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DOCTOR OF PHILOSOPHY
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Addendum

Chapter 3

Section 3.4.3

In reference to individual taxon dynamics and associated water quality parameters, the most notable relationship is that with salinity. In particular, *Staurosirella pinnata* and *Staurosira construens* var. *venter* increase in abundance downstream of Mildura, corresponding with an increase in salinity (rising from a mean of 366 µS/cm to 908 µS/cm at Lake Alexandrina – see table 1.1 for further details). Conversely, the highest abundances of *Aulacoseira ambigua* occur at sites with lowest salinity (Mildura through to Renmark). There is also an obvious relationship between abundances of the estuarine taxa *Actinocyclus normanii* and *Thalassiosira lacustris* and salinity levels, with highest abundances of these taxa occurring in Lake Alexandrina.

Chapter 7

Section 7.2.2

The relationship between rate of sedimentation and diatom dissolution also needs to be considered. The rate of sedimentation can be an important influence over the degree of diatom preservation, with slow sedimentation rates leading to diatoms being exposed to physical breakage and chemical dissolution for longer periods of time. Core 1, which has the slowest sedimentation rate of all three Lake Alexandrina cores, has the worst diatom preservation, while Core 3, with the highest sedimentation rate, has comparatively very well preserved diatoms.

Section 7.2.3.3

Further evidence for the relationship between the abundance of taxa in the Fragilariaceae and turbidity is the absence of *Campylodiscus clypeus* and *Tryblionella compressa* in the upper section of the core. These benthic taxa require light penetration to the lake bed, and thus, as turbidity increased, their habitat availability would have decreased.

Unfortunately, the plankton survey of the lower Murray River did not provide any further insights into the relationship between turbidity and small Fragilariaceae diatoms, with salinity appearing to be a stronger determinant of population dynamics within this modern data set.

Section 7.2.6

Inter – core comparison is a critical issue when examining complex, fluvially dominated lake systems. A comparison of the upper parts of cores 1 and 2 allows this issue to be addressed, as, despite minor differences, these cores do show similar diatom assemblages and downcore trends. In particular, the recent increase in Fragilariaceae taxa in both cores clearly began pre – 1900. When comparing this increase to assemblage changes in core 3, which was more precisely dated, there is most likely an acceleration in the increase of these taxa during the 1950s/1960s. The upper section of core 3 displays similar patterns of change in the recent past although the taxa are rather different, most probably due to

its depositional position near the river mouth. It is also important to note that the diatom assemblages in all three cores, when compared to the data in figure 3.9 (plankton diagram from the River Murray plankton survey), indicate that most of the diatoms appear to be of autochthonous origin.

The issues of core representativeness are not as crucial for Lake Cullulleraine, as it is a simpler limnological system than Lake Alexandrina and, as such, a single core is likely to be representative of lake wide changes.

Section 7.2.6

Additional table summarising the major limnological changes in both Lake Alexandrina and Lake Cullulleraine:

	Lake Alexandrina open water	Lake Alexandrina inflow zone	Lake Cullulleraine
Recent (post 1950)	Further increase in turbidity. No discernible changes in salinity, nutrients or pH.	Small decrease in salinity post river regulation. Sustained turbidity levels. No discernible changes in nutrients or pH.	Increase in nutrients post WWII, with a small decrease within the past 10 years. Increase in salinity, peaking in the late 1970s.
Early European impact (1820 – 1950)	Increase in turbidity. Increase in nutrient concentrations.	Growth in aquatic vegetation. Increase in nutrient concentrations. Increase in turbidity. Increase in salinity between approx. 150 years BP and 70 years BP.	Stabilisation of the lake basin post 1930s resulted in lower turbidity and an increase in littoral vegetation.
Late Holocene to pre – European contact	Lake level deepens. Small decrease in pH between approx. 700 to 900 years BP Eutrophic conditions continue.	Continuing fluctuations in pH.	
Early-mid Holocene (7000 BP to 2000 BP)	No diatom preservation in core 1. Marine incursion at its highest between 7000 BP and approx 2000 years BP Eutrophic conditions throughout this period.	Regular fluctuations in pH, probably ranging between 7.5 and 9.0.	

Section 8.2

The adoption of new, highly sophisticated numerical techniques, as an alternative to Weighted Averaging, is most likely unviable. The exception to this, perhaps, may be numerical methods that are capable of incorporating several controlling factors over time. Additionally, the continued pursuance of quantitative techniques for complex systems, such the lower Murray River, should not occur at the cost of implementation of simpler approaches, which although perhaps more qualitative in nature, may actually provide more useful information.

Section 8.4

The clearest diatom – inferred trend in the Lake Alexandrina cores is changes in turbidity levels, particularly in the recent past. It may therefore be possible to develop a palaeo – turbidity index based on the ratio of *Aulacoseira* to Fragilariaceae valves, applicable to this environment, and that can incorporated into diatom based restoration targets. However, because of its complex nature, and the fact that this study highlighted the dynamic nature of the lower Murray River system prior to non – indigenous arrival, restoration targets must also be dynamic.

Minor corrections

p.4 penultimate line "data is" should be "data are"; also page 19 (pt 1 and final paragraph); p. 100 (last paragraph); p. 164 (line 5).

p. 13 paragraph 1 line 6, "embayement" should be "embayment"

p. 16 paragraph 4 line 1, "palaeolimnogical" should be "palaeolimnological"

p. 19 paragraph 2, "principle" should be "principal" (also p. 23 paragraph 1 line 8; p. 25 paragraph 2)

p. 25 paragraph 4 Sayer should be 2001.

p. 26 paragraph 4 line 11, El Niño not El Ninio

p. 33 paragraph 1 should be e.g. Bennion, not ie. Bennion

p. 179 Zone 1C; inbetween should be in between

A diatom – based palaeolimnological investigation of the lower Murray River (south east Australia).

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Thesis submitted for the degree of Doctor of Philosophy,
School of Geography and Environmental Science,
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Lower Murray River, looking upstream from Mannum

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Abstract

The lower reaches of the Murray River, forming part of the greater Murray – Darling Basin, are an essential domestic, industrial and irrigation water source for much of South Australia and Victoria. Hence, this section of the river is subject to intense limnological investigation. This thesis aims to put the contemporary limnological status of the river in context of the palaeolimnology of the river, of which, there is relatively little known. The primary palaeolimnological objective of this study was to determine baseline water quality (with particular reference to pH, EC and nutrients) prior to the arrival of non – indigenous Australians, and also to establish the degree of change since this time (with particular reference to the impact of river regulation).

To facilitate these objectives, this study incorporates the development of a modern diatom data set which is used, along with information from other published diatom data sets, to make inferences about the autecology of selected diatom taxa that are prevalent in the lower Murray River fossil records. The development of this data set involved the collection of 50 modern lake samples and 51 modern stream samples (both diatoms and water chemistry). Canonical Correspondence Analysis determined that pH was the most influential environmental variable in driving changes to diatom assemblages. Hence, a pH transfer function was developed (using Weighted Averaging regression and calibration), with a correlation co-efficient between measured and predicted pH of 0.64 and a Root Mean Square Error of Prediction of 0.36 pH units.

Two sites were selected from the lower Murray River region as being suitable for diatom-based palaeolimnological investigation: Lake Alexandrina, South Australia, and Lake Cullulleraine, Victoria. Three sediment cores were analysed from Lake Alexandrina (dating to approximately 7000 years b.p.) and one core was analysed from Lake Cullulleraine (dating to approximately 65 years b.p). The primary conclusion from the Lake Alexandrina study is that the lake has been a eutrophic to hypertrophic system, and that pH has ranged between 8 and 9, for the past 7000 years. There is also evidence that the lake has been relatively fresher and deeper from approximately 2000 b.p. to the present. The Lake Cullulleraine record provides a high resolution analysis of changes in water quality since the onset of river regulation. These changes include an increase in nutrient concentrations in the late 1940s and 1950s and also an increase in salinity in the 1960s and 1970s.

This study also demonstrated that although there is a place for quantitative ecological reconstruction in palaeolimnological research, model validity decreases with system complexity. In the case of the lower Murray River, system complexity is immense due to very variable flows, numerous inflows and outflows, high levels of turbidity and nutrients, and, in most regions, an extensive littoral zone.

Declaration

This thesis contains no material which has been accepted for the award of any other degree of diploma in any university or other institution and to the best of my knowledge and belief contains no material previously published or written by another person, except where due reference is made in the text.

[redacted]

Jennie Fluin

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Chapter one - The lower Murray River

1.1 Introduction

The following discussion describes the geographic and historical setting of the lower Murray River region, south eastern Australia. The physical features of the region including location, geomorphology, vegetation, and water quality are briefly outlined. A human history is then chronicled, including information about when humans first arrived in the region and what were the likely and known impacts of their use of the basin. This information is then placed in the context of a summary of current ecological issues in the basin, including a synopsis of palaeoenvironmental research.

1.2 Geographical setting

1.2.1 Location

For the purposes of this study, the lower Murray River is defined as the section of river downstream of Wentworth, or downstream of the Darling River confluence, to the river mouth. However, due to a paucity of information specific to this region, and because upstream environmental modifications have important implications, descriptions of land use and economic data tend to be drawn from the entire basin.

The lower Murray River forms part of the greater Murray - Darling Basin (covering approximately 1 061 469 km^2) incorporating both the Murray and Darling Rivers, and also the Murrumbidgee, Lachlan, and Goulburn Rivers, as shown in Figure 1.1. It is Australia's largest river system and also one of the world's major river systems, ranking fifteenth in terms of length and twenty - first in terms of area (Kurian, 1989). However, despite the large catchment size, discharge rates are very low with a mean discharge of 0.4 Ml/sec compared to 7.0 Ml/sec for both the Danube in Europe and the Nelson River in North America, rivers with comparable catchment sizes (Crabb, 1993).

The Murray River rises in the Australian Alps about 40 km south of Mt Kosciusko, flowing northwest through New South Wales, Victoria and finally South Australia before entering the terminal lake system of Lake Alexandrina, Lake Albert and the Coorong at the river mouth. The main tributary, the Darling River, can be sourced back to the head of the Condamine River near Warwick in Queensland, and contributes approximately 10% of the system's annual discharge (Shiel *et al.*, 1982). The Great Dividing Range forms the limit of the Basin to the south and the east, and the Wimmera region of Victoria and the Mt Lofty Ranges of South Australia form the boundary in the south west. The Bulloo Basin forms part of the north west boundary while the Chesterton and Warrego Ranges bound the north.

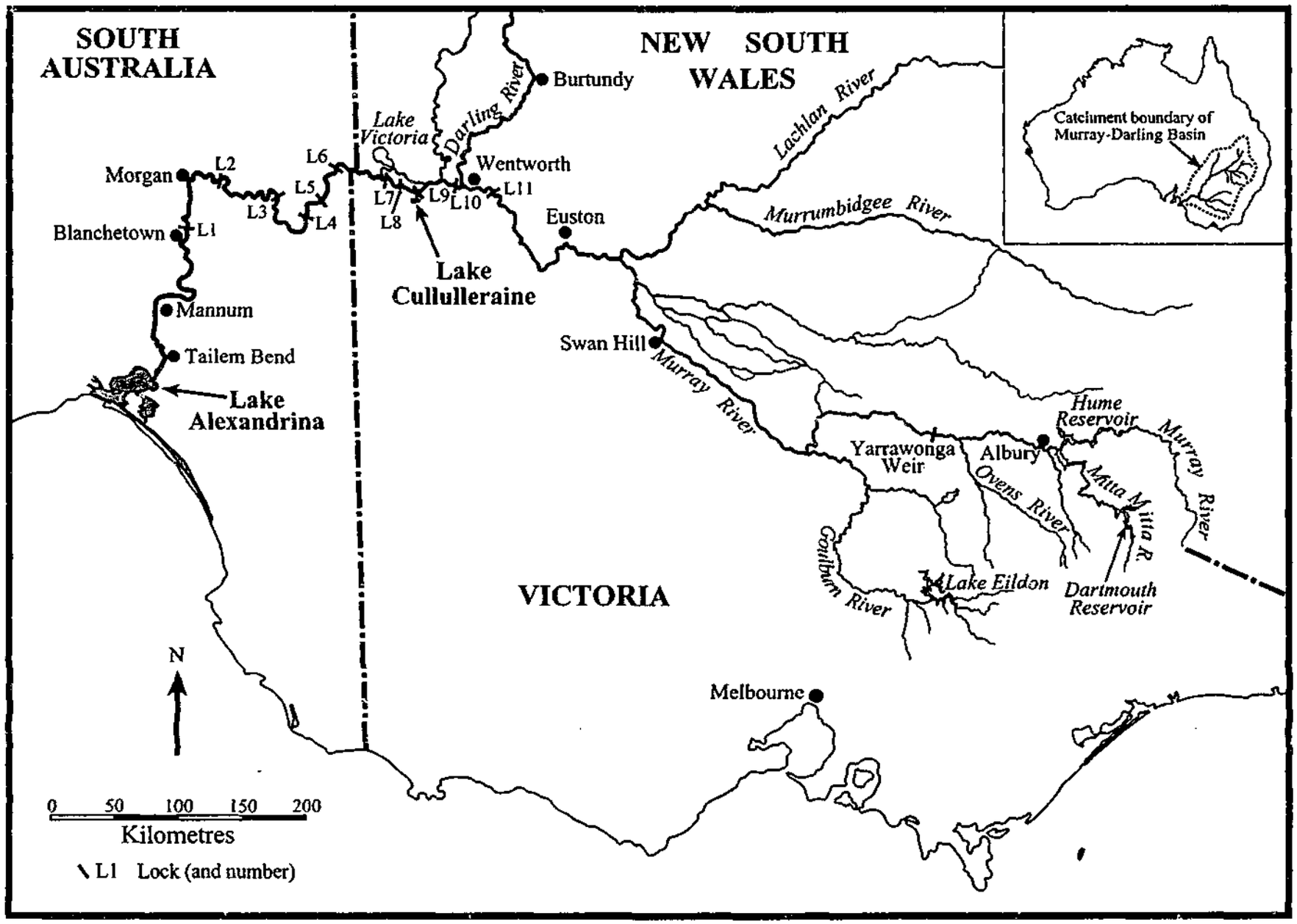


Figure 1.1: *The Murray-Darling Basin, downstream of Lake Hume, showing major locks, weirs and storages. The lower Murray River is defined as being downstream of Wentworth.*

1.2.2 Geomorphology and geology

The lower Murray River can be divided into three distinct geomorphic regions; the mallee trench from Wentworth to Overland Corner, the Murray Gorge from Overland Corner to Mannum and the floodplains from Mannum through to the beginning of the river mouth at Wellington. From the junction of the Darling River to Overland Corner in South Australia, the Murray channel meanders through a five to ten kilometre wide floodplain trench, with remnants of up to four terraces (Gill, 1973). Rutherfurd (1990) found these meanders to have an average wavelength of 4 km. This floodplain contains a complex system of anabranches, wetlands and billabongs and cuts at least 30 m into the late Pliocene sediments. These sediments are named the Parilla Sands Formation, are fluvial - lacustrine in origin and were deposited under Lake Bungunnia (McCord, 1995).

Downstream of Overland Corner the river flows through Miocene limestone (Mannum Formation) and its valley becomes narrow and deeply incised, generally less than 1.5 to 2 km wide and 30 to 40 m deep (Gill, 1973). The Mannum Formation was formed in the late Eocene to early Pliocene when transgression of the basin by the sea saw a variety of fossiliferous limestones and other marine sediments deposited (McCord, 1995). This section is often called the Murray Gorge and is characterised by long straight reaches with almost vertical cliffs (Eastburn, 1990b). These cliffs become much less steep as the river reaches Mannum and the valley begins to widen again, creating a natural plethora of wetlands, most of which have been drained in the last 100 years (Twidale *et al.*, 1978).

The gradient of the lower Murray is extremely low, ranging from a drop of 5 cm/km at Mildura to less than 2 cm/km near the Murray Mouth. In comparison to other sections of the Murray channel, the lower region has been relatively geologically stable for millions of years (Rutherfurd, 1990). There is an extensive sequence of palaeochannels throughout the lower region with a meander wavelength of up to 3.6 km and a width of 1.5 km (Bowler, 1978). Schumm (1968) estimates the migration rate of the channels to have been approximately 15 cm/year, with discharges up to 460% greater than those of the modern river channel.

1.2.3 Vegetation

The lower Murray River passes predominantly through open woodland and mallee vegetation. The most common tree species in the open woodland sections are *Eucalyptus camaldulensis* (River Red Gum) and *Eucalyptus largiflorens* (Murray Black Box), while the most common mallee trees include *Acacia stenophylla* (River cooba) and *Casuarina distyla* (Shrub Sheaoak). The understorey in open woodland is dominated by herbaceous vegetation types, including a wide variety of Asteraceae (daisies), Poaceae (grasses) and Cyperaceae (sedges). Common mallee understorey includes *Cyperus gymnocaulos* (Spiny sedge), *Enchylaena tomentosa* (Ruby saltbush), *Morgania floribunda* (Bluerod), and *Muellenbeckia cunninghamii* (Lignum) (Carnahan, 1990).

In the broader trench section from Wentworth to Overland Corner, River Red Gums tend to dominate the riparian vegetation with Murray Black Box on the flood terraces, while in the gorge section, most of the surrounding flat floodplains are relatively bare due to livestock grazing. Downstream of Mannum there is a return to eucalypt dominance (but not Black Box) interspersed with salt bush and associated vegetation types (Noble 1990 and Bren 1990).

There has been massive vegetation clearance along the entire floodplain of the lower Murray River in the last 200 years (Walker, 2000). Early efforts to clear mallee scrub to provide agricultural holdings proved fruitless as the soil and climate were not conducive to the growing of crops. In most regions the native mallee vegetation has not returned to its original condition and the land has instead been invaded with a mixture of exotic weeds and more salt tolerant bushes and shrubs (DENRb, undated). The effects of this large scale clearance are discussed further in section 1.3.2.2.

Complementary to vegetation clearance has been the planting and subsequent propagation of introduced species. For example, willows (*Salix* spp.) were planted to stabilise levees that were constructed to reclaim swamplands for agriculture. Walker (1985) states that willows now dominate the riverbank in some areas, particularly in the Mannum region, and have displaced the native River Red Gums.

1.2.4 Limnology

Regular water quality monitoring of the Murray River commenced in 1934 at Murray Bridge in South Australia. Comprehensive water quality monitoring, involving sampling from 35 sites, only commenced in 1978, and biological monitoring of macroinvertebrates and phytoplankton was initiated in 1980 (Mackay, 1988). The water quality measures of most concern to both managers and users of lower Murray River water are salinity, turbidity and nutrients.

Along the lower Murray River there are distinct spatial and temporal differences in water quality. As a general rule, water quality declines with distance downstream but some parameters are notably dynamic, reflecting the changing catchment land use and geomorphology of the channel. Table 1.1 summarises salinity (measured as electrical conductivity - EC), turbidity and total phosphorus (TP) data for nine sites downstream of Wentworth that are monitored regularly by the Murray - Darling Basin Commission (MDBC). The data is averaged over a 12 month period from 1994 to 1995.

Table 1.1 Summary EC, TP and turbidity data for the lower Murray River over a 12 month period (1994-1995). DS = downstream. Source: Murray Darling Basin Commission

Site	EC (µS/cm)			TP (µg/L)			Turbidity (NTU)		
	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean
Lock 9	252	500	**366**	29	106	**62**	8	99	**29**
DS Rufus Junction	330	557	**438**	39	170	**68**	8	56	**28**
Lock 5	337	707	**537**	32	440	**67**	10	53	**24**
Lock 3	469	887	**692**	18	155	**58**	3	50	**23**
Waikerie	420	1030	**762**	26	81	**52**	7	43	**19**
Morgan	412	1160	**805**	5	124	**50**	4	43	**16**
Murray Bridge	582	1230	**912**	39	93	**63**	7	38	**19**
Tailem Bend	685	1290	**945**	14	1310	**96**	6	30	**17**
Lake Alexandrina	655	1190	**908**	5	210	**96**	4	78	**30**

Within this section of the Murray River salinity increases with distance downstream reaching a maximum annual mean of 945 µS/cm at Tailem Bend. Total phosphorus also increases downstream, although there is a small decrease between Lock 3 and Morgan. Interestingly, turbidity appears to decrease downstream before rising again in Lake Alexandrina. This is probably due to Lock 9 being the closest site to the Darling River confluence which has much higher turbidities than the Murray River. As the waters from the Darling River become more dilute downstream of the confluence, turbidity would therefore decrease. It is likely that the subsequent increase in turbidity in Lake Alexandrina is due to characteristic high wind stress conditions across the lake. A further reason is that turbidity may increase as a result of evaporation (MacKay, 1988).

Other water quality variables that are pertinent to this study, and which are regularly measured in the Murray River, include ionic composition and pH. Streamflow is also an important variable and is discussed in detail in section 1.3.2.2. The major anions in the lower Murray River are chloride (between 60 and 70%) and bicarbonate (between 20 and 25%). The major cations are sodium (between 50 and 60%) and magnesium (between 20 and 25%). Although the proportion of these ions change along the course of the River, within the lower Murray River they remain relatively stable (Shrafron *et al.* 1990).

Compared to other measured parameters, pH is more stable over the long term with values generally ranging between 7.5 at Lock 9 and 8.5 at Lake Alexandrina. Data collected between 1978 and 1986 showed median pH values for the entire stretch of the lower Murray River range between 7.7 and 8.0 (MacKay *et al.*, 1988). Only during extreme events such as flooding or prolonged droughts are pH levels likely to fall below 7.0 or rise above 9.0.

1.3 Human occupation of the lower Murray River region

1.3.1 Indigenous populations

1.3.1.1 History

Archaeological research is increasing our understanding of Aboriginal migration into the Murray-Darling Basin, with new discoveries leading to earlier estimates of the date of first occupation. Presently it can be said

with confidence that people have lived in the basin for at least 40 000 years (Bowler, 1976). There are more than 10 000 Aboriginal sites in the basin, particularly adjacent to the rivers and on the floodplains, with evidence derived from sources such as shell middens, quarries, rock shelters, stone tools, open camp sites, and burial and ceremonial grounds (Jones, 1988). Recent excavations at Lake Mungo suggest, with some debate, that Aboriginal people lived on the shores of the Willandra Lakes at least 60 000 years ago (Thorne *et al.*, 1999). The remains found at Lake Mungo suggest that early Aboriginal communities fished for *Macquaria ambigua* (Golden Perch) and collected large quantities of Murray River mussel. More recently, when water levels in Lake Victoria were lowered for maintenance in 1994, more than 10 000 Aboriginal burials were revealed, all dating to between 3000 and 4000 years ago (Calibrated ^{14}C dates) (MDBC, 1998).

The shores of Lake Alexandrina and the Coorong appear to have been a permanent home for Aboriginal people for at least 1000 years (Bernt *et al.*, 1993). The Yaraldi, part of the Ngarrindjeri Aboriginal group (which had the highest population density for any Australian Aboriginal group (Jenkin, 1979)), were the most populous community and continue to inhabit the shores today, predominantly at Point McLeay and Murray Bridge. Ethnographic evidence shows a great reliance and subsequent respect for the lower Murray River environment with many myths and stories associated with the changing dynamics of the river. Historically, food was abundant with both marine and freshwater fish, as well as large fauna that would have been attracted to the shores for drinking water (Bernt *et al.*, 1993).

Within 20 years of non - indigenous people first exploring the lower Murray River region the Aboriginal population had been decimated, both because of introduced diseases and violent confrontations (Walker, 1996). Women were bought or simply taken away from their communities and 'given' to the early settlers. Understandably this led to much hatred and antagonism between the ethnic groups and also initiated a breakdown in Aboriginal clan structure.

1.3.1.2 Interaction of Aboriginal people and the river environment

The River provided Aborigines with a permanent water supply, food supply, locations for ceremonies and raw material for shelter, clothing, tools, weapons and transport (Eastburn, 1990). The level of environmental effects of Aboriginal interaction with the River is a highly contentious issue. There is argument for Aboriginal burning, perhaps leading to an increase in surface runoff and turbidity, but evidence is scarce and not as conclusive as in other regions of Australia (Thorne *et al.*, 1999). There has been suggestion that Aboriginal burning in the Barmah Forest influenced the regeneration of river red gum and the distribution of reed beds (Pressey, 1990), but again this is speculative. Impacts cannot be assessed until more is known about Aboriginal way of life, and this may not be possible because of the destruction of many archaeological sites along the river floodplain. However, it can be concluded that the impact of Aboriginal occupation in the lower Murray River was minimal in comparison to the impact of non - indigenous people.

1.3.2 Non - indigenous settlement

1.3.2.1. History

The Murray River was first sighted by non - indigenous Australians in November 1824 by a group led by Sir Charles Sturt. River exploration in the early 1830s concentrated on charting the channel, with particular hopes of finding a suitable seaport that would enable more efficient transportation and communication between Australia and Europe. However, arrival at the mouth proved to be an anticlimax as Encounter Bay was dangerous as a port and the mouth itself unnavigable because of the unpredictability and verociousness of the breakers (Davis, 1978).

Land settlement began as early as 1835 with cattle stations established near Albury and Swan Hill (Lawrence and Kinross Smith, 1975). Settlers began to rapidly descend on the lower Murray River region, originating from Adelaide, Sydney and Melbourne, eventually converging when arable riverside land became scarcer. Transport played a major role in the creation of new stations and towns along the Murray and Darling Rivers, and by 1850 there was a comprehensive paddlesteamer industry in place (Kenderline, 1993). In total, more than 6400 km of the Murray-Darling system was opened up to trade and travel by the paddlesteamer. As key towns such as Murray Bridge, Mildura, and Albury/Wodonga became established, rail networks were developed providing links to the cities and ports (Eastburn 1990a).

Following the initial enthusiasm for settlement along the river banks, migration stabilised until the 1880s when irrigation schemes flaunted the possibilities of fortunes to be made on food production. Subsequent to impressive advertising schemes, vast tracts of land, with Mildura as an epicentre, were cleared and developed for horticultural use (Hallows and Thompson, 1995). The next big wave of settlement came in the post World War periods in the early 1920s and late 1940s when extensive soldier settlement schemes were established, particularly at Morgan, Berri, Cadell and Renmark (Menzies, 1983).

1.3.2.2. Interaction of non - indigenous people and the river environment

The relationship between non - indigenous people and the lower Murray River environment has been well documented. The two principle mechanisms by which non - indigenous people have interacted with the river have been through land use changes in the catchment area and by regulating the flow of the river channel. The impacts of these activities, and others, are discussed in detail in section 1.4. A brief history is provided below.

1.3.2.2.1 Changes in catchment land use

Massive land clearing has taken place since non - indigenous settlement, with a diverse native habitat of eucalypt woodland and shrubland being replaced with crops, pastures and urban areas (Walker *et al.*, 1993).

At least half the Murray - Darling Basin's native vegetation cover has been removed, with many species, particularly those of native grasslands, being lost (Graetz *et al.*, 1995). Beale and Frey (1990) estimate that this devegetation equates to over 20 billion trees with more than 30% of the total land surface of the basin being cleared. The growth of the paddlesteamer trade played a significant role in this vegetation clearance, with large-scale felling of River Red Gums to satisfy the fuel requirements of the boats. Holmes (1948) found that this felling encouraged the growth of the wood-cutting industry and also helped to initiate large scale soil erosion along the river banks.

The section of river downstream of Mannum is one of the most extensively cleared areas in the lower Murray River region. Smith and Smith (1990) state that the River here was once flanked almost continuously with wetlands but since the 1880s most of these have been drained. Levee banks were constructed, creating a large irrigated area for intensive settlement and grazing. By 1929, development works had drained nearly all of the wetlands along the River between Wellington and Mannum, with river flows confined to the main channel (DENRb, undated). As a consequence, the River is now densely fringed by Weeping Willow (*Salix babylonica*) with Red Gums becoming increasing uncommon and virtually absent below Murray Bridge (Walker *et al.*, 1993). Willows crowd out other plants due to their extensive shading properties and, unlike native trees, they shed all their leaves in autumn which decompose in the river channel, adding to the already excessive nutrient loading and turbidity.

The major land uses in the Murray - Darling Basin, as of 1983, are outlined in table 1.2.

Table 1.2: Major land uses in the Murray - Darling basin (Garman, 1983)

Land use	% of total area
Grazing	82.2
Conservation purposes	8.3
Crops	4.4
Forests	3.1
Urban	2.0

Grazing is by far the dominant land use. All grazing animals are selective, preferring the taste of some plants over others, which has led to a further change in vegetation structure. Forested areas are generally not protected and many are still subject to logging concessions. Urban areas are currently stable and in some areas actually decreasing due to a degeneration of small country towns and the trend for migration to larger population centres.

1.3.2.2.2 River Regulation

Australia's variable rainfall means that the Murray - Darling River system is subject to considerable annual variability of flows (McMahon *et al.*, 1992). In the early 1880s Alfred Deakin (Australian Prime Minister)

proposed that the only way to colonise and develop the northern part of Victoria was to install an irrigation system that utilised the waters of the Murray River (Hallows and Thompson, 1995), and the need for regulation of these highly variable flows soon became apparent (Sinclair, 2001). A Royal Commission was set up to examine proposals for irrigation and Deakin became interested in the Canadian Chaffey brothers who had experience with irrigation schemes. George Chaffey arrived in Melbourne in 1886 and proceeded to set up irrigation networks in both Mildura (Victoria) and Renmark (South Australia) in 1887. Davis (1978) writes that it is quite clear from the early writings of this time that the River itself was taken for granted with no thought given to the likely effects on the downstream environment.

Following regular disagreements about water usage and rights, the State Governments of New South Wales, Victoria and South Australia appointed a further Royal Commission to investigate both the problems and possible solutions. The Commission was established in May 1902 and quickly recommended the designation of a permanent commission to control the river (Industrial Australian and Mining Standard, 1994). After a further eleven years of protracted investigations, an interstate working committee, comprised mainly of engineers, concluded that for irrigation to continue and flourish, storage basins were needed to counter the adversities of highly variable flows. The scheme was eventually accepted and the River Murray Waters Agreement was ratified in 1915 by Acts of Parliament in both Federal and State Governments (Sinclair, 2001). This scheme recommended the development of storage at Lake Hume and Lake Victoria, the construction of 26 locks and weirs extending from Blanchetown to Echuca, and the construction of 9 locks and weirs on the Murrumbidgee River. Lake Hume and Lake Victoria were deemed necessary for the long term storage of water to counter the effects of dry years, and the lock and weir system was thought necessary to enable permanent navigation and also to provide steady, regulated pools for irrigation (Menzies, 1983). Table 1.3 outlines the details of structures downstream of Lake Hume, including completion date and storage capacity, and their locations are shown in Figure 1.1.

Since the initial completion of the Hume Dam in 1936, a continuous flow has been maintained throughout the length of the Murray. Without storages and regulation the Murray would almost certainly have ceased to run during the droughts of 1938-39, 1944-45, 1967-68 and 1982-83 (Thompson, 1994). Under regulated flow conditions it takes approximately five weeks for water to travel from Hume Dam to the South Australian off takes (pipelines supplying principally domestic water to South Australian towns). Of the water that would have reached the sea under unregulated conditions, the majority is now diverted for irrigation and domestic purposes. At the Murray Mouth, outflows to the sea have been reduced by 8100 GL/year which is almost two thirds of the natural outflow (Close, 1990). One of the most important ramifications of this reduced flow is that the River is now in a state of drought (as defined by river levels) for more than sixty one in every hundred years, compared to five years per hundred under natural conditions (MDBMC, 1995).

Table 1.3 Details of the major storages downstream of, and including, Lake Hume

Storage Name	Completed	Capacity (Ml)
Lake Hume	1936/1961	3 038 000
Lake Victoria	1928	680 000
Yarrawonga Weir	1939	120 000
Torrumbarry Weir	1924 / 1996	38 000
Lock 15	1937	38 600
Lock 11	1927	37 000
Lock 10	1929	47 000
Lock 9	1926	32 000
Lock 8	1935	24 000
Lock 7	1934	13 000
Lock 6	1930	35 000
Lock 5	1927	39 000
Lock 4	1929	31 000
Lock 3	1925	52 000
Lock 2	1928	43 000
Lock 1	1922	64 000
Goolwa Barrages	1940	1 974 000

Thomson (1994) states that from a basin wide perspective, there has been a shift from a steady variation in mid range flows under natural conditions to a dominance of very low flows and occasional high flow events. For instance, while large floods, such as that of 1956, are reduced only to a limited extent, the small to medium sized floods have been virtually eliminated from the channel. Prior to regulation there were high flows in winter and spring, followed by low flows in summer and autumn. Post regulation, higher than natural flows are maintained in summer and autumn to meet peak demands for irrigation and urban supply, and flows in winter and spring are correspondingly lower (Walker, 2000). Effectively, there has been a seasonal flow reversal.

However, despite the obvious massive ecological consequences, river regulation is still judged to be an essential engineering device and will probably only increase in intensity rather than decrease (Sinclair, 2001). During a normal year Adelaide draws approximately 55% of its water from the Murray River and during droughts this increases to more than 90% (DENRa, undated). Without the present system of river regulation, the population of Adelaide, and many other cities and towns in the Murray - Darling Basin, would be considerably smaller. More than 40% of the national total GDP derived from agriculture comes from the Murray - Darling Basin, a figure that would be drastically reduced if irrigation ceased (GHD, 1992). This then leads to the current dilemma facing the Murray - Darling regulatory authorities of exactly how to allocate

these water flows equitably both between States and between competing industries, and in a sustainable manner that produces the least ecological disruption.

1.4 Current ecological issues in the lower Murray River

1.4.1 Introduction

The ecological issues that are deemed to be the most important, both in context of management of the river and also in terms of the focus of this study, are briefly summarised. These issues include river salinisation, eutrophication, high turbidity and wetland degradation.

1.4.2 River salinisation

Accounts of Captain Sturt reaching the Darling River in 1829 and finding the water too salty to drink are so frequently quoted that the story has now acquired legend status (Gill, 1970). The Murray River has always experienced natural inputs of salt, both from groundwater and tributary flow, due to the leaching of saline soils and the weathering of rocks. A further historical cause of salinity during drought times in the lower Murray was the intrusion of seawater into Lake Alexandrina and extending upstream as far as Mannum (Davis, 1978). The construction of five barrages (between 1935 and 1940) in Lake Alexandrina countered these mechanisms by preventing seawater from flowing into the River during periods of low flow (Cann *et al.* 2000).

The salinity levels of the lower Murray River generally increase with distance downstream. Salinity has been recorded for 50 years in the lower Murray River, with the most continuous records at Morgan and Murray Bridge. It is evident that EC has fluctuated widely over that time with most variation arising through changes in river flow. This wide variation over short time periods (often monthly or even weekly) makes it difficult to determine an underlying trend (Close, 1990).

[illegible], however, that saline inputs to the lower Murray River have increased since non-[illegible]ement. Changes to vegetation cover, particularly the removal of native grasses, shrubs and tree[illegible]nged the natural water balance (Williamson, 1997). Replacement of deep rooted native vegetation with shallow rooted annual crops and pastures have resulted in increased recharge to the groundwater table and as the groundwater level rises, naturally occurring salts are brought to the surface and evaporated. The legacy of this activity is salt scalding which almost always permanently renders the land unusable for conventional agriculture, and also increased saline runoff and groundwater flow to the river, resulting in increased river salinity levels. Allison and Schonfeldt (1989) conclude that dryland salinity from

the Victorian Riverine Plains will contribute an increase of up to 140 µS/cm at Morgan over the next 50 to 100 years. In addition, irrigation greatly exacerbates the impact of salinisation by disrupting the water balance even further and thus increasing groundwater recharge.

Natural variability in riverine EC over the long term, combined with difficulties in trying to detect changes over the short term, have led to disagreement over the level of importance that should be attached to controlling salinity levels (MDBMC, 1999). The native flora and fauna of the river and its floodplain are adapted to fluctuating salinity levels and most can survive extremes during periods of drought. Periods of high EC are essentially a natural state, with species present that are competitively advantaged by these saline events. Concern arises, however, when the effects on introduced species are considered, particularly horticultural crops such as fruit. Many of the citrus and grape plantations of the Sunraysia and Riverland regions are sensitive to even small changes in EC and river salinities are frequently above threshold levels at which the crop yield starts to decline. Anthropogenic activities, such as native vegetation clearance, have led to an increase in saline inputs to the river system. However, river regulation has acted to control natural variability by reducing the effects of drought induced increases in salinity. Theoretically, these two factors may counter each other. This then raises a multitude of philosophical issues, particularly the one of being overly concerned about increasing salinity levels when the natural biota is already well adapted. Obviously, for managers of the basin, this point doesn't rank highly compared to the issues of loss of income due to crop failure.

1.4.3 Eutrophication

Eutrophication refers to the process of enrichment of waters with nutrients, particularly phosphorus but also nitrogen and other elements. Human activity has greatly exacerbated the input of nutrients to the river system by discharges from diffuse sources such as runoff from agricultural land where there is a high fertiliser use, and also point sources such as sewage effluent, industrial activity, fish farms, drainage from irrigation lines and stormwater runoff from urban areas. It has been estimated that between 25% and 75% of the nutrient load entering the Murray - Darling river system originates from diffuse sources, with agriculture being a major contributor (Raisin, 1996). Thoms (1995) states that there are presently more than 50 urban centres that discharge effluent into the Murray River and its tributaries and that the number and magnitude of these centres is likely to expand.

Perhaps the most serious of the problems arising from elevated nutrient levels is the growth of Cyanobacteria throughout the basin. The most common Cyanobacteria in the Murray River are *Anabaena*, *Microcystis* and *Nodularia* which, under favourable conditions of high nutrient availability, low flow and warm temperatures, reproduce at a very high rate to form blooms that can dominate the phytoplankton along large stretches of river (Baker *et al.* 2000). Blue green algae received wide publicity in 1991 when an enormous toxic bloom of

Anabaena circinalis in the Darling River, stretching over 1000 km in length, caused the NSW Government to declare a state of emergency. These algae are not purely an artifact of recent increases in nutrient loading due to fertilisers, manufacturing and effluent discharges. It has been documented that as early as 1878 a bloom of blue-green algae caused the death of livestock that were drinking from Lake Alexandrina and it is highly likely that blooms also existed prior to non - indigenous settlement (Francis, 1878). A study of Holocene sediments from the Cooke Plains embayement, a former extension of Lake Alexandrina, showed evidence of blue green algal remains approximately 7000 years ago (von der Borch and Altman, 1979). As well having an adverse affect on livestock and aquatic flora and fauna, toxins produced by blue - green algae can cause liver damage, digestive problems and nervous system disorders in humans (GHD 1992).

1.4.4 Turbidity

Turbidity is a measure of water clarity encompassing the presence of abiotic suspended material and also phytoplankton and zooplankton. These suspended sediments may also carry nutrients and heavy metals. Turbidity in the lower Murray River is predominantly influenced by the effects of the Darling River, which carries large quantities of fine clays. When river velocity is reduced, these suspended sediments lose buoyancy and settle out, resulting in reduced storage capacity behind weirs and barrages. An estimated 800 000 m^3 of sediment accumulates in Lake Hume every year (Shafron *et al.*, 1990).

High turbidity has an adverse effect on aquatic organisms from all levels of the food chain as it reduces visibility and photosynthetic capacity. Turbidity can also cause mechanical problems on water reticulation and treatment equipment, making treatment an expensive and lengthy process (Sinclair, 2001).

South Australia has turbidity records spanning nearly 50 years but, as with salinity, because of great variability it is not possible to determine a meaningful long term trend (MacKay *et al.*, 1988). In reference to palaeolimnological evidence, Thoms *et al.* (1999) infer increased clay loads within the river since the onset of non – indigenous settlement. This is most probably due to massive land clearance, which, combined with the introduction of hard hoofed fauna and rabbits acting to destabilise soil structure, has increased the sediment loads in runoff.

1.4.5 Wetland degradation

River regulation has also interrupted the flow of natural waters entering floodplains and wetlands, and in some instances, these flows have ceased altogether. A further dramatic impact on wetland environments has been drainage for agricultural and urban uses, with more than 50% of the wetlands being converted to urban development and agricultural holdings (ANCA, 1995). Pressey (1990) indicates that there are currently more than 7000 wetlands between the Murray Mouth and Lake Hume, covering more than 222 000 hectares.

Wetlands (backwaters, billabongs, marshes, swamps, and lakes both natural and constructed) are places of intense biological activity. Wetlands are important for maintaining biological diversity, and providing a complexity of habitat types necessary to satisfy the breeding and refuge requirements of dependent birds, fish, amphibians, reptile, mammal, and aquatic invertebrates (McComb and Lake 1990). As floodwaters pass through wetland areas nutrients and heavy metals are removed through biological, chemical and physical processes. Wetlands can therefore improve water quality and reduce the incidence of algal blooms. They also act to regulate peak floods, releasing flood waters gradually, and during drought providing refuge for wildlife and stock (Sharley and Huggan, 1994). Maintenance of the natural flooding and drying regime is of particular importance for wetlands, as flooding triggers breeding and regeneration of riverine biota.

The process of wetland ephemerality is crucial to the habitat and lifecycle of native fish. At least half the native fish species in the River Murray are migratory, with free movement provided by linked wetlands and the main channel being fundamental for their survival (Sinclair. 2001). Barrages and weirs, and the isolated nature of the now permanent wetlands, have greatly restricted this migration, but somewhat ironically, favour the lifecycle needs of introduced fish. Additionally, regulation has led to a general decline in macrophyte abundance in many wetlands in the Murray - Darling Basin (Ogden, 1996). Macrophytes play an essential role in filtering nutrients and heavy metals, and also in stabilising the sediment, thus reducing turbidity.

1.5 Management of the lower Murray River

Regulation of control over river management was omitted from the 1901 Australian Constitution because of disagreement between the States claiming ownership of the Murray River. Public meetings were held soon after Federation to increase the Federal Government's awareness of the need for an agreement on river management and water sharing. A Royal Commission and negotiations between the Commonwealth, New South Wales, Victorian and South Australian governments finally produced an agreement on river management and water sharing in 1915, and the River Murray Commission was established in 1917 (Industrial Australian and Mining Standard, 1994). This Commission oversaw the construction of infrastructure associated with river regulation amid increasing calls to improve water quality. Eventually, this need required an organisation with broader responsibilities and greater input to the management of the catchments surrounding the rivers. Thus, the Murray - Darling Basin Commission was established in 1988.

The Murray - Darling Basin Commission (MDBC) is responsible for coordinating the efforts of all the governments and communities involved in the management of the Basin. The MDBC reports to the Murray Darling Ministerial Council which brings together land, water and environment ministers from the various governments in the Basin. The commission is responsible for distributing the Murray River waters to the states of New South Wales, Victoria and South Australia, and advising the Ministerial Council on natural resources management issues. The Commission consists of more than 20 working groups drawn from government departments, universities, and private and community organisations.

1.6 Palaeolimnology and the lower Murray River

1.6.1 Introduction

It is surprising how little is known about the natural variability of the Murray – Darling Basin. The following statements summarise this general lack of knowledge:

"...it is astonishing how little is known of the ecology of the old river." (Walker, 1996).

" Defining precisely the natural, or at least pre - regulation condition of a wetland in terms of flora and fauna is usually very difficult because of the general lack of records." (Pressey, 1990).

" An historical perspective on the impact of carp is lacking because of the lack of historical ecological data on Australia's inland rivers " (Roberts *et al.*, 1996).

" ...we have been closely monitoring the numbers of these organisms in the Murray, but again the record is highly variable with no clear evidence of a trend in the frequency or duration of algal blooms " (Mackay, 1990).

Immense temporal and spatial variability in ecological parameters has led to difficulties in determining the degree of change since non - indigenous settlement. For instance, although it is assumed that the River has always been naturally eutrophic and that salinity has historically fluctuated to a greater degree than it does presently, this is mainly based on anecdotal evidence.

There is a long overdue need for extensive palaeolimnological research in the Murray - Darling Basin. It is now common place (and actually forms part of water monitoring guidelines) in European Economic Community countries for hindcasts of water quality to be established before monitoring programs are initiated (Bennion *et al.* 2000). Palaeolimnology offers enormous benefits to all research on contemporary water quality dynamics and these are discussed in detail in chapter 2.

1.6.2 Review of palaeolimnological research in the Murray - Darling Basin

This section reviews what is currently known about the past limnology of the Murray - Darling Basin. As well as being limited to discussion of water quality and ecological changes, it is also confined to the upper Murray River and Murrumbidgee River systems, as this has generally been the focus of research. Although there have been numerous palaeoenvironmental studies in the lower sections of the river, these have generally

concentrated on the determination of past climate fluvial regimes and biostratigraphy, rather than establishing palaeo - aquatic ecology and associated water quality.

Thoms (1995) extracted eight cores from the Barmah Forest region of the Murray River floodplain, downstream of the Yarrawonga Weir, and examined changes in sedimentation rates, sediment mineralogy, and modes of sediment transport. Thoms (1995) found that post non - indigenous settlement sediment accumulation rates were an order of magnitude higher than pre non - indigenous settlement rates. This compares well with earlier findings by this author (Thoms and Walker, 1990) which demonstrated that annual sediment loads in the River Murray have increased by 114% in association with changed catchment land use since the 1970s.

Ogden (2000) conducted a study into the palaeoecology of seven billabongs in the upper Murray River region and inferred historical submerged aquatic macrophyte abundance by examining fossilised Cladocera and Chydoridae assemblages. Reconstructions were based on the theory that there is a positive relationship between the two ecological indicators (see Hann and Karrow, 1993; and Manca and Comoli, 1995), with greater Chydoridae abundance generally indicating more extensive macrophyte coverage. Ogden's (2000) analysis showed that there was likely to have been a decline in the abundance of submerged macrophytes after the late 1800s. Ogden interprets this change as a result of increased shading caused by extensive land clearance leading to increased turbidity levels in the river. He goes further to suggest that the consistently depressed macrophyte abundance has important implications for management of the billabongs as some of these sites have been designated as being control or reference sites for monitoring the health of the river system (Thoms *et al.*, 1999), and they are clearly not currently reflecting the natural biota.

Reid (1997) undertook a palaeolimnogical examination of two adjacent billabongs situated on the floodplain of the Goulburn River in Victoria. A single core was taken from the deepest section of each billabong for diatom and pollen analysis. The cores were radiocarbon dated to between 10 000 and 8 000 B.P. The diatom sequences are similar until approximately halfway through the recorded non - indigenous phase of settlement. Prior to the arrival of non - indigenous Australians, the diatom records suggest a gradual reduction in pH over time. Reid (1997) interprets the decrease in pH as being due to a build up of humic acids derived from surrounding myrtaceous vegetation (as indicated by the pollen records). Soon after the arrival of non - indigenous Australians, the diatom - derived pH levels began to increase, probably resulting from a combination of the removal of much of the floodplain vegetation and also increased inputs of base cations from the local catchment. Reid (1997) suggests that these base cations are likely to have increased both because of increased soil erosion and increased groundwater inputs.

Approximately 100 years ago there was a divergence in the diatom records of the two billabongs. While the sequence in one billabong remained relatively constant, there is an increase in more alkaline taxa in the record of the other billabong. This change is interpreted as a further increase in pH, and is supported by contemporary

water chemistry data. Other minor and recent changes in both billabongs provide evidence for increased eutrophication due to changing land use. Reid (1997) concludes that palaeolimnological analysis demonstrates that agriculture had a dramatic impact on the Goulburn floodplain billabongs. He also argues that there is no equivocal evidence for river regulation having had a direct impact on the ecology of the billabongs.

Still focusing on the upper River Murray region, Tibby (2000) researched the palaeolimnology of Burrinjuck Reservoir, located on the Murrumbidgee River. As part of the study, Tibby produced a diatom - total phosphorus transfer function (method described in section 2.3.4) and was able to reconstruct the TP of the lake with a relatively high degree of confidence. This reconstruction highlighted four key time periods in the reservoir's history. Firstly, from the time of lake filling in 1925 through to the late 1930s, TP and silica were both very low, with an average of < 20 µg/L of TP and lack of silica to the point that planktonic diatoms readily dissolved in the water column. From the late 1930s to the late 1960s diatom inferred TP increased to approximately 30 - 35 µg/L, which is attributed to extensive land clearance in the catchment and associated agriculture. From the late 1960s through to the mid 1970s the diatom record indicates a substantial increase in TP, up to 45 µg/L, which corresponds to a significant increase in photosynthetic fossilised pigments. This change correlates with an extensive increase in the urban area of the largest settlement in the catchment, Canberra. Lastly, from the mid 1970s to the present day, the diatom based TP reconstruction shows a reversal of this trend, returning to TP concentrations of approximately 30 - 35 µg/L. This improvement is attributed to the upgrading of the Lower Molonglo Water Quality Control Centre (sewage treatment) in 1978. The research of both Tibby (2000) and Reid (1997) shows the tremendous potential for quantitatively reconstructing past environments in the Murray – Darling Basin.

As mentioned previously there is a paucity of this type of research in the lower Murray River region. Although sediment records have been taken from Lake Alexandrina (Barnett, 1993) and behind weirs (Thoms and Walker, 1993), most studies have concentrated on the changing geomorphic and stratigraphic sequences rather than the palaeolimnology. Barnett (1994) has published a low resolution diatom record from Lake Alexandrina, covering the past 7000 years, and this record is reanalysed and expanded upon as part of this study.

There is great potential for valid and beneficial benchmark conditions to be determined from the lower Murray River with relative ease and little cost. However, as Smol (1992) states, the lack of communication and joint study between palaeolimnologists and geomorphologists, and the lack of interaction between these researchers and water quality managers, impedes this process.

1.6.3 Aims of this study in relation to investigating the palaeolimnology of the lower Murray River region

The lower Murray River cannot be viewed as a discrete ecoregion. A thorough palaeolimnological investigation of the lower Murray River would not be complete without information derived from all sections of the river, and suggestions for the establishment of this research are made in Chapter 8. Rather, this thesis provides examples of past limnological conditions of two distinct sections of the lower Murray River, concentrating on the following objectives -

1. To source potential sites in the lower Murray River that are considered to have continuous sedimentation, have been permanently inundated with water, and contain preserved biological (diatom) communities,

2. To examine several temporal intervals in detail, including -

   a) the establishment of benchmark water quality data and / or the scope of variation, for conditions prior to the arrival of non - indigenous Australians,

   b) the determination of the degree of water quality change since the arrival of non - indigenous Australians,

   c) and high resolution assessment of the impacts of river regulation on water quality.

Chapter 2 - Diatoms and palaeolimnology

2.1 The discipline of palaeolimnology

Battarbee (1997) defines palaeolimnology as a science concerned with how and why lakes have changed and are changing through time. Smol and Glew (1992) extend this definition further to include wetlands and rivers. Although as ancient as the science of neolimnology, palaeolimnology has only recently begun to experience popularity and wide interest among aquatic researchers and managers. One of the major reasons for this increased interest is the growing realisation that temporal trends in limnology are complex and not well understood, ranging from the short term (hourly or daily changes) to the long term (changes over the millennia). In Australia particularly, where usable water resources are becoming increasingly scarce and degraded, understanding aquatic ecological dynamics is crucial to successful sustainable management of these systems.

There are three principle ways in which palaeolimnology can contribute to contemporary aquatic ecosystem management (Charles *et al.*, 1994) -

1) The determination of past trends in the absence of other consistent data sources (such as long term physico - chemical records). Few aquatic systems in Australia have been monitored, and even then only in a limited sense, for more than ten years (Harris and Baxter, 1986). This data gives no long term perspective on the changes since non-indigenous settlement, and in most cases, even since the onset of river regulation.

2) The determination of whether the measured trends fall outside natural variability. Where restoration is a stated aim, trends recorded over the past couple of decades need to be put in the context of natural variability before conclusions, and subsequent management decisions, can be effectively made. This ensures that ecological restorative targets are achievable and within the bounds of natural variability. Consideration must also be given to incorporating the effects of climate change when establishing these restorative targets.

3) The determination of the extent to which trends are anthropogenic or natural in origin. For instance, after recent increases in salinity in the Murray - Darling River are compared to historical trends in salinity, anthropogenic influence versus natural salinity variability can be established and even quantified.

For most aquatic systems there is a general lack of historical data that is of sufficient accuracy and antiquity for the natural state of an ecosystem to be defined (Battarbee, 1997). Clark (1990) states that while simple description of the aquatic environment over a contemporary time scale is essential, it is also inadequate for

detecting longer term trends and understanding rates, directions and magnitudes of change in complex systems. Ford (1988) asserts that the major requirements for effective ecosystem management include knowledge of aquatic baseline conditions, identification of the point in time where the system began to change, and an assessment of the range of possible trajectories. The capacity of palaeolimnologists to contribute to the understanding of these requirements is fundamental, and has increased immeasurably over the last two decades (Smol, 1990).

2.2 Principles of palaeolimnology

Anderson (1997) states that lake sediments are natural environmental archives which collect a variety of biotic and abiotic information over a range of timescales. The vast majority of lacustrine environments, incorporating estuaries, reservoirs and billabongs, contain organic muds that have accumulated fossilised biological and chemical remains since the time of lake formation. For many aquatic systems in the Northern Hemisphere this accumulation is continuous, but in Australia and other arid regions, these sediment records may be discontinuous due to inherent ephemerality, which obviously has important implications for site selection. In context of the Murray – Darling Basin, one of the primary reasons for focussing on the lower Murray River was a reduced risk of site ephemerality because of a more temperate climate, in comparison to the Darling or Lachlan Rivers which are located in more arid regions.

Longmore (1986) has shown that sediment deposition is not consistent over the lake bed. Processes such as sediment focusing, whereby sediments accumulate in greater volumes within the deepest parts of the lake basin, occur in the majority of lake systems. Anderson (1990a and 1990b) has found differing diatom productivities between littoral and profundal zones and evidence of sediment trapping within macrophyte beds. Anderson (1986) also states that it is important to realise that differences in sedimentation processes are subject to change over time, perhaps due to macrophyte senescence or changing flow regimes of incoming streams. Unfortunately, for the study of temperate south east Australian aquatic systems, these problems of representativeness are often greatest in shallow lakes where entrainment and mixing of sediments affects a greater proportion of the sediment surface and where relatively large areas are classed as littoral (Anderson and Odgaard, 1994; Reid, 1997).

Timescales of palaeolimnological reconstruction are dependent upon the process of mixing at the sediment - water interface and sediment accumulation rates, and may range from seasonal to millennial intervals. Smol (1992) states that recent advances in sampling techniques, including close - interval sediment sectioning, tape peel adhesions from frozen cores, and resinous impregnations, have meant that palaeolimnologists can attain timescales that concur with timescales of greatest interest to aquatic ecosystem managers. Sediments that are varved or laminated provide reconstructions that are generally viewed more confidently than homogeneous sediments, although as Anderson (1995) argues, this is not necessarily the case, with homogeneity not always being indicative of sediment mixing.

The diagnostic tools utilised by palaeolimnologists include physical indicators (e.g. sediment description), chemical indicators (e.g. geochemistry), and biological indicators (e.g. pollen, chrysophytes, cladocerans, ostracods, diatoms). In addition to these indicators, sediments are usually subject to radiometric analysis (e.g. levels of ^{210}Pb and ^{14}C) in order to date the sequences. Diatoms, frequently forming the mainstay of palaeolimnological studies because of their considerable species diversity and because their taxonomically distinct remains are abundant and well preserved (Battarbee, 1997), are the primary palaeolimnological tool used in this study, and are discussed in depth in section 2.3.

2.3 Review of diatom - based palaeolimnology

Diatoms are unicellular siliceous algae (class Bacillariophyceae) that are ecologically sensitive and widely distributed over a range of environmental conditions and habitats (Round, 1981). Diatoms are excellent palaeolimnological indicators because they are taxonomically distinct, are usually abundant, diverse, and are well preserved in sedimentary profiles (Dixit *et al.*, 1992; Charles *et al.*, 1994). Battarbee (1997) states that diatoms are perhaps the most useful and powerful indicators of water chemistry as their ecological optima and tolerances can be quantitatively defined. As such, diatoms are the most commonly used limnological indicators in palaeoecological studies (Moser *et al.*, 1996).

Although diatoms are ecologically sensitive to a wide range of parameters, their relationships to pH, TP and salinity are perhaps the best known and most established. To date, some of the most comprehensive investigations of the autecology of diatoms are the projects that examined the effect of changing pH on diatom communities: PIRLA-I and PIRLA-II (Palaeoecological Investigation of Recent Lake Acidification - Charles and Whitehead, 1986, and Charles and Smol, 1990) in eastern North America, and SWAP (Surface Water Acidification Project - Stevenson *et al.*, 1991) in Europe. Many of the recent advances in palaeolimnological research have been driven from the funding and subsequent success of these programs.

There have been many extensive reviews of methods used in diatom based palaeolimnology (Battarbee 1984, Anderson 1997, Birks *et al.* 1990, Smol 1990 and Birks 1998). Those aspects which are pertinent to this study are outlined to provide a methodological rationale. The review is divided into two categories - non Weighted Averaging approaches and the Weighted Averaging approach. Many present day studies adopt Weighted Averaging techniques as they are regarded as being the most refined tool available for diatom based palaeolimnology (Birks, 1999).

2.3.1 Non Weighted Averaging techniques

Ecological profiles of individual taxa can be interpreted qualitatively based on known ecological characteristics of the taxa. This information can be in the form of observed presence / absence limitations in different chemical environments, and also regional or geographic information. Diatom taxa can also be

combined into ecological groups, usually based on the more commonly measured parameters of pH, TP, salinity, and tolerance to organic pollution. In general, most of these categories are created arbitrarily by dividing an ecological continuum into segments and then assigning taxa to a category on the basis of available ecological knowledge (Charles and Smol, 1994). Many types of indices have been developed based on ratios of the abundance of organisms in these different ecological categories, with conditions expressed as a single univariate index (Birks, 1995).

A discussion on non Weighted Averaging techniques, by necessity, concentrates on the history of pH reconstruction as this was the basis of most of the initial research on diatom based palaeolimnology. One of the earliest, and still most important, studies on the relationship between diatoms and changing pH conditions is the research of Hustedt (1937 - 1939) who examined 650 samples from a range of habitats in south east Asia. Hustedt concluded that the hydrogen ion concentration of the water had the greatest influence on the composition of the diatom flora, a result that has been found to be a general phenomenon in other areas (e.g.: Stevenson *et al.*, 1991; Bennion, 1993; Reid, 1997). Hustedt classified his diatom taxa into 5 categories according to their individual pH tolerances, ranging from alkalibiontic to acidobiontic (see section 5.3 for full explanation). From these groupings, he concluded that the greatest diversity in diatom taxa was to be found within the pH 7 - 8 category, with diversity decreasing as pH declined. Despite minor confusion over Hustedt's definition of the groupings (see Battarbee, 1984 for a review), and the many recent successful advances in quantitative analysis of diatom / pH relationships, this classification remains greatly utilised.

In 1956, Nygaard attempted to quantify Hustedt's classification system, taking into consideration all taxa, in opposition to Hustedt who only included common taxa, arguing that rare taxa, particularly those that showed distinct preferences for the extremes of the pH gradient, often proved to be better ecological indicators. To account for minor taxa, Nygaard weighted the relative frequencies of acidobiontic and alkalibiontic taxa by a factor of five, and then summed the total number of taxa in both of these pH "units" (Nygaard, 1956). Three indices were then derived from these data: α = acid units / alkaline units; ω = acid units / number of acid taxa; and ε = alkaline units / number of alkaline taxa.

Battarbee (1984) justifiably criticises Nygaard's index as the downweighting of circumneutral taxa can greatly under or over estimate actual lake pH conditions, particularly in lakes that contain very few alkaliphilous or alkalibiontic taxa. Meriläinen (1967) also found a flaw in the sampling procedure used by Nygaard to obtain his results, where contemporary pH measurements were taken without adjustment for seasonal fluctuations. In his own study of Finnish lakes, Meriläinen (1967) found a better correlation between predicted and measured pH if values were only used from the autumn overturn period, and concluded that the index based system was capable of predicting pH with an error margin of ± 0.5 pH units. However, despite these criticisms of the index approach, Nygaard's system has been used by many workers, with good correlation between measured and predicted pH (e.g. Foged, 1969; Renberg, 1982).

Furore concerning lake acidification in the 1970s and 1980s promoted the need for more accurate, empirical evidence of recent lake acidification processes (Battarbee, 1984). The pioneering work of Imbrie and Kipp (1971) initiated the development of several numerical techniques to reconstruct palaeoenvironmental variables quantitatively (see Charles, 1982; Davis *et al.*, 1983; and Davis and Anderson, 1984). Birks (1995) states that the fundamental distinction between the developed methods is the underlying taxon – environment response model that is assumed by the different approaches. Principally, there are two main models – linear taxon response models and unimodal taxon - based models. Linear based techniques include methods such as classical and inverse linear regression, principle components regression, canonical correlation analysis, redundancy analysis, and multivariate partial least squares regression. Linear methods are not commonly used for determining diatom taxon – environment relationships due to the non – linear response of most diatoms to environmental forces. ter Braak (1987) states that all taxa tend to occur over a characteristic but limited environmental range, with their environmental optimum occurring within this range, with this relationship being termed the Gaussian response.

Statistical procedures that assume a unimodal taxon response include maximum likelihood, log linear and logit regression and calibration, Weighted Averaging regression and calibration, Weighted Averaging - Partial - Least squares regression, and inverse regression methods based on correspondence analysis (Birks 1995). There has also been recent developments in quantitative inference models that are capable of assuming skewed unimodal or a sigmoid increasing / decreasing reponse, such Artificial Neural Networks, which appear to have a similar predictive power to that of Weighted Averaging (Racca *et al.*, 2001). However, the most commonly used methods, and the ones used in this study because they have been comparatively extensively evaluated, are Weighted Averaging regression and calibration and Weighted Averaging - Partial Least - Squares regression.

2.3.2 Weighted Averaging techniques for palaeolimnological reconstruction

Birks (1998) states that in the last decade palaeolimnology has undergone a fundamental shift in methodology, from being a predominantly qualitative, descriptive subject to a quantitative, analytical science. A common perception is that advances in quantitative techniques, particularly that of Weighted Averaging, has revolutionised palaeolimnological interpretations and that such studies have made their way to the forefront of the developing science of ecological inference (Smol, 1990).

The use of Weighted Averaging (WA) is now widespread within diatom based palaeolimnology and is regarded by many as one of the most sophisticated and accurate tools available (Fritz *et al.*, 1991; Hall and Smol, 1992; Bennion *et al.* 1995; Gasse *et al.*, 1995; Gell, 1995; and Birks, 1998). ter Braak and Juggins (1993) attribute the recent popularity of WA regression and calibration to ecological plausibility, mathematical and computational simplicity, a rigorous underlying theory, and good empirical predictive power. They also state that WA consistently performs as well as, or even better than, maximum likelihood

regression and calibration, which is theoretically more rigorous, as assessed in terms of root mean square error of prediction (RMSEP) in cross validation. However, it is worth noting that Birks (2001) has recently published a correction to the computer program that implements maximum likelihood regression, and that it now appears that this method can perform as well as, or even better than, Weighted Averaging regression.

Weighted Averaging regression is based on the hypothesis that diatom taxon distributions have a point along environmental gradients which represent their optima, or the point at which their abundance is expected to be greatest. Briefly, the method involves estimating the environmental variable (x) optimum for a particular taxon by averaging all the x values for sites in which the taxon occurs, weighted by the taxon's relative abundance. The taxon's tolerance can also be estimated as the weighted standard deviation or abundance weighted standard deviation of x (Birks, 1995).

Weighted Averaging regression involves firstly the development of a diatom / environmental gradient calibration data set which quantifies this relationship between individual diatom taxa and environmental variables. The calibration data set (or training set) is generally designed to represent the maximum range of environmental gradient variation possible, or that is appropriate for the geographical area of interest. Usually between 30 - 300 modern diatom samples (from at least 30 different sites) and data on 1 - 20 environmental variables are collected (Birks, 1998). The relative abundance of each diatom taxon is determined for each of the diatom samples. Multivariate statistics (principally Canonical Correspondence Analysis) are then used to identify which environmental variables are most strongly (and significantly) correlated with diatom taxa distribution. Once this is established, the unimodal statistical technique of Weighted Averaging regression is used to determine the environmental optima of individual diatom taxa. Weighted Averaging regression assumes that the environmental optimum of a taxon occurs at, or close to, its maximum abundance (ter Braak and van Dam, 1989), and that this optimum can be accurately estimated by a simple abundance - weighted average of the taxon abundance along the environmental gradient (Anderson, 1997). The performance of these models is usually reported in terms of the bootstrapped or jackknifed root mean square error of prediction and the r^2 correlation coefficient of determination between measured and inferred variables. These optima are then used to reconstruct past environmental variables from fossil diatom assemblages using Weighted Averaging calibration techniques. Further clarification of the method can be found in Charles and Smol (1994), Bennion (1994), Bennion *et al.* (1995), Birks (1998) and Lotter *et al.* (1998).

2.3.2.2 Limitations of the Weighted Averaging approach

Anderson (1995) argues that the continued pursuance of qualitative palaeolimnological techniques results in neolimnologists assuming that palaeolimnology has little relevance to contemporary problems and process studies. However, he also suggests that although quantitative methods can improve the relationship and respect between neo and palaeolimnologists, there is still considerable room for improvement. Birks (1998) states that the biggest limitation in quantitative environmental reconstructions is the quality and internal

consistency of the modern calibration data set. Fundamental to the usefulness of any transfer function is an accurate understanding of the present day distributions and ecological tolerances of the taxa that are contained within the sediment record.

The principle criticism of Weighted Averaging transfer functions is their lack of sensitivity to the issues of diatom provenance. Sayer (1996 and 2001), Reid (1997) and Tibby (2000) have all demonstrated that changes to the habitat source of diatoms, such as abundances of planktonic taxa versus littoral and epiphytic taxa, can greatly affect the numerical reconstruction. Sayer (1996), in particular, found that large increases in planktonic taxa in the sediment record often generated sharp increases in TP concentrations at times when accurate historical records suggested that TP should be decreasing. Sayer (1996) concluded that the increase in planktonic taxa arose as a result of transition from macrophyte to phytoplankton dominance rather than a nutrient increase. The switches between littoral and planktonic taxa would not be as great a cause for concern if the optima of both sets of taxa were narrow, but many littoral taxa, particularly those that commonly replace planktonic taxa such as *Staurosira* spp., and *Cocconeis* spp. have very wide ecological tolerances which can distort the reconstruction (Bennion *et al.* 2001).

Following on from this point, sometimes the majority of taxa in the sediment record occur over broad ecological gradients and therefore convey little information on specific physico - chemical conditions. This problem of broad ecological tolerance can be illustrated by the nutrient reconstruction of Burrinjuck Reservoir in NSW (Tibby 2000). The presence of littoral taxa in the sediment record greatly exaggerated inferred TP because these taxa were ubiquitous in almost all of the samples in the modern data set and hence exhibited wide tolerances. To put more simply, within a modern data set, a taxon may have a TP optimum of 50 µg/L, but it will also occur at up to 40 % of the total assemblage at both 10 µg/L and 100 µg/L (as was the case with the modern data set applied to Burrinjuck Reservoir). If this optima of 50 µg/L is applied to systems that were actually oligotrophic then the inferred TP will be overstated. Despite the best efforts of statistical innovations, including downweighting of rare taxa, this problem has not been adequately dealt with.

The most commonly encountered taxa that have broad ecological tolerances are the suite of small *Staurosira, Pseudostaurosira* and *Staurosirella* taxa (formerly classified in the genus *Fragilaria*). These taxa are ubiquitous and extremely abundant in many south east Australian aquatic systems (Gell 1995, Reid 1997, Tibby 2000) and so this issue is highly pertinent to this study. Many workers have classified this suite of taxa as non-planktonic, preferring to inhabit the benthos. However, there has also been evidence to show that these taxa are, in some systems, important components of the planktonic community, particularly in shallow environments (Lowe 1974, Barker *et al.* 1994, Tibby and Fluin 1997, Dam *et al.* 2000). Sayer (2000) states that the presence of these taxa in a fossil record means that the effectiveness of the transfer function is much reduced, and goes as far to say that where these taxa dominate the fossil record, it is unwise to use a transfer function to reconstruct environmental variables. A recent study by Bennion *et al.* (2001), which aimed to reconstruct TP values from four shallow lakes in Norfolk, England, emphasises the difficulties in using these taxa in quantitative reconstructions. The dominance of *Staurosira, Pseudostaurosira* and *Staurosirella* spp. in

all four records meant that diatom inferred TP did not reflect historical TP values as documented in the literature, with inferred values greatly underestimating the known TP trends.

2.4 Aims of this study in relation to establishing diatom - water quality relationships

Most of the above discussion is focused on the relevance of palaeolimnology to lacustrine environments as this is where the majority of research has taken place. There is a valid reason for this - lakes are fundamentally more stable, both temporally and spatially, than river and floodplain systems. Sediments in riverine environments are much more likely to be disturbed by slumping, flooding, erratic deposition and human alteration such as bridges, weirs, channel stabilisation, devegetation, reservoir formation and numerous inputs and outputs. This poses a problem for Australia, in particular, where the majority of water storages that are of greatest relevance to managers, are riverine or floodplain based.

However, palaeolimnological techniques do provide an opportunity to investigate benchmark ecological conditions in these riverine systems. Thoms *et al.* (1999) state that pre non - indigenous settlement conditions should be the optimal benchmark for the health of disturbed floodplain ecosystems such as the Murray River. The lower Murray River is a complicated aquatic system with a massive temporal (i.e. drying versus flooding) and spatial (influences of regulation, marine incursion, input from the Darling River system, changes in land use) variability. The principle aim of this study was to attempt to quantify this variability over time.

Diatom water quality relationships used in the determination of this variability were generated by several means, with a detailed methodology provided in chapters 3, 4 and 5. Although the other three major south east Australian diatom palaeolimnological studies of Gell (1995), Reid (1997) and Tibby (2000) developed and utilised a Weighted Averaging transfer function as the primary means of environmental reconstruction, this technique was not appropriate as the sole methodology for this study. Principally, this was due to the complexity of problems that are inherent in the Weighted Averaging transfer function approach, many of which were applicable to the study area in question. In addition, the Weighted Averaging method was formulated and tested on northern hemisphere (particularly western European and North American) aquatic systems which are markedly different from south east Australian aquatic systems. South east Australian aquatic ecosystems are unique due to their high level of variability (due partly to the climatic influence of the El Ninio Southern Oscillation), their ionic composition (due partly to the antiquity of the landscape) and the slow flowing nature of our major river systems which has resulted in an abundance of very shallow lakes and wetlands. As diatom based palaeolimnological research is still in its infancy in Australia, and due to the relative uniqueness of the Murray – Darling Basin, it was decided to incorporate as many tools as possible in generating ecological information. The rationale behind this methodology is that by increasing the amount of ecological information used for determining diatom autecology, a more robust and powerful limnological reconstruction will result.

More than 180 diatom samples were collected from aquatic environments within south east Australia along with associated water chemistry. Constrained ordination techniques were then applied to this data to determine which environmental variables were exerting the greatest influence on diatom assemblages. Weighted Averaging techniques were then applied for key variables in an attempt to derive environmental optima and tolerances. Patterns of taxon abundance across ecological gradients were then compared to the published optima and / or preferences from 22 other key studies from both Australia and overseas (for the more common fossil taxa). After reviewing all available information, a summary, generally qualitative, ecological description is proposed for these taxa, including details on pH, EC and TP preferences and any additional information which was gained from the literature (such as habitat preference and preservation potential).

Chapter 3 - The modern data set

3.1 Introduction

The primary objective of this section of the study was to explore diatom community dynamics toward the development of a modern diatom calibration data set. There was no *a priori* selection of a single water quality variable as the focus of the calibration data set (*sensu* Hall and Smol, 1992). Rather, it was decided to investigate diatom assemblage responses to a wide gradient of water quality conditions. This chapter presents both the diatom data and water quality determinations, and also ordination analysis which explores the spatial patterns of the species data. Calibration data sets are then developed in chapter 4.

3.2 The methodology of the modern diatom data set

3.2.1 Site selection

3.2.1.1 Rationale for site selection

When developing a calibration data set that focuses on one or two variables only, it is desirable that the range of variables which are known to be influential on diatom communities are controlled. For instance, if TP is the variable to be reconstructed, then other variables, principally pH and EC, are kept within a narrow range. This then reduces the influence of these other variables on diatom community response. For this study, however, being of an exploratory nature, limited control was imposed on water quality parameters from sampling sites. Although somewhat subjective, the main determinant used for site selection was that the sites physico-chemical properties must have been within the likely boundaries of past conditions of the lower Murray River. Essentially, this meant that sites with very low pH (< 6) or very high EC (> 15 000 µS/cm) were eliminated because it was very unlikely that the lower Murray River ever experienced these conditions (Bourman and Barnett, 1995). Additionally, these environmental extremes are usually associated with unique diatom assemblages and it was thought that these would be easily recognisable in the fossil record. This process of site selection ensured the effectiveness and accuracy of modern analogues and prevented the generation of superfluous data.

As the lower Murray River is influenced by both lotic and lentic processes, it was decided to sample both lake and stream / river environments in order to develop the ecological data needed for the interpretation of the fossilised diatoms. By exclusively sampling lakes, there would have been a gap in the knowledge of ecological preferences of particular diatoms that are only prevalent in river systems, and vice versa. In terms of both qualitative and quantitative analysis, the lake (n = 50) and stream (n = 51) data sets were initially analysed separately and then together using a combined data set of 101 samples.

3.2.1.2 Lake data set

The lake data set consisted of 50 lakes from both South Australia and Victoria (locations in figures 3.1a and 3.1b). The names of the sites and physical properties are outlined in Appendix 1. There were three main areas of sampling - the Adelaide hills (18 lakes with both diatom and water sampling undertaken by the author), the Kerang region (21 lakes with diatom and water sampling undertaken by L. Radke) and the Mildura region (11 lakes with diatom and water sampling undertaken by I. Sluiter). The Kerang and Mildura samples were included because of the lakes proximity to the Murray River and because of their widely varying TP and EC concentrations. Lakes in the Adelaide hills region were chosen on the basis of both widely varying depth and degrees of disturbance (ranging from protected catchments to urban recreation lakes), and also because the majority of sites receive piped water from the Murray River.

3.2.1.3 Stream data set

The stream data set utilised sampling undertaken by the Australian Water Quality Centre (AWQC) for the South Australian Environment Protection Authority (EPA) as part of the national AUSRIVAS sampling program. During both spring and autumn of 1995 the AWQC routinely collected water samples and diatom scrapes from over 100 different streams throughout South Australia. From this program, 51 streams were chosen to be included in this study and these are illustrated in figure 3.2. The names of the sites and physical properties are presented in Appendix 1. The streams were selected on the basis of similar ionic composition to the Murray River, a wide range of TP and EC, and also a wide variety of flow conditions. After initial screening of the diatom communities found in the streams, it was also decided to include a wide geographic range of sites so as to include as much taxa variability as possible. Therefore, although there are many sites scattered adjacent to the lower Murray system, there are also sites as far north as Coopers Creek (see figure 3.2).

3.2.2 Diatom sampling

3.2.2.1 Lake data set

Essentially there are two principle methods for collection of diatom samples for use in the derivation of ecological information :

a) collection of a centre lake surface sediment sample (commonly the upper 1 cm). The diatom assemblage is then related to the past 12 months of water quality data. Usually at least 6 water chemistry / physical parameter measurements are taken during this preceding 12 month period and these are averaged. This method is often termed the taphenocoenose approach.
b) collection of a littoral zone surface sediment sample (usually a mud or rock scrape). The resultant assemblage is then related to the contemporary water quality only. It is assumed that the littoral zone sample is a modern diatom sample (as opposed to the centre lake sample which is mostly fossilised material) and as such represents the present lake chemistry.

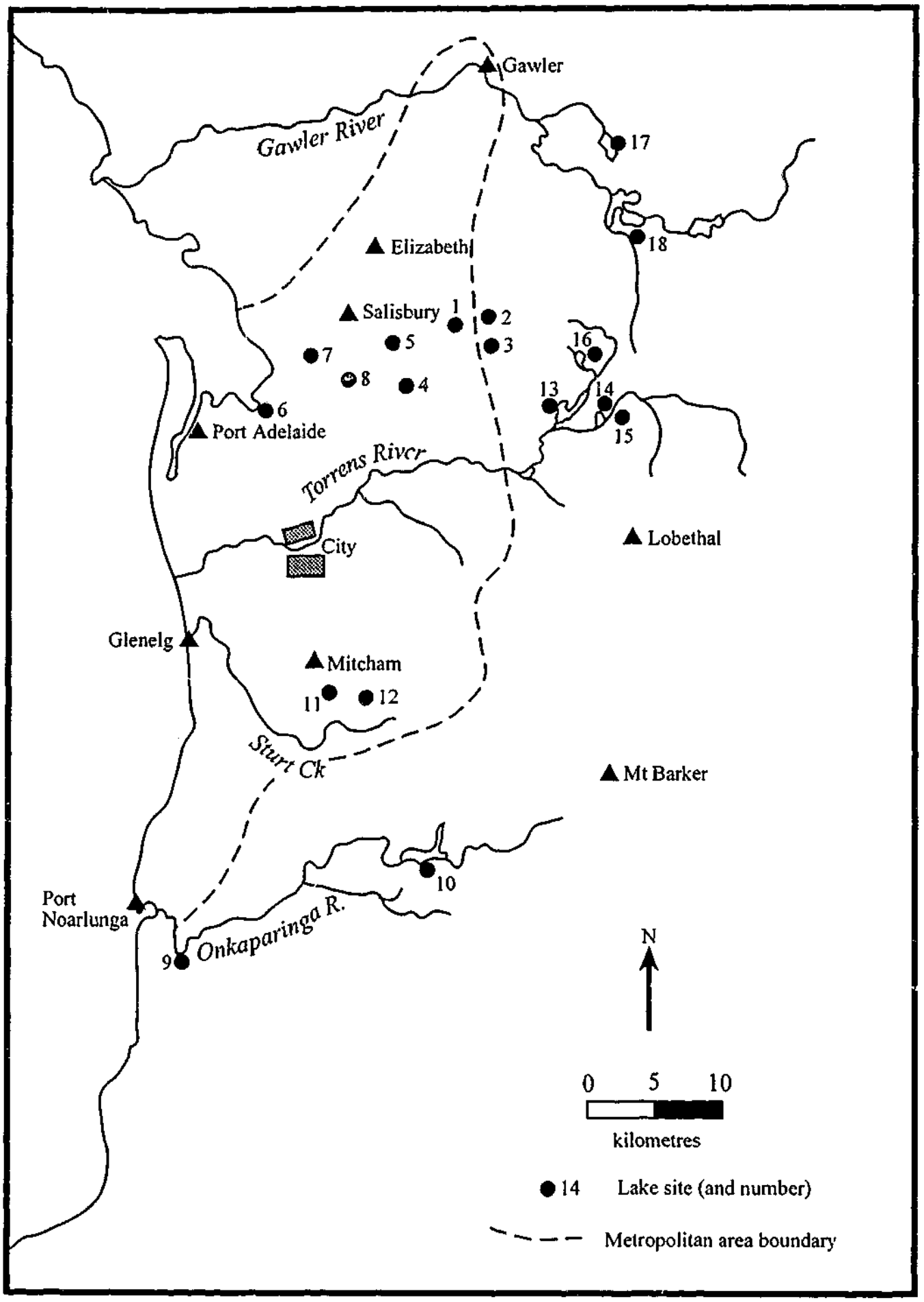


Figure 3.1a: Location of the lake sampling sites for the modern data set (sites 1 – 18).

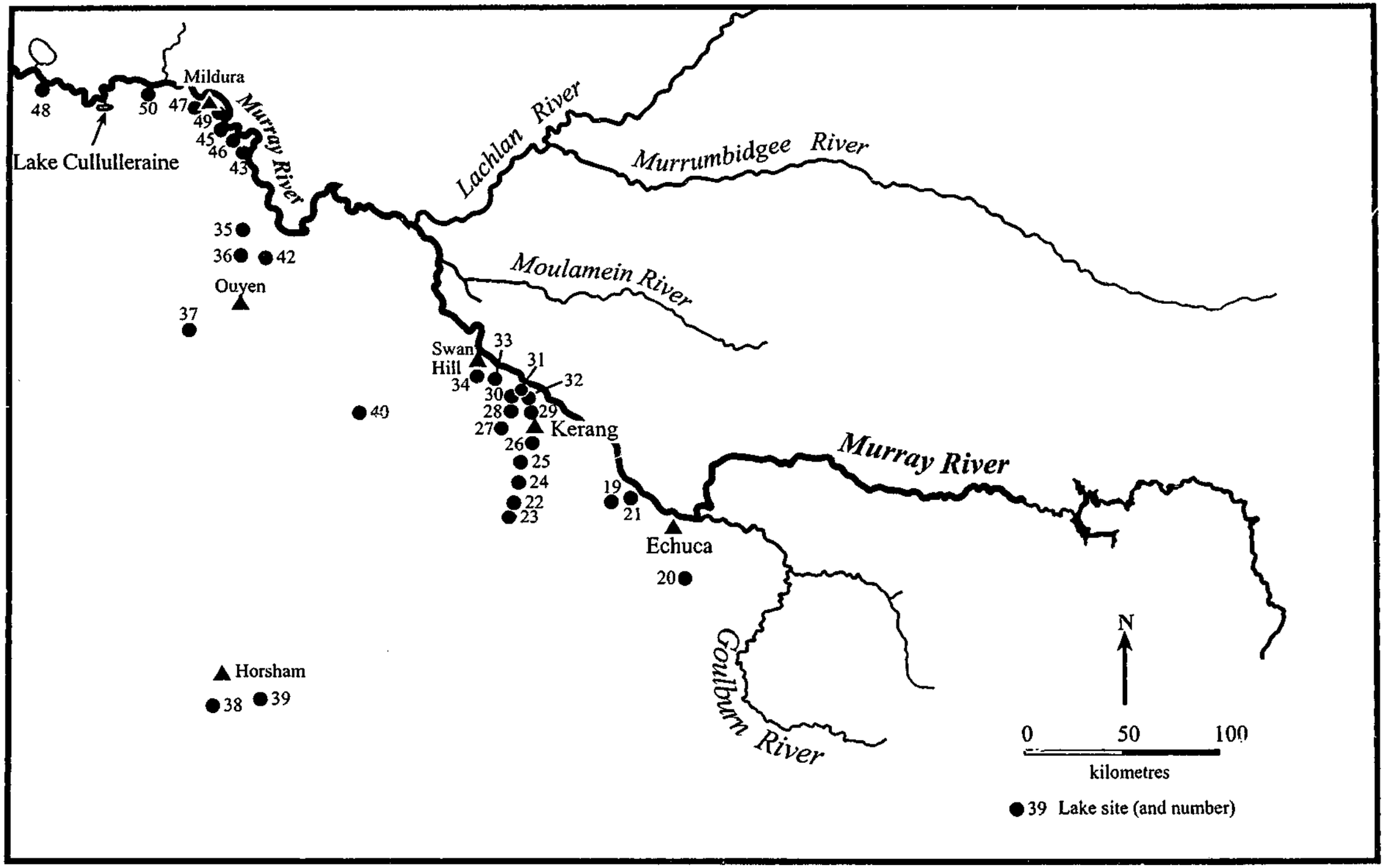


Figure 3.1b: *Location of the lake sampling sites for the modern data set (sites 19 - 50).*

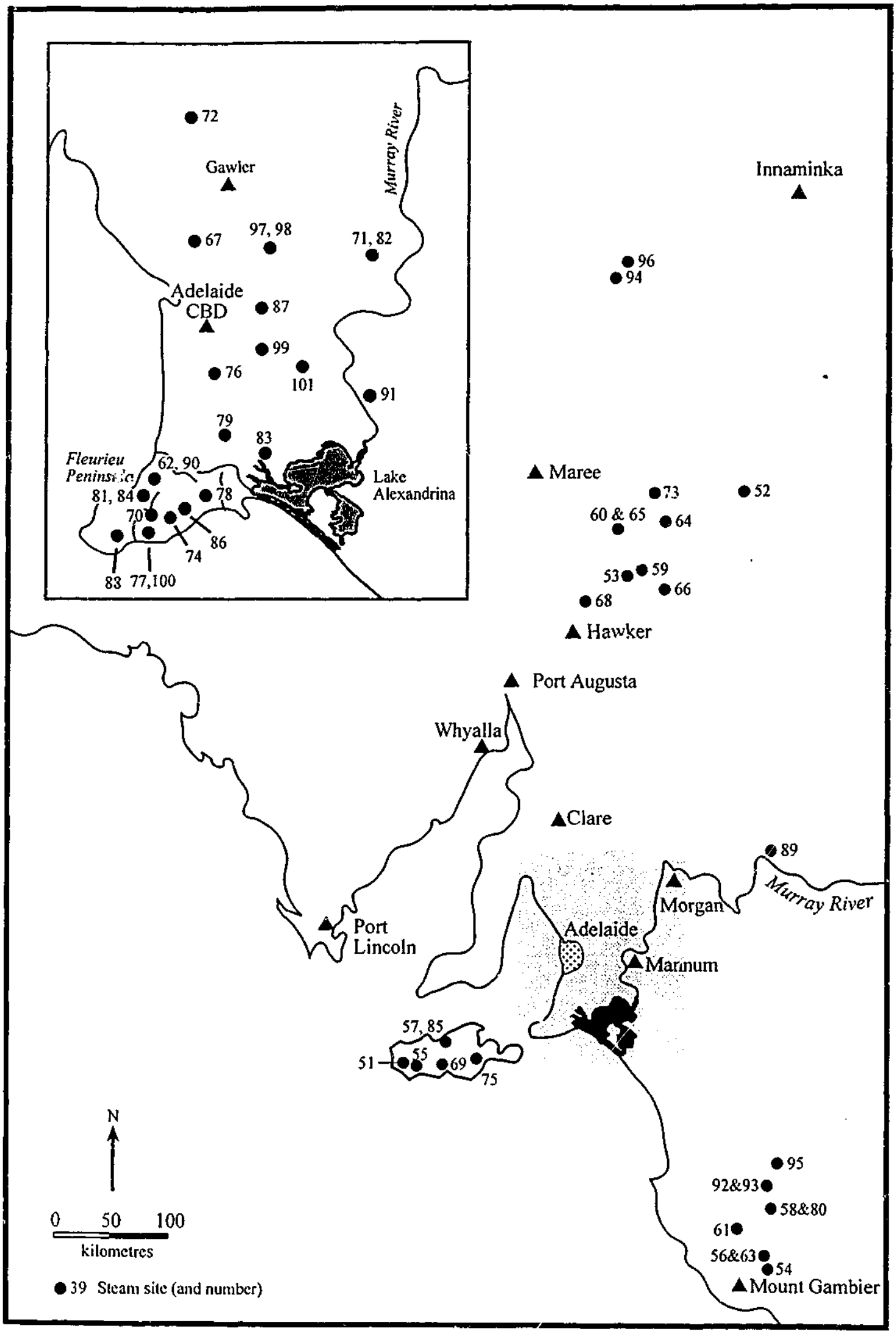


Figure 3.2: Location of the stream sampling sites for the modern data set (sites 51 – 101).

The method of littoral zone sampling was adopted for several reasons. Firstly, there is no guarantee that the upper 1cm of sediment in the centre of a lake represents the past 12 months. Sediment accumulation rates may vary from a fraction of a cm / year to several cm / year. Without extensive dating it is not possible to gauge the rate of sedimentation prior to sampling and so there will always be a high level of uncertainty about the duration of time represented in the surface sample. This is especially pertinent in this study as the lake location and type, and thus sedimentation rates, varied dramatically. Secondly, few lakes in the region have been monitored continuously for 12 months for the full range of environmental parameters. This then vastly narrows the list of suitable lakes for inclusion in the data set. There have been studies (i.e.: Bennion 1993) where the researcher has routinely taken the water samples every month for 12 months but this was impossible in this type of study where the 50 lakes were spread across two states. Additionally, although the taphenocoenose sampling approach is used widely in Europe and North America with considerable success (i.e.: Bennion 1994, Hall and Smol 1992), it may not be appropriate for the relative complexity of south east Australian aquatic systems. Australia's ENSO impacted environment is subject to such high seasonal and interannual variability that the accuracy of the taphenocoenose approach is greatly decreased.

To obtain a littoral diatom sample, a lake edge surface sediment sample was collected from all 50 lakes. Sampling consisted of taking a scrape of the uppermost layer of sediment at the lake margin. By sampling the very top of the surface sediment (usually < 1 mm) it was hoped that the sample would consist of live or very recently dead diatoms, thus accurately reflecting the water quality at the time of sampling. The sample was then placed in a labelled jar and preserved with ethanol.

3.2.2.2 Stream data set

Two samples were taken from each site - a rock scrape and a mud scrape from on top of the rock surface. This is the standard protocol used for sampling stream diatom communities for the National AUSRIVAS program (Gell *et al.*, 1999). These were preserved in the same manner as those taken from lakes.

3.2.2.3 Murray River plankton survey

In order to further investigate the diatom assemblages likely to be found in the fossil records, a river plankton survey was undertaken in November 1997 and April 1998. These months were chosen to ensure collection from both high flow (November) and low flow (April) periods. Nine sites were included in this study; Lake Alexandrina, Tailem Bend, Mannum, Swan Reach, Morgan, Berri, Renmark, Lake Cullulleraine and Mildura (all located on figure 3.1). Tailem Bend and Mannum were not included in the November 1997 survey. Samples were obtained by filling a 10 litre bucket with surface water from the middle of the channel (reached by either boat or by dropping the bucket from the middle of a bridge). Replicate one litre subsamples were then taken and preserved with 10% formaldehyde so that the chloroplasts remained intact, enabling identification of live cells.

3.2.2.4 Laboratory techniques

Standard methods (Battarbee, 1986) were used to prepare the diatom samples. This involved simmering the sampled materials for 2-3 hours in 10% hydrochloric acid to remove all carbonate material. This was followed by repeated washing, settling and decanting in distilled water. Each sample was then heated in 10% hydrogen peroxide for a similar length of time to remove all organic matter. After several subsequent washes, approximately 400 μL of the residue was left to dry on a washed coverslip overnight. Two coverslips were prepared for each sample and these were then mounted onto slides using Naphrax $_{TM}$ mountant.

The majority of counting was undertaken using an Olympus BH-2 microscope with Nomaski Differential Interference Contrast (1250×). Occasionally a Zeiss Axioscope (2000×) was used to gain better resolution when checking identification. Up to 500 diatom valves were counted per sample, with a minimum count of 300 valves, which is the minimum number suggested by Battarbee (1986) as a representative count.

The primary diatom floras consulted to assist identification were Krammer and Lange-Bertalot (1986, 1988, 1991a, 1991b). These were supplemented by Gasse (1986), Germain (1981), Foged (1978), Patrick and Reimer (1978), Archibald (1983), and John (1993). Round *et al.* (1990) proposed extensive taxonomic revisions to some of the taxa described in these floras (as outlined in Sonneman, 2000), and where the distinguishing features were identifiable by light microscope, these revisions were adopted.

3.2.3 Environmental variable data

Fourteen physico - chemical variables were measured and used for gradient analysis of the diatom data set. All sites were analysed for pH, temperature, turbidity, dissolved oxygen, total phosphorus, total Kjeldahl nitrogen, EC, potassium, magnesium, calcium, sulphate, alkalinity, sodium and chloride. An additional three variables were used in the analysis of the streams data set: shade cover, soluble reactive phosphorus and nitrate / nitrite. The combined data set included the 14 variables that were common to the lake and stream data sets.

3.2.3.1 Sampling procedures

All water samples were taken using acid - washed plastic bottles at the time of diatom sampling. Two different agencies were involved in analysing the water samples from the lake data set. The Kerang region lakes were analysed by the Commonwealth Scientific Industrial Research Organisation (CSIRO) division of Land and Water in Canberra. Samples collected from Adelaide and the Mildura region were analysed by the author at the Department of Geography laboratory at the University of Adelaide. For the stream samples, all water samples were collected by the South Australian EPA and analysed at the Australian Water Quality Centre, Adelaide. Methods used for water analysis at these agencies are outlined in Appendix 4. Inter - laboratory quality control is obviously preferable for a study such as this where results are combined from different sources. However, financial constraints meant that using three different laboratories was unavoidable.

3.4 Results from the modern diatom data sets

3.4.1 Summary of diatom taxa results

There were 220 diatom taxa in the lake data set, 199 diatom taxa in the stream data set, with 169 diatom taxa common to both data sets. For the purposes of statistical analyses, TRAN (Juggins, unpublished program) was used to eliminate all taxa that had a maximum abundance across all samples of less than 1%. This then decreased the lake data set to 141 taxa, the stream set to 186 taxa, and the combined data set to 159 taxa. Diatom counts were transformed to percentages using TILIA version 1.12 (Grimm, 1992). These percentages are presented in Appendix 2. A list of authorities and synonyms for all diatom taxa mentioned in the text and / or illustrations are presented in Appendix 3. Table 3.1 summarises the major taxa found in both environments, the number of lakes and / or streams within which they occurred, and their maximum abundance within the data set.

Table 3.1 shows that there are several abundant taxa that occurred predominantly in only one habitat although some of this variation may be due to the type of substrate sampled. For instance, although *Aulacoseira granulata* and *Cyclotella meneghiniana* occur in substantially more lakes than streams this difference may be due to these taxa being planktonic, with little likelihood of being present in the stream rock scrape samples. Other taxa are more common in lake environments than in streams but are generally more abundant in the stream sites (i.e.: *Staurosirella pinnata* has a maximum abundance of > 50% in stream samples and only 16% in lake samples). *Bacillaria paradoxa* and *Rhoicosphenia abbreviata* are perhaps the only two major taxa that occur almost exclusively in one environment, being present in only five lake samples and one lake sample respectively while both occurring in at least 30 stream samples.

Table 3.1: *The major diatom taxa found in the lake and stream data sets, showing the number of sites that they occurred and their maximum abundance.*

Lake data set	***n***	**Max.%**	**Stream data set**	***n***	**Max.%**
Planothidium delicatulum	29	38	*Planothidium delicatulum*	20	76
Aulacoseira granulata	20	92	*Achnanthidium minutissimum*	26	53
Cocconeis placentula	18	25	*Bacillaria paradoxa*	30	34
Cyclotella meneghiniana	26	53	*Cocconeis placentula*	24	22
Pseudostaurosira brevistriata	18	50	*Gomphonema parvulum*	20	32
Staurosirella pinnata	23	16	*Melosira varians*	20	40
Navicula cryptocephala	16	37	*Navicula veneta*	32	28
Navicula veneta	23	15	*Rhoicosphenia abbreviata*	33	90
Tryblionella hungarica	27	26	*Rhopalodia musculus*	23	28
Nitzschia lacuum	21	30	*Surirella brebissonii*	21	32
Nitzschia palea	22	14	*Synedra ulna*	27	30

3.4.2 Statistical analysis of the patterns in diatom taxa data

Pattern analysis within the modern diatom data set was carried out using Detrended Correspondence Analysis (DCA). DCA is an eigenanalysis based ordination technique derived from correspondence analysis but detrends the data to counteract the arch effect, which is common in correspondence analysis (CA) (ter Braak and Prentice, 1991), and which was apparent for this data set. For all three data sets DCA was performed with detrending by segments (26 segments), non-linear axis re-scaling (4 times), and downweighting of rare taxa.

3.4.2.1 Lake data set

The summary statistics for DCA on the lake data set are shown in table 3.2.

***Table 3.2**: Summary statistics of a DCA of the 50 sample lake data set*

Axes	1	2	3	4	Total inertia
Eigenvalues	0.771	0.619	0.532	0.446	12.382
Cumulative% variance of taxon data	6.37	11.2	15.5	19.1	

The first four axes explained 19.1% of cumulative variance in taxon data. The sample and diatom taxa plots are shown in figures 3.3 and 3.4. Figure 3.3 shows that there was a distinct split between the geographical areas, with samples 1 to 18 (Adelaide region) all plotting high on axis 1 and samples 19 to 50 (Kerang and Mildura) all plotting relatively low on axis 1. Within this subset of Adelaide samples, the ordination plot further distinguishes between samples, with Adelaide Hills samples (samples 10 – 18) generally plotting separately to the Adelaide plains samples (samples 1 – 9). This may be due to differences in nutrient concentrations, with samples 1 – 9 all having higher TP and TKN concentrations than the Adelaide Hills samples, and this is explored further in Chapter 4 through Canonical Correspondence Analysis (CCA).

Axis 2, although of a similar length to axis 1, had a less even spread of samples, particularly for the Kerang region (19 – 39), which were more tightly grouped than the Adelaide region samples. This is perhaps due to lakes in the surrounds of Adelaide having a widely varying nature, both physically and chemically, ranging from recently constructed wetlands in urban areas to large reservoirs in semi-protected catchments. The lakes in the Kerang region, while spatially diverse (ranging in size from a couple of hectares to hundreds of hectares), have a similar catchment area with generally all being rural, at low altitude, and quite turbid. It is likely that these circumstances contribute to greater differences in diatom samples in the Adelaide region than the Kerang region.

The Mildura region samples (samples 40 – 50), however, are greatly spaced along axis 2, ranging from sample 45 at 0.0 deviation units, to sample 49 at > 6.0 deviation units. The water chemistry of both of these sites is very similar and so it is likely that the different diatom assemblages, as reflected in the spacing

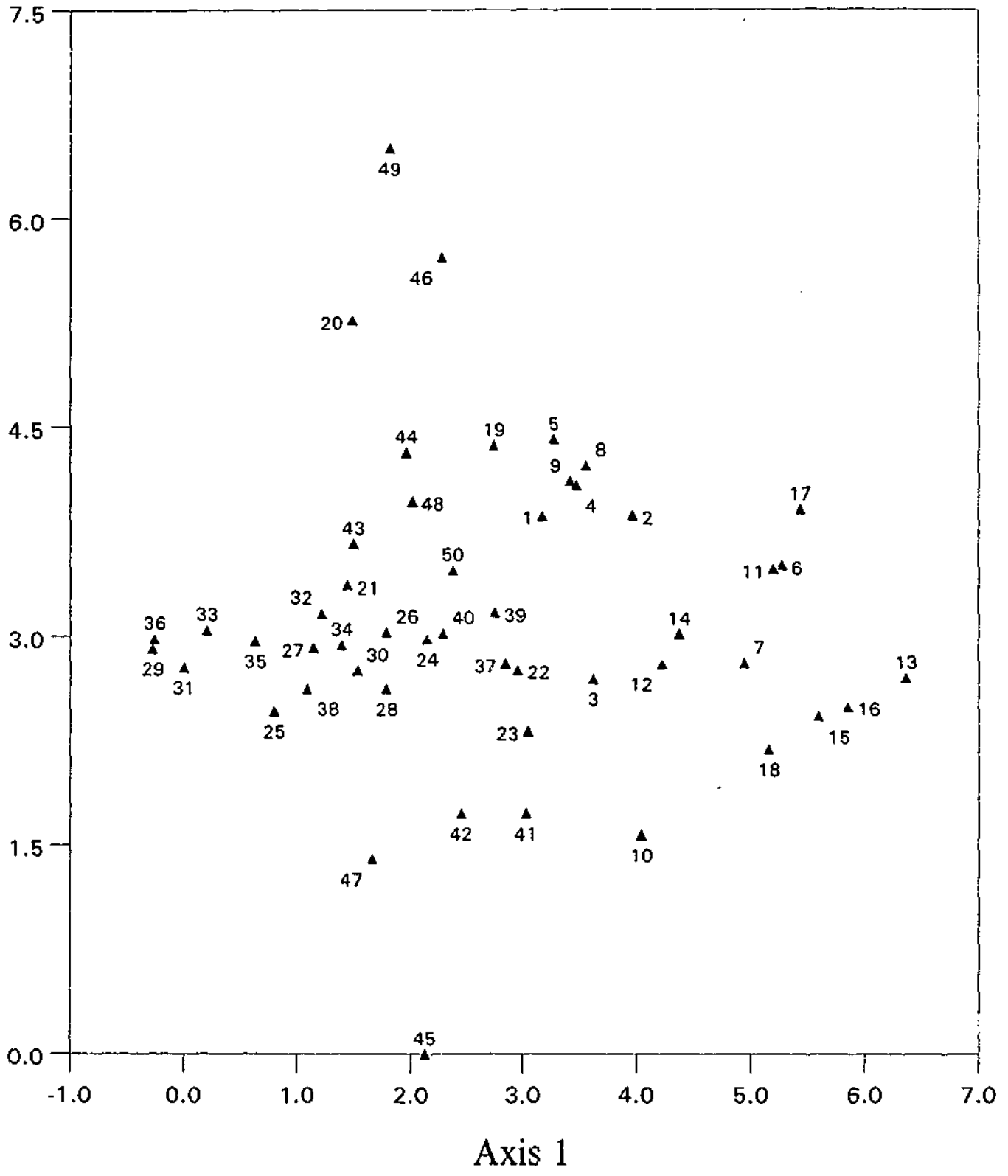


Figure 3.3 Ordination biplot of DCA site scores on axis 1 and axis 2 for the 50 sample lake data set.

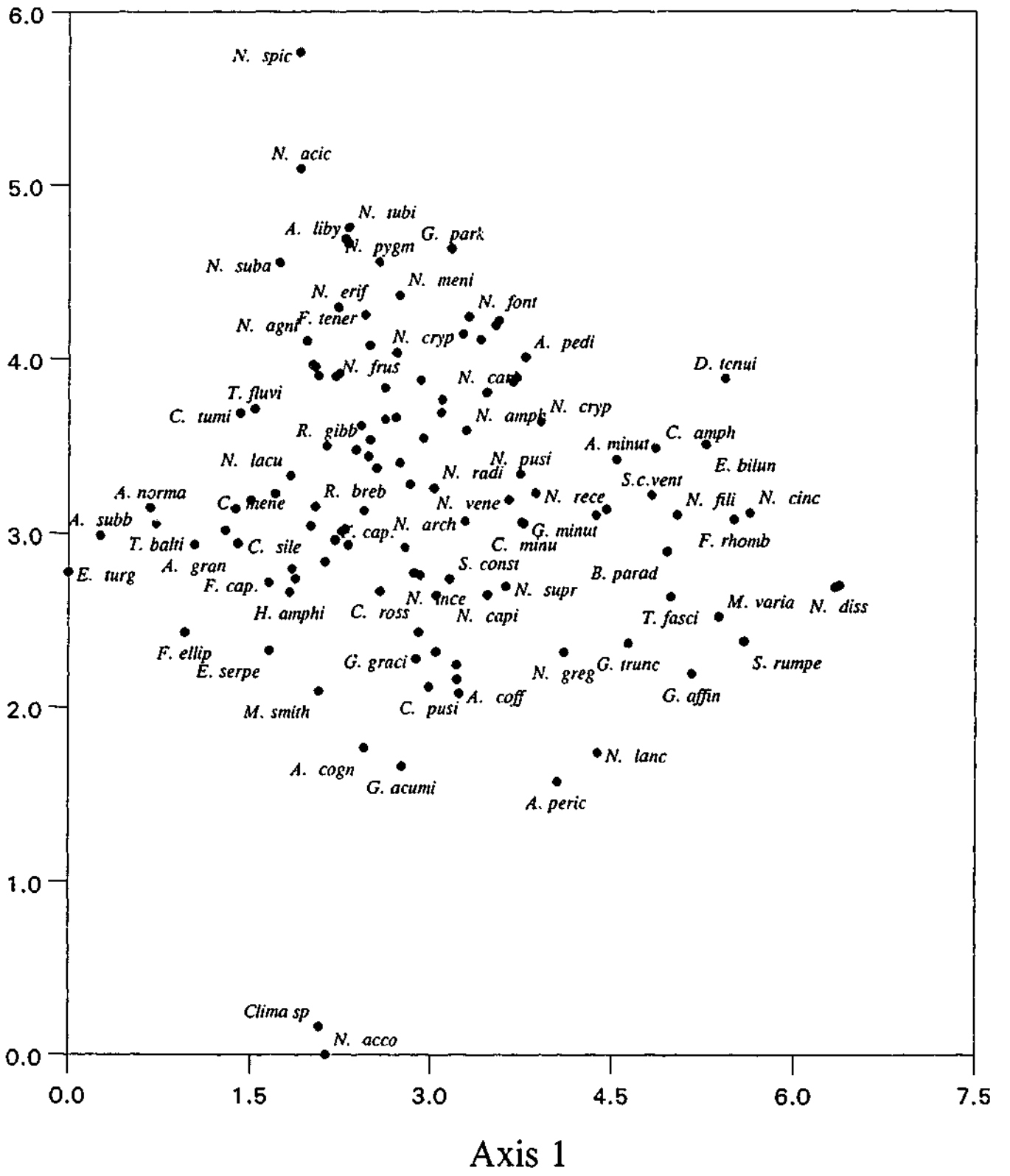


Figure 3.4 Ordination biplot of DCA taxon scores on axis 1 and axis 2 for the 50 sample lake data set

between samples, are due to habitat differences (such as degree of connectivity to the river) or other environmental parameters which were not measured (such as depth).

Figure 3.4, a plot of diatom taxon scores, shows which taxa were associated with these samples. Although not all taxa have been labelled, it can generally be seen that planktonic, facultative planktonic and littoral taxa are not confined to one subset of lakes (such as Kerang or the Adelaide set) and are well spread across the diagram. The outlying taxa *Navicula spicula, Climaconeis* sp. and *Navicula accomoda* are associated with samples 45 and 49 from the Mildura region. These taxa were rare within the data set, but abundant in these individual samples. Apart from these observations, the DCA plot of taxon scores does not appear overly informative, with taxa simply plotting in relation to the samples in which they are most abundant. For instance, *Cyclotella meneghiniana* is most common, and abundant, in the Kerang region samples, which is reflected in the ordination diagram.

3.4.2.2 Stream data set

The summary statistics for DCA on the stream data set are outlined in table 3.3.

***Table 3.3**: Summary statistics for the stream set DCA (51 samples)*

Axes	1	2	3	4	Total inertia
Eigenvalues	0.752	0.568	0.458	0.378	12.163
Cumulative% variance of taxon data	6.2	10.9	14.6	17.7	

The first four axes explained 17.7% of the taxa data, slightly lower than for the lake data set. Figure 3.5 shows the plot of samples scores. Although the spread of samples is greater along axis 1, this is mainly limited to a few samples (89, 91, 54 and 51). Samples 89 and 91 lie at the low end of axis 1, separated from the remaining samples. These samples were both taken from sites in the lower Murray River, a river which is often described as essentially being a series of interconnected pools because of intensive regulation and very low flow velocity (Hotzel and Croome, 1996). Thus, this environment is markedly different to the other temperate, shallow, relatively fast flowing streams in the data set. These samples also have very high nutrient concentrations and turbidity levels. Samples 51 (Breakneck Creek) and 54 (Deep Creek) are located at the high end of axis 1, again relatively separate to the remaining samples. These samples were both taken from well shaded, low flowing sites with very low nutrients and turbidity. The spread of samples along axis 2 is more difficult to interpret as there is an even representation of samples with widely varying physico – chemical properties. However, there does appear to be some grouping according to location of sites, with samples collected from the arid northern Flinders Ranges (52, 53, 60, 64, 65, 68 and 73) all plotting on the high end of axis 2.

