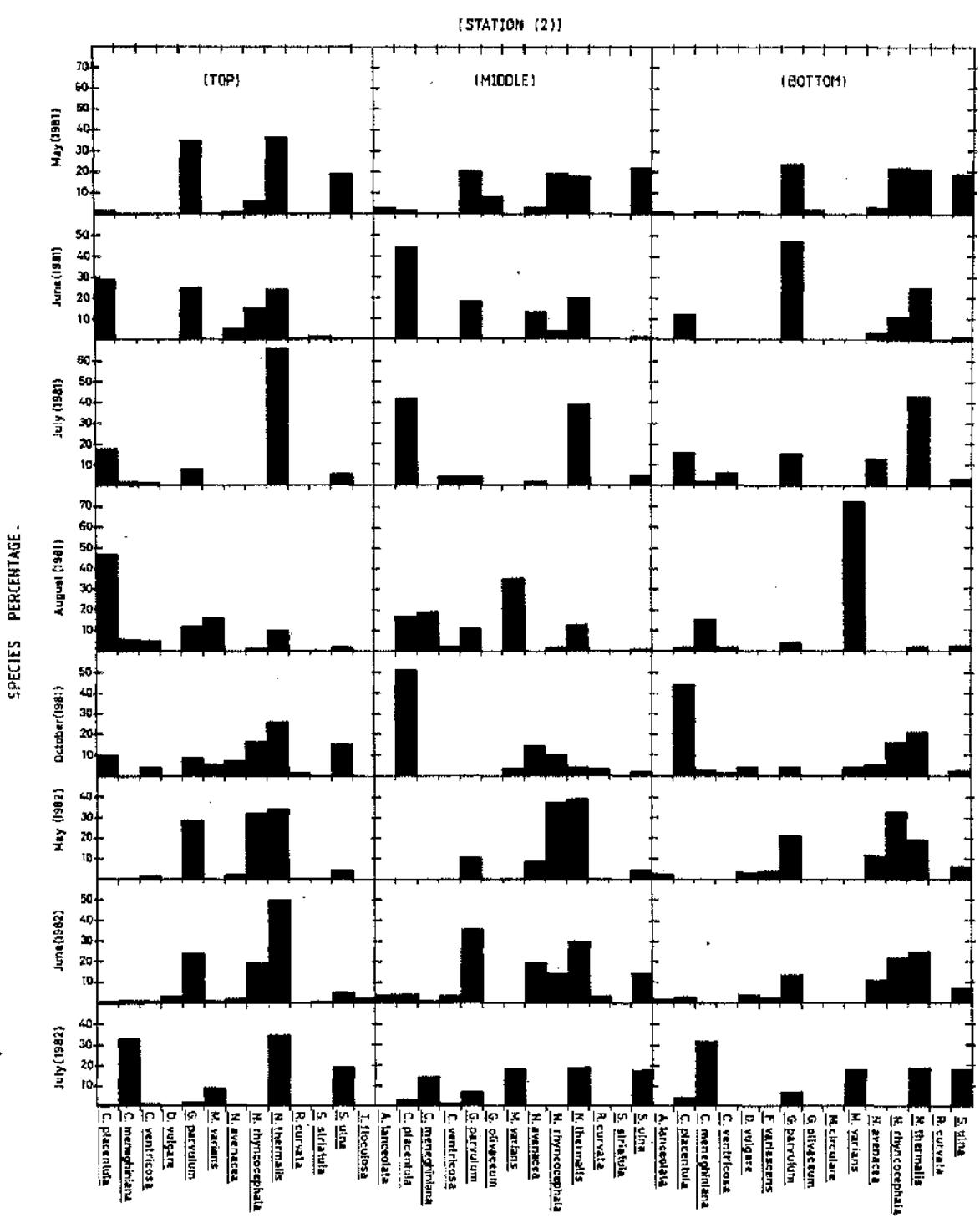


Figure (4.21) Species percentage of the epiphytic diatoms population recorded on Top, Middle and Bottom, of *Potamogeton natans* in the River Kelvin at Station 2 during 1981 and 1982.

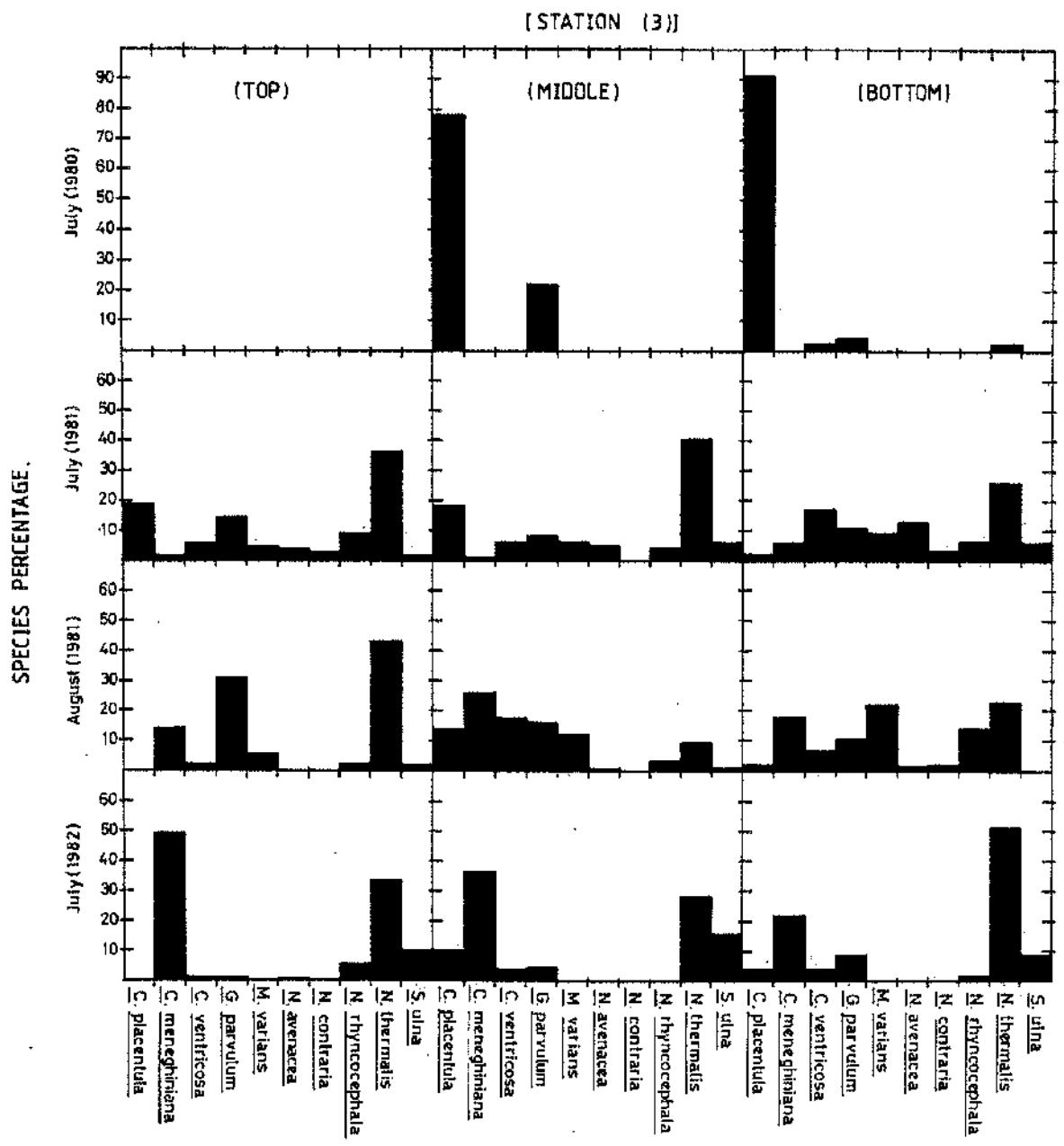


Figure (4.22) Species percentage of the epiphytic diatoms population recorded on Top, Middle and Bottom of *Potamogeton natans* in the River Kelvin at Station 3 during 1980 — 1982.

meneghiniana and *Nitzschia thermalis* occurred on all the supporting plant.

The epiphytic diatom population, during May 1981 at Station 4 (Fig. 4.23) consisted of *Nitzschia thermalis* and *Navicula rhyncocephala* on the top and middle regions, the latter being replaced by *Navicula contraria* on the basal part. The whole population was represented by *Cocconeis placentula* during June and July at this station. It showed a drop in numbers during August, being replaced by *Nitzschia thermalis* with *Cymbella ventricosa* and *Rhoicosphenia curvata* particularly on the middle and basal regions. During 1982 the epiphytic species recorded had completely changed from the previous year, with *Nitzschia thermalis* the most numerous of the species during June on all the regions and *Cyclotella meneghiniana* during July.

At Station 5 (Fig. 4.24) the epiphytic population did not vary much from Station 3 during May 1981. The only noticeable change was in the abundance of *Cymbella ventricosa* on the apical region. *Cocconeis placentula* was brought into the river in high quantities by Luggie Water (Station 4) and found in great numbers during July on the whole supporting plant. It was also recorded during August with a drop in numbers on the distal region, being replaced by *Cymbella ventricosa*. Similar to the previous stations, *Cyclotella meneghiniana* was found in large amounts on all the regions being with *Nitzschia thermalis* particularly on mid and basal portions.

Synedra ulna was recorded in great numbers at Station 6 (Fig. 4.25) during May 1981 on all the regions with *Navicula avenacea*. During June, July and August *Nitzschia thermalis* showed good growth

Figure (4.23) Species percentage of the epiphytic diatoms population recorded on Top, Middle and Bottom of *Potamogeton filiformis* in the River Kelvin at Station 4 during 1981 and 1982.

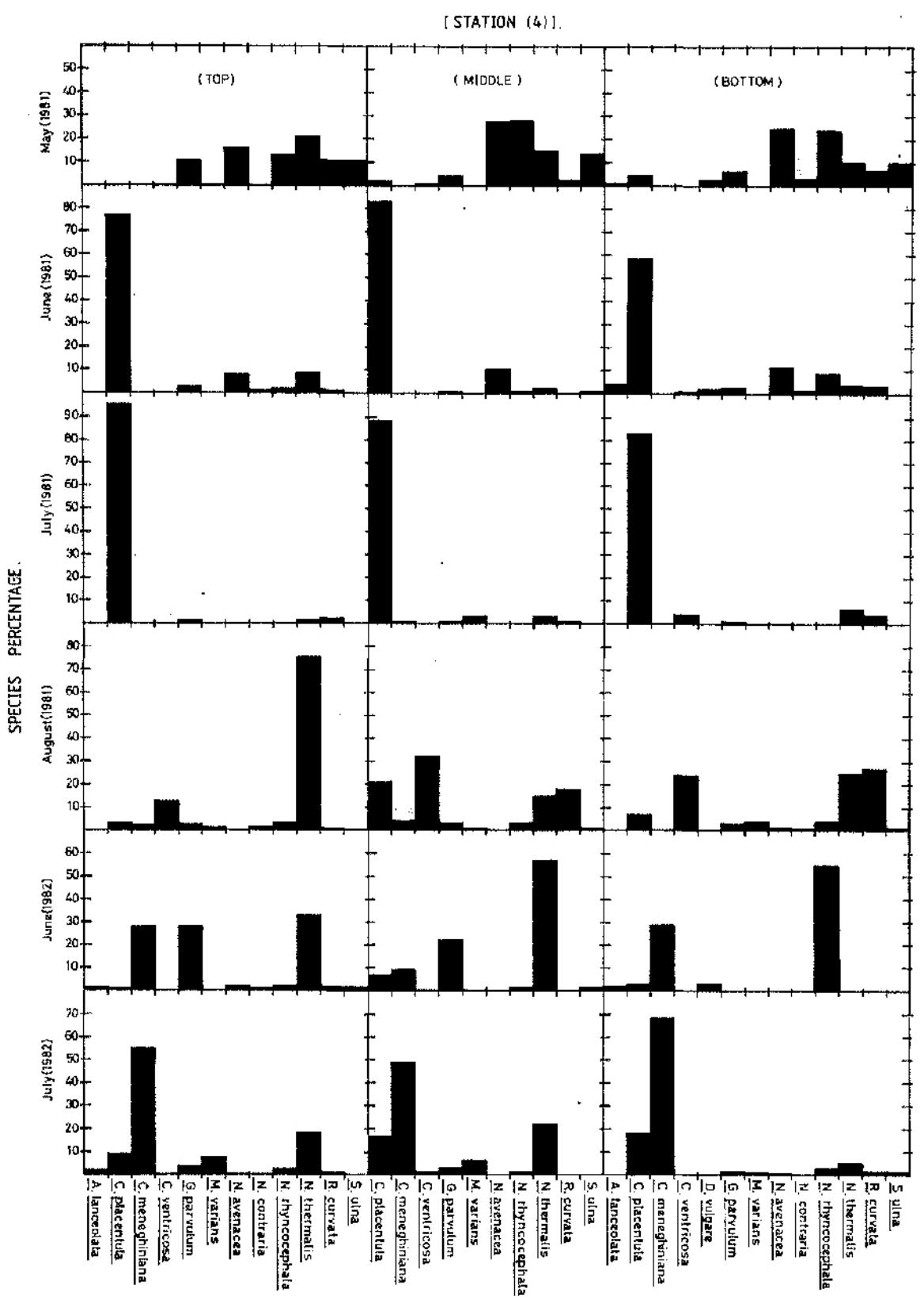


Figure 4.23

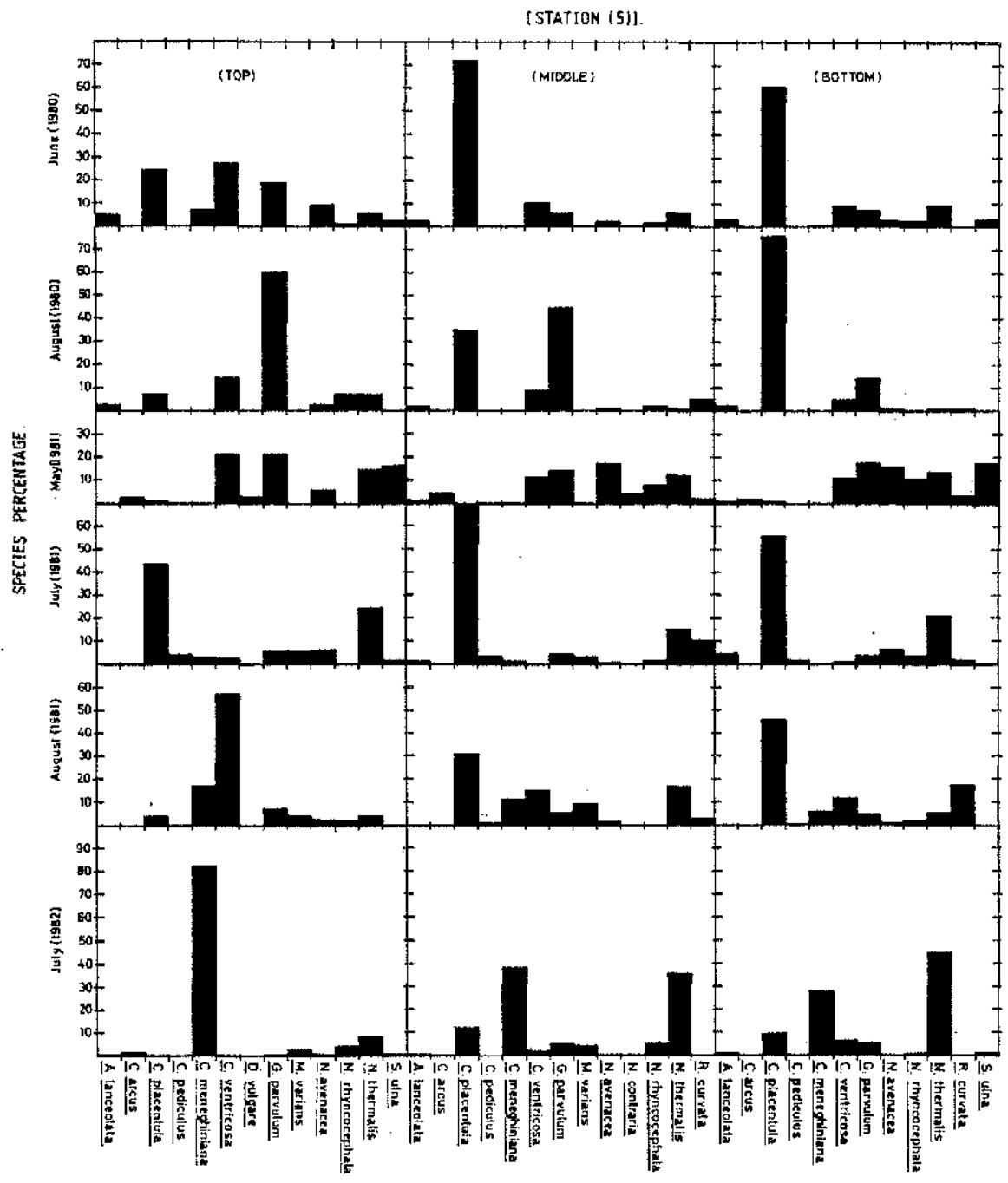


Figure (4.24) Species percentages of the epiphytic diatom population recorded on Top, Middle and Bottom of *Potamogeton natans* in the River Kelvin at Station 5 during 1980 — 1982.

Figure (4.25) Species percentages of the epiphytic diatom population recorded on Top, Middle and Bottom of *Potamogeton natans* in the River Kelvin at Station 6 during 1981 and 1982.

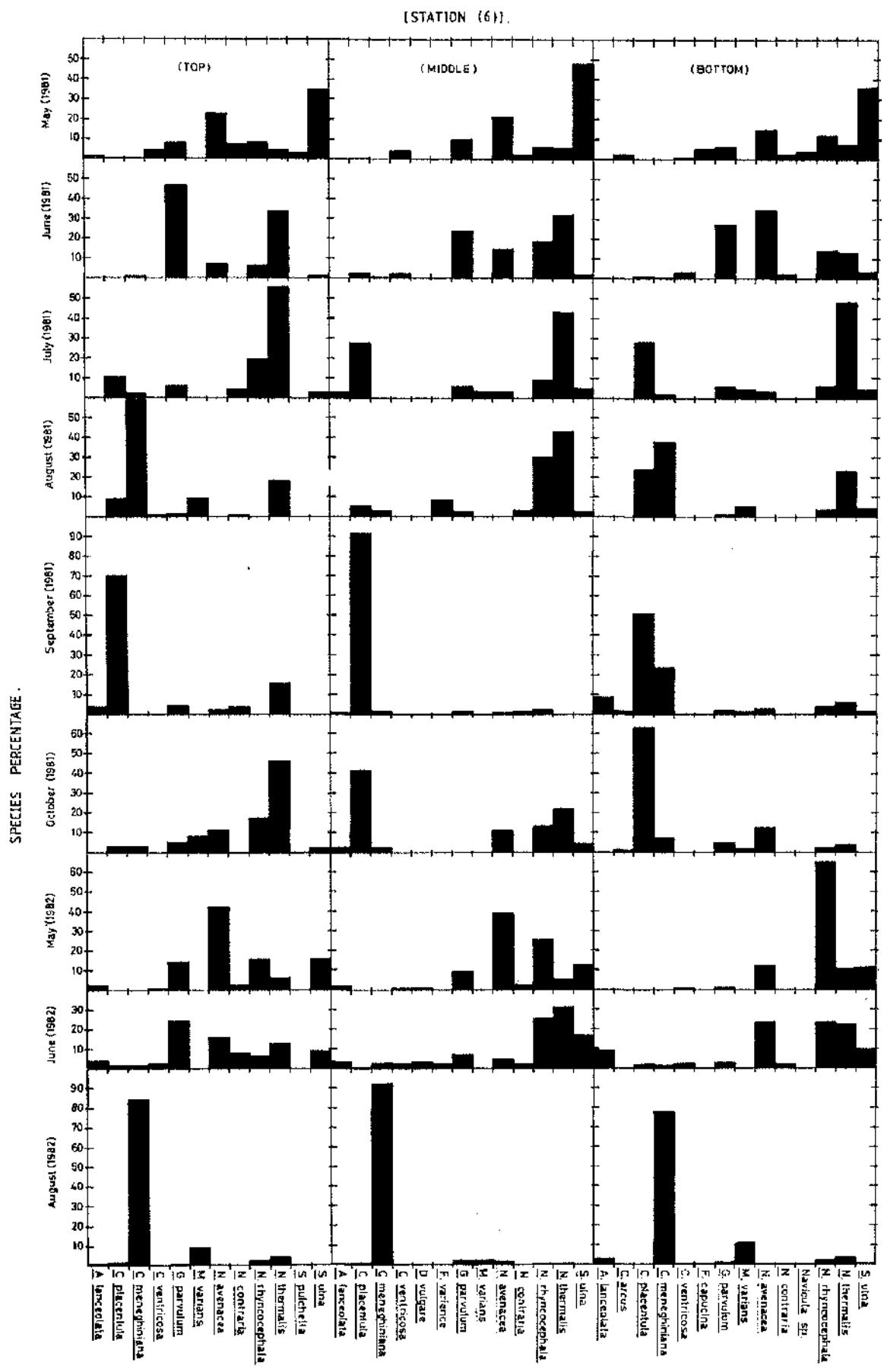


Figure 4.25

on all the regions with *Gomphonema parvulum* during June and *Cocconeis placentula* in July on the whole plant, whilst during August it was found with *Cyclotella meneghiniana*, especially on the top and basal regions and *Navicula rhyncocephala* on the middle. These all dropped in numbers during September, being replaced by *Cocconeis placentula* on all the regions. This was observed during October also except on the apical portion where *Nitzschia thermalis* was found. During May 1982 the population showed alterations with the previous year. *Navicula avenacea* and *Navicula rhyncocephala* composed a great part of the epiphytic population on the whole supporting plant. They were also found during June with *Gomphonema parvulum* on the top and *Nitzschia thermalis* on the mid and basal regions. Similar to the previous stations, a good growth for *Cyclotella meneghiniana* was observed during August.

