Biological indicators of water quality in an urban waterway: Can diatoms reflect short term spatial and temporal changes in water quality?

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ABSTRACT

With water being such a limited resource in South Africa, numerous settlements have developed around the main sources of fresh water – rivers. Rapid urbanization and economic development have resulted in unfavourable changes in the hydrology and ecology of river systems. Rivers remain the only sources of fresh surface water and although numerous, these are often small or ephemeral, or both. Thus, there exists a need for careful management of water with respect to the availability and quality of this resource.

Rivers and streams in urban environments are often characterised by long canalised sections. One of the main reasons for this is flood prevention, especially when riverside property is at risk because existing floodplains and high flood levels were not considered. Canalisation alters the shape and dimensions of a stream's channel which in turn could have major effects on stream ecosystems, such as interruption of lateral and vertical exchange of water and nutrients.

The monitoring of water quality in South Africa has, traditionally been confined to interpretations based on the physical and chemical properties of the water. However, the collection of urban water quality data is costly, while the cumulative impacts of urbanisation on stream impacts may be reflected best by the resident biota. It is often the case that biota commonly used to indicate stream conditions are absent in canalised urban rivers. However, the group of algae known as diatoms are present in the majority of these rivers and were tested in this study for their ability to act as bio-indicators in these particular aquatic environments.

The European and other diatom indices tested in this study were able to reflect the unpredictable and extreme conditions present in the urban channels, with correlation coefficients of correlations between diatom indices and measured environmental variables often exceeding 0.60. It was also shown that diatoms could be used to indicate changes in trace metal concentrations, and that diatoms better indicated the influence of a highly contaminated tributary on the Mooi River than measurement of physio-chemical variables. It was also shown that epilithic and epiphytic substrata could be used interchangeably in urban canals. Thus this study has shown that the diatoms can be successfully applied as bio-indicators of water quality in urban environments.

OPSOMMING

Die skaarsheid van water in Suid Afrika het daartoe aanleiding gegee dat 'n groot hoeveelheid van die dorpe en stede rondom riviere ontstaan het. Die vinnige tempo waarteen stede gegroei het sowel as ekonomiese ontwikkeling, het veroorsaak dat die hidrologie en ekologie van rivier sisteme op 'n onwenslike wyse verander is. Riviere bly steeds die enigste bron van vars water in Suid Afrika en alhoewel groot in getal, is hierdie riviere dikwels klein en nie-standhoudend. Die bogenoemde noodsaak dat water hulpbronne op 'n verantwoordelike wyse bestuur moet word ten opsigte van beskikbaarheid en kwaliteit.

Riviere in stedelike omgewings word gekenmerk deur lang gedeeltes wat gekanaliseer is. Een van die hoof redes hiervoor is om vloede te voorkom, veral wanneer eiendom op die banke van die rivier in gevaar is as gevolg van die feit dat vloedvlaktes en vloedlyne nie in ag geneem is gedurende ontwikkeling nie. Die kanalisering van riviere verander die vorm en dimensies van die natuurlike stroomkanaal, wat weer lei tot impakte op ekosisteme in die riviere, soos byvoorbeeld die onderbreking van laterale en vertikale uitruiling van water en voedingstowwe.

Die monitering van die kwaliteit van water in Suid-Afrika is hoofsaaklik gebaseer op interpretasies van die fisiese en chemiese kenmerke van die water. Dit is egter duur om stedelike water kwaliteit data in te samel, terwyl biota teenwoordig in die rivier die beste beskrywing van die kumulatiewe impakte van verstedeliking op riviere kan weergee. Ongelukkig is die biota wat algemeen vir hierdie doel aangewend word dikwels afwesig in stedelike riviere en kanale. Diatome is egter teenwoordig is die meerderheid van hierdie riviere en kanale en gedurende hierdie studie is daar vasgestel tot watter mate hierdie organismes as bio-indikators in hierdie akwatiese omgewings kan op tree.

Europese en ander diatoom indekse was in staat om die onvoorspelbare en uiterste omstandighede van die stedelike kanale te weerspiëel, veral as die hoë korrelasie koëffisiënte (>0.60) vir korrelasies tussen die diatoom indekse en die gemete omgewingsveranderlikes in ag geneem word. Diatome was ook sensitief vir veranderinge spoorelemente (metale) en diatome was in staat om die invloed van 'n hoogs besoedelde sytak van die Mooi Rivier beter aan te dui as die fisiese en chemiese veranderlikes wat gemeet is. Daar is ook bevind dat epilitiese of epifitiese substrate gebruik kan word in stedelike kanale. Daar kan dus gesê word dat gedurende hierdie studie gevind is dat diatome suksesvol as bio-indikators in stedelike omgewings gebruik kan word.

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ABBREVIATIONS

APDI	Artois-Picardie Diatom Index (Prygiel *et al.*, 1996)
BDI	Biological Diatom Index (Lenoir and Coste, 1996)
CCA	Canonical Correspondence Analysis
CEC	Commission of Economical Community Index (Descy and Coste, 1991)
CEN	Comité Européen de Normalisation
DES	Descy's index (Descy, 1979)
DIN	Dissolved Inorganic Nitrogen
DO	Dissolved Oxygen
DWAF	Department of Water Affairs and Forestry
EC	Electrical Conductivity
FHI	Fish Health Index
GDI	Generic Diatom Index (Coste and Ayphassorho, 1991)
LMI	Leclercq and Maquet's Index (Leclercq and Maquet, 1987)
MR	Mooi River
NTU	Nephelometric Turbidity Units
ROTT	Rott's index (Rott, 1991)
RVI	Riparian Vegetation Index (Kemper, 2001)
SASS	South African Scoring System
SEM	Scanning Electron Microscopy
SHE	Schiefele and Schreiner's index (Schiefele and Schreiner, 1991)
SLA	Slàdeček's index (Slàdeček, 1986),
SPI	Specific Pollution sensitivity Index (Coste in CEMAGREF, 1982)
TDI	Trophic Diatom Index (Kelly and Whitton, 1995)
TDS	Total Dissolved Solids
TN	Total Nitrogen
TWQR	Target water quality range (
WAT	Watanabe index (Watanabe *et al.*,1986; Watanabe, 1990)
WRC	Water Research Commission
WS	Wasgoedspruit

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1 Introduction and Research Aims

1.1 Introduction

Southern Africa has a climate ranging from semi-arid to hyper-arid, with only a few relatively humid regions where the rainfall greatly exceeds 500 mm per year (Davies and Day, 1998). The country has an annual rainfall below the world average, comparatively high temperatures resulting in high evaporation rates, seasonal rainfall and no permanent bodies of standing fresh water (Dallas and Day, 2004; DWAF, 1996). Rivers remain the only sources of fresh surface water and although numerous, these are often small or ephemeral, or both. In addition to this, the increasing population of South Africa has resulted in increased exploitation of rivers as sources of water as well as the increased pollution of rivers. There is a need for careful management of water with respect to the availability and quality of this resource (Dallas and Day, 2004; Davies and Day, 1998). It is for this reason that Eekhout *et al.* (1997) emphasized the need for tools to aid in the management and conservation of riverine ecosystems. Section 137 of the National Water Act (Act 36 of 1998) (DWAF, 1998) calls for the establishment of national monitoring systems for water resources that provide for the collection of appropriate data and information necessary to assess, amongst others, the quality of water resources and the health of aquatic ecosystems.

With water being such a limited resource in South Africa, numerous settlements have developed around the main sources of fresh water – rivers. Rapid urbanization and economic development have resulted in unfavourable changes in the hydrology and ecology of river systems (Duong *et al.*, 2006). Rivers provide various valuable functions to humans, such as fishing, drinking water, irrigation for agriculture, industrial abstraction, protection of flora and fauna, and water sports (John, 2000). The urban population is growing at an accelerating pace and, simultaneously, sources of water supply decrease or, at the best, remain constant in quantity but decrease in quality (Niemczynowicz, 1999). In the urban context, streams or rivers are often viewed as efficient drainage networks serving to protect human life and property from flooding and are managed in terms of drainage only. This management approach often leads to modifications that occur at expense of functions normally provided by urban streams or rivers, such as the provision of aesthetically pleasing zones in a human modified environment and habitats containing a wide variety of organisms (Suren *et al.*, 1998).

Impacts on rivers are numerous, especially in the urban environment. Given time and space, though, rivers have the ability to return to the original state as long as the majority of their abiotic and biotic characteristics are not damaged. This concept is known as the resilience

of rivers (Janse van Vuuren, 1996). For example, sewage effluent will be metabolised by riverine organisms and so cause the river to return to the conditions prevailing upstream of the impact, as long as the effect did not grossly alter the conditions (Janse van Vuuren, 1996). However, the biological recovery processes are complex, and involve a multitude of physical, chemical and biotic interactions resulting in immediate, lagged and threshold responses to changing chemistry which are difficult to model (Monteith *et al.*, 2005). In addition to this it does not necessarily follow that chemical recovery will lead to the re-establishment of the biological structure prior to the impact (Monteith *et al.*, 2005).

The urbanization of rural areas is characterized by an increase in areas impermeable to water, decreased natural storage capacity of water, and canalisation of rivers into drainage canals (Dallas and Day, 2004; Niemczynowicz, 1999). The quality of fresh water in urban areas deteriorates, since runoff in these areas is likely to contain pollutants that will end up in local streams and rivers, especially after heavy rainfall (Dallas and Day, 2004). Hydraulic differences between urban and rural streams include a diminished base flow in urban streams and highly transient stormwater flows, with peaks exceeding flow rates in rural streams by an order of magnitude or more (Novotny and Witte, 1997). The reasons for this include the reduced contribution of ground water and the reduced interchange between surface and ground water (Dallas and Day, 2004) and the high runoff which is characteristic of urban environments (Suren *et al.*, 1998). The traditional management of urban streams as efficient drainage channels, affects the receiving aquatic environment by increasing the concentration of contaminants during rainfall events, in which in turn may accumulate in the sediments of the receiving environments, and by reducing the habitat quality for aquatic organisms (Suren *et al.*, 1998).

Non-point source pollution resulting from stormwater runoff has been identified as one of the major causes of the deterioration of the quality of receiving waters (Lee and Bang, 2000). Lee and Bang (2000) also argue that the runoff from a large storm event in an urban area may carry a greater pollutant load than ordinary effluent. Therefore, the characterization of stormwater runoff pollutants is necessary for the formulation of a water quality management plan for an urban stream (Lee and Bang, 2000). Novotny and Witte (1997) stated that stormwater impacts cannot only be evaluated by considering exceedance of water quality standards, but necessarily by an integrated, comprehensive determination of stormwater impacts on aquatic ecosystems. Biologically available amounts of nutrients are not necessarily reflected by the measured concentration values (Winter & Duthie, 2000). Therefore, when assessing water quality variables, one has to account for the bio-availability

of these variables in the water, which is especially true when assessing levels of metals, which might be complexed, adsorbed or precipitated (Novotny and Witte, 1997).

Runoff from urban areas is diffuse and difficult to quantify and may include pollutants that could interact to increase or decrease the ultimate impacts on aquatic ecosystems (Dallas and Day, 2004). The major pollutants in stormwater runoff are: toxic chemicals (e.g. pesticides, copper sulphate, bleaches, thinners), nutrients including nitrogen and phosphorous from fertilisers, decaying organic matter, oil and other contaminants originating from motor vehicles, particles from paved surfaces, floating litter and sediments, as well as rubbish and household effluent water such as swimming pool water (Bettink and Pearson in John, 2002; Davies and Day, 1998).

Table 1.1 Potential non-point source of pollutants contributing to urban runoff (Dallas and Day, 2004).

Potential sources	**Examples**
Road pavement materials	Degradation products
Motor vehicles	Fuel, lubricants, hydraulic fluids, coolants, linings, exhaust emissions, rust, vehicle components
Atmospheric fallout	From industrial stacks and vents
Litter	Packaging materials, food discards, animal droppings, plant debris
Vegetation	Bark, twigs, leaves, fruit, seeds, pollen, grasses
Spills	Sand, gravel, cement, agricultural and petroleum products, etc.
Domestic spraying	Herbicides, insecticides
Unauthorized dumping or washing	Chemicals such as detergents and oils

Pollutants carried downstream of the city, influence not only the quality of water in a river itself, but also lead to the pollution of flooded land areas downstream. In order to prevent downstream pollution, it is necessary to know what kind, during which activities and where the urban pollution is generated. Thus, in order to plan for pollution mitigation, water quality determinations are important components which should be included in measurement program of any urban basin (Niemczynowicz, 1999).

Vaze and Chiew (2002) noted that stormwater pollution is seen as a two-stage process, pollutant build-up and pollutant wash-off. Build-up is the accumulation of pollutants on the catchment surface during dry periods and wash-off is the removal of the pollutants by rainfall and runoff. Surface pollutant load increases with antecedent dry period (Vaze and Chiew, 2002). The canalising of natural streams results in fast runoff with high peak flows (Niemczynowicz, 1999). Lee and Bang (2000) found that the pollutant concentration peak

occurred before the flow peak and was caused by early runoff, flushing the accumulated pollutants from the streets and sewers. The "initial flush" at the start of a rainfall event transports about 40% of pollutants (Weeks, 1982 in Dallas and Day, 2004).

According to Davies and Day (1998) the most disruptive action that can be taken against a river is canalisation, as it prevents the river from functioning as a healthy ecosystem. In most cases the channel morphology is made straighter, wider and deeper, promoting drainage of low-lying areas, which in turn results in a reduction of flooding of riverfront towns, but results in a substantial loss of aquatic habitat (Allan, 1995). One of the main reasons for canalising a section of a river is flood prevention, especially when riverside property is at risk because existing floodplains and high flood levels were not taken into account (Dallas and Day, 2004).

Canalisation alters the shape and dimensions of a stream's channel which in turn could have major effects on stream ecosystems by affecting light, flow, and temperature regimes and by reducing habitat heterogeneity (Suren *et al.*, 1998). Haslam (1990), stated the following problems with regards to channel modification:

- The enlargement of the river channel provides increased flow capacity, but leads to the removal of biota from the existing channel by reconstruction work.
- Modification of the river channel shape, in profile, plan, and cross section, leads to a loss of microhabitats and associated flora and fauna, as well as reductions in overall community diversity and loss of habitat diversity. Marginal plants are also unable to establish and some macroinvertebrates are lost.
- Bank modifications lead to a loss of shading so that increased light reaching water encourages algal and macrophyte growth. There is also a loss of detrital input during leaf fall and aerial insects for fish food.

Dallas and Day (2004) cite the main problem with regards to the canalisation of rivers to be interruption of lateral and vertical exchange of water and nutrients with the riverbanks and hyporheic zone respectively. They also cite the following problems: 1) isolation of the river from ground water input, 2) a loss of natural habitat diversity, possibly affecting invertebrate density and composition as well as fish spawning and rearing, 3) the removal of protective habitat and feeding sites for adult fish and migration of fish is interfered with and 4) the removal of riparian vegetation leading to increased light penetration and a subsequent increase in water temperature.

The hyporheic zone can be defined as a porous area which connects stream water and subsurface water (ecotone) resulting in an exchange of water between two different media

(Bencala, 2000 in Kazezyilmaz-Alhan and Medina, 2006; Runkel *et al.*, 2003 in Kazezyilmaz-Alhan and Medina, 2006). The interaction between surface and subsurface water has a crucial influence on the biochemistry of stream environments (Boano *et al.*, 2006). Interactions between the surface and subsurface waters influence the downstream water quality significantly because of these exchanges between the two layers as well as biogeochemical reactions that occur between the minerals in the subsurface and those found in the stream (Kazezyilmaz-Alhan and Medina, 2006).

According to Davies and Day (1998) many of the urban rivers in South Africa are no longer functional ecosystems. Hardening of catchments has reduced the capacity of urban environments to absorb water, resulting in greatly accelerated runoff to rivers. The same river corridors have suffered so much damage through building, clearing, use as storm water drains, planting of exotic vegetation and the removal of riverine wetlands that their capacity for regulating their own flows has disappeared. In such circumstances, most urban rivers have been engineered in some way or other, to route storm waters in the most direct way out of the city.

According to Dallas and Day (2004) the term "water quality" was originally used to describe the quality of water from a human perspective, with "good"-quality water being pure and unpolluted and suitable for drinking and stock watering and safe for agricultural and industrial purposes. The monitoring of water quality in South Africa has, therefore, traditionally been confined to interpretations based on the physical and chemical properties of the water (DWA, 1986). Monitoring the water quality of rivers using physical and chemical variables is difficult due to the fact that these variables change rapidly, for example after a rainstorm, when the concentrations of nutrients and pollutants in a small river system could change with one or two orders of magnitude (Eloranta and Kwadrans, 2000). In addition, the collection of urban water quality data is costly and requires sizable investments in instrumentation, data processing and chemical analysis (Niemczynowicz, 1999). Walker and Pan (2006) stated that the cumulative impacts of urbanization on streams may be reflected best by the resident biota.

DWAF (1996) defines water quality as *"the physical, chemical, biological and aesthetic properties of water that determines its fitness for a variety of uses and for the protection of the health and integrity of aquatic ecosystems"*. The assessment of the biota found in aquatic ecosystems, also referred to as bioassessment or biomonitoring, in addition to analyses of the physical and chemical properties has numerous advantages. Biological communities reflect overall ecological quality and integrate the effects of different impacts

and also provide an ecological measurement of fluctuating environmental conditions (Iloipoulou-Georgudaki *et al.* 2003). According to Taylor *et al.* (2005) the main advantage of bioassessment is that such an approach measures the response of organisms that are continuously exposed to water and the pollutants therein and that these organisms reflect the actual effects of the pollutants on the ecosystem. John (2000) also notes that aquatic communities reflect the changes in conditions of streams very effectively. Harding *et al.* (2005) highlighted the inadequacy of conventional analytical approaches, by mentioning that the use of an integrative biological response offsets the inconsistency of rapid changes in water chemistry. Nutrient concentrations in small and moderate sized rivers show rapid and wide ranging fluctuations occasional analyses only provides a rough idea about average nutrient levels (Eloranta and Soininen, 2002). Furthermore, the overall routine monitoring of biological communities is reliable and relatively inexpensive compared to the cost of assessing toxicant pollutants (Iloipoulou-Georgudaki *et al.* 2003).

In 1994 the Department of Water Affairs and Forestry launched the South African River Health Programme (RHP) (Roux, 2004). The main purpose of this programme was that it should serve as source of information concerning the overall ecological status of rivers in South Africa. It makes use primarily of in-stream and riparian biological communities e.g. macro-invertebrates, fish and riparian vegetation for characterising the response of the aquatic environment to multiple disturbances (Roux, 2004). Indices used to assess the health of aquatic ecosystems in the River Health Programme include the South African Scoring System (SASS; Chutter, 1994), the Fish Health Index (FHI; Kleynhans, 1999) and the Riparian Vegetation Index (RVI; Kemper, 2001). SASS has become the standard for rapid bioassessment and forms the backbone of the River Health Programme (Dickens and Graham, 2002). SASS is based on the use of benthic macro-invertebrates. These organisms are recognised as valuable for bioassessments because of their visibility to the naked eye, ease of identification, rapid life cycle often based on the seasons, and their largely sedentary habits (Dickens and Graham, 2002).

Despite the advantageous qualities of these organisms for bioassessments, Round (1991) lists a few reasons why the animal components of an ecosystem may not, in all cases, provide a satisfactory index system:

- Animals have complex reproductive cycles which are often linked to season,
- Animals are largely motile and this may cause difficulties during sampling,
- Animals may have many different life stages and may undergo metamorphosis,
- Animals have specific habitats and niches,

- They are actively grazed; and closely linked to flow conditions and therefore will not usually be evenly distributed from headwaters to estuaries, and
- Water courses, which are too deep to wade across, may prove difficult if not impossible to evaluate using a macroinvertebrate index along the length of the stream.

Thus, the use of aquatic invertebrates as indicators of water quality is limited by hydrology, substrate, habitat, food availability, seasonality and distribution patchiness (Harding *et al.*, 2005). Suren *et al.* (1998) note that, although an urban stream may still support a high density of invertebrates, the presence of invertebrates will not always indicate a stream in good condition. No single group of organisms is always best suited for the detection of a variety of environmental disturbances (Kelly, 2002), and this highlights the need for additional techniques. Harding *et al.* (2005) list the criteria of such techniques as the following:

- Wide-ranging applicability across aquatic ecosystem types and their adjacent (damp) environments.
- Indicators should be retrievable under all hydrological conditions, including stagnant and dry and should lend themselves to forensic interrogation.
- The results obtained should be incontrovertibly linked to water quality.

Cox (1998) also note the standard pre-requisites of any bioassessment system to be 1) that the organisms of choice show a consistent response to environmental variables, 2) they are sufficiently numerous and widely distributed to allow their use at a variety of sites, 3) recognition of taxa is straightforward, reliable and speedy, and 4) quantification is feasible. One group of organisms fulfil these requirements – algae. Algae (including diatoms) have many characteristics that make them suitable as bio-indicators (McCormick and Cairns, 1994): 1) Algae are ubiquitous and ecologically important in the majority of aquatic ecosystems, 2) algae are sensitive to a wide range of environmental stressors, 3) algae provide information distinct from that provided by animal indicators, 4) algae respond rapidly to changes in environmental conditions, 5) using algal communities, historical benchmarks can be established for the estimation of pre-disturbance conditions, and 6) algae provide a cost-effective monitoring tool in terms of information gained per unit of effort. McCormick and Cairns (1994) then go on to recommend the development of indicators based on diatoms (Bacillariophyceae) as a standardized protocol for monitoring aquatic ecosystem change.

Diatoms are a large and diverse group of single celled, occasionally filamentous algae and are distinguished from other algal taxa by the presence of silica cell walls, called frustules. Diatoms are distributed throughout the world in nearly all types of aquatic ecosystems and are one of the most important food resources in marine and freshwater ecosystems (Rott *et*

al., 1998; Potapova and Charles, 2002). Their taxonomy and ecology are well known (Van Dam, 1974) and they exhibit a number of properties which make them useful in the assessment of water quality. These properties will be discussed in the following paragraphs, along with a brief description of diatom research that has been conducted in South Africa.

Eloranta and Kwadrans (2000) stated that diatom analyses are useful for the bioassessment of rivers for the following reasons: 1) they are less likely to be affected by weather events than physical and chemical variables, 2) sampling and laboratory preparation is cheap, 3) the unprocessed material can easily be archived for future reference, and 4) diatom communities have a relatively high diversity in all environments. Diatoms are also sensitive to many environmental variables such as light, moisture conditions, temperature, current velocity, salinity, pH, oxygen, inorganic nutrients (carbon, nitrogen, and silica), organic carbon and organic nitrogen (Van Dam *et al.*, 1994). Diatoms are also valuable as indicators in remote locations subject to pronounced change where access to chemical monitoring is limited (Jüttner et al., 1996).

In light of the above advantages of using bio-indicators to monitoring water quality and the highly suitable nature of diatoms to fulfil this function, the subsequent paragraphs contain a brief overview of the history of diatom research in South Africa, as well as a discussion of the relationships between diatoms and a variety of environmental variables and conditions, especially urban streams and waterways.

Diatoms have been the subject of study in South Africa since the mid nineteenth century Shadbolt, 1854). Some of the earlier studies were conducted on algae in general, with detailed descriptions of some taxa, most notably by Fritsch (1918) and Rich (1932). The contributions to the study of Southern African diatoms by Dr. B.J. Cholnoky were the most significant to date. In a period of 20 years, from 1952 to 1972, Cholnoky conducted research in the fields of diatom taxonomy and autecolgy of species and published over 40 diatom related papers on the southern African region. This included an in depth evaluation of knowledge at the time of the relationship between diatoms and the chemical and physical properties of water bodies and contains numerous contributions to the knowledge of diatom flora of Southern Africa in the form of species descriptions (e.g. Cholnoky, 1958; Cholnoky, 1960). After this period the main body of research on freshwater diatoms was conducted by Dr. R.E.M. Archibald and Dr. F.R. Schoeman, both trained by Cholnoky. Their research was conducted in the area of diatoms as indicators of water quality (e.g. Archibald, 1972; Schoeman, 1976), as well as taxonomy, with their best known contribution being "The diatom Flora of Southern Africa" (Schoeman and Archibald, 1976-1981). In a period from the 1960's

to the early 1980's, Dr. M.H. Giffen was mainly involved in the study of marine diatoms and described many new taxa from South African coastal waters. During this period Schoeman applied the saprobic classification system adapted for diatoms by Lange-Bertalot (1979) to the upper reaches of the Hennops River, with good results, showing the cross continental use of diatom-based indicator systems of water quality to be feasible (Schoeman, 1979). Following this period, little was published on diatoms during the 1980's and 1990's. Diatom research was however, rekindled by efforts of Prof. G.C. Bate (e.g. Van der Molen *et al.*, 1998; Bate *et al.*, 2002) in the late 1990's and early 2000's at the then University of Port Elizabeth now known as the Nelson Mandela Metropolitan University.

Contributions to freshwater diatom research in South Africa in recent years have been made by Dr. J.C. Taylor and Dr. W.R. Harding. Taylor successfully tested numerical diatom indices frequently used in Europe and Asia in the Vaal River in South Africa. This lead the way for the development of standardised methods for sampling, preparation and identification of diatoms as part of the development of Diatom Assessment Protocol by Harding, Taylor and Archibald (Water Research Commission project Project K5/1588). Since 2004, several papers have been published on diatoms and their use as indicators (Archibald and Taylor, 2004; De la Rey *et al.*, 2004; Harding *et al.*, 2005; Taylor *et al.*, 2005a; Taylor *et al.*, 2005b; Taylor and Lange-Bertalot, 2006; Taylor *et al.*, 2007).

As bio-indicators, diatoms have been used to reflect a broad spectrum of environmental variables. Unpolluted waters are usually characterised by a large amount of species each having few individuals (high diversity). With a sudden increase in nutrients, certain species take advantage of their inherent capacity for population increase, the result being a reduced diversity with one or few species dominating (Bahls, 1973). During a study on the diversity of diatom associations and their relation to water quality, Archibald (1972) found that the diversity of the associations did not always reflect water quality and highlighted the danger of using diversity of the community alone as a measure or indication of pollution. Later, De la Rey et al. (2008) found that diatom based aut-ecological indices displayed a significantly better relationship with measured water quality variables than diversity indices and concluded that diatom based aut-ecological indices were more useful in biomonitoring programs of rivers and streams than diversity indices. Evans and Marcan (1976) found that the number of diatom species remained reasonably constant after exposing diatom communities to varying levels of pollution and that variation of the concentration of effluent did not result in a gradual sequence of changes in diatom community composition. Having said this, though, Evans and Marcan (1976) still note that species numbers are generally lower in polluted waters. Vilbaste and Truu (2003) found that in the lowland streams of

Estonia, benthic diatom assemblages were characterised by a small number of species occurring frequently with high abundance and by a large number of occasional taxa represented only by a few specimens.

During the development of their Trophic Diatom Index (TDI), Kelly and Whitton (1995) distinguished between nutrient rich waters and waters containing elevated levels of organic pollution. In the above mentioned studies by Evans and Marcan (1976) and Archibald (1972), the waters in which the studies were conducted contained organic pollution in conjunction with high nutrient levels. Kelly and Whitton (1995) found that high nutrient concentrations were often associated with diverse diatom and macrophyte floras, but that sites dominated by diatoms indicative of organic pollution had much lower species diversity, thereby highlighting the importance of distinguishing between the types of pollution in water bodies, before conclusions can be made concerning diatom diversity.

As a group, diatoms are highly suited for distinguishing between the different types of pollution. As discussed earlier, Eloranta and Soininen (2002) noted that measured nutrient values indicate nutrient concentrations at that particular moment whereas development of the community may have taken a few weeks, a period during which there probably was a fluctuation in nutrient concentrations over one or two orders of magnitude. They further stated that a consistent high number of diatom taxa at a site, provides a stable base for evaluating the trophic status of the water despite all disturbing factors. The trophic level of water, often measured by the concentration of nitrate and phosphate, in conjunction with inorganic carbon and silica, play an important role in governing the structure of benthic diatom assemblages (Vilbaste and Truu, 2003; Van Dam *et al.*, 1994) and changes in diatom assemblages will be valuable as indicators these variables (Winter and Duthie, 2000). Inorganic nutrient supplies are important driving variables for primary production and potentially act in association with disturbance to set the overall habitat template for periphyton growth dynamics in streams (Biggs, 1995).

Eloranta and Soininen (2002) found that conductivity and pH lead to the significant changes in natural communities in Finnish rivers. pH is one of the most important water quality parameters, because of its profound influence on water chemistry, as well as the organisms found in the aquatic environment (Dallas and Day, 2004). Substances dissolved in the water each have their own dissociation constants and pH determines the forms of electrolytes that are found in the water, by influencing the equilibrium of those substances. It, therefore, determines the availability and toxicity of, for instance, metals in water (Dallas and Day, 2004).

Trace metals and their influence on diatom communities have been the focus of several studies (e.g. Rachlin *et al.*, 1983; Morin *et al.*, 2007). Gold *et al.* (2002) and Ivorra *et al.* (2002) showed that diatom communities reacted to changing metal (Cd and Zn) concentrations and that the taxonomic composition of a diatom community could be used as an indicator oF metal pollution. This could be explained in light of the fact that diatoms have relatively high concentration factors for Cd, Hg and Pb, indicating they were good accumulators for these metals (Tien, 2004).

Lange-Bertalot (1979) stated that diatoms are outstanding bio-indicators for different degrees of pollution, if the species-specific tolerance limits are known. Sládeček (1986) stated, however, that the tolerance of diatoms to environmental variables in different catchments may differ. In a recent paper by Dela-Cruz *et al.* (2006) these authors questioned the applicability of diatom indices in the southern hemisphere on the grounds that the ecological tolerance data for most of the commonly used indices had been derived from studies done in the northern hemisphere. However, Dela-Cruz *et al.* (2006) stated that while diatom assemblages are likely to be influenced by local or regional conditions, ecological tolerance data provided by northern hemisphere studies could be applied to southern hemisphere situations, based on the following findings: 1) the presence of a large amount of globally distributed diatom species in their Australian samples, 2) the similarity in the predictions of proportions of pollutant tolerant and intolerant taxa in European and Australian studies based on ecological tolerance data for both continents, and 3) the spatial patterns inferred from metrics based on ecological tolerances were largely persistent over time. The applicability of northern hemisphere ecological tolerance data to diatoms occurring in the southern hemisphere is determined by their "sub-cosmopolitan" distribution (Padisák in Kelly, 1998). "Sub-cosmopolitan" distribution refers to the fact that diatoms occur everywhere a certain set of environmental variables exists and this has been illustrated in South Africa with the recent contribution of Taylor (2004) where European and other indices were used to indicate water quality in the Vaal River with great success.

Wendker (1992) concluded that different diatom communities developed under different current velocities and stressed the importance of this fact for floristic studies as well as the assessment of water quality. In their investigation into the composition of benthic algal mats, Reiter and Carlson (1986) found that benthic algae appeared to be sensitive to water velocity, but that the inevitable alteration of the surroundings of the algae by their growth nullified this sensitivity when current velocity was used to explain the variety of algae found in different parts of the stream. The findings of Biggs (1995) suggested that: 1) the

climate/flood disturbance regime is the primary axis of the habitat template for periphyton, and 2) nutrient resources operate within this by controlling the rate of biomass accrual during periods of stable flow. Biggs (1995) also suggested that the flow variability regime strongly influences nutrient supply to a stream. Townsend and Gell (2005) note that losses of diatoms as a result of sloughing may vary according to species and the position of the algal in a river and its exposure to currents likely to cause sloughing. Although flow rate plays an important part in determining the composition of the diatom community it was not deemed meaningful to measure flow rate in the present study due to the sporadic and unpredictable nature of the flow in the studied stream.

According to Tison *et al.* (2005), geology and relief seem to play an important role in structuring the natural variability of diatom communities. Tison *et al.* (2005) further stated, however, that the influence of natural conditions on diatom communities is overwhelmed by the nature and intensity of any pollution affecting a particular site. Diatom community composition varies between sites due to natural and anthropogenic factors. By first accounting for the natural variability among sites, the accuracy of anthropogenic impact assessment can be increased (Tison *et al.*, 2005). Vilbaste (2004) stated that periphytic diatom community composition and structure are formed over time as a result of biological response to water quality. Maier & Rott (1988) found that the most significant changes in diatom community composition in two small streams were due to heavy domestic waste water loading. The species tolerant of waste water influence (e.g. *Craticula accomoda* (Hustedt) Mann) and *Eolimna minima* (Grunow) Lange-Bertalot) and resistant to waste water influence (e.g. *Mayamaea atomus* (Kützing) Lange-Bertalot) were favoured.

According to Jüttner *et al.* (1996) the decision to sample one or more habitats for diatoms depends on the aims of the sampling program – if the aim is to do a survey of the biodiversity, their results showed that sampling from one habitat alone would only represent a percentage of the diatom species present at a location; whilst if the aim is to detect pollution, then sampling from one habitat may be sufficient, but the authors stress that this habitat should be chosen carefully. Winter and Duthie (2000b) found that there was no noticeable advantage in the sampling of discrete habitats for water quality monitoring when using diatoms with regards to their relationship with water quality. Jüttner *et al.* (1996) concluded that the consistent community change across a range of habitats in the Nepalese Hills suggested that aquatic chemistry was the most important environmental influence on diatoms that occurred in those streams. Jüttner *et al.* (1996) also found no differences in species richness, diversity or evenness between different habitats in the Nepalese Hills.

In a study of the role of substrate type on benthic diatom assemblages in the wet/dry tropics of Australia by Townsend and Gell (2005), there was no clear evidence of substrate specific diatoms, in rivers that had been relatively undisturbed by human activity, included riffle or river run habitats and had low benthic sample depths (<50cm), the only exception being *Psammothidium*, with only the abundance of common species differing between some substrata. Rott *et al.* (1998) found that results obtained from different substrata (grass and stone samples) were very similar and concluded that water quality determination using diatoms is independent of the substratum used.

Kelly and Whitton (1995) concluded that urban runoff and storm sewer overflows affect water quality independent of the other point sources of pollution in a city, such as sewerage treatment works. Therefore, management of water in urban environments requires special water resource management. Institutional action should integrate, on one hand, water resource management, and on the other, urban environmental sanitation (Pompêo, 1999). The focus of this study, therefore, is directed at the application of known techniques to better quantify the impact of these above mentioned effects on the quality of the water in the rivers found in urban environments.

Diatom assemblages have been shown to be good indicators of urban stream conditions (Walker and Pan, 2006). Round's (2001) statement that diatoms can colonise massive rivers, but also "rivers" milimeters deep and centimetres wide, has particular relevance in urban areas where rivers are often canalised, resulting in wide stream channels and shallow stream depths as described above. Unpredictable and unstable environments in rivers (such as those encountered in urban environments) are usually dominated by one or a few common species and/or cosmopolitan diatom species and a smaller number of rarer species (Duong *et al.*, 2006; Kelly, 2002). Duong *et al.* (2006) found the common cosmopolitan species *Nitzschia umbonata* (Ehrenberg) Lange-Bertalot, *Nitzschia palea* (Kützing) W. Smith and *Eolimna minima* (Grunow) Lange-Bertalot to be dominant at a site severely polluted by urban pollution from the Hanoi area in Vietnam.

The ability of diatoms to better reflect the water quality of an urban river was shown by Duong *et al.* (2006). In their study diatom index scores indicated poorer than expected water quality when compared to measured physio-chemical variables. This was ascribed to the fluctuations of the water quality variables, a sample of which can only provide an instantaneous indication of the quality of the waterbody.

Walker and Pan (2006) in their study of urban streams showed that changes in the diatom community composition showed a correlation with conductivity along an urban to rural gradient (Walker and Pan, 2006). In this regard, Newall and Walsh (2005) found strong correlations between diatom indices and electrical conductivity (EC), suggesting changes in response to general water quality, as opposed to changes in response to organic enrichment. Walker and Pan (2006) also concluded that despite the high seasonal variation, diatom communities in urban and rural sites were distinct, despite the similarity of these sites.

During the development of the Diatom Prediction & Classification System for Urban Streams, John (2000) found that diatoms successfully reflected water quality. This study also found the classification of sites based on diatom taxa was confirmed, with the author successfully predicting the diatom species occurring at sites by environmental data. John (2000) concluded that diatoms were able to clearly indicate the status of stream conditions in urban environments. Alkalinity, ground water salinity, catchment land use, riparian damage and colour of water were found to be highly correlated with impacted sites as indicated by diatom distribution.

In a study of the response of epilithic diatom assemblages to urbanization influences, Newall and Walsh (2005) found that the majority of the species that were positively associated with the urbanization gradient were indicative of eutrophication according to the classification of Van Dam (1994). Potapova *et al.* (2005) found that the taxonomic composition of stream algal communities changed along the urban gradient where water chemistry was related to urban intensity. Newall and Walsh (2005) found strong negative correlations between urbanization and European-derived diatom indices and concluded that the indices may be transportable beyond the continent in which it was first derived. They also found significant correlations between connection (defined as the proportion of impervious area directly connected to streams by stormwater pipe) and all the diatom indices used. They further stated that this could be ascribed to the fact that drainage connection may be impacting diatom communities by increased delivery of nutrients during small storm events, which would in turn increase the trophic diatom index scores. Thus there is evidence that European-derived diatom indices may be transportable as indicators of general water quality in urban areas, but also as indicators of the trophic enrichment (Newall and Walsh, 2005). It has also been reported from a study of urban influences on diatom communities in Vietnam, that the majority of the diatoms encountered in these streams, were cosmopolitan (Duong et al., 2006), further strengthening the possible local use of European and other indices to indicate changes in urban water quality.

1.2 Aims

In light of the discussion above, the objectives of this study were to:

1. Measure the impact of the extreme, unpredictable conditions existing in an urban water way, such as the Wasgoedspruit, on diatom community composition,
2. Evaluate the efficacy with which European and other diatom-based indices reflect changes in water quality over the short term in an urban water way, and
3. Evaluate the extent with which changes in water quality in the Wasgoedspruit influence the water quality and diatom community composition of the receiving Mooi River.

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2 MATERIALS AND METHODS

2.1 Study area

The Wasgoedspruit (literal translation "laundry stream") is an urban waterway formed by the confluence of two smaller streams to the west of Potchefstroom. Approximately 4.5 km in length, the Wasgoedspruit flows past the industrial area of Potchefstroom in a natural stream-bed, after which it is canalized to guide the stream into the Mooi River. The canalized section of the Wasgoedspruit was the main focus of this study.

Potchefstroom is located approximately 1350m above sea level and is situated in the eastern half of South Africa which receives predominantly summer rainfall. Potchefstroom has an average rainfall of between 500-700 mm per year (Hoffman and Ashwell, 2001) with thunderstorms a common occurrence in summer. These thunderstorms lead to high urban runoff and a rapid increase in the flow rate of the rivers receiving the storm water.

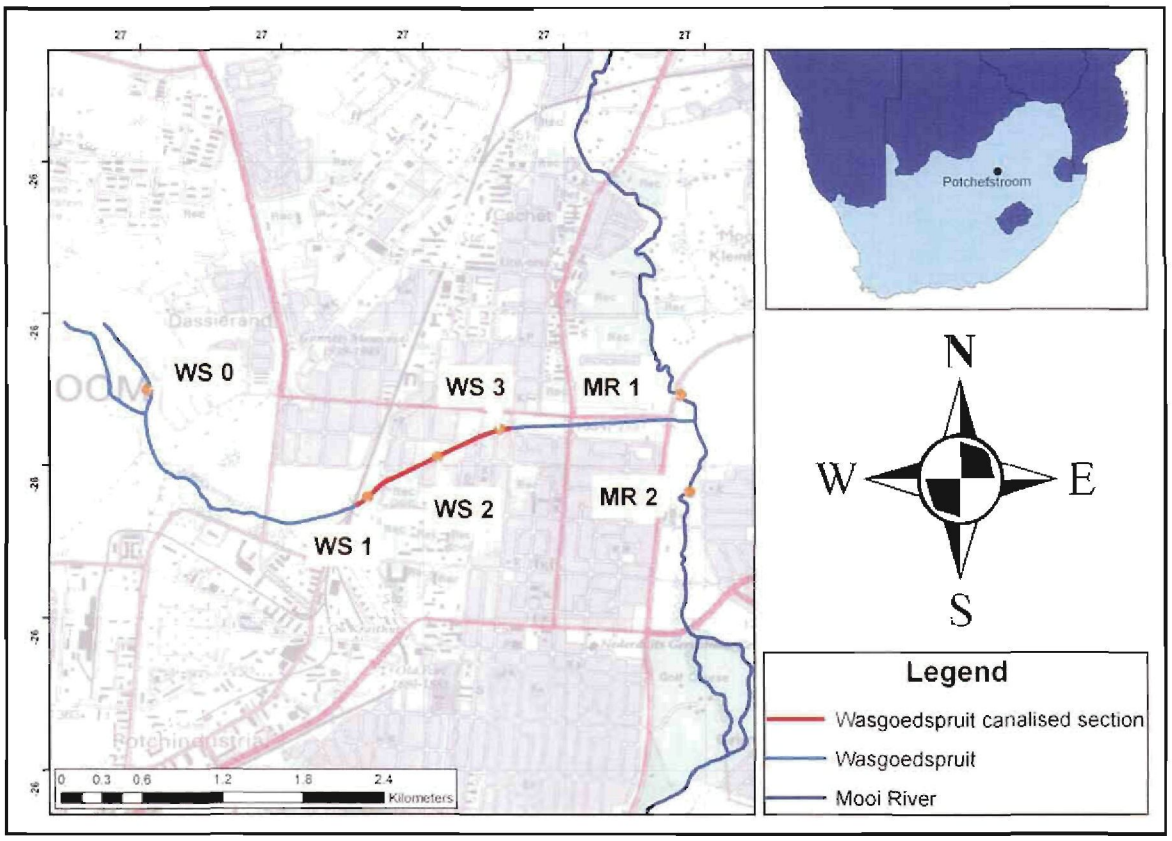

Figure 2.1 Map (1:50 000 Topographical map) showing the location of sites sampled in the Wasgoedspruit and Mooi River

Due to the seasonal summer rainfall of Potchefstroom, the upper reaches of one of the two small streams feeding the Wasgoedspruit are dry for most of the spring period from late

August to late November, following the dry winter (personal observation). During this period the Wasgoedspruit is reduced to a mere trickle, at times covering less than half of the canal floor as it meanders down the concrete section.

