AN INVESTIGATION INTO PHYTOPLANKTON BLOOMS IN THE VAAL RIVER AND THE ENVIRONMENTAL VARIABLES RESPONSIBLE FOR THEIR DEVELOPMENT AND DECLINE

Final Report to the

Water Research Commission

by

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WRC Report No 359/1/97

ISBN 1 86845 330 8

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AN INVESTIGATION INTO PHYTOPLANKTON BLOOMS IN THE VAAL RIVER AND THE ENVIRONMENTAL VARIABLES RESPONSIBLE FOR THEIR DEVELOPMENT AND DECLINE

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When compared with similar studies on rivers in other regions of the world, it is clear that the Vaal River, although unique in many ways, also have characteristics in common with the other rivers. An investigation on the phytoplankton assemblages as well as the associated environmental variables and metabolic and other processes in the Vaal River, however, broadens our understanding of rivers in general. For this reason this particular study on the Vaal River is of general significance to all scientists interested in aquatic ecology and the utilisation and management of inland waters.

The Vaal River is the most important and the most regulated river in South Africa. It is also a eutrophic system on account of high chlorophyll-*a* and inorganic nitrogen and phosphorus concentrations as well as high primary productivity rates. Massive developments of phytoplankton are experienced in certain sections, resulting in aesthetic problems, health hazards, interferences with treatment processes and problems in water distribution systems. Consequently, the causes and consequences of the wax and wane of phytoplankton assemblages in the Vaal River were investigated.

The study showed that a variety of phytoplankton species (at least 124 species and varieties), occurred in the Vaal River during the study period. Green algae showed greatest species diversity. The greatest diversity of Cyanophyceae species were present at the Barrage, while the greatest diversity of Euglenophyceae species were present at the Balkfontein sampling locality. The phytoplankton community was dominated mainly by diatoms and green algae (which succeeded each other), as well as by blue-green algae during warmer periods.

Different species succession patterns were demonstrated for the different sampling positions. At the Barrage, Parys and Balkfontein, different species succeeded one another within relatively small time spans. At Stilfontein a smaller number of species succeeded one another, possibly indicating more stable environmental conditions at this locality in the river.

During the summer periods, diatoms tended to dominate in the Vaal River from January to August of each year, while the green algae were dominant from September to December. Dominance of diatoms during January to April and December, can be ascribed to blooms of *Melosira granulata*. Concentrations of unicellular centric diatoms were usually low during the summer periods, but they often dominated during the cold-water winter periods. Blue-green algae frequently occurred during the mid and late summer months of each year, especially at the Barrage, Stilfontein and Balkfontein sampling localities.

Water temperature also affects solubility of silica and oxygen. Higher silica concentrations were usually present during summer periods when the water temperatures were higher, while lower Si concentrations were present during the winter.

Indications of pollution and eutrophication were reflected in high chlorophyll-*a* concentrations in the Vaal River in comparison with other river systems. Periods of maximum chlorophyll-*a* concentration in the Vaal River occurred from January to March (summer) and again from July to November (winter-spring) of each year. Increased nutrient concentration in the Vaal River apparently increased the capacity of the water to support high production rates and to maintain large standing crops of phytoplankton.

Higher discharge resulted in higher total suspended solids (TSS) concentration, higher nutrient concentration (i.e. N, P and Si), lower total dissolved salts (TDS), higher turbidity and thus in lower euphotic zone (Z_{eu}) and underwater light climate (ULC). Very high levels of discharge can be responsible for a complete wash-out of the phytoplankton.

The turbidity of Vaal River water at Balkfontein was lower during the period 1984 to 1991 than during previous years. Reduced turbidity might be due to the clearing effect of salinity or possibly to the presence of water hyacinths. Water hyacinths could have served as sediment and plant nutrient traps. Since 1991, however, increases in tubidity were observed.

Decreased turbidity resulted in an improved under water light climate (ULC) which, in turn, was associated with reduced daily rates of areal photosynthesis during certain periods.

The total dissolved salts (TDS) concentration in the Vaal River is high, with an average annual maximum of about 650 mg l^{-1} and a minimum concentration of about 400 mg l^{-1}. The mean annual TDS concentration for the study period (1984 - 1993) was about 520 mg l^{-1} which is approximately four times higher than the global mean salinity of river water. An average increase rate of 25 mg l^{-1} per annum was calculated for the study period at both Stilfontein and Balkfontein. Based on the results of the present study, the Vaal River can be classified as a mixohaline system. The order of ionic prominence in the Vaal River at Balkfontein was $SO_4^{2-} >> Ca^{2+} \geq Cl^- \geq Na^+ \geq Mg^{2+}$ and at Stilfontein it was $SO_4^{2-} >> Cl \geq Ca^{2+} \geq Na^+ \geq Mg^{2+}$. The major ionic contributor to the total dissolved salts of the water was the anion SO_4^{2-}.

Salinity was not the primary variable influencing algal growth, but dinoflagellate representatives (responsible for red-tides in the ocean) more frequently occurred in high salinity water, while blue-green algae occurred in water with relatively low salinities.

Growth and carbon assimilation experiments indicated that different algae showed different sensitivities to TDS. Of the three algal species investigated, *Cyclotella meneghiniana* (a diatom) was the most sensitive and *Monoraphidium circinale* (a green alga) the least sensitive to increased dissolved salts. If the total dissolved salts concentration of Vaal River water increases with time, algal species that could persist over a wide range of salinities can be expected to be present most of the time. Of the species investigated, *Monoraphidium circinale* can be expected to become dominant under conditions of increased salinity. *Cyclotella meneghiniana* and *Microcystis aeruginosa* on the other hand, can be expected to be excluded from the water under conditions of increased salinity of 250 mg l^{-1} and above. The recent increased abundance of *Oscillatoria simplicissima* (another blue-green alga) in the Vaal River can possibly be attributed to temporary decreasing levels of salinity.

Reduced annual DIN:DIP ratios suggested that the Vaal River at Balkfontein switched from a generally phosphorus limited system (1986 to 1990) to a potentially nitrogen limited system (1991 and 1993). The decrease in DIN:DIP ratio was caused by relatively low DIN concentrations and relatively high DIP concentrations.

Switching inorganic nitrogen and phosphorus limitation influence algal biomass as well as the composition of the algal assemblages. The switching between Chlorophyceae (green algae) and Bacillariophyceae (diatoms) dominance is apparently partly determined by N:P ratios.

Blue-green algae were most probably favoured by low DIN:DIP and TN:TP ratios. A shift from blue-green algae to other algal groups occurred if the TN:TP ratio increased. Increased phosphorus concentrations in the Vaal River (with a resultant decrease in DIN:DIP and thus TN:TP to less than five) will probably cause a shift in the algal assemblages from diatom and green algal dominance to blue-green algal dominance. The Si:DIP Si:DIP ratio also seemed to be important in influencing the occurrence of diato diatoms usually occurred under conditions of high Si:DIP ratios.

Activities in Vaal River water of two enzymes (Nitrate Reductase or NR and Phosphatases or PASE) studied, appeared to be stimulated by high P levels. No definite correspondence was shown between NO_3-N and NR activities as well as between NO_3-N and PASE activity. In addition, both groups of enzymes appeared to be indirectly proportional to chlorophyll-*a*. NR activity appeared to be corresponding positively with PASE activity. Both enzymes were, therefore, apparently stimulated by similar environmental conditions. Both enzymes seemed to be activated by high PO_4-P concentrations in the Vaal River. These results could suggest that all metabolic activities in algal cells are activated by high PO_4-P concentrations or increased PO_4-P availability. The increased metabolic rates in algal cells could then result in faster division rates and consequently in the development of algal blooms.

All the above results support a conceptual model that assumes that phytoplankton growth in the Vaal River is controlled by fluxes of solar energy, but fluxes of plant nutrients apparently affects the flow of energy into the algal cells.

Highest silica concentrations were reported at the Barrage, whereafter a decrease occurred downstream to Parys, Stilfontein and Balkfontein. A possible explanation for the high Si concentration at the Barrage sampling locality is that the concentration of diatoms is lower at the Barrage than at the Parys, Stilfontein and Balkfontein sampling localities, thereby not removing significant amounts of silicon from the water. It seems as if Si concentration in the Vaal River is primarily determined by temperature and diatom uptake metabolism.

Statistical analysis of the water chemistry, temperature and chlorophyll concentration data showed that major variations in water properties can be ascribed to seasonally-related variations in temperature, suspended solids (turbidity), coupled with directionally opposite variations in dissolved salts (conductivity). Algal growth was greatest in winter when the water is clear and cold and lowest in summer when it is turbid and warm. Seasonal variations in the two forms of inorganic nitrogen (nitrate and ammonium) seemed to be different, with the one being associated with summer and the other with winter conditions.

The application of the principal component analysis (PCA) method to investigate associations between physical, chemical and biological variables resulted, amongst others, in the elucidation of seasonal aspects of algal growth, showing higher algal growth during colder months. Should changes in the seasonality occur, i.e. should summer blooms replace winter blooms or should different years show different associations, PCA analyses will be able to identify and quantify the changes and differences in terms of their statistical significance. However, the statistical comparison between phytoplankton and environmental changes is only the first step in linking species dynamics to changes in a complex environment.

A light-temperature dependent model to simulate algal blooms in the Vaal River was developed, but more information on the ecological behaviour of the river and as well as growth requirements of specific algae is essential. The model is based on two assumptions, namely that the water is eutrophic and that the growth and death of algae are dependent mainly on the available light and temperature of the water. Weekly averaged temperature and the total suspended solids concentration were considered as inputs. The calculated chlorophyll-*a* concentration values agreed fairly well with measured values in qualitative as well as quantitative terms during three years (1985 to 1987) of investigation. By taking into account the possible effects of dissolved silicon concentration, it was possible to improve the quality of simulation of algal growth and algal blooms in the Vaal River.

Two major fields of application for such a mathematical model are possible. Firstly the model can be used to support investigations by providing a consistent splitting of the global data between smaller algal groups assumed to be responsible for blooms, by assisting in the identification of important environmental variables concerning specific algae involved in a bloom, by assisting the researcher to describe the mechanisms involved in the development of a bloom by providing numerical information about experimental data that are not available, and finally by providing a way to test new hypotheses suggested by other investigations. Secondly, mathematical models of this nature can be used in water resource management in which case the model can be used as a planning tool.

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CHAPTER 1: INTRODUCTION AND OBJECTIVES

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Algae are common inhabitants of surface waters exposed to sunlight, where they often give rise to large quantities of organic material. In South Africa problems related to the overabundance of algae (i.e. blooms) in rivers are well recognised, but poorly understood.

The Vaal River is the most important and the most regulated river in South Africa. The Vaal River is also a eutrophic system on account of high chlorophyll-*a* and inorganic nitrogen and phosphorus concentrations as well as high primary productivity rates. Massive developments of phytoplankton are experienced in certain sections, resulting in aesthetic problems, health hazards, interferences with treatment processes and problems in water distribution systems. Attention should, therefore, be given to the causes and consequences of the wax and wane of phytoplankton assemblages and the effect of water transfer and stream regulation on algological and environmental variables in this river.

Our general approach in the development of the project on the ecology of Vaal River phytoplankton is idealistic, and strives to address key theoretical and practical issues. The actual progress of our research is entirely dependent on the availability of financial support as well as interested, enthusiastic and responsible researchers. The project is unique in the South African context, it is of local and international scientific significance and it is very important from a fundamental as well as an applied point of view.

The aims of the programme are

- to qualify and quantify phytoplankton populations and environmental variables and to relate phytoplankton variables to those of the physical and chemical environment in order to understand conditions responsible for the development and decline of specific algal blooms;
- to investigate the seasonal cycles of carbon assimilation and the intrinsic photosynthetic characteristics of the phytoplankton assemblages *in situ*, and also to relate these to environmental conditions;
- to isolate algal species into uni-algal culture and to perform experiments measuring the effect of various treatments on growth and other metabolic rates such as ^{14}C assimilation and enzyme activities; and
- to apply available, and possibly to develop new, mathematical models for algal behaviour and growth in the river for predictive and management purposes.

Approximately 250 algal species have been identified from the Vaal River. Similarities between Vaal River assemblages and that of maturation ponds in South Africa and the River Jordan and Lake Kinneret in Israel, were observed.

Phytoplankton growth in the Vaal River was shown in previous studies to be limited primarily by inorganic nitrogen and secondarily by inorganic phosphorus. Major blooms, i.e by *Micractinium, Stephanodiscus, Carteria* and *Chlamydomonas* species, occur annually between June and September, apparently triggered by increased nitrogen supply. Diversity in phytoplankton composition was related to discharge and discharge-derived variables such as sulfate, silicon, and inorganic nitrogen and phosphorus loading.

From literature it is known that nitrate is most probably limiting algal growth when the N:P ratio is < ca. 10, while phosphate is most probably limiting when the N:P ratio is > ca. 20. Where N:P ratios are between 10 and 20, not one of the two nutrient elements are limiting. The ratios of carbon, nitrogen, phosphorus and chlorophyll in algal cells show fluctuations which could be related to the nutrient conditions that influence the annual patterns of algal blooms. In the Vaal River it was shown that switching N and P limitation influence algal biomass as well as the composition of the algal assemblages. The switching between Chlorophyceae (green algae) and Bacillariophyceae (diatoms) dominance is apparently partly determined by N:P ratios. Different algal species and groups have different N and P requirements affecting their enzymatic activity. Varying enzymatic activities will thus occur in different algal groups and species. An investigation into activities of enzymes such as nitrate reductase and phosphatase will therefore give information about inorganic N and P dynamics in the Vaal River and should explain in more detail environmental control mechanisms exerted on algal biomass and composition by inorganic N and P availability.

All these aspects will serve as background information against which short and long-term changes in community structure due to mineralisation, eutrophication, turbidity, occasional floods and interbasin-transfer, will eventually be evaluated. In this regard an investigation into the application and development of mathematical models are extremely important, especially in relation to water resource management.

CHAPTER 2: PHYTOPLANKTON PRODUCTION AND PHOTOSYNTHETIC CHARACTERISTICS IN RELATION TO ENVIRONMENTAL VARIABLES IN THE VAAL RIVER AT BALKFONTEIN

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1. INTRODUCTION

The quality of middle Vaal River water is declining. The decline is primarily the result of salinisation and eutrophication (DWA, 1986). The enrichment of water by plant nutrients usually leads to excessive plant growth with associated water quality problems. High salinity and eutrophication levels in the Middle Vaal River is a matter of concern to the Department of Water Affairs and other institutions such as those responsible for water treatment.

Phytoplankton productivity has a direct effect on the structure and functioning of aquatic ecosystems because it forms the base of most food chains (Kirk & Akhurst, 1984). Quantitative information on primary production in the Vaal River is essential to understand ecosystem metabolism and the dynamics of phytoplankton populations (Sakamoto *et al.*, 1984).

In the present study phytoplankton biomass and environmental variables that influenced the production of the phytoplankton during 1993, will be emphasised.

2. MATERIAL AND METHODS

2.1 PRIMARY PRODUCTIVITY

In situ primary productivity (measured as photosynthetic carbon fixation) of natural phytoplankton assemblages was determined monthly at Balkfontein in the Vaal River. Photosynthetic rates, for the incubation period (10:00-14:00), were calculated using the standard ^{14}C light and dark bottle technique (Vollenweider, 1969).

Subsurface water samples were collected with a Van Dorn sampler, and 100 ml aliquots were added to round-bottom flasks (100 ml), followed by 1 ml (1 µCi) of $NaH^{14}CO_3$. One dark and two light bottles each were suspended from buoys in the middle of the river at the surface and at 0.125; 0.25; 0.5; 0.75; 1.0; 1.5 and 2.0 m depth intervals. After incubation (4 h), the flasks were placed in a dark box and taken to the river bank for immediate processing. Aliquots (4 ml) were acidified with 0.25 ml of 0.1 N HCl, air was bubbled through each sample for about 3 min. (Schindler *et al.*, 1972), and subsequently 10 ml of scintillation liquid (Insta Gel II) and 0.5 ml of Carbo-sorb were added. The activity in the samples and standards were determined with a Rackbeta liquid scintillation counter (Model 1217). Dark bottle values were subtracted from the ^{14}C-uptake rates in the light bottles.

Calculations of daily rates of photosynthesis (dP_d) were made according to Vollenweider's (1969) method. The initial slopes of the light-saturating curve, $Á^B$, were computed by regression analysis from the chlorophyll-specific photosynthetic rate and the limiting light intensity. Irradiance at onset of light saturation (I_k) was calculated from the relationship, $I_k = P^B{}_m / Á^B$.

2.2 ENVIRONMENTAL VARIABLES

In Situ dissolved oxygen (mg l^{-1}) and temperature (°C) were recorded at the surface and at every 0.5 m interval up to 3 m deep during the midday period (12:00 ± 1 h). A Yellow Springs YSI Model 54A oxygen/temperature meter was used. Surface pH was recorded with an Orion Research Model 231 digital pH meter.

Total inorganic carbon available for photosynthesis was calculated from the total alkalinity, pH and temperature values, using tables given by Wetzel & Likens (1979). Chlorophyll-*a* concentration was measured according to an ethanol extraction method modified from the method described by Lorenzen (1967).

Atmospheric and underwater light intensity (10 cm depth intervals) were measured approximately every hour with a Li-Cor light meter equipped with a scalar probe. The underwater light climate (ULC) was calculated from the total incoming irradiance divided by the extinction coefficient (k) of light in the water-column.

Turbidity was recorded with a Chemtrix Type 10 turbidimeter (1986-1991) and with a AL 1000 (AquaLytic) turbidimeter and quantitatively expressed in terms of Nephelometric turbidity units (NTU). Other chemical and flow data were obtained from the Department of Water Affairs and Forestry, Pretoria. Dissolved inorganic nitrogen (DIN) was calculated as ammonium nitrogen (NH_4-N) plus nitrate (NO_3-N) and nitrite nitrogen (NO_2-N). Total nitrogen (TN) as Kjeldahl nitrogen (organic N + NH_4-N) plus NO_3-N.

3. RESULTS AND DISCUSSION

3.1 DISCHARGE

Down-slope current is a salient feature of streams. Discharge and water velocity have been proved to be important variables influencing river phytoplankton (Petts, 1984). The study period was dominated by the summer floods in 1988 and 1989. The mean discharge during the study period (1986-1993) in the Vaal River ranged between 0.0 and 1 200 $m^3 s^{-1}$ (Fig. 1). The average annual discharge during 1990 to 1993 was fairly constant (20 ± 5 $m^3 s^{-1}$). The major impact of discharge on physical, chemical and biological factors will be discussed in the following section.

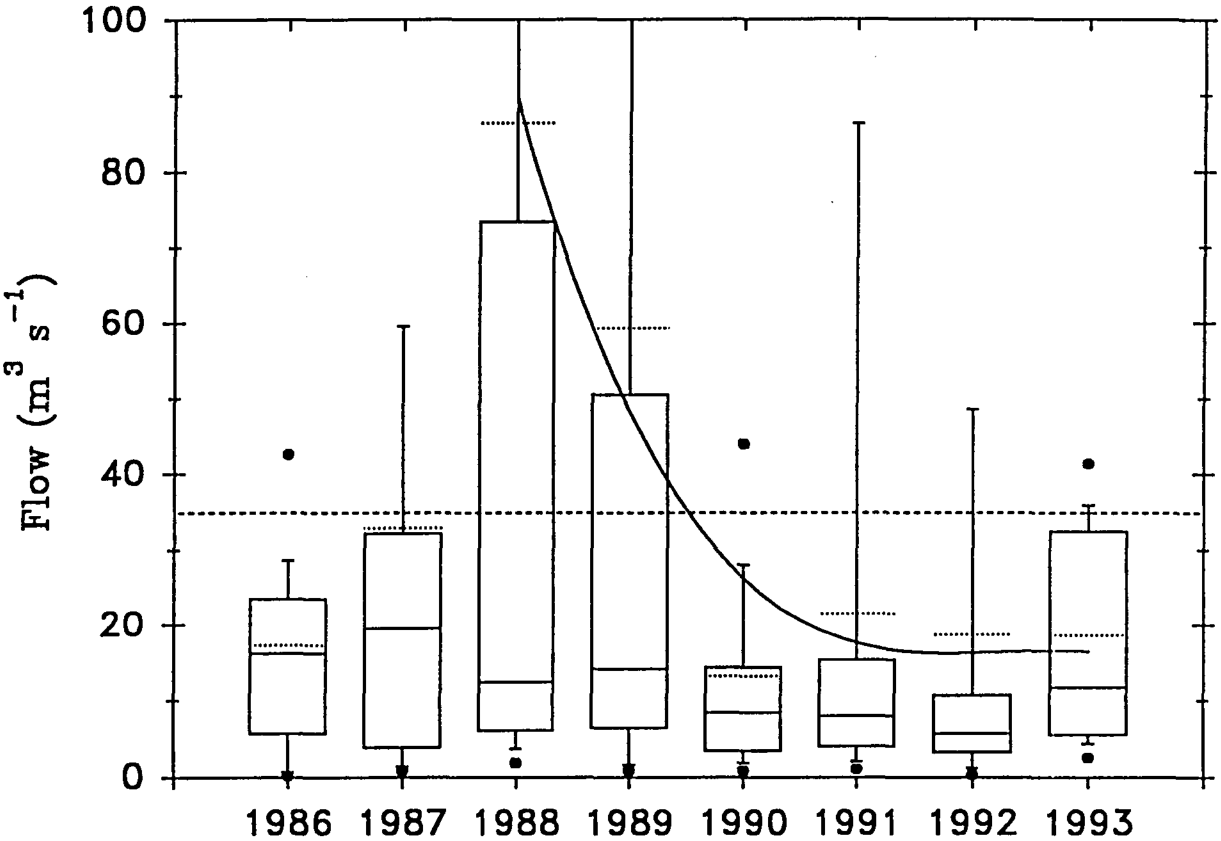


FIGURE 1: Box plot (data distribution) of annual variation of discharge (m^3 s^{-1}) in the Vaal River at Balkfontein. The box represents the 25th through 75th percentiles (i.e. 50 % of data in box). The solid line in the box represents the median and the dotted line the average value. The 5th and 95th percentile points are shown as solid circles. The horizontal dashed line represents the average for the entire study period (1986 - 1993). The increasing or decreasing trends are shown by a polynomial fit on average values (solid line).

3.2 DISSOLVED SOLIDS

A general increase in total dissolved solids (TDS) from 1986 to 1990 was calculated (trend analysis) to be on average about 30 mg l^{-1} a^{-1} (Fig. 2). However, the average annual TDS decreased since 1991 from 660 mg l^{-1} to 477 mg l^{-1} during 1993, which is lower than the desired upper limit of 600 mg l^{-1} for the Vaal River Barrage. During 1993 the TDS at Balkfontein ranged between 158 mg l^{-1} (November, summer, high flow) and 740 (June, winter, low flow) with an average of 477 mg l^{-1} (Fig. 2). TDS in the Vaal River tends to be higher during the dry season, i.e. winter months (Fig. 3).

A significant inverse correlation was demonstrated between discharge and TDS. The TDS reduction by high discharge in the Vaal River is in accordance with major river systems of the world (Hynes, 1970; Harris, 1986, Webb & Walling, 1992). In addition, high TDS was associated with clearer water, i.e. a deep euphotic zone, which could favour algal growth.

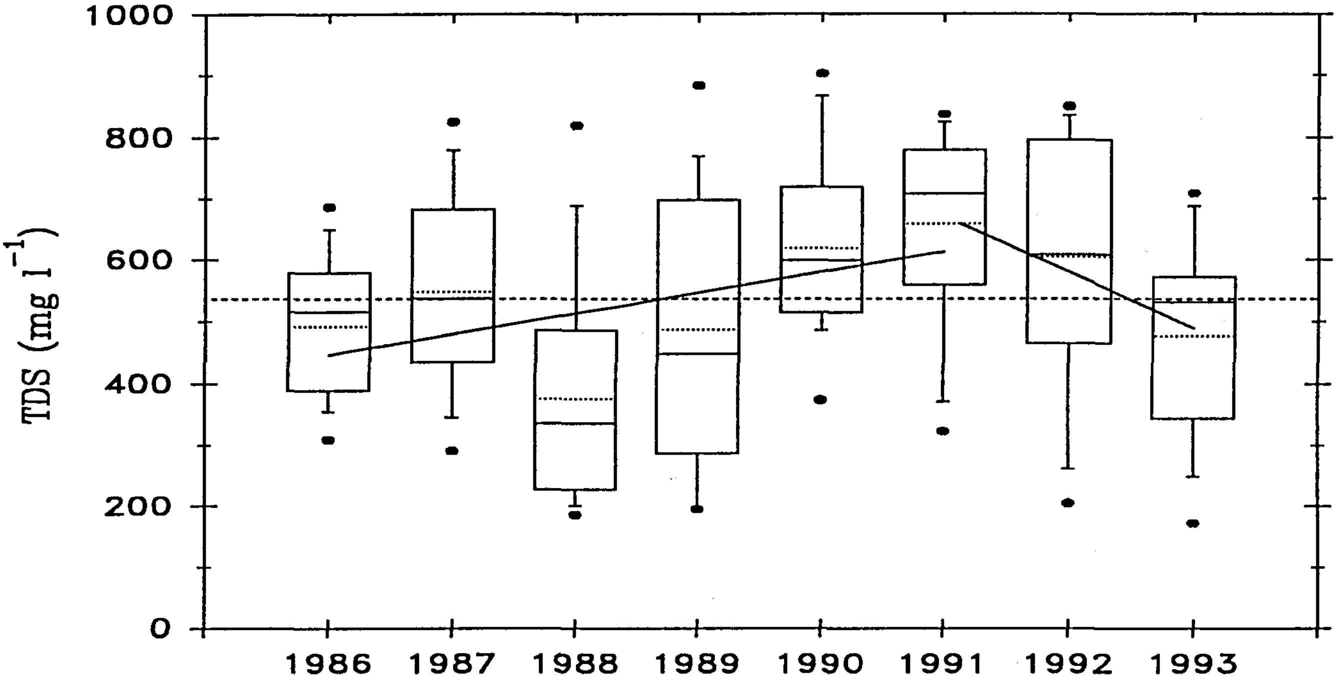


FIGURE 2: Box plot of annual total dissolved solids (TDS) in the Vaal River at Balkfontein.

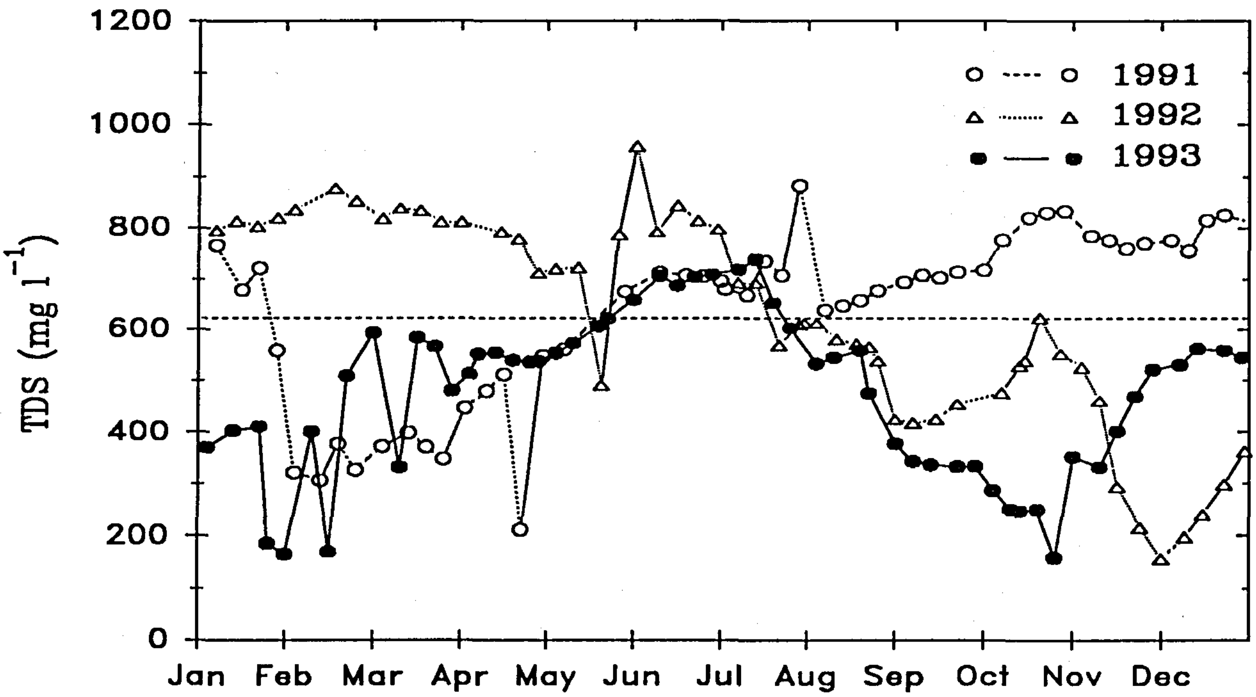


FIGURE 3: Annual variation of total dissolved solids (TDS) in the Vaal River at Balkfontein.

3.3 TURBIDITY AND EUPHOTIC ZONE DEPTH

Turbidity influences both the quantity and quality of light penetration in water and determines the depth of the euphotic zone (Z_{eu}) and the underwater light climate (ULC). Because light is a driving force in primary production, underwater light availability has a significant influence on phytoplankton production and biomass. Turbidity in the Vaal River is primarily a function of suspended sediment concentration which is correlated with discharge (Roos & Pieterse, 1994).

A significant clearing of the Vaal River water at Balkfontein was noticed since 1988 (Figs 4 & 5). This clearing of the water-column probably resulted in more favourable light conditions for photosynthesis and a possible build-up of phytoplankton biomass. However, clearer water was associated with lower nutrient concentrations (especially N, P and Si) that could inhibit phytoplankton growth (Fig. 9).

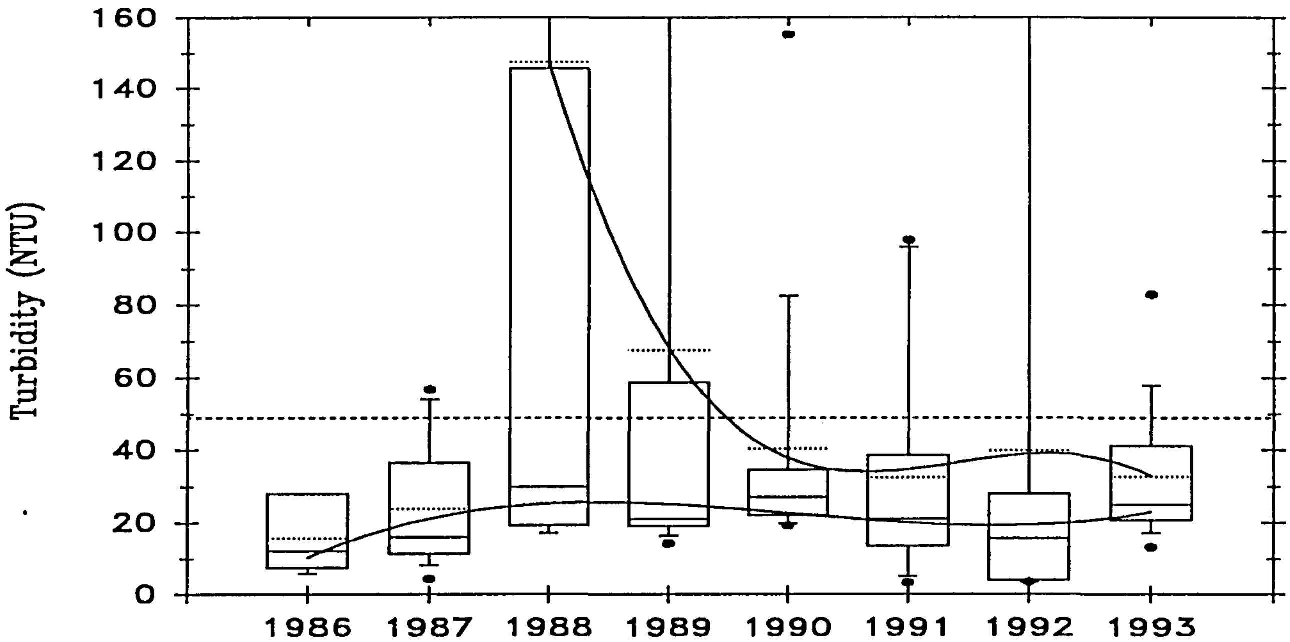


FIGURE 4: Box plot of annual turbidity in the Vaal River at Balkfontein. The high average turbidity for 1988 was caused by floods during February 1988.

Although the average turbidity during 1993 (33 NTU) was lower than during 1992 (40 NTU), the general turbidity level (median and box) during 1993 was higher than in 1992. During 1993, 50 % of the observed values (box) were in the 20 to 40 NTU range compared to the 5 to 25 NTU during 1992 (Fig. 4). The higher turbidity level during 1993 was also displayed by the lower euphotic zone (Fig. 5). The average Z_{eu}, for the study period, of almost 1.5 m, indicates a relatively deep photosynthetic active zone, i.e. a Z_{eu}:Z_m ratio of 0.25 which is lower than the optimum ratio of 0.7 but much higher than the 'critical mixing depth' ratio of 0.05 (Grobbelaar, 1990). The Z_{eu}:Z_m ratio was shown to be the most important factor affecting overall productivity and that nutrients are of secondary importance in turbid waters (Grobbelaar, 1992).

The year 1992 was characterised by relatively low turbidity from January to October (average 14 NTU), with high turbidity observed during November and December (average 172 NTU) occurring after early summer rainfall. The average turbidity of 40 NTU for 1992 was higher than during 1991 (33 NTU), but the average Z_{eu} increased from 1.7 m during 1991 to 1.9 m during 1992 (compare Figs 4 and 5). This apparent contradiction, i.e. higher turbidity associated with higher Z_{eu}, can be explained by the relationship between Z_{eu} and turbidity. The hyperbolic relationship illustrates the situation where relatively low turbidity was associated by high Z_{eu} (e.g. January to October), but little changes in Z_{eu} were noted after large changes in turbidity (e.g. November to December).

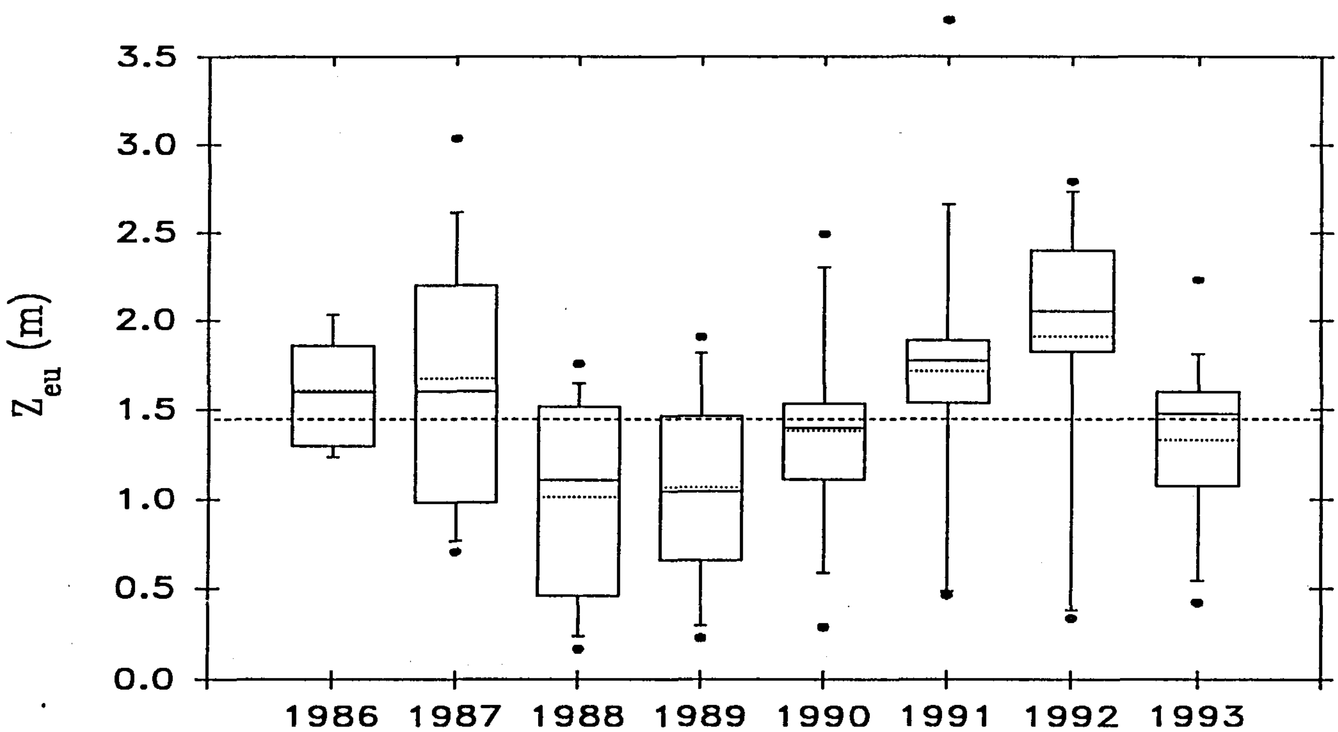


FIGURE 5: Box plot of annual euphotic zone depth (Z_{eu}) in the Vaal River at Balkfontein.

3.4 PHOSPHORUS AND NITROGEN

Because the supply and availability of inorganic phosphorus and nitrogen are causing shifts from less to more productive states in many rivers and lakes, these nutrients are considered important variables. Because diatoms were shown to be an important algal group in the Vaal River, silica is also considered an important nutrient.

The total nitrogen (TN) concentration (i.e. NO_3-N + NH_4-N + organic N) in the Vaal River at Balkfontein was high during 1988 and 1989 (average almost 2 000 $\mu g\ l^{-1}$) but showed a decreasing trend since 1989 (Fig. 6). However, a slight increase in the annual average was observed since 1991 (Fig. 6). TN concentration during 1993 ranged between 659 and 4 722 $\mu g\ l^{-1}$ with an average of 1 521 $\mu g\ l^{-1}$, which is comparable to eutrophic systems (Wetzel, 1983).

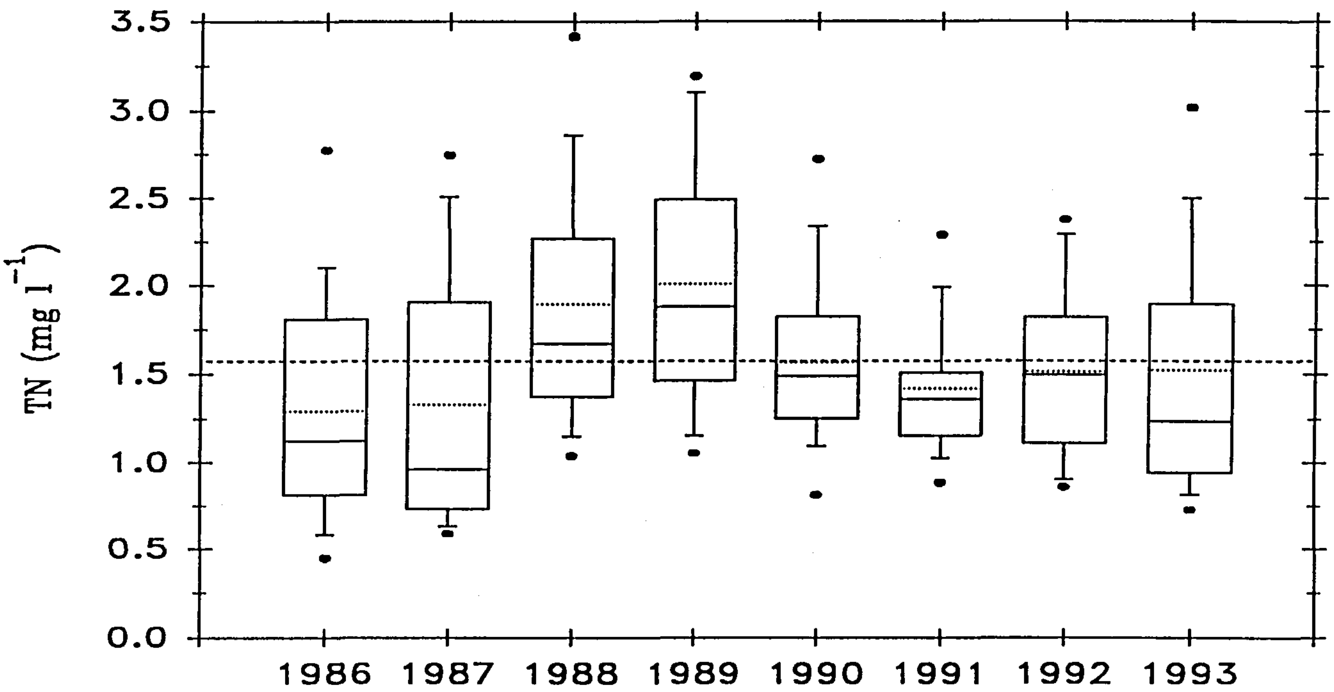


FIGURE 6: Box plot of annual total nitrogen (TN) in the Vaal River at Balkfontein.

The dissolved inorganic nitrogen (DIN) concentration (i.e. NO_3-N + NH_4-N) during 1993 ranged between 20 and 2 284 µg l^{-1} with the average occurring at 402 µg l^{-1} (Fig. 7). The average DIN concentrations also increased since 1991 from 244 to 402 µg l^{-1} during 1993 (Fig. 7). DIN and TN were significantly correlated (positive) with turbidity and discharge, based on annual averages and seasonal variations. Thus turbid waters were associated with higher nutrient concentrations (Fig. 7).

The total phosphorus (TP) concentration was relatively low in 1986 (average 122 µg l^{-1}) but increased to a maximum concentration during 1988 (average 266 µg l^{-1}), followed by a distinct decrease since 1988 to 1991. The average of 128 µg l^{-1} during 1991 was 52 % lower than the average of 266 µg l^{-1} during 1988. Since 1991 the average TP increased from 128 µg l^{-1} in 1991 to 177 µg l^{-1} 1993 (Fig. 7). A significant positive correlation was demonstrated between the annual average TP concentration and turbidity (Fig. 9) as well as discharge (compare Figs 1 & 8).

The dissolved inorganic phosphorus (DIP) concentration (i.e. equivalent to PO_4-P) ranged during 1993 between 12 and 294 µg l^{-1} with the average occurring at 60 µg l^{-1} which is much higher than the overall average of about 27 µg l^{-1} (Fig. 10). The proportion of DIP within TP was relatively low at about 34 % (on average) compared to eutrophied systems which is usually above 60 %. The low % DIP fraction in the Vaal River suggests a rapid uptake of phosphate by phosphorus deficient phytoplankton cells and vascular plants. However, the solubility of DIP in the Vaal River is also limited by high calcium concentrations (Roos, 1992). Thus the decreasing Ca concentration since 1991 could have contributed to the higher DIP concentrations (compare Figs 2 & 10).

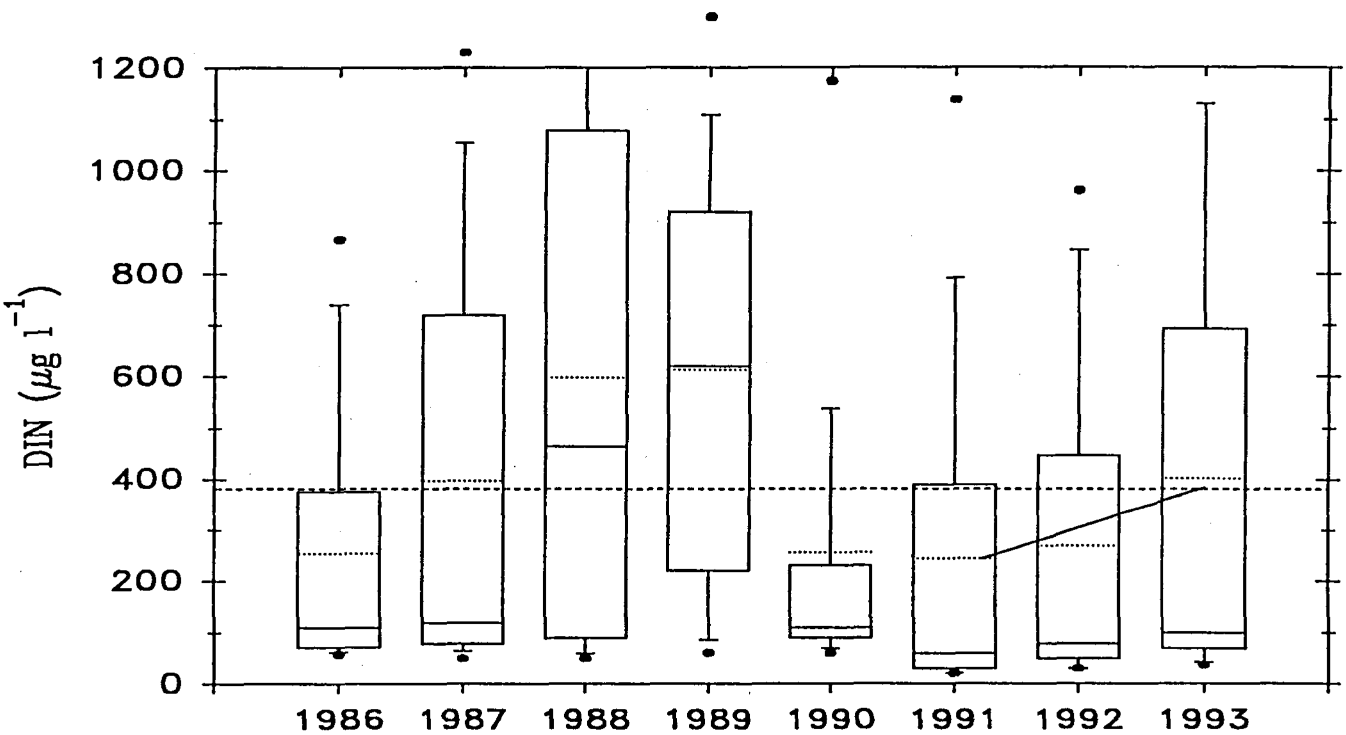


FIGURE 7: Box plot of annual dissolved inorganic nitrogen (DIN) in the Vaal River at Balkfontein.

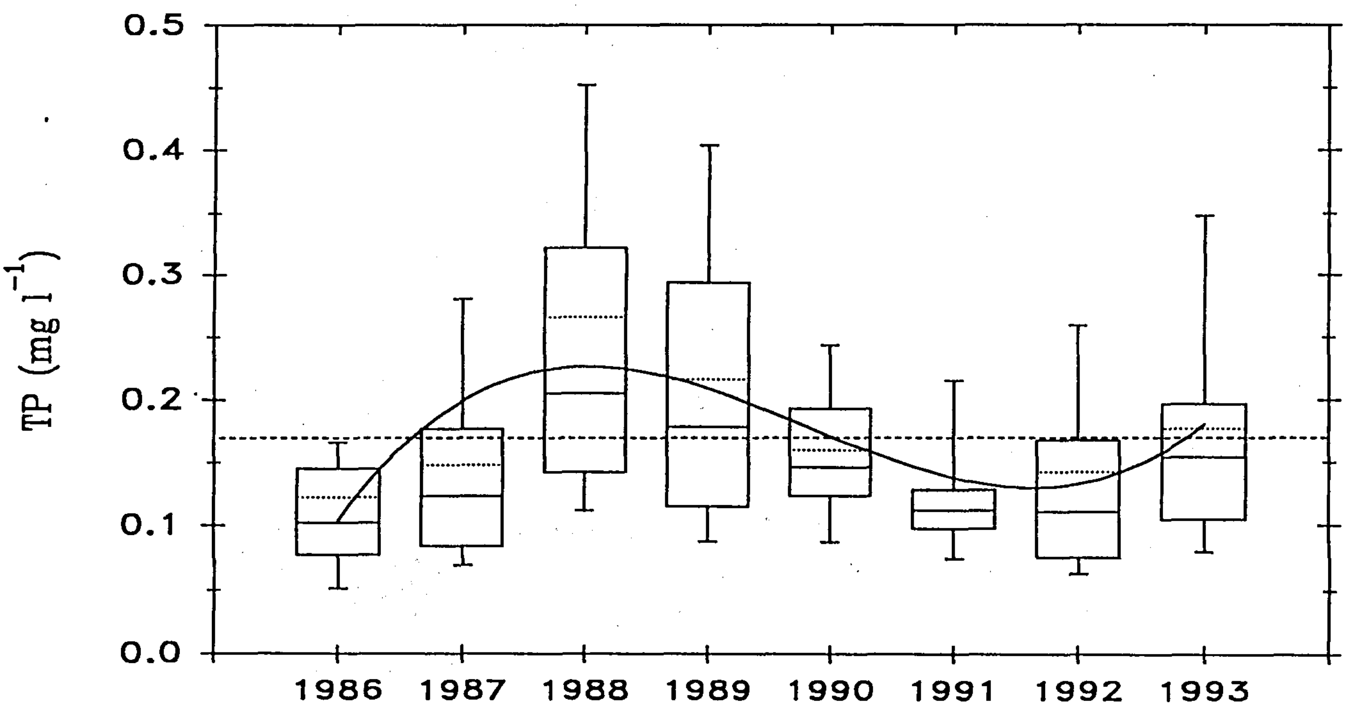


FIGURE 8: Box plot of annual total phosphorus (TP) in the Vaal River at Balkfontein.

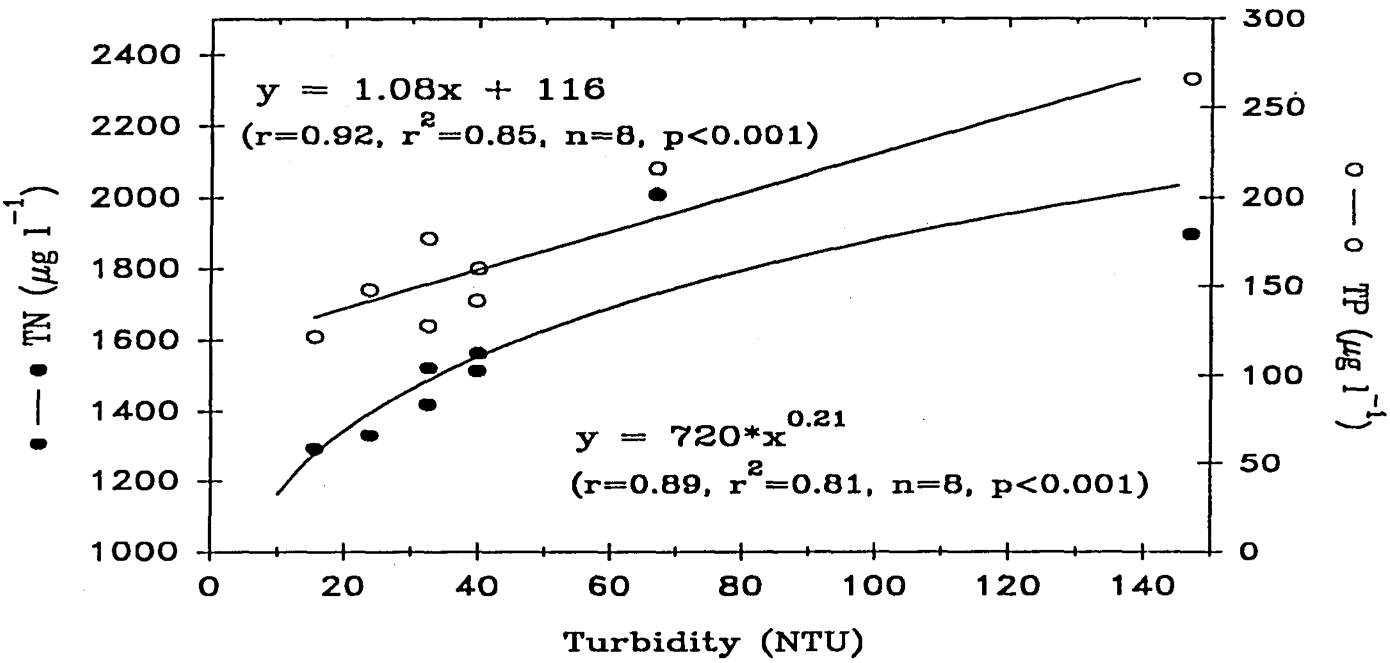


FIGURE 9: Relationship between the annual average turbidity, total nitrogen (TN) and total phosphorus (TP) concentrations in the Vaal River at Balkfontein.

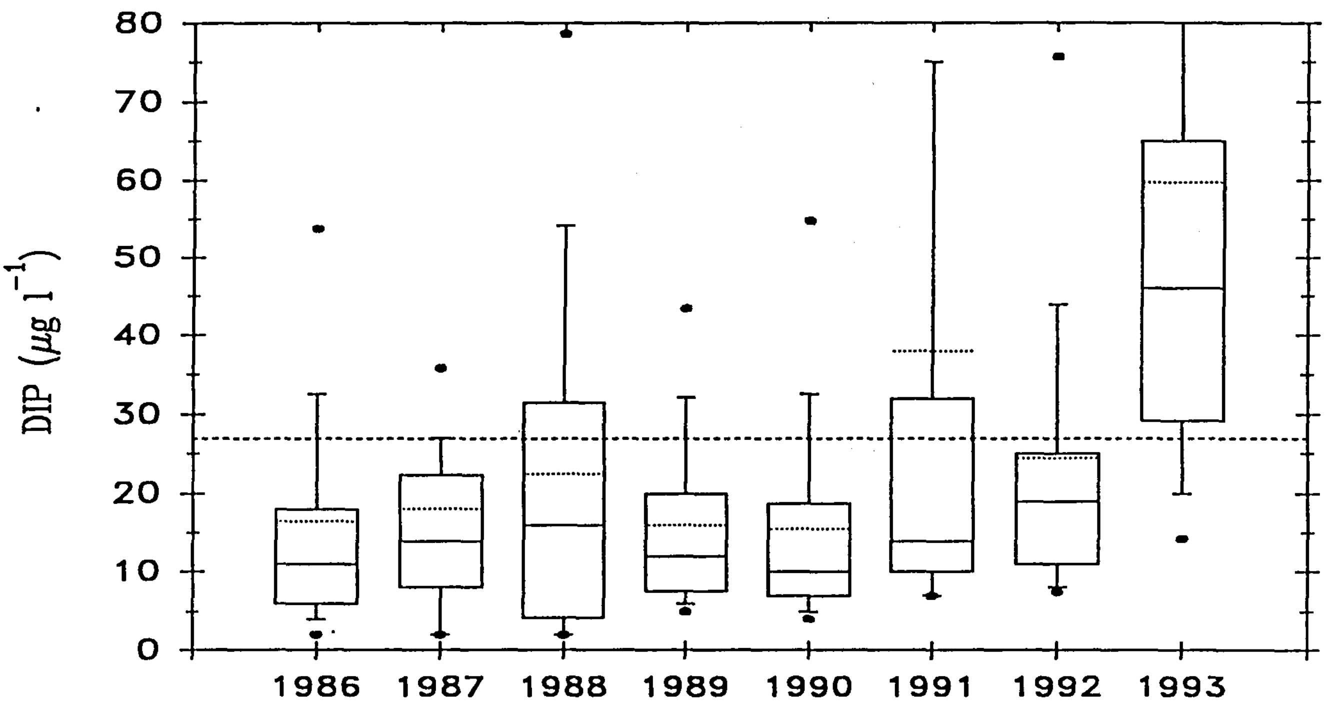


FIGURE 10: Box plot of annual phosphate or dissolved inorganic phosphorus (DIP) in the Vaal River at Balkfontein.

The annual average DIN:DIP ratio decreased since 1989 from 53 to 7.9 for 1993 (Fig. 11). This decrease in DIN:DIP ratio suggested that the Vaal River at Balkfontein switched from a general phosphorus limited system (1986-1990) to a potentially nitrogen limited system (1991-1993). The decrease in DIN:DIP ratio during 1990 was primarily caused by a sharp decrease in DIN concentrations (Fig. 7), while the decrease during 1991 and 1993 was primarily caused by relative higher increase in DIP than in DIN concentrations (Fig. 10).

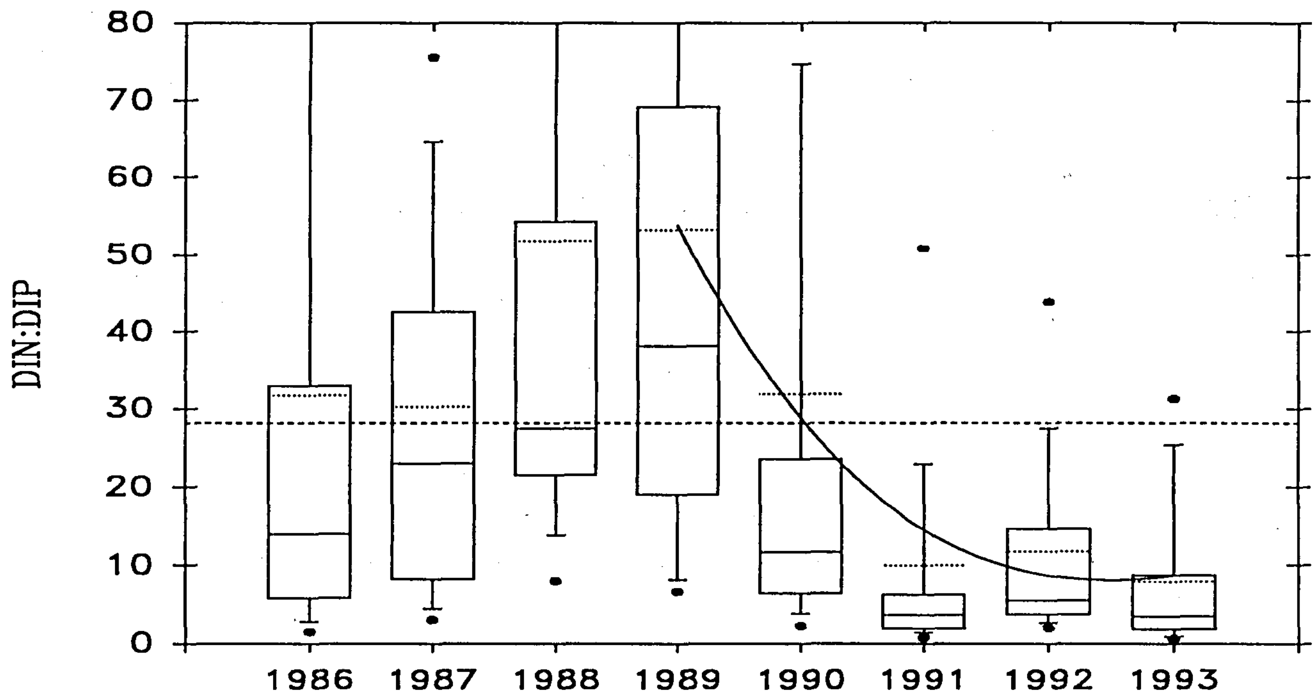


FIGURE 11: Box plot of annual dissolved inorganic nitrogen to dissolved inorganic phosphorus ratio (DIN:DIP) in the Vaal River at Balkfontein.

3.5 SILICA

The average silica silicon (SiO_2-Si) concentration showed a continuous decrease since 1988 (Fig. 12). The Si concentration during 1993 ranged between 0.18 and 5.5 mg l^{-1} with an average of 1.65 mg l^{-1} (Fig. 12). The reduced Si concentrations can partially be ascribed to reduced silt concentrations, which is supported by a positive correlation between Si and discharge and thus turbidity (compare Figs 1 & 12).

The Si concentration in the Vaal River was higher during periods of high discharge and high temperature, i.e. summer months (Fig. 13). The decrease in Si during the winter period (low discharge, clear water and biogenically induced desilification) is in contrast to other major dissolved constituents of river water, that commonly show an inverse relationship between concentration and discharge (Figs 3 & 13).

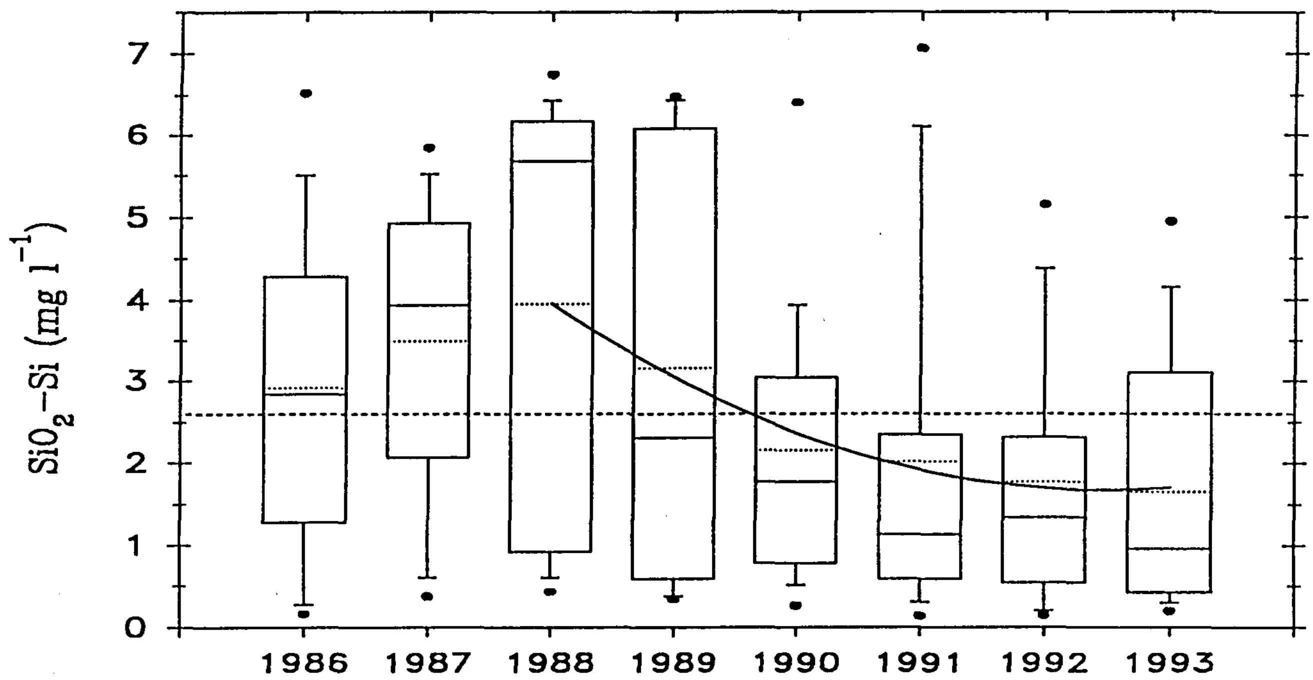


FIGURE 12: Box plot of annual silica silicon concentration (SiO_2-Si) in the Vaal River at Balkfontein.

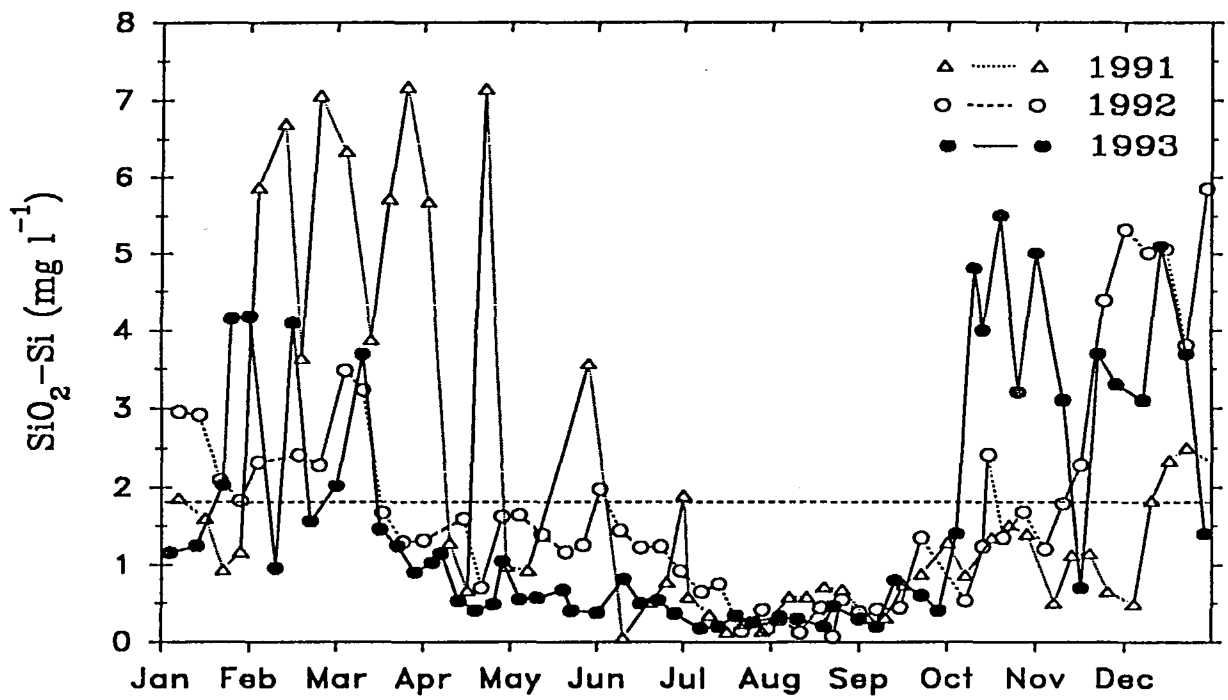


FIGURE 13: Annual variation of silica silicon concentration (SiO_2-Si) in the Vaal River at Balkfontein.

The ability of macrophytes to accumulate large quantities of nutrients from their environment is known from the literature (e.g. Gopal *et al.*, 1984). Only few *Eichhornia crassipes* (water hyacinth) plants were occasionally observed at Balkfontein up to 1989, but since 1990 the quantity increased drastically (visual observation) and heavy infestations were observed during 1991 and 1992. During 1993 the quantities decreased compared to 1991. The reduced phosphorus and nitrogen concentrations of 1990 and 1991 (Figs 6, 7 & 8) can, at least partially, be ascribed to hyacinth growth. The hyacinths could have contributed partially to the clarification of the water (Figs 2 & 3) because suspended material is filtered out by the dense root system. However, hyacinth infestation could also have reduced light availability to the phytoplankton because stretches of the river may have been covered entirely. The influence of water hyacinth on nutrient cycles and as a sediment trap in the Vaal River should be investigated in more detail.

3.6 CHLOROPHYLL-*a*

The chlorophyll-*a* concentration during 1993 ranged between 9 $\mu g\ l^{-1}$ (February) and 80 $\mu g\ l^{-1}$ (September) with the average at 33 $\mu g\ l^{-1}$ (Figs 14 & 16). The decreasing trend of average chlorophyll-*a* concentrations since 1989 was reversed during 1993 (Fig. 14). Although the water was clearer during 1990, 1991 and 1992 (Fig. 5), higher chlorophyll-*a* concentrations did not occur, suggesting that reduced nutrient levels (Figs 6, 8 & 12) limited the abundance of phytoplankton. This is supported by a significant positive correlation between the annual average chlorophyll-*a* and DIN, TP and Si and an inverse trend between annual average chlorophyll-*a* and Z_{eu} (Fig. 16).

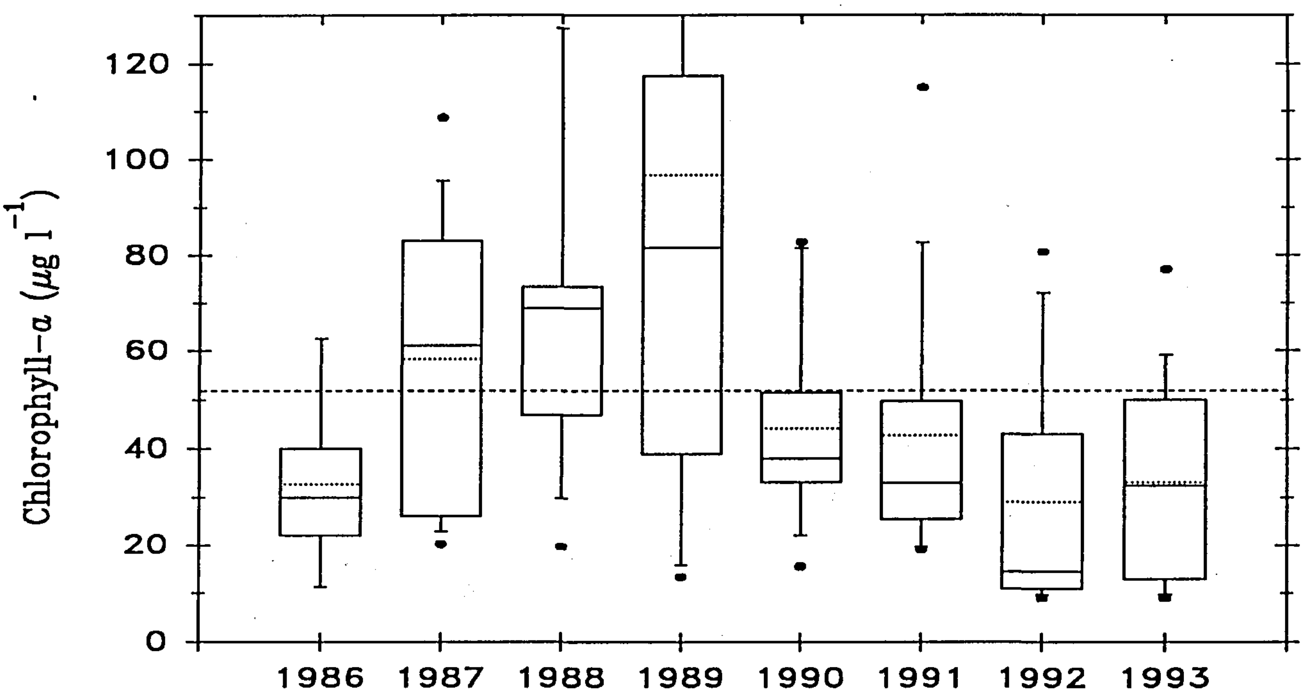


FIGURE 14: Box plot of annual chlorophyll-*a* concentration in the Vaal River at Balkfontein.

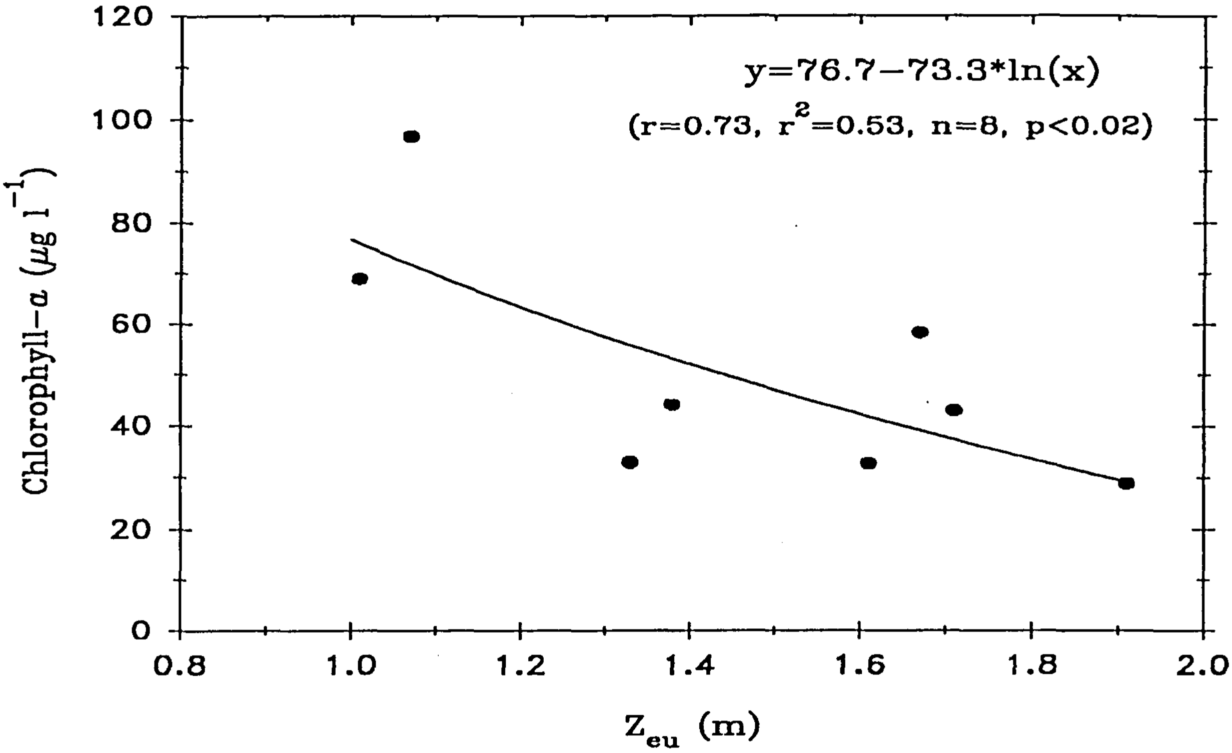


FIGURE 15: Relationship between the annual average euphotic zone (Z_{eu}) and chlorophyll-*a* concentrations in the Vaal River at Balkfontein.

High discharge was usually associated with high turbidity and low chlorophyll-*a* concentrations. On an annual basis, however, a positive correlation was demonstrated between chlorophyll-*a* and discharge (Figs 1 & 14). Higher discharge (especially from summer rainfalls) was associated with higher nutrient concentrations (nitrogen, phosphorus and silicon) which resulted in higher chlorophyll-*a* concentrations. The response of phytoplankton appeared to be related more to nutrient concentration than to light regimes. However, Roos (1992) demonstrated that although nutrients were present in high concentrations during the floods, the phytoplankton could not attain maximum productivity because light penetration was restricted by the turbidity of the water. It was only after several weeks, following the high discharge, that the water transparency increases, permitted a bloom of phytoplankton. Thus in nature, the input of nutrients by higher discharge and the development of an algal bloom is separated by several weeks. For example, the unusually mid summer bloom (maximum 82 $\mu g\ l^{-1}$; December 1992) was probably triggered by the input of high concentrations of DIN (max. 1 070 $\mu g\ l^{-1}$) and DIP (max. 148 $\mu g\ l^{-1}$) during the early summer rains in November 1992 (average flow for week 46 was 178 $m^3\ s^{-1}$)

Maximum chlorophyll-*a* concentrations (blooms) were usually a double peak, encountered during the winter-spring (July to August) period (Fig. 16), commonly dominated first by diatoms followed by a bloom of green algae. However, the intensity of the winter-spring bloom has decreased since 1988, probably because nutrient levels decreased, while summer blooms became more prominent (March 1991, December 1992, Fig. 16). The winter bloom during July 1992 was relatively small, with the chlorophyll-*a* concentrations at 68 $\mu g\ l^{-1}$ (Fig. 16).

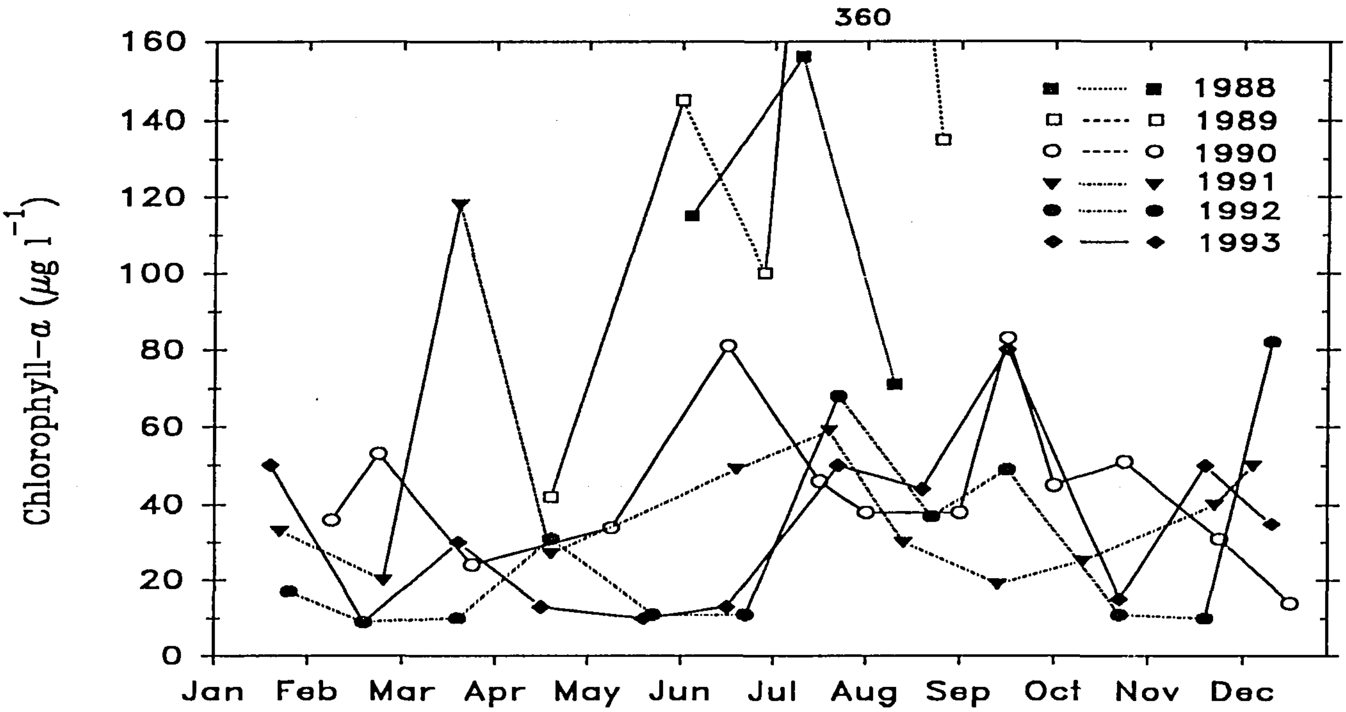


FIGURE 16: Annual variation of chlorophyll-*a* concentration in the Vaal River at Balkfontein.

3.7 PHOTOSYNTHETIC CHARACTERISTICS OF THE PHYTOPLANKTON

The response of phytoplankton to the effect of light through photosynthesis is one of the most important processes in nature. The daily rates of areal photosynthesis (∂P_d) in the Vaal River at Balkfontein during 1993 ranged between 126 and 4 095 mg C m^{-2} d^{-1} with a average of 1 440 mg C m^{-2} d^{-1} (Fig. 17). A decrease of ∂P_d since 1989 can be ascribed to a decrease in chlorophyll-*a* and nutrient concentrations. However, the average values for the study period still fall within the range of eutrophic water bodies (Wetzel, 1983).

Because increased photosynthetic rates were usually associated with increased euphotic zone depths (Roos, 1992), lower daily rates (∂P_d) were expected during 1993. However, a decrease in the average Z_{eu} in the river (Fig. 5) was associated with increased average ∂P_d and P_m (Figs 17 & 20). The higher production values can be ascribed to increased chlorophyll-*a* concentrations (Fig. 14) because 60 % of the variation in ∂P_d during 1993, was associated with the variation in chlorophyll-*a* concentrations. These were probably a result of increased nutrient concentrations.

Several investigators (e.g. Wofsy, 1983) reported turbidity and light intensity (as opposed to nutrient concentration) as the factors most commonly controlling primary production in rivers. The importance of the ULC to photosynthesis was illustrated by Roos (1992). Increased ULC was usually associated with increased P_d values (Fig. 18). However, the production level in the Vaal River appeared to be related more to chlorophyll-*a* concentration than to light regimes (Fig. 18).

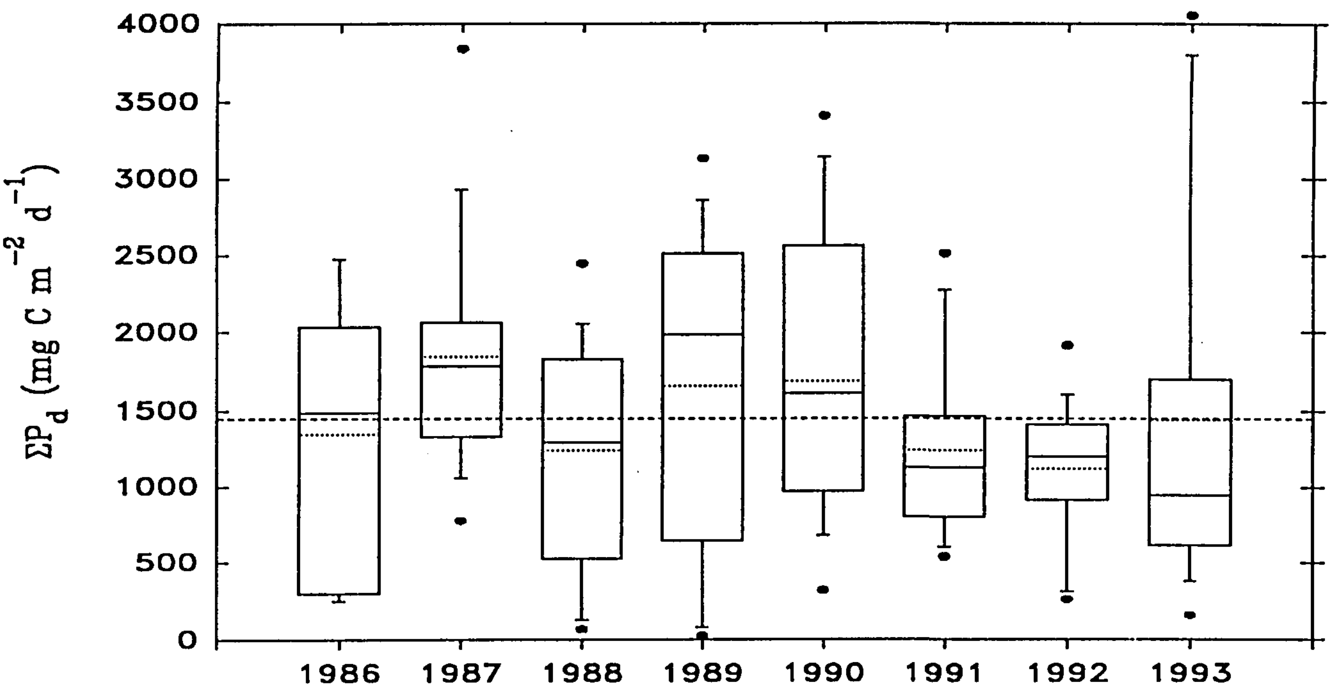


FIGURE 17: Box plot of daily rates of areal photosynthesis (dP_d) in the Vaal River at Balkfontein.

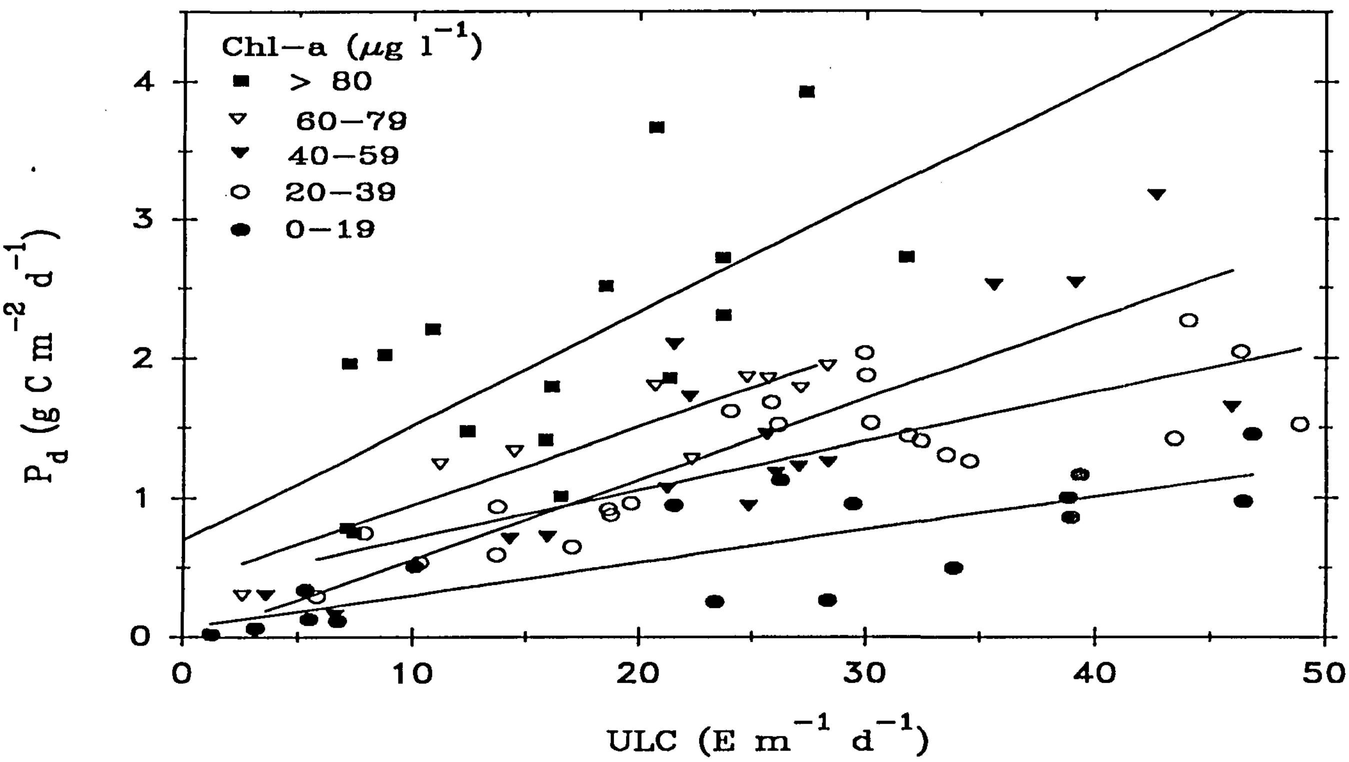


FIGURE 18: Relationship between the underwater light climate (ULC) and daily rates of areal photosynthesis (dP_d) at different chlorophyll-*a* concentrations (with linear regression lines) in the Vaal River at Balkfontein.

The process of photosynthesis involves the conversion of light energy into chemical energy. Photosynthetic efficiency is expressed by the ratio of carbon fixed (normalised to chlorophyll-*a*) by the algae to the light (energy) available. The photosynthetic efficiency (PSE), or total water-column light utilisation efficiency, of phytoplankton in the Vaal River during 1993 ranged between 0.2 and 0.95 mg C (mg Chl*a*)$^{-1}$ mol^{-1} m^2 with the average at 0.49 mg C (mg Chl*a*)$^{-1}$ mol^{-1} m^2. PSE showed an increasing trend since 1989 (Fig. 19), that was associated with a decrease in chlorophyll concentration (Fig. 14). Thus, denser assemblages of algae are photosynthetically less efficient probably because of self-shading.

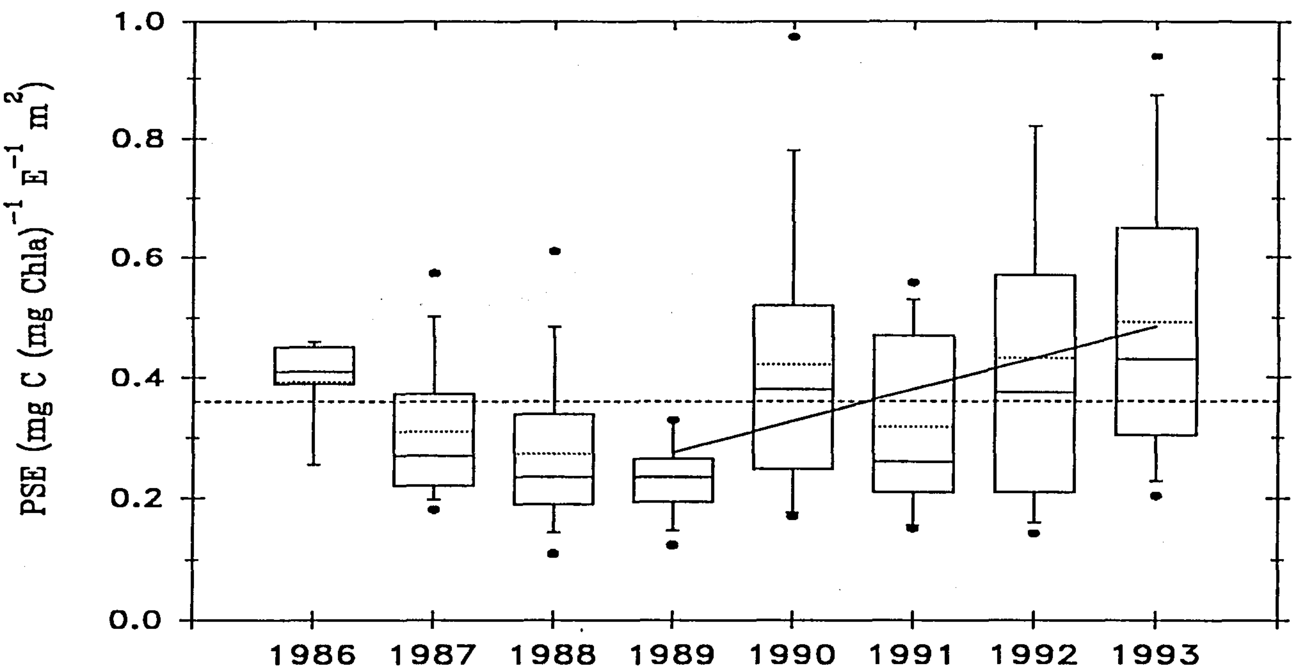


FIGURE 19: Box plot of annual photosynthetic efficiency (PSE) in the Vaal River at Balkfontein.

The maximum rate of photosynthesis per unit volume (P_m) is an expression of the response to optimal irradiance and available nutrients at a particular time and at a specific depth by the biomass present. The relatively high P_m during 1993 (average 230 mg C m^{-3} h^{-1}) and decreasing trend since 1989 could be ascribed to higher nutrient availability and higher chlorophyll-*a* concentrations because P_m was positively correlated with chlorophyll-*a* concentrations.

Irradiance at the onset of light saturation (I_k) represents the index of light adaptation (Talling, 1976). I_k is also an important measure of the physiological adjustment of an algal assemblage, but it does not express photosynthetic efficiency.

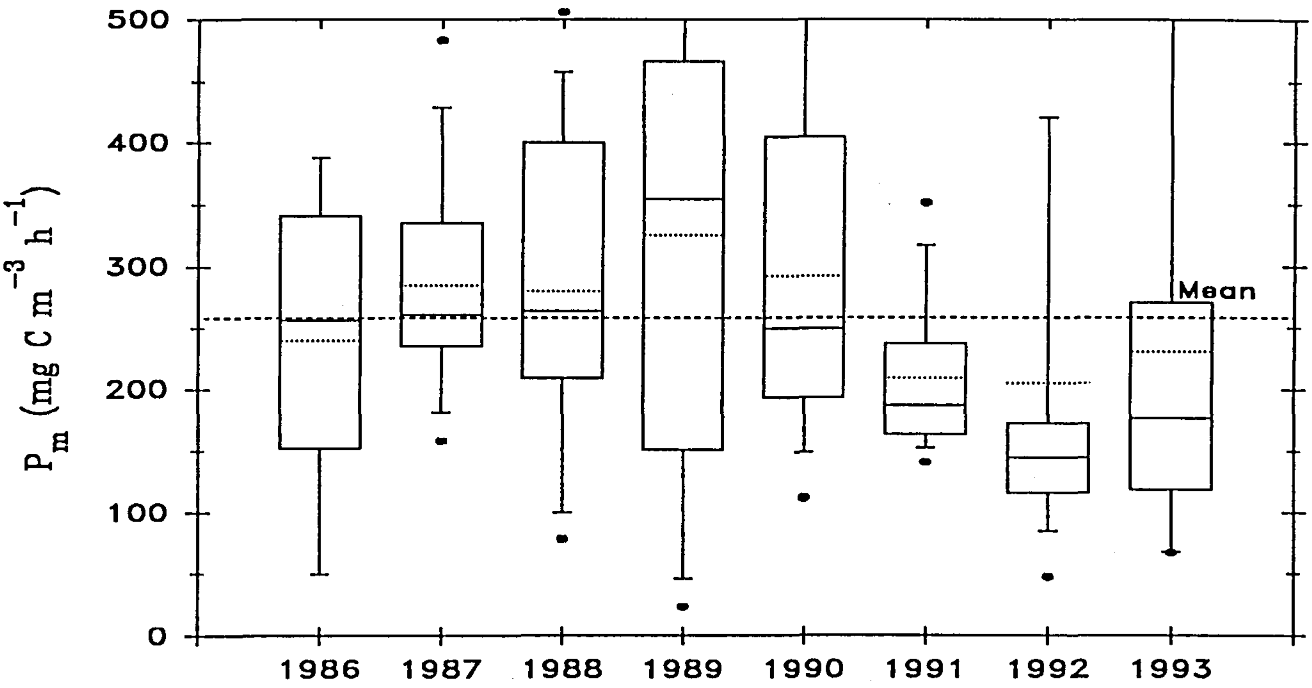


FIGURE 20: Box plot of the annual maximum rate of photosynthesis in depth profile (P_m) in the Vaal River at Balkfontein.

During the study period the average I_k values ranged between 386 and 673 µmol m^{-2} s^{-1}, which is much higher than in other aquatic systems (Harris, 1978). The high I_k values, together with the high light intensities at which P_m occurred, suggested that the Vaal River phytoplankton were adapted to higher light intensities than phytoplankton in other aquatic systems (Roos & Pieterse, 1992). Since 1991 the maximum and average I_k showed a gradual decrease (Fig. 21), that was associated with increased turbidity in the water-column (Fig. 4). Higher I_k values coincide with higher ULC which suggested that the phytoplankton assemblages in the Vaal River are able to adapt to a wide range of light conditions.

The photosynthetic capacity (P^B_m, mg C (mg Chl*a*)$^{-1}$ h^{-1}) represents the ratio of carbon fixed at light saturation to the chlorophyll-*a* concentration of the assemblage. P^B_m is apparently also an indication of photosynthetic efficiency because P^B_m was significantly positively correlated with PSE (compare Figs 19 & 22).

The average P^B_m for 1992 of 8.3 mg C mg Chl*a*$^{-1}$ h^{-1} was the highest since 1986 (Fig. 22). It was shown in the literature (Bannister and Laws, 1980) that P^B_m is independent of nutrients, but sensitive to illumination of the phytoplankton. A less favourable light regime, like that experienced during 1993 (Fig. 5), would therefore lead to lower P^B_m values (Fig. 22). A significant positive correlation was demonstrated between the annual average P^B_m and Z_{eu} (compare Figs 4 & 22). However, an inverse correlation between P^B_m and chlorophyll-*a* concentrations was demonstrated (compare Figs 14 & 22) which is in agreement with the observation that denser assemblages of algae are photosynthetically less efficient.

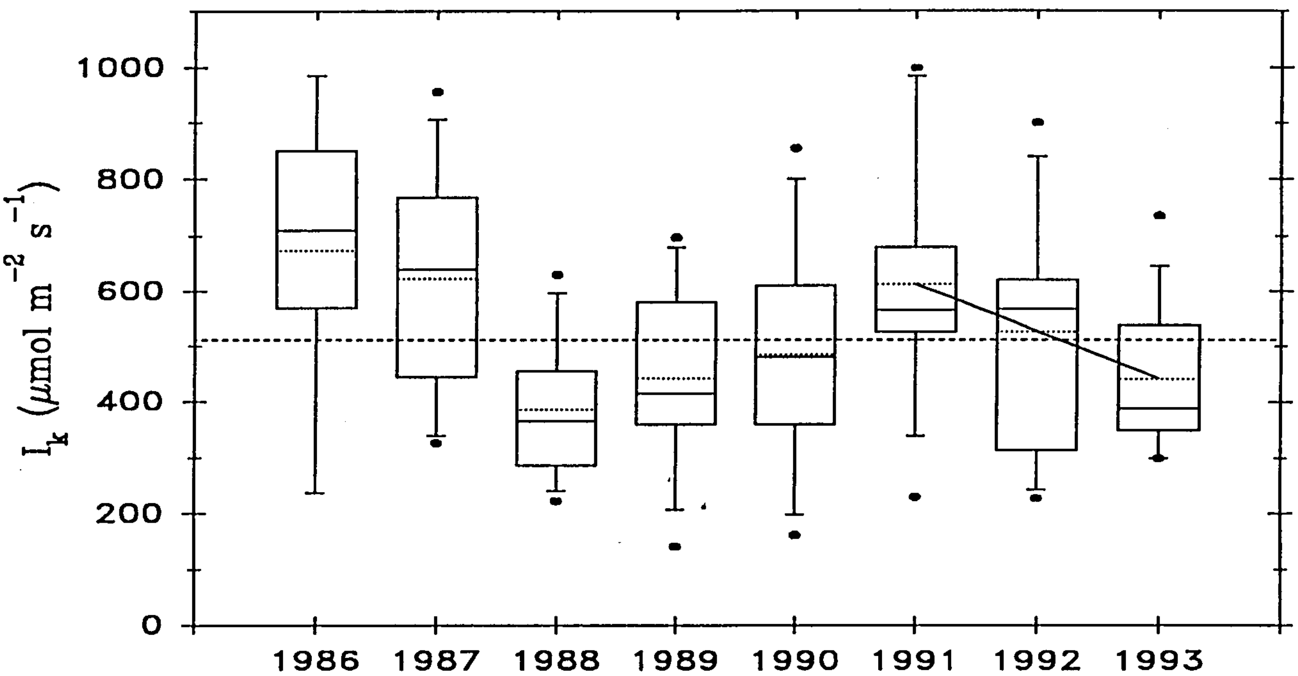


FIGURE 21: Box plot of the annual photoadaptation parameter (I_k) in the Vaal River at Balkfontein.