The epiphytic population at Station 7 (Fig. 4.26) consisted mainly of *Nitzschia thermalis* during the whole period of 1981 and 1982. It accompanied *Gomphonema parvulum* on all the plant during June, whilst during July *Navicula* sp. was recorded on the distal portion only (the only station with great quantities of *Navicula* sp.). During August *Gomphonema parvulum* appeared again but on the top only. Finally, during August 1982 it was recorded with *Cyclotella meneghiniana* on the basal part only.

At Station 8 (Fig. 4.27) *Nitzschia thermalis* was observed in large quantities during summer 1981 on almost all the regions except the basal during May when *Synedra ulna* and *Diatomella elongatum* was found replacing it. The latter two were observed also on the middle

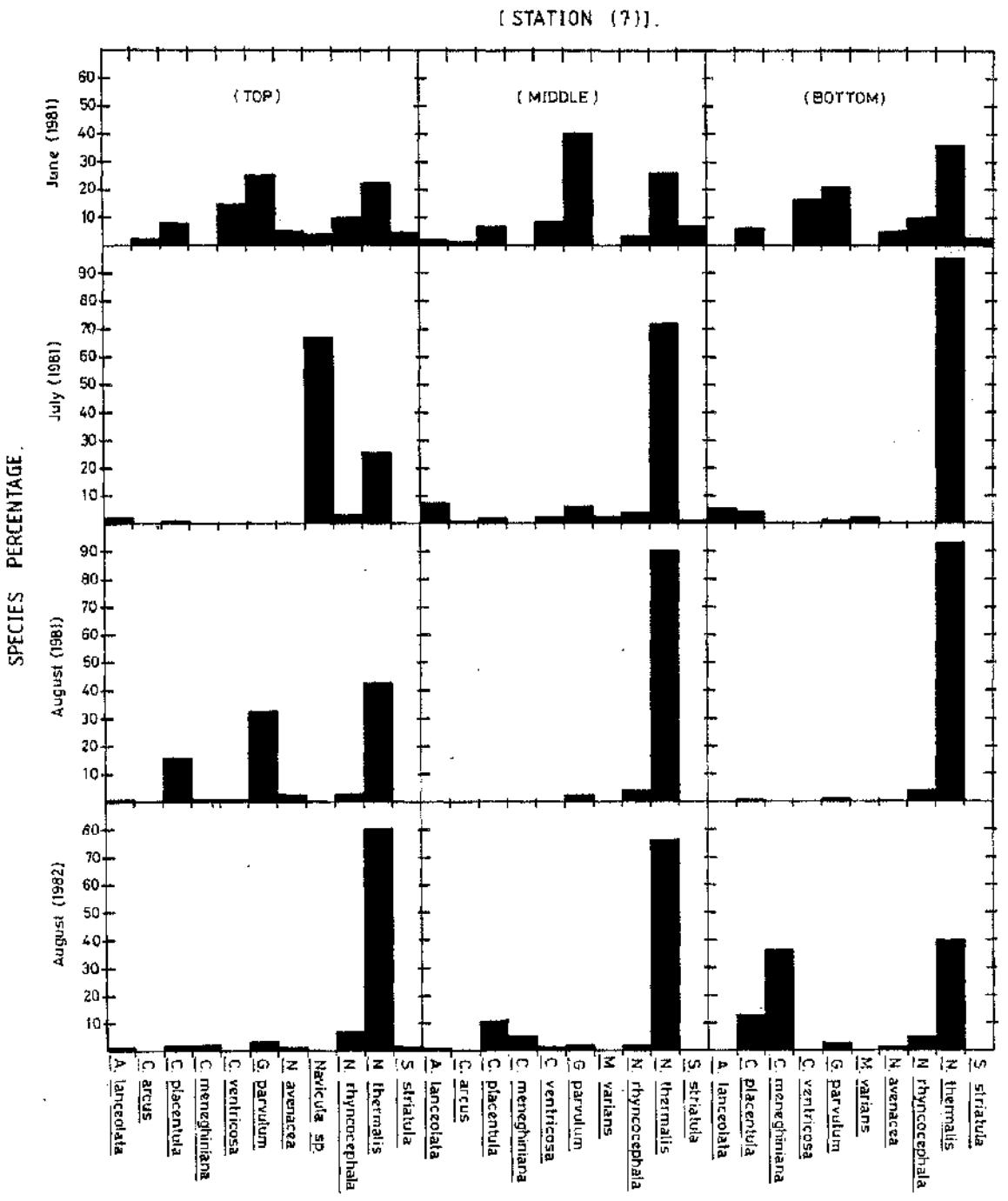


Figure (4.26) Species percentages of the epiphytic diatom population on Top, Middle and Bottom of *Potamogeton natans* in the River Kelvin at Station 7 during 1981 and 1982.

Figure (4.27) Species percentages of the epiphytic diatom population recorded on Top, Middle and Bottom of *Potamogeton natans* in the River Kelvin at Station 8 during 1981 and 1982.

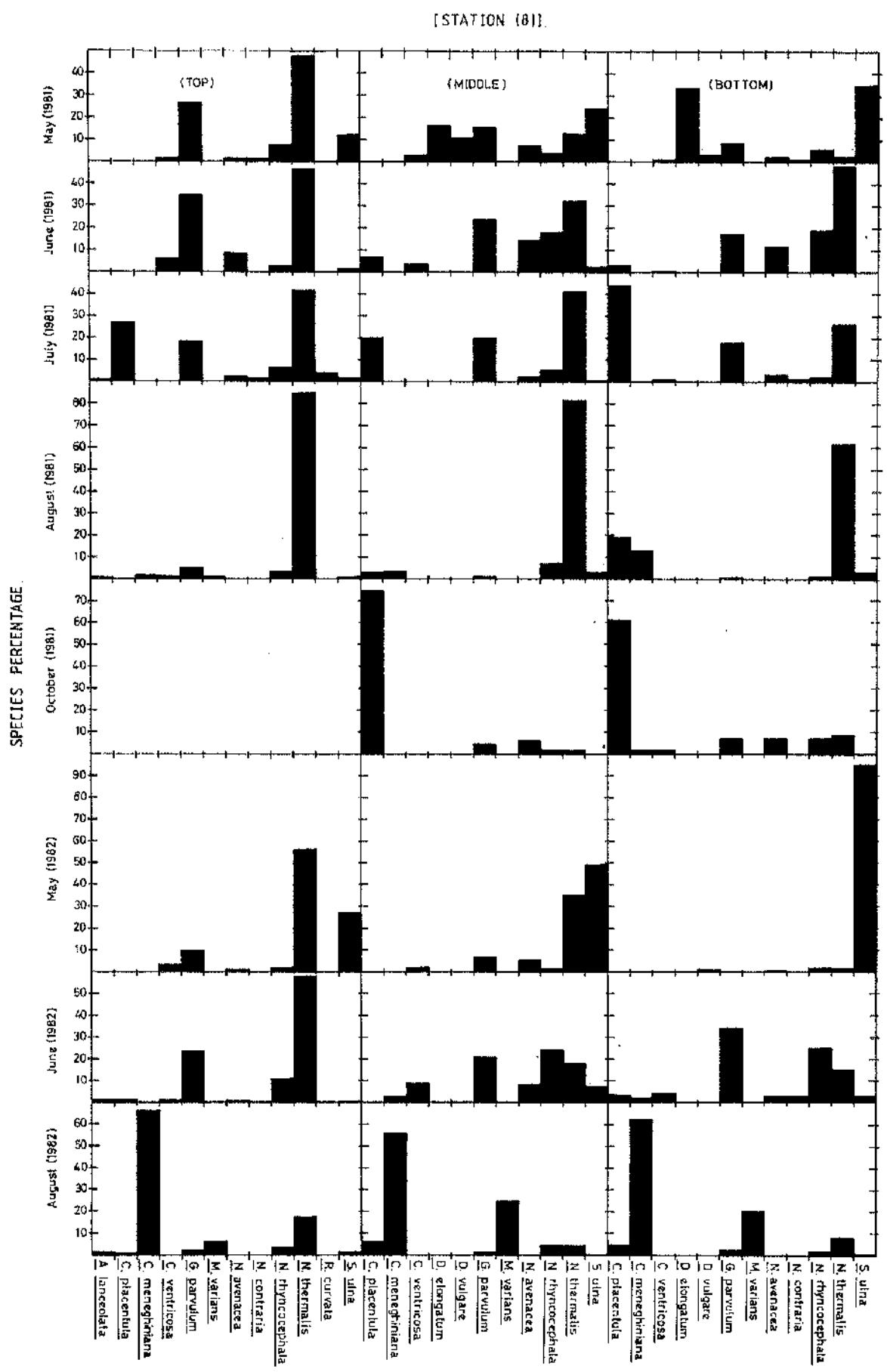


Figure 4.27

portion. *Gomphonema parvulum* accompanied *Nitzschia* during May, June and July. *Coccconeis placentula* showed a growth during July and dropped in numbers for the rest of the summer and reappeared again in October on mid and basal parts only. *Nitzschia thermalis* was recorded during May 1982 particularly on the apical and middle regions accompanied by *Synedra ulna* which formed a high percentage on the basal area. The former was observed during June along with *Gomphonema parvulum* and *Navicula rhyncocephala*. During August the good growth of *Cyclotella meneghiniana* replaced the former diatoms accompanied by *Melosira varians* (this station is the only one with relatively high quantities of *Diatoma elongatum*).

The epiphytic population differed at Station 9 (Fig. 4.28) where *Gomphonema parvulum* was found in large numbers on the top region during May and June 1981. It showed growth on the middle part also during the latter month, being replaced at the same time by *Navicula avenacea* on the basal region. During the former month *Nitzschia thermalis* was found in relatively smaller quantities on the whole plant. The population expanded during July and August with *Coccconeis placentula* on the middle and basal regions in the former month. *Nitzschia thermalis* showed a drop in quantity during October, particularly on the top and middle regions and *Coccconeis placentula* increased in numbers on these regions. These two diatoms also found composing the greatest part of the epiphytic population during December. *Nitzschia thermalis* showed a good growth during 1982 on all the supporting plant with *Synedra ulna* on the middle plus *Navicula avenacea* on the basal regions. During June it accompanied *Navicula*

Figure (4.28) Species percentages of the epiphytic diatom population recorded on Top, Middle and Bottom of *Potamogeton natans* in the River Kelvin at Station 9 during 1981 and 1982.

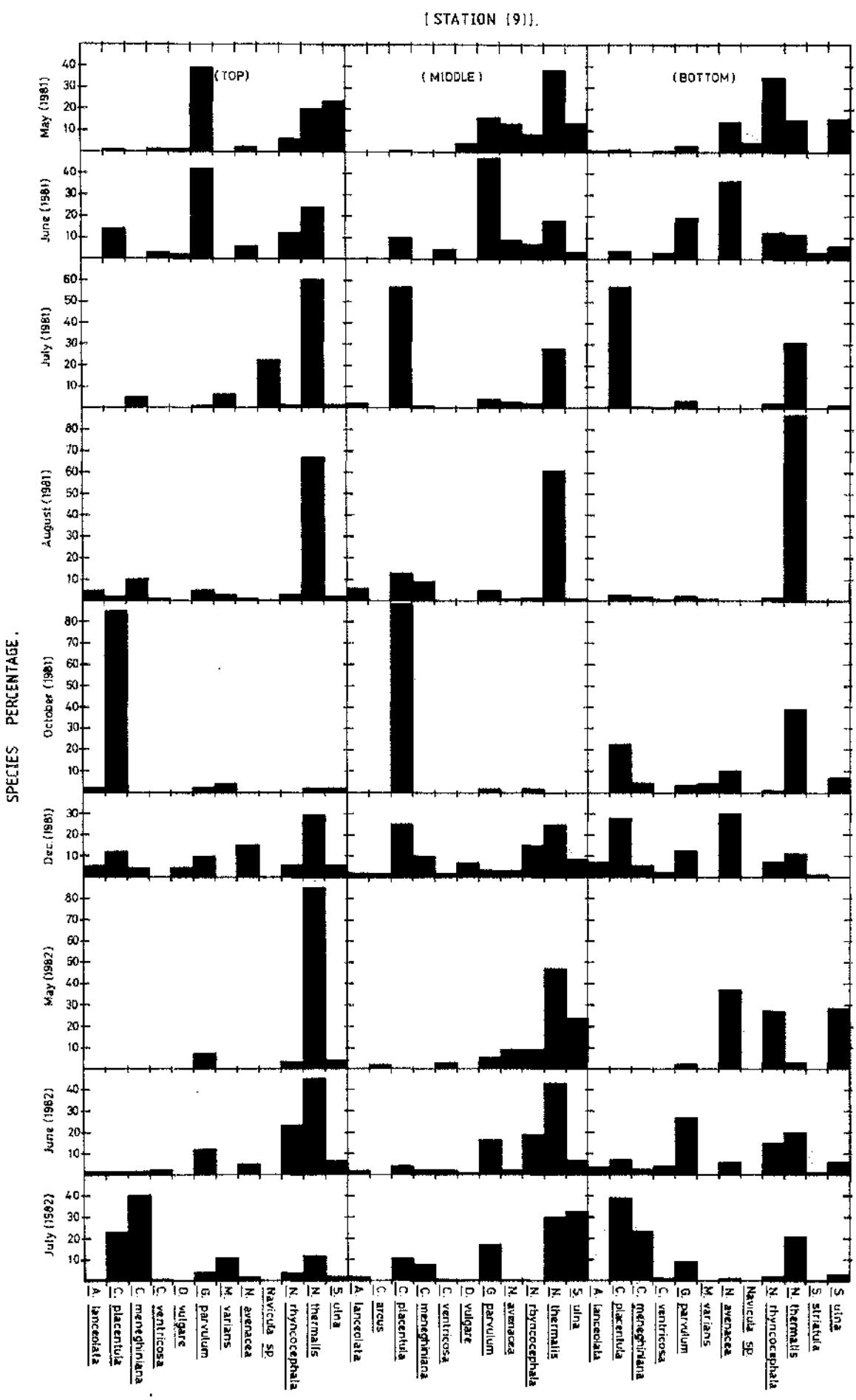


Figure 4.28

rhyncocephala and *Gomphonema parvulum* on all the regions. *Synedra ulna* appeared again during July on the middle part while on the other two portions *Cyclotella meneghiniana* and *Coccconeis placentula* were both found.

4.2.8 Use of natural substrata

The problems in the study of natural periphyton populations have led some workers to study the attached algae which grow on natural or artificial substrates placed in the environment for known lengths of time. The attraction of this method is that the algae grow displayed over a surface, the nature and size of which may be chosen to suit the particular circumstances of the investigation. Also, since a new surface is used every time, some control of substratum conditions is possible.

Due to the problems mentioned in section 3.6.3 only small amounts of data were obtained from this technique. Initially it was found that during the first week of exposing the surfaces only diatoms and few unicellular algae attached themselves to the pebble surfaces. Zoospores and filamentous algae were observed growing on them when they were left in the environment for a longer period. The diatoms recorded were:

Achnanthes lanceolata

Ceratoneis arcus

Cymbella ventricosa

Gomphonema parvulum

Navicula avenacea

*Nitzchia thermalis**Synedra ulna*

with *Navicula avenacea* and *Nitzschia thermalis* the dominant. These species reflected the diatom species in the river but finding four species of the unicellular green algae attached did not show any similarity with the epiphytic flora in the river. These were:

Chlamydomonas sp.

Gleocystis major Gerneck

Pamella mucosa Kutz

Scenedesmus quadricauda (Turp.) de Breb.

The filamentous algae found attached also reflected the same species recorded in the river. They were:

Cladophora fracta

Oedogonium sp.

Oscillatoria sp.

Ulothrix sp.

Vaucheria sp.

They were found growing in a healthy condition in the Bold's basal media inside the containers. *Oedogonium sp.* was found in a mature reproductive stage after 5-6 weeks growing; it was identified as:

Oedogonium crassum (Hass.) Wittrock.

These filamentous algae observed became attached to these natural surfaces during June, July and August whilst during May only diatoms and the unicellular algae were recorded growing on them.

4.2.9 Carbon fixation by the epiphytic diatoms using ^{14}C

This technique was used out in this survey to compare the living activity of the epiphytes from the different regions.

The data for these experiments are shown in Table (4.16) which indicates the carbon fixed by the epiphytes per gram dry weight of the apical, middle and basal regions of *Potamogeton natans* at Stations 6 and 9, Bardawi Bridge and Garscube Estate respectively, during June and July 1982. The data shows higher fixation on the apical regions at both of the stations although more often greater populations were recorded on the mid and the basal areas (Tables 4.14 and 4.15). These data would suggest that the epiphytes near the apical region are more active metabolically than those on other regions of the supporting plant. These results would seem to confirm the use of apical regions of macrophytes for the study of changes in epiphyte populations, as recommended by Moss (1981).

Table (4.16) Carbon fixed in $\text{mg C l}^{-1} 6\text{h}^{-1}$ by the epiphytic diatoms per gram dry weight of *Potamogeton natans* at Stations 6 and 9 during June 1982.

Regions of supporting plant	$\text{mg C l}^{-1} 6\text{h}^{-1}$		cell gm^{-1} dry wt.	
	St. (6)	St. (9)	St. (6)	St. (9)
Top	0.12	0.009	42.5×10^5	4.1×10^5
Middle	0.02	0.003	92.5×10^5	15.6×10^5
Base	0.012	0.0004	43.5×10^5	25×10^5

4.3 Bioassay procedure to assess growth potential of the River Kelvin water

Kelvin water

Fresh water bioassay procedures have been proposed as a fairly accurate and convenient method for determining the growth potential of a range of natural waters. Skulberg (1964) reviewed the use of laboratory cultures of algae to test the fertility of waters. The need for standardisation of algal assay procedure has also been discussed (Oswald and Gaonkar 1969). Subtle changes in water quality, which cannot be detected by many programmes of chemical analysis, are often shown by bioassay tests. There are advantages and limitations to these procedures. It is possible to determine with some accuracy the effect of a given addition to the water upon the test organism and so isolate this from other variables. However, under natural conditions the organisms are exposed to predator pressure and competition and also variable environmental conditions. The bioassay technique can provide much useful information, particularly if it forms part of a larger study programme on the water bodies in question.

