

**THE IMPACT OF ALTERED RIVER  
STRUCTURE ON THE FUNCTION OF  
SELECTED URBAN CAPE TOWN RIVERS**

**NATALIE NICOLETTE NEWMAN**

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**THE IMPACT OF ALTERED RIVER STRUCTURE ON THE FUNCTION OF  
SELECTED URBAN CAPE TOWN RIVERS**

by

**NATALIE NICOLETTE NEWMAN**

**Thesis submitted in fulfilment of the requirements for the degree**

**Master of Technology: Nature Conservation**

**in the Faculty of Applied Sciences**

**at the Cape Peninsula University of Technology**

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**Co-supervisor:** Mr. Victor Harley

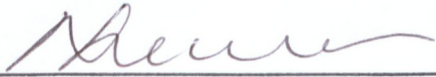
**External supervisor:** Dr. Elizabeth Day

**Cape Town Campus**

December 2010

## DECLARATION

I, Natalie Nicolette Newman, declare that the contents of this thesis represent my own unaided work, and that the thesis has not previously been submitted for academic examination towards any qualification. Furthermore, it represents my own opinions and not necessarily those of the Cape Peninsula University of Technology.



Signed

2012.04.12

Date

## ABSTRACT

Many urban rivers are heavily engineered and no longer function naturally. The City of Cape Town has designed and implemented many stormwater and river management projects. Very little monitoring has occurred as to whether these engineering projects and remediation measures, have had a positive effect on our urban river ecosystem function.

The study investigated the influence of specific engineering interventions such as the placement of rocks in stream to create weirs, gabion lining of stream channels, removal of canal walls, establishment of artificial wetlands, and approaches to urban river management, on river ecosystem function of the Keyzers River, Little Lotus River, Langevlei Canal, Silvermine River, Moddergat River and Big Lotus River, as measured by specific indices including water chemistry and aquatic community structure (macroinvertebrates and diatoms).

Chemical analysis of water samples, analysis of habitat quality, and macroinvertebrate and diatom community studies were undertaken in all six rivers, seasonally, for a period of 1 year. Water chemistry parameters included total suspended solids (mg/l), pH, conductivity (ms/m), chemical oxygen demand (mg/l), total oxidisable nitrogen (mg/l N), total ammonia (mg/L N), soluble nitrite and nitrate (mg/l N), total inorganic nitrogen (mg/l N), total phosphorus (mg/l P), and orthophosphates (mg/l P). These parameters were consistent with the parameters tested historically by the City of Cape Town. The diversity and abundance of macroinvertebrates and diatoms were assessed as well as the habitat quality of the sites sampled.

The results showed that the nutrient enrichment of each river depended on the seasons. Most rivers varied between mesotrophic and eutrophic (Moddergat River, Langevlei Canal), oligotrophic and mesotrophic (Keyzers River, Silvermine River) or eutrophic and hypertrophic (Big Lotus River). The assemblages of diatoms sampled confirmed the water chemistry for each river except for Keyzers River, Moddergat River and site SM3 in the Silvermine River, which was poorer than that expressed by the water chemistry results.

The effects of nutrient enrichment and pollution were visible in the diversity and abundance of macroinvertebrate taxa sampled. These taxa were also mostly pollution tolerant taxa, as classified by Gerber and Gabriel (2002). The survival of these pollution tolerant taxa is dependent on habitat. The Silvermine River was the only river that had a heterogeneous habitat for the colonisation of macroinvertebrates. The habitat of all other rivers was sparse and homogenous. This did have an effect on the macroinvertebrate results and a higher abundance of taxa was often sampled due to the limited availability of habitat.

It was concluded that the dominance of pollution tolerant taxa at all sites, in all rivers, was an indication that water quality is an overriding determinant for macroinvertebrates. Habitat was considered a primary factor in determining the ability of the river to function. Without habitat the survival of the taxa present would have been limited. Therefore while these rivers were subject to very poor water quality, and were severely impacted, they did function ecologically. No one intervention proved superior to the next, and each intervention provided more diversity of habitat, than had it not existed.

The impacts of engineering interventions on our rivers could be further investigated if there was an opportunity to undertake a similar study, prior to, during implementation and after implementation, of such an intervention. This would give a more accurate assessment of the intervention on the urban river, and whether this has in fact improved the ecological function. If, however, the water quality is not addressed at the source, and more habitats provided in our rivers, the recovery of macroinvertebrates could be overcome by pollution.

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## GLOSSARY OF TERMS

Abundance	Is the total number of individual organisms (Dallas and Day, 1993).
Acute effect value	Is the concentration at and above which statistically significant acute adverse effects are expected to occur (DWAF, 1996).
Chronic effect value	Is the concentration limit which is safe for all or most populations even during continuous exposure (DWAF, 1996).
Chemical oxygen demand (COD)	Is the amount of oxygen required to oxidise all organic matter that is susceptible to oxidation by a strong chemical oxidant. (DWAF, 1996).
Diversity	Is the total number of different taxa (Dallas and Day, 1993).
Eutrophic	Is a condition characterised by excess nutrients that stimulate excessive plant and algal growth (DWAF, 1996).
Mesotrophic	Refers to freshwater bodies that contain moderate quantities of plant nutrients and are moderately productive (DWAF, 1996).
Oligotrophic	Refers to freshwater systems with a low productivity, low nutrients, and no nuisance growth of aquatic plants or algae (DWAF, 1996).
Hypertrophic	Is a condition characterised by excessive nutrients that stimulate the growth of aquatic plants and algae blooms, and often include species of algae that are toxic to man, livestock and wildlife (DWAF, 1996).
Nuisance Aquatic Plants	Indigenous or invasive plants which are found in nutrient enriched freshwater systems, which can become detrimental when they cover large areas (DWAF, 1996).
Non-point source pollution	Diffuse discharges of pollutants from surface run off, infiltration or atmospheric sources (DWAF, 1996).
Point source pollution	Discharges of pollutants from discrete sources, e.g. an effluent discharge from an industry. The volume and quantity of such a discharge can normally be measured (DWAF, 1996).

## GLOSSARY OF ABBREVIATIONS

AEV	acute effect value
ANOSIM	analysis of similarities
ANOVA	analysis of variance
ASPT	average score per taxon
CEV	chronic effect value
COCT	City of Cape Town
COD	chemical oxygen demand
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
FIA	flow injection analysis
IHAS	integrated habitat assessment system
NH <sub>3</sub>	Ammonia
NH <sub>4</sub>	Ammonium
NH <sub>4</sub> Cl	ammonium chloride
NRHP	National River Health Program
PRIMER	Plymouth routines in multivariate ecological research
SASS	South African scoring system
SASS5	Version 5 of SASS
SIMPER	similarity of percentages
TON	total oxidisable nitrogen
TSS	total suspended solids
TWQR	Target Water Quality Range
SPI	specific pollution index
PTV	pollution tolerant valves

## GLOSSARY OF UNITS MEASURED

mg/l	milligrams per litre
mS/m	milliSiemens per meter
°C	degrees centigrade (Celsius)

# CHAPTER 1

## INTRODUCTION

### 1.1 Background and literature review

Many of the urban rivers within the City of Cape Town no longer function naturally, and cannot carry out the functions required for humans or ecosystem function. Instead many of these rivers function as nothing more than stormwater drains (Davies and Day, 1998).

According to Findlay and Taylor (2006), the hardening of a catchment is typical of urbanisation, reducing the ability of soils to absorb rain water and runoff. Subsequently, far more water runs off urban development than natural areas thus reducing our rivers to stormwater drains, changing the form and function of our rivers inevitably. Rivers that are gently flowing streams in the dry months become torrents of water flowing in channels that potentially overflow their banks, threatening people, property and infrastructure in the winter months. Urbanisation has thus contributed to most rivers over time being modified to prevent flooding and threat to people, property and infrastructure within our catchments (Davies and Day, 1998).

The average urban river has suffered the loss of riparian vegetation, canalisation, stormwater inflows, bridges, infestation of exotic vegetation and the removal of wetlands, thus becoming a limiting factor on the abundance and diversity of stream ecology (Findlay and Taylor, 2006). Any capacity that rivers once had to regulate their own flows and to provide habitat and water quality buffering, have been destroyed by the engineering modifications to meet the needs of urbanisation, development and over population (Davies and Day, 1998).

Activities in river catchments are reflected in changes in associated riverine ecosystems, alterations or perturbations which can affect the entire length of the river downstream (Dallas and Day, 1993). In many rivers in South Africa flow is strongly seasonal, and there is considerable year to year variability in flow rates. Some rivers even dry up completely, in the non-rainy season. According to Davies and Day

(1998), such rivers are also prone to alteration which leads to unnatural flow regimes and conditions causing stress to the aquatic ecosystems.

The City of Cape Town (COCT) has realised over time that engineering solutions to deal with the problems associated with urbanisation are not always the best solution. The COCT: Catchment Management Department in conjunction with the COCT: Environmental Management Department have started to improve some of the heavily engineered or modified rivers into what is perceived as remediation measures that may be more ecologically sustainable (<sup>1</sup>Thompson, 2010, pers. comm.). However, according to Day and Ractliffe (2002), few in-depth assessments have been done on the effects of these engineering interventions, and whether these interventions are truly improving the ecological function of rivers in an urban context.

Aquatic ecosystems, in this case rivers and wetlands, include numerous species, habitats and processes, all of which are interlinked and interdependent, and which require protection if healthy ecosystem structure and functioning, are to be maintained (DWAF, 1996).

Many streams and rivers draining populated areas are affected in multiple ways and are loaded by organic nutrients and other pollutants, while their flow regimes and habitats are changed by e.g. channelisation, thus having different regulating impacts on macroinvertebrates, and the functioning of urban streams (Sawyer et al., 2004; Soininen and Könönen, 2004).

The four major components of a riverine system that influence the productivity of aquatic organisms are: flow regime, physical habitat structure (channel form and substrate), water quality (temperature, dissolved oxygen) and energy inputs (nutrients and organic matter). The effects of alterations in these system components can be recorded by monitoring riverine macroinvertebrate assemblages (Snaddon, 2009). For this reason various indices exist to assess the ecological functioning of a river, such as water chemistry, habitat quality and diversity, and invertebrate and diatom communities.

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<sup>1</sup> Mr. M. Thompson, Catchment Manager, City of Cape Town: Catchment Management, Cape Town

It is well known that water quality is a vitally important aspect of river ecological health, and that water quality is a standard parameter included in a river health assessment programme (e.g. Triest et al., 2001; Reash, 2003; Donohue et al., 2005; Leira and Sabater, 2005; Findlay and Taylor, 2006; Hirsch et al., 2006; Hu et al., 2006). The then Department of Water Affairs and Forestry (DWAF), in 1996, developed the South African Water Quality Guidelines for Aquatic Ecosystems. At the time there was limited data available with which to assess the water quality requirements of local species, and therefore international literature from Australia, Canada and the United States provided a major part of the information used in developing the guideline. The aim of this project was to develop a set of criteria to assist with the ongoing effort to improve decision making and for the management of the country's resources. These guidelines specifically address the surface water quality required to protect freshwater ecosystems, as well as the quantitative and qualitative criteria to protect ecosystem structure and functioning (DWAF, 1996).

When assessing water quality, it is important to keep in mind that different water constituents support different aquatic biota. Aquatic biota are usually adapted to the natural seasonal cycles of changing water quality which characterise these systems. Changes in the amplitude, frequency and duration of these cycles can greatly affect the ecological and physiological functioning of the aquatic organisms, and the ecological functioning of the system (King et al., 2003; Palmer et al., 2005). The water constituents affect the different macroinvertebrate organisms differently, and can indicate whether the river is functioning sustainably (DWAF, 1996). As described by Palmer et al. (2005) and Snaddon (2009), in South Africa, water quality variables that should be considered when undertaking water chemistry tests include among others, nutrients (phosphates, total nitrogen, nitrates and nitrites, and ammonia) as well as physical variables (turbidity, pH, oxygen demand, and conductivity).

Physical variables such as the concentration of total suspended solids (TSS) which have been studied extensively around the world, could potentially affect the abundance and diversity of macroinvertebrates present in each season (DWAF, 1996; Sawyer et al., 2004; Bilotta and Brazier, 2008). In a review undertaken by Bilotta and Brazier (2008), the damage that suspended solids can cause to the invertebrates and the function of these invertebrates in terms of their functional

feeding groups was discussed, as well as how this may affect higher trophic levels. Similar findings were discussed in several case studies by Cosser (1989), Wood and Armitage (1997), Deksissa et al. (2003), Simon (2003), Dallas and Day (2004), Khalife et al. (2005) and Pagliosa et al. (2005).

According to Palmer et al. (2004), electrical conductivity (which is considered a surrogate measurement for salinity) is also a major threat to the water resources of South Africa due to runoff from naturally saline soils, irrigation return flows from agricultural land, and the influence of saline industrial and sewage effluents. The influence of conductivity on macroinvertebrates was tested by Palmer et al. (2004) in the Olifants River, as well as the Breede and Thukela rivers. This showed the toxicity responses of South African taxa and identified the need for guidelines and methods to protect water resources. There have also been several other case studies which have investigated and shown the impact of increasing conductivity and salinity on aquatic biota, for example Goetsch and Palmer (1997), McNeil and Cox (2000), Nielsen et al. (2003), Oberholster et al. (2008).

Chemical oxygen demand (COD), according to Paterson and Nursall (1975) and Dallas and Day (1993), is the consumption of oxygen during the breakdown (oxidation) of chemicals contained in effluent. COD can determine the level of pollutants in-stream, as the breakdown of chemicals or pollutants result in the lowering of dissolved oxygen concentrations which would certainly have an effect on the survival of natural riverine biota. The effects of oxygen on aquatic biota are well studied (e.g. Paterson and Nursall, 1975; Rader and Richardson, 1992; Dallas and Day, 1993; Dallas and Day, 2004; Rossouw, 2006).

Changes in pH from that which is normally encountered in a system, according to Dallas and Day (1993), may have severe effects upon biota. The extent of acidification and alkalisation is important in determining the degree of severity of the effects of change in pH levels. However, pH does change the toxicity of several chemical variables (e.g. pH levels can affect the toxicity of un-ionised ammonia at certain water temperatures) (DWAF, 1996). Some streams are naturally more acidic than others and their biota are adapted to these conditions (Dallas and Day, 1993).

Most nutrients (nitrogen and phosphorus) are not toxic, (except for nitrite at high concentrations and un-ionised ammonia at low concentrations) (Dallas and Day, 1993; Camargo and Alonso, 2006). High concentrations of nutrients, urban toxins, heavy metals or hydrocarbons can have a significant impact on the structure and functioning of biotic communities. Climate and catchment characteristic can influence the nutrient concentrations in rivers (Dallas and Day, 1993, Findlay and Taylor, 2006). Increased water temperature and pH could cause ammonia to become toxic which, depending on the concentration of ammonia, can have a chronic or acute effect on the aquatic biota. According to Dallas and Day (2004), Deksissa et al. (2004) and De Villiers and Thiar (2007), anthropogenic sources of nutrients may be from point sources or non-point sources of pollution. The principle anthropogenic point sources of nutrients are municipal sewage effluents and overflows (Kay, 1999; Rossouw, 2006; Olberholster et al., 2008). Non-point source pollution includes waste water from livestock farming, runoff from agricultural or fertilised land such as sports fields or golf courses (Kay, 1999; Winter et al., 2002; He et al., 2006; Chen et al., 2009), runoff from construction sites, polluted ground waters and runoff from urban areas (Dodds, 1993; Camargo and Alonso, 2006; Song et al., 2007).

A number of studies used similar water chemistry variables as mentioned above, to determine the relationship of water chemistry to macroinvertebrate and fish community structure (Yee et al., 2000; Malan and Day, 2003; Parr and Mason, 2003; Palmer et al., 2004; Sawyer et al., 2004; Kyriakeas and Watzin, 2006; Wang et al., 2007; Arimoro, 2009; Hall et al., 2009).

In South Africa, Rowntree and Ziervogel (1999) developed a geomorphological index to assess the geomorphological processes, including sedimentation, erosion etc. This index assisted as input towards a National River Health Program (RHP). This programme consisted of seven components. These components assisted with assessing the overall condition of a site. Of these seven components, vegetation, invertebrates and fish together give a good indication of the ecological integrity of a site, while habitat integrity, water quality, hydrology and geomorphology give a good indication of the biophysical habitat.

These components have been developed into indices, and have become techniques that are widely recognised in South Africa (Dickens and Graham, 2002). Seven river health indices now exist: Invertebrates South African Scoring System (SASS), Fish (Fish CII), riparian vegetation, habitat integrity, water quality hydrology and geomorphology (Rowntree and Ziervogel 1999). Dickens and Graham (2002) concluded that by assessing the biota of rivers the condition of the river can be determined. Macroinvertebrates are particularly valuable organisms in bioassessments. They are largely visible to the naked eye, they have a rapid life cycle which is based on seasons, and they are largely sedentary. South Africa has an exemplary history in the field, and has refined the assessment of macroinvertebrates by developing a South African Scoring System (SASS). This system was developed by Chutter (1994), and this forms the basis for determining the health or condition of a river. SASS5 (version 5 of SASS) was adapted by Dickens and Graham (2002) to the Western Cape. The use of biological indices to assess river health and ecosystem functioning has received much attention over the last ten years (Dallas and Day, 1993; Uys, 1994; Davies and Day, 1998).

SASS has been used as a method in South Africa specifically to assess water quality. A number of studies investigate spatial or temporal changes of macroinvertebrates in a river (Dallas and Day, 1993; Dallas, 1995; Dallas, 1996; and Snaddon, 2009), seasonality amongst macroinvertebrate taxa as well as evaluate their implications for SASS (Dallas, 2004). The City of Cape Town (2005) used SASS and habitat integrity to assess the present ecological state of the rivers of the Greater Cape Town, to get an overview of the state of Cape Town's rivers. Chutter (2004) used SASS methodology to determine if there was a relationship between water quality and the macroinvertebrate communities of South Africa. Grobicki (2001), Chutter (2004) and Thiere and Schulz (2004) have used SASS to assess the impact of effluent on streams and macroinvertebrate communities.

As described by Dallas (2007), habitat integrity has also been used in combination with SASS to determine the influence of biotope availability on macroinvertebrate assemblages in South African rivers. Similarly, SASS methodology and the assessment of habitat integrity had been used to monitor the impacts on a catchment

or river, or as bench marking for future rehabilitation and monitoring (Bergman, 2000; Grobicki, 2001; Day and Ractliffe, 2002).

South Africa has successfully combined the Australian, United Kingdom and United States river health indices and has developed the National River Health Program (NRHP) to develop and build up a database on the physical form and condition of our rivers (Rowntree and Ziervogel, 1999). The rivers database is a national storage and management system developed for the NRHP. This database allows river health practitioners to contribute data to the national rivers database. The data collected for macroinvertebrates uses SASS, and includes habitat integrity and biotopes sampled (DWAF, 2007).

Further to the above indices, the assessment of water quality and river health can be complemented by the assessment of diatom communities, in conjunction with physiochemical water quality analysis, and macroinvertebrate sampling (Leira and Sabater, 2005). According to Soininen and Könönen (2004), a complementary assessment of water quality and river health is the assessment of diatom communities and taxa. Diatoms and macroinvertebrate monitoring have been used in several European countries to monitor water quality in conjunction with chemical analysis and river health. Diatom sampling along with macroinvertebrate sampling are considered superior to chemical analysis of water quality due to their integrating nature. Diatoms are regulated by chemical factors such as nutrients and phosphorus, while macro invertebrates are affected by physical aspects such as channel width, habitat provision and flow regime. Diatoms react to water quality within a couple of days, whereas macro invertebrates could take months or years, and chemical analysis only represents a specific moment in time (e.g. Triest et al., 2001; Smolders et al., 2004; Soininen and Könönen, 2004; Griffith et al., 2005; Leira and Sabater, 2005).

Diatoms are considered to be indicators of water quality and water chemistry conditions (Kwandrans et al., 1998; van der Molen, 2000; Bate et al., 2004; Taylor et al., 2005; Taylor et al., 2007; de la Rey et al., 2008; Kriel, 2008; Morin et al., 2010), and are used as indicators of ecosystem conditions (Weilhoefer and Pan, 2006; Weilhofer and Pan, 2007; Jüttner et al., 2010), and environmental characteristics of a

river (Hill et al., 2001; Pan et al., 2004), or as biological monitors of environmental quality of habitat (Hill et al., 2003 Feio et al., 2007) and river condition or health (Sonneman et al., 2001; Triest et al., 2001; Soininen and Könönen, 2004; Taylor et al., 2005; Beyene et al., 2009; Rodrigues et al., 2009).

Diatoms have also been used as an index to measure eutrophication in rivers (Kelly and Whitton, 1995; Kelly, 1998; Kelly, 2003; Beyene, et al., 2009; Justus et al., 2010), to assess short term structural effects of metals in rivers (Gold et al., 2002; Blanco and Bécares, 2010), to develop an index of biotic integrity for acid mine drainage impacted streams (Zalack et al., 2010), and to monitor organic or urban pollution (Biggs, 1989; Duong et al., 2007).

All of the above mentioned indices and parameters are extensively and successfully used in river health assessment programmes worldwide (although adapted in some cases for specific local conditions, as in the case of SASS) (e.g. Department of Water Affairs and Forestry, 2000; Jordan and Vaas, 2000; Triest et al., 2001; Ogbeibou and Oribhabor, 2002; Reash, 2003; Ofenböck et al., 2004; Thiere and Schulz, 2004; Cuffney et al., 2005; Donohue et al., 2005; Friberg et al., 2005; Chessman et al., 2006).

The objective is often not only to assess the present ecological health of a river, but also to use the information in decision making on river restoration or rehabilitation and management decisions, so that government agencies can determine the effectiveness and success of the various management actions (Hart, et al., 2001; Hirsch et al., 2006; Pedersen et al., 2007).

The City of Cape Town to date has designed and implemented many stormwater and river management projects with varying objectives. Most of the projects, interventions or designs are to address stormwater problems in urban areas, erosion, slope stabilisation, energy dissipation, flood control, etc. (Day and Ratcliff, 2002).

Cape Town rivers are heavily engineered due to urbanisation (<sup>1</sup>Thompson, 2010, pers. comm.). To date very little follow-up investigation has occurred as to whether these engineering projects have had a positive effect on our urban river ecosystems.

In many cases ecosystem enhancement is not considered as the number one objective of intervention projects, which are driven primarily by engineering, and sometimes social and/or aesthetic objectives, with implications in terms of ecological sustainability or functioning being assumed, rather than tested in detail. In more cases than not the engineering solution reaches its engineering, aesthetics and amenity objectives, but may not in fact meet its original ecological requirements.

The six rivers considered in the present study are the Keyzers River, the Little Lotus River, the Langevlei Canal, the Moddergat River, the Silvermine River, and the Big Lotus River. Each one of these rivers has had different engineering interventions, some of which have had secondary or tertiary objectives being to improve their ecological functioning as an urban river, and to provide more diverse riverine habitat. Each river has been exposed to significantly different interventions, all affecting channel habitat. Interventions range between canalisation and de-canalisation (Little Lotus River), channelization and gabion stabilisation (Big Lotus River), diversion from a canal into a wetland low-flow habitat (Langevlei Canal), de-channelisation, gabion lining (Moddergat River), and rehabilitation of the lower flood plain and wetland (Silvermine River) (Day and Ractliffe, 2002).

To date there has been very little monitoring of these river interventions, and it is questionable whether the restoration and the altering of the structure of these rivers is actually successfully addressing the ecological functioning of these rivers.

Previously Day and Ractliffe (2002) have carried out broad scale assessments of the ecological effects of some of the above mentioned projects. However, an in-depth analysis of the impacts of these projects on the ecological function of the selected rivers is necessary. The main question addressed in the present study is whether the different engineering interventions of the six selected rivers, have truly improved ecological aspects such as water chemistry, habitat quality and biotic community structure along the river courses within the constraints of an urban environment. While it is accepted that urban rivers are often impacted systems, and may not represent the function of a pristine stream, the questions is do these interventions assist with the ecological functioning of the river, or do all impacted urban rivers not function.

## CHAPTER 2

### RESEARCH QUESTION AND OBJECTIVES

#### 2.1 Research question

Has the altering of river structure of urban rivers, due to various engineering interventions, within the Cape Town City Metropole <sup>1</sup>improved the river ecosystem function, when considering specific indices including macroinvertebrate abundance and diversity, diatom abundance and diversity, and water chemistry?

#### 2.2 Objectives of the study

##### 2.2.1 Main aims

The main aim was to determine the influence of specific engineering interventions and approaches to urban river management, on river ecosystem function of the Keyzers River, Langevelei Canal, Little Lotus River, Big Lotus River, Silvermine River and the Moddergat River, as measured by specific indices including: water chemistry and aquatic community structure (macroinvertebrates and diatoms) and habitat integrity.

##### 2.2.2 Objectives

- To determine the seasonal macroinvertebrate diversity and abundance, at each site, upstream and downstream of an intervention in each river.
  
- To determine the seasonal diatom diversity and abundance, at each site, upstream and downstream of an intervention in each river as an indicator of water quality.
  
- To determine the water chemistry, at each site, upstream and downstream of an intervention in each river as an indication of water quality.
  
- To determine the seasonal influence of interventions on the habitat by assessing the quality and availability of different aquatic biotopes, at each site, upstream and downstream of the intervention in each river, using IHAS (Invertebrate Habitat Assessment System).

- To determine the seasonal influence of altered river structure, at each site, on invertebrate communities by assessing the macroinvertebrate communities of different biotopes, in terms of their taxonomic composition and /or functional roles using SASS5.

<sup>1</sup>Increase in biodiversity of macroinvertebrates and diatoms, increase in habitat diversity or abundance, and water quality being within the natural range.

Water quality, the river structure, and the river biotope were monitored at the following sites: the Sutherland River and the Sutherland Dam. The Sutherland River is a tributary of the Orange River. The Sutherland Dam is a concrete dam with a length of 1.5 km. Samples of water, diatoms, macroinvertebrates, and the river biotope were taken at the following sites: Sutherland Dam, Sutherland River, and Sutherland Dam. The Sutherland Dam is a concrete dam with a length of 1.5 km.

Samples were collected quarterly. The first sampling was in winter in August 2016, the second sampling occasion in spring in November 2016, the third sampling occasion in summer in February 2017, and the fourth sampling occasion in autumn in April 2017. Diatom sampling was only conducted during the summer and winter sampling occasions, since sampling sites could not be accessed, either due to a lack of water or too much flow. Care was taken to ensure that samples were not contaminated by fishing equipment (e.g. nets) thoroughly before use.

The following river health indices and parameters were measured: water chemistry, habitat quality and availability, macroinvertebrates and diatom communities.

## 2.4.1 Macroinvertebrates

### 2.4.1.1 Assessment of macroinvertebrates: Seasonal abundance, seasonal diversity and consistency of taxa

Macroinvertebrates were collected using the SASS5 methodology. This system was developed by Chatter (1974), and adopted by Dickens and Graham (2002). This forms the basis for determining the health or condition of a river (Dickens and Graham, 2002).

Different biotopes were sampled for macroinvertebrates using the rapid, qualitative, multi-biotope, field based method, SASS5 (South African Scoring System Version 5, Dickens and Graham, 2002). A hand held kick-net with a 630 µm mesh sieve

## **CHAPTER 3 CAPETOWN MATERIALS AND METHODS**

### **3.1 Sampling techniques**

Chemical analysis of water samples, analysis of habitat quality, and macroinvertebrate and diatom community studies were undertaken in the Keyzers River, the Little Lotus River, the Langevlei Canal, the Moddergat River, the Silvermine River, and the Big Lotus River, for a period of 1 year. Samples of water chemistry, macroinvertebrates and diatoms were taken at the same sampling points (Chapter 4) upstream, downstream, and at the river intervention.

Samples were collected seasonally. The first sampling occasion was in winter in August 2006, the second sampling occasion in spring in November 2006, the third sampling occasion in summer in February 2007, and the fourth sampling occasion in autumn in April 2007. Diatom sampling was only conducted during the summer and winter sampling. On occasion, some sampling sites could not be sampled, either due to a lack of water or too much flow. Care was taken to ensure that samples were not contaminated, by rinsing equipment (e.g. nets) thoroughly between sites.

The following river health indices and parameters were measured: water chemistry, habitat quality and availability, macroinvertebrate and diatom communities.

#### **3.1.1 Macroinvertebrates**

##### **3.1.1.1 Assessment of macroinvertebrates: Seasonal abundance, seasonal diversity and sensitivity of taxa**

Macroinvertebrates were collected using the SASS5 methodology. This system was developed by Chutter (1994), and adapted by Dickens and Graham (2002). This forms the basis for determining the health or condition of a river (Dickens and Graham, 2002).

Different biotopes were sampled for macroinvertebrates using the rapid, qualitative, multi-biotope, field based method, SASS5 (South African Scoring System Version 5, Dickens and Graham, 2002). A hand held kick-net with a 950 µm mesh sieve

(attached to a 300mm x300 mm wooden frame) was used to sample the available biotopes including marginal vegetation, riffles and submerged riffles. Each biotope was disturbed by kicking and then stirring or sweeping the kick net through the water for at least one minute. The material collected in the net was then emptied into a tray, debris was removed, and organisms were identified in the field, using Gerber and Gabriel (2002).

The tray was assessed for no longer than 15 minutes per biotope, and was stopped if no new macroinvertebrate family were seen after 5 minutes. The macroinvertebrates identified were recorded on the SASS5 data collection sheet. Although the SASS protocol generates three values: the biotic index called the SASS score, the number of taxa, and the average score per taxon (ASPT). Only the collection method of SASS was used and not the interpretive tools or the scoring system, as not all the rivers were suitable for SASS as a bio assessment tool for water quality (e.g. some rivers were canalised), and the data which were obtained were intended to provide a measure of biodiversity rather than water quality.

The SASS protocol assumes different sensitivities to water quality impacts between different macroinvertebrate taxa. "Sensitivity scores" have thus been derived for macroinvertebrate taxa, at the level of family, and these scores form the basis for the derivation of SASS score and ASPT. In the present study, the sensitivity scores themselves were used, as a mechanism for differentiating between different taxa found in different habitats and at different sites. The actual applicators of the SASS methodology was not however deemed appropriate as outlined above.

The diversity of macroinvertebrate taxa (total number of different macroinvertebrate taxa identified to family level) and abundance of macroinvertebrates (total number of macroinvertebrates) for each site were recorded. Macroinvertebrate abundance was recorded in log score categories, namely 1 individual, 2-10 individuals, 11-100 individuals, and 100 – 1000 individuals. The macroinvertebrates were identified, in the field, to family level. This was used to determine the seasonal diversity of macroinvertebrate taxa, and the seasonal abundance of macroinvertebrates supported by different biotopes and water chemistry.

### **3.1.1.2 Habitat assessment**

The method of assessing the variability of aquatic habitat (biotope) quality and availability (habitat diversity) was partially derived from the IHAS (Invertebrate Habitat Assessment System) approach (Dallas, 2005). This method aims to detail the SASS biotope sampled in each habitat (marginal vegetation, riffles, submerged riffles, slack, pool) (Dallas 2005). In the present study, IHAS was used only as a means to distinguish between qualitatively different habitat types. This is because many of these were unnatural habitats (e.g. gabion baskets) they did not lend themselves to actual IHAS application.

An assessment of riverine habitat was also conducted in the reach affected by the intervention, at each site using Section C of the field data sheets for the River Health Programme (Dallas, 2005). These data sheets were completed for each sample site. The data sheets make reference to general site information, stream dimensions, substratum composition, invertebrate biotopes, biotopes sampled, and any fish caught. A photographic record of each site was kept to allow previous visual conditions to be tracked and recorded, and was used as a reference when assessing seasonal differences.

### **3.1.2 Diatoms**

#### **3.1.2.1 Assessment of diatoms: Seasonal abundance and diversity**

Diatom samples were collected upstream and downstream of each intervention, using the methods described by Taylor et al. (2005). Diatoms were collected in summer and winter. According to Taylor et al. (2005), in summer rainfall regions diatom communities are at their peak in mid-winter and under low-flows. Therefore in the present study, diatoms should be at their peak in summer in the Western Cape (<sup>1</sup>Harding, 2010, pers. comm.), which is a winter rainfall region. It is also noted that sampling is not suitable during prolonged rains as high flows can displace diatom communities.

Diatom samples were collected from each biotope that the macroinvertebrates were collected from. Submerged aquatic macrophytes were collected and placed in a plastic packet (zip lock bag) and approximately 50 ml of water was added. The bag

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<sup>1</sup> Dr. W.R. Harding, Freshwater Ecologist, DH Environmental Consulting, Cape Town.

containing the macrophytes and water was closed and shaken vigorously. The resulting brown suspension was then poured into a wide mouthed sample bottle. At least 250 ml per sample was required. Solid substrates were also sampled. Small rocks were placed in a sample tray and the diatoms removed by vigorously scrubbing the upper surface of the substratum with a small brush (toothbrush) to dislodge the diatom community. The rocks were then rinsed in the tray with water and the resulting diatom suspension then poured into a labelled wide mouth plastic sample bottle. At least 250 ml was collected per sample.

The sample bottles were carefully labelled. Vegetation and substratum samples were preserved separately with ethanol as per the method described in Taylor et al. (2005).

These samples were analysed by the University of Potchefstroom, using the Hot HCl and  $\text{KMbO}_4$  method described by Taylor et al. (2005). This method was used as, according to Taylor et al. (2005), it yields best results with samples taken in South Africa which usually have a high content of organic material.

First the diatoms were allowed to settle for twenty four hours after returning to the laboratory. The clear supernatant liquid was then poured from the sample bottle taking care not to lose any material. Five to ten millilitres of the remaining thick suspension was then placed in a heat resistant beaker. Ten millilitres of potassium permanganate solution was added, and the solution left to settle for twenty four hours. In a fume cabinet, five to ten millilitres of HCl (32%) was added and the solution heated at 90 °C for approximately two hours. After oxidation, one millilitre of hydrogen peroxide was added to ensure the oxidation process was complete, and that no organic material remained, in which case the hydrogen peroxide will not cause lasting foaming. Once confirmed, the solution was allowed to cool and transferred to ten millilitre centrifuge tubes. These samples were then rinsed by centrifuging with distilled water. Once rinsed, the supernatant was poured off and the remaining diatoms and small particles were loosened by a jet of distilled water. The beaker was then filled with distilled water, and diatoms and particles left to settle overnight. Again the supernatant was decanted, and the beaker filled with distilled water. This was repeated until the suspension was clear, and it no longer turned blue litmus paper red.

A small pipette was then used to draw a portion of the cleaned solution. This solution was diluted until slightly cloudy to the naked eye. A single drop of ammonium chloride (NH<sub>4</sub>Cl; 10% solution) was added for every ten millilitres of diluted diatom solution to neutralize the electrostatic charges, and thus reduce aggregation of the particles.

Using a pipette, one point five millilitres of this solution was placed on the slide and was allowed to dry at room temperature. Once the water had evaporated, the diatom coated cover slips were placed on a hot plate at 350 °C for three minutes to dry off excess moisture. Once dry, the diatom cover slips were examined under a four hundred times magnification, to determine if the mounting of the slides was correct. Once the diatom-coated cover slip was mounted to the slide, it was viewed under a one thousand times magnification for final viewing and diatom analysis. Only brown algae were analysed.

Prepared slides were examined using a Zeiss light microscope equipped with phase contrast optics. The identification of diatoms was done to species level. Morphological valve characteristics were used as the criteria for identification of the taxa. Diatoms were identified using standard methods. Works by Round (1991) and Taylor et al. (2007) were also consulted.

To produce semi quantitative data from which ecological conclusions could be made, diatoms were enumerated to a total of at least 400 units, as enumeration beyond 400 units has been shown not to significantly affect diatom index scores (Prygiel. 2002). The indices used included the Specific Pollution Sensitivity Index (SPI). Microsoft Excel, 2003 was used for the creation of a database containing all the site information, environmental variable data, diatom abundance data and diatom index results obtained from the OMNIDIA software Package. Opticount software was used for the enumeration of diatom species during diatom sample analyses. The diatom abundance data was then exported to Microsoft Excel 2003, rearranged in the correct format and imported into Omnidia database for the calculation of diatom indices. The Omnidia version 3.1 software package was used for the calculation of

diatom indices and the description of general water quality requirements, of the different species identified.

### **3.1.3 Water chemistry**

#### **3.1.3.1 Assessment of water quality by chemical analysis**

Water samples were collected upstream and downstream of the intervention in each river, from the active channel, using a sterilized one litre plastic container. Whilst sampling, notes were made detailing indications of pollution (litter, smell, oil etc), flow rates, significant rain events in the last week, stormwater outlets, and any other factors that could influence the results of the chemical analysis.

To collect water samples for chemical analysis, the bottle was lowered into the water in the upstream direction. Caution was taken not to disturb any sediment. First the bottle was rinsed with water from the site and then filled. In cases where the sampling site could not be reached (bridge or steep sides), a bucket on a rope was lowered into the water to obtain the sample. Once the samples were collected, they were carefully labelled, placed in a cooler box to keep cool, and delivered directly to the laboratory for analysis. No samples were frozen or stored overnight.

The samples were delivered to Scientific Services of the City of Cape Town Municipality, and analysed immediately.

The parameters analysed were:

- Total suspended solids (TSS)
- pH
- Conductivity
- Chemical oxygen demand (COD)
- Total oxidisable nitrogen (TON)
- Total ammonia (NH<sub>4</sub>-N)
- Soluble nitrite and nitrate (NO<sub>x</sub>-N)
- Total inorganic nitrogen
- Total phosphorus (mg/l P)
- Orthophosphates (PO<sub>4</sub>-P)

Total oxidisable nitrogen, total ammonia, soluble nitrite and nitrate, total phosphorus and orthophosphate were assessed using the Lachat QuickChem FIA+ Automated Ion Analyser (Zellweger Analytics, 1998) methodology. Total suspended solids, pH, conductivity, and chemical oxygen demand were analysed using the standard methods for the examination of water and wastewater as described by Franson (1998).

The parameters analysed were consistent with the ongoing data collection that had been undertaken for the six studied rivers, by the City of Cape Town. It should be noted that no “reference” water quality samples were taken for the present study, as this was not a requirement for the purpose of this study. Therefore the Target Water Quality Range (DWAF, 1996) for several water chemistry parameters could not be determined. I.e. it was not possible to establish whether some parameters exceeded the background concentrations (e.g. Total suspended solid concentrations).

## **3.2 Water chemistry analytical methods**

### **3.2.1 Total suspended solids (TSS)**

TSS were analysed using the laboratory method Total Suspended Solids Dried at 103 °C – 105 °C as detailed in the standard methods for the examination of water and wastewater (Franson, 1998).

A well mixed sample was filtered through a standard glass fiber filter of known weight, and the residue retained on the filter was dried to a constant weight at 103°C to 105°C. The increase in weight of the filter represented the TSS.

### **3.2.2 pH**

pH was measured using the electrometric method as detailed in the standard methods for the examination of water and wastewater (Franson, 1998). This method determines the activity of hydrogen ions by potentiometric measurement, using a glass hydrogen electrode and a reference electrode.

A pH meter consisting of a glass electrode, a reference electrode, and a temperature-compensating device, was used. Most pH meters are accurate at a pH of 0.1 units, with a range of 0-14 pH.

The electrode system was calibrated against standard buffer solutions of known pH. The equilibrium was then established between the electrodes and the sample by stirring the sample to ensure homogeneity (stirring slowly to minimize carbon dioxide entrainment). The electrode was then submerged into the sample, and a pH reading taken. This was repeated once to ensure accuracy.

### **3.2.3 Conductivity**

Conductivity was analysed by using the laboratory method as detailed in the standard methods for the examination of water and wastewater (Franson, 1998). A conductivity meter consisting of a thermometer and a conductivity cell was used. The electrodes were thoroughly rinsed with one or more portions of the sample and then inserted into the samples and conductivity measured.

### **3.2.4 Chemical oxygen demand (COD)**

COD was analysed using the Closed Reflux, Titrimetric Method as detailed in the standard methods for the examination of water and wastewater (Franson, 1998).

A sample was refluxed in a strongly acidic solution with a known excess of potassium dichromate. After digestion, the remaining unreduced potassium dichromate was titrated with ferrous ammonium sulfate to determine the amount of potassium dichromate consumed. The oxidisable matter was then calculated in terms of the oxygen equivalent (Franson, 1998).

### **3.2.5 Total oxidisable nitrogen (TON) and soluble nitrites and nitrates (NOX- N)**

The Flow Injection Analysis (FIA) method was used to analyse total oxidisable nitrogen, nitrites and nitrates (total nitrogen) as detailed by Zellweger (1998).

The FIA is an automated method of introducing a precisely measured portion of liquid sample into a continuously flowing carrier stream. The sample portion is injected into the carrier stream by an injection valve with a fixed volume sample loop (Franson, 1998).

As the sample portion leaves the injection valve it disperses into the carrier stream and forms an asymmetric Gaussian gradient in analyte concentration. This concentration gradient is detected continuously by colour reaction (Franson, 1998).

#### **3.2.5.1 Total oxidisable nitrogen**

The method used was as described by Franson (1998) and Zellweger (1998). The nitrogen compounds were oxidized in line to nitrate using alkaline persulfate digestion. Oxidation of nitrogen containing compounds to nitrate was achieved at 105°C and digestion occurred prior to the injection valve. After digestion, nitrate was quantitatively reduced to nitrite by passage of the sample through a copperised cadmium column. The nitrite (reduced nitrate plus original nitrite) was then determined by diazotization with sulfanilamide under acidic conditions to form diazonium. The diazonium ion was then coupled with N-(1-naphthyl) ethylenediamine dihydrochloride. The resulting pink dye absorbed at 540 nm, and was proportional to total nitrogen. This was detected calorimetrically on the FIA.

#### **3.2.5.2 Soluble nitrate and nitrite (NOX-N)**

The method used is described by Franson (1998) and Zellweger (1998). Nitrate was quantitatively reduced to nitrite by passage of the sample through a copperised cadmium column. The nitrite (reduced nitrate plus original nitrite) was then determined by diazotizing with sulphanylamine, followed by coupling with N-(1-naphthyl) ethylenediamine dihydrochloride. The resulting water soluble dye had a magenta colour which was read at 520 nm. This was detected calorimetrically on the FIA.

#### **3.2.6 Total inorganic nitrogen**

Total inorganic nitrogen was calculated by the addition of total (soluble) ammonia and measured (nitrate and nitrite) data.

#### **3.2.7 Total ammonia (NH<sub>4</sub>-N)**

The FIA method as described in this section 4.2.5 was used to analyse total Ammonia (Franson, 1998; Zellweger, 1998.). This method was based on the Berthelot reaction. Ammonia reacts with alkaline phenol and sodium hypochlorite to form indophenols blue (Franson, 1998; Zellweger, 1998).

Using the FIA method, sodium nitroprusside (nitroferricyanide) was added to each sample to enhance sensitivity. The absorbance of the reaction product was measured at 630 nm. This was detected calorimetrically on the FIA, and was directly proportional to the original ammonia concentrations.

According to Dallas and Day (1993) and DWAF (1996), the ammonia criteria for aquatic ecosystems are calculated from the total ammonia concentration which is the sum of  $\text{NH}_3$  and  $\text{NH}_4^+$  concentrations. The norms for assessing the effects of free ammonia on aquatic ecosystems are chronic and acute toxic effects of ammonia on aquatic organisms.

In the present study the proportion of un-ionised ammonia ( $\text{NH}_3$ ) in each sample was calculated from total (soluble) ammonia using the conversion table of DWAF (1996), and based on measured pH and assumed summer and autumn, spring and winter temperatures of 20°C and 15°C respectively.

### **3.2.8 Total phosphorus and orthophosphates ( $\text{PO}_4\text{-P}$ )**

The FIA method as described in this section 3.2.5 was used to analyse total phosphorus and orthophosphates (Franson, 1998; Zellweger, 1998).

According to Franson (1998) and Zellweger (1998), based on the digestion of various forms of phosphorus, and conversion to phosphates by peroxodisulfate with an inline UV digestion, organic phosphorus was converted to orthophosphate by UV catalyzed persulfate digestion. Polyphosphates were converted to orthophosphate by sulfuric acid digestion prior to the sample valve. A portion of the digested sample was then injected and phosphate concentrations determined by FIA.

After digestion, the orthophosphate ion reacted with ammonium molybdate and antimony potassium tartrate to form a phosphomolybdate complex. This complex was then reduced with ascorbic acid to form a blue complex which absorbed light at 880nm. This was detected calorimetrically on the FIA. The absorbance was proportional to the concentration of orthophosphate in the sample.

### 3.2.9 Dissolved oxygen

Dissolved Oxygen was not measured during the present study. This has been noted as a parameter that will need to be included in any future studies as it plays a vital role when assessing the data and discussing the results.

### 3.3 Statistical analysis

The statistical analysis of the data was undertaken using the Analysis of Variance to determine if there was a significant difference in macroinvertebrate, diatom and water chemistry data for each river. Rivers were then compared for similarities between sites, by undertaking a multivariate analysis of the data, using Plymouth Routines in Multivariate Ecological Research (PRIMER) package to test for significant differences between groups of samples identified a priori by the analysis of similarities to represent relationships between the sites and seasons. A Similarity Percentages (SIMPER) procedure (Clarke, 1993) of Macroinvertebrate abundance between sites, and seasons across all rivers sampled, was carried out to try and identify the taxa contributing to these differences. Abundance data was not transformed in the PRIMER programme since the raw data were in effect collected on a log scale (p 3-3, Section 3.1.1.3).

#### 3.3.1 Analysis of Variance (ANOVA)

There were two factors involved: *Season* (Summer, Autumn, Winter and Spring) and *Site*. To establish *Season* and/or *Site* differences a two-way analysis of variance was, in general, the analysis of choice. The two-way data tables, in this study, had only a single observation per cell and thus there were no degrees of freedom left to test for significance of interaction between the factors. This was circumvented by Tukey's test for additivity. An *interaction graph* is one in which a separate graph is plotted on the same chart for each *Site* for any one of the response variables on the vertical axis, against points representing each *Season* (on the horizontal axis). When the interaction graphs were plotted it was visually clear that in many of the cases the two *Site* profiles were practically parallel. This meant that that there was no interaction between *Site* and *Season*. In such cases two-way additive ANOVA models were applied to test simultaneously for the effect of *Season* and of *Site*.

The ANOVA model included only the main effects of *Season* and the main effects of *Site*. The p-values of the F-tests for *Season* and *Site* were listed for each variable and decisions were based on a significance level of 5%.

In those cases where the profiles crossed over and were apparently different slopes the separate one-way ANOVAs for *Season* and for *Site* were applied.

If a factor showed a significant difference an explanation for the difference was achieved by applying Tukey's test for simultaneous pairwise comparisons (Tukey's HSD test). This procedure was applied by using the software package STATISTICA (data analysis software system), version 10. [www.statsoft.com](http://www.statsoft.com) ).

Due to the small sample sizes, before each of the analyses were done a Box-Cox transformation was applied to the data to establish if the assumptions for ANOVA were met (i.e. normality of the data and constancy of variance). In most cases no transformation was required but in some instances the logarithmic transformation  $Y = \text{Log}_e(X + 1)$  was applied. Others that were required were the square root, inverse square root, power transformation or the regular inverse. These transformations were done in the software package *Design-Expert*<sup>TM</sup> of Stat-Ease Inc., (see [www.statease.com](http://www.statease.com)). In every case, before the transformation was applied, the residual plot of the data was inspected to confirm that the variance was stabilised.

The ANOVA tables are not repeated in the report but only the F-value with its degrees of freedom, e.g. F(m, n), for a given situation together with its p-value are listed ( $p_{\text{site}}$  or  $p_{\text{season}}$ ). Decisions based on these tests were taken at the 5% level of significance. Thus if any p-value is less than 0.05 the hypothesis (of equal mean values, by *Season* or by *Site*) is rejected (the differences are significant) at the 5% level of significance.

### **3.3.2 PRIMER**

A number of routines in the PRIMER (Plymouth Routines in Multivariate Ecological Research) package version 6, as developed at the Plymouth Marine Laboratory, United Kingdom (Clark and Warwick 1994), were used to test for significant differences between rivers, sites and seasons. Initially, multidimensional scaling

(MDS) analysis was carried out on log transformed abundance data. Data shown in the MDS plot were grouped at a 60% level of similarity. Such information was derived from a hierarchical cluster analysis that was carried out on abundance data. Hierarchical cluster analysis fuses samples that have the highest similarities into distinct groupings, and then joins groups into larger clusters at progressively lower levels of similarity, to produce a plot of the natural similarities between groups or samples, irrespective of any group identification assigned to them previously. These results were then represented in a dendrogram with the x axis representing the samples, and the y axis the level of similarity at which successive groupings were formed (Clark and Warwick, 1994; <sup>2</sup>Day, 2010, pers. comm.).

In the present study the clusters suggested by the analysis were used in the SIMPER analysis to identify the macroinvertebrate species that best characterized each cluster as the species that were responsible for the differentiation between clusters. The taxa responsible for 60% within group similarity were identified.

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<sup>2</sup> Dr.E. Day, Freshwater Consultant, Freshwater Consulting Group, Cape Town.

## CHAPTER 4

### STUDY AREA

#### 4.1 Introduction

The six rivers that were considered in this investigation are the Keyzers River, Little Lotus River, Langevlei Canal, Moddergat River, Silvermine River, and Big Lotus River. Each one of these rivers has different engineering interventions with one of the objectives being aesthetics, which may in fact improve the ecological functioning of such “rivers”, and provide more habitats (<sup>1</sup>Thompson, 2010, pers. comm.).

All the rivers sampled are urban rivers that are exposed to moderate to high levels of pollution, are heavily engineered, and act as stormwater conduits for urban runoff. These rivers all run through low to middle income areas, and are often close to light industrial areas (Ractliffe and Day, 2001).

The six rivers selected originate from four catchment areas, namely the Silvermine River Catchment, Sand River Catchment, Zeekoevlei Catchment, and the Eerste River Catchment (Ractliffe and Day, 2001). All of the above rivers are within the Cape Metropole.

River reaches for sampling were selected on the basis of the presence of varying interventions, and the suitability of biotopes present, upstream, at, and downstream from the intervention, along each river course.

#### 4.2 Sampling sites

A summary of the sampling sites, including the biotopes, substratum, channel widths, form, and bank heights are shown in Table 4.1 (p 4-12). All sites were numbered from upstream to downstream. All sites were sampled for diatoms and water quality, and only selected sites depending on the biotope present for macroinvertebrates.

The reason for water quality and macroinvertebrates not being sampled at exactly the same sites is that, to obtain a true representation of the water chemistry results, there needs to be sufficient distance between sample sites so as to depict any

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<sup>1</sup> Mr. M. Thompson, Catchment Manager, City of Cape Town: Catchment Management, Cape Town

change in water chemistry over a reach of river, whereas macroinvertebrate sampling is depicted by the availability of habitat to be sampled.

### **4.3 Habitat assessment**

A summary of the habitat sampled seasonally, at each site and the structure of the river at each site, for all rivers, are shown in Table 4.2 (p 4-19).

## **4.4 Rivers**

### **4.4.1 Keyzers River**

The Keyzers River is a major sub-catchment of the Sand River Catchment, which drains the residential and agricultural areas of Tokai and Constantia, passes through commercial and light industrial areas, drains into Zandvlei Estuary, and into the sea at Muizenberg Beach (Day and Ractliffe, 2002).

As classified in the report compiled by Harding et al. (2002), the study area of the present study for the Keyzers River was classified as a class E river, which identifies that there has been a significant loss of natural habitat, but that there was basic ecosystem function. This reach of river was further prioritised as a category two river, and was identified as having a high ecological priority as it feeds important wetlands downstream (Zandvlei Estuary). When classifying this reach of the river, the water quality was identified by Harding et al. (2002) as being critically impacted.

According to Day and Ractliffe (2002) this river has been severely impacted. Impacts such as soft channelisation, dredging, widespread pollution, litter loading, flooding, and the effects of poor water quality, are contributing factors to its degradation (Day and Snaddon, 2000).

Due to flooding downstream, the lower reach of the Keyzers River was widened during the development of a shopping and residential complex, thus increasing the capacity of the river to accommodate the 1: 50 year flood flow, while at the same time improving the characteristics and amenity value of the river (Day and Ractliffe, 2002).

In the present study, the reach of the Keyzers River that was sampled was from the Tokai Road Extension Bridge to the Military Road Bridge (Figure.4.1, p 4-12). This reach of the river has been widened, and is made up of sections of hardened

substrate (concrete) upstream of and at the Tokai Road Extension Bridge, semi hardened substrate (armourflex) below the Tokai Road Extension Bridge, and soft (earth) canalization between the armourflex below Tokai Road Extension Bridge and Zandvlei Estuary. The constant dredging of this river has resulted in steep vertical embankments, which are subject to erosion. The river is also subject to widespread pollution and litter (Day and Ractliffe, 2000; personal observation).

Currently this reach of the river is being reshaped and replanted with more indigenous vegetation. It is marginally vegetated with Kikuyu grass (*Pennisetum clandestinum*) and Bulrush (*Typha capensis*), with patches of indigenous vegetation where rehabilitation has occurred (personal observation).

A total of three sites (KVS1, KVS2 and KVS3) were selected along this reach of river, including two water sampling sites, three sites where marginal vegetation was sampled for macroinvertebrates, as well as two diatom sampling sites. It was noted by the author that each site was severely impacted by dredging, and litter.

Site KVS1 (earth canal, marginal vegetation) was directly downstream of the Tokai Road Extension Bridge, Tokai. Site KVS2 (earth canal, marginal vegetation) was at the end of Lily Road in Steenberg Industrial Area, upstream of the stormwater outlet. Site KVS 3 (earth canal, marginal vegetation) was directly downstream of the Military Road Extension Bridge, Steenberg.

Table 4.1 (p 4-12) provides a summary of the sampling sites in each river detailing the biotopes, substratum, channel widths, form and bank heights, for each season and each river, also indicating which indices (macroinvertebrates, diatoms and water chemistry) were sampled and investigated at each site.

This reach of the river is subject to many point source and non-point source pollutants which may have an effect on the water chemistry, and as such the quality of the water sampled. Sources of pollution could be associated with dredging, runoff directly from the road bridges at Tokai Road Extension Bridge and Military Road Bridge, release of industrial effluent into the stream, the release of sewage into the

stream, litter loading, runoff from the hardened catchment areas, and erosion or runoff from agricultural areas (Day and Snaddon, 2000).

#### **4.4.2 Little Lotus River**

The little Lotus River is a highly modified environment, and people and their activities have been affecting this system for centuries (Grobicki, 2001). The Little Lotus River forms part of the Zeekoevlei Catchment. It flows from approximately the Royal Cape Golf Course in a southerly direction, passing through the informal settlements of Kanana and Barcelona (which are situated on a “disused waste disposal site”), through high density low income residential areas such as Lotus River and Grassy Park, into Zeekoevlei, and then into False Bay (Moses, 2006).

As classified in the report compiled by Harding et al. (2002), the study area in the present study for the Little Lotus River, was classified as a class F river. This identified that modification on this reach of the river had reached a critical level, the lotic system had been modified completely with almost a complete loss of habitat and biota, and potentially basic ecosystem functions had been destroyed, resulting in changes being irreversible. This reach of river was further prioritised as a category five river, which identified the study area as having a low ecological priority as its rehabilitation potential was very low, and it was canalised. When classifying this reach of the river there was no water quality data, and therefore there could be no assessment of the water quality.

In the present study, the reach of the Little Lotus River that was sampled was at the Fifth Avenue Bridge Lotus River, and downstream of the footbridge which is located approximately 100m upstream of the bridge at Fifth Avenue (Figure 4.2, p 4-22). This reach of the river was a concrete canal with no vegetation. Rocks had been placed just upstream of the Fifth Avenue Road Bridge, and a weir created before the water continues downstream under the bridge. Part of the concrete wall had been replaced with gabions and some marginal vegetation such as Watercress (*Rorippa nasturtium-aquaticum*) and Kikuyu grass (*Pennisetum clandestine*) had re-established along the river edge (personal observation).

The Little Lotus River is subjected to a vast amount of point source and non-point source pollution and litter. Sources of pollution could be associated with the release of sewage into the stream, runoff from hardened catchment areas, litter loading, and runoff from unserviced residential areas upstream (Grobicki, 2001). A total of four sites (LL1, LL2, LL3 and LL4) were selected along this reach of the river, including two sites for water chemistry and diatom sampling, and three sites for macroinvertebrate sampling.

Site LL1 (marginal vegetation), LL2 (riffle) and LL3 (submerged riffle) were directly above Fifth Avenue Road Bridge. LL4 was approximately 100m upstream of site LL1 directly below the foot bridge that crosses the river.

Table 4.1 provides a summary of the sampling sites in each river detailing the biotopes, substratum, channel widths, form and bank heights for each season and each river, also indicating which indices (macroinvertebrates, diatoms and water chemistry) were sampled and investigated at each site.

#### **4.4.3 Langevlei Canal**

The Langevlei River is part of the Sand River Catchment. It flows North to South and originates from the Langevlei detention pond in Retreat, flows through the high density residential area of Retreat and Coniston Park, before draining into Zandvlei, which flows into the sea at Muizenberg Beach (Afridev, 2001). The entire river is canalized and is heavily polluted by windblown litter and runoff from the surrounding urban area (Day and Ractliffe, 2002). The expected catchment area of this canal is approximately 504 ha (Afridev, 2001).

As classified in the report compiled by Harding et al. (2002), the study area of the present study for the Langevlei Canal was classified as a class F river. It therefore identified that modifications on this reach of the Langevlei Canal had reached a critical level, the lotic system had been modified completely with a complete loss of natural habitat and biota, and the ecosystem function had been destroyed. This reach of river was however prioritised as a category two river, which identified this reach of river as having a high ecological priority as it feeds important wetlands downstream (Zandvlei Estuary). There was however no water quality classification for

this reach of the river. It should be noted that the intervention was implemented after the study by Harding et al. (2002).

The reach of the Langevlei Canal that was sampled was downstream of Military Road (Coniston Park), to just upstream of the confluence with the Sand River Canal (Figure 4.3, p 4-23). It was previously a straight concrete canal with steep vertical sides and no vegetation. This canal is subject to litter and pollution (personal observation) Sources of point source and non-point source pollution could be associated with the release of sewage into the stream, runoff from hardened catchment areas, litter loading, and runoff from unserviced residential areas upstream.

In the present study, the canal wall had been breached, allowing high flows to pass through an artificial reed bed to filter the water, before the water re-entered the canal further downstream. This artificial reed bed (wetland) was planted with a diversity of vegetation which included Bulrush (*Typha capensis*), Common reeds (*Phragmites australis*) and a variety of Restio species (personal observation), which provided a filtering mechanism with the intention of improving water quality and habitat.

A total of three sites (LV1, LV2 and LV3) were selected along this reach of river including two water chemistry sampling sites, two marginal vegetation sampling sites for macroinvertebrates, and three diatom sampling sites.

Site LV1 (earth canal) was at the boom gate on Coniston Avenue Coniston Park, directly adjacent to the hole in the canal wall, outside of the canal. Site LV2 (Reed bed) was the same site as LV1 but at the beginning of the reed bed. Site LV3 (marginal vegetation) was directly upstream of the confluence of the Sand River canal at the end of the high flow channel (end of reed bed) before it re-enters the Langevlei Canal.

Table 4.1 provides a summary of the sampling sites in each river detailing the biotopes, substratum, channel widths, form and bank heights for each season and each river, also indicating which indices (macroinvertebrates, diatoms and water chemistry) were sampled and investigated at each site.

#### 4.4.4 Moddergat River:

The Moddergat River originates in the Eerste River Catchment and flows through the residential areas of Firgrove and Macassar, and the small-holding areas between the N2 and the R102 roads, connecting with the Eerste River, in Macassar. The river was a wide, braided, wetland associated system, until it was diverted and confined to a single channel due to urban development encroaching on the 1:50 year floodplain (Day and Ractliffe, 2002).

As classified in the report compiled by Harding et al. (2002), the study area for the Moddergat River in the present study, was classified as a class D river, therefore identifying that the study area was largely modified, and there had been a loss of natural habitat, but that ecosystem functions still occurred. This reach of river was further prioritised as a category one river, which identified this reach of river as having extreme ecological priority, as it has habitat integrity, its relative rehabilitation potential is high, and it displays biological and geomorphological features which are unique among river units of the same "river type". When classifying this reach of the river, the water quality was identified by Harding et al. (2002) as having been largely impacted.

The reach of the Moddergat River that was sampled flows between the N2 and Macassar Road. This reach of the river had been reshaped and gabion-lined to provide bank stabilization (Figure 4.4, p 4-24). Weirs had been introduced to reduce the gradient and facilitate energy dissipation, reducing erosion. Previously this channel was an earth canal with vertical banks infested with Kikuyu grass (*Pennisetum clandestine*), and was heavily polluted with litter (Day and Ratcliff, 2002). In the present study the channel was vegetated along the banks with predominantly Kikuyu grass (*Pennisetum clandestine*) and had marginal vegetation consisting of *Bulboschoenus spp.*, Bulrush (*Typha capensis*) and Watercress (*Rorippa nasturtium-aquaticum*) (personal observation). Sources of point source and non-point source pollution could be associated with runoff from the hardened catchment area, sewage overflows directly into the river, erosion or runoff from the horticultural areas in the broader catchment, as well as litter loading.

A total of seven sampling sites (M1, M2, M3.1, M3.2, M4.1, M4.2, M5, M6, and M7) were selected along this reach of river, five of which were water sampling sites, four marginal vegetation sampling sites, two riffle sites for macroinvertebrate studies, and four diatom sampling sites. It was noted that each site had been heavily impacted by vegetative debris, litter and dumping (personal observation).

Table 4.1 provides a summary of the sampling sites in each river detailing the biotopes, substratum, channel widths, form and bank heights for each season and each river, also indicating which indices (macroinvertebrates, diatoms and water chemistry) were sampled and investigated at each site.

#### **4.4.5 Silvermine River**

The Silvermine River Catchment drains an area of approximately 2260 ha. The source of this river is in the region between the Steenberg Mountain Range and the Constantiaberg Mountain Range. The river flows from the pristine Silvermine Nature Reserve Dam, through the Table Mountain National Park, the Clovelly Golf Course and ultimately between the residential area of Clovelly and Fish Hoek, before entering the sea at Fish Hoek Beach (van der Honert, 2000).

This river was originally an unconfined channel migrating over a wide coastal plain, which historically would have consisted of coastal pans and vleis. Over time the river was confined by urbanization, the construction of roads and bridges, and development within the flood plain. Therefore in order to deal with the stormwater and floodwaters from the surrounding areas, the entire river was altered and the river corridor modified, to create sufficient capacity for the stormwater to escape, thus creating a new 1:50 year floodplain. Along with this engineering solution was intensive rehabilitation of new wetlands that would enhance the ecological functioning and integrity, of the lower reaches of the Silvermine River (Day and Ractliffe, 2002).

As classified in the report by Harding et al. (2002), the study area of the present study for the Silvermine River was classified as a class D river, therefore identifying that this reach of river was largely modified, and there had been a loss and change of natural habitat and biota, which had occurred. However the basic ecosystem

functions were still predominantly unchanged. This reach of river was further prioritised as a category 1 river, which identified this reach of river as having habitat integrity with a high potential for rehabilitation. When classifying this reach of river, Harding et al. (2002) showed the water quality had been moderately to largely impacted.

In the present study, the reach of the Silvermine River that was sampled was between the Pipe Bridge, and the Fish Hoek Main Road (Figure 4.5, p 4-25). A total of four water chemistry sampling sites (SM1, SM2, SM3 and SM4) were selected along this reach of river, including four marginal vegetation sampling sites for macroinvertebrates, as well as three diatom sampling sites, and three water sampling sites.

Sources of point source and non-point source pollution could be associated with runoff from the adjacent hardened catchment, sewage overflow from the adjacent pump station, runoff from adjacent roads, litter loading and potentially runoff from the golf course upstream. As a large portion of this catchment is undeveloped (personal observation), the influence of non-point source pollution in this river is limited (Day and Ractliffe, 2002).

Site SM1 (marginal vegetation) was at the Pipe Bridge on the downstream side of the bridge. SM2 (marginal vegetation) was at the second gabion weir directly upstream of the weir. SM3 (marginal vegetation) was at the outlet of the low flow channel, and SM4 (marginal vegetation) was directly upstream of the Main Road Bridge, outside of the tidally influenced area.

Table 4.1 provides a summary of the sampling sites in each river detailing the biotopes, substratum, channel widths, form and bank heights for each season and each river, also indicating which indices (macroinvertebrates, diatoms and water chemistry) were sampled and investigated at each site.

#### **4.4.6 Big Lotus River**

The Big Lotus River forms part of the Zeekoevlei Catchment. It flows from the Cape Town International Airport in a southerly direction, passing through agricultural areas,

informal settlements, as well as urban and industrial areas, until it discharges into zeekeovlei, and then into False Bay (Kloppers, 2000). The catchment area for the Big Lotus River is so large that it is divided up into several sub-catchments. The sub-catchments that affected the present study are the Phillippi Horticultural Sub-Catchment which is approximately 3500 ha, of which approximately 2279 ha drains towards the Big Lotus River, and 678 ha drains through Grassy Park, Lotus River, and Ottery into the Big Lotus River (Kloppers, 2000).

As classified in the report compiled by Harding et al. (2002), the study area in the present study for the Big Lotus River, was classified as a class E river. This identified that the loss of natural habitat, biota, and basic ecosystem function was extensive, but there was the potential of rehabilitation. This reach of river was further prioritised as a category five river, which identified the study area as having a modest ecological priority, as it has a high relative rehabilitation potential, even though it wasn't situated upstream of an important system. When classifying this reach of the river, the water quality data showed, according to Harding et al. (2002), that the water quality was critically impacted.

In the present study, the reach of the Big Lotus River that was sampled was downstream of the Strandfontein River Bridge (Figure 4.6, p 4- 26). This reach of the river is an earth canal with very steep vertical embankments, and marginal vegetation of Kikuyu grass (*Pennisetum clandestine*), and Watercress (*Rorippa nasturtium-aquaticum*). The sand canal allows water infiltration, and the weir dissipates energy and aerates the water with the hope of improving the water quality (<sup>1</sup>Thompson, 2010, pers. comm.). The river is very heavily polluted and littered. There were also several stormwater outlets entering the system (personal observation). Sources of point source and non-point source pollutants could be associated with runoff and erosion from the adjacent Phillippi Horticultural Area, the over flow of sewage effluent into the river, runoff from unserviced residential areas, runoff from the hardened catchment, runoff from adjacent roads, litter loading and dredging (Grobicki, 2001).

A total of two sampling sites (BL1 and BL2) were selected along this reach of river, which included two sampling sites for macroinvertebrates, and two sampling sites for diatoms and water chemistry. Site BL1 (earth channel and rock weir, marginal



**Table 4.1:** A summary of the sampling sites in each river detailing the biotopes, substratum, channel widths, form and bank heights for each season and each river, also indicating which indices (macroinvertebrates, diatoms and water chemistry) were investigated at each site, VS - Vegetative SASS; RS - Riffle SASS; SRS - Submerged Riffle SASS; av - aquatic vegetation; mv - marginal vegetation. X = not sampled due to a lack of water or too much flow; √ = sampled; – = no sample planned.

