MODELLING THE WATER QUALITY IN DAMS WITHIN THE UMGENI WATER OPERATIONAL AREA WITH EMPHASIS ON ALGAL RELATIONS

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ABSTRACT

Based on many years of water quality (including algal) and water treatment cost data, available at Umgeni Water, a study was undertaken to better understand the water quality relationships in man made lakes within the company's operational area, and to investigate how water quality affected the cost of treating water from these lakes.

The broad aims to the study were to:

- identify the key environmental variables that were affecting algal populations in lakes; and if these were significant to
- establish predictive models relating algae to the water quality; and to
- develop models relating the water quality in lakes to the cost of treating water from the lakes.

Semi-quantitative models were developed relating algal abundances with important environmental variables. In most cases, the models developed were related to algal populations that were known to adversely affect water treatment. Direct algal impact on water treatment processes was through the production of either taste and odour forming compounds (requiring advanced water treatment, such as use of activated carbon), or their ability to clog sand filters and so reduce filter run times (requiring more frequent backwashing of filters).

Thereafter lake water quality parameters (which included water physico-chemistry and algae) were investigated to determine which factors were most significantly impacting on water treatment and hence treatment costs at selected water works (WW) within the Umgeni Water operational area. Models were developed relating raw water quality entering respective water works with costs incurred in treating that water. The models allowed simulations to be developed illustrating how changes in water quality might impact on water treatment costs. The impact of eutrophication and contamination of rivers and lakes, and its subsequent impact on surface water resources, was quantified.

Key words: algae, environment, contaminants, eutrophication, water treatment, water quality, lakes, dams, reservoirs, rivers, cost models, water treatment cost, modelling,

PREFACE AND ACKNOWLEDGEMENTS

Most of this investigation has been reported on in the Water Research Commission (WRC) report "Modelling the Water Quality in Impoundments within the Umgeni Water Operational Area and the Consequences for Potable Water Treatment Costs" (Graham *et al.*, 1998). This investigation emanates from a project funded by the Water Research Commission and conducted primarily at Umgeni Water in Pietermarizburg, kwaZulu-Natal, South Africa, during the period 1990 to 1997. As the principal author and researcher on the WRC project, I subsequently registered the investigation for a PhD study with the Potchefstroom University for Christian Higher Education.

The conception, design, and execution of this work are my own although the steering committee of this WRC project are duly acknowledged for their technical guidance and contributions made during execution of the project. The original report to the WRC has been substantially reorganised and rewritten with the incorporation of various pilot studies and investigations to ensure meeting requirements for a PhD thesis.

As part of the larger project conceptualised and motivated by the author, a pilot study and data analysis for the economic models on the DV Harris waterworks was initially undertaken in collaboration with Dianne Dennison. This work resulted in her MSc (Dennison, 1996). Under the further direction of the author the economic models were subsequently reanalysed and expanded to incorporate all the waterworks reported on in this thesis. The help with analyses of this phase by Dr S. Mbowa (Mbowa, 2003) is duly acknowledged. Unless otherwise acknowledged in the text, interpretation and reporting of this data is that of the author.

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CHAPTER 1: INTRODUCTION

1.1 Background and context of the study

1.1.1 Eutrophication and its impact in the South African and KwaZulu-Natal context

It has often been stated that one of South Africa's principal limiting natural resource is water (e.g. Huntley *et al.*, 1989). The recognition and result of this situation have been policies that have led, amongst others, to: the construction of many reservoirs on almost all available river sites throughout the country; various inter-basin transfer systems; the mandatory return of all effluents to river courses; and a permit system for discharges of effluents back to river courses in critical catchments (with associated standards and receiving water quality guidelines, Walmsley, 2000).

KwaZulu-Natal, and specifically the area served by the parastatal Umgeni Water Board, is no exception to these types of development with, for example, eight major water supply dams and many wastewater effluent discharges to rivers within its operational area.

Although principally designed to ensure the supply of freshwater for the country's developmental needs, the practices highlighted above have often had negative environmental impacts (Department of Water Affairs, 1986), the main one of which has been the increased nutrient enrichment (cultural/anthropogenic eutrophication) of surface waters. Internationally the process has been recognised since the 1920's (Bernhardt, 1992) and is now acknowledged to be a global water resource problem (e.g. Vollenweider, 1968; Australian State of the Environment Advisory Council (ASEC) 1996; United Nations Environment Programme (UNEP) 1999; European Environmental Agency (EEA) 1999).

Nutrient enrichment is most often found in highly populated and developed areas, where agricultural practices and water-borne sewage systems lead to elevated nutrient loads in receiving water systems. These elevated loads generally promote **"eutrophication"** which may be defined as ***"...the nutrient enrichment of waters***

which results in the stimulation of an array of symptomatic changes, amongst which increased production of algae and aquatic macrophytes, deterioration of water quality and other symptomatic changes are found to be undesirable and interfere with water uses" (OECD, 1982).

Excessive eutrophication may cause numerous problems, both in the long- and short-term, as well as from a water treatment, human health, ecological, agricultural, real estate value and aesthetic perspective. In the context of this thesis and from a water board's (e.g. Umgeni Water) point-of-view these include, amongst others:

- increased occurrence and intensity of nuisance algal blooms;
- an increased dominance by blue-green algae;
- increased occurrence of toxic algae;
- clogging of reticulation systems by filamentous benthic algae;
- increased occurrence of taste and odour problems in drinking water;
- increased water treatment costs through increased complexity of treatment processes, for example:
    - the need for activated carbon to eliminate taste and odour compounds;

as well as:

- filter clogging and reduced filter run-times in water treatment works;
- treated water wasted on more frequent backwashing;
- increased occurrence of anaerobic conditions in hypolimnia (bottom waters) of lakes with associated chemical effects (hydrogen sulphide and elevated levels of heavy metals and iron and manganese); and
- interference with recreational use of water bodies.

(Dunst *et al.*, 1974; Palmer, 1980; Lorch, 1987; Dickens *et al.*, 1996; Harding & Paxton, 2001).

The eutrophication problem is partly a human-perception issue and is related to how a community perceives the value of a specific water resource, and the effort and price that it is willing, or able, to pay to solve the problem (Rast and Thornton, 1996). From a water board's perspective, with its mandate to produce bulk potable water for supply to consumers at an acceptable level of quality (e.g. Hodgson *et al.*, 1996), there is clearly a vested interest in ensuring that the raw water which it has to treat is of the "best" quality possible, to minimise its treatment, and hence production costs. Raw water quality is influenced by the factors (and side effects) causing

eutrophication, as well as dissolved and suspended, inorganic and organic particles arising from the various land-use activities, soil and geological formations within catchments. It is widely acknowledged that increased catchment degradation generally leads to increased particle loadings to impoundments. These loads are required to be removed as part of the water purification process. One of the principal costs associated with treating water to an acceptable standard for human consumption, is in chemicals needed for the flocculation (coagulation) of suspended inorganic and organic particles in the water. Figure 1.1 clearly illustrates the contribution of the various classes of water treatment chemicals to the annual water treatment costs incurred by Umgeni Water (financial year ending June 2002).

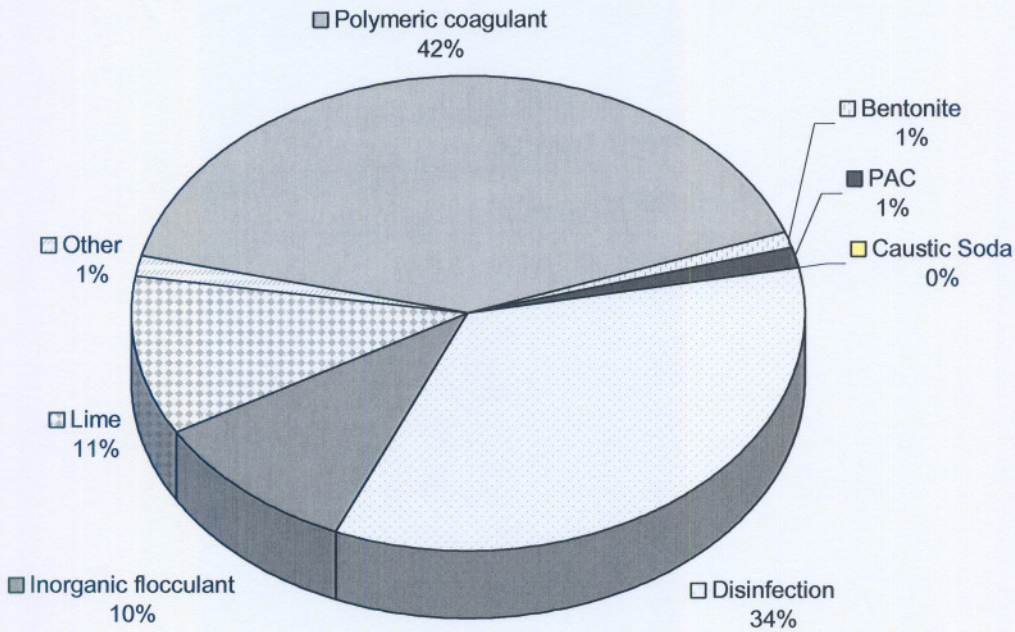

Figure 1.1 Summary of relative contributions of classes of water treatment chemicals to the total cost of treating water at Umgeni Water (for the financial year ending, June 2002). (PAC = Powdered Activated Carbon)

Umgeni Water almost exclusively uses synthetic organic polymers (e.g. polydiallyldimethylammonium chloride), or more colloquially, polymeric coagulants (or simply 'polymers'), for the removal of slow settling colloids in raw water entering its waterworks (WW). The process is one of coagulation and flocculation - with

coagulation neutralising the charge on particles in suspension (destabilisation) and the particles then being brought into intimate contact with one another (flocculation). Particles increase in size, through entrapment (enmeshment) in a precipitate or through interparticle bridging, until they settle out of the water column under gravity. The colloids are primarily inorganic (derived from soils in the catchment and carried into the lake by hydrological processes) and organic (phytoplankton, or algal cells, developed *in situ* within the reservoir). To a large degree most of the coarser inorganic particles sediment-out at the inflow to the lake, whilst it is the finer suspended inorganic fraction and phytoplankton that are generally abstracted in the lake main basins and that have the greatest effect on the water treatment process. These particles need to be removed to provide the consumer with clean water of acceptable quality.

There is not only the need for physical removal of algae but also at times removal and/or treatment of secondary metabolites produced by various algal taxa that may have a profound and, from a cost point-of-view, significant impact on potable water quality (e.g. Dickens *et al.,* 1996; Harding & Paxton, 2001). Blue-green algae (or cyanobacteria) are often found in abundance during eutrophication and have for a long time been recognised as a nuisance in the drinking water industry. This is due to the ability of certain taxa to produce earthy and musty smelling compounds, most notably geosmin, and 2-methylisoborneol (2-MIB), for which odour detection thresholds of less than 10 ng l^{-1} are possible (WHO, 1999). Their other significant contribution to potable water quality issues revolves around their ability to produce highly toxic algal toxins (e.g. microcystin-LR).

The public debate over the abundance of algae, and of the toxic cyanobacteria in particular, came to a head in the summer of 1989 with the deaths of dogs and sheep at Rutland Water, Leicestershire, UK, and then the acute poisoning of soldiers who had been swimming in Rudyard Lake, Staffordshire, UK. Prior to this, however, there had been reported cases of gastro-enteritis from cyanobacteria in the populations of small towns along the Ohio River (USA) in 1931 (Tisdale, 1931). Since then further, well-publicized events involving blooms of toxic cyanobacteria from various corners of the globe have been well-documented with probably the most lethal recorded outbreak occurring in Brazil. Eighty-eight deaths, mostly children, were attributed to cyanobacterial toxins in drinking water from a recently flooded dam that developed an immense cyanobacterial bloom (Teixera *et al.*, 1993, reported in WHO, "*Toxic Cyanobacteria in Water*", 1999).

Various South African incidents of non-human lethal toxicosis have been reported over the years (Harding & Paxton, 2001), with the massive loss of dairy cows in the Tsitsikamma-Kareedouw district in the south-western Cape one of the more dramatic (van Halderen *et al.*, 1995). As yet, there have been no reported human fatalities due to cyanobacterial toxin poisoning in South Africa.

Hence, the greater the intensity of eutrophication and contaminant loading to a water body, the more significant, technologically demanding and expensive, the problems associated with treating that water to acceptable potable water standards. This provides something of the broader South African, and eutrophication, context within which the research for this thesis was undertaken.

1.1.2 The specific impetus for this research at Umgeni Water

Raw water in South Africa is considered a strategic national resource with the responsibility for its management taking place at three tiers, firstly, Central Government, secondly Provincial & Regional Governments and other specified authorities, and thirdly Local Authorities. Umgeni Water plays an important role in both the second and third tier areas of responsibility, and forms a link between the Central Government and Local Government through their engagement in the provision of bulk water services (Umgeni Water web site, http://www.umgeni.co.za).

Umgeni Water is the largest water services provider in the kwaZulu-Natal region of South Africa, with an area of supply of some 24 000 square kilometres and annually producing just over 300,000,000 m^3 of treated, or potable, water (Umgeni Water, 2002). As part of the provision of potable water, there is an extensive water quality monitoring programme (certified to ISO 9002) covering a network of sampling points over 14 potable water works (WW), 3 wastewater works (WWW), 8 dams i.e. man-made lakes (or lakes as referred to in this thesis) and numerous river sites throughout the respective catchments. The monitoring programme, which has been running since 1988, collects approximately 22 750 samples per year with an ISO 17025, 1999, accredited laboratory conducting approximately 375 000 water quality analyses on these samples per year (Moodley & Hodgson 2002). This monitoring programme is used to regularly identify sources of pollution, report on trends, plan

and manage water quality at key operational sites, and to optimise the potable water treatment process to minimise treatment costs.

Likens (1988) has observed that long-term monitoring data are lacking for most aquatic systems and that monitoring programmes must continue for many years before any statistically defensible approaches can detect trends. At the time of undertaking this research, Umgeni Water had more than seven years of regular weekly limnological and algal data from all of the major lakes in Umgeni Water's operational area (Figure 1.2). At the time this probably represented one of the most complete and exhaustive long-term monitoring data sets available in the country for algae and their environment. The data covers eight lakes in Umgeni Water's operational area and represents a range of nutrient states - from typically mesotrophic, in the upper catchments, to eutrophic conditions in the lower reaches. There were also records of the costs incurred in treating these waters for potable consumption.

This, therefore, provided a large and robust data set with which to begin modelling the abundance of algal populations of these lakes, as well as to build economic models of the various algal and water quality loadings on respective water treatment works (WW). There was also the potential to identify those algae (and their abundance) likely to occur under different but specific water quality regimes. This was seen to be of particular economic and planning benefit, where problematic algae (taste and odour forming, filter clogging etc.) impact on the operation of water treatment works.

Previous work in this field of study (e.g. Varis *et al.,* 1989; Dixit *et al.,* 1992) has clearly demonstrated that algae are sensitive to many environmental characteristics, but that certain environmental variables have a larger and more significant impact than others. These variables tend to limit the distribution and abundance of the algae. Once algae/environment models have been established, these may be integrated with economic models to provide a useful and powerful tool for catchment managers responsible for the provision of potable water and maintenance of environmental quality.

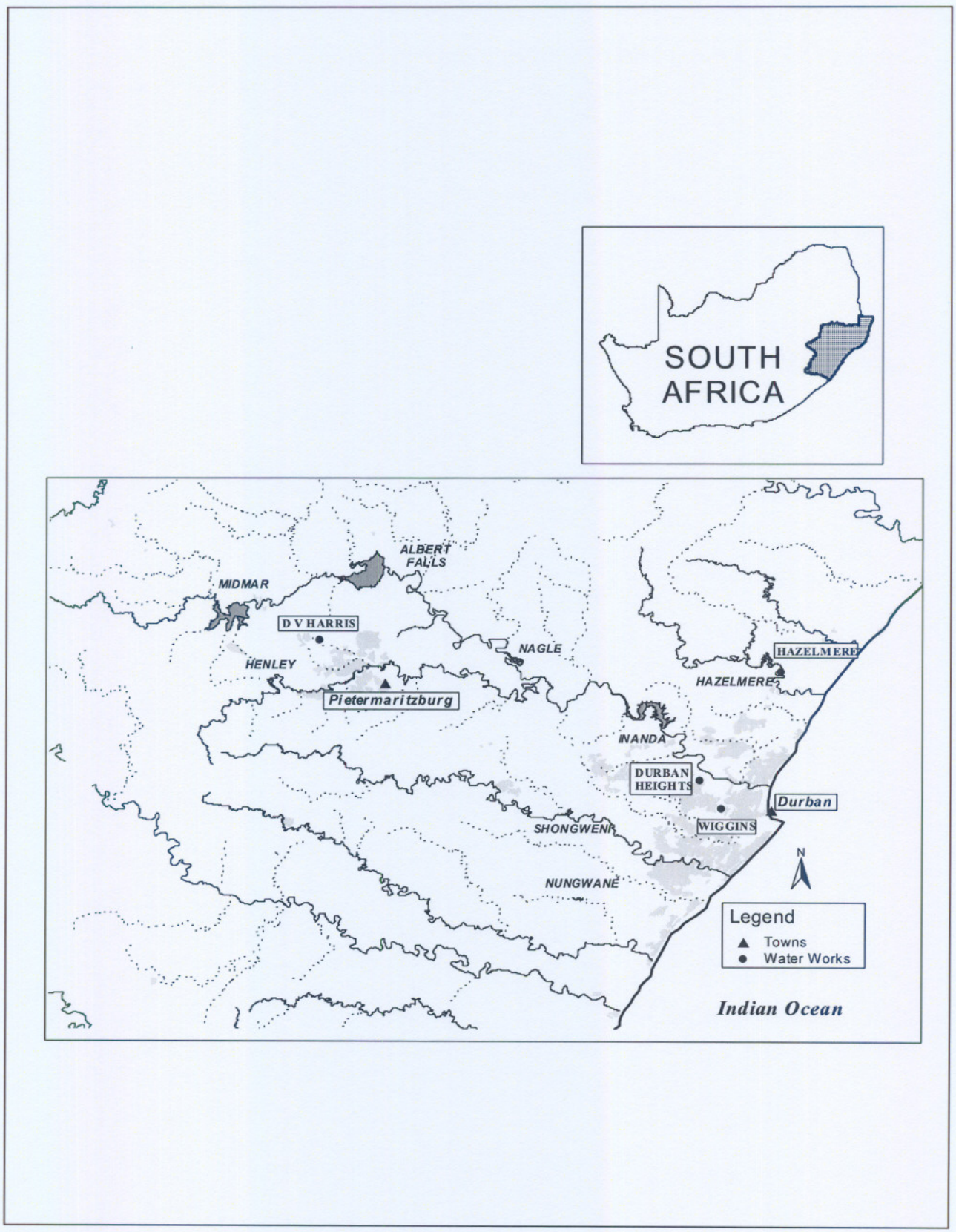

Figure 1.2 Geographical position of lakes and key water works studied in kwaZulu-Natal

Within a South African context, Grobler & Silberbauer (1984) note the dearth of well-defined relationships for water quality variables that can be quantitatively related to water quality problems associated with eutrophication. This is a major limitation in predicting the impact of eutrophication control measures on water quality in South Africa. These authors go further to state that '*research to establish such relationships should receive a high priority.*' More recently Walmsley (2000), in a thorough review of the policy and research needs surrounding eutrophication in surface waters in South Africa, emphasized a number of areas that require immediate national research, namely:

- quantitatively assessing the eutrophication problem in terms of its extent and trends;
- the sources of nutrients and levels entering aquatic systems; and
- the actual social and economic costs of the problem on a national basis.

Walmsley (op. cit.) highlighted that many of the current eutrophication research needs are essentially little different from those published over a quarter of a century ago in a seminal report on inland water ecosystems of South Africa (Noble and Hemens, 1978). With respect to some of the issues identified by these authors, and which this thesis concentrates on at various levels of detail, are the following:

- social and economic impact of eutrophication and its side effects;
- development of systems models for eutrophication management;
- methods for measuring and monitoring nutrient loads in rivers;
- integrated estimations of N and P compound loadings from all point and non-point sources in catchments;
- factors causing the appearance of nuisance plant growths (algae) in aquatic systems;
- algae that produce taste and odours as well as potential toxins.

This thesis is a beginning towards addressing several of the research needs highlighted by the authors Grobler and Silberbauer (1984), and Walmsley (2000), if only at a regional level. Chapter 3 addresses the estimation of catchment nutrient loadings to lakes, whilst Chapters 4 and 5 investigate the factors causing nuisance algal growth in the lakes studied. The water treatment cost models, presented in Chapter 6, may be used practically by operators of water works to improve/optimise the water treatment process. The cost models also quantify for catchment managers, agencies and planners those water quality processes that are likely to

have an impact on the cost of treating water. Various 'what-if' and prediction type scenarios are possible with these derived models and are presented in the thesis.

Post research note: The specific Hazelmere model was explored and applied in practise after the completion of the research for this thesis. Umgeni Water was able to explore the impacts on treatment costs of an application for a proposed sand-winning operation in the uMdloti river (above the inflow to Lake Hazelmere). The sand-winning operation would obviously have affected in-lake water quality (specifically turbidity) and hence treatment costs.

The specific aims of this investigation were therefore to:

- Establish the key environmental variables influencing the distribution and abundance of problematic algae in lakes in the Umgeni Water operational area, (Chapters 4 & 5);
- Establish a predictive model(s) relating algae to the environment for the major lakes in the Umgeni Water operational area, (Chapter 5); and
- Build economic models relating treatment costs to the types and numbers of algae likely to be found in lakes. (Chapter 6).

Models can mean many things to different people, but for the sake of this thesis the following understanding was applied: *"a simplified description of a complex entity or process"* (WordNet ® 1.6, © 1997 Princeton University). The models developed were essentially empirical in nature and relied on appropriate standard multivariate and univariate statistical analyses and investigations. Patterns and correlations identified by these analyses were used to build simple models within a spreadsheet environment. These models went some way to describing the complex water quality and water treatment cost processes within the lake systems and water treatment works studied.

Unfortunately algae were only identified to genus level in this study. This was dictated by the needs of the production type laboratory environment at Umgeni Water, with a high throughput of samples. There is no doubt that had it been possible to use algal data identified to species level, there would have been a much higher correspondence and correlation with environmental variables. This is because a genus is likely to show a wider tolerance to environmental conditions than that pertaining to specific species within that genus. Where algae are identified to

species level, more robust models may be possible and are likely to be more useful. For example, it is likely that only certain species of a typically problematic algal genus (e.g. *Anabaena*) are responsible for taste and odour formation, and hence likely to affect water treatment processes. The non taste and odour forming species within the genus are therefore adding to the "noise" in the data and clouding these types of important relationships.

CHAPTER 2: LAKES STUDIED

2.1 Introduction

For the various lakes studied in this thesis this chapter provides:

- A broad background to the lakes and includes detail on their catchments and physical characteristics.
- A summary of their physico-chemistry as well as algal composition with the primary aim of describing the types of lake systems considered.

Detail on methods of manipulation and handling of the data prior to statistical analyses is presented in the following chapter (Chapter 3).

2.2 Study area

The extent of the study area, as well as the location of the eight lakes and various water works, has been shown in Figure 1.1.

Except for Lakes Inanda and Nungwane, detailed descriptions for all lakes may be found in Walmsley and Butty (1980) and for Midmar in Breen (1983). The Mgeni Catchment Management Plan (MCMP) document (1996) provides supplementary details. A general description of the catchments and their lakes, as they relate to this study, is presented below with more detailed information presented in Table 2.1. Summary statistics for the general physico-chemical condition of respective lakes over the study period are presented in Table 2.2. Further summary statistics are graphically portrayed where appropriate in the body of the thesis.

Table 2.1 Summary catchment and physical characteristics of the lakes studied

Parameters	Henley	Midmar	Albert Falls	Nagle	Inanda	Hazelmere	Nungwane	Shongweni
River system	uMsunduze	uMngeni	uMngeni	uMngeni	uMngeni	uMdloti	Nungwane	Mlazi
Year completed	1943	1963	1975	1963	1989	1975	1978	1927
Altitude (m)	918	1021	630	300	115	61	346	300
Latitude (deg,min,sec)	29 37 23	29 29 42	29 25 50	29 35 31	29 42 29	29 36 00	30 00 26	29 51 39
Longitude (deg,min,sec)	30 14 15	30 12 16	30 25 39	30 37 41	30 52 04	31 02 37	30 44 43	30 43 17
Maximum depth (m)	10	22	26	34	44	24	24	14
Full supply level (FSL) (m)	24.1 *	22.5 *	21.6	24.1	31.2	24.9	16.7	22.5
Max volume (Mm^3)	3.74	177.76	290.72	24.61	251.75	17.81	2.24	7.16
Total catchment (km^2)	219	928	1644	2535	4082	377	58	786
Incremental catchment (km^2)	220	927	721	888	1322	375	59	786
Max lake area (km^2)	0.321	15.64	23.87	1.56	14.63	1.81	0.29	0.81
FSL volume/area ratio	11.64	11.37	12.18	15.77	17.21	9.84	7.72	8.84

* Notes: The dam wall at Henley was raised in 1959 and then lowered in 1991 to 24.1m from its previous height of 31.1m.

The dam wall at Midmar was raised in 2003 to 26m, with an increase in storage capacity to 253 Mm^3. However the pre 2003 'vital statistics' are reflected in Table 2.1 as they were valid for the duration of this study.

2.2.1 General catchment characteristics

The uMngeni River system (which has on it five of the major lakes (or supply reservoirs) studied here) is of particular strategic significance in the province of kwaZulu-Natal (KZN) being one of the most developed catchments in South Africa. The reservoirs on this system have a combined capacity of 745.9 million m^3 and supply the major Pietermaritzburg-Durban complex, home to some 45% of the population of the province. The uMngeni catchment produces approximately 20% of South Africa's gross national product, and 65% of the total economic production in KZN (MCMP, 1996). The other major reservoirs studied (Nungwane and Hazelmere) are important sources of water for the KZN south and north coast regions respectively. Shongweni, on the Mlazi River system, although included in this study, is no longer significant from a water resource point-of-view due its decommissioning in 1993.

The uMngeni River rises in the uMngeni Vlei in the highland plateau area of the KZN Midlands, about 1900 m above sea level. From a reed and grass wetland area water seeps into an incised grass-lined rocky channel whose gradient steepens rapidly as it falls off the plateau. Below the plateau the gradient flattens through the rolling

foothills of the Midlands and plains around Lake Midmar. Midmar is the highest major lake on this system (storage of 177 million m^3, although just recently, 2003, its capacity has been increased to 253 million m^3 with the raising of the dam wall). Other major tributaries of the uMngeni rising in this area are the Karkloof, Lions and uMsunduze Rivers. The catchment and channel characteristics of these rivers are all fairly similar and although not all originating on large wetlands, have a number of wetlands associated with them. The uMsunduze River has Lake Henley (1.5 million m^3) sited in its upper reaches. The soils of these upper catchment areas are generally deep, permeable, well drained and fertile making them agriculturally productive (Scotney, 1970; MCMP, 1996). The relatively low soil erodibility, gentle slopes and generally good vegetation cover traditionally makes for low erosion in this area. More recently, however, the rapid urbanization and influx of people into the uMsunduze valley has reduced vegetation cover and increased soil erosion and soil loss from the catchment.

Downstream of Midmar the uMngeni (and some of its important tributaries) crosses an extensive dolerite dyke marking the edge of a small escarpment. This results in several waterfalls (e.g. Howick and Karkloof Falls) and steep gorges along the rivers downstream of the escarpment. Once over the escarpment, the gradient again flattens as the uMngeni enters Albert Falls (289 million m^3), the largest capacity reservoir on this system. Running parallel to the uMngeni in the next catchment, the uMsunduze River and its tributaries pass through the city and environs of Pietermaritzburg. A combination of the city and the highly modified nature of the uMsunduze have contributed significantly to the degraded ecological health and reduced ability of this river to assimilate waste (e.g. Dickens & Graham 1998).

As these two major rivers enter the Valley of a Thousand Hills the gradients again steepen with the rivers becoming wider and more braided. Progress down the uMngeni catchment is marked by soils becoming shallower with lower permeability and fertility. Lower fertility, combined with reduced vegetation cover, results in increased erosion as the uMngeni flows into the Lake Nagle (23 million m^3), just above the confluence with the uMsunduze. The Mqeku is the only other major tributary of the uMngeni before the system flows into Lake Inanda (242 million m^3), the lowermost lake on the uMngeni system.

The basic characteristics of respective lakes and their catchments are given in Table 2.1 with a statistical summary of the water physico-chemistry for the study period

given in Table 2.2. Further specific information is reflected in analyses and discussions throughout the thesis. Proportional algal genera composition (for respective lakes, for the duration of the study period) is displayed in Figure 2.1, whilst Figures 2.2 to 2.4 reflect summary abundance statistics for some of the key algal genera from respective lakes. Shongweni algal abundances were not reflected on these figures due its high algal counts and the 'swamping' effect this would have had on the data from other lakes.

2.2.1.1 Midmar

Lake Midmar is situated in a broad valley in the upper uMngeni catchment. The uMngeni River here (the principal river feeding this lake) has an estimated mean annual runoff of 158.5 million m^3 of generally good quality water with low suspended solids, low total dissolved solids and low nutrient concentrations (Walmsley & Butty, 1980). Generally the soils in the catchment have a high phosphate (P) binding capacity (Scotney, 1976) which makes P concentrations and loads in the streams low (Furness, 1974; Walmsley & Butty, 1980). Van der Zel (1975) showed that more than 50% of the catchment remains undeveloped grassland with most of the rest comprising forestry, agriculture and stock raising. Scotney (1970) has identified the area as having a high rating for intensive agriculture. Nutrient concentrations entering the lake from other tributaries are generally low except for the Mthinzima Stream that drains the Mpophomeni township.

Walmsley and Butty (1980) describe the lake as a warm monomictic system with Hemens *et al.* (1975) indicating it as oligo-mesotrophic. Walmsley (1977) confirmed this general observation and later noted that although not turbid by South African standards, the lake is unusual in increasing in turbidity in winter. Walmsley & Butty (1980) ascribe the increased turbidity to a combination of fine silt suspension from loose marginal and bottom sediments and intense water circulation of the wind exposed system. The shallow, shelving shoreline and littoral zone is also highly conducive to the re-suspension of sediments. For the duration of the study the mean monthly turbidity over all seasons was only 10.5 NTU. Nutrient concentrations were also low for the corresponding period, with mean monthly total inorganic nitrogen (TIN) and soluble reactive phosphorus (SRP) 0.3 and 0.004 mg/l respectively. The conductivity of the water was also low (mean 6.8 mS/m) reflecting low dissolved salt concentrations in the lake water.

For the duration of the study Midmar, had the lowest algal counts (median total algal count 1 500 cells/ml) of all lakes studied, of which *Chlorella* (monthly median count of 1 000 cells/ml) was frequently occurring and a significant contributor to the total (43% of the total median, Figure 2.1). *Crucigena* (11%) and *Melosira* (11%) were the next most frequent contributing to the total algal count in Midmar. Blue-green algae were less common and this fact probably reflects the "good" water quality in this upper catchment lake.

2.2.1.2 Albert Falls

This, the largest lake on the uMngeni system, is situated downstream of the town of Howick, between Midmar and Nagle. Its primary function is for potable water supply, acting as a storage lake releasing water to the uMngeni River when required for the smaller Lake Nagle further down on the uMngeni River cascade.

The basic characteristics of the lake and its catchment are described in Table 2.1 with the water physico-chemistry for the study period described in Table 2.2. Walmsley and Butty (1980) have classified this clear water, oligotrophic, phosphate-limited lake as a warm monomictic system displaying a normal pattern of summer stratification, autumn turnover and isothermal winter conditions. Unlike Midmar, turbidity in the Albert Falls system does not appear to increase in winter and averaged a low 14.7 NTU for the duration of the investigation. Plant nutrients are also low, with mean monthly total inorganic nitrogen (TIN) and soluble reactive phosphorus (SRP) 0.4 and 0.004 mg/l respectively. The conductivity of the water was also low (mean 7.8 mS/m).

Monthly median algal counts in Albert Falls were 3 800 cells/ml with total counts dominated by *Chlorella* (34%) and *Microcystis* (32%) (Figure 2.1). On occasion counts of the problematic blue-green algae, *Anabaena* and *Microcystis*, can be high (maximums of 29 000 and 14 000 cells/ml respectively, have been recorded).

2.2.1.3 Nagle

Lake Nagle is midway down the uMngeni system, between Albert Falls and Inanda. It was the first reservoir to be built on the uMngeni (1943) to supplement supplies to meet the growing demands of Durban. It has a unique diversion weir and control gates at the head of the lake that allows floodwaters (with high suspended solid and nutrient loads) to be diverted around the lake and thus optimising the in-lake water quality. This diversion system, in combination with Albert Falls and Midmar upstream

of it on the uMngeni, has resulted in Nagle having a unique, and largely unnatural, hydrological regime. It is now essentially a holding reservoir being maintained at almost full supply level by compensation water discharge releases from Albert Falls. From here water is abstracted to a major water treatment works (Durban Heights Water Works) in the Durban metropolitan area.

Walmsley & Butty (1980) describe this reservoir as being a warm monomictic system with summer stratification, autumn turnover and a winter period of isothermal conditions. They go further to describe it as an oligotrophic, phosphate-limited system. This assertion is confirmed in this study with mean monthly total inorganic nitrogen (TIN) and soluble reactive phosphorus (SRP) concentrations low at 0.3 and 0.004 mg/l respectively. Irrespective of these low average nutrient conditions, severe algal related problems have been experienced in Nagle intermittently since 1987 (see Dickens *et al.*, 1996). This thesis illustrates some of the impacts of these algal related problems on water treatment (Chapter 6).

Conductivity of Nagle water is low (mean 8.9 mS/m) as is turbidity (mean 13.7 NTU).

Monthly median algal counts in Nagle were 6 100 cells/ml, with total counts dominated by *Microcystis* (70%) (Figure 2.1). On occasion this alga reached maximum average monthly counts of 31 000 cells/ml with another of the problematic blue-green algae, *Anabaena* similarly high (maximum of 32 000 cells/ml).

2.2.1.4 Inanda

Inanda is the lowermost lake on the uMngeni River and was relatively recently completed in 1989. It boasts a large catchment area (Table 1.1) drained by the uMngeni and its major tributary the uMsunduze river. Activities in this catchment include extensive and intensive agriculture, urban development and informal settlements. The major point sources of nutrient loadings into the lake are from Darvill Wastewater Works in Pietermaritzburg and Cato Ridge Abattoir. Because of the high initial loads of phosphorus (80 tonnes per year), and the retention of up to 80% of this load, Inanda has been classified as eutrophic (Umgeni Water, 1991). It certainly had the highest average SRP loading of all lakes studied in this investigation (1131 kg/ha/yr), even exceeding that of Shongweni (231 kg/ha/yr), acknowledged to be a highly eutrophic system. The relatively large size of this lake, combined with its sinuous nature and prevailing wind direction (easterlies), probably ensures that most of this phosphorus loading remains at the head of the lake where it

is either utilized by algae or locked into sediments. Notwithstanding this though the lake does have relatively high concentrations of algae throughout the year.

For the duration of the study the mean monthly turbidity over all seasons was the lowest of all lakes studied, at only 6.1 NTU. Again the sinuous nature of this lake, with turbidity largely settling out at the inflow to the lake, probably accounts for the low turbidity. Turbidity was measured in the lake main basin, furthest from the inflow to the lake. Nutrient concentrations were also low for the corresponding period, with mean monthly total inorganic nitrogen (TIN) and soluble reactive phosphorus (SRP) 0.4 and 0.004 mg/l respectively. The conductivity of the water was also moderate (mean 21.7 mS/m) and reflecting the cumulative contributions of the various catchment processes, and hence dissolved salts, down the uMngeni cascade.

In terms of temperature and oxygen profiles, this lake may be described as a warm monomictic system with summer stratification, an autumn turnover and uniform conditions during winter.

Monthly median total algal counts in Inanda tended to be lower (7 500 cells/ml) than Nagle but were similarly dominated by the blue-green alga *Microcystis* (comprising 70% of the monthly total median algal count). *Microcystis* reached maximum average monthly counts of 217 000 cells/ml with *Anabaena* also high on occasion (maximum of 93 000 cells/ml). As will be shown later in this study it is these high counts of blue green algae that may cause significant and costly water treatment problems.

2.2.1.5 Shongweni

Shongweni is the oldest man made lake in kwaZulu-Natal (completed in 1927) but which is now essentially decommissioned as far as water supply is concerned. The decommissioning was primarily due to low yields and to the high allocthonous nutrient loads (and excessive algal growth) on this system from an industrial complex and domestic sewage works in the catchments draining into this lake. The combination of these factors made the water from this lake expensive to treat (Wilson, 1992).

The lake is situated in a valley just below the original confluence of the Mlazi, Sterkspruit and WekeWeke rivers in an area dominated by quartzite and shales of the Table Mountain series of the Cape system. There are small outcrops of Dwyka

tillite and shales of the Ecca series of the Karoo system in the upper catchment (Walmsley & Butty, 1980).

Shongweni was included in this study as it was thought useful to represent the extreme end of the enrichment axis based on the extreme loadings to the lake from various catchment sources. Table 2.2 indicates the lake had the highest chlorophyll 'a' concentration (monthly mean 31µg/l) of any lake considered in this study. This agrees with Walmsley and Butty's (1980) observation about its enriched state.

Walmsley and Butty (1980) note that the Mlazi (which drains 81% of the catchment) is frequently rich in nutrients because the catchment is dominated by agricultural activities as well as having a domestic sewage works discharging treated sewage directly into this river. The Sterkspruit (which only drains some 15% of the catchment) is also rich in nutrients (particularly nitrogen) having an industrial WWW discharging into this river. Soluble reactive phosphorus (mean 0.02, min 0.003 and max 0.09 mg/l) and total inorganic nitrogen concentrations (mean 0.8, min 0.06 and max 4 mg/l) were amongst the highest for all lakes studied. Very high conductivities (and hence mineralisation) in this lake (mean 80, min 29 and max 162 mS/m, Table 2.2) are primarily associated with inputs from the Sterkspruit. The mineralised state of Shongweni is highlighted in the present study.

Median monthly average algal counts in Shongweni were the highest of all dams studied (53 000 cells/ml), with a maximum recorded of over 2.3 million cells/ml. The blue-green algae (predominantly *Microcystis* and *Anabaena*) were significant contributors in this regard, at times up to 2.3 million and 853 000 cells/ml respectively. Green algae were minor contributors to the total algal composition of Shongweni.

2.2.1.6 Hazelmere

This coastal lake, completed in 1977, and situated on the Mdloti River, its only major input, drains an essentially undeveloped catchment with sugar cane the dominant agricultural interest. It is one of the younger lakes in this study.

Walmsley and Butty (1980) identified high silt loadings to the system as well as occasionally high nutrient loadings. The former they attribute to poor soil conservation in the catchment making for generally turbid conditions for much of the year. Of all the lakes studied this had the highest recorded average monthly silica

concentrations (9.2mg/l). It is certainly one of the most turbid systems in kwaZulu-Natal (mean monthly turbidity 60 NTU, max 235 NTU). This study highlighted high turbidity as a major consideration affecting water treatment in water abstracted from Hazelmere.

Walmsley and Butty (*op. cit.*) classified the lake as a warm monomictic system with phosphate likely to be the limiting nutrient. Mean nutrient concentrations during the study were total inorganic nitrogen (TIN) and soluble reactive phosphorus (SRP) 0.7 and 0.006 mg/l respectively. The conductivity of the water was moderately low (mean 17.8 mS/m).

Hazelmere had a median monthly total algal count of 3 600 cells/ml, although on occasion this could be higher (maximum recorded 42 400 cells/ml). The blue-green alga, *Microcystis*, was the dominant genera in this lake, comprising 70% of the total median count.

2.2.1.7 Henley

Henley is a relatively small lake (3.7 million m^3) situated in the upper catchment of the uMsunduze River. It was originally designed and built (1943, wall raised 1959) for potable water supply to Pietermaritzburg. The supply had to be supplemented with water from Midmar in more recent years (1964), although it is currently unused for water supply purposes with the wall lowered to its current height of 24.1m in 1991 for safety reasons. The mean annual runoff from the uMsunduze (some 80% of the total inflow to the lake (Walmsley & Butty, 1980)) is seven times the capacity of the lake and therefore greatly influences the quality of the water in the system.

The lake is noted as a warm monomictic system with a long period of summer stratification, a short overturn and a period of isothermal conditions during winter before re-stratification in summer (Walmsley and Butty (*op. cit.*). These authors classify it as oligo-mesotrophic although cautioning that the trophic status may change in dry years when inflows are reduced. The change is primarily because the system is so dependent on its flushing rate (a function of the inflow and small capacity). During the study the mean monthly total inorganic nitrogen (TIN) and soluble reactive phosphorus (SRP) concentrations were 0.8 and 0.004 mg/l respectively. The conductivity of the water was also low (mean 8.3 mS/m) reflecting low dissolved salt concentrations in the lake water. Turbidity was also low at an average of 16 NTU.

As with Midmar, Henly had low algal counts (median total algal count 1 700 cells/ml) for the duration of the study, of which *Chlorella* (monthly median count of 500 cells/ml) was the most frequently occurring and significant contributor to the total (54% of the total median, Figure 2.1). *Crucigena* (18%) was the next most frequent. Blue-green algae were uncommon in Henley and reflects the "good" water quality here.

2.2.1.8 Nungwane

Nungwane is the smallest of the lakes (Table 1.1) studied here. It supplies water to the coastal areas just south of Durban after the water has been treated at Amanzimtoti Waterworks. The lake is relatively high up in its catchment on the Nungwane River. The river has a few minor tributaries before draining into the Lovu River just below the lake. Landuse in the catchment area is mainly (86 %) extensively cultivated land with the remainder natural trees and bush (Umgeni Water, 1994).

Unfortunately there were no gauging weirs to estimate the nutrient loads into the lake during the study period and therefore all the data available on loads was estimated using a model. The model indicates that phosphorus loads into the lake are generally low (maximum of 2 tonnes per year). Monthly average total inorganic nitrogen (TIN) and soluble reactive phosphorus (SRP) concentrations are moderately low, 0.7 and 0.005 mg/l respectively. The conductivity of the water was also moderately low (mean 13.4 mS/m) reflecting low dissolved salt concentrations in the lake water.

The temperature and oxygen profiles indicate that the lake has a warm monomictic system with summer stratification, late autumn turnover and is isothermal during winter.

On the whole Nungwane had low algal counts for the duration of the study (median total algal count of 2 000 cells/ml), with *Chlorella* (monthly median count of 600 cells/ml) the most frequently occurring (43% of the total median). *Crucigena* (19%) and *Cryptomonas* (13%) were the next most frequent contributing to the total algal count here. Blue-green algae were uncommon.

Table 2.2 Summary descriptive statistics of physico-chemistry of respective lakes

Physico-chemical		Albert Falls monthly data summary					Hazelmere monthly data summary					Henley monthly data summary					Inanda monthly data summary				
variable	Units	N	Min	Max	Mean	CV	N	Min	Max	Mean	CV	N	Min	Max	Mean	CV	N	Min	Max	Mean	CV
% DO	%	60	66.6	120.8	97.1	9.9	54	50.8	113.4	88.9	15.2	59	65.9	121.3	94.6	11.9	59	41.0	118.7	92.6	19.2
Alkalinity	mg/l $CaCO_3$	60	21.3	31.1	25.6	10.2	54	24.4	59.7	41.5	23.3	59	12.6	44.2	27.3	22.7	59	24.8	66.1	53.6	15.2
Ammonia	mgN/l	60	0.0	0.1	0.0	58.4	54	0.0	0.2	0.1	73.8	59	0.0	0.8	0.2	90.0	59	0.0	0.2	0.0	85.3
Biological Oxygen Demand	mg/l O_2	60	0.4	9.8	1.9	97.8	54	0.1	7.1	1.3	87.1	59	0.5	4.3	1.3	51.5	59	0.5	2.2	0.9	42.3
Cadmium	ug/l	60	0.5	0.5	0.5	0.0	54	0.5	0.5	0.5	0.0	59	0.5	0.5	0.5	0.0	59	0.5	0.5	0.5	0.0
Calcium	mg/l	60	4.0	5.7	4.7	10.0	54	2.1	8.3	5.9	21.9	59	3.0	7.7	4.7	18.9	59	6.2	16.5	11.0	19.9
Chloride	mg/l	60	4.8	7.9	5.8	12.9	54	19.4	38.9	25.8	15.9	59	5.3	8.1	6.4	10.2	59	14.6	38.0	27.6	22.3
Chlorophyll a	ug/l	60	0.6	10.5	3.8	58.7	54	0.4	42.3	4.6	162.7	59	0.6	25.7	4.7	92.6	59	0.8	23.8	5.8	87.7
Chrome	ug/l	60	2.2	11.4	5.4	47.0	54	3.2	29.1	10.8	47.7	59	0.6	23.6	4.8	81.5	59	0.0	16.5	3.9	77.3
Conductivity	mS/m	60	6.5	9.9	7.8	9.7	54	13.3	24.9	17.8	14.8	59	6.0	12.2	8.3	15.0	59	12.5	29.0	21.7	18.8
Copper	mg/l	60	0.0	0.3	0.0	162.0	54	0.0	0.0	0.0	42.3	59	0.0	0.0	0.0	47.4	59	0.0	0.0	0.0	53.9
Dissolved Oxygen	mg/l O_2	60	5.7	10.0	8.1	12.0	54	4.4	9.7	7.6	15.8	59	5.5	10.4	7.8	13.2	59	3.6	10.8	8.0	18.4
Flouride	ug/l	60	27.9	69.4	41.4	18.3	54	37.5	158.0	59.6	49.2	59	20.7	38.8	35.5	11.1	59	37.5	175.0	62.0	62.1
Hardness	mg/l $CaCO_3$	60	18.3	28.9	24.3	10.0	54	15.4	50.5	37.8	19.2	59	16.5	37.3	24.8	16.6	59	32.1	78.1	53.6	16.6
Inflow	Ml/day	60	65	5185	568	150	54	20	691	150	101	58	7	526	74	121	59	136	8368	778	164
Iron	mg/l	60	0.1	2.5	0.6	73.9	54	0.8	5.5	2.1	55.2	59	0.1	1.8	0.4	77.5	59	0.0	0.7	0.2	78.6
Lead	ug/l	60	0.5	2.3	0.7	42.7	54	0.5	7.2	1.7	66.4	59	0.5	10.0	1.7	102.1	59	0.5	2.5	0.7	63.2
Magnesium	mg/l	60	2.4	3.6	3.0	10.2	54	4.2	7.6	5.6	15.2	59	2.1	4.3	3.1	15.7	59	4.0	8.9	6.2	14.1
Manganese	mg/l	60	0.0	0.1	0.0	52.8	54	0.0	0.2	0.1	74.6	59	0.0	0.3	0.0	100.0	59	0.0	0.2	0.0	87.7
Mercury	ug/l	60	0.3	1.9	0.3	75.0	54	0.3	1.3	0.4	51.7	59	0.3	3.5	0.6	95.2	59	0.0	1.0	0.1	236.9
Nickel	ug/l	60	0.3	10.7	2.4	89.6	54	0.3	10.1	3.6	67.1	59	0.3	8.0	1.9	89.7	59	0.0	18.6	1.7	158.5
Nitrate	mgN/l	60	0.0	0.9	0.3	47.6	54	0.1	1.8	0.6	56.8	59	0.1	1.4	0.6	50.9	59	0.0	0.7	0.3	61.1
Nitrite	mgN/l	60	0.0	0.0	0.0	0.0	54	0.0	0.0	0.0	0.0	59	0.0	0.1	0.0	25.3	59	0.0	0.0	0.0	4.3
Partial retention		60	2	31	13	43	54	1	15	6	52	11	1	6	3	47	59	1	68	20	62
pH		60	7.2	8.4	7.7	3.4	54	7.2	8.1	7.7	2.7	59	6.4	8.1	7.5	3.7	59	6.6	8.5	7.9	4.0
Potassium	mg/l	60	0.8	1.9	1.4	18.3	54	0.9	3.1	2.1	21.9	59	0.5	2.5	1.2	30.0	59	1.4	3.9	2.6	23.8
Secchi	m	60	0.2	2.2	1.3	38.9	54	0.1	1.1	0.4	60.8	59	0.1	4.2	1.7	52.4	59	0.7	2.7	1.7	28.1
Selenium	ug/l	60	0.3	2.1	0.5	64.9	54	0.3	1.2	0.3	43.6	59	0.3	0.6	0.3	19.9	59	0.0	1.1	0.1	236.3
Silicon	mg/l	60	2.4	5.8	4.0	19.7	54	5.9	13.2	9.2	13.4	59	3.5	8.7	5.7	17.1	59	0.5	6.7	2.8	56.3
Sodium	mg/l	60	4.1	7.3	5.3	12.3	54	15.4	26.8	20.3	14.1	59	4.2	8.5	6.2	17.2	59	12.3	31.0	22.5	23.5
Soluble Reactive Phosphorus	ugP/l	60	1.5	10.9	3.5	66.1	54	1.5	19.0	5.9	56.1	59	1.5	11.5	3.9	67.5	59	1.5	16.2	3.5	73.2
SRP load	kg/ha/yr	60	15	934	112	134	54	4	154	33	108	59	1	112	16	119	59	186	6352	1131	104
SRP:Si		60	0.0	0.0	0.0	66.9	54	0.0	0.0	0.0	54.0	59	0.0	0.0	0.0	71.3	59	0.0	0.0	0.0	86.1
Stability	s^{-2}	60	-0.3	9.5	3.1	80.2	54	-0.5	9.2	4.1	72.5	59	0.1	18.9	4.9	92.6	59	-0.2	5.5	2.0	80.4
Sulphate	mg/l as SO_4	60	2.7	4.8	3.5	13.3	54	3.7	12.9	5.5	39.9	59	2.1	4.4	2.8	16.8	59	6.6	20.4	12.3	28.3
Sunhours	hours	60	0.6	10.6	7.3	30.1	54	0.5	11.4	6.4	27.4	59	3.6	11.1	7.9	22.6	59	0.9	10.8	6.9	31.1
Suspended Solids	mg/l	60	2.0	48.2	10.6	91.1	54	2.8	134.0	25.9	96.3	59	2.0	116.7	11.8	145.2	59	2.0	43.1	6.3	101.5
Temperature	degrees	60	13.7	27.1	20.6	18.9	54	15.6	26.8	22.5	15.6	59	11.1	25.9	19.1	21.7	59	15.5	27.9	22.0	15.8
TN load	kg/ha/yr	60	630	67490	7332	154583	54	210	17640	3664	117312	59	860	111860	15266	126379	59	1570	195480	23869	137621
TN:Si		60	0.0	0.2	0.1	33.9	54	0.0	0.2	0.1	43.7	59	0.1	0.4	0.1	37.8	59	0.0	0.9	0.2	90.2
TN:SRP		60	17.0	663.3	141.3	77.2	54	34.4	605.3	204.1	57.8	59	77.4	907.8	354.4	52.6	59	17.4	398.9	168.8	58.2
TN:TP		60	4.1	84.2	24.3	79.4	54	8.4	239.1	37.7	90.3	59	10.0	164.2	55.2	73.3	59	1.8	105.1	35.8	68.1
Total Aluminum	ug/l	60	21.0	1008.3	138.3	122.8	54	53.3	1345.3	483.9	66.1	59	19.0	1136.3	140.0	146.0	59	7.3	179.0	48.7	65.2
Total Dissolved Solids	mg/l	60	30.0	81.6	52.9	16.6	54	35.3	196.4	135.4	15.6	59	32.6	109.5	64.8	23.8	59	48.8	213.6	124.5	19.8
Total Inorganic Nitrogen	mgN/l	60	0.058	0.995	0.358	0.043	54	0.192	1.962	0.704	0.051	59	0.353	1.652	0.800	0.040	59	0.062	0.788	0.360	0.052
Total Kjeldahl Nitrogen	mgN/l	60	0.1	2.1	0.8	59.0	54	0.2	3.9	1.1	77.2	59	0.1	4.2	0.8	83.5	59	0.2	3.4	0.7	88.0
Total Organic Carbon	mg/l as C	60	1.5	4.3	2.4	27.0	54	1.9	7.8	3.5	27.7	59	1.1	3.9	2.4	33.2	59	2.1	5.9	3.3	23.1
Total Phosphorus	ugP/l	60	7.5	94.7	28.4	71.0	54	7.5	98.2	33.4	58.7	59	7.5	98.3	25.1	70.1	59	7.5	53.0	16.5	63.9
TP load	kg/ha/yr	60	78	5840	719	1477	54	16	1439	249	1238	59	3	4133	632	1448	59	430	24241	3015	1371
TP:Si		60	0.0	0.0	0.0	73.4	54	0.0	0.0	0.0	53.7	59	0.0	0.0	0.0	62.9	59	0.0	0.0	0.0	84.2
Turbidity	NTU	60	2.4	57.5	14.7	96.2	54	13.6	235.0	59.7	78.0	59	1.7	193.0	16.0	187.7	59	1.1	79.8	6.1	182.0
Wind direction	degrees	60			84	46.8	54			129	42.8	59			94	40.8	59			129	36.5
Wind speed	m/s	60	1.0	5.7	2.3	41.5	54	1.0	7.1	3.7	36.4	59	1.0	7.0	2.5	44.0	59	1.3	6.5	3.8	33.2
Zinc	mg/l	60	0.0	0.0	0.0	42.9	54	0.0	0.1	0.0	62.1	59	0.0	0.1	0.0	92.9	59	0.0	0.1	0.0	83.9

Physico-chemical		Midmar monthly data summary					Nagle monthly data summary					Nungwane monthly data summary					Shongweni monthly data summary				
variable	Units	N	Min	Max	Mean	CV	N	Min	Max	Mean	CV	N	Min	Max	Mean	CV	N	Min	Max	Mean	CV
% DO	%	59	81.2	118.6	101.5	8.7	59	72.1	133.1	96.6	12.3	59	66.4	127.8	94.4	12.2	58	33.9	159.7	85.2	30.6
Alkalinity	mg/l $CaCO_3$	59	21.2	29.2	25.0	7.9	59	25.3	33.2	28.8	6.5	59	16.8	33.3	24.8	16.1	58	38.6	190.0	106.3	38.1
Ammonia	mgN/l	59	0.0	0.1	0.0	52.6	59	0.0	0.3	0.0	112.2	59	0.0	0.4	0.1	81.0	58	0.0	2.2	0.1	223.0
Biological Oxygen Demand	mg/l O_2	59	0.0	3.9	1.0	69.1	59	0.1	6.8	1.2	79.1	59	0.3	6.4	1.4	60.8	58	0.5	122.0	9.1	200.8
Cadmium	ug/l	59	0.5	1.0	0.5	14.7	59	0.5	2.0	0.7	62.5	59	0.5	0.5	0.5	0.0	58	0.5	1.6	0.5	31.8
Calcium	mg/l	59	3.1	5.3	4.4	9.2	59	4.3	6.5	5.2	10.0	59	2.3	5.8	3.8	18.4	58	6.0	22.7	14.6	25.8
Chloride	mg/l	59	3.5	5.4	4.4	10.9	59	6.1	13.5	7.6	17.4	59	17.4	33.0	23.6	12.6	58	35.5	190.0	95.1	39.2
Chlorophyll a	ug/l	59	1.1	59.3	4.3	177.3	59	0.3	10.8	3.4	69.6	59	1.0	12.9	4.3	53.0	58	1.0	227.9	31.0	136.7
Chrome	ug/l	59	2.2	36.0	7.9	74.8	59	1.3	61.8	7.4	117.2	59	1.7	9.7	4.0	30.1	58	1.0	30.0	6.0	83.4
Conductivity	mS/m	59	5.9	8.0	6.8	8.7	59	5.9	12.5	8.9	10.2	59	10.3	18.1	13.4	12.5	56	28.8	161.6	79.8	46.2
Copper	mg/l	59	0.0	0.0	0.0	47.8	59	0.0	0.0	0.0	47.4	59	0.0	0.0	0.0	47.0	58	0.0	0.1	0.0	76.2
Dissolved Oxygen	mg/l O_2	59	6.7	10.9	8.3	11.1	59	5.9	10.9	8.1	13.1	59	5.9	10.4	8.1	11.5	58	2.7	12.5	7.2	27.5
Flouride	ug/l	59	27.3	46.1	36.9	9.4	59	27.3	63.3	38.6	13.5	59	17.4	83.8	39.3	21.8	58	37.5	179.0	59.2	60.6
Hardness	mg/l $CaCO_3$	59	18.0	25.7	22.0	7.5	59	24.3	33.8	27.6	9.5	59	16.1	35.3	25.5	14.8	58	48.8	108.4	77.2	20.5
Inflow	Ml/day	59	61	4057	484	144	59	144	879	444	34	59	3	473	34	220	60	2	1192	138	142
Iron	mg/l	59	0.2	1.0	0.6	34.5	59	0.2	1.6	0.5	46.0	59	0.5	2.6	1.0	36.5	58	0.1	3.5	1.0	95.5
Lead	ug/l	59	0.5	5.3	1.1	78.6	59	0.5	2.0	0.9	55.3	59	0.5	2.4	0.9	39.4	58	0.5	6.7	2.0	76.6
Magnesium	mg/l	59	2.3	3.2	2.6	7.8	59	3.1	4.3	3.4	9.4	59	2.5	5.5	3.9	12.9	58	6.6	15.9	9.8	17.7
Manganese	mg/l	59	0.0	0.0	0.0	43.3	59	0.0	0.0	0.0	54.3	59	0.0	0.4	0.1	125.8	58	0.0	0.7	0.2	93.6
Mercury	ug/l	59	0.3	2.0	0.4	72.8	59	0.3	0.3	0.3	0.0	59	0.3	0.6	0.3	19.9	58	0.3	2.4	0.4	92.9
Nickel	ug/l	59	0.3	9.9	2.6	83.8	59	0.3	4.8	1.8	83.1	59	0.3	2.9	1.5	32.0	58	0.7	26.5	6.1	86.4
Nitrate	mgN/l	59	0.1	0.7	0.3	34.0	59	0.1	0.6	0.3	34.8	59	0.1	1.4	0.6	54.8	58	0.0	3.2	0.6	106.8
Nitrite	mgN/l	59	0.0	0.0	0.0	4.3	59	0.0	0.1	0.0	17.0	59	0.0	0.0	0.0	6.0	56	0.0	0.0	0.0	4.4
Partial retention		59	2	289	23	157	59	1	3	2	24	59	0	43	6	122	10	93	14562	2293	206
pH		59	7.1	8.1	7.6	3.1	59	7.1	8.1	7.7	3.1	59	7.0	7.9	7.6	2.7	58	7.2	8.8	8.0	4.3
Potassium	mg/l	59	0.7	2.4	1.3	19.2	59	1.0	3.8	1.5	29.6	59	0.7	3.6	1.6	33.2	58	1.3	21.7	9.3	53.7
Secchi	m	59	0.4	4.0	1.3	49.0	59	0.3	2.6	1.2	38.1	59	0.6	1.7	1.1	22.9	58	0.1	2.4	0.9	59.9
Selenium	ug/l	59	0.3	0.3	0.3	0.0	59	0.3	0.9	0.3	34.9	59	0.3	0.3	0.3	0.0	58	0.3	0.3	0.3	0.0
Silicon	mg/l	59	2.6	5.4	4.2	17.7	59	2.6	6.8	4.1	19.6	59	5.5	11.5	7.4	12.2	56	3.2	9.3	5.3	28.3
Sodium	mg/l	59	3.6	5.5	4.4	8.8	59	5.5	9.7	6.6	11.5	59	11.7	20.6	15.6	11.8	58	19.8	298.0	129.3	59.2
Soluble Reactive Phosphorus	ugP/l	59	1.5	10.7	3.9	54.3	59	1.5	14.0	3.9	66.7	59	1.5	12.0	5.3	49.6	58	3.0	94.0	20.6	82.7
SRP load	kg/ha/yr	59	7	1507	141	188	59	12	263	74	66	59	1	393	20	312	58	6	1020	231	116
SRP:Si		59	0.0	0.0	0.0	63.6	59	0.0	0.0	0.0	71.5	59	0.0	0.0	0.0	49.1	58	0.0	0.0	0.0	94.6
Stability	s^{-2}	59	0.0	9.9	2.6	91.6	59	-0.2	8.0	3.1	80.8	59	-0.2	12.9	6.0	69.5	58	-0.3	11.9	4.7	83.4
Sulphate	mg/l as SO_4	59	1.5	3.9	2.1	17.8	59	3.1	7.5	4.0	21.1	59	1.4	11.3	3.6	66.1	58	38.5	355.0	142.3	56.6
Sunhours	hours	59	2.1	10.2	6.9	27.4	59	1.9	11.1	7.4	29.7	59	1.3	36.9	7.3	60.5	58	1.6	10.9	6.9	32.6
Suspended Solids	mg/l	59	2.0	16.5	7.4	50.1	59	2.0	41.5	9.3	77.8	59	2.0	60.3	7.2	108.5	58	2.0	142.6	23.1	123.0
Temperature	degrees	59	10.6	25.6	18.7	21.1	59	13.0	27.6	21.6	17.8	59	12.5	26.3	20.6	18.4	58	11.8	27.6	21.2	20.3
TN load	kg/ha/yr	59	240	35640	5519	138997	59	1240	20730	5578	75485	59	20	15530	978	246863	58	0	26930	3692	150726
TN:Si		59	0.0	0.2	0.1	29.8	59	0.1	0.1	0.1	23.7	59	0.0	0.2	0.1	42.2	58	0.0	0.8	0.1	106.2
TN:SRP		59	36.6	505.6	135.0	54.2	59	30.9	529.8	165.4	48.6	59	51.1	543.4	234.8	52.1	58	1.2	256.3	66.6	82.1
TN:TP		59	5.1	51.8	20.9	54.1	59	5.4	61.7	24.7	70.8	59	4.3	140.2	57.8	63.3	58	0.2	55.6	13.7	91.6
Total Aluminum	ug/l	59	23.3	373.0	108.7	55.0	59	24.7	603.3	113.2	86.4	59	15.7	614.0	79.6	101.8	58	7.0	540.0	112.2	112.0
Total Dissolved Solids	mg/l	59	30.4	113.5	56.6	18.8	59	36.8	150.0	71.9	26.0	59	48.8	165.0	92.3	17.4	58	207.6	940.0	507.5	42.5
Total Inorganic Nitrogen	mgN/l	59	0.208	0.758	0.327	0.029	59	0.142	0.742	0.346	0.035	59	0.275	1.720	0.731	0.043	58	0.063	4.005	0.800	0.107
Total Kjeldahl Nitrogen	mgN/l	59	0.1	11.1	1.5	124.1	59	0.1	4.3	0.8	97.5	59	0.4	4.9	0.9	78.5	58	0.5	4.9	1.8	63.3
Total Organic Carbon	mg/l as C	59	1.5	4.3	2.6	22.4	59	1.4	4.8	2.4	28.9	59	2.1	6.3	3.8	29.7	58	4.9	19.9	8.7	35.9
Total Phosphorus	ugP/l	59	7.5	52.0	24.1	46.1	59	7.5	72.0	25.8	60.8	59	7.5	64.0	21.8	63.0	58	17.2	582.7	101.7	100.3
TP load	kg/ha/yr	59	56	7341	864	1785	59	172	1267	474	495	59	1	993	47	3294	58	12	2617	484	1168
TP:Si		59	0.0	0.0	0.0	59.2	59	0.0	0.0	0.0	63.0	59	0.0	0.0	0.0	67.1	58	0.0	0.1	0.0	103.2
Turbidity	NTU	59	2.3	22.1	10.5	51.9	59	4.4	79.2	13.7	91.8	59	2.5	122.7	10.8	149.6	58	2.2	200.0	30.5	125.8
Wind direction	degrees	59			91	38.4	59			86	36.4	59			128	38.1	58			142	33.7
Wind speed	m/s	59	1.0	5.5	2.4	39.6	59	1.0	4.7	2.3	36.8	59	1.3	7.3	3.9	34.5	58	1.0	6.7	3.7	35.1
Zinc	mg/l	59	0.0	0.1	0.0	74.4	59	0.0	0.0	0.0	58.3	59	0.0	0.1	0.0	70.8	58	0.0	0.4	0.0	207.1

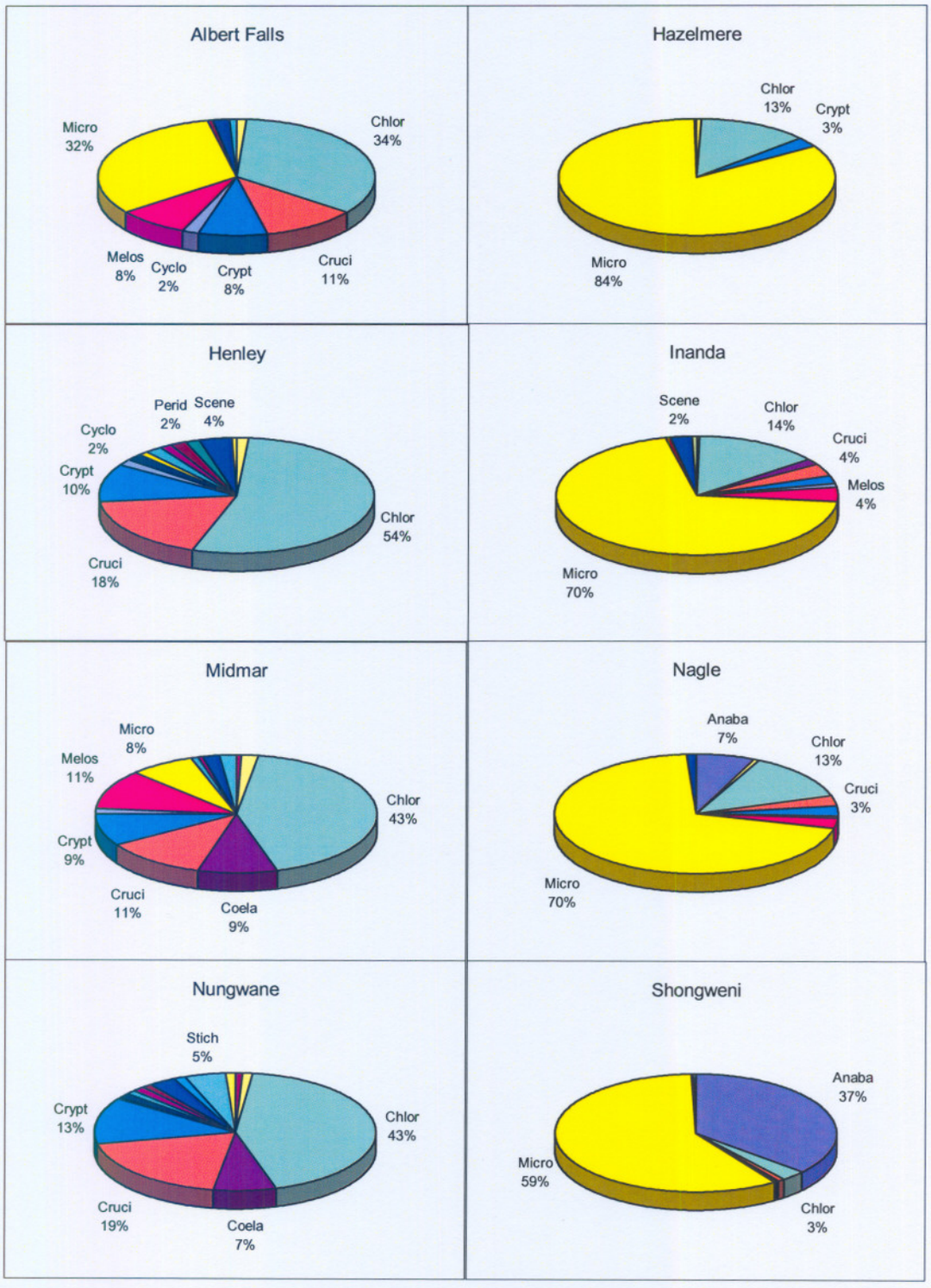

Figure 2.1 Proportional algal genera composition (as a percentage of the total median) for respective lakes for the duration of the study period (Chlor=Chlorella; Crypt=Cryptomonas; Cruci=Crucigena; Melos=Melosira; Anab=Anabaena; Micro=Microcystis; Coel=Coelastrum; Stich=Stichococcus; Perid=Peridinum; Scene=Scenedesmus; Cyclo=Cyclotella).