Most of the bioassay tests are carried out using a single organism in the laboratory (Marvan, Pribil and Lhotsky, 1979) and the usefulness of these methods were pointed out by measuring cell growth and division and chlorophyll a contents of the cultures with less emphasis on other aspects of algal metabolism. A question arising here is: are these growth and pigment measurements adequate as indicators of water quality? As a means of further assessing water quality in this study, the growth of two species of green algae were compared and other relevant aspects of their living activities examined.

The major problem with any bioassay technique is the choice of the test organism. The assays were carried out in this study using two species of green algae. These were

Ankistrodesmus falcatus Plate 4.09, Fig. 5
and *Scenedesmus quadricauda* Plate 4.09, Fig. 1

They have been frequently used in bioassay studies and they were also found occasionally in the river samples. The culture history of these organisms was well known. The optimum conditions for their growth were assessed by growing them under defined laboratory conditions, prior to their use as test organisms, in Bold's basal medium. This is a balanced mineral medium which has been used for algae from eutrophic waters. It produces healthy normal cells and contains principally six salts, e.g. NaNO_3 , $\text{CaCl}_2 \cdot 2\text{H}_2\text{O}$, $\text{MgSO}_4 \cdot 7\text{H}_2\text{O}$, K_2HPO_4 , KH_2PO_4 and NaCl and four other solutions in trace amounts. These were $\text{FeSO}_4 \cdot 7\text{H}_2\text{O}$ in acidified water, H_3BO_3 and finally a mixture of $\text{ZnSO}_4 \cdot 7\text{H}_2\text{O}$, $\text{MnCl}_2 \cdot 4\text{H}_2\text{O}$, MoO_3 , $\text{CuSO}_4 \cdot 5\text{H}_2\text{O}$ and $\text{Co}(\text{NO}_3)_2 \cdot 6\text{H}_2\text{O}$.

The two alga were found to grow very well in this medium and this growth was hence used as a control with every set of river cultures.

4.3.1 Measuring the cell numbers

The algae were exposed to the river water over periods of 15-18 days. The river water was filtered twice through G/CF filter paper and Millipore filters and the samples were inoculated with one week old cultures of the alga and placed in a growth cabinet under controlled conditions of temperature (15°C) and light intensity of continuous illumination of 5 W m^{-2} . The growth in these cultures was represented

Plate (4.09) The unicellular green algae used in the bioassay experiments:

- 1 - *Scenedesmus quadricauda*
- 2 - *Scenedesmus quadricauda* - these cells have lost their starch
- 3 & 4 - *Scenedesmus quadricauda* - dissociated coenobia, 5. *Ankistrodesmus falcatus*
- 6 - *Ankistrodesmus falcatus* showing cell 'clumps'.

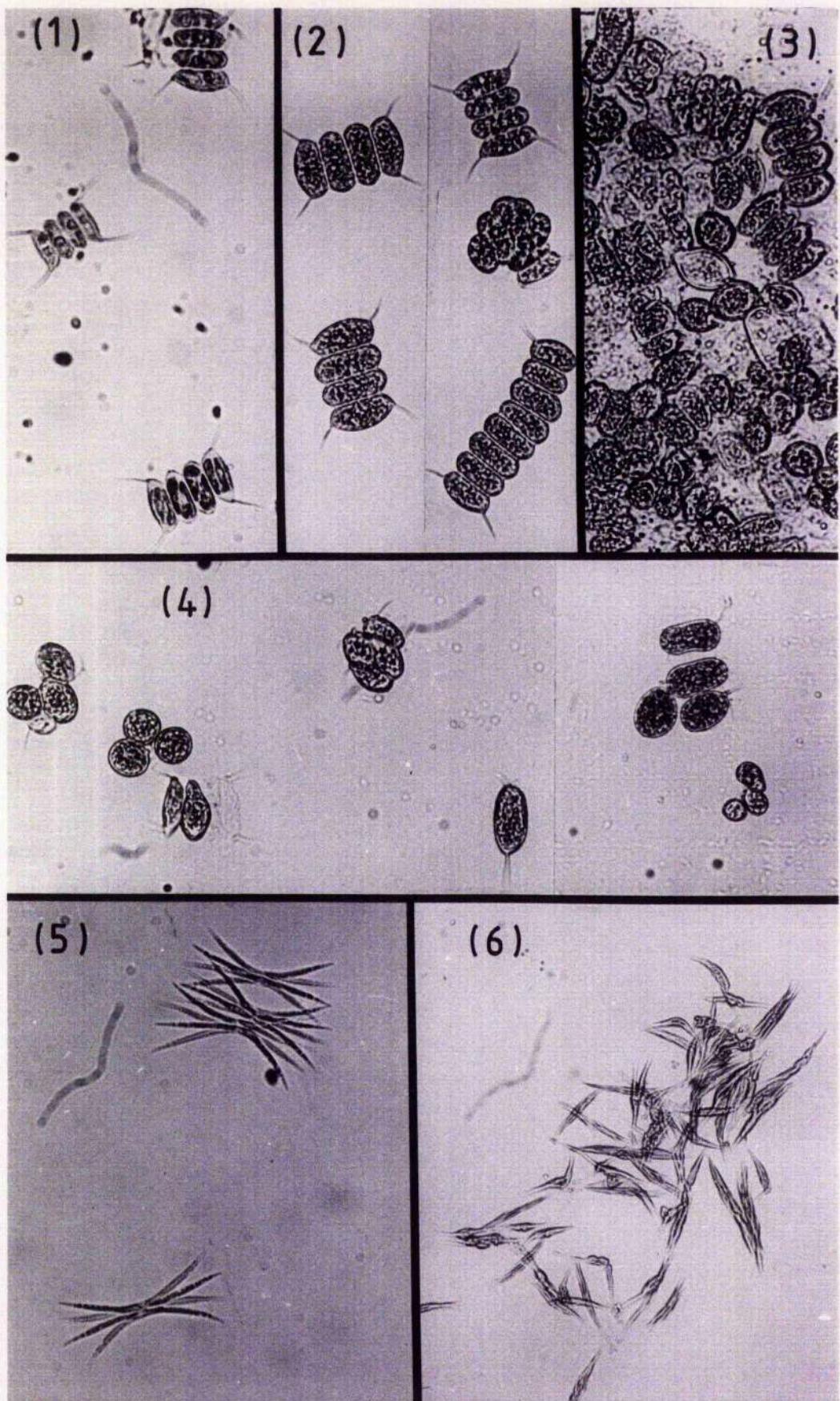


Plate (4.09)

as $\log n_t$ against t , as this gives close values to a straight line (Fogg, 1953), where n_t is the cell number per one ml of medium at any time (t). The exponential growth in the first few days is given by the expression

$$n_t = n_0 e^{rt}$$

where n_0 is the number of cells per unit volume of medium at zero time, e the base of natural logarithms, and r the relative growth constant. This expression may be converted to

$$\log n_t = \log n_0 + rt$$

Figure (4.29) displays the growth of the alga *Ankistrodesmus falcatus* in the river cultures measured as cell numbers per 1 ml volume and expressed as $\log n_t$ against t during December 1980. The peak $\log n_t$ values for all the stations ranged between 5.7-5.8 reached after 12-14 days. The maximum $\log n_t$ value of 5.7 was recorded for the control in 15 days. This indicates that the control needed a longer period for a relatively less growth compared with most of the stations, particularly Station 4, for which the maximum value of 5.8 was observed in 11 days. This increased population declined rapidly without showing any stationary phase. This alga, however, proved unsatisfactory for these experiments due to an accumulation of mucilage between the cells (Plate 4.09, Fig. 6). This caused the formation of large cell "clumps" which were impossible to separate and would have given erroneous estimates of population sizes. Hence *Scenedesmus quadricauda* was used for all further experimental work. This alga also proved to be more sensitive than the former one and counting the cell numbers was much

Figure (4.29) *Ankistrodesmus falcatus* cell number ml^{-1} in
the river cultures from the 10 stations of
the River Kelvin plotted as $\log n_t$ against t
during December 1980.

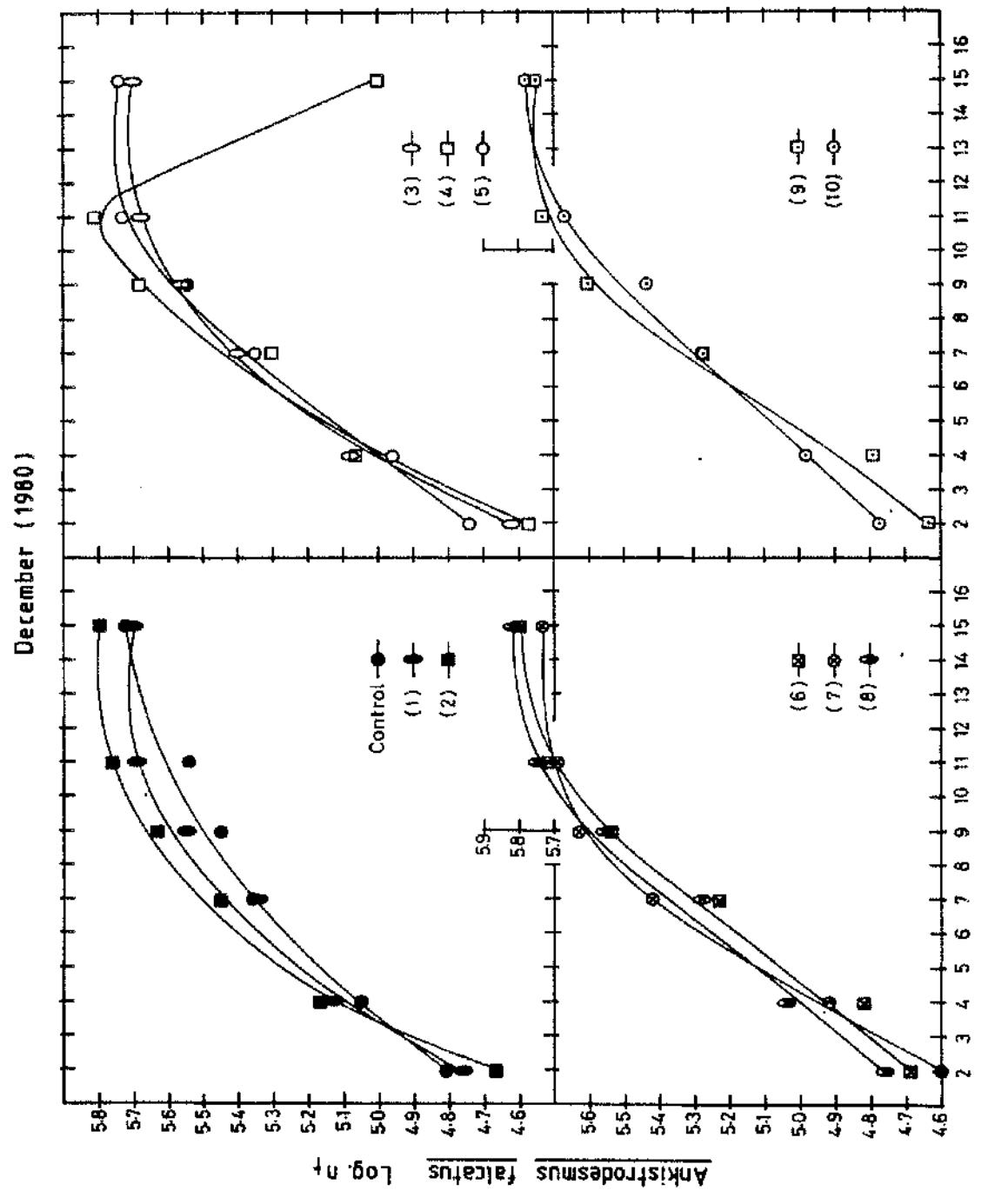


Figure 4.29

easier due to their existence either singly or in coenobia of 2, 4 or 8 cells.

The highest cell numbers obtained for the growth of *Scenedesmus quadricauda* was recorded in water from Station 2 during December 1980 (Fig. 4.30). This was recorded after 4 days of cultivation and the increase in the cell numbers was 5 times greater than the initial inoculum (Table 4.17) whilst the growth in the control observed over the same period was a twofold increase. The rest of the stations showed their growth peaks after 7 days (excluding Stations 9 and 10 where the peak was delayed for a further 2 days). During this period similar growth to the control found in samples from Stations 3, 5 and 7 with about fourfold increases in cell numbers. The growth in water from Station 4 was higher (5 times that of the initial inoculum compared with growth in the control, whilst samples from Stations 1, 6 and 8, the population change was about three times higher). The growth in the control after 9 days was a ninefold increase whilst water from Stations 9 and 10 showed their maximum population changes of 3 and 5 times the initial inoculum respectively. Less growth was recorded in the river cultures during January 1981 (Fig. 4.31) and all samples needed relatively longer times to show their peak values. Stations 1, 2, 3, 5 and 9 water samples showed *Scenedesmus* maximum growth in 8 days, and during this period the growth in the control was by 5 times higher than the initial cell numbers. A higher population than that of the control was recorded for Station 5 (6.5 times the initial inoculum) and less at the others (1, 2, 3 and 9). Water samples from Stations 4, 6, 7, 8 and 10 showed their growth peaks after 11 days when growth in the control was about 9 times higher than the initial cell numbers. The peak value in water from Station 4 was

Figure (4.30) *Scenedesmus quadricauda* cell number ml^{-1} in
the river cultures from the 10 stations of
the River Kelvin plotted as $\log n_t$ against t
during December 1980.

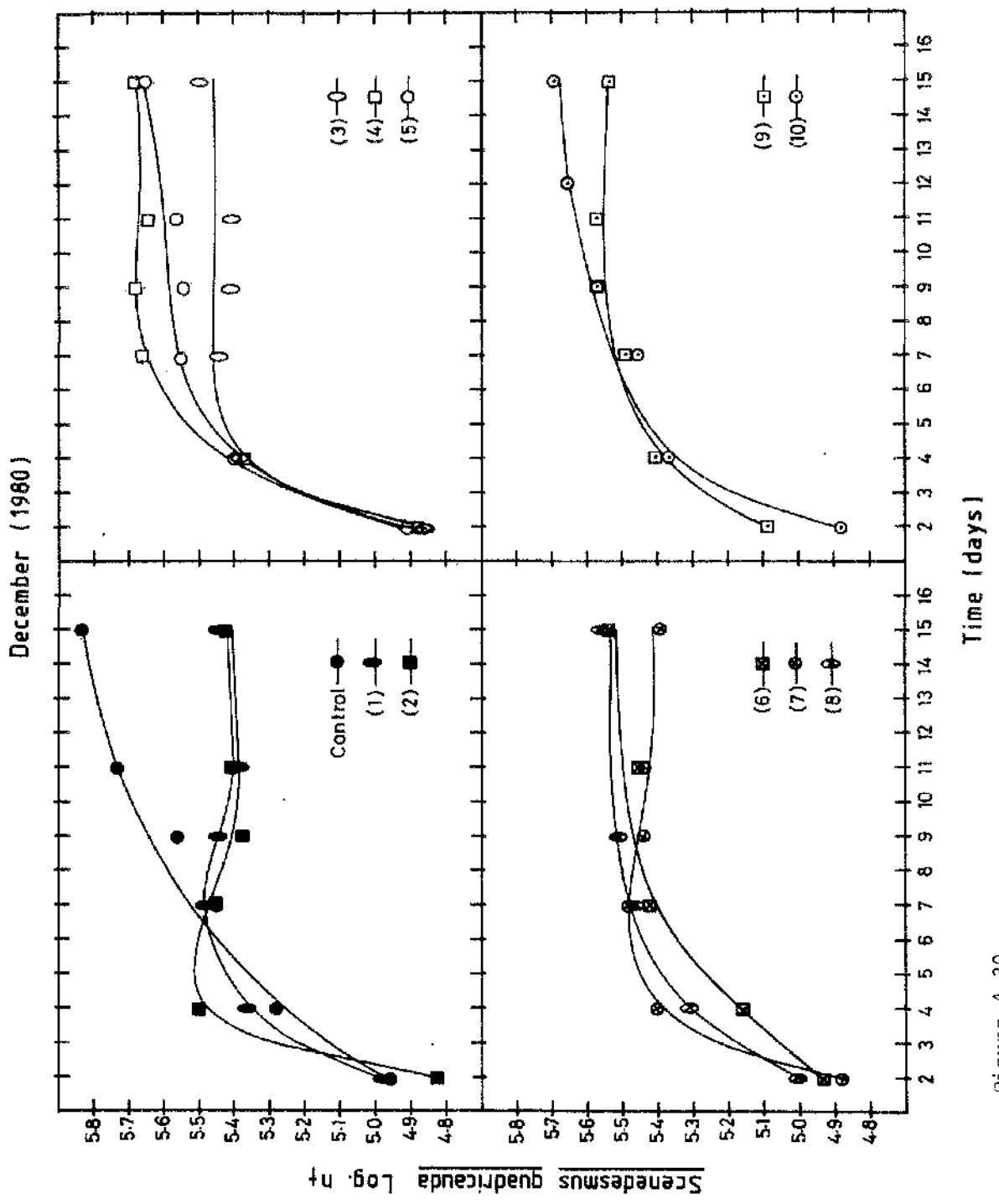


Figure 4.30

Table (4.17) The cell numbers (cells ml^{-1}) in the two-day-old cultures of *Scenedesmus quadrivalvis* (I) and at the growth maxima (M) compared with the control during the period December 1980-April 1981.
 (Figures are $\times 10^4$).

Time (Months)	Control	S T A T I O N S									
		1	2	3	4	5	6	7	8	9	
	Cell no.	Days no.	Cell no.	Days no.	Cell no.	Days no.	Cell no.	Days no.	Cell no.	Days no.	
Dec	I	9.20	9.64	6.67	7.15	10.1	8.02	8.59	7.66	10.1	12.7
Jan	M	35.9	7	30.6	7	32.0	4	25.4	7	46.2	7
Feb	I	7.15	11.6	14.6	8.99	6.67	7.83	13.0	14.6	18.4	20.2
Mar	M	35.9	8	26.6	8	34.3	8	36.7	8	80.4	11
Apr	I	11.1	14.3	14.6	15.7	14.0	17.2	11.6	9.20	17.2	17.2
May	M	42.2	8	36.7	11	56.9	11	46.2	8	106	11
Jun	I	4.51	1.60	1.43	2.01	1.33	1.53	1.43	1.53	1.36	12.7
Jul	M	25.4	10	32.0	10	44.2	10	38.5	10	54.3	10
Aug	I	2.01	2.26	2.54	2.54	2.26	2.26	2.01	2.26	2.54	1.71
Sept	M	23.2	10	28.5	10	50.7	10	40.3	10	128	13
Oct	I										2.06
Nov	M										11
Dec	I										50.7
Jan	M										8
Feb	I										
Mar	M										
Apr	I										
May	M										
Jun	I										
Jul	M										
Aug	I										
Sept	M										
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Figure (4.31) *Scenedesmus quadricauda* cell numbers ml^{-1} in
the River cultures from the 10 stations of
the River Kelvin plotted as $\log n_t$ against t
during January 1981.

