WATER QUALITY OF THE MODDER RIVER

by

NADENE KONING

B.Sc. Hons. (U.O.F.S.)

Dissertation submitted in fulfilment of the
requirements for the degree

MAGISTER SCIENTIAE

in the Faculty of Science
Department of Botany and Genetics
University of the Orange Free State
Bloemfontein

November, 1998

Supervisor: Prof. J.U. Grobbelaar
D.Sc. (U.O.F.S.)

Co-supervisor: Dr. J.C. Roos
Ph.D. (U.O.F.S.)

TABLE OF CONTENTS

HAPTER 3: PHYTOPLANKTON OF THE MODDER RIVER

KNOWLEDGEMENTS:

ish to express my sincere thanks to the following persons and institutions, who made it ssible for me to complete this study.

My supervisor, Prof. J.U. Grobbelaar, for his useful advice, constructive criticism, guidance and encouragement.

My co-supervisor, Dr. J.C. Roos, for his answers on my many questions, his endless patience and constant encouragement.

The University of the Orange Free State for providing me with the opportunity and the facilities to conduct this study.

My family and in particular my parents, to whom I dedicate this dissertation, for their love, support and encouragement.

Mr. J.A. van der Heever, for his help and advice with the toxicity tests, as well as the solutions he provided for many unforeseen problems.

All my friends, in particular Maryna, Lisa, Linda, Hardi and Neil, for their support and friendship.

The Department of Water Affairs and Forestry, for making their national database available to us.

The Free State Technikon, for providing the transport to and from the sample sites, their assistance with the laboratory analyses and the bacterial data they made available to us.

TO MY PARENTS

CHAPTER 1

INTRODUCTION

1.1) GENERAL INTRODUCTION

Rivers contain only a fraction of the world's freshwater (Allan, 1995). Yet they are vital components of the hydrological cycle, annually transporting 32 000-37 000 km^3 of water to the oceans of the world. Around the world, most major cities are situated next to or near rivers. This is because rivers are a source of potable water, serve as recipients of sewage outflow, and in some cases are used for transportation and hydro-electricity generation.

Although South Africa is a country well endowed with natural resources, water is scarce (Toerien *et al.*, 1975). Only 11 percent of the total rainfall reaches the rivers. The remainder is being lost to evaporation and groundwater sources. The total surface run-off in South Africa is on average only 51 km^3 per annum (Koch *et al.,* 1990).

Although water is limited and in spite of effluent quality regulations, salinisation and eutrophication are the major problems threatening water supplies in South Africa. According to Koch *et al.,* (1990), salinisation is common in mines and industries and is mainly due to ignorance. Salinisation is the process by which the concentration of total dissolved solids, in inland waters, is increased. It is not only industrial and mining developments which result in the salinisation of water bodies. Most activities involving the use of water tend to add salt and also to concentrate salt with the consumptive use of the water. Thus, as is the case in South Africa, salinisation is often associated with the intensive use and re-use of limited water supplies.

The importance of salinisation as a measure of water quality lies in the fact that the usefulness of water, for most purposes, decreases with increased salt content. The cost of increased salinity in water supplied to industries varies according to the type of industry involved, but the cost of any increase in general is high. Du Plessis & Van Veelen (1991) calculated that an increase, from 300 to 500 mg/l in the salt content, could cost Rand Water (a water supply authority in South Africa) users R76 million per annum in terms of water purification. A further increase in TDS to 800 mg/l could cost an additional R63 million per annum.

The quantity of water available in a river system as well as development in the catchment area have an impact on the quality and salinity of its natural waters. As the water resources of South Africa are limited, and also many rivers are temporal; salinisation is a problem deserving urgent attention.

Eutrophication is the enrichment of water with plant nutrients, particularly phosphorus and nitrogen and the consequent excessive growth of free-floating algae and floating, or rooted, macrophytes. Such blooms typically turn the water green, due to the presence of high concentration of algae often concentrating as thick surface slicks and scums. These could lead to aesthetic, health and odour problems (Bowling & Baker, 1996).

Eutrophication was only identified as a major problem in South Africa in 1975 when a survey was done on 98 impoundments to determine their trophic status (Toerien *et al.*, 1975). About half of the impoundments were found to be low in plant nutrients, eleven were heavily eutrophic and the rest were mesotrophic. They concluded that urban industrial development, which gives rise to nutrient-rich effluents, was the main cause of eutrophication. It is an important fact that the six most eutrophic impoundments received large quantities of secondary purified sewage effluents. The importance of eutrophication lies in its consequences, namely: impaired water body aesthetics, increased water treatment costs, taste and odour problems, potential health risks and unwanted algal blooms (Hynes, 1970; Westlake, 1975; Du Plessis & van Veelen, 1991).

1.2) THE NATURE OF RIVERS

A river may be viewed as a series of reaches, or sectors, each receiving and discharging water, sediments, organic matter and nutrients (Petts, 1992). A principal ecological process in rivers, distinguishing them from other types of ecosystems, is the unidirectional transport of materials, from the headwaters to the sea. Just as the character of lentic waters reflect the dominant features of light, heat and resulting water masses, so too lotic habitats are best described by flow, erosion deposition and channel form (Jeffries & Mills, 1990). Because all rivers flow between multiple geographical boundaries, which introduce numerous factors such as altitude, climate, topography, geochemistry, hydrology and catchment land use, all in turn influencing the distribution of species, communities and habitats, they are very intricate systems (Hynes, 1970; Beaumont, 1975; Petts, 1992; Jeffries & Mills, 1990; Davies *et al.*, 1993).

Five principles govern our understanding of the characteristics of rivers (Petts, 1992):

i) Rivers are systems characterised by longitudinal, lateral and vertical fluxes

Rivers are complex four-dimensional systems, consisting of longitudinal, lateral and vertical spatial components and time, within which communities must exist and respond to environmental variables. It is widely accepted that within lotic systems physical, chemical and biological processes are dominated by longitudinal fluxes over long time-scales (10000 years), however, over a shorter time-scale (<100 years) the dominance of longitudinal processes are restricted to headwater zones and to lowland floodplain systems, and ecosystem dynamics are dominated by lateral exchanges and influenced by vertical fluxes.

ii) Rivers are systems influenced by hydrology and geomorphology

Lotic ecosystems are largely determined by a range of hydrological, hydraulic and morphological variables: discharge (annual mean, seasonal regime, short-term variability) and hydraulics (flow depth, velocity, shear stress and channel morphology).

Discharge (one of the most important characteristics of a river) is a function of the velocity and cross-section of the river (Hynes, 1970, Beaumont, 1975, Jeffries & Mills, 1990), the formula being (Beaumont, 1975):

$$Q = A \times V$$

where Q = discharge

A = cross-section of channel

V = average velocity of flow

In the hydrological cycle (Hynes, 1970; Beaumont, 1975), precipitation is considered to be the major input in drainage basin systems and evapotranspiration as the most important reason for water loss. Also, the most important water reservoirs in a drainage basin system are soil moisture storage and movement. The three major factors controlling the quantity and movement of moisture in the soil are the size and distribution of the soil pores, the attraction of soil solids to water molecules and the attraction of water molecules to each other.

If the intensity of precipitation is greater than the infiltration rate of the soil, water would accumulate on the ground surface and eventually run over it as overland flow. Overland flow appears to occur more commonly in arid and semi-arid regions, where the vegetation cover is sparse (Beaumont, 1975). Erosion of the land and transport of sediments is greater with overland flow, whereas more dissolved materials are transported by sub-surface flows (Allan, 1995).

iii) Rivers are structured by food webs

River flow creates different habitats. Within each of these habitats, are interacting populations and communities. Habitat diversity, especially the complexity of habitats along the river reaches, on, and within the river bed, has an important influence on the shape of the food web. However, the trophic levels in a river ecosystem are very complex and difficult to investigate. The simple model of a series of trophic levels (supplied with energy by solar radiation) does not apply very well to a river ecosystem, because of the unidirectional flow of water (Hynes, 1970). Everything released into solution by metabolism tends to flow downstream and is not recycled on the spot. Two types of materials are present in rivers: allochthonous material, provided from outside sources and autochthonous material, which is derived from processes in the river itself. In rivers the allochthonous material is the more important source of energy. Since in running water the energy is not all derived from radiation itself, the zone of riparian vegetation largely determines the balance between autotrophy and heterotrophy in headwater streams, through light alteration and supply of allochthonous organic matter (Cummins, 1979)

iv) Rivers have spiralling, delivery and retention characteristics

Rivers ecosystems are characterised not only by downstream transfers but also by important storage characteristics. The dissolved output from a catchment to river water is a function of the inputs, in-system storage and the phase transformations that take place during the constituents residence time in the catchment (Edwards, 1974). Input of nutrients nearly always exceeds output (Golterman, 1975a) and accumulation in sediments accounts for most of the difference. This accumulation should never be regarded as a loss because there are interactions between sediments and overlying waters. Outflow, evaporation and harvest can be considered as losses, with outflow being the greatest.

v) Rivers are characterised by change

The chemistry of freshwater is quite variable, usually more so in rivers than in lakes. Natural spatial variation is mainly determined by the type of rocks available for weathering, the climate and by the composition of rain (Gibbs, 1970, Hynes, 1970, Petts, 1992, Allan, 1995). River chemistry also varies over time, due to the influence of seasonal changes in discharge regime, precipitation input and biological activity (Allan, 1995). Biota respond to these changing factors, and this in turn generates dynamic biological interactions. Large rivers require a great quantity of water to alter their pattern of discharge. Smaller streams are, therefore, much less stable than large ones (Hynes, 1970).

Other than the above, several alternative concepts have been put forward in an attempt to understand river ecosystems. One of the most widely known is the River Continuum Concept (RCC) (Vannote *et al.*, 1980). The RCC logically regards the entire lotic system as a continuous drainage basin gradient and states that, from the headwaters to the mouth of any river, there is a gradation of physical-chemical conditions that trigger a series of responses within riverine populations, which in turn result in a continuum of biotic adjustments and consistent patterns of loading, transport, utilisation and storage of water and matter, along its entire length. Headwaters tend towards detritus-based heterotrophy (primary production(P)/respiration(R) < 1) with little primary production, relying on allochthonous input of organic material for energy. This is because of restricted light, a consequence of shading by riparian vegetation (Cummins, 1979). Downstream, in the mid-sized streams, the system becomes more autotrophic (P/R > 1), with increased production of autochthonous organic material, because of reduced riparian vegetation and relatively shallow, clear, water. Further downstream, in the large rivers, the system becomes heterotrophic again, due to light attenuation by depth and turbidity. However, at the estuary, the velocity of the current decreases and suspended solids flocculate. The invertebrate fauna of the upper reaches are dominated by shredders and collectors, which give way to grazers and collectors in the middle reaches and to collectors in the lower reaches. Species richness maximises in the middle reaches of the stream continuum (Stanford & Ward, 1979) The downstream communities, in the continuum, depend on the inefficient use of nutrients from upstream communities. Thus, lotic ecosystems receive a continuous supply of nutrients from upstream so one would not expect nutrients to exert primary limitation on algal and microbial biomass (Elwood *et al.*, 1981).

The most common criticisms of the RCC are that physical stream parameters are often interrupted by the legacy of events and lithological changes. The RCC notion of succession absence is also rejected (Allanson *et al.*, 1990). Cushing *et al.* (1983), concluded that it is unsatisfactory to classify streams into subjectively defined, isolated reaches, thus losing sight of interactions between reaches and obscuring important ecological similarities. They feel that streams exhibit more continuity along their length and among themselves and that the differences are simply local expressions of general geomorphic processes. There are thus two views regarding the central concepts of stream ecosystem functions. The RCC, and the unstructured view (rivers are an unstructured collection of opportunists, surviving and increasing when conditions are favourable in between floods and droughts).

The RCC was rendered inapplicable on disturbed rivers by the Serial Discontinuity Concept (SDC) (Stanford & Ward, 1979). The SDC assumes the validity of the RCC and proposes that dams act as disruptions to the natural continuum of hydrological, physico-chemical and biotic changes in an impounded river. It is particularly the discharge from these impoundments that appears to be detrimental to riverine biotas.

Two other important hypotheses regarding rivers are the Intermediate Disturbance Hypothesis (IDH) (Connel, 1978) and the Nutrient Spiralling Concept (NSC) (Webster & Patten, 1979; Newbold, 1992). The IDH predicts that biodiversity will be the greatest in communities subjected to moderate levels of disturbance (Connel, 1978). It is based on the fact that a high diversity in temperate streams was found when compared to tropic streams. This has been attributed to new habitats being opened by disturbance. These openings would give competitively inferior species refuge in a community, thus increasing overall community diversity. This disturbance acts at an intermediate level, as intense disturbances completely destroy all members of the community.

The NSC highlights the basic difference between lake and river ecosystems, namely incorporating transport into nutrient cycling. Stream systems, from small headwaters to large rivers, import, produce, process and store organic matter (Cummins *et al.,* 1983). In rivers, nutrients are envisaged as being exported as they move downstream in a helical manner, alternating between organic and inorganic phases rather than remaining in a closed cycle as is the case in lakes. The degree of the cycle's displacement from its conventional closed form is largely determined by water flow, and in general, the greater the flow, the greater the distance between the "loops" of the spiral (Minshall *et al.*, 1983). Different factors control nutrient spiralling; physico-chemical processes, hydrological influences, uptake and assimilation by autotrophs and microbes, and the animal community (Allan, 1995).

1.3) THE CHEMICAL AND PHYSICAL CHARACTERISTICS OF RIVERS

Every chemical component and every physical attribute of a water sample contribute to the "water quality" of the sample (Dallas & Day, 1993). The chemical water quality variables which, potentially, influence river ecosystems and their biota are pH, salinity, conductivity, nutrients, dissolved oxygen and potentially harmful or toxic substances. The physical variables on the other hand, are turbidity, suspended material, flow and temperature. Hynes (1970), however, stated that many other factors influence rivers, such as the morphology of the channel system, the channel pattern, the stream bed material, vegetation and geological rock composition.

1.3.1) Chemical characteristics

All natural surface waters contain dissolved and particulate organic matter and the quantities are surprisingly high (Hynes, 1970). Rivers annually transport 3.9 billion tons of dissolved material to the oceans on a world-wide scale (Holeman, 1968).

There are three major mechanisms that control the world's surface water chemistry (Gibbs, 1970). The first is atmospheric precipitation. The chemical compositions of low-salinity waters are controlled by the quantity of dissolved salts provided by precipitation. These include the tropical rivers of Africa and South America, where the rainfall is very high and the supply of dissolved salts very low. The second mechanism controlling the world's water chemistry is rock dominance. The waters of rock dominant systems are, more or less, in partial equilibrium with the materials in their basins and the composition of ions is dependent on the relief, climate and rock composition of each basin. Evaporation-fractional crystallisation is the third mechanism controlling water chemistry. Rivers controlled by this mechanism are usually located in hot, arid regions. A number of these rivers display evolutionary paths, starting near Ca^{2+} or rock source waters with changes in composition toward Na^{+}-rich, high salinity waters as the rivers flow towards the ocean. These changes in composition are due to evaporation and to the precipitation of $CaCO_3$.

Gibbs (1970) also showed that when all the surface waters of the world are plotted on one graph (in terms of the ratios of Na^{+} to Ca^{2+} and Cl^{-} to CO_3^{2-}, in relation to total dissolved salts), a boomerang shaped envelope of data is produced. This is, if all three mechanisms are of more or less equal importance and the contribution of ancient salts is not taken into account. From left to right, along the lower arm, with a shift of input from rock dominance to a dominance of precipitation. From left to right along the upper arm, Na^{+} and Cl^{-} increase (high salinity) with a input shift from rock dominance to evaporation precipitation dominance. The vertical axis also reflects a gradient from high precipitation and runoff at the base to arid regions at the top (Allan, 1995). Most of the world's rivers are closer to the middle than the ends of this diagram, are low in $Na^{+}/(Na^{+} + Ca^{2+})$, and are dominated by Ca^{2+} and HCO_3^{-} from carbonate dissolution (Allan, 1995). They therefore, show the complicated relationships between pH, CO_2, H_2CO_3, H^{+}, CO_3^{2-}, HCO_3^{-}, Ca^{2+} and Mg^{2+} (Hynes, 1970).

Natural water components can be divided into five classes, namely dissolved inorganic ions and compounds, particulate inorganic compounds, dissolved organic compounds, particulate organic materials and dissolved gases (Golterman, 1975a). The dissolved inorganic constituents may, for convenience, be divided into major constituents, minor constituents, trace elements and gases (Table 1.1) (Hynes, 1970; Golterman, 1975a; Horne & Goldman 1995; Allan, 1995).

Table 1.1: The dissolved inorganic constituents in natural waters (Golterman, 1975a).

Major	Minor	Trace	Gases
Ca^{2+} HCO_3^-	N (as NO_3^-, NO_2^- or NH_4^+)	**Fe^{2+}, Cu^{2+},	O_2
Mg^{2+} SO_4^{2-}	Si (as SiO_2 or $HSiO_3^-$)	Co^{2+} B^{3+},	N_2
Na^+ Cl^-	P (as $H_2PO_4^-$, HPO_4^{2-} or PO_4^{3-})	Mn^{2+}, Mo^{2+}	CO_2
*(NH_4^+) (F^-)		Zn^{2+}, Al^{3+}	
**(Fe^{2+})		etc.	

* The constituents listed in brackets are sometimes regarded as major nutrients.

** Fe^{2+} can be regarded as major and minor nutrient as different algae require different Fe^{2+} concentrations for optimum growth.

The effects of Na^+, K^+ and Cl^- ions are largely ionic and contribute to the conductance of fresh water. These major ions are present, in greater or lesser quantities, in virtually every natural water type (Dallas & Day, 1993). Ca^{2+} is the most abundant cation, in natural freshwaters, and the concentrations thereof depend on the geological formations in the catchment areas (Gibbs, 1970; Dörgeloh *et al;* 1993; Allan, 1995). Ca^{2+} originates, almost entirely, from the weathering of sedimentary carbonate rocks, although pollution and atmospheric inputs constitute small sources (Allan, 1995).

The concentration of Na^+ is lower than both the concentrations of Ca^{2+} and Mg^{2+} (Golterman, 1975a; Allan, 1995; Dörgeloh *et al.*, 1993) while K^+ is a common constituent of many minerals and is always present in considerable quantities (Hynes, 1970). Mg^{2+} originates from the weathering of rocks, particularly Mg-silicate minerals and dolomite sodium which are generally found in association with chloride, while K^+ originates from the weathering of silicate materials, particularly potassium feldspar and mica (Allan, 1995). Other inorganic ions or compounds which are present usually occur at concentration orders of magnitude less than those of the major ions. Important minor nutrients include Mn^{2+}, Co^{2+}, Mo^{2+}, Cu^{2+}, and Zn^{2+} (Horne & Goldman, 1995).

Nitrogen (as NO_3-N) moves readily through most soils and ends up in aquatic ecosystems, while phosphorus, which occurs both as simple ionic orthophosphate (PO_4-P) and as bound phosphate, in soluble and particulate form, is less mobile (Hynes, 1970; Horne & Goldman, 1995). The concentration and rate of supply of nitrate is closely connected to land-use practices in the surrounding watershed. Nitrate also originates from protein decomposition, ammonia and agricultural practices (Hewitt, 1991). On the other hand, the natural occurrence of phosphorus in soil is known to be quite low and remains fairly stable because the majority occurs as insoluble inorganic phosphates (Hughes & Van Ginkel, 1994).

Phosphate does originate from human urine, detergents (Hewitt, 1991) and resuspension of sediments together with the mixing of pore waters with the watercolumn (Hynes, 1970; Hernández-Ayón *et al.*, 1993). The minerals which bind to phosphorus and form insoluble bonds, are Ca^{2+}, Fe^{2+} and Mg^{2+} (Hughes & Van Ginkel, 1994).

Phosphorus is also more readily washed out of sandy, acid or waterlogged soils than it is from clay soils (Walling, 1980); and the quantity of phosphorus in drainage water is controlled as much by the nature of the soil than by the quantity of phosphorus applied to the surface. Only a small quantity of soluble soil phosphate, however, is leached out by rain, and an even smaller quantity of the fixed phosphate in the soil is eroded away (Hughes & Van Ginkel, 1994).

Silicon is an important nutrient and the solute budget of silicon is of interest in the study of chemical weathering processes (Edwards, 1974). Most algae and animals only have a minor need for silica, but in the diatoms, silica forms the rigid algal cell wall or frustule which may account for half the cell's dry weight (Horne & Goldman, 1995). The form of silicon, when used as a structural component, is hydrated to form amorphous silica ($SiO_2.nH_2O$). This rigid material is highly perforated and surrounded on both sides by a thin cell membrane. Reactive silica is probably the only form available for diatom growth.

CO_2, O_2 and N_2 occur, in significant quantities, as dissolved gases in river water (Hynes, 1970; Golterman, 1975a; Allan, 1995; Horne & Goldman, 1995). Biologically, N_2 gas is the least important of the three. Only some cyanobacteria have the ability to fixate atmospheric nitrogen. Both CO_2 and O_2 occur in the atmosphere and dissolve in water, depending on the partial pressure and temperature. Photosynthesis and respiration are two important biological processes that alter the concentration of oxygen and carbon dioxide (Hynes, 1970; Golterman, 1975a). However, the relative proportions of CO_2, HCO_3^- and CO_3^{2-} are pH dependent (Wetzel, 1983). At a pH of below 4.5 only CO_2 and H_2CO_3 are present, and almost no HCO_3^- or CO_3^{2-} can be traced. At higher pH values dissociation of carbonic acid occurs, HCO_3^- and CO_3^{2-} are present and CO_2 and H_2CO_3 are no longer detectable. At intermediate pH values, HCO_3^- predominates and above a pH of 8.3 HCO_3^- declines. These dissociation dynamics are influenced by both temperature and ionic concentrations. In freshwater, this CO_2-HCO_3^--CO_3^{2-} equilibrium serves as the major buffering mechanism, preventing extreme pH values.

On a world-wide scale, natural processes dominate the chemistry of natural waters. However, at present the greatest cause of altered water chemistry, in natural water, is human originated pollution. Influent loading of phosphorus to aquatic systems, in particular, has increased markedly in recent times as a result of man's use of phosphorus for agricultural nutrients, industrial and detergent purposes and its release in domestic waste products (Wetzel, 1983).

1.3.2) Physical characteristics

The presence of suspended sediment or solids in river water are important physical characteristics. The annual sediment load of rivers, world-wide, is 20 billion tons (five times that of the total dissolved load) (Holeman, 1968). These sediment loads of rivers, and the turbidity of their waters, are dependent on complex interactions, between soil characteristics, quantity of precipitation, agricultural practices within the catchment and the flow rates of rivers draining the catchment (Ferrar, 1989).

Suspended sediments can have a direct effect on aquatic life through damage, to organisms and their habitat, and an indirect effect through their influence on turbidity and light penetration (Walling & Webb, 1992). For waters where transparency is governed by suspended inorganic material, turbidity has considerable value because light attenuation is primarily caused by scattering, and turbidity is a measure of this optical property (Walmsley & Bruwer, 1980). The main constituents of suspended sediments (mud) are silicates, carbonates (mostly $CaCO_3$) and organic matter (Golterman, 1975a). Phosphates and metals are the most important compounds adsorbed onto the sediment particles. The particle size of sediments affect their sorption properties. Phosphorus sorption increases as particle size decreases. The surface:volume ratio, acid-extractable Al^{2+}-content and organic matter contents of the sediments are all highly correlated with the P sorption index (Meyer, 1979). Thus, suspended sediments play an important role in the transport of nutrients and contaminants, in water-sediment interactions and as non-point source pollutants (Vaithiyanathan *et al.*, 1992).

Temperature affects other physical properties of rivers, such as dissolved oxygen and suspended solid concentration, and it also influences the chemical and biochemical reactions which take place. One effect of temperature on water is to alter its viscosity, and this causes silt to sink twice as fast at 23°C, as it does at 0°C. Thus, warmer water carries less silt than colder water does, and it also flows a little faster (Hynes, 1970). Large rivers and streams, at some considerable distance from their sources, are usually at more or less the mean air-temperature at the point of measurement. Shallow streams, on the other hand, are particularly subject to short-term temperature variations.

Water velocity and associated physical forces, however, collectively represent perhaps the most important environmental factors affecting organisms in running water. The speed of the current influences the size of particles of the substrate. Current, on the other hand, affects food sources *via* the delivery and removal of nutrients and food items (Allan, 1995). High flow conditions can also cause unfavourable conditions by disturbing the habitat structure of the riverine biota.

1.4) WATER DEMAND AND RESEARCH IN SOUTH AFRICA

Over time, as human activities increased, they have had an increasing impact on the quality of aquatic ecosystems. South Africa is no exception. The rapid growth of South Africa's population and accompanying rural, urban and industrial developments are placing ever increasing demands on water resources, in terms of quantity and quality (DWA, 1986; Wimberley & Coleman, 1993). For example, the total expectec ncrease in water demand for the Greater Bloemfontein (Bloemfontein, Heidedal and Mangaung) in the year 2020 will be 32, 399 x 10^6 l/day (Pretorius & Viljoen, 1997).

Limnological research changed emphasis in South Africa, in the 1980's, from reservoir to river ecosystem studies (Awachie, 1981). Since then, knowledge of South African rivers has been on the increase. To date, inland water research has been dominated by ecologists, most of them with zoological interests. A great deal of research has concentrated on the biota, particularly fish and invertebrates, but there has been little input from chemists, botanists, geomorphologists and hydrologists (Walmsley & Davies, 1991). However, the problem of quantifying the quantity of water required for environmental management demands a multi-disciplinary team approach. Such an approach requires the input of research specialists from numerous disciplines. Thus, a research effort that involves all the expertise necessary will have to be developed.

In the past, the Department of Water Affairs' (DWA's) research related to national water resource utilisation concentrated on economic and socio-economic management criteria, drought management, the deterioration in water quality, the estimation of future requirements and the availability of water from conventional sources (DWA, 1986). Other research concentrated on unconventional sources of potable water and the development of technology for their exploitation, the efficient utilisation of water, the development of water re-use technology, the development of technology to provide safe potable water from eutrophied systems, the influence of water resource development on the ecology and long-range weather forecasting (DWA, 1986).

However, amongst other changes, the Department of Water Affairs and Forestry (DWAF) has adopted a new approach to water pollution control (Van der Merwe & Grobler, 1990). In the past, the DWAF controlled water pollution from point sources by requiring that effluents meet uniform, general and special standards which were set at technologically and economically feasible levels. However, despite these efforts to control pollution, the quality of South Africa's water resources continues to deteriorate. In order to counter this, and to meet the challenges of the future, the following approaches were incorporated into the water quality management: uniform effluent standard, receiving water quality objectives and pollution prevention.

Management and research requirements also underwent some changes when the minister of Water Affairs and Forestry, Prof. Kader Asmal, announced in May 1994 that the Water Law should be subjected to a thorough review. As a result of changing demands the DWAF has shifted its emphasis from resource development to resource management. This shift in emphasis was also accompanied by a greater awareness of water quality and how it should be correctly managed.

Depending on the uses of water, its quality has to comply with standards so that it is not detrimental to human health. For example, recreation is an important use for many of South Africa's water bodies. Recreational management of lakes and dams requires a knowledge of the user's perceptions and behaviour in response to a variety of water quality conditions (Quick & Johanssen, 1992).

Some important points considering the ecological aspects of water ecosystems were made in the reviewed Water Law (National Water Act, No. 36 of 1998):

* Land use and human activities influence and impact upon the hydrological cycle and need to be co-operatively managed.

* Human use of water resources should not individually or cumulatively compromise the long-term sustainability of aquatic ecosystems.

* Aquatic ecosystems may be sustained at different levels of ecological health, depending on human decisions, in able to achieve a balance between development and ecosystem health.

* The ecological reserve in respect of international rivers should include sufficient water of sufficient quality for the full reach of the river.

* Water quality management should ensure that water, of acceptable usable quality, continues to be available to the users thereof and the relevant aquatic ecosystems.

The above-mentioned principles stress that it is important to take a holistic, or all-embracing, view of water management (integrated environmental management), in which a comprehensive spectrum of demands is recognised and evaluated to assess their priority. Integrated environmental management (IEM) is designed to ensure that the environmental consequences of development proposals are understood and adequately considered in the planning process (DWAF & WRC, 1995). This implies that the planning for development should be i) transparent, ii) multi-disciplinary and iii) holistic. i) Transparent: The planning must be open to public participation, so that parties concerned can voice comments, suggestions and problems. ii) Multi-disciplinary: Environmental interactions are very complex and can not be investigate in a single discipline. The investigations must include all factors that could possibly be influenced by the planning process. iii) Holistic: The issues of environmental and resource protection are so broad and interrelated that a holistic view is necessary to recognise all the effects of planned actions and to balance the benefits and costs. Thus, IEM should direct the planning of proposals, and not being considered once the proposal has been planned.

Thus, IEM must address all of the elements of the physical catchment, including impacts on the receiving water bodies and their users (Pegram *et al.*, 1997). This has led to a precautionary approach to water quality management, beginning with options to prevent and minimise pollution (including pollution from stormwater run-off), followed by receiving water quality objectives and holding back remediation of water bodies as a last resort.

Apart from IEM, water quality requirements given for a proposed effluent, must ensure that the water source influenced remain fit for its intended use (DWAF & WRC, 1995). These requirements must be met all times and strict control is desirable. The standard of quality can be defined by identifying constituents of concern that are present in the effluent.

1.5) BOTSHABELO: A SOURCE OF POLLUTION?

Botshabelo (11 00 ha), a low socio-economic urban development, is situated in the catchment of the Modder River, about 60 km east from Bloemfontein and was founded in the 1970's. Urbanisation has resulted in an increasing population of Botshabelo from about 35 000 to a total population of 243 855 in 1995 (Pretorius & Viljoen, 1997). The water demand for Botshabelo is expected to be 8 701 520 m^3/annum for the year 2000, 10 154 504 m^3/annum for the year 2002 and 13 453 159 m^3/annum for the year 2020 (Pretorius & Viljoen ,1997). This was calculated by assuming an average water usage of 600 l/day for informal households.

Sanitation consists mainly of pit latrines for the more than 35 000 stands and of a bucket and collection system. Only about 5% of the residents have water borne sewage. The wastewater treatment facility at Botshabelo releases about 5 megalitre effluent per day into the Klein Modder River. This is a source of minerals, bacteria and nutrients to the Modder River system. Pollution can also originate from sources other than sewage effluents. Wimberley & Coleman (1993) concluded that the large pollution load originating from Alexandra (a low socio-economic settlement in the Gauteng province) could be attributed to the following: the greater quantity of pollutants on the soil surface, the greater percentage of surface runoff, inadequate street cleaning and refuse removal, overflow, disposal of sewage from portable toilets and backyard mechanical operations.

One of the observations that were made during a study on Botshabelo (Grobler *et al.*, 1987), was that the non-point P-load is mainly derived from wash-off of surface phosphorus storage during storm events. As this coincides with relatively high volumes of runoff, which tend not to be retained in downstream impoundments, their impact is relatively minor when compared to the continuous point-source loads from the wastewater treatment facility. Comparing the input loads of Botshabelo with those of Mdantsane (an informal settlement near East London), Hughes & Van Ginkel (1994) found that the higher levels of pollutant input onto the eroded areas of Botshabelo, combined with the higher erosion rate from such areas, with sparse vegetation, contributed to relatively higher mean monthly P-loads from Botshabelo. This higher input was also affected by greater seasonality in the run-off and consequent wash-out in Botshabelo.

Grobler and Toerien (1986) predicted the impact of Botshabelo's sewage effluent on the Modder River system by means of a simulation model. They concluded that the discharge of sewage effluent into the Klein Modder River could result in undesirable eutrophic conditions, in terms of phosphorus, in Mockes Dam and the Mazelspoort Barrage. According to them, problems could arise before 1990, even if a 1000 µg/l P standard on effluent water was enforced. However, if the standard is complied with, conditions of severe nuisance should only arise when the effluent volume increased. The 1000 µg/l P is in accordance with the Special Standard for Phosphate in terms of the Water Act, provided by the DWAF (Taylor, 1984). This standard was implemented to reduce the phosphorus load from wastewater treatment facilities by 80 to 90 %, which in turn would lead to a considerable reduction in eutrophication. To comply with this 1000 µg/l standard required for 95 % of the time for the Botshabelo wastewater treatment facility, an average concentration of 400 µg/l P in the effluent would be required (Grobler & Toerien, 1986). These restrictions could be difficult to enforce with the population increasing in Botshabelo as well as the fact that there are only minimal sanitary services available.

1.6) OBJECTIVES AND MOTIVATION FOR THIS STUDY

Assessment of the ecological state of a river is not complete without an evaluation of the environmental factors which influence the aquatic ecosystem. These include biotic interactions, chemical variables, flow regime and habitat structure (Uys *et al.*, 1996). Additional environmental factors affecting rivers are hydrology, water quality, habitat availability and geomorphology.

Although the Modder River is an important source of potable water for Bloemfontein, it is not considered as one of the major rivers in South Africa. Because of this, limited limnological information is available on the Modder River. Grobbelaar (1985, 1989) reported on primary production as well as Jagals & Grabow (1996) on the effect of pollution on human health. Grobler & Toerien (1986) reported on some of the chemical characteristics of the river. The only other available data, are from the national physical and chemical database from the Department of Water Affairs and Forestry.

Other South African rivers, such as the Orange and Vaal Rivers, have more limnological information, accumulated through years of research as well as historical databases. The Department of Water Affairs and Forestry has established monitoring sites in South Africa's rivers (including the Modder River) in order to acquire information necessary for the formulation of a Water Quality Index (WQI). These sites were also recently recommended for biomonitoring sites, particularly where long-term physico-chemical data are available. This study will contribute to the long-term physico-chemical database of the Modder River, and eventually to the formulation of a WQI. Long-term studies, if provided with sound theoretical underpinning, will develop the understanding needed by decision takers in the pursuit of socio-economic goals (Huntley, 1987). The long-term investigation should not simply become an exercise in data collection, generally uninteresting and unproductive. Data should be regularly scrutinised and analysed, preferably for the development of mathematical models having realistic parameters directly related to the ecology of the subject being studied (Elliot, 1990). With this study we want to contribute to the limnological database of the Modder River and towards the scientific knowledge and resource management of this river. Furthermore, hydrological and water quality indices would give an early warning of widespread, possibly long-term changes in river condition, especially due to changes in land-use and land management or development.

The phytoplankton was investigated to determine whether the algal communities react to environmental changes, including eutrophication. Since the presence of some algae as well as diversity of communities indicates the level of eutrophication (thus the water quality), algal identification can prove to be a useful tool in this regard.

Using information available from the literature, comparisons between different lotic ecosystems were made to place the parameters into perspective and to determine whether common characteristics can be found among the different systems. Because of the increasing population of Botshabelo, and the predictions of severe nuisance conditions, we considered an investigation into the status of the Modder River system of importance. Furthermore, integrated catchment management is not possible without a thorough assessment and, thus, any contribution, irrespective of scope, will be of value.

The water quality of the Modder River also constitutes an important factor influencing the water quality of the Caledon River in the future. As a result of the limited water sources in the immediate vicinity of Bloemfontein, the Caledon River will in the future be the water source for this area (DWA, 1986). A scheme transferring water from the

Caledon River into the Modder River with an off-channel storage dam on a tributary of the Caledon River is presently under construction. Thus, because the movement of water from one river to another also means the concomitant movement of a physico-chemical regime from one area to another (Meador, 1992), it is important that the water quality of the Modder River remains high.

The objectives of this study were to determine whether Botshabelo has a detrimental effect (eutrophication and/or salinisation) on the water quality of the Modder River and what the self-purification potential of the river is. The study also serves to elucidate seasonal and spatial changes in the physical, chemical and biological characteristics of the water. The possible presence of toxic compounds was also investigated. The final objective was to calibrate, verify and determine the validity of a river simulation model (PC-QUASAR)* on the Modder River by using data obtained during the study.

Since the Modder River is a very turbid system, the factor most likely to limit algal growth, is light availability. It will be valuable if a model developed for a clear-water river system (such as PC-QUASAR) can be applied to the turbid systems as well. Very few useful models or theories are available to the planner because ecology has largely been a descriptive science while too little attention has been given to predictability in the field level (Dillon & Rigler, 1974).

If we want to sustain the limited freshwater resources in South Africa, we have to protect our aquatic ecosystems wherever they are. It is important to remember that, although socio-political and economic issues are usually at the root of water quality problems, the physical characteristics and processes in the catchment determine the actual nature of the problems (Pegram *et al.*, 1997).

* Supplied by the Institute of Hydrology, Centre for Ecology and Hydrology, UK.

CHAPTER 2

PHYSICAL AND CHEMICAL CHARACTERISTICS OF THE MODDER RIVER

2.1) INTRODUCTION

The physical attributes and chemical constituents of natural fresh waters differ from continent to continent, and even from region to region, as the result of differences in climate, geomorphology, geology and soils, and aquatic and terrestrial biota (Dallas & Day, 1993; Walmsley & Davies, 1991). The full assessment of water quality, in a river, includes evaluation of the physical, chemical and biological characteristics of that system (Uys *et al.*, 1996). The biggest difference between the physico-chemical and biological characteristics is that the former provide a quick and often accurate assessment of water quality, while the latter integrates changes in the system over time, but may not reflect short-term events.

As discussed in Chapter 1, Gibbs (1970) found that a boomerang-shaped envelope of data is produced when plotting the world's freshwaters according to the ratios of Na^+ to Ca^{2+} and Cl^- to CO_3^{2-}, in relation to total dissolved salts. However, the envelope of data for African waters is shaped like an alchemist's retort rather than as a boomerang (Kilham, 1990). The waters in Africa do not plot in either arm of the boomerang. According to Kilham (1990), the reason for this is that the major mechanism controlling the evolution of African waters, during evaporative concentration, is the precipitation of $CaCO_3$. A substantial loss of carbon occurs during every evaporative concentration step and the precipitation of $CaCO_3$. The alkalinity of these waters is also affected by two additional processes: i.e. reverse weathering decreases alkalinity, while the loss of sulphur to either sediments or the atmosphere increases alkalinity. Thus, although atmospheric precipitation and rock dominance are potentially important mechanisms controlling world water chemistry, atmospheric precipitation plays a lesser role in Africa, while rock dominance is the major mechanism controlling the dilute African waters. It is possible that rock weathering masks the ionic composition of rain or that terrigenous dust and ash, from fires, often determine the composition of rain. African rainwater are also rarely dominated by NaCl or any particular salt, because much of the rain falling in the interior of the continent originates from inland waters (Kilham, 1990).

The mean composition of African river water as determined by Allan (1995) are given in Table 2.1.

Table 2.1: Mean composition of African river water (mg/l)(Allan, 1995)

HCO_3^-	SO_4^{2-}	Cl^-	Ca^{2+}	Mg^{2+}	Na^+	K^+	SiO_2-Si
26.9	4.2	4.1	5.7	2.2	4.4	1.4	12

Examining only the major ions (Na^+, K^+, Mg^{2+}, HCO_3^-, CO_3^{2-}, Cl^- and SO_4^{2-}) and using a single set of observations, Dallas & Day (1993) stated that South African rivers fall into four categories:

Category 1: dominated by Ca^{2+}, Mg^{2+} and HCO_3^-.

Category 2: Ca^{2+}, Mg^{2+} and Na^+ are co-dominant, the major anion being HCO_3^-.

Category 3: Cations are more or less co-dominant and so are anions HCO_3^- and Cl^-.

Category 4: Dominated by Na^+ and Cl^- ions.

The rivers in the Free State are classified as temporary hard carbonate waters with Ca^{2+}, Mg^{2+} and Na^+ being co-dominant cations and HCO_3^-/CO_3^{2-} the dominant anions (Dallas & Day, 1993) i.e. in Category 2.

On a continental basis sediment yield varies considerably (Beaumont, 1975). The measured and estimated sediment yield, as summarised by Holeman (1968), showed that the sediment yield in Africa, Europe and Australia appears to be very low, averaging 112, 144 and 184 tons/km^2 each year, respectively. In Southern Africa, many rivers carry high concentrations of suspended material due to soil erosion as a result of sparse vegetation, erratic rainfall and easily-weathered sedimentary rock (Palmer & O'Keeffe, 1990a, Walmsley & Bruwer, 1980).

Rooseboom (1978) has estimated that in certain areas of South Africa, particularly where shales and mudstones of the Beaufort and Molteno series of the Karoo system predominate, annual sediment production can be as high as 100 t/km^2. Also, most of the waters with Secchi disc transparency lower than 0.5 m are to be found in the Eastern Province, Free State and South-western Gauteng areas where annual sediment production is higher than 201 t/km^2 (Rooseboom, 1978). Therefore, light and not nutrients is considered to be the primary limiting factor for algal growth in many of South African aquatic ecosystems (Grobbelaar, 1985). Thus, when resuspension from sediments or loading from the catchment significantly increases inorganic (non-algal) turbidity and light availability decreases, high production potentials are not realised (Dokulil, 1994).

The chemical characteristics of the Klein Modder and Modder River can be influenced by the sewage and stormwater outflow from Botshabelo. Effluents, from domestic water treatment facilities, may have low dissolved oxygen levels and can contain suspended materials of organic origin. These could still contain concentrations of nitrate, ammonium, phosphate and chloride higher than normally found in river water. Once discharged into a river, flow dilution will occur but the increased concentrations of these factors can be detected and the biotic effects observed in the form of changes in community response (Hewitt, 1991).

Point source pollution (such as sewage effluent) as well as diffuse source pollution influence both the chemical and physical composition of river water. Both the chemical and physical characteristics of the Modder River system were, therefore, taken into account during this investigation.

2.2) MATERIAL AND METHODS

2.2.1) Study site

The Modder River is a relatively small river which drains an area of 7960 km^2, in the central region of the Free State Province, South Africa and has a mean annual run-off of 184 x 10^6 m^3. The catchment area of the Modder River lies between 24° 40' E and 27° 0' E and between 28° 30' S and 29° 25' S (Toerien *et al.*, 1983), and is located in a summer rainfall area, which receives between 600-700 mm per annum, half of which is through thunderstorms (Grobbelaar, 1992). The Modder River catchment lies within an geological area classified as the Molteno, Red Beds and Cave Sandstone Stages of the Stormberg Series and the Beaufort Series, both of the Karoo system (Grobler & Davies, 1981).

Water, from this river, is stored in the Rustfontein, Mockes, Mazelspoort and Krugersdrift Dams (Grobler & Toerien, 1986; Grobbelaar, 1992) (Table 2.2). In the past about 60% of the potable water supply, of the city of Bloemfontein, was provided by the Modder River and the remainder was pumped from the Caledon River which is about 150 km south-east of Bloemfontein (Grobbelaar, 1992). However, at the moment, only about 25% of the potable water supply is provided by the Modder River. The Modder River can be dry for long periods, particularly during the winter months and the impoundments are the only permanent sources of water (Grobbelaar, 1991).

Table 2.2: Morphological and hydrological data of impoundments in the upper Modder River (Grobler & Toerien, 1986)

Reservoir	Full supply level ($m^3 \times 10^6$)	Mean depth (m)	Catchment area (km^2)	Mean annual runoff ($m^3 \times 10^6$)	Mean water retention time (annum)
Rustfontein	76	6.5	950	35	2.2
Mockes	6	1.8	2960	106	0.06
Mazelspoort	0.8	2	3059	106	0.008

This study was conducted in the upper reach of the Modder River which included the region of the Modder River from Rustfontein Dam down to the Mazelspoort Barrage (Figure 2.1). It also included the Klein Modder River which drains from Botshabelo and flows into the Modder River just before Sannaspos. Water samples were taken, fcrtnightly, from February 1996 to December 1997 at ten sampling sites (Figure 2.1 and pp. 23-27), five in the Klein Modder (KM1 - KM5) and five in the Modder River (GM1 - GM5). KM1 was chosen as a reference point for the city's run-off before the effluent of the wastewater treatment facility (KM2) was added. KM3 thus is the collective reference point of the city's total output, including the discharge of surface water run-off. It is important to note that KM2 is not located in the Klein Modder River, but represents the outflow of the Botshabelo wastewater treatment facility. This site was also not taken in consideration when calculating the mean value for any variable for the Klein Modder

River. Both rivers never ran dry during the study period, although water levels were sometimes very low. Because GM1 is well above the populated area, it was used as a reference point (unpolluted water).