2.2 Site description

Below, Table 2.1 provides the site codes, names and exact location of each site. The table is followed by a general description of the site specific characteristics.

Table 2.1 Site codes, names and coordinates

Code	Site	River/stream	Coordinates
WS 0	Origin Dassierand	Wasgoedspruit	26°42'00.2"S; 27°04'07"E
WS 1	Below Railway Bridge	Wasgoedspruit	26°42'26.6"S; 27°05'03.7"E
WS 2	Adjacent Wallis Street	Wasgoedspruit	26°42'12"S; 27°05'22"E
WS 3	Van Riebeeck Street bridge	Wasgoedspruit	26°42'05"S; 27°05'38.5"E
MR 1	Below Mooi River Drive	Mooi River	26°41'57.2"S; 27°06'19.4"E
MR 2	Upstream Retief Street bridge	Mooi River	26°42'23.7"S; 27°06'21.8"E

2.2.1 WS 0 – Wasgoedspruit at Approximate Origin

This site can be described as being the upper-most reach of the Wasgoedspruit with flowing water, sometimes very slow. As stated above, this section of the Wasgoedspruit was dry for most of the spring period from late August to late November 2005.

Figure 2.2 Wasgoedspruit at approximate origin in the Dassierand area of Potchefstroom.

At this site the stream has a natural river bed and steep river banks. As can be seen in Figure 2.1, this site is located upstream of the industrial area (Potchindustria) located to the south of the Wasgoedspruit. Where possible, diatom samples were collected from rocks,

however, the majority of the diatom samples collected at this site were from the submerged parts of plants.

2.2.2 WS 1 – Below Railway Bridge

This site is the first of the three sites located in the canalised section of the Wasgoedspruit and is located approximately 100m downstream from the start of the Wasgoedspruit canal.

Figure 2.3 Wasgoedspruit below the railway bridge in the Dassierand area in Potchefstroom.

This site was frequently characterised by patches of the macro-alga *Cladophora* sp. growing attached to the bottom of the canal surface and covered with diatom material growing epiphytically. A stream depth of greater than 10 cm was rare at this site. The majority of the samples taken at this site were collected directly form the concrete canal surface, since there was a presence of a well developed bio-film growing on the canal surface, dominated by diatoms.

2.2.3 WS 2 – Adjacent to Wallis Street

This site is the second of the three sites located in the canalised section of the Wasgoedspruit and is located in the middle of the Wasgoedspruit canal. The dumping of household rubbish was a common occurrence at this site and the Wasgoedspruit was also crossed by pedestrians, at or downstream of this site because, of its central location in Potchefstroom and the shallow depth at this site. Similar to the conditions at WS 1, the site had a low stream depth and abundant diatom community on the canal surface, from which most of the diatom samples were collected. *Cladophora* sp. growing attached to the canal surface was not observed at this site.

Figure 2.4 Wasgoedspruit adjacent to Wallis Street in the Central area in Potchefstroom.

2.2.4 WS 3 – Upstream Van Riebeeck Street Bridge

This site is the last of the three sites located in the canalised section of the Wasgoedspruit and is located approximately 1.2 km upstream of the confluence with the Mooi River. At this site the Wasgoedspruit formed pools as a result of the transition between the canalised section and the un-canalised (but still artificial) section before the confluence with the Mooi River. This in turn lead to the formation of sand banks and plant communities dominated by *Persicaria* spp., as well as a variety of grasses and other weeds.

Figure 2.5 Wasgoedspruit upstream of the Van Riebeeck Street bridge in the Central Area in Potchefstroom.

Since the grass leaves and *Persicaria* spp. stems were permanently submerged in the shallow stream, diatoms samples were collected from the canal surface as well as from the submerged plant material in order to investigate whether these substrata could be used interchangeably in the this environment.

2.2.5 MR 1 – Below Mooi River Drive

This site is located in the Mooi River approximately 200 m upstream of the confluence with the Wasgoedspruit. The river bed at this site has been stabilised with the aid of rock gabions. These gabions had often deteriorated and where wiring was not present, diatom samples were collected from the rocks. Diatom samples were also collected from rocks and stones up- and downstream of the gabions.

Figure 2.6 The Mooi River below the Mooi Rivier Drive Bridge and upstream of the confluence with the Wasgoedspruit in Potchefstroom.

2.2.6 MR 2 – Upstream of Retief Street Bridge

This site is located in the Mooi River approximately 600 m downstream of the confluence with the Wasgoedspruit. The site is characterised by the presence of a rocky natural stream bed and is shaded on the western side of the stream by the hanging branches of the Willow trees characteristic of the Mooi River as it flows through Potchefstroom. Diatom samples were collected from rocks in this section of the Wasgoedspruit.

Figure 2.7 The Mooi River upstream of the Retief Street Bridge and downstream of the confluence with the Wasgoedspruit in Potchefstroom.

2.3 Sampling

2.3.1 Biological sampling

Fortnightly diatom samples were collected for the period from February 2005 to January 2006. Sampling was conducted in two phases to achieve the objectives as stated in Chapter 1.

The first phase started in February 2005 and was concluded in August 2005, with the main objective being to measure the effect of selected environmental variables on diatom community composition and to evaluate the efficacy with which diatom-based indices reflect changes in water quality over the short term. Diatom samples were collected from 4 sites in the Wasgoedspruit, WS 0, WS 1, WS 2 and WS 3 (Figure 2.1 and Table 2.1). The first site, WS 0, is located in the un-canalised section of the Wasgoedspruit upstream of the canalized section and its confluence with the Spitskopspruit. This site was included to serve as a reference site to compare the conditions prevailing in the stream before the urban impact. The remaining three sites are situated in the canalized concrete section.

The second phase started in September 2005 and ended in January 2006. During this phase the objective was to evaluate the extent with which changes in water quality in the Wasgoedspruit influence the water quality and diatom community composition of the receiving Mooi River. Diatom samples were collected from two sites in the Wasgoedspruit, WS 0 and WS 3, and two sites in the Mooi River, MR 1 and MR 2 (Figure 2.1 and Table 2.1),

which are up- and downstream of the confluence of the Wasgoedspruit and Mooi River respectively.

According to Kelly *et al.* (1998), Prygiel *et al.* (2002) and Taylor *et al.* (2005), the substratum of choice for monitoring water quality is rocks and other hard surfaces. Reasons for this include the widespread availability of this substratum, the ease of collection of diatom material from this substratum (collectively referred to as the epilithon) and the fact that the performance of major diatom-based indices using this substratum are well understood (Kelly *et al.*, 1998). Kelly *et al.* (1998) and Taylor *et al.* (2005) recommend that at least five cobbles or stones be sampled within a 10 metre stretch of the stream or river at a site. Diatoms were removed by vigorously scrubbing of the upper surface (surface continually exposed to the water in the stream) of the substratum with a small brush (toothbrush) to dislodge the diatom community into a plastic sampling tray. The diatom suspension was mixed, then poured into a marked plastic sample bottle and stored in a cool, dark place.

Three of the sites in this study were, however, located within the concrete section of the Wasgoedspruit, which provided ample hard surface for the epilithon to develop, resulting in a well established, diatom dominated, biofilm during most parts of the year. Due to this abundance of diatom material on the concrete canal surface, and the relative lack of cobbles or stones in the canal, diatom samples were collected by diverting the flow of water away from the sampling area and scraping the epilithon from the canal floor with a knife blade.

In the absence of epilithon at a site, alternative substrata should be used for the collection of diatom material (Taylor *et al.*, 2005). This was the case for WS 0 for most of the sampling period, where plant material in the form of grasses and small emergent macrophytes, were found to be suitable alternatives. At WS 3 diatom samples were collected from the canal surface and from plant material, mainly in the form of grasses hanging in the water. This was done to compare the response of diatom communities on different substrata within the canal. At least five plant samples, representative of the particular stretch of the stream, were collected and transferred into a plastic bag with some of the stream water, and shaken in order to dislodge the diatom communities from the plant substratum. The plant material was then carefully removed from the plastic bag and the diatom suspension was poured into a marked plastic sample bottle and stored in a cool dark place.

Samples that were not processed within a few hours after collection were preserved by the addition of ethanol to reach a final concentration of 20% by volume. Sub-samples of the preserved diatom material were archived for future reference.

2.3.2 Sampling for environmental variables

The measurement of environmental variables was done weekly during the same period as stated for the diatom sampling. Environmental variables measured in situ included: temperature, dissolved oxygen concentration, electrical conductivity, turbidity and pH. Temperature and dissolved oxygen were measured with a Yellow Springs YSI Model 54A Oxygen/temperature meter, electrical conductivity with a HI 9033 Multi Range conductivity meter (Hanna Instruments), turbidity with a Hach Company Model 2100P portable turbidity meter and pH with a Wissenchaftlich-Technische Werkstaten WTW Model pH 330/SET-1digital pH meter. Water samples were also collected for laboratory analysis performed by Eco-Analytica. Eco-Analytica is a water quality laboratory using IC and ICP-MS analyses to determine the concentrations of the chemical constituents present in water samples. Analyses were done for the following water quality variables: phosphorus, ammonium, nitrate, sulphate, chorine and fluorine, as well as selected trace metals, including aluminium (Al), arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), manganese (Mn), mercury (Hg), selenium (Se) and zinc (Zn). These trace metals were measured because of the existence of established ecotoxicological values used in the development of the South African Water Quality Guidelines for the protection of natural ecosystems (DWAF, 1996).

It was decided that the results obtained from this laboratory for phosphorus would not be included in the results and the discussion of the environmental variables (Chapter 3), nor the results and discussion of the diatom indices and their relationship to environmental variables (Chapter 4). The reason for this was that the results obtained form this laboratory for phosphorus were deemed to be inaccurate following later comparisons with results from another laboratory. In addition, the results contained a large amount of values that were stated to be below the detection limit, set at 0.005 $mg.l^{-1}$.

2.4 Processing of diatom samples

2.4.1 Removal of organic matter

For accurate identification of the diatom species, the organic matter first has to be removed, by acid oxidation that leaves only the silica diatom frustules. The method used in this study is based on the potassium permanganate and hot hydrochloric acid method described by Taylor *et al.* (2005), which in turn is loosely based on the method described by Hasle (1978). This method has been shown to yield good results for samples collected in South Africa (Taylor, 2004). A variation of the above stated method was used. Instead of using small beakers as stated in the protocol, test tubes were substituted. In doing so a smaller amount of sample is used, requiring the use of less acid. A brief description of the method described

in Taylor *et al.* (2005) is given, followed by alterations made in the volumes of sample and reagent used when using test tubes.

$KMnO_4$ and hot HCl method (Taylor *et al.*, 2005):

1. Let the sample settle allowing at least one hour for each centimetre of water depth (Gell *et al.*, 1999). Decant the supernatant.
2. Homogenize the sample and transfer ca. 20 ml suspension (depending on the density of the suspension) to a small beaker.
3. Depending on the density of the suspension, add 5-10 ml of a saturated potassium permanganate ($KMnO_4$) solution and leave to stand for 24 hours to allow oxidation of the organic material.
4. Add approximately 10 ml hydrochloric acid (HCl) and heat on a hot plate at 90 °C for 1-3 hours or until the sample turns a light yellow colour.
5. Add a few drops of hydrogen peroxide (H_2O_2) to test if oxidation is complete, in which case the hydrogen peroxide will not cause lasting foaming.
6. When oxidation is complete, rinse the samples by centrifugation with distilled water at 2500 rpm for 10 min. Decant the supernatant and rinse a further three times. Care should be taken not to loose any diatom material.
7. Store the cleaned material in small glass vials and in a high concentration of ethanol to prevent the dissolution of the frustules and microbial growth respectively.

Test tube method:

2. Homogenize the sample and transfer ca. 2-4 ml suspension (depending on the density of the suspension) to a glass test tube (10 ml).
3. Depending on the density of the suspension, add 2-3 ml of a saturated potassium permanganate ($KMnO_4$) solution and leave to stand for 24 hours to allow oxidation of the organic material. Due the size and shape of test tubes, it is important to occasionally homogenize this suspension by shaking or using a vortex mixer in order to ensure sufficient contact with the oxidant and so allowing for the complete oxidation of organic material.
4. Add approximately 2 ml hydrochloric acid (HCl) and heat in a water bath at 90 °C for 1-3 hours or until the sample turns a light yellow colour.

It is important to perform all steps of the above stated methods, in a fume cupboard, due to the release of hazardous gasses. Consultation of the relevant material safety data sheets is a necessity. The use of safety glasses, acid resistant lab coats and acid and heat resistant

gloves are strongly recommended and were used during the processing of the diatom samples in this study.

2.4.2 Diatom slide preparation

Due to the siliceous nature of the diatom frustule, it has a refractive index similar to that of glass. For this reason, diatoms have to be mounted in a high refractive index medium, if examination of the morphological characteristics of the frustules is to be done, in order to accurately identify diatoms to species and variation level. Diatoms were mounted in Pleurax (refractive index 1.73; Hanna, 1949), produced by Dr. J.C. Taylor.

A small amount of the cleaned or processed diatom material was diluted until the suspension was only slightly cloudy to the naked eye (Welsh in Taylor, 2005). A drop of 10 % NH_4Cl was added for every 10 ml of suspension in order to neutralize charges on the suspended particles and ensure the even distribution of the material (McBride, 1988). Approximately 1.5 ml of this suspension was placed on a cleaned cover slip and left to dry at room temperature. The cover slips were then placed on a hot plate at 350°C to drive off excess moisture and to sublimate the NH_4Cl residue (Taylor *et al.*, 2005) and allowed to cool. Two drops of the mounting medium (Pleurax) were placed on the cover slip, after which a cleaned glass slide was lowered onto the cover slip, inverted and heated until the media 'boiled' and the mounting media solvent evaporated (Welsh in Taylor, 2005). The slide was allowed to cool and labelled with the following information: date of collection, site name, co-ordinates, habitat, collector and type of mounting media.

2.4.3 Slide and sample archiving

The original sample was retained throughout the preparation process until the final slide had been prepared and examined under the microscope. The density of diatoms on the slide should allow the dominant taxa to be easily identified and enumerated, which is usually in the order of 10–20 valves per field of view (Kelly et al., 1998) at 1000x magnification. Following successful examination a portion of the cleaned material was stored in a labelled glass bottle, with ethanol added to a high as possible concentration, to prevent the growth of micro-organisms. Archival of diatom samples is necessary for the future preparation of slides, verification by other workers and for scanning electron microscope (SEM) studies (Taylor *et al.*, 2005)

2.5 Identification, enumeration, diatom indices and statistical analyses

2.5.1 Identification

Prepared slides were examined using a Zeiss light microscope equipped with phase contrast optics. Photo-micrographs of diatom species were taken with a Nikon DS-U2 Camera, for

verification of identifications and for the production of plates (see Appendix 2). 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 primarily using the works of Prygiel and Coste (2000), Krammer and Lange-Bertalot (1986-1991) and Taylor *et al.* (2007). Other works consulted included Barber and Haworth (1981), Lange-Bertalot and Genkal (1999), Lange-Bertalot (1999) and Round *et al.* (1990).

2.5.2 Counting

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 *et al.*, 2002). No distinction was made between a diatom frustule and valve, because of the difficulty in distinguishing between intact frustules and isolated valves of the frequently encountered small achnanthoid and naviculoid species. The effect that this convention has on the final results has not been formally evaluated, but it is believed to be small (CEN, 2001). Using a 100x objective, all the diatoms encountered within the field of view were identified and enumerated, before moving along a horizontal or vertical traverse to the next field (CEN, 2001). Diatoms appearing in girdle view were identified as accurately possible using distinguishing characteristics visible even in girdle view. Broken valves or frustules were included in the counts if more than 50 % of the valve/frustule was present. The same rule was used for valves/frustules only partially within the field of view. It is important to keep in mind that the precise form of the rule is less important than consistency in the application of the rule during the analyses of samples (CEN, 2001).

2.5.3 Diatom indices

The indices used include Descy's index or DES (Descy, 1979), the Generic Diatom Index or GDI (Coste and Ayphassorho, 1991), Leclercq and Maquet's index or LMI (Leclercq and Maquet, 1987), Sladecek's index (Sladecek, 1986), the Specific Pollution sensitivity Index or SPI (Coste in Cemagref, 1982), the Biological Diatom Index or BDI (Lenoir and Coste, 1996) the Artois-Picardie Diatom Index or APDI (Prygiel *et al.*, 1996), the Commission of Economical Community index or CEC (Descy and Coste, 1991), Schiefele and Schreiner's index or SHE (Schiefele and Schreiner, 1991), the Trophic Diatom Index or TDI (Kelly and Whitton, 1995), Watanabe index or WAT (Watanabe *et al.*, 1986) and Rott's index or ROTT (Rott,1991).

2.5.4 Data analysis

The environmental variable data in this study showed a non-normal distribution (i.e. did not follow a bell-shape distribution; Statsoft, 2005). For this reason non-parametric statistical

methods were employed to represent the general water quality conditions. The median, 25%-75% quartiles and range data was calculated for each of the environmental variables (Chapter 3) and diatom indices (Chapter 4) and presented graphically.

In order to assess the degree with which the different indices used in this study responded to changes in the environmental variable measured, correlation analyses were conducted. Since the data showed a skewed distribution, all environmental variables, except pH, were log_{10} transformed. Standard two-way, correlation analyses were used with a Pearson's correlation coefficient of $P<0.05$ (Snow and Adams, 2006) for the determination of significant correlations. Cluster analysis was also used to determine the degree of similarity between sites as depicted by the diatom index scores. The cluster analysis was performed using Ward's method and Euclidian distances.

An ordination technique was used in this study to determine the response of diatom species encountered in the Wasgoedspruit and Mooi River to general water quality, as changes in the diatom community may lead to changes in diatom index scores. Canonical correspondence analysis (CCA) is a multivariate direct gradient analysis technique, whereby a set of species is related directly to a set of environmental variables. This technique detects the patterns of variation in community composition that can be explained best by the environmental variables (Ter Braak, 1986). CCA is similar to other ordination techniques, in that it provides an integrated description of species-environment relationships by assuming a response model that is common to all species, and the existence of a single set of underlying environmental gradients to which all the species respond (Ter Braak, 1986). Canonical correspondence analysis has an advantage over other ordination techniques because it focuses on the relationships between species and environmental variables (Ter Braak, 1986). Prior to entering the data in Canoco, the data was log_{10} transformed as for the correspondence analysis above and unrestricted permutations were used in the CCA.

2.5.5 Software used

The following software applications were used during this study:

- Microsoft Word 2003 was used for all word processing requirements.
- 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 package 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 the 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. The diatom index values calculated for each diatom sample taken were exported to sample database created in Microsoft Excel 2003.
- Statistical analyses (for correlation analyses, cluster analyses, descriptive statistics and the production of graphs) were carried out with Statistica version 7.
- Canonical correspondence analysis (CCA) was carried out with Canoco for Windows version 4.5 and CCA biplots were drawn with CanoDraw for Windows.
- Photo micrographs were taken during analyses of the diatom samples as describe above in section 2.5.1 and processed with Nikon NIS Elements-F software and exported to Microsoft Word 2003 for the compilation of plates.
- ESRI ArcGIS 9.2 ArcView software package was used for the production for the maps used during this study.

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3 ENVIRONMENTAL VARIABLES

RESULTS AND DISCUSSION

3.1 Introduction

In this chapter, account will be given of the results obtained from the measurements and analyses of the environmental variables as discussed in Chapter 2. The purpose of this chapter will be to highlight trends for the main environmental variables measured during the study period at the sites located in the Wasgoedspruit and Mooi River. In light of this information, Chapter 4 will provide an in depth discussion of the relationship between the measured environmental variables and the results obtained from indices derived from the species composition of the diatom communities.

The study was divided into two periods. In the first period, only the Wasgoedspruit was sampled to determine the short term changes in water quality as reflected by environmental variables, and how the diatom community and the suite of diatom indices calculated by OMNIDIA changed in response to these changes. In the second period sampling was extend to the Mooi River, the receiving water body of the Wasgoedspruit. This was done to investigate the overall effect of the Wasgoedspruit on the water quality of Mooi River and to see whether these hypothetical changes occurring in the Mooi River (see Chapter 1, section 1.2) as a result of the influence of the Wasgoedspruit, could be detected by changes in the diatom community and the resulting indices.

In the rest of this chapter the results of selected environmental variables will be discussed according to the periods. Firstly, account will be given of the physical environmental variables, and thereafter a discussion of the chemical environmental variables will follow. Not all the environmental variables measured or analysed will be discussed. Instead, only those variables that are deemed important by DWAF (1996) and those that aid in the description of the overall conditions encountered in the Wasgoedspruit and Mooi River will be discussed.

3.2 Physical environmental variables

3.2.1 Temperature

A brief discussion of the water temperature for the two periods is given below. The median water temperature values for the period February to August (winter) are illustrated by Figure 3.1. Figure 3.1 shows that the median water temperature tended to increase downstream.

The median water temperature at WS 3 was slightly lower than the median value for WS 2, but still higher than the median value at WS 1. This could be expected as WS 1 is located ca. 200 m downstream of the beginning of the concrete canalised section of the Wasgoedspruit, which has flowed in a natural streambed upstream of this section. The median value of WS 1 shows this very well as it is located almost exactly halfway between the median values for WS 0 and WS 2. A distinct increase in median water temperature can be seen between WS 0 and the two sites located in the canalised section of the Wasgoedspruit.

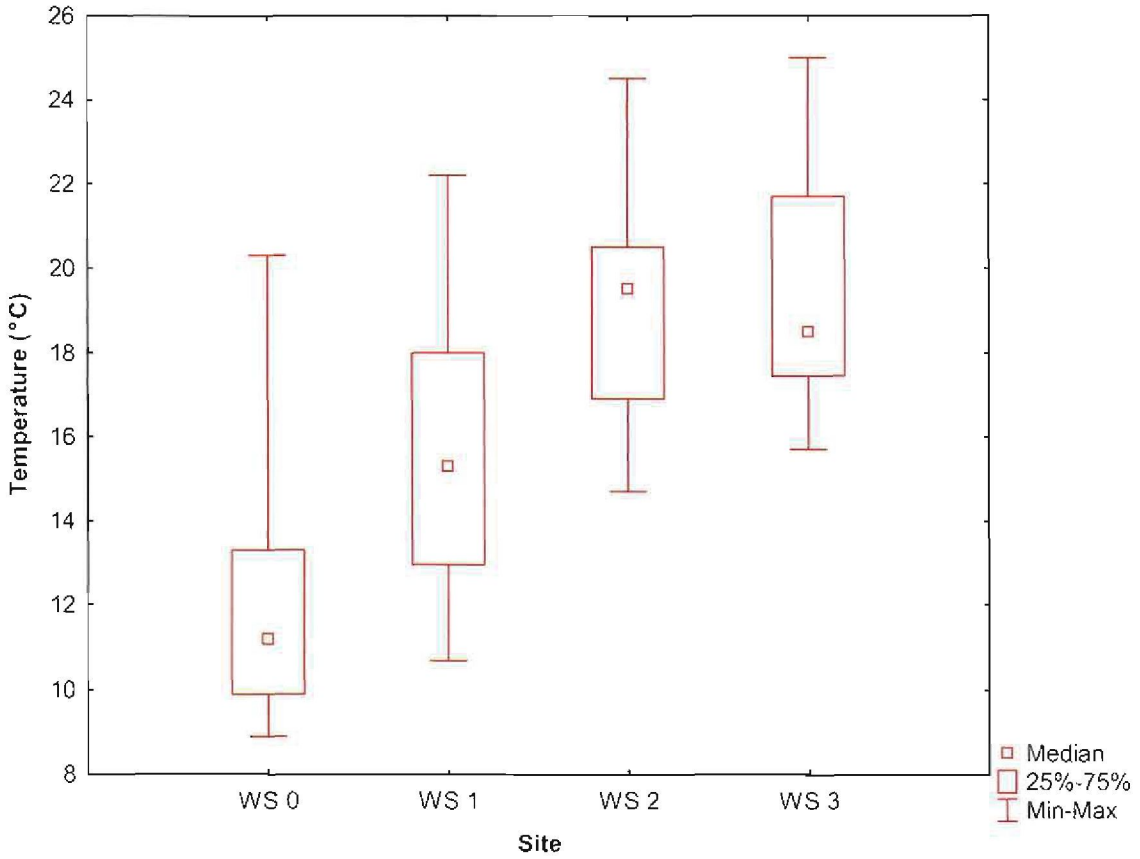

Figure 3.1 Median water temperature values at sites in the Wasgoedspruit for the period February 2005 to August 2005.

Temperatures in this period were lower than those in the second period, since the first period commenced at the end of summer and stretched to the end of winter. Mention should be made of the fact that although changes in water temperature lead to changes in diatom species composition, that this species composition would still reflect the overall water quality (Taylor, 2004).

The median water temperatures for the period September 2005 to January 2006 (summer) are shown in Figure 3.2. When Figure 3.2 is examined, it can be seen that the median water temperatures were higher than in the previous period due to the fact that the second period of the project took place in the spring and summer months. Further examination of the figure shows that the downstream increase in median water temperature was still prevalent in the Wasgoedspruit. The increase was not as great as seen in the first period. This could be ascribed to the fact that on some occasions the flow at WS 0 was severely reduced or at times stagnant, allowing for an increase in temperature. The median water temperature at WS 3 approaches 30°C and other values for this site are distributed close to the median value, showing that this site had constant high temperatures during this time.

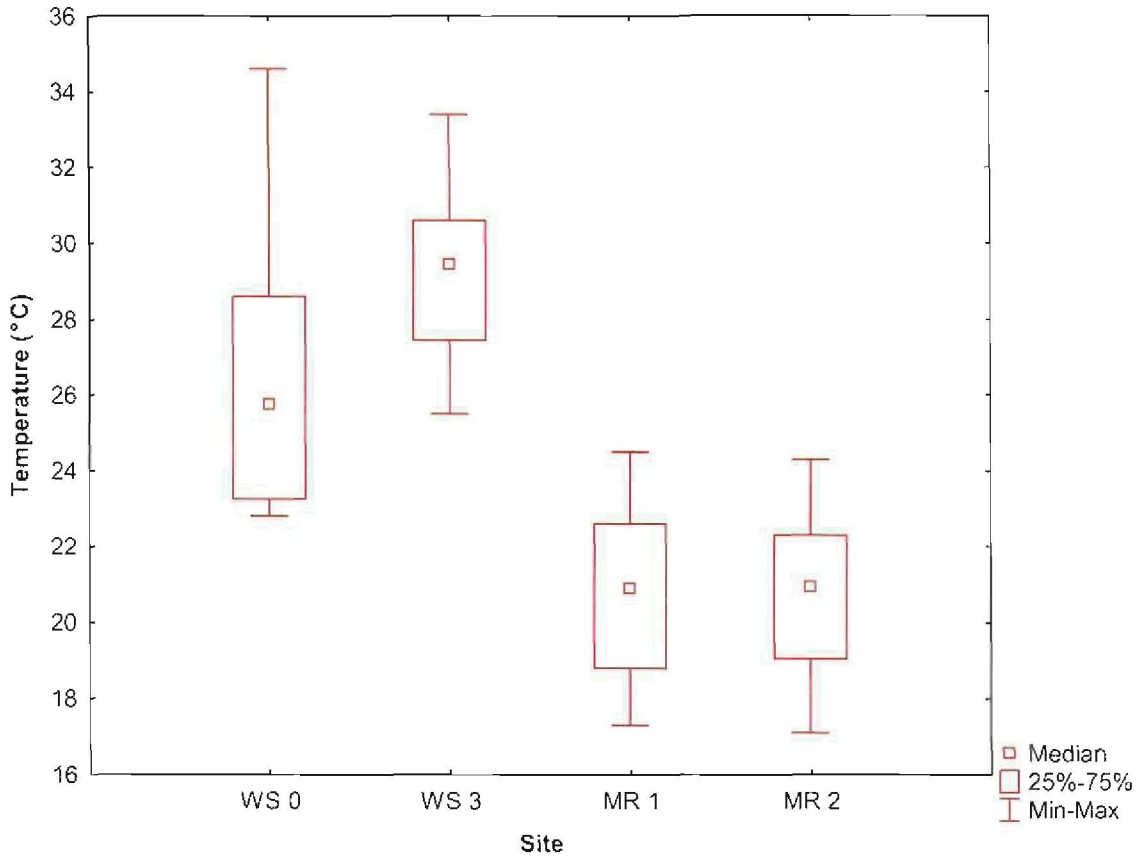

Figure 3.2 Median water temperature values at sites in the Wasgoedspruit and Mooi River for the period September 2005 to January 2006.

Median water temperatures in the Mooi River were lower than those encountered in the Wasgoedspruit. Almost no difference in water temperature could be detected between the two sites located in the Mooi River, MR 1 upstream of the confluence with the Wasgoedspruit and MR 2 downstream of this confluence. Figure 3.2 shows that water temperatures were, in fact, lower at the MR 2, which, from observation, could be ascribed to minor differences inherent at each site, such as increased shading present at MR 2 (see site description –

chapter 2, section 2.1). It appears, therefore, that in terms of temperature the Wasgoedspruit does not have any effect on the receiving Mooi River. This could be explained by the fact that the Wasgoedspruit is a small stream, which rarely exceeds a depth of a few centimetres. Another possible reason could be that just before the confluence of the two water bodies, there is a section where the Wasgoedspruit runs in a natural river bed, which would have a moderating effect on the temperature of the water of the Wasgoedspruit before the confluence.

3.2.2 Turbidity

Turbidity is important in larger streams as it can give a good indication of the amount of light penetrating the water, which affects the primary productivity (Dallas and Day, 2004). In the canalised section of the Wasgoedspruit light penetration was not limited, due to the low turbidity and shallow stream depth. Primary productivity was very high, revealed by the high dissolved oxygen values and the thick biofilm found on the concrete canal, consisting largely of diatoms and other benthic algae. Although turbidity may not have had a major influence on the water quality of the Wasgoedspruit and Mooi River, the results for both periods are illustrated in one figure and discussed below to give an overall picture of the prevailing conditions in these two streams.

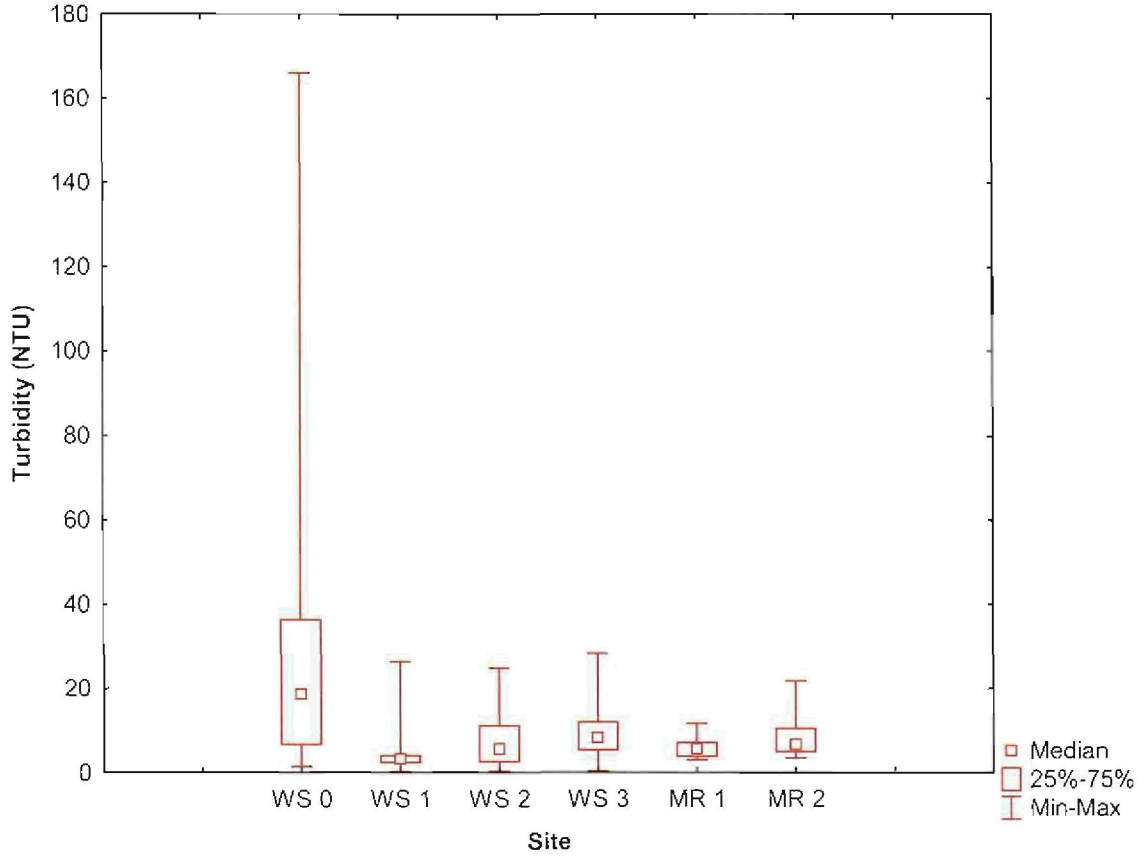

Figure 3.3 Median turbidity values at sites in the Wasgoedspruit and Mooi River (February 2005 to January 2006)

Figure 3.3 shows the median turbidity values observed for both periods. From Figure 3.3 it can be seen that overall the turbidity at the sites located in the canalised section of the Wasgoedspruit (WS 1, WS 2 & WS 3) was low in comparison to larger rivers such as the Vaal River (<29 NTU; Janse van Vuuren, 1996; Taylor, 2004). The low turbidity at these sites can be explained by the lack of a natural stream bed, which has been replaced by a concrete lining. The low turbidity can also be explained if one keeps in mind that the Wasgoedspruit has a low flow rate, with the stream being reduced to a mere trickle that barely covers the cross section of the canal. The result is a very shallow stream of water (ca. 50 mm deep), leading to the deposition of the small amount of suspended material present in the water and a low amount of re-suspension of already deposited material. The median turbidity values did not increase from MR 1 to MR 2.

Although, the turbidity observed at WS 0 was greater than at the other sites in the Wasgoedspruit, it was still low. The turbidity at WS 0 also showed greater variability, which could be ascribed to the fact that this section of the Wasgoedspruit runs in a natural stream bed, resulting in increased suspension of sediments in the water.

3.3 Chemical environmental variables

3.3.1 Dissolved Oxygen

The amount of dissolved oxygen (DO) in the water is at any given time is a function of many factors that include metabolic activity rates, diffusion, atmospheric pressure, temperature and proximity to the atmosphere (Dodds, 2002). The change in altitude from the highest to the lowest site is a mere 5 m, and therefore changing atmospheric pressure would not have affected the DO present in the water between sites.

The DO will be discussed in terms of the percentage saturation, which is the concentration of O_2 relative to the maximum equilibrium concentration of a particular solution. The reason for this is that the water was in most cases supersaturated with oxygen (>100%; Dallas and Day, 2004) and frequently above the detection limit of the dissolved oxygen meter used (>200%) with no specific reading available. In cases where the latter was true, a value of 220% was used. Dallas and Day (2004) also stated that percentage saturation gives a useful indication of the biological activity.

Figure 3.4 shows the median DO values measured during period 1. The main reasons for the elevated levels of DO in the water flowing in the canalised section of the Wasgoedspruit are the shallow stream depth and high primary productivity. The high primary productivity is

a result of the presence of a thick bio-film consisting mainly of diatoms and other algae that release O_2 into the water (Allan, 1995). It is important to keep in mind that this period occurred during the winter months, with the cooler water allowing for more dissolved oxygen in the water (Wetzel, 1983).

From Figure 3.4 it can be seen that the water flowing through the canalised section of the Wasgoedspruit was supersaturated with oxygen in almost al of the cases, with the highest median value found at WS 2, the shallowest site in the canal. The shallow waters (ca. 5 cm) of the canalised section of the Wasgoedspruit allow for greater atmospheric re-aeration, because of the relatively greater surface area of the water exposed to the atmosphere, than would normally be the case in deeper streams (Dallas and Day, 2004).

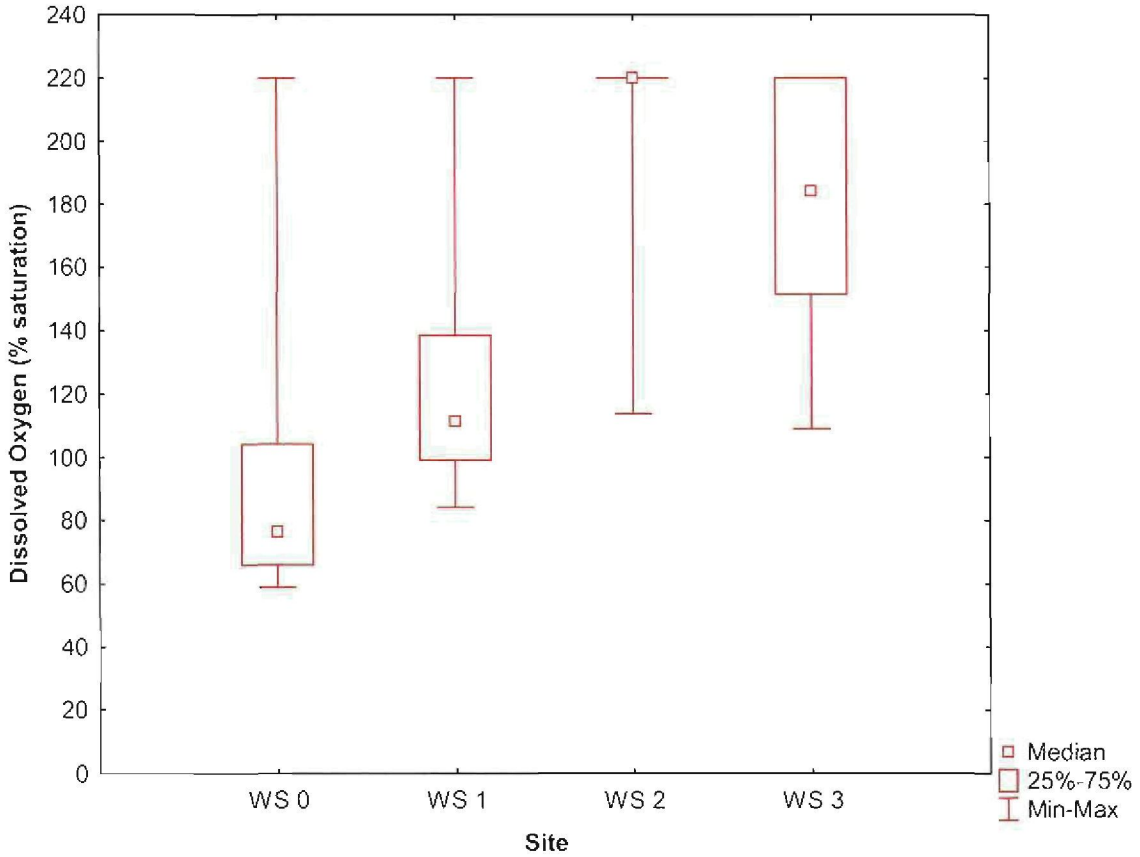

Figure 3.4 Median dissolved oxygen (% saturation) values at sites in the Wasgoedspruit for the period February 2005 to August 2005.

Figure 3.4 also illustrates the transition form the un-canalised section of the stream to the canalised section, with the increases in dissolved oxygen from WS 0 to WS 2. The median value at WS 3 is lower due to the fact that this site had a lot of sedimented material that sometimes lead to a visible reduction in flow rate, as well as a greater stream depth, resulting in less oxygen being mixed into the water. The biofilm present at this site was also not as

well developed as the biofilm present at the other two sites located within the canalised section of the Wasgoedspruit, which could be explained by the fact that this site receives less direct sunlight in the mornings from being shaded by the bridge crossing the canal. This means that there would be a drop in the rate of photosynthesis and therefore less O_2 dissolved into the water than at sites found upstream in the canal.