River	Season	Site Code	Biotope and SASS Site	Habitat	Macroinvertebrates	diatoms	Water chemistry	Channel Form	Water Level	Last Rainfall (prior to sampling)	Turbidity	Macro Channel Width	Active Channel Width	Substratum Composition	
Keyzers River	Summer	KVS1	VS	reeds	√	√	√	Earth canal	Low		Silty/muddy	<1 m	<1 m	Sand/concrete	
		KVS2	VS	av	√	-	-	Earth canal	Low		Silty/muddy	1-2 m	<1 m	Sand	
		KVS3	VS	av	√	√	√	Earth canal	Low	>7 days	Silty/muddy	2-5 m	2-5 m	Sand/concrete	
	Autumn	KVS1	VS	reeds	√	√	√	Earth canal	Low			Silty/muddy	<1 m	<1 m	Sand
		KVS2	VS	reeds	√	-	-	Earth canal	Low			Silty/muddy	1-2 m	<1 m	Sand
		KVS3	VS	av	√	√	√	Earth canal	Low	<7days	Silty/muddy	2-5 m	2-5 m	Sand	
	Winter	KVS1	VS	reeds	√	√	√	Earth canal	Moderate			Clear	2-5 m	2-5 m	Sand/concrete
		KVS2	VS	mv	√	-	-	Earth canal	High			Clear	2-5 m	2-5 m	Sand
		KVS3	VS	av	√	√	√	Earth canal	High	<7days		Clear	2-5 m	2-5 m	Sand
	Spring	KVS1	VS	mv	√	√	√	Earth canal	Low			Clear	2-5 m	2-5 m	Sand/concrete
		KVS2	VS	reed	√	-	-	Earth canal	Moderate			Clear	2-5 m	2-5 m	Sand
		KVS3	VS	mv	√	√	√	Earth canal	Low	<7 days		Clear	2-5 m	2-5 m	Sand

River	Season	Site Code	Biotope and SASS Site	Habitat	Macroinvertebrates	Diatoms	Water chemistry	Channel Form	Water Level	Last Rainfall (prior to sampling)	Turbidity	Macro Channel Width	Active Channel Width	Substratum Composition	
Little Lotus River	Summer	LL1	VS	av	✓	✓	✓	Rip rap	Low	>7 days	Clear -sheen	±1 m	±1 m	Sand/rocks	
		LL2	RS	riffle	✓	✓	-	Rocks	Low		Clear -sheen	±1 m	±1 m	Sand/rocks	
		LL3	SRS	av	✓	-	-	Rip rap	Low		Clear -sheen	±1 m	±1 m	Sand/rocks	
		LL4			-	-	✓	Earth channel	Low		Clear -sheen	N/A	N/A	Concrete	
	Autumn	LL1	VS	mv	✓	✓	✓	Rip rap	Low	<7 days		Clear -sheen	±1 m	±1 m	Sand/rocks
		LL2	RS	riffle	✓	✓	-	Rocks	Low			Clear -sheen	±1 m	±1 m	Sand/rocks
		LL3	SRS	mv	✓	-	-	Rip rap	Low			Clear -sheen	±1 m	±1 m	Sand/rocks
		LL4			-	-	✓	Earth channel	Low			Clear -sheen	N/A	N/A	Concrete
	Winter	LL1	VS	mv	✓	✓	✓	Rip rap	Moderate			Clear	2-5 m	1-2 m	Sand/rocks
		LL2	RS	riffle	✓	✓	-	Rocks	Moderate			Clear	2-5 m	1-2 m	Sand/rocks
		LL3	SRS	riffle	✓	-	-	Rip rap	Moderate			Clear	2-5 m	1-2 m	Sand/rocks
		LL4	-		-	-	✓	Earth channel	Moderate			Clear	N/A	N/A	Concrete
	Spring	LL1	VS	mv	✓	✓	✓	Rip rap	Low			Clear	2-5 m	1-2 m	Sand/rocks
		LL2	RS	riffle	✓	✓	-	Rocks	Low			Clear	2-5 m	1-2 m	Sand/rocks
		LL3	SRS	riffle	✓	-	-	Rip rap	Low			Clear	2-5 m	1-2 m	Sand/rocks
		LL4	-		-	-	✓	Earth channel	Low			Clear	N/A	N/A	Concrete

River	Season	Site Code	Biotope and SASS Site	Habitat	Macroinvertebrates	Diatoms	Water Chemistry	Channel Form	Water Level	Last Rainfall (prior to sampling)	Turbidity	Macro Channel Width	Active Channel Width	Substratum Composition	
Langevlei Canal	Summer	LV1	VS	reed	x	√	x	Earth canal	Dry (pool)		Discoloured	0	0	Sand	
		LV2	VS	reed	-	√	-	Earth canal	Low		Discoloured	0	0	Sand	
		LV3	VS	av	x	x	x	Armourflex	Dry	>7 days	No water	0	0	Sand/armourflex	
	Autumn	LV1	VS	reed	√	√	√	Earth canal	Low			Clear	2-5 m	2-5 m	Sand
		LV2	VS	reed	-	√	-	Earth canal	Low			Clear	2-5 m	2-5 m	Sand
		LV3	VS	mv	√	√	√	Armourflex	Low	>7 days		Clear	2-5 m	<1 m	Sand/armourflex
	Winter	LV1	VS	reed	√	√	√	Earth canal	Moderate			Clear	2-5 m	1-2 m	Sand
		LV2	VS	reed	-	√	-	Earth canal	Moderate			Clear	2-5 m	1-2 m	Sand
		LV3	VS	mv	√	√	√	Run/sand	Low	<7 days		Clear	2-5 m	1-2 m	Sand/armourflex
	Spring	LV1	VS	reed	√	√	√	Earth canal	Low			Clear	2-5 m	1-2 m	Sand
		LV2	VS	reed	-	√	-	Earth canal	Low			Clear	2-5 m	1-2 m	Sand
		LV3	VS	mv	√	√	√	Armourflex	Low	<7 days		Clear	2-5 m	2-5 m	Sand/armourflex

River	Season	Site Code	Biotope and SASS Site	Habitat	Macroinvertebrates	Diatoms	Water Chemistry	Channel Form	Water Level	Last Rainfall (prior to sampling)	Turbidity	Macro Channel Width	Active Channel Width	Substratum Composition	
Moddergat River	Summer	M1		reeds	-	-	√	Earth channel	Low		Clear	N/A	N/A	Sand	
		M2	VS	mv	x	x	x	Gabion	Dry		No water	0	0	-	
		M3.1	VS	mv	x	x	x	Gabion	Dry		No water	0	0	-	
		M3.2	RS	riffle	x	-	-		Rocks	Dry		No water	0	0	-
		M4.1	VS	mv	x	x	x		Gabion	Dry		No water	0	0	-
		M4.2	RS	riffle	x	-	-		Rocks	Dry		No water	0	0	-
		M5	VS	reed	x	x	x	√	Earth canal	Low		Clear	N/A	N/A	Sand
	M6		reed	-	-	x	x	Earth canal	Dry		No water	N/A	N/A	N/A	
	M7		reed	-	-	x	x	Earth canal	Dry	>7 days	No water	N/A	N/A	N/A	
	M1		reeds	-	-	√		Earth channel	Low flow		Clear	N/A	N/A	Sand	
	M2	VS	mv	x	x	x	x	Gabion mattress	Dry		No water	0	0	-	
	M3.1	VS	mv	x	x	x	x	Gabion mattress	Dry		No water	0	0	-	
	M3.2	RS	riffle	x	-	-		Rocks	Dry		No water	0	0	-	
	M4.1	VS	mv	x	x	x	x	Gabion	Dry		No water	0	0	-	
M4.2	RS	riffle	x	-	-		Rocks	Dry		No water	0	0	-		
M5	VS	reeds	x	x	x	x	Earth canal	Dry		No water	N/A	N/A	N/A		
M6		reeds	-	-	x	x	Earth canal	Dry		No water	N/A	N/A	N/A		
M7		reeds	-	-	x	x	Earth canal	Dry	<7 days	No water	N/A	N/A	N/A		



River	Season	Site Code	Biotope and SASS Site	Habitat	Macroinvertebrates	Diatoms	Water Chemistry	Channel Form	Water Level	Last Rainfall (prior to sampling)	Turbidity	Macro Channel Width	Active Channel Width	Substratum Composition
Silvermine River	Summer	SM1	VS	reeds	x	x	x	Earth shallow canal	Dry	> 7 days	No water	0	0	-
		SM2	VS	terrestrial veg	x	-	-	Earth shallow canal	Dry		No water	0	0	-
		SM3	VS	mv	x	x	x	Rip rap	Dry		No water	0	0	-
		SM4	VS	reeds	x	x	x	Earth canal	Dry		No water	0	0	-
	Autumn	SM1	VS	mv	x	x	x	Earth shallow canal	Dry	<7 days	No water	0	0	-
		SM2	VS	mv	x	-	-	Earth shallow canal	Dry		No water	0	0	-
		SM3	VS	mv	x	x	x	Rip rap	Dry		No water	0	0	-
		SM4	VS	reeds	x	x	x	Earth canal	Dry		No water	0	0	-
	Winter	SM1	VS	reeds	√	√	√	Earth shallow canal	High	<7 days	Clear	10-20 m	5-10 m	Sand/mud
		SM2	VS	av	√	-	-	Earth shallow canal	High		Discoloured	10-20 m	5-10 m	Sand
		SM3	VS	mv	√	√	√	Rip rap	High		Discoloured	10-20 m	5-10 m	Sand
		SM4	VS	reeds	√	√	√	Earth canal	High		Clear	5-10 m	2-5 m	Sand
	Spring	SM1	VS	reeds	√	√	√	Earth shallow canal	Low	<7 days	Clear	2-5 m	1-2 m	Sand
		SM2	VS	terrestrial veg	x	-	-	Earth shallow canal	dry		No water	0	0	-
		SM3	VS	mv	√	√	√	Rip rap	Low		Clear	2-5 m	1-2 m	Sand/cobbles
		SM4	VS	reeds	√	√	√	Earth canal	Low		Clear	2-5 m	1-2 m	Sand

River	Season	Site Code	Biotope and SASS Site	Habitat	Macroinvertebrates	Diatoms	Water Chemistry	Channel Form	Water Level	Last Rainfall (prior to sampling)	Turbidity	Macro Channel Width	Active Channel Width	Substratum Composition
Big Lotus River	Summer	BL1	VS	av	✓	✓	✓	Rip rap	Low	>7days	Clear	2-5 m	1-2 m	Sand/rocks
		BL2	VS	mv	✓	✓	✓	Earth canal	Low		Clear	2-5 m	1-2 m	Sand
	Autumn	BL1	VS	av	✓	✓	✓	Rip rap	Low	<7days	Clear	2-5 m	1-2 m	Sand/rocks
		BL2	VS	mv	✓	✓	✓	Earth canal	Low		Clear	2-5 m	1-2 m	Sand
	Winter	BL1	VS	rifle	✓	✓	✓	Rip rap	Moderate	<7days	Clear - odour	5-10 m	2-5 m	Sand/rocks
		BL2	VS	mv	✓	✓	✓	Earth canal	Moderate		Clear - odour	5-10 m	2-5 m	Sand
	Spring	BL1	VS	rifle	✓	✓	✓	Rip rap	Low	<7days	Clear	5-10 m	2-5 m	Sand/rocks
		BL2	VS	mv	✓	✓	✓	Earth canal	Low		Clear	5-10 m	2-5 m	Sand

**Table 4.2:** A summary of the habitats available at each site, including artificial structures, and the habitats sampled seasonally, in each river. SIC = Stones In Current; MV = Marginal Vegetation; AQveg = Aquatic vegetation. Shaded blocks indicate habitat availability and the artificial structures present at each site, in each season.

Season	Site	Canal	Gabion	Armourflex	Rip Rap	Earth Channel	Sand	SIC	MV	AQVeg
Summer	KVS1					■			■	■
	KVS2					■			■	■
	KVS3					■			■	■
Autumn	KVS1					■			■	■
	KVS2					■			■	■
	KVS3					■			■	■
Winter	KVS1					■			■	■
	KVS2					■			■	■
	KVS3					■			■	■
Spring	KVS1					■			■	■
	KVS2					■			■	■
	KVS3					■			■	■
Summer	LL1	■			■				■	
	LL2	■			■				■	
	LL3	■	■		■				■	
Autumn	LL1	■			■				■	
	LL2	■			■				■	
	LL3	■	■		■				■	
Winter	LL1	■			■				■	
	LL2	■			■				■	
	LL3	■	■		■				■	
Spring	LL1	■			■				■	
	LL2	■			■				■	
	LL3	■	■		■				■	
Autumn	LV1					■	■		■	
	LV3			■		■	■		■	
Winter	LV1					■	■		■	
	LV3			■		■	■		■	
Spring	LV1					■	■		■	
	LV3			■		■	■		■	
Winter	M2		■						■	
	M3.1		■						■	
	M4.1		■						■	
	M5		■			■	■		■	
Spring	M2		■				■		■	
	M3.1		■				■		■	

	M4.1								
	M4.2								
	M5								
Summer	SM1								
	SM3								
	SM4								
Autumn	SM1								
	SM3								
	SM4								
Winter	SM1								
	SM2								
	SM3								
	SM4								
Spring	SM1								
	SM3								
	SM4								
Summer	BL1								
	BL2								
Autumn	BL1								
	BL2								
Winter	BL1								
	BL2								
Spring	BL1								
	BL2								

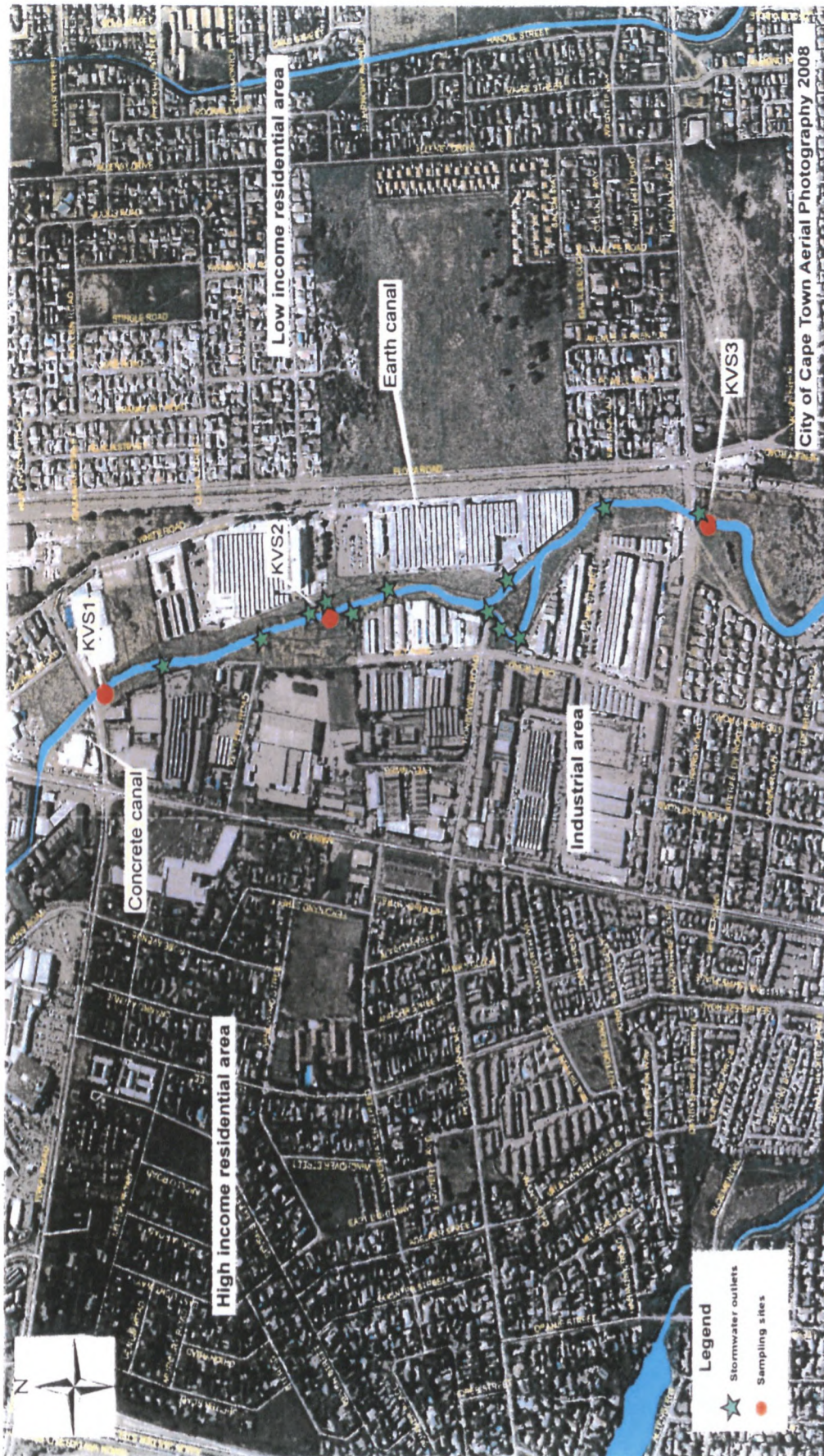


Figure 4.1: The Keyseers River study area and sites.

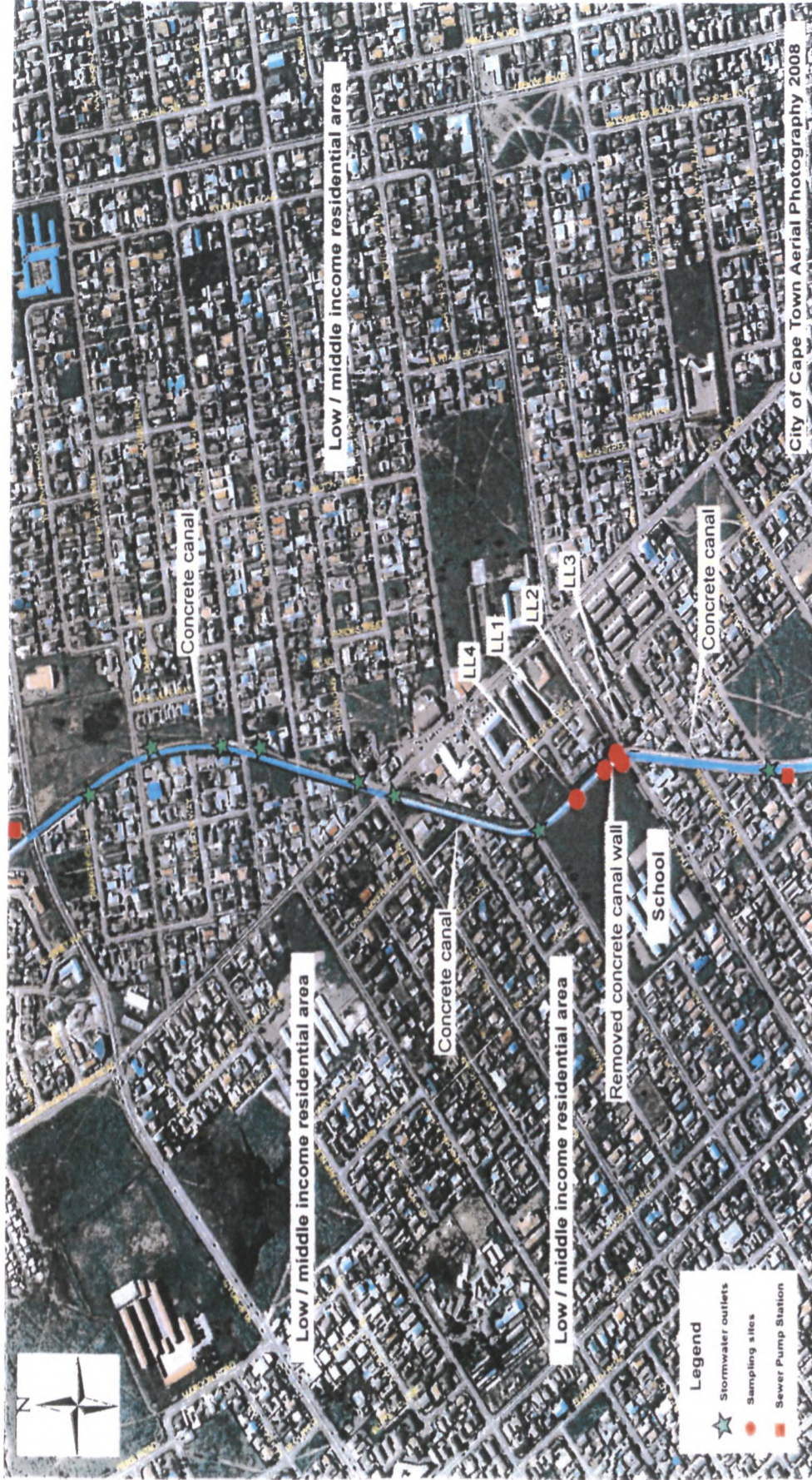


Figure 4.2: The Little Lotus River study area and sites.

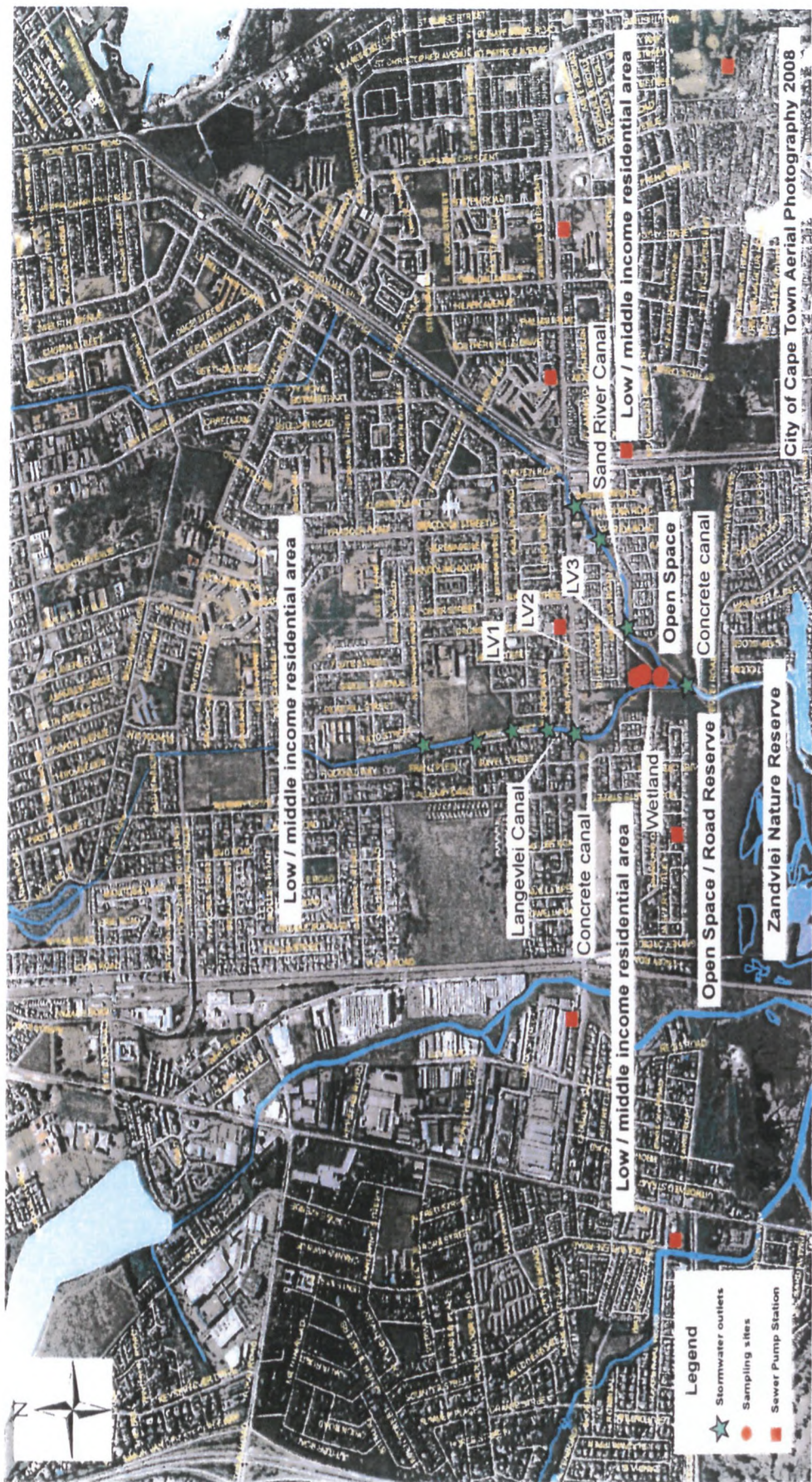


Figure 4.3: The Langevlei Canal Study Area.



Figure 4.4: The Moddergat River study area and sites.

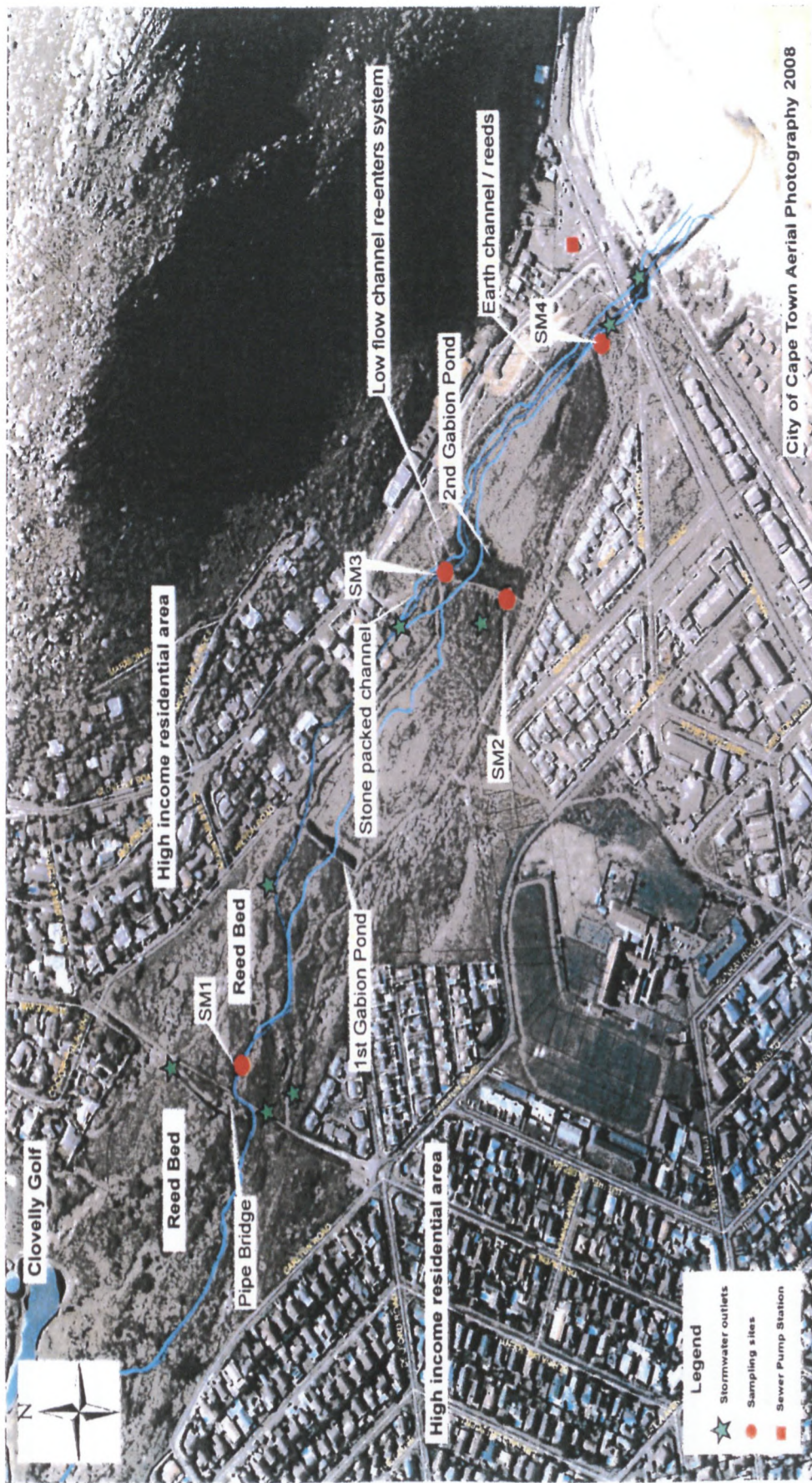


Figure 4.5: The Silvermine River study area and sites.



Figure 4.6: Big Lotus River study area and sites.

**CHAPTER 5**  
**RESULTS: KEYSERS RIVER**

**5.1 Macroinvertebrate results**

**5.1.1 Macroinvertebrates sampled**

Table 5.1 lists the macroinvertebrate taxa sampled in the Keyzers River during summer, autumn, winter and spring at KVS1, KVS2 and KVS3 (KVS1 is the upper most sampling site, KVS2 is the middle sampling site, and KVS3 is the lower most sampling site).

**Table 5.1:** Macroinvertebrate taxa sampled during each season at each site (KVS1, KVS2 and KVS3) within the Keyzers River. × = absence of a taxon; √ = presence of a taxon.

SEASONS	SASS SENSITIVITY	SUMMER			AUTUMN			WINTER			SPRING		
		KVS1	KVS2	KVS3	KVS1	KVS2	KVS3	KVS1	KVS2	KVS3	KVS1	KVS2	KVS3
TAXON	SCORE												
Oligocheata	1	×	×	×	√	√	√	×	×	×	√	×	×
Hirudinea	3	√	√	×	√	√	√	×	×	×	√	×	×
Amphipoda	13	×	×	×	×	×	√	×	×	×	×	×	√
Crabs	3	×	×	√	×	×	×	×	×	×	×	×	×
Baetidae	4	×	×	√	×	×	√	×	√	√	×	√	√
Caenidae	6	×	×	×	×	×	×	×	×	×	√	×	×
Leptophlebiidae	9	×	×	×	×	×	×	×	×	√	×	×	×
Coenagriidae	4	×	√	×	×	×	√	√	√	√	×	×	√
Aeshnidae	8	√	√	×	×	√	√	×	×	×	×	×	×
Gomphidae	6	√	×	×	×	×	×	×	×	×	×	×	×
Libellulidae	4	×	√	×	×	√	√	×	×	√	×	×	×
Corixidae	3	√	×	×	√	√	×	×	×	×	×	√	√
Gerridae	5	√	×	×	×	×	×	×	×	×	×	×	×
Pleidae	4	√	√	√	×	×	×	√	√	√	√	√	√
Veliidae	5	×	×	×	√	√	√	×	×	×	×	×	×
Dytiscidae	5	×	×	×	√	×	×	√	×	×	×	×	×
Gyrinidae	5	×	×	×	×	×	√	×	×	×	×	×	×

Helodidae	12	x	x	x	x	x	x	x	x	x	√	x	x	x
Hydrophilidae	5	x	x	x	√	x	x	x	x	x	x	x	x	x
Ceratopogonidae	5	x	x	x	√	x	√	x	√	x	x	x	x	√
Chironomidae	2	x	√	√	√	√	√	√	√	x	√	√	√	√
Ephydriidae	3	x	x	x	√	x	x	x	x	x	x	x	x	x
Simuliidae	5	√	√	x	x	x	x	x	x	x	x	x	x	√
Tipulidae	5	x	x	√	x	x	x	x	x	x	x	x	x	x
Ancylidae	6	x	x	x	x	x	x	x	√	x	x	x	x	x
Lymnaeidae	3	x	x	x	x	x	x	x	x	x	x	x	x	√
Physidae	3	√	x	x	√	√	x	x	x	√	x	√	√	√
Planorbidae	3	x	x	x	x	x	x	x	√	x	x	x	x	x

Amphipoda, which is accorded a sensitivity score of 13 in the SASS5 methodology, was the highest scoring (i.e. least pollution tolerant) taxa recorded at this site, using the criteria of Dickens and Graham (2001).

### 5.1.2 Seasonal abundance of macroinvertebrates

The means and standard deviations for abundance of macroinvertebrates (total number of individual macroinvertebrates of all taxa sampled) sampled in the Keyzers River during each season, are depicted in Table 5.2.

**Table 5.2:** Total number of macroinvertebrates (abundance) sampled per season, at each site (KVS1, KVS2 and KVS3) in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	512.00	206.00	30.00	230.00
KVS2	232.00	1132.00	140.00	140.00
KVS3	13.00	333.00	142.00	181.00
<b>Mean</b>	252.33	557.00	104.00	183.67
<b>Standard deviation</b>	250.12	502.00	64.09	45.06

There is an apparent interaction present and thus two one-way ANOVA's were done. The Box-Cox computation suggested a square root transformation in the form:  $y=1/\sqrt{x}$ . This stabilised the variance well and the F-test for *season* yielded  $F(3, 8) = 1.53$  with  $p_{\text{season}} = 0.28$ . Thus the available data did not indicate a statistically significant difference in macroinvertebrate abundance between seasons.

In the case of *site*, the calculated value of  $F(2, 9) = 0.61$  with  $p_{\text{site}} 0.56$  did not succeed to show any statistically significant difference in macroinvertebrate abundance between sites in the Keyzers River.

### 5.1.3 Seasonal diversity of macroinvertebrates

The means and standard deviations for diversity of macroinvertebrate taxa (total number of different macroinvertebrate taxa), sampled per season in the Keyzers River, are depicted in Table 5.3.

**Table 5.3:** Total number of different macroinvertebrate taxa (diversity) sampled per season at each site (KVS1; KVS2 and KVS3), in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	10.00	8.00	3.00	5.00
KVS2	7.00	7.00	5.00	5.00
KVS3	4.00	9.00	8.00	9.00
Mean	7.00	8.00	5.33	6.33
Standard deviation	3.00	1.00	2.52	2.31

There is an apparent interaction present and thus two one-way ANOVAs were done. The F-test for *season* yielded  $F(3, 8) = 0.70$  with  $p_{\text{season}} 0.58$ . The ANOVA for *site* gave  $F(2, 9) = 0.42$  with  $p_{\text{site}} 0.67$ . There was no statistically significant difference in macroinvertebrate diversity between seasons or between sites in the Keyzers River.

## 5.2 Water chemistry results

### 5.2.1 Total suspended solids (TSS) (mg/l)

The means and standard deviations for TSS (mg/l), measured in the Keyzers River, for each season are listed in Table 5.4. No water samples were taken at the site KVS2 (middle site in Keyzers River).

**Table 5.4:** Total suspended solids (TSS) (mg/l), measured per season at KVS1 and KVS3, in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	6.00	4.00	4.00	4.00
KVS3	29.00	5.00	2.00	19.00
Mean	17.50	4.50	3.00	11.50
Standard deviation	16.26	0.71	1.41	10.61

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The Box-Cox computation suggested an inverse square root transformation in the form:  $y=1/\text{Sqrt}(x)$ . This stabilised the variance well and the F-test for season yielded  $F(3, 4) = 1.71$  with  $p_{\text{season}} 0.3026$ . Thus the available data did not indicate a statistically significant difference in TSS between seasons.

In the case of site, the calculated value of  $F(1, 6) = 0.48$  with  $p_{\text{site}} 0.5124$  did not succeed to show any statistically significant difference in TSS between sites in the Keyzers River.

### 5.2.2 pH

The means and standard deviations for pH in the Keyzers River for each season are listed in Table 5.5. No water samples were taken at the site KVS2 (middle site in Keyzers River).

**Table 5.5:** pH measured per season at KVS1 and KVS3 in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	7.80	7.40	6.90	7.00
KVS3	7.10	7.20	6.80	6.90
Mean	7.45	7.30	6.85	6.95
Standard deviation	0.49	0.14	0.07	0.07

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 3.91$  and  $p_{\text{season}} 0.1462$  which did not indicate significance at the 5% level in the Keyzers River.

The calculated value of  $F(1, 3) = 3.67$  with  $p_{\text{site}} 0.1514$  did not confirm a statistically significant difference in pH between sites in the Keyzers River.

### 5.2.3 Conductivity (mS/m)

The means and standard deviations for water conductivity in the Keyzers River, for each season, are listed in Table 5.6. No water samples were taken at the site KVS2 (middle site in Keyzers River).

**Table 5.6:** Water conductivity (mS/m), measured per season at KVS1 and KVS3 in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	78.00	77.00	48.00	58.00
KVS3	83.00	64.00	52.00	63.00
<b>Mean</b>	80.50	70.50	50.00	60.50
<b>Standard deviation</b>	3.54	9.19	2.83	3.54

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 8.78$  with  $p_{\text{season}} = 0.053$ , which while it is close to significant, it did not indicate significance at the 5% level in the Keyzers River.

The calculated value of  $F(1, 3) = 0.00$  with  $p_{\text{site}} = 0.9585$  did not confirm a statistically significant difference in water conductivity between sites in the Keyzers River.

#### 5.2.4 Chemical oxygen demand (COD) (mg/l)

The means and standard deviations of COD (mg/l) in the Keyzers River, for each season, are listed in Table 5.7. No water samples were taken at the site KVS2 (middle site in Keyzers River).

**Table 5.7:** Chemical oxygen demand (COD) (mg/l), measured per season at KVS1 and KVS3 in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	43.00	23.00	41.00	29.00
KVS3	29.00	24.00	40.00	43.00
<b>Mean</b>	36.00	23.50	40.50	36.00
<b>Standard deviation</b>	9.90	0.71	0.71	9.90

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The F-test for *season* yielded  $F(3, 4) = 2.17$  with  $p_{\text{season}} = 0.2339$ . The ANOVA for *site* gave  $F(1, 6) = 0$  with  $p_{\text{site}} = 1.00$ . There were no statistically significant differences between seasons, or between sites for COD in the Keyzers River.

### 5.2.5 Total oxidisable nitrogen (TON) (mg/l N)

The means and standard deviations for TON (mg/l N) in the Keyzers River, for each season, are listed in Table 5.8. No water samples were taken at the site KVS2 (middle site in Keyzers River).

**Table 5.8:** Total oxidisable nitrogen (TON) (mg/l N), measured per season at KVS1 and KVS3 in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	0.58	0.65	1.02	2.31
KVS3	0.68	0.61	1.03	3.28
Mean	0.63	0.63	1.03	2.80
Standard deviation	0.07	0.03	0.01	0.69

The Box-Cox computation suggested an inverse square root transformation in the form:  $y=1/\text{sqrt}(x)$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 3) = 74.77$  with  $p_{\text{season}} 0.0026$ . There was a significant difference in total oxidisable nitrogen between seasons in the Keyzers River. The Tukey HSD test for detecting pairwise differences showed summer and autumn to be homogenous and significantly different to winter and spring. Winter was significantly different to all seasons as was spring. Spring had the highest mean total oxidisable nitrogen compared to all other seasons in the Keyzers River.

In the case of site, the calculated value of  $F(1, 3) = 1.41$  with  $p_{\text{site}} 0.3203$  did not succeed to show any statistically significant difference in total oxidisable nitrogen between sites in the Keyzers River.

### 5.2.6 Total inorganic nitrogen (mg/l N)

The means and standard deviations for total inorganic nitrogen (mg/l N) in the Keyzers River, for each season, are listed in Table 5.9. No water samples were taken at the site KVS2 (middle site in Keyzers River).

**Table 5.9:** Total inorganic nitrogen (mg/l N), measured per season at KVS1 and KVS3 in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	0.08	0.07	0.61	1.24
KVS3	0.16	0.04	0.71	2.38
Mean	0.12	0.05	0.66	1.81
Standard deviation	0.06	0.02	0.07	0.81

The Box-Cox computation suggested a natural log transformation in the form:  $y = \log_e(x+1)$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 3) = 21.17$  with  $p_{\text{season}} 0.016$ . There was a significant difference in total inorganic nitrogen between seasons. The Tukey HSD test for detecting pairwise differences showed summer, winter and autumn to be homogenous, and spring was significantly different to summer and autumn. Spring had the highest mean total inorganic nitrogen compared to all other seasons in the Keyzers River.

In the case of site, the calculated value of  $F(1, 3) = 1.83$  with  $p_{\text{site}} 0.2690$  did not succeed to show any statistically significant difference in total inorganic nitrogen between sites in the Keyzers River.

### 5.2.7 Total ammonia (mg/l N)

The means and standard deviations for total ammonia (mg/l N) measured as  $\text{NH}_4\text{-N}$  in the Keyzers River, for each season, are listed in Table 5.10. No water samples were taken at the site KVS2 (middle site in Keyzers River).

**Table 5.10:** Total ammonia (mg/l N) measured per season at KVS1 and KVS3 in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	0.05	0.05	0.11	0.93
KVS3	0.15	0.01	0.13	1.92
Mean	0.10	0.03	0.12	1.43
Standard deviation	0.07	0.03	0.02	0.70

The Box-Cox computation suggested a natural log transformation in the form:  $y = \log_e(x+1)$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 3) = 15.29$  with  $p_{\text{season}} 0.0253$ . There was a significant

difference in total ammonia between seasons. The Tukey HSD test for detecting pairwise differences showed spring was significantly higher compared to all other seasons in the Keyzers River, while summer, winter and autumn were homogenous.

In the case of site, the calculated value of  $F(1, 3) = 1.44$  with  $p_{\text{site}} 0.3164$  did not succeed to show any statistically significant difference in total ammonia between sites in the Keyzers River.

Calculated un-ionised ammonia ( $\text{NH}_3$ ) concentrations coded as to whether they fell within chronic or acute toxicity ranges are highlighted in Table 5.11.

**Table 5.11:** Total ammonia (measured as  $\text{NH}_4\text{-N}$ ) and pH measured during each season at each site within the Keyzers River (KVS). Un-ionised ammonia ( $\text{NH}_3$ ) component calculated for measured pH and at temperatures of  $20^\circ\text{C}$  (summer and autumn estimate) and  $15^\circ\text{C}$  (spring and winter estimate), using pH and temperature dependent conversions supplied DWAf (1996).

Season	River	Site	Soluble (Total) Ammonia (mg/l N)	Temperature	pH	Un-ionised $\text{NH}_3$ (mg/l N)
Sum	Keyzers River	KVS1	0.050	20.00	7.80	0.001
		KVS3	0.146	20.00	7.10	0.002
Aut		KVS1	0.049	20.00	7.40	0.001
		KVS3	0.011	20.00	7.20	0.000
Win		KVS1	0.110	15.00	6.90	0.001
		KVS3	0.132	15.00	6.80	0.008
Spr		KVS1	0.930	15.00	7.00	0.016
		KVS3	1.920	15.00	6.90	0.004

Sum = summer, Aut = autumn, Win = winter and Spr = spring. Acute ammonia toxicity values are highlighted in **dark grey** and chronic ammonia toxicity values in **light grey**. No highlight indicates values were less than the acute or chronic toxicity thresholds as defined by DWAf (1996).

Based on the above, acute and chronic un-ionised ammonia ( $\text{NH}_3$ ) concentrations were calculated for each site to provide an indication of potential ammonia toxicity at the time of sampling. Both concentrations of total ammonia and un-ionised ammonia at all sites during all seasons in the Keyzers River were within DWAf (1996) target water quality range for this variable (Appendix A, Table A1).

### 5.2.8 Soluble nitrite and nitrate (mg/l N)

The means and standard deviations for soluble nitrite and nitrate (mg/l N) in the Keyzers River, for each season, are listed in Table 5.12. No water samples were taken at the site KVS2 (middle site in Keyzers River).

**Table 5.12:** Soluble nitrite and nitrate (mg/l N) measured per season at KVS1 and KVS3 in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	0.00	0.02	0.50	0.31
KVS3	0.01	0.03	0.58	0.46
Mean	0.00	0.02	0.54	0.39
Standard deviation	0.00	0.01	0.06	0.11

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 51.15$  with  $p_{\text{season}} = 0.0045$ . There was a significant difference in soluble nitrite and nitrate between seasons. The Tukey HSD test for detecting pairwise differences showed spring and winter were homogenous as was summer and autumn. Therefore spring and winter were significantly different to summer and autumn.

The calculated value of  $F(1, 3) = 2.41$  with  $p_{\text{site}} = 0.2181$  did not confirm a statistically significant difference in Soluble nitrite and nitrate between sites in the Keyzers River.

### 5.2.9 Total phosphorus (mg/l P)

The means and standard deviations for total phosphorus (mg/l P) in the Keyzers River, for each season, are listed in Table 5.13. No water samples were taken at the site KVS2 (middle site in Keyzers River).

**Table 5.13:** Total phosphorus (mg/l P) measured per season at KVS1 and KVS3 in the Keyzers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	0.17	0.06	0.12	0.24
KVS3	0.46	0.05	0.13	0.27
Mean	0.31	0.06	0.13	0.25
Standard deviation	0.21	0.01	0.01	0.02

The Box-Cox computation suggested an inverse transformation in the form of:  $y=1/x$ . This stabilised the variance well and the two-way additive model ANOVA for season gave a calculated value of  $F(3, 3) = 23.69$  with  $p_{\text{season}} 0.0137$ . There was a significant difference in total phosphorus between seasons in the Keysers River. The Tukey HSD test for detecting pairwise differences showed spring and winter were homogenous as was summer and autumn. Therefore spring and winter were significantly different to summer and autumn.

The calculated value of  $F(1, 3) = 0.10$  with  $p_{\text{site}} 0.7768$  did not confirm a statistically significant difference in total phosphorus between sites in the Keysers River.

### 5.2.10 Orthophosphates (mg/l P)

The means and standard deviations for orthophosphates (mg/l P) in the Keysers River, for each season, are listed in Table 5.14. No water samples were taken at the site KVS2 (middle site in Keysers River).

**Table 5.14:** Orthophosphates (mg/l P) measured per season at KVS1 and KVS3 in the Keysers River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
KVS1	0.03	0.02	0.04	0.12
KVS3	0.14	0.03	0.02	0.10
Mean	0.08	0.02	0.03	0.11
Standard deviation	0.08	0.00	0.01	0.01

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The F-test for *season* yielded  $F(3, 4) = 2.02$  with  $p_{\text{season}} 0.2538$ . The ANOVA for *site* gave  $F(1, 6) = 0.35$  with  $p_{\text{site}} 0.5773$ . There were no statistically significant differences between seasons, or between sites for orthophosphates in the Keysers River.

## 5.3 Diatom results

### 5.3.1 Diatom species sampled

Table 5.15 lists the diatom species sampled in the Keysers River during summer and winter at KVS1 and KVS3. Samples were not taken in autumn and spring.

**Table 5.15:** Diatom species sampled during summer and winter at KVS1 and KVS3 within the Keyzers River. × = absence of diatom species; √ = presence of diatom species.

TAXA	WINTER		SUMMER	
	KVS1	KVS3	KVS1	KVS3
<i>Achnantheidium minutissimum</i>	√	√	√	×
<i>Achnantheidium saprophila</i>	√	×	√	√
<i>Caloneis</i> sp.	√	×	×	×
<i>Cymatopleura solea</i>	×	√	×	×
<i>Cymatopleura solea</i>	×	×	×	×
<i>Cyclotella stelligera</i>	√	√	×	×
<i>Diadesmis confervacea</i>	√	×	×	×
<i>Eunotia flexuosa</i>	×	√	×	×
<i>Eunotia minor</i>	√	×	×	×
<i>Eunotia pectinalis</i>	×	×	×	×
<i>Fragilaria biceps</i>	√	√	×	×
<i>Fragilaria bidens</i>	×	√	×	×
<i>Fistulifera saprophila</i>	×	×	√	×
<i>Fragilaria ulna</i> var. <i>acus</i>	√	×	√	√
<i>Frustulia vulgaris</i>	√	×	×	×
<i>Gomphonema acuminatum</i>	√	√	×	×
<i>Gomphonema affine</i>	×	×	×	√
<i>Gomphonema capitatum</i>	×	√	×	×
<i>Gomphonema insigne</i>	×	×	×	×
<i>Gomphonema laticollum</i>	√	×	×	×
<i>Gomphonema pseudoaugur</i>	×	×	×	√
<i>Hantzschia amphioxys</i>	√	×	×	×
<i>Hippodonta avittata</i>	×	×	×	×
<i>Lemnicola hungarica</i>	×	√	√	√
<i>Mayamaea atomus</i> var. <i>permitis</i>	√	√	×	×
<i>Nitzschia acidoclinata</i>	√	×	×	×
<i>Navicula</i> sp.	√	×	×	×
<i>Nitzschia capitellata</i>	×	×	√	√

<i>Navicula cryptocephala</i>	√	√	√	√
<i>Nitzschia dissipata</i>	x	√	x	x
<i>Navicula erifuga</i>	x	x	x	x
<i>Nitzschia filiformis</i>	x	√	x	x
<i>Navicula germainii</i>	√	√	x	√
<i>Navicula gregaria</i>	√	√	√	√
<i>Nitzschia archibaldii</i>	x	√	x	x
<i>Nitzschia liebetruthii</i>	x	x	√	x
<i>Nitzschia linearis</i>	√	√	x	x
<i>Nitzschia paleacea</i>	x	√	x	x
<i>Nitzschia recta</i>	x	√	x	x
<i>Navicula rhyngocephala</i>	√	√	x	x
<i>Navicula riediana</i>	√	x	x	x
√ <i>Navicula rostellata</i>	√	√	x	x
<i>Navicula salinarum</i>	√	√	x	x
<i>Nitzschia sigma</i>	√	√	x	x
<i>Navicula symmetrica</i>	√	√	x	x
<i>Planothidium dau</i>	√	√	x	√
<i>Placoneis dicephala</i>	√	√	x	√
<i>Planothidium engelbrechtii</i>	√	√	√	x
<i>Psammothidium oblongellum</i>	√	x	x	√
<i>Placoneis placentula</i>	x	x	x	x
<i>Pleurosigma salinarum</i>	x	x	x	√
<i>Pinnularia subbrevistriata</i>	x	x	√	x
<i>Planothidium delicatulum</i>	√	x	x	x
<i>Planothidium lanceolatum</i>	x	√	x	x
<i>Planothidium rostratum</i>	√	√	x	x
<i>Rhopalodia gibba</i>	x	x	x	x
<i>Surirella angusta</i>	√	x	x	x
<i>Staurosirella pinnata</i>	x	x	x	√
<i>Sellaphora seminulum</i>	√	√	√	√
<i>Tryblionella apiculata</i>	x	√	x	x
<i>Tryblionella debilis</i>	x	x	x	x

<i>Tryblionella granulata</i>	x	√	x	x
<i>Amphora veneta</i>	x	√	x	x
<i>Nitzschia</i> sp.	√	√	√	x
<i>Navicula subrhynchocephala</i>	x	x	x	x
<i>Navicula veneta</i>	√	√	√	x
<i>Tabularia fasciculata</i>	x	√	√	√
<i>Nitzschia umbonata</i>	x	x	x	x
<i>Thalassiosira weissflogii</i>	x	√	x	√
<i>Achnantheidium exiguum</i>	√	x	√	√
<i>Nitzschia intermedia</i>	x	x	√	x
<i>Nitzschia microcephala</i>	√	x	x	x
<i>Ctenophora pulchella</i>	x	√	x	√
<i>Hippodonta capitata</i>	√	√	√	√
<i>Nitzschia amphibia</i>	√	√	√	√
<i>Nitzschia frustulum</i>	√	√	√	√
<i>Cocconeis placentula</i>	√	√	√	√
<i>Eolimna minima</i>	√	√	√	√
<i>Melosira varians</i>	√	√	√	√
<i>Nitzschia supralitorea</i>	√	√	√	x
<i>Aulacoseira ambigua</i>	√	x	x	x
<i>Planothidium frequentissimum</i>	√	x	√	√
<i>Sellaphora pupula</i>	x	x	√	√
<i>Nitzschia palea</i>	√	√	√	√
<i>Gomphonema parvulum</i>	√	√	√	√
<i>Fragilaria construens</i> var. <i>subsalina</i>	x	x	√	x
<i>Cyclotella meneghiniana</i>	√	√	√	√
<i>Staurosira elliptica</i>	√	√	√	√

### 5.3.2 Specific Pollution Sensitivity Index and percentage Pollution Tolerant Valves

Table 5.16 shows the diatom index results, as obtained from the OMNIDIA software package, which provides an indication of the Specific Pollution Index (SPI) for each diatom sample taken, in the Keyzers River.

**Table 5.16:** The population, number of diatom species, Specific Pollution Index (SPI) and the percentage of Pollution Tolerant Valves (PTV), for the assemblages of diatoms sampled at KVS1 and KVS3, in winter and summer. As interpreted by Appendix A, Table 2.

Seasons	Sites	Population	No. Species	Specific Pollution Sensitivity Index (SPI)	%Pollution Tolerant Valves (%PTV)	Mean SPI Score	Mean PTV%
Winter	KVS1	402	47	11.9	22.1	11.5	21.85
Winter	KVS3	403	49	11.1	21.6		
Summer	KVS1	407	31	7	24.6	7.85	19.15
Summer	KVS3	409	31	8.7	13.7		

Winter showed a higher mean SPI score, and had a higher mean percentage of PTV's compared to summer. The SPI score in winter was indicative of moderate quality water, as described in Appendix A, Table A2, and had a higher percentage of pollution tolerant valves than summer. The SPI score in summer was indicative of poorer water quality but had less pollution tolerant valves than winter.

### 5.3.3 Seasonal abundance of diatom species

The square root transformed mean ( $\pm$  standard deviation) abundance of diatoms (total number of individual diatoms), sampled in summer and winter in the Keyzers River, is depicted in Table 5.17.

**Table 5.17:** Total number of diatoms (abundance) sampled in summer and winter at KVS1 and KVS3; (KVS1 is the upper most sampling site, and KVS3 is the lower most sampling site), in the Keyzers River.

SITES	WINTER	SUMMER
KVS1	449.00	438.00
KVS3	452.00	440.00
Mean	450.50	439.00
Standard deviation	2.12	1.41

The values were square root transformed to perform the one-way Analysis of variance (ANOVA) test. The calculated value of F at 1 and 2 df did not exceed the critical value of 39. There was no statistically significant difference ( $F_{39}=18.5$ ,  $P > 0.05$ ) in the abundance of diatoms in the Keyzers River between summer and winter.

The calculated value of  $F$  at 1 and 2 df did not exceed the critical value of 39. There was no statistically significant difference ( $F_{39}=18.5$ ,  $P > 0.05$ ) of abundance of diatoms, between sites in the Keyzers River.

#### 5.3.4 Seasonal diversity of diatom species

The square root transformed mean ( $\pm$  standard deviation) diversity of diatom taxa (total number of different diatom taxa), sampled in summer and winter in the Keyzers River, is depicted in Table 5.18.

**Table 5.18:** Total number of different diatom taxa (diversity) sampled in summer and winter at KVS1 and KVS3, in the Keyzers River.

SITES	WINTER	SUMMER
KVS1	47.00	31.00
KVS3	49.00	49.00
Mean	48.00	40.00
Standard deviation	1.00	9.00

The values were square root transformed to perform the one-way Analysis of variance (ANOVA) test. The calculated value of  $F$  at 1 and 2 df did exceed the critical value of 39. There was no statistically significant difference ( $F_{39}= 18.5$ ,  $P >0.05$ ) in diatom taxa, between winter and summer.

The calculated value of  $F$  at 1 and 2 df did not exceed the critical value of 39. There was no statistically significant difference ( $F_{39}=18.5$ ,  $P > 0.05$ ) of diversity of diatoms, between sites in the Keyzers River.

#### 5.3.5 Ecological indicator values

The ecological indicator values as classified using the Cemagref (1982) index as referred to in Van Dam et al. (1994), for the assemblage of diatoms in winter and summer at KVS1 and KVS3, in the Keyzers River, is depicted in Table 5.19.

**Table 5.19:** Classification of ecological indicators, at site KVS1 and KVS3, in summer and winter, in the Keyzers River, as interpreted by Appendix A, Table 3.

Season	Sites	pH	Salinity	N Uptake Mechanism	Oxygen Requirements	Sabrobity	Trophic State
Winter	KVS1	alkaliphilous	fresh-brackish	Nitrogen autotrophic - tolerant	low	$\beta$ -mesosaprobous	eutrophic
Winter	KVS3	alkaliphilous	fresh-brackish	Nitrogen autotrophic - tolerant	low	$\beta$ -mesosaprobous	eutrophic
Summer	KVS1	alkaliphilous	fresh-brackish	Nitrogen autotrophic - tolerant	low	$\alpha$ -meso-polysaprobous	oligo - eutrophic
Summer	KVS2	alkaliphilous	fresh-brackish	Nitrogen autotrophic - tolerant	low	$\beta$ -mesosaprobous	eutrophic

**CHAPTER 6**  
**RESULTS: LITTLE LOTUS RIVER**

**6.1 Macroinvertebrate results**

**6.1.1 Macroinvertebrates sampled**

Table 6.1 lists the macroinvertebrate taxa sampled in the Little Lotus River during summer, autumn, winter and spring at LL1, LL2 and LL3 (LL1 is the riffle up stream of the culvert at Fifth avenue, LL2 is in marginal vegetation up stream of the culvert at Fifth avenue and LL3 is submerged riffle above the marginal vegetation upstream of the culvert at Fifth Avenue).

**Table 6.1:** Macroinvertebrate taxa sampled during each season at each site (LL1, LL2 and LL3) within the Little Lotus River. × = absence of a taxon; √ = presence of a taxon.

SEASONS	SASS SENSITIVITY	SUMMER			AUTUMN			WINTER			SPRING		
		LL1	LL2	LL3	LL1	LL2	LL3	LL1	LL2	LL3	LL1	LL2	LL3
Planarians	3	×	×	×	√	√	√	×	×	×	×	√	×
Oligocheata	1	×	×	×	×	×	×	×	√	×	√	×	√
Hirudinea	3	×	×	×	×	×	×	×	×	×	×	√	×
Crabs	3	×	×	√	×	×	×	×	×	×	×	×	×
Baetidae	4	×	√	×	×	×	×	×	√	√	√	√	√
Coenagriidae	4	×	√	×	×	√	×	×	×	×	×	×	×
Corduliidae	8	×	×	×	×	√	×	×	×	×	×	×	×
Libellulidae	4	×	×	×	×	×	√	×	√	×	×	√	×
Corixidae	3	×	×	×	√	×	×	×	×	×	×	√	√
Ceratopogonidae	5	×	×	×	√	×	×	×	×	×	√	√	√
Chironomidae	2	×	√	√	√	√	√	×	√	√	×	√	√
Psychodidae	1	×	×	×	×	×	×	×	×	×	×	√	×
Simuliidae	5	×	×	×	×	×	×	×	×	×	×	√	√
Ancylidae	6	×	×	×	×	×	×	×	×	×	×	√	√
Lymnaeidae	3	×	×	×	×	√	×	×	×	×	×	×	×
Physidae	3	×	√	×	√	√	×	×	√	√	√	√	×
Planorbidae	3	×	√	√	√	√	√	×	√	×	×	√	×

Corduliidae, which is accorded a sensitivity score of 8 in the SASS5 methodology, was the highest scoring (i.e. Least pollution tolerant) taxon recorded for this site, using the criteria of Dickens and Graham (2001).

### 6.1.2 Seasonal abundance of macroinvertebrates

The mean and standard deviation abundance of macroinvertebrates (total number of individual macroinvertebrates), sampled per season, in the Little Lotus River, as listed in Table 6.2.

**Table 6.2:** Total number of macroinvertebrates (abundance) sampled per season at each site (LL1, LL2 and LL3), in the Little Lotus River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	-	150.00	-	202.00
LL2	230.00	61.00	141.00	281.00
LL3	111.00	40.00	120.00	1420.00
<b>Mean</b>	113.67	83.67	87.00	634.33
<b>Standard deviation</b>	115.02	58.40	76.07	681.55

There was an apparent interaction present and thus two one-way ANOVA's were done. The Box-Cox computation suggested a square root transformation in the form:  $y=1/\sqrt{x}$ . This stabilised the variance well and the F-test for *Season* yielded  $F(3, 6) = 3.39$  with  $p_{\text{season}} 0.095$ . Thus the available data did not indicate a statistically significant difference in macroinvertebrate abundance between seasons in the Little Lotus River.

In the case of *site*, the calculated value of  $F(2, 7) = 0.11$  with  $p_{\text{site}} 0.89$  did not succeed to show any statistically significant difference in macroinvertebrate abundance between sites in the Little Lotus River.

### 6.1.3 Seasonal diversity of macroinvertebrates

The mean and standard deviation diversity of macroinvertebrate taxa (total number of different macroinvertebrate taxa), sampled per season in the Little Lotus River, is listed in Table 6.3.

**Table 6.3:** Total number of different macroinvertebrate taxa (diversity) sampled per season, at each site (LL1; LL2 and LL3), in the Little Lotus River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	-	6.00	-	4.00
LL2	5.00	7.00	6.00	11.00
LL3	3.00	4.00	3.00	7.00
Mean	2.67	5.67	4.50	7.33
Standard deviation	2.52	1.53	2.12	3.51

The two-way additive model ANOVA for *season* yielded  $F(3, 4) = 2.70$  with  $p_{\text{season}} 0.181$ . The ANOVA for *site* gave  $F(2, 4) = 4.47$  with  $p_{\text{site}} 0.096$ . There was no statistically significant difference in macroinvertebrate diversity between seasons or between sites in the Little Lotus River.

## 6.2 Water chemistry results

Water chemistry was sampled at site LL1 and LL4, which was approximately 100m upstream of the Fifth Avenue Bridge.

### 6.2.1 Total suspended solids (TSS) (mg/l)

The means and standard deviations for TSS (mg/l), measured in the Little Lotus River, for each season are listed in Table 6.4.

**Table 6.4:** Total suspended solids (TSS) (mg/l), measured per season at LL1 and LL4 in the Little Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	22.00	54.00	2.00	3.00
LL4	1.00	9.00	2.00	1.00
Mean	11.50	31.50	2.00	2.00
Standard deviation	14.85	31.82	0.00	1.41

The Box-Cox computation suggested a natural log transformation in the form:  $y = \log_e(x+1)$ . This stabilised the variance well. The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The F-test for *season* yielded  $F(3, 4) = 1.66$  with  $p_{\text{season}} 0.31$ . The ANOVA for *site* gave  $F(1, 6) = 2.32$  with  $p_{\text{site}} 0.1788$ .

There were no statistically significant differences between seasons, or between sites for TSS in the Little Lotus River.

### 6.2.2 pH

The means and standard deviations for pH in the Little Lotus River for each season are listed in Table 6.5.

**Table 6.5:** pH measured per season at LL1 and LL4 in the Little Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	7.70	7.70	7.60	7.10
LL4	7.90	7.90	7.60	7.40
Mean	7.80	7.80	7.60	7.25
Standard deviation	0.14	0.14	0.00	0.21

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 17.00$  with  $p_{\text{season}} = 0.022$ . There was a significant difference in pH between seasons in the Little Lotus River. The Tukey HSD test for detecting pairwise differences showed spring to be significantly different to summer and autumn. Summer and autumn were homogenous, as were summer and winter.

The calculated value of  $F(1, 3) = 7.74$  with  $p_{\text{site}} = 0.069$  did not confirm a statistically significant difference in pH between sites in the Little Lotus River.

### 6.2.3 Conductivity (mS/m)

The means and standard deviations for water conductivity (mS/m) in the Little Lotus River, for each season, are listed in Table 6.6.

**Table 6.6** Water conductivity (mS/m), measured per season at LL1 and LL4 in the Little Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	163.00	81.00	75.00	76.00
LL4	163.00	81.00	74.00	75.00
Mean	163.00	81.00	74.50	75.50
Standard deviation	0.00	0.00	0.71	0.71

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 22286.00$  with  $p_{\text{season}} < 0.0001$ . There was a significant difference in conductivity between seasons in the Little Lotus River. The Tukey HSD test for detecting pairwise differences showed summer to have the highest mean conductivity and to be significantly different to all other seasons. Autumn had the lowest mean conductivity and was also significantly different to all other seasons. Winter and spring were homogenous.

The calculated value of  $F(1, 3) = 3$  with  $p_{\text{site}} 0.18$  did not confirm a statistically significant difference in conductivity between sites in the Little lotus River.

#### 6.2.4 Chemical oxygen demand (COD) (mg/l)

The means and standard deviations of COD (mg/l) in the Little Lotus River, for each season, are depicted in Table 6.7.

**Table 6.7:** Chemical oxygen demand (COD) (mg/l), measured per season at LL1 and LL4 in the Little Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	46.00	26.00	71.00	46.00
LL4	51.00	12.00	68.00	51.00
<b>Mean</b>	48.50	19.00	69.50	48.50
<b>Standard deviation</b>	3.54	9.90	2.12	3.54

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 21.31$  with  $p_{\text{season}} 0.016$ . There was a significant difference in chemical oxygen demand between seasons in the Little Lotus River. The Tukey HSD test for detecting pairwise differences showed winter to be significantly different to autumn. Winter had the highest mean COD while autumn had the lowest mean COD in the Little Lotus River. Autumn, spring and summer were homogenous, as was winter, summer and spring.

The calculated value of  $F(1, 3) = 0.15$  with  $p_{\text{site}} 0.72$  did not confirm a statistically significant difference in chemical oxygen demand between sites in the Little lotus River.

### 6.2.5 Total oxidisable nitrogen (TON) (mg/l N)

The means and standard deviations for TON (mg/l N) in the Little Lotus River, for each season, are listed in Table 6.8.

**Table 6.8:** Total oxidisable nitrogen (TON) (mg/l N), measured per season at LL1 and LL4 in the Little Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	2.58	0.74	3.87	2.72
LL4	5.45	0.90	3.64	2.86
Mean	4.01	0.82	3.76	2.79
Standard deviation	2.03	0.11	0.16	0.10

The Box-Cox computation suggested an inverse square root transformation in the form:  $y=1/\text{Sqrt}(x)$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 3) = 23.35$  with  $p_{\text{season}} 0.014$ . There was a significant difference in total oxidisable nitrogen between seasons. The Tukey HSD test for detecting pairwise differences showed autumn to have the highest mean TON compared to all other seasons and to be significantly different to all other seasons in the Little Lotus River.

In the case of site, the calculated value of  $F(1, 3) = 1.91$  with  $p_{\text{site}} 0.26$  did not succeed to show any statistically significant difference in total oxidisable nitrogen between sites in the Little Lotus River.

### 6.2.6 Total inorganic nitrogen (mg/l N)

The means and standard deviations for total inorganic nitrogen (mg/l N) in the Little Lotus River, for each season, are listed in Table 6.9.

**Table 6.9:** Total inorganic nitrogen (mg/l N), measured per season at LL1 and LL4 in the Little Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	1.62	0.10	2.90	1.84
LL4	3.93	0.13	2.85	1.85
Mean	2.78	0.12	2.88	1.84
Standard deviation	1.63	0.02	0.03	0.00

The Box-Cox computation suggested an inverse square root transformation in the form:  $y=1/\text{Sqrt}(x)$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 3) = 134.62$  with  $p_{\text{season}} 0.001$ . There was a significant difference in total inorganic nitrogen between seasons. The Tukey HSD test for detecting pairwise differences showed autumn to be significantly different to all other seasons in the Little Lotus River. Winter, summer and spring were homogenous.

In the case of site, the calculated value of  $F(1, 3) = 2.57$  with  $p_{\text{site}} 0.21$  did not succeed to show any statistically significant difference in total inorganic nitrogen between sites in the Little Lotus River.

### 6.2.7 Total ammonia (mg/l N)

The means and standard deviations for total ammonia (mg/l N) measured as  $\text{NH}_4\text{-N}$ , in the Little Lotus River, for each season, are listed in Table 6.10.

**Table 6.10:** Total ammonia (mg/l N) measured per season at LL1 and LL4 in the Little Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	1.59	0.04	0.12	0.08
LL4	3.91	0.03	0.12	0.13
<b>Mean</b>	2.75	0.03	0.12	0.10
<b>Standard deviation</b>	1.64	0.00	0.00	0.03

The Box-Cox computation suggested an inverse square root transformation in the form:  $y=1/\text{Sqrt}(x)$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 3) = 124.45$  with  $p_{\text{season}} 0.0012$ . There was a significant difference in total ammonia between seasons in the Little Lotus River. The Tukey HSD test for detecting pairwise differences showed summer to be significantly different to all other seasons, and autumn to be significantly different to winter. Winter and spring were homogenous as were spring and autumn. Summer had the lowest mean total ammonia compared to all seasons in the Little Lotus River.

In the case of site, the calculated value of  $F(1, 3) = 1.89$  with  $p_{\text{site}} 0.26$  did not succeed to show any statistically significant difference in total ammonia between sites in the Little Lotus River.

Calculated un-ionised ammonia ( $\text{NH}_3$ ) concentrations coded as to whether they fell within the DWAF (1996) chronic or acute toxicity ranges, are highlighted in Table 6.11.

**Table 6.11:** Total ammonia (measured as  $\text{NH}_4\text{-N}$ ) and pH measured during each season at each site, within the Little Lotus River (LL). Un-ionised ammonia ( $\text{NH}_3$ ) component calculated for measured pH and at temperatures of  $20^\circ\text{C}$  (summer and autumn estimate) and  $15^\circ\text{C}$  (spring and winter estimate), using pH and temperature dependent conversions supplied DWAF (1996).

Season	River	Site	Soluble (Total) Ammonia (mg/l N)	Temperature	pH	Un-ionised $\text{NH}_3$ (mg/l N)
Sum	Little Lotus River	LL1	1.592	20.00	7.70	0.019
		LL4	3.906	20.00	7.90	0.148
Aut		LL1	0.036	20.00	7.70	0.000
		LL4	0.030	20.00	7.90	0.001
Win		LL1	0.120	15.00	7.60	0.001
		LL4	0.122	15.00	7.60	0.001
Spr		LL1	0.083	15.00	7.10	0.001
		LL4	0.125	15.00	7.40	0.001

Sum = summer, Aut = autumn, Win = winter and Spr = spring. Acute ammonia toxicity values are highlighted in **dark grey** and chronic ammonia toxicity values in **light grey**. No highlight indicates values were less than the acute or chronic toxicity thresholds as defined by DWAF (1996).

Based on the above, acute and chronic un-ionised ammonia ( $\text{NH}_3$ ) values were calculated for each site, to provide an indication of actual or potential ammonia toxicity, at the time of sampling. The concentration of total ammonia in the Little Lotus River at LL1 showed a concentration which fell within the range of chronic un-ionised ammonia toxicity during summer, while LL4 showed a concentration which fell within the range of acute un-ionised ammonia toxicity during summer. All other sites during all seasons were within the DWAF (1996) target water quality range for this variable (Appendix A, Table A1).

### 6.2.8 Soluble nitrite and nitrate (mg/l N)

The means and standard deviations for soluble nitrite and nitrate (mg/l N) in the Little Lotus River, for each season, are listed in Table 6.12.

**Table 6.12:** Soluble nitrite and nitrate (mg/l N) measured per season, at LL1 and LL4 in the Little Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	0.03	0.07	2.78	1.76
LL4	0.02	0.10	2.73	1.72
<b>Mean</b>	0.03	0.08	2.76	1.74
<b>Standard deviation</b>	0.01	0.03	0.04	0.03

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 4535.05$  with  $p_{\text{season}} < 0.0001$ . There was a significant difference in soluble nitrite and nitrate between seasons in the Little Lotus River. The Tukey HSD test for detecting pairwise differences showed winter and spring to be significantly different to all seasons in the Little Lotus River, while summer and autumn were homogenous.

The calculated value of  $F(1, 3) = 0.59$  with  $p_{\text{site}} 0.5$  did not confirm a statistically significant difference in soluble nitrite and nitrate between sites in the Little Lotus River.

### 6.2.9 Total phosphorus (mg/l P)

The means and standard deviations for total phosphorus (mg/l P) in the Little Lotus River, for each season, are listed in Table 6.13.

**Table 6.13:** Total phosphorus (mg/l P) measured per season, at LL1 and LL4, in the Little Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	0.78	0.41	0.14	0.13
LL4	1.04	0.11	0.13	0.16
<b>Mean</b>	0.91	0.26	0.13	0.14
<b>Standard deviation</b>	0.19	0.21	0.00	0.02

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 10.26$  with  $p_{\text{season}} 0.044$ . There was a significant difference in total phosphorus

between seasons in the Little Lotus River. However, the Tukey HSD test was unable to detect any pairwise significant differences.

The calculated value of  $F(1, 3) = 0.001$  with  $p_{\text{site}} 0.98$  did not confirm a statistically significant difference in conductivity between sites in the Little Lotus River.

### 6.2.10 Orthophosphates (mg/l P)

The means and standard deviations for orthophosphates (mg/l P) in the Little Lotus River, for each season, are listed in Table 6.14.

**Table 6.14:** Orthophosphates (mg/l P) measured per season at LL1 and LL4, in the Little Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LL1	0.45	0.06	0.10	0.10
LL4	0.69	0.06	0.09	0.10
Mean	0.57	0.06	0.09	0.10
Standard deviation	0.17	0.00	0.00	0.00

The Box-Cox computation suggested an inverse transformation in the form:  $y = 1/x$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 3) = 53.67$  with  $p_{\text{season}} 0.004$ . There was a significant difference in orthophosphates between seasons in the Little Lotus River. The Tukey HSD test for detecting pairwise differences showed that summer had the lowest mean orthophosphate concentrations in the Little Lotus River, and was significantly different to all other seasons. Spring, winter and autumn were homogenous.

In the case of site, the calculated value of  $F(1, 3) = 1.06$  with  $p_{\text{site}} 0.38$  did not succeed to show any statistically significant difference in orthophosphates between sites in the Little Lotus River.

## 6.3 Diatom results

### 6.3.1 Diatom species sampled

Table 6.15 lists the diatom species sampled in the Little Lotus River, during summer and winter, at LL1 and LL2. Samples were not taken in autumn and spring.

**Table 6.15:** Diatom species sampled during summer and winter, at LL1 and LL2, within the Little Lotus River. × = absence of diatom species; √ = presence of diatom species.