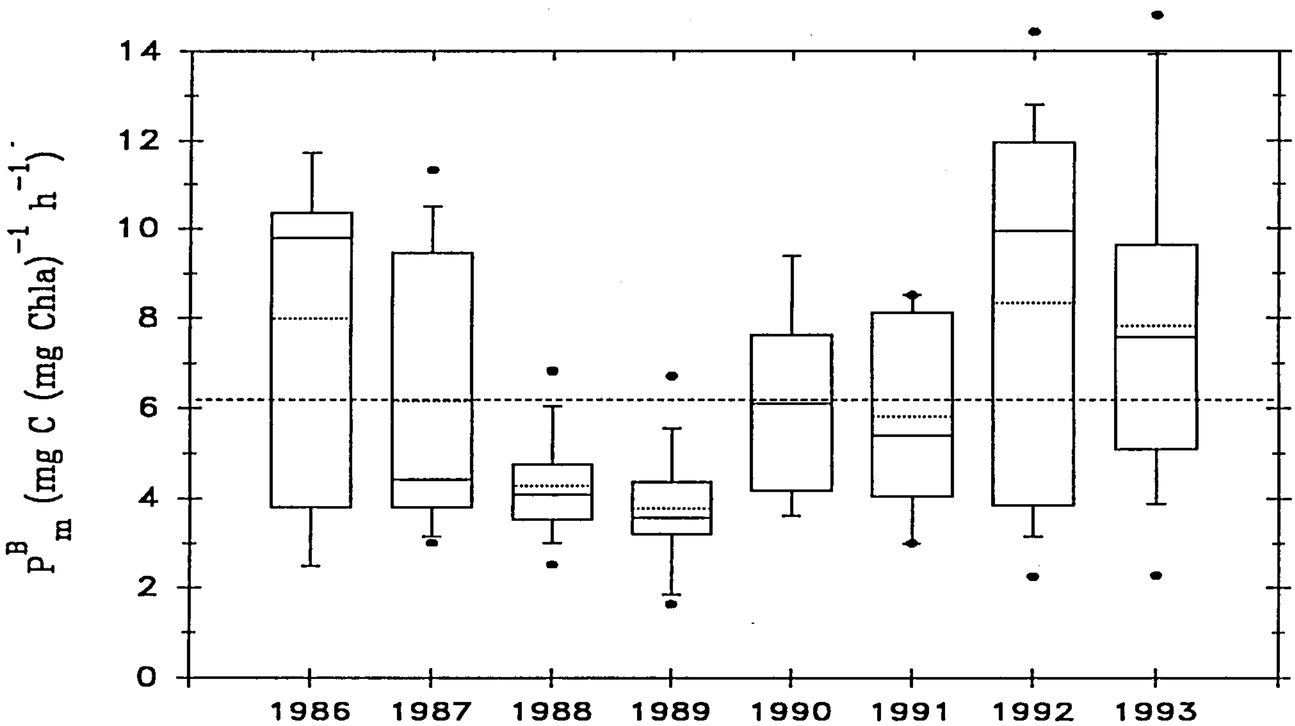
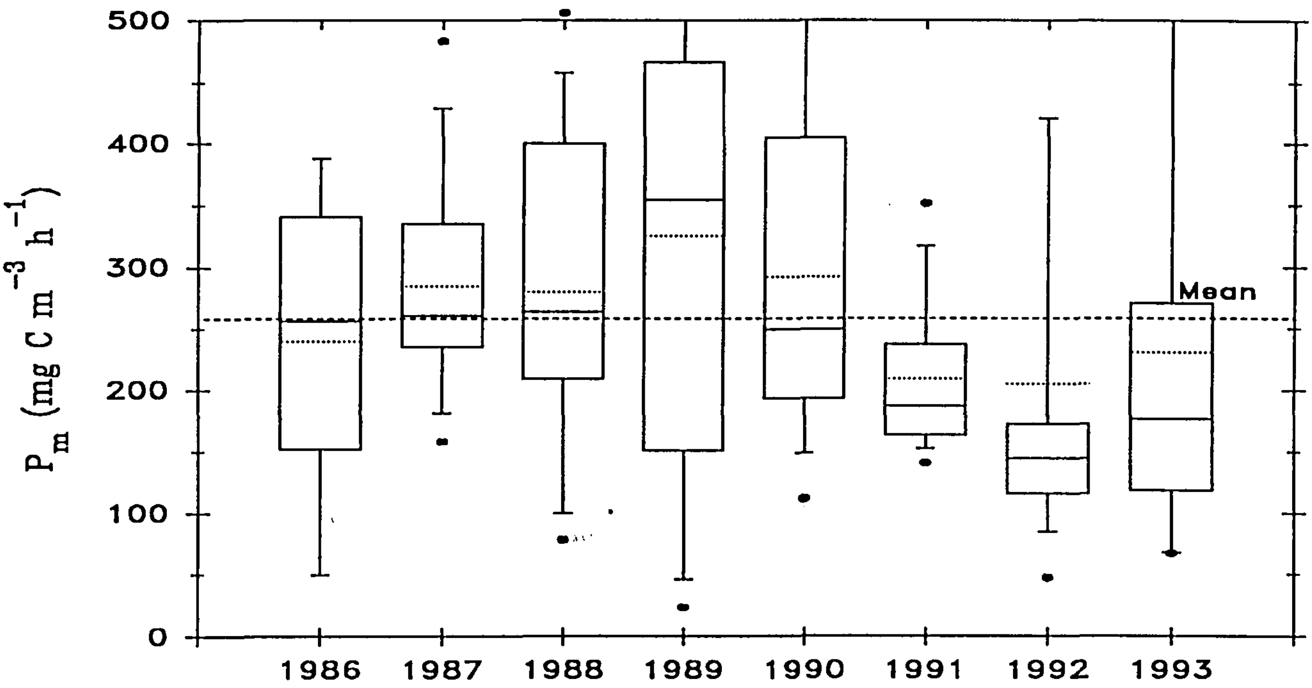


FIGURE 22: Box plot of the annual photosynthetic capacity ($P^B{}_m$) in the Vaal River at Balkfontein.



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During the study period the average I_k values ranged between 386 and 673 µmol m^{-2} s^{-1}, which is much higher than in other aquatic systems (Harris, 1978). The high I_k values, together with the high light intensities at which P_m occurred, suggested that the Vaal River phytoplankton were adapted to higher light intensities than phytoplankton in other aquatic systems (Roos & Pieterse, 1992). Since 1991 the maximum and average I_k showed a gradual decrease (Fig. 21), that was associated with increased turbidity in the water-column (Fig. 4). Higher I_k values coincide with higher ULC which suggested that the phytoplankton assemblages in the Vaal River are able to adapt to a wide range of light conditions.

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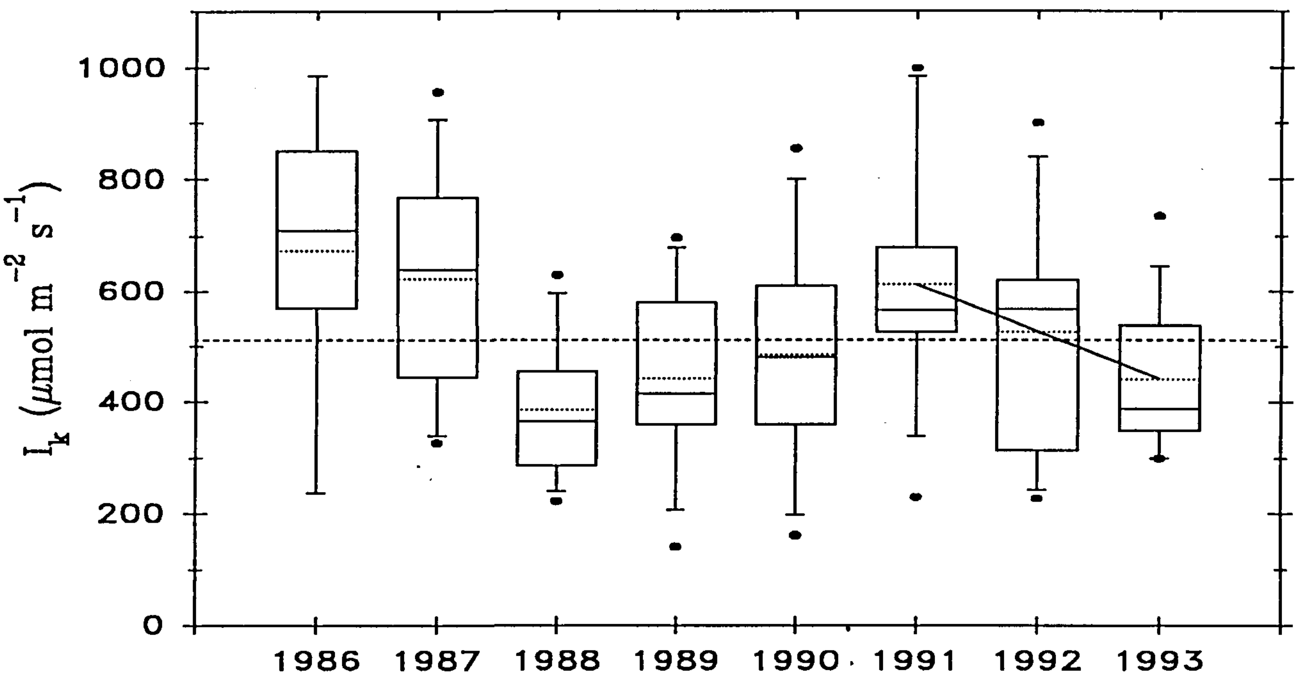

FIGURE 21: Box plot of the annual photoadaptation parameter (I_k) in the Vaal River at Balkfontein.

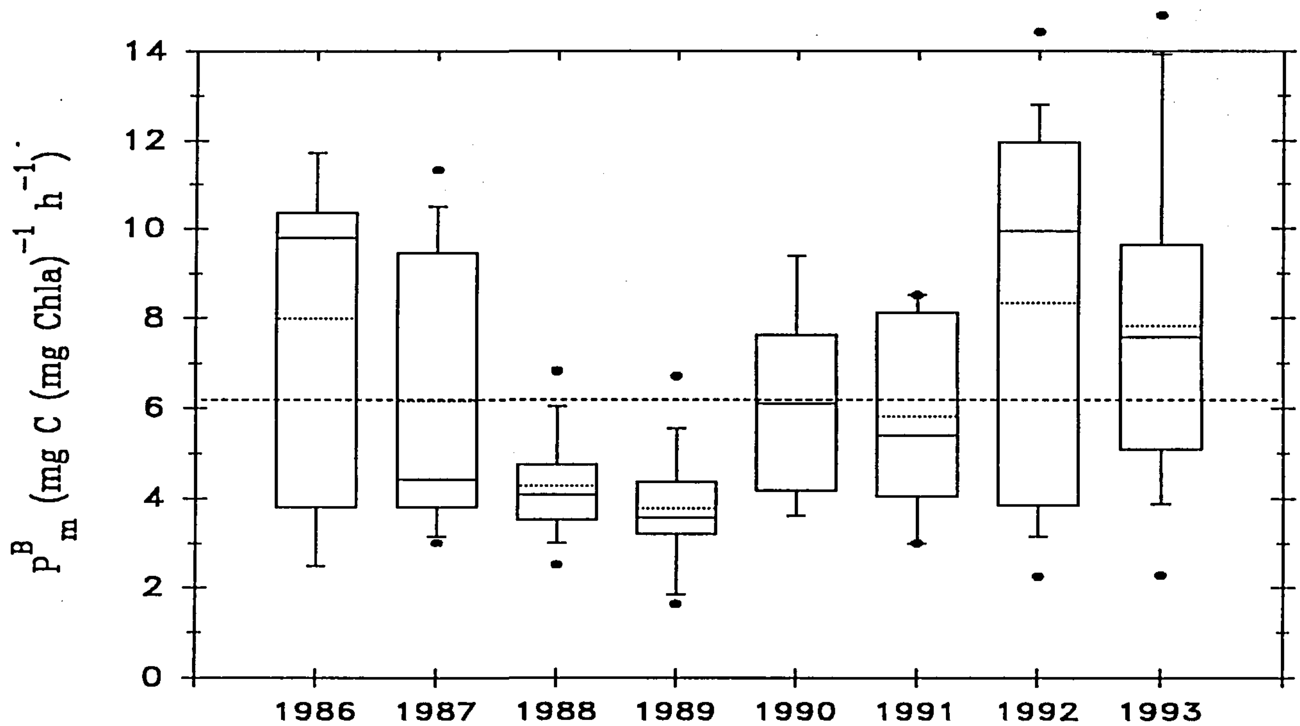

FIGURE 22: Box plot of the annual photosynthetic capacity ($P^B{}_m$) in the Vaal River at Balkfontein.

4. CONCLUSIONS

The hydrology, particularly the episodic inputs from summer rain, plays an important role not only in the chemistry, but also in the biology of the Vaal River (Roos, 1992). Discharge was the most important variable to influence transparency of the Vaal River water. Higher discharge resulted in higher TSS concentrations and in higher turbidity and thus in lower Z_{eu} and ULC.

The average TDS concentration during the study period in the Vaal River (540 mg l^{-1}) is higher than the average TDS in major African rivers (Martins & Probst, 1991), e.g. rivers from arid regions like the Zambezi (113 mg l^{-1}) and Nile (318 mg l^{-1}). The main importance of the high TDS in the Vaal River is evidently its influence on turbidity and the possible clarification of the water-column, which will result in a deeper euphotic zone and thus more favourable light conditions for photosynthesis and a possible biomass build-up during the winter-spring period.

The relatively low turbidities since 1990 (Fig. 3) can be ascribed to the generally high TDS levels and relatively low discharge in the Vaal River (Figs 1 & 2) and possibly to water hyacinth infestation. Water hyacinths evidently served as a biofilter, i.e. a sediment and nutrient trap.

The reduced euphotic zone in the Vaal River during 1993 (Fig. 5) was associated with reduced I_k and $P^B{}_m$ values (Figs 21 & 22), but increased dP_d and P_m values (Figs 17 & 20). The higher P_m and dP_d can be ascribed to increased chlorophyll-*a* concentrations (Fig. 14). However, the photosynthetic efficiency and capacity increased together with decreased chlorophyll-*a* concentrations (Figs 19 & 22).

Reduced annual DIN:DIP ratios suggested that the Vaal River at Balkfontein switched from a general phosphorus limited system (1986 to 1990) to a potentially nitrogen limited system (1991 and 1993). The decrease in DIN:DIP ratio was caused by relatively low DIN concentrations (Fig. 7) and relatively high DIP concentrations (Fig. 10).

Low average chlorophyll-*a* concentrations (Fig. 14) and reduced peak concentrations during the winter-spring period of 1990 to 1993 (Fig. 16) could primarily be ascribed to reduced silica availability (Fig. 12). Reduced nutrient availability (especially DIN), in turn, is possibly caused at least partially by water hyacinth growth since 1990. The reduced nutrient availability apparently limited production and abundance of phytoplankton even when sufficient light was available (Fig. 5). Thus increased nutrient concentration in the Vaal River increases the capacity of the water to support high production rates and to maintain large standing crops of phytoplankton.

The influence of *Eichhornia crassipes* (water hyacinth) on nutrient cycling and sediment accumulation should be investigated to develop a better understanding of nutrient dynamics in the Vaal River ecosystem and the clarification of the water.

The above results support a conceptual model that illustrates biological production as a flux of solar energy. The 'valve' that controls the flow of energy is possibly operated by the flux of plant nutrients.

5. SUMMARY

5.1 The **average annual TDS** in the Vaal River at Balkfontein **decreased** since 1991 from 660 mg l^{-1} to 477 mg l^{-1} during 1993.

5.2 **Higher discharge** resulted in higher TSS concentrations, higher turbidity and thus in lower Z_{eu} and ULC.

5.3 **Turbidity** in the Vaal River is primarily **a function of suspended sediment concentration** which is correlated with discharge

5.4 **High turbidity** was associated with **high nutrient**, especially N & P, concentrations.

5.5 **High TDS** during the winter-spring period in the Vaal River was **associated with low turbidity** thus favourable light conditions for photosynthesis which resulted in a biomass build-up during this period.

5.6 **Maximum chlorophyll-*a* concentrations (blooms) were usually a double peak, encountered during the July to August period,** commonly dominated first by diatoms followed by a bloom of green algae.

5.7 A significant clearing of the Vaal River water at Balkfontein was noticed since 1988 to 1992. **This clearer water was associated with lower nutrient concentrations and thus lower chlorophyll-*a* concentration (annual averages).** Above conditions were reversed during 1993.

5.8 **The reduced nutrient availability apparently limited production and abundance of phytoplankton** even when sufficient light was available.

5.9 Reduced annual DIN:DIP ratios suggested that the **Vaal River** at Balkfontein **switched from a** general **phosphorus limited system** (1986 to 1990) **to a potentially nitrogen limited system** (1991 and 1993).

5.10 **Water hyacinths evidently served as a biofilter,** i.e. a sediment trap (contribute to the clarification of the water) and nutrient trap (reduce nutrient availability).

5.11 **The reduced euphotic zone** in the Vaal River during 1993 was **associated with reduced I_k and P^B_m values, but increased $\text{đ}P_d$ and P_m values.**

5.12 **The higher P_m and $\text{đ}P_d$ can be ascribed to increased chlorophyll-*a* concentrations.** However, the photosynthetic efficiency and capacity increased together with decreased chlorophyll-*a* concentrations.

5.13 **Increased ULC was usually associated with increased P_d values.** However, the production level in the Vaal River appeared to be related more to chlorophyll-*a* concentration than to light regimes.

6. ACKNOWLEDGEMENTS

The authors are grateful to the Water Research Commission for the financial support for the ecological research programme on the Vaal River and to The Department of Water Affairs and Forestry for providing additional chemical data. Erika Coetzee, Danie Traut and Rolien Visser assisted during field trips.

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CHAPTER 3: ENVIRONMENTAL VARIABLES, ABUNDANCE AND SEASONAL SUCCESSION OF PHYTOPLANKTON POPULATIONS

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1. INTRODUCTION

The Vaal River as a water resource is being intensively exploited (Basson & van Rooyen, 1989), and water demands in the catchment area are drastically increasing, making the quality of the water important. Increases in the intensity of utilisation will result in increased regulation and in the need for more intensive monitoring, investigative and management programmes.

The main water quality problems in the Vaal River are increased salinity and eutrophication (Braune & Rogers, 1987). The problem of eutrophication is partly controlled because of the turbid nature of Vaal River water but increasing salinity and accompanying decreases in turbidity can enhance primary productivity (Braune & Rogers, 1987).

The use of the Vaal River as a pathway for the disposal of industrial, mining and domestic wastes, is increasing the concentration of dissolved phosphorus and nitrogen in the water which leads to the eutrophication of the river (Bruwer *et al.*, 1985; Grobler *et al.*, 1983) and results in the development of algal blooms (Pieterse, 1986b). Algal blooms can result, amongst others, in aesthetically unacceptable situations and increased purification costs of water for potable purposes, because filters become clogged and scums are formed in purification plants (Bruwer *et al.*, 1985), increased chemical dosages are necessary and more sophisticated treatment processes are required to remove tastes, odours and other side effects. Carcinogenic trihalomethanes may be formed when water from eutrophic sources is chlorinated during purification. When algal cells die, anaerobic conditions may develop, because of the depletion of oxygen during decomposition, sometimes resulting in fish mortality. Algal growth on canal linings (for example macrophytes) result in loss of hydraulic capacity (DWA, 1986). Algal blooms in open water bodies can be the cause of complaints by persons using the water for recreation purposes.

Clearer water, in conjunction with a high nutrient supply through eutrophication, will result in blooms, in some cases possibly by algal species that have not, until now, caused such blooms (Pieterse, 1986b). One particular, and alarming, possibility is dinoflagellate blooms (which cause red-tides in the ocean; Horstman, 1981), that should be considered potentially toxic (Pieterse, 1986b). During the summers of 1942 and 1943 a serious outbreak of livestock and other animal poisoning occurred around the Vaal Dam. Thousands of cattle, sheep and many other animals died (Steyn, 1943, 1944, 1945a, 1945b; Stephens, 1949 and Louw, 1950). The alga responsible was described as *Microcystis toxica*, a blue-green alga (Stephens, 1949), which was later shown to be a synonym for *M. aeruginosa* fo. *aeruginosa* (Komárek, 1958). *Oscillatoria simplicissima* is another representative of the blue-green algae known for the production of toxins. Both these algae sometimes reach dominant proportions in the Vaal River, indicating the potential for toxic *Microcystis aeruginosa* and *Oscillatoria simplicissima* blooms in the Vaal River.

Specific algal species are responsible for specific problems. It is therefore important to identify the specific algal genera and species causing the problems. Analyses of phytoplankton assemblages in rivers allow, amongst others, investigations of temporal changes in species abundance and species composition, two important aspects or attributes of community structure (Mueller-Dombois & Ellenberg, 1974; Barbour, Burk & Pitts, 1980).

Pieterse (1986b) stated that investigations on Vaal River phytoplankton should, however, also include other sections of the river, because more complete knowledge of the distribution of algal species is necessary in order to improve the understanding of all aspects related to problem causing algae. One of the objectives of this study is, therefore, to provide information about the algal groups and species present in the Vaal River at four sampling localities, namely Barrage, Parys, Stilfontein and Balkfontein. Another objective of the study is to investigate changes in the composition of the algal populations over time (seasonal succession).

The number and kinds of algae which grow in surface waters depend on environmental conditions (Palmer, 1980). Blooms of algal species are thus the result of environmental conditions favourable for growth. Algal problems which relate to providing suitable water supplies, together with the use of some algae in water supply and sewage treatment improvements, clearly indicate a need for more knowledge of the environmental requirements of these organisms (Palmer, 1980). Pieterse & Röhrbeck (1990) and Roos (1992) indicated that seasonal succession in Roodeplaat Dam and the Vaal River respectively, is influenced by physical and chemical variables. It is therefore also important to investigate the environmental variables and their relation to species composition and succession in order to explain the development of blooms. The connection between the environmental variables and the succession patterns of algal species and groups will be discussed in an attempt to explain the development of specific algal blooms.

It is important to know the algal population of a river before any major change is made in the use of the stream. The Lesotho Highlands Water Project, to be completed in the foreseeable future, will supply water to the Vaal River system to meet, for some time, increasing demands. It is not clear, however, what the effect of the Lesotho Highland water will be on the chemical and biological quality of the water in the middle Vaal River region, or how long it will last. The present study could therefore provide more information on the water quality before the implementation of the Lesotho Highland water project. Future studies will show changes in water quality after the importation of the Lesotho Highland water.

The aim of this study is, therefore, to provide more information about the types of algae present, their seasonal patterns of occurrence and environmental factors controlling their growth and succession at four sampling localities in the river, including the highly industrialised PWV complex.

2. STUDY AREA, MATERIALS AND METHODS

2.1 SAMPLING LOCALITIES

For the purposes of this study four sampling localities were selected in the Middle Vaal River, namely at the Rand Water Barrage (near Vanderbijlpark), Parys Municipality (Parys), Western Transvaal Regional Water Company (near Stilfontein), and Goldfield Water at Balkfontein (near Bothaville).

The Rand Water Barrage is situated approximately 80 km downstream from the Vaal Dam and 38 km downstream from Vereeniging. The Vaal River Barrage is 65 km in length with a capacity of 62 000 Ml. The river at the Barrage is approximately 190 m wide (Steynberg, 1986). The water level in the Barrage is controlled at a depth of 7.32 m (1 421.6 m above mean sea level) by means of regulation of the sluice gates. At the Barrage water samples were taken from the Vaal River approximately 50 m upstream from the Barrage sluice gates at approximately equal distances from the Transvaal and Free State banks.

At Parys, approximately 39 km downstream of the Barrage, water samples were taken in the canal that supplies water to the purification plant of the Parys Municipality. The canal is approximately one km long, it has a mean width of 3.5 m and it is approximately 1.2 m deep.

The sampling locality at Stilfontein is at the Western Transvaal Regional Water Company's waterworks, approximately 147 km downstream from the Barrage and 108 km downstream from the Parys sampling locality. The sampling locality is situated in close proximity to the water intake point of the pumping station of the Western Transvaal Regional Water Company, where the river is on average 2 m deep and 68 m wide.

At Balkfontein (± 20 km by road from Bothaville) the sampling locality is situated near the pumping station of the Balkfontein purification plant of Goldfield Water, approximately 265 km downstream from the Barrage and 118 km downstream from the Stilfontein sampling locality. The width of the river at the sampling locality is about 77 m, with a maximum depth of 5 m and an average depth of about 4 m (Pieterse, 1986a).

2.2 PHYTOPLANKTON

Water samples were taken every two weeks from April 1991 to December 1993 at the Barrage and Parys sampling localities and from February 1991 to December 1993 at Stilfontein and Balkfontein. Two liter water samples were filtered through a 10 µm Nybolt net in order to concentrate the algal cells for identification purposes. A 100 ml water sample was also taken every two weeks for quantative analysis. Both the net and the 100 ml samples were fixed with two percent formaldehyde (final concentration).

All algal species present were identified (excepting the centric diatoms), and counted by means of the technique described by Utermöhl (1931, 1958) and modified by Lund *et al.* (1958), which employs an inverted Zeiss Light Microscope. Sedimentation tubes were used for counting of algal units i.e. cells, colonies and filaments. For the purpose of this investigation, the counts were recorded as number of cells, colonies or filaments per unit volume of water (1 ml).

The centric diatoms were not identified while counting (except for the filamentous centric diatom, *Melosira granulata*), but a total count of all other centric diatoms were made. The cell counts, together with the original sub-sample volume transferred to the sedimentation tubes, as well as the number of strips counted in the sedimentation tubes, were then used to calculate the concentration of individual phytoplankton species in cells ml^{-1} with the aid of a standard computer programme (see section 2.4).

Samples were also prepared for investigation with a Jeol Winsem Scanning Electron Microscope (SEM) in order to identify different centric diatom species. Centric diatoms were identified according to the work done by Pienaar (1988). Centric diatoms present were counted and the ratio between the different species encountered in SEM samples was calculated from the SEM counts. These ratios were then used to calculate the concentration of each species in the original sample based on the original counts of total centric diatoms made with the aid of the light microscope. This method was first used by Pienaar (1988).

2.3 ENVIRONMENTAL VARIABLES

Physical, chemical and biological data , measured every two weeks at the Barrage sampling locality, were obtained from Rand Water, Vereeniging. Environmental variables at Parys were measured at Lindequesdrift (station no. C2H008) as well as at the Parys Municipality Water Treatment Plant by the Department of Water Affairs. Data on the chlorophyll-*a* concentration at the Parys sampling locality was obtained from Parys Municipality. Data on environmental variables at the Stilfontein sampling locality were supplied by the Department of Water Affairs as well as the Western Transvaal Regional Water Company. Physical, chemical and biological data, measured at Klipplaatdrift Weir just downstream from the Balkfontein purification plant, were obtained from the Department of Water Affairs as well as Goldfield water. All the automated and manual inorganic analytical procedures, routinely used in Rand Water as well as Western Transvaal Regional Water Company's water quality monitoring programmes, were done according to standard methods (APHA, 1989), while the methods used in the Department of Water Affairs' water quality monitoring programmes, are described by Van Vliet *et al.* (1988).

Physical data include light intensity, temperature, flow (discharge), turbidity and secchi disk depth. Chemical data include conductivity, ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), nitrite nitrogen (NO_2-N), Kjeldahl nitrogen (KN), ortho-phosphate (PO_4-P), total phosphorus (TP), sodium (Na^+), potassium (K^+), magnesium (Mg^{2+}), calcium (Ca^{2+}), chloride (Cl^-), sulphate (SO_4^{2-}), silicate-silicon (SiO_2-Si), pH and dissolved oxygen. Unfortunately no data on the bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}) ions was available during the study period. Biological data include chlorophyll-*a* concentrations. At all four sampling localities, chlorophyll-*a* concentrations were determined according to the method described by Sartory (1982).

Dissolved inorganic nitrogen (DIN) was calculated as NH_4-N plus NO_3-N plus NO_2-N (where available). Total nitrogen (TN) was calculated as KN plus NO_3-N. For the purpose of this study the DIP concentration is considered to be equivalent to the PO_4-P concentration. DIN:DIP, TN:TP and Si:DIP (mass-based) ratios were calculated from the available data.

2.4 COMPUTER PROGRAMMES AND STATISTICAL ANALYSIS OF DATA

Data on algal counts were typed into QUATTRO PRO 3. Graphical presentations were made by SIGMAPLOT 4.0, HARVARD GRAPHICS 2.3 and 3.0 and WTV TIEVEN 3D. Graphics were exported as Hewlett Packard Graphics Language (HPGL) files and as such imported into MS WORD 5.5. All programmes were registered versions available for use in the Vaal River Research group of the Department of Plant and Soil Sciences, PU for CHE, Potchefstroom.

3. RESULTS AND DISCUSSION

3.1 ENVIRONMENTAL VARIABLES

For the purpose of this discussion, environmental variables will be divided into physical variables, which include light, temperature, discharge, turbidity and secchi disk depth as well as chemical variables which include conductivity, total dissolved salts, major ions, phosphorus (DIP, TP), nitrogen (DIN, TN) and silicate-silicon concentrations, pH and dissolved oxygen. Biological variables such as chlorophyll-*a* concentration and total algal units will be discussed in section 3.2.

According to Peterson *et al.* (1987) the three major environmental variables that control photosynthesis are light, temperature and nutrient availability.

3.1.1 PHYSICAL VARIABLES

According to Straskraba (1980) energy affects water bodies mainly in three ways, namely as light, temperature and mechanical energy.

3.1.1.1 LIGHT

Variables such as discharge, sediment load (turbidity) and water chemistry influence the amount of light entering the water. The effects of dissolved organic compounds on the absorption of light energy are substantial (Walmsley & Bruwer, 1980; Wetzel, 1983).

Unfortunately no light measurements were made at the Parys and Stilfontein sampling localities, so that underwater light information was only available for the Barrage and Balkfontein sampling localities.

During 1991 at the Barrage, light measurements commenced during May (Fig. 1A). Relatively deep light penetration occurred from May to December 1991 (if compared with

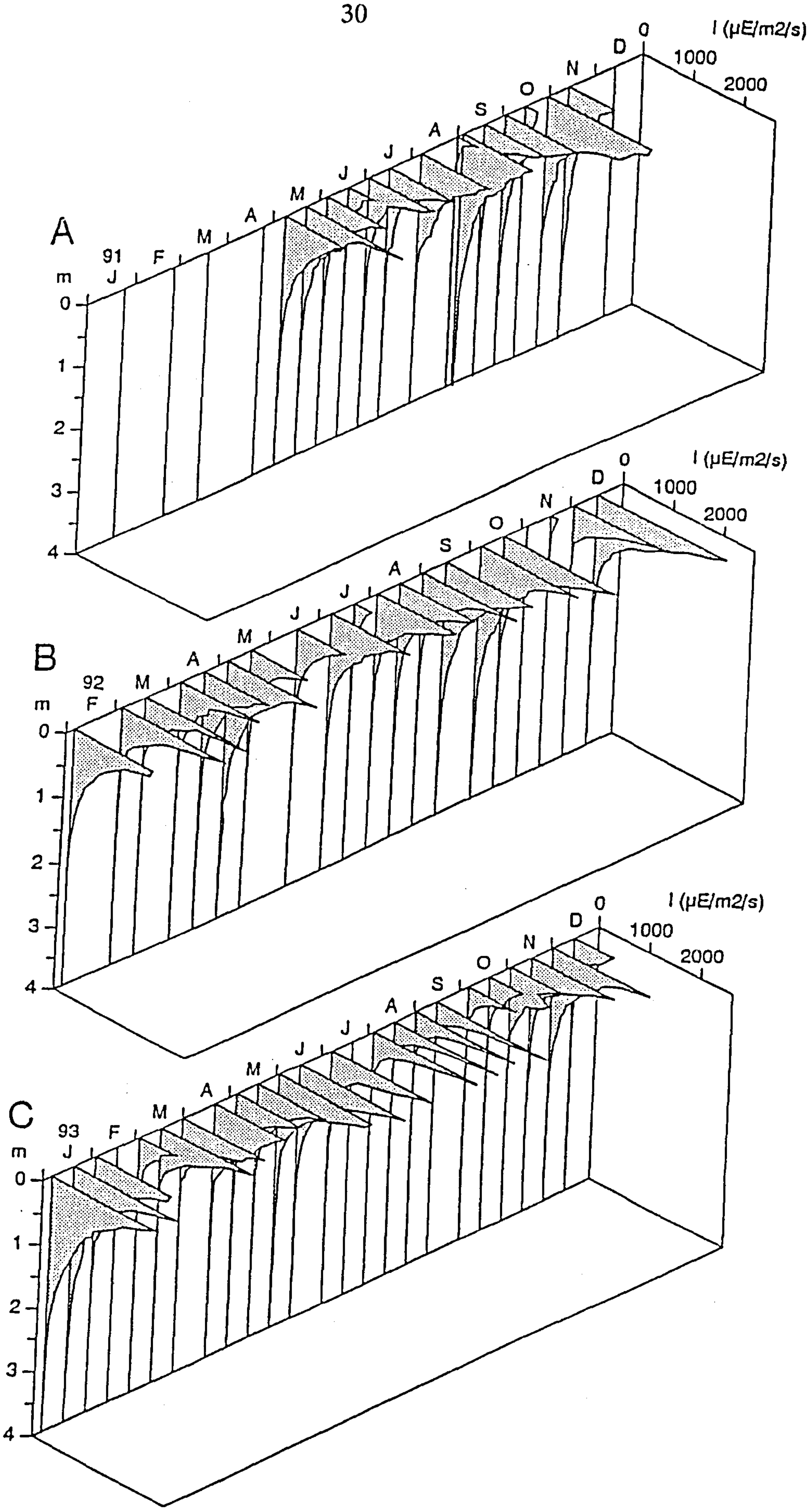

FIGURE 1: Light penetration in the Vaal River Barrage during 1991 (A), 1992 (B) and 1993 (C).

light penetration during 1992 and 1993; Figs 1B & 1C respectively). Deep light penetration can be ascribed to low discharge (Fig. 3) accompanied by low levels of turbidity (Fig. 7) in the river from May to December 1991. Periods of deepest light penetration (September and November 1991; Fig. 1A) coincided with periods when high Secchi disk depth readings occurred (Fig. 7). Low light intensities at the surface as well as low levels of light penetration during June, August and October 1991 can be ascribed to cloudy conditions. During 1992 (Fig. 1B) the low surface light intensities and shallow light penetration during July and November can be ascribed to cloudy conditions during these periods. During November 1992 the Secchi disk depth was also low (± 23 cm) at the Barrage (Fig. 7). Periods of deep light penetration (May, middle of September and October 1992; Fig. 1B) coincided with periods of low discharge and turbidity (Figs 3 and 7 respectively), as well as periods when high Secchi disk depth readings occurred (Fig. 7). Fig. 1C shows that deep light penetration occurred during January and June 1993. These periods were also characterised by low discharge and turbidity (Figs 3 and 7 respectively) as well as high Secchi disk depths (Fig. 7). Fig. 7 shows that from July 1993 an increase in turbidity, accompanied by a decrease in secchi disk depth occurred, which is also reflected in the low levels of light penetration from July to October 1993 (Fig. 1C). From November 1993 onwards, deeper light penetration occurred, because of the lowering in discharge (Fig. 3) and decreasing turbidity (Fig. 7). These data clearly show that the decrease in the depth of light penetration from July to October 1993 (Fig. 1C) is the result of an increase in turbidity (Fig 7), which is a result of an increase in discharge (Fig. 3). When discharge decreased during the beginning of November 1993, the suspended material settled out and the water became clearer (turbidity decreased; Fig. 7). This resulted in more favourable underwater light conditions (Fig. 1C) which resulted in higher algal growth.

At the Balkfontein sampling locality monthly data on light penetration and secchi disk depths were available. Fig. 2A shows that, although very high light intensities were recorded in the surface water during February and March 1991, light penetration during these two months was very shallow and rapid extinction of light penetrating the water was observed. Rapid light attenuation could be ascribed to higher levels of discharge (Fig. 6) and turbidities (Fig. 9) recorded during February and March 1991. During February and March 1991, low secchi disk depths were also recorded in the river at Balkfontein (Fig. 9). The rest of 1991 was characterised by deeper levels of light penetration (Fig. 2), the result of lower levels of discharge (Fig. 6) and turbidities (Fig. 9). High secchi disk depths were also recorded during the rest of 1991 (Fig. 9). During 1992 relative deep light penetration occurred at the Balkfontein sampling locality (Fig. 2B). Unfortunately no light measurements were made during November 1992, a period when high discharge (Fig. 6) and turbidities (Fig. 9) were recorded in the river. Although discharge and turbidity started to decrease during December 1992, relative high levels of discharge and turbidity values were still recorded in the river during December, resulting in the rapid extinction of light penetrating the water during December 1992 (compare Figs 6 & 9 with Fig. 2B). Low secchi disk depths were also recorded during the beginning of December 1992 (Fig. 9). In 1993 at the Balkfontein sampling locality, low levels of light penetration can be observed during February (Fig. 2C). During this stage an increase in discharge (Fig. 6) accompanied by very high levels of turbidity (640 NTU) were recorded in the river (Fig. 9). Fig. 9 also shows that low secchidisk depths were recorded during February. From March to September 1993, relative deep light penetration (Fig. 2C), accompanied by low discharge

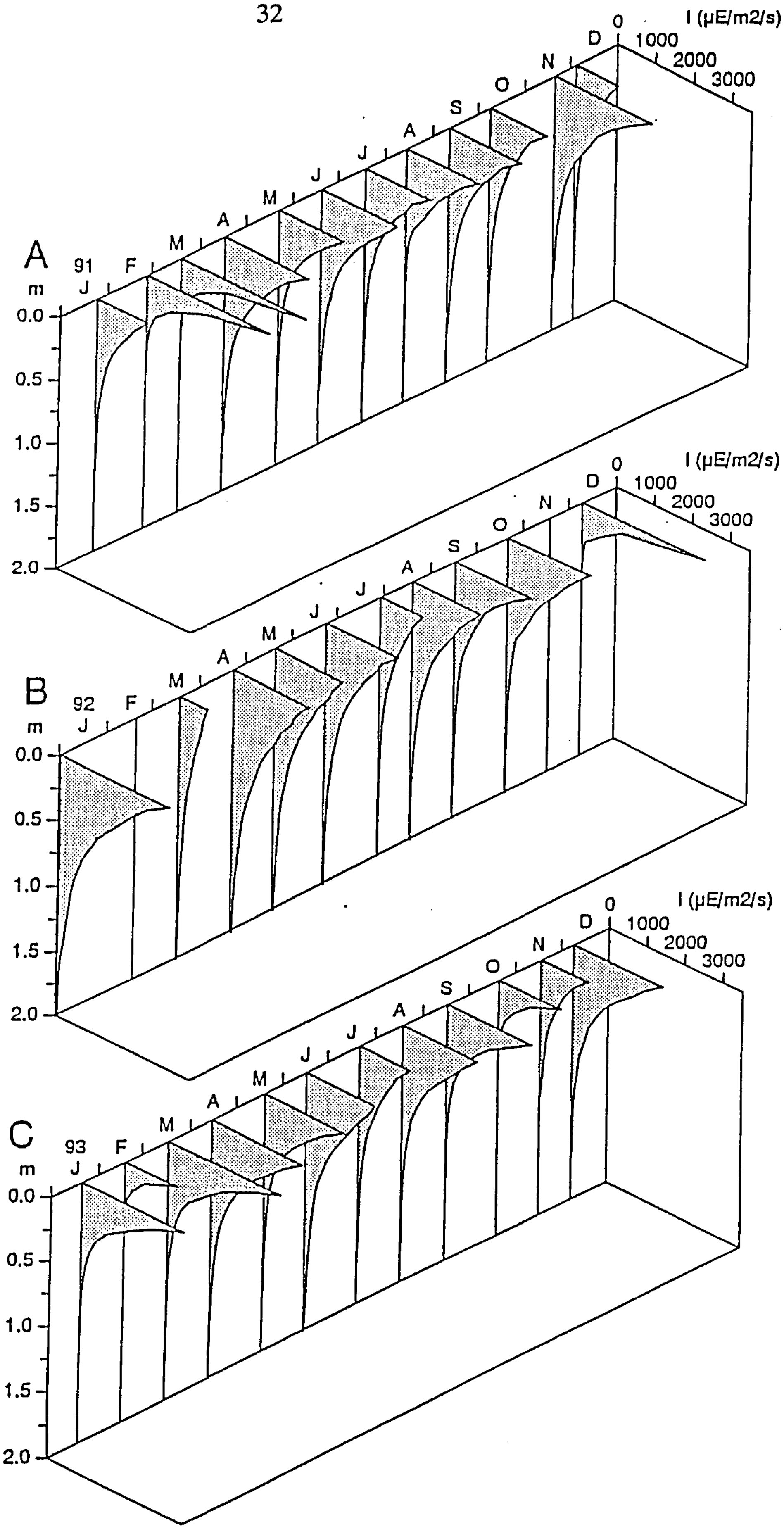

FIGURE 2: Light penetration in the Vaal River at Balkfontein during 1991 (A), 1992 (B) and 1993 (C).

and turbidity levels (Figs 6 & 9) and high secchi disk depths (Fig. 9) were present in the river. During October 1993 lower levels of light penetration (Fig. 2C) and secchi disk depths (Fig. 9) were again present, due to an increase in discharge (Fig. 6) and turbidity (Fig. 9). During November and December 1993 discharge and turbidities decreased, while the secchi disk depths and depth of light penetration showed an increase (compare Figs 6, 9 and 2C).

A comparison of Figs 1 and 2 shows that deeper levels of light penetration were recorded at the Barrage sampling locality than at the Balkfontein sampling locality. This may be ascribed to an increase in turbidity from the Barrage to Balkfontein (Fig. 10; see also 3.1.1.4). However, minimum, average and maximum light intensities recorded in the surface water of the Barrage and Balkfontein sampling localities (Table 1) show that higher light intensities were present at the Balkfontein sampling locality.

The minimum, maximum and average surface light intensities recorded during the three years of the study period at the Barrage and Balkfontein sampling localities are presented in Table 1.

TABLE 1: Minimum, maximum and average light intensities (E m^{-1} s^{-1}) recorded in the surface water of the Barrage and Balkfontein sampling localities for the three years of the study period.

	1991	**1992**	**1993**
BARRAGE:			
Minimum	250	650	650
Maximum	2060	2500	2300
Average	1177	1462	1488
BALKFONTEIN			
Minimum	1100	720	1200
Maximum	3300	3200	2900
Average	2025	1623	1971

3.1.1.2 TEMPERATURE

Water temperature was measured at three of the four sampling localities, namely at the Barrage, Stilfontein and Balkfontein sampling localities. Seasonal changes are indicated by increasing water temperatures during September, October and November (maximum temperatures from December to February) and cooling during April, May and June in the Vaal River (Figs 52, 54 and 55).

Minimum, maximum and average water temperatures at the different sampling localities are presented in Table 2.

TABLE 2: Minimum, maximum and average water temperature (°C) recorded in the surface water of the Barrage, Stilfontein and Balkfontein sampling localities for the three years of the study period.

	1991	1992	1993
BARRAGE:			
Minimum	10	9	8
Maximum	25	25	21
Average	16	17	15
STILFONTEIN:			
Minimum	11	8	10
Maximum	27	25	26
Average	18	19	19
BALKFONTEIN:			
Minimum	10	9	10
Maximum	26	26	25
Average	18	20	19

Fig. 52 shows that change in surface water temperature between winter and summer from 1991 to 1993 at the Barrage was from 8 to 25°C, at Stilfontein from 8 to 27°C (Fig. 54) and at Balkfontein from 9 to 26°C (Fig. 55). This was more or less similar to the situation in South African man-made lakes, e.g. Roodeplaat Dam (range between 13 and 26°C; De Wet, 1986) and Hartbeespoort Dam (range between 13 and 28°C; CSIR, 1985). Calculations made from data of 20 South African impoundments in Walmsley and Butty (1980) revealed similar results, i.e. the surface water temperature fluctuated between 12 ± 2°C in winter and 27 ± 3°C in summer with an annual average of 19.7 ± 2.7°C.

The elevation of water temperature affects oxygen solubility. The relationship between temperature and oxygen concentration in the Vaal River is discussed in more detail in section 3.1.2.7.

Temperature is a significant factor influencing the growth of various types of algae (Roos, 1992). All organisms have a temperature or range of temperatures at which optimal growth, reproduction and general fitness for the environment occur. Organisms (like algae) are very susceptible to changes in water temperature since a 10°C increase results in a doubling of the organism's metabolic rate (Hellawell, 1986). In the Vaal River differences between winter and summer temperatures amounted to almost 20°C, which might be an indication of a certain degree of adaption of Vaal River organisms. Certain species are adapted to very narrow ranges of temperature. As these species disappear from heated water, heat tolerant species increase in number and replace the original species in the ecosystem (Reid & Wood, 1976). The influence of temperature on species succession in the Vaal River will be discussed in section 3.2.4.2.

3.1.1.3 DISCHARGE

Flooding (extreme volumes of water flow) and changes in turbidity are of greater significance in rivers than in lakes (Hynes, 1970). During times of high water discharge, the substratum of the river channel may be scoured off and many animals and plants are suspended as well. Flood periods can thus result in an increase in turbidity, both physical and biological.

In the drainage area of the Vaal River rainfall is almost exclusively a summer phenomenon. Roos (1992) showed that only about 56% of the variation in discharge in the Vaal River at Balkfontein could be explained by the precipitation measured at Bothaville. Precipitation at Bothaville thus had only a negligible effect on flow in the Vaal River. Any correlation is likely to be due to widespread rain effluents. Although precipitation was negligible during the winter months from 1985 to 1989 at Balkfontein, discharge was high (Roos, 1992). The high discharge during winter months is due to the regulation of flow in the system by release of water from the Vaal Dam and the Vaal River Barrage. Minimum, maximum and average discharge values presented in Table 3 indicate that discharge in the Vaal River from 1991 to 1993 was relatively low compared to discharge measured from 1985 to 1989 by Roos (1992), but the phenomenon of winter (non-rainfall season) peaks in discharge could also be seen during the present study, which indicated releases of water from the Vaal Dam.

TABLE 3: Minimum, maximum and average discharge ($m^3 s^{-1}$) recorded in the water of the Barrage, Parys, Stilfontein and Balkfontein sampling localities for the three years of the study period.

	1991	1992	1993
BARRAGE:			
Minimum	5.9	8.4	10.0
Maximum	29.8	55.5	54.5
Average	1.6	19.5	20.8
PARYS:			
Minimum	-	-	7.6
Maximum	-	-	43.9
Average	-	-	18.8
STILFONTEIN:			
Minimum	4.5	2.5	7.1
Maximum	144.4	57.2	66.9
Average	16.0	15.0	19.6
BALKFONTEIN:			
Minimum	0.5	0.3	1.7
Maximum	102.0	102.0	82.6
Average	15.0	12.6	19.1

Periods of high flow rates measured at the Barrage were also reflected in the downstream sampling localities. During 1991 a maximum flow rate at the Barrage was recorded during March (Fig. 3). High flow rates during March were also observed at the Stilfontein (Fig. 5) and Balkfontein (Fig. 6) sampling localities during 1991. During 1992 the flow rate at the Barrage, Stilfontein and Balkfontein sampling localities was low from January to June after which three peaks occurred during July, September and November at the Barrage and Stilfontein (Figs 3 and 5 respectively) and two peaks at Balkfontein (September and November; Fig. 6).

Discharge has also an effect on nutrients and salinity. During periods when an increase in flow rate occurred, decreases in conductivity, TDS concentration and concentration of major ions were observed in the Vaal River (see sections 3.1.2.1 and 3.1.2.2). Discharge peaks coincided with peaks of nutrients (e.g. DIN and DIP; see section 3.1.2.3).

Discharge (according to Lack *et al.*, 1978; Petts, 1984) and water velocity have been proved to be important variables influencing phytoplankton. An increase in flow-rate results in increasing turbidity, which, in turn, results in decreasing light penetration and lower underwater light available for algal growth and photosynthesis. If flooding occurs, the phytoplankton could be completely washed out of the system. The influence of discharge on phytoplankton dynamics will be discussed in section 3.2.4.1.

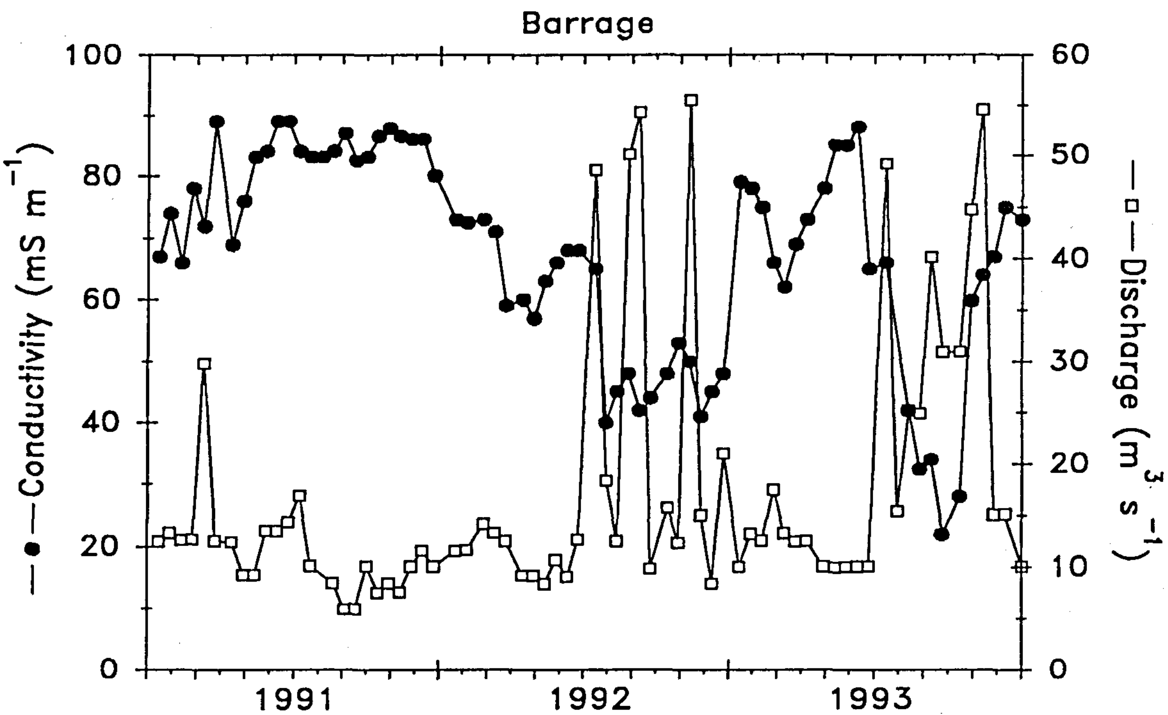

FIGURE 3: Variation in conductivity (mS m^{-1}) and discharge (m^3 s^{-1}) at the Barrage during the study period.

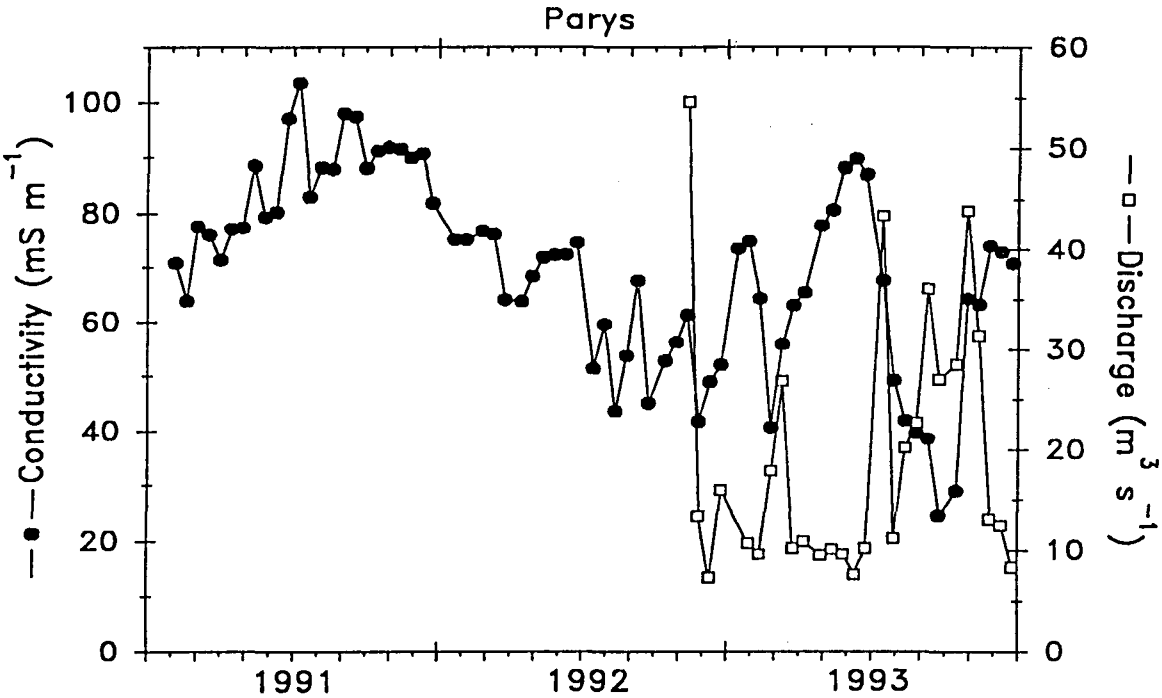

FIGURE 4: Variation in conductivity (mS m^{-1}) and discharge (m^3 s^{-1}) at Parys during the study period.

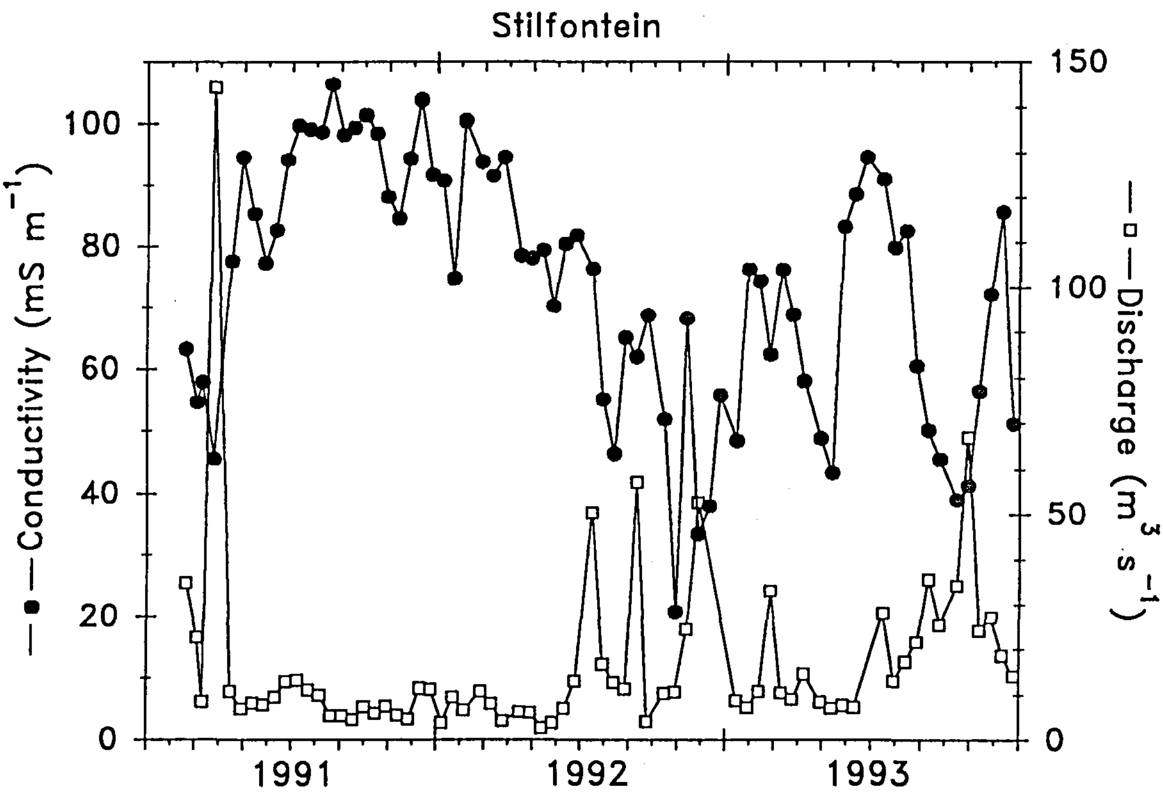

FIGURE 5: Variation in conductivity (mS m^{-1}) and discharge (m^3 s^{-1}) at Stilfontein during the study period.

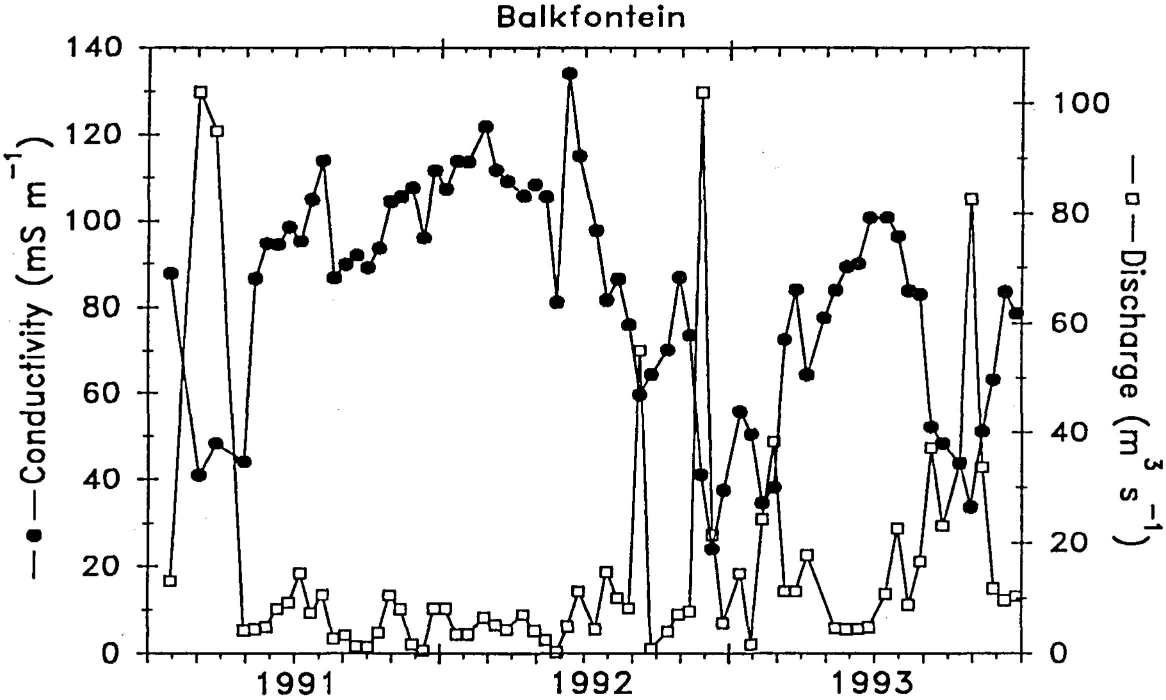

FIGURE 6: Variation in conductivity (mS m^{-1}) and discharge (m^3 s^{-1}) at Balkfontein during the study period.

3.1.1.4 TURBIDITY (NTU) AND SECCHI DISK DEPTH

Turbidity

The most important variable that influences turbidity is discharge (Roos, 1992; WHO, 1984). Natural seasonal variations in rivers often result in changes in turbidity (Harrison & Elsworth, 1958).

In clear water bodies like in certain regions of the northern hemisphere, turbidity is primarily determined by organic suspensoids (Robarts, 1979; Septhon & Harris, 1984; Tilzer & Beese, 1988 and Kirk, 1985). In contrast, it is the concentration of predominantly inorganic particles, derived from soils in the catchment, that dominates in African and Australian waters and thus distinguishes them from turbid waters in the northern hemisphere. An important feature of many South African reservoirs is high turbidity caused by the presence of suspended silt (Walmsley, 1978). For this reason, turbidity in the southern hemisphere is generally considered to be equivalent to some measure of the concentration of suspensoids.

Turbidity is usually much higher in rivers than in lakes, especially during periods of high discharge (Talling & Rzòska, 1967). At the Barrage, Stilfontein and Balkfontein sampling localities a positive correspondence between turbidity and discharge can be seen (compare Figs 7 with 3, 8 with 5 and 9 with 6). During the present study (1991-1993) on the Vaal

River, the average turbidity recorded at the Barrage was 19.5 NTU (Fig. 7), at Stilfontein 17.9 NTU (Fig. 8) and at Balkfontein 59 NTU (Fig. 9).

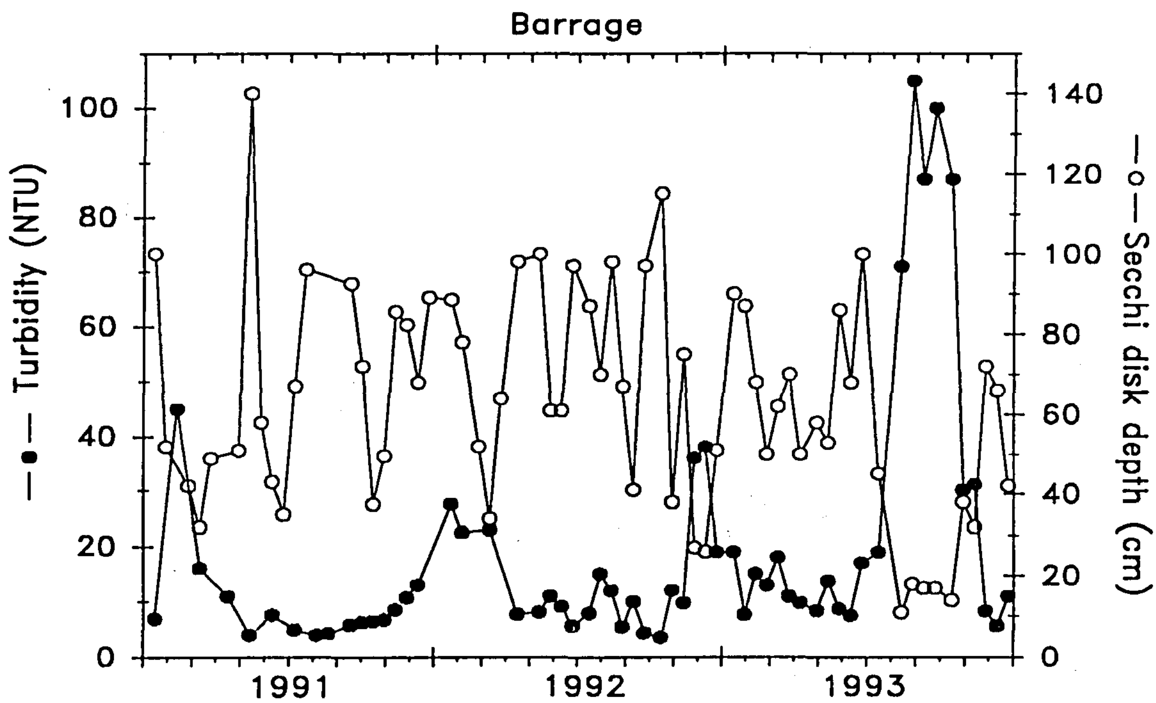

FIGURE 7: Variation in turbidity (NTU) and secchi disk depth (cm) at the Barrage during the study period.

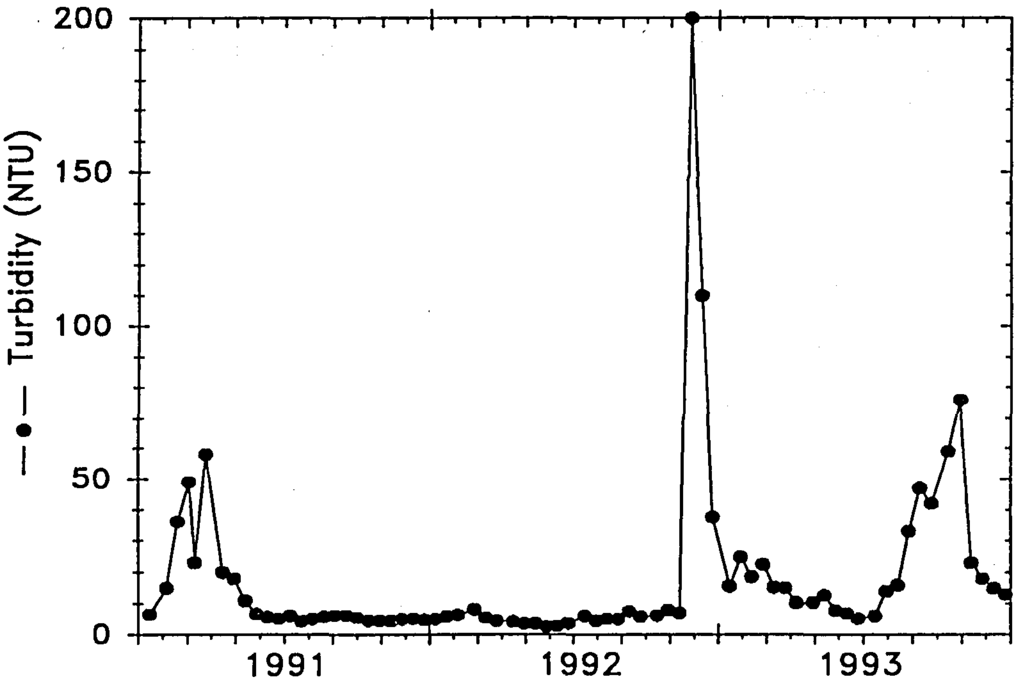

FIGURE 8: Variation in turbidity (NTU) at Stilfontein during the study period.

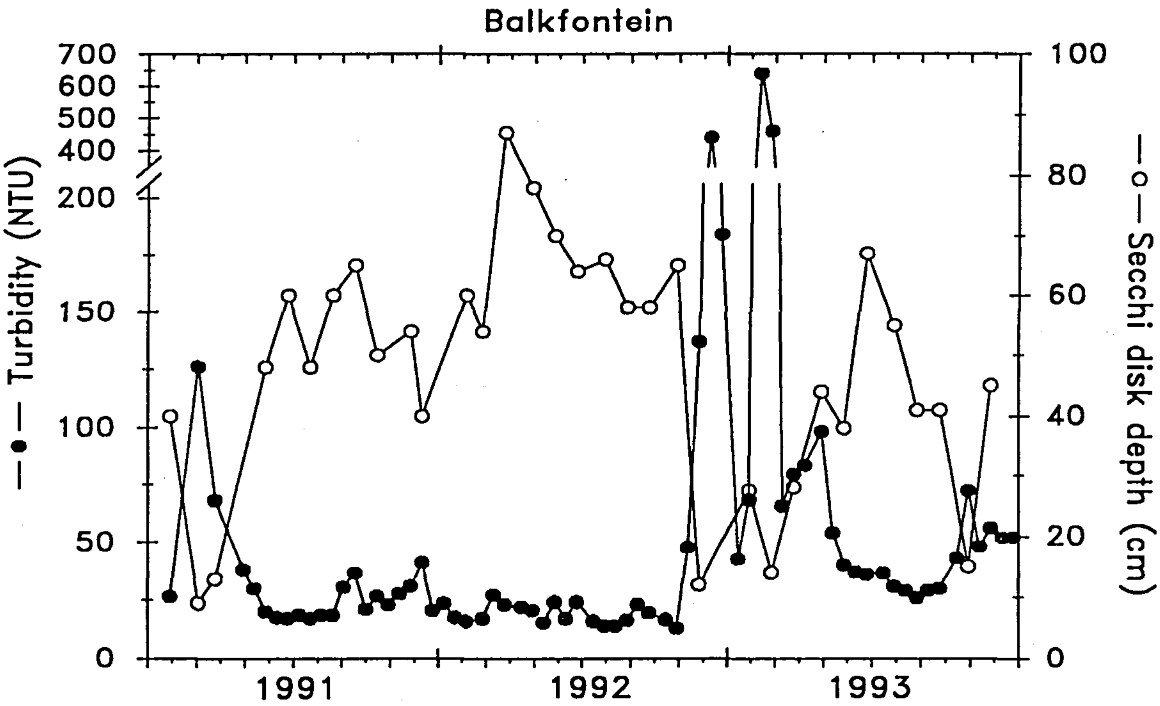

FIGURE 9: Variation in turbidity (NTU) and secchi disk depth (cm) at Balkfontein during the study period.

Minimum, maximum and average turbidities for each year at the different sampling localities are given in Table 4.

TABLE 4: Minimum, maximum and average turbidity (NTU) and Secchi disk depths (cm) recorded in the water of the Barrage, Stilfontein and Balkfontein sampling localities for the three years of the study period.

	1991		**1992**		**1993**	
	Turb.	**Secchi**	**Turb.**	**Secchi**	**Turb.**	**Secchi**
BARRAGE:						
Minimum	4	32	4	26	6	11
Maximum	45	140	38	115	105	100
Average	10	67	14	69	31	53
STILFONTEIN:						
Minimum	4	-	2	-	5	-
Maximum	58	-	200	-	76	-
Average	13	-	19	-	22	-
BALKFONTEIN:						
Minimum	17	9	13	12	26	14
Maximum	126	65	440	87	640	67
Average	32	44	50	61	92	38

Results in Table 4 as well as Fig. 10 clearly indicate that a definite downstream increase in average turbidity occurred from the Barrage to Balkfontein sampling localities during 1991 and 1992. During 1993 the average turbidity at the Barrage sampling locality (30.6 NTU) was a few NTU's higher than the average turbidity recorded at Stilfontein (22 NTU), but from Stilfontein the turbidity increased to Balkfontein (92 NTU). Furthermore, a definite increase in turbidity at all three sampling localities (especially Balkfontein) can be seen from 1991 to 1993 (Fig. 10 and Table 4).

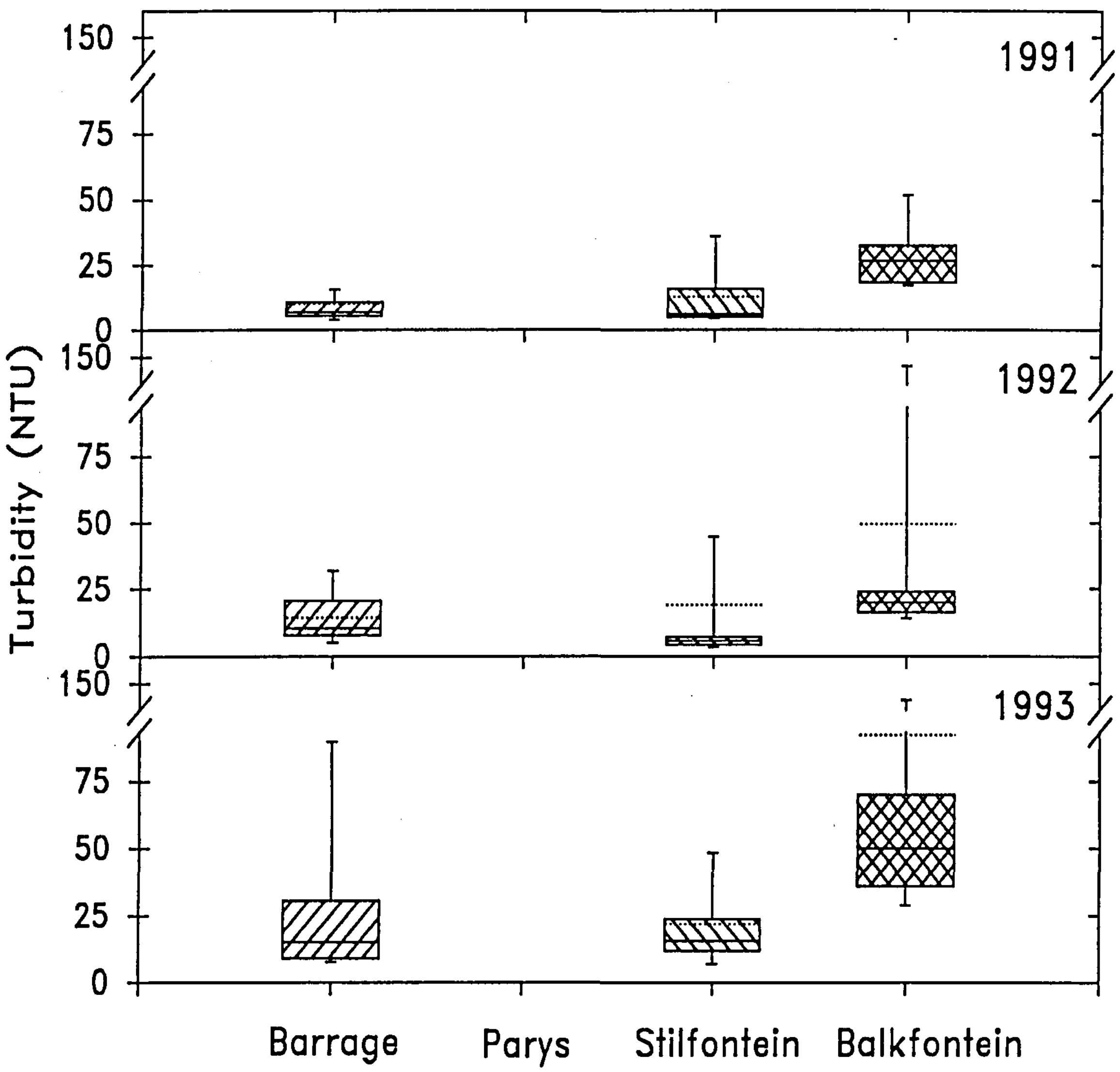

FIGURE 10: Box plot of annual turbidity (NTU) in the Vaal River at the Barrage, Parys, Stilfontein and Balkfontein. The box represents the 25th through 75th percentiles. The solid line in the box represents the median and the dotted line the average value.

Suspensoids have a marked effect on primary production, not only by restricting light penetration, but also by absorbing and re-releasing nitrogen and phosphorus compounds necessary for plant growth (Noble & Hemens, 1978). Nutrients, heavy metals, pesticides and other toxins preferentially adsorb on suspensoids and are transported in the adsorbed form (Roos, 1992).

Turbidity in the Vaal River shows an opposite trend to chlorophyll-*a* concentration, i.e. increases in turbidity accompany decreases in chlorophyll-*a* concentration. Turbidity can be regarded as beneficial from a water quality point of view, because it suppresses algal growth by limiting underwater light availability (Braune & Rogers, 1987). Wofsy (1983) concluded that suspended sediment concentrations above 50 mg l^{-1} prevent significant algal blooms in all but the shallowest streams.

Secchi disk depths

An old, but still very much utilised and valuable method of measuring light penetration, i.e. transparency (influenced by turbidity), is the use of the Secchi disk. Secchi disk depth represents an approximation of light attenuation in water (Golterman, 1975a).

Both discharge and turbidity showed a negative correspondence with the Secchi disk depth (compare Figs 3 & 7 and 6 & 9). Periods of high Secchi disk depth readings (Figs 7 & 9) coincided with periods of low turbidity (Figs 7 & 9) and discharge (Figs 3 & 6).

An increase in total dissolved salts (TDS) concentration or conductivity can also result in a decrease in turbidity (compare Figs 11, 13 & 14 with Figs 7, 8 & 9 respectively). This is in accordance with the findings of Bruwer *et al.* (1985) who mentioned that it appears that higher secchi values are associated with the higher TDS values in the lower Vaal River. Randall and Day (1987) also showed that salinity was correlated with secchi depth in a turbid Louisiana (USA) estuary. More information about the relationship between conductivity and turbidity is given in section 3.1.2.1.

The influence of turbidity on the algal concentration and biomass will be discussed in section 3.2.4.1.

3.1.2 CHEMICAL VARIABLES

3.1.2.1 CONDUCTIVITY AND TOTAL DISSOLVED SALTS

The total amount of material dissolved in water is commonly measured in one of three ways: as total dissolved salts (TDS), as conductivity, or as salinity, all of which correlate closely in most waters.

The most common dissolved substances are usually the cations Na^+, K^+, Ca^{2+} and Mg^{2+} and the anions HCO_3^- (bicarbonate), CO_3^{2-} (carbonate), Cl^- and SO_4^{2-} (Dallas & Day, 1993).

Since the majority of substances dissolved in most waters are ionic, TDS and conductivity usually correlate closely in a particular type of water. In the Vaal River a positive correspondence between conductivity and TDS concentration was obvious for the Parys, Stilfontein and Balkfontein sampling localities (see Figs 12-14).

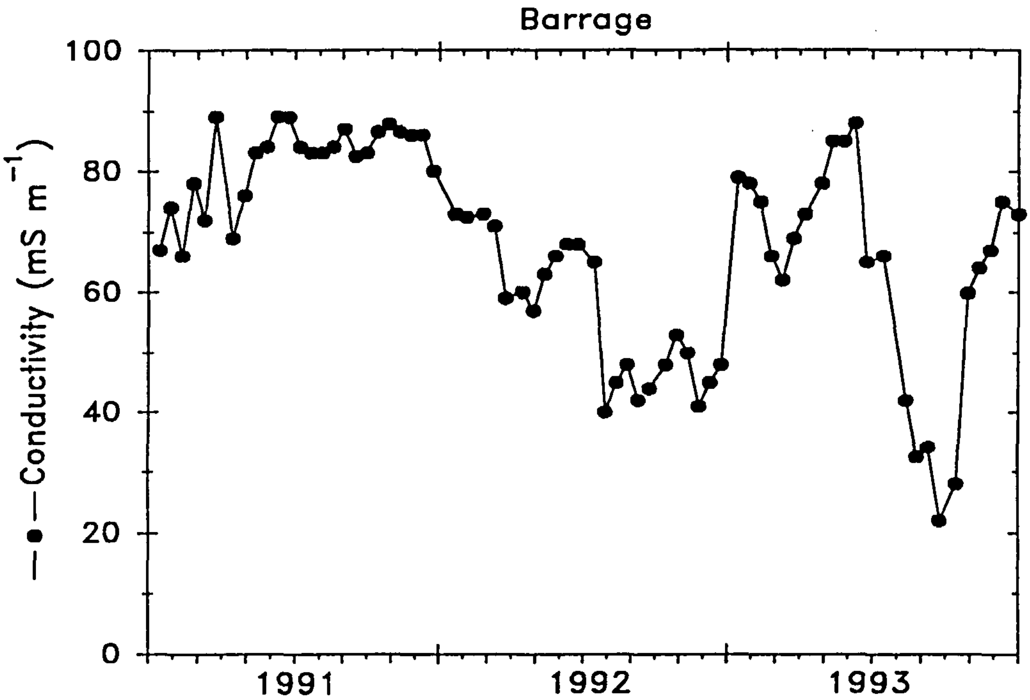

FIGURE 11: Variation in conductivity (mS m^{-1}) at the Barrage during the study period.

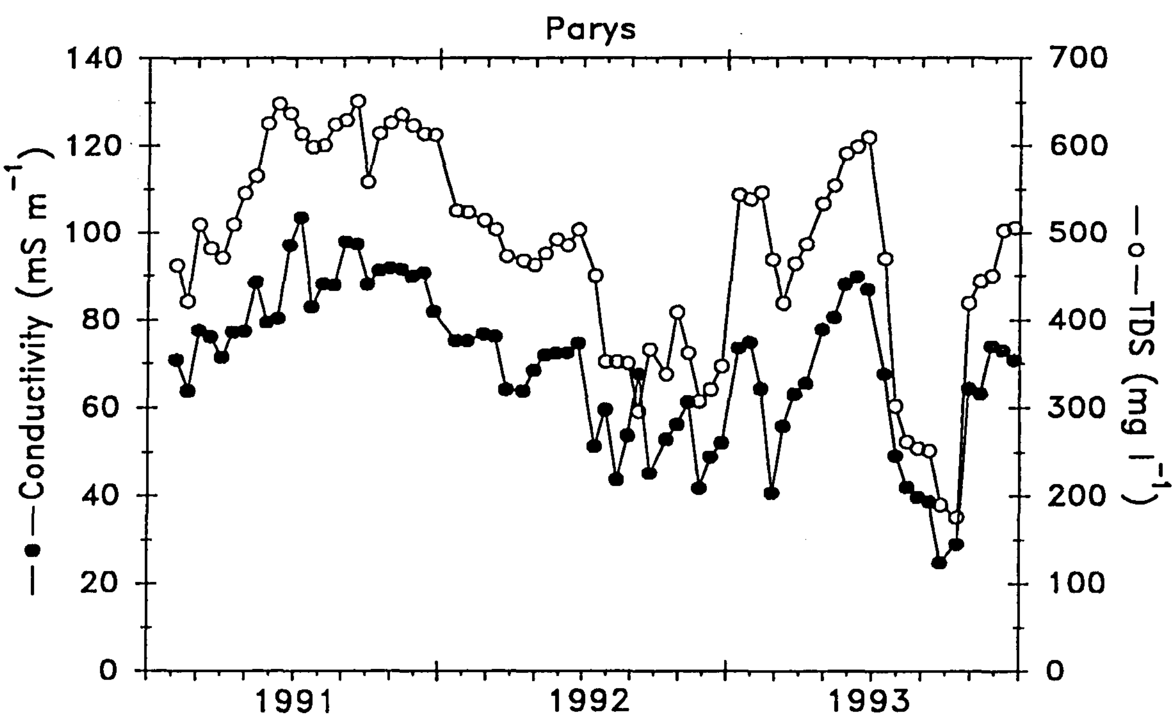

FIGURE 12: Variation in conductivity (mS m^{-1}) and TDS concentration (mg l^{-1}) at Parys during the study period.

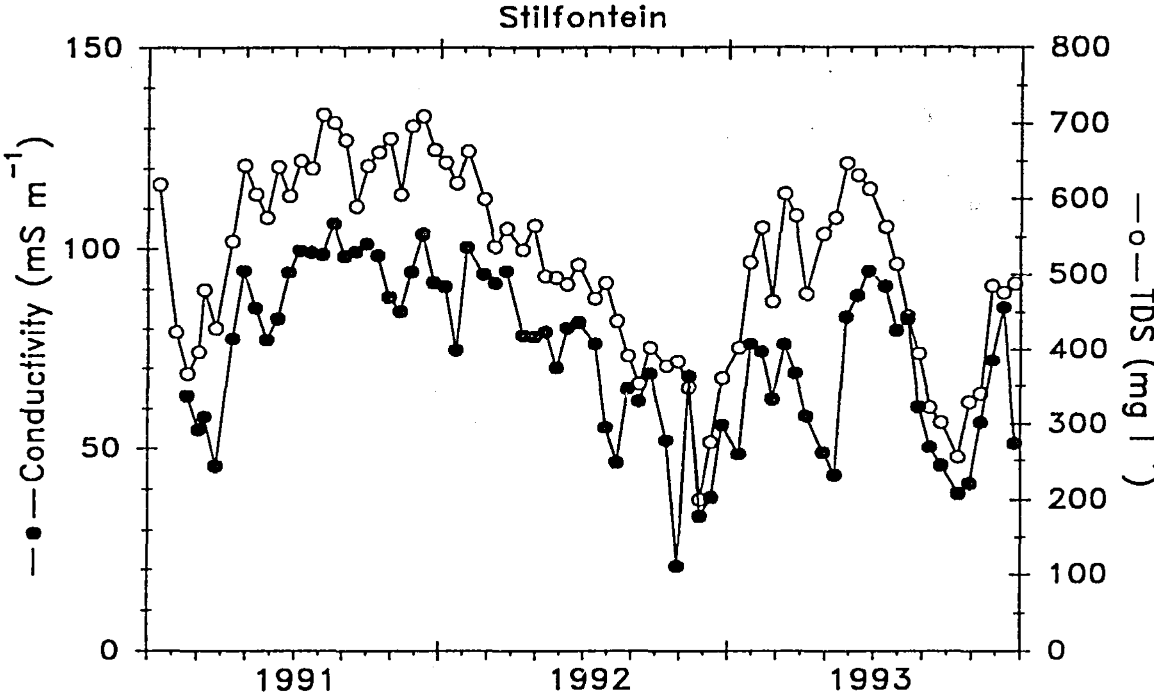

FIGURE 13: Variation in conductivity (mS m^{-1}) and TDS concentration (mg l^{-1}) at Stilfontein during the study period.

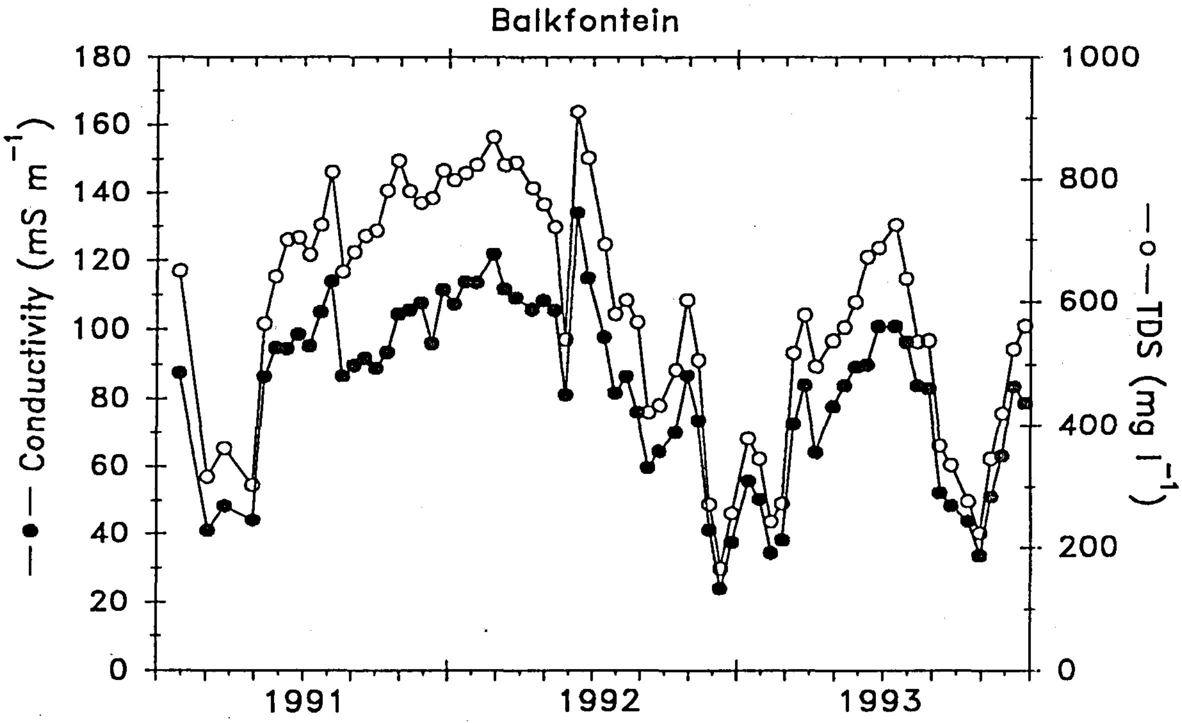

FIGURE 14: Variation in conductivity (mS m^{-1}) and TDS concentration (mg l^{-1}) at Balkfontein during the study period.

Calculation of data from Swedish bicarbonate fresh water systems (Wetzel, 1983) showed that the conversion factor of conductivity to salinity varies between 5.3 and 6.2. Kempster and Van Vliet (1991) specify a conversion factor of 6.5 between electrical conductivity and TDS. The DWA (1986) showed that the electrical conductivity limit for potable water of 70 mS m^{-1} is equivalent to TDS concentrations of between 350 mg l^{-1} and 550 mg l^{-1}, depending on constituents, i.e. a conversion factor between 5 and 7.9. This is in accordance with the situation found in the Vaal River during the present study. From 1991 to 1993 an average conversion factor from conductivity to salinity of 6.9 was recorded at Parys (ranging from 5.9 to 11.5), 7.1 at Stilfontein (ranging from 5.1 and 9.4; one exceptionally high value of 18.4 during 1992 was ignored because of possible methodological problems) and 7.1 at Balkfontein (ranging from 6.3 to 8.4). The above averages show that more or less the same conversion factor was recorded at all three sampling localities where TDS and conductivity were measured. It seems as if a small increase in the conversion factor occurred from the period 1985-1989 to 1991-1993 at the Balkfontein sampling locality, because Roos (1992) found a conversion factor ranging from 5.3 to 8.7 (average 6.72) at Balkfontein from 1985-1989. The exact value of the conversion factor used to relate electrical conductivity to total dissolved salts depends on the composition of the water, especially pH and the bicarbonate content. Low pH waters usually have a factor smaller than 6.5 (DWA, 1993a).

Talling (1980) showed that salinity was inversely related to discharge in the Tigris and Euphrates rivers. The dilution effect of flood-water is commonplace in rivers; the Nile provides another example (Talling, 1976b). During the present study, a negative correspondence between conductivity and discharge was seen at all four sampling localities (Figs 3-6). During 1991 it seems as if there was a trend of increasing conductivity with time, while 1992 showed a decreasing trend in conductivity at all four sampling localities (Figs 3-6). The increasing trend during 1991 and decreasing trend during 1992, were accompanied by a general decrease in flow rate from the beginning to the end of 1991 and a general increase from the beginning to the end of 1992. Maximum discharge for 1991 was recorded during March. At this stage the conductivity levels were low (Fig. 3-6). During 1992 increases in discharge during July, September and November, were accompanied by lower conductivity values. Maximum conductivity for 1993 at all four sampling localities was recorded during June (flow rate low), whereafter a decrease in conductivity occurred, accompanied by a general increase in flow rate, until September. From November 1993 the flow rate decreased, together with an increase in conductivity (Figs 3-6). An increase in discharge has a dilution effect on the TDS, resulting in lower TDS concentrations as well as lower electrical conductance of the water.

Several authors demonstrated a correspondence between conductivity and turbidity of the water. Grobler *et al.* (1983) pointed out that an increase in cation concentration, indicated by increased electrical conductivity, should enhance flocculation of clay particles, and should therefore result in a reduction of turbidity due to an increased settling rate of suspended clay particles. Egborge (1971) also showed a highly positive correlation between transparency and electrical conductivity (in the River Oshun, Nigeria). Therefore, the higher the concentration of suspended matter, the greater the number of ions absorbed and *vice versa.* An increase in conductivity and/or TDS concentration was frequently accompanied by a decrease in the turbidity of the water during the present study on the

Vaal River. At the Barrage, increases in conductivity from December 1992 to January 1993 and from September to December 1993 were probably the reason for decreases in turbidity during the same time (compare Fig. 3 with Fig. 7). At the Stilfontein sampling locality certain periods, showing an increase in conductivity and decrease in discharge (e.g. from March to May 1991, December 1992 to January 1993 and October to December 1993) were accompanied by a decrease in turbidity (compare Fig. 5 with Fig. 8). At the Balkfontein sampling locality the same tendency could be seen, where an increase in conductivity and decrease in discharge (e.g. from the end of February to June 1991, December 1992 to January 1993 and from February to March 1993; Fig. 6) was accompanied by a decrease in turbidity during the same period (compare to Fig. 9). This relationship between conductivity and turbidity was also confirmed by Roos (1992) on work done at Balkfontein from 1985 to 1989.

In South Africa the lowest conductivity values recorded in inland fresh water systems are about 0.9 to 3.6 mS m^{-1} (10-27 mg l^{-1}; Waterkloof stream, Transvaal; DWAF data-base) and 1.8 to 3.1 mS m^{-1} (17-37 mg l^{-1}; the Swartboskloof Stream near Stellenbosch; Britton, 1991). The highest recorded value is that for Burgerspan (9790 mS m^{-1}, ca 65 000 mg l^{-1}) in the south western Cape (Silberbauer & King, 1991). One of the rivers in South Africa, for which the highest salinities have regularly been recorded, is the Sak River near Williston in the Karoo (maximum recorded values 7684 mS m^{-1}; Dallas & Day, 1993). Minimum, maximum and average conductivity and TDS values at the different sampling localities for 1991, 1992 and 1993 are given in Table 5.

TABLE 5: Minimum, maximum and average conductivity (mS m^{-1}) and total dissolved salts (TDS; mg l^{-1}) recorded in the water of the Barrage, Parys, Stilfontein and Balkfontein sampling localities for the three years of the study period.

	1991		**1992**		**1993**	
	Cond.	**TDS**	**Cond.**	**TDS**	**Cond.**	**TDS**
BARRAGE:						
Minimum	66	-	40	-	22	-
Maximum	89	-	73	-	88	-
Average	81	-	57	-	64	-
PARYS:						
Minimum	63	431	42	296	25	176
Maximum	104	652	77	526	90	610
Average	85	579	62	421	62	441
STILFONTEIN:						
Minimum	46	366	21	200	39	255
Maximum	106	712	101	663	95	647
Average	87	598	69	467	66	481
BALKFONTEIN:						
Minimum	41	303	24	166	34	223
Maximum	114	831	134	910	101	724
Average	90	664	89	629	69	474

At the Barrage no TDS measurements were made, but mean conductivity measured at the Barrage from 1991 to 1993 was 67.6 mS m^{-1} (Fig. 11). Average TDS concentration at Parys was 481 mg l^{-1} (conductivity: 70 mS m^{-1}; Fig. 12), at Stilfontein 517 mg l^{-1} (conductivity: 74 mS m^{-1}; Fig. 13) and at Balkfontein 586 mg l^{-1} (conductivity: 82 mS m^{-1}; Fig. 14). From the above averages recorded, as well as Fig. 15 and Table 5, it is clear that a downstream increase in the conductivity and TDS concentration occurred from the Barrage to Balkfontein during all three years of the study period.

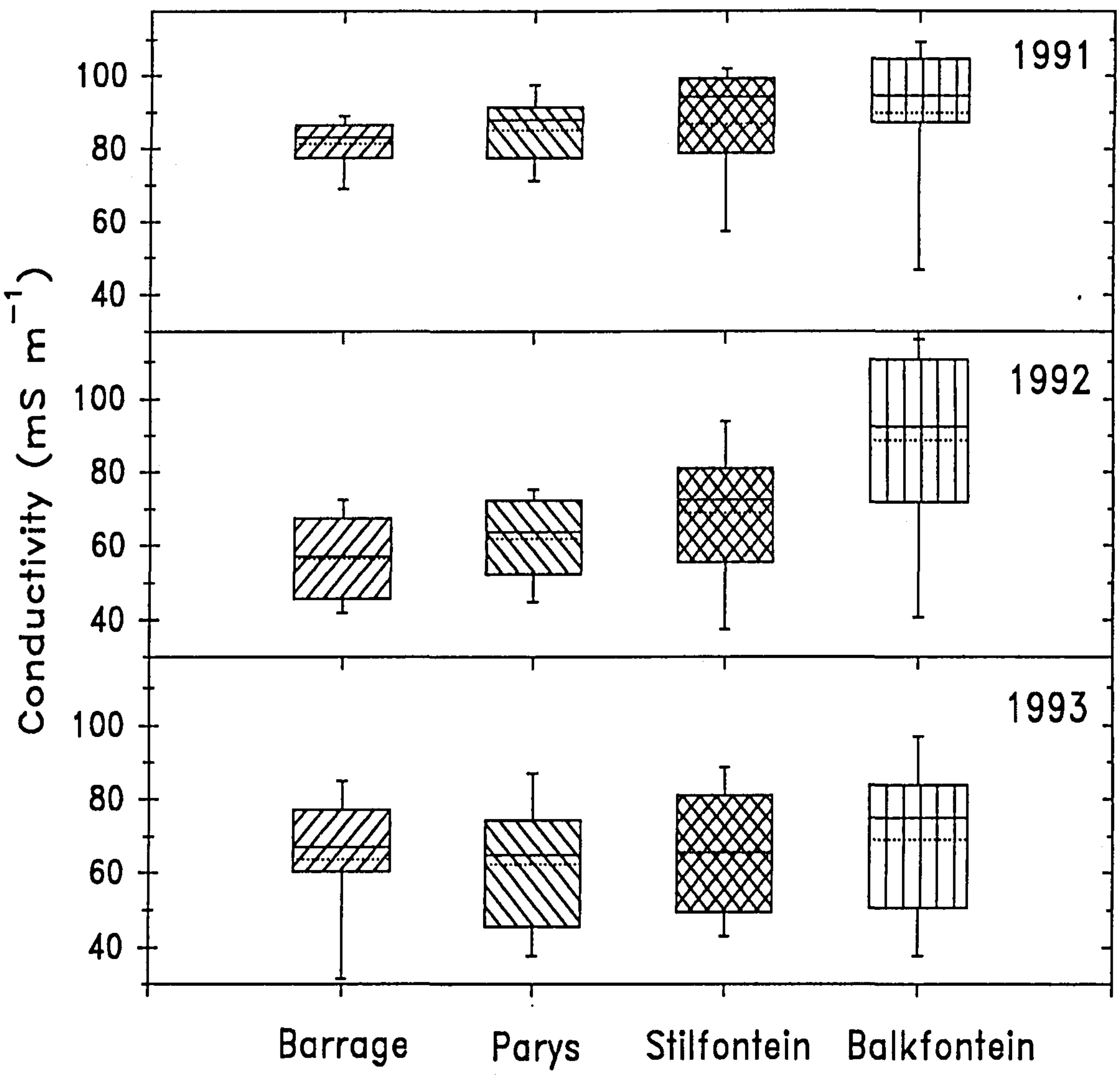

FIGURE 15: Box plot of annual conductivity (mS m^{-1}) in the Vaal River at the Barrage, Parys, Stilfontein and Balkfontein.