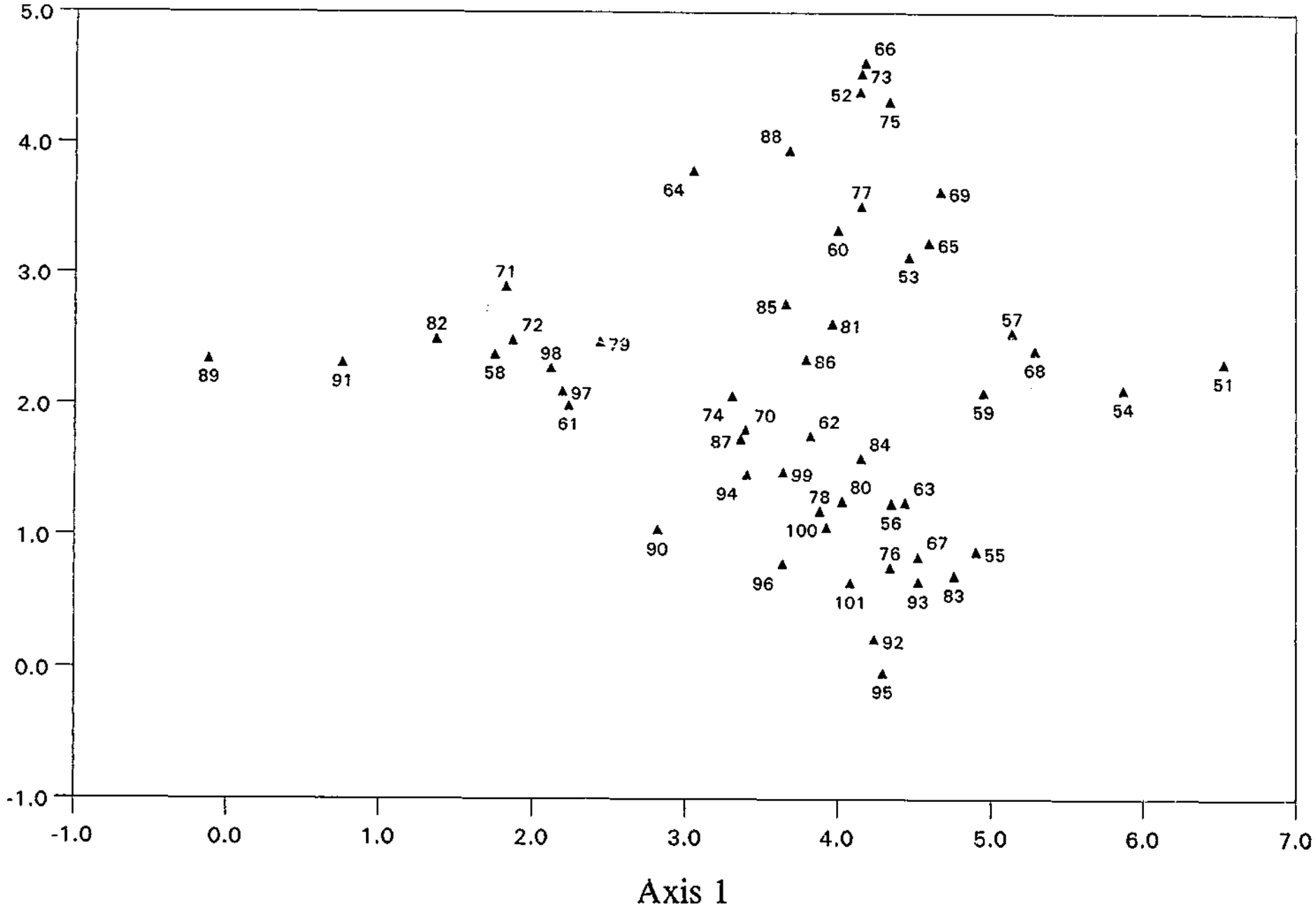


Figure 3.5 Ordination biplot of DCA site scores on axis 1 and axis 2 for the 51 sample stream data set

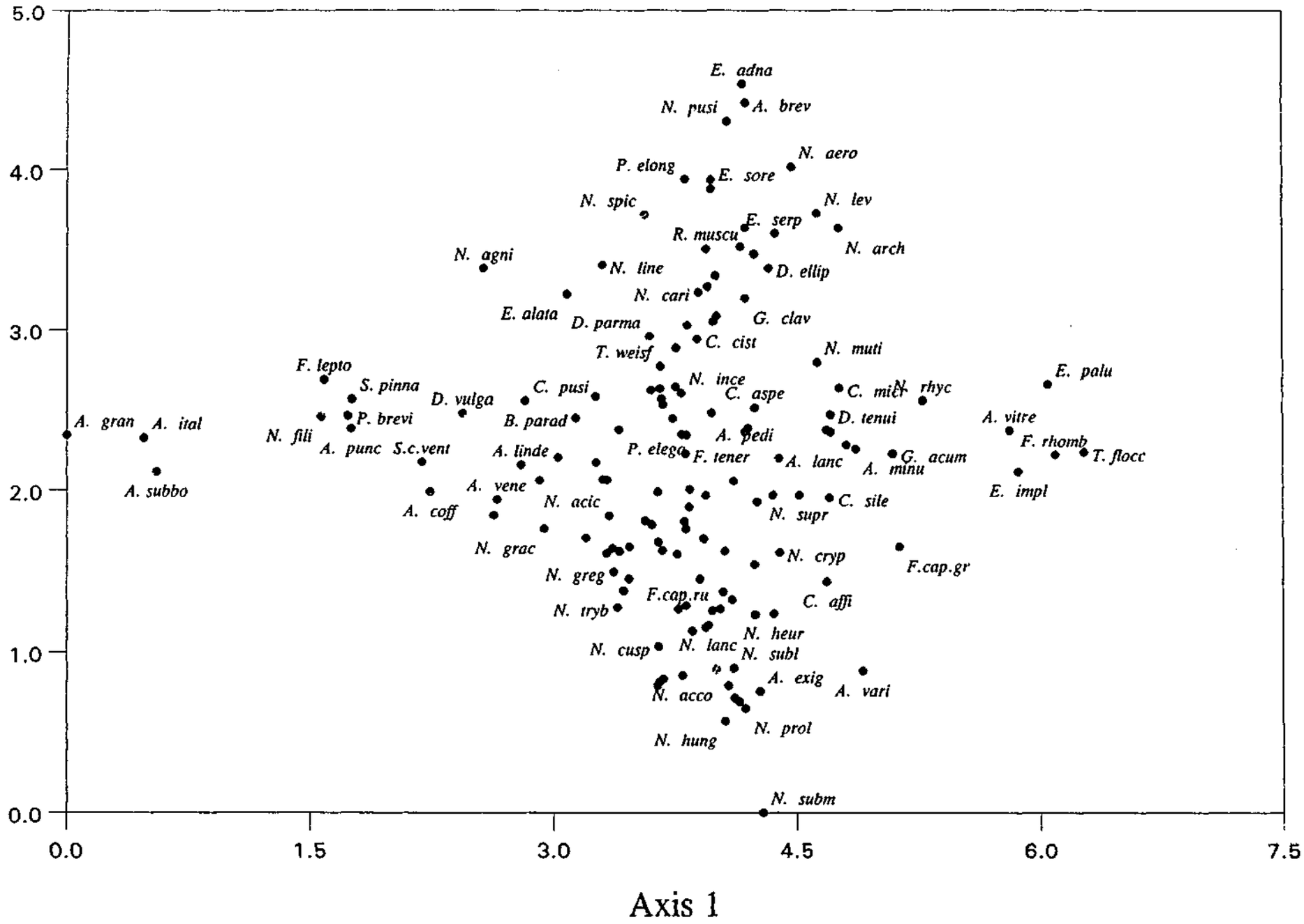


Figure 3.6 Ordination biplot of DCA taxon scores on axis 1 and axis 2 for the 51 sample stream data set

Figure 3.6. illustrates the diatom taxon scores for the stream data set. Among the taxa that plot at the low end of axis 1 are the group of small taxa from the *Staurosira* group, such as *Staurosirella pinnata*, *Pseudostaurosira brevistriata*, and *Staurosira construens* forma *venter*. This group of taxa do not have a well defined ecology (Bennion *et al*., 2001), occur frequently in the data set, and were not restricted to any one geographic region. For instance, *Staurosirella pinnata* was found in 12 streams that ranged in location from upland streams on the southern Fleurieu Peninsula to lowland streams north of the Adelaide plains. Also plotting low on axis 1 are *Aulacoseira granulata* and *Aulacoseira italica*. These taxa plot in a similar location to the Murray River samples (89 and 91), reflecting the fact that these are often the two most abundant taxa in Murray River plankton (Hotzel and Croome, 1996). A group of taxa which prefer alkaline conditions (see Gell 1995, Reid 1997, Gasse *et al*. 1995) such as *Epithemia adnata*, *Epithemia sorex*, *Rhopalodia musculus*, and *Achnanthes brevipes* plot high on axis 2.

3.4.2.3 Combined data set

The summary statistics for DCA of the combined data set of 101 samples is shown in table 3.4.

***Table 3.4**: Summary statistics of the combined data set (101 samples)*

Axes	1	2	3	4	Total inertia
Eigenvalues	0.639	0.559	0.435	0.375	13.083
Cumulative% variance of taxon data	4.9	9.2	12.5	15.4	

There was 15.4% of variance in taxa data explained by the first four axes. This compares to 19.1% and 17.7% respectively for the lake and stream data sets. Therefore increasing the number of samples in the data set did not result in any greater explanation of taxa variance. This is probably due to an increase in the size of the data set resulting in greater heterogeneity within the data set.

The DCA plot of sample scores is shown in figure 3.7. In summary, samples 1 - 50 represent lake sites and samples 51 - 101 represent stream sites. There does not appear to be a distinct split between lake and stream sites, with even representation of both data sets throughout the ordination diagram. There also does not appear to be any spatial or chemical explanation for the patterns of sample scores. For instance, sites with low EC (samples 11 and 68) plot next to sites with high EC (samples 54 and 59), and arid sites (samples 64 and 94) plot adjacent to samples taken from deep reservoirs in the Adelaide Hills (samples 10 and 15).

Figure 3.8 shows a DCA plot of diatom taxa scores. There is a strong grouping of planktonic taxa found low on axis one, including *Aulacoseira* spp., *Cyclotella* spp., *and Thalassiosira* spp., which are all commonly found in deep water lacustrine environments in south - eastern Australia (Gell 1995, Reid 1997, Tibby, 2000). Some of the more common facultative planktonic taxa, including *Pseudostaurosira brevistriata* and *Staurosirella pinnata* also plot low on axis 1. Located high on axis 1 are the majority of littoral taxa including the genera *Navicula, Nitzschia* and *Gomphonema*. Interestingly, although there wasn't a distinct split between lake and stream samples in figure 3.7, diatoms that occur predominantly in streams (*Rhoicosphenia abbreviata*, *Tabularia fasciculata* and *Bacillaria paradoxa*) all plot in similar ordination space.

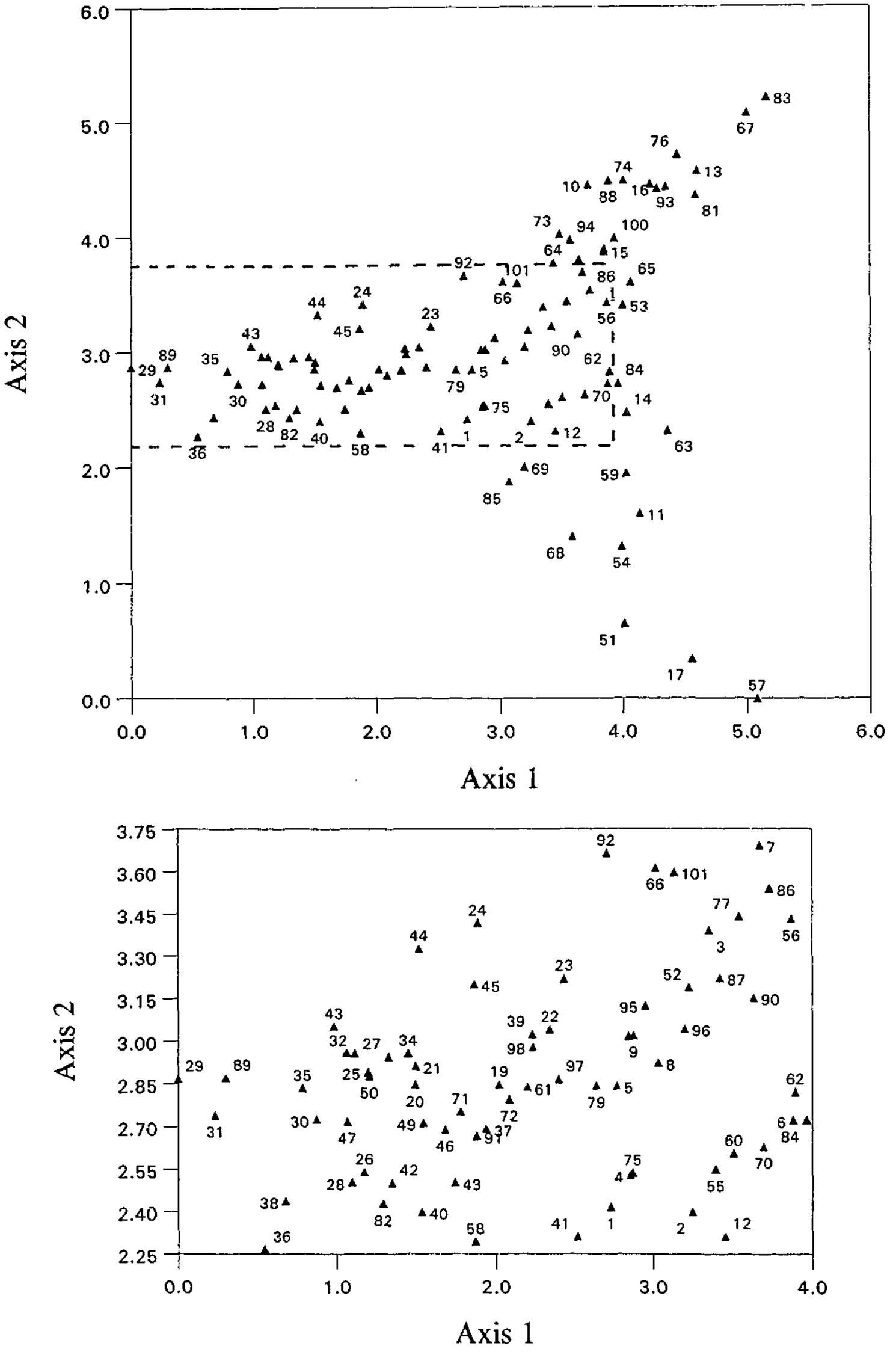


Figure 3.7 Ordination biplot of DCA site scores on axis 1 and axis 2 for the 101 sample combined data set. Inset in top biplot is enlarged in bottom biplot.

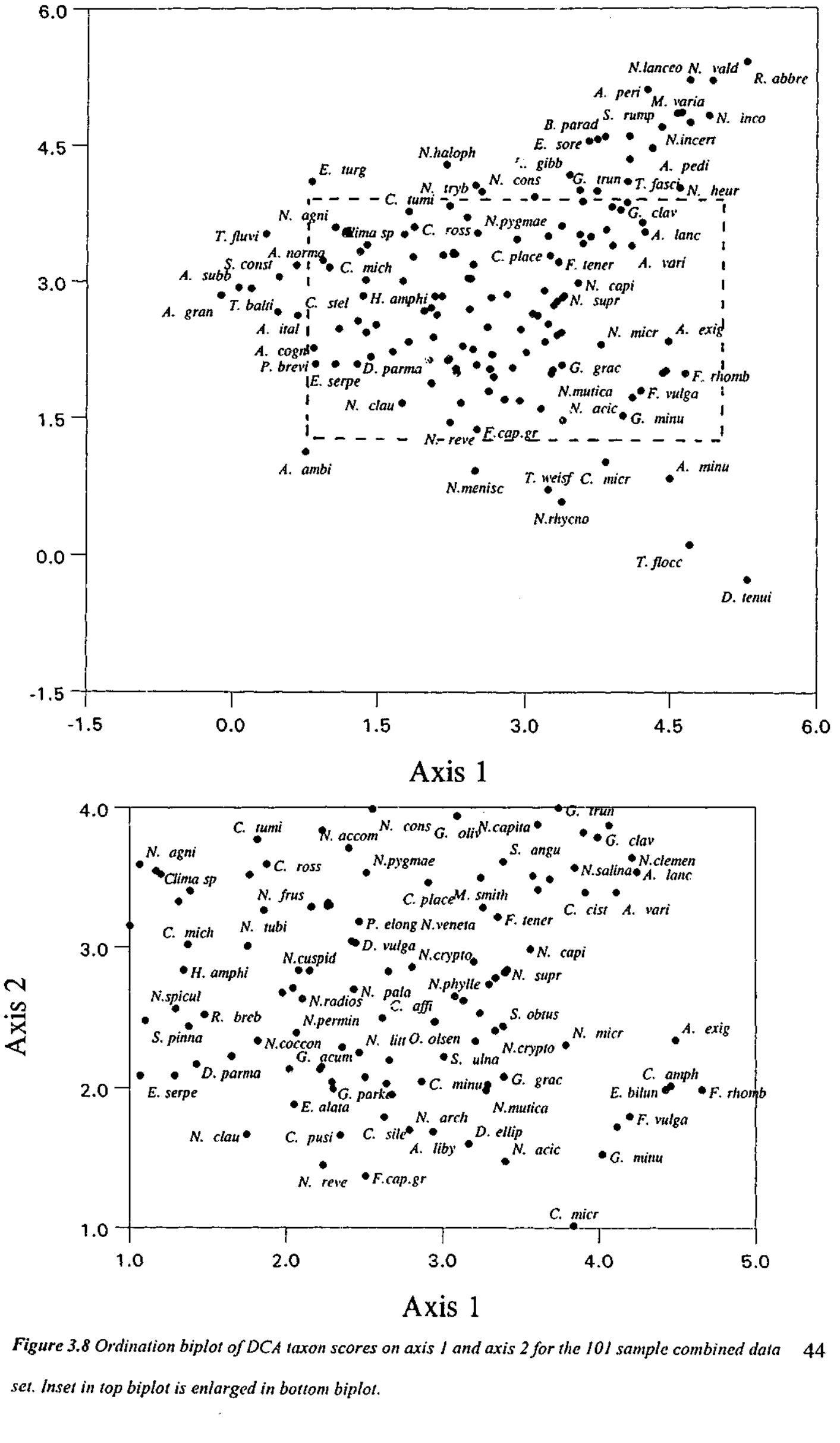


Figure 3.8 Ordination biplot of DCA taxon scores on axis 1 and axis 2 for the 101 sample combined data set. Inset in top biplot is enlarged in bottom biplot.

This pattern thus facilitates an interpretation of the plot of sample scores, as it appears that the samples are separated due to differences in diatom habitat preferences, such as ratios of planktonic and littoral taxa. It also seems likely that these differences could be a reflection of environmental parameters which were not measured. such as depth or flow, as there was little correlation between sites of similar measured chemistry on the sample score plot.

3.4.3 Murray River plankton survey results

Percentage data of the 16 samples is included in Appendix 2. The most common diatom taxa found were *Aulacoseira granulata* and varieties, *Aulacoseira ambigua, Aulacoseira subarctica* forma *subborealis, Staurosirella pinnata, Staurosira construens* varieties, *Pseudostaurosira brevistriata Cocconeis placentula, Cyclotella meneghiniana, Cyclostephanus tholiformis*. and *Actinocyclus normanii*.

Figure 3.9 illustrates the abundance of these major taxa with distance downstream, for the November 1997 and April 1998 sampling periods. It can be seen that *Aulacoseira* spp. dominates at sites closer to Mildura while *Staurosirella pinnata, Staurosira construens* varieties and the littoral *Cocconeis placentula* increasingly dominate closer to the Murray mouth. In reference to habitat origin, *Cocconeis placentula* was probably washed into the river channel from the river edge as it is unlikely this taxon could reproduce within the planktonic community due to its lack of buoyancy. However, the *Staurosirella pinnata, Pseudostaurosira brevistriata* and *Staurosira construens* varieties most probably do constitute part of the planktonic community in this section of the river, as evidenced by the fact that these taxa often formed long chains in the analysed samples, thus enabling buoyancy.

The estuarine taxon *Thallosiosira lacustris* is present in Lake Alexandrina plankton (up to 7%) and extends upstream, decreasing with distance, as far as Morgan. *Actinocyclus normanii,* which along with *Thallosiosira lacustris* has a higher salinity tolerance than the remaining recorded taxa (see Gell, 1995, Gasse *et al.*, 1995) has highest abundances in Lake Alexandrina, but remains present upstream to Mildura.

3.5 Environmental variable sampling results

3.5.1. Overview of the physico - chemistry results

Table 3.5 outlines the range of measured environmental variables for both the lake and stream data sets. Raw data are presented in Appendix 1. Frequency histograms are provided in figure 3.10, illustrating the spread of samples across the range for each environmental variable. Correlation coefficient matrices are presented in tables 3.6, 3.7 and 3.8, showing the degree of correlation between the environmental variables for all three data sets. Ions are expressed as a percentage of total cations and anions by mEq/cm. A discussion is provided summarising the major findings of the physico - chemical analysis, with statistical analysis of the results in Chapter 4.

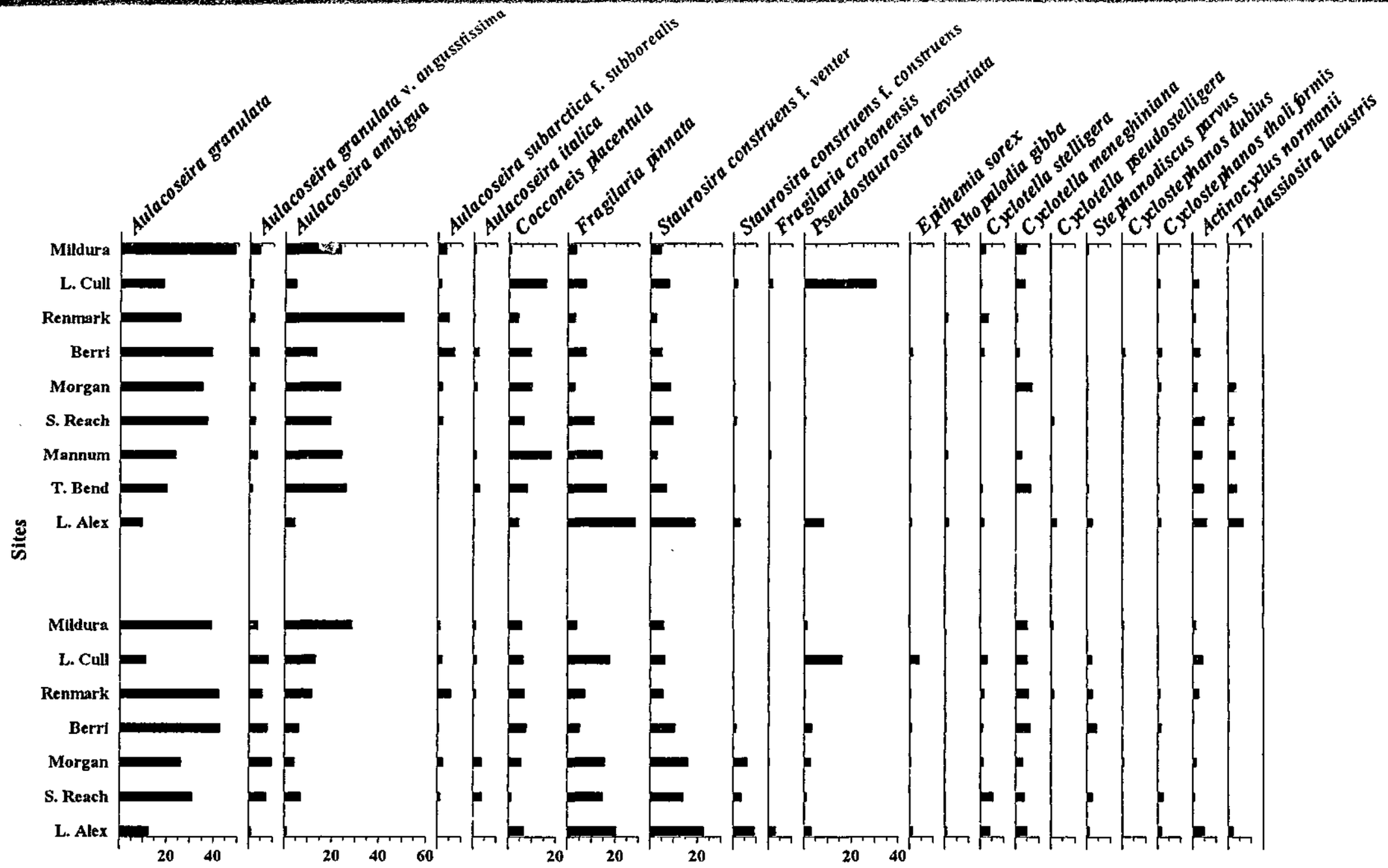


Figure 3.9: TILIA diagram showing the changing relative abundance of planktonic diatom taxa in the lower Murray River, from Mildura to Lake Alexandrina. The top half of the diagram represents the November 1997 sampling period and the bottom half of the diagram represents the April 1998 sampling period. L. Cull = Lake Cullulleraine, S. Reach = Swan Reach, L. Alex = Lake Alexandrina.

Table 3.5: Measured environmental variables and their ranges

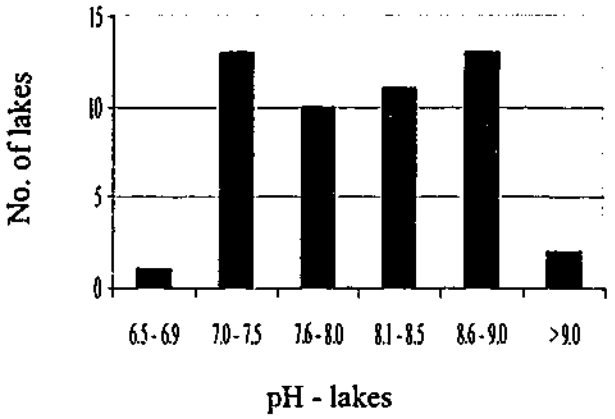
Variable	Lakes	Streams
pH	6.9 - 9.4	7.1 - 8.8
Temperature (°C)	10 - 28.8	12.2 - 28.0
Shade	n/a	0 - 95
Turbidity (NTU)	15 - 467	0.2 - 450
Dissolved oxygen (DO) (mg/L)	5.3 - 14.9	2.1 - 16.4
Soluble reactive phosphorus (SRP) (μg/L)	n/a	3 - 8900
Total phosphorus (TP) (μg/L)	20 - 1000	5 - 9480
Nitrate / nitrite (NO_X) (μg/L)	n/a	10 - 5470
Total Kjeldahl Nitrogen (TKN) (μg/L)	30 - 50 000	20 - 23 600
Electrical Conductivity (EC) (μS/cm)	200 - 10 190	98 - 6460
Sodium (% total cations)	39 - 74%	32 - 80%
Potassium (% total cations)	0.5 - 11%	0.2 - 9.8%
Magnesium (% total cations)	15 - 45%	12 - 39%
Calcium (% total cations)	3 - 29%	3 - 35%
Chloride (% total anions)	35 - 92%	16 - 92%
Sulphate (% total anions)	0.1 - 31 %	0.7 - 29%
Alkalinity (% total anions)	3 - 60%	1.5 - 72%

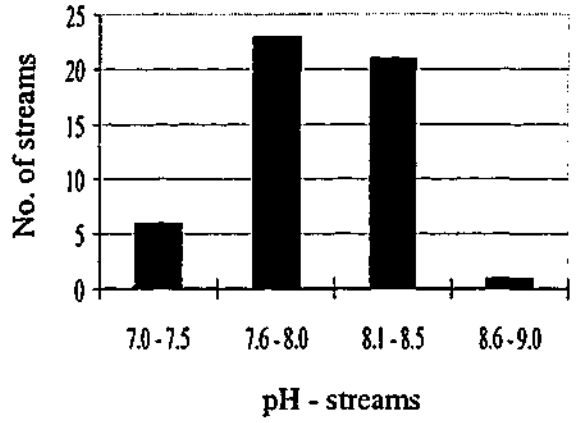
3.5.2. pH, dissolved oxygen, and temperature

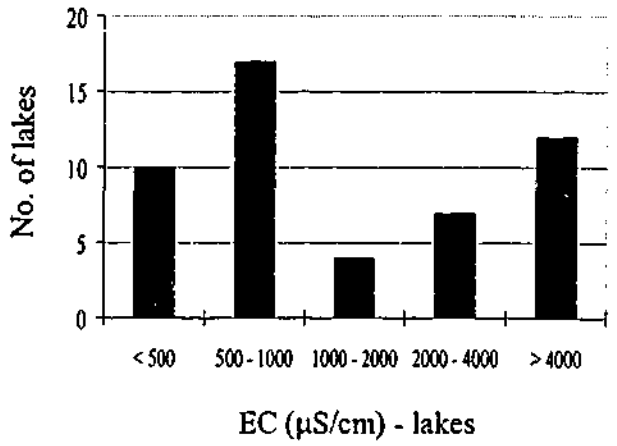
The stream data set has a more tightly constrained pH range than the lake data set, with only 1.7 pH units separating all 51 sites and more than 85% of samples having a pH between 7.6 and 8.5 (see figure 3.1). The 50 lakes are fairly evenly spread between 7.1 and 9.0 pH units, with only 2 samples having a pH > 9.0. There are no samples in either data set with a pH < 6.5. Generally, the lakes with the highest pH are located in the Sunraysia region surrounding Mildura, while the streams with the highest pH were located on the Fleurieu Peninsula, South Australia.

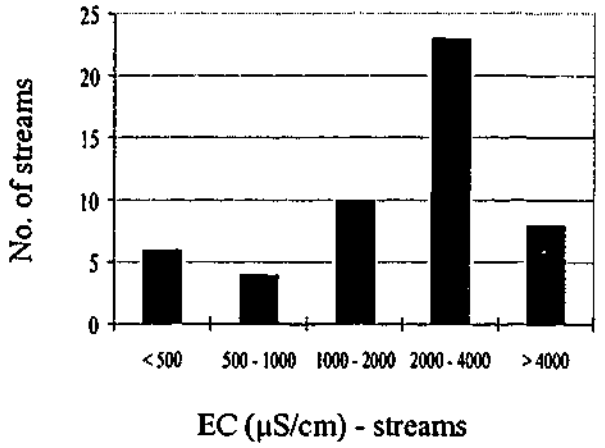
In examining the correlation matrices, pH is most highly correlated with EC and various ions within the lake data set, particularly $\%Cl^-$, and $\%Na^+$ and is most negatively correlated with $\%Ca^{2+}$ and $\%K^+$. The stream data set shows positive correlation between pH, EC and $\%Na^+$. The combined data set shows greatest negative correlation between pH and %Alk and $\%K^+$, with -0.78 and -0.73 respectively.

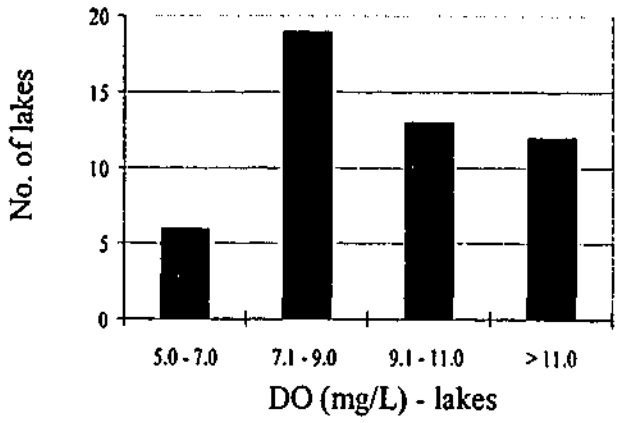
Dissolved oxygen has a wider range in the stream data set with 5 sites registering < 7 mg/L and one site as low as 2.1 mg/L. For both data sets, the majority of samples have a DO of between 8 and 11 mg/L. Lakes sites with the lowest DO (< 8 mg / L) were generally located in the Kerang region of Victoria, while sites with the highest DO (> 10 mg/L) generally located in the Mildura region. This may be due to the fact that most of

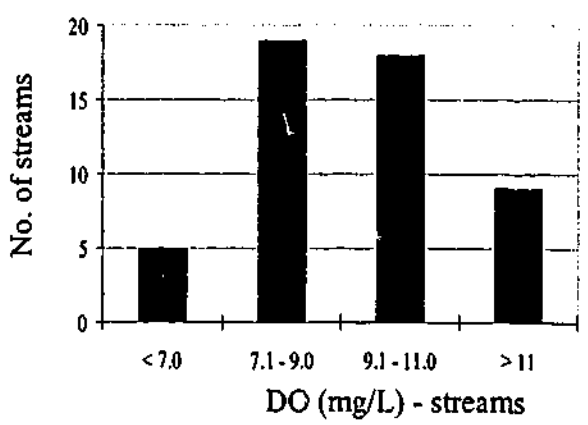


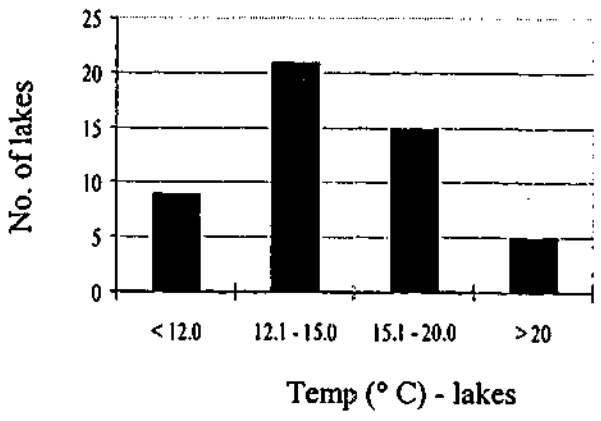












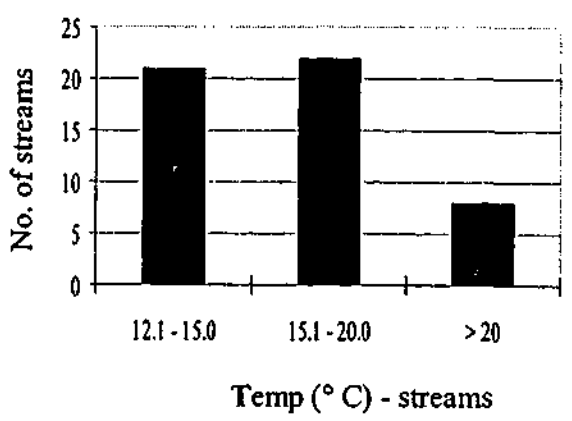
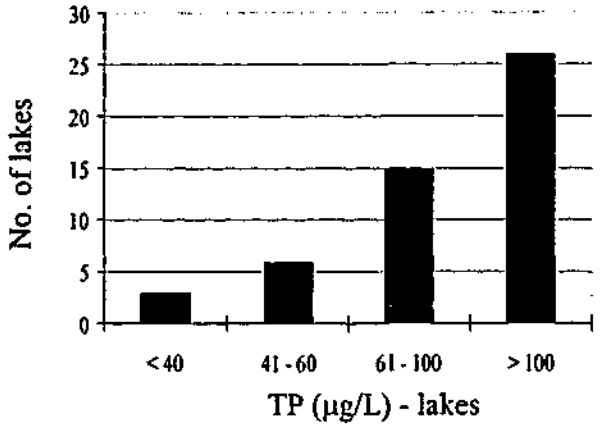
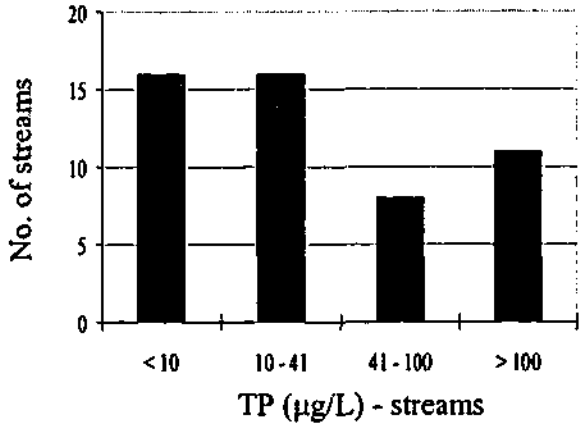
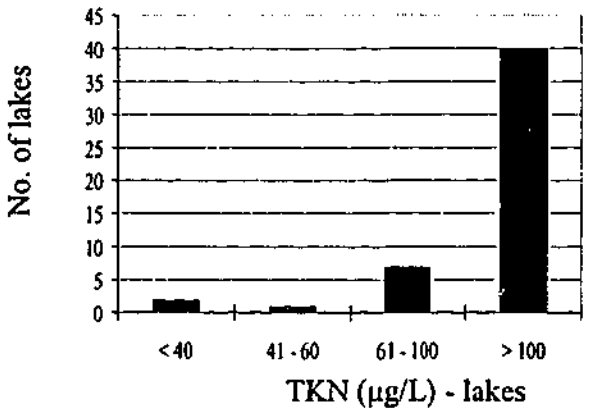
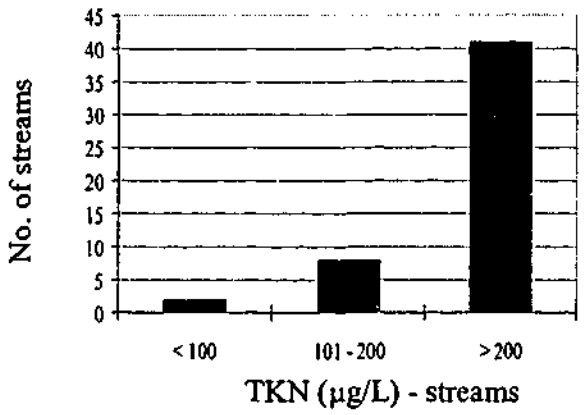


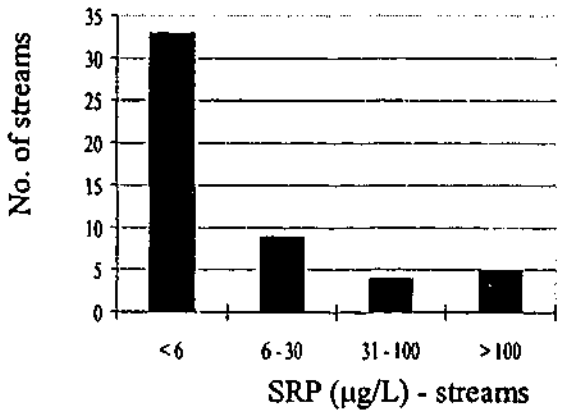
***Figure 3.10** Histograms showing the distribution of samples across the range of physico - chemical parameters for both the lake and stream data sets.*

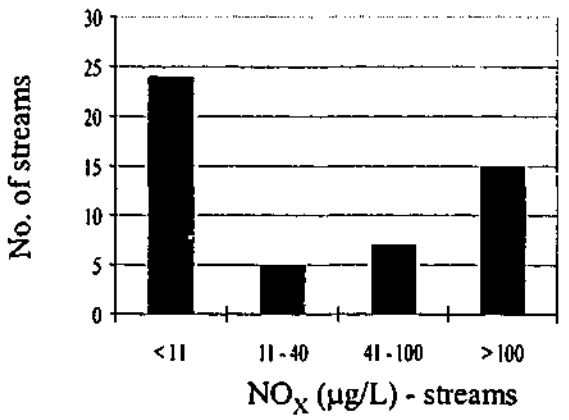


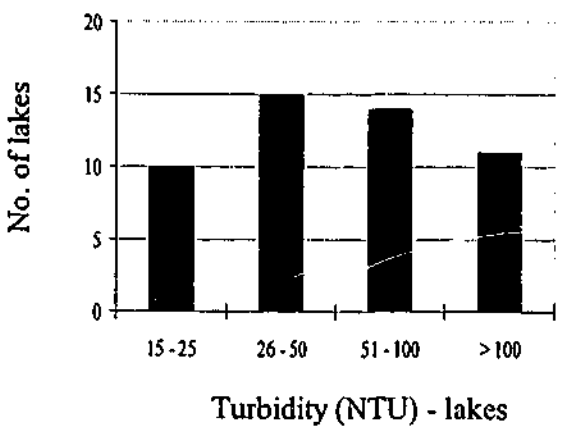












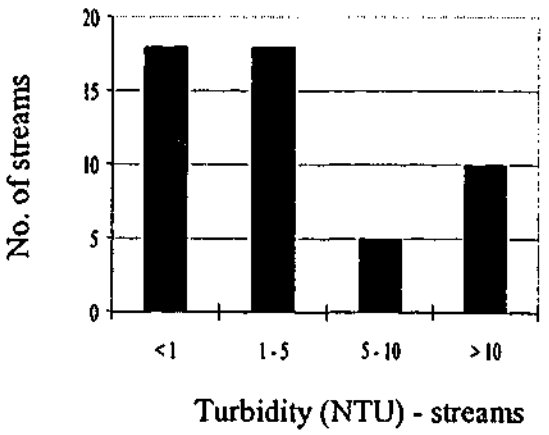
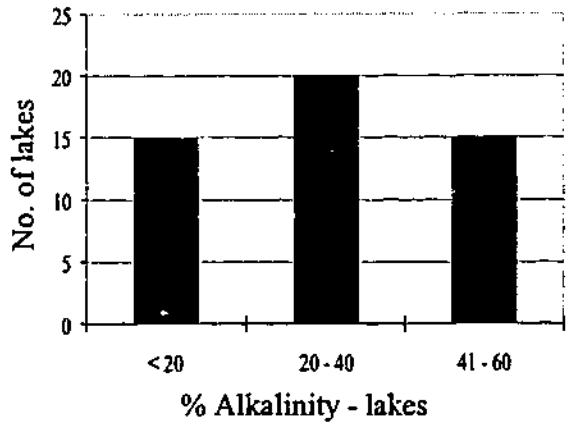
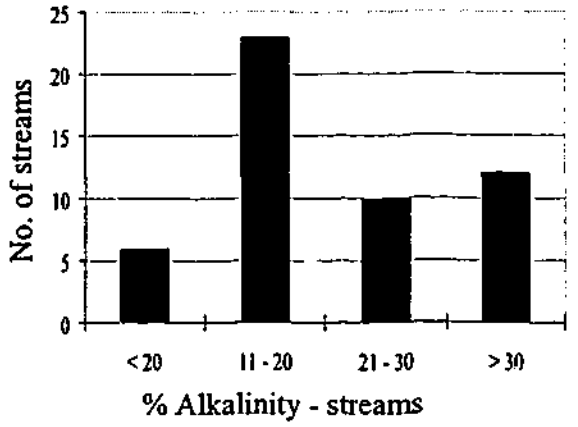
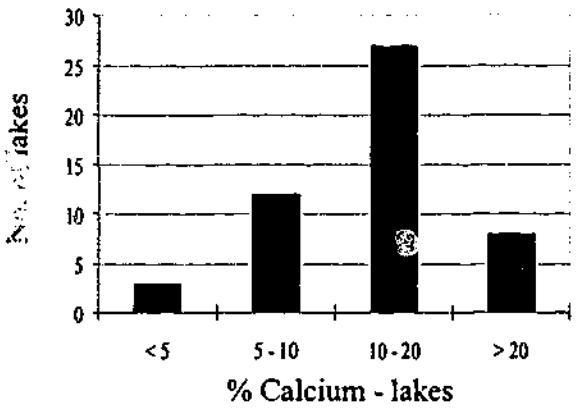
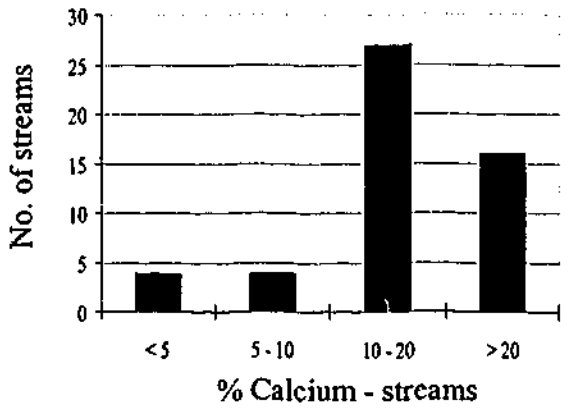


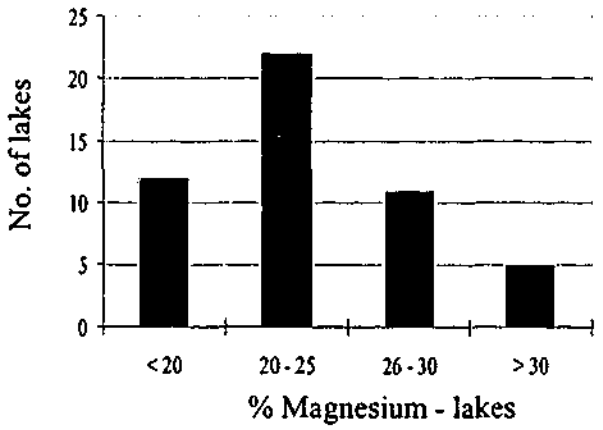
Figure 3.10 (cont): *Histograms showing the distribution of samples across the range of physico - chem'cal parameters for both the lake and stream data sets.*

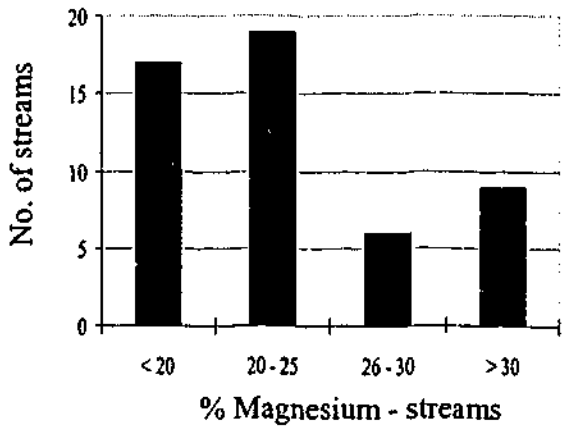


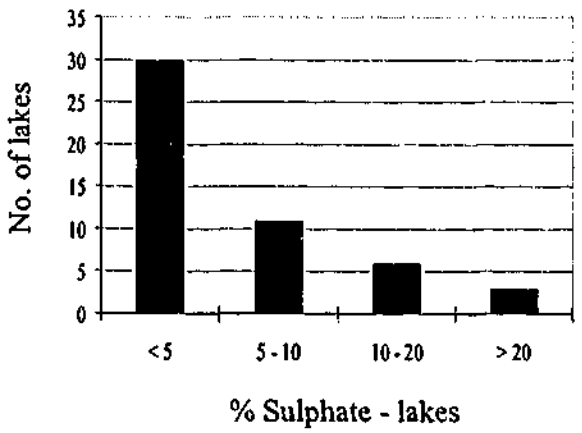












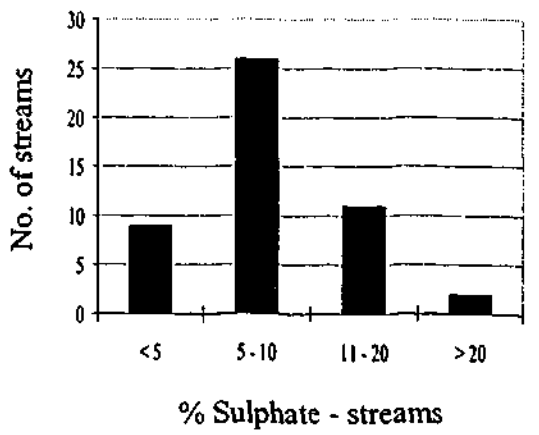
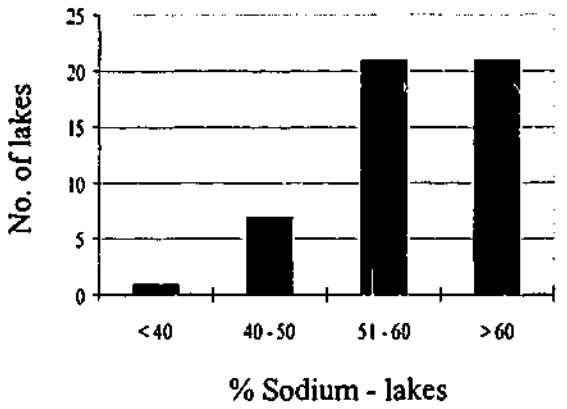
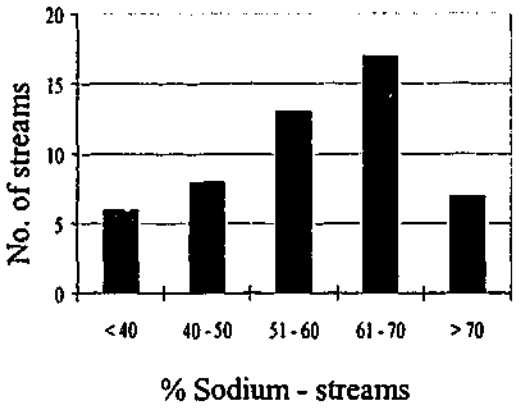
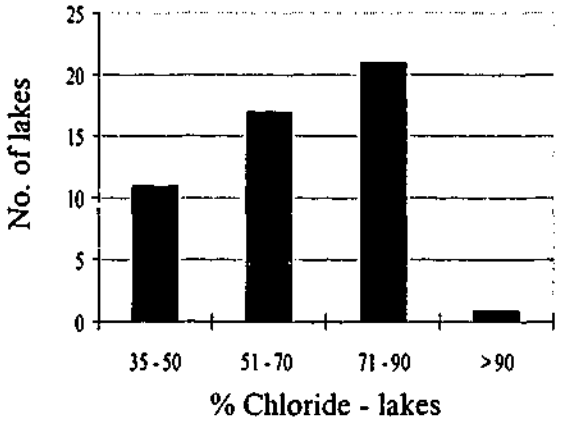
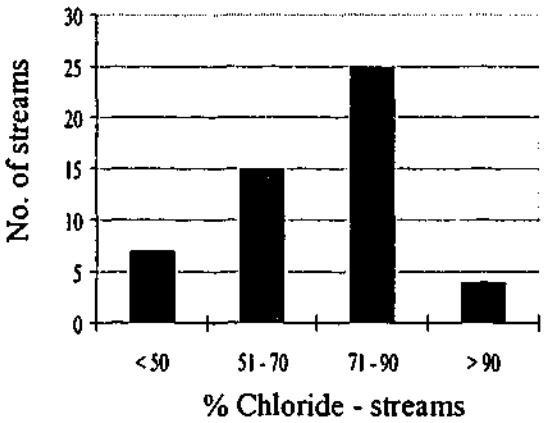


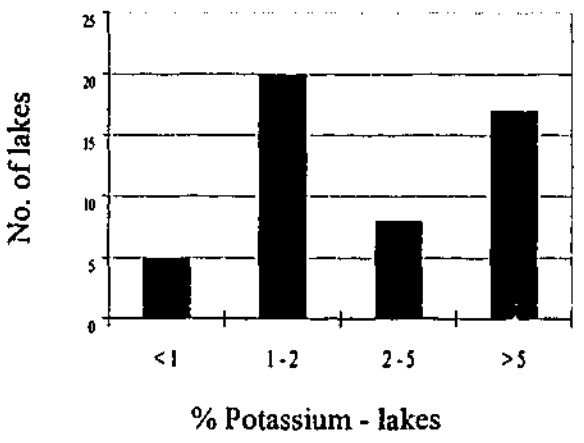
Figure 3.10 (cont): *Histograms showing the distribution of samples across the range of physico - chemical parameters for both the lake and stream data sets.*

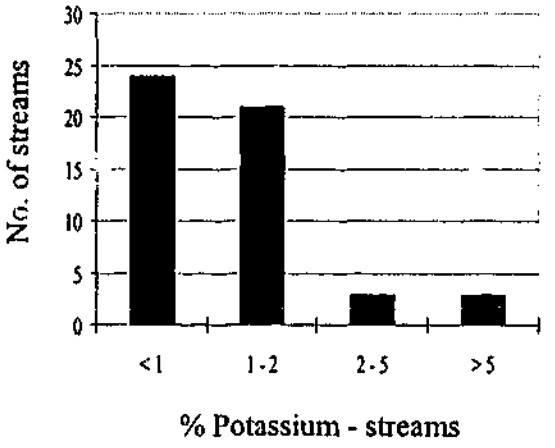












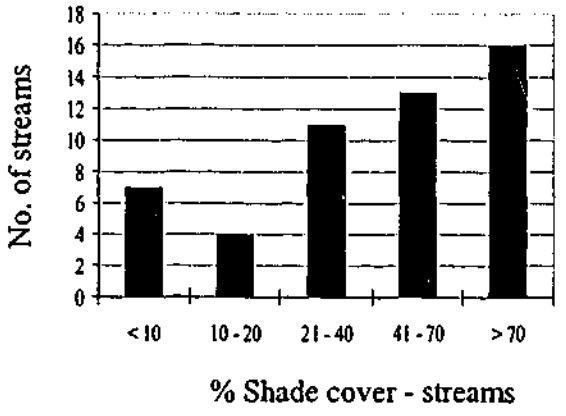


Figure 3.10 (cont): *Histograms showing the distribution of samples across the range of physico - chemical parameters in both the lake and stream data sets.*

Table 3.6: Correlation matrix of the lake data set environmental variables

pH	1													
EC	0.53	1												
DO	0.23	0.17	1											
TP	0.12	0.31	-0.09	1										
TKN	0.08	0.27	-0.21	0.83	1									
Turb	0.13	0.16	-0.18	0.16	0.02	1								
Temp	0.34	0.17	-0.14	0.06	-0.07	0.18	1							
% Alk	-0.41	-0.26	0.14	-0.1	-0.19	-0.2	-0.04	1						
% Cl^-	0.49	0.31	-0.12	0.17	0.25	0.11	0.09	-0.89	1					
% $SO4^{2-}$	-0.17	-0.08	-0.05	-0.2	-0.12	0.08	-0.11	-0.26	-0.2	1				
% Ca^{2+}	-0.52	-0.48	0.02	-0.1	-0.17	-0.3	0.06	0.68	-0.6	-0.12	1			
% Mg^{2+}	-0.06	0.03	-0.05	-0.1	0.04	0.01	-0.19	-0.45	0.47	-0.03	-0.4	1		
% Na^+	0.61	0.46	-0.08	0.22	0.17	0.25	0.13	-0.48	0.47	0.04	-0.7	-0.33	1	
% K^+	-0.49	-0.31	0.27	-0.1	-0.18	-0.2	-0.15	0.76	-0.9	0.18	0.57	-0.21	-0.7	1
	pH	**EC**	**DO**	**TP**	**TKN**	**Turb**	**Temp**	**% Alk**	**% Cl^-**	**% $SO4^{2-}$**	**% Ca^{2+}**	**% Mg^{2+}**	**% Na^+**	**% K^+**

Table 3.7: Correlation matrix of the stream data set environmental variables

Shade	1																
Turb	-0.18	1															
pH	-0.03	0.02	1														
Temp	-0.19	0.10	0.02	1													
DO	-0.16	0.16	0.50	.25	1												
EC	0.06	-0.39	0.10	-0.35	-0.16	1											
SRP	-0.21	-0.04	-0.19	-0.16	-0.60	0.03	1										
TP	-0.22	-0.03	-0.19	-0.17	-0.60	0.03	0.86	1									
NOX	-0.39	-0.08	-0.11	-0.15	-0.16	-0.10	0.55	0.55	1								
TKN	-0.19	-0.01	-0.06	-0.11	-0.48	0.10	0.78	0.79	0.29	1							
% Alk	0.02	0.46	0.10	0.13	0.06	-0.20	-0.07	-0.08	-0.13	-0.01	1						
% Cl^-	-0.09	-0.42	-0.17	-0.11	-0.10	0.20	0.09	0.10	0.15	0.02	-0.96	1					
% $SO4^{2-}$	0.25	0.11	0.29	-0.01	0.14	-0.10	-0.08	-0.09	-0.12	-0.05	0.38	-0.62	1				
% Ca^{2+}	0.02	0.25	0.20	0.00	0.01	0.01	-0.07	-0.08	-0.15	0.04	0.84	-0.85	0.46	1			
% Mg^{2+}	0.12	-0.11	0.10	-0.02	0.11	0.04	-0.16	-0.16	-0.12	-0.16	0.26	-0.38	0.52	0.30	1		
% Na^+	-0.08	-0.21	-0.20	-0.01	-0.07	0.34	0.11	0.12	0.16	0.01	-0.80	0.86	-0.62	-0.91	-0.66	1	
% K^+	0.08	0.57	0.08	0.14	0.05	-0.18	0.10	0.09	0.00	0.24	0.51	-0.52	0.31	0.30	-0.09	-0.32	1
	Shade	**Turb**	**pH**	**Temp**	**DO**	**EC**	**SRP**	**TP**	**NOX**	**TKN**	**% Alk**	**% Cl^-**	**% $SO4^{2-}$**	**% Ca^{2+}**	**% Mg^{2+}**	**% Na^+**	**% K^+**

Table 3.8: Correlation matrix of the combined data set environmental data set

Turb	1													
pH	0.11	1												
Temp	0.07	0.29	1											
DO	0.00	-0.09	-0.06	1										
EC	-0.07	0.33	0.01	-0.01	1									
TP	0.00	-0.08	-0.05	-0.28	0.08	1								
TKN	0.03	0.07	-0.06	-0.24	0.24	0.53	1							
%Alk	0.08	-0.73	-0.20	0.31	-0.23	0.03	-0.08	1						
%Cl^-	-0.22	0.24	0.06	-0.15	0.27	0.08	0.13	-0.65	1					
%$SO4^{2-}$	-0.01	-0.50	-0.15	0.21	-0.13	-0.03	-0.11	0.46	-0.38	1				
%Ca^{2+}	-0.01	-0.06	0.10	-0.02	-0.22	-0.02	-0.08	0.31	-0.67	0.13	1			
%Mg^{2+}	-0.03	0.08	-0.10	-0.02	0.04	-0.06	0.00	-0.13	0.03	0.10	0.00	1		
%Na^+	-0.01	0.31	0.10	-0.13	0.25	0.03	0.08	-0.53	0.71	-0.37	-0.78	-0.47	1	
%K^+	0.06	-0.78	-0.27	0.36	-0.24	0.05	-0.05	0.90	-0.49	0.54	0.06	-0.14	-0.39	1
	Turb	**pH**	**Temp**	**DO**	**EC**	**TP**	**TKN**	**%Alk**	**%Cl^-**	**%$SO4^{2-}$**	**%Ca^{2+}**	**%Mg^{2+}**	**%Na+**	**%K^+**

the lakes in the Sunraysia region form part of a vast irrigation system and as such are relatively well aerated and have shorter residence times than the more natural systems in Kerang. Dissolved oxygen does not appear to be correlated, either positively or negatively, with any of the other variables in either the lake or combined data sets. In the stream data set, however, it negatively correlates with SRP and TP (both -0.6) and TKN (0.48) and positively correlates with pH (0.5).

Temperatures are generally lower in the lake data set although this may be a product of sampling predominantly being undertaken during the winter months while the streams were sampled during spring and autumn. Lowest temperatures are found in the Adelaide Hills subset (all sampled in June and July) while the highest temperatures are found in the Kerang lakes. For the stream data set, lowest temperatures are recorded in the Deep Creek region of the Fleurieu Peninsula, a highly forested area, while the highest are recorded in the far north arid zones of the state. Temperature shows a positive correlation with pH in the lake data set and EC in the river data set.

Turbidity was much lower in the stream data set with over 80% of samples registering < 10 NTU while none of the samples in the lake data set < 15 NTU turbidity. The majority of lakes measured between 20 and 100 NTU. The difference between the two data sets is probably due to the fact that most of the streams have very low volume and velocity with rocky substrates, conditions that are conducive to low turbidities. The lakes, however, are primarily located on the riverine plains with a high density of agriculture, or contain substantial quantities of piped Murray River water, which is characteristically highly turbid (Shafron *et al.* 1990). There were three streams which recorded turbidities > 350 NTU and these are all located in the northern section of the Flinders Ranges. Turbidity is positively correlated with $\%Na^+$ in the lake data set (0.25), and $\%K^+$(0.57), $\%Ca^{2+}$(0.25) and %Alk (0.46) in the stream data set. Interestingly, there appears to be no relationship between turbidity and nutrients, particularly TP and TKN. This may suggest that the undissolved component of these nutrient measures is perhaps not predominantly attached to suspended solids.

Shade was an additional measure for the stream data set and ranged from 0% to 95% coverage. As expected, sites with the lowest shade cover are located in the arid and semi-arid northern regions of South Australia while those with the highest level of cover are located on the Fleurieu Peninsula. Shade was not substantially correlated with any other variables, except perhaps for NO_X which shows a negative correlation at -0.39. This contrasts with findings by Wade (pers. comm.) on the relationship between NO_X and shade on streams in the Flinders Ranges where it was found that as shade was increased, biological uptake of NO_X decreased.

3.5.3. Nutrients

Maximum TP concentrations were much greater in the stream data set than the lake data set (9480 μg/L compared to 1000 μg/L), although the stream data set has a much higher abundance of sites within the oligotrophic range (with 16 sites < 10 μg/L). Nearly 95% of lake sites are classified as being either eutrophic or hypereutrophic, with over 50% of sites having a TP concentration of more than 100 μg/L. The lake data set is therefore greatly skewed towards the upper end of the TP gradient, but when the two data sets are

combined, samples are spread much more evenly across the gradient. Lakes from the different geographic regions were equally represented across the gradient, with no region displaying a greater abundance of low or high TP concentrations. The lake with the highest TP value, at 1000 μg/L, was Lake Lyndger in the Kerang region, while the stream site with the highest TP (9480 μg/L) was Mt. Barker Creek in the Adelaide Hills, which receives effluent from a sewage treatment plant. Total phosphorus was most strongly correlated with TKN in the lake data set (0.83) but does not show any apparent correlation to any other variables. In the stream data set, it is negatively correlated with DO (-0.6), directly correlated with SRP (1.0), and positively correlated with TKN (0.78). In the combined data set, all of these relationships were weaker.

Soluble reactive phosphorus, in the stream data set, had a similar range to that of TP. However, there was a greater abundance of sites with values < 6 μg/L (more than 65% of all sites), with only 5 sites having a SRP concentration greater than 100 μg/L. Soluble reactive phosphorus was strongly positively correlated with all other nutrients, particularly TP where a correlation of 0.86 was obtained.

The majority of sites in both the lake and stream data sets had extremely high TKN values, with more than 80% of sites having concentrations > 100 μg/L. For the stream data set, this contrasts with very low TP values. For instance, while more than 30 stream sites contained < 40 μg/L of TP, only 2 sites had TKN values < 100 μg/L, implying that virtually all the nutrient loading in the streams is in the form of nitrogen rather than phosphorus. Sites with the highest TKN values mirror the sites with the highest TP values for both data sets.

There was notably a greater abundance of sites with < 100 μg/L NO_X than sites with < 100 μg/L of TKN, suggesting that most of the nitrogen in the streams is in the form of particulate or organic material. Eight Mile Creek was an outlier in that NO_X far exceeded TKN (5100 μg/L compared to 60 μg/L). It is interesting that NO_X did not correlate as strongly as SRP with the other nutrients, showing only a 0.55 correlation with SRP and TP and a 0.29 correlation with TKN.

3.5.4. Salinity and ionic concentrations

Salinity was analysed as a measure of Electrical Conductivity rather than Total Dissolved Solids. The lakes had a greater range of EC and also a higher abundance of sites below 500 μS/cm than the stream data set. Over 45% of streams recorded an EC between 2000 and 4000 μS/cm. The geographic spread of lakes with high EC are relatively even across all three regions with Barker Inlet Wetland, on the Adelaide Plains, registering the highest EC (10 190 μS/cm). The stream data set has a similar uniformity of distribution with all regions displaying a range of fresh to brackish sites, with the exception of the northern Flinders Ranges where all streams are relatively fresh.

The relationship between EC, the ions and pH varied. For instance, EC correlates with pH in the lake data set at 0.53, but only 0.1 in the stream data set and 0.33 in the combined data set. Interestingly, there is little or no correlation between EC and the ions in the stream data set (with the exception of $\%Na^+$). Even within the lake data set, the correlation between ions and EC does not exceed 0.48.

Table 3.9 shows the dominant ionic composition for all sites. The dominant cation for all sites was $\%Na^+$ with the second most common being $\%Mg^{2+}$. The most dominant anion was $\%Cl^-$ followed by alkalinity (as measured by $(H)CO_3$).

Table 3.9 Ionic ratios for the lake and stream sites

Site	Dominant ions
1	$Na^+ Ca^{2+}: Cl^- (H)CO_3$
2	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
3	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
4	$Na^+: Cl^- (H)CO_3$
5	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
6	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
7	$Na^+ Mg^{2+}: (H)CO_3 Cl^-$
8	$Na^+ Mg^{2+}: Cl (H)CO_3 SO_4^{2-}$
9	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
10	$Na^+ Mg^{2+} Ca^{2+}: Cl^- (H)CO_3$
11	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
12	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
13	$Na^+ Ca^{2+} Mg^{2+}: Cl^- (H)CO_3$
14	$Na^+ Mg^{2+}: Cl^- SO_4^{2-} (H)CO_3$
15	$Na^+ Mg^{2+}: Cl^- SO_4^{2-} (H)CO_3$
16	$Na^+ Ca^{2+} Mg^{2+}: Cl^- (H)CO_3$
17	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
18	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
19	$Na^+ Mg^{2+}: Cl^-$
20	$Na^+ Mg^{2+}: Cl^-$
21	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
22	$Na^+ Mg^{2+}: Cl^-$
23	$Na^+ Mg^{2+}: Cl^-$
24	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
25	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
26	$Na^+ Mg^{2+}: Cl^-$
27	$Na^+ Mg^{2+}: Cl^-$
28	$Na^+ Mg^{2+}: Cl^-$
29	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
30	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
31	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
32	$Na^+ Mg^{2+}: Cl^-$
33	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
34	$Na^+ Mg^{2+}: Cl^-$
35	$Na^+: Cl^- (H)CO_3$
36	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
37	$Na^+: Cl^- (H)CO_3$
38	$Na^+: (H)CO_3 Cl^-$
39	$Na^+: Cl^- (H)CO_3$
40	$Na^+ Mg^{2+}: Cl^-$
41	$Na^+ Mg^{2+}: Cl^-$
42	$Na^+ Ca^{2+}: Cl^- (H)CO_3$
43	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
44	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
45	$Na^+ Mg^{2+}: Cl^-$
46	$Na^+ Mg^{2+}: Cl^-$
47	$Na^+: (H)CO_3 Cl^-$
48	$Na^+: (H)CO_3$
49	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
50	$Na^+: Cl^- (H)CO_3$

Site	Dominant ions
51	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
52	$Na^+ Mg^{2+}: Cl^-$
53	$Na^+ Mg^{2+}: Cl^- (H)CO_3 SO_4^{2-}$
54	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
55	$Na^+ Mg^{2+}: Cl^-$
56	$Na^+ Mg^{2+}: Cl^-$
57	$Na^+ Ca^{2+} Mg^{2+}: Cl^- (H)CO_3$
58	$Na^+ Mg^{2+}: Cl^-$
59	$Na^+ Ca^{2+}: Cl^- (H)CO_3$
60	$Na^+ Ca^{2+} Mg^{2+}: Cl^- (H)CO_3$
61	$Na^+ Mg^{2+} Ca^{2+}: Cl^- (H)CO_3$
62	$Na^+: Cl^- (H)CO_3$
63	$Na^+: Cl^-$
64	$Na^+ Ca^{2+} Mg^{2+}: Cl^- (H)CO_3$
65	$Na^+ Mg^{2+} Ca^{2+}: Cl^- (H)CO_3$
66	$Na^+ Mg^{2+} Ca^{2+}: Cl^- (H)CO_3$
67	$Na^+ Mg^{2+}: Cl^-$
68	$Na^+ Mg^{2+}: Cl^- (H)CO_3$
69	$Na^+ Mg^{2+}: Cl^-$
70	$Na^+ Mg^{2+}: Cl^-$
71	$Na^+ Mg^{2+}: Cl^-$
72	$Na^+ Ca^{2+}: Cl^-$
73	$Na^+ Mg^{2+}: Cl^-$
74	$Na^+: Cl^- (H)CO_3$
75	$Na^+: Cl^-$
76	$Na^+: Cl^-$
77	$Na^+ Mg^{2+} Ca^{2+}: Cl^- (H)CO_3$
78	$Na^+ Mg^{2+}: Cl^-$
79	$Na^+: Cl^- (H)CO_3$
80	$Na^+ Mg^{2+}: Cl^-$
81	$Na^+: Cl^-$
82	$Na^+ Mg^{2+}: Cl^-$
83	$Na^+: Cl^-$
84	$Na^+ Mg^{2+}: Cl^-$
85	$Na^+: Cl^-$
86	$Na^+ Mg^{2+}: Cl^-$
87	$Na^+: Cl^-$
88	$Na^+ Mg^{2+}: Cl^-$
89	$Na^+: Cl^-$
90	$Na^+ Mg^{2+}: Cl^-$
91	$Na^+: Cl^-$
92	$Na^+ Mg^{2+}: Cl^-$
93	$Na^+ Ca^{2+} Mg^{2+}: Cl^- (H)CO_3$
94	$Na^+ Mg^{2+}: Cl^-$
95	$Na^+ Ca^{2+} Mg^{2+}: (H)CO_3 Cl^-$
96	$Na^+ Ca^{2+}: (H)CO_3$
97	$Na^+ Ca^{2+}: Cl^- (H)CO_3$
98	$Na^+ Mg^{2+}: Cl^-$
99	$Na^+ Mg^{2+}: Cl^-$
100	$Na^+ Mg^{2+}: Cl^-$
101	$Na^+: Cl^-$

Chapter 4 - Development of a Weighted Averaging transfer function

4.1 Introduction

This section of the study follows the relatively standardised procedure for developing diatom transfer functions for environmental variables based on Weighted Averaging regression and calibration (eg. Birks *et al.* 1990, Charles 1990, Fritz *et al.* 1991, Hall and Smol 1992, Bennion 1993, Gasse *et al.* 1995). Canonical Correspondence Analysis was used to identify which environmental parameters explained the greatest diatom variance, with the significance of this influence determined. Weighted Averaging regression techniques were then applied, which allowed the optima and tolerances for different environmental variables to be determined for individual diatom taxa.

The lake and stream data sets were analysed separately to examine whether particular variables were more important in one system than the other in the determination of diatom assemblage. The data sets were then integrated to examine whether expanding the number of samples improved the performance of the transfer functions.

4.2 Canonical Correspondence Analysis

The relationship between diatom assemblages and the measured environmental variables was explored using Canonical Correspondence Analysis (CCA). Canonical Correspondence Analysis is a multivariate direct gradient analysis technique which is based on canonical or constrained ordination. The method assumes that the abundance of a taxon is a symmetrical, unimodal function of position along environmental gradients (ter Braak, 1987). Canonical Correspondence Analysis is an extension of Correspondence Analysis and orders samples and taxa along axes, with the axes constructed to maximise dispersion of the samples and taxa. The biggest advantage of CCA over unconstrained ordination techniques is that the position of a sample or taxon is not only determined by the taxa present in the samples but is also a function of a defined set of environmental variables, thus providing a visual and mathematical expression of the degree of association between taxa and environmental variables (Charles and Smol, 1994).

It must be stressed that the use of CCA was of an exploratory nature only. The main objective behind using this type of ordination analysis in the development of a transfer function is to identify which environmental variable(s) are suitable for Weighted Averaging regression and calibration. It may be argued that this step is somewhat redundant as the cross - validation testing procedures associated with Weighted Averaging (i.e.: jackknifing or bootstrapping) can be used to identify which variables exert significant influence over diatom communities and those that will provide reliable information for the generation of taxon optima. The

advantage of CCA however, is that it may highlight variables that are having a significant influence on diatom taxa but were not considered in the original framework of the study.

For each data set, an initial CCA was conducted with the full range of measured variables, using the program CANOCO 4.02 (ter Braak and Smilauer, 1997). All variables, except pH, were $\log_{10}(x+1)$ transformed for CCA analyses in order to reduce skewness. All default functions were selected (including intersample distance and Hill's Scaling) and rare taxa were downweighted. Results from other key studies at each different stage of the transfer function process are also included. However, not all published studies report the full range of summary statistical results.

For the purpose of diagram interpretation, the CCA biplots are described as having four quadrants. Quadrant one is the upper right quadrant, quadrant two is the bottom right quadrant, quadrant three is the bottom left quadrant and quadrant four is the upper left quadrant.

4.2.1. Analysis of the lake data set

A CCA was undertaken on the lake data set with 14 variables, 50 samples, and 141 diatom taxa. The summary results are supplied in table 4.1.

Table 4.1 Summary statistics for CCA of full lake data set of 50 samples and 14 variables

Axes	1	2	3	4	Total inertia
Eigenvalues	.595	.420	.361	.291	12.581
Taxon - environment correlations	.936	.882	.911	.945	
Cumulative percentage variance -					
of taxon data	4.7	8.1	10.9	13.2	
of taxon - environment relationship	17.3	29.5	40.1	48.5	
Sum of all unconstrained eigenvalues					12.581
Sum of all canonical eigenvalues					3.434

Eigenvalues measure the weight of the axes, and not the significance (which can be assessed at a later stage through Monte Carlo testing). The eigenvalue of axis 1 (or any other axis) divided by the sum of all eigenvalues provides the proportion of variation of the taxon or diatom data explained by axis 1. The sum of all canonical eigenvalues divided by the sum of all unconstrained eigenvalues provides the proportion of variation explained by all the explanatory variables. Furthermore, if the eigenvalue of axis 1 is divided by the sum of canonical eigenvalues, the result is the proportion of variation explained by axis one with respect to the variation explained by all the explanatory variables (ter Braak, 1990).

The summary results (table 4.1) show that the 14 environmental variables explained 13.2% of the variance in taxon data in the first four canonical axes. This compares to results of 29.3% in Bennion's (1993) data set from ponds in south - east England, 9.6% found by Gell (1995) for a lake data set from Western Victoria, 12.8% in Reed (1995) for a lake data set from Spain , 27.1% in Reid (1997) for billabongs along the upper

Murray River, 17% for the SWAP data set (Stevenson *et al.*, 1991), and 38.5% in Tibby (2000) for a reservoir data set from NSW and Victoria. The disparity between the sum of all eigenvalues (12.581) and the sum of canonical eigenvalues (3.434) means that the measured environmental variables accounted for only 27.3% of all variation in diatom assemblages. This compares to 15.4% in Gell (1995), 26.1% in Reed (1995) and 32.7% in Reid (1997). Although this data set compares favourably to these data sets in terms of total variance that can be attributed to the environmental variables, this is still quite low.

Figure 4.1 shows the CCA bi-plot of environmental variables. In this figure, the vector for an environmental variable points in the direction of maximum taxon variation associated with the variable, and its length is proportional to the strength of correlation with changes in the diatom assemblage. Environmental variables with long vectors are more closely related to the patterns of biological variation displayed in the diagram. Therefore, the longer the arrow, the more correlated that variable is with changes to the diatom community. Small angles between environmental variables and axis 1 also suggest high correlations with changes in patterns in the biological data. As suggested by the statistical output, there is no primary variable constrained to axis one, with only 4.7% of taxon variance attributable to the first axis. pH, $\%Na^+$ and $\%K^+$ are the variables most associated with axis 1, with the second axis being represented primarily by EC. Most of the ions are spread between the two axes. Both nutrient variables, TP and TKN are correlated with axis 1 but the short length of their vectors show that they are less important than the other variables in explaining variation in taxon data.

Figure 4.2 shows a CCA biplot of the sample scores for the lake data set. The smallest number of samples occur in quadrant one, principally representing sites with high DO. Most of the samples that plot in quadrant two are samples that occur at the low end of the pH gradient (< 7.5), while samples in quadrant four generally occur at the high end of the pH gradient (> 7.5). For example, samples 4 and 17 both have a pH of 7.4, while samples 20 and 49 have a pH of 8.8 and 9.0 respectively. Quadrant three represents samples with high nutrient concentrations, with sample 33, furthest from the origin, having 200 µg/L TP and 4000 µg/L TKN. Conversely, the majority of samples in quadrant one have relatively low levels of nutrients. Regional differences between samples do not seem to be having a major effect on diatom distributions. The only exception to this is perhaps the Adelaide group of samples which generally plot in quadrants 1 and 2, which is probably due to these samples having a relatively lower pH than samples from the Mildura and Kerang regions.

Figure 4.3 shows a biplot of taxa scores for the lake data set. When examining the diatom taxa spread along the pH vector, it can be seen that there are a number of taxa that are associated with either low or high pH conditions. At the more acidic end of the pH vector are *Diatoma tenuis, Synedra rumpens, Navicula lanceolata* and *Gomphonema affine.* Taxa that occurred in alkaline conditions include *Mastogloia smithii, Nitzschia frustulum, Nitzschia lacuum* and *Cyclotella meneghiniana*. There appears to be an abundance of planktonic taxa associated with the high end of the nutrient vectors, including *Cyclotella meneghiniana, Cyclotella stelligera, Cyclotella michigiana, Cyclotella rossii, Thalassiosira baltica* and *Actinocyclus*

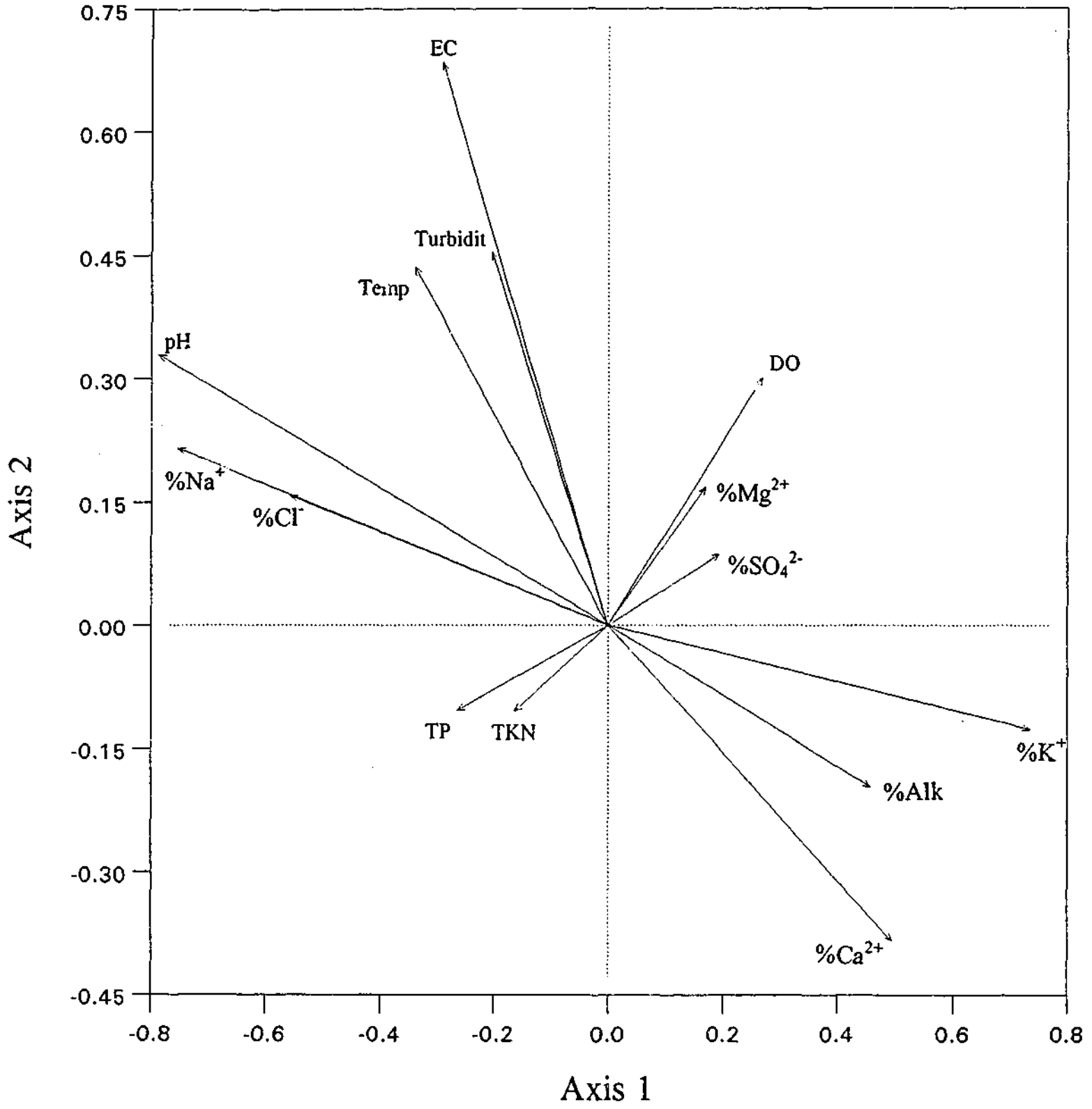


Figure 4.1 *CCA biplot of environmental variables on axis 1 and axis 2 (50 lake data set)*

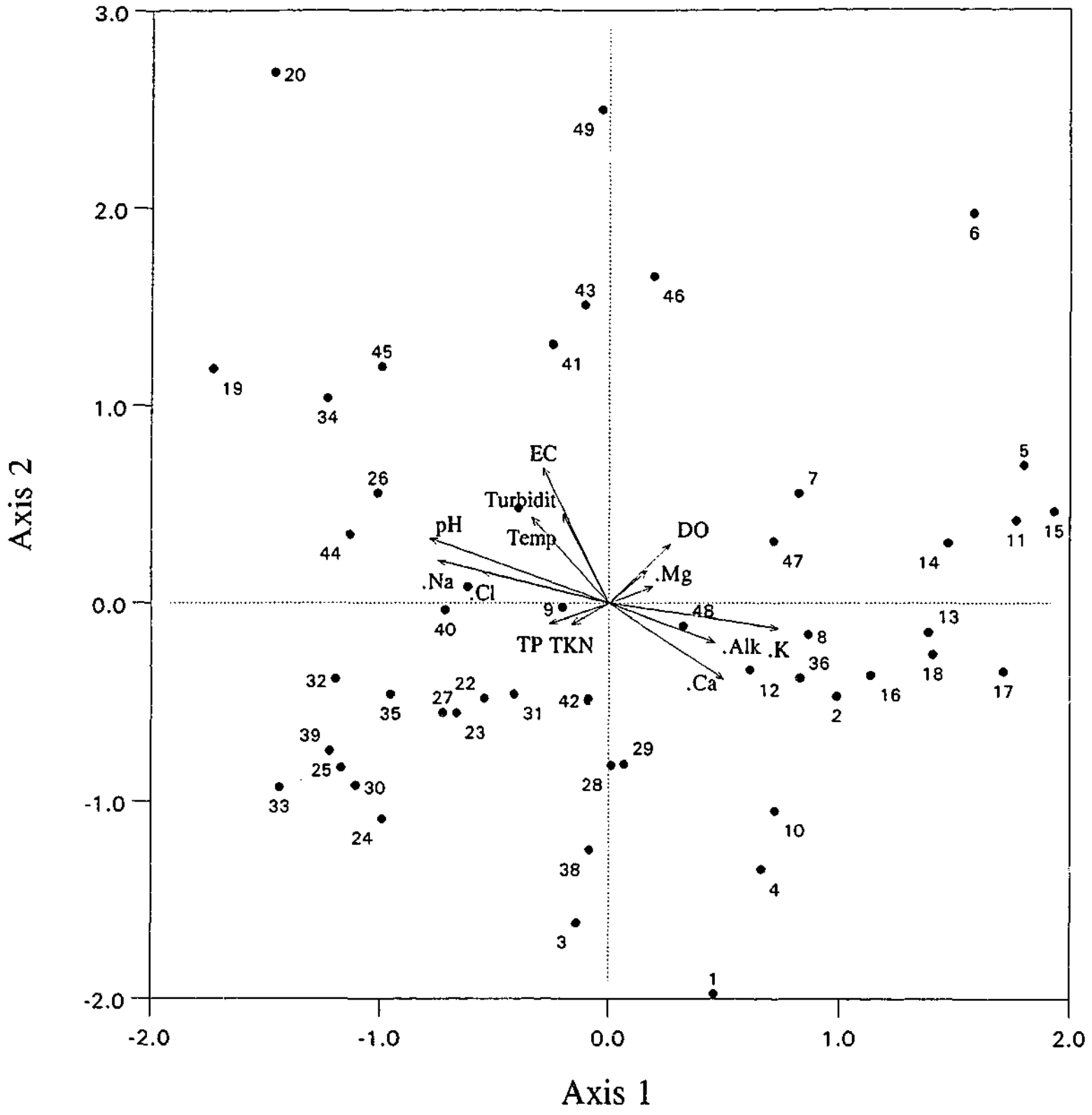


Figure 4.2 CCA biplot of samples scores on axis 1 and axis 2 (50 lake data set)

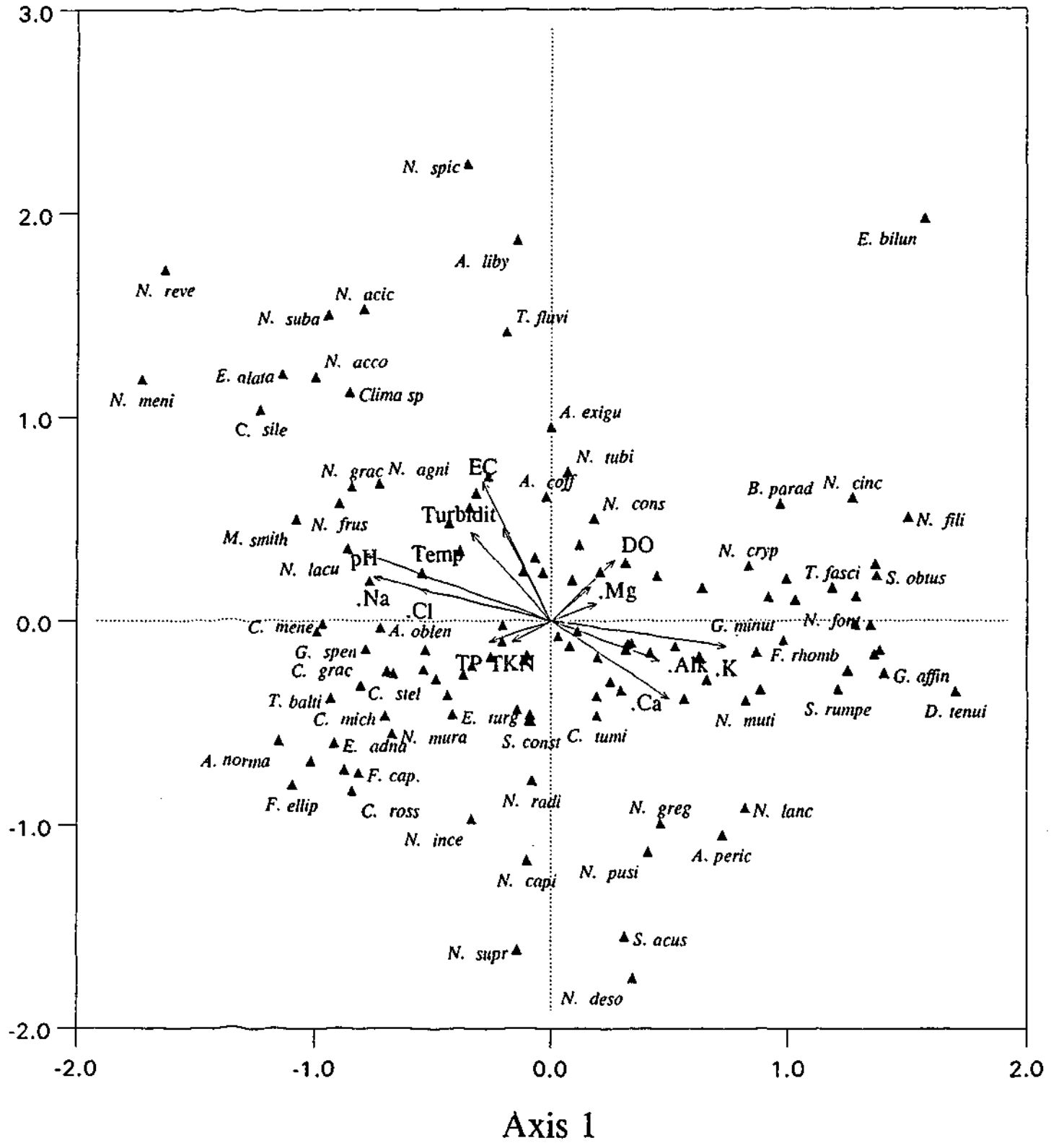


Figure 4.3 CCA biplot of taxon scores on axis 1 and axis 2 (50 lake data set)

normanii. Associated with the high end of the EC vector are several taxa that are known to prefer brackish to marine conditions, including *Navicula spicula, Amphora libyca* and *Thalassiosira fluviatilis* (Fritz *et al.* 1991, Gasse *et al.* 1995, Gell 1995).

4.2.1.2 Forward selection of the lake data set environmental variables

The relative influence and significance of each of environmental variable was determined through the process of forward selection. This process ranks all environmental variables on the basis of the fit for each separate variable. The measure of fit is the first eigenvalue of a CCA which is constrained to a single variable. After the variable that has been ranked first is selected for inclusion in the data set, all other variables are ranked on the basis of fit that each separate variable gives in addition to the variable already selected (ter Braak and Verdonschot, 1995). That is, after each variable has been selected, CANOCO uses this variable as a covariable and perform a partial CCA on the remaining variables separately (Palmer, 1993).

The results of forward selection on the lake data are shown in table 4.2. Here, the difference between percentage variance explained before and after addition to the analysis is due to the effects of covariation between variables. For example, before pH was first selected, the percentage variance attributable to pH also included a portion that was attributable to several of the ions, such as $\%Na^+$, $\%Cl^-$, $\%Mg^{2+}$, and $\%Ca^{2+}$. Therefore the variance explained by these ions decreased when pH was added to the data set as some variance was already accounted for. The *p*-value is the result of Monte Carlo permutation tests for significance, in which 99 unrestricted permutations were used. Variables were considered to be significant at the 0.05 level. The *f*- ratio is the ratio between the sum of all canonical eigenvalues and the residual sum of squares.

***Table 4.2** Results of forward selection of the lake data set environmental variables. Significant variables ($p \leq 0.05$) are emboldened*

Variable	% variance explained before addition	% variance explained after addition	*p* - value	*f* - ratio
pH	3.7	3.7	**0.01**	1.88
$\%Mg^{2+}$	3.3	1.7	0.77	0.88
$\%Ca^{2+}$	3.3	1.1	0.98	0.55
%Alk	3.2	1.7	0.66	0.88
$\%Cl^-$	3.1	1.8	0.59	0.92
$\%Na^+$	2.9	1.7	0.75	0.88
EC	2.7	2.3	0.23	1.15
TKN	2.3	2.1	0.43	1.08
TP	2.3	2.2	0.26	1.11
DO	2.1	2.5	0.10	1.23
$\%K^+$	2.1	1.7	0.69	0.87
Temp	2.1	1.7	0.55	0.91
Turbidity	1.9	1.8	0.55	0.88
$\%SO_4^{2-}$	1.9	1.0	1.00	0.48

The most influential environmental variable identified by forward selection was pH. After addition to the data set, pH explained 3.7% of the taxon variation, DO explained 2.5%, EC explained 2.3%, TP explained 2.2% and TKN 2.1%. Only one of these variables, pH, was statistically significant with a p - value of 0.01. DO is of limited interest to this study as it will usually display great diurnal variation and this was not accounted for in the sampling regime. This variable could not be tested with confidence unless further sampling was undertaken. Although EC was shown to be insignificant, with a p-value of 0.23, it was noted that if EC was selected prior to pH, then it was significant with a p- value of 0.04. The subsequent selection of pH still results in pH having a significant p-value of 0.01. This is most likely due to covariation between pH and EC, and if pH is selected first then the "remaining" variance attributable to EC is reduced and the significance is decreased also. Conducting a partial CCA (variance partitioning) with covariables can shed light on the extent of covariation that exists between variables, but as CCA was being used as an exploratory tool only for this study, there was limited need for extended analyses.

In summary, after conducting a CCA on the lake data set with forward selection it was seen that the variable most correlated with diatom assemblages was pH, followed by EC, TP and TKN. Of these, only pH significantly contributed to taxa variance.

4.2.2 Analysis of the stream data set

A CCA was performed on the river data set with 17 variables, 51 samples, and 186 diatom taxa. The summary results are supplied in table 4.3.

Table 4.3 Summary statistics for CCA of full stream data set of 51 samples and 18 variables

Axes	1	2	3	4	Total inertia
Eigenvalues	0.528	0.495	0.416	0.389	12.763
Taxon - environment correlations	0.926	0.937	0.923	0.904	
Cumulative percentage variance -					
of taxon data	4.1	8.0	11.3	14.3	
of taxon - environment relationship	10.5	20.2	28.5	36.2	
Sum of all unconstrained eigenvalues					12.763
Sum of all canonical eigenvalues					5.055

The amount of taxon variance explained by the first four axis was 14.3%. This is slightly more than the 13.2% explained by the first four axes in the lake data set. The 17 variables in the river data set explained 39.6% of total taxon variance, which is much higher than the 27.3% explained by the 14 variables in the lake data set (calculated as the quotient of the sum of unconstrained eigenvalues and the sum of canonical eigenvalues).

Figure 4.4 illustrates the relative explanatory power and direction of variance of the environmental variables along axes 1 and 2. Axis 1 is associated with pH, shade, and to a lesser extent, turbidity and TKN. Turbidity is probably associated with the nutrient vectors due to its correlation with TP and TKN (by nutrients bound to particulate matter). The other nutrient variable, NO_X, plots separately to the other nutrient variables. Axis 2 is represented by EC, $\%K^+$, $\%SO_4^{2-}$, $\%Na^+$, and $\%Cl^-$. There is little correlation between EC and $\%Na^+$ and

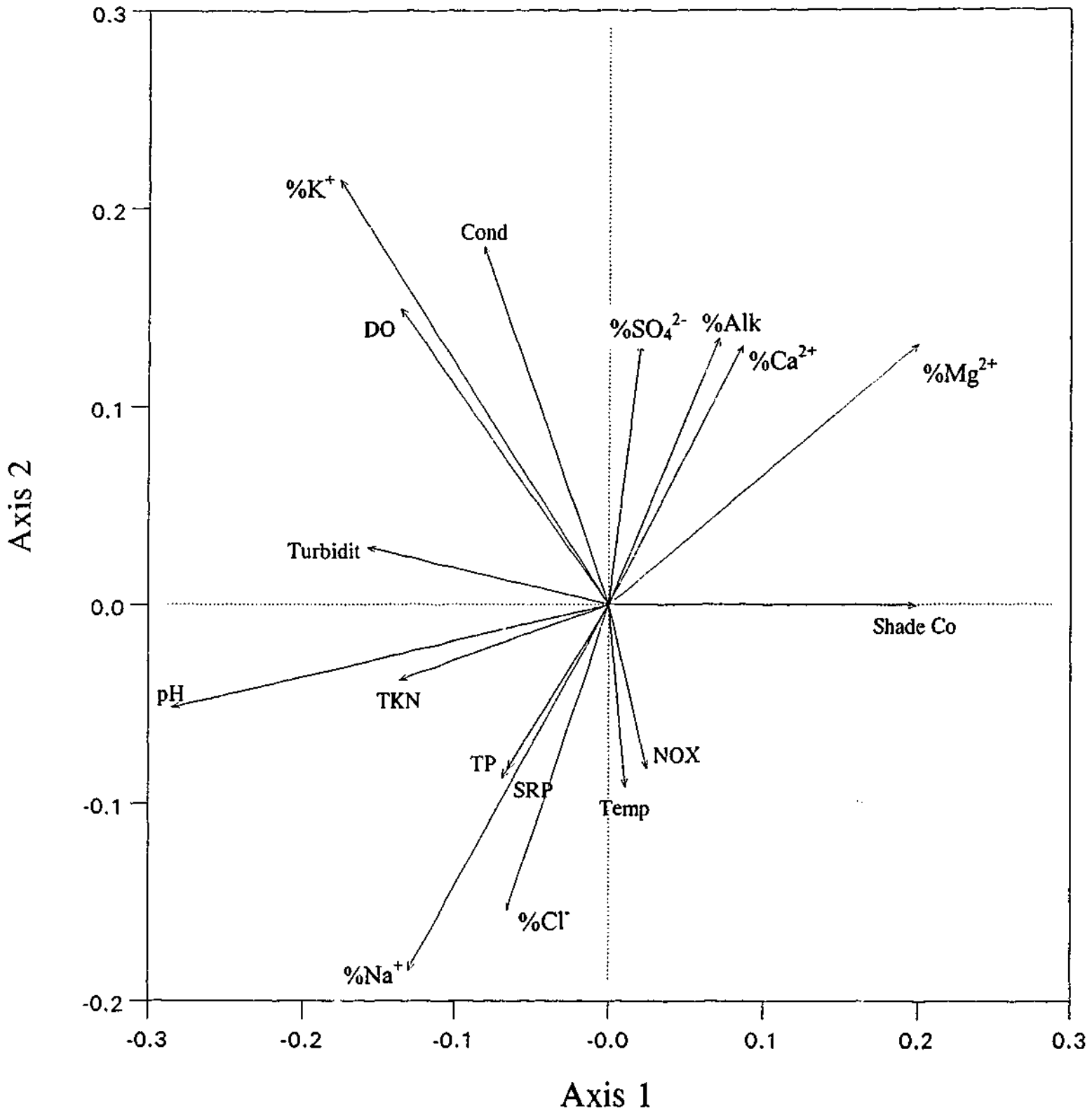


Figure 4.4 CCA biplot of environmental variables on axis 1 and axis 2 (51 stream data set).

%Cl⁻, a result already established by the correlation matrices (table 3.7).

Figure 4.5 shows the distribution of samples in ordination space. It can be seen that samples are more tightly constrained in quadrants two and three (associated with high nutrients and high pH) than they are in quadrants one and four (associated with high EC and low nutrients and pH). The most obvious outliers are those with high nutrient values including samples 89, 91, 95, 97, 99 and 100, and those with low nutrients including samples 51 and 57. The relationship between samples and EC was similar, although not as strong as that of nutrients, with samples with high EC, such as sample 85 with 6460 µS/cm, plotting at the high end of the EC vector but samples with low EC being scattered throughout quadrants two and three.

The relationship between stream samples and environmental variables is similar to that found for the lake data set. That is, nearly all samples appear to be most influenced by pH, with the exception of samples with extreme values of another variable such as EC or TP. These samples then plot at the high end of that particular vector. This may be indicating that diatom communities were more influenced by pH than by EC or TP at the lower end of the EC and TP gradients. However, when EC or TP values exceeded a certain concentration, then these variables became more influential than pH.

The spread of taxa in relation to the environmental variables is illustrated in figure 4.6. It can be seen that the planktonic *Aulacoseira granulata* and *Aulacoseira italica*, both common Murray River taxa, plot at the high end of both the nutrient and pH vectors. Taxa at the low end of these vectors include *Tabellaria flocculosa, Anomoeneis vitrea, Frustulia rhomboides* and *Diatoma tenuis*, all of which have been described as being acidophilous to circumneutral taxa (Stevenson *et al.* 1991, Vyverman 1992, Dixit and Smol 1994, ter Braak and van Dam 1994, Dixit and Smol 1995). Taxa that plot at the high end of the EC gradient include *Navicula spicula* (similar to its position in the lake data set CCA), *Pinnularia elegans, Amphora coffaeformis, Navicula clementis* and *Diploneis parma*. The low end of the EC vector includes taxa such as *Gomphonema acuminatum* and *Amphora libyca*.

4.2.2.1 Forward selection of the stream data set

The relative importance and significance of measured variables was determined using forward selection, the results of which are shown in table 4.4. The most important environmental variables identified by forward selection were pH, turbidity, TP, and $\%Mg^{2+}$. These were then followed by shade, DO, colour, the remaining nutrients and temperature. Only two of these variables, pH and $\%Mg^{2+}$ were statistically significant, with *p*-values of 0.01 and 0.04 respectively. However, as was encountered with the lake data set, problems associated with covariation can affect the amount of taxon variation attributable to each variable and also its significance. For instance, if EC is selected first then it is significant with a *p*-value of 0.05 and explains 2.5% of taxon variance. If pH is subsequently selected then it is still significant (0.01) with a percentage variance explained of 3.1%. Therefore, unlike the lake data set, the variance explained by EC did not covary with pH, as the percentage variance contributed by pH did not alter when EC was selected first. However, once EC was

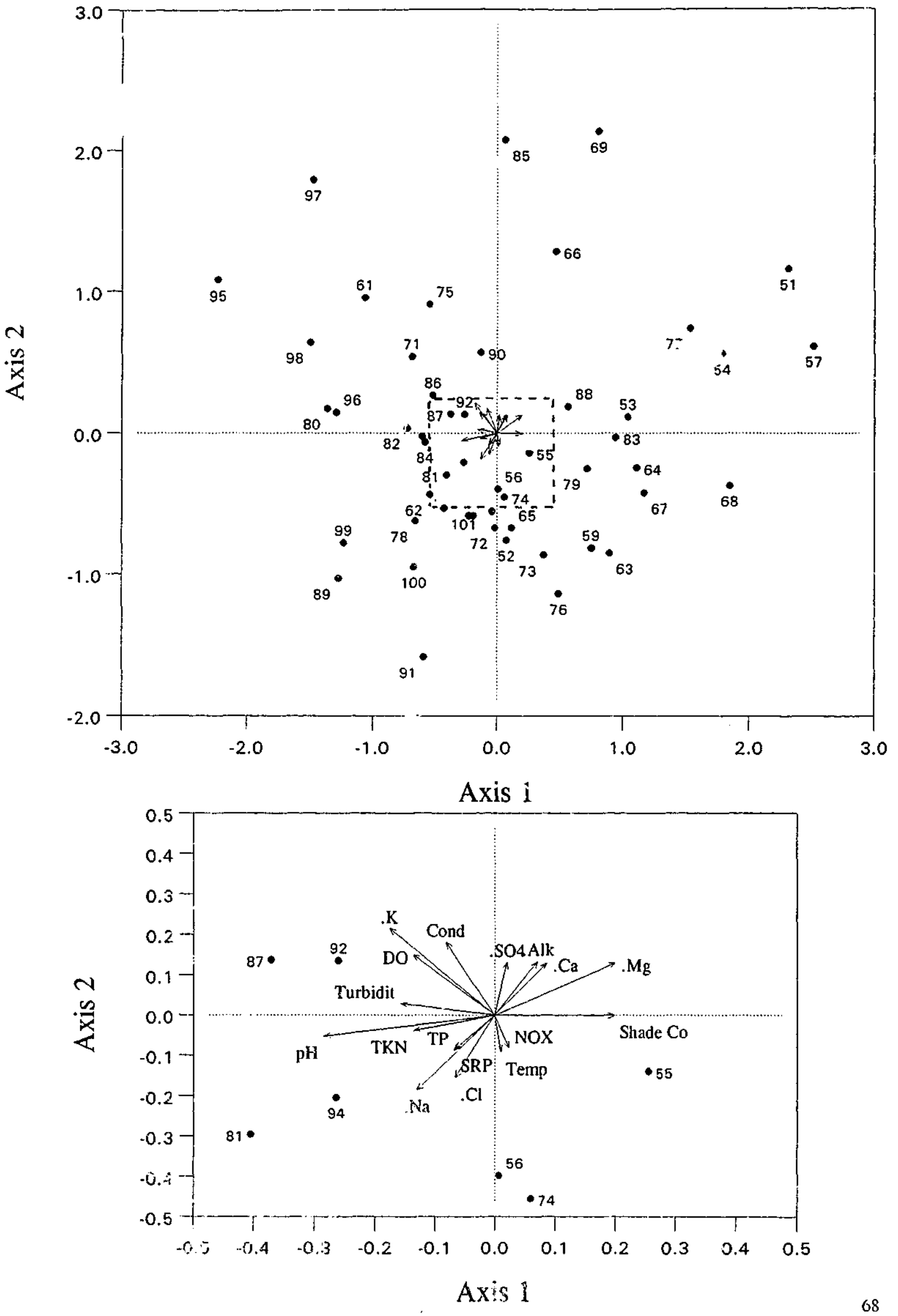


Figure 4.5 CCA biplot of sample scores on axis 1 and axis 2 (51 stream data set). Inset in top biplot is enlarged in bottom biplot.

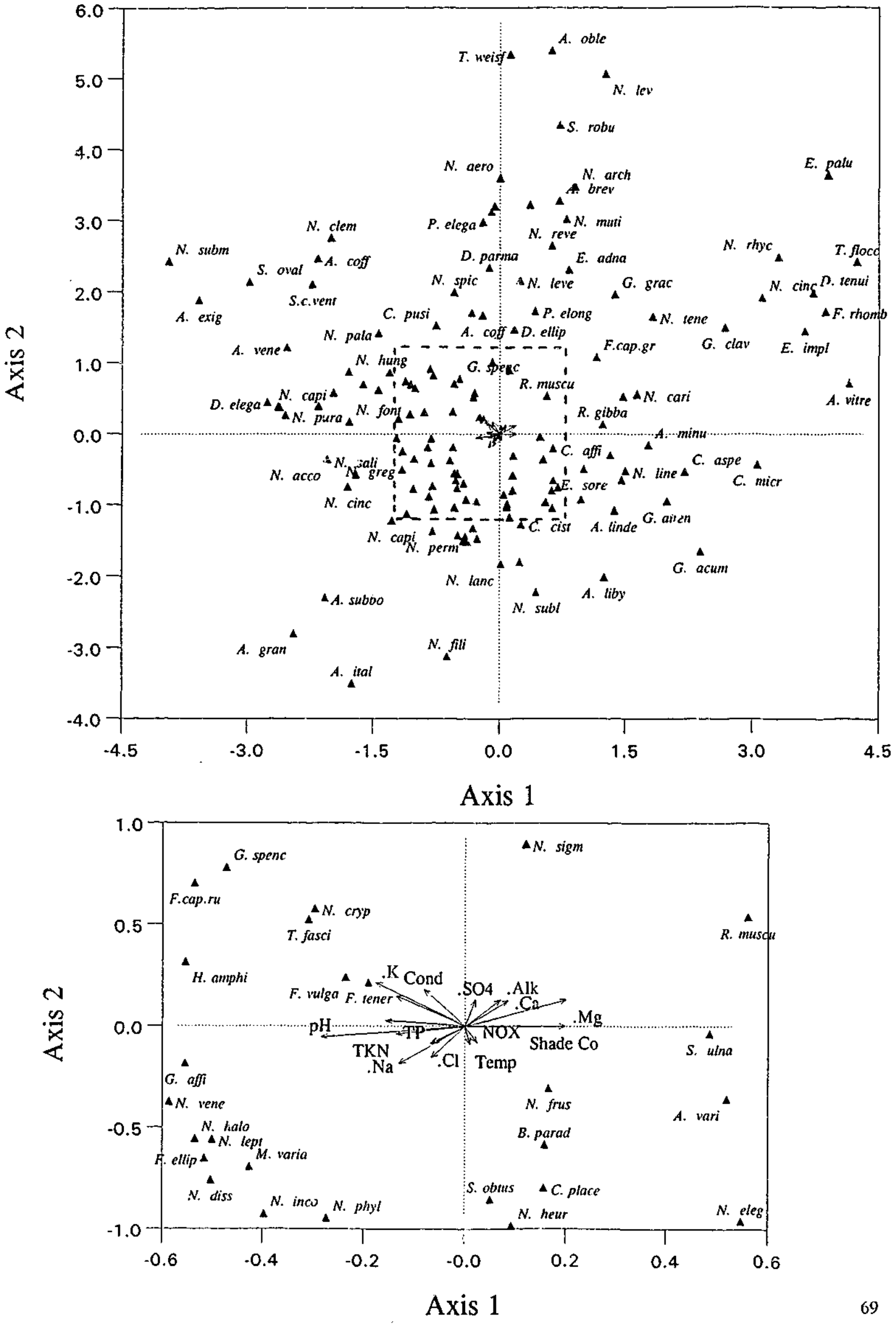


Figure 4.6 CCA biplot of taxon scores on axis 1 and axis 2 (51 stream data set). Inset in top biplot is enlarged in bottom biplot.

selected, the significance of $\%Mg^{2+}$ decreased to 0.18 and its contribution to taxon variance decreased to 2.0%, suggesting that EC is strongly covarying with some of the major ions.

Table 4.4 Results of forward selection of the stream data set environmental variables. Significant variables ≤0.05 are emboldened.

Variable	% variance explained before addition	% variance explained after addition	*p* - value	*f* - ratio
pH	3.1	3.1	**0.01**	1.54
TP	2.8	2.8	0.09	1.46
Turb.	2.8	2.8	0.10	1.44
SRP	2.7	1.4	0.79	0.74
EC	2.5	1.7	0.42	0.91
Shade	2.5	2.5	0.06	1.28
TKN	2.4	1.7	0.5	0.95
Temp	2.4	1.8	0.42	1.00
Colour	2.4	2.4	0.12	1.25
DO	2.4	2.4	0.1	1.23
NO_X	2.1	2.1	0.25	1.14
$\%Na^+$	2.1	1.8	0.43	1.02
$\%SO_4^{2+}$	2.1	2.4	0.12	1.24
$\%K^+$	2.0	2.0	0.32	1.10
%Alk	2.0	2.1	0.19	1.12
$\%Cl^-$	2.0	1.2	0.72	0.83
$\%Ca^{2+}$	2.0	2.3	0.18	1.21
$\%Mg^{2+}$	1.8	2.7	**0.04**	1.42

A similar relationship exists between TP and SRP. Once TP is selected, then the percentage variance contributed by SRP decreases from 2.7 to 1.4. This relationship is an obvious one as TP is composed of two fractions - SRP and organic phosphorus (phosphorus bound to particulate matter). If TP is selected first, then the fraction that is common to both, SRP, is largely accounted for.

4.2.3 Analysis of the combined data set

The summary statistics for a CCA of the combined data set of 15 variables, 101 samples and 159 diatom taxa is shown in table 4.5.