TAXA	WINTER		SUMMER	
	LL1	LL2	LL1	LL2
<i>Achnantheidium minutissimum</i>	√	×	√	×
<i>Achnantheidium saprophila</i>	√	√	×	×
<i>Craticula buderi</i>	√	×	×	×
<i>Diadesmis confervacea</i>	√	√	√	√
<i>Fistulifera saprophila</i>	×	√	×	×
<i>Gomphonema insigne</i>	√	√	√	√
<i>Gomphonema pseudoaugur</i>	×	×	×	√
<i>Hippodonta</i> sp.	√	×	×	×
<i>Mayamaea excelsa</i>	√	√	×	×
<i>Mayamaea atomus</i> var. <i>permitis</i>	√	√	√	×
<i>Nitzschia agnewii</i>	×	√	×	×
<i>Nitzschia communis</i>	×	×	√	×
<i>Nitzschia capitellata</i>	×	√	×	×
<i>Navicula cryptocephala</i>	√	√	×	×
<i>Navicula gregaria</i>	√	√	×	×
<i>Nitzschia commutata</i>	√	×	×	×
<i>Nitzschia pusilla</i>	√	×	×	×
<i>Navicula salinarum</i>	√	×	×	×
<i>Navicula subrotundata</i>	√	×	×	×
<i>Nitzschia subacicularis</i>	√	×	×	×
<i>Navicula vandamii</i>	√	√	×	×
<i>Placoneis dicephala</i>	×	√	×	×
<i>Planothidium engelbrechtii</i>	√	√	√	×
<i>Placoneis placentula</i>	×	×	√	×
<i>Pinnularia subbrevistriata</i>	√	×	×	×
<i>Staurosirella pinnata</i>	×	√	×	×
<i>Sellaphora seminulum</i>	√	×	√	×
<i>Amphora veneta</i>	√	√	×	×

<i>Eolimna subminuscula</i>	√	√	√	×
<i>Nitzschia</i> sp.	√	√	×	×
<i>Navicula veneta</i>	√	√	√	×
<i>Nitzschia umbonata</i>	×	×	√	×
<i>Thalassiosira weissflogii</i>	√	×	×	×
<i>Achnantheidium exiguum</i>	√	√	√	√
<i>Nitzschia intermedia</i>	×	×	√	×
<i>Hippodonta capitata</i>	×	√	√	×
<i>Nitzschia amphibia</i>	√	√	√	√
<i>Nitzschia frustulum</i>	√	√	√	×
<i>Cocconeis placentula</i>	√	√	√	×
<i>Eolimna minima</i>	√	√	√	×
<i>Nitzschia supralitorea</i>	×	×	√	√
<i>Planothidium frequentissimum</i>	√	√	√	×
<i>Sellaphora pupula</i>	√	√	√	×
<i>Nitzschia palea</i>	√	√	√	√
<i>Gomphonema parvulum</i>	√	√	√	√
<i>Cyclotella meneghiniana</i>	√	√	√	√
<i>Staurosira elliptica</i>	√	√	√	√
<i>Achnantheidium minutissimum</i>	√	×	√	×
<i>Achnantheidium saprophila</i>	√	√	×	×
<i>Craticula buderi</i>	√	×	×	×
<i>Diadismis confervacea</i>	√	√	√	√
<i>Fistulifera saprophila</i>	×	√	×	×
<i>Gomphonema insigne</i>	√	√	√	√
<i>Gomphonema pseudoaugur</i>	×	×	×	√
<i>Hippodonta</i> sp.	√	×	×	×
<i>Mayamaea excelsa</i>	√	√	×	×
<i>Mayamaea atomus</i> var. <i>permitis</i>	√	√	√	×
<i>Nitzschia agnewii</i>	×	√	×	×
<i>Nitzschia communis</i>	×	×	√	×
<i>Nitzschia capitellata</i>	×	√	×	×
<i>Navicula cryptocephala</i>	√	√	×	×

<i>Navicula gregaria</i>	√	√	x	x
<i>Nitzschia commutata</i>	√	x	x	x
<i>Nitzschia pusilla</i>	√	x	x	x
<i>Navicula salinarum</i>	√	x	x	x
<i>Navicula subrotundata</i>	√	x	x	x
<i>Nitzschia subacicularis</i>	√	x	x	x
<i>Navicula vandamii</i>	√	√	x	x
<i>Placoneis dicephala</i>	x	√	x	x
<i>Planothidium engelbrechtii</i>	√	√	√	x
<i>Placoneis placentula</i>	x	x	√	x
<i>Pinnularia subbrevistriata</i>	√	x	x	x
<i>Staurosirella pinnata</i>	x	√	x	x
<i>Sellaphora seminulum</i>	x	x	√	x
<i>Amphora veneta</i>	x	x	x	x
<i>Eolimna subminuscula</i>	x	x	√	x
<i>Nitzschia sp.</i>	x	x	x	x
<i>Navicula veneta</i>	x	x	√	x
<i>Nitzschia umbonata</i>	x	x	√	x
<i>Thalassiosira weissflogii</i>	√	x	x	x
<i>Achnantheidium exiguum</i>	√	√	√	√
<i>Nitzschia intermedia</i>	x	x	√	x
<i>Hippodonta capitata</i>	x	√	√	x
<i>Nitzschia amphibia</i>	√	√	√	√
<i>Nitzschia frustulum</i>	√	√	√	x
<i>Cocconeis placentula</i>	√	√	√	x
<i>Eolimna minima</i>	√	√	√	x
<i>Nitzschia supralitorea</i>	x	x	√	√

### 6.3.2 Specific Pollution Sensitivity Index and percentage Pollution Tolerant Valves

Table 6.16 shows the diatom index results, as obtained from the OMNIDIA software package, which provides an indication of the specific pollution index for each diatom sample taken at LL1 and LL2 in the Little lotus River.

**Table 6.16:** The population, number of diatom species, Specific Pollution Index (SPI) and the percentage of Pollution Tolerant Valves (PTV), for the assemblages of diatoms sampled at LL1 and LL2, in winter and summer, in the Little Lotus River, as interpreted by Appendix A, Table A2.

Seasons	Sites	Population	No. Species	Specific Pollution Sensitivity Index (SPI)	%Pollution Tolerant Valves (%PTV)	Mean SPI Score	Mean PTV%
Winter	LL1	418	35	8.3	21.3	8.9	15.95
Winter	LL2	415	33	9.5	10.6		
Summer	LL1	419	25	8.9	9.3	9.65	11.9
Summer	LL2	414	10	10.4	14.5		

Winter showed a lower mean SPI score and had a higher percentage of PTV's compared to summer. The mean SPI score in winter was indicative of Poor water quality, while the mean SPI score in summer was indicative of moderate water quality. Summer had less PTV's than winter.

### 6.3.3 Seasonal abundance of diatom species

The mean ( $\pm$  standard deviation) abundance, square root transformed, of diatoms (total number of individual diatoms) sampled in summer and winter in the Little Lotus River, is depicted in Table 6.17.

**Table 6.17:** Total number of diatoms (abundance) sampled in summer and winter, at LL1 and LL2, in the Little Lotus River.

SITES	WINTER	SUMMER
LL1	453.00	444.00
LL2	445.00	424.00
<b>Mean</b>	449.00	434.00
<b>Standard deviation</b>	5.66	14.14

The values were square root transformed to perform the one-way Analysis of Variance (ANOVA) test. The calculated value of F at 1 and 2 df did not exceed the critical value of 39. There was no significant difference ( $F_{39}=18.5$ ,  $P > 0.05$ ) in the number of diatoms in the Little Lotus River between summer and winter.

The calculated value of  $F$  at 1 and 2 df did not exceed the critical value of 39. There was no significant difference ( $F_{39}=18.5$ ,  $P > 0.05$ ) in the abundance of diatoms between sites in the Little Lotus River.

#### 6.3.4 Seasonal diversity of diatom species

The mean ( $\pm$  standard deviation) diversity, square root transformed, of diatom taxa (total number of different diatom taxa), sampled in summer and winter in the Little Lotus River, is depicted in Table 6.18.

**Table 6.18:** Total number of different diatom taxa (diversity) sampled in summer and winter, at LL1 and LL2, in the Little Lotus River.

SITES	WINTER	SUMMER
LL1	35.00	25.00
LL2	30.00	10.00
Mean	32.50	17.50
Standard deviation	3.54	10.61

The values were square root transformed to perform the one-way Analysis of Variance (ANOVA) test. The calculated value of  $F$  at 1 and 2 df did not exceed the critical value of 39. There is no statistically significant difference ( $F_{39}= 18.5$ ,  $P > 0.05$ ) in diatom taxa between winter and summer.

The calculated value of  $F$  at 1 and 2 df did not exceed the critical value of 39. There was no significant difference ( $F_{39}=18.5$ ,  $P > 0.05$ ) in the diversity of diatoms between sites in the Little Lotus River.

#### 6.3.5 Ecological indicator values

The ecological indicator values index as classified by Cemagraf (1982) and referred to by Van Dam (1994), for the assemblage of diatoms in winter and summer at LL1 and LL2, in the Little Lotus River, is depicted in Table 6.19.

**Table 6.19:** The classification of ecological indicators at site LL1 and LL2 in summer and winter, in the Little Lotus River, as interpreted by Appendix A, Table A3.

Season	Sites	pH	Salinity	N Uptake Mechanism	Oxygen Requirements	Sabrobity	Trophic State
Winter	LL1	alkaliphilous	fresh-brackish	Nitrogen autotrophic - sensitive	continuously high	$\beta$ -mesosaprobous	meso - eutrophic
Winter	LL2	alkaliphilous	fresh-brackish	Nitrogen autotrophic - sensitive	continuously high	$\beta$ -mesosaprobous	meso - eutrophic
Summer	LL1	alkaliphilous	fresh-brackish	Nitrogen autotrophic - sensitive	continuously high	$\beta$ -mesosaprobous	meso - eutrophic
Summer	LL2	alkaliphilous	fresh-brackish	Nitrogen autotrophic - sensitive	continuously high	$\beta$ -mesosaprobous	meso - eutrophic

**CHAPTER 7**  
**RESULTS: LANGEVLEI CANAL**

**7.1 Macroinvertebrate results**

**7.1.1 Macroinvertebrates sampled**

Table 7.1 lists the macroinvertebrate taxa sampled in the Langevlei Canal during summer, autumn, winter and spring at LV1 and LV3 (LV1 is the start of the diversion channel and LV3 is the end of the diversion channel before re-entering the canal).

**Table 7.1:** Macroinvertebrate taxa sampled during each season at each site (LV1 and LV3) within the Langevlei Canal. x = absence of a taxon; √ = presence of a taxon; - = not sampled.

SEASON TAXON	SASS SENSITIVITY SCORE	SUMMER		AUTUMN		WINTER		SPRING	
		LV1	LV3	LV1	LV3	LV1	LV3	LV1	LV3
Oligocheata	1	-	-	x	x	x	x	√	√
Hirudinea	3	-	-	√	x	√	√	√	√
Crabs	3	-	-	x	x	√	x	x	x
Hydrachnellae	8	-	-	x	x	x	x	√	x
Baetidae	4	-	-	x	x	x	x	√	√
Coenagriidae	4	-	-	x	√	x	x	x	x
Zygoptera Juvs	6	-	-	x	x	x	√	x	x
Naucoridae	7	-	-	√	√	√	√	x	x
Corixidae	3	-	-	x	x	x	x	x	√
Notonectidae	3	-	-	√	x	x	x	x	x
Pleidae	4	-	-	√	√	x	x	x	√
Helodidae	12	-	-	x	x	√	√	x	x
Ceratopogonidae	5	-	-	x	√	x	x	x	x
Chironomidae	2	-	-	√	x	x	√	√	x

Helodidae, which is accorded a sensitivity score of 12 in the SASS5 methodology, was the highest scoring (i.e. least pollution tolerant) taxa recorded at this site, using the criteria of Dickens and Graham (2001).

### 7.1.2 Seasonal abundance of macroinvertebrates

The means and standard deviations for abundance of macroinvertebrates (total number of individual macroinvertebrates) sampled in the Langevlei Canal, during each season, is listed in Table 7.2. No macroinvertebrates were sampled in summer as there was no water.

**Table 7.2:** Total number of macroinvertebrates (abundance) sampled per season at each site (LV1 and LV3) in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	-	140.00	113.00	231.00
LV3	-	31.00	212.00	340.00
Mean	-	85.50	162.50	285.50
Standard deviation	-	77.07	70.00	77.07

The two-way additive model ANOVA for *season* gave a calculated value of  $F(2, 2) = 2.69$  and  $p_{\text{season}} 0.27$  which did not indicate significance at the 5% level between seasons for the abundance of macroinvertebrates in the Langevlei Canal.

The calculated value of  $F(1, 2) = 0.22$  with  $p_{\text{site}} 0.69$  did not confirm a statistically significant difference in the abundance of macroinvertebrates between sites in the Langevlei Canal.

### 7.1.3 Seasonal diversity of macroinvertebrates

The means and standard deviations for diversity of macroinvertebrate taxa (total number of different macroinvertebrate taxa), sampled per season in the Langevlei Canal, are shown in Table 7.3. No macroinvertebrates were sampled in summer as there was no flowing water.

**Table 7.3:** Total number of different macroinvertebrate taxa (diversity) sampled per season at each site (LV1 and LV3), in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	-	5.00	5.00	6.00
LV3	-	4.00	5.00	7.00
Mean	-	4.50	5.00	6.50
Standard deviation	-	0.71	0.00	0.71

The two-way additive model ANOVA for *season* gave a calculated value of  $F(2, 2) = 4.33$  and  $p_{\text{season}} = 0.19$  which did not indicate significance at the 5% level between seasons for the diversity of macroinvertebrates in the Langevlei Canal.

The calculated value of  $F(1, 2) = 0.00$  with  $p_{\text{site}} = 1.00$  did not confirm a statistically significant difference in the diversity of macroinvertebrates between sites in the Langevlei Canal.

## 7.2 Water chemistry results

### 7.2.1 Total suspended solids (TSS) (mg/l)

The means and standard deviations for the values of TSS (mg/l), measured in the Langevlei Canal, for each season, are listed in Table 7.4. No water samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.4:** Total suspended solids (TSS) (mg/l), measured per season at LV1 and LV3, in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	13.00	7.00	14.00	9.00
LV3	-	5.00	24.00	10.00
<b>Mean</b>	13.00	6.00	19.00	9.50
<b>Standard deviation</b>	-	1.41	7.07	0.71

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 2) = 3.21$  and  $p_{\text{season}} = 0.25$  which did not indicate significance at the 5% level between seasons for the TSS in the Langevlei Canal.

The calculated value of  $F(1, 2) = 0.69$  with  $p_{\text{site}} = 0.49$  did not confirm a statistically significant difference in TSS between sites in the Langevlei Canal.

### 7.2.2 pH

The means and standard deviations for pH in the Langevlei Canal for each season are listed in Table 7.5. No water samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.5:** The values of pH per season at LV1 and LV3 (LV1 is the start of the diversion channel and LV3 is the end of the diversion channel before re-entering the canal), in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	7.70	7.70	7.00	7.50
LV3	-	7.60	7.10	7.50
Mean	7.70	7.65	7.05	7.50
Standard deviation	-	0.07	0.07	0.00

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 2) = 30.5$  and  $p_{\text{season}} 0.03$ . There was a significant difference in pH between seasons in the Langevlei Canal. The Tukey HSD test for detecting pairwise differences showed winter to be significantly different to summer and autumn. Summer, autumn and spring were homogenous, as was winter and spring.

The calculated value of  $F(1, 2) = 0.00$  with  $p_{\text{site}} 1.00$  did not confirm a statistically significant difference in pH between sites in the Langevlei Canal.

### 7.2.3 Conductivity (mS/m)

The means and standard deviations for water conductivity in the Langevlei Canal, for each season, are listed in Table 7.6. No water samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.6:** Water conductivity (mS/m) measured per season at LV1 and LV3 in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	109.00	100.00	98.00	103.00
LV3	-	116.00	116.00	121.00
Mean	109.00	108.00	107.00	112.00
Standard deviation	-	11.31	12.73	12.73

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 2) = 42.17$  and  $p_{\text{season}} 0.02$ , and for *site* a calculated value of  $F(1, 2) = 676.00$  and  $p_{\text{season}} 0.002$ . There was a significant difference in conductivity between seasons and sites in the Langevlei Canal. Because there were only two sites the Tukey HSD test for detecting pairwise differences indicated the sites were significantly different to each

other. However the Tukey HSD test for detecting pairwise differences could not find a significant difference amongst seasons. This is known as a dissonance that could occur if there is a shortage of data. If one ignores the factor site and a one-way ANOVA for season is done then no significant difference is found. This could be due to both methods using different estimates of residual variance, which could provide a possible explanation.

#### 7.2.4 Chemical oxygen demand (COD) (mg/l)

The means and standard deviations of COD (mg/l) in the Langevlei Canal, for each season, are listed in Table 7.7. No water samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.7:** Chemical oxygen demand (COD) (mg/l) measured per season, at LV1 and LV3 in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	32.00	34.00	35.00	29.00
LV3	-	24.00	41.00	22.00
<b>Mean</b>	32.00	29.00	38.00	25.50
<b>Standard deviation</b>	-	7.07	4.24	4.95

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 2) = 1.54$  and  $p_{\text{season}} = 0.42$  which did not indicate significance at the 5% level between seasons for the COD in the Langevlei Canal.

The calculated value of  $F(1, 2) = 0.56$  with  $p_{\text{site}} = 0.53$  did not confirm a statistically significant difference in COD between sites in the Langevlei Canal.

#### 7.2.5 Total oxidisable nitrogen (TON) (mg/l N)

The mean and standard deviation of TON (mg/l N) in the Langevlei Canal, for each season, are listed in Table 7.8. No water samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.8:** Total oxidisable nitrogen (TON) (mg/l N) measured per season at LV1 and LV3 in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	1.21	1.36	3.91	2.79
LV3	-	1.43	3.42	2.54
Mean	1.21	1.40	3.67	2.67
Standard deviation	-	0.04	0.35	0.18

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 2) = 59.37$  and  $p_{\text{season}} 0.017$ . There was a significant difference in TON between seasons in the Langevlei Canal. The Tukey HSD test for detecting pairwise differences showed winter to be significantly different to summer and autumn. Summer, autumn and spring were homogenous, as was winter and spring.

The calculated value of  $F(1, 2) = 2.00$  with  $p_{\text{site}} 0.29$  did not confirm a statistically significant difference in TON between sites in the Langevlei Canal.

### 7.2.6 Total inorganic nitrogen (mg/l N)

The means and standard deviations for total inorganic nitrogen (mg/l N) in the Langevlei Canal, for each season, are listed in Table 7.9.

**Table 7.9:** Total inorganic nitrogen (mg/l N), measured per season at LV1 and LV3 in the Langevlei Canal. - = not sampled

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	0.56	0.73	3.55	2.08
LV3	-	0.62	3.12	1.93
Mean	0.56	0.67	3.34	2.01
Standard deviation	-	0.07	0.30	0.11

The Box-Cox computation suggested a natural log transformation in the form:  $y = \log_e(x+1)$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 2) = 1255.70$  with  $p_{\text{season}} 0.0008$ , and *site* yielded  $F(1, 2) = 22.96$  with  $p_{\text{site}} 0.041$ . There was a significant difference in total inorganic nitrogen between seasons and sites in the Langevlei Canal. As there were only two sites the Tukey HSD test for detecting pairwise differences showed the sites to be significantly different to each other. The Tukey HSD test for detecting pairwise differences

between seasons showed winter and spring to be significantly different to all other seasons, while summer and autumn were homogenous.

### 7.2.7 Total ammonia (mg/l N)

The means and standard deviations of total ammonia (mg/l N) as  $\text{NH}_4\text{-N}$ , in the Langevlei Canal for each season, are listed in Table 7.10. No water samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.10:** Total ammonia (mg/l N) measured per season at LV1 and LV3 in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	0.50	0.39	0.15	0.45
LV3	-	0.18	0.07	0.41
<b>Mean</b>	0.50	0.29	0.11	0.43
<b>Standard deviation</b>	-	0.15	0.05	0.03

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 2) = 10.87$  and  $p_{\text{season}} 0.085$  which did not indicate significance at the 5% level between seasons for total ammonia in the Langevlei Canal.

The calculated value of  $F(1, 2) = 4.53$  with  $p_{\text{site}} 0.16$  did not confirm a statistically significant difference in total ammonia between sites in the Langevlei Canal.

Calculated un-ionised ammonia ( $\text{NH}_3$ ) concentrations coded as to whether they fell within DWAF (1996) chronic or acute toxicity ranges, are highlighted in Table 7.11.

**Table 7.11:** Total ammonia (measured as NH<sub>4</sub>-N) and pH measured during each season at each site within the Langevlei Canal (LV). Un-ionised ammonia (NH<sub>3</sub>) component calculated for measured pH and at temperatures of 20°C (summer and autumn estimate) and 15°C (spring and winter estimate), using pH and temperature dependent conversions supplied DWAF (1996).

Season	River	Site	Soluble (Total) Ammonia (mg/l N)	Temperature	pH	Un-ionised NH <sub>3</sub>
Sum	Langevlei canal	LV1	0.499	20.00	7.70	0.006
Aut		LV1	0.390	20.00	7.70	0.005
		LV2	0.184	20.00	7.60	0.002
Win		LV1	0.148	15.00	7.00	0.001
		LV2	0.072	15.00	7.10	0.001
Spr		LV1	0.450	15.00	7.50	0.004
		LV2	0.410	15.00	7.50	0.003

Sum = summer, Aut = autumn, Win = winter and Spr = spring. Acute ammonia toxicity values are highlighted in **dark grey** and chronic ammonia toxicity values in **light grey**. No highlight indicates values were less than the acute or chronic toxicity thresholds as defined by DWAF (1996).

Based on the above, acute and chronic un-ionised ammonia (NH<sub>3</sub>) values were calculated for each site during each season to provide an indication of actual or potential un-ionised ammonia toxicity at the time of sampling. The concentration of total ammonia and un-ionised ammonia at all sites during all seasons in the Langevlei Canal were low, and were within the DWAF (1996) target water quality range for this variable (Appendix A, Table A1).

### 7.2.8 Soluble nitrite and nitrate (mg/l N)

The means and standard deviations for soluble nitrite and nitrate (mg/l N) in the Langevlei Canal, for each season, are listed in Table 7.12. No water samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.12:** Soluble nitrite and nitrate (mg/l N) measured per season at LV1 and LV3 in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	0.06	0.34	3.40	1.63
LV3	-	0.44	3.05	1.52
Mean	0.06	0.39	3.23	1.58
Standard deviation	-	0.07	0.25	0.08

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 2) = 134.42$  and  $p_{\text{season}} = 0.007$ . There was a significant difference in nitrite and nitrates between seasons in the Langevlei Canal. The Tukey HSD test for detecting pairwise differences showed winter as well as spring to be significantly different all other seasons. Summer and autumn were homogenous.

The calculated value of  $F(1, 2) = 0.83$  with  $p_{\text{site}} = 0.46$  did not confirm a statistically significant difference in nitrite and nitrates between sites in the Langevlei Canal.

### 7.2.9 Total phosphorus (mg/l P)

The means and standard deviations of total phosphorus (mg/l P) in the Langevlei Canal, for each season, are listed in Table 7.13. No water samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.13:** Total phosphorus (mg/l P) measured per season at LV1 and LV3 in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	0.16	0.09	0.09	0.18
LV3	-	0.12	0.11	0.22
<b>Mean</b>	0.16	0.11	0.10	0.20
<b>Standard deviation</b>	-	0.02	0.01	0.03

The Box-Cox computation suggested a natural log transformation in the form:  $y = \log_e(x+1)$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 2) = 148.00$  with  $p_{\text{season}} = 0.0067$ , and *site* yielded  $F(1, 2) = 41.91$  with  $p_{\text{site}} = 0.023$ . There was a significant difference in total phosphorus between seasons and sites in the Langevlei Canal. As there were only two sites the Tukey HSD test for detecting pairwise differences showed the sites to be significantly different to each other. The Tukey HSD test for detecting pairwise differences between seasons showed autumn and winter to be significantly different to summer and spring. Summer was also homogenous with spring.

### 7.2.10 Orthophosphates (mg/l P)

The means and standard deviations of orthophosphates (mg/l P) in the Langevlei Canal, for each season, are listed in Table 7.14. No water samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.14:** Orthophosphates (mg/l N) measured per season at LV1 and LV3 in the Langevlei Canal. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
LV1	0.05	0.04	0.05	0.15
LV3	-	0.08	0.09	0.18
Mean	0.05	0.06	0.07	0.17
Standard deviation	-	0.03	0.02	0.02

The two-way additive model ANOVA for *season* yielded  $F(3, 2) = 215.20$  with  $p_{\text{season}} 0.0046$ , and *site* yielded  $F(1, 2) = 86.08$  with  $p_{\text{site}} 0.011$ . There was a significant difference in orthophosphates between seasons and sites in the Langevlei Canal. As there were only two sites the Tukey HSD test for detecting pairwise differences showed the sites to be significantly different to each other. The Tukey HSD test for detecting pairwise differences between seasons showed spring to be significantly different to all other seasons. Spring had the highest mean orthophosphate concentrations compared to all other seasons.

### 7.3 Diatom results

#### 7.3.1 Diatom species sampled

Table 7.15 lists the diatom taxa sampled in the Langevlei Canal during summer and winter at LV1, LV2 and LV3 (LV1 is the start of the diversion channel, LV2 is slightly up from start of diversion canal and LV3 is the end of the diversion channel before re-entering the canal). No diatom samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer. Samples were not taken in autumn and spring

**Table 7.15:** Diatom species sampled during summer and winter at LV1, LV2 and LV3 within the Langevlei Canal. × = absence of diatom species; √ = presence of diatom species.

TAXA	WINTER			SUMMER	
	LV1	LV2	LV3	LV1	LV2
<i>Achnanthes coarctata</i>	×	×	√	×	×
<i>Achnanthidium saprophila</i>	√	×	√	×	×
<i>Amphora pediculus</i>	√	×	×	√	×

<i>Craticula ambigua</i>	√	√	√	x	x
<i>Diadismis confervacea</i>	x	√	x	x	x
<i>Fragilaria capucina</i> var. <i>vaucheriae</i>	x	x	√	x	x
<i>Gomphonema laticollum</i>	x	√	x	x	x
<i>Gomphonema</i> sp.	√	x	x	x	x
<i>Gomphonema pseudoaugur</i>	x	√	x	x	x
<i>Lemnicola hungarica</i>	√	x	√	x	x
<i>Mayamaea excelsa</i>	√	x	x	x	x
<i>Nitzschia communis</i>	√	√	x	x	x
<i>Navicula cryptocephala</i>	x	√	√	x	x
<i>Navicula gregaria</i>	√	√	√	√	x
<i>Nitzschia paleacea</i>	x	√	x	x	x
<i>Nitzschia recta</i>	x	√	x	x	x
<i>Navicula riediana</i>	x	√	x	x	x
<i>Navicula rostellata</i>	√	x	√	x	x
<i>Planothidium delicatulum</i>	x	x	√	x	x
<i>Sellaphora seminulum</i>	x	x	√	x	x
<i>Amphora veneta</i>	√	√	√	√	√
<i>Eolimna subminuscula</i>	x	√	x	x	√
<i>Nitzschia</i> sp.	√	√	√	x	√
<i>Navicula subrhynchocephala</i>	x	x	x	x	√
<i>Navicula veneta</i>	√	√	√	√	√
<i>Tabularia fasciculata</i>	x	√	√	x	√
<i>Nitzschia umbonata</i>	x	x	√	√	√
<i>Thalassiosira weissflogii</i>	x	x	x	√	√
<i>Achnantheidium exiguum</i>	√	x	√	√	√
<i>Nitzschia intermedia</i>	x	x	x	√	√
<i>Nitzschia microcephala</i>	√	√	√	x	√
<i>Thalassiosira pseudonana</i>	x	x	x	x	√
<i>Ctenophora pulchella</i>	√	√	√	x	√
<i>Hippodonta capitata</i>	√	√	√	√	√
<i>Nitzschia amphibia</i>	√	√	√	√	√
<i>Nitzschia frustulum</i>	√	√	x	√	√

<i>Cocconeis placentula</i>	√	×	√	×	√
<i>Eolimna minima</i>	√	√	√	√	√
<i>Melosira varians</i>	√	√	√	√	√
<i>Nitzschia supralitorea</i>	×	√	×	√	√
<i>Aulacoseira ambigua</i>	√	√	√	√	√
<i>Planothidium frequentissimum</i>	√	√	√	√	√
<i>Sellaphora pupula</i>	√	√	×	√	√
<i>Nitzschia palea</i>	√	√	×	√	√
<i>Gomphonema parvulum</i>	√	√	√	√	√
<i>Nitzschia aurariae</i>	×	×	×	√	√
<i>Fragilaria construens var.subsalina</i>	√	√	√	√	√
<i>Aulacoseira granulata</i>	√	√	√	√	√
<i>Cyclotella meneghiniana</i>	√	√	√	√	√
<i>Staurosira elliptica</i>	√	√	√	√	√

### 7.3.2 Specific Pollution Index and percentage Pollution Tolerant Valves

Table 7.16 shows the diatom index results, as obtained from the OMNIDIA software package, which provides an indication of the Specific Pollution Index (SPI) for each diatom sample taken, in the Langevlei Canal.

**Table 7.16:** The population, number of diatom species, Specific Pollution Index (SPI) and the percentage of Pollution Tolerant Valves (PTV) for the assemblages of diatoms sampled at LV1, LV2 and LV3 in winter, and LV1 and LV2 in summer. No sample was taken at LV3 in summer. This table is to be interpreted as per Appendix A, Table A2.

Season	Sites	Population	No. Species	Specific Pollution Sensitivity Index (SPI)	%Pollution Tolerant Valves (%PTV)	Mean SPI Score	Mean PTV %
Winter	LV1	411	30	9.4	2.9	9.6	3.8
Winter	LV2	408	36	10.1	2.2		
Winter	LV3	437	32	9.3	6.4		
Summer	LV1	419	25	5.7	33.7	6.9	22.6
Summer	LV2	408	30	8.1	11.5		

Winter showed a higher mean SPI score and a lower PTV percentage than summer. The SPI score in winter was indicative of moderate water quality, as described in

Appendix A, Table A2, and had a lower percentage of pollution tolerant valves than summer. The SPI score in summer was indicative of poorer water quality and had more pollution tolerant valves than winter.

### 7.3.3 Seasonal abundance of diatom species

The square root transformed mean ( $\pm$  standard deviation) for abundance of diatoms (total number of individual diatoms), sampled in summer and winter in the Langevlei Canal, is depicted in Table 7.17. No diatom samples were sampled at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.17:** Total number of diatoms (abundance) sampled in summer and winter at LV1, LV2 and LV3 in the Langevlei Canal. - = not sampled.

SITES	WINTER	SUMMER
LV1	441.00	443.00
LV2	469.00	438.00
LV3	438.00	-
<b>Mean</b>	449.33	440.50
<b>Standard deviation</b>	17.10	3.54

The values were square root transformed to perform the one-way Analysis of Variance (ANOVA) test. The calculated value of  $F$  at 1 and 3 df did not exceed the critical value of 39. There was no statistically significant difference ( $F_{39}=10.13$ ,  $P > 0.05$ ) in diatom abundance between summer and winter.

The calculated value of  $F$  at 2 and 2 df did not exceed the critical value of 87.5. There was no statistically significant difference ( $F_{87.5}=19.00$   $P > 0.05$ ) in the abundance of diatoms between sites in the Langevlei Canal.

### 7.3.4 Seasonal diversity of diatom species

The square root transformed mean ( $\pm$  standard deviation) for diversity of diatom taxa (total number of different diatom taxa sampled), sampled in summer and winter in the Langevlei Canal, is depicted in Table 7.18. No diatom samples were taken at the site LV3 (end of the diversion channel into Langevlei Canal) in summer.

**Table 7.18:** Total number of different diatom taxa (diversity) sampled at LV1, LV2 and LV3 in the Langevlei Canal. - = not sampled.

SITES	WINTER	SUMMER
LV1	30.00	24.00
LV2	32.00	30.00
LV3	30.00	-
<b>Mean</b>	30.67	27.00
<b>Standard deviation</b>	1.15	4.24

The values were square root transformed to perform the one-way Analysis of Variance (ANOVA) test. The calculated value of  $F$  at 1 and 3 df did not exceed the critical value of 39. There was no statistically significant difference ( $F_{39} = 10.13$ ,  $P > 0.05$ ) of diatom taxa between winter and summer.

The calculated value of  $F$  at 2 and 2 df did not exceed the critical value of 87.5. There was no statistically significant difference ( $F_{87.5} = 19.00$ ,  $P > 0.05$ ) in the diversity of diatoms between sites in the Langevlei Canal.

### 7.3.5 Ecological Indicators values

The ecological indicator values as classified using the Cemagref (1982) index as quoted by Van Dam et al. (1994), for the assemblage of diatoms in winter and summer at LV1, LV2 and LV3, in the Keysers River, is depicted in Table 7.19.

**Table 7.19:** Classification of ecological indicators at site LV1, LV2 and LV3 in winter and LV1 and LV2 in summer, in the Langevlei Canal, as interpreted by Appendix A, Table A3.

Season	Sites	pH	Salinity	N Uptake Mechanism	Oxygen Requirements	Sabrobity	Trophic State
Winter	LV1	alkaliphilous	fresh-brackish	Nitrogen autotrophic - sensitive	continuously high	β-mesosaprobous	meso - eutrophic
Winter	LV2	alkaliphilous	fresh-brackish	Nitrogen autotrophic - sensitive	continuously high	β-mesosaprobous	meso - eutrophic
Winter	LV3	alkaliphilous	fresh-brackish	Nitrogen autotrophic - sensitive	continuously high	β-mesosaprobous	meso - eutrophic
Summer	LV1	alkaliphilous	fresh-brackish	Nitrogen autotrophic - sensitive	continuously high	β-mesosaprobous	meso - eutrophic
Summer	LV2	alkaliphilous	fresh-brackish	Nitrogen autotrophic - sensitive	continuously high	β-mesosaprobous	meso - eutrophic

**CHAPTER 8**  
**RESULTS: MODDERGAT RIVER**

**8.1 Macroinvertebrate results**

**8.1.1 Macroinvertebrates sampled**

Table 8.1 lists the macroinvertebrate taxa sampled in the Moddergat River during summer, autumn, winter and spring at M2, M3.1, M3.2, M4.1, M4.2 and M5. (M2 is marginal vegetation downstream of the N2, after first riffle and below foot bridge, M3.1 is marginal vegetation downstream of Zandvliet Road, at footbridge after school, M3.2 the riffle downstream of Zandvliet Road, at footbridge after school, M4.1 is marginal vegetation in the lower section before Macassar Road downstream from the riffle, M4.2 is the riffle in the lower section before Macassar Road, M5 is the start of the reed bed downstream of Macassar Road, before it joins with the Kuils River. No macroinvertebrate samples were taken at M1, M6 and M7 (M1 is upstream of N2, M6 is downstream of Macassar Road just before the junction with Kuils River, at the end of the reed bed, M7 is at the Kuils River, above the junction with the Moddergat River).

No samples were taken in summer and autumn as there was no water. No samples were taken at M3.2 and M4.2 in winter as there was too much flow, and in spring at M3.1 as the site was too terrestrial (too much vegetation).



Hydrachnellae, which is accorded a sensitivity score of 8 in the SASS5 methodology, was the highest scoring (i.e. least pollution tolerant) taxon recorded at these sites, using the criteria of .Dickens and Graham (2001).

### 8.1.2 Seasonal abundance of macroinvertebrates

The means and standard deviations for abundance of macroinvertebrates (total number of individual macroinvertebrates sampled per season) sampled in the Moddergat River during each season, is listed in Table 8.2.

**Table 8.2:** Total number of macroinvertebrates (abundance) sampled per season at each site (M2, M3.1, M3.2, M4.1, M4.2 and M5) in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M2	-	-	130.00	210.00
M3.1	-	-	170.00	142.00
M3.2	-	-	-	-
M4.1	-	-	140.00	130.00
M4.2	-	-	-	142.00
M5	-	-	130.00	221.00
Mean	-	-	142.50	169.00
Standard deviation	-	-	18.93	42.91

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between Site and Season. There was an apparent interaction present and thus two one-way ANOVAs were done.

The F-test for *season* yielded  $F(1, 7) = 1.29$  with  $p_{\text{season}} 0.29$ . The ANOVA for *site* gave  $F(4, 4) = 0.28$  with  $p_{\text{site}} 0.87$ . There was no statistically significant difference in macroinvertebrate abundance between seasons or between sites in the Moddergat River.

### 8.1.3 Seasonal diversity of macroinvertebrates

The means and standard deviations for diversity of macroinvertebrate taxa (total number of different macroinvertebrate taxa), sampled per season in the Moddergat River, is listed in Table 8.3. No macroinvertebrate samples were taken in summer and autumn as there was no water. No samples were taken at M3.2 and M4.2 in winter as there was too much flow, and in spring at M3.1 as the site was too terrestrial (too much vegetation).

**Table 8.3:** Total number of different macroinvertebrate taxa (diversity), sampled per season, at each site (M2, M3.1, M3.2, M4.1, M4.2 and M5), in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M2	-	-	4.00	3.00
M3.1	-	-	8.00	7.00
M3.2	-	-	-	-
M4.1	-	-	5.00	7.00
M4.2	-	-	-	4.00
M5	-	-	4.00	5.00
Mean	-	-	5.25	5.20
Standard deviation	-	-	1.89	1.79

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between Site and Season. There was an apparent interaction present and thus two one-way ANOVAs were done.

The F-test for *season* yielded  $F(1, 7) = 0.002$  with  $p_{\text{season}} 0.97$ . The ANOVA for *site* gave  $F(4, 4) = 5.73$  with  $p_{\text{site}} 0.06$ . There was no statistically significant difference in macroinvertebrate diversity between seasons or between sites in the Moddergat River.

## 8.2 Water chemistry results

In summer water samples were only taken at M1 and M5 and in autumn at M1, as there was no flow at the other sites.

### 8.2.1 Total suspended solids (TSS) (mg/l)

The means and standard deviations for the values of TSS (mg/l), measured in the Moddergat River, for each season, are listed in Table 8.4.

**Table 8.4:** Total suspended solids (TSS) (mg/l), measured per season at M1, M2, M3.1, M4.1, M5, M6 and M7, in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M1	17.00	7.00	12.00	9.00
M2	-	-	12.00	21.00
M3.1	-	-	8.00	7.00
M4.1	-	-	6.00	3.00
M5	17.00	-	4.00	4.00
M6	-	-	10.00	14.00
M7	-	-	12.00	5.00
Mean	17.00	7.00	9.14	9.00
Standard deviation	0.00	0.00	3.24	6.45

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between Site and Season. There was an apparent interaction present and thus two one-way ANOVAs were done.

The F-test for *season* yielded  $F(3, 13) = 1.65$  with  $p_{\text{season}} 0.23$ . The ANOVA for *site* gave  $F(6, 10) = 1.24$  with  $p_{\text{site}} 0.36$ . There was no statistically significant difference in TSS between seasons or between sites in the Moddergat River.

### 8.2.2 pH

The means and standard deviations for pH in the Moddergat River for each season are listed in Table 8.5.

**Table 8.5:** pH measured per season at M1, M2, M3.1, M4.1, M5, M6 and M7 in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M1	7.20	7.70	7.50	7.50
M2	-	-	7.40	7.40
M3.1	-	-	7.40	7.40
M4.1	-	-	7.50	7.20
M5	8.00	-	7.50	7.30
M6	-	-	7.50	7.30
M7	-	-	7.40	7.50
Mean	7.60	7.70	7.46	7.37
Standard deviation	0.57	0.00	0.05	0.11

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between Site and Season. There was an apparent interaction present and thus two one-way ANOVAs were done.

The F-test for *season* yielded  $F(3, 13) = 1.59$  with  $p_{\text{season}} 0.24$ . The ANOVA for *site* gave  $F(6, 10) = 0.38$  with  $p_{\text{site}} 0.88$ . There was no statistically significant difference in pH between seasons or between sites in the Moddergat River.

### 8.2.3 Conductivity (mS/m)

The means and standard deviations for water conductivity (mS/m) in the Moddergat River, for each season, are listed in Table 8.6.

**Table 8.6:** Water conductivity (mS/m), measured per season at M1, M2, M3.1, M4.1, M5, M6 and M7 in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M1	83.00	72.00	59.00	64.00
M2	-	-	60.00	64.00
M3.1	-	-	62.00	66.00
M4.1	-	-	63.00	68.00
M5	43.00	-	70.00	76.00
M6	-	-	68.00	76.00
M7	-	-	86.00	108.00
Mean	63.00	72.00	66.86	74.57
Standard deviation	28.28	0.00	9.35	15.61

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 7) = 0.56$  and  $p_{\text{season}} 0.66$  which did not indicate significance at the 5% level between seasons in the Moddergat River.

The calculated value of  $F(6, 7) = 2.06$  with  $p_{\text{site}} 0.18$  did not confirm a statistically significant difference in pH between sites in the Moddergat River.

### 8.2.4 Chemical oxygen demand (COD) (mg/l)

The means and standard deviations of COD (mg/l) in the Moddergat River, for each season, are listed in Table 8.7.

**Table 8.7:** Chemical oxygen demand (COD) (mg/l), measured per season at M1, M2, M3.1, M4.1, M5, M6 and M7, in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M1	37.00	21.00	67.00	28.00
M2	-	-	60.00	31.00
M3.1	-	-	63.00	27.00
M4.1	-	-	43.00	32.00
M5	54.00	-	53.00	28.00
M6	-	-	60.00	29.00
M7	-	-	69.00	33.00
Mean	45.50	21.00	59.29	29.71
Standard deviation	12.02	0.00	8.88	2.29

The Box-Cox computation suggested a power transformation in the form:  $y=(x+1)^{-1.5}$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 7) = 37.42$  with  $p_{\text{season}} 0.0001$ . There was a significant difference in COD between seasons in the Moddergat River. The Tukey HSD test for detecting pairwise differences showed that autumn was significant to all other seasons, as was spring. Summer and winter were also homogenous.

In the case of site, the calculated value of  $F(6, 7) = 0.46$  with  $p_{\text{site}} 0.82$  did not succeed to show any statistically significant difference in COD between sites in the Moddergat River.

### 8.2.5 Total oxidisable nitrogen (TON) (mg/l N)

The means and standard deviations for TON (mg/l N) in the Moddergat River, for each season, are listed in Table 8.8.

**Table 8.8:** Total oxidisable nitrogen (TON) (mg/l N), measured per season at M1, M2, M3.1, M4.1, M5, M6 and M7 in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M1	1.62	0.48	1.87	1.63
M2	-	-	1.79	1.73
M3.1	-	-	1.76	1.18
M4.1	-	-	1.82	6.12
M5	2.85	-	2.02	1.08
M6	-	-	1.76	0.99
M7	-	-	1.83	3.21
<b>Mean</b>	2.24	0.48	1.84	2.28
<b>Standard deviation</b>	0.87	0.00	0.09	1.86

The Box-Cox computation suggested a power transformation in the form:  $y=(x+0.5)^{-1.3}$ . This stabilised the variance well. The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The F-test for *season* yielded  $F(3, 13) = 8.30$  with  $p_{\text{season}} 0.0024$ . There was a significant difference in TON between seasons in the Moddergat River. The Tukey HSD test for detecting pairwise differences showed that autumn was significant to all other seasons.

In the case of site, the calculated value of  $F(6, 10) = 0.67$  with  $p_{\text{site}} 0.67$  did not succeed to show any statistically significant difference in TON between sites in the Moddergat River.

### 8.2.6 Total inorganic nitrogen (mg/l N)

The means and standard deviations for total inorganic nitrogen (mg/l N) in the Moddergat River, for each season, are listed in Table 8.9.

**Table 8.9:** Total inorganic nitrogen (mg/l N), measured per season at M1, M2, M3.1, M4.1, M5, M6 and M7 in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M1	0.84	0.06	1.11	0.76
M2	-	-	1.10	0.91
M3.1	-	-	1.07	0.48
M4.1	-	-	1.01	0.19
M5	2.40	-	1.08	0.24
M6	-	-	1.08	0.32
M7	-	-	1.09	2.35
Mean	1.62	0.06	1.07	0.75
Standard deviation	1.10	-	0.03	0.75

The Box-Cox computation suggested a natural log transformation in the form:  $y = \log_e(x+1)$ . This stabilised the variance well. The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The calculated F value for *season* yielded  $F(3, 13) = 5.49$  with  $p_{\text{season}} 0.012$ . There was a significant difference in total inorganic nitrogen between seasons in the Moddergat River. The Tukey HSD test for detecting pairwise differences showed autumn to be significant to winter and summer. Autumn and spring were homogenous, as were winter, summer and spring.

In the case of site, the calculated value of  $F(6, 10) = 0.50$  with  $p_{\text{site}} 0.80$  did not succeed to show any statistically significant difference in total inorganic nitrogen between sites in the Moddergat River.

### 8.2.7 Total ammonia (mg/l N)

The means and standard deviations for total ammonia (mg/l N) measured as  $\text{NH}_4\text{-N}$ , in the Moddergat River, for each season, are listed in Table 8.10.

**Table 8.10:** Total ammonia (mg/l N) measured per season at M1, M2, M3.1, M4.1, M5, M6 and M7 in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M1	0.58	0.04	0.08	0.14
M2	-	-	0.06	0.24
M3.1	-	-	0.06	0.04
M4.1	-	-	0.06	0.02
M5	0.32	-	0.05	0.02
M6	-	-	0.06	0.06
M7	-	-	0.08	0.05
<b>Mean</b>	0.45	0.04	0.06	0.08
<b>Standard deviation</b>	0.19	0.00	0.01	0.08

The Box-Cox computation suggested a natural log transformation in the form:  $y = \log_e(x+1)$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(3, 7) = 13.76$  with  $p_{\text{season}} 0.002$ . There was a significant difference in total ammonia between seasons in the Moddergat River. The Tukey HSD test for detecting pairwise differences showed summer as having the highest mean concentration of total ammonia and being significantly different to all other seasons in the Moddergat River. Autumn, winter and spring were homogenous.

In the case of site, the calculated value of  $F(6, 7) = 1.92$  with  $p_{\text{site}} 0.21$  did not succeed to show any statistically significant difference in total ammonia between sites in the Moddergat River.

Calculated un-ionised ammonia ( $\text{NH}_3$ ) concentrations, coded as to whether they fell within DWAF (1996) chronic or acute toxicity ranges, are highlighted in Table 8.11.

**Table 8.11:** Total Ammonia (measured as NH<sub>4</sub>-N) and pH measured during each season at each site within the Moddergat River (M). Un-ionised ammonia (NH<sub>3</sub>) component calculated for measured pH and at temperatures of 20°C (summer estimate) and 15°C (spring / winter estimate), using pH and temperature dependent conversions supplied by DWAF (1996).

Season	River	Site	Soluble (Total) Ammonia (mg/l N)	Temperature	pH	Un-ionised NH <sub>3</sub> (mg/l N)	
Sum	Moddergat River	M1	0.582	20.00	7.20	0.007	
		M5	0.318	20.00	8.00	0.012	
Aut		M1	0.035	20.00	7.70	0.004	
Win		M1	0.076	20.00	7.50	0.001	
		M2	0.063	15.00	7.40	0.001	
		M3.2	0.055	15.00	7.40	0.000	
		M3.1	0.055	15.00	7.40	0.000	
		M4.2	0.058	15.00	7.50	0.000	
		M4.1	0.058	15.00	7.50	0.000	
		M5	0.049	15.00	7.50	0.000	
		M6	0.055	15.00	7.50	0.000	
		M7	0.076	15.00	7.40	0.001	
		Spr	M1	0.143	15.00	7.50	0.001
			M2	0.241	15.00	7.40	0.002
M3.2	0.040		15.00	7.40	0.000		
M3.1	0.040		15.00	7.40	0.000		
M4.2	0.023		15.00	7.20	0.000		
M4.1	0.023		15.00	7.20	0.000		
M5	0.023		15.00	7.30	0.000		
	M6	0.055	15.00	7.30	0.000		
	M7	0.045	15.00	7.50	0.000		

Sum = summer, Aut = autumn, Win = winter and Spr = spring. Acute ammonia toxicity values are highlighted in **dark grey** and chronic ammonia toxicity values in **light grey**. No highlight indicates values were less than the acute or chronic toxicity thresholds as defined by DWAF (1996).

Based on the above, acute and chronic un-ionised ammonia (NH<sub>3</sub>) values were calculated for each site to provide an indication of potential ammonia toxicity at the time of sampling. The concentration of total ammonia and un-ionised ammonia at all sites, during all seasons, in the Moddergat River, were within the DWAF (1996) target water quality range for this variable (Appendix A, Table A1).

### 8.2.8 Soluble nitrite and nitrate (mg/l N)

The means and standard deviations for soluble nitrite and nitrate (mg/l N) in the Moddergat River, for each season, are listed in Table 8.12.

**Table 8.12:** Soluble nitrite and nitrate (mg/l N) measured per season at M1, M2, M3.1, M4.1, M5, M6 and M7 in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M1	0.26	0.02	1.03	0.62
M2	-	-	1.04	0.67
M3.1	-	-	1.01	0.44
M4.1	-	-	0.95	0.17
M5	2.08	-	1.03	0.22
M6	-	-	1.02	0.26
M7	-	-	1.01	2.30
Mean	1.17	0.02	1.01	0.67
Standard deviation	1.29	0.00	0.03	0.75

The Box-Cox computation suggested a natural log transformation in the form:  $y = \log_e(x+1)$ . This stabilised the variance well. The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The calculated F value for *season* yielded  $F(3, 13) = 4.30$  with  $p_{\text{season}} 0.026$ . There was a significant difference in nitrite and nitrates between seasons in the Moddergat River. The Tukey HSD test for detecting pairwise differences showed autumn to be significantly different to winter, while spring and summer were homogenous.

In the case of site, the calculated value of  $F(6, 10) = 0.70$  with  $p_{\text{site}} 0.66$  did not succeed to show any statistically significant difference in nitrite and nitrate between sites in the Moddergat River.

### 8.2.9 Total phosphorus (mg/l P)

The means and standard deviations for total phosphorus (mg/l P) in the Moddergat River, for each season, are listed in Table 8.13.

**Table 8.13:** Total phosphorus (mg/l P) measured per season at M1, M2, M3.1, M4.1, M5, M6 and M7 in the Moddergat River. - = not sampled.8-1

SITES	SUMMER	AUTUMN	WINTER	SPRING
M1	0.11	0.06	0.18	0.19
M2	-	-	0.18	0.23
M3.1	-	-	0.17	0.13
M4.1	-	-	0.18	0.10
M5	0.33	-	0.17	0.12
M6	-	-	0.20	0.16
M7	-	-	0.19	0.88
Mean	0.22	0.06	0.18	0.26
Standard deviation	0.16	0.00	0.01	0.28

The Box-Cox computation suggested a power transformation in the form:  $y=(x+0.1)^{-1.3}$ . This stabilised the variance well. The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The F-test for *season* yielded  $F(3, 13) = 2.95$  with  $p_{\text{season}} 0.07$ , while the F- test for *site* yielded  $F(6, 10) = 1.05$  with  $p_{\text{site}} 0.45$ . Neither season nor site succeeded to show any statistically significant difference in total phosphorus in the Moddergat River.

### 8.2.10 Orthophosphates (mg/l P)

The means and standard deviations for orthophosphates (mg/l P) in the Moddergat River, for each season, are listed in Table 8.14.

**Table 8.14:** Orthophosphates (mg/l P) measured per season at M1, M2, M3.1, M4.1, M5, M6 and M7 in the Moddergat River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
M1	0.03	0.03	0.07	0.15
M2	-	-	0.07	0.12
M3.1	-	-	0.06	0.06
M4.1	-	-	0.07	0.04
M5	0.31	-	0.08	0.08
M6	-	-	0.08	0.09
M7	-	-	0.12	0.80
Mean	0.17	0.03	0.07	0.09
Standard deviation	0.20	0.00	0.02	0.27

The Box-Cox computation suggested a inverse square root transformation in the form:  $y=1/\text{Sqrt}(x)$ . This stabilised the variance well. The interaction graph showed a

distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The F-test for *season* yielded  $F(3, 13) = 0.76$  with  $p_{\text{season}} = 0.54$ , while the F-test for *site* yielded  $F(6, 10) = 1.69$  with  $p_{\text{site}} = 0.22$ . Neither season nor site succeeded to show any statistically significant difference in orthophosphates in the Moddergat River.

### 8.3 Diatom results

#### 8.3.1 Diatom species sampled

Table 8.15 lists the diatom species sampled in the Moddergat River during winter at M2, M3.1, M4.1 and M5. No samples were taken in summer as there was no water. Samples were not taken in autumn and spring.

**Table 8.15:** Diatom species sampled during winter at M2, M3.1, M4.1 and M5 within the Moddergat River. × = absence of diatom species; √ = presence of diatom species. Summer samples could not be taken as there was no water present.

TAXA	WINTER			
	M2	M3.1	M4.1	M5
<i>Achnanthes coarctata</i>	×	×	√	×
<i>Achnantheidium minutissimum</i>	√	√	√	√
<i>Achnantheidium saprophila</i>	√	×	×	×
<i>Aulacoseira granulata</i> var. <i>angustissima</i>	√	×	×	×
<i>Bacillaria paradoxa</i>	√	×	×	×
<i>Cyclotella atomus</i>	×	√	×	×
<i>Cocconeis placentula</i> var. <i>euglypta</i>	√	×	×	√
<i>Fragilaria biceps</i>	√	×	×	√
<i>Fallacia monoculata</i>	×	×	√	×
<i>Fragilaria ulna</i> var. <i>acus</i>	√	√	√	√
√ <i>Gomphonema auritum</i>	√	√	√	√
<i>Gomphonema pseudoaugur</i>	√	√	√	√
<i>Gomphonema pumilum</i>	√	√	√	√
<i>Hantzschia amphioxys</i>	×	×	√	×
<i>Lemnicola hungarica</i>	√	×	×	×
<i>Luticola mutica</i>	×	×	√	×
<i>Mayamaea atomus</i> var. <i>permitis</i>	√	√	×	×
<i>Navicula cincta</i>	√	√	√	√

<i>Nitzschia communis</i>	x	x	x	x
<i>Nitzschia capitellata</i>	√	x	x	√
<i>Navicula cryptocephala</i>	√	x	√	√
<i>Nitzschia filiformis</i>	x	x	√	x
<i>Navicula germainii</i>	x	x	√	√
<i>Navicula gregaria</i>	√	√	√	√
<i>Navicula longicephala</i>	x	x	√	x
<i>Nitzschia linearis</i>	√	x	√	√
<i>Nitzschia recta</i>	√	x	x	x
<i>Navicula riediana</i>	√	√	√	√
<i>Navicula rostellata</i>	√	√	√	x
<i>Navicula salinarum</i>	x	x	√	√
<i>Navicula schroeteri</i>	√	x	x	x
<i>Nitzschia sigma</i>	√	x	√	x
<i>Navicula symmetrica</i>	√	√	√	√
<i>Nitzschia tropica</i>	√	x	x	x
<i>Nitzschia linearis</i> var. <i>tenuis</i>	x	x	x	√
<i>Planothidium engelbrechtii</i>	√	√	√	√
<i>Psammothidium oblongellum</i>	x	x	√	x
<i>Pleurosigma salinarum</i>	√	x	x	x
<i>Planothidium rostratum</i>	√	x	x	√
<i>Surirella angusta</i>	√	x	√	x
<i>Surirella brebissonii</i>	x	x	x	√
<i>Sellaphora seminulum</i>	√	x	√	√
<i>Tryblionella calida</i>	x	x	x	√
<i>Tryblionella debilis</i>	√	x	x	√
<i>Tryblionella hungarica</i>	x	x	√	√
<i>Tryblionella levidensis</i>	√	x	x	x
<i>Amphora veneta</i>	√	√	√	√
<i>Eolimna subminuscula</i>	√	x	x	√
<i>Nitzschia</i> sp.	√	√	√	√
<i>Navicula veneta</i>	√	√	√	√
<i>Tabularia fasciculata</i>	x	x	√	x
<i>Thalassiosira weissflogii</i>	x	√	√	x

<i>Ctenophora pulchella</i>	√	×	×	×
<i>Hippodonta capitata</i>	×	√	×	×
<i>Nitzschia amphibia</i>	√	×	×	×
<i>Nitzschia frustulum</i>	√	√	√	√
<i>Cocconeis placentula</i>	√	√	√	√
<i>Eolimna minima</i>	√	√	√	×
<i>Melosira varians</i>	×	√	√	√
<i>Nitzschia supralitorea</i>	√	√	√	×
<i>Planothidium frequentissimum</i>	√	√	√	√
<i>Sellaphora pupula</i>	√	×	×	√
<i>Nitzschia palea</i>	√	×	√	√
<i>Gomphonema parvulum</i>	√	√	√	√
<i>Cyclotella meneghiniana</i>	√	√	√	√
<i>Staurosira elliptica</i>	×	×	×	√

### 8.3.2 Specific Pollution Index and percentage Pollution Tolerant Valves

Table 8.16 shows the diatom index results, as obtained from the OMNIDIA software package, which provides an indication of the Specific Pollution Index (SPI) for each diatom sample taken, in the Moddergat River.

**Table 8.16:** The population, number of diatom species, Specific Pollution Index (SPI) and the percentage of Pollution Tolerant Valves (PTV) for the assemblages of diatoms sampled during winter at M2, M3.1, M4.1 and M5 within the Moddergat River. As interpreted by Appendix A, Table A2.

Season	Sites	Population	No species	Specific Pollution sensitivity Index (SPI)	%Pollution Tolerant Valves (%PTV)	Mean SPI Score	Mean PTV %
Winter	M2	401	46	8.7	13	9.6	11.9
Winter	M3.1	402	24	11.1	6		
Winter	M4.1	403	39	8.3	19.1		
Winter	M5	416	37	10.3	9.4		

Winter showed a mean SPI score indicative of moderate water quality, as described in Appendix A, Table A2. There was no sample for summer due to no water and therefore there can be no comparison between winter and summer in the Moddergat River.

### **8.3.3 Seasonal abundance of diatom species**

No statistical analysis could be done as there was no water in summer, and therefore no samples could be taken.

### **8.3.4 Seasonal diversity of diatom species**

No statistical analysis could be done as there was no water in summer, and therefore no samples could be taken.

### **8.3.5 Ecological Indicators values**

The ecological indicator values classified using the Cemagref (1982) index as referred to in Van Dam et al. (1994), for the assemblage of diatoms in winter at M2, M3.1, M4.1 and M5 within the Moddergat River, is depicted in Table 8.17

**Table 8.17:** Classification of ecological indicators in winter at M2, M3.1, M4.1 and M5 in the Moddergat River, as interpreted by Appendix A, Table A3.

Season	Sites	pH	Salinity	N Uptake Mechanism	Oxygen Requirements	Sabrobity	Trophic State
Winter	M2	alkaliphilous	fresh-brackish	Nitrogen autotrophic - tolerant	low	$\beta$ -mesosaprobous	eutrophic
Winter	M3.1	alkaliphilous	fresh-brackish	Nitrogen autotrophic - tolerant	low	$\alpha$ -meso-polysaprobous	oligo - eutrophic
Winter	M4.1	alkaliphilous	fresh-brackish	Nitrogen autotrophic - tolerant	low	$\alpha$ -meso-polysaprobous	eutrophic
Winter	M5	alkaliphilous	fresh-brackish	Nitrogen autotrophic - tolerant	low	$\alpha$ -meso-polysaprobous	oligo - eutrophic

## CHAPTER 9

### RESULTS: SILVERMINE RIVER

#### **9.1 Macroinvertebrate results**

No samples were taken in summer and autumn as there was no flow.

##### **9.1.1 Macroinvertebrates sampled**

Table 9.1 lists the macroinvertebrate taxa sampled in the Silvermine River during summer, autumn, winter and spring at SM1, SM2, SM3 and SM4 (SM1 is downstream of the pipe bridge, SM2 is in the flood plain area, SM3 at the low flow gabion outlet and SM4 upstream of main road bridge).

Hydrachnellae, Aeshnidae and Corduliidae, each of which is accorded a sensitivity score of 8 in the SASS5 methodology, were the highest scoring (i.e. least pollution tolerant) taxa recorded at these sites, using the criteria of Dickens and Graham (2001).

##### **9.1.2 Seasonal abundance of macroinvertebrates**

The means and standard deviations for abundance of macroinvertebrates (total number of individual macroinvertebrates sampled per season) sampled in the Silvermine River during each season, is listed in Table 9.2.

**Table 9.1:** Macroinvertebrate taxa sampled during each season at each site (SM1, SM2, SM3 and SM4) within the Silvermine River. x = absence of a taxon; √ = presence of a taxon; - = not sampled.

SEASONS	TAXON	SASS SENSITIVITY SCORE	SUMMER				AUTUMN				WINTER				SPRING			
			SM1	SM2	SM3	SM4	SM1	SM2	SM3	SM4	SM1	SM2	SM3	SM4	SM1	SM2	SM3	SM4
	Hydrachnellae	8	-	-	-	-	-	-	-	-	-	-	-	x	x	x	x	
	Baetidae	4	-	-	-	-	-	-	-	√	√	√	√	x	x	√	x	
	Coenagriidae	4	-	-	-	-	-	-	-	x	x	√	x	x	x	x	x	
	Aeshnidae	8	-	-	-	-	-	-	-	-	-	-	-	x	x	x	√	
	Corduliidae	8	-	-	-	-	-	-	-	-	-	-	-	x	x	√	x	
	Corixidae	3	-	-	-	-	-	-	-	-	-	-	-	x	x	x	√	
	Pleidae	4	-	-	-	-	-	-	-	-	-	-	-	x	x	x	x	
	Leptoceridae	6	-	-	-	-	-	-	-	-	-	-	-	x	x	x	x	
	Dytiscidae	5	-	-	-	-	-	-	-	-	-	-	-	√	x	x	x	
	Hydrophiliidae	5	-	-	-	-	-	-	-	-	-	-	-	x	x	x	x	
	Ceratopogonidae	5	-	-	-	-	-	-	-	-	-	-	-	x	x	√	x	
	Chironomidae	2	-	-	-	-	-	-	-	-	-	-	-	x	x	√	√	
	Simuliidae	5	-	-	-	-	-	-	-	-	-	-	-	x	x	√	√	
	Hydrometridae	6	-	-	-	-	-	-	-	-	-	-	-	x	x	x	x	

**Table 9.2:** Total number of macroinvertebrates (abundance) sampled per season at each site (SM1, SM2, SM3 and SM4) in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	140.00	23.00
SM2	-	-	40.00	-
SM3	-	-	130.00	320.00
SM4	-	-	30.00	202.00
Mean	-	-	85.00	136.25
Standard deviation	-	-	58.02	152.18

There was an apparent interaction present between season and site and thus two one-way ANOVA's were done. The F-test for *Season* yielded  $F(1, 5) = 1.46$  with  $p_{\text{season}} = 0.28$ . The available data did not indicate a statistically significant difference in macroinvertebrate abundance between seasons in the Silvermine River.

In the case of *site*, the calculated value of  $F(3, 3) = 0.79$  with  $p_{\text{site}} = 0.58$  did not succeed to show any statistically significant difference in macroinvertebrate abundance between sites in the Silvermine River.

### 9.1.3 Seasonal diversity of macroinvertebrates

The means and standard deviations for diversity of macroinvertebrate taxa (total number of different macroinvertebrate taxa), sampled per season in the Silvermine River, is listed in Table 9.3.

**Table 9.3:** Total number of different macroinvertebrate taxa (diversity) sampled per season at each site (SM1, SM2, SM3 and SM4) in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	5.00	5.00
SM2	-	-	4.00	-
SM3	-	-	4.00	5.00
SM4	-	-	3.00	4.00
Mean	-	-	4.00	4.67
Standard deviation	-	-	0.82	0.58

The two-way additive model ANOVA for *season* yielded  $F(1, 2) = 4.00$  with  $p_{\text{season}} = 0.18$ . The ANOVA for *site* gave  $F(3, 2) = 4.67$  with  $p_{\text{site}} = 0.18$ . There was no

statistically significant difference in macroinvertebrate diversity between seasons or between sites in the Silvermine River.

## 9.2 Water chemistry results

No water chemistry samples were taken at Site SM2. No water chemistry samples were taken in summer and autumn as there was no flow.

### 9.2.1 Total suspended solids (TSS) (mg/l)

The means and standard deviations of TSS (mg/l), measured in the Silvermine River, for each season are listed in Table 9.4.

**Table 9.4:** Total suspended solids (TSS) (mg/l), measured per season at SM1, SM3 and SM4, in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	5.00	5.00
SM3	-	-	4.00	16.00
SM4	-	-	13.00	3.00
Mean	-	-	7.33	8.00
Standard deviation	-	-	4.93	7.00

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The F-test for season yielded  $F(1, 4) = 0.02$  with  $p_{\text{season}} 0.90$ . Thus the available data did not indicate a statistically significant difference in TSS between seasons in the Silvermine River.

In the case of site, the calculated value of  $F(2, 3) = 0.31$  with  $p_{\text{site}} 0.75$  did not succeed to show any statistically significant difference in TSS between sites in the Silvermine River.

### 9.2.2 pH

The means and standard deviations for pH in the Silvermine River for each season are listed in Table 9.5.

**Table 9.5:** pH measured per season at SM1, SM3 and SM4, in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	6.30	6.90
SM3	-	-	6.80	7.10
SM4	-	-	6.60	7.10
Mean	-	-	6.57	7.03
Standard deviation	-	-	0.25	0.12

A two-way additive model ANOVA for *season* yielded  $F(1, 2) = 28.00$  with  $p_{\text{season}} 0.034$ . There was a significant difference in pH between seasons in the Silvermine River. As there were only two seasons the Tukey HSD test for detecting pairwise differences showed the seasons to be significantly different to each other.

The calculated value of  $F(2, 2) = 5.57$  with  $p_{\text{site}} 0.15$  did not confirm a statistically significant difference in pH between sites in the Silvermine River.

### 9.2.3 Conductivity (mS/m)

The means and standard deviations for water conductivity in the Silvermine River, for each season, are listed in Table 9.6.

**Table 9.6:** Water conductivity (mS/m), measured per season at SM1, SM3 and SM4, in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	28.00	37.00
SM3	-	-	31.00	41.00
SM4	-	-	29.00	38.00
Mean	-	-	29.33	38.67
Standard deviation	-	-	1.53	2.08

The Box-Cox computation suggested a natural log transformation in the form:  $y = \log_e(x+1)$ . This stabilised the variance well and a two-way additive model ANOVA for *season* yielded  $F(1, 2) = 8241.53$  with  $p_{\text{season}} 0.0001$ , and *site* yielded  $F(2, 2) = 396.06$  with  $p_{\text{site}} 0.0025$ . There was a significant difference in conductivity between seasons and sites in the Silvermine River. As there were only two seasons the Tukey HSD test for detecting pairwise differences showed the seasons to be significantly

different to each other. The Tukey HSD test for detecting pairwise differences showed the sites to be significantly different to each other.

#### 9.2.4 Chemical oxygen demand (COD) (mg/l)

The means and standard deviations of COD (mg/l) in the Silvermine River, for each season, are listed in Table 9.7.

**Table 9.7:** Chemical oxygen demand (COD) (mg/l), measured per season at SM1, SM3 and SM4, in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	46.00	47.00
SM3	-	-	48.00	47.00
SM4	-	-	56.00	46.00
Mean	-	-	50.00	46.67
Standard deviation	-	-	5.29	0.58

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The F-test for season yielded  $F(1, 4) = 1.18$  with  $p_{\text{season}} 0.34$ . Thus the available data did not indicate a statistically significant difference in COD between seasons in the Silvermine River.

In the case of site, the calculated value of  $F(2, 3) = 0.66$  with  $p_{\text{site}} 0.58$  did not succeed to show any statistically significant difference in COD between sites in the Silvermine River.

#### 9.2.5 Total oxidisable nitrogen (TON) (mg/l N)

The means and standard deviations for TON (mg/l N) in the Silvermine River, for each season, are listed in Table 9.8.

**Table 9.8:** Total oxidisable nitrogen (TON) (mg/l N), measured per season at SM1, SM3 and SM4, in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	0.28	0.72
SM3	-	-	0.88	2.74
SM4	-	-	0.94	0.71
Mean	-	-	0.70	1.39
Standard deviation	-	-	0.36	1.17

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The F-test for season yielded  $F(1, 4) = 0.95$  with  $p_{\text{season}} 0.38$ . Thus the available data did not indicate a statistically significant difference in TON between seasons in the Silvermine River.

In the case of site, the calculated value of  $F(2, 3) = 1.51$  with  $p_{\text{site}} 0.35$  did not succeed to show any statistically significant difference in TON between sites in the Silvermine River.

### 9.2.6 Total inorganic nitrogen (mg/l N)

The means and standard deviations for total inorganic nitrogen (mg/l N) in the Silvermine River, for each season, are listed in Table 9.9.

**Table 9.9:** Total inorganic nitrogen (mg/l N), measured per season at SM1, SM3 and SM4, in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	0.28	0.16
SM3	-	-	0.26	0.15
SM4	-	-	0.22	0.06
Mean	-	-	0.25	0.12
Standard deviation	-	-	0.03	0.05

A two-way additive model ANOVA for *season* yielded  $F(1, 2) = 106.71$  with  $p_{\text{season}} 0.0092$ . There was a significant difference in total inorganic nitrogen between seasons in the Silvermine River. As there were only two seasons the Tukey HSD test for detecting pairwise differences showed the seasons to be significantly different to each other.

The calculated value of  $F(2, 2) = 15.69$  with  $p_{\text{site}} 0.06$  did not confirm a statistically significant difference in total inorganic nitrogen between sites in the Silvermine River.

### 9.2.7 Total ammonia (mg/l N)

The means and standard deviations for total ammonia (mg/l N) measured as  $\text{NH}_4\text{-N}$ , in the Silvermine River, for each season, are listed in Table 9.10.

**Table 9.10:** Total ammonia (mg/l N) measured per season at SM1, SM3 and SM4, in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	0.06	0.05
SM3	-	-	0.05	0.03
SM4	-	-	0.05	0.04
Mean	-	-	0.05	0.04
Standard deviation	-	-	0.01	0.01

A two-way additive model ANOVA for *season* yielded  $F(1, 2) = 315.57$  with  $p_{\text{season}} 0.0032$ , and *site* yielded  $F(2, 2) = 154.71$  with  $p_{\text{site}} 0.0064$ . There was a significant difference in total ammonia between seasons and sites in the Silvermine River. As there were only two seasons the Tukey HSD test for detecting pairwise differences showed the seasons to be significantly different to each other. The Tukey HSD test for detecting pairwise differences between sites could not find a pair that was significant. This could be a dissonance which is known to occur in situations where there is a shortage of data. If one ignores the factor season and does a one-way ANOVA for site, then no significant difference is found. The two methods use different estimates of residual variance which could be a possible explanation.

Calculated un-ionised ammonia ( $\text{NH}_3$ ) concentrations coded as to whether they fell within DWAF (1996) chronic or acute toxicity ranges, are highlighted in Table 9.11.