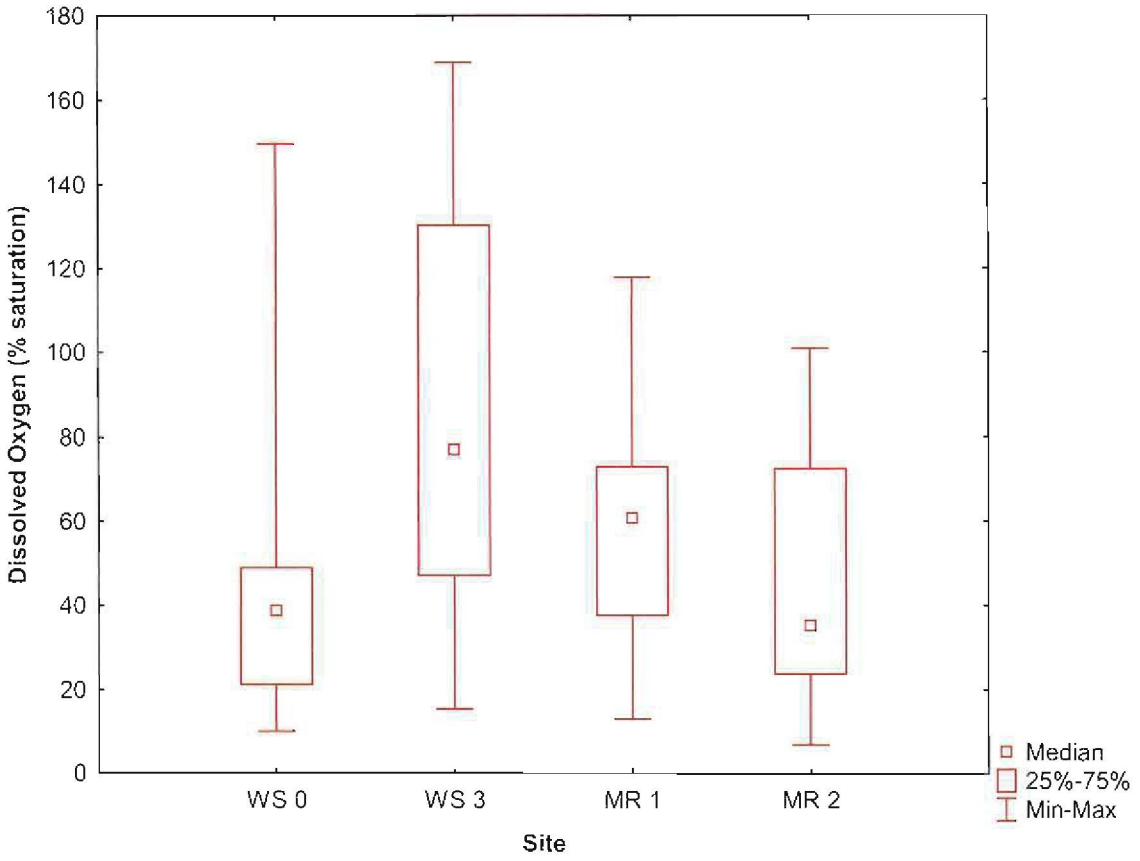

Figure 3.5 Median dissolved oxygen (% saturation) values at sites in the Wasgoedspruit and Mooi River for the period September 2005 to January 2006.

Figure 3.5 illustrates the median DO values measured for the period September 2005 to January 2006. From Figure 3.5 it can be seen that the temperature increase resulted in lower DO median values than those measured for the Wasgoedspruit during winter. The lower DO median values can also be ascribed to reduced primary productivity, as the biofilm present on the concrete substratum was not as prominent in winter. The DO median values decreased from up- to downstream of the Wasgoedspruit (Figure 3.5 MR 1 and MR 2).

3.3.2 Electrical conductivity (EC)

The total amount of material dissolved in a sample of water is generally seen as one of the major descriptors of the quality of the water (Dallas and Day, 2004). The total amount of material dissolved in water is generally given in one of three forms: total dissolved solids (TDS), salinity or electrical conductivity (Dodds, 2002). TDS is a measure of the total amount

of soluble material in a water sample. This includes ionised and un-ionised organic and inorganic material. The majority of this material is generally represented by the so called "major ions" - Na^+, K^+, Ca^{2+}, Mg^{2+}, HCO_3^-, CO_3^{2-}, Cl^- and SO_4^{2-}. TDS is determined by weighing the residue a known amount of filtered water and the unit is generally $mg.l^{-1}$. Electrical conductivity (EC) is a measure of the electrical conductance of water (Allan, 1995) and is measured in $mS.m^{-1}$ or $\mu S.cm^{-1}$, with S being Siemen or the reciprocal of an ohm. The water is able to conduct electric current because of the changed particles (ions) dissolved in it. It follows that an increase in the amount of ions in the water leads to an increase in the conductivity of the water.

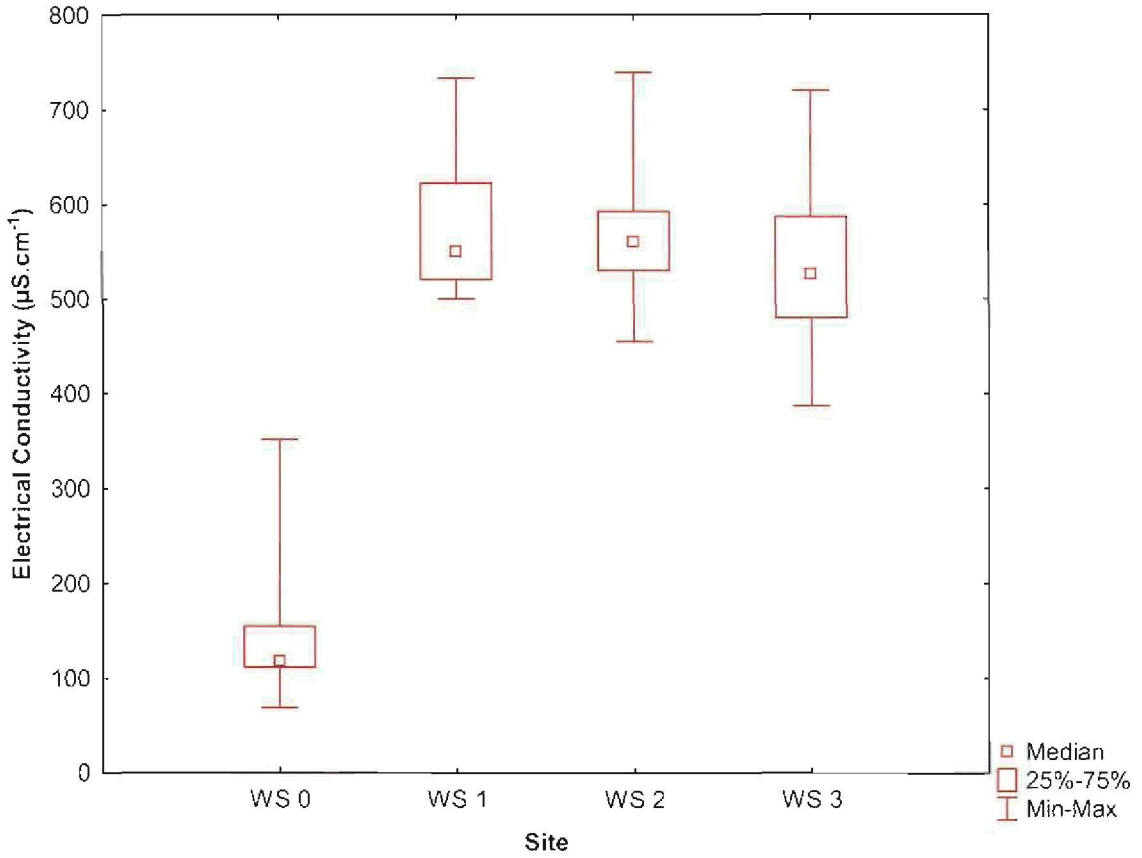

Figure 3.6 Median electrical conductivity values at sites in the Wasgoedspruit for the period February 2005 to August 2005.

As stated above, TDS mainly consists of ions and for this reason EC correlates very well with TDS and is often used as a surrogate for TDS. In a study by Taylor (2004) on the Vaal River, South Africa, TDS and EC showed 100% correlation, although this might not always be the case. In this study, EC was used as a measure of the amount of dissolved material present in the water. The reasons for this are that 1) the correlation of EC and TDS have shown to be very high in South Africa, 2) EC is easy to measure accurately in stream and 3) portable meters were available

Figure 3.6 shows the median values for EC at sites located in the Wasgoedspruit from February to August 2005. Figure 3.6 clearly illustrates the distinct increase in the EC from WS 0 to the sites located in the canalised section. This increase may be ascribed to the inflow of effluent from the industrial area of Potchefstroom, as well as other factors such as those stated in Table 1.1. The increase in median values from WS 0 to WS 1, 2 and 3 is almost five-fold, raising some questions in the light of the 15% maximum change recommended by the Water Quality Guidelines for aquatic ecosystems of South Africa (DWAF, 1996). In the canalised section, the median EC values remain rather constant and vary only slightly between 500 and 600 $\mu S.cm^{-1}$, with EC values reaching maximums in excess of 700 $\mu S.cm^{-1}$.

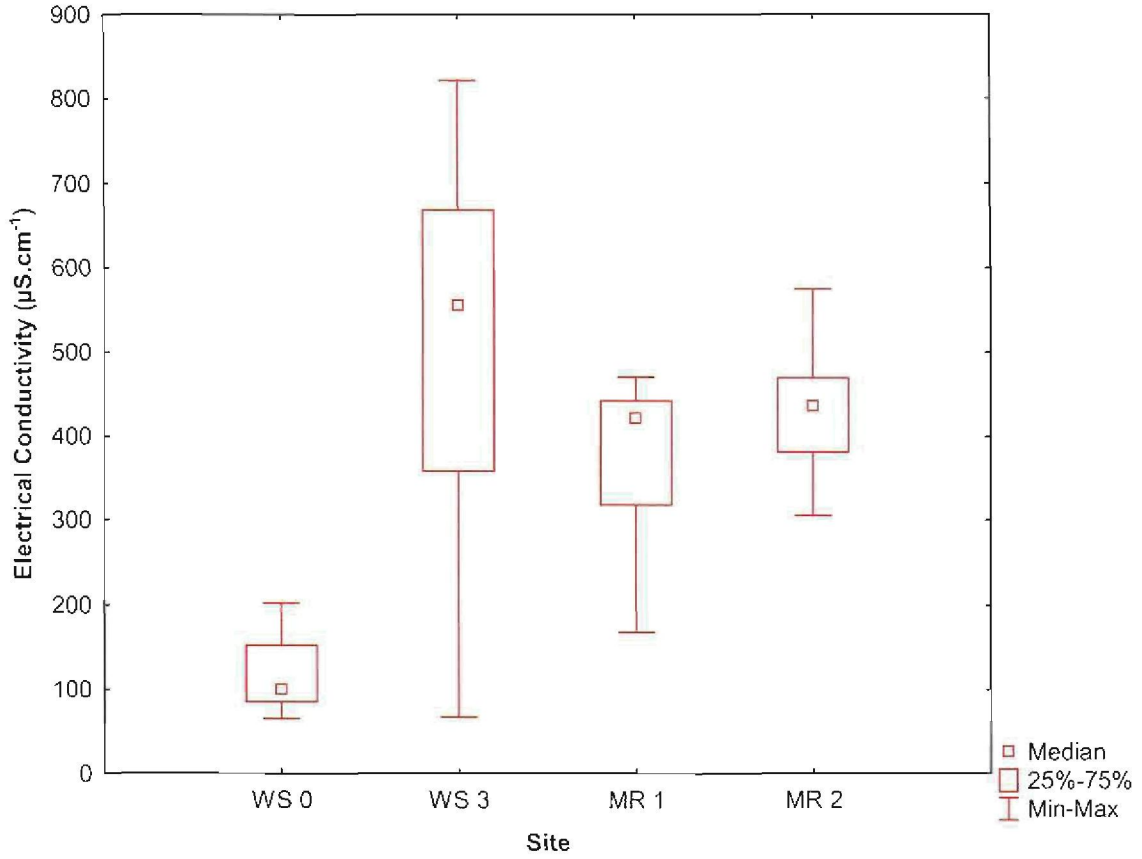

Figure 3.7 Median electrical conductivity values at sites in the Wasgoedspruit and Mooi River for the period September 2005 to January 2006.

Figure 3.7 shows the median values for the EC at sites in the Wasgoedspruit and Mooi River during period 2. The distinct increase in median values from WS 0 to WS 3 is still evident and the median values do not show a great amount of variation from those in period 1. However, WS 3 showed a greater variability which could be explained by the greater amount of precipitation during this period. In fact, prior to October 2005 there had not been any rain

for almost 4 months in Potchefstroom. The increase in precipitation usually leads to a dilution effect, but in the urban environment, precipitation leads to the concentration of material deposited on roads and other impermeable surfaces into the canals designed to conduct "excess" water away from the inhabited areas (Dallas and Day, 2004).

From Figure 3.7 it can also be seen that there is a slight increase in median EC values from MR 1, before the confluence with the Wasgoedspruit, and MR 2 below the confluence. In fact, there was an increase in EC from up- to downstream of the confluence with the Wasgoedspruit on every date of sampling during this period. This almost certainly results from the fact that the Wasgoedspruit adds dissolved inorganic material to the Mooi River, thereby altering the quality of the water as portrayed by EC.

3.3.3 pH

pH can be defined as the negative $\log_{10}$ of the hydrogen ion concentration (activity) and can be calculated be the following equation (Dodds, 2002):

$$pH \approx - \log_{10} [H^+]$$

Because the pH scale is based on negative logarithms, low pH values represent the highest H^+ concentration, and it also follows that a change in one unit of pH represents a 10-fold change in the hydrogen ion activity (Garret & Grisham, 1999). pH is one of the most important water quality parameters, because of its profound influence on water chemistry, as well as the organisms found in the aquatic environment (Dallas and Day, 2004). Substances dissolved in the water each have their own dissociation constants and pH determines the forms of electrolytes that are found in the water, by influencing the equilibrium of those substances. It, therefore, determines the availability and toxicity of, for instance, metals in water. Another example is that of non-metallic ions such as ammonium (NH_4^+) which is the non-toxic form in which nitrogen is taken up by most plants. When the pH increases above approximately 8, the equilibrium between NH_4^+ and NH_3 shift towards NH_3, a highly toxic substance (Dallas and Day, 2004).

In Chapter 4 it will be shown that the majority of the indices tested in this study, showed significant negative correlations with pH, necessitating a description of the conditions in the Wasgoedspruit and Mooi River in terms of pH.

Figure 3.8 displays the median pH values measured in the Wasgoedspruit from February to August 2005. The pH ranged from 6.63 to 10.56. Median values for pH were close to or in

excess of pH 9. When considering that the pH normally found in natural conditions is in the range of pH 6-8 (DWAF, 1996), the pH of the Wasgoedspruit is rather high. DWAF (1996) also recommend that the variation at a site should not be greater than 0.5 of a pH unit. The variations encountered all exceeded 2 pH units, meaning that minimum and maximum hydrogen ion concentrations varied 100-fold.

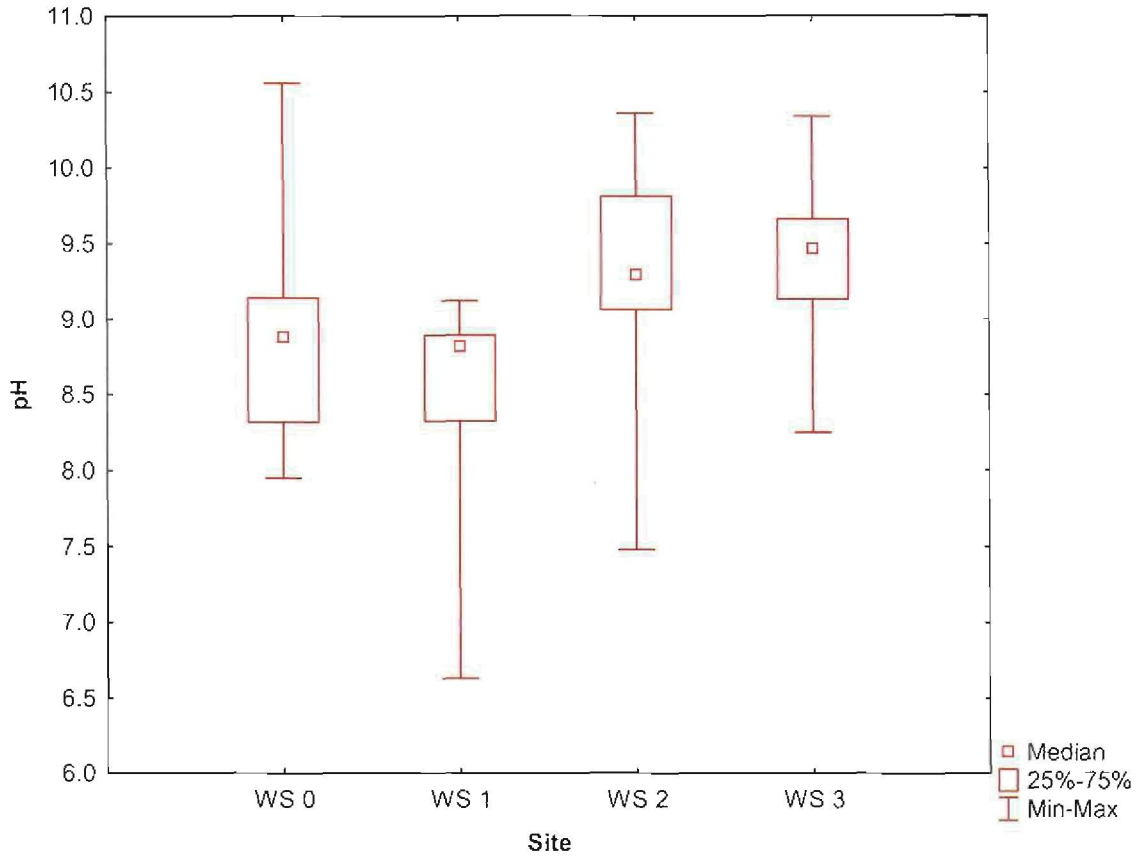

Figure 3.8 Median pH values at sites in the Wasgoedspruit for the period February 2005 to August 2005.

The median pH values encountered in the Wasgoedspruit and Mooi River from September 2005 to January 2006 are shown in Figure 3.9. The pH ranged in the Wasgoedspruit from 5.99 to 10.74 and from 6.91 to 9.54 in the Mooi River. Median values for pH occurred within or just out of the pH 6-8 range stated by DWAF (1996), but the median value for WS 3 exceeded pH 9.5. Variability in the pH of the Wasgoedspruit was higher than that seen during winter with differences greater than 3.5 units visible from Figure 3.9. In the Mooi River the median values for pH did not change to a great extent from up to downstream of the confluence with the Wasgoedspruit. There was, however, a greater variability in pH at MR 2 downstream of the confluence, with the pH ranging between 7.36 and 8.73 at MR 1 and 6.91 and 9.54 at MR 2.

As reflected by the high amount of dissolved oxygen present, the rate of primary productivity was high. It is suspected that this high rate of photosynthesis lead to the removal of large amounts of CO_2 from solution, thereby disturbing the equilibrium existing between CO_2 and carbonic acid (H_2CO_3) and driving it towards CO_2. This formation of CO_2 from carbonic acid would then have caused a reduction in the hydrogen ion concentration, resulting in a higher pH (Dodds, 2002).

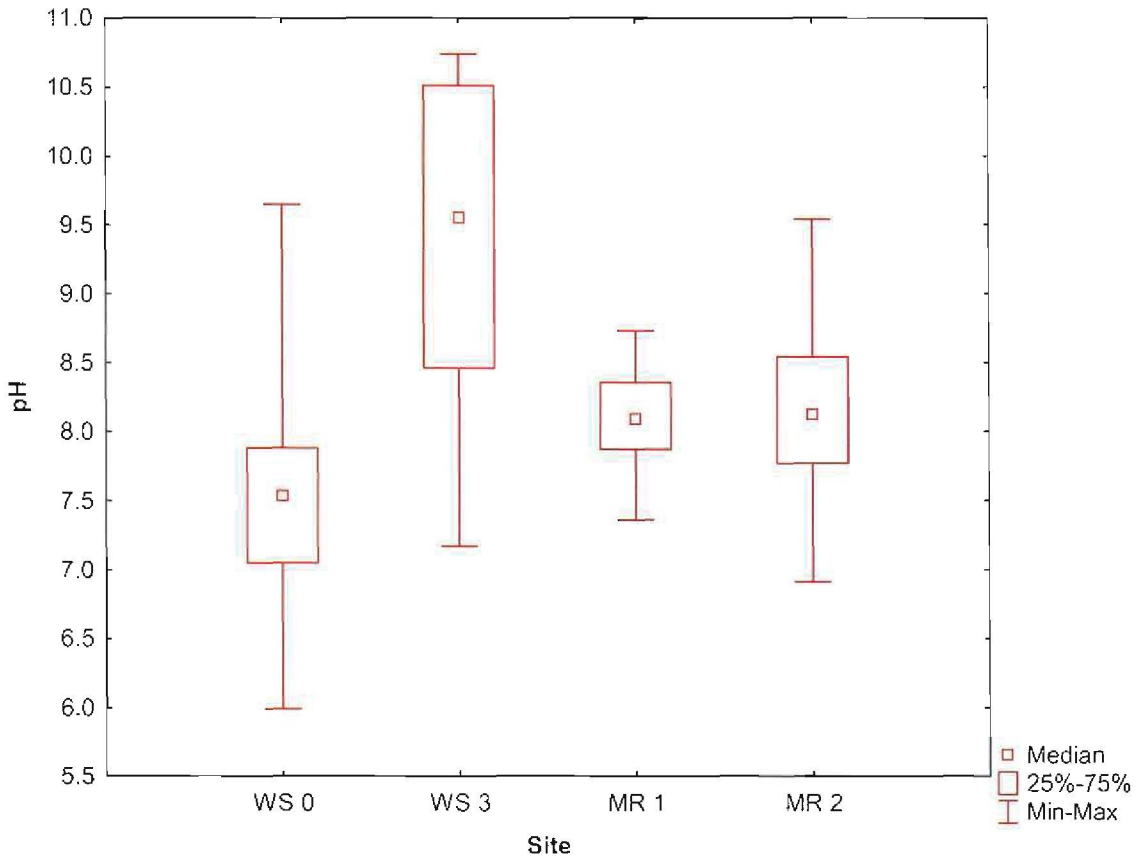

Figure 3.9 Median pH values at sites in the Wasgoedspruit and Mooi River for the period September 2005 to January 2006.

3.3.4 Nutrients

3.3.4.1 Nitrate and ammonia

Nitrogen (N) occurs abundantly in nature and is an essential constituent of many biological structures and biochemical processes, e.g. proteins, chlorophyll, nucleic acids, cell walls. In the aquatic environment nitrogen may be present in many forms, which include inorganic and organic forms. The forms that are measured by the common water quality tests include nitrate (NO_3^-), nitrite (NO_2^-), un-ionized ammonia (NH_3) and ammonium (NH_4^+) (Dallas and Day, 2004). Dodds (2002), stated that the two most important forms of dissolved inorganic nitrogen (DIN) in natural waters are nitrate and ammonium. Nitrite is oxidized to nitrate by nitrifying bacteria under aerobic conditions (DWAF, 1996). Earlier measurements in the

Wasgoedspruit have shown that this stream was highly oxic (De la Rey *et al.*, 2004). For the purposes of this study, therefore, these two forms of inorganic nitrogen were measured and a brief discussion will be given of these two forms of inorganic nitrogen.

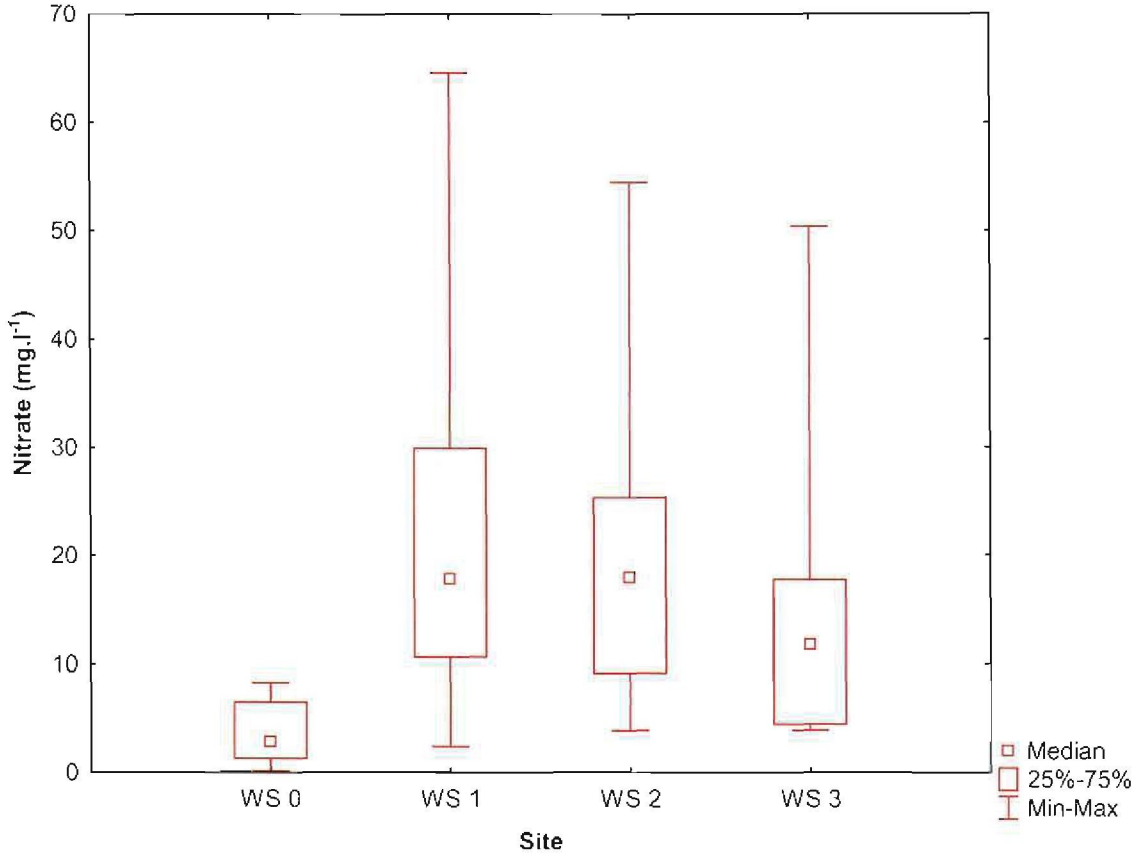

Figure 3.10 Median nitrate (NO_3-N) values at sites in the Wasgoedspruit for the period February 2005 to August 2005.

Nitrate is rarely abundant in natural surface waters because it is incorporated into the cells of organisms or chemically reduced by bacteria to atmospheric nitrogen (Davies and Day, 1998). For assimilation of nitrogen to take place in organisms, nitrogen has to be in the form of ammonium in the cell. Nitrate is taken up by organisms and reduced in the cell to nitrite by the enzyme nitrate reductase. Nitrite is then reduced to ammonium by the enzyme nitrite reductase (Dodds, 2002). Under oxic environments ammonium has a higher potential energy than nitrate, meaning that energy is needed to convert nitrate to ammonium before it can be assimilated by organisms. For this reason, primary producers and bacteria prefer ammonium (Dodds, 2002). However, under conditions of high pH, ammonium is converted to un-ionized ammonium hydroxide (NH_4OH), which is highly toxic (Wetzel, 1983).

The median values for nitrate (NO_3-N) concentrations in the Wasgoedspruit for the period February to August 2005 are shown in Figure 3.10. From Figure 3.10 it can be seen that the

median nitrate concentration in the canalised section of the Wasgoedspruit were between 10 and 20 mg.l-1. The median nitrate concentration increased from WS 0 to the canalised section as in the whole of the Wasgoedspruit and decreased from WS 1 to WS 3.

It can also be seen from Figure 3.10 that there was a greater variability in nitrate concentration at the sites located within the canalised section of the Wasgoedspruit, with maximum values measured in excess of 50 $mg.l^{-1}$ at WS 2 and WS 3, and a maximum of over 60 $mg.l^{-1}$ at WS 1. At WS 0 the nitrate concentration never exceeded 9 $mg.l^{-1}$.

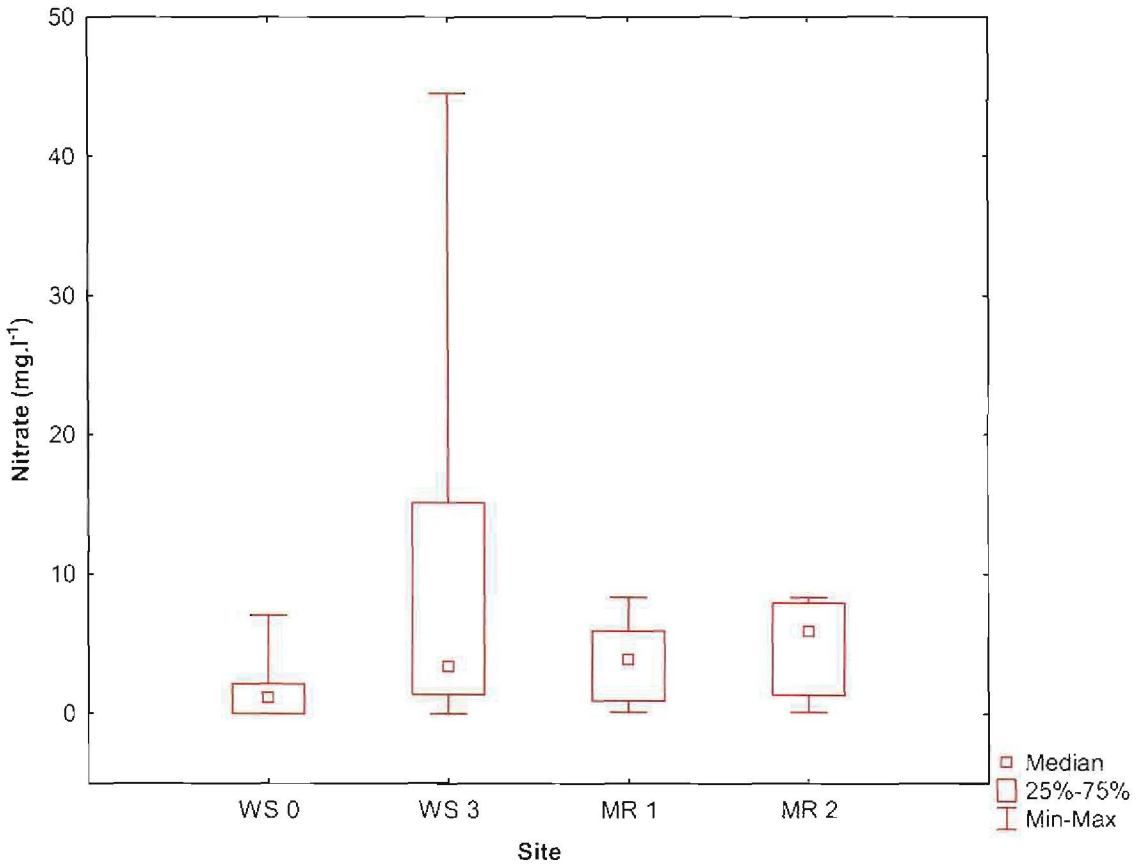

Figure 3.11 Median nitrate (NO_3-N) values at sites in the Wasgoedspruit and Mooi River (September 2005 to January 2006)

The median nitrate concentrations for the Wasgoedspruit and Mooi River for September 2005 to January 2006 are shown in Figure 3.11. From Figure 3.11 it can be seen that the nitrate concentrations measured at WS 0 remained lower than those found in the canalised section of the Wasgoedspruit (WS 3). The median nitrate concentration at WS 3 was lower than that measured during winter. Despite the lower median concentration observed for WS 3, there still was variability in the nitrate concentration at this site. Figure 3.11 also illustrates that the median nitrate concentration increased from upstream of the confluence of the

Wasgoedspruit (MR 1) to downstream of the confluence (MR 2). No concentrations in excess of 9 mg.l^{-1} nitrate were encountered in the Mooi River.

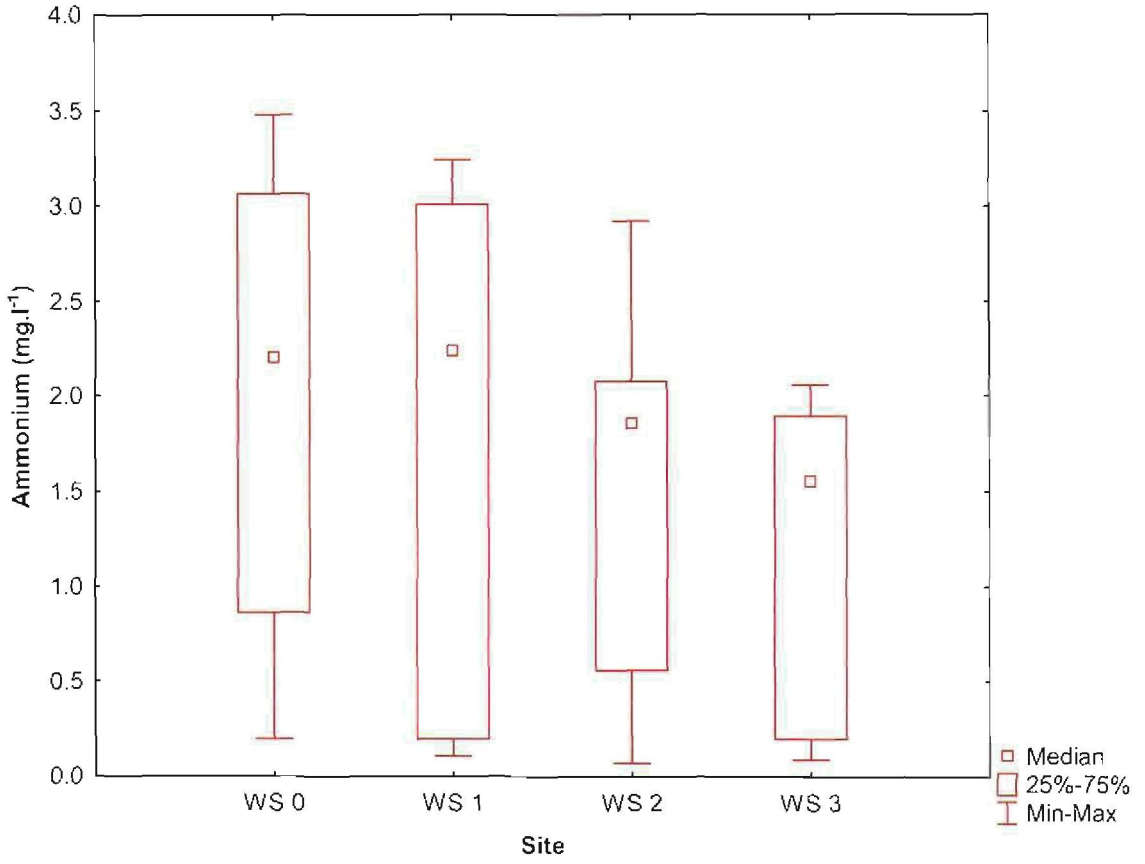

Figure 3.12 Median ammonium (NH_4-N) values at sites in the Wasgoedspruit for the period February 2005 to August 2005.

The nitrate concentration in the Wasgoedspruit was high, when the national water quality guidelines values are taken into consideration, where total dissolved inorganic nitrogen (NO_2-N + NO_3-N + NH_4-N) concentrations of >10 mg.l^{-1} are considered to be characteristic of hypertrophic conditions (DWAF, 1996). When compared to the Vaal River, a highly eutrophic river in South Africa, the nitrate concentrations are very high as Janse van Vuuren (1996) found the highest concentration of dissolved inorganic nitrogen to be 2.1 mg.l^{-1}. More recently, Taylor (2004) found the highest concentration of nitrate in the Vaal River did not exceed 3.5 mg.l^{-1}. In a study on the response of epilithic diatom communities to urbanization influences, Newall and Walsh (2005) found maximum DIN concentration and total nitrogen (TN) concentrations did not exceed 1.9 mg.l^{-1}.

The median ammonium concentrations for February to August 2005 are shown in Figure 3.12. The median ammonium concentrations remained constant from WS 0 to WS 3 during this part of the study, with values ranging from 1.55 mg.l^{-1} to 2.20 mg.l^{-1}. It can also be seen

from Figure 3.12 that there was a great amount of variability at each site with regard to ammonium concentrations.

Figure 3.13 shows the median ammonium concentrations for February to August 2005. From Figure 3.13 it can be seen that there was a sharp decline in the median ammonium concentrations measured in the Wasgoedspruit. Median concentrations were below 0.25 $mg.l^{-1}$. The sites located in the Wasgoedspruit also exhibited a low amount of variability. The sites located in the Mooi River showed median ammonium concentrations close to or in excess of 2 $mg.l^{-1}$, with a high degree of variability.

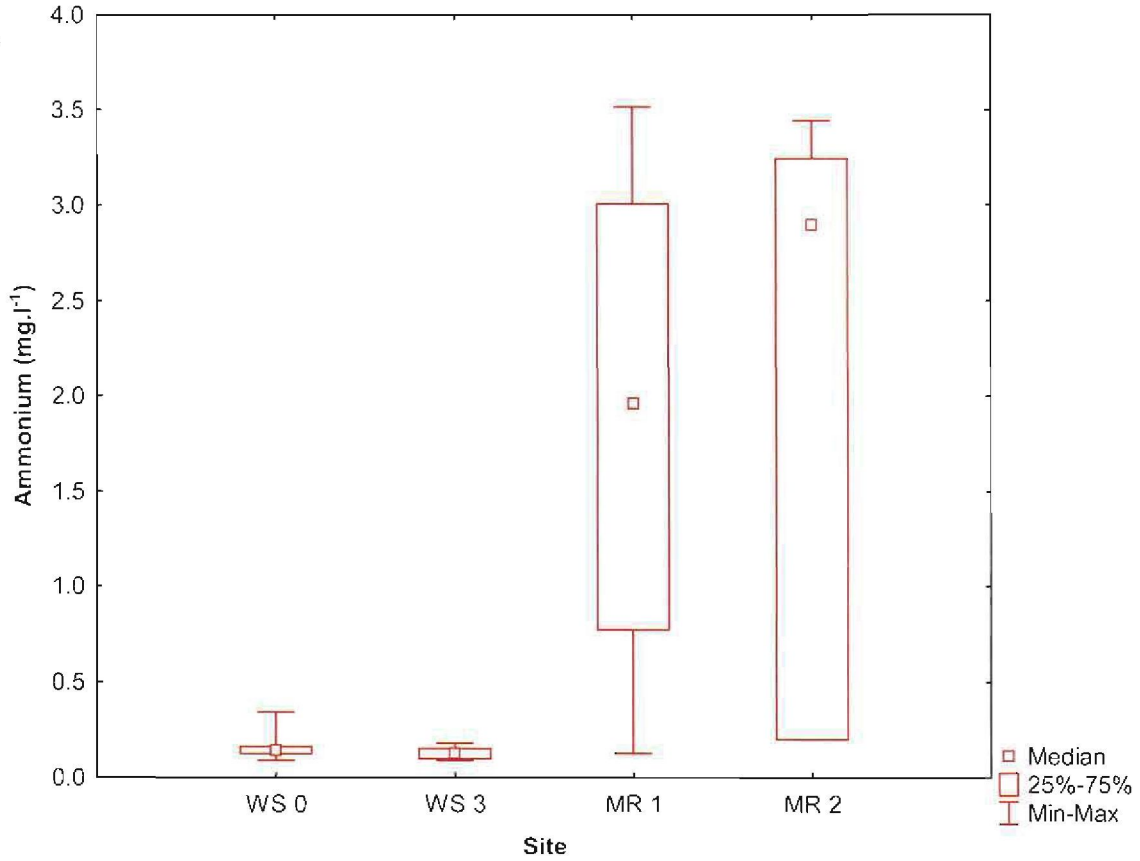

Figure 3.13 Median ammonium (NH_4-N) values at sites in the Wasgoedspruit and Mooi River for the period September 2005 to January 2006.