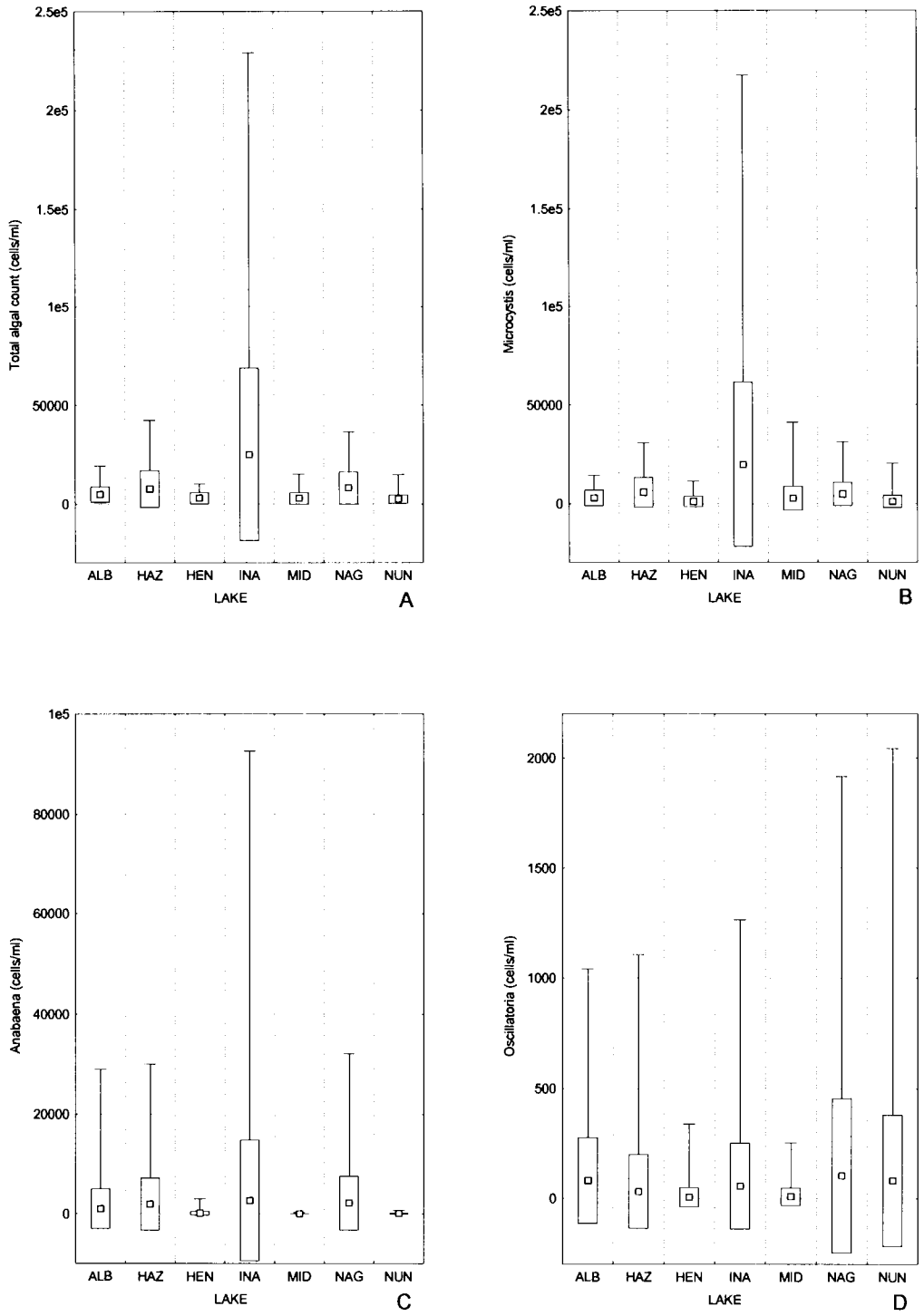

Figure 2.2 Summary abundance statistics (median, 25&75 percentiles & non-outlier minimum & maximum) of Total Algae and key blue-green algal genera in lakes.[1] [2]

[1] Shongweni algal abundances not reflected on Figs. 2.2 – 2.4 due to its high algal counts and the 'swamping' effect this would

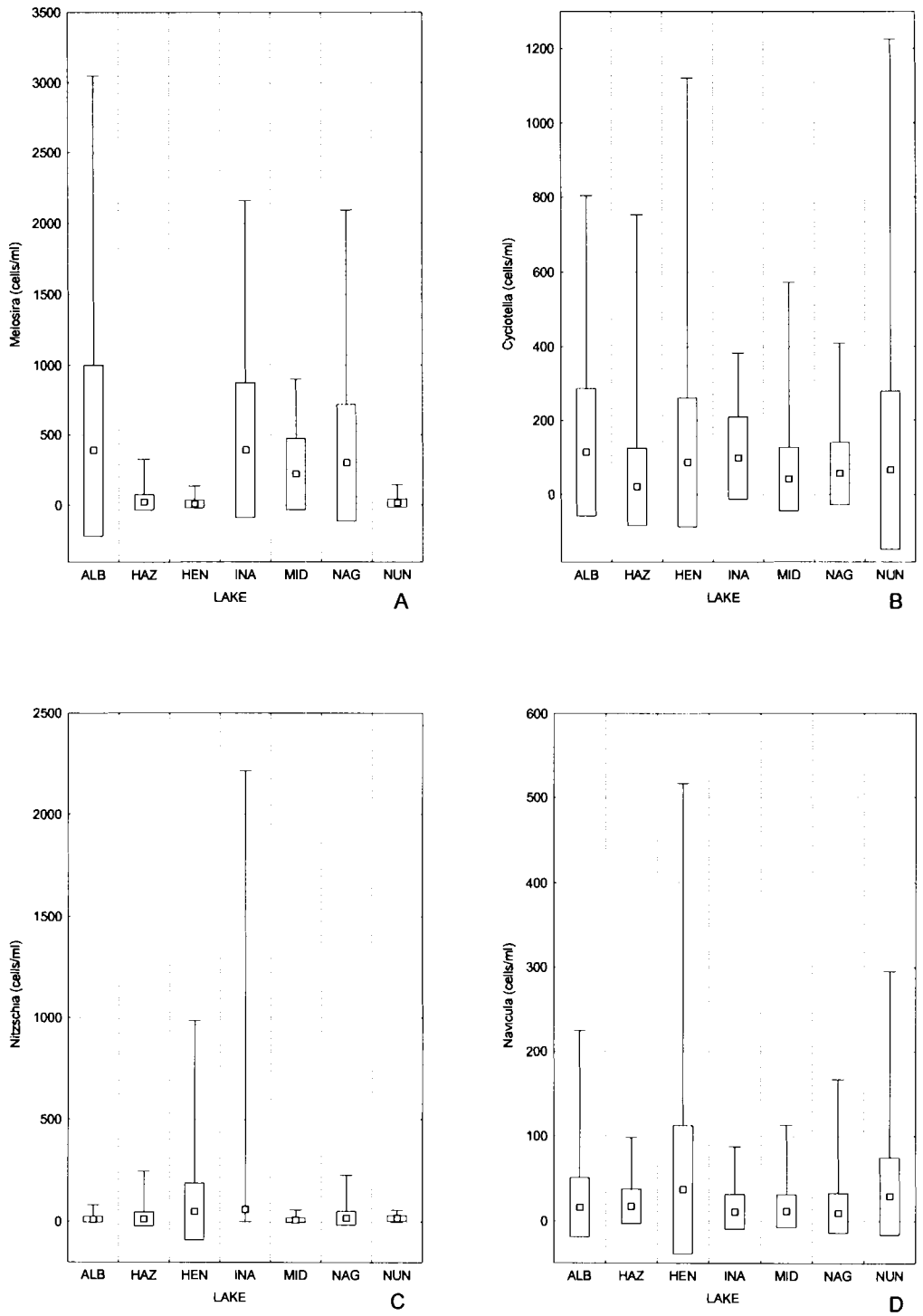

Figure 2.3 Summary abundance statistics (median, 25&75 percentiles & non-outlier minimum & maximum) of key diatom genera in lakes.

have on the display of data from other lakes.

[2] (ALB=Albert Falls, HAZ=Hazelmere, HEN=Henley, INA=Inanda, MID=Midmar, NAG=Nagle, NUN=Nungwane)

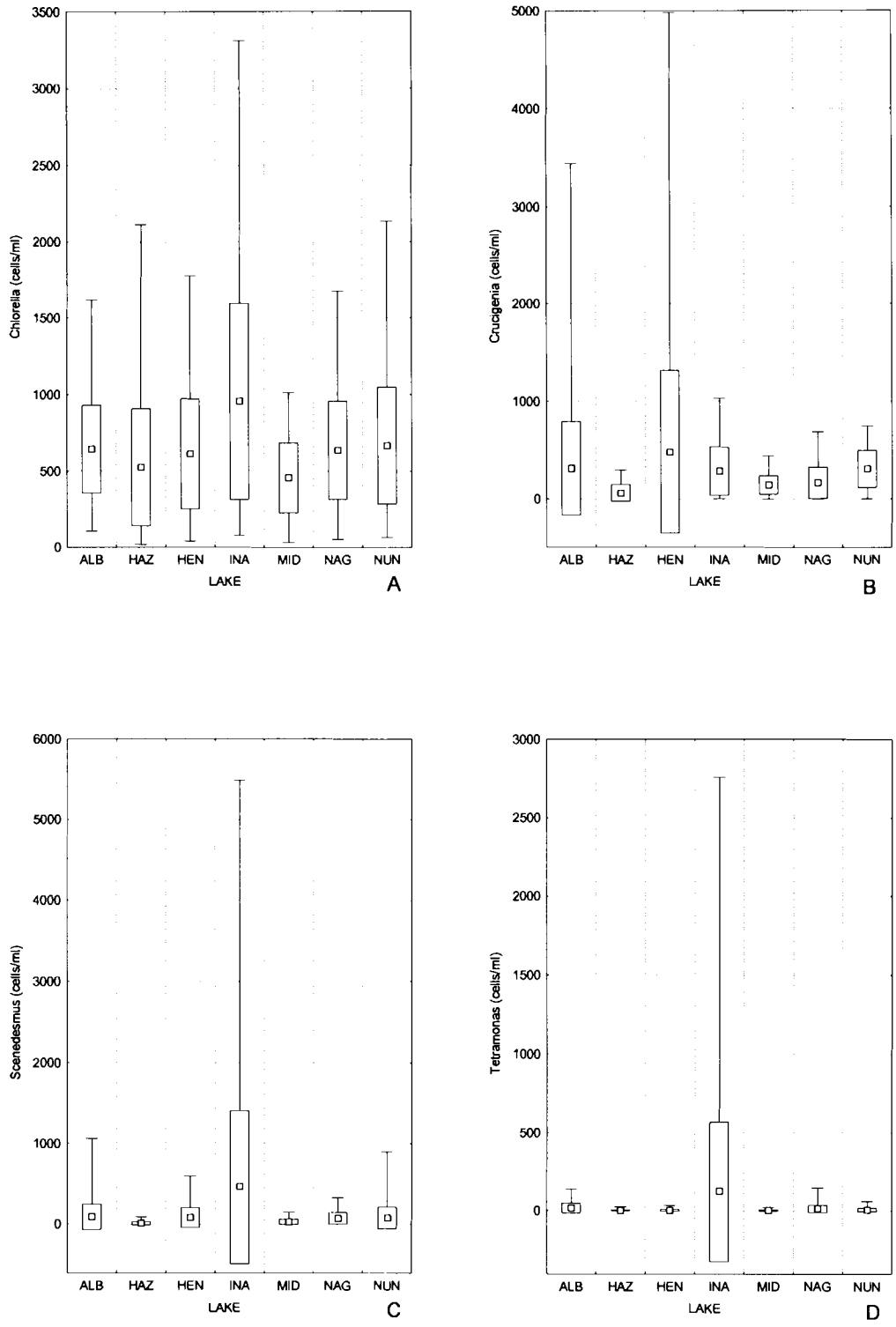

Figure 2.4(A-H) Summary abundance statistics (median, 25&75 percentiles & non-outlier minimum & maximum) of key green algal genera in lakes.
(Figure continued overleaf)

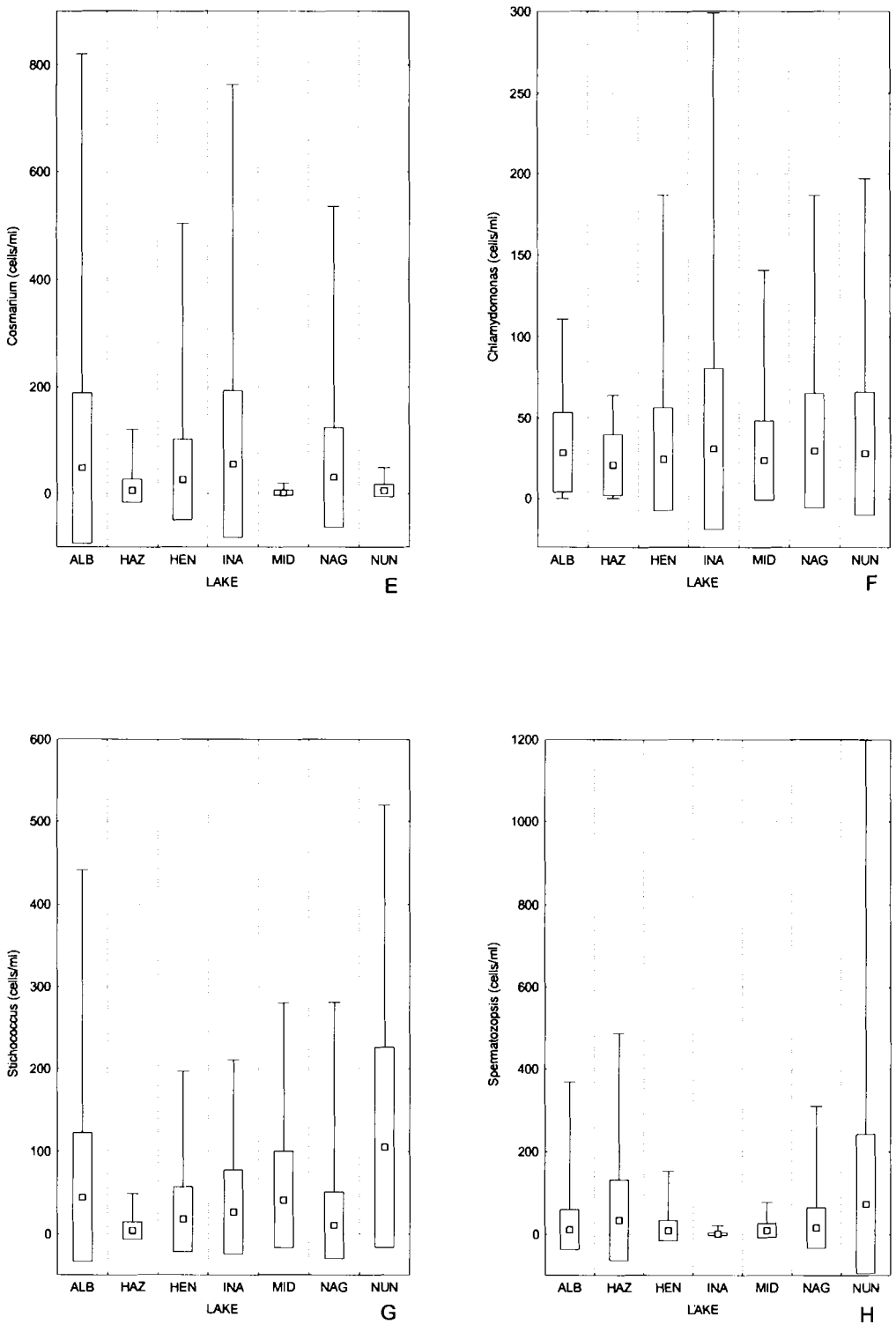

Figure 2.4(A-H) Summary abundance statistics (median, 25&75 percentiles & non-outlier minimum & maximum) of key green algal genera in lakes.

CHAPTER 3: DATA AND ANALYTICAL METHODS

3.1 Introduction

This chapter consists of two principal sections:

- A description of the various sources of data used in this study,
  - including procedures used to handle missing data and data at the detection limit of analytical instrumentation;
  - derivation of supplementary environmental variables; and
  - manipulations of data prior to analyses; and
- An overview of the statistical/analytical methods pursued in this study.

Umgeni Water has an extensive water quality monitoring programme covering a range of sites in all of the major lakes and waterworks (WW) within its operational area. Physico-chemical data from these sites is stored in their in-house laboratory information management system (LIMS) database and provided much of the raw data used in analyses for this study. Water physico-chemistry and algal abundance data (monthly arithmetic means for each determinant/algal genera) used in this study were derived from a SANAS accredited analytical laboratory in terms of ISO/IEC Guide 25, and now most recently to ISO/IEC Guide 17 025, 1999.

The modelling of algae/environment relationships was based on sites monitored in the lakes' *main-basins*, for the five-year period from 1990 to 1994. By contrast, the economic models developed used, wherever possible, the water quality measurements made on the inflow (or *raw-water*) to the respective waterworks (WW) for the seven-year period from 1990 to 1996. Due to their spatial seperation, water quality at the lake abstraction point and lake main-basin is likely to differ in certain respects (e.g. as identified by Dickens *et al.*, 1996).

Further supplementary environmental variables (e.g. key weather parameters at lakes) had to be derived from other sources. Several new determinants[3] were

[3] Determinant – for the purposes of this thesis taken to mean *"a physico-chemical parameter/analyte, measured, analysed or calculated, reflecting or impacting on some aspect of the limnological or water treatment environment".*

derived from those present in the original data set. For example: Total Inorganic Nitrogen (from the summation of nitrite (NO_2), nitrate (NO_3) and ammonia (NH_3)); and ratios from some of the original determinants (e.g. Total inorganic Nitrogen and Total Phosphorus ratios, TN:TP).

Chemical cost data were determined at December 1996 prices (Chapter 6, Table 6.2). Prices relate to the brand of chemicals most frequently used in the treatment of water at respective WW. Costs were expressed in South African Rands per mega-litre (Ml) of water treated, and refer only to expenditure on chemicals. Unfortunately no operational cost data, such as electricity consumption, backwash times etc. were available for respective WW. Data on water quality recorded at each plant were expressed in monthly terms to correspond with average monthly measures of treatment chemical usage.

The names (acronyms, units and an indication of their source) of all environmental variables considered in this study are presented in Table 3.1. Summary descriptive statistics for these data in respective lakes has already been provided in Table 2.2 with an indication of the algal composition in respective lakes shown in Figures 2.1 to 2.4.

Table 3.1 Summary of all environmental variables used in analyses along with their relevant method of extraction, acronyms, units and source

Environmental variables	Extraction technique [4]	Variable acronyms	Units	Derived from LIMS (arrived at by chemical analysis)	Modified from LIMS data (or variables calculated)	Derived from supplementary sources
Alkalinity		alkal	mg/ℓ $CaCO_3$	Yes		
Ammonia		NH_3	mgN/ℓ	Yes		
Biological Oxygen Demand		BOD	mg/ℓ O_2	Yes		
Cadmium	acid	Cd	ug/ℓ	Yes		
Calcium	acid	Ca	mg/ℓ	Yes		
Chloride		Cl	mg/ℓ	Yes		
Chlorophyll 'a'		Chl	ug/ℓ	Yes		
Chrome	acid	Cr	ug/ℓ	Yes		
Conductivity		cond	ms/m	Yes		
Copper	acid	Cu	mg/ℓ	Yes		
Dissolved Oxygen		DO	mg/ℓ O_2	Yes		
% Dissolved Oxygen		% DO	%	Yes		
Fluoride		F	ug/ℓ	Yes		
Hardness		hardness	mg/ℓ $CaCO_3$	Yes		
Inflow (adjusted for lagging effect)		inflowa	Mℓ/day		Yes	Yes (for Shongweni)
Iron	acid	Fe	mg/ℓ	Yes		
Lead	acid	Pb	ug/ℓ	Yes		
Magnesium	acid	Mg	mg/ℓ	Yes		
Manganese	acid	Mn	mg/ℓ	Yes		
Mercury	total acid	Hg	ug/ℓ	Yes		
Nickel	acid	Ni	ug/ℓ	Yes		
Nitrate		NO_3	mgN/ℓ	Yes		
Nitrite		NO_2	mgN/ℓ	Yes		
pH		pH		Yes		
Photic-zone Temperature		Temp	°C		Yes	
Potassium	acid	K	mg/ℓ	Yes		
Secchi depth		Secc	m	Yes		
Selenium	total acid	Se	ug/ℓ	Yes		
Silicon	acid	Si	mg/ℓ	Yes		
Sodium	acid	Na	mg/ℓ	Yes		
Soluble Reactive Phosphorus		SRP	mgP/ℓ	Yes		
Sol. Reactive P : Si ratio		SRP/Si			Yes	
Soluble Reactive P load		SRPLD	kg/ha/yr		Yes	
Sulphate		SO_4	mg/ℓ as SO_4	Yes		
Sun hours		SHrs	hours			Yes
Suspended Solids		SS	mg/ℓ	Yes		
Total Aluminium	acid	TAl	ug/ℓ	Yes		
Total Dissolved Solids		TDS	mg/ℓ	Yes		
Total inorganic Nitrogen		TNE3	mgN/ℓ	Yes		
Total inorganic Nitrogen load		TNLDE-3	kg/ha/yr		Yes	
Total Kjeldahl Nitrogen		TKN	mgN/ℓ	Yes		
Total N : Si ratio		TN/Si			Yes	
Total N : Soluble Reactive P ratio		TN/SRP			Yes	
Total N : Total P ratio		TN/TP			Yes	
Total Organic Carbon		TOC	mg/ℓ as C	Yes		
Total P : Si ratio		TP/Si			Yes	
Total Phosphorus	total acid	TP	ugP/ℓ	Yes		
Total Phosphorus load		TPLDE-1	kg/ha/yr		Yes	
Turbidity		turb	NTU	Yes		
Water column stability index		StabE4	s^{-2}		Yes	
Wind direction		WnDir	°			Yes
Wind speed		WnSp	m/s			Yes
Zinc	acid	Zn	mg/ℓ	Yes		

[4] If not an aqueous extraction technique

3.2 Data screening and manipulations

All data were screened in a spreadsheet environment for obvious outliers or errors in the original data. For example a pH of greater than 14, or less than 1 was not possible and so was highlighted for checking. Where feasible, checks and corrections were made against original laboratory analytical records. After checking it was found that these errors were, however, very rare and constituted a very minor proportion of the data set. Where there was an obvious error, and it was not possible to trace or correct the original data, it was removed from subsequent analyses.

Results recorded at the detection limit for a specific analyte (or determinant) were halved so as not to over-emphasise their significance in statistical analyses. If the detection limit changed during the period of the study, then the lower of the detection limits was selected and 50% of this value returned as some estimate of the actual value for that period. Results that were below the detection limit were often only a minor proportion of the total data set available for that determinant. As the analyses ran, variables with data at their detection limit did not appear to be significant factors in explaining observed algal or treatment cost variability.

3.3 Missing Data

Even with the monthly averaging of certain weekly data, there were a number of environmental variables that still had some missing data. Missing data generally applied to environmental variables that were only monitored on a quarterly basis. Missing data was dealt with by substituting the best possible estimate (by interpolating between data points, by month and lake) for the respective variable in the place of missing data. As Manly (1991) notes, "doing something about missing values is by no means a straightforward matter". This is particularly problematic with multivariate data and his suggestion is to use "common sense" methods of estimation and that this procedure should work satisfactorily "providing that only a small proportion of values are missing."

A summary of the extent of missing data encountered in this data set is presented in Table 3.2. Key findings from this investigation were that only 14% of the total data

set were missing. Furthermore, the majority of this missing data was for heavy metals and other variables which typically did not vary much, and which the literature did not highlight as significantly affecting algal dynamics or water treatment. For this reason these determinants were generally only monitored on a quarterly basis. If this type of data were removed from the calculations of missing data, the percentage of missing data dropped to only 1% of the entire data set (Table 3.2).

Table 3.2 Summary of the percentage of the entire environmental data-set comprising 'missing' data (summarised by lake & environmental variable)

Environmental variables	Midmar	Nagle	Albert Falls	Inanda	Henley	Shongweni	Hazel-mere	Nungwane	Average over all lakes
alkal									
BOD	71	71	68	68	71	64	75	67	69
Ca	3	36	28	2	8	5	7	7	12
Cd	69	73	68	5	68	16	67	67	54
Chl				2	3	9	25		5
Cl									
cond									
Cr	69	73	68	7	69	12	67	67	54
Cu	69	73	68	3	68	16	68	67	54
DO				2			18		3
% DO				2					
F	69	71	68	66	68	67	67	67	68
Fe									
hardness	3	37	28	2	8	5	5	5	12
Hg	69	73	70	5	69	16	67	67	54
Inflowa									
K	3	36	28	2	8	5	5	5	12
Mg	3	36	30	2	8	5	7	7	12
Mn									
Na	3	36	30	3	8	7	5	5	12
NH_3									
Ni	69	73	68	5	69	16	68	67	54
NO_2									
NO_3								2	
Pb	69	75	72	10	68	12	67	70	55
pH			2						
Se	69	73	68	5	68	14	73	67	55
Secc	5	7	32		14		22	43	15
SHrs		2	8	8	8	5		8	5
Si									
SO_4	5	7	7	5	5	7	3	3	5
SRP									
SRP/Si									
SRPLD					2	2			
SS									
StabE4									
TAl			2						
TDS	69	68	68	66	68	69	70	67	68
Temp									
TKN	15	15	17	15	15	17	17	17	16
TN/Si									
TN/SRP								2	
TN/TP								2	
TNE3								2	
TNLDE-3					2	2			
TOC				5		9	5		2
TP									
TP/Si									
TPLDE-1					2	2			
turb									
WnDir		2	8	7	8	5		8	5
WnSp		2	8	7	8	5		8	5
Zn	69	73	68	3	68	9	67	67	53
Overall average									14
Average (excluding quarterly data)									1

3.4 Sourcing and derivation of physico-chemical environmental data

Most of the lake physico-chemical environmental data were available for data analysis in its raw format from the Umgeni Water LIMS. Other data had to be secured from other sources and/or calculated or modified before appropriate analyses could be undertaken.

3.4.1 Data available from LIMS

As noted earlier in the introduction to this chapter, much of the environmental data was available from the Umgeni Water LIMS database system. Most of the environmental variables from the database were used 'as is', although some had to be manipulated for various reasons. Temperature was a case in point with the literature indicating that some measure of the average temperature within the photic-zone is the most significant with respect to phytoplankton productivity (e.g. Wetzel, 2001). Accordingly, new temperature data were derived for respective samples from the average temperature in the water column, limited to the secchi depth. It was assumed that the photic-zone was approximately that of the secchi depth. Table 3.1 reflects the origin of environmental variables.

3.4.2 Nutrient loads on lakes

Nutrient loads for respective lakes were estimated using the FLUX model (Walker, 1987). The model essentially combines a grab sample (for nutrient concentration) from the inflow to respective lakes, with the inflow rate, to create a 'best estimate' of nutrient loads on the system. Monthly nutrient loads were calculated for all lakes for total inorganic nitrogen, total phosphate and soluble reactive phosphate.

3.4.2.1 Derivation of data for use in nutrient load calculations in Shongweni

No nutrient load data were available for Shongweni because of a lack of gauging weirs on its three main catchments (the Umlaas, Sterkspruit and Wekeweke). Without this inflow data nutrient loads were unable to be calculated for the respective rivers draining into Shongweni. Therefore, flows had to be estimated for this system. The ACRU model (ACRU, 1995) was used to generate synthetic flow sequences, and with these flow estimates nutrient loads could be calculated.

One of the rivers flowing into Shongweni was the Sterkspruit. This river flows through the Hammarsdale industrial area and at times has a significant proportion of its flow increased by the addition of return flows from the Hammarsdale Wastewater Works (WWW). The monthly average WWW inflow volumes were added to the respective monthly ACRU estimates for the Sterkspruit to derive an estimated total inflow arriving at Shongweni from this catchment. The Sterkspruit comprises on average 10% WWW return flow with a maximum of 65% and minimum of 3% (for the study period of 1990 to 1994, this study).

3.4.2.2 Lagging of nutrient loads

Previous work by Pillay (1994) on the Inanda system indicated that there might be a lag effect whereby the impact of nutrient loads, calculated at the head (inflow) of the lake, took time to reach the lake main-basin (where the modelling for this study was undertaken). Lagging was only considered an issue in Inanda, due to its length and long retention times, although a smaller lake (Midmar) was also tested and found to not require lagging. A correlation analysis, between nutrient loads and certain numerically dominant algal genera, revealed that a lag between four and six-weeks (Table 3.3) gave the highest correlations with the dominant algae selected and hence all the load data were offset forward by five weeks (the best overall lag period) in all further analyses involving Inanda data.

Table 3.3 Results of a lagged correlation analysis between nutrient loads and three key dominant algal genera in Inanda

	2 week lag			4 week lag			5 week lag			6 week lag		
	Micro	Anab	Chlor	Micro	Anab	Chlor	Micro	Anab	Chlor	Micro	Anab	Chlor
TPLD	0.39	0.29	0.23	0.59	0.41	0.28	0.60	0.40	0.29	0.59	0.35	0.32
SRPLD	0.43	0.43	0.31	0.58	0.53	0.30	0.58	0.52	0.29	0.53	0.45	0.30
TNLD	0.54	0.37	0.31	0.68	0.45	0.28	0.68	0.44	0.26	0.62	0.38	0.28

Micro = *Microcystis*
Anab = *Anabaena*
Chlor = *Chlorella*
TPLD = Total Phosphorus load
SRPLD = Soluble Reactive Phosphorus load
TNLD = Total Nitrogen load

The lag effect is clearly evident in Figure 3.1 with the peaks in algal counts largely corresponding to the peaks in soluble reactive phosphorus (SRP) loads, once lagging had been applied to the data.

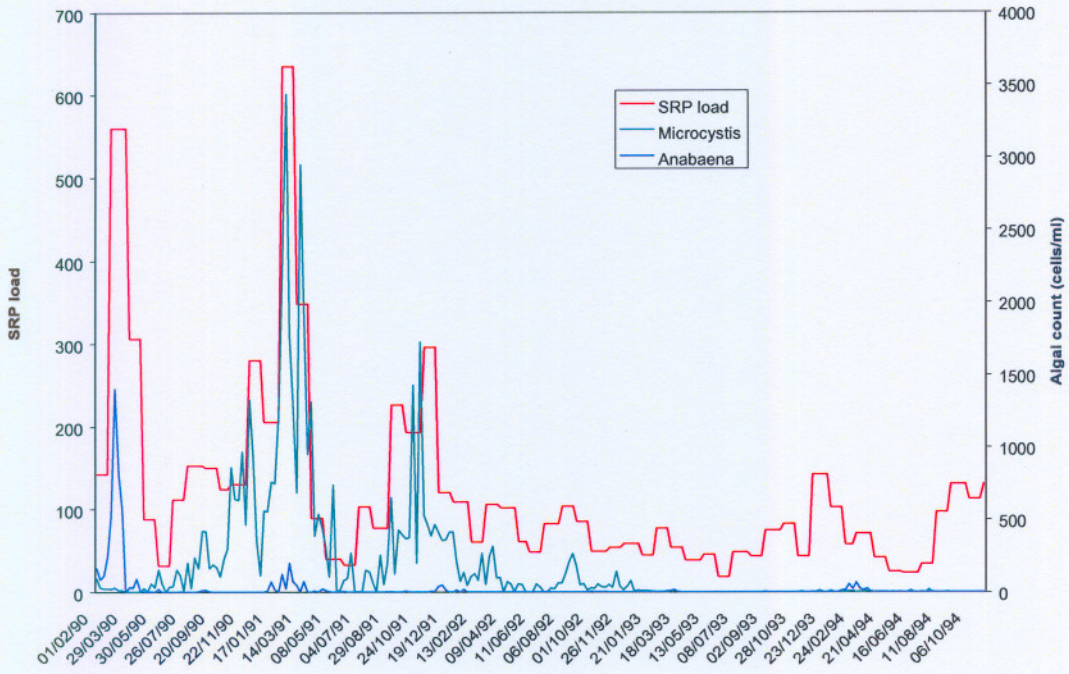

Figure 3.1 *Microcystis* and *Anabaena* algal counts in relation to a five week lag with Soluble Reactive Phosphorus (SRP) loads (kg/ha/yr) in Lake Inanda.

3.4.2.3 Relationships between lake inflow, nutrient concentrations and nutrient loads

Through various iterations of analyses it became apparent that two potentially biologically important variables, nutrient concentration in the water column and mass flow of water (represented as river 'inflow' in this study), were confounded when they were 'collapsed' into a single composite variable called 'nutrient load'[5]. The respective variables were also highly inter-correlated (or multicollinear – see 3.7.2.1) with the nutrient loads and this made interpretations of results questionable. Analyses involving the composite nutrient load variables directly were therefore undertaken with circumspection, with greater emphasis and focus given to 'inflow' and 'nutrient concentration'.

Inflow can affect algae both directly (i.e. the mass movement of water) and indirectly (i.e. through its effect on the concentration/dilution of nutrients in the water column). So whilst on the one hand, increased inflow may increase the nutrient loads on lakes, it may also be diluting the concentration of respective nutrients within a system.

[5] Nutrient loads are a function of the concentration of a nutrient (taken from a grab sample at the inflow to the lake) and the inflow rates (mass flow of water) into the lake.

Figure 3.2 clearly illustrates these relationships for a range of perceived key nutrients (Total Phosphorus, Soluble Reactive Phosphorus and Total Inorganic Nitrogen) from the pooled data from all lakes studied.

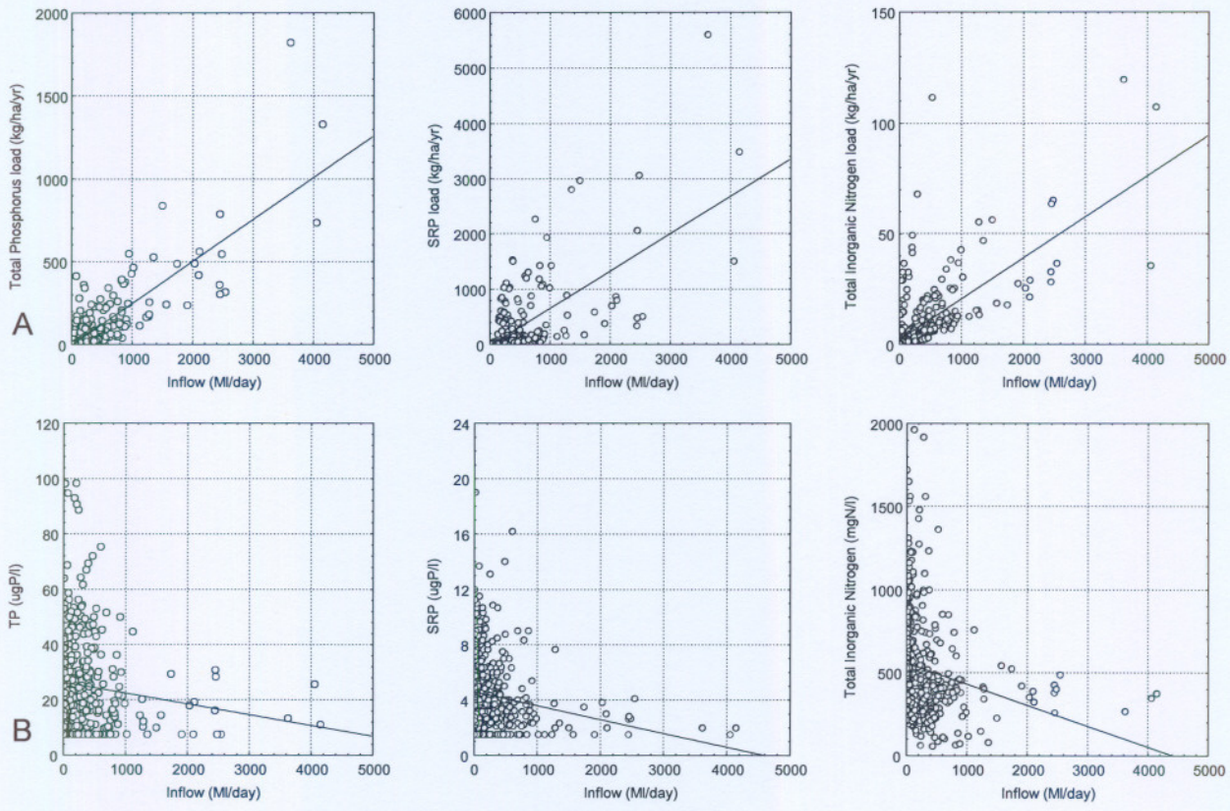

Figure 3.2 Scatterplots (with trend line) illustrating the correlation 'inflow' has with both in lake nutrient loads (A) and in lake nutrient concentrations (B) for all lakes.

3.4.3 Lake weather data

This aspect of the study was designed to identify those South African Weather Service (SAWS – formerly Weather Bureau) stations providing detailed weather data that could be used to infer weather conditions at respective lakes where this data was either unavailable or deficient from local Department of Water Affairs and Forestry (DWAF) lake meteorological stations.

The strength of the correlation between rainfall measured at SAWS and those taken at the different lakes by DWAF was used to establish which SAWS station was best

suited to provide detailed weather records to be used in analyses. A basic assumption made in this aspect of the study was that rainfall was a representative variable co-varying with a range of weather conditions, e.g. temperature, wind and cloud cover (and hence sunshine hours). If there was a good correlation between rainfall at the SAWS and DWAF stations, then it was assumed that the other weather variables of interest for this study could be reasonably extrapolated from the closest SAWS weather station and used to 'patch' the environmental data set derived for the respective lakes. This SAWS data includes wind speed, wind direction and sun hours.

It was reasoned that sun hours would affect algae because of their widely acknowledged photosynthetic dependence on light (e.g. Wetzel, 2001). Wind speed and direction were seen as possibly important variables due to their influence on the stability of the water column in the epilimnion, or upper/surface layers of the lake waters – regions of greatest phytoplankton biomass.

Weather data from various weather stations and lakes in KwaZulu-Natal were obtained from the SAWS and DWAF head offices in Pretoria, South Africa. An initial sorting of weather stations into classes was conducted to isolate those that would be able to provide the most detailed weather information. These classes (first, second and third) followed the SAWS classification.

During the study period DWAF (or their local agents) monitored rainfall and temperature (amongst other parameters) at the following lakes studied: Midmar, Albert Falls, Nagle, Inanda, Hazelmere and Shongweni. Nungwane and Henley were not monitored. The correlation analysis was run taking monthly rainfall data from each of the lakes (Albert Falls, Midmar, Hazelmere and Nagle) and comparing this with data from the SAWS stations for the same period. The SAWS stations used were Cedara, Pietermaritzburg, Durban Airport and Mount Edgecombe. Pearson's *r* correlation coefficient was used to measure the strength of relationships between these two sets of data.

The distribution of the data was checked and found to be generally log-normally distributed. This cautioned against the use of parametric correlation methods (e.g. Pearson's *r*), particularly in the interpretation of the significance of the correlation coefficient. However, as acknowledged in various statistical texts, because the sample sizes were reasonably large (i.e. $n>50$) serious biases are unlikely. The

results of the correlation analysis are therefore acceptable and were worthy of further interpretation and were summarised in the following correlation table.

Table 3.4 Table of correlation coefficients (*r*) for rainfall comparisons between South African Weather Services and Department of Water Affairs and Forestry weather stations

	CedRf	PmbRf	AlbRf	MidmRf	InandRf	HazelRf	NaglRf	ShngRf	DurbRf	MtEdRf
CedRf	1.00									
PmbRf	.90	1.00								
AlbRf	.92	.93	1.00							
MidmRf	.95	.90	.93	1.00						
InandRf	.80	.67*	.71	.78	1.00					
HazelRf	.73	.81	.71	.67	.91	1.00				
NaglRf	.85	.93	.86	.83	.86	.92	1.00			
ShngRf	.61	.70	.64	.58	.73	.73	.71	1.00		
DurbRf	.70	.84	.73	.71	.88	.79	.83	.70	1.00	
MtEdRf	.65	.75	.65	.64	.97	.87	.82	.73	.84	1.00

(*All correlation coefficients significant at *P<0.01* except for PmbRf & InandRf *P=0.07*)

Explanation of codes

Code	Explaination of codes used and source of data
CedRf	Cedara rainfall (South African Weather Services)
PmbRf	Pietermaritzburg rainfall (South African Weather Services)
AlbRf	Albert Falls rainfall (Dept. Water Affairs & Forestry)
MidmRf	Midmar rainfall (Dept. Water Affairs & Forestry)
InandRf	Inanda rainfall (Dept. Water Affairs & Forestry)
HazelRf	Hazelmere rainfall (Dept. Water Affairs & Forestry)
NaglRf	Nagle rainfall (Dept. Water Affairs & Forestry)
ShngRf	Shongweni rainfall (Dept. Water Affairs & Forestry)
DurbRf	Durban Airport rainfall (South African Weather Services)
MtEdRf	Mt. Edgecombe rainfall (South African Weather Services)

Geographically there are two distinct groups of lakes - the inland (Midmar and Albert Falls) and the coastal lakes (Hazelmere and Shongweni). Nagle and Inanda fall somewhere between these two groups. Not unexpectedly the inland lakes are most closely correlated with the Cedara and Pietermaritzburg SAWS stations, whilst the coastal lakes are most closely correlated with the coastal SAWS stations at Durban airport and Mt. Edgecombe.

The results indicate that the Cedara weather station is most closely reflecting weather conditions for the following lakes: Midmar, Albert Falls and Nagle. Hazelmere, Inanda and Shongweni, on the other hand, are more strongly correlated with the Durban Airport and Mt. Edgecombe stations. Inanda, which is geographically lower down in the catchment and hence closer to the coast and

Durban Airport and Mt. Edgecombe weather stations, is more strongly correlated with these coastal SAWS stations, particularly the latter. Unfortunately, however, the Mt. Edgecombe data set was not as complete as the Durban Airport one so this latter station was chosen to represent weather conditions at the various lakes indicated in Table 3.5. Lakes Henley and Nungwane did not have any available DWAF weather data to compare with the SAWS station data but due to their geographical position are expected to be most strongly correlated with Cedara and Durban Airport (or Mt. Edgecombe) weather stations respectively. This analysis indicated that weather data for the various lakes in this study might be inferred with some confidence from the closest high order SAWS weather stations indicated in Table 3.5.

Table 3.5 Groupings of lakes in relation to best available South African Weather Services weather station data records

Cedara Weather Station	Durban Airport Weather Station
Albert Falls	Inanda
Henley	Hazelmere
Midmar	Nungwane
Nagle	Shongweni

3.4.4 Water column stability index

Wetzel (2001) notes that "temperature and density stratification in lakes are dominant regulators of nearly all physico-chemical cycles and consequently of lake metabolism and productivity". Depending on the heat budgets of lakes, their size and morphometry, the lakes will therefore each have a unique pattern of thermal stratification and hence stability (or resistance to mixing) of the water column. Particularly in warm monomictic lake systems, as were common in this study, this stability quantifies the resistance of stratification to disruption by the wind and therefore the extent to which the hypolimnion is isolated from epilimnetic and surface movements. To some degree the water column stability index reflects the integrated effects of wind speed and wind direction.

As a measure of the physical stability of the water column at respective monitoring sites, and over time (sampling occasions) the Brunt-Väsäilä buoyancy frequency squared term (N^2) was calculated (Patterson *et al.*, 1984; Viner, 1985) from equation 1.

$$N^2 = (-g/\rho_0)(\partial\rho/\partial z) \qquad (1)$$

where g = 9.81 ms^{-1} and

$(\partial\rho/\partial z)$ = the density gradient for the entire water column, of mean density ρ_0.

Values of N^2 have units' s^{-2}.

Water density (ρ) was calculated using the following polynomial equation (Hart 1992).

$$\rho = 1000.0873+(0.0215*T)-(0.0057*T^2) \quad (2)$$

where T = temperature (°C) at the respective water depth

Temperature profiles, routinely monitored and captured into LIMS for respective lakes, were used to derive the stability index (using equations 1 and 2).

3.4.5 Flushing rate of lake water

As a measure of the flushing rate of respective lakes, a simple calculation was performed on mean lake storage divided by total outflow. Calculated on a monthly basis this provided a partial water residence time. The inverse of this gave the partial flushing rate to correspond with the monthly measures of all other 'environmental' parameters.

3.5 Algal data

All the algal data used in this study were extracted from the LIMS database. Genera used in analyses (after removal of rare and, from a water treatment point-of-view, unimportant algae) are summarised in Table 3.6. The acronyms used to represent algal genera in various figures in this thesis, as well as their potential impact on water treatment (after Palmer, 1959), are also shown in this table. The complete list of 89 algal genera analysed by Umgeni Water is presented in Appendix 1, Table A1.1.

Table 3.6 Rationalised list of algae used in analyses. Acronyms, generic names, class and potential to cause water treatment problems indicated

Algal genera (acronyms)	Algal genera (scientific name)	Algal class	Potential treatment problems (after Palmer, 1959)
Anaba	*Anabaena*	Cyanophyceae (blue-green)	taste and odour
Ankis	*Ankistrodesmus*	Chlorophyceae (green)	
Botry	*Botryococcus*	Chlorophyceae (green)	
Chlam	*Chlamydomonas*	Chlorophyceae (green)	taste and odour
Chlor	*Chlorella*	Chlorophyceae (green)	filter clogging
Coela	*Coelastrum*	Chlorophyceae (green)	
Cosma	*Cosmarium*	Chlorophyceae (green)	taste and odour
Cruci	*Crucigenia*	Chlorophyceae (green)	filter penetration
Crypt	*Cryptomonas*	Cryptophyceae (brown)	taste and odour
Cyclo	*Cyclotella*	Bacillariophyceae (diatoms)	taste and odour, filter clogging
Dicty	*Dictyosphaerium*	Chlorophyceae (green)	taste and odour, filter clogging
Eugle	*Euglena*	Euglenophyceae	taste and odour
Mallo	*Mallomonas*	Chrysophyceae	taste and odour
Melos	*Melosira*	Bacillariophyceae (diatoms)	filter clogging
Micro	*Microcystis*	Cyanophyceae (blue-green)	taste and odour, algal toxin forming
Navic	*Navicula*	Bacillariophyceae (diatoms)	filter clogging
Nitzs	*Nitzschia*	Bacillariophyceae (diatoms)	filter clogging
Oocys	*Oocystis*	Chlorophyceae (green)	
Oscil	*Oscillatoria*	Cyanophyceae (blue-green)	taste and odour, filter clogging, algal toxin forming
Pando	*Pandorina*	Chlorophyceae (green)	taste and odour
Pedia	*Pediastrum*	Chlorophyceae (green)	taste and odour
Perid	*Peridinium*	Dinophyceae	filter clogging, taste and odour
Ptero	*Pteromonas*	Chlorophyceae (green)	
Scene	*Scenedesmus*	Chlorophyceae (green)	taste and odour
Sperm	*Spermatozopsis*	Chlorophyceae (green)	
Sphae	*Sphaerocystis*	Chlorophyceae (green)	
Stich	*Stichococcus*	Chlorophyceae (green)	
Syned	*Synedra*	Bacillariophyceae (diatoms)	taste and odour, filter clogging
Synur	*Synura*	Chrysophyceae	taste and odour
Tetra	*Tetraedron*	Chlorophyceae (green)	
Trach	*Trachelomonas*	Euglenophyceae	filter clogging

Summary abundance statistics for key genera in all lakes, (summarised by class), and the median proportional composition of algae in respective lakes during the study period, have already been presented in the previous chapter (Figures 2.2 to 2.4).

3.5.1 Rationalisation of the number of algal genera investigated

Due to the size of the data set it was necessary to rationalise the number of algae examined. Therefore those with a low abundance, and particularly those that the literature acknowledged as not adversely affecting water treatment, were removed, and only the more abundant and common problem-causing algae were taken forward in further analyses.

In total 89 genera are potentially enumerated in any one algal sample analysed by Umgeni Water's Hydrobiology laboratory. Many of these genera are only rarely encountered in any sample and, if encountered, are often in low abundance. Typically, a diversity of less than twenty different algal genera will be found in any

one sample. It was therefore postulated that the "rare" algae did not significantly add to the picture or relationships that may be developed for the more commonly encountered and abundant algae. The question was then - "what constituted a 'rare' algae and at what level to draw the distinction between rare and abundant algae?"

To deal with this, monthly averages for all algal genera were considered over all months and over all the study sites (lakes) i.e. a matrix of over 89 X 469 was examined. The number of times the genus was encountered in the data set was compared with the theoretical maximum possible encounters (469). Those rare genera (encountered less than 10% of the time over the entire data set) were eliminated from further analyses after checking that none were known to be important with respect to water treatment. Several of those rare (but acknowledged to be important) were retained in this data set (e.g. *Synura* - a potentially powerful taste and odour forming alga even at low abundances (Palmer, 1959)). This reduced the total data set by some 64% (from 89 to 32 genera).

3.6 Water treatment and cost data

For each respective water treatment works (WW) studied there were two elements to this phase of data collection. Firstly, assembling all the water quality data associated with the raw water entering the waterworks, and then secondly, collating all the corresponding chemical dosage and water treatment data. The water quality data associated with respective WW was relatively easily assembled as it was on LIMS. However the cost data had to be manually captured and checked from respective WW records.

All data used in the economic study of the selected Water Works (WW), Hazelmere, Durban Heights, DV Harris and Wiggins, were sourced from Umgeni Water and covered a period of seven years, i.e. 1990 to 1996. Chemical cost data were measured in December 1996 prices (Chapter 6, Table 6.2). Prices relate to the brand of chemicals most frequently used in the treatment of water at respective WW. Costs were expressed per mega litre (Ml) of water treated, and refer only to expenditure on chemicals. Where appropriate, data on water quality recorded at specified sample points at each plant were expressed in monthly average terms to correspond with the average monthly measures of treatment chemical usage.

For the Hazelmere system, data on water quality parameters recorded at Lake Hazelmere (UW sample point 101.1) and Hazelmere WW (UW raw water sample point 314) were used to analyse this system. Missing data on chemical usage between the months of January to September and December 1993, January, May, June, and September 1994, reduced the number of valid data cases available for analysis from 84 to 71. For the Durban Heights system, water quality data recorded at Lake Nagle (UW sample point number 43.1) and Durban Heights site (UW raw water sample point number 316), recorded over the period March 1990 to February 1997 were used, totalling 84 valid cases. The DV Harris system studied was based on water quality data recorded at the Midmar site (UW sample point 36.1) and the DV Harris site (UW raw water sample point 324). Data used was from January 1990 to December 1996. Some data on chemical dosage for the months of January 1990, July 1990 - April 1991, March 1993, July 1993, February 1994 - May 1994, September 1994, and October 1994 were missing at this plant, reducing the number of valid observations from 84 to 64. Lastly, the Wiggins system was analysed based on 82 valid water quality data observations recorded at Lake Inanda (UW sample point 55.1) and the Wiggins WW (UW raw water sample point 320).

The range of water quality analyses performed on WW raw-water samples is reduced (compared to lake main basins) to the perceived bare essentials as far as water treatment was concerned. These latter analyses are those that are required to run the water treatment plant efficiently. However, from a statistical and analytical point of view, this study sought to consider as wide a range of environmental variables as possible, because these may have *some* effect on treatment costs. Therefore, as a 'best estimate' of the broad spectrum of water quality conditions at the inflow to the respective WW, supplementary environmental variables, measured or calculated for the lake main basin, were patched into the data set available for sample points at the inflow to WW's.

Previous work (Dickens *et al.,* 1996) had indicated that there was likely to be an appreciable number of algae (particularly vacuolate blue-green algae) ruptured in the process of abstraction from Lake Nagle to Durban Heights WW. Routine algal counts, available in the LIMS database from the Durban Heights WW raw water inlet, are therefore likely to be an underestimate of the 'true' algal loading on the treatment process at this particular WW – principally because the ruptured and partially ruptured algae are not easily identified and hence enumerated in samples. To further complicate matters there are two abstraction points (the 'old' and 'new' abstractions)

available at Nagle that merge at Durban Heights WW to provide the raw water for this plant to process. Accurate flow records from these two abstraction points were available from early 1989 to late 1992. These provided an accurate estimate to use in flow-weighting the algal counts up to late 1992. The average proportional flow (derived from the accurate data) in respective abstraction points (respectively 35% and 65% for the old and new abstraction points) was used to estimate flow-weighted algal counts for the remainder of the study period. These therefore provided the 'best estimate' of real algal loadings expected at Durban Heights (to be used in conjunction with all other water physico-chemical data) in the development of the economic models in the study.

3.7 Analytical methods

3.7.1 Introduction

A brief introduction to the methods of data analysis used in the various phases of the study is presented here. All the methods have been widely used in various fields of ecological and econometric research, although some of the methods and their application to environmental problems in the South African context appear little explored.

It is widely acknowledged (e.g. Jongman *et al.*, 1987) that ecosystems are generally complex, consisting of many interacting biotic (biological) and abiotic (non-biological) components. Furthermore, environmental conditions are often difficult to characterise exhaustively. Notwithstanding these limitations and difficulties a range of relatively new analytical/statistical techniques makes it possible to begin to understand some of the key processes affecting observed patterns in ecosystems. Some of these techniques were employed in this study.

3.7.2 Methods used in determining algae/environment relationships

Due to the multivariate nature of this type of this study, i.e. numerous environmental variables with the potential to simultaneously influence numerous members of an algal community in lakes, appropriate multivariate analytical techniques were required (e.g. Manly, 1991). The techniques used to illuminate different aspects of the algae/environment relationships were Principal Components Analysis, Correspondence Analysis and Redundancy Analysis. All of these could be classified as ordination methods.

Ordination is the collective term used to describe multivariate techniques that ***arrange sites*** along axes ***on the basis of the species composition at respective sites***. The general result of an ordination analysis is an ordination diagram in which sites are represented by points. Sites, which are closer together, will have a greater biological similarity compared to sites further apart on the diagram. The ordination represents a graphical summary of the biological data.

Between the classical regression analysis approach (where the primary focus is on the detailed correlation between a single species and the environment), and typical ordination techniques (which provides a more general picture of how a community of species is related to each other), stand the canonical ordination techniques. These deal with many species and many environmental variables with the central aim to detect the main patterns, or relations, between the two. They were the primary analytical tools used in this study - principally to elucidate the relationships between algae observed in the various lakes and their environment.

3.7.2.1 Principal Components Analysis – to detect the principal aspects of variation in the environmental data

It was noted in this study that many of the environmental variables considered were highly inter-correlated (both positively and negatively) with each other (Chapter 4, Table 4.1), or in other words were ***multicollinear***. Multicollinearity is not unusual with environmental data of this nature (Jongman *et al.*, 1987) and, as was also noted in the economic models developed, this causes problems for multiple regression analysis. Principal Components Analysis (PCA) was employed to overcome the problem of multicollinearity (Chatterjee & Price, 1977). The PCA technique converts

the original variables into uncorrelated variables called principal components, PCs. PCs are linear combinations of the original variables as described below:

$$PC_i = \alpha_{i1}X_1 + \alpha_{i2}X_2 + \ldots + \alpha_{ik}X_k \qquad (1)$$

where PC_i = *i*th principal component

α_{ij} = component loadings

X_j = original variables

The principal components must satisfy two conditions:

- they must be orthogonal; and
- the first component (PC_1) should account for the maximum proportion of variation in the original variables.