**Table 9.11:** Total Ammonia (measured as NH<sub>4</sub>-N) and pH measured during each season at each site, within the Silvermine River (SM). Un-ionised ammonia (NH<sub>3</sub>) component calculated for measured pH at temperatures of 20°C (summer and autumn estimate) and 15°C (spring and winter estimate), using pH and temperature dependent conversions supplied DWAF (1996).

Season	River	Site	Soluble (Total) Ammonia (mg/l N)	Temperature	pH	Un-ionised NH <sub>3</sub> (mg/l N)
Win	Silvermine River	SM1	0.063	15.00	6.30	0.001
		SM2	0.063	15.00	6.30	0.001
		SM3	0.049	15.00	6.60	0.000
		SM4	0.045	15.00	6.80	0.000
Spr	Silvermine River	SM1	0.047	15.00	6.90	0.000
		SM3	0.035	15.00	7.10	0.000
		SM4	0.028	15.00	7.10	0.000

Sum = summer, Aut = autumn, Win = winter and Spr = spring. Acute ammonia toxicity values are highlighted in **dark grey** and chronic ammonia toxicity values in **light grey**. No highlight indicates values were less than the acute or chronic toxicity thresholds as defined by DWAF (1996).

Based on the above, acute and chronic un-ionised ammonia (NH<sub>3</sub>) values were calculated for each site to provide an indication of potential ammonia toxicity at the time of sampling. Total ammonia concentrations and un-ionised ammonia concentrations at all sites during all seasons in the Silvermine River were within the DWAF (1996) target water quality range for this variable (Appendix A, Table A1).

### 9.2.8 Soluble nitrite and nitrate (mg/l N)

The means and standard deviations for soluble nitrite and nitrate (mg/l N) in the Silvermine River, for each season, are listed in Table 9.12.

**Table 9.12:** Soluble nitrite and nitrate (mg/l N) measured per season at SM1, SM3 and SM4, in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	0.22	0.11
SM3	-	-	0.17	0.03
SM4	-	-	0.21	0.11
Mean	-	-	0.20	0.08
Standard deviation	-	-	0.03	0.05

The Box-Cox computation suggested a power transformation in the form:  $y=(x+0.5)^{1.5}$ . This stabilised the variance well and a two-way additive model ANOVA

for season yielded  $F(1, 2) = 134.58$  with  $p_{\text{season}} 0.0073$ . There was a significant difference in nitrite and nitrate between seasons in the Silvermine River. The Tukey HSD test for detecting pairwise differences showed that winter was significant to spring.

In the case of site, the calculated value of  $F(2, 2) = 16.61$  with  $p_{\text{site}} 0.06$  did not succeed to show any statistically significant difference in nitrite and nitrate between sites in the Silvermine River.

### 9.2.9 Total phosphorus (mg/l P)

The means and standard deviations for total phosphorus (mg/l P) in the Silvermine River, for each season, are listed in Table 9.13.

**Table 9.13:** Total phosphorus (mg/l P) measured per season at SM1, SM3 and SM4, in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	0.03	0.03
SM3	-	-	0.03	0.05
SM4	-	-	0.05	0.02
Mean	-	-	0.04	0.04
Standard deviation	-	-	0.01	0.01

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between site and season, thus two one-way ANOVAs were done. The F-test for season yielded  $F(1, 4) = 0.004$  with  $p_{\text{season}} 0.95$ . Thus the available data did not indicate a statistically significant difference in total phosphorus between seasons in the Silvermine River.

In the case of site, the calculated value of  $F(2, 3) = 0.12$  with  $p_{\text{site}} 0.89$  did not succeed to show any statistically significant difference in total phosphorus between sites in the Silvermine River.

### 9.2.10 Orthophosphates (mg/l P)

The means and standard deviations for orthophosphates (mg/l P) in the Silvermine River, for each season, are listed in Table 9.14.

**Table 9.14:** Orthophosphates (mg/l P) measured per season at SM1, SM3 and SM4, in the Silvermine River. - = not sampled.

SITES	SUMMER	AUTUMN	WINTER	SPRING
SM1	-	-	0.01	0.01
SM3	-	-	0.01	0.01
SM4	-	-	0.01	0.01
Mean	-	-	0.01	0.01
Standard deviation	-	-	0.00	0.00

There were no significant differences as all data was identical.

### 9.3 Diatom results

No samples were taken in autumn and spring.

#### 9.3.1 Diatom species sampled

Table 9.15 lists the diatom species sampled in the Silvermine River during summer and winter at SM1, SM3 and SM4. Samples were not taken at SM2. No samples were taken in summer as there was no water.

**Table 9.15:** Diatom species sampled during summer and winter at SM1 SM3 and SM4 within the Silvermine River. × = absence of diatom species; √ = presence of diatom species; - = not sampled.

TAXA	WINTER			SUMMER
	SM1	SM3	SM4	
<i>Achnantheidium minutissimum</i>	√	√	√	-
<i>Achnantheidium saprophila</i>	×	√	×	-
<i>Achnanthes rupestris</i>	√	×	×	-
<i>Cyclotella atomus</i>	√	×	×	-
<i>Diadesmis confervacea</i>	√	×	×	-
<i>Diadesmis sp.</i>	×	√	×	-
<i>Denticula elegans</i>	×	√	×	-
<i>Diploneis puella</i>	×	×	√	-
<i>Discostella woltereckii</i>	√	×	×	-
<i>Eunotia flexuosa</i>	√	×	×	-
<i>Eunotia implicata</i>	√	×	×	-
<i>Eunotia incisa</i>	√	×	×	-

<i>Eunotia minor</i>	√	×	√	-
<i>Epithemia</i> sp	√	×	×	-
<i>Eunotia</i> sp.	√	×	√	-
<i>Fragilaria capucina</i>	√	×	×	-
<i>Fragilaria ulna</i> var. <i>acus</i>	×	×	√	-
<i>Frustulia vulgaris</i>	√	×	×	-
<i>Gomphonema capitatum</i>	×	√	×	-
<i>Gomphonema pseudoaugur</i>	√	×	×	-
<i>Hippodonta avittata</i>	√	×	√	-
<i>Lemnicola hungarica</i>	×	√	√	-
<i>Mayamaea agrestis</i>	√	√	×	-
<i>Mayamaea atomus</i> var. <i>permitis</i>	√	√	×	-
<i>Nitzschia acidoclinata</i>	√	×	×	-
<i>Nitzschia capitellata</i>	×	√	×	-
<i>Navicula cryptocephala</i>	×	√	×	-
<i>Navicula germainii</i>	×	√	×	-
<i>Navicula gregaria</i>	√	√	×	-
<i>Nitzschia archibaldii</i>	√	×	×	-
<i>Nitzschia paleacea</i>	×	√	×	-
<i>Nitzschia recta</i>	√	√	√	-
<i>Navicula rhynchocephala</i>	×	√	×	-
<i>Navicula rostellata</i>	×	√	×	-
<i>Nitzschia sigma</i>	×	√	×	-
<i>Navicula tenelloides</i>	√	×	×	-
<i>Planothidium dau</i>	√	√	×	-
<i>Pinnularia</i> sp.	×	×	×	-
<i>Planothidium engelbrechtii</i>	√	√	√	-
<i>Psammothidium oblongellum</i>	√	√	×	-
<i>Planothidium delicatulum</i>	×	√	×	-
<i>Surirella angusta</i>	×	√	×	-
<i>Sellaphora seminulum</i>	√	√	√	-
<i>Tryblionella debilis</i>	√	×	×	-
<i>Amphora veneta</i>	√	√	×	-

<i>Eolimna subminuscula</i>	√	√	×	-
<i>Nitzschia</i> sp.	√	√	√	-
<i>Navicula veneta</i>	√	√	×	-
<i>Achnantheidium exiguum</i>	√	×	×	-
<i>Nitzschia microcephala</i>	√	×	×	-
<i>Thalassiosira pseudonana</i>	×	×	√	-
<i>Ctenophora pulchella</i>	×	√	√	-
<i>Hippodonta capitata</i>	×	√	×	-
<i>Nitzschia amphibia</i>	√	√	√	-
<i>Nitzschia frustulum</i>	√	√	√	-
<i>Cocconeis placentula</i>	√	√	√	-
<i>Eolimna minima</i>	√	√	√	-
<i>Melosira varians</i>	×	√	×	-
<i>Planothidium frequentissimum</i>	×	√	×	-
<i>Sellaphora pupula</i>	×	√	×	-
<i>Nitzschia palea</i>	×	√	×	-
<i>Gomphonema parvulum</i>	√	√	√	-
<i>Fragilaria construens</i> var. <i>subsalina</i>	×	√	√	-
<i>Cyclotella meneghiniana</i>	×	√	√	-
<i>Staurosira elliptica</i>	×	√	√	-

### 9.3.2 Specific Pollution Sensitivity Index and percentage Pollution Tolerant Valves

Table 9.16 shows the diatom index results, as obtained from the OMNIDIA software package, which provides an indication of the specific pollution index for each diatom sample taken, in the Silvermine River. No diatom samples could be taken in summer as there was no water.

**Table 9.16:** The population, number of diatom species, Specific Pollution Index (SPI) and the percentage of Pollution Tolerant Valves (PTV) for the assemblages of diatoms, sampled in winter at sites SM1 SM3 and SM4, within the Silvermine River. As interpreted by Appendix A, Table A2.

Season	Sites	Population	No. Species	Specific Pollution Sensitivity Index (SPI)	%Pollution Tolerant Valves (%PTV)	Mean SPI Score	Mean PTV %
Winter	SM1	411	38	11.2	28	10.7	21.6
Winter	SM3	406	41	7.8	29.8		
Winter	SM4	413	20	13.2	7		

The SPI score in winter was indicative of moderate water quality, as described in Appendix A, Table A2. The SPI score in summer could not be determined as there was no water.

### 9.3.3 Seasonal abundance of diatom species

No statistical analysis could be done as there was no water in summer and therefore no samples could be taken.

### 9.3.4 Seasonal diversity of diatom species

No statistical analysis could be done as there was no water in summer and therefore no samples could be taken.

### 9.3.5 Ecological Indicator values

The ecological indicator values classified using the Cemagref (1982) index as referred to in Van Dam et al. (1994), for the assemblage of diatoms in winter at sites SM1 SM3 and SM4, within the Silvermine River, is depicted in Table 9.17

**Table 9.17:** Classification of ecological indicators in winter at SM1 SM3 and SM4 within the Silvermine River, as interpreted by Appendix A, Table A3.

Season	Sites	pH	Salinity	N Uptake Mechanism	Oxygen Requirements	Sabrobity	Trophic State
Winter	SM1	circumneutral	fresh-brackish	Nitrogen autotrophic - sensitive	continuously high	oligosaprobous	eutrophic
Winter	SM3	alkaliphilous	fresh-brackish	Nitrogen autotrophic - tolerant	very low	$\alpha$ -meso-polysaprobous	eutrophic
Winter	SM4	circumneutral	fresh-brackish	Nitrogen autotrophic - tolerant	continuously high	$\beta$ -mesosaprobous	oligo - eutrophic

**CHAPTER 10**  
**RESULTS: BIG LOTUS RIVER**

**10.1 Macroinvertebrate results**

**10.1.1 Macroinvertebrates sampled**

Table 10.1 lists the macroinvertebrate taxa sampled in the Big Lotus River during summer, autumn, winter and spring at BL1 and BL2 (BL1 is downstream of Strandfontein Road at the gabion weir, BL2 is 100m downstream of Strandfontein Road).

**Table 10.1:** Macroinvertebrate taxa sampled during each season at each site (BL1 and BL2) within the Big Lotus River. × = absence of a taxon; √ = presence of a taxon.

SEASONS TAXON	SASS SENSITIVITY SCORE	SUMMER		AUTUMN		WINTER		SPRING	
		BL1	BL2	BL1	BL2	BL1	BL2	BL1	BL2
Oligocheata	1	×	×	×	×	√	×	×	×
Crabs	3	×	√	×	×	×	×	×	×
Baetidae	4	×	√	×	×	×	×	×	×
Coenagriidae	4	×	√	√	×	×	√	×	×
Belostomatidae	3	×	√	×	×	×	×	×	×
Corixidae	3	×	√	×	×	×	×	×	×
Pleidae	4	×	×	×	×	×	√	×	×
Dytiscidae	5	×	×	×	×	×	√	×	×
Hydrophilidae	5	×	×	√	×	×	×	×	×
Ceratopogonidae	5	×	×	×	√	×	√	×	√
Chironomidae	2	×	√	√	√	√	√	√	√
Culicidae	1	×	×	√	√	×	×	×	×
Physidae	3	×	√	×	√	×	√	√	×
Planorbidae	3	×	×	×	√	×	×	×	×

Dytiscidae, Hydrophilidae and Ceratopogonidae, each of which is accorded a sensitivity score of 5 in the SASS methodology, were the highest scoring (i.e. least pollution tolerant) taxa recorded at this site using the criteria of Dickens and Graham (2001).

### 10.1.2 Seasonal abundance of macroinvertebrates

The means and standard deviations for abundance of macroinvertebrates (total number of individual macroinvertebrates of all taxa sampled per season) sampled in the Big Lotus River during each season, are depicted in Table 10.2.

**Table 10.2:** Total number of macroinvertebrates (abundance) sampled per season at each site (BL1 and BL2), in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	0.00	112.00	101.00	1100.00
BL2	412.00	140.00	150.00	200.00
<b>Mean</b>	206.00	126.00	125.50	650.00
<b>Standard deviation</b>	291.33	19.80	34.65	636.40

The interaction graph showed a distinct non-parallel pattern indicating the possibility of interaction between Site and Season. There was an apparent interaction present and thus two one-way ANOVAs were done. The Box-Cox computation suggested a square root transformation in the form:  $y=1/\text{sqrt}(x)$ . This stabilised the variance well and the F-test for *Season* yielded  $F(3, 4) = 0.86$  with  $p_{\text{season}} 0.5314$ . Thus the available data did not indicate a statistically significant difference in macroinvertebrate abundance between seasons.

In the case of *site*, the calculated value of  $F(1, 6) = 0.02$  with  $p_{\text{site}} 0.8907$  did not succeed to show any statistically significant difference in macroinvertebrate abundance between sites in the Big Lotus River.

### 10.1.3 Seasonal diversity of macroinvertebrates

The means and standard deviations for diversity of macroinvertebrate taxa (total number of different macroinvertebrate taxa), sampled per season in the Big Lotus River, are depicted in Table 10.3.

**Table 10.3:** Total number of different macroinvertebrate taxa (diversity) sampled per season at each site (BL1 and BL2), in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	0.00	4.00	2.00	2.00
BL2	7.00	5.00	6.00	2.00
<b>Mean</b>	3.50	4.50	4.00	2.00
<b>Standard deviation</b>	4.95	0.71	2.83	0.00

There was an apparent interaction present and thus two one-way ANOVAs were done. The F-test for *season* yielded  $F(3, 4) = 0.28$  with  $p_{\text{season}} 0.8362$ . The ANOVA for *site* gave  $F(1, 6) = 4.91$  with  $p_{\text{site}} 0.0686$ . There was no statistically significant difference in macroinvertebrate diversity between seasons or between sites in the Big Lotus River, although the latter p-value shows some promise, being not too far off 5%.

## 10.2 Water chemistry results

### 10.2.1 Total suspended solids (TSS) (mg/l)

The means and standard deviations for TSS (mg/l), measured in the Big Lotus River, for each season are listed in Table 10.4.

**Table 10.4:** Total suspended solids (TSS) (mg/l), measured per season at BL1 and BL2, in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	3.00	10.00	8.00	6.00
BL2	4.00	9.00	9.00	6.00
Mean	3.50	9.50	8.50	6.00
Standard deviation	0.71	0.71	0.71	0.00

The F-tests for the effect of *season* and *site* have F-values of  $F(3, 3) = 31.55$  and  $F(1, 3) = 0.27$  respectively with p-values of  $p_{\text{Season}} 0.0091$  and  $p_{\text{site}} 0.6376$ . According to the Tukey HSD test the seasons differ significantly from each other as follows: summer and spring are homogeneous (not significantly different), spring and winter are homogeneous and winter and autumn are homogeneous. Summer is significantly different to autumn and winter. Summer had the lowest mean TSS compared to all seasons in the Big Lotus River.

### 10.2.2 pH

The means and standard deviations for pH in the Big Lotus River for each season are listed in Table 10.5.

**Table 10.5:** pH measured per season at BL1 and BL2 in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	7.30	7.80	7.60	7.60
BL2	7.50	8.00	7.70	7.50
Mean	7.40	7.90	7.65	7.55
Standard deviation	0.14	0.14	0.07	0.07

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 8.83$  and  $p_{\text{season}} = 0.0533$  which did not indicate significance at the 5% level.

The calculated value of  $F(1, 3) = 2.0$  with  $p_{\text{site}} = 0.2522$  did not confirm a statistically significant difference in pH between sites in the Big Lotus River.

### 10.2.3 Conductivity (mS/m)

The means and standard deviations for water conductivity in the Big Lotus River, for each season, are listed in Table 10.6.

**Table 10.6:** Water conductivity (mS/m), measured per season at BL1 and BL2 in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	88.00	167.00	163.00	160.00
BL2	90.00	167.00	163.00	160.00
Mean	89.00	167.00	163.00	160.00
Standard deviation	1.41	0.00	0.00	0.00

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 5558.3$  with  $p_{\text{season}} < 0.0001$ . There was a highly significant difference in water conductivity between seasons in the Big Lotus River. The Tukey HSD test for detecting pairwise differences showed summer to be significantly lower than the other seasons, autumn to be significantly higher than winter and spring, and the latter two being homogeneous.

The calculated value of  $F(1, 3) = 1$  with  $p_{\text{site}} = 0.391$  did not confirm a statistically significant difference in water conductivity between sites in the Big Lotus River.

#### 10.2.4 Chemical oxygen demand (COD) (mg/l)

The means and standard deviations for COD (mg/l) in the Big Lotus River, for each season, are listed in Table 10.7.

**Table 10.7:** Chemical oxygen demand (COD) (mg/l), measured per season at BL1 and BL2 in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	67.00	15.00	54.00	45.00
BL2	74.00	16.00	53.00	45.00
Mean	70.50	15.50	53.50	45.00
Standard deviation	4.95	0.71	0.71	0.00

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 162.89$  with  $p_{\text{season}} = 0.0008$ . There was a highly significant difference in chemical oxygen demand between seasons. The Tukey HSD test for detecting pairwise differences showed summer to be significantly higher than all other seasons, and autumn to be significantly lower than all other seasons. Spring and winter were homogenous.

The calculated value of  $F(1, 3) = 0.95$  with  $p_{\text{site}} = 0.40$  did not confirm a statistically significant difference in chemical oxygen demand between sites in the Big Lotus River.

#### 10.2.5 Total oxidisable nitrogen (TON) (mg/l N)

The means and standard deviations for TON (mg/l N) in the Big Lotus River, for each season, are listed in Table 10.8.

**Table 10.8:** Total oxidisable nitrogen (TON) (mg/l N), measured per season at BL1 and BL2 in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	14.35	13.57	9.95	8.76
BL2	13.58	13.39	10.45	8.77
Mean	13.97	13.48	10.20	8.77
Standard deviation	0.54	0.13	0.35	0.01

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 92.50$  with  $p_{\text{season}} = 0.0019$ . There was a highly significant difference in total oxidisable

nitrogen between seasons. The Tukey HSD test for detecting pairwise differences showed summer and autumn to be homogeneous, spring and winter to be homogeneous. Summer and autumn were therefore significantly different to winter and spring in the Big Lotus River.

The calculated value of  $F(1, 3) = 0.18$  with  $p_{\text{site}} 0.703$  did not confirm a statistically significant difference in total oxidisable nitrogen between sites in the Big Lotus River.

### 10.2.6 Total inorganic nitrogen (mg/l N)

The means and standard deviations for total inorganic nitrogen (mg/l N) in the Big Lotus River, for each season, are listed in Table 10.9.

**Table 10.9:** Total inorganic nitrogen (mg/l N), measured per season at BL1 and BL2 in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	12.66	13.54	9.41	8.32
BL2	12.26	13.38	9.47	8.33
Mean	12.46	13.46	9.44	8.33
Standard deviation	0.28	0.12	0.04	0.01

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 550.26$  with  $p_{\text{season}} 0.0001$ . There was a highly significant difference in total inorganic nitrogen between seasons. The Tukey HSD test for detecting pairwise differences showed all seasons were significantly different in the Big Lotus River.

The calculated value of  $F(1, 3) = 1.41$  with  $p_{\text{site}} 0.32$  did not confirm a statistically significant difference in total inorganic nitrogen between sites in the Big Lotus River.

### 10.2.7 Total ammonia (mg/l N)

The means and standard deviations for total ammonia (mg/l N) measured as  $\text{NH}_4\text{-N}$  in the Big Lotus River, for each season, are listed in Table 10.10.

**Table 10.10:** Total ammonia (mg/l N) measured per season at BL1 and BL2 in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	11.40	13.26	5.53	6.75
BL2	11.01	13.04	5.40	6.82
Mean	11.21	13.15	5.47	6.79
Standard deviation	0.28	0.16	0.09	0.05

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 1431.59$  with  $p_{\text{season}} < 0.0001$ . There was a highly significant difference in total ammonia between seasons. The Tukey HSD test for detecting pairwise differences showed all seasons to be significantly different in the Big Lotus River .

The calculated value of  $F(1, 3) = 3.06$  with  $p_{\text{site}} 0.1786$  did not confirm a statistically significant difference in total ammonia between sites in the Big Lotus River.

Calculated un-ionised ammonia ( $\text{NH}_3$ ) concentrations coded as to whether they fell within chronic or acute toxicity ranges, as defined by DWAF (1996), are highlighted in Table 10.11.

**Table 10.11:** Total Ammonia (measured as  $\text{NH}_4\text{-N}$ ) and pH measured during each season at each site, within the Big Lotus River (BL). Un-ionised ammonia ( $\text{NH}_3$ ) component calculated for measured pH, and assuming temperatures of 20 °C (summer and autumn estimate) and 15°C (spring and winter estimate), using pH assuming temperature dependent conversions supplied DWAF (1996).

Season	River	Site	Soluble (Total) Ammonia (mg/l N)	Temperature	pH	Un-ionised $\text{NH}_3$ (mg/l N)
Sum	Big Lotus River	BL1	11.400	20.00	7.30	0.137
		BL2	11.010	20.00	7.50	0.132
Aut		BL1	13.260	20.00	7.80	0.504
		BL2	13.040	20.00	8.00	0.496
Win		BL1	5.530	15.00	7.60	0.047
		BL2	5.400	15.00	7.70	0.046
Spr		BL1	6.750	15.00	7.60	0.057
		BL2	6.820	15.00	7.50	0.058

Sum = summer, Aut = autumn, Win = winter and Spr = spring. Acute ammonia toxicity values are highlighted in **dark grey** and chronic ammonia toxicity values in **light grey**. No highlight indicates values were less than the acute or chronic toxicity thresholds as defined by DWAF (1996).

Based on the above, acute and chronic un-ionised ammonia (NH<sub>3</sub>) concentrations were calculated for each site to provide an indication of actual or potential ammonia toxicity at the time of sampling (Appendix A, Table A1). Both total ammonia concentrations and un-ionised ammonia were highest in the Big Lotus River at BL1 in summer and autumn, showing a three to four fold increase over winter and spring concentrations. Summer and autumn concentrations fell within the acute toxicity range for this variable. Winter and spring concentrations at all sites lay above chronic threshold values only. Un-ionised ammonia concentrations increased three to four fold between summer and autumn in the Big Lotus River.

#### 10.2.8 Soluble nitrite and nitrate (mg/l N)

The means and standard deviations for soluble nitrite and nitrate (mg/l N) in the Big Lotus River, for each season, are listed in Table 10.12.

**Table 10.12:** Soluble nitrite and nitrate (mg/l N) measured per season at BL1 and BL2 in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	1.26	0.28	3.88	1.57
BL2	1.25	0.34	4.07	1.51
Mean	1.26	0.31	3.98	1.54
Standard deviation	0.01	0.04	0.13	0.04

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 833.69$  with  $p_{\text{season}} < 0.0001$ . There was a highly significant difference in soluble nitrite and nitrate between seasons. The Tukey HSD test for detecting pairwise differences showed summer and spring to be homogeneous and significantly different to autumn and winter. Autumn was also significantly different to winter. Autumn had the lowest soluble nitrite and nitrate compared to any other season in the Big Lotus River.

The calculated value of  $F(1, 3) = 0.68$  with  $p_{\text{site}} 0.4710$  did not confirm a statistically significant difference in soluble nitrite and nitrate between sites in the Big Lotus River.

#### 10.2.9 Total phosphorus (mg/l P)

The means and standard deviations for total phosphorus (mg/l P) in the Big Lotus River, for each season, are listed in Table 10.13.

**Table 10.13:** Total phosphorus (mg/l P) measured per season at BL1 and BL2 in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	1.57	2.05	0.97	1.17
BL2	1.48	1.99	0.98	1.22
Mean	1.52	2.02	0.98	1.20
Standard deviation	0.06	0.04	0.01	0.04

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 216.05$  with  $p_{\text{season}} = 0.0005$ . There was a highly significant difference in total phosphorus between seasons. The Tukey HSD test for detecting pairwise differences showed all seasons to be significantly different. Autumn had the highest total phosphorus and winter the lowest.

The calculated value of  $F(1, 3) = 0.46$  with  $p_{\text{site}} = 0.5469$  did not confirm a statistically significant difference in total phosphorus between sites in the Big Lotus River.

#### 10.2.10 Orthophosphates (mg/l P)

The means and standard deviations for orthophosphates (mg/l P) in the Big Lotus River, for each season, are listed in Table 10.14.

**Table 10.14** Orthophosphates (mg/l P) measured per season at BL1 and BL2 in the Big Lotus River.

SITES	SUMMER	AUTUMN	WINTER	SPRING
BL1	1.30	1.64	0.75	0.89
BL2	1.22	1.60	0.75	0.95
Mean	1.26	1.62	0.75	0.92
Standard deviation	0.06	0.03	0.00	0.04

The two-way additive model ANOVA for *season* gave a calculated value of  $F(3, 3) = 162.79$  with  $p_{\text{season}} = 0.0008$ . There was a highly significant difference in orthophosphates between seasons. The Tukey HSD test for detecting pairwise differences showed summer to be significant to all seasons, autumn to be significant to all seasons, and winter and spring to be homogenous.

The calculated value of  $F(1, 3) = 0.27$  with  $p_{\text{site}} = 0.6379$  did not confirm a statistically significant difference in orthophosphates between sites in the Big Lotus River.

### 10.3 Diatom results

#### 10.3.1 Diatom species sampled

Table 10.15 lists the diatom species sampled in the Big Lotus River during summer and winter at BL1 and BL2. Samples were not taken in autumn and spring.

**Table 10.15:** Diatom species sampled during summer and winter at BL1 and BL2 within the Big Lotus River. × = absence of diatom species; √ = presence of diatom species. – = not sampled

TAXA	WINTER		SUMMER	
	BL1	BL2	BL1	BL2
<i>Anomoeoneis sphaerophora</i>	×	√	×	-
<i>Craticula accomodiformis</i>	×	×	√	-
<i>Cyclotella atomus</i>	×	√	×	-
<i>Craticula accomoda</i>	√	√	×	-
<i>Diadesmis confervacea</i>	×	√	×	-
<i>Discostella woltereckii</i>	×	√	×	-
<i>Fistulifera saprophila</i>	√	×	×	-
<i>Fragilaria ulna</i> var. <i>acus</i>	√	√	×	-
<i>Gomphonema insigne</i>	×	×	√	-
<i>Gomphonema pseudoaugur</i>	√	√	√	-
<i>Hantzschia amphioxys</i>	×	√	×	-
<i>Mayamaea excelsa</i>	×	√	×	-
<i>Mayamaea atomus</i> var. <i>permitis</i>	√	×	√	-
<i>Nitzschia agnewii</i>	×	√	×	-
<i>Navicula</i> sp.	×	√	×	-
<i>Nitzschia capitellata</i>	√	√	√	-
<i>Navicula gregaria</i>	×	×	√	-
<i>Nitzschia commutata</i>	×	√	×	-
<i>Navicla microcari</i>	×	√	×	-
<i>Staurosirella pinnata</i>	×	√	×	-
Small hyaline centric species aff. <i>Skeletonema</i>	√	√	×	-
<i>Sellaphora seminulum</i>	√	√	√	-
<i>Eolimna subminuscula</i>	√	√	√	-
<i>Navicula veneta</i>	√	√	√	-

<i>Tabularia fasciculata</i>	x	√	x	-
<i>Nitzschia umbonata</i>	√	√	√	-
<i>Thalassiosira weissflogii</i>	x	√	x	-
<i>Achnantheidium exiguum</i>	√	√	√	-
<i>Nitzschia frustulum</i>	√	√	√	-
<i>Eolimna minima</i>	√	√	√	-
<i>Nitzschia supralitorea</i>	√	√	√	-
<i>Sellaphora pupula</i>	√	√	√	-
<i>Nitzschia palea</i>	√	√	√	-
<i>Gomphonema parvulum</i>	x	x	√	-
<i>Nitzschia aurariae</i>	√	√	x	-
<i>Cyclotella meneghiniana</i>	√	√	√	-
<i>Staurosira elliptica</i>	x	x	√	-

### 10.3.2 Specific Pollution Index and Percentage Pollution Tolerant Valves

Table 10.16 shows the diatom index results, as obtained from the OMNIDIA software package, which provides an indication of the Specific Pollution Index (SPI) for each diatom sample taken, in the Big Lotus River.

**Table 10.16:** The population, number of diatom species, Specific Pollution Index (SPI) and the percentage of Pollution Tolerant Valves (PTV) for the assemblages of diatoms sampled at BL1 and BL2 in winter, and at BL1 in summer. As interpreted by Appendix A, Table A2.

Season	Sites	Population	No. Species	Specific Pollution Sensitivity Index (SPI)	%Pollution Tolerant Valves (%PTV)	Mean SPI Score	Mean PTV %
Winter	BL1	416	17	2.4	23.1	3.4	23.4
Winter	BL2	418	28	4.4	23.7		
Summer	BL1	409	19	3.5	59.4	3.5	59.4

Winter showed similar SPI scores, which were indicative of bad water quality, as described in Appendix A, Table A2. However, summer had a much higher percentage of pollution tolerant valves than winter.

### 10.3.3 Seasonal abundance of diatom species

The square root transformed mean ( $\pm$  standard deviation) abundance of diatoms (total number of individual diatoms), sampled in summer and winter, in the Big Lotus River is depicted in Table 10.17.

**Table 10.17** Total number of diatoms (abundance) sampled in summer and winter at BL1 and BL2, in the Big Lotus River.

SITES	WINTER	SUMMER
BL1	416.00	409.00
BL2	417.00	-
Mean	416.50	409.00
Standard deviation	0.71	-

The values were square root transformed to perform the one-way Analysis of Variance (ANOVA). The calculated value of  $F$  at 1 and 1 df did not exceed the critical value of 39. There was no statistically significant difference ( $F_{39}=161.45$ ,  $P > 0.05$ ) in the number of diatoms in the Big Lotus River in summer, compared to winter.

The calculated value of  $F$  at 1 and 1 df did not exceed the critical value of 39. There was no statistically significant difference ( $F_{39}=161.45$ ,  $P > 0.05$ ) in the number of diatoms between sites in the Big Lotus River.

### 10.3.4 Seasonal diversity of diatom species

The square root transformed mean ( $\pm$  standard deviation) diversity of diatom taxa (total number of different diatom taxa), sampled in summer and winter in the Big Lotus River, is depicted in Table 10.18.

**Table 10.18:** Total number of different diatom taxa (diversity) sampled in summer and winter at BL1 and BL2 in the Big Lotus River.

SITES	WINTER	SUMMER
BL1	17.00 (4.12)	19.00 (4.36)
BL2	27.00 (5.20)	-
Mean	22.00 (4.66)	19.00 (4.36)
Standard deviation	7.07 (0.76)	-

The values were square root transformed to perform the one-way Analysis of Variance (ANOVA). The calculated value of  $F$  at 1 and 1 df did exceed the critical

value of 39. There is no statistically significant difference ( $F_{39} = 161.45$ ,  $P > 0.05$ ) in diatom taxa between summer and winter.

The calculated value of  $F$  at 1 and 1 df did exceed the critical value of 39. There is no statistically significant difference ( $F_{39} = 161.45$ ,  $P > 0.05$ ) in diatom taxa between sites in the Big Lotus River.

#### **10.3.5 Ecological Indicator Values**

The ecological indicator values as classified using the Cemagref (1982) index as referred to in Van Dam et al. (1994), for the assemblage of diatoms in winter at BL1 and BL2, and summer at BL1 in the Big Lotus River, is depicted in Table 10.19.

**Table 10.19:** Classification of ecological indicators at site BL1 and BL2 in summer and winter, in the Big Lotus River, as interpreted by Appendix A, Table A3.

Season	Sites	pH	Salinity	N Uptake Mechanism	Oxygen Requirements	Sabrobity	Trophic State
Winter	BL1	alkaliphilous	brackish-fresh	Nitrogen autotrophic - tolerant	very low	$\alpha$ -meso-polysaprobous	eutrophic
Winter	BL2	alkaliphilous	fresh-brackish	Nitrogen heterotrophic - facultative	very low	$\alpha$ -meso-polysaprobous	eutrophic
Summer	BL1	circumneutral	fresh-brackish	Nitrogen heterotrophic - facultative	very low	$\alpha$ -meso-polysaprobous	eutrophic

## CHAPTER 11

### RESULTS: WATER CHEMISTRY ALL RIVERS

#### 11.1 pH

All sites sampled were within the range of 6 and 8 pH as illustrated in Appendix B, Figure B1. The Keysers River and the Silvermine River showed lower levels of pH in winter and spring than any other sites. The pH levels across all sites showed lowest pH values in winter and the highest in summer.

#### 11.2 Conductivity (mS/m)

There was seasonal change in conductivity between seasons or sites in each river as illustrated in Appendix B, Figure B2. In all rivers, conductivity increased in summer, and decreased in winter, with the exception of the Langevlei Canal. The increased salinity in winter in the Langevlei Canal could be explained by the backing up of saline water from Zandvlei Estuary due to increased water levels. The Big Lotus River had the highest conductivity (167mS/m) in autumn and the Silvermine River the lowest conductivity (28mS/m) in winter, compared to all other seasons in all other rivers. Conductivity fell within the range for fresh (as opposed to brackish) conditions (i.e. less than 450mS/m) (Davies and Day 1998) for all of the sites at all seasons.

#### 11.3 Ammonia (Total and un-ionised) (mg/l N)

Concentrations of un-ionised ammonia falling in DWAF (1996) acute and chronic toxicity ranges for this variable have been highlighted. Values of both measured and calculated un-ionised ammonia are shown in Table 11.1.

**Table 11.1:** Total Ammonia (measured as NH<sub>4</sub>-N) and pH measured during each season, at different sampling sites within the Big Lotus River (BL), Little Lotus River (LL), Keysers River (KVS), Langevlei Canal (LV), Moddergat River (M) and Silvermine River (SM). Un-ionised ammonia (NH<sub>3</sub>) component calculated for measured pH, and at temperatures of 20 °C (summer and autumn estimate) and 15°C (spring and winter estimate), using pH and temperature dependent conversions supplied DWAF (1996).

Season	River	Site	Soluble (Total) Ammonia (mg/l N)	Temperature	pH	Un-ionised NH <sub>3</sub> (mg/l N)
Sum	Big Lotus	BL1	11.400	20.00	7.30	0.137
		BL2	11.010	20.00	7.50	0.132
Aut		BL1	13.260	20.00	7.80	0.504

		BL2	13.040	20.00	8.00	0.496	
Win		BL1	5.530	15.00	7.60	0.047	
		BL2	5.400	15.00	7.70	0.046	
Spr		BL1	6.750	15.00	7.60	0.057	
		BL2	6.820	15.00	7.50	0.058	
Sum	Little Lotus River	LL1	1.590	20.00	7.70	0.019	
		LL4	3.910	20.00	7.90	0.148	
Aut		LL1	0.040	20.00	7.70	0.000	
		LL4	0.030	20.00	7.90	0.001	
Win		LL1	0.120	15.00	7.60	0.001	
		LL4	0.122	15.00	7.60	0.001	
Spr		LL1	0.083	15.00	7.10	0.001	
		LL4	0.125	15.00	7.40	0.001	
Sum		Keysers River	KVS1	0.050	20.00	7.80	0.002
			KVS3	0.146	20.00	7.10	0.002
Aut	KVS1		0.05	20.00	7.40	0.001	
	KVS3		0.01	20.00	7.20	0.000	
Win	KVS1		0.11	15.00	6.90	0.001	
	KVS3		0.13	15.00	6.80	0.008	
Spr	KVS1		0.93	15.00	7.00	0.016	
	KVS3		1.92	15.00	6.90	0.004	
Sum	Langevlei canal	LV1	0.499	20.00	7.70	0.006	
Aut		LV1	0.390	20.00	7.70	0.005	
		LV2	0.184	20.00	7.60	0.002	
Win		LV1	0.148	15.00	7.00	0.001	
		LV2	0.072	15.00	7.10	0.001	
Spr		LV1	0.450	15.00	7.50	0.004	
		LV2	0.410	15.00	7.50	0.003	
Sum		Moddergat River	M1	0.582	20.00	7.20	0.002
Aut	M5		0.318	20.00	8.00	0.012	
Win	M1		0.035	20.00	7.70	0.004	
	M1		0.076	20.00	7.50	0.001	
	M2		0.063	15.00	7.40	0.001	
	M3.2		0.055	15.00	7.40	0.001	
	M3.1		0.055	15.00	7.40	0.000	
	M4.2		0.058	15.00	7.50	0.000	
	M4.1		0.058	15.00	7.50	0.000	
	M5		0.049	15.00	7.50	0.000	
	M6		0.055	15.00	7.50	0.000	
	M7		0.076	15.00	7.40	0.000	
	Spr		M1	0.143	15.00	7.50	0.001
			M2	0.241	15.00	7.40	0.002
			M3.2	0.040	15.00	7.40	0.000
			M3.1	0.040	15.00	7.40	0.000
		M4.2	0.023	15.00	7.20	0.000	

		M4.1	0.023	15.00	7.20	0.000
		M5	0.023	15.00	7.30	0.000
		M6	0.055	15.00	7.30	0.000
		M7	0.045	15.00	7.50	0.000
Win	Silvermine River	SM1	0.063	15.00	6.30	0.001
		SM2	0.063	15.00	6.30	0.001
		SM3	0.049	15.00	6.60	0.000
		SM4	0.045	15.00	6.80	0.000
Spr	Silvermine River	SM1	0.047	15.00	6.90	0.000
		SM3	0.035	15.00	7.10	0.000
		SM4	0.028	15.00	7.10	0.000

Sum = summer, Aut = autumn, Win = winter and Spr = spring. Acute ammonia toxicity values are highlighted in **dark grey** and chronic ammonia toxicity values in **light grey** - toxicity thresholds after DWAF (1996). No highlight indicates values were less than the acute or chronic toxicity thresholds as defined by DWAF (1996).

Appendix B, Figure B3 graphically illustrates concentrations of total ammonia across all sites, seasons and rivers, and shows that total ammonia concentrations were lowest in the Keysers River at KVS 3 in autumn, and highest in the Big Lotus River at BL1. There was also a 3 to 4 fold increase in unionized ammonia between summer and autumn in the Big Lotus River (see also Table 11.1).

On the basis of data shown in Table 11.1, samples exceeded acute thresholds at the Big Lotus River in autumn and summer at BL1 and BL2 and at LL4 in the Little Lotus River in summer. In summer, winter and spring, BL1 and BL2 in the Big Lotus River exceeded DWAF's (1996) chronic threshold, as did LL1 in the Little Lotus River in summer. Concentrations of un-ionised ammonia at all other rivers and sites fell within the target water quality range as classified in DWAF (1996) for this variable (Appendix A, Table A1).

#### 11.4 Total inorganic nitrogen (mg/l N)

Total inorganic nitrogen concentrations, calculated by addition of total (soluble) ammonia and measured nitrate and nitrite data, are illustrated graphically in Appendix B, Figure B3, where they are shown in relation to their implications for trophic status (eutrophic, mesotrophic, oligotrophic or hypertrophic categories), using the ranges for each trophic state as described by DWAF (1996) (Appendix A, Table A4).

The data show that the highest concentrations of total inorganic nitrogen (TIN) occurred in the Big Lotus River in autumn at BL1 (13.54 mg/l N). The lowest TIN concentrations occurred in the Keyzers River in autumn at KVS3 (0.04 mg/l N).

All sites in the Big Lotus River during summer and autumn were hypertrophic with respect to nitrogen, with all TIN measurements >10 mg/l N. The Big Lotus River at BL1 and BL2 in winter and spring, LL4 in the Little Lotus River in summer and LL1 and LL4 of the Little Lotus River in winter, as well as LV1 and LV2 of the Langevelei Canal in winter, were all eutrophic (i.e. in the range 2.5- 10 mg/l N).

LL1 of the Little Lotus River in summer as well as LL1 and LL4 in spring, the Keyzers River at KVS1 and KVS3 in winter and spring, the Langevelei Canal at LV1 in summer and LV1 and LV2 in autumn and spring, the Moddergat River at M1 and M5 in summer as well as all sites in winter and M1, M2 and M7 in spring, were mesotrophic with respect to nitrogen, with all measurements of TIN being in the range of 0.5 mg/l N - 2.5 mg/l N.

LL1 and LL4 in the Little Lotus River in autumn, and KVS1 and KVS3 of the Keyzers River in summer and autumn, M1 in the Moddergat River in autumn and M4.2, M4.1, M5 and M6, as well as all sites and seasons in the Silvermine River, were oligotrophic with respect to nitrogen, with all measurements being less than 0.5 mg/l N.

As depicted in Appendix B, Figure B3, total inorganic nitrogen comprised mainly of ammonia nitrogen in the Big Lotus River in all seasons at all sites, although the proportion decreased somewhat in winter at BL1 and BL2 compared to other seasons. Similarly TIN concentrations at all sites, in the Little Lotus River in summer was also dominated by ammonia nitrogen with a definite spike in ammonia at LL4. Ammonia nitrogen (measured as total ammonia  $\text{NH}_4\text{-1}$ ) also dominated the total inorganic nitrogen component at KVS1 and KVS3 in spring and at LV1 and M1 in the Langevelei Canal and Moddergat River respectively in summer, suggesting low levels of aeration.

### 11.5 Total Phosphorus and Orthophosphates (mg/l P)

Appendix B, Figure B4 illustrates total phosphorus and orthophosphate concentrations across all rivers, sites and seasons. These data are shown in relation to the associated trophic states suggested by DWAF (1996) for this variable, as depicted in Appendix A, Table A4.

The data show that the highest orthophosphate concentrations occurred in the Big Lotus River in autumn at BL1 (1.64 mg/l P). The lowest orthophosphate concentrations occurred in the Silvermine River in all seasons, with all sites consistently depicting less than 0.01 mg/l P.

The Big Lotus River at all sites in all seasons, and the Little Lotus River in summer at all sites, fell above DWAF (1996) threshold values for hypertrophic systems, with orthophosphate values greater than 0.25 mg/l P.

Phosphorus concentrations at KVS 3 in the Keyzers River in summer and spring, and M5 in summer, and M7 in spring in the Moddergat River also fell within this hypertrophic range.

Sites that lay within the range of eutrophic conditions, with concentrations of orthophosphates between 0.025 mg/l P and 0.25 mg /l P, included all of the sites in the Little Lotus River except summer (see above), the Keyzers River at all sites except KVS1 in autumn and KVS3 in Winter, as well as all sites and seasons in the Langevelei Canal, the Moddergat River at all sites except M5 in summer and M7 in spring.

Sites that lay within the range of mesotrophic conditions with concentrations of orthophosphates between 0.005 mg/l P and 0.025 mg/l P, were in the Silvermine River at all sites in all seasons, in the Keyzers River at KVS1 in autumn and KVS3 in winter, in the SM3 in winter. No sites were within the range of oligotrophic conditions with less than 0.005mg/l P.

Total phosphorus at most sites comprised mostly orthophosphate (PO<sub>4</sub>-P), that is, the biologically available component (DWAF, 1996). This pattern was not however

exhibited in the Keyzers River at KVS1 in summer, KVS1 and KVS3 in winter and KVS3 in autumn. Neither was it exhibited in the Silvermine River during any seasons.

#### **11.6 Total suspended solids (TSS) (mg/l)**

As illustrated in Appendix B, Figure B5, there was a noticeable seasonal trend of decreasing TSS concentrations between winter and summer, across all rivers. The Big Lotus River showed the highest TSS concentrations in winter at site BL2, and in autumn at BL1, the Little Lotus River in summer and autumn at LL1 and LL4, the Keyzers River in summer and spring at KVS3, the Langevelei Canal in summer at LV1 and winter at LV3, the Moddergat River in summer at site M1 and M5 and in winter at Sites M1, M2, M6 and M7, and the Silvermine River in winter at SM3 and in spring at SM4.

The highest concentration of TSS across all sites and rivers was 54.00 mg/l in the Little Lotus River at LL1 in autumn, and the lowest was 1.00 mg/l in the Little Lotus River at LL4 in summer and spring.

#### **11.7 Chemical oxygen demand (COD) (mg/l)**

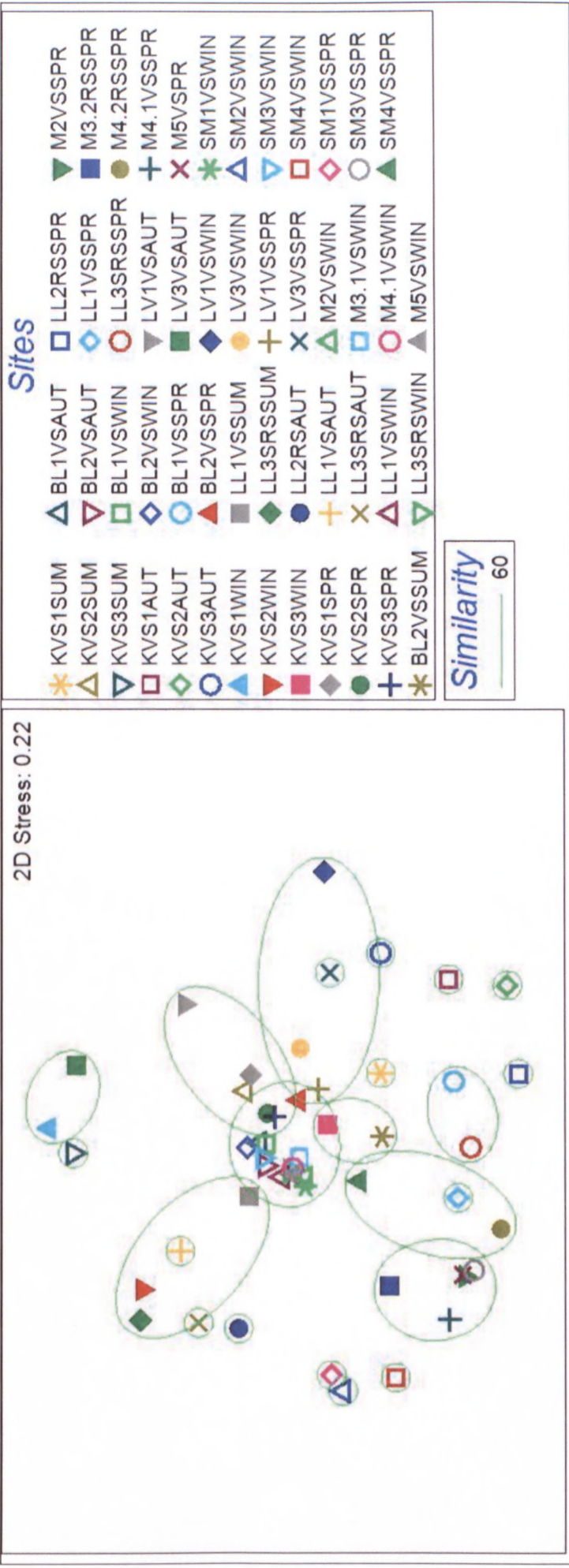
As depicted in Appendix B, Figure B6, there was a clear trend in this variable across all rivers and sites, with elevated COD levels being evident in winter and summer, across all sites. The Big Lotus River had the highest value of COD at BL1 in winter, while the Little Lotus River had the Lowest COD value in autumn at LL4. COD was measured to determine effects of effluents discharged in to the system i.e. constraints on the system from a water quality point of view.

## CHAPTER 12

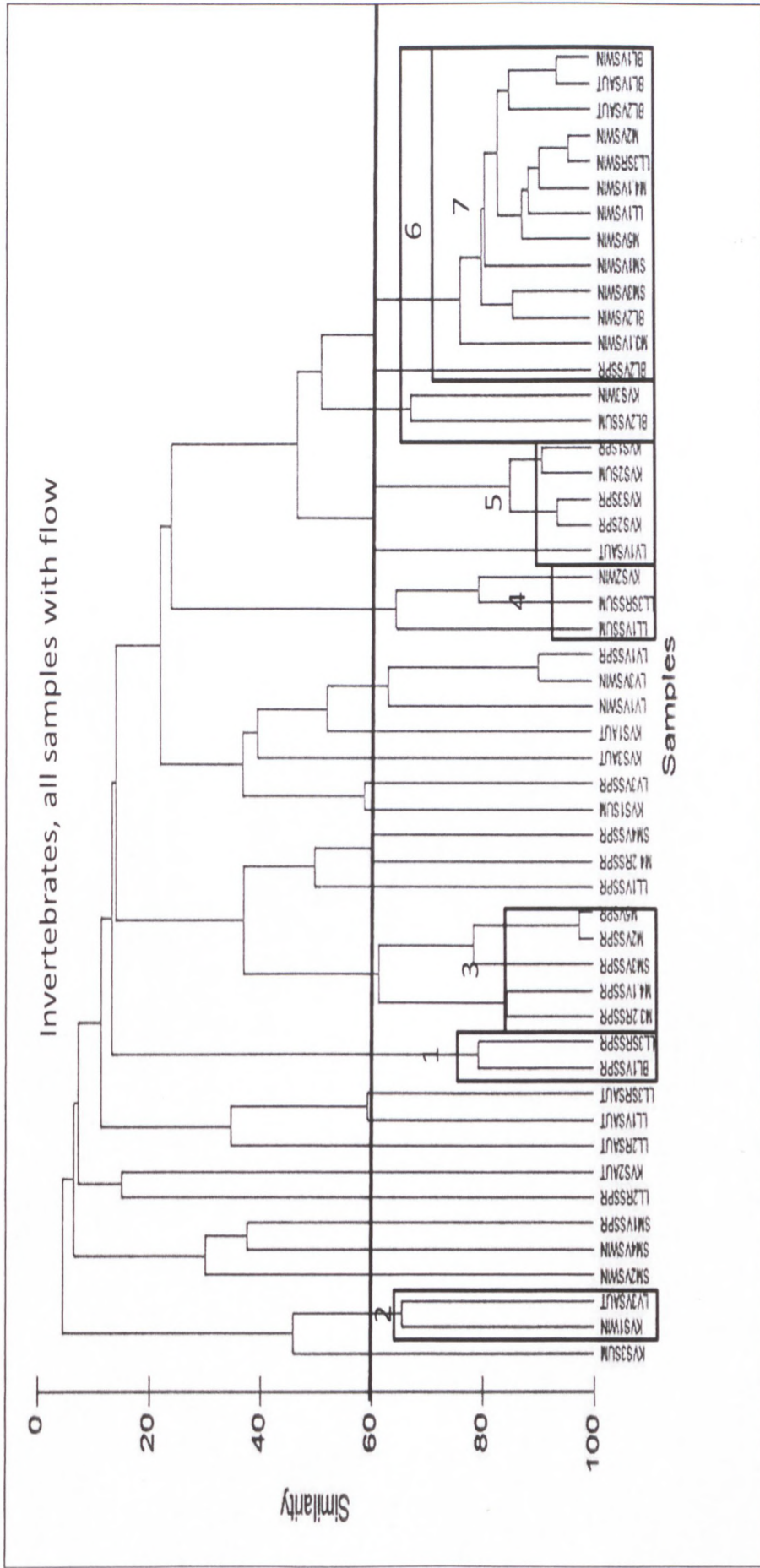
### RESULTS: MACROINVERTEBRATES ALL RIVERS

A number of routines in the PRIMER (Plymouth Routines in Multivariate Ecological Research) package version 6 were used to test for significant differences between rivers, sites and seasons. Initially, multidimensional scaling (MDS) analysis was carried out on log transformed abundance data. The results of the MDS analyses are depicted in Figure 12.1, with the three dimensional distance between sampling points being indicative of levels of similarity – sites that are closest to each other are most similar. Data shown in the MDS plot in Figure 12.1 have been grouped at a 60% level of similarity. Such information was derived from a hierarchical cluster analysis that was carried out on abundance data, and used to generate the dendrogram shown in Figure 12.2. Hierarchical cluster analysis fuses samples that have the highest similarities into distinct groupings, and then joins groups into larger clusters at progressively lower levels of similarity, to produce a plot of the natural similarities between groups or samples, irrespective of any group identification assigned to them previously. As depicted in Figure 12.2, these results are then represented in a dendrogram with the x axis representing the samples, and the y axis the level of similarity at which successive groupings are formed.

Of the samples shown in Figure 12.2, a number of distinct groupings can be identified on the basis of similarity. If samples that group with one or more other samples at similarities greater than or equal to 60% are themselves treated as individual groupings, and numbered as shown in Figure 12.2, then seven distinct groups can be identified. The cluster analysis indicates that within-group similarity for each of these groups is high, and they can thus be regarded as having characterizing invertebrate communities, which should be distinct from other numbered groups. The remainder of the samples separate out at much lower levels of similarity, and invertebrate communities in the latter are thus less markedly distinct from other sites and groups. In order to determine the invertebrate taxa that drive both the close within-group similarities, and the differences between groups, a similarity percentage (SIMPER) analysis of macroinvertebrate abundance was carried out on the numbered groups (Figure 12.2). The results of the SIMPER analysis are shown in Table 12.1.



**Figure 12.1:** Multidimensional scaling (MDS) analysis of macroinvertebrates for each river at each site in each season, depicting clusters of 60% similarity. BL- Big Lotus River, LL - Little Lotus River, LV - Langevelei River, M - Moddergat River, SM - Silvermine River. The biotopes represented are described as VS - Marginal vegetation, SRS - Submerged Riffle, RS - Riffle. Seasons are represented as AUT - autumn, SPR - spring, WIN - winter and SUM - summer.



**Figure 12.2:** Cluster analysis dendrogram, with the x-axis showing the macroinvertebrate samples, and the y axis the level of similarity at which successive groups are formed. The horizontal line highlights the groups identified as having a similarity of 60%. BL- Big Lotus River, LL - Little Lotus River, LV - Langevlei River, K - Keyzers River, M -Moddergat River, SM - Silvermine River. The biotopes represented are described as VS - Marginal vegetation, SRS - Submerged Riffle, RS - Riffle. Seasons are represented as AUT- autumn, SPR - spring, WIN - winter and SUM - summer.

**Table 12.1:** Clusters of sites; biotopes and seasons, across all rivers (Keyzers River (KVS); Little Lotus River (LL); Langevlei Canal (LV); Moddergat River (M); Silvermine River (SM); Big Lotus River (BL)), that are similar and dissimilar. Seven clusters separated out at 60% similarity for which the SIMPER results have been discussed. VS - Marginal vegetation; RS – Riffle and SRS – Submerged Riffle.

Sites	Season	Group Numbers	% Similarity within group	% Dissimilarity to other sites/groups
BL1VS	Spring	1	79%	32% to remaining sites
LL3SRS	Spring			
KVS1	Winter	2	65%	86% to remaining sites
LV3VS	Autumn			
M3.2VS	Spring	3	71%	63% to remaining sites
M4.1RS	Spring			
M2VS	Spring			
M5VS	Spring			
SM3VS	Spring			
LL1VS	Summer	4	70%	76% to remaining sites
LL3SRS	Summer			
KVS2	Winter			
KVS2	Summer	5	77%	53% to remaining sites
KVS1	Spring			
KVS2	Spring			
KVS3	Spring			
LV1VS	Autumn			
BL2VS	Summer	6	62%	76% to Group 5
BL1VS	Autumn			
BL2VS	Autumn			
BL1VS	Winter			
BL2VS	Winter			
BL2VS	Spring			
LL1VS	Winter			
LL1SRS	Winter			
KVS2	Summer			
KVS1	Spring			
KVS2	Spring			
KVS3	Spring			
LV1VS	Autumn			
M2VS	Winter			
M3.1VS	Winter			
M4.1VS	Winter			
M5VS	Winter			
SM1VS	Winter			
SM3VS	Winter			
LL1VS	Summer			

LL3SRS	Summer			
KVS2	Winter			
M2VS	Winter			
M3.1VS	Winter			
M4.1VS	Winter			
M5VS	Winter			
SM1VS	Winter			
SM3VS	Winter			
KVS3	Winter			
LL3SRS	Winter	7	72%	53% to group 6
LL1VS	Winter			
BL1VS	Winter			
BL2VS	Winter			
BL2VS	Spring			
BL1VS	Autumn			
BL2VS	Autumn			
BL2VS	Summer			

The SIMPER results as reflected in Table 12.1 indicated that:

Sites and seasons that fall in group one were 79% similar to each other, and the grouping was best characterized by the prolific abundance of Chironomidae. Group two was 87% dissimilar to the remaining sites in all rivers, and were 32% similar to each other due to the presence of individuals of the families Chironomidae, Baetidae, Simuliidae, Hirudinea and Pleidae taxa.

Sites and seasons that fall in group two were 65% similar to each other. The similarity was characterized by the presence of individuals of Pleidae. Group two was 86% dissimilar to the remaining sites in all rivers (rest). The remaining sites in all rivers were considered 41 % similar to each other due to the presence of individuals of Chironomidae, Hirudinea, Pleidae and Baetidae taxa.

Sites and seasons that fall in group three were 71% similar to each other, and were characterized by the presence of individuals of Baetidae and Simuliidae taxa. Group three was 63% dissimilar to group five. Group five was 53% similar to each other, and was characterized by the presence of Simuliidae taxa.

Sites and seasons that fall in group four were 70% similar to each other, and characterized by the presence of individuals of Planorbidae taxa, and were 76%

dissimilar to the remaining sites (Group 6) which were considered 62% similar to each other, and were characterized by the presence of individuals of Chironomidae and Pleidae taxa.

Sites and seasons that fall in group five were 77% similar to each other, and characterized by the presence of individuals of Pleidae and Chironomidae taxa. The remaining sites (Group 7) were 53.01% dissimilar to group five but 72% similar to each other, and characterized by the presence of individuals of Chironomidae taxa.

When considering the similarity amongst seasons, the SIMPER analysis indicated that autumn and summer samples were only considered 10% similar to each other, and winter and spring samples 21% similar to each other. Autumn and summer were also 87% dissimilar to spring and winter samples. Taxa contributing to autumn and summer similarities were individuals of the family Chironomidae, Physidae, Planorbidae, Hirudinea and Coenagriidae. The SASS5 macroinvertebrate index (Chutter, 1994) allocates these taxa scores of between two and three (Dickens and Graham, 2002). Similarities in taxa in winter and spring were driven by Chironomidae, Baetidae, Simuliidae and Physidae. Chironomidae contributed 65% of the similarity of taxa and had the highest abundance. Low levels of similarity occurred between rivers, with all rivers being less than 25% similar between sites and seasons within the same river, except the Big Lotus River which showed 41% similarity within its sites, across all seasons.

## CHAPTER 13

### DISCUSSION: KEYSERS RIVER

#### 13.1 Water chemistry

The South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996) consists of a water quality 'effect', a Target Water Quality Range (TWQR) and water quality criteria, the Critical Effects Value (CEV), and the Acute Effects Value (AEV) of various parameters. The results of the present study were compared with the South African Water Quality Guidelines for Aquatic Ecosystems water quality 'effects', and TWQR (DWAF, 1996).

No significant differences were found between sites KVS1 and KVS3, for any of the water chemistry parameters measured, suggesting the intervention (i.e. earth canal) had no effect on water quality (Chapter 5, Section 5.2 p 5-3). There were also no significant differences in total suspended solids (TSS), pH, conductivity, chemical oxygen demand (COD), and orthophosphates between seasons, measured per site.

TSS concentrations were not significantly different, but there was a 'spike' in summer (Table 5.4 p 5-3). Whether the results of the present study for TSS concentrations were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined as there was no background data. It was therefore unknown whether these results varied more than 10% to that of the background levels, as stipulated by DWAF (1996) and DWAF (2008). According to Palmer et al. (2005), in the lower reaches of rivers there are no unimpacted sites, and reference conditions are difficult to determine. The results of the present study were less than 100mg/l and therefore, as categorised in DWAF (1996), within the range for all aquatic ecosystems.

Increased levels of TSS between seasons could be as a result of anthropogenic sources such as the discharge of domestic sewage, poor farming techniques, or the removal of riparian vegetation (DWAF, 1996; Grobicki, 2001; Dallas and Day, 2004; Khalife, 2005; Bilotta and Brazier, 2008; Walsh and Wepener, 2009). This was possible in the present study as an increase in runoff after rainfall in the catchment may have been the cause of the higher TSS. The 'spike' in TSS in summer was however, more likely as a result of a reduction in riparian vegetation due to dredging.

pH was considered to be within the TWQR (DWAF, 1996), ranging between 6 and 8 pH (Table 5.5, p 5-4).

COD was not significantly different and exceeded 30 mg/l in all seasons except autumn. According to Grobicki (2001), COD for raw sewage is 15 mg/l – 75 mg/l. COD for the present study was high, and could possibly have been an indication of severely polluted water. However, the COD values of the present study did not appear to have increased when considered with the mean values of COD per season, for the preceding ten years, as given by the City of Cape Town (Appendix C). According to Dallas and Day (1993), COD is used as a routine measurement of effluents and is a measure of the amount of oxygen likely to be used in the degradation of organic waste, however it is unlikely that all organic waste will be fully oxidised in aquatic ecosystems.

The conductivity values for the present study were not significantly different from each other (Table 5.6, p 5-5). Whether the results for conductivity were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined as there were no background levels for conductivity. It was therefore unknown whether these results varied more than 15% from the normal conductivity cycles under unimpacted conditions, as stipulated by DWAF (1996) and DWAF (2008). The conductivity measured in the present study, did not appear to have increased, when considered with the mean values per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

In the present study, the seasonal measurements did not vary more than 15% seasonally, but did exceed the guideline of 75 mS/m for final effluent (DWAF, 1996; Dallas and Day, 2004), in summer. While low levels of conductivity are unlikely to impact on the flora and fauna of the river (Dallas and Day, 1993; Grobicki, 2001; Dallas and Day, 2004), which is not considered to be the case in the present study, the elevated conductivity measured in the present study, in summer, could possibly have influenced the diversity and abundance of macroinvertebrate organisms. According to Newall and Walsh (2005), macroinvertebrates may be sensitive to changes in conductivity. However, according to Dallas and Day (1993), many macroinvertebrate species are able to survive and flourish at relatively high levels of

salinity, but it is often the rate of change of conductivity (salinity), that has the most critical impact on aquatic organisms.

According to DWAF (1996), Deksissa et al. (2003), Nielsen et al. (2003), and Dallas and Day (2004), increases in conductivity are often as a result of anthropogenic influences such as sewage effluent or irrigation runoff, which are often found to occur under low flows, during periods of low rainfall, when evaporation of surface waters could result in an increase in conductivity (salinity) levels. It was considered likely in the present study that the increased levels of conductivity in summer could have been due to greater evaporation rates, as there had been no rainfall in the previous seven days (Appendix D, Table D1) to flush the system, water levels were low, and there was little flow (personal observation). The other possibility was a pollution event such as a sewage spill, or the discharge of effluent into the river that caused the increase in conductivity. According to Dallas and Day (1993), increased salinity from such causes could result in a greater concentration of ions in the water, and as such an elevated conductivity result.

The possibility of the increased levels of conductivity in the present study being as a result of point source pollution is not considered likely, as the seasonal variations were not more than 15%. According to Day and Snaddon (2000), it is the rate of change that is more important to aquatic biota than the absolute values.

In the present study, the lower conductivity measurements in winter were likely due to the rainfall being greater in autumn, and the flow of the river therefore being faster, and the dilution of anthropogenic influences being more. Deksissa et al. (2003) and Nielsen et al. (2003) describe similar findings in other studies, where low flows during drought periods possibly explained poorer water quality conditions, and periods of higher rainfall more improved water quality results.

The concentrations of orthophosphates (Table 5.14, p 5-10) were not significantly different. According to DWAF (1996), an orthophosphate concentration of  $<0.005$  mg/l P in summer is representative of an oligotrophic status,  $0.005 - 0.025$  mg/l P a mesotrophic status,  $0.025 - 0.25$  mg/l P a eutrophic status, and  $>0.25$  mg/l P hypertrophic status. The mean concentrations of orthophosphates were representative of a mesotrophic ( $0.005$  mg/l P –  $0.025$  mg/l P) status in autumn and a

eutrophic (0.025 mg/l P – 0.25 mg/l P) status in all other seasons. When compared to DWAF (2008) and PD Naidoo and Associates (2010), the water quality in all seasons, was considered good in autumn and fair in all other seasons (Appendix A, Table A5).

Orthophosphate concentrations of less than 0.005 mg/l P (DWAF, 1996), are considered to be sufficiently low to reduce the likelihood of algal and plant growth. According to DWAF (1996), the effect of changes in trophic status is accompanied by the growth of algae and other aquatic plants in rivers. Trophic states are the norm in assessing the effects of orthophosphate concentrations on aquatic ecosystems (DWAF, 1996).

There were significant differences between seasons, measured per site, with total oxidisable nitrogen (TON) (mg/l N) showing significant differences between seasons, measured per site, with spring showing the highest mean TON ( $2.80 \pm 0.69$  mg/l N) and autumn ( $0.63 \pm 0.03$  mg/l N) and summer ( $0.63 \pm 0.07$  mg/l N) the lowest mean TON. Total inorganic nitrogen (mg/l N) showed significant differences between seasons, measured per site, with spring showing the highest mean concentration of total inorganic nitrogen ( $1.81 \pm 0.81$  mg/l N) compared to all other seasons. Total ammonia (mg/l N) showed significant differences between seasons, measured per site, with spring showing the highest mean concentration of total ammonia ( $1.43 \pm 0.70$  mg/l N), and autumn the lowest ( $0.03 \pm 0.03$  mg/l N). Nitrites and nitrates (mg/l N) showed significant difference between seasons, measured per site with summer and autumn being significantly different to winter and spring, with winter having the highest ( $0.54 \pm 0.06$  mg/l N) mean nitrite and nitrate concentrations and summer the lowest ( $0.00 \pm 0.00$  mg/l N). Total phosphorus (mg/l N) showed significant difference between seasons measured per site with summer having the highest ( $0.31 \pm 0.21$  mg/l P) mean total phosphorus concentrations compared to all other seasons.

### **13.1.1 Total inorganic nitrogen (mg /l N)**

Total inorganic nitrogen concentrations were significantly different, with spring having the highest mean concentration of total inorganic nitrogen ( $1.81 \pm 0.81$  mg/l N) and being significantly different to summer and autumn. Autumn had the lowest mean concentration of total inorganic nitrogen ( $0.05 \pm 0.02$  mg/l N) (Table 5.9, p 5-7). The elevated levels of concentration in winter and spring and the lower levels of

concentration in summer and autumn were true for both sites KVS1 and KVS3. Total oxidisable nitrogen (mg/l N) was also significantly different with spring being significantly different to all seasons and winter being significantly different to all seasons. Summer and autumn were also homogenous and significantly different to winter and spring.

Concentrations of total inorganic nitrogen were calculated by the addition of total ammonia and measured nitrate and nitrite data (DWAF, 1996). Nitrites and nitrates also showed significant difference between seasons with summer and autumn being significantly different to winter and spring. Both, winter and spring, as well as summer and autumn were homogenous to each other.

According to DWAF (1996), occasional increases in the inorganic nitrogen concentrations above the TWQR, are less of a concern than continuously high concentrations of total inorganic nitrogen. Change in trophic status is the norm in assessing the 'effects' of concentrations of total inorganic nitrogen on aquatic ecosystems (DWAF, 1996).

Whether the results of the present study were within the TWQR could not be determined, as it was unknown whether the concentrations measured, exceeded the natural background concentrations by more than 15%. These concentrations were however higher than the mean concentrations per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1). As categorised by DWAF (2008) and PD Naidoo and Associates (2010), the water quality in summer and autumn was almost natural, while in winter it was good, and spring it was fair (Appendix A, Table A5).

According to DWAF (1996), a concentration of  $<0.5$  mg/l N in summer is representative of an oligotrophic status,  $0.5 - 2.5$  mg/l N of a mesotrophic status,  $2.5 - 10$  mg /l N of a eutrophic status, and  $>10$  mg/l N of a hypertrophic status. In the present study, in terms of the water quality 'effect' on aquatic ecosystems (DWAF, 1996), the mean concentration of total inorganic nitrogen in the water indicated that the status of the water at the sites in summer ( $0.12 \pm 0.06$  mg/l N), and autumn ( $0.05 \pm 0.02$  mg/l N), were oligotrophic, and mesotrophic in winter ( $0.66 \pm 0.07$  mg/l N) and spring ( $1.81 \pm 0.81$  mg/l N).

As shown in Table 5.9 (p 5-7), lower concentrations of total inorganic nitrogen were recorded in autumn and summer during periods of low flow, while higher concentrations of total inorganic nitrogen were recorded in winter and spring, during periods of high flow (Table 4.1, p 4-12). According to Deksissa et al. (2004), the concentration of nitrogen is normally inversely related to flow, with the higher nitrogen concentrations occurring during the dry season when flows are at a minimum, and the lower nitrogen concentrations in the rainy season. However, according to Deksissa et al. (2004) and Chen et al. (2009), if non-point source pollution is greater than point source pollution, this could cause the opposite effect, with higher concentrations of nitrogen being associated with periods of higher flow, rainfall and runoff; and lower concentrations of inorganic nitrogen with periods of lower flow, rainfall and runoff.

The results of the present study supported the findings of Day and Snaddon (2000) and Deksissa et al. (2004). The high flows in winter and spring, could possibly have resulted in an increase in the concentration of total inorganic nitrogen, due to an increase in runoff from the catchment area, which included agricultural lands and light industrial activities (personal observation). As described by Kay (1999), fertilisers used on agricultural lands could lead to substantial increases in the dissolved nutrients in runoff entering streams. In the present study, this was possibly why spring showed a higher concentration of total inorganic nitrogen compared to winter, and why autumn showed the lowest concentration of total inorganic nitrogen. In the present study autumn was associated with the lowest flows of the year. Another possibility is that the area is subjected to more non-point source pollution than point source pollution (personal observation). In the present study it is likely that this could have accounted for the similarities between summer and autumn, and winter and spring with respect to total oxidisable nitrogen and nitrates and nitrite concentrations.