Upstream from the Barrage sampling locality, even lower average TDS concentrations and conductivity levels were recorded, namely 23 mS m^{-1} in the Vaal Dam and 9 mS m^{-1} in the Sterkfontein Dam (Dörgeloh *et al.,* 1993). The downstream increase in TDS and conductivity confirms the results of Van Vliet and Nell (1986) who stated that most of the rivers in the Upper Vaal catchment are generally of a good mineral quality and rivers in upper catchments are characterised by a relatively low TDS concentration. The water quality deteriorates as the concentration of dissolved salts increases in a downstream direction by the input of mining, industrial and treated sewage effluents. The longitudinal distribution of salinity or electrical conductivity has been described by various authors (e.g. Al-Sahaf, 1975; Arndt & Al-Saadi, 1975; Maulood *et al.*, 1979). In the Tigris and Euphrates Rivers the salinity also increased during flow southwards (Talling, 1980).

In the Vaal River a decrease in the average conductivity was observed from 1991 to 1993 (Fig. 15). The decrease in conductivity from 1991 to 1993 could possibly explain the increase in turbidity during the same period (Fig. 10). During 1991 higher TDS concentrations were present, which could have been responsible for flocculation and settlement of suspended material. The lower TDS concentrations present during 1993 could have resulted in less efficient flocculation of suspended material so that higher turbidities were recorded. However, the increase in conductivity and TDS from the Barrage to Balkfontein sampling localities (Fig. 15) cannot explain the downstream increase in turbidity (Fig. 10). This shows that other environmental variables influencing turbidity must also be tak
en into account. Because most rivers and large lakes world-wide have TDS values < 100 mg l^{-1} (mean volume-weighted average for rivers; Hutchinson, 1975), the Vaal River can be regarded as a river with a relatively high salinity.

Perhaps the two environmental variables that most decisively determine the communities of organisms living in a particular aquatic ecosystem are flow rate and salt concentration (Dallas & Day, 1993). Numerous studies (i.e. Wood & Talling, 1988; Hart *et al.*, 1991) have shown that the individual species making up the faunas and floras of lakes and estuaries have very distinctive salinity tolerances. More information about the effect of salinity on riverine organisms is given in section 3.2.4.3 where salinity and phytoplankton populations will be considered.

3.1.2.2 MAJOR IONS

In South Africa the waters of the highveld tend to be dominated by calcium (Ca^{2+}), magnesium (Mg^{2+}) and bicarbonate (HCO_3^-) ions, whereas those of the coastal regions and the arid west tend to be dominated by sodium (Na^+) and chloride (Cl^-) ions (Dallas & Day, 1993). Averages presented in Table 6 show that the concentration of major cations at the Barrage shows proportions of $Ca^{2+} > Na^+ > Mg^{2+} > K^+$ (Fig. 16), while at the other three sampling localities (Parys, Stilfontein and Balkfontein) proportions of $Na^+ > Ca^{2+} > Mg^{2+} > K^+$ existed (Figs 17-19). At all four sampling localities in the Vaal River the proportion of anions were $SO_4^{2-} > Cl^-$ (Figs 20-23).

The cation and anion proportions in the Vaal River differ from the proportions illustrated in freshwater. The mean composition of the major cations of river water has been given as $Ca^{2+} > Mg^{2+} \geq Na^+ > K^+$ (Gibbs, 1970). The relative high Na^+ concentration in the Vaal River is responsible for the difference in proportions. The concentration of major anions of many surface waters of the world tends to exist in the proportion of CO_3-HCO_3^- > $SO_4^{2-} > Cl^-$ (Wetzel, 1983).

TABLE 6: Minimum, maximum and average concentrations of major ions (mg l^{-1}) recorded in the water of the Barrage, Parys, Stilfontein and Balkfontein sampling locations for the three years of the study period.

	1991						1992						1993					
	Ca^{2+}	Na^{+}	Mg^{2+}	K^{+}	SO_4^{2-}	Cl^{-}	Ca^{2+}	Na^{+}	Mg^{2+}	K^{+}	SO_4^{2-}	Cl^{-}	Ca^{2+}	Na^{+}	Mg^{2+}	K^{+}	SO_4^{2-}	Cl^{-}
BARRAGE:																		
Minimum	49	41	17	10	115	48	29	30	13	7	32	25	17	14	6	4	26	5
Maximum	69	70	29	19	275	80	50	60	18	13	128	83	66	61	21	13	200	82
Average	63	61	24	12	176	67	39	43	15	9	73	51	15	43	16	10	137	43
PARYS:																		
Minimum	53	44	17	10	153	48	32	31	12	8	60	27	20	17	7	6	34	18
Maximum	73	82	27	16	137	92	55	70	21	14	148	70	75	74	23	16	211	86
Average	65	69	23	13	197	76	44	51	16	11	106	50	53	51	17	11	138	55
STILFONTEIN:																		
Minimum	42	39	15	9	130	10	26	19	10	5	56	33	24	22	9	6	72	37
Maximum	78	94	42	15	312	130	68	95	34	16	244	85	79	83	35	16	232	88
Average	65	73	31	12	235	64	53	53	26	12	162	64	56	55	21	10	164	60
BALKFONTEIN:																		
Minimum	26	29	13	7	64	30	20	12	6	5	29	11	26	20	9	5	51	17
Maximum	86	98	45	20	331	106	99	112	41	22	319	114	89	83	32	17	278	91
Average	69	77	33	12	260	81	65	75	27	14	199	76	56	52	20	10	156	53

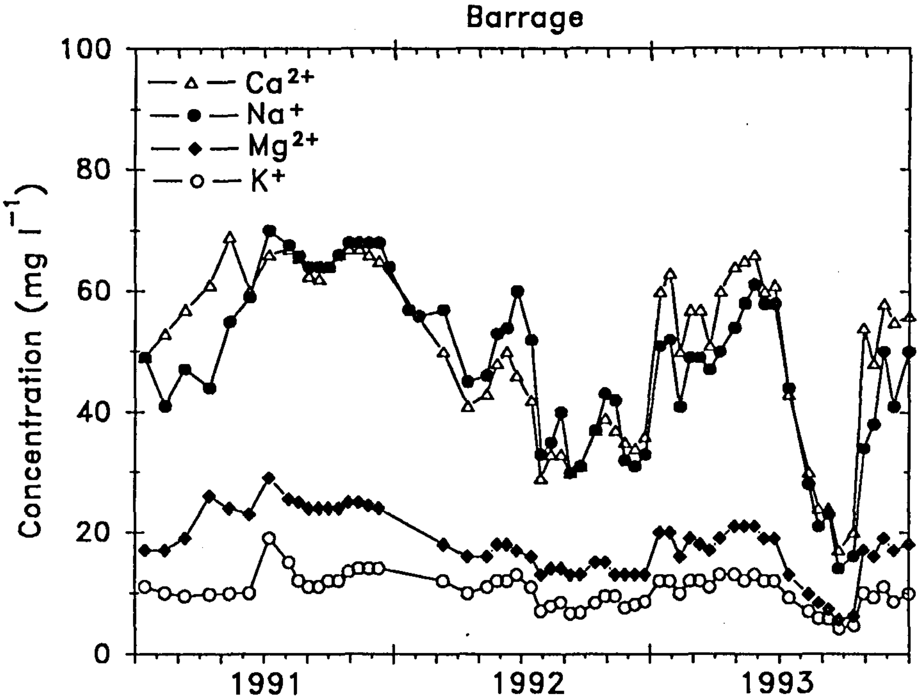

FIGURE 16: Variation in cation concentrations (mg l^{-1}) at the Barrage during the study period.

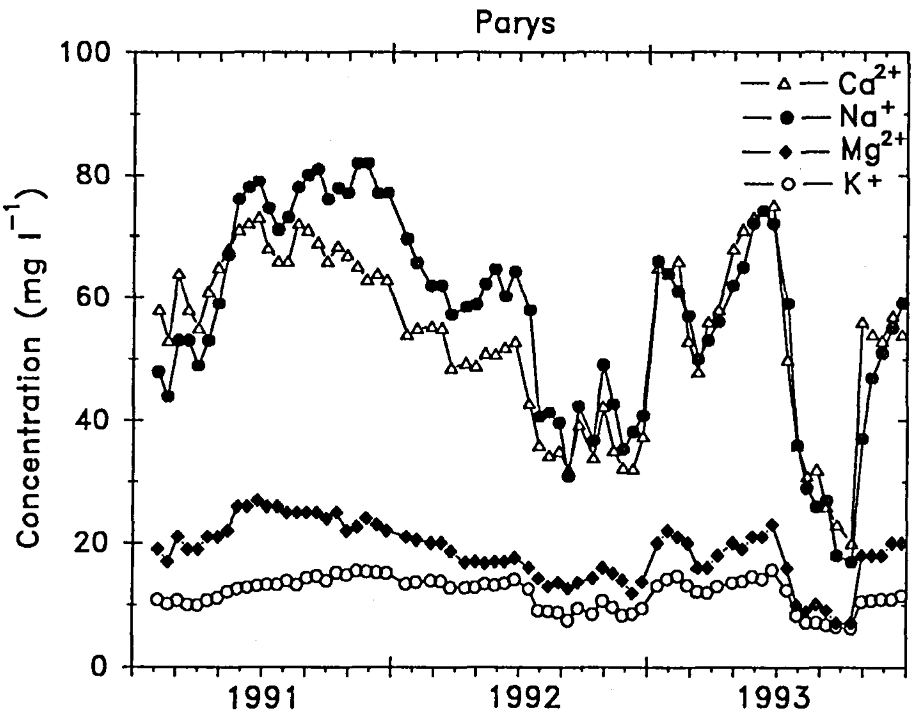

FIGURE 17: Variation in cation concentrations (mg l^{-1}) at Parys during the study period.

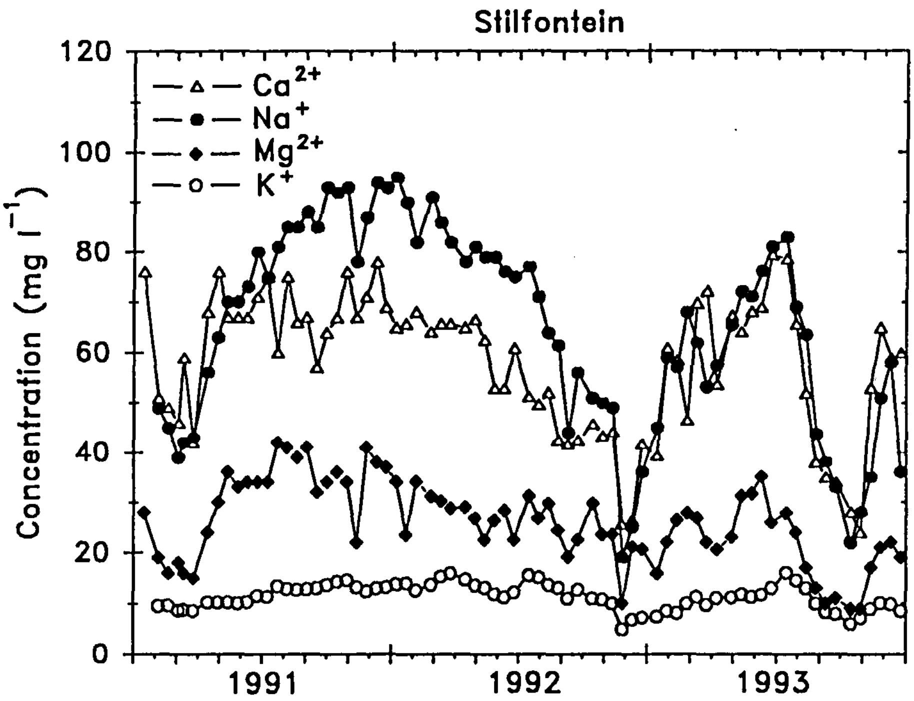

FIGURE 18: Variation in cation concentrations (mg l^{-1}) at Stilfontein during the study period.

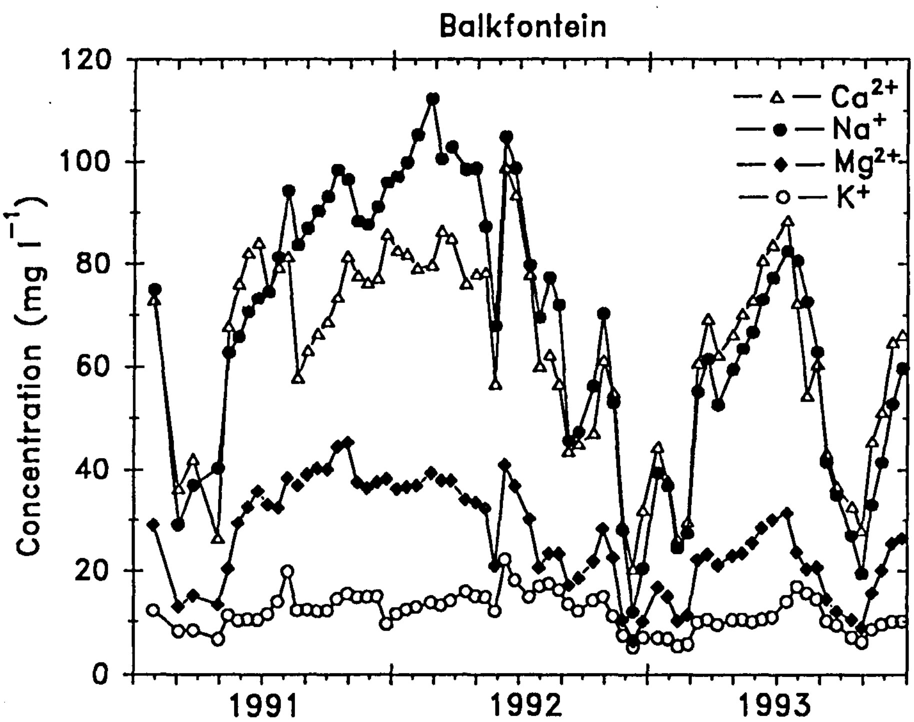

FIGURE 19: Variation in cation concentrations (mg l^{-1}) at Balkfontein during the study period.

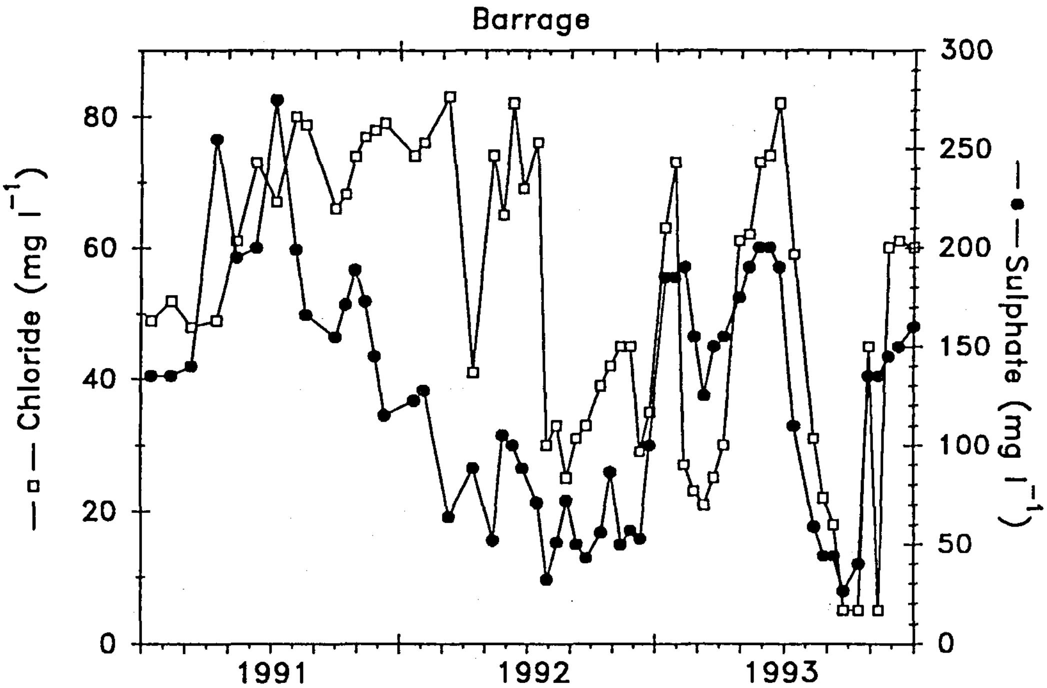

FIGURE 20: Variation in anion concentrations (mg l^{-1}) at the Barrage during the study period.

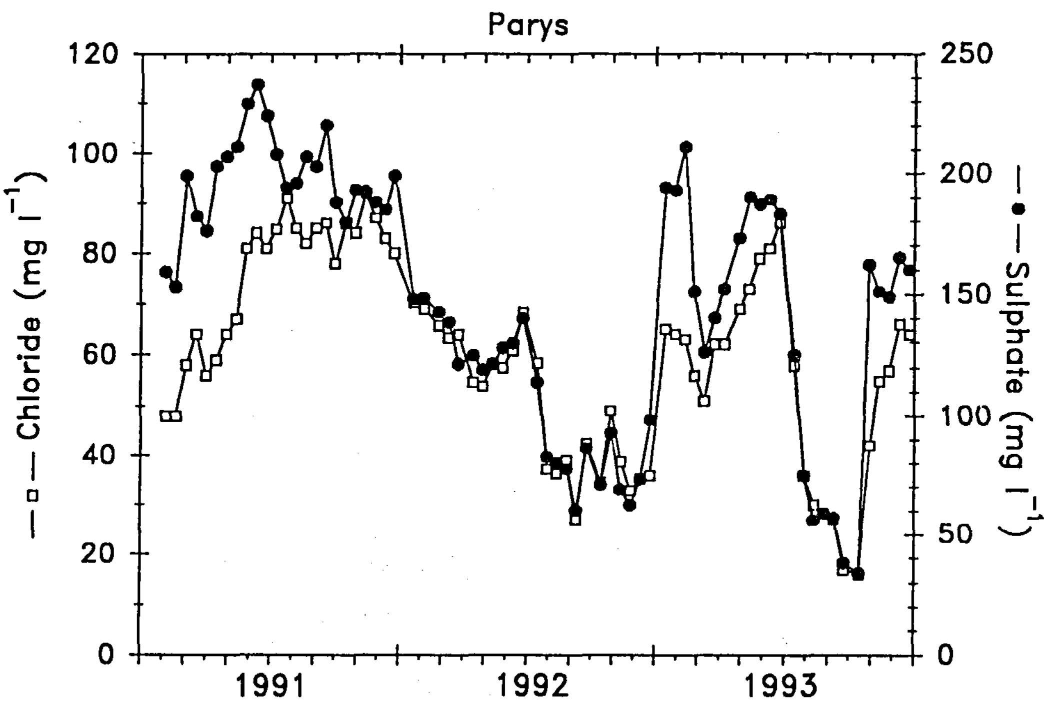

FIGURE 21: Variation in anion concentrations (mg l^{-1}) at Parys during the study period.

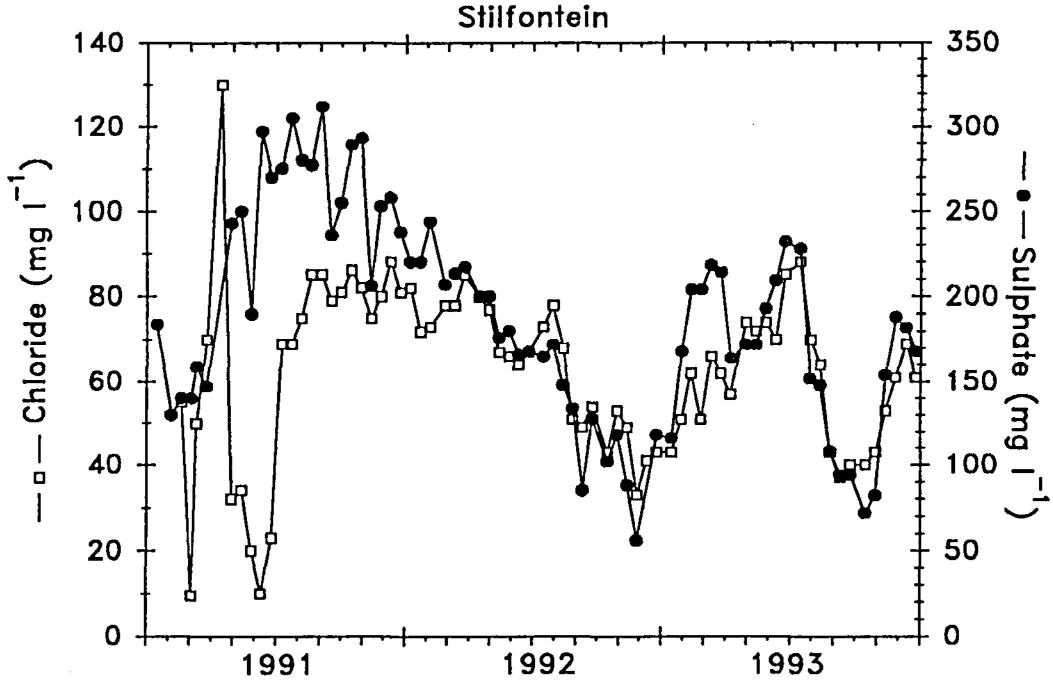

FIGURE 22: Variation in anion concentrations (mg l^{-1}) at Stilfontein during the study period.

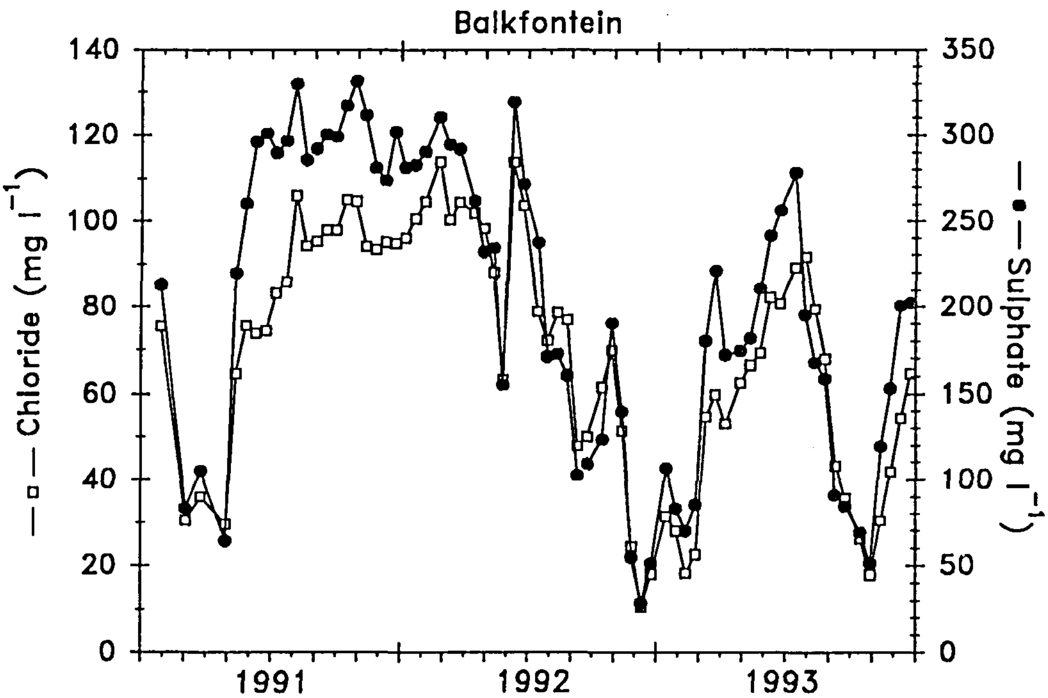

FIGURE 23: Variation in anion concentrations (mg l^{-1}) at Balkfontein during the study period.

The ionic composition of the Vaal River (at Balkfontein) about 40 years ago (1954-1956 - referred to as historical data) given by Stander *et al.* (1962), compared with the results presented here, emphasised the deterioration in mineral quality of Vaal River water. More information on this deterioration will be given later.

Sodium (Na^+) is ubiquitous in natural water and is often the major cation in many South African inland waters (Dallas & Day, 1993). Na^+ was also the second dominant cation at the Barrage sampling locality and was dominant at the other sampling localities during the study period. Na^+ requirements are particularly high in some species of blue-green algae. A threshold level of 4 mg l^{-1} sodium is required for near optimal growth of several species and maximal growth of several blue-green algae was found to occur at 40 mg l^{-1} Na^+ (Wetzel, 1983). The enrichment of water with high levels of sodium and phosphorus was indicated as a potential contributor to effective competition by blue-green algae under bloom conditions (Wetzel, 1983). Higher phosphorus (PO_4-P) concentrations recorded at the Barrage, compared to the other sampling localities (see section 3.1.2.3), together with the abundance of the Na^+ cation can possibly be a reason why intensive blue-green algal blooms are sometimes experienced in the Barrage (Loch Vaal) vicinity. During the present study, dominance of blue-green algae usually occurred at Na^+ concentrations above 40 mg l^{-1} (compare Figs 69, 71 & 72 with Figs 16, 18 & 19 respectively).

The average Na^+ concentration encountered for all three years of the present study was 48 mg l^{-1} at the Barrage (Fig. 16), 57 mg l^{-1} at Parys (Fig. 17), 65 mg l^{-1} at Stilfontein (Fig. 18) and 68 mg l^{-1} at Balkfontein (Fig. 19). The average Na^+ concentration in the Vaal River is much higher than the average for African rivers (11 mg l^{-1}; Wetzel, 1983). The average Na^+ concentration at the Balkfontein sampling locality was three times higher than the historical values (average of 22 mg l^{-1}) recorded from 1954 to 1956. Bruwer *et al.* (1985) concluded that the increased concentration of sodium in the lower Vaal River can be seen as a potential health problem.

Calcium (Ca^{2+}) is one of the major elements essential for living organisms, and is an important ion in the buffering system of lake and river waters (Goldman & Horne, 1983). Calcium ions are often the major cations in inland waters (Dallas & Day, 1993). Ca^{2+} was also the dominant cation found at the Barrage sampling locality in the Vaal River. In the Vaal River an average Ca^{2+} concentration of 50 mg l^{-1} was recorded at the Barrage (Fig. 16), 54 mg l^{-1} at Parys (Fig. 17), 58 mg l^{-1} at Stilfontein (Fig. 18) and 63 mg l^{-1} at Balkfontein (Fig. 19). Minimum, maximum and average Ca^{2+} concentrations for each separate year of the study period can be seen in Table 6. The downstream increase in average Ca^{2+} concentration is reflected in the TDS concentration and conductivity (see section 3.1.2.1). Lewin and Guillard (1963) noted that relatively high Ca^{2+} concentrations are stimulatory to phytoplankton growth.

Magnesium (Mg^{2+}) is an essential element, being found, amongst others, as part of the chlorophyll molecule. Because it is usually found in relatively high concentrations, it is unlikely to act as a limiting nutrient, but very little is known about its effects on aquatic organisms (Goldman & Horne, 1983). The average magnesium concentration from 1991 to 1993 in the Vaal River was high with an average of 18 mg l^{-1} recorded at the Barrage

(Fig. 16), 18.6 mg l^{-1} at Parys (Fig. 17), 26 mg l^{-1} at Stilfontein (Fig. 18) and 26.5 mg l^{-1} at Balkfontein (Fig. 19).

One of the more profound differences between terrestrial plant growth and that of phytoplankton is the minor role that is apparently played by potassium (K^+) in aquatic plants (Goldman & Horne, 1983). Lund (1965) stated that potassium is rarely considered to have an important influence on the ecology of algae. From 1991 to 1993 at the Vaal River Barrage an average K^+ concentration of 10 mg l^{-1} was recorded (Fig. 16), at Parys 11.9 mg l^{-1} (Fig. 17), at Stilfontein 11.4 mg l^{-1} (Fig. 18) and at Balkfontein 12.0 mg l^{-1} (Fig. 19). Compared to the historical data (1954 to 1956) with an average of 4 mg l^{-1}, the average K^+ concentration at the Balkfontein sampling locality increased three-fold during the last 40 years.

Sulphur in water occurs largely in the sulphate (SO_4^{2-}) ion form. Sulphate (SO_4^{2-}) rarely limits the growth or distribution of the aquatic biota (Goldman & Horne, 1983). In excess, however, and under specific chemical conditions, dissolved SO_4^{2-} results in the formation of sulphuric acid, which is a strong acid that reduces pH and can have devastating effects on aquatic ecosystems (Dallas & Day, 1993). The negative effect of sulphuric acid is particularly problematic in water seeping from mines, where sulphate levels can be extremely high. In the Vaal River, at the Barrage sampling locality, the mean sulphate concentration during the three years of the study period was 125 mg l^{-1} (Fig. 20), at Parys it was 148 mg l^{-1} (Fig. 21), at Stilfontein 187 mg l^{-1} (Fig. 22) and at Balkfontein 202 mg l^{-1} (Fig. 23). Historical records showed that the SO_4^{2-} concentration at the Balkfontein sampling locality increased more than 3 fold from 1954-1956 (60 mg l^{-1}) to 1993. The sulphate concentration in the Vaal River is amongst the highest concentrations reported for a river world wide (Roos, 1992). A world average for sulphate in rivers is 20 mg l^{-1} (Goldman & Horne, 1983). The high concentrations suggest sulphate pollution in the middle Vaal River. Only in the Moreau River (S Dakota, USA), higher SO_4^{2-} values, up to 1 460 mg l^{-1}, were reported (Golterman, 1975b). According to Braune & Rogers (1987) the high sulphate concentration in the water is due to high inputs from the tributaries draining the northern part of the catchment that is heavily contaminated by intensive mining and industrial activities. More recently the deposition of atmospheric pollutants (of which SO_4^{2-} is a major constituent) has been identified as a further potential cause of increased salinisation (Van Rooyen & Herold, 1992).

Chloride (Cl^-) is the major anion in sea water and in many inland waters, particularly in South Africa (Dallas & Day, 1993). However, in the Vaal River the Cl^- concentration was much lower than the concentration of sulphate (SO_4^{2-}). Chloride does not appear to limit algal production directly in nature, but may play (as sodium chloride) a major part in determining the types of algae that can grow in the water (Lund, 1965). In the Vaal River an average Cl^- concentration of 52 mg l^{-1} was recorded at the Barrage (Fig. 20), 60 mg l^{-1} at Parys (Fig. 21), 62 mg l^{-1} at Stilfontein (Fig. 22) and 70 mg l^{-1} at Balkfontein (Fig. 23). The average Cl^- concentration during the present study at Balkfontein is more than 4 times higher than the average of 16 mg l^{-1} recorded during the period 1954-1956.

3.1.2.3 DISSOLVED INORGANIC PHOSPHORUS (DIP), DISSOLVED INORGANIC NITROGEN (DIN), AND DIN:DIP RATIOS

The major nutrients that contribute to eutrophication are phosphorus and nitrogen (Wetzel, 1983; Davies & Day, 1986; Dallas & Day, 1993). Nitrogen and phosphorus are used by plants in the inorganic form. During decomposition, microbes convert organic nitrogen and phosphorous to the inorganic form.

Lotic systems are reported to be less susceptible to nutrient enrichment than are lentic systems (Porter, 1975), because there is little retention in the moving water of rivers. The initial nutrient loading of a stream influences the effect that subsequent addition of nutrients will have on the system. It is therefore possible to reduce the effects of nutrient enrichment in rivers by removing or reducing the source of nutrient enrichment, given that the stream or river is flowing reasonably strongly (Dallas & Day, 1993). Thus, nutrient enrichment in slow-flowing riverine ecosystems may result in excessive plant (algal and macrophyte) growth (Cole, 1973). Impoundments reduce the size and number of high-flow events, decrease turbidity and increase summer temperatures (Hynes, 1969). All of these changes encourage primary production and may result in prolific growth of algae in reservoirs, exemplified by the blooms of the buoyant blue-green alga *Microcystis aeruginosa* in Hartbeespoort Dam (NIWR, 1985).

Dissolved inorganic phosphorus (DIP)

Phosphorus has been implicated more widely than nitrogen as a limiting nutrient in freshwater systems (Hart *et al.*, 1992). In water, phosphorus usually occurs in the oxidized state, either as inorganic orthophosphate ions (PO_4^{3-}, HPO_4^{2-}, $H_2PO_4^-$) or in organic, largely biogenic compounds (Reynolds, 1984). Dissolved orthophosphate is evidently the major source of phosphorus directly available to phytoplankton (Wetzel, 1983). Much phosphorus may be unavailable, because it is adsorbed onto suspensoids or bonded to particles in the water (Addiscott *et al.*, 1991).

It was demonstrated in the Vaal River that phosphorus and nitrogen were usually present in higher concentrations after rains, i.e. during periods of relatively high discharge (Roos, 1992). In the Vaal River high flow conditions were frequently the result of water releases from the Vaal Dam. In this case the increase in phosphorus concentration is smaller, compared to the situation when high discharge is caused by high rainfall. Lewis and Grant (1979) also showed increased PO_4-P (DIP) concentrations with increased discharge in the watershed of Como Creek, Colorado. Ravichandran and Ramanibai (1988) reported high soluble reactive phosphate and nitrate nitrogen concentrations in Buckingham Canal (Madras, India), while the peaks coincided with the monsoon season during two years of study. In the Vaal River, at the Stilfontein and Balkfontein sampling localities, peaks of increased DIP concentrations were often accompanied by increases in the discharge (e.g March 1991; compare Figs 26 with 5, and 27 with 6 respectively), but at the Barrage (compare Fig. 24 and Fig. 3) and Parys (compare Figs 25 & 4) sampling localities, this phenomenon was not so obvious. The significance of the correspondence between discharge and DIP concentration at the Balkfontein and Stilfontein sampling localities will, however, be tested by statistical analysis.

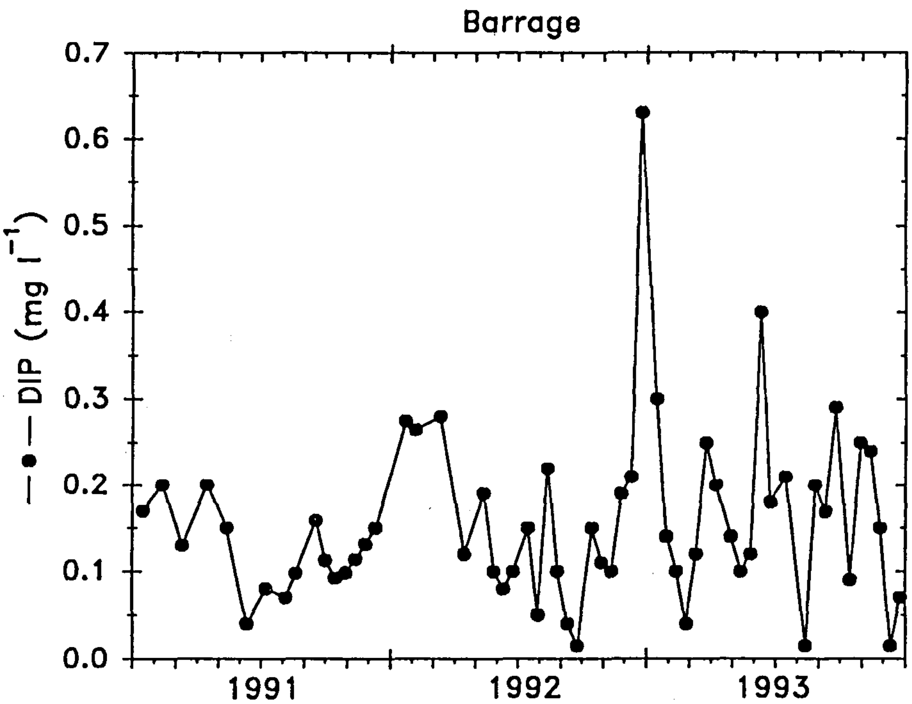

FIGURE 24: Variation in dissolved inorganic phosphorus (DIP) concentration (mg l^{-1}) at the Barrage during the study period.

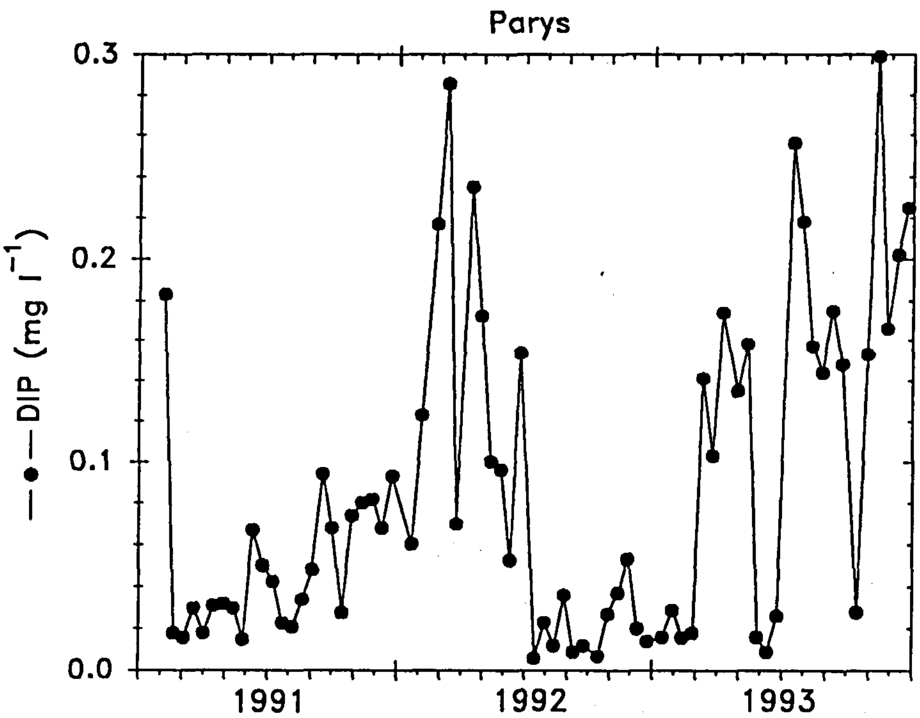

FIGURE 25: Variation in dissolved inorganic phosphorus (DIP) concentrations (mg l^{-1}) at Parys during the study period.

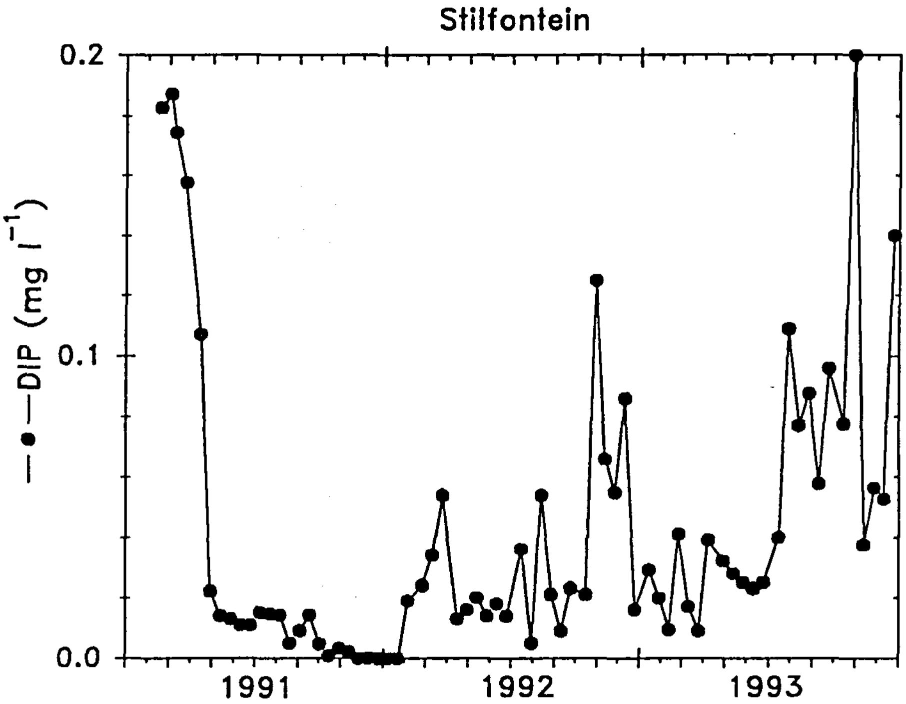

FIGURE 26: Variation in dissolved inorganic phosphorus (DIP) concentrations (mg l^{-1}) at Stilfontein during the study period.

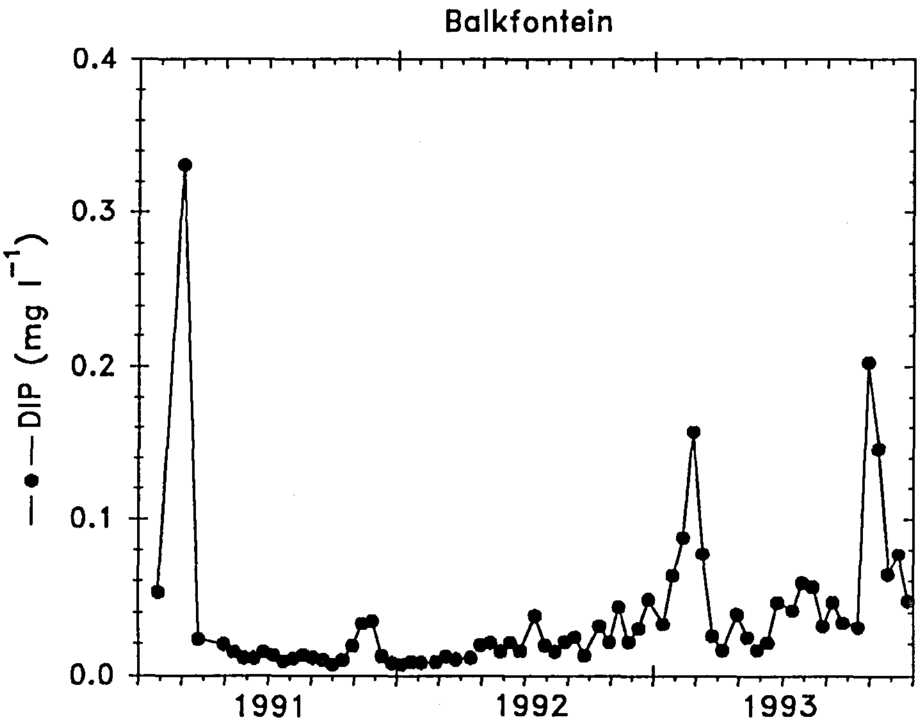

FIGURE 27: Variation in dissolved inorganic phosphorus (DIP) concentrations (mg l^{-1}) at Balkfontein during the study period.

Higher concentrations of phosphorus containing substances are likely to occur in waters that receive sewage, man-made detergents and leaching or runoff from cultivated land (Wheeler & Neushul, 1981).

TABLE 7: Minimum, maximum and average inorganic nitrogen (DIN) and phosphorus (DIP) concentrations in mg l^{-1} as well as DIN:DIP ratios recorded in the water of the Barrage, Parys, Stilfontein and Balkfontein sampling locations for the three years of the study period.

	1991			1992			1993		
	DIP	**DIN**	**DIN:DIP**	**DIP**	**DIN**	**DIN:DIP**	**DIP**	**DIN**	**DIN:DIP**
BARRAGE:									
Minimum	0.04	0.77	5	0.02	0.21	0.8	0.02	0.51	2.00
Maximum	0.20	3.18	57	0.63	2.33	27	0.40	3.07	172
Average	0.12	2.02	20	0.17	0.97	9	0.16	1.53	19
PARYS:									
Minimum	0.02	1.04	9	0.01	0.08	4	0.01	0.09	1
Maximum	0.18	3.42	192	0.29	2.04	341	0.30	3.39	218
Average	0.05	2.30	69	0.08	1.15	48	0.13	1.55	29
STILFONTEIN:									
Minimum	0.00	0.17	12	0.00	0.12	4	0.01	0.06	4
Maximum	0.19	2.55	818	0.13	1.76	352	0.20	2.64	37
Average	0.04	1.00	97	0.03	0.49	32	0.06	0.73	15
BALKFONTEIN:									
Minimum	0.01	0.01	1	0.01	0.02	1	0.02	0.05	1
Maximum	0.33	1.27	82	0.05	0.96	40	0.20	2.10	39
Average	0.03	0.20	12	0.02	0.24	11	0.06	0.46	9

From 1991 to 1993 the average phosphate phosphorus (PO_4-P; DIP) concentration in the Vaal River at the Barrage sampling locality (0.155 mg l^{-1}; Fig. 24) was higher than the average DIP concentrations measured at Parys (0.086 mg l^{-1}; Fig. 25), Stilfontein (0.043 mg l^{-1}; Fig. 26) and Balkfontein (0.04 mg l^{-1}; Fig. 27). The higher DIP concentration measured at the Barrage (upstream sampling locality), was in accordance with the findings of Bruwer *et al.* (1985) who showed that orthophosphate concentrations in the upper reaches of the Vaal River was the highest, ranging between 0.15 and 0.28 mg l^{-1}. Upstream from the Barrage, in the Vaal Dam, a high average DIP concentration (0.15 mg l^{-1}) was also recorded*. The average DIP concentration in the Vaal River at the Barrage sampling locality was comparable to the world average in rivers of 0.1 mg l^{-1} (Goldman & Horne, 1983). It was, however, lower than the average DIP concentration of enriched impoundments like Hartbeespoort Dam (approximate mean 0.35 mg l^{-1}; Robarts, 1984), but quite comparable with the DIP concentration measured in other enriched impoundments, e.g. Roodeplaat Dam (approximate mean 0.15 mg l^{-1}; De Wet, 1986). In contrast to the DIP concentrations measured at the Barrage sampling locality, the

*A. Vermeulen, Department of Plant and Soil Sciences, PU for CHE, Potchefstroom: personal communication

concentrations at the Parys, Stilfontein and Balkfontein sampling localities showed similarities to the average phosphate concentration, calculated from information on 21 South African impoundments given by Walmsley and Butty (1980), namely 0.06 mg l^{-1}. Because Roos (1992) showed that the DIP concentration in a eutrophic system ranged between 0.008 and 0.25 mg l^{-1}, the Vaal River can be regarded as eutrophic (see also Wetzel, 1983). Table 7 and Fig. 28 show that a decrease in average DIP concentration occurred downstream from the Barrage to Balkfontein during all three years of the study period. This indicates higher levels of pollution in the Barrage area, which could possibly be the result of urbanisation and industrialisation in the PWV complex. Further downstream, DIP can be removed from the water by algae as well as macrophytes, such as the water hyacinths, which form thick growths downstream of the Barrage, especially at the Parys and Stilfontein sampling localities.

Steynberg (1986) suggested a phosphorus management strategy for the Vaal River catchment area which involves implementation and maintenance of the 1 mg l^{-1} phosphate standard for sewage effluents in the Zuikerboschrand and Klip River catchments. Steynberg concluded that the application of the phosphorus standard will result in maintaining the annual average raw water chlorophyll concentration below 20 µg l^{-1} and the phosphate concentration below 0.44 mg l^{-1} requirement up to the year 2000.

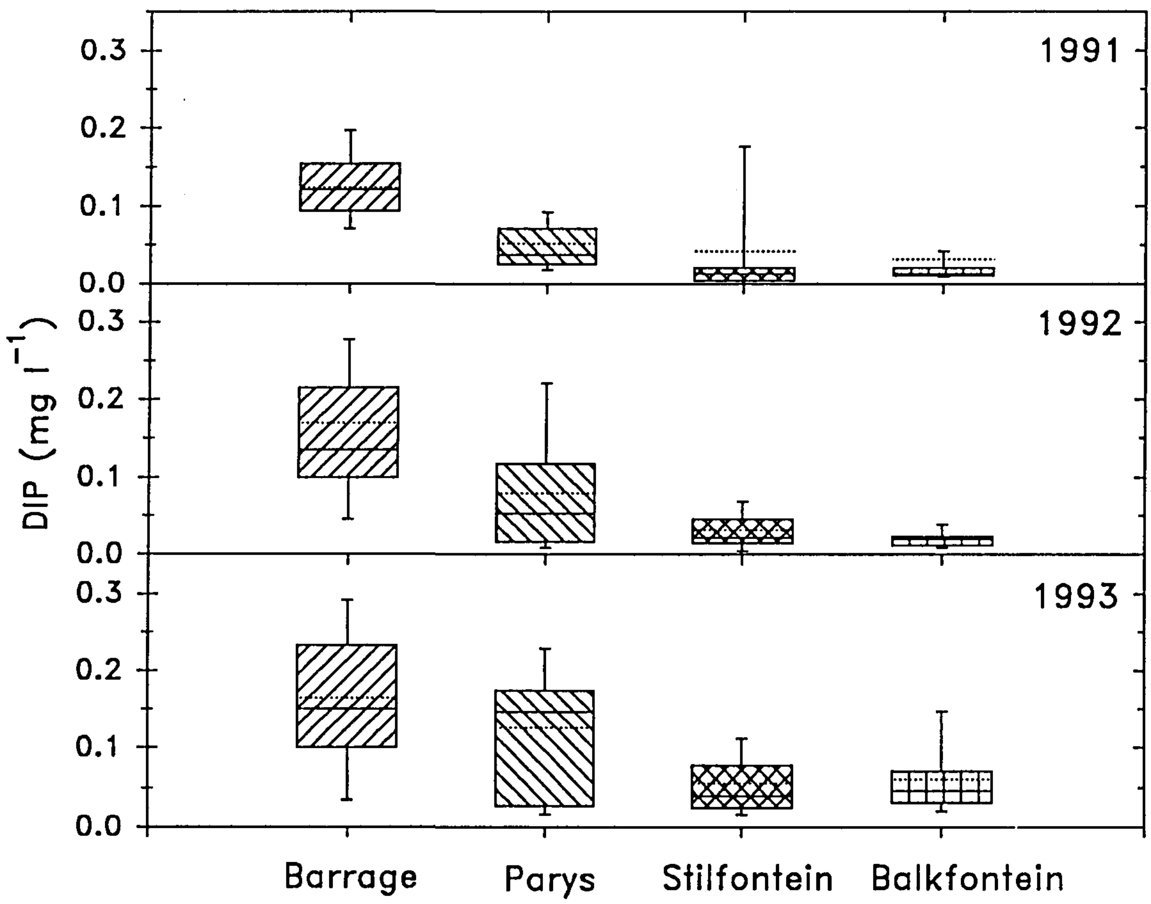

FIGURE 28: Box plot of annual dissolved inorganic phosphorus (DIP) concentration (mg l^{-1}) in the Vaal River at the Barrage, Parys, Stilfontein and Balkfontein.

Dissolved inorganic nitrogen (DIN)

Several forms of nitrogen are available to phytoplankton. These forms include nitrate (NO_3^-), nitrite (NO_2^-) and ammonium (NH_4^+) ions as well as certain dissolved organic nitrogenous compounds, such as urea ($CO[NH_2]_2$), free amino acids and peptides (Reynolds, 1984).

Nitrate (NO_3^{2-}) is normally the most common form of combined inorganic nitrogen in lakes and streams (Roos, 1992). Nitrate may enter the water via fertilizers, agricultural runoff, etc. The average nitrate nitrogen (NO_3-N) concentration recorded in the Vaal River from 1991 to 1993 ranged from 1.252 mg l^{-1} (Barrage; Fig 29) to 1.566 mg l^{-1} (Parys; Fig. 30) to 0.628 mg l^{-1} (Stilfontein; Fig. 31) to 0.22 mg l^{-1} (Balkfontein; Fig. 32). High NO_3-N concentrations were recorded at both the upstream sampling localities (Barrage and Parys; Figs 29 & 30 respectively) and low concentrations at the two downstream sampling localities (Stilfontein and Balkfontein; Figs 31 & 32 respectively), comparable to higher DIP concentrations at the upstream sampling localities. While the average NO_3-N concentrations at the Barrage and Parys sampling localities from 1991 to 1993 were higher than the world average for rivers (1 mg l^{-1}; Goldman & Horne, 1983), the NO_3-N concentrations at the Stilfontein and Balkfontein sampling localities were lower. During the present study increases in NO_3-N concentration were frequently accompanied by increases in discharge at the Barrage, Stilfontein and Balkfontein sampling localities (compare Figs 29 & 3, 31 & 5 and 32 & 6). The significance of this correspondence must, however, be tested by statistical analysis. Roos (1992) also stated that the NO_3-N concentration decreased during flood conditions (i.e. discharge of more than 250 $m^3\,s^{-1}$), probably because of a wash-out action. During the present study (1991-1993) discharge values of more than 150 $m^3\,s^{-1}$ were not recorded.

Nitrite (NO_2^-) is a naturally occurring anion in fresh and saline waters. Nitrite nitrogen (NO_2-N) in the Vaal River was only analysed for at the Barrage, where it was present in almost undetectable low concentrations (average of 0.07 mg l^{-1}; Fig. 29). Harris (1986) stated that nitrite is generally present only in trace quantities in natural waters, probably because of the rapidity of nitrification (nitrite is the intermediate in the conversion of ammonia to nitrate).

Ammonia gas (NH_3) is readily soluble in water (solubility ≈ 100 000 mg l^{-1} at 20°C; Hart *et al.*, 1992) and reacts with the water to form ammonium hydroxide (NH_4OH). NH_4OH dissociates into ammonium (NH_4^+) and hydroxyl (OH^-) ions, which tends to raise the pH of the water. Ammonia is present in water primarily as NH_4^+ ions and as undissociated NH_4OH (Wetzel, 1983). The proportions of NH_4^+ to NH_4OH are dependent on the dissociation dynamics which are governed by pH and temperature, i.e. lower NH_4:NH_4OH ratios occur together with higher pH conditions. The ratio of NH_4^+ to NH_4OH in the Vaal River was calculated to be approximately 30:1 at an average pH of 8.1 (Roos, 1992).

During the study period an average NH_4-N concentration of 0.15 mg l^{-1} was recorded at the Barrage (Fig. 29), 0.111 mg l^{-1} at Parys (Fig. 30), 0.110 mg l^{-1} at Stilfontein (Fig. 31) and 0.08 mg l^{-1} at Balkfontein (Fig. 32). These averages for the Vaal River are higher, especially at the Barrage, Parys and Stilfontein sampling localities, than the average NH_4-N concentration for rivers of the world (0.05 mg l^{-1}; Goldman & Horne, 1983). It is, however, lower than the average NH_4-N concentration calculated from the data of 21 South African impoundments given by Walmsley & Butty (1980), namely 0.163 mg l^{-1}.

The average dissolved inorganic nitrogen (DIN = NH_4-N + NO_3-N + NO_2-N) concentration recorded at the Barrage was 1.472 mg l^{-1} (Fig. 29), at Parys 1.677 mg l^{-1} (Fig. 30), at Stilfontein 0.738 mg l^{-1} (Fig. 31) and at Balkfontein 0.3 mg l^{-1} (Fig. 32). Because NO_3-N was the dominant component of DIN (Figs 29-32), the same tendency regarding averages of NO_3-N and DIN can be seen (Table 7), namely where the highest concentrations occurred at the two upstream sampling localities, decreasing downstream to Stilfontein and Balkfontein. Higher annual averages of DIN concentration can also be seen at the upstream sampling localities (Table 7 and Fig. 33) during all three years of the study period. High DIN concentrations were recorded at the Barrage, after which an increase in average DIN concentration occurred to Parys and then the concentration decreased from Parys downstream to Stilfontein and Balkfontein.

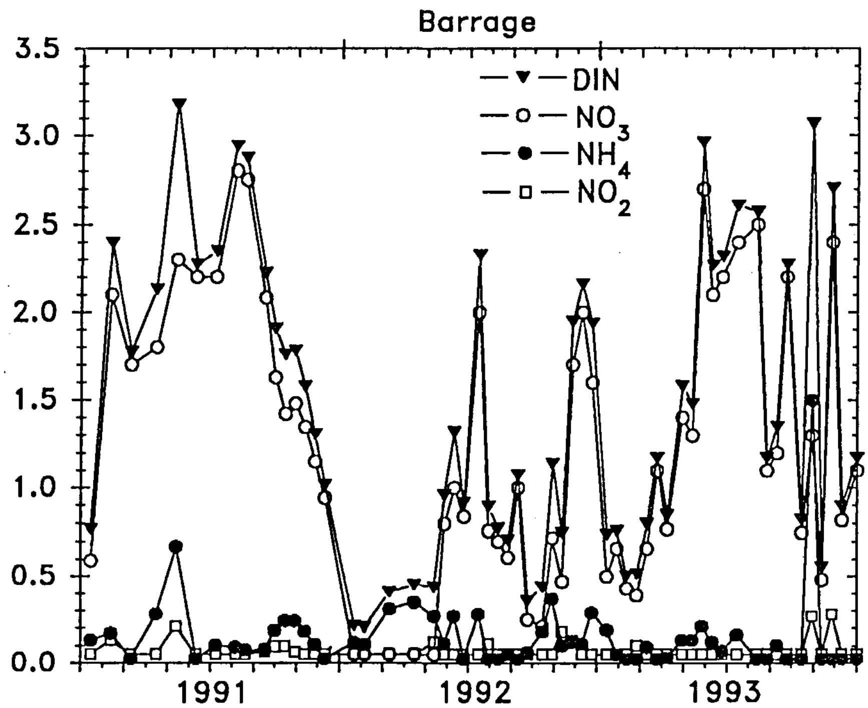

FIGURE 29: Variation in nitrogen concentration (mg l^{-1}) at the Barrage during the study period.

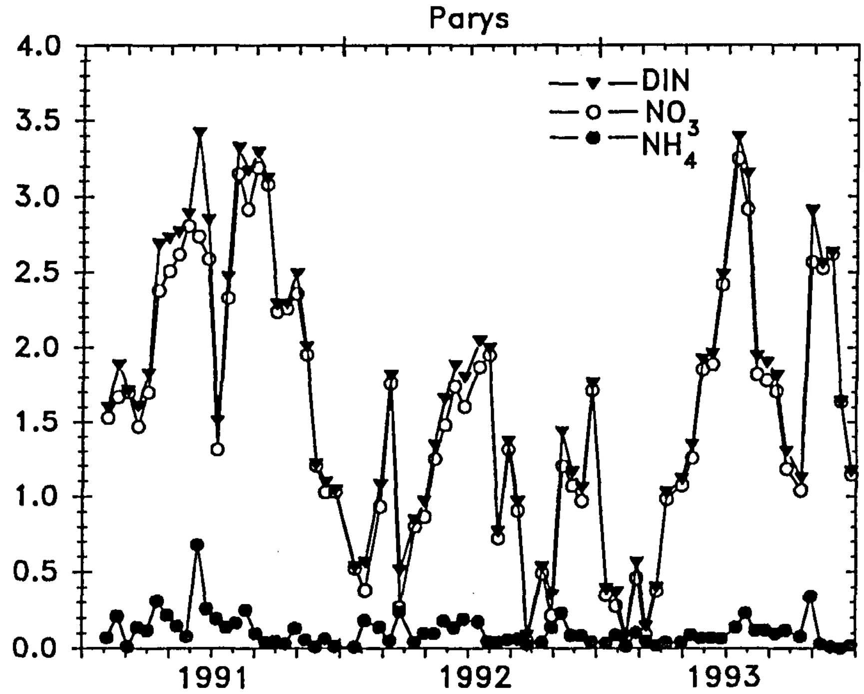

FIGURE 30: Variation in nitrogen concentration (mg l^{-1}) at Parys during the study period.

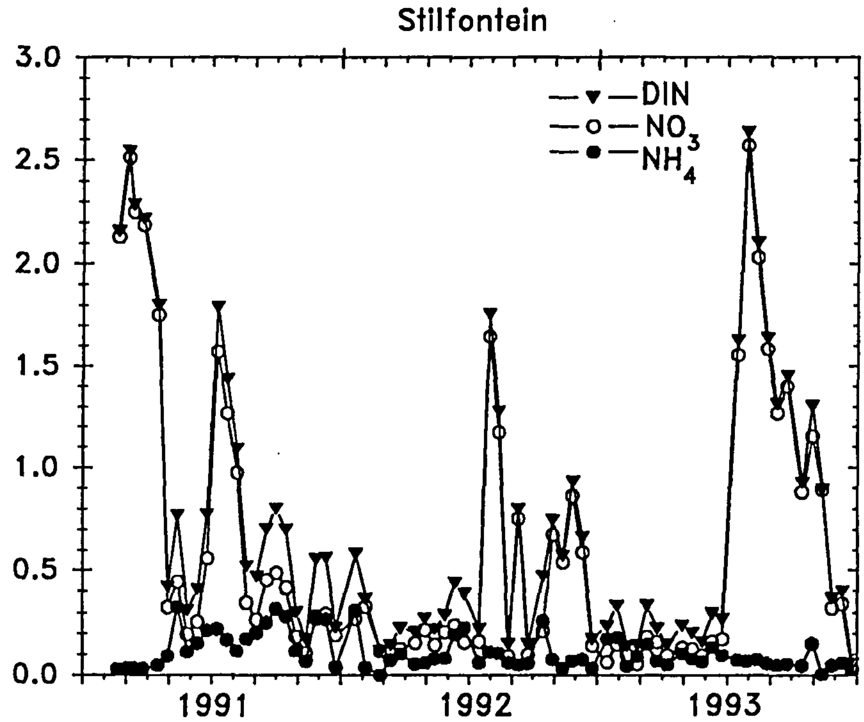

FIGURE 31: Variation in nitrogen concentration (mg l^{-1}) at Stilfontein during the study period.

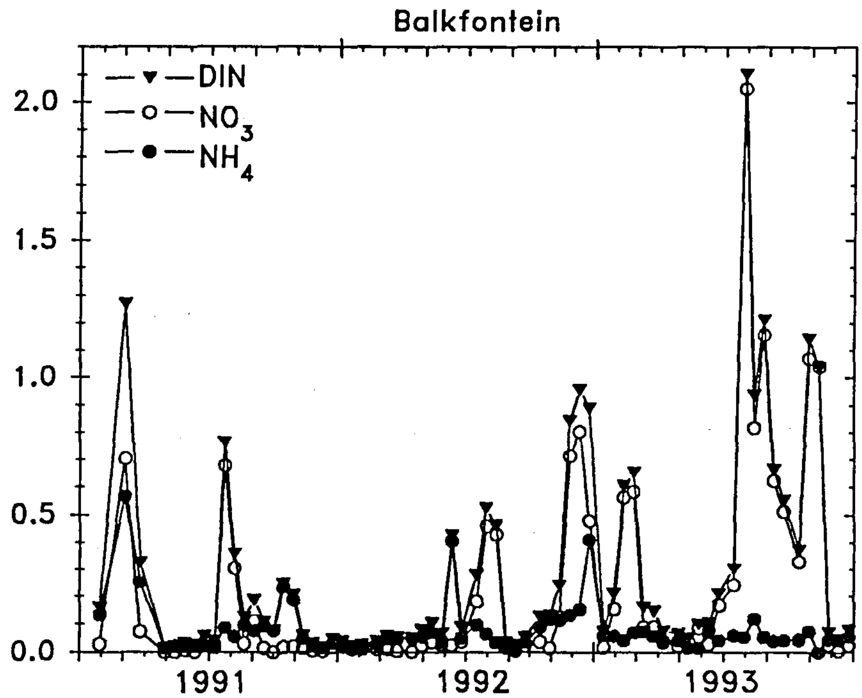

FIGURE 32: Variation in nitrogen concentration (mg l^{-1}) at Balkfontein during the study period.

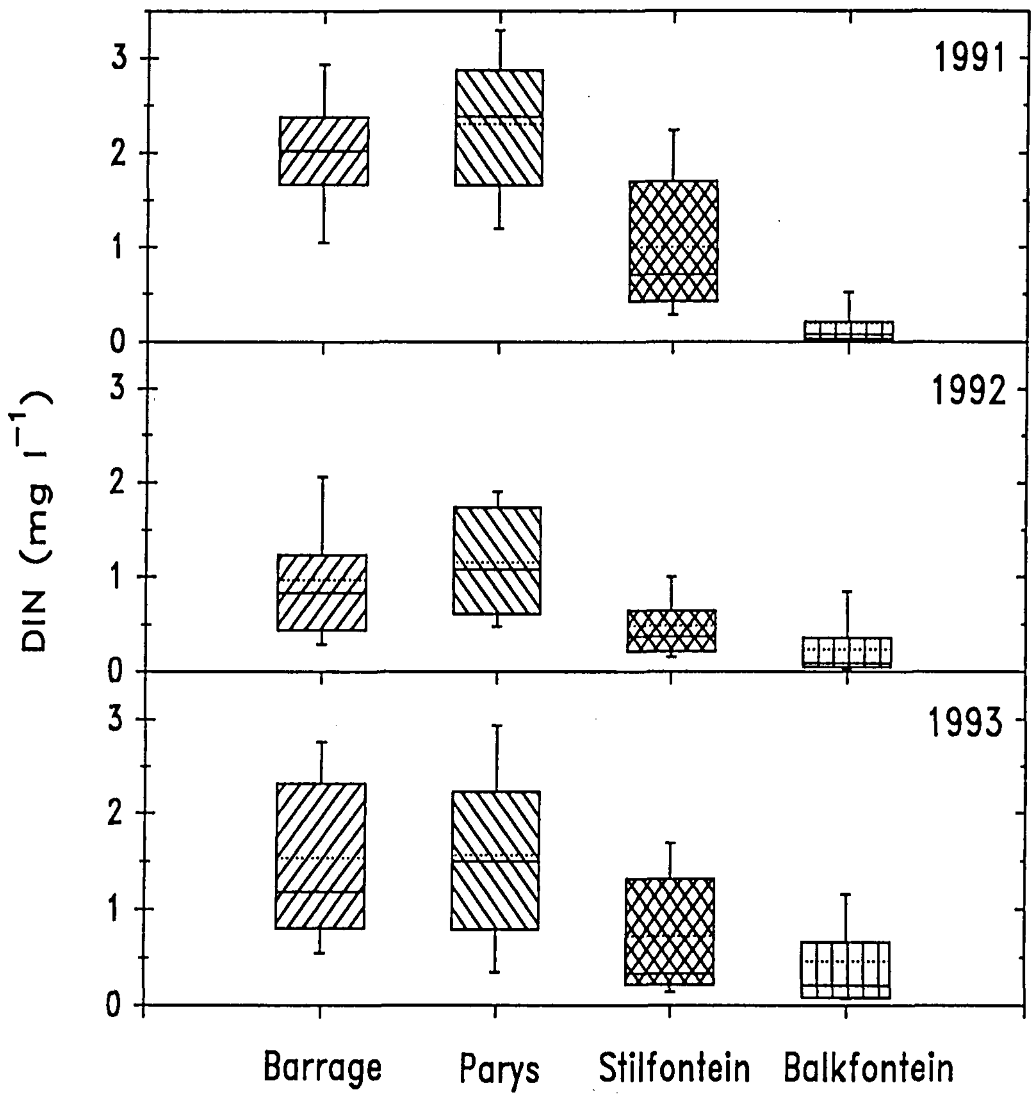

FIGURE 33: Box plot of annual dissolved inorganic nitrogen (DIN) concentration (mg l^{-1}) in the Vaal River at the Barrage, Parys, Stilfontein and Balkfontein.

DIN:DIP ratios

Organic matter from algae and aquatic macrophytes contains nitrogen and phosphorus in approximately the ratio 7N:1P (mass), also called the Redfield ratio (Parsons *et al.*, 1984; Wetzel, 1983). This implies that at N:P (DIN:DIP) ratios of less than 7 in the environment, nitrogen could possibly be limiting plant growth and at DIN:DIP ratios greater than 7, phosphorus could be limiting algal or macrophytic growth. However, different algae prefer different N:P ratios, because algal assemblages consist of many species, each with different optimal N:P requirements (Grimm & Fisher, 1986). Several investigators have used ratios of nitrogen to phosphorus (N:P) to indicate which of these essential nutrients potentially limits production (Grimm & Fisher, 1986).

During the study period, the dissolved inorganic nitrogen (DIN) to phosphate (PO_4-P or DIP) ratio (DIN:DIP) was high with an average of 15.9 at the Barrage (Fig. 34), 48.6 at Parys (Fig. 35), 45.8 at Stilfontein (Fig. 36) and 10.3 at Balkfontein (Fig. 37).

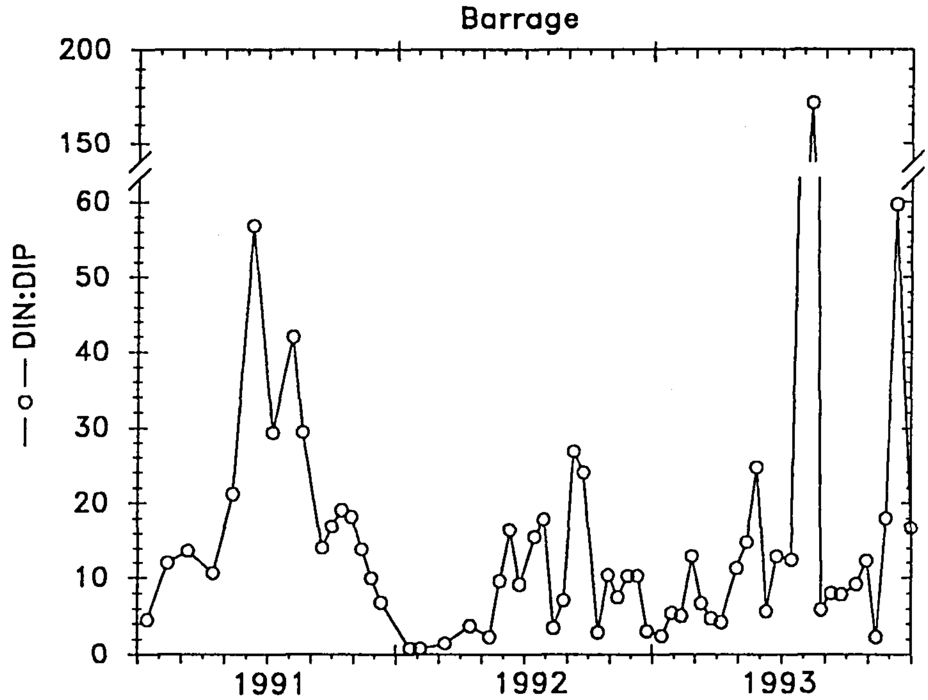

FIGURE 34: Variation in DIN:DIP ratios at the Barrage during the study period.

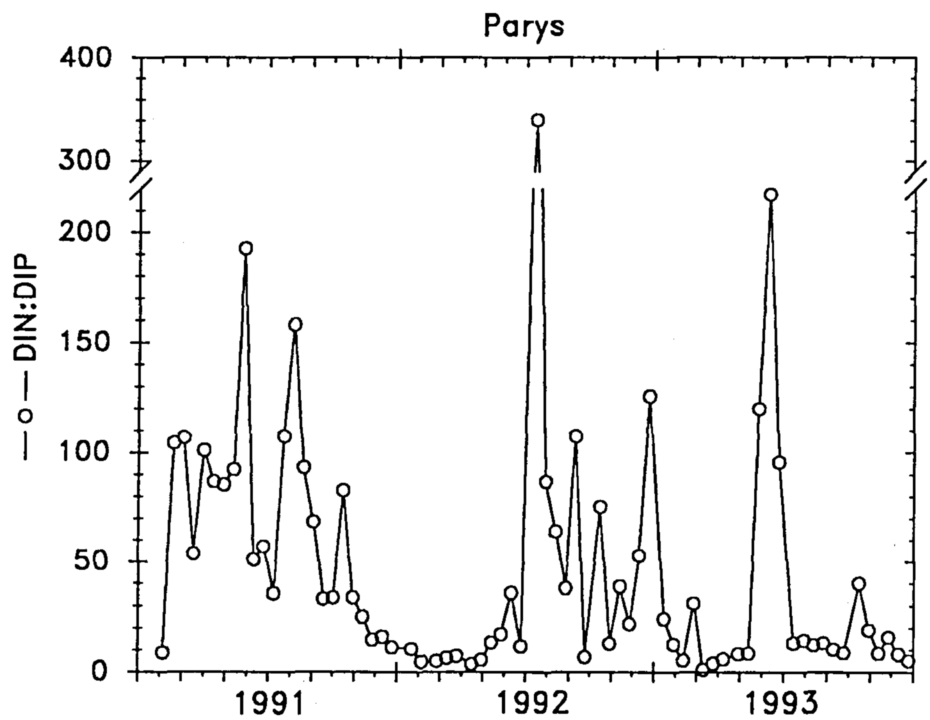

FIGURE 35: Variation in DIN:DIP ratios at Parys during the study period.

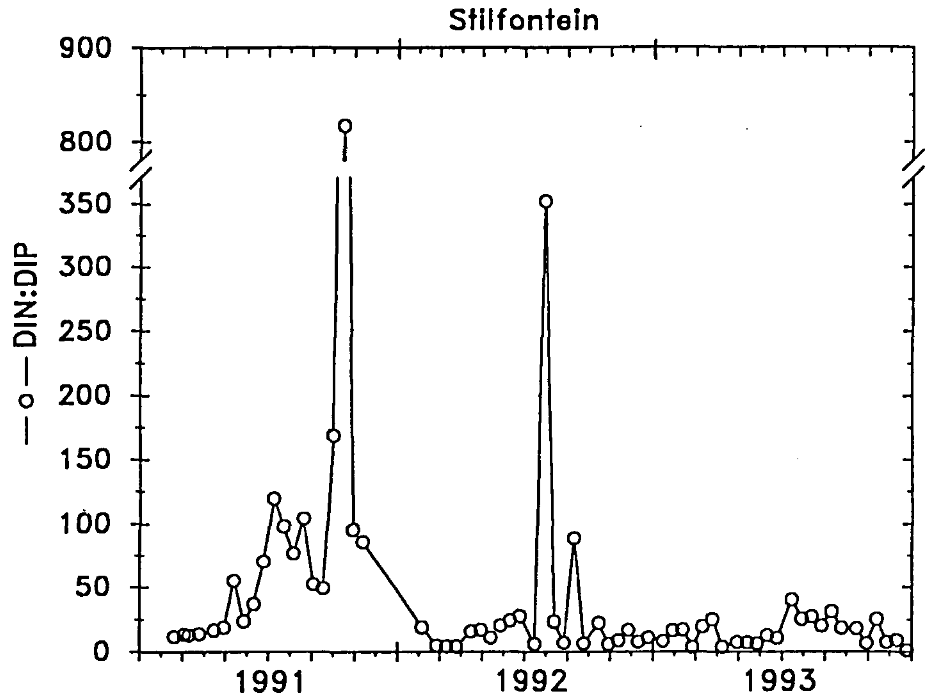

FIGURE 36: Variation in DIN:DIP ratios at Stilfontein during the study period.

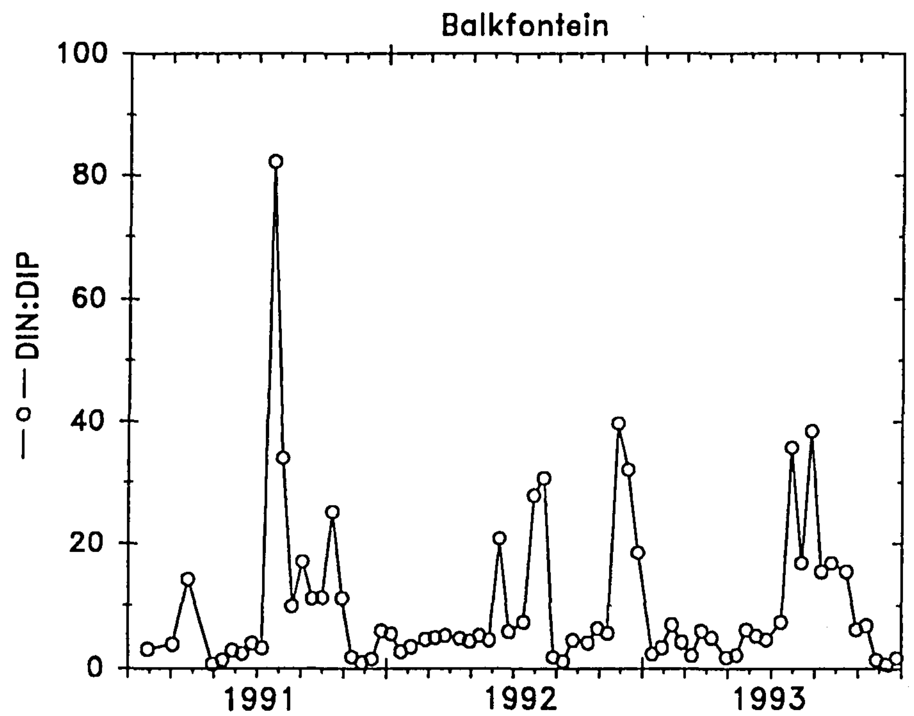

FIGURE 37: Variation in DIN:DIP ratios at Balkfontein during the study period.

The high average DIN:DIP ratios recorded at Parys and Stilfontein can be explained as follows: Fig. 35 shows that two periods of very high DIN:DIP ratios occurred at the Parys sampling locality, namely during July 1992 (DIN:DIP = 341) and June 1993 (DIN:DIP = 217). During these periods the DIN concentration was relative high (Fig. 30), but low DIP concentrations were recorded (Fig. 25). The low DIP concentrations caused high calculated DIN:DIP ratios. At the Stilfontein sampling locality, two peaks of high DIN:DIP ratios were also observed, namely October 1991 (DIN:DIP = 817) and July 1992 (DIN:DIP = 352; Fig. 36). Although the DIN concentration did not reach very high levels during October (Fig. 31), the extremely low DIP concentration (Fig. 26) was the reason for the high DIN:DIP ratio. During July 1992 (Fig. 36) high DIN:DIP ratios can be ascribed to high DIN concentrations (Fig. 31) as well as low DIP concentrations (Fig. 26). DIN:DIP ratios higher than 200 were never recorded at the Barrage and Balkfontein sampling localities. At the Balkfontein sampling locality, where the DIP concentration was low (Fig. 27), DIN:DIP ratios followed a more or less similar pattern than the DIN concentration (compare Figs 37 and 32). At times when a low DIN:DIP ratio accompanied by high DIN concentrations occurred (compare Figs 37 & 32), the DIP concentration was high (e.g. end of February 1991 as well as February and October 1993; Fig. 27). The average N:P ratios from 1991 to 1993 at all four sampling localities suggest that phosphorus is the limiting nutrient in the river.

Minimum, maximum and average DIP and DIN concentrations as well as DIN:DIP ratios for each year of the study period at the different sampling localities are presented in Table 7.

The relationship between nutrient -concentrations (DIP and DIN) and -ratios (DIN:DIP) and algal biomass will be discussed in sections 3.2.4.4 and 3.2.4.5 respectively.

3.1.2.4 TOTAL PHOSPHORUS (TP), TOTAL NITROGEN (TN) AND TN:TP RATIOS

Total phosphorus (TP)

Roos (1992) demonstrated a statistically significant correlation between TP concentration and discharge in the Vaal River at Balkfontein (1985-1989). The best correlation was found after heavy rains. Usually an increase in TP concentration, as a result of an increase in discharge because of rainfall, is more obvious than that of discharge caused by the releasing of water from the Vaal Dam. Increases in discharge during the present study were frequently the result of releases of water from the Vaal Dam rather than rainfall. This might explain why increases in discharge during the present study were not always accompanied by marked increases in TP concentration. However, during the present study certain periods characterised by high TP concentrations were also characterised by high discharge, for example during the end of November 1992 at Stilfontein (compare Fig. 40 with Fig. 5) and during the end of February/beginning of March 1991 and the end of November 1992 at the Balkfontein sampling locality (compare Figs 41 with 6). A statistical analysis must, however, be done to determine the significance of the correspondence between discharge and TP concentration, because the correlation between discharge and TP seems less obvious at the Barrage and Parys sampling localities.

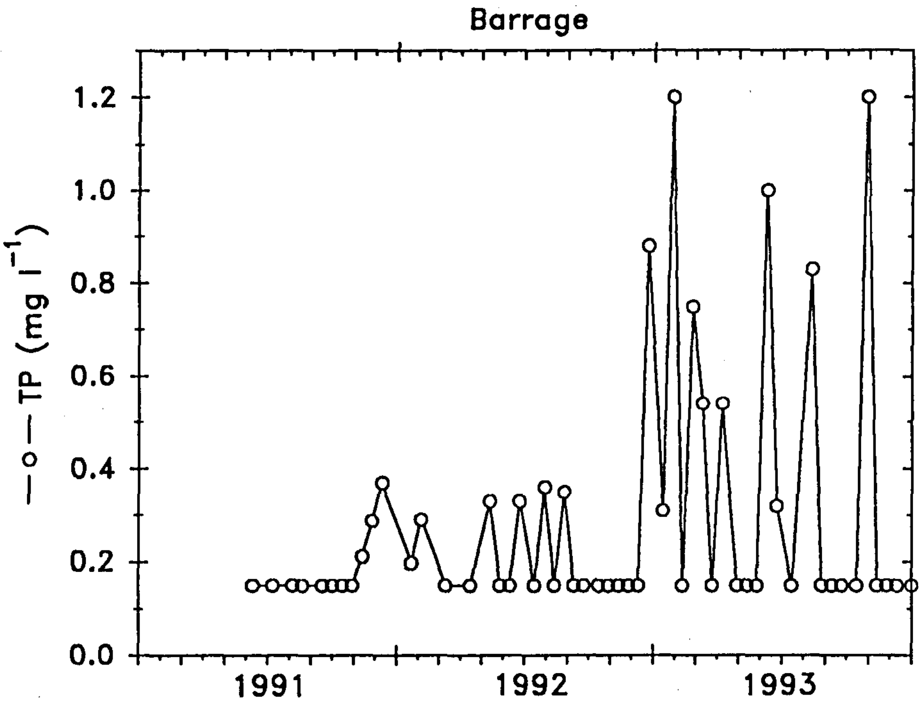

FIGURE 38: Variation in total phosphorus (TP) concentration (mg l^{-1}) at the Barrage during the study period.

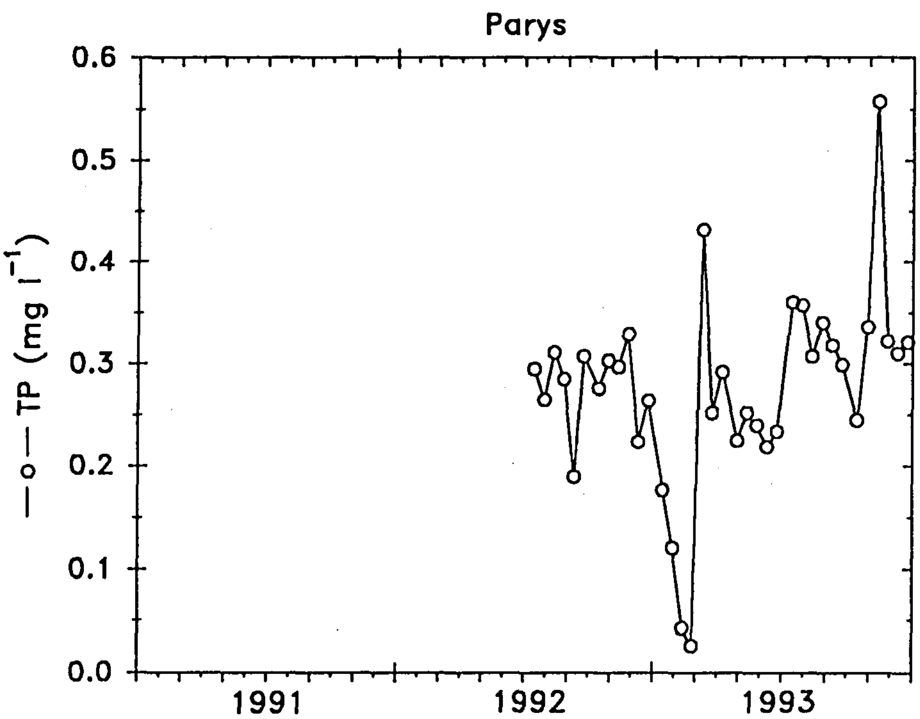

FIGURE 39: Variation in total phosphorus (TP) concentration (mg l^{-1}) at Parys during the study period.

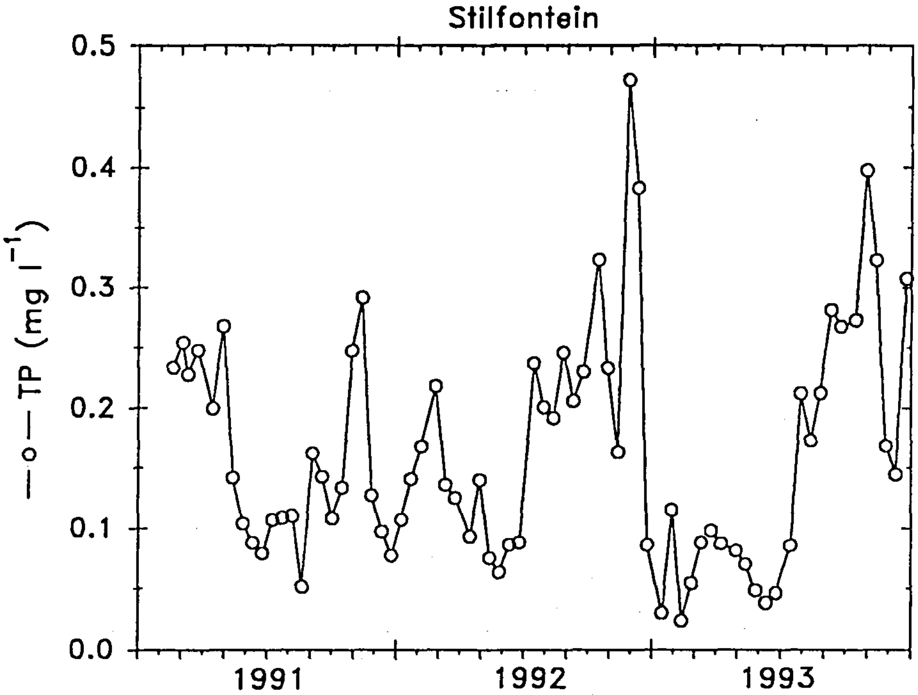

FIGURE 40: Variation in total phosphorus (TP) concentration (mg l^{-1}) at Stilfontein during the study period.

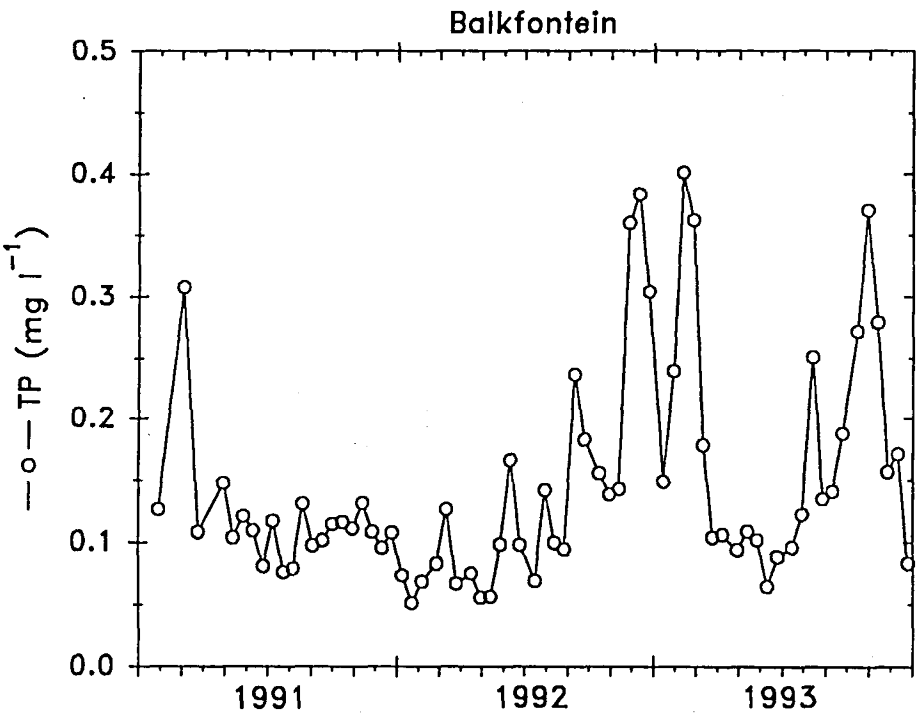

FIGURE 41: Variation in total phosphorus (TP) concentration (mg l^{-1}) at Balkfontein during the study period.

During the present study on the Vaal River, peaks in TP concentrations were frequently accompanied by peaks in turbidity, indicating that processes like erosion possibly represents the main input of phosphorus to the Vaal River. During the end of November 1992 and the end of October 1993 at Stilfontein (compare Figs 40 and 8), as well as the end of February 1991, December 1992 and February 1993 at Balkfontein (compare Figs 41 and 9), peaks of TP concentration coincided with peaks of turbidity. This relation was, however, not as clear at the Barrage sampling locality, and therefore a statistical analysis will be done to test the significance of the correspondence between TP concentration and turbidity. Studies done by Roos (1992) showed a highly significant correlation between total suspended solids and TP concentration. This observation is supported by Golterman (1973) who showed that the amount of phosphorus carried by river water could be considerable, especially if the river is heavily loaded with silt. For example, the Kaisi River (tributary of lake Edward in West Africa) carried 200 mg l^{-1} of suspended matter, mainly as clay material, which contained 0.8 mg l^{-1} of particulate phosphate. Hoyer and Jones's (1983) data for 82 Midwest reservoirs in the USA (Missouri and Iowa), also showed a direct relationship between inorganic suspended solids and TP concentration.

Golterman (1973) stated that not all the phosphorus adsorbed to clay particles is available to algae. However, phosphate adsorbed onto sediments can make up a large proportion of the phosphate available for algal growth in an impoundment (Grobler & Davies, 1981). High chlorophyll-*a* concentrations recorded in the Vaal River (see section 3.2.2), suggest that much of the TP that enters the river eventually becomes available as inorganic phosphate to the algae, thereby sustaining high PO_4-P concentrations. This implies a rapid turnover of phosphate and/or a continuous supply of DIP from the TP pool. Turnover times of inorganic phosphate in temperate oceans can be as little as 1.5 days (Golterman, 1975a).

During the present study on the Vaal River, high TP concentrations usually coincided with high DIP concentrations (compare Figs 38 & 24, 39 & 25, 40 & 26 and 41 & 27). This was in accordance to the findings of Harris (1986) who showed that, as the TP concentration of surface waters increases, the proportion of DIP increases as well. The proportion of DIP within TP is usually less than 10% in oligotrophic lakes, whereas at high TP concentrations (eutrophic conditions), the dissolved inorganic phosphorus pool becomes almost 100% of the total (Harris, 1986). Data from Roodeplaat Dam (De Wet, 1986) and Hartbeespoort Dam (Robarts, 1984), both eutrophic systems, showed that the fraction of DIP was approximately 90% and 66% respectively. During the present study the percentage DIP in TP at the Barrage sampling locality was on average 55%, 25% at Parys, 27% at Stilfontein and 24% at Balkfontein. The higher percentage DIP in TP at the Barrage could possibly be an indication of more eutrophic conditions present at the Barrage than at the other sampling localities.

Large mats of water hyacinths frequently occur in the Vaal River downstream from the Barrage. The lower % DIP fraction at the Parys, Stilfontein and Balkfontein sampling localities suggests a more rapid uptake of phosphate by algal cells and possibly water hyacinths at these sampling localities. Higher DIP concentrations were also present at the Barrage, than at the other sampling localities (see section 3.1.2.3). Hynes (1970) stated that the concentration of phosphate and nitrate in solution in stream or river water are normally low, because the ions are rapidly taken up by plants. Lund (1965) stated that because inorganic phosphorus is utilised rapidly and can be stored in excess of immediate needs, the total amount of phosphorus present may be a better index of the fertility of the water.

The average TP concentration measured in the Vaal River during the study period was 0.288 mg l^{-1} at the Barrage (Fig. 38), 0.276 mg l^{-1} at Parys (Fig. 39), 0.164 mg l^{-1} at Stilfontein (Fig. 40) and 0.146 mg l^{-1} at Balkfontein (Fig. 41). The above averages as well as results in Table 8 and Fig. 42 show a general decrease in the average TP concentration from the Barrage to Balkfontein. During 1992, however, the average TP concentration at Parys was somewhat higher than the average TP concentration recorded at the Barrage, but from Parys it decreased downstream to Balkfontein (Table 8). The TP concentration levels in the Vaal River are relatively high, as Wetzel (1983) indicated that the concentration of TP in flowing waters is generally less than 0.1 mg l^{-1}. The average TP concentrations recorded in the Vaal River during the present study at all four sampling localities fall within the range of TP concentrations for a eutrophic system (0.016 - 0.368 mg l^{-1}) given by Wetzel (1983).

TABLE 8: Minimum, maximum and averages of total nitrogen (TN) and phosphorus (TP) concentrations in mg l^{-1} as well as TN:TP ratios recorded in the water of the Barrage, Parys, Stilfontein and Balkfontein sampling localities for the three years of the study period.

	1991			1992			1993		
	TP	**TN**	**TN:TP**	**TP**	**TN**	**TN:TP**	**TP**	**TN**	**TN:TP**
BARRAGE:									
Minimum	0.15	1.44	4	0.15	0.551	2	0.15	0.98	2
Maximum	0.37	2.58	17	0.88	4.14	23	1.20	4.20	26
Average	0.19	1.94	11	0.23	1.64	9	0.38	2.43	11
PARYS:									
Minimum	-	-	-	0.19	1.38	5	0.03	0.39	2
Maximum	-	-	-	0.33	3.25	12	0.56	4.63	47
Average	-	-	-	0.28	2.32	8	0.27	2.52	11
STILFONTEIN:									
Minimum	0.05	1.06	4	0.06	0.86	5	0.02	0.64	5
Maximum	0.29	3.35	30	0.47	2.93	15	0.40	3.99	38
Average	0.16	1.96	14	0.18	1.54	9	0.15	1.68	15
BALKFONTEIN:									
Minimum	0.08	0.85	7	0.05	0.84	6	0.06	0.66	6
Maximum	0.31	2.27	25	0.38	2.44	19	0.40	4.08	27
Average	0.12	1.42	13	0.14	1.47	13	0.18	1.62	10

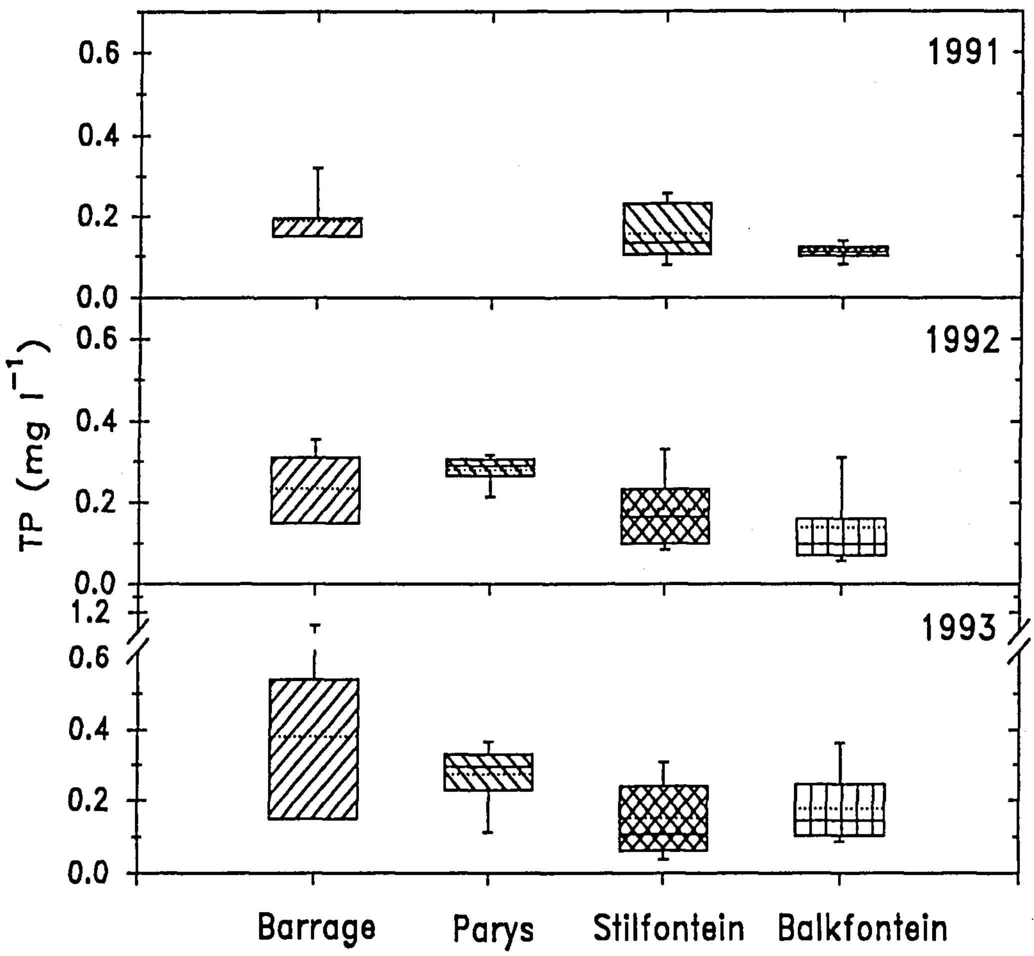

FIGURE 42: Box plot of annual total phosphorus (TP) concentration (mg l^{-1}) in the Vaal River at the Barrage, Parys, Stilfontein and Balkfontein.