3.3.4.2 Sulphate

The main form in which sulphur occurs in waters is sulphate (SO_4^{2-}), which is reduced to sulfhydryl (–SH) groups in protein synthesis. Sulphate is not toxic, but can, in high concentrations, lead to the formation of sulphuric acid, that would lead to a reduction in the pH (Dallas and Day, 2004). Although information is not available from the National Water Quality Guidelines (DWAF, 1996) on acceptable levels for the aquatic environment, mention is made of the impacts that sulphates have in conjunction with other constituents found in the aquatic environment. Sulphates of Cadmium (Cd) and Zinc (Zn), for instance, have additive

effects on fish. Copper, on the other hand, decreases in toxicity in the presence of sulphates (DWAF, 1996). In Chapter 4 it will be shown that sulphates showed high significant negative correlations to the diatom indices tested, necessitating a discussion of the concentrations found in the Wasgoedspruit and Mooi River.

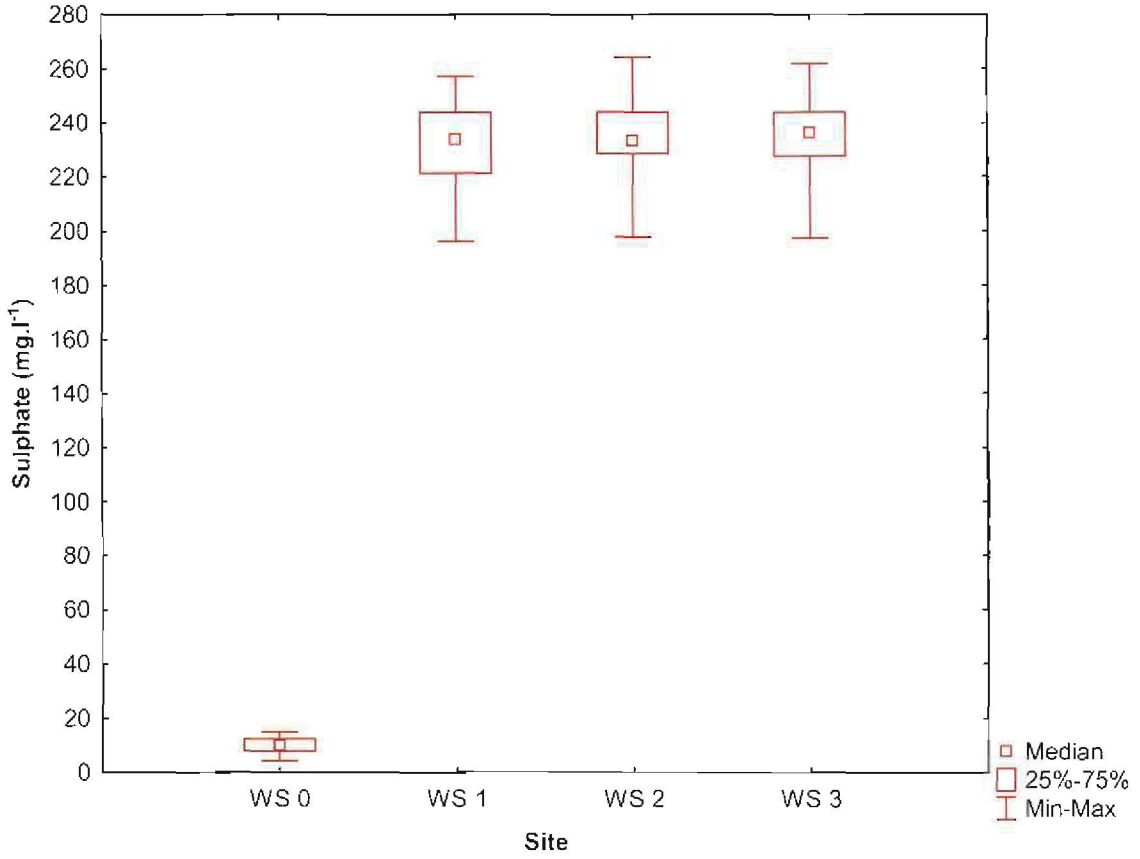

Figure 3.14 Median sulphate (SO_4-S) values at sites in the Wasgoedspruit for the period February 2005 to August 2005.

Figure 3.14 shows the median sulphate values encountered in the Wasgoedspruit for the period February to August 2005. Values for SO_4^{2-} at WS 0 did not exceed 20 $mg.l^{-1}$ for this period. A distinct increase in SO_4^{2-} concentration is observed, with median values in excess of 230 $mg.l^{-1}$at the sites further downstream. In fact, minimum concentrations for SO_4^{2-} never fell below 190 $mg.l^{-1}$.

Figure 3.15 shows the concentrations for SO_4^{2-} for the period September 2005 to January 2006. A sharp increase in the SO_4^{2-} concentration is still evident, although not quite as profound as the increase observed in summer. Median values varied little between MR 1 and MR 2.

When compared to the 0.011 $mg.l^{-1}$ considered by Förstner and Wittmann (1981) to be that found in "normal" river water, and that of Blinn and Bailey (2001) where a maximum of 12 $mg.l^{-1}$ was found, the sulphate concentrations found in the Wasgoedspruit were extremely high. A possible explanation for the high sulphate concentrations could be presence of the gypsum mining activities upstream of the sites monitored in this study.

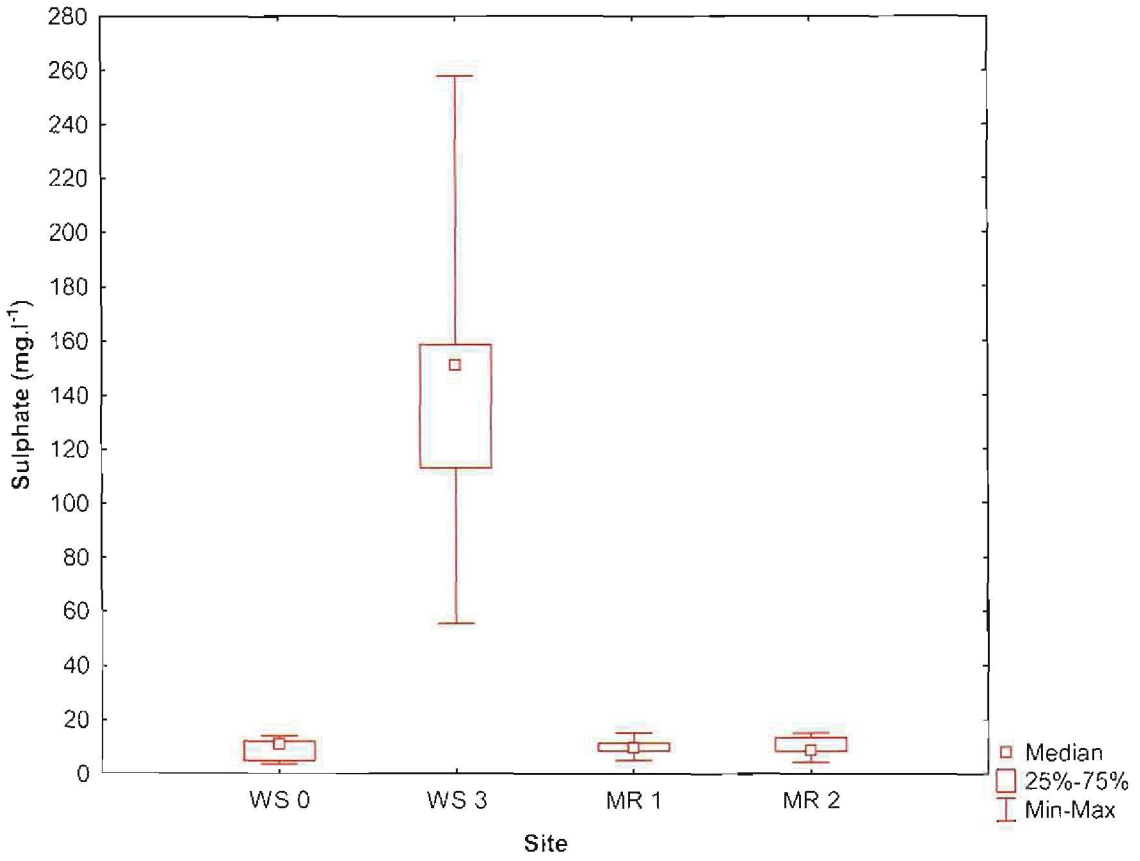

Figure 3.15 Median sulphate (SO_4-S) values at sites in the Wasgoedspruit and Mooi River for the period September 2005 to January 2006.

3.3.5 Selected trace metals

A trace metal as used in this study can be defined as a heavy metal (atomic mass greater than calcium) occurring in trace quantities in the environment (Davies and Day, 1998). The main sources of trace metals in water bodies are geological weathering, the atmosphere, industrial effluents, agricultural runoff and acid mine drainage (Dallas and Day, 2004). Trace metals are often highly toxic, but the level of toxicity of a particular trace metal depends on, amongst others, the following: the chemical species of the metal, the presence of other metals and organic compounds, the flow rate and volume of water in which it occurs, the temperature, the pH and the salinity (Davies and Day, 1998).

As stated in Chapter 2, the trace metals measured in this study were chosen because of the existence of established ecotoxicological values used in the development of the South

African Water Quality Guidelines for the protection of natural ecosystems (DWAF, 1996). A general discussion will follow in order to provide a broad overview of these trace metals and their concentrations measured in the Wasgoedspruit and Mooi River during the whole sampling period.

Table 3.1 Median, Mean, Standard Error, Standard deviation, Minimum and Maximum values measured in the Wasgoedspruit and Mooi River for the period February 2005 to January 2006. Shaded values exceed max. target water quality range (TWQR; DWAF, 1996)

Metal ($mg.l^{-1}$)	Median	Min	Max	TWQR
Al	0.068	0.002	18.000	0.01 *(pH>6.5)*
As	0.0023	0.0001	0.0190	0.01
Cd	0.00015	0.00005	0.00110	0.00015 - 0.0004
Cr	0.02400	0.00034	0.06200	0.007 - 0.012
Cu	0.0330	0.0058	1.3000	0.0003 - 0.0014
Pb	0.00550	0.00034	0.01500	0.0002 - 0.0012
Mn	0.02100	0.00044	2.00000	0.18
Hg	0.00005	0.00001	0.00039	0.00004
Se	0.0084	0.0005	0.0610	0.002
Zn	0.0350	0.0011	0.2600	0.002

The concentrations for selected trace metals are given in Table 3.1. The majority of the trace metals measured had median concentrations in excess of the target water quality guideline concentrations stated by DWAF (1996), with only arsenic, cadmium and manganese not exceeding these guideline concentrations. From Table 3.1 it can be seen that all the trace metals measured had maximum concentrations that at some time during the sampling period exceeded the TWQR, sometimes by several orders of magnitude (e.g. aluminium, copper, zinc). Where median trace metal concentrations exceeded the TWQR, it had at least double the concentration of the upper concentration of the guideline range, with the exception of mercury.

3.4 Summary

From the discussion in this chapter it is evident that physical and chemical environmental variables in urban, canalised streams, such as the Wasgoedspruit and the Mooi River, are prone to extremes. With the exception of turbidity and a few of the trace metals, all other environmental variables displayed high values and/or a high degree of variability in the canalised section of the Wasgoedspruit. Environmental variable values/concentrations were much higher in the Wasgoedspruit than the Mooi River. The influence of the elevated concentrations of the environmental variables of the Wasgoedspruit on those of the Mooi River was also visible, especially in the cases of pH, electrical conductivity, nitrate and ammonium.

With regard to the physical environmental variables, temperature increased downstream in the Wasgoedspruit, with the increase greater in winter than in summer. The water temperature of the Wasgoedspruit did not seem to have a great impact on that of the Mooi River. Turbidity was relatively low in comparison to other larger rivers (e.g. the Vaal River) in South Africa possibly due to the low flow rate and shallow stream depth.

The chemical environmental variables all had values much higher than guideline values in the Wasgoedspruit. Although nutrient concentrations showed a high degree of variability (with the exception of sulphate), these concentrations were still extremely high. Electrical conductivity, pH and dissolved oxygen showed consistently high levels in the Wasgoedspruit. Electrical conductivity and nitrate concentrations from the Wasgoedspruit appeared to have the greatest impact on that of the Mooi River. The majority of the trace metals measured had median concentrations in excess of guideline concentrations, except for arsenic, cadmium and manganese.

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4 DIATOM INDICES AND THEIR RELATIONSHIP TO ENVIRONMENTAL VARIABLES

RESULTS AND DISCUSSION

4.1 Introduction

In this chapter, account will be given of the results obtained from the analyses of the diatom samples collected at the various sites in the Wasgoedspruit and Mooi River. First, a description of the taxonomic composition of the diatom communities at the different sites will be given, followed by a discussion of the results of canonical correspondence analysis (Ter Braak, 1986) of the main environmental variables discussed in Chapter 3 and their relationship to the dominant diatom taxa encountered.

Correlations between selected diatom indices and environmental variables are discussed, followed by a discussion of the results of the diatom indices, calculated with the OMNIDIA software package for each site. Attention is also given to the difference in index scores obtained from different substrata.

4.2 Taxonomic composition

The value generated by a diatom index is a result of the mean of the water quality optima of the diatom taxa in a sample, weighted by the abundance of each taxon (Descy, 1979). It follows from this that the identification of diatom taxa should be made as accurately as possible, to increase the verity of the value generated by a particular index. The diatom taxa encountered in this study are listed in table 4.1. During the course of this study a total of 144 diatom taxa were identified. Of these 144 taxa, a total of 37 taxa were found to be dominant (i.e. having a relative abundance of > 5% of a particular diatom community). Fifty-six of the remaining 104 taxa had relative abundances of < 1%.

4.3 Ecological classification

Diatom species were classified according to Van Dam *et al.* (1994). According to these ecological indicator values, the majority of the dominant taxa (89%) encountered in the Wasgoedspruit and Mooi River are indicative of circumneutral to alkalophilous waters, which agrees with the results of pH measurements as discussed in Chapter 3 (section 3.3.3). According to this same classification system, 30% of the dominant taxa are nitrogen autotrophic (taxa with the ability to tolerate elevated concentrations of organically bound nitrogen), while 43% of the dominant taxa encountered are facultatively/obligately nitrogen-heterotrophic (taxa that need elevated concentrations of organically bound nitrogen).

Table 4.1 Diatom species encountered in the Wasgoedspruit and Mooi River from February 2005 to January 2006 (Bolded taxa were dominant)

Achnanthes inflata (Kützing) Grunow	*Fragilaria leptostauron* (Ehrenberg)	***Nitzschia capitellata* Hustedt in A. Schmidt**
Achnanthidium minutissimum var. *affinis* (Grunow) Bukhtiyarova	*Fragilaria parasitica* (W. Smith) Grunow	*Nitzschia clausii* Hantzsch
***Achnanthidium minutissimum* (Kützing) Czarnecki**	*Fragilaria pinnata* Ehrenberg var. *pinnata*	*Nitzschia communis* Rabenhorst
***Achnanthidium biasolettianum* (Grunow in Cleve & Grunow) Lange-Bertalot**	*Fragilaria ulna* (Nitzsch) Lange-Bertalot	*Nitzschia desertorum* Hustedt
Achnanthidium eutrophilum (Lange-Bertalot) Lange-Bertalot	*Fragilaria ulna* (Nitzsch) Lange-Bertalot var. *acus* (Kützing) Lange-Bertalot	***Nitzschia dissipata* (Kützing) Grunow**
***Achnanthidium saprophila* (Kobayasi & Mayama) Round & Bukhtiyarova**	*Frustulia vulgaris* (Thwaites) De Toni	*Nitzschia fonticola* Grunow in Cleve & Möller
Adlafia bryophila (Petersen) Moser, Lange-Bertalot & Metzeltin	*Gomphonema affine* Kützing	***Nitzschia frustulum* (Kützing) Grunow**
Amphora inariensis Krammer	*Gomphonema angustatum* (Kützing) Rabenhorst	*Nitzschia heufleriana* Grunow
Amphora libyca Ehrenberg	*Gomphonema gracile* Ehrenberg	***Nitzschia inconspicua* Grunow**
Amphora montana Krasske	*Gomphonema minutum* (Agardh) Agardh	***Nitzschia intermedia* Hantzsch ex Cleve & Grunow**
***Amphora pediculus* (Kützing) Grunow**	***Gomphonema parvulum* (Kützing) Kützing**	***Nitzschia liebetruthii* Rabenhorst**
Asterionella formosa Hassall	***Gomphonema pseudoaugur* Lange-Bertalot**	***Nitzschia linearis* (Agardh) W.M. Smith**
Aulacoseira ambigua (Grunow) Simonsen	*Gomphonema pumilum* (Grunow) Reichardt & Lange-Bertalot	*Nitzschia microcephala* Grunow in Cleve & Moller
Aulacoseira granulata (Ehrenberg) Simonsen	*Gomphonema truncatum* Ehrenberg	*Nitzschia nana* Grunow in Van Heurck
Aulacoseira granulata var.*angustissima* (O. Müller) Simonsen	*Gyrosigma acuminatum* (Kützing) Rabenhorst	***Nitzschia palea* (Kützing) W. Smith**
Aulacoseira muzzanensis (Meister) Krammer	*Gyrosigma attenuatum* (Kützing) Rabenhorst	***Nitzschia paleacea* (Grunow) Grunow in van Heurck**
Caloneis bacillum (Grunow) Cleve	*Hantzschia amphioxys* (Ehrenberg) Grunow in Cleve & Grunow	***Nitzschia paleaeformis* Hustedt**
Cymbella kappii (Cholnoky) Cholnoky	*Hippodonta capitata* (Ehrenberg) Lange-Bertalot, Metzeltin & Witkowski	*Nitzschia recta* Hantzsch in Rabenhorst
***Cocconeis pediculus* Ehrenberg**	*Luticola mutica* (Kützing) D.G. Mann	*Nitzschia sinuata* var. *tabellaria* Grunow
Cocconeis placentula Ehrenberg	*Luticola ventricosa* (Kützing) D.G. Mann	*Nitzschia sociabilis* Hustedt
***Cocconeis placentula* var. *euglypta* (Ehrenberg) Grunow**	***Mayamaea atomus* (Kützing) Lange-Bertalot**	*Nitzschia subacicularis* Hustedt in A.Schmidt
***Craticula accomoda* (Hustedt) Mann**	*Mayamaea atomus* var. *permitis* (Hustedt) Lange-Bertalot	*Nitzschia supralitorea* Lange-Bertalot
Craticula ambigua (Ehrenberg) Mann	*Mayamaea fossalis* var. *obsidialis* (Hustedt)	***Nitzschia umbonata* (Ehrenberg) Lange-**

	Lange-Bertalot	**Bertalot**
Craticula halophila (Grunow ex Van Heurck) Mann	*Melosira varians* Agardh	*Pinnularia borealis* f. *rectangularis* Carlson
Cyclotella atomus Hustedt	*Navicula* sp.	*Pinnularia braunii* (Grunow) Cleve
***Cyclotella meneghiniana* Kützing**	*Navicula angusta* Grunow	*Pinnularia brebissonii* (Kützing) Rabenhorst
***Cyclotella ocellata* Pantocsek**	*Navicula antonii* Lange-Bertalot	*Pinnularia gibba* Ehrenberg
Cyclotella pseudostelligera Hustedt	*Navicula arvensis* var. *maior* Lange-Bertalot	*Pinnularia microstauron* (Ehrenberg) Cleve
Cymatopleura solea (Brébisson) W.Smith	*Navicula capitatoradiata* Germain	*Pinnularia subbrevistriata* Krammer
Cymbella cymbiformis Agardh	*Navicula cincta* (Ehrenberg) Ralfs in Pritchard	*Pinnularia viridiformis* Krammer
Cymbella tumida (Brébisson) Van Heurck	***Navicula cryptocephala* Kützing**	*Placoneis dicephala* (W. Smith) Mereschkowsky
Cymbella turgidula Grunow	***Navicula cryptotenella* Lange-Bertalot**	*Placoneis placentula* (Ehrenberg) Heinzerling
***Diadesmis confervacea* Kützing**	*Navicula lanceolata* (Agardh) Ehrenberg	***Planothidium frequentissimum* (Lange-Bertalot) Lange-Bertalot**
Diadesmis contenta (Grunow ex Van Heurck) Mann	*Navicula libonensis* Schoeman	*Psammothidium montanum* (Krasske) Mayama
Diatoma vulgaris Bory	*Navicula longicephala* var. *vilaplanii* Sabater & Lange-Bertalot	*Reimeria uniseriata* Sala, Guerrero & Ferrario
Encyonema minutum (Hilse in Rabhorst) D.G. Mann	*Navicula minuscula* Grunow in Van Heurck	*Rhoicosphenia abbreviata* (C. Agardh) Lange-Bertalot
Encyonema silesiacum (Bleisch in Rabhorst) D.G. Mann	*Navicula molestiformis* Hustedt	*Rhopalodia gibberula* (Ehrenberg) O.Müller
Encyonopsis microcephala (Grunow) Krammer	***Navicula pseudoventralis* Hustedt**	***Sellaphora pupula* (Kützing) Mereschkowksy**
***Eolimna minima* (Grunow) Lange-Bertalot**	***Navicula symmetrica* Patrick**	*Sellaphora seminulum* (Grunow) D.G. Mann
***Eolimna subminuscula* (Manguin) Moser, Lange-Bertalot & Metzeltin**	***Navicula tripunctata* (O.F.Müller) Bory**	*Stauroneis anceps* Ehrenberg
Fallacia pygmaea (Kützing) Stickle & Mann	*Navicula trivialis* Lange-Bertalot	*Stauroneis phoenicenteron* (Nitzsch) Ehrenberg
***Fistulifera saprophila* (Lange-Bertalot & Bonik) Lange-Bertalot**	***Navicula veneta* Kützing**	*Surirella angusta* Kützing
Fragilaria biceps (Kützing) Lange-Bertalot	*Navicula rostellata* Kützing	*Surirella Brébissonii* Krammer & Lange-Bertalot
***Fragilaria brevistriata* Grunow**	*Navicula viridula* (Kützing) Ehrenberg	*Surirella brightwellii* W. Smith
Fragilaria capucina Desmazieres	***NITZSCHIA* A.H. Hassall sp. 1**	*Surirella ovalis* Brébisson
Fragilaria construens (Ehrenberg) Grunow	*Nitzschia acicularis* (Kützing) W.M. Smith	***Thalassiosira weissflogii* (Grunow) Fryxell & Hasle**
Fragilaria construens f. *binodis* (Ehrenberg) Hustedt	***Nitzschia amphibia* Grunow**	*Tryblionella apiculata* Gregory
Fragilaria crotonensis Kitton	***Nitzschia archibaldii* Lange-Bertalot**	*Tryblionella gracilis* W. Smith

Based on the Van Dam *et al.* (1994) classification 70% of the dominant taxa encountered in the Wasgoedspruit and Mooi River are found occurring in meso-eutraphentic to hypereutraphentic waters. On first examination it seems, therefore, that the elevated levels of the various environmental variables in the Wasgoedspruit, and to some extent, the Mooi River (Chapter 3), are reflected by the dominant diatom taxa identified from these waterbodies. This was subject, however, to further analyses (including canonical correspondence analysis and correlation analyses) presented in the sections below.

4.4 Canonical correspondence analysis

Canonical correspondence analysis (CCA) is a multivariate direct gradient analysis technique, whereby a set of species is related directly to a set of environmental variables. This technique detects the patterns of variation in community composition that can be explained best by the environmental variables (Ter Braak, 1986). The purpose of using an ordination technique in this study was to determine the response of diatom species encountered in the Wasgoedspruit and Mooi River to general water quality, as changes in the diatom community may lead to changes in diatom index scores. Similar to other ordination techniques, CCA provides an integrated description of species-environment relationships by assuming a response model that is common to all species, and the existence of a single set of underlying environmental gradients to which all the species respond (Ter Braak, 1986). Canonical correspondence analysis has an advantage over other ordination techniques because it focuses on the relationships between species and environmental variables (Ter Braak, 1986).

The results of canonical correspondence analysis performed on the species and environmental variable data from the Wasgoedspruit and Mooi River from February 2005 to January 2006 are shown in Figure 4.1. From Figure 4.1 three distinct groupings of diatom species can be observed. The first group (Group 1 in the figure) consists mainly of diatom species that are common to streams with elevated pollution (i.e. elevated levels of organically bound nitrogen and high nutrient levels) or bad water quality species that Van Dam *et al.* (1994) describe to be facultatively/obligately nitrogen-heterotrophic and eutraphentic/hypereutraphentic. The species in this grouping include, amongst others: *Eolimna minima* (Grunow) Lange-Bertalot, *Eolimna subminuscula* (Manguin) Moser, Lange-Bertalot & Metzeltin, *Fistulifera saprophila* (Lange-Bertalot & Bonik) Lange-Bertalot, *Cyclotella meneghiniana* Kützing, *Nitzschia palea* (Kützing) W. Smith, *Nitzschia paleacea* (Grunow) Grunow in van Heurck, *Gomphonema parvulum* (Kützing) Kützing. After inspection of the localities in which these species occurred (see Appendix B), it becomes

evident that all of these species were encountered in the canalised section of the Wasgoedspruit as can be seen from Figure 4.2.

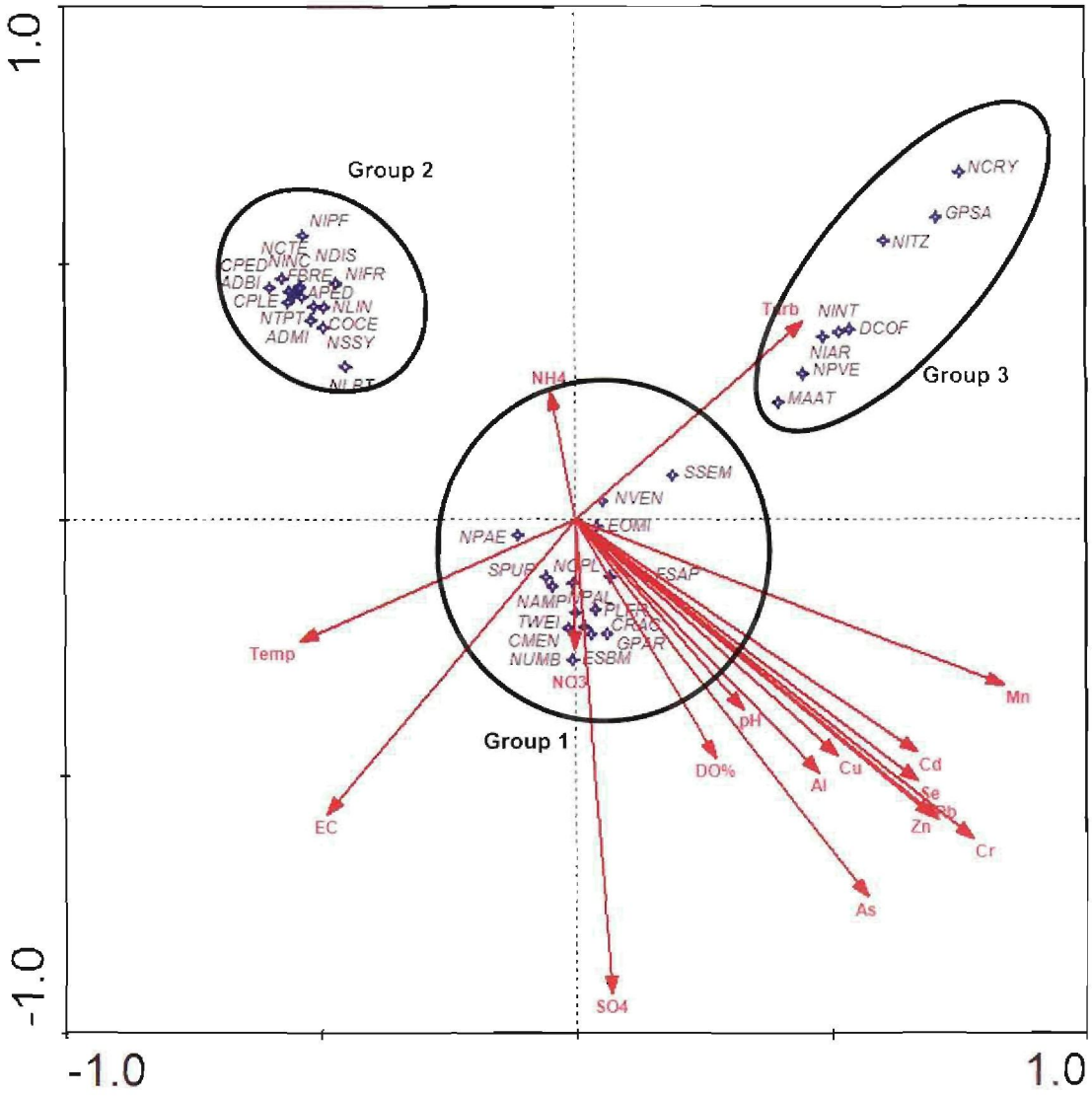

Figure 4.1 *Canonical correspondence analysis biplot showing the relationship between the dominant diatom species and measured environmental variables measured in the Wasgoedspruit and Mooi River from February 2005 to January 2006. Species with a weight range of 2-100% are shown. Acronyms for the species are provided in Appendix B. Refer to Table 4.2 for a summary of the environmental variable data used for this CCA.*

From Figure 4.1 it can also be observed that the main environmental drivers of the diatom species in group 1 were high nutrient levels (NO_3-N, SO_4-S). This was expected, given these species are indicative of eutrophied or highly eutrophied waters. Other environmental variables driving this group of species were high electrical conductivity, high pH and high levels of dissolved oxygen. In the Wasgoedspruit, the source of the high SO_4^{2-} ion concentration was likely to be the ammonium sulphate ($[NH_4]_2SO_4$) discharged by the

fertiliser plant upstream from the sites measured. Ammonium sulphate dissociates in water to form NH_4^+ and SO_4^{2-} ions. It is suspected that the increased concentration of SO_4^{2-} ions would drive the equilibrium existing between SO_4^{2-} ions and HSO_4^- ions towards HSO_4^- and lead to an increase in the HSO_4^- ion concentration. In order for HSO_4^- ions to form from SO_4^{2-} ions, protons (H^+) would need to be removed from the water. This would lead to a reduction in the concentration of H^+ and subsequently lead to an increased pH, as described in section 3.3.3. The high concentration of SO_4-S, together with the high rate of photosynthesis (section 3.3.3), is suspected to have contributed significantly to the high pH and large amount of alkalophilous taxa observed in the Wasgoedspruit. Following this, it is suspected that the high correlations between the diatom indices and SO_4-S concentrations can not be attributed to the presence of high SO_4-S concentrations in itself, but rather reflects the strong influence that SO_4-S had on pH, which in turn dictated the presence of certain diatom species in the Wasgoedspruit.

Eolimna minima, *Navicula veneta* and *Fistulifera saprophila* seemed also to demonstrate a relationship to the trace metals measured in the present study[1], while temperature seems to have had an influence in the abundance of *Nitzschia paleacea* in the Wasgoedspruit.

The second grouping seen in Figure 4.1 consists mainly of species normally indicative of slightly better water quality than those of the first grouping. The majority of the species of this group (with the exception of ADMI) are normally encountered in eutrophic waters, and categorised as nitrogen autotrophic taxa, tolerating elevated concentrations of organically bound nitrogen (Van Dam *et al.*, 1994). This group is made up of the following species: *Achnanthidium minutissimum* (Kützing) Czarnecki, *Cocconeis pediculus* Ehrenberg, *Navicula tripunctata* (O.F.Müller) Bory, *Navicula cryptotenella* Lange-Bertalot, *Fragilaria brevistriata* Grunow, *Cyclotella ocellata* Pantocsek, *Navicula schroeteri* var. *symmetrica* (Patrick) Lange-Bertalot, *Nitzschia frustulum* (Kützing) Grunow, *Nitzschia liebetruthii* Rabenhorst. These species were all found in the Mooi River sites, MR 1 and MR 2.

From Figure 4.1 it can be seen that the main drivers were lower nutrient and trace metal concentrations, as wells as lower pH and dissolved oxygen, since these species occur on the opposite ends of the vectors of those particular environmental variables. These species also were related to a lower electrical conductivity than those encountered in the first group.

There are, however, two species encountered at the Mooi River sites that do not fit in completely with the other species of this second grouping, namely *Nitzschia frustulum* and

[1] For a more detailed discussion of diatom response to trace metals observed in this study, refer to section 4.6.

Nitzschia liebetruthii, which were encountered in greatest abundance in the Mooi River towards the end of December 2005 and throughout January 2006. These two species are usually found in electrolyte rich waters and *N. frustulum* is also tolerant of critical levels of pollution and fluctuations in osmotic pressure (Krammer and Lange-Bertalot 1986-1991).

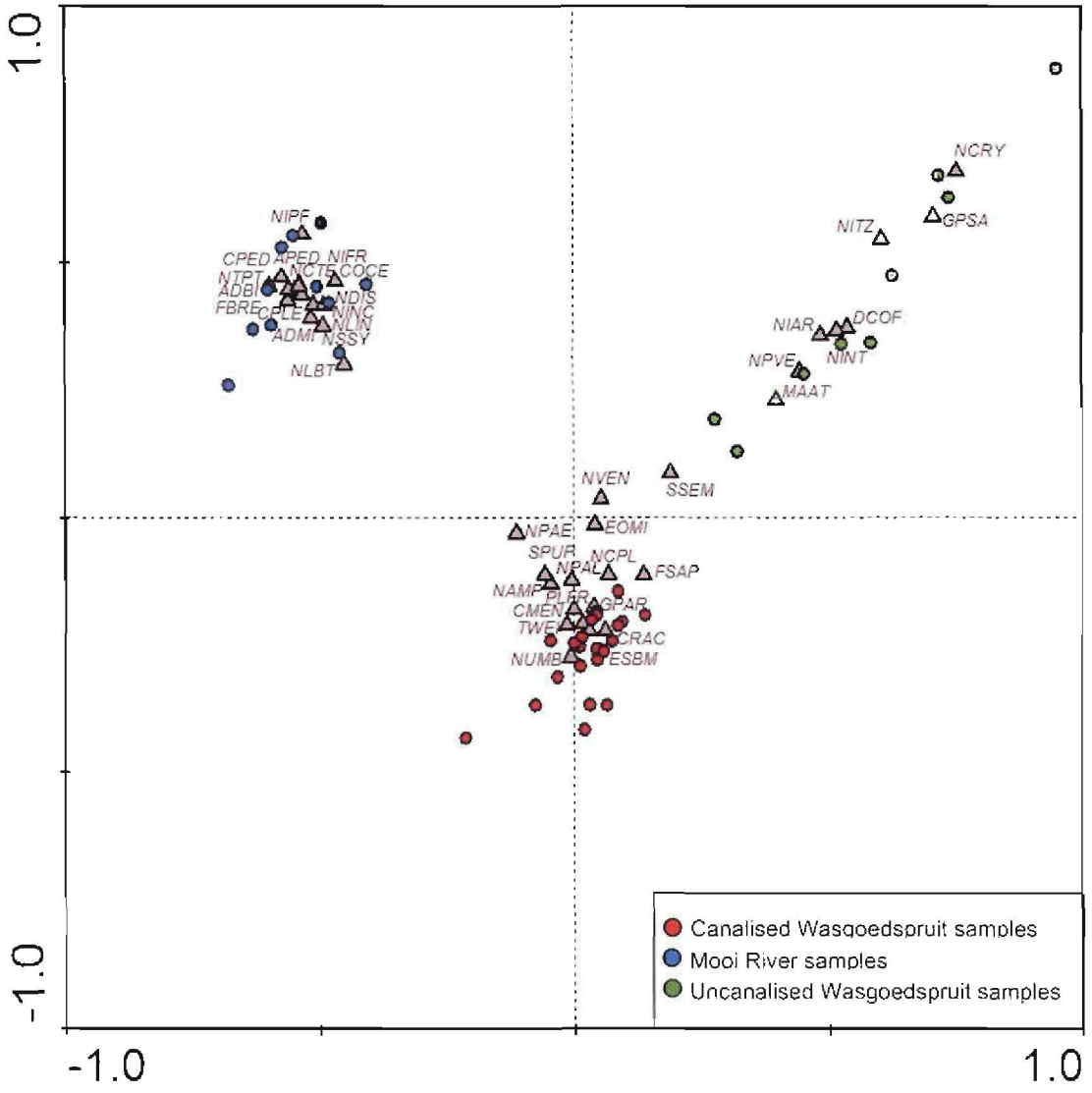

Figure 4.2 Canonical correspondence analysis biplot showing the relationship between the dominant diatom species and diatom samples collected in the Wasgoedspruit and Mooi River from February 2005 to January 2006. Species with a weight range of 2-100% are shown. Acronyms for the species are provided in Appendix B.

Van Dam *et al.* (1994) describe these two species as having the same ecological preferences with regard to their nitrogen uptake metabolism, pH, trophic state, and importantly for this discussion, oxygen requirements. According to this classification, these species require a low amount of oxygen (< 50%). From Figure 4.1 it can be seen that these two species are found on the CCA in a position showing a relationship to low oxygen

concentrations. It was indeed the case that the percentage dissolved oxygen of the Mooi River had declined to levels consistently below 35% towards the end of December 2005 and declined to a minimum of 6% at MR 2 at the end of January 2006. It is interesting to note that the first occurrence of *N. frustulum* was towards the end of December 2005 and this species reached a maximum abundance of 40% at MR 2 at the end of January 2006.

Table 4.2 Summary of the environmental variable data used for the CCA presented in Figure 4.1. All units of measurement in $mg.l^{-1}$ unless otherwise indicated.

	Mean	Standard Error	Median	Standard Deviation	Range	Minimum	Maximum
Temp (°C)	19.73	0.76	19.80	5.57	23.50	9.90	33.40
DO %	117.10	9.40	95.80	68.44	213.94	6.06	220.00
EC (µS/cm)	420.45	23.44	451.00	170.68	737.00	65.00	802.00
pH	8.88	0.11	8.86	0.80	2.95	7.71	10.66
Turbidity (NTU)	13.17	3.17	5.80	23.11	164.63	1.37	166.00
NO_3	9.014606	1.837634	4.800000	13.378174	64.565000	0.005000	64.570000
NH_4	1.576822	0.177331	1.695628	1.290987	3.444110	0.072154	3.516264
SO_4	130.634881	14.476326	141.820000	105.389242	248.620000	3.920000	252.540000
Al	0.957121	0.481445	0.068000	3.504974	17.997900	0.002100	18.000000
As	0.002483	0.000234	0.002400	0.001706	0.006704	0.000096	0.006800
Cd	0.000168	0.000024	0.000145	0.000171	0.001055	0.000045	0.001100
Cr	0.019495	0.001496	0.023000	0.010892	0.040640	0.000360	0.041000
Cu	0.100806	0.034011	0.039000	0.247605	1.294200	0.005800	1.300000
Mn	0.063894	0.030257	0.018000	0.220276	1.599560	0.000440	1.600000
Pb	0.005471	0.000530	0.005000	0.003858	0.013640	0.000360	0.014000
Se	0.008019	0.000561	0.008400	0.004081	0.015500	0.000500	0.016000
Zn	0.052877	0.007276	0.041000	0.052970	0.198900	0.001100	0.200000

The third grouping observable from Figure 4.1 consists mainly of species abundant in the un-canalised section of the Wasgoedspruit (WS 0). The following dominant diatom species were abundant: *Navicula cryptocephala* Kützing, *Navicula pseudoventralis* Hustedt, *Diadesmis confervacea* Kützing, *Gomphonema pseudoaugur* Lange-Bertalot, *Nitzschia intermedia* Hantzsch ex Cleve & Grunow, *Mayamaea atomus* (Kützing) Lange-Bertalot. With the exception of one of the dominant species at this site, i.e. *N. pseudoventralis*, the species encountered are not necessarily indicative of good water quality, as might be expected of this site. These species do, however, often have a wide tolerance range. *Gomphonema pseudoaugur* may occur in meso- to eutrophic waters and is not tolerant of more than critical levels of pollution and *Diadesmis confervacea* may occur in a wide range of waters (Krammer and Lange-Bertalot 1986-1991). *Nitzschia intermedia* may occur in moderate to high electrolyte content (Krammer and Lange-Bertalot 1986-1991). *Mayamaea atomus* is often regarded as one of the most pollution resistant diatom taxa, but is often found to occur in moderate water quality associated with a micro-habitat, e.g. organic detritus (Lange-Bertalot, 2001). *Navicula cryptocephala* has an unusually broad tolerance spectrum, ranging

from oligotrophic, electrolyte poor, weakly acidic waters to eutrophic, moderately electrolyte rich and weakly alkaline (Lange-Bertalot, 2001).

4.5 Correlation analysis

Correlation analysis was performed between the diatom indices calculated from the analysis of diatom samples and the values of the environmental variables measured in the Wasgoedspruit and Mooi River. The correlation analysis was done in order to calculate the efficacy with which a selection of the diatom indices in use in Europe and the rest of the world, indicate changes in the water quality with regard to the environmental variables discussed in Chapter 3.

Three main correlations were carried out between diatom indices and measured environmental variables: 1) diatom indices and environmental data measured on the same date as diatom sampling, 2) diatom indices and the average of the environmental data one month prior to sampling, and 3) diatom indices and the maximum environmental variable values in the month prior to diatom sampling.