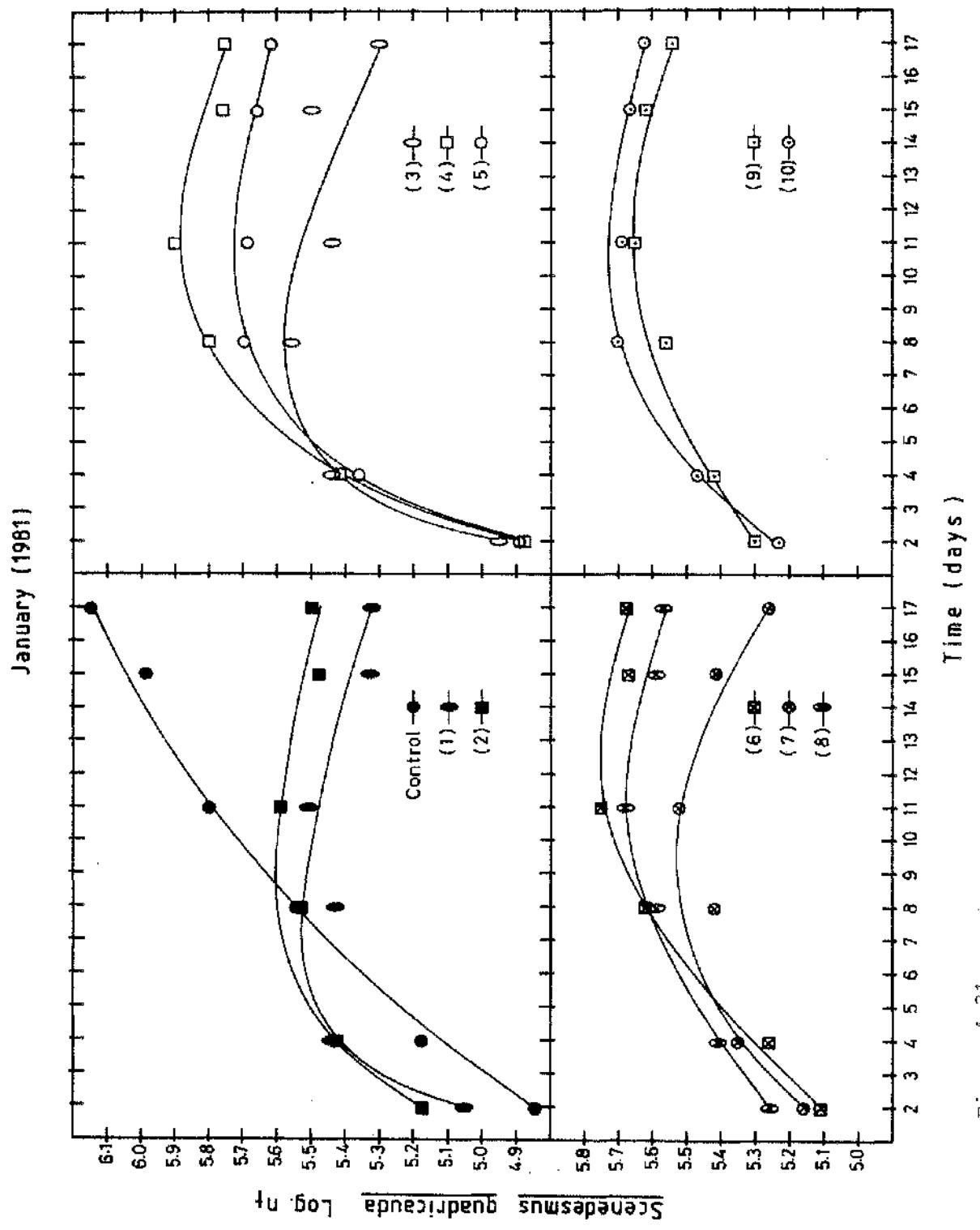


Figure 4.31

of $\log n_t$ value 5.9, and that is an increase in cell number of about 12X but in the sample from the other station the growth was less than the control. The growth in water from Station 4 (Luggie Water) was found to be more than the other stations and the control also during February (Fig. 4.32). The peak for this water sample was about 8 times more than the initial cell numbers, recorded after 11 days of cultivation. During the same period the increase in the control was 5 times greater than the initial cell number and in water from Stations 1 and 2 the growth peak was less than the control being of three and fourfold increases respectively. The maximum growth at the other stations was observed after 8 days during this period when the increase in cell numbers was by 4 times in the control. This was similar to the growth observed in samples from Stations 7 and 8, higher than that from Station 3 (a three-fold increase) and lower than the growth peaks in water from Stations 5, 6, 9 and 10, where cell number increases of 5-6 times were obtained. During March (Fig. 4.33) the peak in almost all the river samples was recorded after 10 days and the growth in the control reached a $\log n_t$ value of 5.4, which is about a 6 times increase. During this month the growth obtained for all the stations samples was higher with the maximum again at Station 4 (population 41X initial inoculum) and the lowest for Stations 1, 3, 7, 9 and 10 (about a twentyfold increase) and for the other station the increase in *Scenedesmus quadricauda* cell numbers was by 25-35 times. The growth in water from Station 4 showed an increase of about 57 times that of the initial cell numbers during April (Fig. 4.34) in 13 days. During this period the growth in the control was less than half of that in Station 4. The same period was required for the *Scenedesmus quadricauda* in the river cultures of the Stations 5, 6, 8, 9

Figure (4.32) *Scenedesmus quadricauda* cell number ml^{-1} in
the river cultures from the 10 stations of
the River Kelvin plotted as $\log n_t$ against t
during February 1981.

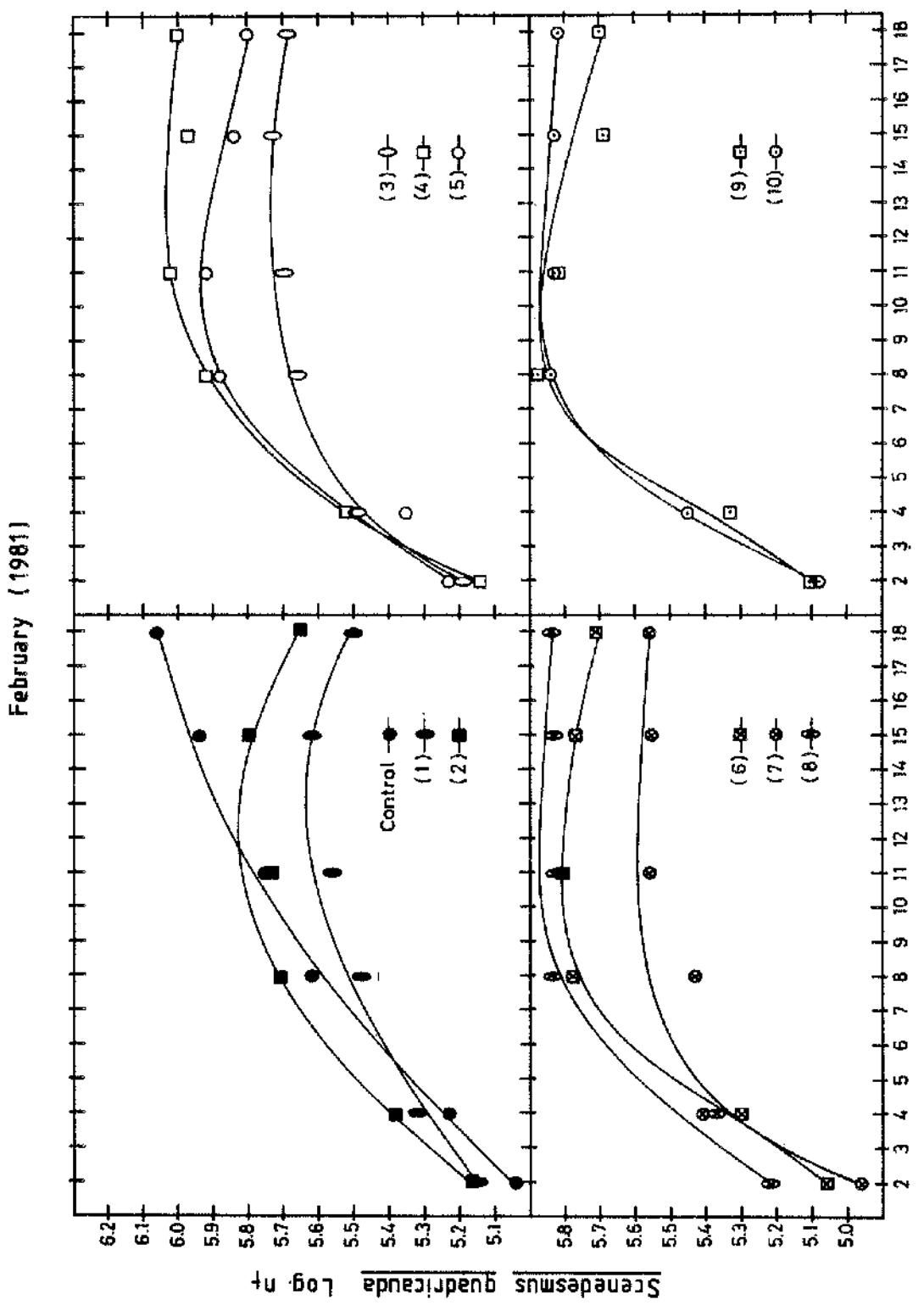


Figure 4.32

Figure (4.33) *Scenedesmus quadricauda* cell number ml^{-1} in
the river cultures from the 10 stations of
the River Kelvin plotted as $\log n_t$ against t
during March 1981.

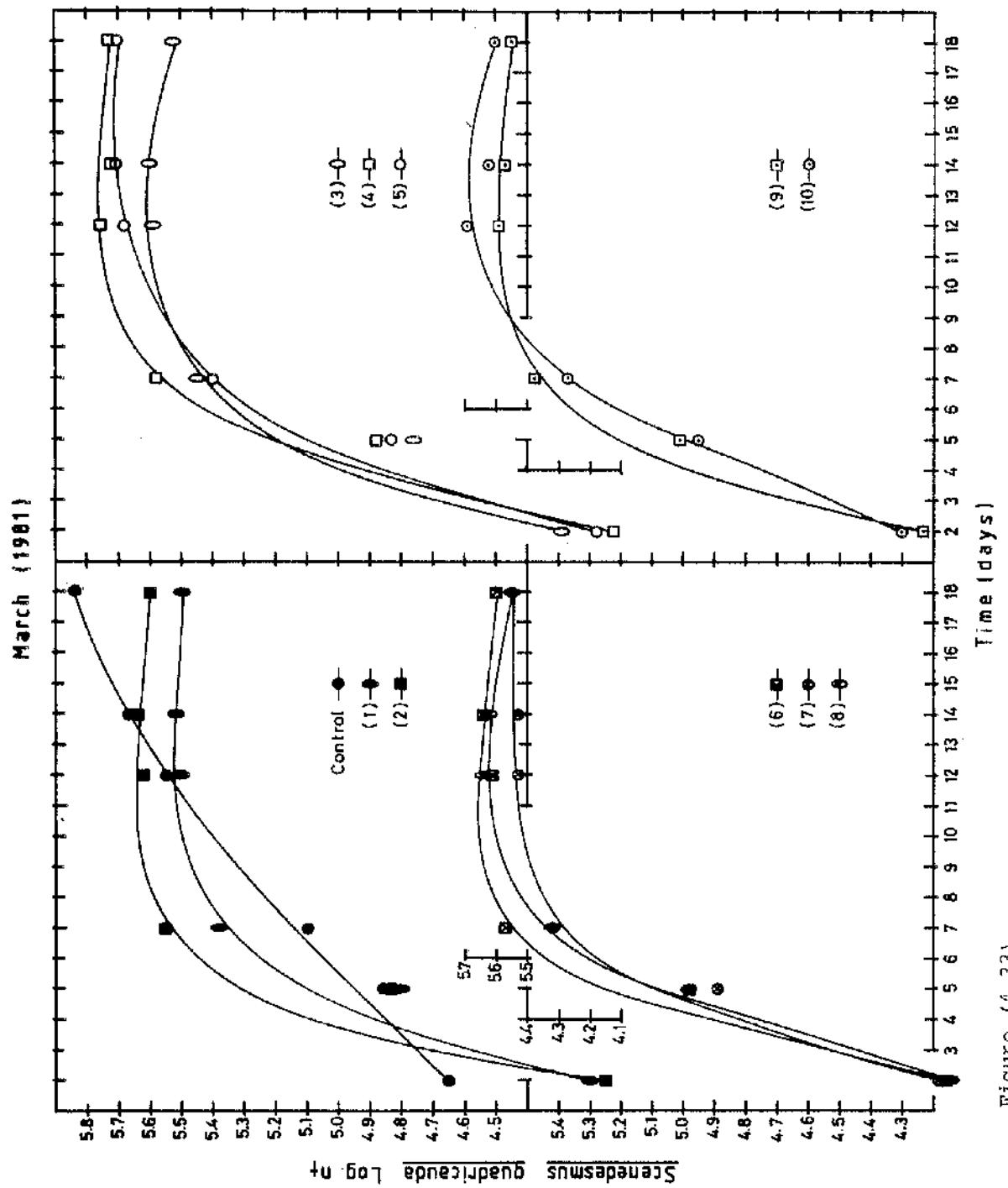


Figure (4.33)

Figure (4.34) *Scenedesmus quadricauda* cell number ml^{-1} in
the river cultures from the 10 stations of
the River Kelvin plotted as $\log n_t$ against t
during April 1981.

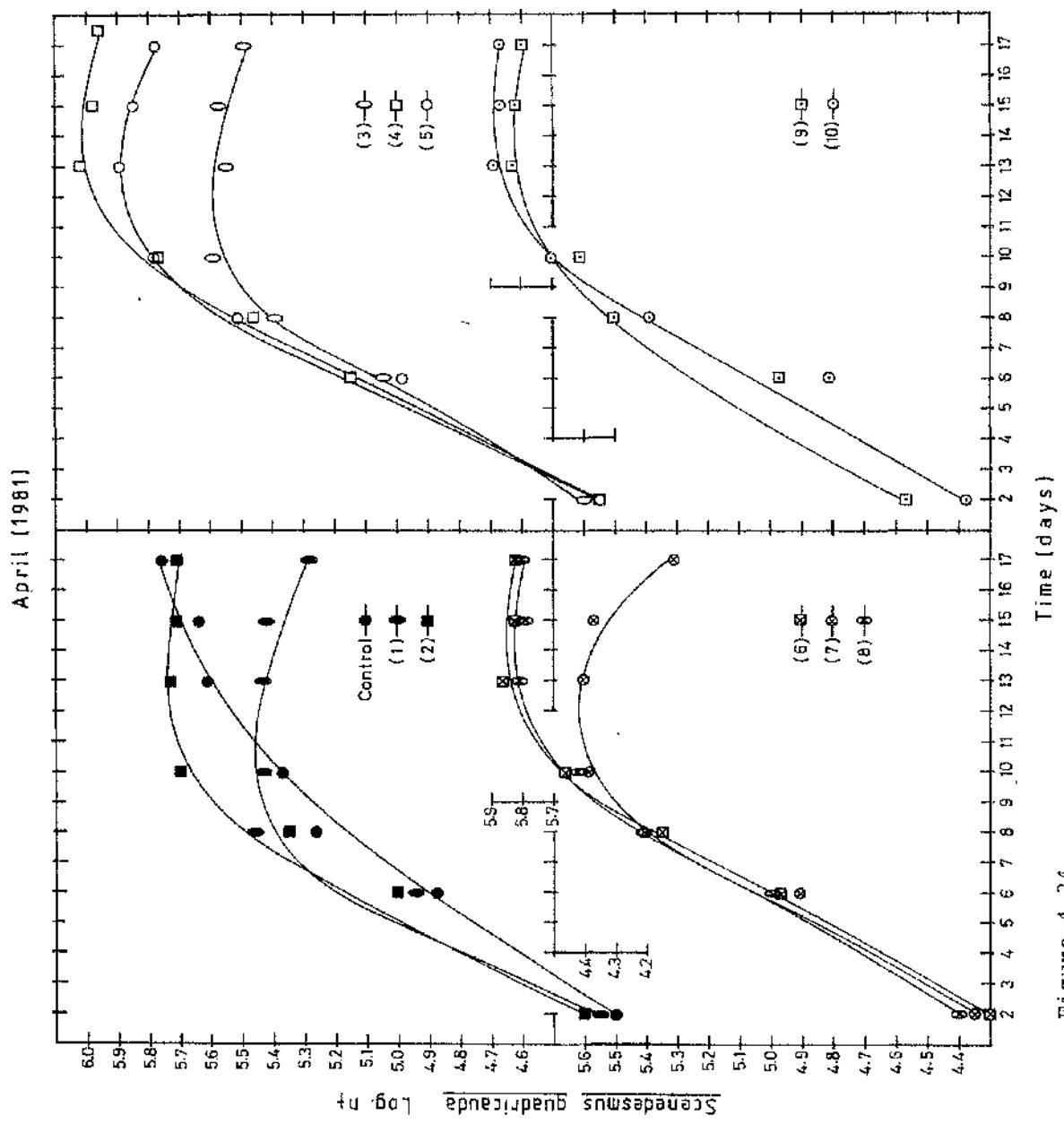


Figure 4.34

and 10 to show their maximum growth being greater than the control at Stations 5, 6 and 10 (35X increases in cell numbers) and 26X at Station 8 whilst the other two stations show a growth almost similar to the control. Samples from Stations 1, 2, 3 and 7 needed 10 days to reach their growth peak and the control showed an increase in the cell numbers to about 12 times during the same period. This was similar to the growth in Station 1 water only and for the others the growth recorded was higher.

The above information indicates more rapid increase in population sizes for the *Scenedesmus quadricauda* in the river water samples usually compared with the control cultures as the spring months approached, especially during April, i.e., when the water enrichment by the nutrients became relatively higher than in the winter months. During this month the spring outburst of the diatoms also starts in the river. The highest growth was always at Station 4 which was much higher than the growth in the control. The difference found between the cultures growing in the river samples and the control was in the growth pattern. The former cultures usually showed their peaks within the first few days of culturing and showing no more growth after that, whilst the latter cultures were growing gradually and continuously.

4.3.2 Cell appearance in river water

Whilst increases in cell numbers were observed, the *Scenedesmus quadricauda* cells did not appear to be very healthy in the river water samples. They appeared in a normal condition in the control but they were found in other shapes in the river cultures. Initially after 2-4 days cultivation in river water samples, the cells of the alga show signs of loss of their pyrenoids (Plate 4.09, Fig. 2) and under further

growth the coenobia of 4 and 8 cells often dissociated into single cells with spherical, oval and cylindrical shapes whilst still possessing their spines (Plate 4.09, Figs. 3 and 4). These showed none of the appearance of the genus *Scenedesmus*. This phenomena increased with increasing age of the cultures. The numbers of these abnormal cells were counted in one week old cultures as the percentage of the total cell numbers in the cultures and compared with the control during the same period of measuring the growth in the cultures (Fig. 4.35).

The results show that the abnormal cell phenomena was more severe during winter months when the highest percentages for all the stations was recorded during December. Slightly less abnormal cells were observed during January and they dropped in numbers from February until April. The control had always the minimum amount of these cells, showing almost the same quantities (about 1-2%) during the whole period.

It was decided to stop counting the cell numbers and the abnormal cell percentages since the cultures were repeating the same growth pattern every time compared with the control and also due to the lack of space in the culture cabinets and time for the other experiments. More attention was paid to other parameters in this study as cell counting alone was considered insufficient to assay water qualities.