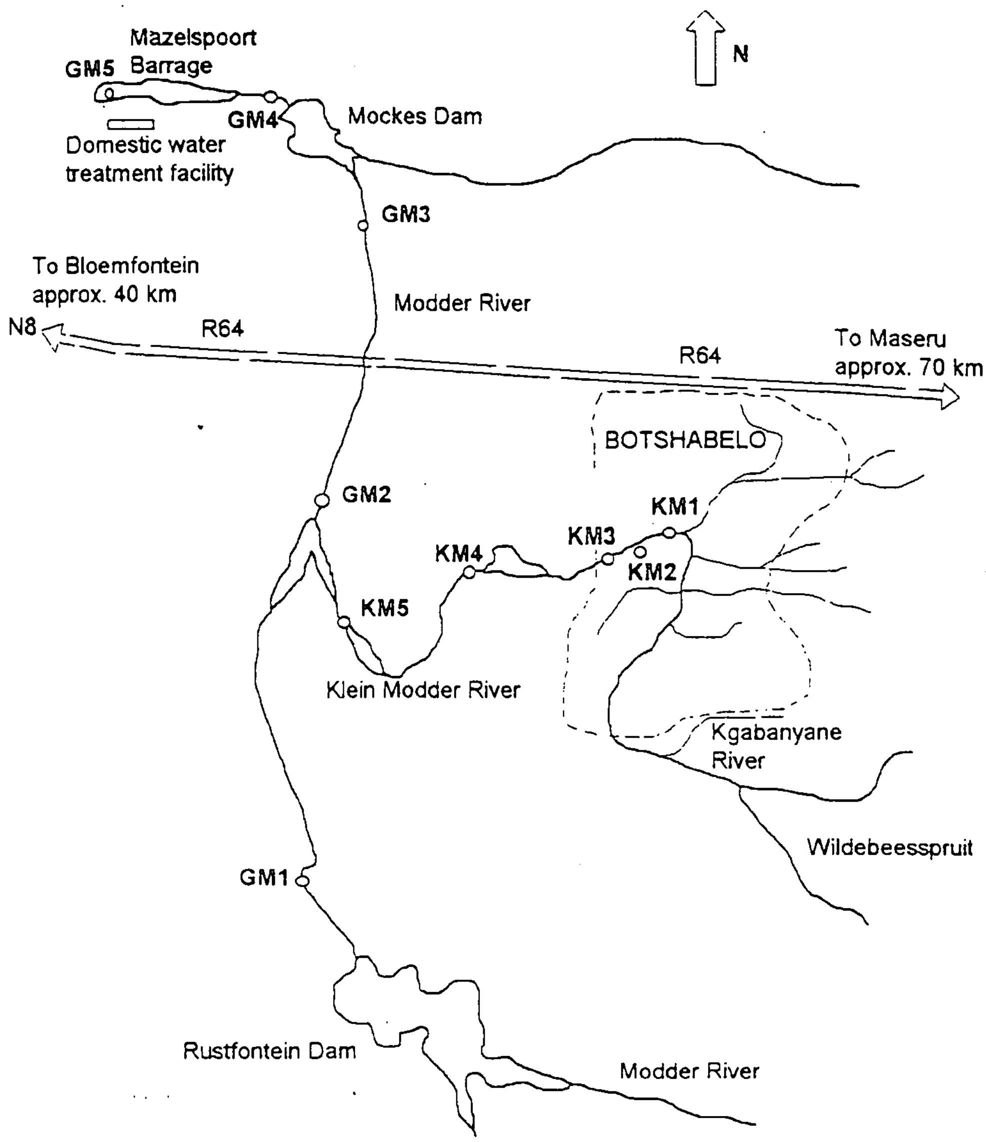


Figure 2.1: Sampling sites in the Klein Modder and Modder Rivers

Klein Modder River:

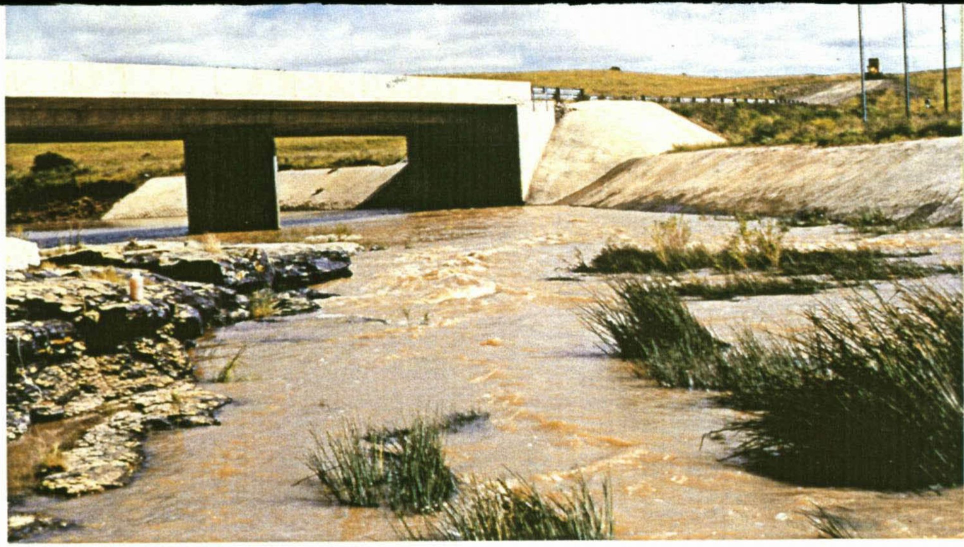
KM1: Reference point for the city's run-off before the effluent are added.
KM2: Outflow of the Botshabelo treatment facility.
KM3: Collective reference point (above Botshabelo Dam) for the city's total output/discharge of surface water run-off.
KM4: In the river beneath Botshabelo Dam.
KM5: On a farm (Vadersgift) above a damwall.

Modder River:

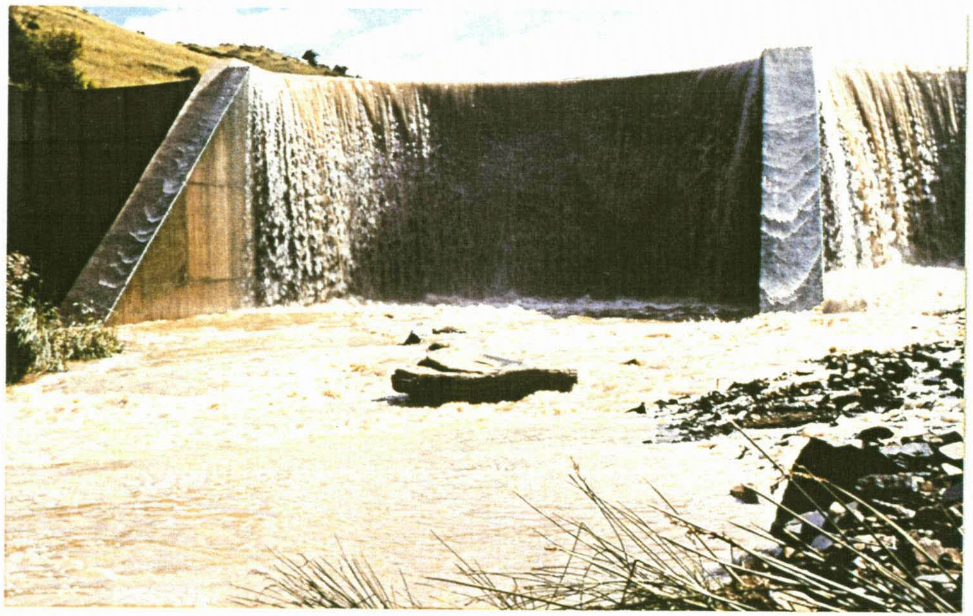
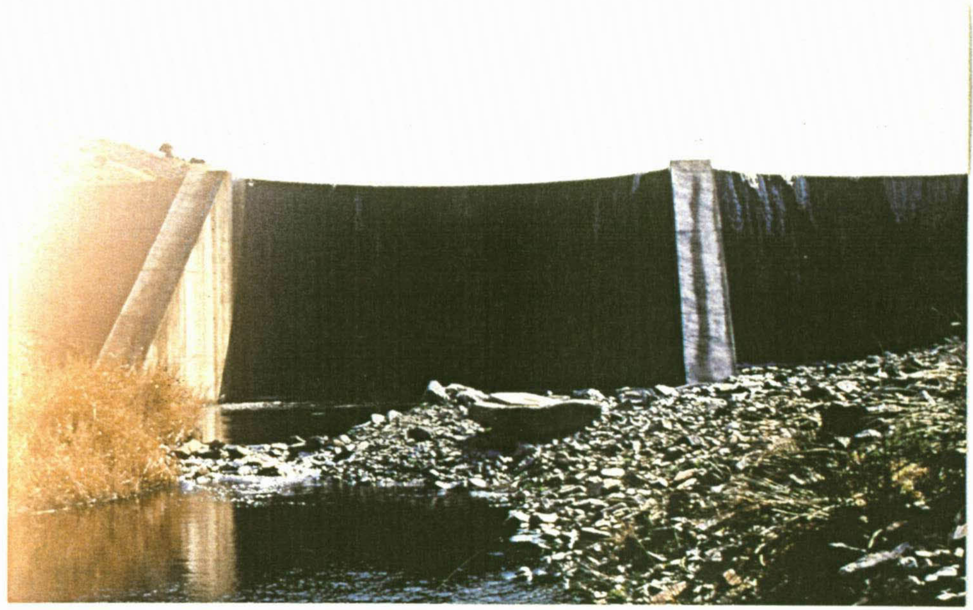
GM1: In the river near Palmietfontein Nature Reserve, below Rustfontein Dam. This site was used as an unpolluted reference point.
GM2: Sannaspos - just after the confluence of the Modder and Klein Modder Rivers.
GM3: A site about 12 kilometers downstream with rocky banks.
GM4: A site below Mockesdam.
GM5: In Mazelspoort Barrage.

KM1: Reference point for the city's run-off before effluent is added (summer and winter)

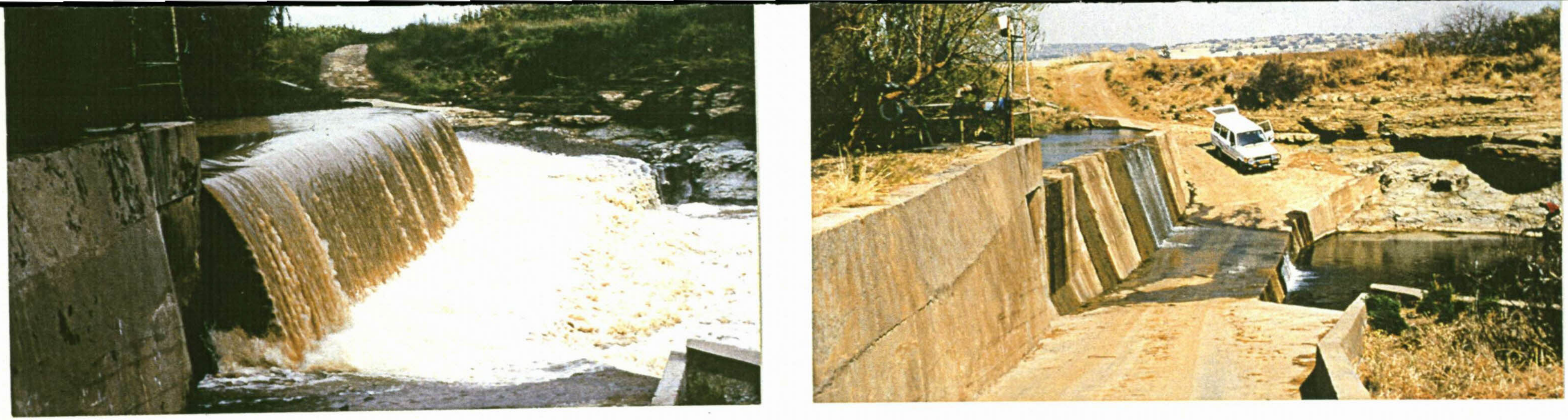
KM2: The treated effluent of the Botshabelo sewage works (all seasons)



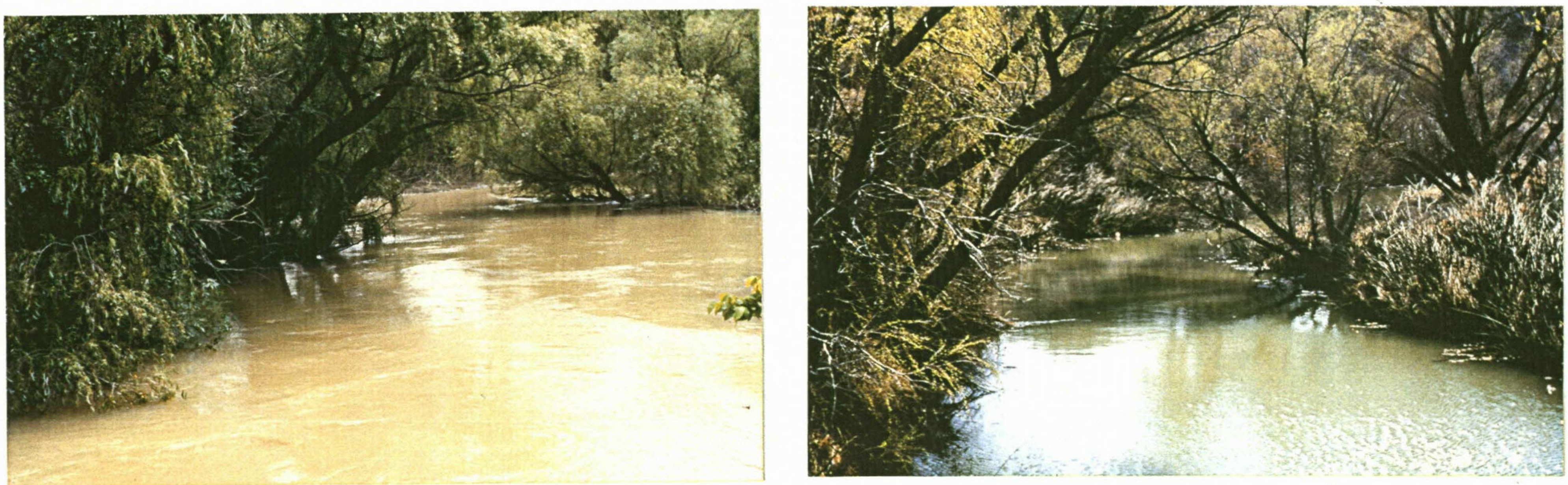

KM3: The sampling point above Botshabelo dam (summer and winter)

KM4: The sampling point just below Botshabelo Dam (summer and winter)

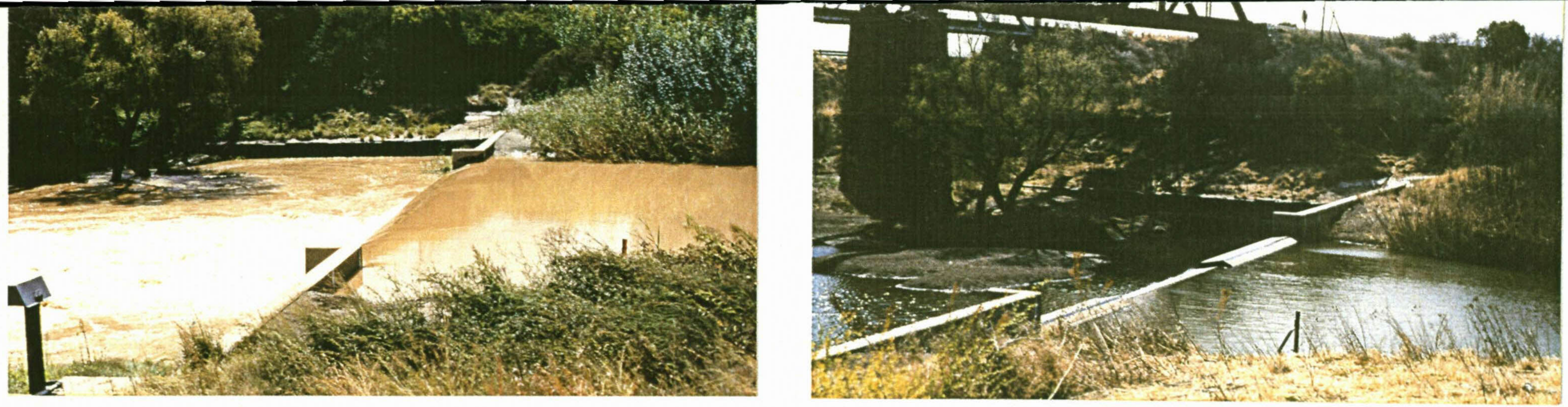


KM5: The sampling point at Vadersgift, just before the confluence with the Modder River (summer and winter)

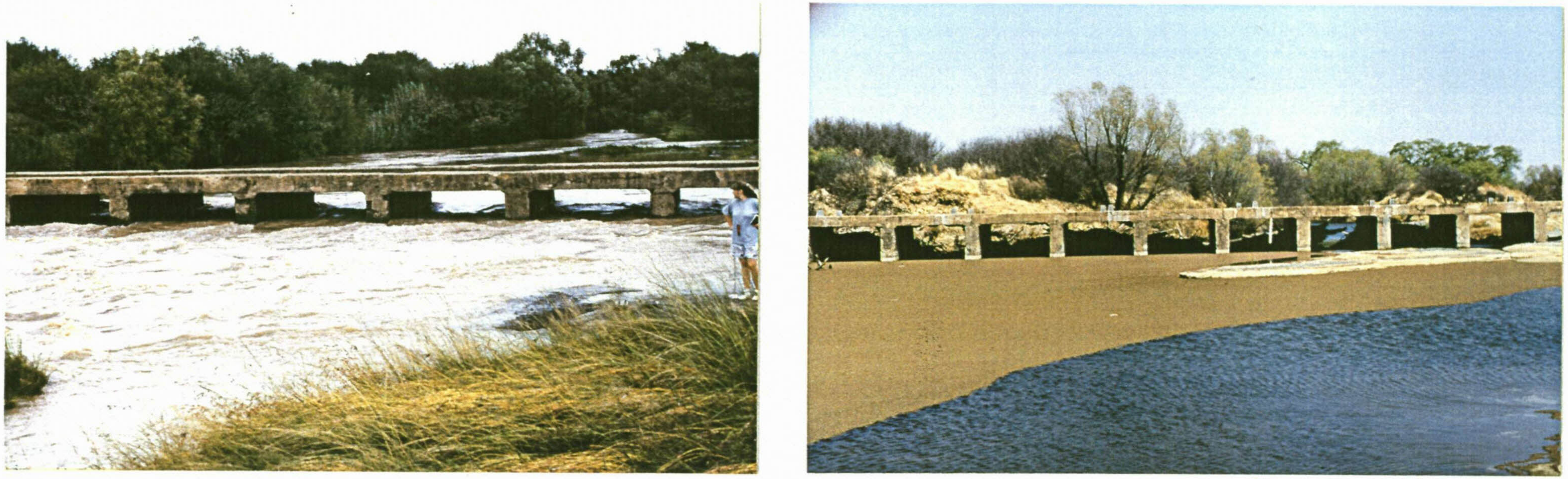


GM1: The sampling point just below Rustfontein Dam, at the Palmietfontein Nature Reserve (reference point)

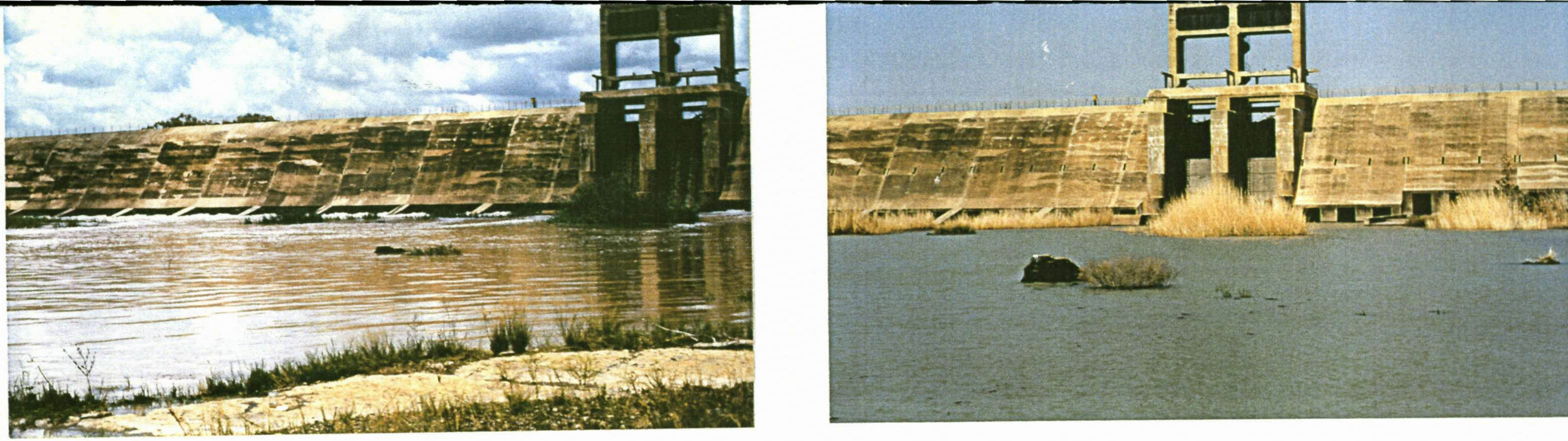
(summer and winter)



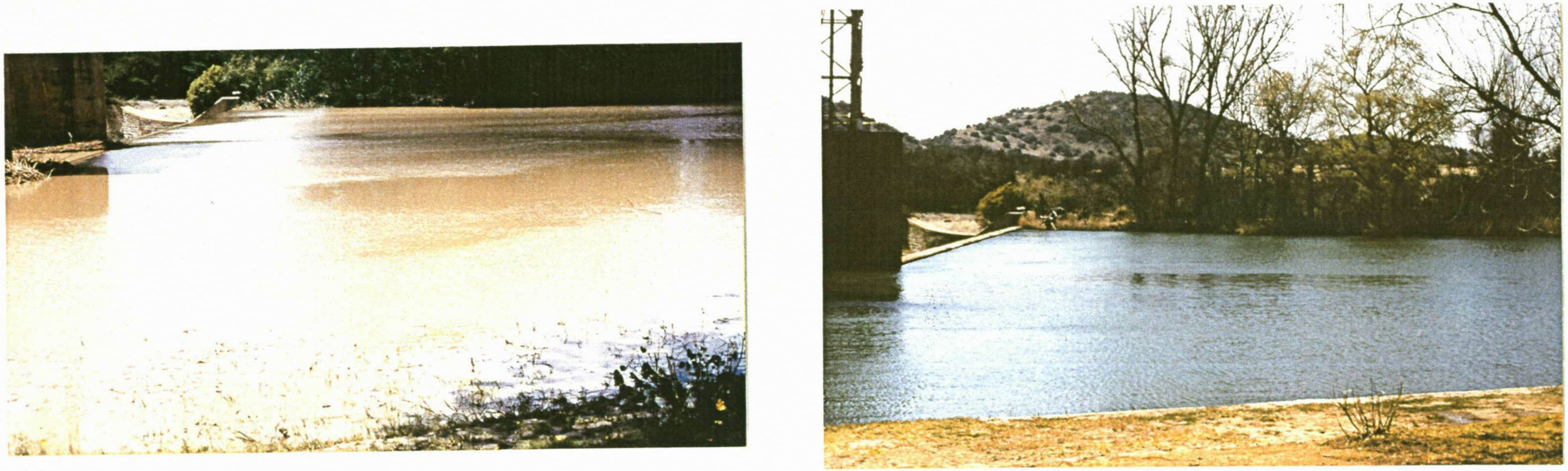
GM2: The sampling point at Sannaspos, just after the confluence with the Klein Modder River (summer and winter)



GM3: The sampling point 15 km downstream from Sannaspos (summer and winter)



GM4: The sampling point just below Mockes Dam, at Philip Sanders Holiday Resort (summer and winter)



GM5: The sampling point in Maselspoort Barrage (summer and winter)

2.2.2) Physical and chemical parameters

In situ measurements were made and subsurface samples (2 litres) were taken, kept on ice and brought to the laboratory for chemical analyses. The analyses were done within 48 hours. Prior to analyses the samples were stored, in the dark, at 4°C to limit metabolic alterations.

The water-temperature (°C) was measured with a YSI Model 50B dissolved oxygen meter (5739 probe) and were done *in situ*. The mean air temperatures for 1996-1997 were obtained from the Weather Bureau in Pretoria. Turbidity is a measurement of the concentration of suspended, organic and biological material in the water and was determined with an Aqualytic Turbidimeter AL1000, expressed as NTU. Flow data as well as the water level in the Modder River at Sannaspos were obtained from the Department of Water Affairs, Pretoria. Water level was used as indication of flow. Rainfall data for the Bloemfontein area from January 1996 until June 1998 were obtained from the Department of Meteorology, Faculty of Agriculture, University of the Free State.

The concentration of dissolved oxygen and percentage of saturation were measured using a YSI Model 50B dissolved oxygen meter (5739 probe). The pH of the water was determined with a HANNA HI 9073C MICROCOMPUTER pH meter. These measurements were done *in situ*.

Conductivity (which serves as an indication of the total dissolved salts in the water), was determined using a T & C Model 2001 conductivity meter, expressed as mS/m.

The total dissolved salt content (different ions) at Sannaspos in the Modder River (GM2) were obtained from the Department of Water Affairs and Forestry, Bloemfontein. The data given for the Orange River were also obtained from the historical database of the Department of Water Affairs, Pretoria.

Nitrate-nitrogen (NO_3-N) was determined as described in Bausch and Lomb (1974) using 10 ml Whatman GF\C filtered water, the addition of 2 ml NaCl and digestion with 10 ml sulphuric acid for 25 minutes at 95°C to give a yellow colour reaction with 0.5 ml brucine sulphate. The absorbance was measured using a Hitachi spectrophotometer at 410 nm.

Both dissolved reactive *orto*-phosphates (PO_4-P) (100 ml GF\C filtered water) and total phosphates were determined by using the methods described in *Standard methods for the examination of water and waste water* (1995) which involved a reaction of 4 ml ammoniummolybdate with 0.5 ml stannochloride to give a blue colour. For total phosphate determination, unfiltered water was pre-digested during 1996 with persulphate and H_2SO_4 in an autoclave at 121°C for 30 minutes, and during the last part of 1997 with an UV oxidation apparatus and then filtered. The absorbance was measured with a Hitachi spectrophotometer at 690 nm.

Silica-silicon (SiO_2-Si) (50 ml GF\C filtered water) was determined by using a modified method as described in *Standard methods for the examination of water and waste water* (1995) - the silica yellow method - which involved a reaction of 1 ml HCl and 2 ml ammoniummolybdate to give a yellow colour that was measured with a Hitachi spectrophotometer at 410 nm. The intensity of the yellow colour is proportionate to the concentration of "molybdate-reactive" silica.

2.3) RESULTS

2.3.1) Turbidity and flow

The turbidity in the Modder River ranged between 10 and 650 NTU's (mean = 86 NTU's). In the Klein Modder River the turbidity was between 1 and 900 NTU's (mean = 97 NTU's). It increased during the rainy season with increased flow (February 1996 and October/November 1996 to May 1997), but was low in both years (<20 NTU's), during the winter period (June to August) (Figure 2.2 & 2.3).

During the rainy season in 1996, the flow in the Modder River can increase up to 19 $m^3\ s^{-1}$ from < 1 m^3s^{-1} and the mean water level changes accordingly, and can increase to 0.55 m (Figure 2.4). Once during the winter (July - August 1996) there was an increase in turbidity when both snow and rainfall occurred (Figure 2.2 & 2.3).

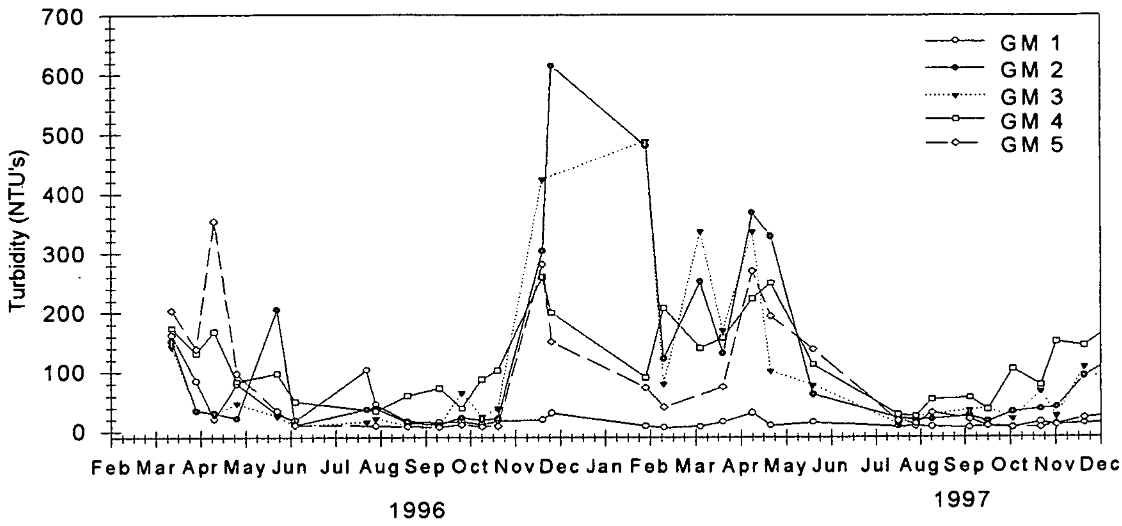


Figure 2.2: Variation in turbidity in the Modder River during the study period.

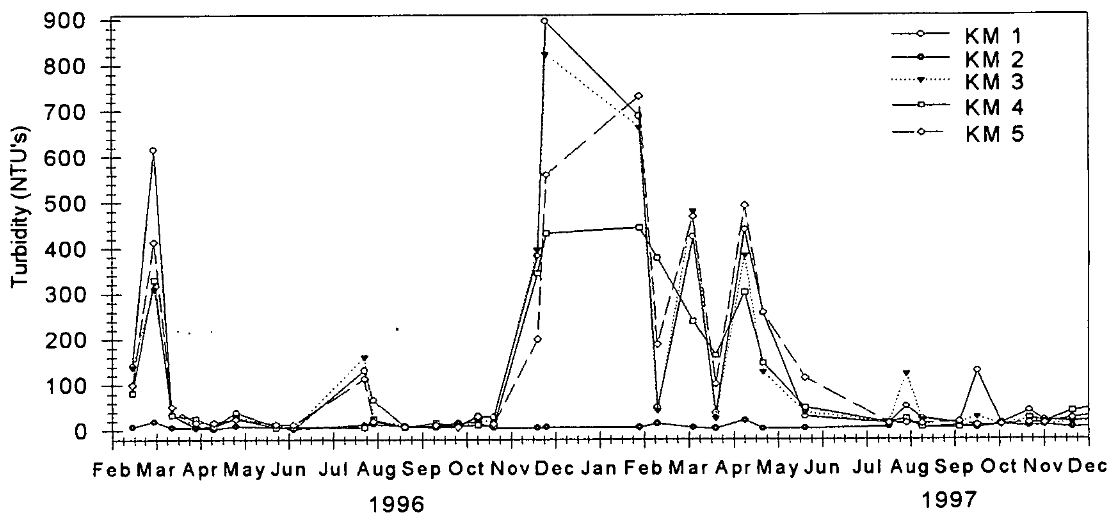


Figure 2.3: Variation in turbidity in the Klein Modder River during the study period.

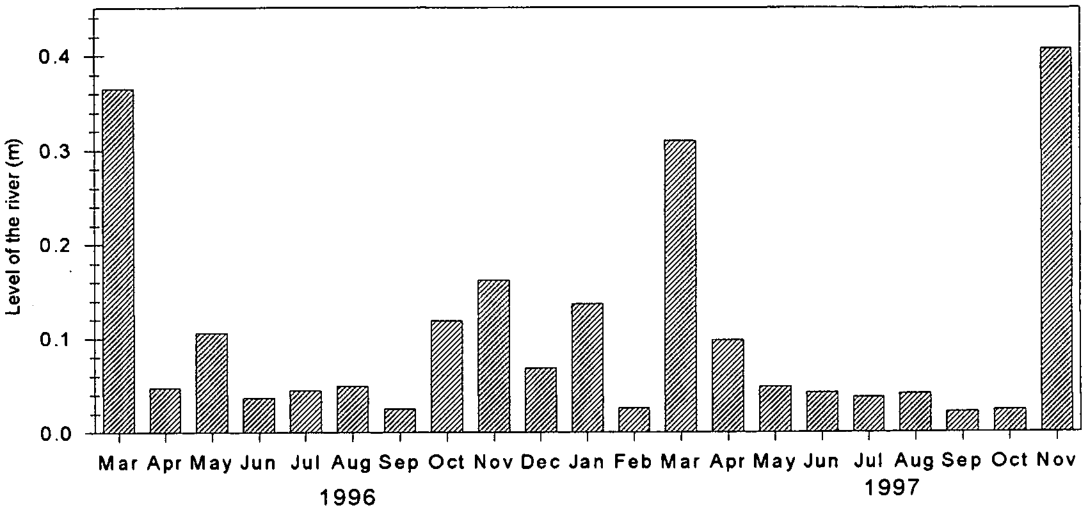


Figure 2.4: The change in water level in the Modder River for the different years of the study period.

In 1996 the total rainfall for Bloemfontein was 642.8 mm and in 1997 it was lower being only 370.2 mm. During 1996 the maximum rainfall occurred during February, March and December. During 1997, the maximum rainfall occurred during February, September, October and December.. Although the rainfall was lower in 1997, the mean turbidity in both the Modder and the Klein Modder River increased from 1996 to 1997 (Figure 2.5 & 2.6).

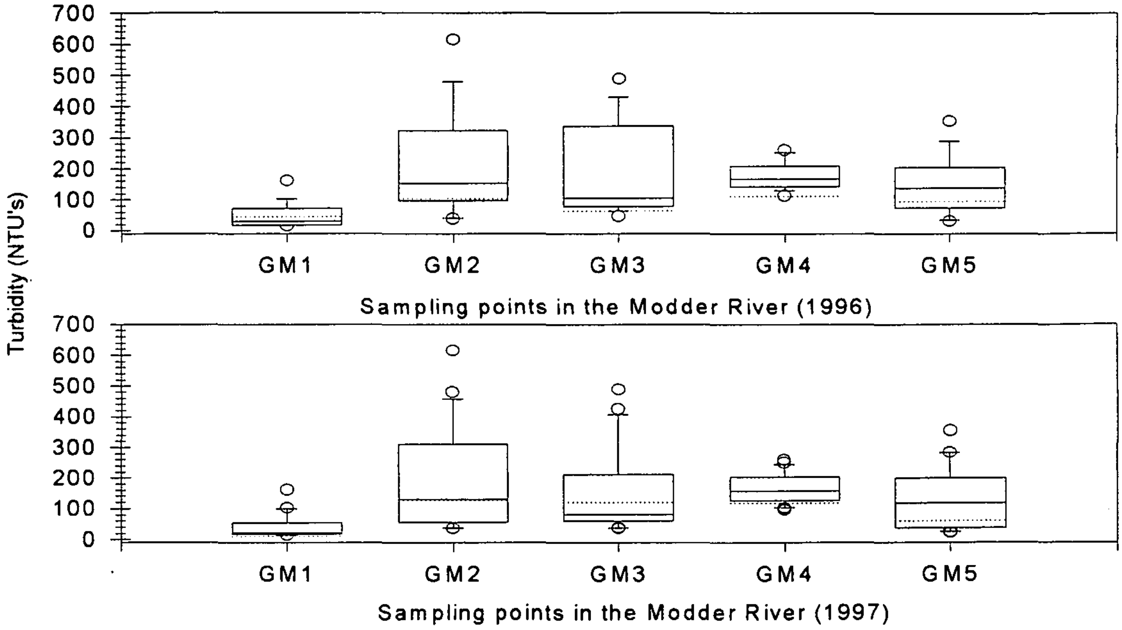


Figure 2.5: The turbidity in the Modder River for 1996 and 1997. (The horizontal lines of the box plots mark the median, 10^{th}, 25^{th}, 50^{th} and 95^{th} percentile points of the data. The box encompasses the 25^{th} through the 75^{th} percentiles. The 5^{th} and the 95^{th} percentiles are shown as symbols (o) below and above the 10 and 90% caps respectively. The dotted line represents the mean value).

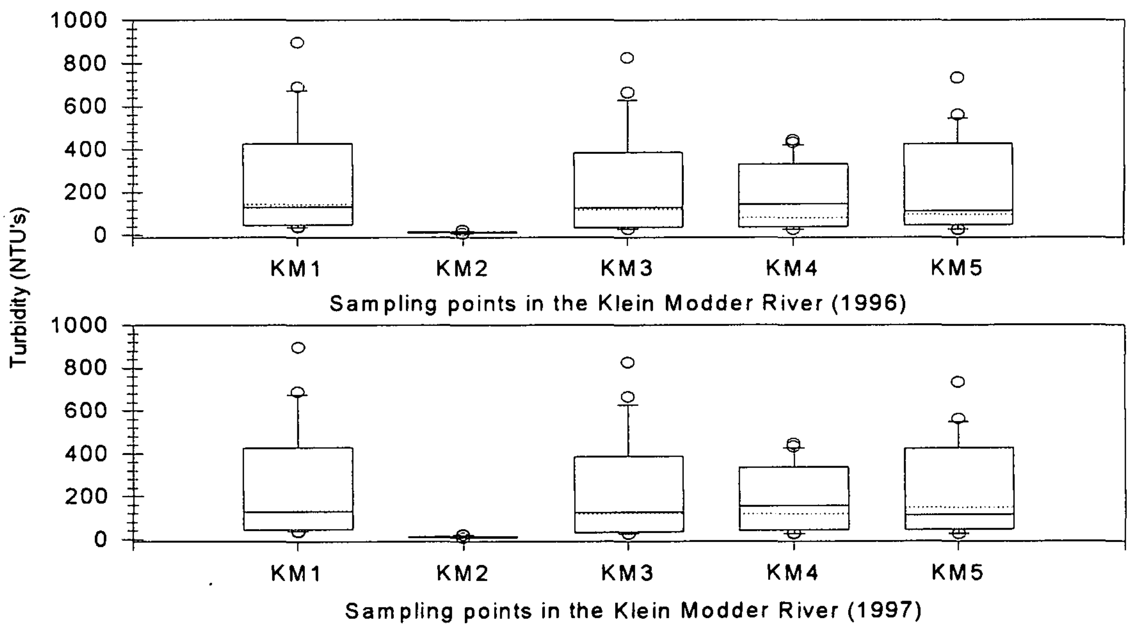


Figure 2.6: The turbidity in the Klein Modder River for 1996 and 1997. For explanation see Figure 2.5.

2.3.2) Conductivity

The minimum value of conductivity in the Modder River was 10 mS/m and the maximum value was 67 mS/m (mean = 36 mS/m) (Figure 2.7)

In the Klein Modder River was the minimum value was 12 mS/m and the maximum was 100 mS/m (mean = 52 mS/m) (Figure 2.8).

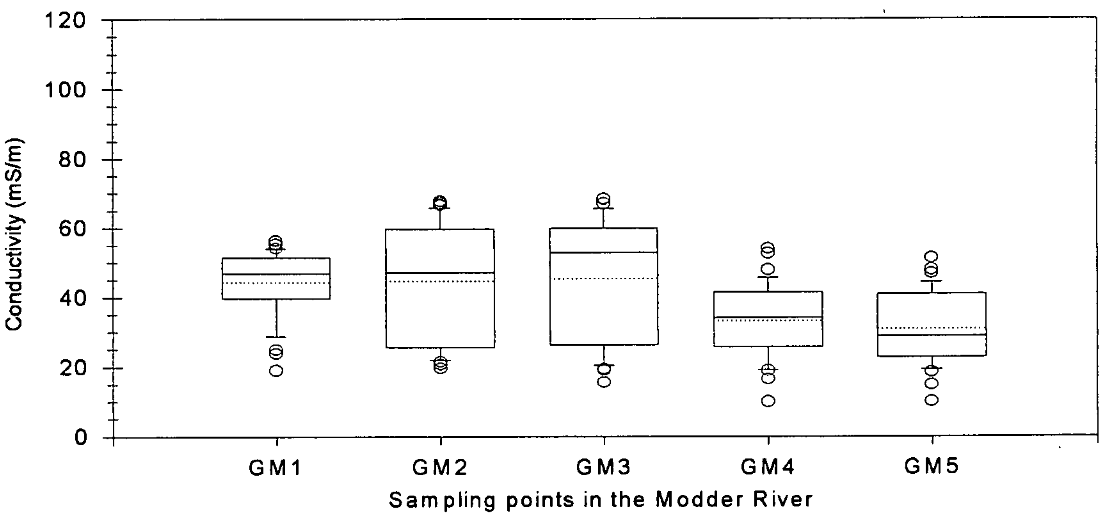


Figure 2.7: The downstream variation in conductivity in the Modder River. For explanation see Figure 2.5.

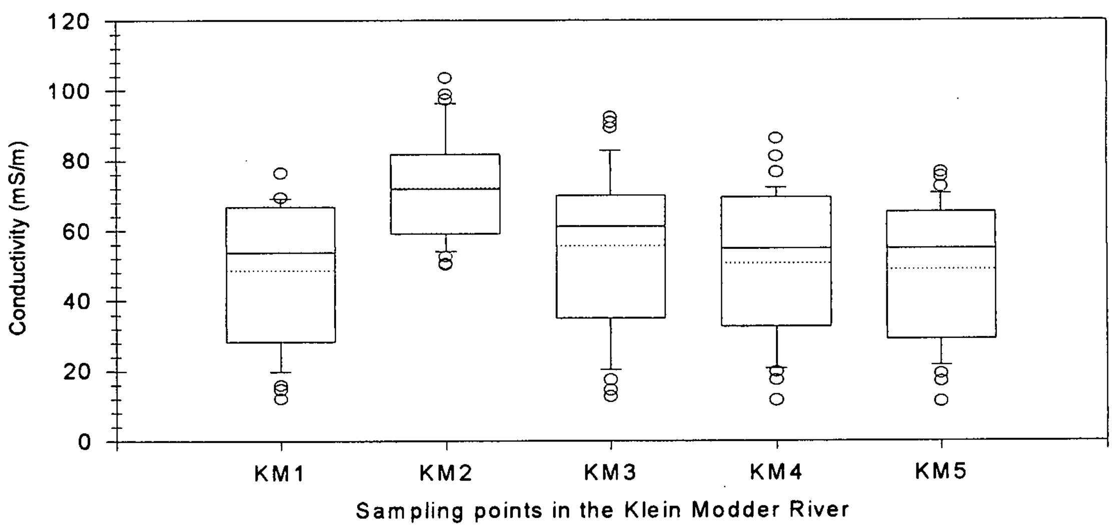


Figure 2.8: The downstream variation in conductivity in the Klein Modder River. For explanation see Figure 2.5.

The composition of total dissolved salts in the Modder River (given above as conductivity), are dominated by Ca^{2+} and Na^{+} as being the dominant cations and Cl^{-} and SO_4^{2-} as being the major anions. Ca^{2+} is the dominant ion, the order of dominance being: $Ca^{2+}>Na^{+}>Cl^{-}>SO_4^{2-}>Mg^{2+}>K^{+}$ (Figure 2.9).

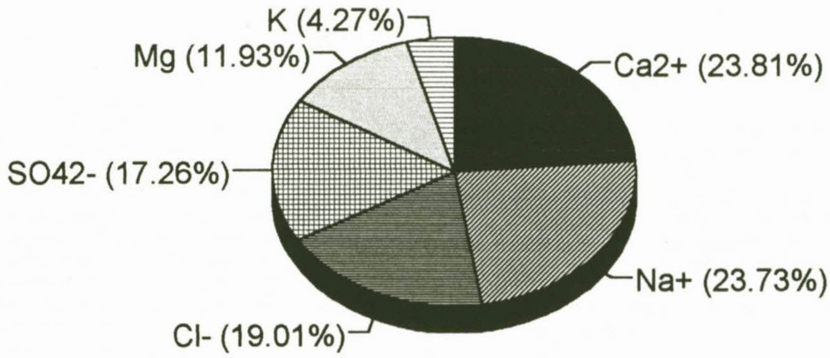


Figure 2.9: Mean ionic composition of the waters of the Modder River.

The mean conductivity at the sewage outflow, KM2 (72 mS/m), was higher than that at sampling points in the river. Conductivity decreased, significantly, from KM3 to KM5 (Figure 2.8). The average conductivity at GM1, GM2 and GM3 was almost the same, being about 42 mS/m. However, there was a definite decrease in conductivity at GM4 and GM5 when compared to GM3 (Figure 2.7).

The conductivity levels at all sampling sites in both the Modder and Klein Modder Rivers followed the same patterns (Figures 2.10 & 2.11). After rain, in February 1996 (129.1 mm) the conductivity decreased. From March to November 1996 conductivity at all the sampling sites increased. In December 1996 (with a rainfall of 128.4 mm), conductivity decreased again. In 1997 conductivity levels increased steadily, excepting for a decrease during March-April. During both years there was a salinity build-up during winter. This build-up occurred from July to September in 1996 and from June to September in 1997 (Figure 2.10 & 2.11).