The PCs are also ordered in such a way that the first PC (PC_1) displays the largest amount of variation, PC_2 the second largest amount of variation, and so on (Manly, 1991).

Therefore, to simplify the understanding of the correlation structure of the data, standardised (mean 0, unit variance) Principal Components Analysis (PCA) was used to produce a few key "variables" i.e. Principal Components, PCs, which describe most of the variation in the environmental data. The result of this approach is a degree of economy in that the variation in the original large number of variables is concisely accounted for by a smaller number of variables or PCs. Another way of viewing this approach is as a "reduction in the dimensionality of the data". The reduced dimensionality of the data also obviously assists in the understanding of the problem as the variability in algae may now be described by fewer key parameters. It is worth reiterating that the reduced set of key parameters (PCs) have close (and known) associations with the underlying environmental variables comprising the PCs.

Essentially PCA takes a number of variables (environmental parameters in this case) and finds linear combinations of these to produce new variables or components (PCs) that are un-correlated. PCA ensures that the components are measuring different 'dimensions' or variability in the data. Because of multicollinearity, with the increase in variance accounted for by successive axes, relatively fewer axes are required to represent the majority of variation in the environmental data set. Figure 3.3 elegantly summarises this phenomenon using data derived from this thesis.

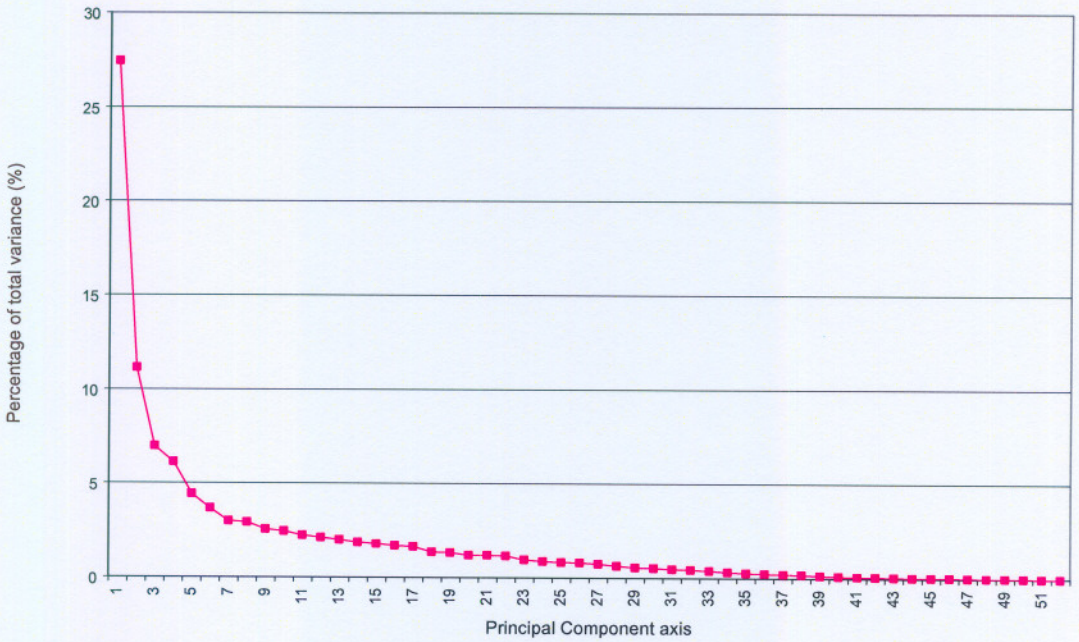

Figure 3.3 Plot illustrating the rapid decline in the percentage of total environmental variance accounted for by successive PCA axes.

3.7.2.2 Redundancy Analysis – for examination of algae/environment relationships

Due to the nature of the data (relatively short environmental axes - likely to represent **linear** relationships between algae and the environment), Jongman *et al.*, 1987, advise Redundancy Analysis (RDA) as the preferred technique to perform the multivariate ordinations on log transformed algal data.

The aim of the ordination analyses was to arrange lake sites (represented by points on an ordination diagram) such that points that are close together correspond with sites that are similar in algal composition. Obviously sites that are further apart on ordinations are more dissimilar in algal composition. The ordination of algae in RDA methods results in an ordination diagram where the arrows correspond with the rate of change in abundance of respective algae over ordinated lake sites. The interpretation of these diagrams is described in section 3.7.2.4.

3.7.2.3 Kriging of environmental and algal data

At each ordinated lake site there are corresponding quantitative values available for environmental variables. In an effort to develop predictive models relating algae to the environment, kriging was employed. Kriging essentially takes quantitative values, for environmental variables of interest, at respectively ordinated lake sites, and creates "best fit" contours of environmental similarity over the ordination diagram. In other words, as the values for an environmental variable change at respective sites over an ordination diagram, contours are created to represent this trend surface (in rate, value and direction) over the ordination. In the same way as kriging was used to fit **environmental** trend-surfaces across the ordination diagrams, an **algal** abundance trend-surface could also be fitted. This approach provides a more general indication than the algae/environment biplot (e.g. Fig. 3.4) as to where algal abundances could be expected to be highest and under what environmental conditions. Kriging therefore provided semi-quantitative models relating algae to key environmental elements. Results for this are summarised in Table 4.1 (summary by genera of algae/environment relations).

3.7.2.4 Interpretation of ordination figures

As many of the results were dependent on this type of analysis, a brief summary to be used in the interpretation of all ordination figures is presented here. To reiterate, the aim of ordination analyses is to arrange lakes sites (represented by points on an ordination diagram) such that points that are close together correspond with sites that are similar in algal composition. As the algal composition changes across lake sites (and hence the ordination diagram), the position of arrows representing respective algae also changes. This corresponds with the rate of change in abundance of algae over ordinated lake sites. As an example Figure 3.4 (taken from this study) illustrates many of the features of ordination diagrams. The interpretation of this diagram is described below.

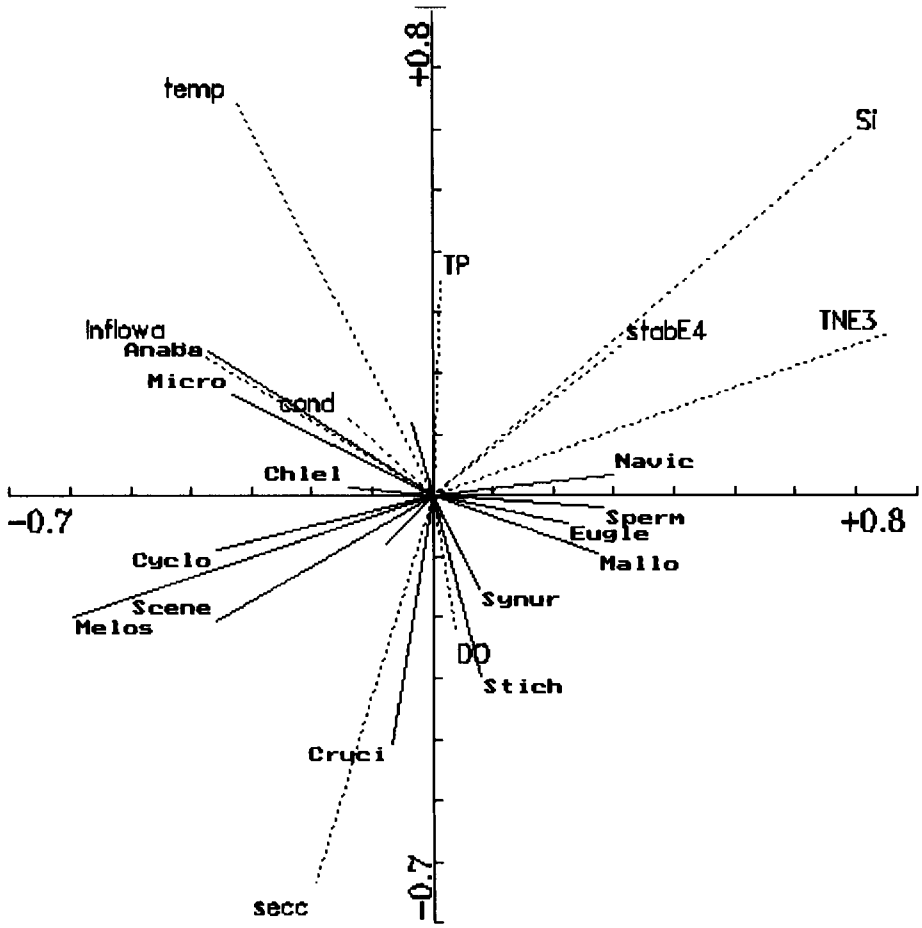

Figure 3.4 Biplot ordination of algae and key environmental variables (summary of Fig. 5.1a & b). Algal genera are: Anaba=*Anabaena*, Micro=*Microcystis*, Navic=*Navicula*, Sperm=*Spermatozopsis*, Eugle=*Euglena*, Mallo=*Mallomonas*, Synur=*Synura*, Stich=*Stichococcus*, Cruci=*Crucigena*, Scene=*Scenedesmus*, Melos=*Melosira*, Cyclo=*Cyclotella* and Chlel=*Chlorella*. Environmental variables are: TP=Total Phosphorus, Si=Silicon, stabE4=stability, TNE3=Total Inorganic Nitrogen, DO=%Dissolved Oxygen, cond=conductivity, Inflowa=inflow and temp=temperature.

In the interpretation of RDA ordination figures, algal genera should strictly be represented as arrows as the rationale underlying the technique assumes that there is a linear change of algal abundance across the ordination diagram. However, for visual clarity, genera in some ordination figures were represented by dots (the position of dots representing the ends of the respective arrows). Algae grouped closely together on an ordination diagram are more similar in their response to their environment compared to algae further apart on an ordination diagram. So, for example in Figure 3.4, the algae *Microcystis* (Micro) and *Anabaena* (Anaba) are grouped together in the upper left quadrant of the ordination diagram. They therefore have similar environmental requirements and, as the ordination diagram shows,

generally tend to favour warmer temperatures and higher inflows compared to the other algae displayed on the ordination. Environmental (and algal) variables are positively correlated (e.g. temperature and inflow) if their arrows subtend a small angle; uncorrelated (independent) if their arrows are at 90° (e.g. temperature and total inorganic nitrogen); and negatively correlated if their arrows are in opposite directions (e.g. temperature and % dissolved oxygen). Environmental variables with the longest arrows relative to an axis have the greatest influence on that axis (e.g. temperature and inflow compared to conductivity). Successive ordination axes account for successively lower proportions of algal variability i.e. the first axis accounts for more of the total algal variability compared to the second axis, and so on. Algae with long arrows (distance from the origin of ordinations) have a greater proportion of their variability explained by the ordination than do algae closer to the origin (e.g. *Anabaena* & *Microcystis* compared to say *Chlorella*). They are consequently responding more strongly (rate-of-change) to the measured environmental variables. The order of response of genera along any environmental gradient is obtained by dropping a perpendicular bisector from the end of the arrows (points), representing algal genera, to the extended line of an environmental arrow.

3.7.3 Methods used in determining relationships between treatment costs and water quality

3.7.3.1 The linear regression model

In order to identify the most important variables (environmental and algal) affecting water treatment costs in water treatment works (WW), a multiple linear regression model (expressed in equation 1) was estimated. This model used all variables that were identified as being significantly correlated with water treatment cost for each of the selected WW systems (i.e., Hazelmere, Durban Heights, DV Harris and Wiggins).

$$Cost = \alpha_0 + \alpha_1 X_1 + \alpha_2 X_2 + \ldots\ldots\ldots\ldots\ldots \alpha_n X_n \qquad (1)$$

where;

Cost = Water treatment cost in 1996 Rands per mega litre (R/Mℓ) terms

X_i = Variables (determinants) selected as affecting water treatment costs

α_i = Estimated regression coefficients

3.7.3.2 Principal Components Analysis (PCA)

Initial attempts to simply conduct multiple regression type modelling of treatment cost against water quality determinants proved unsuccessful. The *t*-values of the original model estimates (before removing the multicollinearity problem) were often extremely low for certain variables. For example, it was anticipated that *t*-values for suspended solids, total organic carbon and total aluminium in the Hazelmere system should have been higher (more significant) than they were. Likewise the signs of the coefficients of these variables do not conform to *a priori* relationships understood to exist with treatment costs. For example in the Hazelmere system, suspended solids would be expected to positively influence treatment costs, indicating that as they increased in the raw water, so too would polymeric coagulant demand, and hence costs. This lack of adherence to *a priori* expectations is a classic symptom of the multicollinearity anticipated in these types of models (Gujarati, 1988). The implication was that the correlation between individual variables and treatment cost could not be robustly estimated. In order to solve this problem, principal component analysis (PCA) was applied to solve the multicollinearity before valid multiple regression analyses could proceed.

CHAPTER 4: WATER QUALITY AND ENVIRONMENTAL CONDITIONS OF THE STUDIED LAKES

4.1 Introduction

This chapter investigates and illustrates:

- how the dimensionality of the environmental data was reduced using Principal Components Analysis to produce a handful of key environmental variables (to use in further analyses to establish algae environment relationships);
- the dominant parameters accounting for variation in the environmental data set;
- how Lake Shongweni was identified as an extreme outlier along the eutrophication gradient (and was subsequently dropped from further analyses);
- how lakes became progressively and successively enriched with 'pollutants' and nutrients down the catchment; and
- how PCA could be used to objectively identify lakes with similar physico-chemical properties (for use in planning and management strategies).

Summary descriptive statistics for respective lakes have already been presented in Tables 2.1 and 2.2. Highlights of this data, as they pertain to respective lakes, have also been narrated in Chapter 2.

4.2 Results of Principal Components Analysis of environmental conditions in all lakes studied

Preliminary investigations of the data indicated that many of the 52 environmental variables assembled for this study were highly inter-correlated (both positively and negatively - Table 4.1 - correlation matrix), or in other words were multi-collinear. As indicated in Chapter 3, this is not unusual with environmental data of this nature (e.g. Jongman *et al.*, 1987) and, as was also noted in the economic models developed later (Chapter 6), causes problems for multiple regression analysis. Principal Components Analysis (PCA) was used to address the multicollinearity issue.

Table 4.1 Correlation matrix for all environmental variables from all lakes

Note: Environmental variables as per Table 2.2, Correlations significant at P < 0.05 - marked *

	STABE4	WNSP	WNDIR	SHRS	TPLDE-1	SRPLD	TNLDE-3	TNE3	TEMP	CHL	SECC	PH	TURB	COND	TAL	ALKAL	HARD	CA	MG	NA	K	FE	MN	SI	NO3	NO2	NH4
STABE4	1																										
WNSP	.32*	1																									
WNDIR	0.01	.45*	1																								
SHRS	-.18*	-.15*	-.20*	1																							
TPLDE-1	0.04	-0.01	0	-0.09	1																						
SRPLD	-0.04	0.07	0.04	-.10*	.91*	1																					
TNLDE-3	0.05	-.09*	-0.03	-0.08	.86*	.78*	1																				
TNE3	-0.02	-0.01	.13*	.10*	-.11*	-.14*	0	1																			
TEMP	.63*	.25*	0	-.18*	.28*	.22*	.27*	-.17*	1																		
CHL	0.04	0.07	.16*	-0.04	0.02	0.07	0.01	.10*	0.05	1																	
SECC	-.10*	-0.06	-0.07	.10*	0.08	.11*	0.05	-.22*	-.13*	-.20*	1																
PH	.13*	.23*	.17*	-0.02	.14*	.19*	0.07	-.16*	.21*	.28*	0	1															
TURB	.14*	0.06	0.06	-.10*	-0.02	-0.06	0.02	.31*	.18*	.34*	-.56*	0.05	1														
COND	0.09	.29*	.36*	-0.03	-0.04	0.04	-.11*	.15*	0.05	.61*	-.13*	.47*	.19*	1													
TAL	.14*	0.09	0	-0.09	-0.04	-.12*	-0.02	.27*	.18*	0.02	-.53*	-0.06	.80*	-0.05	1												
ALKAL	0.02	.31*	.37*	-0.03	-0.01	.10*	-.10*	0.05	0.05	.58*	-0.08	.50*	.14*	.96*	-0.08	1											
HARD	0.01	.36*	.38*	-0.07	0.08	.24*	-0.02	.11*	.12*	.51*	-0.08	.49*	.15*	.86*	-0.05	.92*	1										
CA	-0.03	.30*	.34*	-0.07	.14*	.29*	0.03	0.05	.10*	.53*	0	.49*	.11*	.84*	-.10*	.91*	.98*	1									
MG	0.05	.41*	.41*	-0.07	0.03	.17*	-0.07	.17*	.13*	.47*	-.16*	.47*	.20*	.85*	0.03	.90*	.98*	.92*	1								
NA	0.06	.23*	.34*	-0.03	-0.05	0.02	-.11*	.19*	0.01	.63*	-.14*	.42*	.21*	.97*	-0.04	.92*	.81*	.79*	.80*	1							
K	0.05	.22*	.33*	-0.03	-0.04	0.03	-.11*	.21*	0.01	.66*	-.16*	.41*	.25*	.94*	-0.02	.90*	.82*	.81*	.79*	.95*	1						
FE	0.08	.11*	0.08	-0.06	-.15*	-.17*	-.14*	.45*	0.05	.16*	-.62*	-0.05	.69*	.11*	.74*	0.05	0.06	-0.03	.15*	.14*	.15*	1					
MN	-.14*	0.06	.24*	0.06	-.11*	-0.06	-.12*	.46*	-.17*	.37*	-.21*	0.07	.23*	.54*	0.08	.53*	.52*	.48*	.55*	.56*	.57*	.35*	1				
SI	.17*	.13*	.10*	-0.03	-.17*	-.22*	-0.08	.46*	0.05	0.04	-.47*	-.19*	.45*	-0.03	.48*	-.12*	-.13*	-.27*	0.02	0	-0.03	.63*	.21*	1			
NO3	0	0.01	.11*	0.08	-.10*	-.13*	0.02	.95*	-.12*	0.09	-.26*	-.16*	.34*	.10*	.31*	-0.02	0.08	0.01	.14*	.13*	.15*	.46*	.38*	.47*	1		
NO2	-0.01	0.02	0.01	-0.01	0.02	0	0.01	.11*	0.02	-0.01	0.02	-0.03	0.01	-0.03	0.05	-0.03	-0.03	-0.03	-0.03	-0.03	-0.03	0.01	0	0	0.05	1	
NH4	-0.05	-0.04	.13*	.12*	-0.08	-0.08	-0.06	.60*	-.22*	0.08	0	-0.06	0.04	.23*	0.02	.21*	.15*	.14*	.16*	.26*	.25*	.18*	.44*	.18*	.34*	.21*	1
CL	.10*	.35*	.40*	-0.04	-0.05	0.05	-.12*	.21*	0.07	.58*	-.18*	.46*	.21*	.98*	-0.02	.95*	.88*	.84*	.89*	.95*	.93*	.16*	.57*	0.05	.16*	-0.04	.25*
F	-0.01	0.02	0.03	-.13*	.21*	.34*	.20*	.19*	.13*	-0.02	-.10*	0.02	.17*	0.08	.12*	0.07	.21*	.18*	.24*	0.06	0.07	.18*	.09*	.15*	.22*	-0.02	-0.01
SO4	0.07	.20*	.30*	-0.03	-0.06	0	-.11*	.21*	0	.63*	-.14*	.41*	.21*	.97*	-0.05	.91*	.79*	.78*	.77*	.95*	.95*	.12*	.56*	-0.05	.15*	-0.02	.27*
TP	0.06	.10*	.16*	-0.03	-0.06	-0.05	-.10*	.16*	0.01	.60*	-.24*	.28*	.39*	.62*	.15*	.58*	.48*	.47*	.47*	.61*	.67*	.19*	.35*	0.06	.13*	-0.03	.17*
SRP	0.02	.14*	.22*	-0.04	-0.09	-0.04	-.13*	.24*	-0.05	.43*	-.15*	.29*	.20*	.66*	0.04	.62*	.54*	.51*	.55*	.64*	.67*	.14*	.39*	0.07	.20*	-0.02	.22*
TN:TP	-.11*	-0.02	0.01	0.08	-0.09	-0.06	0.04	.43*	-.20*	-.20*	.18*	-.28*	-.16*	-.24*	-.11*	-.28*	-.25*	-.28*	-.21*	-.22*	-.25*	0	-0.02	.31*	.44*	.13*	.18*
TN:SRP	-0.02	-0.08	-0.04	.12*	-0.06	-.11*	.15*	.40*	-.09*	-.19*	0.08	-.28*	0	-.32*	0.03	-.35*	-.32*	-.33*	-.30*	-.29*	-.30*	0.01	-.10*	.25*	.40*	.12*	.18*
TN:SI	-.15*	0.01	.17*	0.08	-0.01	0.02	0.02	.61*	-.19*	0.05	.10*	0.03	0.02	.15*	-0.01	.16*	.28*	.31*	.24*	.16*	.21*	0.05	.27*	-.18*	.57*	.10*	.41*
TP:SI	0.01	.10*	.16*	-0.03	-0.03	0	-.11*	-0.01	0	.50*	-0.08	.37*	.20*	.60*	-0.01	.60*	.52*	.54*	.47*	.57*	.63*	-0.01	.23*	-.24*	-0.04	-0.03	0.08
SRP:SI	-0.01	.15*	.20*	-0.03	-0.04	0.02	-.12*	0.05	-0.03	.35*	0.01	.36*	0.05	.64*	-0.09	.63*	.57*	.57*	.54*	.60*	.62*	-0.05	.26*	-.21*	0.02	-0.02	.12*
TDS	.11*	.28*	.34*	-0.04	-0.04	0.04	-.10*	.17*	0.08	.61*	-.19*	.44*	.25*	.96*	0.01	.94*	.85*	.82*	.85*	.94*	.92*	.17*	.55*	0.03	.13*	-0.04	.21*
SS	.12*	0.07	0.08	-0.07	0.01	-0.02	0.02	.25*	.16*	.46*	-.47*	.10*	.87*	.28*	.63*	.24*	.24*	.22*	.25*	.31*	.37*	.51*	.25*	.32*	.27*	0.05	0.07
CU	-.11*	-.10*	-0.06	0.05	-0.07	-0.07	-.10*	-0.04	-.12*	0.01	0.01	-0.02	-0.05	-0.05	-0.06	-0.06	-0.05	-0.07	-0.04	-.09*	-0.04	-0.02	-0.04	-0.01	-.12*	-0.03	0.02	-0.03
ZN	-0.03	0.03	0.04	-0.05	-0.02	0.03	-0.01	.26*	-0.01	0.07	-.11*	-0.09	0.04	0.04	-0.02	0	0.06	0.04	0.08	0.03	0.05	0.06	.16*	.14*	.27*	0.01	.11*
PB	0.06	0.08	.12*	-0.01	-0.08	-.11*	-0.07	.22*	-0.02	.18*	-.12*	0.06	.22*	.25*	.15*	.23*	.19*	.16*	.22*	.27*	.27*	.14*	.25*	.21*	.22*	-0.01	.10*
CD	-.10*	-.11*	-0.07	-0.01	-0.03	-0.05	-0.03	-0.04	0.07	-0.02	0.01	-.13*	-0.05	-0.01	-0.07	-0.02	-0.06	-0.04	-0.07	0	-0.02	-0.06	-0.04	-0.06	-0.05	.18*	-0.02
CR	0.01	-.14*	-.12*	-0.07	0.06	0	0.05	.14*	0.02	-0.02	-.31*	-.18*	.37*	-0.08	.37*	-.10*	-.09*	-.12*	-0.06	-0.06	-0.05	.40*	0.07	.27*	.16*	-0.02	0.01
HG	0	-.17*	-0.06	-0.01	-.16*	-.21*	-.13*	.27*	-.13*	.23*	-.20*	-.29*	.23*	0.02	.22*	-0.04	-0.07	-0.08	-0.05	0.04	0.05	.22*	.13*	.24*	.25*	0.06	.17*
SE	0.04	-.18*	-.18*	0.03	-.16*	-.23*	-.14*	0.02	0	-0.06	-.10*	-.18*	.09*	-.11*	.09*	-.17*	-.24*	-.25*	-.21*	-.09*	-.10*	.10*	-0.06	.14*	0.01	-0.02	0.02
NI	.10*	0.04	.09*	-0.06	-0.04	0	-.10*	.21*	0.03	.49*	-.35*	.19*	.49*	.55*	.30*	.50*	.45*	.45*	.43*	.56*	.60*	.37*	.37*	.11*	.17*	0.03	.20*
TOC	.19*	.33*	.31*	-0.02	-0.05	0.02	-.13*	.13*	.11*	.57*	-.24*	.38*	.28*	.88*	0.06	.84*	.77*	.73*	.79*	.88*	.87*	.21*	.48*	0.09	.11*	-0.02	.15*
BOD	-0.01	0.01	0.02	0	-0.04	-0.02	-0.04	.34*	-0.01	0.08	-.15*	0.01	0.09	.22*	0.04	.14*	.20*	.17*	.22*	.20*	.24*	.15*	.23*	0.09	.38*	0.01	0.06
DO	0.04	.13*	-0.04	-0.01	-0.02	-0.01	-0.05	-.25*	-.23*	-0.07	.18*	.23*	-.23*	-0.07	-.22*	-0.07	-.14*	-.12*	-.16*	-0.07	-0.09	-.24*	-.32*	-.17*	-.26*	0.04	-.09*
%DO	.37*	.15*	-.10*	-0.07	.10*	0.04	0.08	-.33*	.18*	-0.07	.18*	.26*	-.19*	-.12*	-.16*	-.13*	-.20*	-.17*	-.23*	-.13*	-.14*	-.28*	-.44*	-.19*	-.32*	0.03	-.19*
TKN	0	-0.05	0.04	-0.01	0.05	0.02	0.04	.13*	-0.02	.28*	-.13*	0.06	.14*	.26*	0.03	.22*	.15*	.14*	.15*	.30*	.29*	.11*	.22*	.14*	.09*	0	.16*
PR FLUSH	0.05	0.06	0.1	0	-0.02	-0.01	-0.04	-0.06	0	.22*	-0.03	.19*	0	.43*	-0.05	.38*	.22*	.21*	.22*	.42*	.31*	-0.05	0.02	-0.03	-0.07	-0.01	0.01

	CL	F	SO4	TP	SRP	TN:TP	TN:SRP	TN:SI	TP:SI	SRP:SI	TDS	SS	CU	ZN	PB	CD	CR	HG	SE	NI	TOC	BOD	DO	%DO	TKN	PR FLUSH
CL	1																									
F	.13*	1																								
SO4	.94*	0.07	1																							
TP	.59*	-0.02	.63*	1																						
SRP	.66*	0.05	.66*	.83*	1																					
TN:TP	-.20*	0.02	-.24*	-.34*	-.21*	1																				
TN:SRP	-.30*	-0.03	-.30*	-.27*	-.37*	.61*	1																			
TN:SI	.18*	.10*	.18*	0.06	.12*	.25*	.27*	1																		
TP:SI	.56*	-0.08	.60*	.91*	.79*	-.38*	-.32*	.16*	1																	
SRP:SI	.61*	-0.02	.61*	.73*	.92*	-.26*	-.38*	.20*	.83*	1																
TDS	.96*	0.07	.94*	.63*	.67*	-.25*	-.32*	.13*	.59*	.62*	1															
SS	.29*	.10*	.31*	.51*	.27*	-.19*	-0.03	0.03	.33*	.14*	.33*	1														
CU	-0.06	-0.07	-0.02	-0.01	-0.04	-0.05	-0.04	0.05	0.02	-0.02	-0.05	-0.03	1													
ZN	0.07	.16*	0.09	0.08	.12*	0.08	0.03	.11*	0.01	0.03	0.06	0.06	0.07	1												
PB	.26*	0.03	.27*	.20*	.22*	-0.07	0.03	0.05	.10*	.13*	.24*	.23*	0.02	.17*	1											
CD	-0.04	-0.04	0.02	-0.03	-0.01	0.01	0.02	-0.04	-0.03	-0.01	-0.01	-0.02	.12*	0.07	.10*	1										
CR	-0.08	.17*	-0.05	0.03	-0.02	-0.05	0.02	-0.05	-0.06	-.10*	-0.02	.27*	0	.10*	0.06	.21*	1									
HG	0	0.01	0.08	0.07	0.05	0.07	.16*	0.06	-0.03	-0.04	0.04	.20*	0.01	.17*	.19*	-0.03	.12*	1								
SE	-.14*	.10*	-0.07	-0.01	-0.04	0.02	0.03	-.14*	-0.07	-.11*	-.11*	0.03	0.03	-0.03	0	.12*	0.09	.32*	1							
NI	.53*	.11*	.60*	.47*	.37*	-.25*	-.21*	0.09	.38*	.29*	.57*	.54*	0.01	.11*	.24*	-0.05	.25*	.20*	0.04	1						
TOC	.89*	.09*	.85*	.60*	.60*	-.29*	-.32*	0.07	.53*	.54*	.89*	.37*	-0.06	0.05	.22*	-0.05	-0.03	0.04	-.12*	.55*	1					
BOD	.25*	.39*	.27*	.16*	.24*	-0.06	-.09*	.20*	0.09	.13*	.22*	.11*	-0.02	.35*	.14*	0.01	0.05	0.08	-0.01	.19*	.25*	1				
DO	-.10*	-.16*	-.10*	-.10*	-.13*	0.01	-0.04	-0.06	0	-0.03	-.13*	-.23*	0.01	-.23*	-0.08	-0.03	-.20*	-.15*	-0.01	-.11*	-.10*	-.17*	1			
%DO	-.16*	-.17*	-.14*	-.11*	-.18*	-.09*	-0.05	-.17*	-0.02	-0.07	-.17*	-.18*	-0.05	-.25*	-.10*	-0.06	-.18*	-.16*	0.02	-.11*	-.11*	-.19*	.87*	1		
TKN	.25*	0	.28*	.27*	.22*	0	0.01	0.02	.19*	.15*	.29*	.17*	-0.06	0.06	0.03	-0.05	.25*	.13*	-0.03	.23*	.25*	0.02	-0.07	-0.05	1	
PR FLUSH	.35*	-0.03	.39*	.20*	.32*	-0.08	-.11*	-0.05	.21*	.32*	.38*	0.01	-0.02	-0.03	.23*	.24*	-0.05	0.04	-0.01	.15*	.39*	.33*	.16*	.15*	0.09	1

Initial analyses included data from all eight lakes identified in the study. Of the 52 potential PC's only the first 13 were worthy of interpretation (i.e. having eigen-values greater than unity (Manly, 1991)). They accounted for some 77% of the variation in the environmental data with the first 5 accounting for 56% of the total variability (Table 4.2).

An examination of the variable loadings on these PCs (Table 4.3) revealed that PC_1 could be described as a "conductivity" axis with many of the 'salts' that contribute to high conductivities in natural waters having high loadings on this axis - in this case Ca, Mg, hardness, TOC, alkalinity, SO_4, Na, K, TDS, conductivity, and Cl. Clearly the use of the term 'salts' here is rather loose as not all the variables are strictly 'salts'. The other key PCs could similarly be colloquially described by the types of environmental variables which had high loadings on the respective axes (Table 4.3) and were respectively PC_2 – 'turbidity', PC_3 – 'N:P ratio', PC_4 – 'nutrient loads' and PC_5 – 'stability'.

Table 4.2 Key results from a standardised Principal Components Analysis (performed on the full (all lakes) environmental correlation matrix) for the first 13 axes. The first 5 axes are named

AXIS	EIGENVALUE	PERCENT OF TOTAL	CUMULATIVE PERCENT	Axis name
1	14.285	27.47	27.47	'Conductivity'
2	5.798	11.15	38.62	'Turbidity'
3	3.624	6.97	45.59	'N:P ratio'
4	3.186	6.13	51.72	'Nutrient loads'
5	2.309	4.44	56.16	'Stability'
6	1.917	3.69	59.85	
7	1.572	3.02	62.87	
8	1.526	2.94	65.80	
9	1.339	2.58	68.38	
10	1.291	2.48	70.86	
11	1.174	2.26	73.12	
12	1.107	2.13	75.25	
13	1.049	2.02	77.26	

To further illustrate the multicollinearity effect already highlighted, and how variables with high loadings on a particular PC axis have high correlations with each other, consideration of the correlation matrix for all environmental variables in all the lakes studied (Table 4.1) was useful. The PC_1 axis could be confirmed as a 'Conductivity' axis on the basis of the following variables having high correlations with Conductivity; Alkalinity (0.96), Hardness (0.86), Calcium (0.84), Magnesium (0.85), Sodium (0.97), Potassium (0.94), Sulphate (0.97), Chloride (0.98), Total Dissolved Solids (0.96) and Total Organic Carbon (0.88). Interestingly, although Total Organic Carbon is not typically recognised as affecting Conductivity, its variability appears to be co-varying with this

variable. Clearly such high correlations indicate close relationships among these variables, and how a single variable (in this instance Conductivity), could usefully capture this aspect of the variation, or trend, in the data. Successive PC axes capture different and unique aspects of variation in the data set.

Table 4.3 Ranked environmental variable loadings on the first five PCs/axes from PCA analysis of data from all lakes. Variables with the highest loadings on (and therefore association with) respective axes are *highlighted.* See Table 2.2 for an explanation of codes used for environmental variables

Variable	PC_1	Variable	PC_2	Variable	PC_3	Variable	PC_4	Variable	PC_5
Conductivity		*Turbidity*		*N:P Ratio*		*Nutrient loads*		*Stability*	
TN/SRP	-0.09	***Fe***	***-0.31***	***temp***	***-0.31***	Se	-0.17	***WnSp***	***-0.42***
TN/TP	-0.08	***NO3***	***-0.30***	***turb***	***-0.27***	TP/Si	-0.14	***stabE4***	***-0.34***
secc	-0.07	***TNE3***	***-0.30***	***TAl***	***-0.27***	TP	-0.14	***%DO***	***-0.30***
%DO	-0.06	***Si***	***-0.30***	***SS***	***-0.24***	DO	-0.12	***DO***	***-0.26***
DO	-0.04	***TAl***	***-0.27***	***stabE4***	***-0.23***	SRP	-0.10	***WnDir***	***-0.24***
Se	-0.04	***turb***	***-0.26***	TPLDE-1	-0.18	Hg	-0.10	***Si***	***-0.21***
TNLDE-3	-0.03	***SS***	***-0.21***	Fe	-0.16	SRP/Si	-0.09	***TN/TP***	***-0.20***
SHrs	-0.02	Hg	-0.19	Cr	-0.15	Cu	-0.08	TN/SRP	-0.17
Cu	-0.01	Cr	-0.19	TNLDE-3	-0.14	%DO	-0.08	Temp	-0.16
Cd	-0.01	TN/SRP	-0.16	%DO	-0.14	SHrs	-0.06	PH	-0.13
TPLDE-1	-0.01	NH4	-0.13	SRPLD	-0.14	Ni	-0.05	NO3	-0.09
NO2	-0.01	Mn	-0.13	Ni	-0.11	chl	-0.04	Mg	-0.09
Cr	0.00	TN/TP	-0.13	pH	-0.10	Cd	-0.04	TNE3	-0.09
Si	0.01	Zn	-0.11	Si	-0.08	Pb	-0.03	Cl	-0.09
Hg	0.02	Ni	-0.11	WnSp	-0.08	SO4	-0.03	TOC	-0.08
SRPLD	0.02	BOD	-0.10	TP	-0.07	Fe	-0.03	Fe	-0.07
temp	0.02	Pb	-0.10	chl	-0.06	SS	-0.03	cond	-0.05
stabE4	0.02	Se	-0.09	F	-0.06	TOC	-0.03	NO2	-0.05
TAl	0.03	TN/Si	-0.08	Se	-0.04	TAl	-0.03	Pb	-0.04
Zn	0.03	F	-0.08	TOC	-0.04	Na	-0.02	NH4	-0.04
F	0.04	TKN	-0.06	TP/Si	-0.03	K	-0.02	hardness	-0.04
NO3	0.05	NO2	-0.03	TKN	-0.01	turb	-0.02	TAl	-0.04
TN/Si	0.06	TP	-0.02	DO	-0.01	cond	-0.01	Na	-0.04
Fe	0.06	SHrs	-0.01	TDS	0.01	TDS	-0.01	TDS	-0.04
TNE3	0.07	chl	-0.01	Pb	0.01	Si	-0.01	alkal	-0.04
NH4	0.07	Cd	-0.01	Cd	0.02	TKN	0.00	TN/Si	-0.03
BOD	0.07	stabE4	0.00	Mg	0.02	stabE4	0.00	K	0.00
TKN	0.08	SRP	0.00	hardness	0.03	alkal	0.01	SO4	0.00
Pb	0.08	Cu	0.01	Ca	0.03	Cl	0.02	secc	0.00
WnSp	0.08	WnDir	0.01	alkal	0.03	Cr	0.02	turb	0.02
turb	0.09	TOC	0.02	K	0.04	pH	0.04	Ca	0.02
WnDir	0.10	K	0.03	SRP	0.04	NO2	0.04	Mn	0.05
SS	0.12	TDS	0.03	Hg	0.04	Mn	0.04	SS	0.06
pH	0.12	SO4	0.03	cond	0.04	NH4	0.05	F	0.06
Mn	0.16	Cl	0.04	Cl	0.04	secc	0.06	Se	0.07
Ni	0.17	temp	0.04	Na	0.05	WnSp	0.06	BOD	0.07
chl	0.18	Na	0.04	NO2	0.05	WnDir	0.09	Ni	0.08
TP/Si	0.18	Mg	0.04	Cu	0.05	BOD	0.09	chl	0.08
SRP/Si	0.18	WnSp	0.05	WnDir	0.05	Zn	0.09	Hg	0.08
TP	0.20	TNLDE-3	0.05	SRP/Si	0.05	Mg	0.11	TKN	0.08
SRP	0.20	Cond	0.06	SO4	0.06	hardness	0.12	SHrs	0.08
Ca	***0.23***	hardness	0.09	BOD	0.07	Ca	0.13	Zn	0.10
Mg	***0.23***	TP/Si	0.09	Zn	0.08	TN/SRP	0.13	SRP	0.12
hardness	***0.23***	alkal	0.10	Mn	0.15	TN/TP	0.14	SRP/Si	0.12
TOC	***0.23***	TPLDE-1	0.10	SHrs	0.15	temp	0.15	TNLDE-3	0.13
alkal	***0.25***	SRP/Si	0.10	TN/SRP	0.17	TNE3	0.15	Cd	0.14
SO4	***0.25***	Ca	0.12	NO3	0.18	NO3	0.16	Cu	0.14
Na	***0.25***	SRPLD	0.13	***secc***	***0.22***	TN/Si	0.18	TP	0.15
K	***0.25***	PH	0.17	***TNE3***	***0.23***	***F***	***0.26***	SRPLD	0.15
TDS	***0.25***	DO	0.17	***NH4***	***0.24***	***TPLDE-1***	***0.43***	TPLDE-1	0.16
cond	***0.25***	%DO	0.19	***TN/TP***	***0.26***	***TNLDE-3***	***0.43***	TP/Si	0.17
Cl	***0.25***	***Secc***	***0.22***	***TN/Si***	***0.29***	***SRPLD***	***0.45***	Cr	0.18

4.3 Detection of outliers (extremes) in the data

As noted by numerous authors (e.g. Gauch *et al.*, 1977; Reyment, 1980) if there are outliers in the data set then it is best to remove these prior to data analysis, this especially so where ordination is used as an analytical technique (Gauch, 1982). Outliers probably represent extreme environmental conditions. To check for the presence of outliers, firstly the physico-chemical and then the biological (algal) similarities between the 8 lakes studied were considered. The more dissimilar sites were, the more likely they were to be outliers and hence likely to cloud the central focus of this study, i.e. in establishing the key environmental variables influencing problematic algae.

To determine the physico-chemical similarity between the various lakes a plot of lake sites along the first two PCA axes (Fig. 4.1) reveals how different Shongweni is from all other lakes (Shongweni located apart from the other clustered lake sites).

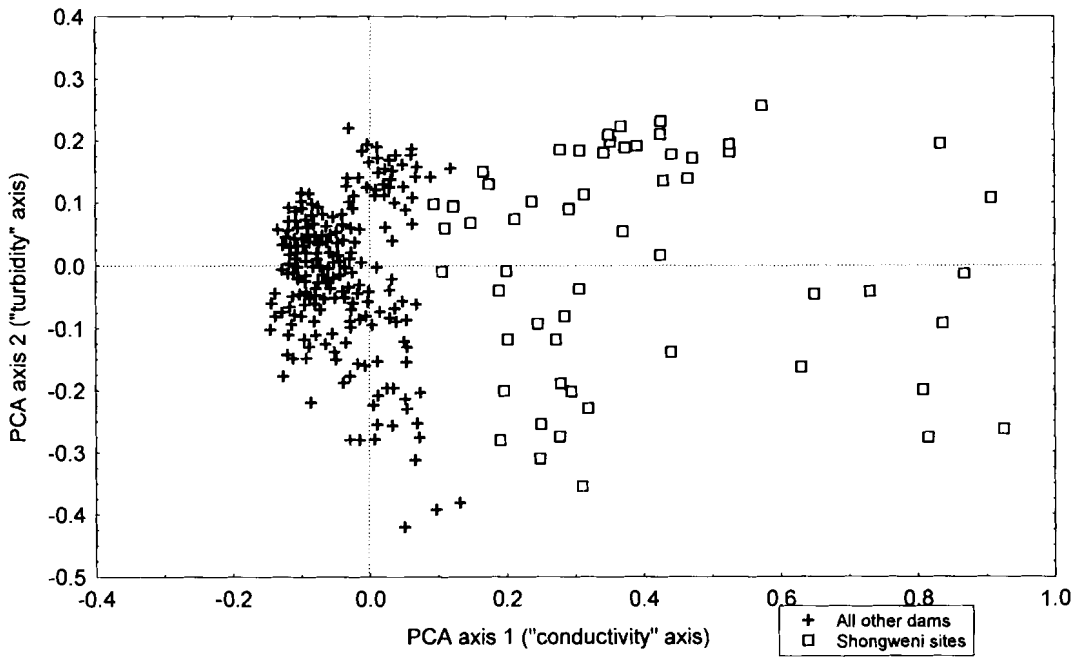

Figure 4.1 PCA ordination plot of Shongweni and all other lakes. (Note: lake sites closely clustered on the ordination diagram have more similar physico-chemical conditions compared to sites further apart on the ordination diagram).

Examining Figure 4.2 (a-f) provided some of the explanation for this difference between Shongweni and all other lakes, with the former characterised by variables (dissolved 'salts') contributing to high conductivities. Examination of the algal (or biological) pattern of differences between Shongweni and the other lakes further emphasised the difference between this lake and all others in the study. Ordination of sites on the basis of their algal composition using Correspondence Analysis (CA) produced an indication of the separation of lake sites (Fig. 4.3).

As in all ordination diagrams, sites that are closer together on the diagram are more similar in terms of their underlying algal composition and abundance compared to sites that are further apart. For example then, Nagle (NAG) and Hazelmere (HAZ) have a greater similarity in their algal community to Shongweni (SH) than do Henley (HEN) and Nungwane (NUN).

It was of interest to note the specific algal genera accounting for the difference, or separation, between Shongweni and all other lakes noted in the Correspondence Analysis ordination of lakes seen in Figure 4.3. This was investigated by examining an ordination of genera over all the sites ordinated in the CA (Fig. 4.4) (i.e. which algal genera dominate the respective areas of the ordination of lake sites in Fig. 4.3). Algal abundances, which distinguish between Shongweni and other lakes, were then graphically investigated (Figs. 4.5 & 4.6 - algal abundance and proportional composition in respective lakes).

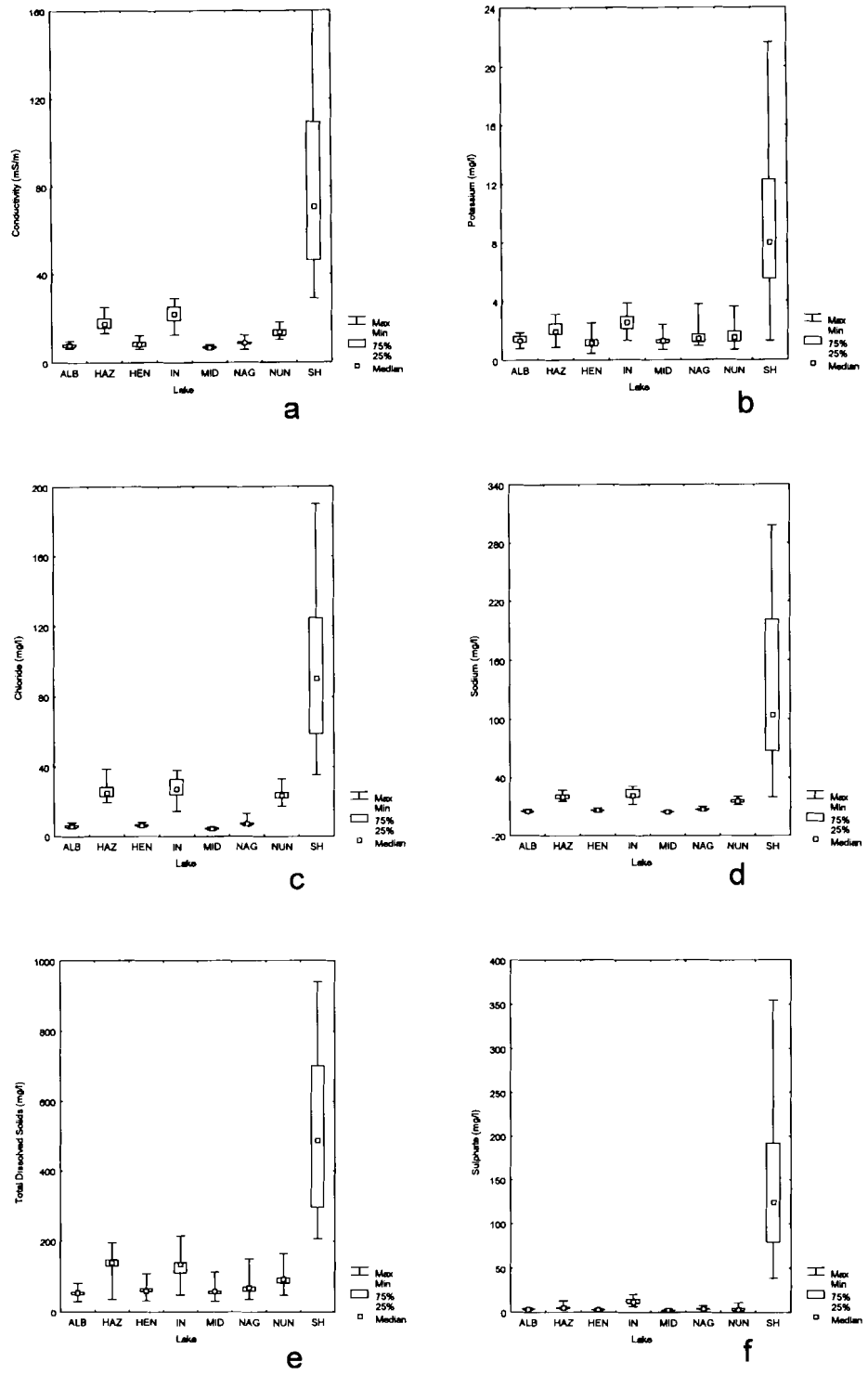

Figure 4.2 Variation of environmental variables with high loading on the first PC axis separating Shongweni sites from other lakes respectively conductivity (a), potassium (b), chloride (c), sodium (d), total dissolved solids (e) and sulphate (f).*

*Note the exceptionally high values of these parameters in Shongweni compared to all other lakes.

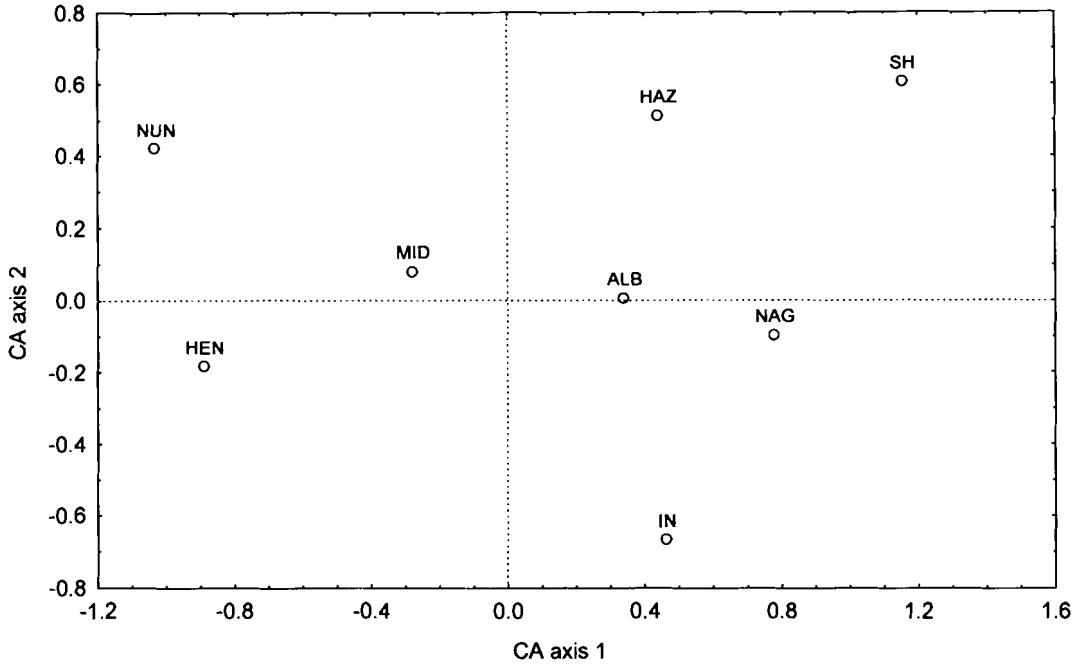

Figure 4.3 Correspondence analysis ordination of biological (algal) data from all lake sites (average site scores (centroids) for lakes have been used as points on the ordination diagram as individual sites were omitted for clarity).

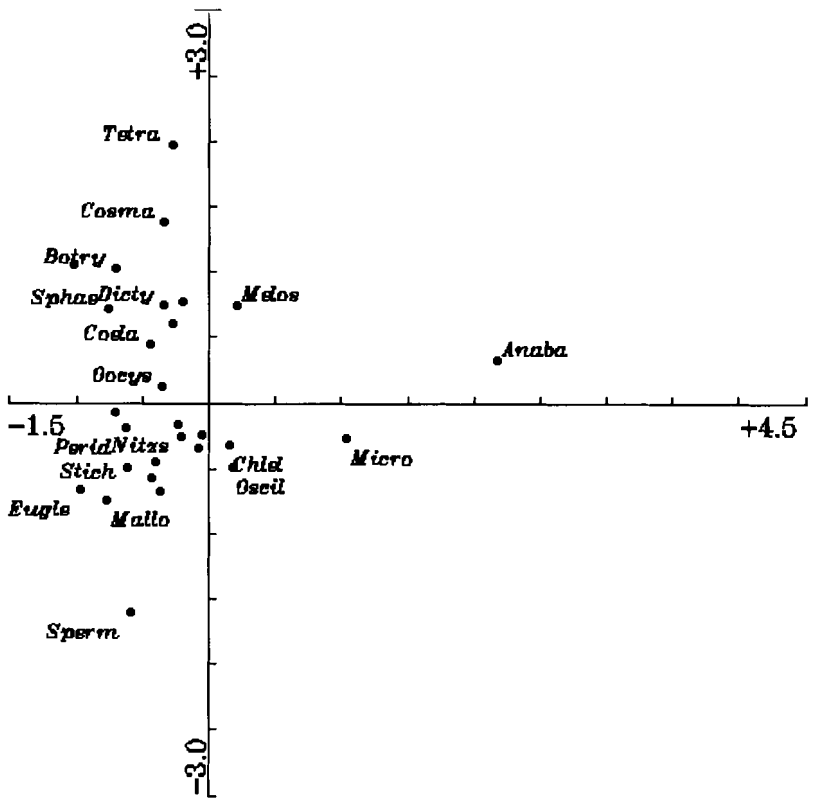

Figure 4.4 Correspondence Analysis ordination of algae over all lakes studied (to be read with site ordinations Fig. 4.3). Algal genera as per Table 2.4.

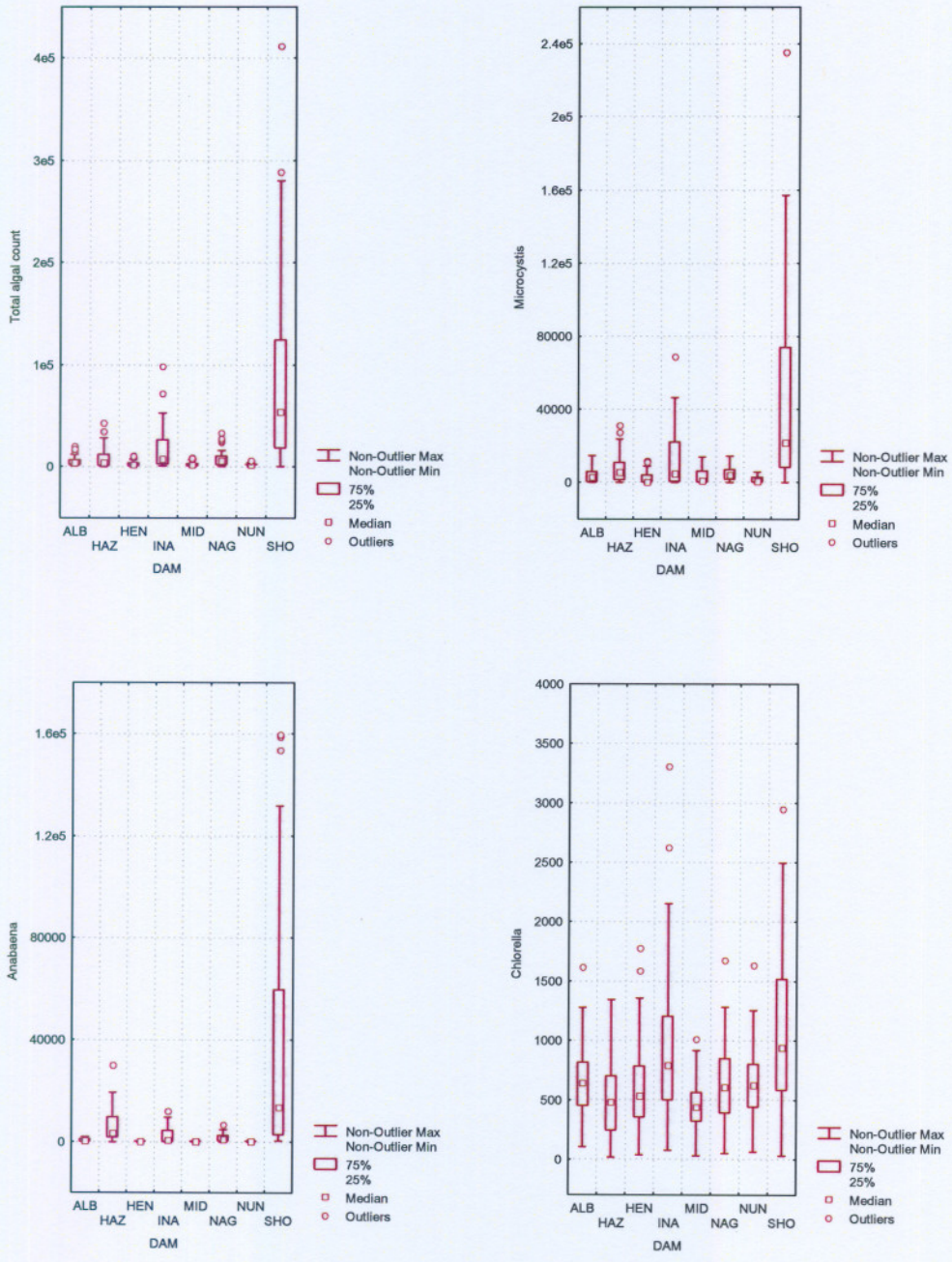

Figure 4.5 Abundance of Total algal populations, and some dominant algal genera recorded from all lakes studied (including Shongweni) during the study period.

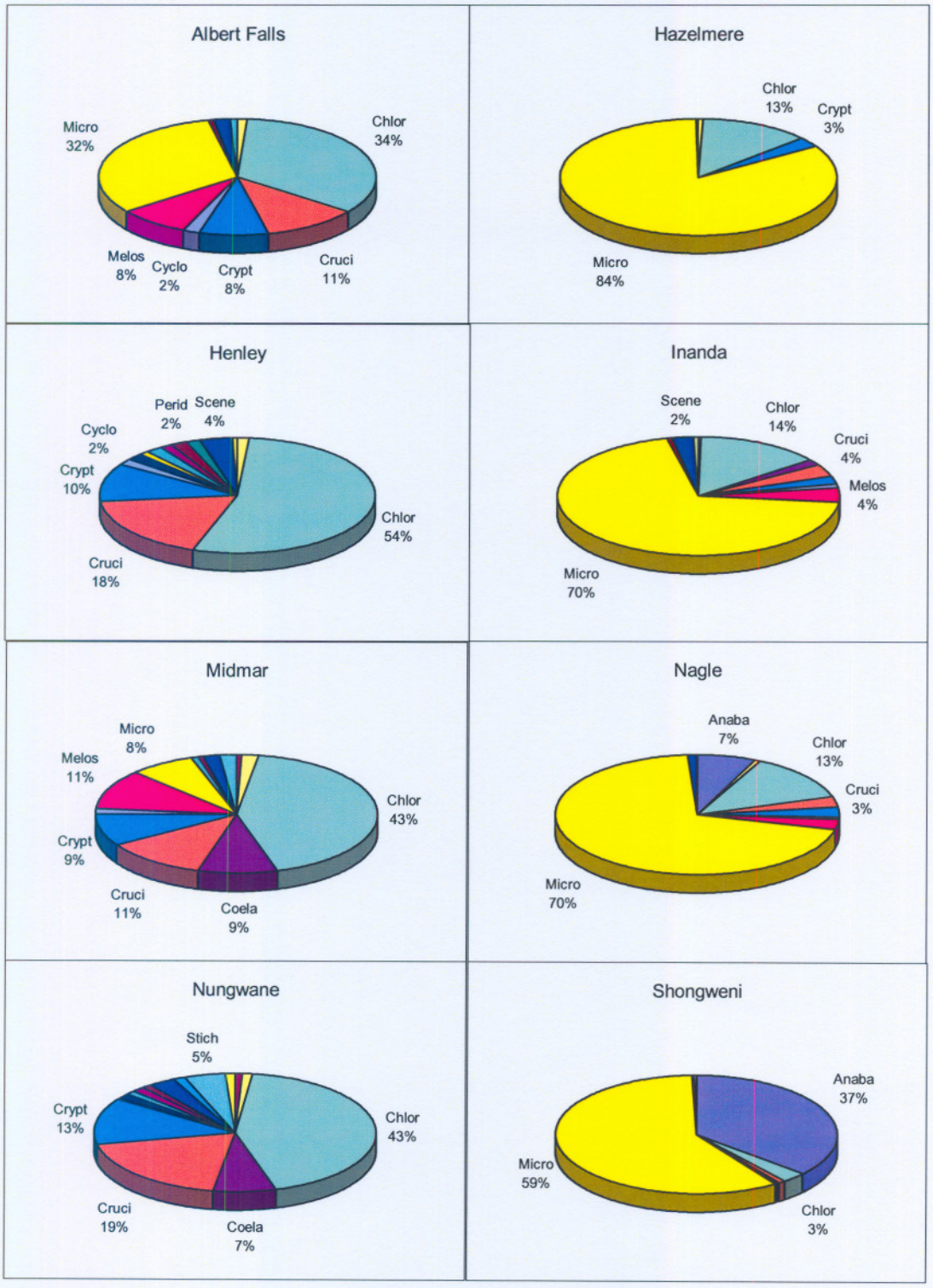

Figure 4.6 Proportional composition (as a percentage of the total median) for respective lakes for the duration of the study period (Chlor=*Chlorella*; Crypt=*Cryptomonas*; Cruci=*Crucigena*; Melos=*Melosira*; Anab=*Anabaena*; Micro=*Microcystis*; Coel=*Coelastrum*; Stich=*Stichococcus*; Perid=*Peridinum*; Scene=*Scenedesmus*; Cyclo=*Cyclotella*).

These figures clearly show Lake Shongweni as numerically and proportionally dominated by the blue-green algae (*Microcystis* and *Anabaena*). The algal counts for *Chlorella*,

another dominant to sub-dominant alga, are more evenly spread over the other lakes in the study.

Indications from these analyses were that Shongweni was particularly different from all the other lakes in the study - from both a physico-chemical as well as algal composition point of view. Shongweni was therefore eliminated from inclusion in further analyses because its very high values for certain environmental and algal variables, would dominate analyses and mask potentially significant relationships in other lakes important from a potable water supply point of view. Although the high counts of *Anabaena* and *Microcystis* in this lake are the key algae that appear to be separating it from all other lakes in the study, these taxa were still well represented in the rest of the data set.

The causes of physico-chemical and algal difference between Shongweni and other lakes in the study are probably related to inputs of industrial effluents and agricultural runoff emanating from large textile (and other) industries and intensively farmed agricultural lands located in the Shongweni catchment. Many of the processes in the textile industry produce high dissolved 'salt' effluents. The resultant and persistent eutrophication problems associated with these effluents contributed to the lake being decommissioned as a regular supplier of potable waters (Wilson, 1992).

4.4 Water quality and environmental conditions in all lakes excluding Shongweni

The environmental data in the remaining seven lakes were again analysed using standardised Principal Components Analysis (PCA) on the correlation matrix to produce a few key Principal Components (PCs) which describe a substantial portion of the variation in the environmental data. The rationale for the use of this method was as previously mentioned, i.e. to reduce the complexity (and dimensionality) of the environmental data to a manageable number of biologically meaningful variables, whilst removing the multicollinearity problem often found with such environmental data sets.