Total nitrogen (TN)

The average TN concentration in the Vaal River at the Barrage was 2.043 mg l^{-1} (Fig. 43), at Parys 2.456 mg l^{-1} (Fig. 44), at Stilfontein 1.724 mg l^{-1} (Fig. 45) and at Balkfontein 1.509 mg l^{-1} (Fig. 46). Studies done by Vermeulen* showed an average TN concentration of 0.96 mg l^{-1} in the Vaal Dam. Higher TN concentrations downstream from the Vaal Dam indicate higher organic and inorganic nitrogen loading (especially between the Vaal Dam and Barrage) possibly because of agricultural, mining and industrial effluents. A comparison of Figs 47 and 33 shows that both TN and DIN concentrations increased from the Barrage to Parys sampling localities, whereafter a decrease occurred downstream to Stilfontein and Balkfontein. This shows that DIN (which consists mainly of NO_3-N) is an important component of the total nitrogen.

According to Wetzel (1983) and Roos (1992) the TN concentration of a eutrophic system ranges between 0.393 and 6.1 mg l^{-1}. The average TN concentrations at all four sampling localities during the present study, fall within this range so that the Vaal River can be classified as a eutrophic system also on account of its TN concentration.

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*A. Vermeulen, Department of Plant and Soil Sciences, PU for CHE, Potchefstroom: personal communication

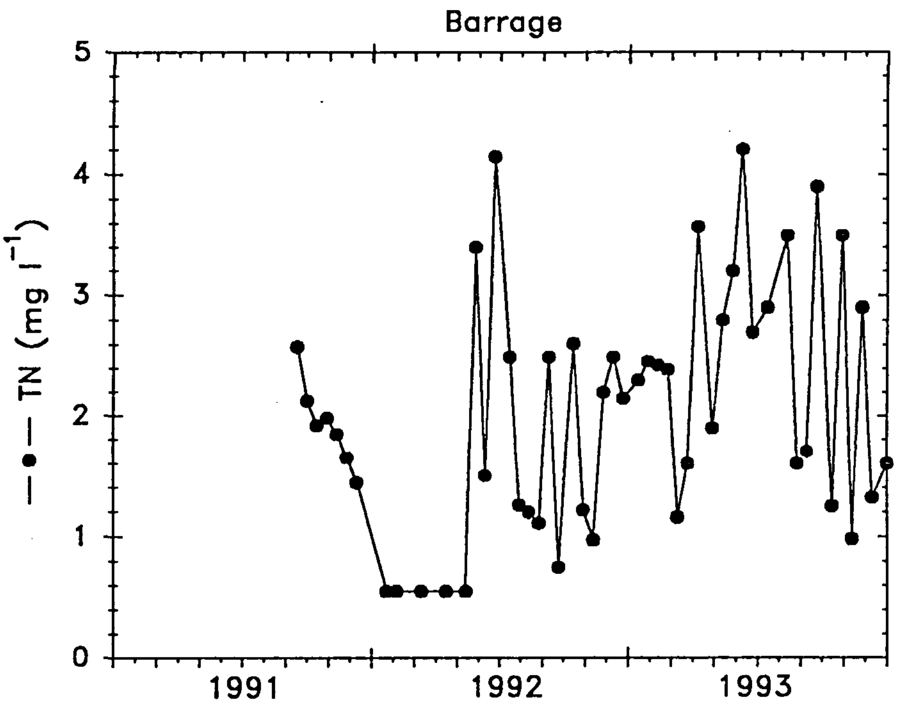


FIGURE 43: Variation in total nitrogen (TN) concentration (mg l^{-1}) at the Barrage during the study period.

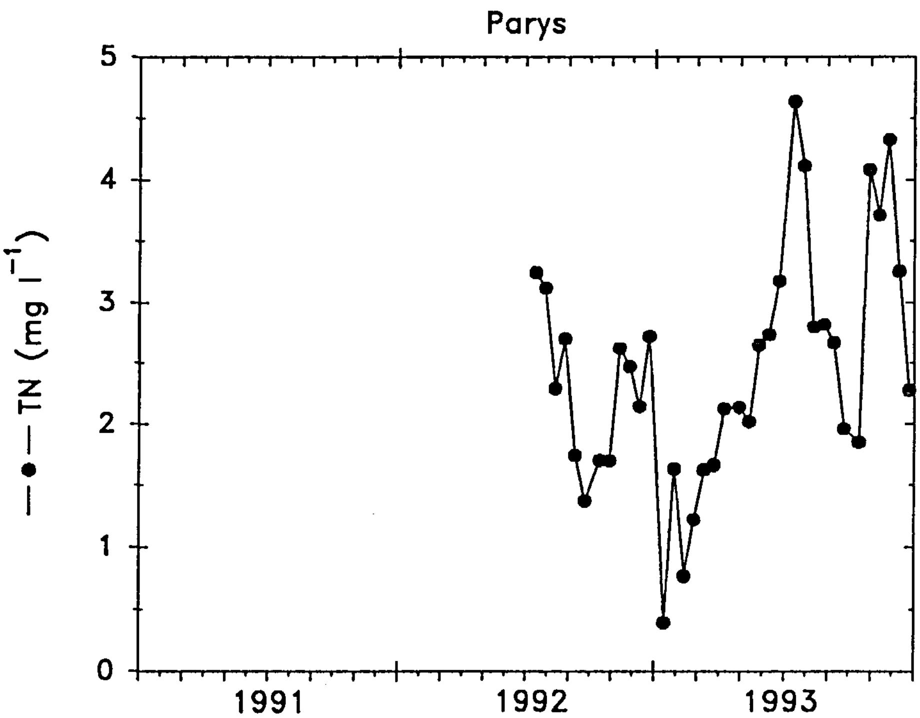


FIGURE 44: Variation in total nitrogen (TN) concentration (mg l^{-1}) at Parys during the study period.

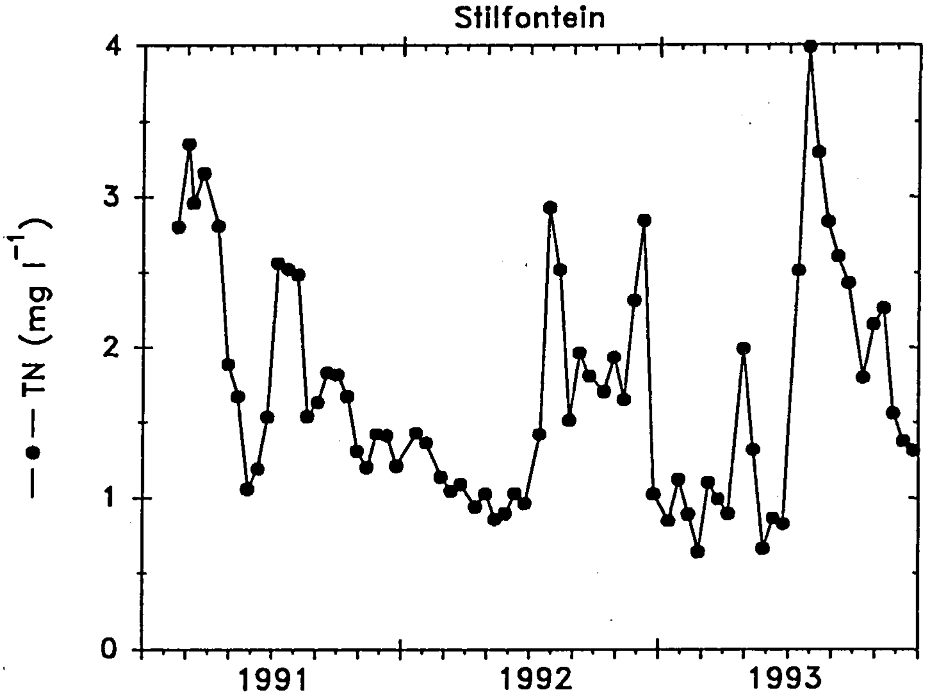


FIGURE 45: Variation in total nitrogen (TN) concentration (mg l^{-1}) at Stilfontein during the study period.

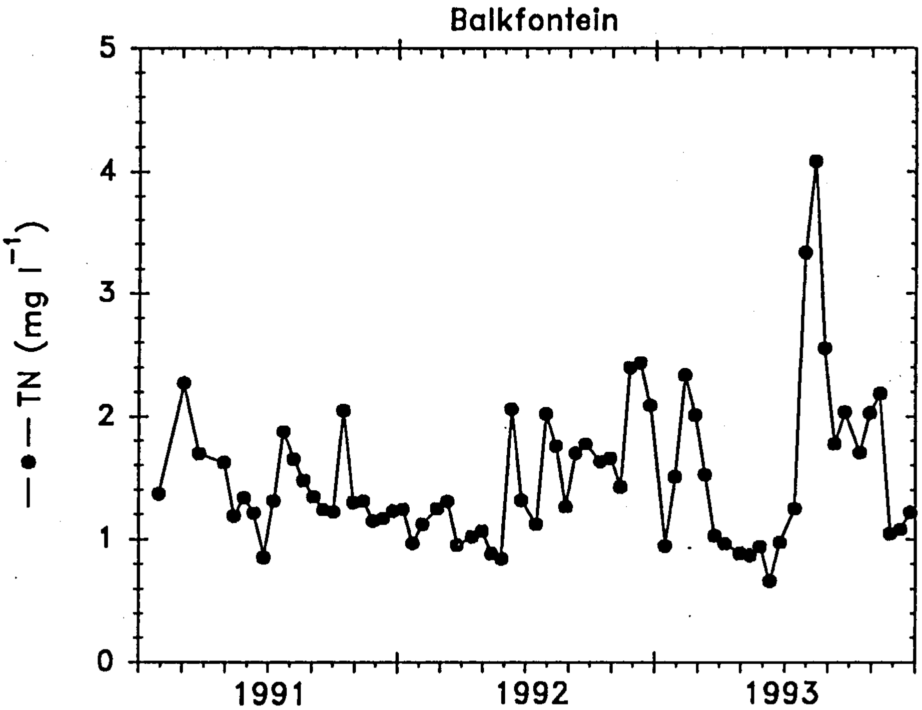


FIGURE 46: Variation in total nitrogen (TN) concentration (mg l^{-1}) at Balkfontein during the study period.

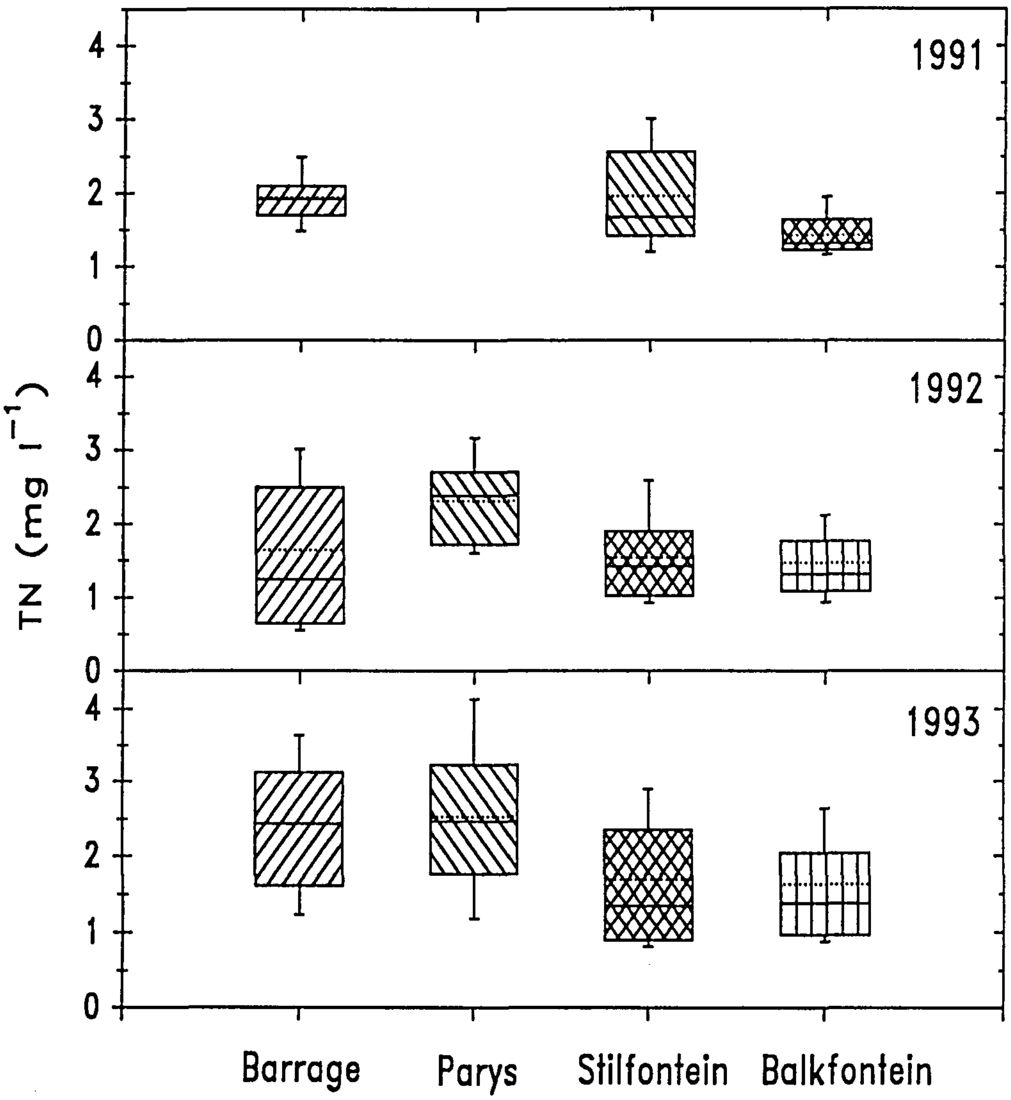


FIGURE 47: Box plot of annual total nitrogen (TN) concentration (mg l^{-1}) in the Vaal River at the Barrage, Parys, Stilfontein and Balkfontein.

TN:TP ratios

The average TN:TP ratios at the different sampling localities in the Vaal River was 9.9 at Barrage (ranging between 1.7 and 26; Fig. 48), 10.2 at Parys (ranging between 2.2 and 47; Fig. 49), 12.9 at Stilfontein (ranging from 4.1 to 37.7; Fig. 50) and 11.9 at Balkfontein (ranging from 5.5 to 27.2; Fig. 51). Correlation analysis done by Roos (1992) showed that the TN:TP ratio in the Vaal River is more sensitive to TP concentrations than to TN concentrations. The downstream decrease in average TP concentration (Fig. 42) was probably responsible for the increase in TN:TP downstream from the Barrage.

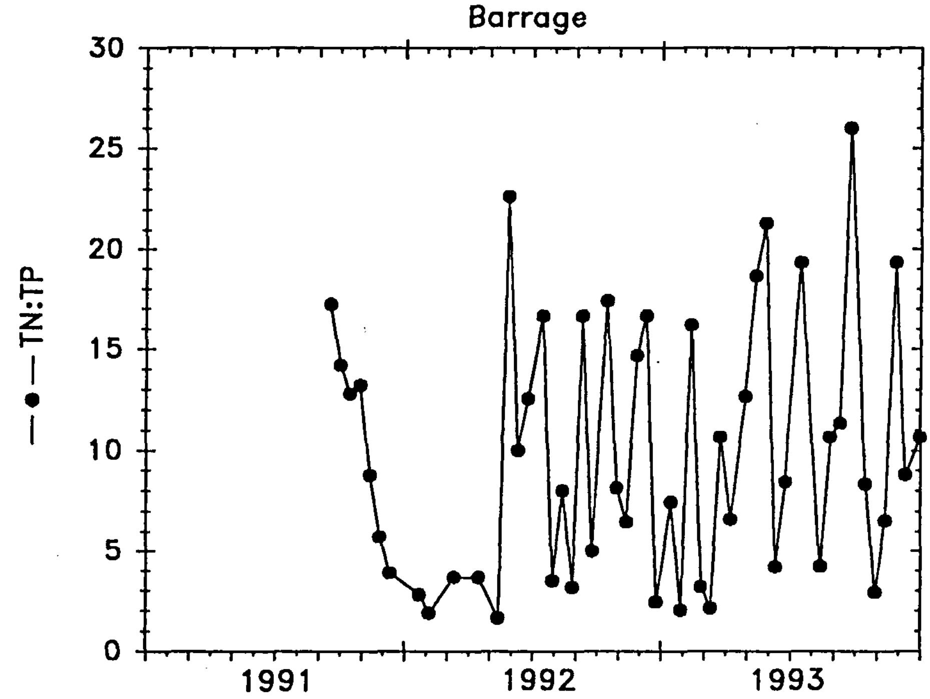


FIGURE 48: Variation in TN:TP ratios at the Barrage during the study period.

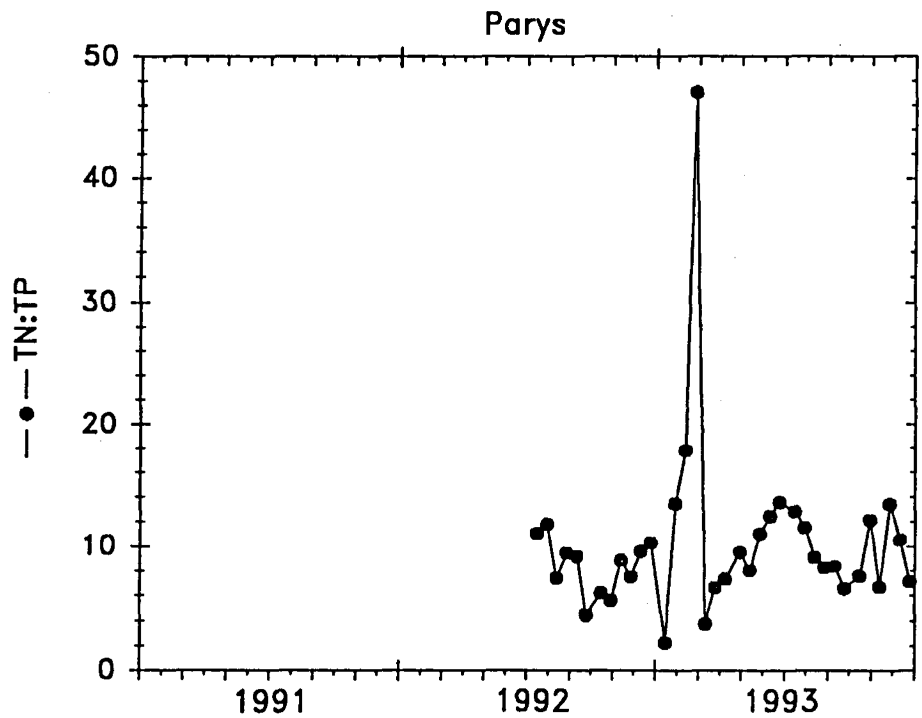


FIGURE 49: Variation in TN:TP ratios at Parys during the study period.

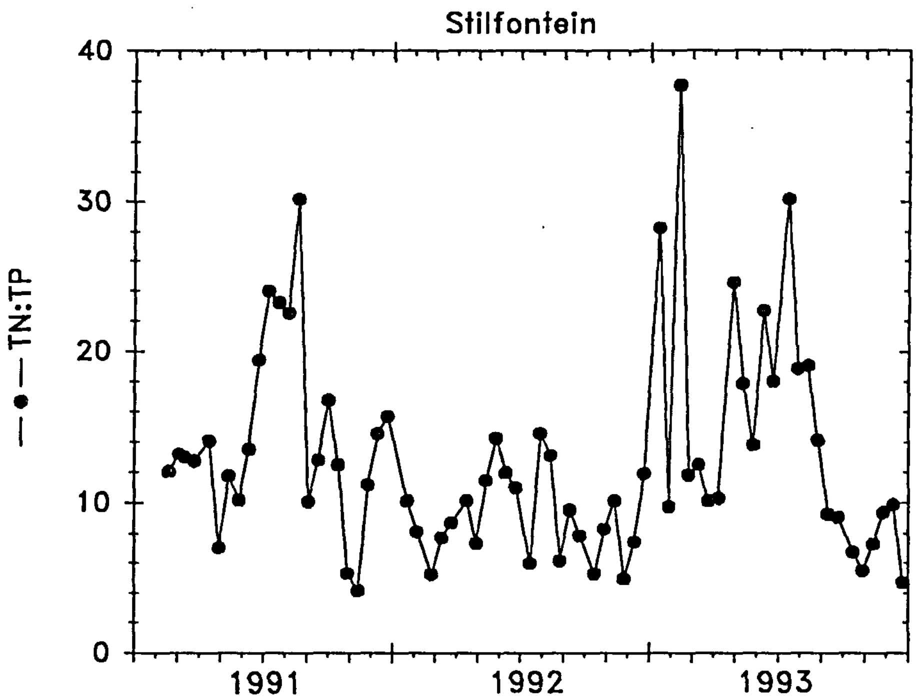


FIGURE 50: Variation in TN:TP ratios at Stilfontein during the study period.

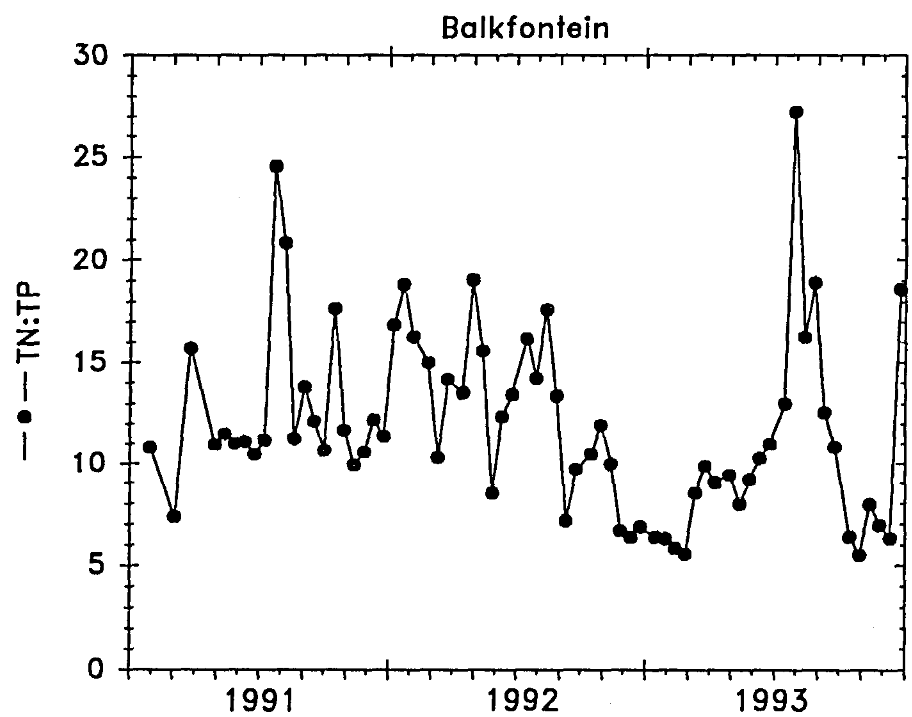


FIGURE 51: Variation in TN:TP ratios at Balkfontein during the study period.

Goldman and Horne (1983) showed that the release of phosphorus into rivers is related to the average slope of the drainage basin. As the flow-rate increases, relatively more phosphorus is released than nitrogen. This was probably the reason why the low TN:TP ratios were frequently encountered during periods of high discharges, especially at the Parys (compare Figs 49 & 4), Stilfontein (compare Figs 50 & 5) and Balkfontein (compare Figs 51 & 6) sampling localities. If statistical analysis on Barrage data show that the correlation of TN to flow was higher than that of TP to flow, it can be explained why the TN:TP ratio was high at the Barrage during periods of high discharge. Point and diffuse sources can also play a role in TN:TP ratios.

The correspondence between TN and TP concentrations and ratios and phytoplankton biomass will be discussed in sections 3.2.4.4 and 3.2.4.5.

Minimum, maximum and average TN and TP concentrations as well as TN:TP ratios at the different sampling localities for each year of the study period, are presented in Table 8.

3.1.2.5 SILICATE-SILICON (SiO_2-Si) AND SI:DIP RATIOS

Silicon (Si) is the second most abundant element in the lithosphere (Cole, 1983). During the present study the average SiO_2-Si concentration in the Vaal River at the Barrage was 3.4 mg l^{-1} (Fig. 52), at Parys 2.6 mg l^{-1} (Fig. 53), at Stilfontein 2.3 mg l^{-1} (Fig. 54) and at Balkfontein 1.7 mg l^{-1} (Fig. 55).

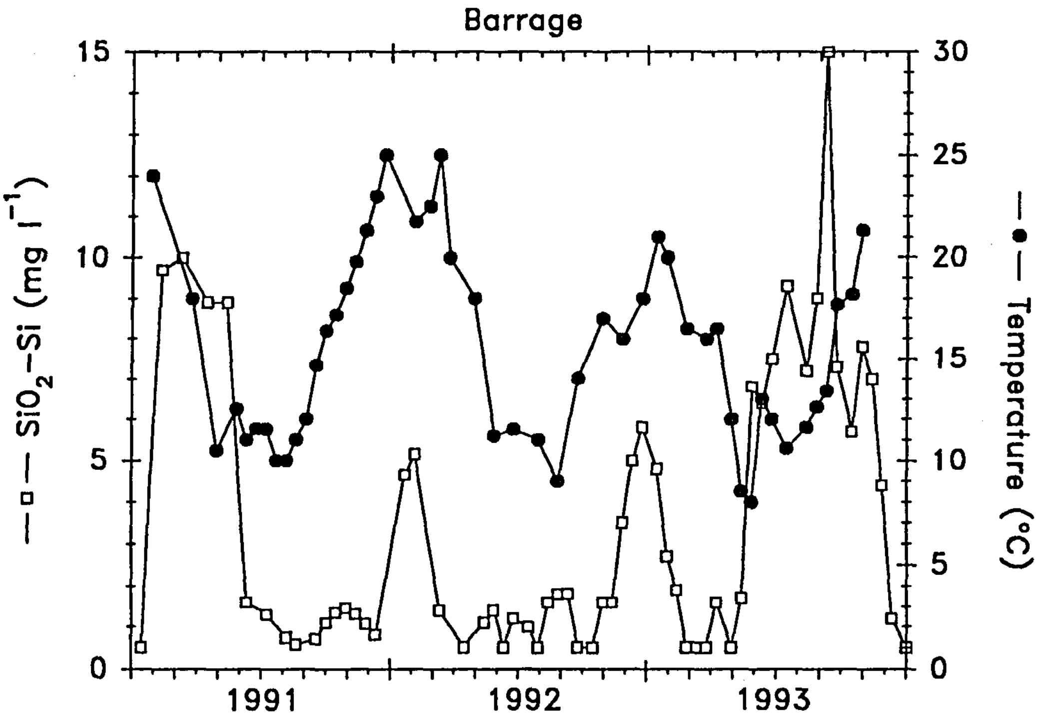


FIGURE 52: Variation in silicate-silicon (SiO_2-Si) concentration (mg l^{-1}) and temperature (°C) at the Barrage during the study period.

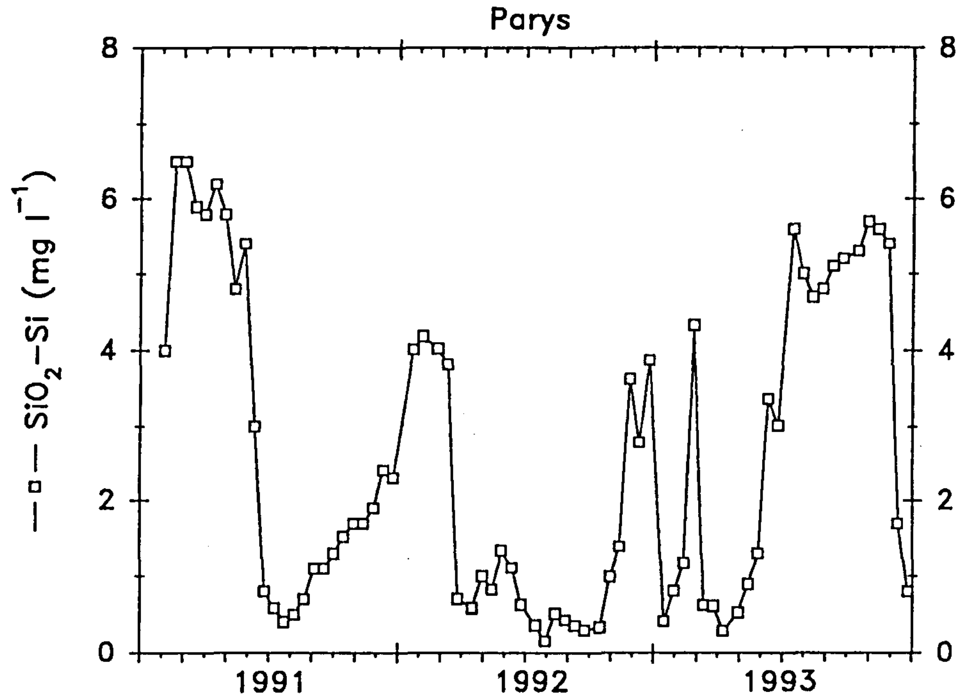


FIGURE 53: Variation in silicate-silicon (SiO_2-Si) concentration (mg l^{-1}) at Parys during the study period.

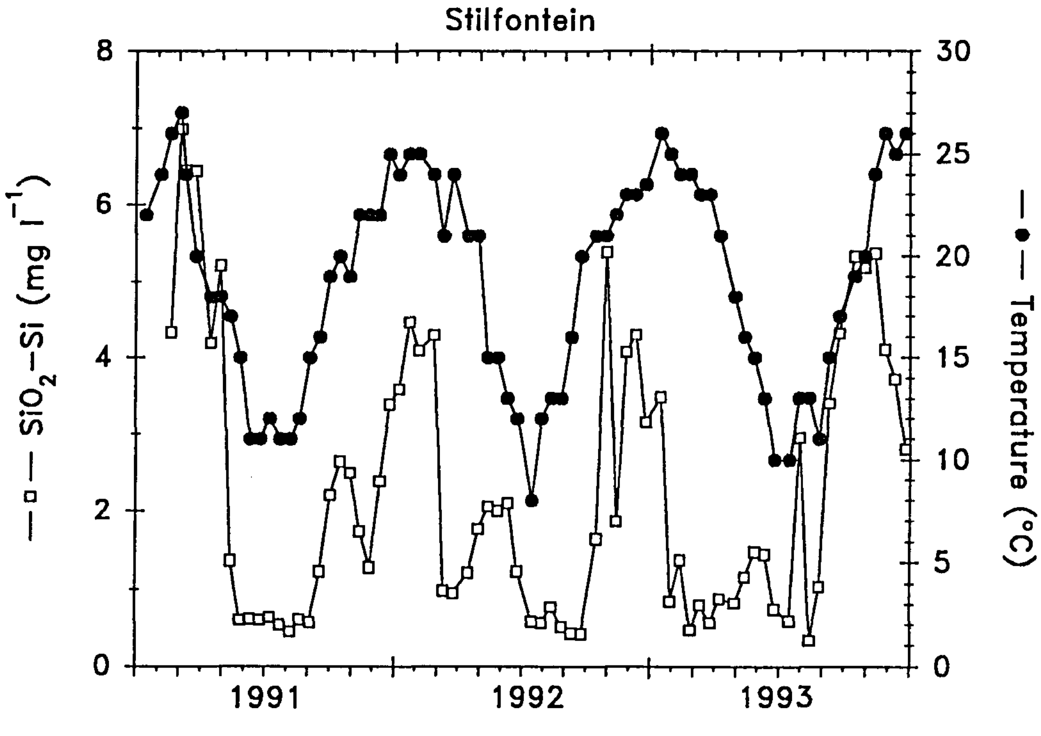


FIGURE 54: Variation in silicate-silicon (SiO_2-Si) concentration (mg l^{-1}) and temperature (°C) at Stilfontein during the study period.

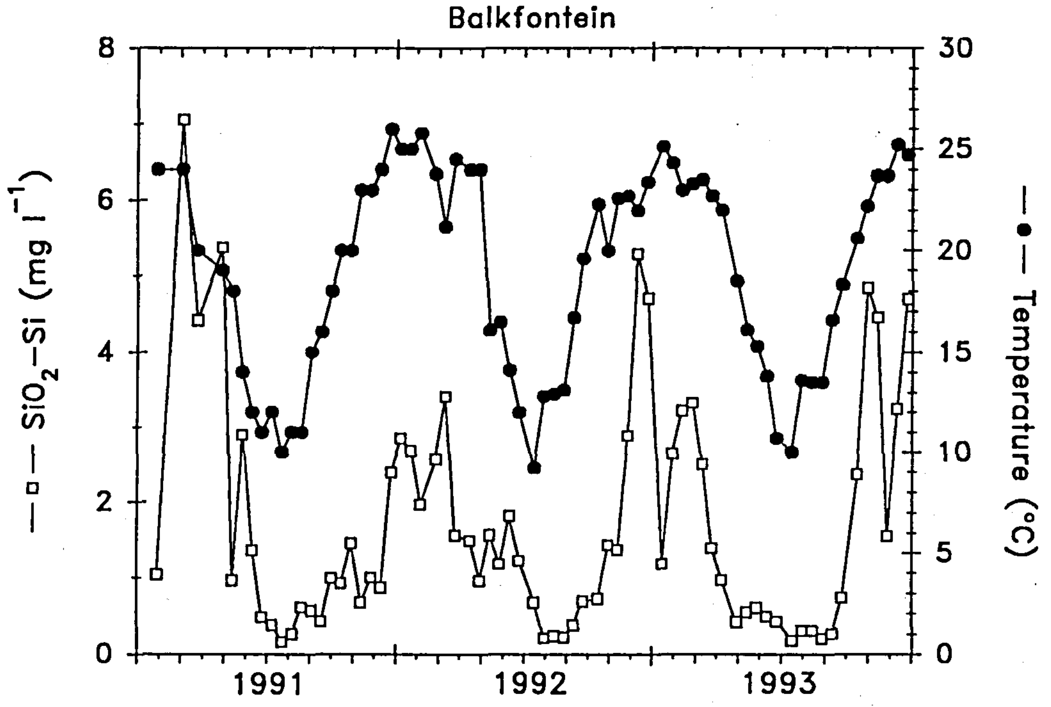


FIGURE 55: Variation in silicate-silicon (SiO_2-Si) concentration (mg l^{-1}) and temperature (°C) at Balkfontein during the study period.

The minimum, maximum and average SiO_2-Si concentration for each year is shown in Table 9. Averages presented in Table 9 as well as in Fig. 56 show that highest SiO_2-Si concentrations were reported at the upstream sampling locality (Barrage), whereafter a decrease occurred downstream to the Parys, Stilfontein and Balkfontein sampling localities. Interesting to note is that the concentration of diatoms at the Barrage was much lower than at the other sampling localities (Figs 82 to 85). Low diatom concentrations remove less SiO_2-Si from the water than high diatom densities. This can provide an explanation for the higher SiO_2-Si concentrations recorded at the Barrage (see section 3.2.4.6).

The average SiO_2-Si concentrations recorded in the Vaal River during the present study are much lower than the world average of 9.0 mg l^{-1} in rivers (Goldman & Horne, 1983) and are also lower than the average of 7.5 mg l^{-1} for European rivers (Wetzel, 1983). The average silica concentrations in 21 South African impoundments were 5.2 mg l^{-1} (calculated from data given by Walmsley & Butty, 1980).

Solubility of silica increases directly with temperature (Cole, 1983). Roos (1992) demonstrated a statistically significant correlation between water temperature and SiO_2-Si concentration in the Vaal River. During the present study at all four sampling localities, higher SiO_2-Si concentrations occurred in the summer periods when the water temperature was higher, while lower concentrations were present during the winter (Figs 52-55).

TABLE 9: Minimum, maximum and average silicate silicon concentration (mg l^{-1}) as well as Si:DIP ratios recorded in the water of the Barrage, Parys, Stilfontein and Balkfontein sampling localities for the three years of the study period.

	1991		1992		1993	
	SiO_2-Si	Si:DIP	SiO_2-Si	Si:DIP	SiO_2-Si	Si:DIP
BARRAGE:						
Minimum	0.5	2	0.5	3	0.5	2
aximum	10	76	5.8	45	15	480
Average	3.12	23	2.06	15	4.77	50
PARYS:						
Minimum	0.4	12	0.15	2	0.29	1.6
Maximum	6.5	406	4.19	276	5.7	374
Average	3	107	1.62	43	3.01	59
STILFONTEIN:						
Minimum	0.45	23	0.42	14	0.33	4
Maximum	6.99	3085	5.39	215	5.37	196
Average	2.48	313	2.18	76	2.21	57
BALKFONTEIN:						
Minimum	0.16	17	0.21	10	0.18	4
Maximum	7.06	299	5.29	396	4.84	99
Average	1.64	92	1.76	120	1.71	33

Although all phytoplankton have a requirement for small amounts of Si, certain chrysophyte and diatom genera in particular build and strengthen their cell coverings with amorphous silica polymers (Reynolds, 1984). The requirement for silica makes it an ecologically important environmental variable for chrysophytes and diatoms (Reynolds, 1984). Several studies indicated that changes in silica concentrations followed the wax and wane of diatom populations (e.g. Lund, 1964; Septhon & Harris, 1984). Together with the water temperature, diatom blooms have a marked impact on silica concentration in the Vaal River (see section 3.2.4.6). When a sudden decrease in SiO_2-Si concentration occurred during the summer period, it can often be ascribed to a bloom of diatoms (section 3.2.4.6). Increases in the diatom concentrations caused decreases in the concentration of SiO_2-Si. Silica concentration in the water is most probably controlled by temperature and diatom blooms.

The silica concentration in river water, according to Edwards and Liss (1973), tends to be remarkably uniform and shows little response to changes in discharge. However, Roos (1992) showed that the silica concentration in the Vaal River was higher during periods of high discharge. Golterman (1975b) reported that Rhine silt consists mainly of SiO_2-Si (also small amounts of PO_4-P and Fe). Because of the high silicon content of silt, Roos (1992) found that there was a statistically significant correlation between silica concentration and total suspended solids (TSS) and consequently with discharge, in the Vaal River. The correlation between silica, discharge and TSS could, however, be partially a coincidental correlation, because the Vaal River generally show higher discharge rates (high TSS) during summer months when the water temperature is high, which resulted in higher solubilities of SiO_2. During the present study no clear correspondence could be seen regarding the SiO_2-Si concentration and discharge. Because peaks of discharge during the present study were frequently recorded during the winter, it is possible that the correspondence between discharge and silica found by Roos (1992) was in fact a coincidental correlation, but a statistical analysis must, however, be done on data of the present study to give an indication of the correlation between discharge and SiO_2-Si concentration.

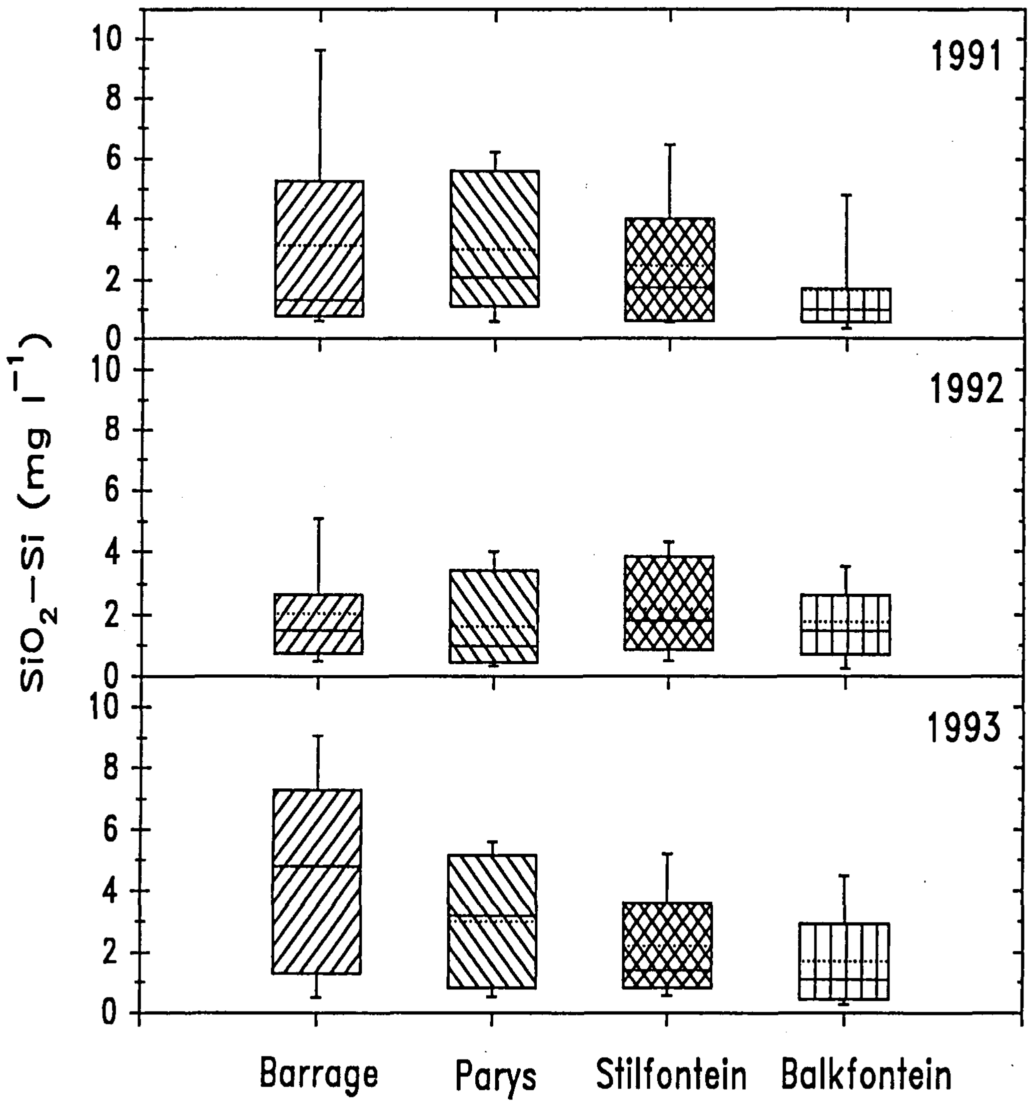


FIGURE 56: Box plot of annual silicate-silicon (SiO_2-Si) concentration (mg l^{-1}) in the Vaal River at the Barrage, Parys, Stilfontein and Balkfontein.

Golterman (1973) calculated that for several rivers in unpolluted areas of Uganda, the $Si:PO_4$ (Si:DIP) ratio was approximately 110. The Si:DIP ratio in the Vaal River was on average 31 at the Barrage (Fig. 57), 70 at Parys (Fig. 58), 141 at Stilfontein (Fig. 59) and 81 at Balkfontein (Fig. 60). The low Si:DIP ratio at the Barrage can be ascribed to enrichment of the river by PO_4-P at, and upstream from, this sampling locality (see section 3.1.2.3; Fig. 28). The higher ratios recorded at the other three sampling localities can be ascribed to lower DIP concentrations (Fig. 28), possibly because of biogenic PO_4-P uptake by algae and hyacinths as discussed in section 3.1.2.3. High average Si:DIP ratios at the Stilfontein sampling locality can also be ascribed to a peak of extremely high Si:DIP ratios (3 085) recorded in October 1991.

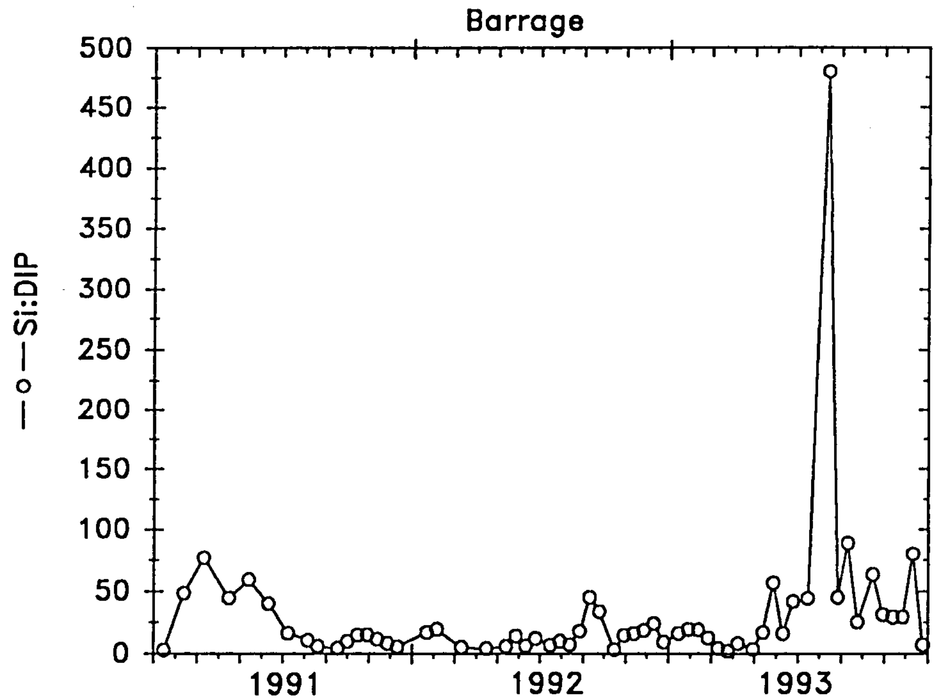


FIGURE 57: Variations in Si:DIP ratios at the Barrage during the study period.

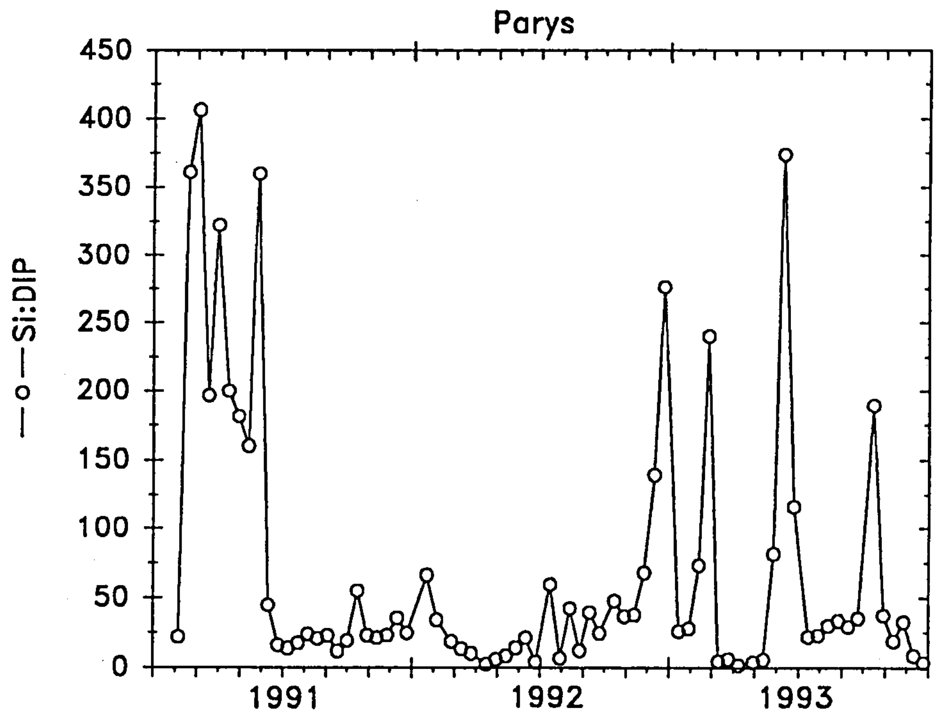


FIGURE 58: Variations in Si:DIP ratios at Parys during the study period.

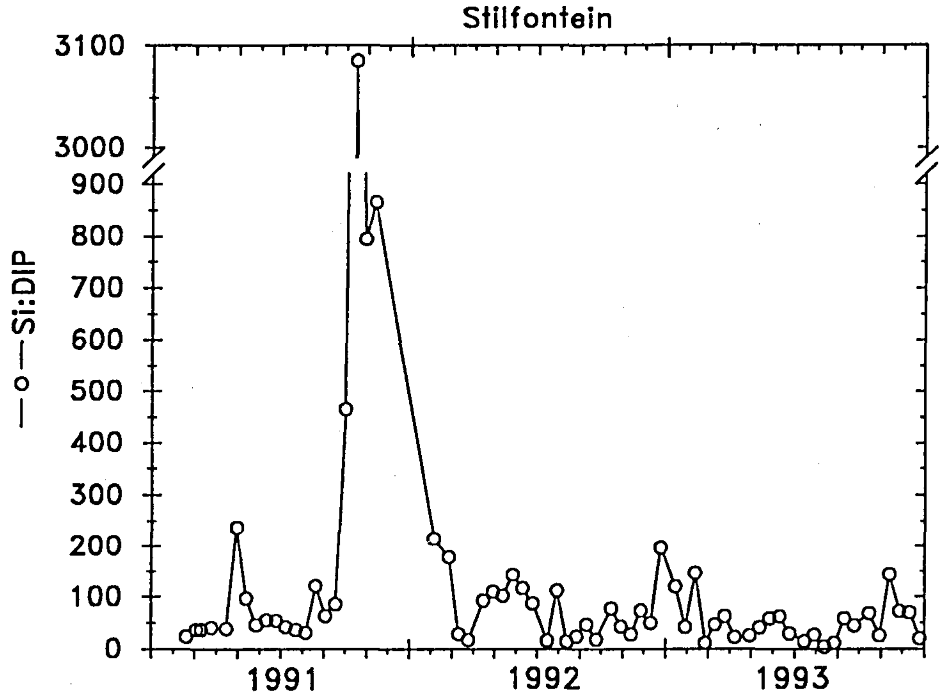


FIGURE 59: Variations in Si:DIP ratios at Stilfontein during the study period.

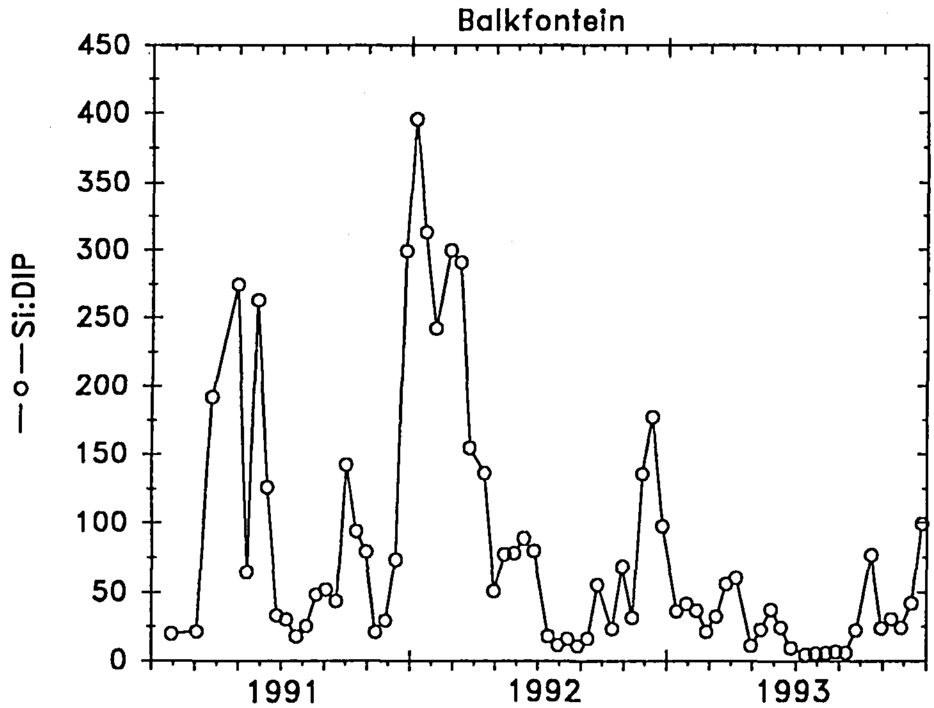


FIGURE 60: Variations in Si:DIP ratios at Balkfontein during the study period.

The relationship between Si:DIP ratios and algal composition, as well as that between silica and diatom abundance, will be discussed in section 3.2.4.6.

3.1.2.6 HYDROGEN ION CONCENTRATION

In natural water pH is governed to a large extent, by the increases in concentration of H^+ ions (from the dissociation of H_2CO_3) or OH^- ions (produced during the hydrolysis of bicarbonate; Wetzel, 1983).

Most fresh waters, including most of those in South Africa, are relatively well buffered and more or less neutral, with pH ranges around 6.5 to 8.5 (WHO, 1984). The pH ranges for fresh water given by WHO (1984) is in accordance with the situation found in the Vaal River during the present study where the average pH at the Barrage was 8.2 (ranging from 6.9 to 9.4; Fig. 61), at Parys 8.1 (ranging from 7.3 to 8.8; Fig. 62), at Stilfontein 8.1 (ranging from 7 to 8.8; Fig. 63) and at Balkfontein 8.2 (ranging from 7.3 to 8.6; Fig. 64). Minimum, maximum and average pH values for each year at the different sampling localities are presented in Table 10.

It was found that, when pH ranged between 6 and 9 in water for domestic purposes, no significant effects on health, or on aesthetics (taste) are expected. Very slight effects on taste may, on occasion, be noticed at the extremes of this range (Kempster and Smith, 1985; Aucamp and Vivier, 1990; Kempster and Van Vliet, 1991). Levels of pH vary widely

TABLE 10: Minimum, maximum and average pH recorded in the water of the Barrage, Parys, Stilfontein and Balkfontein sampling localities for the three years of the study period.

	1991	**1992**	**1993**
BARRAGE:			
Minimum	7.05	6.86	7.35
Maximum	8.81	9.2	9.37
Average	8.02	8.37	8.26
PARYS:			
Minimum	7.3	7.35	7.7
Maximum	8.5	8.8	8.64
Average	8.05	8.3	8.05
STILFONTEIN:			
Minimum	7.63	7.52	7.05
Maximum	8.52	8.85	8.52
Average	8.4	8.24	7.95
BALKFONTEIN:			
Minimum	7.28	8.01	7.76
Maximum	8.6	8.62	8.57
Average	8.05	8.28	8.21

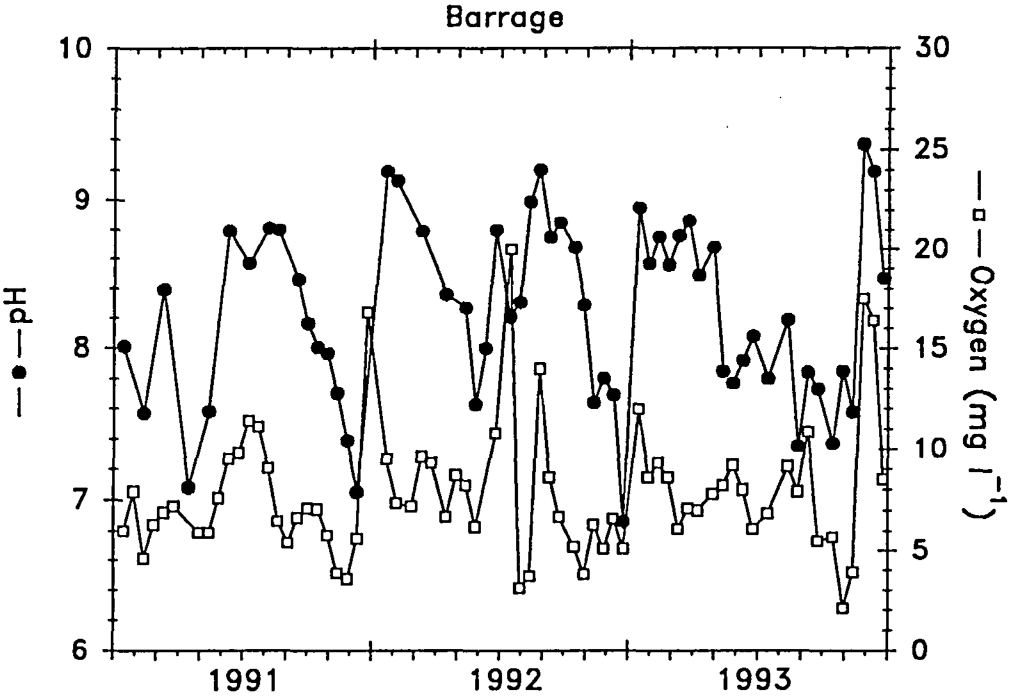


FIGURE 61: Variations in pH and oxygen concentration (mg l^{-1}) at the Barrage during the study period.

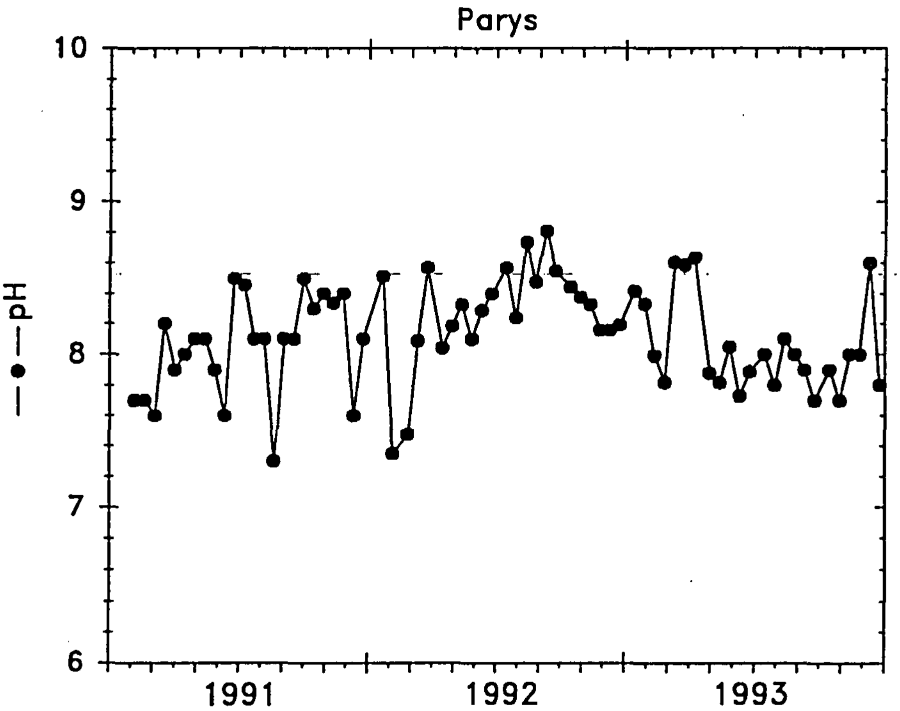


FIGURE 62: Variations in pH at Parys during the study period.

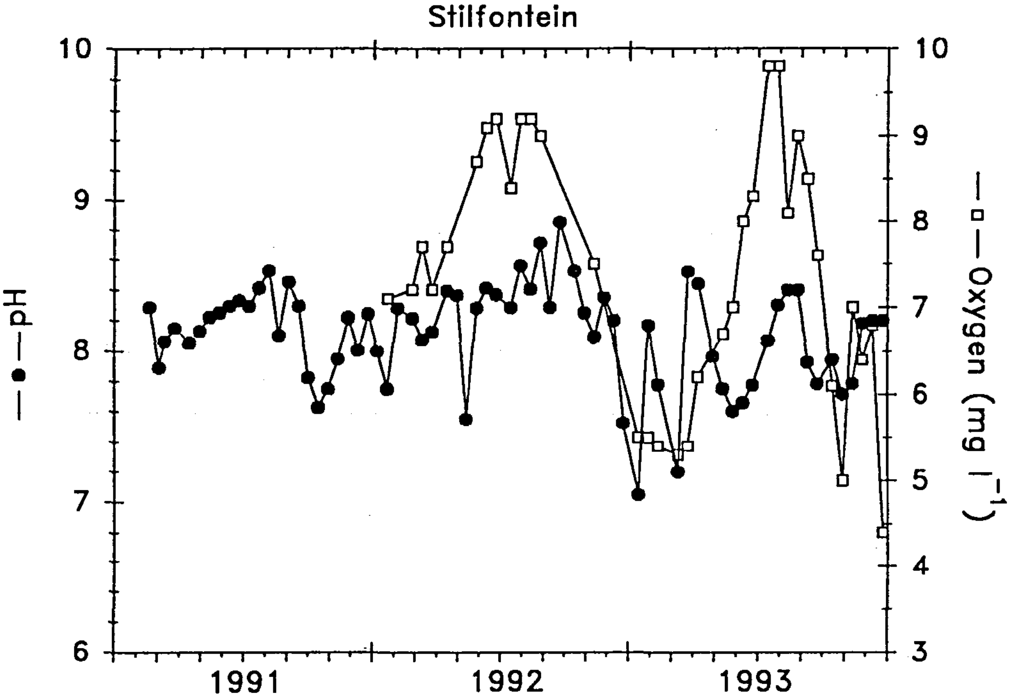


FIGURE 63: Variations in pH and oxygen concentration (mg l^{-1}) at Stilfontein during the study period.

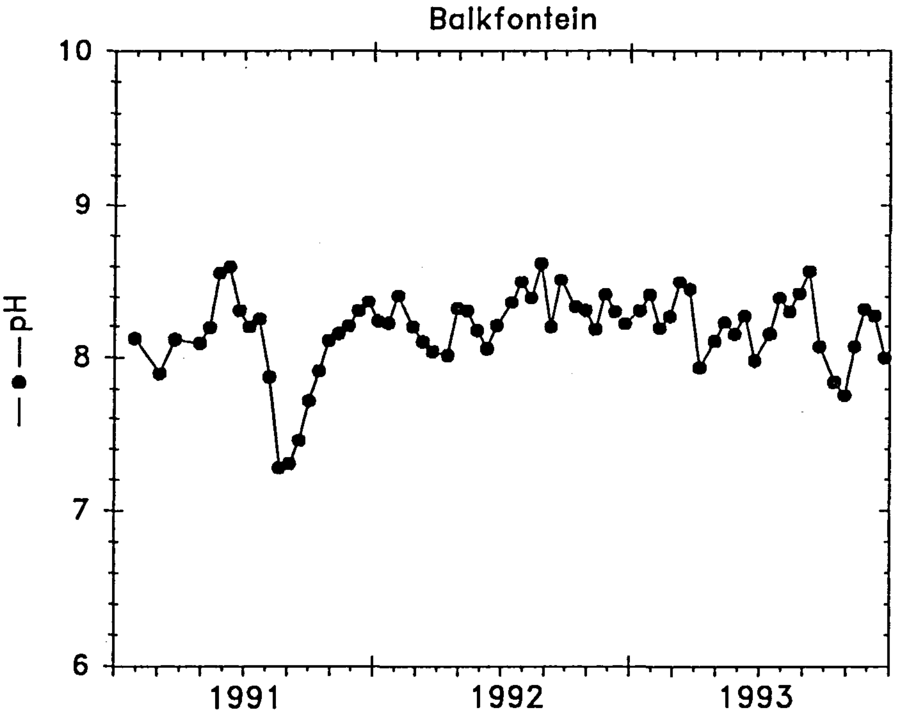


FIGURE 64: Variations in pH at Balkfontein during the study period.

between different rivers and streams, and are greatly influenced by bicarbonate-carbonate alkalinity and the concentration of free carbon dioxide (Talling, 1976a).

Extreme rates of photosynthesis, whether natural or as a result of eutrophication, commonly cause very high pH values in standing water (Dallas & Day, 1993). During daytime, dense populations of phytoplankton photosynthesise and algal cells assimilate carbon dioxide from the water, thus lowering the CO_2 concentration in the water, and increasing the pH. This process can occur only in sunlight. At night, the major biotic processes are respiration and decomposition, which absorb O_2, release CO_2 and result in a decrease in pH. Thus, the pH of eutrophic systems may exhibit wide diurnal fluctuations from < 6 to > 10 (Dallas & Day, 1993). Rivers seldom show these large fluctuations in pH because they rarely exhibit the extremes of eutrophication encountered in standing waters (Dallas & Day, 1993). It was, however, shown by Roos (1992) that algal photosynthesis had an effect on pH in the Vaal River. During the present study at all four sampling localities, increases in pH (Figs 61-64) were frequently recorded during periods when an increase in chlorophyll-*a* concentration and algal units occurred (Figs 77 to 80; see section 3.2.4.7). A statistical analysis will, however, be done to determine the significance of the correspondence between pH and chlorophyll-*a*.

By their photosynthetic activity algae do not only influence pH, but also oxygen concentration. The relation between pH and oxygen during the present study is discussed in section 3.1.2.7. The influence of pH variations on algal communities is discussed in section 3.2.4.7.

Cholnoky (1958), working in South Africa, was one of the first algologists to draw attention to the fact that diatoms exhibit very clear pH preferences. Cholnoky was able to estimate the pH of any stream from examining the diatom community composition found there.

3.1.2.7 DISSOLVED OXYGEN

One of the most important abiotic factors related to the survival of most aquatic organisms is the concentration of dissolved oxygen in the water. The measurement of dissolved oxygen (O_2), therefore, is one of the most frequently used and the most important of all chemical methods available for the investigation of the aquatic environment (Wetzel & Likens, 1979). The behaviour and basic relations of gases in streams follow the same fundamental physical and biochemical laws as in lentic situations (Reid & Wood, 1976).

Conditions modifying the concentration of dissolved oxygen in water were summarised and discussed by Dallas & Day (1993): An increase in dissolved oxygen concentration can be caused by re-aeration from the atmosphere, increase in atmospheric pressure, photosynthesis of aquatic plants and low temperatures. A decrease in dissolved oxygen can be the result of respiration by aquatic organisms, increase in salinity (decreases oxygen solubility), aerobic decomposition of organic material by micro-organisms or the chemical breakdown of pollutants.

Solubility of oxygen is affected non-linearly by temperature, and increases considerably in cold water (Wetzel, 1983). Oxygen concentrations in the Vaal River also showed seasonal trends, with peak values coinciding with low water temperatures and maximum phytoplankton biomass. In the Vaal River at the Barrage higher oxygen concentrations were measured during the winter periods of 1991 and 1992 (Fig. 61) when lower water temperatures were recorded (Fig. 52). During the summer months the solubility of oxygen decreased and lower oxygen concentrations were measured. Rapid increases in oxygen content during summer as well as rapid decreases during winter can be ascribed to increases and decreases of chlorophyll-*a* concentrations respectively (see section 3.2.4.8). The fact that no winter increase in oxygen concentration could be observed during 1993 at the Barrage can probably be ascribed to the very low algal (and chlorophyll-a) concentration present during June and July 1993 (see section 3.2.4.8). During 1992 and 1993 at the Stilfontein sampling locality an increase in dissolved oxygen concentration (Fig. 63) was also observed during the winter months, while it decreased during summer periods with higher water temperatures (Fig. 54). Decreases in oxygen concentrations during the winter periods as well as increases during the summer (Fig. 63) can again be ascribed to lower or higher phytoplankton biomass (see section 3.2.4.8). The average oxygen concentration measured at the Barrage (Fig. 61) was 7.9 mg l^{-1} and at Stilfontein (Fig. 63) it was 7.4 mg l^{-1}. Minimum, maximum and average oxygen concentration for each year at the different sampling localities are presented in Table 11.

Dissolved oxygen concentrations in water fluctuate diurnally in response to photosynthesis by plants (light periods) and respiratory activity (dark and light periods) of all aquatic organisms (Lloyd & Swift, 1976). Under natural conditions the concentration is usually lowest near

TABLE 11: Minimum, maximum and average dissolved oxygen concentration (mg l^{-1}) recorded in the water of the Barrage and Stilfontein sampling localities for the three years of the study period.

	1991	1992	1993
BARRAGE:			
Minimum	3.55.	3	2.12
Maximum	16.8	20	17.5
Average	7.39	8.37	8.26
STILFONTEIN:			
Minimum	-	7.1	9.8
Maximum	-	9.2	4.6
Average	-	8.25	6.9

dawn, it increases during the day, it peaks in the afternoon, and decreases during the night (Roos & Pieterse, 1992; Dallas & Day, 1993). During photosynthesis CO_2 is utilised and O_2 produced. Algal blooms usually result in a reduction of the CO_2 concentration, with an associated increase in pH concentration, accompanied by an increase in oxygen concentration (Roos, 1992). A statistically significant correlation was demonstrated between pH and O_2 by Roos (1992) at the Balkfontein sampling locality from 1985 to 1989.

A statistical analysis will be done to determine the significance of the correspondence between pH and O_2 in the Vaal River during the present study.

Roos (1992) showed that minimum oxygen concentrations in the Vaal River at Balkfontein (1985-1989) were recorded during the summer rain periods. A statistical analysis will be done to determine the significance of the correspondence between discharge and oxygen concentration during the present study.

3.2 PHYTOPLANKTON ASSEMBLAGES

Qualitative and quantitative knowledge of the organisms growing in an ecosystem is of fundamental importance in understanding the functioning of ecosystems (Vollenweider, *et al.*, 1974). From the analysis of the algal communities of a specific water body one can often deduce the chemical environment and the origin of the algal taxa found there (Uherkovich, 1984).

The dynamics of phytoplankton in rivers have not been investigated as intensively as in lakes and estuaries (Roos, 1992). In this section phytoplankton biomass and composition as well as environmental variables that influence their abundance, will be emphasised.

3.2.1 ALGAL GROUPS AND SPECIES REPRESENTATION

Seven major algal groups (taxa) were found in the Vaal River (Table 12) during the study period (1991 to 1993), namely the Cyanophyceae (blue-green algae), Bacillariophyceae (diatoms), Chlorophyceae (green algae), Cryptophyceae (cryptophytes), Chrysophyceae (golden algae), Dinophyceae (dinoflagellates) and the Euglenophyceae (euglenophytes).

Representatives of the centric and pennate diatoms, as well as the chrysophytes were not all identified to species levels, because of the difficulty to distinguish between different species by means of the light microscope. Further studies with the aid of the scanning electron microscope will make it possible to identify species of centric and pennate diatoms as well as the representative species of the Chrysophyceae. At least 13 species of centric diatoms were present in the Vaal River, while representatives of the pennate diatoms and chrysophytes were scarce during the entire study period.

Table 12 shows that the taxon with the greatest species diversity was the Chlorophyceae with a total of 66 identified species and 6 genera which have not yet been identified to species level. The Chlorophyceae was followed by the Euglenophyceae with 28 species (and two genera not identified to species level), the Bacillariophyceae (not less than 13 species), the Cyanophyceae (9 species), the Chrysophyceae (not less than 3 species), the Dinophyceae (2 species) and the Cryptophyceae (2 species). At least 124 species and varieties were identified from the Vaal River during the present study (Table 12). Apart from these 124 species, 12 species could only be identified to genus level.

Table 13 shows the species representation at the different sampling localities. Seventy-nine species (excluding the centric and pennate diatoms) occurred at all the sampling localities (70% of the total number of species identified - again excluding the centric and pennate diatoms). Species which occurred only at Barrage and Parys (upstream section of the study area), include *Anabaena circinalis*, *Microcystis incerta*, *Kirchneriella* sp. and *Spirogyra* sp.. Other species such as *Amphiprora* sp., *Cerasterias irregularis*, *Closterium cornu*, *Selenastrum minutum*, *Phacus ephippion* and *P. longicauda* only occurred at the downstream sampling localities, namely Stilfontein and Balkfontein. Species unique to the Barrage were *Anabaenopsis tanganyikae*, *Carteria cordiformis*, *Chlamydomonas conferta*, *Oedogonium* sp., *Scenedesmus ecornis*, *Mallomonas corymbosa* and *Mallomonas* sp.. *Carteria* sp., *Pediastrum* sp., *Euglena* sp. 2 and *Strombomonas schauinslandii* were unique to the Parys sampling locality. Three species were unique to Stilfontein, namely *Ankistrodesmus fusiformis*, *Monoraphidium pusillum* and *Strombomonas verrucosa* var. *zmiewika*. All the species unique to Balkfontein were euglenophytes, namely *Euglena fusca*, *Phacus orbicularis* and *Strombomonas gibberosa*. However, it cannot be stated categorically that species unique to a specific sampling locality only occurred at that specific locality. The volume of water used for counting purposes, represents only a small part of the total volume of water in the river. It is therefore possible that certain algal cells were not detected in the water sample investigated, although it may have been present in the river. During extended investigations, these species may be found to occur at one or more of the other sampling localities as well. The sampling locality with the smallest variety of species was Balkfontein (98 species; Table 13). At the Stilfontein sampling locality 102 species were present and at Parys and Barrage almost the same amount of species, namely 103, occurred. It seems as if the greatest diversity of Cyanophyceae species (9) occurred at the Barrage sampling locality, while the Balkfontein sampling locality showed the greatest diversity of Euglenophyceae species (28 species).

TABLE 12: List of algal species identified and algal units counted from the Vaal River water column (February 1991 - December 1993).

Algal species	**Algal units counted**
CYANOPHYCEAE	
Anabaena circinalis Rabenhorst	filament
Anabaenopsis tanganyikae (G.S. West) Wolosz *et* Miller	filament
Aphanocapsa littoralis Hansgirg	colony
Merismopedia minima G. Beck	colony
Microcystis aeruginosa Kütz.	colony
Microcystis flos-aquae (Wittr.) Kirchn.	colony
Microcystis incerta Lemm.	colony
Oscillatoria simplicissima Gomont	filament
Raphidocelis curvata F.E. Fritsch	filament
Synechococcus cedrorum Sauvagea	cell
BACILLARIOPHYCEAE	
Amphiprora sp.	cell
Centric sp. 2	cell
Cyclostephanos dubius (Fricke) Round	cell
Cyclotella atomus Hustedt	cell

Cyclotella meduanae Germain	cell
Cyclotella meneghiniana Kützing	cell
Cyclotella pseudostelligera Hustedt	cell
Melosira granulata (Ehr.) Ralfs.	cell
Melosira nyassensis O. Müller	cell
Stephanodiscus hantzschii fo. Stoermer *et* Håkansson	cell
Stephanodiscus invisitatus Hohn *et* Hell.	cell
Stephanodiscus sp.	cell
Thallassiosira duostra Pienaar *et* Pieterse	cell
Thallassiosira weissflogii (Grun.) Fryxel *et* Hasle	cell
Pennate diatoms	cell
CHLOROPHYCEAE	
Actinastrum gracillimum G.M. Smith	cell
Actinastrum hantzschii var. *subtile* Wolosz.	cell
Ankistrodesmus bibraianus (Reinsch) Kors.	cell
Ankistrodesmus falcatus (Corda) Ralfs	cell
Ankistrodesmus fusiformis Corda	colony
Ankistrodesmus stipitatus (Chod.) Kom. -Legn.	cell
Carteria cordiformis (Carter) Dill.	cell
Carteria fornicata Nyg.	cell
Carteria peterhofiensis Kiss	cell
Carteria simplicissima Pasch.	cell
Carteria sp.	cell
Cerasterias irregularis G.M. Smith	cell
Characium ensiforme Herm.	cell
Characium limneticum Lemm.	cell
Chlamydomonas bicocca Pasch.	cell
Chlamydomonas conferta Korsch.	cell
Chlamydomonas incerta Pasch.	cell
Chlamydomonas ulla Skuja	cell
Chlorococcum infusionum (Schrank) Menegh.	cell
Closterium cornu Ehr. ex Ralfs	cell
Coelastrum carpaticum Hind.	colony
Coelastrum pseudomicroporum Kors.	colony
Conococcus elongatus Carter	cell
Crucigenia lauterbornii (Schmidle) Schmidle	colony
Crucigenia tetrapedia (Kirsch.) West et. West	colony
Crucigeniella rectangularis (Näg.) Kom.	colony
Dictyosphaerium elegans Bachm.	colony
Eudorina elegans Ehrenberg	colony
Golenkinia radiata Chod.	cell
Kirchneriella sp.	colony
Lagerheimia balatonica (Scherff in Kol) Hind.	cell
Micractinium pusillum Fres.	colony
Monoraphidium arcuatum (Kors.) Hind.	cell
Monoraphidium circinale (Nyg.) Nyg.	cell

Monoraphidium griffithii (Berk.) Kom.-Legn.	cell
Monoraphidium minutum (Näg.) Kom.-Legn.	cell
Monoraphidium pusillum (Printz) Kom.-Legn.	colony
Oedogonium sp.	filament
Oocystis lacustris Chod.	cell
Oocystis marssonii Lemm.	cell
Oocystis pusilla Hansgirg	cell
Pandorina morum (Müller) Bory	colony
Pediastrum duplex Meyen	colony
Pediastrum simplex (Meyen) Lemm.	colony
Pediastrum sp.	colony
Pediastrum tetras (Ehr.) Ralfs	colony
Phacotus lenticularis (Ehr.) Stein	cell
cf. *Polytomella citri* Kater	cell
Pteromonas aculeata var. lemmermanii Skuja	cell
Pteromonas angulosa var. takedana West	cell
Quadrigula lacustris (Chod.) G.M. Smith	cell
Scenedesmus acuminatus (Lag.) Chodat	cell
Scenedesmus disciformis (Chod.) Fott & Kom.	cell
Scenedesmus dispar (Bréb.) Rabenh.	cell
Scenedesmus ecornis (Ehrenb.) Chod.	cell
Scenedesmus intermedius var. *balatonicus* Hortob.	cell
Scenedesmus lefevrii var. *manguinii* Lefév. *et* Bourr.	cell
Scenedesmus opoliensis var. *mononensis* Chod.	cell
Scenedesmus opoliensis var.*opoliensis* P. Richt.	cell
Schroederia indica Philipose	cell
Selenastrum minutum (Näg.) Collins	cell
Spirogyra sp.	filament
Staurastrum tetracerum var. *validum* W. & G.S. West	cell
Staurastrum tetracerum ?var.	cell
Tetraedron mediocris Hind.	cell
Tetraedron planctonicum G.M. Smith	cell
Tetraedron regulare var. *torsum* (Turner) Brunnthaler	cell
Tetrastrum heteracanthum var. *homoiacanthum* Hub. Pest.	colony
Tetrastrum staurogeniaeforme (Schroed.) Lemm.	colony
Thorakomonas feldmannii Bourr.	cell
Treubaria planctonica (G.M. Smith) Kors.	cell
Treubaria quadrispina (G.M. Smith) Fott & Kovac.	cell
Trochiscia prescotii Sieminska	cell
CRYPTOPHYCEAE	
Cryptomonas ?major	cell
Cryptomonas ?minor	cell
CHRYSOPHYCEAE	
Chrysophyte sp.	cell
Mallomonas corymbosa Asmund *et* Hilliard	cell

Mallomonas sp.	cell
DINOPHYCEAE	
Peridinium penardiforme Lindem.	cell
Sphaerodinium sp. nov. Pieterse *et* Theron	cell
EUGLENOPHYCEAE	
Euglena acus Eherenberg	cell
Euglena allorgei Defl.	cell
Euglena charkowiensis Swir.	cell
Euglena clavata Skuja	cell
Euglena elastica Prescott	cell
Euglena fusca (Klebs) Lemmerman	cell
Euglena hemichromata Skuja	cell
Euglena oblonga Schmitz	cell
Euglena pusilla var. *longa* Playfair	cell
Euglena sp. 1	cell
Euglena sp. 2	cell
Lepocinclis salina Fritsch	cell
Phacus acuminatus Stokes	cell
Phacus ephippion Pochm.	cell
Phacus longicauda Pochm.	cell
Phacus meson Pochm.	cell
Phacus orbicularis Hübner	cell
Phacus pyrum (Ehr.) Stein	cell
Strombomonas fluviatilis (Lemm.) Defl.	cell
Strombomonas gibberosa (Playf.) Defl.	cell
Strombomonas jaculata (Palmer) Defl.	cell
Strombomonas lanceolata (Playf.) Defl.	cell
Strombomanas longicauda (Swir.) Defl.	cell
Strombomonas ovalis (Playf.) Defl.	cell
Strombomonas schauinslandii (Lemm.) Defl.	cell
Strombomonas verrucosa var. *borystheniensis* (Roll.) Defl.	cell
Strombomonas verrucosa var. *zmiewika* (Swir.) Defl.	cell
Trachelomonas hispida (Perty) Stein emend Defl.	cell
Trachelomonas intermedia Dangeard	cell
Trachelomonas scabra Playf.	cell
Trachelomonas volvocina Ehrenberg	cell

TABLE 13: Species representation at the different sampling localities (February 1991 - December 1993).

	BAR	PAR	STF	BALK
CYANOPHYCEAE				
Anabaena circinalis	X	X	-	-
Anabaenopsis tanganyikae	X	-	-	-
Aphanocapsa littoralis	X	X	X	X
Merismopedia minima	X	-	X	X
Microcysts aeruginosa	X	X	X	X
Microcystis flos-aquae	X	X	X	X
Microcystis incerta	X	X	-	-
Oscillatoria simplicissima	X	X	X	X
Raphidocelis curvata	x	x	x	x
Synechococcus cedrorum	X	X	X	X
BACILLARIOPHYCEAE				
Amphiprora sp.	-	-	X	X
Centric sp. 2				
Cyclostephanos dubius				
Cyclotella atomus				
Cyclotella meduanae				
Cyclotella meneghiniana				
Cyclotella pseudostelligera				
Melosira granulata	X	X	X	X
Melosira nyassensis	-	X	X	-
Stephanodiscus hantzschii fo. *tenuis*				
Stephanodiscus invisitatus				
Stephanodiscus sp.				
Thallassiosira duostra				
Thallassiosira weissflogii				
Pennate diatoms	X	X	X	X
Centric diatoms	X	X	X	X
CHLOROPHYCEAE				
Actinastrum gracillimum	X	X	X	-
Actinastrum hantzschii var. *subtile*	X	X	X	X
Ankistrodesmus bibraianus	X	X	X	X
Ankistrodesmus falcatus	X	X	X	X
Ankistrodesmus fusiformis	-	-	X	-
Ankistrodesmus stipitatus	X	X	X	X
Carteria cordiformis	X	-	-	-
Carteria fornicata	X	X	X	X
Carteria peterhofiensis	X	X	X	-
Carteria simplicissima	X	X	X	X
Carteria sp.	-	X	-	-
Cerasterias irregularis	-	-	X	X
Characium ensiforme	X	X	X	-
Characium limneticum	X	X	X	X
Chlamydomonas bicocca	X	X	X	X
Chlamydomonas conferta	X	-	-	-
Chlamydomonas incerta	X	X	X	X
Chlamydomonas ulla	X	X	X	X
Chlorococcum infusionum	X	X	X	X
Closterium cornu	-	-	X	X
Coelastrum carpaticum	X	X	X	X
Coelastrum pseudomicroporum	X	X	X	X

Conococcus elongatus	X	X	X	X
Crucigenia lauterbornii	X	X	-	X
Crucigenia tetrapedia	X	X	X	X
Crucigeniella rectangularis	X	X	X	X
Dictyosphaerium elegans	X	X	X	X
Eudorina elegans	X	X	X	-
Golenkinia radiata	X	X	X	X
Kirchneriella sp.	X	X	-	-
Lagerheimia balatonica	X	X	X	X
Micractinium pusillum	X	X	X	X
Monoraphidium arcuatum	X	X	X	X
Monoraphidium circinale	X	X	X	X
Monoraphidium griffithii	X	X	X	X
Monoraphidium minutum	X	X	X	X
Monoraphidium pusillum	-	-	X	-
Oedogonium sp.	X	-	-	-
Oocystis lacustris	X	X	X	X
Oocystis marssonii	X	X	X	X
Oocystis pusilla	X	X	X	X
Pandorina morum	X	X	X	X
Pediastrum duplex	X	X	X	X
Pediastrum simplex	X	X	X	X
Pediastrum sp.	-	X	-	-
Pediastrum tetras	X	X	X	X
Phacotus lenticularis	X	X	X	X
cf. *Polytomella citri*	X	X	X	-
Pteromonas aculeata	X	X	X	X
Pteromonas angulosa	X	X	X	-
Quadrigula lacustris	X	X	X	X
Scenedesmus acuminatus	X	X	X	X
Scenedesmus disciformis	X	X	X	X
Scenedesmus dispar	X	X	X	X
Scenedesmus ecornis	X	-	-	-
Scenedesmus intermedius	X	X	X	X
Scenedesmus lefevrii	X	X	X	X
Scenedesmus opoliensis var. *mononensis*	X	X	X	X
Schroederia indica	X	X	X	-
Selenastrum minutum	-	-	X	X
Spirogyra sp.	X	X	-	-
Staurastrum tetracerum	X	X	X	X
Staurastrum tetracerum var. ???	X	-	X	X
Tetraedron mediocris	X	X	X	X
Tetraedron planctonicum	X	X	X	X
Tetraedron regulare	X	X	X	X
Tetrastrum heteracanthum	X	X	X	X
Tetrastrum staurogeniaeforme	X	X	X	X
Thorakomonas feldmannii	X	X	X	X
Treubaria planctonica	X	X	X	X
Treubaria quadrispina	X	X	X	X
Trochiscia prescotii	X	X	X	X
CRYPTOPHYCEAE				
Cryptomonas major	X	X	X	X
Cryptomonas minor	X	X	X	X
CHRYSOPHYCEAE				
Chrysophyte sp.	X	X	X	X
Mallomonas corymbosa	X	-	-	-
Mallomonas sp.	X	-	-	-

DINOPHYCEAE				
Peridinium penardiforme	X	X	X	X
Sphaerodinium sp. nov.	X	X	X	X
EUGLENOPHYCEAE				
Euglena acus	-	X	-	X
Euglena allorgei	X	X	X	X
Euglena charkowiensis	X	X	-	X
Euglena clavata	X	X	X	X
Euglena elastica	X	X	X	X
Euglena fusca	-	-	-	X
Euglena hemichromata	X	X	X	X
Euglena oblonga	X	X	X	X
Euglena pusilla var. *longa*	X	X	X	X
Euglena sp. 1	X	X	X	X
Euglena sp. 2	-	X	-	-
Lepocinclis salina	X	X	X	X
Phacus acuminatus	X	X	X	X
Phacus ephippion	-	-	X	X
Phacus longicauda	-	-	X	X
Phacus meson	-	X	X	X
Phacus orbicularis	-	-	-	X
Phacus pyrum	X	X	X	X
Strombomonas fluviatilis	X	X	X	X
Strombomonas gibberosa	-	-	-	X
Strombomonas jaculata	-	X	-	X
Strombomonas lanceolata	X	X	X	X
Strombomonas longicauda	-	X	X	X
Strombomonas ovalis	X	X	X	X
Strombomonas schauinslandii	-	X	-	-
Strombomonas verrucosa var. *borysthenienis*	X	X	X	X
Strombomonas verrucosa var. *zmiewika*	-	-	X	-
Trachelomonas hispida	X	X	X	X
Trachelomonas intermedia	X	X	X	X
Trachelomonas scabra	X	X	X	X
Trachelomonas volvocina	X	X	X	X
TOTAL	**103**	**103**	**102**	**98**

In Table 14 the dominant algal species (present in highest concentration) and the number of times when they were dominant at the different sampling localities, are shown. The most dominant group was the Bacillariophyceae, which dominated 197 times out of the 265 samples investigated (74% of the times). The unicellular centric diatoms dominated 141 times, while the filamentous centric diatom, *Melosira granulata*, was dominant 56 times. The Bacillariophyceae was followed by the Chlorophyceae (green algal representatives) which were dominant 54 times (20% of the time) during the study period. The most dominant Chlorophyceae species was *Chlamydomonas incerta* (dominant 32 times). The Cyanophyceae dominated 10 times from 1991 to 1993. The Cyanophyceae species which dominated most frequently, was *Oscillatoria simplicissma* (dominant 8 times).

TABLE 14: Dominant algal species from the Vaal River water column and their number of occurrence (February 1991 - December 1993)

	BAR	PAR	STF	BAL	TOT
CYANOPHYCEAE					
Oscillatoria simplicissima	3		2	3	8
Microcystis aeruginosa	1				1
Synechococcus cedrorum			1		1
Total Cyanophyceae					**10**
BACILLARIOPHYCEAE					
Melosira granulata	20	23	8	5	56
Unicellular centric diatoms	28	27	49	37	141
Total Bacillariophyceae					**197**
CHLOROPHYCEAE					
Actinastrum hantzschii				1	1
Ankistrodesmus stipitatus				2	2
Carteria simplicissima	3			1	4
Chlamydomonas bicocca				1	1
Chlamydomonas incerta	5	12	4	11	32
Crusigeniella rectangularis		1			1
Scenedesmus lefevrii var. manguinii				2	2
Scenedesmus opoliensis var. mononensis	1	2	1	4	
Schroederia indica	3				3
Dictyosphaerium elegans	1	1			2
Oocystis lacustris			1	1	2
Total Chlorophyceae					**54**
CRYPTOPHYCEAE					
Cryptomonas major	1		1	1	3
Total Cryptophyceae					**3**
DINOPHYCEAE					
Sphaerodinium sp. nov.				1	1
Total Dinophyceae					**1**

BAR = Barrage; PAR = Parys; STF = Stilfontein; BAL = Balkfontein; TOT = Total

3.2.2 PHYTOPLANKTON COMPOSITION AND BIOMASS

The phytoplankton composition, total algal units and chlorophyll-*a* concentrations at the different sampling localities are shown in Figs 65 to 68.

3.2.2.1 BARRAGE

Fig. 65B shows that the phytoplankton community at the Barrage was dominated mainly by diatoms and green algae as well as blue-green algae during the mid and late summer months. A sudden bloom of Cryptophyceae occurred during the middle of August 1991. During this time total algal units (mostly Cryptophyceae cells) reached values of more than 16 000 cells ml^{-1} (Fig. 65A). This bloom will again be referred to when the downstream variation in phytoplankton composition will be discussed in later paragraphs. A bloom of Cyanophyceae (mainly *Anabaena circinalis*) occurred during December 1991, followed by a very large bloom of *Oscillatoria simplicissima* (chlorophyll-*a* concentration 74 µg l^{-1}; Fig. 65A) which formed thick scums on the surface of the water during the end of January 1992. During this period *O. simplicissima* comprised almost eighty percent of the phytoplankton assemblages present. *O. simplicissima* filaments were also present in high concentrations during April 1992. During May a bloom of *Microcystis aeruginosa* occurred. The second dominant species during this period was *Anabaena circinalis*, also a representative of the blue-green algae. The Cyanophyceae comprised seventy percent of the phytoplankton composition during this period. During the end of 1992 and the beginning of 1993 two blooms of *Oscillatoria simplicissima* were again observed. Although Cyanophyceae representatives were present during November and December 1993, they did not reach dominant proportions. It seems as if blue-green algae repeatedly occurred during the mid and late summer months of each year, frequently reaching dominant proportions. Other algal groups were not well represented at the Barrage, although representatives of the Euglenophyceae did occur and representatives of the Cryptophyceae became more important from the middle of 1992 to the end of 1993.

In Fig. 65A a comparison between the chlorophyll-*a* concentration and the total number of algal units ml^{-1} of water can be seen. Several periods of high cell concentrations, accompanied by high chlorophyll-*a* values, can be identified, namely during the middle of August 1991, where high cell concentrations can be ascribed to a bloom of Cryptophyceae as described above (Fig. 65B). High chlorophyll-*a* concentrations during December 1991 were caused by representatives of green- and blue-green algae (*Oscillatoria simplicissima* the dominant species; compare Figs 65B & 69), while high chlorophyll-*a* concentrations recorded during March 1992 were the result of a bloom of *Melosira granulata* (compare Figs 65B & 69). During August 1992 total algal units (mostly *Chlamydomonas incerta* cells) reached values of almost 20 000 cells ml^{-1}. High chlorophyll-*a* concentrations measured during the middle of February 1993 could be ascribed to a bloom of *Oscillatoria simplicissima* (see Figs 65B & 69). During November 1993 a very high chlorophyll-*a* concentration (almost 150 µg l^{-1}), accompanied by a total algal unit concentration of ± 26 000 units ml^{-1}, was the result of a bloom of *Carteria simplicissima* (compare Fig. 69).

The minimum, maximum and average chlorophyll-*a* concentrations measured during the three years of the study period at the Barrage sampling locality, are given in Table 15, while the minimum, maximum and average number of algal units ml^{-1} are given in Table 16.

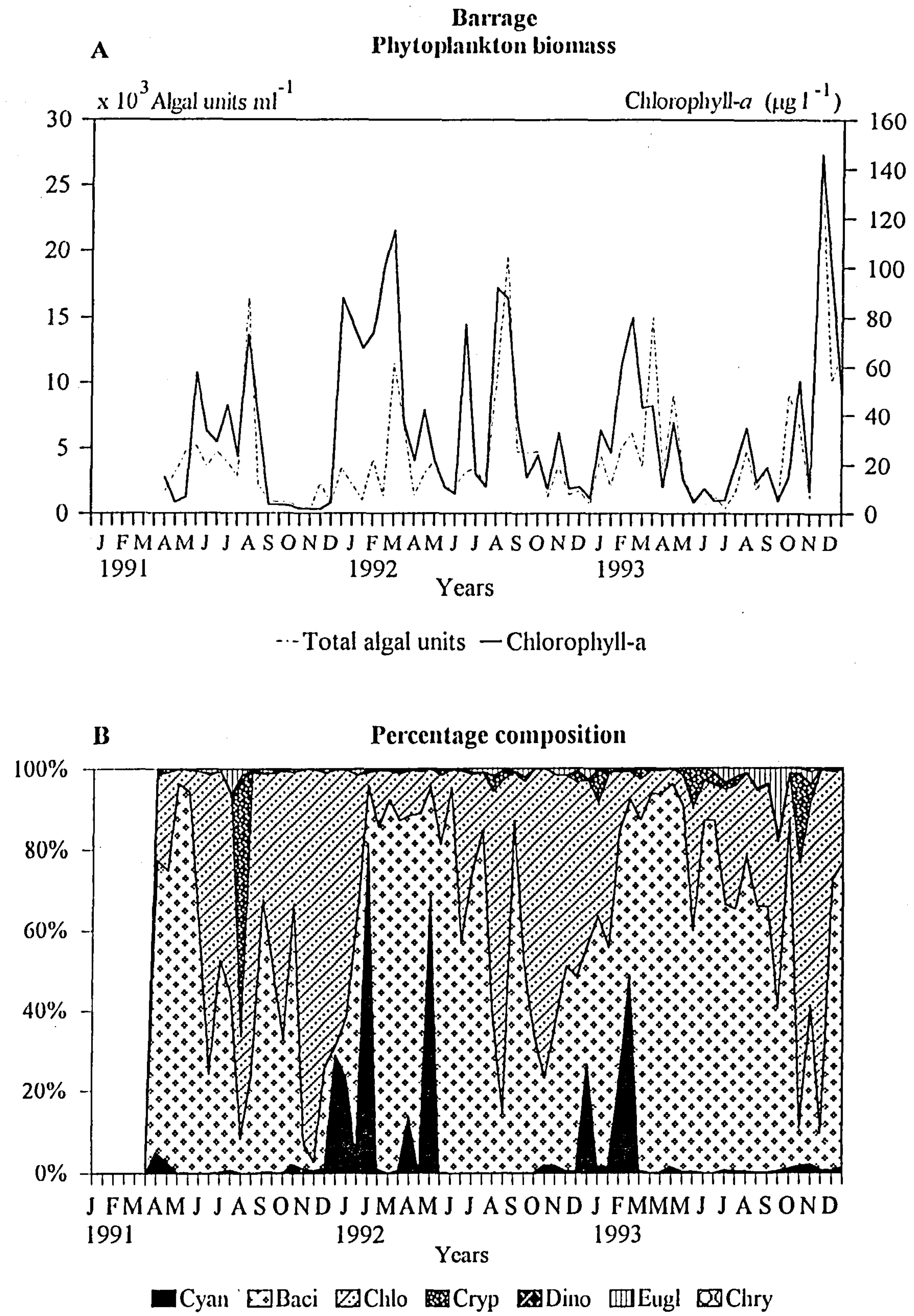


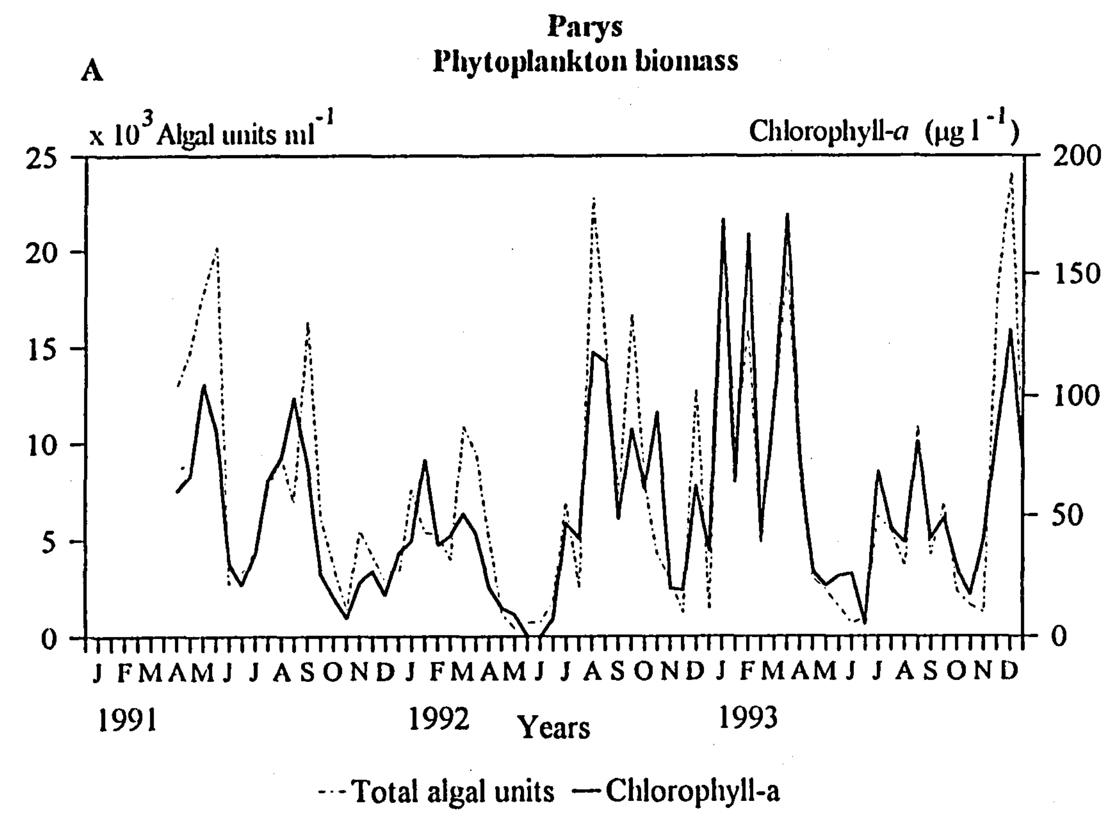
FIGURE 65: Variation in total algal units (x 10^3 ml^{-1}) and chlorophyll-*a* concentration (A), as well as phytoplankton composition (B) in the Vaal River at the Barrage. Cyan = Cyanophyceae, Baci = Bacillariophyceae, Chlo = Chlorophyceae, Cryp = Cryptophyceae, Dino = Dinophyceae, Eugl = Euglenophyceae and Chry = Chrysophyceae.