In the present study, the lower total inorganic nitrogen concentrations in autumn could also possibly be due to the establishment of semi-aquatic vegetation, which could possibly reduce the amount of total inorganic nitrogen in the water, due to the uptake of nitrogen by the vegetation. This was considered a possibility. According to Saunders and Kalff (2001), Camargo and Alonso (2006) and Chen et al. (2009), lower flows could increase the nitrogen retention through the vegetative uptake of

nitrogen by aquatic plants, and provide favourable conditions for denitrification, thus reducing concentrations of nitrogen in the water column. This could explain why autumn was significantly different to all other seasons when considering the concentrations of total inorganic nitrogen.

The total inorganic nitrogen concentrations recorded during the present study, in summer and autumn, were indicative of a low productivity system, with rapid nutrient cycling, and as a result, there was limited growth of 'nuisance aquatic plants' (DWAF, 1996). DWAF (1996) defines 'nuisance aquatic plants' as indigenous or invasive plants that can become a nuisance in rivers that have some form of nutrient enrichment. Large expanses of 'nuisance aquatic plants' can become detrimental to fish and invertebrates depending on the level of growth (DWAF, 1996). In the present study, in winter and spring, the elevated concentrations of total inorganic nitrogen were not considered exceptionally high and as categorised by DWAF (1996) were in fact representative of a productive system with 'nuisance growth' of aquatic plants and blooms of algae (personal observation). According to DWAF (1996), a productive system could be conducive to higher levels of species diversity.

### **13.1.2 Total ammonia (mg/l N )**

Total ammonia concentrations were significantly different, with spring showing the highest mean concentration of total ammonia ( $1.43 \pm 0.70$  mg/l N), compared to all other seasons. Spring was significantly different to all other seasons and all other seasons were considered homogenous. All total ammonia results exceeded the TWQR (0.007 mg/l N) (DWAF, 1996) in all seasons, but were not considered toxic, as the concentrations were lower than the Chronic Effects Value (CEV) and Acute Effects Value (AEV) thresholds. The concentrations of total ammonia thus showed a seasonal trend. The concentrations from the present study were higher than the mean concentrations, per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

In the present study the proportion of un-ionised ammonia ( $\text{NH}_3$ ) in each sample was calculated from total (soluble) ammonia, using the conversion table of DWAF (1996), and based on measured pH, and assumed summer and autumn, spring and winter temperatures of 20°C and 15°C respectively, as indicated in Table 5.11 (p 5-8). Total ammonia criteria for aquatic ecosystems, were calculated from the concentration of

total ammonia, which is the sum of  $\text{NH}_3$  and  $\text{NH}_4^+$  concentrations. The norms for assessing the effects of free ammonia on aquatic ecosystems were chronic and acute toxic effects of ammonia on aquatic organisms (DWAF, 1996).

In the present study, the ammonia concentrations (Table 5.10, p 5-7) showed a similar seasonal trend to that of the concentrations of total inorganic nitrogen, as discussed in Section 13.1.1 (p 13-4), above. Increased ammonia concentrations were associated with periods of high flows in winter and spring, and reductions in concentrations of total ammonia with periods of low flows, in summer and autumn.

Ammonia is known to be a common pollutant, and is one of the nutrients that could contribute to eutrophication of a system and excessive growth of 'nuisance aquatic vegetation' (Dallas and Day, 1993; DWAF, 1996; and Grobicki, 2001). According to DWAF (1996), Kay (1999), Dallas and Day (2004), and Moore and Palmer (2005), high levels of ammonia could come from commercial fertilisers containing highly soluble ammonia, and ammonium salts, or sewage discharge, which is transported through irrigation water, and runoff, into rivers. In the present study, it was likely that the concentrations of total ammonia in spring could have been from fertilisers that were applied to the agricultural lands or from sewage discharge. According to DWAF (1996), Moore and Palmer (2005), and Chen et al. (2009), the use of too much fertilisers which exceed the requirements of the crop, could result in excess fertiliser running off from the cultivated lands into nearby rivers. This was possible in the present study as ammonia, total inorganic nitrogen, total suspended solids and total phosphorus concentrations, showed higher mean concentrations in spring, possibly as a result of runoff containing sediments and fertilisers (nutrients) from agricultural lands. It was also a possibility that the increases in concentrations of ammonia were due to sewage discharge as a result of a malfunctioning pump station or a sewage leak in the area (personal observation). According to Grobicki (2001) and Moore and Palmer (2005), this could potentially affect macroinvertebrates in a river.

In the present study, while autumn had the lowest mean concentration of total ammonia, the sites had similar seasonal conditions to that of the sites sampled in summer. However, the rainfall seven days prior (Appendix D, Table D1) to sampling in autumn, could possibly have flushed the system of ammonia, or diluted the concentrations. According to Deksissa et al. (2003) and Vos and Roos (2005), rainfall

could determine the amount of flow in a river and therefore the degree of dilution of natural chemical constituents, and of pollutants. The lower concentration of total ammonia in summer, autumn and winter could have counted for the similarity between seasons.

### **13.1.3 Total Phosphorus (mg/l P)**

There were significant differences in total phosphorus concentration per season, measured per site. Summer and autumn were homogenous as were winter and spring, however these homogenous pairs were significantly different to each other. Summer had the highest mean concentration of total phosphorus and autumn the lowest.

In the present study, the 'increased in total phosphorus and orthophosphate concentrations, in summer and spring, could have resulted from point source discharges, such as domestic effluents, sewage, and diffuse sources in which the phosphorus load is generated by surface and subsurface drainage. Such non-point source pollution could arise from urban runoff, and in particular, drainage of land on which fertilisers have been applied (DWAF 1996; Kay, 1999; Dallas and Day, 2004; Newall and Walsh, 2005; He et al., 2006; Oberholster et al., 2008). In the present study, this was likely in spring, as phosphorus concentrations were higher than autumn and winter, but was unlikely in summer, due to the lack of rainfall. The increased concentrations of phosphorus and orthophosphates in summer were possibly from point source discharges (pollution), and not from non-point source discharges.

According to Dallas and Day (1993), the phosphorus retention time in streams could also be affected by the flow regime and water residence time, which could result in low flows of a lotic system resembling a lentic system (standing water), with respect to nutrient retention. In the present study it was likely that this was a contributing factor, as the concentrations recorded in summer were high. According to Vos and Roos (2005), phosphorus loads could accumulate in systems in periods of lower flow, and could result in higher phosphorus concentrations in the drier periods. It was also possible that phosphorus remained in the system after periods of rainfall, which would have generated runoff containing higher phosphorus loads, such as the runoff associated with spring in the present study.

### 13.2 Macroinvertebrates

Of the nineteen different taxa identified (Annelida, Crustacea, Decapoda, Ephemeroptera, Odonata, Hemiptera, Coleoptera), the ANOVA showed that there were no significant differences in the abundance, or diversity, of macroinvertebrates between seasons, or sites, in the Keysers River (Table 5.2, p 5-2 and Table 5.3, p 5-3). Even though the data were similar, there was a 'spike' in the number of different taxa, and abundance of macroinvertebrates in summer and autumn compared to winter and spring.

Of the taxa sampled, the average sensitivity score per taxon for all seasons was less than 5 on the sensitivity scale derived from the tolerance of the taxa to pollution, as used in the SASS5 scoring system (Dickens and Graham, 2002). As classified by Gerber and Gabriel (2002), these taxa were generally hardy taxa (Day and Snaddon, 2000). Half the taxa sampled were associated with particularly low oxygen levels and organic enrichment, while the other half sampled scored above 5 on the sensitivity scale and are associated with a moderate tolerance to pollution (Gerber and Gabriel, 2002). There were also a few very sensitive taxa identified (Amphipoda, Aeshnidae, Helodidae), scoring above 10 on the sensitivity scale (Gerber and Gabriel, 2002), which were only associated with low pollution. The sensitivity score per taxon for each season is shown in Table 5.1 (p 5-1), and was similar for all seasons in the present study.

The similarity in the diversity of taxa, and abundance of taxa, in summer and autumn, could possibly be due to the oligotrophic status of the water. According to DWAF (1996), oligotrophic systems are usually associated with moderate levels of species diversity. In the present study, the similarity could also be attributed to the low water levels and sandy substrate which may have allowed for the establishment of 'aquatic in-stream vegetation', thus reducing the active channel and providing more habitat, which was likely to support a larger diversity of taxa. This theory was supported in studies undertaken by Grobicki (2001), Ogbeidu and Oribhabor (2002), King et al. (2003), Bernhardt and Palmer (2007) and Feio et al. (2007). In the present study, of the taxa sampled (Table 5.1, p 5-1), the majority favoured quiet pools, slow flowing streams, coarse sand or muddy substrates, and vegetation, as described by Gerber and Gabriel (2002).

The similarity between the diversity and abundance of macroinvertebrates, sampled in winter and spring, could be due to an increase in runoff or greater flows. According to Dallas and Day (1993), Grobicki (2001) and Dallas and Day (2004), greater flows and scouring events due to increased runoff and rainfall, could result in a lesser number of taxa and abundance of individual macroinvertebrates. This was likely in the present study, as the reduced diversity and abundance of macroinvertebrates, recorded in winter and spring, could be due to greater flows, scouring or flooding events, which would require taxa to recolonise the area. According to Sawyer et al. (2004), TSS concentrations can affect the abundance and diversity of macroinvertebrates. This is unlikely in the present study, as the TSS results were not significantly different, and there was a noticeable seasonal trend, with there being a greater concentration of TSS in summer and spring, than in winter and autumn (Table 5.4, p 5-3).

In the present study, all sites and seasons were similar. Site KVS1 showed a reduced diversity of taxa in winter but it was not significant. This was possibly due to site KVS1 being directly below a road bridge, and at the end of a concrete canal. It could be assumed that the velocity of the water may have been higher at site KVS1 than other sites, in winter (personal observation). According to Dallas and Day (1993) and Grobicki (2001), increased flow and scouring events could result in a reduction in the diversity and abundance of macroinvertebrates at a site. In the present study, the reduction in taxa was possibly due to scouring, but it is assumed that there was sufficient marginal vegetation and aquatic vegetation to provide adequate calmer areas for the protection of some of the taxa, which explains the reduction in taxa but not the total removal of taxa. The increase in taxa at sites KVS2 and KVS3 could be because the river widened, slowing the velocity of the flow (personal observation). This was consistent with the findings of Dallas and Day (1993) as discussed previously. Sawyer et al. (2004) further suggested that a reduction in taxa could be as a result of significant total ammonia results. This is unlikely in the present study, as the total ammonia concentrations discussed in Section 13.1.2 (p 13-7) were not considered toxic.

Site KVS3 in summer reflected a reduced abundance of macroinvertebrates. According to Sawyer et al (2004), TSS could be responsible for the reduction in the abundance of macroinvertebrates at a particular site. This was a possibility in the

present study, when considering the results for TSS in summer (Table 5.4, p 5-3). However, it was not considered the only contributing factor. It was observed on site, that the river was deeper and the banks steeper, resulting in less marginal vegetation to be sampled and less substrate to be included in the sampling. It was also heavily infested with 'nuisance aquatic plants' which may have resulted in fewer taxa and individual macroinvertebrates being sampled.

The similarity of abundance and diversity of macroinvertebrates, in the present study, at all sites, and in other seasons, was likely to be associated with the extent of habitat and the biotopes available. The Keysers River is a sand canal (Chapter 4, p 4-1), and water quality was representative of a productive system (mesotrophic), with substantial marginal and 'nuisance' aquatic growth, which may have resulted in a moderate level of species diversity, with sufficient habitat to support a considerable variety and abundance of macroinvertebrates. According to Grobicki (2001) and several other authors (Ogbeidu and Oribhabor, 2002; King et al., 2003; Bernhardt and Palmer, 2007; Feio et al., 2007), species diversity can be associated with the availability of habitat and the lack of suitable habitat could severely limit the potential for colonisation by aquatic organisms. In the present study, the non-canalisation of this section of the Keysers River had proved successful as it provided a suitable substrate for the establishment of marginal vegetation which provided for a higher diversity of aquatic and semi-aquatic fauna. This study supports similar statements made by Day and Snaddon (2000).

### **13.3 Diatoms**

There were no significant differences in the diversity or the abundance of diatoms, between sites and seasons, summer and winter, in the Keysers River (Chapter 5, Section 5.3, p 5-10).

A total of eighty species representing thirty one genera were identified (Table 5.15, p 5-11). The assemblages were dominated by *Cocconeis placentula var. euglypta* (Ehrenberg), at sites KVS1 and KVS3, in winter. Summer samples were dominated by *Planothidium frequentissimum* (Lange-Bertalot), at site KVS1 in summer, and *Cocconeis placentula* (Ehrenberg), at site KVS3. Of the dominant species found in South African waters, it has been confirmed by Bate et al. (2004), that many of the

species are in fact cosmopolitan species, and are not endemic to South Africa, as they occur and are well recorded internationally (<sup>1</sup>Harding, 2010, pers comm).

The Specific Pollution Sensitivity Index, as referred to by Van Dam et al. (1994) and categorised by Coste in Cemagraf (1982), is widely used as a preferred index to monitor water quality (Newell and Walsh, 2005; Beyene et al., 2009). In the present study, the diatom species were classified ecologically according to Van Dam et al. (1994). The ecological classification is discussed by several other authors, including Taylor et al. (2005), Weilhoefer and Pan (2007), and Resende et al. (2009).

As shown in Table 5.16 (p 5-14) the mean Specific Pollution Index (SPI) assigned to the assemblages for sites KVS1 and KVS3, in the present study in winter, were between 9 and 13, and therefore were indicative of moderate water quality (Appendix A, Table A2). The mean SPI in summer was between 5 and 9, and therefore indicative of poor water quality. The percentage of Pollution Tolerant Values (%PTV) for sites KVS1 and KVS3 in winter and summer were also high, with assemblages in winter being 22.1% tolerant at site KVS1, and 21.6% tolerant at site KVS3, compared to 24.6% tolerant in summer at site KVS1, and 13.7% tolerant in summer at site KVS3. As the SPI scores were similar between sites KVS1 and KVS3 in winter and summer it could be concluded that there were no differences in the ecological integrity of the sites between seasons. This was supported in similar findings in other studies by Beyene et al. (2009).

In the present study in winter, both sites KVS1 and KVS3 showed assemblages of diatoms which were associated with a eutrophic status, having low oxygen requirements (>30% saturation), having occurred in waters with a pH greater than 7 (alkaliphilous), having been tolerant of moderately polluted water ( $\beta$ -mesosaprobous), and which had elevated concentrations of organically bound nitrogen (Nitrogen autotrophic – tolerant) (Table 5.19, p 5-16 and Appendix A, Table A3).

Summer at site KVS1 showed diatom assemblages associated with an oligo-eutrophic status and very highly polluted water ( $\alpha$ -meso – polysaprobous). Site KVS3

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showed assemblages associated with a eutrophic status, with water having been moderately polluted ( $\beta$ - mesosaprobous). Both sites in summer showed assemblages of diatoms having had low oxygen requirements ( $>30\%$  saturation), having occurred in waters with a pH greater than 7 (alkaliphilous), as well as having been tolerant of elevated concentrations of organically bound nitrogen (Nitrogen autotrophic – tolerant). The water chemistry results for winter and summer, as discussed in section 13.1 (p 13-1), were not consistent with the poor water quality conditions identified by the assemblage of diatoms.

The assemblages of diatoms between sites showed no significant differences, and indicated the water quality to be worse than indicated by the water chemistry. The assemblages of the dominant diatom taxa were also representative of assemblages of diatoms that could have been associated with nutrients, and the physical conditions in the river. This was similar to the findings of Weillhoefer and Pan (2007), Taylor et al. (2005) and Resende et al. (2009).

The diatom genera identified as being present in all samples, in the present study, included *Nitzschia spp.*, *Navicula spp.*, *Sellaphora spp.*, *Cyclotella spp.*, *Gomphonema spp.*, *Eolimnas spp.*, *Cocconeis spp.*, *Melosira spp.*, *Hippodonta spp.*, *Staurosira spp.*, and *Planothidium spp.* Of these, some *Nitzschia spp.*, *Gomphonema spp.*, and *Eolimnas spp.*, are common nitrogen heterotrophic taxa and are tolerant of high nutrient levels (nitrogen heterotrophic and nitrogen autotrophic) and moderate to high levels of pollution (polysaprobous and mesosaprobous). Some *Nitzschia spp.*, *Navicula spp.*, *Sellaphora spp.*, *Cocconeis spp.*, *Melosira spp.* and *Cyclotella spp.*, are species consistently found in eutrophic waters and are tolerant of polluted water (Taylor et al., 2004; Weillhoefer and Pan, 2007; Resende et al., 2009).

It is concluded that the diatom assemblages of the present study, and the respective SPI scores, and percentages of PTV's for each site and season, were not entirely consistent with the water chemistry results. The SPI score did decrease between winter and summer. This was possibly due to higher rainfall and the dilution of pollution, therefore implying the water was of a moderate quality in winter, but of a poor quality in summer. The findings of the present study were similar to those discussed by Taylor et al. (2007), when testing the diatom-based indices in the Vaal and Wilge Rivers in South Africa.

### 13.4 Conclusion

The Keysers River has over time been subjected to severe impacts which resulted in poor water quality. In the present study, it was evident that the soft channelisation permitted the establishment of physical habitat, which provided for a potentially higher diversity of aquatic fauna, thus making the Keysers River an important system to preserve. There appeared to be a strong relationship between the mean macroinvertebrate counts, and the availability of suitable habitat, making habitat availability the factor most limiting to the presence of aquatic invertebrates with water quality being subsidiary. According to Grobicki (2001) and Findlay and Taylor (2006), water quality is a limiting factor, but habitat is a greater limiting criterion. According to Grobicki (2001), Ogbeidu and Oribhador (2002) and Bernhardt and Palmer (2007), the provision of suitable habitat is the primary constraint behind the re-establishment of biotic diversity at a site.

According to Soininen and Könönen (2004) and Feio et al. (2007), macroinvertebrates show a higher sensitivity to morphological change in channel and vegetation integrity, while diatoms are more related to water quality. Therefore if habitat was to be reduced in the Keysers River it could be assumed that this would negatively impact on the diversity and abundance of the macroinvertebrates.

In the present study, the Keysers River water quality was fair to good, but subject to nutrient enrichment in winter and spring, with concentrations of total inorganic nitrogen representing a mesotrophic status, while in summer and autumn being representative of an oligotrophic status, when runoff is at its lowest. According to Parr and Mason (2003), eutrophication-induced changes in macroinvertebrate assemblages could occur over a longer period of time compared to that sampled in one study, and there must be sufficient heterogeneity in habitat for assemblages to survive changing conditions. This was likely in the present study, as there were no significant differences in the diversity and the abundance of macroinvertebrates, between sites or seasons.

The assemblage of diatoms identified at each site during winter and summer was indicative of eutrophic conditions and oligo-eutrophic conditions, respectively. The assemblages of diatoms were indicative of very polluted water in summer, and moderately polluted water in winter. This was also the case in the scoping study

completed by Day and Snaddon (2000). While the eutrophication of the sites in the various seasons was consistent with the water chemistry findings, the diatom assemblages indicated that the water quality was possibly not as good in summer and autumn as indicated, and the water may in fact have been more polluted.

In the present study, the action of the soft channelization and the lack of hardened river bed and banks, has allowed for the establishment of marginal vegetation along the banks and substrate of the river. This provided a more diverse habitat for the establishment of macroinvertebrates. Improvements in agricultural practices upstream could possibly have a significant effect on the water quality of the river system. This could result in reducing the level of nutrients in spring, as could the reduction of sewage discharges, and improvement of the water quality of runoff, possibly from the adjacent industrial area, and surrounding roads. This could be done by filtering all runoff through a seasonal wetland before it enters the Keyzers River.

## CHAPTER 14

### DISCUSSION: LITTLE LOTUS RIVER

#### 14.1 Water chemistry

The South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996), consists of a water quality 'effect', a Target Water Quality Range (TWQR) and water quality criteria, the Chronic Effects Value (CEV), and the Acute Effects Value (AEV) of various parameters. The present results were compared with the South African Water Quality Guidelines for Aquatic Ecosystems water quality 'effects' and TWQR (DWAF, 1996).

No significant differences were found between sites LL1 and LL4, for any of the water chemistry parameters measured (Chapter 6, Section 6.2, p 6-3). There were also no significant differences in total suspended solids (TSS), between seasons, measured per site.

TSS concentrations were not significantly different. Whether the results of the present study for TSS concentrations were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined as there were no background data of TSS levels sampled. It was therefore unknown as to whether these results varied more than 10% to that of the background levels, as stipulated by DWAF (1996) and DWAF (2008). According to Palmer et al. (2005), in the lower reaches of rivers there are no unimpacted sites and reference conditions are difficult to determine. The results of the present study, were however less than 100mg/l, and therefore within the range for all aquatic ecosystems, as classified by DWAF (1996).

Increased levels of TSS between seasons could have been as a result of anthropogenic sources, such as the discharge of domestic sewage, or the removal of riparian vegetation (DWAF, 1996; Grobicki, 2001; Dallas and Day, 2004; Khalife, 2005; Bilotta and Brazier, 2008; Walsh and Wepener, 2009). This was a possibility in the present study, as an increase in runoff after rainfall in the catchment, may have been the cause of the higher TSS concentrations. The elevated concentrations of TSS in autumn were however considered more likely to be due to a reduction in

riparian vegetation, as a result of dredging, or a point source pollutant (personal observation).

There were significant differences between seasons, measured per site, for pH with spring having the lowest ( $7.25 \pm 0.21$ ) pH compared to all other seasons. Conductivity (mSm), showed summer having the highest mean conductivity ( $163.00 \pm 0.00$  mS/m), and winter the lowest ( $74.50 \pm 74.50$  mS/m), compared to all other seasons. Chemical Oxygen Demand (COD) showed significant differences between seasons, measured per site, with autumn having the lowest mean COD ( $19.00 \pm 9.90$  mg/l), compared to all other seasons. There were significant differences between seasons, measured per site, for total oxidisable nitrogen (TON), with autumn having the lowest mean TON ( $0.82 \pm 0.11$  mg/l N), compared to all other seasons, while summer showed the highest mean TON ( $4.01 \pm 2.03$  mg/l N). Total inorganic nitrogen showed a significant difference between autumn and all other seasons with autumn having the lowest mean total inorganic nitrogen ( $0.12 \pm 0.02$  mg/l N) compared to any other season. Total ammonia showed significant differences between summer and all other seasons, and between autumn and winter. Summer had the highest mean concentration of total ammonia ( $2.75 \pm 1.64$  mg/l N), compared to all other seasons, and autumn the lowest concentration of total ammonia ( $0.03 \pm 0.00$  mg/l N). There were significant differences between seasons per site for soluble nitrates and nitrites, with winter showing the highest nitrate and nitrites ( $2.76 \pm 0.04$  mg/l N), compared to all other seasons, and summer the lowest ( $0.03 \pm 0.01$  mg/l N). Total phosphorus was significantly different with summer showing the highest mean concentration of total phosphorus ( $0.91 \pm 0.19$  mg/l P) compared to all other seasons, while winter had the lowest mean concentration of total phosphorus ( $0.13 \pm 0.00$  mg/l P). Orthophosphates showed significant differences, with summer having the highest mean orthophosphates concentration ( $0.57 \pm 0.17$  mg/l P) compared to all other seasons, and autumn the lowest mean concentration ( $0.06 \pm 0.00$  mg/l P), compared to all other seasons.

#### **14.1.1 pH**

pH was within the TWQR (DWAF, 1996), range of 6 and 8 pH. The pH measurement for the present study was significantly different between seasons. The mean pH for summer, winter and autumn were homogenous, while the pH for spring was significantly different to the pH for summer and autumn. The pH ranged between 7.25

in spring and 7.8 in autumn and summer (Table 6.5, p6-4). These results were considered to be within the TWQR of 6 and 8pH (DWAF 1996), which in terms of South African freshwaters, are considered neutral. According to DWAF (1996) and Dallas and Day (2004), most freshwaters of South Africa are well buffered. While the TWQR values, according to Grobicki (2001) and Dallas and Day (2004), are considered alkaline as they flow through limestone, the results of the present study are considered normal for rivers on the Cape Flats.

According to Dallas and Day (1993) and DWAF (1996), elevated levels of pH could be associated with an increase in biological activity in eutrophic systems, as well as, with an increase in temperature and lower water levels. This was likely in the present study. According to Brown et al. (1998) and Dallas and Day (2004), high pH values could result in the proportion of ammonia relative to other forms of nitrogen becoming higher, which could be a problem as ammonia is toxic to many organisms. As discussed below, this was the case in the present study.

#### **14.1.2 Conductivity mS/m**

The conductivity values for the present study were significantly different (Table 6.6, p 6-4), between summer and all other seasons, and between autumn and all other seasons. Spring and winter were homogenous. Summer had the highest mean conductivity compared to all other seasons. Whether the results for conductivity were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined, as there were no background levels for conductivity. It was therefore unknown whether these results varied more than 15% from the normal conductivity cycles under unimpacted conditions, as stipulated by DWAF (1996). The conductivity measured in the present study, did not appear to have decreased, when considered with the mean conductivity, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

In the present study, summer showed the highest conductivity ( $163.00 \pm 0.00$  mS/m) and winter the lowest conductivity ( $74.50 \pm 0.71$  mS/m). These results exceeded the guideline (75 mS/m) for final effluent (DWAF, 1996; Dallas and Day, 2004), in summer and autumn, and were considered the equivalent to that of final effluent in

winter and spring. This accounts for the homogeneity of winter and spring conductivity values.

According to Dallas and Day (1993), Grobicki (2001), Dallas and Day (2004), and Newall and Walsh (2005), low levels of conductivity are unlikely to impact on the flora and fauna of a river, but elevated results could possibly influence the diversity and abundance of macroinvertebrate organisms that may be sensitive to changes in conductivity. This was likely in the present study. According to Dallas and Day (1993), many macroinvertebrate species are able to survive and flourish at relatively high levels of salinity, but it is often the rate of change of conductivity (salinity), that has the most critical impact on aquatic organisms.

According to DWAF (1996), Deksissa et al. (2003) and Dallas and Day (2004), increases in conductivity are often as a result of anthropogenic influences such as sewage effluent or irrigation runoff. According to Nielsen et al. (2003) increases in conductivity could be associated with low flows, during periods of low rainfall, when evaporation of surface waters is greatest. This was likely in the present study as levels of conductivity in summer, could have been due to greater evaporation rates, as there had been no rainfall in the previous seven days to flush the system, and the water levels were low with little flow (Table 4.1, p 4-12). It was a possibility that this could also have been possibly because of a pollution event, such as a sewage spill or the discharge of effluent, into the river. According to Dallas and Day (1993), increased salinity from such causes could result in a greater number of ions in the water and as such an elevated conductivity result.

In the present study, autumn was significantly different to winter and spring, with winter having the lowest mean conductivity level (Table 6.6, p 6-4). This was possibly as a result of the rainfall that was recorded seven days prior to sampling. The lower levels of conductivity in winter and spring was likely due to the volume of rainfall being seasonally more than in autumn, the flow of the river being faster, and the dilution of anthropogenic influences being more. According to Deksissa et al. (2003) and Nielsen et al. (2003), low flows during drought periods could explain poorer water quality conditions, and periods of higher rainfall more improved water quality conditions. In the present study, while autumn did not have the highest conductivity level, it was higher than the conductivity measured in winter and spring, which could

possibly have been associated with the first rains (first flush) of the winter season, and therefore the runoff may have contained higher concentrations of effluent or pollution from the surrounding areas.

#### **14.1.3 Chemical oxygen demand (COD) (mg/l)**

COD was significantly different between winter and autumn, winter spring and summer were homogenous (Table 6.7, p 6-5). Autumn had the lowest mean COD ( $19.00 \pm 9.90$  mg/l) and was within the TWQR, as stipulated by DWAF (1996). The mean values of COD, exceeded the acceptable range 30mg/l in summer ( $48.50 \pm 3.54$  mg/l), winter ( $69.50 \pm 2.12$  mg/l), and spring ( $48.50 \pm 3.54$  mg/l). The COD measurements of the present study were considered to be consistent with the mean values of COD, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

According to Dallas and Day (1993), COD is a routine measurement for effluents, and is a measure of the amount of oxygen likely to be used in the degradation of organic waste. It is however unlikely that all organic waste, in aquatic ecosystems, will be fully oxidized. It is therefore recommended that the effluent in a stream should not exceed 30mg/l as stipulated in Government Gazette No.9225, Regulation No 991 18 May 1984. Considering that COD for raw sewage is 15 mg/l – 75 mg/l (Grobicki, 2001), the results of the present study, were considered high and could possibly have been an indication of severely polluted water (personal observation).

According to Walsh and Wepener (2009), an increase in COD measurements could potentially indicate higher levels of pollution in the stream, either due to an increase in temperature, or possibly sewage effluent entering the stream. In the present study, summer, winter and spring had higher COD values, and were considered more polluted than autumn. This was possibly due to the increase in water temperature, and the reduction in flow in summer, and increased runoff and pollution entering the stream in winter and spring, due to more rainfall. This could account for the homogeneity between these seasons.

According to Schulz et al. (2003) and several other authors (Price and Probert, 1997; Song et al., 2007; Li et al., 2008), the filtration of effluent and surface water through reeds and aquatic vegetation, could result in a reduction of effluent in rivers. In the

present study, lower COD measurements in autumn could therefore be attributed to the river being heavily vegetated with marginal vegetation, aquatic and semi-aquatic vegetation, which could have reduced the COD, as the vegetation could have taken up much of the nutrients associated with effluents, from nearby Informal settlements.

#### **14.1.4 Total inorganic nitrogen (mg/l N)**

Total inorganic nitrogen concentrations were significantly different between seasons with autumn having the lowest mean total inorganic nitrogen and being significantly different to all other seasons. TON concentrations were also significantly different, with autumn having the lowest mean TON ( $0.82 \pm 0.11$  mg/l N) compared to all other seasons (Table 6.8, p 6-6). The elevated concentration of TON in summer, winter and spring, and the reduced concentration of TON in autumn, were true for both sites LL1 and LL4. The mean concentrations of total inorganic nitrogen, per season, in the present study, did appear to be higher than the mean concentrations per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1). Whether the results of the present study were within the TWQR could not be determined as it was unknown, whether these concentrations exceeded the natural background concentrations of the specific system, by more than 15%. These levels were not required for the purpose of the present study.

In the present study, concentrations of total inorganic nitrogen were calculated by the addition of total ammonia and measured nitrate and nitrite data, as stipulated in DWAF (1996). The mean concentrations of nitrite and nitrate (Table 6.12, p 6-9) were significantly different between seasons, measured per site with winter being significantly different to all seasons and spring being significantly different to all seasons. Winter showed the highest mean soluble nitrate and nitrite concentrations ( $2.76 \pm 0.04$  mg/l N), compared to all other seasons, and summer the lowest mean concentrations ( $0.03 \pm 0.01$  mg/l N). According to DWAF (1996), Saunders and Kalff (2001) and Dallas and Day (2004), nitrate and nitrite ions are major nutrients that are associated with excessive plant growth and contribute to eutrophication of a system.

The reduction in the concentration of nitrates and nitrites in summer, in the present study, was considered likely, as according to Camargo and Alonso (2006), macrophytes are often associated with the uptake of nitrogen from the water, which could result in a decrease in concentrations of total inorganic nitrogen, and an

increase in plant growth. It was a possibility, in the present study, that the reduction in the nitrates and nitrites in summer may, possibly be associated with the uptake of nutrients by aquatic vegetation. Extensive plant growth was observed by the researcher in the canal during the period of low flow (summer). Summer having the lowest mean concentration of nitrates and nitrites, was not consistent with the trend for the mean concentration of total inorganic nitrogen, and TON, as summer had the highest mean concentration of total inorganic nitrogen, and TON.

According to DWAF (1996) occasional increases in the concentration of total inorganic nitrogen above the TWQR, are less of a concern than continuously high concentrations of total inorganic nitrogen. Change in trophic status is the norm in assessing the effects of nitrogen on aquatic ecosystems (DWAF, 1996). As categorised in DWAF (1996), a concentration of total inorganic nitrogen in summer of  $<0.5\text{mg/l N}$  implies an oligotrophic trophic status,  $0.5 - 2.5 \text{ mg/l N}$  a mesotrophic status,  $2.5 - 10 \text{ mg/l N}$  a eutrophic status, and  $>10 \text{ mg/l N}$  a hypertrophic status. As categorised by DWAF (1996), in terms of the water quality 'effect', the concentration of total inorganic nitrogen in the present study, was considered representative of a eutrophic status in summer ( $2.78 \pm 1.63 \text{ mg/l N}$ ) and winter ( $2.88 \pm 0.03 \text{ mg/l N}$ ), an oligotrophic status in autumn ( $0.12 \pm 0.02 \text{ mg/l N}$ ), and a mesotrophic status in spring ( $1.84 \pm 0.00 \text{ mg/l N}$ ). Autumn showed the lowest concentration of total inorganic nitrogen (Table 6.9, p 6-6), compared to all other seasons. As categorised by DWAF (2008) and PD Naidoo and Associates (2010), the water quality of the present study, was considered fair in summer, winter and spring, and good in autumn (Appendix A, Table A5).

According to Deksissa et al. (2004) and Chen et al. (2009), the concentration of nitrogen is normally inversely related to flow, with higher nitrogen concentrations occurring during the dry season when flows are at a minimum, and the lower nitrogen concentrations in the rainy season. However, if the non-point source pollution is greater than the point source pollution, this could be inversed, and then there would be higher concentrations of nitrogen during periods of higher flow and rainfall, and lower concentrations of nitrogen, during periods of lower flow. This was likely in the present study, as autumn flows were the lowest (personal observation), compared to all other seasons. It is also possible that the Little Lotus River received more non-point source pollution than point source pollution.

The lower nitrogen concentrations in autumn, in the present study, could also possibly be due to the establishment of semi-aquatic vegetation, which could have reduced the concentration of nitrogen in the water, due to the uptake of nitrogen by the vegetation. According to Saunders and Kalff (2001), Camargo and Alonso (2006) and Chen et al. (2009), lower flows increase nitrogen retention through the vegetative uptake of nitrogen by aquatic plants, and provides favourable conditions for denitrification, which could result in reduced nitrogen concentrations in a system. According to Deksissa et al. (2004) and Saunders and Kalff (2001), reduced flows and runoff, and an increase in aquatic vegetation could result in a decrease in the concentration of total inorganic nitrogen. In the present study this was possible, as reduced flows in autumn could have resulted in a decrease in the concentration of total inorganic nitrogen, while in winter and spring, the concentration of total inorganic nitrogen was higher due to non-point source pollution entering the river being greater.

As for summer having equally high concentrations of total inorganic nitrogen as winter, it was a possibility that there were point source pollutants present. In the present study, it was possible that, point source pollutants, and the lack of flow in summer could have resulted in higher concentrations of total inorganic nitrogen. According to Kay (1999) and Deksissa et al. (2004) runoff from irrigated land, could also cause higher concentrations of total inorganic nitrogen in summer.

#### **14.1.5 Total ammonia (mg/l N)**

Total ammonia concentrations were significantly different, with summer showing the highest mean concentration of total ammonia ( $2.75 \pm 1.64$  mg/l N), compared to all other seasons (Table 6.10, p 6-7). Spring and winter were homogenous and autumn was significantly different to winter. All concentrations of total ammonia exceeded the TWQR (0.007 mg/l N), as stipulated by DWAF (1996), in all seasons. In summer at site LL4, the total ammonia exceeded the Acute Effect Value (AEV) (0.1 mg/l N) at a pH of 8, and at LL1 exceeded the Chronic Effect Value (CEV) (0.015) at a pH of 8. While all other seasons exceeded the TWQR (DWAF, 1996), the concentrations of total ammonia were not considered toxic, as the concentrations were lower than the CEV (0.015 mg/l N) and AEV (0.1 mg/l N) thresholds, as stipulated in DWAF (1996). The mean concentrations of total ammonia, of the present study, did not appear to be

higher than the mean data, per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

In the present study, the proportion of un-ionised ammonia ( $\text{NH}_3$ ) in each sample, was calculated from total (soluble) ammonia using the conversion table of DWAF (1996), and based on measured pH and assumed summer and autumn, spring and winter temperatures, of 20°C and 15°C respectively, as indicated in Table 6.11 (p 6-8). Ammonia criteria for aquatic ecosystems were calculated from the concentration of total ammonia which is the sum of  $\text{NH}_3$  and  $\text{NH}_4^+$  concentrations. The norms for assessing the effects of free ammonia on aquatic ecosystems were chronic and acute toxic effects of ammonia on aquatic organisms (DWAF, 1996).

According to Dallas and Day (2004), Deksissa et al. (2004) and Camargo and Alonso (2009), elevated concentrations of total ammonia could be associated with periods of low flows, and a reduction in the concentration of total ammonia, with periods of high flows. This is likely in the present study, as summer showed high concentrations of total ammonia, and was associated with periods of low flows, while winter showed a lesser concentration of total ammonia, and was associated with higher flows. This was also consistent with the personal observations of the researcher, as summer had not received rainfall at least fourteen days prior to sampling, while all other seasons had rainfall seven days prior to sampling (Appendix D, Table D1 and Table 4.1, p 4-12).

According to DWAF (1996), Kay (1999) and Dallas and Day (2004), high concentrations of total ammonia could also be due to commercial fertilisers containing highly soluble ammonia and ammonium salts which are being transported through irrigation water, or possible sewage discharge. According to Grobicki (2001), this could affect macroinvertebrates. In the present study, it was possible that point source pollution such as sewage effluent from the informal settlement, or fertilisers from the adjacent sports field, could have been the cause of elevated concentrations of total ammonia. According to Dallas and Day (1993) and DWAF (1996), the most significant factors that affect the proportion and toxicity of un-ionised ammonia in aquatic ecosystems, are water temperature and pH. In the present study, the increased water temperature, and the low water levels in summer, could possibly

have contributed to higher concentrations of total ammonia than was shown in other seasons.

Ammonia is known to be a common pollutant, and is one of the nutrients that can contribute to eutrophication, and excessive growth of 'nuisance aquatic vegetation' (Grobicki, 2001; DWAF, 1996; and Vos and Roos, 2005). This was not the case in the present study, as while the canal was vegetated, it was not with aquatic weeds but rather semi-aquatic terrestrial vegetation. This vegetation could possibly have restricted flow in summer, and therefore contributed to the increase in temperature of the pools and concentrations of total ammonia. According to Vos and Roos (2005), evaporation could also be associated with higher concentrations of dissolved substances in surface waters. This was also a possibility in the present study. Vos and Roos (2005), also suggest that rainfall determines the amount of flow in a river, and therefore the degree of dilution of natural chemical constituents and of pollutants. In the present study there was less vegetation growth in the canal in autumn than in summer, thus potentially reducing the residual time of the ammonia. While autumn displayed similar seasonal characteristics with regards to water temperature and flow to that of summer, the rainfall seven days prior to sampling, could possibly have flushed the system of the high concentrations of total ammonia, or diluted these concentrations. This is probably why autumn was significantly different to summer and winter. Winter and spring were also considered homogenous as the mean concentrations of total ammonia were similar.

#### **14.1.6 Total phosphorus and orthophosphates (mg/l P)**

Total phosphorus concentrations were significantly different but no pairwise difference could be established between seasons. Orthophosphate concentrations were significantly different between summer and all other seasons, while spring autumn and winter were homogenous. These trophic states are the norm in assessing the effects of orthophosphate concentrations, on aquatic ecosystems (DWAF, 1996). According to this author, an orthophosphate concentration of <0.005mg/l P in summer implies an oligotrophic status, 0.005 – 0.025 mg/l P a mesotrophic status, 0.025 – 0.25 mg/l P a eutrophic status, and >0.25 mg/l P a hypertrophic status.

In the present study, orthophosphate concentrations were representative of a hypertrophic status ( $> 0.25$  mg/l P) in summer, and a eutrophic status ( $0.05$  mg/l P –  $0.25$  mg/l P) in all other seasons. Summer had the highest mean concentrations of total phosphorus ( $0.91 \pm 0.19$  mg/l P) (Table 6.13, p 6-9) and orthophosphates ( $0.57 \pm 0.17$  mg/l P) (Table 6.14, p 6-10). Autumn had the lowest mean concentration of orthophosphates ( $0.06 \pm 0.00$  mg/l P). These mean concentrations were consistent with the data, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1). As categorised by DWAF (2008) and PD Naidoo and Associates (2010), the water quality of the present study, was considered unacceptable in summer, and fair in autumn, winter and spring (Appendix A, Table A5).

According to Dallas and Day (1993) and DWAF (1996), orthophosphates is that phosphorus which is immediately available to aquatic biota. Orthophosphate concentrations of less than  $0.005$  mg/l P are considered to be sufficiently low, to reduce the likelihood of algal and plant growth. The effect of increases in trophic status is accompanied by the growth of algae, and other aquatic plants in rivers (DWAF, 1996; Dallas and Day, 2004).

According to DWAF (1996), Dallas and Day (2004) and He et al. (2006), phosphorus is considered to be the principle nutrient in controlling the degree of eutrophication, and algal blooming is often related to increased concentrations of phosphorus in freshwater systems. In the present study, the concentration of orthophosphates was considered representative of a hypertrophic status in summer. According to DWAF (1996), increased concentrations in total phosphorus and orthophosphates are normally associated with an increase in rainfall and runoff. In the present study, it was likely that the increased concentrations of phosphorus in summer resulted from point source discharges (pollution), such as sewage, and not from non-point source discharges, as there was no rain fourteen days prior to sampling (personal observation), and therefore reduced runoff from the greater catchment. According to Dallas and Day (1993), the total phosphorus and orthophosphate retention time in streams could be affected by the flow regime, and water residence time, which could result in low flows of a lotic system resembling a lentic system, with respect to nutrient retention. This was likely a contributing factor in the present study, as the concentrations for total phosphorus and orthophosphates in summer were high.

According to Vos and Roos (2005), phosphorus loads could also accumulate in a system in periods of lower flow and result in higher phosphorus concentrations in the dry season. This was also a possibility in the present study.

According to Dallas and Day (1993), Grobicki (2001), Vos and Roos (2005) and He et al. (2006), a reduction in the concentration of orthophosphates could be due to higher rainfall, and the dilution of phosphorus loading, or it could be as a result of the phosphorus load being flushed from the system, due to increased flows. This was not likely in the present study, as the rain in autumn was associated with the first flush of the catchment, and was assumed to contain more nutrients and pollution than later rains (<sup>1</sup>Thompson, 2010, pers comm). However, it was considered that the rainfall was not significant enough to facilitate the runoff of phosphorus from the catchment, which could also have possibly explained the low concentrations of orthophosphates in autumn. Furthermore the concentrations of orthophosphates in the river may have been diluted by the rainfall. According to Vos and Roos (2005), rainfall could result in the dilution of chemical constituents. In the present study this may have accounted for the homogeneity in the concentration of orthophosphates for spring, winter and autumn.

According to Rossouw (2006) and Chen et al. (2009), the establishment of semi-aquatic vegetation could adsorb phosphorus thus reducing phosphorus loads in the water column. This may have been possible in the present study. It was also a possibility that seasonal concentrations had no relation to runoff as there was a relatively constant input of pollutants into the system throughout the year. According to de Villiers and Thiart (2007), point sources in contrast to diffuse nutrient sources, can result in seasonal concentrations having no relation to runoff as they provide a relatively constant input to the system throughout the year.

## **14.2 Macroinvertebrates**

Of the seventeen different taxa identified (Annelida, Turbellaria, Decapoda, Ephemeroptera, Odonata, Hemiptera, Diptera and Gastropoda), the ANOVA showed that there were no significant differences, in the abundance or diversity of

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<sup>1</sup> Mr. M Thompson, Catchment Manager, City of Cape Town: Catchment Management, Cape Town.

macroinvertebrates, between season or sites, in the Little Lotus River (Table 6.2 and 6.3, p 6-2 and p 6-3).

Of the taxa sampled the sensitivity score per taxon for all seasons was less than 5 on the sensitivity scale (Table 6.1, p 6-1) derived from the tolerance of the taxa to pollution, as used in the SASS5 scoring system (Dickens and Graham, 2002). As classified by Gerber and Gabriel (2002), the majority of the taxa identified, were pollution tolerant, associated with particularly low oxygen levels and organic enrichment.

In the present study, taxa which are particularly susceptible to low oxygen levels such as Ephemeroptera, Plecoptera and Trichoptera, were not present in the samples obtained from the Little Lotus River (personal observation). This could be true in the present study, but unfortunately dissolved oxygen concentrations were not sampled. According to Griffith et al. (2005), these taxa are widely used as indicators of environmental disturbances including, sedimentation, enrichment and exposure to toxicity of chemicals. According to Grobicki (2001), the absence of these taxa could indicate that oxygen concentrations of a river are often unacceptably low, and the anthropogenic influences high. According to Dallas (2004), the presence or absence of taxa could be associated with the lifecycle stage of the taxa and the preferred habitat of the taxa. In the present study, it was likely that there were a combination of reasons for the absence of more sensitive scoring taxa being sampled. However, these taxa were not collected from samples taken from any season, and therefore it was not likely that the lifecycle stage of the taxa had an influence on the assemblages of taxa sampled.

In the present study, the diversity and abundance of macroinvertebrates were considered similar between sites. No taxa were found in summer at site LL1, possibly due to significantly low water levels (personal and observation), and high nutrient loading and possibly low oxygen, as discussed earlier in the water chemistry section (Section 14.1, p 14-1). According to Dallas and Day (1993), the enrichment of a water body with organic waste, generally could result in a reduction in invertebrate species. However, according to Radar and Richardson (1992), the diversity and abundance of macroinvertebrates could also be associated with enriched water quality rather than

unenriched, when sampling similar habitats, as is thought to be the case in the present study.

In the present study, spring showed a increase in the diversity and abundance of taxa when considered with that of any other season. All sites showed a good diversity of taxa (Planarians; Oligochaeta; Hirudinea; Baetidae; Libellulidae; Corixidae; Ceratopogonidae; Chironomidae; Psycodidae; Simuliidae; Ancyliidae; Phoridae and Planorbidae), of between 3 and 11 different types of macroinvertebrates, depending on the season. The abundance of macroinvertebrates at each site was between 40 and 1420, depending on the season.

According to Grobicki (2001), King et al. (2003) and Bernardt and Palmer (2007), the extent of riparian and marginal vegetation, the reduction in peak flows and the diversification of bed materials, could be associated with the diversity and the abundance of macroinvertebrates present. According to Ogbeidu and Oribhabor (2002), Bernardt and Palmer (2007) and Feio et al. (2007), more homogenous substrates and extent of habitat, could be associated with a reduction in the number of colonizing taxa. In the present study, the placing of rocks in the stream to reduce the energy of the water during periods of high flow, and the removal of the canal walls, could have provided a more diverse substrate for colonisation by taxa, as well as an area for the establishment marginal vegetation, albeit limited.

The abundance of macroinvertebrates in the present study was possibly due to the majority of the taxa sampled consisting of Chironomidae, which suggests high productivity. According to Dickens and Graham (2002), Grobicki (2001), Gerber and Gabriel (2002) and Griffith et al. (2005), Chironomidae is a highly pollution tolerant taxa, they multiply prolifically during favourable conditions such as increased temperature, high nutrient loading, reduction in flow and good diversity of habitat, and are known to occur almost anywhere (streams, ponds, stagnant pools of water and fast flowing water, etc). The proliferation of taxa and the abundance of macroinvertebrates, in the present study, in spring were possibly due to favourable conditions and increases in temperature. The taxa that were in the minority were Ancyliidae and Oligochaeta. According to Gerber and Gabriel (2002), Ancyliidae are sensitive to poor water quality, and Oligochaete prefers muddy substrates. In the present study Ancyliidae were possibly the minority due to poor water quality, and

Oligochaete were not prolific due to insufficient muddy substrate. Most of the site is either rocky substrate or concrete bottom (personal observation). According to Dallas (2004) and Kotze (2008) habitat and substrate do play a role in the spatial variability of taxa.

In the present study winter had a lower abundance of macroinvertebrates and the lowest diversity of taxa. This was possibly as a result of scouring events due to increased flows, recent rainfall, flooding and a reduction in habitat due to higher water levels removing vegetation that could not establish in the canal due to the poor substrate. According to Dallas and Day (1993), Grobicki (2001) and Dallas and Day (2004), scouring events either due to flooding events or river maintenance could result in a reduction in the abundance and diversity of macroinvertebrates. After such an event, macroinvertebrates would need to re-establish themselves. This was likely in the present study. According to Sawyer et al. (2004) a reduction in taxa could also be due to increased levels of TSS. This was considered unlikely in the present study, as the TSS results were not of significance, and there was a noticeable seasonal trend with there being higher TSS in summer and autumn than in winter and spring (Table 6.4, p 6-3).

In the present study autumn and winter were similar with the exception that the taxa identified in winter included Baetidae and Oligochaeta, which were not present in autumn. Autumn included Corixidae, Ceratopogonidae and Lymnaeidae, which were not present in winter. According to Gerber and Gabriel (2002), Baetidae prefer fast flowing water with rocks and plants or coarse sand, this could explain why in the present study, Baetidae were present in winter and not autumn. In the present study, the rocks placed in the canal, during periods of low flow also acted as a weir, and caused the water to pool, providing back pools and calm areas (personal observation) for colonisation by macroinvertebrates. The findings of the present study were similar to those discussed in studies by King et al. (2003) and Bernhardt and Palmer (2007).

The similarities of the sites are shown in Table 4.1 (p 4-12). In the present study, all sites had low flows in summer, autumn and spring, and moderate flows in winter (personal observation). The substrate for all sites except LL4 in all seasons was rocks, with silt deposits, and marginal vegetation. The rocks and marginal vegetation

during periods of low flow (summer, spring, autumn), provided favourable habitat for the majority of the taxa sampled. Since the area of the intervention was small, the seasonal similarity and abundance of macroinvertebrates between sites could be attributed partly, to the small area of habitat that existed on this section of the Little Lotus River (personal observation). This intervention had none the less assisted with improving the ecological functioning of that section of the river. This was shown by the similarity of taxa identified and the diversity of habitat sampled throughout all the seasons. Furthermore, the results of the present study proved that the concrete substrate of the canal did not allow for the establishment of habitat, such as marginal vegetation, and the concrete surface did not provide adequate substrate (homogenous) or habitat (personal observation), for colonisation by macroinvertebrates. This was confirmed by samples collected in the canal at site LL4, which returned no macroinvertebrates.

### 14.3 Diatoms

There were no significant differences in the diversity or abundance of diatoms, between sites or between seasons, in the Little Lotus River (Table 6.17, p 6-14 and 6.18, p 6-15).

A total of forty seven species, representing nineteen genera, were identified (Table 6.15, p 6-11). The assemblages were dominated by *Staurosira elliptica* (Schumann, Williams and Round) at all sites, in all seasons. Other common species that were abundant at all sites, in all seasons were *Nitzschia palea* (Kützing, W.Smith), *Gomphonema parvulum* (Kützing), *Cyclotella meneghiniana* (Kützing), *Achnantheidium exiguum* (Grunow, Czarnecki), and *Gomphonema insigne* (Gregory). According to Bate et al. (2004), of the dominant species found in South African waters, many of the species are in fact cosmopolitan species and not endemic to South Africa as they occur, and are well recorded internationally.

According to Newell and Walsh (2005) and Beyene et al. (2009), the Specific Pollution Sensitivity Index is widely used as a preferred index to monitor water quality. The diatom species, in the present study, were as categorised by Coste in Cemagref (1982) as referred to by Van Dam et al. (1994). Similarly the ecological classification according to Van Dam et al. (1994) is discussed by other authors including Taylor et al. (2005), Weilhoefer and Pan (2007), and Resende et al. (2009).

As depicted Table 6.16 (p 6-14) the mean Specific Pollution Index (SPI) assigned to the assemblages for sites LL1 and LL2, in winter were between 5 and 9 and therefore indicative of bad water quality, and between 9 and 13 in summer, and therefore indicative of moderate quality water (Appendix A, Table A2). When considering the mean seasonal SPI both, winter (9) and summer (10), qualify as having moderate water quality with a SPI of between 9 and 13. The percentage of Pollution Tolerant Values (%PTV) for site LL1 (21.3%) in winter were more than that at site LL2 (10.6%) in winter. Site LL1 (9.3%) in summer, was less than that at site LL2 (14.5%) in summer. Therefore while the water quality was moderate there was a small improvement in the water quality between sites LL1 and LL2 in winter, and a deterioration of the water quality between sites LL1 and LL2 in summer.

In the present study at site LL1, the assemblage of diatom species was less than that at site LL2 in summer. This was likely due to the intervention creating more diversity of substrate, for the diatoms to attach onto. According to Townsend and Gell (2005), the diversity in substrate can be associated with a greater diversity of diatoms. However common species are not substrate specific. In the present study, the reduction in flow associated with the intervention, the marginal vegetation, and aquatic vegetation, could have resulted in the improved water quality which may not have been detected at a water chemistry level, but was definitely evident in the diatom assemblages present at site LL1. These findings of the present study were an indication of a pollution gradient as categorised by Beyene et al. (2009).

In the present study, winter and summer were associated with a meso-eutrophic status, with assemblages of diatoms having had continuously high requirements for oxygen (~100%saturation), having occurred in waters with a pH greater than 7 (alkaliphilous), and having been tolerant of moderately polluted waters ( $\beta$ -mesosaprobous), and with small concentrations of organically bound nitrogen (Nitrogen autotrophic – sensitive) (Table 6.19, p 6-16 and Appendix A, Table A3). Similar findings in other studies were discussed by other authors, including Taylor et al. (2005), Weilhoefer and Pan (2007), and Resende et al. (2009).

The assemblages of diatoms, in the present study, confirmed the water chemistry result as being representative of the water quality in the Little Lotus River. The

assemblages also reflected the relation of the dominant diatom taxa to the nutrients and physical conditions in the river. The diatom genera's identified as having been the most abundant in all samples, included *Nitzschia spp.*, *Navicula spp.*, *Sellaphora spp.*, *Cyclotella spp.*, *Gomphonema spp.*, *Eolimnas spp.*. Of these, some *Nitzschia spp.*, *Gomphonema spp.* and *Eolimnas spp.*, are common nitrogen heterotrophic taxa, and are tolerant of high nutrient levels (nitrogen heterotrophic and nitrogen autotrophic), and moderate to high levels of pollution (polysaprobous and mesosaprobous). Some *Navicula spp.*, *Sellaphora spp.* and *Cyclotella spp.* are also consistently found in eutrophic waters and are tolerant of polluted water (Kelly, 1998; Taylor et al., 2005; Weilhoefer and Pan, 2007; Resende et al., 2009).

In the present study, the assemblages of diatoms were consistent with the environmental variables and water chemistry results as shown by the SPI scores, and the percentages of PTV's, for each site and season (Table 6.16, p 6-14). The relationship between the SPI, and the water chemistry results was considered good, and was similar to the findings by Taylor et al. (2007), when testing the diatom-based indices in the Vaal and Wilge Rivers in South Africa.

#### **14.4 Conclusion**

The Little Lotus River is a much degraded system with poor water quality, poor habitat and biotope selection. According to Grobicki (2001), Ogbeidu and Oribhador (2002), Findlay and Taylor (2006) and Bernhardt and Palmer (2007), water quality is a limiting factor and habitat is a more limiting criterion. The provision of suitable habitat is the primary constraint behind the re-establishment of biotic diversity at a site.

In the present study there was a strong relationship between the mean macroinvertebrate counts, and the lack of suitable habitat. According to Soininen and Könönen (2004) and Feio et al. (2007), macroinvertebrates show a higher sensitivity to morphological change in channel and vegetation integrity, while diatoms are more related to water quality. Therefore if habitat was to be reduced in the Little Lotus River it could be assumed that this would impact negatively on the diversity and abundance of macroinvertebrates.

In the present study, all sites except LL4 showed a diversity of taxa ranging from between 3 and 11 taxa depending on the season. This was possibly due to the placing of rocks and the variety of habitat provided by the intervention, which resulted in the establishment of marginal vegetation, and a more diverse substrate, compared to that of the existing canal. The rocks also acted as an energy dissipater, and aerated the water during periods of high and low flow.

The water quality results showed that the Little Lotus River is nutrient rich, with concentrations of orthophosphates being representative of a hypertrophic status in summer, and a mesotrophic status in autumn, winter and spring. As was suggested by Parr and Mason (2003), eutrophication-induced changes in macroinvertebrate assemblages could occur over a long period of time compared to that sampled in a single study. According to Dallas and Day (1993) and Snaddon (2009), heterogeneity in the habitat could assist assemblages to survive changing conditions or recover from major setbacks such as pollution or periods of very high flow. It was likely in the present study, that the provision of habitat was sufficient for macroinvertebrates to have survived pollution or scouring events, as there were no significant differences in the diversity and abundance of taxa, between sites or seasons.

The assemblage of diatoms sampled at each site during summer and winter supported the long term eutrophic status of the system. The assemblages of diatoms between sites showed no significant differences and confirmed the water chemistry to be representative of the water quality in the Little Lotus River.

In the present study it could be concluded that while the seasonal diversity and abundance of macroinvertebrates, at each site was similar, the action of the intervention, in terms of creating more diverse habitat and biotope, acting as an energy dissipater, and causing the aeration of the water during the periods of high flow, could have resulted in an improved quality of habitat for macroinvertebrates, compared to that of the concrete canal. The action of this intervention (personal observation) had allowed for the establishment of marginal vegetation along the banks of the river, a rockier substrate, and therefore a greater habitat for the establishment of macroinvertebrates and diatoms. However macroinvertebrate community structure was still dominated by large numbers of pollution tolerant fauna

and low diversity, suggesting that water quality issues may over-ride habitat availability.

reeds having reduced the water flow, and the banks of the channel having had a lesser gradient than that of site LV1, more vegetation was able to establish at site LV2. According to Dallas and Day (2004), wetlands (reed beds) also dissipate the energy of the flows during periods of high rainfall, thus reducing the scouring effect of the water. This is likely in the present study, as there was little change to the diversity and abundance of macroinvertebrates in all seasons.

The water quality of the present study was nutrient rich, with concentrations of orthophosphates and total inorganic nitrogen being representative of a mesotrophic status through summer, autumn and spring, and Winter was representative of a eutrophic status due to greater runoff and influence from non-point source pollution.. Orthophosphates concentrations were representative of being of a eutrophic status through all seasons. According to Parr and Mason (2003) eutrophication-induced changes in macroinvertebrate assemblages could occur over a long period of time, and a heterogeneous habitat could assist aquatic organisms in surviving changes. This is likely in the present study, as the water quality did not affect the abundance or diversity of macroinvertebrates in winter, as there were no significant differences with any other seasons. The assemblages of diatoms sampled in the present study, at each site during summer and winter, supported the long term eutrophic status of the system.

In the present study, the intervention created more diverse habitat and biotope, while at the same time acting as a filtering mechanism, with the objective of improving water quality. This seemed to have resulted in an ecological functioning system for aquatic biota, and non-aquatic biota, compared to that of a concrete canal.

## CHAPTER 15

### DISCUSSION: LANGEVLEI CANAL

#### 15.1 Water chemistry

The South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996) consists of a water quality 'effect', a Target Water Quality Range (TWQR) and water quality criteria, the Chronic Effects Value (CEV), and the Acute Effects Value (AEV) of various parameters. The present results were compared with the South African Water Quality Guidelines for Aquatic Ecosystems water quality 'effects', and TWQR (DWAF, 1996).

No significant differences were found between sites LV1 and LV3, for the majority of the water chemistry parameters measured, except conductivity, total phosphorus and orthophosphates (Chapter 7, Section 7.2, p 7-3). There were also no significant differences in total suspended solids (TSS), chemical oxygen demand (COD), total ammonia, between seasons, measured per site.

TSS concentrations were not significantly different, but did show an increase, albeit not significant, in summer and winter, compared to spring and autumn (Table 7.4, p 7-3). Whether the results of the present study, for TSS concentrations were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined, as there were no background data of TSS concentrations. It was therefore unknown whether these results varied more than 10% to that of the background levels, as stipulated by DWAF (1996) and DWAF (2008). According to Palmer et al. (2005), in the lower reaches of rivers there are no unimpacted sites and reference conditions are difficult to determine. The results of the presents study were however less than 100mg/l, and therefore within the range for all aquatic ecosystems (DWAF, 1996).

Increased concentrations of TSS between seasons could be as a result of anthropogenic sources, such as the discharge of domestic sewage, poor farming techniques or the removal of riparian vegetation (DWAF, 1996; Grobicki, 2001; Dallas and Day, 2004; Khalife et al., 2005; Bilotta and Brazier, 2008; and Walsh and Wepener, 2009). This was possible in the present study, in winter, as an increase in runoff after rainfall in the catchment may possible have caused a 'spike' in TSS

concentrations. However, this was not likely in summer, as the increase in TSS concentrations could possibly have been due to very low water levels, and contamination from disturbed silt (suspended material), when collecting the sample.

COD was not significantly different, the mean COD for winter ( $38 \pm 4.24$  mg/l), exceeded the TWQR of 30 mg/l. According to Grobicki (2001), COD for raw sewage is 15 mg/l – 75 mg/l. It is recommended that the effluent in a stream should not exceed 30mg/l as stipulated in Government Gazette No.9225, Regulation No 991 18 May 1984. In the present study, there was little seasonal variation of COD (Table 7.7, p 7-5). In the present study, the 'spike' in COD, albeit not significant, in winter could possibly have been an indication of severely polluted water. According to Walsh and Wepener (2009), an increase in COD could potentially indicate higher levels of pollution in a stream, either due to an increase in temperature, or possibly sewage effluent entering the stream. This was likely in the present study, as the spike in COD could have been due to increased runoff, and influence from non-point source pollutants. COD is a routine measurement for effluent and is a measure of the amount of oxygen likely to be used in the degradation of organic waste, and according to Dallas and Day (1993), it is unlikely that all organic waste will be fully oxidized in aquatic ecosystems.

Concentrations of total ammonia were not significantly different, but did exceed the TWQR (0.007 mg N/L) (DWAF, 1996), in all seasons. The mean concentrations of un-ionised ammonia were not considered toxic, as the concentrations were lower than the CEV and AEV thresholds, as stipulated by DWAF (1996). Ammonia criteria for aquatic ecosystems are calculated from the total ammonia concentration, which is the sum of  $\text{NH}_3$  and  $\text{NH}_4^+$  concentrations. The norms for assessing the effects of free ammonia on aquatic ecosystems are chronic and acute toxic effects, of ammonia on aquatic organisms (DWAF, 1996). In the present study the proportion of un-ionised ammonia ( $\text{NH}_3$ ) in each sample was calculated from total (soluble) ammonia using the conversion table of DWAF (1996) and based on measured pH, and assumed summer and autumn, spring and winter temperatures, of 20°C and 15°C respectively, as indicated in Table 7.11 (page 7-8).

In the present study, while not significant, there was a seasonal trend with a 'spike' in the mean concentration of total ammonia in spring ( $0.43 \pm 0.03$  mg/l N) and

summer ( $0.50 \text{ mg/l N}$ ), and a decrease in mean concentrations in autumn ( $0.29 \pm 0.15 \text{ mg/l N}$ ) and winter ( $0.11 \pm 0.05 \text{ mg/l N}$ ) (Table 7.10, p 7-7). According to DWAF (1996), the proportion and toxicity of un-ionised ammonia in aquatic ecosystems is affected by water temperature and pH. Increases in either pH or temperature could result in a relative proportion of un-ionised ammonia in solution, and hence an increase in toxicity to aquatic organisms (DWAF, 1996). This was likely in the present study, as the increase and decrease in mean concentrations, between seasons, was possibly a seasonal trend.

In the present study, there was also a consistent reduction in the concentration of total ammonia between sites LV1 and LV3 in all seasons. This could possibly have been due to adsorption of nutrients by the artificial wetland (reed bed) between sites LV1 and LV3. According to Davies and Day (1998), Camargo and Alonso (2006) and Chen et al. (2009), wetlands could improve water quality, as the plants in the wetlands slow the flow of the water allowing the adsorption of nutrients. This was possible in the present study, as the area between sites LV1 and LV3 consisted of an artificial reed bed that was planted to act as a wetland (<sup>1</sup>Thompson, 2010, pers. comm.). However, according to Helfield and Diamond (1997) and Davies and Day (1998), the extent of the wetland often determines the capacity of the wetland plants to adsorb nutrients, and act as a sediment trap. The wetland in the present study was very small, and therefore its effectiveness to adsorb nutrients was questionable, and would require further investigation.

There were significant differences between seasons, measured per site, for pH, with winter having the lowest mean pH ( $7.05 \pm 0.07 \text{ pH}$ ) compared to all other seasons. There were significant differences between seasons and sites for conductivity with LV1 showing a lower conductivity than LV3, and the mean conductivity for spring ( $112.00 \pm 12.73$ ) being higher than all other seasons. Total oxidisable nitrogen (TON) showed significant differences between winter and autumn, winter and summer. Summer and autumn, and spring and winter were homogenous. Winter had the highest mean TON ( $3.67 \pm 0.04 \text{ mg/l N}$ ) and summer the lowest mean TON ( $1.21 \text{ mg/l N}$ ). There were significant differences between seasons, measured per site, for concentrations of total inorganic nitrogen. Winter showed the highest mean

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concentration of total inorganic nitrogen ( $3.34 \pm 0.30$  mg/l N), and summer the lowest mean concentration of total inorganic nitrogen ( $0.56$  mg/l N). There were also significant differences between sites LV1 and LV3. LV1 had higher concentrations of total inorganic nitrogen than LV3. Soluble nitrate and nitrite showed significant differences between seasons, measured per site, with winter showing the highest mean soluble nitrate and nitrite ( $3.23 \pm 0.25$  mg/l N), and summer the lowest soluble nitrate and nitrite ( $0.06$  mg/l N). There were significant differences between seasons and sites, for the concentration of total phosphorus and orthophosphates measured per season and per site. LV1 and LV3 were significant to each other, while spring was significant to all other seasons and had the highest concentration of total phosphorus ( $0.20 \pm 0.03$  mg/l P) and orthophosphates ( $0.17 \pm 0.02$  mg/l P) compared to all other seasons.

### 15.1.1 pH

pH was significantly different between seasons, with winter being significantly different to summer and autumn. pH was however not significant between sites (Table 7.5, p 7-4). The mean pH for winter ( $7.05 \pm 0.07$  pH), was considerably lower than the mean pH for all other seasons. These results were considered to be within the TWQR (DWAF, 1996), of 6 and 8 pH. According to DWAF (1996) and Dallas and Day (2004), South African freshwaters are considered neutral, as most freshwaters of South Africa are well buffered. While the TWQR (DWAF, 1996), according to Grobicki (2001) and Dallas and Day (2004), are considered alkaline, as they flow through limestone, the results of the present study are considered normal for rivers on the Cape Flats. According to DWAF (1996), Brown et al. (1998) and Dallas and Day (2004), at high pH values, the proportion of ammonia relative to other forms of nitrogen becomes higher, which could cause un-ionised ammonia to be toxic, and affect many aquatic organisms. This is not the case in the present study. According to Dallas and Day (2004), it is not always possible to determine the effects of lower levels of pH. Lower levels of pH could however be associated with a decrease in the diversity and abundance of aquatic organisms. This was not likely in the present study, as the lower pH was within the TWQR (DWAF, 1996) and was considered neutral for freshwaters of South Africa.

In the present study, the decrease in pH was possibly seasonal. According to Dallas and Day (2004), rain has a lower pH than that of freshwaters. In the present study,

the pH levels gradually increased from winter to summer, and declined again in autumn. This was consistent with the field notes (Table 4.1, p 4-12), as rain was experienced less than seven days prior to sampling in autumn, winter and spring, and more than seven days prior to sampling in summer, which had the highest pH. Due to the present study having occurred in a winter rainfall area, it could be assumed that the largest volume of rain fell in winter, and gradually decreased towards summer, increasing again in autumn (Appendix D, Table D1). This would require further investigation, but could explain the seasonal change between seasons in the present study.

### **15.1.2 Conductivity**

Conductivity results were significantly different between sites, and between seasons. However the Tukey HSD test for detecting pairwise differences could not find a significant difference amongst seasons. This is known as a dissonance that could occur if there is a shortage of data. If one ignores the factor site and a one-way ANOVA for season is done then no significant difference is found. This could be due to both methods using different estimates of residual variance, which could provide a possible explanation. In the present study seasons were therefore not considered significant.

Whether the results for conductivity were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined, as there was no background levels of conductivity. It was therefore unknown whether these results, varied more than 15% from the normal conductivity cycles under unimpacted conditions, as stipulated by DWAF (1996). Conductivity values for the present study, (Table 7.6, p 7-4) did exceed the guideline of 75 mS/m, for final effluent (DWAF, 1996; Dallas and Day, 2004), in all seasons. According to Dallas and Day (1993), Grobicki (2001) and Dallas and Day (2004), low conductivities are unlikely to impact on the flora and fauna of a river. The same authors suggest that elevated results could possibly influence the diversity and abundance of macroinvertebrate and diatom organisms, which may be sensitive to changes in conductivity.

Increases in conductivity are often as a result of anthropogenic influences, such as sewage effluent or irrigation runoff (DWAF, 1996; Deksissa et al., 2003; Dallas and

Day 2004). According to Nielsen et al. (2003) and Deksissa et al. (2003), increases in conductivity could also occur under low flows, during periods of low rainfall, when evaporation of surface waters is greatest. This was likely in the present study, as all the factors as discussed above, could have attributed to the elevated levels of conductivity throughout all the seasons. However this does not explain the significance between sites and it is therefore possible that site LV1 had less anthropogenic influences affecting it than site LV3, throughout all seasons. According to Dallas and Day (1993), many macroinvertebrate species are able to survive, and flourish at relatively high levels of salinity, but it is often the rate of change of conductivity (salinity), that has the most critical impact on aquatic organisms.

### **15.1.3 Total inorganic nitrogen (mg/l N)**

The concentrations soluble nitrates and nitrites and total oxidisable nitrogen (TON) were each significantly different, between seasons, as measured per site. Concentrations of TON showed winter being significant to summer and autumn, as well as spring autumn and summer being homogenous. Concentrations of soluble nitrates and nitrites showed winter and spring being significantly different to all other seasons, and summer and autumn being homogenous.

Concentrations of total inorganic nitrogen were significantly different between seasons and sites, with winter and spring being significantly different to all other seasons. Winter showed the highest mean concentration of total inorganic nitrogen ( $3.34 \pm 0.30$  mg/l N), compared to all other seasons (Table 7.9, p 7-6), and LV1 showed the highest concentration of total inorganic nitrogen compared to LV3.