Table 4.5 Summary statistics for CCA of the combined data set

Axes	1	2	3	4	Total inertia
Eigenvalues	0.466	0.321	0.277	0.242	16.546
Taxon - environment correlations	0.885	0.861	0.800	0.768	
Cumulative percentage variance -					
of taxon data	2.8	4.8	6.4	7.9	
of taxon - environment relationship	16.4	27.7	37.4	45.9	
Sum of all unconstrained eigenvalues					16.546
Sum of all canonical eigenvalues					2.845

The amount of taxon variance explained by the first four axes is 7.9%, which compares to 13.2% in the lake data set and 14.3% in the river data set. The large disparity between the sum of canonical eigenvalues and the sum of all unconstrained eigenvalues means that the measured environmental variables explained only 17.2% of the total taxon variance, which is much lower than the 27.3% explained in the lake data set and the 39.6% explained in the river data set.

Figure 4.7 illustrates the distribution of and relationship between environmental vectors along CCA axes one and two. The most influential variables are pH and $\%K^+$. Turbidity also appears to be relatively important and is correlated with both pH and Na^+, and to a lesser extent, TKN and TP. In keeping with the ordination results of the two individual data sets, pH remains one of the most important variables in determining diatom assemblages, while EC becomes less important when the data sets are combined.

Figure 4.8 shows the location of the 101 samples in ordination space. Samples plotting in quadrant one are generally samples with very high (> 200 µg/L) TP or TKN. These include sample 23 (1000 µg/L TP and 50 000 µg/L TKN), sample 24 (350 µg/L TP and 12 280 µg/L TKN), and sample 33 (200 µg/L TP and 4000 µg/L TKN), all of which are located in the Kerang region of Victoria. Samples located within this quadrant at the high end of axis one also tend to be comparatively turbid. Samples plotting in quadrant two, at the high end of the pH gradient include 19 (pH of 8.9), 20 (pH of 8.8) and 34 (pH of 8.8). Although the most important vector in quadrant three is $\%Ca^{2+}$, the samples located in this space do not appear to represent sites with high$\%Ca^{2+}$. Rather, samples plotting in quadrant three are generally low nutrient sites, or in the case of samples 86, 88 and 98, are samples with high EC. Samples plotting in quadrant four at the high end of axis two are representative of sites located at the low end of the pH gradient for this data set. These include sample 11 (pH of 6.9), sample 2 (pH of 7.2) and sample 4 (pH of 7.4). Samples located at the low end of both axes one and two tend to be sites with relatively high $\%K^+$.

Unlike the individual data sets, there does not appear to be any data point separation based on spatial location (i.e.: Adelaide Hills versus Kerang). However, there does appear to be some separation based on whether the sample was collected from a lake or stream, with the majority of lake samples plotting in quadrants one and four and the majority of stream samples plotting in quadrants two and three. This distinction is probably due to the stream samples being generally more saline than the lake samples, and thus, they tend to plot at the high end of the EC vector.

Figure 4.9 illustrates the spread of diatom taxa across the ordination space. As with the individual data sets, planktonic taxa tend to plot at the high end of the nutrient vectors. These include *Cyclotella rossii, Cyclotella meneghiniana, Cyclotella stelligera, Cyclotella michigiana* and *Actinocyclus normanii*. Apart from this feature, the diatom taxa appear to be well spread throughout the diagram, with rarer taxa occurring at the high end of both vectors and axes (such as *Navicula muralis, Navicula cryptotenella, Tabellaria flocculosa,* and *Navicula spicula*), while more common taxa plot towards the origin of the diagram (including *Cocconeis*

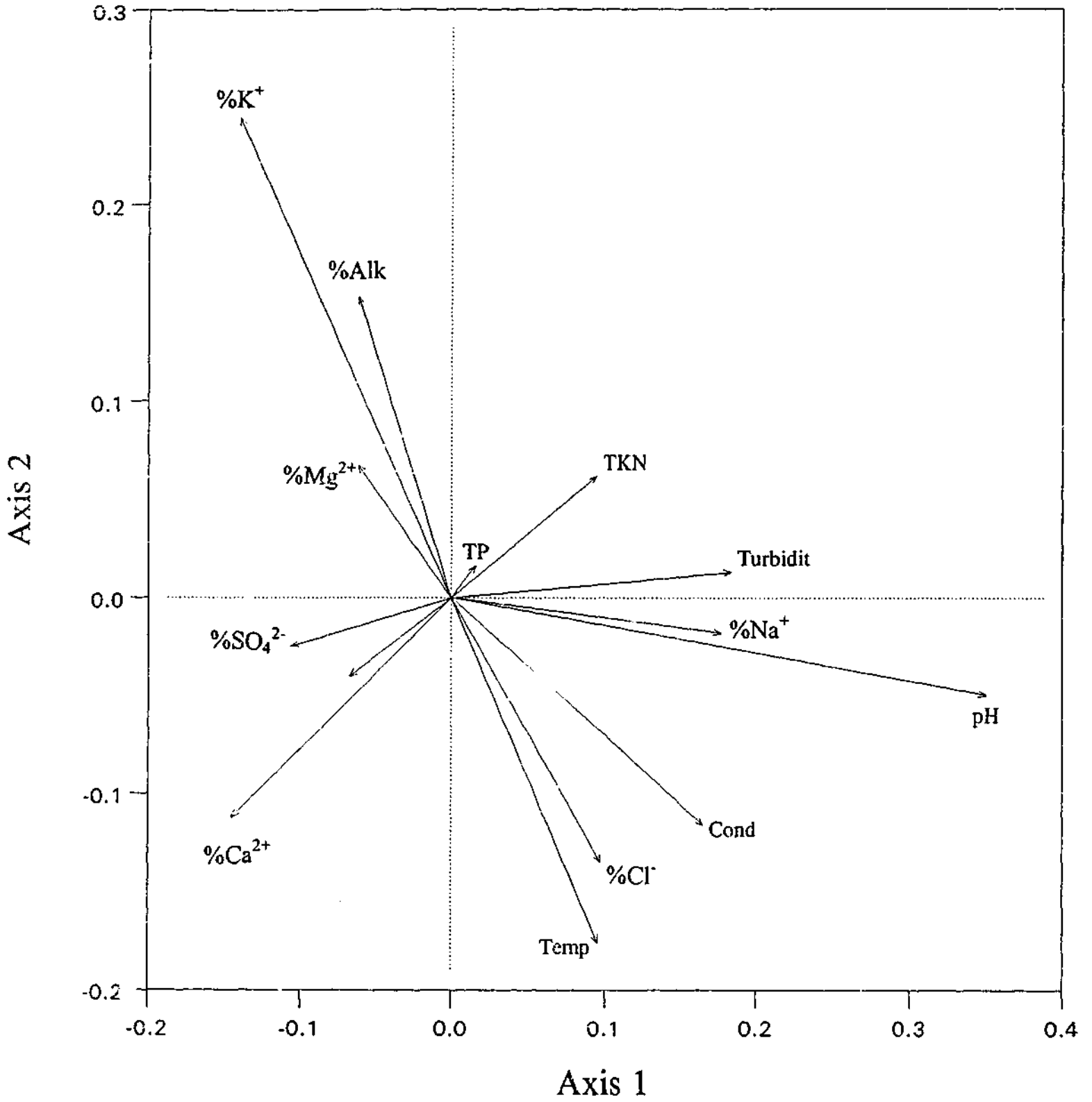


Figure 4.7 CCA biplot of environmental variables on axis 1 and axis 2 (101 sample combined data set)

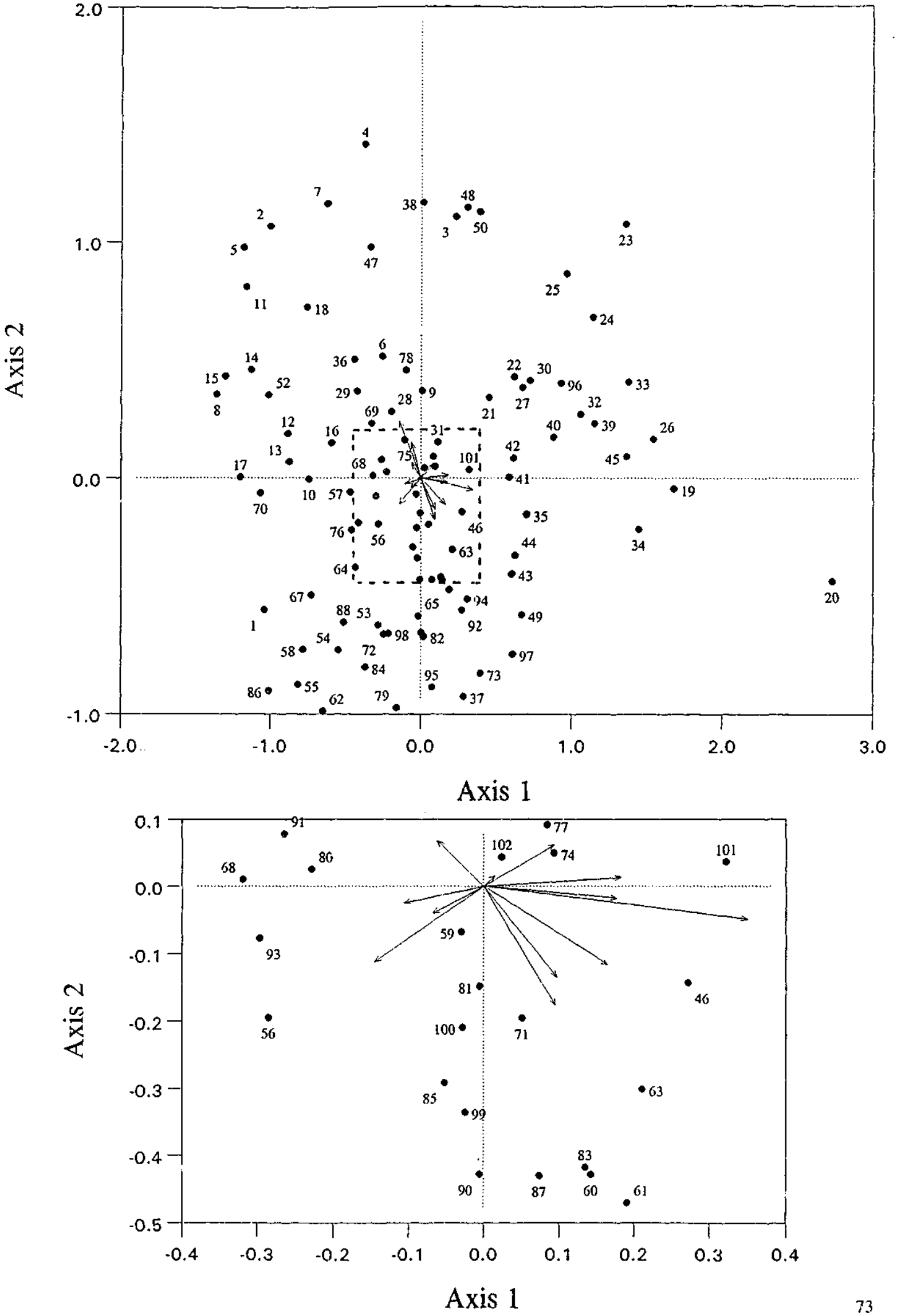


Figure 4.8 CCA biplot of sample scores on axis 1 and axis 2 (101 sample combined data set). Inset in top biplot is enlarged in bottom biplot.

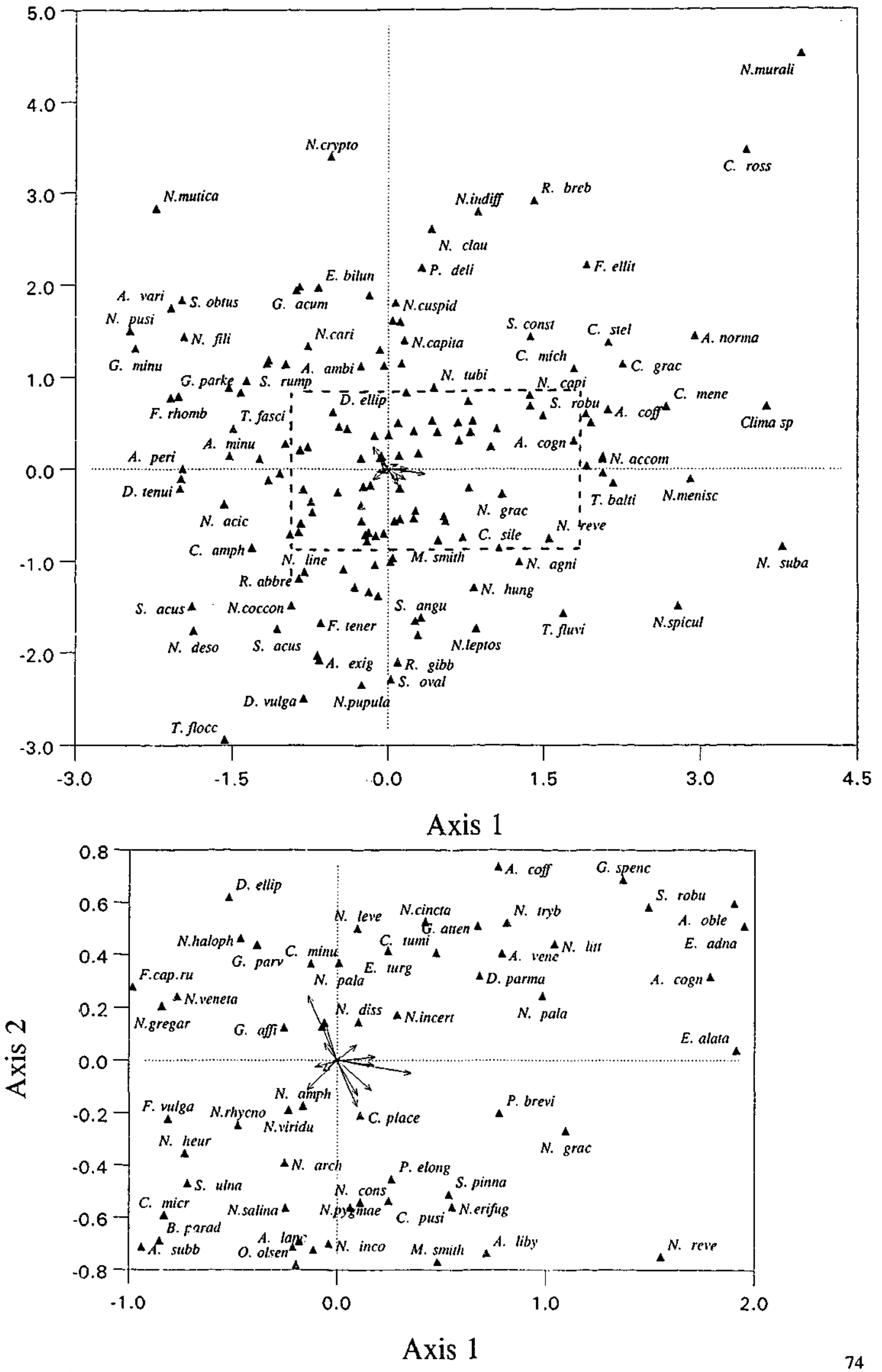


Figure 4.9 CCA biplot of taxon scores on axis and axis 2 (101 sample combined data set). Inset in top biplot is enlarged in bottom biplot.

placentula, Nitzschia palea, Pseudostaurosira brevistriata, Diploneis parma, Navicula veneta, Gomphonema parvulum, Synedra ulna, and *Staurosirella pinnata*).

4.2.3.1 Forward selection of the combined data set

The relative importance of each variable was determined quantitatively by applying forward selection to the data, the results of which are shown in table 4.6. The most influential variables in the data set were pH, turbidity, EC, TKN, and DO. Three variables (pH, EC and DO) were significant at the 95% level. TP accounts for a very small percentage of taxon variance and has a very high *p*-value, thus rendering it insignificant. The large difference between the percentage variance explained both before and after addition for $\%Ca^{2+}$ is due to its covariance with pH. Once pH was selected, the variance attributable to $\%Ca^{2+}$ was greatly decreased.

***Table 4.6** Results of forward selection of the combined data set environmental variables. Significant variables ≤0.05 are emboldened*

Variable	% variance explained before addition	% variance explained after addition	*p* - value	*f* - ratio
pH	2.0	1.9	**0.01**	1.95
TKN	1.5	1.4	0.17	1.41
EC	1.5	1.3	**0.05**	1.34
Turb.	1.5	1.3	0.11	1.34
$\%Ca^{2+}$	1.5	0.7	0.21	0.96
%Alk	1.3	1.0	0.29	1.04
$\%Na^{+}$	1.3	1.1	0.21	1.1
$\%Cl^{-}$	1.3	0.6	1.0	0.65
$\%SO_4^{2-}$	1.3	1.0	0.55	0.98
$\%Mg^{2+}$	1.3	0.8	0.68	0.89
DO	1.1	1.3	**0.02**	1.35
$\%K^{+}$	1.1	0.7	0.94	0.8
Temp	1.0	1.1	0.32	1.1
TP	0.8	1.0	0.49	0.9

4.2.4 Summary of the CCA analysis

The results of the CCA ordinations for all three data sets identified several key variables that were influential in determining diatom taxon variance. Within the lake data set, pH was the most important variable, followed by DO, EC, TP and TKN. Of these, only pH had a significant *p*-value (0.01) when the forward selection process was strictly followed. The most important variable within the stream data set was also pH, with the next most important variables being turbidity, TP and $\%Mg^{2+}$. Only pH and $\%Mg^{2+}$ were significant with *p*-values of 0.01 and 0.04 respectively. The CCA of the combined data set showed pH to be the most important variable in determining taxon variance, followed by turbidity, EC, TKN and DO. Three of these variables, pH, EC and DO were significant at the 95% confidence level.

In summary, pH was the most influential variable in all data sets. This is interesting as the pH range for the lake and stream samples was relatively narrow in comparison to the range of TP and TKN values, and

suggests that in these lake and stream sites, diatoms were responding more to small changes in pH than they were to large changes in nutrient concentrations. The importance of turbidity is also interesting, although perhaps not surprising given the high sediment loading of many lakes and rivers in the data set. Total phosphorus was shown to have a major influence on taxon variance in both the lake and steam data sets (fourth and second most important variables respectively), but was the least important variable in the combined data set. This may be due to differences in nutrient optima for individual taxa in lacustrine and riverine environments which acts to confuse the response when the two data sets are combined. This aspect will be explored more fully in the following section on Weighted Averaging where optima for these variables are derived.

Birks *et al.* (1990) state that environmental variables can be reconstructed as long as there is a statistically significant relationship between modern assemblages and the environmental variables and that there are good biological reasons for believing that the variables being reconstructed either are ecologically important determinants in the system of interest or are linearly related to such determinants. Based on these requirements and taking into consideration the palaeolimnogical aims of this study, three variables were chosen for further analysis: pH, EC, and TP. Although some of these variables were not shown to be significant in influencing diatom communities in all three data sets, they were included because information on these variables was thought to be important for the palaeolimnological analyses, and as explained previously, there was a reticence to omit key variables based purely on exploratory analysis. This is particularly important where covariance with other measured variables may have contributed to insignificant p – values.

4.3 The development of taxon optima and tolerance for environmental variables

4.3.1 Introduction to Weighted Averaging regression

As outlined in Chapter 2, Weighted Averaging (WA) regression is based on diatom taxon responses that are modelled as a Gaussian response curve. The method relies on the assumption that taxa are most abundant in waters close to their optimum for a particular variable. A taxon optimum for an environmental variable can be determined by averaging the variable values of the sites where the taxon occurs, weighted by its relative abundance at each site. Absent taxa carry zero weight (ter Braak, 1987). The mathematical formulae for these calculations are shown below -

$$u_k^* = \sum_{i=1}^{n} y_{ik} x_i \,/\, \sum_{i=1}^{n} y_{ik}$$

where (u_k^*) is the optimum of taxon *k*, x_i is the observed value for the variable *x* at site *I* and y_{ik} is the abundance of taxon *k* at site *I*.

The diatom inferred environmental value (x_i^*) can then be calculated by Weighted Averaging of the optima of the taxa which occur there. For simple WA the equation is:

$$x_i^* = \sum_{k=1}^{n} y_{ik} u_k^* / \sum_{k=1}^{n} y_{ik}$$

A taxon tolerance for a particular variable is determined by the weighted standard deviation of the distribution (Birks *et al*, 1990). The estimated tolerance range of a given taxon (t_k^*) is calculated as:

$$t_k^* = \sqrt{\sum_{i=1}^{n} y_{ik} (x_i - u_k^*)^2 / \sum_{i=1}^{n} y_{ik}}$$

Taxa can then be weighted according to their tolerance range with taxa with a narrow range given more weight (Juggins, 1993). The calculations for this are as follows:

$$x_i^* = (\sum_{k=1}^{n} y_{ik} u_k^* / t_k^{*2}) / (\sum_{k=1}^{n} y_{ik} / t_k^{*2})$$

Weighted Averaging regression and calibration were carried out using the computer program CALIBRATE version 0.85 (Juggins and ter Braak, 1997).

The predictive ability of transfer functions can be examined by comparison of actual values recorded at each site against those values predicted by Weighted Averaging of taxon optima. The performance of the calibration data set can be calculated as a measure of the predictive power, most commonly expressed as the root mean square error of prediction (RMSEP), which represents a reliable measure of the difference between measured and predicted values, and r^2, which is the correlation of determination between measured and predicted values. These measures of strength between measured and predicted values are generated using internal data set testing procedures, whereby the data set is used to predict the measured value of each sample, in turn, within that data set. In order to overcome circularity, a form of cross validation, such as bootstrapping or jackknifing (sometimes called leave-one-out cross validation), which involves the calculation of environmental estimates for each site based on a data set which excludes that site, can be applied. Jackknifing was the method adopted in this study.

Estimates of model bias, based on comparison of model derived and measured variables and derived in CALIBRATE (Juggins and ter Braak, 1997), are also provided. Bias measures the mean differences between

observed and model derived values for reconstructed parameters. In addition to estimates of the bias across the data set (average bias), in this study, the approximate location of maximum bias is identified as which of ten equally divided segments of the data set has the greatest mean bias.

Data deshrinking is necessary as during WA environmental reconstructions, averages are taken twice, thus shrinking the range of inferred values towards the mean of the environmental variable. Data can be deshrunk by either of two methods, classical or inverse. Gasse *et al.* (1995) state that if greatest accuracy is required at the extremes of the environmental gradients, then classical deshrinking is appropriate, but if the emphasis is on a wide range of the environmental variable, then inverse regression should be used. Given the complex and dynamic nature of the lower Murray River, and therefore the need for information over a wide ecological range, it was decided to adopt inverse regression techniques.

Weighted Averaging Partial least squares regression (WA-PLS) was also carried out on each analysis to see whether it improved the predictive capabilities of the data. WA-PLS is designed to take account of residual correlations among the biological data that remain after fitting the environmental variable of interest (Birks, 1995). WA-PLS involves a weighted inverse deshrinking regression which tends to pull the inferred values towards the mean, thus somewhat overcoming the edge effect of simple WA which results in both over and under estimation of optima at the low and high ends of the gradient respectively (Birks, 1998). Consequently, WA-PLS often produces training sets with low RMSEP and low average and maximum bias, as assessed by cross - validation. However, for all parameters analysed in this study, WA-PLS was outperformed by the simple two way WA.

In terms of efficacy for the generation of accurate environmental optima, Weighted Averaging models with r^2 values of < 0.6 and / or RMESP (either minus or plus the predicted value in absolute terms) of > 0.5 pH units and $> 30\%$ for both EC (at EC of 1000 µS/cm) and TP (at 50 µg/L) are rejected. It is acknowledged that these criteria are somewhat arbitrary, but there is a lack of established guidelines for defining model performance limits in the literature. Therefore, these criteria are specific for this study only and are relevant to the focus fossil sites.

4.3.2 Selection of outliers

In order to improve the predictive power of the data set it is considered that outliers, or samples that are poorly related to the variable in question, should be removed from the data set (ter Braak and Juggins, 1993). Gasse *et al.* (1995) recommend the removal of samples whose jackknifed residuals between actual and predicted values exceed 25% of the range of that variable. Although this may be appropriate for those parameters with a small range (such as shade cover or DO) it is not an appropriate method to use for parameters such as TP or EC where the range is high. For instance, 25% of the range of TP in the lake data

set is 250 µg/L and this is not an acceptable error margin for samples with low TP concentrations. Jones and Juggins (1995) suggest an alternative method whereby samples with predicted residual values in excess of the standard deviation of the observed variable are removed. This method is likely to be problematic when applied to variables that have a high range with a concentration of samples at the lower end of the range, such as nutrients in the stream data set, as the standard deviation will be inflated due to the few high values (although, to a certain extent, this is countered by logging of variables).

Regardless of the method used to select outliers, the process of outlier removal results in a loss of ecological information. There should be stronger justification for outlier removal than simple reasoning that the sample had high residual values or that there is an increase in model performance if the sample is omitted (Anderson *et al.*, 1993). Therefore, for this study, outliers were removed on the basis of poor ecological fit, rather than numerical fit. This included samples which were dominated by taxa which did not occur in other samples in the data set, or samples which were obviously reflecting a different water chemistry to that which was measured at the time of diatom collection (eg: flooding may result in lower EC conditions than the existing diatom assemblage suggests).

4.3.3 Development of a pH transfer function

4.3.3.1. Lake data set

Weighted Averaging for pH was initially applied to the full 50 lake data set. The summary statistics for this initial analysis are included in table 4.7.

Table 4.7: Results of pH Weighted Averaging (50 lakes)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.1433	0.3785	0.7449	0	0.243 (5)
Tol. d/w WA	0.1255	0.3543	0.7692	0	0.338 (2)
Jackknifed simple WA	0.302	0.5496	0.3976	-.002453	1.172 (2)
Jackknifed tol. d/w WA	0.2885	0.5371	**0.4088**	-.01059	1.153 (2)

The best performing jackknifed model for the full lake data set was the tolerance downweighted model, with an r^2 of 0.41 and a root mean squared error (RMSEP) of 0.54. Reed (1995) states that poor performance of jackknifed models may be due to a heterogeneous data set, that is, a data set that contains many rare taxa. This will result in water quality being poorly predicted when such samples are left out of the data set in the jackknifing process. This probably only applies in part to this data set as, although there were a few samples dominated by rare taxa, the majority of samples contained common taxa.

Figures 4.10 and 4.11, show a plot of measured versus predicted pH and a plot of predicted pH versus residuals between actual and predicted values. Five samples were identified as outliers that decreased the performance of the model. Details of these samples are provided in Table 4.8.

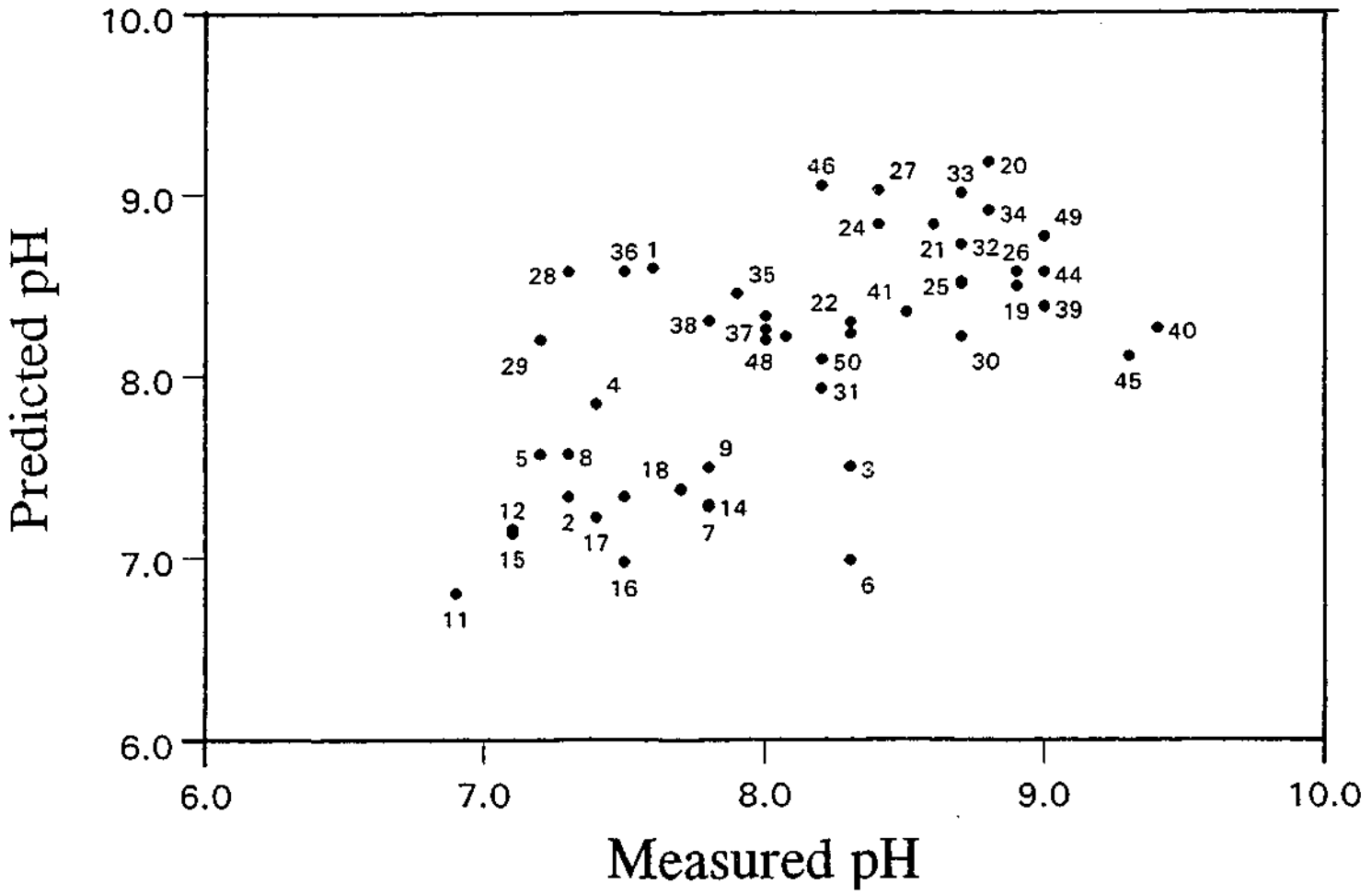


Figure 4.10 Plot of measured pH versus predicted pH (lake data set, 50 samples)

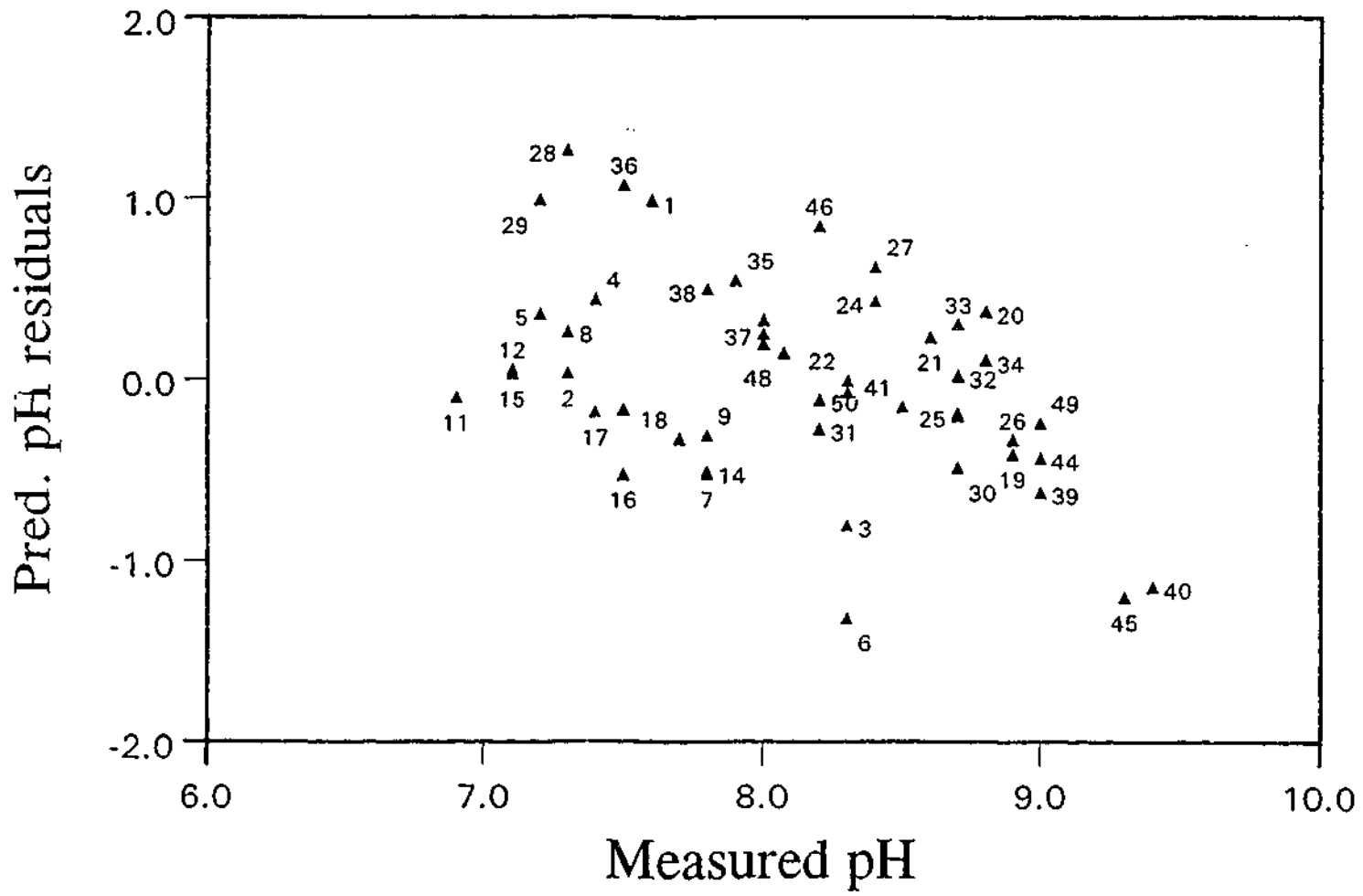


Figure 4.11 Plot of predicted pH versus residuals between actual and predicted values (lake data set, 50 samples)

Table 4.8 Samples with high residual values - pH WA (lake data set)

Sample	Actual pH	Predicted pH	Dominant diatom taxa
6 (Barker Inlet wetland)	8.3	7.2	*Navicula cincta, Navicula cryptocephala, Nitzschia filiformis*
28 (Little Lake Charm)	7.3	8.47	*Aulacoseira granulata, Pseudostaurosira brevistriata, Staurosirella pinnata*
29 (Reedy Lake)	7.2	8.17	*A. granulata*
45 (Cardross Basin)	9.3	8.1	*Climaconeis sp.*
46 (Karadoc Basin)	8.2	8.8	*Navicula spicula*

Two of these samples, 45 and 46, are undoubtedly unique as their diatom assemblages were completely dominated by a single taxon that was rare or absent in the rest of the data set. In fact, *Climaconeis* sp. was only found in sample 45 and so once this sample was removed from the data set there was no further ecological information available for this taxon. This taxon also did not occur in the fossil records so there was no ecological justification for retaining the sample in the data set. Samples 28 and 29 had overestimated pH, and were dominated by the planktonic *Aulacoseira granulata*, which is a common riverine taxon. It has been suggested that the presence of this taxon in a floodplain lacustrine assemblage can be used as an indicator of flooding (Reid *et al.*, 2001), and it is therefore possible that these sites may have experienced recent flooding. Sample 6, a coastal wetland to the north of Adelaide, receives stormwater flows and is subject to periodic drying, factors which increase the risk of collecting an unrepresentative diatom assemblage.

These five samples were omitted from subsequent WA analysis, with the results shown in table 4.9.

Table 4.9 Results of pH Weighted Averaging (45 lakes)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.05876	0.2424	0.8375	0	0.2461 (7)
Tol. d/w WA	0.04172	0.2042	0.8846	0	0.2378 (7)
Jackknifed simple WA	0.1318	0.3631	**0.6358**	0.013	0.4117 (7)
Jackknifed tol. d/w WA	0.1415	0.3801	0.6030	0.03056	0.4376 (3)

The removal of five lakes greatly improves the performance of the data set, with a jackknifed r^2 of 0.64 and RMSEP of 0.36 pH units. The plots of measured pH against predicted pH and predicted pH ± actual residuals are shown in Figures 4.12 and 4.13. The line of best fit between measured and predicted pH is now much stronger with a lesser trend in residual values. Further analysis using WA-PLS did not improve the performance of the data.

This model fulfills the criteria previously established for the generation of reliable optima. These optima are included in appendix 5.

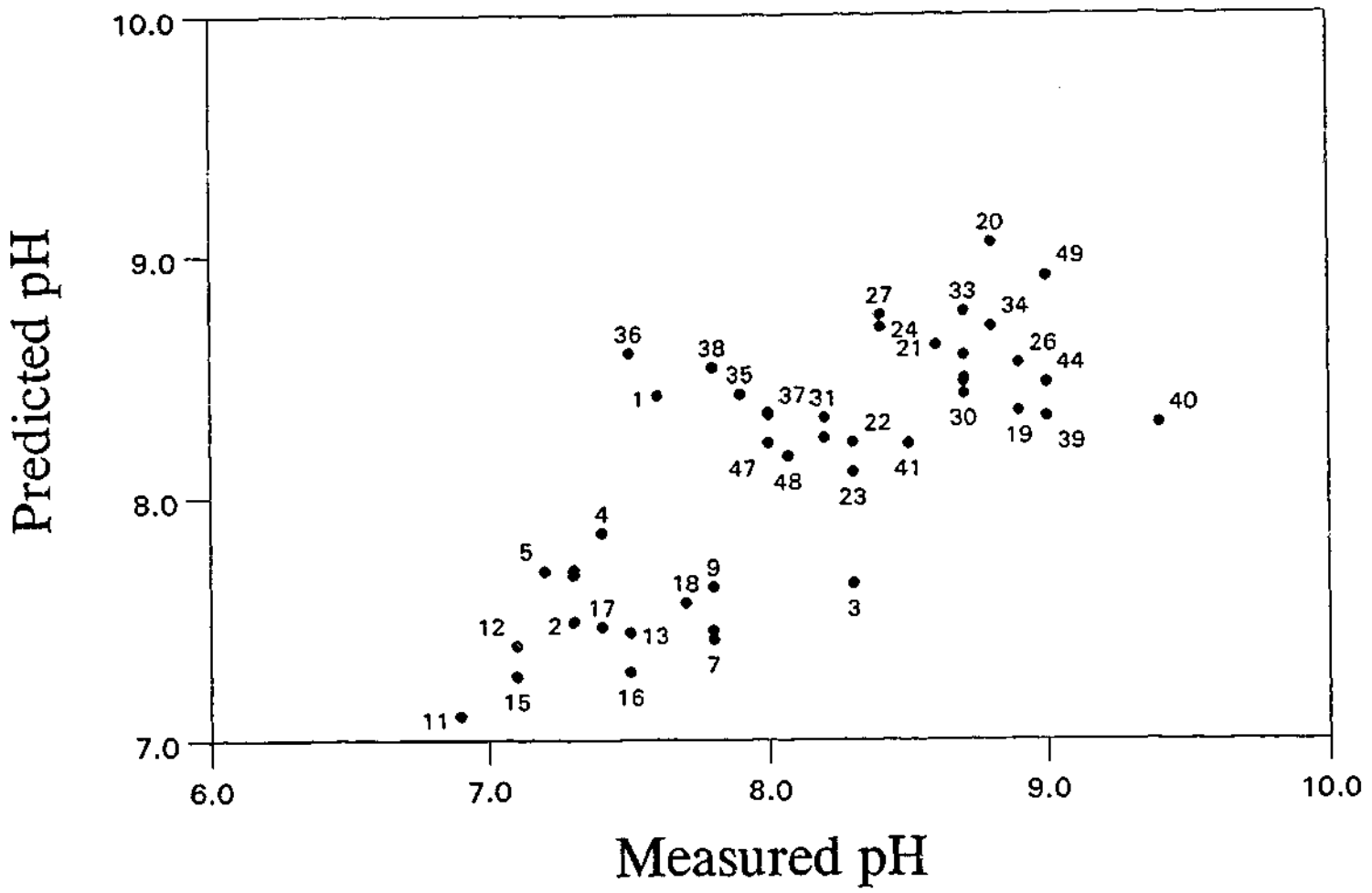


Figure 4.12 Measured pH versus predicted pH (lake data set, 45 samples)

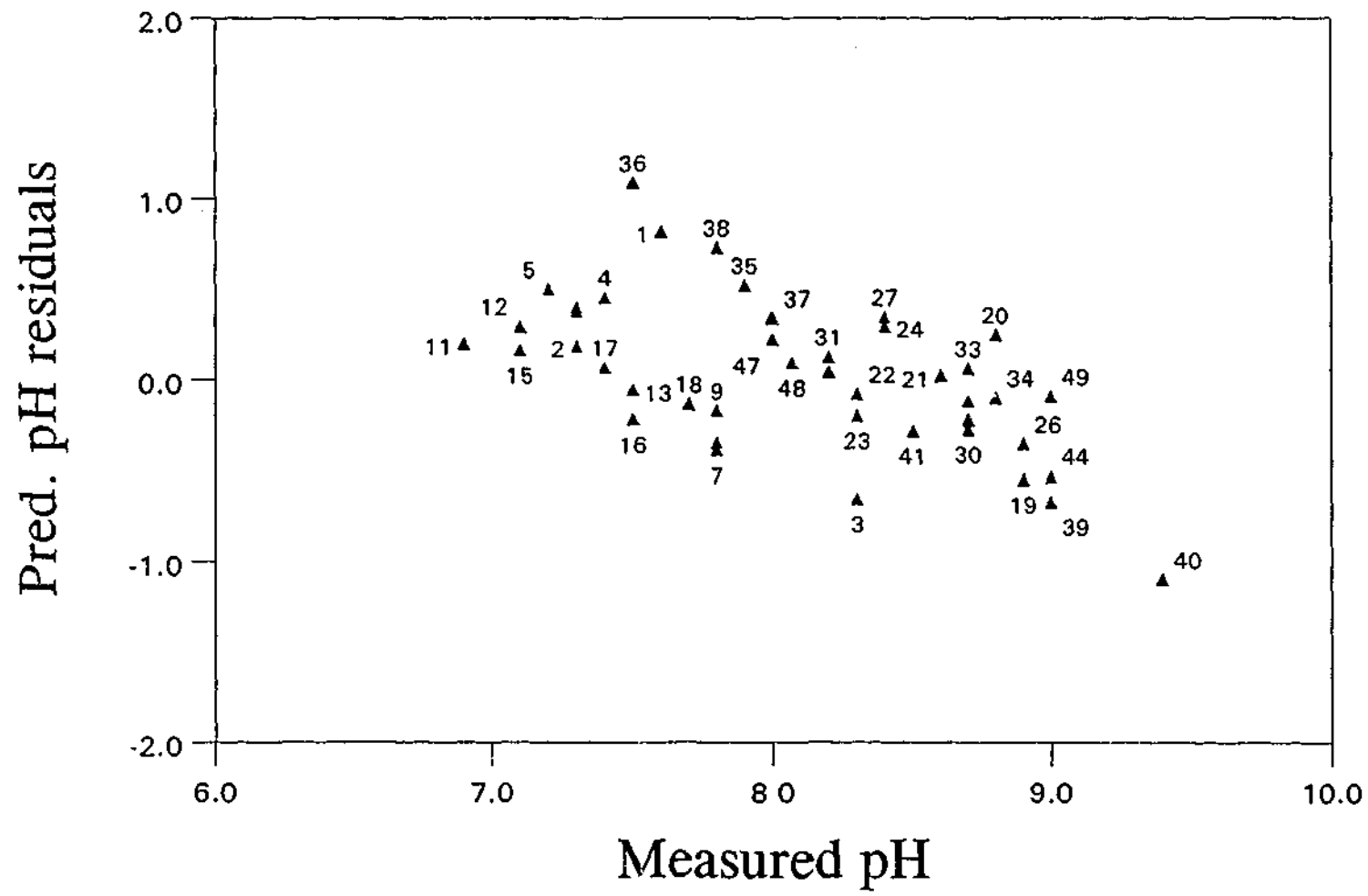


Figure 4.13 Plot of predicted pH versus residuals between actual and predicted values (lake data set, 45 samples)

4.3.3.2. Stream data set

Initial WA analysis for pH was applied to all 51 streams, the results of which are shown in Table 4.10.

Table 4.10 Results of pH Weighted Averaging (51 streams)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.0358	0.1892	0.7025	0	0.2363 (1)
Tol. d/w WA	0.03617	0.1902	0.6994	0	0.3015 (1)
Jackknifed simple WA	0.8922	0.2987	**0.2959**	0.01887	0.5762 (1)
Jackknifed tol. d/w WA	0.1047	0.3235	0.2017	0.05278	0.5862 (1)

The best performing jackknifed model for pH for the full stream data set was the simple WA method, with an r^2 of 0.30 and an RMSEP of 0.30. This r^2 is substantially lower than that for the full lake set for pH, where an r^2 of 0.41 was achieved. This difference concurs with the CCA results which showed that pH accounted for 3.8% of taxon variance in the lake data set and 3.0% of taxon variance in the stream data set. Plots of measured versus predicted pH and predicted pH versus residuals between actual and predicted values (figures 4.14 and 4.15) identified nine samples (78, 79, 83, 85, 92, 94, 95, 99, 101) that were decreasing the performance of the model. After omitting these samples from further analysis, r^2 values were greatly increased, as shown in table 4.11.

Table 4.11 Results of pH Weighted Averaging (stream data set, 42 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.02192	0.1480	0.8138	0	0.3524 (1)
Tol. d/w WA	0.02225	0.1492	0.8116	0	0.2610 (2)
Jackknifed simple WA	0.04824	0.2196	**0.5226**	0.01824	0.3439 (1)
Jackknifed tol. d/w WA	0.05630	0.2373	0.4322	0.03907	0.3559 (1)

This is still lower than the r^2 obtained from the lake data set after outlier removal (0.52 compared to 0.64). Further analysis of WA-PLS did not improve the performance of the data. This model is therefore rejected for use in pH reconstruction.

4.3.3.3 Combined data set

Summary results for pH WA regression for the full combined data set are shown in Table 4.12

Table 4.12 Results of pH Weighted Averaging (combined data set, 101 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.161	0.4013	0.6301	0	0.2901 (2)
Tol. d/w WA	0.14	0.3741	0.6621	0	0.2137 (1)
Jackknifed simple WA	0.2648	0.5146	**0.2954**	0.001107	1.151 (2)
Jackknifed tol. d/w WA	0.2651	0.5149	0.2694	-0.00071	1.293 (2)

The jackknifed r^2 for cross validated simple WA was 0.30, which compares to 0.41 for the full lake data set and 0.30 for the full stream data set. RMSEP values are high, indicating that pH in many samples was poorly predicted. These samples are apparent in the plots of measured pH versus predicted pH and predicted pH

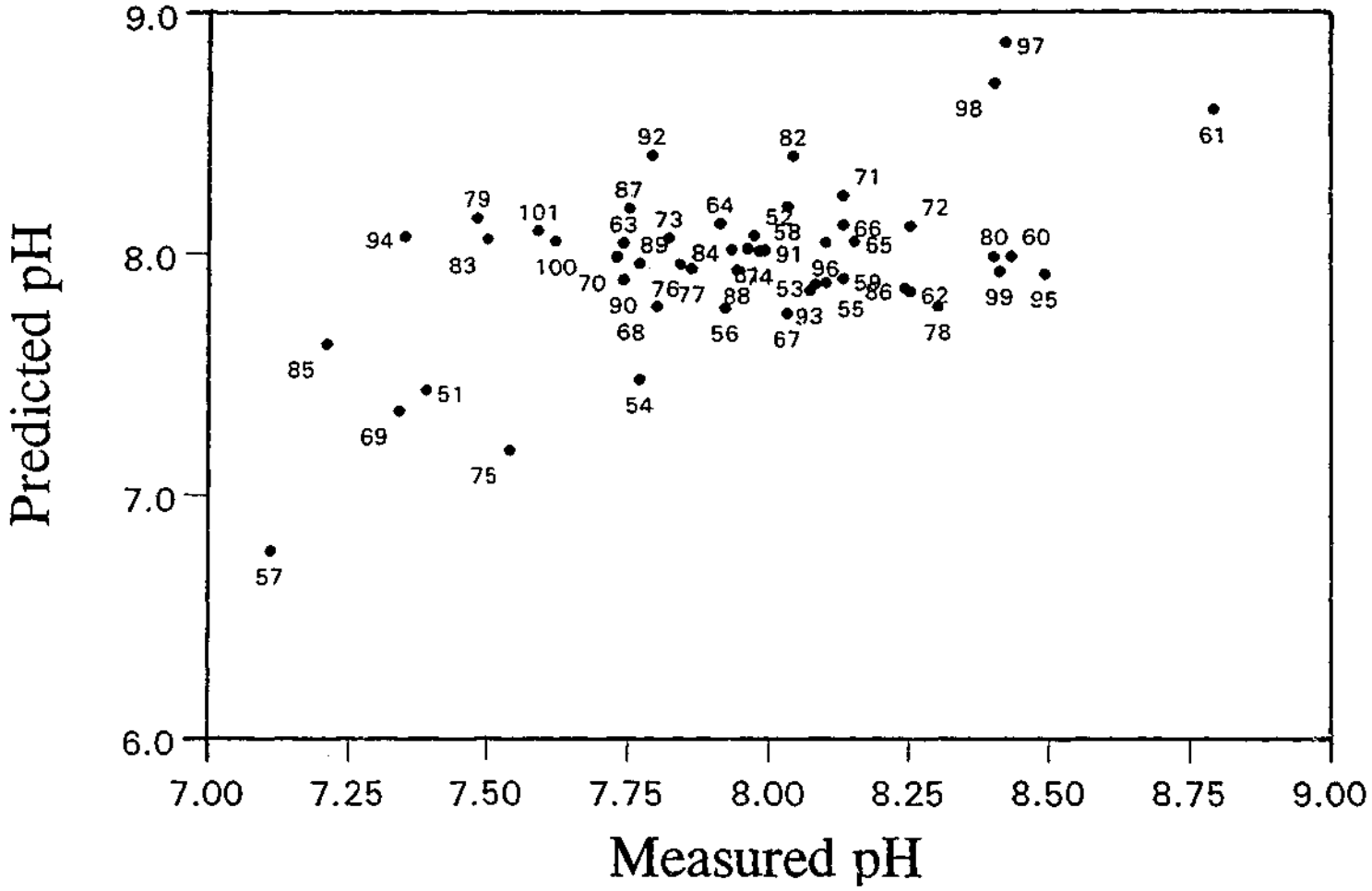


Figure 4.14 Measured pH versus predicted pH (stream data set, 51 samples)

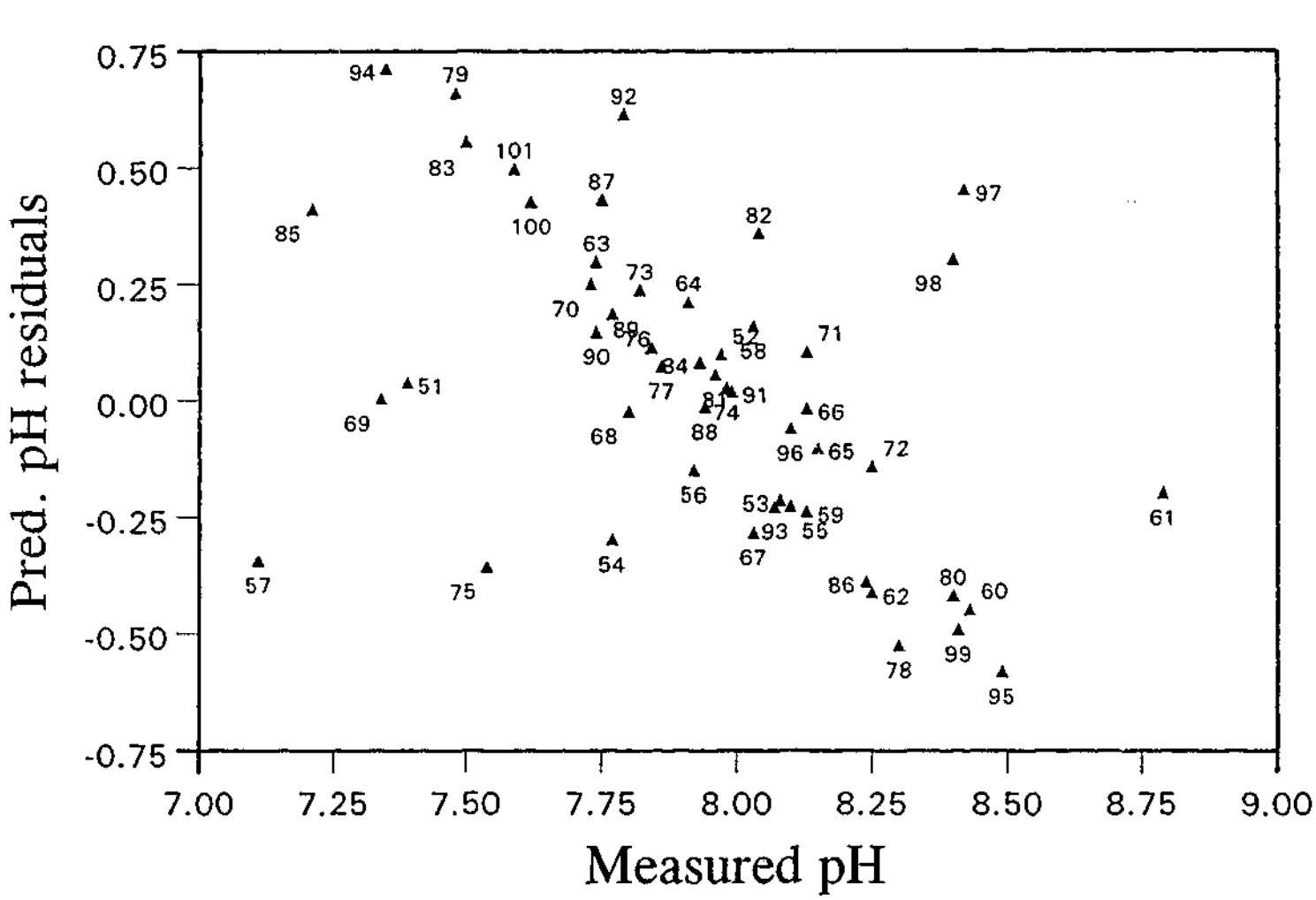


Figure 4.15 Plot of predicted pH versus residuals between actual and predicted values (stream data set, 51 samples)

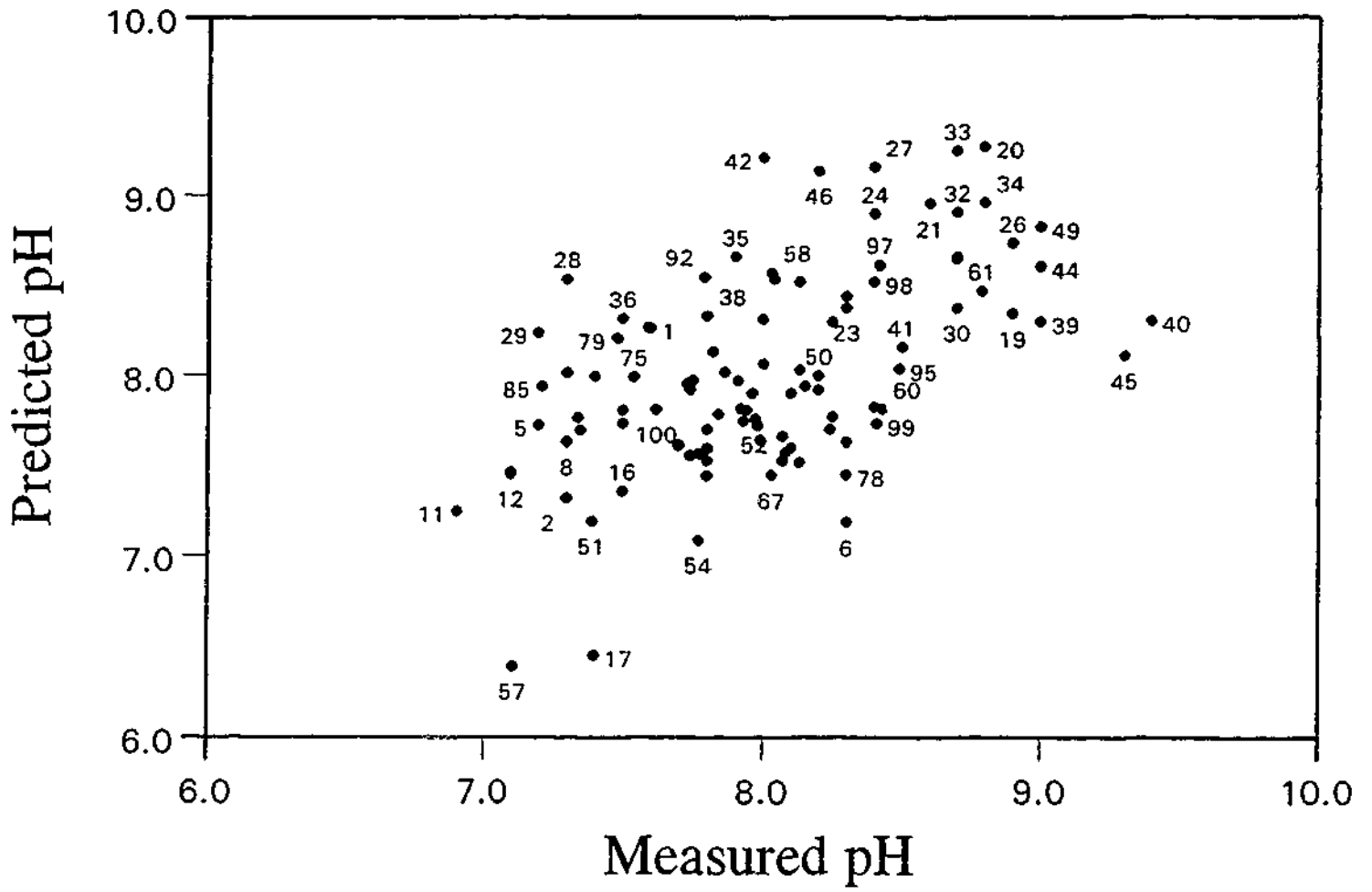


Figure 4.16 Measured pH versus predicted pH (combined data set, 101 samples)

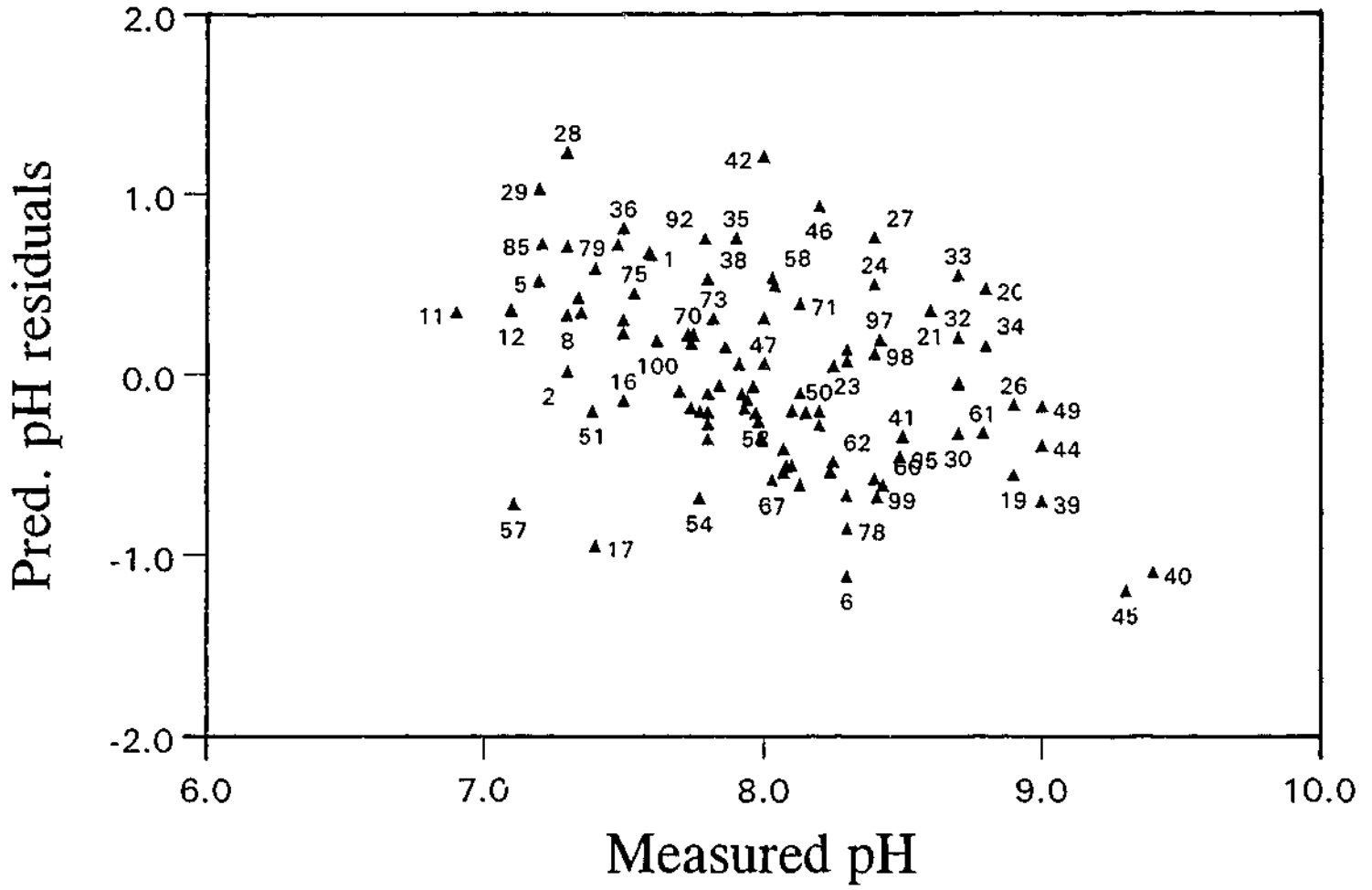


Figure 4.17 Plot of predicted pH versus residuals between actual and predicted values (combined data set, 101 samples)

versus residuals between actual and predicted values (figures 4.16 and 4.17). Twenty four samples were progressively removed from the data set, with the r^2 values increasing slightly upon each removal. The results for pH WA regression for a 77 site data set are shown in Table 4.13.

Table 4.13: Results of pH Weighted Averaging (combined data set, 77 sites)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.0722	0.2650	0.7762	0	0.1653 (7)
Tol. d/w WA	0.0489	0.2213	0.8325	0	0.2341 (7)
Jackknifed simple WA	0.1257	0.3545	**0.5484**	0.009874	0.3272 (5)
Jackknifed tol. d/w WA	0.1181	0.3436	0.5317	-0.01669	0.5207 (6)

Both the jackknifed r^2 and the RMSEP scores have greatly improved but the model is not suitable for accurate reconstruction / calibration purposes.

4.3.4 Development of an EC transfer function

4.3.4.1 Lake data set

Initial Weighted Averaging of diatom – EC relationships included all 50 lakes, the results of which are shown in Table 4.14.

Table 4.14 Results of EC Weighted Averaging (lake data set, 50 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.05984	0.2446	0.7322	0	0.3654 (3)
Tol. d/w WA	0.05556	0.2357	0.7513	0	0.3721 (3)
Jackknifed simple WA	0.2168	0.4657	0.1523	-0.01281	0.7163 (3)
Jackknifed tol. d/w WA	0.1865	0.4319	**0.2505**	-0.01027	0.7481 (3)

The best performing cross validated method included tolerance downweighting (r^2 of 0.25). Plots of measured versus predicted EC and predicted EC versus residuals between actual and predicted values (figures 4.18 and 4.19) highlight nine samples, which, in particular, decrease the performance of the model. These are summarised in Table 4.15.

Table 4.15 Samples with high residual EC values – lake data set (50 samples)

Sample	Actual EC	Predicted EC	Dominant diatom taxa
3 (Wetland, Greenwith)	200 µS/cm	1055 µS/cm	*Navicula capitata, Navicula gregaria*
4 (Wetland, Regent Gds)	310 µS/cm	1626 µS/cm	*Navicula cryptotenella, Planothidium delicatulum*
5 (Wetland, Paddocks)	200 µS/cm	1014 µS/cm	*P. delicatulum, Navicula cryptocephala*
6 (Wetland, Barker Inlet)	10 190 µS/cm	854 µS/cm	*Navicula cincta, N. cryptotenella, Nitzschia filiformis*
23 (Lake Lyndger)	6350 µS/cm	1220 µS/cm	*Amphora veneta, Navicula muraliformis,*
41 (Wargan basin)	4800 µS/cm	760 µS/cm	*Amphora coffaeformis, Cocconeis placentula, Cymbella pusilla*
42 (Hattah Lake 2)	640 µS/cm	4100 µS/cm	*Amphora cognata*
43 (Bullock Swamp)	9330 µS/cm	1250 µS/cm	*Aulacoseira granulata, Cyclotella meneghiniana*
45 (Cardross Basin)	5367 µS/cm	250 µS/cm	*Climaconeis sp.*

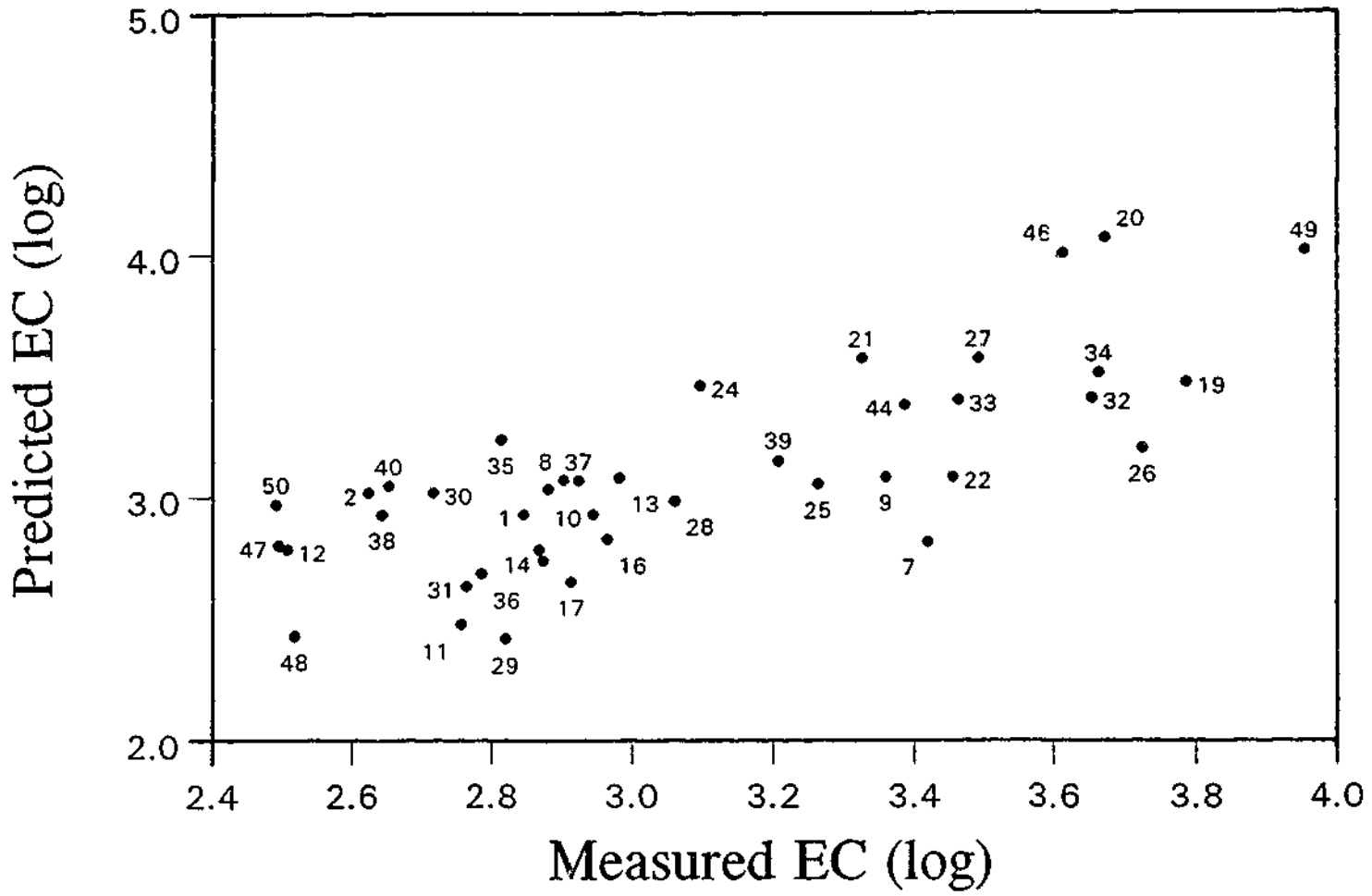


Figure 4.18 Measured EC (log) versus predicted EC (log) (lake data set, 50 samples)

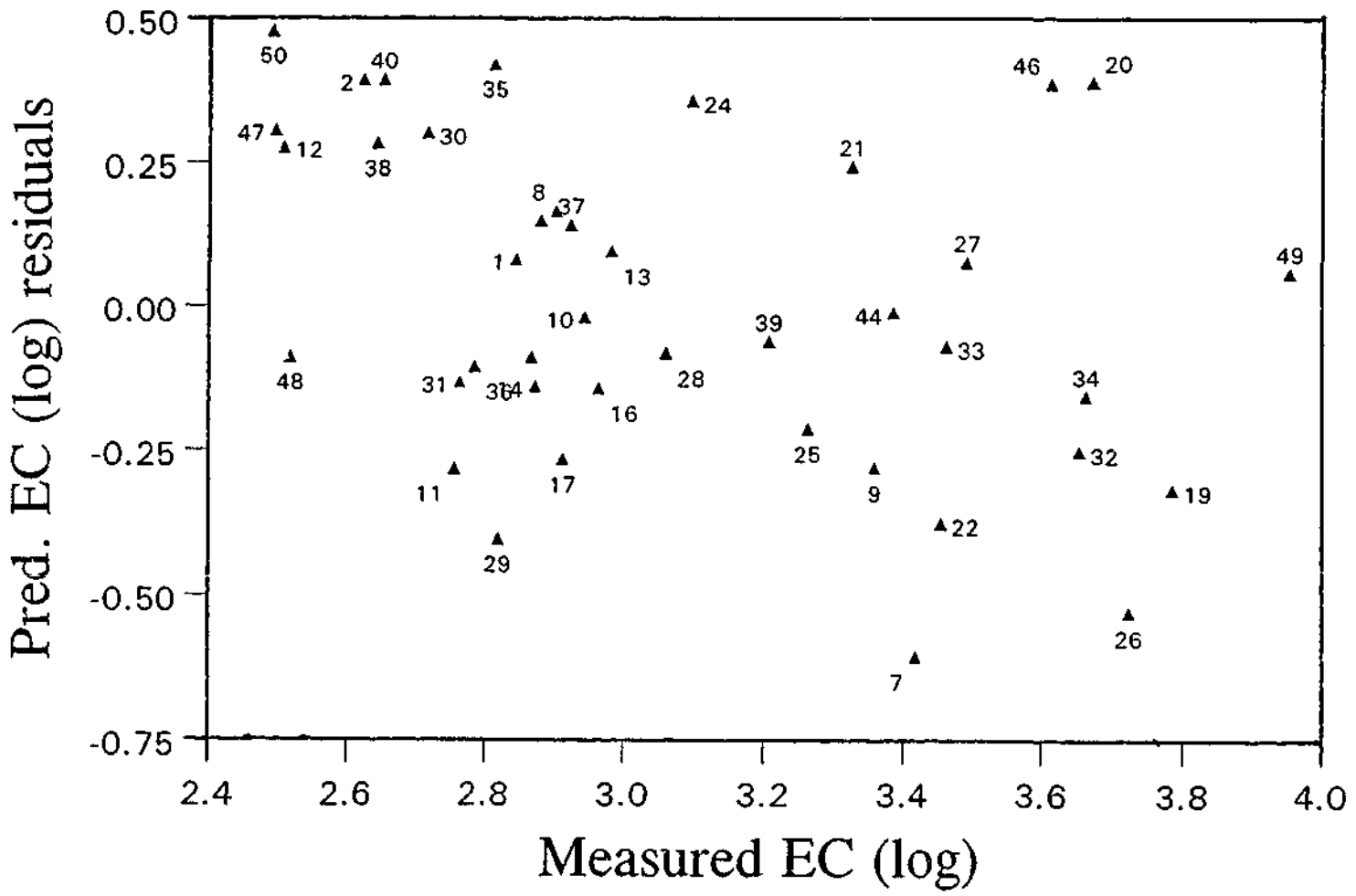


Figure 4.19 Plot of predicted EC (log) versus residuals between actual and predicted values (lake data set, 50 samples)

The difference between measured and predicted EC is very large for these samples, particularly samples 6, 23, 41, 42, 43, and 45. The high residuals at the wetland sites may be due to rapid changes in water salinity because of constant entry of stormwater and periodic flushing, resulting in a diatom flora that is not entirely representative of the modern water chemistry. Sampling of these systems was undertaken in regions where the effects of these processes were thought to be minimilised (i.e.: permanent holding ponds with established aquatic vegetation). However, it was possible that the water chemistry had changed considerably in the period immediately prior to sampling. Samples 23, 41, 43, and 45 all had greatly underestimated EC. Sample 43 was probably underestimated due to the diatom flora being dominated by *Aulacoseira granulata*, a taxon which occurs in over 20 lakes in the data set, the majority of which have a salinity less than 5000 µS/cm.

This illustrates one of the problems with WA that was discussed in Chapter 2, that of broad ecological tolerances. *Aulacoseira granulata* possesses a preference for fresh to brackish aquatic environments but it can also occasionally survive in waters with higher EC levels (Gell, 1995). However, because the majority of sites where it occurred were fresh, then the subsequent generated optimum was relatively low (583 µS/cm). Electrical Conductivity in sample 45 was underestimated due to its unique diatom flora - this was the only lake with *Climaconeis* sp. present and therefore the predicted EC was derived from the small number of other diatoms present in the sample, most of which were abundant in fresher environments (thus highlighting a further problem discussed in Chapter 2, that of adequate samples to generate reliable modern analogues). A similar circumstance exists with sample 42 which was dominated by *Amphora cognata* - a taxon that did not occur in any other sample.

These nine samples were removed from the data set and subsequent WA regression undertaken, the results of which are shown in Table 4.16

Table 4.16 Results of EC Weighted Averaging (lake data set, 41 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.03832	0.1957	0.808	0	0.2437 (1)
Tol. d/w WA	0.03021	0.1738	0.8422	0	0.2092 (1)
Jackknifed simple WA	0.07676	0.2770	0.5753	0.009042	0.2725 (5)
Jackknifed tol. d/w WA	0.06835	0.2614	**0.6220**	-0.00879	0.2593 (5)

The predictive power of the jackknifed WA for EC is much improved, with an r^2 of 0.62 (tolerance downweighted). However, RMSEP remains high (> 30% at 1000 µS/cm) and thus the model is not suitable for quantitative reconstruction. Further analysis using WA-PLS did not improve the performance of the data.

4.3.4.2 Stream data set

Summary statistics for the full stream data set for EC WA regression are provided in table 4.17.

Table 4.17 Results of EC Weighted Averaging on 51 streams

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.04915	0.2217	0.7485	0	0.4281 (2)
Tol. d/w WA	0.03807	0.1951	0.7935	0	0.4095 (2)
Jackknifed simple WA	0.14360	0.3790	**0.1548**	.04312	0.9312 (1)
Jackknifed tol. d/w WA	0.1636	0.4045	0.1139	0.1243	1.03 (1)

The best performing jackknifed model for the full stream data set was the simple WA model, with an r^2 of 0.15 and a RMSEP of 0.38. The summary plots identified eight samples with high residual values - 51, 54, 68, 79, 90, 94, 95 and 96 (figures 4.20 and 4.21). Removal of these outliers did improve the predictive power of the model (from 0.15 to 0.34 - table 4.18) but the r^2 is still low. WA-PLS regression did not improve the performance of the model and hence, no model was developed.

Table 4.18 Results of EC Weighted Averaging (stream data set, 43 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.02487	0.1577	0.7410	0	0.33 (2)
Tol. d/w WA	0.01987	0.1385	0.7878	0	0.2832 (2)
Jackknifed simple WA	0.0575	0.2398	**0.3362**	0.0231	0.3373 (3)
Jackknifed tol. d/w WA	0.0669	0.2586	0.2357	0.0502	0.44 (1)

4.3.4.3 Combined data set

The results of Weighted Averaging regression for EC – diatom relationships on the full 101 sites are shown in table 4.19. Figures 4.22 and 4.23 illustrate these results.

Table 4.19 Results of EC WA (combined data set, 101 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.1038	0.3222	0.6465	0	0.2035 (3)
Tol. d/w WA	0.1004	0.3169	0.6541	0	0.4428 (1)
Jackknifed simple WA	0.2629	0.5127	0.1264	0.02222	0.8692 (1)
Jackknifed tol. d/w WA	0.2889	0.5375	**0.1446**	0.0367	0.8755 (1)

The model performs poorly with low r^2 values and high RMSEP. Sixteen outliers were identified and were subsequently removed from the data set. Results in table 4.20 show improvement in the model, but it remains unsuitable for the derivation of taxon optima.

Table 4.20 Results of EC WA (combined data set, 85 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.07643	0.2765	0.6782	0	0.3302 (2)
Tol. d/w WA	0.1916	0.4377	0.4568	0	0.686 (2)
Jackknifed simple WA	0.1187	0.3445	0.3621	0.02288	0.7572 (2)
Jackknifed tol. d/w WA	0.09695	0.3114	**0.4140**	0.0332	0.7851 (2)

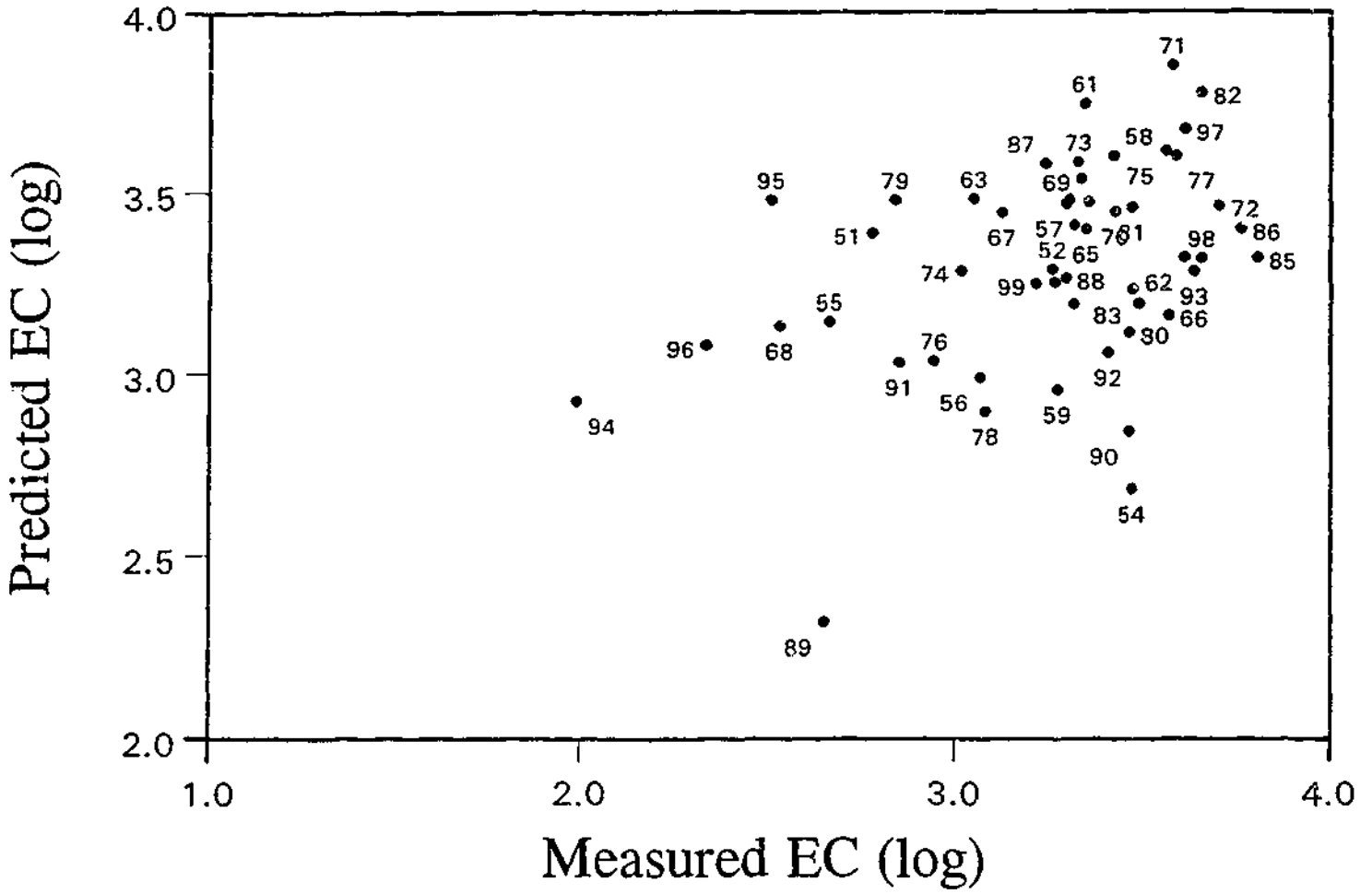


Figure 4.20 Measured EC (log) versus predicted EC (log) (stream data set, 51 samples)

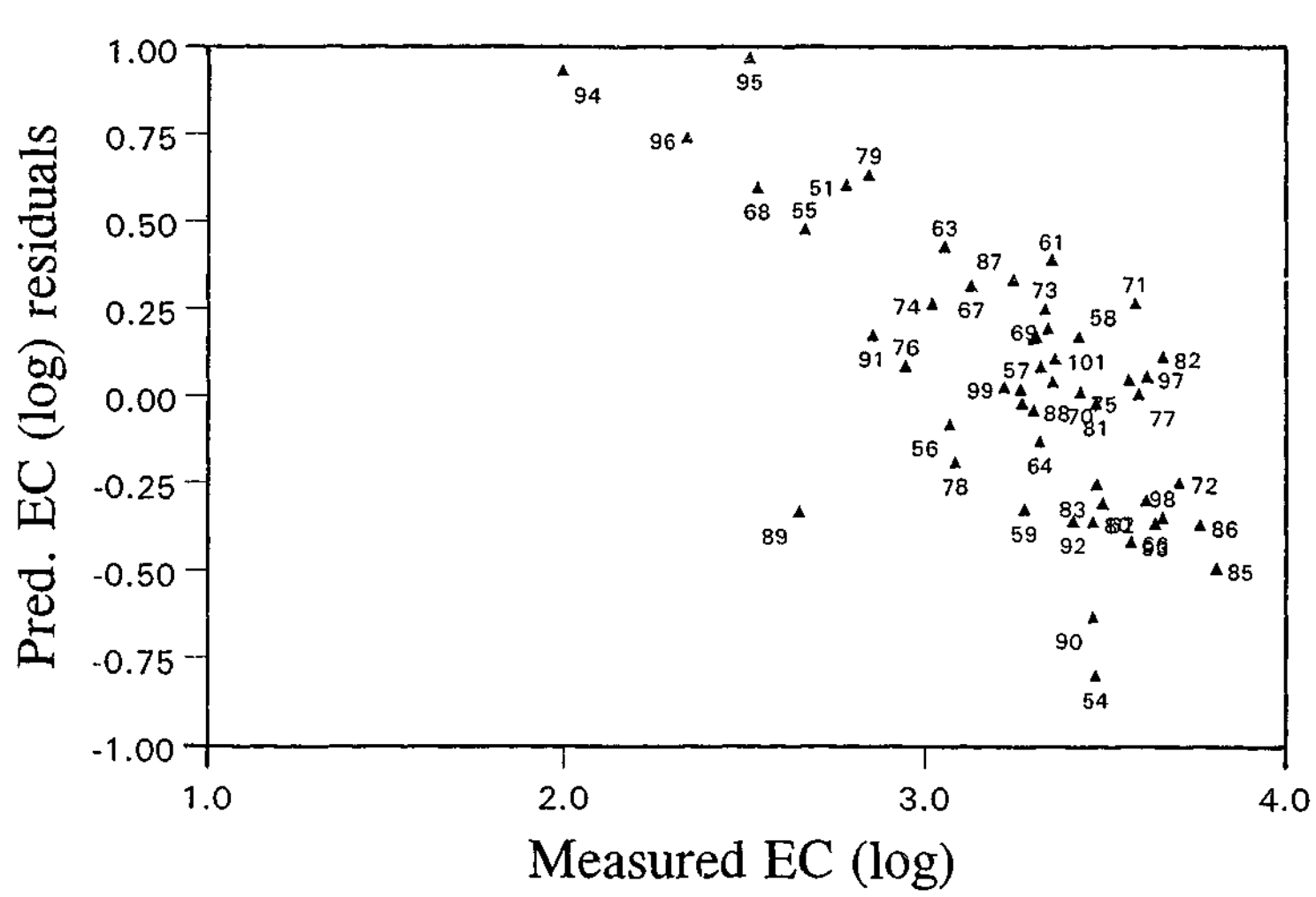


Figure 4.21 Plot of predicted EC (log) versus residuals between actual and predicted values (stream data set, 51 samples)

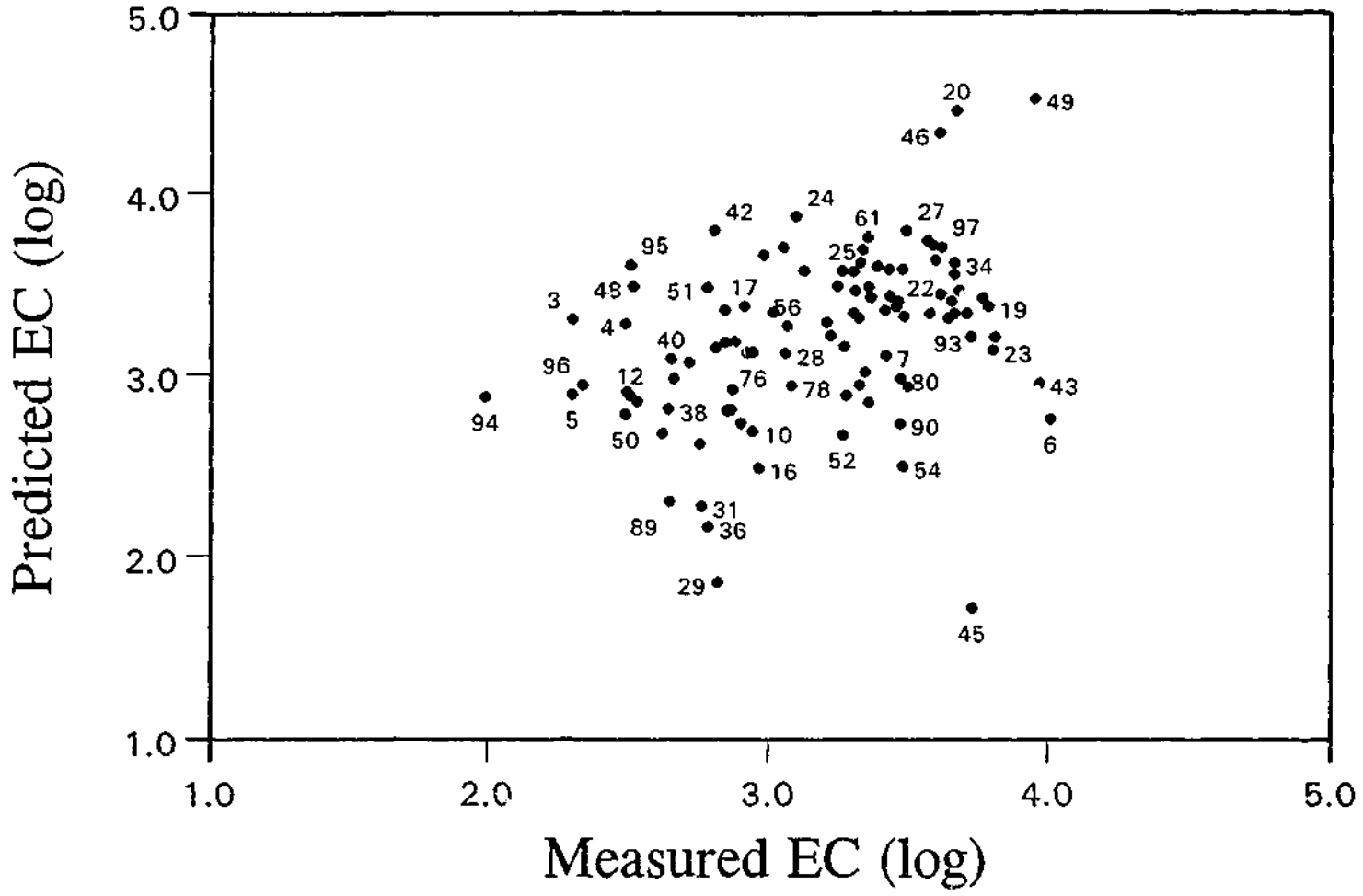


Figure 4.22 Measured EC (log) versus predicted EC (log) (combined data set, 101 samples)

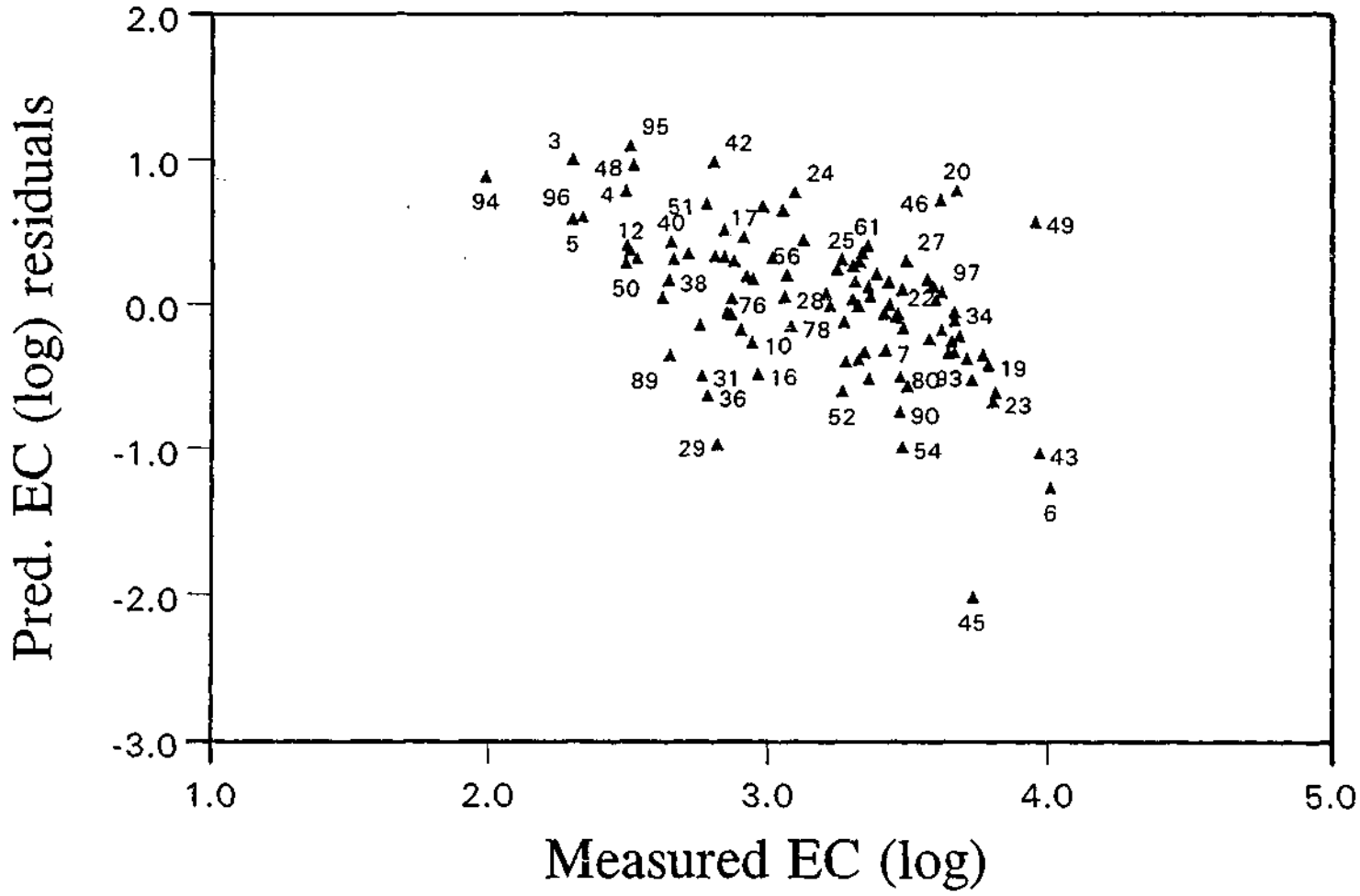


Figure 4.23 Plot of predicted EC (log) versus residuals between actual and predicted values (combined data set, 101 samples)

4.3.5 Development of a TP transfer function

4.3.5.1 Lake data set

Summary statistics for Weighted Averaging of TP for the 50 lake data set are shown in table 4.21. The jackknifed or predicted r^2 of 0.03 was very low. The plots of the relationship between measured TP and predicted TP and the residuals are shown in figures 4.24 and 4.25 Sixteen samples with high residuals were selected for removal from the data set. These samples included 1, 6, 7, 21, 23, 24, 25, 32, 33, 35, 37, 39, with the results of the regression shown in table 4.22.

Table 4.21 Results of TP WA (lake data set, 50 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.03563	0.1887	0.6691	0	0.5405 (1)
Tol. d/w WA	0.03687	0.1920	0.6576	0	0.6596 (1)
Jackknifed simple WA	0.1278	0.3575	0.0240	0.007791	0.8796 (1)
Jackknifed tol. d/w WA	0.1225	0.3500	**0.0337**	-0.01816	0.8855 (1)

Table 4.22 Results of TP Weighted Averaging (lake data set, 34 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.01257	0.1121	0.7819	0	0.1776 (3)
Tol. d/w WA	0.007932	0.0891	0.8623	0	0.1449 (3)
Jackknifed simple WA	0.03835	0.1958	**0.3551**	0.01592	0.2773 (3)
Jackknifed tol. d/w WA	0.04621	0.2150	0.2416	0.01574	0.3611 (1)

Although the model was improved with an increase in predicted r^2 from 0.03 to 0.36, this is still unacceptable for developing reliable taxa optima for TP. As 16 samples were removed from the data set, reducing the size to 34 lakes, it was not considered feasible to remove any more outliers. Further analysis of WA-PLS did not improve the performance of the data set. Therefore, it can concluded that the lake data set was not suitable for the development of TP optima.

4.3.5.2 Stream data set

Summary results of WA regression for TP for the 51 stream samples is shown in table 4.23.

Table 4.23 Results of TP Weighted Averaging (stream data set, 51 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.1663	0.4078	0.7880	0	0.2605 (2)
Tol. d/w WA	0.2229	0.4722	0.7350	0	0.9893 (2)
Jackknifed simple WA	0.4001	0.6326	**0.3855**	-0.007221	1.393 (2)
Jackknifed tol. d/w WA	0.5446	0.7379	0.2400	-0.1232	1.951 (2)

Plots of measured versus predicted TP and predicted TP versus residuals between actual and predicted values (figures 4.26 and 4.27) identified three samples that had very high residuals - 61, 100 and 101. These were removed from the regression analysis, with the results reported in table 4.24.

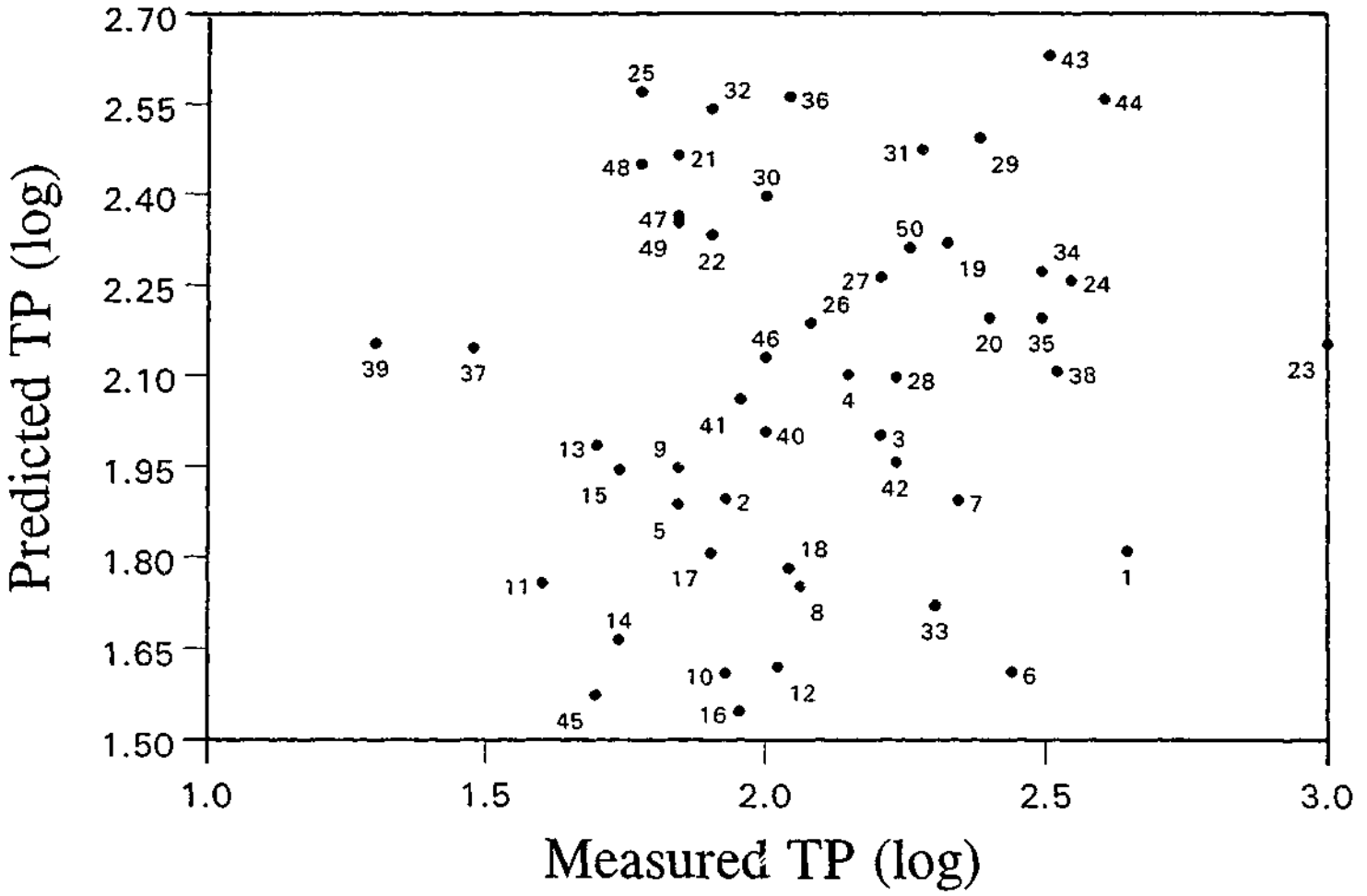


Figure 4.24 Measured TP (log) versus predicted TP (log) (lake data set, 50 samples)

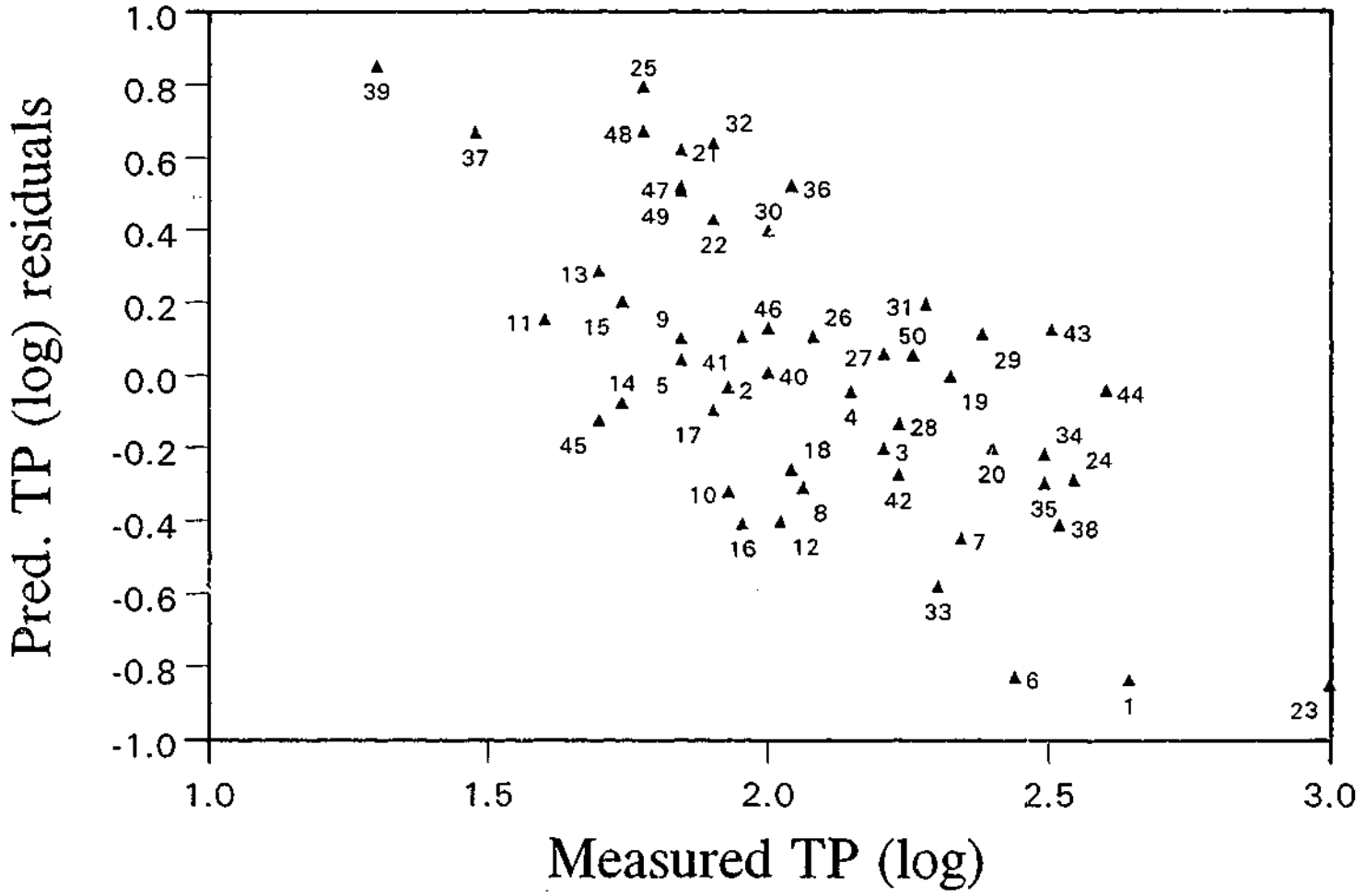


Figure 4.25 Plot of predicted TP (log) versus residuals between actual and predicted values (lake data set, 50 samples)

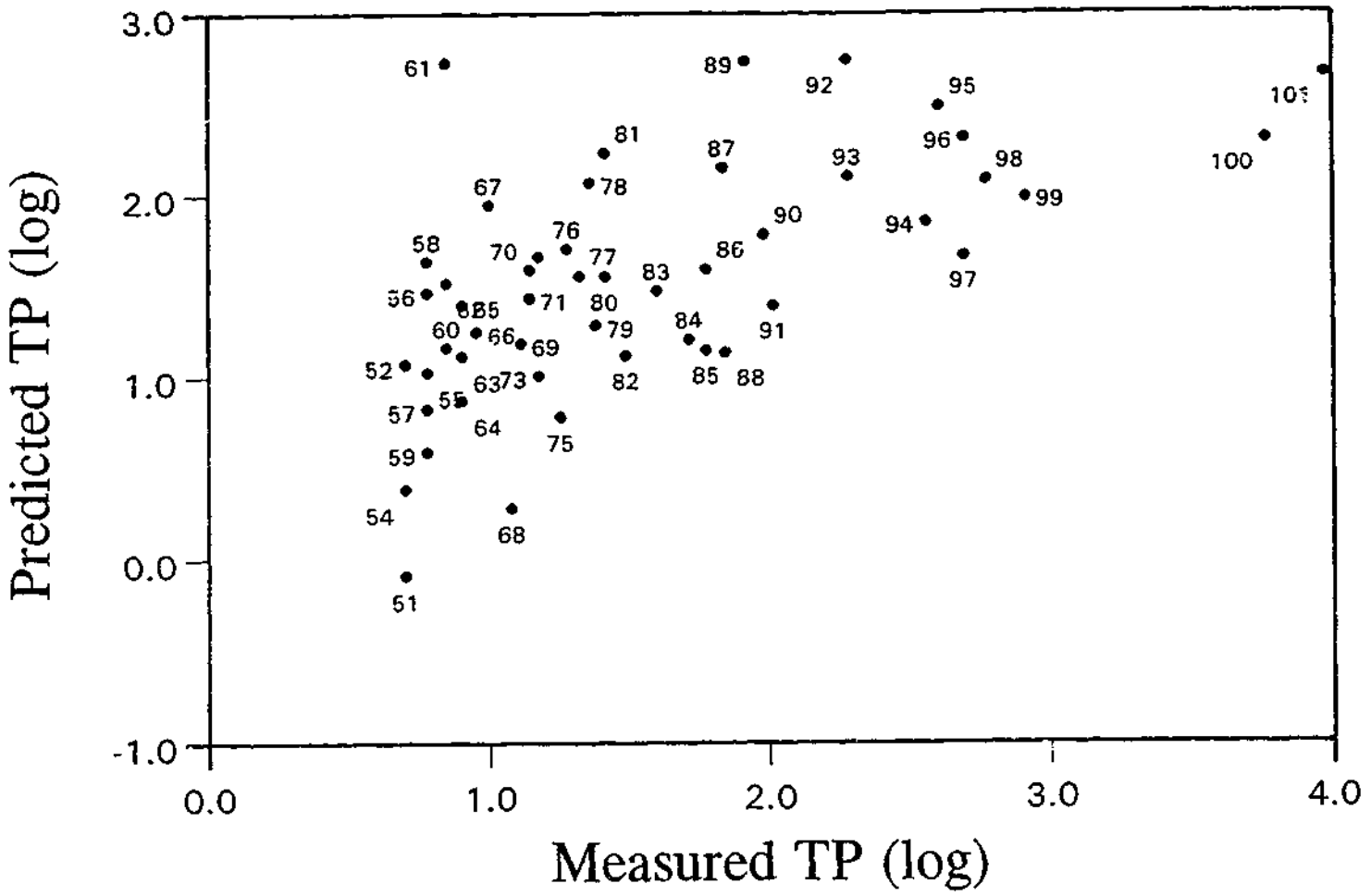


Figure 4.26 Measured TP (log) versus predicted TP (log) (stream data set, 51 samples)

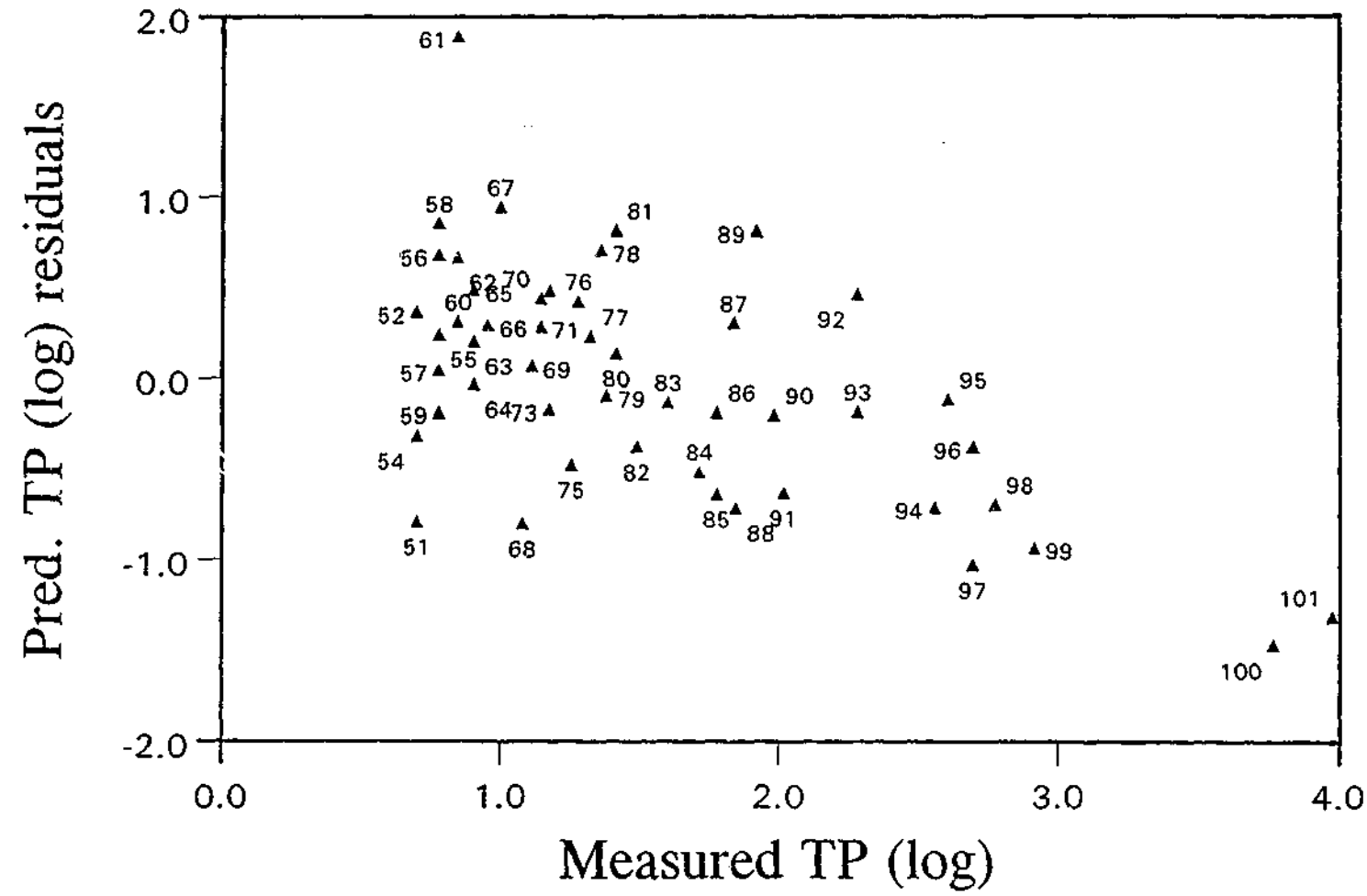


Figure 4.27 Plot of predicted TP (log) versus residuals between actual and predicted values (stream data set, 51 samples)

Table 4.24 Results of TP Weighted Averaging (stream data set, 48 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.105	0.3241	0.7961	0	0.2825 (2)
Tol. d/w WA	0.09925	0.3150	0.8052	0	0.51 (2)
Jackknifed simple WA	0.2363	0.4861	**0.4513**	0.001112	0.7054 (2)
Jackknifed tol. d/w WA	0.3655	0.6046	0.2493	-0.06379	1.008 (2)

Despite samples 61, 100 and 101 having much higher residuals than the other samples, their removal did not result in a great improvement in r^2 (0.39 to 0.45). Additionally, it was found that removal of any further samples, such as 97 or 58, which had the next highest residuals, led to a decrease in r^2 values. Further WA regression by WA-PLS did not result in an improvement in the model. Therefore it can be concluded that the stream data set was not appropriate for the development of diatom TP optima.