3.2.2.2 PARYS

Fig. 66B illustrates the phytoplankton composition at Parys during the study period. The phytoplankton was dominated by diatoms and green algae. A comparison of Figs 70 & 66B shows that representatives of the Bacillariophyceae and Chlorophyceae succeeded one another rapidly. From April to November 1991 unicellular centric diatoms and *Chlamydomonas incerta* dominated and succeeded each other alternatively. This alternating succession of centric diatoms and *Chlamydomonas incerta* as the dominants was again observed from July to December 1992 and from June to November 1993 (Figs 70 & 66B). During the first six months of 1992 and 1993 *Melosira granulata* was the dominant species. During the end of 1991 and the beginning of 1992, Cyanophyceae representatives (mainly *Oscillatoria simplicissima*) were present, although they did not reach such high concentrations as the *O. simplicissima* bloom at the Barrage during the same time. Fig. 66B shows that the summer periods of 1992/1993 and 1993/1994 were also characterised by relatively low concentrations of Cyanophyceae representatives (compared to the percentage composition of Cyanophyceae at the Barrage; Fig. 65B). Although all seven algal groups identified were present at the Parys sampling locality, the Cryptophyceae and Euglenophyceae were the only other groups (besides the Bacillariophyceae, Chlorophyceae and Cyanophyceae) present in significant quantities (Fig. 66B).

A positive correspondence between chlorophyll-*a* concentration and total algal units is shown in Fig. 66A. Fig. 66A shows that several periods of high algal concentrations existed. During the end of May/beginning of June 1991, total algal units reached values of approximately 20 000 cells ml^{-1} due to a bloom of centric diatoms (compare Figs 66 A & B). During August 1992, a total algal unit concentration of more than 22 000 cells ml^{-1} was recorded. During this stage *Chlamydomonas incerta* was the dominant species (compare Fig. 70). During January, February and March 1993, chlorophyll-*a* concentrations of more than 170 $\mu g\ l^{-1}$ were recorded. These high chlorophyll-*a* concentrations can be ascribed to blooms of *Melosira granulata* (compare Fig. 70). A bloom of *M. granulata* was again responsible for high algal unit concentrations (almost 25 000 units ml^{-1}) during the beginning of December 1993.

The minimum, maximum and average chlorophyll-*a* concentrations measured during the three years of the study period at the Parys sampling locality, are given in Table 15 and the minimum, maximum and average number of algal units ml^{-1} are given in Table 16.



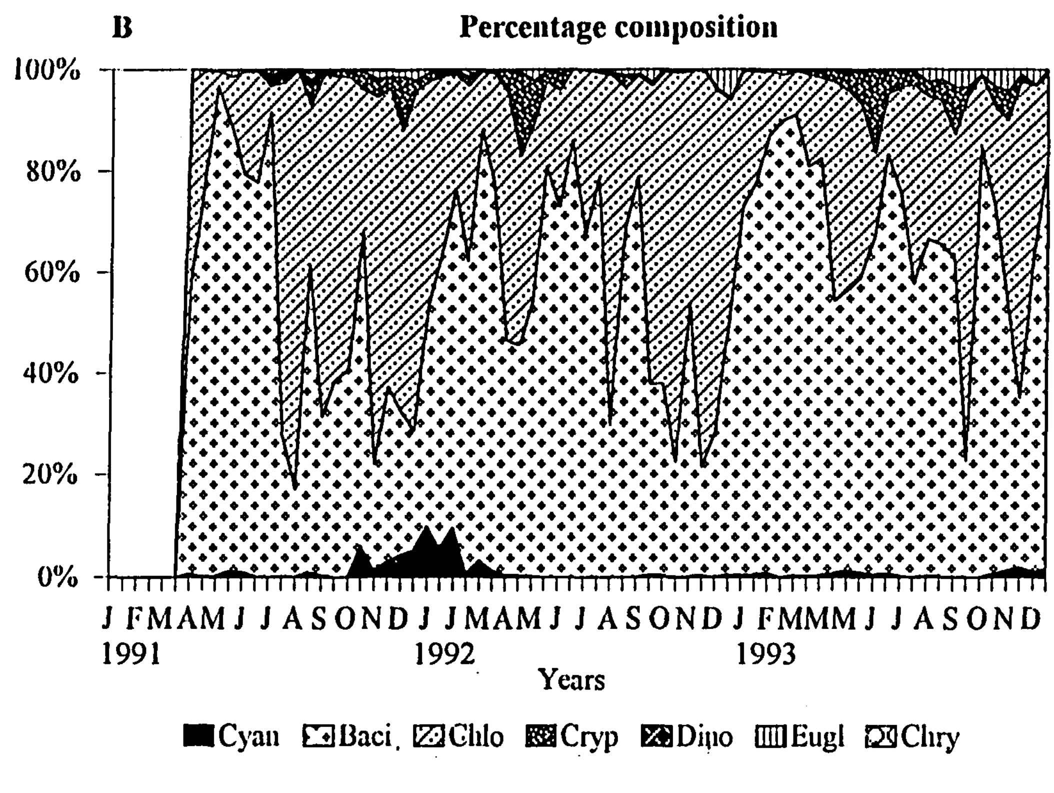


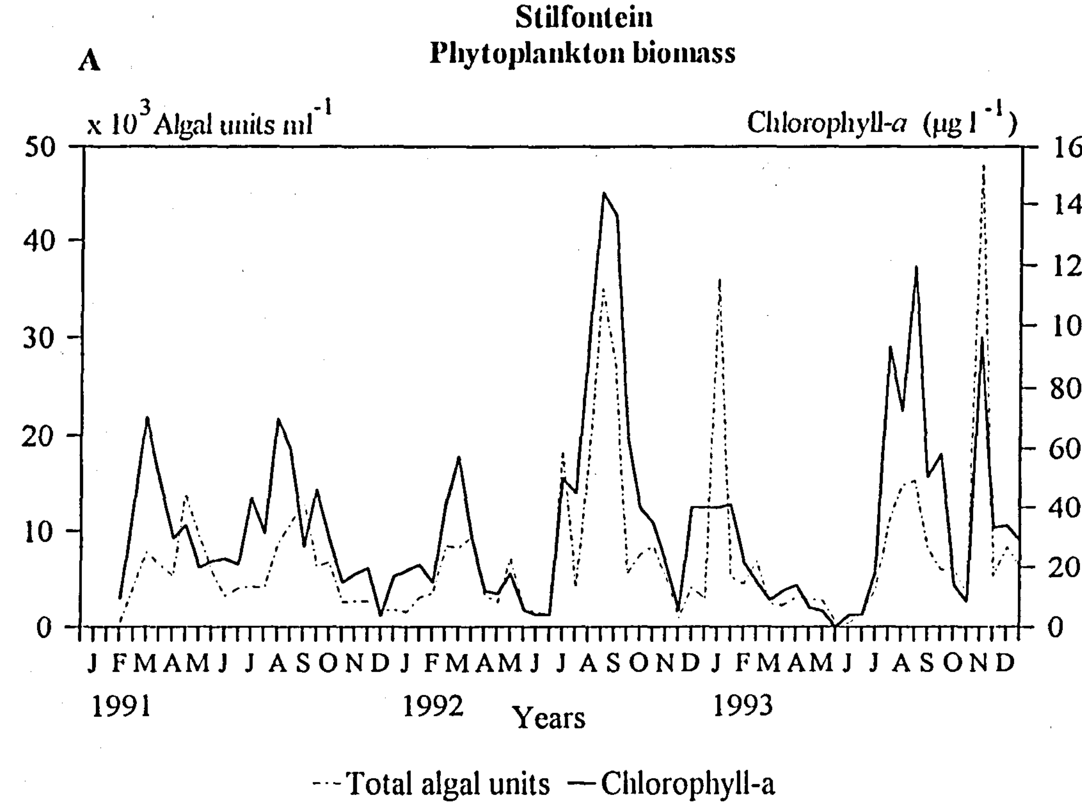
FIGURE 66: Variation in total algal units (x 10^3 ml^{-1}) and chlorophyll-*a* concentration (A), as well as phytoplankton composition (B) in the Vaal River at Parys. Cyan = Cyanophyceae, Baci = Bacillariophyceae, Chlo = Chlorophyceae, Cryp = Cryptophyceae, Dino = Dinophyceae, Eugl = Euglenophyceae and Chry = Chrysophyceae.

3.2.2.3 STILFONTEIN

At Stilfontein five major groups or taxa were present in significant quantities, namely the Cyanophyceae, the Bacillariophyceae, the Chlorophyceae, the Cryptophyceae and the Euglenophyceae (Fig. 67B). Bacillariophyceae and Chlorophyceae were the most dominant groups, but Cryptophyceae and Euglenophyceae representatives were present throughout the study period. During the summer months of 1991 a bloom of Cyanophyceae (*Oscillatoria simplicissima*) occurred in the Vaal River at Stilfontein. Although the total algal units did not reach very high concentrations (± 8 000 units ml^{-1}; Fig 67A), the Cyanophyceae representatives comprised approximately 70% of the phytoplankton population during this period. Comparable to the situation at Parys, centric diatoms and *Chlamydomonas incerta* cells succeeded each other alternatively from April to December 1991 and from April to September 1992 (compare Fig. 71). During 1991, both at Parys and Stilfontein, centric diatoms dominated from April to July, before they were replaced by *Chlamydomonas incerta*. Comparable to the situation at the Barrage, a bloom of Cyanophyceae was observed from October 1992 to the beginning of 1993. It differs from the Cyanophyceae bloom at the Barrage in so far as the bloom at Stilfontein was caused mainly by *Synechococcus cedrorum* cells (compare Fig. 71).

A positive correspondence between total algal units and chlorophyll-*a* concentration at the Stilfontein sampling locality is illustrated in Fig. 67A. Five periods of very high cell or chlorophyll-*a* concentrations occurred, each the result of a bloom of centric diatoms (compare Fig. 71). During the middle of August 1992 total cell numbers of 35 000 ml^{-1} were recorded, accompanied by a chlorophyll-*a* concentration of almost 150 $\mu g\ l^{-1}$. During January 1993 a bloom of very small centric diatoms occurred. Although the centric diatoms reached concentrations of more than 36 000 cells ml^{-1}, the chlorophyll-*a* concentration was relatively low, reaching values of approximately 40 $\mu g\ l^{-1}$. The same tendency could also be seen during November 1993 (algal unit concentration of 48 000 units ml^{-1} accompanied by a chlorophyll-*a* concentration of less than 100 $\mu g\ l^{-1}$). During July and August 1993 high chlorophyll-*a* concentrations occurred as the result of a bloom of centric diatoms.

The minimum, maximum and average chlorophyll-*a* concentrations measured during the three years of the study period at the Stilfontein sampling locality, are given in Table 15. The minimum, maximum and average number of algal units ml^{-1} are given in Table. 16.



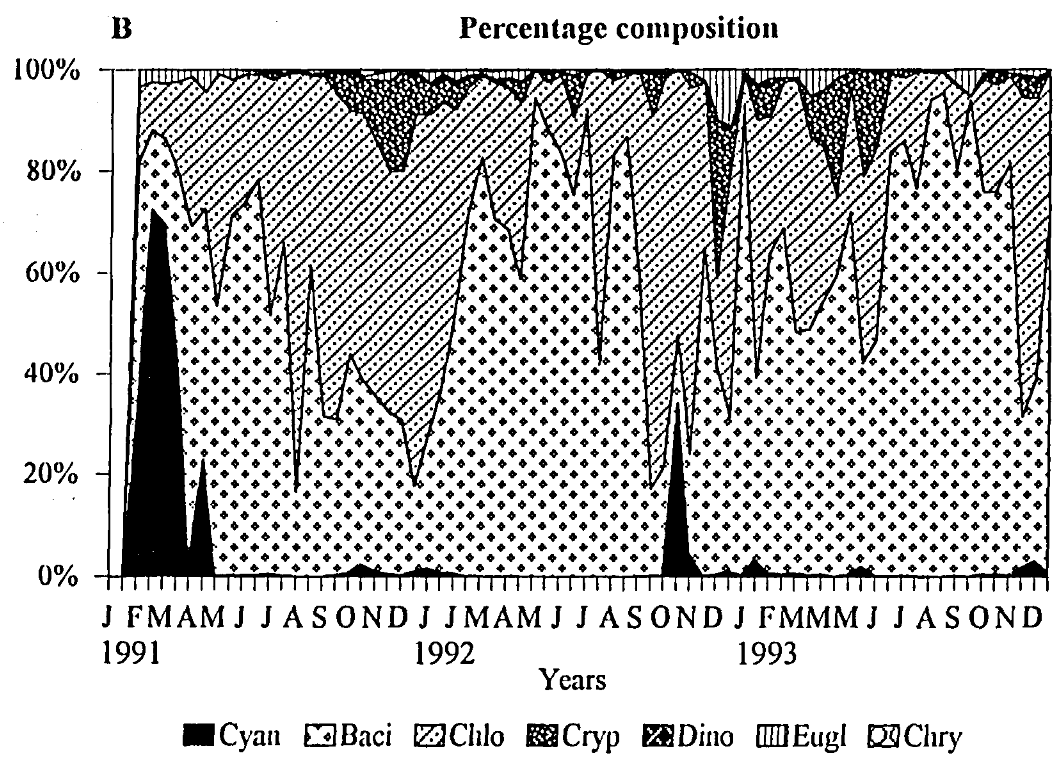


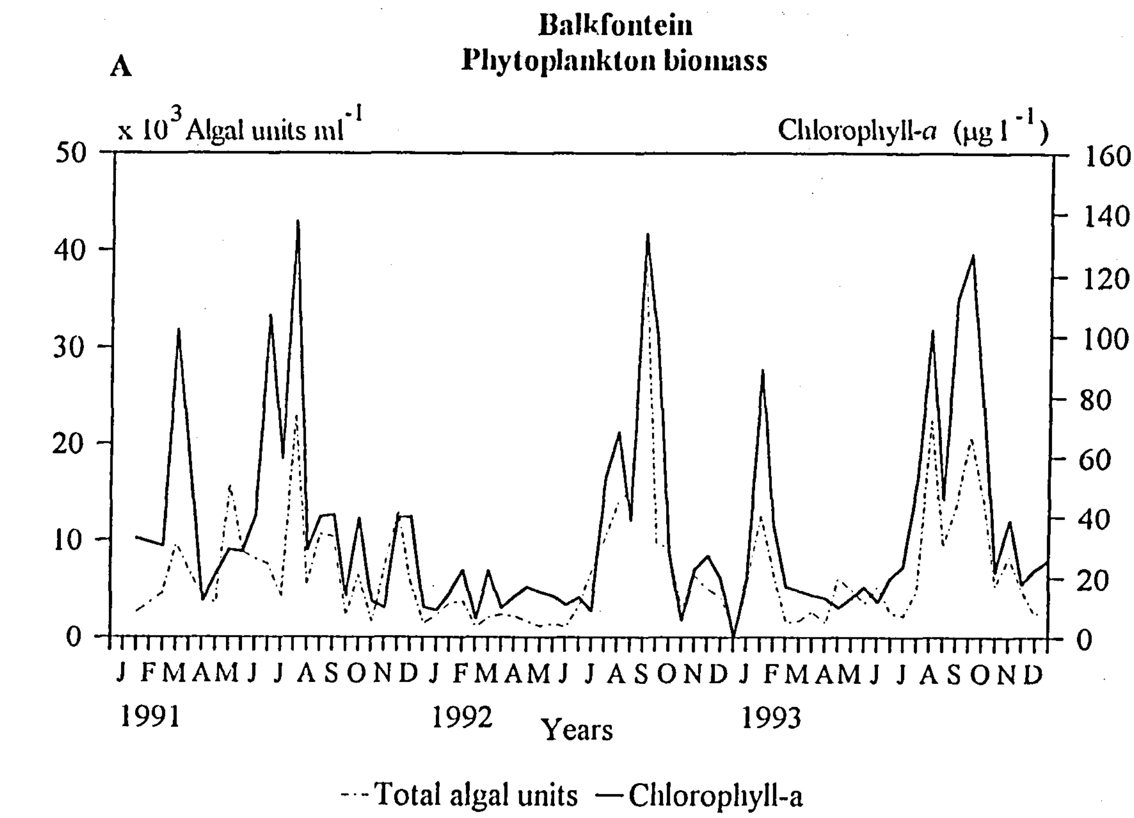
FIGURE 67: Variation in total algal units (x 10^3 ml^{-1}) and chlorophyll-*a* concentration (A), as well as phytoplankton composition (B) in the Vaal River at Stilfontein. Cyan = Cyanophyceae, Baci = Bacillariophyceae, Chlo = Chlorophyceae, Cryp = Cryptophyceae, Dino = Dinophyceae, Eugl = Euglenophyceae and Chry = Chrysophyceae.

3.2.2.4 BALKFONTEIN

Six of the seven algal groups that were found in the study area, occurred in considerable quantities at the Balkfontein sampling locality (Fig. 68B). The Euglenophyceae as well as the Dinophyceae were more dominant in relation to the other phytoplankton groups at Balkfontein than at the other sampling localities. The Balkfontein sampling locality showed a much more rapid succession of dominant species than the other sampling localities (compare Fig. 72 with Figs 69-71). Cyanophyceae representatives (mainly *Oscillatoria simplicissima*; Fig. 72) occurred during the beginning of 1991, comparable with *O. simplicissima* blooms at the Barrage and Stilfontein. The Cyanophyceae bloom was followed by a rapid, alternating succession of centric diatoms and *Chlamydomonas incerta* as the dominants until September 1991, when *Cryptomonas major* reached dominant proportions. During February 1992 a dinoflagellate, *Sphaerodinium* sp. nov. was the dominant species in the Vaal River at Balkfontein. This bloom was followed by an alternating succession of diatom- and green algal species as the dominants until the end of 1993. During the end of 1992 and beginning of 1993, Cyanophyceae representatives were again present, but they did not reach dominant proportions.

Seven periods of chlorophyll-*a* concentrations above 100 $\mu g\ l^{-1}$ were observed during the study period (Fig. 68A). During the middle of March 1991 a chlorophyll-*a* concentration of 102 $\mu g\ l^{-1}$ was recorded. During this time the phytoplankton assemblage was mainly composed of centric diatoms, but Chlorophyceae and a smaller amount of Euglenophyceae representatives were also present (Fig. 68B). Fig. 68A shows that high chlorophyll-*a* concentrations were also recorded during July 1991, due to blooms of *Chlamydomonas incerta* (compare Figs 72 & 68B). The greater part of 1992 was characterised by low chlorophyll-*a* concentrations and cell numbers, but the algal biomass started to increase during July to reach very high concentrations during the beginning of September. A chlorophyll-*a* concentration of 133 $\mu g\ l^{-1}$ was recorded during this time, and the total algal units (comprising mostly centric diatoms, but also Chlorophyceae; Fig. 68B) reached levels of almost 40 000 cells ml^{-1} (Fig. 68A). After this peak period the biomass started to decline. During the beginning of August 1993, a bloom of *Chlamydomonas incerta* occurred, accompanied by Bacillariophyceae and Cryptophyceae representatives. During the middle of September chlorophyll-*a* concentrations of almost 130 $\mu g\ l^{-1}$ could be ascribed to a bloom of unicellular centric diatoms (compare Fig. 72).

The minimum, maximum and average chlorophyll-*a* concentrations measured during the three years of the study period at the Balkfontein sampling locality, are given in Table 15, while the minimum, maximum and average number of algal units ml^{-1} are given in Table 16.



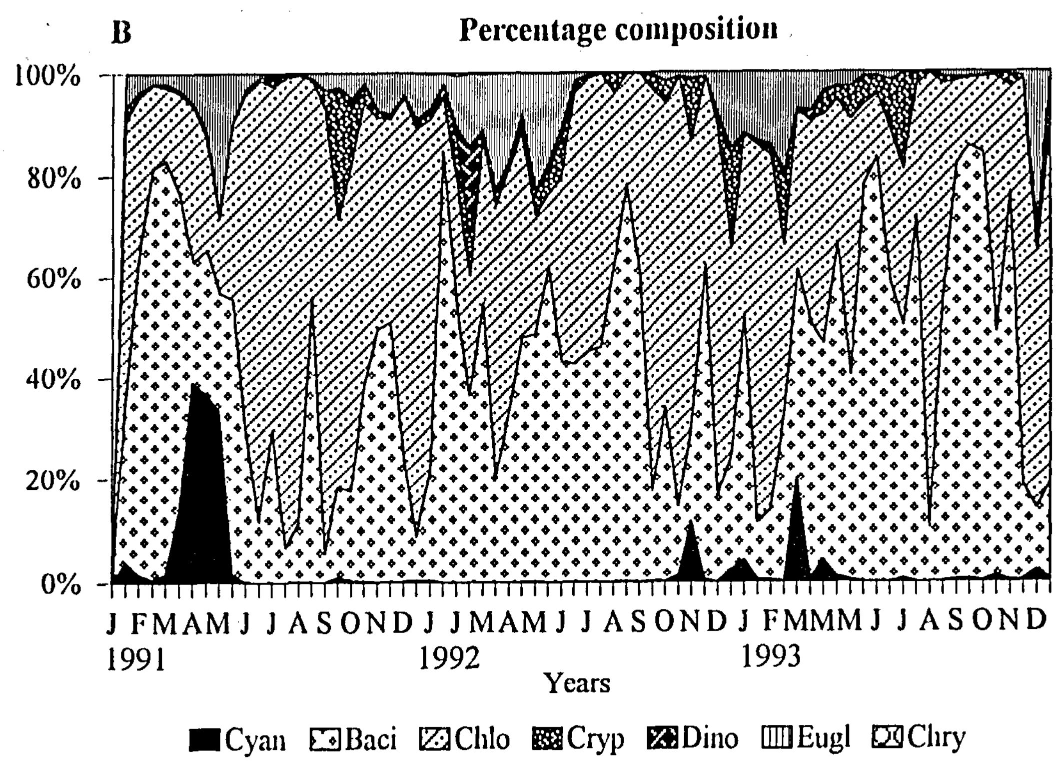


FIGURE 68: Variation in total algal units (x 10^3 ml^{-1}) and chlorophyll-*a* concentration (A), as well as phytoplankton composition (B) in the Vaal River at Balkfontein. Cyan = Cyanophyceae, Baci = Bacillariophyceae, Chlo = Chlorophyceae, Cryp = Cryptophyceae, Dino = Dinophyceae, Eugl = Euglenophyceae and Chry = Chrysophyceae.

TABLE 15: Minimum, maximum and average chlorophyll-a concentrations ($\mu g\ l^{-1}$) recorded in the water of the Barrage, Parys, Stilfontein and Balkfontein sampling localities for the three years of the study period.

	1991	1992	1993
BARRAGE:			
Minimum	1.9	6.3	4.6
Maximum	88	11.5	146
Average	24.1	46.4	33.1
PARYS:			
Minimum	7.64	0	5
Maximum	105	118	175
Average	47.8	45	66.2
STILFONTEIN:			
Minimum	3.7	4	0
Maximum	70	144	119.4
Average	30.5	40.4	33.3
BALKFONTEIN:			
Minimum	10	0	10.1
Maximum	137.2	133	127
Average	41.6	28.2	39.5

TABLE 16: Minimum, maximum and average algal units ml^{-1} recorded in the water of the Barrage, Parys, Stilfontein and Balkfontein sampling localities for the three years of the study period.

	1991	1992	1993
BARRAGE:			
Minimum	16513	694	373
Maximum	306	19526	25776
Average	3281	4225	5593
PARYS:			
Minimum	1369	433	676
Maximum	20192	22780	24175
Average	7971	6437	8282
STILFONTEIN:			
Minimum	465	902	404
Maximum	13949	35043	48041
Average	5791	7879	8558
BALKFONTEIN:			
Minimum	1422	803	1492
Maximum	22980	38921	22559
Average	7526	6266	7025

The complexity and variability of rivers and streams as habitats discourage systematic investigation, and much less is known about blue-green algae (Cyanophyceae) in streams than in lakes (Fogg *et al.,* 1973). Fogg *et al.* (1973) stated that blue-green algae are not usually part of the phytoplankton of small or medium sized-rivers, but they may occur in large ones such as the Nile. In the Nile blue-green algae and diatoms were the dominant groups (Talling, 1976b). It was found in the river Jordan that the Cyanophyceae (represented by *Microcystis* and *Anabaena* species) was the principal contributor to algal biomass, with the diatoms taking up the second place (Pollingher, 1978). As mentioned previously, blue-green algae were relatively abundant in the Vaal River during summer periods (especially at the Barrage, Stilfontein and Balkfontein sampling localities), sometimes reaching dominant proportions. The Cyanophyceae biomass, as well as species diversity, was higher at the Barrage sampling locality than at the other sampling localities (compare Fig. 65B with Figs 66B-68B). At the Barrage sampling locality nine Cyanophyceae species were present, at Parys seven species and at Stilfontein and Balkfontein six species each (Table 13). This could be an indication of more favourable conditions for Cyanophyceae growth prevailing at or upstream from the Barrage sampling locality.

According to Round (1985) the diatoms (Bacillariophyceae) and coccoid green algae (Chlorophyta) tend to be the common organisms in rivers, and of the diatoms the small centric species of the genera *Stephanodiscus* and *Cyclotella* are undoubtedly the most common (e.g., in the River Thames, Lack, 1971; the River Lee, Stour and Severn, Swale, 1969; the upper Mississippi River, Baker & Baker, 1981 and the River Avon, Aykulu, 1978). Dominance of diatoms was also found in the Vaal River (Pieterse & Roos, 1987). Pieterse and Roos showed that the diatoms and chlorophytes were the best represented algal groups for 1985 and 1986. This was also the situation found during the present study from 1991 to 1993. Representatives of the Cryptophyceae, Dinophyceae, Euglenophyceae and Chrysophyceae were present during the study period but seldom reached dominant proportions.

An indication of the quality of a waterbody can be given by the species present in the water. Palmer (1980) stated that the presence of cryptophytes and chrysophytes were indicators of clean, unpolluted water. Representatives of both these groups were found in the Vaal River, but their concentrations remained low, thus showing indications of pollution in the river. According to Palmer (1980) an absence of blue-green algae are also indicative of clean, unpolluted water. This group was abundant in the Vaal River, especially during certain periods, again supporting the fact that the Vaal River shows high levels of pollution.

As mentioned earlier, the Chlorophyceae was the phytoplankton group showing the greatest diversity of species, followed by the Euglenophyceae (Table 12). Studies done by Vermeulen* on the Vaal Dam showed that the Chlorophyceae was also the group showing the largest diversity of species in the Vaal Dam (upstream of the Barrage sampling locality). The resemblance in phytoplankton species composition between the Vaal River and the Vaal Dam confirms that the Vaal Dam is the main supplier of water to the Vaal

*A. Vermeulen, Department of Plant and Soil Sciences, PU for CHE, Potchefstroom: personal communication

River. This resemblance also indicates that conditions in the Vaal Dam have a major influence on conditions in the Vaal River below the dam.

The fact that the Euglenophyceae is relatively abundant in the Vaal River is interesting, because it is known that Euglenophyceae representatives are relatively scarce in other rivers of the world, e.g. rivers Jordan (Pollingher, 1978), Nile (Talling, 1976b) as well as the Tigris and Euphrates (Talling, 1980). A possible explanation for the relatively richness in euglenophytes in the Vaal River may be that the river is polluted with heavy metals and other inorganic substances (suggested by Pieterse, 1986a). Certain Euglenophyceae species (e.g. *Strombomonas* and *Trachelomonas* species) impregnate iron and manganese in their loricas, a type of cell covering (Walne, 1980). The high iron and manganese concentrations in the Vaal River most probably play an important role in the presence and selection of certain Euglenophyceae representatives. If Figs 65B-68B are compared, it seems as if the concentration, and therefore the importance of the Euglenophyceae biomass relative to other groups, increased from the Barrage downstream to Balkfontein. At Balkfontein a fairly large amount of Euglenophyceae biomass occurred throughout the study period. The diversity of the Euglenophyceae also increased downstream (Table 13). More Euglenophyceae species were present at the Balkfontein sampling locality (28 species) than at the upstream sampling localities (e.g. Barrage, 19 species). It is known that excessive manganese concentrations occur in the Klerksdorp-Orkney-Stilfontein area, originating from mines adjacent to the Vaal River (Verhoef, 1981; Bruwer *et al.,* 1985). The higher biomass and diversity of Euglenophyceae at the downstream sampling localities could, therefore, possibly be a result of higher levels of heavy metal and inorganic pollution at and upstream from these localities.

A considerable number of algal species are capable of producing blooms. Some of the genera most frequently involved are *Microcystis, Anabaena* and *Oscillatoria* (blue-green algae); *Chlorella, Ankistrodesmus* and *Chlamydomonas* (green algae); *Synedra* and *Cyclotella* (diatoms) and *Euglena* (euglenophytes; Palmer, 1980). Most of these genera were present in the Vaal River during the study period (Table 12), and if conditions for their growth becomes favourable, they may form blooms. Blooms of algae can cause numerous problems, ranging from aesthetically unacceptable situations to poisoning of animals by certain blue-green algae (e.g. *Microcystis aeruginosa*). It has been the experience in Natal that tastes and odours from blue-green algae occur at relatively low concentrations of chlorophyll-*a* (5-10 $\mu g\ l^{-1}$; DWA, 1993a). Algal species present in the Vaal River (Table 12) which could be responsible for taste and odour problems include *Anabaena circinalis, Oscillatoria simplicissima, Microcystis aeruginosa, Pandorina morum* and certain Cryptophyceae species (compare results of Wnorowski, 1992).

The determination of chlorophyll-*a* concentration represents a general method for the estimation of phytoplankton biomass (Vollenweider, 1974). The phytoplankton biomass (chlorophyll-*a* concentration) in the middle Vaal River has increased significantly over the last 20 years (DWA, 1986). In 1973, 92% of samples from the Vaal Barrage had chlorophyll concentration levels below 5 $\mu g\ l^{-1}$ (DWA, 1986). During the present study only 14% of the samples taken at this sampling locality had chlorophyll-*a* levels below 5 $\mu g\ l^{-1}$. By 1982, 87% of samples had chlorophyll concentration levels exceeding 15 $\mu g\ l^{-1}$, while 34% of samples exceeded 35 $\mu g\ l^{-1}$ (DWA, 1986). If this is compared to the recent situation where 61% of the samples showed chlorophyll concentrations exceeding 15 µg l^{-1} and 39% exceeding 35 µg l^{-1}, it seems as if a further increase in chlorophyll-*a* concentration occurred during the recent nine to eleven years. Average chlorophyll-*a* concentrations of 35.5, 53.5, 34.9 and 36.2 µg l^{-1} were recorded at the Barrage, Parys, Stilfontein and Balkfontein sampling localities between 1991 and 1993 (see Table 15 for ranges of each year). Guidelines by the DWA (1993b) for water used for recreational use indicates that at a mean chlorophyll-*a* concentration of more than 30 µg l^{-1} (as the case was in the Vaal River during the present study), severe algal blooms (scums) as well as other symptoms of eutrophication occur and contact recreational users may experience skin irritation from water contact and gastroenteritis if algal-laden water is ingested (especially if blue-green algae are dominant). Annual averages in chlorophyll-*a* concentrations are given in Table 15 and Fig. 81. Fig. 81 indicates that the highest average chlorophyll-*a* concentration during each year was present at the Parys sampling locality. Highest DIN and TN concentrations were also present at the Parys sampling locality (compare Figs 33 & 47 respectively). High chlorophyll-a concentrations at Parys were probably the result of high available nitrogen concentrations present in the river at and upstream from this sampling locality.

The chlorophyll-*a* concentrations in the Vaal River are high in comparison with other river systems such as rivers Meuse (0.2 to 120 µg l^{-1}; Descy *et al.*, 1987), Ogeechee (0.05 to 1.19 µg l^{-1}; Edwards & Meyer, 1987), Thames (1.2 to 219 µg l^{-1}; Kowalczewski & Lack, 1971), Kennet (0.6 to 38.2 µg l^{-1}; Kowalczewski & Lack, 1971) and Lot (0.1 to 45; Décamps *et al.,* 1979, 1983) . If Figs 65A to 68A are compared, it seems as if periods of maximum chlorophyll-*a* concentration in the Vaal River occurred from January to March and again from July to November of each year. In the Vaal River at all four the sampling localities the observed chlorophyll-*a* concentration ranges (see Table 15) fall within the range of eutrophic systems (Wetzel, 1983) confirming observations made by Roos (1992). The DWA (1993a) stated that chlorophyll-*a* concentrations in raw water vary from less than 1 µg l^{-1} in clear waters to over 50 µg l^{-1} in severe nuisance conditions.

Fig. 67A shows periods where high algal unit concentrations were accompanied by low chlorophyll-*a* concentrations (e.g. January and November 1993). Although the biomass reached concentrations of more than 36 000 cells ml^{-1} during January and more than 48 000 cells ml^{-1} during November (dominated by centric diatoms), the chlorophyll-*a* concentrations was relatively low, reaching values of less than 50 µg l^{-1} (during January 1993) and less than 100 µg l^{-1} (during November 1993). Low chlorophyll-*a* concentrations, accompanied by high cell numbers, can be explained as follows. Diatom cells contain less chlorophyll-*a* per cell unit volume than for example *Chlamydomonas* cells (unpublished information). This is the reason why diatom blooms of the same cell concentration as, for example, green algal blooms, show lower chlorophyll-*a* concentrations. Another reason is that the blooms of centric diatoms at Stilfontein during January and November 1993 consisted of very small diatom cells. Because the cells were so small, a significant number of them are necessary to reach the same chlorophyll-*a* concentration than less, but larger, centric diatom cells. This seems to be of particular importance, because blooms of larger centric diatoms also occurred (e.g. August 1992, July and August 1993; Fig. 67A), accompanied by high chlorophyll-*a* concentrations.

3.2.3 SUCCESSION OF ALGAL SPECIES

The succession of the dominant phytoplankton species at the different sampling localities is shown in Figs 69-72. Figs 69-72 show that the Bacillariophyceae (especially the unicellular centric diatoms and *Melosira granulata*) and Chlorophyceae (especially *Chlamydomonas incerta*), were the main phytoplankton groups and species which dominated and succeeded each other (compare Table 14).

If the phytoplankton composition at all four sampling localities in the Vaal River (Figs 65B to 68B) is viewed superficially, it seems as if the diatoms tended to dominate from January to August of each year, while the green algae were dominant from September to December. Successional patterns of phytoplankton species where maximum concentrations occurred in winter and spring, dominated by diatoms and followed by dominance of green algae in the autumn and summer periods, were found in many rivers of the world (Lack, 1971; Egborge, 1974; Baker & Baker, 1981,; Descy *et al.*, 1987). Descy *et al.* (1987) indicated that the succession of dominant algal groups in the River Meuse is small cold-water diatoms in spring and autumn *versus* large warm-water diatoms and green algae in summer. Exactly the same tendency was found during the present study on the Vaal River. Figs 73 to 76 show the succession of unicellular centric diatoms (usually small species) and *Melosira granulata* (a large, filamentous centric diatom) at the four sampling localities respectively. It is clear that dominance of diatoms during the summer periods (January to April) could be ascribed to high concentrations of *Melosira granulata*. Concentrations of unicellular centric diatoms were usually low during the summer periods, but they often dominated during the cold-water winter periods (Figs 73-76). At the Barrage, Parys and Stilfontein sampling localities (Figs 65B-67B) dominance of diatoms during December 1993 was the result of blooms of *Melosira granulata* (compare Figs 73-75 with Figs 69-71). It therefore seems as if *M. granulata* prefers warmer water temperatures than the unicellular centric diatoms.

Simultaneous development and dominance of algal species occurred at the different sampling localities during the present study. The downstream sampling localities (Stilfontein and Balkfontein), as well as the upstream sampling localities (Barrage and Parys) showed similarities regarding the succession of dominant species. On 16 September 1992 centric diatoms at both the downstream sampling localities were replaced by *Oocystis lacustris* as the dominant species (Figs 71 & 72). During the same period centric diatoms dominated at both the upstream sampling localities (Barrage and Parys), but were replaced by *Dictyosphaerium elegans* as the dominant species on 21 October 1992 (Figs 69 & 70). *Chlamydomonas incerta* as well as representatives of centric diatoms frequently dominated during the same periods at different sampling localities (e.g. 5 August 1992 and 1-15 December 1993 at the Barrage and Parys; compare Figs 69 & 70). Simultaneous development and dominance of algae at different sampling localities were also demonstrated by Lack *et al.* (1978) along a whole section (21 km) of the River Thames.

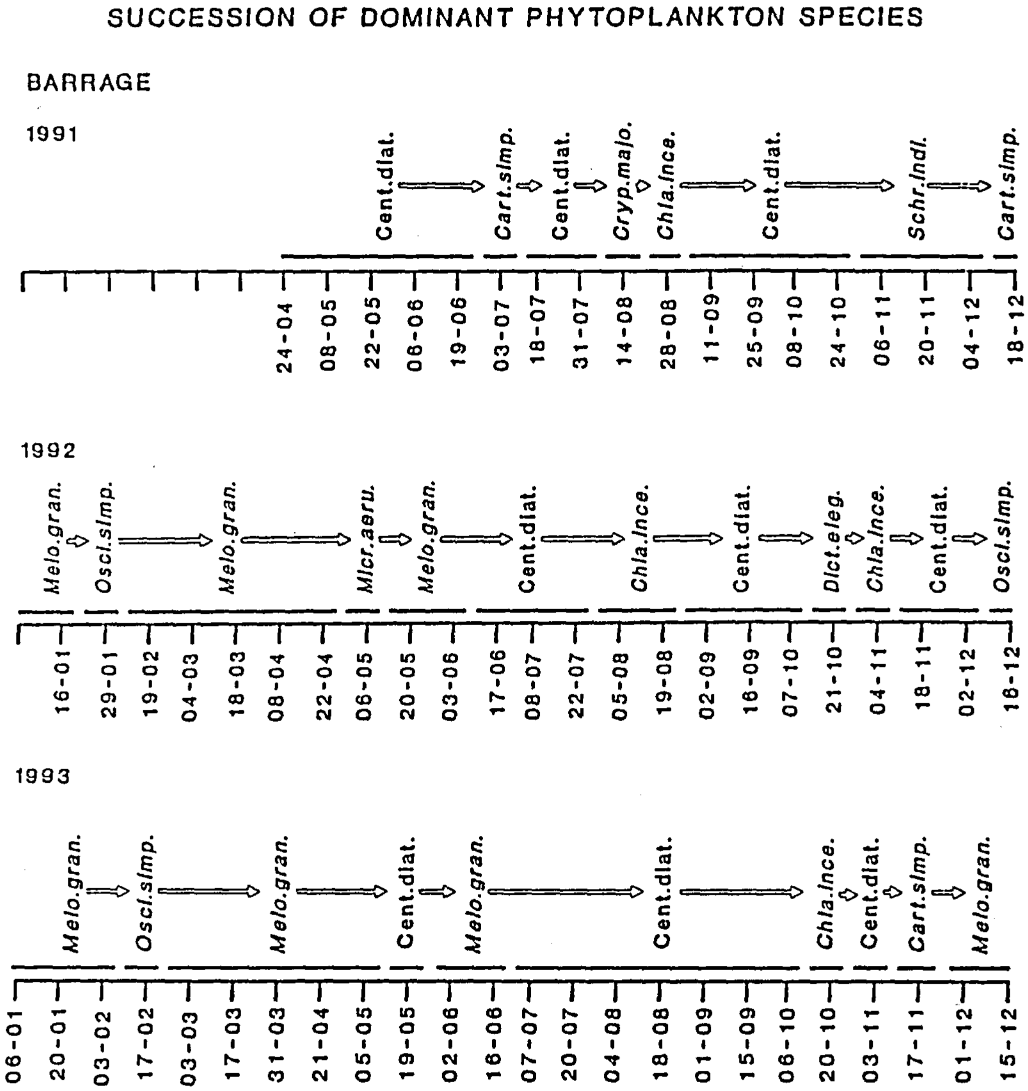


FIGURE 69: Succession of dominant phytoplankton species in the Vaal River at the Barrage. *Cart.simp.* = *Carteria simplicissima*, Cent.diat. = Unicellular centric diatoms, *Chla.ince.* = *Chlamydomonas incerta*, *Cryp.majo.* = *Cryptomonas ?major*, *Dict.eleg.* = *Dictyosphaerium elegans*, *Melo.gran.* = *Melosira granulata*, *Micr.aeru.* = *Microcystis aeruginosa*, *Osci.simp.* = *Oscillatoria simplicissima*, *Schr.indi.* = *Schroederia indica*.

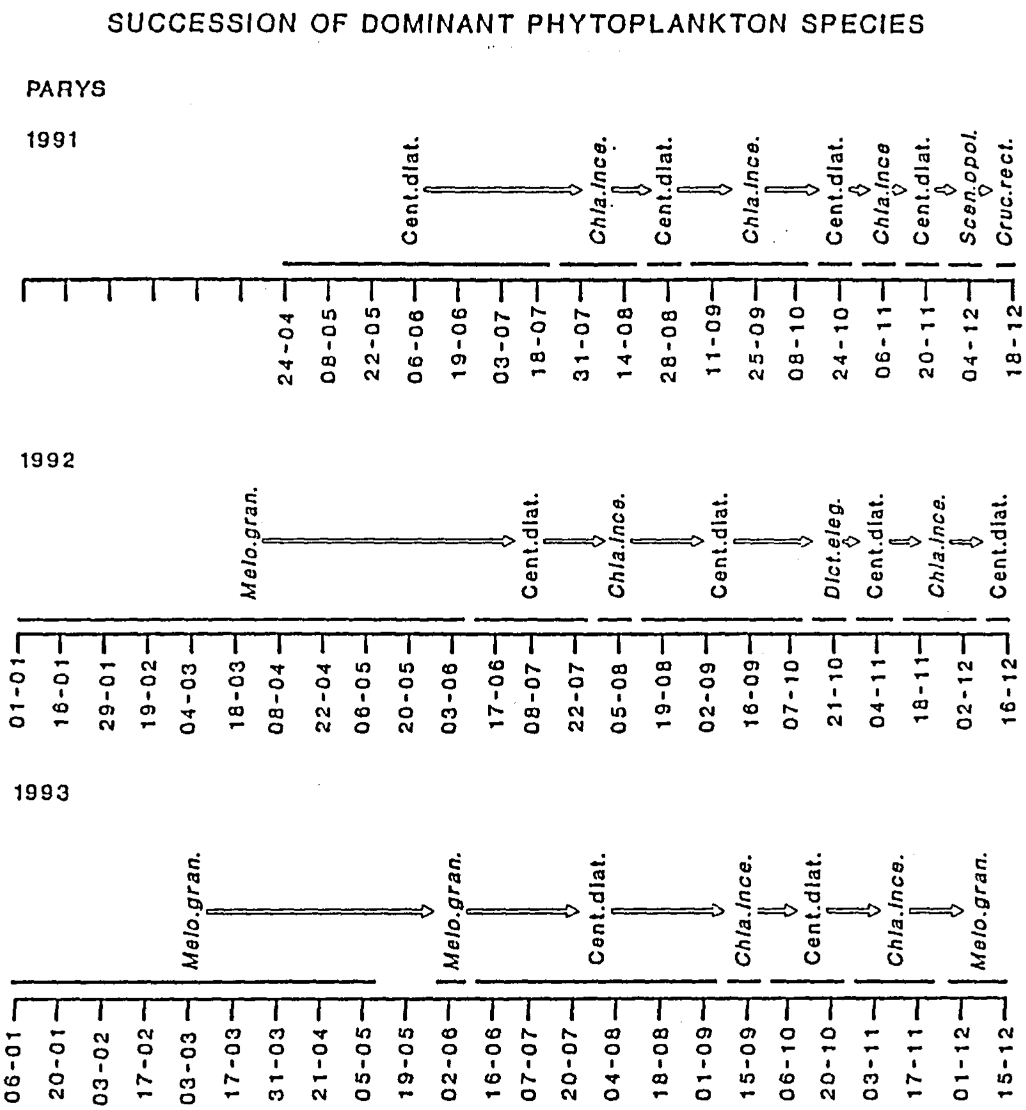


FIGURE 70: Succession of dominant phytoplankton species in the Vaal River at Parys. Cent.diat. = Unicellular centric diatoms, *Chla.ince.* = *Chlamydomonas incerta*, *Cruc.rect.* = *Crucigeniella rectangularis*, *Cryp.majo.* = *Cryptomonas ?major*, *Dict.eleg.* = *Dictyosphaerium elegans*, *Melo.gran.* = *Melosira granulata*, *Scen.opol.* = *Scenedesmus opoliensis*.

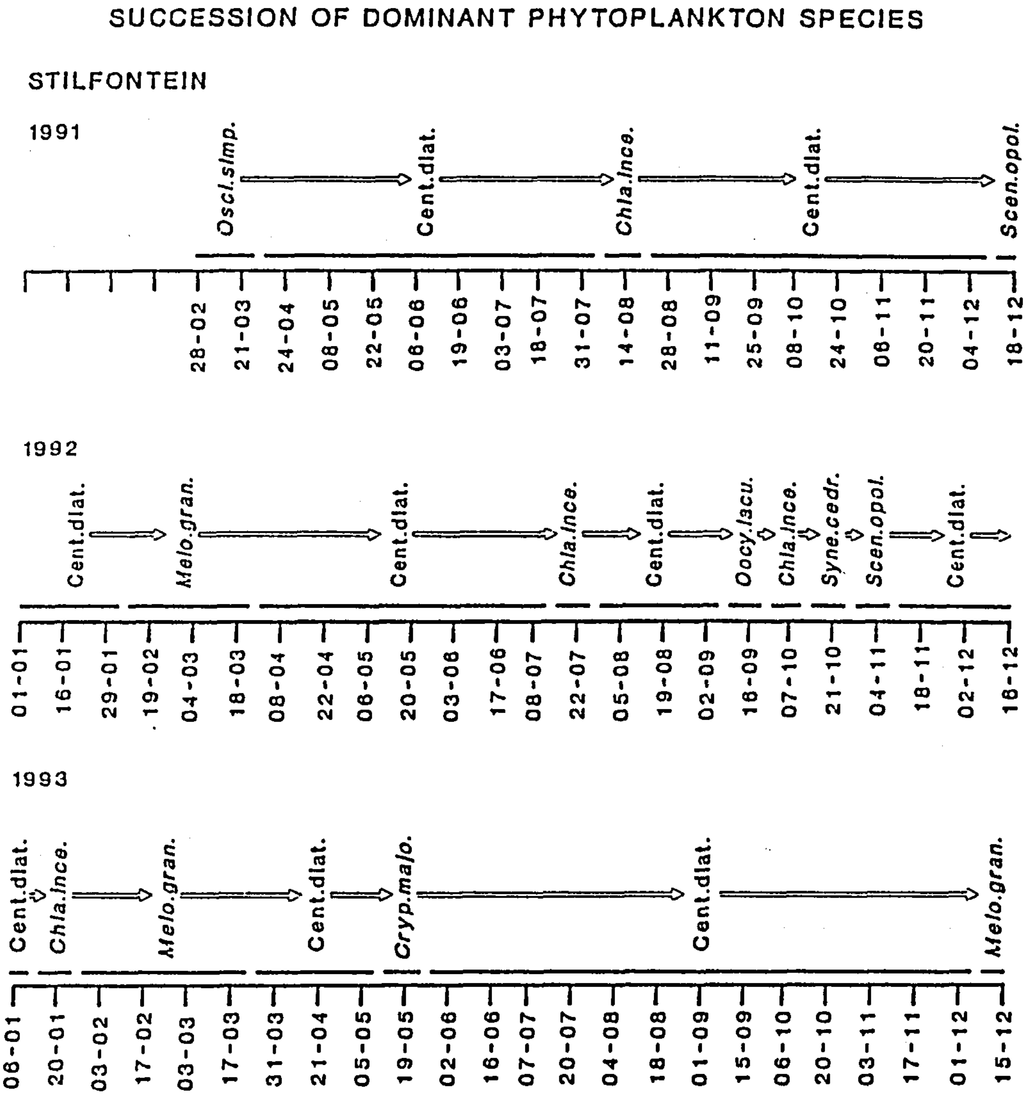


FIGURE 71: Succession of dominant phytoplankton species in the Vaal River at Stilfontein. Cent.diat. = Unicellular centric diatoms, *Chla.ince.* = *Chlamydomonas incerta*, *Melo.gran.* = *Melosira granulata*, *Oocy.lacu.* = *Oocystis lacustris*, *Osci.simp.* = *Oscillatoria simplicissima*, *Scen.opol.* = *Scenedesmus opoliensis*, *Syne.cedr.* = *Synechococcus cedrorum*.

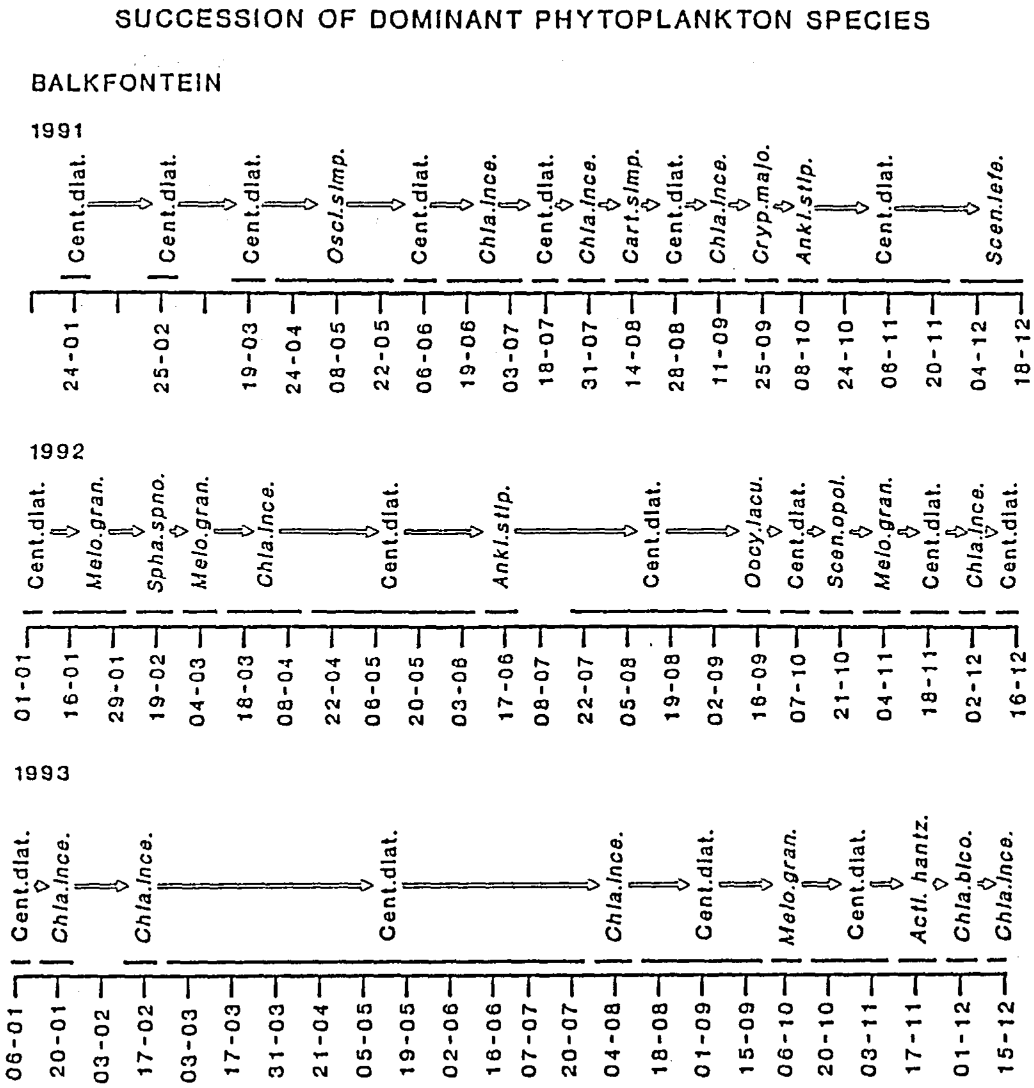


FIGURE 72: Succession of dominant phytoplankton species in the Vaal River at Balkfontein. *Acti.hant.* = *Actinastrum hantzschii*, *Anki.stip.* = *Ankistrodesmus stipitatus*, *Cart.simp.* = *Carteria simplicissima*, Cent.diat. = Unicellular centric diatoms, *Chla.bico.* = *Chlamydomonas bicocca*, *Chla.ince.* = *Chlamydomonas incerta*, *Cryp.majo.* = *Cryptomonas ?major*, *Melo.gran.* = *Melosira granulata*, *Oocy.lacu.* = *Oocystis lacustris*, *Osci.simp.* = *Oscillatoria simplicissima*, *Scen.lefe.* = *Scenedesmus lefevrii*, *Scen.opol.* = *Scenedesmus opoliensis*, *Spha.spno.* = *Sphaerodinium* sp. nov.

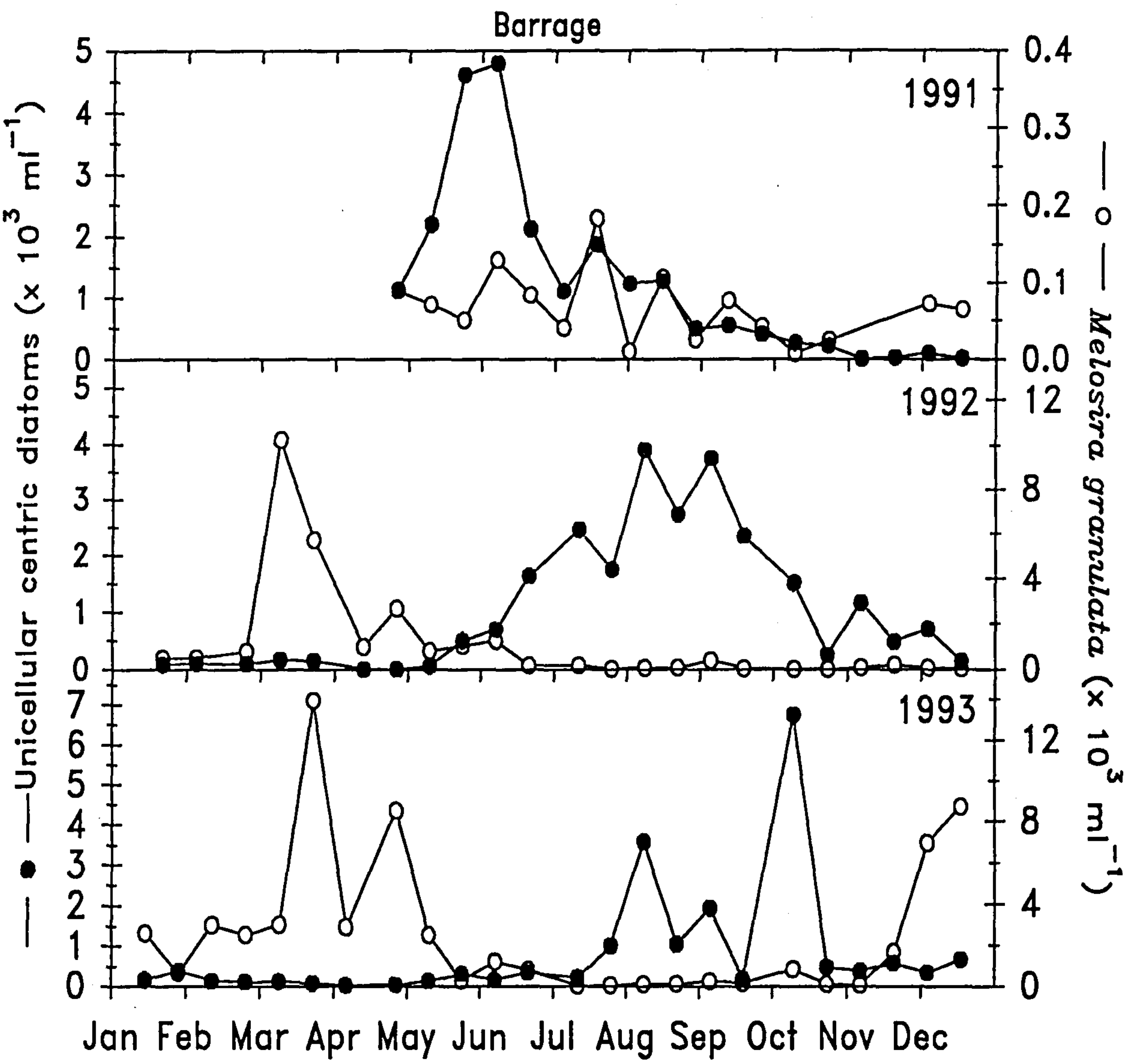


FIGURE 73: Variation in the concentration of unicellular centric diatoms and *Melosira granulata* (x 10^3 ml^{-1}) at the Barrage during the study period.

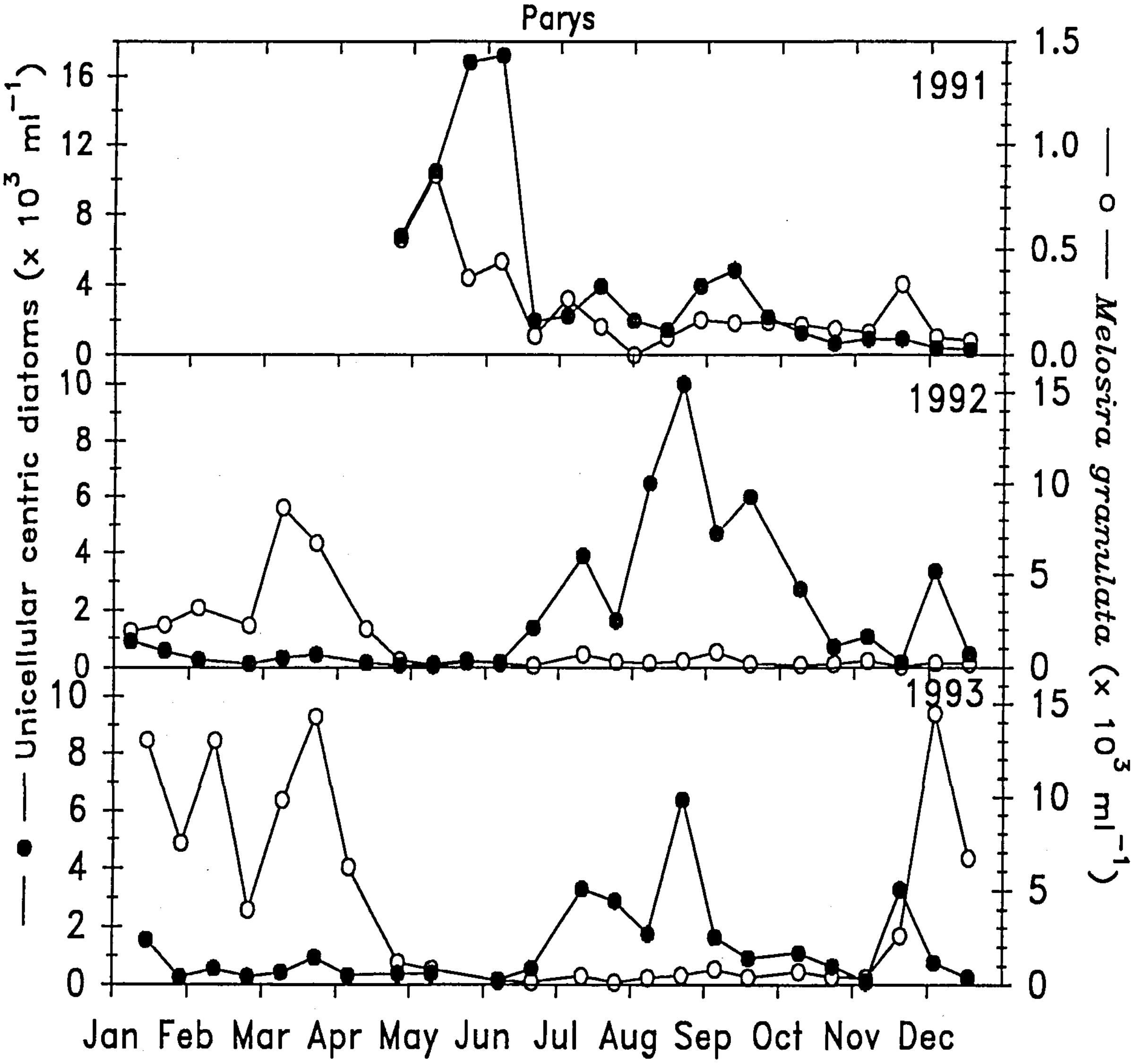


FIGURE 74: Variation in the concentration of unicellular centric diatoms and *Melosira granulata* (x 10^3 ml^{-1}) at Parys during the study period.

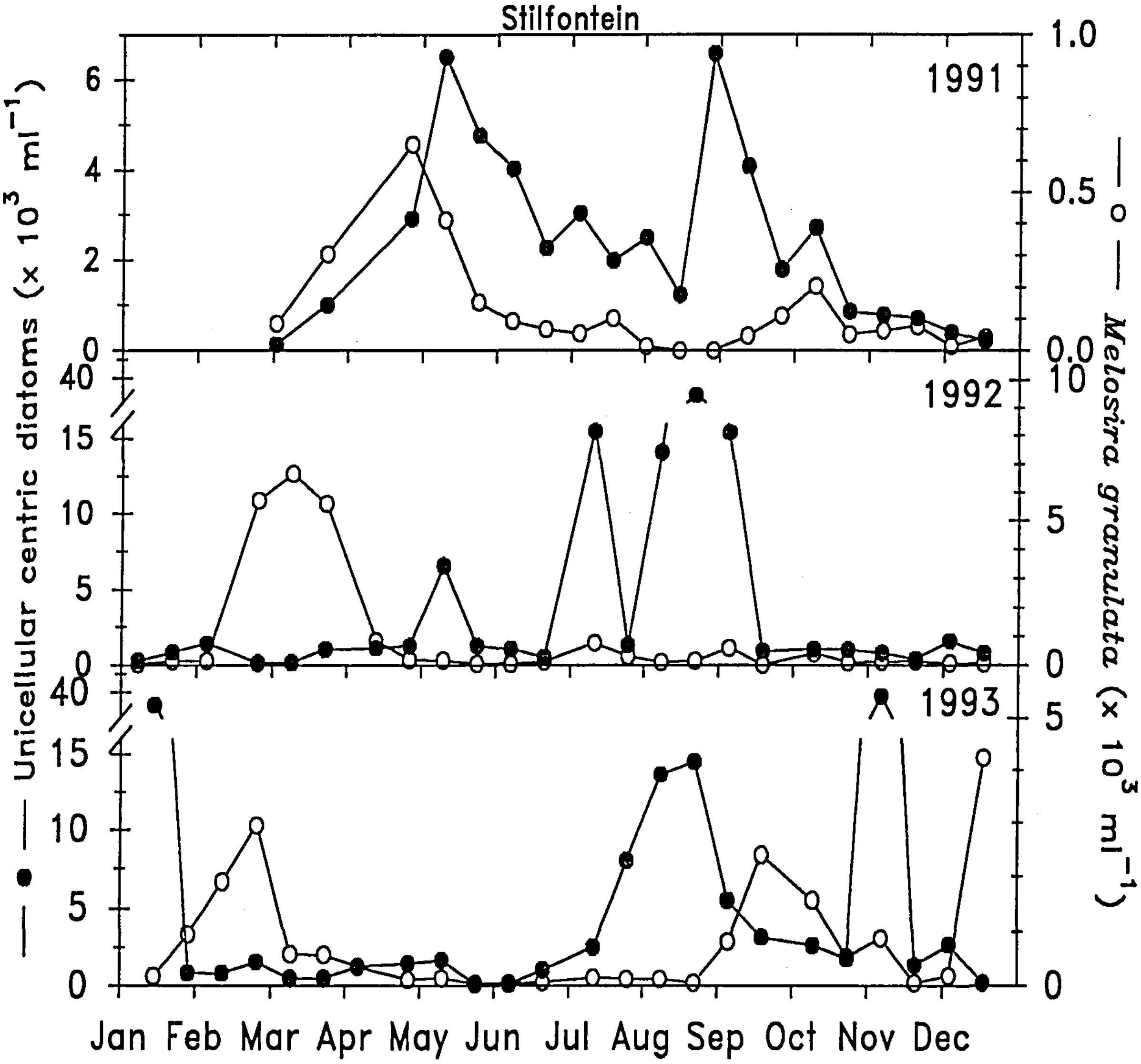


FIGURE 75: Variation in the concentration of unicellular centric diatoms and *Melosira granulata* (x 10^3 ml^{-1}) at Stilfontein during the study period.

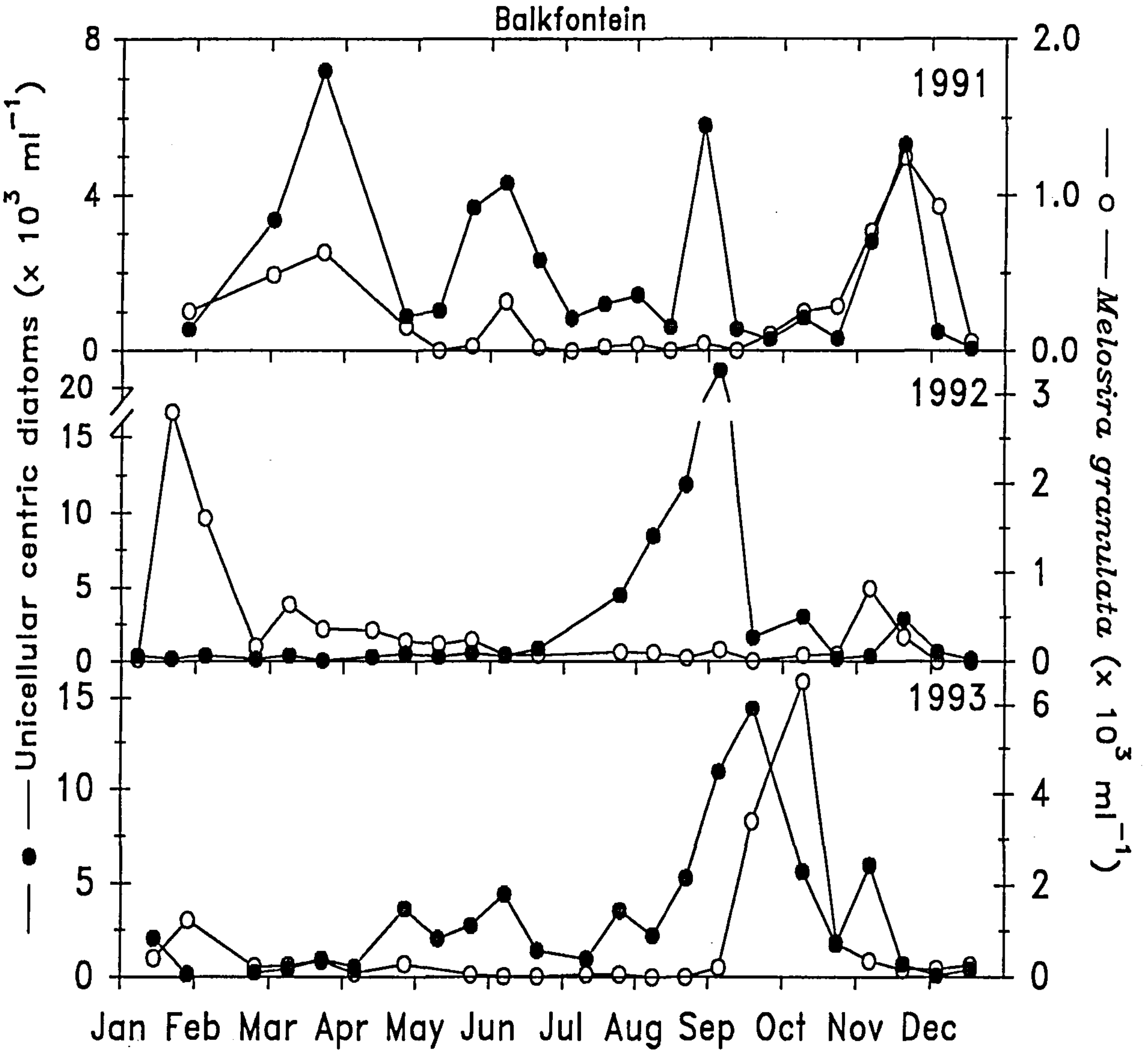


FIGURE 76: Variation in the concentration of unicellular centric diatoms and *Melosira granulata* (x 10^3 ml^{-1}) at Balkfontein during the study period.

Resemblances regarding the succession of dominant species could be seen at the different sampling localities. At three of the sampling localities (Barrage, Parys, and Stilfontein; Figs 69-71) centric diatoms dominated from 24 April 1991, while *Oscillatoria simplicissima* was dominant at Balkfontein during the same time. *Oscillatoria simplicissima* was also dominant at Stilfontein just prior to the centric diatoms (28 February 1991 to 21 March 1991). It is possible that *Oscillatoria simplicissima* was drifting downstream from Stilfontein and reached Balkfontein during April, because *Oscillatoria simplicissima* was also replaced by centric diatoms at Balkfontein. At three of the sampling localities, namely Parys, Stilfontein and Balkfontein, the centric diatoms were replaced by *Chlamydomonas incerta* as the dominant species (Figs 70-72). There are also a correspondence between the succession of dominant species at Parys and Stilfontein. *Scenedesmus opoliensis* was the dominant species at Parys on 4 December 1991 (Fig. 70). It seems as if *S. opoliensis* drifted downstream from Parys to reach Stilfontein on 18 December 1991 where it also reached dominance (Fig. 71).

There are also other indications that phytoplankton communities are moving downstream in specific stretches of water. Fig. 65B shows a cryptophyte bloom at the Barrage on 14 August 1991. At Parys (Fig. 66B) a small peak caused by cryptophytes occurred during the end of August 1991. Although cryptophytes were present at Stilfontein on 11 September 1991, the percentage composition reached by cryptophytes increased during November (Fig. 67B). At Balkfontein a peak of Cryptophyceae representatives occurred on 25 September 1991 (Fig. 68B). During the beginning of December 1992 a peak of cryptophytes developed at Stilfontein (Fig. 67B). Two peaks of cryptophytes were observed at Balkfontein (Fig. 68B), namely on 16 December 1992 and on 17 February 1993. It is possible that the Cryptophyceae peaks at the different sampling localities were the same cryptophyte population that drifted or moved downstream from the Barrage to Balkfontein. Unfortunately the retention time of the river between the Barrage and Balkfontein is not known so that it is not possible to know whether it was in fact the same population reaching the different localities as the water moved downstream.

On occasion apparent upstream development of algal assemblages took place. On 21 October 1992 *Scenedesmus opoliensis* was dominant at Balkfontein (Fig. 72). At Stilfontein *S. opoliensis* was dominant on the 4th of November 1992 (Fig. 71). During August 1993 *Chlamydomonas incerta* was the dominant species at the Balkfontein sampling locality (Fig. 72). At the Parys sampling locality, however, *C. incerta* dominated during September 1993 (Fig. 70) and at the Barrage during October 1993 (Fig. 69). If Figs 82 to 85 are compared, it seems as if peaks of diatoms occurred earlier at the Balkfontein sampling locality than at the Barrage. Fig. 82 shows that a peak in diatom concentration occurred during the end of March and the beginning of April 1993 at the Barrage. At the Parys sampling locality (Fig. 83) peaks of diatoms were observed during January, February and March 1993. At the Stilfontein sampling locality (Fig. 84) the diatom peak was observed during January 1993, while at the Balkfontein sampling locality (Fig. 85) a peak of diatoms occurred during September 1992. The tendency of upstream development could also be seen during the end of 1993, where a diatom peak could be observed during the end of December 1993 at the Barrage (Fig. 82), the middle of December 1993 at Parys (Fig. 83), the middle of November 1993 at the Stilfontein sampling locality (Fig. 84) and the end of September 1993 at the Balkfontein sampling locality (Fig. 85). This apparent upstream movement of algal species may be ascribed to an upstream shift in, or development of, environmental conditions favourable for growth.

3.2.4 PHYTOPLANKTON AND ENVIRONMENTAL VARIABLES

3.2.4.1 PHYTOPLANKTON, DISCHARGE, TURBIDITY AND LIGHT

Discharge is usually the main component influencing turbidity in the water. Turbidity, in turn, influences the underwater light conditions in the water. Discharge and water velocity have been proved to be important variables influencing phytoplankton (Lack *et al.*, 1978; Petts, 1984). The relationship between phytoplankton and discharge, which is the main diagnostic feature of rivers when compared to lentic habitats, seems not yet fully understood (Reynolds, 1984). Descy *et al.* (1987) pointed out that only general correlations between phytoplankton concentration (chlorophyll-a) and flow rate have been described.

Rai & Hill (1984) stated that phytoplankton production apparently does not reach its potential until flow is slowed or stopped.

From July 1993 an increase in discharge (Fig. 3), accompanied by an increase in turbidity, occurred (Fig. 7) at the Barrage sampling locality. The increase in turbidity is also reflected in the low levels of light penetration from July to October 1993 (Fig. 1C). The low underwater light climate (together with relative low DIP concentrations) could be responsible for relatively low algal biomass during this period (Fig. 77). From November 1993 onwards deeper light penetration occurred (Fig. 1C), most probably because of decreasing flow rates (Fig. 3). The lower flow rates were responsible for the settling of suspended material and therefore a decrease in turbidity (Fig. 7). More favourable underwater light conditions prevailing, together with relatively high nutrient concentrations, caused a bloom of phytoplankton at the Barrage during the middle of November 1993 (mainly *Carteria simplicissima*; compare Fig. 69) with chlorophyll-*a* concentrations reaching almost 150 $\mu g\ l^{-1}$ and total algal units reaching approximately 26 000 cells ml^{-1} (Fig. 77).

At the Stilfontein sampling locality, a sharp increase in discharge occurred during March 1991 (Fig. 5). The increase in silt-load caused higher turbidities during this period (Fig. 8), usually accompanied by reduced light penetration and thus underwater light climate. Dissolved inorganic nitrogen (DIN; Fig. 31) and especially dissolved inorganic phosphorus (DIP; Fig. 26) were increased to high concentrations, resulting in low DIN:DIP ratios (Fig. 36). High DIN and DIP concentrations usually serves as an important source of nutrients that stimulates algal growth. Silica concentrations were also high, because of the high silt load as well as high water temperatures (Fig 54). Thus, as far as the nutrients were concerned, potentially favourable conditions for algal growth were created during the high flow conditions. Although nutrients were present in high concentrations in the Vaal River at Stilfontein during this period, the phytoplankton could not attain maximum biomass (biomass did increase to chlorophyll-*a* levels of ± 70 $\mu g\ l^{-1}$; Fig. 79), because light penetration was restricted by the turbidity of the water (Fig. 8). The weeks following these high flow conditions were characterised by low discharge (Fig. 5), and an increase in water transparency (Fig. 8). It can be expected that more favourable underwater light conditions could result in increases in the algal concentration, but algal growth was probably suppressed by low DIN (Fig. 31) and especially DIP (Fig. 26) concentrations in the river. During the end of November and beginning of December 1992 at the Stilfontein sampling locality, an increase in discharge (Fig. 5) caused an increase in turbidity (Fig. 8), accompanied by relatively high DIN, DIP and SiO_2-Si concentrations (Figs 31, 26 and 54 respectively). A comparison of Figs 8 and 79 show that the period of high turbidity (low underwater light conditions) was characterised by extremely low algal biomass. As soon as the flow rate and turbidity (Figs 5 and 8) decreased, algal biomass (small unicellular centric diatoms; compare Fig. 71) increased to reach a chlorophyll-*a* concentration of 110 $\mu g\ l^{-1}$ (Fig. 79). The same tendency could also be seen at the Balkfontein sampling locality during the end of November and beginning of December 1992 (compare Figs 6, 9 and 80 for discharge, turbidity and algal biomass and Figs 32, 27 and 55 for DIN, DIP, and SiO_2-Si concentrations). In an earlier study done by Roos (1992), major blooms by *Micractinium, Stephanodiscus, Carteria* and *Chlamydomonas* species have been reported between June and September. Apparently these blooms were also caused by a decrease in flow and clearer water conditions due to reduced suspended material and high nutrient concentrations especially nitrogen, phosphorus and silicate.

Roeder (1977) demonstrated an inverse correlation between rate of flow and the concentration of centric diatoms in a central Iowa stream; diatoms were common only when the flow rate was less than 1.7 $m^3 s^{-1}$. In spite of the fact that flow rates as low as 1.7 $m^3 s^{-1}$ were seldom recorded in the Vaal River, diatoms were frequently the dominant component of the phytoplankton assemblage. It must, however, be remembered that the Vaal River is bigger and therefore it seems as if it is more important to look at the current velocity ($m s^{-1}$) than at the discharge (measured in $m^3 s^{-1}$).

The turbulent nature of lotic systems could also be an important variable that influences the occurrence of blue-green algae. Krüger (1978) observed that an artificial increase in turbulence leads to a increased viability of *Microcystis* species. This may have ecological implications as far as the onset of *Microcystis* sp. blooms are concerned, because of the mixing phenomenon of lotic systems. However, Steinberg and Hartmann (1988) showed that if the turbulence of the water-column is high (mixing depth much greater than euphotic depth), or if the mixing pattern is irregular, as in slow flowing or regulated rivers, blue-green algae are out-competed. *Microcystis aeruginosa* was dominant only once during

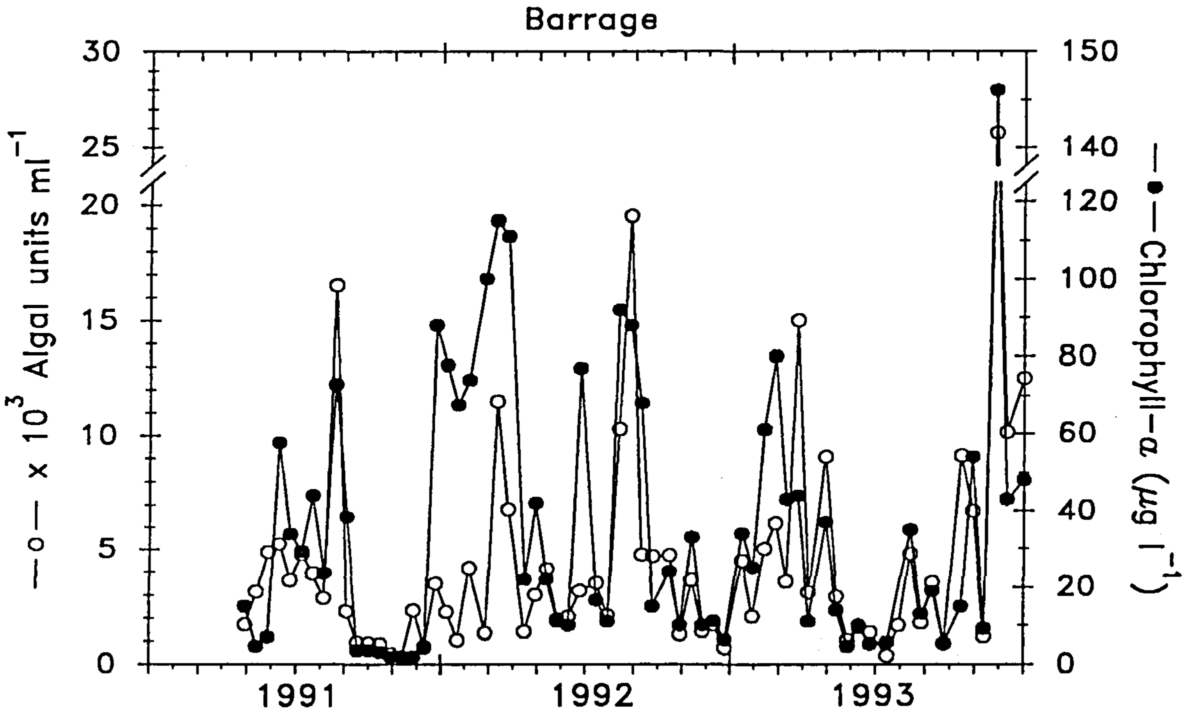


FIGURE 77: Variation in total algal unit (x 10^3) and chlorophyll-*a* concentration ($\mu g\ l^{-1}$) at the Barrage during the study period.

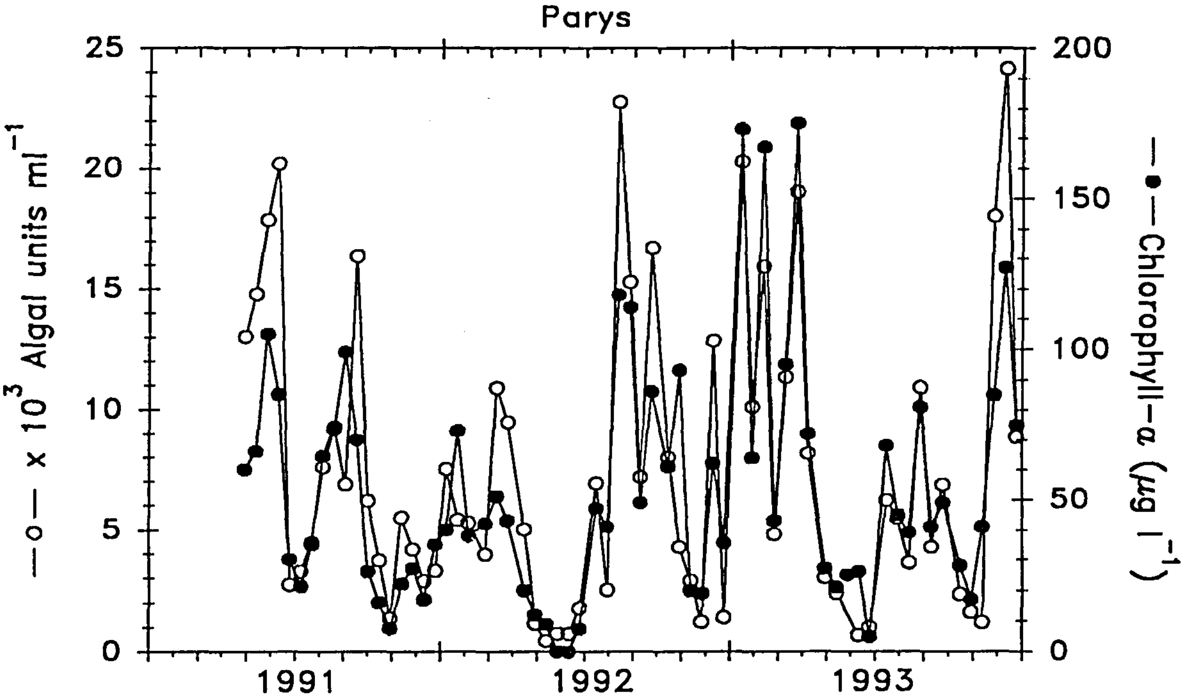


FIGURE 78: Variation in total algal unit (x 10^3) and chlorophyll-*a* concentration ($\mu g\ l^{-1}$) at Parys during the study period.

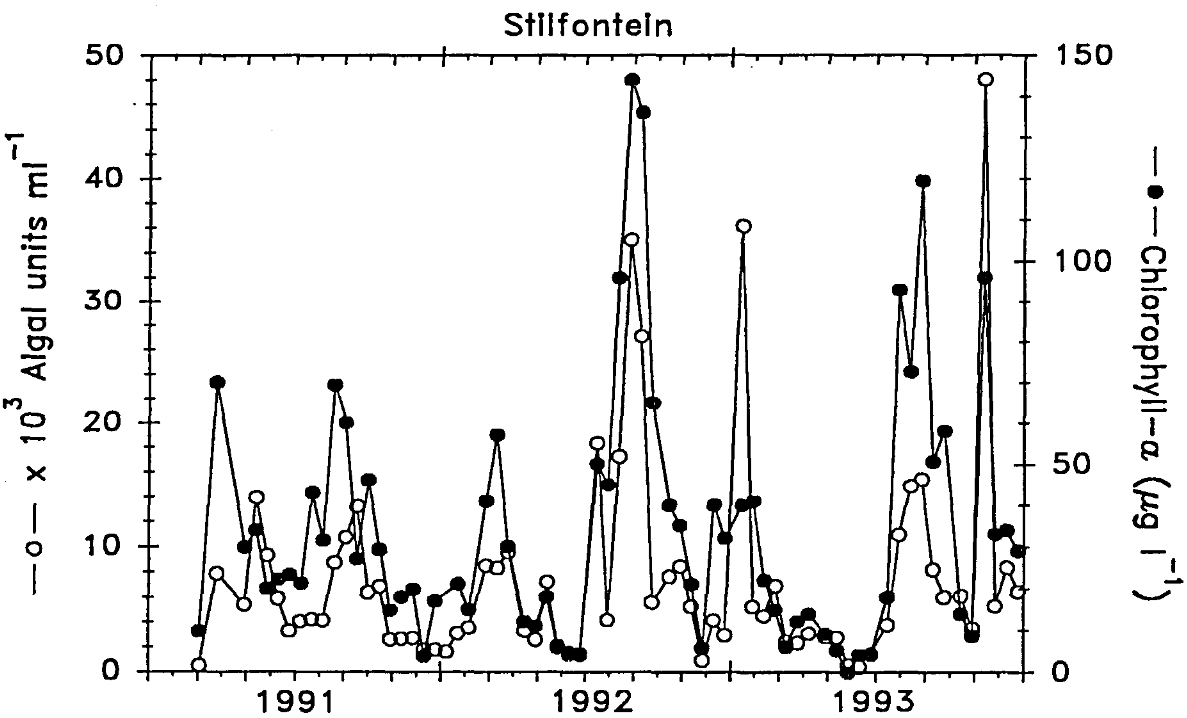


FIGURE 79: Variation in total algal unit (x 10^3) and chlorophyll-*a* concentration ($\mu g\ l^{-1}$) at Stilfontein during the study period.

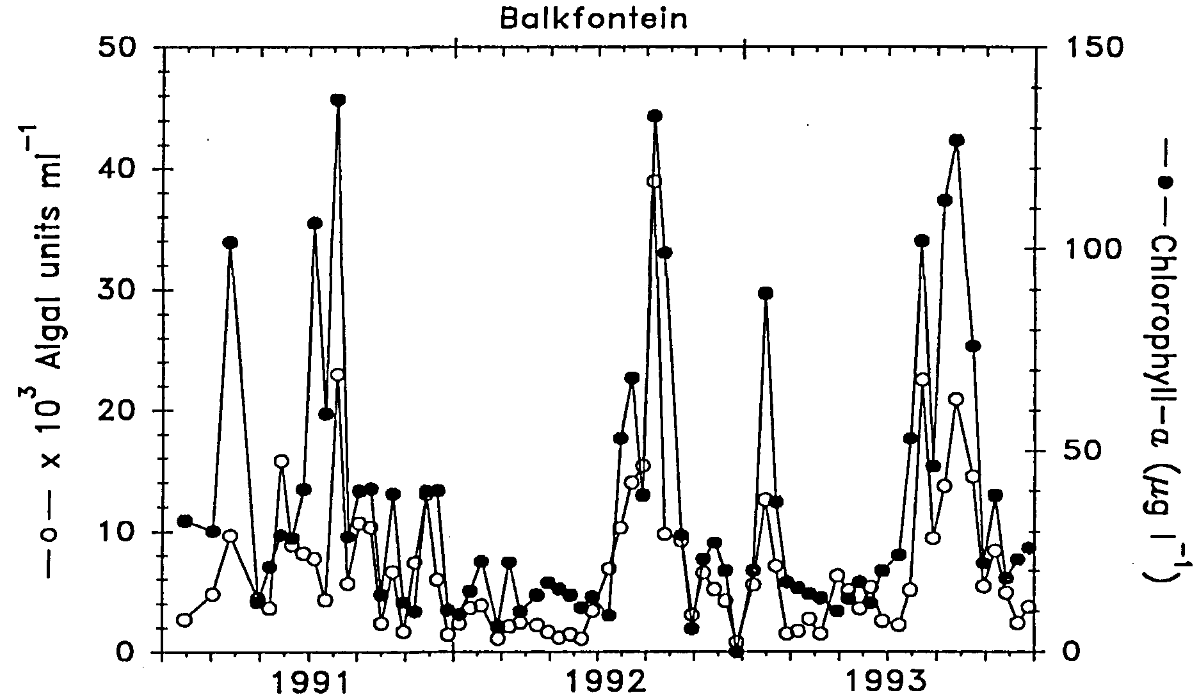


FIGURE 80: Variation in total algal unit (x 10^3) and chlorophyll-*a* concentration ($\mu g\ l^{-1}$) at Balkfontein during the study period.

the study period at the Barrage during the middle of November 1993 (Fig. 69). At that stage the turbidity of the water at the Barrage was relative low (compare Fig. 7).

Lack (1971) showed that the spring peaks of phytoplankton cell density occurred in the River Thames when the discharges were between 12.7 and 16.1 $m^3\ s^{-1}$. On almost every occasion the decline of a peak of cell numbers was coincident with a rise in discharge to a value exceeding 40 $m^3\ s^{-1}$. Lack (1971) argued that after an initial dilution of the phytoplankton (with increased discharge), the cells were swept away faster than they could divide. Statistical analysis on Vaal River data will give more information on the range of discharge at which maximum phytoplankton concentrations occur.

According to Hancock (1973) heavy thunder storms (which cause erosion of surface soil leading to an abrasive action of water in the streams) were the cause of low algal concentrations in the Klip River (tributary of the Vaal River).

It was shown by Roos (1992) that periods of floods almost completely removed phytoplankton populations (chlorophyll-*a* concentrations were reduced to less than 18 $\mu g\ l^{-1}$). The rapid and abundant growth (blooming) of phytoplankton is most likely to occur in streams in which the discharge is sufficiently low to permit the build-up of a considerable phytoplankton biomass (e.g., Lack, 1971; Reid & Wood, 1976; Petts, 1984; Ertl, 1985).

3.2.4.2 PHYTOPLANKTON AND TEMPERATURE

Rosemarin and Hart (1978) concluded that water temperature was the most important factor in determining seasonal variation in phytoplankton primary productivity and biomass in the Ottawa River.

In South African impoundments only one out of 33 displayed a maximum chlorophyll-*a* concentration during the winter-spring (June to September) period (Walmsley, 1984). Maximums were most frequently recorded during January. In other words, maximum chlorophyll-*a* concentrations coincide with high water temperatures in the majority of South African impoundments.

A view of all four sampling localities in the Vaal River during the present study (Figs 77 to 80), show that higher chlorophyll-*a* concentrations always occurred from January to March (summer period) and from July to November (winter-spring period), while low chlorophyll-*a* concentrations usually occurred during May and June of each year. High chlorophyll-*a* concentrations from January to March were frequently the result of blooms of *Melosira granulata* and sometimes *Oscillatoria simplicissima* (compare Figs 69-72), while high chlorophyll-*a* concentrations from July to November could be ascribed to blooms of unicellular centric diatoms and green algae (especially *Chlamydomonas incerta*; compare Figs 69-72). A comparison of Figs 73-76 show that *Melosira granulata* frequently dominated from December to May (summer periods) at all four sampling localities, while unicellular centric diatoms (succeeded by green algae) usually dominated during the winter periods. It was therefore concluded that *Melosira granulata* (filamentous centric diatom) prefers warmer water temperatures than the unicellular centric diatoms. It is known that high dominance by diatoms is often restricted to periods when the temperature is low (less than 15 °C), i.e. during the winter-spring period (Munawar & Munawar, 1975; Løvstad & Bjørndalen, 1990). During the present study winter minima (8-11°C) in water temperatures were often accompanied by dominance of unicellular centric diatoms, e.g. end of May to end of July 1991, beginning of July 1992 and June to July 1993 at the Stilfontein sampling locality (compare Figs 54 and 71) as well as during July 1991, 1992 and 1993 at the Balkfontein sampling locality (compare Figs 55 & 72). In northern marine European waters, the annual diatom maxima and temperature minima also tend to coincide.

During high water temperatures, blooms of blue-green algae were frequently observed in the Vaal River at all four sampling localities (compare Figs 52, 54 & 55 with Figs 69, 71 & 72 respectively). Temperature (together with nutrient availability) seems to be one of the most important variables regulating Cyanophyceae growth and succession in the Vaal River.

Maximum chlorophyll-*a* concentrations occurred when the water temperature started to increase after the winter minimum was reached, as well as during periods when high water temperatures were recorded. This tendency was found at the Barrage (compare Figs 77 and 52), Stilfontein (compare Figs 79 and 54) and Balkfontein sampling localities (compare Figs 80 and 55). At the Parys sampling locality no data on water temperature was available.

3.2.4.3 PHYTOPLANKTON, SALINITY AND MAJOR IONS

The DWA (1986) pointed out that the quality of many water sources in the RSA is declining, primarily because of salinisation, and the salinity in the Vaal River is particularly also gradually increasing. This probably cause flocculation of suspended matter and therefore a decrease in turbidity. The water becomes clearer and more light can penetrate the water to deeper levels. Due to these changes, an increase in salinity can result in underwater light climate (ULC) becoming more favourable for algal growth.

Very little information is available on the salinity tolerances of riverine organisms. It seems as if many species are able to survive at relative high salinities because certain freshwater blue-green algae can apparently adapt to TDS concentrations between 700 and 14 000 mg l^{-1} (Hart *et al.*, 1991). Tolerance to high salt concentrations enables blue-green algae to dominate the plankton of saline lakes, whether these be of the chloride, sulphate or carbonate type (Fogg *et al.*, 1973). Cleave *et al.* (1981) suggested that an increase in salt concentration could provide a competitive advantage to certain blue-green algae. In the Vaal River, however, it was found that relatively low conductivity values and TDS concentrations (less than 500 mg l^{-1}) were present during periods dominated by *Oscillatoria simplicissima* and *Microcystis aeruginosa* (compare Figs 11, 13 and 14 with Figs 69, 71 & 72). This shows that although blue-green algae, in general, are able to tolerate wide ranges of salinities, blue-green algal species present in the Vaal River usually dominate within narrow ranges of salinity. It must, however, be stated that salinity is not necessarily the primary variable influencing Cyanophyceae growth. Variables such as high temperature and nutrient availability (especially high inorganic nitrogen and phosphorus concentrations) are probably more important under favourable light conditions (see sections 3.2.4.1, 3.2.4.2 and 3.2.4.4). The occurrence of Cyanophyceae during periods of relatively low salinity in the Vaal River can be explained as follows: Cyanophyceae blooms were possibly initiated by increasing temperature and nutrient availability (as primary driving forces) during the summer period. Since summer rainfall occurs in the catchment of the Vaal River, an increase in discharge during this period brought about a dilution of the Vaal River water and lower salt concentrations are therefore recorded in the river. Taking this into account, it seems as if the rising temperatures played a more important role in initiating blue-green algal blooms than did salinity. However, more investigation on the relation between salinity and algal population composition are needed to better understand the effect of salinity on the composition of algal assemblages.