4.3.3 Measurements of chlorophyll a contents

The colour of the river cultures was green during the first week of culturing, but turned to a yellowish-green after that and finally to a yellowish colour after the second week of their cultivation. Thus chlorophyll a was measured in the *Scenedesmus quadricauda* cultures in one week old and two week old cultures using hot methanol for extraction

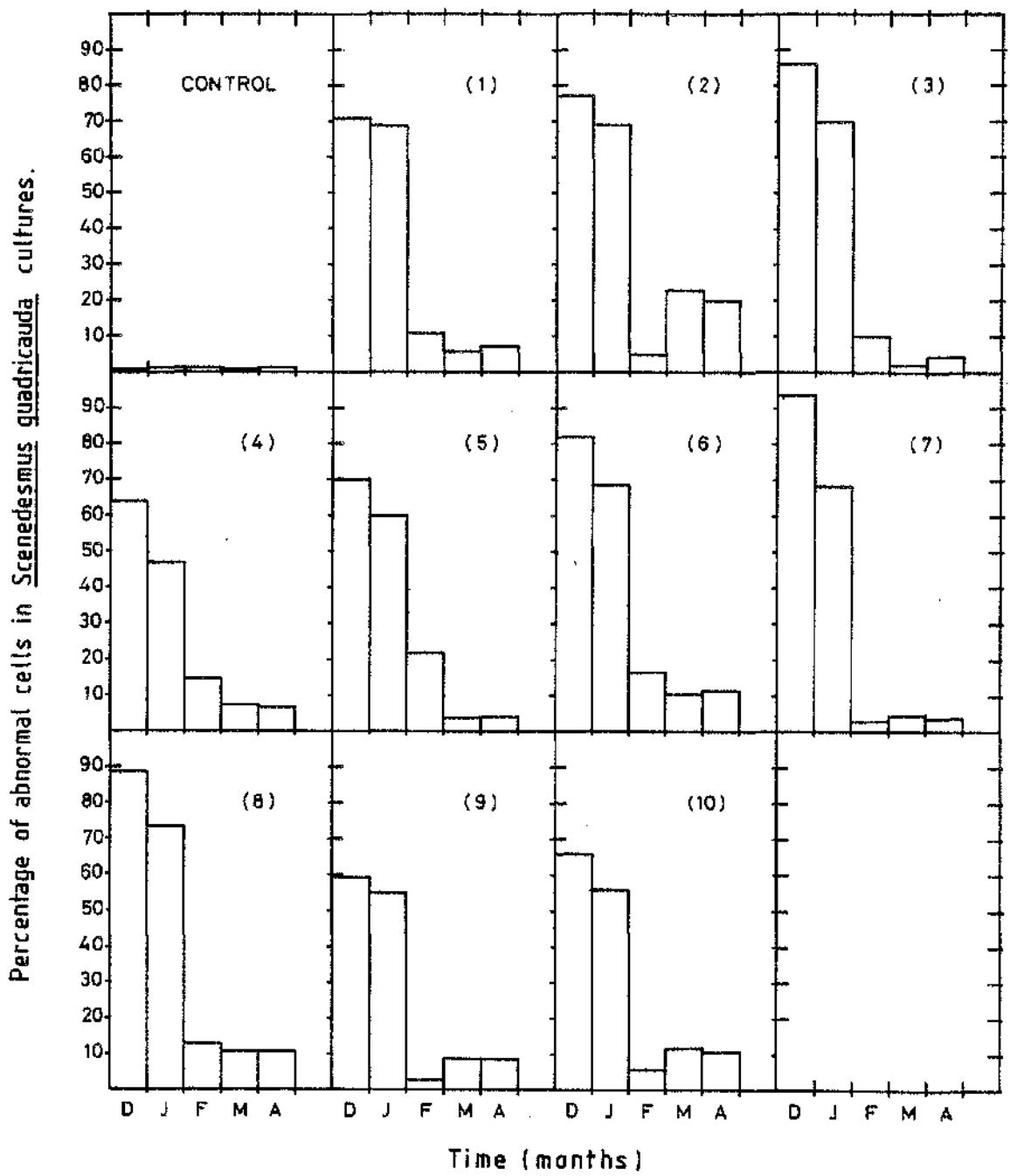


Figure (4.35) The percentage of the abnormal cells formed in the coenobia of the *Scenedesmus quadricauda* exposed to the river samples after one week of cultivation during the period December 1980 - April 1981.

(see section 3.8.2 as this method gives a better extraction than acetone) during the period March 1981–February 1982 and compared with the control (Fig. 4.36).

The quantities in the control samples gave higher values during the second week compared with the first but the water samples from the stations differed. Stations 1, 2, 3 and 7 contrasted with the control during the whole period, i.e. lower chlorophyll a values were recorded during the second week of cultivation. The rest of the stations varied. During the spring and summer period they coincided with the control whilst during the autumn-winter period they differed from it. At these stations the change in these two phases was mostly during September when a drop in the chlorophyll a content for almost all the stations, excluding Station 1, was recorded. The maximum contents were observed in water from Station 4 during the spring-summer period and after 2 weeks of growth. Again the death of cells during the second week and low chlorophyll a content generally for all the stations (excluding Station 1 where the quantities were always low) was low during the autumn-winter period.

4.3.4 Phaeopigment measurements

Due to the formation of the yellowish colour in the river cultures, which indicated degradation of the chlorophyll pigment, both chlorophyll a and phaeophytin were measured in the two week old cultures using acetone for extraction, as this method is the most reliable for estimating the phaeopigments. The results are shown in Fig. (4.37), which displays the amounts of chlorophyll a in mg m^{-3} and the phaeopigments as the percentage of the former. These results varied slightly from the

Figure (4.36) Chlorophyll a in mg ml^{-1} measured in the cultures of *Scenedesmus quadricauda* after one week and 2 weeks cultivation during the period March 1981 - February 1982.

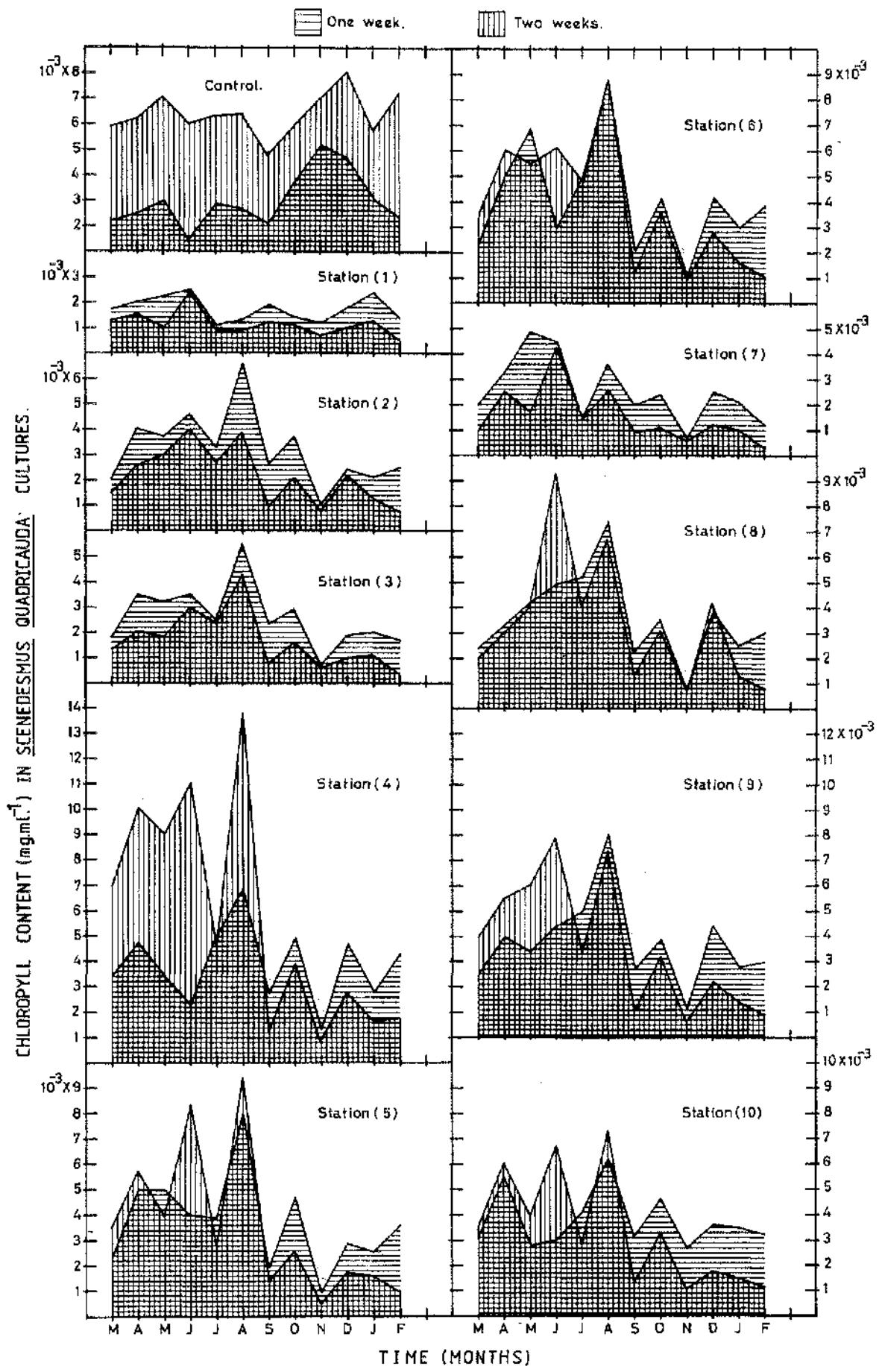


Figure 4.36

Figure (4.37) The chlorophyll a and the phaeopigments in mg m^{-3}
measured in the *Scenedesmus quadricauda* cultures
using 90% acetone for extraction after 2 weeks
cultivation during the period June - February 1981.
(The phaeopigments expressed as percentages of
chlorophyll a).

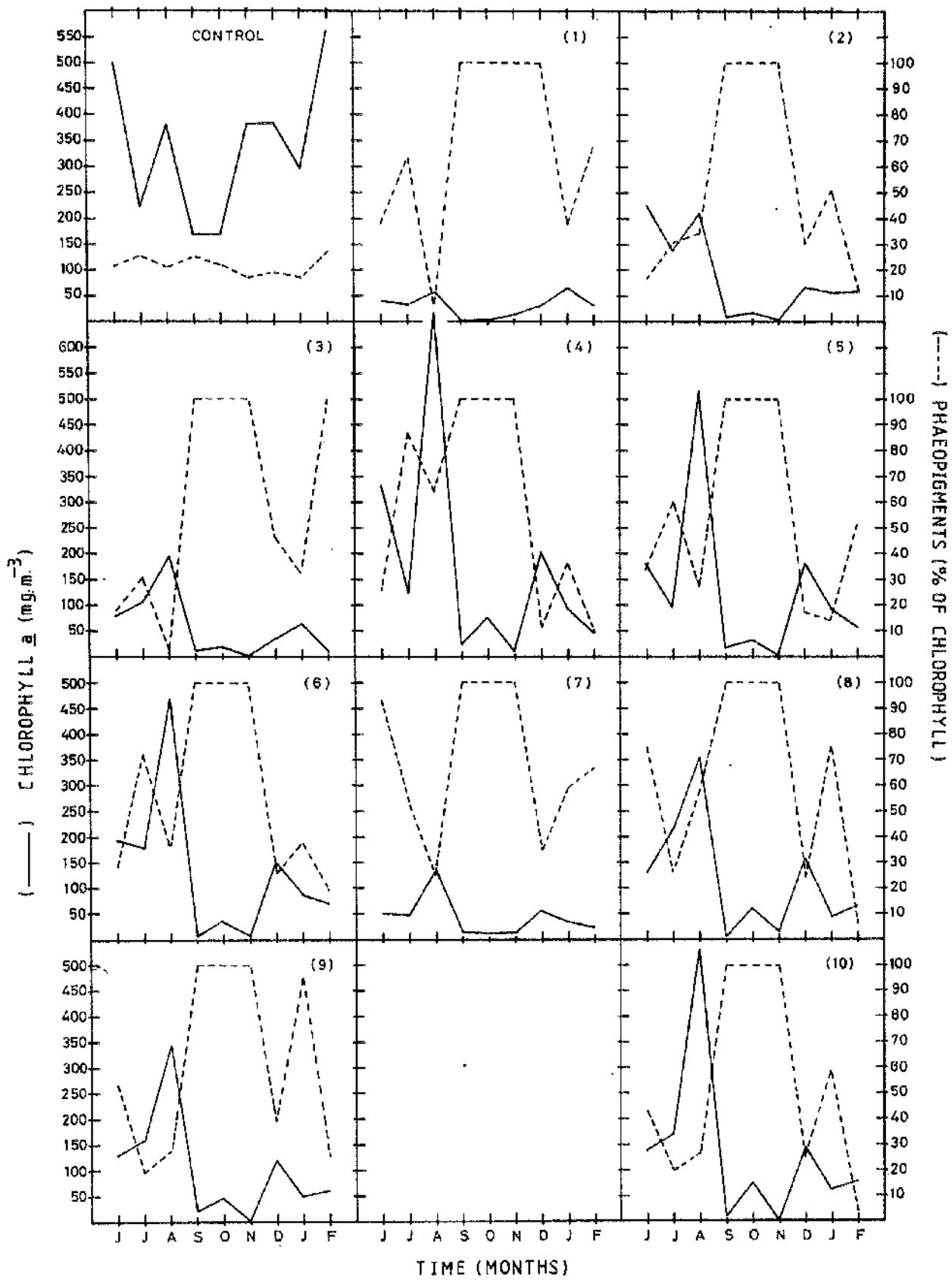


Figure 4.37

previous one due to the different procedures used for extraction but, in general, the two methods showed similarity. The control had always a fairly constant quantity for the phaeopigments which was lower than the chlorophyll a measurements during the whole period. The stations varied but they all showed a peak for chlorophyll a during August, being highest at Station 4 (similar to the results with the previous method of extraction). The other similarities observed were the severe drop in the chlorophyll a amounts during September for all the stations when the phaeopigments formed 100% of the extracted pigments for all the stations. This continued until November when there was a further drop obtained in chlorophyll a quantities. A small peak for chlorophyll a was obtained, using both methods of extraction and at almost all the stations, excluding Station 1, during December, when relatively high nutrient loads were recorded in the river due to the relatively low flow rate and ice formation.

4.3.5 Rates of photosynthesis

The photosynthesis rates in the cultures of *Scenedesmus quadricauda* were measured using the oxygen electrode, as described in sec. 3.8.3. This was considered an additional means of detecting how healthy the cells were in the cultures. The estimations were carried out on one week old cultures and the results are shown in Fig. (4.38) which displays the rates of photosynthesis of *Scenedesmus quadricauda* in river water samples expressed as the percentage of the rates recorded for the control (in Bold's basal medium) during the period March 1981–February 1982. The results showed that almost all the stations had nearly the same seasonal pattern. A noticeable increase in the oxygen production

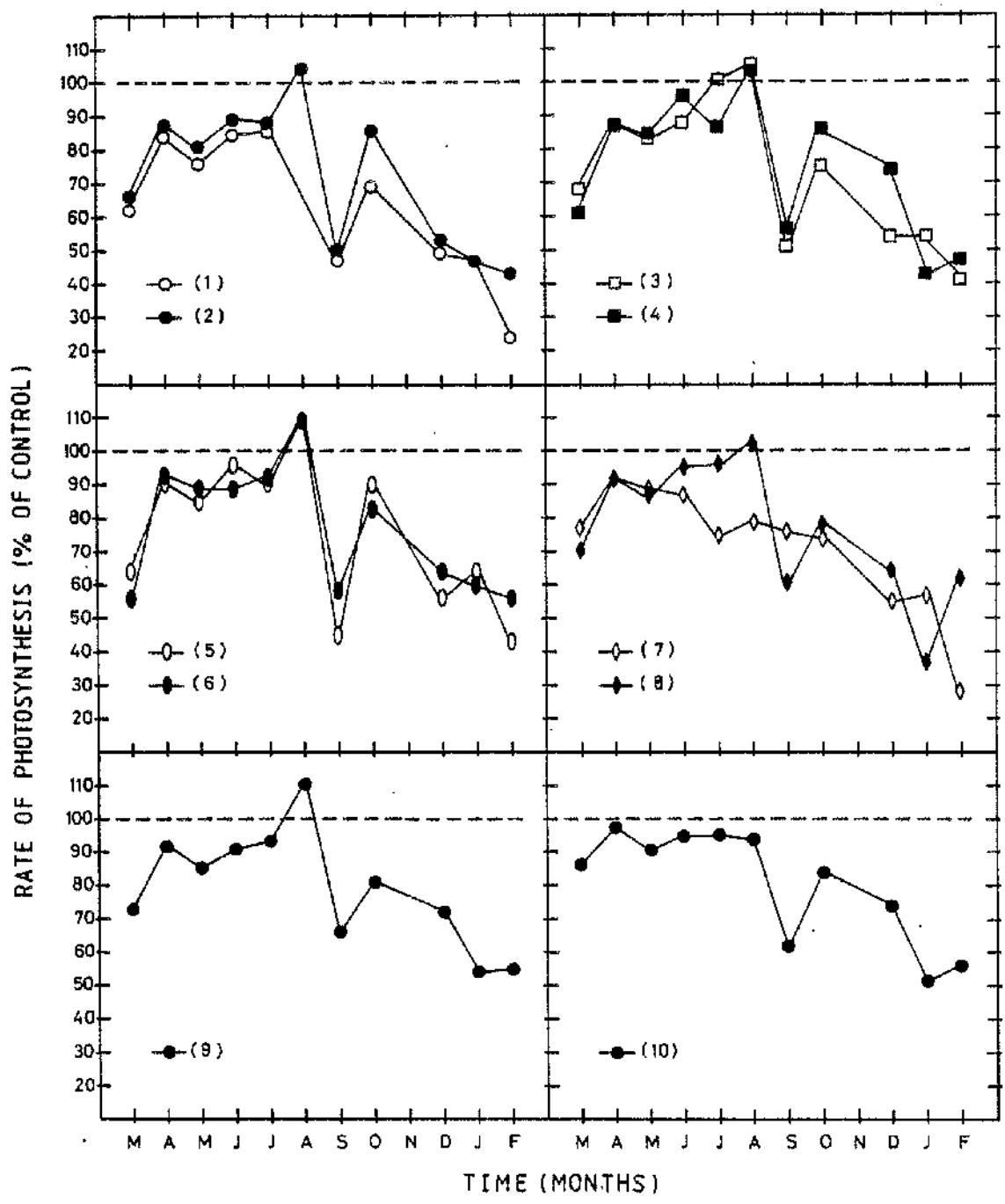


Figure (4.38) The photosynthesis rate expressed as the percentage of the control in the cultures of *Scenedesmus quadricauda* after one week cultivation during the period March 1981 - February 1982.

by the cells in all the cultures was observed during April. There was only one period (August) when the photosynthesis in the river cultures for almost all the stations (excluding Stations 1, 7 and 10) were higher than photosynthetic activities in the control. This coincided with the distinct peak in chlorophyll a and the fall in phaeopigments at all the stations during the same period, as reported in the previous two stations (4.3.3 and 4.3.4). There was a change in the response of *Scenedesmus* in September, shown here as a drop in the oxygen production in the cultures, whilst a dramatic fall in the chlorophyll a values and an increase in the phaeophytin content observed at almost all the stations during the same period. The cultures showed an improvement during October but they then showed a continuous decline throughout the winter period at all the stations.

CHAPTER FIVE

General Discussion

The River Kelvin with its average flow of 152 mgd carries a large amount of sewage effluent and a moderate amount of industrial effluent. It also carries large percentages of undissolved solids and wastes connected with farming. The local authority sewage treatment works form the major polluting load discharges into the Kelvin and its tributaries. There is very little dilution in the river although most of these sewage works have good quality effluents. In some cases (e.g. Bishopbriggs Burn) almost all the flow in the river consists of sewage effluent.

Upstream of Kilsyth, the water quality is satisfactory until its confluence with the seriously polluted Dock Water. This water was very badly affected by the poor quality effluent from Kilsyth sewage treatment works during summer 1982 when the work was vandalised. This occurred for only a few days; the repairs were carried out and the river returned to its original condition (C.R.P.B. Annual Report 1982). Dock Water adversely affects the River Kelvin downstream of the confluence. At Kirkintilloch, the good quality water from the Glazert Water leads to an improvement in the main river. Within a short distance, it is joined by the polluted Luggie Water which brings $0.05 \text{ m}^3 \text{ s}^{-1}$ and $0.1 \text{ m}^3 \text{ s}^{-1}$ of sewage effluent discharges respectively from Deerdykes and Auchengeich sewage works. During July 1982 a number of dead trout were found in the Luggie Water below Deerdykes sewage treatment works. This was related to the very low dissolved oxygen levels which brought about a combination of heavy weed growth, prolonged low flows and high temperature (C.R.P.B. Annual Report 1982). Downstream of Luggie Water the Kelvin

received approximately $0.16 \text{ m}^3 \text{ s}^{-1}$ of sewage effluent from Kirkintilloch Sewage Works, which is generally marginal quality as it is volumetrically overloaded.

The Kelvin is then joined by the polluted Bishopbriggs Burn, as mentioned above, then by Allander Water. This is a clean tributary of the Kelvin which brings only about $0.1 \text{ m}^3 \text{ s}^{-1}$ good quality sewage effluent into the main river. Downstream of the Allander there is a general improvement in the condition of the Kelvin and it is of "fairly good quality" * throughout Glasgow until it finally reaches the River Clyde.