Of the 52 potential PCs only the first 14 were worthy of interpretation. They accounted for 77% of the variation in the environmental data with the first five accounting for over half of the total variation (Table 4.4). As with the previous analyses, an examination of the loadings of environmental variables on these axes (Table 4.5 - loading table)

revealed that the first five key PCs (in terms of the proportion of environmental data variability explained) could be described as follows: PC_1 – 'conductivity', PC_2 – 'turbidity' or 'secchi depth', PC_3 – 'N & P', PC_4 – 'nutrient loads or inflow volumes' and PC_5 – 'water column stability'. Therefore even after the removal of the Shongweni data from the analyses, there was still a similar pattern of variation (or structure) in the environmental data in the remaining seven lakes. Other potentially important environmental variables associated with these PC axes were silicon, temperature and % dissolved oxygen.

All the nutrient loadings (i.e. total phosphorus, soluble reactive phosphorus and total inorganic nitrogen) came out strongly together at the one end of PC_4 (Table 4.5). The implication is that some common environmental parameter is underlying the variability of the nutrient loads. This variable was obviously 'inflow' volume into the lakes as it was used in conjunction with a discrete water quality sample in the calculation of all nutrient loads. Chapter 3, section 3.4.2 discusses the relationship between inflow, nutrient concentration and nutrient loads in greater detail.

Table 4.4 Key results from a standardised Principal Components Analysis for the first 14 axes (performed on the environmental correlation matrix with the exclusion of all Shongweni sites from the data set). The first 5 axes are named

AXIS	EIGENVALUE	PERCENT OF TOTAL	CUMULATIVE PERCENT	Axis name
1	10.199	19.61	19.61	'Conductivity'
2	6.701	12.89	32.50	'Turbidity'
3	4.408	8.48	40.98	'N&P'
4	3.132	6.02	47.00	'Nutrient loads' or 'Inflow'
5	2.400	4.62	51.61	'Stability'
6	2.252	4.33	55.94	
7	1.687	3.24	59.19	
8	1.574	3.03	62.22	
9	1.523	2.93	65.14	
10	1.396	2.68	67.83	
11	1.308	2.52	70.34	
12	1.201	2.31	72.65	
13	1.100	2.12	74.77	
14	1.024	1.97	76.74	

Table 4.5 Ranked environmental variable loadings on the first five PCs/axes from PCA analysis of all lakes excluding Shongweni

Variable	PC_1	Variable	PC_2	Variable	PC_3	Variable	PC_4	Variable	PC_5
Conductivity		*Turbidity*		*N:P ratio*		*Nutrient loads or inflow*		*Stability*	
Se	-0.13	***secc***	***-0.24***	***TP***	***-0.30***	***TNLDE-3***	***-0.47***	***stabE4***	***-0.33***
Hg	-0.11	Ca	-0.13	***TP/Si***	***-0.26***	***TPLDE-1***	***-0.45***	***WnSp***	***-0.33***
Cd	-0.05	DO	-0.12	Ni	-0.19	***SRPLD***	***-0.41***	***%DO***	***-0.32***
TKN	-0.04	%DO	-0.12	SS	-0.18	temp	-0.24	***DO***	***-0.23***
TN/SRP	-0.04	SRPLD	-0.11	TAl	-0.18	F	-0.20	***Si***	***-0.21***
SHrs	-0.04	SRP/Si	-0.11	turb	-0.18	chl	-0.19	***WnDir***	***-0.20***
NH4	-0.04	TP/Si	-0.09	temp	-0.13	TKN	-0.14	***temp***	***-0.19***
Zn	-0.04	TPLDE-1	-0.08	SRP/Si	-0.13	Cr	-0.13	Cl	-0.14
Cu	-0.03	pH	-0.08	SRP	-0.12	stabE4	-0.11	TOC	-0.14
%DO	-0.03	alkal	-0.07	Cr	-0.11	TN/SRP	-0.10	Zn	-0.11
TNE3	-0.03	SO4	-0.06	%DO	-0.11	turb	-0.06	pH	-0.09
Cr	-0.03	hardness	-0.06	BOD	-0.11	SS	-0.06	Na	-0.09
TN/TP	-0.02	Cu	-0.05	chl	-0.10	%DO	-0.05	chl	-0.09
Si	-0.02	TNLDE-3	-0.05	pH	-0.10	NO3	-0.04	TDS	-0.08
NO3	-0.02	TN/Si	-0.02	stabE4	-0.09	Zn	-0.04	SRP	-0.06
NO2	-0.01	Cd	-0.02	Fe	-0.09	TN/TP	-0.03	cond	-0.04
Pb	-0.01	SHrs	-0.01	TOC	-0.08	Cd	-0.03	Fe	-0.03
DO	-0.01	K	0.00	TPLDE-1	-0.05	TNE3	-0.03	Pb	-0.01
secc	-0.01	chl	0.01	Se	-0.04	Si	-0.02	TN/TP	-0.01
BOD	-0.01	NO2	0.02	Cu	-0.02	pH	-0.02	Mg	0.00
TP	-0.01	Zn	0.03	K	-0.02	TAl	-0.01	BOD	0.01
stabE4	0.00	cond	0.03	DO	-0.02	Ni	-0.01	Se	0.02
Ni	0.02	BOD	0.03	SRPLD	-0.02	NO2	-0.01	TKN	0.03
Fe	0.03	Mg	0.03	TNLDE-3	0.00	Ca	-0.01	NO2	0.04
SRP	0.03	WnDir	0.04	F	0.00	SO4	-0.01	K	0.04
Mn	0.03	F	0.05	Ca	0.01	TDS	0.00	secc	0.05
chl	0.03	temp	0.06	Pb	0.01	Se	0.00	NO3	0.06
SS	0.03	Se	0.06	TDS	0.02	secc	0.01	alkal	0.07
TAl	0.04	TKN	0.06	Cd	0.02	hardness	0.02	Cd	0.07
turb	0.04	WnSp	0.07	hardness	0.03	Fe	0.02	NH4	0.07
TNLDE-3	0.06	Na	0.07	TKN	0.03	Pb	0.03	Cu	0.07
TP/Si	0.07	TN/TP	0.08	alkal	0.03	Hg	0.03	Hg	0.08
temp	0.09	Cl	0.09	SO4	0.03	Na	0.04	TNE3	0.08
TPLDE-1	0.09	NH4	0.11	Hg	0.04	WnDir	0.05	TPLDE-1	0.08
TN/Si	0.11	Pb	0.11	WnSp	0.04	Mg	0.05	SRPLD	0.09
F	0.11	stabE4	0.11	Mg	0.05	cond	0.05	hardness	0.09
SRP/Si	0.11	SRP	0.11	cond	0.07	NH4	0.05	Mn	0.09
WnDir	0.13	Hg	0.12	NO2	0.07	TOC	0.05	TNLDE-3	0.10
pH	0.13	TN/SRP	0.12	SHrs	0.08	Cl	0.06	TAl	0.10
SRPLD	0.14	TDS	0.13	Si	0.09	WnSp	0.06	TN/SRP	0.10
WnSp	0.16	TOC	0.13	Na	0.09	Cu	0.07	F	0.11
TOC	0.16	Ni	0.14	Cl	0.09	K	0.07	TP	0.11
TDS	***0.25***	TP	0.15	WnDir	0.09	alkal	0.08	SO4	0.11
K	***0.27***	Mn	0.16	Zn	0.12	DO	0.09	turb	0.12
Ca	***0.27***	Cr	0.16	Mn	0.15	BOD	0.09	Ni	0.13
Cl	***0.27***	***NO3***	***0.26***	secc	0.16	SHrs	0.09	Cr	0.14
alkal	***0.27***	***TNE3***	***0.26***	TN/Si	0.18	Mn	0.09	SRP/Si	0.15
SO4	***0.28***	***SS***	***0.27***	***NH4***	***0.22***	TP	0.09	SS	0.15
Na	***0.29***	***TAl***	***0.30***	***NO3***	***0.22***	TN/Si	0.09	Ca	0.16
hardness	***0.30***	***turb***	***0.31***	***TNE3***	***0.26***	TP/Si	0.17	***SHrs***	***0.20***
Mg	***0.30***	***Si***	***0.31***	***TN/SRP***	***0.31***	SRP	0.18	***TP/Si***	***0.23***
cond	***0.30***	***Fe***	***0.32***	***TN/TP***	***0.40***	***SRP/Si***	***0.22***	***TN/Si***	***0.27***

Table 4.5 note: Variables with the highest loadings on (and therefore association with) respective axes highlighted in ***bold***. See Table 2.2 for an explanation of codes used for environmental variables.

A summary of the key environmental variables associated with respective PC axes is presented in Table 4.6.

Table 4.6 Reduced key environmental data set representing axes of major variation in the environmental data set (also reflecting major physico-chemical limnological processes occurring in lakes)

Principal Component axis	PC_1	PC_2	PC_3	PC_4	PC_5
Environmental variables	Conductivity	Secchi Silicon	Total Inorganic Nitrogen Total Phosphorus	Inflow Temperature	% DO Stability

An examination of the distribution of lakes, in the absence of Shongweni data, by means of a PCA ordination diagram (Fig. 4.7), reveals how Inanda sites tended to dominate the 'high end' of the first PC axis, which represented a conductivity gradient (Tables 4.5 & 4.6), whilst the other lakes were arranged further back along this axis. This distribution of sites followed the expected pattern for these lakes, i.e. along an increasing pollution gradient of conductivity (and hence salts). 'Cleaner' inland lakes (Henley, Midmar, Albert Falls and Nagle) were at the low end of the conductivity gradient with coastal lakes (Nungwane, Hazelmere and Inanda), lower down in the catchment, coming out at the 'high' conductivity end of PC_1.

Interestingly, all the uMngeni River lakes were closely grouped at the low end of PC_1 with Inanda, a distinct outlier, at the high end of the 'conductivity' gradient. With few other major tributaries, other than the polluted uMsunduze River, coming into the uMngeni system between Nagle and Inanda (Fig. 1.1 - Study area), the relative impact of this tributary on the water chemistry of Inanda is demonstrated by this analysis. The uMsunduze is acknowledged to be a polluted river system, passing as it does through the urbanised Pietermaritzburg area with its associated inputs from industry, sewage effluent and storm-water runoff (MCMP, 1996).

The second PC axis, identified in this analysis as a turbidity gradient (Tables 4.5 & 4.6), had all the uMngeni River lakes grouped at its lower end (PC_2, Fig. 4.7). On the other hand, Hazelmere, on the Mdloti River, is clearly separated out at the upper end of this axis. This lake is acknowledged to be a highly turbid system, e.g. Walmsley & Butty (1980), and as narrated in Chapter 2. Nungwane (and to a lesser extent Henley) is also separated from the uMngeni system lakes by this turbidity feature on the second axis.

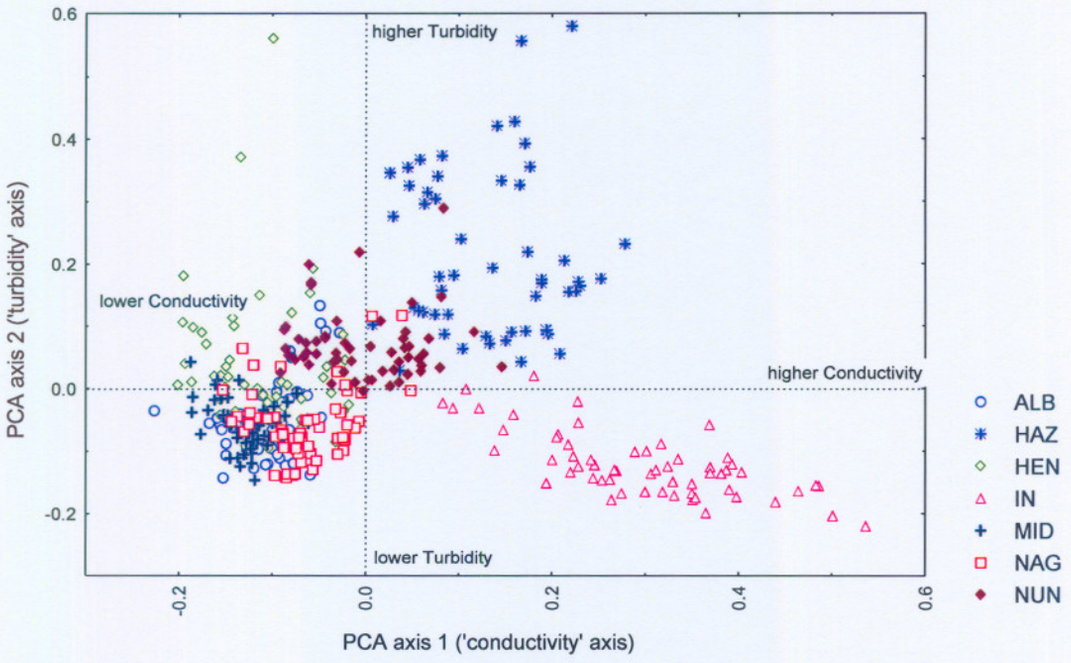

Figure 4.7 Principal Components Analysis ordination of lake sites on the first two PC axes ('conductivity' and 'turbidity'). (ALB = Albert Falls, HAZ = Hazelmere, HEN = Henley, IN = Inanda, MID = Midmar, NAG = Nagle, NUN = Nungwane).

The other noteworthy feature of Figure 4.7 is the close proximity and progression of sites, from Midmar through to Albert Falls and then on to Nagle, which respectively lie successively downstream from one another on the uMngeni River system. This analysis indicates that their general water quality is more alike, as a group of lakes, compared to all the other lakes studied. Henley, although generally a little more turbid, is fairly similarly grouped to this set just mentioned, although in a different sub-catchment (the uMsunduze River). Its physico-chemistry is obviously similar with its source waters originating in a comparable area of the kwaZulu-Natal Midlands as Midmar and Albert Falls. The coastal lakes, in different catchments (Nungwane on the Nungwane River and Hazelmere on the Mdloti River) are not surprisingly different in their overall water chemistry from the uMngeni system lakes, and are hence further apart on Figure 4.7.

Ordination of lake sites on the third and fourth PCA axes (respectively identified as gradients in nitrogen and phosphorus, and 'inflow volumes' (Fig. 4.8 - PCA ordination with 3rd and 4th axes)) did not give as clear a separation of sites in relation to these gradients as was obtained with the first two ordination axes. The lack of clear gradients was not surprising given that successive axes in PCA account for progressively less variation in the underlying environmental data set (Table 4.4). Table 4.4 indicates that

the third and fourth axes only account for 8% and 6% of the variability in the environmental data respectively whilst the first two axes accounted for 20% and 13% respectively.

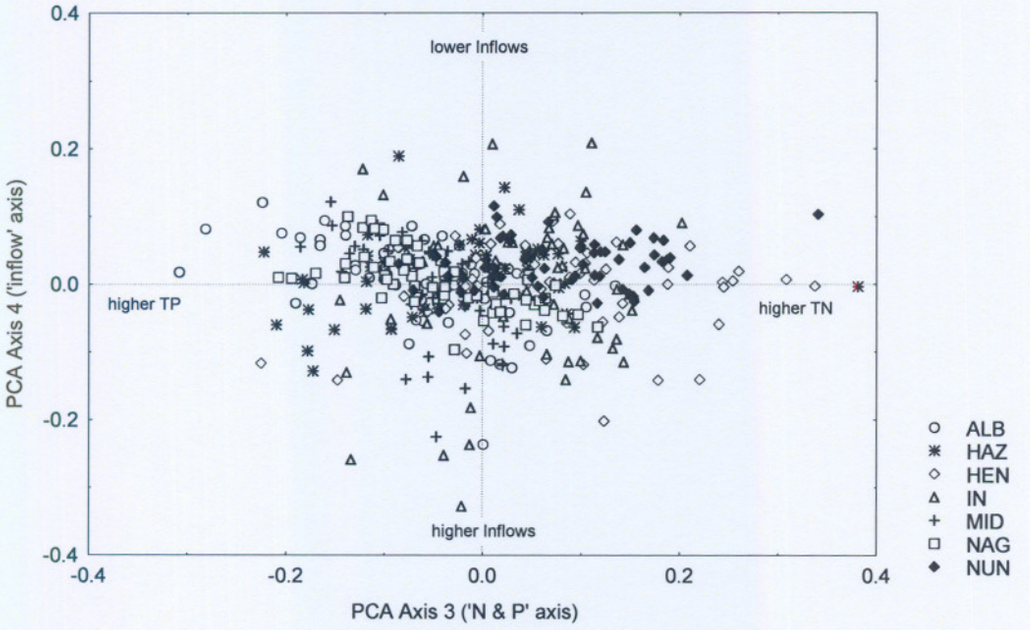

Figure 4.8 Principal Components Analysis ordination of lake sites on the third and fourth PCA axes ('N & P' and 'inflow volumes' respectively). (ALB = Albert Falls, HAZ = Hazelmere, HEN = Henley, IN = Inanda, MID = Midmar, NAG = Nagle, NUN = Nungwane).

Although there was relatively poor separation of sites, there did appear to be a few interesting trends worthy of comment. The majority of Midmar, Nagle and Albert Falls sites appeared to be tending towards the higher Total Phosphorus end of PC_3 axis, whilst Nungwane, and to a lesser extent Henley, were at the end with higher Total Nitrogen. Walmsley & Butty (1980) also noted that Henley is influenced by high nitrogen values. Inanda sites were widely spread over both ends of all axes indicating it experiences a range of these environmental conditions. No clear trends existed for the separation of lakes along the 'inflow' gradient (PC_4).

4.5 Summary

Many of the 52 environmental variables assembled for this study were highly inter-correlated (both positively and negatively), or in other words were multi-collinear. This feature of the data causes problems for multiple regression analysis. Principal

Components Analysis (PCA) was used to address the multicollinearity issue. Successive PC axes capture different and unique aspects of variation in the data set.

From both a physico-chemical as well as algal composition point of view Shongweni appears to be significantly different from all the other lakes in the study. Shongweni is numerically and proportionally dominated by the blue-green algae (*Microcystis* and *Anabaena*) and by high nutrient and dissolved salt concentrations. The causes of this difference between Shongweni and other lakes in the study are probably related to inputs of industrial effluents and agricultural runoff emanating from large textile (and other) industries and intensively farmed agricultural lands located in the catchment. Shongweni was therefore eliminated from inclusion in further analyses. Reasons for its exclusion was that its very high values for certain environmental and algal variables would probably dominate analyses and mask potentially significant algal/environmental relationships in other lakes, important from a potable water supply point of view. With the removal of the Shongweni data the potentially problematic blue-green algal taxa, *Anabaena* and *Microcystis,* were still well represented in the rest of the data set.

Analyses (after the exclusion of Shongweni data) indicated that most of the variability in the remaining environmental data-set could be explained by five key axes or PCs. These captured some 52% of the variability in the environmental data. Certain 'key' environmental variables (viz. conductivity, secchi depth (or water column turbidity), silicon, total inorganic nitrogen and total phosphorus, inflow, temperature, % dissolved oxygen and stability), which represented primary components of variation in the data-set and acted as surrogates for this variation, were uniquely and strongly associated with these PCs.

Lake Inanda sites tended to dominate the 'high end' of a conductivity gradient with the distribution of other lakes following an expected pattern along an decreasing pollution gradient of conductivity (and hence salts). 'Cleaner' inland lakes (Henley, Midmar, Albert Falls and Nagle) were all at the low end of the conductivity gradient, with coastal lakes (Nungwane, Hazelmere and Inanda), lower down in the catchment, at the 'high' conductivity end of the gradient.

All the uMngeni River lakes were closely grouped as low conductivity lakes with Inanda, a distinct outlier, at the high end of this gradient. With few other major tributaries, other than the polluted uMsunduze River, coming into the uMngeni system between Nagle and Inanda, the relative impact of this tributary on the water chemistry of Inanda is

demonstrated by this analysis. The uMsunduze is acknowledged to be a polluted river system, contributing inputs from industry, sewage effluent and storm-water runoff.

The second major axis of variability in the environmental data is a turbidity gradient. All the uMngeni River lakes are grouped at its lower end (clear water with higher secchi depths). Hazelmere, on the Mdloti River, is clearly confined to the upper end of this axis, and is acknowledged to be a highly turbid system, e.g. Walmsley & Butty (1980), and as narrated in Chapter 2. Nungwane is also separated from the uMngeni system lakes by its higher turbidity.

Midmar, Nagle and Albert Falls tended to be dominated by higher total phosphorus concentrations, whilst Nungwane, and to a lesser extent Henley, had with higher total inorganic nitrogen concentrations. Walmsley & Butty (1980) also noted that Henley is influenced by high nitrogen values. Inanda sites appear to experience a range of these environmental conditions.

The analyses reflect that as a group the uMngeni system lakes have similar water quality compared to all the other lakes studied. Henley, although more turbid, is fairly similarly grouped to this set just mentioned, although in a different sub-catchment (the uMsunduze River). The coastal lakes, in different catchments (Nungwane on the Nungwane River and Hazelmere on the Mdloti River), are most dissimilar to the uMngeni system lakes.

Use of PCs helped to objectively identify lakes that could be considered to have similar physico-chemical (water quality) properties and therefore would be likely to respond similarly to management/perturbation – for example reduction in nutrient loadings.

The reduced data set of key environmental variables, representing axes of major variation in the environment, were taken forward into further analyses to determine which aspects, and to what degree, the environment may be influencing the distribution and abundance of problematic algae in the lakes within the Umgeni Water operational area (first objective of the study).

CHAPTER 5: RELATIONSHIPS BETWEEN ALGAE AND KEY ENVIRONMENTAL VARIABLES

5.1 Introduction

One of the primary aims of this study was "establishing the key environmental variables influencing the distribution and abundance of problematic algae in lakes within the Umgeni Water operational area". To achieve this a series of investigations were undertaken to better understand the relationships between observed algal heterogeneity and the corresponding environment.

The heterogeneity that may be observed in communities between spatially different sites generally has various origins (Borcard *et al.*, 1992). Two classic models have been given in the literature to describe this spatial heterogeneity (e.g. May, 1984). Firstly, the environmental control model (where environmental variables (in the strictest sense of the word) have the primary control over the observed variations) and secondly the biotic control model (where links among organisms, e.g. competition or predation, are considered to be the primary factors structuring communities (Southwood, 1987)). These models have often been viewed as competing or mutually exclusive hypotheses (May, 1984). Quinn and Dunham (1983) propose viewing the alternative models in terms of the relative contribution of each to explaining the causes of community variation.

Within this understanding of the range and complexity of potential models, analyses were conducted to determine which facets of the environment, i.e.:

- zooplankton predation,
- quantified (i.e. measured in this study) environmental variables,
- un-quantified effects unique to respective lakes, and
- seasonality,

were playing the most significant part in explaining the observed variability in the algal populations.

Each of these analyses revealed a different aspect of how the heterogeneity in observed algal communities could be explained, and hence addressed the first objective of this

study. From this, interpretations could be made about the conditions favouring problematic algae.

Due to the multivariate nature of the data , i.e. numerous different types of algae and numerous environmental effects potentially affecting them, analyses based on ordination techniques were used. Essentially algae are not independent, i.e. they do not exist secluded in aquatic systems, as there is always a range of biological interactions taking place in these communities - be it competitive exclusion, luxury nutrient uptake, predation, etc. (e.g. Lampert *et al.,* 1986; Sommer, 1989; Shapiro, 1997). Therefore attempts to model individual algal genera, as independent factors, against a range of environmental parameters will always be fraught with problems if the interaction effects are ignored. Soudant *et al.* (1997) noted that due to the inherent complexity of this type of algal data and the high variabilities often associated with algae over time (generated by processes occurring at different spatial and temporal scales), static regression models are not suitable for this type of analysis – although they were investigated in this study. Multiple regression ordination techniques elegantly surmount this problem and consider the response of algae to the environment at the community level i.e. how are ***all*** the algae responding to the environment as a whole?

Analyses in the preceding chapter had identified a reduced number of environmental variables (Conductivity, Secchi, Silicon, Total Inorganic Nitrogen, Total Phosphorus, Inflow, Temperature, %DO, Stability) that appeared to describe the principal elements of variability in the environmental data set. The reduced set of variables had intuitive biological appeal in that they are often implicated in the literature as influencing algae in some way. There was also greater innate attraction to using 'real' environmental variables, over statistically defined PC axes (composite environmental gradients), for use in establishing algae/environment relationships. Most of the major physico-chemical limnological processes known to affect algae (nutrients, energy, water column stability and light regime (e.g. Sommer, 1989; Viner, 1985; Tilman *et al.*, 1986; Wetzel, 2001) were represented in some measure by the selected variables. It was assumed that the selection of the reduced set of 'key' environmental variables would in fact effectively represent the major limnological processes as they influenced algae in the lakes in this study. However not all of them were necessarily related to the observed algal variability.

Prior to relationships being investigated between the physico-chemical variables and algae, and based on a relatively small data set (Hart, pers. comm.), a small pilot study

was undertaken to determine the relative contribution that zooplankton could be making to an understanding of the observed algal heterogeneity.

5.2 The effect of zooplankton on algae

The literature has indicated that zooplankton may affect the presence and abundance of algae (e.g. Kimmel *et al.,* 1990; Hart, 1992). The possible effect of predation is not an unreasonable proposal because algae are primarily predated upon by zooplankton. However, algae have equally been shown to be not the only resource, or food base, for zooplankton (e.g. Sanders & Porter, 1990).

Due to the relatively meagre zooplankton data set available (only 3, Midmar, Albert Falls & Inanda, of the 7 lakes studied in any detail, and only available from June 1990 to June 1992, compared to 1990 to 1994 for the full environmental/algal dataset) a small pilot study was conducted on Lake Inanda to gain some understanding of this biotic pressure on observed algal heterogeneity. For valid analyses environmental and algal datasets were reduced to the same period. Weekly environmental and algal data was used to increase the number of sample points (N) in this particular set of analyses (as opposed to monthly used in all subsequent analyses).

Results indicate that with the inclusion of the zooplankton, soluble and total phosphorus loads emerged as significant factors accounting for observed algal heterogeneity. When analyses were re-run without the zooplankton, but using the otherwise identical environmental and algal datasets, the phosphorus load variables again appeared to be the most influential variables affecting algal heterogeneity. This suggested that zooplankton were not statistically important in the Inanda system as determinants of algal heterogeneity. To confirm this, another set of analyses were run with zooplankton, and no other environmental data, to examine the relationship zooplankton alone were having with algae. The outcome of this analysis was Monte Carlo permutation tested. The results indicated algal heterogeneity to be apparently statistically non-significantly influenced by available zooplankton data.

It was concluded that observed algal abundances were poorly explained by the zooplankton records. As previously mentioned, work by some authors has indicated that algal abundance in aquatic systems is generally affected by three controlling factors viz. physical and chemical factors, and biotic factors (e.g. Kimmel *et al.,* 1990). The results

of this analysis appears to show that it is probably something in the realm of physico-chemical factor/s within the lake environments that are the most important in terms of driving the observed algal community composition - although the biotic effects of parasitism and competition have not been examined. Subsequent analyses sought to uncover these 'other factors' or environmental conditions.

Based on the structure of the available data the following analyses were undertaken to determine which facets of the environment, i.e.:

- Quantified (i.e. measured in this study) environmental variables (section 5.3),
- Un-quantified effects unique to respective lakes (section 5.4), and
- Seasonality (months, section 5.5)),

were playing the statistically most significant part in explaining the observed heterogeneity in the algal populations. Each set of analyses revealed different aspects of how the algae appeared to be responding to the environment - within lakes as a whole but also between lakes and across seasons.

5.3 The effect of measured environmental conditions on observed algal heterogeneity

To briefly summarise the results of the preceding chapter, it was found that many of the 52 environmental variables assembled for this study were highly inter-correlated, or in other words were multicollinear. Principal Components Analysis (PCA) was used to address the multicollinearity issue, where successive PC axes captured different and unique aspects of variation in the data set. Analyses (after the exclusion of Shongweni data) indicated that most of the variability in the remaining environmental data-set could be explained by five key axes or PCs. These captured some 52% of the variability in the environmental data and certain 'key' environmental variables (viz. conductivity, secchi depth (or water column turbidity), silicon, total inorganic nitrogen and total phosphorus, inflow, temperature, % dissolved oxygen and stability), were uniquely and strongly associated with these PCs. The reduced number of environmental variables appeared to describe the principal elements of variability in the environmental data set, and in subsequent analyses are referred to as 'key' environmental variables. It should be emphasised that all further results and analyses presented in this thesis pertain to analyses performed after the removal of Lake Shongweni data.

Redundancy Analysis (RDA), with Monte Carlo permutation testing, was the principal analytical tool used for this aspect of the analysis. This would provide some indication on possible response relationships between the environment and the various algal genera. The Monte Carlo permutation testing was to establish the probability that the relationship was not just due to chance (ter Braak, 1987-1992).

5.3.1 Effects of 'key' environmental variables

Key environmental variables were fitted to the algal data with the spatial effects, of respective lakes and the temporal effects of months, overlain as passive variables. The 'fitting' of key environmental data was to determine which of these elements appeared to be statistically significantly related to the pattern of variability observed in the algal data. 'Passive' plotting of variables was to examine the relationship of these variables to the algae and the key environmental variables of interest but without restricting the derivation of the ordination axes with their inclusion (ter Braak, 1987-1992).

5.3.2 Results and Discussion

Results from these analyses are summarised in Table 5.1 and Figures 5.1 & 5.2.

Table 5.1 Results from analyses where 'key' environmental variables were related to algae in all lakes except for Shongweni

Axes	1	2	3	4	Total variance
Eigenvalues	0.08	0.03	0.02	0.01	1.00
Algae-environment correlations	0.69	0.52	0.51	0.42	
Cumulative % variance					
of algal data.	8.0	11.0	12.8	14.0	
of algae-env. relation	51.2	69.6	81.3	88.9	
Sum of all unconstrained eigen values					1.00
Sum of all canonical eigenvalues					0.16
Algal variability explained by analysis					15.7%

Key findings and observations were that the first 2 axes accounted for only 11% of algal data variation and 70% of the algal/environment relationship (Table 5.1). It should be noted that this low proportion of the algal variance accounted for is not atypical of multi-species data sets, particularly when there are many zero values (e.g. Stevenson *et al.*, 1991; Agbieti, 1992) and that it can still provide a useful description of observed pattern (ter Braak, 1987-1992). Gauch (1982) notes that ordination diagrams that explain only a low percentage of species variability may be quite informative, with ter Braak (1990) indicating that the low explained variance is not as important as the significance testing.

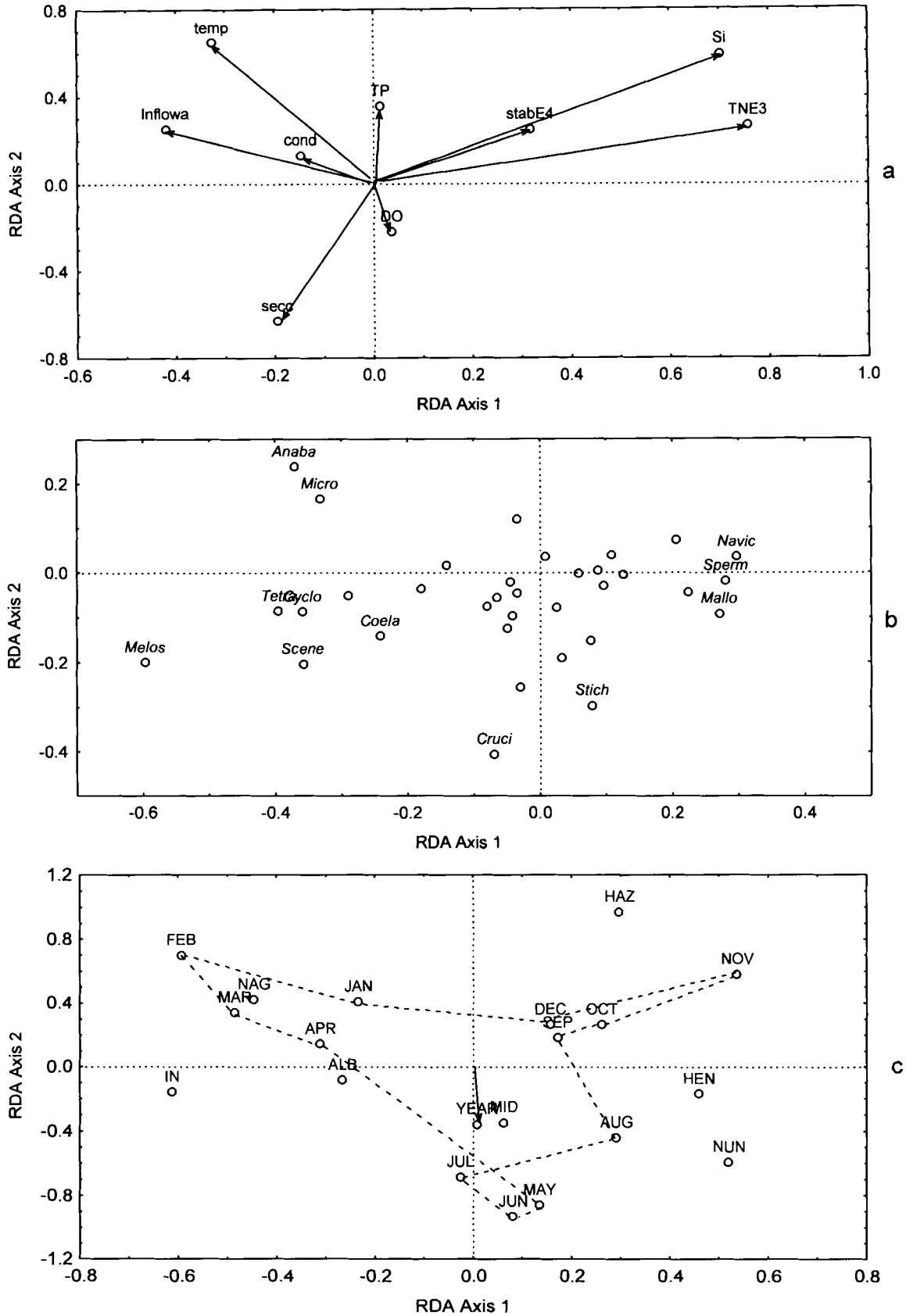

Figure 5.1 Ordinations where key environmental variables influencing algae were examined (a). Lakes and months (seasons) were plotted as passives (c). Algal genera with >10% of their variation explained by the analysis are plotted and labelled (b). Abbreviations as per Tables 2.1 & 2.4 and interpretations of the figures as detailed in §3.7.2.4 & Figure 3.4.

As a check on the significance of this apparently low explained variability, as in all facets of the analyses, the environmental effects were Monte Carlo permutation tested (ter Braak, 1987-1992) and found to be statistically significantly related ($P < 0.01$) to the variation in the algal data. In other words, environmental variables identified in the analyses as explaining the variability in the species data (total inorganic nitrogen, silicon, temperature, inflow (on the first axis) and total phosphorus and secchi (on the second axis)) account for a statistically significant proportion of the variability in the observed algal heterogeneity. Put another way, of all the environmental variables available for analysis, these just listed, form the primary environmental gradients (axes) along which the observed algae appear to be responding.

Algal genera which had a significant proportion (>10%) of their variance explained by the environmental variables in this analysis are labelled on the ordination (Fig. 5.1b) and summarised in Table 5.2. These results show the ranked response of genera (in terms of variance explained) to the supplied environmental variables. Clearly *Melosira* has more of its variability explained by the supplied environmental variables than does *Euglena*. The higher this variance, the greater the confidence that the supplied environmental variables are important in explaining observed variability in the algae.

Table 5.2 Ranked response of genera (in terms of variance explained) to the supplied key environmental variables

Genus	Cumulative % variance explained
Melosira	49.42
Anabaena	22.39
Tetraedron	22.26
Cyclotella	21.94
Crucigenia	20.26
Scenedesmus	19.48
Peridinium	18.81
Spermatozopsis	16.98
Microcysits	16.68
Stichococcus	16.09
Mallomonas	15.83
Cosmarium	14.10
Coelastrum	13.28
Navicula	12.72
Cryptomonas	12.24
Botryococcus	11.26
Euglena	10.87

Note - only genera with >10% of their variance explained are included in above table.

Trend surface plots for key environmental variables (Figs. 5.3 to 5.9) and responsive algae (Figs. 5.10 to 5.16) over all sites in the ordinations were fitted using kriging (see Ch 3 - Methods). This resulted in semi-quantitative models relating algae to the environment. The kriging approach provided best fit 'contours' for environmental variables across the ordination diagram. When read in conjunction with the algae/environment biplot (Fig. 5.2), they provide some indication of the levels of key environmental variables when respective algae are abundant.

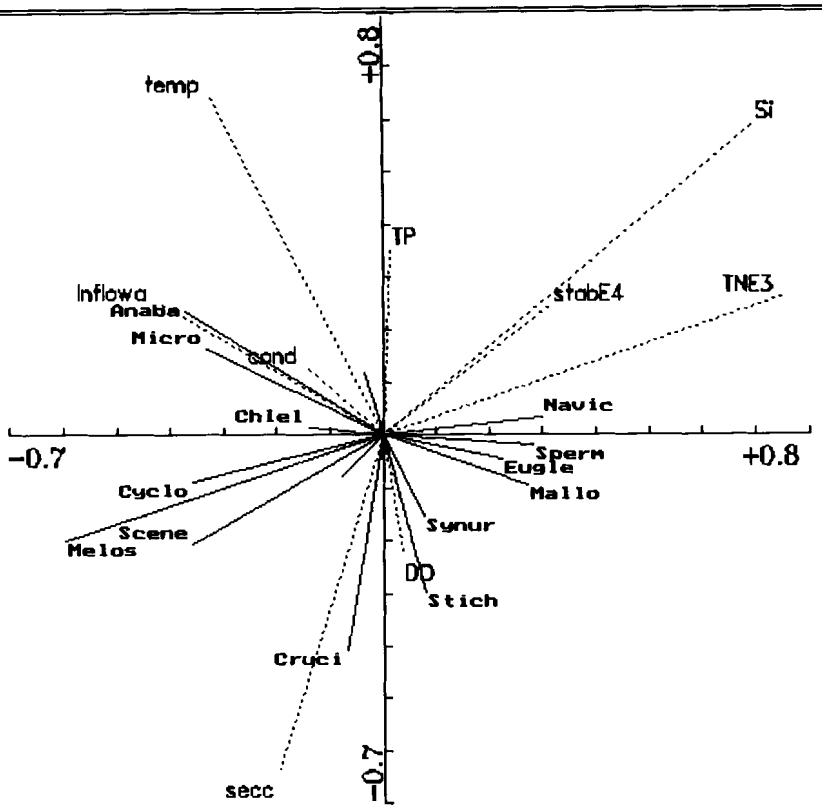

Figure 5.2 Biplot ordination of algae and key environmental variables (summary of Fig. 5.1a & b). Algal genera are: Anaba=*Anabaena*, Micro=*Microcystis*, Navic=*Navicula*, Sperm=*Spermatozopsis*, Eugle=*Euglena*, Mallo=*Mallomonas*, Synur=*Synura*, Stich=*Stichococcus*, Cruci=*Crucigena*, Scene=*Scenedesmus*, Melos=*Melosira*, Cyclo=*Cyclotella* and Chlel=*Chlorella*. Environmental variables are: TP=Total Phosphorus, Si=Silicon, stabE4=stability, TNE3=Total Inorganic Nitrogen, DO=%Dissolved Oxygen, cond=conductivity, Inflowa=inflow, and temp=temperature.

Figure 5.2 illustrates the joint variation of key environmental variables along with algae which had a reasonable proportion of their variability explained by the supplied environmental variables. Note the length of the environmental arrows indicating their importance, rate and direction of change in explaining observed algal variability. Of these variables silica, total nitrogen, temperature, inflow and secchi are important (relatively long arrows). The angle subtended by arrows indicates the degree of correlation between variables e.g. temp and inflow are positively correlated, TP and secchi negatively correlated (on different sides of the origin) whilst temp and silicon (or total inorganic nitrogen) are not correlated. Higher temperature and inflow favour the blue-green algae *Anabaena* and *Microcystis* whilst the opposite set of conditions favour *Stichococcus, Mallomonas, Euglena* and *Spermatozopsis.* High total inorganic nitrogen and silicon tends to favour *Navicula, Stichococcus, Mallomonas, Euglena* and *Spermatozopsis*, whilst low readings for these variables favour *Cyclotella, Melosira* and *Scenedesmus.* Clear water with a high secchi disk reading and generally low TP favours *Crucigena.*

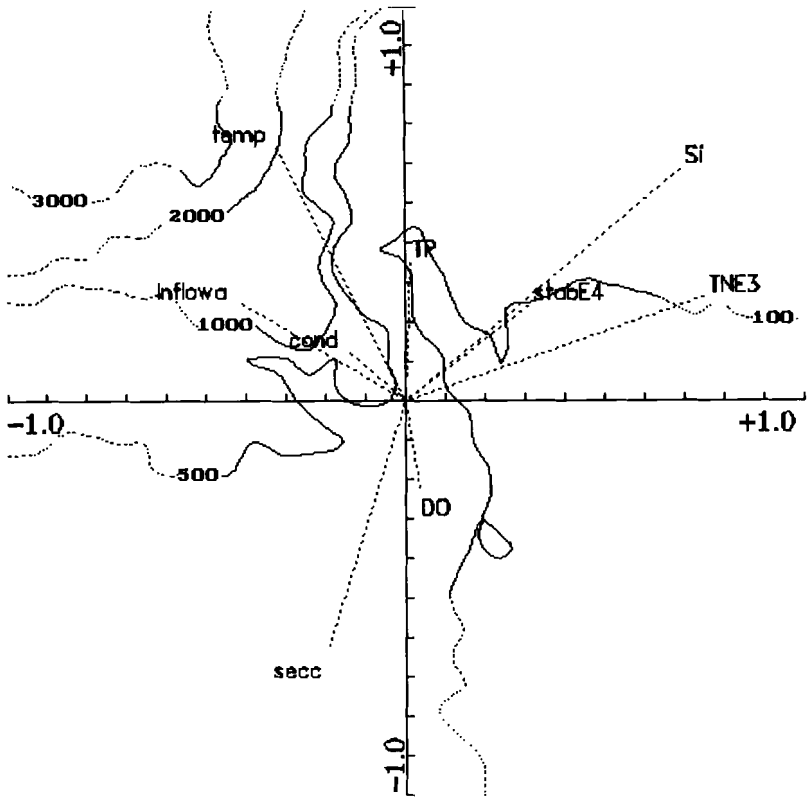

Figure 5.3 Inflow (Mℓ/day) trend data fitted to the ordination of sites from all seven lakes.

In conjunction with Figure 5.2, Figure 5.3 illustrates the trend of inflow data over all the sites underlying this ordination. Clearly inflow is highest in the top left quadrant of the ordination diagram. This corresponds with that grouping of lake sites sites having a large proportional abundance of the blue-green algae *Anabaena* and *Microcystis*. Optimal inflow conditions favouring these algae appear to be >1000Mℓ/day. Lowest flows (<500Mℓ/day) are in the bottom right quadrant of the ordination figure.

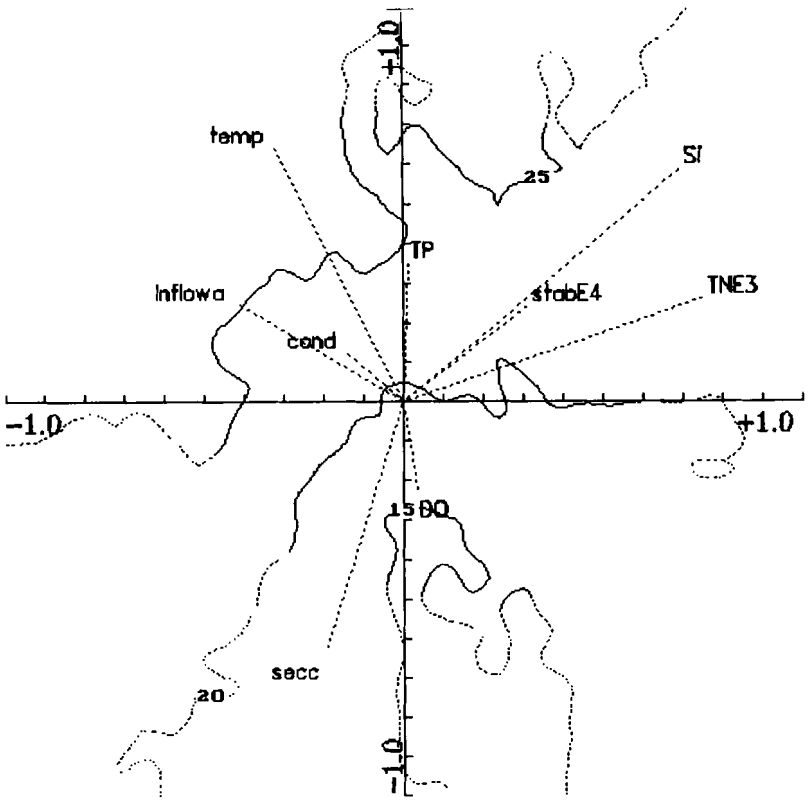

Figure 5.4 Temperature (°C) trend data fitted to the ordination of sites from all seven lakes.

Figure 5.4 (to be interpreted in conjunction with Fig. 5.2) illustrates the trend of temperature data over all the sites underlying this ordination. Temperature is highest in the top left quadrant of the ordination diagram. This corresponds with that grouping of lake sites having a large proportional abundance of the blue-green algae *Anabaena* and *Microcystis*. Optimal temperature conditions favouring these algae appear to be >20°C. *Melosira, Cyclotella* and *Scenedesmus* appear to favour temperatures between 20° and 25°C whilst *Spermatozopsis, Euglena, Synura, Stichococcus* and *Mallomonas* favour temperatures <20°C.

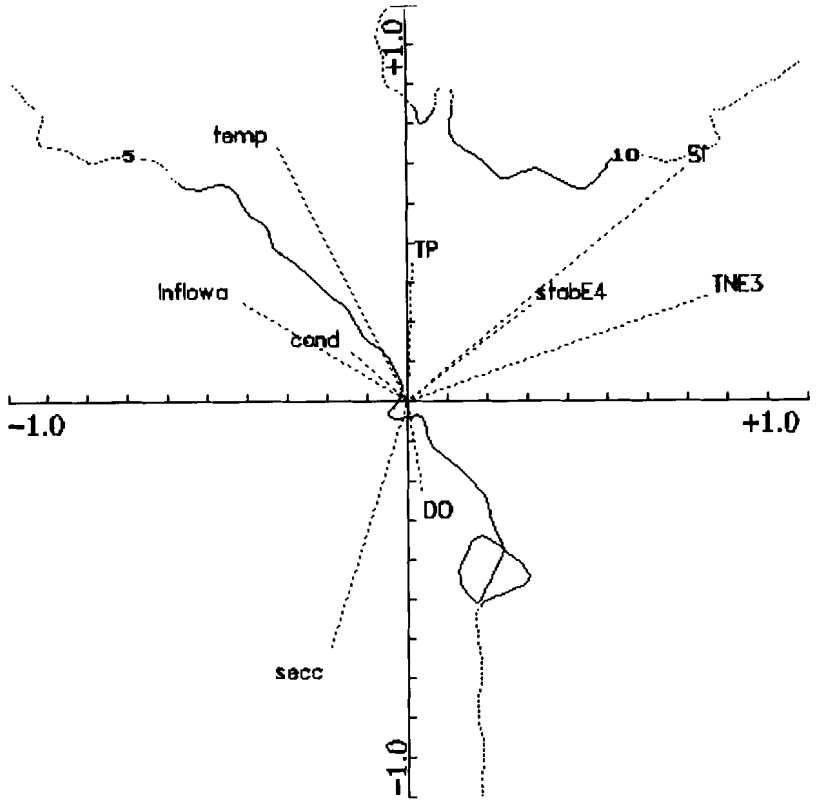

Figure 5.5 Silicon (mg/ℓ) trend data fitted to the ordination of sites from all seven lakes.

Figure 5.5 (to be interpreted in conjunction with Fig. 5.2) illustrates the trend of silica data over all the sites underlying this ordination. Silicon concentration is highest in the top right quadrant of the ordination diagram whilst lowest in the bottom left quadrant. *Melosira, Cyclotella, Crucigena, Microcystis, Anabaena* and *Scenedesmus* appear to favour silicon concentrations <5mg/ℓ.

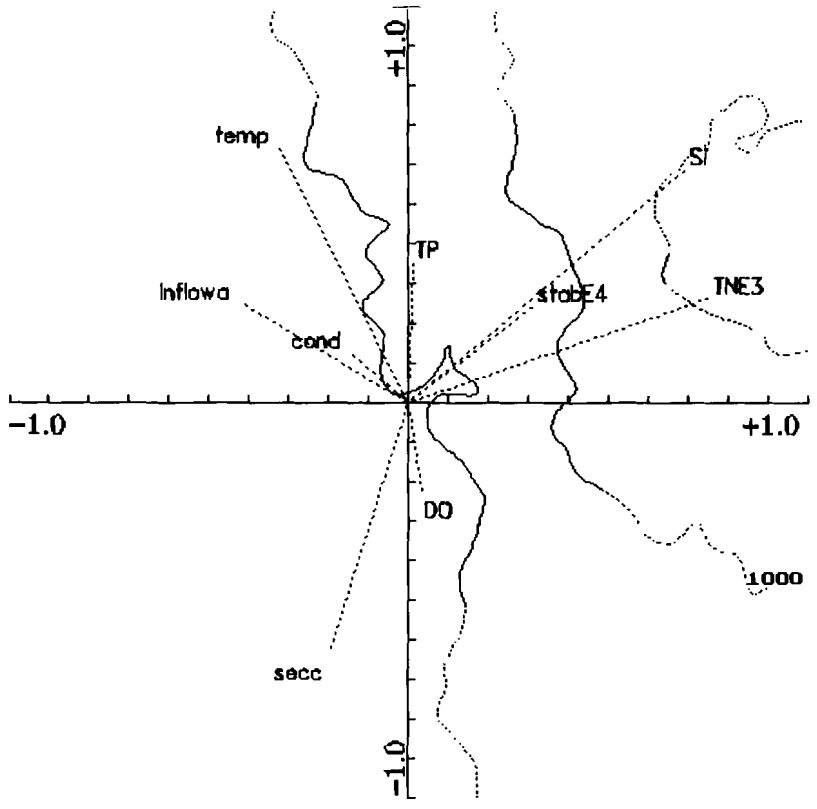

Figure 5.6 Total inorganic nitrogen (TNE3 mg/ℓ) trend data fitted to the ordination of sites from all seven lakes. Note contours (at 500, 1000 & 1500) on this ordination multiplied by a factor of 1000 i.e. 1000 contour = 1mg/ℓ N.

Figure 5.6 (to be interpreted in conjunction with Fig. 5.2) illustrates the trend of total nitrogen data over all the sites underlying this ordination. Total inorganic nitrogen concentration is highest in the top right quadrant of the ordination diagram whilst lowest in the bottom left quadrant. *Melosira, Cyclotella, Crucigena, Microcystis, Anabaena* and *Scenedesmus* appear to favour total inorganic nitrogen concentrations <0.5mg/ℓ.

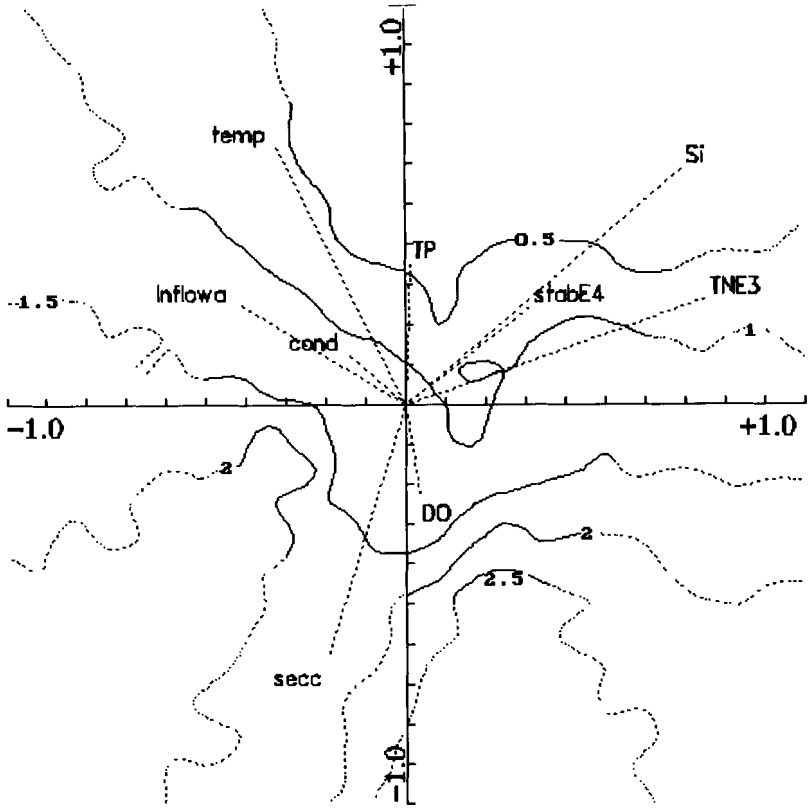

Figure 5.7 Secchi depth (metres) trend data fitted to the ordination of sites from all seven lakes.

Figure 5.7 (to be interpreted in conjunction with Fig. 5.2) illustrates the trend of secchi depth data over all the sites underlying this ordination. Sites in the lower half of the ordination diagram have clearer water (high secchi i.e. >1.5m), which appears to favour *Melosira, Cyclotella, Scenedesmus*, *Stichococcus* and particularly *Crucigena.* Moderately turbid water (secchi 1 to 1.5m) appears to favour the important blue-green algae.

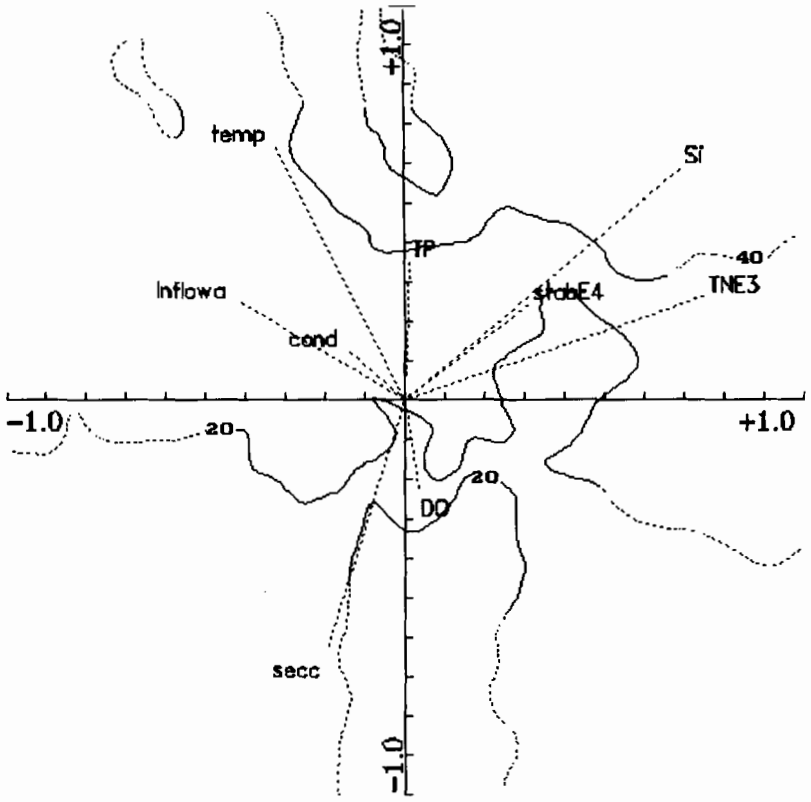

Figure 5.8 Total phosphorus (ug/ℓ) trend data fitted to the ordination of sites from all seven lakes.

Figure 5.8 (to be interpreted in conjunction with Fig. 5.2) illustrates the trend of total phosphorus (TP) data over all the sites underlying this ordination. TP concentration is highest in the top half of the ordination diagram whilst lowest in the bottom half. An increase in TP (i.e. between 20 to 40 ug/ℓ) favours the blue-green algae whilst most other algae appear to prefer TP concentrations <20 ug/ℓ.

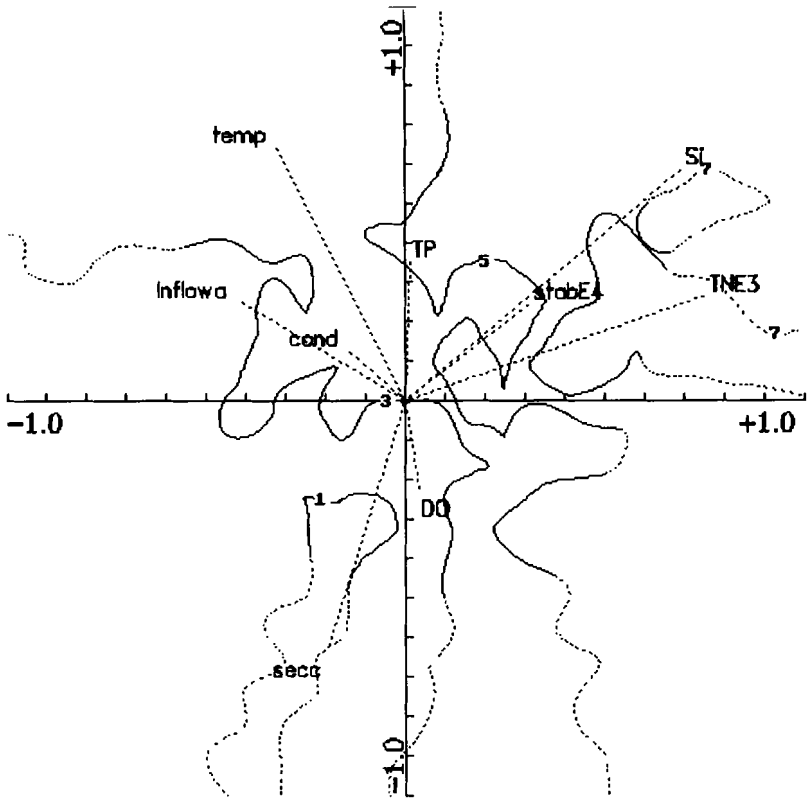

Figure 5.9 Stability (s^{-2}) trend data fitted to the ordination of sites from all seven lakes.

Figure 5.9 (to be interpreted in conjunction with Fig. 5.2) illustrates the water column stability trend over all the sites underlying this ordination. Sites in the upper right quadrant generally have more stable water column conditions whilst those sites in the lower left quadrant are more unstable. Low column stability appears to particularly favour *Crucigena* and to a lesser extent *Cyclotella, Melosira* and *Scenedesmus*. Moderate stability favours most other algal genera.

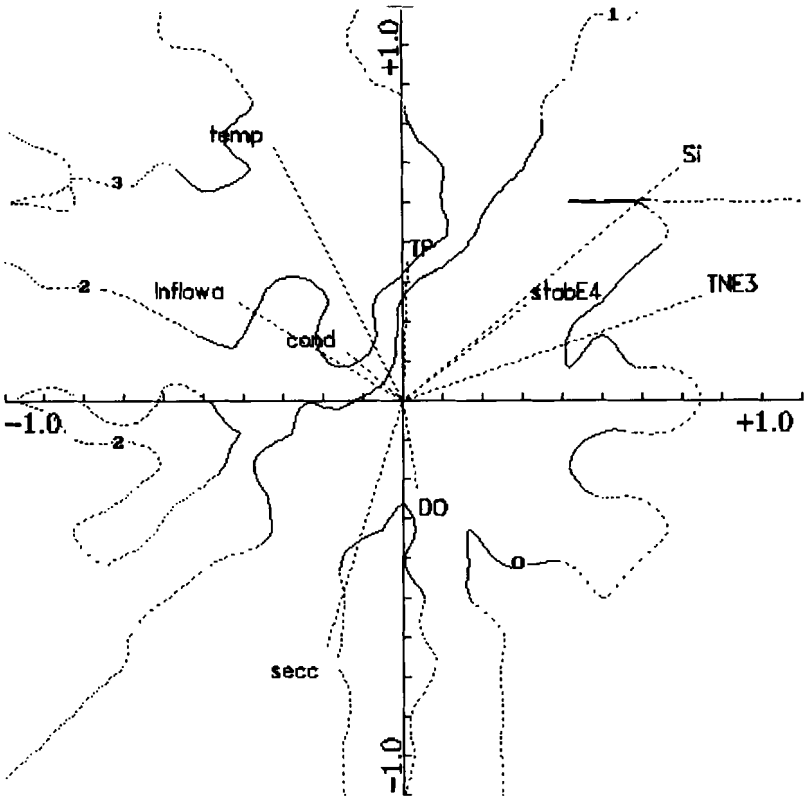

Figure 5.10 Trends of *Anabaena* counts (log data) fitted to the ordination of sites from all seven lakes.

Figure 5.10 (to be interpreted in conjunction with Fig. 5.2) illustrates the change in log abundance data of *Anabaena* over all the sites underlying this ordination. Sites in the upper left quadrant generally have high counts (100 to 1000 cells/mℓ) for this genus whilst those sites in the lower right quadrant have low counts (<1 cell/mℓ).