4.5.1 Correlation between diatom indices and concurrent environmental variable values

The correlation between diatom indices and concurrent environmental data was done with the aim of reflecting a situation similar to that often encountered in biological sampling regimes, where sampling for environmental variables commonly happens on the same day as biological sampling. It has been discussed in Chapter 1 that the measurement of environmental variables reflecting water quality might only represent a "snap shot" of the water quality conditions prevailing at that particular time. The results of a correlation between diatom indices and concurrent environmental variable values would therefore provide a measure of the efficacy of diatoms as bio monitoring tools in cases where once-off samples are taken.

The results of a Pearson correlation between diatom indices and concurrently measured environmental variable values are shown in Table 4.3. From Table 4.3 it can be seen that significant negative correlations ($p < 0.05$) existed between diatom indices and the majority of the environmental variables measured for. The Generic Diatom Index (GDI; Coste and Ayphassorho, 1991) seems to be the exception, with the only significant correlation of this index being with arsenic (As). This could be ascribed to the fact that this index uses diatom genera, and not species, to infer general water quality. From Table 4.1 it can be seen that there was a total of 42 genera encountered in the Wasgoedspruit and Mooi River during the

whole period of sampling. The canalised section of the Wasgoedspruit, where the bulk of the samples were taken, exhibited low species diversity with the amount of species ranging from 12 to 29. In addition to this, the species encountered in the Wasgoedspruit and Mooi River often belonged to genera comprising a relatively large number of species, e.g. *Nitzschia* and *Navicula*. There would thus be a lower amount of response perceived when using this index, resulting in weaker correlations between GDI and measured environmental variable values.

Table 4.3 Pearson correlations coefficients between diatom indices and concurrent environmental variable values in the Wasgoedspruit and Mooi River. Coloured values indicate significant correlations at $p < 0.05$. Abbreviations for the indices are given in Appendix C.

	SPI	SLA	DES	LMI	SHE	WAT	TDI	EPI	ROT	GDI	CEC	BDI	APDI
DO%	-0.21	-0.43	-0.33	-0.40	-0.36	-0.51	0.38	-0.30	-0.46	0.13	-0.21	-0.13	-0.22
EC	-0.19	-0.16	-0.04	-0.22	-0.18	-0.31	0.29	-0.04	-0.16	-0.11	0.03	-0.46	-0.21
pH	-0.21	-0.50	-0.31	-0.39	-0.19	-0.42	0.46	-0.38	-0.39	0.02	-0.44	-0.22	-0.22
Turb	0.30	0.16	0.26	0.28	0.38	0.24	-0.20	0.15	0.31	0.27	0.02	0.26	0.25
NO_3	-0 10	-0.14	0.06	-0.11	-0.11	-0.14	0.24	-0.08	-0 11	0.24	0.00	-0.17	-0.27
SO_4	-0.68	-0.75	-0.65	-0.79	-0.74	-0.80	0.83	-0.65	-0.79	-0.30	-0.55	-0.69	-0.75
Al	-0.53	-0.68	-0.60	-0.66	-0.60	-0.68	0.64	-0.66	-0.69	-0.14	-0.58	-0.31	-0.55
As	-0.76	-0.86	-0.87	-0.89	-0.82	-0.83	0.78	-0.84	-0.92	-0.34	-0.77	-0.47	-0.71
Cd	-0.67	-0.65	-0.75	-0.71	-0.69	-0.71	0.66	-0.74	-0.75	-0.22	-0.69	-0.32	-0.59
Cr	-0.75	-0.86	-0.85	-0.85	-0.78	-0.83	0.80	-0.87	-0.91	-0.26	-0.83	-0.41	-0.71
Cu	-0.52	-0.76	-0.66	-0.72	-0.60	-0.68	0.64	-0.69	-0.73	-0.18	-0.67	-0.29	-0.49
Hg	-0.59	-0.72	-0.66	-0.71	-0.62	-0.70	0.64	-0.70	-0.73	-0.22	-0.61	-0.40	-0.61
Mn	-0.51	-0.73	-0.73	-0.67	-0.54	-0.56	0.56	-0.69	-0.72	-0.18	-0.78	-0.17	-0.45
Pb	-0.69	-0.86	-0.79	-0.84	-0.72	-0.79	0.76	-0.83	-0.86	-0.25	-0.78	-0.42	-0.67
Se	-0.59	-0.75	-0.78	-0.74	-0.63	-0.72	0.61	-0.70	-0.79	-0.19	-0.74	-0.26	-0.49
Zn	-0.72	-0.83	-0.83	-0.83	-0.74	-0.79	0.75	-0.85	-0.86	-0.30	-0.80	-0.40	-0.68

From Table 4.3 it can also be seen that almost all of the indices showed weak, insignificant correlations with electrical conductivity (EC), while there were no significant correlations between any of the diatom indices and nitrate (NO_3-N). Despite the fact that high values for these environmental variables were encountered in the Wasgoedspruit and the Mooi River, the diatom indices did not seem to primarily reflect the changes in EC and nitrate concentrations, environmental variables used often in the routine water quality monitoring. Stronger, significant negative correlations existed between the diatom indices and DO%, pH, SO_4-S and trace metal concentrations. These environmental variables were consistently high (Chapter 3). The diatom community that would develop in response to these conditions would therefore result in index values consistent with the prevailing water quality.

4.5.2 Correlation between diatom indices and averages of environmental variable values

Correlation analysis was done between diatom indices and the average environmental data one month prior to diatom sampling in order to determine whether the indices derived from the diatom community composition would reflect the average water quality conditions in the

Wasgoedspruit and the Mooi River that prevailed in the period before biological sampling took place.

The results of a Pearson correlation between diatom indices and the average of the environmental variable values are shown in Table 4.4. From Table 4.4 it can be seen that significant negative correlations existed between the majority of diatom indices used in this study and the average environmental data one month prior to diatom sampling. Similar to the results of the correlations between the diatom indices and concurrent data, the Generic Diatom Index (GDI; Coste and Ayphassorho, 1991) was again the exception, only showing a significant negative correlation with arsenic (As) and the explanation may be assumed to be the same as in section 4.5.1.

Table 4.4 Pearson correlations coefficients between averages of environmental variable values and diatom indices in the Wasgoedspruit and Mooi River. Coloured values indicate significant correlations at $\rho < 0.05$. Abbreviations for the indices are given in Appendix C.

	SPI	SLA	DES	LMI	SHE	WAT	TDI	EPI	ROT	GDI	CEC	BDI	APDI
DO%	-0.37	-0.69	-0.50	-0.59	-0.48	-0.72	0.57	-0.58	-0.64	0.04	-0.43	-0.17	-0.35
EC	-0.46	-0.22	-0.22	-0.34	-0.45	-0.45	0.46	-0.27	-0.35	-0.37	-0.04	-0.62	-0.45
pH	-0.30	-0.60	-0.37	-0.47	-0.31	-0.56	0.56	-0.56	-0.49	0.01	-0.43	-0.09	-0.32
Turb	0.43	0.02	0.18	0.24	0.43	0.23	-0.23	0.13	0.23	0.39	-0.19	0.69	0.43
NO_3	-0.44	-0.44	-0.20	-0.38	-0.26	-0.44	0.47	-0.38	-0.34	-0.21	-0.42	-0.47	-0.35
SO_4	-0.82	-0.81	-0.76	-0.86	-0.82	-0.91	0.88	-0.82	-0.87	-0.47	-0.61	-0.66	-0.76
Al	-0.65	-0.74	-0.75	-0.77	-0.61	-0.72	0.68	-0.71	-0.74	-0.41	-0.67	-0.38	-0.59
As	-0.78	-0.91	-0.91	-0.92	-0.80	-0.87	0.80	-0.87	-0.94	-0.36	-0.80	-0.48	-0.69
Cd	-0.62	-0.76	-0.86	-0.78	-0.68	-0.68	0.64	-0.73	-0.78	-0.32	-0.72	-0.35	-0.56
Cr	-0.69	-0.92	-0.88	-0.88	-0.73	-0.83	0.77	-0.86	-0.90	-0.29	-0.82	-0.34	-0.61
Cu	-0.59	-0.85	-0.73	-0.78	-0.64	-0.81	0.72	-0.80	-0.80	-0.20	-0.70	-0.23	-0.54
Hg	-0.57	-0.82	-0.67	-0.78	-0.59	-0.76	0.71	-0.77	-0.74	-0.14	-0.62	-0.37	-0.61
Mn	-0.49	-0.77	-0.77	-0.71	-0.51	-0.63	0.56	-0.68	-0.71	-0.16	-0.83	-0.16	-0.40
Pb	-0.70	-0.92	-0.83	-0.87	-0.70	-0.85	0.77	-0.87	-0.87	-0.27	-0.83	-0.39	-0.63
Se	-0.67	-0.89	-0.88	-0.88	-0.70	-0.82	0.73	-0.82	-0.88	-0.24	-0.79	-0.37	-0.59
Zn	-0.69	-0.91	-0.87	-0.88	-0.71	-0.83	0.76	-0.87	-0.87	-0.31	-0.81	-0.35	-0.65

Table 4.4 shows a similar situation to that observed in the previous section, with weak, insignificant correlations existing between EC, turbidity and nitrate (NO_3-N), while significant negative correlations existed between the majority of the diatom indices and DO%, pH, SO_4-S and trace metal concentrations. The correlations between the diatom indices and the average environmental data were stronger than those calculated between the diatom indices and the environmental data measured concurrently. These stronger correlations of the diatom indices with the average environmental data of the month preceding diatom sampling, show the value of diatoms as a bio-monitoring tool in that it suggests that diatom communities (and the indices derived from the community composition) are representative of

the prevailing conditions at a particular site over a longer period (Harding *et al.*, 2005 & Taylor, 2004).

4.5.3 Correlation between diatom indices and maximum environmental variable values

Correlation analysis was done between diatom indices and the maximum values measured for environmental variables in the month prior to diatom sampling to investigate whether the diatom indices would be able to reflect significant changes/fluctuations in environmental variables that may have occurred in the period prior to diatom sampling.

The results of a Pearson correlation between diatom indices and maximum environmental variable values measured in the month prior to diatom sampling are shown in Table 4.5. Table 4.5 shows that the same significant correlations exist between the various diatom indices and the environmental variables as that of Table 4.3 and Table 4.4. Once again it would appear that the strongest correlations exist between the diatom indices and DO%, pH, SO_4-S and trace metal concentrations.

Table 4.5 Pearson correlations coefficients between diatom indices and maximum environmental variable values in the Wasgoedspruit and Mooi River. Coloured values indicate significant correlations at $p < 0.05$. Abbreviations for the indices are given in Appendix C.

	SPI	SLA	DES	LMI	SHE	WAT	TDI	EPI	ROT	GDI	CEC	BDI	APDI
DO%	-0.29	-0.63	-0.43	-0.50	-0.38	-0.64	0.52	-0.47	-0.55	0.09	-0.45	-0.10	-0.26
EC	-0.50	-0.25	-0.26	-0.38	-0.50	-0.48	0.50	-0.30	-0.39	-0.38	-0.01	-0.68	-0.52
pH	-0.22	-0.63	-0.37	-0.46	-0.21	-0.52	0.53	-0.48	-0.47	0.15	-0.49	-0.06	-0.20
Turb	0.39	-0.02	0.14	0.20	0.39	0.18	-0.18	0.10	0.20	0.39	-0.22	0.63	0.38
NO_3	-0.44	-0.44	-0.21	-0.40	-0.27	-0.45	0.47	-0.38	-0.35	-0.18	-0.41	-0.50	-0.35
SO_4	-0.83	-0.82	-0.78	-0.87	-0.83	-0.92	0.89	-0.82	-0.89	-0.48	-0.64	-0.67	-0.78
Al	-0.63	-0.72	-0.72	-0.75	-0.57	-0.70	0.67	-0.67	-0.72	-0.38	-0.64	-0.39	-0.58
As	-0.75	-0.88	-0.88	-0.90	-0.77	-0.86	0.77	-0.84	-0.91	-0.34	-0.77	-0.47	-0.67
Cd	-0.53	-0.66	-0.79	-0.69	-0.59	-0.55	0.53	-0.63	-0.68	-0.28	-0.61	-0.31	-0.50
Cr	-0.69	-0.92	-0.87	-0.88	-0.72	-0.82	0.77	-0.87	-0.89	-0.28	-0.82	-0.35	-0.62
Cu	-0.55	-0.82	-0.70	-0.74	-0.59	-0.76	0.68	-0.76	-0.75	-0.19	-0.67	-0.18	-0.52
Hg	-0.53	-0.76	-0.61	-0.72	-0.51	-0.67	0.63	-0.71	-0.66	-0.14	-0.61	-0.37	-0.59
Mn	-0.45	-0.74	-0.73	-0.66	-0.45	-0.59	0.51	-0.64	-0.66	-0.12	-0.81	-0.12	-0.36
Pb	-0.69	-0.93	-0.81	-0.86	-0.68	-0.84	0.76	-0.87	-0.85	-0.25	-0.80	-0.39	-0.64
Se	-0.61	-0.86	-0.82	-0.83	-0.62	-0.78	0.68	-0.79	-0.82	-0.19	-0.79	-0.33	-0.55
Zn	-0.68	-0.88	-0.84	-0.86	-0.70	-0.81	0.75	-0.85	-0.85	-0.31	-0.77	-0.34	-0.65

From Table 4.5 it can also be seen that correlation coefficients between the various diatom indices and the maximum environmental variable values show a great degree of similarity to those calculated between the diatom indices and average environmental variable values, with slight increases and decreases present. This similarity between the correlation coefficients of the diatom indices with the average and maximum environmental variable

values can almost certainly be explained by the consistently high environmental variable values encountered in the Wasgoedspruit and Mooi River. These consistently high values would mean that the diatom community would not have significantly changed during times of "peaks" in environmental variable values, and therefore the diatom indices would have remained relatively unchanged.

4.6 Relationship between diatom species/indices and trace metals

Trace metals often fulfil functions as essential micronutrients to microflora. For example, iron and manganese are functional components of nitrate assimilation in the Hill reaction of photosynthesis and iron is essential component in the pathways of chlorophyll and protein synthesis (Wetzel, 1983). However, trace metals in elevated concentrations are known to detrimentally affect aquatic organisms, often by being toxic (DWAF, 1996). Examples of this include the inhibition of pigment transformations in the diatom cycle by cadmium (Bertrand et al., 2001), the interference of arsenic with energy metabolism and the activity of a variety of essential enzymes (DWAF, 1996), the genotoxic and mutagenic properties of chromium, well as its high rate of accumulation in algae (Dallas and Day, 2004), to mention but a few.

The diatom indices showed strong negative correlations to the measured environmental variables as discussed above. However, the strongest correlations were shown between the diatom indices and the trace metals measured for in this study. These variables are often neglected during routine monitoring of aquatic ecosystems. Referring to Figure 4.1, it can be observed that the dominant diatom species identified in group 2 were located opposite to the trace metals during canonical correspondence analysis. According to Figure 4.1, the group 3 species were more closely associated with, amongst others, the trace metal concentrations, showing that the diatom community structure was influenced to a high degree by the concentration levels of trace metals in solution.

Trace metals and their influence on diatom communities have been the focus of several studies (e.g. Rachlin *et al.*, 1983; Morin *et al.*, 2007), but these studies have seldom taken into account the influence of a wide variety of trace metals on the diatom communities or the resulting indices. The focus of these studies has often been on trace metals that were known to have high concentrations in the particular area of study and cause toxicological effects. Diatoms have been shown to have relatively high concentration factors for Cd, Hg and Pb, indicating they were good accumulators for these metals (Tien, 2004). Chromium has also been found to accumulate in algae (Dallas and Day. 2004). Diatoms have also been shown to be sensitive to changing metal (Cd and Zn) concentrations (Gold *et al.*, 2002). For

this reason these organisms are suitable indicators of contamination with such metals (Tien, 2004) an argument that is supported in light of the above results.

4.7 Diatom indices

4.7.1 Diatom index scores in the Wasgoedspruit and Mooi River

The diatom indices used to assess water quality in this study have already been discussed in Chapter 1 and Chapter 2. As stated earlier, the value generated by a diatom index is a result of the mean of the water quality optima of the diatom taxa, weighted by the abundance of each taxon (Descy, 1979). For the majority of the diatom indices used in this study, the index score is given as a value on a scale between 0 and 20, with 0 representing the lowest water quality and 20 the highest water quality. In the case of the TDI of Kelly and Whitton (1995) the index scores increase with an increase in the trophic level of the water (positive correlations in above correlation tables).

The results of the different indices that were calculated from the diatom community composition by the OMNIDIA software package, are shown in Figure 4.3 and Figure 4.4 below. All the diatom indices displayed in the below figures showed the same trend: a decrease in the diatom index scores between WS 0 and WS 1 and a slight increase in diatom index scores between WS 2 and WS 3 (with a converse relationship between TDI and the environmental variable values). These changes in diatom index scores are in agreement with the results discussed in Chapter 3 where there was an increase in the observed environmental variable values between WS 0 and WS 1 and a slight decrease between WS 2 and WS 3.

Figure 4.3 and Figure 4.4 also show a decrease in diatom index score values between MR 1 and MR 2. This also follows the trend of the environmental variable values, where the general trend was to increase in value from MR 1 to MR 2 located downstream of the confluence of the Wasgoedspruit with the Mooi River.

Eloranta and Soininen (2002) distinguished certain water quality and trophic classes based on the SPI scores (Coste in Cemagref, 1982) and TDI respectively. The following classes were identified:

high quality / oligotrophy	(SPI >17 / TDI <7);
good quality / oligo-mesotrophy	(SPI 15–17 / TDI 7–10);
moderate quality / mesotrophy	(SPI 12–15 / TDI 10–13);
poor quality / meso-eutrophy	(SPI 9–12 / TDI 13–16);
bad quality / eutrophy	(SPI <9 / TDI >16).

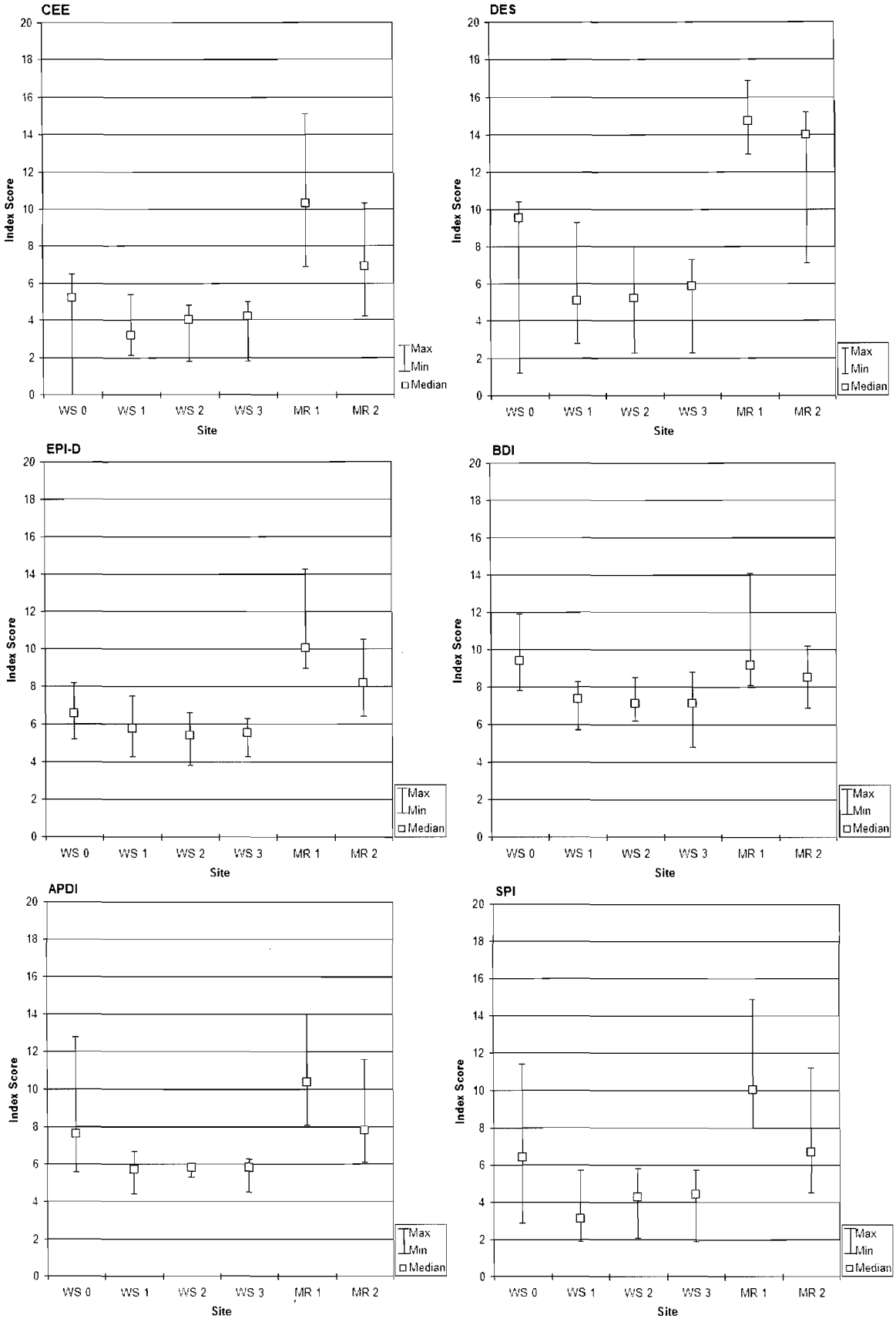

Figure 4.3 Median diatom index scores at the different sites in the Wasgoedspruit and Mooi River for the period February 2005 to January 2006.

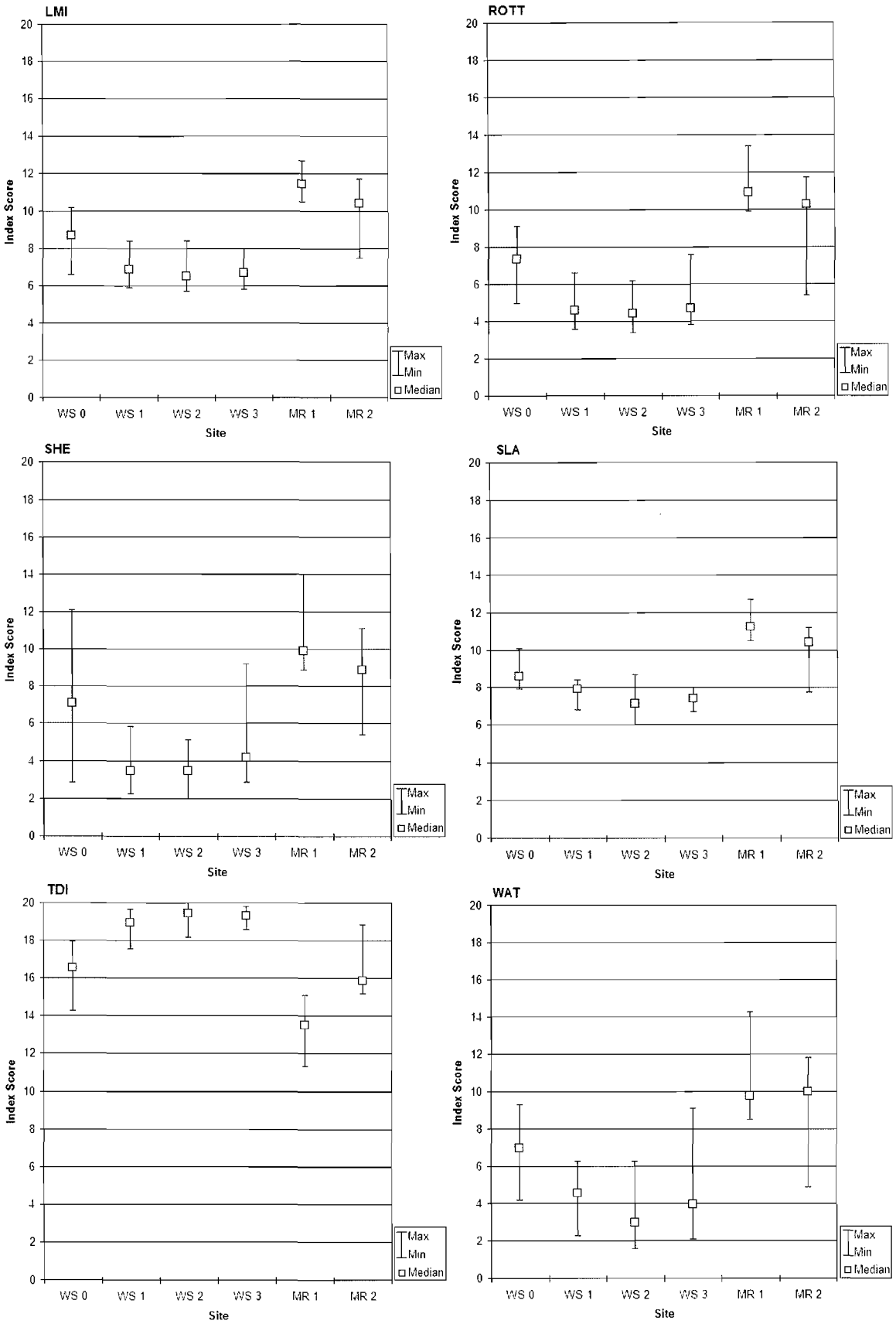

Figure 4.4 Median diatom index scores at the different sites in the Wasgoedspruit and Mooi River for the period February 2005 to January 2006.

When this classification is applied to the results of the diatom index scores from the different sites in the Wasgoedspruit and Mooi River (Figure 4.5), it became clear the water of the Wasgoedspruit was of very poor water quality and also highly eutrophic. The figures also show that the water quality of the Mooi River decreases from poor (above the confluence with the Wasgoedspruit) to poor (below the confluence with the Wasgoedspruit), while the trophic state of the water also changes from a low meso-eutrophy to a high meso-eutrophy above and below the confluence with the Mooi River respectively.

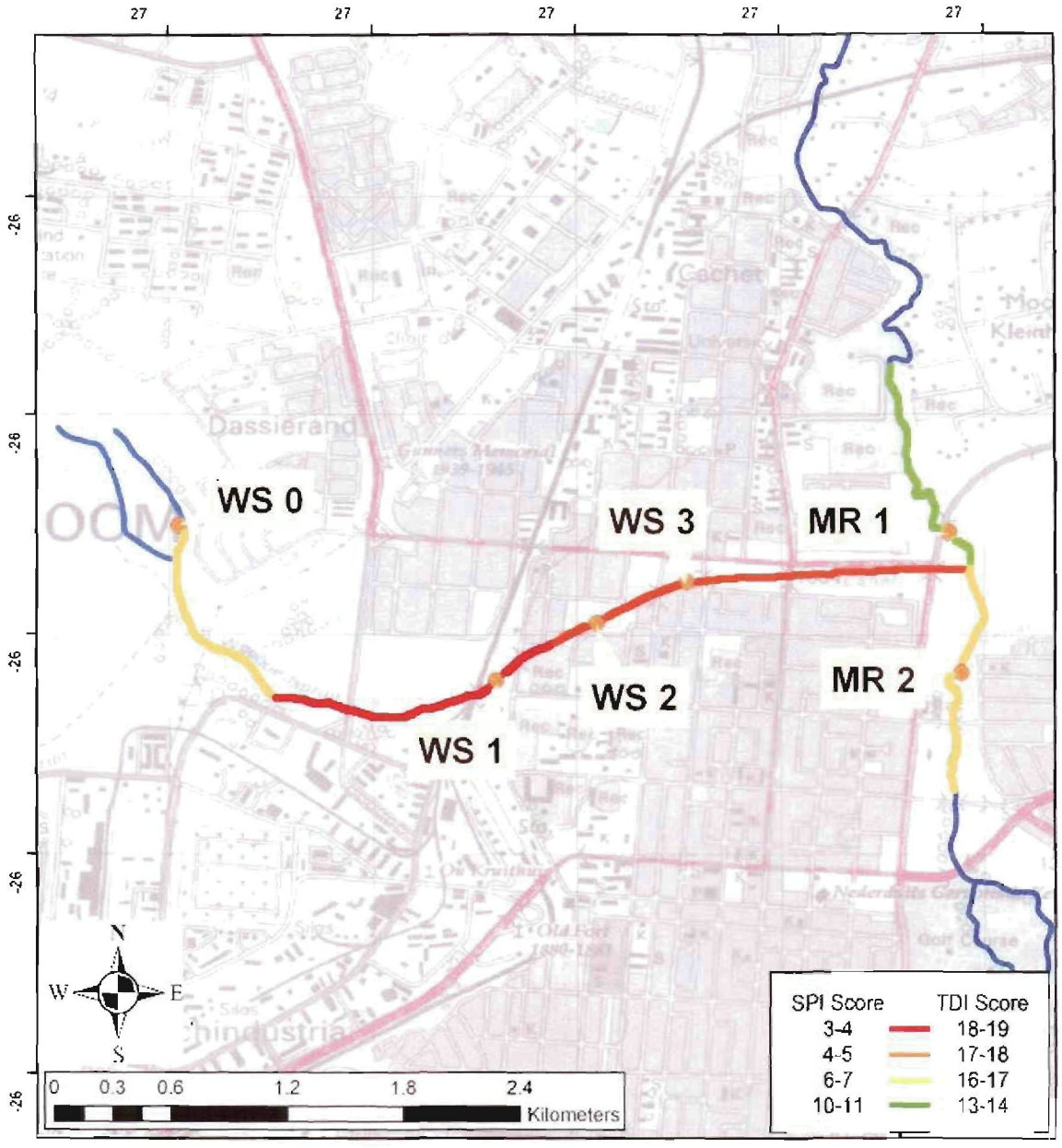

Figure 4.5 Map showing the diatom inferred water quality of sites sampled in the Wasgoedspruit and Mooi River

Figure 4.6 shows the similarity of the sites according to the results of the indices. Referring to the description and location of the sites as shown in Chapter 2, at first glance it would be

expected that the sites that would group together would be those that are canalised (WS 1, WS 2 & WS 3) vs. those that occur in natural river beds (WS 0, MR 1 & MR 2). This arrangement would be expected based not only on their physical characteristics but also the quality of the water flowing through the particular section of stream.

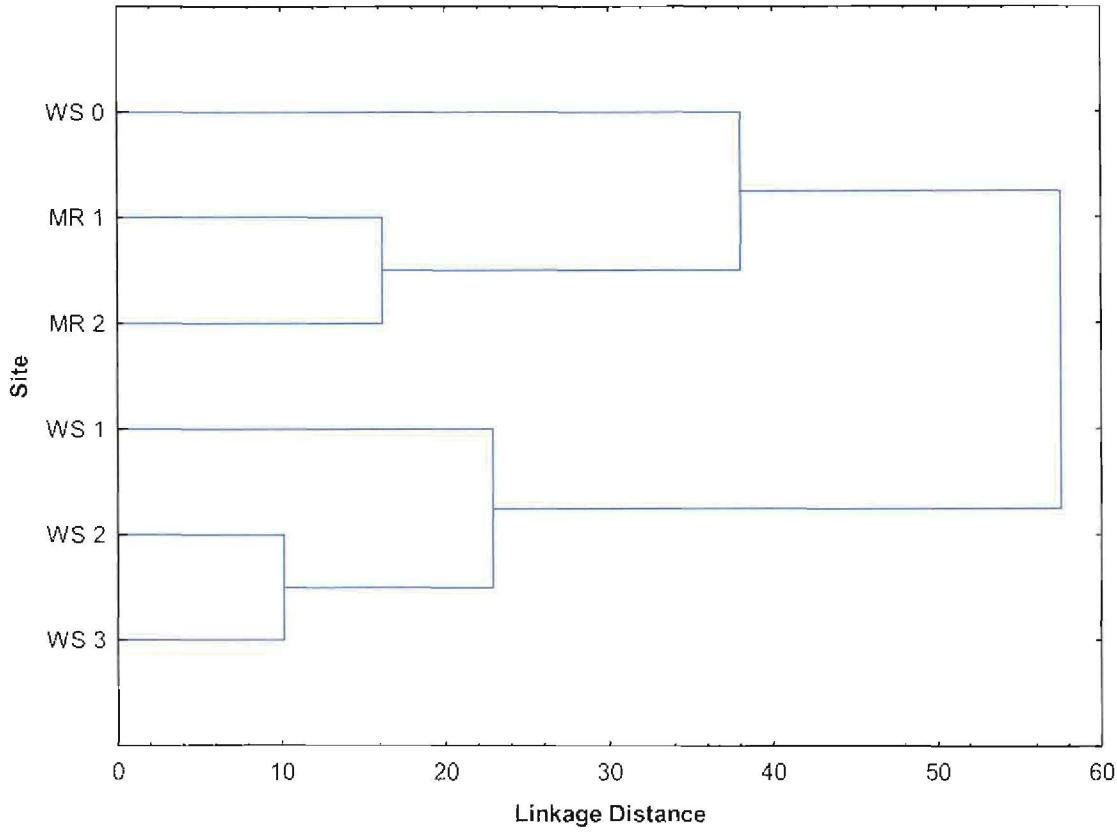

Figure 4.6 Cluster analysis showing the similarity between the sites according to diatom scores. The analysis was done using Ward's method and Euclidian distances.

As would be expected, when the diatom scores are grouped using cluster analysis, this grouping of the sites is clearly shown. From Figure 4.6 it can be seen that according to the diatom index scores, the three sites occurring in natural river beds are similar with the two sites in the Mooi River showing the greatest similarity. This agrees with what is expected from the environmental variable values for the particular sites. The nutrient and dissolved oxygen (DO) levels were lower at these sites occurring in the natural river beds. However, there were some differences in the electrical conductivity (EC) between the Mooi River and the Wasgoedspruit (refer to Chapter 3). It was also evident that the Wasgoedspruit site had lower pH conditions than those occurring in the Mooi River.

Figure 4.6 also shows that according to the diatom index scores the sites in the canalised section of the Wasgoedspruit are clustered together and here there is a greater similarity

between the two downstream sites, WS 2 & WS 3. From the environmental variable values, as shown in Chapter 3, this is to be expected. Again, the nutrient levels were consistent, and the changes in the environmental variable values were confined largely to differences in pH, dissolved oxygen and temperature. These differences between the environmental variable values at WS 1 and those and WS 2 and WS 3, can almost certainly be ascribed to the transition from the natural stream bed to the canalised section of the Wasgoedspruit. It can be seen that the diatom communities and the resultant diatom index score values derived from the community composition, clearly reflect the changes in general water quality and trophic state of the water in the Wasgoedspruit and Mooi River.

4.7.2 Comparison of index scores from epilithic and epiphytic samples collected from the Wasgoedspruit.

Due to the shallow nature of the Wasgoedspruit, the availability of a suitable substratum for the collection of diatoms was often restricted, despite the abundance of diatoms on the canal surface. Therefore, any available substratum on which diatoms could establish successfully would be valuable as a source of diatom material for the determination of water quality. The Wasgoedspruit formed pools in certain sections and this in turn lead to the formation of sand banks and the development of plant communities dominated by *Persicaria* spp., as well as a variety of grasses. This was especially true at WS 3 where the grass leaves and *Persicaria* stems were permanently submerged in the shallow stream. Diatoms samples were collected from epilithic and epiphytic substrata at this site in order to investigate whether the results obtained from the different substrata would differ, and whether these substrata could be used interchangeably in the this environment.

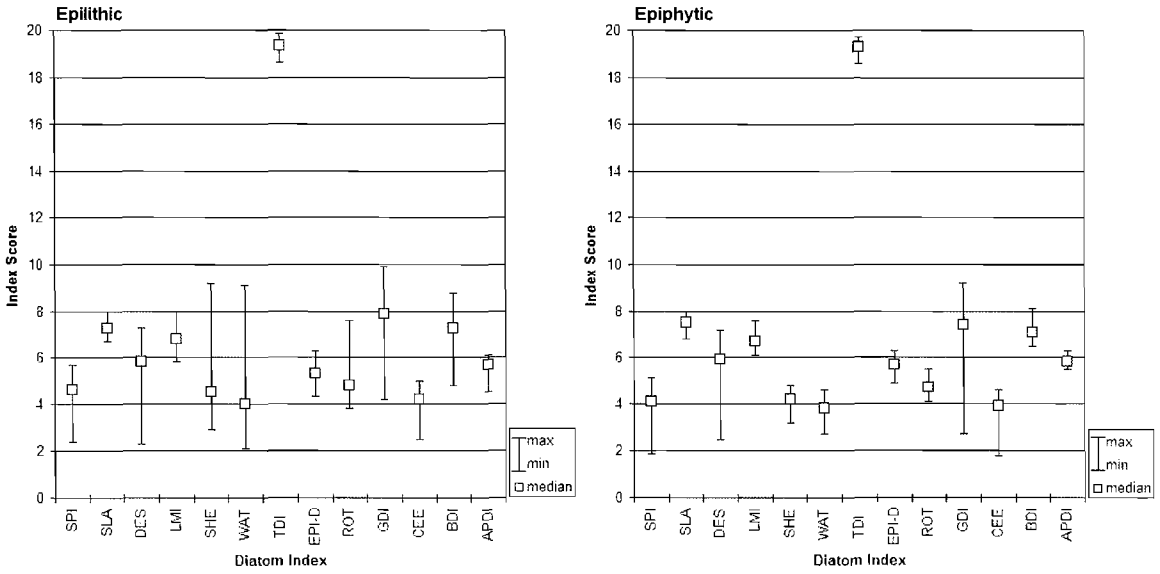

Figure 4.7 Median diatom index scores for the different indices at WS 3 for the period February 2005 to January 2006.

The results of the diatom index calculations from epilithic and epiphytic samples for the different indices used in this study are shown in Figure 4.7. As can be seen from Figure 4.7, there is a high degree of similarity between the results obtained from epilithic samples and those obtained from epiphytic samples, with the only major difference between the two results being the difference in minimum and maximum values for the SHE and WAT.

Townsend and Gell (2005) reported similar results in their study of the role of substrate type on benthic diatom assemblages in the wet/dry tropics of Australia, in rivers that had been relatively undisturbed by human activity, included riffle or river run habitats and had low benthic sample depths (<50cm). They concluded that there was no clear evidence of substrate specific diatoms, the only exception being *Psammothidium*, with only the abundance of common species differing between some substrata. Similarly, Rott et al. (1998) found that results obtained from different substrata (grass and stone samples) were very similar and concluded that water quality determination using diatoms is independent of the substratum used. From the above it can be concluded that the results obtained from epilithic and epiphytic samples, in terms of diatom index scores generated, are similar. And in turn it can then be concluded that these two substrata could be used interchangeably in the canalised section of the Wasgoedspruit.

4.8 Summary

Thirty seven diatom taxa were found to be dominant in the Wasgoedspruit and Mooi River and of these, 89% are usually indicative of circumneutral to alkalophilous waters, 30% of the dominant taxa are nitrogen autotrophic, 43% are facultatively/ obligately nitrogen-heterotrophic, and 70% of the dominant taxa are found occurring in meso-eutraphentic to hypereutraphentic waters. In light of the environmental variable values discussed in Chapter 3, the dominant diatom taxa gave a good indication of the prevailing water quality conditions in the Wasgoedspruit and Mooi River. The following paragraphs provide a summary of the main findings in this chapter supporting this statement.

Canonical correspondence analysis of the environmental data revealed three distinct groupings of diatom species in the Wasgoedspruit and Mooi River. The first group (Group 1 in the Figure 4.2) consisted mainly of diatom species that are common to streams with elevated pollution and the main environmental variables that influenced their presence were found to be high nutrient levels (NO_3-N, SO_4-S) and trace metals. The second group consisted mainly of species normally indicative of slightly better water quality, but that still occur in eutrophic waters and that can tolerate elevated concentrations of organically bound

nitrogen. The main drivers were lower nutrient and trace metal concentrations, as wells as lower pH and dissolved oxygen. The third group consisted of species abundant in the un-canalised section of the Wasgoedspruit. These species, however, often have a wide tolerance range. They were also not necessarily indicative of good water quality as would be expected at this upper stream site of the Wasgoedspruit.

Correlation analysis was done in order to calculate the efficacy with which a selection of the diatom indices indicated changes in the water quality of the studied waterbodies. Correlations of diatom indices with concurrent environmental variable values showed weak correlations with electrical conductivity and NO_3-N. Stronger, significant negative correlations existed between the diatom indices and DO%, pH, SO_4-S and trace metal concentrations, suggesting that the diatom community that would develop in response to these variables would therefore result in index values consistent with the prevailing water quality. Correlations of diatom indices with average environmental variable values yielded stronger correlations than those calculated between the diatom indices and the environmental data measured concurrently. These stronger correlations of the diatom indices with the average environmental data of the month preceding diatom sampling, highlighted the value of diatoms as a bio-monitoring tool in that it suggests that diatom communities (and the indices derived from the community composition) are representative of the prevailing conditions at a particular site over a longer period. The similarity between the correlation coefficients of the diatom indices with the average and maximum environmental variable values was attributed to the consistently high environmental variable values encountered in the Wasgoedspruit and Mooi River.