Concentrations of total inorganic nitrogen were calculated by the addition of total ammonia and measured nitrate and nitrite data (DWAF, 1996). Whether the results for the concentration of total inorganic nitrogen, TON and soluble nitrate and nitrite, were within the TWQR (DWAF, 1996) could not be determined, as it was unknown whether the concentrations measured, exceeded the natural background concentrations of the specific system, by more than 15%. According to DWAF (1996), occasional increases in the inorganic nitrogen concentrations above the TWQR, are less of a concern than continuously high concentrations of total inorganic nitrogen. Change in trophic status is considered the norm used in assessing the 'effects' of concentrations of total inorganic nitrogen, on aquatic ecosystems (DWAF, 1996).

According to DWAF (1996), a concentration of total inorganic nitrogen of  $<5\text{mg/l N}$  in summer is representative of an oligotrophic status,  $0.5 - 2.5\text{ mg/l N}$  a mesotrophic status,  $2.5 - 10\text{ mg/l N}$  a eutrophic status, and  $>10\text{ mg/l N}$  a hypertrophic status. The present study showed that, in terms of the water quality 'effect' on aquatic ecosystems (DWAF 1996), the concentrations of total inorganic nitrogen, were representative of a mesotrophic status in summer ( $0.56 \pm \text{mg/l N}$ ), autumn ( $0.67 \pm 0.07\text{ mg/l N}$ ) and spring ( $2.01 \pm 0.11\text{ mg/l N}$ ), and a eutrophic status in winter ( $3.34 \pm 0.30\text{ mg/l N}$ ). As categorised by DWAF (2008) and PD Naidoo and Associates (2010), the water quality in summer and autumn was good, while in all other seasons it was considered fair (Appendix A, Table A5). This was possibly due to the change in trophic status between seasons.

As described by Deksissa et al. (2004) and Chen et al. (2009), the concentration of nitrogen is normally inversely related to flow, with the higher nitrogen concentrations occurring during the dry season when flows are at a minimum, and the lower nitrogen concentrations in the rainy season. However, if the non-point source pollution was greater than the point source pollution, this could be inversed, meaning there would be higher concentrations of nitrogen during periods of higher flow and rainfall, and less during periods of lower flow. This was likely in the present study, as winter was associated with the highest flows, and the river was subjected to more non-point source pollution than point source pollution, which could have explained the eutrophic status in winter.

In the present study, summer and autumn showed the lowest concentrations of total inorganic nitrogen, which could explain their homogeneity. This could have indicated that due to a lack of runoff, there could have been a reduced amount of nitrogen (non-point source pollution) entering the system. While the amount of nitrogen entering the system is considered less than other seasons, the concentration of total inorganic nitrogen in summer and autumn, were still considered elevated, as the water quality was of a mesotrophic status. According to DWAF (1996), a mesotrophic status would be indicative of moderately polluted water. According to Dallas and Day (2004), nitrate and nitrite ions are considered major nutrients that are associated with extensive plant growth and contribute to eutrophication.

In the present study the sites were significantly different, lower concentrations of total inorganic nitrogen were shown at site LV3 when compared to site LV1 for each season. According to Dallas and Day (2004) and Davies and Day (1998), wetlands can be created to filter water, which will mitigate the movement of nutrients through a system. This was likely in the present study, as site LV1 was at the start of the artificial wetland, and site LV3 at the end of the wetland (Chapter 4, p 4-1). The water flowed from the canal through the artificial wetland and returned to the canal. In the present study, lower concentrations of total inorganic nitrogen in summer and autumn, could have been due to the reed bed, which possibly reduced the amount of nitrogen in the water, due to the uptake of nitrogen by the vegetation. According to Saunders and Kalff (2001), Camargo and Alonso (2006), and Chen et al. (2009), lower flows increase nitrogen retention through the vegetative uptake of nitrogen by aquatic plants, and provide favourable conditions for denitrification, which could result in reduced concentrations of nitrogen. This was considered possible in the present study.

The present study supported the findings by Deksissa et al. (2004), as the increased flows and runoff could possibly have resulted in an increase in the concentration of total inorganic nitrogen, in the Langevlei Canal during winter and spring, while in summer and autumn the concentrations of total inorganic nitrogen were lower, due to less non-point source pollution entering the river, and possible nutrient uptake by the wetland and aquatic vegetation. This therefore showed a seasonal trend in total inorganic nitrogen concentrations.

#### **15.1.4 Total phosphorus and orthophosphates (mg/l P)**

The concentration of total phosphorus and the concentration of orthophosphates were significantly different between seasons and sites. Site LV1 was significantly different to LV3 with LV1 showing the lowest concentration of total phosphorus and orthophosphates. The mean concentration of total phosphorus in summer was significantly different to that of autumn and winter, and was homogenous with spring. The mean concentration of orthophosphates showed spring being significantly different to all other seasons.

The mean concentrations of orthophosphates were representative of a eutrophic (0.025 mg/l P – 0.25 mg/l P) status in all seasons. As categorised by DWAF (2008)

and PD Naidoo and Associates (2010), the water quality in all seasons was considered fair (Appendix A, Table A5). While sites LV1 and LV3 were significantly different there was no explanation for this.

Phosphorus is regarded as the principle nutrient which controls the degree of eutrophication, and algal blooming which is often related to increased phosphorus in freshwater systems (DWAF, 1996; He et al., 2006). According to DWAF (1996), orthophosphate concentrations of less than 0.005 mg/l P are considered to be sufficiently low, to reduce the likelihood of algal and plant growth. The effect of increases in trophic status is accompanied by the growth of algae, and other aquatic plants in rivers (DWAF, 1996). These trophic states are the norm used to assess the effects of orthophosphate concentrations on aquatic ecosystems (DWAF, 1996). According to DWAF (1996), an orthophosphate concentration of <0.005mg/l P in summer implies an oligotrophic status, 0.005 – 0.025 mg/l P a mesotrophic status, 0.025 – 0.25 mg/l P a eutrophic status, and >0.25 mg/l P a hypertrophic status.

In the present study, the concentration of orthophosphates in the Langevelei Canal, during all seasons was representative of a eutrophic status (Table 7.14, p 7-10). According to Dallas and Day (1993) and DWAF (1996), a eutrophic status is associated with a highly productive system, moderate levels of pollution, and a low level of species diversity. In the present study, spring showed a increase in the mean concentration of total phosphorus ( $0.20 \pm 0.03$  mg/l P) and the mean concentration of orthophosphates ( $0.17 \pm 0.02$  mg/l P), compared to all other seasons. According to DWAF (1996), increased concentrations in total phosphorus and orthophosphates, are normally associated with an increase in rainfall and runoff. Phosphorus concentrations could also be from non-point source pollution arising from urban runoff, and sewage discharge (DWAF 1996; Kay, 1999; Dallas and Day, 2004; Newall and Walsh, 2005; He et al, 2006; Oberholster et al., 2008). In the present study, the increase in concentrations of phosphorus could possibly be from diffuse sources, in which the phosphorus load was generated by surface and subsurface drainage, as there was rainfall seven days prior to sampling (Appendix D, Table D1 and Table 4.1, p 4-12).

According to Dallas and Day (1993), the total phosphorus and orthophosphate retention time in streams could also be affected by the flow regime, and water

residence time, which could result in low flows of a lotic system resembling a lentic system, with respect to nutrient retention. In the present study, it was likely that this was a contributing factor, as the concentrations recorded in spring were elevated, compared to other seasons, and the flow had slowed when compared to that of winter (personal observation). It was also a possibility that the wetland could have slowed the flow of the water down considerably, and as such, allowed a longer nutrient retention time in the water column.

According to Dallas and Day (1993), Grobicki (2001), Vos and Roos (2005) and He et al. (2006), a reduction in concentrations of total phosphorus and orthophosphates could be due to higher rainfall and flows potentially diluting and distributing the phosphorus load, phosphorus loads being flushed from the system, and the nutrient retention time of the phosphorus at a site being reduced. In the present study, the orthophosphate concentrations in autumn were less than other seasons. This was not likely due to rain in autumn, as runoff in autumn was associated with the first flush of the catchment, and was considered to contain more nutrients and pollution than later rains (<sup>1</sup>Thompson, 2010, pers comm). However, it was possible that the sites were not adjacent to phosphorus producing sources, or the rain was not significant enough to facilitate the runoff of phosphorus from the catchment, into the river. It is also a possibility that runoff from the first rains may not have contained high levels of phosphorus, and the concentration of phosphorus in the river, may have been diluted by precipitation. According to Vos and Roos (2005), rainfall could result in the dilution of chemical constituents.

Furthermore, according to Rossouw (2006) and Chen et al. (2009), established wetlands and semi-aquatic vegetation could absorb some of the phosphorus, thus reducing phosphorus loads in the water column. This was a possibility in the present study. According to de Villiers and Thiar (2007), point sources of pollution, in contrast to diffuse nutrient sources, can result in seasonal concentrations having no relation to runoff as they provide a relatively constant input to the system throughout the year. This was most likely in the present study.

## **15.2 Macroinvertebrates**

Of the fourteen different taxa identified (Annelida, Decapoda, Hydracarina, Ephemeroptera, Odonata, Hemiptera, Coleoptera and Diptera), the one-way ANOVA

showed that there were no significant differences in the abundance or diversity of macroinvertebrates sampled, between seasons or sites, in the Langevlei Canal (Table 7.2 and 7.3, p 7-2).

Of the taxa sampled, the majority were pollution tolerant taxa, scoring between 1-5 on the sensitivity scale derived from the taxa's tolerance to pollution, as used in the SASS5 scoring system (Dickens and Graham, 2001).the sensitivity score per taxon was between 1 and 5 in autumn and spring, and between 5 and 10 in winter (Table 7.1 p 7-1). No sample could be taken in summer due to no flow. These taxa were associated with particularly low oxygen levels and organic enrichment, as categorised by Gerber and Gabriel (2002). Winter had the highest average sensitivity score (6) per taxon, possibly due to higher water levels and greater habitat availability.

In the present study, taxa that were particularly susceptible to low oxygen levels, such as Ephemeroptera, Plecoptera and Trichoptera were not collected in any samples obtained from the Langevlei Canal. This could be true in the present study, but unfortunately dissolved oxygen concentrations were not sampled. According to Griffith et al. (2005), these taxa are widely used as indicators of environmental disturbances, including sedimentation, enrichment, and exposure to toxicity of chemicals. The absence of these taxa could indicate that oxygen concentrations are often unacceptably low, and the anthropogenic influences are high (Grobick, 2001). According to Dallas and Day (1993), the enrichment of a water body with organic waste, generally could result in a reduction in invertebrate species. According to Dallas (2004), the presence or absence of taxa could be associated with the lifecycle stage of the taxa and the preferred habitat of the taxa. In the present study, it was likely that there were a combination of reasons for the absence of more sensitive scoring taxa being sampled. However, these taxa were not sampled in any season, and therefore it was not likely that the lifecycle stage of the taxa had an influence on the assemblages of taxa sampled.

In the present study, the macroinvertebrate data sampled was similar. No taxa were sampled in summer, due to there being no flow. The diversity of taxa was good at all sites. There was between 4 and 7 taxa at each site, depending on the season. Spring at site LV3 showed a small 'spike' in the diversity and abundance of

macroinvertebrates, compared to all other seasons, while autumn at site LV3 showed a decrease in the diversity of taxa and abundance of macroinvertebrates, compared to all other seasons.

The taxa sampled at site LV3 in spring, were dominated by Corixidae, Pleidae and Hirudinea, of which, Oligochaeta, Hydrachnellae, and Baetidae did not occur in any other season. These taxa, according to Gerber and Gabriel (2002), are all pollution tolerant taxa, which prefer shallow water, quiet areas of rivers, marginal vegetation, or coarse sand. According to Grobicki (2001) and Griffith et al. (2005), macroinvertebrates are known to multiply prolifically during favourable conditions, such as increased temperature, high nutrient loading, reduction in flow and good habitat diversity. This was possibly the reason for the spike in diversity and abundance of macroinvertebrates, in the present study.

In the present study, it was also possible that the reed bed (separating sites LV1 and LV3), provided a reduction in flow, sediments and potentially nutrients, therefore facilitating the establishment of more marginal vegetation, and more favourable conditions for colonisation by macroinvertebrates, at site LV3. According to Grobicki (2001) and several other authors (Ogbeidu and Oribhabor, 2002; King et al., 2003; Dallas and Day, 2004; Bernardt and Palmer, 2007; Feio et al., 2007), the diversity of habitat and substrate is essential for establishment of a diversity of taxa. According to Dallas and Day (2004), a reduction in flow could reduce the scouring effects of a river, thus reducing the loss of macroinvertebrates. This was a possibility in the present study.

In the present study, autumn at site LV3 showed a decrease in the abundance of macroinvertebrates and diversity of taxa (Naurcoridae, Pleidae and Chironomidae). As classified by Gerber and Gabriel (2002), all the taxa identified preferred low flows, dense vegetation, and were considered moderately tolerant of pollution. In the present study, it was observed that the reed bed had expanded, and the marginal vegetation decreased, providing a more homogenous, yet denser habitat. While this was not the only explanation for a reduction in the diversity of taxa and abundance of taxon, a reduction in water, habitat diversity and substrate diversity, according to several authors (Grobicki, 2001; King et al., 2003; Dallas and Day, 2004; Bernardt

and Palmer, 2007; Feio et al., 2007), could be the result of a decrease in the diversity and abundance of macroinvertebrates.

In the present study, at site LV1 the abundance and diversity of macroinvertebrates was similar in all seasons. This was possibly due to the limited marginal vegetation, and large expanse of reeds (*Typha capensis*) that made up the majority of site LV1, in all seasons. Taxa that were identified at site LV1, for all seasons, included Hirudinea, Notonectidae, and Chironomidae. As classified by Gerber and Gabriel (2002), all these taxa are associated with pools, still water, dense vegetation, and fine sand, mud or silt. In the present study, this was characteristic of site LV1 in all seasons. According to Grobicki (2001) and several other authors (King et al., 2003; Dallas and Day, 2004; Bernardt and Palmer, 2007; Feio et al., 2007), the reduction in the diversity of taxa at a site, could be due to the reduction in habitat diversity, substrate diversity and water. While, this was likely in the present study, it was not confirmed.

Flows were diverted through the artificial wetland, with the intention of improving water quality (<sup>1</sup>Thompson, 2010, pers. Comm.). In the present study the wetland had created an ecologically functioning system, compared to that of the concrete canal (Chapter 4, section 4.4.3, p 4-5). The area through which the water was diverted and, the establishment of the artificial wetland, had proven to provide a more diverse habitat, and a variety of biotope conditions, which permitted the establishment of a more ecologically diverse assemblage of macroinvertebrates. However, the low diversity of taxa in all seasons of the present study was possibly due to the wetland being too monotypic (*Typha capensis*), and there having been not enough diversity of habitat. This could have attributed to the similarities of the present study. According to Rader and Richardson (1992), diversity and abundance of macroinvertebrates could be associated with enriched water quality, rather than unenriched when sampling in similar habitats. This was not likely in the present study, as the water chemistry (Chapter 15.1, p 15-1), was representative of a mesotrophic status, which according to DWAF (1996), and could be associated with high levels of species diversity.

In the present study, it was observed that the concrete substrate of the canal, did not allow for the establishment of habitat such as marginal vegetation, nor did the

concrete surface provide adequate substrate, for colonisation by macroinvertebrates. This was confirmed by undertaking SASS in the canal, adjacent to site LV1, which returned no macroinvertebrates.

While the area of the intervention in the present study was small, and the abundance of macroinvertebrates could be attributed partly to the small area of habitat that existed (personal observation), it was none the less an intervention that improved the ecological functioning of that section of the river. It is also a possibility that the artificial wetland could reduce flows, and nutrients downstream, thus permitting the establishment of marginal vegetation, and improved conditions for the establishment of taxa. This was similar to the finding in the case studies by King et al. (2003).

### 15.3 Diatoms

There were no significant differences in the diversity of diatom taxa or abundance of diatoms, between sites or seasons (summer and winter), in the Langevlei Canal (Table 7.17 and 7.18, p 7-13 and 7-14).

A total of fifty species, representing twenty three genera were identified (Table 7.16, p 7-12). The assemblages were dominated by *Staurosira elliptica* (Schumann, Williams and Round) at all sites, in all seasons. Other common species that were abundant at all sites, in all seasons, were *Nitzschia palea* (Kützing, W.Smith), *Gomphonema parvulum* (Kützing), *Cyclotella meneghiniana* (Kützing), *Achnanthisidium exiguum* (Grunow, Czarnecki), *Hippodonta capitata* (Ehrenberg) and *Aulacoseira granulata* (Ehrenberg, Simonsen). Of the dominant species found in South African waters it has been confirmed by Bate et al. (2004) that many of the species are in fact cosmopolitan species and not endemic to South Africa as they occur and are well recorded internationally.

The Specific Pollution Sensitivity Index, as referred to by Van Dam et al. (1994) and categorised by Cemagraf (1982), is widely used as a preferred index to monitor water quality (Beyene et al., 2009; Newall and Walsh, 2005). The diatom species were classified ecologically according to, the classification system of Van Dam et al. (1994) (Table 7.20, p 7-16). Similarly the ecological classification according to Van Dam et al. (1994), is discussed by several other authors, including Taylor et al. (2004), Weilhoefer and Pan (2007), and Resende et al. (2009).

In the present study the mean Specific Pollution Index (SPI), assigned to the assemblages for winter at sites LV1, LV2 and LV3, were between 9 and 13, and were indicative of moderate water quality (Appendix A, Table A2). In summer the mean SPI assigned to the assemblages for sites LV1 and LV2 was between 5 and 9, and indicative of poor water quality. No diatom samples were taken in summer at site LV3 due to no water. The percentage Pollution Tolerant Values (%PTV) for sites LV1 (2.9%) and LV2 (2.2%) in winter, were lower than that at site LV3 (6.4%) in winter. Site LV1 in summer (33.7%), was higher than that at site LV2 (11.5%) in summer. There were therefore more pollution tolerant species in summer than winter. This was likely due to the flow being slower in summer (personal observation), and the low levels of water causing ponding and pooling, at site LV1.

In the present study, it was observed that there was more litter, and point source pollutants at site LV1 compared to site LV2. The reduction in pollution tolerant species at LV2 in summer could be due to the improvement in water quality, and greater diversity of substrate. Winter and summer showed the same meso - eutrophic state, with assemblages of diatoms having had continuously high requirements for oxygen (~100%saturation), having occurred in waters with a pH greater than 7 (alkaliphilous), and having been tolerant of moderately polluted waters ( $\beta$ -mesosaprobous), with low concentrations of organically bound nitrogen (Nitrogen autotrophic – sensitive). The assemblages of diatoms also reflected the relationship of the dominant diatom taxa with the nutrients, and physical conditions in the river. This is similar to the findings of Taylor et al. (2005), Weilhoefer and Pan (2007), and Resende et al. (2009).

The diatom genera identified in the present study, as being most abundant in all samples, included *Sellaphora spp.*, *Cyclotella spp.*, *Gomphonema spp.*, *Eolimnas spp.*, *Fragilaria spp.*, *Aulacoseira spp.* and *Staurosira spp.*. Of these, some *Gomphonema spp.* and *Eolimnas spp.* are common nitrogen heterotrophic taxa, and are tolerant of high nutrient levels (nitrogen heterotrophic and nitrogen autotrophic), and moderate to high levels of pollution (polysaprobous and mesosaprobous). While Some, *Sellaphora spp.* and *Cyclotella spp.* are consistently found in eutrophic waters, and are tolerant of polluted water (Taylor et al., 2005; Weilhoefer and Pan, 2007; Resende et al., 2009).

In the present study, the assemblages of diatoms showed similar results to that of the water chemistry results, as was indicated by the SPI scores, and the percentages of PTV's, for each site and season. As discussed by Townsend and Gell (2005), the diversity of substrate can be associated with a greater diversity of diatoms. In the present study, the common species were not substrate specific, and it is a possibility that the lack of difference, between diversity and abundance of taxa between sites, was due to the wetland providing a homogenous substrate (personal observation), for the establishment of the different diatom taxa.

#### **15.4 Conclusion**

The Langevlei Canal is a very impacted, and a much degraded system, with poor water quality, poor habitat and biotope selection. In the present study the intervention of diverting the water through an artificial wetland had possibly improved the habitat and biota, with an attempt at improving the ecological function. Findlay and Taylor (2006) suggest that water quality is a limiting factor, but that habitat is a more limiting criterion. According to Grobicki (2001), Ogbeidu and Oribhador (2002), and Bernhardt and Palmer (2007), the provision of suitable habitat is the primary constraint behind the re-establishment of biotic diversity at a site. However the macroinvertebrate community was still dominated by pollution tolerant fauna and low diversity, suggesting that the water quality may override habitat availability.

The present study showed a definite relationship between the lack of suitable habitat, and the diversity and abundance of macroinvertebrates. According to Soininen and Könönen (2004) and Feio et al. (2007), macroinvertebrates show a higher sensitivity to morphological change in channel and vegetation integrity, while diatoms are more related to water quality. Therefore, if the habitat of the diversion canal, in the present study, ceased to exist, it could be assumed that there would have been a negative impact on the diversity and abundance, of the macroinvertebrates, in the Langevlei Canal.

In the present study, all sites showed a good diversity of macroinvertebrate taxa ranging from between 4 to 7 taxa, depending on the season. This was possibly due to the variety of habitat provided by the intervention (artificial wetland), and the establishment of reeds at site LV1, and marginal vegetation at site LV2. Due to the

reeds having reduced the water flow, and the banks of the channel having had a lesser gradient than that of site LV1, more vegetation was able to establish at site LV2. According to Dallas and Day (2004), wetlands (reed beds) also dissipate the energy of the flows during periods of high rainfall, thus reducing the scouring effect of the water. This is likely in the present study, as there was little change to the diversity and abundance of macroinvertebrates in all seasons.

The water quality of the present study was nutrient rich, with concentrations of orthophosphates and total inorganic nitrogen being representative of a mesotrophic status through summer, autumn and spring, and Winter was representative of a eutrophic status due to greater runoff and influence from non-point source pollution.. Orthophosphates concentrations were representative of being of a eutrophic status through all seasons. According to Parr and Mason (2003) eutrophication-induced changes in macroinvertebrate assemblages could occur over a long period of time, and a heterogeneous habitat could assist aquatic organisms in surviving changes. This is likely in the present study, as the water quality did not affect the abundance or diversity of macroinvertebrates in winter, as there were no significant differences with any other seasons. The assemblages of diatoms sampled in the present study, at each site during summer and winter, supported the long term eutrophic status of the system.

In the present study, the intervention created more diverse habitat and biotope, while at the same time acting as a filtering mechanism, with the objective of improving water quality. This seemed to have resulted in an ecological functioning system for aquatic biota, and non-aquatic biota, compared to that of a concrete canal.

## CHAPTER 16

### DISCUSSION: MODDERGAT RIVER

#### 16.1 Water chemistry

The South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996), consists of a water quality 'effect', a Target Water Quality Range (TWQR) and water quality criteria, the Chronic Effects Value (CEV) and the Acute Effects Value (AEV), of various parameters. The present results were compared with the South African Water Quality Guidelines for Aquatic Ecosystems water quality 'effects', and TWQR (DWAF, 1996).

No significant differences were found between sites M1, M2, M3.1, M4.1, M5, M6, M7 for any of the water chemistry parameters measured, suggesting that the structural intervention (i.e. gabions) had no effect on water quality (Chapter 8, Section 8.2, p 8-4). There were also no significant differences in total suspended solids (TSS), pH, conductivity, total phosphorus and orthophosphates, between seasons, measured per site.

The TSS concentrations were not significantly different, but did show a increase in mean concentrations in summer. Whether the results of the present study for TSS concentrations were, natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined as there were no background data of TSS concentrations sampled. It was unknown whether these results varied more than 10% to that of the background concentrations, as specified by DWAF (1996). According to Palmer et al (2005), in the lower reaches of rivers there are no unimpacted sites, and reference conditions are difficult to determine. The results of the present study were less than 100mg/l, as stipulated DWAF (1996), and therefore within the range for all aquatic ecosystems. The mean TSS concentrations in the present study were less than the mean concentrations, for the proceeding ten years, as given by the City of Cape Town (Appendix C, Table C1).

In the present study, the increase in summer concentrations could have been due to very little water in the river, and sampling only being possible at two sites. These concentrations are therefore not representative of the TSS concentrations in the river

in summer, due to the low water levels and the possibility of the sample being contaminated with sediments that may have been disturbed during sampling.

The pH levels were not significantly different between seasons (Table 8.5, p 8-5). The pH levels were within the TWQR of 6 and 8 pH (DWAF, 1996), which in terms of South African freshwaters was considered neutral, as most freshwaters of South Africa are well buffered (Dallas and Day, 2004). In the present study pH showed a seasonal trend, with higher pH measurements in summer than winter. According to DWAF (1996), an increase in pH can be associated with an increase in temperature.

The conductivity results of the present study were not significant. Whether the results for conductivity were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined, as there were no background levels of conductivity. It is therefore unknown whether these measurements varied more than 15% from the normal conductivity cycles under unimpacted conditions (DWAF, 1996). The mean conductivity in the present study was consistent with the mean conductivity levels, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1). Conductivity measurements for the present study (Table 8.6, p 8-6), did not vary considerably between seasons, nor exceed the guideline of 75 mS/m (DWAF, 1996), for final effluent (Dallas and Day, 2004), in any seasons.

As low conductivities are unlikely to impact on the flora and fauna of the river (Dallas and Day, 1993; Grobicki, 2001; Dallas and Day, 2004), it is unlikely that the results of the present study would have influenced the diversity and abundance of macroinvertebrate and diatom organisms, which may be sensitive to changes in conductivity. According to Dallas and Day (1993), many macroinvertebrate species are able to survive and flourish at relatively high salinities, but it is often the rate of change of conductivity (salinity) that has the most critical impact on aquatic organisms.

Total phosphorus concentrations, and orthophosphate concentrations were not significantly different. The mean orthophosphate concentrations were representative of a eutrophic status in all seasons (spring 0.09mg/l P, summer 0.17 mg/l P, autumn 0.03 mg/l P and winter 0.07 mg/l P). As categorised by DWAF (2008) and PD Naidoo

and Associates (2010), in summer, the water quality was considered poor, while in autumn, winter and spring it was considered fair (Appendix A, Table A5). This could have been as a result of the change in trophic status. These concentrations appeared to be consistent with the mean concentrations, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

Orthophosphate concentrations of less than 0.005 mg P/l (DWAF, 1996), are considered to be sufficiently low to reduce the likelihood of algal and plant growth. According to DWAF (1996) and Dallas and Day (2004), the effect of changes in trophic status can be accompanied by the growth of algae and other aquatic plants in rivers. These trophic states, as defined by DWAF (1996), are the norm in assessing the effects of orthophosphate concentrations on aquatic ecosystems. According to DWAF (1996), an orthophosphate concentration of <0.005mg/l P in summer implies an oligotrophic status, 0.005 – 0.025 mg/l P a mesotrophic status, 0.025 – 0.25 mg/l P a eutrophic status, and >0.25 mg/l P a hypertrophic status.

The concentrations for both total phosphorus and orthophosphates in autumn were not considered representative, as only site M1 was sampled due to no flow at any other sites.

In the present study, a increase in total phosphorus and orthophosphate concentrations in spring, albeit not significant, may have been from point source discharges (domestic and industrial effluents) and from diffuse sources, in which a phosphorus load was generated by surface and subsurface drainage. According to DWAF (1996), and several other authors (Kay, 1999; Dallas and Day, 2004; Newall and Walsh, 2005; He et al., 2006; Oberholster et al., 2008), such non-point source pollution could have been from urban runoff, drainage of agricultural land, and in particular drainage of land on which fertilisers have been applied. This was considered likely in the present study, as there were agricultural lands and vineyards in the greater catchment area.

In the present study, the elevated concentrations of total phosphorus in summer were possibly as a result of the retention time, of the total phosphorus and orthophosphate in the water column. According to Dallas and Day (1993), streams could be affected by the flow regime and water residence time, which could result in low flows of a lotic

system resembling a lentic system, with respect to nutrient retention. This was likely for the present study, but data with respect to summer were not considered representative, as samples were only taken at sites M1 and M5, due to no water being present at the other sites.

The concentrations of total phosphorus and orthophosphates in winter are normally associated with an increase in rainfall and runoff (DWAF, 1996; Dallas and Day 2004; Khalife et al., 2005; Newall and Walsh, 2005; He et al., 2006). This was likely in the present study, as while the concentrations were not as elevated as summer or spring, the concentrations were considered representative of a eutrophic status, and associated with moderately polluted water. Concentrations of phosphorus in the present study could therefore be associated with an increase in runoff, as well as non-point source and point source pollution from the surrounding area, due to rainfall.

In the present study, the reduction in orthophosphate concentrations in winter, spring and autumn, could have been due to the dilution of pollution by increased rainfall. It was also a possibility that the residual time of the phosphorus in the system was reduced due to increased flows. According to Dallas and Day (2004) and Deksissa et al. (2004), rainfall could determine the amount of flow in a river and therefore the degree of dilution of natural chemical constituents, and pollutants. According to Dallas and Day (1993), the phosphorus retention time in streams could be affected by the flow regime and water residence time, and could result in low flows of a lotic system, resembling a lentic system, with respect to nutrient retention. According to Vos and Roos (2005), evaporation could also be associated with higher concentrations of dissolved substances in surface waters. This was likely in the present study.

There were however, significant differences between seasons, measured per site, for chemical oxygen demand (COD) with autumn having the lowest COD ( $21.00 \pm 0.00$  mg/l) measurements compared to any other season. There were significant differences between seasons, measured per site for total inorganic nitrogen, with autumn showing the lowest mean total inorganic nitrogen ( $0.06$  mg/l N) concentration, compared to all other seasons. There were also significant difference between seasons, measured per site for total oxidisable nitrogen (TON), autumn showed lowest mean TON ( $0.48 \pm 0.00$  mg/l N) compared to all other seasons. Nitrite

and nitrate concentrations were also significantly different between seasons, measured per site with autumn showing the lowest mean concentration of nitrite and nitrates ( $0.02 \pm 0.00$  mg/l N) compared to any other season. Total ammonia concentrations was significantly different, with summer showing the highest mean concentration of total ammonia ( $0.45 \pm 0.19$  mg/l N), compared to all other seasons. As these parameters showed significant differences amongst the seasons sampled, these are discussed in sections 16.1.1, 16.1.2 and 16.1.3 below.

#### **16.1.1 Chemical oxygen demand (COD) (mg/l)**

COD was significantly different between autumn and all other seasons and between spring and all other seasons. Summer and winter were homogenous. COD also exceeded the Target Water Quality Range (TWQR) (DWAF, 1996) of 30 mg/l, during winter and summer (Table 8.7, p 8-7). According to Grobicki (2001), COD for raw sewerage is 15 mg/l – 75 mg/l COD. COD in the present study, was high and could possibly have been an indication of severely polluted water. It is recommended that the effluent in a stream should not exceed 30mg/l as stipulated in Government Gazette No.9225, Regulation No 991 18 May 1984. The COD of the present study did not appear to have increased when considered with the mean concentration of COD, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

According to Dallas and Day (1993), COD is a routine measurement for effluent, and is a measure of the amount of oxygen likely to be used in the degradation of organic waste. In aquatic ecosystems it is unlikely that all organic waste will be fully oxidized (Dallas and Day, 1993). According to Walsh and Wepener (2009), an increase in COD could potentially indicate higher levels of pollution in-stream. In the present study, summer and winter were more polluted than autumn and spring. This could be due to very low water levels in summer, with a greater concentration of effluent, due to greater evaporation, and more runoff of pollution in winter with a increase in rainfall. According to Deksissa et al. (2004) and Vos and Roos (2005), evaporation could be associated with higher concentrations of dissolved substances in surface waters. According to Walsh and Wepener (2009), an increase in COD could be attributed to a greater source of non-point source pollution, due to an increase in runoff. In the present study, this could account for the increase in COD in winter.

### 16.1.2 Total inorganic nitrogen (mg/l N)

The concentrations of total inorganic nitrogen, total oxidisable nitrogen and nitrite and nitrate showed significant differences, with autumn having the lowest concentrations compared to all other seasons for all parameters. Total inorganic nitrogen autumn was significantly different to winter and summer. Autumn showed the lowest mean total inorganic nitrogen concentration, compared to all other seasons (Table 8.9, p 8-9). Total oxidisable nitrogen showed autumn to have the lowest mean TON and to be significantly different to all other seasons. Nitrite and nitrate concentrations were also significantly different with autumn being significant to winter.

Total inorganic nitrogen concentrations were calculated by the addition of total ammonia and measured nitrate and nitrite data (DWAF, 1996). According to DWAF (1996), occasional increases in the inorganic nitrogen concentrations above the TWQR are less of concern, than continuously high concentrations of total inorganic nitrogen. Whether the results of the present study were within the TWQR could not be determined, as it was unknown whether the concentrations measured exceeded the natural background concentrations of the specific system by more than 15%. These concentrations were consistent with the mean concentrations, per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1). As categorised by DWAF (2008) and PD Naidoo and Associates (2010), the water quality in summer and winter was fair, in spring it was good, and in autumn it was almost natural. Change in trophic status is considered the norm in assessing the 'effects' of total inorganic nitrogen concentrations on aquatic ecosystems (DWAF, 1996).

According to DWAF (1996), a total inorganic nitrogen concentration in summer of  $<0.5\text{mg/l N}$  is representative of an oligotrophic status,  $0.5 - 2.5\text{ mg/l N}$  a mesotrophic status,  $2.5 - 10\text{ mg/l N}$  a eutrophic status, and  $>10\text{ mg/l N}$  a hypertrophic status. The present study showed, that in terms of the water quality effect (DWAF, 1996), the mean concentrations of total inorganic nitrogen were considered to be of a mesotrophic status in summer ( $1.62 \pm 1.10\text{ mg/l N}$ ), winter ( $1.07 \pm 0.03\text{ mg/l N}$ ) and spring ( $0.75 \pm 0.75\text{ mg/l N}$ ), and an oligotrophic status in autumn ( $0.06 \pm \text{mg/l N}$ ).

The oligotrophic status for autumn, in the present study, may not be representative of the entire River, as there was only water at site M1. It was also a possibility that there

were fewer inputs of point source and non-point source pollution at site M1 as the area upstream was not developed (personal observation).

According to Deksissa et al. (2004), the concentration of nitrogen is normally inversely related to flow, with higher inorganic nitrogen concentrations occurring during the dry season when flows are at a minimum, and lower inorganic nitrogen concentrations in the rainy season. In the present study, summer was associated with the lowest flows of the year, but had a higher concentration of total inorganic nitrogen compared to all other seasons. This was possibly due to less water being present, and the concentration of nitrogen having been more, due to the influence of point source pollution. According to Deksissa et al. (2004) and Vos and Roos (2005), evaporation due to an increase in temperature, could also be associated with higher concentrations of dissolved substances in surface waters. In the present study, the concentration of total inorganic nitrogen at site M5 in summer was very high, compared to that of any other site sampled. This was possibly due to point source pollution between sites M1 and M5, which may have accumulated at site M5 due to no flow. According to Dallas and Day (1993), the total inorganic nitrogen retention time in streams could be affected by the flow regime and water residence time, which could result in low flows of a lotic system resembling a lentic system, with respect to nutrient retention. However, in the present study, summer was not considered representative of the entire river, as samples were only taken at sites M1 and M5, due to no water being present at any other site.

The concentration of total inorganic nitrogen in the present study in winter indicated that winter had a lower concentration of nitrogen than summer, but still represented a mesotrophic status. The concentrations of total inorganic nitrogen were still elevated compared to spring and autumn. This could have been as a result of a greater influence from non-point source pollution and increased runoff from surrounding areas due to rainfall. According to Deksissa et al. (2004) and Chen et al. (2009), if non-point source pollution is greater than point source pollution, this could cause the opposite effect, with higher concentrations of nitrogen being associated with periods of higher flow, rainfall, and runoff, and lower concentrations of nitrogen with periods of lower flow. This was likely in the present study and could explain the significance of the lower concentrations in autumn.

In the present study, there was no improvement in the concentration of total inorganic nitrogen between M1 and M6 in winter, and very little improvement in the concentrations between M1 and M6 in spring. The little improvement in spring could be associated with a reduction in runoff from surrounding areas, and possibly a greater uptake of nitrogen by in-stream vegetation. The reduced flows in spring could also have increased the residual time of the nitrogen in the water column and therefore created more favourable conditions for the uptake of nutrients by the vegetation. According to Saunders and Kalff (2001), Camargo and Alonso (2006) and Chen et al. (2009), lower flows increase nitrogen retention through the vegetative uptake of nitrogen by aquatic plants, and provides favourable conditions for denitrification, which could result in reduced concentrations of total inorganic nitrogen. Similarly, in the present study, the lack of improvement in the concentrations of total inorganic nitrogen in winter, between sites M1 and M6, could have been due to the retention time in-stream being too little. This could have been due to faster flowing water, or a lack of vegetation growth in the channel. This could have limited the uptake of the nutrients by the vegetation.

In the present study, the in-stream vegetation was observed to be less where the Moddergat River was channelled by gabion mattresses, compared to that at sites M5, M6 and M7, where it was a reed channel and the reeds were well established and senescent. The senescence of the reeds was possibly the reason why there had been little improvement in the total inorganic nitrogen concentrations, between sites M5 and M6. Also, site M6 was adjacent to a road bridge, which could have resulted in runoff from the road, contributing to the concentration of total inorganic nitrogen. Therefore, any improvements in the concentrations of total inorganic nitrogen, associated with the reed bed, may not have been evident

### **16.1.3 Total ammonia (mg/l N)**

Total ammonia concentrations were significantly different, with summer showing the highest mean total ammonia concentration ( $0.45 \pm 0.19$  mg/l N), compared to all other seasons (Table 8.10, p 8-10). The concentrations of total ammonia exceeded the TWQR (0.007 mg/l N) (DWAF, 1996), for all sites in summer, and at site M2 in spring. These concentrations of un-ionised ammonia were not considered toxic, as the concentrations were lower than the CEV and AEV thresholds (DWAF, 1996). The mean concentrations of total ammonia were consistent with the mean concentrations,

per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

In the present study, the proportion of un-ionised ammonia ( $\text{NH}_3$ ) in each sample was calculated from total (soluble) ammonia using the conversion table of DWAF (1996), and based on measured pH and assumed summer and autumn, spring and winter temperatures, of 20°C and 15°C respectively, as indicated in Table 8.11 (p 8-11).

Ammonia criteria for aquatic ecosystems were calculated from the concentration of total ammonia, which is the sum of  $\text{NH}_3$  and  $\text{NH}_4^+$  concentrations. The norms for assessing the effects of free ammonia on aquatic ecosystems were chronic and acute toxic effects of ammonia, on aquatic organisms (DWAF, 1996).

Ammonia is known to be a common pollutant, and is one of the nutrients that can contribute to eutrophication, and excessive growth of 'nuisance' aquatic vegetation (Dallas and Day 1993; DWAF, 1996; Grobicki, 2001). According to DWAF (1996), Kay (1999), Dallas and Day (2004) and Moore and Palmer (2005), high concentrations of ammonia could come from commercial fertilisers containing highly soluble ammonia and ammonium salts, which are transported through irrigation water and runoff, or from sewage discharge. According to Grobicki (2001), these factors could affect macroinvertebrates.

In the present study, the increased concentration of total ammonia in summer could have been due to periods of low flows. The reduction in the concentration of total ammonia in winter, spring and autumn, could have been due to periods of higher flows, which could account for the homogeneity of the results. This was possible as summer had no rainfall at least fourteen days prior to sampling, while all other seasons had rainfall within seven days prior to sampling (Table 4.1, p 4-12 and Appendix D, Table D1). According to Dallas and Day (2004) and Deksissa et al. (2004), rainfall can determine the amount of flow in a river, and therefore the degree of dilution of natural chemical constituents and pollutants. According to Vos and Roos (2005), evaporation could also be associated with higher concentrations of dissolved substances in surface waters. According to Dallas and Day (1993), the retention time of nutrients in streams could be affected by the flow regime and water residence time, which could result in low flows of a lotic system resembling a lentic system

(standing water), with respect to nutrient retention. In the present study, this was likely, as the point source pollution could have caused elevated concentrations of total ammonia in summer. The restricted flow and shallow depth of the water in summer could also have increased the temperature of the pools, resulting in elevated concentrations of total ammonia.

## **16.2 Macroinvertebrates**

Of the sixteen different taxa identified (Annelida, Plecoptera, Ephemeroptera, Odonata, Hemiptera, Coleoptera, Diptera and Gastropoda), the one-way ANOVA showed that there were no significant differences in the abundance or diversity of macroinvertebrates, between seasons or sites, in the Moddergat River (Table 8.2, p 8-3 and 8.3, p 8-4). No taxa were sampled in summer or autumn, at any sites, or in spring at site M3.2, due to their having been no water in the river. No samples were taken at sites M3.2 and M4.2 in winter, due to flows being too fast and water levels too high.

The majority of the taxa sampled, were pollution tolerant taxa, which scored between 1-5 on the sensitivity scale, as used in the SASS5 scoring system, classified by Dickens and Graham (2002). According to these authors, the sensitivity scores per taxon are derived from the tolerance of the taxa to pollution. The average sensitivity score per taxon for all seasons, in the present study was less than 5 (Table 8.1, p 8-1). According to Day and Snaddon (2000), these taxa are generally hardy taxa. As classified by Gerber and Gabriel (2002), the majority of the taxa identified are pollution tolerant, scoring between 1-5 on the sensitivity scale, and are associated with particularly low oxygen levels, and organic enrichment.

In the present study, taxa that were particularly susceptible to low oxygen levels such as Ephemeroptera, were present at all sites during winter and spring. According to Grobicki (2001), these taxa are associated with acceptable oxygen concentrations. This could be true in the present study although dissolved oxygen was not measured. According to Dallas (2004), the presence or absence of taxa could also be associated with the lifecycle stage of the taxa and the preferred habitat of the taxa. In the present study, it was likely that there were a combination of reasons for the absence of more sensitive scoring taxa being sampled. However, these taxa were not sampled in any season, and therefore it was not likely that the lifecycle stage of

the taxa had an influence on the assemblages of taxa sampled in the present study, winter samples were dominated by Chironomidae, and spring samples by Simuliidae and Baetidae.

All sites in the present study showed a good diversity of taxa, ranging from between 4 and 8, depending on the season and the site. Site M2 had a similar diversity of taxa in winter and spring, but a much higher abundance of macroinvertebrates in spring, when compared to winter. Site M2 in spring was dominated by Chironomidae, Simuliidae and Baetidae, which under favourable conditions, according to Gerber and Gabriel (2002), could multiply prolifically. While these taxa were also found in winter, the reduction in flow, seasonal increase in temperature, and the establishment of more vegetation due to lower flows in spring, could possibly explain the increased abundance of macroinvertebrates at site M2 and other sites.

In the present study, in winter and spring, site M3.1 showed a 'spike' in the abundance of macroinvertebrates and diversity of taxa, compared to all other sites. The taxa present included Hirudinea, Hydrachnellae, Baetidae, Corixidae, Pleidae, Chironomidae, Lymnaeidae and Physidae. According to Gerber and Gabriel (2002), these taxa prefer habitats of aquatic vegetation, pools or slower flows. In the present study, it was a possibility that the weirs acted as energy dissipaters. This could have provided for calmer conditions upstream and downstream of the weirs, allowing for the establishment of a greater variety of marginal vegetation, and therefore greater colonisation by taxa (personal observation). The establishment of marginal vegetation downstream of these gabion rock weirs was possibly due to there having been a greater deposition of silt (personal observation), as the weirs acted as energy dissipaters. In turn, the establishment of a more ecological diverse habitat allowed for the establishment of a greater assemblage of macroinvertebrate taxa and abundance of macroinvertebrates, due to the protection offered by the vegetation.

According to Grobicki (2001) and Snaddon (2009), diversity of habitat and substrates could protect macroinvertebrates from predators and scouring. The diversity and abundance of macroinvertebrates could also be associated with enriched water quality rather than unenriched when sampling (Rader and Richardson, 1992). This is a possibility in the present study, as the diversity of taxa and abundance of macroinvertebrates at site M4.2 (gabion weir) in spring was less, compared to site

M5 downstream of the weir. In the present study, it is likely that sites that were downstream of the gabion weirs (M3.2 and M4.2) are similar and have more favourable habitat than the weirs themselves. This was however an assumption as there were no data for the weirs themselves due to increased flow or a lack of flow as discussed previously. Based on the above it was possible that the gabion lined channel, with the weirs and establishment of marginal vegetation, were functioning ecologically. The rocks and vegetation, as well as the low flow and high flow channels, provided a good diversity of biotope for colonisation by various taxa. The present study supported previous case studies discussed in King et al. (2003).

### 16.3 Diatoms

There were no summer samples as there was no water present, and therefore no statistical analysis could be undertaken.

In the present study in winter, a total of sixty seven species, representing twenty eight genera were identified (Table 8.15, p 8-14). The assemblages were dominated by *Plamnothidium frequentissimum* (Lange-Bertalot) at all sites. Site M2 was also dominated by *Cocconeis placentual var. Euglypta* (Ehrenberg). Species that were abundant at all sites were *Nitzschia frustulum* (Kützing), *Gomphonema parvulum* (Kützing), *Navicula veneta* (Kützing), *Nitzsichia sp*, *Amphora veneta* (Kützing), *Planothidium engelbrechtii* (Cholnoky), *Navicula symmetrica* (Patrick), *Navicula gregaria* (Donkin), and *Navicula cinta* (Ehrenberg). According to Bate et al. (2004) and Newall and Walsh (2005), of the dominant species found in South African waters, many of the species are in fact cosmopolitan species, are not endemic to South Africa, and are well recorded internationally.

According to Newell and Walsh (2005) and Beyene et al. (2009), the Specific Pollution Sensitivity Index is widely used as a preferred index to monitor water quality. The diatom species, in the present study, as discussed in Van Dam et al. (1994) were categorised using Coste in Cemagref (1982). Similarly the ecological classification according to Van Dam et al. (1994), is discussed by other authors including Taylor et al. (2005); Weilhoefer and Pan (2007); and Resende et al. (2009). In the present study the Specific Pollution Index (SPI) assigned to the assemblages for winter at sites M2 and M4.1 was between 5 and 9, and was indicative of poor water quality. The SPI assigned to the assemblages for sites M5 and M3.1 was

between 9 and 13, and indicative of moderate water quality. The mean SPI for winter was 9.4, and therefore indicated a moderate water quality overall (Table 8.16, p 8-16).

The percentage Pollution Tolerant Values (%PTV), for site M2 were 13% and 19.1% at site M4.1, which indicated that these sites had a high percentage of PTV's. Site M3.1 had 6 % PTV, and site M5 9.4 % PTV's. Site M3.1 therefore had the least number of PTV, which indicated an improved water quality to that of the other sites. This was consistent with the SPI discussed above.

As shown in Table 8.17 (p 8-18), diatom assemblages at site M2 were associated with a eutrophic status, low requirements for oxygen (>30%saturation), occurring in waters with a pH greater than 7 (alkaliphilous), having been tolerant of moderately polluted waters ( $\beta$ - mesosaprobous), and having had elevated concentrations of organically bound nitrogen (Nitrogen autotrophic – tolerant). Diatom assemblages at sites M3.1 and M5 were associated with an oligo-eutrophic status, low requirements for oxygen (>30%saturation), having occurred in waters with a pH greater than 7 (alkaliphilous), and having been tolerant of heavily polluted waters ( $\alpha$ - meso – polysaprobous) with elevated concentrations of organically bound nitrogen (Nitrogen autotrophic – tolerant). Diatom assemblages at site M4.1 were associated with a eutrophic status, low requirements for oxygen (>30%saturation), occurring in waters with a pH greater than 7 (alkaliphilous), and having been tolerant of heavily polluted waters ( $\alpha$ - meso – polysaprobous) and elevated concentrations of organically bound nitrogen (Nitrogen autotrophic – tolerant). Similar findings in other studies were discussed by other authors, including Taylor et al. (2004), Weilhoefer and Pan (2007), and Resende et al. (2009).

In the present study the assemblages of diatoms were representative of the water chemistry results, in that they were tolerant of organically bound nitrogen. However, the assemblages of diatoms show the water quality to be more polluted water than that represented by the water chemistry (Section 16.1, p 16-1). The assemblages of diatoms indicated the water was heavily polluted to very heavily polluted, whereas the water chemistry only indicated it was moderately polluted. This was confirmed by the presence of some *Nitzschia spp.*, *Navicula spp.*, *Sellaphora spp.*, *Cyclotella spp.*, *Gomphonema spp.*, and *Eolimnas spp.* According to Taylor et al. (2005), Weilhoefer

and Pan (2007), and Resende et al. (2009), some *Nitzschia spp.*, *Gomphonema spp.* and *Eolimnas spp.*, are common nitrogen heterotrophic taxa, which are tolerant of high nutrient levels (nitrogen heterotrophic and nitrogen autotrophic) and moderate to high levels of pollution (polysaprobous and mesosaprobous). While some, *Navicula spp.*, *Sellaphora spp.* and *Cyclotella spp.*, are consistently found in eutrophic waters, and are tolerant of polluted water.

It is concluded from the present study, that when considering the ecological values of the assemblages of diatoms, the water appears to have been more polluted, than when referring to the SPI, which indicated the water quality was poor to moderately polluted. The improvement of the water quality between sites M2 and M3.1 could possibly be due to the aeration of the water by the gabion weir at site M3.2, and the establishment of vegetation at site M3.1. The degradation of the water quality between sites M3.1, M4.1 and M5 in winter is possibly due to point source pollution, at some point, as this decline in water quality and increase in nutrients was not evident in the water chemistry for those sites.

#### **16.4 Conclusion**

In the present study, the Moddergat River is a much degraded system with poor water quality, and a high level of pollution (personal observation). According to Findlay and Taylor (2006) and Grobicki (2001), water quality is a limiting factor, and habitat is more of a limiting criterion. The extensive channel alteration with gabion mattresses, gabion weir energy dissipaters and anti erosion measures was to reduce flooding and improve aesthetics (Day and Ractliffe, 2002). In the present study this was successful, in that the energy dissipating weirs slowed the flow during periods of high flow, and created changes in habitat from that of the active channel, which allowed for the establishment of a diversity of habitat (personal observation). The weirs also acted as riffles, which aided in diversifying the biotopes along this section of the river. According to Grobicki (2001), Ogbeidu and Oribhabor (2002), Bernhardt and Palmer (2007), the provision of suitable habitat is the primary constraint behind the re-establishment of biotic diversity at a site. This has been confirmed and supported by the data collected in the present study.

The Moddergat River water chemistry as measured in the present study was moderately to heavily polluted, and there was little change in the quality of the water

along various sections of the river. It is therefore concluded that, the diversity in habitat superseded the water quality as a limiting criterion, for the ecological functioning of the river. In the present study, there appeared to be a strong relationship between the mean macroinvertebrate counts, and the lack of suitable habitat. The macroinvertebrates did not vary considerably in winter and spring, and the sites below the energy dissipating structures provided a better diversity of biotopes (personal observation), and therefore a greater diversity and abundance of macroinvertebrates.

In the present study the water quality was considered nutrient rich with orthophosphate concentrations being representative of a eutrophic status in all seasons. Total inorganic nitrogen concentrations were representative of a mesotrophic status in summer, winter and spring, and an oligotrophic status in autumn. Eutrophication-induced changes in macroinvertebrate assemblages could occur over a long period of time (Parr and Mason, 2003). Heterogeneity in the habitat could also provide macroinvertebrates protection from changing conditions such as pollution, or periods of very high flow. This was likely in the present study. The assemblage of diatoms sampled in the present study, at each site during winter, supported the long term eutrophic status of the river. Unfortunately with no summer data, these results were not representative across all seasons, but did confirm the water chemistry results.

Therefore while the seasonal diversity and abundance of macroinvertebrate taxa at each site were not significantly different, the action of the intervention in terms of creating more diverse habitat and biotope, has improved quality of habitat for macroinvertebrates when compared to that of a concrete canal, or a gabion-lined channel with no energy dissipating weirs. However, improvement in the quality of the water and the habitat, could potentially improve the diversity and abundance of the macroinvertebrates.

## CHAPTER 17

### DISCUSSION: SILVERMINE RIVER

#### 17.1 Water chemistry

The South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996) consists of a water quality 'effect', a Target Water Quality Range (TWQR) and water quality criteria, the Chronic Effects Value (CEV), and the Acute Effects Value (AEV) of various parameters. The results of the present study were compared with the South African Water Quality Guidelines for Aquatic Ecosystems, water quality 'effects', and TWQR (DWAF, 1996).

No significant differences were found between sites SM1, SM3 and SM4, for the majority of the water chemistry parameters measured except conductivity and total ammonia. No Samples were taken during summer and autumn due to no water being present.

No significant differences were found between seasons measured per site for total suspended solids, chemical oxygen demand (COD), total oxidisable nitrogen, total phosphorus and orthophosphates.

TSS concentrations were not significantly different. Whether the results for TSS were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined as there were no background levels of TSS sampled. It was therefore unknown whether these results varied more than 10% to that of the background levels, as stipulated by DWAF (1996). According to Palmer et al. (2005), in the lower reaches of rivers, there are no unimpacted sites, and reference conditions are difficult to determine. In the present study the TSS concentrations were less than 100mg/l, and as classified by DWAF (1996), were within the range for all aquatic ecosystems.

In the present study, TSS concentrations were not more than 10 mg/l for special standard rivers, or more than 25 mg/l for general standard rivers, as categorised by Dallas and Day (1993) and as required by the National Water Act, Government Gazette No. 20526, 8 October 1999 (Appendix E). The concentrations were therefore generally acceptable. According to Dallas and Day (1993), Dallas and Day (2004)

and Bilotta and Brazier (2008), natural variations in rivers often result in changes in TSS concentrations, the extent of which is governed by the hydrology and geomorphology of a particular region or stream. The similarities in the present study were possibly due to the lack of seasonal variation, hydrology and geomorphology of the river between seasons. The seasonal conditions of the river were considered similar in winter and spring.

There were no significant differences in COD between winter and spring. COD measurements exceeded 30 mg/l, in winter and spring (Table 9.7, p 9-6). According to Grobicki (2001), COD for raw sewerage is 15 mg/l – 75 mg/l. It is recommended that the effluent in a stream should not exceed 30mg/l as stipulated in Government Gazette No.9225, Regulation No 991 18 May 1984. According to Dallas and Day (1993), COD is a routine measurement for effluents, and is a measure of the amount of oxygen likely to be used in the degradation of organic waste. According to Dallas and Day (1993), it is unlikely in aquatic ecosystems, that all organic waste will be fully oxidized. An increase in COD could also indicate higher levels of pollution in-stream. In the present study, the COD measurements were considered high, and could possibly have been an indication of polluted water. This could possibly have been due to increased runoff from surrounding areas, due to more rainfall in winter than spring. The COD measurements did not appear to have increased, when considered with the mean COD measurements, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

Total phosphorus concentrations and orthophosphate concentrations were not significantly different. Orthophosphate concentrations of less than 0.005 mg/l P, according to DWAF (1996), are considered to be sufficiently low to reduce the likelihood of algal and plant growth. According to DWAF (1996) and Dallas and Day (2004), the effects of change in trophic status are accompanied by the growth of algae and other aquatic plants in rivers. These trophic states are the norm in assessing the effects of orthophosphates on aquatic ecosystems (DWAF, 1996). As stipulated in DWAF (1996), an orthophosphate concentration of <0.005mg/l P in summer is representative of an oligotrophic status, 0.005 – 0.025 mg/l P a mesotrophic status, 0.025 – 0.25 mg/l P a eutrophic status, and >0.25 mg/l P a hypertrophic status.

In the present study, the mean concentrations of orthophosphates were representative of a mesotrophic status ( $0.01 \pm 0.00$  mg/l P) in winter and spring (Table 9.14, p 9-11). As categorised by DWAF (2008) and PD Naidoo and Associates (2010), the water quality in summer and winter, was considered good (Appendix A, Table A5). This could have been due to the greater catchment area being mostly undeveloped (personal observation). These concentrations were consistent with the mean concentrations, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

According to DWAF (1996) and He et al. (2006), phosphorus is the principle nutrient in controlling the degree of eutrophication, and algal blooming is often related to increased phosphorus in freshwater systems. In the present study, the concentration of orthophosphates was representative of a productive system, with 'nuisance plants' and moderate levels of species diversity. It could not be determined if the concentration of phosphorus measured in winter and spring would increase, or decrease in summer or autumn, as no samples were taken due to no flow.

According to Dallas and Day (1993), phosphorus retention time in streams could also be affected by the flow regime and water residence time, which could result in low flows of a lotic system resembling lentic systems, with respect to nutrient retention. According to DWAF (1996), and several other authors (Kay, 1999; Dallas and Day, 2004; Newall and Walsh, 2005; He et al., 2006; Oberholster et al., 2008) elevated levels of phosphorus could occur from point source discharges such as domestic and industrial effluents. It could also come from non-point source pollution, such as runoff or particularly drainage from land on which fertilisers are used. This was likely in the present study.

In the present study, winter and spring were periods associated with high flow and high rainfall. This could have flushed the phosphorus from the system, due to a greater volume of water entering the system. According to Vos and Roos (2005), rainfall could determine the amount of flow in a river, and therefore the degree of dilution of natural chemical constituents, and pollutants. Based on the low concentrations of total phosphorus and orthophosphates, in the present study, phosphorus has little influence on the water quality in winter and spring, in the Silvermine River.

There were significant differences between seasons, measured per site, for pH, with spring having the highest mean pH ( $7.03 \pm 0.012$  pH), and winter the lowest mean pH ( $6.57 \pm 0.25$  pH). Measurements for conductivity were significantly different with summer having a higher measurement of water conductivity ( $36.67 \pm 2.08$  mSm) than winter. Concentrations of total inorganic nitrogen were significantly different, with winter having the highest mean concentration of total inorganic nitrogen ( $0.25 \pm 0.03$  mg/l N). Soluble nitrates and nitrites were significantly different, with winter showing the lowest mean concentration of soluble nitrates and nitrites ( $0.20 \pm 0.03$  mg/l N). There were also significant differences between seasons, measured per site, for total ammonia with winter having the highest concentration of total ammonia ( $0.05 \pm 0.01$  mg/l N) compared to spring. The above parameters showed significant differences, between winter and spring, as measured per site, and have been discussed in sections 17.1.1 to 17.1.4 below.

#### **17.1.1 pH**

The pH was significantly different between season, and not between sites. The pH ranged between 6.57pH in winter and 7.03 pH in spring (Table 9.5, p 9-5). These measurements were within the TWQR of 6 and 8 pH. According to DWAF (1996) and Dallas and Day (2004), in terms of South African freshwaters, this is considered neutral, as most freshwaters of South Africa are well buffered. While the TWQR values, according to Grobicki (2001) and Dallas and Day (2004), are considered alkaline as they flow through limestone, the measurements of pH for the present study, are considered normal for rivers on the Cape Flats.

In the present study, the mean pH for winter ( $6.57 \pm 0.25$  pH) was less than the mean pH for spring ( $7.03 \pm 0.12$ ). According to Dallas and Day (1993) and DWAF (1996), elevated levels of pH could be associated with an increase in biological activity in eutrophic systems, as well as with an increase in temperature at lower water levels. In the present study, the change in pH was possibly due to an increase in temperature and seasonal variations. It was not likely due to eutrophication, as the system was oligotrophic in status, and therefore a low productivity system. According to Dallas and Day (2004), rain has a lower pH than freshwaters. In the present study, the largest volume of rainfall was in winter and this gradually decreased towards summer. This could have been the reason for the reduced pH measurements,

compared to that of spring, when there was less rain, and less dilution. This would require further investigation to be proven.

### **17.1.2 Conductivity (mS/m)**

The conductivity values between winter and spring were significantly different for season and site. There were significant seasonal difference with the lowest mean conductivity being in winter ( $29.33 \pm 1.53$  mS/m), and the highest mean conductivity being in spring ( $38.67 \pm 2.08$  mS/m) (Table 9.6, p 9-5). There were also significant differences between sites and each site was considered significantly to each other. Site SM3 showed the highest measurement of conductivity in winter and spring, and SM1 the lowest. Whether the results for conductivity were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined as there was no background levels for conductivity. It was therefore unknown whether these results varied more than 15% from the normal conductivity cycles under unimpacted conditions (DWAF, 1996). The mean conductivity measured in the present study, was consistent with the mean conductivity measurements, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C).

In the present study, conductivity in both winter and spring were not considered high, as the results were well below 75 mS/m, which according to DWAF (1996) and Dallas and Day (2004), is representative of final effluent. According to Dallas and Day (1993), Grobicki (2001) and Dallas and Day (2004), low conductivities are unlikely to impact on the flora and fauna of a river. The conductivity measured in the present study was therefore unlikely to have had an influence on the diversity and abundance of macroinvertebrate organisms that may be sensitive to changes in conductivity. However, according to Dallas and Day (1993), the rate of change of conductivity (salinity), could have more of an impact on aquatic organisms, than a low or high conductivity. In the present study the increase in conductivity measurements at site SM3 could have been due to a point source pollutant which would have indicated why this site may be significantly different to SM1 and SM4.

Increases in conductivity are often as a result of runoff from irrigation, or sewage effluent entering a system (Dallas and Day, 2004). In the present study, the higher levels of conductivity in spring could have been due to increased irrigation runoff from

the golf course upstream. Due to rainfall having been less in spring, irrigation of the golf course may have increased, resulting in increased runoff. It is also a possibility that there was effluent from a point source, which had entered the system. The potential of the increased levels being as a result of point source pollution are not considered likely as the seasonal variations were not more than 15% when compared to each other. The lower levels of conductivity in winter could have been due to greater rainfall, which resulted in the flow of the river having been faster, and the dilution of anthropogenic influences being more. According to Deksissa et al. (2003) and Nielsen et al. (2003), low flows during drought periods possibly explain poorer water quality and periods of higher rainfall more improved water quality. According to Day and Snaddon (2000), it is the rate of change that is more important to aquatic biota, than the availability of absolute values.

### **17.1.3 Total inorganic nitrogen (mg/l N)**

Concentrations of total inorganic nitrogen, and soluble nitrate and nitrite concentrations, were significantly different. Lower concentrations of total inorganic nitrogen, and soluble nitrate and nitrites were recorded in spring compared to winter (Table 9.9, p 9-7 and 9.12, p 9-9). No sampling was possible in summer and autumn as there was no flow, and therefore it was not possible to determine if there was a seasonal trend. Total oxidisable nitrogen was not significantly different between seasons, measured per site.

Concentrations of total inorganic nitrogen were calculated by the addition of total ammonia and measured nitrate and nitrite data. According to DWAF (1996), occasional increases in the inorganic nitrogen concentrations above the TWQR (DWAF 1996), are less of concern than continuously high concentrations of total inorganic nitrogen. According to DWAF (1996) and Dallas and Day (2004), changes in trophic status are the norm in assessing the effects of total inorganic nitrogen on aquatic ecosystems.

Whether the results of the present study are within the TWQR (DWAF, 1996), could not be determined, as it was unknown whether these concentrations exceeded the natural background thresholds of the specific system by more than 15%. The mean concentrations of total inorganic nitrogen, in the present study, were higher than the

mean concentrations, per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

According to DWAF (1996), a concentration of total inorganic nitrogen of  $<0.5 \text{ mg/l N}$  in summer is representative of an oligotrophic status,  $0.5 - 2.5 \text{ mg/l N}$  a mesotrophic status,  $2.5 - 10 \text{ mg/l N}$  a eutrophic status, and  $>10 \text{ mg/l N}$  a hypertrophic status. The present study showed, that in terms of the water quality 'effect' on aquatic ecosystems (DWAF, 1996), the mean concentration of total inorganic nitrogen was of an oligotrophic status ( $<0.5 \text{ mg/l N}$ ), in winter and spring, with the highest mean concentration being in winter ( $0.25 \pm 0.03 \text{ mg/l N}$ ).

In the present study, lower concentrations of total inorganic nitrogen were recorded in spring, compared to winter. According to Deksissa et al., (2004), concentrations of total inorganic nitrogen in winter could be higher if non-point source pollution is greater than point source pollution. This was likely in the present study, as there was more rainfall, flow and runoff from the catchment during winter, which could have resulted in higher concentrations of total inorganic nitrogen entering the stream. This was also consistent with the recorded concentrations of ammonia. In the present study it is assumed that these concentrations of total inorganic nitrogen are from non-point sources of pollution from the golf course upstream, and adjacent urban areas. According to Kay (1999), fertilisers used on agricultural lands can lead to substantial increases in dissolved nutrients in runoff. Similarly this could be assumed for golf courses (Moss et al., 2006; Baird et al., 1996).

In the present study, the concentration of total inorganic nitrogen at site SM4 was lower than all other sites. This was possibly due to site SM4 having been at the outlet of the wetland (reed bed). It was therefore a possibility, that the concentrations at SM4 were consistently lower than other sites, due to nutrients being taken up by the vegetation. According to Davies and Day (1998) and Dallas and Day (2004), wetlands could filter water which could mitigate the movement of nutrients through a system. Saunders and Kalff (2001), Camargo and Alonso (2006) and Chen et al. (2009), also indicated that slowed flows could increase the nitrogen retention through the vegetative uptake of nitrogen by aquatic plants. This could provide favourable conditions for denitrification, resulting in reduced nitrogen concentrations. This was likely in the present study, and possibly explained the reduction in the concentration

of total inorganic nitrogen between sites SM1 and SM4 even though there was no significance between concentrations.

#### **17.1.4 Total ammonia (mg/l N)**

Total ammonia concentrations were significantly different between seasons measured per site, and between sites, measured per season. Winter had the highest mean concentration of total ammonia compared to spring. While sites were identified in the two-way ANOVA as being significantly different, the Tukey HSD test for detecting pairwise differences between sites could not find a pair that was significant. This could be a dissonance which is known to occur in situations where there is a shortage of data. If one ignores the factor season and does a one-way ANOVA for site, then no significant difference is found. The two methods use different estimates of residual variance which could be a possible explanation.

Total ammonia did however exceed the TWQR (0.007 mg/l N) (DWAF, 1996), in winter and spring (Table 9.10, p 9-8) at all sites. In the present study, the concentrations of un-ionised ammonia, in winter and spring were not toxic, as the concentrations were lower than the CEV and AEV thresholds as categorised by DWAF (1996). The mean concentrations of total ammonia were consistent with the mean concentrations, per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

Ammonia criteria for aquatic ecosystems are calculated from the total ammonia concentration which is the sum of  $\text{NH}_3$  and  $\text{NH}_4^+$  concentrations. The norms for assessing the effects of free ammonia on aquatic ecosystems are chronic and acute toxic effects of ammonia on aquatic organisms (Dallas and Day, 1993; DWAF, 1996). In the present study the proportion of un-ionised ammonia ( $\text{NH}_3$ ) in each sample, was calculated from total (soluble) ammonia using the conversion table of DWAF (1996), and based on measured pH, and assumed summer and autumn, spring and winter temperatures of 20°C and 15°C respectively, as shown in Table 9.11 (p 9-9).

The increased ammonia concentrations in the present study were associated with periods of high flows in winter and spring. Unfortunately, due to no flow in summer and autumn it could not be determined whether there would have been a reduction or an increase in the ammonia concentrations with periods of low flows. According to

Dallas and Day (1993), DWAF (1996) and Grobicki (2001), ammonia is a common pollutant, and is one of the nutrients that can contribute to eutrophication and excessive growth of 'nuisance' aquatic vegetation. Elevated concentrations of ammonia could result from commercial fertilisers containing highly soluble ammonia and ammonium salts being transported through irrigation water and runoff, or from sewage discharge into rivers (DWAF, 1996; Kay, 1999; Dallas and Day, 2004; Moore and Palmer, 2005). According to Grobicki (2001), this could affect macroinvertebrates. In the present study, this could have been possible, as increased rainfall, runoff and flows, from the golf course located upstream, were likely to have elevated the concentrations of total ammonia in winter and spring. As described by Deksissa et al. (2004), concentrations of total ammonia in winter could be higher, if the non-point source pollution is greater than the point source pollution.

## **17.2 Macroinvertebrates**

Of the thirteen different taxa identified (Hydracarina, Trichoptera, Ephemeroptera, Odonata, Hemiptera, Coleopteran and Diptera), the one-way ANOVA showed that there were no significant differences in the abundance or diversity of macroinvertebrates, between seasons or sites, in the Silvermine River (Table 9.2, p 9-3 and 9.3, p 9-3).

Of the taxa sampled, half were pollution tolerant taxa which scored between 1 and 5 on the sensitivity scale, and half were moderately pollution tolerant taxa which scored between 6 and 10 on the sensitivity scale. The sensitivity scale was derived from the tolerance of the taxa to pollution as used in the SASS5 scoring system developed by Dickens and Graham (2002). The sensitivity score per taxon in each season is depicted in Table 9.1 (p 9-1). In the present study, taxa that were particularly susceptible to low oxygen levels, such as Trichoptera, were present. According to Grobicki (2001), the presence of these taxa could possibly indicate that oxygen concentrations in a river are acceptable. Oxygen was unfortunately not measured in the present study.

In the present study, even though there was a similarity in the diversity of taxa and abundance of macroinvertebrates in winter and spring, there was a 'spike' in the number of different taxa at site SM1 in winter and spring, compared to at site SM4 in

winter and spring. This could have been due to better habitat at site SM1 than site SM4 (personal observation).

The pollution tolerances of different taxa are classified in Gerber and Gabriel (2002). In the present study, there were no differences in the number of taxa between SM1 in winter and spring. However, in winter at site SM1 there was a higher abundance of pollution tolerant macroinvertebrates (Chironimidae, Simuliidae), when compared to that of site SM1 in spring. Site SM1 in spring had more taxa that were moderately tolerant of pollution (Leptoceridae, Hydrophilidae, Ceratopogonidea). This was consistent with the water chemistry results discussed in Chapter 17.1 (p 17-1). Site SM4 showed an increase in the taxa diversity and abundance of macroinvertebrates in spring, compared to winter. All the taxa identified were highly pollution tolerant (Chironomidae, Simuliidae). Taxa that are highly pollution tolerant taxa, as classified by Gerber and Gabriel (2002), are known to occur almost anywhere in freshwater systems (streams, ponds, stagnant pools of water and fast flowing water, etc). According to Dickens and Graham (2002) and Gerber and Gabriel (2002), these taxa are known to multiply prolifically during favourable conditions such as, increased temperature, high nutrient loading and reduction in flow. This could possibly have been true for the present study, as flows were less in spring, and temperatures increased. The reduction of taxa in the present study, at site SM4 in winter, could also have been due to flooding or a scouring event. According to Dallas and Day (1993) and Grobicki (2001), scouring events either due to flooding or river maintenance, could result in a reduction in the abundance and the diversity of macroinvertebrates, as they would have to re-establish after such an event. This was also a possibility, in the present study, at site SM3.