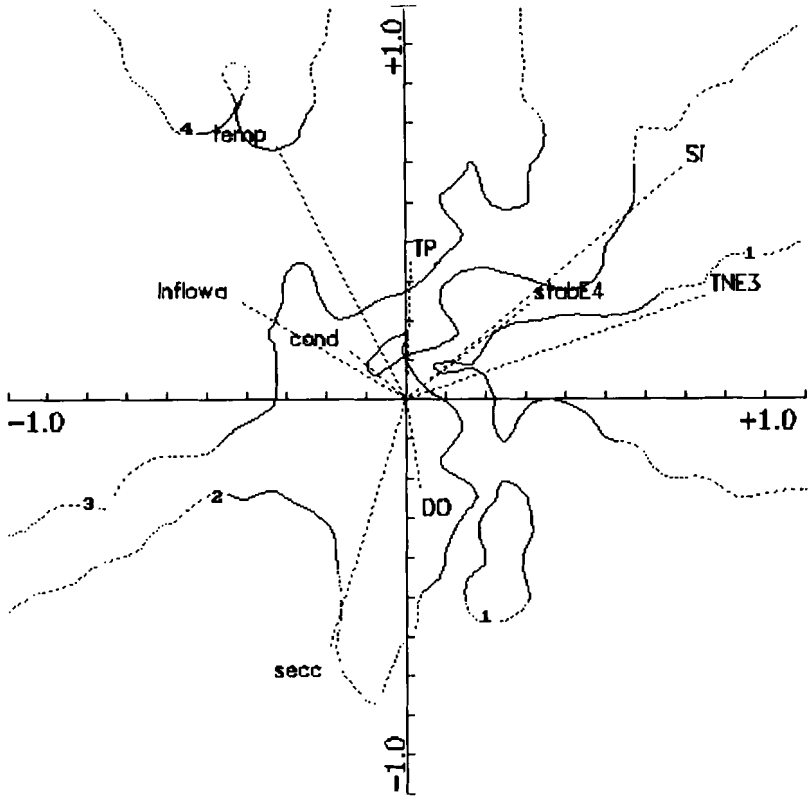

Figure 5.11 Trends of *Microcystis* counts (log data) fitted to the ordination of sites from all seven lakes.

Interpretation of Figure 5.11 (to be read in conjunction with Fig. 5.2) illustrates the change in log abundance data of *Microcystis* over all the sites underlying this ordination. Sites in the upper left quadrant generally have high counts (1000 to 10000 cells/mℓ) for this genus whilst those sites in the lower right quadrant have low counts (<10 cells/mℓ).

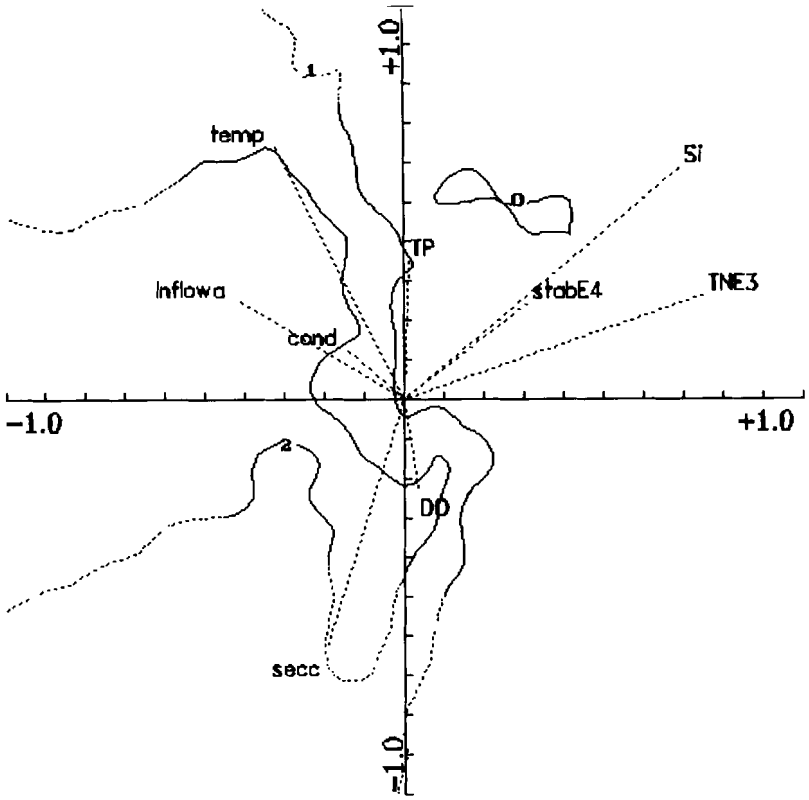

Figure 5.12 Trends of *Melosira* counts (log data) fitted to the ordination of sites from all seven lakes.

Interpretation of Figure 5.12 (to be read in conjunction with Fig. 5.2) illustrates the change in log abundance data of *Melosira* over all the sites underlying this ordination. Sites in the lower left quadrant generally have relatively high counts (>100 cells/mℓ) for this genus whilst those sites in the upper right quadrant have low counts (<1 cells/mℓ).

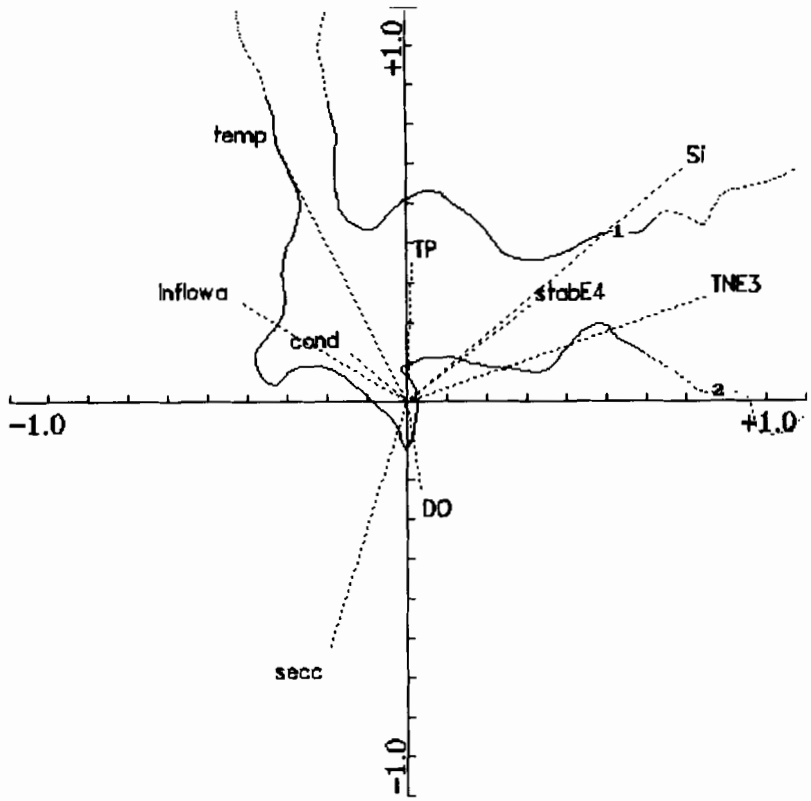

Figure 5.13 Trends of *Crucigena* counts (log data) fitted to the ordination of sites from all seven lakes.

Interpretation of Figure 5.13 (to be read in conjunction with Fig. 5.2) illustrates the change in log abundance data of *Crucigena* over all the sites underlying this ordination. Sites in the lower left quadrant generally have relatively high counts (>100 cells/mℓ) for this genus whilst those sites in the upper right quadrant have low counts (<1 cells/mℓ).

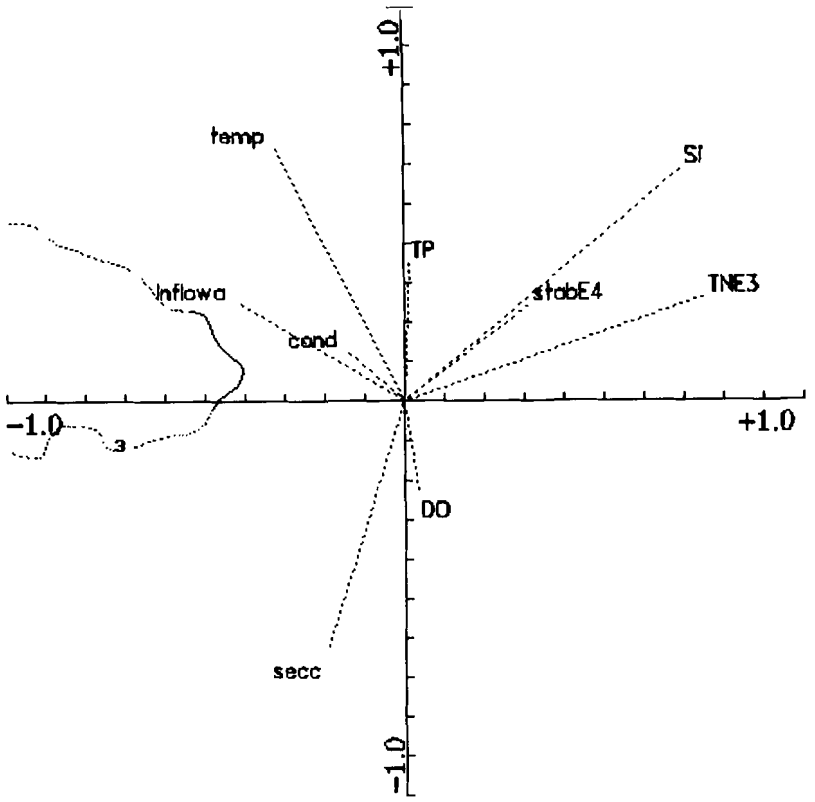

Figure 5.14 Trends of *Chlorella* counts (log data) fitted to the ordination of sites from all seven lakes.

Interpretation of Figure 5.14 (to be read in conjunction with Fig. 5.2) illustrates the change in log abundance data of *Chlorella* over all the sites underlying this ordination. As noted in the analyses this genus did not vary much across the sites and if anything tended to most dominant in sites on the extreme left of the ordination diagram.

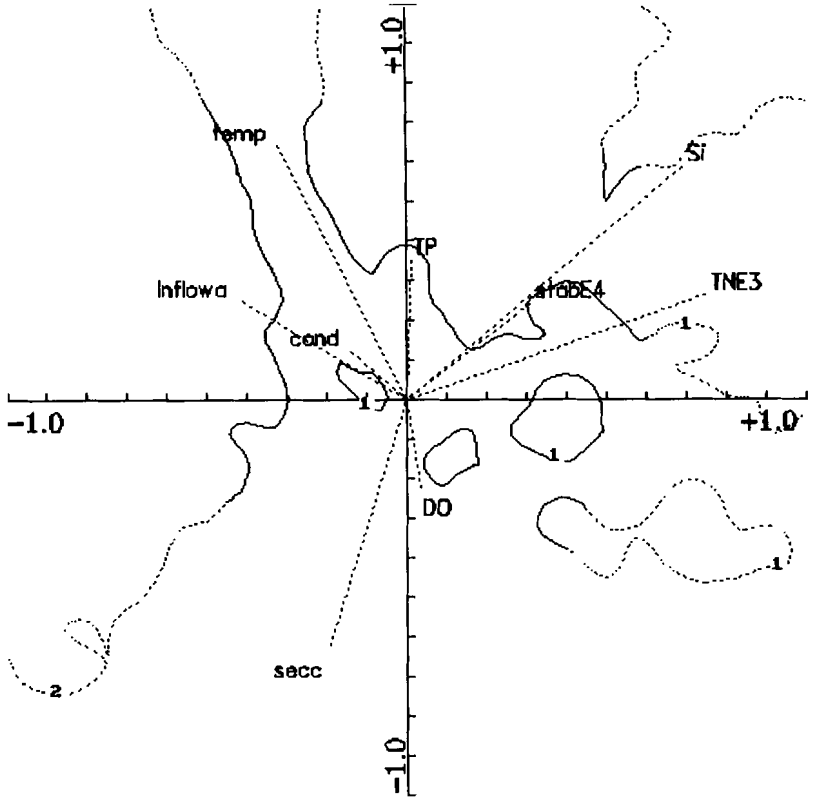

Figure 5.15 Trends of *Scenedesmus* counts (log data) fitted to the ordination of sites from all seven lakes.

Interpretation of Figure 5.15 (to be read in conjunction with Fig. 5.2) illustrates the change in log abundance data of *Scenedesmus* over all the sites underlying this ordination. Sites in the extreme left of the ordination have relatively high counts (>100 cells/mℓ) for this genus whilst those sites in the right have low counts (<1 cells/mℓ).

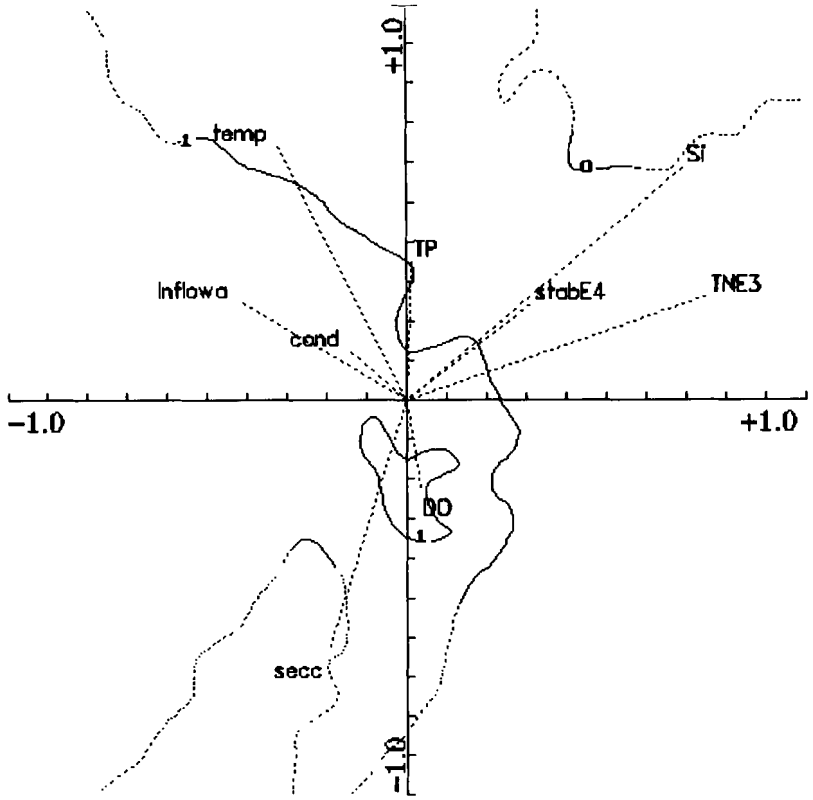

Figure 5.16 Trends of *Cyclotella* counts (log data) fitted to the ordination of sites from all seven lakes.

Interpretation of Figure 5.16 (to be read in conjunction with Fig. 5.2) illustrates the change in log abundance data of *Cyclotella* over all the sites underlying this ordination. Generally if this genus occurred it was only in low abundance. Sites in the extreme left of the ordination have relatively high counts (>10 cells/mℓ) for this genus whilst those sites in the right have low counts (<1 cells/mℓ).

5.3.2.1 Discussion of results from the kriging of key environmental variables and algal genera

Of the algal genera apparently responding to the key environmental gradients (Fig.5.1a&b and Fig. 5.2), *Anabaena* and *Microcystis* (blue-green algae) were most abundant (100-1000 and 1000-10000 cells/mℓ respectively) at the higher temperatures (20-25°C), inflows (>1000Ml/day) and conductivities that one would expect with summer conditions whilst favouring lower silicon (Si <5mg/ℓ) and total inorganic nitrogen (TN <1mgN/ℓ) values. Secchi disk depths (or water transparency) also tended to be lower when these genera were most dominant. These results were similar to the pattern noted by Lund and Davis (2000, development of blooms of *Microcystis aeruginosa* under summer conditions with shallow Secchi disk depths). The study supports the hypotheses of various authors (e.g. Reynolds, 1984; Smith, 1986; Havens *et al.*, 1998) who noted that blue-green algae dominate phytoplankton communities under conditions of reduced light.

The temperatures (20 to 25°C, Table 5.3) leading to the greatest abundances of the blue-green algae *Anabaena* and *Microcystis* in this study were in accordance with those noted in the literature (e.g. Romo & van Tongeren, 1995, Bahnwart *et al.*, 1999; An & Jones, 2000). Hammer (1964) found *Anabaena* blooming at 15 to 20°C with *Microcystis* blooming in a wide range of temperatures, from 19 to 25°C. Romo & van Tongeren (1995) had an even wider temperature tolerance with reports of *Microcystis incerta* reaching a maximum at 19 to 31 °C. Reynolds & Walsby (1975), Dawes *et al.* (1987) and Varis (1993) also noted this dominance of blue-green algae in the warmer summer months with Zohary & Robarts (1989) indicating that the growth of *Microcystis* is inhibited at temperatures below 15 °C.

Although with lower amounts of their variability explained by the key environmental variables (cf *Anabaeana* and *Microcystis*), *Navicula, Spermatozopsis* and *Mallomonas* appeared responsive to higher Si (> 5mg/ℓ) and TN (> 0.5mg/ℓ) and lower temperatures (15-20°C) and inflows (<100 Ml/day). Cooler (<20°C), clear waters (secchi >1.5m) with low total phosphorus (<20μg/ℓ) favour *Crucigena,* whilst waters generally low in TN (<0.5mgN/ℓ) and Si (<5mg/ℓ) have *Melosira, Scenedesmus, Cyclotella* and *Tetraedron* predominating. These waters are also generally clearer (higher secchi depths) and have lower water column stability. The low water column stability favouring the potentially important filter clogging diatoms *Melosira* and *Cyclotella,* and non-motile green algae *Scenedesmus* and *Tetraedron,*

is in accordance with Reynolds & Walsby's (1975) observation that conditions of stable stratification selects against diatoms and non-motile green algae. Therefore in water-columns with low stability (for example during late summer and winter, and in systems subjected to wind induced mixing) these diatoms may be expected to dominate. *Scenedesmus* and *Tetraedron* are not particularly important from a filter clogging point of view, although *Scenedesmus* has been implicated in taste and odour formation in some waters (e.g. Palmer 1959).

As with this study, Descy (1987) also noted a successional pathway of blue-green, and then *Melosira,* dominance in late summer. Various other authors (e.g. Bahnwart *et al.*, 1999, Kohler & Hoeg, 2000) have noted the dominance of *Melosira* during summer in European systems with Bormans & Webster (1999) illustrating a similar response for this genus in the River Murray.

Unfortunately little may be said about *Chlorella*, a dominant or sub-dominant species in many lakes (Fig. 2.1, proportional algal composition figures for lakes), as its variability is not well explained by the measured environmental variables.

It should be emphasised that the few 'key' environmental variables reflected in this RDA analysis, and subsequent ordinations, are closely associated/correlated with specific groups of other variables, i.e. their variations in the environmental data set are not singular, and because of this, all the associated variables are useful in interpretations of algae and environment. For example, during the PCA analysis of purely environmental data, all the nutrient loadings (i.e. total phosphorus, soluble reactive phosphorus and total nitrogen) came out strongly together at the one end of the fourth principal component or environmental axis. The common environmental parameter underlying this variability was 'inflow' volume into the lakes. Hence significance of 'inflows' (noted in this latter RDA analysis) needs to be seen in the light of its being strongly correlated to nutrient loads. Another example is the total inorganic nitrogen axis positively correlated with the TN:TP ratio. This implies that at low TN concentrations there are also likely to be low TN:TP ratios. The blue-green algae (*Anabaena* and *Microcystis*) in this study appeared to be more abundant at the lower end of this composite TN/TN:TP gradient (low TN:TP ratio, < 20) (see Figure 5.6). The literature supports this observation (e.g. Schindler 1977; Tilman *et al.*, 1982, 1986) with Jensen *et al.* (1994) noting blue-green algae seldom dominant at TN:TP ratios of 32 and higher. The level of around 20 noted in this study is also not far off the critical level of 29 observed by Smith (1983) for differentiating blue-green dominated lakes.

Figure 5.17 provides descriptive statistics for the N:P ratio in the respective lakes. Clearly some lakes had generally higher TN:TP ratios than others. The observed ranking of the lakes according to these ratios (from high to low median values) was as follows: Nungwane, Henley, Inanda, Hazelmere, Midmar, Albert Falls, Nagle and Shongweni. To some degree this also reflects the perceived eutrophication gradient for these respective lakes. Shongweni, with its associated blue-green dominated algal communities, was included in this figure to illustrate the extreme end of the eutrophication gradient.

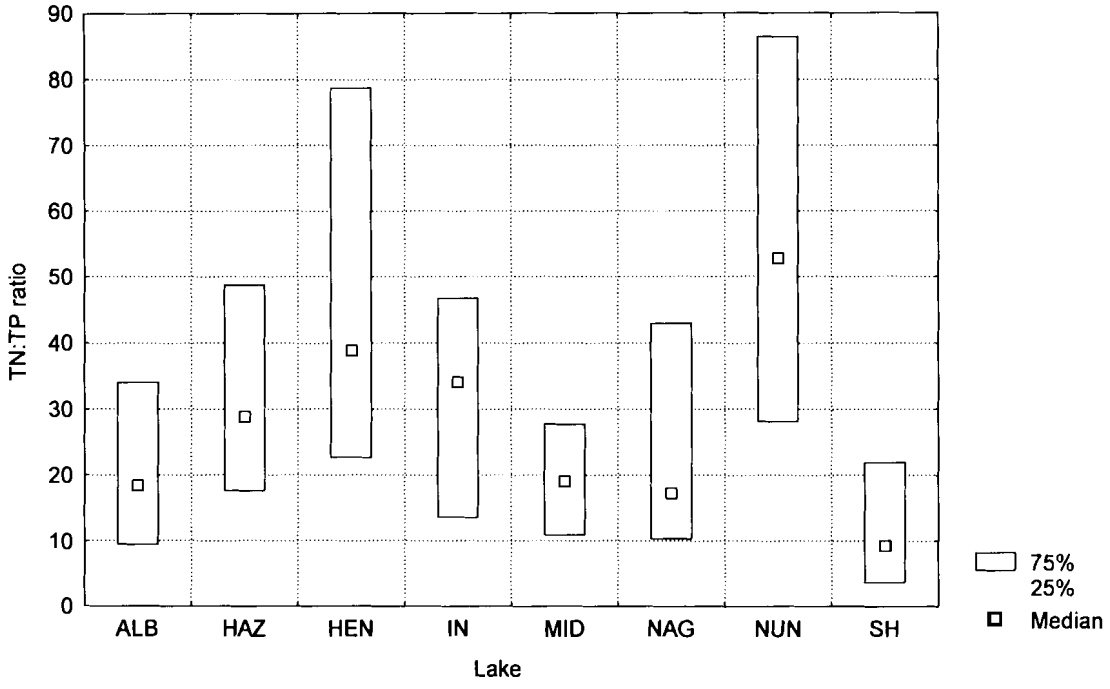

Figure 5.17 Median and 25 & 75percentiles for the Total Inorganic Nitrogen/Total Phosphorus (TN:TP) ratios in all lakes studied. Note Shongweni (as an example of a highly eutrophic lake) is included in this figure for comparative purposes. (ALB = Albert Falls, HAZ = Hazelmere, HEN = Henley, IN = Inanda, MID = Midmar, NAG = Nagle, NUN = Nungwane, SH = Shongweni).

The passively plotted lakes, discussed below, also reflected how Nagle, and to a lesser degree Midmar and Albert Falls, were at the low end of the total inorganic nitrogen (TN:TP) gradient (median values < 20). The figure indicating the proportional composition of the different types of algae in the respective lakes (Fig. 2.1 - proportional algal composition) shows the importance this TN:TP ratio appears to have with occurrence of the blue-green algae *Anabaena,* and particularly *Microcystis.* All the lakes with a low TN:TP ratio had a composition proportionally high in these algae. Midmar was an apparent anomaly, and hence interesting in this

respect as its TN:TP ratio indicates its **potential** to have blue-green algae problems. Its generally lower ambient water temperatures (another important variable explaining the variability of the blue-greens) probably prevents these algae from becoming dominant, and hence problematic, however. The implications of global warming, and associated increased water column temperatures, with potentially increased blue-green algal abundances, are worth noting for this important supply lake on the uMngeni river system.

Lakes Henley and Nungwane, with low proportions and abundance of blue-green algae (Figs. 2.1 & 2.2) were at the high TN:TP ratio (>35) end of the axis. Their algal population composition was dominated by green algae. The literature generally supports this observation, i.e. green algae favouring higher TN:TP ratios compared to blue-green algae (e.g. Jensen *et al.,* 1994).

A summary of the key environmental conditions favouring respective key algae is presented in Table 5.3. Only the more abundant algal genera, with significant amounts of their variability explained by analyses, were included.

Table 5.3 Summary of key environmental variable conditions favouring the maximal abundance of algae (only more abundant algal genera with appreciable amounts of their variability explained by key environmental variables (long arrows on ordination figures) are presented)

Variables/ parameters	Units	*Anabaena*	*Microcystis*	*Melosira*	*Crucigena*	*Scenedesmus*
Algal abundance	cells/ml	>100	>1000	>100	>100	>100
# Silicon (2)	mg/l	<5	<5	<5 (1-4)	<5	<5 (0.5-3)
# Secchi (2)	m	0.5-1.5	0.5-1.5	>2.0	>2.0	>2.0
Total Aluminum (2)	ug/l				(<200)	
Turbidity (2)	NTU				(<20)	
Iron (2)	mg/l				(<1)	
# Total Inorganic Nitrogen (3)	mgN/l	<0.5	<0.5	<0.5 (<0.25)	<0.5	<0.5 (<0.25)
TN:TP ratio (3)			(<10)			
TN:SRP ratio (3)			(<50)			
Nitrate (3)	mgN/l			<0.5 (<0.25)		
# Total Phosphorus (3)	ug/l	20-40	20-40	<20	>20	<20
TP:Si ratio (3)				(>0.01)		
# Temperature (4)	degrees	20-25 (26-27)	20-25 (25-27)	20-25	15-20	20-25
# Inflow (4)	Ml/day	>1000 (500-1000)	>1000 (500-1500)	100-500	100-500	100-500
Stability (5)	s-2			<0		

Notes:
Algal abundance in the table refers to the minimum algal count that could be expected in a sample under specified key environmental conditions.
Key environmental variables identified in RDA ordination analyses
Numbers in brackets (i.e. 2-5) after environmental variables refer to the Principal Component (PC) to which the environmental variable is most closely associated (from PCA analysis). For example in the table above, silicon, secchi, total aluminium, turbidity & iron are associated with PC axis 2 and is essentially a water clarity or turbidity axis.
Only algae displaying appreciable responses to the respective environment variables are displayed.
Values in brackets (under the algae) refer to their response after direct gradient analysis/graphing against respective environmental variables (§ 5.3.3)

To further supplement and add value to Table 5.3 results a series of Direct Gradient Analyses were performed (§ 5.3.3).

5.3.2.2 Direct gradient analysis

As a supplement to the interpretation of the multivariate analyses, classical direct gradient analysis (DGA) was performed to relate individual algal responses to important environmental variables (identified in the preceding Redundancy Analysis or which were strongly correlated with key environmental parameters identified in the Principal Components Analysis). This information was used in both confirming and supplementing the summary of algal responses to key variables as shown in Table 5.3. Only algae displaying a strong and meaningful response to the different environmental parameters are plotted in these figures. A polynomial best-fit line (with 95% confidence bands) was applied to the data. The formula for the best-fit line is given above each figure.

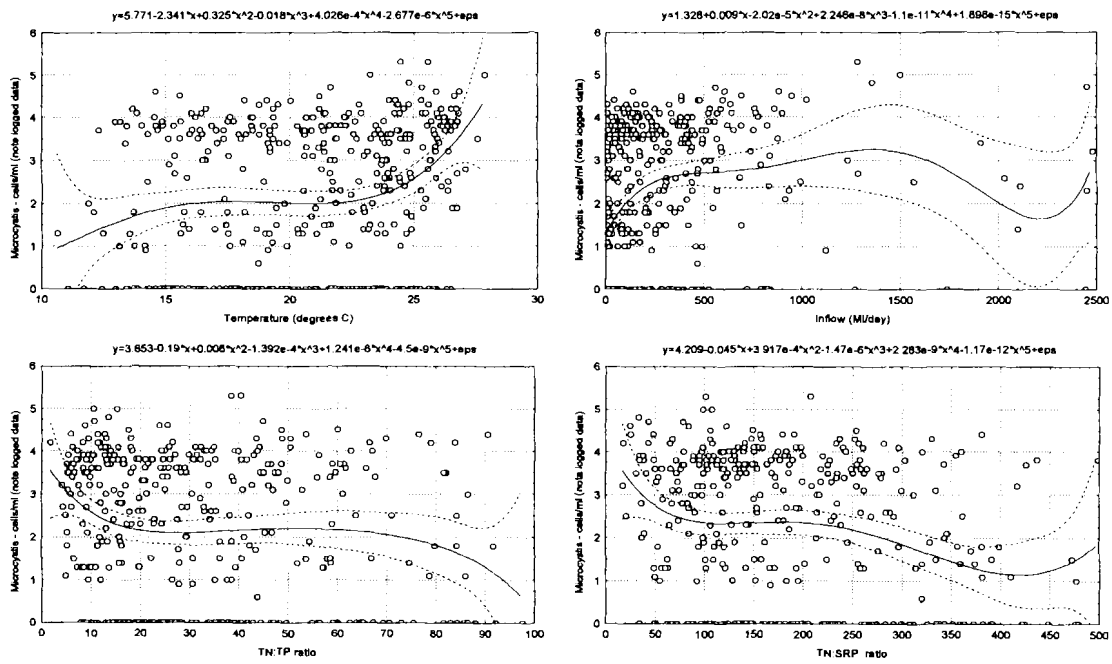

Figure 5.19. Direct gradient analysis plots (with 95% confidence bands) of *Microcystis* against various important environmental variables. Formula for the polynomial best-fit line given above respective graphs.

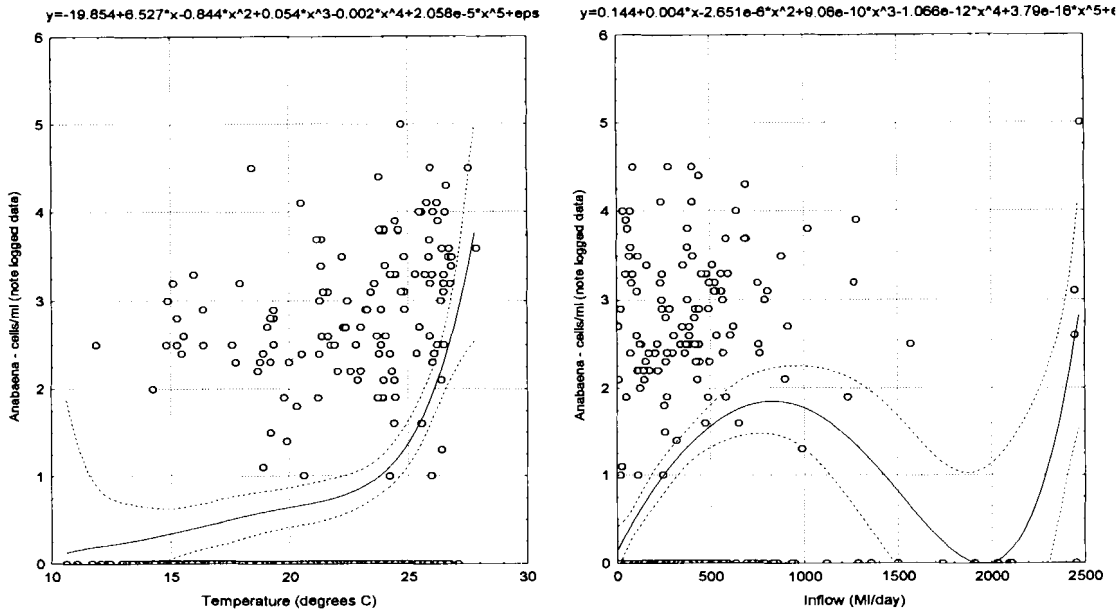

Figure 5.20. Direct gradient analysis plots (with 95% confidence bands) of *Anabaena* against various important environmental variables. Formula for the polynomial best-fit line given above respective graphs.

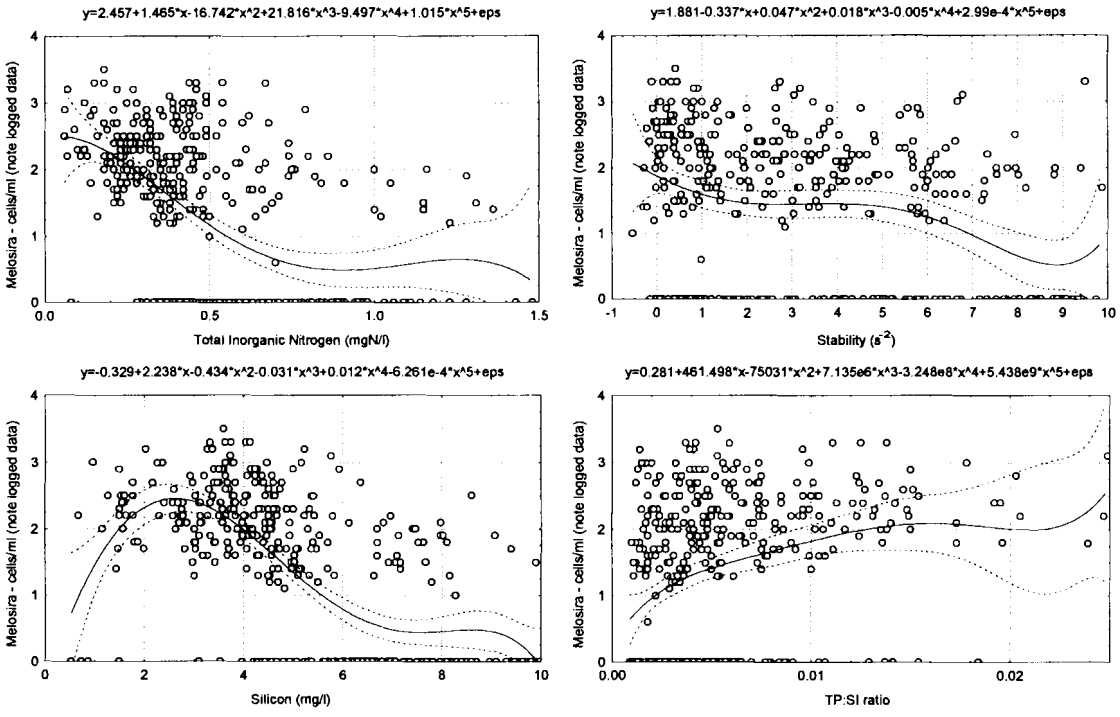

Figure 5.21. Direct gradient analysis plots (with 95% confidence bands) of *Melosira* against various important environmental variables. Formula for the polynomial best-fit line given above respective graphs.

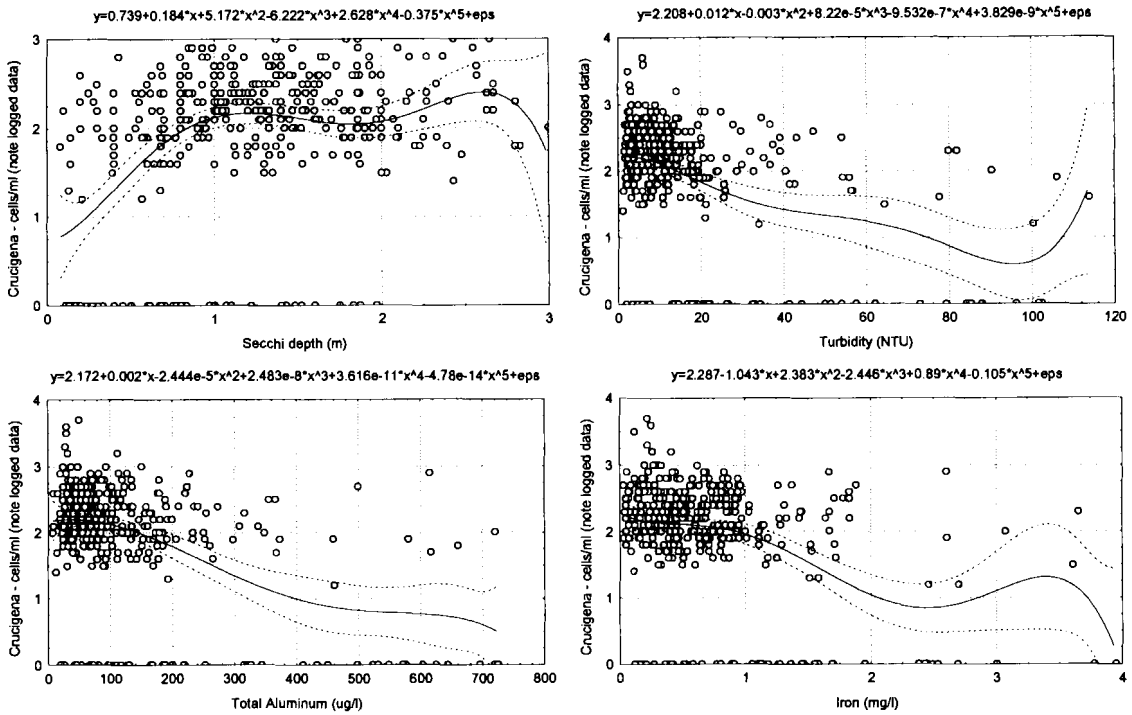

Figure 5.22. Direct gradient analysis plots (with 95% confidence bands) of *Crucigena* against various important environmental variables. Formula for the polynomial best-fit line given above respective graphs.

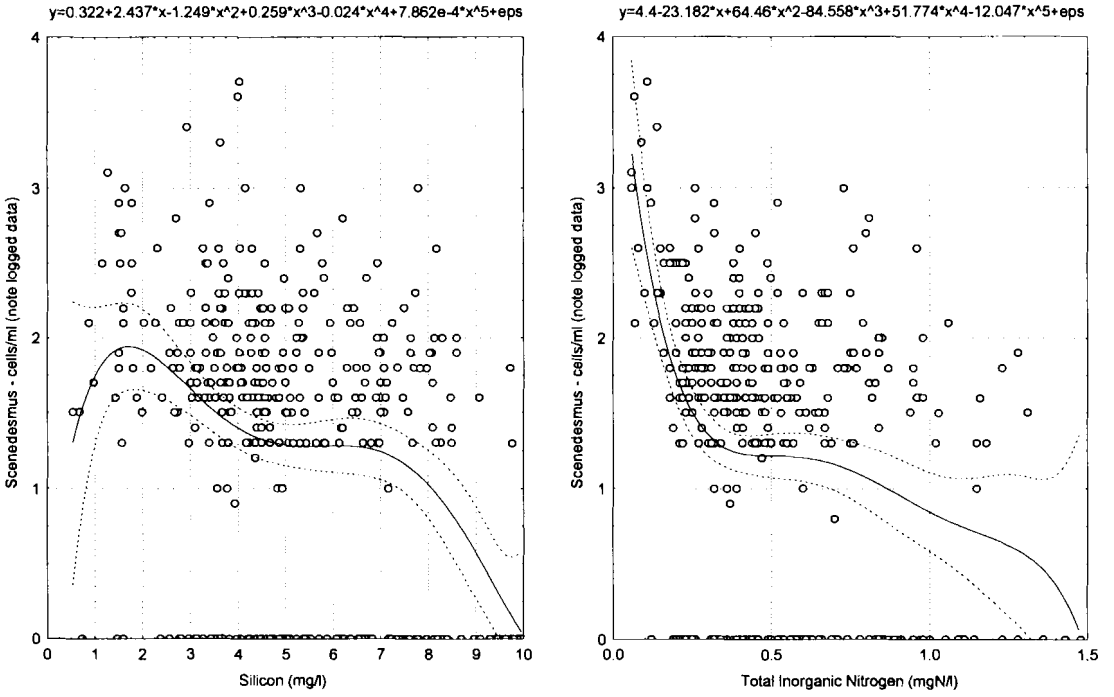

Figure 5.23. Direct gradient analysis plots (with 95% confidence bands) of *Scenedesmus* against various important environmental variables. Formula for the polynomial best-fit line given above respective graphs.

The direct plotting of algae against the different environmental variables also gives an indication of the range of applicability of the models developed, i.e. the

abundances (of algae) and values (of environmental variables) for which the semi-quantitative ordination models (and kriging) holds true.

5.3.2.3 Results and discussion of the passive plotting of 'lakes' and 'months'

After analysing the response of algae to 'key' environmental variables, it was possible to examine the distribution of the actual lakes and seasons (months) in relation to these same variables (Figure 5.1c). It is apparent that the spatial variation among lakes is more important than seasonal or annual differences. Hazelmere (Fig 5.1c), in the early part of the summer rainy season (September to December), is dominated by conditions associated with high silicon and total inorganic nitrogen and low water clarity (low secchi and high turbidity). The algal variability observed in Henley and Nungwane appear to be dominated by relatively high silicon and total inorganic nitrogen and correspondingly low temperatures and inflow volumes associated with spring (August to September). Having just come through the winter, where water temperatures and inflows are at their lowest, this would be a reasonable explanation for the observed pattern. August is traditionally one of the windiest months of the year in KwaZulu-Natal, and combined with a non- or weakly-stratified water column at the end of winter, would probably have higher silicon concentrations from colloidal sediments entrained by water column mixing. Inanda and Nagle, on the other hand, appear to have higher temperatures and inflows and lower silicon and total inorganic nitrogen dominating their algal distribution and hence ordination, particularly during the mid to late summer period from January through to April.

5.4 Intrinsic, 'lake-unique', differences and its effect on observed algae

5.4.1 Introduction

It has to be acknowledged that not all of the parameters that may be affecting algae in a lake can always be measured or quantified in a biologically meaningful way. These parameters may be of a biological, physico-chemical or seasonal nature (or various combinations of all of these). It is this un-quantified element of variability in a lake that prevents one from being able to fully understand, and therefore predict, all the dynamics and processes that result in the abundance and variation in algal populations observed. Often certain parameters may be unique to (e.g. basin morphometry) or vary uniquely in (e.g. local weather patterns) respective lakes, thus giving that lake its unique or intrinsic set of characteristics affecting resident algal populations. Analyses to determine the response of algae to this 'uniqueness' of

respective lakes (intrinsic or lake-unique differences) were conducted. In this way lakes with similar characteristics could be expected to have similar algal heterogeneity.

Concurrent with this analysis, environmental variables were examined to determine possible environmentally based reasons for the differences (or similarities) observed between lakes. Results also helped to determine if there were processes occurring in these lakes that had not been accounted for in the environmental variables measured/quantified for this project.

The mean seasonal (monthly) effects/trends were removed as co-variables, with environmental variables overlain as passive variables. The rationale for the plotting of passive variables was as described earlier. The focus, with passive plotting of environmental parameters, was on the spatial differences between lakes and how, in an unrestricted manner, environmental effects may be related to these spatial differences.

5.4.2 Results and Discussion

Results are given in Table 5.4 and Figure 5.24.

Table 5.4 Key results from analyses where the lake-unique effects were related to algae in all lakes except for Shongweni

Axes	1	2	3	4	Total variance
Eigenvalues	0.09	0.05	0.02	0.01	1.00
Algae-environment correlations	0.77	0.67	0.61	0.43	
Cumulative % variance					
of algal data.	10.1	15.3	17.1	18.0	
of algae-env. relation	53.1	79.9	89.5	94.5	
Sum of all unconstrained eigen values (after fitting covariables)					0.92
Sum of all canonical eigenvalues (after fitting covariables)					0.18
Algal variability explained by analysis					17.6%

Key findings here were that eigen-values and algae/environment correlations were marginally higher than where only key environmental variables were related to algal variability (Table 5.4 vs. Table 5.1). The implication is that effect/s or process/es unique to different lakes are slightly better able to describe the observed algal heterogeneity than the key environmental variables alone. These processes, captured by the spatial separation of the lakes, were statistically significant. It would therefore appear that something, which varies amongst lakes, and which was either un-quantified in this study, or was not included in the reduced 'key' environmental data set, must be affecting the observed algal heterogeneity. Further interpretation

of the data (§5.4.4 - passive plotting of PCA axes) indicates that these un-quantified effects were probably due to currently unmonitored environmental variables.

On the basis of its algal composition, and how this is related to spatial differences between lakes, Hazelmere again appears distinctly different from all other lakes (Fig 5.24a), ordinated as it is at the extreme end of the second RDA axis. This difference appears primarily associated with higher turbidities (low secchi Fig. 5.24c). Lakes Hazelmere, Nungwane and Inanda (and to a lesser extent Nagle) are all at the extreme edges of the ordination (Fig. 5.24a), illustrating their relative differences. Given that they are all on entirely different river systems (except Nagle and Inanda on the uMngeni River), with different water physico-chemistry, lake morphometry, local climatic regime, etc., this is not surprising.

There is an interesting trend in lakes from generally upper catchment (Henley – on the right, Fig. 5.24a), down the uMngeni system sequentially to the coastal lakes (Inanda – on the left, Fig. 5.24a). Nungwane is an apparent anomaly in this trend as it is generally classified as a 'coastal' lake but is positioned at the upper end of its catchment. From its relative positioning on the ordination diagram (Fig. 5.24a), Nungwane appears to have a greater affinity to the upper catchment. Catchment position would, therefore, appear to play a more important role on observed algae in the lakes than does proximity to the coast. This picture was repeated in the PCA ordination of lake sites based on their environmental variables (Chapter 4).

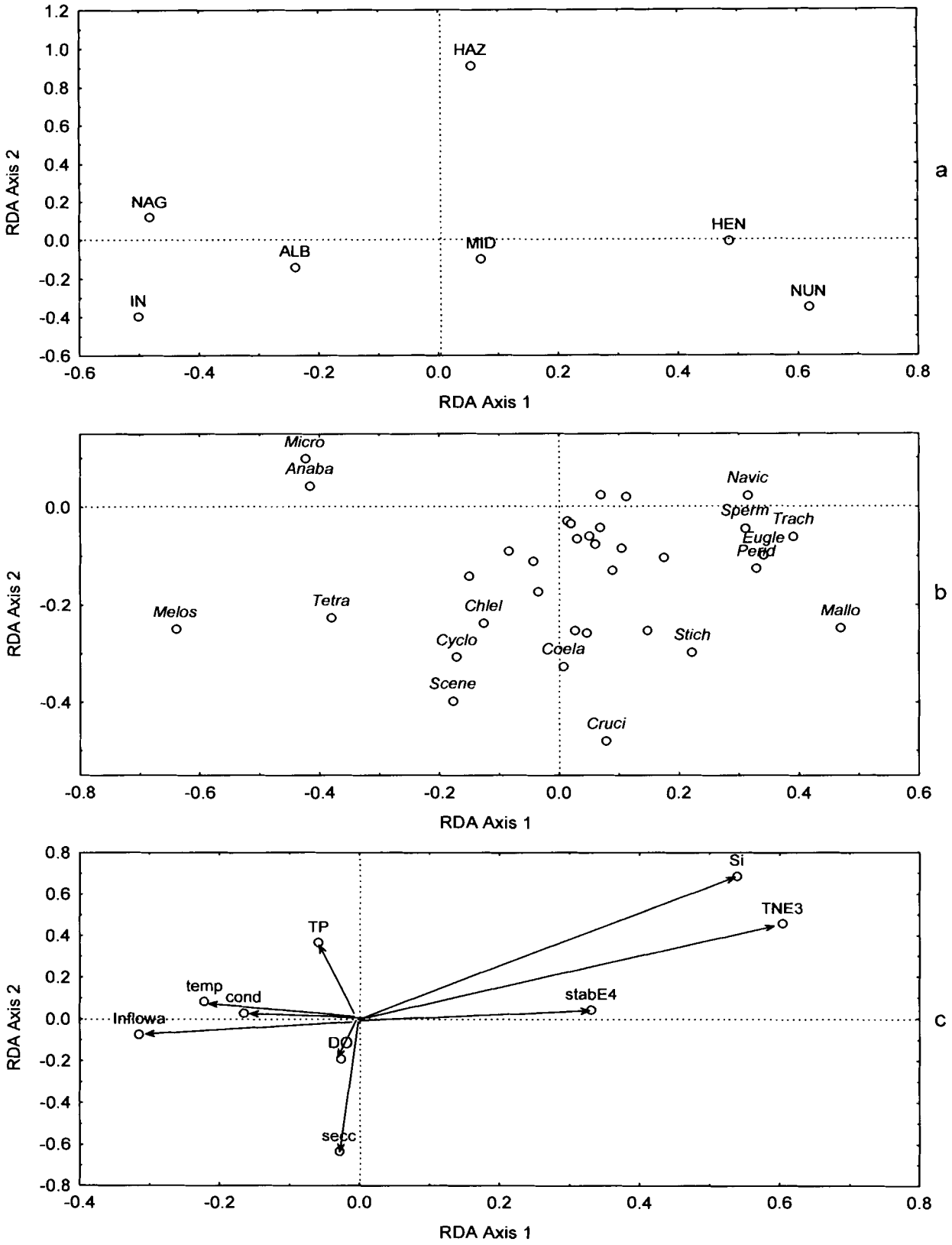

Figure 5.24. Ordinations where the effects unique to lakes, and how these were influencing algae, were the focus of interest (a). Environmental variables were plotted as passives (c). Genera with >10% of their variation explained by the analysis plotted and labelled (b). Abbreviations as per Tables 2.1 & 2.4.

5.4.2.1 Discussion of results from passively ordinated key environmental variables

Passive plotting of environmental variables over the ordinated algal data (analysed and ordinated with lakes the constraining external or explanatory variables) indicates that the observed trend/difference between lakes appears to be associated with progression down the catchment and is related to factors associated with increasing inflow, temperature and conductivity. The progressive trend down catchments is particularly evident for lakes on the uMngeni River system. There are also parallel decreases in stability and lowered silicon and total nitrogen values down the catchment. This response of lakes was similar to that noted where environmental variables were the key variables of interest and lakes had been plotted as passive variables. Again Henley and Nungwane are characterised by higher stabilities, silicon and total inorganic nitrogen and lower inflows and temperatures, whilst the lower uMngeni River lakes (Nagle and Inanda) are characterised by the opposite set of conditions. It should again be noted however, that the analysis does underscore there being some 'unmonitored' factors that also account for differences between lakes.

Silicon and total nitrogen concentrations (long arrows on the first axis of Fig 5.24c) appear to be the most important variables associated with the observed spatial differences among the lakes (Fig. 5.24a) - particularly Henley and Nungwane compared to all the other uMngeni system lakes. This implies that these two environmental variables are important in distinguishing between these groups (Henley and Nungwane, and all others). The lakes also appear to have more stable water column conditions compared to the other lakes. Water column stability may be affected by local wind conditions and certainly Henley is acknowledged to be a sheltered lake, situated as it is in the hills above Pietemaritzburg (Walmsley & Butty 1980).

The warmer conditions in lakes at the lower end of the uMngeni River system (Inanda and Nagle) appear to produce less stable water column conditions. The typically steady onshore easterly coastal breezes, combined with the east – west valley orientation (funnelling the breezes up from the coast), may be a possible explanation for the reduced stability in these more coastal lakes. Unfortunately no measure combining duration, direction and speed of wind was available, or derived, to test their effect on water column stability.

The algal distribution on the ordination (Fig 5.24b, and in relation to passively plotted environmental variables) is not markedly different from that observed in the previous analysis examining algae and restricted environmental effects.

Algal genera with a significant proportion (>10%) of their variance explained by the spatial effects associated with the different lakes are indicated in Table 5.5. As with the previous analysis, *Melosira* again has a significant proportion of its variance explained by the spatial effects associated with different lakes. Figures 2.1 (proportional composition) and 2.3 (abundance) indicate that this genus is only really in appreciable numbers and proportionally abundant in Lakes Midmar, Albert Falls, Nagle and Inanda. This genus is potentially problematic to the water industry, where it can reduce filter runs, and therefore appears to be most significant on the uMngeni system lakes. *Navicula, Euglena, Spermatozopsis, Trachelomonas, Peridinium* and *Mallomonas* appear to be restricted to Henley and Nungwane. Their abundance (Tables 2.3 & 2.4) and relative proportion (Fig. 2.1) in these lakes is never significantly high and, *ceteris paribus*, are therefore unlikely to negatively impact water treatment.

Table 5.5 Ranked algal responses (in terms of variance explained) to lake unique effects (only genera with >10% variance explained shown)

Genera	Cumultive % variance explained
Melosira	54.42
Mallomonas	32.58
Peridinium	26.46
Tetraedron	26.13
Crucigenia	25.72
Scenedesmus	23.66
Anabaena	22.92
Stichococcus	22.22
Microcystis	19.48
Spermatozopsis	18.57
Trachelomonas	18.12
Coelastrum	15.17
Euglena	15.12
Cyclotella	14.86
Navicula	10.67
Chlorella	10.65

5.4.2.2 Passive plotting of PCA axes

Preceding analyses identified certain conditions, which varied amongst lakes, and which were either,

- un-quantified in this study, or were
- not included in the reduced 'key' environmental data set,

and which appeared to have a significant effect on observed algal heterogeneity.

To assist with understanding which of these scenarios was most likely, Principal Component (PC) axes (from analyses on the purely environmental data set, Chapter 4) were passively analysed in relation to the species axes in RDA analyses - where lake differences and seasonal trends were removed. Indications from Table 5.6 (correlations of passively plotted PCA axes) are that including further key environmental variables from the original data set, or using the original PCAs will not help identify these unknown factors. The first five PCs (with associated key environmental variables with high loadings on respective axes, Chapter 4, Table 4.5), when passively overlain onto the algal data in RDA, are the only axes to show a reasonably strong co-relation with observed algal variability.

The implication of this result is that further axes (with associated environmental variables with high loadings on these axes) will not add significantly to the explanation of algal variability. It therefore appears that some parameter/s which were un-quantified or unmeasured in this study, were accounting for the greater proportion of observed algal heterogeneity.

Table 5.6 Correlation of passively plotted PCA axes with species axes in RDA analyses where lake differences and seasonal trends were removed.

	SPEC AX1	SPEC AX2	SPEC AX3	SPEC AX4
PC_1	-.2238	-.0176	-.1049	.2001
PC_2	.3280	.5775	.0235	-.0083
PC_3	-.3417	.0722	.2061	.0065
PC_4	-.2253	.0873	-.1791	-.0301
PC_5	-.2200	.1580	-.0040	.0453
PC_6	-.1448	.0487	.0833	-.0075
PC_7	-.0462	.1378	.1254	-.0743
PC_8	-.0751	-.0172	-.0354	-.0478
PC_9	.0895	-.1954	-.0806	-.0859
PC_{10}	-.0184	-.1512	-.1064	-.0257
PC_{11}	.0826	-.0531	.1351	.0306
PC_{12}	.1699	-.0912	.2028	-.0098
PC_{13}	-.0416	.0169	-.0143	-.1543
PC_{14}	-.0662	.0748	.0911	-.0527
PC_{15}	.0991	-.1044	-.0327	-.1059

Note: only the first 5 PCA's have a reasonable association with lake differences (especially PC_2 and PC_3 - turbidity/secchi and N&P gradients respectively).

5.5 Seasonal changes in lakes and its effects on algal populations

5.5.1 Introduction

Seasonal trends (both within the year, i.e. on a monthly basis, and annually i.e. over the duration of the study) were examined to determine if these were significantly

affecting observed algal populations in the study lakes. Again RDA was used to establish the relationship. The mean spatial effects/trends of individual lakes were removed as co-variables, with environmental variables overlain as passive variables.

5.5.2 Results and discussion

Results are reported in Table 5.7 and Figure 5.25

Table 5.7 Key results from analyses where the temporal effects of months were related to algae in all lakes except for Shongweni

Axes	1	2	3	4	Total variance
Eigenvalues	0.03	0.02	0.01	0.01	1.00
Algae-environment correlations	0.56	0.55	0.55	0.41	
Cumulative % variance					
of algal data.	3.3	5.7	7.0	7.6	
of algae-env. Relation	35.7	61.7	76.1	83.4	
Sum of all unconstrained eigen values (after fitting covariables)					0.82
Sum of all canonical eigenvalues (after fitting covariables)					0.08
Algal variability explained by analysis					7.5%

Key findings here were that the first two axes of Figure 5.25a had eigen values, species/environment correlations and cumulative percentage variance of species/environment relations (Table 5.7) lower than all other effects investigated (key environmental and lake unique effects - Tables 5.1 & 5.4 respectively). However, although the values were low, Monte Carlo permutation tests indicated that the effects were statistically significant ($P<0.01$, ter Braak 1987-1992). The implication is that when seasonal effects are compared to all other effects (key environmental, and lake-unique effects), seasonality accounts for a relatively small amount of algal variability. There is nevertheless still a clear and statistically significant seasonal pattern in the observed algal data, which although unmeasured and un-quantified in this study, is affecting the observed algal heterogeneity.

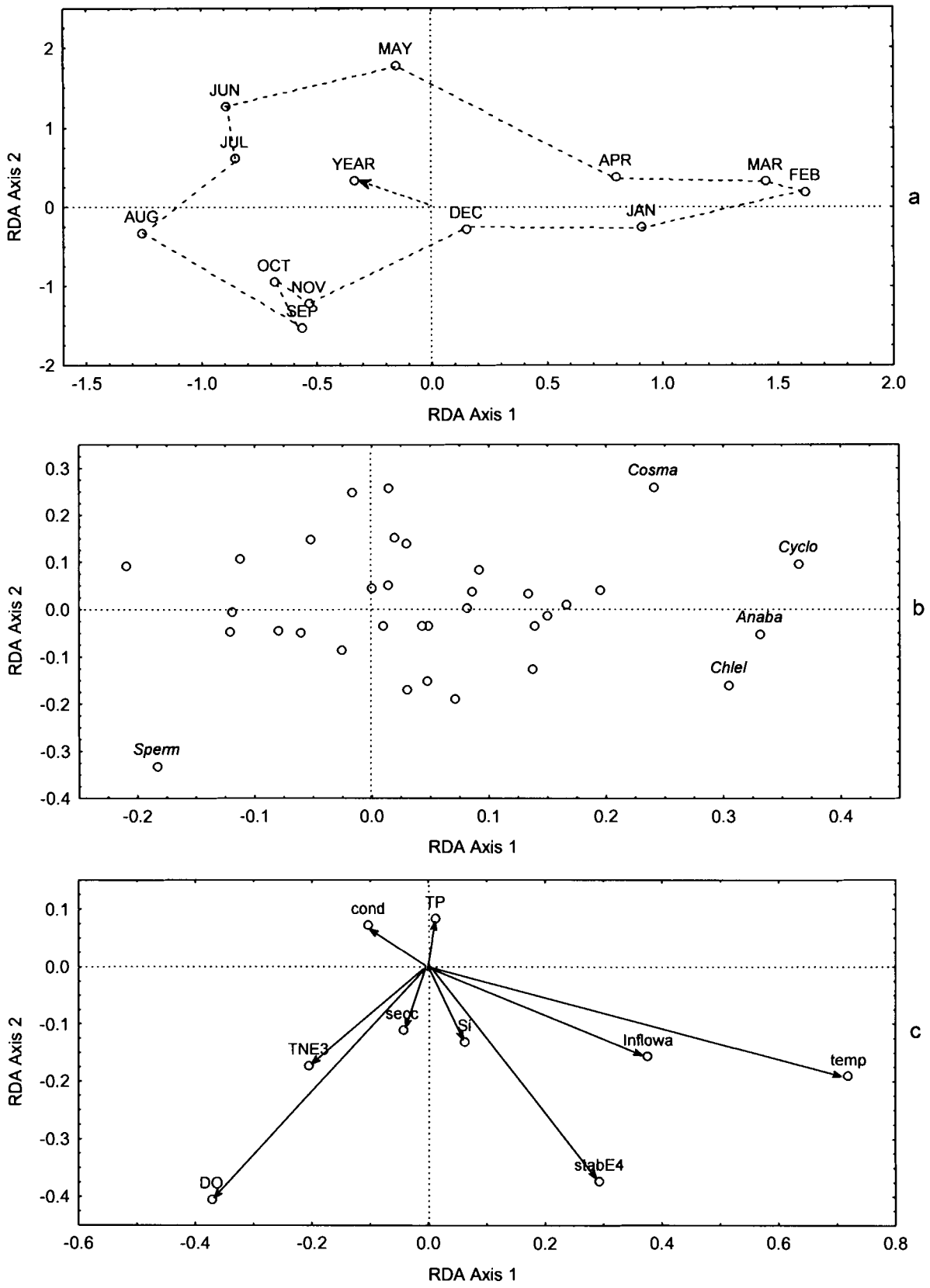

Figure 5.25 Ordinations from RDA where the temporal effects of months were the focus of interest (a). Environmental variables plotted as passives (c). Genera with >10% of their variation explained by the analysis plotted and labelled (b). Abbreviations as per Tables 2.1 & 2.4

For the duration of the study, long-term directional algal community changes (succession) that may have been happening within the lakes (represented by YEAR on Fig. 5.25a) are less significant than monthly changes (shorter term successional patterns). The relatively short length of the YEAR axis (longer-term succession), compared with the distance monthly centroids are apart (shorter-term succession) on Figure 5.25a, provides the evidence for this conclusion. The implication of this result is that there is no overall successional trend in the algal populations in the lakes under study. In other words, the systems are relatively stable within the annual seasonal changes represented by months and are not showing directional movement toward a particular type or community of algae dominating the systems over time. Had the annual successional change been larger than seasonal successional change, and in a particular direction of algal community, then this may have been some cause for concern – particularly if the algal community structure were dominated by problematic genera.

The implications of these results (little current successional change in the current algal communities) need not lull the relevant catchment management agencies into a false sense of comfort, as the Shongweni example has shown. In this latter case poor water quality from catchment processes and activities resulted in the decommissioning of the dam as a potable water supply (Wilson 1992).

Algal genera that appear to have a significant proportion of their variability (>10%) explained by seasonal effects are shown in Table 5.8. Fewer genera were found, with significant portions of their variability explained, compared to either of the previous two analyses (key environmental and lake unique effects). This emphasises the lower significance seasonal (monthly) effects have on algal populations compared to pure environmental and spatial effects (associated with different lakes).