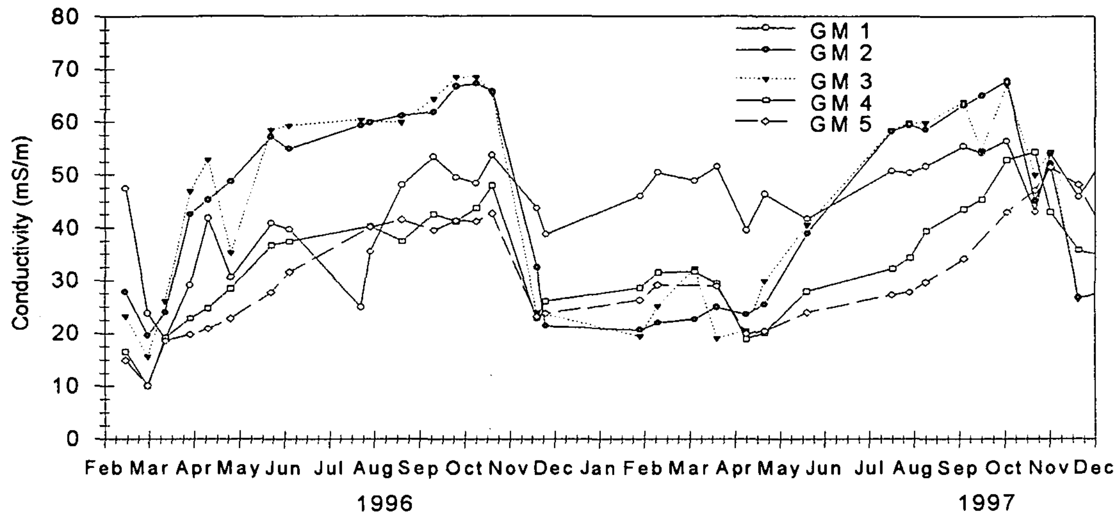


Figure 2.10: Variation in conductivity in the Modder River during the study period.

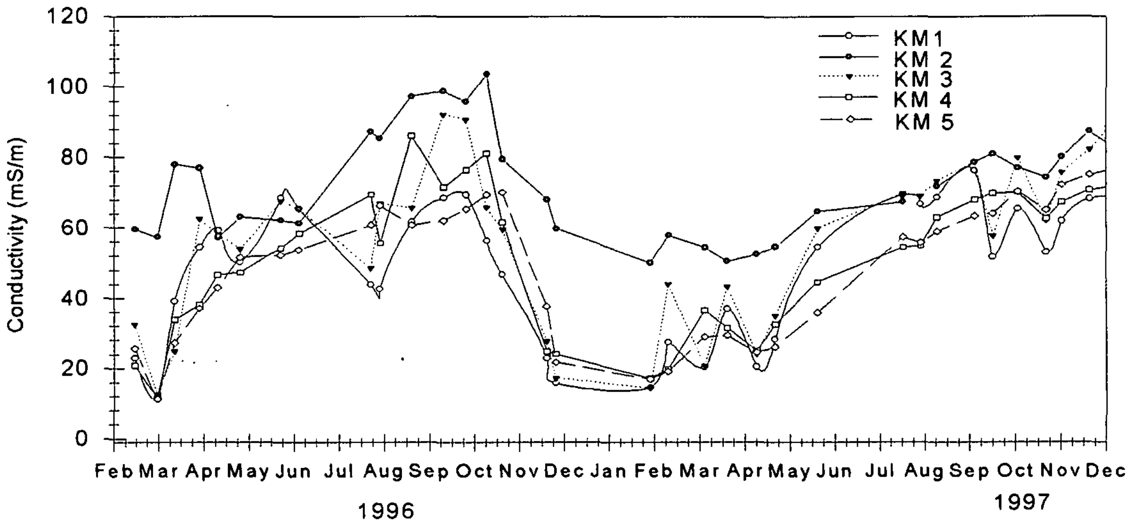


Figure 2.11: Variation in conductivity in the Klein Modder River during the study period.

2.3.3) Turbidity vs. conductivity vs. rainfall

A statistically significant inverse correlation was demonstrated between conductivity and turbidity in the Klein Modder and Modder Rivers. This relationship was determined by using individual data. Sixty-eight percent of the variation in conductivity was associated with the variation in turbidity ($r = 0.680$, $p < 0.001$) (Figure 2.12).

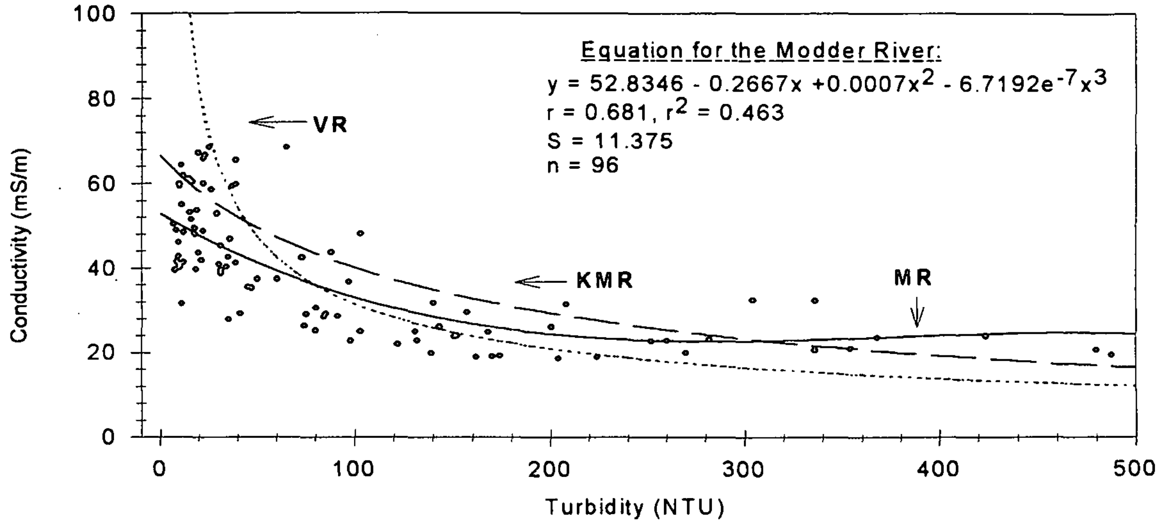


Figure 2.12: The inverse relationship between conductivity and turbidity in the Modder River (MR) (solid line) The broken line represents this relationship in the Klein Modder River (KMR) and the dotted line the relationship in the Vaal River (VR).

2.3.4) Nitrate-nitrogen (NO_3-N) and phosphate-phosphorus (PO_4-P)

During the study period the NO_3-N concentration in the Klein Modder River displayed large variations (from <20 µg/l to 4 571 µg/l) with a mean of 813.2 µg/l. A maximum of 4 571 µg/l was recorded at KM2 following storm events when the sewer system received large quantities of urban drainage. NO_3-N concentration in the Modder River was much lower, with an average of 191.2 µg/l and it also displayed large variations (from <20 µg/l to 816 µg/l). However, most of the increases in NO_3-N concentration, in both the Klein Modder and Modder Rivers, was associated with rainfall (Table 2.3 & 2.4).

Table 2.3: Variations in NO_3-N in the Klein Modder River associated with rainfall

Month	Monthly rainfall	Monthly mean NO_3-N concentration
April 1996	60.1 mm	1157.5 µg/l
August 1996	32.3 mm	1364 µg/l
March 1997	121.6 mm	823 µg/l
April 1997	30 mm	1719 µg/l

Table 2.4: Variations in NO_3-N in the Modder River associated with rainfall

Month	Monthly rainfall	Monthly mean NO_3-N concentration
March 1996	149.2 mm	495.2 µg/l
November 96	112.2 mm	444.25 µg/l
March 1997	121.6 mm	560.2 µg/l
April 1997	30 mm	64.6 µg/l

The PO_4-P concentrations in the Modder and Klein Modder Rivers showed no seasonal patterns and large variations occurred during the study period. The PO_4-P concentration in the Modder averaged 64.4 µg/l and in the Klein Modder River it was 197.4 µg/l. However, most of the higher PO_4-P concentrations, as in the case of NO_3-N, were associated with periods of rainfall. The total phosphorus (TP) concentration was very high in both the Modder and the Klein Modder Rivers, throughout the study period, with an average of 283.95 µg/l and 519.95 µg/l, respectively.

There was a definite increase in NO_3-N, PO_4-P and TP concentrations at GM2, compared with GM1 and were ascribed to inflow from the Klein Modder River (Figure 2.13, 2.14 & 2.15). Downstream in the Modder River all these concentrations decreased to same or almost the same levels than at GM1 (reference point). In the Klein Modder River, these concentrations also decreased downstream from KM3 to KM5, but was still higher than at GM1, thus causing increasing concentrations at GM2.

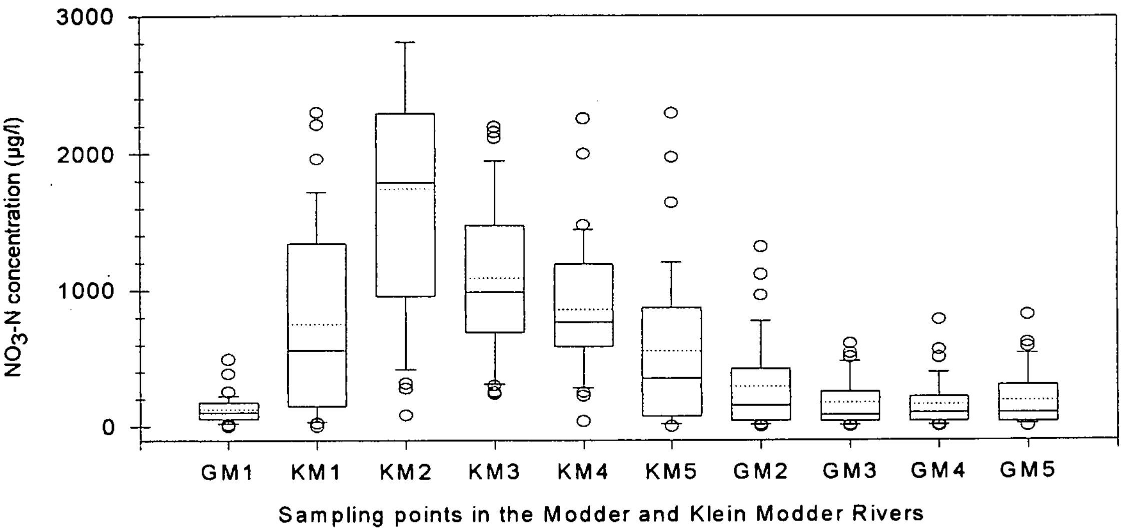


Figure 2.13: The downstream variation in NO_3-N in the Klein Modder and Modder Rivers. For explanation see Figure 2.5.

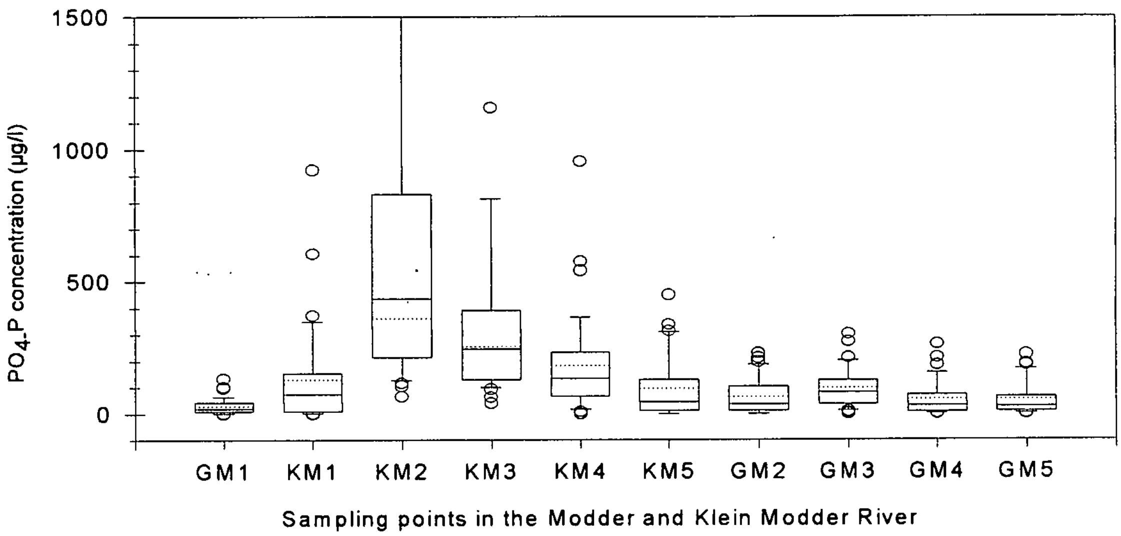


Figure 2.14: The downstream variation in PO_4-P concentration in the Modder and Klein Modder Rivers. For explanation see Figure 2.5.

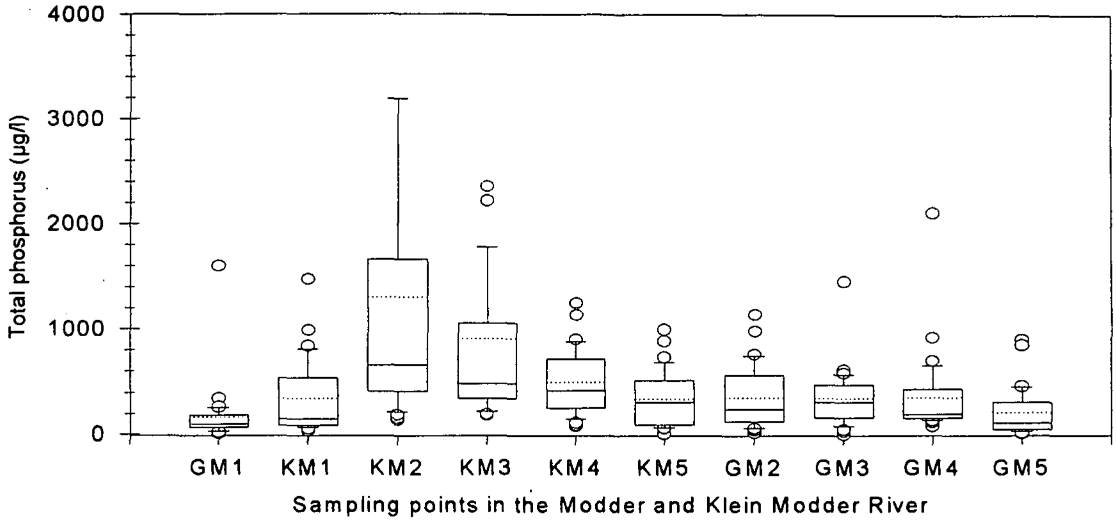


Figure 2.15 The downstream variation in TP in the Modder and Klein Modder Rivers. For explanation see Figure 2.5.

In the Modder River, the average NO_3-N and PO_4-P concentration were lower during 1997 than in 1996 (Figure 2.16 & 2.17). NO_3-N decreased from 250 µg/l to 132 µg/l and PO_4-P decreased from 78 µg/l to 51 µg/l.

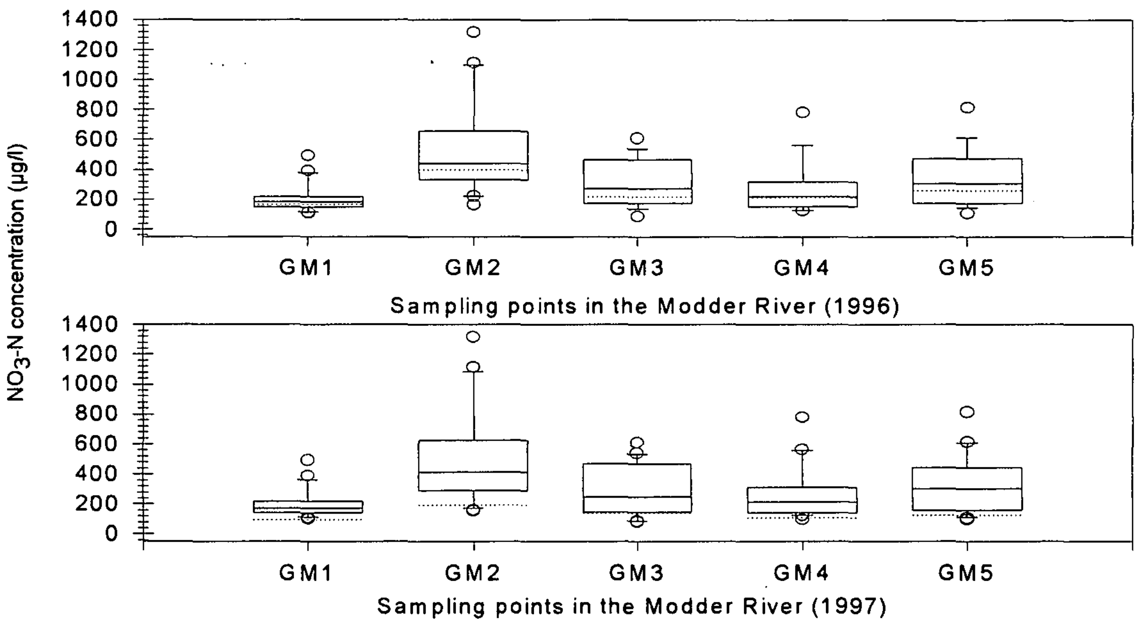


Figure 2.16: The downstream variation in NO_3-N concentration in the Modder River for 1996 and 1997. For explanation see Figure 2.5.

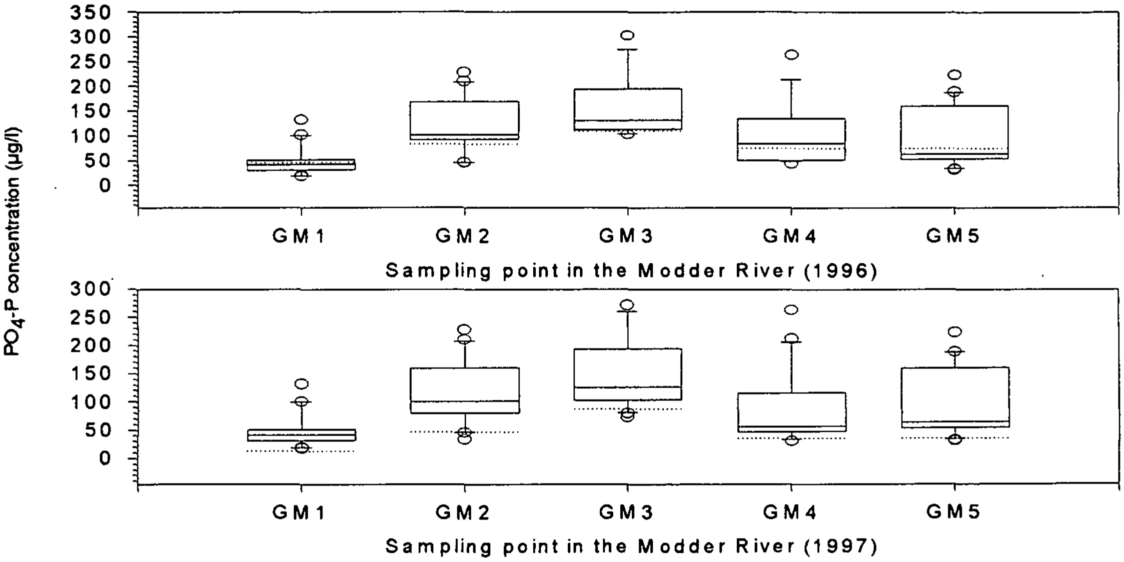


Figure 2.17: The downstream variation in PO_4-P in the Modder River for 1996 and 1997. For explanation see Figure 2.5.

2.3.5) Silica-silicon (SiO_2-Si)

The SiO_2-Si concentration in the rivers ranged between 2.6 - 5.4 mg/l with an average of 3 mg/l in the Modder River and 3.8 mg/l in the Klein Modder River (Figure 2.18 & 2.19). The highest concentrations were at KM2 (6.4 mg/l) and GM1 (5.2 mg/l). At the rest of the sampling points, both in the Modder and Klein Modder River, the concentrations were nearly the same.

No seasonal pattern was observed, however, at most of the sampling points in both the Modder and the Klein Modder Rivers, the lowest SiO_2-Si values occurred during the winter (August).

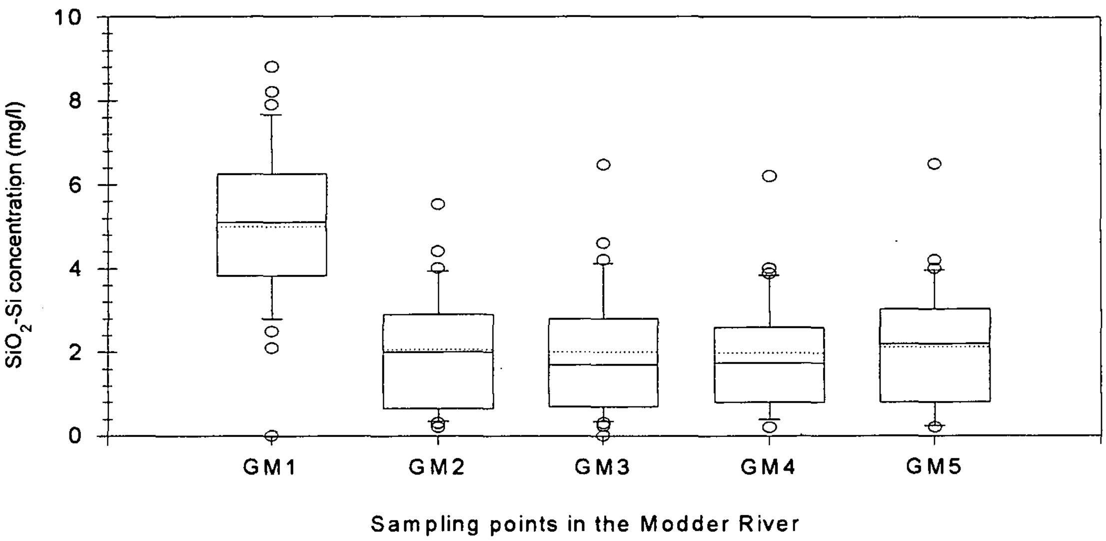


Figure 2.18: The downstream variation in SiO_2-Si concentration in the Modder River. For explanation see Figure 2.5.

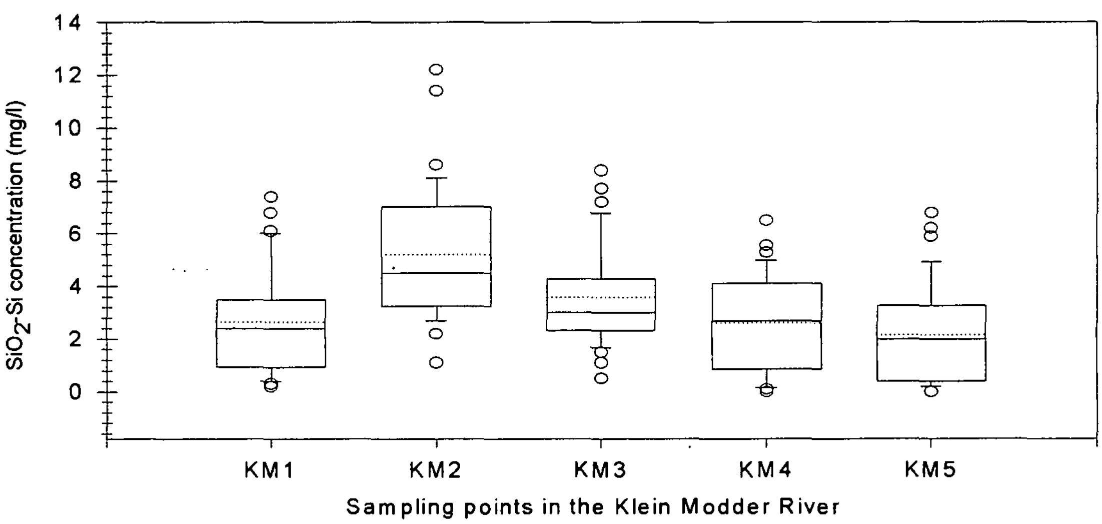


Figure 2.19 : The downstream variation in SiO_2-Si concentration in the Klein Modder River

2.3.6) Oxygen, temperature and pH

The dissolved oxygen concentration, in the Modder River, varied between 4 and 8 mg/l (Figure 2.20) and in the Klein Modder River between 6 and 8 mg/l (Figure 2.21). The mean percentage saturation at the sampling points in both rivers were high, except at

KM2 (the sewage outflow) and at GM1. At GM1 the growth of *Azolla filiculoides* (Lam.) decreased the percentage oxygen saturation to almost zero at some stages and this account for the low average. The following mean percentages oxygen saturation was found: In the Klein Modder River - 90% and in the Modder River- 84%.

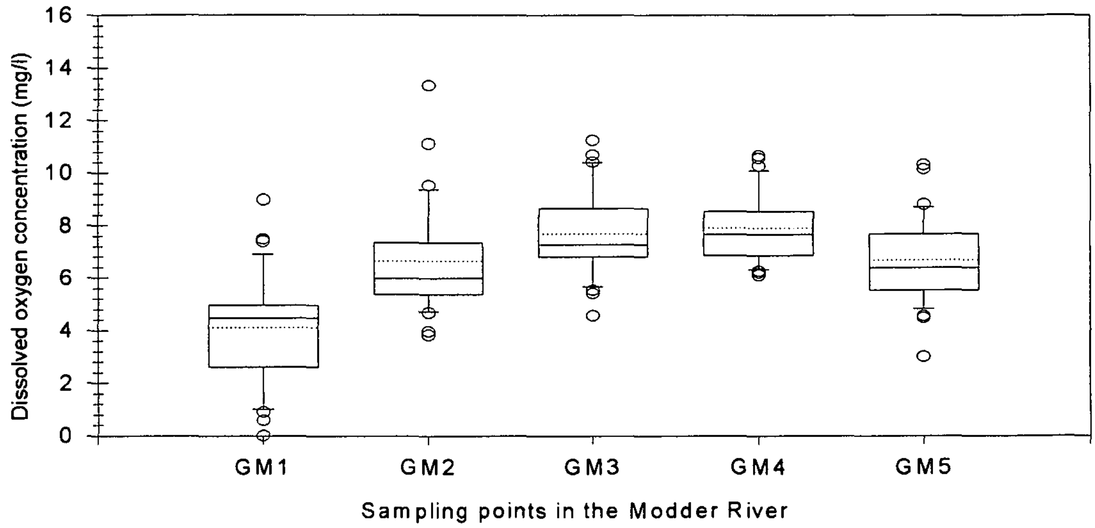


Figure 2.20: The dissolved oxygen concentration downstream in the Modder River. For explanation see Figure 2.5.

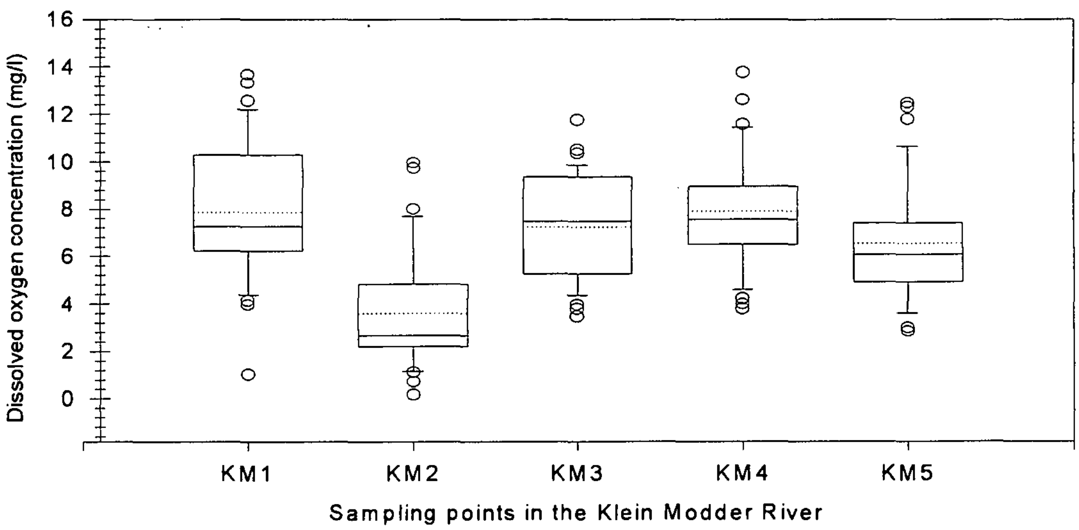


Figure 2.21: The downstream variation in dissolved oxygen concentration in the Klein Modder River. For explanation see Figure 2.5.

The temperature in the Modder and Klein Modder Rivers followed the seasons, being high (27°C in the Modder River and 23°C in the Klein Modder River) in the summer (September - May) and low (±7°C in both the Modder and Klein Modder Rivers) in the winter (June - August), with a pattern similar to that of the mean air-temperature (Figure 2.22 & 2.23).

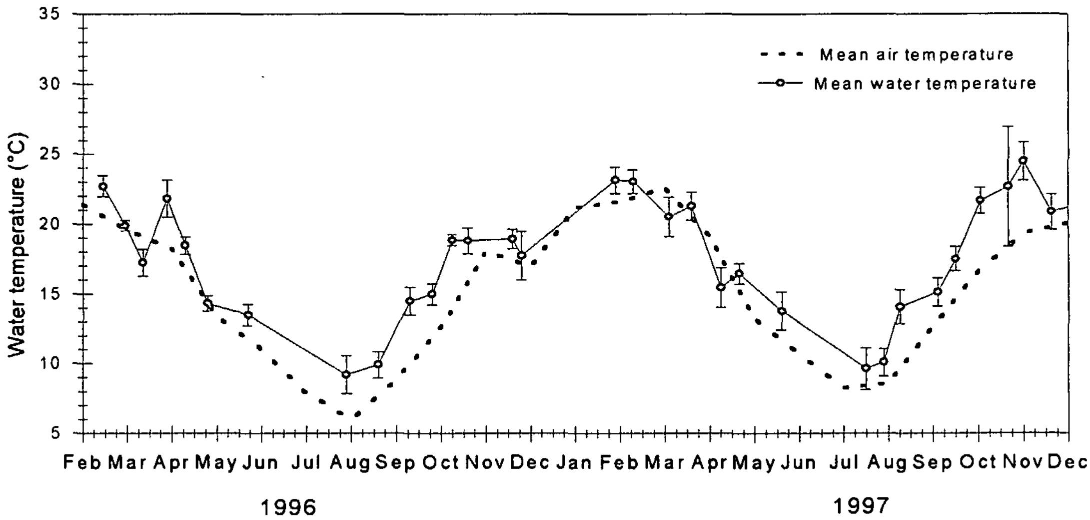


Figure 2.22: <u>Seasonal variation in mean water temperature in the Modder River</u>

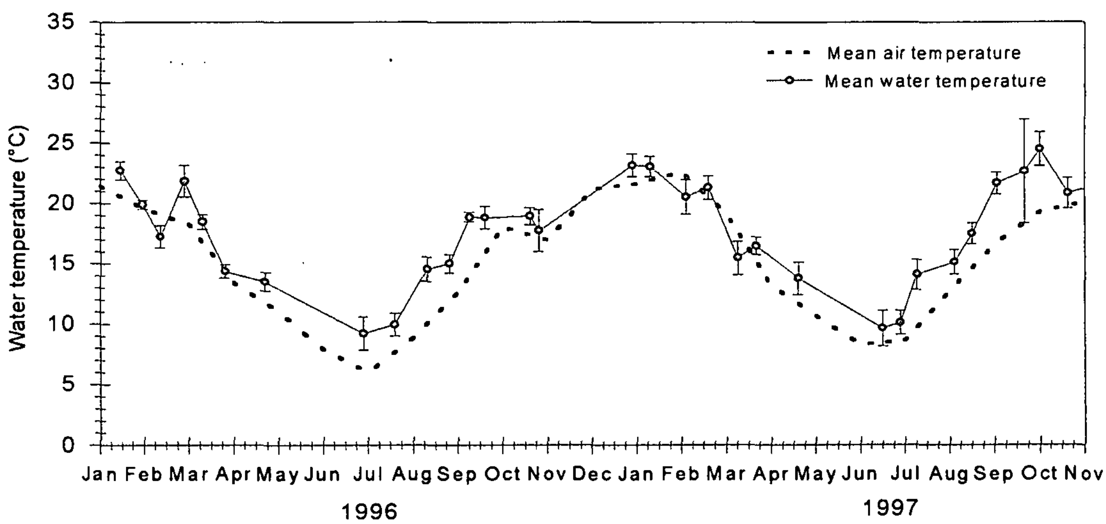


Figure 2.23: <u>Seasonal variation in mean water temperature in the Klein Modder River</u>

The pH in the Modder River (mean = 8.06) (Figure 2.24) and the Klein Modder River (mean = 8) (Figure 2.25), indicate that the water was mainly alkaline. The highest mean pH occurred at GM3, KM3 and KM4. It is important, however, to note that these measurements (dissolved oxygen, temperature and pH) were made at a given time and that they can change, rapidly, both spatially and over time.

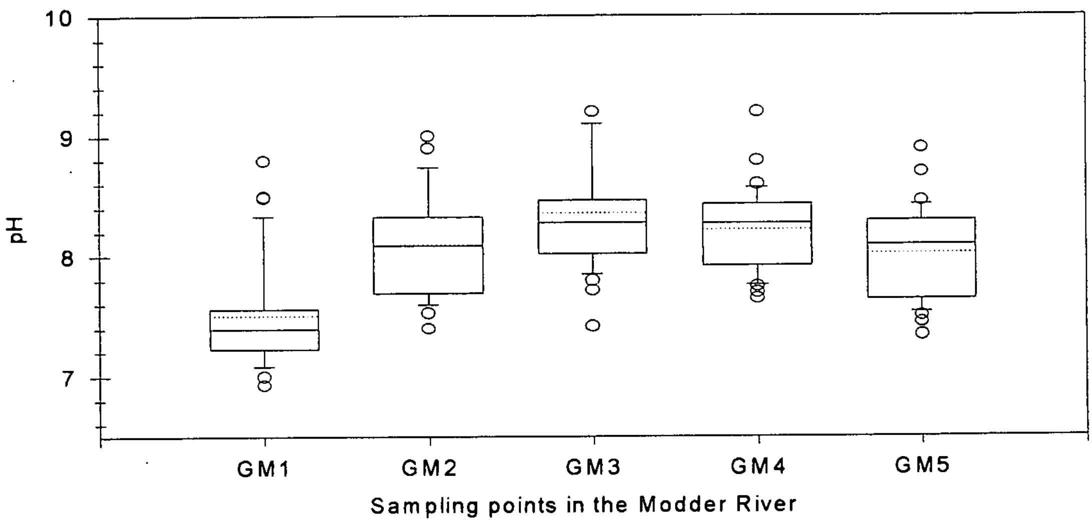


Figure 2.24: The variation in pH downstream in the Modder River

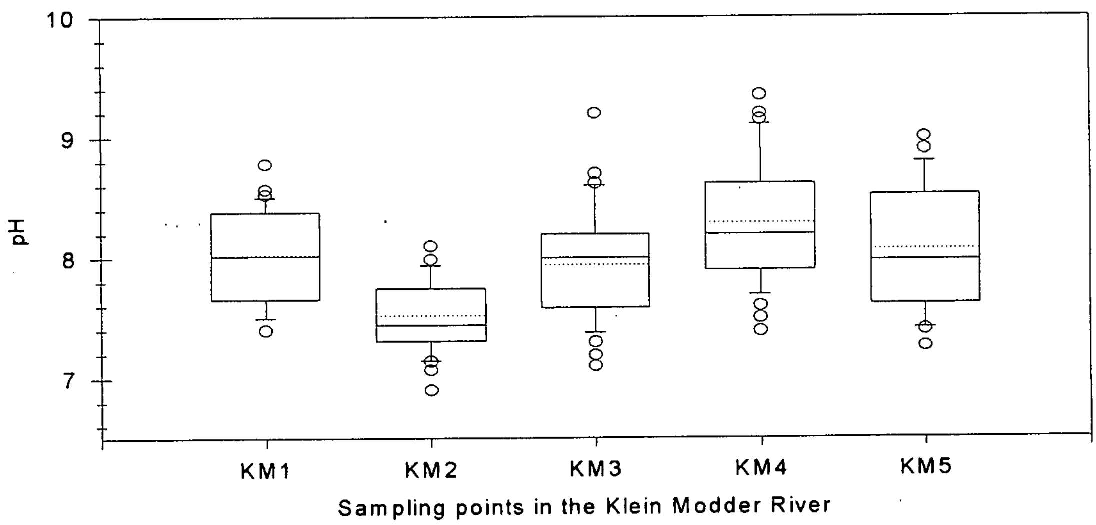


Figure 2.25: The variation in pH downstream in the Klein Modder River

2.4) DISCUSSION

2.4.1) Turbidity and flow

The significance of turbidity as a limnological optical measurement has been questioned as it does not account for the absorption of light by all the components (Walmsley & Bruwer, 1980). However, for waters where transparency is governed by suspended inorganic material (such as in the Klein Modder and Modder Rivers), turbidity is of considerable value, because light attenuation is primarily caused by scattering, and turbidity is a measure of this optical property.

The higher turbidity in the Klein Modder River (mean = 97 NTU's) when compared to the Modder River (mean = 86 NTU's) demonstrates the fact that the Klein Modder River is much shallower and more turbulence occurs which suspends solid materials in the water. Turbidity increased after rainfall in both rivers and thus reflects the mobilisation of topsoil during stormwater run-off. This concurs with the Vaal River where the most important factor that influencing the turbidity, and thus the euphotic zone, is discharge (Roos & Pieterse, 1995a). However, stream steady-state, where the solid material transported from stream basins is considered equal to that eroded from upland slope, cannot be assumed (Trimble, 1975). The higher turbidities at the start of the rainy season in both rivers (November 1996 - February 1997) compared to other rainfall periods could be ascribed to more turbid stormwater run-off. After the dry season, when almost no rain occurred, more topsoil was mobilised by the first rain than during the rest of the rainy season.

Long-term variations in flow are caused by climatic variations as well as by human impact (Burt, 1992). These human influences can be classified into two groups: i.e. direct impacts such as construction and operation of reservoirs, direct abstractions for domestic supply or irrigation and indirect influences such as agricultural practices, deforestation, urbanisation and drainage of wetlands. The rapid urbanisation of Botshabelo influences the catchment area of the Modder River, the sparse vegetation cover causes increasing stormwater run-off with higher turbidity values. The flow regime of the upper Modder River is also influenced by the construction of impoundments at various sites e.g. Rustfontein Dam, Mockes Dam and Mazelspoort Barrage. In the Klein Modder River, Botshabelo Dam as well as the Vadersgift impoundment influences the flow. Thus, human actions coincide with natural variations in flow. Unfortunately, a quantitative evaluation of human impact on streamflow is complicated because of the simultaneous operation of numerous factors e.g. storm events, droughts, agricultural and industrial practices.

2.4.2) Conductivity

In many instances an excellent relationship can be established between total dissolved salts and conductivity. Grove (1972) found a strong correlation between dissolved salts and conductance in West African rivers. Differences in conductivity mainly result from the concentration of the changed ions in solution and, to a lesser degree, from ionic composition and temperature (Allan, 1995).

The average conductivity for the Klein Modder was 52 mS/m and for the Modder River it was 36 mS/m. This is well within the target guideline range proposed by the South African Water Quality Guidelines for Recreational Use (DWAF, 1993a). No health, aesthetic or treatment effects are associated within this range (0-70 mS/m). The average value of typical, unpolluted rivers in general is approximately 35 mS/m. (Webb & Walling, 1992). However, the conductivity in both the Modder and Klein Modder Rivers was much lower than, for example, that of the Vaal River, South Africa (Roos & Pieterse, 1995b) with an average of 76 mS/m. The conductivity of the Orange River, South Africa, for 1977 to 1997, was much lower, being between 18-30 mS/m (DWAF, 1997).

The higher conductivity, during the dry season, in both the Klein Modder and Modder Rivers is in accordance with Awachie (1981), who states that in African rivers the conductivity is normally higher during the dry season. This does not necessarily mean that the total quantity of salts is less during flood periods, but that it rather is a reflection of the more diluted conditions. Alternatively, ion concentration might not change greatly with fluctuations in discharge. This is expected when water chemistry reaches an equilibrium with the soil through which it percolates, or when concentrations approach saturation values. Additional to these two common patterns, however, some ions have been found to increase in concentration with rising discharge (Allan, 1995; Golterman, 1975b).

The mean conductivity at KM2 (70 mS/m), was higher than at the other sampling points and can be ascribed to the salt content in treated sewage as well as chlorination of the effluent before discharge. Although the degree of salinisation is less extreme than with mine effluents, treated sewage effluents contain much higher concentrations of salts than are found in domestic water supplies (Ferrar, 1989). Effluents usually have a high chloride content (originating from urine) because the works do not alter it in transit, or because of a final chlorination step. Rivers, therefore, tend to show increased chloride concentrations (and thus conductivity) below sewage treatment works and industrial effluent discharge points (Hewitt, 1991). This phenomenon can be observed in the Klein Modder River. Conductivity decreased gradually in the Klein Modder River, from KM3 to KM5, probably due to dilution of the sewage outflow.

Between sampling points GM3 and GM4 a 47% decrease in conductivity was observed. Mockes Dam lies between GM3 and GM4. Impoundments store water from peak floods and this water is usually low in conductivity. Thus, conductivity levels below a impoundment will usually be lower than upstream, as is the case at GM3 (before Mockes Dam) and GM4 (after Mockes dam). In the Great Fish River, South Africa, conductivity dropped from 210 mS/m to 130 mS/m after an impoundment (Palmer & O'Keeffe, 1990a)

Although conductivity in the upper Modder River (the study area) was relatively low (40 mS/m), it can increase in the future due to increase in effluent discharge (in volume) as the city expands its waterborne sewage reticulation system to replace the pit latrines.

2.4.3) Turbidity vs. conductivity vs. rainfall

The turbidity of a system also depends on factors other than rainfall, such as total dissolved salts (TDS), water flow, geological formations and marginal vegetation cover (Ferrar, 1989).

Conductivity was inversely related to turbidity in both the Klein Modder and Modder Rivers. In the Modder River 68% percent of variation in conductivity could be ascribed to the variations in turbidity. This relationship was also found by Roos & Pieterse (1995b) who found that salinity in the Vaal River, South Africa, displayed seasonal changes that were strongly influenced by turbid conditions following rainfalls, and that 56% percent of the variation in conductivity in this river was associated with variation in turbidity. They also stated that most rivers exhibit decreasing conductivity with increasing flow (high rainfall).

Ferrar (1989) reported that immediately after the onset of a storm, conductivity levels in rivers increase sharply due to salts being flushed down the system and as dilution occurs, they then rapidly return to pre-flood, or even lower, levels. Conductivity in the Modder River and Klein Modder River decreased dramatically after rain (high flow conditions), while turbidity increased. However, these measurements were taken days after the occurrence of heavy rains. The conductivity in the Klein Modder and Modder Rivers probably increased briefly during or just after rainfall. Thus, it is possible that the high flush-down pulse, during rainfall, was missed and only the following lower conductivity levels were measured.

2.4.4) Nitrate-nitrogen (NO_3-N) and phosphate-phosphorus (PO_4-P)

The average NO_3-N concentration of the Modder River (230 µg/l) was much higher than the 100 µg/l for unpolluted world rivers (Webb & Walling, 1992), but lower than for the eutrophied Vaal River, South Africa (400 µg/l) (Roos & Pieterse, 1995a). The mean NO_3-N concentration in the Klein Modder River (860 µg/l) was more than twice that of the Vaal River. In the Orange River, South Africa, the mean NO_3-N concentration was 390 µg/l, for 1977 to 1997 (DWAF, 1997).

The mean PO_4-P concentration in the Klein Modder River (260 µg/l) was much higher than in the eutrophied Vaal River (18 µg/l) (Roos, 1991) as well as in the Orange River (27 µg/l) (DWAF, 1997). At the site of urban and industrial effluents, flowing into the Buffalo River, Eastern Cape, TP exceeded 1000 µg/l on 70-80 % of sampling occasions and maximum soluble reactive phosphates concentrations were between 2 770 and 9 300 µg/l, much higher than that of the Klein Modder River (Palmer & O'Keeffe, 1990b). The TP concentration in unpolluted water should be between 10 - 50 µg/l (Wetzel, 1983). The higher NO_3-N and PO_4-P concentrations in the Klein Modder River than in the Modder River, once again emphasise the effect of enrichment, due to run-off, from Botshabelo on the system.

Roos & Pieterse (1995a) found that in the Vaal River, South Africa, most of the annual load of soluble nutrients, such as phosphorus and nitrogen, are transported during periods of high discharge and also state that the main sources of nitrates in streams are surface run-off from the catchment areas, following rainfall. This phenomenon was also illustrated in other river systems, such as the Humber Rivers, England (House *et al.*, 1997), the Barwon-Darling River, Australia (Bowling & Baker, 1996), the Swan River, Australia (Thompson & Hosja, 1996) and the upper Orange River, South Africa (Keulder, 1979). All this is in accordance with the findings in the Modder River system where NO_3-N and PO_4-P increased with increased discharge. However, during flood conditions NO_3-N decreased, probably due to wash-out (e.g. March 1997).

Two mechanisms may be involved in the high concentration of PO_4-P after rains, namely percolation and erosion (Golterman, 1975b). Percolation is a source of substantial quantities of P probably due to wash-out from fertilised agricultural soils. Erosion, on the other hand, occurs where the soil is geologically unstable, or plant cover is sparse or absent. Since erosion represents the removal of solid materials, such as clay particles, the phosphate adsorbed to the particles is carried off as silt (Horne & Goldman, 1995). The phosphate transported during erosion, is usually much lower than transported through percolation.