In the present study, taxa identified in winter at SM2 and SM3 were similar. SM3 did show a higher abundance of macroinvertebrates than at SM2. This was possibly due to the flood waters having covered more terrestrial vegetation, and therefore habitat was more limited. Re-colonisation of the site by macroinvertebrates was also possibly slower, as this site only had water during winter (personal observation). In spring site SM2 could not be sampled as the flood waters had receded, and there was no water. Site SM3 in spring, had the same diversity and abundance of macroinvertebrates, as site SM1 in spring. This was possibly due to a similarity in the diversity of habitat. However, the taxa were still biotope and habitat specific. Taxa identified at site SM1

(Ceratopogonidae, Hydrophilidae, Leptoceridae and Baetidae), did not occur at site SM3 (Simuliidae, Chironomidae, Corixidae and Aeshnidae). As classified in Gerber and Gabriel (2002), in the present study, taxa identified at site SM1, were all associated with shallow, muddy pools amongst vegetation, while at site SM3, the taxa were more associated with shallow, flowing streams. The consistency of the Chironomidae in the present study, at all sites, in winter and spring, was similar. According to Grobicki (2001) Chironomidae are the most tolerant primary colonisers of streams, thus allowing them to proliferate without any competition.

In the present study, the seasonal variations between the abundance of macroinvertebrates, and diversity of taxa were not noticeable. The Silvemine River flood alleviation scheme provides a large expanse of habitat. The generally lower diversity of taxa, and abundance of macroinvertebrates, could have been associated with a larger habitat (personal observation), and a greater dispersion of taxa and individuals, which limited competition between organisms. The dense vegetation and habitat availability did provide a more ecologically diverse assemblage of macroinvertebrates, and appeared to offer more protection from predators and heavy flows.

The unenriched waters of the present study did not appear to be associated with an abundance, or diversity of macroinvertebrates. According to Rader and Richardson (1992) the diversity and abundance of macroinvertebrates are associated with enriched water quality rather than unenriched when sampling in similar habitats. This possibly explains the diversity of taxa, and the relatively low abundance of macroinvertebrates, when compared to an enriched system.

### **17.3 Diatoms**

It could not be determined if there were significant differences in the diversity of diatom taxa or the abundance of diatoms, between sites and seasons, as there was no water in the river in summer, and therefore it was only possible to sample in winter.

A total of sixty five species representing thirty genera were identified (Table 9.15, p 9-11). The assemblages were dominated by *Planthidium engelbrechtii* (Cholnoky) at sites SM1, SM3 and SM4 in winter. Site SM4 also had a high abundance of

*Achnantheidium minutissimum* (Kutzing). According to Bate et al. (2004), of the dominant species found in South African waters, many of the species are in fact cosmopolitan species and not endemic to South Africa, as they occur and are well recorded internationally. The diatom species in the present study were classified ecologically according to Van Dam et al. (1994) (Table 9.17, p 9-15). Similarly, the ecological classification according to Van Dam et al. (1994), is discussed by other authors, including Taylor et al. (2005), Weihhoefer and Pan (2007) and Resende et al. (2009). According to Newell and Walsh (2005) and Beyene et al. (2009), the Specific Pollution Sensitivity Index, as discussed by Van Dam et al. (1994) and categorised by Coste in Cemagref (1982), is widely used as a preferred index to monitor water quality.

In the present study, the mean Specific Pollution Index (SPI) assigned to the assemblages for site SM1 in winter, were between 9 and 13, for site SM3 between 5 and 9, and for site SM4 between 13 and 17. Site SM1 was indicative of moderate water quality, site SM3 poor water quality, and site SM4 good water quality. The percentage of Pollution Tolerant Values (%PTV) for sites SM1 (28%) and SM3 (29.8%) were high, and at site SM4 were low (7%). This was considered consistent with the water chemistry and SPI scores, as sites SM1 and SM2 had a higher percentage of PTV's than that at site SM4. This could indicate that there was a difference in the ecological integrity between sites.

As depicted in Table 9.16 (p 9-14), site SM1 had a eutrophic status, with assemblages of diatoms identified as having continuously high oxygen (~100%) requirements, having occurred in waters with a pH of about 7 (circumneutral), having been tolerant of small amounts of organically bound nitrogen (nitrogen – autotrophic – sensitive), and having preferred unpolluted to slightly polluted water (oligosaprobous). Site SM3 was considered to have been of a eutrophic status, with assemblages of diatoms identified as having had low requirements of oxygen (>30%), having occurred in waters with a pH of mainly 7 (alkaliphilous), having been tolerant of elevated levels of organically bound nitrogen (nitrogen – autotrophic – tolerant), and having preferred very heavily polluted water ( $\alpha$ -meso - polysaprobous). Site SM4 was considered to have been of an oligo-eutrophic status, with assemblages of diatoms identified as having required continuously high oxygen (~100%), having occurred in waters with a pH of about 7 (circumneutral), having had

elevated levels of organically bound nitrogen (nitrogen – autotrophic –tolerant), and having preferred moderately polluted water ( $\beta$ -mesosaprobous). Similar findings in other studies were discussed by Taylor et al. (2005), Weilhoefer and Pan (2007), and Resende et al. (2009).

In the present study, the diatom assemblages indicated good quality to lightly polluted water at site SM1. This was consistent with the water chemistry, and could possibly have been because the water had filtered through large reed beds higher upstream. The assemblage of diatoms that indicated heavily polluted water at site SM3 was not consistent with the water chemistry, and could possibly have indicated that a point source pollutant had been discharged, between sites SM1 and SM3. This was a possibility as site SM3 was adjacent to a residential area, and a road. It was also a possibility that because site SM3 was located at the end of the low-flow channel, there was not much absorption of nutrients by in-stream vegetation, as the low-flow channel was gabion-lined, and not as well vegetated as the other sites. The assemblage of diatoms at site SM4 showed improvements in water quality from that at site SM3. This was thought to be possible due to the water having been filtered through numerous reed beds, before reaching site SM4 (personal observation).

The diatom genera identified in the present study, as being most abundant in all samples included *Nitzschia spp.*, *Sellaphora spp.*, *Planothidium spp.* and *Achnantheidium spp.* According to Taylor et al. (2005), Weilhoefer and Pan (2007) and Resende et al. (2009), some *Nitzschia spp.*, are common nitrogen heterotrophic taxa, and are tolerant of high nutrient levels (nitrogen heterotrophic and nitrogen autotrophic) and moderate to high levels of pollution (polysaprobous and mesosaprobous). Some *Sellaphora spp.* are consistently found in eutrophic waters, and are tolerant of polluted water, while *Achnantheidium spp.* are associated with oligotrophic waters. The presence of these diatom species, in the present study, at each site, is indicative of the water quality at each site as discussed by Van dam et al. (1994) and categorised in Coste in Cemagref (1982). The assemblages of diatoms, in the present study, were in accordance with the environmental variables, but indicated that the water could have been more polluted than what has been identified by the water chemistry results.

#### 17.4 Conclusion

Silvermine River has been subject to extensive works, and was part of a flood prevention scheme for the greater Fish Hoek valley. The lower reaches are a rehabilitated system with moderate water quality and much heterogeneity of habitat (Day and Ractliffe, 2002). The flood alleviation measures and rehabilitation project, was likely to have contributed to a more diverse and abundant habitat with a higher heterogeneity of biotopes than that of a project that may have had no rehabilitation. According to Grobicki (2001), the provision of suitable habitat is the primary constraint behind the re-establishment of biotic diversity at a site, and water quality a secondary limiting criterion. From the results and discussion above this has been supported by the data collected in the present study even though the data did not provide any significant results for the diversity and abundance of macroinvertebrates. However, seasonal variations in the abundance and the diversity of macroinvertebrates were also not noticeable as no taxa could be sampled in summer or autumn due to no water.

In the present study it was expected that due to the large expanse of the lower reach of the river, and the large area rehabilitated, there would have been a higher diversity of taxa and abundance of macroinvertebrates. As described by Rader and Richardson (1992), the diversity and abundance of macroinvertebrates could be associated with enriched water quality rather than unenriched when sampling in similar habitats. In the present study, the lower number of taxa was possibly due to the water quality having been oligotrophic and mesotrophic in status and not particularly nutrient rich. It was also a possibility that the large habitat allowed for the macroinvertebrates to be more dispersed, thus limiting competition between organisms, or higher flows displacing individuals, and as such fewer organisms were present per sample area. The different individual taxa sampled were also higher scoring taxa and these could have been associated with the moderate to good water quality, associated with the sites sampled.

However, this was not the case in the present study, when the diatom assemblages were assessed. The assemblages of diatoms reflected that the sites in winter were more polluted than indicated by the water chemistry results. This was possibly due to a point source pollutant discharge between sites SM1 and SM3, which indicated that site SM3, had a poorer water quality. Unfortunately due to it not having been possible

to sample in summer it remained unclear as to whether this was a seasonal trend or not.

The present study has indicated that the action of undertaking significant engineering works and ploughing substantial amounts of money into the rehabilitation of a system could be beneficial. It could result in an ecologically functioning system, provided the diversity of habitat is maintained and the water quality is not extremely poor. The action of the wetland did appear to improve water quality. The less nutrient rich water from the greater catchment provided for a greater diversity, of more pollution sensitive taxa. Without summer and autumn samples, the present study was not seasonally representative. Winter and spring seasons are not very different and the therefore the data was expected to be similar.

## CHAPTER 18

### DISCUSSION: BIG LOTUS RIVER

#### 18.1 Water chemistry

The South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996) consists of a water quality 'effect', a Target Water Quality Range (TWQR) and water quality criteria, the Critical Effects Value (CEV), and the Acute Effects Value (AEV) of various parameters. The present results were compared with the South African Water Quality Guidelines for Aquatic Ecosystems water quality 'effects', and TWQR (DWAF, 1996).

No significant differences were found between sites BL1 and BL2, for any of the water chemistry parameters measured, suggesting that the structural intervention (i.e. rock weir) had no effect on the water quality (Chapter 10, Section 10.2, p 10-3) between sites

There were significant differences between seasons, measured per site, for all the water chemistry parameters except, pH.

The pH measurement for the present study was not significantly different between seasons, or between sites. The mean pH for summer, winter and spring were similar. The pH ranged between 7.3 in summer and 8 in autumn (Table 10.5, p 10-4). These results were considered to be within the TWQR of 6 and 8pH (DWAF 1996), which in terms of South African freshwaters, are considered neutral. According to DWAF (1996) and Dallas and Day (2004), most freshwaters of South Africa are well buffered. While the TWQR values, according to Grobicki (2001) and Dallas and Day (2004), are considered alkaline as they flow through limestone, the results of the present study are considered normal for rivers on the Cape Flats.

According to Dallas and Day (1993) and DWAF (1996), elevated levels of pH could be associated with an increase in biological activity in eutrophic systems, as well as, with an increase in temperature and lower water levels. This was likely in the present study. According to Brown et al. (1998) and Dallas and Day (2004), high pH values could result in the proportion of ammonia relative to other forms of nitrogen becoming

higher, which could be a problem as ammonia is toxic to many organisms. As discussed below, this was the case in the present study.

There were significant differences between seasons, measured per site, for total suspended solids (TSS), with summer having the lowest mean TSS ( $3.50 \pm 0.71$  mg/l) compared to all other seasons. There were significant differences between seasons, measured per site, for chemical oxygen demand (COD), with summer having the highest mean COD ( $70.50 \pm 4.95$  mg/l) and autumn the lowest mean COD ( $15.50 \pm 0.71$  mg/l). Total oxidisable nitrogen (TON) showed significant differences between seasons, measured per site, with summer showing the highest mean TON ( $13.97 \pm 0.54$  mg/l N), and spring the lowest mean TON ( $8.77 \pm 0.01$  mg/l N). There were significant differences between seasons measured per site, for total nitrogen, with spring having the lowest mean total nitrogen ( $8.33 \pm 0.01$  mg/l N), and with total ammonia with autumn having the highest mean total ammonia ( $13.15 \pm 0.16$  mg/l N). Nitrite and nitrate showed significant differences between seasons measured per site with autumn having the lowest mean nitrite and nitrates ( $0.31 \pm 0.04$  mg/l N). There were significant differences between seasons, measured per site, for total phosphorus and orthophosphates, with autumn showing the highest mean orthophosphates ( $1.62 \pm 0.03$  mg/l P) and total phosphorus ( $2.02 \pm 0.04$  mg/l P).

#### **18.1.1 Total suspended solids (TSS) (mg/l)**

The TSS concentrations between summer, and autumn and winter, and between autumn and spring, were significantly different. Whether these results for TSS were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined as there were no background data for TSS. It is therefore unknown whether these results varied more than 10% to that of the background concentrations, as stipulated by DWAF (1996). The concentrations of TSS in the present study, were however less than 100mg/l, and therefore within the range for all aquatic ecosystems (DWAF, 1996). The mean concentrations of TSS in the present study were lower than the data for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C). This possibly indicates an improvement in TSS concentrations.

Summer showed the lowest mean TSS ( $3.50 \pm 0.71$  mg/l), and autumn the highest mean TSS ( $9.50 \pm 0.710$  mg/l). While these results were significantly different, seasonal differences in the results for TSS did not differ statistically between sites. The concentrations of TSS were not greater than 10mg/l at any sites, during any seasons, and were considered generally acceptable, according to the TWQR for general standard rivers, and special standard rivers, as classified by Dallas and Day (1993) and DWAF (1996), and as required by the National Water Act, Government Gazette No. 20526, 8 October 1999 (Appendix E).

According to Dallas and Day (1993), Dallas and Day (2004) and Bilotta and Brazier (2008), natural variations in rivers often result in changes in TSS, the extent of which is governed by the hydrology and geomorphology of a particular region or stream. An increase in TSS could be as a result of anthropogenic sources, such as the discharge of domestic sewage, physical perturbations from roads, bridges, poor farming techniques, removal of riparian vegetation or stormwater runoff, which could result in increased loads of suspended solids in urban rivers (DWAF, 1996; Grobicki, 2001; Dallas and Day, 2004; Khalife et al., 2005; Bilotta and Brazier, 2008; Walsh and Wepener, 2009).

According to Malan and Day (2003) and Kyriakeas and Watzin (2006), a reduction in TSS could possibly be due to less runoff from surrounding areas and the broader catchment, due to less precipitation in the non rainy season. Similarly, increases in TSS could possibly have been due to greater runoff from the surrounding areas and broader catchment, due to increased levels of precipitation. It was therefore likely that the reduction in TSS concentrations, in the present study in summer, could have been due to lower rainfall, and a reduction in sediment transport from adjacent areas, as could the lower results in spring. The increased TSS concentrations in winter and autumn were possibly due to increased rainfall and runoff from the catchment and surrounding areas. Increased concentrations in TSS, according to Dallas and Day (1993), Grobicki (2001), Dallas and Day (2004), Sawyer et al. (2004) and Kyriakeas and Watzin (2006), could be detrimental to the diversity of macroinvertebrates, and could potentially result in a reduction of habitat.

### 18.1.2 Conductivity

The conductivity values in summer were significantly different to all other seasons, as was autumn. Autumn showed the highest conductivity compared to all other seasons and summer the lowest. Winter and spring were however also homogenous. Whether the results for conductivity were natural, good, fair, poor or unacceptable, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), could not be determined as there were no background data of conductivity. It was therefore unknown whether these results varied more than 15% from the normal conductivity cycles, under unimpacted conditions (DWAF, 1996). The conductivity values in the present study had not increased, when considered with the mean conductivity, per season, for the preceding ten years as given by the City of Cape Town (Appendix C, Table C1).

Conductivity values for the present study (Table 10.6, p 10-4) did exceed the guideline of 75 mS/m (DWAF, 1996) for final effluent (Dallas and Day, 2004), in all seasons. According to Dallas and Day (1993), Grobicki (2001), and Dallas and Day (2004), low conductivities are unlikely to impact on the flora and fauna of the river. The elevated values in the present study could therefore possibly have influenced the diversity and abundance of macroinvertebrate and diatom organisms that may be sensitive to changes in conductivity. According to Dallas and Day (1993) and Newall and Walsh (2005), many macroinvertebrate species are able to survive, and flourish at relatively high salinities, but it is often the rate of change of conductivity (salinity), that has the most critical impact on aquatic organisms. This was likely in the present study.

According to DWAF (1996), Deksissa et al. (2003) and Dallas and Day (2004), increases in conductivity are often as a result of anthropogenic influences such as sewage effluent, or irrigation runoff. In the present study this was likely as the spring and winter concentrations were elevated and both these seasons experienced higher rainfall and runoff than that of autumn and summer, this could also explain why these seasons were homogenous. However, in the present study the reduced conductivity values in summer could also be due to reduced rainfall and runoff and a reduction in anthropogenic influences entering the stream.

However, According to Nielsen et al. (2003), increases in conductivity, could also occur under low flow conditions, during periods of low rainfall, when evaporation of surface waters could result in an increase in conductivity (salinity) levels. In the present study it was likely that the elevated conductivity in autumn was significantly different to all other seasons as the lack of rainfall and the increase in temperatures at the end of summer may have increased evaporation and therefore elevated the conductivity.

### 18.1.3 Chemical oxygen demand (COD) (mg/l)

The COD values were significantly different between autumn, and all other seasons, and between summer and all other seasons. Summer had the highest mean COD ( $70.50 \pm 4.95$  mg/l) and autumn the lowest mean COD ( $15.50 \pm 0.71$  mg/l) compared to all other seasons. COD exceeded the TWQR of 30 mg/l (DWAF, 1996), in summer, winter and spring (Table 10.7, p 10-5). The mean COD values, of the present study, had not increased considerably when compared with the mean concentration of COD, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C).

In aquatic ecosystems, it is unlikely that all organic waste will be fully oxidized (Dallas and Day, 1993; DWAF, 1996). It is therefore recommended that the effluent in a stream should not exceed 30mg/l as stipulated in Government Gazette No.9225, Regulation No 991 18 May 1984.

The mean values of COD exceeded the acceptable range 30mg/l in summer ( $70.50 \pm 4.95$  mg/l), winter ( $53.50 \pm 0.71$  mg/l), and spring ( $45.00 \pm 0.00$  mg/l). Autumn was within the acceptable range, with a mean value of  $15.50 \pm 0.71$  mg/l. According to Grobicki (2001) COD for raw sewage is between 15 mg/l and 75 mg/l. The COD values of the present study were considered very high, and could possibly be an indication of severely polluted water. *Potamogeton pectinatus* was however present in very small numbers at site BL1 in all seasons, which could have been associated with an increase in dissolved oxygen. According to Dallas and Day (1993) and Grobicki (2001), *Potamogeton pectinatus* could possibly be associated with an increase in dissolved oxygen concentrations due to photosynthesis. Unfortunately dissolved oxygen concentrations were not measured, but it is possible that the weir

may have provided sufficient aeration of the water for the establishment of the *Potamogeton pectinatus*

An increase in COD could potentially indicate increased levels of pollution in-stream, either due to an increase in temperature, or possibly sewage effluent entering the stream (Walsh and Wepener, 2009). In the present study, summer had the highest COD values which could possibly be attributed to lower flows, and an increase in temperature, resulting in higher concentrations of pollution. The COD levels in winter and spring were homogenous and could have been due to a pollution incident, such as sewage effluent entering the stream, or increased runoff from the surrounding catchment due to increased precipitation. In the present study, the low COD levels in autumn could also be due to a lack of runoff from the surrounding catchment or the first rains could have resulted in a dilution of pollution and therefore lower COD levels.

#### **18.1.4 Total inorganic nitrogen (mg/l N)**

Total nitrogen concentrations were significantly different between all seasons. Total nitrogen showed seasonal trends between all seasons, in the Big Lotus River, at both sites BL1 and BL2. The concentrations of total nitrogen in the present study were less than the mean concentrations, per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1). The river was however still considered eutrophic in status, as categorised by DWAF (1996) and therefore the results could be considered similar.

The concentration of total nitrogen was calculated by the addition of total ammonia and measured nitrate and nitrite data (DWAF, 1996). Nitrate and nitrite concentrations were also significantly different with autumn as well as winter being significant to all other seasons, while summer and spring were homogenous. Total oxidisable nitrogen showed summer and autumn being homogenous and winter and spring being homogenous, and these pairs being significantly different to each other.

According to DWAF (1996), occasional increases in the inorganic nitrogen concentrations above the TWQR are less of a concern, than continuously high concentrations of total inorganic nitrogen. A change in trophic status of a water body

is the norm in assessing the 'effects' of the concentration, of total inorganic nitrogen on aquatic ecosystems (DWAF, 1996).

According to DWAF (1996) a concentration of  $<0.5\text{mg/l N}$  in summer is representative of water with an oligotrophic status,  $0.5 - 2.5\text{ mg/l N}$  a mesotrophic status,  $2.5 - 10\text{ mg/l N}$  a eutrophic status, and  $>10\text{ mg/l N}$  a hypertrophic status. The results of the present study indicated that, in terms of the water quality 'effect' (DWAF, 1996), the mean concentrations of total inorganic nitrogen were considered hypertrophic in summer ( $12.46 \pm 0.28\text{ mg/l N}$ ) and autumn ( $13.46 \pm 0.12\text{ mg/l N}$ ), and eutrophic in winter ( $9.44 \pm 0.4\text{ mg/l N}$ ), and spring ( $8.33 \pm 0.01\text{ mg/l N}$ ). The highest mean concentration was in autumn and the lowest in spring.

As shown in Table 10.9 (p 10-6), lower concentrations of total inorganic nitrogen were recorded in winter and spring during periods of high flow, and higher concentrations in summer and autumn, during periods of low flow. This could possibly have been due to the flow and water level in summer and winter. According to Deksissa et al. (2004), the concentration of nitrogen can be inversely related to flow, with the higher inorganic nitrogen concentrations occurring during the dry season, when flows are at a minimum, and the lower inorganic nitrogen concentrations in the rainy season as flow increases. This is possible in the present study, as the reduced flows could have resulted in an increase in the concentration of total inorganic nitrogen, in summer and autumn, while in winter and spring the concentrations of total inorganic nitrogen were lower, due to the dilution of the nitrogen as a result of increased flows. As noted by Kay (1999), fertilisers used on agricultural lands can also lead to substantial increases in the dissolved nutrients, in the runoff entering the stream. In the present study, it was also possible that the high concentrations in summer and autumn could have been due to irrigation runoff from the adjacent Phillippi Horticultural Area.

In the present study, the concentrations of total inorganic nitrogen were high for all seasons, and while there were seasonal trends and the concentrations during times of low flow can be explained, the concentrations during high flows were also high. This could possibly have been due to a greater non-point source contribution of total inorganic nitrogen into the river during the rainy season, and more runoff being

generated from the broader catchment which included agricultural areas, roads, and informal settlements (personal observation). In the present study it is therefore likely that the significant differences between all seasons was related to rainfall patterns and seasonal change.

#### **18.1.5 Total ammonia (mg/l N)**

Total ammonia concentrations were significantly different in all seasons. These concentrations also exceeded the TWQR (0.007 mg/l N) (DWAF, 1996), in summer and winter (Table 10.10, p 10-7). In summer the concentration of total ammonia exceeded the Critical Effect Value (CEV) and the Acute Effect Value (AEV) in autumn. Concentrations of total ammonia, in winter and spring exceeded the TWQR, but were not considered toxic as the concentrations were lower than the CEV and AEV thresholds, as defined in DWAF (1996). The concentrations of total ammonia in the present study were lower than the mean concentrations per season, for the preceding ten years, as given by the City of Cape Town (Appendix C, Table C1).

Ammonia criteria for aquatic ecosystems are calculated from the total ammonia concentration, which is the sum of  $\text{NH}_3$  and  $\text{NH}_4^+$  concentrations. The norms for assessing the effects of free ammonia on aquatic ecosystems are chronic and acute toxic effects, of ammonia on aquatic organisms (Dallas and Day, 1993; DWAF, 1996). In the present study, the proportion of un-ionised ammonia ( $\text{NH}_3$ ) in each sample was calculated from total (soluble) ammonia, using the conversion table of DWAF (1996), and based on measured pH, and assumed summer and autumn, spring and winter temperatures, of 20°C and 15°C respectively, as indicated in Table 10.11 (p 10-7).

The concentration of total ammonia in the present study, showed a similar seasonal trend to that of the concentrations of total nitrogen (Chapter 10, Section 10.2, p 10-3). Increases in the concentration of total ammonia were associated with periods of low flows (summer), and a reduced concentration of total ammonia with periods of high flows (winter).

While the trends of the present study were considered seasonal and were significant, when assessing the concentration of free ammonia at each site, during each season,

autumn and summer showed the un-ionised concentration of total ammonia, at BL1 and BL2 in autumn was acute, and all other sites, and seasons, were within the chronic effect range. This was possibly due to the elevated pH levels at sites BL1 and BL2 in summer and autumn. According to DWAF (1996), the most significant factors that affect the proportion, and toxicity of un-ionised ammonia in aquatic ecosystems, are water temperature and pH. An increase in either pH, or temperature, could result in a relative proportion of un-ionised ammonia in solution, and hence an increase in toxicity to aquatic organisms (Dallas and Day, 1993; DWAF, 1996).

ammonia is a common pollutant, and is one of the nutrients that could contribute to eutrophication of a system and excessive growth of 'nuisance aquatic vegetation' (Dallas and Day, 1993; DWAF, 1996; and Grobicki, 2001). High concentrations of total ammonia and un-ionised ammonia could come from commercial fertilisers containing highly soluble ammonia and ammonium salts, which are transported through irrigation water and runoff, or sewage discharge into rivers (DWAF, 1996; Kay, 1999; Dallas and Day, 2004; Moore and Palmer, 2005). According to Grobicki (2001), this could affect the diversity and abundance of macroinvertebrates. The increase in the concentration of total ammonia in autumn, compared to all other seasons, was possibly due to irrigation runoff from the adjacent Phillippi Horticultural Area, flowing into the Big Lotus River, as well as the increase in water temperature. Spring showed the greatest decrease in concentrations of total ammonia. This was possibly due to the increase in rainfall, water level, and flow from the surrounding catchment. This most likely provided a dilution of pollution resulting in reduced concentrations of total ammonia. According to Vos and Roos (2005), rainfall could determine the amount of flow in a river, and therefore the degree of dilution of natural chemical constituents and pollutants.

#### **18.1.6 Total phosphorus and orthophosphates (mg/l P)**

Concentrations of total phosphorus were significantly different between all seasons, while orthophosphates showed summer as well as autumn to be significantly different to all other seasons, and winter and spring to be homogenous. The concentration of orthophosphates was representative of a hypertrophic status (>0.25 mg/l P) in all seasons (Table 10.14, p 10-9). The concentrations of orthophosphates had not increased when compared with the data, for the preceding ten years, as given by the

City of Cape Town (Appendix C, Table C). The City of Cape Town data showed the seasonal water quality to be of a similar trophic status, to that of the present study (personal observation). As categorised by DWAF (2008) and PD Naidoo and Associates (2010), the water quality of the present study was considered unacceptable in all seasons (Appendix A, Table A5).

According to Dallas and Day (1993) and DWAF (1996), orthophosphates is that phosphorus component which is immediately available to aquatic biota. Orthophosphate concentrations of less than 0.005 mg/l P are considered to be sufficiently low to reduce the likelihood of algal and plant growth (DWAF, 1996). According to the same author and Dallas and Day (2004), the effect of an increase in trophic status is accompanied by the growth of algae and other aquatic plants in rivers. Trophic states are the norm in assessing the effects of orthophosphate concentrations on aquatic ecosystems (DWAF, 1996). According to this author, an orthophosphate concentration of <0.005mg /l P in summer implies an oligotrophic status, 0.005 – 0.025 mg/l P a mesotrophic status, 0.025 – 0.25 mg/l P a eutrophic status and >0.25 mg/l P a hypertrophic status.

In the present study, the orthophosphate concentrations were representative of a hypertrophic status through all seasons. The lowest mean concentration of orthophosphates was in winter ( $0.75 \pm 0.00$  mg/l P) and spring ( $0.92 \pm 0.04$  mg/l P), and the highest mean concentration of orthophosphates was in summer ( $1.26 \pm 0.06$  mg/l P) and autumn ( $1.62 \pm 0.03$  mg/l P). Autumn had the highest mean concentration compared to all other seasons.

According to DWAF (1996), Dallas and Day (2004) and He et al. (2006), phosphorus is the principle nutrient in controlling the degree of eutrophication, and algal blooming is often related to increased phosphorus in freshwater systems. Increased concentrations in orthophosphates and phosphorus are normally associated with an increase in rainfall and runoff (DWAF, 1996, He et al, 2006). In the present study, the elevated levels of phosphorus may have resulted from point source discharges, such as domestic and industrial effluents and from diffuse sources (non-point discharges), in which the phosphorus load was generated by surface and subsurface drainage. Non-point source pollution can arise from urban runoff, drainage of agricultural land,

and in particular drainage of land on which fertilisers have been applied (DWAF 1996; Kay, 1999; Dallas and Day, 2004; Newall and Walsh, 2005; He et al, 2006; Oberholster et al., 2008). This was likely in the present study, as the Big Lotus River is in close proximity to the Phillippi Horticultural Area and receives runoff from the agricultural fields.

In the present study, the high concentrations of phosphorus and orthophosphates in autumn were possibly due to the first rains, and as such runoff from surrounding areas could have contained higher concentrations of phosphorus and orthophosphates. This was likely in the present study as runoff would have been limited in autumn as the rainfall was less than in winter (Appendix D, Table D1) and therefore runoff was expected to be less.

In the present study, the eutrophication of all the sites, in all seasons, could also have been as a result of phosphorus already in the system. According to DWAF (1996) and Dallas and Day (1993), if phosphorus loads in the system are already high and the phosphorus loading of the sediments is at capacity, no more phosphorus will be able to be adsorbed and the concentrations of phosphorus in the water column will be elevated. According to Dallas and Day (1993) and Grobicki (2001) a reduction in orthophosphate concentrations could be due to higher rainfall and flows potentially diluting and distributing the phosphorus load, as well as phosphorus loads being flushed from the system and the nutrient retention time of the phosphorus at a site being reduced. This was possible in the present study, as winter and spring had the highest rainfall and associated flows, and the lowest concentrations of total phosphorus and orthophosphates, when compared to that of summer and autumn. This could account for the winter and spring being considered homogenous and summer and autumn being significantly different to all other seasons.

## **18.2 Macroinvertebrates**

Of the fourteen different taxa identified (Annelida, Decapoda, Ephemeroptera, Odonata, Hemiptera, Coleopteran, Diptera and Gastropoda), the ANOVA showed that there were no significant differences in the abundance, or diversity of

macroinvertebrates, between seasons or sites, in the Big Lotus River (Table 10.2 and 10.3, p 10-2).

Of the taxa identified, the majority were pollution tolerant taxa and had a sensitivity score ranging between 1 and 5 on the sensitivity scale as derived from the tolerance of the taxa to pollution, as used in the SASS5 scoring system (Dickens and Graham, 2002). As classified by Gerber and Gabriel (2002), these taxa were associated with particularly low oxygen levels and organic enrichment. The average sensitivity score per taxon during each season is shown in Table 10.1 (p 10-1). Taxa which were particularly susceptible to low oxygen levels, such as, Plecoptera and Trichoptera were not present in the samples obtained from the Big Lotus River, in the present study. According to Griffith et al. (2005), these taxa are widely used as indicators of environmental disturbances, including sedimentation, enrichment, and exposure to toxicity of chemicals. According to Grobicki (2001), the absence of these taxa could indicate that oxygen concentrations are often unacceptably low and the anthropogenic influences high. According to Dallas (2004), the presence or absence of taxa could be associated with the lifecycle stage of the taxa and the preferred habitat of the taxa. In the present study, it was likely that there were a combination of reasons for the absence of more sensitive scoring taxa being sampled. However, these taxa were not sampled in any season, and therefore it was not likely that the lifecycle stage of the taxa had an influence on the assemblages of taxa sampled. According to Dallas and Day (1993), the enrichment of a water body with organic waste, generally could result in a reduction in invertebrate species.

In the present study, the diversity of different taxa at site BL2 was greater than at site BL1 for all seasons as was the abundance of macroinvertebrates at site BL1 compared to site BL2. Site BL1 comprised of more pollution tolerant taxa than site BL2. No macroinvertebrates were found in summer at site BL1. This was possibly due to significantly low water levels (personal observation). Site BL2 in summer showed a greater diversity and abundance of macroinvertebrates, when compared to that of BL1. Taxa sampled consisted of Potamonautidae, Baetidae, Coenagriidae, Belostomatidae, Corixidae, Chironomidae, Physidae (seven different taxa). This was possibly due to the higher water levels (personal observation) below the rock weir, and a higher suitability of habitat (marginal vegetation) sampled. As classified by

Gerber and Gabriel (2002), all the taxa identified favoured marginal vegetation and sandy substrates, while the minority of the taxa, namely Potamonautidae, Belostomatidae and Corixidae, preferring rocks, fine sand, mud, or shallow pools. In the present study, the site as described in Chapter 4, Section 4.4.6 (p 4-10), had medium flow, was an earth channel with marginal vegetation, and had a sandy substrate (personal observation), which was the preferred habitat of the dominant taxa sampled.

In the present study, spring at site BL1 showed a greater number of macroinvertebrates (1100 individuals), when compared to site BL2. This was possibly due to the majority of the taxa sampled consisting of Chironomidae and Culcidae, which according to Gerber and Gabriel (2002), are highly pollution tolerant. According to Dickens and Graham (2002) and Gerber and Gabriel (2002), these taxa are also known to occur almost anywhere in freshwater systems (streams, ponds, stagnant pools of water, fast flowing water, etc), and to multiply prolifically during favourable conditions, such as increased temperature, high nutrient loading and reduction in flows.

In the present study, the marginal vegetation sampled at site BL2 in spring, did not provide for a greater diversity of taxa, but rather the same diversity of taxa as identified at site BL1. This was possibly due to the high nutrient loading of the river, as described in Chapter 10, Section 10.2 (p 10-3), and possible scouring events, prior to sampling, that may have resulted from increased flows due to rainfall. According to Dallas and Day (1993) and Grobicki (2001), scouring events either due to flooding or river maintenance, could result in a reduction of the number of macroinvertebrates and the diversity of taxa, as macroinvertebrates would require to re establish themselves after such an event. This was likely in the present study.

According to Sawyer et al. (2004), a reduction in the abundance and diversity of macroinvertebrates could be attributed to increased levels of TSS. In the present study this was unlikely, as the TSS concentrations were within the TWQR, as discussed previously. Sawyer et al. (2004) also suggested that as total ammonia concentrations increase, the diversity of macroinvertebrates decrease. While the un-ionised ammonia concentrations in the present study in autumn and summer reached

acute toxicity levels, this did not appear to influence the abundance or diversity of macroinvertebrates at site BL1 or BL2, and was therefore not likely.

In the present study, sites BL1 and BL2 in autumn, had a similar diversity and abundance of macroinvertebrates, but the diversity was high. Common taxa to both sites were Chironomidae and Physidae. Site BL2 included Ceratopogonidae, Physidae and Planorbidae, and site BL1 Hydrophilidae and Coenagriidae. According to Gerber and Gabriel (2002), these taxa all prefer marginal vegetation biotopes on the edge of streams and in shallow pools or back waters, and they are all pollution tolerant taxa. In the present study, it is more likely the weir (engineering intervention), may have provided backwaters and pools. The seasonally lower flows could also have reduced the active channel, thus providing less flow on the edges of the river, and improved conditions (personal observation), for the establishment of macroinvertebrates. This is supported by the case studies discussed in King et al. (2003), and by Bernhardt and Palmer (2007).

In the present study winter showed fewer different taxa at site BL1 when compared to site BL2, but a similar abundance of individuals. At site BL1 only Chironomidae and Oligochaetae were present, while at site BL2 Coenagriidae, Pleidae, Dytiscidae, Ceratopogonidae were also identified. As classified by Gerber and Gabriel (2002), the taxa in the present study identified at site BL1 are much more pollution tolerant than the taxa identified at site BL2. In the present study it is considered possible, that the lack of variation in biotope provided by the weir at site BL1 could have been a contributing factor to the lack of diversity of taxa. As classified by Gerber and Gabriel (2002) the taxa that were identified at site BL2, prefer marginal vegetation, edges of streams and calm pools or back waters. In the present study, the reduced flow of the river downstream of the weir is considered favourable to the establishment of macroinvertebrates, whereas the riffle created by the weir, and the fast flowing waters as were observed in winter (rainfall season), could be considered less favourable habitat.

According to Grobicki (2001) and Griffith et al. (2005), Chironomidae are the most tolerant colonisers of streams, and are primary colonisers of polluted streams, thus allowing them to proliferate without any competition. This could explain the

similarities amongst the diversity and abundance of macroinvertebrates in the present study, between sites and seasons.

According to Grobicki (2001) and <sup>1</sup>Thompson pers. comm. (2010), the rock weir was installed as an energy dissipater during high flows. It was not expected to have any importance with regards to the ecological functioning of the river, as it is an engineering solution to reduce the flow of the water. In the present study, the weir during low flow periods possibly created more diverse habitat and variety of biotope conditions, downstream of site BL1. This permitted the establishment of a more ecological diverse assemblage of macroinvertebrates at site BL2, and offered more protection from predators. According to Grobicki (2001) and Snaddon (2009), better habitat conditions could provide greater protection from predators and scouring events. Furthermore, Rader and Richardson (1992) stated that the diversity and abundance of macroinvertebrates could be associated with enriched water quality rather than unenriched, when sampling in similar habitats. Finally, the abundance of macroinvertebrates often mirrors the nutrient loading of a stream or reach. Therefore, the higher the degree of eutrophication the higher the abundance of macroinvertebrates (Rader and Richardson, 1994; Haase and Nolte, 2008). This was consistent with the water chemistry results of the present study, and could explain the similarities between abundance and diversity of macroinvertebrates.

### 18.3 Diatoms

There were no significant differences in the diversity or abundance of diatoms, between sites or seasons, summer and winter, in the Big Lotus River (Table 10.17 and 10.18, p 10-12). A total of forty one species representing twenty two genera were identified. The assemblages were dominated by *Navicula veneta* (Kutzing) at site BL1 and *Cyclotella meneghiniana* (Kutzing) at site BL2, in winter. Summer samples were dominated by *Gomphonema parvulum* (Kutzing). It should be noted that of the dominant species found in South African waters, it has been confirmed by Bate et al. (2004), that many of the species are in fact cosmopolitan species, and are not

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<sup>1</sup> Mr. M. Thompson, Catchment Manager, City of Cape Town: Catchment Management, Cape Town.

endemic to South Africa as they occur and are well recorded internationally (<sup>2</sup>Harding, 2010, pers. comm.).

The Specific Pollution Sensitivity Index, as referred to by Van Dam et al. (1994) and categorised by Coste in Cemagraf (1982), is widely used as a preferred index to monitor water quality (Newell and Walsh, 2005; Beyene et al., 2009). In the present study the diatom species were classified ecologically using the Cemagraf (1982) index as referred to in Van Dam et al. (1994), (Table 10.19, p 10-14). Similarly the ecological classification according to Van Dam et al. (1994) is discussed by other authors, including Taylor et al. (2005), Weilhoefer and Pan (2007), and Resende et al. (2009).

As shown in Table 10.16 (p 10-11), the Specific Pollution Index (SPI) assigned to the assemblages for sites BL1 and BL2, of the present study, in winter and summer were >5, and therefore indicative of bad water quality. The percentage Pollution Tolerant Values (%PTV) for sites BL1 and BL2 in winter and summer were also high. Assemblages in winter were 23.1% tolerant at site BL1, and 23.7% tolerant at site BL2, compared to 59.4% tolerant in summer. As the SPI in the present study were similar at sites BL1 and BL2, it could be concluded that there is no difference in the ecological integrity of the sites between seasons. As shown in Table 10.19 (p 10-14), winter was representative of a eutrophic system with assemblages of diatoms identified as being tolerant of very low oxygen levels, occurring in waters with a pH greater than 7 (alkiphilous), and being tolerant of heavily polluted waters ( $\alpha$ -mesopolysaprobous). Summer also represented a eutrophic system, with assemblages of diatoms identified as being tolerant of very low oxygen levels, mainly occurring in water with a pH of 7 (alkiphilous), and having been tolerant of heavily polluted water.

The water chemistry results for winter and summer, as discussed previously, are consistent with the poor water quality conditions identified by the assemblage of diatoms. The assemblages of the dominant diatom taxa were also representative of assemblages of diatoms that could be associated with the nutrients and physical

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<sup>2</sup> DR. W.R.Harding, Freshwater Ecologist, DH Environmental Consulting, Cape Town.

conditions in the river. Similar findings in other studies were discussed by Taylor et al. (2005), Weilhoefer and Pan (2007), and Resende et al. (2009).

In the present study, the diatom genera identified as being most abundant in all samples included *Nitzschia spp.*, *Navicula spp.*, *Sellaphora spp.*, *Cyclotella spp.*, *Gomphonema spp.*, *Eolimnas spp.* Of these, some *Nitzschia spp.*, *Gomphonema spp.* and *Eolimnas spp.* are common nitrogen heterotrophic taxa, and are tolerant of high nutrient levels (nitrogen heterotrophic and nitrogen autotrophic) and moderate to high levels of pollution (polysaprobous and mesosaprobous). According to Taylor et al. (2005), Weilhoefer and Pan (2007) and Resende et al. (2009), some *Navicula*, *Sellaphora* and *Cyclotella* species are consistently found in eutrophic waters, and are tolerant of polluted water.

In the present study, it was concluded that the diatom assemblages and the respective SPI scores, and percentages of PTV's for each site and season, are consistent with the water chemistry results as discussed in the previous section. This was similar to the findings discussed by Taylor et al. (2007) when testing the diatom-based indices in the Vaal and Wilge Rivers in South Africa. In the present study the SPI score did improve in winter, from 2.4 at site BL1 to 4.4 at site BL2. This showed that there was an improvement in water quality, but it was not significant enough to be identified at a water chemistry level. It is possible that this could be due to the weir, which is possibly indirectly contributing to the improvement of water quality downstream. This was however not substantial enough to show a change in trophic status.

The reason for the improvement in the SPI score will need to be investigated more specifically. This was not relevant to the present study as the saprobity of sites BL1 and BL2 in winter, are both a-meso-polysaprobous, indicating that the water quality at both these sites was very heavily polluted. This is consistent with the water chemistry results as discussed previously.

#### **18.4 Conclusion**

The Big Lotus River is a much degraded system with poor water quality and poor habitat and biotope selection. As stated by Grobicki (2001), the provision of suitable habitat is the primary constraint behind the re-establishment of biotic diversity at a

site. The present study showed a strong relationship between the mean macroinvertebrate counts and the lack of suitable habitat, making habitat availability the factor most limiting to the presence of aquatic invertebrates, with water quality being subsidiary. However, macroinvertebrate community structure was still dominated by large numbers of pollution tolerant fauna, and low diversity, suggesting that water quality may over-ride habitat availability. If habitat were to be reduced in the Big Lotus River, it could be assumed that this could also negatively impact on the diversity and abundance of the macroinvertebrates.

The diversity of macroinvertebrates was greater at sites BL2 than BL1 in all seasons, and the taxa at site BL1 more pollution tolerant, than at site BL2. As discussed above this is assumed to have been due to the marginal vegetation and slower flows at site BL2, compared to site BL1.

The water quality data showed that the Big Lotus River is nutrient rich, with orthophosphate concentrations being representative of a hypertrophic system throughout all seasons at each site. Concentrations of total inorganic nitrogen fluctuated between hypertrophic in summer and autumn when flows were least, and eutrophic in winter and spring when flows were highest. The hypertrophic status in summer and autumn was possibly attributed to the high pH and temperature. The assemblage of diatoms sampled at each site during summer and winter, supported the long-term eutrophication of the system. The assemblages of diatoms between sites, showed no significant differences, and confirmed the water chemistry results as being representative of the water quality. The water quality in the present study was similar to that determined by Grobicki (2001) in the Big Lotus River.

The action of the weir as an energy dissipater to high flows, resulted in an improved quality of habitat for macroinvertebrates at site BL2, when compared to that at site BL1. Due to urban runoff, and runoff from the Phillippi Horticultural Areas, the Big Lotus River continued to be nutrient rich. Therefore it was likely that the weir had no influence on water quality between sites BL1 and BL2. However, while the water chemistry results are not detailed enough to identify subtle changes in water quality, the saprobity index of the diatoms sampled, reflected that both sites in winter were extremely polluted. There was however an improvement in the SPI at site BL2,

downstream of the weir. It may therefore be possible that the weir was contributing, on a small scale, to the improvement of water quality. This would need to be investigated further. The weir did act as an informal litter trap (personal observation), limiting the amount of pollution downstream of sites BL1 at BL2.

## CHAPTER 19

### DISCUSSION: ALL RIVERS

#### **19.1 Water chemistry:**

All the rivers (Keysers River, Little Lotus River, Langevlei Canal, Moddergat River, Silvermine River, and Big Lotus River) in the present study, showed a level of eutrophication, and were polluted to some degree. The highest concentration of total nitrogen (mg/l N) was measured in the Big Lotus River in autumn, and the lowest concentration of total nitrogen was measured in the Keysers River in autumn (Appendix B, Figure B3).

The concentration of total nitrogen in the Big Lotus River, as categorised by DWAF (2008) and PD Naidoo and Associates (2010), ranged between unacceptable and poor water quality all year. The Little Lotus River, Keysers River, Moddergat River and the Langevlei Canal, showed fair to good water quality, while the Silvermine River showed good to 'natural' water quality. The Big Lotus River showed the most impacted water quality, compared to all other rivers, and the Silvermine River the least impacted. This was most likely due to the different sources of non-point source and point source pollutants, and the different activities that occurred in the various catchments (Chapter 4, pg 4-1).

As categorised by DWAF (2008) and PD Naidoo and Associates (2010), orthophosphate concentrations in the Little Lotus River showed 'fair' to 'unacceptable' water quality. The Keysers River, Moddergat River and the Langevlei Canal showed 'fair' to 'poor' water quality. The Silvermine River showed 'good' water quality, and the Big Lotus River 'unacceptable' water quality, as classified by DWAF (2008) and PD Naidoo and Associates (2010) (Appendix A, Table A5 and Appendix B, Figure B4). A lower concentration of orthophosphates in the Silvermine River compared to the Big Lotus River was possibly due to the fact that, the Silvermine River flows through a mostly natural catchment whereas, the Big Lotus River flows through a highly modified catchment that is heavily influenced by agriculture and urbanisation. The findings of the present study, supports those of Grobicki (2001) for the Big Lotus River, and Day and Ractliffe (2000) for the Silvermine River. The similarity, in the present study, of concentrations of orthophosphates between the Little Lotus River, Keysers River and the Langevlei Canal, could have been due to

similar activities having occurred within the various catchments. This is a possibility as both the Keyesers River and the Langevlei Canal originates in the Sand River Catchment. The Little Lotus River originates in the Zeekoevlei Catchment, which is the same catchment as the Big Lotus River, except that the Little Lotus River is less influenced by agriculture and more influenced by runoff from urban areas, which could be assumed to be similar to that of the Sand River Catchment.

The pH for all the rivers, in all seasons, was between 6 and 8 (Appendix B, Figure B1), which, according to DWAF (1996), is acceptable for rivers of the Western Cape. In the present study, the Big Lotus River, at a pH of 7.5 in summer and at a pH of 8 in autumn, showed total ammonia concentrations to be of acute toxicity as classified by DWAF (1996) While in winter and spring at a pH of 7.5 total ammonia concentrations were of chronic toxicity. The only other river that showed an acute and chronic toxicity of total ammonia was the Little Lotus River, presumably linked to low oxygen in summer. For all other seasons, the Little Lotus River and all other rivers showed low concentrations of total ammonia, below toxic thresholds (Appendix B, Figure B3). According to Dallas and Day (1993) and several other authors (DWAF, 1996; Kay, 1999; Dallas and Day, 2004; Moore and Palmer, 2005), ammonia is a common pollutant. High concentrations of ammonia can originate from commercial fertilisers containing highly soluble ammonia and ammonium salts which are transported through irrigation runoff, or from sewage discharge into the rivers. This is possibly the case in the present study, as the Big Lotus River is influenced mostly by the agricultural areas located in the catchment area, compared to that of all the other rivers. It is also a possibility that the concentration of total ammonia could have been due to sewage that was discharged into the river. The elevated concentrations of total ammonia in the Little Lotus River were also possibly related to sewage discharge, which could have been due to point source pollution entering the system from the informal settlements located higher up in the catchment.

### **Biotope and habitat availability**

While water quality determines which species could survive the water quality in a particular habitat, the availability and quality of such habitat could also limit whether such species could inhabit a section of river (Dallas, 1995; <sup>1</sup>Day, 2010, pers. comm.).

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<sup>1</sup> Dr. E. Day, Freshwater Ecologist, Freshwater Consulting Group, Cape Town

According to Grobicki (2001), even though water quality is a limiting factor on the type of organisms present at a site, habitat is an even more of important factor in limiting the presence of such organisms at a specific site. However, habitat and lifecycle also dictate which organisms occur where and when.

In the present study, the availability and quality of habitat was sparse in the Big Lotus River, Little Lotus River and the Langevlei Canal. It was diverse in the Keysers River, the Moddergat River and the Silvermine River. The quality of this habitat was however poor and homogenous in all rivers except the Silvermine River, where it was heterogeneous and provided a vast area for colonisation by organisms. These observations supported the river and vlei assessment and monitoring in the Cape Metropolitan Area (Harding et al., 2002). According to Harding et al. (2002), the Big Lotus River and the Keysers River were both categorised as class E rivers, and were identified as having a loss of natural habitat, biota, and basic ecosystem function, but had rehabilitation potential. The Little Lotus River and the Langevlei Canal were classified as class F rivers, and were identified as having reached a critical level in that the lotic system had been modified completely with a complete loss of natural habitat and biota, as well as ecosystem function. The Moddergat River and the Silvermine River were classified as class D rivers and were identified as rivers that had been largely modified, had a loss of natural habitat, but ecosystem function could still have occurred.

While, the classes of the rivers are not relevant to this study the classifications of the classes are. The present study confirmed the habitat availability and ecological function of the rivers, as classified by Harding et al. (2002), except that the Langevlei diversion channel showed that the Langevlei Canal had potential for more complex or natural ecological function if the canal walls were removed.

In the present study, it was noted that the substrate of the Keysers River, Big Lotus River, Silvermine River and Langevlei Canal were all similar, as they were earth channels. The Moddergat River was not similar to any other river as it was gabion-lined, and the Little Lotus River was not similar, as it was a concrete canal (Table 4.2, p 4-19). A section of the Little Lotus River and the Silvermine River were similar to that of the Moddergat River, as these sections had been gabion-lined. The Langevlei Canal and the Little Lotus River were also similar, as they both had sites that were

concrete canals. The Keyzers River and the Big Lotus River were similar in that they were both earth channels. The diversion channel of the Langevlei canal, that was sampled, was also an earth channel, but was not considered similar to the Keyzers River and the Big Lotus River, as the diversion channel was more representative of a lentic system than a lotic system, and is considered to be more similar to the Silvermine River. The Langevlei Canal itself was however similar to that of the Little Lotus River, as both rivers are canalised.

The Moddergat River, Silvermine River and the Langevlei Canal diversion channel, experienced seasonal flow and dried up in summer and autumn, therefore also limiting the availability of habitat. The Keyzers River, Big Lotus River and the Little Lotus River experienced flow all year, but due to the river systems being modified, the water levels and volume of flow were not necessarily representative of how the system should function under natural conditions.

## **19.2 Macroinvertebrates**

In the present study, using the similarity percentage (SIMPER), seven groupings (Table 12.1, pg 12-4) separated out at 60% similarity as illustrated in Figure 12.2 (pg 12-3). This determined the similarities between assemblages of macroinvertebrates between sites and seasons, across all rivers sampled.

All sites and seasons that fall in grouping one were considered 79.5% similar, and consisted of site BL1 in spring and site LL3 in spring. These similarities were characterised by the abundance of Chironomidae, which were likely to have been associated with the onset of spring conditions. As described by Dickens and Graham (2002), Gerber and Gabriel (2002) and Dallas and Day (2004), Chironomidae are known to occur almost anywhere in freshwater, are highly pollution tolerant and are known to multiply prolifically in favourable conditions, such as an increase in water temperature. In the present study, the presence and proliferation of Chironomidae at site LL3 in the Little Lotus River in spring, and site BL1 in the Big Lotus River in spring, could also have been influenced by water quality, as both rivers were equally polluted, as discussed in Chapter 14 (pg 14-1) and 18 (pg 18-1). This may have influenced the diversity of taxa as Chironomidae are pollution tolerant and often thrive in conditions of organic enrichment. The biotopes were also similar at site LL3 and site BL1 in spring (Table 4.1, p 4-12), as there were stones in current at both

sites. In the present study, it was a possibility that the similarity between sites and seasons were not related to habitat, as Chironomidae according to Grobicki (2001), Gerber and Gabriel (2002) and Dallas and Day (2004), are not specific to any particular biotope or substrate, and occur anywhere in freshwater. The habitat availability and quality in each of these rivers, at both sites, was sparse, which could have resulted in more macroinvertebrates being present in a small area due to habitat restrictions.

The sites and seasons that fall in grouping two were 51% similar to each other and 86% dissimilar to the remaining sites in all rivers. The similarity was characterized by the presence of individuals of Simuliidae and Baetidae. All the sites in this grouping were sampled in spring and were from the Moddergat River and the Silvermine River and the Langevelei Canal. These rivers were all associated with fair to good water quality which could explain the similarity of the seasons, but the habitat was different therefore possibly explaining why this group was not 60% similar. Group two was 86% dissimilar to the remaining sites in all rivers (rest). The remaining sites in all rivers were considered 41% similar to each other due to the presence of individuals of Chironomidae, Hirudinea, Pleidae and Baetidae taxa.

Grouping three was considered 71% similar within the grouping. All the sites in this grouping were sampled in spring in the Moddergat River, except site SM3 which was sampled in the Silvermine River in spring. This similarity could have been associated with the biotope sampled and geomorphology and characteristics of the rivers at these particular sites. According to Grobicki (2001), Dickens and Graham (2002) and Gerber and Gabriel (2002), species can proliferate with an increase in temperature which could be associated with a seasonal change such as spring. Baetidae and Simuliidae were both found at each site, in this grouping. These taxa, according to Gerber and Gabriel (2002), are generally associated with rocks, plants and moderately flowing streams. The habitat, biotope and structure of the Moddergat River, and site SM3 of the Silvermine River, were representative of the preferred habitat of Baetidae and Simuliidae. Site SM3 in the Silvermine River, and the sites in the Moddergat River were similar in habitat, with both consisting of a stone channel with marginal vegetation, and moderately flowing water. This was therefore a possible reason for the similarity between sites, and as such between the dominant species. While water quality was an unlikely driving factor, as the Silvermine River

showed better water quality than the Moddergat River, site SM3 in the Silvermine River did show a poorer water quality than the rest of the sites sampled in the Silvermine River. This suggested that the water quality could possibly have been a contributing factor to the similarity amongst species. According to Grobicki (2001) and Findlay and Taylor (2006), water quality is a limiting factor but habitat is more of a limiting criterion. In this case it appears that habitat (i.e. flowing water) was a driving factor, but species were very pollution tolerant.

Grouping four was considered 70% similar between sites LL1 and LL3 of the Little Lotus River in summer, and site KVS2 of the Keyzers River in winter. This similarity was characterised by Planorbidae. This taxon, according to Gerber and Gabriel (2002), is associated with aquatic vegetation, gravel beds and fast flowing streams. While all the sites in the grouping represented such conditions, the main similarity between the sites was the presence of aquatic vegetation. As discussed in Chapter 19.1 (pg 19-1), even though the water quality for both the Keyzers River and the Little Lotus River was fair to good, which was possibly a driving factor, Planorbidae are pollution tolerant and therefore it was unlikely that water quality was the contributing factor to the similarity between sites, as there would then have been similarities between all the rivers that had a fair to good water quality.

Grouping five was considered 77 % similar between sites in the Keyzers River at site KVS2 in summer, and KVS1, KVS2 and KVS 3 in spring, as well as site LV1 in the Langevlei Canal, in autumn. The water quality, as classified by DWAF (2008) and PD Naidoo and Associates (2010), for both the Keyzers River and the Langevlei Canal were fair to good. The samples were characterised by Chironomidae and Pleidae at all the sites in this grouping. Both these taxa are tolerant of polluted water. The similarity amongst sites in the Keyzers River was expected, as there was little difference in biotopes and habitat sampled between sites, and the characteristics of each site were similar (personal observation) (Table 4.1, p 4-12). The Langevlei Canal in autumn was similar to the Keyzers River as there was an abundance of marginal vegetation, the slopes were sandy and the water was shallow and clear. This may have created a similar habitat to that which was associated with the Keyzers River. Baetidae prefer rocks, plants and coarse sand and Chironomidae occur anywhere in freshwaters. The dense vegetation in both the Langevlei diversion channel and the Keyzers River could have provided similar habitat for the

establishment of similar taxa. The Keyzers River and the Langevlei Canal diversion channel are both sand channels which would also have assisted in the establishment of in-stream aquatic vegetation as well as marginal vegetation, compared to that of a concrete canal or gabion-lined river, such as the Little Lotus River or the Moddergat River.

Grouping six showed a 62 % similarity between sites and seasons at BL2 in the Big Lotus River in all seasons, site BL1 of the Big Lotus River in autumn and winter, sites LL1 and LL2 of the Little Lotus River in winter, the Keyzers River in spring at all sites, and at site KVS2 in summer and winter, in the Langevlei Canal in autumn at site LV1, all sites in the Moddergat River in winter, sites SM1 and SM3 in the Silvermine River in winter, and the Little Lotus River at sites LL1 and LL2 in summer (Figure 12.2, p 12-3). This similarity was characterised by Chironomidae and Pleidae. According to Gerber and Gabriel (2002), both Chironomidae and Pleidae are tolerant of poor quality water. According to the same authors, Chironomidae are found almost anywhere in freshwaters and Pleidae cling to submerged vegetation, and prefer shallow and clear water. All the sites sampled had marginal vegetation which may have been the driving factor for the similarity amongst these sites. Group six also consists of a seasonal similarity amongst sites which could be associated with seasonal changes, or simply a similarity in seasonal taxa. This is however unlikely as the characteristic species are specifically pollution tolerant. The relationship between these sites is therefore not considered truly similar.

The sites and seasons within grouping seven were 72 % similar. These included, the Moddergat River in winter at all sites, the Silvermine River in winter at all sites, the Keyzers River at site KVS3 in winter, the Little Lotus River at sites LL1 and LL2 in winter, and the Big Lotus River in winter and autumn at sites BL1 and BL2, as well as at site BL2 in summer, and site BL2 in spring (Figure 12.2, p 12-3). These rivers and sites were mostly similar in winter, and were characterised by an abundance of Chironomidae. As winter is associated with the highest rainfall and therefore the greatest volume of water and rate of flow, the majority of other macroinvertebrate taxa could have been removed during a scouring event such as fast flows or flooding. Or these taxa may not be fauna that occur in winter and therefore were not very abundant. This would have allowed Chironomidae to proliferate as there would be less competition from other species. While Chironomidae dominated, this was not

truly significant, as Chironomidae can occur anywhere in fresh waters (Gerber and Gabriel (2002). The water quality in these rivers for these seasons were not considered a factor as while these rivers and sites were all nutrient rich they all had varying trophic states and were polluted to varying degrees.

From the above groupings it is likely that the driving factor for the similarities between sites was habitat and water quality. While habitat was required for the taxa to colonise the area, the water quality also played a role in determining the type of taxa that were present, as different taxa are tolerant to different levels of pollution. The majority of the taxa that characterised the various groups were indeed pollution tolerant taxa such as Chironomidae. According to Gerber and Gabriel (2002), these taxa occur almost anywhere, are pollution tolerant, and can multiply easily under favourable conditions.

Seasonality having been a factor for similarity was not considered likely in the present study, as there were only similarities between winter and spring, with Chironomidae contributing 65% of the similarity of the taxa, and having the highest abundance. Again the similarity was driven by one taxon. As suggested by Grobicki (2001) and Findlay and Taylor (2006), while water quality is a limiting factor, habitat is more of a limiting criterion. Autumn and summer were dissimilar to winter and spring. When considering the sites and seasons of all the rivers together, these were less than 25% similar. When determining how similar the sites within each river were, the Big Lotus River showed the highest similarity (41%) across all seasons. Sites within all other rivers were only 25 % similar. Seasonality is thus probably a greater determinant of invertebrate community in these rivers than are the actual rivers themselves. Furthermore, water quality could also be an overriding factor when looking at species diversity in these polluted rivers.

## CHAPTER 20

### CONCLUSION

The previous chapters gave a description and discussion of the results of the relationship between measured environmental variables such as habitat availability and water quality, to the composition of macroinvertebrates and diatoms of the Keysers River, Little Lotus River, Langevlei Canal, Moddergat River and the Big Lotus River. These results were used to discuss whether the engineering interventions of each river, were successful in improving the ecological function of the river. Several conclusions could be drawn from these discussions, and are presented below.

When undertaking the present study, some constraints were experienced. More samples needed to be taken at each site to achieve larger data sets, which would have made the data more statistically representative. Data collected over a longer period of time would have shown better seasonal results, and dissolved oxygen measurements should have been included for reference with other research.

With regards to water chemistry, all the rivers experienced eutrophication. There was a great influence from point source pollutants and non-point source pollutants, in all the rivers. The Big Lotus River and the Little Lotus River were the most nutrient rich, and the Silvermine River, the least. The nutrient enrichment of the rivers did depend on the seasons. When considering the orthophosphate concentrations most rivers varied between eutrophic and hypertrophic (Little Lotus River, Moddergat River, and Big Lotus River), while others varied between mesotrophic and eutrophic (Keysers River, Langevlei Canal, Silvermine River), depending on the season, and the volume of rainfall and runoff entering the catchment. When considering the total inorganic nitrogen concentrations, most rivers varied between oligotrophic and eutrophic (Little Lotus River), oligotrophic and mesotrophic (Keysers River and Moddergat River), and, mesotrophic and eutrophic (Langevlei Canal) depending on the season, and the volume of rainfall and runoff entering the catchment.

The Big Lotus River was hypertrophic all year and the Silvermine River was oligotrophic all year, regardless of the season, rainfall and catchment runoff.

The effects of nutrient enrichment and pollution were visible in the macroinvertebrate results. The seasons that represented water quality of an oligotrophic status showed a moderate level of diversity of taxa, a eutrophic status a low diversity of taxa, and a mesotrophic status a high level of diversity of taxa. While these rivers supported various levels of diversity of taxa, the pollution sensitivity score of these taxa per season, per site never exceeded five. This indicates that all the taxa were highly pollution tolerant taxa. This could also have shown that while there was a diversity of taxa, it was the only taxa that could survive in these polluted waters.

However, the survival of these pollution tolerant taxa is dependent on habitat. Of all the rivers, the Silvermine River was the only river that had a large complex heterogeneous habitat for colonisation by macroinvertebrates. The Keyzers River and the reach sampled on the Langevelei canal, were well vegetated, but the habitat was homogenous, less complex, and only covered a small area. The Big Lotus River, Moddergat River and the Little Lotus River had very sparse, homogenous habitat. This lack of habitat was reflected in the macroinvertebrate results as there were fewer taxa than that of a natural system.

The abundance of macroinvertebrates also varied according to season, and was highest when the river was most nutrient rich compared to when there were fewer nutrients in the river. These seasonal changes weren't significant, but spring did show the most 'spikes' in the abundance of macroinvertebrates, and winter the most decreases, compared to all other seasons. It is a possibility that due to favourable conditions such as increased water temperature and reduced flows, the macroinvertebrates were more prolific in spring than in any other seasons. Winter showed the least macroinvertebrates in most rivers. This is assumed to have been as a result of scouring and flood events, the effects of which were greatest in the concrete canals (Little Lotus River), due to increased volumes of rainfall. This could also have assisted taxa with proliferating in spring, as there would be limited competition due to macroinvertebrates having been removed from the system due to high flows.

Diatoms were less sensitive to changes in river habitat and heterogeneity. There were no seasonal trends for diatom assemblages, but these did indicate similar water quality results, to that of the water chemistry in all rivers. The diatoms provided a look into the longer term water quality of each river, whereas the water chemistry only provided a snapshot of the water quality for a specific moment in time. In some rivers (the Keyzers River, Moddergat River, and at site SM3 in the Silvermine River) the assemblage of diatoms showed the water quality to be worse than that expressed by the water chemistry results. This is considered to be due to a pollution event, or changes that had occurred over time prior to sampling, which were not able to be determined from the water chemistry analysis.

The present study showed macroinvertebrates as being better indicators of catchment disturbance, and having a higher sensitivity to morphological change of a river, whereas diatoms are better indicators of water quality. However, it was determined that the presence of habitat was the secondary constraint in limiting the diversity and abundance of macroinvertebrates, and water quality was primary. Macroinvertebrates could cope with a decline in water quality at the expense of other more sensitive taxa, which are unable to tolerate highly polluted or eutrophic conditions. The survival of these taxa, however were dependent on the availability and quality of habitat. In the present study it was evident that the removal of habitat would further impact on the diversity and abundance of macroinvertebrates, as habitat provides macroinvertebrates protection from scouring and predators.

Therefore, from the dominance of pollution tolerant taxa at all sites, in all rivers, water quality was an overriding determinant for the abundance and diversity of macroinvertebrates. Habitat was therefore considered a secondary factor in determining the ability of the river to function ecologically, as without habitat the survival of the taxa present would have been limited.

Of the interventions that were assessed, no single intervention was superior over the other. Each intervention provided more diversity of habitat, than had it not existed. The implementation of a 100% re-vegetation after a project, as was applied to the Silvermine River, proved successful and was colonised by macroinvertebrates as it provided a complex habitat. Rivers that are earth canals proved more favourable for

the establishment of vegetation (habitat) than those that were concrete canals. The earth canals (Keysers River, Langevelei Canal diversion Channel and the Big Lotus River) allowed for the establishment of marginal vegetation, and in low flow areas the establishment of reeds, all of which provided vertically complex habitat. The placement of rocks in some of the rivers (Big Lotus River, Moddergat River and Little Lotus River) to slow flows provided improved conditions for the establishment of more complex and sheltered habitat, downstream of the intervention. These rock weirs also acted as informal litter traps and limited solid waste from moving downstream and increased aeration. The rocks did provide a greater diversity of substrate, but this was not significant as the water quality was the overriding factor, and therefore there was not a greater diversity of taxa. The removal of canal walls (Little Lotus River) also proved successful, as marginal vegetation could establish compared to upstream which was a concrete wall. Gabion-lining (Moddergat River) provided more diversity of habitat but did limit growth of vegetation.

If water quality is not addressed, the potential for recovery of macroinvertebrates could be overcome by the pollution. This could limit the potential of a river to recover, and that river could have no ecological function in the future. The present study clearly indicated that while these systems are currently "functioning", albeit at a very low level of ecological functioning, the potential for these rivers to recover, even if they had suitable habitat, could be severely hampered by the impact of the poor water quality. Improvements in water quality upstream and at the source need to become a focus in terms of improving the ecological functioning of urban rivers.

Therefore interventions do improve habitat diversity and so increase invertebrate diversity, albeit not to a large scale, because of water quality limitations. However, water quality can be improved over time, and an increase in the biological activity within channels adds to biodiversity by establishing a strong food chain for urban birds and frogs that still use water bodies.

If the water quality was improved, the colonisation of the river with a richer diversity of macroinvertebrate taxa may still be constrained not only because of the water quality being historically polluted but also because of the colonisation process. If measures to improve water chemistry are to be considered as means to enhance

macroinvertebrate taxa and thereby attain ecological targets, this may be adversely affected or severely delayed, particularly where water is polluted at the source.

The degree of “naturalness” of a system or an ecosystem is equated with ecosystem integrity and health. When considering the rivers in this study, their present ecological state is poor. However notwithstanding the above our urban rivers act as ecological corridors and links between the sea, wetlands and the mountains and these are needed to maintain ecological function as a whole. Therefore, even though it is not possible to preserve corridor function for the entire length of each river, these rivers provide vitally important links in the greater biodiversity of an area.

Looking beyond this study it is recommended that to improve the ecological functioning of urban river systems, water quality needs to be addressed at the source. Concrete canals that have wide riparian corridors should have the canal walls removed and the banks shaped appropriately to promote the establishment of marginal vegetation, even if this is only implemented in sections. Flow should be controlled to allow the establishment of vegetation and prevent scouring, but at the same time should mimic the seasonal variations expected (low flows and flooding). Riparian corridor management is critical to the distribution of benthic taxa, and therefore good quality and diversity of vegetation adjacent to rivers must be protected and enhanced, which will in turn improve stream communities. Streams should only be dredged when they no longer have capacity to convey stormwater without causing flooding, and not because they are part of a maintenance schedule.

As there was no background data for each river sampled it would be useful in future studies to assess the ecological functioning of a stream before an intervention is implemented, during the implementation and after the implementation to assess the true impact of such an intervention on the ecological function of the river.

Due to the fact that there are very few options for the mitigation of water quality to protect the ecological functioning of a river system the precautionary approach must be adopted. Therefore to minimise risk to the environment, decision makers must adopt the precautionary principle in planning and development.

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## **APPENDIX A**

## APPENDIX A

**Table A1:** The DWAF 1996 Target Water Quality Range for un-ionised ammonia in aquatic ecosystems.

TWQR and Criteria	Un-ionised Ammonia Concentration (mg/l N)
<i>Target Water Quality Range (TWQR)</i>	0.007
<i>Chronic Effect Value (CEV)</i>	0.015
<i>Acute Effect Value (AEV)</i>	0.1

**Table A2:** Specific Pollution Sensitivity Index and percentage Pollution Tolerants Values: Interpretation of Index scores

Interpretation of index scores	
Index score	Class
>17	high quality
13 to 17	good quality
9 to 13	moderate quality
5 to 9	poor quality
<5	bad quality

**Table A3:** Classification of ecological indicators interpretation

<b>Classification of Ecological Indicators</b>			
<b>pH</b>			
acidobiontic	optimal occurrence at pH <5.5		
acidophilous	mainly occurring at pH <7		
circumneutral	mainly occurring at pH values about 7		
alkaliphilous	mainly occurring at pH >7		
alkalibiontic	exclusively occurring at pH >7		
indifferent	no apparent optimum		
<b>Salinity</b>			
	<b>Cl<sup>-</sup> (mg l<sup>-1</sup>)</b>	<b>Salinity (‰)</b>	<b>Cond. mS/m</b>
fresh	<100	<0.2	<3
fresh-brackish	<500	<0.9	<139
brackish-fresh	500-1000	0.9-1.8	139-277
brackish	1000-5000	1.8-9.0	277-1385
<b>Nitrogen uptake mechanism</b>			
Nitrogen autotrophic - sensitive	tolerating very small concentrations of organically bound nitrogen		
Nitrogen autotrophic - tolerant	tolerating elevated concentrations of organically bound nitrogen		
Nitrogen heterotrophic - facultative	needing periodically elevated concentrations of organically bound nitrogen		
Nitrogen heterotrophic - obligatory	needing continuously elevated concentrations of organically bound nitrogen		
<b>Oxygen requirements</b>			
continuously high	~100% saturation		
fairly high	>75% saturation		
moderate	>50% saturation		
low	>30% saturation		
very low	~10% saturation		
<b>Saprobity</b>			
	<b>Pollution</b>	<b>Oxygen saturation</b>	<b>BOD<sub>5</sub> (mg l<sup>-1</sup>)</b>
oligosaprobous	Unpolluted to slightly polluted	>85	<2
β-mesosaprobous	Moderately polluted	70 - 85	2 - 4
α-mesosaprobous	Strongly polluted	25 - 70	4 - 13
α-meso-polysaprobous	Very heavily polluted	10 - 25	13 - 22
polysaprobous	Extremely polluted	<10	>22

**Table A4:** Summary of ranges of phosphorus and/or nitrogen based nutrients associated with different trophic conditions in aquatic ecosystems. Nitrogen and Phosphorus ranges from DWAF (1996).