Farming also causes pollution problems in the Kelvin, either by silage effluent, which is the water and sap squeezed from the crop during fermentation and storage, or slurry, the concentrated wastes of animals kept inside during winter time. These can cause vigorous bacterial growths which can lead to deoxygenation of the water.

There are very few direct discharges of industrial effluent into the river as most of the factories are located in or around the towns and their drainage is connected to the local authority sewage system for treatment at the sewage works. One of the main discharges from industrial premises emanates from the Gartcosh Steel Works to the Bothlin Burn (one of the Luggie Water's branches) where equipment has

* The chemical classification employed in Scotland uses the expression "fairly good quality" for waters with limiting criteria of D.O. saturation $> 40\%$, $\text{BOD} < 9 \text{ mg l}^{-1}$ and non-toxic to fish, based on National Water Council (1978). In England and Wales a "doubtful quality" designation is used instead, according to Lewin (1981).

been provided by the company to prevent oil pollution to the stream.

Pollution often occurs as a result of discharges of contaminated drainage from domestic refuse tips. When rainfall enters the refuse a very highly polluting leachate is produced which can be serious.

Due to the large volume of sewage effluent discharged daily into the River Kelvin, recreational use of the river is restricted. Nevertheless, there are several stretches of the river which support trout populations and people enjoy fishing at Glazert Water, Allander Water and River Kelvin upstream of Dock Water and also at Kirkintilloch. (The above unpublished information was provided by C.R.P.B.).

The data provided by the C.R.P.B. for the River Kelvin indicated that the stream is clean until about 5 km from the source and becomes moderately polluted for the rest of its length. The average dissolved oxygen for the whole river was 70-90%. BOD_5 averages never exceeded 5 mg L^{-1} and ammoniacal nitrogen average was < 2500 $\mu\text{g L}^{-1}$. Slightly higher BOD_5 values were presented by Mulla-Ali (1981) using two different methods for measuring BOD - the rapid microbial method and the modified Winkler's method. He recorded a mean value of 5.7 mg L^{-1} for 254 river water samples of low BOD_5 values, i.e. not the raw sewage. He observed parallel BOD values in both methods.

The normal parameters for measuring pollution in the present work, when applied to the Kelvin and its tributaries, were within a satisfactory range of levels where there were no sewage effluent discharges.

The changes in the quality of the river in the present work were assessed by measurements of BOD_5 , phosphate P, total nitrate + nitrite N,

ammonia and dissolved oxygen. The effect of pollution depends upon the types and quantities of the pollutants, and if there is no further pollution, the quality of a stream water improves and returns back to almost its natural quantities by the "self cleaning" process. This occurs when the pollutants pass downstream and the oxygen content increases as the river passes over stones and down waterfalls. With the improvement in the oxygen content of the water the amount of organic pollution decreases as the distance from the pollution source increases. Oxygen will also be produced by algae and plants in the stream. High concentration of ammonia also makes demands on the oxygen resources of the river and nitrification takes place, i.e. oxidising ammonia to nitrites and then to nitrates.

The water quality at the river's head, Station 1, was generally satisfactory and within the normal (clean) range, although the BOD_5 values were fairly high, reflecting the nature of the drainage area near the source. The range of the nutrient levels recorded were $PO_4 P$ 0.26-4.6 $\mu g l^{-1}$, $NO_3 N$ 12.0-105 $\mu g l^{-1}$ and NH_3 13-670 $\mu g l^{-1}$. At Station 2, levels of $PO_4 P$ and NH_3 increased by approximately 5 and 6 times respectively. This reflected the influence of the Dock Water bringing in its sewage treatment work effluent which also caused a slight increase in $NO_3 N$ levels (about one and a half times). At Station 3 the main change was a slight decrease in $PO_4 P$ quantities to about 4 times that of the river's head, the $NO_3 N$ staying almost the same but the concentrations of ammonia increased further to 7 times over the source. This increase in ammonia levels may be due to the effluents from Milton of Campsie and Lennoxtown. The latter caused major pollution problems

in the past (see Chapter 2). These changes were due to the entry of Glazert Water. Downstream of this station, the river was joined by the polluted Luggie Water (Station 4) with its high nutrient levels with the PO_4 P greater by sixfold over Station 1 and the NO_3 N nearly 3 times, but the concentrations of ammonia did not vary very much with the previous station. These high levels were due to the unsatisfactory sewage effluents from Greengairs and also Deerdykes and Auchengeich. They caused a rise in the nutrient loads in the main river at Station 5 where they destroyed the slight PO_4 P dilution caused by Glazert Water and the levels were again almost the same as at Station 2. The NO_3 N concentrations were twice as high as the head but the ammonia levels were higher than Luggie Water itself, being about 9 times greater than at Station 1. This might have resulted from the action of bacteria which break down inorganic compounds to ammonia or by demineralization, i.e. when the anaerobic bacteria take the oxygen from NO_3 and NO_2 and reduce it to nitrogen then to ammonia as a self purification process which takes place in river waters. Further increase in PO_4 P levels to 7 times that of the river's head and NO_3 N to almost the same quantities as Luggie Water were brought about in the Kelvin downstream due to the influence of the polluted Bishopbriggs Burn which was affected by unsatisfactory sewage effluents. The maximum ammonia concentrations here were 10 times greater than the source. The good quality of Allander Water, Station 7, with its PO_4 P and NO_3 N levels approximately the same as Station 3 and its relatively low ammonia quantities (3 times that of Station 1) did not have a great effect on the main river in reducing its nutrient load at Station 8. The only noticeable reduction

was in the amount of NO_3^- N when the same levels as Station 5 were observed (twice that at the Kelvin's head). PO_4^{2-} P and NH_3 decreased very slightly but they were still high (6 and 9 times, respectively, compared with Station 1). The river continued down towards Stations 9 and 10, receiving no further pollutants and with almost the same water quality as the previous station.

From the above data it is obvious that sewage effluent discharges are the major nutrient sources in the River Kelvin and its tributaries. Phosphorus acts as a builder in detergents, Keup (1968) related the high PO_4^{2-} P levels in streams to urbanization and waste water. Land drainage would probably not contribute much to the stream as the Kelvin passes through flat lands. Eck et al (1957) found that land gradient had a great effect on the erosion potential and losses of phosphorus and other nutrients into streams. Another likely source of the phosphorus in the Kelvin could be the fertilizers washed from surrounding farmlands (unpublished reports of C.R.P.B.). Casey (1969) observed that large increases in phosphorus levels in the River Frome occurred after the addition of fertilizers to nearby fields and, on another survey on the same river (Casey, 1975), it was found that 70% of the phosphate in the river could be attributed to sewage effluents. Nitrate could also be contributed by land drainage. Cook and Williams (1970, 1973) considered that drainage from well-farmed arable land in England will contain on average 10 mg l^{-1} NO_3^- N. Light land will lose nitrate to give drainage water with larger and more constant nitrate concentrations during the year, by contrast to heavy land. Large concentrations of nitrate may be found in drainage from heavy soils in spring. They

also stated that the nitrate loss from productive land cannot be prevented because more nitrogen will be mineralized from the soil by microbial action than the crops can take up at some times in the year. Drainage would contain $5 \text{ mg l}^{-1} \text{ NO}_3^- \text{ N}$ even where fertilizers were not used on arable land. Casey (1975) related the increase in nitrate in the River Frome to the increase in fertilizers usage and he found that the contribution of effluent nitrate to the total throughput was low and stable. Although the main source for ammonia is sewage effluents, this could be oxidized to nitrite and nitrate or the latter two might be reduced to nitrogen and then ammonia, depending upon the quantities of oxygen in the river. Lester (1975) attributed the presence of ammonia in the River Trent to sewage and some industrial wastes and he also added that the higher the degree of purification, the lower the ammonia concentration of the effluent due to its oxidization to nitrate.

Comparing the water quality in the River Kelvin with other rivers covered by C.R.P.B.'s routine pollution survey, their data for ammoniacal nitrogen gave a range of $7\text{-}6600 \mu\text{g l}^{-1}$ for the Kelvin. According to the C.R.P.B.'s annual report (1981) the rivers varied in their ammoniacal N content. In some of them there were lower ammonia N levels than was observed in the Kelvin. These were (i) the River Clyde* (a river recovered from severe pollution) which had a range of $10\text{-}2400 \mu\text{g l}^{-1}$, (ii) South Calder Water $10\text{-}4700 \mu\text{g l}^{-1}$ and (iii) Black Cart Water $5\text{-}2100 \mu\text{g l}^{-1}$. North Calder Water had almost the same levels of

* The River Clyde is the main river flowing in that area with the Kelvin, South Calder Water, North Calder Water, White Cart Water and Black Cart Water forming the main tributaries.

ammoniacal N as the Kelvin ($10\text{--}6500 \mu\text{g l}^{-1}$) whilst White Cart Water had contents of about 3 times that of the Kelvin ($10\text{--}16400 \mu\text{g l}^{-1}$). No phosphate P or nitrate N measurements were published by the C.R.P.B.

The River Falloch and Endrick Water flow into Loch Lomond (located north west of the City of Glasgow). The River Falloch is the largest single inflow into the Loch at its northern point and Endrick Water has the largest drainage area of all the inflows to the loch, entering at its south east corner. The data for these were taken from near their inflow into the loch (Maulood, 1974). Low nutrient loads were recorded for the River Falloch where PO_4^{P} never exceeded $1.3 \mu\text{g at l}^{-1}$ and NO_3^{N} was at the maximum of $24.6 \mu\text{g at l}^{-1}$. No measurements for ammonia are available. Endrick Water flows through fertile arable land and similar to the Kelvin, there were increases in the phosphate levels, accompanied by high nitrate content (at the maximum of $4.0 \mu\text{g at l}^{-1}$ and $75.8 \mu\text{g at l}^{-1}$ respectively). The relatively high phosphate content in the main loch was related to this inflow. These levels were still lower than the levels observed in the Kelvin.

Considering other rivers in the United Kingdom, the River Trent in England is a highly polluted river, due to sewage and industrial effluents (Lester, 1975). The range of the ammoniacal N recorded was $500\text{--}16500 \mu\text{g l}^{-1}$, which is about three times that of the Kelvin. No data for phosphate P or nitrate N were available. The River Frome, also in England (Casey, 1975), had average PO_4^{P} levels of $4.2 \mu\text{g at l}^{-1}$ and $\text{NO}_3^{\text{N}} 0.22 \mu\text{g at l}^{-1}$. These levels are considered as clean, compared with the Kelvin.

The River Kelvin is less eutrophic compared with the River Clyde and its tributaries, excluding the upper reaches of the Clyde and Black Cart Water. The biotic indices for these streams, as represented by C.R.P.B. annual report for 1981, indicate that the North and South Calder Waters, with the lower reaches of the Clyde, are between polluted to heavily polluted streams. White Cart Water had similar indices to the Kelvin, whilst Black Cart Water was considered as a clean river. Rivers Falloch and Endrick are to be considered as oligotrophic and mesotrophic, respectively, when compared with the Kelvin. The River Frome, which was similar to Endrick Water in its phosphate P levels, but with very low NO₃ N content, can also be considered as mesotrophic. The River Trent is a highly polluted river. Due to its very high ammoniacal N levels it would appear to be hypereutrophic, and is also being thermally polluted at certain points as its water is used as a cooling water by industry and the Central Electricity Generating Board.

Whilst the eutrophic condition of the Kelvin and its tributaries was shown by the high levels of phosphate P, nitrate N and ammonia, the river did not give signs of deoxygenation. Dissolved oxygen is a good parameter for indicating the pollution effects if a river contains no toxic constituents. No evidence of the heavy metal pollution (Zn, Cd and Pb) was obtained for the Kelvin. The overall range of the dissolved oxygen was 6-8 mg l⁻¹ (the saturation level for water at 14°C being average 10.3 mg l⁻¹). In fact, there was not a great difference in dissolved oxygen concentrations among the stations. Station 1 had the lowest levels for which a range of 2.30-8.95 µg l⁻¹ was recorded. This low value was probably due to its very slow water flow and also to its

minimal content of plants. The levels increased slightly at Station 2, despite its higher nutrient loads, by approximately 2 mg l^{-1} . Due to the influx of the clean Glazert Water the oxygen content increased very slightly ($< 1 \text{ mg l}^{-1}$ compared with the previous station). This level stayed almost constant for the rest of the river, excluding Stations 7 and 10. The former station, due to its good water quality, Allander Water, and also to its being faster flowing with Station 10 than the other stations, particularly Station 10 with its small water falls which help the aeration of the water.

The differences in dissolved oxygen content between the curves (Fig. 4.04) represents the seasonal variations. These were mainly due to the temperature and diffusion from the air and plant activity. The former factor affected the river during summer time, June-August, when relatively low oxygen content was recorded despite the growth of algae and plants. The diffusion from the air had a great effect at winter time, the period of maximum oxygen levels in the river. Butcher, Pentelow and Woodley (1930) explained that the temperature was an important factor in determining the minimum oxygen values and they related the lower oxygen values to the highest water temperatures and also to high nutrient loads. The bacteria growing in the river would then be using the oxygen for decomposition of the organic material as a process of self-purification during the growth periods. They also related the high oxygen levels during summer months to the abundance of the plants and filamentous algae (oxygen producers). The data for the dissolved oxygen concentration in the Kelvin in the present work were similar to the concentrations recorded by C.R.P.B.

BOD_5 is also another good parameter for measuring pollution.

Lester (1975) stated that "the oxygen profile is a delayed mirror image of the BOD profile; the higher the BOD, the lower the oxygen concentrations". The range for BOD_5 levels at Station 1 was $0.95\text{--}3.98 \text{ mg l}^{-1}$ which is relatively high for a river head. This was increased to approximately twice that value at Station 2 due to the influx of Dock Water but the levels decreased, by approximately 1 mg l^{-1} from Station 2, at Station 3 after the influence of the good quality Glazert Water. Stations 4, 5, 6, 8 and 9 had the same levels as Station 2 but Stations 7 and 10 showed another increase by 3 mg l^{-1} higher than Station 1. The BOD_5 profile (Fig. 4.05) showed that the overall average for the BOD_5 levels did not vary very much at the different stations, being the highest at Station 4, which differed by about only 1 mg l^{-1} average content higher than Station 1. The difference with the rest of the stations was by the average of only 0.5 mg l^{-1} . These results were comparable with the data provided by the C.R.P.B. (average values of $< 5 \text{ mg l}^{-1}$ always). The data presented by Mulla-Ali (1981) showed BOD_5 values of 5.67 mg l^{-1} as the average of 254 samples from the river. This value is slightly higher than the values recorded during this survey (about 1 mg l^{-1}) which, in turn, indicates the river improvement since the period of Mulla-Ali's survey. The high BOD_5 levels during the growth periods were due to the oxygen consumption by the microorganisms suspended in the water which resulted as a response to the high nutrient levels. The high BOD_5 values observed occasionally during winter were partly due to the relatively low currents (e.g. October 1980 and December 1981) which caused more nutrient load concentration. Mulla-Ali related these

increases to the release of larger volumes of the effluents into the river from the sewage works to avoid flooding. He also related the seasonal variations in BOD_5 levels to the differences in the river water temperature between summer and winter, the amount of rainfall and the quantity and quality of the sewage effluents discharged into the river which might affect the breakdown of non-soluble to soluble organic substances over a period of 5 days. He also discussed the numerous environmental factors which might interfere with the BOD_5 measurements (e.g. the abundant growth of filamentous algae and aquatic plants during the growth periods, the microbial contents caused by effluent discharges and the variation of the bacterial contents in the river).

Comparing the Kelvin, with its mean dissolved oxygen saturations of 70-90% (as recorded by C.R.P.B.) and average BOD_5 values of $< 5 \text{ mg l}^{-1}$, with other local rivers, the River Clyde had an average oxygen saturation similar to the Kelvin but its BOD_5 averages were higher by about 1 mg l^{-1} at its lower reaches. In South Calder Water the oxygen saturation was lower than the Kelvin, especially at its upper reaches, 40%, where the BOD_5 value was of the average 11 mg l^{-1} , which is more than twice that of the Kelvin. The averages of the BOD_5 levels reached about three times that of the Kelvin (15 mg l^{-1}) at the lower reaches of the North Calder Water, whilst its oxygen saturation was always greater than 80%. Black Cart Water had BOD_5 averages similar to the Kelvin while its oxygen saturation was slightly higher ($> 80\%$). White Cart Water had BOD_5 mean values again similar to the Kelvin, except at its very upper reaches, where average values of 11 mg l^{-1} were recorded. The oxygen saturation was 80-100% except at its very lower reaches where it was down to about 60%.

In the polluted River Trent (Lester, 1975) the average oxygen contents recorded were 2-9.5 mg l⁻¹ which was less than the Kelvin by more than 1 mg l⁻¹ (the average for the Kelvin as a whole was about 7 mg l⁻¹).

The seasonal variation of dissolved silica (the very important element for diatoms) was different from that of phosphorus and nitrogen. The increase in these two elements in the Kelvin was not accompanied by an increase in the dissolved silica levels as its production by man was nil. Farming activities and the progressive increase of the effluents did not affect the silicate silicon values and its main source was from land drainage and the contribution of the soil (Owens, 1970). Unlike the other nutrients, the highest levels for dissolved silica were at Station 1 (108.3-263.7 µg at l⁻¹). This reflected the drainage area near the source and was also due to the very small diatom numbers available at that station for consuming the silicate. The values decreased very slightly at Station 2 and decreased more at Station 3 by about 75 µg at l⁻¹ less than Station 1, probably due to the higher diatom growths at these stations (Casey, Clark and Marker, 1981). At Station 4 the levels decreased by a further 10 µg at l⁻¹ less than the previous station and at Station 5 by about 100 µg at l⁻¹ less than Station 1. The levels at Station 6 were similar to those at Station 3, but at Station 7 values were down to about 140 µg at l⁻¹ less than Station 1. At Station 8 the maximum values were as high as Station 1 but Stations 9 and 10 had similar levels again to Station 3. The dissolved silica profiles, Fig. (4.11), show that the overall average contents at the stations did not vary much, being highest at Station 1, with a slight

decrease on passing downstream. The seasonal variation in the dissolved silica content was also due to its usage by the diatoms during the growth periods especially during the diatom "outbursts". The relatively higher silicon concentration recorded during the whole period of 1980 was probably due to the low standing crop of the diatoms compared with the subsequent years. Owens (1970) commented that diatom growth was often limited by the dissolved silicon concentration and the effect of pollution would show up in non-diatomaceous algal growth than in more diatoms.

The seasonal variations measured for the other nutrients, $\text{PO}_4 \text{ P}$, $\text{NO}_3 \text{ N}$ and NH_3 were mainly due to the current flow in addition to effluent input. This was caused by the increased water volume due to the heavy rainfall and subsequent dilution of these nutrients in the water. Thus low concentrations were observed during the autumn-winter periods. The unusually low current flows recorded occasionally during this period (e.g. December, 1981) were indicated by small peaks of the above nutrients during the same period. Flooding and high rainfall would bring more soil into the water, therefore more dissolved silica was measured.

The regular seasonal variations for pH values being maximum at summer and minimum during winter may be explained by the photosynthesis of masses of angiosperms and algae which tend to bring CO_2 out of solution and produce oxygen (Curtis and Harrington, 1970). The slight increase in the average pH values, particularly at Station 4 (Fig. 4.03) was mainly due to the sewage discharges which contain soap and other detergents of an alkaline nature. At Station 10 the increase may be partly due to its relatively high oxygen content.

The seasonal variation in the water temperature was expected in relation to the weather conditions obtained at that time. The slightly higher temperature recorded at Station 6 (Fig. 4.03) may be due to the effluent discharges. Relevant data were also given by C.R.P.B.

The enrichment of the Kelvin and its tributaries by the former nutrients mentioned will certainly enhance the growth of plants. This is similar to land plants where the growth increases by adding fertilizers into the soil, except when the soil is sufficiently fertilized.