Table 5.8 Ranked algal responses (in terms of variance explained) to the effects of season (months) (only genera with >10% variance explained shown)

Genera	Cumulative % variance explained
Chlorella	20.52
Spermatozopsis	20.13
Cyclotella	17.23
Cosmarium	15.37
Cryptomonas	13.43
Anabaena	12.71

Where *Chlorella* previously had a low proportion of its variability explained by key environmental (7.42%), and lake unique effects (10.65%), the analysis of seasonal effects accounted for a greater proportion of its variability (20.52%). This result for *Chlorella* was significant considering that the alga was dominant to subdominant in many lakes (Figs. 2.1 (proportional composition) and 2.4 (abundance)). There is clearly some un-quantified seasonal factor(s) in the environment, of either biotic or abiotic origin, which is affecting some of this algas' variability.

As a generalisation, in terms of the successional pattern observed within the year (i.e. from month to month), the sequence of abundance is from *Chlorella* and *Anabaena* (in the height of summer in the early part of the calendar year, when inflows and temperatures are greatest) through to *Cyclotella* and *Cosmarium* in autumn (March/April). *Spermatozopsis* is likely to be more abundant in spring (August to October) when factors associated with dissolved oxygen concentrations in the water appear to be higher.

Reference to Figures 2.1 - 2.4 (proportional algal abundance and abundances) indicate that it is only *Chlorella* and *Anabaena* which are ever really numerically and proportionally abundant. Of these, it is the latter that is the most significant affecting water treatment processes due to its potential to form taste and odour forming compounds (see Chapter 6). The analysis indicates that January through to April are months when this genus may be expected to become most abundant and, therefore, cause treatment problems. Water treatment plant operational experience supports this observation.

The passive plotting of environmental variables onto this analysis indicates that elevated temperatures and inflows (and correlated factors) are closely associated with this mid- to late-summer bloom of *Anabaena* and *Chlorella*.

Another noteworthy feature of the analysis is the apparently relatively large shift in composition (or succession) occurring at certain times of the year and clearly associated with changes in season. Compositional change is particularly evident during autumn (March/April to May/June), and then again in early summer (September to November) through to late summer (January/February) when conditions appear to change quite dramatically from the earlier period. The shift is seen in Figure 5.25 in the distance centroids (representing months) are apart from

one another (the greater this distance, the greater the degree of successional change in the algal communities in lakes).

The shift in seasonal pattern is most probably associated with changes in the physical stability of the water column (establishment and breakdown of thermal stratification within the water column). This conclusion is supported by the extent to which environmental variables, which may traditionally be associated with physical conditions in the water column (temperature, inflow, stability and dissolved oxygen), are related to algal variability uniquely associated with this analysis of seasonal effects. Typically, stratification of the water column in the lakes studied occurs in early summer (September/October) with de-stratification in autumn (April/May). An example of this is given in Figure 5.26, which illustrates the thermal stratification pattern (temperature isopleths) in Inanda during 1993. Water column stratification are typically periods of major physico-chemical change in the water column that could be expected to have a significant impact on algae resident there (e.g. Wetzel, 2001). For example, it is during periods of de-stratification that plant nutrients are typically entrained from bottom, nutrient rich waters, brought to the photic zone of the water column and available to algae present in this zone. Algal blooms are often most prevalent during these periods.

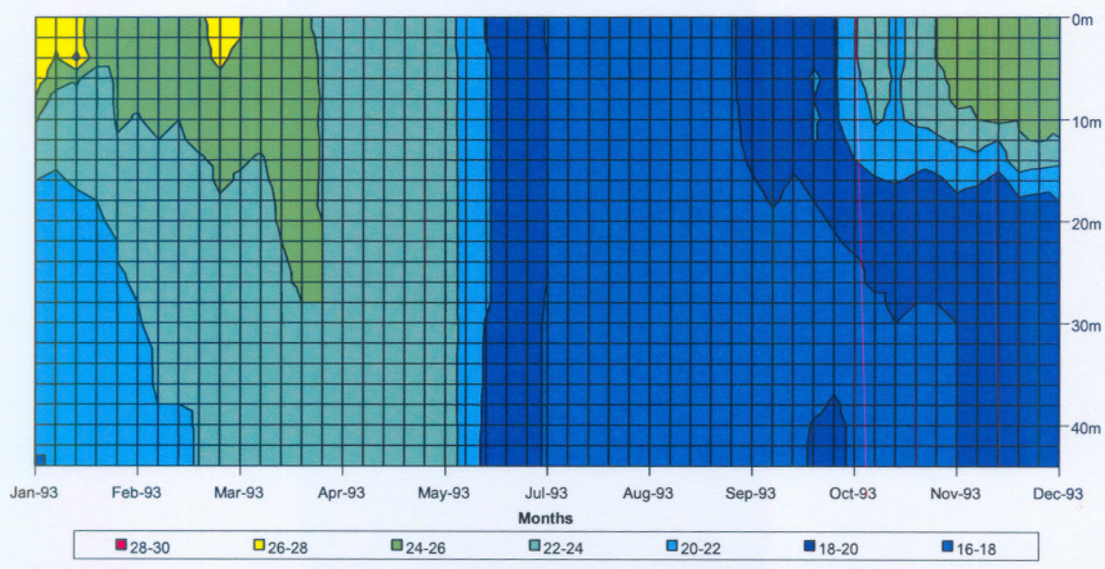

Figure 5.26 Temperature isopleths (thermal stratification patterns) in Inanda main basin during an example year (1993) of the study period (to illustrate typical stratification of the water column).

In previous analyses (environmental and spatial effects) dissolved oxygen appears to make a limited contribution to the explanation of variability underlying the algal populations (Figs. 5.1a & 5.25c – relatively short arrows on ordination diagrams). Examination of environmental variables in the analysis probing seasonal effects, however, indicated that DO is closely related to seasonality in the algal variability (Fig. 5.25c – long arrow). This observation is in accordance with the understanding that the oxygen profile in water columns changes significantly during stratification. Water column 'stability' is also strongly influenced by the stratification patterns of the water column and as such also appears to be more strongly related to the seasonal aspects of algal variability than had been noted in previous analyses.

5.6 Establishing statistically rigorous predictive models relating algae to the environment

Attempts at classical multiple regression modelling, relating specific algae to measured environmental parameters, were not particularly successful. The principal reasons were due to: the distribution of the algal genera and total algal abundance data being not normally distributed and hence skewed in some way (see for example Chapter 2, Fig. 2.2). This is to be expected and is common in such biological data with relatively occasional occurrences of high counts. Therefore, if the actual data are modelled then one of the main assumptions of the regression model, namely constant error variance, is violated. The standard procedure is to then transform the response variate, in this case the algal count. The Box-Cox procedure was used to identify the best transformation. When the Box-Cox procedure was applied, then the problem of non-constant errors disappears, but strangely, the predictive ability (i.e. the variance accounted for by the model) drops drastically. A University of Natal statistician (Dicks 1997, pers. com.) is not sure why this happened, as a transformation suggested by the Box-Cox algorithm usually leads to an improvement (or at least no change) in model fit as well as variance stability. A check on the collinearity that may exist among some of the predictor (x) environmental variables indicates that they are not causing the problem. Nor are the instances of few very high algal counts. Attempts to formulate the model within a Generalized Linear Model framework, where one specifies a particular distribution for the errors (not normal), rather than transforming the y variable, did not solve the problem of the

skewed y variate. It appears that temporal autocorrelation and differences between lakes, where counts are high in only certain lakes at rare intervals, may be contributing to the problem.

To pursue the multiple regression modelling approach, would require each lake to be modelled separately, accounting for both the temporal cycle (through fitting appropriately shaped harmonic functions), and the effects of variables which are not related to the seasonal cycle. Daily, or weekly data may be useful and whilst this approach is exhaustive, it would have to be balanced against how such models could be used. More importantly, as others have suggested (e.g. van Tongeren *et al* 1992), it may be an impossible goal to try and predict the individual contribution to algal abundance of a number of factors that are intrinsically related (e.g. temperature and stability). Soudant *et al.* (1997) noted that with the inherent complexity of this type of algal data, and the high variabilities often associated with algae over time (generated by processes occurring at different spatial and temporal scales), static regression models are not suitable for this type of analysis. Multivariate analyses, where the joint contribution of variables can be examined, are perhaps more useful for understanding the pattern of algal community variation in the lakes under consideration (again observed by others in the literature, such as, van Tongeren *et al.,* 1992).

As noted by Borcard *et al.* (1992), in relation to multi-species and environmental data, as in this study, the amount of unexplained variation in the biotic data is always fairly high. Borcard and co-workers also noted that it may not always be possible to measure all the environmental variables (covering biological interactions and external environmental condition) that are relevant in an ecological study such as was undertaken here. Lynam (1997) elegantly summarises the problem of unmeasured and interacting environmental variables when he states: *"The more variables there are affecting the behaviour of a system, the more complex the web of interactions and dependencies. As analysts we can either turn a blind eye to these dependencies or we can attempt to identify their relative contributions to system performance. It is however, an extremely difficult task analytically, to separate complex interactions in ways that are reliable and have meaning in the world in which we make our observations."* These sentiments are further echoed by Wetzel, who notes in the preface to the 3rd edition of his seminal work on the subject of limnology (Wetzel, 2001), *"Because of the dynamic properties of metabolism, growth, and reproductive capacities of organisms as they respond to the bewildering array of*

dynamic environmental factors influencing growth and development, variability is enormous and often difficult to fathom. Modelling at the present level of understanding is unable to cope with so many nonlinear variables that are changing rapidly."

Notwithstanding the problems of applying the classical multiple regression modelling approach to the type of data available in this study, these models were investigated but found to be inappropriate to furthering the understanding of algae and their responses to measured environmental variables. The semi-quantitative empirical models developed in the ordination analyses had to suffice for producing predictive models relating algae to their environment.

CHAPTER 6: ECONOMIC EVALUATION OF THE EFFECTS OF LAKE WATER ON WATER TREATMENT COSTS

6.1 Introduction

As noted in the introductory chapter, one of the main consequences of eutrophication is the nutrient enrichment of surface waters resulting in abundant algal growth. Eutrophication may lead to increased water treatment costs. As per the aims of this study, this chapter has two objectives, namely:

- to identify the main factors or contaminants[6] affecting treatment costs at key waterworks (WW) in the Umgeni Water operational area, and
- to predict treatment costs from observed levels of contaminants.

Four waterworks (WW) systems were investigated in this regards, namely: Hazelmere WW, which draws water from Lake Hazelmere; DV Harris WW, which draws water from Lake Midmar; the Durban Heights WW, which draws water from Lakes Nagle and Inanda; and the Wiggins WW which draws water primarily from Inanda. The latter three constituted the most important WW operating within the Umgeni Water operational area during the study period (1990 - 1996), both in terms of volumes of water treated/processed and costs incurred in producing this potable water. Figure 6.1 (for the 1997 period[7]) illustrates the relative contribution these WW made to total volumes of potable water produced by Umgeni Water and their respective chemical treatment costs. Since completing the analytical phase of this study, Midmar WW came 'on-line' (1998) and is now also a major producer of potable water. Hazelmere was included in the study due to the uniquely turbid nature of the raw water entering this WW.

[6] Contaminants refers to those water quality variables or factors which impact on the cost of treating water (these may be physico-chemical and/or biological e.g. algae).

[7] 1997 was chosen to reflect the relative contributions of WW, as it was the period closest to the final phase of data collection and analysis for this study.

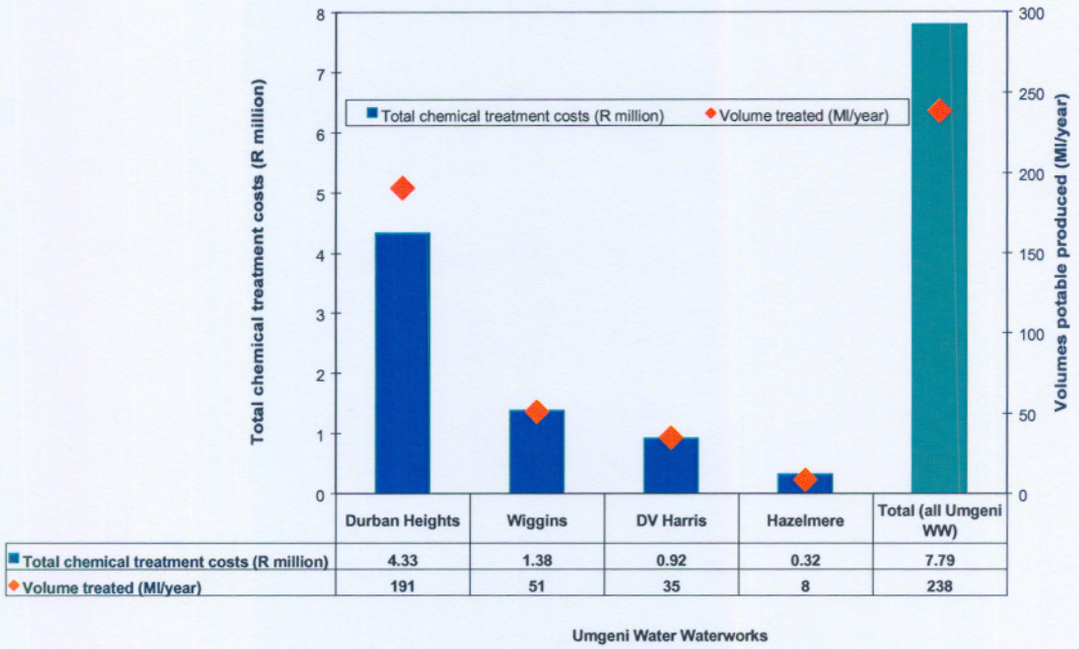

Figure 6.1. Total chemical treatment costs and volumes of potable water produced by key waterworks in the Umgeni Water operational area during 1997.

Treatment costs refer to financial costs incurred in ensuring that the water meets the appropriate potable water standards. The statutory need for monitoring to these standards is contained within section 9(1) of the Water Services Act, 1997 (Act No. 108 of 1997), the Regulations Relating To Compulsory National Standards And Measures To Conserve Water (SEC9DREG -7 March 2001). The regulations state that a *"Water Services Authority must include a suitable programme for sampling the quality of potable water provided to consumers in its water services development plan"* (section 5.1) and that a *"Water Services Institution must compare the results obtained from the testing of samples with SABS 241: Specifications for Drinking Water, or the South African Water Quality Guidelines published by the Department of Water Affairs and Forestry"* (section 5.3). The South African Bureau of Standards, SABS 241:2001 – Drinking Water (SABS, 2001), is the principal guideline document against which Umgeni Water assesses its final (potable) drinking water quality. An example of typical raw and final (potable) water for the Hazelmere WW (extracted from the Umgeni Water annual report 2001/2002), with these attendant standards, is presented in Table 6.1.

Table 6.1 Typical raw and final (potable) water statistics and the relevant "standards" to which the water is treated for the Hazelmere Waterworks - treating 17 Mℓ/d (extract from the Umgeni Water annual report 2001/2002)

		Raw water			Final (potable) water			
DETERMINAND	Units	Minimum	Average	Maximum	Minimum	Average	Maximum	Standard
Alkalinity	mg/ℓ	21	34	41	34	44	50	-
Aluminium	µg/ℓ	29	420	1 097	< 10	33*	69	300
Ammonia	mg/ℓ	0.01	0.07	0.22	< 0.01	0.23*	0.65	1
Antimony	µg/ℓ	< 2	< 2	< 2	< 2	< 2	< 2	10
Arsenic	µg/ℓ	< 2	< 2	< 2	< 2	< 2	< 2	50
Cadmium	µg/ℓ	< 1	< 1	< 1	< 1	< 1	< 1	5
Calcium	mg/ℓ	4.2	5.3	6.0	9.0	10.6	12.5	150
Chloride	mg/ℓ	16.8	20.8	24.2	19.6	23.4	26.8	200
Chlorine (free)	mg/ℓ	-	-	-	< 0.05	0.05*	0.20	-
Chlorine (total)	mg/ℓ	-	-	-	1.10	1.55	2.00	-
Chromium	µg/ℓ	< 3	1.8*	3.1	< 3	< 3	< 3	100
Coliforms	per 100 mℓ	0	84	800	0	0	18	0
Colony Count @ 37°C	per mℓ	-	-	-	0	0	7	100
Colour	°H	1.2	83.3	224	< 1	2.0*	4.4	20
Conductivity	mS/m	11.0	14.8	17.7	14.2	17.3	20.1	150
Copper	mg/ℓ	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	1
Cyanide (free)	µg/ℓ	-	-	-	< 10	< 10	< 10	50
Cyanide (recoverable)	µg/ℓ	< 10	< 10	< 10	< 10	< 10	< 10	200
E.coli	per 100 mℓ	0	44	450	0	0	0	0
Fluoride	µg/ℓ	< 100	128*	189	< 100	129*	162	1 000
Iron	mg/ℓ	0.27	1.06	3.31	< 0.02	0.01*	0.12	0.2
Lead	µg/ℓ	< 4	2.5*	4.6	< 4	< 4	< 4	50
Magnesium	mg/ℓ	3.5	4.6	5.2	3.2	4.5	5.2	70
Manganese	mg/ℓ	< 0.01	0.04*	0.42	< 0.01	0.01*	0.01	0.1
Mercury	µg/ℓ	< 0.5	< 0.5	< 0.5	< 0.5	< 0.5	< 0.5	2
Nickel	µg/ℓ	< 3	< 3	< 3	< 3	< 3	< 3	150
Nitrate + Nitrite	mg/ℓ	0.38	0.58	0.69	0.40	0.56	0.73	10
pH	pH units	6.5	-	8.0	8.5	-	9.5	5 - 9.5
Phenols	µg/ℓ	< 5	< 5	< 5	< 5	< 5	< 5	10
Potassium	mg/ℓ	1.3	1.6	1.8	1.3	1.6	1.8	50
Selenium	µg/ℓ	< 1	< 1	< 1	< 1	< 1	< 1	20
Sodium	mg/ℓ	13.8	17.8	21.0	13.8	17.8	21.0	200
Sulphate	mg/ℓ	4.1	4.7	5.3	3.8	4.6	5.3	400
Suspended Solids	mg/ℓ	< 4	61*	349	-	-	-	-
Total Dissolved Solids	mg/ℓ	108	147	216	96	118	154	1 000
Total Hardness	mg/ℓ	25	32	37	37	45	53	20 – 300#
Total Organic Carbon	mg/ℓ	1.5	4.9	12.4	< 0.7	2.0*	3.6	10$
Total Trihalomethanes	µg/ℓ	-	-	-	3	12	34	200
Turbidity	NTU	3.8	145	758	0.11	0.32	0.63	1
Vanadium	µg/ℓ	-	-	-	< 10	< 10	< 10	200
Zinc	mg/ℓ	< 0.03	0.08*	0.20	< 0.03	< 0.03	< 0.03	5

Standard: Standards have been adopted from the SABS 241: 2001 Drinking Water Specification. Class I limits are shown which are suitable for lifetime consumption.
* Statistics for constituents that had some results below the detection limit were calculated using half the detection limit
$ SABS 241: 2001 specifies Dissolved Organic Carbon
# Umgeni Water standard

Figure 6.1 highlights that during the 1997 report period alone almost R8 million (R7.79 million) was spent on chemicals for the purification of drinking water in the

WW focussed on for this study (Umgeni Water, 2001). Obviously savings may be possible with a better understanding of how raw water quality entering respective WW was impacting on the treatment costs.

Statistical methods (involving linear regression and principal components analysis PCA) were used to analyse the cost of treating water at the waterworks. The water quality of raw (untreated) water from the different storage impoundments (lakes) were all generally quite different from one another. For example Lake Hazelmere is highly turbid, while water in the more coastal lakes (Nagle and Inanda) is eutrophic relative to the 'clean water' in Lake Midmar (see Chapter 2 of this thesis).

Respective WW were designed, built and upgraded over numerous years, partly in relation to the types of raw water quality that they were receiving, but also as water treatment technology has advanced. Consequently, treatment processes and costs differ between the respective WW. Treatment costs were therefore analysed separately at each WW. A trend variable was considered in each analysis to determine if there was a gradual increase or decrease in treatment costs at respective works. An increase in treatment costs over time would indicate deterioration in raw water quality entering WW (and *vice versa*).

The December 1996 water treatment chemical prices used in analyses are presented in Table 6.2 (based on information supplied by the Umgeni Water, Finance and Administration Division). Prices are expressed as South African Rands/kg.

Table 6.2 Water treatment chemical prices for respective waterworks studied (expressed in December 1996 Rands/kg prices)

	Chemical (R/kg)								
Water Works	**Chlorine**	**Granular Chlorine (HTH)**	**Ammonia**	**Polymeric coagulant**	**Lime**	**Bentonite**	**Activated Carbon**	**Ferric Chloride**	**Sodium Chlorite**
Hazelmere	2.82	8.57		3.06	0.53				3.62
Durban Heights	2.82	8.57		3.34	0.78	0.68	3.25		
DV Harris	2.82	8.60	5.66	4.07	0.53	0.69		0.59	
Wiggins	2.82	8.57		3.34	0.53	0.65	3.25		2.20

Treated water in the inland (Pietermaritzburg – DV Harris plant) regions is dosed with lime to achieve a precipitation potential of 4mg/ℓ of calcium carbonate ($CaCO_3$). This limits the corrosive nature of water that is too 'soft'. In the case of the coastal Durban Heights/Wiggins WW systems lime is also dosed. However, the amount added is generally lower due to a request from the Durban Metro (the major bulk purchaser of water in the region) who prefer to have a lower pH water to improve the disinfection

efficiency of their oxidants (generally chlorine). To perform valid cost comparisons between the respective WW, the theoretical additional lime dose (based on the raw water quality entering the coastal plants) was calculated to achieve a similar precipitation potential (4 mg/ℓ $CaCO_3$) to that achieved at the DV Harris plant (Thompson pers. comm., 2003). The additional calculated dose was then added to that 'actually' used in the Durban Heights and Wiggins WW prior to analyses.

The previous chapter of this thesis presented a model that relates algal populations to various environmental factors. Potentially the results of the economic and algae/environment models could be combined in a future study to explore links between land use activities, water quality and treatment costs. Reliable information about the origin of high treatment costs is required to inform and guide both policy and planning decisions, as well as to assist with the refinement of potable water treatment processes.

6.2 Description of selected water treatment systems

For the Hazelmere system, data on water quality parameters recorded at Lake Hazelmere (UW lake main basin sample point 101.1) and Hazelmere WW (UW raw water sample point 314) were used to analyse this system. Missing data on chemical usage between the months of January to September and December 1993, January, May, June, and September 1994, reduced the number of valid data cases available for analysis from 84 to 71.

For the Durban Heights system, water quality data recorded at Lake Nagle (UW sample point number 431) and Durban Heights WW (UW raw water sample point number 316), were recorded over the period March 1990 to February 1997, totalling 84 valid cases.

Study of the DV Harris system was based on water quality data recorded at the Midmar site (UW sample point 36.1) and the DV Harris site (UW raw water sample point 324). Data used were from January 1990 to December 1996. Unfortunately, some data on chemical dosage for the months of January 1990, July 1990 - April 1991, March 1993, July 1993, February 1994 - May 1994, September 1994, and October 1994 were missing, reducing the number of valid observations from 84 to 64.

Lastly, the Wiggins system was analysed; based on 82 valid water quality data observations recorded at Lake Inanda (UW sample point 55.1) and the Wiggins WW (UW raw water sample point 320).

A broad description of each of the catchments and the water quality of respective lakes is provided in Chapter 2. More specific information, as it pertains to the economics of water treatment, is presented here for each of the raw water supply systems and their respective WW.

6.2.1 The Hazelmere system

As the analysis of algae and environmental variables showed (Chapters 4 & 5), Hazelmere is a turbid lake, with the mean and maximum turbidity, recorded at the WW over the period March 1990 to February 1997, 87 NTU and 540 NTU respectively (Table 6.3). The turbid water is persistent for most of the summer stratification period after heavy rains result in inflow maxima (Archibald *et al.*, 1980). The rapid flushing rate and the high turbidity of the water may offset to some extent the summer nutrient input and consequent potential for algal growth (Archibald *op. cit.*). Information describing the quality of the water in the Hazelmere system is summarised in Table 6.3.

Table 6.3 Descriptive statistics showing characteristics of raw water treated at Hazelmere WW (March 1990 - February 1997)

Water quality indicator	Units	Minimum	Mean	Maximum
Treatment cost	R/Mℓ	6.00	40.59	111.00
Turbidity	NTU	11.80	87.47	540.90
Total Organic Carbon	mg/l	0.00	3.86	10.80
Suspended Solids	mg/l	2.50	38.99	264.00
Total Aluminium	ug/l	102.00	553.8	2027.00
Alkalinity	mg/l	17.70	37.61	54.20
Nitrate	mgN/l	0.00	0.54	1.32
Sulphate	mg/l	0.00	5.77	10.40
Potassium	mg/l	1.20	2.20	4.00
Inflow	Mℓ/dav	12.00	198.50	1769.00
Chrome	ug/l	0.00	9.83	61.20
Total Dissolved Solids	mg/l	0.00	132.3	286.00
Iron	mg/l	0.48	2.07	6.39
Manganese	mg/l	0.01	0.06	0.35
pH		6.66	7.32	7.77
Chlorella	cells/ml	0	77	352
Cryptomonas	cells/ml	0	26	149
Microcystis	cells/ml	0	59	829
Nitzschia	cells/ml	0	3	23
Scenedesmus	cells/ml	0	1	21

The average cost of treating water at Hazelmere WW over the study period was R40/Mℓ. This cost ranged between R6/Mℓ and R111/Mℓ in response to changes in water quality (Table 6.3). On average, the cost of polymeric coagulant constitutes the largest proportion (66%) of treatment costs (Figures 6.2 and 6.3).

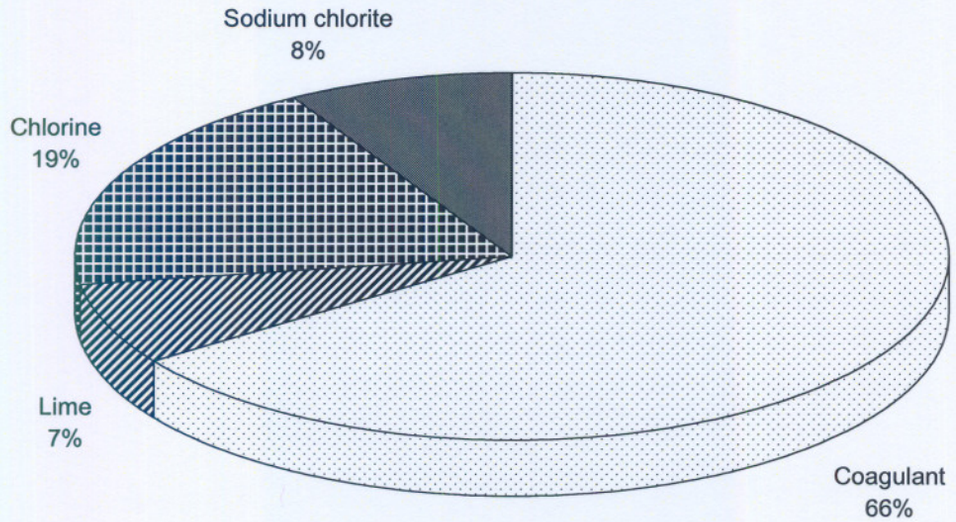

Figure 6.2 Average chemical cost composition at Hazelmere WW (March 1990 - February 1997).

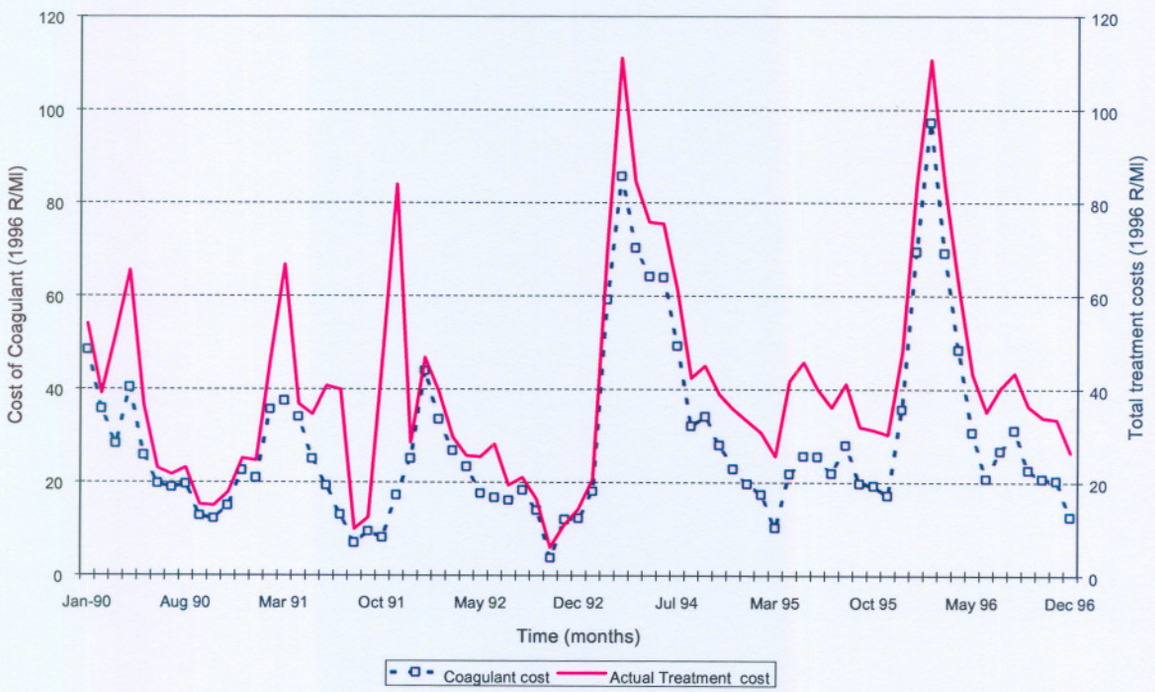

Figure 6.3 Total treatment costs and cost of polymeric coagulant at Hazelmere WW.

6.2.2 The Durban Heights system

The Durban Heights WW plant draws water mainly from Lake Nagle and occasionally from Lake Inanda. Water quality conditions in Lake Nagle are conducive to the formation of occasional blooms of the algal genera *Anabaena* and *Microcystis*. These algae have the potential to form taste and odour compounds requiring advanced (and costly) water treatment processes. A maximum of 16869 cells/mℓ and 6117 cells/mℓ of *Anabaena* and *Microcystis* counts respectively, were recorded in Lake Nagle (Table 6.4).

Table 6.4 Descriptive statistics showing characteristics of raw water treated at the Durban Heights WW (January 1990 - February 1997)

Water quality indicator	Units	Minimum	Mean	Maximum
Treatment cost	R/Mℓ	14.17	24.90	65.24
Turbidity	NTU	6.76	18.28	89.98
Total Organic Carbon	mg/l	1.65	2.74	5.84
Suspended Solids	mg/l	3.28	13.04	44.25
Total Aluminium	ug/l	47.20	158.20	915.30
Nitrate	mgN/l	0.09	0.30	1.38
Secchi	m	0.25	1.18	2.36
Conductivity	mS/m	6.43	9.23	14.58
Sulphate	mg/l	3.09	4.21	9.37
Chloride	mg/l	6.30	8.02	16.74
Sodium	mg/l	5.48	6.85	12.60
Total Hardness	mg/l $CaCO_3$	24.32	28.31	36.93
Calcium	mg/l	4.28	5.36	6.57
Magnesium	mg/l	3.06	3.57	5.09
Potassium	mg/l	0.97	1.65	5.07
Silica	mg/l	1.43	4.05	7.38
Dissolved oxygen	mg/l O_2	6.23	8.23	10.28
Sol. Reactive Phosphorus	ugP/l	1.50	4.08	19.67
Iron	mg/l	0.21	0.58	1.26
Manganese	mg/l	0.01	0.04	0.13
pH		7.19	7.66	7.86
Coliforms	colony counts/100	3	71	566
E.coli	colony counts/100	1	32	239
Anabaena	cells/ml	0	873	16869
Microcystis	cells/ml	0	848	6117
Chlorella	cells/ml	31	407	1023

The cost of treating water at Durban Heights WW was on average is R25/Mℓ, however, this can rise as high as R65/Mℓ (Table 6.4). On average the cost of polymeric coagulant constitutes 45% of treatment costs, followed by the cost of chlorine at 38% (Figure 6.4). This illustrates that treatment costs at Durban Heights are mainly driven by changes in coagulant requirement and water disinfection. The average treatment cost at the Durban Heights WW increases to R28/Mℓ if the final water is adjusted to achieve a final precipitation potential of 4mg/ℓ $CaCO_3$ (more

lime). This would change the average cost structure at the Durban Heights WW as illustrated in Figure 6.5. This cost structure again changes when powdered activated carbon (PAC) is dosed during periods of taste and odour problems, resulting from blooms of *Anabaena* or *Microcystis* in Lake Nagle (Figure 6.6). For example, in March 1994, the average cost of treating water was R65/Mℓ during a particularly severe period of *Anabaena* generated taste and odour problems.

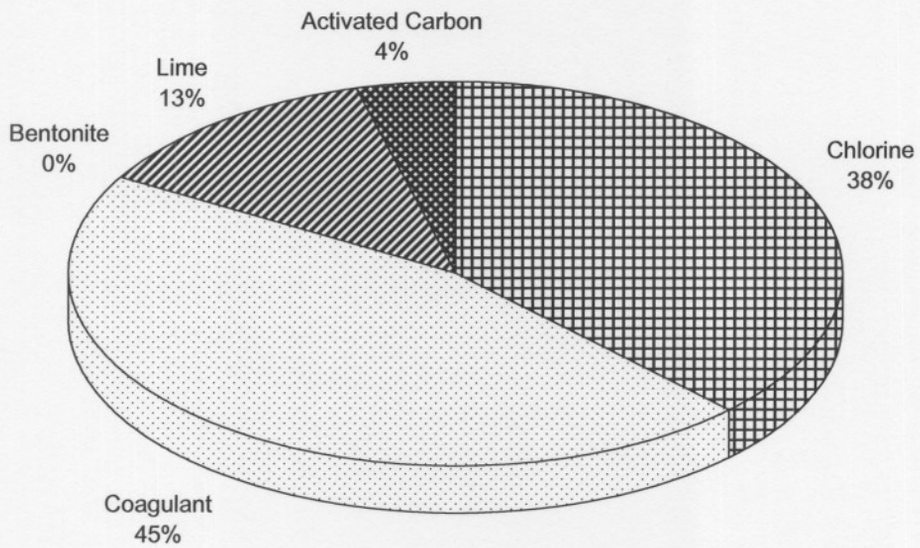

Figure 6.4 Average chemical cost composition at Durban Heights WW (January 1990 – February 1997).

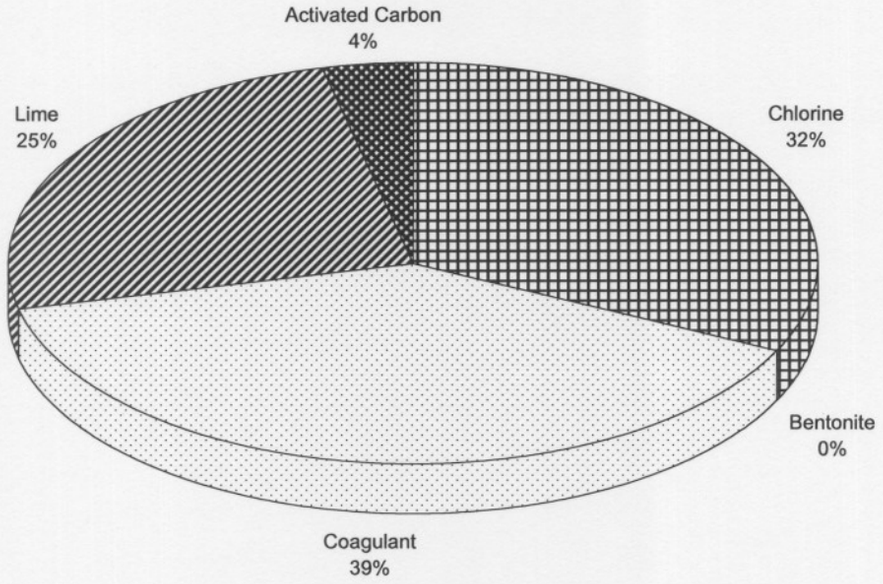

Figure 6.5 Average water treatment cost composition at Durban Heights WW (January 1990 – February 1997) after correction for lime dosages to attain a precipitation potential of 4mg/ℓ $CaCO_3$.

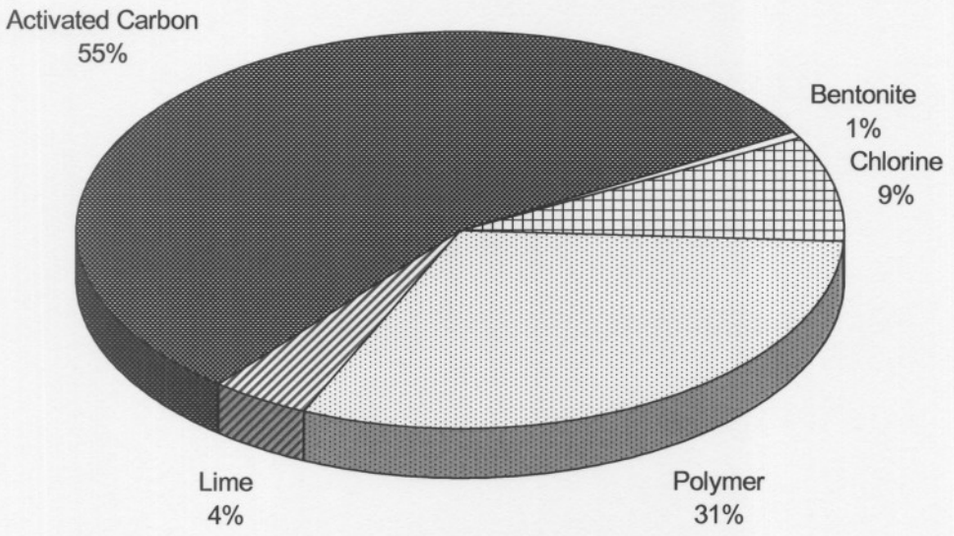

Figure 6.6 Average water treatment cost composition at Durban Heights WW during a particularly severe period of taste and odour formation (March 1994)

6.2.3 The DV Harris system

The DV Harris WW plant draws water from Lake Midmar, an inland impoundment characterised by generally 'clean' water (compared to that treated at Hazelmere and

Durban Heights WW). Maximum turbidity of water treated at DV Harris recorded between January 1990 to December 1996 was 52 NTU. The descriptive statistics showing the condition of raw water treated at DV Harris are given in Table 6.5.

Table 6.5 Descriptive statistics showing the characteristics of raw water treated at the DV Harris WW

Water quality indicator	Units	Minimum	Mean	Maximum
Treatment cost	R/Mℓ	22.64	28.30	49.83
Turbidity	NTU	4.63	23.65	52.46
Total Organic Carbon	mg/l	1.82	2.86	7.85
Suspended Solids	mg/l	3.82	14.96	36.70
Alkalinity	mg/l	19.73	24.83	29.20
Nitrate	mgN/l	0.07	0.31	1.09
Potassium	mg/l	0.69	1.38	2.22
Temperature	^{o}C	11.03	17.20	21.93
Conductivity	mS/m	5.97	6.92	8.09
Total Hardness	mg/l $CaCO_3$	18.02	21.96	25.73
Calcium	mg/l	3.11	4.40	5.31
Magnesium	mg/l	2.24	2.63	3.16
Sodium	mg/l	3.24	4.48	5.55
Silicon	mg/l	2.30	4.53	5.92
Dissolved oxygen	mg/l O_2	6.63	8.39	10.88
Manganese	mg/l	0.01	0.01	0.03
Coliforms	colony counts/100ml	0	16	136
E.coli	colony counts/100ml	0	5	53
Chlorella	cells/ml	12	221	644

The average cost of treating water at DV Harris plant over the period January 1990 to December 1996 was R28/Mℓ, with the cost ranging between R22/Mℓ and R49/Mℓ with changes in water quality (Table 6.5). The cost of disinfecting the water constituted approximately 50% of total treatment costs (Figure 6.7). The cost of water treatment at DV Harris would therefore appear to be strongly influenced by changes in the requirements of chlorine and ammonia to disinfect the water. A more credible explanation, however, is that the polymeric coagulant costs in the relatively clean Lake Midmar water are relatively low, compared to the contribution of disinfection costs. This is artificially inflating the influence of disinfection on total costs by comparison with more turbid (coagulant) driven systems.

The removal of cost of ammonia, which is only required for distribution purposes of treated water at DV Harris, reduces the average treatment cost to R25/Mℓ. This also changes the cost structure of the DV Harris system as shown in Figure 6.8, compared to that shown in Figure 6.7.

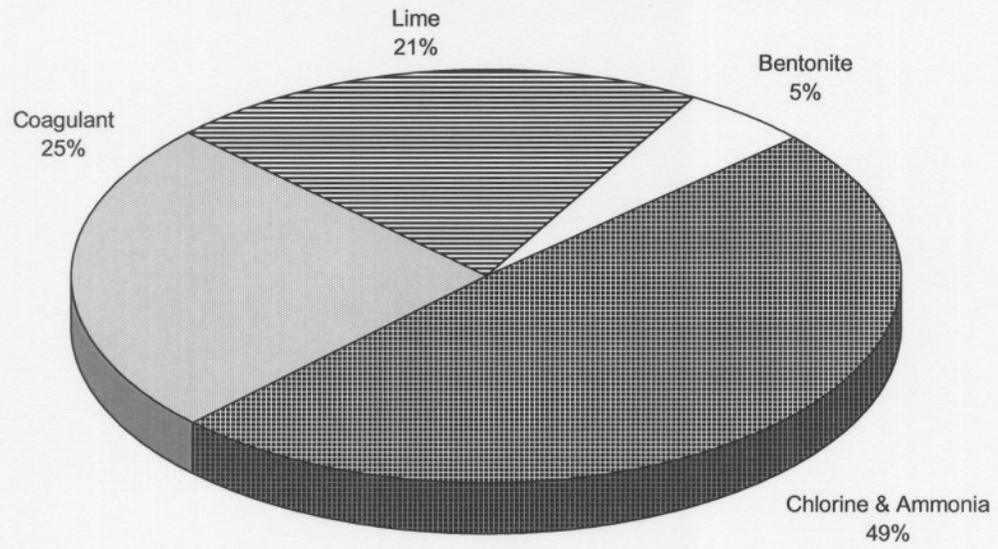

Figure 6.7 Average water treatment cost composition at DV Harris WW.

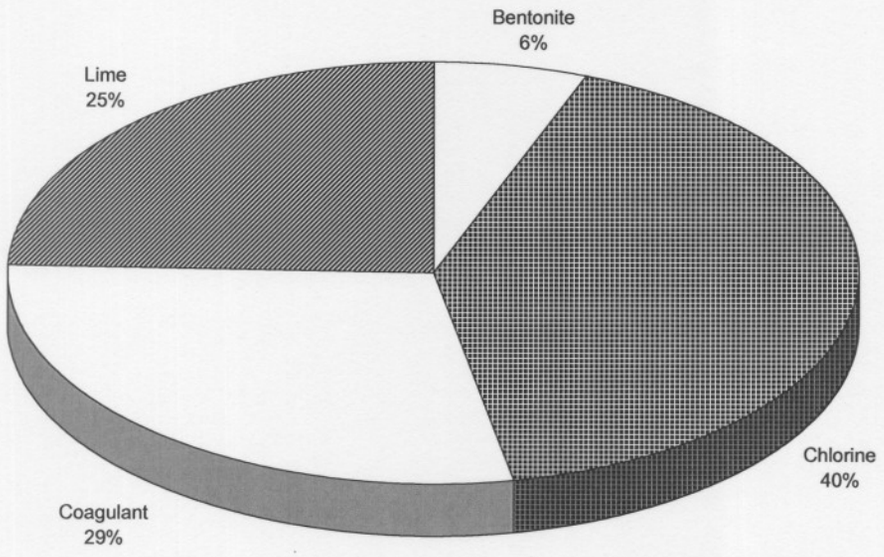

Figure 6.8. Average treatment cost composition at DV Harris WW (with the exclusion of ammonia).

6.2.4 The Wiggins system

The Wiggins WW plant draws water mainly from Lake Inanda and occasionally from Lake Nagle. Water quality conditions in the lake are such that occasional blooms of

the algae *Anabaena* and *Microcystis* occur. A maximum of 6310 cells/ml *Anabaena* was recorded in Lake Inanda during the study period (Table 6.6).

Table 6.6 Descriptive statistics showing the characteristics of raw water treated at Wiggins WW (January 1990 - March 1997)

Water quality indicator	UNITS	Minimum	Mean	Maximum
Treatment cost	R/Ml	3.06	20.84	75.32
Turbidity	NTU	2.36	19.19	141.13
Total aluminium	ug/l	3.40	85.46	519.50
Iron	mg/l	0.10	0.56	1.89
Suspended Solids	mg/l	2.00	15.14	88.50
Nitrate	mgN/l	0.03	0.44	2.35
Total Organic Carbon	mg/l	1.62	3.09	5.45
Total Dissolved Solids	mg/l	39.20	117.13	224.00
pH		7.32	7.71	8.11
Secchi	m	0.40	1.65	2.90
Silicon	mg/l	0.50	2.98	7.60
Potassium	mg/l	1.00	2.83	15.00
Coliforms	colony counts/100ml	2.00	157.23	1472.00
Sulphate	mg/l	3.20	11.13	22.00
Temperature	°C	15.22	20.80	25.97
Trachelomonas	cells/ml	0	0	4.00
Chlorella	cells/ml	6	247	726
Cryptomonas	cells/ml	0	35	130
Crucigena	cells/ml	0	74	819
Anabaena	cells/ml	0	151	6310

The average cost of treating water at the Wiggins WW during the period January 1990 to April 1997 was R20/Mℓ. The cost however, ranged between R3/Mℓ and R75/Mℓ, with changes in water quality. The cost of flocculation constituted approximately 44% of total water treatment costs, followed by disinfection costs, 38% (Figure 6.9). As with the Durban Heights system the average treatment cost increases (to R22/Mℓ) if the lime dosage is adjusted to attain a precipitation potential of 4mg/ℓ $CaCO_3$. This would change the average cost structure at Wiggins WW to that shown in Figure 6.10.

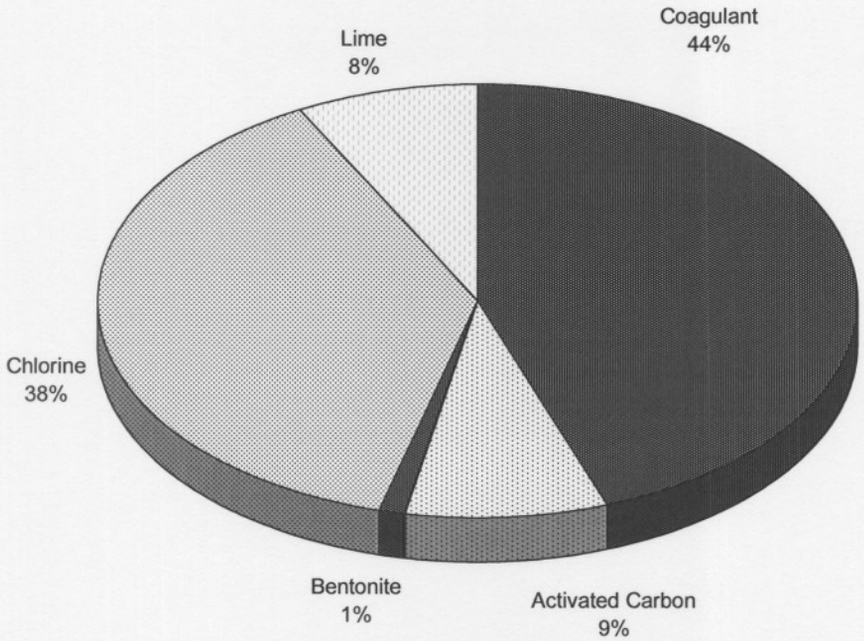

Figure 6.9 Average water treatment cost composition at Wiggins WW (Jan 1990 – Feb 1997).

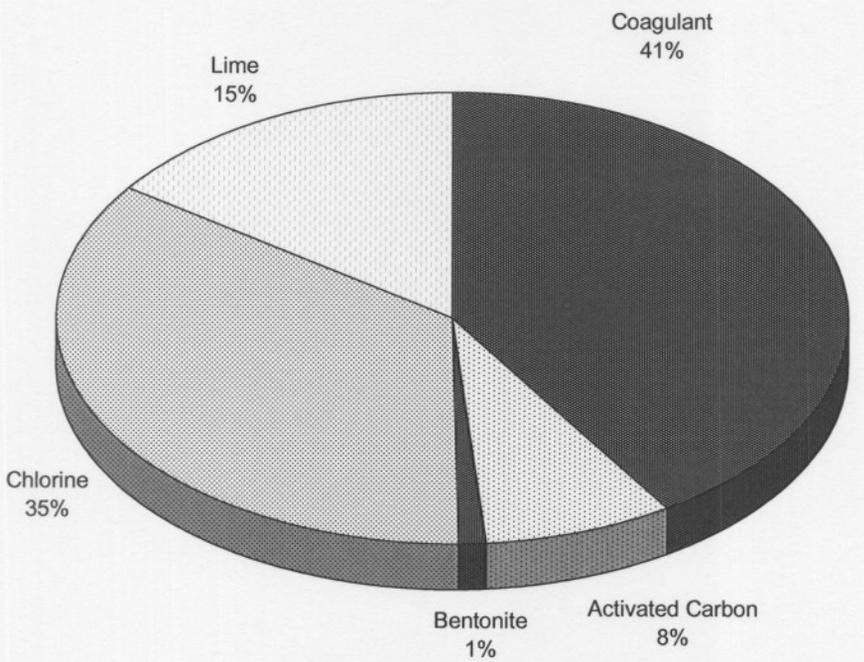

Figure 6.10 Average chemical cost composition at Wiggins WW (Jan 1990 - Feb 97) after adjusting for extra lime required to achieve the 4mg/ℓ $CaCO_3$ precipitation potential.

6.3 Results and discussion

6.3.1 Selection of water quality contaminants for the modelling of costs at selected waterworks

In order to isolate conditions most closely associated with water treatment costs at the selected WW (Hazelmere, Durban Heights, DV Harris and Wiggins), zero-order correlation analysis was performed relating treatment costs with some of the contaminants anticipated to influence water treatment. Contaminants with statistically significant correlation (r^2) coefficients were selected for further analysis.

On the basis of the correlation analysis, contaminants that were anticipated to be affecting the cost of treating water at the selected WW are given in Table 6.7.

Table 6.7 Correlation coefficients (r^2) for important contaminants (environmental variables & algae) with treatment costs at respective WW

Contaminant	Units	Hazelmere	Durban Hts	DV Harris	Wiggins
Alkalinity	mg/l	-0.5686**		-0.615**	
Boron	ug/l	0.5665**[8]			
Chrome	ug/l	0.3060*			
Coliforms	counts/100ml		0.4068**	0.604**	0.279*
Conductivity	mS/m		0.5655**	-0.323**	
Dissolved oxygen	mg/l O_2		-0.3394**	-0.310*	
E. coli	counts/100ml		0.3757**	0.562**	
Inflow	Ml/day	0.4170**			
Iron	mg/l	0.2735*	0.5442**		0.613**
Manganese	mg/l	0.2360*	0.3449**	-0.359**	
Nitrate	mgN/l	0.5577**	0.5177**	0.436**	0.534**
pH		-0.2591*	-0.3873**		-0.402**
Potassium	mg/l	0.4937**	0.4937**	0.456**	0.321**
Secchi	m		-0.5244**		-0.377**
Silicon	mg/l		0.5044**	0.329**	0.335**
Soluble Reactive Phosphorus	ugP/l		0.4271**		
Sulphate	mg/l	0.5151**			0.245*
Suspended Solids	mg/l	0.6741**	0.6135**	-0.205	0.577**
Temperature	°C		0.3662**	0.267*	0.245*
Total Aluminium	ug/l	0.6552**	0.4563**		0.702**
Total Dissolved Solids	mg/l	0.2899*			0.422**
Total Hardness	mg/l $CaCO_3$		0.5200**	-0.528**	
Total Kjel Nitrogen	mgN/l		0.2484*		
Total Organic Carbon	mg/l	0.6857**	0.5540**	0.225	0.481**
Total Phosphorus	ugP/l		0.4126**		
Trend Variable	Month			0.437**	0.318**
Turbidity	NTU	0.7619**	0.7345**	-0.189	0.871**
ALGAL GENERA					
Anabaena	cells/ml		0.3234**		0.089
Chlorella	cells/ml	-0.4749**	-0.2589*	-0.479*	-0.365**
Crucigenia	cells/ml				-0.218*
Cryptomonas	cells/ml	-0.3216**			-0.254*
Microcystis	cells/ml	-0.2705*	0.4092**		
Nitzschia	cells/ml	-0.2637*			
Scenedesmus	Cells/ml	-0.2495*			
Trachelomonas	Cells/ml				0.574**

Note: * Significant at 5%, ** Significant at 1%

6.3.2 Contaminants affecting treatment costs at respective waterworks

Results of the linear regression models fitted for the respective WW are presented in Tables 6.8 to 6.11. Due to multicollinearity, whereby many environmental variables were strongly correlated with each other, there were problems with developing robust

[8] Although cost was statistically significantly correlated with Boron ($r^2 = 0.567$), this contaminant was omitted owing to a large number of missing values. Boron had only 21 valid cases as opposed to 71 cases recorded for other variables

statistical models relating water treatment costs to water quality variables (original model estimates). After removing multicollinearity (final model estimates), the final models were considered a more robust predictor of treatment costs. The lowest R^2 in these final models was a reasonable 64%. The figures indicating the match between actual costs and those predicted by the various models, illustrate the robustness of these models (e.g. Fig. 6.11). Where appropriate, selected additional explanations and graphical presentations are made in the text of some of the important relationships between water quality variables and treatment costs.

In only the DV Harris and Wiggins WW systems was the trend variable[9] found to be significantly affecting treatment costs. This implies an increase in treatment costs over time and a possibly a deterioration in raw water quality entering these respective WW.

6.3.2.1 The Hazelmere system

Results of the regression model calculated for the Hazelmere WW are presented in Table 6.8. The final model fitted for this system captures 79% of the information on factors affecting water treatment costs with a good match shown between actual and predicted treatment costs (Figs. 6.11 and 6.12).

Table 6.8 Estimated regression coefficients of factors affecting water treatment costs at Hazelmere WW (before and after removing the multicollinearity problem)

Explanatory Variable	Original model estimates		Final model estimates	
	Coefficients	t-values	Coefficients	t-values
Constant	50.100	1.16	62.330	
pH	-8.150	-1.36	-9.296	-1.84
Turbidity	0.180	4.19**	0.026	5.78**
Total Aluminium	-0.001	-0.18	0.004	3.24*
Alkalinity	-0.147	-0.63	-0.310	-1.53
Potassium	7.210	2.48*	8.026	2.80*
Manganese	145.000	5.95**	135.031	5.67**
Nitrate	23.970	2.93*	22.677	1.01
Sulphate	1.880	2.24*	1.653	2.26*
Suspended Solids	-0.105	-1.55	0.049	4.10**
Total Organic Carbon	-0.420	-0.30	1.633	3.82*
Chlorella	-0.032	-1.71	-0.031	-1.71
R^2		82%		79%
Adj. R^2		80%		76%
Number of cases	71			

* Sig. at 5%, and ** Sig. at 1%.

[9] Trend variable – to determine progressive increasing or decreasing treatment costs at respective works. An increase in treatment costs over time would indicate deterioration in raw water quality entering WW (and *vice versa*).

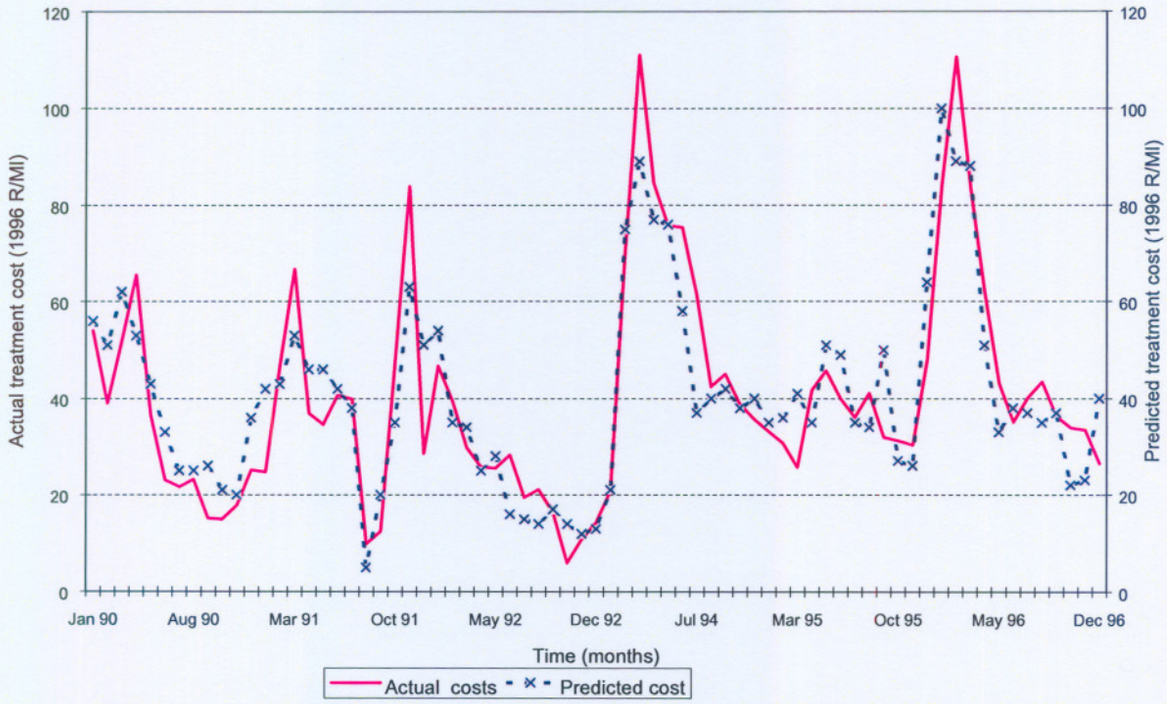

Figure 6.11 Actual versus predicted treatment costs (over time) estimated for the final Hazelmere WW model.

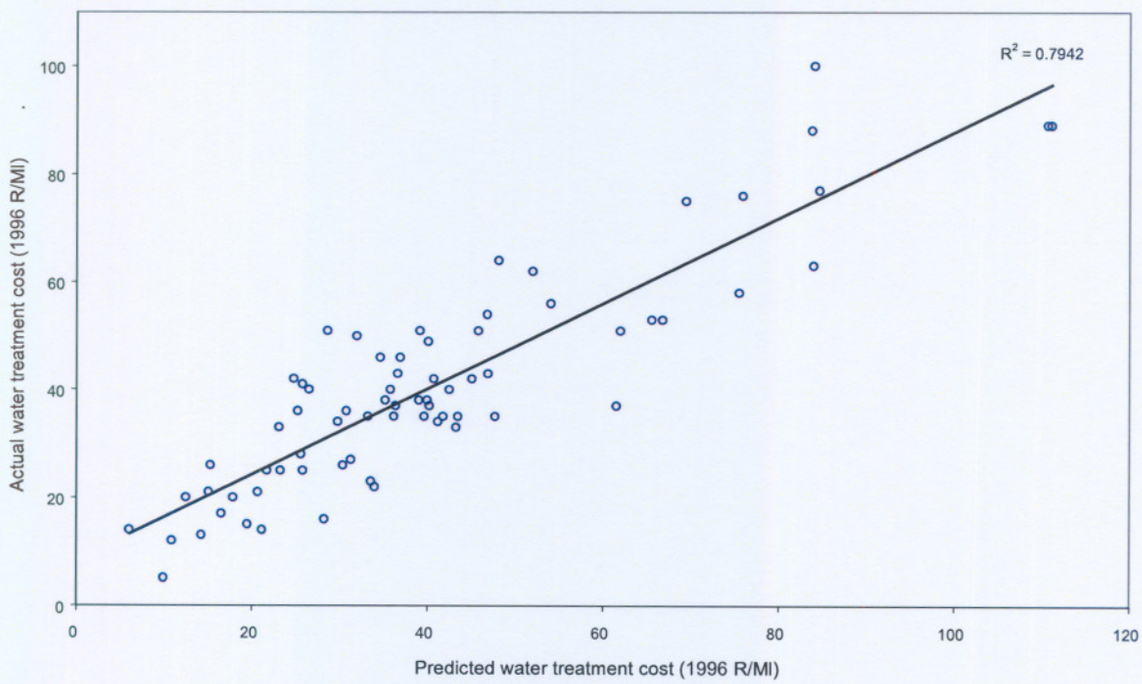

Figure 6.12 Scatterplot of actual versus predicted treatment costs for the Hazelmere WW model.

The algebraic form of the model used to predict water treatment cost at Hazelmere WW is as follows:

$$Cost = 62.3 + 0.026Turb + 135.0Mn + 0.049SS + 1.633TOC + 0.004Al + 1.653SO_4 + 8.026K - 9.296pH - 1.71\textit{Chlorella} - 0.310\ Alk + 22.7NO_3$$

Where:

Cost	=	Predicted water treatment cost
Turb	=	Turbidity
Mn	=	Manganese
SS	=	Suspended Solids
TOC	=	Total Organic Carbon
Al	=	Total Aluminium
SO_4	=	Sulphates
K	=	Potassium
pH	=	pH
Alk	=	Alkalinity
NO_3	=	Nitrates

The results (Table 6.8) suggest that real treatment costs at the Hazelmere WW are closely related to water turbidity in Lake Hazelmere. Increases in cost arise as a result of increased use of polymeric coagulants to treat the elevated turbidity in the raw water (Figure 6.13). It should be noted that highly turbid water in Lake Hazelmere has correspondingly high levels of suspended solids, total organic carbon and total aluminium.