Work by Prinsloo* & Pieterse** showed that increases in TDS concentration in the middle Vaal River have been accompanied by decreases in turbidity (probably as a result of flocculation). They have also shown that growth and carbon assimilation in a green alga, *Monoraphidium circinale*, increased at TDS values between 500 and 2500 mg l^{-1}, while that of a diatom *Cyclotella meneghiniana* and the blue-green alga, *Microcystis aeruginosa*, were inhibited at these concentrations (see Chapter 4). It is often the rate of change rather than

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**Department of Plant and Soil Sciences, PU for CHE, Potchefstroom: personal communication.

the final salinity that is most critical (Dallas & Day, 1993). Many organisms are able to adjust to slow change by a process of acclimation that cannot be accomplished if an environmental change is rapid (Dallas & Day, 1993).

It was observed in our laboratories that the concentration of dinoflagellates (the group that causes red tides in the ocean) increased with an increase in salinity. During 1990 the total dissolved salts (TDS) concentration at Balkfontein increased on occasion to 1 030 mg l^{-1}. During this period one of the dinoflagellate species, *Sphaerodinium* sp. nov., was the dominant species in the river (unpublished information). Fig. 68B shows that the Dinophyceae was abundant during February 1992 at Balkfontein. At this stage the salt concentration in the river was high (867 mg l^{-1}; Fig. 14). It can therefore be predicted that the concentration of dinoflagellates may increase in future if the salinity increases in the river. In order to prevent dinoflagellate blooms, which might be toxic (Pieterse, 1986b), the salinity level in the Vaal Barrage should be regulated by means of water releases from the Vaal Dam and Vaal River Barrage. If the TDS concentration increases to levels above 600 mg l^{-1}, additional water is released from the Vaal Dam in order to dilute the water in the Barrage.

The ions Na^+, K^+, Mg^{2+}, Ca^{2+}, SO_4^{2-} and Cl^- are possibly present in large amounts in the Vaal River relative to plant needs. The question of the importance of the ratio of monovalent (Na^+ and K^+) to divalent (Mg^{2+} and Ca^{2+}) cations (M:D) is particularly interesting in respect to the distribution and dynamics of algae (Wetzel, 1983). For example, diatoms apparently prefer waters with a M:D ratio of below 1.5 and have a wide flexibility towards different Ca^{2+} to Mg^{2+} ratios. Higher M:D ratios are apparently favourable to desmids (Wetzel, 1983). The M:D ratios in the Vaal River during the present study display relatively small variations and were well below 1.5, with a mean of 0.8 at the Barrage (min. = 0.6; max. = 1), 0.9 at Parys (min. = 0.6; max. = 1.2), 0.9 at Stilfontein (min. = 0.6; max. = 1.2) and 0.9 at Balkfontein (min. = 0.6; max. = 1.2). Low M:D ratios recorded could possibly give an explanation for the high diatom biomass recorded in the Vaal River during the present study. Roos (1992) showed that the onset of diatoms blooms was accompanied by high Si and low M:D ratios. It is therefore possible that, together with other conditions, the resulting shift in M:D ratio, which is usually followed by a shift to mixed populations of predominantly green algae with diatoms, influenced phytoplankton succession in the Vaal River.

3.2.4.4 PHYTOPLANKTON, PHOSPHORUS (P) AND NITROGEN (N)

Schindler (1978) has argued that the maximum biomass in many temperate lakes is ultimately limited by the inorganic phosphate phosphorus (PO_4-P) supply, much in the same way as phytoplankton in the ocean is limited by the inorganic nitrogen (especially NO_3-N) supply. Several studies have shown a limiting effect of PO_4-P for phytoplankton growth in temperate lakes and reservoirs, but in the tropics N seems to be a critical nutrient (Henry *et al.*, 1984). The conventional wisdom is that inorganic N is limiting in the oceans and inorganic P is limiting in freshwater (Harris, 1986).

Increased nutrient levels frequently result in increased phytoplankton standing crop, i.e. supported by whole-lake fertilisation with phosphorus and nitrogen (e.g. Schindler, 1977;

Schindler *et al.*, 1978; Walmsley, 1980; Henry *et al.*, 1984; Welch *et al.*, 1989). The essence of the quantification of the effect of eutrophication is to determine "how much phytoplankton" for "how much nutrients" (Reynolds, 1984).

A correlation between the average chlorophyll-*a* concentration and the TP average has been documented by a number of investigators (e.g. Jones & Bachmann, 1976; Schindler *et al.*, 1978; OECD, 1982; Roos, 1992; Welch *et al.*, 1989). Studies showed that phosphorus availability probably controlled algal biomass (chlorophyll-*a* concentrations) in a variety of aquatic systems, including the Vaal River (Roos, 1992). The slope of a regression line drawn between chlorophyll-*a* and TP by Roos (1992) implied that a 1 mg l^{-1} increase in the concentration of TP will probably result in an almost 1 mg l^{-1} increase in chlorophyll-*a* concentration which could make it extremely simple to predict the standing crop in the Vaal River. Schindler *et al.* (1978) indicated that more chlorophyll was produced per unit phosphorus in the lakes of the experimental lake area (ELA) than in other lakes. This is probably because all the phosphorus added to the ELA lakes was in the form of PO_4-P, while in other lakes a high proportion of inputs was almost certainly not biologically available. Statistical analysis on the data of the present study will provide more information on the effect of an increase in phosphorus concentration on an increase in chlorophyll-*a* concentration during the present study.

The relationship between mean annual chlorophyll-*a* concentration and mean TP concentration can possibly be used for controlling eutrophication in the Vaal River, because much of the TP eventually becomes available. Limiting water fertility, preferably by controlling phosphorus supplies, is generally regarded to be the most desirable eutrophication control strategy (Lund, 1970; Dillon & Rigler, 1974; Jones & Bachmann, 1976; Schindler *et al.*, 1978; Jones & Lee, 1982; Premo *et al.*, 1985; Riley & Prepas, 1985; Young *et al.*, 1985; Auer *et al.*, 1986; Lukatelich & McComb, 1986; Steynberg, 1986; Botha, 1989), because phosphorus appears to be the key element limiting primary productivity. In addition to the phosphorus management strategy proposed by Steynberg (1986) discussed earlier, Botha (1989) also proposed a model (for Hartbeespoort Dam), to evaluate the economic effect of eutrophication and recommended the introduction of a 1 mg l^{-1} phosphate standard to all sewage works in the catchment.

Roos (1992) stated that eutrophication of the middle Vaal River may be alleviated or reversed by reducing the TP supply to the river. When sewage effluent was diverted from Lake Washington, Seattle, USA, a fairly rapid and parallel decrease in phosphorus concentrations and phytoplankton biomass was demonstrated, while nitrate decreased less rapidly (Edmondson, 1970).

Roos (1992) did not find a significant correlation between TN and chlorophyll-*a* concentration from 1985 to 1989 in the Vaal River. The relationship for phosphorus was better than that for nitrogen. The better correlation with PO_4-P is probably a reflection of the possibility that phosphorus was more often a factor limiting algal growth than nitrogen (Roos, 1992). However, when Figs 33 and 47 are compared with Fig. 81, it seems as if nitrogen availability played an important role in determinating the chlorophyll-*a* concentration at Parys, because highest DIN and TN concentrations were present at the Parys sampling locality - the sampling locality where the highest chlorophyll-a concentrations were recorded. Statistical analysis on data from the present study, will show if there was a correlation between nitrogen concentration and chlorophyll-*a*.

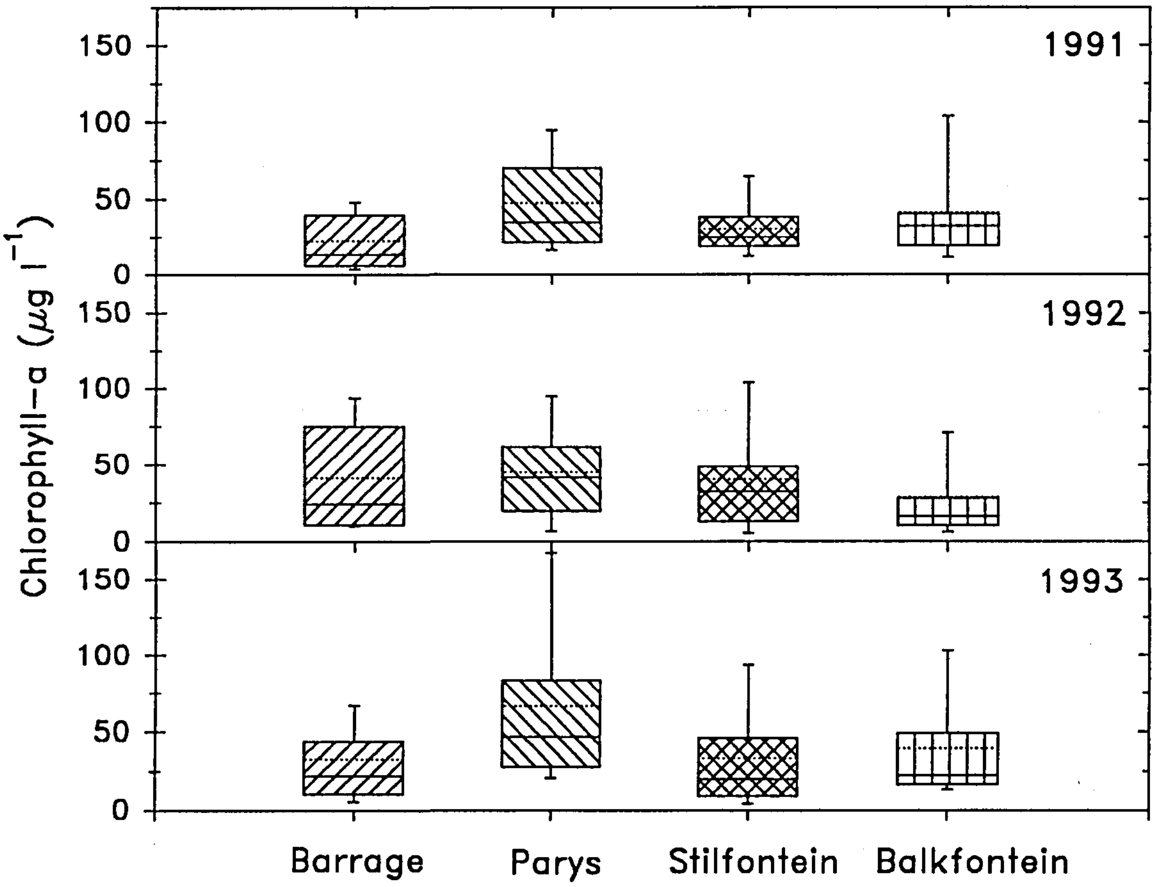

FIGURE 81: Box plot of annual chlorophyll-*a* concentration (μg l^{-1}) in the Vaal River at the Barrage, Parys, Stilfontein and Balkfontein.

Pieterse and Toerien (1978) demonstrated a significant correlation between average chlorophyll-*a* and PO_4-P concentrations in Roodeplaat Dam. Pieterse (1986b) also showed a statistically significant correlation for 1984 data on the Vaal River. Roos (1992), however, did not illustrate a statistically significant correlation between PO_4-P and chlorophyll-*a* concentration; an inverse tendency was rather demonstrated, probably because a luxury uptake of inorganic phosphorus by phytoplankton may mask any relationship between nutrient availability and chlorophyll-*a* concentration (Lehman *et al.*, 1975; Fogg, 1980). It was observed by Fogg (1980) that blooms of planktonic algae appear in freshwater when concentrations of phosphate and nitrate are at their lowest. High algal biomass accompanied by low concentrations of phosphorus and/or nitrogen can also be seen during the present study. A comparison of Fig. 77 with Figs 24 and 29 show that this tendency was frequently observed during bloom periods at the Barrage (e.g. beginning of March 1992 - low N concentrations, end of August 1992, February 1993 - low P and N and November 1993 - low P). At the Parys sampling locality high algal biomass was also frequently accompanied by low phosphorus and/or nitrogen concentrations, e.g. June 1991 - low P concentration, August 1992, January & February 1993 - low P and N, December 1993 - low N (compare Fig. 78 with Figs 25 and 30). At the Stilfontein sampling locality (compare Fig. 79 with Figs 26 and 31), peaks of algal biomass accompanied by low values of phosphorus and nitrogen concentrations occurred during the end of August 1992, beginning of January and November 1993. At the Balkfontein sampling locality high algal biomass was also frequently accompanied by low phosphorus and nitrogen concentrations, e.g. March, July and August 1991, September 1992 and January & September 1993 (compare Fig. 80 with Figs 27 and 32). A direct measurement of phosphate in water rarely gives an accurate measure of phosphorus available to algae (Fogg, 1980; Reynolds, 1984). A measurement of the concentration of a nutrient is not a sufficient test of whether or not it is limiting. As Brylinsky and Mann (1973) pointed out, the rate of cycling of nutrients is important, not only the concentration in the water at any one time.

Several authors have sought to relate mean summer standing crops (chlorophyll-*a* concentration) to the phosphorus concentration obtained at the beginning of the growing season, i.e. at overturn or during spring in lakes. Inherent in this relationship is a time delay of 2 to 4 months (Sakamoto, 1966; Edmondson, 1970; Lund, 1970; Dillon & Rigler, 1974; Riley & Prepas, 1985). The same tendency was also shown for estuaries e.g. Lukatelich and McComb (1986) showed, in the Peel-Harvey estuary system, that the magnitude of a *Nodularia* sp. bloom clearly depends on the amount of phosphorus entering the system in winter, but the two events are separated in time by some 2 to 3 months. According to Cushing and Dickson (1976), aquatic plants and plants in general are integrators of environmental variables and they respond to changes in environmental factors with a certain lag period occurring between the environmental stimulation and its response. Harris (1986) showed that some time-lags were of the order of four to eight weeks, indicating that long time-lags may occur between the initiation of growth and the appearance of the species as dominant in the assemblage. Vollenweider (1953) stated that it is the factors which cause the bloom that are important, not the conditions during the development of the bloom. This phenomenon makes it difficult to compare phosphorus and nitrogen concentrations in relation to algal blooms by inspecting temporal graphs. Although the retention time of the Vaal River is not known, it seems as if a similar significant correlation was demonstrated between summer maximum TP during the annual high flow in the river and the average spring chlorophyll-*a* concentration (Roos, 1992). During the present study blooms of phytoplankton were, in several occasions, preceded by high phosphorus concentrations (with a time lag of 3 to 8 weeks). A very high DIP concentration was present in the water at the Barrage during the end of December 1992 (Fig. 24). This high DIP concentration can be responsible for an increase in algal biomass which reached a peak two months later during the end of February 1993 (Fig. 77). Since DIP is an important component of the TP, especially at the Barrage (55%), blooms of phytoplankton preceded by high TP concentrations can also be seen when the algal biomass at the Barrage (Fig. 77) is compared with the TP concentration (Fig. 38). Peaks in algal biomass during the end of February and the end of November 1993 can be ascribed to high TP concentrations (probably resulting in better PO_4-P supply) prevailing during the end of January and during the end of October 1993, respectively.

At the Parys sampling locality the tendency of algal blooms preceded by high phosphorus concentrations can also occasionally be seen. An increase in DIP concentration during June 1992 (although very high values were not recorded) might have been the cause of an increase in total algal unit concentration which reached values of approximately 23 000 units ml^{-1} during the beginning of August 1992 (compare Figs 25 and 78). High DIP concentrations during the beginning of November 1993 (Fig. 25) as well as high TP concentrations during the same period (Fig. 39) might possibly be the reason for an algal bloom reaching total algal unit concentrations of almost 25 000 units ml^{-1} during the beginning of December 1993 (Fig. 78).

At the Stilfontein sampling locality three periods of high algal biomass preceded by higher DIP concentrations could have been identified. High DIP concentrations during the end of February/beginning of March 1991 (Fig. 26) could be responsible for the higher algal biomass recorded during the end of March/beginning of April 1991 (Fig. 79). A comparison of Figs 26, 40 and 79 also show a peak in DIP, and especially TP concentrations, during November 1992 that was followed by an increase in total algal units until it peaked during January 1993. During the end of October 1993 very high DIP and TP concentrations were recorded (Figs 26 & 40). These high concentrations could be the cause of an algal bloom during November 1993, a stage at which total algal units of approximately 49 000 units ml^{-1} were recorded (Fig. 79).

At Balkfontein only one peak in DIP concentration could be identified which might have been the cause of an algal bloom approximately three weeks later. During the end of February 1991 high DIP concentrations were recorded at the Balkfontein sampling locality (Fig. 27). This was followed by an increase in algal biomass until a peak in chlorophyll-*a* could be observed during March 1991 (Fig. 80). High TP concentrations during the beginning of December 1992 (Fig. 41) could possibly have been the cause of an increase in chlorophyll-*a* concentration which formed a peak during the end of January 1993 (Fig. 80).

3.2.4.5 PHYTOPLANKTON, DIN:DIP AND TN:TP RATIOS

As discussed in section 3.1.2.3, the Redfield ratio implies that at an N:P (DIN:DIP) ratio of less than 7 in the environment, inorganic N could be limiting and at an N:P ratio greater than 7, inorganic P could be limiting to algal growth.

Sakamoto (1966) concluded that chlorophyll yield (in Japanese lakes) was possibly dependent only on TN when the TN:TP ratio was less than 10, and only on TP when the TN:TP ratio was greater than 17. However, Hoyer and Jones (1983) indicated that in Midwest USA reservoirs (Missouri) with TN:TP less than 10, nitrogen accounted for the same amount of variance in chlorophyll-*a* as did phosphorus. The average TN:TP ratios recorded in the Vaal River during the three years of the study period was 9.9 at the Barrage, 10.2 at Parys, 12.9 at Stilfontein and 11.9 at Balkfontein (Figs 48 to 51). These averages indicate that TN:TP ratios during the present study were always below 13. In section 3.2.4.4 it was illustrated that the average annual chlorophyll-*a* concentrations followed similar patterns to the average annual TN concentration (highest average recorded at Parys; compare Fig. 81 with Fig. 47). It therefore seems as if the average chlorophyll concentrations are primarily dependant on the average TN concentrations at low TN:TP ratios (< 13). This is in accordance with the findings of Sakamoto (1966) as indicated above. An analysis of the total nitrogen (TN) and total phosphorus (TP) data from 55 lakes by Harris (1986) revealed that the TN:TP ratio varied between above 200 in oligotrophic lakes and less than 15 in eutrophic lakes. The average TN:TP ratios recorded in the Vaal River during the three years of the study period were all below 15, indicating eutrophic conditions.

Roos (1992) indicated that the chlorophyll-*a* concentration from 1985 to 1989 in the Vaal River at Balkfontein was positively correlated with the TN:TP ratio. This suggested that the phosphorus concentration was, relative to nitrogen, lower when the chlorophyll-*a* concentration was high, resulting in a high TN:TP ratio and *vice versa*. This observation suggested that phosphorus may limit algal growth during high phytoplankton concentrations. Statistical analysis done on the data of the present study, will reveal if this was also the tendency occurring during the present study.

Smith (1982) indicated by a multiple regression analysis model that the chlorophyll yield in 127 north latitude lakes was related to both TP and TN. This was also found by Roos (1992) on studies done on the Vaal River at Balkfontein.

The occurrence of blue-green algae are generally associated with high loading of nitrogen and phosphorus that favours algae which do not require silica (Goldman & Horne, 1983). Fogg *et al.* (1973) stated that blue-green algae can be used as indicators of pollution in rivers. Eutrophication of water bodies is often followed by significant shifts in phytoplankton composition towards blue-green algae (Steinberg & Hartmann, 1988). In eutrophic South African impoundments like Hartbeespoort Dam and Roodeplaat Dam a blue-green alga *Microcystis aeruginosa*, is the most frequent bloom-forming algal species (e.g. Robarts & Zohary, 1984; CSIR, 1985; De Wet, 1986; Pieterse & Röhrbeck, 1990). Although not the dominant group in the Vaal River, blue-green algal representatives, for example *Oscillatoria simplicissima*, *Microcystis aeruginosa* and *Synechococcus cedrorum*, frequently occurred, sometimes (especially in the late summer periods) reaching dominant proportions (compare Figs 65B to 68B).

As discussed in sections 3.2.4.1, 3.2.4.3 and 3.2.4.7 several conditions in the Vaal River (e.g. turbidity, salinity and pH) can be considered favourable to blue-green algal growth. However, it seems as if the key factor determinating blue-green algal dominance is the nitrogen to phosphorus ratio. Sommer (1985) showed that where diatoms compete with Chlorophyta and Cyanophyta, diatoms do better at high Si:DIP ratios but are favoured by low DIN:DIP ratios; Chlorophyta do best at low Si:DIP ratios but are favoured by high DIN:DIP ratios, and Cyanophyta do best at low Si:DIP ratios but are favoured by low DIN:DIP ratios. In the Vaal River at all four sampling localities, it was found that, when high Si:DIP ratios occurred, the Bacillariophyceae (mainly centric diatoms) was the dominant phytoplankton group. At the Barrage sampling locality higher Si:DIP ratios occurred from January to June 1991 and from July to October 1993 (Fig. 57), periods that were also characterised by dominance of centric diatoms (compare Fig. 69). At the Parys sampling locality several periods of higher Si:DIP ratios occurred, namely from March to June 1991, from December 1992 to July 1993 and a smaller peak in Si:DIP ratio also occurred during October 1993 (Fig. 58). During all three these periods centric diatoms were dominant at the Parys sampling locality (Fig. 70). At the Stilfontein sampling locality an increase in Si:DIP ratio occurred from September to reach extremely high Si:DIP ratios (due to very low DIP concentrations) during October and November 1991 (Fig. 59). Centric diatoms were dominant at Stilfontein from September to November 1991 (Fig. 71). At the Balkfontein sampling locality certain periods of high Si:DIP ratios (e.g. January 1992 and December 1992 to January 1993) were also dominated by centric diatoms (compare Figs 60 and 72). In contrast to the situation described by Sommer (1985), high Si:DIP ratios in the Vaal River were usually accompanied by high DIN:DIP ratios, because of very low dissolved inorganic phosphorus (DIP) concentrations during these periods. Centric diatoms thus frequently dominated during periods of high Si:DIP and DIN:DIP ratios. Chlorophyceae usually dominated during periods characterised by relatively low Si:DIP and DIN:DIP ratios at all four sampling localities (compare Figs 69 to 72 with Figs 57 to 60 respectively). The statement of Sommer (1985) that Cyanophyceae do best at low Si:DIP ratios, but are favoured by low DIN:DIP ratios, was confirmed by results from the present study, except for the Balkfontein sampling locality (Fig. 60) where relatively high Si:DIP ratios occurred during the end of April and May 1991 - a period when *Oscillatoria simplicissima* was the dominant species in the river. During the present study blooms of blue-green algae were always associated with water showing low DIN:DIP ratios. At the Barrage sampling locality *Oscillatoria simplicissima* was dominant during the end of January 1992 (DIN:DIP ratio of 0.7), December 1992 (DIN:DIP = 3.07) and February 1993 (DIN:DIP = 12.8; compare Fig. 69 with 34). *Microcystis aeruginosa* was also dominant during the beginning of May 1992 at the Barrage (Fig. 69) when a DIN:DIP ratio of 2.3 was recorded (Fig. 34). At the Stilfontein sampling locality *Oscillatoria simplicissima* dominated during February and March 1991 (Fig. 71). At this stage DIN:DIP ratios of 14 were recorded in the river (Fig. 36). Another representative of the blue-green algae, *Synechococcus cedrorum* dominated during October 1992 when a DIN:DIP ratio of 5.9 was recorded at the Stilfontein sampling locality (compare Fig. 71 with Fig. 36). *Oscillatoria simplicissima* was also dominant at the Balkfontein sampling locality from the end of April until the end of May 1991. During this period low DIN:DIP ratios of 0.7, 1.4 and 2.9 were recorded (compare Fig. 72 with Fig. 37). Harris (1986) showed that the large summer blooms in Blelham Tarn in recent years were associated with a shift towards blue-greens that resulted from lower DIN:DIP ratios. Smith (1982) illustrated an optimal DIN:DIP ratio of approximately four for the blue-green alga, *Microcystis* sp.. Recently, in Hartbeespoort Dam, the absence of the *Microcystis aeruginosa* bloom during 1988/89 was ascribed to the low epilimnetic phosphate concentration and the resultant increasing DIN:DIP ratios, i.e. from about 4 to 10 (Chutter, 1989).

Shapiro (1990), on the other hand, stated that the DIN:DIP ratio was not the primary variable influencing blue-green algal dominance or abundance. He placed emphasis on the importance of CO_2 and the pH of the water. He related the effectiveness of DIN:DIP ratios, in changing algal dominance to blue green algae, to CO_2. Lower DIN:DIP ratios appear to be directly related to increasing eutrophication (Forsberg, 1979). Therefore, as productivity increases and CO_2 becomes limited there will develop an apparent relation between low DIN:DIP and blue-green algal dominance. However, several other authors (e.g. Schindler, 1977; Barica *et al.*, 1980; Smith, 1982 and Harris, 1986) states that blue-green algae, both nitrogen-fixing and non-fixing, are benefited by low DIN:DIP ratios. During spring blooms in the Vaal River (dominated by *Chlamydomonas incerta* and centric diatoms; Figs 69-72) relative high pH levels of 8 to 9 were frequently recorded (Figs 61-64) without replacement by blue-green algae as the dominant species. This shows that high pH values alone was not responsible for a shift from diatoms and green algae to blue-green algae as the dominants. Shifts were rather caused by a combination of environmental variables favouring blue-green algal growth, e.g. high temperatures, low DIN:DIP ratios and relative high pH values (see section 3.2.4.7).

Smith (1983) analysed data from 17 lakes and concluded that, although blue-green participation could range 0-100% in lakes with low TN:TP ratios (less than 29/1), at higher ratios blue-greens would represent but a small proportion of the population. More recently, McQueen & Lean (1987), in attempting to understand blue-green dominance in Lake St. George, Ontario, showed that there was no correlation between percent blue-greens and the TN:TP ratio. However, they did find that if they used the ratio as NO_3:TP there was an inverse relationship with blue-green dominance. In the Vaal River periods of blue-green algal dominance were always characterised by low TN:TP ratios (compare Figs 69-72 with Figs 48-51 respectively). Compiled data from 12 north-temperate lakes (Smith, 1982) clearly show shifts from blue-green algae to green algae and diatoms as the total nitrogen to total phosphorus (TN:TP) ratio in the lakes increased. This tendency could also be seen during the present study at two of the three sampling localities where blue-green algae reached dominance, namely at the Barrage and Stilfontein. From May 1992 a sharp increase in TN:TP ratio was observed at the Barrage sampling locality (Fig. 48). This increase in TN:TP was accompanied by a shift from *Microcystis aeruginosa* (blue-green alga) to *Melosira granulata* (filamentous centric diatom) as the dominant species (Figs 65B and 69). The same phenomenon was observed at the Stilfontein sampling locality where an increase in TN:TP ratio occurred from May 1991 (Fig. 50). At this stage blue-green algal representatives (mainly *Oscillatoria simplicissima*) were replaced by other unicellular centric diatoms (compare Figs 67B and 71). From December 1992 an increase in TN:TP ratio again occurred (Fig. 50), accompanied by a transition from blue-green algal dominance (*Synechococcus cedrorum*) to green algal species and then to diatom species (Figs 67B & 71). These results show that nutrient ratios affect the composition of mixed-species assemblages in lakes and rivers.

Smith (1983) reported a tendency for blue-green algal blooms to occur when epilimnetic TN:TP ratios decreased to below about 29:1 by mass, and that blue-green algae were rare when the TN:TP ratio is in excess of this value. Harris (1986) showed that at a TN:TP ratio of less than 10, there is a 40 to 50% probability that *Microcystis* sp. will occur. Low TN:TP ratios in the Vaal River possibly indicated more favourable conditions for blue-green algal growth, but the probability that blue-greens would occur is still less than 50% (Harris, 1986).

It is clear from the above discussion that low DIN:DIP or TN:TP ratios most probably favour blue-green algal growth. The average DIN:DIP ratios in the Vaal River was 15.9 at the Barrage, 48.6 at Parys, 45.8 at Stilfontein and 10.3 at Balkfontein. If a comparison of the phytoplankton composition at the four sampling localities is made (Figs 65B to 68B), it seems as if Cyanophyceae biomass (and diversity - see section 3.2.2) was the greatest at the Barrage sampling locality (Fig. 65B) and lowest at the Parys sampling locality (Fig. 66B). Higher blue-green algal biomass at the Barrage can be ascribed to, amongst others, lower average DIN:DIP ratios than at the Parys sampling locality. However, at the Stilfontein and Balkfontein sampling localities more or less the same amount of blue-green algal biomass occurred despite the differences in DIN:DIP ratios (compare Figs 67B and 68B).

Increased phosphorus concentrations in the Vaal River (with a resultant decrease in DIN:DIP and thus TN:TP to less than five) will probably cause a shift in the algal assemblages from diatom and green algal dominance to blue-green algal dominance.

3.2.4.6 PHYTOPLANKTON AND SILICATE-SILICON (SiO_2-Si)

Because of the lag period occurring between environmental stimulations and the responce of algal assemblages, some inputs (e.g. from terrestrial sources) may take time before inorganic nutrients become available for re-cycling by way of mineralisation.

Harris (1986) showed that some time-lags were of the order of four to eight weeks indicating that long time-lags may occur between the initiation of growth and the appearance of the species as a dominant in the assemblage. Descy *et al.* (1987) showed for the River Meuse that the phytoplankton growth phase began in spring and maxima were reached during the summer period, i.e. approximately three months later.

An example of this kind of time-lag in the Vaal River during the present study can be seen in the illustrated inverse relationship between silicon and diatom concentration (Figs 82 to 85).

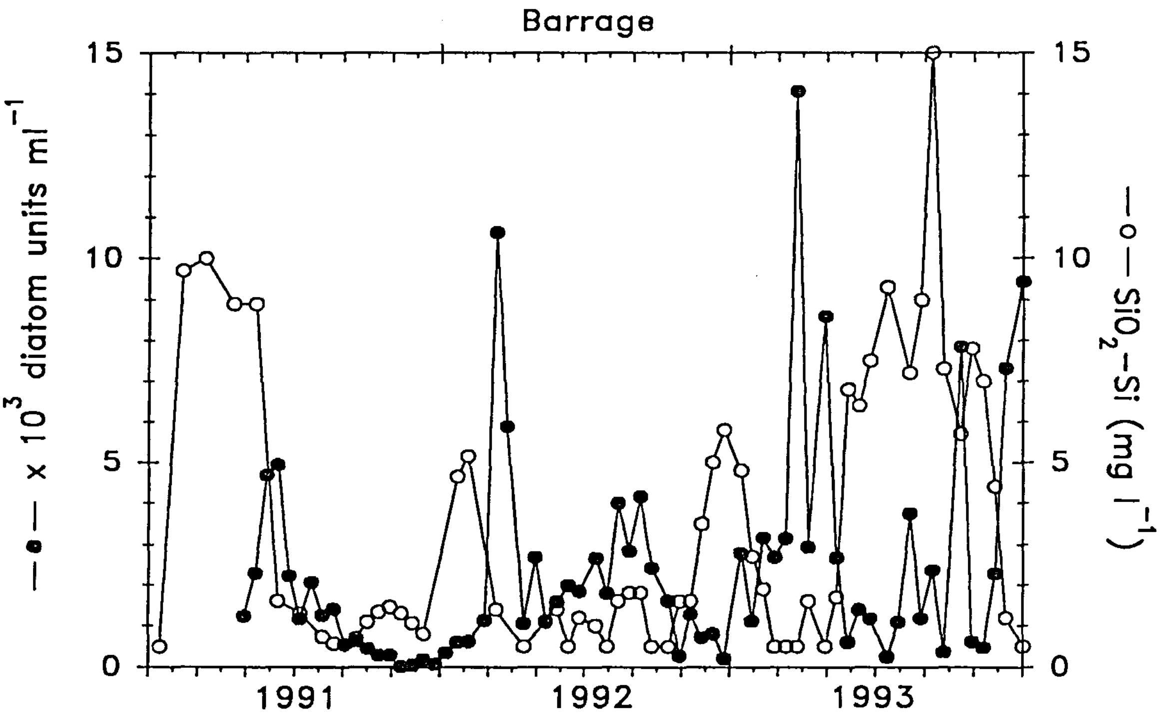

FIGURE 82: Variation in diatom unit (x 10^3) and silicate-silicon (SiO_2-Si) concentration (mg l^{-1}) at the Barrage during the study period.

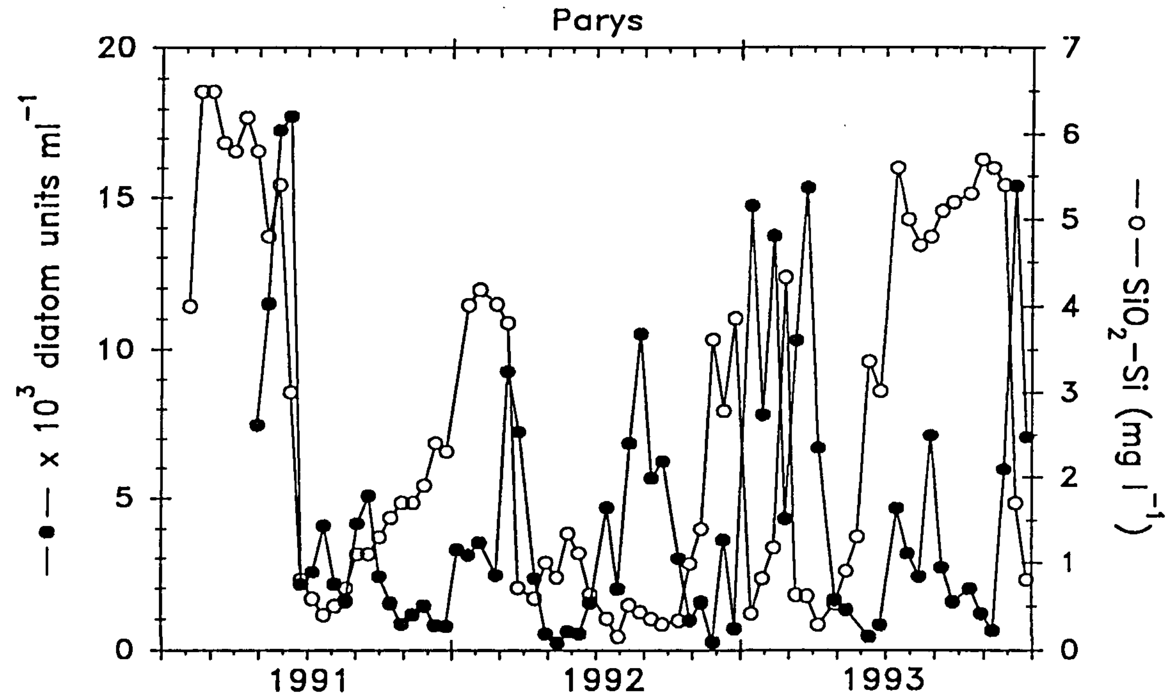

FIGURE 83: Variation in diatom unit (x 10^3) and silicate-silicon (SiO_2-Si) concentration (mg l^{-1}) at Parys during the study period.

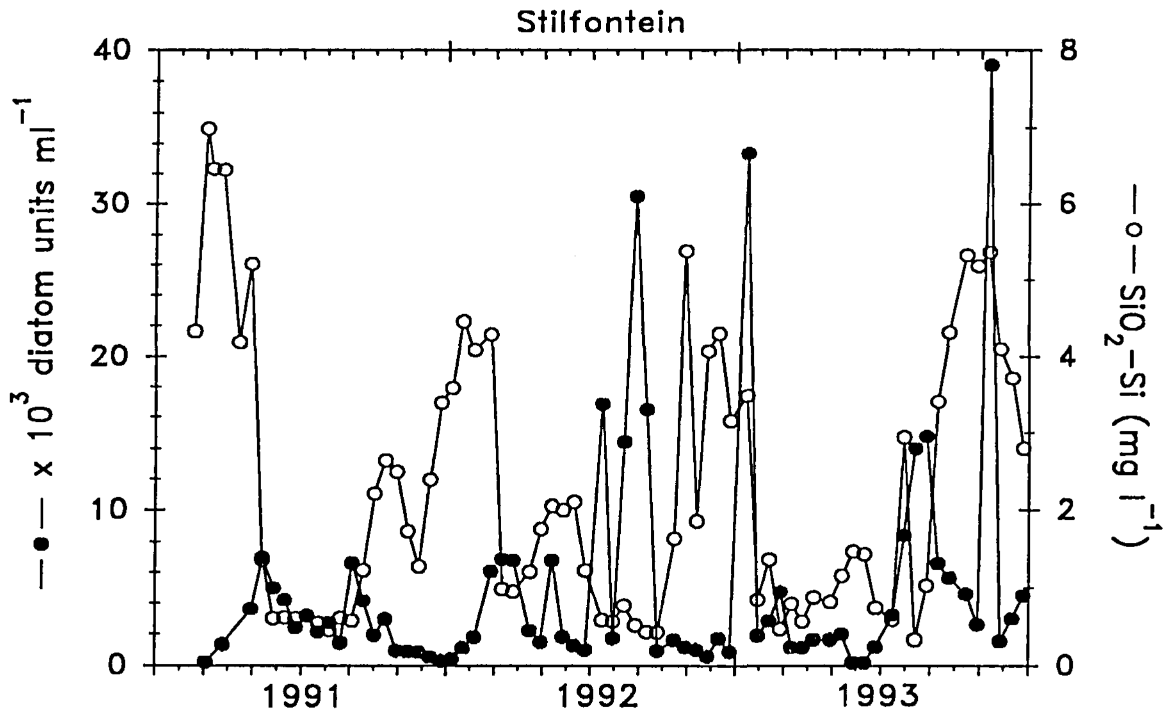

FIGURE 84: Variation in diatom unit (x 10^3) and silicate-silicon (SiO_2-Si) concentration (mg l^{-1}) at Stilfontein during the study period.

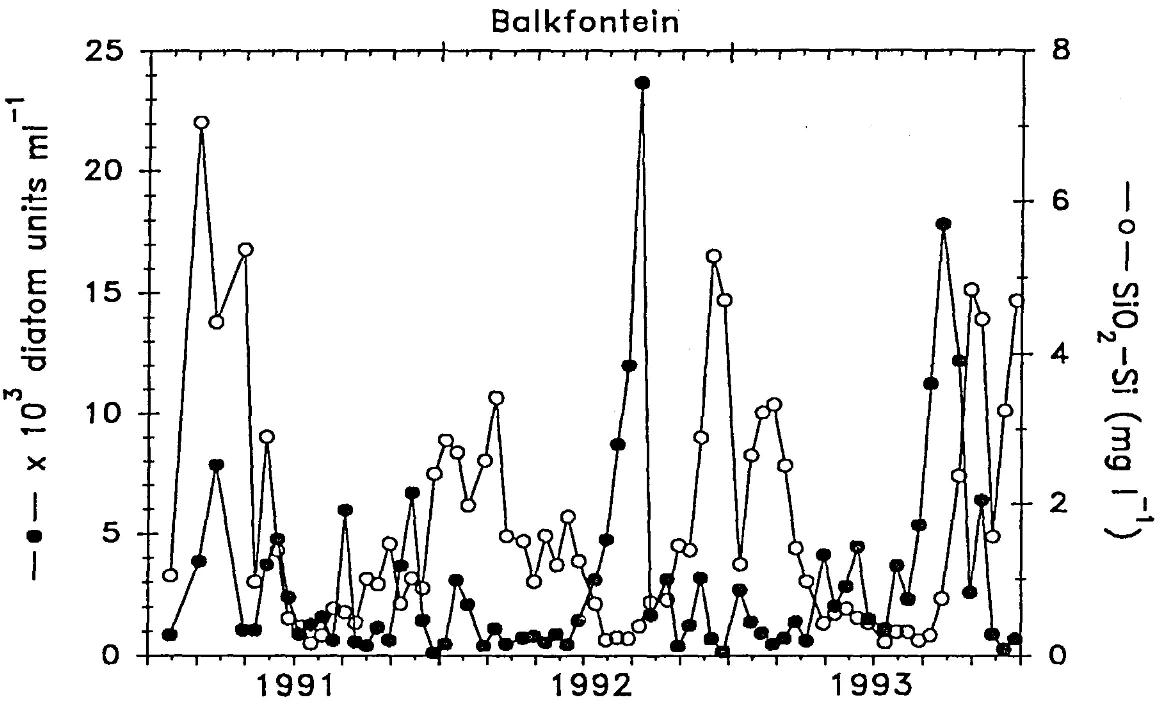

FIGURE 85: Variation in diatom unit (x 10^3) and silicate-silicon (SiO_2-Si) concentration (mg l^{-1}) at Balkfontein during the study period.

At all four sampling localities (to a lesser extent at Stilfontein), low silicon concentrations occurred during periods when the biomass of diatoms was high and *vice versa* (Figs 82 to 85). Sudden decreases in SiO_2-Si concentration can frequently be ascribed to sudden increases in diatom numbers, e.g. June 1991, August and October 1993 at the Barrage (Fig. 82), March and December 1992 as well as January, March and December 1993 at the Parys sampling locality (Fig. 83). At the Stilfontein sampling locality decreases in SiO_2-Si concentration accompanied by increases in diatom biomass can be observed during May 1991, March and June 1992, July to August 1993 (Fig. 84) and at Balkfontein during March 1991 and June 1992 (Fig. 85). Observations showing sharp decreases in silicon concentrations during diatom blooms were also made during other investigations in various aquatic systems (Lund, 1964; 1965; Jones, 1977; Lack *et al.*, 1978; Lukatelich & McComb, 1986; Descy *et al.*, 1987). Figs 82-85 show that blooms of diatoms were often preceded by peaks in the SiO_2-Si concentration. The time-lag in the Vaal River apparently varied from about one month at the upstream sampling localities to periods of almost six months at Balkfontein. At the Barrage sampling locality (Fig. 82) the time difference between peaks in SiO_2-Si concentration and maximum diatom biomass ranged from approximately one month (e.g. peak of SiO_2-Si during beginning of February 1992, followed by maximum diatom biomass during March 1992) to three months (e.g. peak of SiO_2-Si during beginning of January 1993, followed by maximum diatom biomass during the end of March/beginning of April 1993). At the Parys sampling locality time-lags of one to two months can be observed (Fig. 83), while the time difference between peaks of SiO_2-Si and diatom units ranged from one to two and a half months at the Stilfontein sampling locality (Fig. 84). It is not always certain exactly which of the preceding peaks in SiO_2-Si concentration are responsible for the peak in diatom concentration, but in general it seems as if the time-lags recorded at the downstream sampling locality at Balkfontein (Fig. 85) were longer than those recorded at the more upstream sampling localities. It seems as if time-lags of almost six months occurred at the Balkfontein sampling locality during the present study (e.g. peak in SiO_2-Si concentration during March 1992 and peak in diatom numbers during September 1992; Fig. 85). Time-lags of approximately four months between chlorophyll-*a* concentration and SiO_2-Si concentration were demonstrated by Roos (1992) at the Balkfontein sampling locality from 1985 to 1989. The observations on the Vaal River support the observation made by Vollenweider (1953), namely that it is the factor(s) which cause(s) the bloom that are important, not the conditions during the occurrence of the bloom. Such time-lags interfere with attempts to correlate the species abundance with ecological conditions at any point in time (Harris, 1986).

Harris (1986) indicated that spring diatom blooms in freshwater may be terminated in several ways, but a combination of grazing, silicate depletion, thermal stratification and sedimentation usually brings the bloom to an end. Menzel *et al.* (1963) suggested that silicate depletion is the primary mechanism leading to species succession of non-silicious flagellates following the spring diatom bloom in sub-tropical waters. Figs 82 to 85 show that diatom blooms were also responsible for the depletion of SiO_2-Si in the surface waters of the Vaal River. During several periods of diatom blooms, SiO_2-Si concentrations dropped to levels likely to limit growth, i.e. concentrations frequently declining to levels below 0.5 mg l^{-1}. Reid & Wood (1976) stated that the development of certain diatom populations is limited, at least partially, by concentrations from 0.5 to 0.8 mg SiO_2 l^{-1}.

Lack *et al.* (1978) calculated that an algal crop of around 20 * 10^6 cells l^{-1}, required a silica concentration of 800 μg l^{-1} to enable cell division. Higher diatom biomass was recorded during the present study at the Stilfontein and Balkfontein sampling localities (see Figs 84 & 85). At the Stilfontein sampling locality relatively high SiO_2-Si concentrations were observed during periods of diatom blooms. Figs 84 shows that diatom concentrations of more than 30 000 units ml^{-1} were recorded during the beginning of September 1992 (SiO_2-Si = ± 0.5 mg l^{-1}), January 1993 (SiO_2-Si = ± 3.5 mg l^{-1}) and November 1993 (SiO_2-Si = ± 5.5 mg l^{-1}) at the Stilfontein sampling locality. Despite these relatively high SiO_2-Si concentrations, it seems as if the SiO_2-Si was still limiting diatom division, because a decline in the diatom biomass occurred immediately after high concentrations were reached. At the Balkfontein sampling locality (Fig. 85) diatom concentration exceeded 23 000 cells ml^{-1} during September 1992. At this stage an extremely low SiO_2-Si concentration (less than 0.5 mg l^{-1}) was recorded.

There are indications that the abundance of diatoms at certain sampling localities may have an influence on the SiO_2-Si concentration at the particular sampling locality. From 1991 to 1993 the average diatom biomass at the Barrage, (2 383 units ml^{-1}; Fig. 82), was lower than at the downstream sampling localities (Parys 4 560 units ml^{-1}, Stilfontein 4 963 units ml^{-1}, Balkfontein 3 092 units ml^{-1}; compare with Figs 83, 84 and 85 respectively). Lower diatom biomass at the Barrage could probably be the reason why the highest average SiO_2-Si concentrations were recorded at the Barrage (3.4 mg l^{-1}), decreasing downstream to the Parys (2.56 mg l^{-1}), Stilfontein (2.29 mg l^{-1}) and Balkfontein (1.7 mg l^{-1}) sampling localities (compare with Fig. 56).

3.2.4.7 PHYTOPLANKTON AND HYDROGEN ION CONCENTRATION

During daytime aquatic plants (especially phytoplankton) photosynthesise and assimilate carbon dioxide from the water. The assimilation of carbon dioxide lowers the CO_2 concentration in the water, with an associated increase in pH. During the night respiration (where oxygen is consumed and CO_2 released) is more important than photosynthesis. The increase in CO_2 concentration during respiration results in lower pH values during the night.

Data from the present study shows periods when an increase in algal biomass was accompanied by an increase in the pH level. At the Barrage sampling locality the sharp increase in algal biomass during the end of November 1993 (Fig. 77) could be responsible for the increase in pH during the same period (approximately 1.8 pH unit; Fig. 61). At the Parys sampling locality, increases in pH levels from the end of February to the beginning of May 1993 as well as from the beginning of November to the beginning of December 1993 (Fig. 62) were probably the result of increases in algal biomass during the same time (Fig. 78). At the Stilfontein sampling locality the highest pH level (8.8) was recorded during September 1992 (Fig. 63), a period also characterised by a very high algal unit concentration (Fig. 79). At the Balkfontein sampling locality, peaks in pH accompanied by peaks in algal biomass could not be demonstrated clearly (compare Figs 64 with 80).

Harris (1986) stated that the ability of blue-green algae to grow under high pH conditions appears to be correlated with their ability to produce late summer blooms in eutrophic waters. The relatively high pH of the Vaal River water at the Barrage and Balkfontein (average 8.2) as well as Parys and Stilfontein (average 8.1) sampling localities could therefore have been beneficial to blue-green algal growth. During periods when the blue-green algae were more important (although not necessarily dominant), the pH varied between 6.9 and 9.4 at the Barrage, between 7.3 and 8.6 at Parys, between 7.3 and 8.3 at Stilfontein and between 8.1 and 8.6 at Balkfontein. During periods of blue-green algal dominance, the pH was usually above 8 in the river at all four sampling localities. It can, however, not be concluded that high pH was the most important variable responsible for blue-green algal blooms, because certain periods characterised by diatom and green algal dominance also showed relatively high pH values of 8 and 9. A combination of several environmental variables are responsible for a shift from diatom and green algal dominance to blue-green algal dominance (see section 3.2.4.5).

3.2.4.8 PHYTOPLANKTON AND DISSOLVED OXYGEN

Algae by their photosynthetic activity, do not only influence the pH (see section 3.2.4.7), but also the oxygen concentration in the water.

Oxygen concentration in the Vaal River showed seasonal trends, with peak values coinciding with low water temperatures and maximum phytoplankton biomass. At the Barrage rapid increases in oxygen content during summer periods (e.g. December 1991 and November 1993) were ascribed to increases in chlorophyll-*a* concentrations during the same time (compare Figs 61 & 77), while rapid decreases during winter (e.g. end of July

1992; Fig. 61) were ascribed to decreases in algal and chlorophyll-*a* concentrations (Fig. 77). During 1993 at the Barrage sampling locality no winter increase in oxygen concentration was observed (Fig. 61). A possible explanation can be the very low algal concentration (and chlorophyll-*a*) present during June and July 1993 (Fig. 77). When the algal concentration increased during the beginning of November 1993, an increase in oxygen concentration also occurred (compare Figs 61 and 77). During 1992 and 1993 at the Stilfontein sampling locality an increase in dissolved oxygen concentration was observed during the winter months, while it decreased during the summer period (Fig. 63). Sudden decreases in oxygen concentrations during the winter periods (e.g. July 1992 and August 1993; Fig. 63), were the result of decreases in algal concentrations (Fig. 79), whereafter the algal and oxygen concentrations increased again. Higher oxygen concentrations measured during November 1993, were ascribed to a high algal unit concentration of approximately 49 000 units ml^{-1} (compare Figs 63 and 79). Thus, a great deal of the variation in oxygen concentration in the Vaal River during the present study can possibly be explained by the photosynthetic activity of the phytoplankton as well as the temperature of the water. Similar observations were made by Lukatelich and McComb (1986) in the Peel-Harvey estuary system, i.e. peak surface oxygen coinciding with maximum phytoplankton biomass and minimum values were recorded in late summer, after a collapse of an algal bloom. Roos (1992) demonstrated a statistically significant correlation for the relationship between oxygen and chlorophyll-*a* concentrations in the Vaal River at Balkfontein from 1985 to 1989.

4. CONCLUSIONS

At least 124 species and varieties, belonging to seven major algal groups, were identified in the Vaal River during the present study. Most of the species were present at all four sampling localities, but certain species were found to be unique to one or more sampling locality(s). The greatest diversity of Cyanophyceae species were present at the Barrage, while the greatest diversity of Euglenophyceae species were present at the Balkfontein sampling locality. Blue-green algal blooms in the Vaal River at the Barrage can possibly be ascribed to a combination of environmental variables favourable for growth at and upstream from this sampling locality for example, high inorganic phosphate and nitrogen concentrations, low DIN:DIP ratios, deep levels of light penetration as well as abundance of the Na^+ ion.

At all four sampling localities the phytoplankton community was dominated mainly by diatoms and green algae (which succeeded each other) as well as blue-green algae during certain periods. Euglenophyceae biomass increased from the Barrage downstream to Balkfontein. Representatives of the Cryptophyceae, Chrysophyceae and Dinophyceae were relatively scarce throughout the study period. It is known that an indication of the quality of water can be given by the species composition of the water. The scarceness of certain groups (e.g. Chrysophyceae and Cryptophyceae) and the abundance of other (e.g. Cyanophyceae) show that the Vaal River is a polluted and eutrophic system.

Indications of pollution and eutrophication in the Vaal River are also reflected in chlorophyll-*a* concentrations which are high in comparison with other river systems. It seems as if periods of maximum chlorophyll-*a* concentrations in the Vaal River occurred from January to March (summer) and again from July to November (winter-spring) of each year. The highest average chlorophyll-*a* concentration was present at the Parys sampling locality, probably the result of high available nitrogen and phosphorus concentrations present in the river at, and upstream from, this sampling locality.

One of the most important variables influencing the organisms living in the Vaal River, is discharge. An increase in discharge often results in an increase in turbidity (mainly inorganic suspensoids derived from soils in the catchment) and therefore a decrease in underwater light availability. Even under conditions of high nutrient supply, algal blooms were sometimes possibly prevented by low underwater light conditions as a result of increasing discharge. Very high levels of discharge can be responsible for a complete wash-out of the phytoplankton. Increases in discharge in the Vaal River were frequently the result of releases of water from the Vaal Dam and the Vaal River Barrage. Discharge did not only influence turbidity and underwater light conditions, but also showed a positive correspondence to nutrient (phosphorus and nitrogen) concentrations and a negative correspondence to salinity.

Water temperature is another important environmental variable, because specific organisms have definite ranges of temperature at which maximum growth and reproduction occurs. Diatoms tended to dominate from January to August of each year, while the green algae were dominant from September to December. Dominance of diatoms during summer periods (January to April and December) can be ascribed to blooms of *Melosira granulata*.

Concentrations of unicellular centric diatoms were usually low during the summer periods, but they often dominated during the cold-water winter periods. It therefore seems as if *Melosira granulata* prefers warmer water temperatures than the unicellular centric diatoms. Blue-green algae frequently occurred during the mid and late summer months of each year, especially at the Barrage, Stilfontein and Balkfontein sampling localities. Water temperature also affects solubility of silica and oxygen. Higher silica concentrations were usually present during summer periods when the water temperatures were higher, while lower Si concentrations were present during the winter. The solubility of oxygen is higher in colder water so that higher oxygen concentrations were observed during winter periods.

Besides eutrophication, salinity is the major problem in the Vaal River. The best quality of water is found in the upstream section of the Vaal River, showing a rapid downstream deterioration as the concentration of dissolved salts increases from the Barrage to Balkfontein. Inputs of mining, industrial and treated sewage effluents are probably the major contributors to increasing salinity. The sulphate concentration in the Vaal River is amongst the highest concentrations reported for rivers world wide. High sulphate concentrations in the Vaal River is probably due to high inputs from the tributaries draining the northern part of the catchment that is heavily contaminated by intensive mining and industrial activities. Increases in TDS concentration cause floc formation of suspended material. The flocs settle out, the water becomes clearer and more underwater light becomes available, which could, in turn, result in intensive blooms by problem-causing algal species. During the present study salinity was not the primary variable influencing algal growth, although Dinophyceae more frequently occurred in high salinity water and Cyanophyceae in water with a relatively low salinity.

Results of the present study show that eutrophication (nutrient enrichment) is particularly a problem at the two upstream sampling localities (Barrage and Parys). Higher DIP and TP concentrations were measured at the Barrage and Parys sampling localities, decreasing in a downstream direction. High phosphorus concentrations at the two upstream sampling localities could be a result of urbanisation and industrialisation in the PWV complex, while lower concentrations in the downstream section of the river can be ascribed to removal of DIP by thick mats of water hyacinths and algae, as well as adsorption. High DIN and TN concentrations were also recorded at the Barrage and especially at the Parys sampling localities decreasing downstream to the Stilfontein and Balkfontein sampling localities. High nitrogen concentrations downstream from the Vaal Dam indicate higher organic and inorganic nitrogen loading possibly because of agricultural, mining and industrial effluents that are released into the Vaal River. DIP, TP and TN concentration ranges as well as high TN:TP ratios in the Vaal River fall within the ranges for eutrophic systems. Higher percentage DIP in TP at the Barrage sampling locality than at the Parys, Stilfontein and Balkfontein sampling localities, could also be an indication of more eutrophic conditions present at the Barrage than at the other sampling localities. The highest silica concentrations were also reported at the Barrage, whereafter a decrease occurred downstream to Parys, Stilfontein and Balkfontein. A possible explanation for the high Si concentration at the Barrage sampling locality is that the concentration of diatoms is lower at the Barrage than at the Parys, Stilfontein and Balkfontein sampling localities, thereby not removing similar amounts of silicon from the water. It seems as if Si concentration in the Vaal River is primarily determined by temperature, diatom uptake metabolism as well as settling of suspended solids.

Algal blooms were frequently preceded by high nutrient (DIN, DIP and Si) concentrations, but when the bloom reached its peak, low nutrient concentrations were recorded. Blue-green algae were most probably favoured by low DIN:DIP and TN:TP ratios. A shift from blue-green algae to other algal groups occurred if the TN:TP ratio increased. Increased phosphorus concentrations in the Vaal River (with a resultant decrease in DIN:DIP and thus TN:TP to less than five) will probably cause a shift in the algal assemblages from diatom and green algal dominance to blue-green algal dominance. The Si:DIP ratio also seemed to be important in influencing the occurrence of diatoms, since diatoms usually occurred under conditions of high Si:DIP ratios.

It is possible to reduce the effects of eutrophication (nutrient enrichment) in the river by removing or reducing the sources of nutrient enrichment by means of the implication of standards such as the 1 mg/l phosphate standard proposed by the Department of Water Affairs (1986).

Average pH values at the different sampling localities in the Vaal River were relatively high. pH was influenced to a great extent by the presence of phytoplankton. Blooms of phytoplankton, resulting in extreme rates of photosynthesis, commonly caused high pH values.

Oxygen concentration in the Vaal River was mainly influenced by temperature and phytoplankton biomass. Oxygen concentrations showed seasonal trends with peak values coinciding with low water temperatures and maximum phytoplankton biomass.

5. SUMMARY

Seasonal succession, growth and the development of algal blooms in the Vaal River are is influenced by physical and chemical variables.

5.1 **One of the most important variables influencing the organisms living in the Vaal River, is discharge.** An increase in discharge often results in an increase in turbidity which is responsible for low levels of light penetration, because of the rapid extinction of light penetrating the water. Even under conditions of high nutrient supply, algal blooms were sometimes possibly prevented by low underwater light conditions as a result of increasing discharge. When the discharge decreases, the suspended material settle out and the water becomes clearer (turbidity decrease). More light can penetrate the water to deeper levels, resulting in high algal growth. If flooding occurs, phytoplankton can be completely washed out of the system.

5.2 **An increase in discharge also has a dilution effect on TDS, resulting in lower TDS and major ion concentrations as well as lower electrical conductance of the water.** It can be predicted that the concentration of dinoflagellates will increase with an increase in TDS concentration to more than 900 mg l^{-1}. Since the majority of substances dissolved in most waters are ionic, an increase in TDS concentration causes an increase in conductivity. An increase in TDS concentration or conductivity enhance the flocculation of clay particles, which result in a decrease in turbidity. When the water becomes clearer, more light can penetrate the water to deeper levels, and the underwater light climate becomes more favourable for algal growth.

5.3 **Discharge** did not only influence turbidity, underwater light conditions and TDS, but **also showed a positive correspondence to nutrient** (especially phosphorus and nitrogen) **concentrations.** An increase in discharge (rainfall or releases of water from the Vaal Dam) cause an increase in suspended solids to which nutrients and other substances adsorb. During flood conditions with flow rates of more than 250 $m^3 s^{-1}$, however, a wash-out action can occur, resulting in decreases in nutrient concentrations.

5.4 Discharge is, however, not the only factor influencing nutrient concentrations. **The disposal of industrial, mining and domestic wastes increases the dissolved phosphorus and nitrogen concentration in the water** (eutrophication) which results in blooms by algal species causing various problems (with serious financial implications).

5.5 **The specific algal species which form blooms depend, to a great extent, on the temperature of the water.** During the summer periods blue-green algae (e.g. *Oscillatoria simplicissima*; filamentous) and *Melosira granulata* (filamentous centric diatom) usually dominated, while unicellular centric diatoms and green algal representatives (especially *Chlamydomonas incerta*) usually dominanted and succeeded each other during the winter. It can, therefore, be concluded that *Melosira granulata* prefers warmer water temperatures than unicellular centric diatoms. Maximum chlorophyll-*a* concentrations were recorded when the water temperature started to increase after the winter minimum was reached (July to November) as well as during periods when high water temperatures were recorded (January to March), while low chlorophyll-*a* concentrations usually occurred during May and June of each year.

5.6 **Temperature also influenced both silicon and oxygen concentrations in the water.** An increase in water temperature leads to a decrease in the solubility of oxygen and an increase in the solubility of silicon. Lower oxygen and higher silicon concentrations were therefore recorded in summer and higher oxygen and lower silicon concentrations in winter.

5.7 **Diatom concentrations also have a major influence on silicon concentrations in the water.** Increases in diatom numbers usually results in decreases in silicon concentrations and *vice versa*. Silicon concentration in water is most probably controlled by diatoms and water temperature.

5.8 **A combination of high temperatures and nutrient (DIN and DIP) concentrations as well as low DIN:DIP and TN:TP ratios (less than 5) apparently favour blue-green algal growth.** Sharp increases in TN:TP ratios were accompanied by a shift from blue-green algal groups as the dominants to other algal groups such as diatoms or green algae. Diatoms are favoured by high Si:DIP ratios while green algae usually dominated under conditions of low DIN:DIP and Si:DIP ratios.

5.9 **During peaks in algal biomass, nutrient depletion often occurred resulting in low concentrations of phosphorus and nitrogen.** Blooms of diatoms were also responsible for low silicon concentrations. Sudden decreases in silicon concentration could frequently be ascribed to sudden increases in diatom numbers. A combination of grazing, silica depletion and sedimentation usually brings a diatom bloom to an end.

5.10 **Increases in pH were frequently recorded during periods when an increase in chlorophyll-**a **concentration and algal biomass occurred.** Extreme rates of photosynthesis, whether natural or as a result of eutrophication, can cause very high pH condition because of the uptake of CO_2.

5.11 **Photosynthesis of aquatic plants can also cause an increase in oxygen concentration,** because oxygen is released during photosynthesis. During the summer, oxygen concentrations are usually low because oxygen is less soluble in warm water than in cold water. Sudden increases in oxygen concentration during summer periods can be ascribed to increases in algal biomass and chlorophyll-*a* concentrations, while rapid decreases during the winter could be ascribed to decreases in algal biomass and chlorophyll-*a* concentration. A great deal of the variation in oxygen concentration during the present study can be explained by the photosynthetic activity of phytoplankton as well as temperature of the water.

6. ACKNOWLEDGMENTS

The assistance of the following persons and institutions in the execution of the sampling programme is gratefully acknowledged: Mr MC Steynberg of Rand Water, Vereeniging, Mr W du Preez of the Parys Municipality, Parys, Mrs M Krüger, Mr J Pietersen and Mr K Morgan of the Western Transvaal Regional Water Company, Stilfontein, and Mrs N Basson and Mr H du Preez of Goldfield Water, Balkfontein, Bothaville. Financial support of the Water Research Commission made the study possible.

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CHAPTER 4: THE EFFECT OF INCREASED CONCENTRATIONS OF TOTAL DISSOLVED SALTS ON ALGAL SPECIES FROM THE VAAL RIVER

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1. INTRODUCTION

Salinity is the chemical term for the ionic composition of fresh waters (Wetzel, 1983). Salinisation, however, is the process by which the concentration of total dissolved salts (TDS) in inland waters is increased (Williams & Noble, 1984). Salinity represents an important environmental variable as it is generally correlated with levels of primary production (Likens, 1975).

Salinity is a world-wide problem and the Department of Water Affairs and Forestry pointed out that the quality of many water sources in the RSA is declining, primarily because of salinisation (DWA, 1986). There are two major activities of man that accelerate the input of salts to inland waters from essentially natural sources: irrigation and dry-land farming following on the removal of natural vegetation. Salts added directly from non-natural sources include those from sewage effluents, industrial discharges, mining drainage water and atmospheric pollutants (Williams & Noble, 1984). The salinity caused by human disturbances, as the above, is often referred to as secondary salinisation.

The problem of salinisation of the Vaal River already became apparent in 1949 (DWA, 1986). In the middle Vaal region, the Vaal River and tributaries draining the southern Witwatersrand and West Rand areas are dominated by high TDS sources, and the mineral water quality conditions are the worst of the entire Vaal River catchment (Van Vliet & Nell, 1986). According to Braune & Rogers (1987), irrigation return-flow is making a major contribution to increased salinity in the Vaal River, particularly below the Barrage. Apparently another contributor to the TDS concentration is the precipitation of atmospheric pollutants of which sulphate is a major constituent.

Grobler *et al.* (1986) stated that increased salinity levels can affect downstream users and could also result in decreased turbidity in the greater part of the Vaal River. Clearer water due to salinisation and the high nutrient supply (eutrophication), will probably result in more intensive algal blooms, because deeper light penetration could result in increased primary productivity. Increased primary productivity is already reflected in the extensive stands of rooted underwater macrophytes in sections of the lower Vaal River (Grobler *et al.*, 1986).

According to Brock (1985) high salinity conditions could affect autotrophic macrophytic communities in the following way. As the salinity levels increase, the diversity of macrophyte species declines. Little is known about the response of algal communities to increased salinity in running waters. As algae is a major contributor to problems in the Vaal River, especially during water purification for household and industrial usage, it is important to investigate the effect of increased TDS on freshwater algal species in order to provide specific information in this regard.

For this report an historical overview on the change in TDS concentration was done at two study points in the Vaal River, namely Stilfontein and Balkfontein, for the period 1984 to 1993 at Stilfontein and 1985 to 1993 at Balkfontein. In this way annual, seasonal and spatial trends were determined and the composition of the major ionic constituents to the total dissolved salts were determined.

The average TDS values determined in the historical overview were then used to do algal growth and carbon assimilation rates experiments. The growth experiments were done with *Cyclotella meneghiniana* (Bacillariophyceae), *Microcycstis aeruginosa* (Cyanophyceae) and *Monoraphidium circinale* (Chlorophyceae). These algal species were chosen as they frequently occur in the Vaal River and could cause problems during water purification.

2. MATERIAL AND METHODS

For the determination of annual, seasonal and spatial trends in TDS concentrations, as well as the ionic composition of the TDS in the Vaal River, average weekly data received from Stilfontein and Balkfontein were used to calculate monthly and annual averages and the data were then put into graphs.

The growth experiments were done with uni-algal cultures. Algae used as experimental material were *Cyclotella meneghiniana* Kutz. (diatom), *Monoraphidium circinale* (Nyg.) Nyg. (green alga) and *Microcystis aeruginosa* Kutz. (blue-green alga). The first two species were isolated from the Vaal River, while the last species was isolated from the Hartebeespoort Dam.

The three algal species were introduced to different laboratory salt concentrations as well as different Vaal River salt concentrations in a GBG-11 growth medium (Krüger, 1978) for a period of 14 days. The laboratory salts mix consisted of calcium chloride ($CaCl_2.2H_2O$) and sodium sulphate ($Na_2SO_4.10H_2O$). The total salt concentration added ranged between 100 and 2000 mg l^{-1}. Turbidity was used as a measure of growth. At day 8 of the experiment, water samples were taken and chlorophyll-*a* was measured by using the pigment extraction method (Sartory, 1982) as well as primary productivity by the ^{14}C-uptake method. ^{14}C-uptake was done by the acid-bubbling method as described by Schindler *et al.* (1972).

3. RESULTS AND DISCUSSION

3.1. HISTORICAL OVERVIEW

3.1.1. TOTAL DISSOLVED SALTS AT BALKFONTEIN FOR THE PERIOD 1985 - 1993

During the study period 1985 to 1993 at Balkfontein , the average annual TDS concentration ranged between 368.5 mg l^{-1} and 635 mg l^{-1} with a mean of 472.1 mg l^{-1} (Fig. 1). A general increase of about 29 mg $l^{-1}.a^{-1}$ was observed since 1985 to 1993 (Fig. 1).

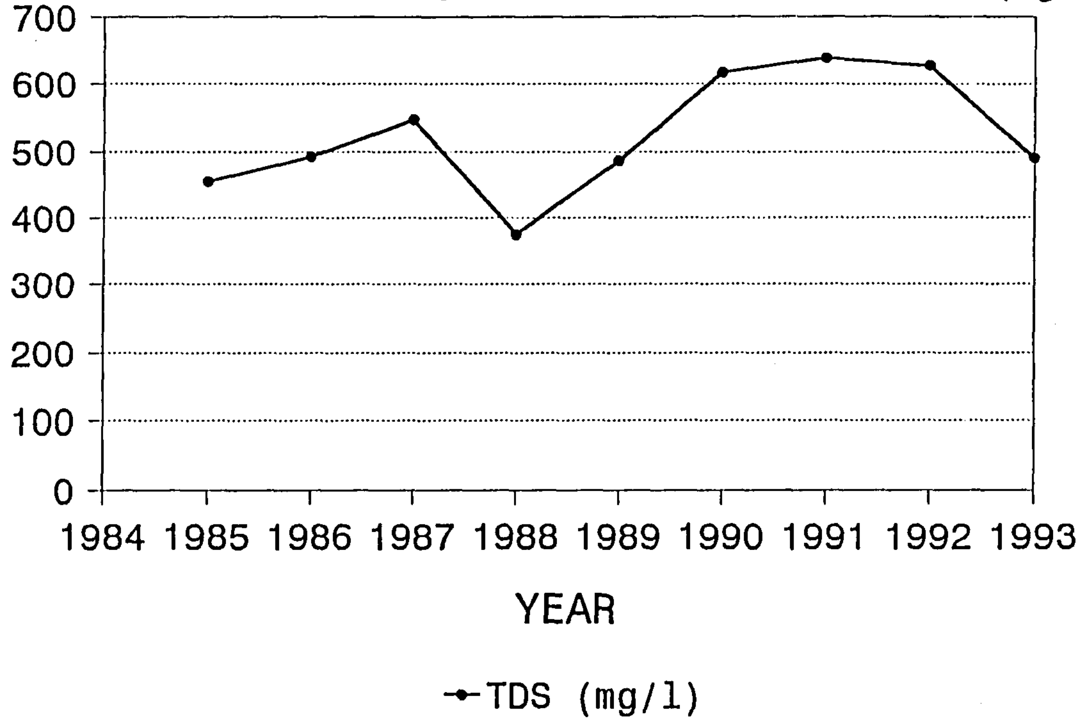


FIGURE 1: The average annual TDS concentration (mg l^{-1}) from 1985-1993 at Balkfontein (n = 52 or 53)

The average monthly TDS concentration for 1985 at Balkfontein (Fig. 2a) shows that the highest TDS concentrations occurred during the winter to spring period with the highest average TDS concentration in July (655 mg l^{-1}). The lowest average TDS concentration for 1985 occurred in November (257.3 mg l^{-1}) and the mean TDS concentration for 1985 was 454.3 mg l^{-1}. Figure 2b shows the average monthly ionic composition for 1985. It is evident that sulphate (SO_4^{2-}) was the major constituent followed by calcium (Ca^{2+}), sodium (Na^+), chloride (Cl^-), and magnesium (Mg^{2+}). The highest ionic composition occurred during the winter to spring period which is in accordance with the average monthly TDS values.

Fig. 3a shows the average monthly TDS concentration at Balkfontein for 1986. Here it is evident that the highest TDS concentration occurred during May with an average value of 650.5 mg l^{-1}. The lowest TDS concentration occurred in November (297.5 mg l^{-1}). The major ionic components consisted of sulphate, calcium, sodium, chloride and magnesium (Fig. 3b).

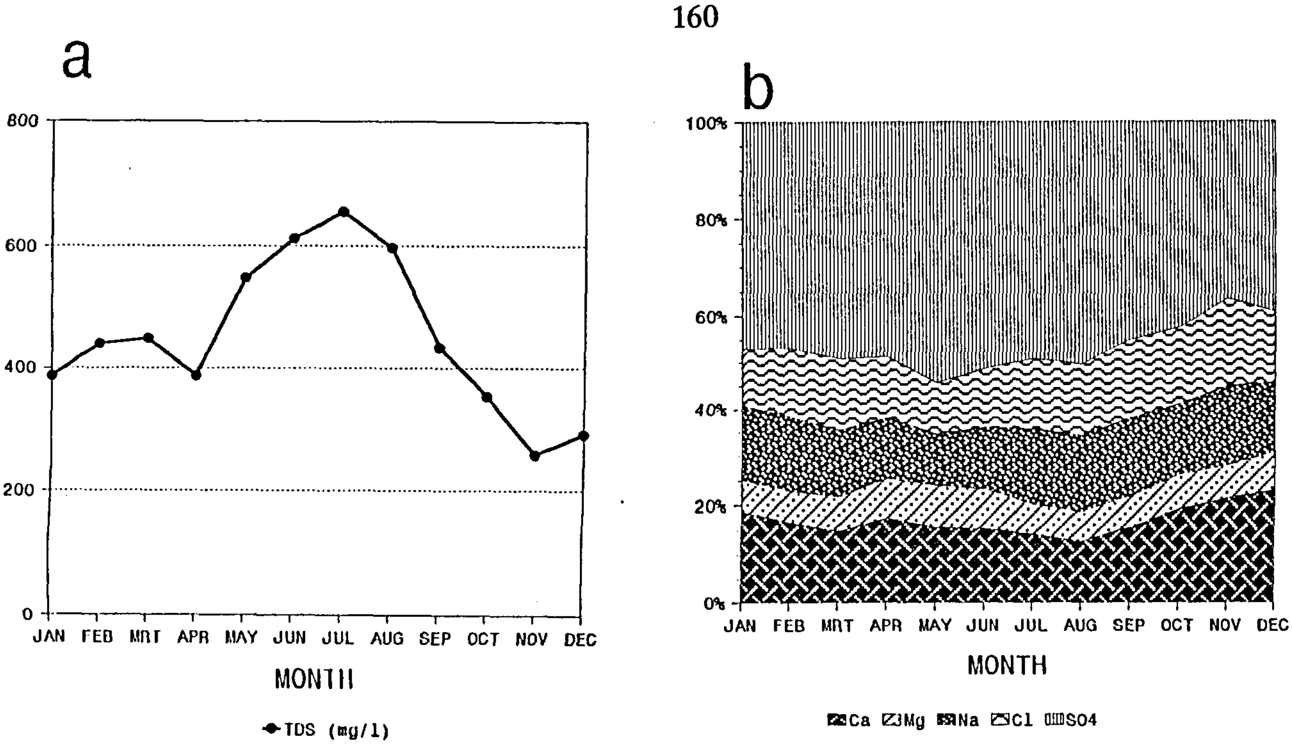


FIGURE 2: The average monthly TDS concentration for 1985 at Balkfontein (a). The average monthly ionic composition for 1985 at Balkfontein (b).

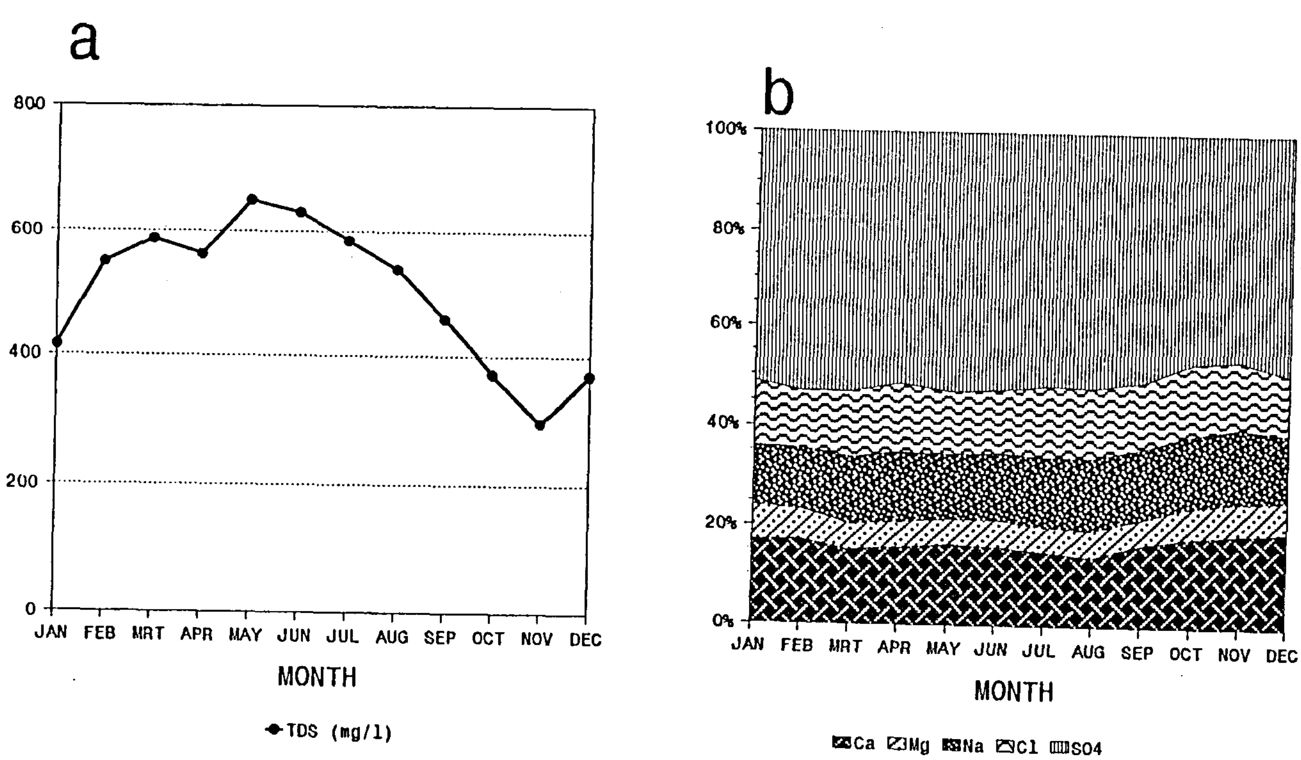


FIGURE 3: The average monthly TDS concentration for 1986 at Balkfontein (a). The average monthly ionic composition for 1986 at Balkfontein (b).

For 1987 (Fig. 4a) the highest average TDS concentration was recorded in June (828.6 mg l^{-1}) and the lowest TDS concentration in November (319.8 mg l^{-1}). The mean monthly TDS concentration for 1987 was 547.4 mg l^{-1}. The major ionic components consisted of sulphate, the major contributor to the total dissolved salts, followed in decreasing order by calcium, sodium, chloride and magnesium. The highest individual ionic value for each ion occurred during the winter period with the highest average monthly values recorded in June (Fig. 4b).

Fig. 5a shows the monthly TDS concentration for 1988. It is evident that the highest TDS concentration occurred during September (737.8 mg l^{-1}). The lowest average monthly TDS concentration was recorded during March (205 mg l^{-1}) and this correlates with the flood period of 1988. If one looks at the average monthly ionic composition for 1988 (Fig. 5b), it is evident that sulphate was once again the major contributor to the composition of the TDS salts in the river. The highest individual ionic concentration was recorded in September and the lowest concentration during March. Sulphate is followed in decreasing order by calcium, sodium, chloride and magnesium.

In Fig. 6a the average monthly TDS concentration for 1989 is illustrated. The highest average TDS concentration recorded for 1989 occurred in October (852.2 mg l^{-1}) and the lowest average TDS in February (197.3 mg l^{-1}). The major ionic composition consisted of sulphate, calcium, sodium, chloride and magnesium in decreasing order (Fig. 6b). The average total dissolved salts as well as the concentrations of the major ionic constituents for February 1989 was very low and could be attributed to the floods of February 1989.

The average monthly TDS concentration for 1990 was at its highest during the summer period and an average monthly TDS concentration of 911.7 mg l^{-1} was recorded (Fig. 7a). The lowest monthly average was recorded in April (491.4 mg l^{-1}), which is much higher than the lowest monthly averages recorded for the study period (Fig. 7a). The ionic components for 1990 consisted of sulphate, chloride, sodium, calcium and magnesium in decreasing concentration order (Fig. 7b).

The average monthly TDS concentration for 1991 showed a general increase from February to October with the highest average TDS concentration recorded in October (814.5 mg l^{-1}, Fig. 8a). The lowest TDS concentration was recorded in February (266.6 mg l^{-1}) and the average annual TDS concentration for 1991 was 639.1 mg l^{-1} (Fig. 8a and Fig. 1). Fig. 8b shows the major ionic constituents for 1991 and here it is evident that sulphate was the major contributor to the total dissolved salts followed in decreasing order by chloride, sodium, calcium and magnesium.

For 1992 (Fig. 9a) the highest average monthly TDS concentration was recorded in February (855 mg l^{-1}) and the lowest average TDS concentration in December (306 mg l^{-1}). The annual average TDS concentration for 1992 was 627 mg l^{-1} (Figs. 1 and 9a). The major ionic contributors was sulphate followed, in decreasing order, by sodium, calcium, chloride, magnesium, potasium and fluoride.

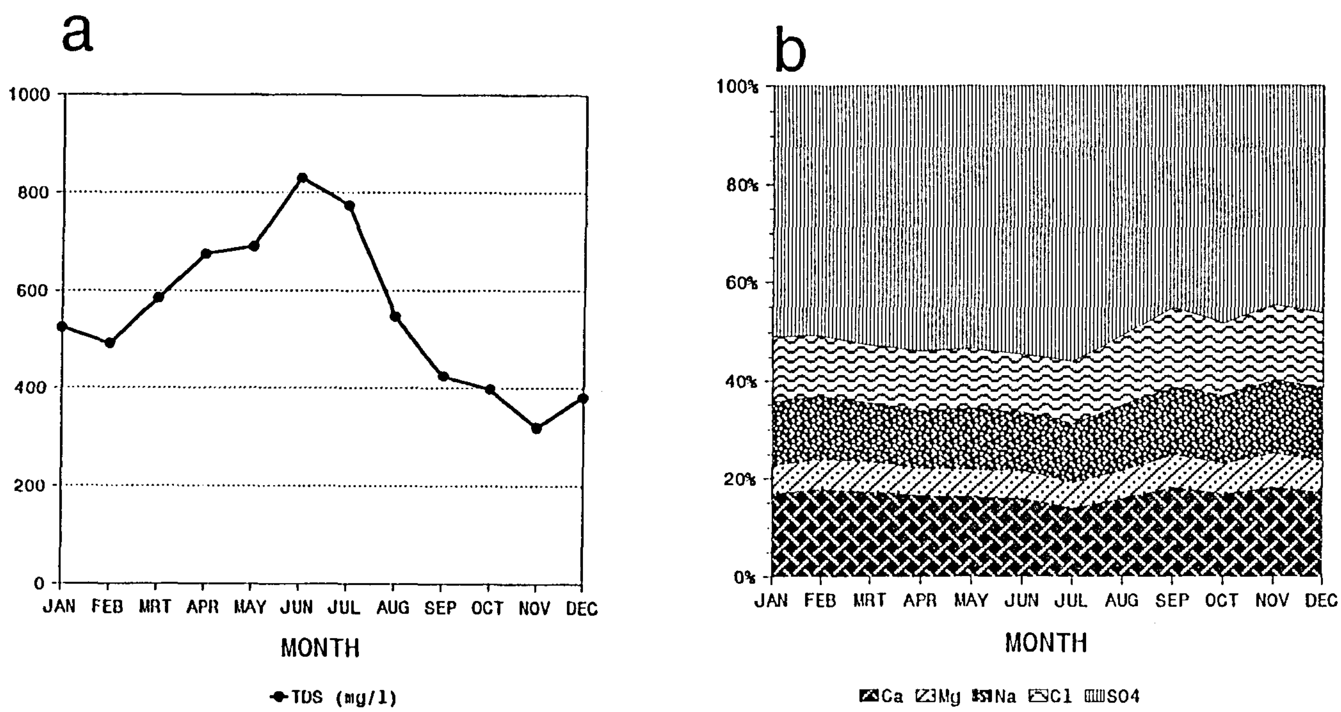


FIGURE 4: The average monthly TDS concentration for 1987 at Balkfontein (a). The average monthly ionic composition for 1987 at Balkfontein (b).

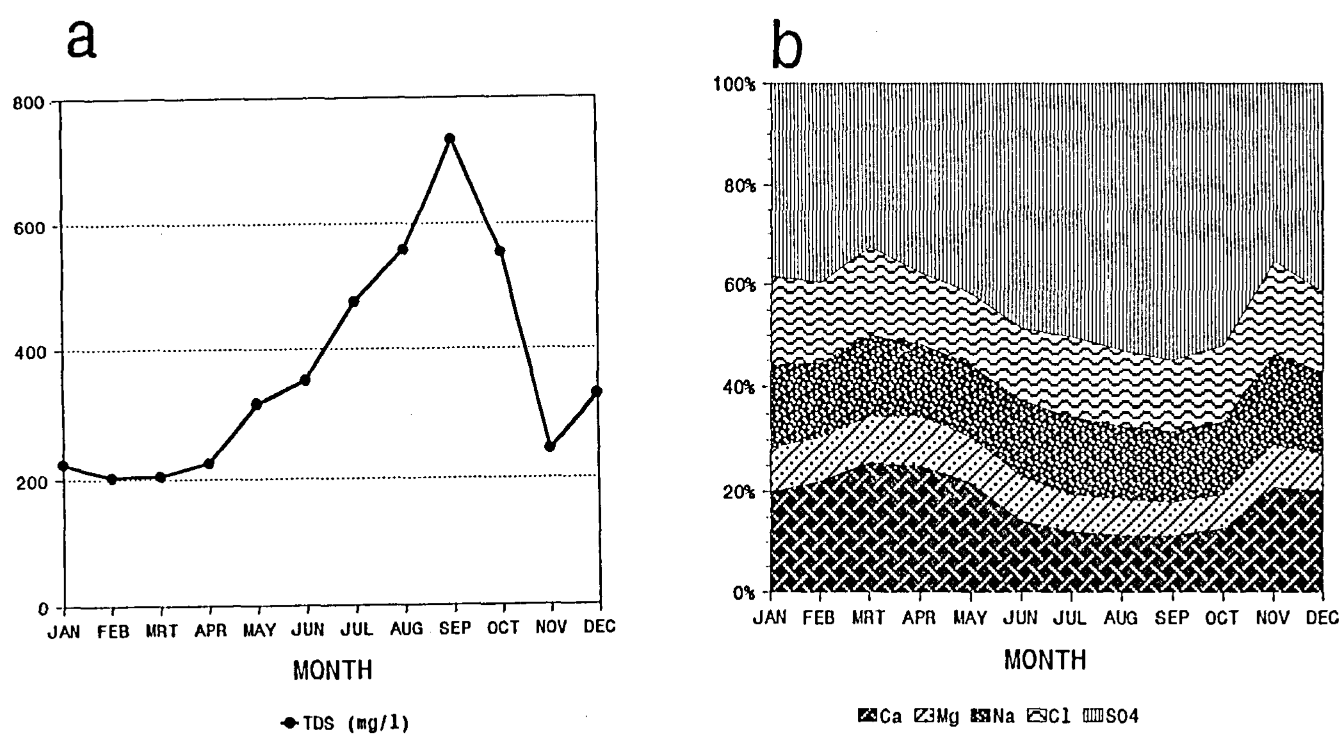


FIGURE 5: The average monthly TDS concentration for 1988 at Balkfontein (a). The average monthly ionic composition for 1988 at Balkfontein (b).

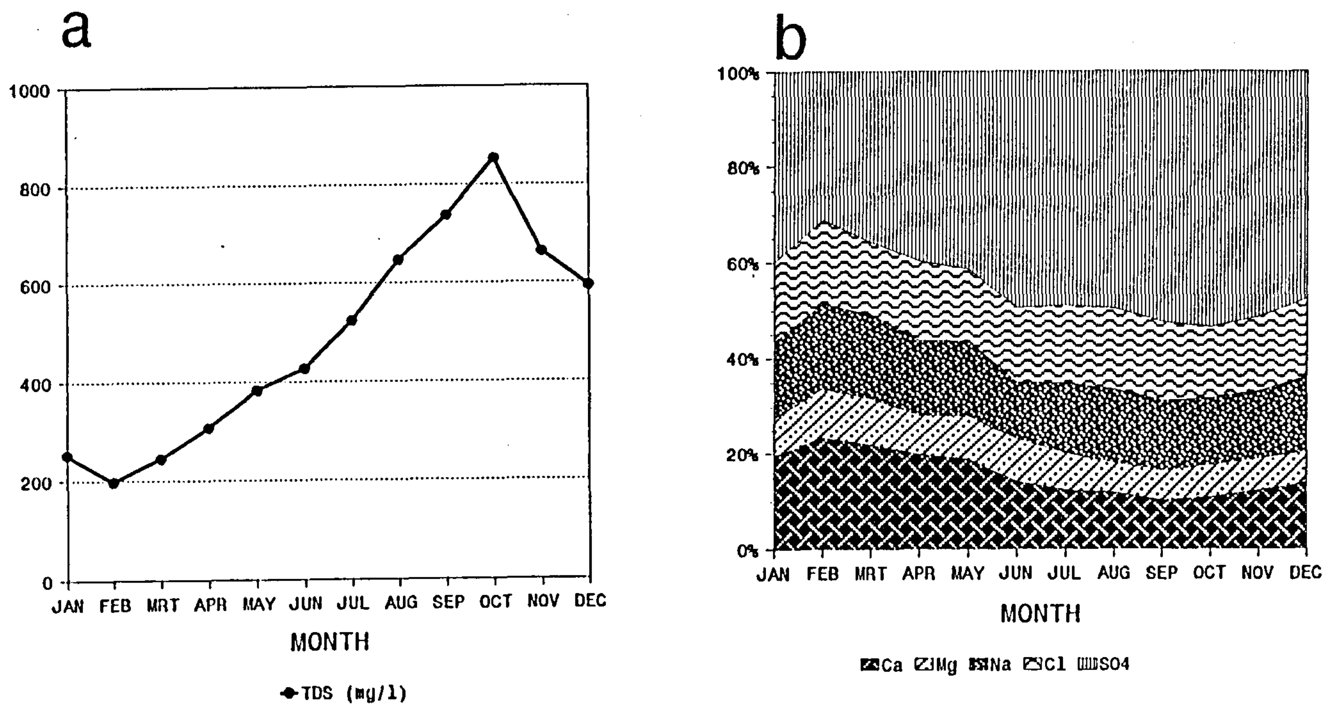


FIGURE 6: The average monthly TDS concentration for 1989 at Balkfontein (a). The average monthly ionic composition for 1989 at Balkfontein (b).

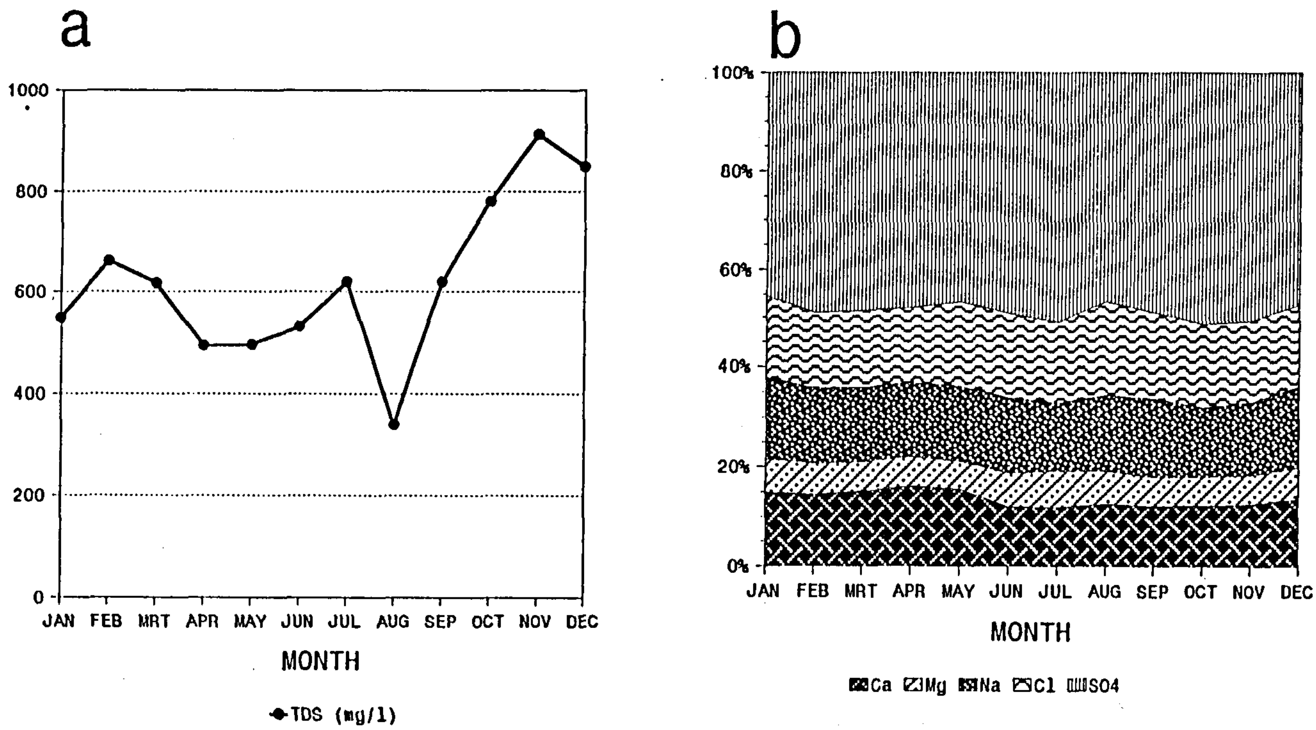
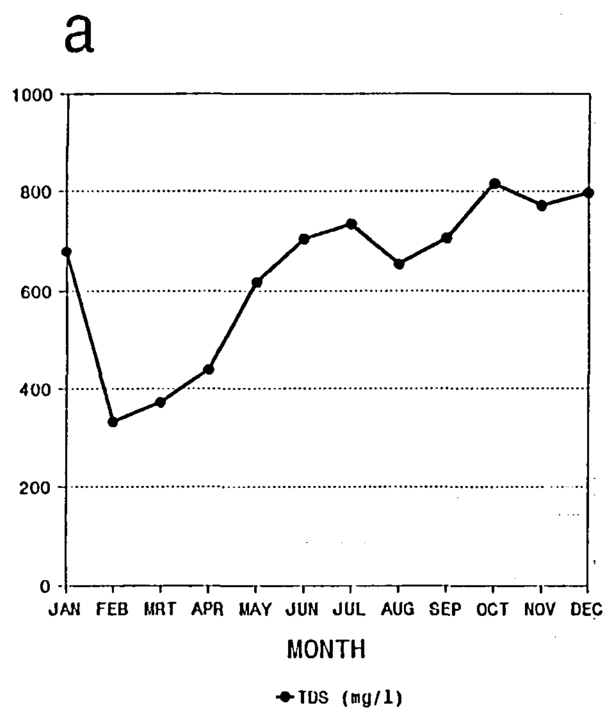


FIGURE 7: The average monthly TDS concentration for 1990 at Balkfontein (a). The average monthly ionic composition for 1990 at Balkfontein (b).



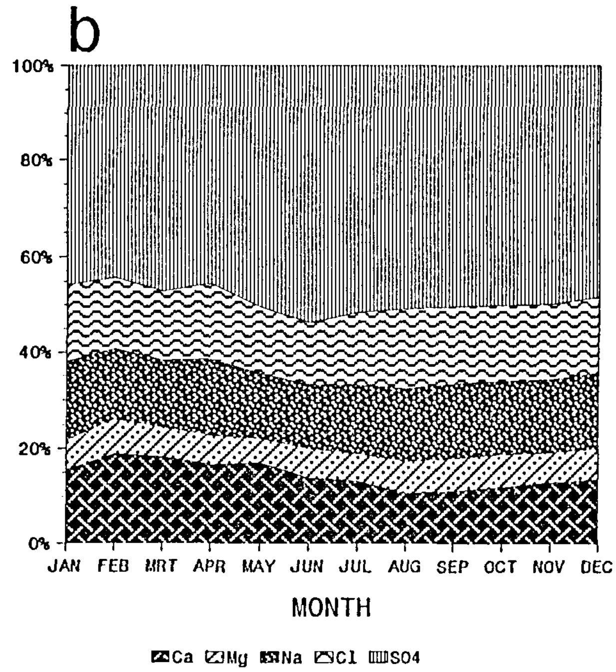
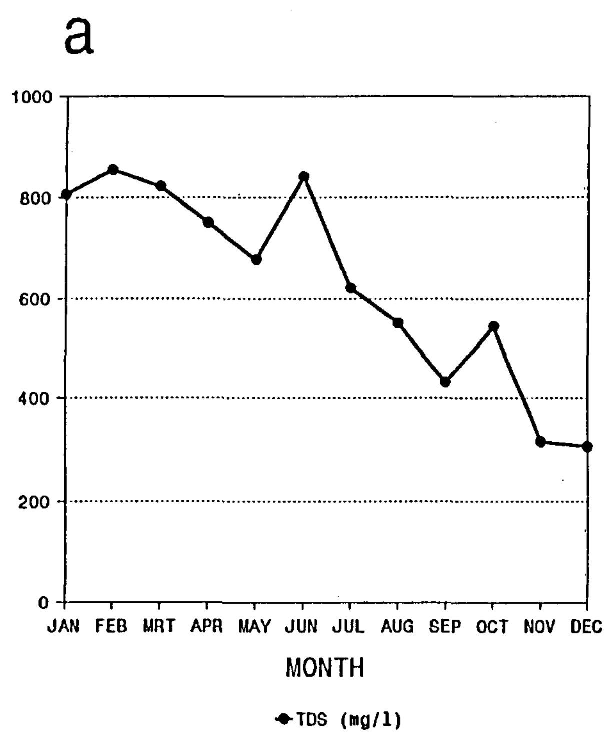


FIGURE 8: The average monthly TDS concentration for 1991 at Balkfontein (a). The average monthly ionic composition for 1990 at Balkfontein (b).