Mulligan and Baranowski (1969) found a good correlation between the growth of algae and vascular plants and the available nutrient levels in experimental ponds. They applied different nutrient levels of high ($15-50 \text{ mg l}^{-1}$ N; $5 \cdot \text{mg l}^{-1}$ P), middle ($0.172-1.522 \text{ mg l}^{-1}$ N; $0.065-0.515 \text{ mg l}^{-1}$ P) and low levels ($0.037-0.172 \text{ mg l}^{-1}$ N; $0.02-0.065 \text{ mg l}^{-1}$ P) and found the best growth for phytoplankton, filamentous algae and vascular plants, respectively, at the different levels.

The effects of influencing river plant growths by nutrients have been mentioned by many investigators. Jolly and Chapman (1966) found heavy growths of the green alga *Stigeoclonium* and diatoms below the source of pollution in Farmer's Creek and Cox's River in South Wales; Benson-Evans and Williams (1970) suggested that the entrance of detergents into the River Usk reduced the diatom population but stimulated the growth of *Cladophora* and *Lemanea*. Whitton (1975) concluded that certain species of algae may define their environmental conditions (e.g. massive growths of *Cladophora* may indicate a form of nutrient eutrophication). Casey and Ladle (1976) related the rich flora in Winterborne to the addition of nutrients by man. Mitchell (1980) observed that the organic

enrichment is the most important factor of influencing diatom and macrophyte distribution in South Wales rivers. Say and Whitton (1981) related the floristic changes in stream sites in northern England to changes in zinc levels. Foster (1982 a and b) found *Microspora* and *Spirogyra* communities respectively growing in high and low metal pollution in the Rivers Hayle and Gannel in Cornwall.

The phytoplankton in the River Kelvin are mainly benthic forms which have been detached from their substrata by physical factors such as current flow. The benthic origin of the diatoms was confirmed in this study by finding similar species suspended in the water and attached to the macrophytes. This was parallel to surveys on other British rivers (Butcher 1947; Moss 1973; Whitton 1975; Wetzel 1975; Marker and Gunn 1977 and Moore 1977 b and c). Among these, Moss also used the word "tychoplankton" for these kinds of algae which have been resuspended from attached algal communities on rocks and sediments by water movement. Zacharias (1898) used "potamoplankton" for these types of algae which he recorded from some German rivers. The existence of "true" plankton has been doubted by investigators for many years (e.g. Kofoid, 1903; Hawkes, 1975). The existence of phytoplankton species in rivers is thought to be brought about by their being inoculated from a neighbouring lake, pond or reservoir. Very few species of truly planktonic diatoms were recorded in the River Kelvin, being mainly the centric diatom *Cyclotella meneghiniana* and *Asterionella formosa*. The former diatom was observed during the periods of minimum flow rates and this was similar to records from other British rivers (e.g. Whitton and Dalpra, 1968 for the River Tees; Swale, 1969 on the rivers Severn and Stour; Lack, 1971 on the

River Thames; Holmes and Whitton, 1981a for the River Tees). They related this to the slow flowing of the river water so that some species, mainly centric diatoms, can maintain population growths. The existence of *Cyclotella* in the Kelvin was likely to be due to the above reason since it occurred during the period of low flow, June-August (Fig. 4.01). The presence of the other planktonic diatom, *Asterionella formosa* in the Kelvin mostly during February and September (Fig. 4.13) may be due to chance (Symoens, 1951) or as Blum (1960) concluded that in nearly all streams, the usual phytoplankton pulses are during the warm season but individual plankters may frequently show growth maximum in winter.

The main spring "outburst" in the Kelvin was probably caused mainly by the increased day length (Blum 1956 and Moore 1976) since considerable quantities of nutrients (especially silicate) were available in the river even during winter. The spring-summer period was characterized by the abundant growth of diatoms and certain macrophyte species. Similar observations were recorded on the Rivers Rheidol and Melindwr in Cardiganshire (Reese, 1937). The planktonic algal flora in the Kelvin showed generally a single annual cycle with a maximum mostly in spring and a drop in midsummer. The community was dominated mainly by the diatoms *Cyclotella meneghiniana*, *Gomphonema parvulum*, *Navicula avenacea*, *Nitzschia thermalis* and *Synedra ulna*. They differed in their quantities through the whole period and along the river as a whole, and their peak growths did not occur at the same time every year. A similar result was obtained by Butcher (1946b) in a study of the effect of effluents on the River Churnet, where he found a high increase in the growth of diatoms, mainly *Gomphonema parvulum* and *Nitzschia palea* and *Cocconeis*.

in addition to other algae (*Stigeoclonium* and *Chamaesiphon*). However, the maximum growth in the Kelvin occurred during the warmest months and during the period of low discharge. This was parallel, respectively, to results for the River Wye system (Furet, 1979) and the River Thames (Lack 1971, 1973 and Berrie, 1972).

The relatively low standing crop of the phytoplankton in the River Kelvin, where high populations of filamentous algae were to be found, was also described by Berrie (1972) for the River Thames, Moore (1977b) in a canal in southern England and Aykulu (1978) for the Bristol Avon. Fitzgerald (1969) has related this correlation to antagonism and competition between the two populations for nutrients. As the nutrients existed in the Kelvin, always in large amounts, this could be due to other environmental factors (Moore, 1977b). It must be remembered that the amounts of nutrients available in addition to the flushing and flooding of the water were the main factors affecting the composition and the distribution of the phytoplankton in the River Kelvin. Similar observations suggested by Moore (1976) on the River Avon, Moore (1977a) on a eutrophic stream in southern England and Moss (1977) on the River Wye. These factors (particularly nutrient loading) in addition to the substrate on which the diatoms were found to grow were the main factors affecting the difference in species composition and distribution at the different stations on the Kelvin. Whitton (1975) suggested that the distribution of benthic algae in streams is affected by size, surface texture and chemistry of the substratum. He also found better algal colonization on coarse sandstone surfaces as the particles can be ~~adhered~~
^{to} ~~adhered~~ by the algae better than on a smooth surface. The substratum

in the River Kelvin consists of unstable mud and pebbles at some sites and frequently the sediment is covered with rubbish accumulation. Patches of mud were found at most of the stations during the time of low flow but they were washed away by fast flows and floods. Attempts were made to sample the mud surface for the epipelic algal colonization by drawing a glass tube over the mud surface (Aykula, 1982; see sec. 3.6.3). This was not successful due to the unlevelled sediment and also to the rubbish accumulation which prevented the tube moving freely over the surface in addition to the filamentous algae which were found covering the pebbles (if there were any) and the rubbish surfaces.

The very low standing crop at Station 1 and the relatively low diatom populations at Stations 3 and 7 were mainly due to their low nutrient loading (Figs. 4.10 and 4.11) compared with the Luggie Water and Bardawi Bridge (Stations 4 and 6 respectively). The latter two stations had almost always the highest diatom population. Thus the increase in the population downstream was mainly due to the influence of sewage treatment works, but the extra pulses occurring at Station 10 (during May and August 1982 - Figs. 4.12 a and b) were probably due to the washing of the diatoms from the upper reaches and accumulating downstream at the last station. The upstream and downstream stations did not show any variation in the occurrence of their diatom "pulses" and this contrasted with the study on the River Tees (Holmes and Whitton, 1981a) where the spring diatom outburst occurred one month earlier in upstream sites than in downstream ones.

Some diatoms were more likely to be brought into the main river by its tributaries. *Ceratoneis arcus* was found in the Kelvin after its

confluence with Glazert Water, whilst *Nitzschia dissipata* and *Nitzschia thermalis* were obtained after joining with Bishopbriggs Burn.

The maximum chlorophyll a values recorded in the River Kelvin did not coincide with the times the peaks occurred for the phytoplankton during the spring "outburst" and the diatoms variations at the different stations. The station with the highest number of diatoms (Station 4) did not show maximum chlorophyll a values. However, the chlorophyll a concentrations recorded were relatively low, but the phaeophytin was high. As already explained, this may be due to the presence of the large amounts of suspended matter in the river which could interfere with the analysis and help produce the large quantities of phaeophytin. The relatively large amounts of phaeopigments observed in the river as a background for this estimation supports the above reason. It was stated by Marker (1977) that when a large amount of phaeophytin is present in a sample, the chlorophyll a extraction of that sample in acetone is not suitable for measurement, due to the interference between 630 nm and 670 nm. Although methanol is superior to acetone for extraction, chlorophyll a extraction in 90% acetone in the presence of phaeopigments is still the most reliable method since extraction in methanol has some disadvantages and has been criticized by many investigators. Chlorophyll a is less stable in methanol (Marker, 1972, 1977 and Tett *et al* 1975). During the acidification, the rate of chlorophyll a conversion to phaeophytin is slow, which might prohibit the analysis in addition to the formation of precipitates which might occur (Marker *et al* 1980). It was also found by Tett *et al* (1977) that using methanol for extraction in the presence of other pigments would lead to overestimation of phaeophytin a.

and underestimation of the chlorophyll. Marker and Gunn (1977) stated that the method of using methanol for differentiating between chlorophyll and phaeopigments does not allow for the presence of intermediate degradation products (e.g. chlorophyllide). However, the seasonal variations recorded in the chlorophyll a in the River Kelvin being low during autumn-winter and relatively high during spring-summer period was due at least in part to the growth of the phytoplankton.

Marker and Gunn (1977) measured the suspended chlrophyll a and phaeopigments in two eutrophic streams, Bere stream and the River Frome, and also a soft stream, Ober Water, in southern England. They observed maximum chlorophyll concentrations during April, being of mean maxima for the whole period of 22.5 mg m^{-3} for Bere stream, 29.4 mg m^{-3} for the River Frome and for Ober Water, 7.0 mg m^{-3} . They related these high values to the pennate diatoms detached from the benthic flora. Low chlorophyll values and high phaeopigments were observed during July and August, due to the rise in the degradation process. These chlorophyll values were much higher than the concentrations recorded for the River Kelvin but the low concentrations obtained during winter were similar to the above streams with occasional chlorophyll and phaeopigment increases almost in proportion.

The ^{14}C fixation by the phytoplankton in water samples from the River Kelvin varied for the different stations and did not fit the total numbers of diatoms available (Fig. 4.12b and Table 4.12), but fitted more closely with the chlorophyll a values (Fig. 4.14). The minimum fixation observed at Station 1 was due to both its low phytoplankton population and chlorophyll concentrations. The fixation at Station 4,

Luggie Water, was relatively low although the highest phytoplankton populations were almost always obtained there. The highest ^{14}C fixation for most of the period was recorded in samples from Station 6. This low fixation level at Station 4 is probably a measure of cell activities although the water quality at this station is highly loaded with nutrients which leads to an expansion in the population sizes. These may not be of healthy cells as indicated by their measurement of photosynthetic activity. More fixation was obtained with Station 6 samples although the cell numbers were higher at Station 4, excluding May and June when the population was greater at Station 6 by 10,000 cells m^{-1} during April and by about 1,000 cells m^{-1} during June, August and September (Fig. 4.12b). Findenegg (1971) suggested that in productivity studies the carbon content is a relevant way of estimating populations since the production is given as the amount of carbon fixed per unit time. He also considered the phytoplankton cell sizes as a source of error rather than cell numbers for a true estimation of a population biomass (e.g. volume of the species, *Asterionella formosa* is $700 \mu\text{m}^3$, *Melosira granulata* $60,000 \mu\text{m}^3$ and *Microcystis aeruginosa* $100,000 \mu\text{m}^3$). Cell sizes do not seem to be a source of error in the ^{14}C fixation in the River Kelvin since the species observed were the same at the different stations. Fogg *et al* (1965), Fogg (1977) stated that the excretion of the organic products by healthy phytoplankton cells makes another source of error in estimating the total primary production by ^{14}C method. He also suggested (Fogg, 1971) that this excretion increases with increasing oligotrophy of the water. The fraction of the total organic carbon fixed released as extracellular material being 1% in highly eutrophic

freshwaters, but in oligotrophic waters this rises to 35%. Since the Kelvin is a eutrophic stream, this may not be a major source of error. However, Peterson (1980) stated that although alternate methods of measuring carbon flow in phytoplankton used (e.g. change in ATP, $^{14}\text{C}-\text{CO}_2$ uptake by bacteria, change in particle number and volume), the results of these suggest that the underestimation which occurs often in ^{14}C uptake could be for reasons not completely understood. He also concluded that in spite of all the modified ^{14}C techniques, the original method (Vollenweider, 1971) remains the standard and basic method for estimating aquatic primary production.

The River Kelvin supports few species of filamentous algae, being mainly, *Cladophora fracta* and *Cladophora glomerata* at the eutrophic sites, accompanied by *Oedogonium* sp. and *Vaucheria* sp. (Sec. 4.2.2). These algae were the common filamentous types recorded at most other British rivers (e.g. Thames, Swale, Avon, etc.). The stations differed in their contents of these types of algae with no growth of filamentous green algae at Station 1, and with relatively poor growth at Stations 3 and 7, compared with Station 4, Luggie Water, which had the highest amount of these algae, mainly *Cladophora*, forming a green blanket during the summer (Plate 4.05) and this species was also dominant at the rest of the stations. The variations among the stations could be due mainly to the difference in the water quality. Thus the Kelvin at Station 1 supported no algal growth and at Stations 3 and 7 small amounts of filamentous algae were found compared with the other stations. This may be due to the relatively low nutrient contents at these stations, particularly Station 1, being mainly PO_4^3- P and NO_3^- N (Figs. 4.10, 4.11). The highest levels of PO_4^3- P at Station 4 was probably the main cause for the extensive

Cladophora Growths. Fogg (1973) concluded that the growth of plants in aquatic habitats could be controlled by the available phosphorus compounds. Carole, Pitcairn and Hawkes (1973) correlated the standing crop of *Cladophora* with phosphorus concentrations in river waters and they found that in culture experiments the *Cladophora* growth in waters upstream of the sewage discharge could be increased to that of downstream sites by addition of phosphorus. They also suggested an interaction between the phosphate phosphorus and nitrate nitrogen in natural waters and observed different available maximum phosphorus concentrations for *Cladophora* growth being 2.5 mg P l^{-1} at 3.2 mg N l^{-1} and 0.95 mg P l^{-1} at 5.25 mg N l^{-1} , and found that the *Cladophora* growth in synthetic media would be enhanced by the highest levels of NO_3^- (7.7 mg N l^{-1}) at the lowest phosphorus levels (0.5 mg P l^{-1}) but the growth was reduced at the higher phosphorus levels. In addition, they obtained good *Cladophora* growth in waters with pH 7-8 and observed that the *Cladophora* is sensitive to heavy metals. The sensitivity of this alga to zinc was also suggested by Whitton (1980) and Harding, Say and Whitton (1981). In a survey on the River Etherow by the latter, the absence of *Cladophora* at some of the sites was related to zinc pollution (mean maximum Zn concentrations 72.9 mg l^{-1}) but abundant growths occurred at stations with average levels below 0.5 mg Zn l^{-1} . They also suggested that this alga almost always grows in nutrient-enriched waters. The study of the River Etherow (a river recovered from pollution) confirms the present work on the River Kelvin since the latter is a nutrient-enriched stream with no evidence of zinc pollution. This is parallel to studies on other British rivers (e.g. River Swale, Holmes and Whitton, 1977;

River Tame, Harding 1979; River Douglas, Harding 1980 and River Etherow, Harding, Say and Whitton 1981). The non-existence of *Cladophora* at Station 7 is probably due to its low phosphorus loading but its domination by *Oedogonium* could be related to unknown factors. The disappearance of the filamentous algae at Station 5 during the second half of this survey may be due to the interference and cutting by man at the time of building the new bridge. The growth peaks occurred during the summer and were probably due to the increase in day length and high nutrient levels (Blum 1960; Whitton 1975).

The most species of macrophytic angiosperms mainly found dominating the river were *Potamogeton natans* and *Potamogeton filiformis* (Sec. 4.2.3). These are among the common angiosperms growing in many British rivers (Haslam 1978). The distribution of these flowering plants in the Kelvin (Table 4.13) could be due to the water quality, current flow and type of substratum. It was concluded by Moyle (1945) "water chemistry appears to be the most important single factor influencing the general distribution of aquatic plants but the type of bottom soil and the physical nature of the body of water greatly influences the local distribution of a species within its range of chemical tolerance". The chemical nature of lake water was also correlated with the macrophytic assemblages (Seddon, 1972). He related the distribution of lake species to total dissolved solids, conductivity and hardness. Spence (1969) related the distribution of aquatic angiosperms in freshwater lochs in Scotland to the source of carbon (i.e. some species use free carbon dioxide but not bicarbonate, e.g. *Potamogeton polygonifolius*). Westlake (1975) stated that the carbon may be an important limiting factor in addition to other nutrients (mainly

phosphorus and nitrogen). Haslam (1978) provides a very good table which lists the aquatic angiosperms under the nutrient concentration to which they are related and she also mentioned all the important factors affecting the macrophytic vegetations in water sources, being mainly nutrient supply, light and current flow. The latter two factors were the main ones affecting the seasonal variations of the macrophytic angiosperms in the River Kelvin. Their growth peak during the summer months was mainly due to the increase in day length and lowered flow rates, reaching the growth maximum during July and August. The decrease in day length with the high discharges were the factors affecting the disappearance and flushing away of the branches during the autumn. According to Cook et al (1974) *Potamogeton* and *Sparganium* are perennial herbs regenerated from winter buds produced by rhizomes, but *Elodea canadensis* regenerates well from fragments (Haslam 1978). Ham et al (1981 and 1982) found that discharges affected the growth of *Ranunculus* in the River Lambourn by influencing the underlying substrata and they related the increase in the *Ranunculus* in the same river during spring to the mean discharges at that time. Seddon (1972) stated that *Elodea canadensis* is among the species tolerant of oligotrophic waters and *Potamogeton natans* is a species tolerant of a wide range of conditions. Spence (1964) found *Potamogeton filiformis* to be of a wide range of tolerance, from moderately rich to rich waters.

It was stated by Haslam (1978) that *Potamogeton natans* (broad-leaved pondweed) is frequent in medium sandstone streams with moderate flow on clay in upper reaches of essentially eutrophic waters. *Sparganium emersum* (strapweed) is characteristic of clay streams in eutrophic and semi-eutrophic streams with moderate water volumes, and *Elodea canadensis*

grows on clay catchments and absent from very oligotrophic water it grows in swift water but prefers still and slow moving clear waters with some eutrophic confluence. It was also stated by Haslam, Sinker and Wolseley (1975) that *Potamogeton filiformis* and *Potamogeton natans* are the characteristic species of eutrophic waters. *Sparganium* was found growing in both oligotrophic and eutrophic waters and *Sparganium emersum* was found at Station 1, probably due to its relatively low nutrient loading and the muddy substrata. At most of the other stations on the main river *Potamogeton natans* and *Sparganium emersum* were mostly found (excluding Station 10), probably due to their high nutrient loading and their moderate flow. The existence of *Potamogeton filiformis* at Station 4, Luggie Water, may be due to its high nutrient loading. This species seemed to be brought into the main river via Luggie Water since it was found in the main stream downstream of Luggie's confluence. This was similar to the source of *Elodea canadensis*, which was found in the River Tees to have been transferred from the Tyne system (Holmes and Whitton 1977a). The absence of the angiosperms at Station 10 could be related to the fast water flow as this station has the fastest flowing currents and is the only station with small waterfalls and the sediment was covered by rubbish and pieces of metal which were found covered by *Cladophora glomerata* during the whole growth period. Butcher (1933); Dawson (1973); Westlake (1975); Holmes and Whitton (1975) and Holmes and Whitton (1977a) related the absence of angiosperms in water sources are mainly due to the rocky substrata and the water velocity. This absence was similar in the upper reaches of the River Swale (Holmes and Whitton 1977b) due to the above mentioned factors. Unlike the rivers

Tees and Swale (Holmes and Whitton 1977a and b) where no macrophytic angiosperms were found on the upper reaches and started to appear in the lower reaches, the River Kelvin had supported perennial angiosperms along the whole stretch of the stream starting from Station 1 and increasing downstream (excluding Station 10), they were found rooted in mud patches on the sediment which formed during the period of low discharges.