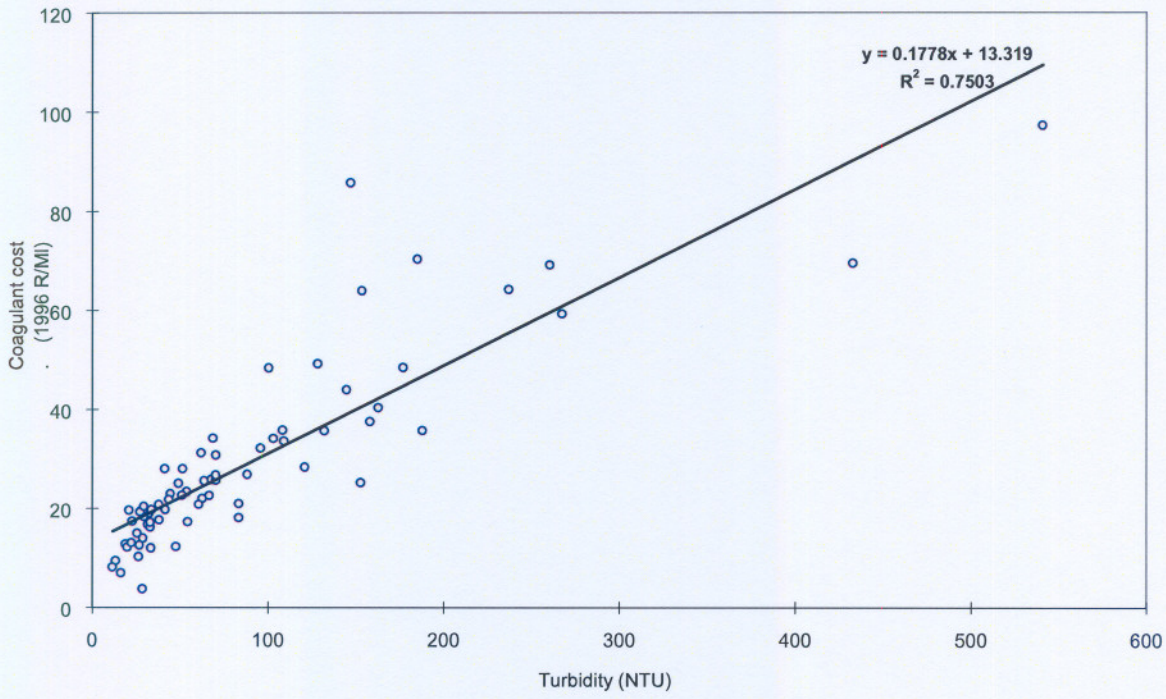

Figure 6.13 Cost of polymeric coagulant versus raw water turbidity at Hazelmere WW.

The positive and statistically significant relationship between treatment costs and manganese shows that manganese is a potentially serious contaminant in Lake Hazelmere water. Manganese in the reduced form may cause aesthetic problems, for example staining of laundry and household appliances. It may also exacerbate bio-film production through the formation of deposits in distribution networks. Treatment for problems associated with manganese normally requires the use of a strong oxidant e.g. chlorine dioxide (formed through the combination of sodium chlorite and chlorine, Freese, pers. comm., 2003).

The results presented in Table 6.8 (negative coefficient and *t*-value) suggest that treatment costs at Hazelmere WW diminish with an increase in abundance of the common green alga *Chlorella.* Lake Hazelmere, unlike other lakes e.g. Midmar and Nagle, has generally highly turbid water that probably suppresses the growth of planktonic algae. *Chlorella* abundance, although relatively low, is most apparent during periods of low turbidity when treatment costs are lowest. *Chlorella* is therefore probably a surrogate for water clarity in Lake Hazelmere and helps to explain the negative relationship between treatment costs and increases in *Chlorella* counts in the lake.

There is an apparent inverse relationship between the raw water alkalinity and treatment costs at the Hazelmere WW. This relationship arises from lower lime dosages required (to maintain water with low corrosivity) as raw water alkalinity levels rise, and *vice versa.* The relationship between pH and treatment costs follows a very similar trend to that of alkalinity for the same reasoning.

6.3.2.2 The Durban Heights system

Results of the regression model estimated for the Durban Heights WW are presented in Table 6.9. The final model fitted for the Durban Heights WW captures a reasonable amount of information (R^2=64%) on factors affecting water treatment costs with a good match shown between actual and predicted treatment costs (Figs. 6.14 and 6.15).

Table 6.9 Estimated regression coefficients of contaminants affecting water treatment costs at Durban Heights WW (before (original) and after removing the multicollinearity problem (final))

Explanatory Variable	Original model estimates		Final model estimates	
	Coefficients	t-values	Coefficients	t-values
Constant	-106.524	-1.43	67.890	
Temperature	0.692	2.10*	0.383	3.20*
Coliforms	-0.006	-0.71	0.005	2.10*
pH	9.659	1.11	-8.681	-3.62*
Turbidity	0.178	1.39	0.084	8.33**
Iron	-2.461	-0.39	1.613	1.60
Suspended Solids	0.059	0.36	0.084	3.62*
Secchi	-2.239	-0.93	-0.748	-1.42
Conductivity	-0.252	-0.19	0.617	3.14*
Total hardness	0.680	1.49	0.256	3.74*
Potassium	0.420	0.23	0.823	2.58*
Silica	2.922	2.52*	0.577	2.56*
Nitrate	5.431	0.54	3.177	2.40*
Total Organic Carbon	0.314	0.21	0.618	2.65*
Dissolved Oxygen	1.268	1.17	-0.940	-3.30*
Microcystis	0.020	2.11*	0.021	3.25*
Anabaena	0.013	1.97	0.001	2.86*
R^2		70%		64%
Adj. R^2		62%		62%

*Sig. at 5%, and **Sig. at 1% level.

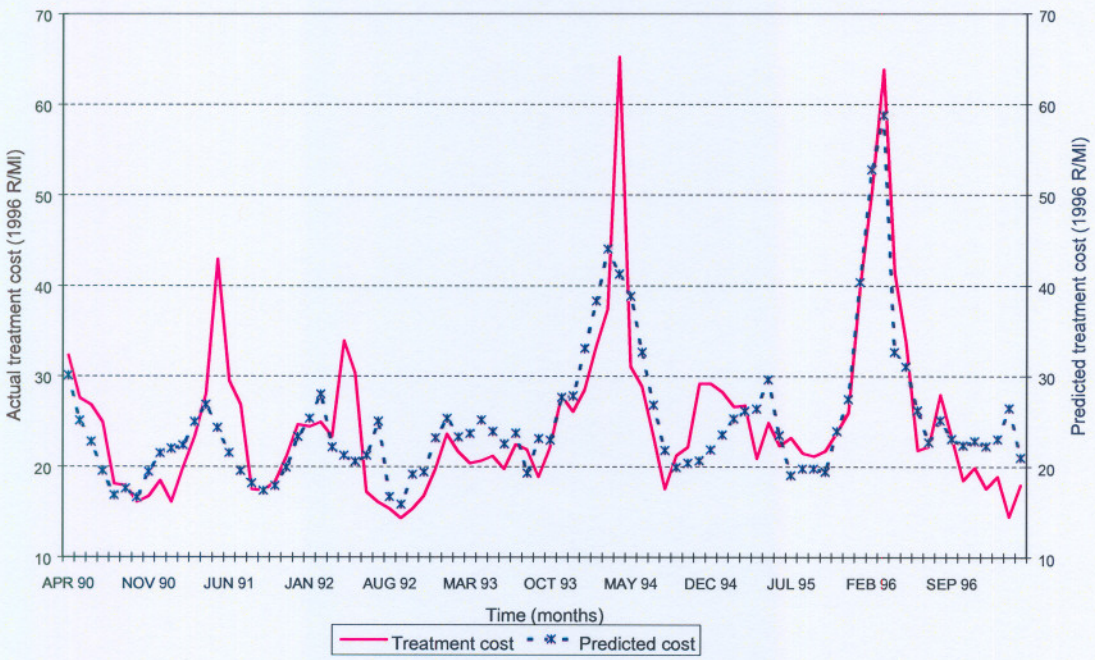

Figure 6.14 Actual versus predicted treatment costs over time fitted using coefficients of contaminants estimated in the final Durban Heights WW model.

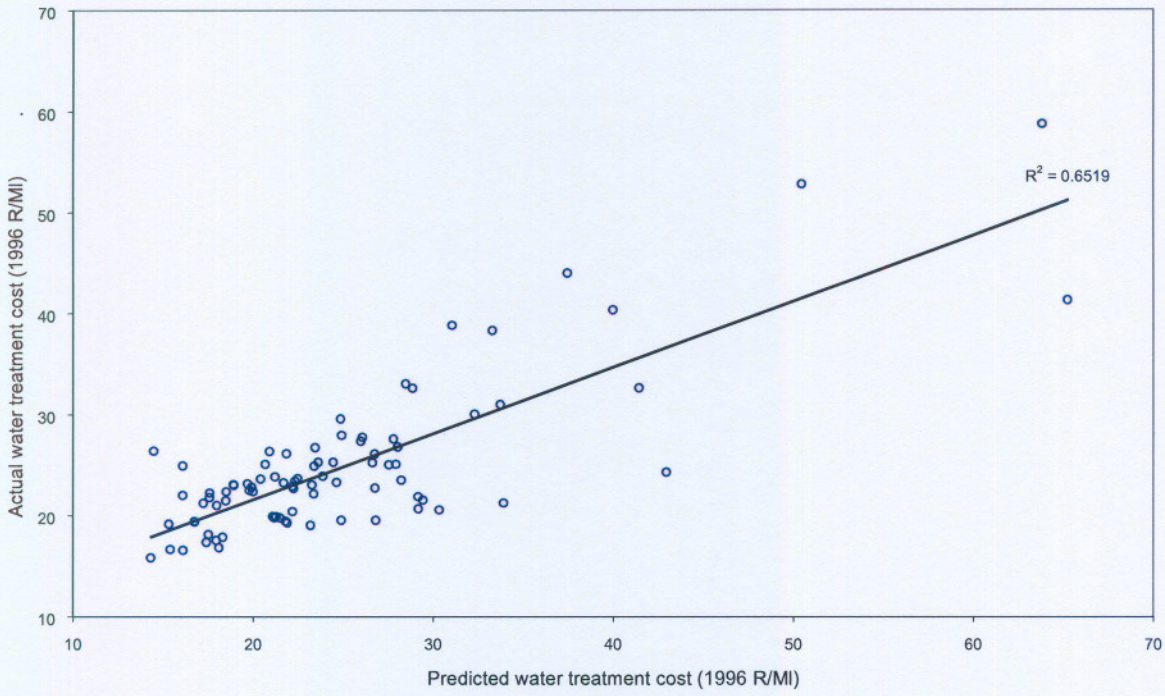

Figure 6.15 Scatterplot of actual versus predicted treatment costs for the Durban Heights WW model.

The algebraic form of the model estimated for the Durban Heights WW is as follows:

$$Cost = 67.9 + 0.383Temp + 0.005Colif - 8.681pH + 0.084Turb + 1.613Fe + 0.084SS - 0.748Secchi + 0.617Cond + 0.256Hard + 0.823K + 0.577Si + 3.177NO_3 + 0.618TOC - 0.940DO + 0.021\textit{Microcystis} + 0.001\textit{Anabaena}$$

Where;

Cost	=	Predicted water treatment cost			
Temp	=	Temperature	Colif	=	Coliforms
pH	=	pH	Turb	=	Turbidity
Fe	=	Iron	SS	=	Suspended Solids
Cond	=	Conductivity	Hard	=	Hardness
K	=	Potassium	Si	=	Silicon
NO_3	=	Nitrates	TOC	=	Total Organic Carbon
DO	=	Dissolved Oxygen			

The results in Table 6.9 show that treatment costs at the Durban Heights WW increase with an increase in water turbidity, suspended solids, silicon, total organic carbon, coliform numbers, conductivity, total water hardness, potassium, nitrates, *Microcystis* and *Anabaena*. These factors are directly related to water pollution of both natural (soil erosion, siltation) and anthropogenic (accelerated soil erosion, sewage and agricultural nutrient runoff) origin. Conversely, results show that

treatment costs diminish with an increase in water pH and dissolved oxygen in the water.

The cost increases associated with higher water turbidity are a direct result of increased use of polymeric coagulants. This is functionally the same relationship as observed in the Hazelmere WW system. The positive coefficient estimated for temperature confirms that treatment costs rise during summer, probably owing to increased rainfall runoff and the attendant deterioration in water quality (generally due to increased turbidity).

High coliform counts are often associated with high runoff (after rains) which washes faecal matter into watercourses and hence lakes. Concurrently turbidity, which is one of the primary determinants of cost, also increases with runoff. Therefore coliforms tend to co-vary with turbidity. Again, as with the Hazelmere WW system, there is an inverse relationship between pH and treatment costs at the Durban Heights WW. The reasoning for this is also the same as expressed for the former system.

Both the blue-green algal genera, *Anabaena* and *Microcystis*, carry positive and significant regression coefficients (Table 6.9), implying that an increase in their abundance leads to increased treatment cost at the Durban Heights WW. Figures 6.16 and 6.17 illustrate this positive relationship between the respective algae and water treatment cost at the WW. *Anabaena*, and to a lessor extent *Microcystis*, are algae widely recognised as causing taste and odour problems (Table 3.6). Advanced (and relatively expensive) treatment procedures e.g. powdered activated carbon (PAC) treatment, are required to remove taste and odour substances from water (e.g. Tebbutt, 1992).

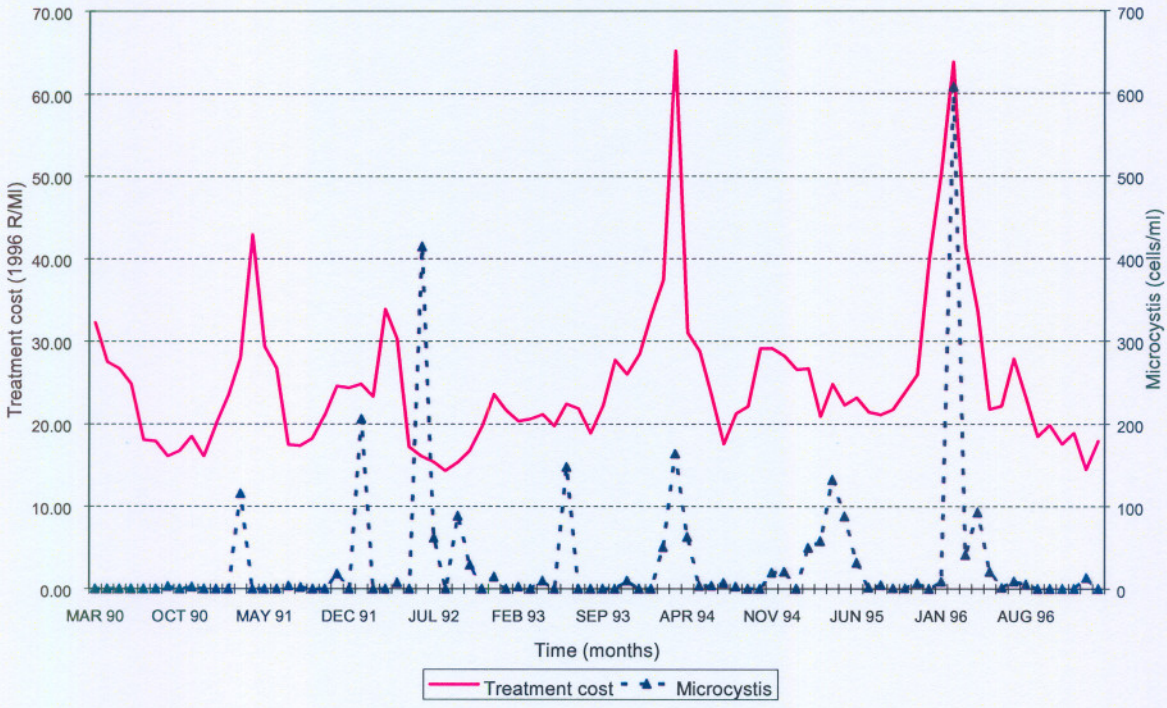

Figure 6.16 Treatment cost versus *Microcystis* counts in the raw water at the Durban Heights WW.

In view of previous work which indicates that a large proportion of algae, contained in water abstracted from Lake Nagle, disappears during transport from the lake to the treatment plant at Durban Heights (Dickens, *et al.* 1996), the Durban Heights model was originally estimated using flow-weighted algal data collected from the two Lake Nagle abstraction points (UW sample points 17 and 18). These results showed that *Anabaena* was the more important algal genus influencing water treatment cost at Durban Heights WW, while *Microcystis* was insignificant. However, the overall fit of the model was low (R^2 = 58 %). The reason for the low explanatory power could be attributed to a poor estimate of flow weighted algal counts expected in the raw water at the Durban Heights WW. The algal abundance at Durban Heights may in fact have been somewhat different and affected by scarce flow data at the two Lake Nagle abstraction points. The results given here for the Durban Heights WW are based on actual algal data collected at the Durban Heights raw water sample point (UW point 316), because the model provided a better fit.

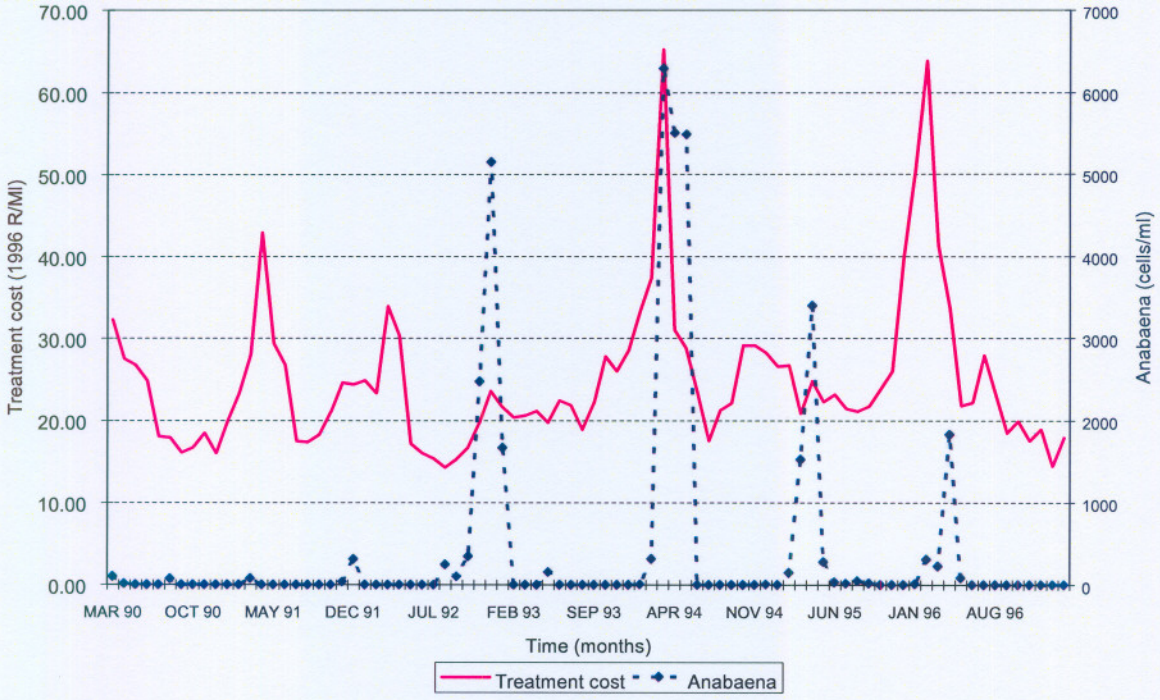

Figure 6.17 Treatment cost versus *Anabaena* counts in the raw water at the Durban Heights WW.

Clearly the peaks in algal data do not always appear to correspond with peaks in treatment costs indicating that not all blue-green algal blooms have associated taste and odour treatment problems. This fact is widely acknowledged in the literature (e.g. Harding & Paxton, 2001; WHO, 1999).

6.3.2.3 The DV Harris system

Results of the regression model estimated for the DV Harris WW are presented in Table 6.10. The final fitted model captures a reasonable amount of information (R^2=67%) on contaminants affecting water treatment costs at this WW.

Table 6.10 Estimated regression coefficients of factors affecting water treatment costs at DV Harris WW (before and after removing the multicollinearity problem).

CONTAMINANT	Original model estimates		Final model estimates	
	Coefficients	t-values	Coefficients	t-values
Constant	44.030	2.67*	45.500	
Alkalinity	-0.352	-0.85	-0.415	-6.70**
Coliforms	0.073	2.70*	0.028	6.13**
E.coli	0.026	0.37	0.075	6.12**
Total Hardness	-0.632	-1.44	-0.345	-6.16**
Potassium	3.242	1.24	2.150	3.02*
Trend Variable	0.119	3.09*	0.024	2.86*
Nitrate	6.608	1.98	5.075	3.37*
Manganese	103.724	1.01	-97.369	-4.11*
Conductivity	0.291	0.16	-0.508	-2.19*
Silica	0.008	0.01	0.285	1.50
Dissolved Oxygen	-0.682	-0.86	-0.463	-2.94*
Temperature	-0.051	-0.18	0.165	3.69*
Total Organic Carbon	-0.369	-0.64	0.234	1.27
Suspended Solids	-0.039	-0.45	-0.021	-0.55
Turbidity	0.032	0.43	-0.010	-0.61
Chlorella	-0.003	-0.63	-0.006	-6.53**
R^2		75%		67%
Adj. R^2		67%		64%
Number of cases	64			

*Sig. at 5% and **Sig. at 1%

Figures 6.18 and 6.19 show a reasonably good match between actual costs and those predicted by the model.

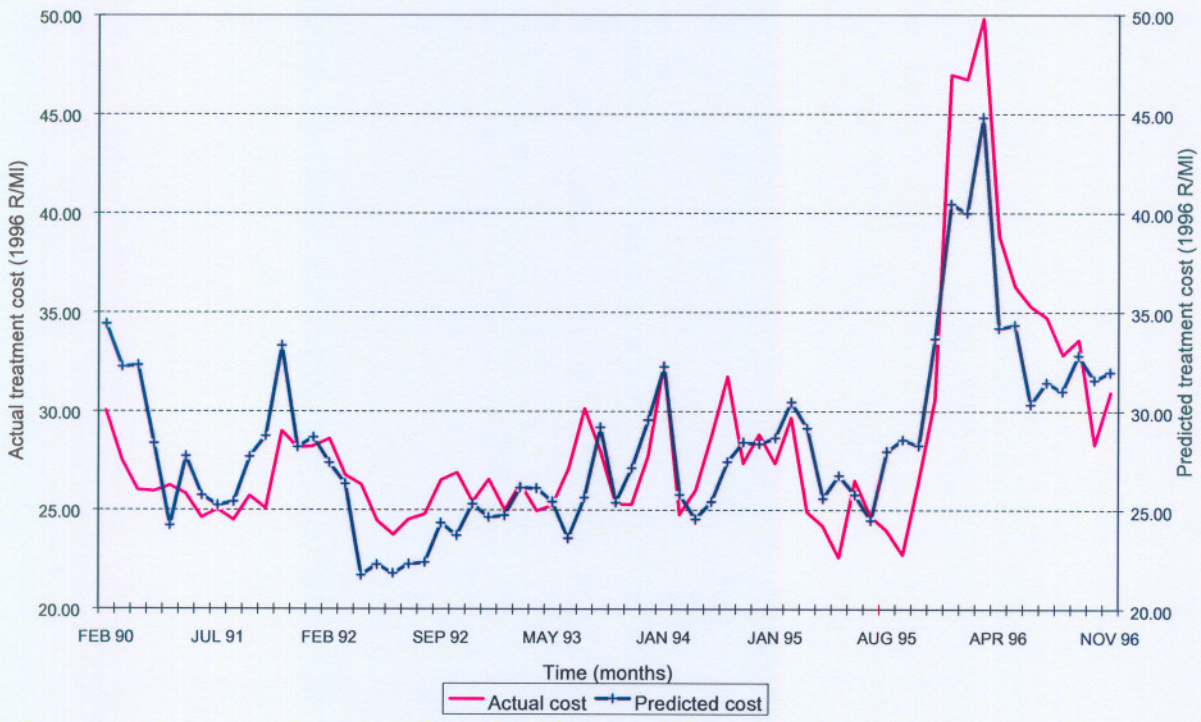

Figure 6.18 Actual versus predicted treatment costs over time fitted using coefficients of contaminants estimated in the final DV Harris WW model.

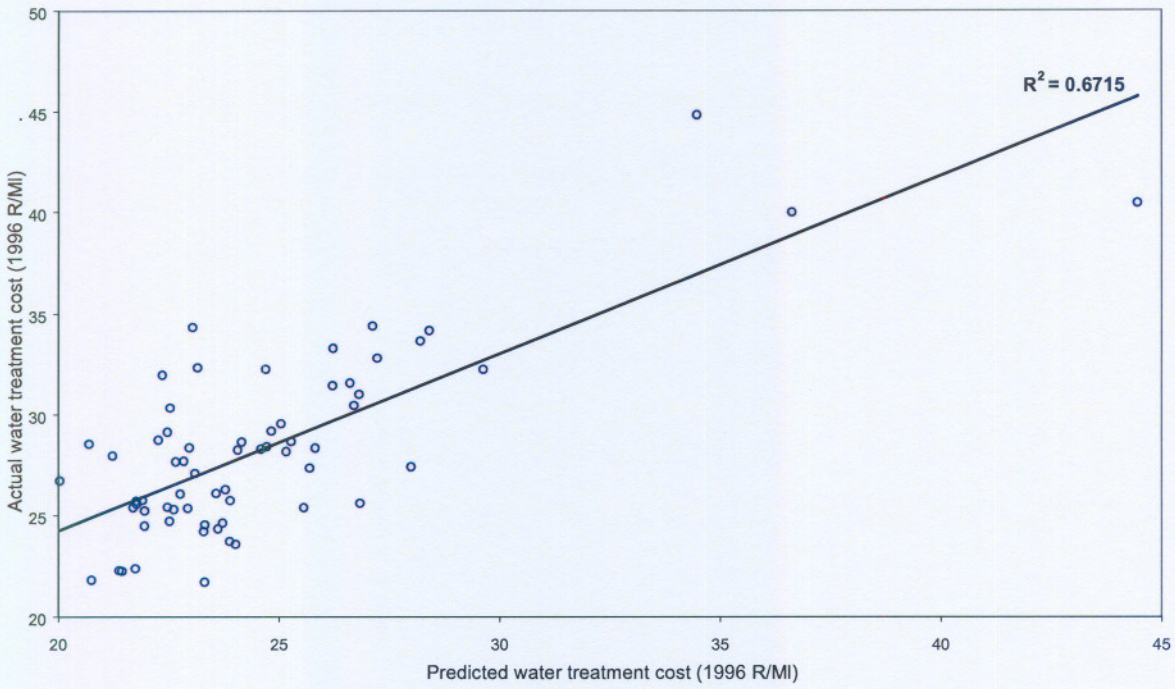

Figure 6.19 Actual versus predicted treatment for the DV Harris WW model.

The algebraic form of the model used to predict water treatment cost at DV Harris WW is as follows:

Cost = 45.5 - 0.415Alk + 0.028Colif + 0.075 E.coli - 0.345Hard + 0.024K + 0.024Trend + 5.075NO_3 - 97.4Mn - 0.508Cond + 0.285Si -0.463DO + 0.234TOC - 0.021SS - 0.010Turb - 0.006*Chlorella*

Where:

Cost	=	Predicted water treatment cost			
Alk	=	Alkalinity	Colif	=	Coliforms
E.coli	=	*E.coli*	Hard	=	Hardness
K	=	Potassium	Trend	=	Trend variable
NO3	=	Nitrates	Mn	=	Manganese
Cond	=	Conductivity	Si	=	Silicon
DO	=	Dissolved Oxygen	Temp	=	Temperature
TOC	=	Total Organic Carbon	SS	=	Suspended Solids
Turb	=	Turbidity			

The results suggest that real treatment costs at the DV Harris plant diminish with an increase in the alkalinity, *Chlorella* and Total Hardness, in Lake Midmar. As with the previous systems analysed, an increase in alkalinity (and hardness) of the water means a reduction in lime dosage and hence treatment costs. Likewise the presence of the alga *Chlorella* acts as a surrogate for clean water, so that when the water is turbid, *Chlorella* is less abundant, and the costs of treating such water increase.

Water treatment costs at DV Harris appear to rise significantly with an increase in coliforms and *E.coli* (Figure 5.19) in Midmar. Traditionally cost increases associated with coliforms and *E.coli* in raw water may be expected to be because of a higher chlorine-demand in this water. However, a plot of these contaminants against chlorine costs did not reveal any obvious relationships. An alternative explanation may be that these bacteriological variables generally increase with increased rainfall and runoff from catchments. Along with this comes an increase in turbidity and other variables that have a potential chlorine-demand and will therefore increase treatment costs. A plot of total organic carbon (which may have been anticipated to be related to chlorine demand) against chlorine costs did not reveal a close relationship.

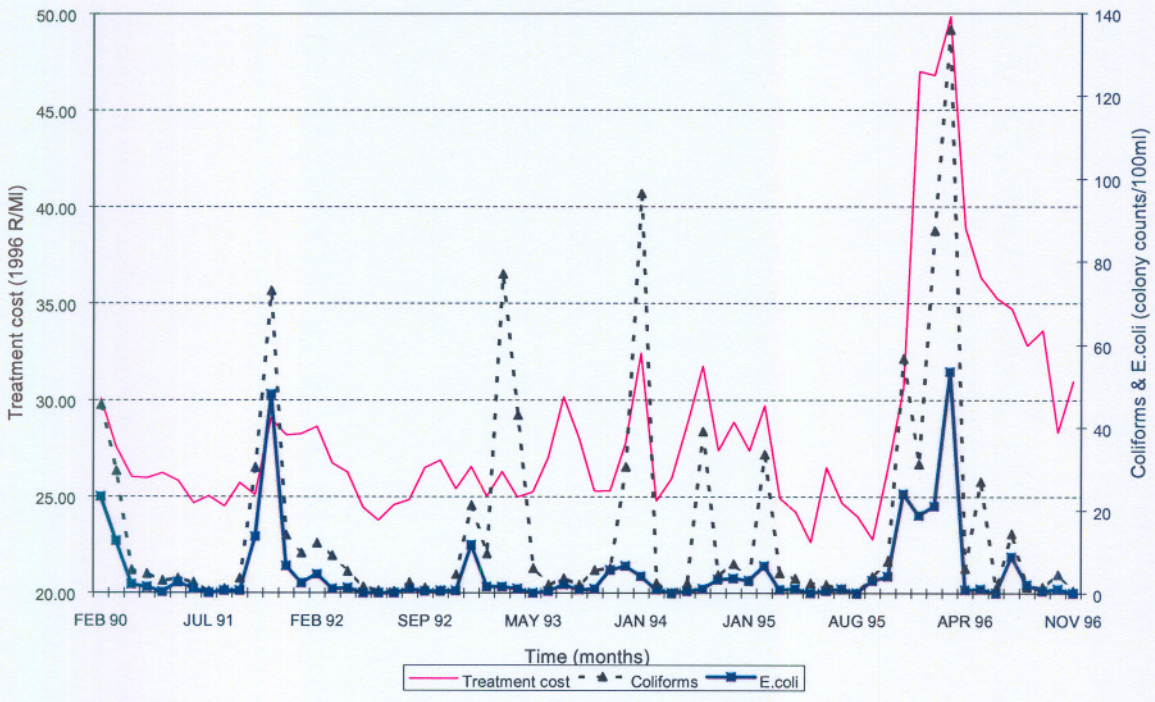

Figure 6.20 Water treatment cost versus number of coliforms & *E.coli* in the raw water to DV Harris WW.

Unlike all other WW considered, results indicate that turbidity is not significant in influencing treatment costs in the DV Harris system. In fact costs may increase under low turbidity conditions as bentonite has to be added to the raw water to increase the efficacy of polymeric coagulants.

The positive coefficient estimated for temperature confirms that treatment costs rise during summer, probably owing to increased rainfall runoff and the attendant

deterioration in water quality. The significance of the "trend" variable in the DV Harris model shows that over time treatment cost have been increasing in this system. The increase might be associated with factors other than those included in the model having a statistically significant effect on treatment costs at the DV Harris WW.

6.3.2.4 The Wiggins system

The final model fitted for the Wiggins WW (Table 6.11) captures a reasonable amount of information (R^2=79%) on factors affecting water treatment costs at this WW. Figures 6.21 and 6.22 show a good match between actual costs and those predicted by the model fitted for this WW.

Table 6.11 Estimated regression coefficients of factors affecting water treatment costs at Wiggins WW (before and after removing the multicollinearity problem)

CONTAMINANT	Original model estimates		Final model estimates	
	Coefficients	t-values	Coefficients	t-values
Constant	-34.288	-0.77	34.087	1.96*
Turbidity	0.303	6.21***	0.091	13.88***
Total Aluminium	0.020	1.35	0.016	6.96**
Iron	4.432	1.55	4.253	4.67***
Suspended Solids	-0.015	-0.19	0.104	3.76***
Nitrate	-3.77	-1.45	3.156	4.33***
Total Organic Carbon	-0.327	-0.30	1.196	5.15***
Total Dissolved Solids	-0.011	-0.47	0.024	3.90***
PH	4.497	0.79	-5.279	-2.32**
Silicon	0.654	1.18	0.486	3.23***
Trend variable	0.084	1.87*	0.032	2.25**
Coliforms	0.006	2.34**	0.009	3.79***
Sulphates	0.404	2.14**	0.090	1.78*
Temperature	-0.051	-0.18	0.465	2.42**
Anabaena	0.001	1.71*	0.001	2.01*
R^2:		92%		79%
Adj. R^2:		85%		77%
Number of cases:	82			

*Sig. at 10%, **Sig. at 5% and ***Sig. at 1%

The algebraic form of the predictive model estimated for the Wiggins WW is as given below:

$$Cost = 34.1 + 0.091Turb + 0.016Al + 4.253Fe + 0.104SS + 3.156NO_3 + 1.196TOC + 0.024TDS - 5.279\ pH + 0.486Si + 0.032Trend + 0.009Colif + 0.090SO_4 + 0.465Temp + 0.001Anabaena$$

Where:

Cost = Predicted water treatment cost; Turb = Turbidity; Al = Total Aluminium; Fe = Iron; SS = Suspended Solids; NO_3 = Nitrates; TOC = Total Organic Carbon; TDS = Total Dissolved Solids; pH = pH; Si = Silicon; Colif = Coliforms; SO_4 = Sulphates; Temp = Temperature

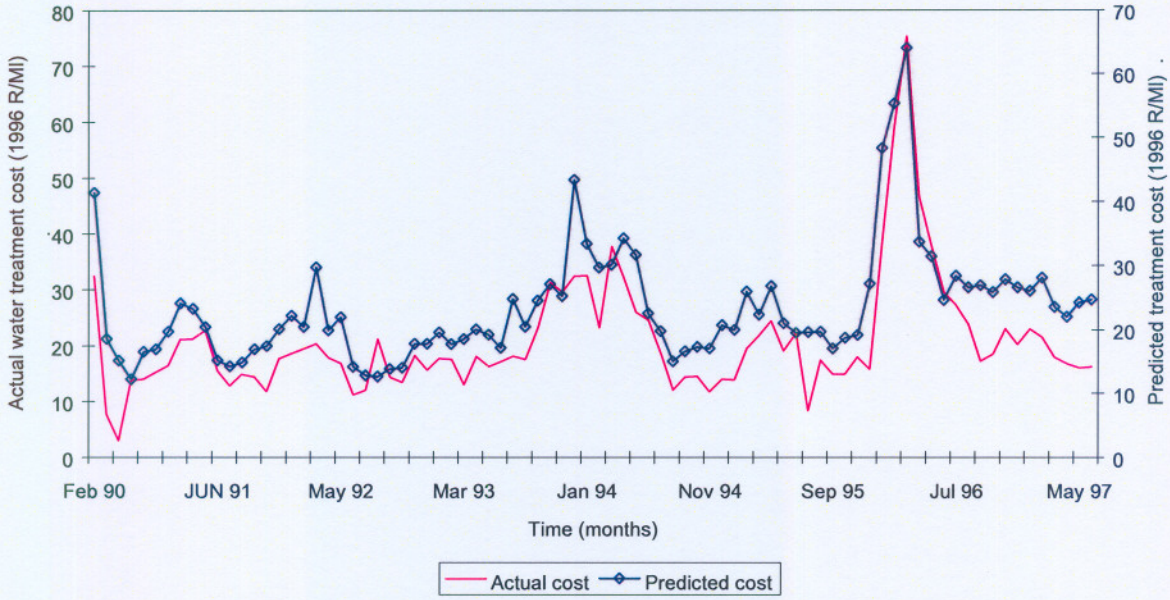

Figure 6.21 Actual versus predicted treatment costs over time fitted using coefficients of contaminants estimated in the final Wiggins WW model.

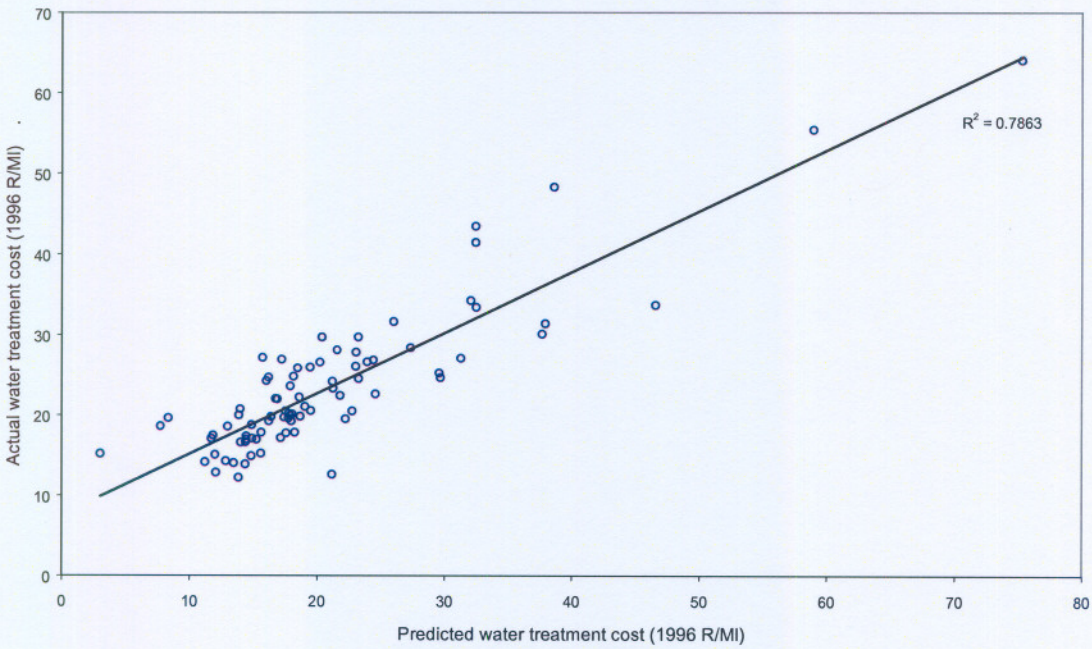

Figure 6.22 Scatterplot of actual versus predicted treatment costs for the Wiggins WW.

The results in Table 6.11 show that treatment costs at the Wiggins WW increase with an increase in water turbidity (e.g. Figure 6.23), total aluminium, iron, suspended solids, nitrates, total organic carbon, total dissolved solids, silicon, coliform numbers,

conductivity, total water hardness, potassium, nitrates, *Microcystis* and *Anabaena*. Conversely, results show that treatment costs diminish with an increase in water pH and dissolved oxygen in the water. As with the DV Harris system the significance of the "trend variable" shows that over time treatment costs appear to have been increasing in this system.

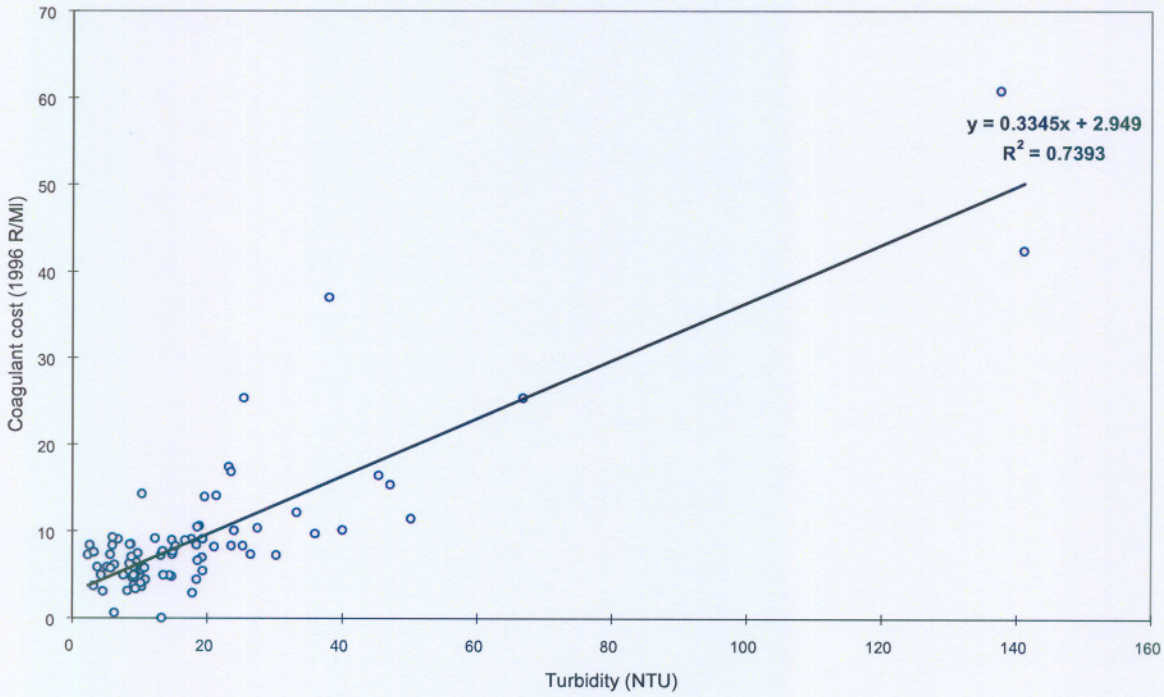

Figure 6.23 Polymeric coagulant cost versus raw water turbidity at the Wiggins WW.

6.4 Practical application of water treatment cost models

The water treatment cost models presented in section 6.3 may be used practically by WW operators to improve/optimise the water treatment process. They also quantify for catchment managers, agencies and planners those water quality processes that are likely to have an impact on the cost of treating water. Various 'what-if' and prediction type scenarios are possible with these derived models.

It should be emphasised that the derived models can be used to investigate numerous other scenarios, and this would obviously depend on the specific questions being asked about a particular system. To illustrate this approach two scenarios are presented here.

6.4.1 Predicting changes in water treatment costs with changes in raw water turbidity – the Hazelmere system example

The predictive model developed for Hazelmere WW is a useful management tool. Regression coefficients estimated using ordinary least squares (OLS) and principal components analysis (PCA) quantified the independent effects of each of the contaminants identified as statistically significantly affecting treatment costs. The model can therefore be used to predict the outcome of a management-induced change in one or more of the significant/explanatory contaminants. Plausible scenarios can be predicted rapidly using a standard spreadsheet package. As an example, the model was used to predict treatment costs assuming that turbidity is held at different levels. In lakes with abstraction towers, allowing a choice of abstraction levels, this would be feasible. Modifications to catchment land-use may also induce changes to lake turbidities, although in a less predictable manner. For example, sand-winning operations in the Mdloti River, at the inflow to this lake, are known to have an impact on in-lake turbidity.

Turbidity was identified as playing a significant role in affecting the cost of water treatment at Hazelmere Dam. At the same time it is acknowledged that controlling lake turbidity would have a direct impact on the levels of suspended solids and total aluminium in the lake and that, in any scenario planning exercise, these should also be allowed to co-vary in concert with turbidity. Figures 6.24 & 6.25 illustrate the respective relationship turbidity has with suspended solids and total aluminium.

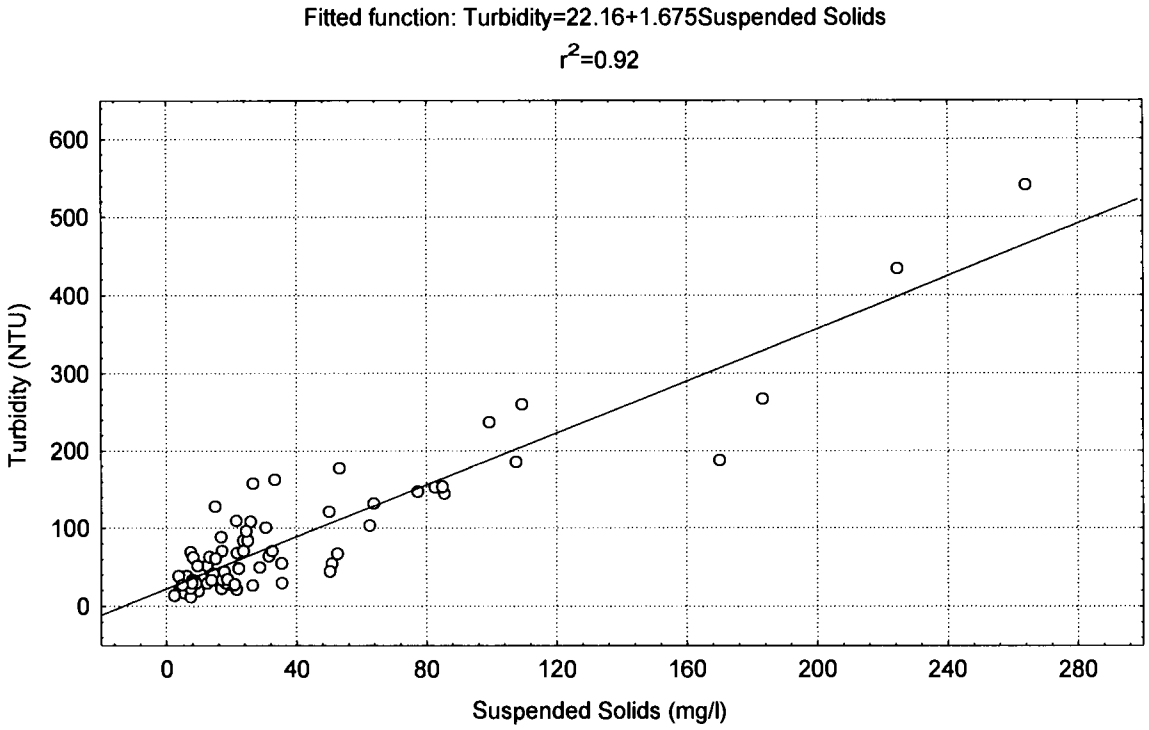

Figure 6.24 Linear relationship between Turbidity and Suspended Solids: Hazelmere WW raw water.

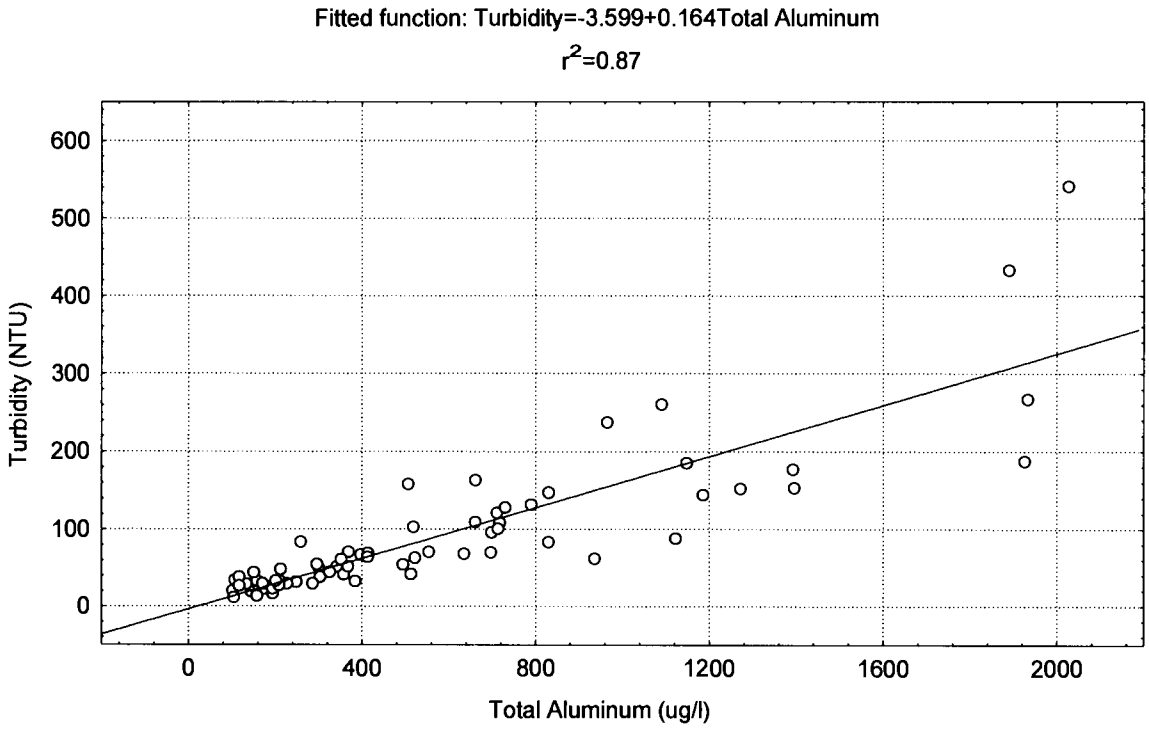

Figure 6.25 Linear relationship between Turbidity and Total Aluminium: Hazelmere WW raw water.

Assuming a strategy could be put in place whereby turbidity could be held constant (or selected for, e.g. variable off-take abstraction tower at the dam) and whilst holding

suspended solids and total aluminium at corresponding levels (as defined by the relationships in Figs. 6.24 & 6.25), the impact on treatment costs of such an approach could be examined. The results of such a prediction exercise are given in Table 6.12, and graphically presented in Figure 6.26.

Table 6.12 Predicted treatment cost, holding turbidity (and corresponding levels of suspended solids and total aluminium) constant, but allowing other factors to vary with real data at Hazelmere WW

Turbidity (NTU)	Suspended Solids (mg/ℓ)	Total Aluminium (ug/ℓ)	Predicted mean cost (1996 R/Mℓ)	Difference in cost from the mean cost (R 40.59)
0	0	22	34.36	6.23
10	0	83	34.85	5.74
20	0	143	35.34	5.25
30	5	204	36.08	4.51
40	11	265	36.87	3.72
50	17	326	37.66	2.93
60	23	387	38.45	2.14
70	29	448	39.24	1.35
80	35	508	40.02	0.57
90	41	569	40.81	-0.22
100	46	630	41.55	-0.96
.	.	.	.	.
.	.	.	.	.
200	106	1238	49.44	-8.85
.		.	.	.
.		.	.	.
550	315	3367	76.98	-36.39

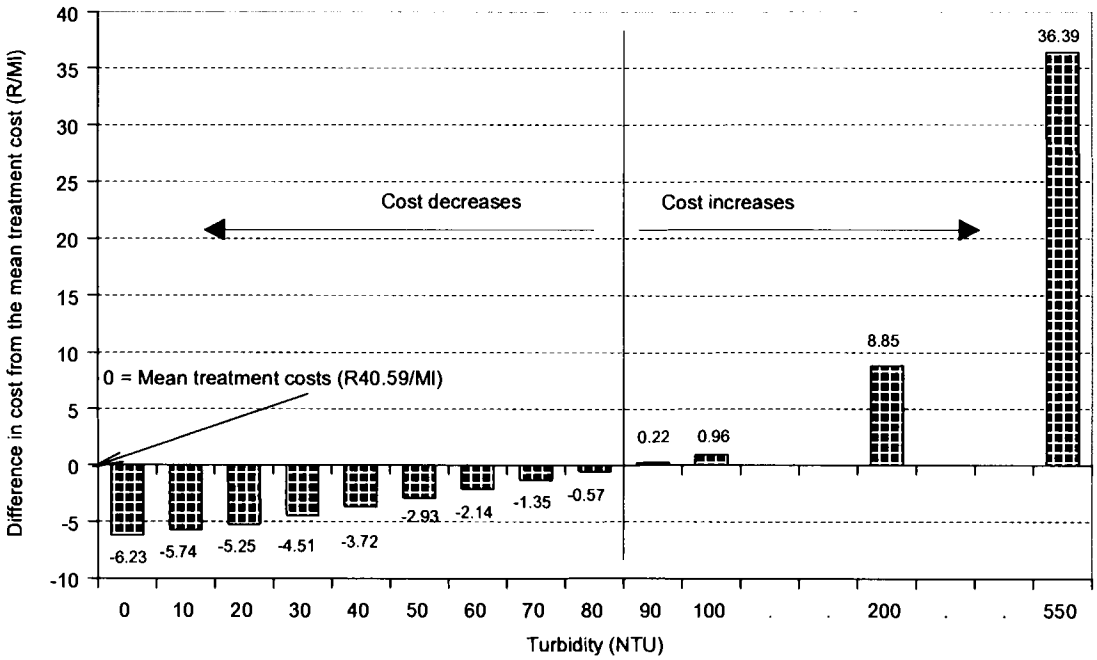

Figure 6.26 Difference in predicted costs from mean treatment cost (R40.59/ℓ) at different turbidity levels in the Hazelmere WW.

Figure 6.26 illustrates the differences in predicted treatment cost from the mean treatment cost with varying turbidity scenarios. Water treatment costs in this figure are observed to rise above the average (R40.59/Mℓ), recorded at Hazelmere WW over the period January 1990 to December 1996, when turbidity levels exceed 80NTU. Implications are that any management action that either, keeps the turbidity level below 80 NTU in Lake Hazelmere, or allows for selection of water from the water column below 80 NTU, would lower the average water treatment costs at Hazelmere WW. For example, if turbidity is held at 20 NTU, average treatment cost would be lowered by R5.25/Mℓ (*ceteris paribus*). From a financial perspective (based on 1996 chemical treatment costs and approximately 8500Mℓ treated) this could amount to a theoretical cost saving of approximately R45 000 annually.

6.4.2 Predicting changes in water treatment costs with changes in raw water turbidity and *Anabaena* abundance – the Durban Heights system

The raw water abstracted from Lake Nagle (and occasionally Inanda) and fed to Durban Heights, is characterised as relatively turbid with occasional blooms of *Anabaena* and *Microcystis* that may significantly affect water treatment costs. The model estimated for Durban Heights WW was used to predict possible cost scenarios given changes in water turbidity and *Anabaena* counts.

As with the Hazelmere system, a turbidity/suspended solids relationship exists in raw water entering Durban Heights as described in Figure 6.27. Hence for any changes in turbidity, suspended solids levels are co-varied in accordance with this observed relationship. However, unlike the Hazelmere system, results from analyses of water treatment costs and contaminants (Table 6.9) show that total aluminium does not statistically significantly affect treatment costs at the Durban Heights WW.

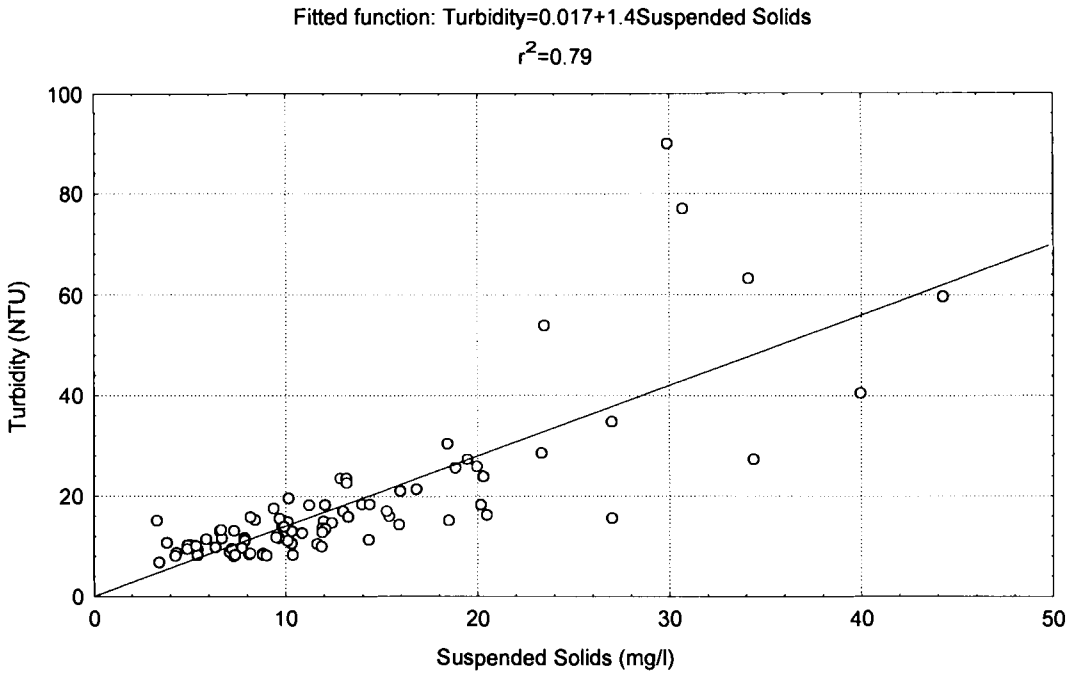

Figure 6.27 Linear relationship between Turbidity and Suspended Solids: Durban Heights WW raw water.

Treatment costs over the period are predicted with turbidity (and corresponding levels of suspended solids) held at different levels (Table 6.13), and allowing other factors to vary with real data.

Table 6.13 Predicted treatment cost, holding turbidity (and corresponding levels of suspended solids) constant, but allowing other factors to vary with real data at Durban Heights WW

Turbidity (NTU)	Suspended Solids (mg/ℓ)	Predicted mean cost (1996 R/Mℓ)	Deviation from mean cost (R 40.59)
0	0	22.23	2.61
6	4	23.06	1.78
10	7	23.65	1.19
20	14	25.06	-0.22
30	21	26.48	-1.64
40	29	27.98	-3.14
50	36	29.40	-4.56
60	43	30.81	-5.97
70	50	32.23	-7.39
80	57	33.65	-8.81
90	64	35.07	-10.23

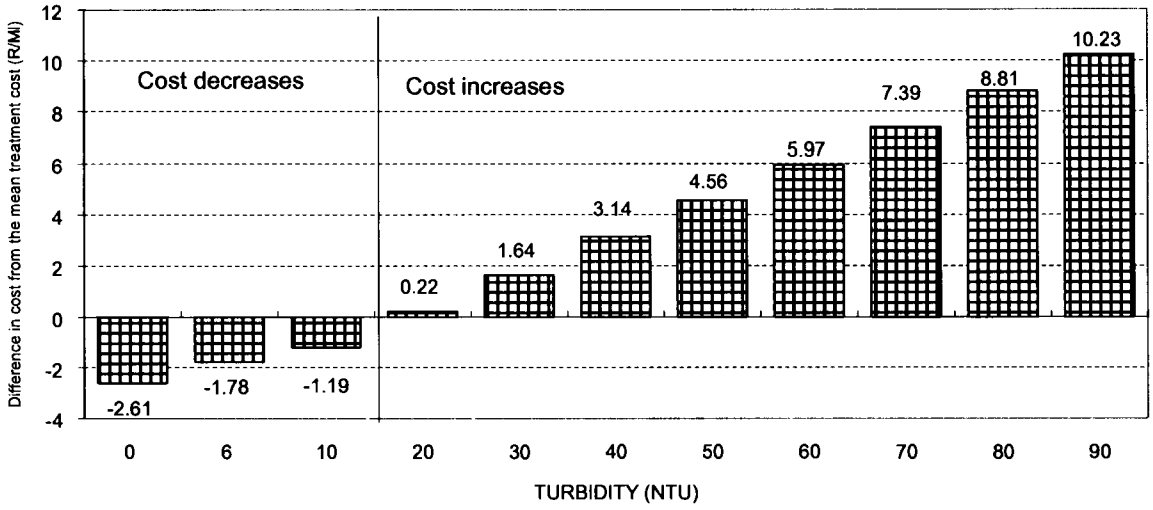

Figure 6.28 Differences in mean treatment cost at different turbidity levels at Durban Heights WW.

Figure 6.28 shows that water treatment costs are observed to rise above the mean (≈R25/Mℓ) water treatment cost recorded at the Durban Heights WW when turbidity levels exceed 20NTU. Implications are that any management action that keeps the turbidity level below 20NTU at this WW will lower the mean water treatment costs. For example, if turbidity is held at 6NTU, this could lower the mean treatment cost by R1.78/Mℓ. From a financial perspective (based on 1996 figures) this could amount to a saving of approximately R350 000 annually, with a total of ≈190 000Mℓ treated. Conversely, as turbidity rises, mean treatment cost will increase by the magnitudes shown in Figure 6.28.

In a similar way the Durban Heights model is also used to predict the possible effects of different numbers of the alga *Anabaena* on cost. Results of these simulations are illustrated in Figure 6.29.