Trophic state	Effects	Average summer inorganic <u>nitrogen</u> concentrations (mg/l)	Average summer inorganic <u>phosphorus</u> concentrations (mg/l)
Oligotrophic	Moderate levels of species diversity; usually low productivity systems with rapid nutrient cycling; no nuisance growth of aquatic plants or the presence of blue-green algae	< 0.5	< 0.005
Mesotrophic	Usually high levels of species diversity; usually productive systems; nuisance growth of aquatic plants and blooms of blue-green algae; algal blooms seldom toxic	0.5-2.5	> 0.005-0.025
Eutrophic	Usually low levels of species diversity; usually highly productive systems; nuisance growths of aquatic plants and blue-green algae, algal blooms may include species that are toxic to man, livestock and wildlife	2.5-10	>0.025-0.25
Hypertrophic	Usually very low levels of species diversity; usually very highly productive systems; nuisance growths of aquatic plants and blue-green algae, algal blooms may include species that are toxic to man, livestock and wildlife	>10	> 0.25

**Table A5:** Ecological reserve water quality Criteria.as sourced by PD Naidoo (2010) from DWAF (2004) and DWAF (2008)

Variable	Units	Natural	Good	Fair	Poor	Unacceptable	Comments
Temperature*	°C	Depends on background (upper boundary – 90 <sup>th</sup> percentile; Lower boundary = 10 <sup>th</sup> percentile); Good ± 2°C; Fair ± 4°C; Poor ± >4°C					Need to determine typical background water quality- not essential for prioritisation exercise.
TSS*	Mg/l	Depends on background ( not more than 10 % higher than background)					Need to determine typical background water quality- not essential for prioritisation exercise.
DO*	Mg/l	>8	8-6	6-4	4-2	<2	Also dependent on background levels to some extent. No unacceptable range given but if one selects equal bands then 2 mg/l is the next logical band and is applicable to assessing the actual data.
Conductivity#	mS/m	Depends on background (not more than 15% different form normal cycle)					Need to determine the typical background water quality – not essential for prioritisation exercise.
pH*	units	8-6.5	9-8 or 6.5-5.75	10-9 or 5.75-5	>10; <5		Need to determine typical background water quality- not essential for prioritisation exercise.
SRP*	mg/l	<0.005	0.005 - 0.025	0.025 - 0.125	0.125 - 0.250	>0.250	Ranges as recommended in the latest water quality benchmarks for the Ecological Reserve (DWAF 2005).
TIN*	mg/l	<0.25	0.25-1	1-4	4-10	>10	
Ammonia*	mg/l	<0.015	0.015-0.058	0.058-0.1	0.1-0.2	>0.2	No unacceptable range given but if one selects equal bands then 0.2 mg/l is the next logical band and is applicable to assessing the actual data.
Algae*	µg/l	<10	10-20	20-30	30-40	>40	No unacceptable range given but if one selects equal bands then 40 µg/l is the next logical band and is applicable to assessing the actual data.

# South African Water Quality Guidelines (DWAF, 2004), \*Ecological Reserve water quality benchmarks (2005).

## **APPENDIX B**







Figure B4

### Total Phosphorus and Orthophosphates

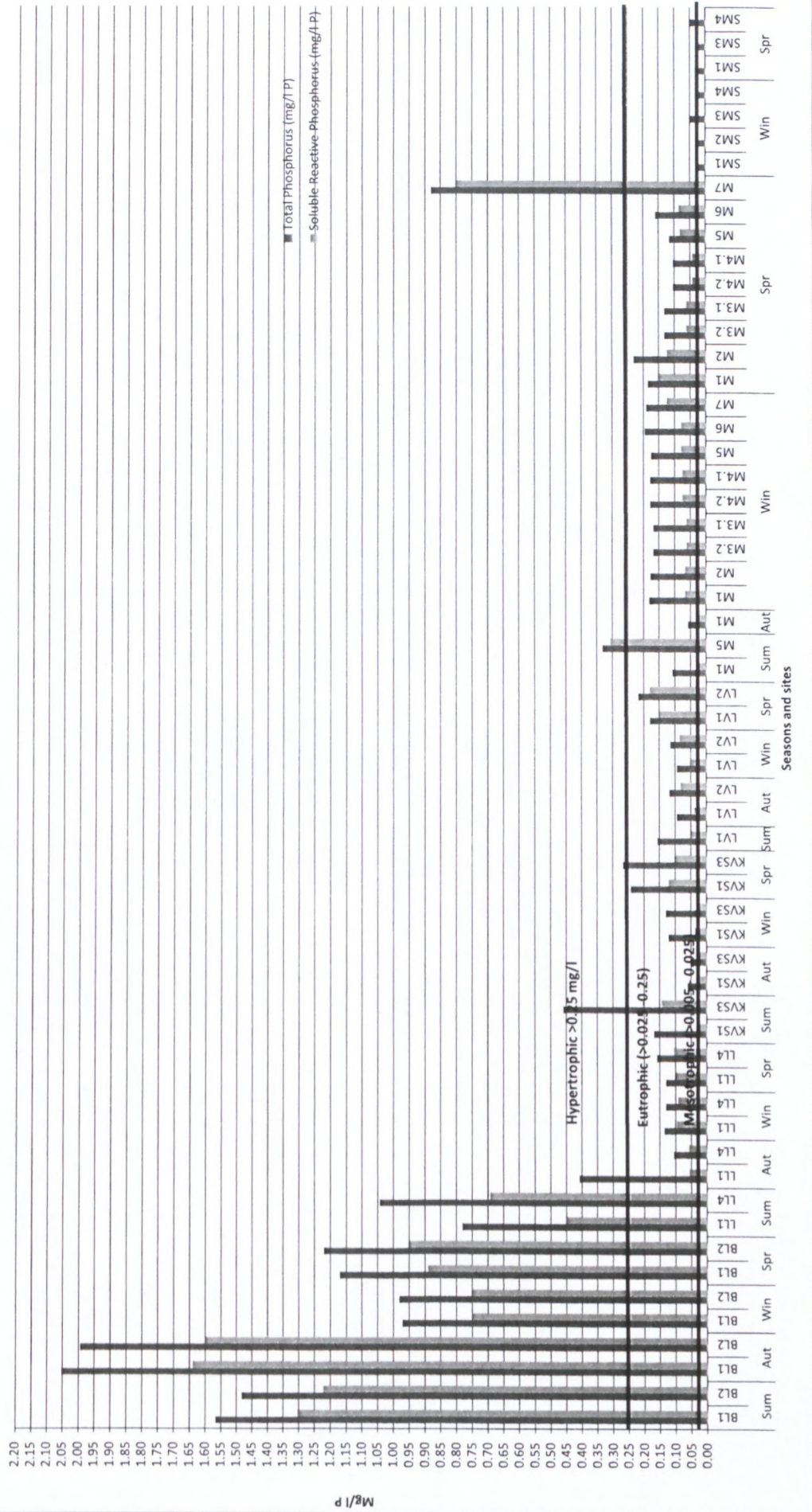


Figure B4: Total phosphorus and orthophosphates (mg/l P) for all rivers, measured per season, and per site. BL – Big Lotus River, LL – Little Lotus River, LV – Langevelei Canal, M – Moddergat River, SM – Silvermine River, BL – Big Lotus River. Sum- summer, Aut – autumn, Win – winter, Spr – spring.

Figure B5

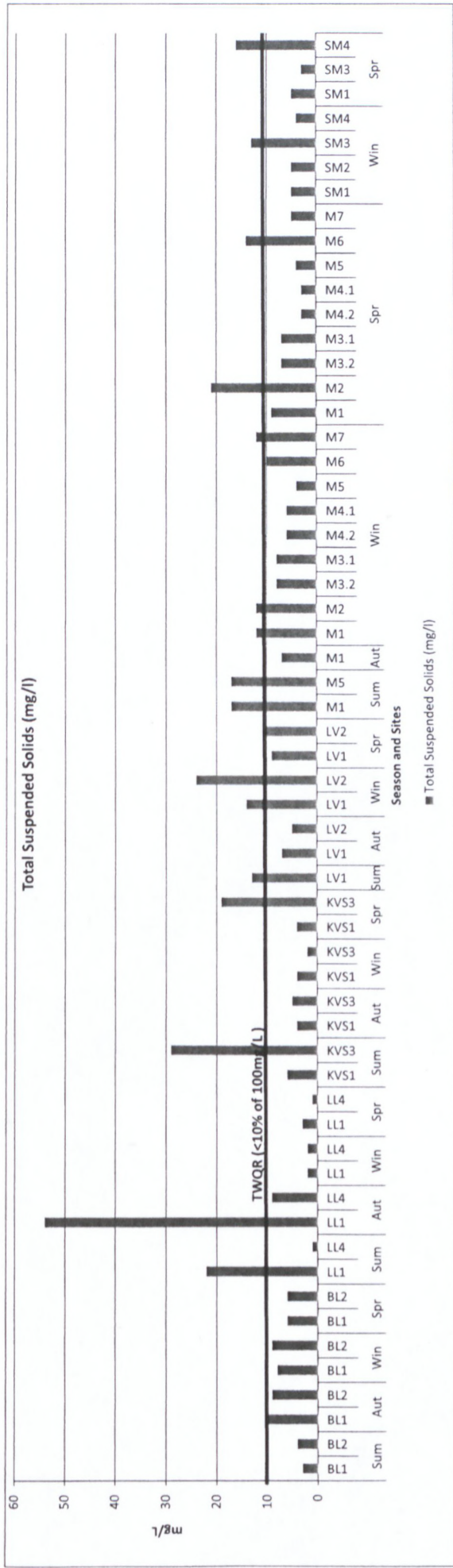


Figure B5: Total suspended solids (TSS) (mg/l) for all rivers, measured per season, and per site. BL – Big Lotus River, LL – Little Lotus River, LV – Langevelei Canal, M – Moddergat River, SM – Silvermine River, BL – Big Lotus River. Sum- summer, Aut – autumn, Win – winter, Spr – spring.



## **APPENDIX C**

**APPENDIX C: CITY OF CAPE TOWN DATA FOR THE PRECEDING TEN YEARS**

**Table C1: Water quality data as given by the City of Cape Town for the preceding 10 years (Raw data as provided by Catchment Management Department).**

River	COCT River Code	Date	water temperature	dissolved oxygen	oxygen saturation	total suspended solids	PH	conductivity	chem oxygen demand	total N	soluble ammonia	un-ionised ammonia	nitrite+nitrate	TON	total P	orthop
Keyser river	CR20	2000/01/27	20.80	3.50	39.00	0.00	8.80	86.00	60.00	0.76	0.09	0.02	0.84	0.04	0.02	0.01
Keyser river	CR20	2000/02/22	19.40	10.00	108.00	5.00	7.80	102.00	41.00	0.75	0.06	0.00	0.82	1.35	0.01	0.01
Keyser river	CR20	2000/03/30	17.00	2.90	30.00		7.30	65.00	34.00	0.64	0.12	0.00	0.75	0.01	0.01	0.00
Keyser river	CR20	2000/04/26				1.00	8.00	59.00	18.00	1.68	0.14		1.82	0.75	0.05	0.02
Keyser river	CR20	2000/05/23	14.40	5.60	55.00	2.00	7.30	70.00	22.00	1.68	0.09	0.00	1.77	0.04	0.03	0.01
Keyser river	CR20	2000/06/29	12.60	11.30	106.00	1.00	7.20	78.00	22.00	4.87	0.08	0.00	4.95	1.78	0.14	0.02
Keyser river	CR20	2000/07/27	10.30	8.40	74.00	4.00	6.90	82.00	43.00	1.43	0.01	0.00	1.44	0.79	0.05	0.02
Keyser river	CR20	2000/08/25	12.70	5.60	53.00	6.00	6.90	64.00	24.00	1.58	0.12	0.00	1.69	0.54	0.04	0.02
Keyser river	CR20	2000/09/27	13.40	7.50	71.00	22.00	7.00	45.00	44.00	1.80			1.80	0.36	0.08	0.04
Keyser river	CR20	2000/10/26	14.70	6.30	62.00	12.00	7.70	53.00	24.00	0.77	0.16	0.00	0.93	0.27	0.07	0.03
Keyser river	CR20	2000/11/30	20.70	4.80	53.00	1.00	7.90	70.00	21.00	0.32	0.01	0.00	0.32	0.04	0.07	0.02
Keyser river	CR20	2001/02/22	20.80	4.80	54.00	1.00	7.70	71.00	25.00	0.84	0.42	0.01	1.25	0.02	0.03	0.01
Keyser river	CR20	2001/03/29	17.10	4.00	41.00	5.00	7.10	83.00	27.00	0.72	0.11	0.00	0.83	0.01	0.02	
Keyser river	CR20	2001/04/26	14.10	1.10	11.00	4.00	7.60	76.00	26.00	0.17	0.02	0.00	0.19	0.05	0.02	0.01
Keyser river	CR20	2001/05/31	16.90	5.40	56.00	2.00	6.90	61.00	28.00	1.07	0.12	0.00	1.19	0.44	0.06	0.02

Keyser river	CR20		2001/06/28	13.40	6.70	64.00	4.00	7.20	65.00	26.00	0.97	0.01	0.00	0.97	0.42	0.05	0.02
Keyser river	CR20		2001/07/26	11.40	8.30	76.00	17.00	7.20	51.00	31.00	1.87	0.07	0.00	1.94	1.13	0.10	0.08
Keyser river	CR20		2001/08/31	12.00	12.30	113.00	80.00	7.20	33.00	35.00	1.69	0.14	0.00	1.83	0.77	0.25	0.08
Keyser river	CR20		2001/09/20	16.00	12.50	126.00	39.00	7.40	52.00	17.00	1.76	0.13	0.00	1.89	0.02	0.08	0.03
Keyser river	CR20		2001/10/25	17.10	11.00	114.00	24.00	7.20	50.00	21.00	1.20	0.06	0.00	1.26	0.51	0.12	0.01
Keyser river	CR20		2001/11/29	20.70	8.40	94.00	15.00	7.20	57.00	35.00	1.10	0.07	0.00	1.16	0.32	0.12	0.03
Keyser river	CR20		2001/12/20	21.90	10.10	115.00	18.00	7.90	63.00	66.00	0.34	0.26	0.01	0.61	0.00	0.08	0.01
Keyser river	CR20		2002/01/31	21.30	2.40	27.00		7.10	66.00	35.00	0.66	0.05	0.00	0.71	0.01	0.02	0.01
Keyser river	CR20		2002/02/28	22.70	2.20	26.00	2.00	7.10	71.00	49.00	0.63	0.03	0.00	0.66	0.01	0.03	0.02
Keyser river	CR20		2002/03/19	20.40	3.10	34.00	14.00	6.90	62.00	42.00	0.52	0.01	0.00	0.53	1.35	0.05	0.01
Keyser river	CR20		2002/04/25	17.70	1.20	13.00	4.00	6.40	48.00	76.00	0.92	0.27	0.00	1.19	0.17	0.05	0.03
Keyser river	CR20		2002/05/30	11.70	6.50	60.00	3.00	6.50	44.00	65.00	1.39	0.27	0.00	1.66	0.33	0.05	0.04
Keyser river	CR20		2002/06/27	10.70	6.60	59.00	32.00	6.60	40.00	64.00	1.75	0.31	0.00	2.06	0.45	0.12	0.09
Keyser river	CR20		2002/07/25	12.90	7.80	73.00	20.00	6.80	51.00	26.00	1.89	0.21	0.00	2.10	1.24	0.07	0.00
Keyser river	CR20		2002/08/29	13.10	5.30	50.00	20.00	7.30	55.00	60.00	1.58	0.27	0.00	1.85	0.67	0.08	0.02
Keyser river	CR20		2002/09/26	14.90	5.40	53.00	58.00	6.80	53.00	36.00	1.02	0.27	0.00	1.30	0.47	0.06	0.02
Keyser river	CR20		2002/10/31	16.40	5.40	55.00	42.00	6.90	57.00	12.00	0.43	0.08	0.00	0.51	0.27	0.18	0.02
Keyser river	CR20		2002/11/28	23.00	2.50	29.00	16.00	6.70	60.00	34.00		0.27	0.00	0.27		0.18	0.02
Keyser river	CR20		2002/12/19	20.20	3.20	35.00	34.00	6.80	60.00	63.00		0.14	0.00	0.14	0.19	0.14	0.02
Keyser river	CR20		2003/01/30	20.70	2.90	32.00	10.00	7.20	65.00	23.00		0.01	0.00	0.01	0.01	0.08	0.01
Keyser river	CR20		2003/02/27	22.30	3.80	44.00	9.00	7.20	43.00	24.00	0.61	0.02	0.00	0.63	0.01	0.08	0.08
Keyser river	CR20		2003/03/27	16.70	2.50	26.00		6.70	38.00	39.00	0.76	0.01	0.00	0.76	0.15	0.04	0.01
Keyser river	CR20		2003/04/24	16.30	4.40	45.00	2.00	5.80	54.00	15.00	0.90	0.03	0.00	0.93	0.22	0.03	0.00
Keyser river	CR20		2003/05/29	20.10	8.10	89.00	25.00	6.30	45.00	26.00		0.04	0.00	0.04	0.58	0.12	0.02
Keyser river	CR20		2003/06/25	11.40	5.10	46.00	141.00	6.30	58.00	19.00	1.04	0.18	0.00	1.22	0.61	0.74	0.04
Keyser river	CR20		2003/07/23	12.50	6.00	56.00	181.00	6.60	63.00	31.00	0.90	0.03	0.00	0.93	0.67	0.45	0.00
Keyser river	CR20		2003/08/28	10.50	6.90	61.00	10.00	6.20	56.00	34.00	1.10	0.15	0.00	1.25	0.73	0.07	0.02
Keyser river	CR20		2003/10/29	18.70	3.90	42.00	53.00	6.80	60.00	25.00	0.88	0.16	0.00	1.04	0.28	0.16	0.01
Keyser river	CR20		2003/11/27	24.00	7.70	92.00	16.00	8.30	104.00	29.00	0.82	0.03	0.00	0.84	0.41	0.10	0.02

Keyser river	CR20	2003/12/17	18.50	3.50	37.00	94.00	6.70	56.00	73.00	0.67	0.03	0.00	0.70	0.02	0.23	0.00
Keyser river	CR20	2004/01/29	16.70	1.30	13.00	28.00	7.30	82.00	19.00	0.58	0.06	0.00	0.64	0.03	0.29	0.00
Keyser river	CR20	2004/02/26	20.10	8.00	88.00	10.00	8.50	99.00	22.00	1.37	0.02	0.00	1.39	0.51	0.12	0.01
Keyser river	CR20	2004/03/25	18.30	1.20	13.00	75.30	7.10	98.00	47.00	0.88	0.03	0.00	0.91	0.04	0.65	0.07
Keyser river	CR20	2004/04/29	15.60	1.50	15.00	7.60	6.70	84.00	30.00	0.30	0.06	0.00	0.36	0.02	0.04	0.02
Keyser river	CR20	2004/05/27	11.90	4.00	37.00	2.00	7.00	87.00	17.00	0.71	0.04	0.00	0.75	0.09	0.04	0.03
Keyser river	CR20	2004/06/24	13.70	5.00	48.00	7.20	7.30	70.00	62.00	0.75	0.05	0.00	0.80	0.52	0.05	0.02
Keyser river	CR20	2004/07/29	13.40	8.00	76.00	7.20	7.30	92.00	28.00	2.23	0.31	0.00	2.54	1.55	0.12	0.05
Keyser river	CR20	2004/08/19	9.40	8.90	77.00	21.60	6.90	64.00	33.00	1.43	0.06	0.00	1.49	0.74	0.09	0.03
Keyser river	CR20	2004/09/30	17.20	7.00	73.00	25.20	7.00	67.00	5.00	1.12	0.18	0.00	1.30	0.53	0.11	0.02
Keyser river	CR20	2004/10/28	17.50	7.70	80.00	33.20	9.30	57.00	42.00	1.35	0.02	0.01	1.37	0.51	0.20	0.04
Keyser river	CR20	2004/11/25	21.00	5.70	64.00	10.00	7.40	90.00	12.00	2.46	0.04	0.00	2.50	1.56	0.09	0.02
Keyser river	CR20	2004/12/20	21.10	3.90	44.00	21.20	7.30	83.00	33.00	0.58	0.06	0.00	0.64	0.05	0.10	0.03
Keyser river	CR20	2005/01/27	21.10	2.40	27.00	11.20	7.40	107.00	18.00	0.58	0.01	0.00	0.59	0.00	0.05	0.01
Keyser river	CR20	2005/04/26	17.60	6.70	70.00	22.40	8.00	66.00	16.00	0.87	0.06	0.00	0.93	0.34	0.11	0.02
Keyser river	CR20	2005/05/24	11.40	4.90	45.00	1.60	7.20	53.00	49.00	1.39	0.23	0.00	1.62	0.52	0.08	0.05
Keyser river	CR20	2005/06/15	13.10	9.80	93.00	10.40	6.60	48.00	48.00	1.13	0.09	0.00	1.22	0.56	0.10	0.05
Keyser river	CR20	2005/07/19	11.70	7.20	66.00	24.80	7.10	31.00	13.00	1.34	0.28	0.00	1.61	0.47	0.18	0.05
Keyser river	CR20	2005/08/18	11.60	7.60	69.00	61.60	7.50	34.00	45.00	1.63	0.10	0.00	1.72	0.70	0.21	0.04
Keyser river	CR20	2005/09/12	15.10	7.60	75.00	20.40	6.90	59.00	19.00	1.29	0.13	0.00	1.42	0.77	0.12	0.01
Keyser river	CR20	2005/10/11	11.70	10.40	95.00	28.00	7.50	60.00	51.00	1.28	0.11	0.00	1.39	0.48	0.13	0.01
Keyser river	CR20	2005/11/07	15.20	5.00	50.00	79.00	7.20	66.00	21.00	0.81	0.11	0.00	0.92	0.22	0.20	0.04
Keyser river	CR20	2005/12/06	17.70	10.00	105.00	13.20	7.90	76.00	35.00	0.58	0.01	0.00	0.59	0.00	0.10	0.01
Keyser river	CR20	2006/01/24	15.70	2.70	27.00	6.80	7.80	79.00	43.00	0.84	0.05	0.00	0.89	0.01	0.07	0.01
Keyser river	CR20	2006/02/20	16.10	1.60	16.00	14.00	7.80	99.00	6.00	0.74	0.02	0.00	0.76	0.02	0.07	0.02
Keyser river	CR20	2006/03/13	13.70	0.50	5.00	28.00	6.50	84.00	57.00	2.01	0.41	0.00	2.42	0.01	2.80	1.49
Keyser river	CR20	2006/04/24	12.70	1.30	12.00	8.40	6.90	54.00	32.00	0.72	0.13	0.00	0.85	0.12	0.04	0.02
Keyser river	CR20	2006/05/29	11.70	7.40	68.00	5.00	7.30	53.00	48.00	0.89	0.13	0.00	1.02	0.32	0.09	0.03
Keyser river	CR20	2006/06/20	11.10	2.20	20.00	3.00	7.10	70.00	24.00	1.07	0.08	0.00	1.15	0.53	0.03	0.02

Keyser river	CR20	2006/07/18	9.10	10.00	86.00	7.00	7.00	53.00	31.00	1.18	0.12	0.00	1.30	0.44	0.10	0.04
Keyser river	CR20	2006/08/15	11.50	6.50	59.00	34.40	7.00	43.00	69.00	1.17	0.08	0.00	1.25	0.48	0.16	0.05
Keyser river	CR20	2006/09/14	13.90	2.50	24.00	3.00	7.10	64.00	39.00	1.19	0.11	0.00	1.30	0.52	0.07	0.01
Keyser river	CR20	2006/10/12	12.70	5.50	52.00	26.00	7.10	62.00	23.00	1.31	0.12	0.00	1.43	0.44	0.17	0.01
Keyser river	CR20	2006/11/23	21.10	3.40	38.00	21.00	7.80	65.00	26.00	0.84	0.02	0.00	0.86	0.12	0.22	0.01
Keyser river	CR20	2006/12/18	18.10	4.70	50.00	12.00	8.00	83.00	39.00	0.78	0.09	0.00	0.87	0.03	0.38	0.05
Keyser river	CR20	2007/01/08	19.30	2.00	22.00	7.80	7.40	72.00	33.00	0.46	0.05	0.00	0.51	0.03	0.10	0.04
Keyser river	CR20	2007/02/15				26.00	7.80	57.00	21.00	0.46	0.01		0.47	0.01	0.22	0.03
Keyser river	CR20	2007/03/15				21.00	8.00	67.00	30.00	0.67	0.03		0.70	0.02	0.13	0.00
Keyser river	CR20	2007/05/15	16.30	2.30	23.00	17.00	7.40	54.00	25.00	0.82	0.01	0.00	0.83	0.39	0.06	0.04
Keyser river	CR20	2007/06/18	9.70	7.30	64.00	17.00	7.20	54.00	24.00	0.82	0.13	0.00	0.95	0.11	0.17	0.05
Keyser river	CR20	2007/07/16	12.90	7.50	71.00	11.20	7.40	66.00	16.00	1.46	0.09	0.00	1.55	0.68	0.10	0.02
Keyser river	CR20	2007/08/07	11.10	10.50	95.00	90.00	7.00	45.00	50.00	1.35	0.09	0.00	1.44	0.64	0.24	0.04
Keyser river	CR20	2007/09/04	14.10	7.40	72.00	54.00	7.50	51.00	40.00	1.53	0.07	0.00	1.60	0.70	0.17	0.05
Keyser river	CR20	2007/10/02	12.70	4.40	41.00	56.00	7.60	65.00	28.00	1.08	0.06	0.00	1.14	0.61	0.17	0.02
Keyser river	CR20	2007/11/15	13.70	5.20	50.00	25.00	7.30	63.00	37.00	1.24	0.15	0.00	1.38	0.52	0.29	0.11
Keyser river	CR20	2007/12/12	19.70	12.10	132.00	6.00	9.00	74.00	24.00	1.95	0.04	0.01	2.00	1.28	0.05	0.02
<b>Mean</b>	<b>CR20</b>		<b>16.71</b>	<b>5.56</b>	<b>64.03</b>	<b>24.91</b>	<b>7.40</b>	<b>72.90</b>	<b>34.28</b>	<b>1.17</b>	<b>0.11</b>	<b>0.00</b>	<b>0.33</b>	<b>0.44</b>	<b>0.11</b>	<b>0.03</b>
<b>Summer</b>	<b>CR20</b>		<b>21.19</b>	<b>4.24</b>	<b>61.00</b>	<b>21.28</b>	<b>7.62</b>	<b>139.87</b>	<b>26.06</b>	<b>0.87</b>	<b>0.09</b>	<b>0.00</b>	<b>0.29</b>	<b>0.24</b>	<b>0.10</b>	<b>0.03</b>
<b>Winter</b>	<b>CR20</b>		<b>11.68</b>	<b>13.96</b>	<b>71.73</b>	<b>25.60</b>	<b>15.59</b>	<b>51.21</b>	<b>42.07</b>	<b>1.11</b>	<b>0.11</b>	<b>0.02</b>	<b>0.46</b>	<b>0.57</b>	<b>0.09</b>	<b>0.02</b>
Little Lotus River	LR11	2000/01/13	20.00	13.40	148.00		8.40	108.00	40.00		6.34	0.65	6.98	0.90	0.70	0.41
Little Lotus River	LR11	2000/02/10	23.10	16.50	194.00	12.00	8.50	165.00	64.00	7.57	2.06	0.31	2.37	0.77	0.44	0.38
Little Lotus River	LR11	2000/03/02	19.50	11.20	123.00	10.00	8.40	140.00	48.00	3.75	0.12	0.01	0.13	0.22	0.72	0.48
Little Lotus River	LR11	2000/04/17	19.00	17.90	194.00	34.00	8.60	139.00	46.00	10.34	2.88	0.42	3.30	2.44	0.54	0.36
Little Lotus River	LR11	2000/05/29	15.20	4.40	44.00	21.00	7.60	40.00	23.00	3.12	1.28	0.02	1.29	0.96	0.21	0.18
Little Lotus River	LR11	2000/06/21	14.60	11.30	112.00	12.00	8.00	141.00	51.00	13.05	3.82	0.11	3.93	6.16	0.44	0.37
Little Lotus River	LR11	2000/07/19	11.80	8.40	77.00	22.00	7.70	109.00	43.00	14.71	1.35	0.02	1.36	5.92	0.53	0.06
Little Lotus River	LR11	2000/08/01	13.40	14.40	138.00	10.00	7.50	171.00	54.00	13.58	3.85	0.03	3.88	6.78		0.51



Little Lotus River	LR11		2003/01/09	19.40	11.60	127.00	5.00	8.30	130.00	52.00		0.91	0.07	0.98	2.03	1.40	1.10
Little Lotus River	LR11		2003/02/06	21.80	3.40	39.00	2.00	7.50	75.00	44.00		0.24	0.00	0.24	0.12	0.17	0.10
Little Lotus River	LR11		2003/03/06	21.40	5.00	56.00	0.00	7.30	46.00	52.00	3.04	0.15	0.00	0.15	0.04	0.04	0.04
<b>Mean</b>				<b>18.27</b>	<b>8.71</b>	<b>97.61</b>	<b>16.73</b>	<b>8.00</b>	<b>91.66</b>	<b>56.85</b>	<b>4.59</b>	<b>0.59</b>	<b>0.03</b>	<b>0.36</b>	<b>1.49</b>	<b>0.30</b>	<b>0.16</b>
<b>Summer</b>	<b>February</b>			<b>22.32</b>	<b>8.70</b>	<b>101.24</b>	<b>8.15</b>	<b>8.28</b>	<b>92.58</b>	<b>56.93</b>	<b>1.93</b>	<b>0.26</b>	<b>0.02</b>	<b>0.16</b>	<b>0.22</b>	<b>0.14</b>	<b>0.06</b>
<b>Winter</b>	<b>August</b>			<b>14.26</b>	<b>8.94</b>	<b>86.89</b>	<b>9.31</b>	<b>7.72</b>	<b>67.91</b>	<b>62.04</b>	<b>5.82</b>	<b>0.37</b>	<b>0.01</b>	<b>0.30</b>	<b>2.67</b>	<b>0.30</b>	<b>0.22</b>
Langevlei Canal - NO DATA																	
Moddergat River	ek18		2001/01/18	22.60	7.80	90.00	59.00	8.90	144.00	58.00	2.79	0.46	0.14	0.25		0.19	0.08
Moddergat River	ek18		2001/02/15	22.60	7.80	90.00	59.00	8.90	144.00	58.00	2.79	0.46	0.14	0.25		0.19	0.08
Moddergat River	ek18		2001/06/07	15.40	7.70	77.00	10.00	8.10	83.00	47.00	4.25	0.08	0.00	1.28		0.13	0.09
Moddergat River	ek18		2001/07/05	11.40	9.60	87.00	3.00	7.60	42.00	48.00	4.31	0.28	0.00	3.01		0.51	0.29
Moddergat River	ek18		2001/07/09	15.40	7.70	77.00	10.00	8.10	83.00	47.00	4.25	0.08	0.00	1.28		0.13	0.09
Moddergat River	ek18		2001/08/02	13.20	8.00	76.00	10.00	7.70	67.00	36.00	2.58	0.15	0.00	1.63		0.36	0.10
Moddergat River	ek18		2001/09/06	13.70	13.10	126.00	35.00	7.90	49.00	41.00	2.29	0.00	0.00	1.33		0.38	0.07
Moddergat River	ek18		2001/10/04	16.80	14.90	153.00	14.00	7.70	64.00	33.00	5.59	0.04	0.00	0.91		0.17	0.06
Moddergat River	ek18		2001/11/01	19.90	12.80	140.00	14.00	8.10	55.00	34.00	2.24	0.04	0.00	1.32		0.21	0.12
Moddergat River	ek18		2001/12/06	23.20	12.70	149.00	16.00	8.30	71.00	11.00	0.54	0.24	0.02	0.24		0.11	0.03
Moddergat River	ek18		2002/01/24	20.50	7.90	88.00	18.00	8.20	64.00	39.00	0.81	0.13	0.01	0.08		0.06	0.05
Moddergat River	ek18		2002/02/21	23.20	15.10	178.00	21.00	7.70	94.00	52.00	1.51	0.13	0.00	0.37		0.06	0.03
Moddergat River	ek18		2002/03/14	22.80	13.80	161.00	16.00	7.90	150.00	106.00	5.28	0.84	0.04	0.93		0.30	0.27
Moddergat River	ek18		2002/04/18	21.70	7.30	83.00	243.00	7.90	98.00	102.00	20.79	13.38	0.52	1.77		6.49	3.72
Moddergat River	ek18		2002/05/23	14.40	9.90	96.00	14.00	7.70	57.00	95.00	2.95	0.60	0.01	0.65		0.19	0.13
Moddergat River	ek18		2002/06/20	10.80	8.10	73.00	1.00	7.90	67.00	43.00	6.43	1.07	0.02	0.96		0.33	0.28
Moddergat River	ek18		2002/07/18	10.80	8.10	73.00	10.00	7.60	57.00	53.00	3.25	0.34	0.00	1.31		0.22	0.14
Moddergat River	ek18		2002/08/22	13.20	7.80	74.00	12.00	7.40	63.00	45.00	2.12	0.22	0.00	0.83		0.15	0.10
Moddergat River	ek18		2002/09/19	17.70	7.90	83.00	11.00	8.00	49.00	32.00	2.11	0.04	0.00	0.38		0.14	0.07
Moddergat River	ek18		2002/10/24	17.50	10.90	114.00	5.00	8.40	60.00	30.00	1.91			0.24		0.08	0.07

Moddergat River	ek18	2002/11/21	19.20	10.30	111.00	8.00	8.20	80.00	35.00	3.71	0.12	0.01	0.73		0.10	0.08
Moddergat River	ek18	2002/12/12	22.30	5.10	59.00	105.00	7.40	92.00	72.00		14.29	0.19	0.34		1.82	0.83
Moddergat River	ek18	2003/01/23	16.50	6.60	67.00	13.00	7.50	110.00	39.00		0.08	0.00	0.31		0.16	0.10
Moddergat River	ek18	2003/04/10	13.70	4.60	44.00	86.00	7.20	39.00	114.00	1.93	0.02	0.00	0.19		0.03	
Moddergat River	ek18	2003/05/15					7.00	64.00	25.00		0.02		0.17		0.14	0.09
Moddergat River	ek18	2003/06/12	7.80	9.00	75.00		6.30	57.00	33.00	2.48	0.04	0.00	0.38		0.08	0.08
Moddergat River	ek18	2003/07/24	8.20	7.40	62.00		7.00	63.00	26.00	1.06	0.03	0.00	0.18		0.05	0.03
Moddergat River	ek18	2003/08/21	8.20	9.90	83.00	14.00	6.20	60.00	40.00	3.01	0.04	0.00	1.53		0.16	0.10
Moddergat River	ek18	2003/09/18	14.60	8.70	85.00	40.00	6.70	67.00	26.00	2.41	0.03	0.00	0.37		0.08	
Moddergat River	ek18	2003/10/16	14.10	5.50	53.00	10.00	7.20	63.00	26.00	1.25	0.03	0.00	0.66		0.22	0.11
Moddergat River	ek18	2003/11/13	15.30	4.10	41.00	0.00	6.80	67.40	48.00	0.56	0.03	0.00	0.04		0.08	0.06
Moddergat River	ek18	2003/12/18	19.50	10.60	115.00	2.00	7.00	62.70	18.00	6.51	2.74	0.01	2.67		3.60	3.27
Moddergat River	ek18	2004/01/15	19.70	7.50	82.00	12.00	7.10	60.00	17.00	0.41	0.04	0.00	0.15		0.14	0.08
Moddergat River	ek18	2004/05/13	12.90	6.40	60.00	310.00	7.40	77.10	39.00	0.67	0.06	0.00	0.12		0.91	0.14
Moddergat River	ek18	2004/06/10	11.20	7.50	68.00	6.80	7.30	74.40	24.00	0.82	0.03	0.00	0.24		0.10	0.08
Moddergat River	ek18	2004/07/08	8.40	6.10	52.00	6.40	8.10	77.60	33.00	0.77	0.05	0.00	0.35		0.08	0.05
Moddergat River	ek18	2004/08/05	11.50	8.40	77.00	49.60	6.60	24.20	3.00	2.00	0.32	0.00	0.64		0.81	0.24
Moddergat River	ek18	2004/09/02	19.80	7.80	85.00	8.00	7.80	75.60	32.00		6.12	0.17	0.56		0.09	0.08
Moddergat River	ek18	2004/10/14	17.30	8.10	84.00	8.80	7.20	70.50	30.00	1.62	0.06	0.00	0.64		0.15	0.09
Moddergat River	ek18	2004/11/11	21.70	6.90	78.00		7.60	80.80	13.00	0.77	0.03	0.00	0.11		0.18	0.09
Moddergat River	ek18	2004/12/09	20.40	6.70	74.00	12.00	6.50	19.30	39.00	0.74	0.08	0.00	0.28		0.09	0.08
Moddergat River	ek18	2005/04/14	17.40	3.80	40.00	4.40	7.80	136.90	25.00	1.06	0.05	0.00	0.09		0.24	0.17
Moddergat River	ek18	2005/05/12	15.20	5.50	55.00	39.20	7.60	95.20	33.00	1.20	0.05	0.00	0.04		0.06	0.04
Moddergat River	ek18	2005/06/09	10.40	9.10	81.00	37.60	7.70	63.40	21.00	2.81	0.07	0.00	1.77		0.28	0.12
Moddergat River	ek18	2005/07/14	13.70	6.80	65.00	19.60	8.10	86.20	22.00	1.58	0.06	0.00	0.71		0.11	0.06
Moddergat River	ek18	2005/08/04	8.70	9.80	84.00	14.40	8.10	70.30	37.00	1.82	0.05	0.00	1.01		0.20	0.10
Moddergat River	ek18	2005/09/08	14.30	10.10	98.00	14.40	7.80	67.00	28.00	2.89	0.04	0.00	0.73		0.16	0.09
Moddergat River	ek18	2005/10/17	16.10	8.70	88.00	72.00	7.80	89.70	35.00	1.04	0.08	0.00	0.42		0.29	0.04
Moddergat River	ek18	2005/11/14	20.90	6.40	72.00	3.00	7.10	68.90	34.00	0.90	0.02	0.00	0.18		0.09	0.04

Moddergat River	ek18		2006/05/18	15.60	3.60	36.00	12.00	7.50	67.00	40.00	1.06	0.08	0.00	0.26		0.19	0.12
Moddergat River	ek18		2006/06/29	12.20	5.80	54.00	40.00	7.70	77.00	36.00	1.38	0.06	0.00	0.57		0.13	0.06
Moddergat River	ek18		2006/07/27	13.40	6.00	57.00	7.00	7.80	72.20	42.00	2.19	0.22	0.00	1.40		0.22	0.10
Moddergat River	ek18		2006/08/24	15.70	5.70	57.00	7.00	7.80	73.00	96.00	2.09	0.10	0.00	1.09		0.24	0.12
Moddergat River	ek18		2006/09/21	14.20	5.90	57.00	8.00	7.60	71.00	45.00	0.75	0.03	0.00	0.26		0.15	0.10
Moddergat River	ek18		2006/10/19	18.50	2.20	23.00	8.00	7.40	142.00	34.00	2.36	0.99	0.01	0.60		0.21	0.14
Moddergat River	ek18		2006/11/16	19.40	3.90	42.00	10.00	7.70	89.00	36.00	0.74	0.05	0.00	0.17		0.14	0.12
Moddergat River	ek18		2006/12/14	15.50	3.60	36.00	20.00	7.80	156.00	39.00	1.68	0.77	0.02	0.29		0.33	0.21
Moddergat River	ek18		2007/05/17	16.40	5.70	58.00	11.00	7.50	82.00	30.00	1.10	0.09	0.00	0.15		0.11	0.09
Moddergat River	ek18		2007/06/14	12.20	6.90	64.00	9.00	7.80	50.00	34.00	2.14	0.02	0.00	1.68		0.22	0.20
Moddergat River	ek18		2007/07/12	9.70	6.20	54.00	4.00	7.80	63.00	33.00	1.70	0.02	0.00	0.88		0.11	0.07
Moddergat River	ek18		2007/08/23	9.60	4.20	37.00	18.00	7.90	94.40	53.00	2.19	0.03	0.00	1.53		0.17	0.08
Moddergat River	ek18		2007/09/13	14.20	6.70	65.00	10.00	7.80	65.70	32.00	1.45	0.02	0.00	0.62		0.20	0.09
Moddergat River	ek18		2007/10/18	18.30	5.70	60.00	7.00	7.70	69.00	33.00	4.84	0.14	0.00	0.36		0.17	0.11
Moddergat River	ek18		2007/11/15	19.20	5.00	54.00	6.00	7.70	74.00	54.00	0.74	0.04	0.00	0.09		0.67	0.08
Moddergat River	ek18		2007/12/13	15.40	5.40	54.00	6.00	7.70	99.00	38.00	0.81	0.03	0.00	0.29		0.13	0.09
<b>Mean</b>				<b>15.71</b>	<b>7.64</b>	<b>77.09</b>	<b>27.22</b>	<b>7.62</b>	<b>76.42</b>	<b>40.80</b>	<b>2.46</b>	<b>0.72</b>	<b>0.02</b>	<b>0.69</b>		<b>0.37</b>	<b>0.22</b>
<b>Summer</b>				<b>16.84</b>	<b>6.76</b>	<b>75.29</b>	<b>19.57</b>	<b>6.50</b>	<b>83.33</b>	<b>34.86</b>	<b>1.19</b>	<b>0.22</b>	<b>0.02</b>	<b>0.23</b>		<b>0.14</b>	<b>0.08</b>
<b>Winter</b>				<b>10.01</b>	<b>6.73</b>	<b>61.00</b>	<b>15.63</b>	<b>6.46</b>	<b>56.49</b>	<b>38.75</b>	<b>1.98</b>	<b>0.11</b>	<b>0.00</b>	<b>1.03</b>		<b>0.26</b>	<b>0.11</b>
Silvermine River	sil04		2000/08/22	13.10	8.90	84.00		6.50	33.40	39.00		0.17	0.00	0.22		0.02	0.01
Silvermine River	sil04		2000/09/20	13.00	7.60	72.00	1.00	5.80	23.40	64.00	1.11	0.17	0.00	0.15		0.03	0.01
Silvermine River	sil04		2000/10/17	14.80	7.60	75.00	1.00	7.10	29.20	55.00	0.82	0.00	0.00	0.18		0.03	0.02
Silvermine River	sil04		2000/12/12	17.90	6.70	70.00	10.00	7.50	55.80	35.00	0.46			0.11		0.06	0.02
Silvermine River	sil04		2001/05/24	13.00	9.60	91.00	4.00	7.30	50.10	24.00	0.44	0.08	0.00	0.02		0.04	0.01
Silvermine River	sil04		2001/06/06	13.00	8.60	81.00	11.00	5.60	28.90	66.00	1.29	0.46	0.00	0.17		0.06	0.02
Silvermine River	sil04		2001/07/04	14.30	10.60	103.00	38.00	5.40	17.90	66.00	1.34	0.09	0.00	0.18		0.07	0.06
Silvermine River	sil04		2001/08/08	13.20	13.70	130.00	1.00	5.90	18.30	44.00	1.82	0.13	0.00	0.53		0.25	0.08
Silvermine River	sil04		2001/09/05	12.90	13.20	124.00	8.00	6.40	17.70	66.00	1.10	0.03	0.00	0.22		0.06	0.03

Silvermine River	sil04		2001/10/10	17.10	11.40	118.00	3.00	7.40	38.10	43.00	1.11	0.04	0.00	0.33		0.07	0.05
Silvermine River	sil04		2001/11/22	20.00	10.10	111.00	3.00	7.70	53.90	44.00	1.13	0.02	0.00	0.36		0.07	0.04
Silvermine River	sil04		2001/12/04	18.90	12.40	133.00	2.00	7.00	61.70	36.00	1.29	0.06	0.00	0.43		0.03	0.03
Silvermine River	sil04		2002/01/23	21.50	5.00	57.00	15.00	6.60	48.50	43.00	0.74	0.02	0.00	0.10		0.06	0.04
Silvermine River	sil04		2002/02/20	20.50	6.80	75.00	2.00	7.40	72.10	35.00	0.95	0.06	0.00	0.10		0.03	0.01
Silvermine River	sil04		2002/03/20	18.00	12.90	136.00	10.00	7.80	78.90	37.00	0.61	0.00	0.00	0.04		0.02	0.02
Silvermine River	sil04		2002/04/17	16.80	8.60	88.00	7.00	7.60	77.10	34.00	0.71			0.06		0.02	0.02
Silvermine River	sil04		2002/05/21	14.90	7.60	75.00	6.00	7.70	76.30	58.00	0.84	0.05	0.00	0.20		0.03	0.03
Silvermine River	sil04		2002/06/24	12.00	7.60	70.00	3.00	6.20	32.60	62.00	0.64	0.06	0.00	0.22		0.02	0.02
Silvermine River	sil04		2002/07/22	11.70	7.40	68.00	2.00	5.60	30.20	54.00	1.10	0.09	0.00			0.02	0.01
Silvermine River	sil04		2002/08/14	11.60	7.30	67.00	3.00	5.10	27.60	78.00	1.22	0.01	0.00	0.39		0.19	0.18
Silvermine River	sil04		2002/09/16					5.80	31.70	55.00	1.20	0.07		0.35		0.03	0.02
Silvermine River	sil04		2002/10/22	15.80	6.90	69.00	4.00	6.50	43.50	34.00	0.98	0.02	0.00	0.24		0.04	0.02
Silvermine River	sil04		2002/11/18	14.70	5.70	56.00	3.00	6.60	50.10	33.00	0.69	0.00	0.00	0.15		0.02	0.01
Silvermine River	sil04		2002/12/09	14.70	5.70	56.00	2.00	7.30	63.40	34.00		0.03	0.00	0.11		0.04	0.02
Silvermine River	sil04		2003/01/20	18.30	7.80	83.00		7.40	68.10	31.00		0.07	0.00	0.07		0.02	0.02
Silvermine River	sil04		2003/04/02	16.10	7.10	72.00	8.00	7.30	40.00	25.00		0.01	0.00	0.17		0.07	0.02
Silvermine River	sil04		2003/05/05	16.70	8.80	90.00	11.00	7.20	67.90	36.00		0.03	0.00	0.13		0.11	0.03
Silvermine River	sil04		2003/06/02	14.20	9.90	96.00	39.00	3.80	30.80	23.00	0.53	0.02	0.00	0.14		0.09	0.01
Silvermine River	sil04		2003/07/03	8.40	10.00	85.00	2.00	6.90	62.70	17.00	0.40	0.02	0.00	0.04		0.03	0.01
Silvermine River	sil04		2003/08/13	11.90	8.80	81.00		5.30	38.20	42.00	0.73	0.03	0.00	0.18		0.03	0.01
Silvermine River	sil04		2003/09/02	8.10	9.80	82.00	2.00	2.60	14.70	55.00	0.53	0.03	0.00	0.03		0.02	0.01
Silvermine River	sil04		2003/10/15	13.70	6.50	62.00	2.00	6.40	35.20	39.00	0.84	0.03	0.00	0.13		0.03	0.01
Silvermine River	sil04		2003/11/12	14.50	10.60	104.00	2.00	6.90	49.10	34.00	0.88	0.04	0.00	0.07		0.01	0.01
Silvermine River	sil04		2003/12/01	8.10	9.80	82.00	2.00	6.80	62.90	41.00	0.51	0.01	0.00	0.02		0.01	0.00
Silvermine River	sil04		2004/02/10	22.70	7.70	89.00	1.00	4.40	32.00	9.00	0.47	0.04	0.00	0.04		0.01	0.01
Silvermine River	sil04		2004/05/03	15.20	6.60	65.00	5.20	7.50	87.10	101.00	0.30	0.01	0.00	0.01		0.02	0.02
Silvermine River	sil04		2004/06/07	13.50	7.80	74.00	0.00	7.90	71.70	9.00	0.48	0.02	0.00	0.02		0.03	0.02
Silvermine River	sil04		2004/07/05	7.90	7.50	63.00	1.20	6.70	38.10	34.00	0.35	0.04	0.00	0.05		0.01	0.00

Silvermine River	sil04	2004/08/02	8.60	7.30	62.00	1.60	4.90	23.50	68.00	1.02	0.06	0.00	0.15		0.03	0.02
Silvermine River	sil04	2004/09/09	12.00	10.10	93.00	4.40	6.80	55.50	57.00	1.22	0.05	0.00	0.26		0.05	0.03
Silvermine River	sil04	2004/10/11	14.90	6.90	68.00	2.80	6.50	58.60	83.00	1.30	0.04	0.00	0.06		0.05	0.03
Silvermine River	sil04	2004/11/08	14.20	8.40	81.00	4.00	6.80	53.20	102.00	1.02	0.06	0.00	0.13		0.04	0.02
Silvermine River	sil04	2004/12/02	16.40	8.70	89.00	3.20	7.00	64.20	30.00	0.73	0.04	0.00	0.23		0.06	0.02
Silvermine River	sil04	2005/01/04	16.50	8.10	83.00	64.80	3.90	86.30	16.00	0.61	0.04	0.00	0.05		0.19	0.00
Silvermine River	sil04	2005/02/03	21.50	5.40	61.00	18.00	7.60	40.70	17.00	1.36	0.04	0.00	0.65		0.10	0.05
Silvermine River	sil04	2005/04/11	15.60	10.60	106.00	10.80	7.40	64.30	35.00	1.70	0.05	0.00	0.70		0.28	0.15
Silvermine River	sil04	2005/05/09	13.10	9.10	86.00	2.90	6.70	44.20	54.00	0.71	0.02	0.00	0.06		0.05	0.02
Silvermine River	sil04	2005/06/06	13.00	7.90	75.00	0.40	6.20	28.90	36.00	0.89	0.04	0.00	0.13		0.03	0.01
Silvermine River	sil04	2005/07/04	12.90	6.90	65.00	2.80	6.30	33.90	24.00	1.13	0.07	0.00	0.35		0.04	0.02
Silvermine River	sil04	2005/08/11	10.10	8.50	75.00	3.20	7.00	25.80	101.00	0.80	0.04	0.00	0.16		0.07	0.01
Silvermine River	sil04	2005/09/08	9.70	11.30	99.00	3.60	6.80	22.00	58.00	0.74	0.04	0.00	0.16		0.02	0.02
Silvermine River	sil04	2005/10/10	13.30	12.30	117.00	2.00	7.50	34.00	99.00	0.73	0.08	0.00	0.17		0.04	0.00
Silvermine River	sil04	2005/11/07	15.50	7.70	77.00	5.00	7.10	43.00	47.00	0.75	0.04	0.00	0.16		0.07	0.02
Silvermine River	sil04	2005/12/06	18.20	8.40	89.00	2.80	7.80	62.70	55.00	0.67	0.02	0.00	0.16		0.04	0.01
Silvermine River	sil04	2006/04/24	12.70	9.50	89.00	21.20	7.70	81.60	32.00	0.91	0.06	0.00	0.03		0.13	0.08
Silvermine River	sil04	2006/05/29	12.50	7.50	70.00	3.00	6.10	28.00	60.00	0.68	0.06	0.00	0.12		0.03	0.00
Silvermine River	sil04	2006/06/26	13.60	7.00	67.00	6.00	6.20	34.00	51.00	0.94	0.05	0.00	0.18		0.03	0.00
Silvermine River	sil04	2006/07/13	12.50	8.50	79.00	3.00	5.60	22.00	92.00	0.60	0.04	0.00	0.07		0.02	0.01
Silvermine River	sil04	2006/08/21	14.10	6.10	59.00	6.00	6.10	23.00	49.00	0.90	0.09	0.00	0.21		0.04	0.00
Silvermine River	sil04	2006/09/19	13.30	8.70	83.00	5.00	7.00	36.00	49.00	1.03	0.05	0.00	0.14		0.02	0.00
Silvermine River	sil04	2006/10/17	16.30	5.60	57.00	4.00	7.20	43.20	63.00	0.73	0.04	0.00	0.14		0.04	0.01
Silvermine River	sil04	2006/11/14	18.20	7.40	78.00	3.00	7.50	42.00	51.00	0.57	0.04	0.00	0.11		0.03	0.00
Silvermine River	sil04	2006/12/11	13.70	6.40	61.00	3.00	8.00	64.00	36.00	0.56	0.02	0.00	0.07		0.02	0.01
Silvermine River	sil04	2007/01/09	19.50	7.40	80.00	3.40	7.60	60.40	22.00	0.39	0.04	0.00	0.04		0.02	0.01
Silvermine River	sil04	2007/05/28	13.70	8.80	84.00	3.00	7.80	33.00	52.00	0.66	0.03	0.00	0.15		0.04	0.00
Silvermine River	sil04	2007/06/28	12.70	9.10	85.00	2.00	6.30	22.00	63.00	0.76	0.06	0.00	0.13		0.02	0.01
Silvermine River	sil04	2007/07/26	12.60	7.30	68.00	9.00	7.00	18.00	76.00	1.16	0.04	0.00	0.19		0.02	0.02

Silvermine River	sil04		2007/08/30	12.70	5.60	53.00	4.00	7.10	28.00	25.00	0.97	0.03	0.00	0.24		0.03	0.03
Silvermine River	sil04		2007/09/27	12.70	1.80	17.00	4.00	7.30	32.00	42.00	0.75	0.05	0.00	0.24		0.03	0.00
Silvermine River	sil04		2007/10/31	20.20	7.90	87.00	10.00	7.40	39.40	40.00	0.88	0.06	0.00	0.23		0.05	0.02
Silvermine River	sil04		2007/11/19	19.20	5.10	55.00	3.00	7.40	37.00	31.00	0.87	0.11	0.00	0.15		0.04	0.02
Silvermine River	sil04		2007/12/11	19.10	6.30	68.00	30.00	7.40	37.00	51.00	0.69	0.03	0.00	0.17		0.09	0.02
<b>Mean</b>				<b>14.61</b>	<b>8.23</b>	<b>80.32</b>	<b>6.90</b>	<b>6.67</b>	<b>44.17</b>	<b>47.25</b>	<b>0.85</b>	<b>0.05</b>	<b>0.00</b>	<b>0.17</b>		<b>0.05</b>	<b>0.02</b>
<b>Summer</b>				<b>19.32</b>	<b>7.08</b>	<b>76.33</b>	<b>7.24</b>	<b>7.55</b>	<b>59.97</b>	<b>32.50</b>	<b>0.77</b>	<b>0.05</b>	<b>0.00</b>	<b>0.19</b>		<b>0.04</b>	<b>0.02</b>
<b>Winter</b>				<b>11.08</b>	<b>7.60</b>	<b>70.64</b>	<b>3.20</b>	<b>5.63</b>	<b>26.35</b>	<b>49.45</b>	<b>0.95</b>	<b>0.06</b>	<b>0.00</b>	<b>0.21</b>		<b>0.06</b>	<b>0.03</b>
Big Lotus River	lr03		2000/01/13	23.30	3.80	45.00		8.10	121.00	45.00	10.97	7.81	0.52	0.85		0.72	0.40
Big Lotus River	lr03		2000/02/10	26.30	3.80	47.00	16.00	7.90	130.00	68.00	17.34	14.08	0.72	0.47		0.49	0.28
Big Lotus River	lr03		2000/03/02	23.30	1.80	21.00	60.00	8.10	131.00	128.00	89.95	38.86	2.58			3.12	2.75
Big Lotus River	lr03		2000/04/17	20.20	2.30	25.00	17.00	8.10	141.00	64.00	45.40	27.87	1.52	0.21		1.32	1.13
Big Lotus River	lr03		2000/05/29	16.60	3.20	33.00	32.00	7.60	79.00	44.00	8.97	5.51	0.08	2.67		0.54	0.53
Big Lotus River	lr03		2000/06/21	15.90	9.60	98.00	13.00	8.00	125.00	44.00	17.55	9.96	0.33	6.57		0.49	0.33
Big Lotus River	lr03		2000/07/19	12.20	7.40	69.00	63.00	7.60	84.00	42.00	22.40	2.27	0.02	0.31		0.92	0.40
Big Lotus River	lr03		2000/08/01	15.10	11.10	111.00	14.00	7.80	148.00	49.00	24.55	9.21	0.18	9.05		0.40	0.40
Big Lotus River	lr03		2000/09/05	16.00	4.60	46.00	12.00	8.20	128.00	54.00	22.06	7.02	0.36	6.81		0.55	0.55
Big Lotus River	lr03		2000/10/24	20.20	7.20	79.00	12.00	8.20	126.00	66.00	22.32	17.67	1.20	2.06		1.45	1.07
Big Lotus River	lr03		2000/11/23	22.00	5.90	67.00	7.00	8.10	139.00	40.00	19.91	15.15	0.93	2.82		0.75	0.68
Big Lotus River	lr03		2001/01/11	25.90	8.70	108.00	25.00	8.30	148.00	64.00	17.39	12.34	1.45	1.95		0.77	0.43
Big Lotus River	lr03		2001/01/19				63.00	7.60	84.00	42.00	22.40	2.27		0.31		0.92	0.40
Big Lotus River	lr03		2001/02/08	19.10	6.00	65.00	42.00	7.90	128.00	121.00	44.95	26.72	0.87	0.47		4.45	3.29
Big Lotus River	lr03		2001/03/08	21.20	3.70	42.00	33.00	7.90	144.00	56.00	31.84	22.61	0.85	0.74		1.37	1.17
Big Lotus River	lr03		2001/04/11				84.00	7.50	115.00	133.00		29.18		0.18		3.79	1.86
Big Lotus River	lr03		2001/05/10	16.70	8.60	88.00	9.00	7.80	125.00	63.00	26.76	15.79	0.35	3.68		1.14	1.05
Big Lotus River	lr03		2001/06/14	15.40	7.00	71.00	7.00	7.90	157.00	56.00	27.89	13.33	0.34	6.53		0.97	0.65
Big Lotus River	lr03		2001/07/12	13.80	8.70	85.00	22.00	7.70	134.00	63.00	26.57	7.54	0.11	6.63		0.57	0.20
Big Lotus River	lr03		2001/08/14	16.10	7.40	75.00	33.00	7.70	132.00	43.00	21.60	0.89	0.02	6.35		1.48	1.20



Big Lotus River	lr03	2004/01/22	22.10	4.30	49.00	20.00	7.70	139.00	48.00	20.01	19.24	0.49	0.48	1.62	1.38
Big Lotus River	lr03	2004/02/19				9.00	7.80	136.00	53.00	23.45	21.95		0.53	1.54	1.11
Big Lotus River	lr03	2004/03/18	20.40	3.40	38.00	25.20	7.70	133.00	54.00	27.85	24.99	0.57	0.34	2.29	1.78
Big Lotus River	lr03	2004/04/22	21.00	7.10	80.00	19.00	7.90	116.00	56.00	12.64	8.59	0.32	3.47	0.81	0.53
Big Lotus River	lr03	2004/05/20	14.40	5.30	52.00	6.00	7.90	158.00	42.00	18.72	16.61	0.39	1.10	1.08	0.77
Big Lotus River	lr03	2004/06/17	15.80	5.30	53.00	8.80	7.60	117.00	49.00	13.35	7.81	0.10	4.27	0.65	0.54
Big Lotus River	lr03	2004/07/15	9.50	6.00	52.00	8.80	7.70	155.00	45.00	20.80	12.26	0.13	4.42	0.77	0.66
Big Lotus River	lr03	2004/08/12	17.20	7.20	75.00	14.40	7.80	104.00	20.00	19.26	5.37	0.12	7.35	0.92	0.48
Big Lotus River	lr03	2004/09/16	20.10	8.30	92.00	10.80	8.10	153.00	28.00	21.21	10.33	0.56	5.32	0.58	0.50
Big Lotus River	lr03	2004/10/21	18.30	6.80	72.00	338.50	7.80	118.00	41.00	10.96	3.57	0.09	5.39	0.75	0.30
Big Lotus River	lr03	2004/11/18	23.70	9.70	116.00	13.20	8.10	178.00	60.00	22.48	15.80	1.08	2.41	0.67	0.34
Big Lotus River	lr03	2004/12/09	25.80	4.30	53.00	14.00	7.90	173.00	45.00	25.47	18.61	0.93	1.07	0.95	0.43
Big Lotus River	lr03	2005/01/20	22.70	7.70	90.00	10.00	8.10	152.00	44.00	24.08	21.70	1.39	0.42	1.33	0.73
Big Lotus River	lr03	2005/02/17	20.70	5.10	57.00	21.60	8.20	183.00	65.00	27.16	21.88	1.54	1.06	1.79	0.99
Big Lotus River	lr03	2005/03/15	25.50	10.40	128.00	18.40	8.30	176.00	51.00	27.86	22.47	2.59	0.77	1.53	0.99
Big Lotus River	lr03	2005/04/21	17.90	6.10	64.00	46.00	8.00	111.00	8.00	13.40	6.49	0.24	4.18	0.56	0.32
Big Lotus River	lr03	2005/05/17	13.40	5.90	56.00	36.40	8.00	13.00	68.00	15.13	10.17	0.27	3.92	1.02	0.60
Big Lotus River	lr03	2005/06/20	14.60	6.10	60.00	6.80	7.90	132.00	45.00	15.67	4.97	0.12	7.14	2.44	0.26
Big Lotus River	lr03	2005/07/21	15.60	8.70	88.00	11.60	8.50	153.00	18.00	15.26	4.98	0.47	8.51	0.40	0.27
Big Lotus River	lr03	2005/08/22	11.50	5.30	48.00	16.40	8.40	121.00	20.00	16.57	8.23	0.47	5.93	0.56	0.32
Big Lotus River	lr03	2005/09/20	14.90	4.70	46.00	8.80	7.70	133.00	56.00	15.57	11.38	0.18	3.84	0.52	0.40
Big Lotus River	lr03	2005/10/24	19.20	4.60	50.00	28.00	8.00	160.00	53.00	27.14	19.28	0.79	2.94	0.83	0.54
Big Lotus River	lr03	2005/11/22	20.80	6.80	76.00	33.00	8.10	151.00	49.00	20.40	18.10	1.03	1.81	1.43	0.73
Big Lotus River	lr03	2005/12/15	21.70	17.80	203.00	22.00	8.30	157.00	41.00	28.96	20.46	1.89	1.56	1.60	0.81
Big Lotus River	lr03	2006/01/19	16.70	3.30	34.00	16.00	8.10	155.00	83.00	28.52	23.73	1.03	0.16	19.81	14.07
Big Lotus River	lr03	2006/02/14	23.90	13.70	164.00	25.00	8.20	163.00	34.00	29.20	25.93	2.21	0.61	1.89	1.43
Big Lotus River	lr03	2006/03/09	19.10	6.70	73.00	30.00	8.40	167.00	55.00	24.85	22.16	2.15	1.00	1.96	1.32
Big Lotus River	lr03	2006/04/06	18.30	2.50	27.00	60.00	7.80	156.00	72.00	23.53	21.68	0.54	0.59	2.33	1.45
Big Lotus River	lr03	2006/05/04	18.90	1.00	11.00	176.00	7.90	127.00	150.00	39.40	30.98	1.00	0.19	4.85	3.05

Big Lotus River	lr03	2006/06/01	14.70	8.50	83.00	25.00	7.90	121.00	40.00	11.74	7.40	0.18	3.96	0.53	0.33
Big Lotus River	lr03	2006/07/04	18.90	1.00	11.00	392.00	7.70	32.00	40.00	4.92	2.49	0.05	0.96	2.13	0.54
Big Lotus River	lr03	2006/08/03	16.30	4.30	44.00	25.00	7.70	181.00	50.00	12.75	7.39	0.13	3.89	0.57	0.39
Big Lotus River	lr03	2006/09/07	14.70	3.70	36.00	10.00	7.40	156.00	122.00	17.40	14.35	0.11	2.60	0.98	0.67
Big Lotus River	lr03	2006/10/05	22.40	1.30	15.00	31.00	7.80	158.00	52.00	25.13	18.87	0.62	1.87	1.56	0.85
Big Lotus River	lr03	2006/11/02	20.00	2.70	30.00	52.00	7.70	105.00	42.00	10.15	7.15	0.16	2.07	0.99	0.43
Big Lotus River	lr03	2006/12/07	16.70	1.60	16.00	18.00	7.90	153.00	57.00	16.40	14.17	0.39	0.78	1.49	0.81
Big Lotus River	lr03	2007/01/11	25.90	3.80	47.00	30.80	8.20	145.00	66.00	13.95	0.47	0.05	1.14	1.63	1.03
Big Lotus River	lr03	2007/02/13	25.30	1.40	17.00	17.00	8.00	123.00	37.00	11.71	8.84	0.53	0.84	1.21	0.66
Big Lotus River	lr03	2007/03/13	23.30	6.10	72.00	10.00	8.10	129.00	47.00	16.17	14.07	0.94	1.01	1.46	1.03
Big Lotus River	lr03	2007/04/04	16.10	0.90	9.00	43.00	8.10	131.00	45.00	16.95	16.60	0.69	0.19	2.17	1.59
Big Lotus River	lr03	2007/05/10	18.70	1.80	19.00	45.00	8.00	99.00	64.00	10.84	9.90	0.39	0.58	2.00	1.03
Big Lotus River	lr03	2007/06/07	15.30	4.50	45.00	14.00	8.00	121.00	39.00	12.96	6.13	0.19	5.82	0.40	0.37
Big Lotus River	lr03	2007/07/09	9.50	2.60	23.00	83.00	7.90	131.00	47.00	13.84	6.71	0.11	6.14	0.96	0.35
Big Lotus River	lr03	2007/08/27	11.10	4.10	37.00	73.00	7.80	114.00	16.00	10.29	4.05	0.06	4.48	0.84	0.45
Big Lotus River	lr03	2007/09/18	17.20	4.50	47.00	13.00	8.00	128.00	68.00	14.33	8.79	0.32	4.15	0.65	0.50
Big Lotus River	lr03	2007/10/16	20.60	3.80	42.00	18.00	8.00	132.00	35.00	15.75	12.74	0.58	2.20	1.06	0.71
Big Lotus River	lr03	2007/11/29	22.10	3.30	38.00	13.00	7.90	139.00	48.00	17.76	15.53	0.62	2.35	1.02	0.61
Big Lotus River	lr03	2007/12/12	22.30	3.80	44.00	16.00	8.00	146.00	46.00	17.39	16.28	0.82	1.04	1.43	0.61
Mean			17.82	6.22	65.40	28.74	7.62	107.95	57.35	16.18	10.55	0.44	2.11	1.15	0.74
Summer		February	22.63	5.70	66.25	16.18	7.89	125.44	61.56	20.94	16.05	0.84	0.74	1.54	1.03
winter		August	12.37	6.08	58.83	26.97	6.84	106.22	40.33	15.45	5.13	0.11	5.00	0.82	0.49

## **APPENDIX D**

APPENDIX D: RAINFALL DATA

Table D1: Rainfall data, measured in millilitres (ml), as provided by the South African Weather Bureau (Raw data as provided by the South African Weather Bureau).

Station No. Location	0004695 X				0004728 2				0004874 X				0005609 8			
	Silvermine Aug-06	Silvermine Nov-06	Silvermine Feb-07	Silvermine Apr-07	Fishhoek Aug-06	Fishhoek Nov-06	Fishhoek Feb-07	Fishhoek Apr-07	Ronde-vlei Aug-10	Ronde-vlei Nov-06	Ronde-vlei Feb-07	Ronde-vlei Apr-07	Strand Aug-06	Strand Nov-06	Strand Feb-07	Strand Apr-07
1	9.0				4.0				6.7	1.2			3.4			
2		11.5			2.0	8.3			0.9	11.8			0.8	19.8		
3	14.8	13.5			14.0	27.1			15.5	10.3			18.8	24.0		
4										4.4				0.8		
5							0.2									
6																
7	10.5								15.2				4.8			
8	7.0		9.5			8.5			0.5		13.5		5.4		19.0	
9			36.2								16.5				12.2	
10			12.6	14.5		7.5			5.5		20.0	45.0			7.4	7.0
11	32.3		8.5			32.0			31.0		4.0	10.0	13.2			1.4
12																
13	18.0				13.5				7.5				0.4			
14	14.5				4.5	0.3			10.0				16.4	0.6		
15			2.6				5.2							0.2	0.8	
16				0.3			1.0									
17																
18	10.7				8.5				7.5				6.0			
19						0.3				0.4				0.2		



## **APPENDIX E**

**APPENDIX E: DEPARTMENT OF WATER AFFAIRS – GENERAL AND SPECIAL  
AUTHORISATION**

**Table E1:** Discharge limits and conditions set out in terms of Section 39 of the National Water Act, Government Gazette No. 20526, 8 October 1999 (<sup>1</sup>National Water Act, 1998). Wastewater limit values applicable to discharge of wastewater into a water resource.

<b>SUBSTANCE/PARAMETER</b>	<b>GENERAL LIMIT</b>	<b>SPECIAL LIMIT</b>
Faecal Coliforms (per 100ml)	1000	0
Chemical Oxygen Demand (mg/l)	75	30
pH	5,5-9,5	5,5-7,5
Ammonia (ionised and unionised) as nitrogen (mg/l)	3	2
Nitrate and Nitrite as Nitrogen (mg/l)	15	1,5
Chlorine as Free Chlorine (mg.l)	0,25	0
Suspended solids (mg/l)	25	10
Electrical Conductivity (mS/m)	70 mS/m above intake to a maximum of 150 mS/m	50 mS/m above background receiving water, to a maximum of 100 mS/m.
Orthophosphates and phosphorus	10	1 (median and 2,5 (maximum))

(National Water Act, Government gazette No. 20526, 8 October 1999)

<sup>1</sup> National Water Act, 1998 (Act no. 36 of 1998), Government Gazette No. 20526, 8 October 1999.

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