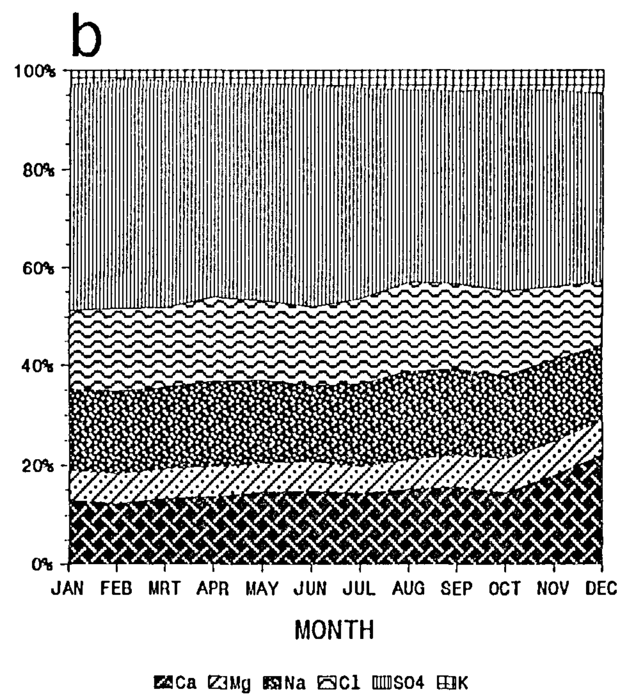
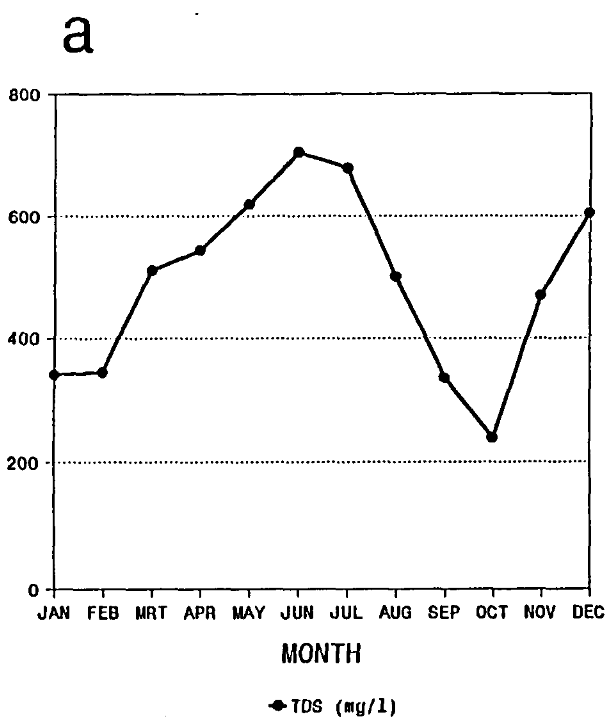


FIGURE 9: The average monthly TDS concentration for 1992 at Balkfontein (a). The average monthly ionic composition for 1992 at Balkfontein (b).

In 1993 (Fig. 10a) the highest monthly TDS concentration was recorded in June (703 mg l^{-1}) and the lowest monthly average in October (238 mg l^{-1}). The ionic contributors were sulphate followed by sodium, chloride, calcium, magnesium and potasium.



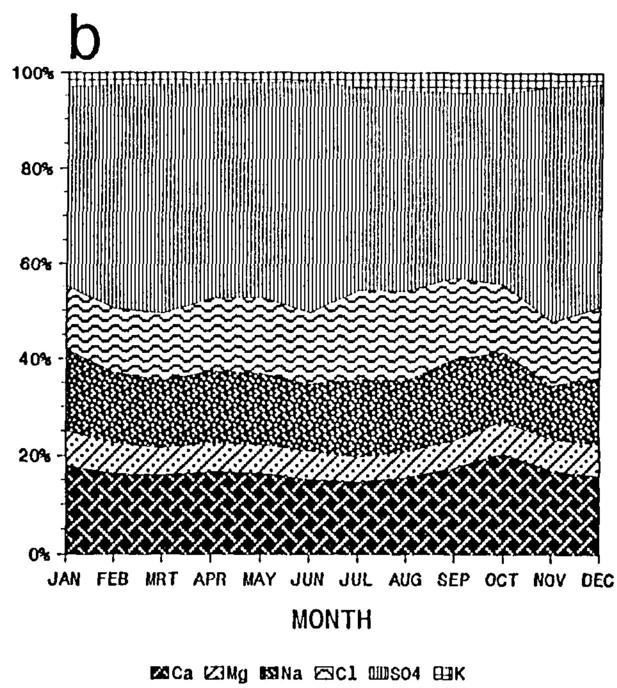


FIGURE 10: The average monthly TDS concentration for 1993 at Balkfontein (a). The average monthly ionic composition for 1993 at Balkfontein (b).

3.1.2 TOTAL DISSOLVED SALTS AT STILFONTEIN FOR THE PERIOD 1984 - 1993

During the study period 1984 to 1993 at Stilfontein the average annual TDS concentration ranged between 419 mg l^{-1} and 632 mg l^{-1} with a mean annual value of 504.2 mg l^{-1} (Fig. 11). At Stilfontein a general increase of 21.3 mg $l^{-1}.a^{-1}$ was observed for the study period (Fig. 11).

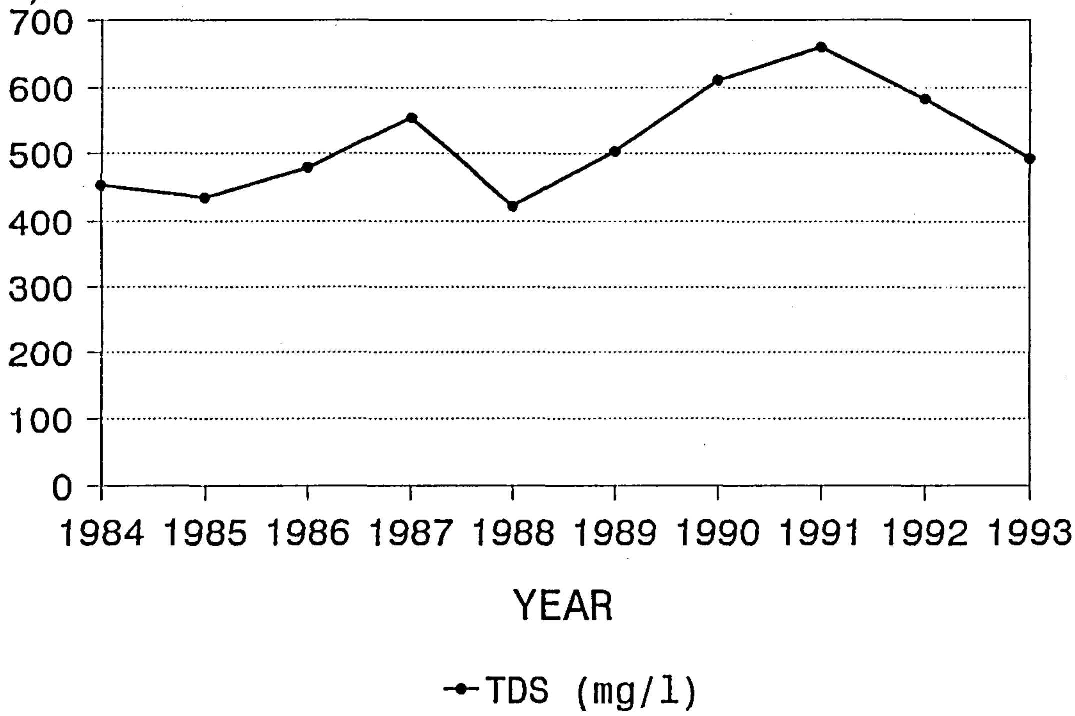


FIGURE 11: The average annual TDS concentration (mg l^{-1}) from 1984-1993 at Stilfontein (n = 52 or 53)

Inspection of the average monthly TDS values for each individual year, the following was found: For 1984 the highest average monthly TDS concentration was recorded in March (708.17 mg l^{-1}) and the lowest average monthly TDS was recorded during October (294.3 mg l^{-1}, Fig. 12a). The major ionic components for 1984 consisted of chloride, calcium, sodium and magnesium in decreasing sequence for each individual month (Fig. 12b).

Fig. 13a shows the average monthly TDS values for 1985. Here it is evident that the highest average monthly TDS concentration was recorded during the winter period, i.e. 628.5 mg l^{-1} was recorded in July. The lowest average monthly TDS concentration was 276.8 mg l^{-1} which was recorded in October (Fig. 13a). The major ionic components consisted of sulphate, the major contributor to the total dissolved salts in the river, followed by chloride, calcium, sodium and magnesium (Fig. 13b).

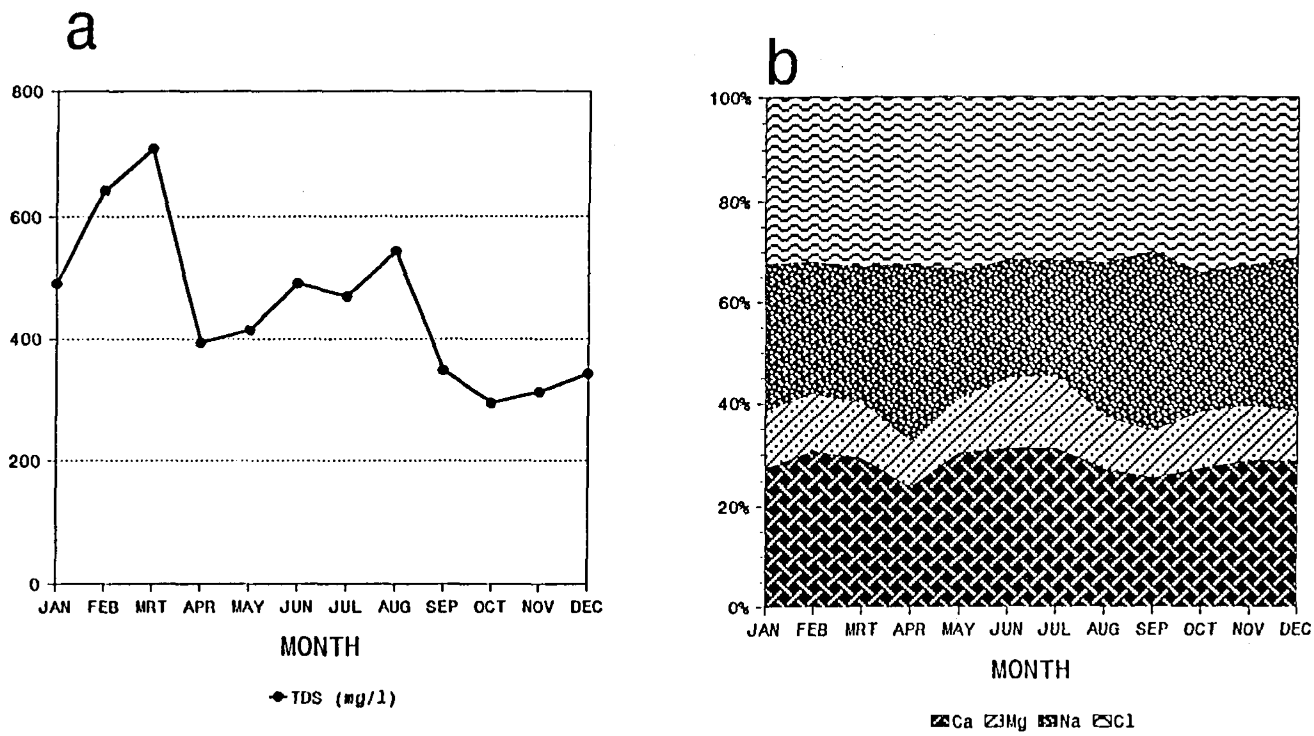


FIGURE 12: The average monthly TDS concentration for 1984 at Stilfontein (a). The average monthly ionic composition for 1984 at Stilfontein (b).

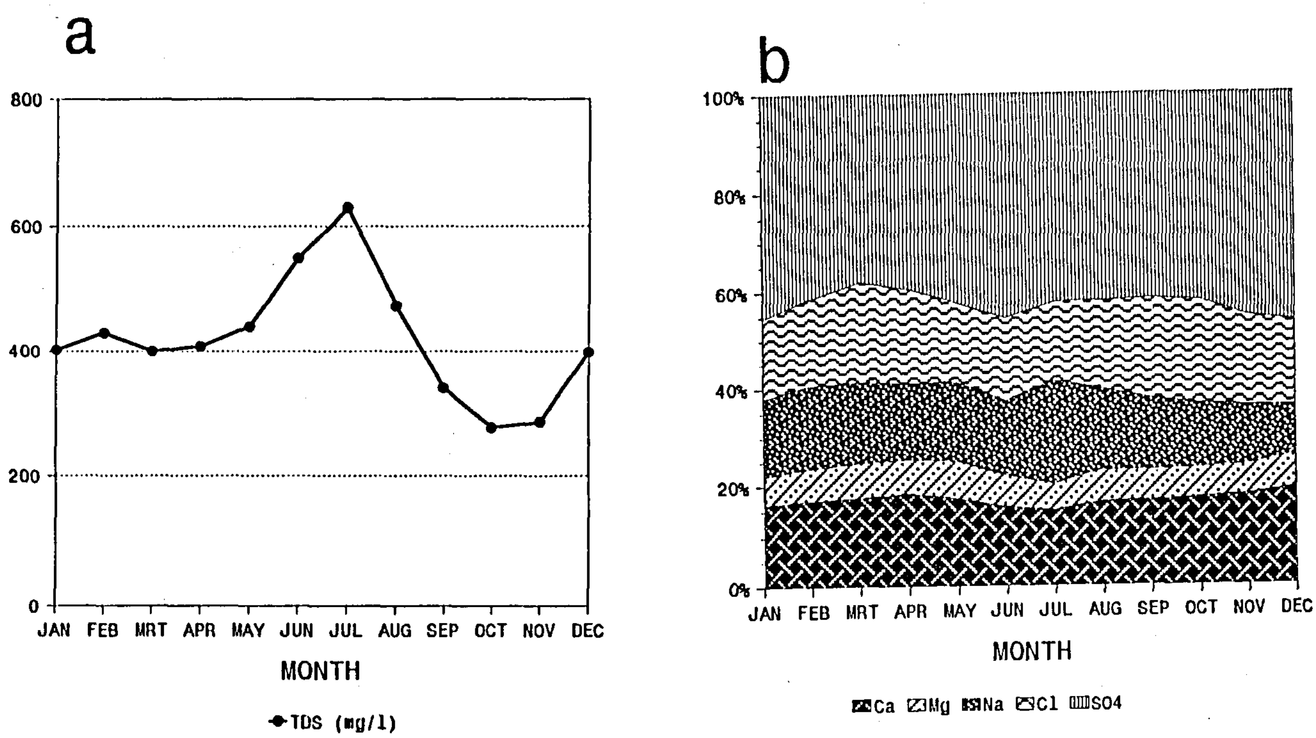


FIGURE 13: The average monthly TDS concentration for 1985 at Stilfontein (a). The average monthly ionic composition for 1985 at Stilfontein (b).

For 1986 (Fig. 14a) the average monthly TDS concentration was high for the period March to July, with the highest average value recorded in April (590.6 mg l^{-1}). The lowest monthly average was recorded in November (326.8 mg l^{-1}). Sulphate was once again the major ionic contributor to the TDS followed by chloride, calcium, sodium and magnesium (Fig. 14b).

Fig. 15a shows the average monthly TDS concentrations for 1987. Here it is evident that the highest total dissolved salt concentration was 794.5 mg l^{-1} which was recorded during June. The lowest TDS concentration for 1987 was 371.4 mg l^{-1} which was recorded in September (Fig. 15a). The ionic components consisted of sulphate, calcium, chloride, sodium and magnesium in decreasing order (Fig. 15b).

For 1988 (Fig. 16a) the lowest average monthly TDS concentration was 218.4 mg l^{-1} (March 1988) while the highest average TDS concentration was recorded in September (772.4 mg l^{-1}). The ionic components for 1988 consisted of sulphate, calcium, sodium, chloride and magnesium (Fig. 16b).

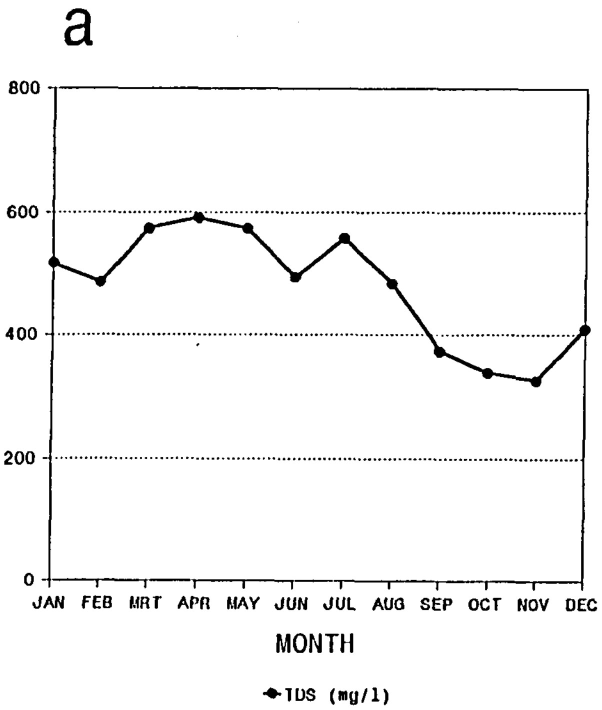
Fig. 17a shows the average monthly TDS concentration for 1989. The highest TDS value was recorded in September (769.25 mg l^{-1}) and the lowest average monthly TDS value in February (256.9 mg l^{-1}). This correlates with the floods of February 1989. The average monthly ionic composition for 1989 was also at its lowest during February while the major ionic contributors to the total dissolved salts were sulphate, chloride, sodium, calcium and magnesium in decreasing order (Fig. 17b).

The TDS concentration for 1990 was high for most of the year with the highest average value recorded during the spring period (Fig. 18a). The highest monthly TDS value was recorded in October (711.8 mg l^{-1}) and the lowest average TDS concentration was 467.6 mg l^{-1}, recorded in July (Fig. 18a). Sulphate, sodium, chloride, calcium and magnesium were the main contributors to the total dissolved salts, with sulphate as the major contributor (Fig. 18b).

For 1991 (Fig. 19a) the highest average monthly TDS concentration was 816 mg l^{-1} (October 1991) and the lowest average TDS concentration was recorded in February (421 mg l^{-1}). The ionic composition concisted of sulphate, chloride, sodium, calcium, magnesium and potasium (Fig. 19b).

Fig. 20a shows the average monthly TDS concentration for 1992. The lowest monthly TDS concentration was recorded in December (247 mg l^{-1}) and the highest value was recorded during January (732 mg l^{-1}). The ionic composition consisted of sulphate the major ionic contributor followed in decreasing order by calcium, sodium, chloride, magnesium and potasium (Fig. 20b).

The average monthly TDS concentration for 1993 (Fig. 21a) showed that the highest monthly average was in June (713 mg l^{-1}) and the lowest average concentration recorded in October (246 mg l^{-1}). The ionic constituents consisted of sulphate, sodium, chloride, calcium, magnesium and potasium (Fig. 21b).



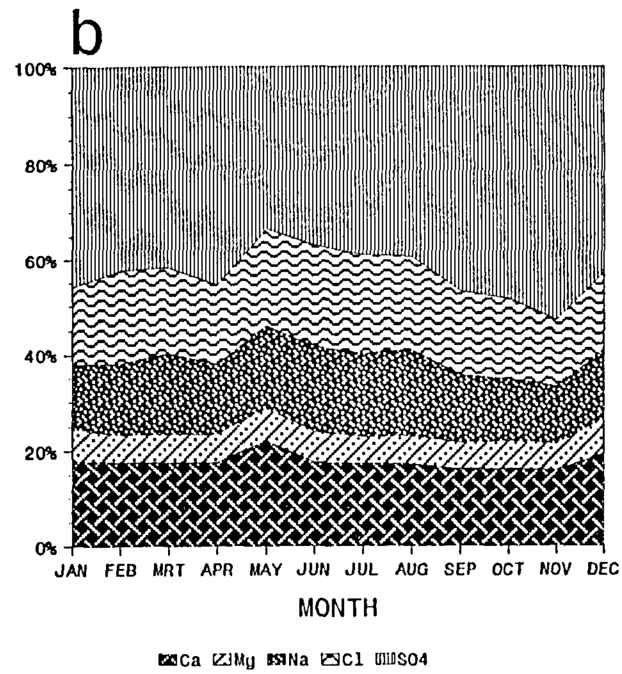
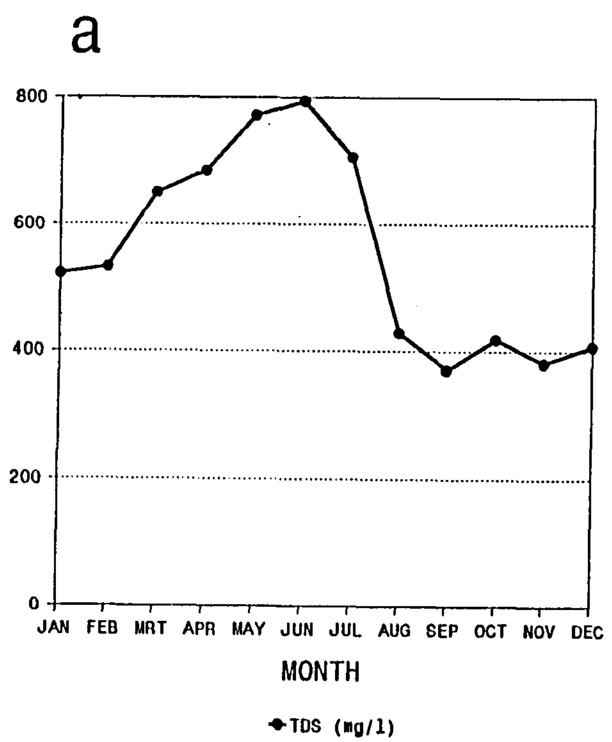


FIGURE 14: The average monthly TDS concentration for 1986 at Stilfontein (a). The average monthly ionic composition for 1986 at Stilfontein (b).



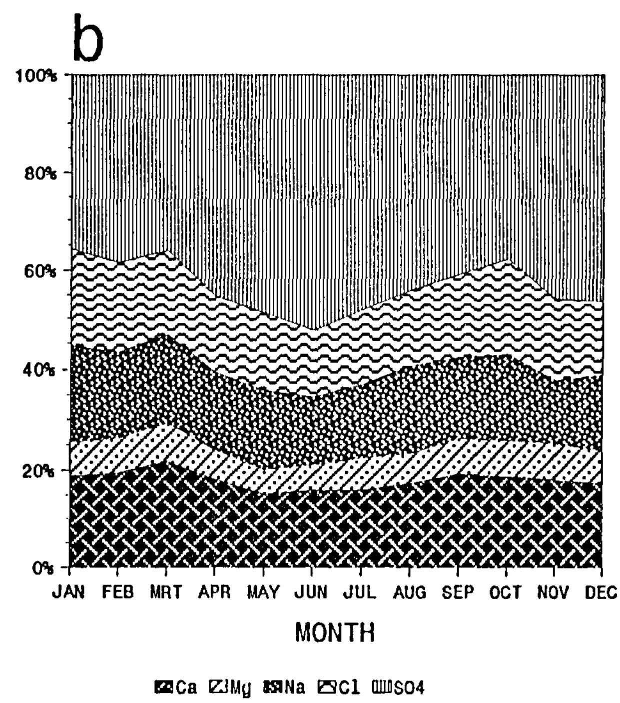


FIGURE 15: The average monthly TDS concentration for 1987 at Stilfontein (a). The average monthly ionic composition for 1987 at Stilfontein (b).

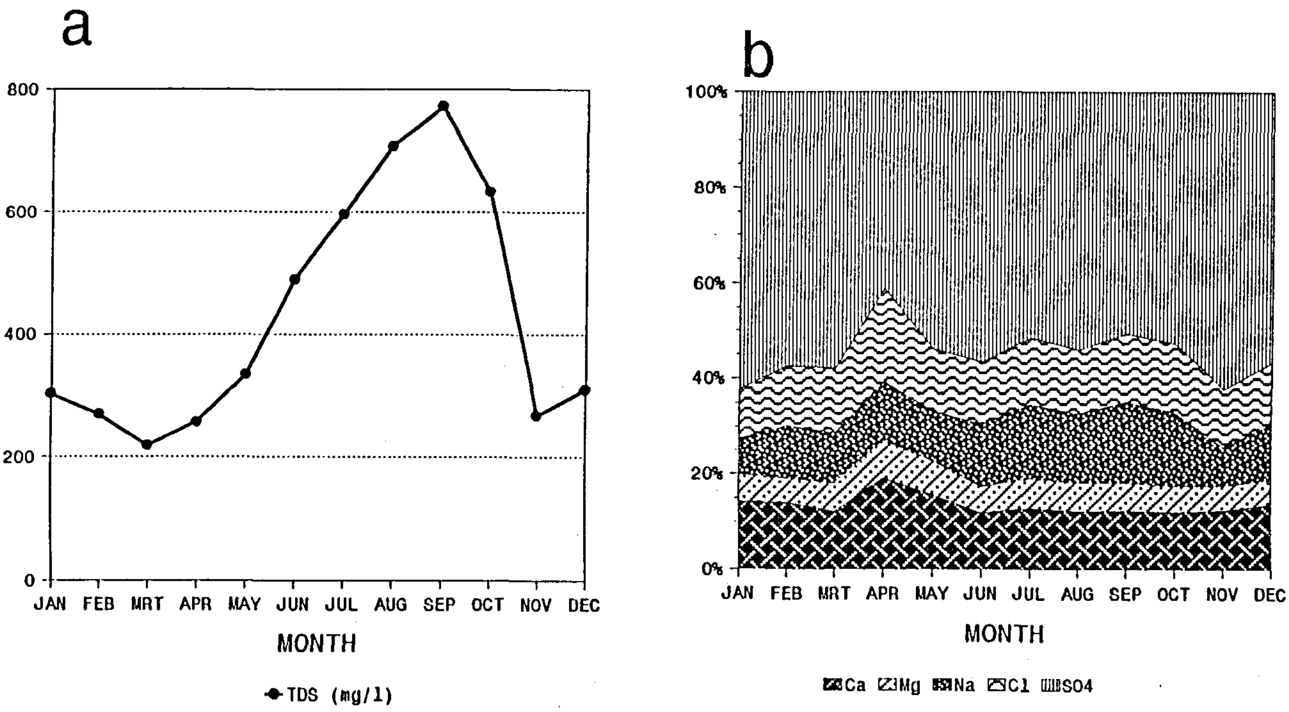


FIGURE 16: The average monthly TDS concentration for 1988 at Stilfontein (a). The average monthly ionic composition for 1988 at Stilfontein (b).

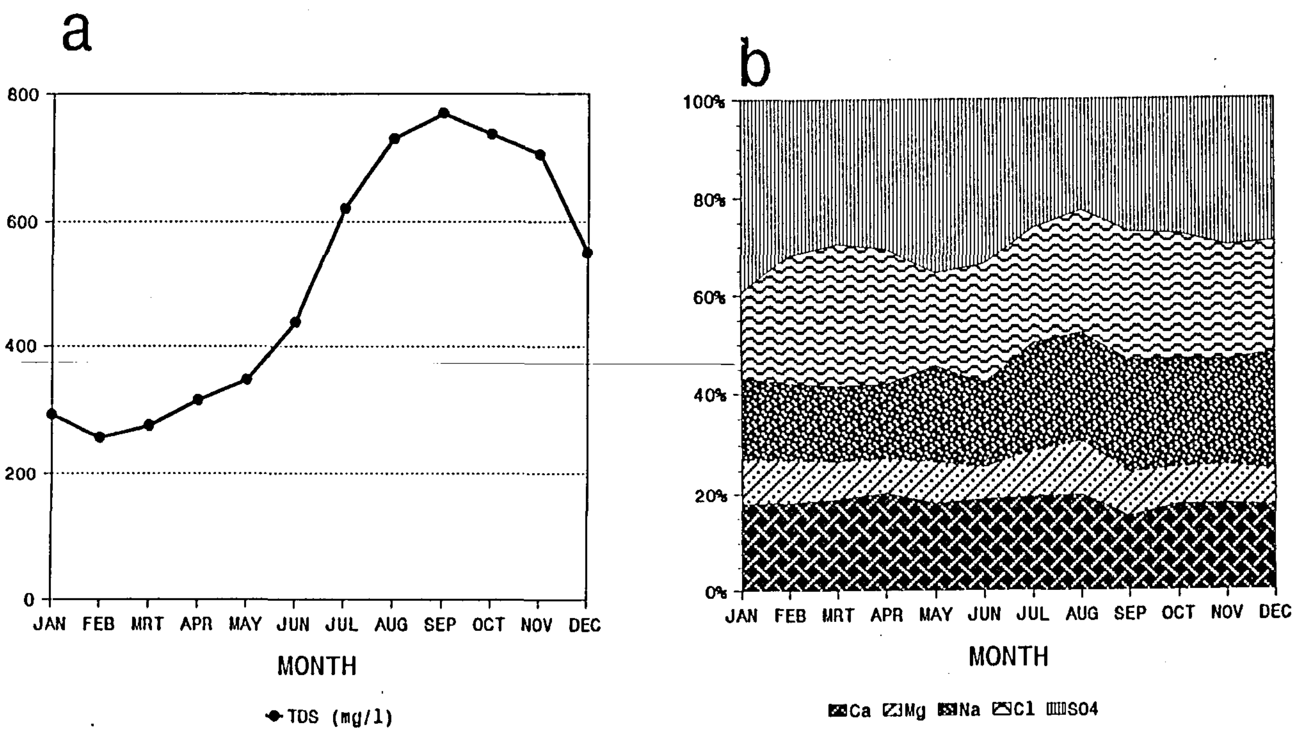


FIGURE 17: The average monthly TDS concentration for 1989 at Stilfontein (a). The average monthly ionic composition for 1989 at Stilfontein (b).

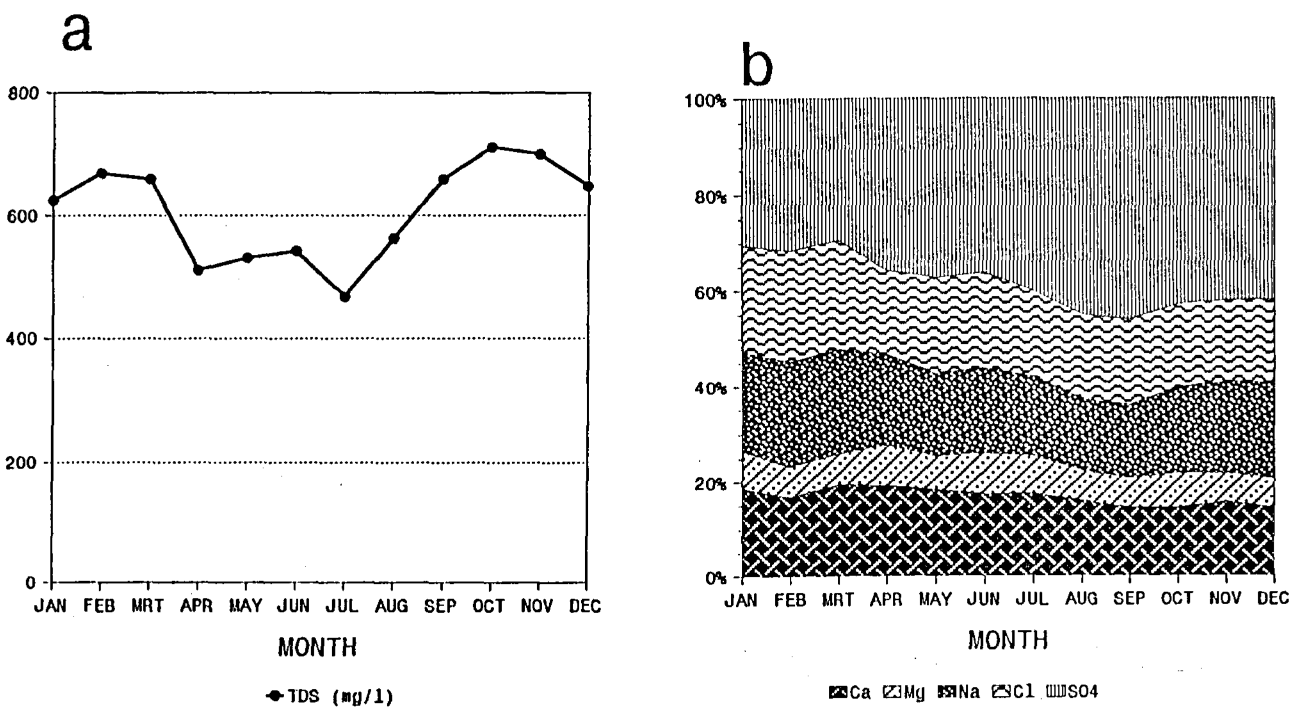


FIGURE 18: The average monthly TDS concentration for 1990 at Stilfontein (a). The average monthly ionic composition for 1990 at Stilfontein (b).

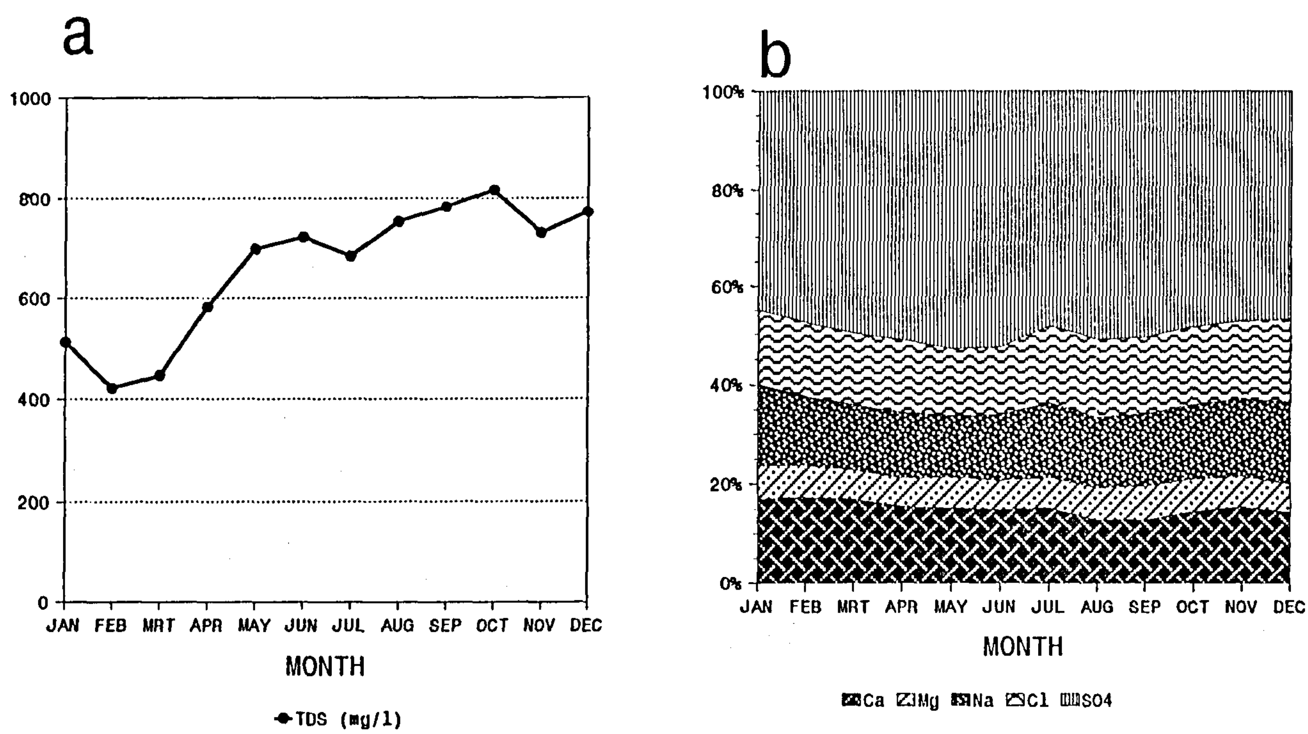


FIGURE 19: The average monthly TDS concentration for 1991 at Stilfontein (a). The average monthly ionic composition for 1991 at Stilfontein (b).

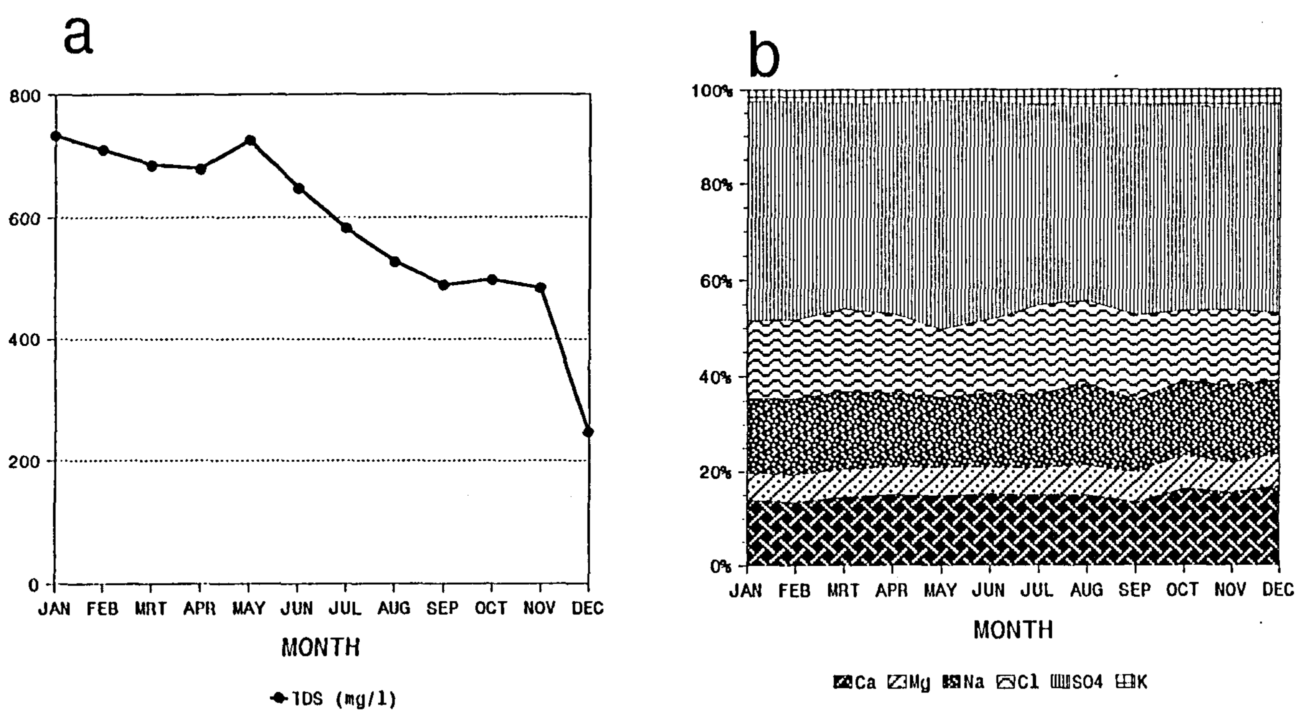


FIGURE 20: The average monthly TDS concentration for 1992 at Stilfontein (a). The average monthly ionic composition for 1992 at Stilfontein (b).

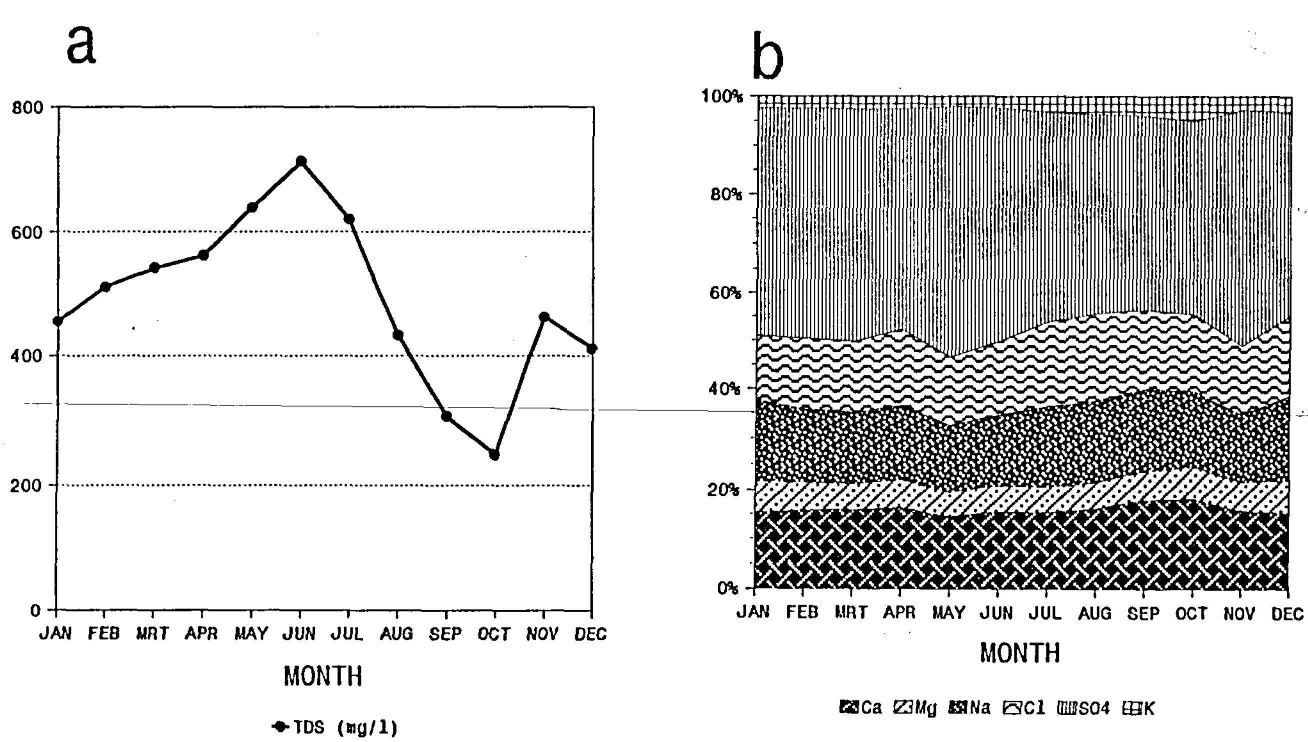


FIGURE 21: The average monthly TDS concentration for 1993 at Stilfontein (a). The average monthly ionic composition for 1993 at Stilfontein (b).

3.1.3. CONCLUSIONS

If results from the two study localities are compared for the period 1984 to 1993, it is evident that for Balkfontein a general increase of 29 mg $l^{-1}.a^{-1}$ was observed and that the mean annual TDS concentration for this period was 516.04 mg l^{-1}. At Stilfontein there was an annual increase of 21.3 mg l^{-1} and the mean annual TDS concentration was lower than at Balkfontein (504.2 mg l^{-1}). The highest salinity, for both Balkfontein and Stilfontein, usually occurred during the winter to spring period. During the years of flooding (March 1988 and February 1989), highest salinity values were recorded during spring. The salinity was usually low during summer months (November to March) with lowest salinity occurring during the flood months (March 1988 and February 1989). After the dilution of the salts by the floods, the salinity again increased sharply. Salinity at both the study localities displayed seasonal changes that were strongly influenced by the discharge of the river , especially discharge that originated from rainfalls. Salinity reduction by high discharge in the Vaal River is in accordance with the major river systems of the world (Harris, 1986).

The order of ionic prominence in mg l^{-1} in the Vaal River at Balkfontein was: SO_4^{2-} » Ca^{2+} ≥ Cl^- ≥ Na^+ ≥ Mg^{2+} » K and at Stilfontein : SO_4^{2-} » Cl ≥ Ca^{2+} ≥ Na^+ ≥ Mg^{2+} » K. For Balkfontein and Stilfontein the major ionic contributor to the total dissolved salts of the Vaal River was the anion SO_4^{2-}. Gibbs (1970) concluded that the three mechanisms controlling the composition of dissolved salts of the world's waters were atmospheric precipitation, rock dominance, and evaporation-crystalisation processes. The dominance, however, of SO_4^{2-} in the Vaal River indicated that pollution, evidently of mining and industrial origin, is the main source of the total dissolved salts.

The dissolved salts concentration in the Vaal River is very high, with an average annual maximum of 647.3 mg l^{-1} and a minimum TDS concentration of 393.5 mg l^{-1}. The mean annual TDS concentration for the study period was 518 mg l^{-1} which is approximately four times higher than the global mean salinity of river water (Wetzel, 1983). An increase rate (average) of 25 mg l^{-1} per annum was calculated for the study period 1984 to 1991 at both Stilfontein and Balkfontein. The Vaal River can therefore, be classified as mixohaline, while its mean salinity permits a classification of mixo-oligohaline in accordance with Roos (1991).

The origin of the high salinity in the middle Vaal River can be attributed to various sources, but intensive gold-mining activities and industrial discharges in the catchment area are probably the major sources.

The concentration of the major ionic components existed in the proportion of SO_4^{2-} » Ca^{2+} ≥ Cl^- ≥ Na^+ ≥ Mg^{2+} » K. The high sulphate concentration (average = 203.2 mg l^{-1}) indicated sulphate pollution in the middle Vaal River. All the dissolved ions in the Vaal River showed decreased concentrations with increased run-off. A constant variation pattern between the various ions suggested that the dissolved substances originated from the same source.

The total dissolved salts concentration in the Vaal River tend to be higher during the winter months. This is in accordance with most other African rivers (Wetzel, 1983). The main importance of the high salinity in the Vaal River is the apparent influence on turbidity and the clarification of the water column, which could result in more intensive algal blooms.

3.2. GROWTH AND CARBON ASSIMILATION EXPERIMENTS

3.2.1. EFFECTS ON GROWTH

Fig. 22a and b shows the effect of increased concentrations of total dissolved salts (TDS) on the growth (in Klett units) of *Microcystis aeruginosa.* In the Vaal River salts experiment the growth of *Microcystis aeruginosa* was inhibited after 5 days at concentrations of 500 mg l^{-1} and above, while the laboratory salts experiment showed no inhibitory effect. These observations are in agreement with information given by Wetzel (1983) indicating that most freshwater bacteria and blue-green algae (cyanobacteria) are sensitive to increased salinity, but can adapt to increasing salinity by means of genetic change.

The growth of *Cyclotella meneghiniana* (Fig. 23a & b) was stimulated after 8 days at a concentration of a 100 mg l^{-1}. At salinity concentrations between 250 mg l^{-1} and 2000 mg l^{-1} no growth occurred after 2 to 5 days, indicating that *Cyclotella meneghiniana* is sensitive to an increase in dissolved salts concentration. This was the case for the experiments done with the laboratory salts as well as for the Vaal River salts mix experiment. The observed sensitivity of *Cyclotella meneghiniana*, corresponds with Kolbe's (1932) classification of *Cyclotella meneghiniana* as an oligohalobien species that have optimum growth under conditions of low salt concentration. The low growth rate shown for *Cyclotella meneghiniana* could, however, be influenced by temperature (23°C) at which the cells were incubated. According to Patrick (1971) the development of populations of a given diatom species is correlated with temperature; some species are growing best under cool water conditions while others grow best under warmer conditions. The temperature requirements for *Cyclotella meneghiniana* are not known at present but it is possible that optimal temperature for growth is below 23°C, the temperature under which the experiments were preformed.

Figs. 24a & b shows the effect of increased TDS concentrations on the growth of *Monoraphidium circinale.* With an increase in TDS concentration, the growth of this species was neither stimulated nor inhibited for both the laboratory salts mixture and the Vaal River salts mixture. The apparent lack of effect of dissolved salts within the range investigated, can be attributed to the fact that *Monoraphidium circinale*, which is also present in Lake Kinneret in Israel, might be able to adapt to a wide range of salinities. According to Wetzel (1983) most algal species in waters from semi-arid and arid regions, like in Lake Kinneret, are adapted genetically to persist over a much wider range of salinities.

Fig. 25a & b shows the chlorophyll-*a* concentrations in *Cyclotella meneghiniana, Microcystis aeruginosa* and *Monoraphidium circinale* cultures after 8 days of growth. With an increase in the salinity concentration the chlorophyll-*a* concentration in *Cyclotella meneghiniana, Microcystis aeruginosa*

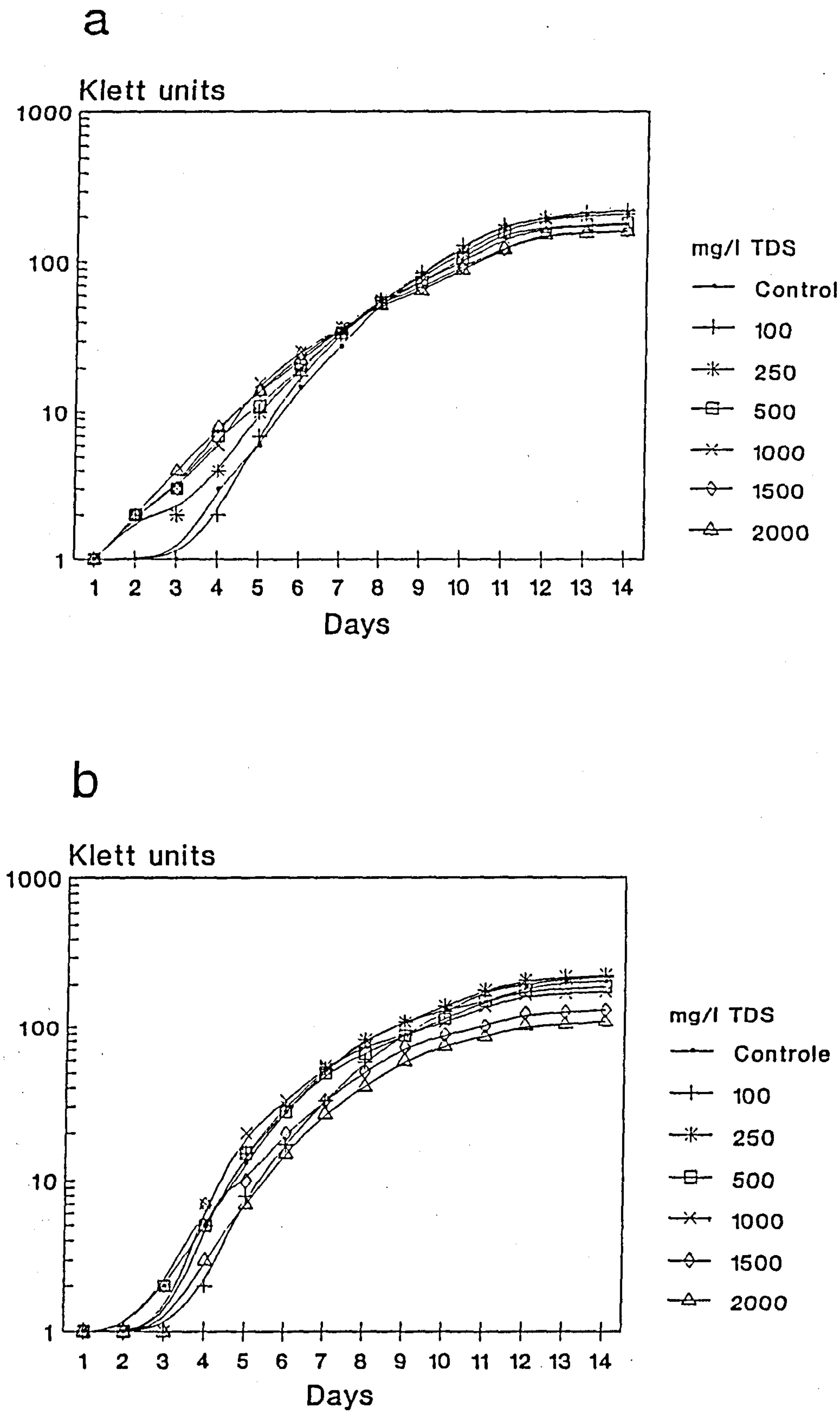
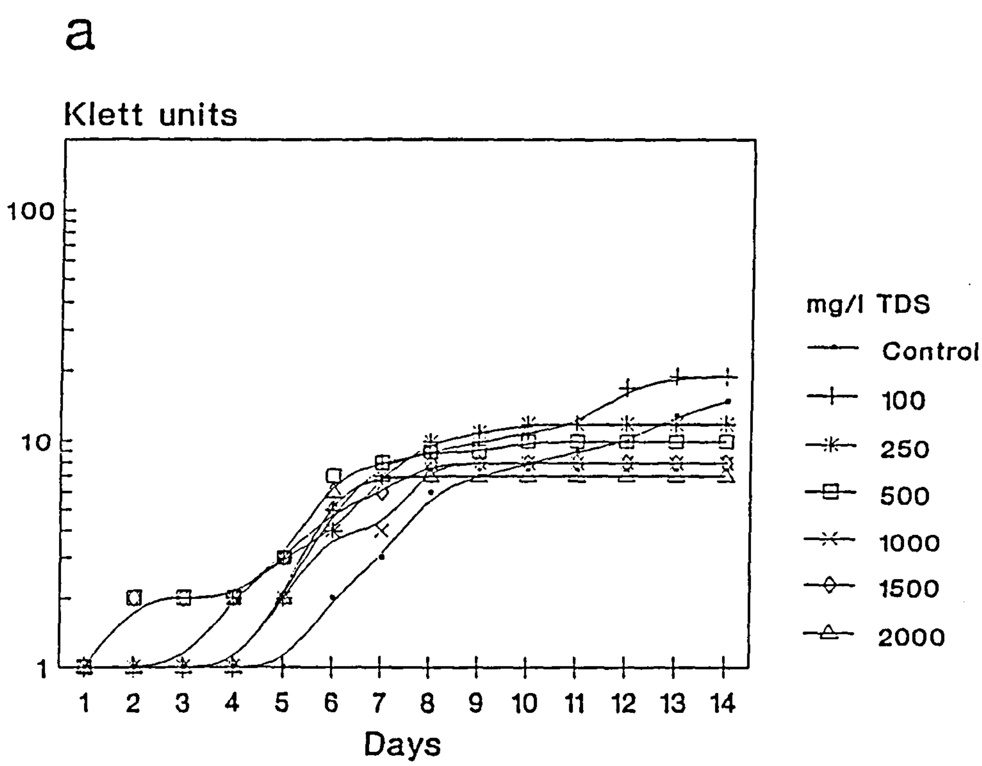


FIGURE 22: The effect of increased TDS concentration lab. salts (a) and Vaal River salts (b) on the growth of *Microcystis aeruginosa* over a 14 day period.



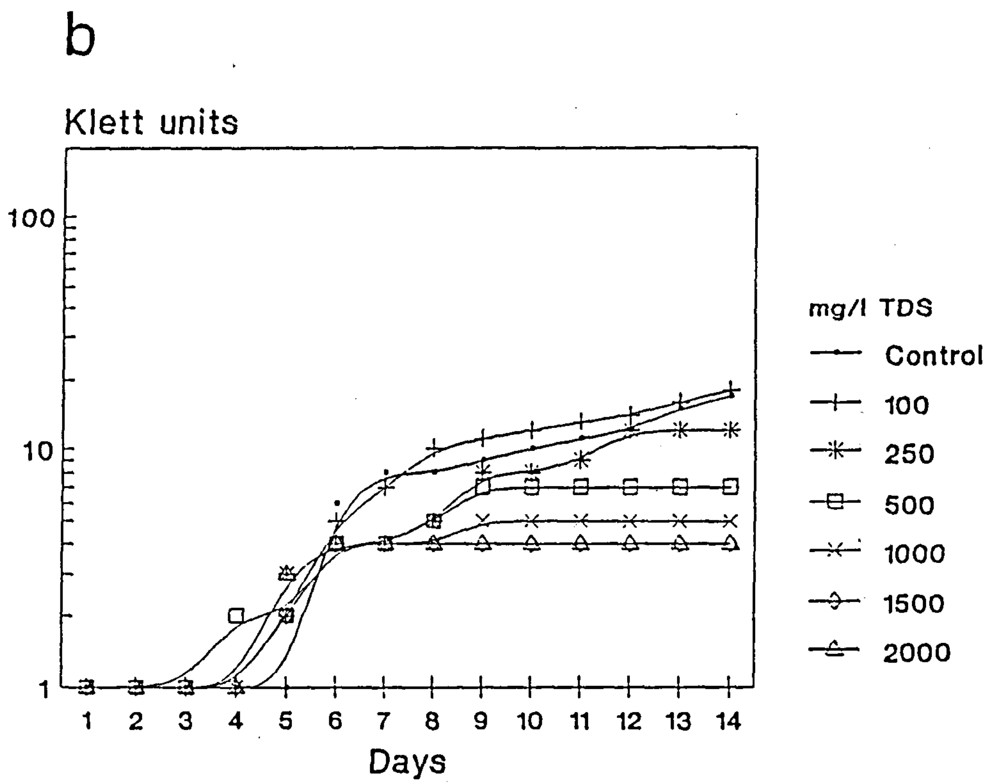


FIGURE 23: The effect of increased TDS concentration lab. salts (a) and Vaal River salts (b) on the growth of *Cyclotella meneghiniana* over a 14 day period.

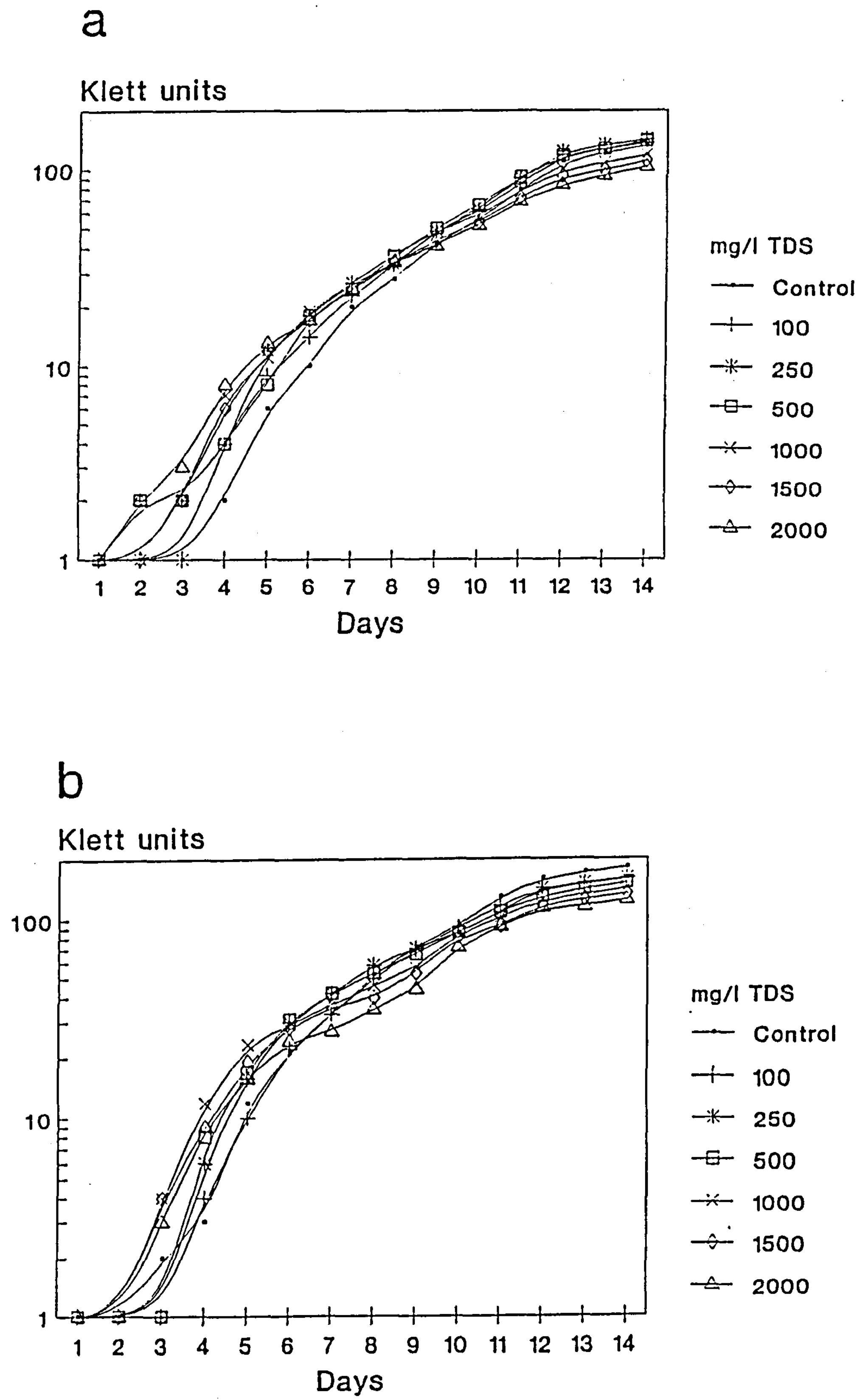
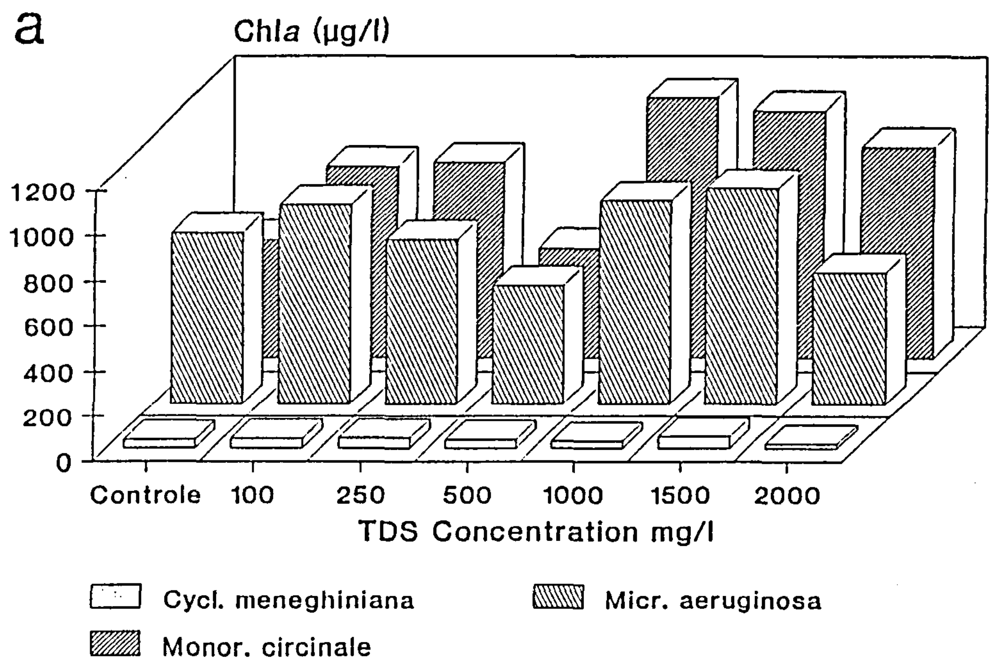


FIGURE 24: The effect of increased TDS concentration lab. salts (a) and Vaal River salts (b) on the growth of *Monoraphidium circinale* over a 14 day period.



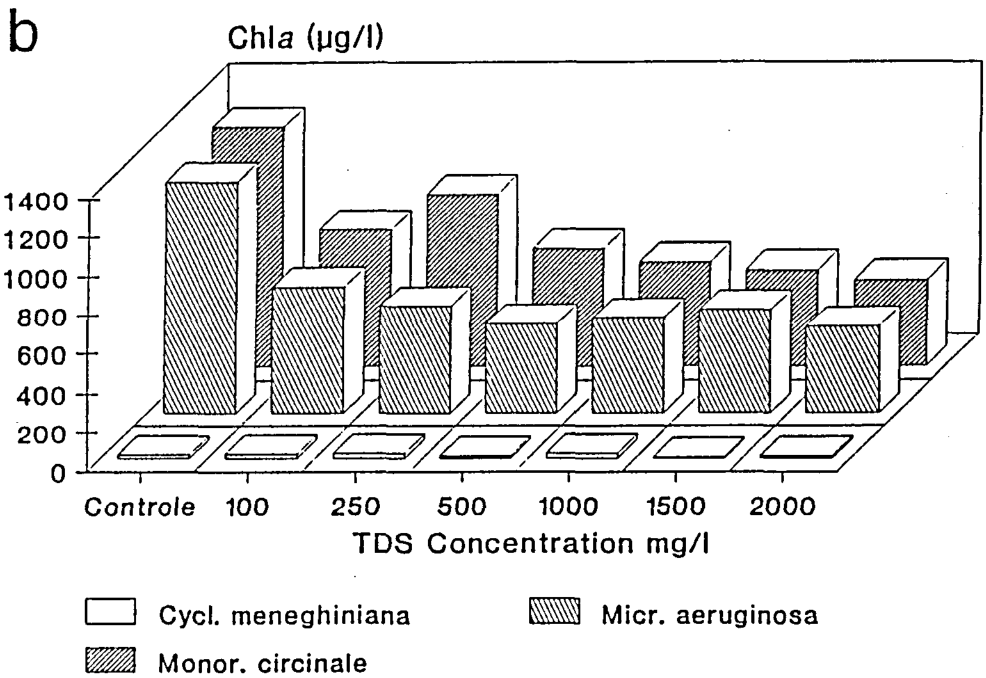


FIGURE 25: Chlorophyll-*a* concentrations in *Cyclotella menghiniana*, *Microcystis aeruginosa* and *Monoraphidium circinale* for (a) lab. salts and (b) Vaal River salts.

and *Monoraphidium circinale* cultures generally increased at TDS concentrations up to 500 mg l^{-1} and then decreased at concentrations between 1000 mg l^{-1} and 2000 mg l^{-1} (Fig. 25a). The chlorophyll-*a* concentration in the experiment done with the Vaal River salts mix shows an increase in the chlorophyll-*a* concentration with an increase in the salinity concentration up to 250 mg l^{-1}, and then decreases at salinities between 500 and 2000 mg l^{-1} (Fig. 25b).

3.2.2. EFFECTS ON CARBON ASSIMILATION

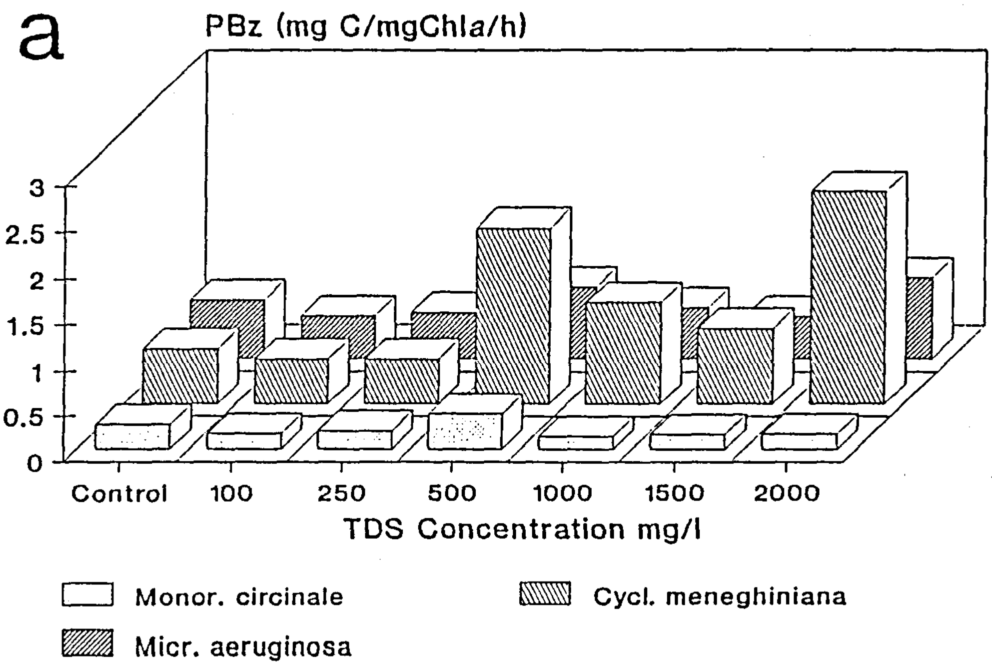
Fig. 26a & b shows that the carbon assimilation rate of *Cyclotella meneghiniana* decreased with an increase in the TDS concentration in correspondence with the growth results illustrated in Fig. 23a & b. The carbon assimilation rate of *Monoraphidium circinale* was low at TDS concentrations of 250 mg l^{-1} and above, while TDS showed no effect on the growth of this species (Fig. 24a & b). The carbon assimilation of *Microcystis aeruginosa* generally increased with an increase in the concentration of TDS (Fig. 26a & b) in contrast to the effect TDS had on the growth of this species (Fig. 22a & b).

3.2.3 THE EFFECT OF NUTRIENTS IN THE VAAL RIVER SALTS MIX ON THE GROWTH OF THE ALGAL SPECIES

The growth in both *Cyclotella meneghiniana* and *Microcystis aeruginosa* (Figs. 27a & b) could only be sustained for a period of 5 days after which no growth occurred. It is possible that the salts added, together with the growth substances in the natural salts mix, could have inhibited the growth of these two species as in the growth experiment done with the GBG-11 growth medium (see Figs 22a & b and Figs 23a & b). The growth of *Monoraphidium circinale* (Fig. 27c) showed no real effect with an increase in the natural salts mix concentrations added, which correlated with the results of the experiment done with the GBG-11 growth medium.

The final biomass after 14 days of growth (Fig. 27d) was very low for the three species investigated, which indicated that the growth substances could only sustain growth for a short period of time. The results of this experiment clearly showed that the nutrients present in the natural salts mix had no effect on the results of the experiment in which the effect of an increase in TDS concentration was investigated on growth of the algal species.

If the different growth results of this experiment are compared, it is evident that the growth of the three species investigated was stimulated with the addition of the natural salts mix, which indicated that growth substances were present in the natural salts mix used.



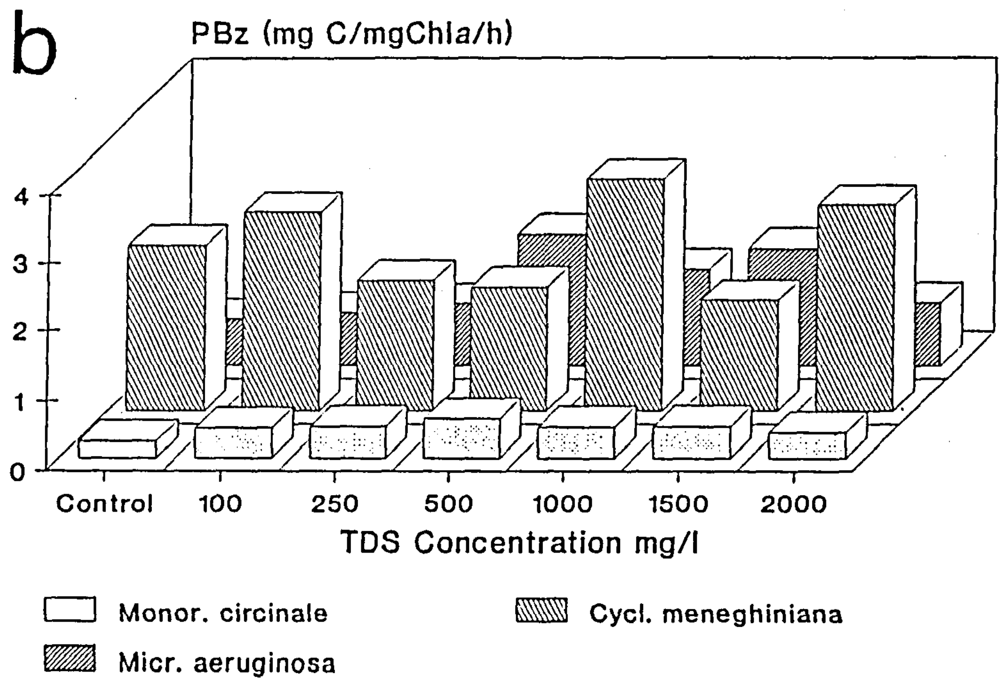


FIGURE 26: Carbon assimilation by *Cyclotella meneghiniana, Microcystis aeruginosa* and *Monoraphidium circinale* for (a) lab. salts and (b) Vaal River salts in mg C mg Chl-a^{-1} h^{-1}.

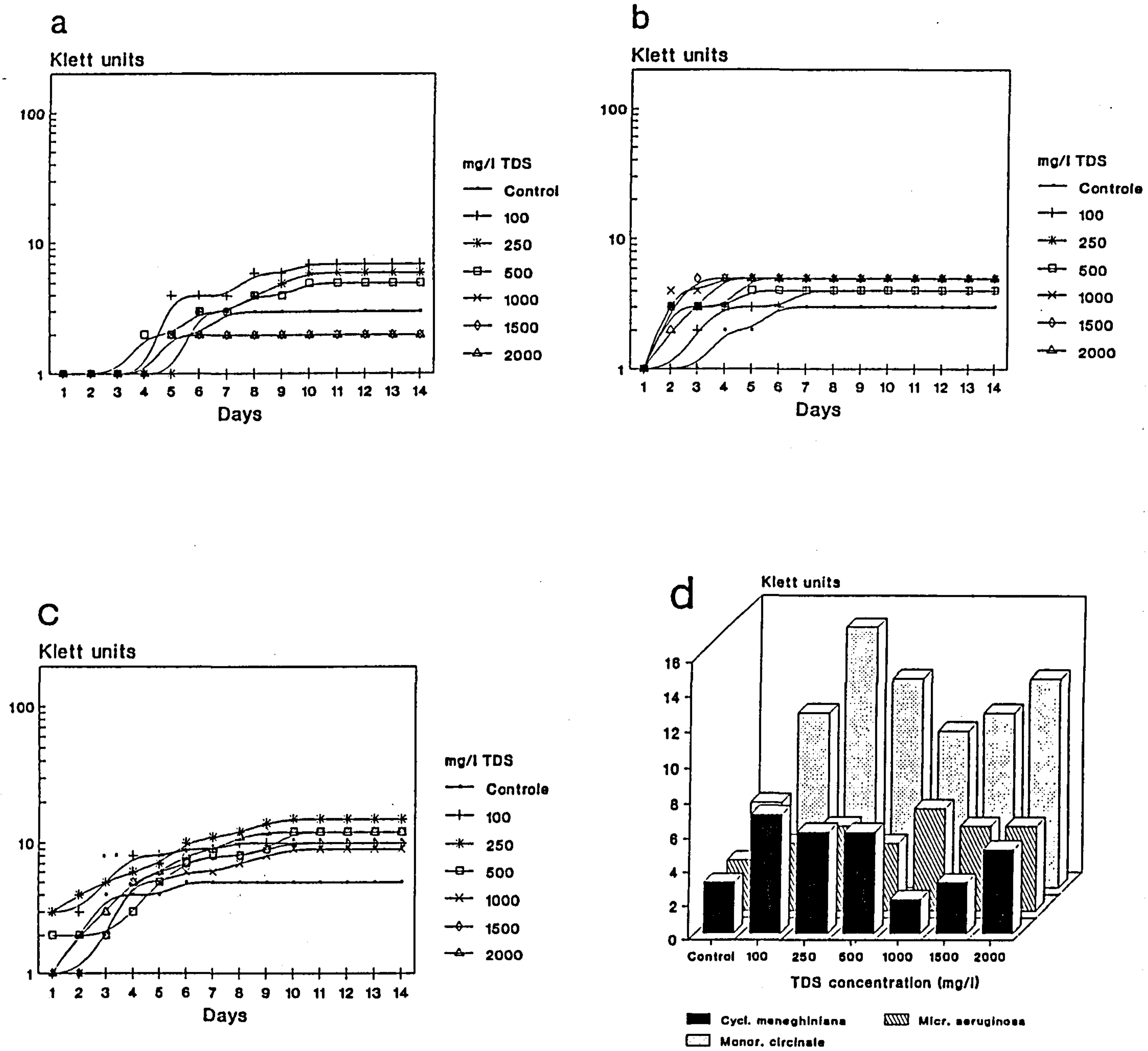


FIGURE 27: The effect of increased concentrations of total dissolved salts (TDS) on the growth of *Cyclotella meneghiniana* (a), *Microcystis aeruginosa* (b) and *Monoraphidium circinale* (c) as well as the final biomass (d) after 14 days of growth.

3.2.4. CONCLUSIONS

Different algae showed different sensitivities to TDS. Of the three algae investigated *Cyclotella meneghiniana* was the most sensitive and *Monoraphidium circinale* the least sensitive to increased dissolved salts.

If the salt concentration of Vaal River water increases with time, algal species that could persist over a much wider range of salinities can become dominant. Of the species investigated, *Monoraphidium circinale* can be expected to become dominant under conditions of increased salinity. *Cyclotella meneghiniana* and *Microcystis aeruginosa* on the other hand, can be expected to be excluded from the water under conditions of increased salinity to above 250 mg l^{-1}.

According to Sanet Janse van Vuuren (Department of Plant and Soil Sciences, PU for CHE, Potchefstroom) the Cyanophyceae, of which *Microcystis aeruginosa* is a member, is usually dominant during the summer months when the TDS concentration is low. This is in agreement with the results of the growth experiments, at the low TDS concentrations there was no real inhibition of growth but at the high TDS concentrations growth was inhibited. The Chlorophyceae, of which *Monoraphidium circinale* is a member are dominant for most of the year and showed no preference for a specific season. The fact that an increase in TDS concentration showed no real effect on the growth of *Monoraphidium circinale* is in agreement with the dominance of this group (Chlorophyceae) for most of the year.

4. SUMMARY

4.1 HISTORICAL OVERVIEW

4.1.1 **Total dissolved salts** (TDS) concentration for 1984 - 1993 was high - average annual maximum (= 647 mg l^{-1}) and average annual minimum (= 395 mg l^{-1}).

4.1.2 **Mean annual TDS concentration** (= 519 mg l^{-1}) was four times higher than the global mean salinity for river water.

4.1.3 **Major ionic components** existed in the porportion of $SO_4^{2-} >> Ca^{2+} \geq Cl^- \geq Na^+ \geq Mg^+ >> K^+$.

4.1.4 **High sulphate concentration** (= 203.2 mg l^{-1}) indicates sulphate polution in the river.

4.1.5 **Constant variation pattern** between the various ions suggest that the substances originate from the same source.

4.1.6 **Seasonal variation** - TDS was higher in the winter months and showed a decrease with an increase in run-off during the summer months.

4.2 GROWTH AND CARBON ASSIMILATION EXPERIMENTS

4.2.1 **Different algal species showed different sensitivities with imcreased TDS concentrations** - *Cyclotella meneghiniana* (diatom) was the most sensitive and *Monoraphidium circinale* (green alga) the least sensitive.

4.2.2 With increased TDS concentration above 250 mg l^{-1} *Monoraphidium circinale* can become dominant and *Microcystis aeruginosa* (blue-green alga) and *Cyclotella meneghiniana* will be excluded from the water.

5. ACKNOWLEDGMENTS

Mrs M Krüger, Mr J Pietersen and Mr K Morgan of the Western Transvaal Regional Water Company, Stilfontein, and Mr G Quibell of the Department of Water Affairs and Forestry, Pretoria, for making data available for use.

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CHAPTER 5: NUTRIENT AVAILABILITY: PHOSPHATASE AND NITRATE REDUCTASE ACTIVITY IN THE VAAL RIVER AND ITS PHYTOPLANKTON

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1. INTRODUCTION

South Africa lies in a dry girdle. The western parts of the country are drier than the eastern parts and as rainfall decreases, rainfalls tend to become less reliable. The major limiting resource in South Africa is thus undoubtedly water. Preservation of water quality and sound management of the river are therefore of utmost national importance.

The Vaal River is the most important freshwater river system in the SA heartland. It is geographically strategically situated to serve one-fifth of the South African population (8 million people) in an area covering only 1,4 % of the total land surface (DWA, 1986). Its catchment area is 17 000 km^2 and it supplies water to 60 % of the country's industries. With the ever increasing water demand, it has been estimated that the maximum delivering capacity of the Vaal River will be exceeded by the end of this century (DWA, 1986). We are thus obliged to conserve and optimise the remaining Vaal River water source.

The future useage of Vaal water will be influenced by the following factors, namely the increase in population, the rise in living standards of the population, the increase in agricultural and industrial water demands and the urbanisation of rural areas.
Future economic growth in the New South Africa is thus largely dependant on an efficient water supply from the Vaal River (DWA, 1986).

Purification processes of natural waters are affected by algal growth which is enhanced in the Vaal River by extensive eutrophication. The increasing human population is largely responsible for the large amounts of pollutants released into the river. Although not all algae such as *Microcystis* and *Oscillatoria* species (Steyn, 1945 and Ressom *et al.*, 1993) are toxic, many can impart unpleasant tastes and smells to the water and can block agricultural filters leading to a reduction in water pumping capabilities. Algal blooms are the cause of tedious and expensive purification processes.

Because algal growth plays an important role in the aesthetic and biological quality of Vaal River water, the development of sound management strategies will be possible if the causes of algal blooms are known. For this reason the research of the present study, i.e. investigating inorganic nitrogen and phosphorous availability to algae through estimates of enzyme activities, is of direct relevance to one of the most urgent problems facing SA today, namely the supply of water of a sufficient quality for urban and industrial use.

In order to understand the development of algal blooms, research has been done on physical and chemical factors which influence the patterns of algal development (Wetzel, 1983). Nitrogen and phosphorus are two elements necessary for algal growth in aquatic

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and terrestrial systems. Sources of nitrogen and phosphorus in aquatic systems include agricutural, domestic and urban run-off. These sources are usually the cause of blooms due to the large quantities of inorganic N and P enriching the water. However, both can also be limiting to algal growth; inorganic P is generally growth limiting in freshwater and inorganic N is generally growth limiting in marine waters and in certain tropical lakes (Wetzel, 1983). Algal species generally have different N:P ratio requirements for optimal growth. Not only is the N:P ratio important for growth, but also the inorganic nitrogen and phosphorus concentrations as well as their rate of supply, and therefore inorganic N and P availability. In addition, the intensity and quality of light influences the optimum N:P ratio in phytoplankton and this effect is species specific (Wynne & Rhee, 1988).

From literature it is known that nitrate most probably limits algal growth when the N:P ratio is < 10, while phosphate is most probably limiting when the N:P ratio is > 20. Where the N:P ratios are between 10 and 20, not one of the two nutrients are limiting. The ratios of carbon, nitrogen, phosphorus and chlorophyll in algal cells show fluctuations which can possibly be related to the nutrient conditions that influence the annual patterns of algal blooms. Chlorophyll concentration (an indication of biomass) is dependant on both phosphate and inorganic N concentrations as well as the total N to total P (TN:TP) and inorganic N to P (inorganic DIN:DIP) ratios. It was suggested by Sakamoto (1966) that chlorophyll is a log-function of both total phosphorus and total nitrogen.

Two enzymes which play a vitally important role in making these essential elements, nitrogen and phosphorus, available to algae are Nitrate Reductase (NR) and Phosphatase (PASE) respectively. The function of enzymes is to increase the rate of biochemical reactions. Enzymes are regulated; that is, they are capable of changing back and forth from a state of low activity to one of high activity (Mathews & van Holde, 1990).

Nitrate reductase and Phosphatase are inducible enzymes, which means that their activity is regulated by extracellular nutrient concentration (Wynne, 1981; Hochman *et al.*, 1986). Both enzymes are metalloenzymes, requiring metals for optimal activity, i.e. zinc for phosphatase and molybdenum for nitrate reductase (Campbell & Smarrelli, 1986; Sabbioni *et al.*, 1976). Enzymes can ensure a supply of essential nutrients to growing cells that would otherwise be unavailable. These two enzymes and the role which they play in nutrient availability to phytoplankton, form the core of this study.

Nitrogen is an essential element for many metabolic processes in all living organisms. One of its functions is the formation of amino acids to form proteins. N is thus essential for growth.

Algae can utilise NH_4^+, urea-N and nitrate (NO_3-N) as nitrogen source, but NH_4^+ is always taken up preferentially (Berman *et al.*, 1984). The reason for this is that NH_4^+ is the only inorganic nitrogen source to be taken up and metabolised directly by phytoplankton. NO_3^- must first be reduced to NH_4^+ by enzymes Nitrate Reductase and Nitrite Reductase (Guerrero *et al.*, 1981), before the nitrogen can be incorporated into the cells. NO_3^- assimmilation consists of two closely linked steps, namely the uptake of nitrate and its reduction to NH4+. Although NH_4^+ does not directly inhibit NR activity, both the uptake of NO_3^- and the *de novo* synthesis of NR is usually inhibited even at low (< 2 µm) ambient concentrations of NH4+.

When NH_4^+ levels are low and NO_3^- levels are high, then a large portion of the total nitrogen flux comes from NO_3^- (Berman *et al.*, 1984). NO_3^- thus seems to be an emergency back-up source of nitrogen and Nitrate Reductase is the enzyme ensuring that this essential nutrient is available for utilisation by the algae.

Nitrate Reductase activity can be used as an indicator of the nitrogen source and reflects ongoing nitrate uptake. Discernable activities of this enzyme reflect the use of nitrate rather than NH_4^+ as the source of nitrogen. The levels of Nitrate Reductase could be used to predict actual rates of nitrate uptake (Hochman, 1986).

In the present investigation, the role of Nitrate Reductase in the phytoplankton from the Vaal River at Balkfontein was examined.

Phosphate is one of the more general phytoplankton growth-limiting elements, even though it is only needed in small amounts.

Inorganic orthophosphate (PO_4-P) is the main P source directly available to phytoplankton. However, cells are able to synthesise alkaline phosphatase, an enzyme which will hydrolyse organophosphate esters, thereby liberating orthophosphate for subsequent metabolism by the algae (Wynne, 1981). Two types of phosphatase occur, namely acid and alkaline phosphatase. Both these enzymes occur in a soluble and particulate fraction. Acid phosphatase activity may be involved in the carbon photosynthetic pathway (Holm-Hansen, 1962) and this could explain its apparent correlation with the chlorophyll content of cells (Wynne, 1977).

Alkaline phosphatase is involved in bone and fish scale formation. This implicates its involvement in protein synthesis and other metabolic processes, not directly connected with PO_4-P nutrition, possibly via their phosphotransferase activity (Barman, 1969). The alkaline phosphatase enzyme also shows diurnal fluctuations suggesting a role in cellular metabolism (Wynne, 1981). A possible connection between enzymatic activity and cell division has been demonstrated. The maximum division rate of *Peridinium cinctum* cells in Lake Kinneret corresponded approximately with the time of maximal alkaline phosphatase activity of the cells therefore implying a connection between enzyme activity and cell division (Pollinger & Serruya, 1976; Pollinger, 1978). The presence of alkaline phosphatase in lake water appears to reflect the degree of eutrofication of a lake (Jones, 1972).

In the present study, the role of the different enzymes in the phytoplankton from the Vaal River at Balkfontein was examined.

2. MATERIALS AND METHODS

The following chemicals were used in the present study: NADH obtained from Boehringer (Mannheim, Germany), Sulfanilamide and $ZnSO_4$ from BDH Chemicals Ltd., England and

NaOH from PAL Chemicals. Toluene was purchased from NT Laboratory Supplies (Pty.) Ltd. and K_2HPO_4 and KH_2PO_4 from SaARchem.

Axenic cultures of and natural phytoplankton samples from the Vaal River at Balkfontein, were examined for Nitrate Reductase and Phosphatase activity. For samples of natural phytoplankton, near surface water samples were collected from the Vaal River at approximately 14h00 once a month. The samples were transported to the laboratory in plastic jars and kept in a cool room overnight before analysis the following day.

Various techniques and experimental procedures were carried out in the laboratory in order to find the best method suited to the given conditions. These procedures preceded the determination of enzyme activity from the Vaal River. The enzymes need to be extracted from the cells and the efficient breaking of the cells is therefore crucial for determining enzyme activity.

As part of a long-term monitoring programme, ambient concentrations of major nitrogen and phosphorus nutrients were measured once a month at Balkfontein. These findings were then compared with enzyme activities. The following nitrogen and phosphorus components were investigated. DIN (dissolved inorganic nitrogen) which is the available inorganic nitrogen and includes nitrate nitrogen (NO_3-N) and ammonium nitrogen (NH_4-N) and TN (total available nitrogen) which includes the inorganic and organic nitrogen components. The phosphorus nutrients examined were DIP (dissolved inorganic phosphate; PO_4-P) which is the available inorganic phosphate and TP (total available phosphorus) which is the inorganic and organic phosphorus components. Chlorophyll-*a* concentration is an indication of algal biomass and was also included in this study.

2.1 NITRATE REDUCTASE ASSAYS

The determination of NR activity was based upon the assay method of Hochman *et al.* (1986). Two techniques for the collection of cells for the assay of NR activity were applied. Cells were collected by centrifugation and filtration.

Natural Vaal River phytoplankton suspensions were centrifuged at 3000 rpm for 10 minutes in a *Sorvall GLC-I* centrifuge. After discarding the supernatant, the samples were resuspended in 4,5 ml PO_4-buffer (150 mM, pH 7.6) made up of K_2HPO_4 and KH_2PO_4. The cell slurry was homogenised on ice for 2 minutes with a *Heidolph* glass-Teflon homogeniser.

In addition, natural Vaal River phytoplankton suspensions were filtered on 4.7 cm GF/C glass microfibre filters. 1.5 ml PO_4-buffer (150 mM, pH 7.6) was added to the filtrate on the filters, immediately followed by 60 µl of toluene. One set of samples were vortexed for 1 minute and then homogenised and one set was only homogenised for 2 minutes.

After collecting the cells by either filtration or centrifugation, the enzyme reaction was initiated by adding 0.3 ml NADH (6.5 mM - 5 mg per 1 ml PO_4 buffer (150 mM, pH 7,6)) and 0.2 ml 0.1 M KNO_3 to 1.5 ml homogenate. The samples were gently shaken on ice on a *Gerhardt* shaking apparatus for 30 minutes. During this time a beaker of water was boiled on a Bunsen burner into which test tubes containing 1.7 ml 0.13 M $ZnSO_4$ were placed. To terminate the reaction, 1 ml of the cell slurry was withdrawn and rapidly mixed into a test tube containing 1.7 ml 0.13 M $ZnSO_4$ at 97°C and boiled for 60 seconds. Test tubes were cooled on ice and 0.2 ml of 1N NaOH was added. The samples were centrifuged for 10 minutes at 13 000 rpm on a *Beckman Model J2-21M* centrifuge. The supernatant was then analysed for NO_2.

Nitrite was determined by a diazo coupling method (Strickland & Passons, 1972). Two ml of the supernatant was removed and assayed for NO_2 by adding, with mixing, 0.4 ml of Reagent A* and 0.4 ml of Reagent B**. Absorbance at 540 nm was determined after 10 minutes (Nicholas & Nason, 1957).
* Components of Reagent A: 2.5 g Sulfanilamide dissolved in 250 ml of diluted HCl (25 ml HCl with 225 ml H_20).
** Components of Reagent B: 0.25 g N-(1-Naphthyl)-ethylenediamine dihydrochloride GR (NED) in 250 ml H_20.

2.2 PHOSPHATASE ASSAYS

All reagents used were of analytical grade or equivalent. P-nitrophenyl phosphate (PNPP) was obtained from Sigma.

Four different phosphatase activities were examined, namely Alkaline soluble and particulate Phosphatase and Acid soluble and particulate Phosphatase.

Cell samples (2 ml) were filtered using (GF/C) filter paper. The filtrate was used to determine soluble Phosphatase activities while the filter was used to determine particulate Phosphatase activities. Phosphatase activities were measured by a spectrophotometric method, using P-nitrophenyl phosphate (PNPP) as substrate (Wynne, 1977; 1981).

Particulate Phosphatase activity was determined in the following way. The filter was mixed with 2.6 ml acetate buffer (0.2 M, pH 5) consisting of acetic acid and sodium acetate for acid Phosphatase activity and 2.6 ml Tris-HCl buffer (0.05 M, pH 8.6) and 0.03 ml 0.1 M $MgCl_2$ for alkaline Phosphatase activity. To both acid and alkaline samples 2 drops of chloroform and 0.1 ml 10 mM PNPP was added prior to incubation of 2 hrs. The reaction was then terminated by the addition of 0.3 ml 1 M NaOH and the Particulate Phosphatase activity calculated from the absorbance of released P-nitrophenyl read at 410 nm on a Beckman Model J2-21M centrifuge.

Soluble Phosphatase activity was determined in the following way. The filtrate was mixed with 0.27 ml acetate buffer (1 M, pH 5) for acid phosphatase activity and 0.27 ml Tris-HCl buffer (0.5 M, pH 8.6) and 0.027 ml 0.1 M $MgCl_2$ for alkaline phosphatase activity. The volumes of both alkaline and acid samples were then made up to 2.6 ml with distilled water and 0.1 ml 10 mM PNPP was added. Incubation and termination followed as described for the particulate fraction. Soluble Phosphatase activities were then determined in the same way as for Particulate Phosphatase activities.

2.3 CHLOROPHYLL-a CONCENTRATION

Vaal River water samples were filtered with 0.6 µm GF/C filter papers. Ten ml of 96% ethanol was added, and the samples were placed directly into a water bath at 78 °C for 5 minutes. The samples were then left to cool in the dark after which absorbance was read at 750 nm, 665 nm and 665 nm (acidified) (Holm-Hansen *et al.*,1965).

3. RESULTS

Dissolved inoganic phosphate (DIP; PO_4-P) and dissolved nitrate nitrogen (NO_3-N) for the study period are shown in Fig. 1. High NO_3-N concentrations were found in August and from December to March. From September to December NO_3-N concentrations decreased constantly and then also in April. PO_4-P concentrations increased from September to October and then again from December to February. PO_4-P concentrations decreased from August to September, from October to December and then from February to March.

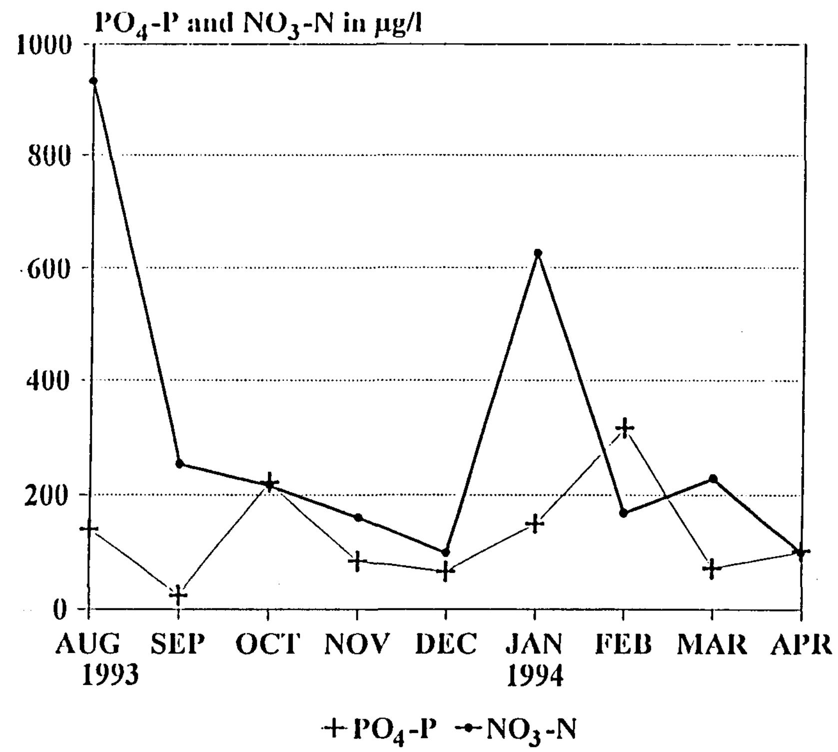

FIGURE 1: Dissolved inorganic phosphate (DIP; PO_4-P) and dissolved nitrate nitrogen concentrations (NO_3-N) in the Vaal River at Balkfontein.

In Fig. 2 total phosphorus (TP) and total nitrogen (TN) are shown. TP values remain fairly constant over the entire study period, reaching low concentrations in September, December and March. The TP values follow the same trend as PO_4-P, which implies that dissolved inorganic phosphate was the dominating P source during the present study. The TN followed a similar trend to NO_3-N, with an exception in October when the NO_3-N concentration was relatively low.

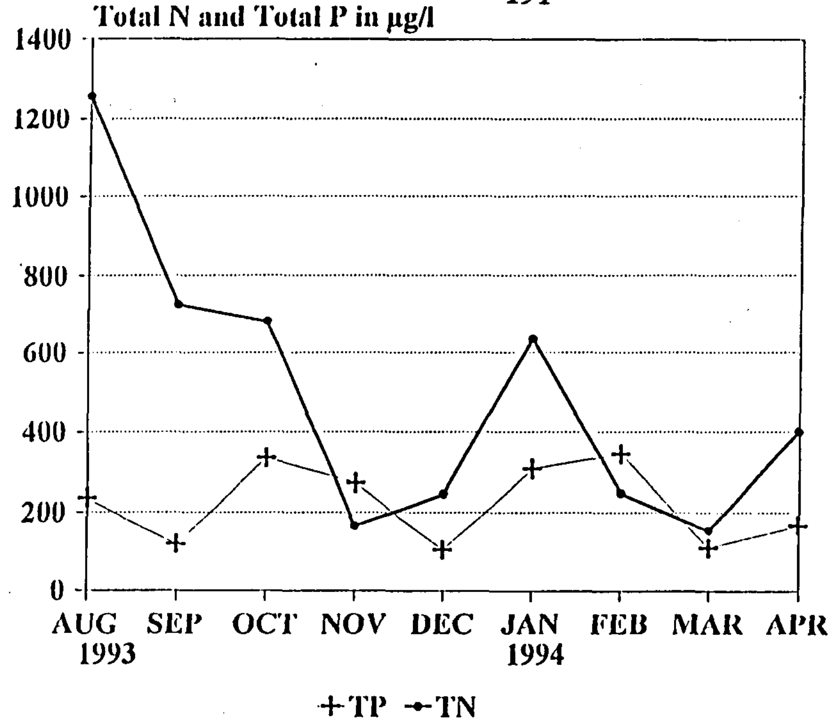

FIGURE 2: Total phosphorus (TP) and total nitrogen (TN) concentrations in the Vaal River at Balkfontein.