PO_4-P constitutes 38% of the TP fraction in the Klein Modder River and 21% of the TP fraction in the Modder River. In the Vaal River, a statistically significant positive correlation between PO_4-P and TP was illustrated (r^2 = 0.46, p >0.001) (Roos & Pieterse, 1995a). Roodeplaat Dam (De Wet, 1986) and Hartbeespoort Dam (Robarts, 1984), both eutrophic systems, showed that the PO_4-P fraction was approximately 90% and 66% of the TP fraction, respectively. The ratio of PO_4-P to TP is usually less in oligotrophic waters (lakes), whereas at high TP concentrations, the dissolved inorganic phosphorus pool reaches almost 100% of the TP (Harris, 1986).

Grobler and Toerien (1986) predicted the possible total phosphorus concentration in Mockes Dam by assuming a P limitation of 1000 μg/l and runoff of 0.2 times the mean annual runoff. They predicted a TP concentration of 276 μg/l in 1990 and 351μg/l in the year 2000. A mean value of 340 μg/l was observed during 1996 at GM2 and GM3. The predicted value was thus an underestimation of the observed value.

GM2 is the sampling point just below the confluence of the Modder and Klein Modder Rivers. At this point, the increase in concentrations of NO_3-N and PO_4-P when compared to GM1 is a direct result of treated sewage which flows into the Modder River *via* the Klein Modder River. There is a tendency for the nutrients to decrease downstream, which is an indication of dilution and the self-purification capacity of the system. In the River Cam, England, increasing concentrations of phosphate, nitrate, chloride, ammonia and detergents were found in the river at a sewage outflow, but at a sampling site downstream, the river had purified itself, excepting for chloride and phosphate contents (Hewitt, 1991).

Because the throughputs in lotic ecosystems are unusually high, they provide a natural cleansing ability mainly through dilution. Consequently, rivers have some natural recovery capabilities which facilitates the restoration of these ecosystems (Allan, 1995), called the assimilative capacity (Webster *et al.,* 1983). In contrast to the high resilience exhibited by stream ecosystems, low resistance to disturbances is apparent (Webster & Patten, 1979). Disturbances involving addition to or removal from streams, such as organic pollutant discharge or flood scouring, influence the streams dramatically. These effects, however, are rapidly diminished by resiliency mechanisms.

At KM2 (the sewage outflow) the nutrient concentrations were always high but the algal biomass was relatively low (Chapter 3). This could be due to the possible high concentration of chlorine, in the treated water, which is toxic to living organisms or to the short residence time of the out-flowing water, limiting the development of the algae.

The lower NO_3-N and PO_4-P concentrations in the Modder River, during the second year of the study period, can be ascribed to lower rainfall, which resulted in a decrease in leaching and erosion. However, the PO_4-P and TP concentration at the sewage outflow (KM2) also decreased in 1997. At this point, the PO_4-P decreased from 821.9 µg/l in 1996 to 320.6 µg/l in 1997 and the TP from 1753.4 µg/l to 844.5 µg/l. The different methods used for the pre-digestion of TP, during the two years of the study period, can also possibly have caused the difference in TP concentration.

2.4.5) Silica-silicon (SiO_2-Si)

The average silica concentration in the Modder River (3 mg/l) and the Klein Modder River (3.8 mg/l), is much lower than both the world average (9 mg/l) and the average for African rivers (11.24 mg/l) (Golterman, 1975b, Allan, 1995) but comparable to other South African rivers, like the Vaal River (3 mg/l). In the Orange River, South Africa, the average silica value was 8.09 mg/l, for 1977 to 1997 (DWAF, 1997). In the Norfolk Rivers, England, silicon values ranged between 4.7 - 5.9 mg/l (Edwards, 1974) and in the River Moselle, France the average silica concentration was only 0.6 mg/l (Descy, 1993).

At some of the sites in the Modder and Klein Modder Rivers, silica concentration decreased when algal growth (chl-*a*) increased (Chapter 3). It can be assumed that silica was incorporated into the cellwalls of diatoms, because during the study period, pennate diatom species frequently dominated the algal assemblage. Silica concentrations are also dependent on temperature, having lower values at low temperatures (in the winter). This is demonstrated by the fact that the lowest SiO_2-Si values were measured in June in both the Klein Modder and the Modder Rivers.

2.4.6) Oxygen, temperature and pH

2.4.6.1) Oxygen

The mean percentage saturation of dissolved oxygen in the Modder (84%) and Klein Modder Rivers (90%) shows that oxygen limitation is not a problem for the fish and invertebrates in this system. In fact, dissolved oxygen is usually not a limiting factor in running water, except in situations where the depletion of oxygen is caused by intensive biological activity (Awachie, 1981; Allan, 1995). The Klein Modder and Modder Rivers are both shallow, turbulent rivers. This could explain the relatively high oxygen concentrations, as recovery from oxygen deficiency occurs more rapidly in shallow and turbulent reaches where there is maximum contact between water and air (Walling & Webb, 1992).

The highest dissolved oxygen concentrations coincided with maximum chl-*a* concentrations. For example, during the spring increase at GM2 and GM3 in September (80 µg/l chl-*a*) (Chapter 3), dissolved oxygen concentrations were 15 and 9 mg/l respectively, representing supersaturation levels. This trend was also recorded by Roos & Pieterse, (1996) in the Vaal River, South Africa.

2.4.6.2) Temperature

The highly significant linear relationships found between air and water temperature in both the Klein Modder and Modder Rivers, as well as the marked seasonal variation, showed that the temperatures of rivers and streams vary far more rapidly than those of lakes but, quite often this variation is over a much smaller range than that of at least the shallower parts of still water (Hynes, 1970). Generally, the surface water temperature follows ambient air temperatures but is also influenced by insulation, substrate composition, turbidity, vegetation cover, ground water and rainwater inflows (Awachie, 1981; Walling & Webb; 1992, Allan, 1995). In both the Modder and Klein Modder Rivers, there were a "lag" phase of 1-2 weeks before water temperatures reached the prevailing air temperature. The water temperatures never dropped lower or rise higher than those of the air, supporting the fact that water provide a stable environment in terms of temperature.

2.4.6.3) pH

Levels of pH vary greatly between different rivers and streams and are influenced by bicarbonate/carbonate alkalinity and the concentration of free CO_2 (Wetzel, 1983). The alkaline water of the Modder (mean = 8.06) and Klein Modder (mean = 8.0) Rivers is within the range of the prescribed Water Quality Guidelines for Recreational Use (DWAF, 1993a). Grassland rivers in Africa, such as the Modder and Klein Modder River tend to be neutral or slightly alkaline (Awachie, 1981). The pH of both the Modder and Klein Modder Rivers compares well with the mean pH of the Vaal River, South Africa, which varied between 6.3 and 9.2 (mean 8.1) and the Orange River, with a mean pH of 8.2 (Roos & Pieterse, 1995a). The highest mean pH values occurred at GM3, KM3 and KM4. Since algal blooms occurred at these points (Chapter 3), the increase in average pH could be ascribed to photosynthetic activity.

2.5) APPLICATION OF SOME OF THE RIVER CONCEPTS TO THE KLEIN MODDER AND MODDER RIVERS

2.5.1) The River Continuum Concept

In South Africa most rivers are modified in some way, whether by impoundment, extraction of water, or pollution. This is also the case in the Modder and Klein Modder Rivers. Both these rivers are modified by their impoundments as well as by pollution originating from Botshabelo. Ward and Stanford (1983) state that with increasing distance downstream from a dam, energy flows are reset and diversity patterns should be followed in the context of the river continuum. However, in the upper reach of the Modder River, there are many impoundments, relatively close to each other, each of them discontinuing the continuum. Thus, the RCC can not be successfully applied to these rivers, because the natural continuum and condition of the river is constantly being disturbed. The balance between the organic matter that is processed by a river order and that exported to the next order, a fundamental feature of river communities, is lost (Cummins, 1977).

2.5.2) The Serial Discontinuity Concept

All the nutrients in both the Klein Modder and the Modder Rivers decreased in concentrations downstream. This decrease in nutrient concentrations could either be because of dilution or to the presence of impoundments in the rivers. In the Klein Modder River there are impoundments before KM4 and KM5. In the Modder River there are impoundments before GM2 and GM4. Dams are physical barriers in river systems, and they and their associated impounded waters can result in changes to the natural flow regime, water temperature, sediment loads, and the chemistry of the receiving river (Stanford & Ward, 1979; Palmer & O'Keeffe, 1990b). As the river progresses downstream from the natural conditions are restored. In the Buffalo River, impoundments situated in the upper catchment caused reduction in the levels of phosphate and nitrate (Palmer & O'Keeffe, 1990b) as is the case in the Klein Modder and Modder Rivers. No downstream recovery rate for the Klein Modder and Modder Rivers were determined, but it is likely that it will be masked by stormwater run-off and pollution from urban and industrial areas. The impoundments acted as sink for nutrients originating from urban effluents and agricultural run-off.

Preliminary conclusions on the SDC in South Africa, based on investigations of the Palmiet and Buffalo Rivers are the following (Allanson *et al.*, 1990):

i) Small impoundments in the upper reaches increase average water temperatures and reduce the annual temperature range. Changes in available nutrients are slight and depend on whether water is released from the surface or from a depth.

ii) Medium-sized impoundments, in the middle reaches of the river, considerably reduced both average temperature and temperature ranges downstream. All major nutrient concentrations increased below the dams.

Thus, the downstream effects of impoundments depend on the parameter examined and the position of the impoundment along the river (Palmer & O'Keeffe, 1990b). The impoundments in the upper Modder and Klein Modder Rivers are relatively small, with short retention times (Table 2.2). Thus, they can act as a sink for nutrients for a short period of time, but they are not the major factors controlling the water quality. Dilution together with wash-out probably plays a greater role in influencing the concentration of nutrients downstream. Also, the impoundments do not seem to influence the temperature regime as is the case in the Palmiet and Buffalo Rivers.

2.6) CONCLUSION AND RECOMMENDATIONS

The physico-chemical characteristics of the Modder and Klein Modder Rivers show distinctive local variations without clear-cut seasonal trends, except for temperature. In terms of the nutrient content of the waters of the Modder and Klein Modder Rivers, these rivers can be classified as eutrophic (Mason, 1991). Both these rivers are also very turbid systems.

Nutrient concentrations in the Klein Modder River were consistently higher than in the Modder River, due to enrichment from treated sewage effluent from Botshabelo, as well as from polluted stormwater run-off. Nutrient concentrations in the Modder River were the highest at GM2 and much lower at GM5. This decrease could be due to i) dilution and/or ii) self-purification through assimilation/sedimentation of pollutants. The impoundments could also make a small contribution towards the downstream "purification" of the system.

The detrimental influence of Botshabelo on the water quality of the Modder River is considerable. The inflow of the Klein Modder River into the Modder River caused a 112 % increase in PO_4-P, and a 171 % increase in NO_3-N River from GM1 to GM2. It is important to remember that the dilution effect, observed during this study, can be ascribed to an above average rainfall and high flush-out rate during the study period.

The long-term effect of Botshabelo on the system can be seen by comparing the observed values with those predicted by Grobler and Toerien (1986). When comparing the values obtained, with those of 10 or 20 years ago, a definite increase can be seen. With increasing population numbers, the nutrient loading of the Modder River system will increase continuously and the quality of the water will continue to deteriorate. A thoroughly integrated management plan is needed to limit the effects of pollution. This includes strict control over nutrient content of effluents as well as the volumes released.

Limitation of one essential nutrient is able to prevent algal growth. For example, no algae can grow without phosphate, although this does not mean that all excessive algal growth is due to the input of TP (Golterman, 1975b). The obvious solution to remedy excessive algal growth and eutrophication in rivers such as the Modder River is to restrict inputs of phosphate. The delay in the replacement of phosphates by non-phosphate containing elements has probably done more damage than any other source of pollution (Golterman, 1975b).

Some case studies regarding the restoration of eutrophied rivers by P limitation through more efficient sewage plants are, for instance, the Thames (England) and Red Cedar (America) Rivers (Jeffries & Mills, 1990; Ball & Bahr, 1975). The Thames River's water quality has declined in the nineteenth century due to pollution from human and animal wastes to such an extent that 1858 was called "The year of the great stink". By implementing modernised, for that time, sewage networks, the river started to recover. However, in 1955 the increased volume of sewage, due to population growth and urbanisation, once again reduced the water quality. No fish were found in a radius of 70 km around London, due to the anoxic conditions in the Thames River. Further improvements were made to the treatment of organic wastes and by 1973, 62 species of fish were found in the river.

The Red Cedar River also underwent the change from a pristine river supporting trout population to a grossly polluted one carrying the wastes of tens of thousands of people. Urbanised areas have had a profound impact on the ecology and the human population of the catchment approached 150 000 in 1975. The largest single source of P to the Red Cedar River was from sewage treatment plant effluents (44%) and urban run-off (34%). Effluent from plating and sewage plants have been eliminated and the discharges of an adverse nature from the urban areas have been minimised. The river has since recovered to almost its original condition.

Looking at the above-mentioned case studies, it can be clearly seen that rivers have the capability to restore themselves. Thus, if the needed P limitation is implemented in the Modder River system, the system will not deteriorate to such an extent that the water becomes unusable. However, it is important to identify point and/or diffuse source dominance before phosphorus limitation can be successfully implemented. In Botshabelo, stormwater run-off probably constitutes a large part of the pollution reaching the Klein Modder River.

There is no simple recipe, however, applicable to all cases, to solve the eutrophication problem (Vollenweider, 1981). Policies and programs to counteract and prevent eutrophication in water bodies should become a fundamental part of water resource planning. Together with this, an objective assessment of the significance of eutrophication should be made in relation to the particular usage, purpose or function of the different water bodies.

CHAPTER 3

PHYTOPLANKTON OF THE MODDER RIVER

3.1) INTRODUCTION

Algae are quantitatively the most important group of primary producers in aquatic environments (Golterman, 1975b), acting as an internal energy supply by providing food for fish and other aquatic organisms (Cummins, 1974). Small standing crops of periphyton are capable of supporting relatively large standing crops of consumer organisms. This is due to the rapid turnover rates of algae in comparison with the slower turnover rates of animals and occurs even under conditions of low light intensity (Minshall, 1978). Excess algae, or undesirable algal types can, however, become a nuisance and interfere with the uses of a water body. They can also cause taste and odour problems as well as gastro-enteritis and skin irritations (Bowling & Baker, 1996).

Rivers carry a diverse assemblage of suspended algae and their presence, in lowland rivers, has been recognised for almost a century (Köhler, 1994). Since then, investigations have continuously been done on the dynamics of algal growth in rivers and the influencing factors thereof. Thus, for a better understanding of the functioning of aquatic ecosystems and to evaluate the relationships between their various components, data on the dynamics of algae are needed. This can be done in various ways. Analyses of phytoplankton assemblages in rivers allow, amongst others, investigations of temporal changes in species abundance and species composition, two important aspects of community structure (Pieterse, 1987). However, factors providing favourable conditions for algal growth, such as high light availability and high nutrient concentrations, must also be investigated.

In the Modder River, the phytoplankton was investigated to determine whether the algal communities react to environmental changes, including eutrophication. Since the presence of some algae as well as diversity of communities indicates the level of eutrophication (thus the water quality), algal identification can also prove to be a useful tool in this regard.

3.2) FACTORS INFLUENCING ALGAL GROWTH IN THE MODDER RIVER

The physiological and biochemical properties of algae are dependent on external conditions (Kozitskaya, 1990) and through their adaptive strategies, they survive in a variable environment (Köhler, 1994). Various of these variables are discussed in the following sections.

3.2.1) Flow

One of the major problems that algae experience in rivers, is the persistent and unidirectional passage of water (Hynes, 1970; Golterman, 1975b). Variability in flow may lead to movement or suspension of material with an increase in turbidity and attenuation of light. Thus, periods of relatively stable environmental conditions necessary for the achievement of an equilibrated steady-state are much shorter due to shifts in river morphometry, velocity and in trophic interactions along a river (Vannote *et al.*, 1980). Descy (1993) and Dokulil (1994) also stated that flood episodes are major disturbances in otherwise continuously mixed environments (rivers), affecting the composition and biomass of the plankton. However, it was found that some algae require current for optimal growth (Hynes, 1970), illustrating their evolutionary adaptive strategies. To support this, Lock & John (1979) found, by determining the effect of flow on P uptake, that water movement caused an increase in uptake of ^{32}P. No stimulation was found with river water containing 105 µg/l PO_4-P, suggesting that at such high concentrations the demand for P can be satisfied by molecular diffusion processes alone.

In the Modder and the Klein Modder Rivers, flow is constantly changing and in the rainy season the rivers experience periods of relatively high flow (up to 19 m^3s^{-1}) and very low flow (< 1 m^3s^{-1}) in the dry winter period.

Together with flow, several environmental factors are responsible for the regulation of the spatial and temporal growth of phytoplankton. The most basic requirements for algal growth are temperature and light, although inorganic and organic nutrients also play critical roles in algal growth (Hynes, 1970; Horne & Goldman, 1995).

3.2.2) Light

Different algal genera have different light requirements. For instance, many diatoms appear to be fairly indifferent to light, while many Chlorophycaceae require fairly high intensities of light (Hynes, 1970).

Underwater light attenuation in turbid systems is largely a function of suspended particle concentration and size (Dokulil, 1994). Consequently, Secchi depth, euphotic zone and the extinction coefficient can all be related to the quantity of suspended solids.

The Modder River is a very turbid system (modder = mud) (Chapter 2), thus the quality of penetrating light can be altered and light availability is restricted. It is possible that in this system primary production is limited by light rather than by nutrient availability (Grobler & Toerien, 1986), because the energy required for phytoplankton growth is strongly regulated by availability of underwater light which depends on the critical mixing depth, fluctuating light intensities and the passive or active movement of algae in the water (Dokulil, 1994). Phytoplankton primary productivity essentially follows the same pattern in turbid waters as in clear waters, excepting that productivity profiles are compressed, due to the rapid attenuation of light (Grobbelaar, 1985). The critical mixing depth is also important for overall productivity, as it determines the time phytoplankton spend in the light and dark. Thus, the unfavourably low photic zone coupled with the mixing depth are the important factors that control overall primary productivity in these type of systems (Grobbelaar, 1989).

3.2.3) Temperature

Temperature is a major factor controlling the role of photosynthesis in all plants (Davison, 1991). Arrhenius proposed the following relationship:

$\mu = Ae^{-E/RT}$ or $\log_{10}\mu = \log_{10}A - (E/2.303R)^1/T$ (Du Preez, 1978)

where T = absolute temperature, °K

R = universal gas constant

E = activation energy

A = entropy constant

Temperature optimum curves are typically based on growth vs. temperature data for a single species. These data generally show no growth at very low temperatures, followed by an exponential increase in growth with increased temperature over a greater part of the temperature range. However, the growth rate eventually levels off to a maximum value at the optimum temperature, and then begins to decline at very high temperatures until growth finally ceases at an upper temperature limit.

Greater algal respiration often occurs in shallow waters due to higher water temperatures when compared to deeper systems (Dokulil, 1994). A temperature elevation of 5°C can produce a heavy growth of green algae when the community, light, nutrients, flow and grazers are the same (Cummins, 1974). It is difficult, though, to disentangle the effect of temperature from that of light. It has been illustrated that many stream algae have been driven by evolutionary circumstances to operate at temperatures well below optimum, because of the effect of shading in running waters (Hynes, 1970). The waters of the Modder and Klein Modder Rivers followed the same pattern of variation in air temperature, creating varying conditions for algal species to survive, as well as seasonal changes in community composition.

3.2.4) Nutrients

Many resources limit phytoplankton growth rates. Liebig's law of the minimum states that only a single growth factor can be limiting at any given time. Since different organisms in a population could be limited by different limiting resources, it is necessary to identify the limiting factors for the various populations of a community (Melack, 1995). Important concepts in understanding interactions between algal growth and nutrient dynamics are the optimum nutrient ratio and the nutrient loading concept.

The optimum nutrient ratio is the ratio at which a transition from one nutrient limitation to another occurs (both could thus be limiting), or where the resource requirement is such that the resource is not in short supply relative to another (Rhee & Gotham, 1980). If the optimum N:P for two species are 20 and 10 respectively, then both will be P limited when the ratio is >20. However, the second species will be more P-limited than the first. If they have similar μ_{max} (maximum growth rate) values, the first species will eliminate the second species at N:P ratios > 20. In principle, the concentration of a limiting nutrient, can be estimated from the Redfield ratio (106C:16N:1P) for the key nutrients. If N, P and Si are in excess of requirements for the algae, the ratios are irrelevant.

However, in rivers and lakes other factors influencing algal physiology, together with the concentration of the key nutrients, determine the relative growth rates and structure of the community. Although N and C are the key nutrients for most algae, the availability of a third nutrient may limit the growth of particular species e.g. the availability of silica can limit the growth of diatoms (Grobbelaar & House, 1995; Kilham, 1971). However, it is unlikely that SiO_2-Si will ever be a limiting factor in running water as most rocks and soils contain silicates (Hynes, 1970). Other important aspects complicating nutrient and algal interactions are the interactive effects between nutrients and trace components e.g. vitamins or trace metals, as well as the chemical forms of N and P (Grobbelaar & House, 1995). Different algal species prefer different forms of N (e.g. NO_3-N or NH_4-N) as well as different forms of P.

The nutrient loading concept states that a quantifiable relationship exists between the quantity of nutrients reaching a water body and its trophic degree measurable with some kind of trophic scale index (Vollenweider, 1976) Thus, chlorophyll-*a* could be predicted from the loading characteristics of lakes, by relating phosphorus concentration to loading, with the following equation:

$[P]_\lambda = (L_p/q_s).(1/ 1 + \sqrt{z}/q_s)$

where $[P]_\lambda$ = P concentration

(L_p/q_s) = average inflow concentration

z = mean depth

q_s = hydraulic load (which expresses the relationship between the hydrologic properties of the basin and the lake).

Since it has been demonstrated that a relationship exists between the spring overturn phosphorus concentration and the average chlorophyll build-up during the following season (Sakamoto, 1966), Vollenweider (1976) postulated that the above equation could be read in the sense of a production equation in which critical loading is measured by a biological parameter such as chlorophyll. In order to test this hypothesis, average epilimnetic chlorophyll concentrations for 60 lakes have been plotted against P-concentration (Vollenweider, 1976). Although there were a few uncertain values, the relationship was clear, with a correlation coefficient of 0.868. Furthermore, each lake plotted has been characterised as oligo-, meso- or eutrophic. Oligotrophic lakes would be found to the left of the 10 mg/m^3 critical loading P concentration, and eutrophic lakes would be found to the right of the 20 mg/m^3 critical loading P concentration. Thus, the relationship between phosphorus loading and chlorophyll can be used to predict biomass in terms of chlorophyll, in relation to the specific P loading characteristics. The prediction is probably quite good for low and medium productive lakes, yet less certain for highly productive lakes.

It has been observed that algal growth increases at points where rivers flow through areas that are rich in nitrate (NO_3-N) and phosphate (PO_4-P), contributed by agricultural or industrial practices (Hynes, 1970), as is the case with the Modder River. The concentration for limitation of the growth rate of algae, by inorganic phosphate, is less than 3 μg P/l (Wetzel, 1983). Because nitrogen in most natural waters exceeds phosphorus by an order of magnitude, phosphorus is most often the limiting factor. However, the applicability of nutrient limitation in flowing waters has long been questioned, based on the reasoning that currents transport a fresh supply of nutrients to the vicinity of an individual cell. Thus, no matter how low the concentration of nutrients in the water, the cell's environment is "physiologically enriched" by the flow of water carrying dissolved material over its surface (Hynes, 1970; Allan, 1995). The nutrient concentrations in the Modder and Klein Modder Rivers were always high during the study period (Chapter 2).

Other factors that influence algal growth in lotic systems are alkalinity, pH, the substratum, scouring and grazing by animals (Hynes, 1970). Thus, algal growth is a function of a combination of different factors. In this study, the seasonal and spatial variations in algal growth were determined, taking into consideration factors that could possibly influence it.

3.3) MATERIAL AND METHODS

3.3.1) Study site

The chlorophyll-*a* concentration of the Klein Modder River and the Modder River were measured fortnightly, from February 1996 to December 1997 at ten sampling points, five in the Klein Modder (KM1 - KM5) and five in the Modder River (GM1 - GM5) (Chapter 2, Figure 2.1).

3.3.2) Determination of chlorophyll-*a* and algal species

The use of chlorophyll-*a* as an index of trophic status is based on the fact that it is normally the most abundant and important pigment in phytoplankton cells. Thus, measurements provide a convenient estimation of the algal biomass (Walmsley, 1984). Chlorophyll-*a* in the Modder and Klein Modder Rivers was measured using a modified method, as described by Sartory & Grobbelaar (1984), and involved filtering a known volume of water, whereafter the filter paper was boiled in 10 ml 95% ethanol at 78°C. The absorbance was measured at 665nm and 750 nm. After adding 100µl of 0.3 N HCl the absorbance was measured again after 2 minutes at 665nm and 750 nm with a Philips UV\Visible Spectrophotometer PU8700 Series. The following formula was used:

Chlorophyll-*a* in extract (mg/l) = $(A_{665} - A_{665a}) \times 28.66$ where:

A_{665} = absorbance of ethanol extract at 665 nm before it was acidified minus absorbance at 750 nm.

A_{665a} = absorbance of the acidified ethanol extract at 665 nm minus the absorbance at 750 nm.

The concentration of chlorophyll-*a* in the original sample:

$$\text{Concentration } (\mu g/l) = \frac{\text{concentration of extract} \times 10 \text{ ml(extract volume)}}{\text{volume of sample in litre}}$$

The dominant algal species were identified with a Zeiss light microscope after fixation with 2% formaldehyde. The number of a specific algal species was determined in a known volume of water, counting the individuals occurring in 20 blocks of known dimensions. The result was multiplied by a constant to obtain the total counts. Algal species was determined as a percentage of the total community.

3.3.3) Determination of biochemical oxygen demand (BOD)

The BOD was determined in duplicate by placing 100ml of the collected sample in a BOD flask and kept at 20°C in the dark for 5 days. The initial dissolved oxygen was measured again after 5 days. No dilution water was added (*Standard methods for the examination of water and waste water,* 1995). BOD was determined at GM1 (reference point), GM2 (after the confluence with the Klein Modder River to determine any variations), and GM5 (downstream in the river to determine whether water was restored to its original quality in terms of BOD).

3.4) RESULTS

3.4.1) Spatial and seasonal variation in chlorophyll-*a*

The chl-*a* concentration of the Klein Modder River (excepting for KM2) ranged between 1 and 251 µg/l (mean = 43 µg/l) (Figure 3.1) and that of the Modder River between 1 and 201 µg/l (mean = 20 µg/l) (Figure 3.2). Thus, the average chlorophyll-*a* concentration in the Klein Modder River was more than twice that in the Modder River. However, as described in Chapter 2, the nutrient concentrations in the Klein Modder River were much higher than those of the Modder River. There was a 50 % increase in the mean chl-*a* concentration in the Modder River from GM1 to GM2, with a gradual decrease from GM3 to GM5 (Figure 3.2).

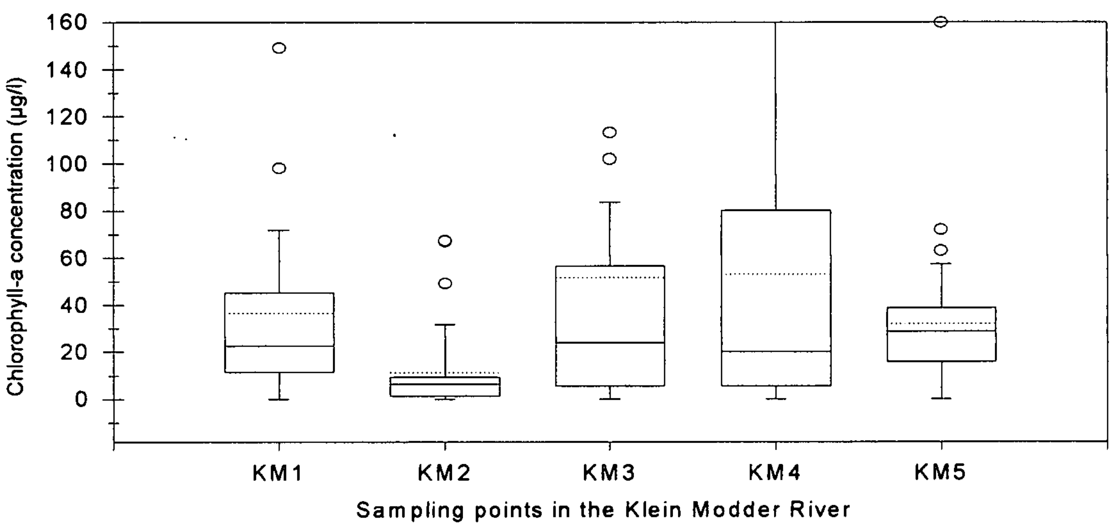


Figure 3.1: Variation in chlorophyll-*a* concentration downstream in the Klein Modder River during the study period. For explanation see Figure 2.5.

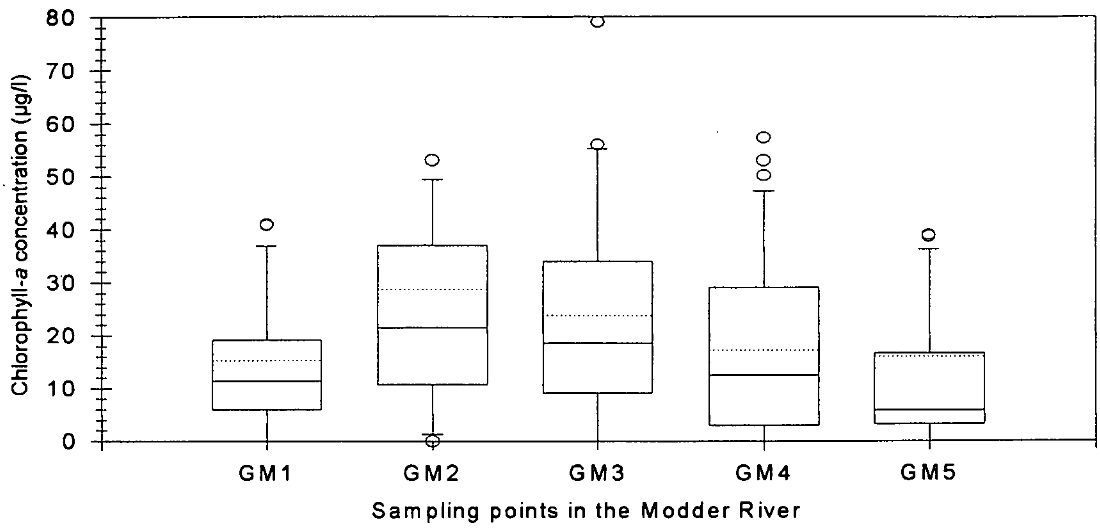


Figure 3.2: Variation in chlorophyll-*a* concentration downstream in the Modder River during the study period. For explanation see Figure 2.5.

The algal biomass in the sewage discharge (KM2) was low in comparison to the other sampling points during the study period (Figures 3.1 & 3.3). At the other sites in the Klein Modder River, the chl-*a* concentration showed comparable patterns with the Modder River (Figure 3.4). In the Klein Modder River chl-*a* peaks were observed during March and September 1996, and during March and July 1997 (Figure 3.3). No consistent seasonal pattern between the sampling sites were observed. The lowest chl-*a* concentrations did occur during the dry winter months, though. At all the sampling points in th Klein Modder River (except at KM2), the chlorophyll was high, but at KM5 it decreased.

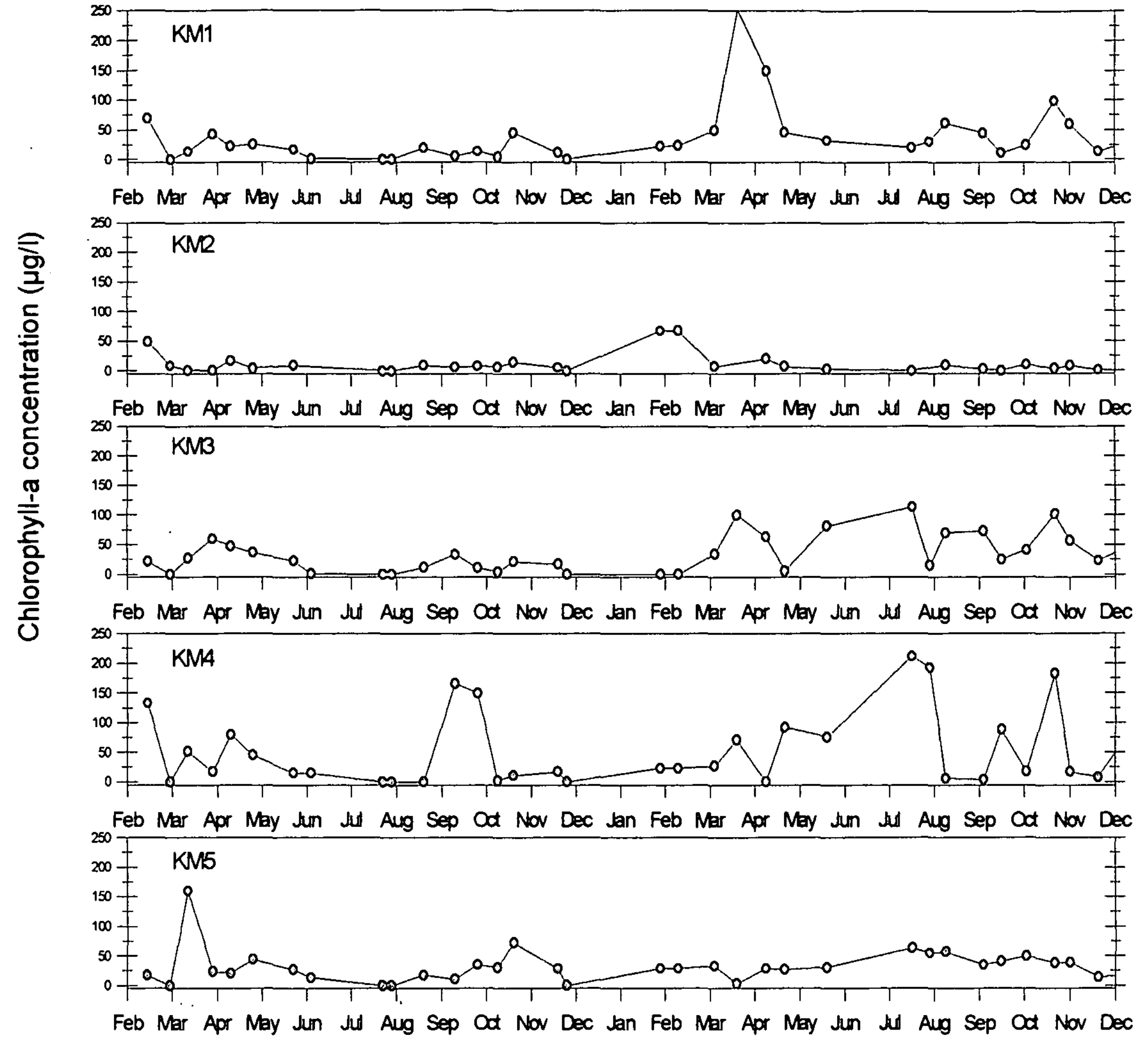


Figure 3.3: Seasonal variation in chlorophyll-*a* concentration at the different sampling points in the Klein Modder River

In the Modder River the chl-*a* at all of the sites varied throughout the study period (Figure 3.4). There were chl-*a* peaks at all the sites in August and November 1996, as well as in March, April and September 1997. The maximum chl-*a* concentrations were observed at GM2 (201 µg/l) during April 1996 and GM5 (180 µg/l) during September 1997.

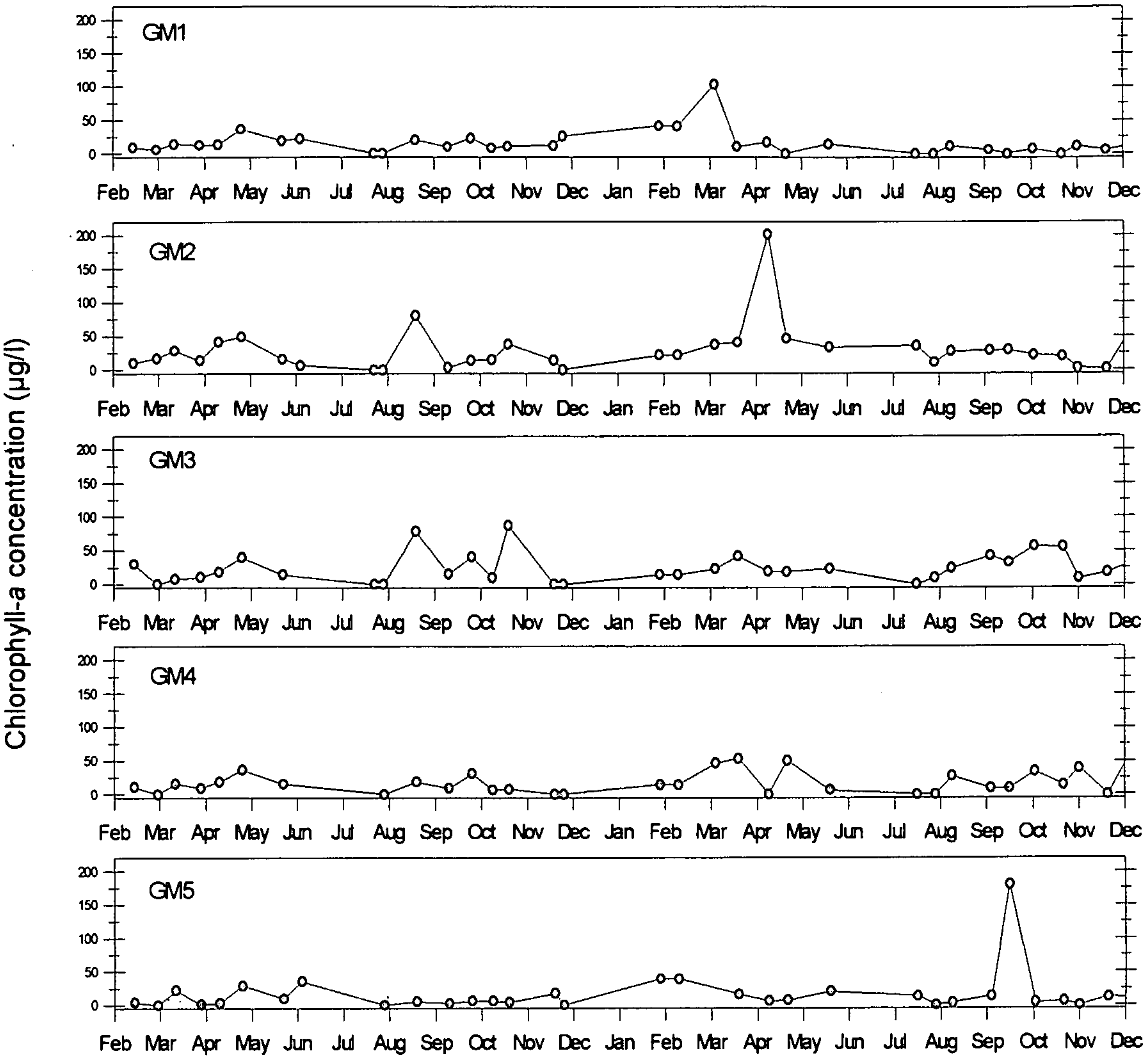


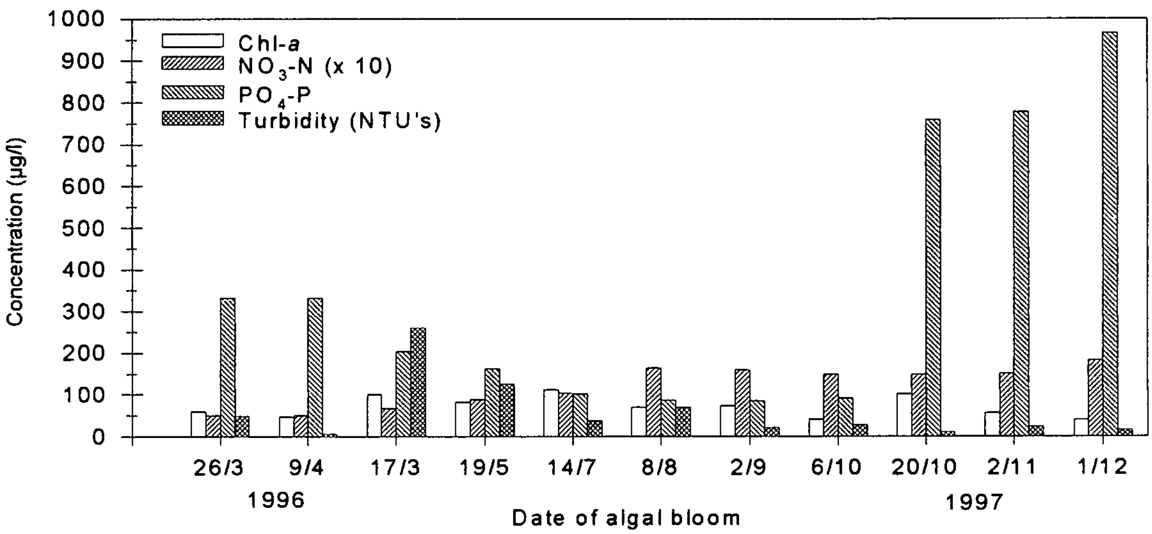
Figure 3.4: Seasonal variation in chlorophyll-*a* at the different sampling points in the Modder River

The South African Water Quality Guidelines for Recreational Use (DWAF, 1993a) states that chlorophyll-*a* concentrations above 30 µg/l, for free-floating algae, cause severe nuisance algal blooms, thus chlorophyll-*a* concentrations higher than 40 µg/l were considered as algal blooms. The majority of algal blooms in both rivers occurred when nutrient concentrations (NO_3-N and PO_4-P) were high and turbidity was low. However, some of the blooms coincided with higher turbidities. Very few algal blooms occurred in the Modder River, compared to the Klein Modder River. The chl-*a* concentrations of these blooms in the Modder River were also much lower than that of the Klein Modder

River. In the Klein Modder River, most blooms occurred at KM3 and KM4 (Figures 3.5a & 3.5b). These points are below the sewage inflow of the Botshabelo sewage works. In the Modder River the most algal blooms occurred at GM2 and GM3, the sampling sites just below the confluence with the Klein Modder River (Figures 3.6a & 3.6b).

NO_3-N and PO_4-P were calculated as the mean concentration of these nutrients two to three months preceding the bloom. Turbidity was calculated as the mean turbidity as determined two weeks to a month preceding the bloom.

3.5a) KM3



3.5b) KM4

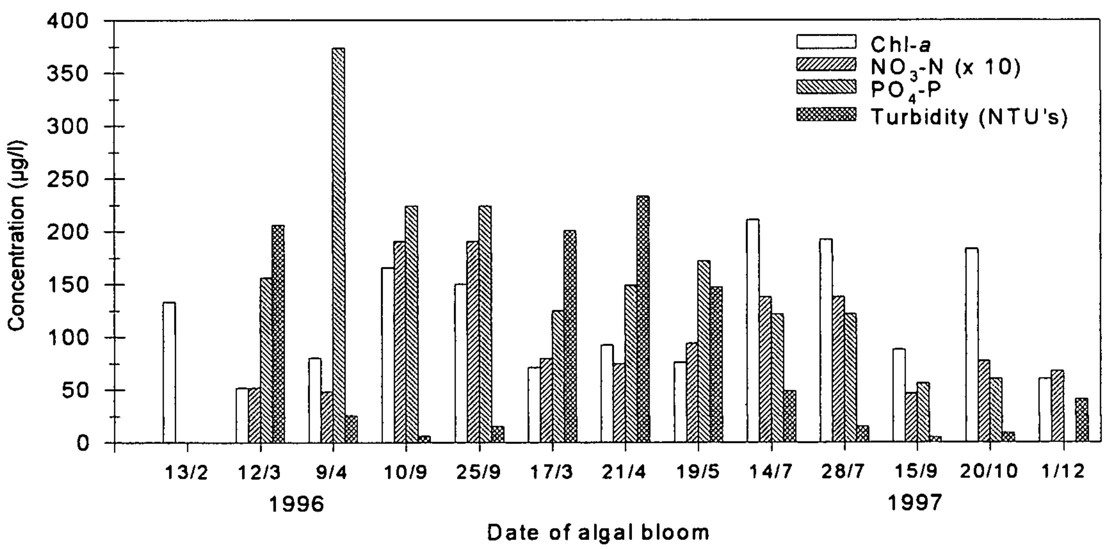
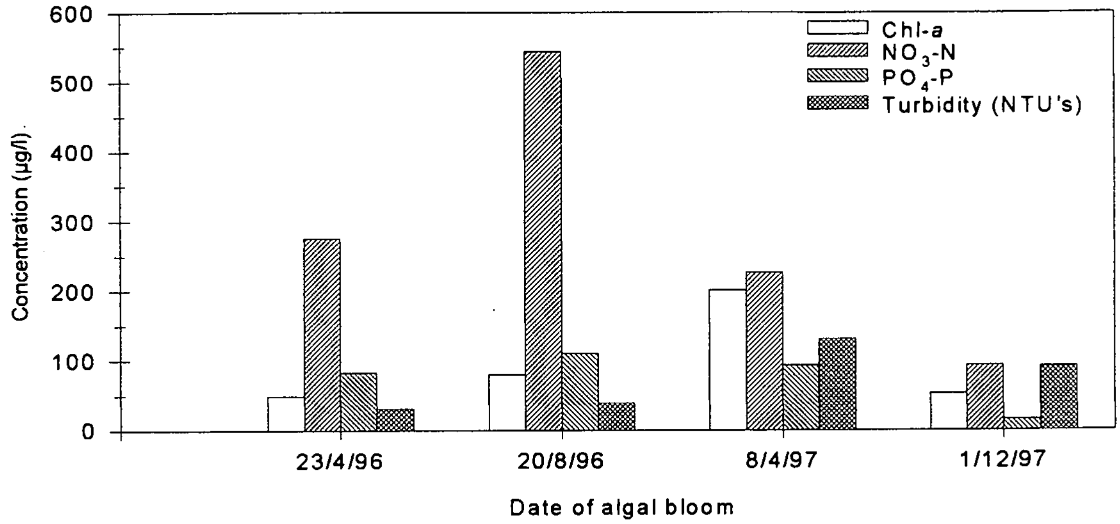


Figure 3.5: Sites in the Klein Modder River with the most frequent algal blooms and conditions preceding the blooms.