A noteworthy relationship that was identified in this chapter was that between diatom indices and trace metal concentrations. In fact, the strongest correlations were shown between the diatom indices and the trace metals measured for in this study, showing that the diatom community structure was influenced to a high degree by the concentration levels of trace metals in solution. It was concluded that diatoms are suitable indicators of contamination with such metals.

Diatom indices also followed the trends of the environmental variables as increases in variable values resulted in decreases in diatom index scores at the specific sites in the Wasgoedspruit and Mooi River. The diatom index scores also confirmed that the water of the Wasgoedspruit was of very bad water quality and also highly eutrophic and also showed that the water quality of the Mooi River decreased from poor to bad below the confluence

with the Wasgoedspruit, while the trophic state of the water also changed from low meso-eutrophy to high meso-eutrophy.

When the diatom scores for the particular sites were grouped using cluster analysis the sites in the canalised section of the Wasgoedspruit were clustered together and the cluster analysis of diatom index scores also showed a greater similarity between the two downstream sites, WS 2 and WS 3. The three sites occurring in natural river beds were also clustered together, with the two Mooi River sites showing the greatest similarity according to diatom index scores. The diatom communities and the resultant diatom index score values derived from the community composition, reflected the changes in general water quality and trophic state of the water in the Wasgoedspruit and Mooi River.

The difference between the diatom index results of epilithic and epiphytic samples was also investigated. It was found that the results obtained from epilithic samples and those obtained from epiphytic samples were similar. From this it was concluded that these two substrata could be used interchangeably in the canalised section of the Wasgoedspruit.

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5 CONCLUSIONS AND RECOMMENDATIONS

"The project of canalizing Wasgoedspruit, left an indelible imprint in the minds of the inhabitants, and emphasized the value of preserving the natural heritage within the built-up areas."

Bawcombe (1988)

The preceding chapters gave a description and discussion of the results of the physical and chemical measurements of the Wasgoedspruit and Mooi River (Chapter 3), as well as the relationship between measured environmental variables and the diatom community composition and the indices derived from the diatom community composition. Several conclusions can be drawn from these discussions and these will be presented below.

Chapter 3 highlighted the unpredictable nature of the urban aquatic environment in that the extremes of the physical and chemical environmental variables were clearly shown. Environmental variables displayed high values and/or a high degree of variability in the canalised section of the Wasgoedspruit and these values were much higher in the Wasgoedspruit than the Mooi River. The influence of the elevated concentrations of environmental variables of the Wasgoedspruit on of the Mooi River was also visible, especially in the case of pH, electrical conductivity, nitrate and ammonium.

The physical environmental variables of the Wasgoedspruit did not seem to have a great impact on the Mooi River. The chemical environmental variables in the Wasgoedspruit all had values much higher than the target water quality range (TWQR) as recommended in the National Water Quality Guidelines set by the Department of Water Affairs and Forestry (DWAF, 1996). It was also observed that the chemical environmental variables showed a high degree of variability. The majority of the trace metals measured had median concentrations in excess of guideline concentrations

In Chapter 4 illustrated that diatoms could be used as effective indicators of water quality in the urban environment of Potchefstroom. In light of the environmental variable values discussed in Chapter 3, the dominant diatom taxa gave a good indication of the prevailing water quality conditions in the Wasgoedspruit and Mooi River. Canonical correspondence analysis confirmed this point by clearly distinguishing three groups of diatom species in the Wasgoedspruit and Mooi River, each of these groups showing clear preferences towards

certain environmental variable conditions that were consistent with the ecological preferences of these species.

Correlation analysis showed that stronger correlations existed between the diatom indices and DO%, pH, SO_4-S and trace metal concentrations, suggesting that the diatom community that would develop in response to these variables would therefore result in low index values consistent with the prevailing water quality. Correlations of diatom indices with average and maximum environmental variable values in the month prior to sampling yielded stronger correlations than those calculated between the diatom indices and the environmental data measured concurrently, which highlighted the value of diatoms as a bio-monitoring tool in that it suggests that diatom communities (and the indices derived from the community composition) are representative of the prevailing conditions at a particular site over a longer period. From the results of the correlation analysis it can also be concluded that diatoms are suitable indicators of contamination with trace metals, as it was observed that correlations between the diatom indices and the trace metals were the strongest of all correlations between environmental variables and diatom indices. The diatom community structure was, therefore, influenced to a high degree by the concentration levels of trace metals in solution.

Diatom indices also followed the trends of the environmental variables as increases in variable values resulted in decreases in diatom index scores at the specific sites in the Wasgoedspruit and Mooi River and the influence of the low water quality of the Wasgoedspruit on the Mooi River was evident from the resultant drop in diatom index scores above and below stream of the confluence of these two streams. The diatom indices also better indicated the influence of the Wasgoedspruit on the Mooi River than measurement of physio-chemical variables. Cluster analysis revealed that the diatom communities and the resultant diatom index score values derived from the community composition, reflected the changes in general water quality and trophic state of the water in the Wasgoedspruit and Mooi River, as sites expected to have similar water quality were grouped together according to the diatom index scores.

From comparisons drawn between the diatom index scores generated from concurrent epilithic and epiphytic samples it can be concluded that these two substrata could be used interchangeably in the canalised section of the Wasgoedspruit. This would increase the chances of successful reflection of the water quality, by diatoms, which, in the urban canals can in some circumstances often be restricted, despite the abundance of diatoms on the canal surface.

Although this study was conducted on a relatively small scale, the high frequency of sampling and measuring of environmental variables interrogated the results obtained and showed that the diatom communities in the Wasgoedspruit were impacted by the conditions prevailing in this canalised urban stream. The European and other diatom based indices were also successful in reflecting the overall (Figures 4.3 and 4.4) and short term (Tables 4.3, 4.4 and 4.5) trends in environmental variable values. The diatom community composition (Figures 4.1 and 4.2) and the diatom indices calculated from them (Figures 4.3, 4.4 and 4.5) successfully showed the influence of the small urban tributary Wasgoedspruit on the relatively larger Mooi River. It can thus be said that the objectives of this study as set out in section 1.2 have been achieved.

However, looking beyond this study, it is recommended that the efficacy with which diatom indices are able to accurately reflect water quality be tested on a larger scale in South Africa, such as large urban centres of Guateng and other cities such as Cape Town, Durban, and Port Elizabeth. This distribution of cities would allow for the testing of indices in different climatic and geographical regions, where the quality of the aquatic environments may differ as a result of the influence of vegetation, flow rates, environmental variable concentration levels, etc. These urban areas would also subject the use of diatom indices as a measure of water quality to different anthropogenic impacts on the aquatic environments, since these cities are characterised by large industrial, residential and commercial areas. These areas have impacts inherent to them, and the response of the diatom communities to the cumulative effects of these areas and their impacts may provide a more holistic picture of the suitability of diatoms for use in urban environments.

This study has shown that bio-assessment, especially with diatoms, is feasible in urban environments. By assessing the condition of our urban aquatic environments, it is hoped that the preservation of *"...the natural heritage within the built-up areas"* can be achieved with greater success.

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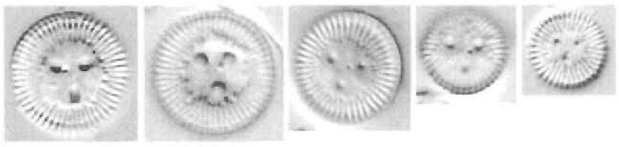
Appendix A

Taxonomic Plates

Centrics



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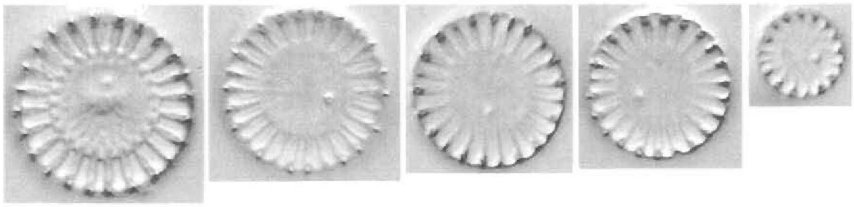

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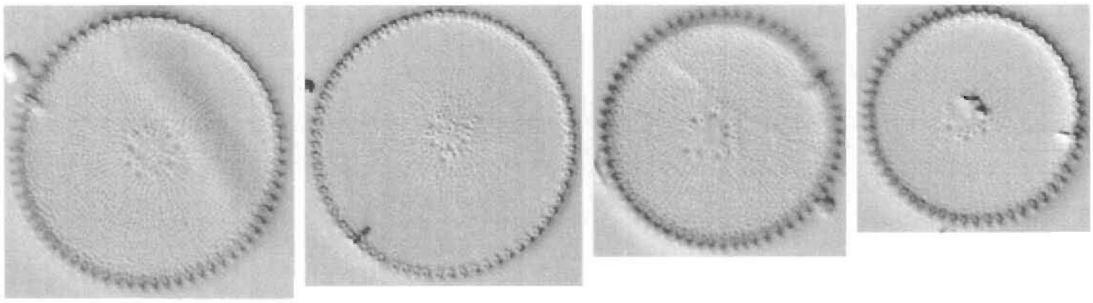
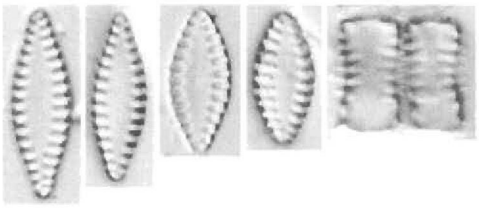


11 12 13 14

Araphideae



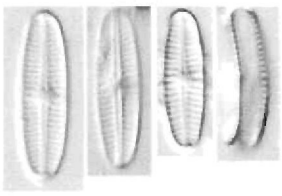
15 16 17 18 19

10 µm

1-5 *Cyclotella ocellata Pantocsek*
6-10 *Cyclotella meneghiniana Kützing*
11-14 *Thalassiosira weissflogii* (Grunow) Fryxell & Hasle
15-19 *Fragilaria brevistriata* Grunow

PLATE 1

Monoraphideae



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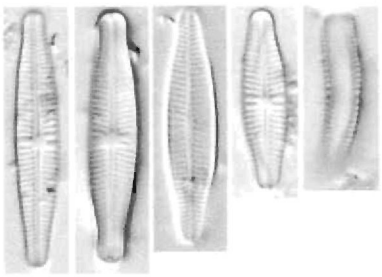
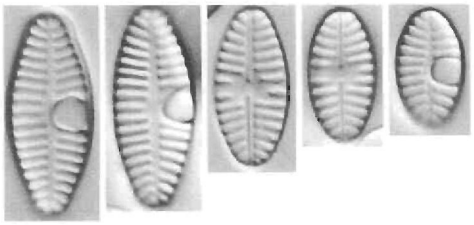
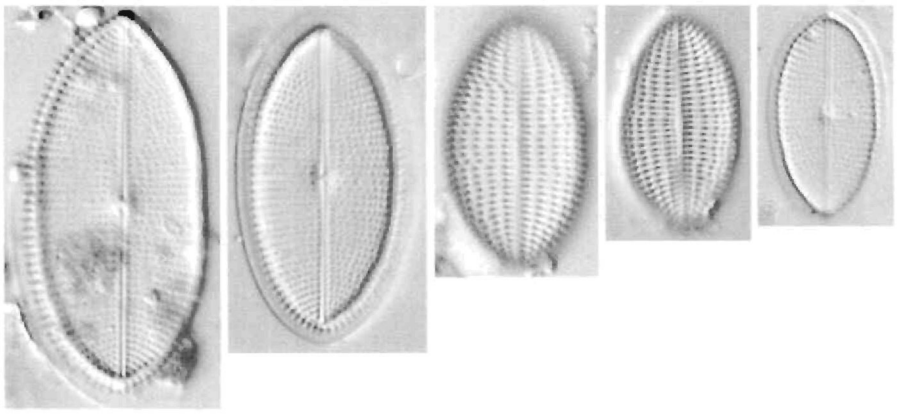


24 25 26 27 28



29 30 31 32 33



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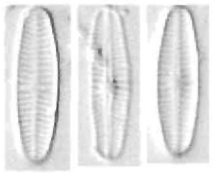
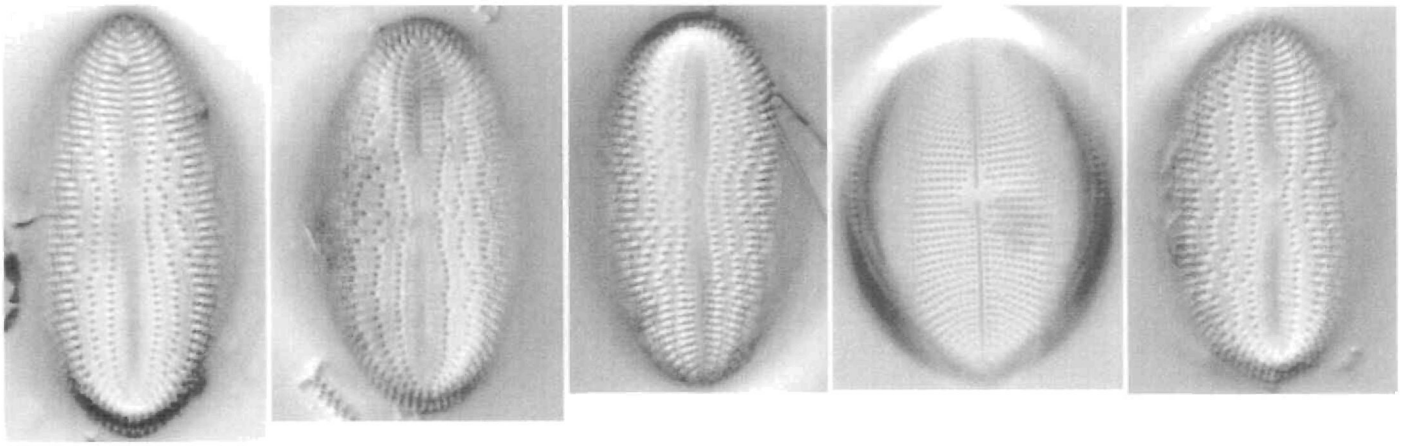


39 40 41



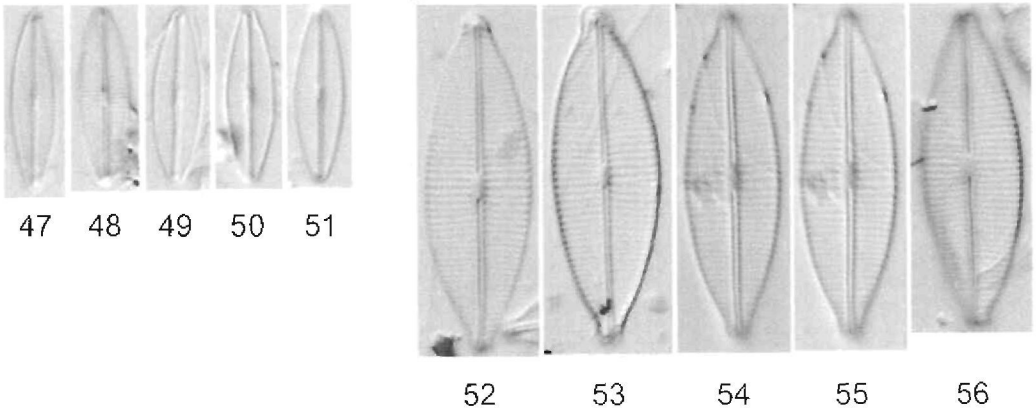
42 43 44 45 46

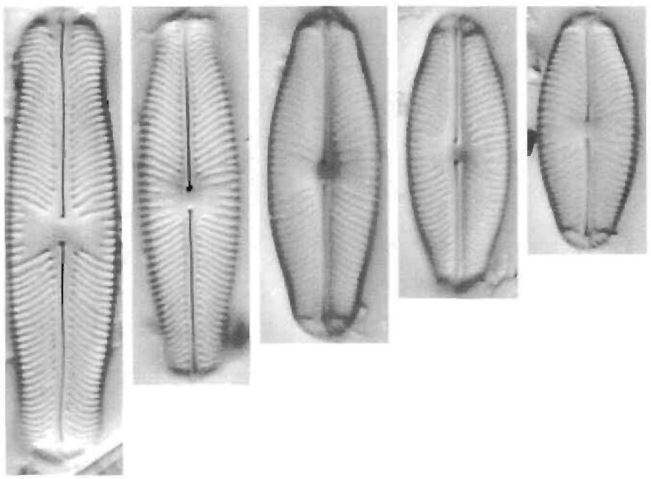
10 µm

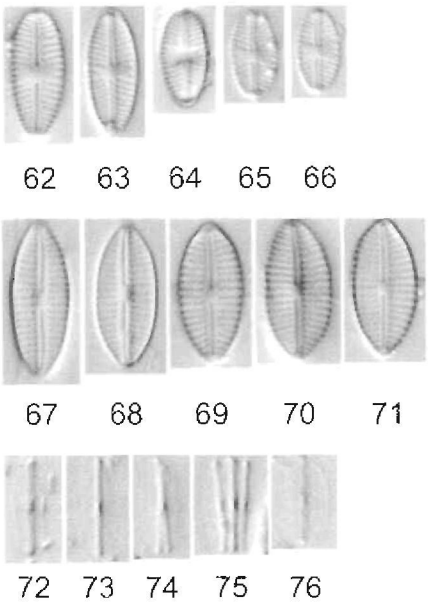
20-23 *Achnanthidium biasolettianum* (Grunow in Cleve & Grunow) Lange-Bertalot
24-28 *Achnanthidium minutissimum* (Kützing) Czarnecki
29-33 *Planothidium frequentissimum* (Lange-Bertalot) Lange-Bertalot
34-38 *Cocconeis placentula var. euglypta* (Ehrenberg) Grunow
39-41 *Achnanthidium saprophila* (Kobayasi & Mayama) Round & Bukhtiyarova
42-46 *Cocconeis pediculus* Ehrenberg

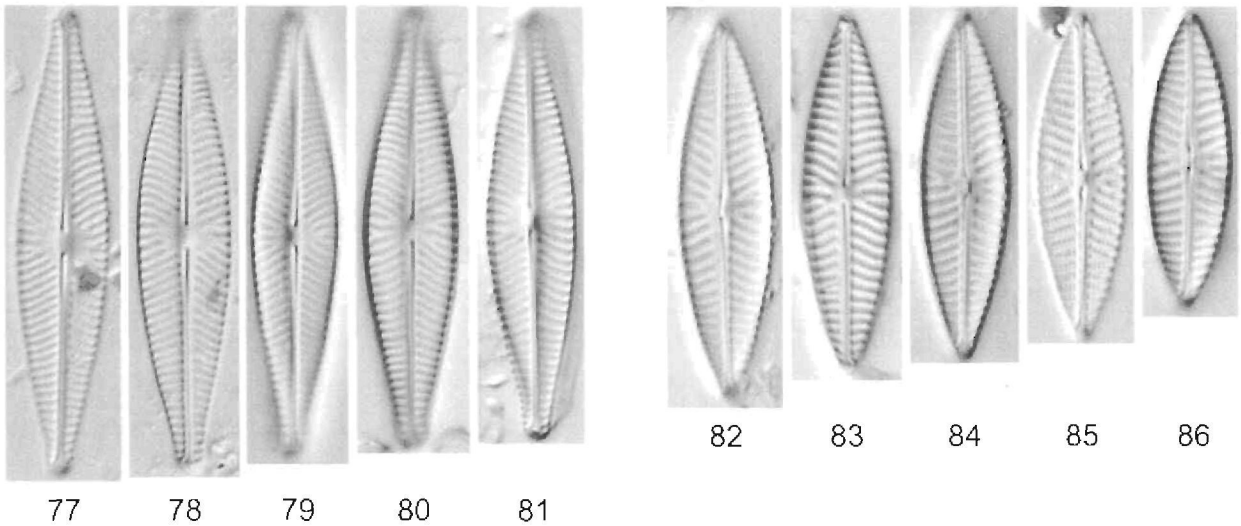
PLATE 2

Biraphideae







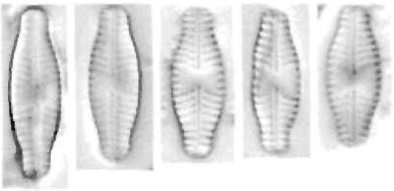


10 µm

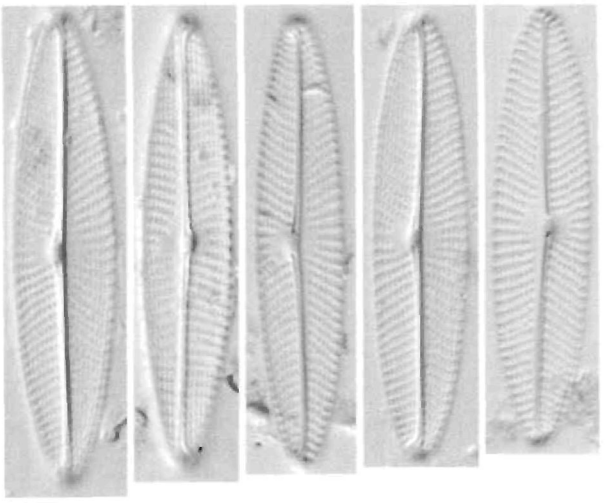
47-51 *Craticula molestiformis* (Hustedt) Lange-Bertalot
52-56 *Craticula accomoda* (Hustedt) Mann
57-61 *Sellaphora pupula* (Kützing) Mereschkowksy
62-66 *Eolimna minima* (Grunow) Lange-Bertalot
67-71 *Eolimna subminuscula* (Manguin) Moser, Lange-Bertalot & Metzeltin
72-76 *Fistulifera saprophila* (Lange-Bertalot & Bonik) Lange-Bertalot
77-81 *Navicula cryptocephala* Kützing
82-86 *Navicula cryptotenella* Lange-Bertalot

PLATE 3

Biraphideae



87 88 89 90 91



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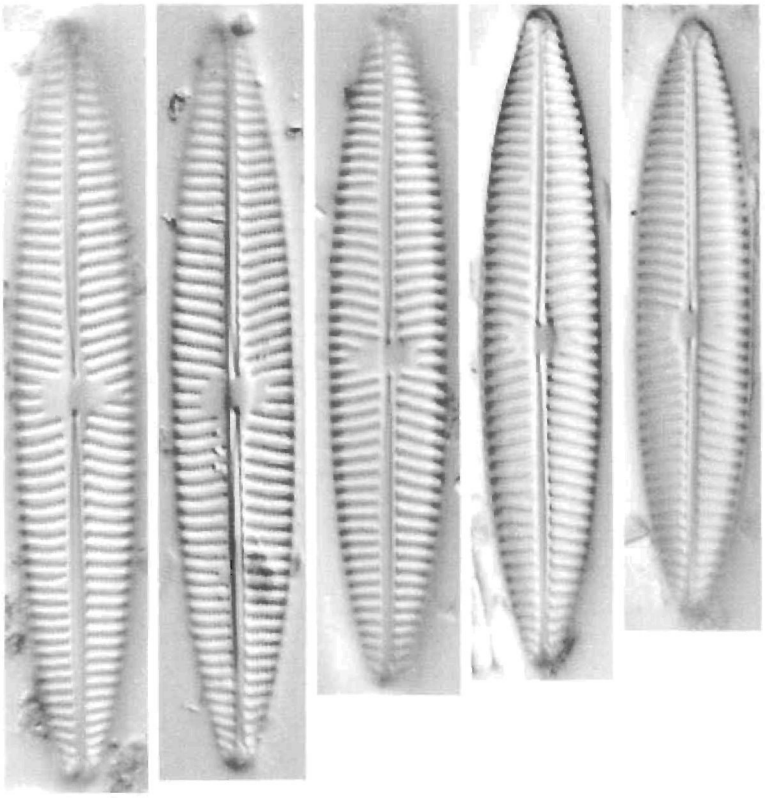
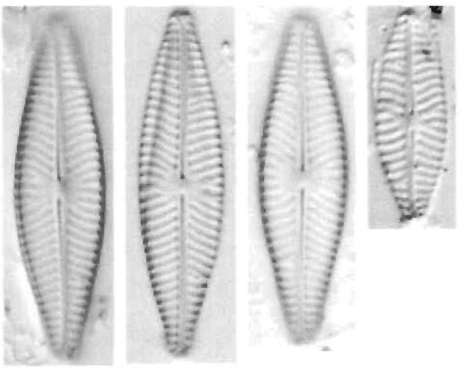


97 98 99 100 101



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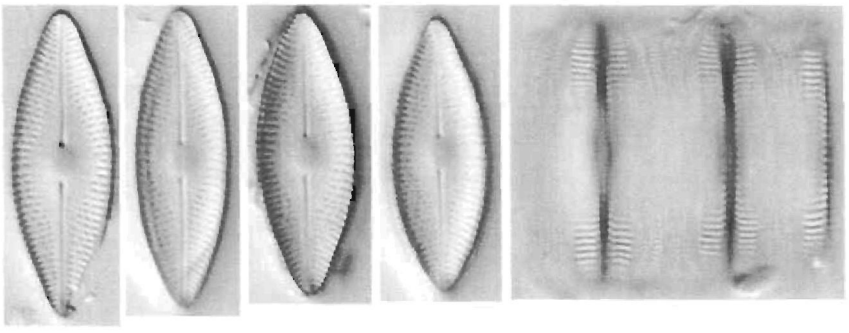

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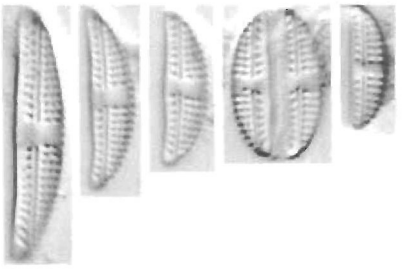


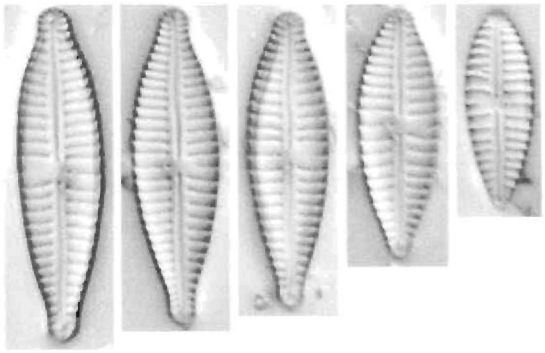
111 112 113 114 115

10 µm

87-91 *Navicula* cf. *pseudoventralis* Hustedt
92-96 *Navicula symmetrica* Patrick
97-101 *Navicula tripunctata* (O.F.Müller) Bory
102-105 *Navicula veneta* Kützing
106-110 *Diadesmis confervacea* Kützing
111-115 *Amphora pediculus* (Kützing) Grunow

PLATE 4

Biraphideae



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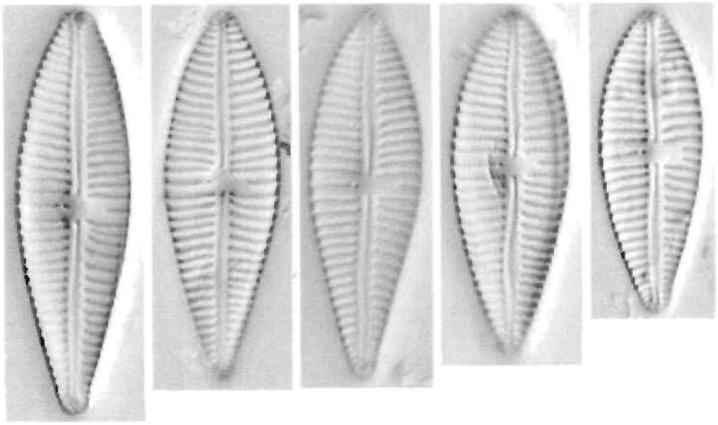

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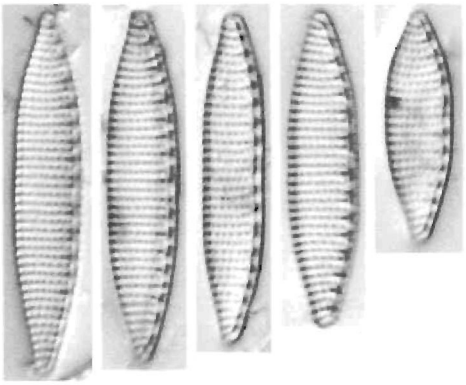

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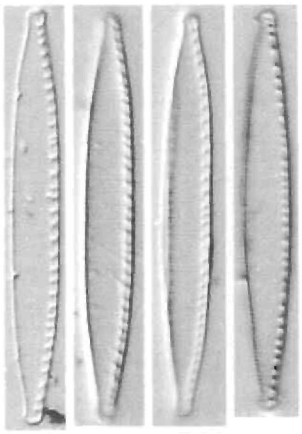

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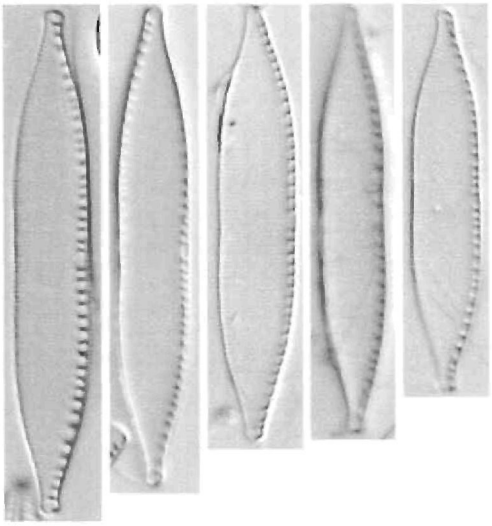

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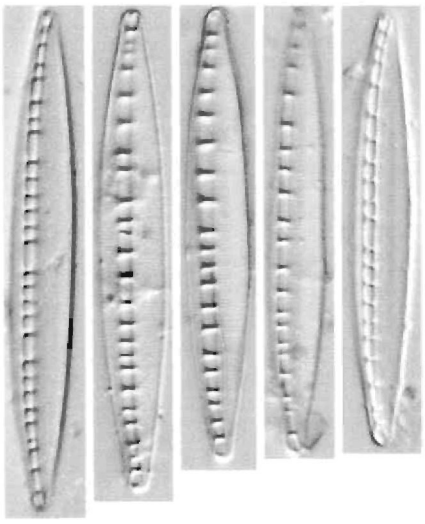

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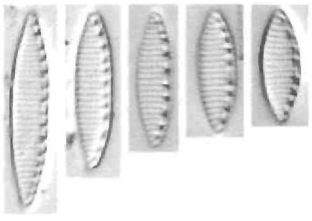


145 146 147 148 149

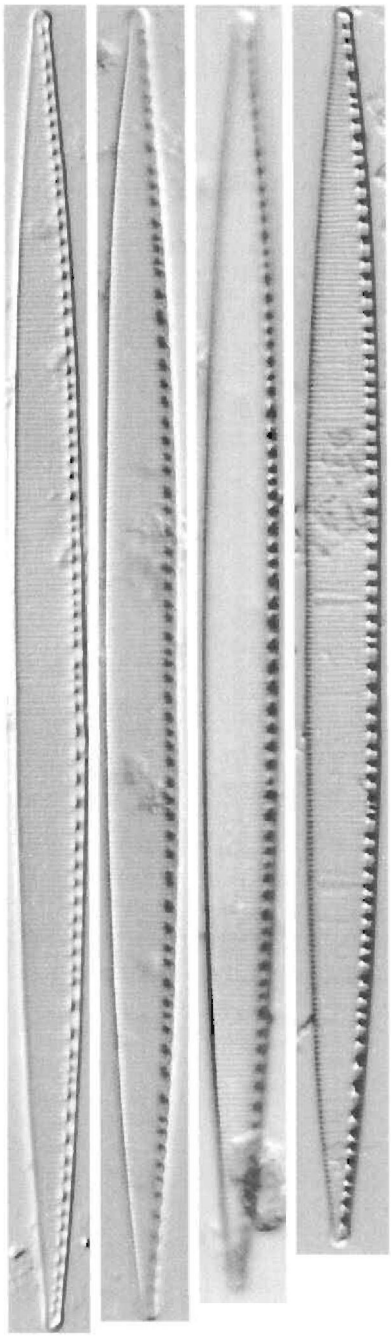
150 151 152 153

10 µm

116-120 *Gomphonema parvulum* (Kützing) Kützing
121-125 *Gomphonema pseudoaugur* Lange-Bertalot
126-130 *Nitzschia amphibia* Grunow
131-134 *Nitzschia archibaldii* Lange-Bertalot
135-139 *Nitzschia capitellata* Hustedt in A. Schmidt
140-144 *Nitzschia dissipata (Kützing)* Grunow
145-149 *Nitzschia frustulum (Kützing)* Grunow
150-153 *Nitzschia inconspicua* Grunow

PLATE 5

Biraphideae



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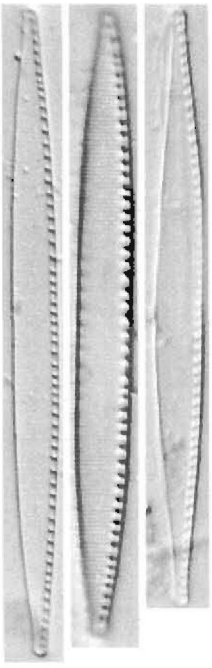

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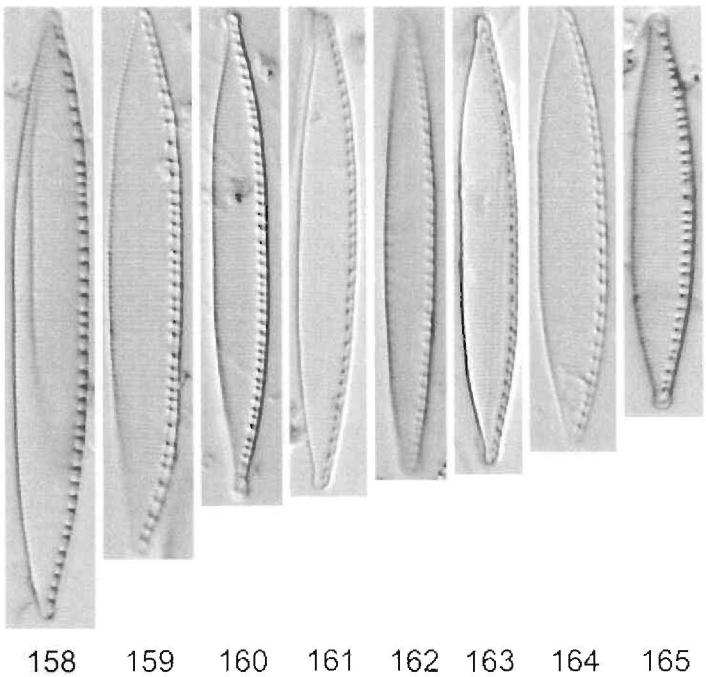

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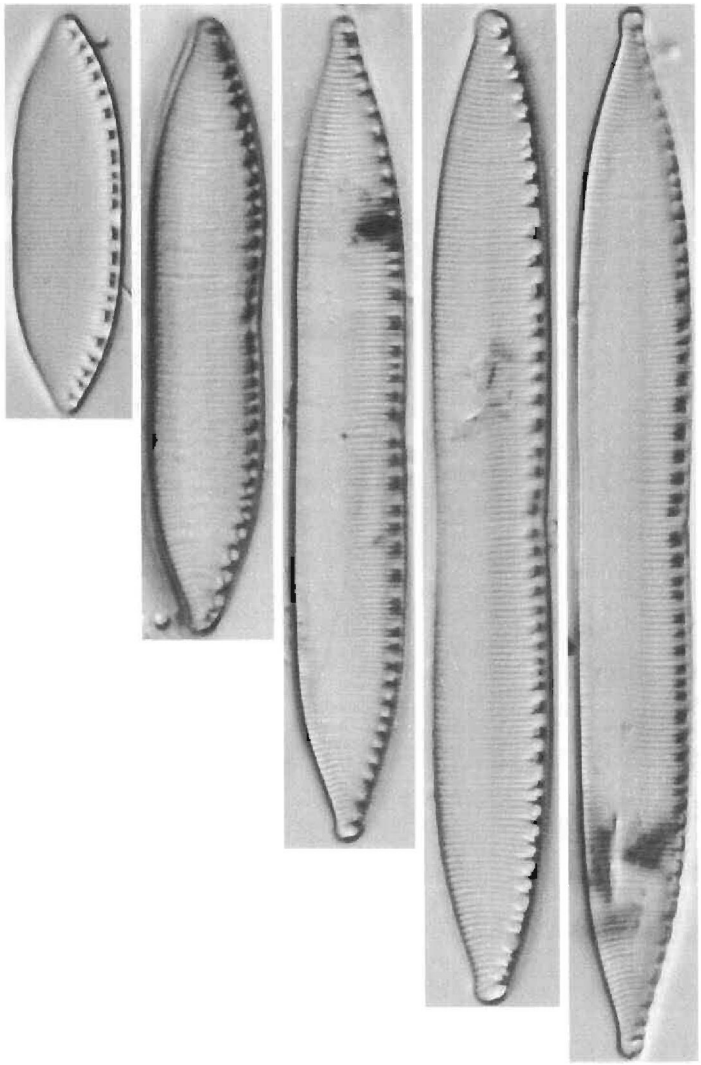
166 167



171 172 173 174 175

10 µm

154-157 *Nitzschia intermedia* Hantzsch ex Cleve & Grunow
158-165 *Nitzschia palea* (Kützing) W. Smith
166-167 *Nitzschia paleacea* (Grunow) Grunow in van Heurck
168-170 *Nitzschia paleaeformis* Hustedt
171-175 *Nitzschia umbonata* (Ehrenberg) Lange-Bertalot

PLATE 6

Appendix B

Species Abundance Data

Acronyms for the species used	
Acronym	**Species**
AINF	*Achnanthes inflata* (Kützing) Grunow
ADMF	*Achnanthidium minutissimum* (Kützing) Czarnecki var. *affinis* (Grunow) Bukhtiyarova
ADMI	*Achnanthidium minutissimum* (Kützing) Czarnecki
ADBI	*Achnanthidium biasolettianum* (Grunow in Cleve & Grunow) Lange-Bertalot
ADEU	*Achnanthidium eutrophilum* (Lange-Bertalot) Lange-Bertalot
ADSA	*Achnanthidium saprophila* (Kobayasi et Mayama) Round & Bukhtiyarova
ABRY	*Adlafia bryophila* (Petersen) Moser, Lange-Bertalot & Metzeltin
AINA	*Amphora inariensis Krammer*
ALIB	*Amphora libyca* Ehrenberg
AMMO	*Amphora montana* Krasske
APED	*Amphora pediculus* (Kützing) Grunow
AFOR	*Asterionella formosa* Hassall
AAMB	*Aulacoseira ambigua* (Grunow) Simonsen
AUGR	*Aulacoseira granulata* (Ehrenberg) Simonsen
AUGA	*Aulacoseira granulata* (Ehrenberg) Simonsen var.*angustissima* (O. Müller) Simonsen
AMUZ	*Aulacoseira muzzanensis* (Meister) Krammer
CBAC	*Caloneis bacillum* (Grunow) Cleve
CKPP	*Cymbella kappii* (Cholnoky) Cholnoky
CPED	*Cocconeis pediculus* Ehrenberg
CPLA	*Cocconeis placentula* Ehrenberg var. *placentula*
CPLE	*Cocconeis placentula* Ehrenberg var. *euglypta* (Ehrenberg) Grunow
CRAC	*Craticula accomoda* (Hustedt) Mann
CAMB	*Craticula ambigua* (Ehrenberg) Mann
CHAL	*Craticula halophila* (Grunow ex Van Heurck) Mann
CATO	*Cyclotella atomus* Hustedt
CMEN	*Cyclotella meneghiniana* Kützing
COCE	*Cyclotella ocellata* Pantocsek
CPST	*Cyclotella pseudostelligera* Hustedt
CSOL	*Cymatopleura solea* (Brebisson) W.Smith
CCYM	*Cymbella cymbiformis* Agardh
CTUM	*Cymbella tumida* (Brebisson) Van Heurck
CTGL	*Cymbella turgidula* Grunow
DCOF	*Diadesmis confervacea* Kützing
DCOT	*Diadesmis contenta* (Grunow ex V. Heurck) Mann
DVUL	*Diatoma vulgaris* Bory
ENMI	*Encyonema minutum* (Hilse in Rabh.) D.G. Mann
ESLE	*Encyonema silesiacum* (Bleisch in Rabh.) D.G. Mann
ENCM	*Encyonopsis microcephala* (Grunow) Krammer
EOMI	*Eolimna minima* (Grunow) Lange-Bertalot
ESBM	*Eolimna subminuscula* (Manguin) Moser Lange-Bertalot & Metzeltin
FPYG	*Fallacia pygmaea* (Kützing) Stickle & Mann
FSAP	*Fistulifera saprophila* (Lange-Bertalot & Bonik) Lange-Bertalot
FBCP	*Fragilaria biceps* (Kützing) Lange-Bertalot
FBRE	*Fragilaria brevistriata* Grunow
FCAP	*Fragilaria capucina* Desmazieres
FCON	*Fragilaria construens* (Ehrenberg) Grunow f. *construens*
FCBI	*Fragilaria construens* f. *binodis* (Ehrenberg) Hustedt
FCRO	*Fragilaria crotonensis* Kitton
FLEP	*Fragilaria leptostauron* (Ehrenberg) Hustedt var. *leptostauron*
FPAR	*Fragilaria parasitica* (W. Smith) Grunow var. *parasitica*
FPIN	*Fragilaria pinnata* Ehrenberg var. *pinnata*
FULN	*Fragilaria ulna* (Nitzsch) Lange-Bertalot var. *ulna*
FUAC	*Fragilaria ulna* (Nitzsch) Lange-Bertalot var. *acus* (Kützing) Lange-Bertalot
FVUL	*Frustulia vulgaris* (Thwaites) De Toni