4.3.5.3 Combined data set

WA regression was applied to the combined data set with the results reported in table 4.25.

Table 4.25 Results of TP WA (combined data set, 101 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.2417	0.4917	0.6492	0	0.9528 (2)
Tol. d/w WA	0.293	0.5413	0.6043	0	1.224 (2)
Jackknifed simple WA	0.4003	0.6327	**0.3263**	-0.001627	1.62 (2)
Jackknifed tol. d/w WA	0.4833	0.6952	0.2689	0.006513	1.751 (2)

The jackknifed r^2 was higher than that for the initial WA regression from the lake data set (0.03) and was slightly lower than the score for the initial WA regression for the stream data set (0.39). RMSEP was very high at 0.63, a result of many samples with high residual values (as seen in figures 4.28 and 4.29). It was found that the model performed best after the removal of 19 samples, which are detailed in table 4.26.

The majority of outliers are stream samples (16), with no obvious explanation for this result. There is the possibility that common diatom taxa in the lake data set had a different response to TP than the same taxa in streams. The only common characteristic between the measured physio-chemistry of the 19 samples is that they all had a relatively high EC, with 13 of the samples having EC > 2000 µS/cm. This suggests that diatom taxa assemblages in these samples are more influenced by EC than nutrients. However, there were 17 lake samples and 40 stream samples, other than the above outliers, which also had EC > 2000 µS/cm but didn't have high residuals between measured and predicted TP. Data set heterogeneity is unlikely to be a possible explanation for poor TP prediction, as most of the dominant diatom taxa in these samples occurred in at least 5 other samples (with the exception of sample 42). It is likely then that these outliers were responding to unmeasured environmental variables, such as water depth, degree of vegetation cover, or perhaps heavy metal concentrations.

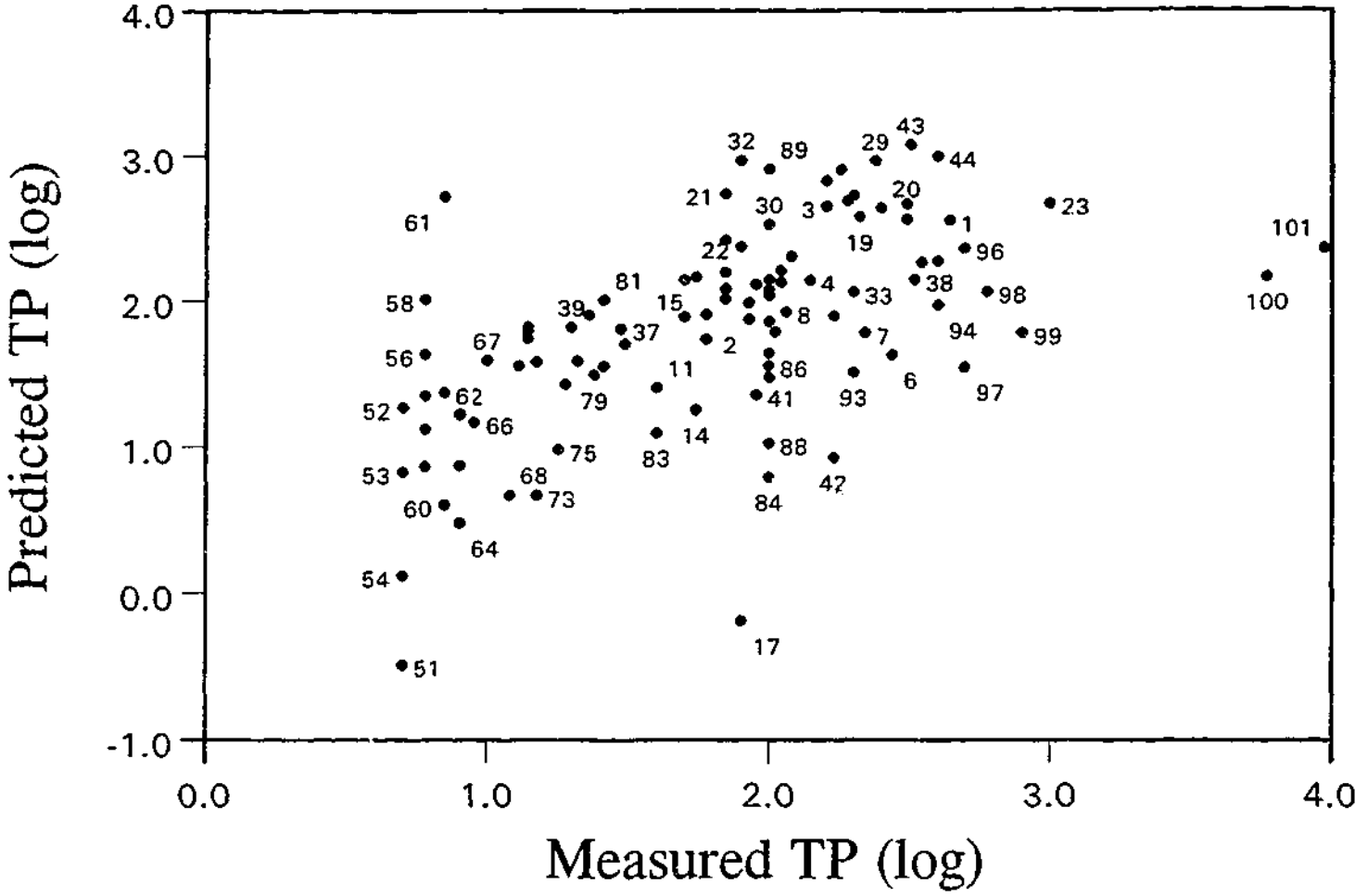


Figure 4.28 Measured TP versus predicted TP (combined data set, 101 samples)

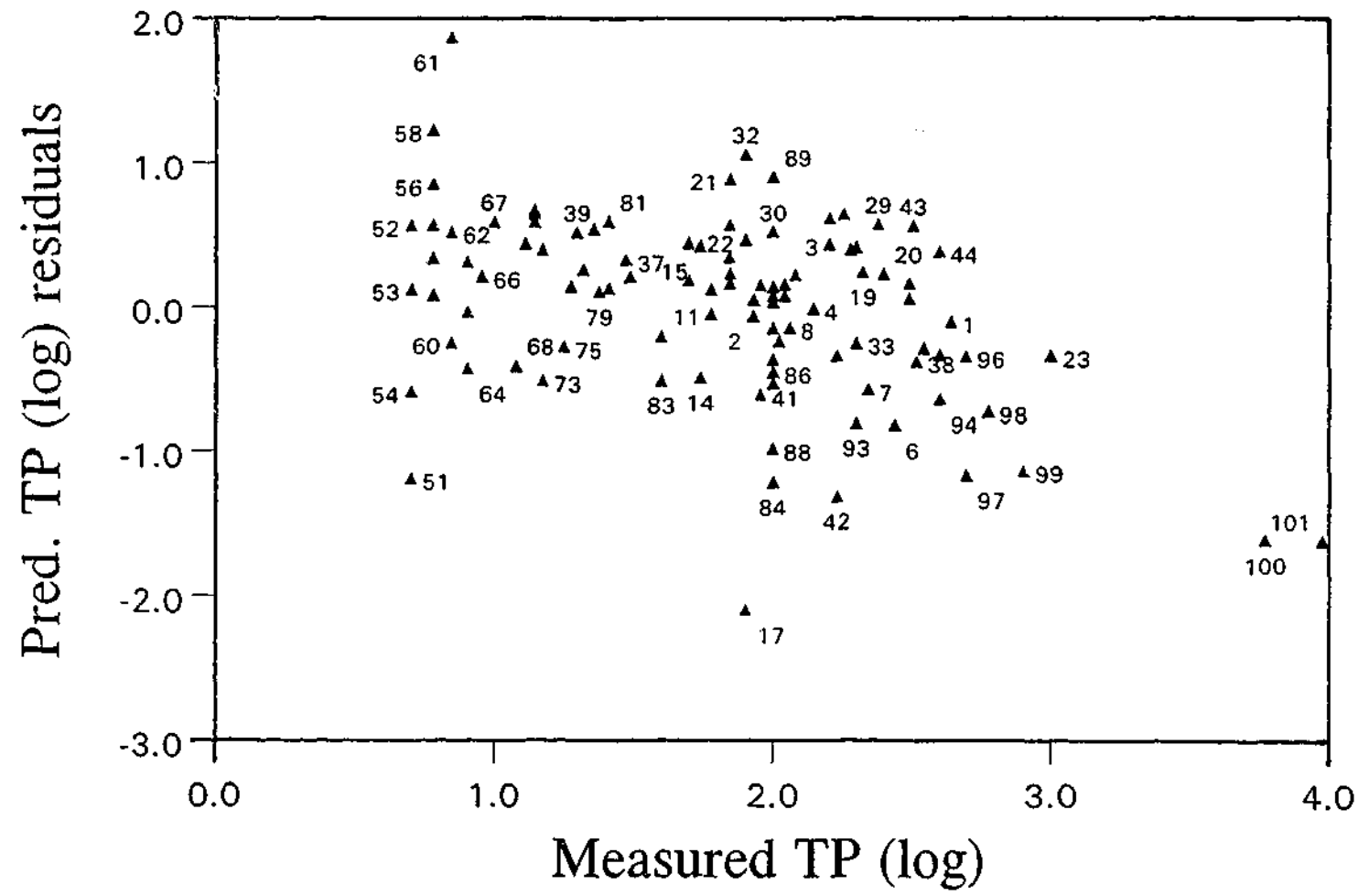


Figure 4.29 Plot of predicted TP versus residuals between actual and predicted values (combined data set, 101 samples)

Table 4.26 Details of outliers - TP WA (combined data set)

Sample	TP	Predicted TP	Dominant taxa
6 (Wetland, Barker Inlet)	275	41	*Achnanthidium minutissimum, Navicula cincta, Navicula cryptotenella*
17 (Barossa Reservoir)	80	1	*Achnanthidium minutissimum, Diatoma tenuis*
21 (Greens Lake)	70	538	*Actinocyclus normanii*
32 (Lake Charm)	80	907	*Cyclotella meneghiniana*
42 (Hattah Lake B)	170	8	*Amphora cognata*
56 (Eight Mile Creek)	6	42	*Cocconeis placentula, Melosira varians*
58 (Mosquito Creek)	6	101	*Cymbella pusilla, Pseudostaurosira brevistriata*
61 (Callender Drain)	7	515	*Staurosira construens* forma *venter*
70 (Inman River)	14	65	*Nitzschia acicularis, Surirella brebisonii*
72 (Wakefield River)	14	61	*Bacillaria paradoxa, Pseudostaurosira brevistriata*
84 (Yankalilla River)	52	6	*Achnanthes exigua*
88 (Deep Creek)	100	10	*Epithemia sorex, Melosira varians*
89 (Murray River)	83	793	*Aulacoseira granulata*
93 (Naracoorte Creek)	190	31	*Tryblionella constricta, Rhoicosphenia abbreviata*
97 (North Para River)	490	34	*Staurosira construens* forma *venter*
98 (North Para River)	598	112	*Staurosira construens* forma *venter, Amphora veneta*
99 (Mt Barker Creek)	821	58	*Navicula gregaria, Rhoicosphenia abbreviata*
100 (Inman River)	5860	143	*Gomphonema parvulum*
101 (Mt Barker Creek)	9480	226	*Gomphonema parvulum, Nitzschia frustulum.*

These 19 samples were removed from the data set, with the WA regression results shown in table 4.27.

Table 4.27 Results of TP WA (combined data set, 82 samples)

	MSE	RMSEP	r^2	AvgBias	MaxBias
Simple WA	0.0766	0.2768	0.8199	0	0.2579 (1)
Tol. d/w WA	0.07899	0.2810	0.8153	0	0.2819 (1)
Jackknifed simple WA	0.1366	0.3695	**0.6218**	-0.001744	0.8115 (1)
Jackknifed tol. d/w WA	0.1966	0.4434	0.4850	0.04256	0.8171 (1)

Removal of these outliers results in a dramatic improvement in the simple WA model with jackknifed r^2 increasing from 0.33 to 0.62. The values for RMSEP decrease from 0.6327 to 0.3695. Further analysis using WA-PLS did not improve the performance of the data set, although results were comparable. Although this result fulfils part of the established criteria for the generation of accurate environmental optima, the RMSEP is too high (more than 30% at 50 µg/L), and thus the model is rejected.

4.3.8 Summary of the Weighted Averaging regression results

This study followed the established procedure for the development of a WA based transfer function. In summary, this involved firstly the use of CCA to determine which variables were responding the most to changes in diatom assemblages. For all three data sets the most influential variable was pH. In order of decreasing magnitude, the next most influential variables in the lake data set were DO, EC, TP and TKN. For the stream data set, they were turbidity, TP, $\%Mg^{2+}$, and shade, and for the combined data set they were TKN, turbidity, EC and DO.

Weighted Averaging regression was carried out on three environmental parameters (pH, EC and TP) for three data sets (lake, stream and combined). All nine models initially performed poorly with $r^2 < 0.50$. In most analyses, simple WA models (jackknifed) produced better results than tolerance downweighted models. The WA models could not be improved by WA-PLS regression (with 6 components), although results were comparable to those of simple cross-validated WA. All models were improved by the removal of outlying samples which were selected on the basis of their residual values and ecological relevance. Outliers were omitted sequentially in order of magnitude until the model could no longer be improved. The number of samples removed varied for each model, with no data set reduced to fewer than 34 samples.

In terms of model predictive power, following outlier deletion, pH performed best for both the lake and stream data sets (with better results for the lake data set), and was the second best performing variable in the combined data set. Similar trends were observed for EC, where best results were derived from the lake data set, with the stream and combined data sets showing a weaker relationship between measured and predicted EC. Total phosphorus WA regression performed poorly in both the lake and stream data sets, with r^2 values of 0.36 and 0.45 respectively, but performed relatively well in the combined data set with an r^2 of 0.62. However, the RMSEP was too high to generate accurate reconstructions.

When comparing the suitability of the lake and stream data sets for the development of a reliable transfer function, as a generalisation, it was found that the lake data set provided better r^2 results for most parameters. Combining the two data sets, and thus extending the range of each variable and the number of samples, did not improve data set performance (with lower RMSEP for all variables). This may be due to diatom taxa responding differently to environmental parameters in lakes than they do in streams and therefore, when the data sets were combined, there were conflicting results for taxa optima and tolerances, which weakened the relationship. Additionally, combining data sets may have resulted in data congestion at certain points of the environmental gradients, rather than acting to "fill in the gaps". For data sets with a concentration of samples at either end of the environmental gradients, there is a greater risk of over or under estimating taxon optima. This factor certainly contributed to poor WA performance for the parameters pH and EC.

In summary, the only model with predictive capabilities considered high enough to produce reliable and accurate environmental optima for the lower Murray River fossil records, was pH.

4.3.9 Weighted Averaging calibration of the fossil records

The majority of transfer function studies involve establishing a data set of lakes / streams where all parameters, other than the parameter that is the focus, remain relatively unchanged (Birks, 1995). For instance, if pH is the variable to be reconstructed, then TP and EC values are kept within a tight range so that there is a decreased chance of these parameters having an influence on taxon variance. The major problem with this, however, is that there is a possibility that these other parameters would not have remained constant or within a narrow range in the past. If the aim is to reconstruct pH values, then it must be considered that TP or EC may have also changed. This holds particularly true for water bodies in Australia because of their highly variable nature. It is not sufficient to know what taxon response to pH is under unchanging TP and EC conditions – a palaeolimnological study needs to be able to incorporate how individual diatom taxa respond under all conditions. It is therefore imperative that information is gained about a taxon's "ecological" niche, that is, its preference for a particular suite of parameters. In lakes dominated by littoral floras (such as lower River Murray aquatic systems), it is likely that taxon abundances are determined by a more complex set of interactions than those of deeper plankton dominated lakes (Reid 1997). This problem of multiple variable influence has been discussed by many workers, including Anderson *et al.* (1993), Fritz *et al.* (1993), Jones and Juggins (1995) and Hall *et al.* (1997).

Given the known complexity of the lower Murray River, and that the Weighted Averaging analysis resulted in information for pH only, it was deemed imperative that more data be gained about the nutrient and EC preferences of the more common fossil record taxa. It was therefore decided to reconstruct the fossil records using both quantitative and qualitative means, as quantitative means alone would not be sufficient to reconstruct likely historical limnological conditions. To facilitate this, the following chapter includes a review of pH, EC and TP optima for eighteen common taxa from more than twenty key transfer function studies, as well as descriptive results from ten additional studies. A summary of this information is then used to qualitatively describe the changes in the fossil record.

Chapter 5 - A review of the autecology of select diatom taxa

5.1 Introduction

This chapter examines the modern results of this study and similar studies to further investigate individual diatom species response to specific environmental variables, namely pH, EC and TP. The primary aim of this chapter is to gain as much ecological information as possible about select diatom taxa in order to facilitate a qualitative reconstruction of the lower Murray River fossil records. More than twenty studies are reviewed, data on species pH, EC and TP optima and ranges extracted, and then these data are compared to ecological responses from this study's data set. Where possible (i.e.: for those studies that have published the relevant data), information on number of samples and maximum abundances from which optima and ranges are derived is also included.

Results from this study are examined further by graphing diatom taxon relative abundance across environmental gradients (figures 5.1 through to 5.18). This is intended to capture and assess individual taxon response, rather than the "community" response which is assessed in CCA and Weighted Averaging regression. For instance, with the WA regression analysis presented in Chapter 4, certain taxa may exhibit a strong ecological response to a particular variable, but the strength of the response (as evaluated by the r^2 values) may have been downweighted by a weaker response by other taxa. To illustrate this, across a diatom / environmental variable data set there may be a small number of taxa that display a narrow response to a particular variable. However, this information is often lost if the remainder of the taxa within the data set have a weak response to the variable, thus leading to poor correlation coefficients. Graphing the individual taxon responses, although simple in approach, therefore allows a more thorough examination of ecological relationships.

Eighteen diatom taxa were chosen for autecological description. Fifteen taxa were selected on the basis of being dominant in the lake and stream data sets and also dominant and / or important in the sediment cores from the lower Murray River (fossil records are presented in the Chapters 6 and 7). These taxa include *Achnanthidium minutissimum, Aulacoseira granulata, Cocconeis placentula, Cyclotella meneghiniana, Gomphonema* [illegible] *Melosira varians, Navicula cryptocephala, Navicula veneta, Nitzschia palea, Planothidium* [illegible] *Pseudostaurosira brevistriata, Staurosira construens* forma *venter, Staurosirella pinnata,* [illegible] *Ulnaria fasciculata*. A further three taxa, although not dominant in either the lake or [illegible] data sets, were chosen for inclusion because they were abundant in the palaeolimnological records. These taxa comprise *Aulacoseira subarctica* forma *subborealis, Epithemia adnata,* and *Rhopalodia gibba*.

For all 18 diatom taxa, ecological data from a range of studies is presented, including a comparison between overseas and Australian results. A summary ecological description is then proposed, including details on pH,

EC and TP preferences and any additional information which was gained from the literature, such as habitat preference and preservation potential. Where taxa appear to have a well defined response to an environmental variable, in both the literature and this study, a quantitative summary is proposed. For those taxa where there is little agreement between the literature and where the graphs illustrating environmental gradient distribution for this study show a wide ranging distribution, a more descriptive, and in some cases inconclusive, summary is proposed.

Clear specimens of these reviewed taxa were photographed and are included in Appendix 6.

5.2 Summary of reviewed literature

A diverse range of diatom studies were examined, spanning five continents (Europe, Asia, North America, Australia and Africa). Methods used in these studies for the development of ecological data included WA regression techniques, twinspan classification, and multiple linear regression. Two studies (van Dam *et al.* 1994, and Denys 1991/92) that assigned broad based ecological preferences to diatom taxa based on findings from surface samples in the Netherlands and Belgium respectively, were also included. Numerous other studies were reviewed for additional qualitative information, including Cholnoky (1968), Shroeman (1972), Sládecek (1986), Whitmore (1989) and Hotzel and Croome (1996).

Details of the reviewed quantitative literature, including number of samples, range of parameters, methods used, correlation coefficients (if applicable), and region are provided in table 5.1. Direct comparisons of correlation coefficients are complicated by some studies not cross-validating the WA regression through either bootstrapping or jackknifing. The difference between non cross-validated and cross-validated regressions can be great with the predictive errors of the later being much more robust, as seen in this study. Gasse *et al.* (1995) found that jackknifed estimates of the prediction error (RMSEP) were between 18% and 32% higher than the corresponding apparent errors (non cross-validated), which the authors state highlights the importance of using a method of cross-validation to estimate likely error when the transfer functions are applied to unknown samples. Gasse *et al.* (1995) further state that while apparent measures allow direct comparisons with other published transfer functions, they are less reliable indicators of the true predictive ability of the transfer function as they are more biased by sample resubstitution.

In order to maintain a degree of taxonomic consistency between studies, diatom taxon plates / micrographs were examined from some of the reviewed studies. These include Vyverman (1992), Reavie *et al.* (1995), Gasse (1986), Gell (1995), Reid (1997) and Tibby (2000). Consultation was also undertaken with the authors of Bennion (1993), Bennion *et al.* (1996), and some of the participants of Stevenson *et al.* (1991), where samples from both sets of studies were examined and compared using a Zeiss Axioskop 2 FS light microscope.

Table 5.1: Details of the reviewed studies, including range of studied parameters, number of samples and methods used.

Author	Environmental parameters studied	Range of parameters	Number of samples	Methods used	r^2	Region
Agbeti, 1992	TP	2 - 63 µg/L	30	WA (not cross -validated)	0.86	Canada
Bennion, 1993	TP	25 - 646 µg/L	31	WA (bootstrapped)	0.79	SE England
Bennion *et al*, 1996	TP	5 - 1190 µg/L	152	WA (jackknifed)	0.91	Europe
Dixit and Smol, 1994	pH TP	5.3 - 9.0 (pH) 0.8 - 154.5µg/L (TP)	66	WA (not cross -validated)	TP = 0.62 pH = 0.86	NE USA
Dixit and Smol, 1995	pH	4.3 - 7.8	20	WA (not cross - validated)	0.89	NE USA
Dixit *et al.*, 1999	pH TP	n/a	309	WA (bootstrapping)	pH = 0.89 TP = 0.55	NE USA
Fritz *et al.* 1993	salinity	0.65 - 270 g/L	66	WA (jackknifed)	0.66	Northern Great Plains, US
Gasse *et al.* 1995	pH EC	5.5 - 10.9 (pH) 40 - 99 060 µS/cm (EC)	249 (pH) 274 (EC)	WA (jackknifed)	pH = 0.63 EC = 0.81	Africa
Gell, 1995	salinity	0.5 - 133.0 g/L	156	WA (jackknifed)	0.87	SE Australia
Reavie *et al.* 1995	TP	5 - 138 µg/L	64	WA (bootstrapped)	0.46	British Columbia
Reed, 1995	salinity	0.15 - 338 mS/cm	70	WA (jackknifed)	0.57	Spain
Reid, 1997	TP pH	20 - 3100 µg/L (TP) 5.38 - 8.8 (pH)	31 (TP) 42 (pH)	WA (jackknifed)	pH = 0.68 TP = 0.71	SE Australia
Sincock (1997)	PO_4	13 - 1142 µg/L	80	WA (jackknifed)	0.42	SE Australia
Stevenson *et al.* 1991	pH	4.23 - 7.25	167	WA (bootstrapped))	0.91	Europe
ter Braak and Van Dam, 1989	pH	3.3 - 7.3	97	WA (tested on external data set)	0.33 - 0.79	Western Europe
Tibby (2000)	TP	7 - 451 µg/L	29	WA (jackknifed)	0.80	SE Australia
Vyverman, 1992	pH	3.4 - 9.5	145	Twinspan		PNG
Whitmore, 1989	TP (TSI)		30	Multiple regression and TROPHI (not cross-validated)	0.85 and 0.83	Florida
This study	pH	6.9 - 9.0	45	WA (jackknifed)	0.64	SE Australia

The major pH studies reviewed were Stevenson *et al.* (1991), Vyverman (1992), Dixit and Smol (1994), ter Braak and van Dam (1994), Dixit and Smol (1995), Gasse *et al.* (1995), Reid (1997), Dixit *et al.* (1999) and this study. TASDIAT (Vyverman *et al.* 1995), a pH data set derived from Tasmanian lakes, was considered for inclusion as it is currently the only study to publish pH optima for Australian aquatic systems, but was rejected as the pH range was greatly skewed towards acidic sites. The best performing data set was the SWAP data set (Stevenson *et al.*, 1991) with a bootstrapped r^2 of 0.91. Correlation coefficients from other results which were cross validated include 0.89 in Dixit *et al.*(1999), 0.68 in Reed (1997) and 0.63 in Gasse *et al.* (1995).

The major EC / salinity studies reviewed include Denys (1991/92), Fritz *et al.* (1993b), van Dam *et al.* (1994), Gasse *et al.* (1995), Gell (1995), and Reed (1995). Out of these, the best performing cross-validated data set was that of Gell (1995) with an r^2 of 0.87. This was followed by Gasse *et al.* (1995) with an r^2 of 0.81, Fritz *et al.* (1993b) with 0.66, then Reed (1995) with 0.57.

The major TP studies reviewed were Whitmore (1989), Agbeti (1992), Bennion (1993), Dixit and Smol (1994), Reavie *et al.* (1995), Bennion *et al.* (1996), Reid (1997) and Tibby (2000). The results of Sincock (1997), although they were based on PO_4 observations and not TP, were also included because of the paucity of information from south east Australian aquatic systems. The best performing cross-validated WA regression was the Bennion *et al.* (1996) study with an r^2 of 0.91. This was followed by a cross-validated r^2 of 0.80 in Tibby (2000), 0.79 in Bennion (1993), 0.71 in Reid (1997), 0.55 in Dixit *et al* (1999), 0.46 in Reavie *et al.* (1995) and 0.42 in Sincock (1997). It should also be noted that the published results of Bennion *et al.* (1996) did not include taxa optima, but this information was provided upon request (Bennion *et al.*, pers. comm., 2000).

5.3 Ecological definitions

Several different terms are used to describe diatom taxa ecology. The pH classification used here was adapted from Hustedt (1937 -39) -

acidobiontic	pH < 5.5
acidophilous	pH < 7.0
circumneutral	pH values around 7.0
alkaliphilous	pH > 7.0
alkalibiontic	exclusively occurring at > 7.0

The selection of a salinity classification system was confined by the likely range of past salinity conditions in the lower Murray River (see Barnett, 1994; Cann *et al.*, 2000), and also the fact that for the majority of the fossil records the salinity was of marine influence. Accordingly, the system of Van Der Werff and Huls (1957 - 1974) was adopted (with parts per thousand converted to μS/cm), as follows -

fresh	< 300 µS/cm
fresh - brackish	< 1320 µS/cm
brackish - fresh	1320 - 2650 µS/cm
brackish	2650 - 13 235 µS/cm

Trophic status was classified according to a ranking system developed by the OECD (1982) -

Ultra-oligotrophic	< 4.0 µg/L TP
Oligotrophic	4.0 - 10.0 µg/L TP
Mesotrophic	10 - 35µg/L TP
Eutrophic	35 - 100µg/L TP
Hypertrophic	> 100 µg/L TP

5.4 The autecology of *Achnanthidium minutissimum*

A summary of ecological data from the literature is provided in table 5.2.

Table 5.2 Comparisons of optima and preferred environment data for Achnanthidium minutissimum

Study	pH	EC	TP
Bennion (1993)			66.1 µg/L
Bennion *et al.* (1996)			17 µg/L
Charles (1985)	7.2		
Denys (1991/92)	circumneutral	Fresh - brackish	ultraoligotrophic - eutrophic
Dixit and Smol (1994)	7.9		12.5 µg/L
Dixit and Smol (1995)	7.44		
Dixit *et al.* (1999)	7.8		13 µg/L
Gasse *et al.* (1995)	7.93	263 µS/cm	
Gell (1995)		1983 µS/cm	
Reavie *et al.* (1995)			13.5 µg/L
Reed (1995)		1880 µS/cm	
Reid (1997)	6.86		111 µg/L
Sincock (1997)			87.1 µg/L (PO_4)
Stevenson *et al.* (1991)	6.3		
ter Braak and van Dam *et al.*(1989)	6.8		
van Dam *et al.*. (1994)	circumneutral	< 1323 µS/cm	oligotrophic - eutraphentic
Whitmore (1989)	circumneutral		oligotrophic - eutraphentic
Vyverman (1992)	6.3 - 8.0	280 µS/cm	
This study	7.37		

Achnanthidium minutissimum is a common taxon in most of the reviewed studies, highlighting its cosmopolitan nature. Estimated pH optima for this taxon ranges from 6 3 to 7.9. Stevenson *et al.* (1991) provide the lowest optimum of 6.3, probably a result of the range of pH in the data set being limited to values below 7.25, which is much lower than for the other studies. The only Australian studies to determine pH optima were Reid (1997), who established an optimum of 6.86, based on presence in 45 lakes with a maximum occurrence of 39% and this study which generated an optimum of 7.37 based on presence in 12 lakes with a maximum abundance of 46%. Gell (1995) found the species in the most acidic of the lakes in his data set (pH range 6.3 to 10.95) with an upper pH tolerance of 9.3, which was based on presence in 18 lakes with a maximum abundance of 15%. The three qualitative studies identify *A. minutissimum* as preferring circumneutral pH conditions. As there are no optima > 8.0, it can be assumed that in the above studies *A. minutissima* was not abundant in waters with pH > 8.0.

Figure 5.1 shows the distribution of *A. minutissimum* across the pH gradient for both the lake and stream data sets in this study. On these graphs, every tick mark on the X axis represents a single sample. It should also be noted that the graphs do not have a linear scale for the environmental variables. This was intentional as the spread of samples across the environmental gradients was uneven and linear scaling would have resulted in data congestion. Within the lake data set, there is an apparent limit to the taxons presence above approximately 8.0, with samples that have high percentages of *A. minutissimum* mostly having a pH below this level. Abundances in the stream data set do not appear to be restricted by pH, with the taxon being present across the gradient.

Estimated EC optima for *A. minutissimum* indicate a preference for fresh to brackish waters with a limit to presence above an EC of approximately 2000 µS/cm. Gasse *et al.* (1995) produced the lowest EC optima for this taxon of 263 µS/cm which was based on 62 lakes and a maximum abundance of 83%, and Gell (1995) produced the highest of 1983 µS/cm (based on 18 lakes with a maximum abundance of 15%). Due to lake systems in the Northern Hemisphere being mostly dominated by a different brine type (potassium or magnesium) than south-eastern Australian systems (sodium - chloride dominated), more confidence was placed in the Australian results for this study. It should be noted, however, that the results of Reed (1995) were very similar to that found by Gell (1995), and although the lakes in this data set were located in Spain, they were of the same brine type to those in Gell (1995).

Figure 5.1 shows the distribution of *A. minutissimum* across the EC gradient for both the lake and stream data sets in this study, where it can be seen that the majority of occurrences were below an EC of 850µS/cm in the lake samples and 3010 µS/cm in the stream samples. However, it should be noted that *A. minutissimum* was present up to 16% in the lake data set in a sample with the highest EC (10 190 µS/cm).

Estimated TP optima for this taxon range from 12.5 µg/L (Dixit and Smol, 1994) to 111 µg/L (Reid, 1997). Both Whitmore (1989) and van Dam *et al.* (1994) state that the taxon is present in oligotrophic to eutrophic

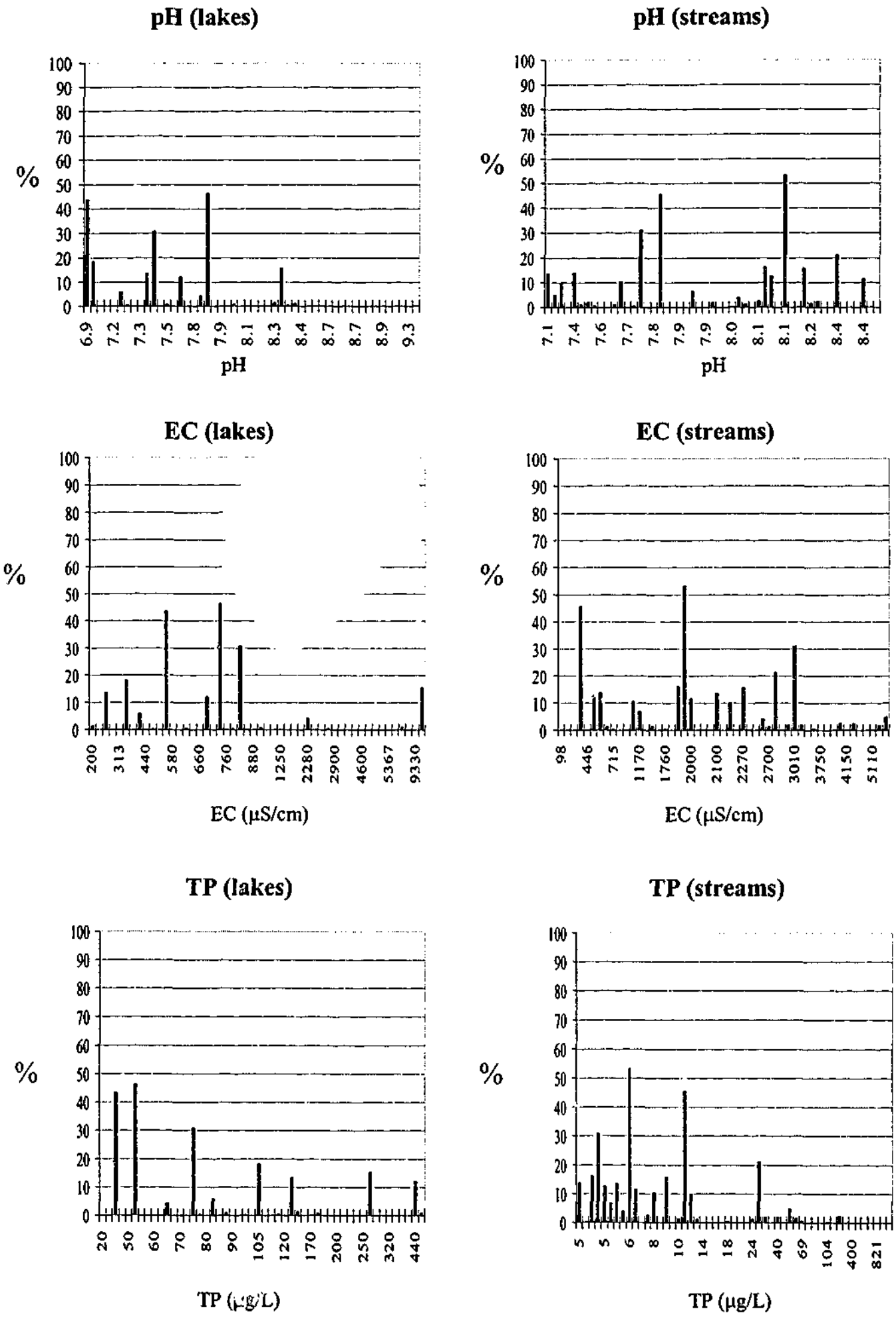


Figure 5.1: The distribution of Achnanthidium minutissimum *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

conditions, while Denys (1991/92) states a preference for oligotrophic to ultra-oligotrophic waters. Australian studies which examined TP response include Reid (1997) and Sincock (1997) who produced an optimum of 87.1 μg/L (PO_4). The optimum in Reid (1997) was based on 45 samples, while Sincock (1997) was based on 40 samples. Gell (pers. comm.) has stated that the presence of *A. minutissimum* in high nutrient waters in the Sincock (1997) data set was most probably due to very high dissolved oxygen levels, for which it has been suggested that this taxon has a preference (see Gasse, 1986). The TP optimum derived from Bennion *et al.* (1996), the best performing of the data sets, was 17 μg/L, which was derived from 140 samples with a maximum abundance of 41%.

Figure 5.1, illustrating the distribution of this taxon across the TP gradients in this study, shows a wide distribution across the lake samples (from 50 to 500 μg/L with a negligible abundance at 1000 μg/L) and a narrower distribution for the stream samples (presence limited to between 5 and 13 μg/L). It is unclear whether this pattern of distribution within the stream data set is a response to TP or simply a reflection of the skewed distribution of samples across the TP gradient, with 25 samples having a TP concentration < 20 μg/L. It is unlikely that the difference in abundance between the lake and stream data sets is due to environmental dissimilarity between the two hydrological systems, as the optimum in Sincock (1997) was derived from stream samples also.

Additional information

Denys (1991/92) found that *A. minutissimum* lives primarily attached to a substratum such as vegetation or rocks. He also states that it can be a dry, aerophilous taxon and that it has a low potential for preservation in sediments. Figure 5.1 shows that *A. minutissimum* is common in both lake and stream environments. In studies of south east Australian streams, Sonneman (pers. comm.) has found that there appears to be two groups of this taxon, one that is abundant in polluted waters and one that flourishes in more oligotrophic conditions. Sonneman also suspects that *A. minutissimum* is less subject to grazing pressure as it is a taxon that is able to exploit micro-refuges.

Summary

Based on the above discussion, it can be assumed that *A. minutissimum* prefers pH conditions below 8.0 in lakes, although in stream environments the taxon appears to be able to tolerate pH conditions above this level. There were no EC optima derived for this taxon above 2000 μS/cm and it can therefore be assumed that the taxon has a preference for fresh to fresh - brackish waters. In terms of TP preferences, it seems that *A. minutissimum* has a wide tolerance to TP, with optima ranging from oligotrophic to eutrophic conditions. The ecological preferences that will be adopted for *A. minutissimum* for the reconstruction of palaeolimnology in the lower Murray River are -

pH - generally circumneutral, predominantly occurring in waters < 7.5 pH

EC - fresh to fresh – brackish

TP - wide tolerance up to 100 µg/L, but limited in distribution for TP concentrations greater than this

5.5 The autecology of *Aulacoseira granulata*

The relationship between *A. granulata* and pH is not clear with estimated optima ranging from 6.8 (Reid, 1997) to 8.24 (this study) (table 5.3). However, the range of the pH gradient in the Reid (1997) data set (5.38 to 8.8, with a mean of 6.7) is more limited that the range of pH in this studies data set, which may influence this results. Stevenson *et al.* (1991) establish a pH optimum of 7.1, although this is based on presence in 3 lakes only. Gasse *et al.* (1995) publish an optimum of 7.66 which is based on abundances in 60 lakes. Both van Dam *et al.* (1994) and Denys (1991/92) describe *A. granulata* as being alkaliphilous. Considering these results, it appears that this taxon is alkaliphilous, that is, it occurs mainly at pH levels > 7.0.

Table 5.3:Comparisons of optima and preferred environment data for Aulacoseira granulata

Study	pH	EC	TP
Agbeti (1992)			22.44 µg/L
Bennion *et al.* (1996)			232 µg/L
Denys (1991/92)	alkaliphilous	fresh - brackish	eutrophic
Dixit *et al.* (1999)	7.8		29 µg/L
Fritz (1993)		1475 µS/cm	
Gasse (1995)	7.66	363 µS/cm	
Gell (1995)		2738 µS/cm	
Reavie *et al.* (1995)			36 µg/L
Reid (1997)	6.8		300 µg/L
Sincock (1997)			75 µg/L (PO_4)
Stevenson *et al.* (1991)	7.1		
Tibby (2000)			42 µg/L
van Dam *et al.* (1994)	alkaliphilous	< 1300 µS/cm	eutraphentic
Vyverman (1992)	6.7 - 9.5	382 µS/cm	
This study	8.24		

When examining the distribution of *A. granulata* across the pH range in this data set (figure 5.2), it can be seen that the sites where *A. granulata* is present all have a pH > 7.2, with the greatest abundances occurring at pH > 7.7. Results from this study are based on presence in 18 lakes with a maximum abundance of 68%.

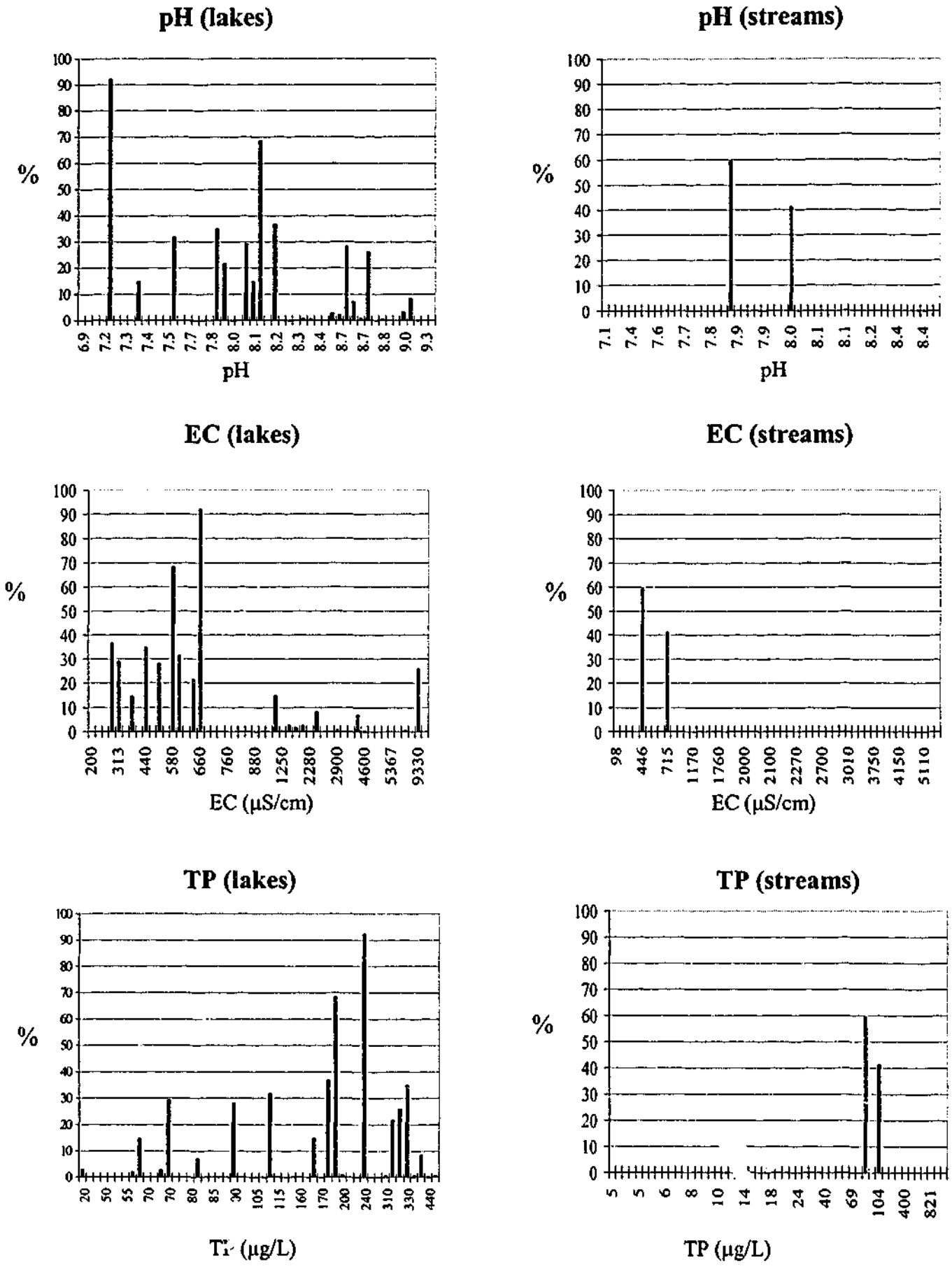


Figure 5.2: The distribution of Aulacoseira granulata *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

The EC requirements of *A. granulata* appear to be clearly defined, with all studies showing a preference for fresh to fresh - brackish conditions, with the highest optimum being 2738 µS/cm (Gell, 1995), based on presence in 7 samples with a maximum abundance of 5%. Gasse *et al.* (1995) produce the lowest optimum of 363 µS/cm which was based on presence in 65 samples with a maximum abundance of 90%. Both Denys (1991/92) and van Dam *et al.* (1994) describe a preference for fresh - brackish waters. The plot illustrating the distribution of *A. granulata* across the EC gradient in the lake set (figure 5.2) emphasises this relationship with the majority of samples with high abundances of this taxon having an EC < 700 µS/cm.

The optima derived for TP generally show that *A. granulata* has a preference for high nutrient conditions. Agbeti (1991) and Reavie *et al.* (1995) are exceptions to this, with both producing optima < 40 µg/L. The TP ranges of these studies, however, are both narrow, particularly Agbeti (1991) which has a range between 1 and 63 µg/L. The three south east Australian studies all produce a TP optima > 40 µg/L, with Reid (1997) establishing an optimum of 300 µg/L (based on 34 samples). Figure 5.2, illustrating the distribution of *A. granulata* across the TP gradient, shows a distinct preference for samples that have a TP concentration > 100 µg/L.

Additional information

Surveying by the Murray - Darling Basin Commission has shown that *A. granulata* is the dominant alga within the Murray River phytoplankton, a river which commonly has TP concentrations exceeding 80 µg/L. Hötzel and Croome (1996) state that the population dynamics of this species in the Murray River are primarily related to flow conditions, turbidity and silica concentrations. The authors also found that this taxon was most abundant in highly turbulent waters (upper and mid reaches) and was less abundant in the lower Murray where flow is impeded by the river regulation.

The study by Hötzel and Croome (1996) should be treated with caution, however, as *A. granulata* is easily confused with the similar taxonomic forms of *Aulacoseira ambigua* and *Aulacoseira italica*. As part of this study, sixteen plankton samples were taken from different sections of the lower Murray and it was found that all three of these species occurred concurrently in the water column, with *A. ambigua* often being the dominant taxon. *Aulacoseira ambigua* and *Aulacoseira italica* were not reported as being present in Murray River plankton in the Hötzel and Croome (1996) study.

Denys (1991/92) states that *Aulacoseira granulata* is euplanktonic, living almost exclusively in the water column. He also states that the taxon has moderate preservation potential. Figure 5.2 shows that *A. granulata* is principally a lake diatom within this study's data set, occurring in only 2 stream sites (both Murray River), which as discussed in chapter 3, are ecologically more similar to a lacustrine environment than a riverine one.

Summary

Aulacoseira granulata is an alkaliphilous taxon that prefers waters with pH > 7.0. It can be found in high abundances in pH up to 9.0. *Aulacoseira granulata* is a freshwater taxon although it can tolerate slightly brackish conditions. *Aulacoseira granulata* is generally a eutrophic taxon, commonly occurring in waters with TP concentrations > 100 µg/L. The ecological limits that will be adopted for palaeolimnological reconstructions in this study are-

pH – predominantly occurring in pH conditions > 7.0
EC - fresh
TP - a eutrophic to hypertrophic taxon, commonly occurring in waters with TP concentrations > 100 µg /L.

5.6 The autecology of *Aulacoseira subarctica* forma *subborealis*

Aulacoseira subarctica forma *subborealis* is a diatom that is prevalent in many of the inland waters of south east Australia. It has been found in high abundances by Tibby (2000) in deep mesotrophic to eutrophic Victorian lakes that are particularly turbid, and also by Reid (1997) in billabongs located along the Murray River downstream of Hume Reservoir. The taxon found in this study is of the same morphological type as that found by Tibby (2000) and Reid (1997). A similar taxon, *Aulacoseira alpigena*, is described in Krammer & Lange-Bertalot (1991a), but differs slightly in morphology (finer striae and punctae density) than the taxon identified in the south east Australian studies. It should also be noted that *A. subarctica* forma *subborealis* is also taxonomically similar to *Aulacoseira distans*, which has been identified in the Murray Darling Basin Commissions phytoplankton surveys from the Murray River (Hotzel and Croome, 1996). These taxonomic inconsistencies indicate that the results from overseas studies are inappropriate for comparison with Australian results. Therefore, only Australian studies are reviewed.

Table 5.4: Comparisons of optima and preferred environment data for Aulacoseira subarctica *forma* subborealis

Study	pH	EC	TP
Reid (1997)	6.8		318 µg/L
Tibby (2000)			32 µg/L
This study	7.9		

The pH optimum derived from this study is higher than that derived by Reid (1997), but the mean of the pH gradient in Reid (1997) study was much lower also. The taxon was present in 37 lakes in Reid (1997) with a maximum abundance of 36%. Figure 5.3, showing the distribution of *A. subarctica* forma *subborealis* is not overly revealing because of the small number of sites. However, both the lake and river data sets show distribution centred around a pH of 8.0.

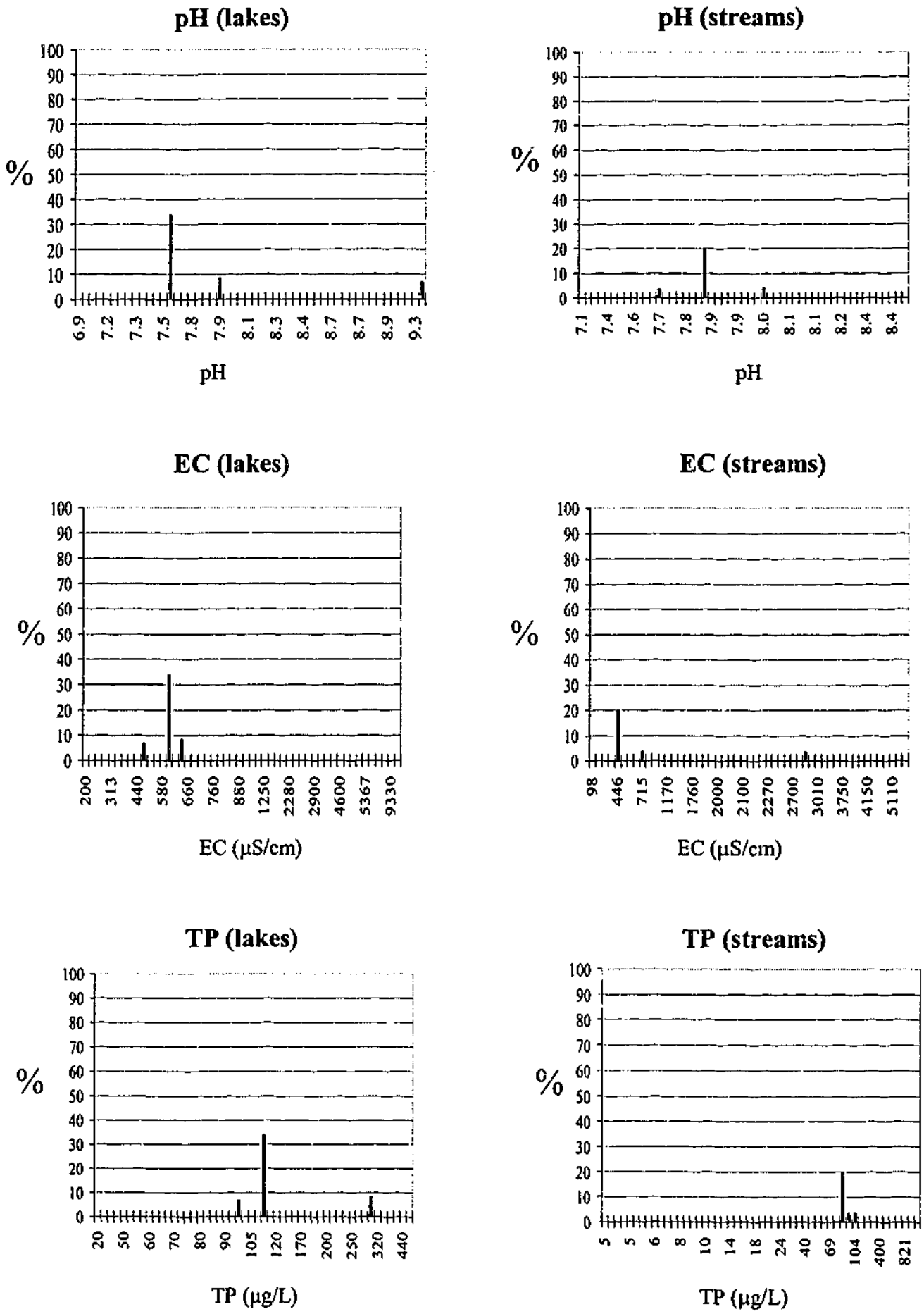


Figure 5.3: The distribution of Aulacoseira subarctica *forma* subborealis *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

Although Tibby (2000) did not produce EC optima, he did find that this taxon only occured in sites with EC < 500 µScm (pers. comm.), which concur with the distribution found in this study (see figure 5.3).

Reid (1997) produced a TP optimum for this taxon of 318 µg/L, while Tibby (2000) produced a much lower optimum of 32 µg/L . The results of Tibby (2000) and Reid (1997) are both comprehensive with results based on a large number of samples (both > 30), however they produce vastly different TP optima. With a lack of other data, the only conclusion is that *A. subarctica* forma *subborealis* is abundant in mesotrophic to hypertrophic waters.

Summary

Aulacoseira subarctica forma *subborealis* is a planktonic taxon that is found in both deep lakes (Tibby, 2000) and shallow lakes and streams (Reid 1997, and results from this study). It is a freshwater taxon, with an apprently lower EC tolerance than *A. granulata*. *Aulacoseira subarctica* forma *subborealis* is a mesotrophic to hypertrophic taxon. The ecological preferences for *A. subarctica* forma *subborealis* that will be adopted for the palaeolimnological reconstruction of the lower Murray River is -

pH - generally circumneutral
EC - fresh water, most commonly < 500 µS/cm
TP - mesotrophic to hypertrophic

5.7 The autecology of *Cocconeis placentula*

Table 5.5 outlines the different environmental optima and ecological information presented in the reviewed studies.

Cocconeis placentula appears to be more abundant in alkaline waters, with five WA regression studies (Dixit and Smol 1994, Gasse *et al.* 1995, Reid 1997, Dixit *et al.* 1999 and this study) all producing pH optima > 7.0. Van Dam *et al.* (1994) and Denys (1991/92) also describe the taxon as alkaliphilous. The exceptions to this are Stevenson *et al.* (1991), a study where the range of pH didn't exceed 7.25, and Whitmore (1989), which was primarily a nutrient study, with both studies concluding that *C. placentula* prefers circumneutral conditions. Figure 5.4, illustrating the abundance of this taxon across the pH gradient in studied lakes, shows a wide distribution from 7.4 to approximately 9.0. The abundances of *C. placentula* throughout this section of the gradient are relatively evenly spread, however, with no obvious optimum.

Table 5.5: Optima and preferred environment data for Cocconeis placentula

Study	pH	EC	TP
Agbeti (1992)			20.88 µg/L
Bennion (1993)			89.9 µg/L
Bennion *et al.* (1996)			232 µg/L
Denys (1991/92)	alkaliphilous	fresh - brackish	eutrophic
Dixit and Smol (1994)	8.1		22.1 µg/L
Dixit *et al.* (1999)	8.0		28 µg/L
Fritz *et al.*. (1995)		3400 µS/cm	
Gasse *et al.*. (1995)	8.31	467 µS/cm	
Gell (1995)		17 680 µS/cm	
Reavie *et al.*. (1995)			18.3 µg/L
Reid (1997)	7.17		119 µg/L
Sincock (1997)			319 µg/L (PO_4)
Stevenson *et al.*. (1991)	6.7		
Tibby (2000)			201 µg/L
van Dam *et al.* (1994)	alkaliphilous	fresh - brackish	eutrophic
Whitmore (1989)	circumneutral		eutrophic
This study	8.17		

The reviewed studies indicate that *C. placentula* is tolerant of EC conditions ranging from fresh to brackish. The lowest EC optima of 467 µS/cm was derived by Gasse *et al.* (1995) and was based on presence in 74 samples with a maximum abundance of 99%. The highest EC optima of 17 680 µS/cm was produced by Gell (1995), based on occurrences in 118 samples, also with a maximum abundance of 99%. The vast difference between the optima of Gasse *et al.* (1995) and Gell (1995) cannot be easily explained, although two possibilities can be suggested. Firstly, the range of EC in Gell (1995) was much higher than that in Gasse *et al.* (1995) (195 000 µS/cm compared to 99 000 µS/cm). Secondly, *C. placentula* may be sensitive to brine type, and therefore has different ecological preferences in south east Australia to that found in Africa because of the different brine types that are predominant in these regions. Figure 5.4 shows a wide distribution of samples with *C. placentula* throughout the EC gradient in this study, based on presence in > 40 samples (lakes and streams combined) with a maximum abundance of 26%. The above results appear to suggest that *C. placentula* is highly tolerant of a wide range of salinity conditions.

The relationship between this taxon and TP is reported to be quite varied in the reviewed studies. Optima range from 18.3 µg/L in Reavie *et al.* (1995) to 319 µg/L in Sincock (1997). Again, however, the optima are influenced by the range of TP in the respective data sets, with Reavie *et al.* (1995) having a range of 5 – 138 µg/L and Sincock (1997) a range of 13 - 1142 µg/L. Reid (1997) produced an optimum of 119 µg/L which was based on presence in 39 billabongs with a maximum abundance of 19%, and Tibby (2000)

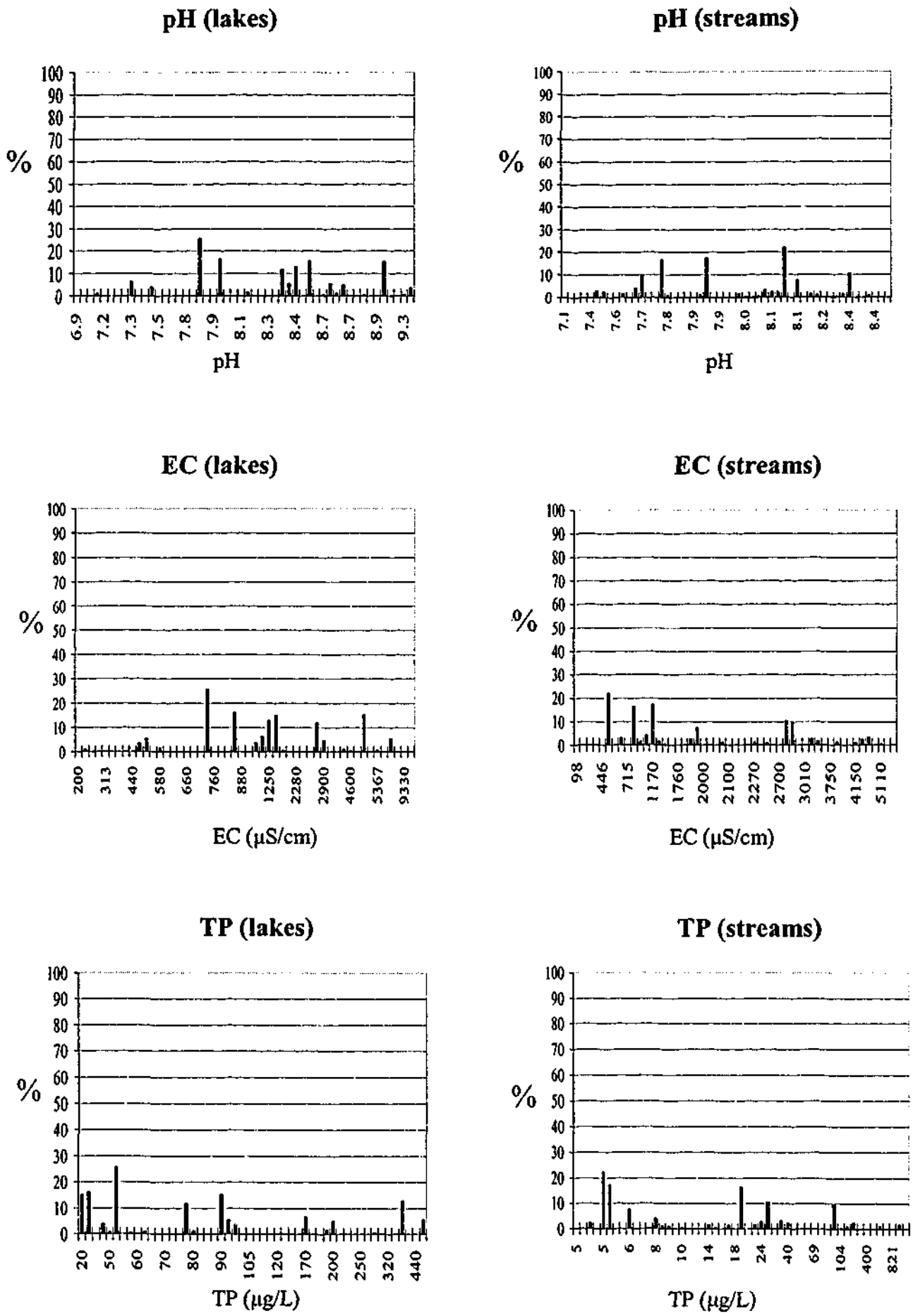


Figure 5.4: The distribution of Cocconeis placentula *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

produced an optimum of 201 µg/L based on presence in 16 lakes with a maximum abundance of 7%. With three south east Australian studies producing such wide ranging results, it can only be concluded that *C. placentula* is abundant in oligotrophic to hypertrophic waters.

Additional information

Cocconeis placentula is an epontic taxon, living primarily attached to aquatic plants, but also the benthos and rocks. Denys (1991/92) states that it can be common in ephemeral waters and that it has moderate preservation potential. Figure 5.4 shows that the taxon is equally present in both lake and stream environments.

Summary

Cocconeis placentula is primarily an alkaliphilous taxon with a wide abundance between pH of 7.0 and 9.0. *Cocconeis placentula* does not appear to have a narrow EC tolerance, with the taxon being abundant in waters ranging from 100 µS/cm to 20 000 µS/cm. A similar relationship exists between TP and this taxon, with a wide ranging distribution in oligotrophic to hypertrophic waters.

pH – alkaliphilous
EC – fresh to brackish
TP – oligotrophic to hypertrophic

5.8 The autecology of *Cyclotella meneghiniana*

The optima and preferences for the environmental variables pH, EC and TP determined from reviewed modern diatom studies are shown in table 5.6.

The pH preference of *Cyclotella meneghiniana* varies widely from Vyverman's (1992) acidophilous values to the alkaliphilous values of Denys (1991/92), van Dam *et al.* (1994) and Whitmore (1989) and the alkalibiontic values of Gasse *et al.* (1995) and this study. The other south east Australian study of Reid (1997), derived an optimum of 6.8, based on abundances in 31 samples. The distribution of this taxon across the pH gradient in the lake study (figure 5.5), shows an obvious preference for sites with high pH (above 8.0). Abundances of *C. meneghiniana* below a pH of 8.0 are low, with the greatest abundances occurring around a pH of 8.4.

All of the reviewed studies that examine the relationship between *C. meneghiniana* and EC (with the exception of Vyverman, 1992) indicate that *C. meneghiniana* has a preference for fresh - brackish to brackish conditions with estimated EC optima ranging from 1300 to 8640 µS/cm. The south east Australian study of Gell (1995) developed an optima of 4983 µS/cm based on 60 occurrences with a maximum abundance of

35%. Figure 5.5 supports this conclusion, showing that the majority of lakes with *C. meneghiniana* abundant have an EC above 1000 µS/cm.

Table 5.6: Comparison of optima and preferred environment conditions for Cyclotella meneghiniana

Study	pH	EC	TP
Bennion *et al.* (1996)			712 µg/L
Denys (1991/92)	alkaliphilous	brackish - fresh	eutrophic
Dixit *et al.* (1999)	8.3		66 µg/L
Fritz *et al.* (1993)		4970 µS/cm	
Gasse (1995)	8.85	6025 µS/cm	
Gell (1995)		4983 µS/cm	
Reavie *et al.* (1995)			10.7 µg/L
Reed (1995)		8640 µS/cm	
Reid (1997)	6.8		197 µg/L
Sincock (1997)			212 µg/L (PO_4)
Stevenson *et al.* (1991)	6.9		
Tibby (2000)			45 µg/L
van Dam *et al.* (1994)	alkaliphilous	1300 - 2650 µS/cm	eutraphentic
Vyverman (1992)	5.5 - 6.5	92 µS/cm	
Whitmore (1989)	alkaliphilous		eutrophic
This study	8.57		

The relationship between this taxon and TP appears to indicate a preference for high nutrient waters. Results from three Australian studies are varied, with Reid (1997) producing an optimum of 197 µg/L, Sincock (1997) an optimum of 212 µg/L (PO_4), and Tibby an optimum of 45 µg/L. Although the optimum of Tibby (2000), which is based on presence in 25 lakes with a maximum abundance of 40%, is still indicative of eutrophic conditions, it is considerably lower than the results of the other south east Australian studies. Bennion *et al.* (1996) produce a very high optimum of 712 µg/L, based on 75 samples, with a maximum abundance of 30%. The descriptive studies of Denys (1991/92), van Dam *et al.* (1994), and Whitmore (1989) all classify the taxon as being eutrophic. The distribution across the TP gradient in this study, shown in figure 5.5, shows that while the greatest abundance of samples with *C. meneghiniana* have TP concentrations above 100 µg/L, there are samples with substantial percentages of this taxon in the eutrophic range also.

Additional information

Cyclotella meneghiniana is predominantly a planktonic taxon although it can survive on the benthos. Denys (1991/92) states that it is a tychoplanktonic taxon, meaning that although it readily occurs in the plankton it is derived primarily from other habitats, and it often metabolises and reproduces in the plankton to the same

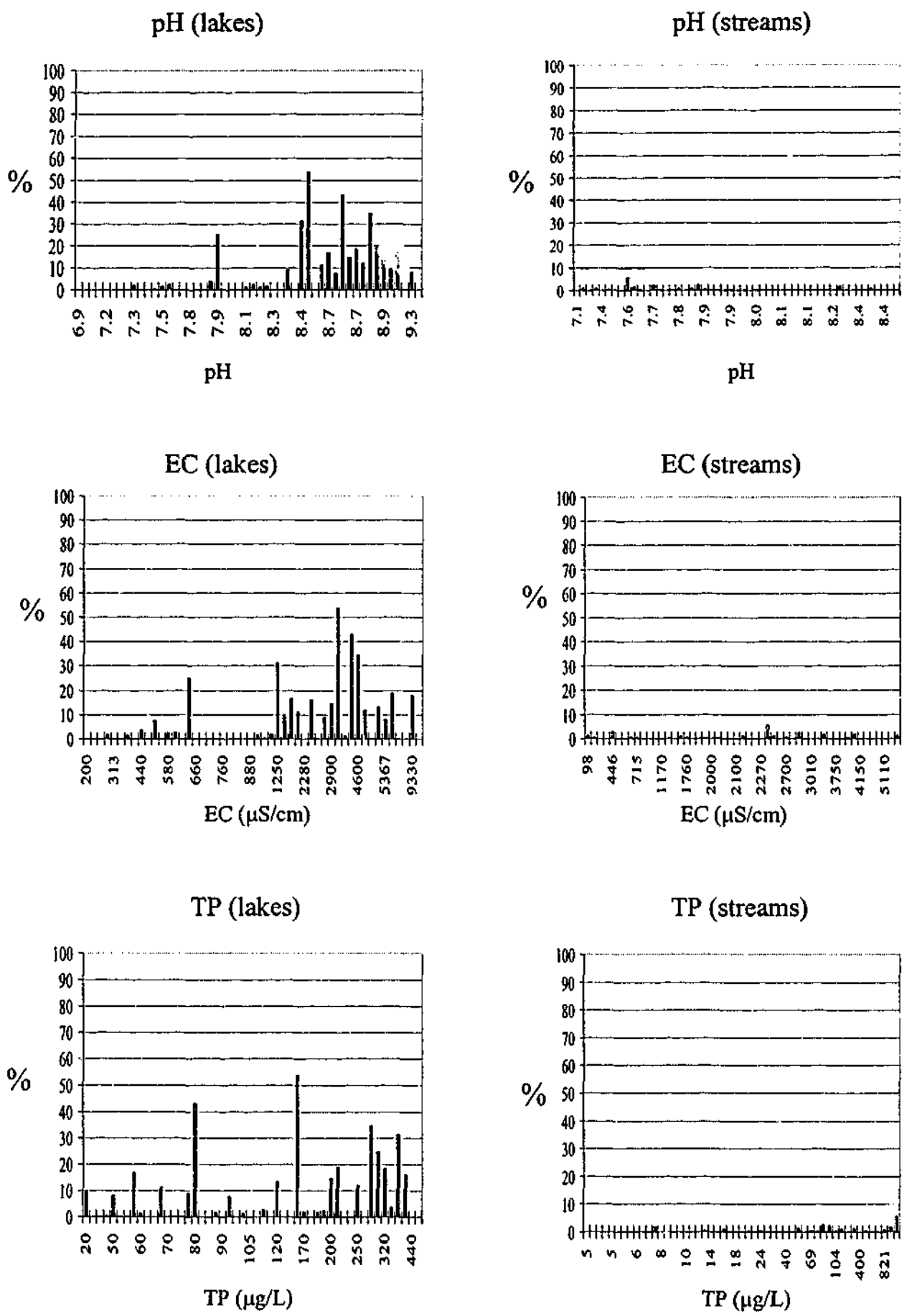


Figure 5.5: The distribution of Cyclotella meneghiniana *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

degree that it did in its original habitat. Denys (1991/92) further states that *C. meneghiniana* commonly occurs in ephemeral waters and that it has moderate preservation potential. Cholnoky (1968) regards this taxon as being a reliable indicator of oxygen rich conditions in weakly to moderately alkaline waters. In a study on diatoms from sewage works in South Africa, Shroeman (1972) found that *C. meneghiniana* was a facultative nitrogen heterotroph that had a pH optimum > 8.0. Sonneman (pers. comm.) has found this taxon mainly in lowland south east Australian phytoplankton populations, occasionally dropping out to survive in the epibenthos. Figure 5.5 shows that in this study, *C. meneghiniana* was much more abundant in lacustrine environments than riverine environments.

Summary

Cyclotella meneghiniana appears to be most abundant in waters with pH above 8.0, although it can also be present in circumneutral waters. Reid (1997) developed a pH optimum that was much lower than this but the pH range for that study was skewed towards more acidic samples. A clear relationship was shown between *C. meneghiniana* and EC, with the majority of studies concluding that it prefers brackish - fresh to brackish waters. While the relationship between this taxon and TP is not as clear as that for pH or EC, the Australian studies all agree that it prefers high nutrient waters, but can also be present in oligotrophic waters. The ecological preferences that will be used for *C. meneghiniana* for palaeolimnological reconstructions in chapters 6 and 7 are -

pH – circumneutral to alkaliphilous, most abundant in waters with pH > 8.0
EC - prefers fresh to brackish waters
TP - eutrophic to hypertrophic

5.9 The autecology of *Epithemia adnata*

Comparisons between published environment optima and other ecological information for *Epithemia adnata* are provided in table 5.7.

Table 5.7: Optima and preferred environment data for Epithemia adnata

Study	pH	EC	TP
Denys (1991/92)	alkalibiontic - alkaliphilous	fresh - brackish	eutrophic - mesotrophic
Fritz *et al.* (1993b)		3190 µS/cm	
Gasse *et al.* (1995)	8.27	263 µS/cm	
Gell (1995)		1525 µS/cm	
Reid (1997)	6.8		148 µg/L
van Dam *et al.* (1994)	alkalibiontic	fresh - brackish	mesotrophic - eutrophic
Vyverman (1992)	6.3 - 8.0	280 µS/cm	
This study	8.43		

There is a paucity of information on the ecology of *E. adnata*, with it cited in only seven of the reviewed studies. However, despite this lack of data, there is still enough information contained in the studies to conclude that *E. adnata* has a preference for alkaline waters. Both Gasse *et al.* (1995) and this study produce optima above 8.0 (8.27 and 8.43 respectively), with the results of this study based on 11 samples with a maximum abundance of 14.5%, and Gasse *et al.* (1995) based on presence in 25 lakes with a maximum abundance of 77%. Denys (1991/92) and van Dam *et al.* (1995) also describe this taxon as being alkalibiontic to alkaliphilous. Reid (1997) is the exception to this with an optimum of 6.8 but results are based on presence in only 3 lakes with a maximum abundance of 4.8%. Figure 5.6, illustrating the distribution of *E. adnata* across the pH gradient in this study, shows a clear increase in samples with this taxon present above a pH of 8.0 (for the lake data set).

Epithemia adnata appears to have a wide tolerance to EC in the range of fresh to brackish - fresh conditions. Gasse *et al.* (1995) establishes the lowest optimum of 263 μS/cm, while Fritz *et al.* (1993b) produce the highest optimum of 3190 μS/cm. Gell (1995) produces an optima of 1525 μS/cm, based on presence in 9 samples with a maximum abundance of 21%. Figure 5.6 shows that the samples with the highest abundances of *E. adnata* in this study (albeit relatively small values) occur in samples with EC concentrations > 1000 μS/cm.

The only study to quantitatively examine the relationship between *E. adnata* and TP is the south east Australian study of Reid (1997) who produced an optimum of 148 μg/L derived from three samples. The two descriptive studies of Denys (1991/92) and van Dam *et al.* (1994) conclude that the taxon is mesotrophic - eutrophic. Figure 5.6 shows that while the greatest abundance of this taxon occurs in eutrophic to hypertrophic samples, there is a sample with 12% of this taxon present that has a TP concentration of 9 μg/L of TP. Due to a lack of more comprehensive data, the relationship between *E. adnata* and TP is inconclusive.

Additional information

Denys (1991/92) has found *E. adnata* surviving in ephemeral waters, and states that it has a high preservation potential. This taxon is an adnate form, meaning that it primarily lives attached to plants and rocks.

Summary

Data on *E. adnata* are sparse and therefore the conclusions made here must be used with caution. From the data available, it seems that this taxon is predominantly alkalibiontic with a preference for waters with pH levels > 8.0. This taxon is abundant in waters with EC ranging from fresh to brackish - fresh conditions. The relationship between *E. adnata* and TP is not clear, although the taxon was present in samples ranging from oligotrophic to hypertrophic conditions. The ecological preferences that will be adopted for palaeolimnological reconstruction are -

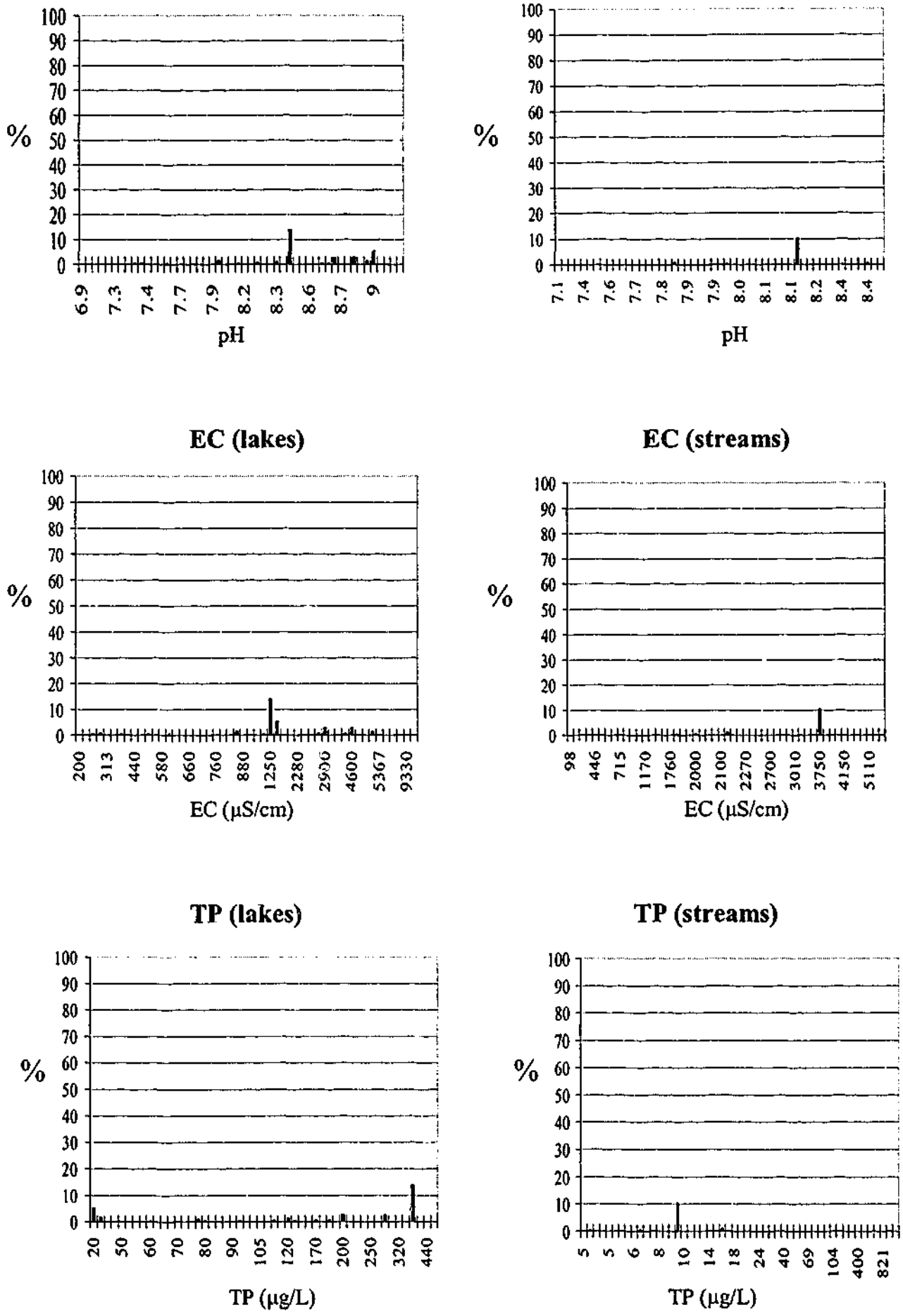


Figure 5.6: The distribution of Epithemia adnata *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

pH - alkalibiontic, most abundant in pH > than 8.0

EC – fresh to brackish - fresh conditions

TP - oligotrophic to hypertrophic

5.10 The autecology of *Gomphonema parvulum*

Comparisons between published environment optima and other ecological information for *Gomphonema parvulum* are provided in table 5.8.

Table 5.8: Optima and preferred environment data for Gomphonema parvulum

Study	pH	EC	TP
Bennion (1993)			138.4 µg/L
Bennion *et al.* (1996)			129 µg/L
Denys (1991/92)	alkaliphilous to circumneutral	fresh - brackish	eutrophic to mesotrophic
Fritz *et al.* (1995)		3660 µS/cm	
Gasse *et al.* (195)	7.63	263 µS/cm	
Gell (1995)		2030 µS/cm	
Reavie *et al.* (1995)			19.2 µg/L
Reid (1997)	6.72		188 µg/L
Sincock (1997)			123 µg/L (PO_4)
Stevenson *et al.* (1991)	6.2		
Tibby (2000)			74 µg/L
van Dam *et al.* (1994)	circumneutral	< 1323 µS/cm	eutrophic
Vyverman (1992)	6.5 - 7.0	86 µS/cm	
Whitmore (1989)	acidophilous - circumneutral		oligotrophic
This study	7.65		

The above studies all appear to indicate that *G. parvulum* is a circumneutral to alkaliphilous taxon, with a pH optimum between 6.5 and 7.5. Gasse *et al.* (1995) and the results from this study produce the highest optima of 7.63 (based on 90 samples with a maximum abundance of 59%) and 7.65 (based on 12 samples with a maximum abundance of 15%) respectively, while Stevenson *et al.* (1991) produce the lowest optima of 6.2. While not estimating pH optima, Gell (1995) found that this taxon tolerates pH levels up to 9.4, however the maximum concentration of *G. parvulum* in his study was only 4.8%. Figure 5.7, illustrating the distribution of *G. parvulum* across the pH gradient in this study, shows that the majority of samples containing this taxon all have a pH < 8.3. Samples in the stream data set show a wider distribution throughout the gradient but with the taxon reaching its highest concentrations in samples with pH levels around 7.5.

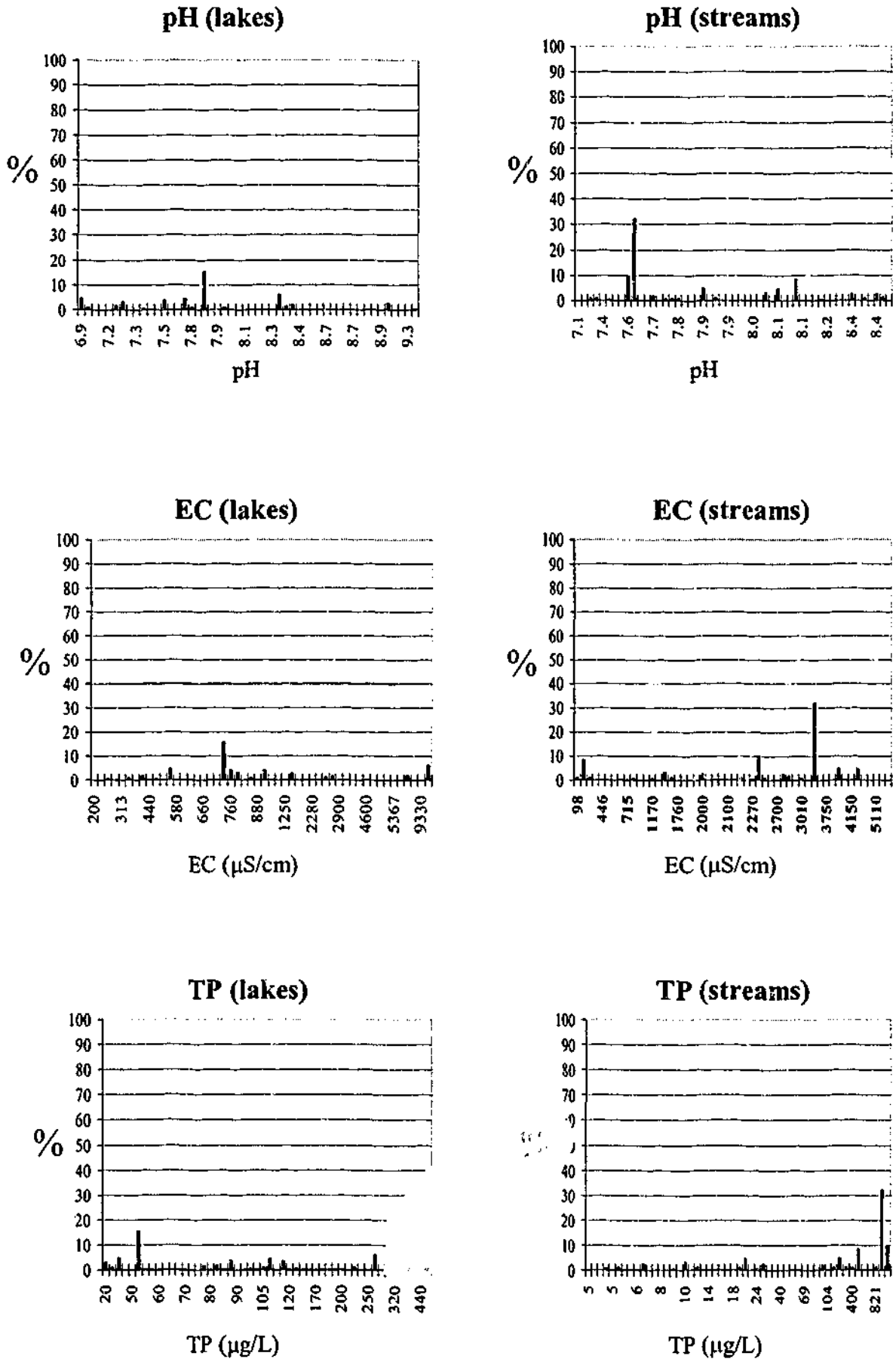


Figure 5.7: The distribution of Gomphonema parvulum *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

The reviewed studies also appear to indicate that *G. parvulum* prefers fresh to fresh - brackish waters, with an optimum < 1000 µS/cm The exceptions to this are Fritz *et al.* (1993b) who produced an optimum of 3660 µS/cm and Gell (1995) who produced an optimum of 2030 µS/cm. The results of Fritz *et al.* (1993b) and Gell (1995) are probably not as comprehensive of those of Gasse *et al.* (1995) because of low abundances of this taxon within their data sets (2.47% and 4.8% respectively), compared to the 59% found in the Gasse *et al.* (1995) data set. Because of the lack of more conclusive data on the ecology of this taxon in south east Australia, the most judicious conclusion would be that this taxon occurs predominantly in waters with an EC < 1000 µS/cm, but is able to tolerate conditions of at least 3400 µS/cm.

The relationship between TP and *G. parvulum* is more straightforward with the majority of studies indicating a preference for high TP conditio . The results from Reid (1997) were based on taxon presence in 49 samples with a maximum abundance of 23%. Sincock (1997), another south east Australian study, derived an optimum of 123 µg/L which was based on 40 samples but with a maximum abundance of only 6%. *Gomphonema parvulum* was also found to be the dominant taxon in a study of the algae inhabiting the Werribee sewage ef uent ponds which treat Melbourne's wastewater (Tibby, pers. comm.). The descriptive studies conclude differently, with van Dam *et al.* (1994) stating a preference for eutrophic waters while Whitmore (1989) states a preference for oligotrophic conditions.

Figure 5.7 shows an interesting distribution of this taxon across the TP gradient in both the lake and stream data sets. Although the taxon is fairly evenly spread across the TP gradient in the lake data set, the highest abundance occurs in a sample with a TP concentration of 54 µg/L, while in the stream data set, there is an obvious concentration of samples at the high end of the TP gradient, particularly at concentrations > 800 µg/L. Both results still show a preference for waters with TP concentrations higher than 50 µg/L, which would place the taxon within the eutrophic to hypertrophic classification.

Additional information

Gomphonema parvulum is a stalked taxon, meaning that it most commonly lives attached to plants. Denys (1991/92) states that it can also survive in moist subaerial environments. Sonneman (pers. comm.) has found that this taxon is generally more abundant in urban polluted waters and that it prefers epiphytic to epilithic communities. Sonneman (pers. comm.) also states that that there are several taxonomically similar varieties of *G. parvulum* in south east Australian waters and that one of them, *G. parvulum* var. *parvulius* is highly indicative of oligotrophic waters and *G. parvulum* var. *lagenula* is highly indicative of phosphorus rich waters.

Summary

It can be concluded that *G. parvulum* is a circumneutral to alkaliphilous taxon, with a preferred pH range between 6.5 and 7.5. The above discussion showed that this taxon occurs predominantly in waters with an EC < 1000 µS/cm, although it can be present in south east Australian waters at EC at least 3500 µS/cm.

Gomphonema parvulum is abundant in eutrophic to hypertrophic waters with highest abundances between 50 and 150 µg/L.

pH – circumneutral to alkaliphilous

EC – prefers EC < 1000 µS/cm, although can be present in higher salinities.

TP – eutrophic to hypertrophic

5.11 The autecology of *Melosira varians*

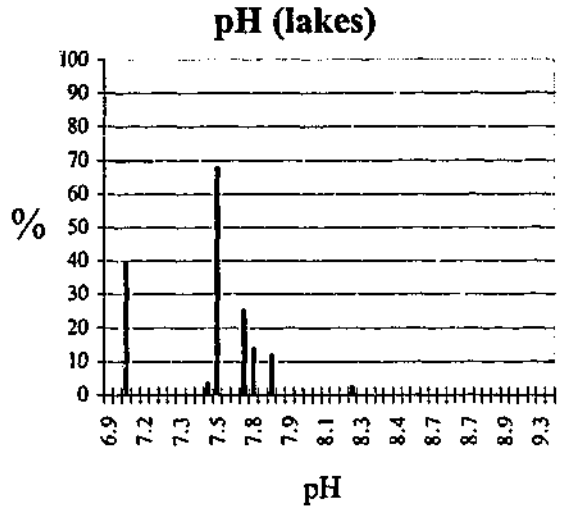
Comparisons between published environment optima and other ecological information for *Melosira varians* are provided in table 5.9.

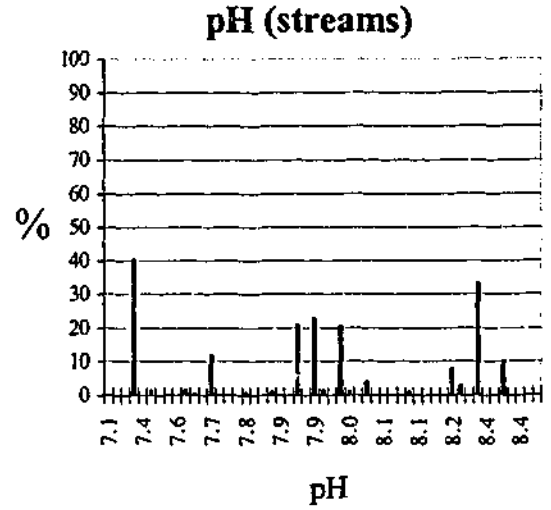
Table 5.9: Comparisons of optima and preferred environment data for Melosira varians

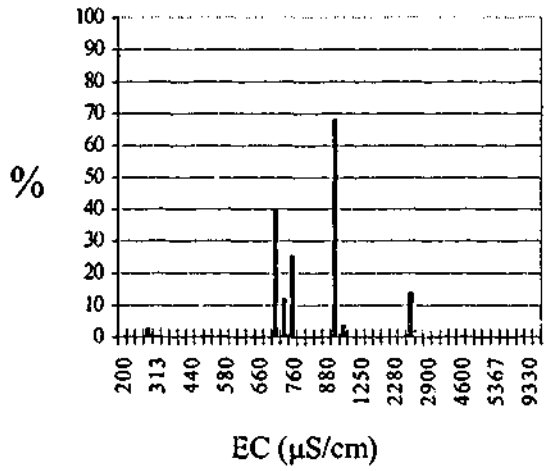
Study	pH	EC	TP
Bennion *et al.* (1996)			80 µg/L
Denys (1991/92)	alkaliphilous	fresh - brackish	eutrophic
Gasse *et al.* (1995)	7.44	166 µS/cm	
Reid (1997)	6.69		213 µg/L
Sincock (1997)			72 µg/L (PO_4)
van Dam *et al.* (1994)	alkaliphilous	fresh - brackish	eutrophic
Whitmore (1989)	alkaliphilous		eutrophic
This study	7.48		

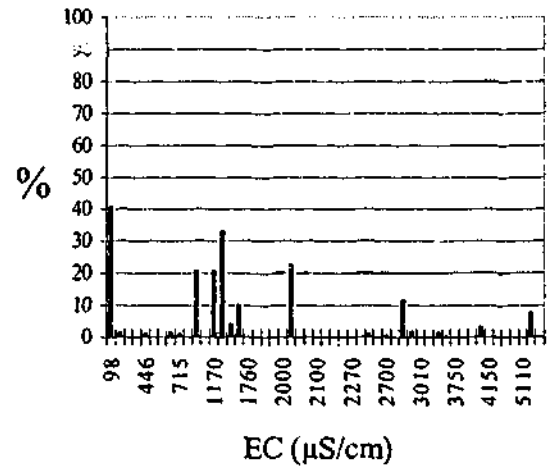
With the exception of Reid (1997), the above studies indicate that *M. varians* has a preference for slightly alkaline waters (pH between 7.0 and 8.0). The optima derived in this study of 7.48 (based on presence in 7 samples with a maximum abundance of 68%) is identical to the optima derived by Gasse *et al.* (1995), based on 9 samples and a maximum abundance of 6%. Figure 5.8, illustrating the distribution of samples across the pH gradient in this study, shows that within the lake data set the taxon has a clear preference for waters with pH less than 7.8. However, the stream data set shows a wider distribution with samples containing up to 40% of this taxon having a pH > 8.0.

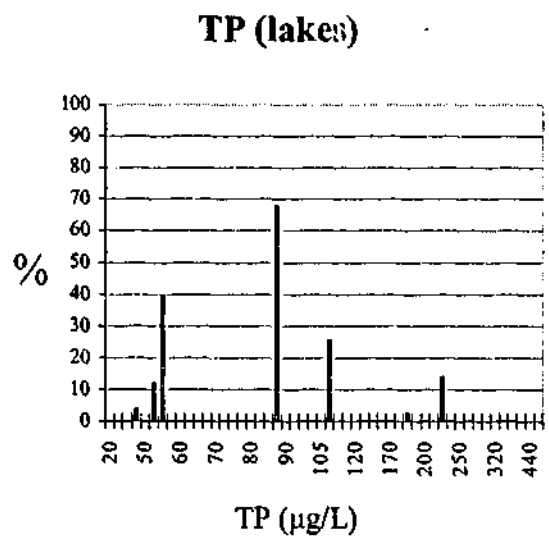
Melosira varians is a fresh - brackish taxon with all of the reviewed studies stating a preference and / or optimum for EC < 1000 µS/cm. Figure 5.8 shows a concentration of samples with *M. varians* abundant between 700 and 900 µS/cm in the lake data set. Again, this taxon's distribution in the stream data set is more widespread, however, the samples with the highest abundances occur at the lower end of the EC gradient, generally < 2100 µS/cm.











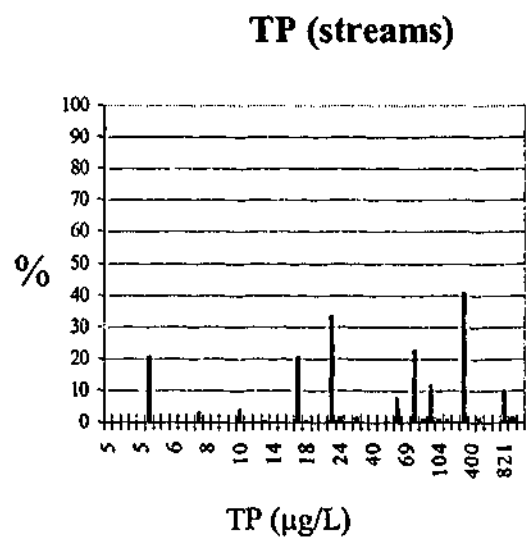


Figure 5.8: The distribution of Melosira varians *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

The relationship between *M. varians* and TP is also quite clear, with all of the reviewed studies indicating a preference for eutrophic waters. The optimum derived by Reid (1997) is much higher than that of the other south east Australian study (Sincock 1997) which is probably an artifact of the comparatively high range of TP (up to 3100 µg/L) in Reid (1997). Figure 5.8 shows a wide distribution of samples across the TP gradient in both the lake and stream data sets, although samples with the highest abundances of this taxon occur at TP concentrations indicative of eutrophic conditions.

Additional information

Denys (1991/92) states that this taxon is tychoplanktonic, that is, it is facultative planktonic, surviving equally well in both the plankton and on the benthos. He also states that the taxon can inhabit ephemeral waters and that is has high preservation potential. Sonneman (pers. comm.) has commonly found this taxon in slow moving urban streams, particularly those that are ephemeral.

Summary

It can generally be stated that *M. varians* has a preference for alkaliphilous waters. The reviewed studies show a clear relationship between *M. varians* and fresh - brackish waters, being most abundant in waters < 1000 µS/cm. *Melosira varians* has a preference for eutrophic to hypertrophic waters.

pH - alkaliphilous

EC - most abundant < 1000 µS/cm.

TP - eutrophic to hypertrophic

5.12 The autecology of *Navicula cryptocephala*

The relationship between pH and *N. cryptocephala* is not well defined with the reviewed studies (table 5.10) stating preferences ranging from acidophilous to alkalibiontic. Figure 5.9, illustrating the distribution of samples containing *N. cryptocephala* across the pH gradient in the lake data set, shows that the greatest abundance of samples occurs in samples with pH between 7.2 and 7.3.

The results from those reviewed studies which examine the relationship between this taxon and EC are also somewhat conflicting, with estimated optima ranging between 407 µS/cm (Gasse *et al.* 1995) to 4500 µS/cm (Gell 1995). For comparison, the optimum from Gasse *et al.* (1995) was derived from 47 samples with a maximum abundance of 14.9%, while the optimum of Gell (1995) was based on 38 samples with a maximum abundance of 50%. The two descriptive studies of Denys (1991/92) and van Dam *et al.* (1994) state that this taxon prefers fresh - brackish conditions (< 1300 µS/cm). Figure 5.9 does not clarify the relationship any further, with the greatest taxon distribution spread between 250 µS/cm and 2400 µS/cm.

Table 5.10: Optima and preferred environment data for Navicula cryptocephala

Study	pH	EC	TP
Bennion (1993)			130 µg/L
Bennion *et al.* (1996)			170 µg/L
Denys (1991/92)	alkaliphilous	fresh - brackish	eutrophic - oligotrophic
Dixit and Smol (1994)	8.2		27.5 µg/L
Dixit *et al.* (1999)	8.1		23 µg/L
Fritz *et al.* (1995)		3190 µS/cm	
Gasse *et al.* (1995)	7.62	407 µS/cm	
Gell (1995)		4500 µS/cm	
Reavie *et al.* (1995)			9.5 µg/L
Reid (1997)	6.85		156 µg/L
Sincock (1997)			159 µg/L (PO_4)
Stevenson *et al.* (1991)	6.6		
van Dam *et al.* (1994)	circumneutral	fresh - brackish	oligotrophic - eutrophic
Whitmore (1989)			ultra - oligotrophic
This study	7.39		

The relationship between *N. cryptocephala* and TP is also unclear, with the reviewed studies developing optimum that range between 9.5 µg/L (Reavie *et al.*, 1995) and 170 µg/L (Bennion *et al*, 1996).

Additional information

Denys (1991/92) states that this taxon prefers benthic environs such as bottom muds and rocks, although it can tolerate moist subaerial conditions as well. It is rated as having moderate preservation potential.

Summary

Although the relationship is not strongly defined, it appears that *N. cryptocephala* has a preference for alkaliphilous waters, with a preferred range between 7.0 and 7.5. Its relationship with EC cannot be confidently defined as the results are inconclusive, although the taxon is present in fresh - brackish to brackish conditions. The taxon also appears to live equally well in oligotrophic to hypertrophic conditions.

pH – preference for conditions between 7.0 and 7.5.

EC – fresh – brackish to brackish

TP - oligourophic to hypertrophic waters

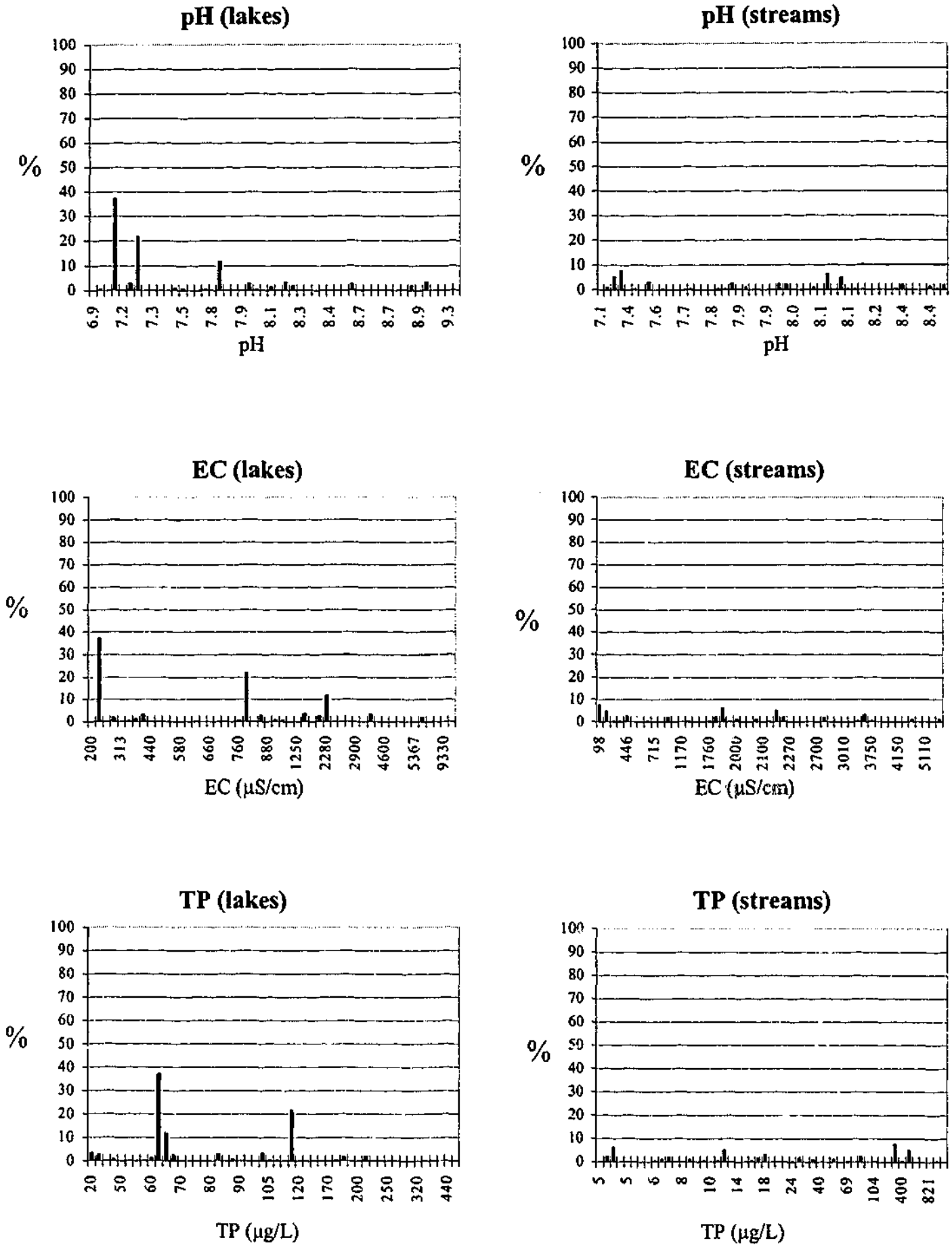


Figure 5.9: The distribution of Navicula cryptocephala *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

5.13 The autecology of *Navicula veneta*

Comparisons between published environment optima and other ecological information for *Navicula veneta* are provided in table 5.11.

Table 5.11: Comparisons of optima and preferred environment data for Navicula veneta

Study	pH	EC	TP
Denys (1991/92)	alkaliphilous	brackish - fresh	eutrophic
Fritz *et al.* (1993b)		7264 µS/cm	
Gell (1995)		3527 µS/cm	
Reed (1995)		9350 µS/cm	
Reid (1997)	7.02		100 µg/L
Sincock (1997)			630 µg/L (PO_4)
van Dam *et al.* (1994)	alkaliphilous	brackish - fresh	oligotrophic - eutrophic
This study	7.61		

There is consensus between the reviewed studies that *N. veneta* is an alkaliphilous taxon, preferring pH conditions between 7 .0 and 8.0. The optima derived from this study was based on occurrences in 19 samples with a maximum abundance of 15%. Figure 5.10, illustrating the distribution of this taxon across the pH gradient in this study, suggests an upper pH limit of around 8.4 within the lake data set. There is no apparent limit for the stream data set (although the pH gradient only extends to 8.4).

All five studies that examine the relationship between *N. veneta* and EC show a preference for brackish - fresh to brackish waters. The distribution across the EC gradients in both the lake and stream data sets (shown in figure 5.10) concurs with these results, with taxon presence relatively even in fresh to brackish waters.

The reviewed studies appear to indicate that *N. veneta* can survive in a wide range of nutrient concentrations. The very high optimum generated by Sincock (1997) cannot be confidently considered as it is based on presence in one sample with a maximum abundance of 0.6%. Other optima include 100 µg/L (Reid, 1997), based on 23 samples with a maximum abundance of 4%. The two descriptive studies (van Dam *et al.*, 1994; Denys 1991/92) state that the taxon has a preference for oligotrophic to eutrophic conditions. Figure 5.10 does not clarify the relationship any further with samples containing this taxon being generally evenly spread across the TP gradient.

Additional information

Navicula veneta is a benthic taxon, preferring to inhabit surface substrates and rocks. It can survive in moist subaerial conditions and has moderate preservation potential (Denys, 1991).