3.6a) GM2



3.6b) GM3

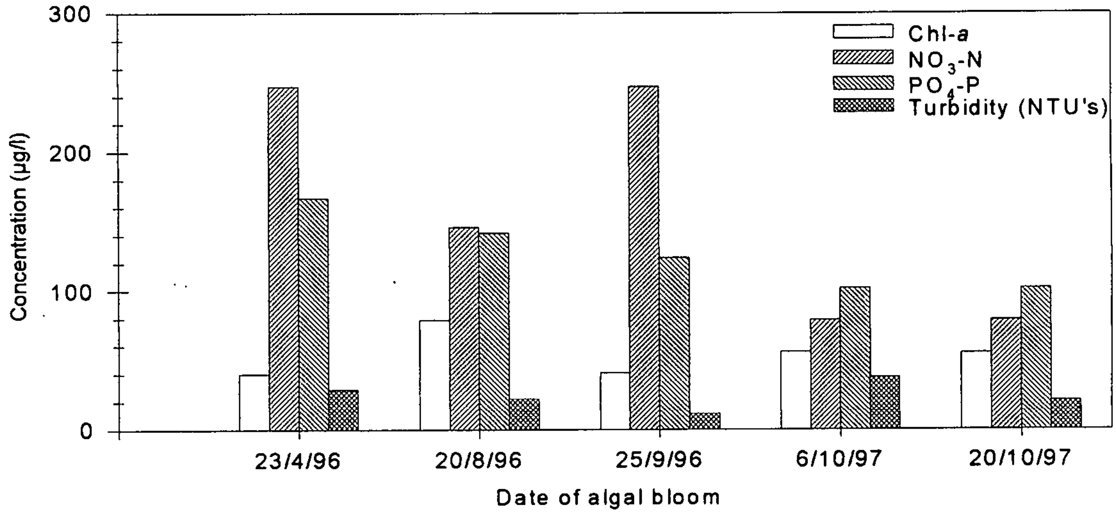


Figure 3.6: <u>The sites in the Modder River with the most frequent algal blooms and the conditions preceding the blooms.</u>

The above-mentioned data thus clearly show that the most algal blooms occurred when nutrient concentrations were high and turbidity was relatively low.

3.4.2) Algal species identified

The algal species identified most often, in both the rivers, were (in alphabetical order):

Aphanocapsa sp.

Centric diatoms (*Cyclotella* sp., *Stephanodiscus* sp.)

Chlamydomonas sp.

Chlorococcum sp.

Euglena sp.

Melosira sp.

Pennate diatoms (*Navicula* sp., *Nitzschia* sp., *Pinnularia* sp.)

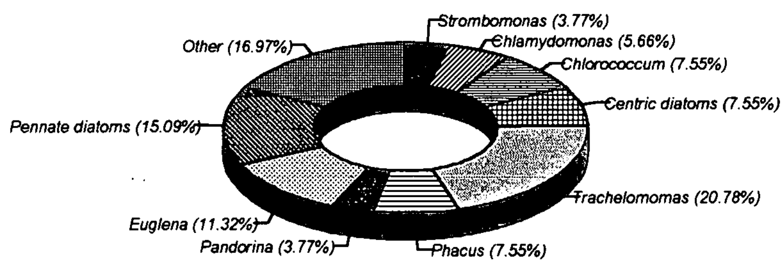
Phacus sp.

Strombomonas sp.

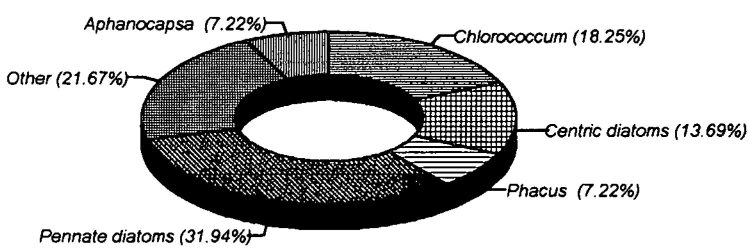
Trachelomonas sp.

Figures 3.7 and 3.8 show the dominant species at the different sampling sites in the Klein Modder and Modder Rivers. Diatoms constituted a large portion of the algal populations at all of the sampling points.

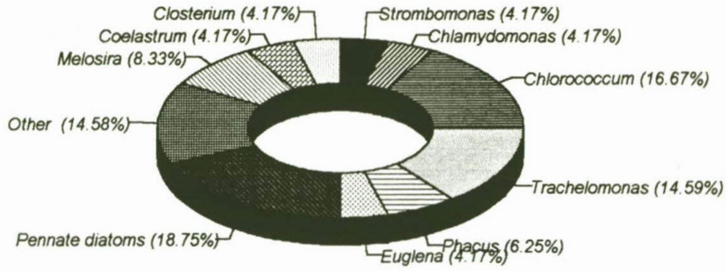
3.7a) KM1:



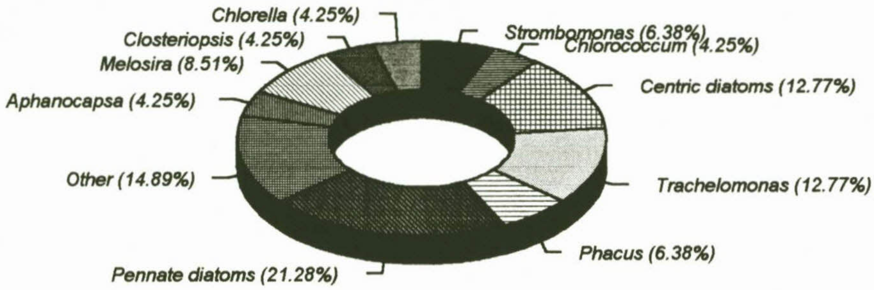
3.7b) KM2



3.7c) KM3:



3.7d) KM4:



3.7e) KM5

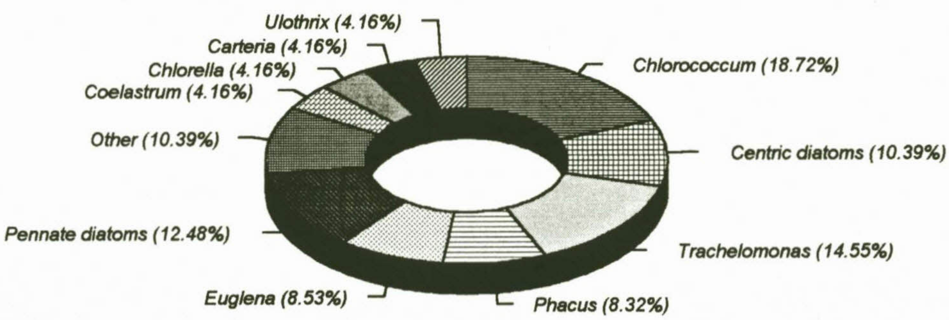
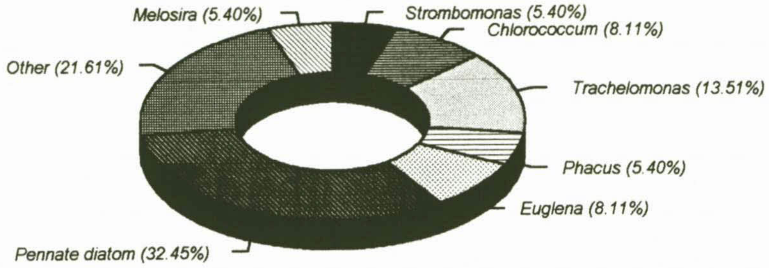
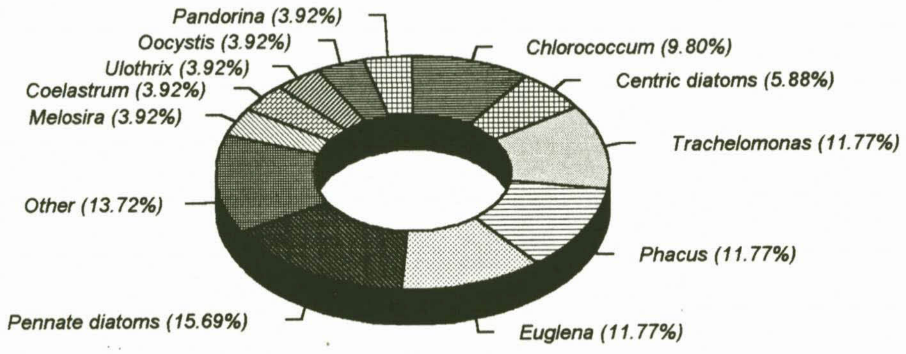


Figure 3.7: Algal species composition of the different sampling points in the Klein Modder River

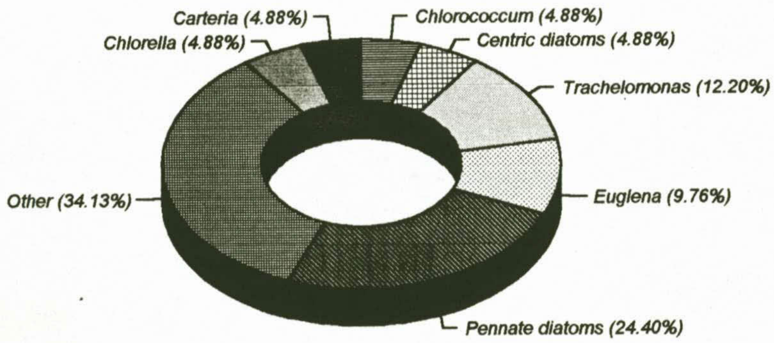
3.8a) GM1:



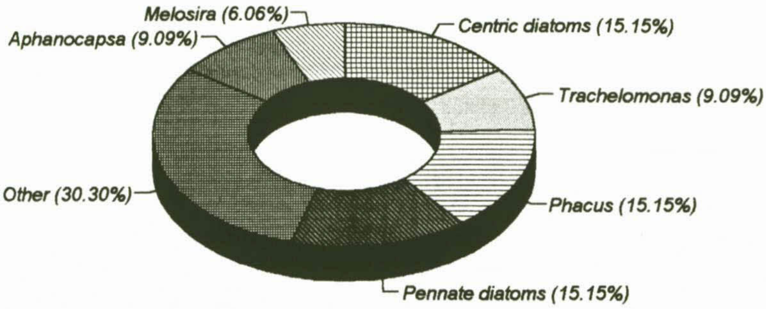
3.8b) GM2:



3.8c) GM3:



3.8d) GM4:



3.8e) GM5:

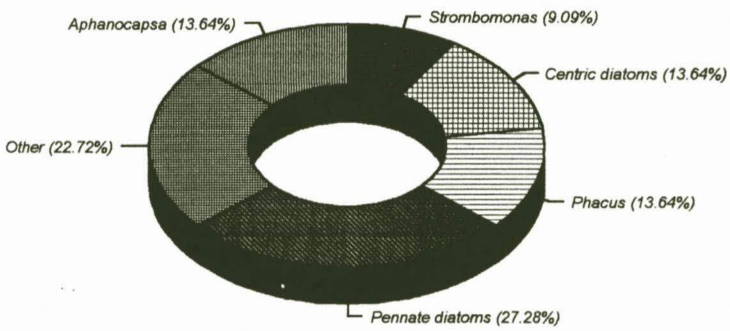


Figure 3.8: Algal species composition of the different sampling points in the Modder River

In the Klein Modder and Modder Rivers the algal species varied from season to season and there was always a diversity of species present. At KM3, KM4 and KM5 there were a greater diversity of algal species than at the other sampling points in the Klein Modder River. In the Modder River, the greatest algal diversity was found at GM2, after the confluence with the Klein Modder River. No distinct seasonal succession was observed, but the following species were found to dominate during the different seasons (Table 3.1):

Table 3.1: Seasonal domination of algal species in the Modder River system

Season	Algal species
Autumn	*Chlorococcum* sp., *Phacus* sp., *Euglena* sp., pennate diatoms
Winter	Low chl-*a* concentrations. Single pennate diatoms and *Trachelomonas* sp.
Spring	*Euglena* sp., *Trachelomonas* sp., *Phacus* sp., pennate diatoms
Summer	Pennate diatoms, *Trachelomonas* sp.

3.4.3) Relationship between chlorophyll-*a* and other variables

3.4.3 a) Chlorophyll-*a* and TP

A linear relationship between $\log_{10}$TP and $\log_{10}$chl-*a* was found in the Modder River system ($r = 0.961$, $p < 0.001$) (Figure 3.9a). The average chl-*a* and the average TP concentrations of the two different years of the study period at the five sampling points in the Modder River and four sampling points in the Klein Modder River (without KM2) were used for this regression. The equation obtained was:

$$\log_{10}[\text{chl-}a] = 0.52736441(\log_{10}[\text{TP}]) + 0.10807407$$

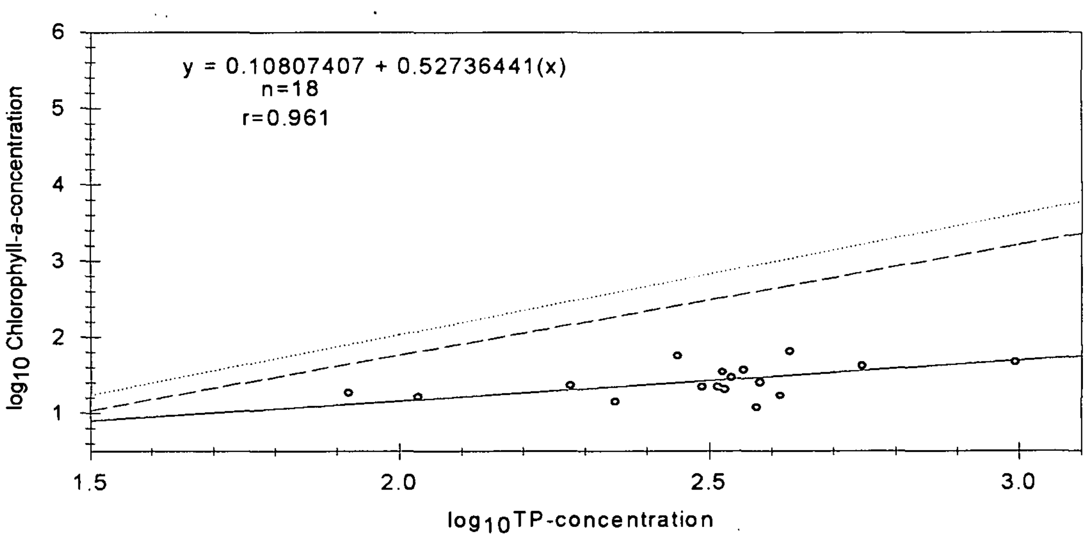


Figure 3.9a: The linear log relationship between chlorophyll-a concentration and TP in the Modder River system (solid line) The dotted line represents the relationship found by Sakamoto (1966) and the broken line the relationship found by Dillon & Rigler (1974)

When the N:P ratio in the Modder River (calculated as 6.6) is taken into account and plotted with the equation found by Smith (1982), (constituting the particular phosphorus concentration [PP] with total phosphorus concentration for the Modder River) the following graph is obtained (Figure 3.9b):

$\log_{10}$[chl-a] = 0.986(log10[TP]) + 0.0202 (TN:TP)
and TN:TP = 6.6 in the Modder River

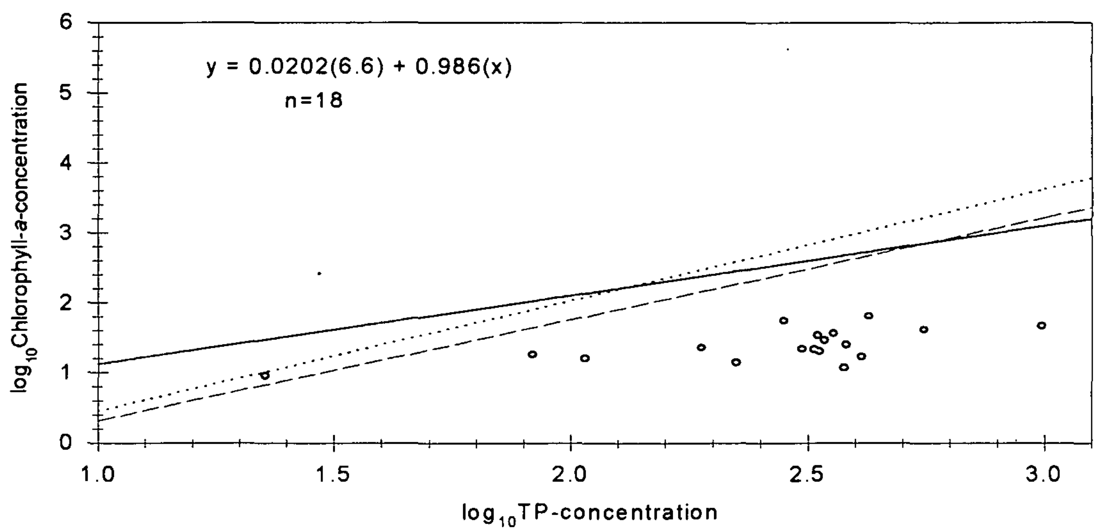


Figure 3.9b: The linear log relationship between TP and chlorophyll-a in the Modder River, taken N:P ratio (6.6) into account (solid line). The dotted line represents the relationship found by Sakamoto (1966) and the broken line the relationship found by Dillon & Rigler (1974) with N:P ratios. The real data are given as dots.

3.4.3 b) Chlorophyll-*a* and SiO_2-Si

An inverse relationship between SiO_2-Si and chlorophyll-*a* was found in the Modder River ($r = 0.927$, $p < 0.001$) (Figure 3.10). The average SiO_2-Si concentration at all the sampling sites in the Modder River for each of the two years of the study period was plotted against the average chlorophyll-*a* concentration of the sampling sites for the two years.

$$[\text{Chl-}a] = 1/(-6.7884 + 6.8196\ [\text{Si-SiO}_2]\ x^{0.00354})$$

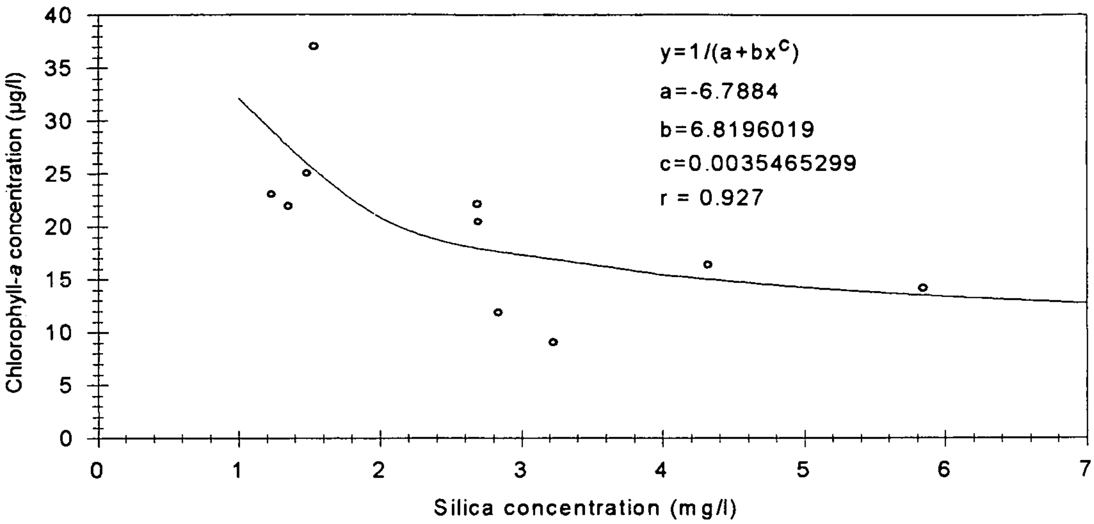


Figure 3.10: The inverse relationship between silica concentration and chlorophyll-*a* concentration in the Modder River

3.4.4) Biochemical oxygen demand (BOD)

The BOD in the Modder River increased from GM1 to GM2 with 2.2 mg/l, after the confluence with the Klein Modder River, but the BOD at GM5 (3.7 mg/l) was almost the same as at GM1 (3.2 mg/l) (Figure 3.11).

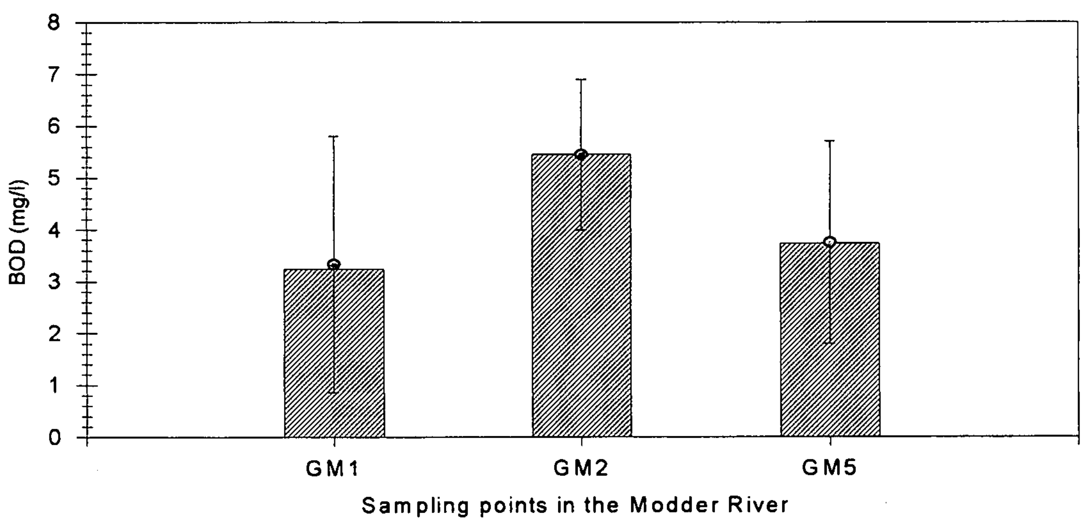


Figure 3.11: The average BOD at some of the sampling points in the Modder River

3.5) DISCUSSION

3.5.1) Spatial and seasonal variation in chlorophyll-*a*

The river environment is always more or less disturbed, and never experiences periods of stability as occur in lakes and the changes in community structure and diversity are clearly connected with the amplitude of weather and discharge fluctuations (Hynes, 1970; Descy, 1993). This is illustrated in both the Modder and Klein Modder Rivers where high flow was mostly associated with low chlorophyll-*a* concentrations, probably due to wash-out, dilution and higher turbidities. However, Grobler & Davies (1981) reported that sediments suspended in the water column, in turbid waters, might act as a direct source of phosphate to algae, whereas in shallow waters there is the possibility that the bottom sediments can be resuspended in the water column to act as potential phosphate sources to algae. This can explain the fact that high chlorophyll-*a* concentrations sometimes occurred despite high turbidities.

In the Klein Modder River more algal blooms occurred and at higher concentrations in 1997 than in 1996 (Figure 3.3). In the Modder River at GM2 (chl-*a* = 201 µg/l) and GM5 (chl-*a* = 180 µg/l) algal blooms with very high maxima occurred in 1997 (Figure 3.4). The fact that more algal blooms occurred in 1997 and that chl-*a* concentrations, in both rivers, were higher despite higher turbidities, can be explained by the low flow conditions experienced in the rivers during drier periods. The total rainfall for 1997 was 370.2 mm and for 1996, 642.8 mm. Thus, during low flow conditions, the time the algae spend at a specific site in the river increases, since flow does not disturb the river environment to the extent that periods of flooding do. It is also possible that, during low flow conditions, light limitation is lifted by turbulence (the mixing depth becomes greater than the euphotic zone, resulting in an increase in photosynthetic activity) or by the fact that light can penetrate to the bottom of the river, as was the case at some of the sampling sites.

The predicted average chl-*a* concentration for Mockes Dam, by Grobler & Toerien (1986), was 38 µg/l in 1990 and 46 µg/l for the year 2000. The average value during the study period at GM3 (above Mockes Dam) was 24 µg/l. At GM2 (just after the confluence of the two rivers) the average chl-*a* concentration was 28 µg/l, higher than at GM3 (Figure 3.2). It seems thus that algal growth occurred in the river itself and not, as expected, only in the impoundments.

The higher mean chl-*a* concentration in the Klein Modder River, than in the Modder River, can be ascribed to higher nutrient (NO_3-N, PO_4-P and SiO_2-Si) concentrations creating favourable conditions for algal growth, as well as the availability of sufficient light, and low-flow conditions during the drier periods. Of the dissolved substances that may possibly influence algal diversity in the Vaal River, inorganic N, P and Si also seem to be of particular importance (Pieterse & Van Zyl, 1988). Inorganic N and P could limit algal growth and Si availability could control diatom growth. Diatom, chlorophytes and total phytoplankton concentrations were positively correlated during 1984 in the Vaal River, South Africa, with inorganic N/P ratios (Pieterse & Van Zyl, 1988). Chlorophyte concentration was similarly correlated with monthly inorganic N loading (Pieterse & Van Zyl, 1988).

Although the PO_4-P concentration in the Modder River was sometimes low before the occurrence of algal blooms, it must be taken into consideration that algae are capable of utilising other forms of soluble P than PO_4-P. PO_4-P (available P) only constitutes a dissolved fraction, (in the Modder River 38% and in the Klein Modder River 21%), of the TP and is available to algae. However, many investigations showed that algal phosphatase that are bound to algal membranes, increase in activity when phosphorus becomes deficient (Wetzel, 1983). Thus, to determine the fertility of fresh waters, TP must be taken into account, because inorganic phosphorus is rapidly utilised and can be stored, in excess of immediate needs.

NO_3-N concentration in both rivers was high (> 70 µg/l) before algal blooms occurred. In the River Spree, Germany, concentrations of dissolved reactive phosphorus > 10 µg/l, dissolved inorganic nitrogen > 2300 µg/l and dissolved reactive silicon > 220 µg/l were always high in the catchment where algal blooms occurred (Köhler, 1994). Phytoplankton assimilation caused declining concentrations of dissolved inorganic phosphorus and nitrogen along the river course (Köhler, 1994).

Low temperatures (10 -15°C) during the winter could be a limiting factor for algal growth, despite the relatively high light availability (low turbidity). Increased algal growth also occurred when temperatures became more favourable e.g. during the increase in spring.

The increase in chl-*a* at GM2 and GM3, when compared to GM1, can be ascribed to the nutrient-rich water that flows from the Klein Modder River into the Modder River, providing, together with high light availability and low flow, favourable conditions for algal growth. The fact that the highest chl-*a* concentration, during the study period occurred at

GM2 in March 1997, gives an indication of the degree to which the Klein Modder River enriches the Modder River with nutrients.

There was a gradual decrease in chl-*a* from GM3 to GM5, probably due to dilution of the nutrients. Rapid water renewal had significant and similar negative impacts on both the abundance and predictability of suspended algae in all systems investigated in the USA (Søballe & Kimmel, 1987). Thus, in rivers and rapidly flushed impoundments, algal abundance often depends more on the variation in physical characteristics (such as turbidity) than on nutrient concentrations. In the Modder and Klein Modder Rivers, physical characteristics (like turbidity) as well as nutrient availability plays an important role in the limitation of algal growth.

GM5 showed a totally different seasonal pattern from than any of the other points in the Modder River. This could be due to the fact that this point is the Mazelspoort Dam which is lentic with a deep water-column. Thus, it creates a totally different environment e.g. deep water column and mixing, than a fast-flowing, turbid river. During September 1997, only one algal bloom occurred at GM5.

3.5.2) Algal species identified

Africa has special characteristics concerning the seasonality of phytoplankton (Talling, 1986). Unlike other continents it extends almost equally to northern (37°) and southern (35°) latitudes. Although the non-equatorial areas are often arid, they do include a variety of inland waters in which seasonality is pronounced for hydrological or other climatic reasons. This is also the case in the Modder River, where the algal communities reacted to climatic as well as hydrological changes.

Some of the species found to be dominant in the Modder River, pose problems to water purification, while others act as indicators of water pollution. Since the Modder River is a source of potable water for the city of Bloemfontein, these undesirable characteristics can interfere with the process of water purification. These characteristics of the dominant species in the Modder River are listed in Table 3.2.

At the sewage outflow, there was the least diversity of species, with pennate diatoms (some of these species are associated with water rich in organic matter) dominating. Algal growth was probably limited by the presence of chlorine. Species diversity increased downstream in the Klein Modder River, with the greatest diversity at KM3, KM4 and KM5. In the Modder River species diversity was relatively high at all the sampling sites except at GM5, Mazelspoort Barrage. However, species diversity was the highest at GM2 (after the confluence with the Klein Modder River), where an increase in nutrients was observed (Chapter 2).

The algal species that dominated most often during the algal blooms were pennate diatoms spesies. Sometimes *Trachelomonas* sp. and *Chlamydomonas* sp. were also dominant.

Table 3.2: Some characteristics and problems caused of the dominant species found in the Klein Modder and Modder River (Carter-Lund & Lund, 1995)

Species	Characteristics
Strombomomas sp.	- Common in fresh waters
Euglena sp.	- Commonest in water rich in organic matter - Occur in waters with low dissolved oxygen
Melosira sp.	- Filter- and screen-clogging algae
Aphanocapsa sp.	- Common in fresh waters
Navicula sp.	- Filter- and screen-clogging algae - Live on a surface, not attached
Nitzschia sp.	- Fresh water pollution - Second largest genus of freshwater algae
Pinnularia sp.	- Clean water - Live on mud and other substrata
Cyclotella sp.	- Common, mainly planktonic - Filter- and screen-clogging algae
Stephanodiscus sp.	- Common in fresh waters
Chlorococcum sp.	- Widespread in aquatic habitats and in soil
Trachelomonas sp.	- Commonest in water rich in organic matter - Filter- and screen-clogging algae - Occur in waters low in dissolved oxygen
Phacus sp.	- Commonest in water rich in organic matter - Fresh-water pollution - Occur in water with low dissolved oxygen

In rivers, diatoms are the most abundant and species-rich primary producers, occurring in all habitats from source to mouth (Jüttner *et al.*, 1996). During spring and autumn phytoplankton of large rivers are generally dominated by diatoms (*Asterionella* and *Stephanodiscus* sp.), while green algae and cryptomonads may develop best in summer conditions (Descy, 1993). Eventually, at very low discharges, typical succession leading to cyanobacterial dominance may be observed (Descy, 1993). During all seasons, most of the sampling sites, in both the Klein Modder and Modder Rivers, phytoplankton communities were dominated by pennate and centric diatoms (Bacillariophyceae). The algal blooms that occurred, were also mostly dominated by diatoms. *Trachelomonas* was the other species which were often present. Euglenophyceae, Chlorophyceae and, during one occasion, Cyanophyceae, were also present. Algal blooms, in the Modder and Klein Modder Rivers, also occurred during times when temperatures were more favourable. This is all in accordance with the Vaal River, South Africa, where the phytoplankton community was represented by seven algal groups: Dinophyceae, Euglenophyceae, Cyanophyceae, Bacillariophyceae, Chlorophyceae and Cryptophyceae (Pieterse, 1987). Chrysophytes occurred only in September and October in the Vaal River. Numerically, diatoms were also dominant in the Vaal River in April and August (autumn and spring) while green algae were dominant during the other months.

The highest individual numbers were reached by the following species in the Vaal River (Pieterse, 1987): *Trachelomonas* (March), *Cyclotella* (April, May), *Ankistrodesmus* (June, July), *Stephanodiscus* (August), *Micratinium* (September) and *Chlamydomonas* (October). The greatest species diversity was demonstrated during the warmer months. The least diversity occurred during August and September, two months during which phytoplankton concentration was the highest and similarity between the community structure was the greatest.

It can thus be concluded that algal diversity as well as seasonal patterns vary from ecosystem to ecosystem. However, the algal families most often found in South African waters appear to be Bacillariophyceae, Chlorophyceae and Cyanophyceae.

Although high chl-*a* concentrations occurred, at regular intervals, in both the Klein Modder and the Modder Rivers, no cyanobacterial blooms were found. This can be explained by different factors, as stated by Steinberg and Hartmann (1988). They state that above a threshold of 10 µg/l TP, the development of cyanobacteria can be described by physical factors, such as water column stability. When the turbulence of the water column is rather low, as it is in sheltered or meromictic lakes, cyanobacteria can build up dense populations. If the turbulence of the water column is high (mixing depth much greater than euphotic depth) or the mixing pattern is irregular, as in the case of the Klein Modder and Modder Rivers, cyanobacteria are outcompeted. Cyanobacterial blooms also occur more readily at elevated water temperatures, high pH values and low N:P ratios. The N:P ratio in the Modder River (6.6) was low, and water temperatures were sometimes high, but the turbulence of its waters probably prevented the growth of cyanobacteria. It is thus possible that blue-green algae will develop in stable environments.

3.5.3) Relationship between chlorophyll-*a* and other variables

3.5.3 a) Chlorophyll-*a* and TP

Chlorophyll-*a* vs. TP (log-scale) gave a linear relationship in the Modder River. This is confirmed by the linear relationship found between these variables for some Japanese lakes (Sakamoto, 1966) as well as for 19 lakes in Ontario (Dillon & Rigler, 1974). The equation derived by Sakamoto (1966), was (without the two lowest N:P ratios):

$$\log_{10}[\text{chl-a}] = 1.583\log_{10}[P] - 1.134$$

The equation derived by Dillon and Rigler (1974) to demonstrate the relationship between chl-*a* and TP, is:

$$\log_{10}[\text{chl-}a] = 1.449 \log_{10}[P] - 1.136$$

When comparing the linear relationship between $\log_{10}$chl-*a* and $\log_{10}$TP found in the Modder River with the relationships found by Sakamoto (1966) and Dillon & Rigler(1974), it is observed that the data of the Modder River lies far beneath the other. Thus, at similar concentrations of TP, Sakamoto's and Dillon & Rigler's equations predict chlorophyll-*a* concentrations up to three times higher than those found in the Modder River system. These lower chl-*a* concentrations per unit TP could be ascribed to the high turbidity of the system. Although the TP concentration is high, the efficiency of algae is probably limited by low light availability. Another factor that could explain the lower chlorophyll-*a* concentrations, is the fact that the algal communities in the Modder and Klein Modder Rivers were mostly dominated by diatoms. Since diatoms contain less chlorophyll-*a* per cell volume than other algae, the prediction of the total chlorophyll-*a* in terms of TP could be an underestimation of the real algal biomass.

Sakamoto (1966) noted that the chl-*a* yield in Japanese lakes was a logarithmic function of both TP and TN. He concluded that over the range 10<TN:TP<17 by weight, chl-*a* yield was very nearly balanced with respect to both TP and TN, but that chlorophyll was dependent only on TN when TN:TP<10, and only on TP when the TN:TP>17. Variability in the TN:TP ratio, thus, may provide a major source of variability in TP:chl-*a* relationships. When the N:P ratio drops, and nitrogen becomes more limiting, the concentration of dissolved phosphorus (PO_4-P) should increase and represent a greater proportion of the TP (Smith, 1982). As a result, TP would no longer represent the mean particulate phosphorus concentration and a regression of the mean chl-*a* concentration on TP would become variable.

The equation by Smith (1982), taking TN:TP into account:

$$\log_{10}[\text{chl-a}] = 0.986\log_{10}PP + 0.0202\ TN{:}TP$$

The equation derived for the Modder River, constituting PP with TP (N:P = 6.6)

$$\log_{10}[\text{chl-}a] = 0.986\ \log_{10}TP + 0.133$$

The N (dissolved inorganic nitrogen):P (dissolved inorganic phosphorus) ratio in the Modder River, for the study period, was 6.6 (<10), indicating that algal growth is dependent on N. When the relationship between $\log_{10}$[chl-*a*] and $\log_{10}$[TP] for the Modder River was plotted with the N:P ratio taken into account, the line obtained was much nearer to that given by Sakamoto (1966) and Dillon & Rigler (1974). It is thus important in systems where N is the limiting factor, to take N:P ratios into account for TP:chl-*a* relationships.

Three impoundments in the Modder River, Mockes, Mazelspoort and Rustfontein Dam were investigated by Toerien and co-workers in 1975. In Mockes Dam (N:P ratio - 1.4) and Mazelspoort (N:P ratio - 2.7) nitrogen was found to be the primary limiting nutrient and in Rustfontein Dam phosphorus (N:P ratio not determined) was the limiting nutrient.

3.5.3 b) Chlorophyll-*a* and SiO_2-Si

Since the dominant algal species in the Modder River was mainly diatoms, silica constitutes an important part of the nutrient balance in this river. The inverse relationship between SiO_2-Si and chl-*a*, in the Modder River, can be ascribed to the uptake of SiO_2-Si by the diatoms to built their cellwalls. This inverse relationship between silica and diatoms species was also reported by Kilham (1971). Diatom growth is common in North America, Europe, South America and Africa in waters with high silica concentrations. Different diatoms, however, require different concentrations of silica, i.e. *Melosira granulata* was consistently found in water with high silica concentrations, while *Stephanodiscus astraca* occurred in waters with silica concentrations of less than 1 mg/l (much lower than the silica concentrations in the Klein Modder and Modder Rivers). Kilham (1971) concluded that it is possible that the occurrence of freshwater eutrophic diatoms is principally determined by the silica concentration of the water.

3.5.4) Biochemical oxygen demand

The biochemical oxygen demand (BOD) is the utilisation of dissolved oxygen by aquatic microbes to metabolise organic matter, oxidise reduced nitrogen, and oxidise reduced mineral species such as ferrous iron (Bowie *et al.*, 1985). Concentrations of reduced minerals in streams receiving waste water are usually variable, and thus BOD is commonly divided into two fractions: that exerted by carbonaceous matter (CBOD) and that exerted by nitrogenous matter (NBOD). The increase in BOD at GM2 is an indication of the influence that the sewage outflow of Botshabelo into the Klein Modder River has on the Modder River after their confluence. Thus, more dissolved oxygen was utilised by the aquatic organisms to break down organic and inorganic matter that increased due to the sewage effluent. However, at GM5 the mean BOD was almost the same as at GM1 (reference point). The Modder River had a self-purification effect, and downstream dilution occurred to restore the water to its original quality in terms of BOD.

3.6) CONCLUSION

Algal growth is a function of temperature, light and nutrients. The algal blooms that occurred in both the Modder and the Klein Modder Rivers could be attributed, amongst other factors, to high nutrient concentrations at some of the sampling sites during the spring and summer period. The chl-*a* concentration sometimes reached typically eutrophic levels in the Klein Modder River (> 100 µg/l) and could cause aesthetic or environmental health problems.

However, the algal growth in the Modder and Klein Modder Rivers is probably influenced by more than just the high nutrient concentrations, because the Modder River is a system limited by light rather than by nutrients (Grobbelaar, 1985) during the rainy seasons of the year. Turbid systems often have high potential trophic states due to high nutrient availability but their carrying capacity is not reached because of strong light limitation (Dokulil, 1994). The decrease in turbidity, during the dry season, could cause the light penetration to increase and cause more favourable light conditions for photosynthesis which stimulates primary productivity and leads to algal blooms (Toerien *et al.,* 1983, Roos & Pieterse, 1995b). Turbulence in the shallow waters of the Modder River is also an important factor, because it can lift light limitation by creating a mixing depth greater than the euphotic depth.

The N:P ratio is also an important factor when calculating the chl-*a* yield. The calculated chl-*a* yield was much lower for the Modder River than for other systems, when the N:P ratio was not taken into account. The turbidity of the Modder River also has a limiting effect on the chl-*a* yield of the system.

It can be seen, from the results, that temperature, nutrient concentrations and turbidity (all of which that are caused by seasonal changes) are the most important interacting factors controlling algal growth in the Klein Modder and Modder Rivers. This is confirmed in the Barwon-Darling River, Australia, where warm water temperatures, elevated pH, reduced turbidity and improved water transparency contributed to increased algal growth during spring (Bowling & Baker, 1996).

Although the measured TP reached the predicted values (Chapter 2), chl-*a* concentrations were lower than predicted. It is important to note that higher chl-*a* values were measured upstream from Mockes Dam than downstream. Thus, eutrophication of the system occurs not only in the reservoirs, but also in the river itself. Given the hot, dry summer climate, the potential for further drought and low flows, and the high nutrient concentrations of its waters, future blooms in the Modder river system are still highly probable.

CHAPTER 4

MICROBIAL QUALITY OF THE MODDER RIVER AND POSSIBLE TOXICITY OF ITS WATER

I) DETERMINATION OF TOXIC COMPOUNDS BY USE OF *SELENASTRUM CAPRICORNUTUM* AND *DAPHNIA PULEX*

4.1) INTRODUCTION

Many new chemicals are synthesised or identified each year and enter the commercial use in quantities large enough to be of environmental concern for humans and animals (Giesy and Graney, 1989). There are two main needs for measuring the impact of these chemicals on an ecosystem. First, to anticipate the impact and secondly, to assess the changes that take place when chemicals are released into an ecosystem (Calow, 1993).

If a toxic substance enters a water ecosystem, it can adversely affects the biota in the system, as well as the living organisms which are dependent on the water for life. It must be kept in mind, however, that the properties of an effluent start to change as soon as it mixes with receiving water. Not only the physico-chemical characteristics of the environment but also the chemical form of the pollutant and the physiological state of the biota affect toxicity (Babich & Stotzky, 1980). Furthermore, toxicity caused by contaminants in the effluent are only part of the many influences that determine the health of a biological community. Influences from substrate differences and physical conditions, such as dissolved oxygen, temperature and availability of suitable habitat, can also adversely affect the biotic community (Roux, 1994).

At present, regulation and control of effluent quality is carried out primarily by chemical analyses of water samples. However, a chemical specific approach alone to assess toxicity has the following limitations (Roux, 1994): environmental factors influence toxicity, toxic compounds are sometimes a complex blend of organic chemicals, transformations may occur and chemicals may produce toxicological effects at concentrations not detectable by chemical analyses. Toxicity testing can bridge this gap and play a significant role in improving water quality in the years to come, especially through its application in effluent regulation.

One of the ways to assess toxicity of effluents into river systems, is by means of bioassays. Various aquatic organisms respond differently to hazardous chemicals that are deposited into a river or stream. A variety of organisms are currently being used to test the toxicity of effluent and wastewater, ranging from algae to copepods, fish and daphnias. For example, *Chlorella* was less sensitive than fish and *Daphnia* to a variety of chemicals (Lewis, 1990). In contrast, algae and duckweed have been found to be more sensitive to several detergent surfactants, textile effluents, acridine, dyes, synfuels and a variety of other compounds than are invertebrates and fish.

4.2) TOXICITY TESTING BY *in vivo* FLUORESCENCE OF *SELENASTRUM CAPRICORNUTUM*

Algal toxicity tests are rapid, inexpensive, and sensitive, and can be used effectively to assess those toxic substances which are found in concentrations too low for effective detection by higher trophic levels organisms (Munawar *et al.*, 1989). Toxicity tests using algae as test organisms provide an important method of assessing the effects of elements in solution on biological systems (Van der Heever & Grobbelaar, 1996). Since algae are at the base of most aquatic food webs, any factor having a detrimental effect on algae also affects the rest of the community.

There are some distinct characteristics required from an algae used in bioassays. For instance, it should have a broad nutrient response; a distinct shape and an uniform size. It also should divide distinctly, should not attach to glass or surfaces, it should stay in suspension and should normally be associated with oligotrophic waters (Porcella *et al*, 1970). *Selenastrum capricornutum* meets all these requirements. The distinct morphology, limited variation in form, and the fact that cells generally occur singularly, stay in free suspension and are obligate autotrophic make this algae an ideal test organism.

There is a strong correlation between light-saturated photosynthetic rates and a relatively simple measure of photochemical capacity, the increase in *in vivo* fluorescence. All photosynthetic algae, including cyanobacteria, contain chlorophyll-*a*, which is a blue-green water insoluble pigment. When a solubilised chlorophyll-*a* pigment is exposed to a blue light the solution fluoresces red (Falkowski & Kiefer, 1985). The absolute quantum yield of chlorophyll fluorescence is constant and is due to the relaxation of electrons in the orbitals of the conjugated ring system. When an electron is excited, if the energy of the transition state is not stabilised or converted to chemical energy, the electron of the chlorophyll-a molecule returns to the ground state, re-emitting light. Thus, the fluorescence of chlorophyll-*a in vivo* should be an appropriate indicator of photosynthetic competence.