The epiphytic algal flora in the River Kelvin, being mainly diatoms, varied at the different stations and almost always the highest population was obtained at Station 6 but with an uneven distribution over the different regions of the host at all the sites (Tables 4.15 and 4.16). The epiphytic species failed to show a uniform seasonal pattern during the whole period 1980-1982. A likely source of significant error difficulties could lie in the sampling method (by grapling) imposed by the nature of the river and river banks. *Cocconeis placentula* and *Gomphonema parvulum* were the most numerous epiphytic species during 1980 at almost all the stations but during 1981 and 1982 the former was replaced by *Cyclotella meneghiniana*, *Navicula avenacea* and *Nitzschia thermalis* at almost all the stations (Figs. 4.16-4.28). The existence of planktonic algae as epiphytes "loosely attached" were observed in lakes by Cattaneo and Kalff (1978) and the trapping of phytoplankton by macrophyte beds in streams was described by Chandler (1937). It was suggested by Moss and Abdel Karim (1969) that this trapping was not permanent in lakes and the diatoms may return to the phytoplankton because of their loose attachment. The presence of the centric diatom *Cyclotella meneghiniana* in the Kelvin's epiphytic flora during August (1982) may be due to the low flow rates

and trapping by the macrophytes.

The differences recorded for the epiphytic diatom populations in the River Kelvin could be related to many factors, being mainly, current flow, type of substratum and nutrient loading. Godward (1934, 1937) suggested that the distribution of epiphytes was affected by the age and rate of growth of the host and also by the nature of the host surface. She found maximum growth of epiphytes on the oldest living leaves of the host (parallel to the present study) and also observed more epiphytes on the upper leaf surfaces than the lower. Godward also stated that the dominancy of *Eunotia incisa* and *Cocconeis placentula* on the under side and upper side respectively of the leaves of *Potamogeton alpinus* and *Elodea* in Lake Windermere were produced by different light requirements. Bell (1976) confirmed the epiphyte distribution suggested by Godward (1934) over *Elodea canadensis* in a eutrophic canal near Liverpool. He related the different epiphyte density at various stations with the time when *Elodea canadensis* started to appear at the sites. His correlation suggested that the epiphytic algal peaks were due to colonization by algae which quickly occupied the small initially available surface areas of the host. Cattaneo (1978) and Cattaneo and Kalff (1978) stated that the differences in the distribution of the epiphytes on the upper and lower surfaces of *Potamogeton richardsonii* in a lake was due to the distinct CaCO_3 encrustation on the upper sides. Thus the species which favours CaCO_3 will colonize that area. They also found that this CaCO_3 precipitation may prevent a dense growth of epiphytes either by shading or reducing the available space for colonization. They obtained denser diatom growths on the margins of leaves, probably due to some physical

advantages, e.g. illumination or nutrient renewal. Cattaneo and Kalff (1978) related the lower epiphyte biomass on the upper portion of *Potamogeton richardsonii* to the limited time allowed for epiphyte colonization on the growing tips. The most important factor which governs the epiphytic populations in streams is current flow. This was stated by Douglas (1958), Whitford (1960) and McIntire (1966). They found that the attached algae are more abundant in faster flowing streams. The latter two authors suggested that the water is "physiologically richer" due to the renewal of materials in solution near the surface of the organisms. The type of substratum on which the epiphytes will grow, as mentioned earlier, is another major factor controlling the epiphyte distribution. Round (1965) found *Cocconeis* / *Epithemia* association on the underside of *Lemna* leaves, whilst *Comphonema* and *Achnanthes* were found on *Potamogeton* leaves. The relatively low epiphytic population recorded at Station 4 (excluding July and August 1981) in the present work was probably due to the different species of *Potamogeton* found growing there although the nutrient loading was high. It was also found from the preliminary experiments on comparing the epiphytes growing on *Potamogeton* and *Sparganium* in the present study that the latter supported either very few or no diatoms whilst the colonization on the former angiosperm could exceed 150×10^5 cells gm^{-1} dry wt. The greater population recorded mostly at Station 6 may be due to the dissolved substances available, particularly nitrate N, phosphate P and dissolved silica. The effect of nutrients on the epiphyte distribution received little attention. Butcher (1946b) related the difference in the epiphytic

algae in the River Tees, dominated by *Achnanthes microcephala*, *Chaetopeltis* and *Diatoma hiemale* at the upper (oligotrophic) reaches and the replacement by *Cocconeis ulevilla* - *Chamaesiphon* community at the lower reaches, to be due to the sewage drain flows into the river. In another survey on the effect of tar distilling industries on the River Trent (Butcher, 1947), *Nitzschia palea* and *Stigeoclonium tenue* were found growing throughout the polluted reaches whilst *Cocconeis placentula* was restricted to the very lower reaches where oxidation of the wastes, dilution and mixing were suitable for its growth. Fitzgerald (1969) stated that the available nitrogen affects the growth of epiphytes on *Cladophora* and he also suggested the antagonistic factor between the macrophytes and the epiphytes for the competition for limited nitrogen compounds. On the contrary Cattaneo and Kalff (1979) suggested that *Potamogeton richardsonii* transfers a small amount of nutrient to epiphytes in lakes. Moore (1977c) related the low standing crop and the growth rate of the microflora in a woodland stream to the low alkalinity since nitrates, phosphates and dissolved silica were always available in relatively large amounts. Moss (1981) found that changes in the periphyton development in a lake and experimental tubes are due to the difference in the nitrogen available in waters enriched by phosphorus. He also found that the epiphytic diatom flora were of two species groups, the first was restricted to the periphyton and the second species group was reflected by the phytoplankton. In the present study (Sects. 4.2.1 and 4.2.6) the diatom species constituting the epiphytic flora were also represented in the river phytoplankton and more species diversities for the phytoplankton and the epiphytic flora were recorded during the

seasons 1981 and 1982 compared with 1980. This was more in agreement with the second species group of Moss above. However, Moss's studies were on lake vegetation. The seasonal changes in the epiphytic flora in the River Kelvin which occurred in the flora as a whole, as well as in individual species, could be related to the above mentioned factors, in addition to light and temperature. The effect of the latter two factors is difficult to separate in seasonal field studies. The seasonal changes in the epiphytic flora has received a number of studies (e.g. Butcher 1938, 1940), who found fewer seasonal differences of growth on glass slides in the slow flowing Hampshire Avon and the River Hull in England than in the swifter flowing rivers Tees, Itchen and Lark. He also observed maximum colonization in the latter river during May with periods of rapid reproduction for individual dominant species from March-October, but in cold winter months the growth was slower (as with the Kelvin). Blum (1960) also observed maximum development during warm months, followed by a minimum in cold months in streams and he also stated that while a single benthic alga could be found dominating a long stretch of a stream, a number of different dominant species could be found over other reaches of the same stream. Moore (1977a) related the seasonal succession of the epiphytic algae to changes in light conditions and he correlated large variations in streams to flow rates.

However, despite the wide range of papers referring to the ecology of periphyton no clear picture can be drawn. Although the importance of periphyton in some freshwater ecosystems are known it is difficult to investigate the physical and biological parameters regulating growth.

The problem of getting reliable information about the seasonal patterns occurring in natural periphyton communities led some investigators to use natural and artificial substrata and to determine the attached algae growing on them (Sladeckova 1962; Sladecek and Sladeckova 1963, 1964; Tippet 1970; Schwoerbel 1970; Cattaneo 1978). The epiphytes found growing on the natural substrata placed in the River Kelvin (Sec. 4.2.8) probably represented partly the natural communities (excluding the unicellular green algae, *Chlamydomonas* sp., *Gleocystis major* and *Pamella mucosa*). In these short term experiments the natural communities were reflected by the diatom colonization at the beginning and the filamentous algae later and they were of the same species which were observed colonizing the angiosperms. Sladecek and Sladeckova (1963, 1964) found the method of submerged glass slides convenient for periphyton production studies. Wetzel (1965) stated that the use of artificial substrata for determining algal colonization had numerous sources of error (e.g. selectivity of substrata by particular organisms, colonization rates and mechanical losses). The artificial substrata will be suspended in the main river flow, which would not necessarily represent the natural growth conditions of the periphyton. Wetzel also concluded that many estimates of periphytic production represented only colonization rates of certain of the phytoplankton and may be entirely unrelated to true productivity by sessile producers. Tippet (1970) found that natural epiphytic diatom populations and those on glass-slides were different in the seasonal patterns of growth of both the whole population and the seasonal growth of individual species. Cattaneo (1978) found distinctly different colonization on the upper and undersides of

living plants compared to that on nearby plastic aquarium plants.

However, despite the loss of material, in the present survey, these substrata were found to be more suitable for these studies.

The productivity of the periphyton on the different regions of *Potamogeton natans* in the Kelvin was measured by ^{14}C uptake (Table 4.17). The results showed that although larger diatom populations were often found on the basal regions, the population on the distal regions showed more photosynthetic activity. This may be due to either their position near the water surface, being exposed to higher light intensity than those on the basal and mid regions or to increased rates of gaseous exchange and enhanced nutrient supply for epiphytes on the distal region due to the effect of their rapid and continuous movement by the water current more than the other regions. This movement could be termed the 'flag effect' which might also cause more release of extracellular material from the distal region. The productivity measurements of periphyton communities have been attempted by some investigators (e.g. Wetzel 1964, 1965; Vollenweider and Samaan 1972) using the ^{14}C uptake method and the results suggest that the heterogeneity and the distribution of these epiphytes could be a large source of error in such estimations.

Marvan (1979) stated that the algal bioassay is the most natural way to evaluate the causes of eutrophication separately from the complex factors which are brought about in nature. He also suggested that the algal bioassay is a source of quantitative information which could corroborate the water chemical analysis results about the trophic value of water. Most of the methods used for estimating the growth of algal cultures have been either direct methods e.g., dry weight, cell volume...

or indirect methods e.g., optical density and extracted pigments.

Lukavský, Simmer and Kubín (1979) reviewed the most widely used bioassay methods and also the newly developed methods. These methods included the dry weight, packed cell volume, protein determination, spectrophotometric determinations of chlorophyll a and b, determination of nucleic acids and cell number estimation. None of these methods measured the photosynthesis of the algal cultures. Matulova (1979) applied the bioassay with *Scenedesmus* to determine the available nutrient levels in the water examined (i.e. eutrophication level). Measuring the cell numbers, he found that increasing $\text{PO}_4\text{-P}$ concentrations in the water caused more *Scenedesmus* growth. Žáková (1979) stated that "in spite of the fact that the trophic potential is found to be in most cases dependent on the phosphorus and nitrogen concentrations found by chemical analysis, chemical analysis alone cannot give an unbiased picture of the water acting as a complex of substances on the biological activity of water plants". He also suggested that the methodology of the trophic level potential which he tested (e.g., determination of trophic levels in surface water, nutrient supply from different sources, nutrient limitation for water plant development, providing basic data for the implementation of practical measures against high eutrophication in water sources) show a wide application of algal tests in hydrobiological studies. We may ask how applicable are these methods to lotic systems where water flow rates cannot be simulated in laboratory cultures? Also Žáková suggested that "these determinations may provide important data for practical measures ensuring the reduction of nutrient supply to rivers (such as deciding about the necessity of tertiary treatments,

elimination of nutrients, wastewater dilution, etc.)". The reason for applying algal bioassays in the present investigation was to examine whether there were "hidden" factors which might affect the growth and metabolism of algae and which could become significant at times when flow rates are very much reduced and the concentrations of the nutrients become higher, particularly during summer. An artificial balanced medium (Bold's Basal) was used as a control for the bioassay experiment due to the optimal growth of the test algae in the medium prior to their inoculation in the river samples and the fact that the test algae had been grown in the medium. The water samples from Station 1 differed in their nutrient status from the remaining stations (with the exception of the dissolved silica levels), due to the increased effluent loading down the river. However, initial tests showed that even at Station 1 a large number of abnormal cells were formed. The results of the algal bioassay experiments showed the same response to the water quality in the River Kelvin as the natural populations. The population changes for the experiments were expressed as total cell numbers of *Scenedesmus quadricauda* per ml of river cultures (Figs. 4.30-4.34). Relatively low populations were obtained in the cultures of water samples from Stations 1, 3 and 7 and the growth was higher in the water samples from the other stations being always the maximum at Station 4, Luggie Water, where the initial growth obtained was even more extensive than that of the control. This variation is probably due to the amounts of dissolved nutrients in the water samples, particularly phosphate and nitrate. Matulova (1979) studied the effects of the eutrophication (phosphate P and nitrate N) on cultures of *Scenedesmus quadricauda*. He found that by washing the $\text{PO}_4^{-\text{P}}$

from the cultures the vitality of the cells was decreased and he observed growth inhibition in the cultures with no $\text{PO}_4^{3-}\text{-P}$. He obtained a linear relationship for the growth by increasing the phosphate P concentration in a range of $0.043\text{--}0.51 \text{ mg P l}^{-1}$ and he found that increasing the concentrations above this level did not strongly affect the slope of the growth curve but it affects the growth maximum. He also obtained significant development in *Scenedesmus quadricauda* cultures with nitrate N levels of a range $1.4\text{--}7.0 \text{ mg l}^{-1}$ but increased growth rates were not obtained in concentrations $> 7.0 \text{ mg l}^{-1}$. He observed maximum growths at the highest nutrient levels. This was parallel to the River Kelvin's cultures for which the maximum growths (in terms of cell numbers) were always in the water samples from Station 4, where the highest levels for the nutrients were almost always recorded. Marvan and Přibil (1979) proposed other factors limiting algal growth in cultures, these included either insufficient illumination, low temperature, low concentrations of important nutrients in the medium and the low uptake by the cells from the medium. However, the apparent extensive growth for the coenobia of *Scenedesmus quadricauda* obtained in the River Kelvin's cultures based on cell number increases alone was not regarded as a reliable measure of water quality. Cell appearances and metabolism should also be taken into account. Sulek (1979) stated that "if a number of different parameters is to be recorded in the assays with algae one must not fail to follow the appearance of the cells". In the present investigation the formation of the cell "clumps" by the alga *Ankistrodesmus falcatus* (Sec. 4.3.1) was probably a response to water quality. This is probably the same reason

for the formation of the abnormal cells in the coenobia of *Scenedesmus quadricauda* (Sec. 4.3.2; Pl. 4.09, Fig. 2) and these cells were responsible for the growth inhibition in the cultures and their rapid decline after the first few days rapid growth. Matulova (1979) found irregular cells in the coenobia of *Scenedesmus quadricauda* when they grow in phosphate-starved medium. Necas (1976), Necas and Sulek (1977) and Sulek (1979) found that monstrous coenobia of *Scenedesmus quadricauda* could be obtained under favourable conditions of growth but abnormal cells will form when the organism is under stress of improper combinations of temperature and levels of irradiance or low nutrient conditions. They named these abnormal cells "developmentally stopped and stunted cells" and they were characterized by morphological and cytological changes in the cell structure (e.g. changes in the cell size, changes in the cell wall and the formation of single cells, unfavourable alterations in cell nuclei and changes and destroying the chloroplast structure in the cells and consequently the disappearance of the cell pigments). They stated that these changes could happen during any phase of the life cycle and can cause a total cessation of further development and growth. Fig. 4.35 shows that higher percentages for abnormal cell formation were obtained during winter and these decreased towards spring. The reasons for the high number of abnormal cells are difficult to explain. Whilst nutrient levels are lower in winter than in spring and summer (especially phosphorus), the N:P ratios for 1980-1981 vary from 13:1-79:1 (December), 18:1-116:1 (January), 4:1-20:1 (February), 22:1-114:1 (March) and 1:1-23:1 (April). Whether these cell abnormalities are a response to nutrient imbalance, or to undetermined toxic effects cannot be explained.

The chlorophyll a measurements are indirect estimations of biomass (Lukavský, Simmer and Kubín, 1979). Such measurements are not reliable without taking the breakdown products into account. In the River Kelvin's cultures the changes in the chlorophyll a amounts and the changes in the colour of the cultures occurred after the first week of cultivation (Sec. 4.3.3; Fig. 4.36) made it very necessary to measure the phaeopigments (Sec. 4.3.4; Fig. 4.37). The results show high phaeopigment quantities during autumn and winter at the time of the formation of high percentages of abnormal cells. Thus the reduction in the chlorophyll a amounts and the increase in the phaeopigments could be referred to the formation of these abnormal cells. This could happen through the destruction of the chloroplast of the developmentally stopped and stunted cells, as mentioned above, which leads the cells to lose their pigments. The high chlorophyll a quantities and the relatively low phaeopigment amounts during August may also be correlated with the relatively low percentages of the abnormal cells. Chlorophyll a is the principal factor in the photosynthetic processes in any plant cell. Due to the unhealthy conditions of the *Scenedesmus quadricauda* cells in the river cultures it was thought advisable to measure the photosynthetic activities of the cells. The results of these measurements (Fig. 4.38) reflected the chlorophyll a and the phaeopigment measurements in that the cells in the river cultures showed lower rates of photosynthesis than the controls at all times of the year, with the exception of August, with a dramatic fall during September. This peak of photosynthetic activity would seem to reflect the high nutrient loading observed during the same month. It must also be remembered that September is the critical time when the natural populations in the river show a dramatic decline.

The results of these experiments indicate changes in water quality of the river water whilst standing and the causes of the dramatic increases in cell numbers, abnormal cells, low chlorophyll a and photosynthetic rate could be mainly due to the nutrient balance since there is a difference between the winter and summer results when the nutrient concentrations (N and P) are at their maximum during summer. There were no significant quantities of heavy metals measured in the river during the period of this study and there are no records of other toxic substances in the C.R.P.B. reports. The proliferation of bacteria in standing cultures producing toxic products may have induced cell changes. However, the bioassay experiments did correlate with our information on the Kelvin's water quality and it reflected the actual growth in the natural habitat being more extensive and healthy during spring and summer and vice versa during autumn and winter although other natural factors come into play, e.g., high flow rates, temperature, day length and available radiant energy. The main likely cause of the fall in the photosynthesis rates in the cultures during September is probably due to the decomposition of the macrophytes and microphytes in the river and it could also be due to the relatively lower nutrient concentrations because of the high flow rates.

The value of bioassay experiments may be summarized as follows:-

1. They have underlined some aspects of water quality in producing enhanced growth rates in samples from stations with high nutrient loading but have also shown that cell numbers alone are inadequate assessments.

2. Measurements of abnormal cells and chlorophyll a content for Station 4 (Luggie Water) have underlined the field data throughout. The water in this locality appears in some ways to be over productive.
3. The dramatic fall in photosynthetic activity in September reflects a massive change in river appearance, principally through the breakdown and defoliation of the macrophytes. In the field data a dramatic fall is seen in the number of suspended diatoms and those epiphytic. The 'hidden' nature of the change in water quality is not emphasized in the field data.

It would seem that the bioassay data has given an extra dimension to the field data, although the experiments have been carried out with standing water environments.

Pollution in the River Kelvin thus takes the form of eutrophication due to the addition of treated domestic sewage and industrial effluent. These effluents make the water "physiologically rich", and combined with flow rates, enable the river to support an abundant population of macrophytic angiosperms and massive populations of filamentous green algae. These in turn support a diverse diatom flora which reflects the genera and species growing on other substrata and suspended in the water; and the quantities of diatoms can be related to the nutrient loading at the different stations. Despite the eutrophication and dense weed populations, oxygen levels remain relatively high for most of the year. On occasions a sudden increase in summer flow rates has cleared the river bed of the algal 'mat' at some stations, and local accumulation of rotting weed can cause localized fish kills (as in the summer of 1982). The green algae and diatoms in the River Kelvin are typical of family 'clean'

rivers in other parts of the United Kingdom, and of the 'recovery region' of those rivers which are more severely polluted. Regular observations on the algal population would thus give useful information on the quality of the river water, and on possible long term changes or short term 'accidents' in effluent discharge control.

On September 29th, 1885, an article in the "Glasgow News" in describing the Botanic Gardens, made reference to the River Kelvin, viz. "The Botanic Garden so-called is a very pleasant and lovely park of many green lawns and old trees, fronting to a noble roadway, facing ranges of houses of much architectural merit, and sloping back to a stream which will, no doubt, in time be pure and limpid".

Pollution of the River Kelvin in those days was mainly due to the industrial and urban expansion of Glasgow in the Maryhill district. In nearly 100 years the river may not be "pure and limpid" but it would seem to be in reasonable condition if one takes into account the suburban growth on its banks and those of its tributaries.

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