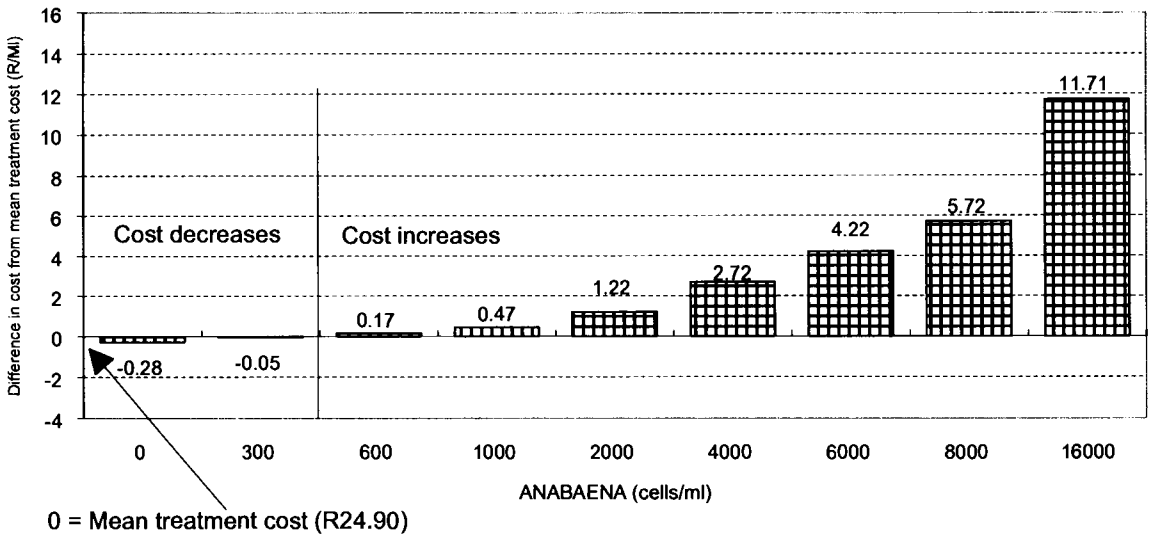

Figure 6.29 Differences in mean treatment cost with different abundances of *Anabaena* in the raw water treated at the Durban Heights WW.

Figure 6.29 illustrates that an increase in the abundance of *Anabaena* beyond ≈600 cells/mℓ would result in an increase in mean treatment cost. For example, treatment costs would increase by approximately R4.22/Mℓ (over the mean treatment cost of ≈R25/Mℓ) if the *Anabaena* counts in the raw water entering Durban Heights WW were 6000 cells/mℓ, and causing taste and odour problems. Currently, at >1000 cells/mℓ of *Anabaena,* operators of the WW are advised to be aware of potential taste and odour problems associated with this alga. This 1000 cells/mℓ of *Anabaena* cut-off was derived empirically and corresponds well with the results of this simulation.

6.5 Summary discussion

Clearly lake water quality has a significant impact on the cost of treating water in each of the major WW examined in this study. The study has shown that the cost of treating water per mega litre (Mℓ) is highest at the Hazelmere WW (R41) followed by Durban Heights WW (R28), DV Harris WW (R25) and Wiggins WW (R22). These latter three WW are all in a similar 'ballpark,' with respect to treatment costs, whilst the costs in the Hazelmere system are almost double the others.

Important conditions contributing to changes in real treatment costs at each water plant are identified in this study. Physico-chemical contaminants generally affected water treatment more significantly than algae except where potentially powerful taste and odour forming algae (e.g. *Anabaena*) were present, as for example in the Nagle/Durban Heights WW system.

Physico-chemical contaminants (especially those associated with elevated turbidity levels) clearly increase treatment costs at the Hazelmere WW. Algae have a relatively minor impact on treatment costs at this WW. It is surmised that the generally high turbidity conditions in Lake Hazelmere only allows for limited algal growth.

The model derived for the Hazelmere WW explains 79% of the variation in chemical treatment costs. The model predicts actual costs quite well and can be easily applied in simulation exercises. The study shows that a management strategy that reduces the turbidity of Lake Hazelmere would reduce water treatment costs at the Hazelmere WW. Control of turbidity in Lake Hazelmere to levels below 10 NTU could, however, lead to possible algal growth problems. Nevertheless, changing the Hazelmere system from a "turbidity-driven" system to an "algal-driven" system could lower the costs of treating such water. However, costs could increase if conditions created in Lake Hazelmere favoured the growth of taste and odour forming algae like *Anabaena.*

Post research note: The specific Hazelmere model was explored and applied in practise after the completion of the research for this thesis. Umgeni Water were able to explore the impacts on treatment costs of an application for a proposed sand-winning operation in the uMdloti river (above the inflow to Lake Hazelmere). The sand-winning operation would obviously have affected in-lake water quality (specifically turbidity) and hence treatment costs.

The model estimated for the Durban Heights WW explains some 64% of the variation in chemical treatment costs. The model predicts actual costs well (except during occasional peak cost periods) and can be easily applied in simulation exercises. The study identifies some important contaminants contributing to treatment costs at the Durban Heights WW. Results show that environmental (physico-chemical) and algal contaminants have a marked impact on treatment costs. An increase in *Anabaena*

and *Microcystis* numbers in Lake Nagle have a major impact on treatment costs at the Durban Heights WW, particularly when they are producing taste and odour compounds.

The model developed for the DV Harris WW explains 67% of the variation in chemical treatment costs. Paradoxically, clean water - typical of Lake Midmar is expensive to treat, due to the need to dose bentonite. The significance of the "trend variable" in the DV Harris model shows that over time treatment costs have been increasing in this system. This might be associated with factors, other than those included in the model, having a significant effect on treatment costs at the DV Harris WW.

The model fitted for the Wiggins WW explains 79% of the variation in chemical treatment costs. Again results indicate that over time treatment costs have been increasing in this system. This could be from either a deterioration in raw water quality coming into this WW or from a reduction in efficiency in the operation of the plant and hence a need to dose more chemical to achieve the potable water standards.

This chapter of the thesis identifies the key contaminants affecting water treatment costs at respective WW. It also quantifies how an integrated catchment management strategy, that would reduce both point and diffuse nutrient loads and turbidity in the lakes, could reduce water treatment costs. The impact of such control strategies (if they were implemented) could also be monitored to measure their success (in terms of impacts on water treatment) with the tools developed here. The tools also illustrate how treatment costs may be predicted from observed levels of contaminants.

CHAPTER 7: SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

7.1 Summary

7.1.1 Introduction

Chapter 1 of this thesis discussed the major impacts of eutrophication on water resources in South Africa. Kwa-Zulu Natal, and specifically the area managed by Umgeni Water, is not immune to these impacts. Previous work, for example, Noble & Hemens (1978), Grobler & Silberbauer (1984), and most particularly Walmsley (2000), has highlighted the following policy and research needs surrounding eutrophication of surface waters in South Africa:

- quantitatively assessing the eutrophication problem in terms of its extent and trends;
- the sources of nutrients and levels entering aquatic systems; and
- the actual social and economic costs of the problem on a national basis.

The investigation reported on in this thesis goes some way towards addressing the aforementioned research needs, particularly at a regional level.

The broad aims were to:

- identify the key environmental variables that were affecting algal abundance and diversity in lakes; and if these can be used,
- to establish predictive models relating algae to specific water quality parameters; and finally
- to develop models relating the water quality in lakes to the cost of treating that water.

In the present investigation, semi-quantitative models were developed relating algal abundances with important elements in their environment. The models developed were essentially empirical in nature and relied on appropriate standard multivariate and univariate statistical analyses and investigations. Patterns and relationships identified by these analyses were used to build simple models within a spreadsheet environment. These models went some way to describing the complex water quality and water treatment cost processes within the lake systems and water treatment works studied. In most cases, the models developed were related to algae that were known to adversely affect water treatment

(e.g. Palmer, 1959). Direct algal impact on water treatment processes was through the production of either, taste and odour forming compounds (requiring advanced water treatment processes, e.g. use of activated carbon), or their ability to clog sand filters and so reduce filter run times (requiring more frequent backwashing of filters).

Thereafter lake water quality parameters (which included water physico-chemistry and algae) were statistically examined to determine which factors were impacting on water treatment and hence treatment costs at selected water works (WW) within the Umgeni Water operational area. Models were developed relating raw water quality entering respective water works with costs incurred in treating that water. Various 'what-if' scenarios were developed illustrating the practical application of these models.

7.1.2 Lake Shongweni identified as an extreme case of eutrophication

Prior to analyses relating algae to environmental conditions it was shown (Chapter 4, section 4.3) that Lake Shongweni appeared to be an outlier at the extreme end of an eutrophication gradient, both in terms of its physico-chemistry, as well as its algal composition and abundance. It was therefore excluded in further analyses. The distinction between Shongweni and other lakes in the study was based primarily on its high conductivity waters. High conductivities in the Shongweni waters reflected a range of variables (e.g. potassium, chloride, sodium, total dissolved solids and sulphate) that had highly elevated concentrations compared to all other lakes studied. This condition of poor water quality in Shongweni could be traced to catchment activities (primarily textile industries), which contributed to the significant conductivity loadings on this lake. Highly elevated algal counts of the blue-green algae *Microcystis* and *Anabaena* appeared to account for the biological distinction between Shongweni and the other lakes. It was on the basis of its poor water quality that Lake Shongweni was decommissioned as a routine water supply lake (Wilson, 1992).

With Lake Shongweni removed from the investigation the focus of the study was then concentrated on the remaining seven lakes viz. Midmar, Albert Falls, Nagle, Inanda, Henley, Nungwane and Hazelmere (Chapter 5).

7.1.3 Reduction in the number of environmental variables used in analyses

Many of the environmental variables investigated in this study were statistically significantly correlated with each other resulting in a situation of mulicollinearity (Chapter 4, section 4.4).

This created problems in analysis of the data and appropriate analytical techniques were therefore employed to counter the multicollinearity. The result was that a degree of economy was achieved in that the variation of the original large number of environmental variables could be reasonably accounted for by a smaller number of 'key' environmental variables. These were identified as conductivity, secchi depth, silicon, total inorganic nitrogen and total phosphorus, lake inflow volumes, temperature, percentage saturation of dissolved oxygen and water column stability. The selected variables represented the primary aspects of variation in the environmental data. This set of nine environmental variables also had intuitive biological appeal in that they are often implicated in the literature as influencing algal populations (e.g. Tilman *et al.*, 1982; Wetzel, 2001).

7.1.4 Physico-chemical differences between lakes studied

A preliminary investigation of purely physico-chemical differences and similarities between lakes produced useful findings (Chapter 4, sections 4.4 & 4.5). These had potential management and policy implications, as lakes that could be considered to have similar physico-chemistry would be likely to respond similarly to management actions or catchment modifications.

A distinct group, of relatively low conductivity, low turbidity, 'inland' lakes (Midmar, Albert Falls, Nagle and Henley) were identified. These were distinct from more turbid lakes (Hazelmere and to a lesser extent Nungwane) in the coastal area. Inanda appeared different from either of these previous groupings in that it was characterised by generally higher conductivities but with low turbidity. From a physico-chemical point of view, Inanda and Hazelmere are the most dissimilar lakes in the study (excluding Shongweni). On the other hand Midmar, Albert Falls and Nagle are relatively similar when compared to the other lakes in the study. They demonstrate the change in water quality within lakes with progression down the uMngeni River catchment. Not surprisingly, progression down catchments was characterised by increased conductivities. Inanda, the last lake on the uMngeni system cascade, had the highest conductivities.

Further (though weaker) distinctions between lakes could be identified in relation to gradients characterised by total inorganic nitrogen, total phosphorus and inflows. Midmar, Nagle and Albert Falls appeared to be characterised by higher total phosphorus, whilst Nungwane and Henley had higher total inorganic nitrogen. Inanda was relatively 'mixed', experiencing a range of values for total inorganic nitrogen, total phosphorus and inflows.

7.1.5 Relationships between algae and their environment

A reduced set of 'key' environmental variables (conductivity, secchi, silicon, total inorganic nitrogen, total phosphorus, inflow, temperature, % dissolved oxygen and stability) (Chapter 5, section 5.3), lake unique effects (section 5.4), and seasonal effects (section 5.5), were then used to determine which aspects of the environment were playing the most significant part in determining observed algal heterogeneity in the respective lakes. Each of these analyses revealed a different aspect of how environmental variables were influencing the distribution and abundance of algae (first objective of this study). Interpretations were then made about the conditions favouring problematic algae.

Chapter 5, section 5.3, highlights how the reduced set of 'key' environmental variables explained some 16% of the variability in the algal data. Of these variables, total inorganic nitrogen, silicon, temperature, inflow and secchi appeared to be the parameters accounting for the major portion of the variability in the algal data explainable by available environmental data. The low amount of explained variation is not unusual for large multi-genera ecological data. As a check on the significance of the relatively low variability explained, the environmental effects were Monte Carlo permutation tested and found to be statistically significantly related to the variation in the algal data ($P < 0.01$). Therefore these variables (total inorganic nitrogen, silicon, temperature, inflow and secchi) appear to account for a statistically significant proportion of the variability in the observed algal data.

Of the algal genera apparently responding to the environmental gradients, the blue-green algal genera, *Anabaena* and *Microcystis,* were responding to higher temperatures and inflow volumes that one would expect with summer conditions whilst they were less affected by higher silicon and total inorganic nitrogen values. The temperatures (20–25°C) leading to the highest abundance of the blue-green algae *Anabaena* and *Microcystis* were in accordance with those noted in the literature (e.g. Romo & van Tongeren, 1995, Bahnwart *et al.*, 1999; An & Jones, 2000).

It was recognised (and should be emphasised) that the reduced set of 'key' environmental variables are closely associated with particular groups of other variables and accordingly all the associated variables are useful in interpretations of algae and environment. For example, positively associated with total inorganic nitrogen (TN) was the TN:TP ratio. This implies that at low TN concentrations there are also likely to be low TN:TP ratios. The blue-

green algae (*Anabaena* and *Microcystis*) appeared to be more abundant at the lower end of this TN (low TN:TP ratio, < 20) gradient. The literature again supported this finding (e.g. Schindler 1977; Tilman *et al.*, 1982, 1986).

Although with lower amounts of their variability explained by the key environmental variables, conversely *Navicula, Spermatozopsis* and *Mallomonas* were responding to higher silicon (>5mg/ℓ) and total inorganic nitrogen (> 0.5mg/ℓ) and lower temperatures (15-20°C) and inflows. Cool (15-20°C), clear waters (with high secchi depths i.e. >1.5m), with low total phosphorus (<20µg/ℓ), favour *Crucigena,* whilst waters generally low in total inorganic nitrogen (<0.5mgN/ℓ) and silicon(<5mg/ℓ) favours *Melosira, Scenedesmus, Cyclotella* and *Tetraedron*. These waters are also generally clear (higher secchi depths, i.e. > 2.0m) and have lower water column stability.

Unfortunately not much can be said about *Chlorella*, a dominant or sub-dominant alga in many lakes, as its variability is not well explained by the measured environmental variables. Other results (Chapter 5, section 5.5) appear to indicate that there may be some unmeasured (in this investigation) environmental parameter which varies seasonally and which is affecting *Chlorella*. This aspect is in need of further investigation.

7.1.6 The effects of differences unique to individual lakes ('lake unique' differences) on algae

It has to be acknowledged that not all of the parameters that may be affecting algae in a lake can always be measured or quantified in a biologically meaningful way (e.g. Soudant *et al.*, 1997). These 'other' parameters may be of a biological, physico-chemical or seasonal nature (or various combinations of all of these). It is this un-quantified element of variability in a lake that prevents full understanding, and therefore prediction of, all the dynamics and processes that result in the abundance and variation in algal populations observed. Often certain parameters may be unique to (e.g. basin morphometry, water quality) or vary uniquely in respective lakes (e.g. local weather patterns), thus giving that lake its unique set of characteristics affecting local algal populations. Analyses to determine the response of algae to this 'uniqueness' of respective lakes (lake unique differences) were conducted (Chapter 5, section 5.4). In this way lakes with similar characteristics could be expected to have a similar composition and abundance of algal populations. Concurrent with this,

environmental variables were examined to determine possible environmentally based reasons for the differences (or similarities) observed between lakes.

Hazelmere is distinctly different from all other lakes studied. This difference appears primarily associated with low water clarity (high turbidities/low secchi depths). Of all the lakes studied Hazelmere, Nungwane and Inanda (and to a lesser extent Nagle) are biologically most dissimilar from each other. Given that they are all on entirely different river systems (except Nagle and Inanda on the uMngeni River) with different water physico-chemistry this is not surprising.

There is an interesting trend, as quantified by the algal heterogeneity of lakes, from generally upper catchment (Henley), down the uMngeni system sequentially to the coastal lakes (Nagle and Inanda). Nungwane is an apparent anomaly in this trend as it could be geographically classified as a 'coastal' lake. However, it appears to have a greater affinity to the upper catchment lakes and indeed is situated relatively high up in its catchment. Examination of the environmental data indicates that this trend down the catchment is related to conditions associated with increasing inflow, temperature and conductivity approaching the coast on the uMngeni system with parallel decreases in stability and lowered silicon and total inorganic nitrogen values.

Conditions associated with silicon and total inorganic nitrogen concentrations appear to be the most important variables associated with the observed spatial differences between the lakes - particularly Henley and Nungwane compared to all the other UMngeni system lakes.

The relatively high importance attached to silicon, as it distinguished between the different lakes in the study, could possibly be related to the respective sizes of the lakes. The main basins of smaller lakes (e.g. Henley and Nungwane) are likely to be more prone to, and therefore influenced to a larger degree by, seasonal fluctuations related to inflow. Larger lakes on the other hand (with their main basins often some distance from the inflow) tend to 'absorb' the impact of floods more readily, with the bulk of the silicon loadings generally confined to and processed within the lake inflow area. Associated with seasonally variable inflows (particularly flood type events) are the clay and silt loadings that will affect the silicon concentration in the water. The hypothesised dynamics between lake size and silicon variability appears to be borne out in this study and help elucidate the importance silicon has in explaining the difference (as measured by the algal populations dominating them) between Henley and Nungwane and the other larger lakes. Diatoms, which are typically

associated with higher silicon concentrations, do not appear to be responding particularly strongly in Henley and Nungwane.

Lakes at the lower end of the uMngeni River system (Inanda, Nagle and Albert Falls) appear to have less stable water column conditions. The relative size of lakes may explain this as Henley and Nungwane were the smallest studied and therefore potentially less prone to the effects of winds. Generally the smaller the dimensions of the lake, the lower the fetch (uninterrupted distance wind is able to travel across the water body) and hence the lower the impact of wind induced mixing currents on the water column, which obviously affects stability. Notwithstanding this explanation, however, there is also the counter argument around the relative inertia a larger water body would have over a smaller. Because of their relatively higher inertia, a larger water body would require a greater input of energy to induce mixing compared to the smaller.

The warmer conditions in lakes at the lower end of the uMngeni River system (Inanda and Nagle) appear to produce less stable water column conditions. The typically steady onshore coastal breezes, combined with the east – west valley orientation (funnelling the breezes up from the coast), may be a possible explanation for the reduced stability in these more coastal lakes. Unfortunately no measure combining duration, direction and speed of wind was available, or derived, to verify this conclusion and this aspect is in need of further investigation.

7.1.7 The effects of 'season' on algae

Chapter 5, section 5.5 illustrates how 'seasonal' effects account for a relatively small proportion of the observed algal variability, as compared to that of the 'key' environmental and 'lake-unique' variables. That said however, the effect of season is still clear and accounts for a statistically significant portion of the algal variability. Seasonal successional trends (within the year, i.e. on a monthly basis) and annually (i.e. over the duration of the study) were examined to determine how these may be affecting algal populations in the lakes studied.

The annual successional trends are less significant than monthly patterns and show no strong unidirectional trend within the time period of the study. This indicates that there was no overall successional trend in the algal populations in the lakes under study.

Fewer algal genera have significant portions of their variability explained in the 'seasonal' analysis compared to either of the previous two analyses (key environmental and lake-unique variables). This emphasises the lower importance seasonal (monthly) effects have on algal populations compared to pure environmental and effects unique to different lakes.

Notwithstanding the fact that seasonal effects accounted for a relatively small proportion of the observed algal variability, *Chlorella* had a greater proportion of its variability explained by seasonality than could be explained by either key environmental or lake unique effects. That seasonality is significant for *Chlorella* is important when it is considered that this alga was a dominant to subdominant genus in many lakes. The implication is that there is some unmeasured seasonal condition, of either biotic or abiotic nature, which is explaining some of this genus' observed variability.

Even though seasonality explained a small proportion of total algal variability, a number of individual genera had appreciable amounts of their variability explained by these effects. Of these the abundance of *Chlorella* and *Anabaena* is highest in summer, in the early part of the calendar year, when inflows and temperatures are greatest. *Cyclotella* and *Cosmarium* are abundant in autumn (March/April) whilst *Spermatozopsis* is likely to be more abundant in spring (August to October), when dissolved oxygen concentrations in the water are at their highest.

In terms of numbers that may be expected to impact on water treatment, and in terms of the algae responding to seasonal changes, it is only *Chlorella* and *Anabaena* which are numerically and proportionally abundant. Of these two the latter is the more significant to the water treatment process due to its potential for producing taste and odour compounds. The analysis indicates that this genus may be expected to become sufficiently abundant and therefore cause treatment problems from January through to April. Elevated temperatures and inflows are closely associated with the summer bloom of *Anabaena* and *Chlorella*.

Another noteworthy result of the seasonal analysis is the shift in algal communities at certain times of the year associated with changes in season. The changes in algal populations in lakes is most apparent during the late summer period (February/March) through to midwinter (June/July) with another large shift occurring in early summer (September to November) through to late summer. The following genera exemplify these shifts: *Chlorella* and *Anabaena* (increase in abundance in late summer), *Cyclotella* and *Cosmarium* (increase in autumn - March/April) and *Spermatozopsis* (increase in spring). The shift in seasonal

pattern is likely to be associated with changes in the water column physical stability (establishment and breakdown of stratification within the water column). Within the study area stratification of the water column typically occurs in early summer with destratification in late summer/winter. These are periods of major physico-chemical change in the water column and which the literature reflects as having a significant impact on algal populations resident there (e.g. Wetzel, 2001).

7.1.8 Establishing statistically rigorous predictive models relating algae to the environment

Classical multiple regression modelling of important algae against environmental variables was attempted to gain a statistically rigorous model of how the abundance of the algae was affected by the environment. However, the predictive ability of these models was poor (R_a^2<0.5), even with appropriate transformations of the response variables (algal abundance data). There were indications that interactions among environmental variables, spatial locations (lakes) and algae themselves were important. In other words, the effects of environmental variables on the different algal genera, were not consistent over all lakes. The inconsistency was not surprising given the variation in physico-chemistry between lakes.

The statistically rigorous multiple regression modelling of key algae against environmental variables was therefore not successful in this study. The semi-quantitative empirical models developed in the ordination analyses had to suffice as far as better understanding algae/environment relationships within lakes studied.

7.1.9 Economic models relating water quality to treatment costs at waterworks

The analysis of water quality and water treatment costs was undertaken to identify the main factors (contaminants) affecting treatment costs at selected water works (WW) in the Umgeni Water (UW) operational area. The aim was to develop models to predict treatment costs from observed measures of water quality. Models of treatment costs were estimated using ordinary least squares (OLS) linear regression and principal component analysis (PCA) and are discussed in detail in Chapter 6 of this thesis.

Lake water quality clearly has a significant impact on the cost of treating water in each of the major WW examined in this study. In addition the study has shown that the cost of treating water per mega litre (Mℓ) is highest at the Hazelmere waterworks (WW) (R41) followed by

Durban Heights WW (R 28), DV Harris WW (R25) and Wiggins WW (R22). The latter three WW are all in a similar 'ballpark' with respect to treatment costs, whilst the costs in the Hazelmere system are almost double that of the others.

Important factors contributing to changes in treatment costs at each water plant are identified in this study. Physico-chemical water quality conditions generally affected water treatment more significantly than algae, except where potentially powerful taste and odour forming algae (e.g. *Anabaena*) were present, as for example in the Nagle/Durban Heights WW system.

Physico-chemical water quality conditions have a particularly significant impact on treatment costs at the Hazelmere WW. Treatment costs increase when turbidity, total aluminium, manganese, suspended solids, potassium, sulphates, and total organic carbon concentrations increase in Lake Hazelmere water. Likewise, costs rise with lower water pH and alkalinity levels. Algae have a relatively minor impact on treatment costs at Hazelmere WW. This is mainly because the high turbidity conditions in Lake Hazelmere allow for only limited algal growth. Lower turbidity corresponds with low inflows to the lake and this is generally also during winter when algal growth is limited.

The model derived for the Hazelmere WW explains 79% of the variation in chemical treatment costs. The model predicts actual costs quite well and can be easily applied in simulation exercises. The study shows that a management strategy that reduces the turbidity of Lake Hazelmere would reduce water treatment costs at the Hazelmere WW. Control of turbidity in Lake Hazelmere to levels below 10 NTU could, however, lead to possible algal growth problems. Nevertheless, changing the Hazelmere system from a turbidity driven system to an algal determined system could lower the costs of treating such water as costs of treating water in the less turbid systems like Durban Heights and DV Harris are lower. However, costs could increase if conditions created in Lake Hazelmere favoured the growth of taste and odour forming algae like *Anabaena.*

The specific Hazelmere model was explored and applied in practise after the completion of the research for this thesis. Umgeni Water were able to explore the impacts on treatment costs of an application for a proposed sand-winning operation in the uMdloti river (above the inflow to Lake Hazelmere). The sand-winning operation would obviously have affected in-lake water quality (specifically turbidity) and hence treatment costs.

The model estimated for the Durban Heights WW explains some 64% of the variation in chemical treatment costs. The model predicts actual costs well (except during occasional peak cost periods) and can be easily applied in simulation exercises. The study identifies some important water quality variables contributing to treatment costs at the Durban Heights WW. Results show that environmental and algal contaminants have a marked impact on treatment costs. Treatment costs increased when levels of turbidity, suspended solids, total organic carbon, conductivity, total water hardness, potassium, and nitrates rise in the raw water. Costs also rise with higher numbers of coliform bacteria in the water. Treatment costs rise with a fall in raw water pH and alkalinity (more acidic conditions, requiring greater lime dosages). An increase in *Anabaena* and *Microcystis* numbers in Lake Nagle have a major impact on treatment costs at the Durban Heights WW, particularly when they are producing taste and odour compounds. Costs also appear responsive to an increase in temperature and a fall in dissolved oxygen concentration. The results show that iron, manganese, total phosphorus, and *E.coli,* do not add significantly to treatment costs, other factors held constant. Initially the policy implications may be that catchment management resources should not be wasted on trying to limit the magnitude of these variables. However it is a well-acknowledged fact that phosphorus has the potential to have a marked effect on algal abundances (particularly the blue-green algae which traditionally cause the taste and odour problems) (e.g. Pillay, 1994). The results obtained from using the algae/environment models presented in Chapter 5 of this thesis support this view. It is possible that a critical phosphorus concentration has to be first exceeded before it becomes a significant factor in future water quality related problems.

The model developed for the DV Harris WW explains 67% of the variation in chemical treatment costs. The results show that treatment costs increase when levels of alkalinity, total hardness, manganese and conductivity in Lake Midmar decrease. Treatment costs also rise when concentrations of the alga, *Chlorella,* decline. Conversely, costs rise with higher concentrations of potassium and concentrations of coliforms and *E.coli.* Results show that turbidity, silicon, suspended solids and total organic carbon do not significantly affect treatment costs at the DV Harris WW. Paradoxically, relatively clean water, typical of Lake Midmar, is expensive to treat and reflects the need to add additional bentonite to the clean water to act as a nucleus for effective flocculation.

The significance of the trend variable[10] in the DV Harris model shows that over time, treatment cost have been increasing in this system. This might be associated with factors other than those included in the model having a significant effect on treatment costs, or alternatively, reduced treatment efficiency at the DV Harris WW.

The model fitted for the Wiggins WW explains 79% of the variation in chemical treatment costs. The results show that treatment costs at the Wiggins WW increase with an increase in water turbidity, total aluminium, iron, suspended solids, nitrates, total organic carbon, total dissolved solids, silicon, coliform numbers, conductivity, total water hardness, potassium, nitrates, *Microcystis* and *Anabaena*. These factors are directly related to water pollution of both natural (soil erosion, siltation) and anthropogenic (sewage and agricultural nutrient runoff) origin. Conversely, results show that treatment costs diminish with an increase in water pH and dissolved oxygen in the water. The significance of the trend variable shows that over time, treatment costs have been increasing in this system. This could be from either a deterioration in raw water quality coming into this WW, or from a reduction in efficiency in the operation of the plant and hence a need to dose more chemicals to achieve acceptable potable water standards.

The study quantifies how an integrated catchment management strategy, that would reduce both point and diffuse nutrient loads and turbidity in the lakes, could reduce water treatment costs at the respective WW.

7.2 Conclusions

7.2.1 Conclusions related to algae and their environment

The study has highlighted the use of a suite of statistical techniques that has assisted in reducing a complex multivariate problem (many different types of algae and many environmental variables) into something more comprehensible and useful. The complex community of algae in lakes were graphically summarised into associations of algae showing similar responses to important environmental variables.

[10] Trend variable – to determine progressive increasing or decreasing treatment costs at respective works. An increase in treatment costs over time would indicate deterioration in raw water quality entering WW (and *vice versa*).

Lake Shongweni appears to be an outlier at the end of an eutrophication gradient, both in terms of its physico-chemistry and algal composition - the result of which were problems in data analyses. It was therefore excluded from analyses.

The remaining seven lakes viz. Midmar, Albert Falls, Nagle, Inanda, Henley, Nungwane and Hazelmere formed the focus of the investigation.

Certain environmental variables investigated were highly correlated with each other which created problems in data analysis. Specific statistical techniques were implemented to counter this. The original large number of environmental variables (53) could be reasonably accounted for by a smaller number of 'key' environmental variables identified as conductivity, secchi depth, silicon, total inorganic nitrogen, total phosphorus, inflow, temperature, percentage dissolved oxygen and stability. These variables represented the primary aspects of variation in the environmental data.

A distinct group, of relatively low conductivity, low turbidity, 'inland' lakes (Midmar, Albert Falls, Nagle and Henley) were identified as distinct from more turbid 'coastal' lakes (Hazelmere and to a lesser extent Nungwane). Inanda was characterised by generally higher conductivities but low turbidities.

Inanda and Hazelmere are the most dissimilar lakes in the study (excluding Shongweni). Midmar, Albert Falls and Nagle are relatively similar when compared to the other lakes in the study and clearly demonstrate the change in water quality within lakes with progression down the uMngeni River catchment. Progression downstream in catchments was characterised by increased conductivities. Inanda, the last lake on the uMngeni system cascade, had the highest conductivities.

Weaker distinctions between lakes could be identified in relation to gradients characterised by total inorganic nitrogen, total phosphorus and inflows.

The reduced set of 'key' environmental variables explained some 16% of the variability in the algal data. Of these variables, total inorganic nitrogen, silicon, temperature, inflow and secchi were the most important explaining algal variability. The environmental effects were significantly related ($P<0.01$) to the variation in the algal data.

Anabaena and *Microcystis* (blue-green algae) were responding to the higher temperatures and inflow volumes that one would expect with late summer conditions. They are also more abundant at the lower end of the total inorganic nitrogen (low TN:TP ratio, < 20) gradient.

With lower amounts of their variability explained *Navicula, Spermatozopsis* and *Mallomonas* were responding to higher silicon (> 5mg/ℓ) and total inorganic nitrogen (> 0.5mg/ℓ) and lower temperatures (15-20°C) and inflows. Cool (15-20°C), clear waters (with high secchi depths i.e. >1.5m) with low total phosphorus (<20µg/ℓ) favour *Crucigena.* Waters low in total inorganic nitrogen (<0.5mgN/ℓ) and silicon (<5mg/ℓ) favours *Melosira, Scenedesmus, Cyclotella* and *Tetraedron.* These waters are also generally clear (secchi > 2.0m) and have lower water column stability.

Chlorella, a dominant or sub-dominant species in many lakes, does not have its variability well explained by measured environmental variables. Its variability is better explained by some (un-quantified) seasonal factor(s).

Hazelmere is distinctly different from all other lakes studied – the difference primarily associated with low water clarity (or high turbidity). Furthermore Hazelmere, Nungwane and Inanda (and to a lesser extent Nagle) are biologically most dissimilar from each other.

'Seasonal' effects account for a relatively small proportion of the observed algal variability compared to 'key' environmental and 'lake unique' variables. The effect of season is still clear, however, and accounts for a statistically significant portion of the algal variability.

The annual trends are less significant than monthly patterns and show no strong overall successional trend in the algal populations in the lakes under study.

January through to April are the most significant months where *Anabaena* may be expected to become sufficiently abundant and therefore cause treatment problems. Conditions associated with elevated temperatures and inflows are closely associated with summer highs of *Anabaena* and *Chlorella.*

Algal populations (and hence diversity) shift during the late summer period (February/March) through to midwinter (June/July) with another shift occurring in early summer (September to November) through to late summer. The following genera most exemplify these shifts:

Chlorella and *Anabaena* (increase in abundance in late summer), *Cyclotella* and *Cosmarium* (increase in autumn - March/April) and *Spermatozopsis* (increase in spring). The seasonal pattern is most probably associated with changes in the water column physical stability (establishment and breakdown of stratification within the water column).

Classical multiple regression modelling of important algae (dominant or statistically significantly affecting water treatment processes) against environmental variables was unsuccessful with the predictive ability of all multiple regression models poor ($R_a^2 < 0.5$).

Of the approaches taken, the semi-quantitative empirical models developed in ordination analyses were the best available predictive models for understanding algae/environment relationships in lakes studied.

In terms of the research needs surrounding eutrophication of South African surface waters identified in Chapter 1 (by amongst others Noble & Hemens, 1978, Grobler & Silberbauer, 1984, and Walmsley, 2000), the extent and trend of the problem in the Umgeni Water operational area has been partially answered. Some of the sources of nutrients, and their levels entering aquatic systems, have also been quantified. The following summary addresses some of the actual social and economic costs of the problem, at least at a provincial level.

7.2.2 Conclusions related to the cost of water treatment

Lake water quality clearly has a significant impact on the cost of treating water in each of the major WW examined.

The cost of treating water per mega litre (Mℓ) is highest at the Hazelmere WW (R41) followed by the Durban Heights (R 28), DV Harris (R25) and Wiggins WWs (R22). The cost in treating water in the Hazelmere system are almost double the others and appears to be related to the higher turbidity water experienced by Lake Hazelmere.

Within the range of the data analysed, physico-chemical water quality factors generally affected water treatment more significantly than algae, except where potentially powerful taste and odour forming algae (principally *Anabaena*) were present. The Nagle/Durban Heights WW system is most affected by taste and odour problems.

The model derived for the Hazelmere WW explains 79% of the variation in chemical treatment costs. The model predicts actual costs quite well and can be easily applied in simulation exercises.

Physico-chemical water quality factors have a particularly significant impact on treatment costs at the Hazelmere WW. Treatment costs increase when turbidity, total aluminium, manganese, suspended solids, potassium, sulphates, and total organic carbon concentrations in Lake Hazelmere water increase. Likewise, costs rise with lower water pH and alkalinity levels. Algae have a relatively minor impact on treatment costs at Hazelmere WW.

Hazelmere is the only system analysed that appeared to suffer from problems associated with manganese (necessitating the use of a powerful oxidant such as chlorine dioxide).

A management strategy that reduces the turbidity of Lake Hazelmere would reduce water treatment costs at the Hazelmere WW. During periods of lake turnover (when the stratification of the water column breaks down) manganese (in the reduced form) should be carefully monitored to reduce its potential impact on water treatment (and hence costs).

The model estimated for the Durban Heights WW explains some 64% of the variation in chemical treatment costs. The model predicts actual costs well (except during occasional peak cost periods) and can be easily applied in simulation exercises. Treatment costs increased when levels of turbidity, suspended solids, total organic carbon, conductivity, total water hardness, potassium, nitrates and coliform bacteria rise in the raw water. Treatment costs rise with a fall in raw water pH and alkalinity (more acidic conditions, requiring greater lime dosages).

An increase in *Anabaena* and *Microcystis* numbers in Lake Nagle have a major impact on treatment costs at the Durban Heights WW, particularly when they are producing taste and odour compounds. The results show that iron, manganese, total phosphorus, and *E. coli* do not add significantly to treatment costs, other factors held constant. Initially the policy implications may be that resources should not be wasted on these apparent problems. However, it is a known fact that phosphorus has the potential to have a marked effect on algal abundances (particularly the blue-green algae which traditionally cause the taste and odour problems). The results of algae/environment models presented in Chapter 5 of this thesis support this view.

The model developed for the DV Harris WW explains 67% of the variation in chemical treatment costs. The results show that treatment costs increase when levels of alkalinity, total hardness, manganese and conductivity in Lake Midmar decrease. Treatment costs also rise with declining numbers of *Chlorella*. Conversely, costs rise with higher concentrations of potassium and numbers of coliforms and *E.coli*. Results show that turbidity, silicon, suspended solids and total organic carbon do not significantly affect treatment costs at the DV Harris WW. This indicates that raw water in this system is traditionally clean with costs principally driven by the need to disinfect and stabilise (lime) the water. There are indications that over time treatment costs have been increasing in the DV Harris WW system.

The model fitted for the Wiggins WW explains 79% of the variation in chemical treatment costs. The results show that treatment costs at the Wiggins WW increase with an increase in water turbidity, total aluminium, iron, suspended solids, nitrates, total organic carbon, total dissolved solids, silicon, coliform numbers, conductivity, total water hardness, potassium, nitrates, *Microcystis* and *Anabaena*. Conversely, treatment costs diminish with an increase in water pH and dissolved oxygen in the water. The significance of the "trend variable" shows that over time treatment cost have been increasing in this system.

The study quantifies how an integrated catchment management strategy that would reduce both point and diffuse nutrient loads and turbidity in the lakes, could reduce water treatment costs at the respective WW.

7.3 Recommendations

The following recommendations for future research and management strategies were highlighted by this work.

7.3.1 Recommendations with respect to algae and their environment

Lakes should be grouped into similar entities based on their water quality. These groups are likely to respond more similarly to changes in water quality because of policy or management

changes than are lakes in different groupings. Changes in water quality would also affect algae resident in lakes.

Water temperature and inflow to lakes (and factors associated with these environmental variables) appear to be the more important variables controlling the occurrence and abundance of the important blue-green algae, *Anabaena* and *Microcystis*. These variables are effectively beyond management "control". However, they also appeared to favour low total inorganic nitrogen concentrations (and low TN:TP ratios, i.e. <20). Therefore, wherever possible, management should aim to keep the TN:TP ratio above 20 to avoid incidences of these problematic algae.

The database should be enlarged and then re-analysed to determine whether the trends and associations identified in analyses still hold, or whether systems are changing and responding to different patterns in water quality and climate. The different drought cycles (and their impact on water quality) should also be considered in this analysis. For example, at the end of the analysis phase of this study the regional climate system appeared to be entering a wet phase, with overall lake capacities at an all-time high (1996 –1997), with lakes approaching drought condition levels during the write up of this thesis in 2003. The statistical tools to perform these analyses are now sufficiently well understood (and documented here) to perform this at any future date.

Efforts need to be applied to determining what other environmental parameters are important which may assist in better explaining algae/environment relationships i.e. attempt to improve the amount of algal variability explainable by environmental parameters. These parameters may be of either an abiotic (purely physico-chemical) or biotic (e.g. predation, competitive exclusion etc.) nature. The degree of interaction between algae (i.e. successional dynamics, and to what extent this is accounting for observed algal variability) also needs further investigation.

As algae were only identified to genus level there is little doubt that species level identifications would lead to a much higher correspondence and co-relation with environmental variables. This is because of the narrower environmental tolerances respective species have compared to an entire genus. In the future, where algae are identified to species level, more robust models may be possible and likely to be more useful. For example, it is likely that only certain species of typically problematic algal genera (e.g. *Anabaena*) are responsible for taste and odour formation, and hence likely to affect water

treatment processes. The non taste and odour forming species within the genus are therefore adding to the "noise" in the data and clouding these types of important relationships.

Further work is required on elucidating features of the environment, which were unique, or sufficiently different between lakes, to account for the algal heterogeneity observed between different lakes.

A reduced suite of variables, that explains the main axes of variation in the environmental data set almost as well as the entire data set, was identified. From a water quality monitoring perspective this may allow a reduction in monitoring intensity and hence costs for sample analyses. With a known correlation structure in the environmental data set it appears that monitoring a reduced number of 'key' environmental variables would allow a reasonable picture to be determined of how other unmonitored variables may be co-varying.

The algae/environment models developed should be tested against 'new' data to determine their predictive ability.

The models should also be tested in other systems to determine their applicability in those systems.

7.3.2 Recommendations with respect to water treatment

Treatment cost models were developed based on only the chemicals used for coagulation, stabilisation and disinfection. Obviously "dirtier" water would require more frequent backwashing of filters and as such would incur increased WW treatment plant operational costs. Unfortuntately for the period of this study, records of electricity used or backwash times, for the relevant treatment periods, were generally unmonitored, inaccessible, or unable to be isolated and differentiated from the general electricity usage. Therefore to improve the application and accuracy of the models developed, attempts to build in these factors should be made. This improvement would obviously be dependent on the availability of these records.

Models were developed describing the impact of in-lake processes on water treatment costs for most of the major WW in the Umgeni Water operational area. To elucidate the link between catchment management activities and water treatment costs, investigations are

required to determine the relationships between catchment and in-lake processes. This should, as far as possible, be determined quantitatively. For example: how are catchment activities (e.g. farming practices, sand winning/mining, etc.) influencing in-stream turbidity and hence in-lake turbidity. Changes to in-lake turbidity now have a known impact on treatment costs. With these relationships better understood, there is a greater possibility of determining the "costs" (from a water treatment point-of-view) of catchment management (or lack thereof). This will allow valid 'cost/benefit' analyses to be made in terms of alternative catchment management strategies.

The specific Hazelmere model was explored and applied in practise after the completion of the research for this thesis. Umgeni Water were able to explore the potential impacts on treatment costs of a proposed sand-winning operation in the uMdloti river (above the inflow to Lake Hazelmere). The sand-winning operation would obviously have affected in-lake water quality (specifically turbidity) and hence treatment costs.

Other examples of possible application of these treatment cost models would be:

- in determining the impact of inter-basin transfers (of water of widely differing quality),
- raising dam-walls,
- etc.,

on treatment costs. This would be on the assumption that these activities had a known and quantified impact on in-lake processes.

The sensitivity of the treatment cost models, with and without the trend variable (where it was shown to be a significant factor affecting cost – the DV Harris and Wiggins WW) needs to be tested. The future application of these models would be more useful without having to insert an estimate for the 'trend variable' into the equation for the prediction of costs.

An economic analysis of water quality variables over the combined data set from all waterworks systems would possibly be more generally useful to other water authorities. This would result in a more generalised model of how water quality factors are impacting on water treatment costs. Within this type of analysis, sub-models to cater for the various aspects of the treatment process should be formulated, i.e. coagulation, stabilisation (liming) and disinfection (chlorination) sub-models.

Economic models developed should be tested for their predictive ability against raw water quality and WW systems in other parts of the country.

Future research should look into integrating water treatment cost and water quality data from around South Africa, using the methods developed here, to develop more universally applicable treatment cost models.

On a more general note, the accessibility of water treatment dosage (and hence cost) information was in some instances difficult, and once located, rather patchy. This system should be revised and standardised across WW to assist in updating treatment cost models as well as making valid comparisons across WW. A central, electronic database of this information would assist in this process. Recording of electricity usage, to reflect filter backwash times, would also enhance the validity and accuracy of future models which may be developed.

REFERENCES

ACRU 1995. Water Research Commission Report, TT69/95. ACRU Report 43.

Agbeti MD. 1992. Relationship between diatom assemblages and trophic variables: a comparison of old and new approaches. *Can. J. Fish. Aquat. Sci.* 49: 1171-1175.

An KG & Jones JR. 2000. Factors regulating bluegreen dominance in a reservoir directly influenced by the Asian monsoon. *Hydrobiologia*, 432: 34-48.

Archibald CGM, Warwick RJ, Fowles BK, Muller MS & Butler AC. 1980. Hazelmere Dam. Institute for Water Research, Council for Scientific and Industrial Research. Durban.

Australian State of the Environment Advisory Council (ASEC). 1996. http://www.environment.gov.au/

Battarbee RW, Mason J, Renberg I & Talling JF. 1990. Eds. Paleolimnology and lake acidification. *Phil. Trans. R. Soc. Lond.* B 223-445.

Bahnwart M, Hubener T & Schubert H. 1999. Downstream changes in phytoplankton composition and biomass in a lowland river-lake system (Warnow River, Germany). *Hydrobiologia* 391: 99-111.

Bernhardt H. 1992. Introduction and opening remarks, in *Eutrophication: Research and Application to Water Supply*. Freshwater Biologcial Association.

Borcard D, Legendre P, & Drapeau P. 1992. Partialling out the spatial component of ecological variation. *Ecology* 73(3): 1045-1055

Bormans M & Webster IT. 1999. Modelling the spatial and temporal variabilitity of diatoms in the River Murray. *Journal of Planton Research* 21: 581-598.

Breen CM. 1983. Limnology of Lake Midmar. *South African National Scientific Programmes Report* No. 78

Chatterjee S & Price B. 1977. Regression Analysis by Example. John Wiley & Sons. New York.

Dawes CJ, Cowell BC, Gardiner WE & Scheda SM. 1987. Limnological characteristics of two eutrophic and four mesotrophic lakes in west-central Florida. *Int. Revue ges. Hydrobiol.* 72: 171-203.

Dennison DB. 1996. An economic evaluation of water treatment costs in Umgeni catchment area. Unpublished Master of Agricultural Management thesis, Department of Agricultural Economics, University of Natal, South Africa.

Descy JP. 1987. Phytoplankton composition and dynamics in the River Meuse (Belgium). *Arch. Hydrobiol. Suppl.* 78: 225-245.

Department of Water Affairs. 1986. Management of water resources of the republic of South Africa. CTP Book Printers, Capte Town.

Department of Water Affairs and Forestry and Umgeni Water. 1996. Mgeni Catchment Management Plan. Report No. WQM U200/00/0196

Dicks H. 1997. University of Natal, Pietermaritzburg, South Africa, Statistics Dept.

Dickens CWS, Graham PM & Freese S. 1996. Algal rupture during abstraction from reservoirs and the consequences for water treatment.
Water Research Commission Report No. 558/1/96

Dickens CWS & Graham PM. 1998. Biomonitoring for effective management of wastewater discharges and the health of the river environment. *Aquatic Ecosystem Health and Management* 1: 199-217.

Dixit SS, Smol JP, Kingston JC & Charles DF. 1992. Diatoms: powerful indicators of environmental change. *Environ. Sci. Technol.*, 26: 23-33.

Dunst RC, Born SM, Uttormark PD, Smith SA, Nichols SA, Peterson JO, Knauer DR, Serns SL, Winter DR & Wirth TL. 1974. Survey of lake rehabilitation techniques and experiences. Technical Bulletin No 75. Department of Natural Resources, Madison, Wisconsin.

European Environment Agency (EEA) 1999. Nutrients in European ecosystems. Environmental assessment report No 4. European Environment Agency, Copenhagen.

Freese SD. 2003. Scientist, Process Services, Umgeni Water, PO Box 9, Pietermaritzburg, 3200, South Africa.

Furness HD. 1974. Eutrophication studies in the catchment area of Midmar Dam. MSc Thesis, University of Natal, Pietermaritzburg.

Gauch HG. 1982. *Multivariate analysis in community ecology.* Cambridge Studies in Ecology.

Gauch HG, Whittaker RH & Wentworth TR. 1977. A comparative study of reciprocal averaging and other ordination techniques. *Journal of Ecology* 65: 157-174.

Graham PM, Dickens CWS. & Mbowa S. 1998. Modelling the Water Quality in Impoundments within the Umgeni Water Operational Area and the Consequences for Potable Water Treatment Costs. Water Research Commission Report No. 615/1/98.

Grobler DC & Silbauer MJ. 1984. Impact of eutrophication control measures on the trophic status of South African impoundments. Water Research Commision Report.

Gujarati DN. 1988. Basic Econometrics (second edition). McGraw-Hill, New York.

Gujarati DN. 1992. Essentials of Econometrics, Economic series (International edition). McGraw-Hill, Singapore.

Hammer VT. 1964. The succession of 'bloom' species of blue-green algae and some causal factors. *Verh. int. Ver. Limnol.* 15: 829-836.

Harding WR & Paxton BR. 2001. Cyanobacteria in South Africa: A review. Water Research Commission Report No: TT 153/01.

Hart RC. 1992. Aspects of comparative plankton ecology in cascading Mgeni River reservoirs (Midmar, Albert Falls and Nagle): an overview. *Southern African Journal Aquatic Sciences* 18: 20-41.

Havens KE, Phlips EJ, Cichra MF & Li B. 1998. Light availability as a possible regulator of cyanobacteria species composition in a shallow subtropical lake. *Freshwater Biol.* 39: 547-562.

Hemens J, Simpson DE & Warwick RJ. 1977. Nitrogen and phosphorus input to the Midmar Dam, Natal. *Water SA.* 3(4): 193-201.

Hodgson KG, Pillay M & Ramjatan A. 1996. Rapid transfer of water quality information to ensure the maintenance of potable water standards.
Poster presentation: Water Institute of Southern Africa - biennial conference.

Huntley B, Siegfried R & Sunter C. 1989. South African Environments into the 21st Century. Human, Rousseau & Tafelberg, Cape Town.

Jensen JP, Jeppesen E, Olrik K & Kristensen P. 1994. Impact of nutrients and physical factors on the shift from cyanobacterial to chlorophyte dominance in shallow Danish lakes. *Can. J. Fish. Aquat. Sci.* 51:1692-1699.

Jongman RHG, ter Braak CJF, van Tongeren OFR. 1987. Eds. Data Analysis in Community and Landscape Ecology; Pudoc: Wageningen, The Netherlands.

Kimmel BL, Lind OT & Paulson LJ. 1990. Reservoir primary productivity.
Pages 133-193 in "Reservoir Limnology". Eds. Thornton KW, Kimmel BL & Payne FE.
Wiley and Sons.

Kohler J. & Hoeg S. 2000. Phytoplankton selection in a river-lake system during two decades of changing nutrient supply. *Hydrobiologia*, 424: 13-24.

Lampert W, Fleckner W, Rai H & Taylor BE. 1986. Phytoplankton control by grazing zooplankton; a study on the spring clear water phase.
Limnology and Oceanography 31: 478-490.

Likens GE. 1988. Ed. Long-Term Studies in Ecology; Springer-Verlag: New York.

Lorch W. 1987. Handbook of Water Purification. Second Edition.
Halsted Press (Division of John Wiley)

Lund MA & Davis JA. 2000. Seasonal dynamics of plankton communities and water chemistry in a eutrophic wetland (Lake Monger, Western Australia): Implications for biomanipulation. *Mar. Freshwater Res.* 51: 321-332.

Lynam TJP. 1997. An analysis of some observational and philosophical problems of modelling human needs satisfaction systems in the communal areas of the Zambezi Valley of Zimbabwe.
In: Hatch G.P. & Morley T. Third Developing Areas Symposium Proceedings: Multiple Land-Use Options for Arid and Semi-Arid Rangelands.
Grassl. Soc. Sth. Afr.

Manly BJ. 1991. Multivariate Statistical Methods - A primer. Chapman & Hall.

May RM. 1984. An overview: real and apparent patterns in community structure. Pages 3-16, in DR Strong, D Simberloff, LG Abele & AB Thistle, eds Ecological communities: conceptual issues and the evidence. Princeton University Press, Princeton, New Jersey, USA.

Mbowa S. 2003. Dept of Agricultural Economics, Makerere University, P.O. Box 7062, Kampala, Uganda.

Moodley J & Hodgson KG. 2002: Implementation of an Advanced Sample Scheduling System at Umgeni Water: LV-TAPS. Poster presentation at Water Institute of Southern Africa (WISA) 2002 conference, Durban, South Africa.

Noble RG. & Hemens J. 1978. Inland water ecosystems in South Africa - a review of research needs. South African National Scientific Programmes Report No. 34: 150 pp.

OECD. 1982. *Eutrophication of Waters - Monitoring, Assessment and Control.* Organisation for Economic Co-operation and Development. Paris.

Palmer CM. 1959. Algae in water supplies. US Department of Health, Education and Welfare. Bureau of State Services, Division of Water Pollution Control. Cincinnati, Ohio.

Palmer CM. 1980. Algae and Water Pollution. Castle House Publications Ltd.

Patterson JC, Hamblin PF & Imberger J. 1984. Classification and dynamic simulation of the vertical density structure of lakes. *Limnology and Oceanography* 29: 845-861.

Pillay M. 1994. Detergent phosphorus in South Africa: impact on eutrophication with specific reference to the Umgeni catchment. Master of Science in Engineering Thesis, University of Natal, South Africa.

Quinn JF & Dunham AE. 1983. On hypothesis testing in ecology and evolution. *American Naturalist* 122: 602-617.

Rast W & Thornton JA. 1996. Trends in eutrophication research and control. *Hydrobiological Processes* 10: 295-313.

Reyment R. 1980. Multivariate analysis in statistical paleoecology. pp211–235. In Multivariate Methods in Ecological Work. Eds. L. Orloci, C. R. Rao and W. M. Stiteler: Vol. 7 in Statistical Ecology. International Co–operative Publishing House, Maryland, U. S. A.

Reynolds CS & Walsby AE. 1975. Water Blooms. *Biol. Rev.* 50: 437-481.

Reynolds CS. 1984. Phytoplankton periodicity: the interactions of form, function and environmental variability. *Freshwater Biology* 14: 111-142.

Romo S & van Tongeren O. 1995. Multivariate analysis of phytoplankton and related environmental factors in a shallow hypertrophic lake. *Hydrobiologia* 299: 93-101.

Round FE. 1973. The Biology of the Algae. Arnold UK.

Sanders R.W. & Porter KG. 1990. Bacterivorous flagellates as food resources for the freshwater crustacean zooplankter *Daphnia ambigua*. *Limnol. Oceanogr.* 35: 188-191.

Schindler DW. 1977. Evolution of phosphorus limitation in lakes. *Science* 195: 260-262.

Scotney DM. 1970. Soils and Landuse in the Howick extension area. PhD Thesis. University of Natal, Pietermaritzburg, South Africa. Unpublished.

Shapiro J. 1997. The role of carbon dioxide in the initiation and maintenance of blue-green dominance in lakes. *Freshwater Biology* 37: 7-323

Smith VH. 1983. Low nitrogen to phosphorus ratios favour dominance by blue-green algae in lake phytoplankton. *Science* 221: 669-670.

Smith VH. 1986. Light and nutrient effects on the relative biomass of blue-green algae in lake phytoplankton. *Can. J. Fish aquat. Sci.* 43: 148-153.

Sommer U. (ed). 1989. Plankton ecology. Succession in plankton communities. Springer Verlag, Berlin.

Soudant D, Beliaeff B & Thomas G. 1997. Dynamic linear Bayesian models in phytoplankton ecology. *Ecological Modelling* 99:161-169.

South African Bureau of Standards. 2001. SABS 241:2001, South African Standard. Specification – Drinking Water.

Southwood TRE. 1987. The concept and nature of the community. In JHR Gee & PS Giller, Eds. Organization of communities past and present. Blackwell Scientific.

Stevenson AC, Juggins S, Birks HJB, Anderson DS, Anderson NJ, Battarbee RW, Berge F, Davis RB, Flower RJ, Haworth EY, Jones VJ, Kingston JC, Kreiser AM, Line JM, Munro MAR & Renberg I. 1991. The surface water acidification project paleolimonlogy programme: modern diatom/lake-water chemistry data set. ENSIS Publishing, London, England.

Tebbutt THY. 1992. Principles of Water Quality Control. Fourth Edition. Pergamon Press, Oxford, New York Seoul, Tokyo.

Teixera MGLC, Costa MCN, Carvalho VLP, Pereira MS & Hage E. 1993. Gastroenteritis epidemic in the area of the Itaparica Dam, Bahia, Brazil. *Bull. of Pan. Am. Hlth. Org.* 27: 244–253.

ter Braak CJF. 1987-1992. CANOCO - a FORTRAN Program for Canonical Community Ordination. Microcomputer Power, Ithaca, New York, USA.

ter Braak CJF. 1990. Update notes: CANOCO version 3.10. Agricultural Mathematics Group, Wageningen, 35 pp

Thompson P. 2003. Scientist, Process Services, Scientific Services, Umgeni Water. PO Box 9, Pietermaritzburg, 3200, South Africa.

Tilman D, Kilham SS & Kilham P. 1982. Phytoplankton community ecology: the role of limiting nutrients. *Annu. Rev. Ecol. Syst.* 13: 349-382.

Tilman D, Kiesling R, Sterner R, Kilham SS & Johnson FA. 1986. Green, blue-green, and diatom algae: taxonomic differences in competitive ability for phosphorus, silicon, and nitrogen. *Arch. Hydrobiol.* 106: 473-485.

Tisdale E. 1931. Epidemic of intestinal disorders in Charlestown, Wva, occurring simultaneously with unprecented water supply conditions. *Am. J. Public Health*, 21, 198-200.

Umgeni Water. 1991. Internal Report November 1991.
Eutrophication status of Inanda Dam, Natal, South Africa, during the first two and a half years of impoundment. Umgeni Water, Water Quality Department, Scientific Services

Umgeni Water 1994 Internal Report WQP 1.3/94.
Proposal to collect water quality and flow data using automatic flow recording and sampling equipment at selected phase 1 weirs. Umgeni Water, Water Quality Department, Scientific Services.

Umgeni Water. 1996/97. Technical and Scientific Statistics, Supplement to Umgeni Water's 24th Annual Report, 1996/97

Umgeni Water. 2000/2001. Technical and Scientific Statistics, Supplement to Umgeni Water's 26th Annual Report, 2000/2001.

Umgeni Water. 2002. Technical and Scientific Statistics, Supplement to Umgeni Water's 27th Annual Report, 2001/2002.

United Nations Environment Programme (UNEP). 1999. Planning and Management of Lakes and reservoirs: An Integrated Approach to Eutrophication [Technical Publication Series 11] 375pp.

van der Zel DW. 1975. Umgeni River catchment analysis. *Water SA* 1(2): 70-75

van Halderen A, Harding WR, Wessels JC, Schneider DJ, Heine EWP, van der Merwe J and Fourie JM. 1995. Cyanobacterial (blue-green algae) poisoning of livestock in the western Cape province of South Africa. *J. S. Afr. Vet. Assoc.* 66, 260-264.

van Tongeren OFR, van Liere L, Gulati RD, Postema G & Boeswinkel-de Bruyn PJ. 1992. Multivariate analysis of the plankton communities in the Loosdrecht lakes: relationship with chemical and physical environment. *Hydrobiologia* 233: 105-117

Varis O. 1993. Cyanobacterial dynamics in a restored Finnish lake: a long term simulation study. *Hydrobiologia* 268: 129-145.

Varis O, Sirvio H & Kettunen J. 1989. Multivariate analysis of lake phytoplankton and environmental factors. *Arch. Hydrobiol.* 117: 163-175.

Viner AB. 1985. Thermal stability and phytoplankton distribution. *Hydrobiologia* 125: 47-69.

Vollenweider RA. 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. OECD, Paris. Tech. Rpt. DA 5/SCI/68.27. 250p

Walker WW Jr. 1987. Empirical methods for predicting eutrophication in impoundments. Report 4 Phase III: Applications Manual.
Department of the Army, US Army Corps of Engineers, Washington DC.

Walmsley RD. 1977. Limnological studies on Midmar Dam. M.Sc. Thesis Botany Department, University of Natal, Pietermaritzburg, South Africa.

Walmsley RD. 2000. Perspectives on Eutrophication of Surface Waters: Policy/Research Needs in South Africa. Water Research Commission. Report No. KV129/00

Walmsley RD & Butty M. 1980. Limnology of some selected South African impoundments. Report by Water Research Commission and National Institute for Water Research Council for Scientific and Industrial Research

Water Services Act. 1997. Republic of South Africa, Water Services Act, Act 108 of 1997.

Wetzel RG. 2001. Limnology: Lake and River Ecosystems (3rd Edition). Academic Press.

Wilson AJ. 1992. *Quo vadis* Umlaas System?
Umgeni Water Corporate Services Division, Technical Report 21 April 1992.

WHO (World Health Organisation). 1999. *Toxic Cyanobacteria in Water.* Ed. Chorus I. & Bartram J.

Zohary T. & Robarts RD. 1989. Diurnal mixed layers and the long-term dominance of *Microcystis aeruginosa*. *J. Plankton Res.* 11: 25-48.

APPENDIX 1

Table A1.1 Complete alphabetical list of algal genera routinely monitored by Umgeni Water in lakes within its operational area

Genus	Genus
Achnanthes	*Lepocinclis*
Actinastrum	*Mallomonas*
Amphipleura	*Melosira*
Amphiprora	*Meridion*
Anabaena	*Merismopedia*
Ankistrodesmus	*Micractinium*
Ankyra	*Micrasterias*
Asterionella	*Microcystis*
Biddulphia	*Microspora*
Botryococcus	*Navicula*
Ceratium	*Nitzschia*
Chaetoceros	*Oedogonium*
Chlamydomonas	*Oocystis*
Chlorella	*Oscillatoria*
Chlorogonium	*Pandorina*
Chodatella	*Pediastrum*
Chroococcus	*Peridinium*
Cladophora	*Phacotus*
Closterium	*Phacus*
Cocconeis	*Pleurococcus*
Coelastrum	*Pteromonas*
Cosmarium	*Scenedesmus*
Crucigenia	*Schroederia*
Cryptomonas	*Selenastrum*
Cyclotella	*Siderocelis*
Cymatopluera	*Spermatozopsis*
Cymbella	*Sphaerocystis*
Diatoma	*Spirogira*
Dictyosphaerium	*Staurastrum*
Didymogenes	*Stephanodiscus*
Dinobryon	*Stichococcus*
Elakatothrix	*Stigeoclonium*
Euastrum	*Surirella*
Eudorina	*Synedra*
Euglena	*Synura*
Fragilaria	*Tabellaria*
Gloeocapsa	*Tetraedron*
Golenkinia	*Tetrastrum*
Gomphonema	*Thalassiosira*
Gomphosphaeria	*Trachelomonas*
Gonium	*Ulothrix*
Gyrosigma	*Volvox*
Haematococcus	*Westella*
Hydrodictyon	
Kirchneriella	