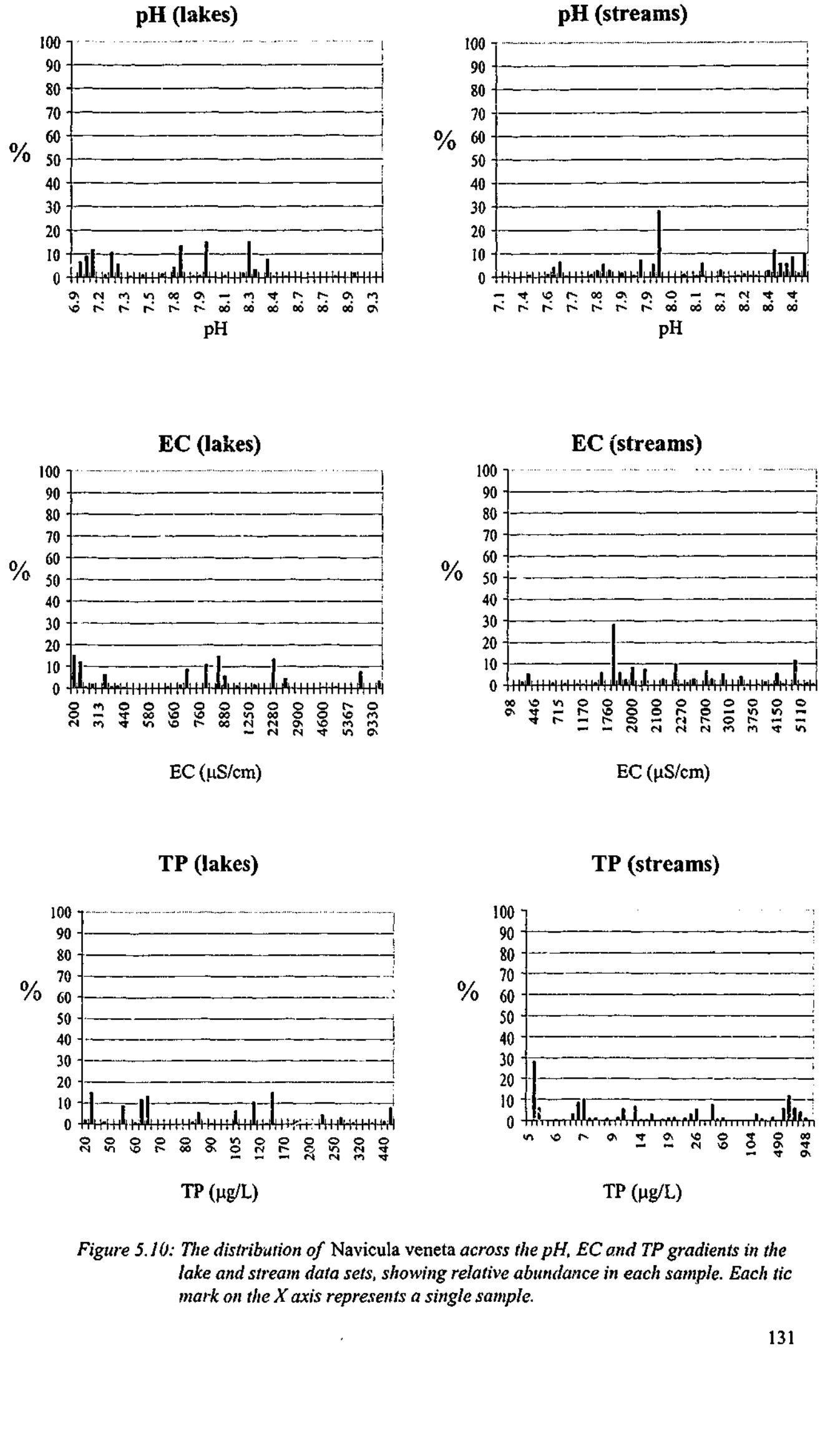


Figure 5.10: The distribution of Navicula veneta *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

Summary

Navicula veneta is an alkaliphilous taxon, with a preference for pH conditions between 7.0 and 8.0. From the information available it appears that *N. veneta* has a wide tolerance to EC and is present in fresh to brackish - fresh waters. *N. veneta* also has a wide ranging tolerance to TP, including oligotrophic to hypertrophic conditions.

pH - alkaliphilous
EC - fresh to brackish - fresh waters
TP - oligotrophic - hypertrophic conditions.

5.14 The autecology of *Nitzschia palea*

The reviewed studies (table 5.12) seem to indicate a pH preference between circumneutral and alkaliphilous. The exception to this is the results from this study which produce an alkalibiontic optima of 8.44. However, this result is based on presence in 14 lake samples with a maximum abundance of 8% and therefore not sufficiently thorough to be conclusive. Figure 5.11 emphasises the lack of strength in the relationship, illustrating that the majority of samples containing *N. palea* in both the lake and river data sets are fairly evenly spread across the pH gradient. Thus, with the reviewed results not contradicted, it would appear that *N. palea* has a pH preference for alkaliphilous to circumneutral waters.

***Table 5.12:** Optima and preferred environment data for* Nitzschia palea

Study	pH	EC	TP
Bennion (1993)			129 µg/L
Bennion *et al.* (1999)			199 µg/L
Denys (1991/92)	alkaliphilous - circumneutral	fresh - brackish	eutrophic
Dixit and Smol (1994)	7.6		10.3 µg/L
Dixit *et al.* (1999)	7.8		81 µg/L
Fritz et al. (1993b)		4250 µS/cm	
Gasse *et al.* (1995)	7.81	891 µS/cm	
Gell (1995)		4980 µS/cm	
Reavie *et al.* (1995)			17.9 µg/L
Reed (1995)		8540 µS/cm	
Reid (1997)	6.99		128 µg/L
Sincock (1997)			796 µg/L (PO_4)
Stevenson *et al.* (1991)	6.2		
van Dam *et al.* (1994)	circumneutral	fresh - brackish	hypertrophic
Vyverman (1992)	6.7 - 9.5	fresh	
Whitmore (1989)	circumneutral		oligotrophic - eutrophic
This study	8.44		

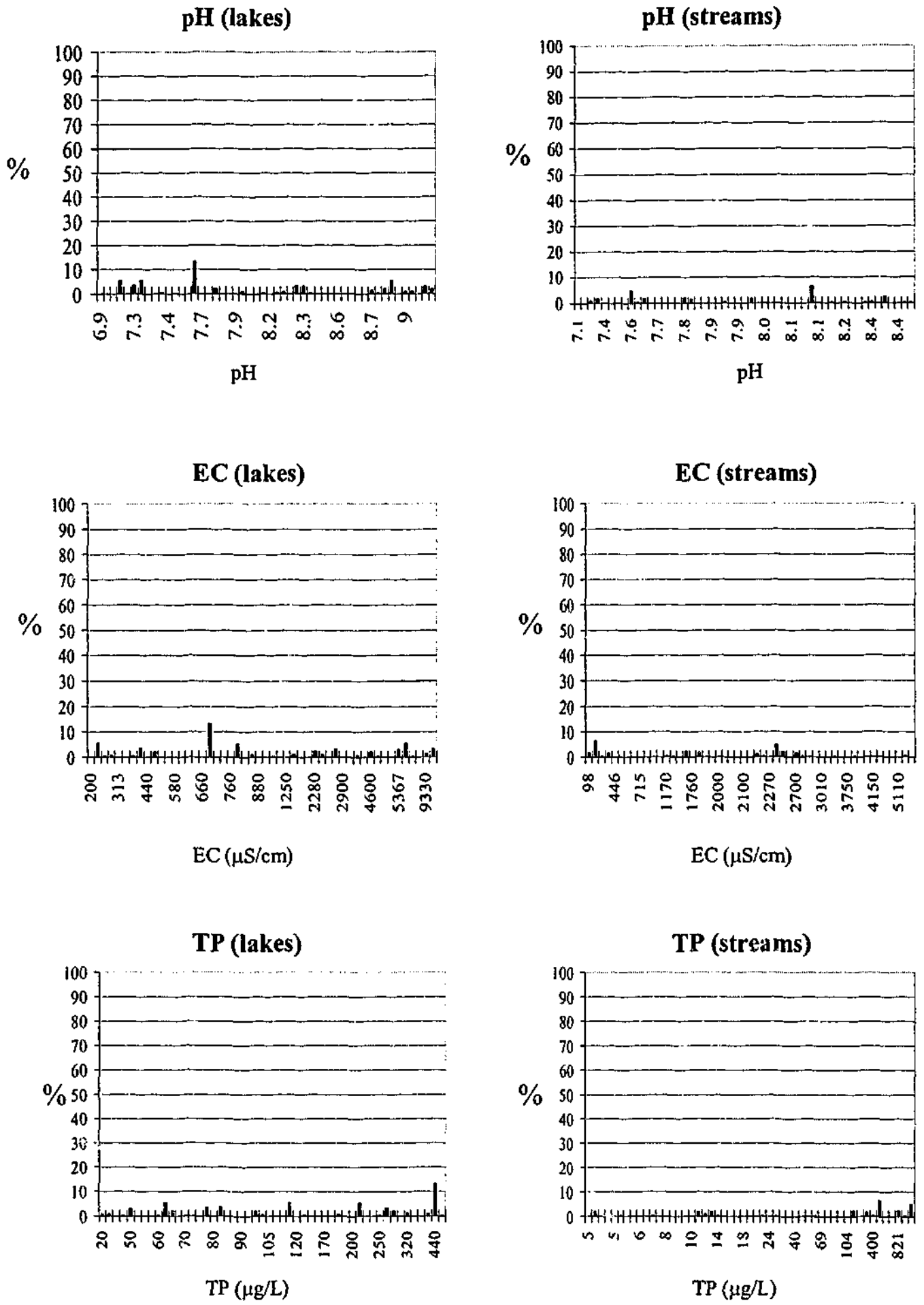


Figure 5.15: The distribution of Nitzschia palea *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

The relationship between *N. palea* and EC is not well defined, with established optima ranging between 891 µS/cm (Gasse *et al.*, 1995) to 8540 µS/cm (Reed, 1995). Gell's (1995) south east Australian study produced an optimum of 4980 µS/cm based on presence in 64 lakes with a maximum abundance of 50%. All of the descriptive studies state that this taxon prefers fresh to fresh - brackish waters. The low optimum produced by Gasse *et al.* (1995) was based on presence in 120 lakes with a maximum abundance of 48%. Therefore the results from Gell (1995) and Gasse *et al.* (1995) can both be regarded as robust and comprehensively derived, which presents a problem as they are so different. In terms of palaeolimnological reconstruction, more confidence could be placed in the results of Gell (1995) as they derived from south east Australian systems with similar brine types to those lakes that are the focus of the palaeoreconstruction in this study, as opposed to the different brine types found in African lakes in Gasse *et al.* (1995). In the absence of other data however, it would seem appropriate to assign a wide EC tolerance to this taxon, rather than limiting it to the results of one study only.

With the exception of Dixit and Smol (1994), Reavie *et al.* (1995) and Dixit *et al.* (1999), the reviewed studies indicate that *N. palea* is competitive in hypertrophic conditions. Bennion (1993) and Reid (1997) produce similar TP optima of 129 µg/L. and 128 µg/L respectively. Sincock (1997) produces a very high PO_4 optimum of nearly 800 µg/L, which was based on 19 samples with a maximum abundance of 4%. Sonneman (pers. comm.) has also found the taxon to be common in eutrophic streams. It would thus appear that *N. palea* is a hypertrophic taxon that is able to be competitively advantaged in very high concentrations of TP, although the results of Dixit and Smol (1994), Reavie *et al.* (1995) and Dixit *et al.* (1999) demonstrate that it can also be found in oligotrophic environments.

Additional information

Denys (1991/92) states that this taxon can live both on the benthos and on rocks and plants. He also states that it is common in moist subaerial environments and that it has moderate preservation potential.

Summary

There is a general consensus between the reviewed studies that *N. palea* prefers circumneutral to alkaliphilous waters. This taxon is abundant in fresh to brackish - fresh waters, although the relationship is not well established. *N. palea* appears to be tolerant of hypertrophic conditions but can also live in oligotrophic conditions.

pH - circumneutral - alkaliphilous

EC - fresh to brackish - fresh waters

TP - oligotrophic - hypertrophic, but most abundant in hypertrophic waters

5.15 The autecology of *Planothidium delicatulum*

A summary of ecological data from the literature is provided in table 5.13. Only six studies have published optima / ecological preferences for *P. delicatulum*. The only study to develop a pH optimum is this study, with an optima of 7.84 based on 26 lakes with a maximum abundance of 38%. Denys (1991/92) and van Dam *et al.* (1994) describe this taxon as being alkaliphilous and alkalibiontic respectively. Figure 5.12, illustrates a broad distribution of this taxon across the pH gradients in the lake and stream data sets, although highest abundances in both environments occur at pH < 8.0.

Table 5.13:Comparisons of optima and preferred environment data for P. delicatulum

Study	pH	EC	TP
Denys (1991/92)	alkaliphilous	brackish	eutrophic
Gell (1995)		8081 µS/cm	
Reavie *et al.* (1995)			19.5 µg/L
Reed (1995)		9010 µS/cm	
Sincock (1997)			72 µg/L (PO_4)
van Dam *et al.* (1994)	alkalibiontic	2640 - 13 000 µS/cm	
This study	7.84		

Information on the response to EC is more extensive with all reviewed studies showing a tolerance to brackish waters. The south east Australian study of Gell (1995), developed an optimum of 8081 µS/cm, which was based on 65 samples. Gell (1995) notes that this taxon also displays a widely scattered response to EC. Figure 5.12, illustrating taxon abundances across EC gradients in this study, shows a relatively uniform distribution throughout the EC range, with high abundances in fresh waters, suggesting that this taxon can survive equally well in fresh and brackish waters.

Total phosphorus optima include 19.5 µg/L in Reavie *et al.* (1995) and 72 µg/L in Sincock (1997). Denys (1991/92) describes the nutrient preferences of this taxon as eutrophic. The distribution of *P. delicatulum* across the TP gradient in this study can be seen in Figure 5.12. The results are similar to that for pH and EC, that is, the taxon displays a wide tolerance across the gradient. However, the taxon does occur less frequently at the higher end of the gradient (i.e.: TP concentrations > 200 µg/L), although this may be an artefact of the skewed nature of the gradient (see figure 5.12).

Additional information

Denys (1991/92) states that *P. delicatulum* lives primarily attached to various substrates. It has also been found in waters that are periodically dry and has a moderate preservation potential in sediments. Figure 5.2 shows that this taxon is equally common in both lake and stream environments.

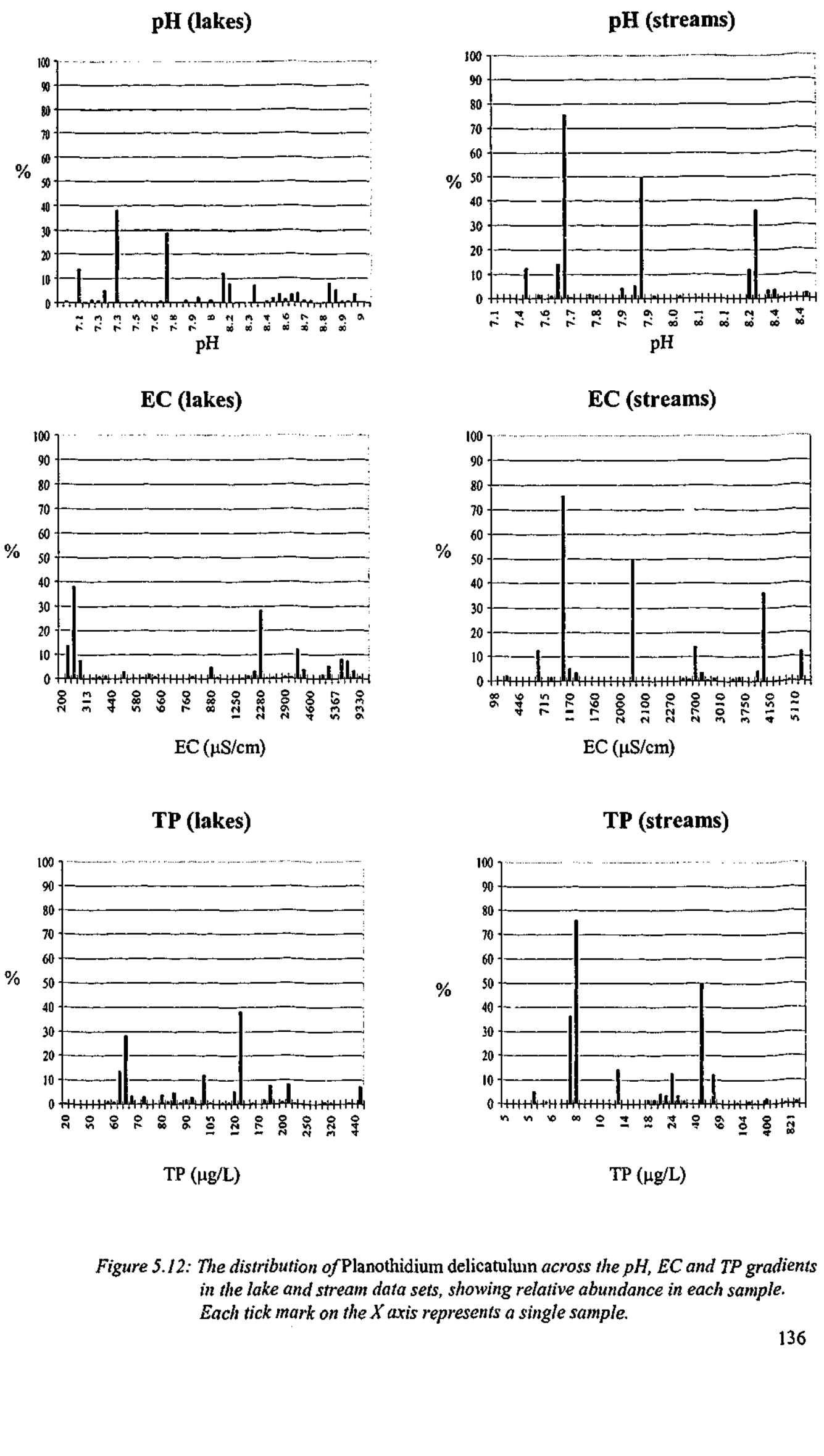


Figure 5.12: The distribution of Planothidium delicatulum *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tick mark on the X axis represents a single sample.*

Summary

There is a lack of studies that have researched the relationship between *P. delicatulum* and pH, although the limited data available seem to indicate a preference for alkaliphilous waters. The relationship between EC and this taxon has been more thoroughly researched with the majority of studies stating a preference for brackish - fresh to brackish waters. Although *P. delicatulum* was present in fresh waters in this study, both Gell (1995) and Reed (1995) developed optima > 8000 µS/cm. It would therefore seem that this taxon has a wide EC tolerance. *Planothidium delicatulum* also displays a wide tolerance to TP ranging from 0 – 200 µg/L, with figure 5.12 showing a general absence of this taxon at TP concentrations > 200 µg/L in either data set.

pH – an alkaliphilous taxon

EC – a brackish – fresh to brackish taxon, most commonly occuring in a range between 2000 µS/cm and 8000 µS/cm.

TP – oligotrophic to hypertrophic

5.16 The autecology of *Pseudostaurosira brevistriata*

Information on preferred ecological environments for *P. brevistriata* was derived from sixteen studies (table 5.14).

Table 5.14: Optima and preferred environment data for P. brevistriata

Study	pH	EC	TP
Agbeti (1992)			34.48 µg/L
Bennion (1993)			95.7 µg/L
Bennion *et al.* (1996)			180 µg/L
Denys (1991/92)	alkaliphilous	fresh - brackish	eutrophic to mesotrophic
Dixit and Smol (1994)	8.0		12.6 µg/L
Dixit *et al.* (1999)	7.7		13 µg/L
Fritz *et al.* (1995)		2530 µS/cm	
Gasse *et al.* (1995)	7.82	575 µS/cm	
Gell (1995)		2802 µS/cm	
Reavie *et al.* (1995)			12.2 µg/L
Reed (1995)		1550 µS/cm	
Reid (1997)	6.69		165 µg/L
Stevenson *et al.* (1991)	6.5		
Tibby (2000)			19 µg/L
van Dam *et al.* (1994)	alkaliphilous	< 1323 µS/cm	oligotrophic - eutrophic
Whitmore (1989)	alkaliphilous		eutrophic - hypertrophic
This study	8.3		

Eight of these studies examined the relationship between taxon abundance and pH, with six studies showing a preference for alkaline environments. Although not producing optima, Gell (1995) found this taxon in lakes with a pH > 8.4 but < 9.5. Figure 5.13, showing the distribution of *P. brevistriata* across the pH gradient in this study, adds strength to these findings by illustrating that the majority of samples with this taxon present have a pH > 7.8.

All of the reviewed studies seem to indicate that *P. brevistriata* has a preference for fresh - brackish through to brackish - fresh waters, with EC optima ranging from 575 µS/cm (Gasse *et al.*, 1995) which was derived from occurrences in 57 lakes with a maximum abundance of 70%, to 2802 µS/cm (Gell, 1995) which was based on presence in 12 samples with a maximum abundance of 11%. Figure 5.13, illustrates a wide ranging distribution of *P. brevistriata* across the EC gradient, with substantial abundances of this taxon occurring in both fresh (< 500 µS/cm) and also brackish waters (> 2650 µS/cm) . It therefore seems that, generally, *P. brevistriata* is tolerant of fresh to brackish conditions.

The relationship between TP and this taxon is somewhat confused by the optima derived by Dixit and Smol (1994), Dixit *et al.* (1999) and Reavie *et al.* (1995) of 12.6 µg/L, 13.0 µg/L and 12.2 µg/L respectively, which are substantially lower than the optima from the remaining studies. Once again, this is probably a result of the low range of TP within their data sets. The south east Australian study (Reid 1997) produces an optimum that indicates that this taxon has a preference for eutrophic to hypertrophic waters, a result which was also found by Bennion (1993) and Bennion *et al.* (1996). Both Denys (1991/92) and van Dam *et al.* (1994) describe *P. brevistriata* as having a wide ranging tolerance to TP, while Whitmore (1989) found that the taxon prefers eutrophic to hypertrophic waters.

Examination of Figure 5.13 complicates the relationship between TP and *P. brevistriata*, as it did with EC. Although most of the samples with this taxon present have TP concentrations around 100 µg/L. there is a sample containing nearly 50% of this taxon that has a TP concentration of 30 µg/L. This result, combined with the findings from the reviewed studies that produced the comparatively low optimas, suggests that this taxon can survive equally well in low or high nutrient conditions.

Additional information

Denys (1991/92) found that this taxon can live both on bottom muds (benthic) and attached to the littoral environment (plant, rock etc.). However, in a survey of Lake Alexandrina and Murray River plankton , *P. brevistriata* was found living abundantly within the water column, suggesting that in this environment it can be a facultative planktonic taxon. Both Lake Alexandrina and the lower Murray River are characterised by high turbidity levels, suggesting that perhaps this taxon can out compete other more common planktonic taxa (*Cyclotella* spp. and *Aulacoseira* spp.) which may prefer clearer waters. Figure 5.13 shows that the taxon is present in both lake and stream environments, although it is more common in the lake data set. Denys (1991/92) states that this taxon has moderate preservation potential.

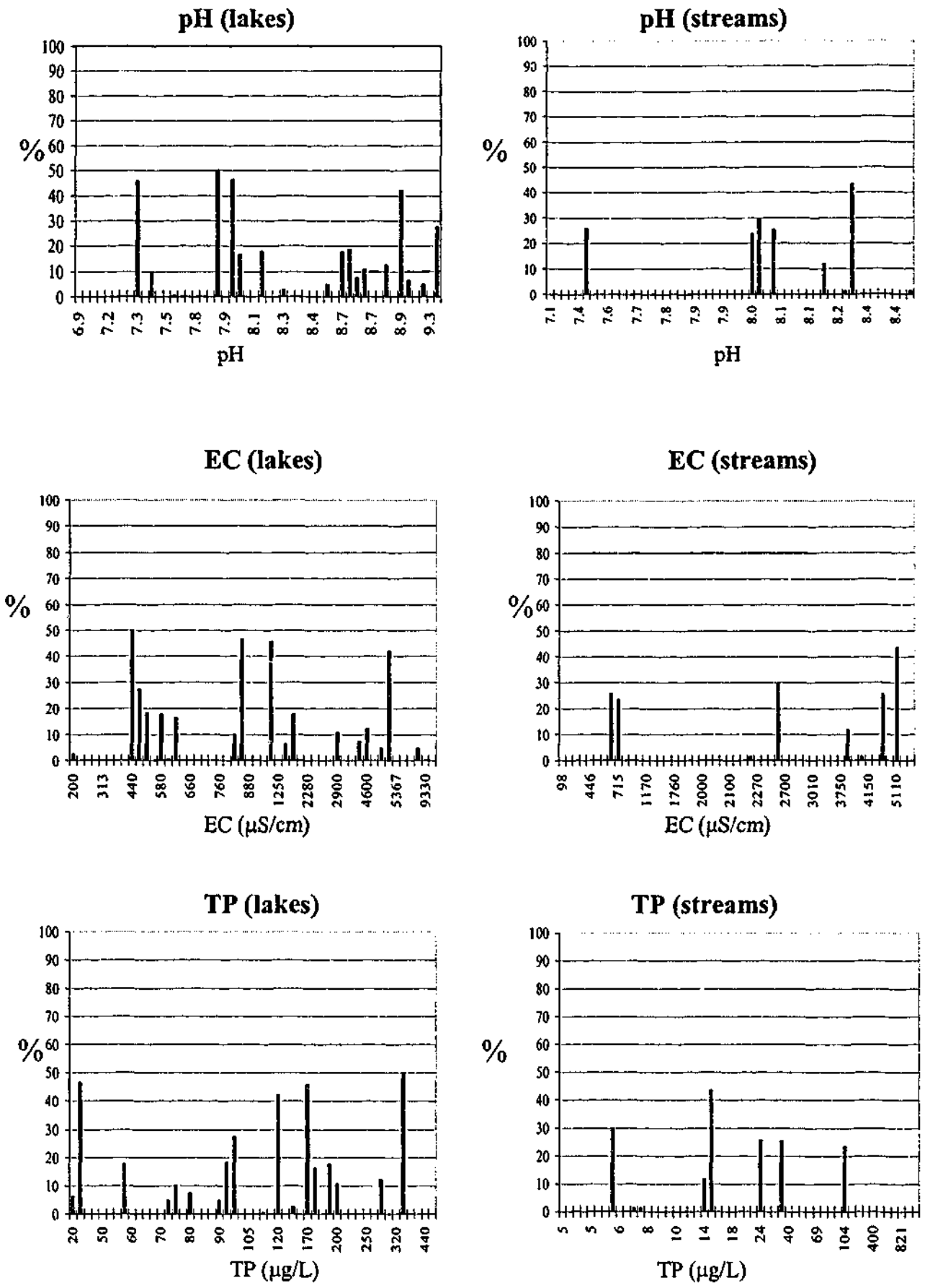


Figure 5.13: The distribution of Pseudostaurosira brevistriata *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

Summary

The reviewed literature shows that *P. brevistriata* has a preference for alkaliphilous to alkalibiontic waters, with three studies producing pH optima > 7.5. Despite several of the studies producing EC optima that indicate a preference for brackish waters, figure 5.13 showed that substantial percentages (up to 50%) of this taxon are present in fresh waters and brackish waters as well. Tolerance to TP appears to be wide, although greater abundances seem to occur in eutrophic waters. In conclusion, the ecological preferences that will be adopted for *P. brevistriata* for the reconstruction of palaeolimnology of the lower Murray River are -

pH - predominantly occurring in alkaliphilous waters
EC - is abundant in fresh to brackish.
TP - wide range of TP tolerance, but most commonly present in eutrophic waters.

5.17 The autecology of *Rhopalodia gibba*

Comparisons between published environment optima and other ecological information for *Rhopalodia gibba* are provided in table 5.15.

Table 5.15:Comparisons of optima and preferred environment data for Rhopalodia gibba

Study	pH	EC	TP
Denys (1991/92)	alkaliphilous	fresh - brackish	eutrophic
Fritz *et al.* (1993b)		5250 µS/cm	
Gasse *et al.* (1995)	8.14	630 µS/cm	
Gell (1995)		1522 µS/cm	
Reavie *et al.* (1995)			11.9 µg/L
van Dam et al. (1994)	alkalibiontic	fresh - brackish	eutrophic
Whitmore (1989)	alkalibiontic	fresh - brackish	mesotrophic - eutrophic
This study	7.66		

All of the reviewed studies conclude that *R. gibba* is an alkaliphilous to alkalibiontic taxon. The pH optima produced from lakes in this study can not be used confidently as it is based on presence in 4 samples only with a maximum abundance of 8%. Figure 5.14, which illustrates taxon distribution across the pH gradient in this study, shows that while presence is minor in the lake data set , presence is much greater in the stream data set. *Rhopalodia gibba* occurs in 14 streams with a maximum abundance of 43%, with the distribution of these samples being largely confined to between 7.8 and 8.2 pH units. This information, combined with the results from the reviewed literature, demonstrate that this taxon has a preference for alkaliphilous to alkalibiontic environments.

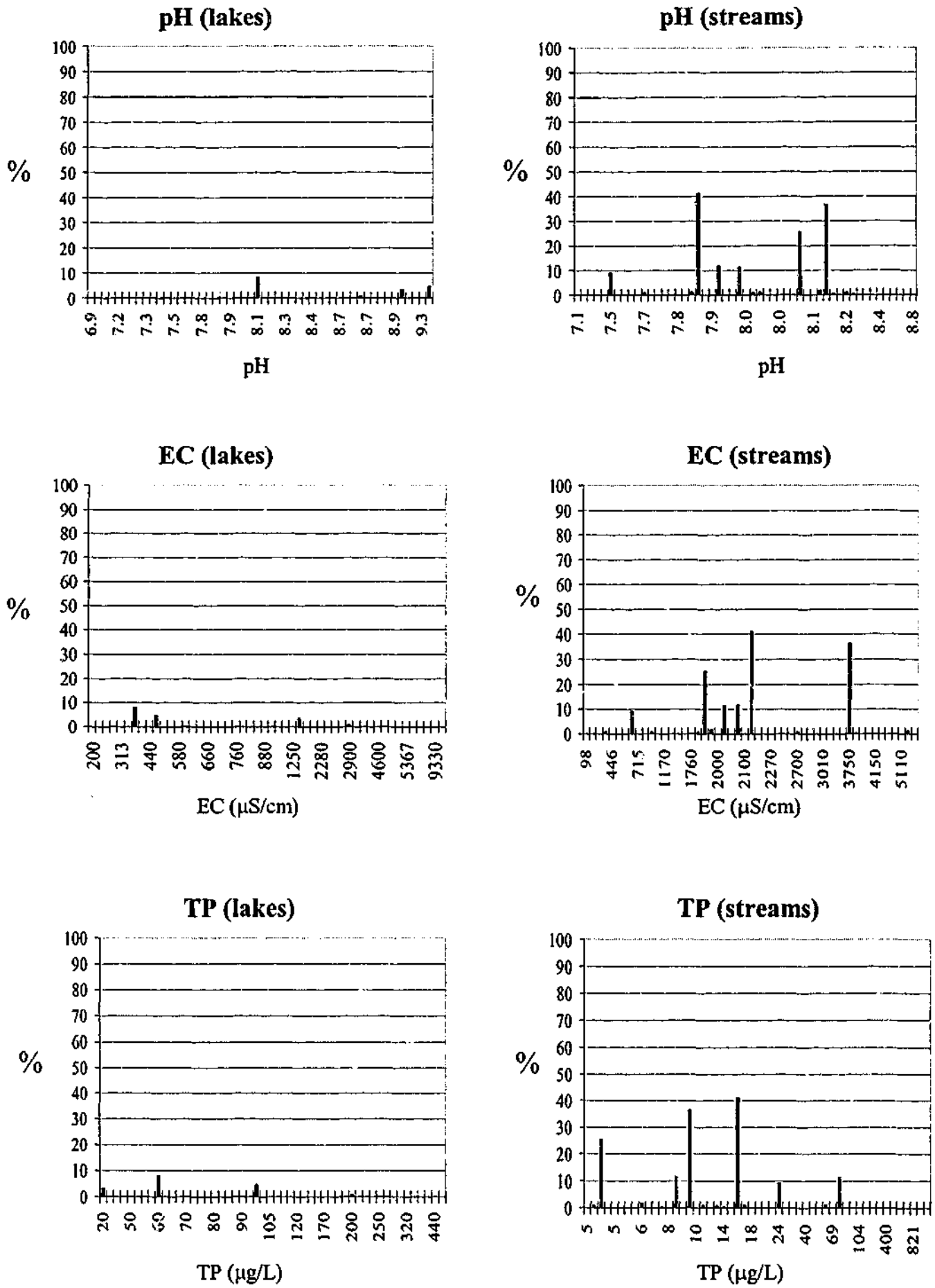


Figure 5.14: The distribution of Rhopalodia gibba *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

The relationship between *R. gibba* and EC is not narrowly defined with five of the reviewed studies stating a preference for fresh - brackish waters and two of the studies stating a preference for brackish - fresh to brackish waters (Fritz *et al.* 1993b and Gell 1995)). For comparison, the results of Fritz *et al.* (1993b) are based on 21 samples with a maximum abundance of 10%, while Gell (1995) is based on 12 samples with a maximum abundance of 4.2% and Gasse *et al.* (1995) is based on 23 samples with a maximum abundance of 11.4%. However, as was the case with pH, more conclusive information can be gained by examining the taxon distribution in the stream data set (figure 5.14). Here, it can be seen that *R. gibba* tends to generally dominate brackish - fresh streams. This distribution strengthens the results found by Fritz *et al.* (1993b) and Gell (1995), and therefore it can be fairly confidently concluded that this taxon prefers brackish - fresh conditions in south east Australian waters.

The relationship between TP and *R. gibba* is also not clear with the reviewed studies showing wide ranging results ranging from a preference for oligotrophic to eutrophic conditions. The quantitative study of Reavie *et al.* (1995) produces a TP optimum indicative of mesotrophic conditions. Figure 5.14 illustrates that the majority of samples containing *R. gibba* occur at TP concentrations between 8 µg/L and 23 µg/L. These results are in contrast to the results of the qualitative studies such as Denys (1991/92), van Dam *et al.* (1994) and Whitmore (1989) who all state that this taxon has a preference for mesotrophic to eutrophic waters. As such, the most conclusive statement that can be made about the TP preferences of *R. gibba* is that it can be present in oligotrophic to eutrophic waters.

Additional information

Rhopalodia gibba generally lives firmly attached to a stratum including benthos muds, plants, and rocks. Denys (1991/92) states that it can be common in periodic waters and that it has moderate preservation potential.

Summary

Rhopalodia gibba is abundant in waters with a pH between 7 and 9. This taxon is predominantly present in brackish - fresh waters. *Rhopalodia gibba* appears to have a wide tolerance to TP, ranging from oligotrophic to eutrophic conditions.

pH - alkaliphilous to alkalibiontic
EC - brackish - fresh conditions
TP - oligotrophic to eutrophic conditions.

5.18 The autecology of *Staurosira construens* forma *venter*

Details of environmental optima and ecological information from the reviewed studies are provided in table 5.16.

Table 5.16:Comparisons of optima and preferred environment data for S. construens *forma* venter

Study	pH	EC	TP
Bennion (1993)			71.1 µg/L
Bennion *et al.* (1999)			93 µg/L
Charles (1985)	7.3		
Denys (1991/92)	alkaliphilous to circumneutral	fresh - brackish	eutrophic to ultraoligotrophic
Dixit *et al.* (1999)	7.3		8 µg/L
Fritz *et al.* (1995)		3590 µS/cm	
Gell (1995)		4340 µS/cm	
Reavie *et al.* (1995)			12.5 µg/L
Reed (1995)		830 µS/cm	
Reid (1997)	6.56		182 µg/L
Stevenson *et al.* (1991)	6.2		
ter Braak and van Dam *et al.* (1989)	5.6		
van Dam *et al.* (1994)	alkaliphilous	< 1323 µS/cm	eutrophic to ultraoligotrophic
Whitmore (1989)	alkaliphilous		mesotrophic to eutrophic
This study	6.94		

The relationship between *S. construens* forma *venter* and pH is not clear, with estimated optima ranging between 5.6 (ter Braak and van Dam *et al.*, 1989) and 7.3 (Dixit *et al.*, 1999). However, again, some of the data are misleading due to the limited environmental ranges and / or abundances of some studies. Reid (1997), Stevenson *et al.* (1991) and ter Braak and van Dam *et al.* (1989), for instance, all have a pH range that is dominated by samples with pH < 7.0. Furthermore, in the Australian study of Reid (1997) the taxon was found at a maximum of 1.5% in the data set. The Dixit *et al.* (1999) study generated an optimum of 7.3 which was based on 48 occurrences with a maximum abundance of 24.5%. However, unlike some of the previously described taxa, there are no optima generated that are > 7.5, suggesting that this taxon may prefer circumneutral waters.

The distribution of taxon abundance across the stream data set (figure 5.15) contradicts this finding however, showing a distinct preference for sites with pH > 8.8. *Staurosira construens* forma *venter* was present in 12 stream samples with a maximum abundance of 70%. This relationship between high taxon abundance and pH over 8.0 was also found by Gell (1995), who stated that the taxon was found in the pH range of 7.6 - 10.0. It

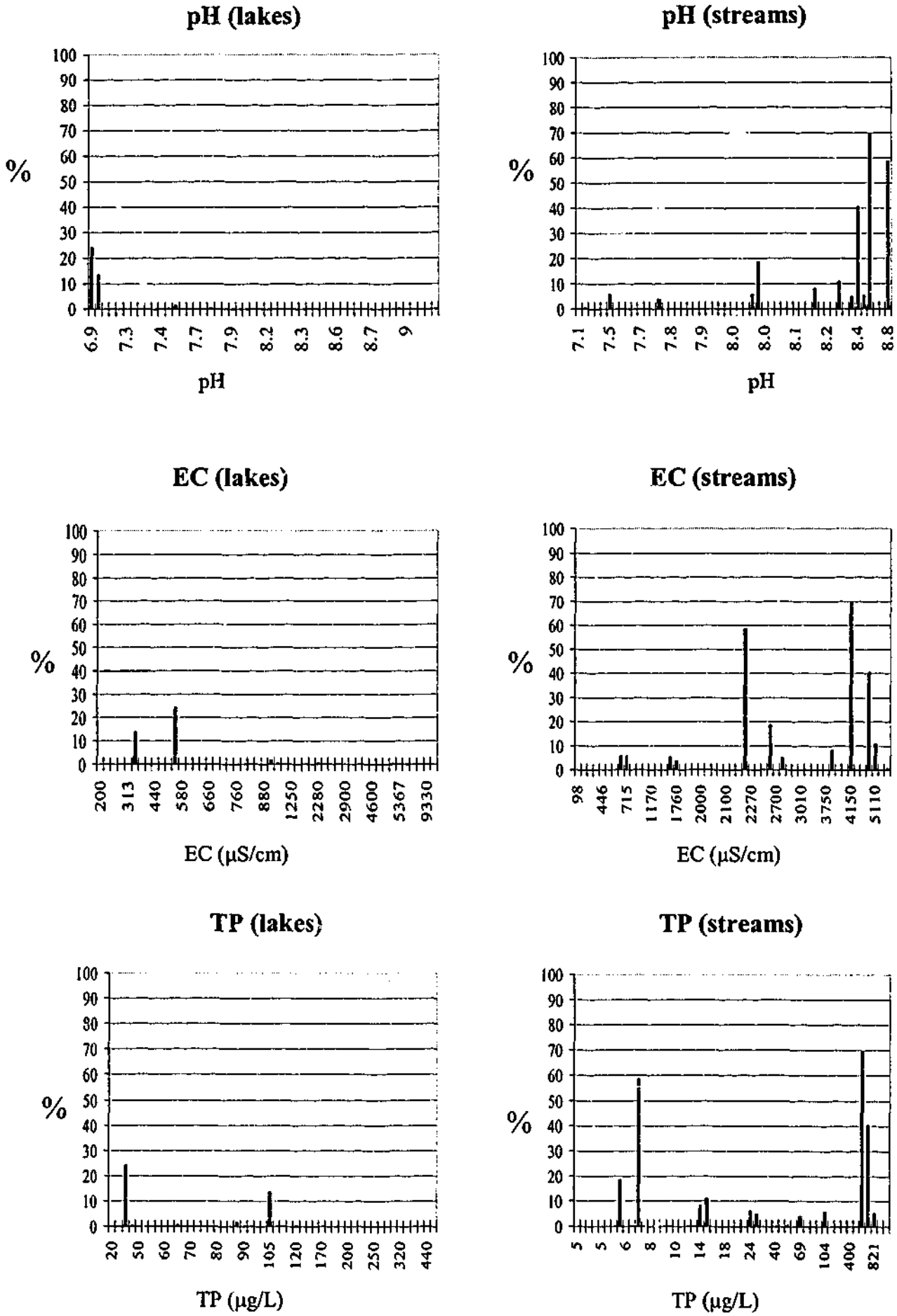


Figure 5.15: The distribution of Staurosira construens *forma* venter *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

therefore appears likely that this taxon has a wide tolerance for pH, as with the taxonomically similar *Pseudostaurosira brevistriata*.

The relationship between *S. construens* forma *venter* and EC is also somewhat confusing with derived optima ranging from 512 µS/cm (Gasse *et al.*, 1995) to 4340 µS/cm (Gell, 1995). *Staurosira construens* forma *venter* was found in 40 samples in Gell (1995) with a maximum abundance of 32%, providing substantially more information than most of the other studies, including this study. Abundances of this taxon in the stream data set (figure 5.15) provides results which are in accord with those found by Gell (1995), with the majority of samples with high abundances of this taxon having an EC of > 2000 µS/cm.

Staurosira construens forma *venter* does not have a clear TP preference as defined by the reviewed studies. Optima range from 8.0 µg/L (Dixit *et al.* 1999) to 182 µg/L (Reid, 1997). Figure 5.15, illustrating the distribution of samples containing this taxon across the TP gradient, shows a wide ranging tolerance to TP concentrations in streams with high abundances occurring at both 7 µg/L and 700 µg/L. Although the optima of Dixit *et al.* (1999) and Reavie *et al.* (1995) are biased due to the skewed nature of the TP gradient, their results show that the taxon can be present at these low concentrations (present in 48 samples and 47 samples respectively).

Additional information

Denys (1991/92) states that *S. construens* forma *venter* is a tychoplanktonic taxon, meaning that although it is a benthic taxon it can survive in the water column. The plankton survey of the lower Murray and Lake Alexandrina (results in Chapter 3) found this taxon in abundances up to 20% living in the plankton. It should be noted that this taxon is difficult to taxonomically separate from *P. brevistriata* and *Staurosirella pinnata* which may mean that results from different researchers may be troublesome to utilise without prior taxonomic harmonisation.

Summary

The results from the reviewed studies on the relationship between pH and *S. construens* forma *venter* showed a preference for circumneutral waters. However, the results illustrated in figure 5.15 showed a strong connection between high taxon abundance and high pH (> 8.0). The results of Gell (1995) combined with the distribution of this taxon in the stream samples, appear to demonstrate that *S. construens* forma *venter* has a preference for brackish - fresh to brackish waters. It seems that this taxon has a wide tolerance to TP, with it being abundant in oligotrophic to hypertrophic waters.

pH - circumneutral to alkalibiontic.
EC – brackish fresh to brackish waters
TP - tolerant of oligotrophic to hypertrophic waters.

5.19 The autecology of *Staurosirella pinnata*

Comparisons between published environment optima and other ecological information for *Staurosirella pinnata* are provided in table 5.17.

Table 5.17: *Optima and preferred environment data for* Staurosirella pinnata

Study	pH	EC	TP
Agbeti (1992)			26.5 µg/L
Bennion (1993)			94.0 µg/L
Bennion *et al.* (1996)			194 µg/L
Denys (1991/92)	alkaliphilous - circumneutral	fresh - brackish	eutrophic
Dixit and Smol (1994)	7.9		14.4 µg/L
Dixit and Smol (1995)	5.8		
Dixit *et al.* (1999)	7.6		14 µg/L
Fritz *et al.* (1995)		4340 µS/cm	
Gasse *et al.* (1995)	7.97	144 µS/cm	
Gell (1995)		5721 µS/cm	
Reavie *et al.* (1995)			21.8 µg/L
Sincock (1997)			203 µg/L (PO_4)
Stevenson *et al.* (1991)	6.3		
Tibby (2000)			8.7 µg/L
van Dam *et al.* (1994)	alkaliphilous	< 1323 µS/cm	oligotrophic - eutrophic
Vyverman (1992)	7.1 - 8.5	590 µS/cm	
Whitmore (1989)	alkaliphilous		mesotrophic - hypertrophic
This study	8.33		

With the exception of Stevenson *et al.* (1991) and Dixit and Smol (1995), the above studies indicate that *S. pinnata* is an alkaliphilous taxon. This study produced the highest pH optima of 8.33, based on presence in 16 lakes with a maximum abundance of 16%. While Gell (1995) did not produce an optimum for *S. pinnata*, he found that it had a pH range between 8.2 and 9.6 (based on 51 occurrences). Figure 5.16, illustrating the distribution of samples with *S. pinnata* across the pH gradient, shows that the majority of lake sites with the taxon present have pH > 8.0. However, the site with the highest percentage of *S. pinnata* has a pH of 7.2. The stream sites show a much stronger relationship between high pH (> 8.0) and substantial abundances of this taxon. Furthermore, the stream sites contain much higher abundances of *S. pinnata* than the lake sites (maximum abundance of 58% compared to 16%). To further complicate the relationship, however, recent palaeolimnological analysis on 30 Scottish lochs showed that this taxon can be abundant (> 40% of the diatom assemblage) in waters with pH between 5.0 and 6.5 (Bennion *et al.*, 2000). It is concluded that *S. pinnata* can tolerate a wide range of pH conditions.

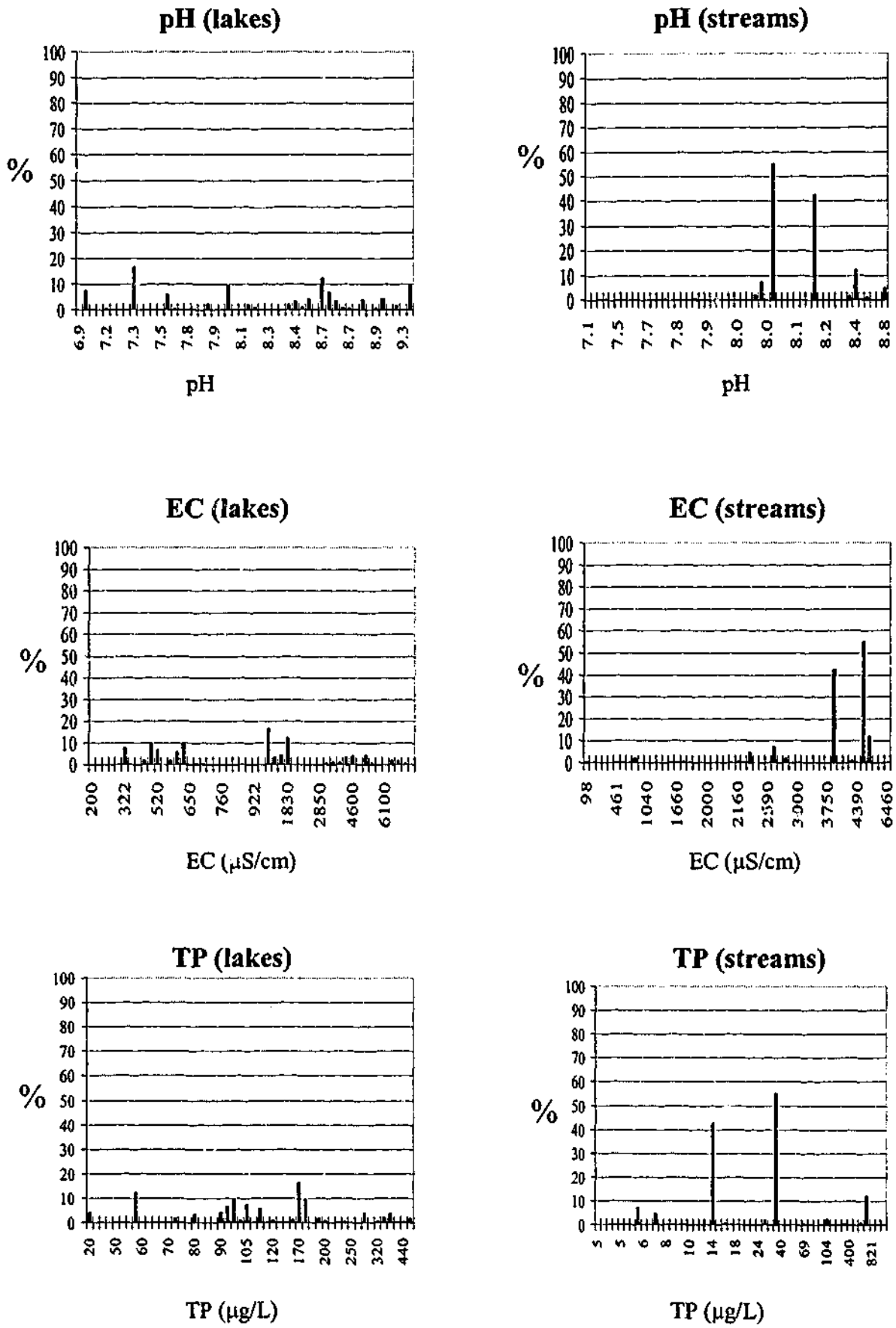


Figure 5.16: The distribution of Staurosirella pinnata *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

Based on the results in table 5.17, *S. pinnata* appears to have a wide tolerance for EC in the range of fresh to brackish waters. EC optima range from 144 µS/cm (Gasse *et al.*, 1995) to 5721 µS/cm (Gell, 1995), with results from Gasse *et al.* (1995) based on presence in 26 lakes and a maximum abundance of 5.7%, and Gell (1995) based on 51 lakes and a maximum abundance of 65%. Gell (1995) also states that this taxon shows a higher salinity tolerance than either the taxonomically similar *P. brevistriata* or *S. construens* forma *venter*. Based on the high number of samples and the concentrations of this taxon in the Gell (1995) data set, these results can be used confidently. The optimum of Gell (1995) is also quite similar to that derived by Fritz *et al.* (1995), although the maximum abundance of the taxon in that data set never exceeds 1.8%. Figure 5.6 supports the results of Fritz *et al.* (1995) and Gell (1995) as it implies that although this taxon occurs widely across the EC gradient in the lake data set, it has a well defined preference for waters with EC > 3500 µS/cm in the stream data set.

The relationship between *S. pinnata* and TP is much weaker than that of pH or EC as indicated by the results of the above studies. For instance, TP optima in the reviewed studies range from values < 20 µg/L (Dixit and Smol 1994, Reavie *et al.* 1995, Dixit *et al.* 1999, and Tibby 2000), to a value of 203 µg/L which was derived by Sincock (1997). The result of Sincock (1997), however, was based on presence in 4 samples with a maximum abundance of 1.3%. However, Bennion *et al.* (1996) produces a similar optimum of 194 µg/L, based on presence in 99 samples with a maximum abundance of 15%. The descriptive studies of van Dam *et al.* (1994) and Whitmore (1989) place the taxon within a wide ranging classification from oligotrophic to hypertrophic. Data in figure 5.16 supports the conclusions of these descriptive studies as it shows a relatively even taxon distribution across the TP gradient in both the lake and stream data sets.

Additional information

Denys (1991/92) states that this taxon is of benthic origin (lives attached to plants, rocks, surface muds). However, as with *P. brevistriata* and *S. construens* forma *venter*, the survey of lower Murray River plankton (figure 3.9) in this study showed that *S. pinnata* is present (and living) in the river plankton. Denys (1991/92) also found that this taxon is commonly moist subaerial and that it has moderate preservation potential.

Summary

The above results indicate that the pH preferences of *S. pinnata* are wide ranging, although in south east Australian waters the optima are slightly higher than elsewhere, with the taxon being most abundant in waters that have pH > 8.0. Results of Gell (1995) and this study show a preference for brackish waters with high abundances occurring at an EC of more than 3500 µS/cm, although it can also survive in fresh conditions. There did not appear to be a well defined taxon response to TP with results from the reviewed studies and figure 5.16 showing a wide ranging distribution from oligotrophic to hypertrophic waters.

pH - wide tolerance, although yet to be found in abundance in south east Australian waters with pH < 7.0
EC - preference for brackish - fresh to brackish waters
TP - abundant in oligotrophic to hypertrophic waters

5.20 The autecology of *Synedra ulna*

Comparisons between published environment optima and other ecological information for *Synedra ulna* are provided in table 5.18.

Table 5.18: *Comparisons of optima and preferred environment data for* Synedra ulna

Study	pH	EC	TP
Bennion (1993)			155 µg/L
Bennion *et al.* (1996)			137 µg/L
Denys (1991/92)	alkaliphilous - circumneutral	fresh - brackish	eutrophic
Dixit and Smol (1994)	8.0		20.1 µg/L
Dixit *et al.* (1999)	7.9		15 µg/L
Fritz *et al.* (1993b)		2400 µS/cm	
Gasse *et al.* (1995)	7.7	389 µS/cm	
Gell (1995)		3072 µS/cm	
Reed (1995)		1591 µS/cm	
Reid (1997)	6.6		160 µg/L
Sincock (1997)			90.9 µg/L (PO_4)
van Dam *et al.* (1994)	alkaliphilous	fresh - brackish	oligotrophic - eutrophic
This study	7.94		

The reviewed studies show that *S. ulna* has a pH preference for circumneutral to alkaliphilous environments. Gasse *et al.* (1995) provide the most comprehensive results with the taxon present in 103 samples with a maximum abundance of 41%, developing an optimum of 7.7. Figure 5.17, illustrating the distribution of this taxon across the pH gradient in this study, shows a wide response with samples spread across the gradient in both the lake and stream data sets. It can thus be concluded that *S. ulna* has a wide tolerance to pH within the circumneutral to alkaliphilous range.

The results from the reviewed studies on the relationship between *S. ulna* and EC are also conflicting, with optima ranging between 389 µS/cm (Gasse *et al.*, 1995) to 3072 µS/cm (Gell, 1995). As with pH, Gasse *et al.* (1995) provide the most thorough derivation of optima, with results from Gell (1995), for example, based on presence in 10 lakes with a maximum abundance of 4.4%. Reed (1995) produces an optimum of 1591 µS/cm, midway between those of Gasse *et al.* (1995) and Gell (1995), which was based on presence in 16 lakes with a maximum abundance of 13%. Figure 5.17 illustrates the distribution of this taxon across the EC gradient in the stream data set for this study, where abundances are higher than those for the lake data set. With the exception of a sample which contains 20% *S. ulna* at an EC of 3000 µS/cm, it could be said that *S. ulna* generally prefers EC concentrations between 300 µS/cm and 1860 µS/cm. There is a reticence however, to

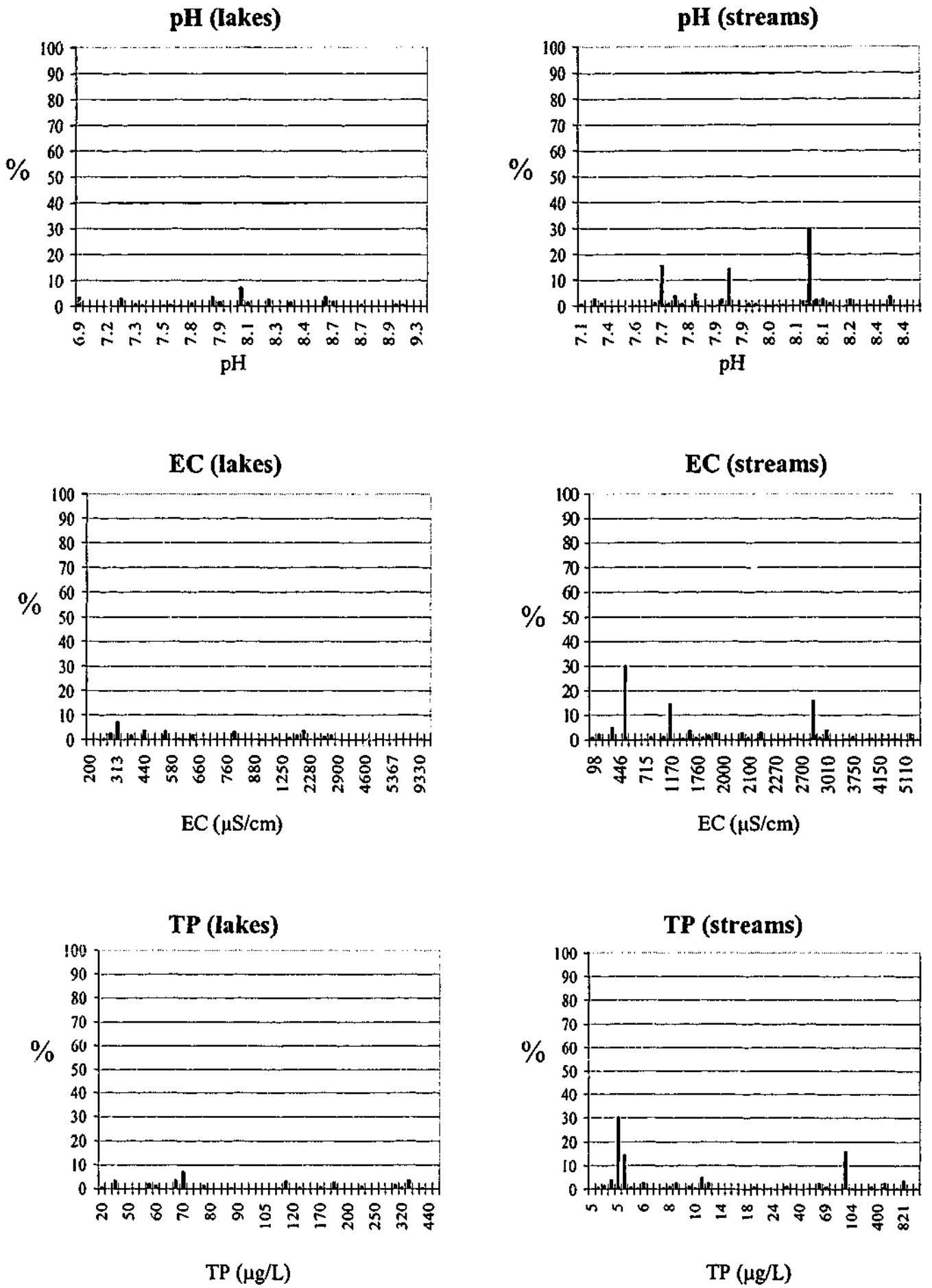


Figure 5.17: The distribution of Synedra ulna *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

disregard this sample as an anomaly, primarily because of the findings in Gell (1995), which, although not based on extensive taxon presence, do suggest that in south east Australian waters this taxon can survive in more brackish environments.

The results on the relationship between *S. ulna* and TP range from a preference for oligotrophic waters (Dixit and Smol 1994, van Dam *et al.* 1994 and Dixit *et al.* 1999) to a preference for eutrophic to hypertrophic waters (Denys, 1991; Bennion, 1993; Reid, 1997; Sincock, 1997; and Bennion *et al.*; 1999). Figure 5.17 illustrates that taxon distribution in this study, particularly in the stream data set, is generally limited to sites with TP concentrations indicative of oligotrophic conditions. In the absence of more conclusive data, it appears that *S. ulna* is present in waters ranging from oligotrophic through to hypertrophic.

Additional information

Sonneman (pers. comm.), in his study of urban stream diatom communities, found that *S. ulna* has high light requirements and is rarely found in shaded sites, although Gell (pers. comm.) has found this taxon in abundances of up to 10% in streams in the Adelaide Hills with only 20% shade cover. Denys (1991/92) states that this taxon lives primarily attached to both the substrate and plants and rocks, however, the planktonic survey undertaken in this study (see results in Chapter 6) shows that *S. ulna* is present (albeit only up to a maximum of 5%) in Murray River plankton.

Summary

The above studies show that *S. ulna* has a wide tolerance to pH between circumneutral to alkaliphilous conditions (primarily occurring at pH levels around 7.0 up to 8.0). *Synedra ulna* is present in waters with EC ranging between 300 to > 3000 µS/cm. This taxon also has a wide tolerance to TP, being present in oligotrophic to hypertrophic waters.

pH - circumneutral to alkaliphilous
EC - fresh - brackish to brackish - fresh conditions
TP - oligotrophic to hypertrophic waters

5.21 The autecology of *Tabularia fasciculata*

All of the reviewed studies which examine the relationship between *T. fasciculata* and pH conclude that this is a taxon which prefers alkaliphilous conditions (Table 5.19). The results of the two optima based studies were based on a similar number of samples, with the results of Gasse *et al.* (1995) produced from 11 samples with a maximum abundance of 36% , and the results of this study based on 7 samples with a maximum abundance of 34%. Figure 5.18, which illustrates the distribution of this taxon across the pH gradient in this study, shows that although taxon presence is limited to samples with pH below 7.8 in the lake data set, it is

more widely distributed in the stream data set, occurring in 13 samples with a pH > 8.0. However, these results still concur with the alkaliphilous classification.

Table 5.19: Comparisons of optima and preferred environment data for Tabularia fasciculata

Study	pH	EC	TP
Bennion (1993)			119 µg/L
Denys (1991/92)	alkaliphilous	brackish	eutrophic
Fritz *et al.* (1993b)		9950 µS/cm	
Gasse *et al.* (1995)	7.45	6025 µS/cm	
Gell (1995)		4869 µS/cm	
van Dam *et al.* (1994)	alkaliphilous	brackish	eutrophic
This study	7.63		

There is strong agreement that *T. fasciculata* has a preference for brackish waters. Fritz *et al.* (1993b), Gasse *et al.* (1995), and Gell (1995) all produce EC optima > 2650 µS/cm , the boundary used in this study to delineate brackish - fresh waters from brackish waters. Gell (1995) provides the most comprehensive results for this taxon, with EC optima derived from presence in 45 samples with a maximum abundance of 46%. These samples were also taken from lakes with a similar brine type to that of the lakes which are the focus of the palaeolimnological section of this study.

The relationship between TP and *T. fasciculata*, as indicated by the above studies, appears to be that this taxon has a preference for eutrophic to hypertrophic waters. Sonneman (pers. comm.), in a survey of urban stream diatoms in Victoria, Australia, primarily found this taxon in high nutrient environments (> 50 µg/L TP). Figure 5.18, illustrating the distribution of this taxon across the TP gradient in this study, shows that although it is present across the stream gradient, those samples with the highest abundances of this taxon are generally eutrophic to hypertrophic.

Additional information

Denys (1991/92) states that this taxon is an epontic taxon, that is, it lives almost exclusively attached to the substrate. He also states that it can be common in ephemeral waters and that it has moderate preservation potential.

Summary

Tabularia fasciculata is an alkaliphilous taxon, predominantly occurring at pH > 7.0. This taxon prefers living in brackish waters . *Tabularia fasciculata* is abundant in eutrophic to hypertrophic waters.

pH - alkaliphilous

EC - brackish

TP - eutrophic to hypertrophic

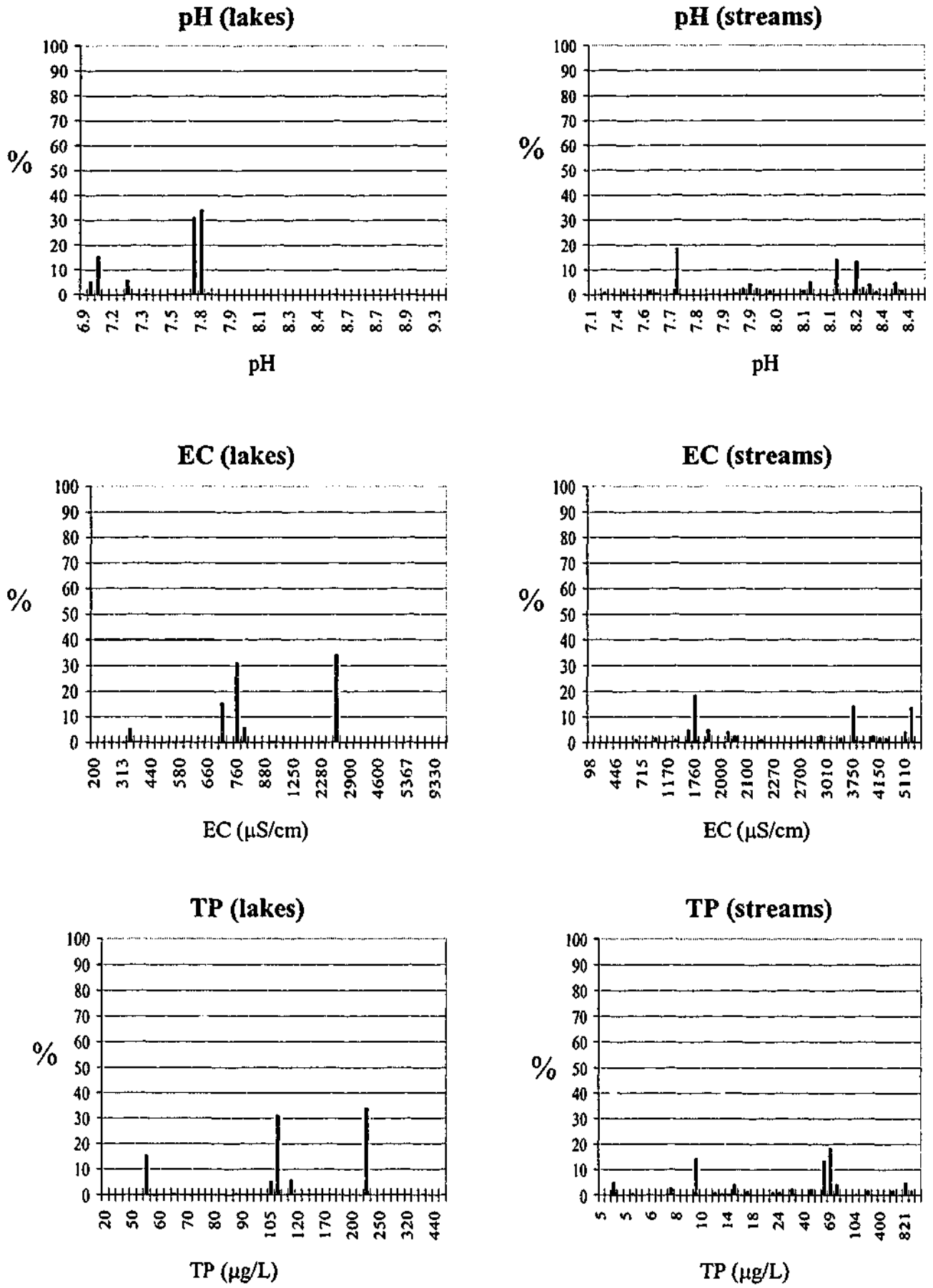


Figure 5.18: The distribution of Tabularia fasciculata *across the pH, EC and TP gradients in the lake and stream data sets, showing relative abundance in each sample. Each tic mark on the X axis represents a single sample.*

5.22 Implications of this review

Weighted Averaging regression produces environmental optima for individual diatom taxa and these optima are then used to calibrate or reconstruct that variable in the fossil record. However, as this chapter has shown, not all taxa have clearly defined optima for individual variables. Rather, taxa often have wide ranges where they are most abundant. For instance, *C. placentula* has optima between 467 µS/cm and 17 680 µS/cm. However, despite this taxon being relatively evenly distributed across this gradient, WA regression is designed to generate an optimum for EC. Essentially, it is an arithmetic optima rather than an ecological optimum, and this optimum may grossly overestimate or underestimate past EC conditions when the fossil record is calibrated.

This review of literature on diatom / environmental variable relationships commonly demonstrated conflicting results between studies. When these results were analysed in more detail it could be seen that most of this disagreement was due to differences in environmental variable ranges and also the number of samples that those species were present, and their maximum abundances in the data sets. Not only does the range of variables have an enormous influence over derived optima, but the distribution of samples within this range also has a substantial influence. For instance, the TP range of the Reavie *et al.* (1995) study covered oligotrophic to hypertrophic conditions, but most of the samples were concentrated within the oligotrophic to mesotrophic range (51 out of 59 samples), and thus TP optima were nearly always much lower than other studies with similar ranges, but with a more even spread of samples. Ideally, what is needed for accurate derivation of environmental optima is broad gradients with minimal data gaps. This point is constantly emphasised by ecological statisticians (see Birks 1995 and 1998), but it is often difficult to employ, particularly in a nation where water quality monitoring is not comprehensive and / or consistent, and also where there is a relative paucity of inland aquatic environments (such as semi – arid south east Australia).

This review of autoecological information emphasises the pressing need for continuing research on the complex relationship between diatoms and environmental relationships. Significant correlation coefficients are an indication of Weighted Averaging model efficacy. However, the question must be raised, if diatoms do display true Gaussian responses to external forces, then why can two (or more) studies with similar correlation coefficients produce environmental optima that are at different ends of the environmental gradients ? In attempting to address this problem, there is a need for merging and integration of existing modern diatom data sets, perhaps starting with the south east Australian studies (i.e. Gell, 1995; Reid, 1997; Tibby, 2000; and this study) leading to integration with Northern Hemisphere (i.e. Cameron *et al.*, in press) merged data sets. This is explored further in chapter 8.

Chapter 6 - Construction of palaeolimnological records from the lower Murray River region

6.1 Introduction

This chapter describes the methodology and results of the palaeolimnological techniques applied to four sediment cores taken from two sites in the lower Murray River region. These results are interpreted in chapter 7.

6.2 Core collection

6.2.1. Site selection

As stated in chapter 1, the palaeolimnological aims of this thesis were (in relation to the lower Murray River):

a) to establish benchmark limnological data for conditions prior to the arrival of non - indigenous Australians.
b) to determine the degree of limnological change since the arrival of non - indigenous Australians.
c) to undertake high resolution analysis of the impacts of river regulation on water quality.

To achieve these aims, several criteria were established to select suitable sites for palaeolimnological analyses. Firstly, the location of sites had to be in a region of undisturbed sediment. For chronological integrity it is essential that sedimentation is continuous and undisturbed. The main channel of the Murray River was therefore avoided because of the risk of sediment movement and slumping. Although the risk is decreased in weir pools, these were not deemed to be suitable because of the possibility of mass movement during flood or high release events. Billabongs were also avoided because of the risk that they were once ephemeral, resulting in a discontinuous diatom profile, and also the risk that they were subject to mostly autochthonous processes (see Reid, 1997) and therefore not a true reflection of the ecology of the main river channel.

It was also essential [illegible] ple sites were selected because of the different time frames in question. For instance, for the section of the study that concentrates on comparing pre and post non-indigenous settlement environments, sites with records of at least 300 years were needed. The site(s) selected as the focus for river regulation assessment ideally needed a reasonably high sedimentation rate so that annual or biannual data could be extracted. Additionally, it was preferable that all sites were in regions where there were existing data on the sedimentology (i.e.: stratigraphy, geochemistry, dating), to complement the fossil diatom records of this study.

Two sites that fulfilled the above criteria were Lake Alexandrina, South Australia (plate 1 and figure 6.1) and Lake Cullulleraine, Victoria (plate 2 and figure 6.3).

Plate 1: Lake Alexandrina, South Australia. Northern embayment.

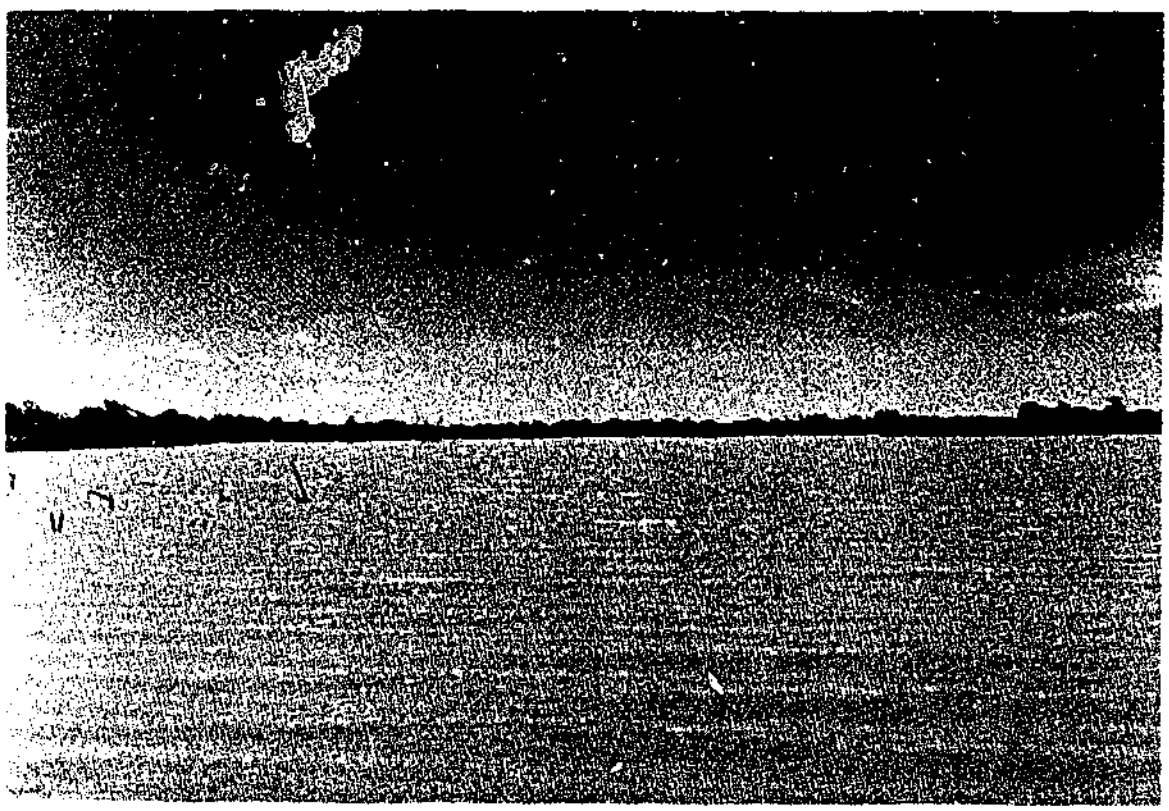


Plate 2: Lake Cullulleraine, Victoria.

6.2.2 Lake Alexandrina

6.2.2.1 Site characteristics

Lake Alexandrina is a broad shallow lake located at the mouth of the Murray River, in South Australia (see figure 6.1). It is over 660 km^2 in area and currently has a water volume of 1.66 10^6 ML. It forms part of the Murray River terminal lakes system which also incorporates the adjacent Lake Albert and Coorong. During the early Holocene the terminal lakes were an estuarine system measuring almost 75 000 hectares, characterized by very variable flows (Barnett, 1993). Geologically, the system formed in response to eustatic sea level rise following the Last Glacial Maximum (c. 18 000 years ago, stabilizing c. 6000 years ago). Sprigg (1952) found that the location of the lakes reflects ongoing subsidence of the local region in relation to the Mount Lofty Ranges and the Robe - Naracoorte area. Bourman and Barnett (1995) state that the Murray River channel is incised through Tertiary sediments when it enters Lake Alexandrina, and through calcreted Pleistocene sediments between Points Sturt and MacLeay (see figure 6.1). Magnetic reversal studies of these Pleistocene sediments indicate that this has been the position of the river channel for at least the last 780 000 years (Bourman and Murray-Wallace, 1991).

Controlling tidal flows to the terminal lakes systems was crucial to the success of river regulation and consequently five barrages were constructed between Lakes Alexandrina and Albert and the Coorong (Cann *et al.*, 2000). The barrages have a total length of 7.6 km and were completed in 1940. The barrages have transformed Lakes Alexandrina and Albert into permanent freshwater bodies, while the Coorong, in the absence of a natural flushing regime, has become more saline.

The most pertinent limnological characteristics of Lake Alexandrina are that it is shallow (maximum depth of 4.05 m and mean depth of 2.86 m) and highly turbid. The dominant cation is sodium and the dominant anions are chloride and carbonate / bicarbonate (Geddes, 1984a). The abiogenic turbidity is mostly derived from the Darling River tributary which carries a very high load of suspended fine clays (see Chapter 1 for further discussion). Salinity has been regularly measured in Lake Alexandrina since the early 1940s, nutrients and have been monitored since 1977 and turbidity and temperature since 1980. Yearly averages for conductivity (since 1943), and three monthly moving averages for pH and total phosphorus (since 1978) are shown in figures 6.2a, 6.2b and 6.2c.

Geddes (1984a, 1984b, and 1988) has conducted detailed biological and chemical analysis of Lake Alexandrina and concluded that the ecology of the lake is controlled by abiogenic turbidity which affects light penetration and also nutrient levels (with significant correlation between turbidity and TP and TKN). Perhaps more importantly for this study, these changes in photic depth impact upon phytoplankton concentration and diversity.

The lake edge flora is dominated by *Typha, Phragmites* and *Myriophyllum*, with minor communities of *Triglochin* in the shallows. Mats of filamentous algae, including *Cladophora*, *Enteromorpha* and *Oscillatoria* are often recorded in the main basin and northern lagoon, occasionally covering much of the lake bottom

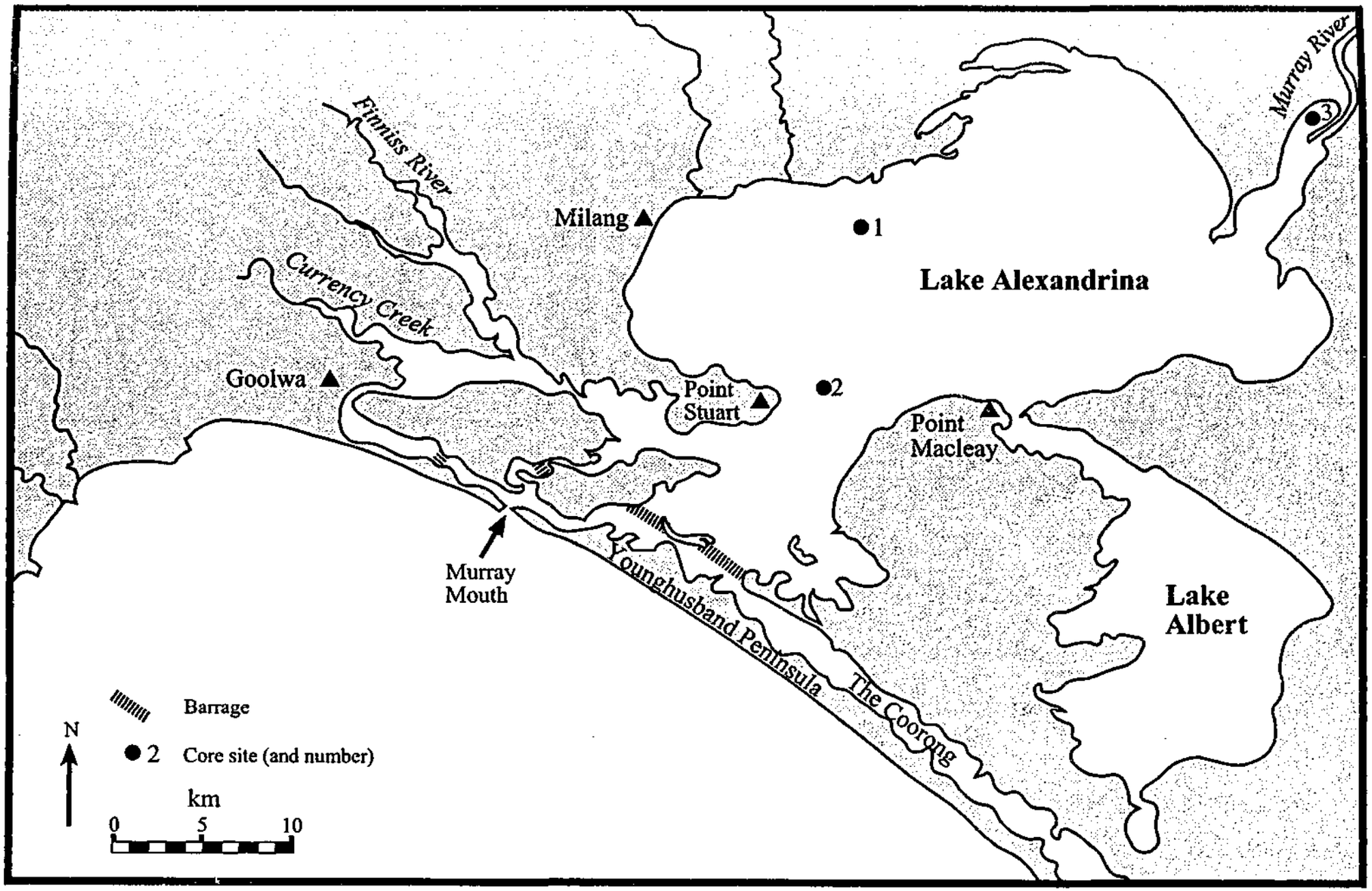


Figure 6.1: Map of Lake Alexandrina, South Australia, with location of coring sites.

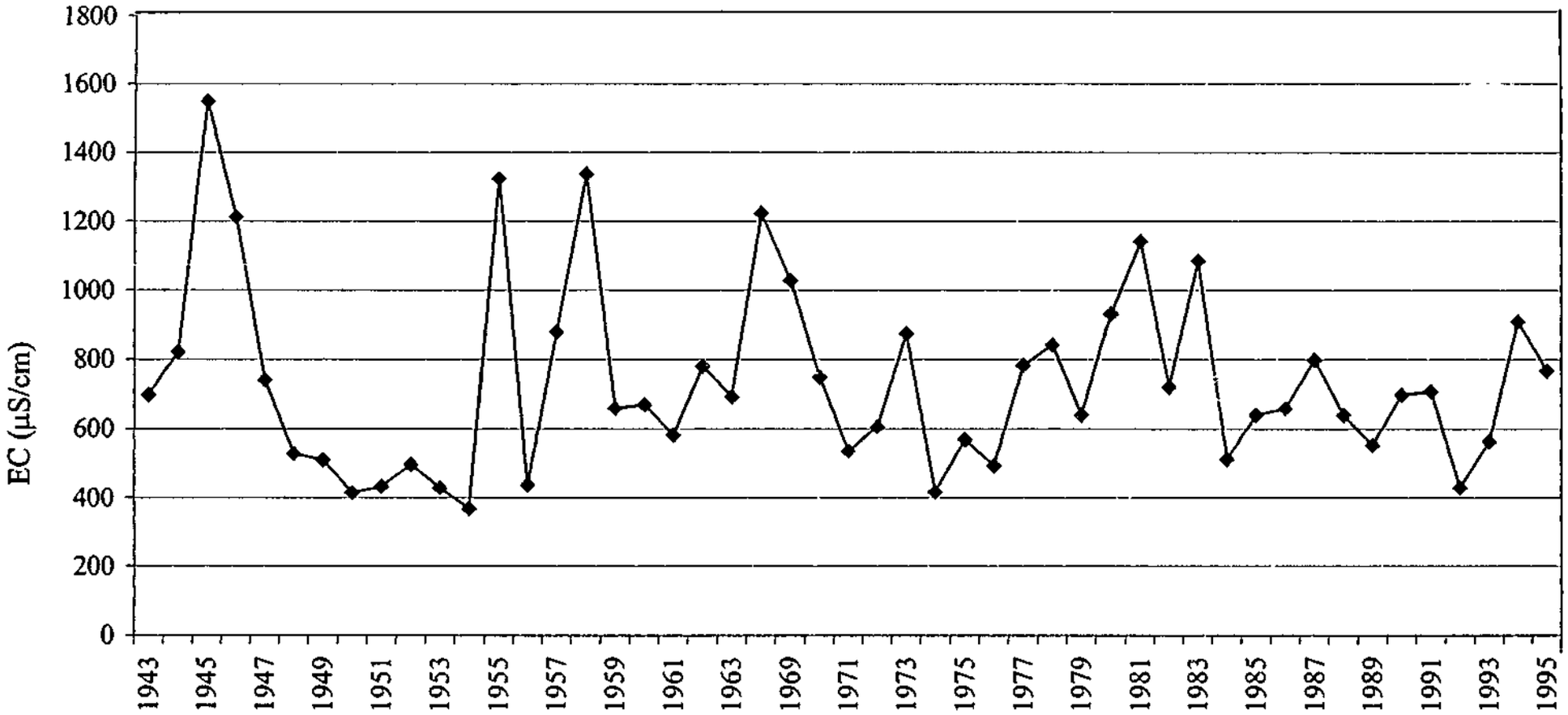


Figure 6.2a: *Yearly averages of EC data from Lake Alexandrina. Measurements were taken from Milang. Source of data: Murray Darling Basin Commission*

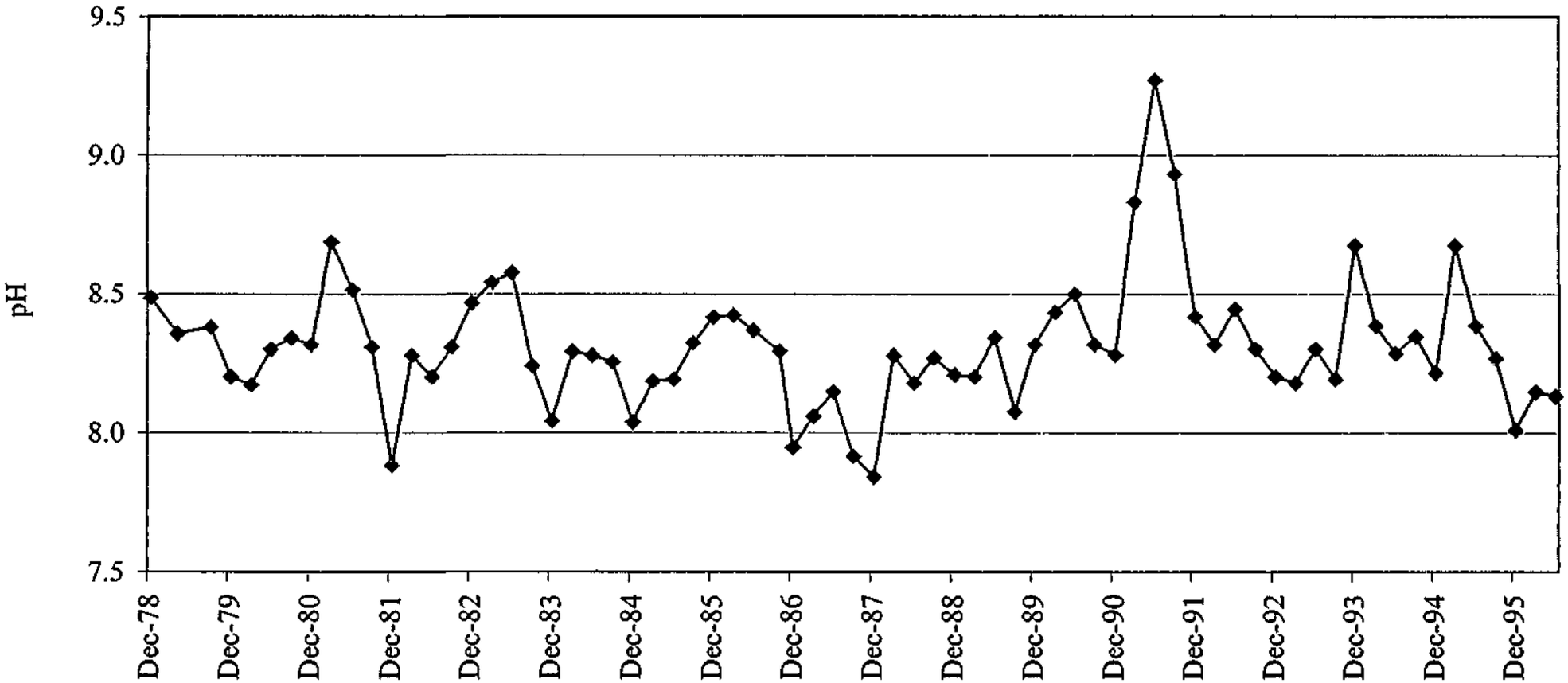


***Figure 6.2b**: Three monthly moving averages of pH data for Lake Alexandrina. Measurements were taken from Milang. Source of data: Murray Darling Basin Commission*

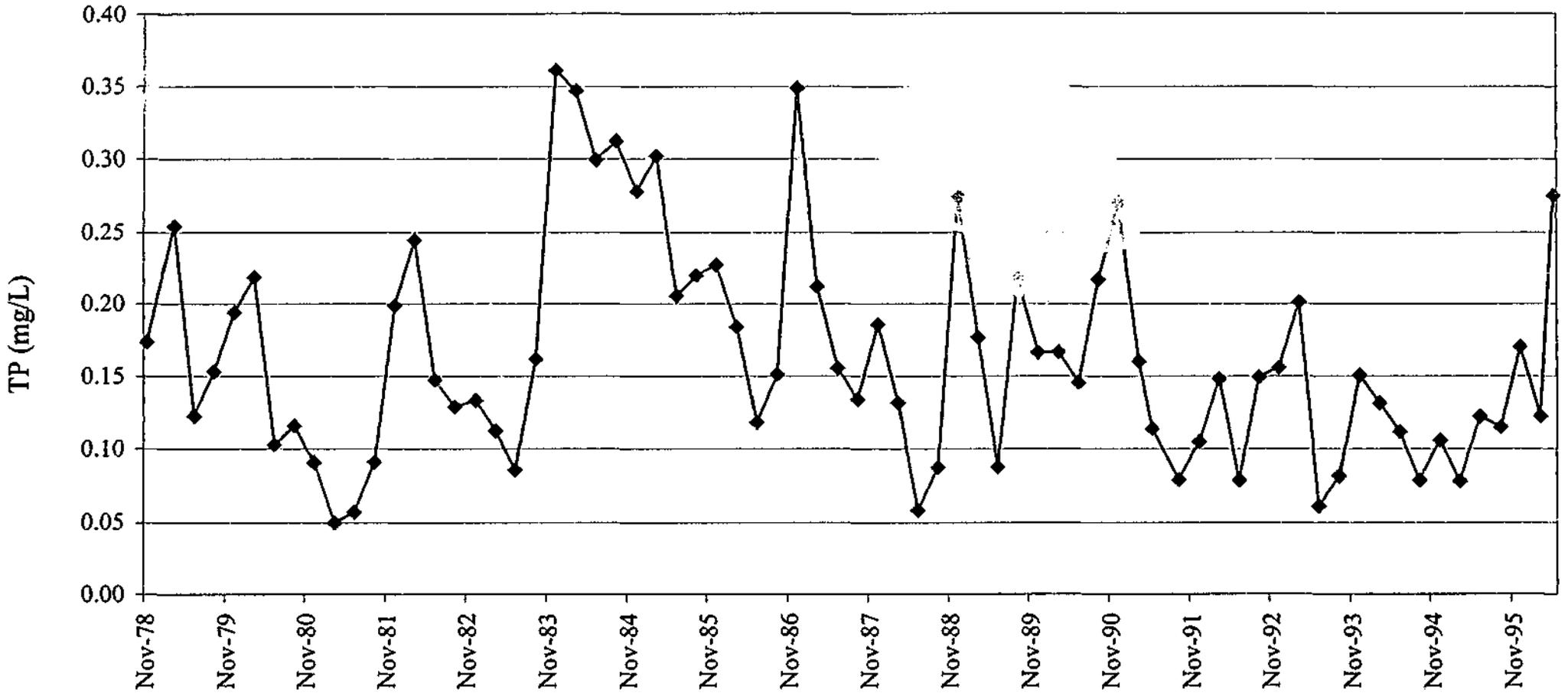


***Figure 6.2c**: Three monthly moving averages of TP data from Lake Alexandrina. Measurements were taken from Milang. Source of data: Murray Darling Basin Commission.*

(Edyvane *et al.*, 1996). The lake and its surrounds also support a diverse bird community, with the area recognised as a "Wetland of International Importance" under the Ramsar Convention. It is also included on the register of the National Estate (DENRa, undated).

6.2.2.2. Coring methodology

Lake Alexandrina was selected as a site for investigating natural variability in lower Murray River water quality as existing research has shown that sediments have accumulated throughout most of the Holocene period (Barnett, 1993; Von der Borch and Altman, 1979). Barnett (1993) analysed the recent sedimentology of Lake Alexandrina from 33 piston cores. Barnett's research showed that the majority of the sediments in Lake Alexandrina have been unaffected by slumping or channelisation and appear contiguous. The results also showed that the southern sections of the lake were less likely to represent regional catchment processes with substantial marine interference whereas the northern and central sections would most likely represent riverine processes with negligible marine incursion. The cores were dated using both ^{210}Pb and ^{14}C methods which resulted in sedimentation rates being established for different sections of the lake, ranging between 1.25 and 1.7 mm/yr.

The cores obtained by Barnett (1993) were sought for use in this study, but upon examination it was found that the sediments were substantially dried with large cracks disrupting core integrity. One core, however, taken from west of the main basin had been stored separately at Monash University and was in excellent condition (core 16 in Barnett, 1993). This core, referred to as Core 1 in this study, was opened, lithographically described and the diatom content analysed. Also located at Monash University were sediment samples from an additional core taken from the centre of the main basin. This core, referred to as Core 22 in Barnett (1993), was the primary focus of Barnett's (1993) study, and as such was dated extensively and lithographically described in detail. There was sufficient remaining sediment for diatom analyses and so this core was also included in this study (referred to as Core 2).

There were no cores available from the northern region of Lake Alexandrina available for further examination from the Barnett (1993) study. Analysis of this section of the lake was deemed essential to an understanding of lower Murray River palaeolimnology because of its proximity to the entrance of the river channel. Therefore, four cores, ranging in length from 1.1 m to 1.3 m, were extracted from the northern sheltered embayment in March 1997 using an impact corer. The longest of these cores was selected for diatom analysis and lithostratigraphic description (named Core 3).

The location of the coring sites are shown in figure 6.1.

6.2.3. Lake Cullulleraine

6.2.3.1 Site characteristics

Lake Cullulleraine is an artificial storage lake which was constructed in 1919, and is located approximately 50 km downstream of Mildura (see figure 6.3). The lake is maintained by pumping water from Lock 9 on the Murray River and was developed as part of the river regulation system. It supplies domestic and stock water

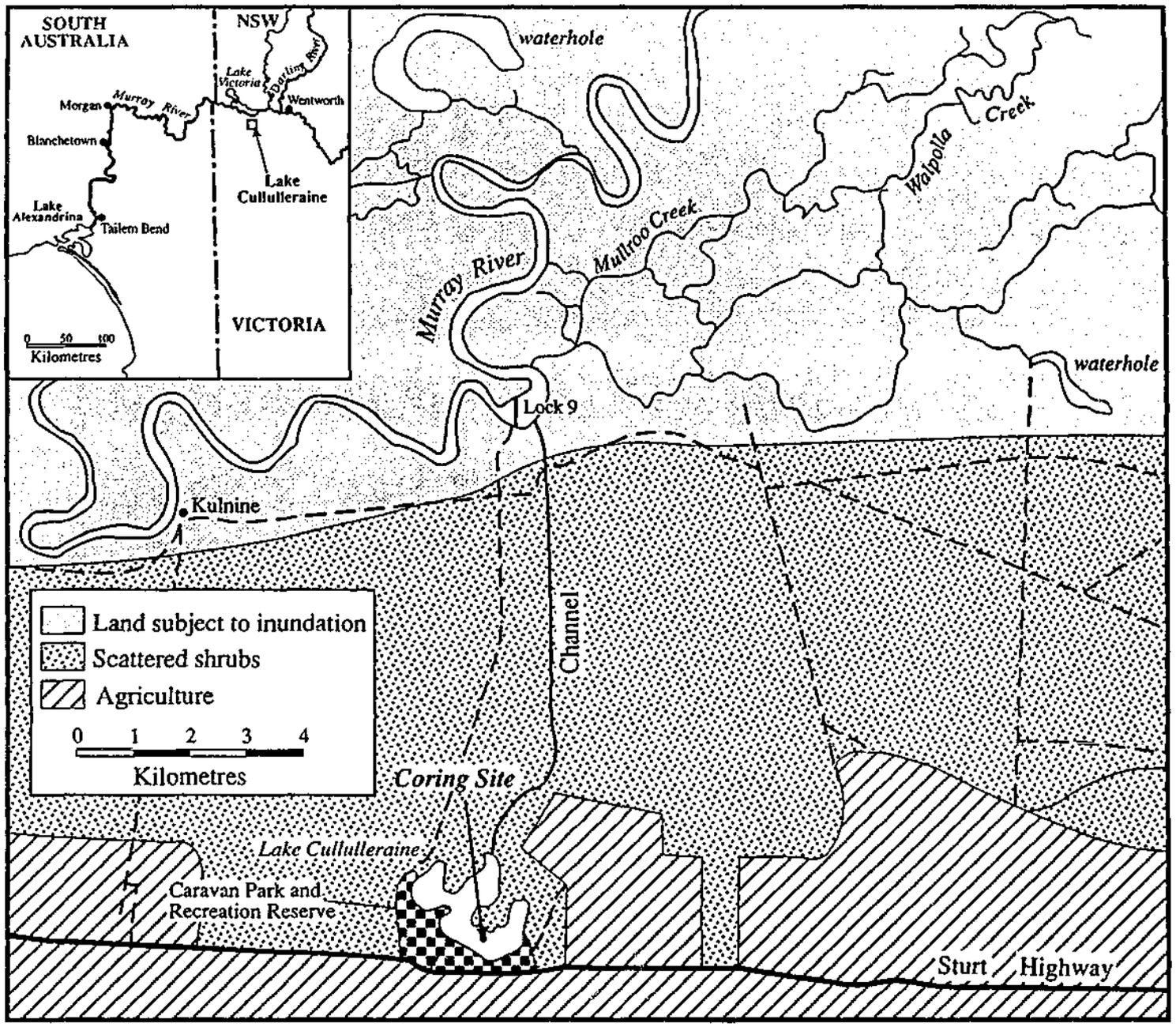


Figure 6.3: *Map of Lake Cullulleraine, Victoria, with location of coring site.*

to the Millewa district west of Mildura and is also a primary storage for the Millewa Pipeline Scheme (a 664 km pipeline network). The lake is surrounded by both red sand hills and agricultural land (primarily vineyards), and there is a small camping ground located on the southern shore. Although water quality has been monitored at Lock 9 since the early 1960s, it has not been consistently monitored within Lake Cullulleraine. The only water quality data available is quarterly measurements since 1996 (Australian Water Quality Centre, Bolivar, Adelaide) and these are presented in table 6.1. There is also a lack of geomorphic or physical data available. From the authors own observations, the lake has very similar physical attributes to Lake Alexandrina, that is, it is a shallow (sounding indicated a maximum depth of 4.5 m), wind stressed, turbid lake. At the time of coring, the lake edge sustained healthy communities of *Typha* and *Phragmites*.

Table 6.1: Water quality data for Lake Cullulleraine, 1996 - 2000

Date	pH	EC (µS/cm)	Turb (NTU)	DO (mg/L)	Temp	TP (µg/L)	TKN (µg/L)
14.10.96	8.53	656	76	13.69	20.6	80	800
3.12.96	8.5	504	45	8.1	20.4	100	1000
23.2.97	8.37	602	119	8.5	29.6	90	900
2.5.97	8.51	597	182	10.6	18	80	800
26.8.97	8.91	628	175	12.81	13.4	40	600
22.12.97	8.55	569	385	9.34	25.3	20	500
20.2.98	8.87	606	86	12.14	19.9	20	600
22.5.98	8.78	553	135	10.32	13.3	40	500
2.9.98	8.37	525	54	11.42	15.6	127	1160
19.11.98	8.67	535	93	11.53	18.6	123	1090
6.2.99	8.53	567	49	7.93	26.4	49	920
11.6.99	8.77	524	30	13.5	12.7	74	1050
21.10.99	8.3	520	150	9.4	20	92	940
3.12.99	8.1	470	97	7.7	23	93	1100
4.2.00	8.43	551	123	8.37	25.9	152	450
13.5.00	8.11	523	62	6.17	14.6	105	1130

6.2.3.2 Coring methodology

Two cores (46 and 48 cm in length) were taken from the centre of Lake Cullulleraine using a soft sediment corer. The longer of these two cores, called Core 4, was selected for diatom and sediment analysis. The location of the coring sites are shown in figure 6.3.

6.3 Core analyses

6.3.1 Dating

The efficacy of palaeolimnological records relies on accurate dating techniques, a variety of which are available to palaeolimnologists. Dating of sediments is critical for most palaeolimnological studies as it can provide absolute dates for levels as well as allowing the calculation of mean sedimentation rates. A number of techniques are available for dating lake sediments, including qualitative stratigraphic, palynological, and radioactive isotope analyses (e.g.: ^{14}C, ^{210}Pb, ^{137}Cs). The first two dating methods are relatively simple and

inexpensive, while radioactive isotope analysis is often applied more strategically due to the high cost involved. This project was fortunate in that both core 1 and core 2 were already extensively dated by both ^{210}Pb and ^{14}C (Barnett, 1993). Forming part of this study's analysis, core 3 was dated using ^{210}Pb and ^{14}C and core 4 was dated using ^{210}Pb analysis. An attempt was also made to date core 1 and core 3 using palynological methods.

6.3.1.1 Radiocarbon dating

Radiocarbon dating utilises organic material in the sediments and dates them according to the half life of contained ^{14}C. The method is based on the fact that upon the death of an organism ^{14}C ceases to be assimilated and decays with a half life of approximately 5730 years. The main assumption of the method is that the ^{14}C : ^{12}C ratio in the atmosphere has remained constant from the time of death until measurement of the ^{14}C concentration (Libby *et al.*, 1949). However, over time, variations in the ^{14}C : ^{12}C ratio have occurred, including changes in the production rate of ^{14}C with increases due to atomic bomb testing from the 1950s onwards, and it is common for these variations to be accounted for in the calculation of dates (i.e.: Klein *et al.*, 1982).

Sediment samples that are either too small (typically < 20 g dry weight) for conventional ^{14}C dating, or have a low carbon content, can be radiocarbon dated using Accelerator Mass Spectrometry (AMS). The major difference between AMS and conventional ^{14}C dating is that the radiocarbon atoms are directly detected instead of waiting for them to decay, and thus, sample sizes can be up to 1000 times smaller.

All three cores from Lake Alexandrina were radiocarbon dated using both conventional and AMS dating methods, as outlined in table 6.2.

Table 6.2 Sampling depth interval, lithology and carbon source for Cores 1, 2 and 3.

Core	Depth (cm)	Lithology	Carbon	Method
1	119 - 125	Wood fragment in soil	Org C	Conventional[1]
2	50 - 60	Lacustrine mud	Org C	Conventional[1]
	122 - 129	Lacustrine mud	Org C	Conventional[1]
	251 - 259	Lacustrine mud	Org C	AMS[3]
	251 - 259	Lacustrine mud	Org C	Conventional[1]
	402 - 410	Lacustrine mud	Org C	Conventional[1]
	492 - 494	Lacustrine mud	Org C	AMS[2]
3	40 – 43	Lacustrine mud	Org C	AMS[4]
	65 - 68	Lacustrine mud	Org C	AMS[4]
	82 - 85	Lacustrine mud	Org C	AMS[4]

[1] Conventional ^{14}C from the Division of Water Resources, CSIRO, Adelaide.

[2] AMS ^{14}C age from the Division of Exploration Geoscience, CSIRO, Sydney.

[3] AMS ^{14}C age from the department of Nuclear Physics, Australian National University

[4] AMS ^{14}C age from the University of Waikato Radiocarbon Dating Laboratory

6.3.1.2. ^{210}Pb dating

Dating based on ^{210}Pb analysis uses the ^{238}U decay series in which ^{210}Pb is an intermediate daughter with a half life of 22.26 years. It is therefore only appropriate for dating sediments younger than c. 150 years (with the absolute date able to be determined dependant upon the concentration of ^{210}Pb at the top of the core). Unsupported ^{210}Pb in modern sediments is derived from aerosols (^{222}Rn) in the atmosphere, where it resides for up to 30 days. This unsupported ^{210}Pb is then incorporated into lake sediments through direct precipitation or catchment erosion and inwash. There are also supported forms of ^{210}Pb in lake sediments which are derived from minerals containing uranium and its daughters including ^{226}Ra, and is present at all depths of the sediment (Krishnaswamy *et al.*, 1971). Lead dating of sediments requires a constant supply of unsupported ^{210}Pb and also needs to be calibrated for the presence of supported ^{210}Pb. Calibration is undertaken by measuring the amount of ^{226}Ra or by examining the amount of ^{210}Pb at depths that exceed a background stable level so that only supported ^{210}Pb remains in the sediment and then calculating the activity of excess ^{210}Pb (see Krishnaswamy *et al.*, 1971 for further explanation).

Lead dating was undertaken on core 2 by Barnett (1993). The process involved taking samples in the field at 2 cm intervals. The samples were then sealed in preweighed containers and oven dried at 60 °C. All analyses were undertaken by Barnett (1993) with a further six samples submitted to AMDEL laboratories, Adelaide, for ^{226}Ra analysis by the Rn emanation method to determine support levels of ^{210}Pb.

The ^{210}Pb dating on cores 3 and 4 was conducted by the Australian Nuclear Science and Technology Organisation (ANSTO). Eleven samples were obtained from core 3: a surface sample, 2 cm, 4 cm, 6 cm, 8 cm, 10 cm, 12 cm, 14 cm, 16 cm, 18 cm and 20 cm. Six samples were obtained from core 4: a surface sample, 8 cm, 16 cm, 25 cm, 32 cm, and 40 cm.

6.3.1.3 Dating using palynological techniques

A method for extracting all *Pinus* grains in a sediment sample, and thus using the presence of *Pinus* pollen as a dating tool, was developed by Ogden (1996) and is outlined in Appendix 7. As *Pinus* is an introduced tree in Australia, its pollen is useful as a stratigraphic marker for estimating the arrival of non-indigenous Australians. *Pinus* species are prolific pollen producers and the pollen is efficiently transported long distances (Faegri and Iverson, 1975). Pines were introduced, primarily by Europeans, in the late 1850s with widespread plantings in the 1860s and 1870s throughout the Murray - Darling Basin. Consequently, *Pinus* is an abundant taxon in many post non-indigenous contact records in south eastern Australia. Ogden (1996) states that *Pinus* species begin to produce pollen after 5 - 10 years of growth and so it is likely that *Pinus* pollen was being deposited into the sediments of Lake Alexandrina from about 1870 onwards. The source of the *Pinus* pollen in Lake Alexandrina sediments could possibly originate from a very large catchment area as it has been shown that in depositional sites on large Australian rivers, the majority of pollen grains are fluvially transported (Clark, 1990b).

Two cores from Lake Alexandrina (cores 1 and 3) were processed for pine pollen detection, with samples taken every 5 cm (total of 20 samples for each core). There was no extra sediment available from Core 2 for this method.

6.3.2 Lithostratigraphic analyses

6.3.2.1 Sediment description

The triangular textural plot sediment description method of Folk (1968), used by Barnett (1993), was adopted in this study so as to achieve descriptive consistency between the different cores. This method involved classifying the inorganic component of the sediment into five units. In order to reduce the number of sediment lithologies described, several similar categories were combined. These included combining silty sand and sandy silt with sand, clayey sand and sandy clay combined with sandy mud to muddy sand, and sediments with a clay : silt ratio greater than 2:1 or less than 1:2 being classified as mud. The units and a brief description, including colour, based on the Munsell soil colour charts (Geological Society of America, 1994), are outlined below -

a) Sand - predominantly fine to medium sand - sized quartz. Yellowish grey to light brown.

b) Sandy Mud to Muddy Sand - generally a transitional unit from sand to mud. Ranges from predominantly silt and fine to medium sand sized quartz to an abundance of clay minerals such as smectite, illite and kaolinite (Barnett, 1993). Colours include light olive grey, olive grey, light greenish grey, dark greenish grey, medium light grey and medium dark grey.

c) Mud - consisting mostly of clay and silt in varying proportions. The clay fraction is as for unit b), while the silt fraction is mostly quartz. Dark greenish grey to olive grey bands and lenses.

d) Peaty Sandy Mud to Muddy Sand - consists predominantly of clay and silt sized quartz grains with minor mica, zircon and rutile (Barnett 1993). Numerous reeds and cellullose fragments are also included in this unit. Greyish brown to dusky brown.

e) Brownish Sandy Mud to Muddy Sand - similar in lithology to the sandy to muddy sand unit but differing in colour (brownish grey to brownish black).

Sediment descriptions were undertaken by the author for cores 1, 3 and 4, while core 2 was described by Barnett (1993).

6.3.2.2. Dry weight and Loss on Ignition

Material for dry weight and Loss on Ignition (LOI) analyses was extracted at the same intervals as those for diatom sampling for cores 1, 2 and 4 (see table 6.3). Sediment analyses on core 3 were undertaken by Barnett (1993) at a 10 cm sampling interval.

Homogenised sediment was packed into a pre - weighed ceramic crucible with an internal volume of approximately 2 cm^3, and then weighed. Dry weight was determined by drying the sediment overnight at 90°C to its constant weight, then cooled in a desiccator at room temperature before re-weighing. Water content was expressed as a percentage of wet weight. Organic content (as measured by loss on ignition) is often used as an indicator of palaeoproductivity and nutrient availability within a lake, and was determined by loss on ignition after firing each dried sample at 550°C for 2 hours in a muffle furnace. Samples were then cooled and re-weighed as before. LOI was expressed as a percentage of dry weight.

6.3.3 Diatom analyses

Table 6.3 shows the diatom sampling resolution for the four cores. Samples from cores 1, 3 and 4 were taken contiguously, with samples taken as a scrape across the entire interval and then well mixed. Diatom samples were processed using the same method as outlined in section 3.2.3. Up to 500 diatom valves were counted per sample, with a minimum count of 300 valves, which is the minimum number suggested by Battarbee (1986) as a minimum representative count. Broken valves were only counted if the central area was present. Most of the counting was undertaken using an Olympus BH-2 microscope with a Nomaski Differential Interference Contrast component. Occasionally a Zeiss Axioscope was used to gain better resolution when checking identification. The primary diatom floras followed were Krammer and Lange-Bertalot (1986, 1988, 1991a, 1991b), supplemented by Foged (1978), Germain (1981), Archibald (1983), Gasse (1986), and John (1993).

Table 6.3 Sampling intervals for Cores 1, 2, 3, and 4.