Acronyms for the species used	
Acronym	**Species**
GAFF	*Gomphonema affine* Kützing
GANG	*Gomphonema angustatum* (Kützing) Rabenhorst
GGRA	*Gomphonema gracile* Ehrenberg
GMIN	*Gomphonema minutum* (Agardh) Agardh f. *minutum*
GPAR	*Gomphonema parvulum* (Kützing) Kützing var. *parvulum*
GPSA	*Gomphonema pseudoaugur* Lange-Bertalot
GPUM	*Gomphonema pumilum* (Grunow) Reichardt & Lange-Bertalot
GTRU	*Gomphonema truncatum* Ehrenberg
GYAC	*Gyrosigma acuminatum* (Kützing)Rabenhorst
GYAT	*Gyrosigma attenuatum* (Kützing) Rabenhorst
HAMP	*Hantzschia amphioxys* (Ehrenberg) Grunow in Cleve et Grunow
HCAP	*Hippodonta capitata* (Ehrenberg) Lange-Bertalot, Metzeltin & Witkowski
LMUT	*Luticola mutica* (Kützing) D.G. Mann
LVEN	*Luticola ventricosa* (Kützing) D.G. Mann
MAAT	*Mayamaea atomus* (Kützing) Lange-Bertalot
MAPE	*Mayamaea atomus* var. *permitis* (Hustedt) Lange-Bertalot
MFOB	*Mayamaea fossalis* var. *obsidialis* (Hustedt) Lange-Bertalot
MVAR	*Melosira varians* Agardh
NAVI	*NAVICULA* J.B.M. Bory de St. Vincent
NAAN	*Navicula angusta* Grunow
NANT	*Navicula antonii* Lange-Bertalot
NAMA	*Navicula arvensis* Hustedt var. *maior* Lange-Bertalot
NCPR	*Navicula capitatoradiata* Germain
NCIN	*Navicula cincta* (Ehrenberg) Ralfs in Pritchard
NCRY	*Navicula cryptocephala* Kützing
NCTE	*Navicula cryptotenella* Lange-Bertalot
NLAN	*Navicula lanceolata* (Agardh) Ehrenberg
NLIB	*Navicula libonensis* Schoeman
NLOV	*Navicula longicephala* Hustedt var. *vilaplanii* Sabater & Lange-Bertalot
NMIS	*Navicula minuscula* Grunow in Van Heurck
NMLF	*Navicula molestiformis* Hustedt
NPVE	*Navicula pseudoventralis* Hustedt
NSSY	*Navicula schroeteri* Meister var. *symmetrica* (Patrick) Lange-Bertalot
NTPT	*Navicula tripunctata* (O.F.Müller) Bory
NTRV	*Navicula trivialis* Lange-Bertalot var. *trivialis*
NVEN	*Navicula veneta* Kützing
NVRO	*Navicula viridula* (Kützing) Ehrenberg var. *rostellata* (Kützing) Cleve
NVIR	*Navicula viridula* (Kützing) Ehrenberg
NITZ	*NITZSCHIA* A.H. Hassall
NACI	*Nitzschia acicularis* (Kützing) W.M. Smith
NAMP	*Nitzschia amphibia* Grunow f. *amphibia*
NIAR	*Nitzschia archibaldii* Lange-Bertalot
NCPL	*Nitzschia capitellata* Hustedt in A. Schmidt
NCLA	*Nitzschia clausii* Hantzsch
NCOM	*Nitzschia communis* Rabenhorst
NDES	*Nitzschia desertorum* Hustedt
NDIS	*Nitzschia dissipata* (Kützing) Grunow var. *dissipata*
NFON	*Nitzschia fonticola* Grunow in Cleve et Möller
NIFR	*Nitzschia frustulum* (Kützing) Grunow var. *frustulum*
NHEU	*Nitzschia heufleriana* Grunow
NINC	*Nitzschia inconspicua* Grunow
NINT	*Nitzschia intermedia* Hantzsch ex Cleve & Grunow
NLBT	*Nitzschia liebetruthii* Rabenhorst
NLIN	*Nitzschia linearis* (Agardh) W.M. Smith

Acronyms for the species used	
Acronym	**Species**
NMIC	*Nitzschia microcephala* Grunow in Cleve & Moller
NNAN	*Nitzschia nana* Grunow in Van Heurck
NPAL	*Nitzschia palea* (Kützing) W. Smith
NPAE	*Nitzschia paleacea* (Grunow) Grunow in van Heurck
NIPF	*Nitzschia paleaeformis* Hustedt
NREC	*Nitzschia recta* Hantzsch in Rabenhorst
NSIT	*Nitzschia sinuata* (Thwaites) Grunow var. *tabellaria* Grunow
NSOC	*Nitzschia sociabilis* Hustedt
NSUA	*Nitzschia subacicularis* Hustedt in A.Schmidt
NZSU	*Nitzschia supralitore*a Lange-Bertalot
NUMB	*Nitzschia umbonata* (Ehrenberg) Lange-Bertalot
PBRT	*Pinnularia borealis* Ehrenberg f. rectangularis Carlson
PBRA	*Pinnularia braunii* (Grunow) Cleve
PBRE	*Pinnularia brebissonii* (Kützing) Rabenhorst var. brebissonii
PGIB	*Pinnularia gibba* Ehrenberg
PMIC	*Pinnularia microstauron* (Ehrenberg) Cleve
PSBV	*Pinnularia subbrevistriata* Krammer
PVIF	*Pinnularia viridiformis* Krammer
PDIC	*Placoneis dicephala* (W. Smith) Mereschkowsky
PPLC	*Placoneis placentula* (Ehrenberg) Heinzerling
PLFR	*Planothidium frequentissimum* (Lange-Bertalot) Lange-Bertalot
PMTN	*Psammothidium montanum* (Krasske) Mayama
RUNI	*Reimeria uniseriata* Sala, Guerrero & Ferrario
RABB	*Rhoicosphenia abbreviata* (C. Agardh) Lange-Bertalot
RGBL	*Rhopalodia gibberula* (Ehrenberg) O.Müller
SPUP	*Sellaphora pupula* (Kützing) Mereschkowksy
SSEM	*Sellaphora seminulum* (Grunow) D.G. Mann
STAN	*Stauroneis anceps* Ehrenberg
SPHO	*Stauroneis phoenicenteron* (Nitzsch) Ehrenberg
SANG	*Surirella angusta* Kützing
SBRE	*Surirella brebissonii* Krammer & Lange-Bertalot
SBRI	*Surirella brightwellii* W. Smith
SOVI	*Surirella ovalis* Brebisson
TWEI	*Thalassiosira weissflogii* (Grunow) Fryxell & Hasle
TAPI	*Tryblionella apiculata* Gregory
TGRL	*Tryblionella gracilis* W. Smith

Wasgoedspruit at Approximate Origin (WS 0)										
Taxon	Date									
	27/04/2005	05/05/2005	22/05/2005	02/06/2005	19/06/2005	03/07/2005	31/07/2005	13/08/2005	11/01/2006	26/01/2006
Achnanthes inflata (Kützing) Grunow				0.25			0.50		0.50	
Adlafia bryophila (Petersen) Moser, Lange-Bertalot & Metzeltin	0.37									
Caloneis bacillum (Grunow) Cleve			4.50	0.75						0.50
Craticula ambigua (Ehrenberg) Mann					0.25	0.25				
Craticula halophila (Grunow ex Van Heurck) Mann			0.50							
Cyclotella atomus Hustedt					0.25	0.25				
Cyclotella meneghiniana Kützing	8.99	1.50	2.00	2.00	0.50	0.50	0.50	0.25		
Cyclotella ocellata Pantocsek	0.12		1.00	0.25						
Diadesmis confervacea Kützing	1.97	41.65	2.00	3.75	4.74	4.74	1.25	0.25	0.50	0.50
Diadesmis contenta (Grunow ex V. Heurck) Mann					0.25	0.25				
Encyonema minutum (Hilse in Rabh.) D.G. Mann	0.37		0.25	0.25						
Encyonema silesiacum (Bleisch in Rabh.) D.G. Mann	0.49									
Eolimna minima (Grunow) Lange-Bertalot	3.79	2.99	6.25	7.98	3.99	3.99	3.23	1.49	6.00	2.00
Eolimna subminuscula (Manguin) Moser Lange-Bertalot & Metzeltin	9.85	1.50	1.00		1.50	1.50				
Fistulifera saprophila (Lange-Bertalot & Bonik) Lange-Bertalot	5.42		0.75	18.73	15.72	15.72	3.72	7.44	0.50	2.00
Fragilaria ulna (Nitzsch) Lange-Bertalot var. *acus* (Kützing) Lange-Bertalot		1.00		1.50	0.50	0.50				
Gomphonema affine Kützing				0.50	0.75	0.75	0.25	0.25		2.00
Gomphonema angustatum (Kützing) Rabenhorst	4.93	4.99								
Gomphonema gracile Ehrenberg	0.25	1.50		1.25	0.25	0.25				
Gomphonema minutum (Agardh) Agardh f. *minutum*					0.25	0.25				
Gomphonema parvulum (Kützing) Kützing var. *parvulum*	3.79	3.24	3.25	9.23	4.74	4.74	1.25			1.50
Gomphonema pseudoaugur Lange-Bertalot	0.12	4.99	0.50	4.49	8.98	8.98	6.45	6.45	0.50	0.50
Gomphonema pumilum (Grunow) Reichardt & Lange-Bertalot			0.25	0.50	2.24	2.24				
Hantzschia amphioxys (Ehrenberg) Grunow in Cleve et Grunow					1.00	1.00			1.50	2.50
Hippodonta capitata (Ehrenberg) Lange-Bertalot, Metzeltin & Witkowski				0.50						
Luticola mutica (Kützing) D.G. Mann				0.50						0.50
Luticola ventricosa (Kützing) D.G. Mann										1.00
Mayamaea atomus (Kützing) Lange-Bertalot	21.92	16.29	2.25	3.24	3.49	3.49			4.00	17.00
Mayamaea atomus var. *permitis* (Hustedt) Lange-Bertalot	2.94		1.75	2.74						
Mayamaea fossalis var. *obsidialis* (Hustedt) Lange-Bertalot				0.25						
NAVICULA J.B.M. Bory de St. Vincent		0.25								
Navicula arvensis Hustedt var. *maior* Lange-Bertalot				0.25	0.25	0.25				
Navicula capitatoradiata Germain			1.25							
Navicula cryptocephala Kützing	2.46	1.75	16.25	13.97	23.94	23.94	7.72	75.19		
Navicula pseudoventralis Hustedt	1.48	4.24	3.25	4.74	0.50	0.50	0.25		11.00	26.50
Navicula veneta Kützing	1.70	1.50	11.25		6.23	6.23	6.23	4.96	1.00	
Navicula viridula (Kützing) Ehrenberg	0.12									
NITZSCHIA A.H. Hassall					1.22	1.22	0.25	1.25	3.00	1.50
Nitzschia acicularis (Kützing) W.M. Smith			0.75		0.25	0.25				
Fragilaria pinnata Ehrenberg var. *pinnata*	0.12									
Nitzschia archibaldii Lange-Bertalot		1.50	4.50		1.25	1.25	0.50		5.50	1.50
Nitzschia capitellata Hustedt in A. Schmidt	0.74		6.00	2.00	0.25	0.25	0.25	0.25	3.00	2.00
Nitzschia clausii Hantzsch	0.12									
Nitzschia desertorum Hustedt	0.25									
Nitzschia inconspicua Grunow	0.12									
Nitzschia intermedia Hantzsch ex Cleve & Grunow	0.25	0.50	8.25	6.73	2.00	2.00	1.49	0.50	3.00	0.50
Nitzschia liebetruthii Rabenhorst	0.37									
Nitzschia linearis (Agardh) W.M. Smith			1.00	0.25	0.25	0.25				
Nitzschia palea (Kützing) W. Smith	22.78	4.99	13.25	4.99	2.24	2.24		0.74	58.00	3.00
Nitzschia paleacea (Grunow) Grunow in van Heurck	1.70	2.49	3.25	5.24	1.50	1.50	0.74			
Nitzschia supralitorea Lange-Bertalot	0.12									
Nitzschia umbonata (Ehrenberg) Lange-Bertalot	0.86									
Pinnularia borealis Ehrenberg f. *rectangularis* Carlson					0.25	0.25	0.74		0.50	1.00
Pinnularia braunii (Grunow) Cleve			0.25							1.00
Pinnularia microstauron (Ehrenberg) Cleve									0.50	1.00
Pinnularia subbrevistriata Krammer		0.25	1.00				0.99			
Pinnularia viridiformis Krammer			0.75				0.50	0.50		
Placoneis dicephala (W. Smith) Mereschkowsky		0.25	0.75		0.25	0.25			1.00	0.50
Planothidium frequentissimum (Lange-Bertalot) Lange-Bertalot	0.37									
Sellaphora pupula (Kützing) Mereschkowksy	2.96		1.25	0.75	0.25	0.25	0.25			2.50
Sellaphora seminulum (Grunow) D.G. Mann		2.49	0.50	1.75	0.75	0.75		0.25		
Stauroneis anceps Ehrenberg		0.25	0.25	0.75				0.25		1.00
Stauroneis phoenicenteron (Nitzsch) Ehrenberg										0.50
Surirella angusta Kützing					0.25	0.25				0.50
Thalassiosira weissflogii (Grunow) Fryxell & Hasle	0.62									

Wasgoedspruit Below Railway Bridge (WS 1)																
Taxon	Date															
	20/04/2005	05/05/2005	22/05/2005	02/06/2005	19/06/2005	19/06/2005	03/07/2005	03/07/2005	17/07/2005	17/07/2005	31/07/2005	31/07/2005	13/08/2005	13/08/2005	28/08/2005	28/08/2005
Achnanthidium minutissimum (Kützing) Czarnecki														1.20	1.93	0.74
Achnanthidium saprophila (Kobayasi et Mayama) Round & Bukhtiyarova					0.25											
Amphora montana Krasske							1.24									
Craticula accomoda (Hustedt) Mann									0.98		1.25	5.49	0.98			0.99
Craticula ambigua (Ehrenberg) Mann							0.25				0.25		0.25		0.48	
Craticula halophila (Grunow ex Van Heurck) Mann													1.97	1.44	1.25	1.23
Cyclotella atomus Hustedt										0.25						
Cyclotella meneghiniana Kützing	1.00	17.89	22.22	16.27	8.75	29.18	8.66	21.59	6.62	2.00	12.50	8.35	12.98	54.36	3.62	8.13
Cyclotella ocellata Pantocsek		0.25				0.25		0.25								
Diadesmis confervacea Kützing										0.75	0.75					
Eolimna minima (Grunow) Lange-Bertalot	5.75	1.48	3.46	1.18	1.50	4.49	1.24	4.71	1.48	1.23	3.25	2.39	1.48	0.96	4.82	1.23
Eolimna subminuscula (Manguin) Moser Lange-Bertalot & Metzeltin	24.25	13.73	19.75	17.22	25.50	5.99	24.55	3.97	4.92	0.75	3.50	2.86	1.48	0.24	4.96	2.79
Fistulifera saprophila (Lange-Bertalot & Bonik) Lange-Bertalot	7.25	1.78	9.14	4.94	2.75	3.24	2.48	8.93	5.88	4.44	6.50	3.13	1.54	0.72	6.27	18.72
Fragilaria ulna (Nitzsch) Lange-Bertalot var. *ulna*			0.25	0.24	0.25	0.25				0.25	0.50					
Fragilaria ulna (Nitzsch) Lange-Bertalot var. *acus* (Kützing) Lange-Bertalot											0.25			2.88		
Gomphonema affine Kützing											0.75					
Gomphonema parvulum (Kützing) Kützing var. *parvulum*	3.25	5.15	9.63	5.90	1.75	1.47	1.73	18.61	8.88	9.38	13.50	0.72	1.97	4.78	11.87	1.23
Mayamaea atomus (Kützing) Lange-Bertalot		0.25	0.25				0.25									
Mayamaea atomus var. *permitis* (Hustedt) Lange-Bertalot	2.25	1.23	0.49			0.50										
NAVICULA J.B.M. Bory de St. Vincent											0.75		0.25			
Navicula arvensis Hustedt var. *maior* Lange-Bertalot							0.25				0.25					
Navicula capitatoradiata Germain								0.25								
Navicula cryptotenella Lange-Bertalot							0.25									
Navicula trivialis Lange-Bertalot var. *trivialis*										0.25						
Navicula veneta Kützing	6.25	1.72	3.74	6.37	5.25	5.24	8.17	1.74	1.48	0.49	3.50	1.19	0.49	0.48	3.61	2.79
Navicula viridula (Kützing) Ehrenberg	0.25	0.25			0.25											
NITZSCHIA A.H. Hassall						0.75		0.25								
Nitzschia amphibia Grunow f. *amphibia*	1.25	0.25		1.66	0.25	2.99	1.00	0.74			1.00			0.24		0.49
Nitzschia capitellata Hustedt in A. Schmidt		0.74	0.25	0.94	3.25	0.75	5.20	1.49	22.59	3.96	6.50	17.18	5.64	0.48	1.69	3.45
Nitzschia communis Rabenhorst											0.50	1.19	1.48			2.46
Nitzschia desertorum Hustedt	1.00	0.49	0.75	0.24	1.00	1.00	0.25	0.25	0.74	0.75	0.50	0.48	0.49	0.24		
Nitzschia inconspicua Grunow		0.25														
Nitzschia intermedia Hantzsch ex Cleve & Grunow													0.25	0.24	0.96	0.49
Nitzschia liebetruthii Rabenhorst	0.25	0.49								1.73		0.24				
Nitzschia microcephala Grunow in Cleve & Moller	0.25															
Nitzschia palea (Kützing) W. Smith	28.50	35.29	19.26	25.47	37.00	6.48	29.46	14.89	23.28	2.99	15.75	4.95	34.31	5.24	7.95	31.77
Nitzschia paleacea (Grunow) Grunow in van Heurck										7.65	1.75	7.40	18.38	1.48	12.29	16.75
Nitzschia supralitorea Lange-Bertalot	0.50	0.25	0.75			0.50		1.99			0.75	0.48				
Nitzschia umbonata (Ehrenberg) Lange-Bertalot	1.00	1.72	0.75	11.79	7.25	0.50	3.71	0.74	1.72	5.68	1.25	3.13	3.19	0.48	0.72	2.22
Fragilaria pinnata Ehrenberg var. *pinnata*	6.00	0.74	0.99	0.94	0.75	12.72	0.50	9.68	19.36	9.63	11.25	3.58	2.94	2.15	7.72	3.22
Sellaphora pupula (Kützing) Mereschkowksy	2.00	5.88	7.47	6.13	3.25	0.75	1.15	0.25	0.25	0.75	0.25	0.24	0.25		0.96	0.25
Sellaphora seminulum (Grunow) D.G. Mann				0.24		0.25		0.25	0.49	0.75		0.72	0.49			
Surirella angusta Kützing						0.50			0.98	0.49	0.25	0.24	0.25			0.74
Surirella brightwellii W. Smith												0.24	0.25		0.25	
Thalassiosira weissflogii (Grunow) Fryxell & Hasle		1.23	0.99	1.42	1.00	13.22	0.74	9.43	1.72	9.88	3.75	0.72	0.74	14.11	2.66	0.49

Wasgoedspruit Adjacent to Wallis Street (WS 2)											
Taxon	Date										
	30/03/2005	30/03/2005	20/04/2005	05/05/2005	19/05/2005	02/06/2005	19/06/2005	03/07/2005	17/07/2005	31/07/2005	13/08/2005
Achnanthidium saprophila (Kobayasi et Mayama) Round & Bukhtiyarova							0.50				
Craticula accomoda (Hustedt) Mann								1.24	0.25	1.70	
Craticula ambigua (Ehrenberg) Mann								0.25		0.24	
Craticula halophila (Grunow ex Van Heurck) Mann											2.14
Cyclotella meneghiniana Kützing	6.83	6.88	2.88	3.38	15.42	11.50	17.76	9.45	24.50	12.86	27.62
Cyclotella ocellata Pantocsek								0.25			
Eolimna minima (Grunow) Lange-Bertalot	0.24	0.25	0.48	0.24	1.00	1.25	2.74	2.74	1.25	7.28	2.14
Eolimna subminuscula (Manguin) Moser Lange-Bertalot & Metzeltin	15.85	13.27	67.73	3.43	32.84	34.50	27.43	5.72	4.00	11.65	8.33
Fistulifera saprophila (Lange-Bertalot & Bonik) Lange-Bertalot	24.39	12.29	17.22	25.13	6.72	8.25	2.24	1.45	1.50	11.65	5.24
Fragilaria ulna (Nitzsch) Lange-Bertalot var. *ulna*										0.49	
Gomphonema affine Kützing								0.75			
Gomphonema parvulum (Kützing) Kützing var. *parvulum*	4.15	4.67	3.35	18.12	8.76	1.00	11.22	45.25	2.75	6.68	4.76
Mayamaea atomus (Kützing) Lange-Bertalot				0.48				0.50		0.24	
Mayamaea atomus var. *permitis* (Hustedt) Lange-Bertalot	1.22	1.72	4.78		0.50		0.25	0.25			
Navicula arvensis Hustedt var. *maior* Lange-Bertalot							0.75				
Navicula veneta Kützing			0.24	0.48	2.74	4.00	5.99	2.24	0.75	2.18	1.67
Navicula viridula (Kützing) Ehrenberg						0.50	2.49				
NITZSCHIA A.H. Hassall							0.50				
Nitzschia amphibia Grunow f. *amphibia*	0.24	0.25	1.67	0.24	0.25	2.50	0.75	0.75		2.67	0.71
Nitzschia capitellata Hustedt in A. Schmidt	1.22			0.48	1.24		1.00	1.24	8.00	4.13	0.24
Nitzschia desertorum Hustedt	4.63	0.25		0.72	0.75		0.25	0.50	1.25	1.94	0.71
Nitzschia inconspicua Grunow					0.25						
Nitzschia liebetruthii Rabenhorst	0.24	0.25					1.50		0.50		
Nitzschia palea (Kützing) W. Smith	27.32	48.43	0.24	11.84	11.94	1.75	11.47	11.44	45.50	2.15	13.90
Nitzschia paleacea (Grunow) Grunow in van Heurck									1.25	4.85	25.71
Nitzschia supralitorea Lange-Bertalot		0.25		0.24	0.25		0.75	0.75	0.50	1.70	
Nitzschia umbonata (Ehrenberg) Lange-Bertalot	9.76	8.85	0.24	1.93	5.97	1.25	3.99	0.25	1.25	0.98	1.19
Planothidium frequentissimum (Lange-Bertalot) Lange-Bertalot			0.48			1.00	0.25	2.49	1.25	4.37	2.14
Sellaphora pupula (Kützing) Mereschkowksy	3.92	2.73	0.72	6.28	11.44	13.25	5.49	1.74	1.75	0.98	0.95
Sellaphora seminulum (Grunow) D.G. Mann							0.25	0.50	0.25	0.49	0.24
Surirella angusta Kützing									0.50	0.98	0.24
Thalassiosira weissflogii (Grunow) Fryxell & Hasle						1.25	2.49	1.49	3.00	2.43	2.14

Wasgoedspruit Upstream of Van Riebeeck Street Bridge (WS 3)																	
Taxon	Date																
	30/03/2005	20/04/2005	20/04/2005	05/05/2005	19/05/2005	19/05/2005	02/06/2005	02/06/2005	19/06/2005	19/06/2005	03/07/2005	17/07/2005	17/07/2005	31/07/2005	31/07/2005	13/08/2005	13/08/2005
Achnanthidium minutissimum (Kützing) Czarnecki var. *affinis* (Grunow) Bukhtiyarova			0.25														
Achnanthidium minutissimum (Kützing) Czarnecki																0.49	
Achnanthidium saprophila (Kobayasi et Mayama) Round & Bukhtiyarova							2.79	0.25	0.75								
Amphora montana Krasske			0.25								0.25						
Amphora pediculus (Kützing) Grunow																	
Aulacoseira ambigua (Grunow) Simonsen		0.75															
Aulacoseira granulata (Ehrenberg) Simonsen var.angustissima (O. Müller) Simonsen																	
Caloneis bacillum (Grunow) Cleve																	
Craticula accomoda (Hustedt) Mann											0.25	0.25		1.00	0.75	0.73	1.72
Craticula ambigua (Ehrenberg) Mann																	0.25
Craticula halophila (Grunow ex Van Heurck) Mann																0.49	0.25
Cyclotella meneghiniana Kützing	1.24	15.75	8.29	6.23	8.63	17.79	0.99	11.22	5.50	12.50	8.76	3.93	12.93	7.43	15.72	5.70	11.79
Cyclotella ocellata Pantocsek		0.25	0.25				0.25		0.75			0.25	0.25				
Diadesmis confervacea Kützing		1.25								1.00							
Eolimna minima (Grunow) Lange-Bertalot		1.50	3.23	0.75	2.22	1.24	3.22	1.00	2.25	1.75	5.47	5.16	4.71	6.44	5.99	13.41	7.86
Eolimna subminuscula (Manguin) Moser Lange-Bertalot & Metzeltin	4.50	28.25	42.40	41.90	37.68	34.46	44.83	41.40	42.00	36.50	31.34	35.14	23.77	2.50	19.77	37.73	24.32
Fistulifera saprophila (Lange-Bertalot & Bonik) Lange-Bertalot	8.29	8.75	15.92	18.95	4.19	8.42	13.35	4.74	5.50	3.75	6.22	11.79	1.67	10.00	9.48	1.49	5.90
Fragilaria biceps (Kützing) Lange-Bertalot																	
Fragilaria brevistriata Grunow																	
Fragilaria leptostauron (Ehrenberg) Hustedt var. *leptostauron*													0.25				
Fragilaria ulna (Nitzsch) Lange-Bertalot var. *ulna*			0.50														
Fragilaria ulna (Nitzsch) Lange-Bertalot var. *acus* (Kützing) Lange-Bertalot																0.24	
Frustulia vulgaris (Thwaites) De Toni																	
Gomphonema parvulum (Kützing) Kützing var. *parvulum*	5.72	1.00	9.45	12.47	14.39	11.88	22.17	16.46	12.75	15.75	18.66	23.34	6.70	10.00	8.73	4.63	5.65
Gomphonema pumilum (Grunow) Reichardt & Lange-Bertalot																	
Mayamaea atomus (Kützing) Lange-Bertalot					0.25								0.25	0.25	0.25		0.25
Mayamaea atomus var. *permitis* (Hustedt) Lange-Bertalot	0.25	3.00	0.75		0.25		0.99			1.00							
Navicula arvensis Hustedt var. *maior* Lange-Bertalot		0.25															
Navicula minuscula Grunow in Van Heurck																	
Navicula molestiformis Hustedt																	
Navicula schroeteri Meister var. *symmetrica* (Patrick) Lange-Bertalot																	
Navicula trivialis Lange-Bertalot var. *trivialis*																	
Navicula veneta Kützing	0.25	3.00	1.24	1.50	5.17	2.48	1.48	2.24	2.75	2.50	4.73	1.47	2.23	3.71	1.50	2.68	3.93
Navicula viridula (Kützing) Ehrenberg			0.50	0.25	0.25						0.50						
NITZSCHIA A.H. Hassall		0.25															
Nitzschia acicularis (Kützing) W.M. Smith																	
Nitzschia amphibia Grunow f. *amphibia*	1.49	1.50	1.49		8.37	1.49	2.22	1.75	3.75	2.50	1.00	1.47	1.49	3.47	2.99	0.98	2.21
Nitzschia capitellata Hustedt in A. Schmidt	0.50	1.00	0.25			0.25		0.50	0.25	1.00	0.75	2.21	0.99	0.74	2.00	0.73	3.44
Nitzschia communis Rabenhorst																	0.49
Fragilaria pinnata Ehrenberg var. *pinnata*		1.00	0.25	0.25	0.49	1.24		1.50	1.25	0.75	1.24	0.74	2.73	0.25	1.25	0.49	0.25
Nitzschia dissipata (Kützing) Grunow var. *dissipata*																	
Nitzschia fonticola Grunow in Cleve et Möller														1.73	0.25		
Nitzschia frustulum (Kützing) Grunow var. *frustulum*												0.25		0.25			
Nitzschia heufleriana Grunow																	
Nitzschia inconspicua Grunow		0.75	0.50														
Nitzschia intermedia Hantzsch ex Cleve & Grunow																	0.25
Nitzschia liebetruthii Rabenhorst	0.25	0.25		1.00		0.25	0.74		1.75								
Nitzschia linearis (Agardh) W.M. Smith																	
Nitzschia microcephala Grunow in Cleve & Moller																0.49	0.25
Nitzschia nana Grunow in Van Heurck		0.25															
Nitzschia palea (Kützing) W. Smith	36.32	14.25	6.22	1.72	5.67	8.92	1.72	6.23	6.25	9.25	11.94	9.83	19.63	16.89	15.72	7.85	1.57
Nitzschia paleacea (Grunow) Grunow in van Heurck												0.74	0.50	2.72	3.24	7.32	1.32
Nitzschia paleaeformis Hustedt																	
Nitzschia supralitorea Lange-Bertalot			0.25		0.49			1.00	1.75	0.50		0.25		0.74	1.50	0.49	
Nitzschia umbonata (Ehrenberg) Lange-Bertalot	3.23	2.00	1.99	0.75	0.99	1.73	0.74	2.99	2.25	1.75	1.00	0.49	0.50	1.73	2.00	0.24	0.25
Pinnularia gibba Ehrenberg																	
Pinnularia microstauron (Ehrenberg) Cleve																	
Planothidium frequentissimum (Lange-Bertalot) Lange-Bertalot	0.25	0.25	1.00	0.25	1.48	1.00	0.25	1.25	1.50	1.50	0.50	0.74	2.73	6.19	2.74	1.46	5.45
Psammothidium montanum (Krasske) Mayama													0.25				
Rhopalodia gibberula (Ehrenberg) O.Müller					0.25												
Sellaphora pupula (Kützing) Mereschkowksy	2.24	5.75	5.47	4.99	8.37	9.46	4.43	6.73	7.00	4.75	5.22	1.72	6.95	4.46	2.00	1.77	1.97
Sellaphora seminulum (Grunow) D.G. Mann					1.23	0.25				0.25	0.25		1.25	0.25	0.50	0.98	0.49
Surirella angusta Kützing													0.25	0.25	0.50		0.25
Surirella ovalis Brebisson																	
Thalassiosira weissflogii (Grunow) Fryxell & Hasle								0.75	2.00	3.00	1.99	0.25	1.74	2.48	3.24	1.46	1.97
Tryblionella apiculata Gregory																	

Taxon	Wasgoedspruit Upstream of Van Riebeeck Street Bridge (WS 3)														
	Date														
	28/08/2005	28/08/2005	11/09/2005	11/09/2005	09/10/2005	09/10/2005	26/10/2005	26/10/2005	10/11/2005	13/12/2005	27/12/2005	27/12/2005	05/01/2006	11/01/2006	11/01/2006
Achnanthidium minutissimum (Kützing) Czarnecki var. *affinis* (Grunow) Bukhtiyarova															
Achnanthidium minutissimum (Kützing) Czarnecki		0.24	0.25	1.25	0.75	0.25	0.50		0.25			0.50	0.50		0.25
Achnanthidium saprophila (Kobayasi et Mayama) Round & Bukhtiyarova					0.50	0.25	0.25	0.25		0.74		0.75	0.50	0.25	0.50
Amphora montana Krasske															
Amphora pediculus (Kützing) Grunow													0.25		
Aulacoseira ambigua (Grunow) Simonsen															
Aulacoseira granulata (Ehrenberg) Simonsen var.angustissima (O. Müller) Simonsen				0.25											
Caloneis bacillum (Grunow) Cleve												0.25			
Craticula accomoda (Hustedt) Mann		0.97			2.50	1.25	1.00	1.47	1.25		0.22	0.25	0.25		0.50
Craticula ambigua (Ehrenberg) Mann											0.22				
Craticula halophila (Grunow ex Van Heurck) Mann	0.25	1.95	0.50	0.75	0.25							0.25			
Cyclotella meneghiniana Kützing	5.24	14.60	2.99	12.25	6.75	9.25	6.25	7.62	6.25	7.20	5.72	5.00	3.25	1.24	3.73
Cyclotella ocellata Pantocsek				0.50		1.25									0.50
Diadesmis confervacea Kützing					0.25	0.50									
Eolimna minima (Grunow) Lange-Bertalot	9.73	9.24	6.48	6.50	13.75	9.00	13.50	1.82	2.50	0.99	4.82	3.25	1.25	4.46	1.99
Eolimna subminuscula (Manguin) Moser Lange-Bertalot & Metzeltin	32.67	2.19	48.88	23.50	2.00	13.75	16.25	18.92	13.00	2.23	13.82	6.75	6.25	3.96	6.47
Fistulifera saprophila (Lange-Bertalot & Bonik) Lange-Bertalot	1.22	5.19	1.47	8.00	3.50	4.25	2.75	0.49	2.00		2.19	1.25	2.50	1.00	1.74
Fragilaria biceps (Kützing) Lange-Bertalot										0.25			0.25		0.25
Fragilaria brevistriata Grunow													0.25		
Fragilaria leptostauron (Ehrenberg) Hustedt var. *leptostauron*															
Fragilaria ulna (Nitzsch) Lange-Bertalot var. *ulna*															
Fragilaria ulna (Nitzsch) Lange-Bertalot var. *acus* (Kützing) Lange-Bertalot				0.25											
Frustulia vulgaris (Thwaites) De Toni						0.50									
Gomphonema parvulum (Kützing) Kützing var. *parvulum*	9.48	7.79	8.73	4.25	4.00	8.00	7.25	1.74	8.50	13.90	2.19	5.00	1.50	0.74	1.74
Gomphonema pumilum (Grunow) Reichardt & Lange-Bertalot					0.75					0.25					
Mayamaea atomus (Kützing) Lange-Bertalot						0.50	1.25	0.74	0.25	0.25	0.22	0.25	1.00	0.74	0.75
Mayamaea atomus var. *permitis* (Hustedt) Lange-Bertalot					0.25										
Navicula arvensis Hustedt var. *maior* Lange-Bertalot												0.25	0.25	0.50	0.25
Navicula minuscula Grunow in Van Heurck					0.75										
Navicula molestiformis Hustedt															0.25
Navicula schroeteri Meister var. *symmetrica* (Patrick) Lange-Bertalot										0.25					
Navicula trivialis Lange-Bertalot var. *trivialis*							0.25		0.25			0.25			
Navicula veneta Kützing	4.74	2.43	2.00	2.00	6.25	2.00	8.75	4.67	6.00	4.47	4.82	3.00	0.75	24.55	2.99
Navicula viridula (Kützing) Ehrenberg			0.50												
NITZSCHIA A.H. Hassall															
Nitzschia acicularis (Kützing) W.M. Smith															0.25
Nitzschia amphibia Grunow f. *amphibia*	4.99	3.65	4.49	2.00	8.75	6.25	1.75	6.14	1.25	5.22	3.72	2.75	0.75	53.47	3.99
Nitzschia capitellata Hustedt in A. Schmidt	0.50	1.22		1.75	1.75	3.75	3.75	2.21	1.00		2.86	1.75	3.00	0.74	1.74
Nitzschia communis Rabenhorst		0.24	0.50	0.25		0.50		0.25		0.25		0.25	0.50		0.25
Nitzschia desertorum Hustedt		0.73	0.25	0.50	0.75	0.25	0.50	0.25	1.00		0.88		1.50	0.25	0.50
Nitzschia dissipata (Kützing) Grunow var. *dissipata*							0.50								
Nitzschia fonticola Grunow in Cleve et Möller															
Nitzschia frustulum (Kützing) Grunow var. *frustulum*												0.50			
Nitzschia heufleriana Grunow					0.50	0.25									
Nitzschia inconspicua Grunow															
Nitzschia intermedia Hantzsch ex Cleve & Grunow			0.25	0.50	0.25										
Nitzschia liebetruthii Rabenhorst										0.25	0.22			0.50	0.50
Nitzschia linearis (Agardh) W.M. Smith						0.75	0.25			0.25	0.22	0.50			
Nitzschia microcephala Grunow in Cleve & Moller					0.25										
Nitzschia nana Grunow in Van Heurck															
Nitzschia palea (Kützing) W. Smith	6.23	9.73	2.99	11.25	6.75	18.25	5.25	2.88	13.75	44.67	37.29	48.25	57.00	3.22	57.71
Nitzschia paleacea (Grunow) Grunow in van Heurck	5.49	8.27	5.74	1.50	8.75	4.75	11.00	3.93	4.25	1.25	1.75	9.75	8.50	2.48	1.74
Nitzschia paleaeformis Hustedt														0.50	0.50
Nitzschia supralitorea Lange-Bertalot		0.97	0.75	1.50	0.50	0.25		0.74	0.50						
Nitzschia umbonata (Ehrenberg) Lange-Bertalot	1.75	0.49	0.25	3.25	1.50	1.50	3.50	4.18	4.50	9.18	6.58	5.50	6.75	0.50	5.22
Pinnularia gibba Ehrenberg							0.25								
Pinnularia microstauron (Ehrenberg) Cleve		0.24													
Planothidium frequentissimum (Lange-Bertalot) Lange-Bertalot	5.24	7.56	2.74	5.00	5.50	2.50	2.75	3.69	3.75	1.25	1.32	1.00	1.50		3.99
Psammothidium montanum (Krasske) Mayama															
Rhopalodia gibberula (Ehrenberg) O.Müller															
Sellaphora pupula (Kützing) Mereschkowksy	2.74	2.92	0.50	2.50	2.75	5.25	3.00	2.21	1.50	7.20	1.75	1.50	1.50	0.25	1.24
Sellaphora seminulum (Grunow) D.G. Mann	0.25	0.73	0.75	0.25	0.50		0.25		0.50		0.44	0.25		0.74	
Surirella angusta Kützing		0.24			0.25										
Surirella ovalis Brebisson								0.49							
Thalassiosira weissflogii (Grunow) Fryxell & Hasle	0.50	1.22		1.25	1.00	4.50			0.75		0.44	1.00			0.50
Tryblionella apiculata Gregory						0.50	0.25						0.25		