It has been postulated (Cullen *et al.*, 1986) that, by measuring the increase of fluorescence upon addition of the photosynthetic inhibitor DCMU (3-(3,4-dichlorophenyl)-1,1,-dimethyl urea), one can obtain a relative measure of the capacity for photosynthesis, or, more specifically, operational PSII reaction centres.

This can be explained by the following (Van der Heever & Grobbelaar, 1998): In photosynthetic algae there are two types of reaction centres, namely PSI and PSII. The fluorescence yield from PSI and its antennae is very large so that almost all of the chlorophyll fluorescence of an algal cell is associated with PSII. When a photon is absorbed by antenna pigments and is transferred to PSII, it can drive an electron to Q_A. If Q_A is oxidised the PSII "trap" is said to be "open". Under these conditions the electron can reduce Q_A to Q_A^- and excitation energy is converted to vibrational energy (heat) or another "trap". It is believed that Q_A is a quinone tightly bound to a protein. Q_A transfers electrons to a secondary quinone called Q_B. Q_B loosely binds to or near Q_A and can also dissociate after it receives a pair of electrons and protons. Many herbicides, including DCMU, compete with Q_B for a binding site on Q_A. In the presence of such an inhibitor the complex Q_A-I is formed and PSII traps cannot turn over more than once. Thus, even in the presence of light, electron flow from water to Q_B is blocked, and fluorescence rises to a maximum value. DCMU-induced fluorescence is probably under the control of factors additional to non-cyclic electron flow capacity and variation in fluorescence may reflect secondary photochemical responses rather than direct changes in the primary driving forces of photosynthesis (Vincent, 1980).

The index of photosynthetic capacity (Ft) (which can also be seen as an index of toxicity of a pollutant) can be calculated as:

$F_t = (F_m - F)/(F_m - F_0)$

F_m = maximum fluorescence

F_0 = natural fluorescence

F = measured fluorescence of a sample

F_t = the index of toxicity

In vivo fluorescence were measured as the algae's response to possible toxic compounds.

4.3) TOXICITY TESTING BY USING *DAPHNIA PULEX*

Another species that was used in bioassays was *Daphnia pulex*. The freshwater cladoceran *Daphnia*, commonly known as the water flea, has, for over a century, been used in freshwater toxicity studies. The average life span of *D. pulex* at 20°C is approximately 50 days and populations consist almost exclusively of females during most of the year; males are abundant only in spring or autumn (*Standard methods for the examination of water and waste water*, 1995). The advantages of using these organisms, as test species, include their short life cycle, the ease of laboratory culturing, their wide distribution and ecological significance, their low space and water volume requirements and their sensitivity to chemicals (Roux *et al*, 1993). Kenaga and Moolenaar (1979) found that for a large number of chemicals, animals generally are more sensitive indicators of acute toxicity than plants. Thus, water quality limitations based on toxicity data for fish and daphnids should be sufficiently restrictive to protect algae and aquatic vascular plants. Toxicity tests with animals are carried out using a limited number of test subjects, each with an assumed individual tolerance towards the particular toxic test material (Nyholm *et al*, 1992) and the response obtained is often quantal or categorical (i.e. death).

4.4) MATERIAL AND METHODS

Approximately 400 ml of the water, from different selected sampling sites at different dates (October and November 1997), in the Modder River were freeze-dried with a Virtis Freezemobile II, Virtis Company Inc. The reason why water was sampled during these two months is because the water level increased from 0.025 m in October 1997 to 0.4 m in November 1997. It was thought that it will give a good indication of toxicity during low flow as well as high flow conditions. A known mass (varying from 0.3 to 1.5 g) of the dried product was dissolved in 50 ml deionised water and used for the toxicity tests. Since no toxicity was expected in the Modder River system, because of the absence of any chemical factories in the catchment area, screening was performed using the maximum possible concentrations of these unknown substances (total dissolved solids), to determine whether toxic compounds were present.

4.4.1) Acute toxicity tests (screening) performed with *S. capricornutum*

Selenastrum capricornutum was the algal test species used in this study, and can be considered to be of "medium sensitivity". The algae were grown in GBG 11 (Krüger, 1978) (Table 4.1) at a light intensity of 300 µmol Quanta/m^2/s and a temperature of 21°C. PAAP-medium was not used for the screening, because the algae do not grow in this medium. Nyholm and Kallqvist (1989) stated that toxicity data obtained from growth-optimised systems were more readily comparable than data obtained with growth-limited organisms. A three week old culture of *Selenastrum capricornutum* was used during the screening. The pH was 7.5 ± 1 and it remained within this range during the 24 h test period.

Table 4.1: The composition of GBG11 medium (Krüger, 1978)

Components	Concentration g/l
$NaNO_3$	0.15
K_2HPO_4	0.069
$MgSO_4.7H_2O$	0.075
$CaCl_2.2H_2O$	0.036
Na_2SiO_3	0.1
Na_2CO_3	0.2
EDTA	0.001
Citric acid	0.012
$FeSO_4.7H_2O$	0.011
Minor elements*	

*Composition of minor element solution (GBG 11)

Component	Concentration g/l
H_3BO_4	0.00286
$MnCl_2.4H_2O$	0.00113
$ZnSO_4.7H_2O$	0.00022
$NaMoO_4.5H_2O$	0.00039
$Co(NO_3)_2.6H_2O$	0.000049
$CuSO_4.5H_2O$	0.000079

Screening was done, in duplicate, by adding 5 ml of the unknown substances (100% concentration) to 5 ml of *S. capricornutum* (1000 cells/ml) in GBG11 -medium (2x dilution). The final concentrations of the samples are shown in Table 4.2. Deionised water was used as control and a concentration of 100mg/l DCMU was taken as "maximum fluorescence". Fluorescence measurements were taken at three different exposure times (2 h after exposure, 4 h after exposure and 24 h after exposure), measured with a Hitachi Model F-2000 fluorescence spectrophotometer (Hitachi Ltd., Tokyo, Japan). Excitation and emission wavelengths were set at 430 and 680 nm, respectively, as determined by Van der Heever & Grobbelaar (1998). Between measurements the samples were kept under identical conditions as the stock culture. Before each measurement the samples were placed in the dark for 20-30 minutes to adapt. Samples were handshaken during the course of the experiments, especially just before measurements after dark adaptation.

Table 4.2: Concentrations of the unknown substances (total dissolved solids) added to *Selenastrum capricornutum* cultures

Sample number	Final concentration TDS in measured sample (g/l)
KM1 (20/10/97)	0.35
KM1 (3/11/97)	0.75
KM2 (20/10/97)	1.0
KM2 (3/11/97)	2.5
KM3 (20/10/97)	0.8
KM3 (3/11/97)	1.8
KM4 (20/10/97)	0.85
KM4 (3/11/97)	1.5
KM5 (3/11/97)	1.1
GM2 (20/10/97)	0.65
GM2 (3/11/97)	1.3
GM3 (20/10/97)	0.7
GM3 (3/11/97)	1.1

4.4.2) Toxicity tests (screening) performed with *Daphnia pulex*.

Daphnia pulex are indigenous to the Modder River and the stock cultures were maintained in moderately - hard reconstituted water (Table 4.3):

Table 4.3: Moderately - hard reconstituted water for *Daphnia sp.* (Truter, 1990).

Reagent (Analytical grade)	Concentration (mg/l deionized water)
$NaHCO_3$	96.0
$CaSO_4.2H_2O$	60.0
$MgSO_4.7H_2O$	123.2
KCl	4.0

The pH was constant at around 8 ± 0.5 and temperatures were kept at 20°C. The daphnias were fed with a suspension of fish pellets, alfalfa and yeast. All tests were conducted in the same controlled conditions as described for the stock culture, but no food was given during the toxicity tests.

Healthy organisms were used for toxicity tests and therefore no daphnia were taken from cultures producing ephippia (resting eggs that are produced during unfavourable conditions). Adult females, bearing embryos, were removed from the stock cultures 24 h preceding the initiation of the test. Ten females were placed in a 400 ml beaker containing 300 ml of medium and 0.5 ml of prepared food. The young which were found in the beaker the following day (24 hours old) were used for the toxicity tests.

The test was for acute lethality, because death is an easily detected effect. For the maximum possible concentration of the freeze-dried product, a total of 10 organisms were distributed among 2 vessels (50 ml) with 10 ml of test solution and 10 ml deionised water. The final concentrations of the samples were the same as that used for *Selenastrum capricornutum* (Table 4.2). A control was set up with each test and it consisted of the same dilution water, test conditions, procedures, and organisms used in testing the sample. The test vessels were checked, for mortality, after 24 and 48 hours. If the mortality of the organisms exposed to 100% of the sample exceeded 10% at any time during the first few hours, the sample was considered to exhibit acute toxicity and a definitive test was conducted. For a valid test, control mortality must not exceed 10%.

4.5) RESULTS:

4.5.1) Acute toxicity tests performed with *Selenastrum capricornutum*

When comparing the fluorescence obtained upon addition of the unknown substances with the maximum concentration and the control, no fluorescence measurements were higher than the control (thus, $F_t > 1$), except at GM2 (3/11/97) and GM3 (20/10/97) where $F_t < 1$ (Table 4.4 and Figures 4.1 to 4.4).

Table 4.4: Toxicity index for the different samples obtained from the Klein Modder and Modder Rivers ($Ft=(F_m - F)/(F_m - F_0)$)

Sample number	1 hour exposure	4 hours exposure	24 hours exposure
KM1 (20/10/97)	1.15	1.07	1.16
KM1 (3/11/97)	1.22	1.13	1.15
KM2 (20/10/97)	1.26	1.17	1.10
KM2 (3/11/97)	1.28	1.17	1.08
KM3 (20/10/97)	1.20	1.15	1.11
KM3 (3/11/97)	1.19	1.15	1.09
KM4 (20/10/97)	1.19	1.11	1.07
KM4 (3/11/97)	1.19	1.06	1.12
KM5 (3/11/97)	1.2	1.16	1.07
GM2 (20/10/97)	1.18	1.15	1.1
GM2 (3/11/97)	1.01	0.91	0.78
GM3 (20/10/97)	1.23	1.16	0.96
GM3 (3/11/97)	1.21	1.10	1.09

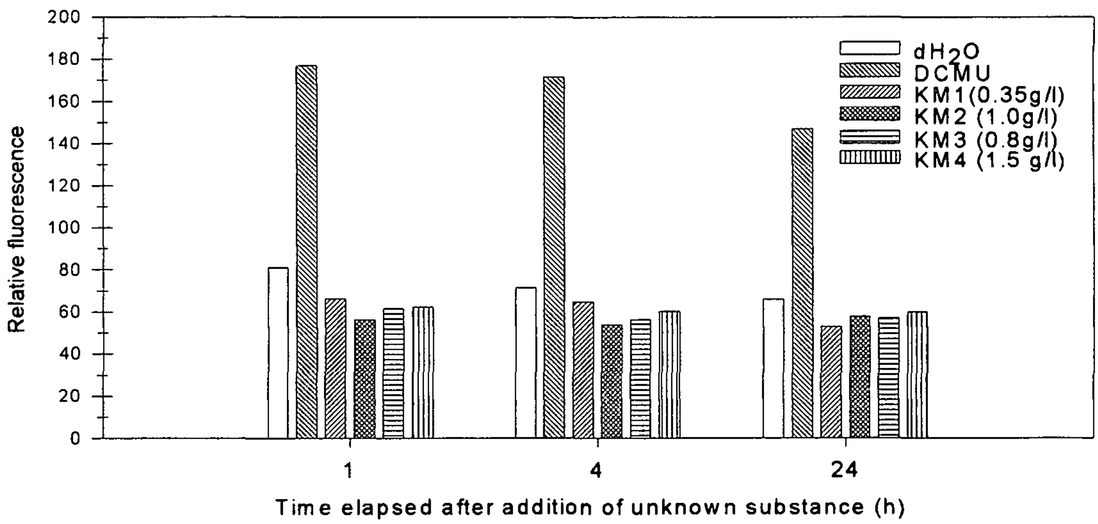


Figure 4.1: The fluorescence of *Selenastrum capricornutum* after addition of freeze-dried samples obtained from the Klein Modder River on 20/10/97.

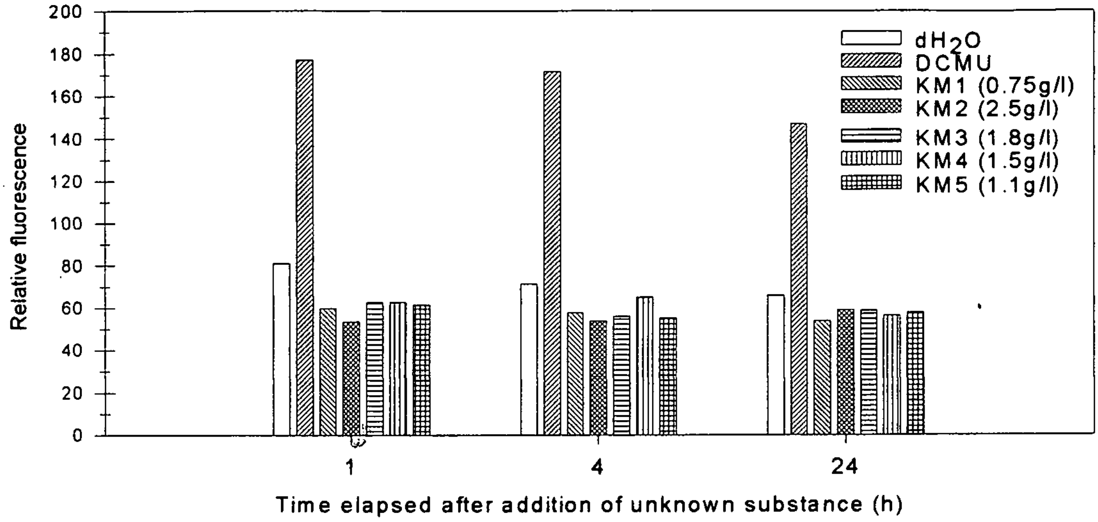


Figure 4.2: The fluorescence of *Selenastrum capricornutum* after addition of freeze-dried samples obtained from the Klein Modder River on 3/11/97.

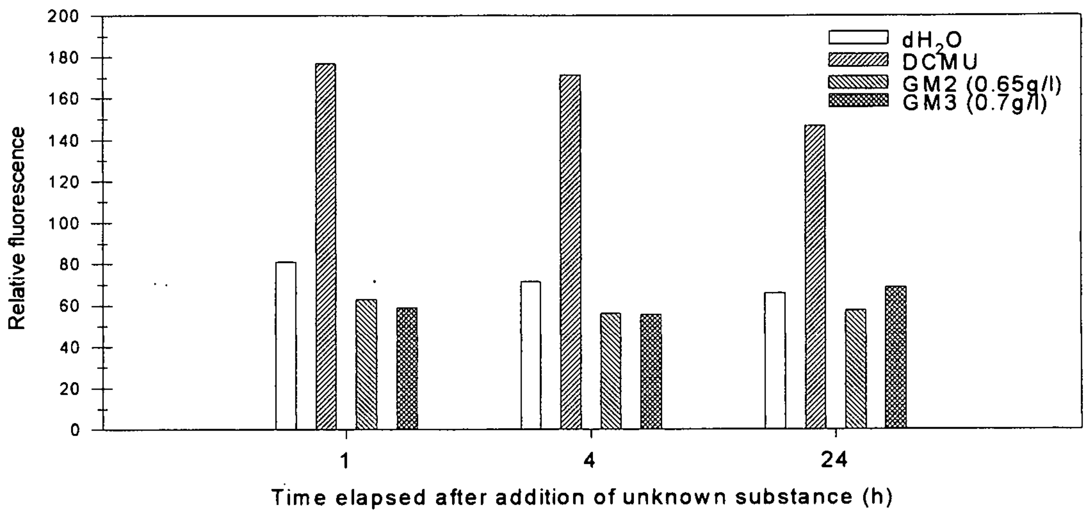


Figure 4.3: The fluorescence of *Selenastrum capricornutum* after addition of freeze-dried samples obtained from the Modder River on 20/10/97.

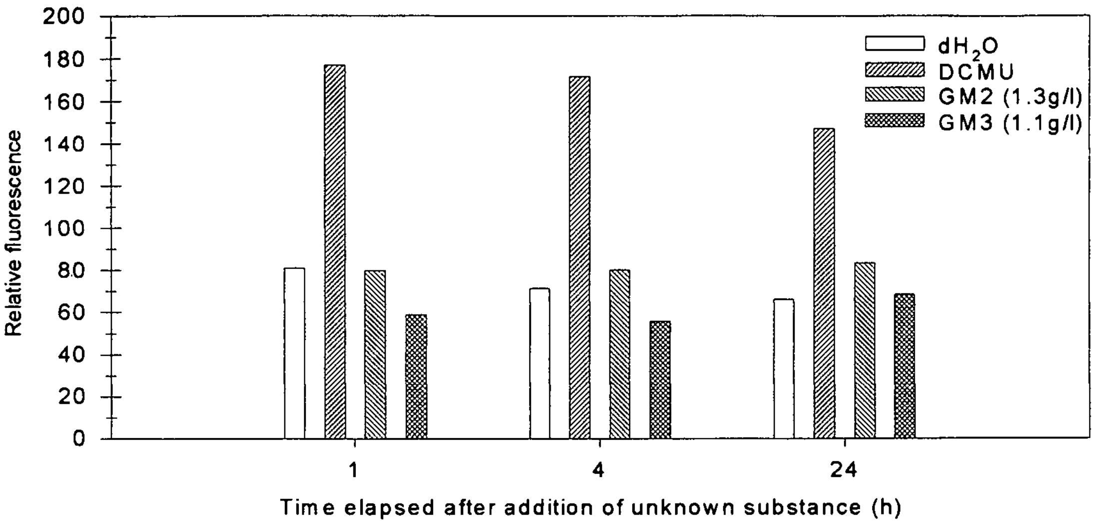


Figure 4.4: The fluorescence of *Selenastrum capricornutum* after addition of freeze-dried samples obtained from the Modder River on 3/11/97.

The freeze-dried product of GM2 (3/11/97), which showed some toxicity after only 4 hours, was analysed by the Institute for Groundwater Studies (U.O.F.S.), for copper, iron, lead and mercury. All of these metals can be considered as potentially toxic. The following concentrations were found (Cruywagen, 1998), (Table 4.5).

Table 4.5: The concentrations of some selected heavy metals in the water of GM2 on 3/11/97 (Institute for Groundwater Studies, U.O.F.S)

	Cu^{2+} (mg/l)	Fe^{2+} (mg/l)	Pb^{2+} (mg/l)	Hg^{2+} (mg/l)
Proposed boundaries	0.5	0.1	0.05	5
Maximum limit	1	1	0.1	10
Crisis limit	2	2	0.2	20
SABS specifications	1	2	0.05	5
Sample (GM2)	0.021	0.007	<0.025	<1

4.5.2) Toxicity tests performed with *Daphnia pulex*

No mortalities, in any of the test vessels, were observed after 24 hours of exposure to the unknown substances. However, a mortality was observed in KM1 (3/11/97) after 48 hours. It was considered as being non-significant, since the mortality only occurred in one of the duplicate test vessels. It is possible that the layer of dissolved solids which flocculated to the bottom of the test vessel, "trapped" one of the daphnias.

II) BACTERIA AS A THREAT TO HUMAN AND ANIMAL HEALTH

4.6) INTRODUCTION

Although no toxic compounds were measured, microbial polluted water has long been associated with the transmission of infectious diseases such as gastro-enteritis, amoebiasis, giardiasis, salmonellosis, dysentery, cholera, typhoid fever and hepatitis A (DWAF, 1993b). Faecal coliforms and *Escherichia coli* (*E. coli*) are rarely found in soil and water which has not been subjected to faecal pollution. Soil contaminated by animal faecal pollution has been shown to contribute significantly to pollution of stormwater run-off, and receiving water bodies. Run-off from informal residential areas (such as Botshabelo) is also usually contaminated with faecal coliforms and pathogens.

Stormwater run-off, together with the discharge of treated or untreated domestic wastewater, is the major source of faecal coliforms and *E. coli* in the aquatic environment. Thus, faecal coliforms, and more specifically *E. coli* are the most common bacterial indicators of faecal pollution, and hence of the possible presence of faecal-associated pathogens in water supplies.

4.7) MATERIAL AND METHODS

Bacterial counts were not determined during this study, but data were obtained from the Department for Environmental Sciences, Free State Technikon, to provide a global picture of the possible health threat of the water of the Modder River. *Escherichia coli* (*E. coli*) were enumerated on Chromocult® Coliform Agar for the detection of *E. coli* in water samples with the membrane filtration technique (Merck, 1996). The enumeration were done in triplicate 90 ml Petri-dishes. The *E. coli* counts were taken from January 1997 - December 1997. Only the *E. coli* counts were taken into account, because they act as important indicators of faecal pollution and associated increased concentrations of faecal pathogens.

4.8) RESULTS

E. coli counts varied throughout 1997 and no seasonal patterns were observed (Figures 4.5 & 4.6). However, during March in the Klein Modder River, *E. coli* counts increased at all the sampling points, with a decrease in April and in the following winter months. In the Modder River *E. coli* counts also decreased during the winter. *E. coli* counts, in the Modder River, were not as often above the limit for recreational water (150 N/100 ml), as that of the Klein Modder River (Figures 4.5 & 4.6). The mean *E. coli* counts were much higher in the Klein Modder River (about 39 600/100 ml) than in the Modder River (about 3 340/100ml) (Figures 4.7 & 4.8). In the Modder River there was a 167% increase in *E. coli* counts from GM1 to GM2.

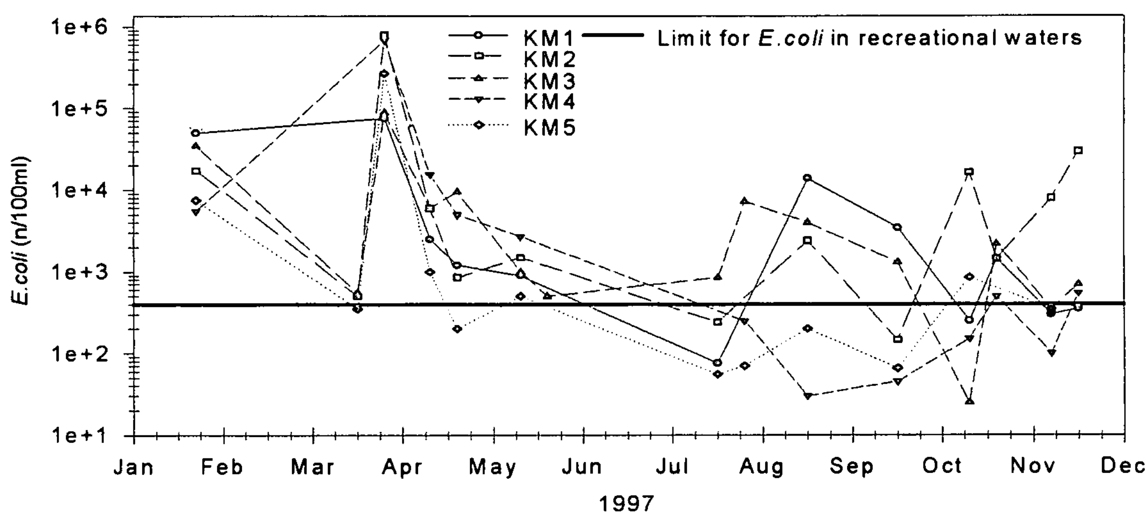


Figure 4.5: Seasonal variation in *E.coli* in the Klein Modder River during 1997

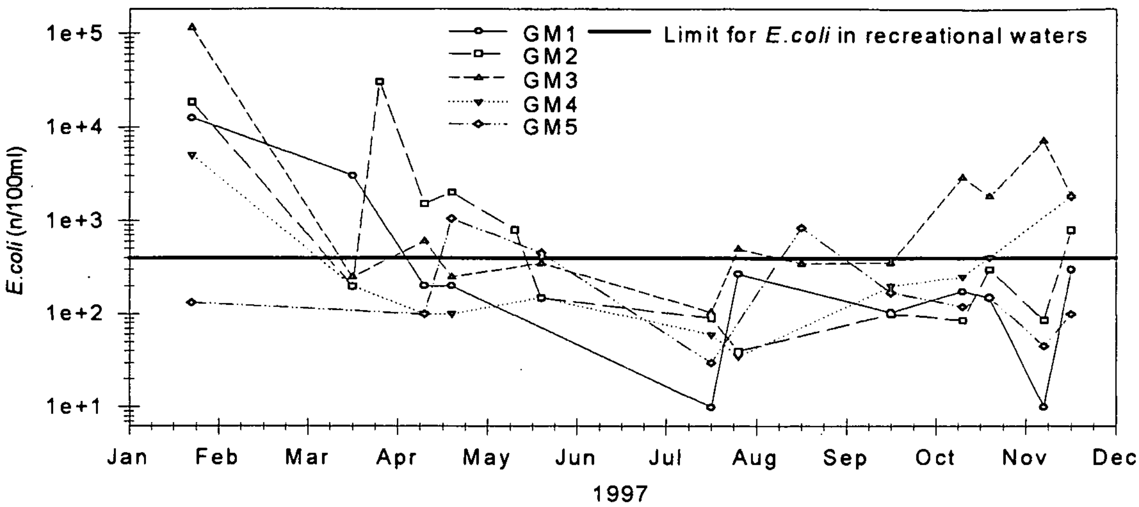


Figure 4.6: <u>Seasonal variation in *E. coli* in the Modder River during 1997.</u>

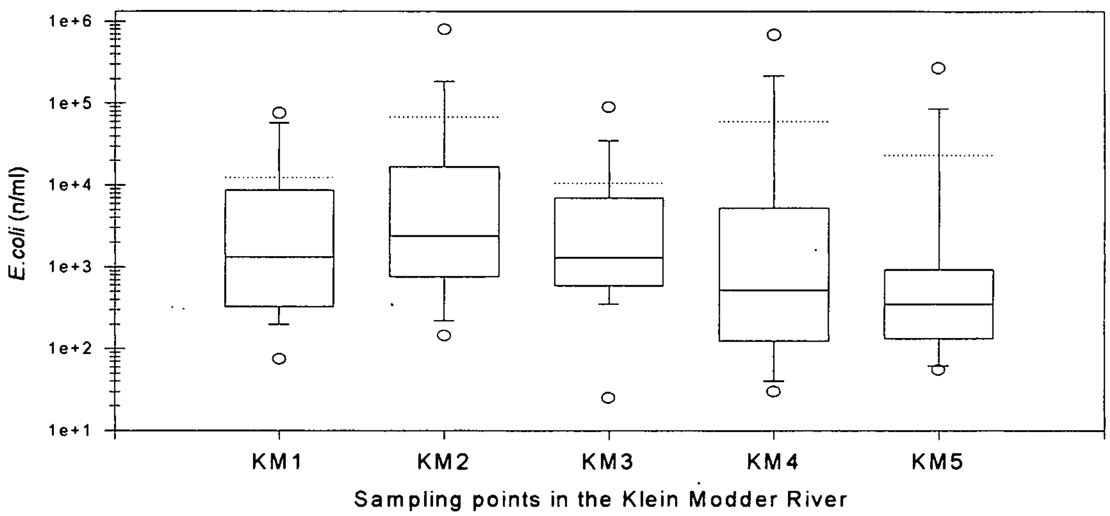


Figure 4.7: <u>Variation in *E. coli* counts downstream in the Klein Modder River</u>

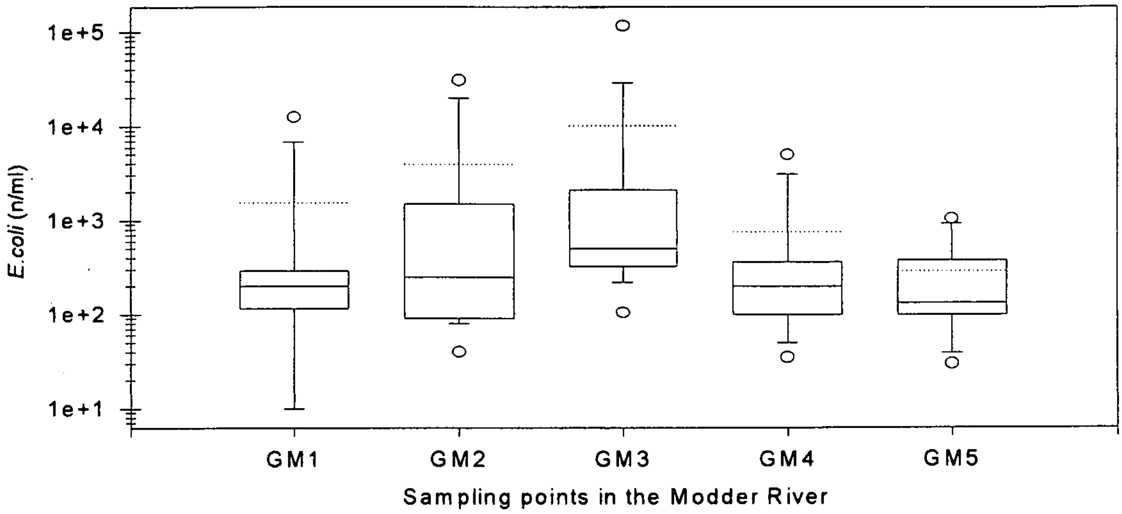


Figure 4.8: Variation in *E. coli* counts downstream in the Modder River

4.9) DISCUSSION AND CONCLUSION

4.9.1) Toxicity

Based on the preliminary screening tests on *Daphnia pulex*, the water of the Modder River was considered as non-toxic. *Daphnia* is sensitive to most toxic substances Kenaga & Moolenaar, 1979) and since no mortalities occurred in any of the Modder River samples, it was assumed that no toxic compounds were present.

The heavy metals that were determined in the water of GM2 (3/11/97), that gave a slight fluorescence reaction with *S. capricornutum*, were within the proposed boundaries as well as within the SABS specifications (Table 4.5) (Institute of Groundwaterstudies, U.O.F.S.). The South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996a), however, give much lower values at which acute effects can occur due to heavy metals:

Table 4.6: Comparison between the heavy metal concentration in the Modder River and the acute effect values according to DWAF (1996a)

Heavy metal	Concentration at GM2 (µg/l)	Acute effect value (µg/l)
Copper	21	7.5
Mercury	<1	1.7
Lead	<25	13
Iron	7	not given

However, toxicity test performed in the literature on *S. capricornutum*, gave EC (effective concentration of the toxic compound) values higher than the acute effect values proposed by the DWAF. Van der Heever & Grobbelaar (1996) found that copper concentrations of 1000 µg/l completely inhibited growth of *S. capricornutum* and a concentration of 200 µg/l showed some inhibition after 24 hours. Christensen & Nyholm (1984) reported EC values for copper of 0.0263, 0.0485, and 0.0717 mg/l (EC_{10}, EC_{50} and EC_{90} respectively). They also reported a EC_{50} value of 0.033 mg/l for mercury. The values obtained at GM2 were within or lower than these values. GM2 (3/11/97), and to some extent GM3 (20/10/97), were the only samples that showed some toxicity, and the heavy metals determined were within a safe range comparing to the literature. Thus, it can be assumed that the waters of the Modder River are non-toxic in terms of heavy metals. Also, GM2 (20/10/97) and GM3 (3/11/97), showed no toxic characteristics indicating that if there are toxic metals in the Modder River, they occur in relatively low concentrations.

It can be concluded that it is difficult to determine the exact toxicity of an unknown substance, because different factors affect the toxicity of chemical compounds (Babich & Stotzky, 1980). These include the pollutant characteristics, environmental factors and the physiological state of the biota. Clay particles can also influence the toxicity of metals. Heavy metals introduced into the environment may be exchanged for cations on the exchange complex of clays, and, thereby, the metal toxicants are removed, at least temporarily, from solution and their uptake by the microbiota reduced (Babich & Stotzky, 1980). This is possible in the Modder River, since the water is very turbid.

It is important to remember that results from toxicity tests are better to compare the toxicity of chemicals and determine the sensitivities of different algae than to predict environmental impact (Calow, 1993). Therefore, the major limitation of the algal toxicity test method is the uncertainty of environmental significance on the results and usually several different test species are required to provide a better indication. Attention must be focused on the physico-chemical characteristics of the specific environment into which the pollutants is deposited if a clearer and more meaningful understanding of the impacts of pollutants on the biota is to be obtained (Babich & Stotzky, 1980).

4.9.2) Bacteria

E. coli counts in the Klein Modder River were very high (about 39 600/100 ml). Since *E. coli* is used to indicate potential of infection by associated pathogens such as *Salmonella* sp. it can be assumed that the water of the Klein Modder and Modder Rivers poses a threat to human health. The *E. coli* limit for recreational use is 300/100ml. At this concentration there is a risk for gastro-intestinal illnesses (DWAF, 1996b). Above 2000/100ml *E. coli* counts, there exists a very significant risk for the occurrence of gastro-intestinal illnesses (DWAF, 1996b). In the Klein Modder River and sometimes in the Modder River, the *E. coli* counts exceeded these limits. This indicated pollution of faecal origin, that can be ascribed to the sewage effluent from the Botshabelo sewage works as demonstrated by the increase of *E. coli* at GM2 in the Modder River. The *E. coli* counts at GM3 were higher than at GM2. This could be ascribed to the fact that there occurred less turbulence at GM3, providing a more stable environment as well as a longer retention time for bacterial growth. Downstream dilution did occur to GM5, and bacterial counts decreased to levels lower than at the reference point.

However, sewage effluent is not the only contributor to the pollution. The high *E. coli* counts measured at KM1, indicates that diffuse effluents (such as surface run-off) from the city itself, were also major contributors. At KM3, the E. coli counts were much lower than at KM4 and KM5. Again, this can be explained by the fast-flowing water at KM3, creating unfavourable, unstable conditions for growth.

The limit for *E. coli* counts in water for domestic use, is 0/100ml (DWAF, 1996c). The *E. coli* in the water of the Modder and Klein Modder Rivers were far above this limit, thus the water was not fit for domestic use.

Because *E. coli* acts as an indicator organism for several pathogens, it could be possible that these pathogens are present in the Klein Modder River, as the *E. coli* counts were very high. The high *E .coli* counts in the Klein Modder River, indicating potential infection by associated pathogens, posed a threat to animal health as well. If *E. coli* counts in water consumed by livestock exceed 1000-5000 for more than 50% of the samples a significant risk is associated with the use (DWAF, 1996d).

Thus, for most of the time, the water of the Klein Modder River is not fit for animal or human use, nor for recreation. In the Modder River, dilution has a significant decreasing effect on the *E. coli* counts in the water, but judged by the results, the bacterial concentration often still exceeds safe levels.

CHAPTER 5

MODEL CALIBRATION AND VERIFICATION WITH THE WATER QUALITY DATA OF THE MODDER RIVER

5.1) INTRODUCTION

Models can be defined as *tools of thinking to formulate hypotheses which have to be verified by observations* (Vollenweider, 1975). The use of mathematical models to simulate ecological and water quality interactions in surface waters, has grown dramatically during the 1970's and 1980's (Bowie *et al.*, 1985) and have proved to be invaluable for improving our understanding of the processes operating in the water environment. They can also be used to assist with management and planning decisions. It is foreseen that water quality models will in future become increasingly refined and integrated with the increasingly complex water quality problems we can expect to face in the future (Du Plessis & Van Veelen, 1991).

Models, in order to be satisfactory should meet three essential criteria (Vollenweider, 1975). They should be general, realistic and precise. Models that meet all these criteria to the same extent, are seldom realised. In general, one has to sacrifice one property to maximise the others.

The ecological impact of polluting substances is important when determining the level of water pollution in rural areas. Modelling forms an integral part of present-day research and plays an important role to predict and forecast situations that can arise in dynamic ecosystems. The use of PC-QUASAR can give an indication whether models that are developed overseas can be used on South African river ecosystems, (especially one as turbid as the Modder River), to predict correctly the effects of pollution, eutrophication or other events on a lotic system.

5.2) DISTINGUISHING BETWEEN MODELS

The choice of a model for impact assessment on aquatic ecosystems should be based on the hydrological or water resources problems to be studied (Arnell, 1992). Since there are at present such a large array of river water-quality models available, there are several ways of distinguishing between them. This is done in terms of the following (Crockett, 1992):

5.2a) Modelling purpose

Models can be used with different perspectives in mind. For instance, it can be used to gain insight into physical, chemical and biological characteristics of a river (as in the case of the Modder River), or to help managers and planners to control a system. Before selecting a model, the model user must take the purpose for which the model will be used, into consideration. PC-QUASAR (**P**ersonal **C**omputer - **QUA**lity **S**imulation **A**long **R**ivers), one such model, developed by the Institute of Hydrology, United Kingdom, was designed for use by river regulatory authorities and water/sewage utility companies in the implementing and monitoring of River Quality Objectives.

5.2b) Hierarchy of the model

Different models are used for the completion of a variety of tasks, ranging from simple tasks to more complex. Planning models can evaluate investment programmes, design models can give an evaluation of planning options and operational models can be used in the everyday management of rivers. Each model, depending on its function, has a different position in this "hierarchy" of models.

5.2c) Formulation of the model

The formulation of any model can be either deterministic or mechanistic. Stochastic models contain stochastic input disturbances and random measurement errors (Jørgensen, 1994). If they are both assumed to be zero, the stochastic model will reduce to a deterministic model. A deterministic model assumes that the future response of the system is completely determined by a knowledge of the present-state and future measured inputs. Thus, in a deterministic model the parameters are presented as exact numbers, where stochastic models present parameters in terms of statistical distributions. PC-QUASAR can be adjusted to be both a stochastic and an deterministic model.

5.2d) Physical and hydrological representation

As the complexity of the model's representation of the system increases, the quantity of data required will also increase. Most river water-quality models use a one-dimensional approximation of a river by only simulating longitudinal differences in a river. Thus, the system geometry is formulated conceptually as a linear network of segments or volume segments (Bowie *et al.*, 1985). PC-QUASAR models a river as a series of reaches usually defined by the locations of tributary confluences, weirs, public water supply intakes or effluent discharge. Each reach is subdivided into a number of subreaches; each modelled as a stirred tank reactor. Models can also have parameters that are either steady-state (constant over time), dynamic (variable over time) or a mixture of both. To assess short-term impacts, a dynamic model will be needed, while a steady-state model is better to predict long-term effects (DWAF & WRC, 1995).

5.2e) Model generation

The simplest forms of models are the empirical formulae. Second was the development of site-specific numerical models, while generalised main-frame models were the third generation. PC-QUASAR is a fourth generation model, being a menu-driven PC model. The fifth generation model will soon be developed, which will collaborate water quality models with expert systems.

5.3) PC-QUASAR

The legislative background of pollution control in the rivers of England and Wales has produced models with a unique structure. As consent standards are set within a framework the model is required to predict the 95 percentile discharge quality with the achievement of a 95 percentile river quality (Crockett, 1992). As a result, several consent-setting models have been developed to enable the statistically based discharge consents to be set on a catchment scale. Currently three models are available: TOMCAT (developed by Thames Water), SIMCAT (developed by NRA Anglian Region) and QUASAR (developed by the Institute of Hydrology, UK).

PC-QUASAR is a river network, water quality and flow, model. The model comprises a set of equations describing the changes in water quality and flow over time. At the input to a reach, a mass balance is performed on all the inputs or abstractions and the resulting river quality is routed down the reach. During their passage through the reach, the concentrations of the water quality parameters are modified according to instream physical and chemical processes.

PC-QUASAR can be run in two modes: a stochastic mode for planning purposes and an operational mode for pollution event and time series studies. In its operational (dynamic) mode, time series data are put into the model and flow and quality estimates are generated at each boundary over time. Travel times are incorporated so that pollution pulses can be tracked downstream. Two sets of output data are generated in this mode. One is based on "observed" values and the other on user-edited values. The user-edited values give the prediction of changes in interactive parameters by means of graphic results. In the stochastic (planning) mode, a Monte Carlo simulation approach is used to provide distributions of flow and quality at key sites of interest. Repeated model runs using randomly selected values from these distributions are used to generate downstream distributions.

In dynamic mode the simulated water quality and flow can be viewed, either as a profile along the river system, or against time at any reach of interest. In the planning mode cumulative frequency and distribution curves are generated at any point.

5.4) WHY PC-QUASAR?

A model must not be too complex for the system the model user is planning to simulate. In fact, simplicity is the most desirable attribute of mathematical models (provided it is a reasonable representation of the real system) (Petrie, 1978). PC-QUASAR was used on the Modder River to simulate the effects of the Botshabelo sewage effluent on the system as well as an increase in algal growth. The parameters modelled by PC-QUASAR are flow, nitrate, dissolved oxygen, ammonia, temperature, *E. coli*, pH, BOD (biochemical oxygen demand) and a conservative pollutant or tracer. All these parameters, except ammonium (which was obtained from the Department of Water Affairs and Forestry) were determined during the study, thus the model seemed to be ideal to simulate the water quality of the Modder River from the measurements made. Phosphate, a major algal nutrient was unfortunately not included in the model. The data that were not available, such as algal respiration slopes, algal respiration offset, rates of denitrification, etc., could be derived from other similar systems or by implementing present knowledge of the system. Sometimes, only an average value (within the range given by PC-QUASAR) was selected.

PC-QUASAR is a model that runs on *Microsoft Windows* version 3.1 or higher, *Windows 95* or *Windows NT*. It is entirely menu-driven, with interactive data preparation and editing using menus and forms. The program has also been designed to be easy to use with no requirement to understand the computer operating system or the structure of data files. Thus, it is very user-friendly, providing the minimum of problems. It is often better to use well-established modelling software that has been tried and tested, rather than develop a new model.

The sensitivity analysis, calibration, validation and application of PC-QUASAR were the aim of this exercise.

5.5) CALIBRATION AND SENSITIVITY ANALYSIS PROCEDURE

5.5.1) PC-QUASAR and software

PC-QUASAR was obtained from the Institute of Hydrology, Centre for Ecology and Hydrology, Maclean Building, Wallingford. It was run on a 486 PC, with 12 Mb RAM, and > 5Mb of hard disk space (for program and data storage). The PC had *Microsoft Windows* 3.11 for Workgroups installed on the hard drive.

5.5.2) Steps involved in running a PC-QUASAR simulation

i) Define river network and store in mapfile
ii) Create index file
iii) Load map file to create parameter set
iv) Select planning or dynamic mode
v) Choose river(s) to be modelled
vi) Edit parameters as required
vii) Save parameter set to index file

viii) Run model

ix) Produce plots required

5.5.3) Parameter estimation

Parameters can be obtained by using measured field data, from the literature or deducted from current knowledge of the system. Several sets of parameters were tested by the calibration, that is, the various model outputs of state variables were compared with measured or observed values of the same state variables (Jørgensen, 1994).

One of the features of PC-QUASAR is the presence of rate coefficients for each reach in the river. Since all of this data was not always available or not measured experimentally, values were estimated.

5.5.4) Calibration and sensitivity analysis

During this experiment an informal calibration procedure was done by choosing some parameter values. The simulated model performance was then compared with the observed behaviour of the Modder River system. The reach rate coefficient values of PC-QUASAR were selected within a range obtained from the literature and other South African River systems, such as the Vaal River. The parameters were changed to try and obtain the most reliable simulation as observed during the field measurements. If PC-QUASAR was found to be inadequate in its characterisation of reality, the parameter values were adjusted.

The major goal of sensitivity analysis is to establish the relative sensitivity of the model predictions to the uncertainty of the model parameters or in the input database. If the output is completely insensitive to a specific parameter, it means that it will be impossible to find a reliable estimate for that parameter during calibration. If the output is very sensitive to a specific parameter or certain input data, special effort should be taken to obtain a good parameter estimate or reliable input data (DWAF & WRC, 1995). Biological parameters are generally more sensitive to interactive, environmental factors as well as several feedback biochemical mechanisms (Jørgensen, 1994). Thus, when simulating biological parameters, it is important to select and adjust influencing parameters to reflect this sensitivity.

For calibration of a model, it is of great importance that the observations reflect the dynamics of the system. If the objective of the model is to give a good description of one or a few state variables, it is essential that the data can show the dynamics of just these internal variables. The frequency of the data collection should therefore reflect the dynamics of the state variables in focus.

5.6) RESULTS AND DISCUSSION

All the simulations were done in dynamic analytical mode. The changes in the rate coefficients did not affect the user-edited simulations and all the simulations were nearly the same. No managerial questions were answered and only a few representative examples, are given, to demonstrate the insensitivity of the model. The run time step was set to 240 minutes, the output timestep to 1440 minutes (24h) and the run output length range was set to 200. To simulate the immediate effect of the power failure at the Botshabelo sewage works, the run time step was step to 15 minutes, the output time step to 60 minutes (1 h) and the run output length range was set to 8. The following parameters were edited during calibration:

5.6.1) Direct discharge increase

Direct discharge from the Botshabelo sewage works was increased from 0.0578 m^3/s (5Ml a day) to 0.12 m^3/s (100% increase). Because of the increasing population in South Africa and the increasing development of urban and industrial areas, it is likely that the sewage effluent of Botshabelo will increase in the future. It was thought that by simulating the a 100 % increase in the wastewater that the treatment facility will treat, a good indication will be given of the sensitivity of the model for changes in a direct discharge source. In the Modder River no change in either nitrate, dissolved oxygen, biochemical oxygen demand (BOD), ammonia or *E. coli* was observed between the observed run and the user-edited run (Figure 5.1 - 5.5). However, the different reach profiles in the Klein Modder River showed increases in all the above-mentioned parameters during the user-edited run. The nitrate concentration in the Sewage works/Botshabelo Dam reach increased during January to June (Figure 5.6), The dissolved oxygen concentration increased from March to June (Figure 5.7), the BOD showed an increase From March to June (Figure 5.8), the ammonia increased from January to June (Figure 5.9) and the *E. coli* increased from January to June, with a large increase during April and May (Figure 5.10). The Botshabelo Dam/ Vadersgift reach profile showed exactly the same pattern.