Core	Sampling interval
1 - West basin, Lake Alexandrina	1.0 cm sampling from 0 - 30 cm 2.0 cm sampling from 20 - 106 cm
2 - Central channel, Lake Alexandrina	0, 4, 8, 20, 40, 60, 80, 100, 120, 160, 180, 220, 260, 300, 340, 380, 40, 460, and 490 cm
3 - Northern channel, Lake Alexandrina	1.0 cm sampling from 0 - 30 cm 2.0 cm sampling from 30 to 100 cm
4 - Lake Cullulleraine	1.0 cm sampling from 0 - 48 cm

6.4 Results

6.4.1. Dating results

6.4.1.1 ^{14}C results

Table 6.4 details the results of the ^{14}C analyses for cores 1, 2 and 3. The radiocarbon ages are expressed as uncalibrated values, due to the uncertainty about the magnitude of the marine reservoir correction factor that should be applied when dating estuarine materials (Alan Hogg, pers. comm., 2001), and therefore sedimentation rates are an approximation.

Table 6.4: Results of the ^{14}C analyses for cores 1, 2 and 3.

Core	Depth (cm)	^{14}C Age yr BP
1	119 - 125	7610 ± 140
2	50 - 60	130% modern
	122 - 129	110% modern
	251 - 259	1940 ± 140
	251 - 259	2661 ± 240 (AMS)
	402 - 410	4940 ± 250
	492 - 494	7000 ± 300
3	40 – 43	6644 ± 76
	65 - 68	6499 ± 69
	82 - 85	6765 ± 76

The difference in ages provided by the conventional and AMS ^{14}C ages for 251 - 259 cm in Core 2 is maybe due to different fractions or sources of carbon being dated. Barnett (1993) explains this by stating that less than 6 mg of pure carbon was prepared for use in the AMS dating method, whereas up to 2 g of carbon (at 0.95%) of the sediment was used in the conventional dating method. In accordance with Barnett's (1993) conclusions, it was decided to take an average date of 2301 ± 190 years for this level of sediment.

6.4.1.2 ^{210}Pb results

In core 2, excess ^{210}Pb ($^{210}Pb_{ex}$) was calculated from the total ^{210}Pb minus the supported value (1.52 dpm/g), the average of values below 16 cm, where the total ^{210}Pb activity remains relatively stable (standard deviation 0.333 dpm/g) (Barnett, 1993). From the slope of $^{210}Pb_{ex}$ (figure 6.4), Barnett (1993) estimates a sedimentation rate of 1.71 mm/year for the central region. A date of 1946 (the closest ^{210}Pb sampled depth to the onset of river regulation) can then be placed at approximately 7cm (see age – depth model in figure 6.5).

In cores 3 $^{210}Pb_{ex}$ was calculated by subtracting ^{226}Ra values (equivalent to the supported ^{210}Pb) from the total ^{210}Pb (inferred from the granddaughter isotope ^{210}Po). The plot of $^{210}Pb_{ex}$ for core 3, extracted in 1997, is quite variable (figure 6.6). In many ways this may be expected in systems where a substantial amount of the ^{210}Pb is delivered from river catchments (Wasson *et al.*, 1987). In this context, different ^{210}Pb signatures may be derived from variation in the proportion of subsoil (often depleted in $^{210}Pb_{ex}$) erosion and from different areas within the catchment. Changes in the spatial provenance are often more readily detected due to changes in parent radionuclides such as ^{226}Ra or ^{222}Rn (Olley *et al.*, 1997). As can be seen by examination of figure 6.8, there is relatively little variation in the core 3 ^{226}Ra profile, aside from the two lowest values (18-19 and 20-21 cm) which have notably lower ^{226}Ra.

Age-depth relationships were derived using the constant initial concentration model applied to the upper 9 cm, with the sample at 6-7 cm, excluded as an outlier (see figures 6.6 and 6.7). Ages from this part of the record, though probably not exact, may be regarded with reasonable confidence. The only other ^{210}Pb age which may derived is that where $^{210}Pb_{ex}$ falls below detection levels (0.001 dpm/g at the ANSTO laboratory, Heijnis pers. comm.). The age this value represents varies dependent on the initial $^{210}Pb_{ex}$ concentration in the sediment at each site and, thus the number of half-lives expired to reach below detection limits. A corollary is that this age can only be determined on sediments which have similar ability to scavenge radioisotopes.

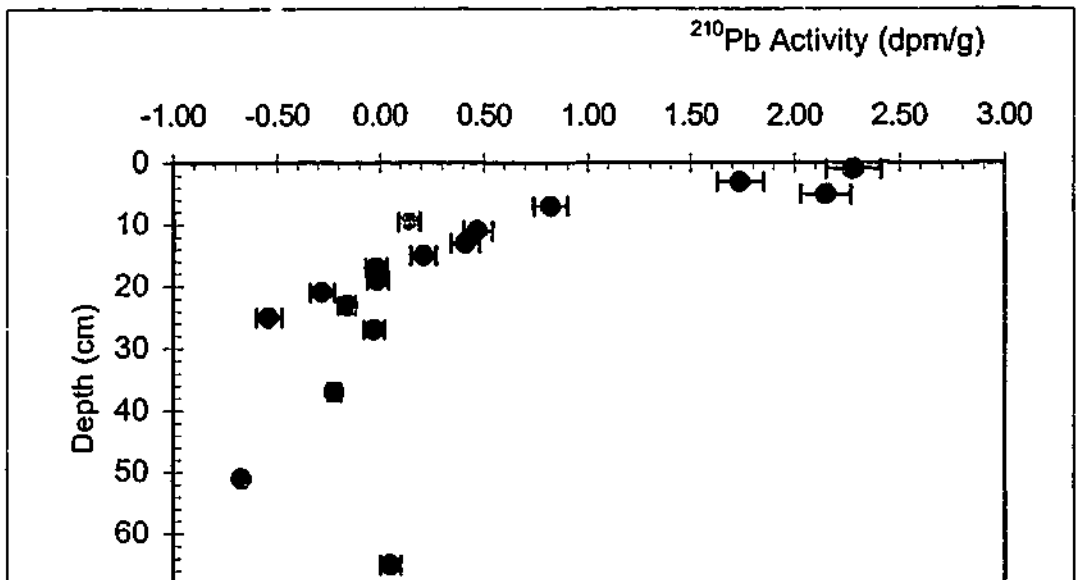


Figure 6.4. Excess ^{210}Pb from Lake Alexandrina core 2. The grey data point was excluded from calculation of the age-depth model. See section 6.4.1.2 for details regarding the calculation of these values.

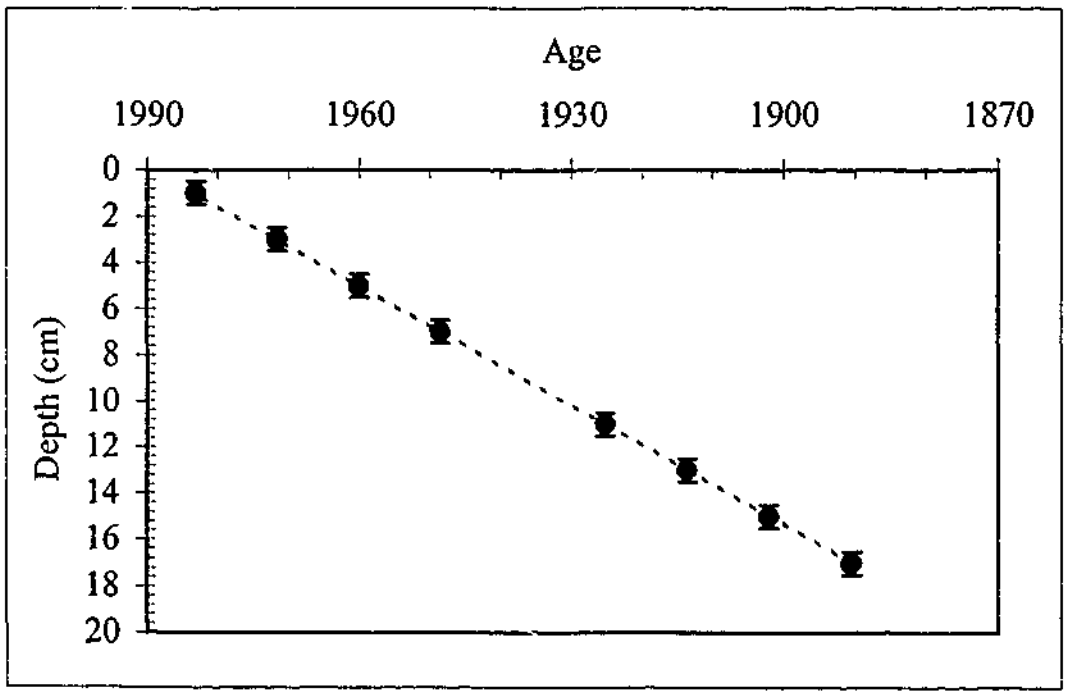


Figure 6.5. Age-depth relationships derived from excess ^{210}Pb decay in core 2. This curve was derived from the overall slope of the excess ^{210}Pb profile (with the 9-10 cm sample omitted), rather than from estimation of ages for individual samples. Age error bars, which are at times smaller than the symbols, are not displayed.

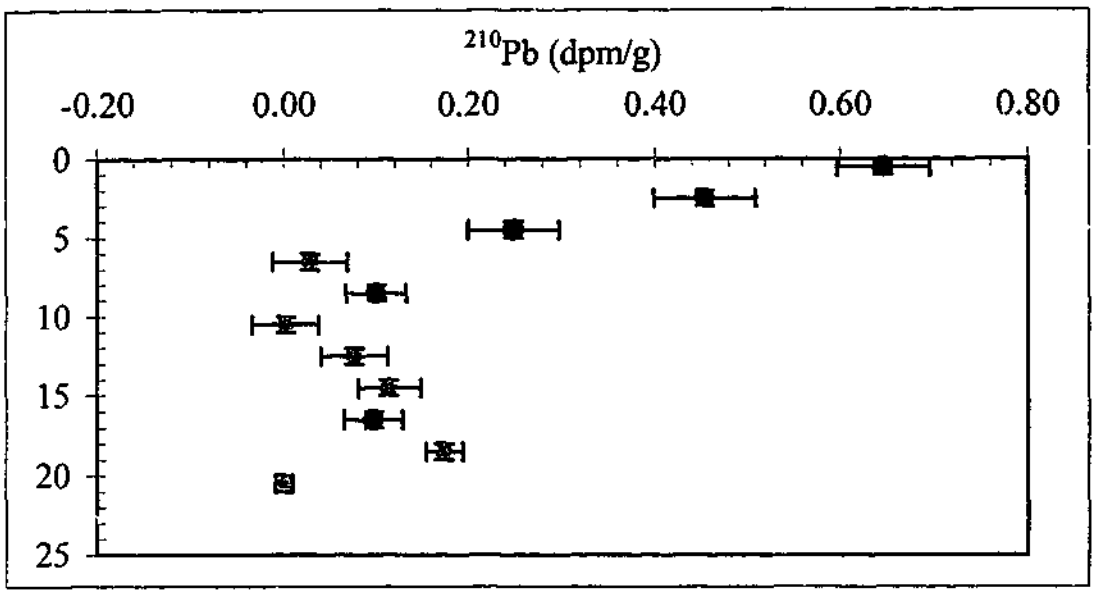


Figure 6.6. Excess ^{210}Pb for core 3 (linear scale). See section 6.4.1.2 for details regarding the calculation of these values. Note that only enlarged, black, data points were used to derive the age-depth model.

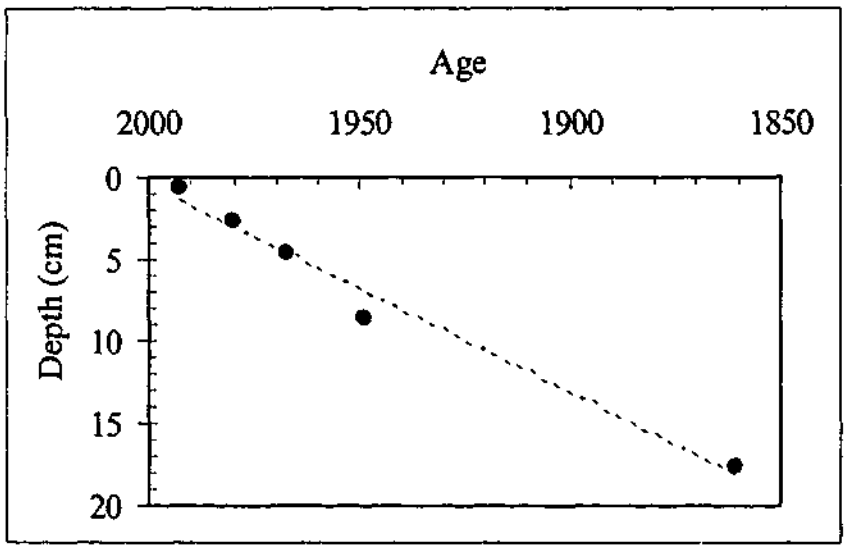


Figure 6.7: Age-depth relationship for core 3. See section 6.4.1.2 for details of how relationship has been derived.

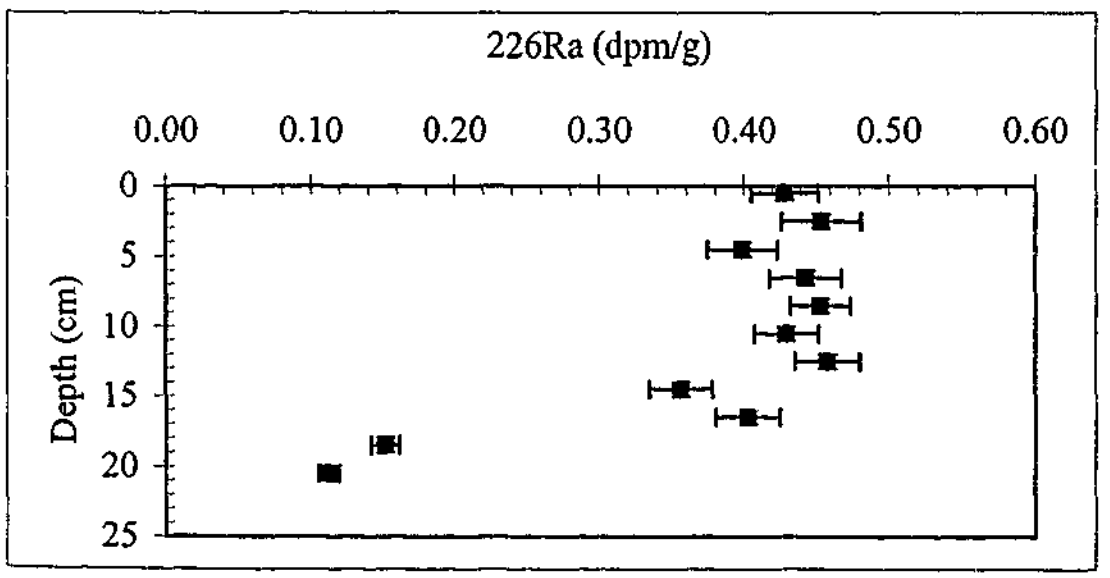


Figure 6.8. ^{226}Ra values for core 3.

Hence, in core 3, the bottom two samples (18-19 and 20-21 cm) cannot be utilised as they have considerably lower ^{226}Ra than the sediments above. Thus, for the purpose of constructing an age-depth model, the core 3 detection limit for $^{210}Pb_{ex}$ is assumed to occur just below 16-17 cm (at 17.5 cm). The age these sediments were deposited is therefore approximately 139 years before extraction.

In core 4 (Lake Cullulleraine) $^{210}Pb_{ex}$ was calculated by subtracting ^{226}Ra values (equivalent to the supported ^{210}Pb) from the total ^{210}Pb (inferred from the granddaughter isotope ^{210}Po). In core 4, ^{226}Ra activity did not vary substantially (standard deviation 0.115). This, combined with an exponential decay of $^{210}Pb_{ex}$ (figure 6.9) and a basal age determination which largely coincides with the formation of the lake, provides a relatively high degree of confidence in the derived age-depth model (figure 6.10).

6.4.1.3 Pinus dating

There were no preserved *Pinus* spores in the samples prepared for pollen analysis from cores 1 and 3. This conclusion was verified by three experienced palynologists. This was an unexpected result because of the large catchment area of the lake and also the presence of extensive pine plantations within a 50 km radius of the lake (particularly on the Fleurieu Peninsula). The absence of *Pinus* pollen did not appear to be a result of poor preservation as there were many other types of pollen grains and spores present in the samples. This result may be due to inappropriate preparation techniques (including problems with sieving), or it may cast doubt on the regional fetch of *Pinus* spores. It could also be a consequence of poor sinking capabilities of the spores. A study on differential floating dynamics of pollen spores (Hopkins, 1950), found that *Pinus* remained buoyant in the water column for longer periods of time than other common pollen types, tending to be deposited at the extremities of experimental floatation tanks.

6.4.2 Lithostratigraphy

6.4.2.1 Core 1 (Lake Alexandrina)

The lithostratigraphy of core 1 is illustrated in figure 6.11. The sediment is composed of dark greenish grey mud with greenish grey to medium grey irregular bands (5 GY 4/1, 5 GY 6/1 and N5). There are lighter greenish grey bands at 35.5, 47 to 47.5, 80.8 to 81.2, and 84 cm, and darker bands at 72 and 89.5 to 90 cm. Water content shows a small increase with depth, ranging from 59% at the surface sediments to 77% at 44 cm, before decreasing to 60% at the core base (110 cm). Organic matter does not exceed 2.0%, and displays an opposite trend to that of water content by generally decreasing with depth.

6.4.2.2 Core 2 (Lake Alexandrina)

The lithostratigraphy of core 2 is illustrated in figure 6.12. The surface sediments to 64 cm are composed of a light grey to medium dark grey olive mud (5 Y 5/2 N6 to N4). From 64 cm to 79 cm there is a light grey to medium dark grey (5 Y 4/1 to N4) sandy mud to muddy sand. From 79 cm to 100 cm the sediment is mostly medium bluish grey, dark greenish grey, olive grey and medium dark grey mud (5 B 5/1, 5 GY 4/1, 5 Y 4/1),

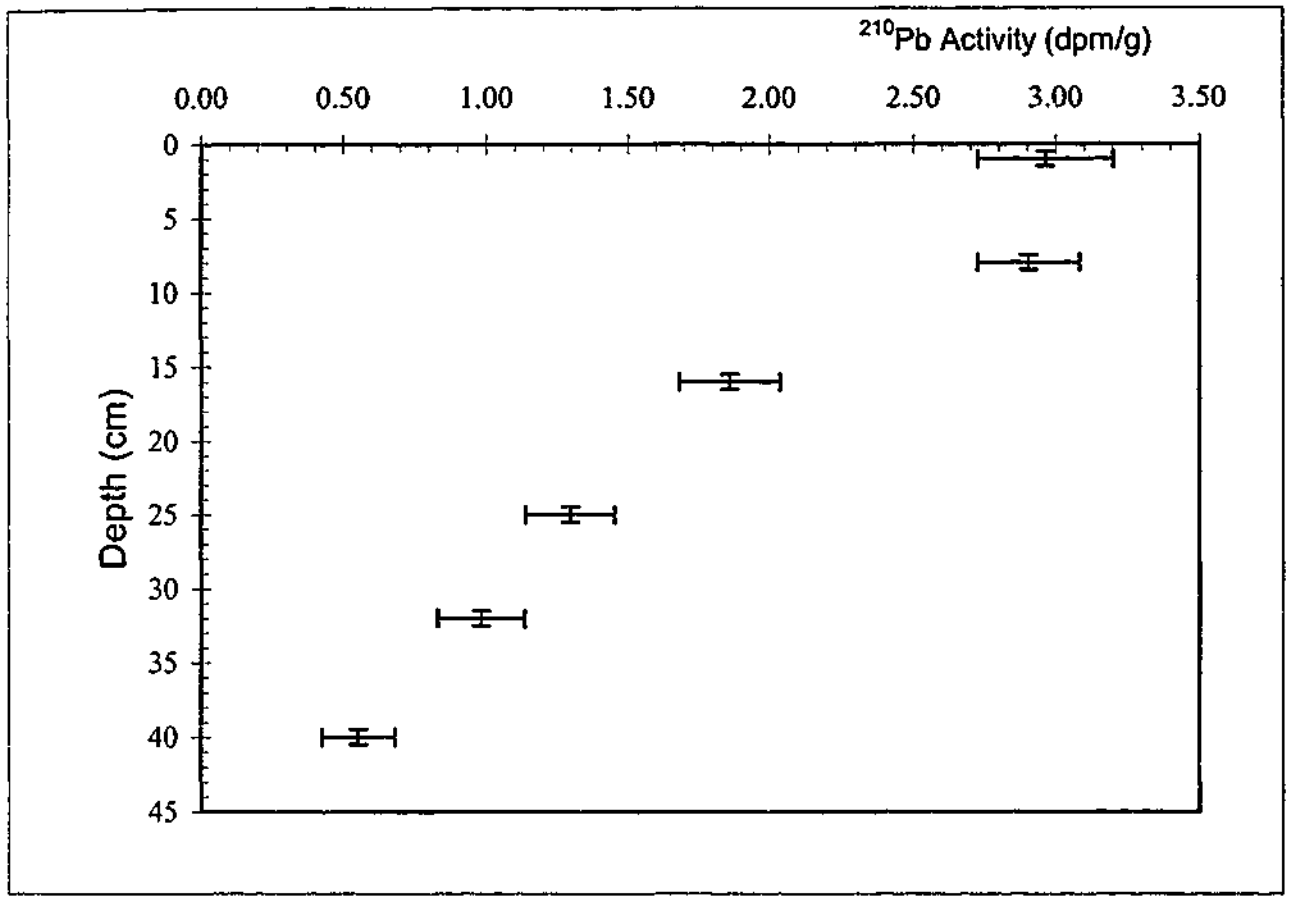


Figure 6.9: Excess ^{210}Pb from the Lake Cullulleraine core.

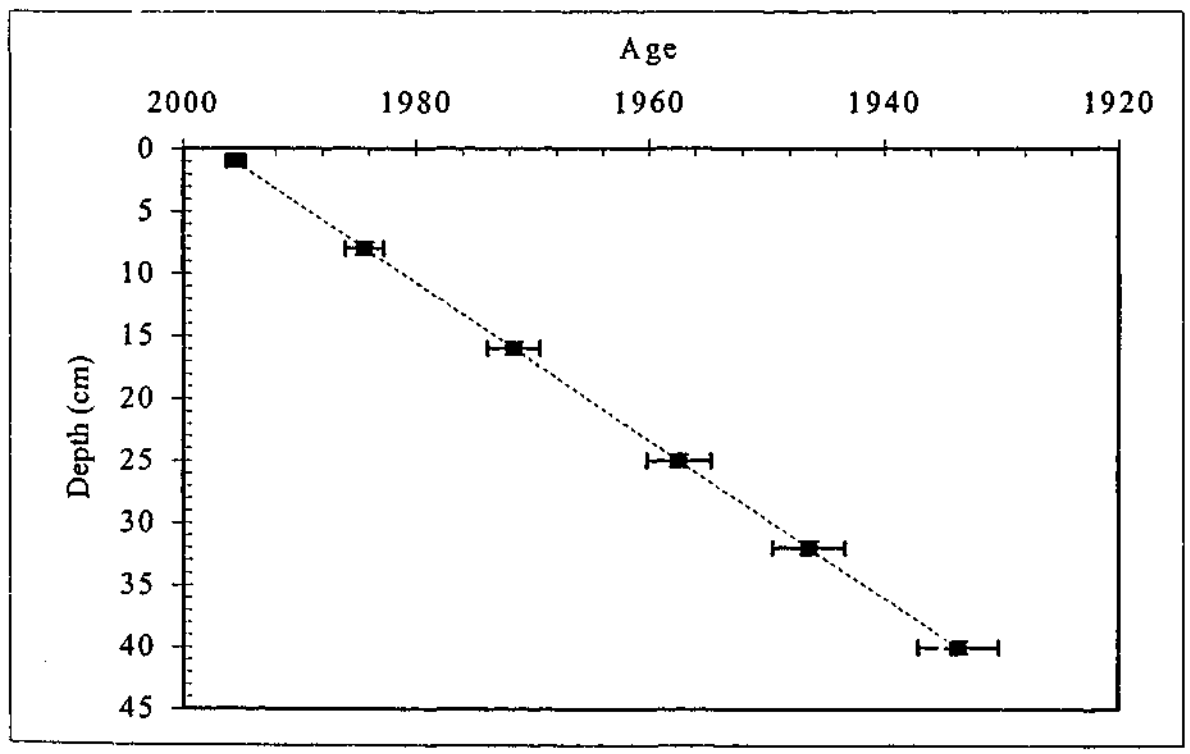


Figure 6.10: Age-depth relationship for Lake Cullulleraine core based on excess ^{210}Pb data.

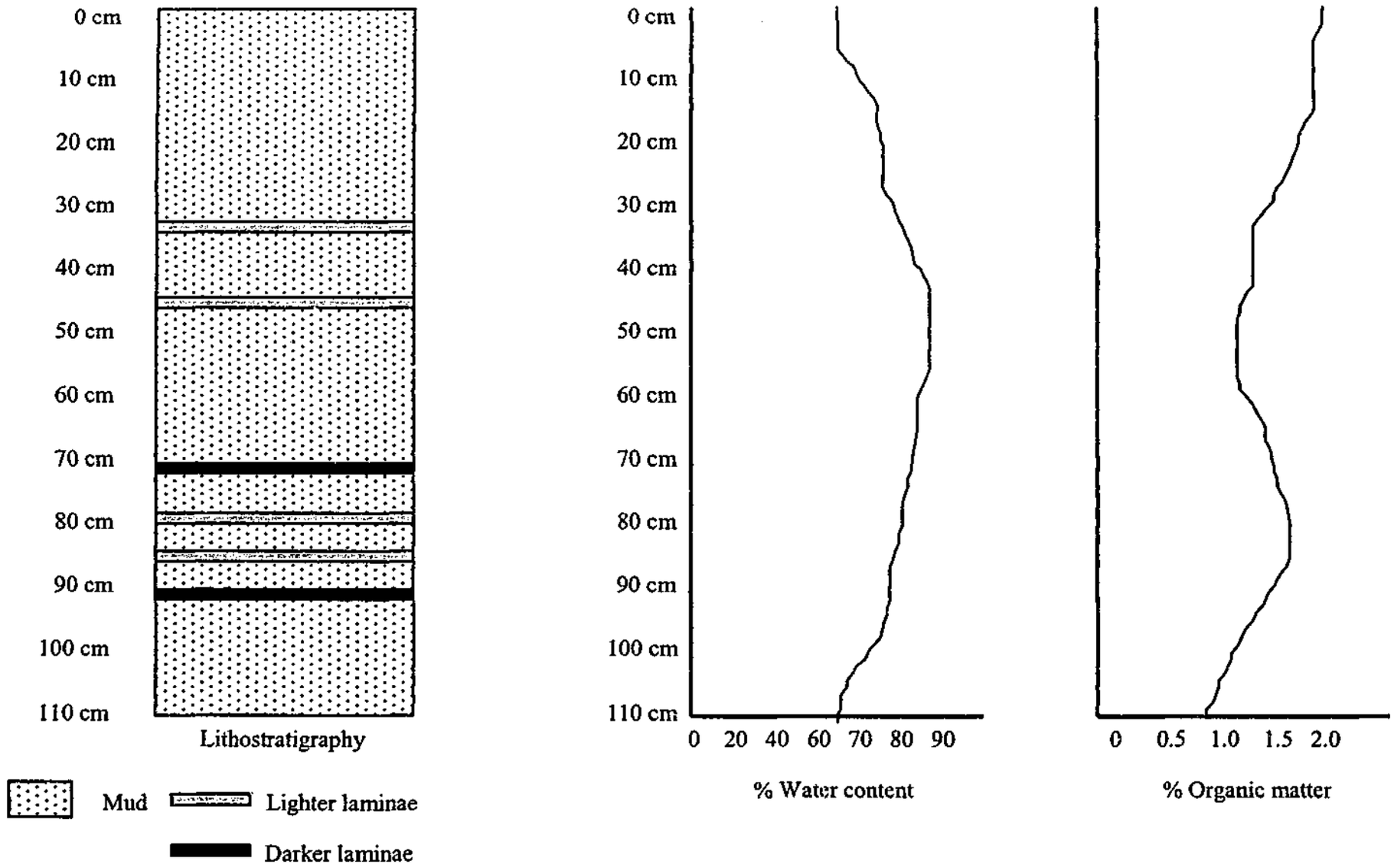


Figure 6.11: Lithostratigraphy, water content and organic matter content of Core 1, Lake Alexandrina

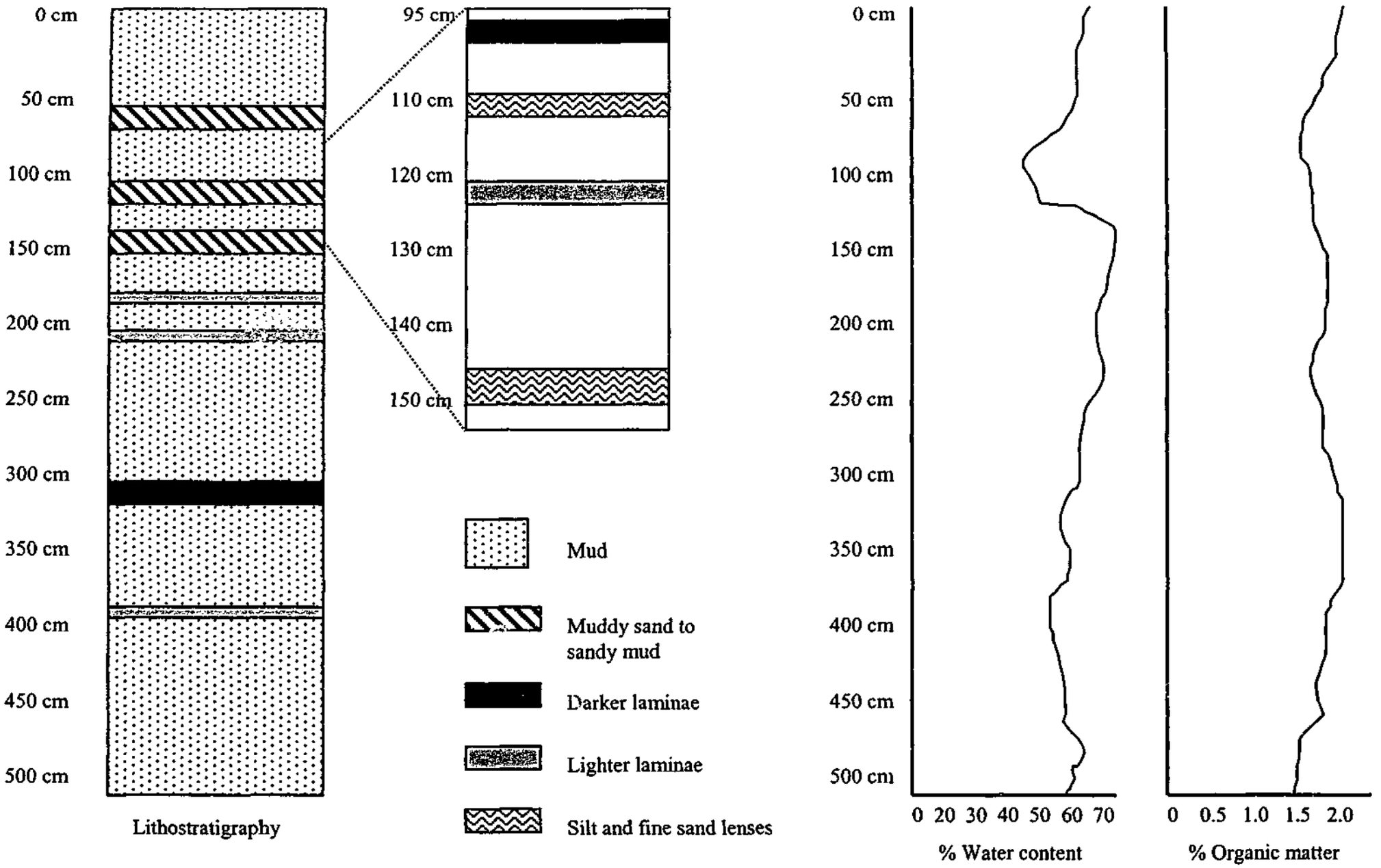


Figure 6.12: *Lithostratigraphy, water content and organic matter content for Core 2, Lake Alexandrina (only the major laminae and lenses are included)*

with medium dark grey bands at 96 to 100 cm (5 Y 4/1). Between 110 cm to 123 cm there is light olive grey to medium dark grey (5Y 6/1 to N4) sandy mud to muddy sand, with silt and fine sand irregular bands from 110 cm to 113 cm. Between 121 cm to 123 cm there are lighter greenish grey to olive grey (5 GY 6/1 and 5 GY 4/1) mud bands. Dark greenish grey to olive grey mud (5 GY 4/1 to 5 Y 4/1) is present between 123 cm and 138 cm. From 138 cm to 158 cm there is light olive grey to darkish greenish grey (5 Y 6/1 to 5 GY 4/1) sandy mud to muddy sand, with irregular bands of silt and fine sands at 146 cm to 150 cm. From 158 cm to the base of the core the sediment is composed of a relatively uniform dark greenish grey, olive grey, and medium light grey to dark grey (5 GY 4/1, 5 Y 4/1, N5 and N4). There are also light greenish grey to olive grey to olive grey (5 G 6/1 to 5 Y 4/1) bands at 185 to 190 cm and 214 to 218 cm; light grey (N6) bands at 405 to 408 cm, dark greenish grey to olive grey (5 GY 4/1 to 5 Y 4/1) fine sand irregular bands at 160 to 164 cm, olive grey (5 Y 4/1) silt rich mud bands at 306, 306.5, 309, 323, 332, and 343 cm, dark grey (N3), striations between 290 and 302 cm, crosscutting bands from 306 to 350 cm and between 440 to 446 cm, and a dark grey (N3) mud layer between 310 to 321 cm with faint veins included. There are also *Corbiculina* shells present from 160 cm to the bottom of the core, and crustacean fragments at 256 and 462 cm.

Water content is approximately 60% in the surface sediments and then decreases sharply at 60 cm. After reaching a low of 39% at 110 cm, it then increases to between 60 and 70%, remaining relatively constant for the rest of the core. Organic matter is highest in the surface sediments at 2.15% and then falls to < 2.0% downcore. There is a small decrease in organic matter at the same time as the sharp decrease in water content.

6.4.2.3 Core 3 (Lake Alexandrina)

The lithology of Core 3 is illustrated in figure 6.13. The sediment is predominantly olive grey mud (5 Y 4/1 to 5 GY 4/1) from the sediment surface to 50 to 60 cm depth. There is then a gradual change between 50 and 60 cm to sandy muds to muddy sands which continue to the base of the core. This layer of sediment is also olive grey although slightly darker than the overlying muds. There are two darker coloured, coarse textured bands present at 55 - 56 cm, and 78 - 80 cm. Water content decreases slightly downcore, from 59% at the surface to 40% at 100 cm. Organic matter also decreases downcore with 3.5% at the surface and 0.5% at 100 cm.

6.4.2.4 Core 4 (Lake Cullulleraine)

This core is 48 cm in length with two very distinct types of sediment, as presented in figure 6.14. The surface to 46 cm is dark greenish grey mud (5 Y 4/1), while 46 - 48 cm is gravelly, coarse sand. The transition between the sediment types is very sharp, as are the changes in water content and organic matter content. Between 0 and 46 cm water content ranges between 60 and 75% while between 46 and 48 cm it is less than 30%. The shift in organic matter content is as abrupt as that of water content, with greater concentrations in the mud section of the core (average of 25% compared to 12%).

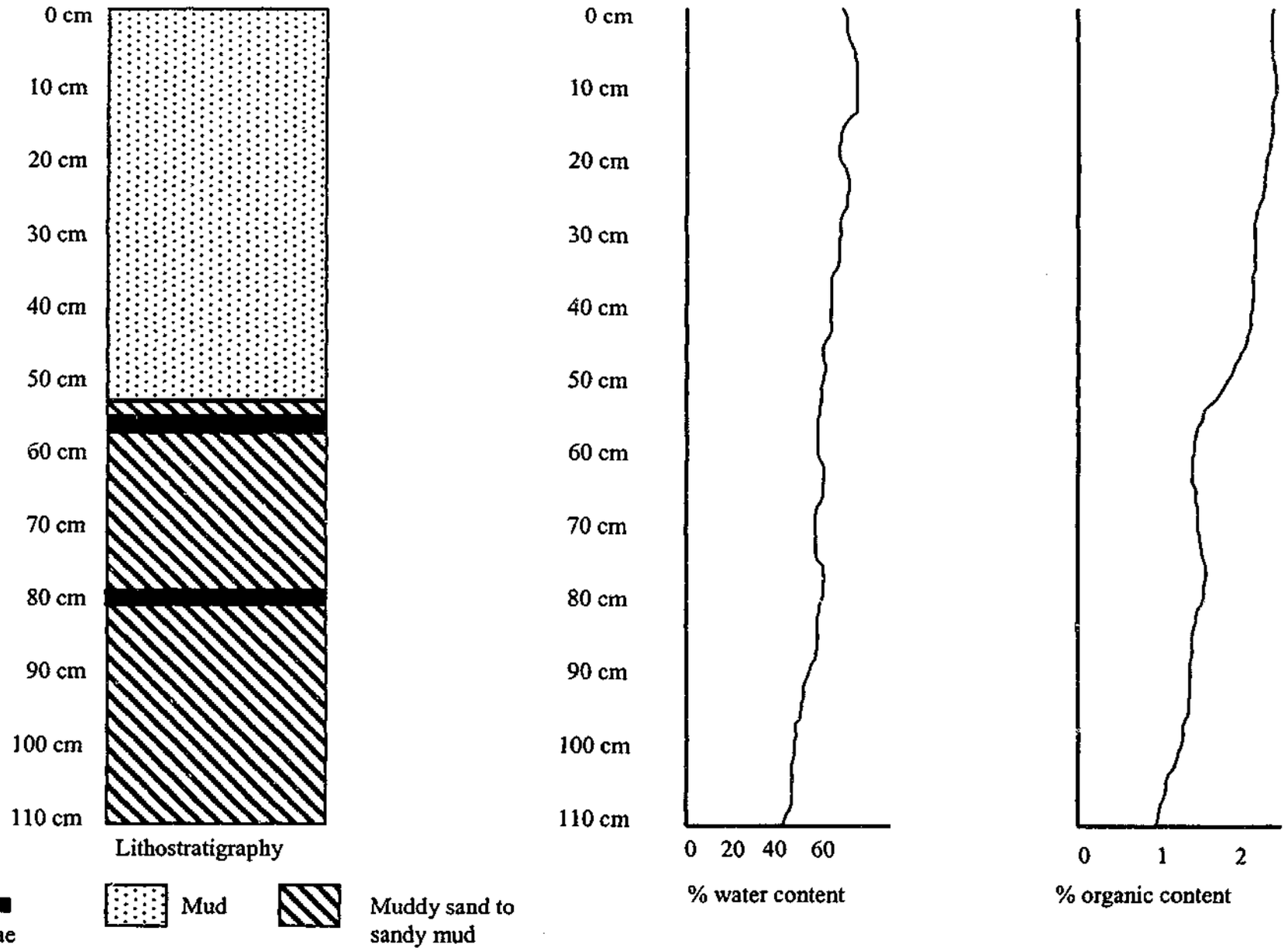


Figure 6.13: Lithostratigraphy, water content and organic matter content of Core 3, Lake Alexandrina.

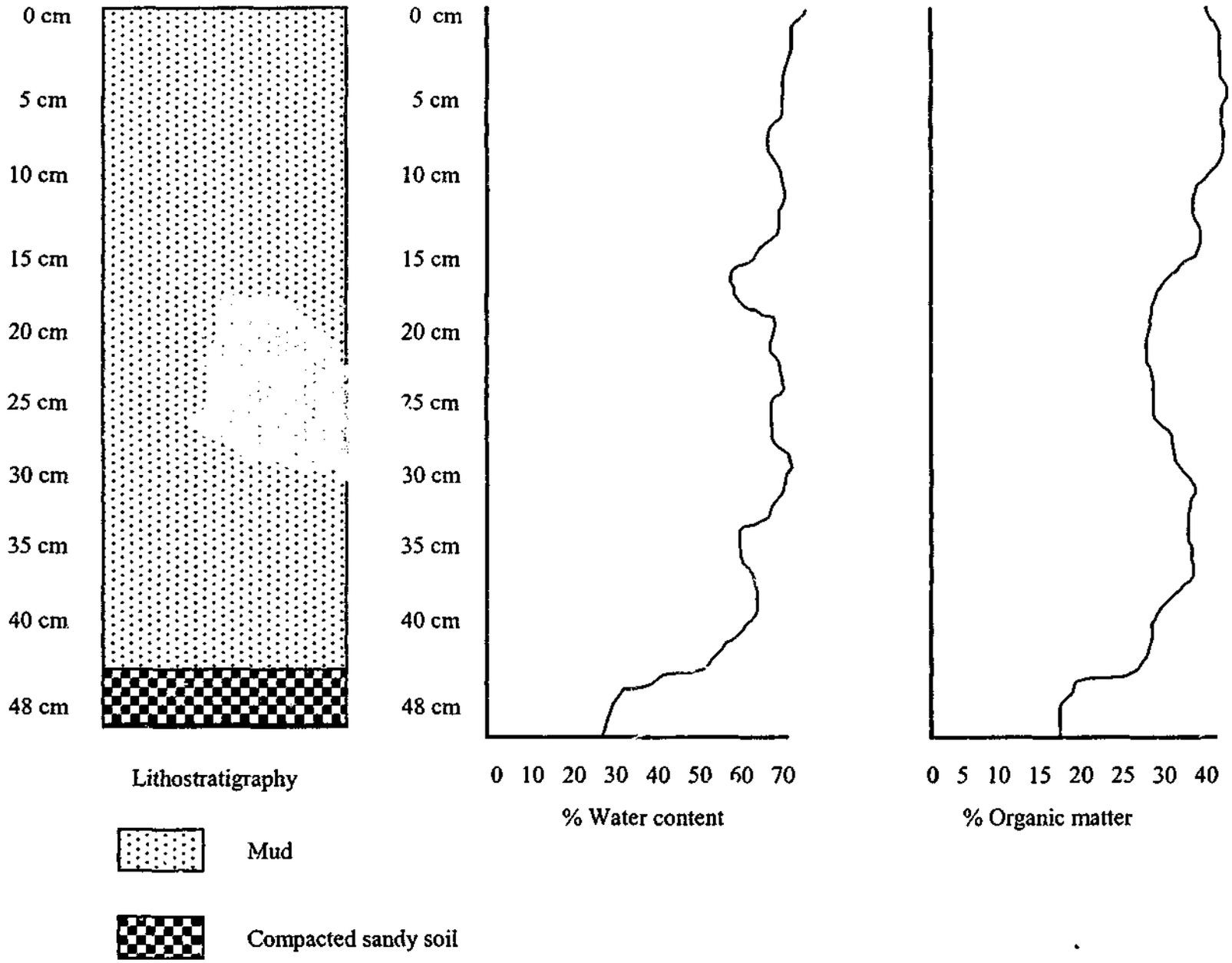


Figure 6.14: Lithostratigraphy, water content and organic matter content for Core 4, Lake Cullulleraine.

6.4.3 Fossil diatom analysis

All fossil relative abundance diatom data are contained in Appendix 8. In each record (figure, 6.15 – 6.18) only the most abundant taxa for each zone (taxa that occur > 5% in the fossil record) and / or those taxa that experience a shift in abundance, either prior or subsequent to this zone, are discussed. Average abundances of diatoms from different habitat types are also provided (calculated from the total species present not just those graphed). CONISS classification (Grimm, 1987), using Euclidean distance, was applied to all four cores on taxa ≥ 1%.

6.4.3.1 Core 1 (Lake Alexandrina)

Core 1, located in the west of the central basin, only had preserved diatoms in the top 24 cm of sediment. Additionally, within this 24 cm there were two sections where diatoms were absent, between 13 and 14 cm and 17 to 19 cm. The dominant diatoms in this core were the planktonic *Aulacoseira granulata* and the facultative planktonic *Staurosirella pinnata*. With the aid of CONISS classification, four zones were identified. The diatom diagram, illustrating the major taxa, is presented in figure 6.15.

Zone 1D (24 - 18 cm)

Planktonic taxa account for an average of 67% of the diatom assemblage in this zone, while facultative planktonic taxa account for 24%. This zone is composed of three samples, and is dominated by *Aulacoseira granulata* (average abundance of 52%). However, this taxon's abundance is not constant throughout the zone, accounting for > 95% of the assemblage at the base of the zone and then decreasing to < 25% at 20 cm. *Aulacoseira granulata* is succeeded by *Staurosirella pinnata*, which although absent in the bottom sample, accounts for nearly 40% of the assemblage at the top of the zone. Minor taxa present in this zone include *Staurosira construens* forma *venter*, which displays a similar pattern to *S. pinnata*, *Tryblionella compressa* and *Pinnularia borealis*.

Zone 1C (18 - 14 cm)

This zone is comprised of two samples which lie inbetween two zones of diatom discontinuity. Planktonic taxa increase to 78% in this zone, while facultative planktonic taxa decrease to 2%. *Aulacoseira granulata* returns to dominance in this zone, accounting for 60% at 16 cm and 86% of the assemblage at 15 cm. As with the previous zone, this is countered by a decrease in *Staurosirella pinnata*, which is present only at 15 cm (< 2%). *Tryblionella compressa* peaks in this zone, reaching an abundance of 13% at 16 cm, as does *Campylodiscus clypeus* (maximum of 7%).

Zone 1B (14 - 7 cm)

The diatom assemblages within this zone are quite variable, although there is a general decrease in planktonic taxa (to 50%) and an increase in facultative planktonic taxa (to 42%). *Aulacoseira granulata* is subdominant

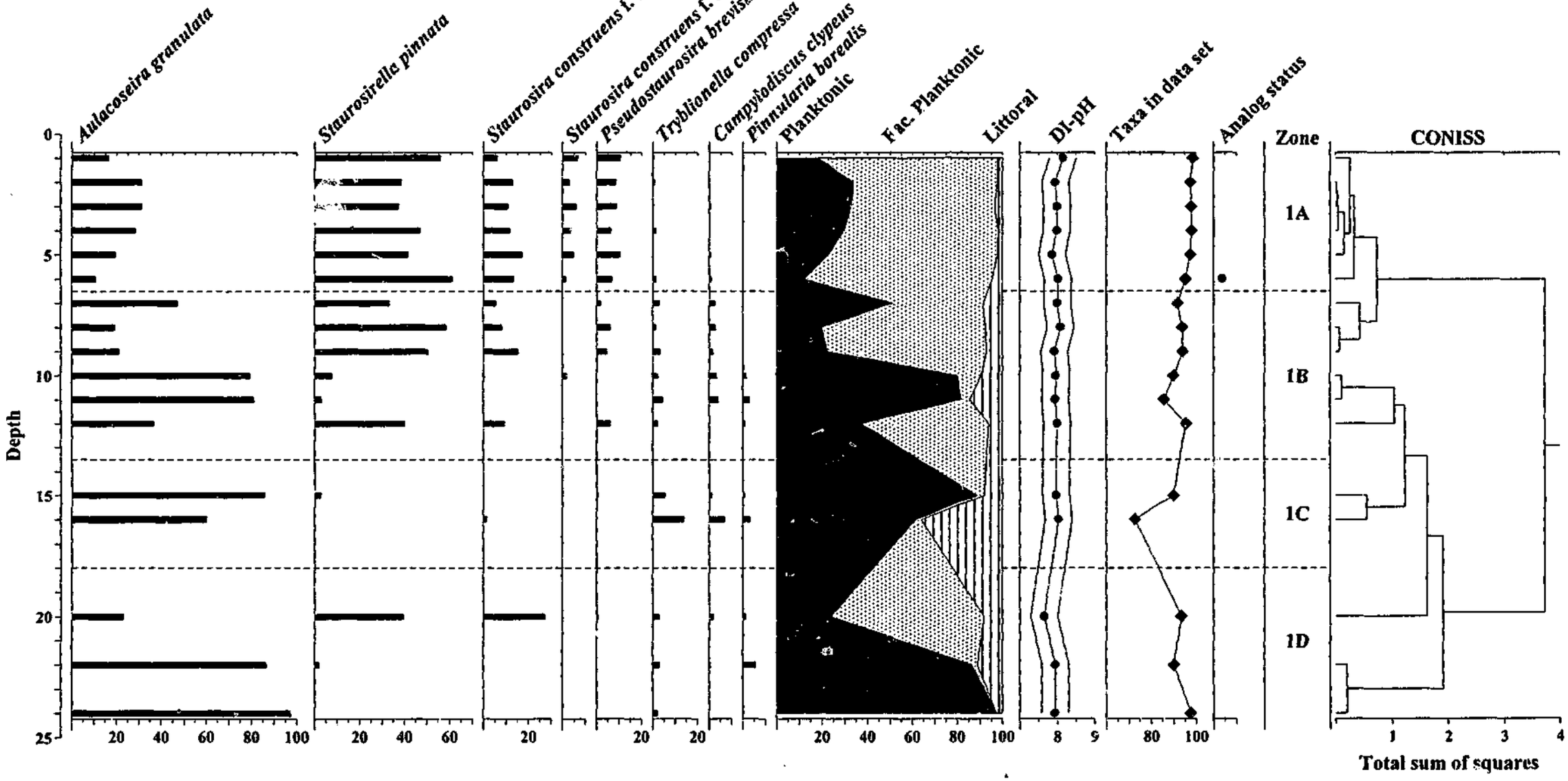


Figure 6.15: Core 1 diatom stratigraphy. The pH reconstruction and associated positive and negative root-mean-squared-error of prediction (RMSEP) is included. Also shown is the total proportion of diatoms found in the lake data set. Samples with no symbol and "•" have, respectively, good and fair analogues in the modern data set (see Jones and Juggins, 1995).

at 37% at 12 cm, then increases to > 80% between 10 and 12 cm, decreases to < 20% at 8 cm, and then increases again to 47% at 7 cm. Abundance of *Staurosirella pinnata* shows an inverse pattern to that of *A. granulata*. *Pseudostaurosira brevistriata* makes its first consistent appearance in this zone, accounting for an average of 3%, and showing a similar pattern to that of *S. pinnata*. *Campylodiscus clypeus* has low but generally consistent values in this zone, while *Tryblionella compressa* has low but generally consistent representation.

Zone 1A (7 - 0 cm)

This zone is dominated by facultative planktonic taxa with *Staurosirella pinnata*, *Staurosira construens* forma *venter*, *Pseudostaurosira brevistriata* and *Staurosira construens* var. *construens* jointly accounting for over 70% of all taxa. The remainder of the assemblage is represented almost solely by the planktonic *Aulacoseira granulata*, averaging 22.5% and peaking at 31% at both 2 and 3 cm. *Campylodiscus clypeus* and *Tryblionella compressa* decrease to negligible concentrations (< 1.0). It is worth noting that there is a comparable lack of variability within the zone.

Summary

Out of a total core length of 100 cm, only the top 24 cm contained any diatoms, and even this section contained sections of diatom discontinuity. The bottom 2 cm of the section of core containing diatoms are dominated by *A. granulata*, with a shift to *S. pinnata* dominance between 20 and 22 cm. There is then absence of diatoms between 20 and 17 cm, before a return to a *A. granulata* dominated assemblage between 15 and 17 cm. The remainder of the core is variable, with continued switches between these two taxa, although the diatom assemblage appears to stabilise towards the top of the core, becoming increasingly dominated by *S. pinnata*. Diatom inferred pH (DI – pH) reconstructions for core 1 are also presented in figures 6.15. Root mean square errors of prediction are represented by the solid lines either side of the reconstruction. There is little change in DI – pH for core 1, with reconstructions between 7.9 and 8.1 pH units for all depths, except 20 cm, which has a DI – pH of 7.6.

6.4.3.3 Core 2 (Lake Alexandrina)

Core 2 is dominated primarily by planktonic and facultative planktonic taxa. The major planktonic taxa present are *Aulacoseira granulata*, *Cyclotella striata*, and *Thalassiosira lacustris*. The major facultative planktonic taxa present are *Staurosirella pinnata, Pseudostaurosira brevistriata* and *Staurosira construens* forma *venter*. With the aid of CONISS, four zones were identified. The diatom stratigraphy for core 2 is presented in figure 6.16.

Zone 2D (490 - 240 cm)

The degree of dissimilarity between most samples combined with the high level chaining indicates that this is a very variable zone, but one which is not easy to subdivide without separation of individual samples. With the exception of the bottom sample, dominated by *Cocconeis placentula* (28.6%), the major taxon in this zone is *S. pinnata* (average of 17%). The next most common taxon is the planktonic *Thalassiosira lacustris*,

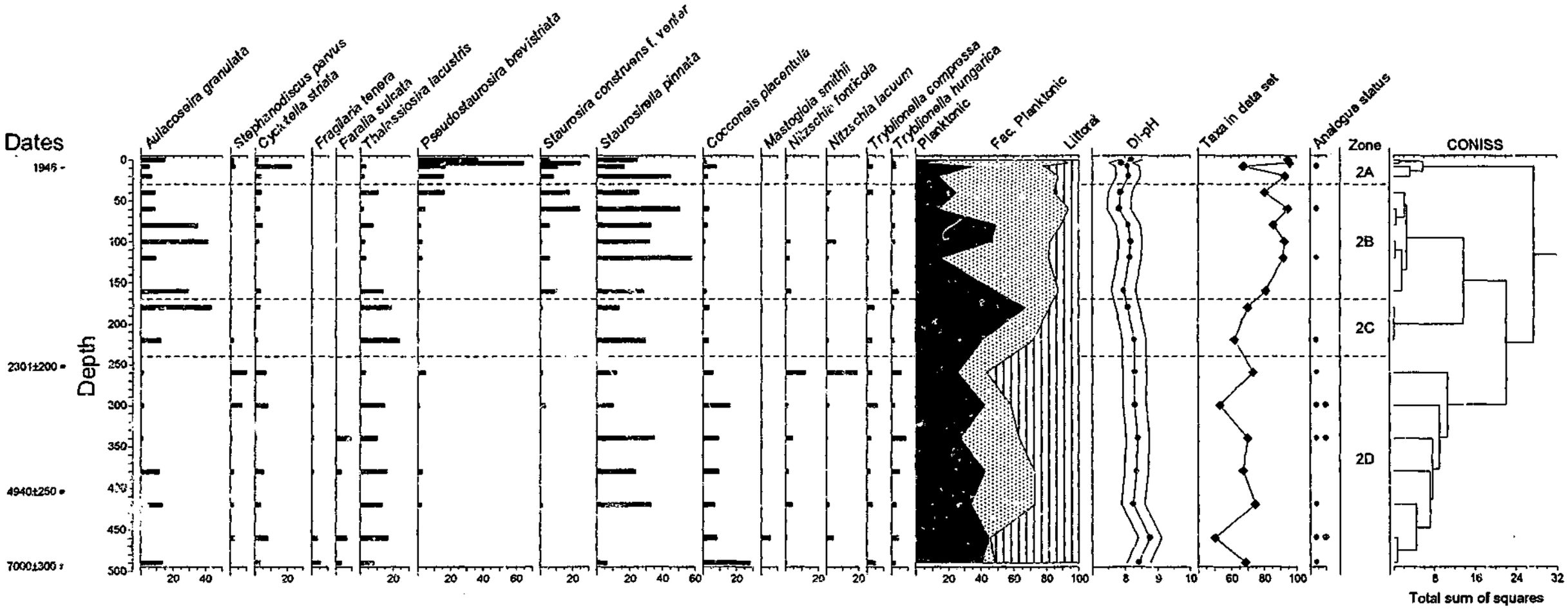


Figure 6.16: Core 2 diatom stratigraphy. The pH reconstruction and associated positive and negative root-mean-squared-error of prediction (RMSEP) is included. Also shown is the total proportion of diatoms found in the lake data set. Samples with no symbol, "•" and "••" have, respectively, good, fair and poor analogues in the modern data set (see Jones and Juggins, 1995).

which has an average abundance of > 11%. There are three other minor planktonic taxa present; *Aulacoseira granulata*, *Cyclotella striata* and *Stephanodiscus parvus*. *Aulacoseira granulata* has its highest abundances at 490 cm, 420 cm, and 380 cm (all > 11%), and lowest values of < 2% at depths of 340 cm, 300 cm, and 260 cm. *Cyclotella striata* has an average abundance of 5% throughout the zone, peaking at 8% at 460 cm. *Stephanodiscus parvus* has an average abundance of < 3%, increasing in abundance towards the top of the zone. Littoral taxa are prevalent in this zone, particularly *C. placentula*, *Paralia sulcata*, *Nitzschia fonticola*, and *Nitzschia lacuum*. After peaking in the bottom sample, *C. placentula* remains relatively constant in abundance, averaging 10%. *Paralia sulcata* does not display any particular patterns and is present at < 10% in all samples. Both *N. fonticola* and *N. lacuum* peak in abundance at the top of the zone, accounting for 11.7% and 18.8% respectively.

Zone 2C (240 - 170 cm)

This zone consists of two samples only, at 220 cm and 180 cm depth. It appears that CONISS clusters this section of the core as a distinct zone because of an increase in the abundance of *T. lacutris*, which accounts for 24% of the diatom assemblage at 220 cm and 18% at 180 cm, and also *A. granulata* which increases to 12% and 43% respectively. The other common planktonic taxon, *C. striata*, decreases in this zone, however, averaging < 3%. *Cocconeis placentula* also decreases in abundance in this zone, averaging < 3%. There is little change in *S. pinnata* from its abundance in zone 2D. *Paralia sulcata* is absent from this zone, while *N. fonticola* and *N. lacuum* decrease to an average of < 1% abundance.

Zone 2B (170 - 30 cm)

This zone consists of 6 samples: 160 cm, 120 cm, 100 cm, 80 cm, 60 cm and 40 cm. The most abundant taxon from zone 2C, *A. granulata*, decreases in abundance at 120 cm (to 9%), increases again to > 35% at 100 and 80 cm depths, before decreasing at 60 and 40 cm depth to < 9%. *Thalassiosira lacustris* declines considerably in this zone, falling to below < 2% at 60 cm. However, it should be noted that at 80 cm and 40 cm this taxon increases to 8% and 11% respectively. *Staurosirella pinnata* peaks in abundance at 120 cm accounting for 51% of the total assemblage, with the average abundance increasing to 38% (from 24% in zone 2C and 17% in zone 2D). *Staurosira construens* forma *venter* increases in abundance in this zone, with highest representation in the top 2 samples, while *P. brevistriata* has low but consistent representation until a sharp increase to 16% in the top sample of the zone.

Zone 2A (30 - 0 cm)

This zone is comprised of four samples, at 0 cm, 4 cm, 8 cm, and 20 cm, with most diatoms being facultative planktonic (average of 69.4%) in origin. Of these facultative planktonic taxa, *P. brevistriata* is the most abundant, accounting for 36.6% of the total count at the surface, 64.4% at 4 cm, 16.4% at 8 cm and 15.8% at 20 cm. *Staurosirella pinnata* reaches a maximum of 44.8% at 20 cm, while *S. construens forma venter* reaches a maximum of 24.8% at 4 cm. Of the planktonic taxa, *A. granulata* occurs at an average of 6.5% throughout the zone (lower than its average abundance from zone 2) and *C. striata* occurs at < 5% at all

depths with the exception of 8 cm depth where it is peaks at 22.2%. *Thalassiosira lacustris* is also present at < 5%, consistently lower than in earlier zones.

Summary

The basal three metres of this core are dominated by the planktonic *A. granulata* and *T. lacustris,* the facultative planktonic *S. pinnata*, and the littoral *C. placentula. Thalassiosira lacustris* shows a decrease in abundance from 160 cm to the surface sediments, while *A. granulata* increases from 180 cm depth to approximately 80 cm, and then decreases towards the surface. *Cocconeis placentula* and other littoral taxa decrease in abundance towards the top of the core. The surface sediments of this core are dominated by the facultative planktonic taxa *Pseudostaurosira brevistriata, Staurosirella pinnata* and the planktonic *Aulacoseira granulata*. Perhaps the most dramatic change in the uppermost sediments is the peak in *P. brevistriata* at 4 cm (64.7%).

Diatom inferred pH is slightly more dynamic in core 2, with estimates ranging between 7.8 to 8.7. There is a small decrease in pH towards the surface, particularly after 220 cm, and then again after 80 cm.

6.4.3.3 Core 3 (Lake Alexandrina)

A diagram illustrating the abundances of the most common diatom taxa in Core 3 is shown in figure 6.17. The most common groups of diatom species are littoral and facultative planktonic. The most abundant diatom species are *Cocconeis placentula* (littoral), *Epithemia adnata* (littoral) and *Staurosirella pinnata* (facultative planktonic). There are five sections of the core where there are no diatoms present, between 17-20 cm, 39-44 cm, 53-55 cm, 77-80 cm, and 85-100 cm.

Zone 3G (85 - 78 cm)

Littoral and facultative planktonic taxa are present in equal abundances in this zone (average of > 46%), with only a minor presence of planktonic taxa (7%). The dominant taxon is *Staurosirella pinnata,* which has an average abundance of 36%. The littoral taxa *Epithemia adnata* and *Cocconeis placentula* are the next most abundant with averages of 16% and 7% respectively. Minor taxa, including *Staurosira construens* forma *venter*, *Pseudostaurosira brevistriata* and *Rhopalodia gibba,* are also consistently present. There are no diatoms present between 80 and 77 cm.

Zone 3F (78 - 55 cm)

Littoral taxa decrease to an average of 39.1% in this zone (from 46.3% in Zone 3G), resulting mainly from a decrease in *E. adnata* from an average of 16% in zone 3G to 9.5% in this zone. *Rhopalodia gibba* gradually increases towards the top of the zone, reaching its maximum abundance for the entire core of 68% at 56 cm. An increase in *S. pinnata* abundance from an average of 36% in zone 3G to 44% in this zone, contributes to an overall increase in facultative planktonic taxa to 52.7% (from 46.3% in the previous zone). *S. pinnata* abundance is highly variable within the zone, however, being present at 44% at 76 cm, increasing to 73% at

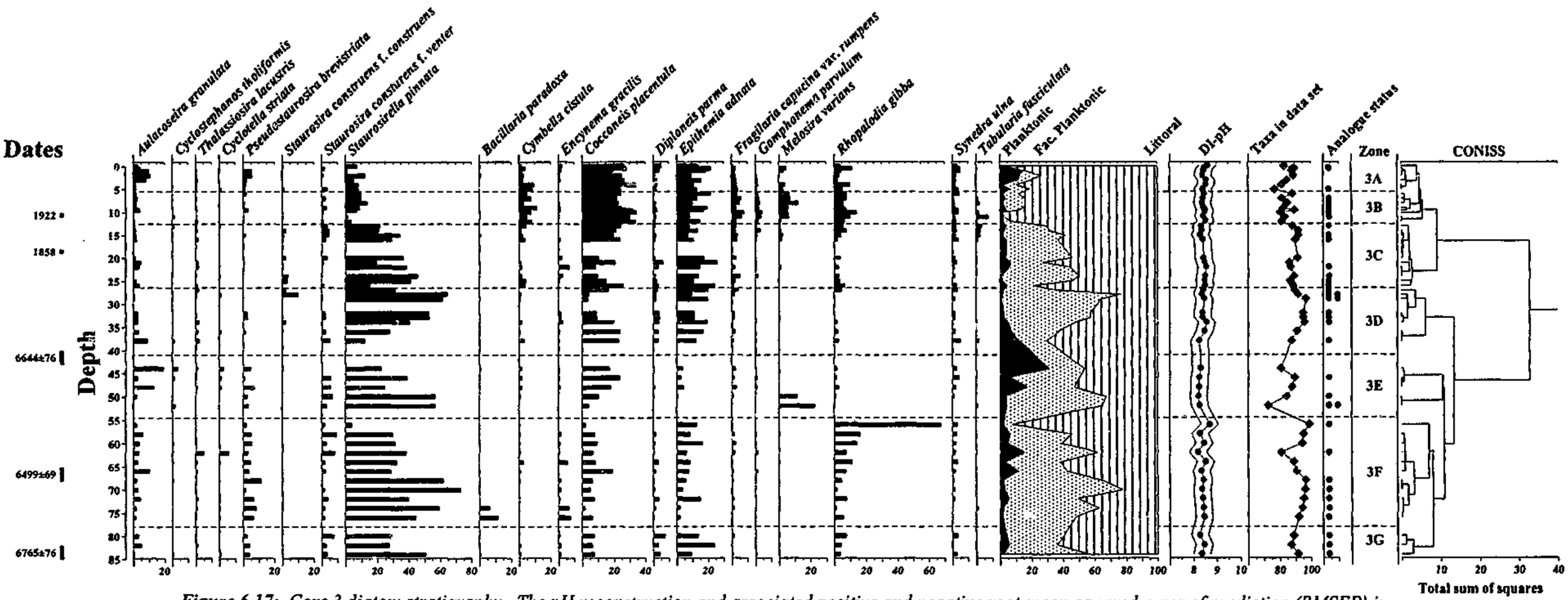


Figure 6.17: Core 3 diatom stratigraphy. The pH reconstruction and associated positive and negative root-mean-squared-error of prediction (RMSEP) is included. Also shown is the total proportion of diatoms found in the lake data set. Samples with no symbol, "•" and "••" have, respectively, good, fair and poor analogues in the modern data set (see Jones and Juggins, 1995).

70 cm, then decreasing to 8% at 56 cm. There is little change in the percentage of planktonic taxa between this zone and the previous zone.

Zone 3E (55 - 42 cm)

There are no diatoms present between 55 and 53 cm. Littoral taxa continue their general decrease in this zone, accounting for an average of 28.8% of the total diatom assemblage (from 39.1% in Zone 3F). This is despite an increase in *C. placentula* abundance from 7% in zone 3F to 18% in this zone. The decrease is largely due to a lower abundance of *E. adnata,* which falls to its lowest abundance in this zone from an average of 7% in Zone 3F to only 2% in this zone, and a decrease in *Rhopalodia gibba* abundance from an average of 29% to 1%. The littoral taxa are replaced by higher values in planktonic taxa, from 6% to 19.5%. This is generated by an increase in *Aulacoseira granulata* abundance from 4% in zone 3F to 7%, and also an increase in the abundance (up to 3% each) of other centric taxa including *Cyclotella striata*, *Cyclostephanos tholiformis*, and *Thalassiosira lacustris*, which were absent in previous zones.

Zone 3D (42 - 27 cm)

There are no diatoms present between 42 and 38 cm. Zone 3D incorporates a section containing no diatoms (29-31 cm). This zone is characterised by a substantial increase in littoral taxa (from 28.8% to 42.3%), and decreases in both facultative planktonic and planktonic taxa. The increase in littoral taxa, however, is not uniform across all species, with *C. placentula* decreasing from an average of 18% in Zone 3E to 14% in this zone, while *E. adnata* increases in average abundance from 2% to 16%. *Staurosirella pinnata,* still the dominant facultative planktonic taxon, remains constant in average abundance (39% in this zone and 40% in zone 3E), but is, however, still highly variable within the zone. Both *Staurosira construens* var. *venter* and *Pseudostaurosira brevistriata* decrease from an average of 4% in zone 3E to 1% in this zone. Conversely, *Staurosira construens* var. *construens* makes its first appearance in the record in this zone, with an average of 2% and a maximum abundance of 11%. Planktonic taxa decrease in abundance to 4.6%, due to the disappearance of the minor planktonic taxa that were present in the previous zone and also a decrease in *A. granulata* from 7% to 3%.

Zone 3C (27 - 13 cm)

There are no diatoms present between 20 and 17 cm. The mean representation of littoral taxa increase again in this zone (to 56%) with a corresponding decrease in facultative planktonic taxa (down to 39%). Interestingly, at the beginning of this zone there are higher and more consistent values for several of the minor littoral taxa. These include *Cymbella cistula* (increasing from 1.5% in zone 3D to 4% in this zone), *Gomphonema parvulum*, *Melosira varians* and *Tabularia fasciculata*. The biggest change within the facultative planktonic taxa is a decrease in *Staurosirella pinnata,* after reaching a peak abundance of 65% near the top of the previous zone. There is little change in the abundance of the planktonic taxa, remaining at < 5.0%.

Zone 3B (13 - 6 cm)

The abundance of littoral taxa is higher still in this zone, reaching an average of 74.1%. This is due to small increases in both *Cocconeis placentula* (to an average of 28%) and *E. adnata* (to an average of 14%) and also to further increases in the aforementioned minor littoral taxa which began to increase in the previous zone. These include *Cymbella cistula* (average of 8%), *Melosira varians* (6%), *Tabularia fasciculata* (7%) and to a lesser extent, *Gomphonema parvulum* (2%). Facultative planktonic taxa decrease in abundance markedly, from an average of 39% in zone 3C to 16.7% in this zone. This decline is accounted for primarily by *S. pinnata* which decreases overall to an average of 7.5% (from 31% in the previous zone). This decrease is very abrupt at the zone 3C / zone 3B boundary. Planktonic taxa increase to an average of 9.1% in this zone, from 4.6% in zone 3C. This increase is caused mainly by a cumulative increase in minor taxa (ungraphed), including *Cyclotella striata* and *Cyclostephanos tholiformis*.

Zone 3A (6 - 0 cm).

Littoral diatoms account for over 72% of all the taxa in this uppermost zone of the core. The three dominant taxa *Cocconeis placentula*, *Epithemia adnata* and *Staurosirella pinnata* remain relatively constant in abundance in this zone, changing little from the previous zone. *Pseudostaurosira brevistriata* increases slightly from 1% to 3%, while *Rhopalodia gibba* decreases from an average of 8% to 5%. The minor littoral taxa which increased in abundance in zone 2 all decrease in this zone, with *Cymbella cistula* decreasing from an average of 7% to 3%, *G. parvulum* decreasing from 2% to 1%, *M. varians* decreases from 6% to 2% and *T. fasciculata* from 1.5% to 0%. The planktonic *A. granulata* increases from an average of 2% in zone 3B to an average of 8% in this zone, with a maximum of 11% at 2 - 3 cm.

Summary

This core is characterised by numerous sections of sediment where there are few diatoms present with the thickness of these sections ranging between 3 and 8 cm. The major changes in this core are the shifts between littoral and facultative planktonic taxa dominated assemblages. There is an equal abundance of the two habitat groups at the base of the core, with a switch to a facultative planktonic dominated assemblage (principally *S. pinnata*) from 77 to 38 cm. The zone between 38 and 27 cm is represented equally by the two groups of taxa, and then between 26 cm and the surface there is a switch to a littoral taxa dominated assemblage (principally *C. placentula* and *E. adnata*). The other obvious change is a concurrent increase in abundance of several littoral taxa (*C. cistula*, *G. parvulum*, *M. varians* and *T. fasciculata*) between 26 cm and the surface, with the greatest abundance between 12 and 5 cm.

Core 3 shows remarkably little change in DI – pH throughout the entire record. Diatom inferred pH ranges from 8.2 to 8.7 with no obvious patterns.

6.4.3.4 Core 4 (Lake Cullulleraine)

This diatom record (figure 6.18) has very well defined zones with distinct differences between the top and bottom sections. Although the entire length of the core (45 cm length) was sampled and prepared, diatoms were only found to be present above 35 cm in any abundance. In all samples between 45 and 35 cm depth, the only diatoms present were aerophilous taxa, mainly *Pinnularia borealis* and *Luticola mutica*, in abundances too low for effective counting.

Zone 4E (35 - 32 cm)

Zone 4E is dominated by *Aulacoseira subarctica* forma *subborealis*, which is present at > 60% in all samples. The next most common taxon is *Staurosira construens* forma *venter*, which is present at 9% at 35 cm, increasing to 14% at 32 cm. Minor taxa present at < 2% include *Cocconeis placentula*, *Epithemia sorex* and *Epithemia adnata*.

Zone 4D (32 - 20 cm)

The most obvious features of this zone are an increase in *S. construens* forma *venter* and the concurrent decrease in *A. subarctica* forma *subborealis*. Both of these shifts are gradual and constant. By the top of this zone, *A. subarctica* forma *subborealis* has decreased to < 2% while *S. construens* forma *venter* abundance peaks at 23 cm (60%) and then decreases towards the top of the zone with an abundance of 25% at 20 cm. As the abundance of *S. construens* forma *venter* decreases, it is replaced predominantly by another facultative planktonic taxon, *Pseudostaurosira brevistriata*, which has an abundance of 17% at 20 cm. Other changes include small increases in the littoral taxa *Epithemia sorex*, *Cocconeis placentula*, and the facultative planktonic *Staurosirella pinnata*. Additionally, the littoral taxon *Tabularia fasciculata* and the planktonic *Aulacoseira granulata* appear almost for the first time in the upper section of this zone.

Zone 4C (20 - 11 cm)

A. subarctica forma *subborealis* decreases to an average of < 1% abundance in this zone. *Staurosira construens* forma *venter* also gradually decreases in this zone, until it reaches an abundance of 10% at 11 cm. These two taxa are replaced by taxa from all three different habitats. Within the littoral taxa group, *Cocconeis placentula* and *Epithemia sorex* increase in abundance (to an average of 18.3% and 11.1% respectively), with the increases occurring early in the zone (defining the zone boundary) and then remaining relatively constant. *Tabularia fasciculata* maintains its abundance from the previous zone, while *Gomphonema truncatum* increases to an average abundance of 3%, compared to < 1% in zone 4D. *Pseudostaurosira brevistriata*, a facultative planktonic taxon decreases at the beginning of the zone, before increasing again, peaking at 20% abundance at 14 cm and then decreasing to < 10% at the top of the zone. *Staurosirella pinnata* abundance remains unchanged from the upper part of zone 4D. Planktonic taxa,

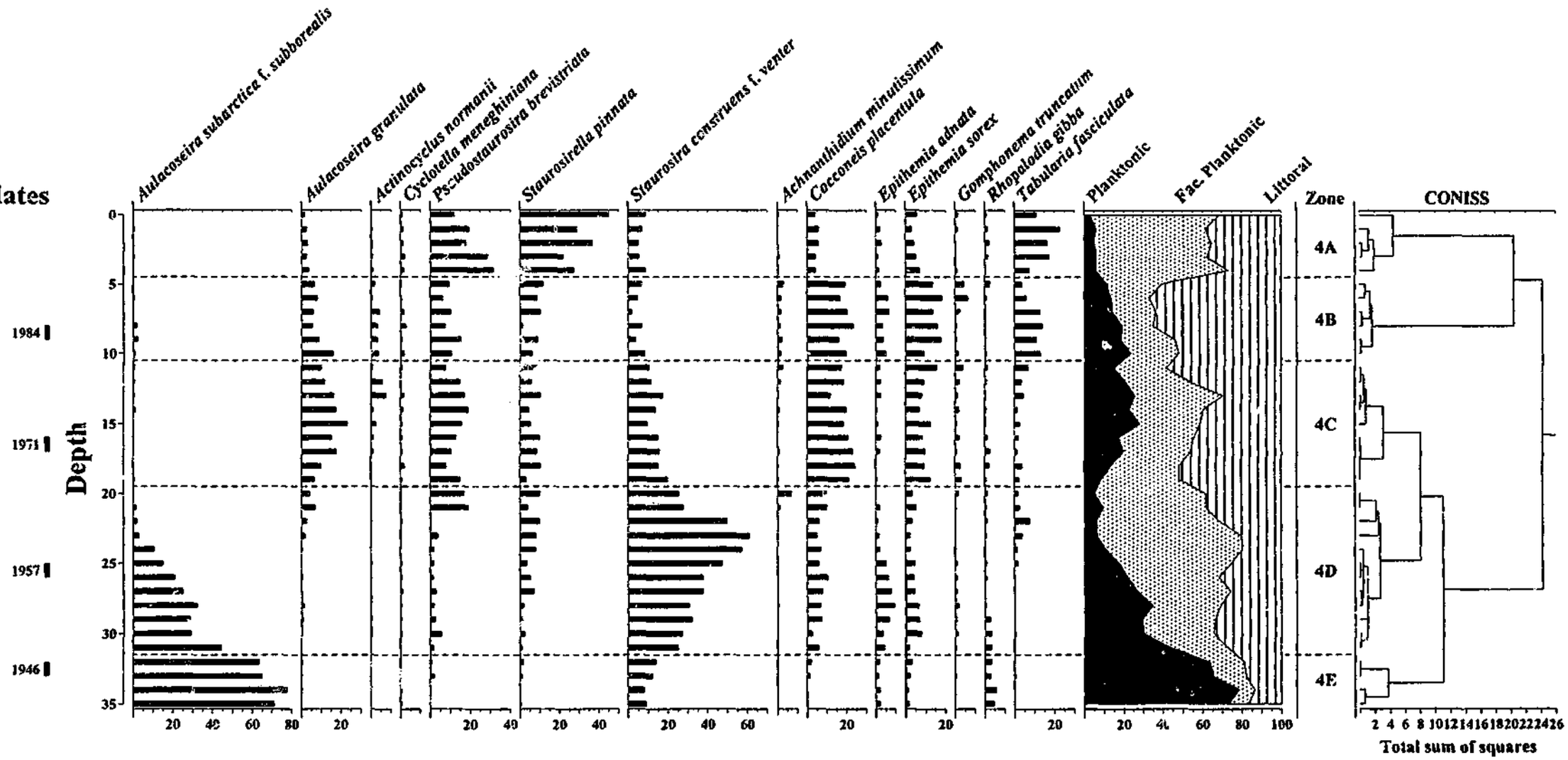


Figure 6.18: Core 4 diatom stratigraphy.

primarily *A. granulata*, increase substantially in this zone. *Aulacoseira granulata* peaks in abundance at 15 cm with 23% of the total assemblage, before gradually decreasing again towards the top of the zone (11 cm) where it accounts for 10% of the assemblage. On average, *A. granulata* represents 14% of the diatom assemblage in zone 4C. In regards to other planktonic taxa, *Actinocyclus normanii* also increases, from < 0.5% in zone 4D to 2.2% in this zone, while *Cyclotella meneghiniana* has fairly consistent representation.

Zone 4B (11 - 5 cm)

There are three major changes between the previous zone and this zone: decreases in both *Aulacoseira granulata* and *Staurosira construens* forma *venter*, and an increase in *Tabularia fasciculata*. *Aulacoseira granulata*'s abundance decreases to an average of 8% from its average of 14% in the previous zone. *Staurosira construens* forma *venter* continues the decline shown in zone 4C, decreasing to an average of < 10%. *Tabularia fasciculata* sharply increases in abundance in this zone, reaching a peak of 14% at 8 cm.

Zone 4A (5 - 0 cm)

There is a sharp boundary between this zone and zone 4B. At the base of the zone there are simultaneous increases in *Staurosirella pinnata* and *P. brevistriata*. *Staurosirella pinnata* is the dominant taxon, accounting for an average of 32% of the diatom assemblage across the zone, and shows an increasing trend towards the surface. Conversely, *P. brevistriata* shows a decreasing trend towards the surface. Planktonic taxa decrease in this zone, with *A. granulata*'s abundance falling from an average of 11.2% in zone 4B to < 3%, and *A. normanii* abundance falling to < 1% from 2% in zone 4B. The two principle littoral taxa, *C. placentula* and *E. sorex*, decrease markedly in this zone (averages of 4.2% and 5% respectively) but, because of the increase in *T. fasciculata*, the total representation of littoral taxa doesn't alter substantially (33.6% in this zone compared to 35% in zone 4B).

Summary

This stratigraphy displays sharp and distinct boundaries between zones. The basal sediments of the core are dominated almost entirely by *Aulacoseira subarctica* forma *subborealis*. This taxon then gradually decreases in abundance upcore and is replaced by *Staurosira construens* forma *venter*. After peaking at 23 cm, *S. construens* forma *venter* decreases in abundance towards the top of the diagram and is replaced by (primarily) *Pseudostaurosira brevistriata, Aulacoseira granulata, Cocconeis placentula* and *Epithemia sorex*. These taxa are then replaced by *Staurosirella pinnata*, *Tabularia fasciculata* and *Pseudostaurosira brevistriata* between 5 cm and the sediment surface.

It was not considered feasible to undertake a DI – pH reconstruction for Lake Cullulleraine due to a lack of adequate analogues in the modern data set. For instance, *Aulacoseira subarctica* forma *subborealis*, which is the major taxon in the lower half of the Lake Cullulleraine fossil record, at certain depths > 80% of the total diatom assemblage, is only present in one sample in the modern data set

Chapter 7 - The palaeolimnology of Lakes Alexandrina and Cullulleraine

7.1 Introduction

This chapter presents interpretations of the palaeolimnology of Lake Alexandrina and Lake Cullulleraine. The zones ascribed in Chapter 6 will be used to aid the interpretations as the results of the dating analysis are not sufficiently precise to enable a discussion based purely on temporal boundaries. However, it is acknowledged that there are limitations to numerically determined zone boundaries in that they reinforce the idea of community integrity when it is possible that species respond individually to environmental or biological influences. For this reason, and the desire to examine diatom assemblages in relation to key human impacts, principally pre and post non - indigenous settlement and river regulation, which may not conform to zone boundaries, interpretation is not based solely on zonation structure. For the purposes of this interpretation, the onset of non-indigenous settlement is assumed to have commenced in 1824 (with the first sighting of the Murray River by a non-indigenous Australian being in 1824), and the onset of river regulation commencing in 1922 (with the completion of the first lock at Blanchetown). Diatom ecological preferences are based on the autecological descriptions made in Chapter 5, with quantitative diatom inferred pH reconstructions also provided for cores 1, 2 and 3. For taxa not reviewed in Chapter 5, ecological inferences, unless otherwise stated, are drawn from the same reviewed literature used for the major taxa.

7.2 Lake Alexandrina

7.2.1 Sediment record chronology

Core 1 has a much slower sedimentation rate than cores 2 or 3, based on the ^{14}C results, with an age of 7610 ± 140 years determined at 119 - 125 cm, equating to an average sedimentation rate of 0.16 mm/year. This is a similar age to that determined for the base of core 2 at nearly 5 m depth. The core 1 site is located at least 5 km from the main channel, while core 2 is located within the channel (figure 6.1). If sediment was being deposited in a fan from the channel, then it seems logical that the coring site furthest away would have the slowest sedimentation rate. This is supported by Barnett's (1993) study which shows that with increasing distance from the palaeochannel, the depth to the Pleistocene - Holocene boundary decreases.

Based on the ^{14}C derived sedimentation rate for core 1, river regulation would have occurred at approximately 1.2 cm, while the onset of non-indigenous settlement would have occurred at approximately 2.6 cm. These depths are almost definitely greatly underestimated, particularly as cores 2 and 3 show evidence of more rapid sedimentation in the past 150 years using ^{210}Pb dating. Additionally, this sedimentation rate is based upon a single radiocarbon date and therefore doesn't take into account the likely changes in sediment input, due to factors such as climate change, over the 7600 ^{14}C year record.

The age of core 2 is similar to that of core 1, with ^{14}C placing an age of 7000 ± 300 on the bottom sediments. Barnett (1994) states that the two modern ^{14}C dates at 50 – 60cm and 122 – 129cm could be the result of organic carbon redistribution, as ^{14}C at similar depths from other cores within the same region of the lake yielded ages greater than 1000 yr BP. The ^{210}Pb dating of core 2 determined a sedimentation rate of 1.71 mm/year (figures 6.4 and 6.5). These ^{210}Pb results show a much slower sedimentation rate than those suggested by the ^{14}C results (1.71 mm/year compared to 0.69 mm/year). This difference in sedimentation rates suggests a relative increase in sedimentation during at least the past 150 years compared to at least the past 7000 years. However, as Barnett (1994) discusses, while the change in slope between the mean ^{14}C based and ^{210}Pb based rates suggests accelerated accumulation of the upper sediments, part of this difference may be an artefact of the time intervals (see Sadler 1981 for further explanation). ^{210}Pb derived sedimentation rates equate to an approximate depth of 9cm for the onset of river regulation and 28cm for the onset of non-indigenous settlement. As the sediments at the base of core 2 were classed as being lacustrine muds and contained diatoms (Barnett, 1993), it can be stated that, at least for the channel section, Lake Alexandrina has been a permanent water body for at least 7000 years. There is obviously a chance of lake dryness, particularly prior to river regulation but this could not be determined because of the coarse sampling resolution undertaken by Barnett (1993).

The ^{14}C analysis of core 3 presents a dilemma in that all three sampled depths are apparently of similar age (approximately 6000 yr BP, see table 6.4). The dilemma is whether to accept these dates as reliable, which would most probably indicate that the sediment between at least 40 cm and 90 cm was deposited very rapidly, or to consider that the dates are unreliable and are a consequence of external forces. Liaison with the Radiocarbon Dating Laboratory at the University of Waikato (Alan Hogg, pers. comm., 2001) who processed and analysed the samples, seemed to suggest that given the very small quantities of organic carbon in the sediment (typically < 1% of the sediment weight), and the fact that other data (such as the diatom record and sedimentology) indicated that the sediment between these depths was not homogeneous, these dates were not an accurate reflection of the true age of the sediment. Rather it would appear that the ^{14}C analysis was probably affected by the hard - water effect (Shotton, 1972), which refers to sediments being contaminated with older carbon. This process can take place when the materials to be dated, such as aquatic plants, take up carbon from water containing bicarbonate derived from old, inert sources, and is common in aquatic environments where calcareous rocks occur. Consequently, this dilution of ^{14}C levels causes ages to appear older than their true age (Roberts, 1998). Although the influence of the hard water effect cannot be verified without further analysis, it does appear to be a likely scenario, especially given the high level of bicarbonates in the water column, and also the presence of fossilised plant matter in the core. Perhaps the most prudent conclusion to make is that the lower sediments (40 cm to the core base) cannot be reliably dated, which has obvious ramifications for the interpretation of the diatom fossil record.

The ^{210}Pb for core 3 does assist, however, in applying timescales to the more recent sediments. This is fortunate as the impact of cultural events such as non – indigenous settlement and river regulation are the prime focus of this study, as opposed to the secondary focus of the impact of climatic events. Following consultation with ANSTO, who undertook the ^{210}Pb analysis (Heijnis, pers. comm., 2001), a sedimentation rate of 1.3 mm / year can be derived from the slope between samples at the surface, 2cm, 4cm and 8cm. This sedimentation rate can then be extrapolated to 17.5cm, at which point the levels of ^{226}Ra change (figure 6.8),

indicating a substantial shift in sediment type. This equates to an age of 1858 at 17.5cm. The ^{210}Pb results suggest that the onset of river regulation (1922) occurred at approximately 10cm, and the onset of non – indigenous settlement occurred at a depth below 18cm (unable to be calculated due to the change in sediment type).

7.2.2. Discontinuities in the diatom records

Core 1 contains three sections of diatom discontinuity; between 13 and 15 cm, 17 and 19 cm and 25 and 100 cm (the base of the core) (figure 6.15). There are five sections within core 3 which contained insufficient diatoms to count; 17-20 cm, 39-44 cm, 53-56 cm, 77-80 cm and 85-100 cm. All samples analysed from Core 2 contained diatoms, but as the sampling resolution was quite coarse, the possibility of barren sections cannot be dismissed. There are several possibilities for the lack of diatoms, including problems of valve preservation, and lake dryness.

Perhaps the most likely explanation for an absence of diatom preservation in the lake sediments is valve dissolution. Dissolution is caused by hydrolysis of the metal carbonates, thus producing hydroxyl ions which then liberate oxygen from the silicate module, leaving silica in its dissolved state (Barker, 1992). Dissolution of siliceous valves is influenced by a number of factors including alkalinity, salinity, temperature, dissolved silica concentrations, grazing and groundwater ingression (Hurd and Birdwhistell, 1983, Flower, 1993). Ryves (1996) found that the single most important factor affecting diatom dissolution is pH. In experimental studies he found that above a pH of about 9.0 silica dissolved rapidly and often completely. However, Flower (1993) mentions that high pH sites can contain good diatom records and that the form of carbonate present is an important factor in diatom dissolution, with deterioration being greatest in sodium carbonate rich systems and poorest in calcium dominated systems. The current pH of Lake Alexandrina ranges between 7.0 and 9.2 (figure 6.2b) and its dominant ions are sodium carbonate. The lake also receives in the order of 300 m^3/day of groundwater discharges from the adjacent Pleistocene sediments (Barnett and Stadter, 1991), which has been shown to contain high concentrations of sulphate (Wasson pers. comm., 1998), which can also have a negative impact on diatom preservation

Dissolution initially affects weakly silicified and / or fragile valves such as small *Nitzschia* and *Navicula* species. If, in sections of the core where diatoms are preserved, these weakly silicified valves are absent, or if there are partially dissolved diatoms (particularly striae), then this is a good indication that the lake could be subject to diatom dissolution processes. There was evidence of valve dissolution in the sections of cores 1 and 3 which contained diatoms, with between 20 and 35% of all cells either partially dissolved or broken (calculated from scanning 100 valves). There was also an absence of weakly silicified diatoms in all three cores, but this may be an artefact of the ecology of the lake rather than one of dissolution. The high degree of valve breakage in these cores may also be due to the shallow exposed nature of the lake, as Flower (1990) found that diatom breakage is also common in high energy, wind stressed environments.

The sections of cores 1 and 3 that displayed a high level of valve dissolution tended to be located either side of the zones of diatom absence. To illustrate this, the following table describes the degree of dissolution

within zone 3F (77 - 56 cm) from core 3, and also the taxon richness for each sample (raw data presented in appendix 8). In the absence of any suitable method to quantify the level of dissolution, dissolution is ranked arbitrarily based on the difficulty of valve identification relative to levels where the valves were very well preserved. Species richness is included to briefly examine whether there is a relationship between dissolution and taxa richness, which may be important in reference to differential dissolution. However, there appears to be little correlation.

Table 7.1 Degree of valve preservation throughout zone 3F from core 3.

Depth (cm)	Degree of valve preservation	*n* taxa
54	diatoms absent	-
56	poor	17
58	very poor	27
60	poor	31
62	poor	29
64	excellent	34
66	excellent	28
68	excellent	20
70	excellent	18
72	excellent	24
74	poor	23
76	very poor	31
78	diatoms absent	-

Lake dryness can also affect diatom production and preservation, and there is a possibility that the zones of diatom discontinuity represent periods of dryness at particular sections of the lake. Early accounts of the River Murray often cite times when the river was reduced to a series of pools, with a completely dry river bed in certain places (Beale and Frey 1990, Close 1990). On a longer time scale, drier conditions have been indicated in Australia between 3000 and 2000 years BP (Bowler, 1981; Chivas *et al.*, 1985; Cann *et al.* 2000). Of particular relevance to this study is the research of Cann *et al.* (2000), which examined fossilised foraminifera as indicators of estuarine and oceanic influences in sediments from sections of the Coorong and Lake Alexandrina. Cann *et al.* (2000) conclude that there was a 2 m decrease in sea level between approximately 5200 yr BP and 3600 yr BP, as evidenced by both a foraminifera inferred increase in salinity within the estuary, and also exposure of wave cut cliffs containing shells of intertidal molluscs.

This event is contemporaneous with other evidence for drier conditions from various sites in south eastern Australia. Palaeoclimatic research on Lake Tyrell, located in the Kerang region of the Murray – Darling basin, indicated that the lake was dry after 2200 yr BP, with modern conditions commencing at 800 BP (Luly, 1993). Other evidence for a late Holocene dry phase between 2000 – 1000 yr BP includes palaeoclimatic records from Lake Keilambete, Victoria (Dodson, 1974) and Caledonia Fen, in the Victorian Alps (McKenzie, 1997). The recent research of Cupper (2000) also suggests a heightened severity or frequency of drought events after 2000 years ago, as evidenced by a decline in *Callitris* in the Warrananga and Tooperoopna Basins, south – western New South Wales.

Core 1 was taken away from the main channel of the river (figure 6.1), in an embayment which, in Aboriginal folklore, was named as swampland (Bernt *et al.*, 1993). The most recent period of diatom

absence from core 3 (17 - 20 cm) would be approximately 140 years ago, at least 70 years prior to river regulation. Lake dryness could also explain why there is no absence of diatoms from core 2, as this core was taken from the main channel (or palaeo-channel) of the Murray River and hence, the sediment probably would have been continuously saturated. However, the fundamental limitation with this theory is that, although possible, it is unlikely that sediment would continue to accumulate in periods of lake dryness, which it appears to do in core 3. In fact, lakes such as Lake Alexandrina, which are characterised by extreme exposure to wind, are more often subject to deflation during periods of dryness.

Converse to the idea of lake dryness is the impact of flooding events on diatom deposition and preservation. Although there is no apparent sedimentological evidence for past flooding events, flooding has undoubtedly played an important role in the lake's history (Rutherfurd, 1990; Thompson, 1994; McCord, 1995). A flood may flush large quantities of sediment down the river channel, resulting in a flux of sediment being deposited in a single event. This may provide conditions unsuitable for both diatom deposition and preservation, and / or may dilute the diatom seston. The repeated pattern of diatom sucession in between the zones of diatom absence, particularly evident for *Staurosirella pinnata* in core 3 (figure 6.17), is similar to the process described by Salo *et al.* (1986), whereby new successional sequences are commenced post flooding events. However, given the lack of sedimentological evidence for flooding, it is difficult to ascertain the role that flooding may have played in affecting fossilised diatom assemblages.

It would thus seem that chemical diatom dissolution is the most plausible and probable explanation for diatom absence in Lake Alexandrina, with lake dryness and flooding perhaps playing secondary roles. Dissolution almost certainly occurred, particularly as current pH generally ranges between 8 and 9. Reduced freshwater input may have also raised the pH to > 9 and dissolved biogenic silica. In light of the early descriptions of dry river beds in the Murray River channel during droughts the concept of lake dryness is also likely. Adding weight to this theory is the fact that there are no zones of diatom discontinuity during the period covered by river regulation. However, although dryness is almost a certainty in this environment, it is unlikely to explain the majority of sections of diatom absence, principally because the sediment continues to accumulate rather than more conceivably deflating. Perhaps the most judicious conclusion to make is that zones of diatom absence in Lake Alexandrina do not have a single determinant, with all the discussed factors playing important roles at different times in the past.

7.2.3 Core 1

The dominant taxa in core 1 are *Aulacoseira granulata* and *Staurosirella pinnata* (see figure 6.15). There is a greater proportion of planktonic taxa in core 1 than in either core 2 or 3. The presence of *S. pinnata* alongside *A. granulata*, combined with the lack of littoral taxa, strengthens the argument that *S. pinnata* is facultative planktonic in this environment (see section 7.2.5). If this taxon was originating from the lake margin then there probably would have been a similar supply of epiphytic taxa, such as *Cocconeis placentula* and *Epithemia adnata*, as it is highly likely that the lake shore always sustained a substantial macrophyte community (based on present day distributions and also evidence from cores 2 and 3).

Interpreting this core is somewhat more straightforward than interpreting cores 2 or 3 because of the lack of

littoral taxa. Unlike cores 2 and 3, this then negates the possibility of changes in habitat availability and therefore diatom recruitment to the sediment budget, being responsible for changes in diatom abundance. Although this isn't necessarily a problem as it can identify important changes in lake habitat (as it does in core 3), it can interfere with and / or mask changes due to chemical factors. Therefore, as most of the taxa in this core are recruited from the same habitat, the changes in abundance are more likely to be due to lake or river chemistry dynamics.

Although core 1 was sampled to a depth of 106 cm, diatoms were only present in the top 24 cm. Forty eight levels were sampled and scanned for diatoms between 25 and 106 cm with only 10 samples containing any evidence of diatoms, and these were in very low concentrations (no more than 1- 2 valves per transect). The sediment stratigraphy in this core is composed of mud for the entire sequence, with little change that could indicate a possible terrestrial environment from 25 – 106 cm depth. A further justification against lake dryness being the cause of diatom absence is the presence of *A. granulata*, to almost 100% abundance, in the lowest sample containing diatoms at 24 cm. If the lake was dry and was gradually being inundated at this stage then a greater proportion of littoral and perhaps aerophilous taxa would be expected, rather than a taxon which is indicative of turbulent waters more than 1 - 2 m deep. However, there is also the possibility that the diatom assemblage at 24 cm is mostly allochthonous material originating from the river inflow. As previously discussed, fossilised diatoms deposited in Lake Alexandrina are derived both autochthonously and allochthonously, with no accurate means of identifying the likely source.

7.2.3.1 Zone 1D

This section of Lake Alexandrina appears to have been eutrophic to hypertrophic as indicated by the presence of *A. granulata*. If the lake had been oligotrophic or mesotrophic the fossil record is likely to have contained planktonic taxa less competitive in high nutrient conditions, such as *Cyclotella* spp or *Aulacoseira ambigua*.

There are two brackish to saline taxa (*Thalassiosira lacustris* and *Campylodiscus clypeus*) present in this section of the core. Both of these taxa do not exceed 5% in any one sample and so do not indicate any substantial marine incursion. Interestingly, they are both absent from the bottom two samples, suggesting that salinities were lower at this time. This could indicate entire closure of the river mouth at this time, preventing mixing of marine and fresh waters.

Immediately overlying zone 1D (19 - 17 cm) is a period of diatom discontinuity which is likely to be due to lake drying as the abundance of *A. granulata* decreases rapidly immediately prior to this zone and is replaced by *S. pinnata* which can probably survive in shallower environments. Potential chemical changes for this transition should also be examined. While *S. pinnata* is an ecologically tolerant taxon, *A. granulata* has a comparatively quite narrow ecological range. For instance, *S. pinnata* can tolerate EC of up to 5000 µS/ cm while *A. granulata* only seems to be abundant in EC < 1000 µS/ cm, and it can also tolerate slightly lower pH conditions than *A. granulata*. The diatom inferred pH (DI – pH) reconstruction shows that it is unlikely that pH altered during this time. This switch between the two taxa is discussed in greater detail in section 7.2.3.3.

7.2.3.2 Zone 1C

Thalassiosira lacustris and *Campylodisus clypeus* both increase in abundance in the 15 and 16 cm samples. Another taxon which is indicative of brackish conditions, *Tryblionella compressa*, also increases at this stage, reaching a maximum of 14% at 16 cm. This assemblage probably indicates a complete opening of the Murray Mouth, perhaps due to a flooding event, and hence, greater marine incursion.

Immediately overlying zone 1C is a further period of diatom discontinuity (14 - 13 cm). This zone of absence is probably more due to valve dissolution because *A. granulata* remains in high abundance until the barren phase and there is relatively little change in the other taxa. However, the same caution applied to the interpretation of valve absence below 24 cm should be applied here also.

7.2.3.3. Zone 1B

Midway through zone 1B there is a change in the diatom assemblage, with *A. granulata* being replaced by *S. pinnata*. This switch in taxa is sustained for the remainder of the core. In conjunction with the increase in *S. pinnata* are more consistent values for *Staurosira construens* forma *venter* and *Pseudostaurosira brevistriata*. The change in dynamics of *S. pinnata* and *A. granulata* is difficult to resolve on either habitat or chemical grounds. For instance, chemical tolerance differences between the two taxa contradict what is known about recent water quality (i.e.: fresher conditions due to cessation of marine incursion).

A plausible explanation for this switch in dominant taxa is an increase in lake turbidity / suspended solids. It has been suggested (Flower *et al.* 1989 and Sayer 2000) that many of the small *Staurosiraceae* have adapted well to living in turbid environments and will become abundant in these conditions. They may not necessarily prefer turbid waters but are simply more tolerant than most other taxa. An increase in turbidity in the lake is quite likely post non-indigenous settlement because of massive land clearance. South Australia pioneered the techniques of mallec clearance, beginning in the 1870s and continuing in earnest for at least 80 years, which combined with the concurrent introduction of hard hoofed animals, would have greatly increased the sediment load of the Murray River (Sinclair, 2001). Given the available evidence, it would appear that the increase in small *Staurosiraceae* in the section of this core, probably representing non-indigenous settlement, is due to an increase in turbidity.

As for other physical / chemical changes, the estuarine taxa present prior to this time period (*Campylodiscus clypeus*, *Tryblionella compressa* and *Thalassiosira lacustris*) either remain stable or decrease in abundance throughout this zone.

7.2.3.4 Zone 1A

There is little substantial change between this zone and the previous zone. There is a small increase in *Aulacoseira granulata* towards the surface but *Staurosirella pinnata* remains the dominant taxon. The only evidence of ecological impact due to river regulation is a small decrease in the estuarine taxa, *C. clypeus* and *T. compressa* between 6 cm and the surface.

7.2.3.5 Summary

The most obvious characteristics of core 1 are the lack of diatom presence and / or preservation below 24 cm and two sections of diatom absence between 17 – 19 and 13 – 14 cm. The absence below 24 cm is probably due to valve dissolution, perhaps because of high pH conditions, with DI – pH estimating a pH of 7.9 at 24 cm. The section of diatom absence between 17 and 19 cm is more likely to be due to lake dryness while is it hypothesised that the absence between 13 – 14 cm is again due to valve dissolution.

This coring site was subject to minor marine influence, particularly prior to non-indigenous settlement. Probably around the time of arrival of non- indigenous Australians, there is an increase in *Staurosirella pinnata* which continues until the present day. It is surmised that this is due to an increase in turbidity, most likely due to massive land clearance throughout the basin. All of the taxa present in this core are indicative of eutrophic to hypertrophic conditions. Specific changes in chemical conditions through time cannot be made because of the similarity in tolerances between the major taxa.

7.2.4 Core 2

Core 2 represents a much longer time period than cores 1 and 3, with the core base dating to 7000 ± 300 years B.P. The dominant taxa in this core are planktonic and facultative planktonic in origin with sporadic peaks in littoral taxa (figure 6.16). As most of the peaks in littoral taxa occur prior to 2000 years ago, there is a strong possibility that the lake margins were subject to substantial change because of fluctuating sea levels and also river discharge, and so there may have been a shallow littoral zone within close proximity to the coring site. A further likely scenario is that the lake was less turbid during this period, with a deeper photic zone which was conducive to the *in situ* growth of littoral taxa. There is also the possibility that the littoral taxa were fluvially transported from higher reaches of the river and the lake, although this seems less likely given that present day river plankton assemblages have a very low littoral component despite the riparian zone being well covered with aquatic vegetation.

7.2.4.1 Zone 2D

The base of the core is dominated by *Cocconeis placentula* and *Aulacoseira granulata*, and to a lesser extent *Thalassiosira lacustris*. The presence of *A. granulata* indicates a flowing, relatively deep river / lake system while the *C. placentula* (nearly 30%) was probably transported from a shallow, well vegetated shoreline within close proximity. *Thalassiosira lacustris* indicates definite, although minor, marine influence at this time.

At 420 cm, there is a sudden rise in *Staurosirella pinnata*, from negligible abundance in the previous samples to nearly 40%. This increase is not associated with any appreciable decrease in other planktonic taxa such as *A. granulata* or *Cyclotella striata*, which would perhaps indicate a drop in lake level. Rather, the increase is associated with a decrease in various minor littoral taxa. Similar to core 1, this may suggest an increase in turbidity at this time, conceivably then leading to a reduction in photic depth and subsequent decrease in littoral taxa. High values of *S. pinnata* are sustained until 300 cm depth where they decrease and are replaced by littoral taxa. *Aulacoseira granulata* also decreases between 380 cm and the top of the zone.

Thalassiosira lacustris has lower values at 260 cm, however, the other major planktonic taxon, *Cyclotella striata*, does not decrease in abundance in this section of sediment. As *C. striata* and *T. lacustris* have higher EC tolerance than either *S. pinnata* or *A. granulata*, and are common in estuaries, this change in diatom community is likely to represent a decrease in lake level and increased penetration of seawater. This would then lead to lower abundance of riverine / fresh lake planktonic diatoms and an increase in littoral taxa, particularly benthic taxa, and a mixing or incursion of more brackish to marine planktonic taxa from areas closer to the river mouth.

While there are definite habitat changes and changes in EC during this time period, changes in nutrients and pH are more difficult to quantify. The major taxa are indicative of eutrophic to hypertrophic waters and the DI – pH suggests that pH varied between 8.1 and 8.7 during this time period.

7.2.4.2 Zone 2C

There is a recovery in planktonic taxa at 220 cm, particularly *A. granulata* which increases from < 5% to 43% at 180 cm, suggesting a deeper lake phase. Interestingly, *C. striata* decreases at this time, perhaps suggesting that this taxon originates from the more mixed waters further towards the river mouth and is predominantly present when river levels are low.

7.2.4.3 Zone 2B

The decrease in *T. lacustris* during this period almost definitely reflects a further increase in lake level and fresher water conditions. The decrease in *A. granulata* from 60 cm is associated with an increase in *Staurosira construens* forma *venter*, and may again be due to an increase in turbidity levels. Associated with these changes is a sustained decrease in *Cocconeis placentula* from zone 2C, which may be due to loss of macrophytes related to an increase in turbidity. Diatom inferred pH shows a slight decrease in pH towards the top of the zone, concomitant with the increase in *Staurosira construens* forma *venter*.

7.2.4.4 Zone 2A

The most obvious and dramatic change to diatom communities in core 2 occurred in the recent past. The change is almost solely represented by a massive increase in *Pseudostaurosira brevistriata* abundance, to a peak of 65% at 4 cm depth, following the estimated time boundary for the onset of river regulation, as determined by ^{210}Pb analysis. This increase is associated with a small decrease in *Staurosirella pinnata*, and also numerous littoral taxa. The most important factors in explaining this change lie in the ecological differences between *S. pinnata* and *P. brevistriata*, which as previously discussed, is problematic. The ecological review in Chapter 5 suggested that *P. brevistriata* has a slightly lower EC tolerance than *S. pinnata*, which is consistent with the known ecological history of this time period (i.e.: formation of barrages leading to fresher water). However, both these taxa are ubiquitous and very abundant in aquatic systems throughout the world, and their published ecological tolerances are very similar. Until more research is conducted into the ecological / biological preferences of these taxa (and also various other small forms from the *Staurosiracea* group), it is impossible to state, with any degree of confidence, what shifts in these taxa may actually be representing.

7.2.4.5 Summary

In terms of water chemistry, as with core 1, most of the taxa present throughout this core are indicative of eutrophic to hypertrophic waters and alkaliphilous to alkalibiontic conditions. Attempts to interpret changes within the core have proved problematic because, unfortunately, all of these taxa have wide, overlapping preferences. However, there are definite changes in EC levels, with salt water intrusion being at its maximum below 160 cm depth (prior to approximately 1800 years ago). Diatom inferred pH shows a small decrease in pH from 80 cm depth (approximately 700 – 900 years ago), rising again slightly towards the surface. In terms of lake level, it appears that at this site in Lake Alexandrina, lake level was at its deepest between approximately 2000 years ago and the present.

7.2.5 Core 3

Despite the core 3 site being at least 500 m from the shores of Lake Alexandrina (figure 6.1), the diatom record is strongly dominated by littoral (primarily epiphytic) and facultative planktonic taxa (figure 6.17). This characteristic is normally present in very shallow (< 1 m) lake sites where either the lake area was very small or the core was taken close (i.e.:< 50 m) to the shore, or where there was enough light penetration to allow growth on the benthos (see Bennion *et al.*, 2000). In reference to this site, there are at least two reasons why these taxa, and not planktonic taxa, dominate the fossil record of this core. Firstly, the shoreline of the northern embayment of Lake Alexandrina provides a vast seed bed of epiphytic diatoms as there is at least a 5 m wide band of fringing *Phragmites* and *Typha* bordering most of the shore. With highly turbulent waters and wind stress, these epiphytic diatoms would be readily incorporated into the water column and again, because of the high degree of mixing, would be able to travel long distances before being deposited into the sediment.

The second reason relates to the facultative planktonic diatoms (the small *Staurosiraceae* group), which although generally described and accepted as not being true plankton, preferring to inhabit the benthos, are most probably planktonic in this environment. The discussion of River Murray and Lake Alexandrina plankton in Chapter 3 showed that the plankton community was, at some sites (particularly the Lake Alexandrina samples), dominated by small *Staurosira, Staurosirella and Pseudostaurosira spp.*, and that analysis of live samples showed these specimens to be living and not simply dead frustules that were being fluvially transported. Therefore, these taxa (i.e.: *Staurosirella pinnata*, *Staurosira construens* varieties, and *Pseudostaurosira brevistriata*) are able to form a component of the contemporary lake plankton in Lake Alexandrina. Obviously this may not have been the situation during the past, and this possibility is considered within the interpretation. These taxa remain classified as facultative planktonic on figure 6.13, however, in order to differentiate from true planktonic taxa.

7.2.5.1 Zone 3G

There are no diatoms between 100 cm and 85 cm in this core. As there were no signs of valve dissolution directly above this section of the core, and because the diatom communities adjacent to this section do not seem to be indicating recovery from an extremely alkaline period (i.e.: by the presence of taxa with high pH tolerances), it would seem that other fluvial processes are responsible for this absence. Most likely, this

section of the core represents a period of lake dryness or a period when the northern embayment of Lake Alexandrina was very shallow swampland.