Mooi River Below Mooi River Drive (MR 1)										
Taxon	Date									
	24/09/2005	09/10/2005	26/10/2005	10/11/2005	26/11/2005	13/12/2005	27/12/2005	05/01/2006	11/01/2006	26/01/2006
Achnanthidium minutissimum (Kützing) Czarnecki	1.25	18.00	9.23	0.99	1.50	1.48	1.99	1.49		
Achnanthidium biasolettianum (Grunow in Cleve & Grunow) Lange-Bertalot	3.25	1.25	3.75							
Achnanthidium eutrophilum (Lange-Bertalot) Lange-Bertalot		0.50		2.96	1.75	0.99	1.00	0.50	0.50	0.49
Achnanthidium saprophila (Kobayasi et Mayama) Round & Bukhtiyarova		3.75	2.20	25.68	11.50	2.00	14.93	11.94	8.25	4.17
Amphora libyca Ehrenberg								0.25	0.75	
Amphora pediculus (Kützing) Grunow	5.00	1.50	8.73	1.62	11.50	15.39	12.69	12.69	11.50	2.94
Asterionella formosa Hassall	0.25			0.49				0.25		
Aulacoseira ambigua (Grunow) Simonsen								0.25		
Aulacoseira granulata (Ehrenberg) Simonsen var.angustissima (O. Müller) Simonsen	2.50	0.50	0.25							
Aulacoseira muzzanensis (Meister) Krammer	0.75					0.25				
Caloneis bacillum (Grunow) Cleve		0.75	0.25		0.25	0.75	0.50	0.25	0.50	0.25
Cymbella kappii (Cholnoky) Cholnoky		4.00	0.75	1.73	2.50	1.23	0.50			
Cocconeis pediculus Ehrenberg	14.00	4.25	2.74	1.23		1.23			0.75	2.70
Cocconeis placentula Ehrenberg var. *placentula*	3.00	1.75			0.25					
Cocconeis placentula Ehrenberg var. *euglypta* (Ehrenberg) Grunow				3.46	0.25				0.75	0.49
Craticula ambigua (Ehrenberg) Mann									0.25	
Craticula halophila (Grunow ex Van Heurck) Mann	0.25		0.25							
Cyclotella meneghiniana Kützing	2.50	0.25		1.48	0.75	0.25	0.25	0.25	1.25	0.49
Cyclotella ocellata Pantocsek	11.25	2.75	1.75	1.48	2.25	1.48	1.49	3.48	1.75	1.72
Cyclotella pseudostelligera Hustedt	0.25	0.75								
Cymatopleura solea (Brebisson) W.Smith	1.75	0.75							0.75	
Cymbella cymbiformis Agardh		0.50		0.49	3.50	0.75	1.49			
Cymbella tumida (Brebisson) Van Heurck			0.25	0.49	0.25				0.75	1.23
Cymbella turgidula Grunow									0.25	
Diadesmis confervacea Kützing	0.25		0.25							1.72
Diatoma vulgaris Bory	1.00		0.75		0.25					
Encyonema silesiacum (Bleisch in Rabh.) D.G. Mann						0.25				
Encyonopsis microcephala (Grunow) Krammer	1.75	1.25			0.25		0.25	0.75	0.25	
Eolimna minima (Grunow) Lange-Bertalot	2.75	0.75	1.75	1.98	3.75	2.47	2.74	7.21	1.50	1.48
Eolimna subminuscula (Manguin) Moser Lange-Bertalot & Metzeltin	1.25		0.50		0.25			0.50	0.25	
Fallacia pygmaea (Kützing) Stickle & Mann	0.25				0.25		0.75			
Fistulifera saprophila (Lange-Bertalot & Bonik) Lange-Bertalot								0.25	0.25	0.49
Fragilaria biceps (Kützing) Lange-Bertalot					0.25		0.25	0.25		
Fragilaria brevistriata Grunow	6.50	3.25	2.99	3.74	1.50	5.93	7.71	5.72	8.25	8.88
Fragilaria construens (Ehrenberg) Grunow f. *construens*			1.25							
Fragilaria construens f. *binodis* (Ehrenberg) Hustedt			0.25							
Fragilaria crotonensis Kitton		0.25								
Fragilaria leptostauron (Ehrenberg) Hustedt var. *leptostauron*						0.75	0.50			0.25
Fragilaria parasitica (W. Smith) Grunow var. *parasitica*	0.25		0.25						0.25	
Fragilaria pinnata Ehrenberg var. *pinnata*	0.50	0.50	2.00	1.23	2.50	2.96	1.74	0.75	0.25	1.72
Fragilaria ulna (Nitzsch) Lange-Bertalot var. *ulna*	0.25									
Frustulia vulgaris (Thwaites) De Toni									0.50	1.23
Gomphonema affine Kützing						0.25				
Gomphonema gracile Ehrenberg					0.50					
Gomphonema parvulum (Kützing) Kützing var. *parvulum*	3.00	1.00	1.00	0.25	1.75	0.75	1.24	0.50	1.25	2.94
Gomphonema pumilum (Grunow) Reichardt & Lange-Bertalot	2.75	0.75	1.00	0.49	0.25	0.99		1.24	0.25	0.98
Gomphonema truncatum Ehrenberg				0.25	1.00			0.25		
Gyrosigma acuminatum (Kützing)Rabenhorst	0.25	0.50	0.25		0.25	0.49	0.75			
Gyrosigma attenuatum (Kützing) Rabenhorst									0.75	
Hippodonta capitata (Ehrenberg) Lange-Bertalot, Metzeltin & Witkowski		0.25		0.75	0.25	1.48	0.75	0.75	1.00	
Mayamaea atomus (Kützing) Lange-Bertalot							0.25			0.98
Melosira varians Agardh	0.75								0.50	
Navicula antonii Lange-Bertalot	0.50		0.50	0.75	0.50	1.48	1.24	1.24	0.75	2.46
Navicula capitatoradiata Germain	1.50	1.00	1.50	0.75	1.50	0.49	0.50			
Navicula cincta (Ehrenberg) Ralfs in Pritchard									0.25	0.25
Navicula cryptocephala Kützing	0.75	0.50	0.75	0.25	0.50	0.25	0.25	0.25	1.25	
Navicula cryptotenella Lange-Bertalot	1.50	1.00		2.47	1.25	2.96	1.00	0.75	7.50	5.64
Navicula lanceolata (Agardh) Ehrenberg										0.25
Navicula schroeteri Meister var. *symmetrica* (Patrick) Lange-Bertalot				0.49		0.25	0.25	1.24	2.00	3.68
Navicula tripunctata (O.F.Müller) Bory	14.25	1.25	8.23	7.47	4.50	2.72	3.48	3.23	6.50	2.70
Navicula trivialis Lange-Bertalot var. *trivialis*					0.25		0.75			
Navicula veneta Kützing	1.00	0.75	1.50	0.49	2.00	1.73	1.49	1.49	0.50	2.70
Navicula viridula (Kützing) Ehrenberg var. *rostellata* (Kützing) Cleve						0.75			0.25	
Navicula viridula (Kützing) Ehrenberg					0.75					
Nitzschia acicularis (Kützing) W.M. Smith				0.25	2.75	0.75				

Mooi River Below Mooi River Drive (MR 1)										
Taxon	Date									
	24/09/2005	09/10/2005	26/10/2005	10/11/2005	26/11/2005	13/12/2005	27/12/2005	05/01/2006	11/01/2006	26/01/2006
Nitzschia amphibia Grunow f. *amphibia*	0.25	1.00	1.50	0.99	0.25	1.23	0.25	3.23	9.00	0.74
Nitzschia capitellata Hustedt in A. Schmidt	0.25	0.50	1.00	1.48	0.25		0.50	0.50	0.50	0.25
Nitzschia desertorum Hustedt			0.50	1.48	0.50		1.00	0.50	0.75	
Nitzschia dissipata (Kützing) Grunow var. *dissipata*	0.50	2.25	5.24	2.72	0.75	0.99	0.75	0.75	0.25	1.48
Nitzschia frustulum (Kützing) Grunow var. *frustulum*							4.23	6.47	4.25	28.19
Nitzschia heufleriana Grunow	4.00	2.50	1.75							
Nitzschia inconspicua Grunow	0.50	0.50	2.49	1.23	2.00	2.72				
Nitzschia liebetruthii Rabenhorst						1.48	1.99	3.73	4.25	2.46
Nitzschia linearis (Agardh) W.M. Smith				3.30	7.50	4.69	1.99	1.49	0.25	0.98
Nitzschia microcephala Grunow in Cleve & Moller						0.75				
Nitzschia palea (Kützing) W. Smith	1.00	0.25	2.49	4.69	8.00	3.30	12.94	7.21	5.50	7.60
Nitzschia paleacea (Grunow) Grunow in van Heurck		1.75	4.74	5.68	7.00	7.65	9.24	11.94	8.00	
Nitzschia paleaeformis Hustedt		2.25			5.00				2.75	1.72
Nitzschia sinuata (Thwaites) Grunow var. *tabellaria* Grunow			0.25							
Nitzschia sociabilis Hustedt						0.25	0.25	0.25		0.49
Nitzschia subacicularis Hustedt in A.Schmidt							1.00	0.25	0.25	0.25
Nitzschia supralitorea Lange-Bertalot		1.50	0.25		1.00					
Nitzschia umbonata (Ehrenberg) Lange-Bertalot	0.25				0.25					
Pinnularia brebissonii (Kützing) Rabenhorst var. *brebissonii*					0.25				0.25	
Pinnularia microstauron (Ehrenberg) Cleve			0.25							
Planothidium frequentissimum (Lange-Bertalot) Lange-Bertalot	0.50					0.49	0.25		0.25	0.49
Reimeria uniseriata Sala, Guerrero & Ferrario		1.50	0.75	0.25	1.00	0.25		0.25	0.50	0.74
Rhoicosphenia abbreviata (C. Agardh) Lange-Bertalot	1.00	0.25	0.25	0.25				1.00		0.74
Sellaphora pupula (Kützing) Mereschkowksy	0.25	2.75	3.49	2.96	2.25	3.46	3.23	3.23	0.75	0.25
Sellaphora seminulum (Grunow) D.G. Mann		0.25					0.75			
Surirella angusta Kützing										0.25
Thalassiosira weissflogii (Grunow) Fryxell & Hasle	0.25			0.25	0.25	0.49	0.25	0.25	0.25	0.25
Tryblionella apiculata Gregory	4.00	1.50	2.24	2.22	0.50	0.99	0.75	1.24		1.23
Tryblionella gracilis W. Smith	0.25									

Mooi River Upstream of Retief Street Bridge (MR 2)									
Taxon	Date								
	24/09/2005	09/10/2005	10/11/2005	26/11/2005	13/12/2005	27/12/2005	05/01/2006	11/01/2006	26/01/2006
Achnanthidium minutissimum (Kützing) Czarnecki	1.25	2.49	0.50	1.00	1.00	0.75	0.25		0.75
Achnanthidium biasolettianum (Grunow in Cleve & Grunow) Lange-Bertalot		0.75							
Achnanthidium eutrophilum (Lange-Bertalot) Lange-Bertalot		0.75	0.50		0.25	0.25			
Achnanthidium saprophila (Kobayasi et Mayama) Round & Bukhtiyarova		0.50	2.49	2.25	2.50	2.47	0.75	2.24	1.00
Amphora inariensis Krammer					0.25				
Amphora pediculus (Kützing) Grunow		6.72	7.23	12.50	3.75	5.43	3.24	4.98	6.75
Asterionella formosa Hassall			1.25	0.25	0.25	0.25			
Aulacoseira granulata (Ehrenberg) Simonsen			0.50		0.25		0.50		
Aulacoseira granulata (Ehrenberg) Simonsen var.angustissima (O. Müller) Simonsen		1.74	0.50	0.75	0.25	0.49			
Aulacoseira muzzanensis (Meister) Krammer	0.25								
Caloneis bacillum (Grunow) Cleve			0.25	3.00	0.75	0.25	0.25		
Cymbella kappii (Cholnoky) Cholnoky		0.50	0.25		0.25				
Cocconeis pediculus Ehrenberg	0.25	8.76	14.96	2.75	4.25	1.73	1.75	2.74	
Cocconeis placentula Ehrenberg var. *placentula*		2.99							0.25
Cocconeis placentula Ehrenberg var. *euglypta* (Ehrenberg) Grunow			3.75	12.25	4.25	2.22	0.50	1.99	2.00
Craticula accomoda (Hustedt) Mann	1.75	0.75							
Cyclotella atomus Hustedt				0.25					
Cyclotella meneghiniana Kützing	25.75	6.47	4.24	3.50	2.25	2.96	1.25	1.49	0.75
Cyclotella ocellata Pantocsek	0.25	13.18	8.48	2.25	6.75	3.46	1.50	0.50	1.50
Cyclotella pseudostelligera Hustedt		0.50							
Cymatopleura solea (Brebisson) W.Smith			0.25						
Cymbella tumida (Brebisson) Van Heurck							0.25		
Diadesmis confervacea Kützing						0.25			
Diatoma vulgaris Bory		1.00	0.25	0.25	0.75	0.25	0.25		
Encyonema silesiacum (Bleisch in Rabh.) D.G. Mann				0.25					
Encyonopsis microcephala (Grunow) Krammer		1.00	0.75			0.25	0.25	0.25	
Eolimna minima (Grunow) Lange-Bertalot	7.25	6.47	3.99	11.25	2.50	1.98	1.00	1.99	4.50
Eolimna subminuscula (Manguin) Moser Lange-Bertalot & Metzeltin	13.50	2.74	1.25	3.00	1.75	4.44	3.99	1.99	8.00
Fallacia pygmaea (Kützing) Stickle & Mann						0.49			
Fistulifera saprophila (Lange-Bertalot & Bonik) Lange-Bertalot	1.25		0.75	1.00		0.25	2.00	1.74	1.00
Fragilaria biceps (Kützing) Lange-Bertalot			0.25		0.75	0.75	0.25		
Fragilaria brevistriata Grunow	0.25	1.99	3.24	1.50	2.25	2.47	1.00	1.49	1.75
Fragilaria capucina Desmazieres			0.25						
Fragilaria construens (Ehrenberg) Grunow f. *construens*								0.25	
Fragilaria pinnata Ehrenberg var. *pinnata*		0.25	0.25	0.25		0.49			0.25
Frustulia vulgaris (Thwaites) De Toni				0.50	0.25	0.25		0.25	
Gomphonema parvulum (Kützing) Kützing var. *parvulum*	11.00	4.48	1.75	1.75	4.25	3.96	2.74	0.50	4.50
Gomphonema pumilum (Grunow) Reichardt & Lange-Bertalot		1.49	1.75		0.75	1.73	2.00		2.25
Gomphonema truncatum Ehrenberg						0.25			
Fragilaria pinnata Ehrenberg var. *pinnata*		0.25			1.75	0.49			
Hippodonta capitata (Ehrenberg) Lange-Bertalot, Metzeltin & Witkowski		0.25				0.49	0.50		
Luticola ventricosa (Kützing) D.G. Mann								0.25	
Mayamaea atomus (Kützing) Lange-Bertalot				0.25					0.25
Mayamaea atomus var. *permitis* (Hustedt) Lange-Bertalot		0.75							
Melosira varians Agardh				0.25	0.25		0.50		
Navicula angusta Grunow						0.25			
Navicula antonii Lange-Bertalot		0.25	0.75	0.25	1.00	0.75	1.25	1.00	0.75
Navicula arvensis Hustedt var. *maior* Lange-Bertalot								0.75	0.75
Navicula capitatoradiata Germain		1.74	0.50	0.25	0.75	1.48	0.50	0.25	
Navicula cryptocephala Kützing			0.25		0.25	1.23	0.50		0.25
Navicula cryptotenella Lange-Bertalot		1.00	1.50	1.50	2.25	1.23	1.00	1.99	0.75
Navicula libonensis Schoeman						0.25			
Navicula longicephala Hustedt var. *vilaplanii* Sabater & Lange-Bertalot									0.50
Navicula schroeteri Meister var. *symmetrica* (Patrick) Lange-Bertalot			0.25	1.00	2.25	1.73	12.72	1.20	2.00
Navicula tripunctata (O.F.Müller) Bory	0.75	1.45	8.48	3.00	7.00	1.98	3.49	2.24	0.75
Navicula trivialis Lange-Bertalot var. *trivialis*				0.25	0.25	1.48	2.24	1.49	
Navicula veneta Kützing	3.25	1.74	0.75	1.50	2.50	4.20	8.98	7.96	4.50
Navicula viridula (Kützing) Ehrenberg var. *rostellata* (Kützing) Cleve						0.75	0.50		
NITZSCHIA A.H. Hassall									0.25
Nitzschia aciculans (Kützing) W.M. Smith				0.25	0.75				
Nitzschia amphibia Grunow f. *amphibia*	6.75	1.00	1.50	1.00	0.75	1.23	1.25	1.24	0.75
Nitzschia capitellata Hustedt in A. Schmidt	1.50	1.49	0.25		0.50	0.75	1.00	0.75	
Nitzschia desertorum Hustedt	0.50	0.25	0.50	2.00	0.75	1.98	2.24	0.50	0.25
Nitzschia dissipata (Kützing) Grunow var. *dissipata*		0.25	0.50		0.25	0.49	0.25		0.75
Nitzschia frustulum (Kützing) Grunow var. *frustulum*					5.00	2.96	5.24	8.29	4.00

Mooi River Upstream of Retief Street Bridge (MR 2)									
Taxon	Date								
	24/09/2005	09/10/2005	10/11/2005	26/11/2005	13/12/2005	27/12/2005	05/01/2006	11/01/2006	26/01/2006
Nitzschia heufleriana Grunow	0.25	1.49							
Nitzschia inconspicua Grunow		1.24	2.74	4.50				16.67	
Nitzschia intermedia Hantzsch ex Cleve & Grunow	1.25	0.25							
Nitzschia liebetruthii Rabenhorst					1.50	2.72	9.48	1.70	1.00
Nitzschia linearis (Agardh) W.M. Smith			1.25		2.25	3.46	1.25		0.25
Nitzschia palea (Kützing) W. Smith	9.50	2.49	4.99	5.50	11.00	17.28	13.22	6.22	6.00
Nitzschia paleacea (Grunow) Grunow in van Heurck	4.75	2.99	8.23	1.25	9.75	6.42	3.75	2.74	
Nitzschia paleaeformis Hustedt		1.00						0.75	0.25
Nitzschia recta Hantzsch in Rabenhorst				0.25					
Nitzschia sociabilis Hustedt					0.25		0.50		0.25
Nitzschia subacicularis Hustedt in A.Schmidt						1.48	0.25	0.25	
Nitzschia supralitorea Lange-Bertalot		0.50		0.25					
Nitzschia umbonata (Ehrenberg) Lange-Bertalot	3.50			1.00		0.99		0.25	
Placoneis placentula (Ehrenberg) Heinzerling		0.25							
Planothidium frequentissimum (Lange-Bertalot) Lange-Bertalot	3.75	0.75	1.50	3.00	1.50	0.25	1.50	1.74	3.00
Psammothidium montanum (Krasske) Mayama		0.75							
Reimeria uniseriata Sala, Guerrero & Ferrario		0.50	2.00	0.50	1.75	0.49	1.00	0.50	
Rhoicosphenia abbreviata (C. Agardh) Lange-Bertalot			0.25		0.25		0.25		0.25
Sellaphora pupula (Kützing) Mereschkowksy	1.50	1.24	2.24	1.75	2.00	1.73	1.50	0.25	0.50
Sellaphora seminulum (Grunow) D.G. Mann				0.25		0.49			0.50
Surirella angusta Kützing					0.75		0.25		
Surirella brebissonii Krammer & Lange-Bertalot		0.25							
Thalassiosira weissflogii (Grunow) Fryxell & Hasle			0.75	0.25	0.50	0.75	0.25		
Tryblionella apiculata Gregory		1.49	1.25	0.50	2.00	2.22	1.00	0.75	0.25

Appendix C

Diatom Index Scores

Wasgoedspruit at Approximate Origin (WS 0)										
Diatom Index	**Date**									
	27/04/2005	**05/05/2005**	**22/05/2005**	**02/06/2005**	**19/06/2005**	**03/07/2005**	**31/07/2005**	**13/08/2005**	**11/01/2006**	**26/01/2006**
Population	406	401	400	400	401	401	403	403	400	400
Number of species	21	22	32	30	34	34	20	15	17	25
SPI	6.1	3.3	5.8	6.7	7.0	7.0	11.2	11.4	2.9	6.0
SLA	9.0	10.1	7.9	9.3	8.6	8.6	8.4	8.4	8.0	8.9
DES	9.1	9.9	9.2	9.2	9.9	9.9	10.4	10.3	1.2	2.9
LMI	9.3	9.6	8.6	8.2	8.7	8.7	10.2	10.0	6.6	7.5
SHE	4.5	6.4	7.7	6.4	8.0	8.0	12.1	11.8	2.9	3.2
WAT	9.3	4.9	7.9	6.1	7.0	7.0	9.0	8.9	4.2	6.8
TDI	77.4	84.3	78.1	89.3	82.7	82.7	71.4	74.9	89.8	83.1
EPI-D	5.6	6.3	6.6	6.5	6.6	6.6	8.2	8.2	5.2	6.5
ROT	5.1	7.0	7.8	7.0	7.7	7.7	9.1	9.0	5.0	5.5
GDI	9.2	10.7	7.7	8.7	9.0	9.0	12.0	11.8	3.4	7.1
CEC	6.5	5.6	4.6	5.6	5.4	5.4	4.0	0.0	1.8	5.0
BDI	11.5	9.2	8.7	9.9	9.4	9.4	11.6	11.9	7.8	8.4
APDI	6.7	5.6	7.1	8.0	8.4	8.4	12.5	12.8	7.3	6.2

Acronyms for the Diatom Indices Used	
Acronym	**Diatom index**
DES	Descy's Index
GDI	Generic Diatom Index
EPI-D	Eutrophication pollution Index Germany
LMI	Leclercq and Maquet 's Index
SLA	Sladecek's Index
SPI	Specific Pollution Sensitivity Index
BDI	Biological Diatom Index
APDI	Artois-Picardie Diatom Index
CEC	Commission of Economical Community Index
SHE	Schiefele and Schreiner's Index
TDI	Trophic Diatom Index
WAT	Watanabe Index
ROT	Rott's Index

Wasgoedspruit Below Railway Bridge (WS 1)											
Diatom Index	**Date**										
	20/04/2005	**05/05/2005**	**22/05/2005**	**02/06/2005**	**19/06/2005**	**19/06/2005**	**03/07/2005**	**03/07/2005**	**17/07/2005**	**17/07/2005**	**31/07/2005**
Population	400	408	405	424	401	401	404	403	408	405	400
Number of species	17	21	17	16	18	21	19	19	16	21	26
SPI	3.2	3.0	3.9	2.8	2.3	5.2	2.9	4.4	2.9	3.7	4.1
SLA	6.9	7.4	7.4	7.0	6.8	8.0	6.9	8.4	7.8	8.4	8.4
DES	3.8	4.1	5.7	4.6	2.8	7.7	4.3	5.9	4.6	6.4	6.3
LMI	6.6	6.6	6.9	6.1	5.9	7.8	6.4	6.9	6.7	7.3	7.5
SHE	3.2	3.2	3.5	5.1	3.9	3.9	4.5	2.6	2.3	3.9	3.5
WAT	3.0	2.3	2.7	3.8	3.0	5.7	3.1	5.3	6.3	5.8	6.3
TDI	97.7	98.5	98.3	95.0	94.9	97.4	95.5	98.4	94.5	94.4	94.4
EPI-D	5.8	5.6	5.7	5.3	4.7	7.5	5.0	7.2	6.0	7.1	7.0
ROT	4.1	4.3	4.6	3.7	3.6	4.8	4.6	4.4	3.8	4.8	5.1
GDI	6.9	6.2	8.4	6.5	5.4	9.9	6.2	8.9	5.6	6.6	7.0
CEC	3.1	2.7	3.7	2.9	2.7	3.7	3.3	3.3	2.5	3.1	3.9
BDI	7.3	7.1	7.3	5.7	5.9	7.5	5.8	8.0	6.6	7.4	7.9
APDI	5.6	5.7	5.6	5.1	5.1	5.9	4.9	6.2	6.7	6.2	5.9

Diatom Index	**Date**				
	31/07/2005	**13/08/2005**	**13/08/2005**	**28/08/2005**	**28/08/2005**
Population	419	408	418	415	406
Number of species	21	23	18	18	20
SPI	1.9	2.8	5.7	5.2	2.9
SLA	7.6	8.2	8.4	8.3	8.1
DES	3.2	5.1	9.3	8.4	5.0
LMI	6.1	7.3	8.4	8.2	7.2
SHE	2.6	3.5	5.8	4.2	3.5
WAT	4.5	4.6	5.1	5.4	4.5
TDI	91.6	91.7	87.8	92.2	91.4
EPI-D	4.3	5.6	7.2	7.2	5.5
ROT	4.2	5.0	6.6	5.8	4.9
GDI	3.4	3.7	9.9	10.1	3.9
CEC	2.1	3.1	5.4	4.8	3.3
BDI	5.7	7.7	7.4	8.3	8.2
APDI	4.4	5.7	6.0	6.2	5.7

Wasgoedspruit Adjacent to Wallis Street (WS 2)											
Diatom Index	**Date**										
	30/03/2005	**30/03/2005**	**20/04/2005**	**05/05/2005**	**19/05/2005**	**02/06/2005**	**19/06/2005**	**03/07/2005**	**17/07/2005**	**31/07/2005**	**13/08/2005**
Population	410	407	418	414	402	400	402	402	400	412	420
Number of species	13	13	12	14	15	13	23	22	19	22	18
SPI	2.7	2.1	5.8	4.4	4.2	4.9	4.3	4.5	2.2	3.6	4.6
SLA	6.9	7.1	6	7.2	6.9	7.1	7.1	8.7	7.7	7.7	8.3
DES	3.3	2.3	7.3	5.1	6.8	6.9	6.9	5.2	3.6	5	8
LMI	5.7	5.7	6	6.1	6.5	6.9	6.6	6.2	6.7	6.7	8.4
SHE	3.5	3.2	3.5	3.2	5.1	5.1	4.8	2	2.3	3.5	4.8
WAT	2.9	2.4	1.6	3.2	2.9	3	4.3	6.3	3	4.1	5
TDI	96.1	97.7	99.9	99.3	97.3	98.7	95.3	99.1	95.4	95.6	91
EPI-D	4.4	4.9	3.8	4.6	5.1	5.4	5.5	6.6	5.6	5.7	6.6
ROT	3.4	3.5	3.8	4.2	4.4	5.1	5	4.6	4.2	4.7	6.2
GDI	4.8	4	9.2	8.2	8.3	8.8	8.8	10	5	6.4	6.4
CEC	2.7	2.1	4.8	4.2	4	4.4	4	4.4	1.8	3.5	4.6
BDI	7.1	6.7	8.2	8.2	6.5	7	6.7	7.8	6.2	7.6	8.5
APDI	5.8	5.8	5.8	5.7	5.5	5.5	5.3	5.5	5.8	5.9	5.8

Wasgoedspruit Upstream of van Riebeeck Street Bridge (WS 3)											
Diatom Index	**Date**										
	30/03/2005	**20/04/2005**	**20/04/2005**	**05/05/2005**	**19/05/2005**	**19/05/2005**	**02/06/2005**	**02/06/2005**	**19/06/2005**	**19/06/2005**	**03/07/2005**
Population	402	400	402	401	406	404	406	401	400	400	403
Number of species	13	22	21	13	18	15	15	16	19	18	19
SPI	2.7	4.1	5.1	4.6	5.2	4.8	5.7	4.8	5.0	4.5	4.3
SLA	6.7	7.1	6.8	6.8	7.1	7.1	7.0	6.9	7.0	7.1	7.3
DES	2.3	5.9	6.8	5.7	6.7	7.1	6.8	6.8	7.0	6.1	5.5
LMI	5.8	6.7	6.5	6.3	6.8	6.9	6.3	6.4	6.3	6.6	6.6
SHE	3.2	3.9	4.2	3.5	5.1	4.5	3.9	4.5	5.1	4.2	3.5
WAT	2.1	3.0	2.7	2.6	4.0	2.9	3.6	3.7	3.9	3.8	3.9
TDI	99.1	98.2	98.5	99.2	98.5	98.7	96.8	98.1	97.1	98.7	98.4
EPI-D	4.3	5.2	4.9	4.5	5.5	5.4	4.6	5.2	5.3	5.4	5.3
ROT	3.8	4.5	4.4	4.3	5.2	4.7	4.5	4.5	5.0	4.5	4.6
GDI	5.6	8.3	8.9	8.6	8.9	9.1	9.9	9.2	8.5	9.0	8.9
CEC	3.1	3.9	4.6	4.2	4.4	4.4	4.8	4.4	4.4	4.2	4.2
BDI	7.0	7.1	7.7	7.9	6.8	7.1	8.0	7.0	7.0	7.0	7.3
APDI	5.7	5.5	5.8	5.6	5.4	5.6	5.9	5.6	5.7	5.6	5.5

Diatom Index	**Date**										
	17/07/2005	**17/07/2005**	**31/07/2005**	**31/07/2005**	**13/08/2005**	**13/08/2005**	**28/08/2005**	**28/08/2005**	**11/09/2005**	**11/09/2005**	**09/10/2005**
Population	407	403	404	401	410	407	401	411	401	400	400
Number of species	19	21	22	21	22	24	16	23	20	24	29
SPI	4.4	4.0	4.3	3.9	4.8	4.4	5.0	4.9	5.5	4.6	4.8
SLA	7.3	7.4	7.5	7.5	7.1	7.4	7.2	7.6	6.8	7.5	7.6
DES	4.9	5.3	5.8	5.6	6.5	6.7	6.8	7.0	7.1	7.2	6.8
LMI	6.3	6.9	6.8	6.7	7.2	7.5	7.3	7.6	7.0	7.3	7.7
SHE	2.9	3.9	4.2	3.9	4.2	4.2	4.5	4.5	4.5	4.8	5.8
WAT	4.0	3.0	4.4	3.9	3.2	4.4	4.1	4.5	3.6	4.5	5.2
TDI	99.0	98.0	96.5	96.9	96.3	95.4	97.6	95.8	97.7	93.3	94.0
EPI-D	4.8	5.7	6.2	5.7	5.0	5.7	5.6	6.3	5.0	6.1	6.1
ROT	4.3	4.7	5.0	4.6	4.7	5.1	4.8	5.4	4.8	5.1	5.9
GDI	8.8	7.5	7.4	10.1	7.9	7.4	8.2	7.9	8.4	7.1	7.3
CEC	4.2	3.7	3.9	3.9	4.6	4.2	4.4	4.2	5.0	4.2	4.6
BDI	7.8	7.6	7.9	7.5	8.8	7.8	8.0	7.7	8.3	8.1	7.5
APDI	5.9	5.8	5.9	5.9	6.1	5.7	5.9	6.1	5.9	6.3	5.8

Wasgoedspruit Upstream of van Riebeeck Street Bridge (WS 3)										
Diatom Index	**Date**									
	09/10/2005	**26/10/2005**	**26/10/2005**	**10/11/2005**	**13/12/2005**	**27/12/2005**	**27/12/2005**	**05/01/2006**	**11/01/2006**	**11/01/2006**
Population	400	400	407	400	403	456	400	400	404	402
Number of species	27	24	20	21	19	21	26	24	19	27
SPI	4.1	4.6	3.4	3.9	2.4	2.5	2.3	1.9	4.2	2.0
SLA	8.0	7.8	7.5	7.9	7.8	7.5	7.8	7.6	8.0	7.5
DES	5.6	7.3	4.7	5.4	3.8	4.1	3.8	2.9	5.8	2.5
LMI	7.2	7.7	6.6	7.1	6.2	6.7	6.7	6.3	8.0	6.1
SHE	4.8	5.8	4.5	5.1	4.5	4.2	3.9	3.2	9.2	3.2
WAT	4.6	6.0	4.6	5.0	4.5	4.2	4.2	3.7	9.1	3.5
TDI	95.4	93.3	96.4	96.5	96.0	93.8	93.4	93.0	96.3	95.8
EPI-D	6.0	5.7	5.7	6.0	6.3	5.3	5.8	5.5	6.3	6.0
ROT	5.5	5.8	4.7	5.0	4.5	4.5	4.7	4.2	7.6	4.1
GDI	6.6	7.0	6.7	6.8	4.8	4.2	3.8	2.7	4.6	3.3
CEC	3.9	4.6	3.5	3.9	2.5	2.9	2.7	2.5	3.5	1.8
BDI	7.1	7.1	6.8	7.1	5.9	6.9	6.9	6.9	4.8	6.5
APDI	6.0	5.7	5.8	6.1	5.3	5.4	5.8	6.1	4.5	5.6

Mooi River Below Mooi River Drive (MR 1)										
Diatom Index	**Date**									
	24/09/2005	**09/10/2005**	**26/10/2005**	**10/11/2005**	**26/11/2005**	**13/12/2005**	**27/12/2005**	**05/01/2006**	**11/01/2006**	**26/01/2006**
Population	400	400	401	405	400	405	402	402	401	408
Number of species	46	43	43	40	51	44	44	43	51	43
SPI	13.2	14.9	12.5	10.5	9.6	10.6	8.2	8.6	9.6	8.0
SLA	11.1	12.7	11.6	11.2	10.6	11.4	10.5	11.3	11.3	11.2
DES	16.7	16.9	16.1	15.3	14.2	15.4	13.0	13.3	13.0	14.2
LMI	12.3	12.7	12.0	11.5	10.8	11.7	10.5	10.6	11.4	10.9
SHE	13.0	14.0	10.5	9.2	9.9	9.9	8.9	9.2	10.8	9.9
WAT	12.6	14.3	10.6	9.0	9.6	9.5	8.5	9.2	10.2	9.9
TDI	69.6	56.7	59.2	59.8	69.2	65.2	65.8	70.4	75.4	72.3
EPI-D	11.7	14.3	11.8	9.8	9.9	10.4	9.9	9.4	10.2	9.0
ROT	13.2	13.4	10.9	9.9	11.0	10.8	10.4	10.6	11.6	11.9
GDI	11.6	12.6	11.9	11.1	8.8	10.2	8.7	7.9	8.2	7.4
CEC	12.6	15.1	12.6	10.9	9.6	10.9	8.8	8.8	9.7	6.9
BDI	10.8	14.1	11.2	9.1	9.8	9.2	9.1	8.9	9.0	8.1
APDI	12.9	14.0	12.1	11.3	10.2	10.5	9.5	9.3	10.1	8.1

Mooi River Upstream of Retief Street Bridge (MR 2)									
Diatom Index	**Date**								
	24/09/2005	**09/10/2005**	**10/11/2005**	**26/11/2005**	**13/12/2005**	**27/12/2005**	**05/01/2006**	**11/01/2006**	**26/01/2006**
Population	400	402	401	401	400	405	401	402	400
Number of species	23	47	46	45	50	55	48	38	38
SPI	4.5	11	11.2	9.1	8.6	6.5	5.9	6.7	6.7
SLA	7.7	10.6	11.2	10.7	10.4	9.1	9.5	10.6	10.4
DES	7.1	15.2	14.5	14.1	12.6	10.8	12.2	14.1	14
LMI	7.5	11.1	11.7	11.2	10.8	9.4	9.8	10.4	10.4
SHE	5.4	10.5	11.1	8.9	8.9	7.7	7.7	8.9	8.3
WAT	4.9	11	11.8	10.6	10	8.8	9.2	10	9.6
TDI	94.4	76.8	75.9	78.6	77.9	79.3	81.8	81.1	82.8
EPI-D	6.4	10.2	10.5	9.2	10.1	8.2	7.4	7.5	7.6
ROT	5.4	11.7	11.6	10.1	10.3	9	9.4	10.5	10.6
GDI	8.4	10.7	9.7	8.6	8.3	7.2	7.3	6.1	5.8
CEC	4.2	9.9	10.3	9.6	8	6.9	5.8	5.8	6.5
BDI	6.9	9.7	9.9	10.2	9.4	8.5	7.7	7.2	7.2
APDI	6.1	11.3	11.6	9.8	9.5	7.8	7.1	7	6.7

Appendix D

MSSA 2005 Contribution

MORPHOLOGY AND TAXONOMY OF DIATOMS IN AN URBAN STREAM

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The Wasgoedspruit is a small urban stream formed by the confluence of two smaller streams found to the west of Potchefstroom. Approximately 4.5 km in length, the Wasgoedspruit flows past the industrial area of Potchefstroom in a natural stream-bed after which it is canalized to guide the stream into the Mooi River as it flows through Potchefstroom. The canalized section of the Wasgoedspruit was the main focus of this study.

Diatoms (Bacillariophyta) make up a significant part of the algal biomass in rivers[1]. Since aquatic organisms (diatoms) respond directly to nutrients, stressors (toxicants) and physical conditions in the water body[2], these organisms reflect and are sensitive to changes in water quality. Indices derived from the tolerance limits of individual diatom cells to water quality variables provide a good indication of the trophic state and pollution levels of water bodies.

While other biomonitoring techniques are used in South Africa, such as those based on macro-invertebrates, these techniques are limited by factors such as hydrology, substrate, habitat, food availability, seasonality and distribution patchiness[2]. Periphytic algae are one of a few groups of aquatic organisms that can successfully establish in the hostile environment within canalized streams. Diatoms growing on solid substrata (e.g. cobbles, boulders, pebbles, concrete, etc.) are referred to as the epilithon. The epilithon is the preferred communities for monitoring water quality[3]. These epilithic communities are in abundance in the canalized section of this particular urban waterway, further strengthening the case for diatoms as the bioindicators of choice. Diatoms, however, have a complex taxonomy and have to be identified correctly if the results of diatom-based water quality indices are to be accurate. The aim of the present study was to correctly identify the different diatom species in the Wasgoedspruit using different microscopy techniques.

Diatoms samples were collected from 3 sites in the 1.2 km canalized stretch of the Wasgoedspruit. A control sample was collected upstream from the confluence of the two smaller streams.

Living diatom material was studied under the light microscope and the diatoms photographed to record chloroplast structures (Fig. 1). The samples were then processed and the organic matter removed using acid-oxidation according to standard methods[4] leaving only the silica diatom frustules. Permanent slides were made by mounting the processed material in the high resolution diatom mountant Pleurax (refractive index 1.73). The general characteristics of diatoms, such as size and shape were studied under the light microscope (Fig. 2).

The processed material was also prepared for the Scanning Electron Microscope (SEM) by suspension in 96% ethanol and filtering the suspension through a 1.5 µm Millipore® membrane, which was left to air dry and then sputter coated with gold palladium on an aluminium SEM stub. Using SEM, the fine structures of the diatom frustules were examined in order to document the characteristic features of the diatom species encountered. Photomicrographs were taken of the different species and their taxonomically important characteristics (Fig. 3).

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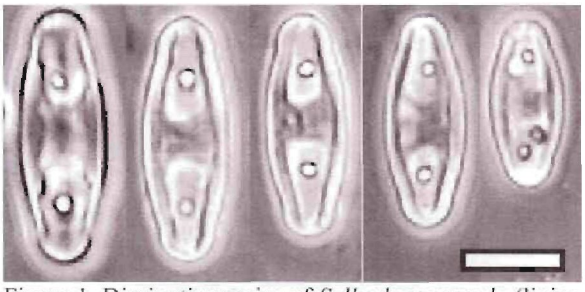

Figure 1. Diminutive series of *Sellaphora pupula* (living diatom material). Scale bar = 10µm

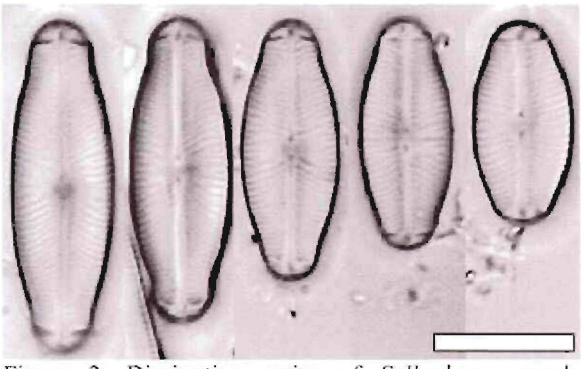

Figure 2. Diminutive series of *Sellaphora pupula* (processed diatom material). Scale bar = 10µm.

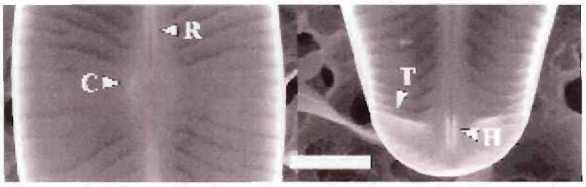

Figure 3. Fine structure C = central nodule, R = raphe, H = helictoglossa, T = thickenings. Scale bar = 2 µm.

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Appendix E

IDS 2006 Contribution

BIOLOGICAL INDICATORS OF WATER QUALITY IN AN URBAN WATERWAY: CAN DIATOMS REFLECT SHORT TERM SPATIAL AND TEMPORAL CHANGES IN WATER QUALITY?

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The Wasgoedspruit is a small urban waterway formed by the confluence of two small streams to the west of Potchefstroom in the North-West Province of South Africa. Approximately 4.5 km in length, the Wasgoedspruit (lit. "the laundry stream") flows past the industrial area of Potchefstroom in a natural stream-bed, after which it is canalized to guide the stream into the Mooi River. The canalized section of the Wasgoedspruit was the main focus of this study.

Indices derived from the tolerance limits of individual diatom taxa to water quality variables provide a good indication of the trophic state and the degree of pollution in a water body. Periphytic algae are one of a few groups of aquatic organisms that can successfully establish within the hostile environment of canalized streams. The diatom community growing on solid substrata (e.g. cobbles, boulders, pebbles, concrete, etc.) is referred to as the epilithon, this is the preferred community for monitoring water quality. These epilithic communities are abundant in the canalized section of this particular urban water way. Previous studies, in South Africa and elsewhere, have shown that both long and short term changes in water quality in rivers were successfully reflected by the index scores derived from the combined tolerance levels of epilithic diatom taxa. The objectives of this study were, firstly, to evaluate the efficacy with which diatom-based indices reflect changes in water quality over the short term in an urban water way, and secondly, to what extent do changes in water quality in the Wasgoedspruit influence the water quality and diatom community composition of the Mooi River.

Physical and chemical samples were collected weekly from three sites in the 1.2 km canalized stretch of the Wasgoedspruit, while biological samples were collected bi-weekly at the same sites. During the second phase of the study, the same sampling routine was applied to one site in the Wasgoedspruit and one site upstream and one site downstream of its confluence with the Mooi River. Diatom index scores were be calculated for each site and compared, using statistical techniques, to the values obtained from physical and chemical water quality analyses. The relationship between diatom community composition and water quality variables was determined using Canonical Correspondence Analysis. The results of this study will be presented and discussed and the value of diatoms as indicators in urban canals will be evaluated.