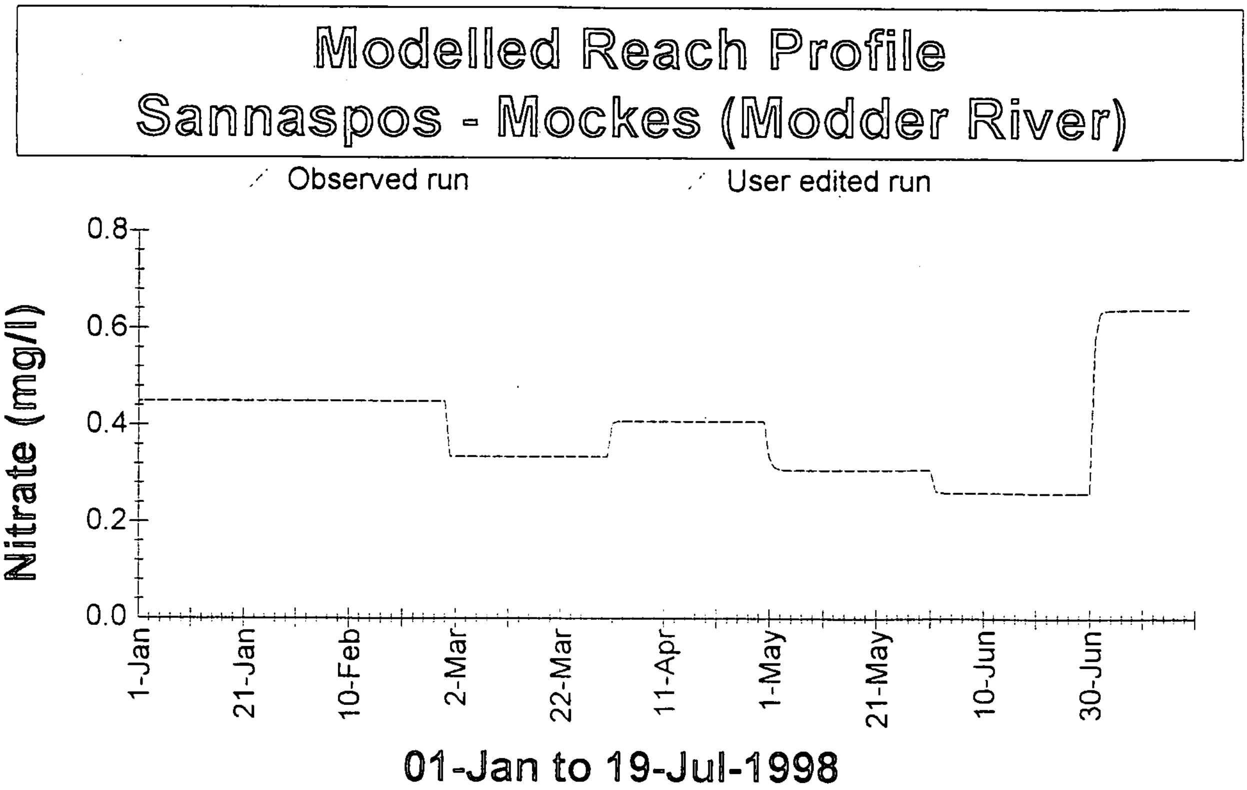


Figure 5.1: Variation in nitrate concentration in the Modder River with an increase in sewage discharge

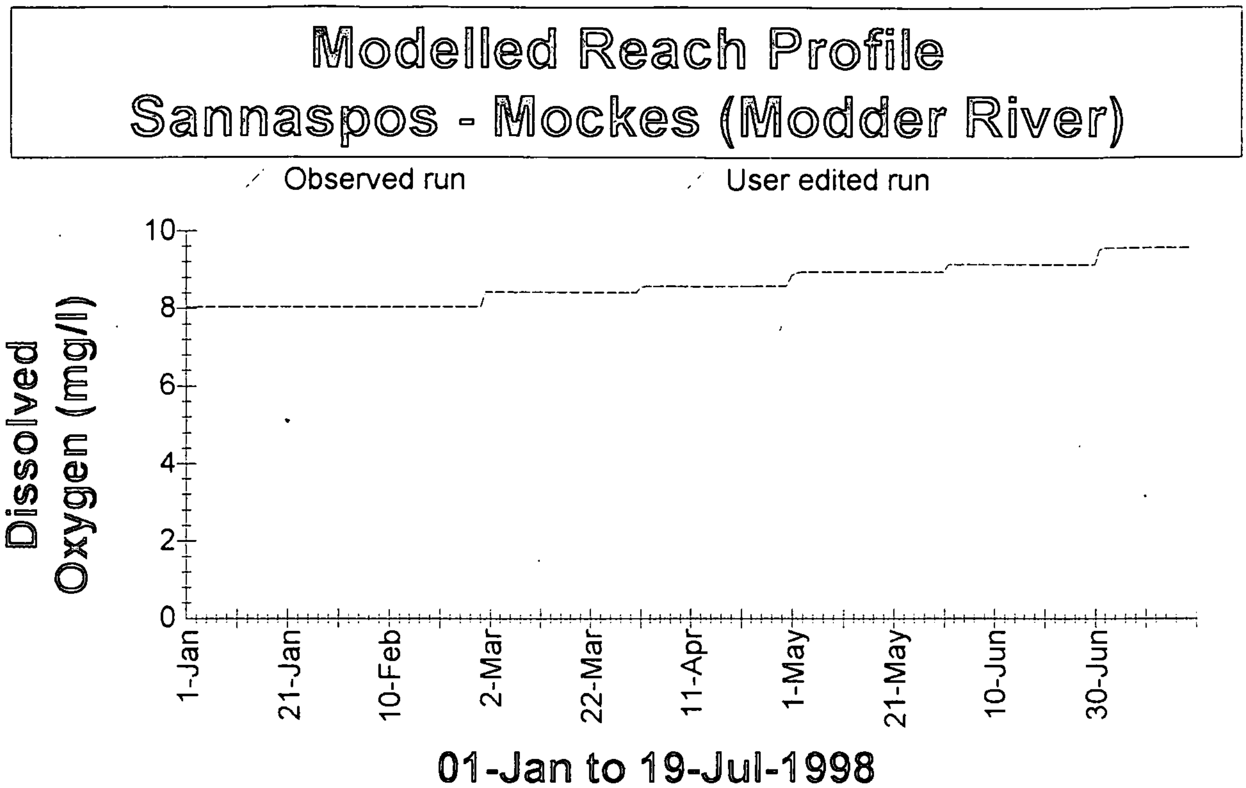


Figure 5.2: Variation in dissolved oxygen in the Modder River with an increase in sewage discharge

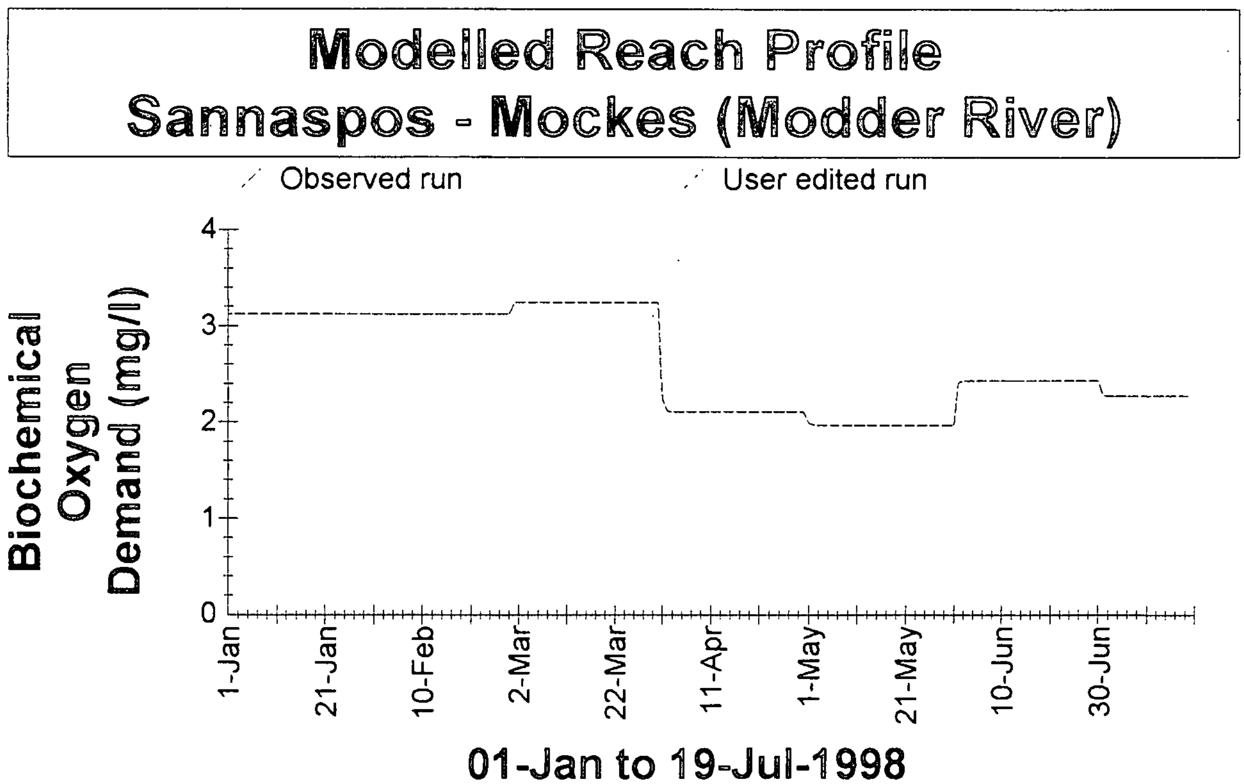


Figure 5.3: Variation in BOD in the Modder River with an increase in the sewage discharge

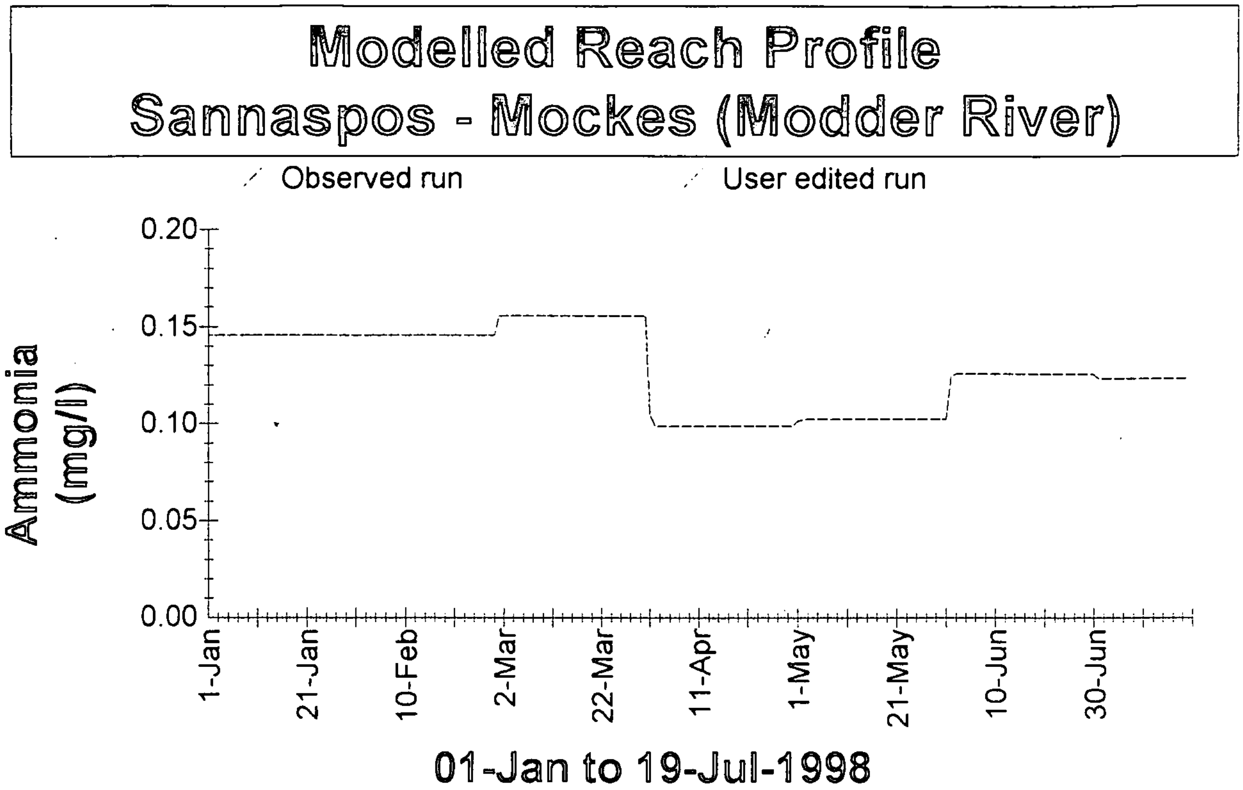


Figure 5.4: Variation in ammonia concentration in the Modder River with an increase in sewage disharge

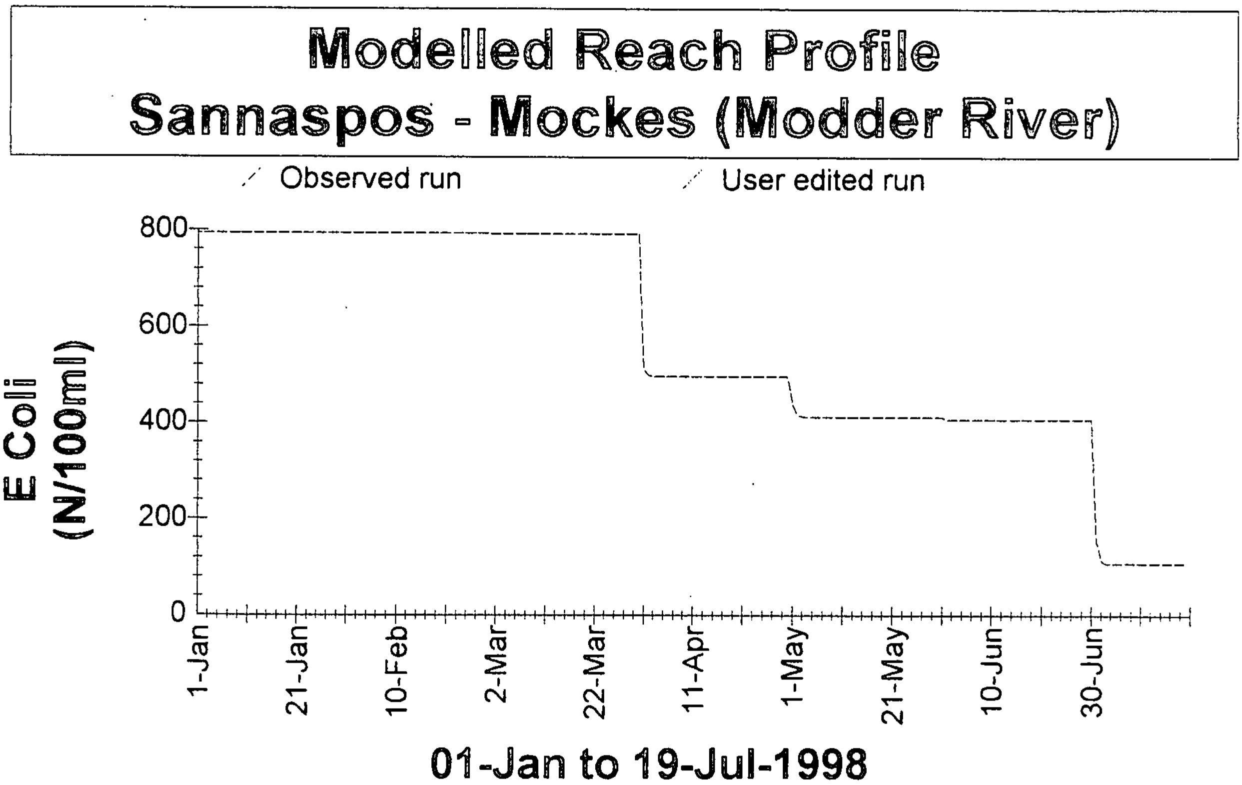


Figure 5.5: Variation in *E. coli* in the Modder River with an increase in sewage discharge

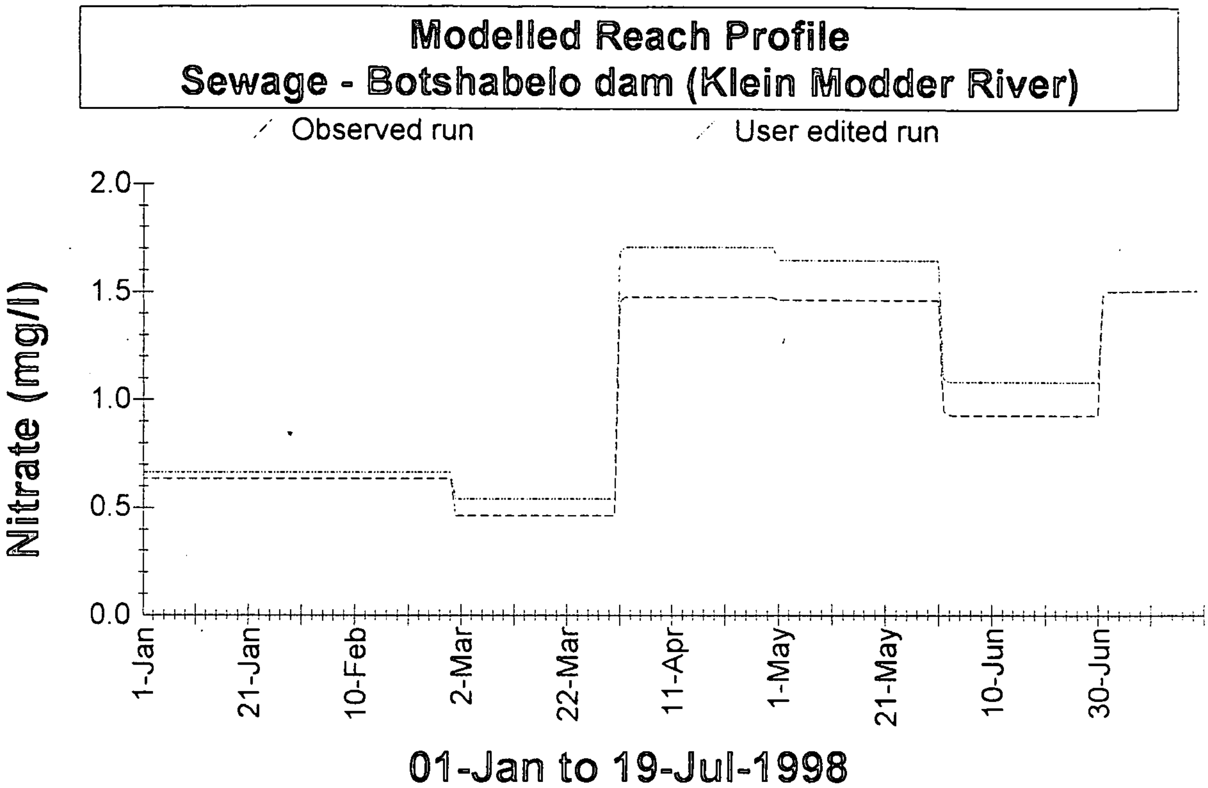


Figure 5.6: Variation in the nitrate concentration in the Klein Modder River with an increase in sewage discharge

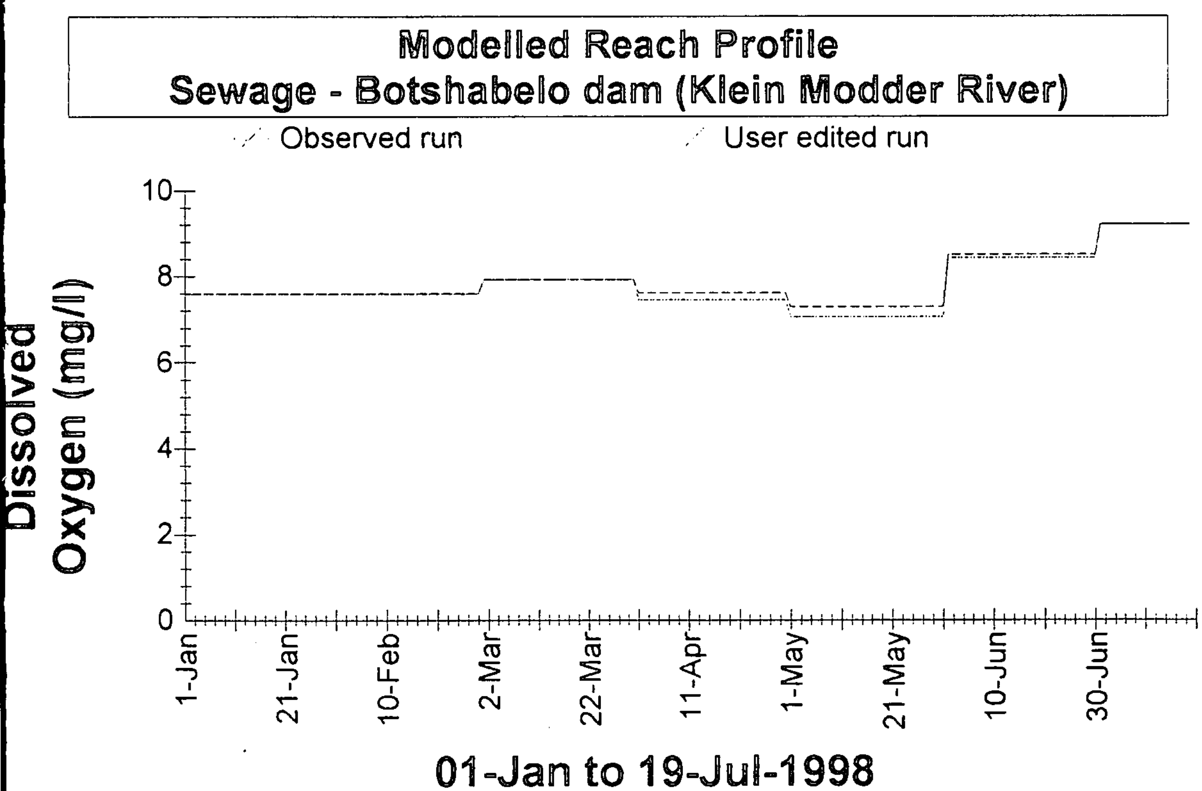


Figure 5.7: Variation in dissolved oxygen in the Klein Modder River with an increase in sewage discharge

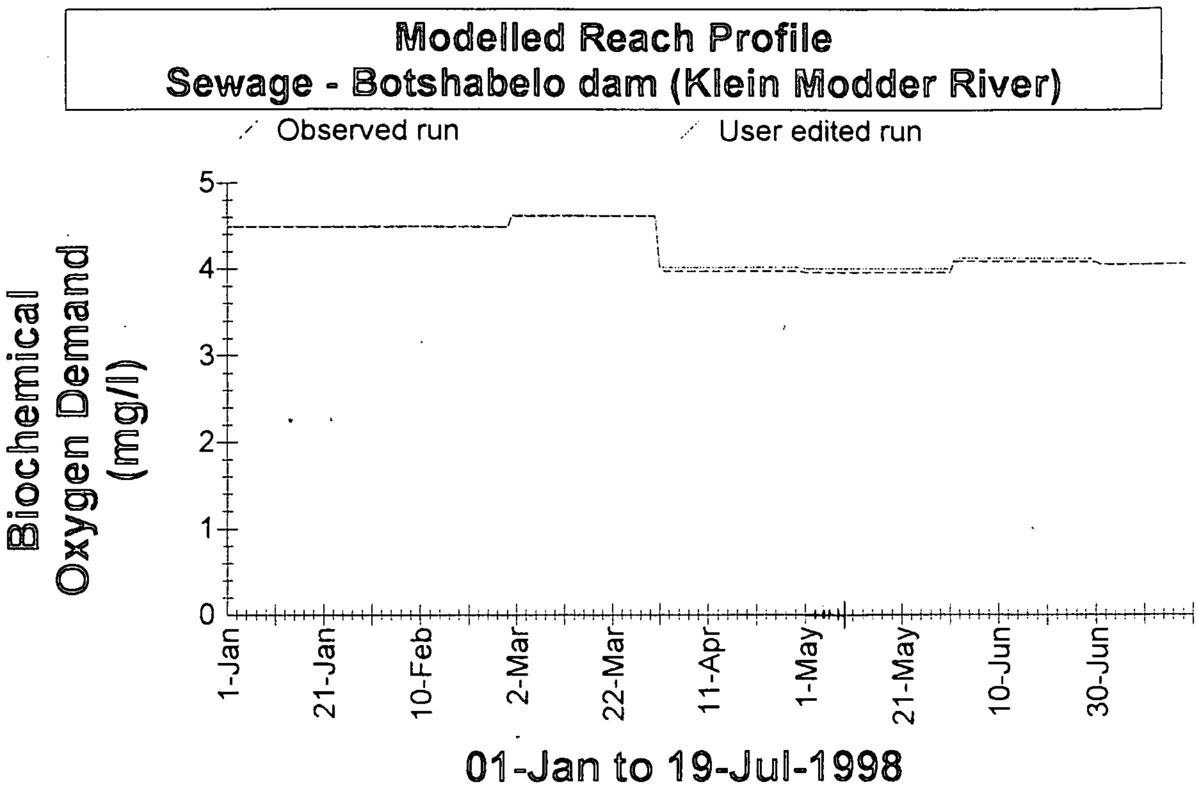


Figure 5.8: Variation in BOD in the Klein Modder River with an increase in sewage discharge

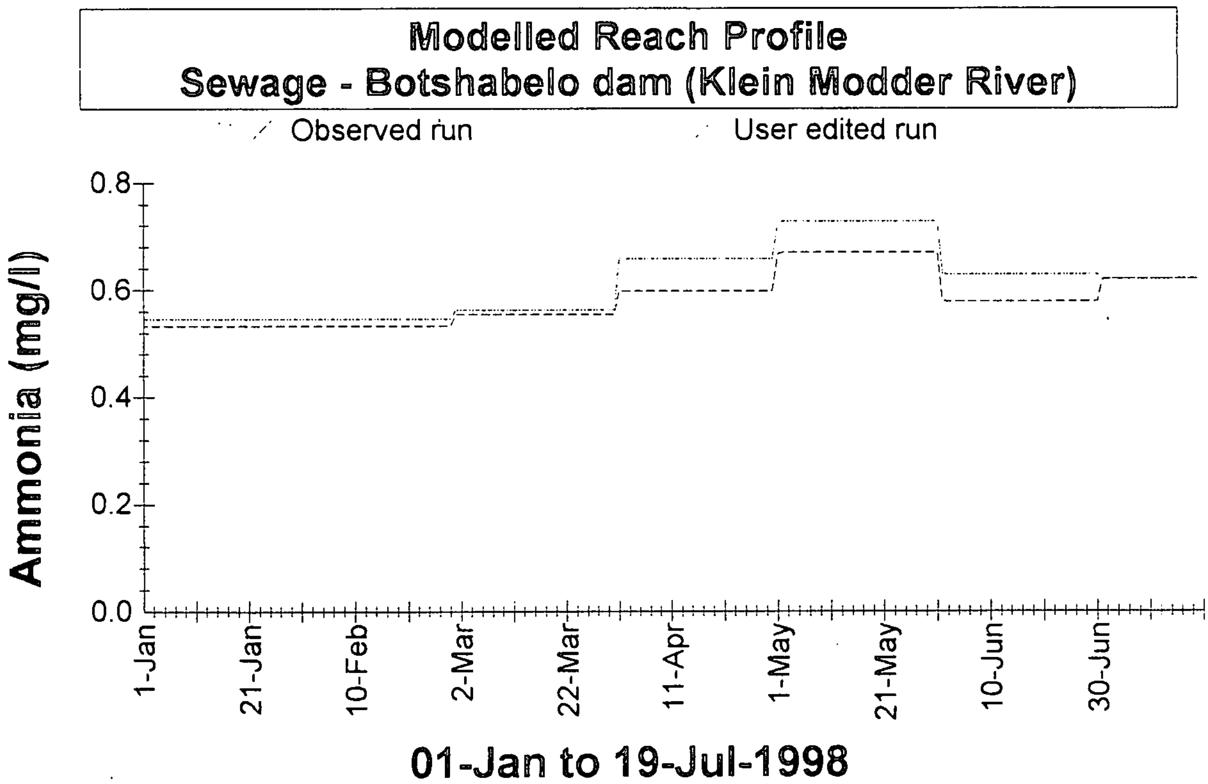


Figure 5.9: Variation in ammonia concentration in the Klein Modder River with an increase in sewage discharge

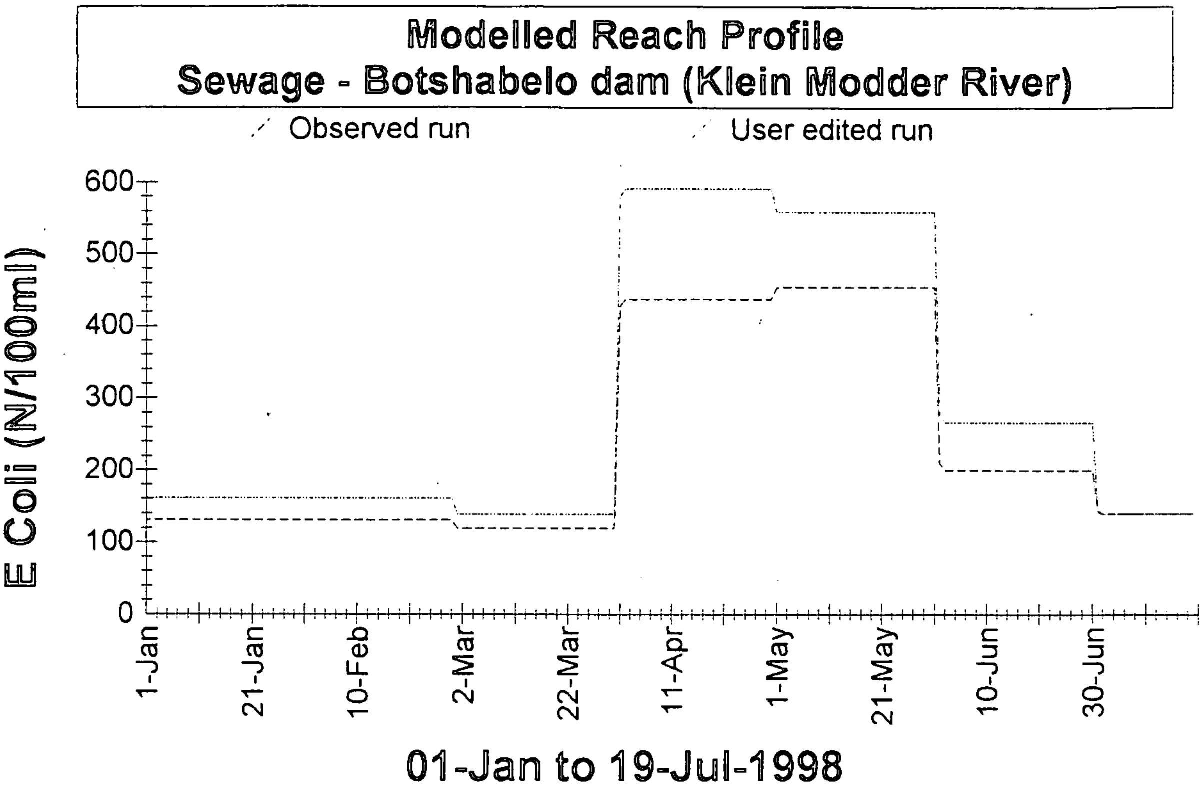


Figure 5.10: Variation in *E. coli* in the Klein Modder River with an increase in sewage discharge

The increased discharge at the sewage plant showed an effect on the water quality of the Klein Modder River. However, no effect was observed in the first reach of the Modder River. The model thus simulates the effect in the river that receives the effluent (Klein Modder River), but it does not simulate the effect into the second-order river, in this case the Modder River. During the study the effect of the polluted Klein Modder River on the water quality of the Modder River was clearly illustrated (Chapters 2 & 3). An according simulation was therefore expected, especially with high nutrient concentrations, but was not found during calibration.

5.6.2) Increase in chlorophyll-*a*

Since the nutrient concentrations of both the Modder and Klein Modder Rivers are high and algal blooms occurred during low flow conditions when light availability was high, the chlorophyll-*a* concentration in all the reaches of both the Klein Modder and Modder Rivers was increased to 10 000 µg/l. This is a very high increase (250%), and was done to determine the sensitivity of other parameters to an increase in chl-*a*. In the Klein Modder River no changes were observed in BOD, dissolved oxygen, pH or nitrates. In the Modder River the different reaches showed different changes. The Rustfontein/Sannaspos reach showed a 0.1 mg/l increase in dissolved oxygen concentration during May (Figure 5.11), as well as a 0.2 mg/l increase in BOD from April to May (Figure 5.12). The Sannaspos/Mockes Dam reach showed a 0.4 mg/l increase in dissolved oxygen during April and a small increase in BOD from April to June. In the Mockes Dam/Mazelspoort reach no changes were observed in the dissolved oxygen, but the BOD was 0.25 mg/l higher from April to June for the user-edited run. There were no changes in the nitrates or pH for the user-edited run for any of the reaches in the Modder River. Since the increase in chlorophyll-*a* to 10 000 µg is unrealistic and will not occur under natural conditions, the absence of results was disappointing. Thus, the model should be rejected, because of its lack of sensitivity to the excessive growth of algae.

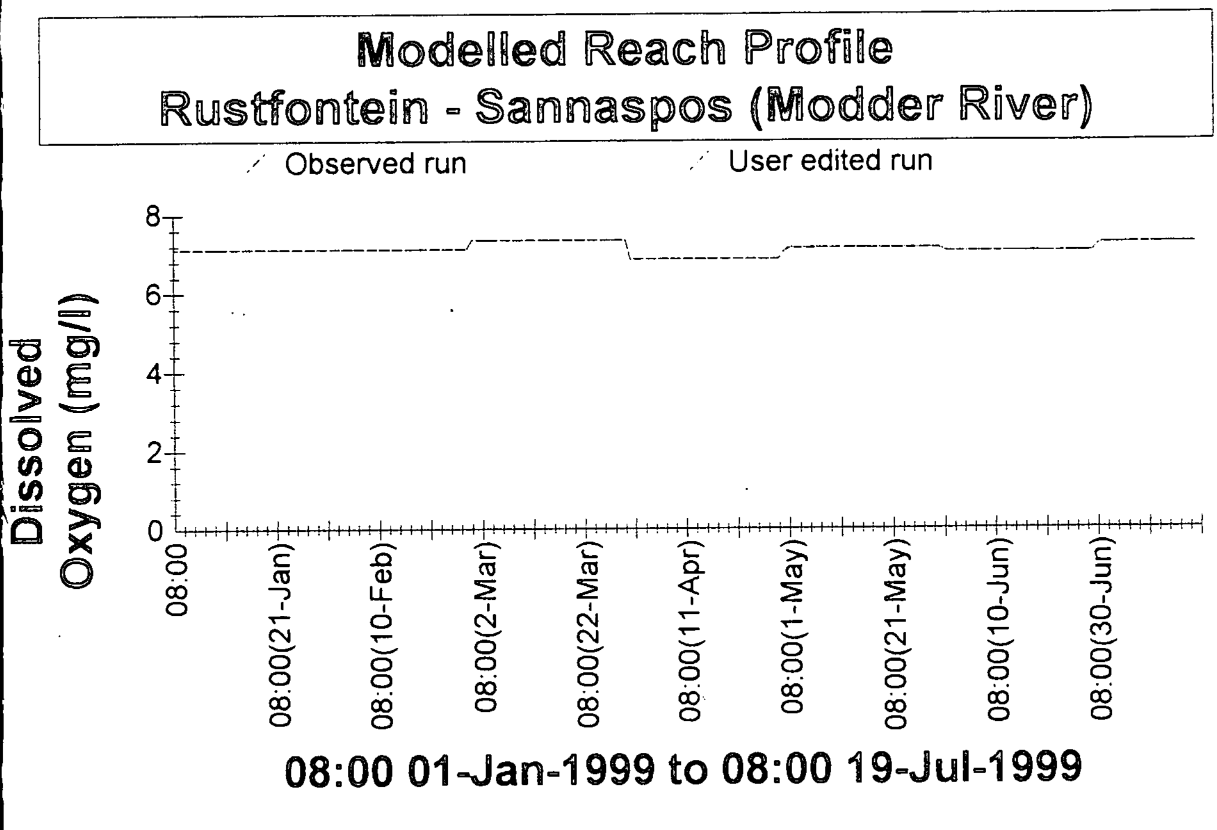


Figure 5.5: <u>Variation in dissolved oxygen in the first reach of the Modder River with an increase in chl-*a*</u>

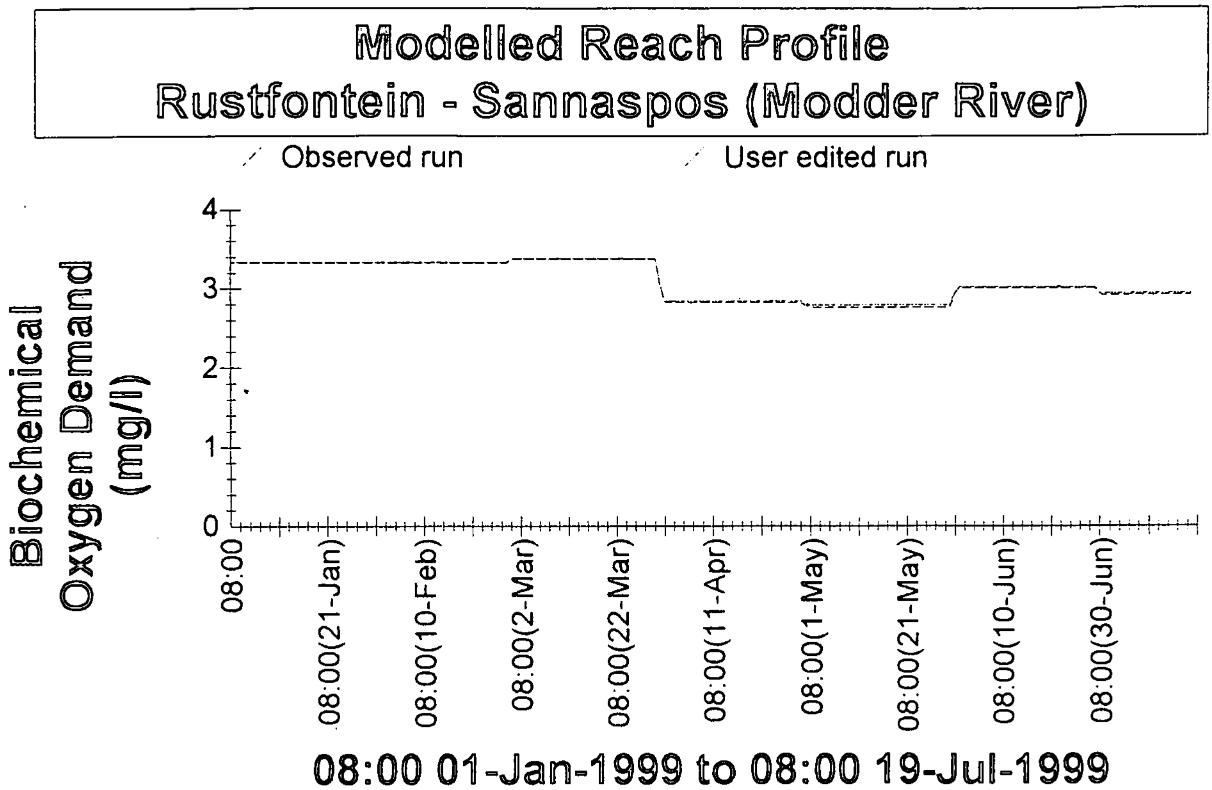


Figure 5.6: Variation in BOD in the first reach of the Modder River with an increase in chl-*a*

It was found that chlorophyll-*a* is only linked to dissolved oxygen by PC-QUASAR through respiration and photosynthesis (Williams, 1998, personal communication). It is not linked to either pH or nitrates, thus, chlorophyll-*a* in PC-QUASAR can be considered as a relatively insensitive biological parameter. BOD is a measure of the consumption of oxygen caused by the decay of organic matter (carbonaceous BOD) together with the oxygen consumed by the nitrification of ammonium (nitrogenous BOD) and is probably the most sensitive biological parameter of PC-QUASAR. PC-QUASAR models the processes influencing carbonaceous BOD and nitrogenous BOD separately. Changes in carbonaceous BOD are due to decay, sedimentation and the addition of dead algae. BOD is also modelled by PC-QUASAR to be temperature dependent.

5.6.3) Creating an impulse - power failure at the Botshabelo sewage works

A hypothetically impulse was created (a power failure at the Botshabelo sewage works), with the following parameters:

Flow = 0.0578 m^3/s

Nitrate = 10 mg/l

Conservative (a possible pollutant) = 4 mg/l

Dissolved oxygen = 2 mg/l

Biochemical oxygen demand = 10 mg/l

Ammonia = 10 mg/l

Temperature = 20°C

E. coli = 10 000 000 N/100ml

pH = 6.4

No changes were observed for any parameters in the Modder River (nitrate, dissolved oxygen, ammonia, *E. coli* and BOD). In the Klein Modder River, all the parameters changed that were changed in the sewage effluent, including nitrate, dissolved oxygen, BOD, ammonia, and *E. coli*. As an example only the variation in nitrate concentration in the Modder River (Figure 5.13) and in the Klein Modder river (Figure 5.14) are given. All the parameters showed the same increase ("pulse")in the Klein Modder River at 08:00 on 1 January, with a rapid decrease during the day.