Diatoms are then present for three sampling levels before a zone of diatom absence between 80 and 77 cm. It is difficult to surmise why diatoms are absent from this section as there is little change in the diatom community either before or after this phase (relative to the other zones of discontinuity), which would provide evidence for either an increase in pH or lake drying.

In terms of water chemistry, all taxa are indicative of eutrophic to hypertrophic conditions and the DI – pH suggests a pH ranging between 8.0 and 9.0 throughout this period. The presence of *C. clypeus*, which is absent from all other zones, suggests that this period of the fossil record had the highest EC levels.

7.2.5.2 Zone 3F

The diatom assemblage from 77 cm is dominated by *S. pinnata*. The abundance of *S. pinnata* peaks at 70 cm (78%) and then decreases to < 10% at the top of the zone, being replaced by a multitude of benthic and epiphytic taxa (including minor taxa that are not graphed).

Overlying this zone is a further period of diatom absence, and there is evidence for changes in pH being a determining factor for this absence. *Staurosirella pinnata* is primarily replaced by *Epithemia adnata* and *Rhopalodia gibba*, which both have a higher pH tolerance. Out of all the taxa present in core 3, *R. gibba* has the highest pH tolerance, with *E. adnata* the second. These two taxa alone account for more than 70% of the total assemblage at 56 cm. This would then suggest an increase in pH towards the phase of diatom absence, to perhaps > 9 pH units, thus being conducive to diatom dissolution. This interpretation is strengthened by the DI – pH reconstruction which suggests a small increase in pH towards the top of the zone.

There appears to be little change in EC within this zone. All of the major taxa are indicative of fresh to brackish waters. A few of the taxa, including *C. placentula* and *Nitzschia frustulum,* have EC optima above the fresh to brackish range but as they survive equally well in these lower salinities, their presence does not refine the interpretation.

There also appears to be little change in trophic status within the zone as most of the taxa present are indicative of mesotrophic to eutrophic waters. *Cocconeis placentula* has the highest nutrient optimum but as this taxon is abundant in a large number of alkaline aquatic environments, regardless of water quality, small changes in abundance cannot ideally be used as an interpretative tool.

In terms of the general ecology of the lake environment, the diatom community is equally dominated by planktonic and littoral taxa. It is unknown whether the planktonic component originate from the Murray River. The large presence of littoral taxa suggest substantial fringing vegetation, particularly as most of these littoral taxa are stalked and therefore have a habitat preference for aquatic vegetation.

7.2.5.3 Zone 3E

Staurosirella pinnata increases again at the start of this zone (increasing to > 50%) and then decreases towards the top of the zone (22% at 44 cm). It is unlikely that this decrease is due to lake shallowing as there is a simultaneous increase in *Aulacoseira granulata* at 44 cm, which occurs immediately prior to a zone of diatom absence. As a true planktonic taxon, *A. granulata* should not be increasing in abundance if the lake level is dropping. A possible explanation is that as that the lake level drops, so does the velocity or turbulence of the water flowing from the main channel, encouraging the sedimentation of *A. granulata* valves. The study by Hotzel and Croome (1996) on the dynamics of this taxon within the Murray River, showed that it is dependant on turbulent waters for survival because of its relative heavy density, and that it quickly sinks if turbulence is decreased. However, the same argument could be made for *S. pinnata*, which can also be heavily silicified, and, as this does not increase in abundance prior to the zones of absence, this proposition seems unlikely.

There is a similar pattern of *S. pinnata* representation to that in the previous zone. However, where *R. gibba* replaced *S. pinnata* in zone 3F, thus providing evidence of an increase in pH, in this zone *S. pinnata* is replaced by *C. placentula*. Given that *A. granulata* can also tolerate alkaliphilous conditions, it is likely that this assemblage change is also due to an increase in pH. The substitution of *R. gibba* by *C. placentula* may be due to a slight change in ionic composition or perhaps an increase in nutrients (which would also help to explain the increase in *A. granulata*).

7.2.5.4. Zone 3D

Zone 3D is dominated by *S. pinnata*, which increases towards the top of the zone, mirroring a small decrease in *C. placentula*. It is likely that this change is either a reflection of altered habitat availability with perhaps an decrease in the area of the littoral zone and thus, less recruitment of littoral taxa, and / or is representing a decrease in pH (which isn't indicated by the DI – pH reconstruction). Other than this change, there appears to be little chemical or physical change within this zone.

7.2.5.5. Zone 3C

The middle of zone 3C coincides with the estimated time boundary of the onset of non - indigenous settlement (based on ^{210}Pb analysis). *Staurosirella pinnata* begins a sustained decline and is replaced by *C. placentula* and *E. adnata*. Possible reasons for this switch are discussed in section 7.3.5.6 (as the change becomes more apparent in the overlying zone). There are also increases in *R. gibba* and *N. frustulum*, which may be indicating an increase in pH.

7.2.5.6 Zone 3B

The beginning of this period marks a time of major change in the diatom community. The most obvious changes are a more substantial decrease in abundance of *Staurosirella pinnata* and an increase in abundance of *Cocconeis placentula* and several other littoral taxa including *Cymbella cistula*, *Gomphonema parvulum*, *Melosira varians*, *Rhopalodia gibba*, and *Tabularia fasciculata*.

The biggest shift in zone 3B is the switch between *S. pinnata* and *C. placentula* dominance. These taxa have

similar wide ranging water quality tolerances and so the change is most likely due to modification of diatom habitat availability. *Staurosirella pinnata* is facultative planktonic in this environment while *C. placentula* is epiphytic. The water level of Lake Alexandrina was artificially lowered after the completion of barrage formation (Cann *et al.* 2000), which according to the ^{210}Pb dating, occurred at approximately 10 cm depth. This, combined with more stable water levels, would have resulted in greater swamp development around the fringes of the lake, particularly in the region where core 3 was extracted. This, in turn, would likely have led to a greater supply of *C. placentula* and other epiphytic forms to the sediment diatom budget because of an expansion in the area of aquatic vegetation. An increase in sediment recruitment of *C. placentula* doesn't necessarily mean that the concentration of *S. pinnata* in the water column has changed, but just that it is diluted by the greater production of epiphytic diatoms. This highlights the need for multiple core analysis from large lakes because of the difference in dynamics, and also competition, between habitat changes and chemical forcing.

Additionally, the increase in abundance of some of the minor littoral taxa (*Cymbella cistula*, *Gomphonema parvulum*, *Melosira varians*, *Rhopalodia gibba*, and *Tabularia fasciculata*) is almost definitely due to an increase in nutrient concentrations within the lake. All of these taxa are indicative of eutrophic to hypertrophic conditions, and because most of them appear for the first time at around the same level (they actually start to increase at the top of the previous zone), probably in the late 1800s, it seems likely that their presence is due to anthropogenic eutrophication. This increase in nutrients may have also encouraged the growth of aquatic vegetation

Tabularia fasciculata, as well as having a high nutrient optimum, also has a high tolerance for brackish waters, suggesting an increase in salinity during this period. Most of south eastern Australia was subject to extensive vegetation clearance soon after non-indigenous settlement (Menzies, 1983), which would have raised groundwater tables in many areas of the Murray - Darling Basin and thus increased saline discharge into the river. There is also documented evidence of a severe drought in south eastern Australia between 1914 and 1915, with salinity levels after this period at Morgan reaching 10 000 EC (Eastburn, 1990). According to the ^{210}Pb dating, this event would have occurred at approximately 11 cm depth, which is where *T. fasciculata* peaks in abundance.

7.2.5.7 Zone 3A

The eutrophic littoral taxa that increased in abundance in the previous zone, decrease in this zone. The other major changes in the top 5 cm are an increase in *Aulacoseira granulata* and *Pseudostaurosira brevistriata* numbers, which are both still indicative of high nutrient concentrations. The decrease in littoral taxa may represent an improvement in trophic status, but the concurrent increase in eutrophic planktonic and facultative planktonic taxa indicate that nutrient concentrations remain high. The switch in taxa may be due to slightly different nutrient thresholds, or perhaps may be an artefact of a fully regulated river system (as opposed to barrage formation only). This latter explanation is the more likely as the onset of the increase in these taxa coincides with the period of time when river regulation reached its full capacity (late 1950s), with the ^{210}Pb analysis dating this time period to around 5cm depth. In ongoing research on the palaeolimnology of the upper Murray River, Reid (pers. comm.) is investigating the potential of using the onset of *Aulacoseira granulata* in fossil records as a marker of river regulation.

7.2.5.8 Summary

It would seem that during the period prior to the approximate onset of non - indigenous settlement, water quality in Lake Alexandrina, at the site of core 3, was characterised by regular fluctuations in pH, most probably ranging from 7.5 to > 9.0. There are several zones of diatom discontinuity which suggest that at times pH was too high to adequately preserve diatom valves. There is no evidence to indicate that the pH was ever lower than this. The water was fresh to brackish and most probably eutrophic to hypertrophic.

Following non-indigenous settlement, Lake Alexandrina probably experienced extensive growth in aquatic vegetation. There is also evidence of an increase in nutrient levels and salinity concentrations. This increase in nutrients appears to be sustained until the present time.

7.2.6 The water quality history of Lake Alexandrina

The Lake Alexandrina cores are widely separated (more than 10 km) and, as such, represent different phases in lake development and maturation. Because of this, as can be seen from the above discussion, these cores are very different in terms of sedimentation rates, sediment type and diatom assemblages. Compounding these differences is the fact that rarely could unambiguous conclusions be made about past lake level or ecology because the majority of taxa present have wide tolerances to ecological parameters. Most of the fossil taxa (major and minor) have very similar, or at least overlapping, requirements for EC, pH and TP and none of these three parameters appear to have altered greatly over the past 7000 years. Shifts in the diatom fossil record may represent small changes in these ecological parameters, as proposed in the above discussions, but due to the lack of unique autecologies, these interpretations are forwarded tentatively. Adding to the complexity is the influence of changing fluvial and tidal forces, and subsequent shifts in the ionic composition of the lake due to fluctuating groundwater, seawater and riverine inputs, all of which can affect the diatom communities.

Lake Alexandrina plays a critical role in both contemporary and palaeolimnological understanding of the Murray – Darling Basin, as the river, upon entry to the lake, represents an accumulation of all upstream processes. In summary, the major palaeolimnological features of Lake Alexandrina are outlined as follows -

1) Diatoms have been present in Lake Alexandrina for at least 7000 years, as evident in core 2.

2) Both cores 1 and 3 suggest evidence of river / lake dryness during the past. The most extensive phases of implied lake dryness are between 24 cm and 100 cm for core 1 and between 85 and 100 cm depth in core 3. The base of core 1 (100 cm), according to ^{14}C dating, is approximately 6250 ± 140 years ago. The age of the sediment at 24 cm cannot be reliably dated but is likely to be in the order of approximately 500 to 1000 years ago. The base of core 3 could not be reliably determined.

3) Most remaining phases of diatom absence in both cores 1 and 3 are almost definitely representing diatom dissolution because of the increasing presence of alkalibiontic taxa immediately prior to the

zone of absence and also because of evidence of partially dissolved diatom valves. Therefore, there are periods in Lake Alexandrina's past where, in some parts of the basin, pH was probably > 9.

4) Shifts in littoral and planktonic taxon representation suggest change in the abundance of aquatic vegetation, both in the littoral zone and submerged. This is particularly evident in core 3 where the diatom community suggests a substantial increase in aquatic vegetation in the post non – indigenous settlement period.

5) There is evidence for fluctuations in turbidity in all cores at various times in the past, particularly in core 1 where turbidity appears to have increased proceeding the arrival of non- indigenous Australians.

6) Core 3 shows a clear increase in nutrients in the last 200 years, shown by an (analogous) increase in the abundance of diatom taxa competitive in high nutrient environments. Core 2 shows a similar pattern.

7) Core 3 also shows a suggested increase in EC between the period of approximately 150 years and 70 years ago, becoming less obvious post river regulation. Apart from this, there is no major evidence to show any major fluctuations in EC in the region of core 3, particularly from marine incursion. Throughout the record occasional values of estuarine taxa were found (*Thalassiosird lacustris* and *Campylodiscus clypeus*) but these never accounted for > 2% of any total assemblage.

8) There is stronger evidence for fluctuating EC levels in cores 1 and 2. There is an increase in estuarine taxa at 16 – 17 cm in core 1, perhaps suggesting a period of low river flow and greater marine incursion (probably at least 500 years ago). In core 2, salt water incursion is at its maximum below 160 cm, although there is some suggestion that it may have also increased during the last 100 years.

9) Most of the taxa present in all three cores are indicative of eutrophic to hypertrophic conditions, suggesting that Lake Alexandrina has always been a naturally eutrophic system. The exception to this is the increase in relative abundance of eutrophic / hypertrophic taxa in the top sections of cores 2 and 3 which most likely represents a further enhancement of these natural high nutrient conditions post non – indigenous settlement.

7.3 Lake Cullulleraine

The sediment core from Lake Cullulleraine provides a detailed, high resolution record of changes since river regulation (figure 6.18). It is uncertain, however, whether these changes reflect the lower Murray River *per se*, or are peculiar to the lake under study. Lake Cullulleraine is directly fed by the main channel of the Murray River, and as such, the water quality should generally be indicative of that of the main channel. However, this is complicated by autochthonous processes. For instance, there is the possibility that the decrease in water velocity once the channel enters the lake basin has an effect on the composition of the

planktonic community. The existence of an extensive littoral zone may also have an effect on the relative abundance of diatoms originating from both the river channel and those growing *in situ*, and may influence the degree of biological nutrient uptake. All of these factors will undoubtedly affect the fossilised diatom community and as such, must be considered throughout the interpretation.

It should also be noted that the Lake Cullulleraine palaeolimnological record may not be representative of all connected billabongs / water storages in this region. Studies by Sheil (1980), Hillman (1986) and Boon (1990) have all shown substantial differences in water chemistry and biology between billabongs located relatively near to each other. So perhaps the most interesting initial observation to make about the Lake Cullulleraine diatom record is its similarity to the Lake Alexandrina diatom records in terms of species presence. Major diatoms that are common to both sites include *Staurosirella pinnata, Pseudostaurosira brevistriata, Staurosira construens* forma *venter, Aulacoseira granulata, Tabularia fasciculata, Cocconeis placentula, Epithemia adnata*, and *Rhopalodia gibba*. This degree of similarity suggests that perhaps individual lake autochthonous processes are not as influential as riverine input, particularly in water bodies that are permanently connected to the main channel. The major diatom species differences between the two sites are the dominance of *Aulacoseira subarctica* forma *subborealis* (planktonic) and *Epithemia sorex* (primarily epiphytic) in Lake Cullulleraine, both of which are closely related, in terms of associated presence and ecology, to other generic forms (i.e.: *A. granulata* and *E. adnata*) that are dominant in Lake Alexandrina.

The second most apparent observation to make about this core is the distinct trends in diatom representation. As well as facilitating interpretation, these clear patterns throughout the record suggests that lake sedimentation has been continuous and constant, and that it has been relatively undisturbed.

7.3.1 Core chronology

The ^{210}Pb analysis provided an excellent dating profile of the core, which suggests an annual sedimentation rate of 6.3 mm/year (figures 6.9 and 6.10). The sediment at 40cm depth was calculated as being approximately 63 years old, meaning that deposition occurred in about 1934, and the surface sediments were calculated to be recent, although slightly mixed, material. The slope of excess ^{210}Pb activity versus depth (figure 6.9) shows that the sedimentation rate appears to have been relatively constant throughout this time period. Extrapolation of this sedimentation rate to the base of the core (at 48cm) is not possible because of the change in sediment type from 46cm to 48cm. However, it is likely that the basal sediments do approximately represent the time period around when the lake was filled.

At 46 cm depth the sediment changed from soft lacustrine mud to grey, unsaturated gravel (figure 6.14). This sequence was repeated in all of the probes conducted on the lake bed in different central locations. This sedimentological pattern is mirrored by sediments core obtained by Walker and Thoms (1993) from weir pools in the lower Murray River. In these cores, the pre-river regulation sediments consisted of predominantly coarse sands, while the post regulation sediments were fine silts and clays. However, complicating what seems to be an obvious transition from a dry riverine plain to an inundated lake is the fact

that the basal 12 cm (between 48 cm and 36 cm) is virtually devoid of diatoms. Valves are not totally absent but those that are present are badly preserved, have very low abundance (< 5 per transect) and are mostly aerophilous / benthic rather than planktonic / epiphytic.

Several hypotheses can be proposed as to why diatoms are absent from this early part of the sediment record. Firstly, it is not uncommon for lakes that are suddenly inundated (such as reservoirs) to not preserve diatoms for at least 10 years (Donar *et al.*, 1996; Bradbury and van Metre, 1997). Burrinjuck Reservoir in NSW, which has been extensively dated, has a lack of planktonic diatoms in the first 20 years of lake development, with the sediment also containing only sparse, aerophilous taxa. It has been suggested (Tibby, 2000) that this is due to silica deficit in the water column as the sediment has preferential uptake of silica. It is likely that there was still planktonic diatom production during this time period but that the valves readily dissolved in the water column before they had a chance to be fossilised. It is unlikely that this scenario applies to Lake Cullulleraine because the lake is so shallow. Burrinjuck Reservoir was probably at least 35 m deep during this early period, a factor which is conducive to planktonic dissolution in the water column. Lake Cullulleraine is only 2 – 3 m deep so planktonic diatom sinking time is a lot shorter and thus less liable to be subject to dissolution.

Another possibility is that of dissolution due to elevated pH levels, particularly given the evidence of dissolution in Lake Alexandrina and the similarity between diatom sequences. However, the diatom assemblage at 35 cm is dominated by *Aulacoseira subarctica* forma *subborealis*, a taxon which has a lower pH tolerance than most of the other taxa present in the core, and which is often described as preferring circumneutral conditions (see Chapter 5 discussion). If dissolution was occurring due to high pH conditions then the recovery phase would be expected to be dominated by alkalibiontic to alkaliphilous taxa, and not by a circumneutral taxon.

It can also be postulated that prior to the construction of Lake Hume (in the late 1930s), Lake Cullulleraine was probably not a stable, permanently inundated lake basin. However, without further research, in particular the examination of archived records from the 1920s and 1930s (records which are currently not accessible), it is impossible to determine whether a dynamic lake level and possible periodic drying contributed to diatom absence.

Obviously there are more possible interpretations of the chronology and sedimentation pattern of this bottom 12 cm section of core material. However, with the information available, the third option seems the most likely. It will therefore be assumed that the onset of diatom presence represents the time of permanent lake inundation.

7.3.2 Zone 4E

The diatom community of Lake Cullulleraine at 35 cm is almost completely dominated by *Aulacoseira subarctica* forma *subborealis*. It seems likely that the initial source of *A. subarctica* forma *subborealis* in the sediment record is from the main Murray River channel, and is transported into the lake during the lake filling process. The second most abundant taxon is *Staurosira construens* forma *venter*. This diatom

community suggests that pH was slightly lower than contemporary pH in the lake, and that nutrient levels may also have been lower. The fact that the flora is dominated by planktonic forms indicates the lack of an established littoral zone and perhaps an absence of aquatic vegetation.

7.3.3 Zone 4D

After dominating Zone 4E, *Aulacoseira subarctica* forma *subborealis* steadily decreases in abundance throughout this zone, becoming absent by the top of the zone. As *A. subarctica* forma *subborealis* decreases in abundance it is steadily replaced by *Staurosira construens* forma *venter* and to a lesser extent *Cocconeis placentula, Epithemia sorex, Epithemia adnata, Pseudostaurosira brevistriata* and *Staurosirella pinnata.* It needs to be considered whether the decline in *A. subarctica* forma *subborealis* is due to autochthonous processes taking precedent over river input or whether it represents a change in the river planktonic community. It is therefore necessary to examine the modern day dynamics of this taxon and also the diatom community that replaces it.

Cocconeis placentula, Epithemia sorex and *Epithemia adnata* are almost definitely derived from the littoral zone of the lake and do not originate from the Murray channel. A plankton survey of the lower Murray River (see section 3.4.3) showed that *Staurosira construens* forma *venter Pseudostaurosira brevistriata* and *Staurosirella pinnata* are capable of surviving in the channel environment although abundances do not appear to reach a total of more than 30%. The fact that they can presently survive in the channel suggests that they would have also been able to do so in the past, perhaps forming more substantial populations, and that this presence could have provided the impetus for continued growth within the confines of Lake Cullulleraine. The most likely scenario is that the *Staurosira* group (and most probably *A. subarctica* forma *subborealis*) were initially sourced from the Murray channel but that population growth continued autochthonously, most likely because of similar water quality between the two systems at the time. There is a possibility that the switch from *A. subarctica* forma *subborealis* to *S. construens* forma *venter* represents a change from a largely riverine input to largely autochthonous processes but it is impossible to determine definitely given the present similarities between the Murray river plankton, Lake Cullulleraine plankton, and also the plankton 400 km downstream in Lake Alexandrina (figure 7.9).

It is therefore important to concentrate on the different chemical and physical preferences of the two taxa. *Aulacoseira subarctica* forma *subborealis* is not common in present day planktonic communities of either the lower Murray or Lake Cullulleraine. A similar morphological taxon, *Aulacoseira distans,* has been recorded in routine plankton sampling of the Murray river but abundances are nearly always below 10% and are predominantly sourced from sites upstream of the Darling River confluence (Hotzel and Croome, 1996). Currently, the only places where *A. subarctica* forma *subborealis* has been found in substantial abundances (> 30%) is in aquatic systems that have a similar turbidity, lower nutrient concentrations and slightly lower pH than either the contemporary lower Murray River or Lake Cullulleraine (Reid, 1997; and Tibby, 2000).

Staurosira construens forma *venter,* as previously discussed within the Lake Alexandrina interpretation, has a wide ecological tolerance. However, there are some differences in comparison to *A. subarctica* forma

subborealis, principally in its ability to thrive in waters with higher EC levels and greater turbidity (Reid, 1997; Tibby, 2000; and results from this study).

It is possible that as water flow to the lake became more regulated (with the completion of Lake Hume, in 1961, occurring at approximately 23 cm depth) the lake became more stable, encouraging the growth of aquatic vegetation. This, in turn, would have decreased water turbulence by reducing wind stress and also reduced the disturbance of lake sediment, therefore leading to a reduction in turbidity. It is also likely that *Staurosira construens* forma *venter* then became the dominant facultative plankton because of its higher tolerance for EC, which is thought to have increased within the basin due to the onset of river regulation and the associated increase in irrigation dependant agriculture (Allison and Schonfeldt, 1989).

7.3.4 Zone 4C

A further major switch in the fossil diatom assemblage is completed in zone 4C. At approximately 22 cm, abundance of *S. construens* forma *venter* starts to decrease and the taxon is replaced by a combination of epiphytic taxa (*C. placentula* and *E. sorex*), and also the planktonic *Aulacoseira granulata* and facultative planktonic *Pseudostaurosira brevistriata*. This change is almost definitely due to an increase in nutrient concentrations. Following the Second World War the Sunraysia region, particularly around Mildura, was developed as part of the War Service Land Settlement Scheme (Menzies, 1983). The increase in both urban and rural dwellings and associated services would have resulted in a substantial increase in effluents and agricultural runoff entering the river system. It was also a time of economic boom and the food products of the floodplains, most importantly fruits and vegetables were in great demand (Sinclair, 2001).

The increases in *C. placentula* and *E. sorex* most probably reflect an increase in aquatic vegetation abundance, which may have arisen due to elevated nutrient levels. *Aulacoseira granulata* and *Pseudostaurosira brevistriata* also increase during this section of the sediment core. Both taxa, but particularly *A. granulata*, have a narrower nutrient preference than *S. construens* forma *venter*, with *A. granulata* predominantly being abundant in eutrophic to hypertrophic waters while *S. construens* forma *venter* is often found in oligotrophic waters (Bennion *et al.*, 2000). Core 2 from Lake Alexandrina demonstrated that *P. brevistriata* is likely to be more competitively advantaged than the other small forms of *S. construens* forma *venter* and *S. pinnata* in high nutrient conditions.

The presence of *Actinocyclus normanii* indicates a possible further rise in EC during this time period (discussed further below in section 7.3.5).

7.3.5 Zone 4B

There is a further decrease in *S. construens* forma *venter* throughout this zone, and also a small decrease in *A. granulata*. The abundance of littoral diatoms is at its highest during this zone, perhaps suggesting that this was the period of maximum aquatic vegetation growth. There also appears to have been quite substantial changes in EC levels as indicated by a steady increase in *Tabularia fasciculata* and sustained

populations of *Actinocyclus normanii*. *Actinocyclus normanii* is common in brackish to fresh and brackish waters and appears to have the narrowest EC tolerance of all taxa present in this core (*C. placentula* is present in waters with much higher EC but is also tolerant of much lower EC). The abundance of *A. normanii*, starts to increase in the late 1960s, reaching a peak in the late 1970s and then decreasing gradually towards the surface, and is virtually absent in the surface sediments. Relating this to historical events, there was a severe drought in south - eastern Australia in 1967 and 1968 resulting in widespread increases in salinity in both the main Murray - Darling channels and also in most of the tributaries. This led to increased awareness of the problems of salinisation with consultants producing the Murray Valley Salinity Investigation for the River Murray Commission during 1968 (MDBMC, 1999).

7.3.6 Zone 4A

The final major change in the diatom community occurs in the top 5 cm of sediment, which would represent approximately the last 8 years (since 1989). *Staurosirella pinnata* and *Pseudostaurosira brevistriata* both increase substantially during this time period, while *A. granulata*, *C. placentula*, and *E. sorex* decrease. This change suggests that there may have been a decrease in nutrient concentrations during this time period. The decreases in *E. sorex* and *C. placentula* probably also indicates a reduction in aquatic vegetation, particularly submerged vegetation. This probable decrease in nutrients within the last decade correlates to increasing public awareness of the relationship between nutrients and water quality, and also progress in effluent disposal legislation which has seen many primary and secondary treatment plants along the river channel replaced with tertiary plant systems (Sinclair, 2001).

It is interesting that *T. fasciculata* continues to increase in abundance throughout this upper 5 cm, suggesting a sustained increase in EC. However, it should also be remembered that *T. fasciculata* has a much wider tolerance to EC than *A. normanii* and so can not be used as confidently in EC reconstructions without additional evidence.

7.3.7 Summary

Analysis of the sediment core from Lake Cullulleraine provided the following insights into the lakes limnological history –

1) Lake formation of Lake Cullulleraine, resulting in lower turbidity, and a concurrent increase in EC levels within the lake, most likely led to *S. construens* forma *venter* replacing *A. subarctica* forma *subarctica* as the dominant lake diatom taxon.

2) The stabilisation of the lake basin after the late 1930s also resulted in the development of littoral vegetation, as reflected by an increase in littoral flora.

3) During the post Second World War period (1948 onwards) there was a substantial shift in the diatom

community in response to increasing nutrient concentrations within the lake. Consequently, aquatic vegetation growth also increased.

4) Starting in the 1960s, and possibly peaking in the late 1970s, there was a further increase in salinity within the lake.

5) The diatom sequence in the last 10 years shows a likely decrease in nutrient concentrations.

Chapter 8 - Conclusions

8.1 Conclusions generated from the modern diatom study

Modern diatom assemblages were collected and analysed from more than 100 samples. From these samples, three data sets were generated; one comprised of lake samples, one of stream samples, and a combined data set. Both the lake and stream data sets yielded similar numbers of diatom taxa, with few taxa only present in either environment. Ordination analysis of the data, which related the diatom assemblages to water physico-chemistry, demonstrated that the most influential variable in all three data sets was pH. This result establishes the important role of pH in determining diatom assemblages in south east Australian aquatic systems, and perhaps, the lesser role that EC and nutrients play.

A transfer function model was developed for pH with a correlation coefficient and RMSEP (used as a measure of the models efficacy and accuracy) that was comparable with other key published studies. There is potential for this model to applied to other sites from within the Murray – Darling Basin, providing an accurate reconstruction of pH throughout the circumneutral to alkalibiontic range.

In order to gain more information about the EC and nutrient preferences of the diatom taxa that were abundant in the fossil records, patterns of abundance were examined across the environmental gradients within the modern study, and then compared to published results from more than twenty key studies. There were some obvious common patterns of ecological preference (such as preference for fresh or brackish waters), but the patterns generally only allowed limited precision in inferences about likely past limnological conditions. Based on these results, it would seem that many of the more common diatom taxa within the lower Murray – Darling Basin have wide EC and nutrient tolerances. This is most probably due to diatom communities, both historical and contemporary, having adapted to the highly variable nature of this aquatic system.

8.2 Potential of quantitative palaeolimnological research in Australia

Quantitative palaeolimnological methods are an exciting advancement and have enormous potential to refine our understanding of past aquatic environments. However, as noted in Chapter 5, accurate and reliable quantitative reconstruction will not be achieved until there is greater symmetry within the body of knowledge about diatom autecology, and also the likely spatial scale of this symmetry. Additionally, the postulated ubiquitous nature of diatoms needs to be examined, and also whether this ubiquity relates to autecology as well as taxonomy.

Quantitative palaeolimnology is regularly and confidently utilized in many regions of the world, and the results of this study should not cast doubt on the usefulness of the method. Sediment cores from lakes in Europe that have a long history of water quality monitoring have been used to validate transfer function

models (with reconstructed values compared with measured values) and correlations can be very reasonable (see Bennion *et al.*, 1995). In Australia, TP reconstruction of Burrinjuck Reservoir, NSW, a mesotrophic deep water system, using a transfer function developed largely from characteristically similar reservoirs (Tibby, 2000), also compares well to documented TP history. Additionally, when the pH transfer function developed in this study was used to quantitatively reconstruct pH in Lake Alexandrina (cores 1, 2 and 3), the reconstructed pH of the recent sediments matched measured pH over the same time period (with an error of 0.2 pH units). What this study does demonstrate is that caution is needed, and perhaps other reconstruction methods need to be considered, such as the analysis of additional limnological indicators (i.e.: ostracods, cladocera, chironomids), particularly in complex, lowland systems.

To summarise, it appears that there is a place for quantitative ecological reconstruction in palaeolimnological research, but that model validity decreases with system complexity. Aquatic systems such as crater lakes and reservoirs, that are usually deep and clear with relatively small littoral zones are not as ecologically complicated as the shallow, turbid, highly eutrophic systems that are characteristic of the Murray - Darling Basin. Additionally, the former systems tend to be dominated by planktonic diatoms, which generally, have narrower responses to environmental forces than littoral diatoms (Jones and Juggins, 1995). It would therefore seem that the more complex an aquatic system the greater the need for multiple reconstruction tools. In the case of the lower Murray River, system complexity is immense due to very variable flows, numerous inflows and outflows, high levels of turbidity and nutrients, and, in most regions, an extensive littoral zone.

There is a continued need for the development of diatom nutrient, EC and pH transfer functions in Australia, as not all systems are as complex as Lakes Alexandrina and Cullulleraine. Transfer function generation may also result in refinement in some of the autecological information. This is particularly important for some of the previously discussed *Aulacoseira, Fragilaria, Staurosira, Staurosirella* and *Pseudostaurosira* genera, as these taxa appear to dominate many of the modern assemblages (not just in the Murray - Darling Basin) throughout south eastern Australia, and also fossil records.

8.3 Summary of the palaeolimnological study

This study examined the palaeolimnology of two lakes located within the lower Murray River region – Lake Alexandrina and Lake Cullulleraine. Lake Alexandrina, in particular, plays a critical role in both contemporary and palaeolimnological understanding of the Murray – Darling Basin, as the mouth of the river represents an accumulation of all upstream processes. However, by the very nature of its location, Lake Alexandrina is a highly multifarious system due to the influence of tidal forces and also extremely variable freshwater inflows (prior to river regulation). The lakes immense size (relative to other Australian systems), coupled with its shallow depth, adds to this complexity by affecting the integrity of sediment records. The palaeolimnological information gained from this study, although perhaps not as clearcut as information

gained from less complex systems upstream, does provide some background data and hopefully also provides the impetus for further, more detailed, research.

Some of the more pertinent conclusions include evidence of river / lake dryness during the past 7000 years. The most extensive phases of implied lake dryness are between 5000 and 2000 years B.P., which concurs with the findings of Cann *et al.* (2000) which show this period of time to be drier than the present, for this region. There is also evidence of high pH events in Lake Alexandrina (> 9.0), as demonstrated by the presence of diatom taxa with high pH tolerance associated with dissolution of other, less tolerant, taxa.

Shifts in representation of littoral and planktonic taxa within the three Lake Alexandrina records suggest change in the abundance of aquatic vegetation, both in the littoral zone and submerged. This is particularly evident in core 3 where the diatom community suggests a substantial increase in aquatic vegetation in the post non – indigenous settlement period. There is also evidence for fluctuations in turbidity in all cores at various times in the past, particularly in core 1 where turbidity appears to have increased proceeding the arrival of non- indigenous Australians. In reference to nutrients, most of the taxa present in all three Lake Alexandrina cores are indicative of eutrophic to hypertrophic conditions, suggesting that Lake Alexandrina has always been a naturally eutrophic system. However, core 3 also shows a clear increase in nutrients in the last 200 years.

The sediment core from Lake Cullulleraine provides a detailed, high resolution record of changes since river regulation. The most pertinent conclusions that can be made from this palaeolimnological record include evidence for a reduction in lake (and possibly river) turbidity, and also an increase in littoral and submerged vegetation, coinciding with the period of time when the Murray – Darling Basin regulation system became fully operational. There is also evidence for an increase in nutrients in the post Second World War period, and an increase in salinity during the 1960s and 1970s.

8.4 Palaeolimnological potential of the lower Murray River

Walker and Thoms (1993) discuss three possible management strategies for the lower Murray River (after Pressey, 1990); a) preservation of the status quo, b) special purpose management such as manipulating selected wetlands for specific ecological targets, and c) restoration of the natural environment. Maintaining present ecological conditions is both not possible, because the system has yet to establish an equilibrium, or feasible because of the unsustainability of the system. Perhaps the best possible management strategy involves a combination of b) and c). Although it is not possible to restore the lower Murray River to its baseline condition, specific ecological manipulation techniques could be used to achieve restorative targets which may fall somewhere between contemporary conditions and pre - impact conditions.

However, this thesis has highlighted that the determination of baseline conditions is not a straightforward process, with the lower Murray River basin proving a multifaceted and complex system. The palaeolimnological conclusions of this study, although important in contributing to the assessment of both

baseline conditions and also the degree of limnological change as a result of impact events, could not be quantified (with the exception of pH). In fact, this study highlighted some inherent problems in the generation and application of quantitative transfer functions to complex, lowland systems.

The most common fossilised diatoms in both Lakes Alexandrina and Cullulleraine were ecologically tolerant diatoms, that is, diatoms that are generally able to thrive in a wide range of limnological conditions. However, some external factor, chemical or physical, must have been driving the population shifts. The modern plankton surveys of this study, and also of Hötzel and Croome (1996), demonstrate that different species and even subspecies of the genera *Aulacoseira, Fragilaria, Staurosirella, Staurosira and Pseudostaurosira* vary dramatically in different sections of the river channel. The fossil records of Lakes Alexandrina and Cullulleraine also show that shifts in these taxa are not random, but instead, are systematic. Consequently, and as previously mentioned, more research needs to be undertaken to refine the understanding of the ecological requirements of some of these common taxa.

Complementary to research on the modern dynamics of these taxa, there is also the need for greater analysis of the fossil records in which they occur. Palaeolimnological emphasis is commonly on examining the contemporary dynamics of diatom taxa and then relating this information to environmental parameters. However, there is the potential for fossil records to be more relevant in terms of providing information about population dynamics and autecology as they represent the end result of taphonomic processes. However, for this to be an option, there is firstly the need to locate sites with well documented physical and / or ecological histories. For example, successful future palaeolimnological research on the Murray - Darling Basin will need to have a more comprehensive understanding about the relationship between the *Aulacoseira* spp., channel flow and turbidity. Potentially, an intensively dated fossil record obtained from a site with a long history of flow data could provide substantially more information than a modern study. Obviously, one of the main advantages of utilising the fossil record to increase our understanding of autecology is the longevity of the study. Some sections of the Murray - Darling basin have more than 40 years of flow data, which if correlated to a fossil record, provides a very substantial time series study.

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Appendix 1

Measured water quality parameters – Lake data set

		pH	EC (µS/cm)	DO (mg/L)	TP (µg/L)	TKN (µg/L)	Turbidity (NTU)	Temp (°C)	%Alk	%Cl^-	%SO_4^{2-}	%Ca^{2+}	%Mg^{2+}	%Na^+	%K^+
1	Wetland 1 Golden Grove, SA	7.6	700	12.2	440	1200	45	12.2	27.7	71.8	0.5	23.2	17.3	55.3	4.3
2	Golden Grove Lake, SA	7.3	420	8.5	85	225	15	11	23.5	72.8	3.7	9.7	42.8	44.0	3.6
3	Wetland 1 Greenwith, SA	8.3	200	9.2	160	745	26	11.5	43.6	51.5	4.9	17.8	24.9	50.2	7.0
4	Wetland 1 Regent Gardens, SA	7.4	310	7.9	140	455	30	11	60.3	35.4	4.3	18.1	17.4	55.8	8.7
5	Paddocks wetland, SA	7.2	200	11.5	70	180	110	12.2	40.4	54.0	5.6	11.5	30.3	51.1	7.1
6	Barker Inlet wetland, SA	8.3	10190	14.6	275	920	85	11	56.4	42.1	1.6	12.1	19.8	58.3	9.8
7	Greenfields wetland, SA	7.8	2620	13.2	220	1120	30	14	58.6	39.1	2.3	11.8	21.8	58.6	7.8
8	Montague Farm wetland, SA	7.3	800	9.6	115	750	25	14.3	28.1	40.7	31.1	18.2	22.9	52.3	6.7
9	Noarlunga wetland, SA	7.8	2280	8.9	70	215	48	12.5	21.9	70.0	8.0	7.0	22.3	69.8	0.9
10	Mt Bold Reservoir, SA	7.3	880	8.5	85	120	16	13.1	45.0	54.1	0.9	26.2	19.7	50.4	3.8
11	Wittunga Lake, SA	6.9	570	7.8	40	45	122	11.2	22.4	74.0	3.6	10.9	45.3	40.0	3.9
12	Belair NP Lake, SA	7.1	322	8.6	105	340	78	12.3	21.6	67.0	11.4	12.0	23.6	62.4	2.0
13	Kangaroo Creek Reservoir, SA	7.5	960	10.6	50	135	40	14.6	44.4	53.7	1.8	24.6	21.2	47.7	6.4
14	Chains of Ponds 1 Reservoir, SA	7.8	747	11.5	55	80	63	14.5	27.7	41.2	31.2	18.1	21.5	49.8	10.6
15	Chain of Ponds 2 Reservoir, SA	7.1	738	8.4	55	140	115	15.1	27.7	41.2	31.2	18.1	21.5	49.8	10.6
16	Kersbrook Reservoir, SA	7.5	922	9.5	90	285	84	13.9	46.1	51.8	2.1	24.9	21.9	46.9	6.3
17	Barossa Reservoir, SA	7.4	820	10.8	80	340	26	14.8	47.5	50.4	2.1	29.1	23.7	39.6	7.6
18	Little Para Reservoir, SA	7.7	760	10.1	110	255	18	15.3	37.6	59.5	2.9	17.8	29.2	44.3	8.7
19	Windy Lake, Vic	8.9	6100	7.6	210	210	68	22.1	17.8	73.4	8.8	3.6	21.2	74.7	0.5
20	Lake Cooper, Vic	8.8	4700	6.6	250	250	467	21.8	18.4	73.1	8.6	3.4	22.3	73.7	0.6
21	Greens Lake, Vic	8.6	2110	12.5	70	70	95	10.05	22.3	69.6	8.1	6.7	22.5	69.9	0.8
22	Lake Boort, Vic	8.3	2850	7.8	80	90	34	13	19.9	76.3	3.9	8.9	26.5	63.0	1.6
23	Lake Lyndger, Vic	8.3	6350	6.9	1000	50000	50	14.5	16.0	84.0	0.1	6.8	24.9	66.5	1.8
24	Lake Leagher, Vic	8.4	1250	7.1	350	12280	155	12	24.5	68.8	6.7	13.6	28.6	55.9	2.0
25	Little Lake Meering, Vic	8.7	1830	8.1	60	360	19	12.5	27.2	68.6	4.2	10.6	32.2	56.0	1.3
26	Lake Lookout, Vic	8.9	5300	8.9	120	1440	149	12.5	15.5	77.3	7.3	4.2	21.2	73.5	1.1
27	Lake Bael Bael, Vic	8.4	3100	8.4	160	2560	20	13.5	17.7	79.4	2.8	7.8	25.2	65.8	1.2
28	Little Lake Charm, Vic	7.3	1150	6.3	170	2890	87	12	17.3	71.9	10.8	10.2	22.1	66.4	1.4

		pH	EC (µS/cm)	DO (mg/L)	TP (µg/L)	TKN (µg/L)	Turbidity (NTU)	Temp (°C)	%Alk	%Cl^-	%SO_4^{2-}	%Ca^{2+}	%Mg^{2+}	%Na^+	%K^+
29	Reedy Lake, Vic	7.2	660	6.7	240	5853	85	12.5	19.2	69.4	11.4	11.8	26.4	60.6	1.3
30	Kangaroo Lake, Vic	8.7	520	10	100	90	69	12.5	22.7	66.3	11.0	12.4	24.2	61.6	1.9
31	Racecourse Lake, Vic	8.2	580	11.1	190	3610	109	12	19.0	70.6	10.4	11.4	23.5	63.7	1.4
32	Lake Charm, Vic	8.7	4500	7.4	80	640	34	12.5	12.3	72.3	15.5	7.3	22.2	69.4	1.1
33	Lake Boga,Vic	8.7	2900	6.8	200	4000	39	15.4	22.8	71.8	5.4	10.7	20.2	67.4	1.7
34	Lake Mannaor, Vic	8.8	4600	8.7	310	310	163	21.7	18.1	76.9	5.0	7.7	26.5	64.0	1.8
35	Lake Bitterang, Vic	7.9	650	5.3	310	310	40	28.3	48.9	50.4	0.6	26.1	19.2	50.1	4.6
36	Lake Hattah 1, Vic	7.5	610	8.1	110	80	37	16.7	39.2	56.7	4.1	18.4	23.2	51.8	6.6
37	Lake Walpeup, Vic	8	840	8.3	30	30	16	25.2	29.5	70.1	0.3	23.9	18.0	57.7	0.4
38	Suspicious Dog, Vic	7.8	440	9	330	720	56	15.4	60.2	35.5	4.3	18.1	17.3	55.9	8.7
39	Dock lake, Vic	9	1610	10.6	20	40	15	13.7	29.5	68.8	1.7	13.0	15.2	70.2	1.6
40	Green Lake, Vic	9.4	450	12.7	100	70	51	17.3	3.2	92.6	4.2	7.6	34.1	55.3	3.1
41	Wargon Basin, Vic	8.5	4800	9.9	90	1400	40	18.7	18.4	76.7	4.9	7.6	27.2	63.4	1.8
42	Lake Hattah 2, Vic	8	640	8.1	170	2000	170	16	49.7	49.7	0.6	25.4	19.0	50.8	4.8
43	Bullock Swamp, Vic	8.7	9330	11.4	320	7000	97	14.3	24.5	68.8	6.7	13.6	28.6	55.9	2.0
44	Calender Swamp, Vic	9	2430	14.9	400	3800	110	18	22.8	71.8	5.4	10.7	20.2	67.4	1.7
45	Cardross Basin, Vic	9.3	5367	10.5	50	1000	20	19	18.4	76.7	4.9	7.6	27.2	63.4	1.8
46	Karadoc Basin, Vic	8.2	4100	10.7	100	1000	107	18	18.4	76.7	4.9	7.6	27.2	63.4	1.8
47	Kings Billabong, Vic	8	313	12.3	70	500	20	17	60.2	35.5	4.3	18.1	17.3	55.9	8.7
48	Mullaroo Lagoon, Vic	8.1	330	9.5	60	100	73	16	60.2	35.5	4.3	18.1	17.3	55.9	8.7
49	Pysche Bend Lagoon, Vic	9	9000	12.3	70	1400	40	18.7	24.5	68.8	6.7	13.6	28.6	55.9	2.0
50	Walpolla Basin, Vic	8.2	310	9.8	180	400	63	17	60.2	35.5	4.3	18.1	17.3	55.9	8.7

Measured water quality parameters – Stream data set

		Shade (%)	pH	Temp (°C)	DO (mg/L)	EC (µS/cm)	SRP (µg/L)	TP (µg/L)	NOX (µg/L)	TKN (µg/L)	Turb. (NTU)	%Alk	%Cl^-	%SO_4^{2-}	%Ca^{2+}	%Mg^{2+}	%Na^+	%K^+
51	Breakneck R., SA	70	7.4	16.4	10.1	605	5	5	10	240	0.8	38.6	51.1	10.3	18.6	33.7	46.1	1.6
52	Italowie Ck., SA	75	8.0	28	11.9	1840	5	5	10	260	0.23	5.4	89.9	4.7	3.2	21.0	74.3	1.5
53	Oraparinna Ck., SA	90	8.1	17.9	8.9	1860	5	5	30	150	0.36	29.8	40.8	29.4	33.5	29.9	35.2	1.5
54	Deep Ck., SA	5	7.8	16.6	10.2	3010	5	5	3000	110	0.3	39.8	48.9	11.3	34.0	30.6	35.2	0.2
55	Rocky R., SA	80	8.1	13.8	10.9	461	5	5	10	130	1.2	16.9	75.4	7.7	15.2	20.8	62.6	1.4
56	Eight Mile Ck., SA	0	7.9	16.3	12	1170	5	6	5150	160	0.25	6.2	89.3	4.4	3.1	20.8	74.7	1.4
57	Middle R. (1), SA	60	7.1	17.1	6.6	2100	5	6	10	250	0.27	43.0	51.2	5.9	33.0	21.4	44.5	1.2
58	Mosquito Ck., SA	20	8.0	13.5	8.9	2680	5	6	600	310	1.3	4.1	91.2	4.7	7.1	25.5	66.6	0.8
59	Oraparinna Ck., SA	95	8.1	12.8	8.3	1890	5	6	10	240	0.2	28.3	65.2	6.5	24.6	18.9	55.8	0.7
60	Parachilna Ck., SA	10	8.4	16.3	8.5	2000	5	7	280	130	3.2	36.4	48.1	15.5	32.1	29.9	37.6	0.3
61	Drain M Callender, SA	5	8.8	18.2	16.4	2260	5	7	10	480	1.3	25.9	58.5	15.5	26.5	30.8	42.2	0.6
62	Yankalilla R. (1), SA	70	8.3	17.8	10.2	4130	5	7	10	460	0.44	24.2	68.7	7.0	12.9	19.9	66.4	0.8
63	Eight Mile Ck., SA	15	7.7	16.1	10	1130	5	8	5100	60	0.21	12.6	81.9	5.5	11.8	18.8	68.5	0.8
64	Brachina Ck, SA	50	7.9	14.8	7.2	2090	5	8	10	140	0.3	44.3	50.4	5.4	35.0	22.4	41.5	1.0
65	Parachilna Ck., SA	20	8.2	22	8.6	2270	5	8	10	90	0.59	30.7	55.2	14.2	26.3	28.2	45.1	0.4
66	Eregunda Ck., SA	60	8.1	12.4	11.6	3750	5	9	10	210	0.6	24.9	56.9	18.2	24.1	32.1	43.3	0.6
67	Little Para R., SA	95	8.0	15.7	9.2	1340	10	10	120	240	0.39	18.5	62.4	19.1	16.4	31.8	51.4	0.4
68	Wilpena Ck., SA	80	7.8	22.6	9.4	340	5	12	10	200	11	38.6	51.1	10.3	18.6	33.7	46.1	1.6
69	Stunsail Boom R., SA	60	7.3	18.7	9.2	2200	5	13	10	460	5.7	16.7	72.7	10.6	13.3	29.2	52.9	4.6
70	Inman R. (1), SA	45	7.7	18.4	7.3	2700	5	14	10	280	1.8	4.4	91.2	4.5	5.0	20.4	73.7	1.0
71	Marne R (1), SA	30	8.1	13.6	10.2	3850	5	14	10	400	0.55	16.0	74.2	9.9	18.7	22.4	58.2	0.7
72	Wakefield R, SA	95	8.3	14.2	7.5	5110	5	14	40	450	0.9	18.8	76.4	4.8	22.0	18.1	59.1	0.8
73	Aroona Ck., SA	50	7.8	21.3	5.7	2160	5	15	10	150	0.58	15.2	67.0	17.8	12.9	31.0	55.0	1.1
74	Coolawang Ck., SA	70	8.0	13.6	9.6	1040	5	15	10	470	5	30.8	53.5	15.7	16.1	19.4	63.9	0.6
75	Stunsail Boom R., SA	40	7.5	13.6	9	3690	5	18	10	510	2.2	10.7	81.8	7.4	7.0	16.2	75.2	1.5
76	Brownhill Ck., SA	95	7.8	15.5	7.2	885	8	19	40	130	3.1	3.2	90.6	6.2	4.4	19.3	75.4	1.0
77	Inman R. (2), SA	45	7.9	14.7	7.5	3930	5	21	10	240	1.5	56.6	35.4	8.0	27.5	38.8	32.7	1.0
78	Hindmarsh R., SA	20	8.3	23.2	11.9	1210	20	23	10	440	1.2	13.6	80.4	6.0	17.7	21.0	60.4	0.9
79	Finniss R., SA	50	7.5	15	7.6	697	5	24	20	320	3.4	25.4	72.1	2.5	14.1	17.1	67.5	1.2
80	Mosquito Ck., SA	20	8.4	13.3	12.1	2950	5	26	810	790	6.2	18.2	75.5	6.3	11.2	21.5	66.2	1.0

		Shade (%)	pH	Temp (°C)	DO (mg/L)	EC (µS/cm)	SRP (µg/L)	TP (µg/L)	NOX (µg/L)	TKN (µg/L)	Turb. (NTU)	%Alk	%Cl^-	%SO_4^{2-}	%Ca^{2+}	%Mg^{2+}	%Na^+	%K^+
81	Yankalilla R. (2), SA	20	8.0	16.7	8.5	3000	5	26	20	580	3	26.4	67.4	6.2	22.1	17.9	59.5	0.6
82	Marne R (2), SA	40	8.0	12.6	8.1	4600	5	31	10	630	0.69	12.6	83.0	4.4	13.5	20.3	65.5	0.6
83	Currency Ck., SA	70	7.5	18	7	3020	12	40	100	680	2	14.4	79.6	6.0	19.0	18.9	60.6	1.4
84	Yankalilla R. (3), SA	10	7.9	14.8	8.5	2040	9	52	10	580	8.5	12.9	83.3	3.8	13.8	22.5	62.6	1.0
85	Middle R. (2),	20	7.2	16.2	9.5	6460	5	60	310	950	2.2	13.8	80.8	5.4	12.9	19.4	66.3	1.4
86	Waitpinga Ck., SA	60	8.2	14.7	8.9	5830	23	60	50	760	3.7	1.4	92.5	6.1	9.0	30.2	59.9	0.8
87	Torrens R, SA	30	7.8	18	11	1760	5	69	10	840	8.3	10.5	84.2	5.3	10.6	17.1	71.4	0.9
88	The Deep Ck. Waterfall, SA	90	7.9	12.7	10.8	2000	17	70	10	740	1.7	14.0	80.1	5.9	12.0	22.7	64.4	0.9
89	Murray R., Murtho, SA	5	7.8	22.9	10.6	446	5	83	20	620	46	13.7	80.7	5.6	6.0	12.9	80.2	0.9
90	Yankalilla R. (4), SA	50	7.7	15.7	8	2950	11	96	10	1410	4	22.0	66.8	11.2	14.0	25.2	58.5	2.4
91	Murray R, Woods Point, SA	5	8.0	23.6	9.4	715	21	104	40	810	48	16.7	82.6	0.7	11.6	19.1	69.0	0.3
92	Naracoorte Ck. (1), SA	70	7.8	13.7	8.8	2590	43	189	300	1870	11	15.7	71.0	13.3	12.2	20.3	65.7	1.8
93	Naracoorte Ck. (2), SA	80	8.1	12.2	7.2	4390	5	190	60	2220	31	25.7	69.2	5.0	23.1	20.3	55.6	1.0
94	Cooper Ck. (1), SA	5	7.4	12.4	10.1	98	73	359	60	730	450	21.2	72.7	6.1	17.1	23.6	58.2	1.1
95	Morambro Ck. SA	80	8.5	15.3	9.8	322	30	40	20	2600	360	61.9	16.4	21.8	31.4	23.2	35.6	9.8
96	Cooper Ck (2)	0	8.1	25.4	12.4	217	99	49	250	800	370	71.7	19.4	8.9	34.3	17.3	42.9	5.5
97	North Para R. (1), SA	40	8.4	18.9	11.5	4150	130	49	130	9400	4.6	42.2	44.3	13.5	34.4	19.4	39.9	6.2
98	North Para R. (2), SA	30	8.4	12.6	10.8	4600	500	598	40	620	0.6	17.2	73.4	9.4	14.5	24.6	59.8	1.0
99	Mt. Barker Ck., SA	20	8.4	14.6	11.7	1660	500	821	1900	5820	21	15.7	75.9	8.4	14.3	23.8	60.7	1.1
100	Inman R (2), SA	20	7.6	15.9	2.4	3130	5400	5860	310	23600	1.4	18.1	76.3	5.6	18.1	21.4	58.9	1.6
101	Mt. Barker Ck., SA	15	7.6	13	2.1	2310	8900	9480	5470	12300	10	17.8	75.0	7.2	13.6	18.0	66.0	2.4

Appendix 2

Diatom counts from lake data set

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Achnanthes brevipes	0	2	0	0	2	0	0	0	2	0	0	0	2	0	0	0	0	0	0	0
Achnanthes exigua	0	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthes oblongella	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthes pericava	0	0	0	0	0	0	0	0	0	20	0	0	0	0	0	0	0	0	0	0
Achnanthidium minutissimum	53	19	5	42	0	54	0	2	14	0	180	62	0	130	0	4	85	0	0	0
Actinocyclis normanii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphora coff var. borealis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphora coff var. coff	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	34	0	0
Amphora cognata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphora libyca	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphora pediculus	0	0	0	4	4	0	0	8	10	0	0	0	4	0	0	0	0	0	0	0
Amphora veneta	2	2	0	8	2	0	0	0	0	18	0	10	0	0	21	2	0	0	6	0
Anomoneis spaerophora	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulacoseira ambigua	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulacoseira subarctica forma *subborealis*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulacosiera granulata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulacosiera italica	0	11	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bacillaria paxillifer	0	0	0	0	0	8	44	4	0	0	0	0	0	0	0	4	0	6	0	0
Bercalia rutilans	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Climaconeis sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cocconeis placentula	0	0	0	0	4	0	0	0	0	0	0	0	16	72	0	0	0	0	0	0
Craticula accomoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Craticula halophila	6	0	0	0	0	0	0	0	0	50	0	2	0	0	0	0	0	2	2	0
Cyclotella atomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyclotella meneghiniana	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	8	0	0	69	39
Cyclotella michigiana	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Cyclotella rossii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyclotella stelligera	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymatapleura solea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella amphicephala	0	0	0	0	0	0	0	0	4	0	20	0	0	0	0	0	0	2	0	0
Cymbella cistula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella tumida	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diatoma tenuis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	162	2	0	0
Diatoma vulgaris	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Diploneis elliptica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diploneis ovalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diploneis parma	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Encyonema gracilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Encyonema minutum	0	4	0	0	0	8	0	0	0	0	0	12	0	0	0	0	0	0	0	0
Encyonema silesiacum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Encyonopsis microcephala	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Entomoneis alata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	23	4
Epithemia adnata	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Epithemia sorex	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Epithemia turgida	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eunotia bilunaris	0	0	0	0	0	19	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eunotia serpentina	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fallacia pygmaea	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fallacia tenera	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Fragilaria cap. var. gracilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fragilaria cap. var. rumpens	13	0	0	0	0	0	6	0	0	0	0	4	0	0	0	0	0	2	0	0
Frustulia rhomboides	0	12	0	0	0	0	0	2	0	0	0	0	34	0	0	0	0	2	0	0

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Frustulia vulgaris	0	0	0	0	0	36	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema acuminatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	8	0	0
Gomphonema affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	12	0	0
Gomphonema augur	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Gomphonema clavatum	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema gracile	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Gomphonema minutum	0	0	0	0	0	0	0	6	0	0	20	33	0	0	0	0	0	0	0	0
Gomphonema olivaceum	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema parvulum	0	6	0	2	0	22	4	14	0	0	20	4	0	44	0	18	0	16	0	0
Gomphonema truncatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	30	0	0
Gyrosigma attenuatum	0	0	0	2	0	0	0	14	0	0	0	0	0	0	0	0	0	0	4	0
Gyrosigma parkerii	0	0	0	2	0	0	0	8	0	0	0	0	0	0	0	0	0	0	0	0
Gyrosigma spencerii	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	4	0
Hantzschia amphioxys	0	4	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	4	2
Hippodonta capitata	0	0	120	2	0	0	0	0	0	16	0	24	0	0	0	2	0	0	0	0
Luticola mutica	0	68	5	0	2	0	0	45	0	0	0	0	0	0	0	0	0	0	0	0
Mastagloia smithii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Melosira varians	0	0	0	0	0	0	40	0	0	0	0	0	16	34	115	305	0	90	0	0
Navicella pusilla	0	0	18	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Navicula angusta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula capitidigorada	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cari	0	12	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	2	0	0
Navicula cincta	0	0	0	0	0	54	13	6	8	0	0	0	116	0	0	0	0	0	0	0
Navicula cincta var. minuta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula clementis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cocconeiformis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cryptocephala	0	10	0	0	136	0	0	88	39	0	0	2	4	0	0	4	0	2	8	0
Navicula cryptotenella	4	45	22	72	0	55	10	0	0	0	0	0	0	0	0	0	0	2	16	0
Navicula cuspidata	0	0	0	7	16	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Navicula erifuga	0	0	0	2	0	0	0	18	0	0	0	0	0	0	0	0	0	0	0	0
Navicula gregaria	0	2	33	0	0	0	0	0	10	55	0	8	14	0	0	0	0	2	0	0
Navicula incertata	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula indifferens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula lanceolata	0	0	0	0	0	0	0	0	0	114	0	0	26	0	0	0	0	0	0	0
Navicula leptostriata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula menisculus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0
Navicula muraliformis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula perminuta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula phyllepta	0	32	0	10	2	0	4	12	16	0	0	2	0	0	0	0	0	0	0	0
Navicula radiosa	16	0	0	0	0	0	0	0	0	0	0	2	4	0	0	0	0	0	0	0
Navicula recens	0	0	0	0	0	0	2	6	0	0	0	8	0	0	0	0	0	0	0	0
Navicula rhynchocephala	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula salinarum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula spicula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	11	132
Navicula veneta	7	4	45	2	44	12	13	44	44	18	0	22	6	0	26	0	0	0	0	0
Navicula viridula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia acicularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Nitzschia agnita	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
Nitzschia amphibia	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia archibaldii	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	2	0	0	0	0
Nitzschia bacillum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia calida	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia capitellata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia clausii	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia compressa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia constricta	18	0	0	4	2	0	2	4	20	0	0	0	4	0	0	0	0	0	0	0
Nitzschia desortorum	125	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia dissipata media	0	0	0	0	0	0	0	0	0	0	0	0	8	0	0	0	0	0	0	0

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Nitzschia filiformis	0	27	0	0	0	55	2	10	2	0	57	20	2	0	83	20	0	0	0	0
Nitzschia fonticola	11	6	0	4	43	0	0	4	6	0	0	0	0	0	0	0	0	0	6	0
Nitzschia frustulum	24	0	6	0	0	0	0	2	12	0	0	0	0	0	0	0	0	0	55	43
Nitzschia gracilis	0	0	0	0	8	0	0	4	0	0	0	0	0	0	0	0	0	0	32	6
Nitzschia heurfleuriana	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia hungarica	10	0	0	4	0	0	4	2	6	0	0	0	0	0	0	0	0	0	4	4
Nitzschia inconspicua	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
Nitzschia lacuum	28	0	3	2	0	0	4	2	0	0	0	0	0	0	0	2	0	0	6	38
Nitzschia levendensis	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia linearis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0
Nitzschia littoralis	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia microcephala	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia palacea	0	2	5	5	0	0	0	10	0	0	0	0	0	0	0	0	0	6	32	4
Nitzschia palea	57	12	0	2	20	12	0	22	8	0	0	0	2	0	0	0	0	0	20	2
Nitzschia pusilla	0	6	5	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia reversa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	2
Nitzschia sigma	0	0	0	2	0	0	0	4	0	0	0	0	2	0	0	0	0	0	0	4
Nitzschia subacicularis	2	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	2	34
Nitzschia supralittorea	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia tubicola	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia valdecostata	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
Nitzschia valdestriata	0	0	0	0	0	0	0	0	0	0	0	0	12	0	0	0	0	0	0	0
Opephora olsenii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planothidium delicatulum	0	4	0	117	50	0	2	4	92	15	0	2	0	0	0	4	0	0	30	0
Planothidium lanceolatum	0	2	0	0	0	0	0	0	0	0	0	0	2	0	0	2	0	0	0	0
Pleurosigma elongatum	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0
Pseudostaurosira brevistriata	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	28	0	0	0
Rhoicosphenia abbreviata	0	0	0	0	0	0	0	0	0	0	0	0	130	0	0	14	0	0	0	0

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Rhopalodia brebisonii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhopalodia gibba	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhopalodia musculus	0	0	0	0	0	0	6	0	4	0	0	0	0	0	0	0	0	0	0	0
Sellaphora pupula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stauroneis obtusa	0	11	0	0	20	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
Stauroneis pachycephala	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurosira con. var. venter	0	0	0	0	0	0	0	0	2	0	100	46	2	0	0	8	0	0	0	0
Staurosira construens	27	0	0	0	0	0	20	0	0	0	0	22	0	0	0	0	0	0	0	0
Staurosira elliptica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurosirella pinnata	2	0	0	0	0	0	0	0	0	0	0	26	0	0	0	0	0	0	0	0
Stephanodiscus hantzschii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella angusta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0
Surirella brebisonii	0	0	0	0	2	0	4	0	0	0	0	0	0	0	0	10	0	6	4	0
Surirella gracilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella ovalis	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0
Surirella patella	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella robusta	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	4
Synedra acus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synedra acus angustissima	18	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
Synedra rumpens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	26	0	8	0	0
Synedra ulna	0	0	0	2	0	0	4	14	0	0	15	0	0	0	0	2	0	0	0	0
Tabellaria flocculosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tabularia fasciculata	0	0	0	0	0	0	97	24	2	0	0	18	0	0	45	2	0	110	0	0
Thalassiosira baltica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Thalassiosira fluvialis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Thalassiosira weisflogii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tryblionella calida	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0

	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40
Achnanthes brevipes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthes exigua	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthes oblongella	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	12
Achnanthes pericava	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthidium minutissimum	0	2	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Actinocyclis normanii	79	0	0	0	0	0	0	0	0	10	0	0	101	0	0	0	0	0	0	0
Amphora coff var. borealis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	18
Amphora coff var. coff	0	0	2	2	2	2	2	0	0	0	0	0	0	0	0	0	4	2	0	0
Amphora cognata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphora libyca	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphora pediculus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphora veneta	2	28	65	0	0	0	0	2	0	0	0	0	0	2	0	0	0	2	0	7
Anomoneis spaerophora	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulacoseira ambigua	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulacoseira subartica var. subborealis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	18	91	0	0	0	38
Aulacosiera granulata	8	0	2	2	6	0	0	41	250	90	160	22	2	2	45	85	0	90	12	0
Aulacosiera italica	0	0	0	0	2	0	0	0	0	0	0	0	0	0	8	15	0	0	0	4
Bacillaria paxillifer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bercalia rutilans	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Climaconeis sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cocconeis placentula	0	42	20	42	2	0	0	18	0	18	4	4	10	2	0	0	54	0	60	20
Craticula accomoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Craticula halophila	10	2	2	0	0	4	0	0	0	0	0	0	0	0	2	0	0	0	0	0
Cyclotella atomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyclotella meneghiniana	33	32	0	102	50	50	163	6	0	25	6	134	30	117	52	8	0	10	39	0
Cyclotella michigiana	0	34	0	0	0	2	6	0	0	2	0	0	0	0	8	0	0	0	0	0
Cyclotella rossii	0	0	28	28	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyclotella stelligera	20	0	2	10	2	55	34	5	0	19	4	18	14	10	0	30	0	2	4	50
Cymatapleura solea	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40
Cymbella affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7
Cymbella amphicephala	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella cistula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella tumida	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0
Diatoma tenuis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diatoma vulgaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diploneis elliptica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diploneis ovalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diploneis parma	35	2	0	0	0	0	31	0	0	4	0	0	0	0	0	0	0	0	0	2
Encyonema gracilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	24
Encyonema minutum	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	2
Encyonema silesiacum	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	0
Encyonopsis microcephala	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Entomoneis alata	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
Epithemia adnata	0	4	0	46	0	6	0	2	0	2	0	2	6	10	0	2	6	0	22	0
Epithemia sorex	0	6	0	40	0	0	0	7	0	2	0	0	0	8	0	0	2	0	14	0
Epithemia turgida	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0
Eunotia bilunaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eunotia serpentina	0	0	0	0	0	0	2	2	1	0	3	0	2	0	0	0	0	0	0	0
Fallacia pygmaea	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fallacia tenera	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0
Fragilaria cap. var. gracilis	0	0	4	4	0	2	0	0	8	30	0	0	0	0	0	0	0	0	0	0
Fragilaria cap. var. rumpens	0	0	0	0	0	8	0	4	0	4	0	0	0	0	0	0	0	2	0	22
Frustulia rhomboides	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frustulia vulgaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema acuminatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema augur	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema clavatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40
Gomphonema gracile	0	4	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema minutum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema olivaceum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema parvulum	0	6	8	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	12	4
Gomphonema truncatum	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0
Gyrosigma attenuatum	4	4	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	26	6
Gyrosigma parkerii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gyrosigma spencerii	0	0	10	10	2	0	0	0	0	0	0	0	0	4	0	0	0	0	14	0
Hantzschia amphioxys	4	0	0	0	0	0	4	0	0	0	0	0	0	0	4	2	2	0	0	0
Hippodonta capitata	2	20	12	0	0	0	0	0	0	0	0	0	0	0	6	0	4	0	4	0
Luticola mutica	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	7
Mastagloia smithii	0	0	0	0	2	0	0	0	0	0	0	0	0	8	0	0	0	0	10	0
Melosira varians	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicella pusilla	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula angusta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula capitidigorada	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cari	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cincta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cincta var. minuta	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula clementis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cocconeiformis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cryptocephala	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0	14	0
Navicula cryptotenella	0	4	5	2	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cuspidata	0	6	2	2	0	0	0	0	0	0	0	0	0	0	2	0	0	2	4	0
Navicula erifuga	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula gregaria	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula incertata	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula indifferens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula lanceolata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula leptostriata	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0

	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40
Navicula menisculus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula muraliformis	0	0	59	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula perminuta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula phyllepta	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	4	0
Navicula radiosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	4	0	4	12
Navicula recens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Navicula rhynchocephala	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula salinarum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula spicula	0	0	0	0	0	4	0	0	0	0	0	0	0	2	0	0	0	0	0	0
Navicula veneta	0	0	28	2	0	0	0	0	0	0	0	0	0	2	2	0	50	2	8	0
Navicula viridula	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia acicularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia agnita	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia amphibia	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia archibaldii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0
Nitzschia bacillum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia calida	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
Nitzschia capitellata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia clausii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia compressa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia constricta	0	0	4	0	2	0	2	2	0	0	0	0	0	0	0	2	0	0	0	0
Nitzschia desortorum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	12
Nitzschia dissipata media	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia filiformis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia fonticola	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia frustulum	4	4	12	0	0	0	2	0	0	0	0	0	0	0	0	0	2	0	2	8
Nitzschia gracilis	0	0	0	0	0	4	0	0	0	2	0	12	0	5	0	5	0	0	0	0
Nitzschia heurfleuriana	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia hungarica	2	6	12	0	4	0	6	2	0	0	2	2	0	0	0	0	4	0	5	4
Nitzschia inconspicua	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia lacuum	36	14	16	4	4	38	6	4	0	0	0	40	2	89	2	0	6	0	2	14

	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40
Nitzschia levendensis	2	22	0	0	0	0	0	4	2	0	0	0	0	0	0	0	0	0	0	0
Nitzschia linearis	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	4	0	0	0
Nitzschia littoralis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	18	0
Nitzschia microcephala	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
Nitzschia palacea	2	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	4	0	0	10
Nitzschia palea	0	12	2	0	0	0	0	0	0	0	0	0	0	8	0	0	4	0	4	12
Nitzschia pusilla	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia reversa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia sigma	0	2	3	0	0	0	0	0	0	2	0	0	0	0	0	2	0	0	10	2
Nitzschia subacicularis	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia supralittorea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia tubicola	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia valdecostata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia valdestriata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Opephora olsenii	0	0	4	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planothidium delicatulum	10	0	26	0	4	20	2	0	0	10	0	12	2	0	2	2	0	0	2	0
Planothidium lanceolatum	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pleurosigma elongatum	2	12	4	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	2	0
Pseudostaurosira brevistriata	0	0	0	0	53	156	0	127	0	60	42	24	22	42	0	2	154	130	26	148
Rhoicosphenia abbreviata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhopalodia brebisonii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhopalodia gibba	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	14	26
Rhopalodia musculus	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Sellaphora pupula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stauroneis obtusa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stauroneis pachycephala	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurosira con. var venter	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurosira construens	0	51	0	2	2	0	0	0	0	18	0	0	0	0	0	0	0	0	0	2
Staurosira elliptica	0	0	2	2	112	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Staurosirella pinnata	0	0	8	12	37	4	4	46	2	22	5	12	2	14	0	16	0	6	18	53
Stephanodiscus hantzschii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Surirella angusta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40
Surirella brebisonii	10	6	0	0	0	0	30	0	0	0	0	2	2	2	0	0	0	0	0	0
Surirella gracilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella ovalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella patella	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella robusta	0	4	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	18	0
Synedra acus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synedra acus angustissima	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0
Synedra rumpens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synedra ulna	11	6	0	0	6	0	0	3	0	0	0	0	0	0	4	2	0	10	4	0
Tabellaria flocculosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tabularia fasciculata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Thalassiosira baltica	0	0	0	0	0	0	0	0	0	0	0	20	0	0	48	4	0	0	0	0
Thalassiosira fluvialis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Thalassiosira weisflogii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tryblionella calida	0	0	2	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	41	42	43	44	45	46	47	48	49	50
Achnanthes brevipes	0	0	0	0	0	0	0	0	0	0
Achnanthes exigua	32	0	0	0	0	0	0	0	0	0
Achnanthes oblongella	0	0	0	0	0	0	0	0	0	0
Achnanthes pericava	0	0	0	0	0	0	0	0	0	0
Achnanthidium minutissimum	0	6	0	0	0	0	0	0	0	0
Actinocyclis normanii	0	0	0	0	0	0	0	0	0	0
Amphora coff var. borealis	0	0	0	0	0	0	0	0	0	0
Amphora coff var. coff	80	0	0	0	10	0	0	0	0	10
Amphora cognata	0	320	0	0	0	2	0	0	0	0
Amphora libyca	2	0	0	0	0	0	0	0	8	0
Amphora pediculus	0	0	0	0	0	0	0	0	0	0

	41	42	43	44	45	46	47	48	49	50
Amphora veneta	0	57	0	0	24	0	0	0	0	2
Anomoneis spaerophora	2	0	0	0	2	0	0	0	0	0
Aulacoseira ambigua	0	0	0	0	0	0	0	0	0	0
Aulacoseira subartica var. subborealis	0	0	0	0	0	0	0	0	0	0
Aulacosiera granulata	0	0	65	25	0	0	40	60	0	75
Aulacosiera italica	0	6	20	12	0	12	12	10	0	16
Bacillaria paxillifer	0	0	0	0	0	0	0	0	0	0
Bercalia rutilans	0	0	0	0	0	0	0	0	0	0
Climaconeis sp.	0	0	0	0	156	0	8	0	0	0
Cocconeis placentula	49	0	0	0	2	0	0	0	0	0
Craticula accomoda	0	0	0	0	6	0	0	0	0	0
Craticula halophila	0	0	6	10	0	0	6	4	0	0
Cyclotella atomus	0	0	0	0	0	0	0	0	0	0
Cyclotella meneghiniana	0	0	46	49	20	7	0	6	0	4
Cyclotella michigiana	0	0	0	0	0	0	0	0	0	0
Cyclotella rossii	0	0	0	0	0	0	0	0	0	0
Cyclotella stelligera	0	2	0	0	0	0	0	0	0	6
Cymatapleura solea	0	0	0	0	0	0	0	0	0	0
Cymbella affinis	0	0	0	0	0	0	0	0	0	0
Cymbella amphicephala	0	0	0	0	0	0	0	0	0	0
Cymbella cistula	0	0	0	0	0	0	0	0	0	0
Cymbella tumida	0	0	0	0	0	0	0	6	0	0
Diatoma tenuis	0	0	0	0	0	0	0	0	0	0
Diatoma vulgaris	0	0	0	0	0	0	0	0	0	0
Diploneis elliptica	0	0	0	0	0	0	0	2	0	0
Diploneis ovalis	0	0	0	0	0	0	0	2	0	0
Diploneis parma	0	0	0	0	0	0	0	0	0	0
Encyonema gracilis	0	0	0	0	0	0	0	0	0	2
Encyonema minutum	0	0	0	0	2	0	0	6	0	0
Encyonema silesiacum	0	0	0	0	0	0	0	0	0	0
Encyonopsis microcephala	0	0	0	0	0	0	0	0	0	0

	41	42	43	44	45	46	47	48	49	50
Entomoneis alata	16	0	0	0	0	0	0	0	0	0
Epithemia adnata	0	0	0	0	0	0	0	2	0	2
Epithemia sorex	0	0	0	0	0	0	0	0	0	0
Epithemia turgida	0	0	0	0	0	0	0	0	0	0
Eunotia bilunaris	0	0	0	0	0	0	0	0	0	0
Eunotia serpentina	0	0	8	7	0	5	31	12	0	3
Fallacia pygmaea	0	0	0	2	0	12	0	0	0	0
Fallacia tenera	0	0	0	0	0	8	0	0	0	4
Fragilaria cap. var. gracilis	2	0	0	0	0	0	0	0	0	0
Fragilaria cap. var. rumpens	10	0	0	0	0	0	0	7	0	0
Frustulia rhomboides	0	0	0	0	0	0	0	0	0	0
Frustulia vulgaris	0	0	0	0	0	0	0	0	0	0
Gomphonema acuminatum	0	0	0	0	0	0	9	0	0	0
Gomphonema affinis	0	0	0	0	0	0	0	0	0	0
Gomphonema augur	0	0	0	0	0	0	0	0	0	0
Gomphonema clavatum	0	0	0	0	0	0	0	0	0	0
Gomphonema gracile	0	0	0	0	2	0	0	0	0	0
Gomphonema minutum	0	0	0	0	0	0	0	0	0	0
Gomphonema olivaceum	0	0	0	0	0	0	0	0	0	0
Gomphonema parvulum	0	0	0	0	0	0	0	0	0	0
Gomphonema truncatum	0	0	0	0	0	0	0	0	0	0
Gyrosigma attenuatum	0	0	0	0	0	6	0	2	0	0
Gyrosigma parkerii	0	0	0	0	0	5	0	0	0	0
Gyrosigma spencerii	0	0	0	0	0	10	0	0	0	0
Hantzschia amphioxys	0	0	0	0	0	0	11	0	0	0
Hippodonta capitata	0	0	0	0	0	0	0	0	0	0
Luticola mutica	0	0	0	0	0	0	0	0	0	0
Mastagloia smithii	2	0	0	0	8	0	0	0	0	0
Melosira varians	0	0	0	0	0	0	0	0	0	6
Navicella pusilla	75	0	0	4	0	0	0	0	0	0
Navicula angusta	0	0	0	0	0	0	0	0	0	0

	41	42	43	44	45	46	47	48	49	50
Navicula capitidigorada	0	0	0	0	0	0	0	0	0	0
Navicula cari	0	0	0	0	0	0	0	8	0	0
Navicula cincta	0	0	0	0	0	0	0	0	4	0
Navicula cincta var. minuta	0	0	0	0	0	0	0	0	0	0
Navicula clementis	0	0	0	0	0	0	0	0	0	0
Navicula cocconeiformis	0	0	0	0	0	2	0	0	0	0
Navicula cryptocephala	0	2	0	0	0	16	0	6	0	4
Navicula cryptotenella	0	0	4	0	0	0	0	0	0	0
Navicula cuspidata	0	0	4	0	0	0	0	0	0	0
Navicula erifuga	0	4	14	28	0	12	0	0	4	0
Navicula gregaria	0	0	0	0	0	0	0	0	0	0
Navicula incertata	0	2	0	0	0	0	0	0	0	0
Navicula indifferens	0	0	0	0	0	0	0	0	0	8
Navicula lanceolata	0	0	0	0	0	0	0	0	0	0
Navicula leptostriata	0	0	0	0	0	0	0	0	0	0
Navicula menisculus	0	0	0	0	0	0	0	0	0	0
Navicula muraliformis	0	0	0	0	0	0	0	0	0	0
Navicula perminuta	0	0	0	0	0	0	0	17	0	0
Navicula phyllepta	0	0	0	0	0	0	0	0	0	0
Navicula radiosa	0	0	0	0	0	0	0	0	0	0
Navicula recens	0	0	0	0	0	0	0	0	0	0
Navicula rhynchocephala	0	0	0	0	0	0	0	2	0	0
Navicula salinarum	0	0	0	0	0	0	0	0	0	0
Navicula spicula	0	0	0	0	0	235	0	0	210	0
Navicula veneta	2	0	0	0	0	0	0	4	0	4
Navicula viridula	0	0	0	0	0	0	0	0	0	0
Nitzschia acicularis	0	0	0	4	0	6	0	0	0	0
Nitzschia agnita	0	0	10	30	0	2	0	0	0	0
Nitzschia amphibia	0	0	0	0	0	0	0	0	0	0
Nitzschia archibaldii	0	0	0	0	0	0	0	0	0	0
Nitzschia bacillum	0	0	0	0	0	0	0	0	0	0
Nitzschia calida	0	0	0	0	0	0	0	0	0	0

	41	42	43	44	45	46	47	48	49	50
Nitzschia capitellata	0	0	0	0	0	0	0	0	0	0
Nitzschia clausii	0	0	0	0	0	0	0	163	0	2
Nitzschia compressa	0	0	0	0	0	0	0	0	0	0
Nitzschia constricta	0	0	14	2	0	0	10	0	26	0
Nitzschia desortorum	0	0	0	0	0	0	0	0	0	0
Nitzschia dissipata media	0	0	0	0	0	0	0	0	0	0
Nitzschia filiformis	0	0	0	0	0	0	0	8	0	0
Nitzschia fonticola	0	0	0	0	0	0	0	0	0	0
Nitzschia frustulum	0	0	0	0	0	0	0	0	0	14
Nitzschia gracilis	0	0	0	0	0	0	0	0	0	2
Nitzschia heurfleuriana	0	0	0	0	0	0	0	0	0	0
Nitzschia hungarica	0	0	6	88	0	0	0	0	6	0
Nitzschia inconspicua	0	0	0	0	0	0	0	0	0	0
Nitzschia lacuum	0	2	6	0	0	0	0	2	0	2
Nitzschia levendensis	0	0	0	4	0	11	0	0	0	6
Nitzschia linearis	0	0	0	0	0	0	0	0	0	0
Nitzschia littoralis	0	0	0	0	0	0	0	0	0	0
Nitzschia microcephala	0	0	0	0	0	0	0	0	0	0
Nitzschia palacea	0	2	0	4	0	16	0	13	4	2
Nitzschia palea	0	0	4	4	8	4	0	3	0	2
Nitzschia pusilla	0	0	0	0	0	0	0	0	0	0
Nitzschia reversa	0	0	0	0	0	0	0	0	0	0
Nitzschia sigma	0	0	9	5	0	7	0	0	4	2
Nitzschia subacicularis	0	0	8	14	0	2	0	0	0	2
Nitzschia supralittorea	0	0	0	0	0	0	0	0	0	0
Nitzschia tubicola	0	0	0	0	0	6	0	0	0	2
Nitzschia valdecostata	0	0	0	0	0	0	0	0	0	0
Nitzschia valdestriata	0	0	0	0	0	0	0	0	0	0
Opephora olsenii	0	0	0	0	0	0	0	0	0	0
Planothidium delicatulum	6	14	2	2	0	57	0	4	10	16
Planothidium lanceolatum	0	0	0	0	0	0	0	0	0	0
Pleurosigma elongatum	0	0	0	0	0	12	0	0	0	0

	41	42	43	44	45	46	47	48	49	50
Pseudostaurosira brevistriata	16	107	0	3	0	0	0	0	15	0
Rhoicosphenia abbreviata	0	0	0	0	0	0	0	0	0	0
Rhopalodia brebisonii	0	0	0	0	4	0	0	18	0	0
Rhopalodia gibba	0	0	0	0	0	0	0	34	0	0
Rhopalodia musculus	2	0	0	0	0	0	0	0	0	0
Sellaphora pupula	0	0	0	0	0	0	0	0	0	0
Stauroneis obtusa	0	0	0	0	0	0	0	0	0	0
Stauroneis pachycephala	0	0	0	0	0	0	0	0	0	0
Staurosira con. var venter	0	0	0	0	0	0	0	0	0	0
Staurosira construens	6	42	0	0	0	0	0	0	0	0
Staurosira elliptica	0	16	0	0	0	0	0	0	0	0
Staurosirella pinnata	14	62	2	0	0	6	0	0	6	0
Stephanodiscus hantzschii	0	0	0	0	0	0	0	0	0	0
Surirella angusta	0	0	0	0	0	0	0	0	0	0
Surirella brebisonii	0	0	0	0	0	0	0	0	0	0
Surirella gracilis	0	0	0	0	0	0	0	0	0	0
Surirella ovalis	0	0	0	0	0	0	0	0	0	0
Surirella patella	0	0	0	0	0	0	0	0	0	0
Surirella robusta	0	0	0	0	0	0	0	0	0	0
Synedra acus	0	0	0	0	0	0	0	0	0	0
Synedra acus angustissima	0	0	0	0	0	0	0	0	0	0
Synedra rumpens	0	0	0	0	0	0	0	0	0	0
Synedra ulna	0	0	2	2	0	0	10	7	0	6
Tabellaria flocculosa	0	0	0	0	0	0	0	0	0	0
Tabularia fasciculata	0	0	0	0	0	0	0	0	0	0
Thalassiosira baltica	0	0	0	0	0	0	0	0	0	0
Thalassiosira fluvialis	0	0	20	2	0	0	0	0	0	0
Thalassiosira weisflogii	0	0	0	0	0	0	0	0	0	0
Tryblionella calida	0	0	0	0	0	0	0	0	0	0

Diatom counts from stream data set

	51	52	53	54	55	56	57	58	59	60	61	62	63	64	65	66	67	68	69	70
Achnanthes amoena	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthes brevipes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	36	0	0	0	0
Achnanthes exigua	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthes oblongella	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	12	0
Achnanthes punctulata	0	0	0	0	0	0	0	20	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthidium minutissimum	58	0	50	118	40	22	54	15	222	33	0	16	30	0	48	0	4	148	34	4
Amphipleura lindeheimeri	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphora coff var. borealis	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0
Amphora coff var. coff	2	0	0	0	0	0	0	4	0	0	0	0	0	0	6	0	0	0	0	0
Amphora libyca	0	0	0	0	0	0	0	0	18	2	0	0	0	0	0	0	0	0	0	0
Amphora pediculus	0	0	0	0	0	2	0	8	0	0	0	0	8	0	84	0	2	0	0	0
Amphora veneta	0	0	0	0	0	0	0	2	0	4	0	8	0	0	0	0	0	0	0	2
Anomoneis spaerophora	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
Anomoneis vitrea	34	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	36	0	0
Aulacoseira subartica var. subborealis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulacosiera granulata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulacosiera italica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulacosiera varians	0	0	0	0	48	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bacillaria paxillifer	0	0	0	0	0	0	0	0	0	2	0	8	0	52	0	0	10	0	0	10
Bercalia rutilans	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caloneis alpostris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cocconeis placentula	0	0	8	0	70	56	0	0	32	0	0	2	12	4	4	0	2	0	0	0
Cocconeis placentula var. euglypta	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0
Craticula accomoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Craticula halophila	0	4	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyclotella atomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyclotella meneghiniana	0	0	0	0	0	0	0	0	0	0	0	12	0	0	0	0	0	0	0	2

	51	52	53	54	55	56	57	58	59	60	61	62	63	64	65	66	67	68	69	70
Cymatapleura solea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella affinis	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella amphicephala	0	0	0	0	0	0	0	0	0	0	0	4	0	0	2	0	0	0	0	0
Cymbella aspera	0	0	2	0	0	0	0	4	0	0	0	0	0	0	0	0	0	8	0	0
Cymbella cistula	0	0	0	0	0	6	0	0	2	14	0	0	0	8	14	0	0	0	0	0
Cymbella helvetica	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0
Cymbella silesiaca	0	0	0	8	0	2	0	0	0	0	0	0	0	2	0	0	0	0	0	0
Cymbella tumida	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Denticula elegans	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Denticula kuetzingii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
Diatoma moniliformis	2	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diatoma tenuis	52	0	0	0	0	0	322	0	0	0	0	2	0	0	0	0	0	0	14	24
Diatoma vulgaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diploneis elliptica	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0
Diploneis parma	2	2	0	0	0	0	0	2	0	0	0	0	0	0	0	4	0	0	0	0
Encyonema minutum	0	0	0	0	0	0	0	4	0	4	0	0	0	0	0	2	0	0	0	0
Encyonopsis microcephala	2	0	12	4	0	0	4	0	0	0	0	0	0	6	0	0	0	42	0	2
Entomoneis alata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0
Epithemia adnata	0	2	0	0	0	0	0	0	0	2	0	0	0	0	0	30	0	0	0	0
Epithemia sorex	0	0	0	0	0	0	0	0	0	0	0	2	0	0	44	0	0	0	0	0
Epithemia turgida	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eunotia implicata	0	0	0	34	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eunotia naegelli	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eunotia paludosa var. paludosa	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Eunotia serpentina	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0
Eunotia tecta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fallacia tenera	0	0	0	0	0	0	0	0	0	0	0	14	0	0	0	0	0	0	0	0
Fallacia tenuis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fragilaria cap. var gracilis	22	0	0	8	12	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fragilaria cap. var rumpens	4	0	2	10	8	2	2	0	0	0	0	6	0	0	0	0	0	6	0	0

	51	52	53	54	55	56	57	58	59	60	61	62	63	64	65	66	67	68	69	70
Fragilaria con. var venter	0	0	0	0	0	0	0	68	0	0	209	0	0	0	0	0	0	0	0	0
Fragilaria leptostauron	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fragilaria tenera	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fragilariforma virescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
Frustulia cruezburgensis	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Frustulia rhomboides	14	0	0	8	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0
Frustulia vulgaris	0	0	0	0	0	2	0	0	0	2	0	24	0	0	0	0	0	2	0	0
Gomphonema acuminatum	0	0	0	0	0	0	0	0	12	0	0	0	0	0	0	0	0	6	0	0
Gomphonema affinis	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Gomphonema augur	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema clavatum	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema gracile	0	0	2	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0
Gomphonema minutum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema olivaceum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema parvulum	0	0	0	4	0	4	0	0	0	8	0	0	0	0	2	0	10	0	4	0
Gomphonema truncatum	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gyrosigma attenuatum	0	0	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0	10	0	0
Gyrosigma parkerii	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2	0
Gyrosigma spencerii	0	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0
Gyrosigma wansbeckii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hantzschia amphioxys	0	0	0	0	0	0	0	0	2	0	4	0	0	2	0	0	0	0	0	0
Luticola mutica	2	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	8	0
Luticola pygmaea	0	0	0	0	0	0	0	0	0	0	0	7	0	4	0	0	0	0	0	0
Mastagloia smithii	0	14	0	2	0	0	0	0	2	5	0	0	0	10	35	4	0	0	2	0
Melosira varians	0	0	0	0	0	67	0	0	0	0	0	20	0	0	0	0	12	0	0	2
Navicella pusilla	0	0	0	0	0	0	0	50	0	8	0	0	0	0	0	2	0	0	0	0
Navicula angusta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula capitata	0	0	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	2
Navicula capitatoradiata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cari	0	7	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	4
Navicula cincta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	51	52	53	54	55	56	57	58	59	60	61	62	63	64	65	66	67	68	69	70
Navicula cincta var. minuta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula clementis	0	0	0	0	0	0	0	0	0	0	0	48	0	0	0	0	0	0	0	0
Navicula clementoides	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cocconeiformis	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cryptocephala	0	10	20	0	0	0	0	0	2	4	8	0	0	4	0	2	0	2	18	0
Navicula cryptotenella	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	2	0
Navicula cuspidata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Navicula elegans	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula elegenensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula expecta	0	0	0	0	0	0	0	0	2	0	0	0	0	2	0	0	0	0	0	0
Navicula gregaria	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	22
Navicula incertata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula indifferens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula lanceolata	0	0	2	0	0	2	0	0	4	0	0	2	0	4	0	0	0	0	0	2
Navicula leptostriata	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula menisculus	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula molestiformis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula perminuta	0	0	0	0	0	0	0	0	0	0	0	14	0	0	0	0	0	0	0	0
Navicula phyllepta	0	0	0	0	8	0	0	0	0	4	0	0	0	2	0	0	0	0	0	6
Navicula recens	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0
Navicula rhycnocephala	6	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Navicula salinarum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Navicula spicula	0	2	0	0	0	0	0	4	0	0	0	0	0	4	0	0	0	0	2	0
Navicula subminiscula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula subplacentula	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula tenelloides	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Navicula trivialis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula veneta	0	112	18	0	0	2	0	0	12	24	35	4	2	0	2	0	4	18	0	20
Navicula viridula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Neidum amplicatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia acicularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	48
Nitzschia aerophila	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	84	0

	51	52	53	54	55	56	57	58	59	60	61	62	63	64	65	66	67	68	69	70
Nitzschia agnita	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia amphibia	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	0
Nitzschia archibaldii	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0
Nitzschia bacillum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia capitellata	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	4	0
Nitzschia clausii	0	4	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia constricta	0	0	0	0	0	0	0	0	0	2	0	2	0	0	0	0	0	0	0	2
Nitzschia desortorum	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	4
Nitzschia dissipata media	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
Nitzschia diversa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia filiformis	0	0	0	0	0	0	0	0	6	2	0	0	0	12	0	0	0	0	0	8
Nitzschia fonticola	0	0	0	4	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	7
Nitzschia fossilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia frustulum	0	0	0	0	0	2	0	0	0	4	2	2	0	0	4	0	0	0	0	2
Nitzschia graciliformis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia gracilis	2	0	0	2	0	0	0	30	0	0	0	2	0	2	0	0	0	0	0	0
Nitzschia heurfleuriana	0	0	0	0	0	48	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia hungarica	0	0	0	0	0	0	0	0	0	4	22	0	0	2	0	0	0	0	2	2
Nitzschia inconspicua	0	0	0	0	0	0	0	0	0	0	0	12	2	0	38	0	0	0	0	0
Nitzschia lacuum	4	0	0	0	0	0	0	0	2	6	4	2	0	0	0	0	0	2	6	0
Nitzschia linearis	0	2	0	0	0	0	0	0	2	10	0	0	0	102	0	6	4	4	0	4
Nitzschia littoralis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	2
Nitzschia microcephala	0	22	0	0	0	4	0	0	0	78	0	12	0	0	4	0	0	0	0	0
Nitzschia palacea	0	0	0	0	0	0	0	0	0	0	8	0	0	4	0	0	0	0	8	6
Nitzschia palae	0	8	0	0	0	2	0	0	0	0	2	0	0	0	0	0	0	6	4	6
Nitzschia perminuta	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
Nitzschia prolongata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia pura	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia pusilla	0	168	0	0	0	0	0	0	0	2	0	0	0	8	0	0	0	0	0	0
Nitzschia reversa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
Nitzschia sigma	2	0	0	24	0	0	0	0	0	0	6	24	0	0	0	0	2	2	4	0
Nitzschia sigmodea	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia subacicularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	51	52	53	54	55	56	57	58	59	60	61	62	63	64	65	66	67	68	69	70
Nitzschia sublinearis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia supralittorea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0
Nitzschia thermaloides	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
Nitzschia tubicola	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia valdecostata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia valdestriata	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Opephora olsenii	0	0	0	0	14	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pinnularia brebisonii	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
Pinnularia elegans	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pinnularia gibba	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planothidium delicatulum	0	0	0	0	0	17	0	4	0	0	0	216	214	0	0	0	0	0	0	42
Planothidium lanceolatum	0	0	26	10	0	0	0	0	30	4	0	2	2	0	4	0	0	0	0	0
Pleurosigma elongatum	0	14	0	0	0	0	0	0	0	0	0	0	0	4	0	29	0	0	0	0
Pseudostaurosira brevistriata	0	0	0	0	0	0	0	108	0	0	6	10	0	0	0	0	0	0	0	0
Rhoicosphenia abbreviata	0	0	56	0	12	14	0	4	22	0	0	12	0	0	6	0	220	0	0	4
Rhopalodia gibba	0	4	78	0	0	0	0	0	8	0	0	2	0	34	0	104	0	4	0	3
Rhopalodia musculus	0	10	4	0	0	0	0	0	0	38	0	30	0	6	4	14	0	0	4	6
Sellaphora pupula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stauroneis kreigerii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stauroneis obtusa	0	2	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stauroneis pachycephala	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
Stauroneis smithii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurosira construens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurosirella elliptica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurosirella pinnata	0	0	0	0	0	0	0	27	0	0	18	0	0	0	0	0	0	0	0	0
Surirella angusta	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella brebisonii	0	0	0	0	0	0	6	0	0	0	16	16	0	0	0	0	4	0	4	42
Surirella ovalis	0	0	0	0	0	0	0	0	6	0	2	6	0	0	0	0	0	0	2	0
Surirella patella	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella perisonis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella robusta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0
Synedra acus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	51	52	53	54	55	56	57	58	59	60	61	62	63	64	65	66	67	68	69	70
Synedra acus angustissima	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Synedra rumpens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synedra ulna	0	4	6	16	95	47	4	2	12	0	0	4	4	8	0	4	2	16	10	0
Tabellaria flocculosa	200	0	0	102	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tabularia fasciculata	0	0	16	0	0	2	0	0	0	0	0	16	0	0	0	41	0	0	4	2
Thalassiosira weisflogii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tryblionella calida	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tryblionella debilis	0	0	0	0	0	0	0	0	0	0	6	2	0	0	0	0	0	0	0	0
Tryblionella levendensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	12	0
Tryblionella levendensis var. *victorae*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	37	0

	71	72	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89	90
Achnanthes amoena	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0
Achnanthes brevipes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthes exigua	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthes oblongella	0	0	0	0	0	0	0	0	0	0	0	0	0	0	21	0	0	0	0	0
Achnanthes punctulata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Achnanthidium minutissimum	0	0	0	0	2	0	0	0	4	60	6	0	6	0	14	5	2	0	0	0
Amphipleura lindeheimeri	0	0	0	0	0	2	0	0	8	0	0	0	0	0	0	0	0	0	0	0
Amphora coff var. borealis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphora coff var. coff	2	0	0	0	0	0	0	0	0	0	0	0	0	0	12	0	0	0	0	0
Amphora libyca	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphora pediculus	4	0	0	0	0	8	0	0	2	8	0	0	4	0	0	2	0	0	0	0
Amphora veneta	0	2	0	0	0	0	2	0	4	0	0	0	0	0	0	0	2	0	0	2
Anomoneis spaerophora	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anomoneis vitrea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulacoseira subartica var. subborealis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	56	12
Aulacosiera granulata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	167	0

	71	72	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89	90
Aulacosiera italica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	28	0
Aulacosiera varians	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bacillaria paxillifer	24	45	0	118	0	0	0	10	6	0	0	2	0	4	0	44	24	34	0	4
Bercalia rutilans	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caloneis alpostris	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
Cocconeis placentula	6	2	0	6	0	58	0	6	10	30	0	12	8	0	0	0	0	0	0	31
Cocconeis placentula var. euglypta	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Craticula accomoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Craticula halophila	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0
Cyclotella atomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
Cyclotella meneghiniana	0	0	4	0	0	0	2	0	2	0	0	0	0	0	4	0	2	2	8	8
Cymatapleura solea	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0
Cymbella affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
Cymbella amphicephala	0	0	0	0	0	0	0	0	0	0	0	0	0	32	0	0	0	0	0	2
Cymbella aspera	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella cistula	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella helvetica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella silesiaca	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella tumida	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Denticula elegans	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0
Denticula kuetzingii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diatoma moniliformis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diatoma tenuis	0	0	0	0	8	0	0	0	0	0	0	0	0	0	34	0	0	0	2	17
Diatoma vulgaris	0	0	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0	0
Diploneis elliptica	0	0	2	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Diploneis parma	2	0	0	0	4	0	0	0	2	0	0	2	0	0	10	8	0	0	0	2
Encyonema minutum	0	0	0	0	0	0	0	2	6	0	0	0	0	0	6	0	0	0	0	0
Encyonopsis microcephala	0	2	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Entomoneis alata	14	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Epithemia adnata	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Epithemia sorex	6	0	104	0	0	2	36	2	0	0	0	0	0	0	0	0	0	215	0	0

	71	72	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89	90
Epithemia turgida	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eunotia implicata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eunotia naegelli	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eunotia paludosa var. paludosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eunotia serpentina	0	0	0	0	16	0	0	0	2	0	0	0	0	0	4	0	0	0	0	0
Eunotia tecta	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Fallacia tenera	0	0	0	0	0	0	18	0	2	2	0	0	0	35	0	18	0	0	0	0
Fallacia tenuis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fragilaria cap. var gracilis	0	0	0	0	0	0	0	0	0	14	0	0	0	0	0	0	2	0	0	0
Fragilaria cap. var rumpens	0	0	0	0	0	0	0	0	6	0	0	0	0	0	4	0	16	2	0	42
Fragilaria con. var venter	31	22	0	0	0	0	0	0	20	14	0	0	0	0	2	0	12	0	0	0
Fragilaria leptostauron	12	0	0	0	0	0	0	0	0	0	0	16	0	0	0	0	0	0	0	0
Fragilaria tenera	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0
Fragilariforma virescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frustulia cruezburgensis	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Frustulia rhomboides	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frustulia vulgaris	0	2	2	0	0	0	0	0	0	0	0	0	0	0	6	2	0	0	0	0
Gomphonema acuminatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema affinis	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	4	0	0	0
Gomphonema augur	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema clavatum	0	0	0	0	0	0	12	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema gracile	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema minutum	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Gomphonema olivaceum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gomphonema parvulum	0	0	0	0	0	4	16	0	2	8	0	0	0	0	0	0	0	0	0	7
Gomphonema truncatum	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gyrosigma attenuatum	2	0	0	0	0	0	0	0	4	0	0	2	0	0	0	0	0	0	0	0
Gyrosigma parkerii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gyrosigma spencerii	0	0	0	0	0	4	0	4	0	0	0	2	0	0	4	0	0	0	0	4
Gyrosigma wansbeckii	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	71	72	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89	90
Hantzschia amphioxys	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Luticola mutica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Luticola pygmaea	0	0	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0	0
Mastagloia smithii	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Melosira varians	0	0	0	72	2	0	0	112	6	0	6	0	0	0	0	24	0	105	4	38
Navicella pusilla	0	0	0	0	0	0	0	0	2	8	0	2	0	0	26	0	0	2	0	0
Navicula angusta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula capitata	0	0	0	4	0	0	0	12	6	10	2	0	0	8	0	0	16	0	0	0
Navicula capitatoradiata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cari	0	0	0	0	0	0	21	0	0	0	0	2	0	0	0	0	0	0	0	0
Navicula cincta	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	2	0
Navicula cincta var. minuta	0	0	0	0	0	0	32	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula clementis	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	20	0	0	0
Navicula clementoides	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cocconeiformis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cryptocephala	0	0	2	8	12	0	0	2	2	6	0	4	0	0	4	0	2	0	8	0
Navicula cryptotenella	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0
Navicula cuspidata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
Navicula elegans	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula elegenensis	0	0	0	0	0	0	0	4	8	0	0	0	0	0	0	0	0	0	0	0
Navicula expecta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula gregaria	0	0	0	4	0	8	0	8	0	0	0	2	0	0	0	4	8	0	0	26
Navicula incertata	0	0	0	0	0	0	0	0	0	0	28	0	0	0	0	0	0	0	0	0
Navicula indifferens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula lanceolata	0	0	0	18	0	92	0	32	0	0	0	0	0	0	0	0	0	0	0	8
Navicula leptostriata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula menisculus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula molestiformis	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Navicula perminuta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula phyllepta	0	12	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula recens	0	0	0	0	0	0	0	2	0	0	0	0	0	4	4	0	0	0	0	0
Navicula rhycnocephala	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0

	71	72	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89	90
Navicula salinarum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula spicula	8	0	0	0	24	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0
Navicula subminiscula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula subplacentula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula tenelloides	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0
Navicula trivialis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula veneta	2	2	10	0	2	4	6	0	4	8	17	0	0	24	2	4	0	0	0	0
Navicula viridula	0	0	0	0	4	4	0	0	0	0	0	0	0	0	6	0	0	0	2	0
Neidum amplicatum	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
Nitzschia acicularis	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia aerophila	0	0	0	0	124	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0
Nitzschia agnita	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia amphibia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia archibaldii	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia bacillum	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Nitzschia capitellata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia clausii	0	2	0	0	8	0	0	0	0	0	0	0	0	0	30	4	0	0	0	0
Nitzschia constricta	0	0	0	2	0	0	4	14	2	22	0	0	0	4	0	0	0	0	0	8
Nitzschia desortorum	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0
Nitzschia dissipata media	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia diversa	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Nitzschia filiformis	0	0	0	4	0	0	0	0	0	2	0	0	0	0	0	0	0	8	0	0
Nitzschia fonticola	0	0	0	0	0	0	0	2	0	14	0	0	0	0	0	4	0	0	0	0
Nitzschia fossilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia frustulum	2	4	0	0	0	0	22	0	0	0	0	0	0	12	0	0	0	0	0	0
Nitzschia graciliformis	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0
Nitzschia gracilis	2	0	0	0	0	0	0	2	0	2	2	0	0	0	0	0	4	0	0	4
Nitzschia heurfleuriana	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
Nitzschia hungarica	0	2	0	0	0	0	0	5	0	0	0	0	0	2	0	2	6	0	0	0
Nitzschia inconspicua	0	0	8	0	0	0	36	19	0	0	197	0	0	0	0	0	0	0	0	0
Nitzschia lacuum	0	0	0	4	2	0	0	4	2	2	0	0	0	4	4	4	0	0	0	0
Nitzschia linearis	10	0	0	0	4	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia littoralis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0

	71	72	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89	90
Nitzschia microcephala	0	0	0	0	0	0	0	0	0	0	6	6	0	0	0	0	0	0	0	0
Nitzschia palacea	4	0	0	0	0	0	0	2	0	2	0	0	0	0	6	0	0	0	0	0
Nitzschia palae	2	0	0	0	0	0	0	2	0	2	0	0	0	0	0	2	0	0	0	0
Nitzschia perminuta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0
Nitzschia prolongata	0	0	0	0	0	0	0	0	0	2	0	0	0	2	0	0	0	0	0	0
Nitzschia pura	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia pusilla	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia reversa	0	0	0	0	0	0	0	0	0	0	0	2	0	0	4	0	0	0	0	0
Nitzschia sigma	2	6	0	0	12	0	4	0	0	2	0	0	0	4	2	12	8	0	0	4
Nitzschia sigmodea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia subacicularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Nitzschia sublinearis	0	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia supralittorea	0	0	0	0	2	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia thermaloides	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia tubicola	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0
Nitzschia valdecostata	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Nitzschia valdestriata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Opephora olsenii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pinnularia brebisonii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pinnularia elegans	0	2	0	0	0	0	0	0	0	0	0	0	0	0	12	0	8	0	0	0
Pinnularia gibba	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planothidium delicatulum	0	0	0	0	6	6	14	12	41	10	4	0	0	158	0	36	0	0	0	2
Planothidium lanceolatum	0	0	0	8	4	0	0	4	0	0	8	0	0	2	0	0	0	2	0	2
Pleurosigma elongatum	8	0	0	0	0	0	0	0	0	0	0	0	0	6	4	0	0	2	0	0
Pseudostaurosira brevistriata	45	142	0	0	0	0	0	0	84	0	0	86	0	0	0	0	0	0	0	0
Rhoicosphenia abbreviata	2	2	2	45	2	132	0	42	8	8	8	0	278	4	6	14	0	10	0	19
Rhopalodia gibba	2	0	132	4	0	0	0	0	30	0	0	0	0	0	0	4	0	53	0	0
Rhopalodia musculus	6	18	44	4	78	0	91	0	0	0	2	2	0	0	6	12	2	0	2	0
Sellaphora pupula	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0
Stauroneis kreigerii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0
Stauroneis obtusa	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0
Stauroneis pachycephala	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	71	72	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89	90
Stauroneis smithii	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Staurosira construens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0
Staurosirella elliptica	0	10	0	17	0	0	0	0	0	2	0	0	0	0	0	8	0	0	0	0
Staurosirella pinnata	160	0	2	0	0	0	0	0	0	6	0	186	0	0	0	0	2	0	0	0
Surirella angusta	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Surirella brebisonii	2	4	0	0	0	0	0	2	4	2	6	0	0	0	10	0	95	0	0	22
Surirella ovalis	0	0	0	4	0	2	2	0	2	14	0	0	0	0	10	0	0	0	0	0
Surirella patella	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella perisonis	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surirella robusta	2	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synedra acus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	16	0	0	0	0
Synedra acus angustissima	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synedra rumpens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synedra ulna	0	0	0	0	0	4	0	2	0	0	4	0	0	0	0	8	4	0	0	51
Tabellaria flocculosa	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Tabularia fasciculata	2	14	0	6	0	0	0	4	4	0	8	0	0	8	0	40	56	20	0	0
Thalassiosira weisflogii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	12	0	0	0	0	0
Tryblionella calida	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	2	0	0	0
Tryblionella debilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tryblionella levendensis	0	0	0	8	12	0	0	2	8	0	0	0	0	2	6	2	0	0	0	0
Tryblionella levendensis var. victorae	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	91	92	93	94	95	96	97	98	99	100	101
Achnanthes amoena	0	0	0	0	0	0	0	0	0	0	0
Achnanthes brevipes	0	0	0	0	2	0	0	0	0	0	0
Achnanthes exigua	0	0	0	0	10	0	0	0	0	0	0
Achnanthes oblongella	0	0	0	0	0	0	0	0	0	0	0
Achnanthes punctulata	0	0	0	0	0	0	0	0	0	0	0
Achnanthidium minutissimum	2	0	14	0	0	0	0	0	0	0	0

	91	92	93	94	95	96	97	98	99	100	101
Amphipleura lindeheimeri	0	0	0	0	0	0	0	0	0	0	0
Amphora coff var. borealis	0	0	0	0	0	0	0	0	0	0	0
Amphora coff var. coff	0	0	2	0	0	0	0	0	6	0	4
Amphora libyca	0	0	0	0	0	0	0	0	0	0	0
Amphora pediculus	0	0	22	0	0	2	0	0	43	0	0
Amphora veneta	0	0	2	0	8	28	4	95	0	0	0
Anomoneis spaerophora	0	0	0	0	0	0	0	0	0	0	0
Anomoneis vitrea	0	0	0	0	0	0	0	0	0	0	0
Aulacoseira subartica var. subborealis	24	0	0	0	0	0	0	0	0	0	0
Aulacosiera granulata	40	0	0	0	0	0	0	0	0	0	0
Aulacosiera italica	88	0	0	0	0	0	0	0	0	0	0
Aulacosiera varians	0	0	0	0	0	0	0	0	0	0	0
Bacillaria paxillifer	6	6	0	0	0	0	0	0	0	4	2
Bercalia rutilans	0	2	0	0	0	0	0	0	0	0	0
Caloneis alpostris	0	0	0	0	0	0	0	0	0	0	0
Cocconeis placentula	2	4	14	0	0	0	4	0	0	6	0
Cocconeis placentula var. euglypta	0	0	0	0	0	0	0	0	0	0	0
Craticula accomoda	0	2	0	0	0	4	0	0	4	0	0
Craticula halophila	0	0	0	0	2	0	0	0	0	0	0
Cyclotella atomus	0	0	0	0	0	0	0	0	0	0	0
Cyclotella meneghiniana	4	4	0	4	0	2	0	0	6	6	32
Cymatapleura solea	0	0	0	0	0	0	0	0	0	0	0
Cymbella affinis	0	0	0	0	0	0	0	0	0	0	0
Cymbella amphicephala	0	0	0	0	0	2	0	0	2	0	0
Cymbella aspera	0	0	0	0	0	0	0	0	0	0	0
Cymbella cistula	2	0	0	4	0	0	0	0	0	0	0
Cymbella helvetica	0	0	0	0	0	0	0	0	0	0	0
Cymbella silesiaca	0	0	0	0	0	4	0	0	0	0	0
Cymbella tumida	0	0	0	10	0	2	0	0	0	0	0
Denticula elegans	0	0	0	0	0	0	0	0	0	0	0
Denticula kuetzingii	0	0	0	0	0	0	0	0	0	0	0

	91	92	93	94	95	96	97	98	99	100	101
Diatoma moniliformis	0	0	0	0	0	0	0	0	0	0	0
Diatoma tenuis	0	0	0	0	0	0	0	0	0	0	0
Diatoma vulgaris	0	0	0	0	0	0	0	0	0	0	0
Diploneis elliptica	0	0	0	0	0	0	0	0	0	0	0
Diploneis parma	0	0	0	0	0	0	0	0	0	0	0
Encyonema minutum	0	0	0	2	0	2	0	0	0	0	0
Encyonopsis microcephala	0	0	0	0	0	0	0	0	0	0	0
Entomoneis alata	0	0	0	0	0	0	0	0	0	0	0
Epithemia adnata	0	0	0	0	0	0	0	0	0	0	0
Epithemia sorex	0	0	0	0	0	0	2	0	0	4	0
Epithemia turgida	0	0	0	0	0	0	0	0	0	0	0
Eunotia implicata	0	0	0	0	0	0	0	0	0	0	0
Eunotia naegelli	0	0	0	0	0	0	0	0	0	0	0
Eunotia paludosa var. paludosa	0	0	0	0	0	0	0	0	0	0	0
Eunotia serpentina	0	0	0	0	0	0	0	0	0	0	0
Eunotia tecta	0	0	0	0	0	0	0	0	0	0	0
Fallacia tenera	0	0	0	0	0	0	0	0	0	0	0
Fallacia tenuis	0	0	0	0	0	0	0	0	0	0	0
Fragilaria cap. var gracilis	2	0	0	0	0	0	0	0	0	0	0
Fragilaria cap. var rumpens	2	0	18	0	14	40	0	0	8	0	0
Fragilaria con. var venter	36	0	0	0	0	0	221	132	30	0	0
Fragilaria leptostauron	0	0	0	0	0	0	0	0	0	0	0
Fragilaria tenera	0	0	0	0	0	0	0	0	0	0	0
Fragilariforma virescens	0	0	0	0	0	0	0	0	0	0	0
Frustulia cruezburgensis	0	0	0	0	0	0	0	0	0	0	0
Frustulia rhomboides	0	0	0	0	0	0	0	0	0	0	0
Frustulia vulgaris	0	0	0	0	0	0	0	0	0	0	0
Gomphonema acuminatum	0	0	0	0	0	0	0	0	0	0	0
Gomphonema affinis	0	0	0	6	0	0	0	0	6	2	8

	91	92	93	94	95	96	97	98	99	100	101
Gomphonema augur	0	0	0	2	0	0	0	0	0	0	0
Gomphonema clavatum	0	0	0	0	0	0	0	0	0	0	0
Gomphonema gracile	0	0	8	2	0	0	0	0	0	0	0
Gomphonema minutum	0	0	0	0	0	0	0	0	2	2	0
Gomphonema olivaceum	2	0	2	0	0	0	0	0	0	0	0
Gomphonema parvulum	2	4	26	4	5	34	0	0	8	104	55
Gomphonema truncatum	0	0	0	0	0	0	0	0	2	0	0
Gyrosigma attenuatum	0	0	0	0	0	0	0	0	0	0	0
Gyrosigma parkerii	0	0	0	0	0	0	0	0	0	0	0
Gyrosigma spencerii	0	0	0	0	0	0	0	0	0	0	0
Gyrosigma wansbeckii	0	0	0	0	0	0	0	0	0	0	0
Hantzschia amphioxys	0	0	0	0	0	2	0	0	0	0	2
Luticola mutica	0	0	0	0	0	4	0	0	0	0	0
Luticola pygmaea	0	2	4	0	0	0	0	0	2	0	0
Mastagloia smithii	0	0	0	0	0	0	0	0	0	0	2
Melosira varians	8	4	2	124	0	6	0	0	58	6	0
Navicella pusilla	0	0	0	0	0	0	0	0	0	0	0
Navicula angusta	0	0	0	0	0	0	0	0	0	0	0
Navicula capitata	0	0	8	0	6	0	0	0	0	62	0
Navicula capitatoradiata	0	0	0	0	0	0	0	0	2	0	0
Navicula cari	0	0	0	2	0	0	0	0	0	0	0
Navicula cincta	0	0	0	0	0	0	0	6	0	6	0
Navicula cincta var. minuta	0	0	0	0	0	0	0	0	0	0	0
Navicula clementis	0	0	0	0	0	0	0	0	0	6	0
Navicula clementoides	0	0	0	0	0	0	4	0	0	0	0
Navicula cocconeiformis	0	0	0	0	0	0	0	0	0	0	0
Navicula cryptocephala	2	0	0	24	2	20	0	0	0	2	0
Navicula cryptotenella	0	0	0	0	0	2	0	0	0	0	0
Navicula cuspidata	0	0	0	0	2	6	0	0	0	0	0
Navicula elegans	0	0	0	0	0	0	0	0	0	0	2
Navicula elegenensis	0	0	0	0	0	0	0	0	0	0	0
Navicula expecta	0	0	0	0	0	0	0	0	0	0	0

	91	92	93	94	95	96	97	98	99	100	101
Navicula gregaria	0	0	0	0	0	0	8	0	62	0	0
Navicula incertata	2	0	0	0	0	0	0	0	0	0	0
Navicula indifferens	0	0	0	0	0	0	0	0	0	0	5
Navicula lanceolata	0	0	0	8	0	16	0	0	8	4	0
Navicula leptostriata	0	0	0	27	0	0	0	0	0	0	0
Navicula menisculus	0	0	0	0	0	0	0	0	0	0	0
Navicula molestiformis	0	0	0	0	0	27	0	0	0	0	0
Navicula perminuta	0	0	0	0	0	0	0	0	0	0	0
Navicula phyllepta	0	8	4	0	0	0	0	0	8	0	0
Navicula recens	2	0	0	0	0	0	0	0	0	0	0
Navicula rhycnocephala	0	0	0	0	0	0	0	0	0	0	0
Navicula salinarum	0	0	0	0	0	0	0	0	14	0	0
Navicula spicula	0	0	0	0	0	0	0	0	0	0	0
Navicula subminiscula	0	8	0	0	48	0	0	0	0	0	0
Navicula subplacentula	0	0	0	0	0	0	0	0	0	0	0
Navicula tenelloides	0	0	0	0	0	0	0	0	0	0	0
Navicula trivialis	2	0	2	0	0	0	0	0	0	0	0
Navicula veneta	0	10	4	0	6	0	18	38	33	14	8
Navicula viridula	0	0	0	0	0	0	0	0	0	0	0
Neidum amplicatum	0	0	0	0	0	0	0	0	0	0	0
Nitzschia acicularis	0	0	0	0	0	0	0	0	0	0	0
Nitzschia aerophila	0	0	0	0	0	0	0	0	0	0	0
Nitzschia agnita	0	0	0	0	0	0	0	0	0	0	0
Nitzschia amphibia	0	0	0	0	0	0	0	0	0	0	2
Nitzschia archibaldii	0	0	0	0	0	0	0	0	0	0	0
Nitzschia bacillum	0	0	0	0	0	0	0	0	0	0	0
Nitzschia capitellata	0	0	0	0	2	28	0	0	2	4	0
Nitzschia clausii	0	0	0	4	0	2	0	0	0	0	0
Nitzschia constricta	0	101	45	0	0	0	2	0	18	8	0
Nitzschia desortorum	0	0	0	2	0	0	0	0	0	0	14
Nitzschia dissipata media	0	0	0	4	0	8	0	0	0	0	0
Nitzschia diversa	0	0	0	0	0	0	0	0	0	0	0
Nitzschia filiformis	190	0	0	2	0	0	0	0	0	0	0

	91	92	93	94	95	96	97	98	99	100	101
Nitzschia fonticola	0	0	0	0	8	2	0	0	0	4	0
Nitzschia fossilis	0	0	0	0	0	4	0	0	0	0	0
Nitzschia frustulum	0	0	0	0	0	0	0	0	2	0	55
Nitzschia graciliformis	0	0	0	0	0	0	0	0	0	0	0
Nitzschia gracilis	2	0	0	6	4	16	0	0	0	0	0
Nitzschia heurfleuriana	0	[illegible]	0	0	0	0	0	0	0	0	0
Nitzschia hungarica	0	[illegible]	30	2	78	0	0	0	2	0	30
Nitzschia inconspicua	12	0	6	0	0	0	0	4	0	42	4
Nitzschia lacuum	0	0	0	26	4	0	0	2	32	4	15
Nitzschia linearis	0	0	0	0	0	6	0	0	2	0	0
Nitzschia littoralis	0	0	0	0	0	0	0	0	0	0	0
Nitzschia microcephala	0	0	0	0	0	0	0	0	2	0	0
Nitzschia palacea	0	4	0	10	14	37	0	2	0	0	2
Nitzschia palae	0	8	0	6	2	26	0	0	14	2	28
Nitzschia perminuta	0	2	26	0	0	0	0	0	10	0	170
Nitzschia prolongata	0	8	0	0	0	0	0	0	0	0	0
Nitzschia pura	0	0	0	0	0	40	0	0	0	0	0
Nitzschia pusilla	0	0	0	2	0	0	0	0	0	0	0
Nitzschia reversa	0	0	0	0	0	0	0	0	0	0	0
Nitzschia sigma	0	16	2	0	0	2	2	0	4	0	0
Nitzschia sigmodea	0	0	0	0	0	0	0	0	0	0	0
Nitzschia subacicularis	0	0	0	0	0	0	0	0	0	0	0
Nitzschia sublinearis	0	0	0	2	0	0	0	0	0	0	2
Nitzschia supralittorea	0	0	0	0	0	0	0	0	0	0	6
Nitzschia thermaloides	0	0	0	0	0	0	0	0	0	0	0
Nitzschia tubicola	0	0	[illegible]	0	0	0	0	0	0	0	0
Nitzschia valdecostata	0	0	[illegible]	0	0	0	0	0	0	0	0
Nitzschia valdestriata	0	0	0	0	0	0	0	0	0	0	0
Opephora olsenii	0	0	0	0	0	0	0	0	0	0	0
Pinnularia brebisonii	0	0	0	0	0	0	0	0	0	0	0
Pinnularia elegans	0	0	0	0	0	0	0	0	0	0	0
Pinnularia gibba	0	0	0	0	0	0	0	0	2	0	0
Planothidium delicatulum	0	4	2	0	8	0	0	2	0	4	0

	91	92	93	94	95	96	97	98	99	100	101
Planothidium lanceolatum	0	0	0	0	0	0	0	0	2	2	35
Pleurosigma elongatum	0	0	0	0	0	0	0	0	0	0	0
Pseudostaurosira brevistriata	142	0	0	0	0	0	0	0	0	0	0
Rhoicosphenia abbreviata	2	6	259	0	2	0	4	0	62	16	44
Rhopalodia gibba	0	0	0	0	0	0	0	0	0	0	0
Rhopalodia musculus	0	4	8	0	0	0	0	0	0	0	0
Sellaphora pupula	0	0	0	0	0	0	0	0	0	0	0
Stauroneis kreigerii	0	0	0	0	0	0	0	0	0	0	0
Stauroneis obtusa	2	0	2	0	0	0	0	0	2	2	0
Stauroneis pachycephala	0	0	0	0	0	0	0	0	0	0	0
Stauroneis smithii	0	0	0	0	0	0	0	0	0	0	0
Staurosira construens	0	0	0	0	0	0	0	0	0	0	0
Staurosirella elliptica	0	0	0	0	0	0	0	2	0	0	0
Staurosirella pinnata	15	0	0	2	0	0	4	40	2	0	0
Surirella angusta	0	16	0	12	10	2	0	0	0	0	0
Surirella brebisonii	0	10	2	0	17	0	36	0	48	0	10
Surirella ovalis	0	4	6	0	94	0	2	0	0	0	0
Surirella patella	0	0	0	0	0	2	0	0	0	0	0
Surirella perisonis	0	0	0	0	0	0	0	0	0	0	0
Surirella robusta	0	0	0	0	0	0	0	0	0	0	0
Synedra acus	0	0	0	0	0	0	0	0	0	0	12
Synedra acus angustissima	0	0	0	0	0	0	0	0	0	0	0
Synedra rumpens	0	0	0	0	0	4	0	0	0	0	0
Synedra ulna	2	0	0	4	0	10	0	2	22	0	2
Tabellaria flocculosa	0	0	0	0	0	0	0	0	0	0	0
Tabularia fasciculata	2	0	10	0	0	0	6	0	28	6	0
Thalassiosira weisflogii	0	0	0	0	0	0	0	0	0	0	0
Tryblionella calida	0	0	0	0	0	0	0	0	0	0	0
Tryblionella debilis	0	2	4	0	0	0	0	0	0	0	0
Tryblionella levendensis	0	0	0	0	0	0	0	0	0	0	0
Tryblionella lev. victorae	0	0	0	0	0	0	0	0	0	0	0

Appendix 3

Authorities for diatom taxa mentioned in the text and / or illustrated.

Taxon name	Authority
Achnanthidium minutissimum	(Kützing) Czarnecki 1994
Achnanthes aff. *saccula*	J.R. Carter in J.R. Carter & Watts 1981
Achnanthes cleveii	Grunow in Cleve and Grunow 1880
Achnanthes oblongella	Ostr. 1902
Achnanthes petersonii	Hustedt 1936
Achnanthes levanderi	Hustedt 1936
Achnanthes pusilla	Grnow in Cleve and Grunow 1880
Amphora libyca	Ehrenberg ex Kützing 1844
Amphora pediculus	(Kützing) Grunow ex A. Schmidt 1975
Amphora veneta	Kützing 1844
Anomoeoneis vitrea	(Grunow) R Ross in Patrick & Reimer 1966
Asterionella formosa	Hassall 1850
Aulacoseira subarctica forma *subborealis*	(O. Muller) Haworth 1988
Aulacoseira ambigua	(Grunow in Van Heurck) Simonsen 1979
Aulacoseira granulata	(Ehrenberg) Simonson 1979
Aulacoseira italica	(Ehrenberg) Simonsen 1979
Bacillaria paradoxa	(Muller) Hendey 1951
Cocconeis placentula	Ehrenberg 1838
Cyclostephanos aff. *tholiformis*	Stoermer, Håkansson and Theriot 1987
Cyclostephanos invisitatus	(Frick in A. Schmidt) Round 1982
Cyclotella atomus	Hustedt 1937
Cyclotella meneghiniana	Kützing 1844
Cyclotella pseudostelligera	Hustedt 1939
Cyclotella stelligera	(Cleve & Grunow in Cleve) Van Heurck 1882
Cymbella affinis	Kutzing 1844
Cymbella aspera	(Ehrenberg) H. Perag. in Pell
Cymbella cistula	(Ehrenberg in Hempr & Ehrenberg) Kirchner 1878
Encyonema gracilis	(Rabenhorst) Cleve 1894
Cymbella microcephala	Grunow in Van Heurck 1880
Encyonema minuta	Hilse ex Rabenhorst 1862
Denticula tenuis	Hilse ex Rabenhorst 1844
Diatoma tenuis	Agardh 1812
Diploneis elliptica	(Kutzing) Cleve 1894
Diploneis parma	Cleve 1891
Epithemia adnata	(Kutzing) Rabenhorst 1853
Epithemia sorex	Kützing 1844
Eunotia serpentina	Ehrenberg 1844
Eunotia bilunaris	(Ehrenberg) F.W. Mills 1934
Eunotia exigua	(Breb, ex Kutz.) A. Berg 1939
Eunotia incisa	W. Smith ex Greg 1854
Eunotia paludosa	Grunow 1862
Eunotia naegelii	Migula 1907
Pseudostaurosira brevistriata	(Grunow in Van Heurck) Williams and Round 1987
Fragilaria capucina var. *gracilis*	(Oestrup) Hustedt 1950
Fragilaria capucina var. *rumpens*	(Kützing) Lange-Bertalot 1991
Fragilaria capucina var. *capucina*	Desm 1825
Fragilaria parasitica	(W.Smith) Grunow in Van Heurck 1881
Fragilaria tenera	(W. Smith) Lange – Bertalot 1980
Staurosira construens var. *construens*	(Ehrenberg) Grunow 1862
Staurosira construens var. *venter*	(Ehrenberg) Hustedt 1957
Staurosirella pinnata	(Ehrenberg) Williams and Round 1987

Gomphonema clavatum	Ehrenberg 18 2
Gomphonema affine	Kutzing 1844
Gomphonema gracile	Ehrenberg 1832
Gomphonema pseudoaugur	Lange – Bertalot 1979
Gomphonema olivaceum	(Hornemann) Brébisson 1838
Gomphonema parvulum	(Kützing) Kützing 1849
Gomphonema truncatum	Ehrenberg 1832
Gyrosigma acuminatum	(Kützing) Rabenhorst 1853
Hantzschia amphioxys	(Ehrenberg) Grunow 1877
Karayevia clevei	(Grunow in Cleve & Grunow) Round and Bukhtiyarova 1996
Melosira varians	Agardh 1827
Maymaea atomus	(Kützing) Lange Bertalot 1997
Hippodonta capitata	(Ehrenberg) Lange Bertalot, Witkowski and Metzeltin 1996
Navicula clementis	Grunow 1882
Navicula confervacea	(Kützing) Grunow in Van Heurck 1880
Navicula cryptocephala	Kützing 1844
Navicula cryptotenella	Lange-Bertalot 1985
Navicula elginensis	(Gre.) Ralfs in Pritch. 1861
Navicula erifuga	Lange-Bertalot 1985
Navicula gregaria	Donkin 1861
Craticula halophila	(Grunow ex Van Heurck) Mann 1990
Navicula halophiloides	Hustedt 1959
Navicula heimansii	Van Dam & Kooyman 1982
Navicula incertata	Lange – Bertalot 1985
Navicula lanceolata	(Agardh) Kutzing 1838
Navicula menisculus	Schum 1867
Navicula minuscula	Grunow in Van Heurck 1880
Navicula obsoleta	Hustedt 1942
Sellaphora pupula	(Kützing) Mereschkowsky 1902
Fallacia pygmaea	(Kützing) Stickle and Mann 1990
Navicula radiosa	Kützing 1844
Navicula rhyncocephala	Kützing 1844
Navicula schroeterii	Meister 1932
Navicula seminulum	Grunow 1860
Navicula subminuscula	Manguin 1941
Fallacia tenera	(Hustedt) Mann 1990
Navicula veneta	Kützing 1844
Navicula viridula	(Kutzing) Ehrenberg 1836
Nitzschia acidoclinata	Lange Bertalot 1976
Nitzschia amphibia	Grunow 1862
Nitzschia capitellata	Hustedt 1930
Nitzschia clausii	Hantzsch 1860
Nitzschia dissipata	Hustedt 1944
Nitzschia filiformis	(W. Smith) Van Heurck 1896
Nitzschia filiformis var. *conferta*	(Richter) Lange-Bertalot 1987
Nitzschia fonticola	Grunow in Van Heurck 1880
Nitzschia gracilis	Hantzsch 1860
Nitzschia intermedia	Hantzsch ex Cleve & Grunow 1880
Nitzschia lacuum	Lange-Bertalot 1980
Nitzschia liebetruthii	Rabenhorst 1864
Nitzschia linearis	W. Smith 1853
Nitzschia palea	Kützing 1844
Nitzschia paleacea	(Grunow in Cleve & Grunow) Grunow in Van Heurck 1881
Nitzschia pumila	Hustedt 1954
Nitzschia solita	Hustedt 1953
Nitzschia supralitorea	Lange-Bertalot 1979
Pinnularia borealis	Ehrenberg 1843
Pinnularia obscura	Krasske 1932

Pinnularia viridis	(Nitzsch) Ehrenberg 1843
Planothidium delicatulum	(Kützing) Round and Bukhtiyarova (1996)
Planothidium lanceolatum	(Brébisson) Round and Bukhtiyarova (1996)
Rhoicosphenia abbreviata	Lange – Bertalot, 1980
Rhopalodia brebissonii	Krammer 1987
Rhopalodia gibba	(Ehrenberg) O. Muller 1895
Rhopalodia musculus	(Kutz.) O. Mull.,1899
Stephanodiscus parvus	Stoermer & Hakansson 1984
Surirella brebisonii	Krammer and Lange – Bertalot 1987
Synedra ulna	(Nitzsch) Lange-Bertalot 1980
Synedra acus	Kutzing (1844)
Tabellaria flocculosa	(Roth) Kützing 1844
Tabularia fasciculata	(Agardh) Williams and Round 1986
Thalassiosira lacustris	Cleve 1873
Thalassiosira sp. 2	Cleve 1873
Tryblionella hungarica	(Grunow) Mann 1990
Tryblionella levidensis	W. Smith 1856

Appendix 4

Water quality measurement methods

Variable	University of Adelaide	CSIRO Land and Water, Canberra	Australian Water Quality Centre, South Australia
pH	Measured in field, using a Metrohm M588 pH meter	Measured in field, using a Metrohm Herisau pH meter	Measured in field using a Orion 230Aplus portable pH meter
Temperature	Measured in field using a CDM80 conductivity meter	Enviro-safer NIST	WQ101 field thermometer
Colour	HACH DR/800 colorimeter	Aquafast11 colorimeter	HANNA H193727 Colormeter
Turbidity (NTU)	HACH turbidometer (2100A)	WQ 70 turbidity sensor	HACH 2100P turbidimeter
Dissolved oxygen (DO)	YSI dissolved oxygen probe	YSI dissolved oxygen probe	DRT titration, Winkler method
Soluble phosphorus (SRP)	ascorbic acid molybdate (APHA, 1992)	Method adapted from the ascorbic acid molybdate analysis (APHA, 1992)	Automated ascorbic acid molybdate colorimetric analysis (adapted from APHA, 1992)
Total phosphorus (TP)	Persulfate digestion followed by the Molybdate Blue method (after Murphy and Riley, 1962)	Persulfate digestion APHA (1995) followed by APHA 4500 (1995)	APHA method 4500 PE
Nitrate / nitrite (NOX)	Cadmium Reduction Method (Wetzel and Likens, 1991)	Cadmium Reduction Method (APHA, 1992)	APHA method 4500 NO3
Total Keldjhal Nitrogen (TKN)	Acid digestion using Hg catalyst followed by colorimetric analysis using salicylate hypochlorite (adapted from APHA, 1992)	Acid digestion using Hg catalyst followed by APHA (1995) 4500	Acid digestion using Hg catalyst followed by APHA (1995) 4500
Electricai Conductivity (EC)	Measured in field using a CDM80 conductivity meter, calibrated to 25°C	Orion 105Aplus portable conductivity meter	Orion 135A advanced conductivity meter
Sodium (% total cations)	Calculated as difference between total anions and total cations	APHA (1992) method 3111B	APHA (1992) method 3111B
Potassium (% total cation)	Tetraphenylborate Method method (adapted from APHA, 1992)	APHA (1992) method 3111B	APHA (1992) method 3111B
Magnesium (% total cations)		APHA (1992) method 3111B	APHA (1992) method 3111B
Calcium (% total cations)	ManVer 11 Buret titration (adapted from APHA, 1992)	APHA (1992) method 3111B	APHA (1992method 3111B
Chloride (% total anions)		APHA (1992) method 4500 Cl	APHA (1992) method 4500 Cl
Sulphate (% total anions)	SulfaVer 4 method (adapted from APHA, 1992)	APHA (1992) method 3111 SO_4	APHA (1992) method 3111 SO_4
Alkalinity (% total anions)	Buret Titration Method (APHA, 1992)	APHA (1992) method 2320	APHA (1992) method 2320

Appendix 5

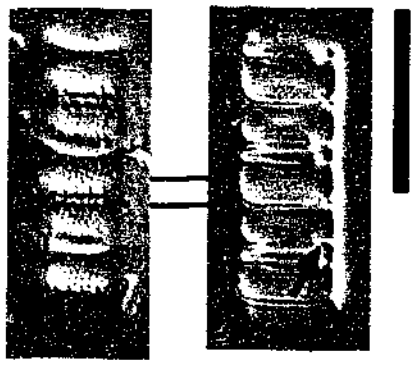
Diatom taxon optima and tolerance for pH, derived from a 45 lake data set

Taxon Name	No. of samples	Max	*N2*	Jack-Opt
Achnanthes brevipes	4	0.64	3.9578	**7.46**
Planothidium delicatulum	26	38.11	7.4042	**7.84**
Achnanthes exigua	2	10.13	1.4751	**8.96**
Achnanthes lanceolata	4	0.64	3.9294	**7.79**
Achnanthidium minutissimum	12	46.43	5.0105	**7.37**
Achnanthes oblengella	1	6.54	1	**7.30**
Achnanthes pericava	3	50.25	1.9799	**8.69**
Actinocyclus normanii	11	25.32	2.8587	**8.38**
Amphora coffaeformis var *coffaeformis*	1	49.69	1	**8.00**
Amphora coffaeformis var. *borealis*	3	2.69	1.9015	**9.07**
Amphora pediculus	5	3.1	4.1107	**7.55**
Amphora veneta	15	18.36	6.1727	**7.95**
Aulacoseira subbartica forma *subborealis*	2	0.63	1.995	**7.9**
Aulacoseira ambigua	1	8.7	1	**7.90**
Aulacoseira granulata	16	68.38	7.701	**8.24**
Aulacoseira italica	9	8.76	6.3433	**8.23**
Bacillaria paradoxa	4	15.55	1.4848	**7.92**
Climaconeis sp.	1	5.84	1	**8.00**
Cocconeis placentula	14	25.71	7.3352	**8.17**
Cyclotella atomus	21	53.97	11.9067	**8.27**
Cyclotella meneghiniana	42	54.40	2.4587	**8.57**
Cyclotella stelligera	14	14.91	7.6991	**8.63**
Cymatapleura solea	1	0.68	1	**8.60**
Cymbella affinis	3	4.85	1.7473	**6.72**
Cymbella cistula	1	0.98	1	**8.20**
Cymbella gracilis	2	0.68	1.9922	**8.05**
Cymbella microcephala	4	3.58	3.1103	**7.37**
Cymbella minutum	4	23.73	1.3393	**8.40**
Cymbella pusilla	1	2.97	1	**8.80**
Cymbella silesiaca	1	1.47	1	**8.07**
Cymbella tumida	2	58.91	1.0193	**7.12**
Diatoma tenuis	4	11.99	2.309	**8.52**
Diploneis elliptica	4	6.35	2.7185	**8.75**
Diploneis parma	5	12.35	1.9273	**8.38**
Epithemia adnata	18	13.10	1	**8.43**
Epithemia sorex	8	22.63	2.3302	**7.99**
Eunotia bilunaris	2	1.96	1.5203	**7.97**
Eunotia serpentina	13	50.39	8.3053	**8.30**
Fallacia tenera	5	9.38	1.8223	**8.75**
Pseudostaurosira brevistriata	26	50.1	6.4454	**8.30**
Fragilaria capucina var *gracilis*	6	24.27	2.1969	**6.94**
Fragilaria capucina var *rumpens*	8	14.53	4.9452	**8.08**
Staurosira construens var *venter*	17	70.0	1.1971	**6.94**
Fragilaria construens var *construens*	16	12.5	9.7328	**8.33**
Fragilaria elliptica	4	8.25	2.0757	**7.49**
Staurosirella pinnata	16	16.14	1.9278	**8.33**

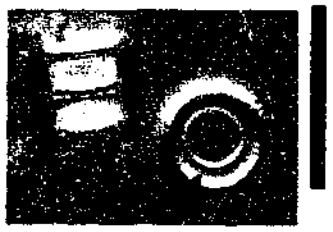
Taxon Name	No. of samples	Max	N2	Jack-Opt
Frustulia rhomboides	1	3.41	1	**7.70**
Frustulia vulgaris	1	0.6	1	**7.10**
Gomphonema acuminatum	3	1.14	2.7249	**8.10**
Gomphonema affinis	3	9.85	2.1351	**7.08**
Gomphonema clavatum	1	1.49	1	**7.30**
Gomphonema gracile	12	15.71	5.4875	**7.65**
Gomphonema minutum	3	8.52	1.5064	**7.53**
Gomphonema olivaceum	7	3.47	4.6992	**7.89**
Gomphonema parvulum	12	15.0	1.5903	**7.65**
Gomphonema truncatum	6	3.09	4.5165	**8.38**
Gyrosigma parkerii	9	8.03	4.1015	**8.01**
Gyrosigma spencerii	3	2.37	2.0911	**8.80**
Hantzschia amphioxys	7	68.23	3.8246	**7.48**
Mastagloia smithii	9	7.16	5.6088	**7.68**
Melosira varians	7	68.12	2.8351	**7.48**
Navicula angusta	5	28.16	1.7584	**7.40**
Navicula capitata	1	0.62	1	**8.40**
Navicula cincta	14	37.26	3.8711	**7.39**
Navicula cincta var. minuta	9	23.45	3.2861	**7.52**
Navicula cocconeiformis	9	4.38	5.6246	**7.69**
Navicula cryptocephala	39	38.8	3.4284	**7.39**
Navicula cryptotenella	6	17.97	2.2478	**7.30**
Navicula erifuga	13	16.34	4.0054	**7.70**
Navicula gregaria	2	0.54	1.8636	**8.81**
Navicula halophila	1	3.92	1	**8.20**
Navicula incertata	2	37.25	1.3293	**7.19**
Navicula lanceolata	1	0.66	1	**8.40**
Navicula leptostriata	1	2.21	1	**8.90**
Navicula menisculus	1	16.67	1	**8.30**
Navicula mutica	4	21.73	1.9455	**7.29**
Navicula phyllepta	1	4.17	1	**8.07**
Navicula pupula	8	10.22	4.0001	**7.44**
Navicula pygmaea	4	1.3	3.4977	**7.97**
Navicula radiosa	4	1.21	3.7946	**7.73**
Navicula rhycnocephala	3	2.39	2.4973	**7.20**
Navicula spicula	5	70.71	2.0228	**8.97**
Navicula veneta	19	15.15	9.7917	**7.61**
Navicula viridula	1	0.57	1	**8.30**
Nitzschia agnita	2	1.34	1.9963	**8.91**
Nitzschia archibaldii	3	10.07	1.8205	**9.04**
Nitzschia bacillum	2	1.49	1.9652	**7.60**
Nitzschia capitellata	3	1.82	2.1419	**7.99**
Nitzschia clausii	1	0.85	1	**8.20**
Nitzschia constricta	2	39.95	1.049	**7.95**
Nitzschia desortorum	13	8.75	6.1008	**8.33**
Nitzschia filiformis	1	1.21	1	**8.00**
Nitzschia fonticola	1	1.94	1	**7.50**
Nitzschia frustulum	10	28.62	3.9894	**7.14**
Nitzschia gracilis	6	11.78	2.5151	**7.24**
Nitzschia hungarica	10	15.19	4.513	**8.64**

Taxon Name	No. of samples	Max	N2	Jack-Opt
Nitzschia lacuum	9	8.84	4.4814	**8.65**
Nitzschia linearis	17	29.53	3.09	**8.90**
Nitzschia littoralis	22	26.41	7.4769	**8.66**
Nitzschia microcephala	5	6.27	2.7832	**8.31**
Nitzschia palacea	2	1.21	1.6534	**8.09**
Nitzschia palae	14	8.0	1	**8.44**
Nitzschia subacicularis	1	0.61	1	**8.00**
Nitzschia tryblionella	14	8.84	6.6582	**8.44**
Opephora olsenii	16	5.52	9.8485	**7.97**
Pleurosigma elongatum	2	1.92	1.6075	**7.28**
Rhiocosphenia abbreviata	2	1.1	1.8555	**8.89**
Rhopalodia brebisonii	11	3.6	7.3106	**8.51**
Rhopalodia gibba	4	8.0	3.4235	**7.66**
Rhopalodia musculus	4	0.71	3.9681	**8.29**
Stauroneis obtusa	1	0.98	1	**8.20**
Stauroneis pachycephala	1	0.62	1	**7.80**
Stephanodiscus hantzschii	1	2.91	1	**7.50**
Surirella angusta	2	1.23	1.9964	**8.36**
Surirella brebisonii	5	3.42	3.492	**8.42**
Surirella ovalis	2	31.55	1.1965	**7.50**
Surirella robusta	1	4.41	1	**8.07**
Synedra acus	2	8.33	1.2367	**7.66**
Synedra acus angustissima	4	2.12	3.1424	**7.94**
Synedra rumpens	3	5.48	2.161	**7.23**
Synedra ulna	18	7.9	1	**7.94**
Tabellaria fasciculata	7	34.0	4.6714	**7.63**
Tabellaria flocculosa	1	1.24	1	**7.80**
Thallosiasira weisflogii	4	1.23	3.4567	**8.49**
Thallossiora fluvialis	2	0.97	1.8736	**7.88**

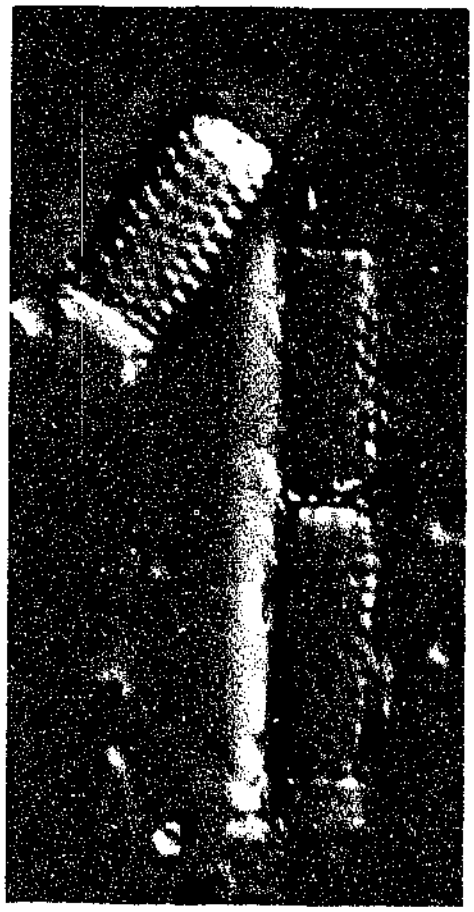
a.



b.

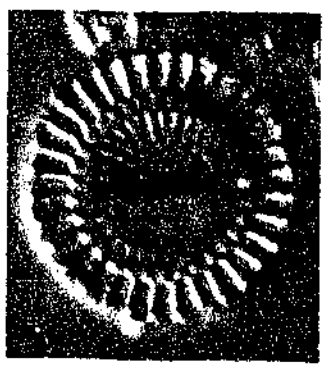


a. & b. *Aulacoseira subarctica* f. *subborealis*
(the position of the ringleiste is indicated by an arrow)

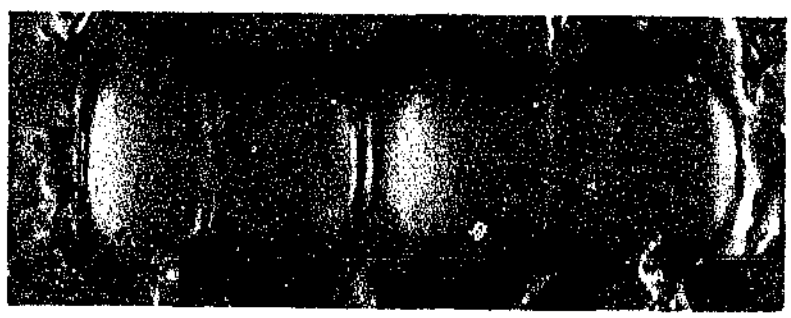


d. *Aulacoseira granulata*
(the separation spine is indicated by an arrow)

c. *Synedra ulna*



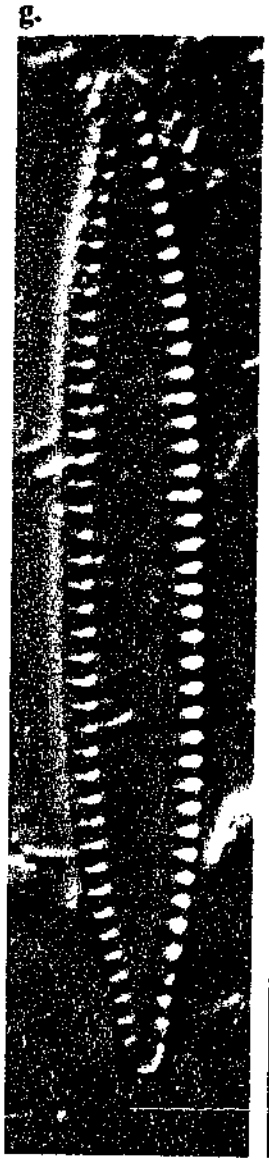
e. *Cyclotella meneghiniana*



f. *Melosira varians*

Appendix 6: Diatom photomicrographs of the taxa described in Chapter 5. Black bar line represents 10 microns

g.

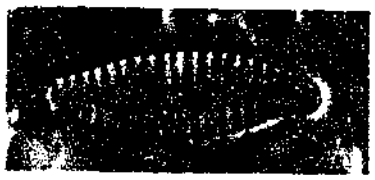


h.

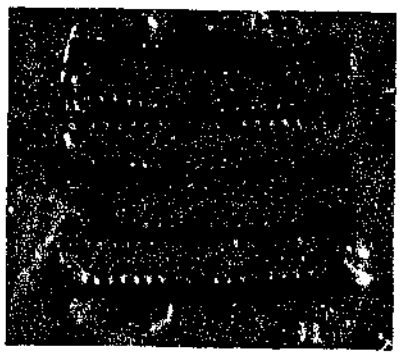
i.

h. & i. *Staurosirella pinnata*
(valve and girdle views are shown, respectively)

j.



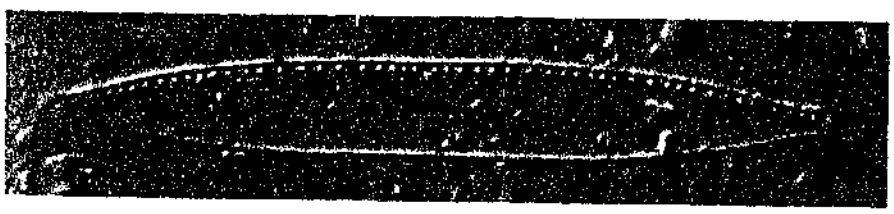
k.



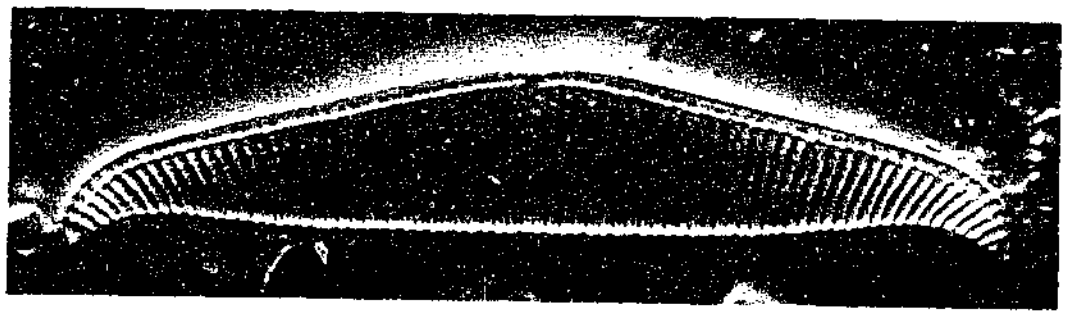
j. & k. *Staurosira construens* f. *venter*
(valve and girdle views are shown, respectively)

g. *Tabularia fasciculata*

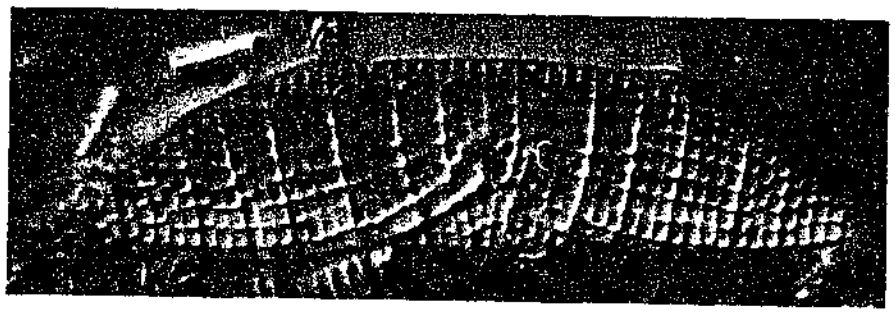
l. *Pseudostaurosira brevistriata*



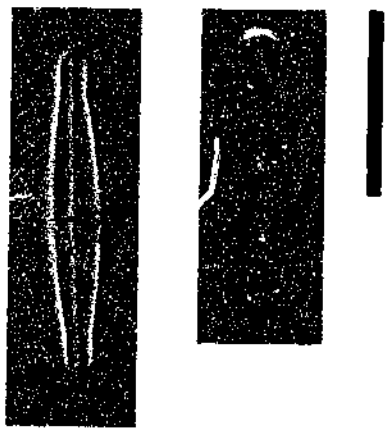
m. *Nitzschia palea*



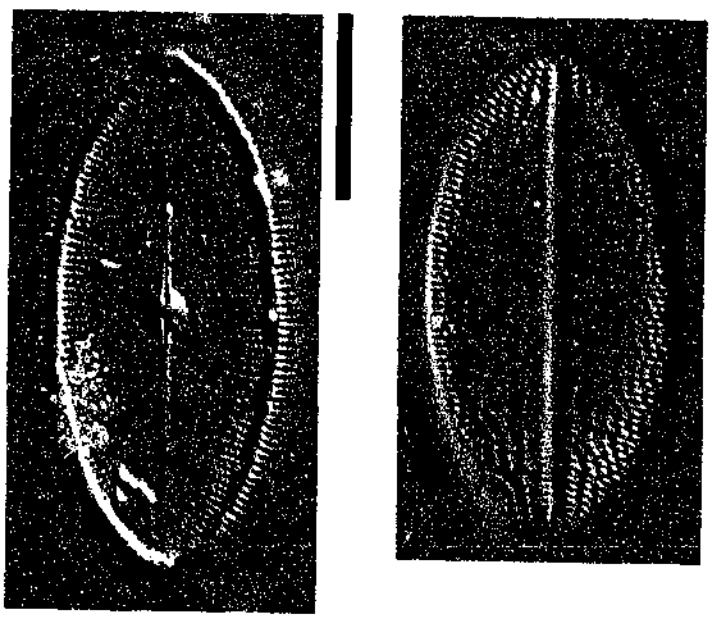
n. *Rhopalodia gibba*



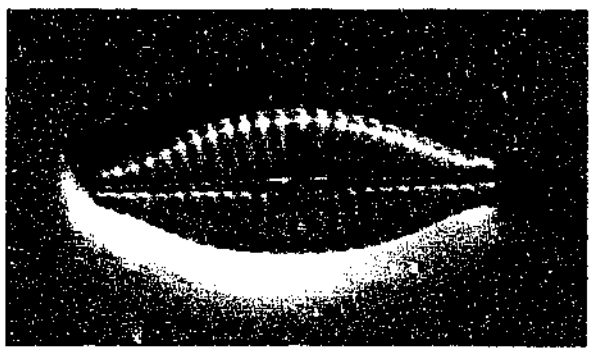
o. *Epithemia adnata*



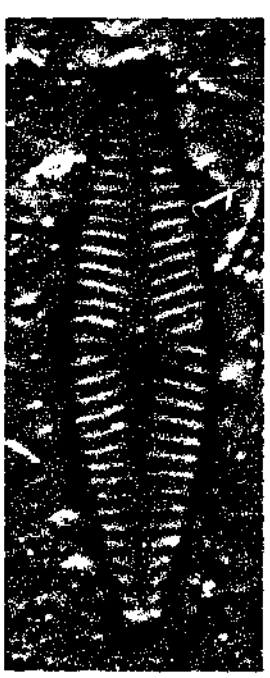
p. *Achnanthidium minutissimum*
(valve and girdle views are shown, respectively)



q. *Cocconeis placentula*
(raphid and araphid valves are shown, respectively)

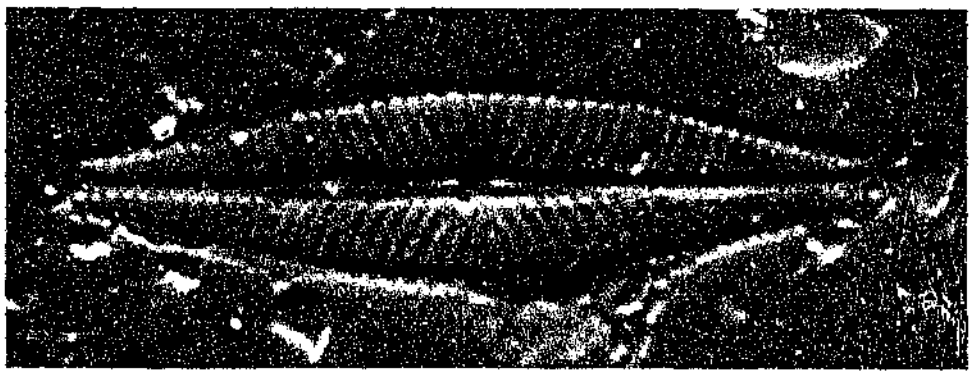


r. *Planothidium delicatulum*



s. *Navicula veneta*

t. *Gomphonema parvulum*



u. *Navicula cryptocephala*

Appendix 7

Preparation of core sediment samples for the purposes of detecting *Pinus* pollen.

Sampling

A 1 cm sample was extracted at each sampling depth. Material was taken from the middle of the core at each level so as to minimise the potential for contamination by surface material transported downwards by the passage of the coring tube. The weight of each sample was recorded.

Dispersion and staining

Following weighing, the sample was placed in a 250 ml conical flask containing 100 ml of 10% KOH and 10 drops of saffranin stain. A magnetic flea was added to the flask which was then covered b inverting a small beaker over the opening. The flask was then placed on a hotplate with a magnetic stirrer and heated at 100°C for 30 minutes.

Isolation of silt sized particles

After the 30 minute heating period, the dispersed sediment was washed through a 65 µm sieve using distilled water. The material passing through the sieve was collected and it was then washed through a 35 µm sieve.

Separating minerogenic material

The contents of the 35 µm sieve were then washed into a settling column of approximately 120 cm depth. The column consisted of two titration columns joined together by means of a sleeve of clear plastic tubing. Each of the columns had been cut so as to create the smoothest possible join. In the case of the lower column, the top lip was removed, while for the upper column, the base was removed. In most cases the process of washing the sieve contents into the column was sufficient to produce a 120 cm water column in the settling column. Further water was added if the resultant column was less than 120 cm. Material was allowed to settle in the column for 8 minutes, after which time the bottom 10 ml was collected and discarded and the remaining material collected. This material was then washed once more through a 35 µm sieve. The contents of the sieve were subsequently washed into a 15 ml centrifuge tube. Depending on the concentration of material, samples in each tube were then concentrated by centrifuging and decanting. In most cases 10 ml yielded a concentration suitable for detecting *Pinus* grains.

Counting *Pinus* grains

The sample (or subsample) was then pipetted onto a glass microscope slide and the material examined using a dissecting microscope at around 160 x magnification. This magnification was varied according to the nature of the material. In order to facilitate reliable counting, the glass slide was placed over a white ceramic tile marked with vertical lines placed at 5 mm intervals. The sample was then examined along transects of 5 mm width. For each sample, the number of *Pinus* grains in the entire sample was counted. If subsampling was used, each subsample was washed into a fresh centrifuge tube following a completed scan and a further subsample was pipetted onto the slide. This process was continued until the whole sample had been examined.

Appendix 8 – Fossil diatom counts

Core 1 (Lake Alexandrina)

	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	10cm	11cm	12cm	15cm	16cm	20cm	22cm	24cm
Aulacoseira granulata	16.6	31.3	31.5	28.5	19.6	10.7	47.2	19.2	21.2	79.4	81.1	36.6	85.6	60.2	23.0	86.5	97.5
A. subbartica var.subborealis	0.0	0.0	0.0	0.0	1.2	0.0	0.9	0.4	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planothidium delicatulum	0.3	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	1.1	0.6	0.0
Achnanthes lanceolata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. oblongella	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amphora coffaeformis	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0
Camplyiodiscus clypeus	0.0	0.0	0.6	0.0	0.6	0.9	2.6	2.8	1.6	3.4	4.2	0.0	1.1	7.0	1.8	0.6	0.0
Cocconeis placentula	0.3	0.3	0.0	0.3	0.0	0.0	0.9	0.4	0.6	0.4	0.0	0.7	1.1	1.6	0.7	0.0	0.0
Cyclostephanos tholiformis	0.0	0.0	0.0	0.0	0.3	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cyclotella meneghiniana	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
C. striata	0.0	0.6	0.3	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cymbella gracilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.1	0.0	0.0	0.0
Diploneis parma	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.1	0.0	0.0	0.0
Actinocyclus normanii	0.9	1.2	0.3	0.3	0.6	0.0	0.4	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Epithemia adnata	0.3	0.0	0.0	0.0	0.0	0.0	0.4	0.4	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eunotia serpentina	0.0	0.	[illegible]	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fallacia tenera	0.0	0.	[illegible]	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurosirella pinnata	56.0	38	[illegible]	[illegible]	41.6	61.3	33.0	58.4	50.5	7.7	3.0	40.1	2.7	0.0	39.6	1.8	0.0
Staurosira construens forma venter	6.0	[illegible]	[illegible]	[illegible]	17.3	13.4	5.6	8.2	15.4	0.4	0.0	9.3	0.0	1.6	27.2	0.0	0.0
Pseudostaurosira brevistriata	10.5	8.8	8.9	[illegible]	10.6	6.8	1.7	6.0	4.5	0.0	0.0	6.1	0.5	0.0	0.7	0.0	0.0
Staurosira construens var. construens	6.6	3.3	6.0	3.5	5.0	1.5	0.0	0.0	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria elliptica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0
Gyrosigma parkerii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hyaladiscus sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.9	1.1	0.7	0.0	0.0	0.0	0.0	0.0
Mastogloia smithii	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Paralia sulcata	0.3	0.0	0.3	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.8	0.4	0.0	0.8	0.4	0.6	0.0
Navicula converacea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0
N. cryptocephala	0.0	0.0	0.3	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	10cm	11cm	12cm	15cm	16cm	20cm	22cm	24cm
N. phyllepta	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. veneta	0.3	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nitzschia compressa	0.3	0.9	0.3	1.4	0.3	1.5	3.0	1.4	3.2	2.6	4.5	2.2	5.3	14.1	2.8	3.1	2.2
N. constricta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0
N. hungarica	0.0	0.9	0.6	0.3	0.0	0.3	0.0	0.4	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. intermedia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lacuum	0.0	0.0	0.3	0.0	0.3	0.3	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0
N. subacicularis	0.3	0.0	0.3	0.0	0.3	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Opephora olsenii	0.0	0.0	0.0	0.0	0.0	0.3	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pinnularia sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7	3.0	0.7	0.5	3.1	1.1	5.5	0.4
Pleurosigma elongatum	0.0	0.0	0.0	0.0	0.3	0.0	0.4	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0
Rhopalodia musculus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6	0.0	1.2	0.0
Stephanodiscus parvus	0.0	0.0	0.3	0.3	0.3	0.0	0.4	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Surirella robusta	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Synedra ulna	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0
Tabularia fasciculata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.7	0.0	1.6	0.0	0.0	0.0
Thalasiossira lacustris	0.9	0.9	0.6	0.5	0.6	1.2	1.7	0.4	0.6	0.4	0.8	0.4	3.2	1.6	0.7	0.0	0.0
Tryblionella levendensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0

Core 2 (Lake Alexandrina)

	0cm	4cm	8cm	20cm	40cm	60cm	80cm	100cm	120cm	160cm	180cm	220cm	260cm	300cm
Achnanthes brevipes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.3	0.0	0.0	0.0
A. oblongella	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0
Achnanthidium lanceolatum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.3
Actinocyclus normanii	3.5	0.6	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.3	0.3	0.6	1.4
Amphora coffaeformis	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.3	0.0	0.4	0.3	0.6	0.0	0.0
A. libyca	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aulacoseira ambigua	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. granulata	14.8	0.0	5.0	6.4	8.8	8.6	35.0	41.1	9.0	29.1	43.0	12.2	1.2	1.7
A. italica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.6	0.0	0.7
Cocconeis placentula	1.0	1.9	7.7	2.5	0.3	3.3	2.6	0.6	0.7	1.5	3.5	3.1	5.8	16.3
Cyclotella heteroides	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. meneghiniana	0.0	0.0	0.0	1.0	0.0	0.4	0.0	0.0	0.3	0.8	0.0	0.0	1.2	2.1
C. mesiana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
Cymbella gracile	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	0cm	4cm	8cm	20cm	40cm	60cm	80cm	100cm	120cm	160cm	180cm	220cm	260cm	300cm
C. pusilla	0.0	0.0	0.4	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	0.0
C. silisiaca	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
C. striata	1.5	1.3	22.2	3.4	3.7	2.9	4.5	2.2	1.0	3.2	2.9	1.4	6.4	7.9
Cyclostephanos invisitatus	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.3	0.3	2.3	3.3
Cyclotella atomus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Diploneis parma	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
D. psuedovalis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Epithemia adnata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
E. sorex	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eunotia serpentina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.3	0.0	0.0	0.0
Fallacia pygmaea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.3	4.7	1.2	0.0
F. tenera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
Fragilaria capensis	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. elliptica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7
F. capucina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
F. leptostriata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gomphonema parvulum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
Gyrosigma attentuatum	0.0	0.0	0.8	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. spencerii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Hantzschia amphioxys	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mastagloia smithii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
Navicula angusta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. amplicatum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. cari	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
N. cincta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
N. concentrica	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. cryptocephala	0.5	0.0	1.9	0.5	0.3	0.0	0.0	0.6	3.0	0.0	0.0	1.7	0.6	1.5
N. cryptotenella	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.3	0.3	0.0	0.0
N. cuspidata	0.0	0.0	1.9	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. durrenbergiana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
N. erifuga	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. gregaria	0.5	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.7	0.0	0.0	0.3	0.0	0.0
N. halophila	0.0	0.0	0.0	0.0	1.7	0.0	0.8	0.0	0.3	0.0	0.3	0.0	1.7	0.7
N. menisculus	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. phyllepta	0.0	0.0	0.0	0.0	1.3	0.0	0.0	0.6	1.3	0.0	0.0	0.3	0.0	0.0
N. ponduriformis lata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. rhynocephala	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	0cm	4cm	8cm	20cm	40cm	60cm	80cm	100cm	120cm	160cm	180cm	220cm	260cm	300cm
N. spicula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.3	0.0	0.0
N. trivialis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. veneta	0.0	0.0	0.0	3.0	0.3	0.0	0.0	0.0	0.3	0.4	0.9	0.8	1.2	1.4
Nitzschia agnita	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0
N. archibaldii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
N. calida	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.3	0.0	2.2	1.4	1.2	2.1
N. capitellata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7
N. compressa	0.5	0.6	3.1	1.0	3.4	0.0	0.0	0.9	1.2	0.4	4.1	2.8	2.6	6.2
N. constricta	0.0	0.0	0.4	0.0	0.0	0.0	0.4	0.0	0.0	0.8	0.0	0.0	0.6	0.7
N. dissipata media	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. filiformis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
N. fonticola	0.5	0.0	0.0	1.0	0.0	0.0	0.0	2.5	0.0	1.1	0.0	0.3	5.2	1.2
N. fonticola var. pelagia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.2	1.7	0.6	0.0	6.7	0.3
N. frustulum	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.3
N. gracilis	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	1.0	0.0	0.0	0.0	0.0	0.0
N. hungarica	0.5	1.9	2.7	0.0	1.7	0.0	1.9	1.9	0.0	4.1	1.2	1.7	5.8	2.1
N. intermedia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lacuum	0.0	0.6	0.0	0.0	2.7	1.6	0.4	5.7	1.3	0.0	0.0	1.4	18.9	1.7
N. linearis	0.0	0.0	0.0	0.0	0.3	0.0	0.4	0.0	0.7	0.0	0.0	0.0	0.0	0.0
N. palacea	0.5	0.0	0.0	0.0	0.3	0.0	1.1	1.6	2.3	0.0	0.0	0.0	1.2	0.3
N. palae	0.0	0.0	0.0	1.0	0.7	0.0	0.0	0.0	0.0	0.4	0.0	0.3	1.7	0.3
N. pusilla	0.5	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. reversa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
N. sigma	0.5	1.3	0.0	0.5	0.0	0.0	0.0	0.3	0.0	0.4	0.0	1.0	0.6	0.0
N. subacicularis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Opephora olsenii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
Planothidium delicatulum	0.0	0.0	0.4	1.5	0.0	0.8	0.0	0.6	0.3	0.4	1.2	1.4	0.0	1.0
Pimmularia divergitissima	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
P. viridis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
Pleurosigma elongatum	0.0	0.0	0.4	0.0	0.0	0.0	0.8	0.0	0.3	0.0	0.3	3.5	0.6	0.5
Paralia sulcata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pseudostaurosira brevistriata	36.6	64.4	16.4	15.8	16.2	4.1	1.5	2.4	2.3	0.8	0.3	0.8	4.7	1.0
Rhopalodia gibba	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
R. musculus	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.3	0.0	0.7	0.0	0.0	0.7
Surirella angusta	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stephanodiscus parvus	1.5	1.9	2.7	0.5	1.0	0.0	0.8	0.0	0.7	0.4	0.6	1.9	9.9	7.2
Staurosirella pinnata	24.6	0.0	16.6	44.8	25.8	50.6	33.1	32.0	57.8	28.2	13.7	29.9	11.9	10.3

	0cm	4cm	8cm	20cm	40cm	60cm	80cm	100cm	120cm	160cm	180cm	220cm	260cm	300cm
Surirella robusta	0.0	0.0	0.8	0.0	0.3	0.4	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Staurosira construens var. construens	2.5	0.0	0.0	4.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurosira construens var. venter	6.0	24.8	10.6	7.9	17.9	24.4	5.6	0.9	5.7	10.2	0.9	0.3	1.2	3.4
Stephanodiscus hantzschii	0.0	0.6	1.2	0.5	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	2.6
Synedra ulna	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
Tabularia fasciculata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Thalasiossira baltica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	1.7	1.0
T lacustris	0.5	0.0	3.1	2.0	10.8	1.6	7.9	2.5	3.4	13.7	18.7	23.6	2.9	14.9
T. weisflogii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tryblionella levendensis	0.0	0.0	0.0	0.0	0.7	0.4	0.8	0.9	0.7	0.8	0.6	0.8	0.0	0.0

	340cm	380cm	420cm	460cm	490cm
Achnanthes brevipes	0.0	0.0	0.0	0.4	0.0
A. oblongella	0.0	0.0	0.0	0.0	0.0
Achnanthidum lanceolata	0.5	0.0	0.0	0.4	0.0
Amphora coffaeformis	0.0	0.7	0.0	4.4	1.3
A. libyca	0.5	0.0	0.0	0.0	0.4
Aulacoseira ambigua	0.0	0.0	0.0	0.0	0.0
A. granulata	1.0	11.6	13.3	0.0	13.6
A. italica	1.0	0.0	1.7	0.8	3.1
Actinocyclus normanii	0.0	0.4	1.1	4.4	0.9
Cymbella gracile	0.0	0.0	0.0	0.4	0.0
C. pusilla	0.5	0.0	0.0	0.0	0.4
C. silisiaca	0.0	0.0	0.0	0.0	0.0
Cyclotella atomus	0.0	0.0	0.0	2.0	0.0
C. heteroides	0.0	0.0	0.0	3.2	3.1
C. invisitatus	1.5	0.4	0.0	3.2	0.9
C. meneghiniana	0.5	0.4	0.6	0.4	1.3
C. riesiana	0.0	0.0	0.0	0.0	0.0
C. striata	1.0	5.3	3.4	8.1	3.6
Cocconeis placentula	9.4	9.5	6.8	8.3	28.6
Diploneis parma	0.0	0.0	0.0	0.0	0.4
D. pseudovalis	0.0	0.0	0.0	0.0	0.0
Epithemia adnata	0.0	0.0	1.1	0.0	0.0

	340cm	380cm	420cm	460cm	490cm
E. sorex	0.0	0.7	0.0	0.0	0.0
Eunotia serpentina	0.0	0.0	0.0	0.0	0.0
Fallacia pygmaea	0.0	0.0	0.0	0.0	0.0
F. tenera	1.0	1.1	0.0	3.8	5.8
Fragilaria capensis	0.0	0.0	0.0	0.0	0.0
F. capucina	0.0	0.0	0.0	0.0	0.0
F. elliptica	0.0	1.1	0.0	0.0	0.0
F. leptostriata	0.0	0.7	1.7	0.0	0.0
Gyrosigma attentuatum	0.0	0.0	0.0	0.0	0.0
G. spencerii	0.0	0.4	0.0	0.0	0.0
Gomphonema parvulum	0.0	0.0	0.0	0.0	0.0
Hantzschia amphioxys	0.0	0.0	0.0	0.0	0.0
Mastagloia smithii	0.0	0.4	0.0	5.6	0.0
Paralia sulcata	9.4	3.2	0.0	6.7	3.6
Navicula angusta	0.0	0.7	0.0	0.0	0.0
N. cari	0.0	0.0	0.0	0.0	0.0
N. cincta	0.0	0.0	0.0	0.0	0.0
N. concentrica	0.0	0.0	0.0	0.0	0.0
N. cryptocephala	3.6	0.7	0.6	1.6	0.4
N. cryptotenella	0.0	0.0	0.0	0.0	0.0
N. cuspidata	0.0	0.0	0.0	0.0	0.0
N. durrenbergiana	1.0	0.0	0.0	0.0	0.0
N. erifuga	0.0	0.0	0.0	0.0	0.0
N. gregaria	0.0	0.0	0.0	0.0	0.0
N. halophila	0.0	0.0	0.0	0.4	0.4
N. menisculus	0.0	0.0	0.0	0.0	0.0
N. phyllepta	1.0	0.0	0.6	0.4	0.0
N. ponduriformis lata	0.0	0.0	0.6	0.0	0.0
N. rhynocephala	0.0	0.0	0.0	0.0	0.0
N. trivialis	0.0	0.0	0.0	0.0	0.4
N. veneta	0.0	0.4	0.6	0.8	0.0
Nitzschia agnita	0.0	0.0	0.0	0.0	0.0
N. amplicatum	0.0	0.0	0.0	0.4	0.0
N. archibaldii	0.0	0.0	0.0	0.0	0.0
N. calida	1.5	0.7	2.3	1.6	0.9
N. capitellata	0.0	0.0	0.0	0.0	0.0
N. compressa	2.0	1.8	2.8	0.0	4.9

	340cm	380cm	420cm	460cm	490cm
N. constricta	0.5	0.0	1.1	1.6	0.9
N. dissipata media	0.0	0.0	0.0	0.0	0.9
N. filiformis	0.0	0.0	0.0	0.0	0.0
N. fonticola	2.5	0.7	2.3	0.0	0.0
N. fonticola var. pelagia	1.5	1.1	1.7	0.8	0.9
N. frustulum	0.0	0.0	0.0	0.0	0.0
N. gracilis	0.0	0.0	0.0	0.0	0.0
N. hungarica	8.6	4.9	2.8	4.4	2.2
N. intermedia	1.0	0.9	0.0	0.0	0.0
N. lacuum	0.5	0.0	1.7	4.4	0.0
N. linearis	0.0	0.0	0.0	0.0	0.0
N. palacea	0.0	0.0	0.0	0.0	0.0
N. palae	0.0	0.0	0.0	0.4	0.0
N. pusilla	0.0	0.0	0.0	0.0	0.0
N. reversa	0.0	0.0	0.0	0.0	0.0
N. sigma	0.0	0.0	0.0	4.4	0.4
N. spicula	0.0	0.0	0.0	0.0	0.0
N. subacicularis	0.0	0.0	0.0	0.0	0.0
Opephora olsenii	1.0	0.0	1.1	0.0	0.9
Pseudostaurosira brevistriata	0.0	2.8	2.3	0.0	0.4
Planothidium delicatulum	1.0	0.7	0.0	1.2	1.3
Pinnularia divergitissima	0.0	0.0	0.0	0.0	0.0
P. viridis	0.0	0.0	0.0	0.0	0.0
Pleurosigma elongatum	0.5	2.8	0.0	1.4	0.9
Rhopalodia gibba	0.0	0.0	0.0	0.0	0.0
R. musculus	0.5	0.0	1.1	3.6	1.8
Surirella angusta	0.0	0.0	0.0	0.0	0.0
S. robusta	0.0	0.0	0.0	0.0	0.0
Staurosira construens var. construens	0.0	1.9	0.0	0.0	0.0
Stephanodiscus hantzschii	0.0	0.0	0.0	0.0	0.0
S. parvus	1.0	2.1	1.7	2.8	0.4
Staurosirella pinnata	35.0	23.6	32.9	0.0	6.3
Synedra ulna	0.5	0.0	0.0	0.0	0.0
Staurosira construens var. venter	0.0	0.0	1.1	0.4	0.0
Thalasiossira baltica	0.0	2.8	0.0	0.0	0.0
T. lacustris	10.2	16.0	13.0	16.9	8.1

	340cm	380cm	420cm	460cm	490cm
T. weisflogii	0.0	0.0	0.0	0.4	0.0
Tabularia fasciculata	0.0	0.0	0.0	0.4	0.0
Tryblionella levendensis	0.0	0.0	0.0	0.0	0.0

Core 3 (Lake Alexandrina)

	0cm	0.5cm	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	9.5cm	10cm	11cm	12cm	13cm	14cm
Achnanthes brevipes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. exigua	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. oblongella	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. saccula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Achnanthdium lanceolatum	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.5	1.0	0.0	0.0	0.0	0.5	0.9	0.0	0.0
Actinocyclus normanii	0.5	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amphora coffaeformis	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.5	0.5	0.0	0.0
A. libyca	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. pediculus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
A. veneta	0.0	0.0	0.6	1.7	0.0	0.0	1.1	1.5	1.4	0.5	0.6	0.0	1.9	2.9	1.4	3.5	0.5
Aulacoseira granulata	3.1	5.0	10.1	11.1	8.3	1.7	1.6	2.0	2.7	2.0	2.3	4.0	0.0	0.0	0.9	1.2	1.6
A. italica	1.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0
Bacillaria paradoxa	0.5	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camplyiodiscus molaris	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0
C. clypeus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cocconeis placentula	26.9	28.2	19.4	25.1	26.8	34.1	24.6	24.6	22.0	23.1	24.9	31.9	34.3	30.2	33.8	24.5	21.3
Cyclostephanos tholiformis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	1.2	0.0
C. invisitatus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cyclotella atomus	1.9	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. meneghiana	0.0	0.0	1.1	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0
C. pseudostelligera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. stelligera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. striata	0.0	0.0	0.6	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0
Cymbella amphicephala	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0
C. aspera	0.5	0.6	0.0	0.0	0.6	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.5	0.0	0.0
C. caespitosa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. cesatii	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	0cm	0.5cm	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	9.5cm	10cm	11cm	12cm	13cm	14cm
C. cistula	1.7	4.1	2.3	3.5	2.5	9.6	8.5	5.0	7.7	3.0	11.3	5.0	7.5	4.9	8.0	4.7	1.6
C. gracilis	2.4	1.2	1.7	0.6	1.3	0.0	0.0	0.0	0.0	0.5	1.1	1.0	0.0	0.0	0.0	0.0	0.0
C. mesiana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.6	0.0	0.0	0.0	0.0
C. minutum	0.0	0.0	0.0	1.2	0.0	0.0	0.0	1.0	0.0	0.5	0.0	0.5	0.0	0.0	0.0	0.0	0.5
C. pusilla	1.9	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. silesiaca	0.0	1.2	0.0	0.6	0.0	0.0	0.0	1.0	0.9	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. tumida	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.5	0.0	0.0
Diploneis elliptica	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
D. parma	5.3	4.1	3.9	1.2	2.5	2.8	2.2	0.5	0.0	1.5	1.7	2.5	1.3	1.5	1.4	1.2	0.5
D. subcontricta	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Entomoneis alata	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Epithemia adnata	13.9	21.8	18.6	9.6	15.9	11.3	12.8	21.6	13.2	10.3	19.5	17.3	8.2	8.3	14.9	8.3	13.4
E. sorex	0.0	0.0	0.0	0.0	0.6	0.6	0.0	0.5	0.0	0.0	0.6	0.0	0.0	0.5	0.9	0.0	0.0
E. turgida var. granulata	0.0	1.2	1.7	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0
Eunotia bilunaris	2.4	0.0	3.4	0.0	2.5	1.7	1.6	0.5	2.3	0.0	0.6	1.5	0.0	0.0	1.9	0.3	0.0
E. pectinalis var. undulata	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
E. praerupta bigibba	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
E. serpentina	0.0	0.6	3.4	2.9	3.2	0.0	1.1	0.5	0.0	0.0	0.0	0.5	0.0	1.0	0.0	1.2	0.0
E. soleirolli	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria capucina car. gracilis	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. elliptica	1.4	0.0	0.0	0.0	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
F. leptostriata	1.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. capucina rumpens	0.5	2.4	1.7	2.3	3.2	3.4	3.8	3.5	5.9	5.0	2.8	1.5	7.5	6.1	2.8	2.9	2.6
F. lata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	0.0	0.0
F. vaucherie	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gomphonema augur var.turis	0.0	1.8	0.6	0.0	0.0	0.6	1.1	1.5	1.4	1.0	0.0	0.5	1.9	2.4	0.5	0.6	1.6
G. clavatum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. gracile	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.5
G. insigne	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. minutum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. parvulum	0.0	0.0	0.0	0.6	1.3	1.7	0.5	1.5	2.3	3.0	2.3	0.5	4.4	3.4	1.9	0.6	3.2
Gyrosigma attenuatum	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.5	0.0	0.6	0.0
G. spencerii	1.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. nodiferum	2.4	1.2	0.0	0.6	1.3	0.0	0.5	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0
Hantzschia amphioxys	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Luticola mutica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Melosira sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	0cm	0.5cm	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	9.5cm	10cm	11cm	12cm	13cm	14cm
M. varians	0.0	4.7	2.8	0.6	0.0	3.9	2.7	6.5	7.3	12.1	6.2	4.0	5.7	6.1	1.2	0.6	1.3
M. circulare	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Navicula absoluta	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. amphibola	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. atomus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. capitata	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. capidigitoradiata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. clememtoides	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. converacea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. cryptocephala	0.5	0.0	0.6	0.6	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. cryptotenella	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. elgenensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0
N. gotlandica	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. halophila	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lanceolata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. ordinaria	0.0	0.0	0.6	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0
N. pupula	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. radiosa	1.4	0.6	0.0	0.6	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	1.0	0.9	0.0	0.0
N. recens	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. salinarum	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. seminulum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. subminiscula	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. submuralis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. subplacentula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. veneta	0.5	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.5
N. viridula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0
Nitzschia agnita	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. amphibia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. angustata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. calida	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
N. compressa	0.0	0.6	0.0	2.3	0.0	1.1	2.2	1.0	0.0	1.0	1.7	0.5	0.0	1.0	1.4	0.6	0.5
N. constricta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. fonticola	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. fonticola var. pelagia	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	1.2	0.0
N. frustulum	1.4	0.0	2.8	2.3	0.6	2.3	4.4	1.5	4.1	4.0	1.7	1.5	0.6	1.0	2.4	1.2	3.9
N. frustulum var. bulnhemii	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	1.9	1.5	0.9	0.0	0.5
N. gracilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.5	0.0	0.0	0.9	0.0	0.0	0.0	0.0

	0cm	0.5cm	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	9.5cm	10cm	11cm	12cm	13cm	14cm
N. hungarica	1.9	0.0	0.0	1.7	1.9	0.0	0.0	0.5	0.9	0.0	0.0	0.0	0.0	0.5	0.5	0.0	0.0
N. incognita	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. inconspicua	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lacuum	0.0	0.0	0.0	0.0	1.3	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. levendensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. levendensis salinarum	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. levendensis var. victoriae	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. liebetruthii	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. linearis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. littoralis	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
N. lorenziana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. palae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. palacea	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	1.4	2.4	0.0
N. pusilla	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. shoenfeldii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. sigma	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. subacicularis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.3	3.6	0.9	2.4	0.0
Opephora olsenii	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Paralia solcata	1.9	0.0	1.1	0.0	1.9	0.0	2.7	0.0	0.5	0.0	0.0	0.0	1.3	0.0	0.0	0.0	0.5
Pinnularia appendiculata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
P. elegans	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
P. nobilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planothidium delicatulum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.5
Pleurosigma elongatum	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pseudostaurosira brevistriata	0.0	0.0	5.6	5.8	2.5	1.7	1.1	2.0	0.0	1.0	0.6	3.0	0.0	0.0	0.5	0.0	0.0
Rhopalodia gibba	4.1	11.2	4.5	3.5	2.5	10.1	5.5	7.9	7.7	6.0	6.8	7.8	13.8	10.0	6.6	5.9	7.9
R. gibba var. minuta	1.0	0.0	0.0	0.0	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R. musculus	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stauroneis acuta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. obtusa	0.0	0.6	0.6	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. wislouchii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurosira construens	0.5	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4
Staurosira construens var.venter	0.0	1.8	0.0	1.2	0.0	0.0	3.3	0.0	2.3	0.5	2.8	0.5	0.6	0.0	0.9	3.5	4.7
Staurosirella pinnata	7.2	0.0	1.1	12.2	5.7	8.5	8.2	9.7	10.0	13.6	9.1	7.0	0.0	1.0	3.8	21.8	21.1
Stephanodiscus parvus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Surirella angusta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. brebissonnii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	0cm	0.5cm	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	9.5cm	10cm	11cm	12cm	13cm	14cm
S. gracilis	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. ovalis	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. robusta	1.4	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1
Synedra acus v. angustissima	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0
S. ulna	3.4	5.0	5.1	2.3	2.5	2.3	3.8	4.5	0.9	1.5	1.7	3.8	3.8	1.5	2.4	2.4	2.6
Tabularia fasciculata	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.9	2.3	0.0	0.5	1.9	7.5	0.9	3.5	2.6
Thalasiossira lacustris	0.0	0.0	0.6	0.6	1.3	0.6	0.5	0.0	0.5	0.0	0.6	0.5	0.0	0.5	1.4	0.6	1.1
Tryblionella levendensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	15cm	16cm	20cm	21cm	22cm	24cm	25cm	26cm	27cm	28cm	29cm	32cm	33cm	34cm	36cm	38cm	44cm
Achnanthes brevipes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0
A. exigua	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. oblongella	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. saccula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Achnanthdium lanceolatum	0.0	0.0	0.0	0.6	0.0	0.0	0.3	0.0	0.4	0.0	0.0	1.7	1.7	0.0	0.6	0.0	0.5
Actinocyclus normanii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0
Amphora coffaeformis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0
A. libyca	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	2.1	0.7	2.7	0.7	0.7	0.0	0.6	0.0	1.0
A. pediculus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. veneta	0.0	0.0	0.6	0.0	2.7	0.0	1.3	1.4	0.0	0.0	0.0	0.7	0.7	0.0	0.0	3.5	1.5
Aulacoseira granulata	1.8	2.2	1.4	4.5	3.5	0.0	1.9	4.0	0.0	0.3	0.6	2.7	2.7	2.8	3.3	9.0	19.1
A. italica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1
Bacillaria paradoxa	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.5
Camplyiodiscus molaris	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. clypeus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cocconeis placentula	20.0	24.0	10.6	20.8	8.4	10.0	15.5	25.9	16.5	4.5	3.7	9.3	9.3	19.6	23.7	23.3	17.0
Cyclostephanos tholiformis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	3.1
C. invisitatus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
Cyclotella atomus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. meneghiana	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0	1.0
C. pseudostelligera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
C. stelligera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. striata	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.4	1.4	0.0	0.0	0.0	0.0	1.2	0.0	2.6
Cymbella amphicephala	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. aspera	0.0	0.0	0.6	0.0	0.0	0.3	0.6	1.7	1.6	0.0	0.0	0.0	0.0	0.0	0.6	0.6	1.0
C. caespitosa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	15cm	16cm	20cm	21cm	22cm	24cm	25cm	26cm	27cm	28cm	29cm	32cm	33cm	34cm	36cm	38cm	44cm
C. cesatii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. cistula	2.3	3.8	0.0	1.2	1.6	1.7	4.5	4.3	0.0	1.4	0.6	0.0	0.0	1.8	0.6	3.5	1.5
C. gracilis	0.0	1.1	2.2	0.0	6.5	2.8	0.6	0.0	0.0	0.0	0.0	1.4	1.4	2.8	0.6	0.6	0.0
C. mesiana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. minutum	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0
C. pusilla	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. silesiaca	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.0	0.5
C. tumida	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Diploneis elliptica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
D. parma	2.0	1.1	1.1	6.3	3.0	1.4	2.6	3.7	2.9	1.7	1.8	4.1	4.1	3.5	3.0	3.5	0.5
D. subcontricta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Entomoneis alata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Epithemia adnata	7.6	11.1	17.4	25.9	14.6	13.9	11.0	24.4	20.2	9.8	19.8	11.3	11.3	19.3	16.9	12.5	3.6
E. sorex	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.5
E. turgida var. granulata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eunotia bilunaris	0.0	0.0	4.2	2.4	2.2	1.1	3.6	1.1	1.6	2.4	0.0	0.0	0.0	0.0	1.8	1.7	0.5
E. pectinalis var. undulata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
E. praerupta bigibba	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
E. serpentina	0.5	1.1	1.1	0.6	0.0	0.6	1.0	0.9	0.0	0.0	0.9	1.4	1.4	0.0	2.7	1.7	0.5
E. soleirolli	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria capucina car. gracilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. elliptica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.0	0.0	0.0	0.0	0.0	0.0
F. leptostriata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. capucina rumpens	3.5	3.5	2.8	0.0	1.6	5.0	2.6	1.4	4.1	1.7	0.0	1.4	1.4	0.0	1.8	1.2	2.1
F. lata	0.0	0.0	0.0	1.5	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. vaucherie	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gomphonema augur var.turis	1.0	0.0	0.0	1.5	0.0	0.0	0.0	1.4	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.6	1.0
G. clavatum	0.0	0.5	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. gracile	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. insigne	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. minutum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. parvulum	1.0	0.5	0.3	0.6	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0
Gyrosigma attenuatum	0.0	0.5	0.0	0.6	0.0	0.0	1.6	0.0	0.0	0.0	0.0	0.3	0.3	0.4	0.0	0.0	0.0
G. spencerii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. nodiferum	0.0	0.0	0.0	3.6	1.6	0.0	0.0	0.0	0.0	0.0	0.0	1.4	1.4	0.0	0.6	0.0	2.1

	15cm	16cm	20cm	21cm	22cm	24cm	25cm	26cm	27cm	28cm	29cm	32cm	33cm	34cm	36cm	38cm	44cm
Hantzschia amphioxys	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.5
Luticola mutica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Melosira sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
M. varians	2.5	1.6	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0
M. circulare	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Navicula absoluta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. amphibola	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. atomus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. capitata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. capidigitoradiata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. clememtoides	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0
N. converacea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
N. cryptocephala	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. cryptotenella	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0
N. elgenensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. gotlandica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. halophila	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lanceolata	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. ordinaria	0.5	0.0	0.0	0.0	0.0	1.1	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. pupula	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. radiosa	1.0	0.5	0.6	0.0	0.0	0.0	0.0	0.0	1.6	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
N. recens	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. salinarum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. seminulum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. subminiscula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. submuralis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. subplacentula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. veneta	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. viridula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nitzschia agnita	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. amphibia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
N. angustata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. calida	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. compressa	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.6	0.0
N. constricta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. fonticola	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. fonticola var. pelagia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	15cm	16cm	20cm	21cm	22cm	24cm	25cm	26cm	27cm	28cm	29cm	32cm	33cm	34cm	36cm	38cm	44cm
N. frustulum	1.0	2.2	1.7	0.3	1.6	2.8	2.6	1.7	0.0	0.0	0.0	2.1	2.1	2.8	0.0	2.3	0.0
N. frustulum var. bulnhemii	0.3	0.0	0.6	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. gracilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. hungarica	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.5
N. incognita	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. inconspicua	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lacuum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. levendensis	0.0	0.0	1.1	0.0	1.6	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0
N. levendensis salinarum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. levendensis var. victoriae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. liebetruthii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. linearis	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. littoralis	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lorenziana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. palae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0
N. palacea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. pusilla	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. shoenfeldii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. sigma	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. subacicularis	1.3	1.1	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Opephora olsenii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Paralia solcata	0.0	0.5	1.1	1.2	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
Pinnularia appendiculata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
P. elegans	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
P. nobilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planothidium delicatulum	0.5	1.1	0.0	0.0	0.0	0.0	0.0	0.3	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.5
Pleurosigma elongatum	0.0	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	1.4	1.4	0.0	0.0	0.0	0.0
Pseudostaurosira brevistriata	1.5	0.0	0.6	0.6	0.0	0.0	0 0	0.0	0.0	0.0	0.0	2.7	2.7	0.0	1.2	0.0	4.6
Rhopalodia gibba	4.3	2.2	3.9	0.6	2.7	2.8	3.9	6.8	4.9	0.0	0.3	0.0	0.0	1.8	2.1	2.9	1.5
R. gibba var. minuta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R. musculus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stauroneis acuta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. obtusa	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. wislouchii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurosira construens	1.0	0.5	2.2	0.0	0.0	3.6	3.2	0.0	2.5	10.1	0.0	0.0	0.0	2.1	0.0	0.0	0.0

	15cm	16cm	20cm	21cm	22cm	24cm	25cm	26cm	27cm	28cm	29cm	32cm	33cm	34cm	36cm	38cm	44cm
Staurosira cconstruens var.venter	4.6	2.7	3.4	0.9	0.0	0.0	0.0	1.1	0.0	0.0	0.0	2.1	2.1	0.0	2.4	5.2	0.0
Staurosirella pinnata	34.4	29.4	36.7	19.6	38.2	45.8	41.4	15.3	31.7	64.7	61.9	52.9	52.9	40.7	27.8	12.2	22.2
Stephanodiscus parvus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
Surirella angusta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. brebissonnii	0.5	0.0	0.6	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
S. gracilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. ovalis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. robusta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Synedra acus v. angustissima	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. ulna	2.0	3.8	1.7	3.0	1.1	1.7	0.0	2.8	3.3	0.0	1.5	1.7	1.7	0.0	1.8	5.2	2.6
Tabularia fasciculata	2.5	1.1	0.6	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0
Thalasiossira lacustris	0.0	1.6	0.0	1.2	1.6	0.3	0.0	1.1	2.1	0.3	0.0	0.0	0.0	2.1	1.2	2.3	1.0
Tryblionella levendensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	46cm	48cm	50cm	52cm	56cm	58cm	60cm	62cm	64cm	66cm	68cm	70cm	72cm	74cm	76cm	80cm	82cm
Achnanthes brevipes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.5	0.4	0.0	0.0	0.0	0.0	0.0	0.0
A. exigua	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. oblongella	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
A. saccula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Achnanthdium lanceolatum	0.0	0.4	0.6	0.0	0.4	0.0	1.1	0.9	1.2	0.9	0.0	1.2	0.4	0.0	0.4	0.7	0.0
Actinocyclus normanii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amphora coffaeformis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
A. libyca	0.0	0.0	0.3	0.0	0.0	0.0	0.4	0.0	0.0	0.9	0.0	0.4	0.4	0.0	0.0	0.7	0.0
A. pediculus	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. veneta	0.0	1.3	0.0	0.3	0.0	0.4	0.4	0.0	0.4	2.7	0.0	0.0	0.0	0.4	0.0	0.7	0.0
Aulacoseira granulata	2.7	12.9	1.9	0.0	1.9	6.0	3.7	3.4	2.7	10.4	2.1	2.9	4.2	1.6	1.6	3.4	4.9
A. italica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Bacillaria paradoxa	0.5	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.0	11.7	0.0	0.0
Camplyiodiscus molaris	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. clypeus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.1	2.2
Cocconeis placentula	23.7	17.9	10.3	4.2	2.3	8.1	9.6	6.9	8.5	18.9	6.4	5.3	7.5	1.6	6.5	7.6	6.6
Cyclostephanos tholiformis	1.4	0.0	0.0	1.9	0.0	0.0	0.4	0.4	0.4	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0

	46cm	48cm	50cm	52cm	56cm	58cm	60cm	62cm	64cm	66cm	68cm	70cm	72cm	74cm	76cm	80cm	82cm
C. invisitatus	0.0	1.8	0.6	1.0	0.0	0.4	0.7	0.4	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.7	0.0
Cyclotella atomus	0.0	0.4	1.0	0.6	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. meneghiana	0.0	0.0	0.3	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. pseudostelligera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. stelligera	0.5	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0
C. striata	1.4	1.3	0.0	0.3	0.0	0.0	0.0	6.0	0.4	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
Cymbella amphicephala	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. aspera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.4	0.5	0.4	0.0	0.0	0.0	0.0	0.0	0.5
C. caespitosa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. cesatii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. cistula	1.8	1.8	1.0	0.0	0.4	0.9	0.4	0.9	2.7	0.9	0.4	0.0	0.0	0.0	0.0	0.0	1.1
C. gracilis	0.5	0.9	0.0	0.3	0.0	1.7	0.7	0.4	5.8	0.0	1.3	1.6	0.8	6.5	7.7	0.7	1.6
C. mesiana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. minutum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. pusilla	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. silesiaca	0.0	0.0	0.3	0.0	0.0	1.3	1.5	0.0	0.0	0.5	0.0	0.0	0.8	0.0	0.0	0.0	0.0
C. tumida	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Diploneis elliptica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
D. parma	0.0	0.9	0.3	0.0	0.0	1.7	1.1	0.4	3.5	1.4	0.4	0.8	3.3	1.2	0.4	7.6	3.3
D. subcontricta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Entomoneis alata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Epithemia adnata	2.7	4.5	1.0	0.3	12.7	8.5	16.3	6.0	8.8	7.7	5.5	3.7	15.1	2.0	2.0	13.8	24.0
E. sorex	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
E. turgida var. granulata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eunotia bilunaris	0.0	0.4	0.0	0.0	0.0	0.4	0.0	1.3	0.4	1.8	0.0	0.4	0.8	0.0	0.0	2.8	3.8
E. pectinalis var. undulata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
E. praerupta bigibba	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
E. serpentina	0.9	1.8	0.0	0.0	0.4	0.9	0.4	0.9	0.4	1.8	0.4	0.0	1.7	0.0	0.8	1.4	2.7
E. soleirolli	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria capucina car. gracilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. elliptica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. leptostriata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. capucina rumpens	1.4	2.7	1.0	0.0	0.0	1.3	2.2	1.7	0.4	0.9	0.4	0.0	0.8	1.2	0.0	1.4	0.0
F. lata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. vaucherie	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.8	0.0	0.0

	46cm	48cm	50cm	52cm	56cm	58cm	60cm	62cm	64cm	66cm	68cm	70cm	72cm	74cm	76cm	80cm	82cm
Gomphonema augur var.turis	0.9	0.4	0.0	0.0	0.0	0.9	0.4	0.0	0.4	0.0	0.4	0.0	0.0	0.0	0.4	0.0	0.0
G. clavatum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. gracile	0.0	0.4	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. insigne	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. minutum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
G. parvulum	0.9	0.9	0.0	0.0	0.0	0.4	0.4	1.3	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gyrosigma attenuatum	0.0	0.0	0.0	0.3	0.0	0.4	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.4	0.4	0.0	2.2
G. spencerii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. nodiferum	0.5	0.0	0.0	0.0	0.0	0.4	0.4	0.0	0.4	0.0	0.4	0.4	0.8	1.6	0.4	0.0	0.5
Hantzschia amphioxys	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Luticola mutica	0.5	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
Melosira sp.	0.0	0.0	11.3	22.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
M. varians	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
M. circulare	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Navicula absoluta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. amphibola	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. atomus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6	0.4	0.7	0.5
N. capitata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. capidigitoradiata	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. clememtoides	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. converacea	0.5	0.4	0.0	0.3	0.0	0.0	0.0	0.0	0.8	0.5	0.0	0.0	0.4	0.0	0.0	0.0	0.0
N. cryptocephala	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0
N. cryptotenella	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
N. elgenensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. gotlandica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. halophila	0.5	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lanceolata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. ordinaria	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. pupula	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. radiosa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.4	0.0	0.0
N. recens	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. salinarum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. seminulum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	1.1
N. subminiscula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. submuralis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
N. subplacentula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0

	46cm	48cm	50cm	52cm	56cm	58cm	60cm	62cm	64cm	66cm	68cm	70cm	72cm	74cm	76cm	80cm	82cm
N. veneta	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0
N. viridula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nitzschia agnita	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
N. amphibia	2.3	0.4	0.3	0.6	0.8	0.0	1.1	1.3	0.4	0.5	0.0	0.0	0.0	0.8	0.4	0.0	0.0
N. angustata	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. calida	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. compressa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.8	0.9	0.4	1.6	0.0	0.0	2.0	0.7	1.1
N. constricta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. fonticola	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. fonticola var. pelagia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. frustulum	0.9	0.4	0.0	0.6	0.8	0.4	0.4	0.4	0.0	0.0	0.0	0.0	0.8	0.0	0.4	0.7	0.0
N. frustulum var. bulnhemii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. gracilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. hungarica	0.5	0.4	0.3	0.0	0.4	0.4	0.4	0.4	0.4	0.0	0.4	0.0	0.0	0.8	0.0	0.0	0.0
N. incognita	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. inconspicua	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lacuum	0.0	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. levendensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. levendensis salinarum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. levendensis var. victoriae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.4	0.0	0.0	0.0	0.0
N. liebetruthii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. linearis	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. littoralis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lorenziana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
N. palae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. palacea	0.0	0.4	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. pusilla	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0
N. shoenfeldii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
N. sigma	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0
N. subacicularis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Opephora olsenii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Paralia solcata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pinnularia appendiculata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
P. elegans	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
P. nobilis	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	46cm	48cm	50cm	52cm	56cm	58cm	60cm	62cm	64cm	66cm	68cm	70cm	72cm	74cm	76cm	80cm	82cm
Planothidium delicatulum	0.9	0.9	0.0	0.0	0.0	1.3	0.0	0.0	0.0	0.9	0.0	0.8	0.4	1.2	0.0	1.4	0.5
Pleurosigma elongatum	0.0	0.4	0.0	0.0	0.0	0.0	0.7	0.0	1.5	0.0	0.9	0.0	0.0	0.4	0.0	0.0	1.1
Pseudostaurosira brevistriata	0.9	7.1	2.6	3.6	1.9	5.1	5.2	3.0	4.6	3.6	11.5	1.6	6.7	8.1	6.5	2.8	3.8
Rhopalodia gibba	1.4	2.7	1.0	0.0	67.6	15.8	14.1	7.3	11.5	7.2	4.7	1.6	7.5	1.2	5.7	7.6	4.4
R. gibba var. minuta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R. musculus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stauroneis acuta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. obtusa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. wislouchii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurosira construens	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurosira construens var.venter	5.5	5.4	6.8	3.2	1.5	9.4	2.6	8.2	1.9	2.3	0.0	1.6	3.8	1.6	1.6	7.6	2.7
Staurosirella pinnata	38.8	24.6	56.8	57.0	3.9	29.9	31.5	38.4	32.7	28.8	62.1	73.3	39.7	59.3	44.5	29.0	27.9
Stephanodiscus parvus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Surirella angusta	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. brebissonnii	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. gracilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. ovalis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. robusta	0.0	0.0	0.3	0.0	0.0	0.0	0.4	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Synedra acus v. angustissima	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. ulna	4.1	2.2	1.3	0.0	3.1	2.6	2.2	1.7	3.5	1.8	0.9	1.6	1.3	0.0	0.4	3.4	0.0
Tabularia fasciculata	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0
Thalasiossira lacustris	0.5	0.9	0.0	0.3	0.0	0.0	0.0	5.2	0.8	0.9	0.0	0.0	1.3	0.4	0.0	0.0	1.1
Tryblionella levendensis	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.5

Core 4 (Lake Cullulleraine)

	0cm	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	10cm	11cm	12cm	13cm	14cm	15cm	16cm
Achnanthes exigua	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Achnanthidium minutissimum	0.5	0.0	0.0	0.9	0.0	3.2	2.1	1.7	1.3	2.2	1.5	2.7	1.5	0.4	1.1	0.4	0.0
Actinocyclus normanii	0.0	0.3	0.5	0.9	0.7	1.9	0.5	4.2	3.4	3.3	3.5	0.8	5.7	7.6	1.1	2.6	1.3
Amphipleura rutilans	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0

	0cm	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	10cm	11cm	12cm	13cm	14cm	15cm	16cm
Amphora coffaeformis var. coffaeformis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. libyca	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. pediculus	0.2	0.0	0.0	0.0	0.0	0.0	0.0	2.5	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0
A. veneta	0.0	0.0	0.0	0.0	0.3	0.6	1.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.8	0.0	0.0
A. sphaerophora	0.2	0.3	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
A. formosa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aulacoseira subartica var. subborealis	0.2	0.6	0.0	0.3	0.0	0.0	0.5	0.0	2.0	2.2	0.5	0.8	0.8	0.4	1.1	0.0	0.0
A. granulata	1.7	2.6	3.0	2.6	3.5	6.5	8.2	5.9	5.4	8.8	15.8	10.1	11.8	16.3	17.5	23.2	15.3
Bacillaria paradoxa	0.2	0.0	0.0	0.0	0.0	0.6	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cocconeis placentula	4.1	5.8	5.7	4.4	4.2	19.4	16.9	20.3	23.5	15.9	19.8	17.5	18.6	11.8	20.2	18.4	20.9
Cyclostephanos tholiformis	0.0	0.3	0.3	0.0	0.3	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cyclotella meneghiniana	0.7	1.3	1.1	2.0	1.4	0.0	0.5	1.7	2.7	0.5	1.5	0.8	0.0	1.5	1.1	0.9	0.7
C. stelligera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cymatapleura solea	0.2	0.0	0.8	0.6	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.4	0.4	0.0	0.0	0.0	0.7
Cymbella amphicephala	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
C. aspera	0.0	0.0	0.0	0.3	0.0	1.9	1.0	0.0	0.0	0.0	1.0	0.4	0.4	0.0	1.1	0.4	1.7
C. cistula	0.5	0.3	0.3	0.9	1.0	3.2	1.0	0.8	0.0	0.0	0.0	0.8	0.4	0.0	1.1	0.0	2.0
C. gracile	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. microcephala	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0
C. minutum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. silisiaca	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0
Diploneis parma	0.0	0.3	0.8	1.7	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Epithemia adnata	1.0	1.9	2.7	0.9	1.4	2.6	6.2	6.8	3.4	3.8	5.0	1.9	2.3	2.3	0.8	0.9	2.7
E. sorex	5.5	2.9	4.1	5.2	7.0	13.5	18.5	13.6	16.1	17.6	8.9	15.6	9.1	8.0	6.8	12.7	8.3
Eunotia serpentina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria elliptica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0
F. tenera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. capucina gracilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. capucina rumpens	1.2	0.0	0.3	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7
F. capucina vaucherie	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gomphonema angustatum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.3	0.0	0.0	0.0	0.0	0.0
G. minutum	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. truncatum	1.4	1.0	0.3	0.0	0.7	4.5	6.7	2.5	0.7	0.5	1.5	3.9	2.3	1.1	1.9	0.9	2.0
G. parvulum	0.5	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.7	0.0	0.5	0.0	0.4	0.0	0.0	0.0	0.0

	0cm	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	10cm	11cm	12cm	13cm	14cm	15cm	16cm
Gyrosigma attentuatum	0.0	0.0	0.8	0.6	0.0	0.6	0.0	0.0	0.0	1.1	0.0	0.4	0.8	0.0	0.4	0.4	0.0
G. nodiferum	0.7	0.6	0.0	0.0	0.3	0.0	1.5	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.4	0.0
G. parkerii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. spencerii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hantzschia amphioxys	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Luticola mutica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Melosira varians	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.4	0.0	0.0	0.0
Navicula capitata	0.0	0.0	0.0	0.3	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
N. capitigidorada	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. cryptocephala	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.4	0.0	0.4	0.0	0.0
N. cuspidata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. elgensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. leptostriata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. radiosa	0.2	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.4	1.0
N. rhynocephala	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
N. phyllepta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. stiolata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. trivialis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. veneta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0
N. viridula	1.0	0.0	0.0	0.3	0.7	0.6	1.0	0.0	0.7	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0
N. vulpina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nitzschia acicularoides	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. compressa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.4	0.0	0.0	0.4	0.0	0.0
N. constricta	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. dissipata media	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. fonticola	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. frustulum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	2.7	0.0	0.5	0.0	0.4	0.0	0.8	0.4	0.0
N. hungarica	0.5	0.0	0.0	0.9	0.3	0.6	0.5	0.0	0.0	0.0	0.0	0.4	0.0	0.4	0.4	0.4	0.0
N. lacuum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. levendensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0
N. linearis	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lorenziana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. palae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0
N. palacea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. sigma	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. subacicularis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
Planothidium delicatulum	0.2	0.0	0.5	1.5	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0

	0cm	1cm	2cm	3cm	4cm	5cm	6cm	7cm	8cm	9cm	10cm	11cm	12cm	13cm	14cm	15cm	16cm
Pleurosigma elongatum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.3
Pseudostaurosira brevistriata	12.0	19.4	17.7	28.9	31.5	9.7	6.2	10.2	7.4	15.4	10.4	7.8	14.8	17.1	19.0	15.8	13.0
Rhopalodia gibba	0.7	1.3	2.2	1.5	0.0	2.6	1.0	0.0	0.0	0.5	0.5	1.2	1.1	0.0	0.8	0.4	1.3
R. musculus	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stauroneis obtusa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurosira construens var. venter	8.7	6.8	4.9	5.2	8.4	6.5	4.6	1.7	6.7	3.3	7.9	10.5	11.4	17.1	13.7	9.2	15.0
Staurosirella pinnata	45.0	29.1	36.8	21.9	27.3	11.6	8.7	10.2	1.3	8.8	5.9	8.2	5.7	10.3	4.6	5.3	10.0
Stephanodiscus parvus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Surirella angusta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. robusta	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Synedra acus angustissima	0.0	0.0	0.0	0.0	0.0	0.6	1.5	0.8	2.7	2.7	0.5	1.9	1.9	0.0	1.5	1.3	0.3
S. ulna	0.5	1.0	0.3	0.3	0.0	1.9	1.5	1.7	3.4	1.1	2.0	0.8	1.1	0.0	0.8	0.4	0.0
Tabularia fasciculata	11.1	22.7	16.3	17.2	7.3	3.9	6.2	12.7	14.1	11.0	12.9	7.0	3.4	4.6	1.5	2.6	2.0
Thalasiossira lacustris	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tryblionella levendensis	0.0	0.3	0.0	0.6	0.0	0.0	0.5	0.0	0.7	0.5	0.0	0.8	0.4	0.0	0.0	0.9	0.7

	17cm	18cm	19cm	20cm	21cm	22cm	23cm	24cm	25cm	26cm	27cm	28cm	29cm	30cm	31cm	32cm	33cm	34cm	35cm
Achnanthes exigua	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Achnanthidium minutissimum	0.7	0.3	0.3	7.1	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Actinocyclus normanii	1.0	0.3	0.0	0.3	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amphipleura rutilans	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amphora coffaeformis var. coffaeformis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.4
A. libyca	0.0	0.7	1.0	0.0	0.8	2.0	0.3	0.9	0.6	0.3	0.0	0.9	1.3	1.8	1.8	1.1	0.3	0.0	0.0
A. pediculus	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. veneta	1.0	1.3	0.3	1.7	0.6	0.3	0.5	0.6	0.6	0.3	0.0	0.9	0.0	0.6	0.3	0.0	0.0	0.0	1.3
A. sphaerophora	0.0	0.7	0.0	0.0	0.3	0.7	1.3	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
A. formosa	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aulacoseira subartica var. subborealis	0.0	0.3	0.7	0.0	1.4	1.7	3.1	10.7	15.1	20.9	25.0	32.4	28.5	29.0	44.4	63.5	65.1	77.7	71.2

	17cm	18cm	19cm	20cm	21cm	22cm	23cm	24cm	25cm	26cm	27cm	28cm	29cm	30cm	31cm	32cm	33cm	34cm	35cm
A. granulata	17.5	10.1	6.6	4.2	7.2	2.6	1.8	0.6	0.0	0.3	0.0	1.5	0.5	0.3	0.0	0.4	0.3	0.0	0.0
Bacillaria paradoxa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cocconeis placentula	23.0	24.6	21.2	9.9	9.9	6.0	5.1	7.0	6.0	10.3	7.8	6.7	7.2	2.4	5.9	2.1	1.3	0.0	0.4
Cyclostephanos tholiformis	0.0	0.0	0.0	0.0	0.0	0.7	1.5	0.3	0.9	0.0	0.0	0.9	0.3	0.0	0.0	0.0	0.3	0.5	0.4
Cyclotella meneghiniana	0.7	2.0	0.7	0.0	0.3	0.7	0.0	0.0	0.6	0.6	0.9	0.3	0.3	0.3	0.0	0.0	0.0	0.5	0.4
C. stelligera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.6	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cymatapleura solea	0.3	0.7	0.0	0.8	0.0	0.3	0.0	0.0	0.0	0.3	0.0	0.9	0.3	1.8	0.3	1.1	0.3	0.0	0.0
Cymbella amphicephala	0.3	0.3	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. aspera	0.7	0.3	1.3	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. cistula	2.1	1.7	2.0	0.3	0.6	0.3	0.5	0.3	0.3	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. gracile	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
C. microcephala	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. minutum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
C. silisiaca	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0
Diploneis parma	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.3	0.0	0.0
Epithemia adnata	1.7	0.0	0.7	0.0	1.9	0.7	1.5	2.1	5.1	6.1	7.8	9.3	6.7	3.6	4.4	1.8	1.3	2.1	2.1
E. sorex	9.6	9.4	12.6	3.4	5.2	3.0	2.0	2.4	4.5	4.2	4.4	6.4	6.7	7.8	1.8	3.2	2.0	1.0	1.3
Eunotia serpentina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria elliptica	0.0	1.0	1.7	4.0	1.1	3.0	0.0	1.8	4.3	2.0	0.0	0.0	1.3	1.2	0.9	1.1	1.3	0.0	0.4
F. tenera	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.3	0.4	0.0	0.0	0.0
F. capucina gracilis	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
F. capucina rumpens	0.3	0.0	0.7	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
F. capucina vaucherie	0.0	0.0	0.0	0.3	0.0	0.3	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gomphonema angustatum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. minutum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. truncatum	0.7	2.7	3.3	1.7	0.3	0.0	0.3	0.3	0.9	0.8	0.9	1.7	0.8	1.2	0.6	0.0	0.7	0.0	0.0
G. parvulum	0.3	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gyrosigma attentuatum	0.0	0.0	0.7	0.0	0.3	0.0	0.8	0.0	0.3	0.8	0.0	0.3	0.3	2.4	1.2	1.4	0.7	0.0	1.7
G. nodiferum	0.3	0.7	0.3	0.0	0.0	0.0	0.0	0.3	0.3	0.6	0.0	0.0	0.5	1.2	1.5	1.1	0.7	0.0	0.0
G. parkerii	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G. spencerii	0.0	0.0	0.0	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	17cm	18cm	19cm	20cm	21cm	22cm	23cm	24cm	25cm	26cm	27cm	28cm	29cm	30cm	31cm	32cm	33cm	34cm	35cm
Hantzschia amphioxys	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.0
Luticola mutica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.3	0.0	0.9	0.4	0.0	0.0	0.0
Melosira varians	0.0	0.0	0.0	0.6	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Navicula capitata	0.3	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.3	0.0	0.3	0.0	0.0
N. capitigidorada	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.6	0.3	0.6	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. cryptocephala	0.3	0.3	0.7	0.8	1.4	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. cuspidata	0.0	0.0	0.0	0.0	1.1	0.0	0.5	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0
N. elgensis	0.0	0.0	0.0	0.0	0.0	2.0	0.8	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. leptostriata	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. radiosa	0.0	0.0	0.0	0.0	0.3	0.7	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4
N. rhynocephala	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. phyllepta	0.0	0.0	0.0	0.3	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4
N. stiolata	0.0	0.0	0.0	0.3	0.3	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0
N. trivialis	0.0	0.0	0.0	0.0	1.7	1.7	0.0	0.0	0.0	0.3	0.0	0.3	3.7	4.5	0.0	0.7	1.3	0.0	0.0
N. veneta	0.0	0.0	0.0	0.0	1.1	0.0	0.3	0.0	0.0	0.8	1.3	0.0	0.0	0.3	0.0	0.0	0.3	1.6	1.7
N. viridula	0.0	0.0	0.7	0.3	0.6	0.3	0.8	1.5	0.3	1.1	1.6	0.3	0.3	0.0	1.5	0.7	3.4	3.1	1.7
N. vulpina	0.0	0.0	0.0	0.6	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.0
Nitzschia acicularoides	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. compressa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. constricta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. dissipata media	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. fonticola	0.0	0.0	0.0	0.3	0.6	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
N. frustulum	0.0	0.0	0.0	0.3	0.0	0.3	0.3	0.3	0.0	0.3	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.4
N. hungarica	0.3	0.0	0.7	1.1	0.0	0.3	0.5	0.0	0.9	1.7	0.0	0.0	0.5	0.3	0.0	0.0	0.3	0.0	0.0
N. lacuum	0.0	0.0	0.0	1.4	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. levendensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
N. linearis	0.0	0.0	0.0	0.0	0.3	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. lorenziana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
N. palae	0.0	0.0	0.0	0.3	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.4	0.3	0.0	0.0
N. palacea	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N. sigma	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.3	0.4	0.0	0.0	0.0
N. subacicularis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planothidium delicatulum	0.0	0.3	1.0	1.4	0.3	0.3	0.0	0.3	0.6	0.3	0.0	0.0	1.6	0.0	0.3	0.0	0.0	0.0	0.0

	17cm	18cm	19cm	20cm	21cm	22cm	23cm	24cm	25cm	26cm	27cm	28cm	29cm	30cm	31cm	32cm	33cm	34cm	35cm
Pleurosigma elongatum	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pseudostaurosira brevistriata	10.3	8.1	14.9	16.9	19.1	0.0	3.8	1.8	1.7	1.4	2.8	1.7	2.4	5.4	1.8	0.7	2.0	0.0	0.0
Rhopalodia gibba	2.4	2.4	0.7	1.4	0.6	1.0	0.8	0.0	0.9	0.8	0.0	0.3	2.7	3.3	3.6	3.5	3.0	5.7	4.7
R. musculus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stephanodiscus parvus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
Stauroneis obtusa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Staurosira construens var. venter	15.5	15.2	19.9	25.4	27.6	49.7	61.1	57.3	47.4	37.7	37.5	30.6	32.0	26.9	25.1	13.8	12.1	7.8	9.0
Staurosirella pinnata	8.2	10.4	3.3	10.2	4.1	9.6	8.1	7.9	3.4	5.0	6.9	1.7	1.1	2.1	0.9	1.8	1.0	0.0	0.4
Surirella angusta	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
S. robusta	0.0	0.3	0.0	0.0	0.0	0.3	0.3	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
Synedra acus angustissima	0.7	0.7	0.3	0.6	1.1	0.3	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
S. ulna	0.0	0.3	0.3	0.3	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.3
Tabularia fasciculata	1.0	4.0	2.3	1.7	2.5	7.6	3.8	1.5	1.7	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.3	0.0	0.0
Tryblionella levendensis	0.3	0.0	0.7	0.3	0.6	0.0	0.0	0.0	0.0	0.3	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Thalasiossira lacustris	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0