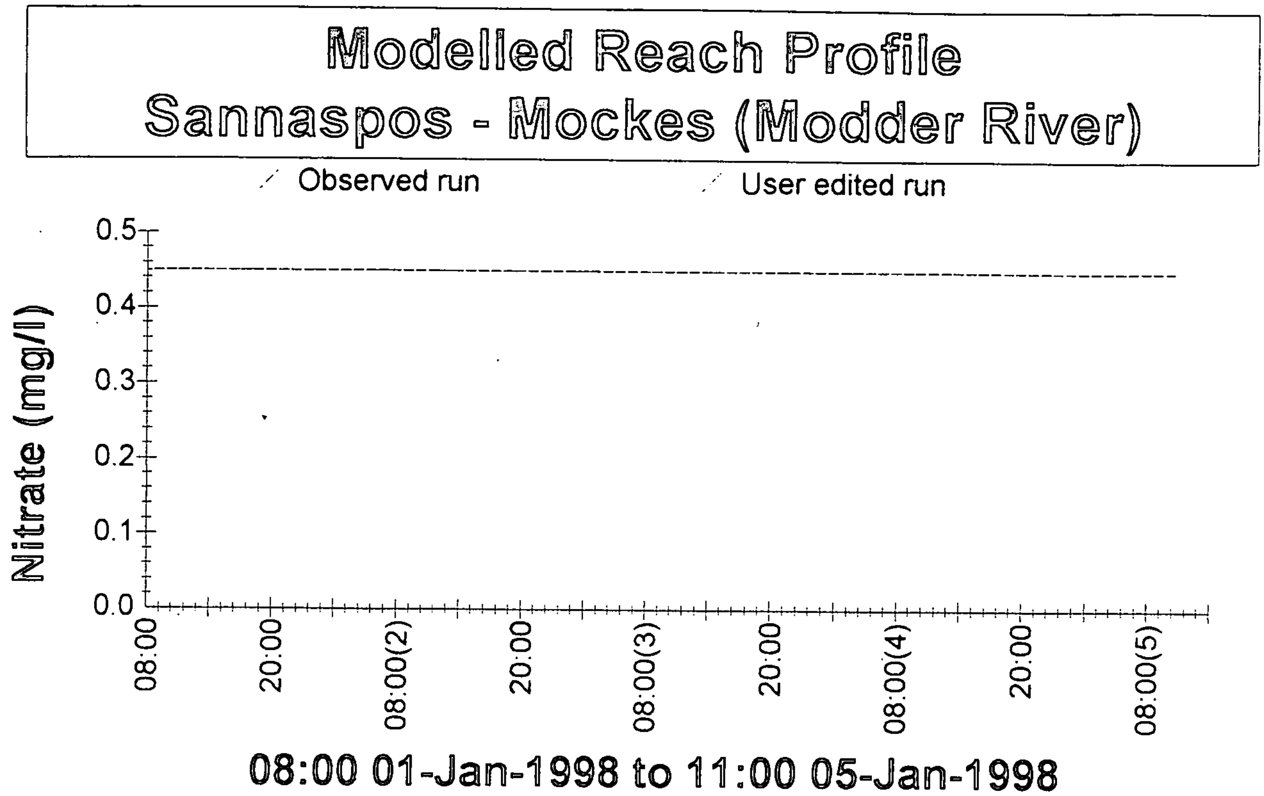


Figure 5.13: Variation in nitrate concentration in the Modder River with an impulse created, e.g. a power failure at the wastewater treatment facility

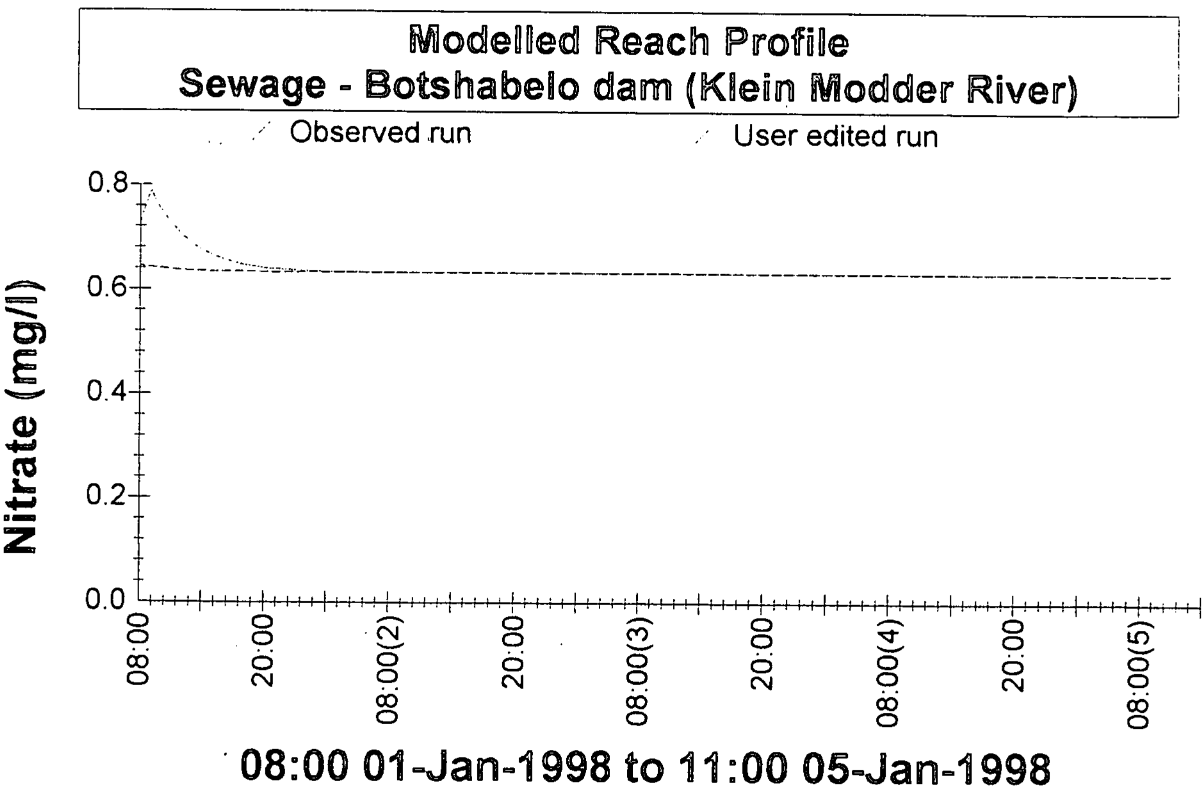


Figure 5.14: Variation in nitrate concentration in the Klein Modder River with an impulse created, e.g. a power failure at the wastewater treatment facility

5.6.4) Insertion of a weir at the Rustfontein/Sannaspos reach

Since the Modder River is an important source of potable water for Bloemfontein, there are many impoundments for water storage. Three of the impoundments (Rustfontein Dam, Mockes Dam, Mazelspoort Barrage) were in the upper part of the Modder River where the study was conducted. To test the sensitivity of the river when impounded, an extra weir was "hypothetically" inserted at the Rustfontein/Sannaspos reach to test its effects on the water quality and flow of the river. This "hypothetical" free-falling type weir with a height of 5 metres, was higher than the weir at both Mockes Dam and Mazelspoort Barrage. The user-edited simulation showed an increase of average 1.5 mg/l in the dissolved oxygen in the Rustfontein/Sannaspos reach (Figure 5.15). There was no change in the flow, nitrates, BOD, ammonia, temperature and *E. coli*. The dissolved oxygen concentration of the Sannaspos/Mockes Dam reach also increased with an average 0.6 mg/l. No change in dissolved oxygen was observed for the Mockes Dam/Mazelspoort reach. As the height of the weir was increased, the resulting dissolved oxygen concentrations also increased. The hypothetically inserted weir was higher than any of the weirs present in the study area at the moment. The effect of impoundment thus was not as significant as expected in terms of effects other than dissolved oxygen. In South African rivers, changes in for example conductivity, temperature and/or nutrient concentrations were observed downstream of an impoundment (Palmer & O'Keeffe, 1990a; Palmer & O'Keeffe, 1990b) and the model did not support this view.

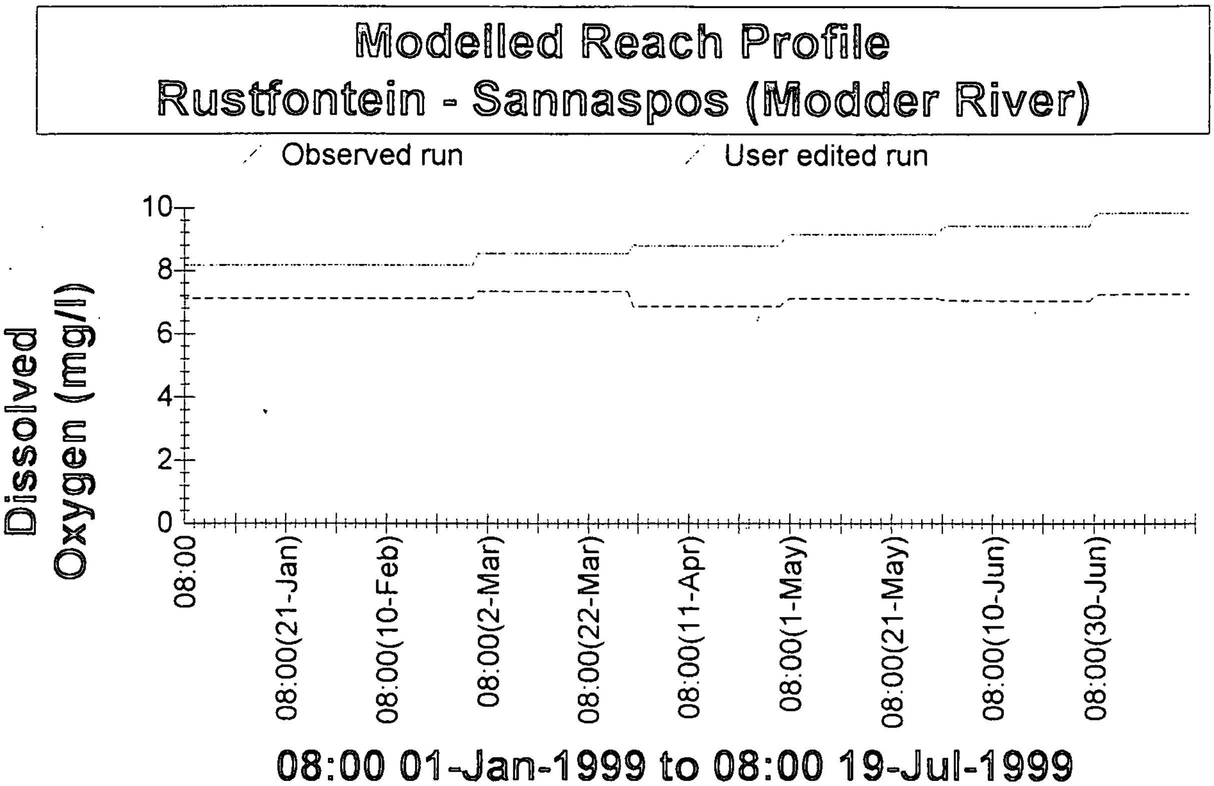


Figure 5.7: Increase in dissolved oxygen in the Modder River after insertion of a weir

5.7) CONCLUSION

Although PC-QUASAR is a user-friendly model, it was found to be unsuitable for modelling the water quality dynamics of the Modder River, since calibration of the model was not possible with the data obtained during this study. Changing of the rate coefficients within the range selected, did not give simulations similar to that of the observed values. The cause of this problem is probably twofold.

Firstly, the data derived from this study only made provision for the parameters simulated by PC-QUASAR. None of the reach rate coefficients were determined experimentally or otherwise. Jørgensen (1994) stated that if it is impossible to calibrate a model properly, this is not necessarily due to an incorrect model, but may be due to the poor quality of the data. The quality as well as the quantity of the data is thus crucial for calibration.

Secondly, several parameters of PC-QUASAR were found to be too insensitive for the objectives in mind. Turbidity, one of the most important factors influencing algal growth (by restricting light availability) in the Modder River (Chapter 3), is not incorporated in the model. Another major limitation of the model, is the absence of phosphorus as one of the parameters modelled. Globally, phosphorus is considered as the major source of eutrophication. Thus, a 1 mg/l P standard is enforced in "sensitive" areas (Chapter 1). Putting phosphorus values in the place of the "conservative" of PC-QUASAR, could partly solve the problem, but a conservative must not be changed biochemically. It will also then not be regarded as a plant nutrient.

Another unexpected, disappointing feature of the model was the limited effect of increased chlorophyll-*a* on the water quality of the Klein Modder and Modder River. The only parameters that showed change, were the dissolved oxygen concentration and BOD. Nitrates or ammonia and pH that were expected to decrease and increase respectively with an increase in algal growth, did not show any changes (not even at chlorophyll-*a* concentrations as high as 10 mg/l), since these parameters were not coupled with chlorophyll-*a*. The Modder River also showed an insensitivity to the effluent discharge into the Klein Modder River, contradictory to the observations made during the study.

Thus, PC-QUASAR is a model probably best used as a consent model, and in systems where turbidity is not an important factor, as is the case with the Modder River. The major limitations making it impossible to calibrate and verify this model with water quality data of the Modder River, were the lack of sufficient data and the insensitivity of model parameters considered as important during the field observations.

Some of the problems were discussed with the Institute of Hydrology, but no solution was reached yet.

REFERENCES

Allan. J.D. 1995. *Stream ecology. Structure and function of running waters.* Chapman and Hall. London. pp.388.

Allanson, B.R., Hart, R.C., O'Keeffe, J.H. & Robarts, R.D. 1990. *Inland waters of southern Africa. An ecological perspective.* Kluwer Academic Publishers. Dordrecht. pp.120-129.

Arnell, N.W. 1992. Hydrology and climate change. In: *The Rivers Handbook*, P. Calow & G.E Petts (eds.). Volume II. *Hydrological and Ecological Principles.* Blackwell Scientific Publications. Oxford. pp.173-186.

Awachie, J.B.E. 1981. Running water ecology in Africa. In: *Perspectives in running water ecology*, M.A. Lock & D.D. Williams (eds.). Plenum Press. New York. London. pp.339-367.

Babich, H. & Stotzky, G. 1980. Environmental factors that influence the toxicity of heavy metal and gaseous pollutants to micro-organisms. *Critical Reviews in Microbiology.* October 1980: 99-145.

Ball, R.C. & Bahr, T.G. 1975. Intensive survey: Red Cedar River, Michigan. In: *River ecology*, B.A. Whitton (ed.). Blackwell Scientific Publications. Oxford. pp. 431-461.

Bausch & Lomb. 1974. *Bausch and Lomb Analytical Systems Division.* Bausch & Lomb Inc. New York. pp.148

Beaumont, P. 1975. Hydrology. In: *River ecology,* B.A. Whitton (ed.). Blackwell Scientific Publications. Oxford. pp. 1-39.

Bowie, G.L., Mills, W.B., Porcella, D.B., Campbell, C.L., Pagenkopf, J.R., Rupp, G.L., Johnson, K.M., Chan, P.W.H., Gherini, S.A. & Chamberlin, C.E. 1985. *Rates, constants, and kinetics formulations in surface water quality modelling.* Second edition.

Environmental research laboratory. Office of Research and Development. U.S.E.P.A., Athens, Georgia. EPA/600/3-85/040. pp.455.

Bowling L.C. & Baker, P.D. 1996. Major cyanobacterial bloom in the Barwon-Darling River, Australia, in 1991, and underlying limnological conditions. *Mar. Freshwater. Res.* **47**: 643-657.

Burt, T.P. 1992. The hydrology of headwater catchments. In: *The Rivers Handbook*, P. Calow & G.E. Petts (eds.). Volume I. Blackwell Scientific Publications, Oxford, UK:3-29.

Calow, P. 1993. *Handbook of ecotoxicology.* Volume One. Blackwell Science Ltd. pp.478.

Carter-Lund, H. & Lund, J.W.G. 1995. *Freshwater algae - their microscopic world explored.* Biopress Ltd. pp. 360.

Christensen, E.R. & Nyholm, N. 1984. Ecotoxicological assays with algae: Weibull Dose-Response Curves. *Environ. Sci. Technol.* Vol **18**(9):713-718.

Connel, J.H. 1978. Diversity in tropical rain forests and coral reefs. *Science* **199**: 1302 - 1310.

Crockett, P. 1992. Water-Quality Modelling. In: *The Rivers Handbook,* P. Calow & G.E Petts (eds.). Volume II. *Hydrological and Ecological Principles.* Blackwell Scientific Publications. Oxford. pp. 213-226.

Cruywagen, L. 1998. Personal communication. University of the Orange Free State. Institute for Groundwaterstudies.

Cullen, J.J., Zhu, M., & Pierson, D.C. 1986. A technique to assess the harmful effects of sampling and containment for determination of primary production. *Limnol. Oceanogr.* **31**(6): 1364-1373.

Cummins, K.W. 1974. Structure and function of stream ecosystems. *Bioscience* **24**(11):631-641.

Cummins, K.W. 1977. From headwater streams to rivers. *The American Biological Teacher.* May 1977:305-311.

Cummins, K.W. 1979. The natural stream ecosystem. In: *The ecology of regulated rivers*, J.V. Ward & J.A. Stanford (eds.). Plenum Press. pp.7-25.

Cummins, K.W., Sedell, J.R., Swanson, F.J., Minshall, G.W., Fisher, S.G., Cushing, C.E., Petersen, R.C. & Vannote, R.L. 1983. Organic matter budgets for stream ecosystems: Problems in their evaluation. In: *Stream ecology: Application and testing of general ecological theory*, J.R. Barnes & G.W. Minshall (eds.). Plenum Press. New York and London. pp. 299-354.

Cushing, C.E., McIntire. C.D., Cummins, K.W., Minshall, G.W., Petersen, R.C., Sedell, J.R. & Vannote, R.L. 1983. Relationships among chemical, physical and biological indices along rivercontinua based on multivariate analyses. *Arch. Hydrobiol* **98**(3):317-326.

Dallas H.F. & Day J.A. 1993. The effect of water quality variables on riverine biota: a review. Draft Report to the Water Research Commission, Pretoria, 230 pp.

Davies, B.R, O'Keeffe, J.H. & Snaddon, C,D. 1993. A synthesis of the ecological functioning, conservation and management of South African river ecosystems. WRC Report No. TT 62/93. pp.232.

Davison, I.R. 1991. Environmental effects on algal photosynthesis. *J. Phycol.* **27**: 2-8.

Department of Water Affairs (DWA). 1986. Management of the water resources of the Republic of South Africa. CTP Book Printers, Cape Town. pp.467.

Department of Water Affairs and Forestry (DWAF). 1993a. *South African Water Quality Guidelines*. Volume 2. Recreational use. First edition. pp. 45-67.

Department of Water Affairs and Forestry (DWAF). 1993b. *South African Water Quality Guidelines*. Volume 1. Domestic use. First edition. pp. 93-117.

Department of Water Affairs and Forestry (DWAF). 1996a. *South African Water Quality Guidelines*. Volume 7. Aquatic ecosystems. First edition. pp.157.

Department of Water Affairs and Forestry (DWAF). 1996b. *South African Water Quality Guidelines*. Volume 2. Recreational Use. Second edition. pp.88.

Department of Water Affairs and Forestry (DWAF). 1996c. *South African Water Quality Guidelines*. Volume 1. Domestic Use. Second edition. pp.197.

Department of Water Affairs and Forestry (DWAF). 1996d. *South African Water Quality Guidelines*. Volume 5. Agricultural Use: Livestock watering. Second edition. pp.163.

Department of Water Affairs and Forestry (DWAF), Pretoria. 1997. Personal Communication. Orange River Historical Database.

Department of Water Affairs and Forestry (DWAF) & Water Research Commision (WRC). 1995. South African Water Quality Management Series. *Procedures to assess effluent discharge impacts*. First edition. pp.163-183.

Descy, J-P. 1993. Ecology of the phytoplankton of the River Moselle: Effects of disturbances on community structure and diversity. *Hydrobiologia* **249**:111-116.

De Wet, J.S. 1986. Fitoplanktonproduktiwiteit en faktore wat dit in Roodeplaatdam beinvloed. - Unpubl. Ph.D. Thesis, UOFS, Bloemfontein, RSA. pp. 234.

Dillon, P.J. & Rigler, F.H. 1974. The phosphorus-chlorophyll relationship in lakes. *Limnol. Oceanogr.* **19**(5): 767-773.

Dokulil, M.T. 1994. Environmental control of phytoplankton productivity in turbulent, turbid systems. *Hydrobiologia* **289**: 65-72.

Dörgeloh, W.G., Seaman, M.T. & Gaigher, I.G. 1993. The physical and chemical limnology of Sterkfontein Dam, Eastern Orange Free State, South Africa. *Water SA* **19**(3):177-184.

Du Plessis, H.M. & van Veelen, M. 1991. Water quality: salinization and eutrophication time series and trends in South Africa. *Suid Afrikaanse Tydskrif vir Wetenskap*. Vol **87**: 11-16.

Du Preez, J.C. 1978. The effect of temperature on the growth of *Acinetobacter calcoaceticus*. *Water SA* **4**(1):10-13.

Edwards, A.M.C. 1974. Silicon depletion in some Norfolk rivers. *Freshwat. Biol.* **4**:267-274.

Elliot, J.M. 1990. The need for long-term investigations in ecology and the contribution of the Freshwater Biological association. *Freshwater Biology* **23**:1-5.

Elwood, J.W., Newbold, J.D., Trimble, A.F. & Stark, R.W. 1981. The limiting role of phosphorus in a woodland stream ecosystem: Effects of P enrichment on leaf decomposition and primary producers. *Ecology* **62**(1): 146-158.

Falkowski, P. & Kiefer, D.A. 1985. Chlorophyll-*a* fluorescence in phytoplankton: relationship to photosynthesis and biomass. *Journal of Plankton Research* **7**(5):715-731.

Ferrar, A.A. 1989. Ecological flow requirements for South African rivers. South African National Scientific Programmes, Report no. 162. pp.118.

Gibbs, R.J. 1970. Mechanisms controlling world water chemistry. *Science* **17**: 1088-1090.

Giesy & Graney, R.L. 1989. Recent developments in and intercomparisons of acute and chronic bioassays and bioindicators. *Hydrobiologia* **188/189**: 21-60.

Golterman, H.L. 1975a. Chemistry. In: *River ecology,* B.A. Whitton (ed.). University of California Press. Berkeley. CA. p 39-81.

Golterman H.L. 1975b. *Developments in water science 2. Physiological limnology - An approach to the physiology of lake ecosystems.* Elsevier Scientific Publishing Company. Amsterdam - Oxford- New York. pp. 489.

Grobbelaar, J.U. 1985. Phytoplankton productivity in turbid waters. *Journal of Plankton Research.* Vol **7**(5): 653-663.

Grobbelaar, J.U. 1989. The contribution of phytoplankton productivity in turbid freshwaters to their trophic status. *Hydrobiologia* **173**: 127-133.

Grobbelaar, J.U. 1991. Primary production and water quality of the polluted Modder River, South Africa, before and after flooding. *Verh. Internat. Vertin. Limnol.* **24**: 1460-1463.

Grobbelaar, J.U. 1992. Nutrients *versus* physical factors in determining the primary productivity of waters with high inorganic turbidity. *Hydrobiologia* **238**: 177-182.

Grobbelaar, J.U. & House, W.A. 1995. Phosphorus as a limiting resource in inland waters, interactions with nitrogen. In: *Phosphorus in the Global Environment*, H. Tiessen (ed.). John Wiley and Son, Ltd. pp. 255 -273.

Grobler, D.C., Ashton. P.J., Mogane, B & Rooseboom, A. 1987. Assessment of the impact of low-cost, high density urban development at Botshabelo on water quality in the Modder River catchment. Report to the Department of Water Affairs, Pretoria, 47pp.

Grobler, D.C. & Toerien, D.F. 1986. The need to consider water quality in the planning of new urban development: a simulation study. *Water SA* **12**(1): 27-30.

Grobler, D.C & Davies, E. 1981. Sediments as a source of phosphate: A study of 38 impoundments. *Water SA* **7**(1):54-60.

Grove, A.T. 1972. The dissolved and solid load carried by some West African Rivers: Senegal, Niger, Benue and Shari. *Journal of Hydrology* **16**: 277-300.

Harris, G.P. 1986. *Phytoplankton ecology*. Structure, function and fluctuation. Chapman & Hall, London. 384pp.

Hewitt, G. 1991. River quality investigations, Part 2: Physical, chemical and microbial determinands of water quality. *Journal of Biological Education* **25**(3): 201-208.

Hernández-Ayón, J.M., Galindo-Bect, M.S., Flores-Báez, B.P. & Alvarez-Borrego, S. 1993. Nutrient concentrations are high in the turbid waters of the Colorado River Delta. *Estuarine, Coastal and Shell Science* **57**:595-602.

Holeman, J.N. 1968. The sediment yield of major rivers of the world. *Wat. Res. Research*. **4**(4): 737-747.

Horne, A.J. & Goldman, C.R. 1995. *Limnology*. McGraw Hill. 576pp.

House, H.A., Leach, D., Warwick, M.S., Whitton, B.A., Pattinson, S.N., Ryland, G., Pinder, A., Ingram,J., Lishman, J.P., Smith, S.M., Rigg, E. & Denison, F.H. 1997. Nutrient transport in the Humber Rivers. *The Science of the Total Environment* **194/195**: 303-320.

Hughes, D.A. & Van Ginkel, C. 1994. Nutrient loads from developing urban areas, a simulation approach and identification of information requirements. *Water SA* **20**(2):139-150.

Huntley, B.J. 1987. Ten years of co-operative ecological research in South Africa. *Sth. Afr. J. Sci.* **83**:72-79.

Hynes, H.B.N. 1970. *The Ecology of Running Waters.* Liverpool University Press. 555 pp.

Jagals, P. 1998. Personal communication. Free State Technikon. Department of Environmental Sciences.

Jagals, P. & Grabow, W.O.K. 1996. An evaluation of sorbitol-fermenting bifidobacteria as specific indicators of human faecal pollution of environmental water. *Water SA* **22**(3):235-238.

Jeffries, M. & Mills, D. 1990. *Freshwater ecology: principles and applications.* Belhaven Press. London and New York. 285pp.

Jørgensen, S.E. 1994. *Fundamentals of ecological modelling.* Second edition. Elsevier Publishers. pp. 4-106.

Jüttner,I., Rathfritz, H. & Ormerod, S.J. 1996. Diatoms as indicators of river quality in the Nepalese Middle Hills with consideration of the effects of habitat-specific sampling. *Freshwater Biology* **36**:475-486. Applied Issues.

Kenaga, E.E. & Moolenaar, R.J. 1979. Fish and *Daphnia* toxicity as surrogates for aquatic vascular plants. *Environmental Science and Technology* **13**(12): 1479-1480.

Keulder, P.C. 1979. Hydrochemistry of the upper Orange River catchment. *J. Limnol. Soc. Sth. Afr.* **5**(1):36-46.

Kilham, P. 1971. A hypothesis concerning silica and the freshwater planktonic diatoms. *Limnol. Oceanogr.* **16**(1): 10-18.

Kilham, P. 1990. Mechanisms controlling the chemical composition of lakes and rivers: Data from Africa. *Limnol. Oceanogr.* **35**(1): 80-83.

Koch, E., Cooper, D. & Coetzee, H. 1990. *Water, waste and wildlife - the politics of ecology in South Africa*. Penguin Forum series. Penguin Books. 63pp.

Köhler, J. 1994. Origin and succession of phytoplankton in a river-lake system (Spree, Germany). *Hydrobiologia* **289**:73-83.

Kozitskaya, V.N. 1990. Effect of illumination and temperature on algal growth: A survey. *Hydrobiological Journal* **25**(6): 53-67.

Krüger, G.H.J. 1978. The effect of physico-chemical factors on growth relevant to the mass culture of *Microcystis* under sterile conditions. Unpublished Ph.D Thesis. University of the Orange Free State. South Africa. 134pp.

Lewis, M.A. 1990. Are laboratory-derived toxicity data for freshwater algae worth the effort? *Environmental Toxicology and Chemistry* **9**: 1279-1284.

Lock, M.A. & John, P.A. 1979. The effect of flow patterns on uptake of phosphorus by river periphyton. *Limnol. Oceanogr.* **24**(2): 365-375.

Mason, C.F. 1991 *Biology of freshwater pollution*. Second edition. Longman Singapore Publishers. 356pp.

Meador, R.M. 1992. Interbasin Water Transfer: Ecological Concerns. *Fisheries* **17**(2):17-22.

Melack, J.M. 1995. Transport and transformations of P, fluvial and lacustrine ecosystems. In: *Phosphorus in the Global Environment*, H. Tiessen (ed.). John Wiley and Sons, Ltd. pp. 245 -254.

Merck Corporation. 1996. Coliforms show their true colours. CAT. No. 1.10426 E. Merck 64271. Darmstadt. Germany.

Meyer, J.L. 1979. The role of sediments and bryophytes in phosphorus dynamics in a headwater stream ecosystem. *Limnol. Oceanogr.* **24**(2) 365-375.

Minshall, G.W. 1978. Autotrophy in stream ecosystems. *Bioscience* **28**(12): 767-771.

Minshall, G.W., Petersen, R.C., Cummins, K.W., Bott, T.L., Sedell, J.R., Cushing, C.E. & Vannote, R.L. 1983. Interbiome comparison of stream ecosystem dynamics. *Ecol. Monogr.* **53**(1):1-25.

Munawar, M., Munawar, I.F., & Leppard, G.G. 1989. Early warning assays: an overview of toxicity testing with phytoplankton in the North American Great Lakes. *Hydrobiologia* **188/189**: 237-246.

National Water Act. No. 36 of 1998. Published in *Government Gazette* No. 19182, 26 August 1998.

Newbold, J.D. 1992. Cycles and spirals of nutrients. In: *The Rivers handbook*, P. Calow & G.E. Petts (eds.). Volume I. Blackwell Scientific Publications, Oxford, UK. pp. 371-379.

Nyholm, N & Kallqvist, T. 1989. Methods for growth inhibition toxicity tests with freshwater algae. *Environmental Toxicology and Chemistry* **8**: 689-703.

Nyholm,N., Sørensen, P.S., Kusk, K.O., & Christensen, E.R. 1992. Statistical treatment of data from microbial toxicity tests. *Environ. Toxicol. and Chem.* **11**: 157-167.

Palmer, R.W. & O'Keeffe, J.H. 1990a. Downstream effects of a small impoundment on a turbid river. *Arch. Hydrobiol.* **119**(4):457-473.

Palmer, R.W. & O'Keeffe, J.H. 1990b. Downstream effects of impoundments on the water chemistry of the Buffalo river (Eastern Cape), South Africa. *Hydrobiologia* **202**: 71-83.

Pegram, G.C., Görgens, A.H.M. & Otterman, A.B. 1997. A framework for addressing the information needs of catchment water quality management. *Water SA* **23**:13-20.

Petrie, C.J.S. 1978. The Modelling of Engineering systems - Mathematical and Computational Techniques. In: *Mathematical Models in Water Pollution Control*, A. James (ed.). John Wiley & Sons Ltd. Great Britain. pp.39-64.

Petts, G.E. 1992. Rivers: Dynamic components of catchment ecosystems. In: *The Rivers Handbook* (eds.) P.Calow & G.E. Petts. Volume II. Blackwell Scientific Publications, Oxford, UK. pp. 3-23.

Pieterse, A.J.H. 1987. Observations on temporal trends in phytoplankton diversity in the Vaal River at Balkfontein, South Africa. *J. Limnol. Sth. Afr.* **13**(1):1-6.

Pieterse, A.J.H & Van Zyl, J.M. 1988. Observations on the relation between phytoplankton diversity and environmental factors in the Vaal River at Balkfontein, South Africa. *Hydrobiologia* **169**:199-207.

Pretorius, C.J. & Viljoen, M.F. 1997. Projektering van waterbehoeftes in stedelike gebiede (Deel 4): Struktuuranalise as makro-projeksiemodel. *Water SA* **23**(1): 31-44.

Porcella, D.B., Grau, P., Huang, C.H., Radimsky, J., Toerien, D.F. & Pearson, E.A. 1970. Provisional Algal Assay Procedures. First Annual Report. Sanitary Engineering Research Laboratory. College of Engineering and School of Public Health. Berkeley. 180pp.

Quick, A.J.R. & Johanssen, A.R. 1992. User assessment survey of a shallow freshwater lake, Zeekoeivlei, Cape Town, with particular emphasis on water quality. *Water SA* **18**(4): 247-254.

Rhee, G-Y. & Gotham, I.J. 1980. Optimum N:P ratios and coexistence of planktonic algae. *J. Phycol.* **16**:486-489.

Robarts, R.D. 1984. Factors controlling primary production in a hypertrophic lake (Hartbeespoort Dam, South Africa). *J. Plankton Res.* **6**:91-105.

Roos, J.C. 1991. Primary productivity of the Vaal River phytoplankton. Unpublished Ph.D. Thesis. University of the Orange Free State. South Africa. pp. 265.

Roos, J.C. & Pieterse, A.J.H. 1995a. Nutrients, dissolved gases and pH in the Vaal River at Balkfontein, South Africa. *Arch. Hydrobiol.* **133**(2):173-196.

Roos, J.C. & Pieterse, A.J.H. 1995b. Salinity and dissolved substances in the Vaal river at Balkfontein, South Africa. *Hydrobiologia* **306**: 41-51.

Roos, J.C. & Pieterse, A.J.H. 1996. Seasonal variation of phytoplankton biomass in the Middle Vaal River, South Africa. *Water SA* **22**:33-42.

Rooseboom, A. 1978. Sedimentafvoer in Suider- Afrikaanse Riviere. *Water SA* **4**(1): 14-17.

Roux, D.J. 1994. The use of toxicity testing in effluent regulation and control: a new dimension to assessment and management of water quality in South Africa. *Sth. Afr. J. aquat. Sci.* **20**(1/2): 62-78.

Roux, D.J., Kempster, P.L., Truter, E. & van der Merwe, L. 1993. Effect of cadmium and copper on survival and reproduction of *Daphnia pulex*. *Water SA* **19**(4): 269-274.

Sakamoto, M. 1966. Primary production by phytoplankton community in some Japanese lakes and its dependence on lake depth. *Arch. Hydrobiol.* **62**(1): 1-28.

Sartory, D.P. & Grobbelaar, J.U. 1984. Extraction of chlorophyll-*a* from freshwater phytoplankton for spectrophotometric analysis. *Hydrobiologia* **114**: 177-187.

Søballe, D.M. & Kimmel, B.L. 1989. A large-scale comparison of factors influencing phytoplankton abundance in rivers, lakes and impoundments. *Ecology* **68**(6):1943-1954.

Smith, V.H. 1982. The nitrogen and phosphorus dependence of algal biomass in lakes: An empirical and theoretical analysis. *Limnol. Oceanogr.* **27**(6):1101-1112

STANDARD METHODS. American Public Health Association. American Water Works Association. Water Environment Federation. *Standard methods for the examination of water and waste water.* 1995. Nineteenth edition. American Public Health Association. Washington. pp. 4.112-4.113 & 4.118-4.120.

Stanford, J.A. & Ward, J.V. 1979. Dammed rivers of the world: symposium rationale. In: *The ecology of regulated streams*, J.V. Ward & J.A. Stanford (eds.). Plenum Press. pp. 1-7

Steinberg, C.E.W. & Hartmann, H.M. 1988. Planktonic bloom-forming Cyanobacteria and the eutrophication of lakes and rivers. *Freshwater Biology* **20**:279-287.

Taylor, R., Best, H.J. & Wiechers, H.N.S. 1984. The effluent phosphate standard in perspective: Part 1. Impact, control and management of eutrophication. IMIESA **9**:43-56.

Talling. J.F. 1986. The seasonality of phytoplankton in African lakes. *Hydrobiologia* **138**:139-160.

Thompson, P.A. & Hosja, W. 1996. Nutrient limitation of phytoplankton in the Upper Swan River Estuary, Western Australia. *Mar. Freshwater. Res.* **47**: 659-667.

Toerien, D.F., Barnard, J.H. & Pieterse. A.J.H. 1983. Waterkwaliteit in die Modderrivier opvanggebied: 'n sintese. Report: University of the Free State, Bloemfontein, RSA. 120pp.

Toerien, D.F., Hyman, K.L. & Bruwer, M.J. 1975. A preliminary trophic status classification of some South African impoundments. *Water SA* **1**: 15-23.

Trimble, S.W. 1975. Denudation studies: Can we assume stream steady state? *Science* **188**(20):1207-1208.

Truter, E. 1990. Toxicity assessment using *Daphnia pulex* as test organism. DWA. Hydrological Research Institute. 11pp.

Uys, M.C., Goetsch, P-A. & O'Keeffe, J.H. 1996. National Biomonitoring Programme for Riverine Ecosystems: Ecological Indicators: A review and recommendations. NBP Report Series no.4. Institute for Water Quality Studies, Department of Water Affairs and Forestry. Pretoria, South Africa. 93pp.

Vaithiyanathan, P., Ramanathan Al & Subramanian, V. 1992. Sediment transport in the Cauvery River basin: sediment characteristics and controlling factors. *Journal of Hydrology* **139**: 197-210.

Van der Heever, J.A. & Grobbelaar, J.U. 1996. The use of *Selenastrum capricornutum* growth potential as measure of toxicity of a few selected compounds. *Water SA* **22**(2): 183-191.

Van der Heever, J.A. & Grobbelaar, J.U. 1998. *In vivo* chlorophyll-*a* fluorescence of *Selenastrum capricornutum* as a screening bioassay in toxicity studies. *Arch. Environ. Contam. Toxicol.* **35**:281-286

Van der Merwe, W. & Grobler, D.C. 1990. Water quality management in the RSA. Preparing for the future. *Water SA* **16**(1):49-53.

Vannote, R.L. Minshall, G.W., Cummins. K.W., Sedell, J.R., and Cushing, C.E. 1980. The River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Sciences* **37**: 130-137.

Vincent, W.F. 1980. Mechanisms of rapid photosynthetic adaptation in natural phytoplankton communities II. Changes in photochemical capacity as measured by DCMU-induced chlorophyll fluorescence. *J. Phycol.* **16**: 568-577.

Vollenweider, R.A. 1975. Input-output models (with special reference to the phosphorus loading concept in limnology). *Schweiz. Zeitschrift fur Hydrologie* **37**:53-84.

Vollenweider, R.A. 1976. Advances in defining critical loading levels for phosphorus in lake eutrophication. *Mem. Ist. Ital. Idrobiol:* **33**:53-83.

Vollenweider, R.A. 1981. Eutrophication - a global problem. *Water Quality Bulletin* **6**: 59-89.

Walling, D.E. 1980. Water in the catchment ecosystem. In: *Water quality in the catchment ecosystem,* A.M. Gower (ed.). John Wiley & Son. Ltd pp.1-49.

Walling D.E. & Webb, B.W. 1992. Water quality I. Physical characteristics. In: *The Rivers handbook*, P. Calow & G.E. Petts (eds.). Volume I. Blackwell Scientific Publications, Oxford, UK. pp. 48 - 73.

Walmsley, R.D. 1984. A chlorophyll-*a* trophic status classification system for South African impoundments. *J. Environ. Qual.* **13**(1):97-104.

Walmsley, R.D. & Bruwer, C.A. 1980. Water transparency characteristics of South African impoundments. *J. Limnol. Soc. Sth. Afr.* **6**(2):69-76.

Walmsley, R.D. & Davies, B.R. 1991. An overview of water for environmental management. *Water SA* **17**(1):67-76.

Ward, J.W. & Stanford, J.A. 1983. Insect species diversity as a function of environmental variability and disturbance in stream systems. In: *Stream ecology: Application and testing of general ecological theory* (eds.) J.R. Barnes & G.W. Minshall. Plenum Press. New York and London. pp. 265-278.

Webb, B.W. & Walling, D.E. 1992. *Water quality II. Chemical characteristics*. In: *The Rivers Handbook*, P. Calow, & G.E. Petts (eds.). Volume I. Blackwell Scientific Publications, Oxford, UK:73-100.

Webster, J.R., Gurtz, M.E., Hains, J.J., Meyer, J.L., Swank, W.T., Waide, J.B., Wallace, J.B. & Wallace, J.B. 1983. Stability of stream ecosystems. In: *Stream ecology:*

Application and testing of general ecological theory, J.R. Barnes & G.W. Minshall (eds.). Plenum Press. New York and London. pp. 355-396.

Webster J.R. & Patten, B.C. 1979. Effects of watershed perturbation on stream potassium and calcium dynamics. *Ecol. Monogr.* **49**(1) :51-72.

Westlake, D.F. 1975 Macrophytes. In: *River ecology* (eds.) B.A. Whitton. University of California Press, Berkeley. Ca. pp. 106-129.

Wetzel, R.G. 1983. *Limnology*. W.B. Saunders Company. 743 pp.

Williams, R.J. 1998. Institute of Hydrology, Wallingford, OXON.

Wimberley, F.R. & Coleman, T.J. 1993. The effect of different urban development types on stormwater run-off quality: A comparison between two Johannesburg catchments. *Water SA* **19**(4):325-330.

SUMMARY

The Modder River is a relatively small river which drains an area of 7 960 km^2, in the central region of the Free State Province, South Africa and has a mean annual runoff of $184 \times 10^6 m^3$. Botshabelo is a city which was developed in the catchment area of the river and its sewage outflows are discharged into the Klein Modder River, a tributary of the Modder River. This study was conducted in order to determine seasonal and spatial patterns in the system, the influence of Botshabelo's sewage outflow on the water quality of the river, as well as the presence of any toxic compounds. It was determined that the Modder and Klein Modder Rivers do not follow distinctive seasonal patterns in terms of chemical parameters, however, NO_3-N and PO_4-P concentrations usually increased with increasing flow in the river. Physical parameters such as turbidity, flow and temperature followed distinctive seasonal patterns. Turbidity and flow was high during the rainy season and temperature followed the air temperature.

The Modder River is a very turbid system, influenced by physical (flow, turbidity and temperature) as well as chemical factors (high nutrient availability). The Modder River showed similarities with other South African rivers, such as the Vaal and Orange Rivers, in terms of turbidity/conductivity relationships and the range of physical and chemical parameters.

Phytoplankton growth also showed distinctive seasonal patterns, with low chlorophyll-*a* concentrations in the winter and higher chlorophyll-*a* concentrations when temperatures became more favourable. Diatoms (especially *Cyclotella* sp., *Stephanodiscus* sp. and *Nitzchia* sp.) dominated the algal community in both the Klein Modder and Modder Rivers for most of the time, with the euglenophyte, *Trachelomonas* dominating occasionally. In the Klein Modder River, algal blooms occurred more frequently, as well as with higher concentrations than in the Modder River. This could be ascribed to the higher nutrient concentrations in the Klein Modder River, which, together with low flow conditions, provide favourable conditions for algal growth.

There were periods when the nutrient concentrations in the waters of the Modder and Klein Modder Rivers were low, however, Botshabelo has an enrichment effect on the water quality, in terms of the nutrient concentrations. The inflow of the Klein Modder River into the Modder River caused on average, a 112 % increase in PO_4-P, a 171 % increase in NO_3-N nitrates and a 50 % increase in chlorophyll-*a* concentration. However,

the Modder River showed a self-purification capacity and nutrient concentrations decreased significantly downstream to Mazelspoort, restoring the water to almost the quality of the "unpolluted" reference point.

Based on toxicity tests performed with *Selenastrum capricornutum* and *Daphnia pulex*, no high concentrations of potentially toxic compounds were found in either the Klein Modder or Modder River. However, the occasional presence of heavy metals can not be excluded. Bacteria concentrations were high in both rivers and may pose a threat to human and animal health.

The use of a water quality model (PC-QUASAR) on the Modder River system, showed no results to predict the conditions in the rivers and for planning and management purposes, since the model could not be calibrated with the available data. The parameters of the model also showed great insensitivities regarding manipulation of important parameters. Because the Modder River is a very turbid system, it is also important that light availability be taken into account in any forecasting procedures.

OPSOMMING

Die Modderrivier (met 'n afvloei van 184 x 10^6 m^3 per jaar) is 'n redelike klein rivier wat 'n area van 7 960 km^2 in die sentrale deel van die Vrystaat, Suid-Afrika dreineer. Botshabelo is 'n stad wat ontwikkel is in die opvanggebied van die Modderrivier en die riooluitvloeisel van die stad vloei in die Klein Modderrivier ('n sytak van die Modderrivier) in. Hierdie studie is gedoen om die seisoenale en ruimtelike veranderinge in die sisteem te bepaal. Die invloed van Botshabelo op die rivierstelsel is ook ondersoek, sowel as die teenwoordigheid van moontlike toksiese stowwe wat in die water mag wees. Die resultate het getoon dat die Modder- en Klein Modderriviere geen seisoenale patrone volg in terme van chemiese parameters nie, alhoewel die NO_3-N en PO_4-P konsentrasies gewoonlik toegeneem het met hoër reënval en dus hoër vloei. Fisiese parameters soos troebelheid, vloei en temperatuur het duidelike seisoenale patrone gevolg. Die troebelheid en vloei was hoog gedurende die reënseisoen en die temperatuur het die wisselinge in lugtemperatuur gevolg.

Die Modderrivier is 'n baie troebel stelsel, en word beïnvloed deur beide fisiese (soos vloei, troebelheid en temperatuur) en chemiese faktore (hoë voedingstofkonsentrasies). Die Modderrivier toon egter ooreenkomste met ander Suid-Afrikaanse rivierstelsels, soos die Vaal en Oranje, in terme van die troebelheid/geleiding verhouding asook die fisiese en chemiese parameters.

Fitoplanktongroei het ook duidelike seisoenale patrone getoon, met lae chlorofil-*a* konsentrasies in die winter en hoër chlorofil-*a* konsentrasies in die somer, wanneer temperature meer gunstig geraak het. Diatome (veral *Cyclotella* sp., *Stephanodiscus* sp. en *Nitzchia* sp.) het die alggemeenskap in beide die Modder- en Klein Modderrivier gedomineer. *Trachelomonas* ('n euglenofiet) was somtyds dominant. Daar het meer algopbloeie voorgekom in die Klein Modderrivier as in die Modderrivier. Die konsentrasies van hierdie opbloeie was ook hoër. Dit kan moontlik toegeskryf word aan die hoë voedingstofkonsentrasies in die Klein Modderrivier, wat, saam met lae vloei, gunstige toestande skep vir alggroei.

Daar is tye wanneer die voedingstofkonsentrasies in die Modder- en Klein Modderrivier laag is, maar Botshabelo het 'n nadelige effek op die water van beide riviere, in terme van die voedingstofkonsentrasies. Die invloei van die Klein Modderrivier na die Modderrivier het gemiddeld die volgende verhogings in voedingstof- en

chlorofilkonsentrasies in die Modderrivier tot gevolg gehad: 'n 112 % verhoging in PO_4-P, 'n 171 % verhoging in NO_3-N en 'n 50 % verhoging in chlorofil-*a*. Die Modderrivier het egter 'n self-reinigingsvermoë en die voedingstofkonsentrasies het stroom-af verminder.

Gebaseer op die resultate van die toksisiteitstoetse met *Selenastrum capricornutum* en *Daphnia pulex*, was daar geen hoë konsentrasies van moontlik toksiese verbindings teenwoordig in enige van die twee riviere nie. Swaar metale kan egter van tyd tot tyd voorkom. Die konsentrasie van bakterieë was hoog in beide riviere en kan gesondheidsgevare vir mense en diere inhou.

Toets van 'n watergehalte model (PC-QUASAR) op die Modderrivier, het geen aanvaarbare resultate gelewer om die toestande in die rivier te voorspel en om bestuursbesluite te neem nie. Die model kon nie gekalibreer word met die beskikbare data nie en sommige belangrike parameters was heeltemal onsensitief vir manipulasie. Omdat die Modderrivier egter 'n baie troebel stelsel is, is dit belangrik om die beskikbaarheid van lig ook in ag te neem voor enige voorspellings gemaak word.