DIN:DIP and TN:TP ratios are shown in Fig. 3. The DIN:DIP ratios were high until December, with the exception of October. From December to April, low DIN:DIP ratios occurred. There is a fluctuation in the TN:TP ratios, but the ratios are generally always lower than the corresponding DIN:DIP ratios. In August and September, the TN:TP ratios were high, but from September onwards these ratios decrease and remained low.

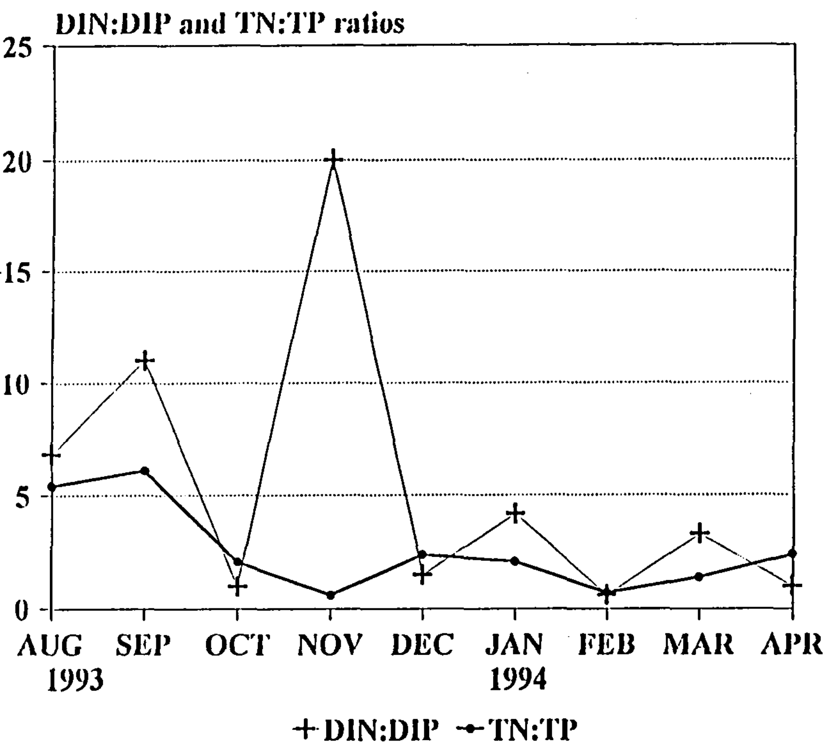

FIGURE 3: Dissolved inorganic nitrogen (DIN) to dissolved inorganic phosphorus (DIP) and total nitrogen (TN) to total phosphorus ratios in the Vaal River at Balkfontein.

The fluctuation of chlorophyll-*a* during the study period is shown in Fig. 4. Chlorophyll-*a* peaks occurred in September, November and March.

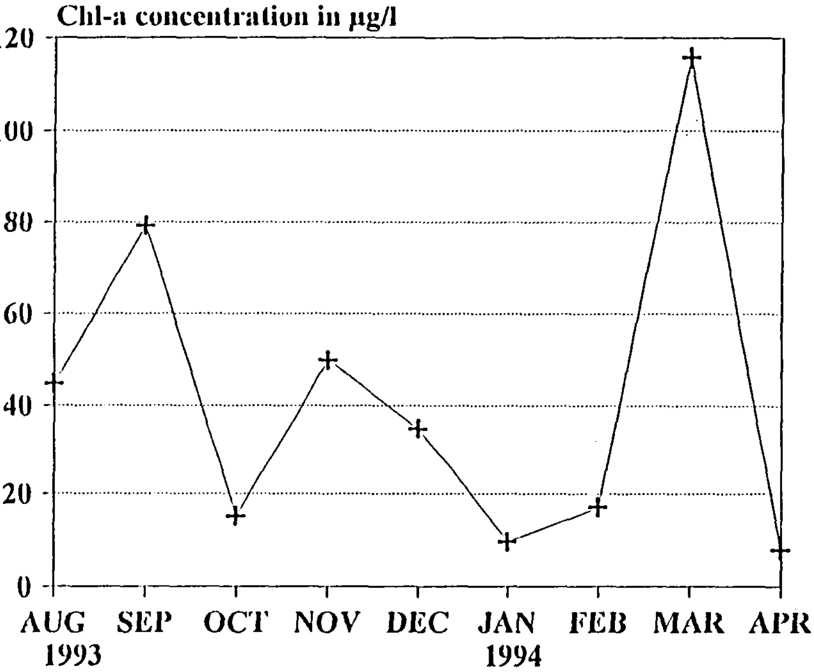

FIGURE 4: Chlorophyll-*a* concentration in the Vaal River at Balkfontein.

In Fig. 5 the specific NR activity is shown. Peaks in the enzyme activity occur in August, October, January and April.

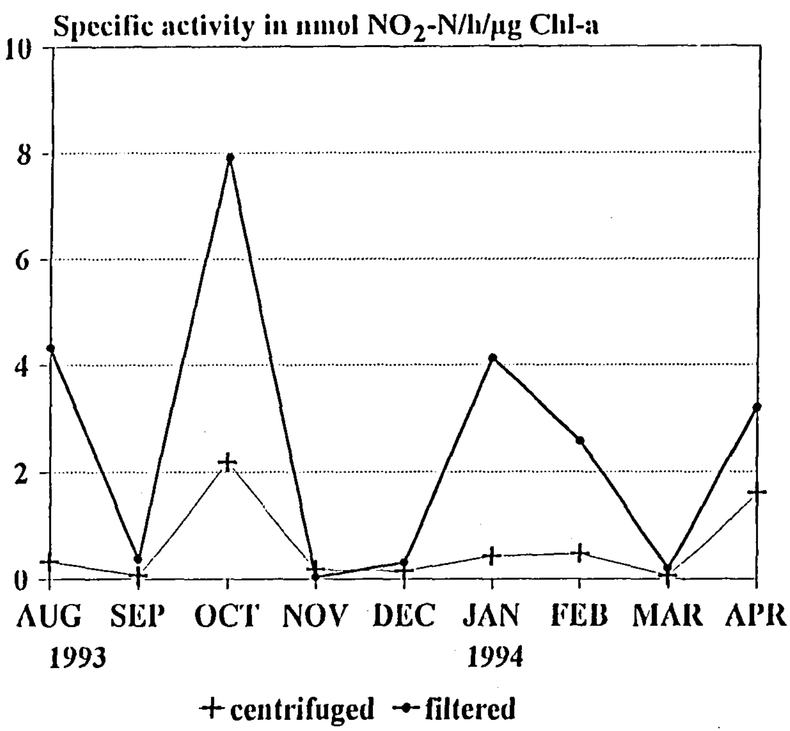

FIGURE 5: Specific Nitrate Reductase activity in the Vaal River at Balkfontein.

In Figure 6 the specific PASE activities are shown. The four different phosphatases follow the same general trend with the exception of the activity of soluble alkaline activity which increases in February in contrast to the decreases of the other three enzymes. Peaks in the enzyme activities occur in October, January and April.

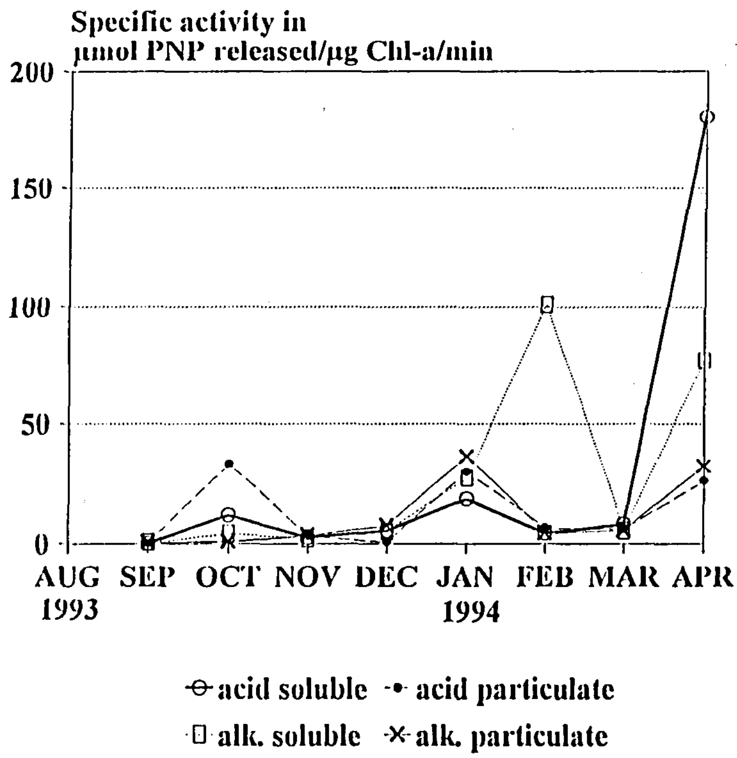

FIGURE 6: Specific Phosphatase activity in the Vaal River at Balkfontein.

4. DISCUSSION AND CONCLUSIONS

4.1 Nitrate Reductase (NR) activity

Two techniques, namely centrifugation and filtration were used to prepare samples for NR analysis. The results of both methods follow the same trend (Fig. 5). The filtration method, however, produced higher NR activity values than the centrifugation method. This means that the filtration method extracts NR more effectively from the cells, implying that the cells are broken more efficiently. This may be due to the glass microfibre filters used. When these filters are homogenised, the glass microfibres probably poke and cut the algal cells, thereby assisting in the breaking of the cells, and subsequently in releasing the enzyme.

Increases in chlorophyll-*a* concentrations, are associated with decreases in NR activity (compare Figs 4 and 5). In September 1993 (Chl-*a* 79.1 μg/l), November 1993 (Chl-*a* 49.7 μg/l) and March 1994 (Chl-*a* 115.8 μg/l) significant Chl-*a* concentration peaks can be seen in Fig. 4. The corresponding NR activities on these dates were low. In October, January and April, low Chl-*a* concentration values have corresponding high NR activities.

NR activity is thus inversely proportional to phytoplankton biomass indicated by the Chl-*a* concentration.

In Fig. 1 the dissolved nitrate nitrogen (NO_3-N) and dissolved inorganic nitrogen (DIP; PO_4-P) for the study period are illustrated. No definite correlation was shown between NO_3-N and NR activity (compare Figs 1 and 5). In August 1993, and January 1994, high nitrate nitrogen concentrations corresponded to high NR activities. This may indicate that when NO_3-N concentrations reach a certain maximum value, in this case above 600 µg/l, the NR activity increases. However, although NR activity might be expected to rise with increasing ambient NO_3-N concentrations, this cannot be assumed from the present results. In a similar experiment (Wynne & Berman, 1990), NR activity showed no correlation with NO_3^- concentrations. Similarly, there is no correlation between NR activities and TN (compare Figs 2 and 5). When NO_3^- tended to become the sole inorganic N source, the likelihood that the phytoplankton population would exhibit NR activity tended to increase, although no direct relationship was found between this ratio and the magnitude of NR activity (Wynne & Berman, 1990). Our results suggest, therefore, that control of NR in natural phytoplankton populations is complex and does not directly depend on ambient NO_3^- and/or NH_4^+ concentrations. NR activity in natural phytoplankton populations appears to depend on several environmental factors.

There is a positive correspondence between NR activities and DIP (PO_4-P) concentrations (compare Figs 1 and 5). A high PO_4-P concentration corresponds to a high NR activity. Similarly, a positive correlation exists between NR activities and TP concentrations (compare Figs 2 and 5). A high TP value corresponds to a high NR activity. Thus, high P levels apparantly activate NR activities.

In Fig. 3 DIN:DIP and TN:TP ratios are shown. No meaningful correspondence could be made concerning the N:P ratios and NR activities. In culture, the N:P ratio has been shown to be critical to the growth of various algae, but it is difficult to determine accurately in natural waters because of the multiple forms of N and P (Wynne, 1990). The N:P atomic ratio, however, has been shown to be an important ecological parameter because it can account for dominance and succession of algal species (Rhee, 1978). N:P ratios also determine structure and dynamics of phytoplankton populations in Lake Kinneret. High N:P ratios, approximately coincided with the onset of the *Peridinium* bloom. Moreover, certain enzyme activities, such as alkaline phosphatases (Wynne & Rhee, 1988) are also sensitive to the external N:P ratio.

4.2 Phosphatase (PASE) activity

In Fig. 6 the four different phosphatase enzyme activities are shown. These enzyme activities all follow the same trend. As was the case with NR activity, PASE activities were high when chlorophyll-*a* concentrations were low (compare Figs 4 and 6). PASE activity appears to be indirectly proportional to phytoplankton biomass as indicated in chlorophyll-*a* concentrations. Both groups of enzymes examined, namely PASE and NR, appear to be indirectly proportional to chlorophyll-*a*.

PASE activities appeared to correspond positively with NR activities (compare Figs 5 and 6). High NR activity had corresponding high PASE activity. The only exception is in February 1994 where the soluble alkaline phosphatase activity was high and the NR activity was low. NR activity decreased from February 1994 to March 1994 while the corresponding phosphatase activity increased slightly.

A positive correspondence existed between phosphatase activities and DIP (PO_4-P) concentrations. A high PO_4-P concentration corresponds to a high phosphatase activity. Similarly, a positive correlation exists between phosphatase activities and TP (total phosphorus) concentrations (compare Figs 2 and 6).

High TP concentrations correspond to high phosphatase activities. A high PO_4-P concentration also corresponds to a high NR activity. Thus, high P levels activate phosphatase and NR activities. Enzyme activity of the two enzymes studied, is thus stimulated by high P levels.

In Fig. 3 DIN:DIP and TN:TP ratios are shown. No significant correlations could be made concerning these N:P ratios and phosphatase activities.

5. SUMMARY

5.1 **Enzyme activity appeared to be inversely proportional to algal biomass** as indicated by chlorophyll-*a* concentrations.

5.2 No definite **correlation** was shown **between NO_3-N and enzyme activities.**

5.3 **High PO_4-P concentrations** corresponded to **increased PASE and NR activities.**

5.4 **NR activity** appeared to **correspond positively with PASE activity.**

5.5 **Both enzymes** seem to be **stimulated by similar conditions.**

6. ACKNOWLEDGEMENTS

We wish to thank Johnn Prinsloo for analysing water samples for the major nutrients and for technical assistance.

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CHAPTER 6: PHYTOPLANKTON BIOMASS AND ENVIRONMENTAL VARIABLES IN THE VAAL RIVER AT BARRAGE AND STILFONTEIN

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1. INTRODUCTION

Physical, chemical and biological environmental variables affect phytoplankton growth and productivity. In general, however, physical, chemical and biological environmental variables affect organisms in a mutually interdependent fashion (Odum, 1971).

In the Vaal River, aspects of the mutual interdependence of environmental variables have been investigated by Pieterse & van Zyl (1988). Temporal changes in phytoplankton structure were shown to be related to discharge, N and P loading and turbidity. In addition, Pieterse (1993) postulated that interrelationships most probably exist between phytoplankton species composition and biomass and different phases of the water purification process. However, conditions unique to the environment in which the algae grow, most probably select for specific algae that show characteristics that cause specific problems in the different phases of purification. Because of the interrelationary nature of the different aspects referred to, conditions in the treatment plant are therefore apparently primarily determined by conditions in the river (Pieterse, 1993).

An integrated multifaceted approach including all relevant aspects in the environment must be investigated with the view to develop a better understanding of the conditions responsible for the growth of phytoplankton populations in the river. Therefore, in order to explain changes in phytoplankton density as well as changes within phytoplankton species assemblages, all environmental variables that possibly influence the phytoplankton, have to be investigated and evaluated.

In the study statistical correlation evaluations of Vaal River chemistry, temperature and chlorophyll data for a period of almost eight years at four sampling localities representing the upper and lower sections of the middle Vaal River region have been made. The statistical approaches followed by Caljon (1983), Edson & Jones (1988), Gould *et al.* (1986), Bartell *et al.* (1987) and Claflin (1987) were used in this study. The association of variables of Vaal River Barrage water immediately upstream of the Rand Water Barrage sluice gates were compared with that of approximately 7 m deep water released into the Vaal River, as well as with that of water at Lindeques Drift after it flowed approximately 9 km downstream from the Barrage. In addition, associations of variables of these waters were compared with those of water at Stilfontein after it flowed another approximately 140 km downstream from Lindeques Drift. The section of the river between Lindeques Drift and Stilfontein is most probably subjected to intensive pollution effects.

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* Work done at the Department of Botany and Genetics, UFS, Bloemfontein

2. METHODS

Water chemistry measurements were made at the Rand Water Barrage from January 1984 to December 1991 at two different sampling localities. The first was on surface water samples taken immediately upstream of the Barrage sluice gates (Top 0) where the water is approximately 7 m deep. The second was on samples taken immediately downstream of the Barrage sluice gates (V17). Because bottom water is released by the sluice gates, these measurements represent the water that is released from the Vaal River Barrage into the Vaal River. Water chemistry measurements were made at Lindeques Drift (V18), approximately 9 km downstream of the Barrage. The two Barrage and one Lindeques Drift samples were analysed by Rand Water. At the two localities downstream from the Barrage (V17 and V18) measurements were taken weekly and at the locality immediately upstream of the sluice gates of the Barrage (Top 0) only bimonthly. The following 15 variables were measured, using standard techniques: temperature, turbidity, pH, chlorophyll, alkalinity, sulfate, total dissolved solids, conductivity, ammonium, phosphate, nitrate, Ca, Mg, Na, Cl, and oxygen. The same variables were tested at V17 and V18, but at Top 0 NH_4 rather than total dissolved solids (TDS) were included in the analyses.

At Stilfontein water chemistry measurements were made from January 1984 to September 1991 on the Vaal River by the Western Transvaal Regional Water Company. By using standard techniques, eighteen chemical variables were measured, some daily (temperature, suspended solids, turbidity, pH, chlorophyll, alkalinity, sulfate, total dissolved solids and conductivity) and others weekly (phosphorus, nitrate, ammonium, Ca, Mg, Na, Cl, Mn and Cu).

A computerised data base of the available values for the three sampling points was constructed for the water samples taken at the Barrage and Lindeques Drift. Simple regressions yielded a correlation matrix for the 15 variables. Principal component analysis (PCA; Pielou, 1984) of all the variables, excluding chlorophyll, was used to extract the main components explaining the variation across the data set. Component scores were calculated for the first 6 components for each of the sampling points for the number of months sampled.

For the Stilfontein samples a computerised data base of weekly values for the 18 variables was constructed (weekly means were calculated for those variables measured daily). Weeks which had missing data for one or more variables were excluded from the analyses. This reduced the total number of weeks for which data was used from 407 to 224. Simple regressions yielded a correlation matrix for the 18 variables. Principal components analysis (PCA; Pielou, 1984) of all the variables, excluding chlorophyll, was used to extract the main components explaining the variation across the data set. Component scores were calculated for the first six components for each of the 224 weeks. These were regressed against the weekly chlorophyll concentration values to assess whether any of the principal components derived from the chemistry data explained a significant proportion of the variation in chlorophyll concentrations over the almost eight year period. Chlorophyll concentration was taken to be the index of algal biomass concentration in the water.

3. RESULTS AND DISCUSSION

3.1 VAAL RIVER AT BARRAGE AND LINDEQUES DRIFT

The correlation matrixes are given in Tables 1 to 3. Positive and negative correlations significant at $P \leq 0.05$ are shown. There are two groups of highly intercorrelated variables, namely turbidity and orthophosphate on the one hand and NO_3, TDS, chlorophyll, pH, conductivity, alkalinity and ionic species (Ca, Mg, Na, Cl and SO_4) on the other hand. In the case of Barrage Top 0 it is only turbidity and orthophosphate that correlated against chlorophyll, pH, conductivity, alkalinity and the ionic species. In all three cases the two groups are negatively correlated with each other.

TABLE 1: Correlation matrix for Vaal River water chemistry and temperature data immediately above the Barrage sluice gates (representing surface water; Top 0), January 1984 to December 1991. Only significancies of $P \leq 0.05$ are shown.

	Temperature	Turbidity	Phosphate	Nitrate	Ammonium	Chlorophyll	pH	Conductivity	Alkalinity	Calcium	Magnesium	Sodium	Chloride	Sulfate	Oxygen
Temperature															
Turbidity	–														
Phosphate		+													
Nitrate	–														
Ammonium															
Chlorophyll															
pH				–		+									
Conductivity		–													
Alkalinity		–						+							
Calcium		–	–					+	+						
Magnesium		–	–					+	+	+					
Sodium		–						+	+	+	+				
Chloride		–						+	+	+	+	+			
Sulfate		–	–					+	+	+	+	+	+		
Oxygen							+		+						

The association of the variables into two groups can be interpreted as a phenomenon when water is turbid and the orthophosphate concentration is high, while the concentration of dissolved ions, chlorophyll, pH and alkalinity are low. It may show that when rainfall is high, the water is turbid, discharge is high and the concentration of dissolved ionic species in the water is low. During periods of lower rainfall, water is less turbid, discharge is lower and the retention time of water in the river is longer, thus higher concentrations of dissolved ions occur in the water.

At all three sampling localities chlorophyll was strongly positively correlated with pH. At V17 (water released from the Rand Water Barrage) chlorophyll shows positive correlations with conductivity, Ca, Na, Cl, SO_4 and O_2. Chlorophyll is not correlated significantly with temperature, orthophosphate or any form of nitrogen. The positive correlation between algal growth indicated by high chlorophyll concentration and high pH might be a cause-effect relationship where algal growth depletes CO_2 and increases OH^- levels in the water.

TABLE 2: Correlation matrix for Vaal River water chemistry and temperature data immediately below the Barrage sluice gates (representing water released from 7 m deep; V17), January 1984 to December 1991. Only significancies of $P \leq 0.05$ are shown.

	Temperature	Turbidity	Phosphate	Nitrate	TDS	Chlorophyll	pH	Conductivity	Alkalinity	Calcium	Magnesium	Sodium	Chloride	Sulfate	Oxygen
Temperature															
Turbidity															
Phosphate	+	+													
Nitrate	–	–	–												
TDS	–	–		+											
Chlorophyll					+										
pH					+	+									
Conductivity		–		+	+	+	+								
Alkalinity		–			+		+	+							
Calcium	–	–		+	+	+	+	+	+						
Magnesium	–	–	–	+	+		+	+	+	+					
Sodium	–	–		+	+	+	+	+	+	+	+				
Chloride	–	–			+	+	+	+	+	+	+	+			
Sulfate		–		+	+	+	+	+	+	+	+	+	+		
Oxygen	–		–		+	+	+	+	+	+	+	+	+	+	

TABLE 3: Correlation matrix for Vaal River water chemistry and temperature data at Lindeques Drift, V17), January 1984 to December 1991. Only significancies of P ≤ 0.05 are shown.

	Temperature	Turbidity	Phosphate	Nitrate	TDS	Chlorophyll	pH	Conductivity	Alkalinity	Calcium	Magnesium	Sodium	Chloride	Sulfate	Oxygen
Temperature															
Turbidity	+														
Phosphate	+	+													
Nitrate		–	–												
TDS		–	–	+											
Chlorophyll															
pH	–				+	+									
Conductivity	–	–	–	+	+		+								
Alkalinity	–	–	–	+	+		+	+							
Calcium		–		+	+		+	+	+						
Magnesium	–	–	–	+	+		+	+	+	+					
Sodium	–	–	–	+	+		+	+	+	+	+				
Chloride	–	–	–	+	+		+	+	+	+	+	+			
Sulfate		–	–	+	+		+	+	+	+	+	+	+		
Oxygen	–		–	+			+								

The first six components yielded by the PCA explained 92.05 % (Table 4), 94.88 % (Table 5) and 93.09 % (Table 6) respectively for Barrage water above the sluice gates (Top 0), Barrage water below the sluice gates (V17) and water at Lindeques Drift (V18) of the total variation in the complete data set (excluding chlorophyll concentration). Variable weightings in the components, with an interpretation of the vectors represented by die components, is given in Tables 4 to 6.

In the Barrage water above the sluice gates (Table 4; Top 0), the strongest component (47.78% of the variation) shows when the turbidity is high while there is a low concentration of ions in the water. Component 2 shows high pH and O_2 concentration *versus* low nitrogen (NH_4 and NO_3) concentration. This relationship explains the situation in the correlation matrix where high pH and chlorophyll concentrations are correlated with each other. Growing algae will reduce the nitrogen concentration in the water, hence the negative correlation. Component 3 shows that in summer with high temperatures low nitrate concentrations were present. In the fourth component it is orthophosphate *versus* SO_4 concentrations. Component 5 deals with a summer/winter relationship. When temperature and turbidity was high, the ammonia concentration was low.

TABLE 4: Principal components from the PCA of the Vaal River water chemistry and temperature data immediately above the Barrage sluice gates (representing surface water; Top 0), January 1984 to December 1991.

Component No	% Variation accounted for	Gradient	Interpretation
1	47.78	(+)Turb (-)Ca,Mg,Na,SO_4,Cl	Turbid waters *versus* high conc. of dissolved salts
2	13.84%	(+)pH,O_2 (-)NH_4,NO_3	Oxygen rich water with high pH when N is deficient
3	11.49	(+)Temp (-)NO_3	High water temp. in summer *versus* high NO_3 concentrations
4	8.32	(+)phosphate (-)SO_4	High orthophosphate *versus* high SO_4 conc.
5	6.45	(+)Temp,Turb (-)NH_4	Warm, turbid water *versus* high NH_4 conc.
6	4.17	(+)Turb,O_2 (-)phosphate	Turbid, oxygen rich water *versus* high phosphate conc

In the water released from the Barrage (Table 5; V17) the first component shows orthophosphate *versus* conductivity, TDS, and dissolved ions. Component 2 shows high temperatures in summer as well as high nitrate concentration. In winter with low temperatures, pH and O_2 concentration were high. High pH indicates algal growth and increasing algal growth commences in autumn and generally develops into blooms in winter and spring. Component 3 of V17 shows the same trend as that of Top 0 - namely low nitrogen concentrations in summer. The rest of the components weighted low and are rather "pick outs" of certain sampling times rather than trends in the data.

At Lindeques Drift (Table 6; V18) component 1 shows the same trend as component 1 of V17. Component 2 have high temperatures in summer with corresponding low oxygen concentrations in the river. When pH and orthophosphate are high, there is a deficiency of nitrate in the water. High pH indicates algal growth which needs nitrate for growth. The rest of the components weighted low and again are "pick outs" rather than data trends.

TABLE 5: Principal components from the PCA of the Vaal River water chemistry and temperature data immediately below the Barrage sluice gates (representing water released from 7 m deep; V17), January 1984 to December 1991.

Component No	% Variation accounted for	Gradient	Interpretation
1	61.38	(+)phosphate (-)conductivity, TDS,Ca,Mg,Na, Cl, SO_4	High phosphate *versus* high conc of dissolved salts
2	12.06	(+)Temp,NO_3 (-)pH,O_2	Warm, NO_3 rich water *versus* high pH water rich in O_2
3	9.42	(+)Temp (-)NO_3	Warm, summer water *versus* water rich in nitrate
4	5.11	(+)phosphate (-)pH	High phosphate *versus* high pH waters
5	4.23	(+)O_2 (-)pH	Oxygen rich water with low pH
6	2.68	(+)pH (-)alkalinity	High pH *versus* alkaline water

TABLE 6: Principal components from the PCA of the Vaal River water chemistry and temperature data at Lindeques Drift (V17), January 1984 to December 1991.

Component No	% Variation accounted for	Gradient	Interpretation
1	59.15	(+)Turb (-)conductivity, TDS,Ca,Mg,Na, Cl,SO_4	Turbid water *versus* high conc of dissolved salts
2	13.32	(+)Temp (-)O_2	Warm water which is oxygen deficient
3	8.17	(+)pH,phosphate (-)NO_3	High pH, phosphate *versus* high NO_3 conc
4	5.49	(+)phosphate (-)alkalinity	High phosphate *versus* alkaline water
5	3.72	(+)O_2 (-)Turb	Oxygen rich water *versus* turbid water
6	3.24	(+)alkalinity (-)O_2	Alkaline *versus* oxygenated water

3.2 VAAL RIVER AT STILFONTEIN

The correlation matrix for the Vaal River Stilfontein is given in Table 7. Positive and negative correlations significant at $P \leq 0.05$ are shown. There are two main groups of highly intercorrelated variables, namely temperature, suspended solids, turbidity and phosphorus on the one hand, and ionic species (Na, Cl, SO_4 etc.), along with associated properties such as conductivity, total dissolved solids and alkalinity (hence also pH) on the other hand. The two groups are negatively correlated with each other. NH_4-N and NO_3-N are intermediate between the two groups, with NO_3-N being more closely related to the temperature/suspended solid group and NH_4-N having closer affinities to the conductivity/ionic species group.

TABLE 7: Correlation matrix for Vaal River water chemistry and temperature data at Stilfontein for the period January 1984 to September 1991.

	Temperature	Susp solids	Turbidity	Phosphate	Nitrate	Ammonium	pH	Chlorophyll	Alkalinity	Calcium	Magnesium	Sodium	Chloride	Sulphate	TDS	Conductivity
Temperature																
Susp solids	+															
Turbidity	+	+														
Phosphate	+	+	+													
Nitrate		+	+													
Ammonium																
pH	–	–	–	+	+	+										
Chlorophyll	–				+		+									
Alkalinity	–	–	–	–	+		+									
Calcium	–	–	–	–	+		+		+							
Magnesium	–	–	–	–	–	+	+	+	+	+						
Sodium	–	–	–	–	–		+	+	+	+	+					
Chloride	–	–	–	–	–		+		+	+	+	+				
Sulfate	–	–	–	–			+	+	+	+	+	+	+			
TDS	–	–	–	–	–		+	+	+	+	+	+	+	+		
Conductivity	–	–	–	–		+	+		+	+	+	+	+	+	+	
Mn	–	–	–		–		+		+	+	+	+	+	+	+	+
Cu	–															

The association of variables into two main groups at Stilfontein is interpreted as a summer *versus* winter phenomenon, i.e. summer rains in the highveld region occur in short, intense spates and the resulting surface flow from the catchment has a high erosive capability, possibly resulting in high silt loading of the river. In winter, rain is infrequent and water flow from the catchment will be mainly subterranean, with a low silt concentration. Even surface flow will be less energetic and erosive in winter. The longer retention of water in the catchment in winter allows more salts to dissolve into the water, hence the greater conductivity, dissolved ionic species etc., in winter.

At Stilfontein chlorophyll concentration was strongly positively correlated with pH ($r=0.62$, $P=0.00001$) and negatively with temperature ($r=-0.43$, $P=0.0001$). It showed weaker, but still significant, correlations with NO_3-N, total dissolved solids, Na, Mg and SO_4. This suggests that algal growth is greater in winter than in summer and there might be a cause and effect relationship between chlorophyll concentration and pH and alkalinity, i.e., algal growth depletes CO_2 and increases OH^- levels in the water.

The first six components yielded by the PCA explained 81 % of the total variation in the whole Stilfontein data set (excluding chlorophyll concentration). Variable weightings in the components, with an interpretation of the vectors represented by the components, is given in Table 8. All 6 components reflect in some aspect the strong summer-winter division of the data shown in the correlation matrix.

TABLE 8: Principal components from the PCA of the Vaal River water chemistry and temperature data at Stilfontein for the period January 1984 to September 1991.

Component No	% Variation accounted for	Gradient	Interpretation
1	45.7	(+)Turb,Suspsol, Temp,phosphate (-)TDS,Cond,Ca,Cl, Na,Mg,SO_4,Alk	Summer high siltload *versus* winter with high conc of dissolved salts
2	9.4	(+)NH_4,Suspsol,Turb pH,phosphate (-)alkalinity	Turbid, high pH water rich in phosphate *versus* alkalinity
3	7.8	(+)Temp (-)NO_3	Summer warm water *versus* high NO_3
4	7.0	(+)pH (-)Temp	High pH *versus* warm water
5	6.0	(+)Mn,Alk,NH_4 (-)Cu	Alkaline, NH_4, Mn rich water *versus* high Cu
6	5.3	(+)NO_3 (-)Suspsol, Turb	NO_3 rich *versus* turbid water

The strongest component (1) is a straightforward gradient of warm, highly turbid water to cold water with higher concentrations of dissolved salts. Component 2 is difficult to interpret and distinguishes periods when the water was turbid and possessed moderate to high concentrations of NH_4-N. Alkalinity was weakly negative on this vector. Component 3 distinguishes situations when the water was warm and NO_3 deficient and when it was colder but possessed higher NO_3 concentrations. Component 4 is strongly seasonally related and represents the gradient of cold, high pH winter waters and warm, acidic winter waters.

The remaining components yielded by the PCA do not relate to the overall variation in the data set but either "pick out" certain weeks (e.g. component 5) or separate small groups of sites at one end of a previous component (e.g. component 6 separates cases at the positive end of component 2).

To test whether the vectors represented by components 1 to 6 can account for the variation in chlorophyll concentrations (i.e. algal growth) at Stilfontein, their component scores for each week were regressed against the corresponding chlorophyll values (Table 9). Component 3 scores were highly negatively, and component 4 scores highly positively, correlated with chlorophyll concentrations. Both indicate that algal growth is greater in winter, especially when NO_3 concentrations are high (component 3) and the greater algal growth is associated with high pH (component 4). As suggested above, the latter might be a cause and effect relationship. The summer-winter definition is also strongly associated with the significant negative correlation between component 1 scores and chlorophyll concentrations. Algal growth is greatest in the colder, clearer, ionically rich winter water than in turbid, nutrient poor summer water, as evidenced by the significant negative correlation of chlorophyll with scores on component 1. The positive correlation of chlorophyll with the component 2 scores possibly indicate periods of enhanced algal growth when the water is turbid but when NH_4 and PO_4 concentrations are moderate to high.

TABLE 9: Percentages of the variation in chlorophyll concentrations accounted for by components 1 to 6 from the PCA of the Vaal River water chemistry and temperature data at Stilfontein for the period January 1984 to September 1991.

Component	Association	% Variation accounted for	Significance
1	-ve	3.01	P = 0.01
2	+ve	2.86	P = 0.05
3	-ve	25.75	P = 0.0001
4	+ve	10.68	P = 0.0001
5	-ve	1.17	P = 0.11
6	-ve	0.03	P = 0.79

The time courses of chlorophyll concentrations and the weekly component scores from January 1984 to September 1991 at Stilfontein clearly showed seasonal variation in chlorophyll concentrations, with maximum values occurring from midwinter to spring. Summer values are generally low. This is almost the reciprocal pattern to those shown by the scores on components 1 and 3, but complimentary to those shown by the scores on components 2 and 4. At some times this complimentarity is lost, e.g. early in 1989 in the case of component 2 and early in 1988 in the case of component 4.

4. CONCLUSIONS

The data base used in this study had many missing values. Key variables such as inorganic nitrogen and phosphorus have oftentimes not been measured. A suitable sampling strategy must therefore be developed. Whatever sampling strategy is decided upon, special effort must be made to ensure that those variables which are only measured weekly are, in fact, measured each week. This has not been the case in the past and the useful data base for 1894 to 1991 has been reduced by half because of it.

Analysis of the water chemistry, temperature and chlorophyll concentration data set for the Vaal River at Barrage, Lindeques Drift and Stilfontein from 1984 to 1991 shows that the major variations in the water properties can be ascribed to seasonally-related variations in temperature, suspended solids (and hence turbidity), coupled with directionally opposite variations in dissolved salts (and hence conductivity). The importance of the effect of temperature was also shown for Lake Michigan (Claflin, 1987). In Lake Michigan the summer phytoplankton assemblages varied directly with temperature and pH, and inversely with nitrogen and silica, while autumn phytoplankton varied directly with total phosphorus as well as with PO_4-P. In the Vaal River, however, algal growth (i.e. chlorophyll concentration) was greatest in winter when the water is clear and cold and lowest in summer when it is turbid and warm. In the Vaal River there was also a significant seasonal variation in water pH, which was higher in winter, and this is thought to be a direct result of enhanced algal growth. A similar seasonality of chlorophyll concentration was demonstrated at Lindeques Drift and at the Barrage below the sluice gates.

In general, the correlation matrices for Vaal River variables at the Barrage below the sluice gates and Lindeques Drift showed similar associations which were, in turn, different from those of the surface water in the Barrage immediately above the sluice gates. The differences between the surface and bottom waters in the Barrage indicated that vertical depth-related processes and conditions over a depth of approximately 7 m have a more significant effect on variables than horizontal distance-related processes and conditions along a distance of approximately 9 km.

Seasonal variations in the two forms of inorganic N at Stilfontein seem to be different, with the one being associated with the "summer" regime (NH_4) and the other with the "winter" regime (NO_3) of the river water. This might be associated with seasonality of rainfall, run-off patterns and agricultural practices and/or to N transformation processes in the soil. Alternatively, it could be related to eutrophication or industrial pollution. In Finnish lakes Varis *et al.* (1989) and Varis (1991) showed that algal populations had a strong positive correlation with total phosphorus and water temperature and a negative correlation with N:P ratios and ammonia. In the Vaal River at Lindeques Drift and the Barrage between the sluice gates NO_3 appears to have been strongly associated with the "winter" regime. The reason for the differences between the upper and lower sections of the middle Vaal River region regarding the seasonality of NO_3 is not clear.

Considered overall, the roles of NH_4-N and NO_3-N in the results of the analyses presented here are ambiguous and hard to interpret. The close correlation of suspended solids and turbidity suggests that, for modelling purposes, only one of these will be necessary. Similar recommendations apply for conductivity, total dissolved solids and possibly alkalinity.

In the above paragraphs reference have been made to the relationships between environmental variables and phytoplankton biomass as indicated by chlorophyll. Edson & Jones (1988), using PCA analysis on phytoplankton species composition, demonstrated seasonal patterns of phytoplankton succession with blue-green dominance in summer, diatoms and chrysophytes in spring and fall, and cryptophytes in winter in Lake Fairfax, a small reservoir in Virginia, USA. In the context of Vaal River research, phytoplankton species composition and seasonal succession patterns must also be investigated in relation to the environmental variables employed in this study.

The application of the PCA method to investigate associations between physical, chemical and biological variables resulted, amongst others, in the elucidation of seasonal aspects of algal growth, showing higher algal growth during colder months. Should changes in the seasonality occur, i.e. should summer blooms replace winter blooms or should different years show different associations, PCA analyses will be able to identify and quantify the changes and differences in terms of their statistical significance. However, Bartell *et al.* (1987) concluded that the statistical comparison between phytoplankton and environmental changes is only the first step in linking species dynamics to changes in a complex environment.

5. SUMMARY

5.1 **Statistical evaluations of Vaal River chemical, temperature and chlorophyll data** for a period of almost eight years at four sampling localities **were made** in the upper and lower sections of the middle Vaal River

5.2 Two groups of highly intercorrelated variables were demonstrated:

5.2.1 At the **Barrage and Lindeques Drift** these two groups were turbidity and orthophosphate on the one hand and NO_3, TDS, chlorophyll, pH, conductivity, alkalinity and ionic species on the other hand. The two groups were negatively correlated with each other. The association into two groups can be interpreted as a phenomenon when the water is turbid and the orthophosphate concentration is high, while the concentration of dissolved ions, chlorophyll, pH and alkalinity are low. In addition, chlorophyll was strongly positively correlated with pH, a cause-effect relationship where algal growth depletes CO_2 and increases OH^- levels in the water.

5.2.2 At the **Stilfontein** sampling locality the two main groups were temperature, suspended solids, turbidity and phosphorus on the one hand, and ionic species (Na, Cl, SO_4, etc.) along with associated properties (conductivity, TDS and alkalinity) on the other hand. The association into two main groups can be interpreted as a summer *versus* winter phenomenon During summer rains, erosion results in high silt loadings of the river. During winter, rain is infrequent and the water will have a low silt concentration. The longer retention of water during winter allows more salts to dissolve, and therefore greater conductivity, dissolved ionic species, etc. during winter months. In addition chlorophyll was strongly positively correlated with pH, and negatively with temperature.

5.3 Algal growth (measured by chlorophyll concentration) was greatest in winter when the water was clear and cold. pH was also higher during winter which was thought to be a direct result of the enhanced algal growth during winter.

5.4 Correlation matrices for Vaal River variables at the Barrage below the sluice gates and Lindeques Drift showed similar associations, but were different from the associations of the surface water in the Barrage immediately above the sluice gates. **Vertical depth-related processes and conditions over a depth of approximately 7 m therefore have a more significant effect on variables than horizontal distance-related processes and conditions along a distance of approximately 9 km.**

5.5 **The close correlation between suspended solids and turbidity as well as between conductivity, total dissolved solids and alkalinity suggest that, for modelling purposes, only one of each of these will be useful.**

6. ACKNOWLEDGMENTS

Mr M Steynberg and Ms A Maritz, Rand Water, Vereeniging, provided chemical and physical data on the Vaal River at the Rand Water Barrage and Lindeques Drift, while data used in the analysis on chemical and physical conditions in the Vaal River at Stilfontein was provided by Mrs M Krüger and Mr J Pietersen, Western Transvaal Regional Water Company, Stilfontein.

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CHAPTER 7: MODELLING ALGAL GROWTH IN THE VAAL RIVER

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1. INTRODUCTION

As indicated in the introduction to this report, the prediction of the development of algal blooms in a river is of great importance in water resource management. In this part of the report, the foundation of a mathematical model is described. This basic model is intended to eventually assist researchers and the relevant authorities by providing them with a tool which can lead to a better understanding of aspects of algal growth, and which might enable meaningful short and long-term water quality predictions.

Originally, algal growth in the Vaal River was modelled by taking only water temperature and underwater light climate variations into account. Within the framework of this assumption, and assuming that assemblages containing up to *n* algal groups, are responsible for the blooms, we derived a basic scheme that may be used for the modelling of algal growth (Cloot *et al.,* 1992). This scheme is illustrated in Figure 1.

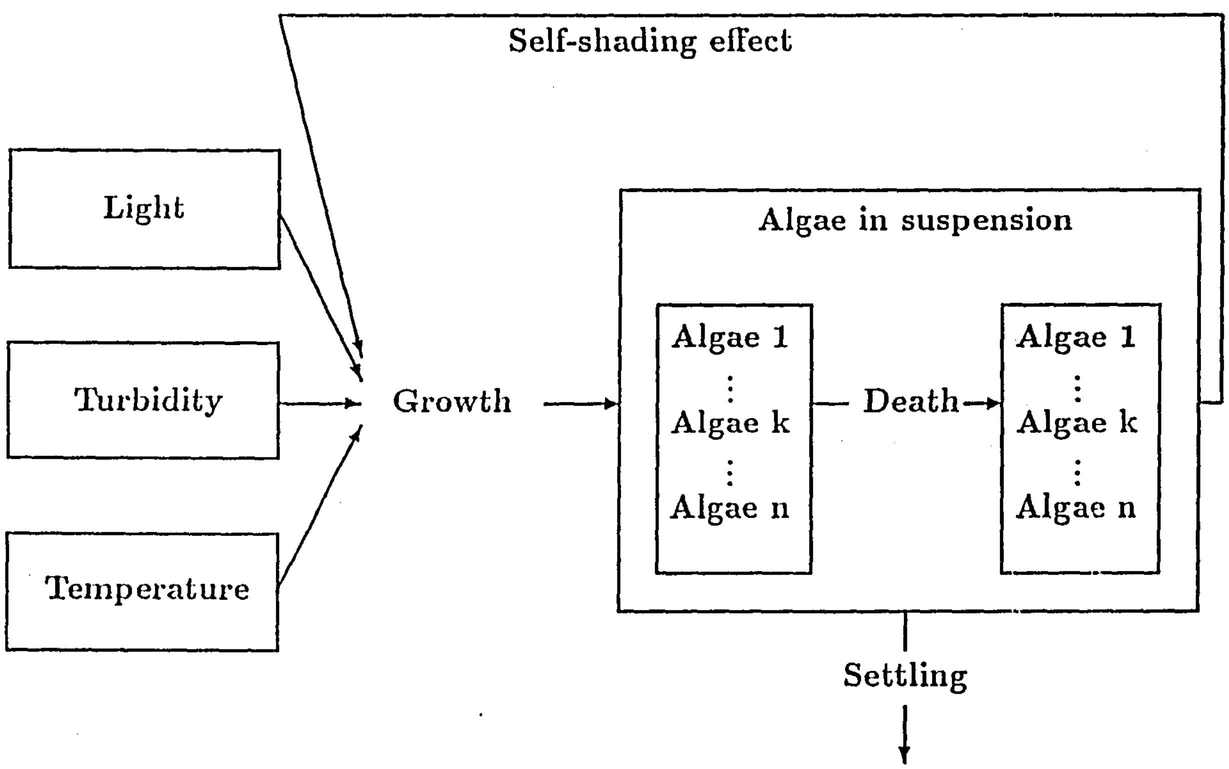

FIGURE 1: A schematic representation of the basic light-temperature Vaal River n-algal growth model

The transcription of the scheme in Fig. 1 in terms of mathematical relations lead to a system of n pairs of coupled non-linear differential equations (Cloot & Schoombie, 1994)

$$\begin{aligned} \dot{x}_{1_i} &= [-k_{D_i} + k_{S_i}) + k_{G_i}(T, I, \underline{K}_j; j = 1, N)]x_{1_i} \\ \dot{x}_{2_i} &= k_{D_i}x_{1_i} - k_{S_i}x_{2_i} \end{aligned} \tag{1}$$

that depend explicitly on parameters which could be classified in two categories, namely (1) environmental variables and (2) variables that are specific for a particular algal group. The environmental parameters that were needed within the frame of this basic model, are described in Table 1.

TABLE 1: Environmental parameters

I_{max}	:	maximal irradiance available on the water surface
$\mu(t)$	:	Cosine of solar zenith angle
T	:	the water temperature
S(t)	:	total concentration of inorganic material suspended in the water
Z_O	:	depth of the mixed layer
k_w	:	light extinction coefficient for pure water
c_s	:	light extinction coefficient for suspended inorganic solids

Furthermore, the reactions of algae to variations in environmental variables differ from group to group, and the sensitivity of an algal group "i" can be represented by a set of parameters $\underline{K}_i$ whose components are described in Table 2.

TABLE 2: Parameter set representing the i-th algal category

I_{opt}	:	optimal light intensity for growth
k_G	:	maximum algal growth rate
T_{opt}	:	optimal temperature for algal growth
T_{min}	:	minimum temperature for algal growth
k_D	:	algal dying rate
k_S	:	algal settling rate
k_x	:	self-shading coefficient for algae

(The index "i" has been omitted here for convenience).

This model was calibrated on data available from the Vaal River and tested on *in situ* measurements from the Stilfontein site, during a three year period starting January 1985.

Typically, during the winter to spring period of each of these years, two algal blooms were observed. The first of these blooms was due mainly to diatoms, while the second was mainly caused by green algae. Assuming that only assemblages containing 2 algal types were possible, and using the weekly averaged water temperature and suspended inorganic solid concentration as input data, we solved the system of equations (1) numerically for the winter-spring period of each of the 3 years and computed the total algal biomass in terms of the chlorophyll-*a* concentration. As an example, Fig. 2 shows the model simulation (solid line) together with measurements made during 1986 at Stilfontein.

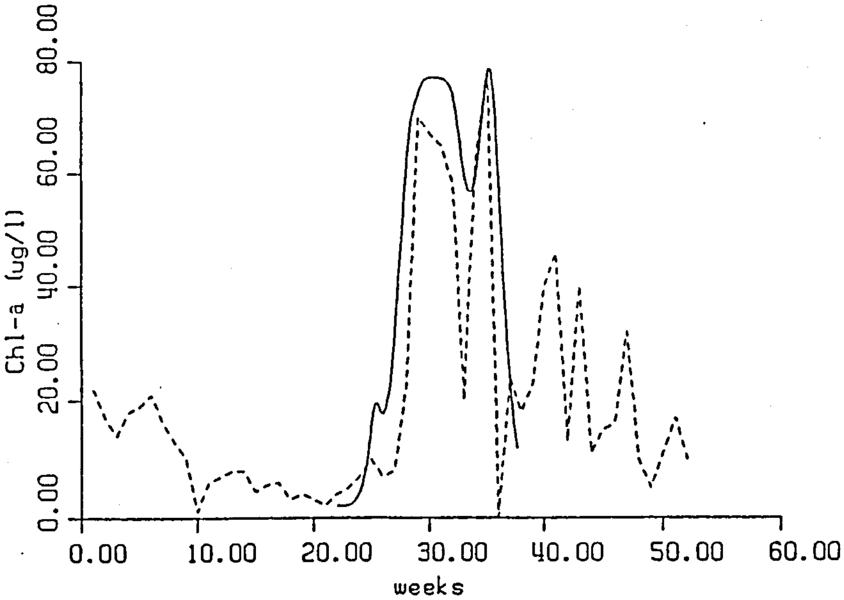

FIGURE 2: Simulated (solid line) and actual (dashed line) total chlorophyll-*a* concentration in the Vaal River at Stilfontein during the winter-early spring period of 1986. (Week 0 starts on 1 January)

The agreement between simulated and measured values was, overall, fairly good during the winter algal bloom period. However, it was obvious that better results were obtained in the modelling of the second algal bloom than the first one. In an attempt to improve the situation, an additional step was taken by explicitly including the possible effects of variations in dissolved silicon concentration in the water on the growth coefficient. More details are provided in the next section.

2. DIATOMS, SILICON UPTAKE AND DISSOLVED SILICON CONCENTRATION

Unlike other types of algae, diatoms are able to absorb the element silicon dissolved in the water in the form or orthosilisic acid $Si(OH)_4$. The availability of this element was shown to be essential in the formation of the protective covering of diatom cells, which acts as a shield against predators and parasites present in the environment, and also as a protection against disturbances of a mechanical nature (Werner, 1977). Should the dissolved silicon concentration in the water be depleted, diatom growth and- development would be negatively affected.

Since only algae belonging to the diatom group are able to affect the concentration of dissolved silicon in the water, the monitoring of this component of the ecological system during an algal bloom could provide an elegant and powerful means to retrieve, a posteriori, the extent to which algae belonging to the diatom group were responsible for an algal bloom. It should be noted that the usefulness of silicon ass an indicator of the presence of diatoms is higher than any other nutrient. In particular, it is far more effective than simply taking into account the water temperature observed during the bloom, since there is no overlap between the range of temperatures favourable for growth of diatoms and green algae, for example. Furthermore, if we assume that the necessary information on the saturation concentration of dissolved silicon in the water is available, the measuring of available Si concentrations in the water will allow the researcher to obtain information about the relative abundance of diatoms during a mixed bloom, i.e. an algal bloom compound of algae belonging to more than one algal group.

The modelling of the dissolved silicon concentration is done in two phases: Firstly the determination of dissolved silicon concentration in the water as a mechanical and chemical process. This concentration will be referred to as the saturation concentration, Si^{sat}.

Secondly the modelling of the silicon uptake by diatoms, with particular attention given to the influence of silicon uptake by diatoms on the dissolved Si concentration in the water, and the effect of dissolved Si concentration on the growth coefficients of the diatoms.

2.1 DISSOLVED SILICON CONTENT OF THE WATER AS A MECHANICAL AND CHEMICAL PROCESS

It is well known that the silicon uptake by diatoms is in the form of orthosilisic acid, $Si(OH)_4$ (Werner, 1977). Thus, the following path for silicon uptake is possible:

$$SiO_2 \ (+H_2O) \rightleftharpoons Si(OH)_4 \rightarrow \text{Diatom uptake} .$$

Assuming that the concentration of dissolved SiO_2 is governed by a chemical process only, we consider the following equilibria

$$SiO_2|_{amorphous} + 2H_2O \rightleftharpoons Si(OH)_4 \qquad (2)$$

characterised by the equilibrium constant

$$K(T)|_{am} = \frac{[Si(OH)_4]}{[SiO_2]_{am}[H_2O]} \qquad (3).$$

where [.] stands for the concentration of element "." (in Mole/l) and "am" is an abbreviation of "amorphous".

TABLE 3: Balkfontein: experimental data for 1986-1987

Date	Chl-*a* (μg/l)	Algal group	Temperature (°C)	Si (mg/l)	Q (m^3/sec)
86-02-18	25	Green + Diatoms	26	2.8	17
86-04-03	22	Green + Diatoms	21	3.83	12
86-06-24	10 *	?	14	3.37	15
86-08-29	65	Diatoms	14.5	0.52	25
86-10-20	40	?	1.8	1.87	22
86-12-09	35	Green + Diatoms	24	4.4	17
87-02-26	38	Green + Diatoms	27	2.57	1.6
87-03-24	25	?	24	2.56	1.5
87-04-21	25 *	?	23	4.94	4
87-05-25	30 *	Green	20	4.07	0.7
87-06-30	20 *	?	10	3.14	20
87-08-04	80	Diatoms	11.1	0.62	26
87-08-17	110	Diatoms	15	0.56	27
87-09-16	61	Diatoms + Green	21.5	0.51	12
87-10-13	83 *	Green	20	4.14	13
87-11-18	86 *	Green	24	4.71	32
87-12-08	83 *	Green	26	4.98	136

Furthermore, since the $Si(OH)_4$ concentration is small compared to that of pure water, and since SiO_{2am} is in a solid form it can be assumed that the hypothesis of infinite dilution is applicable, and therefore (3) reduces to

$$K_{am}(T) \sim [Si(OH)_4] \qquad (4)$$

where $K_{am}(T)$ takes the value

$$K_{am}(25°C) \sim 10^{-2.7}, \qquad (5)$$

if the pressure is 1 Atmosphere. This latter result can be considered insensitive to pH variations up to

$$pH \sim 9,$$

(Stumm & Morgan, 1970) and is independent of pressure, as long as the pressure is less than 1 000 atmosphere (and the temperature remains well below 100 °C; Barrer, 1982). Similar results can be shown for $SiO_{2quartz}$, but these were not included in the present study because the percentage of $SiO_{2quartz}$ is very small compared to that of $SiO_{2amorphous}$. Also, the equilibrium constant for quartz is smaller than $K_{am}(T)$ by a factor 10.

The temperature dependence of the equilibrium coefficient $K_{am}(T)$ is modelled, in agreement with classical chemical practice, by means of an Arrhenius relation

$$K_{am}(T) = Ae^{-\frac{\Delta H_f}{RT}} \tag{6}$$

where ΔH_f is the enthalpy of formation of $Si(OH)_4$, R is the universal gas constant, T is the temperature (expressed in Kelvin), while A is a constant to be determined.

For a given substance, the enthalpy of formation has a fixed value and relation (6) contains only one unknown parameter, namely A. If only chemical processes are taken into account, this remaining parameter will be a constant. However, rivers are dynamic systems and it is possible that the movement of the water mechanically perturbs the characteristics of the chemical equilibrium by affecting the validity of the infinite dilution hypothesis. Therefore, we assumed *a priori* that parameter A was no longer a constant, but could be an (unknown) function of the discharge Q, i.e.

$$A = A(Q),$$

to be determined. In order to gain some information about this function, important in the Vaal River context, the environmental data available from the Balkfontein site for the period 1986-1987 was taken in account, namely: water temperature, silicon concentration, flow and chlorophyll-*a* concentration (see Table 3). At this stage of the modelling process, only the data corresponding to either a low concentration of algae or to an algal bloom realised by algae not belonging to the diatom group are relevant (indicate data by "*" in Table 3). The collection of relevant data sets is given in Table 4.

TABLE 4: Si saturation concentration: field data

Temperature (°C)	Si^{exp} (10^{-3}mol.l^{-1})	Q ($m^3.s^{-1}$)
14	0.1204	15
23	0.17643	4
20	0.14536	0.7
10	0.112143	20
20	0.14786	13
24	0.168214	32
26	0.17786	136

With the aid of relations (4) and (6) it is possible to eliminate the effect of temperature from the field data to get an idea of the value of A(Q) by means of the relation

$$\frac{Si^{exp}}{e^{\frac{-\Delta H}{RT_{exp}}}} = \frac{K(T_{exp})}{e^{\frac{-\Delta H}{RT_{exp}}}} = A(Q)|_{exp} \qquad (7)$$

The results of this computation are given in Table 5.

TABLE 5: Experimental values of A(Q)

Si^{exp} (10^{-3}mol.l^{-1})	A(Q) (mol.l^{-1})	Q (m^3.s^{-1})	Si^{proj} (10^{-3}mol.l^{-1})
0.1204	1.856	15	0.1234
0.17643	2.029	4	0.1654
0.14536	1.839	0.7	0.1503
0.112143	1.982	20	0.1076
0.14786	1.871	13	0.1503
0.168214	1.875	32	0.1707
0.17786	1.862	136	0.1816

From this reduced set of data, it is clear that the magnitude of the flow does not have a significant influence on the parameter A, at least for the period covering the years 1986-1987 at the Balkfontein site. The value of the coefficient, A, was thus chosen as the average of the values presented in Table 5, i.e.

$$A(Q) = A = 1.902 \ \ mol.l^{-1} \qquad (8)$$

The projected dissolved silicon values, obtained by means of relation (6), are given in the last column of Table 5. Comparing these with the measured values (see column 1, Table 5), it is clear that a good agreement between observed and predicted values is achieved whenever the concentration in algae belonging to the diatom group is not too high. However, should the concentration of diatoms increase, the effect of these algae on the dissolved silicon concentration has to be taken into account.

2.2 MODELLING THE SILICON UPTAKE BY DIATOMS

During the last 2 decades, mechanisms involved in the uptake of silicon by diatoms have been studied (Eppley, 1977). Some general features of primary importance when modelling this aspect of algal metabolism, will be reviewed briefly.

The rate of silicon uptake by a diatom cell depends on the species involved as well as on the orthosilisic acid concentration observed in the immediate vicinity of the cell. Usually, the rate of uptake is a non-linear function of the Si concentration which exhibits a saturation phenomenon beyond a relevant level of dissolved Si, Si_{up}^{cr}, and which is discontinued if the orthosilisic acid concentration in the water falls below a certain level, Si_{up}^{min} (Jorgensen, 1979). This behaviour suggests that a relevant function to model silicon uptake by a given diatom cell "i", may be of the form

$$V_i = \begin{cases} V_{max_i} & \text{if } Si \geq Si_{up_i}^{cr} \\ V_{max_i} L(Si, Si_{up_i}^{cr}, Si_{up_i}^{min}) & \text{if } Si_{up_i}^{cr} > Si \geq Si_{up_i}^{min} \\ 0 & \text{if } Si < Si_{up_i}^{min} \end{cases} \tag{9}$$

where V_{max_i} represents the maximum rate for silicon uptake, which is only reached if the dissolved Si concentration exceeds the critical level, $Si_{up_i}^{cr}$. The transfer function L modulates this maximum rate of uptake when the dissolved Si concentration is below the optimal level. The behaviour of this function is illustrated in Fig. 3, for a realistic choice (Werner, 1977) of parameters V_{max_i} = 30 pgSi.(cell.h)$^{-1}$, $Si_{up_i}^{cr}$= 2 mg.l^{-1}, $Si_{up_i}^{min}$ = 0.5 mg.l^{-1}.

As far as the order of magnitude of the different parameters from function L is concerned, little information is available from the literature. Therefore, only an overall estimate for the range of variation of the rate of uptake

$$V_{max_i} \in [10^{-1} - 10^{-5}] \quad mgSi(\mu gchl - a.h)^{-1}$$

could be reached. On the other hand, the field data seems to indicate that the following orders of magnitude for the remaining parameters, $Si_{up_i}^{cr}$, $Si_{up_i}^{min}$, are acceptable:

$$\begin{array}{lcl} Si^{cr} & \sim & 2 \ mgSi.l^{-1} \\ Si^{min} & \sim & 0.5 \ mgSi.l^{-1}. \end{array} \tag{10}$$

Note that these values are derived from a rather small set of experimental data and that more information on this aspect of the problem is necessary.

Combining the two aspects of the problem, i.e. the uptake of silicon by diatoms and the existence of a dissolved silicon saturation concentration, Si^{sat}, we obtain a differential equation describing the behaviour of dissolved silicon in the water

$$\frac{dSi}{dt} = Prod_{Si}(Si^{sat}(T), Si, k_{Si}^{prod}) - \sum_{i=diatom} V_i x_{1i}, \tag{11}$$

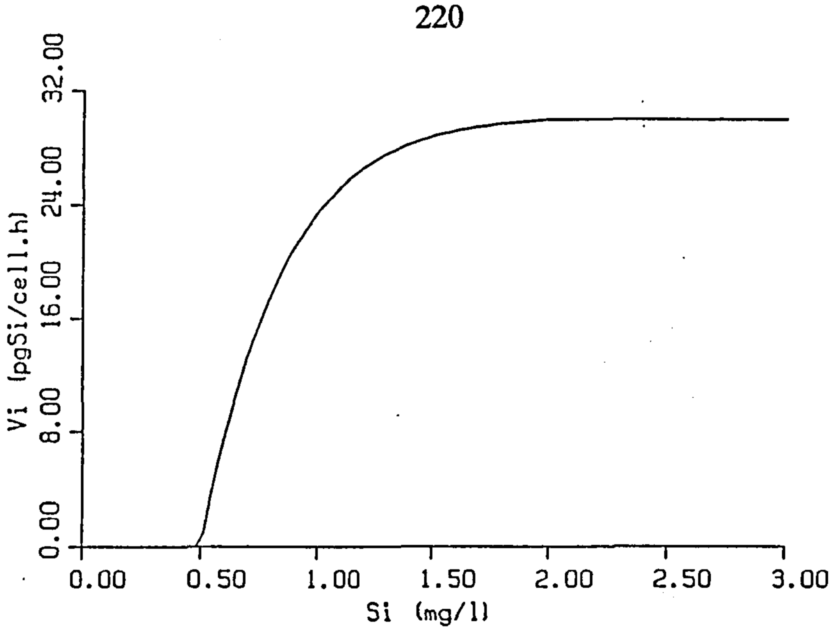

FIGURE 3: Evolution of the uptake function V_i with Si^{sat} = 2 mg l^{-1}, V_{max} = 30 µg Si (cell h)$^{-1}$

where Si^{sat}, the saturation concentration (in the absence of diatoms), is defined by relation (5), while the V_i's are given by (9). Finally, the coefficient, k_{Si}^{prod}, is the inverse of the characteristic time of restitution for saturated Si concentration under the condition that no diatoms are present in the water. From field measurements, it appears that a relevant order of magnitude for this coefficient, in the Vaal River context, is

$$k_{Si}^{prod} \sim 0.1429 \ day^{-1} \ . \tag{12}$$

This means that the time of recovery for the Si concentration is about one week, at least. Note that, once again, this additional equation describing the evolution of dissolve Si concentration in the river is completely non-linear and coupled to other variables like the temperature and the concentration of living diatom.

In this second part of the development of the model, only the effect of diatoms on the dissolved Si concentration has been considered. In order to be complete, we also have to consider the effect of dissolved orthosilisic acid levels in the water on the behaviour (the growth) of different diatom species. Indeed, if the algae have to grow in an environment where the silicon concentration is unfavourable, the metabolism of these algae will be affected and their growth rate will deviate from its optimal value. Since metabolic activities and cell division rates are correlated, it is natural to consider that, for a given diatom I, the growth rate will be affected in a way similar to the silicon uptake rate, when the concentration of the orthosilisic acid is varied. This suggests that the effect of silicon concentration on the growth rate coefficient k_{G_i} can be taken to be of the form

$$k_{G_i}(I,T,Si) = \begin{cases} k_{G_i}(I,T)L(Si, Si_{G_i}^{cr}, Si_{G_i}^{min}) & \text{if } \ Si_{G_i}^{cr} \geq Si > Si_{G_i}^{min} \\ k_{G_i}(I,T) & \text{if } \ Si \geq Si_{G_i}^{cr} \\ 0 & \text{if } \ Si \leq Si_{G_i}^{min}. \end{cases} \tag{13}$$

This function compares well with relation (9). However, the coefficient values are allowed to differ from those relevant to the Si uptake, i.e. it is possible that $Si^{cr}_{up_i}$, $Si^{min}_{up_i}$ may differ, usually slightly, from $Si^{cr}_{G_i}$, $Si^{min}_{G_i}$. The distinction between coefficients is made necessary in order to account for the fact that most of the diatom species are capable of absorbing Si in excess and using then the reserve to prevent or delay the effect of silicon depletion, should it occur. Furthermore, the coefficient $k_{G_i}{}'$ (I,T) corresponds with the growth rate coefficient depending only on light and temperature, as defined in earlier studies (Cloot & Schoombie, 1994).

3. THE MATHEMATICAL MODEL: UPDATED VERSION

When silicon uptake by diatoms is taken into account, as was described in the previous section, the schematic representation of the basic version of the mathematical model presented in Fig. 1 is no longer valid and has to be replaced by the scheme illustrated in Fig. 4.

The updated version presented in the schematic representation have to be implemented in a basic set of equations, (1), which now takes the form of a system of 2N + 1 non-linear differential equations

$$\begin{aligned} \dot{x}_{1_i} &= [-k_{D_i} + k_{S_i}) + k_{G_i}(T, I, Si, \underline{K}_j; j = 1, n)]x_{1_i} \\ \dot{x}_{2_i} &= k_{D_i}x_{1_i} - k_{S_i}x_{2_i} \\ \dot{Si} &= Prod_{Si}(Si^{sat}(T), Si, k^{prod}_{Si}) - \sum_{i=diatom} V_i x_{1i} \end{aligned} \qquad (14)$$

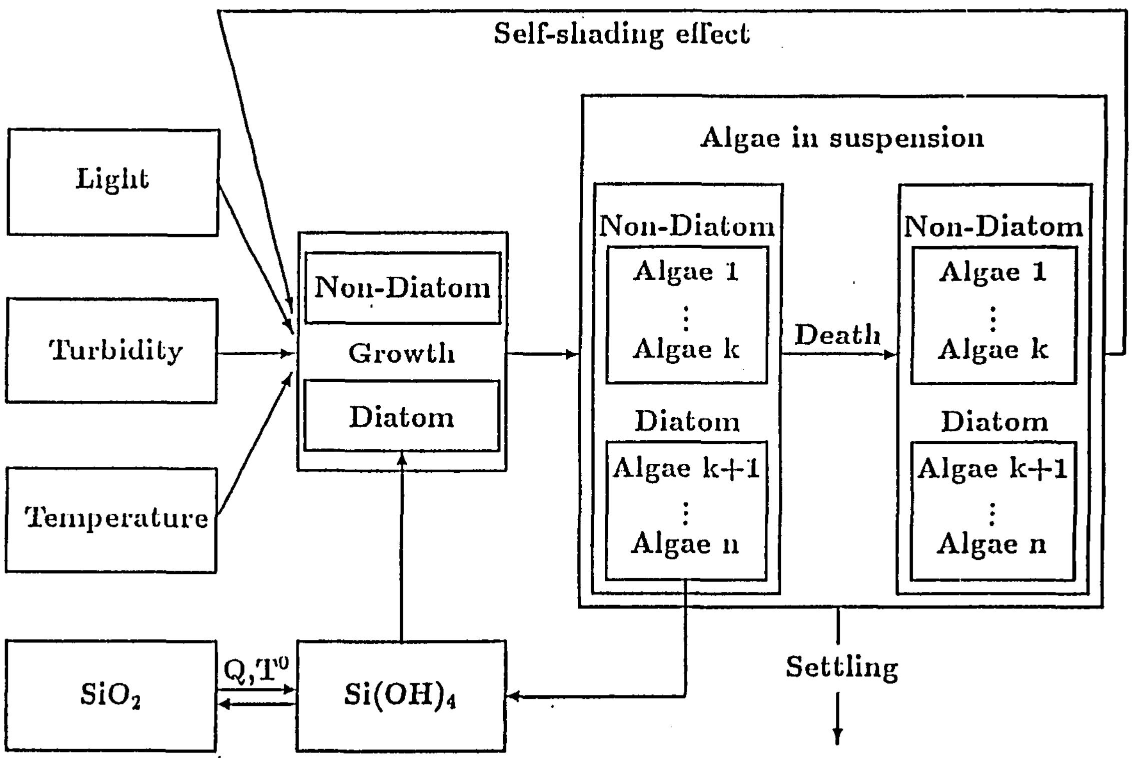

FIGURE 4: Vaal River n-algal growth model including the dissolved silicon effect: a schematic representation

3.1 NUMERICAL SIMULATION

In order to illustrate the influence of silicon depletion in the river on diatom growth, we did the same computation experiments which were performed with the aid of the basic light-temperature dependent model described in an earlier section, but included the possible effect of silicon concentration on the rate of algal growth. As mentioned in the introduction, it is well known that the winter algal blooms, in the Vaal River context, are usually initiated by diatoms, followed by green algae, In the light of this experimental evidence, we thus assumed that the first algal considered in the simulation of the winter period of 1986 was a diatom, i.e. sensitive to silicon variations in the water, while the second algal bloom was due to a non-diatom alga. Keeping the light and temperature parameters of the fist algal at their original values, we defined an additional subset of parameters relation to silicon uptake (see Table 6) and the program was reran to see if silicon effects could account for discrepancies observed within the frame of the original light-temperature model.

TABLE 6: Optimal parameter set for the light-temperature and light-temperatue-silicon dependent models

	Light-temperature model		Light-temperature-silicon model	
Parameter	Alga 1	Alga 2	diatom	non-diatom
k_G	1.28	1.46	1.28	1.40
k_D	0.15	0.15	0.15	0.15
I_{opt}	0.12	0.08	0.12	0.08
T_{opt}	10	13	10	13
T_{min}	0	5	0	5
Si^{cr}_{up}			2	
Si^{min}_{up}			0.5	
V_{max}			0.01	
Si^{cr}_{G}			1	
Si^{min}_{G}			0.5	

The result of this numerical experiment is shown in Fig. 5. Comparing these results with the results obtained from the basic model (see Fig. 2), it is clear that the taking into account of the effects of silicon significantly improves the fitting of the calculated values to the measured values.

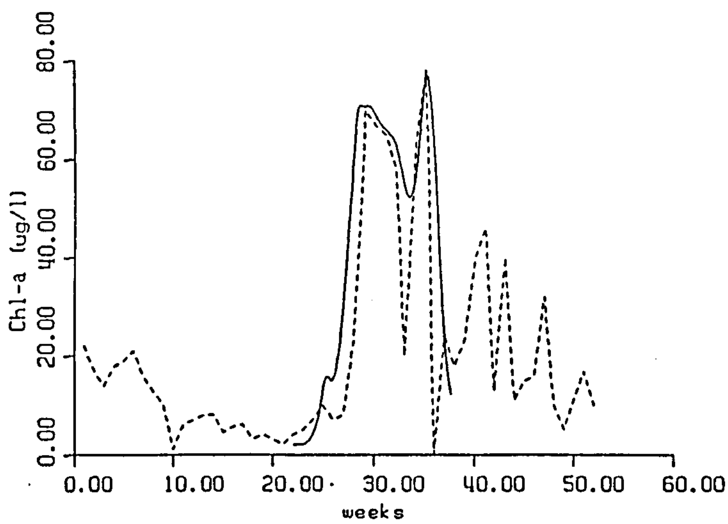

FIGURE 5: Same as Fig. 2, except that silicon uptake by the diatoms is taken into account.

Indeed, if the evolution of the values of the growth coefficient of alga 1 (i.e. the diatom) for both cases (see Fig. 6), together with the projected behaviour of the dissolved silicon concentration in the water is computed (see Fig. 7), it can be observed that as long as the diatom concentration remains at a low level, depletion of silicon by algal uptake is more or less counterbalanced by silicon production, while the silicon concentration remains above the saturation concentration. Also, no effect on the growth coefficient is observed. However, with increasing diatom concentrations, the balance can not be maintained and silicon depletion occurs to such an extent that it affects the growth coefficient. In this case the dissolved silicon concentration becomes a limiting factor for growth.

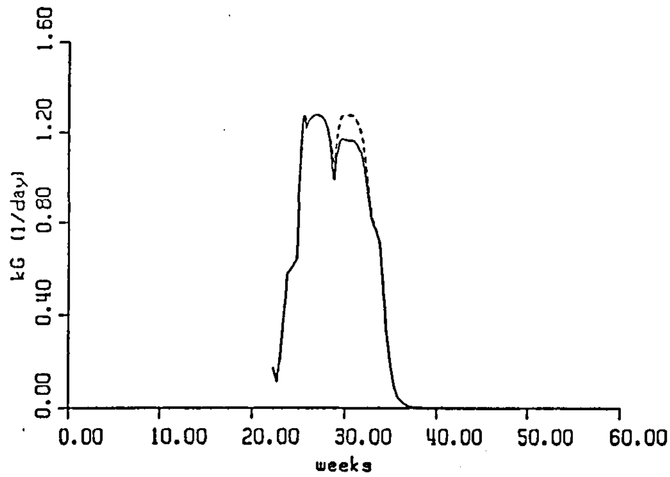

FIGURE 6: Time-evolution of the growth coeffient of the diatoms assuming that: only light and temperature affect this coefficient (dashed line); light, temperature and Si determine the amplitude of k_G (solid line). Note that most of the time solid (Si) and dashed lines (no Si) overlap and then only a solid line is visible.

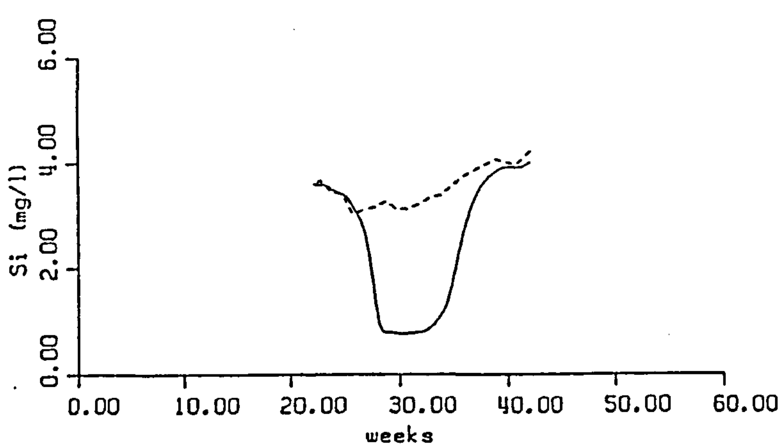

FIGURE 7: Evolution of dissolved Si concentration during 1986 winter algal blooms as predicted by the model: the dashed line represents the saturation concentration reachable in absence of diatom growth, while the solid line represents the dissolved Si curve as computed by the model.

4. CONCLUSIONS

By taking into account the possible effects of variations in dissolved silicon concentration in the water, it was possible to improve the quality of the simulation of algal growth and algal blooms in the Vaal River. However, some discrepancies still remain between simulated and measured values of the total chlorophyll-*a* concentrations, that are not explained by the silicon effect alone. Thus it is obvious that the influence of other factors will have to be implemented in the mathematical mode. The next steps in the development of the model will be to include the effect of salinity, pH, phosphate and nitrate concentration, as well as that of dissolve oxygen and carbon dioxide. The more inclusive approach should give a better quantitative agreement between simulated and observed values, and would also give a better indication of the overall interaction between the algae and their environment.

5. SUMMARY

5.1 The **prediction of the development of algal blooms** in a river is of great importance in water resource management.

5.2 During a three-year period, starting January 1985, **two algal blooms were observed** during the winter to spring periods. The first of these blooms was mainly caused by diatoms, while the second was mainly caused by green algae.

5.3 A **good agreement** existed between **projected dissolved silicon** values **and measured values.** The agreement was **achieved whenever the concentration of diatoms was not high.** Should the concentration of diatoms increase, the effect of these algae on the dissolved silicon concentration has to be taken into account.

5.4 An **equation describing the changes in concentration of dissolved Si concentration** in the river was **non-linear** and **coupled to other environmental variables** like the temperature and the concentration of living diatom cells.

5.5 In order to illustrate the influence of silicon depletion in the river on diatom growth, the possible **effect of silicon concentration on the rate of algal growth** was incorporated in the model. It was assumed that the first algal peak of the season was due to a diatom (which was sensitive to a concentration variation in silicon in the water), while the second algal bloom was formed by species other than diatoms.

5.6 When the **effects of silicon** was taken into account, the fitting of calculated values to the measured values was improved significantly.

5.7 As long as the diatom concentration remains at a **low level, depletion of silicon by algal uptake is more or less counterbalanced by the replacement of silicon through solution,** while the silicon concentration remains above the saturation concentration. With increasing diatom concentrations, the balance cannot be maintained and silicon depletion occurs to such an extent that it negatively affects the growth of diatoms. In this case the dissolved silicon concentration may become a limiting factor for the growth of diatoms.

5.8 **The influence of other environmental factors**, such as the effects of salinity, pH, phosphate and nitrate concentrations, dissolved oxygen and carbon dioxide, **will have to be implemented** in a more advanced mathematical model.

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CHAPTER 8: SUMMARY AND CONCLUSIONS

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The Vaal River is the most important and the most regulated river in South Africa. The Vaal River is also a eutrophic system on account of high chlorophyll-*a* and inorganic nitrogen and phosphorus concentrations as well as high primary productivity rates. Massive developments of phytoplankton are experienced in certain sections, resulting in aesthetic problems, health hazards, interferences with treatment processes and problems in water distribution systems. Attention had, therefore, been given to the causes and consequences of the wax and wane of phytoplankton assemblages.

Four sampling localities were selected in the Middle Vaal River, namely at the Rand Water Barrage (Vanderbijlpark), Parys, Western Transvaal Regional Water Company (Stilfontein) and Goldfield Water (Balkfontein, Bothaville). Algal samples were prepared for investigations with the inverted light and scanning electron microscopes. Physical, chemical and biological data were obtained from the Department of Water Affairs, Rand Water, and the Western Transvaal Regional Water Company. Chemical analyses, primary productivity, as well as inorganic nitrogen and phosphorus availability studies, were done on water samples taken at Balkfontein.

The study showed that a variety of phytoplankton species (at least 124 species and varieties), occurred in the Vaal River during the study period. The green algae showed greatest species diversity. The greatest diversity of Cyanophyceae species were present at the Barrage, while the greatest diversity of Euglenophyceae species were present at the Balkfontein sampling locality. Almost two thirds of the identified species occurred at all the sampling localities. There were, however, some species which were unique to the Barrage and Parys (upstream) sampling localities, while others were unique to the Stilfontein and Balkfontein (downstream) localities.

Different species succession patterns were demonstrated for the different sampling positions. At the Barrage, Parys and Balkfontein, different species succeeded one another within relatively small time spans. At Stilfontein a smaller number of species succeeded one another, possibly indicating more stable environmental conditions at this locality in the river.

Water temperature is an important environmental variable because specific organisms have definite ranges of temperature at which maximum growth and reproduction occur. Diatoms tended to dominate in the Vaal River from January to August of each year, while the green algae were dominant from September to December. Dominance of diatoms during summer periods (January to April and December) can be ascribed to blooms of *Melosira granulata*. Concentrations of unicellular centric diatoms were usually low during the summer periods, but they often dominated during the cold-water winter periods. It therefore seemed as if *M. granulata* prefers warmer water temperatures than the unicellular centric diatoms. Blue-green algae frequently occurred during the mid and late summer months of each year, especially at the Barrage, Stilfontein and Balkfontein sampling localities.

Water temperature also affects solubility of silica and oxygen. Higher silica concentrations were usually present during summer periods when the water temperatures were higher, while lower Si concentrations were present during the winter. The solubility of oxygen is higher in colder water so that higher oxygen concentrations were observed during winter periods.

At all four sampling localities the phytoplankton community was dominated mainly by diatoms and green algae (which succeeded each other), as well as by blue-green algae during warmer periods. Euglenophyceae biomass increased from the Barrage downstream to Balkfontein. Representatives of the Cryptophyceae, Chrysophyceae and Dinophyceae were relatively scarce throughout the study period.

The species composition of the water can give an indication of the quality of the water. The scarceness of certain groups (e.g. Chrysophyceae and Cryptophyceae), and the abundance of others (e.g. Cyanophyceae), show that the Vaal River is a polluted and eutrophic system.

Indications of pollution and eutrophication were also reflected in high chlorophyll-*a* concentrations in the Vaal River in comparison with other river systems. Chlorophyll-*a* concentration gives an indication of phytoplankton biomass. Periods of maximum chlorophyll-*a* concentration in the Vaal River occurred from January to March (summer) and again from July to November (winter-spring) of each year. The highest average chlorophyll-*a* concentration was present at the Parys sampling locality, probably the result of high available nitrogen and phosphorus concentrations present in the river at, and upstream from, this sampling locality. The annual chlorophyll-*a* concentration was positively correlated with total phosphorus (TP), dissolved inorganic carbon (DIN) and silica (SiO_2-Si) concentration. Thus, increased nutrient concentration in the Vaal River apparently increased the capacity of the water to support high production rates and to maintain large standing crops of phytoplankton.

One of the most important variables influencing the organisms living in the Vaal River, is discharge. Higher discharge resulted in higher total suspended solids (TSS) concentration, higher nutrient concentration (i.e. N, P and Si), lower total dissolved salts (TDS), higher turbidity and thus in lower euphotic zone (Z_{eu}) and underwater light climate (ULC). Hydrology, therefore, plays an important role, not only in the chemistry, but also in the biology, of Vaal River water.

Under conditions of high nutrient supply, algal blooms were sometimes possibly prevented by low underwater light conditions as a result of increasing discharge. Very high levels of discharge can be responsible for a complete wash-out of the phytoplankton. Discharge did not only influence turbidity and underwater light conditions, but also showed a positive correspondence to nutrient (phosphorus and nitrogen) concentrations and a negative correspondence to salinity.

The turbidity of Vaal River water at Balkfontein was generally lower during the period 1984 to 1991 than during previous years. Since 1991 an increase in turbidity was observed. Reduced turbidity might be due to the clearing effect of salinity or possibly to the presence of water hyacinths. Water hyacinths could have served as sediment and plant nutrient traps.

Decreased turbidity resulted in an improved under water light climate (ULC) which, in turn, was associated with reduced daily rates of areal photosynthesis during certain periods. However, increases in photosynthetic rates were expected as a result of the improved ULC. The unexpected reduced rates of photosynthesis could be due to reduced chlorophyll-*a* concentrations which most probably were the result of reduced nutrient availability. Reduced nutrient availability, in turn, was possibly at least partially caused by water hyacinth growth.

The total dissolved salts (TDS) concentration in the Vaal River is high, with an average annual maximum of about 650 mg l^{-1} and a minimum concentration of about 400 mg l^{-1}. The mean annual TDS concentration for the study period (1984 - 1993) was about 520 mg l^{-1} which is approximately four times higher than the global mean salinity of river water. An average increase rate of 25 mg l^{-1} per annum was calculated for the study period at both Stilfontein and Balkfontein. Based on the results of the present study, the Vaal River can be classified as a mixohaline system.

The highest salinity during the period 1985 to 1991 for both Balkfontein (average about 520 mg l^{-1}) and Stilfontein (average about 500 mg l^{-1}), usually occurred during the winter to spring period. During the years of flooding (March 1988 and February 1989), the highest salinity values were recorded during spring. The salinity was usually low during the summer months (November to March) in accordance with most other African rivers. After the dilution of the salts by the floods, the salinity increased sharply.

Salinity at the Stilfontein and Balkfontein localities displayed seasonal changes that were strongly influenced by discharge in the river, especially discharge that originated from rainfalls. The order of ionic prominence in the Vaal River at Balkfontein was $SO_4^{2-} \gg Ca^{2+} \geq Cl^- \geq Na^+ \geq Mg^{2+}$ and at Stilfontein it was $SO_4^{2-} \gg Cl \geq Ca^{2+} \geq Na^+ \geq Mg^{2+}$. For both Balkfontein and Stilfontein the major ionic contributor to the total dissolved salts of the water was the anion SO_4^{2-}.

Besides eutrophication, salinity is the major problem in the Vaal River. The best quality of water is found in the upstream section of the Vaal River, showing a rapid downstream deterioration as the concentration of dissolved salts increases from the Barrage to Balkfontein. Inputs of mining, industrial and human effluents are probably the major contributors to increasing salinity. The sulphate concentration in the Vaal River is amongst the highest concentrations reported for rivers world-wide. Salinity was not the primary variable influencing algal growth, but dinoflagellate representatives (responsible for red-tides in the ocean) more frequently occurred in high salinity water, while blue-green algae occurred in water with relatively low salinities.

The main importance of the high TDS in the Vaal River is evidently its influence on turbidity and the possible clarification of the water-column, which will result in a deeper Z_{eu} and thus more favourable underwater light conditions for photosynthesis and possible biomass build-up. However, on an annual basis, it was shown that clearer water was associated with lower nutrient (N, P and Si) concentrations, lower primary productivity and lower chlorophyll-a concentration. Reduced nutrient availability, therefore, apparently limited production and abundance of phytoplankton even when sufficient light was available.

Relatively low turbidities since 1990 can be ascribed to the generally high TDS levels and relatively low discharge in the Vaal River and possibly to water hyacinth *(Eichhornia crassipes)* infestation. Water hyacinths evidently served as a biofilter, i.e. a sediment and nutrient trap.

Growth and carbon assimilation experiments indicated that different algae showed different sensitivities to TDS. Of the three algal species investigated, *Cyclotella meneghiniana* (a diatom) was the most sensitive and *Monoraphidium circinale* (a green alga) the least sensitive to increased dissolved salts. If the total dissolved salts concentration of Vaal River water increases with time, algal species that could persist over a wide range of salinities can be expected to be present most of the time. Of the species investigated, *Monoraphidium circinale* can be expected to become dominant under conditions of increased salinity. *Cyclotella meneghiniana* and *Microcystis aeruginosa* on the other hand, can be expected to be excluded from the water under conditions of increased salinity of 250 mg l^{-1} and above.

The Cyanophyceae (blue-green algae), to which *M. aeruginosa* belongs, is usually dominant during summer months when the TDS concentration is low. This is in agreement with the results of growth experiments where the growth of *M. aeruginosa* was inhibited at increased TDS concentrations. The Chlorophyceae (green algae), to which *Monoraphidium circinale* belongs, is dominant for most of the year in the Vaal River and showed no preference for a specific season. The fact that an increase in TDS concentration showed no effect on the growth of *Monoraphidium circinale*, is in agreement with the presence of the Chlorophyceae group for most of the year. It must be remembered, however, that other Cyanophyceae and Chlorophyceae species may differ in their response to different salinities from the ones used in this study. The recent increased abundance of *Oscillatoria simplicissima* (another bluegreen alga) in the Vaal River can possibly be attributed to temporary decreasing levels of salinity.

Reduced annual DIN:DIP ratios suggested that the Vaal River at Balkfontein switched from a generally phosphorus limited system (1986 to 1990) to a potentially nitrogen limited system (1991 and 1993). The decrease in DIN:DIP ratio was caused by relatively low DIN concentrations and relatively high DIP concentrations.

Switching inorganic nitrogen and phosphorus limitation influence algal biomass as well as the composition of the algal assemblages. The switching between Chlorophyceae (green algae) and Bacillariophyceae (diatoms) dominance is apparently partly determined by N:P ratios. Algal blooms were frequently preceded by high nutrient (DIN, DIP and Si) concentrations, but when the bloom reached its peak, low nutrient concentrations were recorded. Blue-green algae were most probably favoured by low DIN:DIP and TN:TP ratios. A shift from blue-green algae to other algal groups occurred if the TN:TP ratio increased. Increased phosphorus concentrations in the Vaal River (with a resultant decrease in DIN:DIP and thus TN:TP to less than five) will probably cause a shift in the algal assemblages from diatom and green algal dominance to blue-green algal dominance. The Si:DIP ratio also seemed to be important in influencing the occurrence of diatoms, because diatoms usually occurred under conditions of high Si:DIP ratios.

Nitrogen and phosphorus are two nutrient elements necessary for algal growth in aquatic and terrestrial systems. Sources of nitrogen and phosphorus in the aquatic environment include agricultural, domestic and urban run-off. These sources are usually the cause of blooms due to the large quantities of inorganic N and P enriching the water. Two enzymes which play a vitally important role in making these essential elements, namely nitrogen and phosphorus, available for the growth of algae are Nitrate Reductase (NR) and Phosphatase (PASE) respectively. An investigation into the activities of enzymes responsible for the uptake of nitrogen and phosphorus gave information about the dynamics of inorganic nitrogen and phosphorus in the Vaal River and explained in more detail environmental control mechanisms exerted on algal biomass and composition.

High PO_4-P concentrations corresponded to increased Phosphatase and Nitrate Reductase activities. Activities of the two enzymes studied, appeared thus to be stimulated by high P levels. No definite correspondence was shown between NO_3-N and NR activities. In a similar experiment performed in Israel, NR activity did not appear to reflect changes in either NO_3 or NH_4 concentrations in Lake Kinneret. In the present study, no apparent relationship between NO_3-N and PASE activity occurred. In addition, both groups of enzymes, namely PASE and NR, appeared to be indirectly proportional to chlorophyll-*a*.

NR activity appeared to be corresponding positively with PASE activity. Both enzymes seem, therefore, to be stimulated by similar environmental conditions.

Both enzymes seemed to be activated by high PO_4-P concentrations in the Vaal River. These results could suggest that all metabolic activities in algal cells are activated by high PO_4-P concentrations or increased PO_4-P availability. The increased metabolic rates in algal cells could then result in faster division rates and consequently in the development of algal blooms.

All the above results support a conceptual model that assumes that phytoplankton growth in the Vaal River is controlled by fluxes of solar energy, but fluxes of plant nutrients apparently affects the flow of energy into the algal cells.

Results of the present study show that eutrophication (nutrient enrichment) is particularly a problem at the two upstream sampling localities (Barrage and Parys). Higher DIP and TP concentrations were measured at the Barrage and Parys sampling localities, decreasing in a downstream direction. High phosphorus concentrations at the two upstream sampling localities could be a result of urbanisation and industrialisation in the PWV complex, while lower concentrations in the downstream section of the river can be ascribed to removal of

DIP by thick mats of water hyacinths. High DIN and TN concentrations were also recorded at the Barrage and especially at the Parys sampling localities decreasing downstream to the Stilfontein and Balkfontein sampling localities. High nitrogen concentrations downstream from the Vaal Dam indicate higher organic and inorganic nitrogen loading possibly because of agricultural, mining and industrial effluents released into the Vaal River. DIP, TP and TN concentration ranges as well as high TN:TP ratios in the Vaal River fall within the ranges for eutrophic systems.

Highest silica concentrations were reported at the Barrage, whereafter a decrease occurred downstream to Parys, Stilfontein and Balkfontein. A possible explanation for the high Si concentration at the Barrage sampling locality is that the concentration of diatoms is lower at the Barrage than at the Parys, Stilfontein and Balkfontein sampling localities, thereby not removing significant amounts of silicon from the water. It seems as if Si concentration in the Vaal River is primarily determined by temperature and diatom uptake metabolism.

Statistical analysis of the water chemistry, temperature and chlorophyll concentration data showed that major variations in water properties can be ascribed to seasonally-related variations in temperature, suspended solids (turbidity), coupled with directionally opposite variations in dissolved salts (conductivity). Algal growth was greatest in winter when the water is clear and cold and lowest in summer when it is turbid and warm. Seasonal variations in the two forms of inorganic nitrogen (nitrate and ammonium) seemed to be different, with the one being associated with summer and the other with winter conditions.

Correlation matrices for Vaal River variables at the Barrage below the sluice gates (representing 7 m deep bottom water in the Barrage) and Lindeque's Drift (9 km downstream of the Barrage), showed similar associations which were, in turn, different from those of the surface water in the Barrage immediately above the sluice gates. The observed differences between the surface and bottom waters in the Barrage indicated that vertical, depth-related processes and conditions over a depth of approximately 7 m, have a more significant effect on variables than horizontal, distance-related processes and conditions along a distance of approximately 9 km.

The application of the principal component analysis (PCA) method to investigate associations between physical, chemical and biological variables resulted, amongst others, in the elucidation of seasonal aspects of algal growth, showing higher algal growth during colder months. Should changes in the seasonality occur, i.e. should summer blooms replace winter blooms or should different years show different associations, PCA analyses will be able to identify and quantify the changes and differences in terms of their statistical significance. However, the statistical comparison between phytoplankton and environmental changes is only the first step in linking species dynamics to changes in a complex environment.

A light-temperature dependent model to simulate algal blooms in the Vaal River was developed, but more information on the ecological behaviour of the river and as well as growth requirements of specific algae is essential. The model is based on two assumptions, namely that the water is eutrophic and that the growth and death of algae are dependent mainly on the available light and temperature of the water. Weekly averaged temperature and the total suspended solids concentration were considered as inputs. The calculated chlorophyll-*a* concentration values agreed fairly well with measured values in qualitative as well as quantitative terms during three years (1985 to 1987) of investigation.

By taking into account the possible effects of dissolved silicon concentration, it was possible to improve the quality of simulation of algal growth and algal blooms in the Vaal River. However, some discrepancies still remain between simulated and measured values of chlorophyll concentration that are not explained by the silicon effect alone. Other factors will therefore have to be incorporated into the mathematical model.

Two major fields of application for such a mathematical model are possible. Firstly the model can be used to support investigations by providing a consistent splitting of the global data between smaller algal groups assumed to be responsible for blooms, by assisting in the identification of important environmental variables concerning specific algae involved in a bloom, by assisting the researcher to describe the mechanisms involved in the development of a bloom by providing numerical information about experimental data that are not available, and finally by providing a way to test new hypotheses suggested by other investigations. Secondly, mathematical models of this nature can be used in water resource management in which case the model can be used as a planning tool.

ACKNOWLEDGMENTS

The authors are grateful to the Water Research Commission for the financial support without which the research on the Vaal River would be impossible. The Department of Water Affairs and Forestry, the Rand Water Board and the Western Transvaal Regional Water Company provided data on environmental variables.

The assistance of a number of persons in the execution of the research programme is deeply appreciated. These are Dr FC Viljoen, Mr MC Steynberg and Mr J Meyer of the Rand Water Board, Vereeniging, Mr W du Preez of the Parys Municipality, Parys, Mrs M Krüger, Mr J Pietersen and Mr K Morgan of the Western Transvaal Regional Water Company, Stilfontein, as well as Mrs N Basson and Mr H du Preez of Goldfield Water, Balkfontein, Bothaville. In addition, Dr S Mitchell, former Chairman of the Steering Committee of this project, as well the present Chairman, Mr G Offringa, of the Water Research Commission, have always supported our work and were accommodating towards our needs. In addition, special interest in our work have been shown by Mr G Quibell, Department of Water Affairs and Forestry, Pretoria, and Mr N Rossouw, Watertek, Pretoria.

GLOSSARY OF SYMBOLS

Notation - Symbols, definitions and units used in the text.

B	:	Biomass (chlorophyll-*a*) of phytoplankton (μg Chl-*a* l^{-1} = mg Chl-*a* m^{-3}).
I	:	Instantaneous scalar light intensity at the water surface, measured as photosynthetic photon flux fluence rate, mean over spectrum of PAR (μE m^{-2} s^{-1} ≡ μmol m^{-2} s^{-1}).
I_k	:	Irradiance at onset of light saturation (μmol m^{-2} s^{-1}).
k	:	Vertical light extinction (attenuation) coefficient, mean value over spectrum of PAR (m^{-1}).
NTU	:	Nepholemetric turbidity units.
Â^B	:	Initial slope of the chlorophyll-specific P vs. I curve, also termed photosynthetic efficiency (obtained by linear regression analysis; mg C mg Chl*a*$^{-1}$ E^{-1} m^2).
đI_d	:	Integral areal daily irradiance (μE m^{-2} d^{-1}).
đP_d	:	Integral daily rate of areal photosynthesis (mg C m^{-2} d^{-1}).
đP_h	:	Average (hourly) integral rate of areal photosynthesis during incubation period (mg C m^{-2} h^{-1}).
PAR	:	Photosynthetically available radiation between 400 and 700 nm (μmol m^{-2} s^{-1}).
$P^B{}_m$	:	Photosynthetic capacity i.e. maximum light-saturated, chlorophyll-specific photosynthetic rate, also termed assimilation number (mg C mg Chl*a*$^{-1}$ h^{-1}).
P_m	:	Maximum (light-saturated) volumetric rate of photosynthesis in depth profile (mg C m^{-3} h^{-1}).
PSE	:	Photosynthetic efficiency (PSE), or total water-column light utilisation efficiency
TN:TP	:	Total nitrogen to total phosphorus concentration ratio (weight per weight)
TP	:	Total phosphorus (organic and inorganic) concentration (μg l^{-1} or mg l^{-1}).
TSS	:	Total suspended solids (mg l^{-1}).
ULC	:	Underwater light climate i.e. light availability, defined by đI_d / k (E m^{-1} d^{-1}).

Z_{eu}	:	Euphotic depth, defined by light level where PAR is 1 % of value immediately below water surface (m).
Z_m	:	Depth of mixed layer (m).
Z_{sd}	:	Secchi disk depth (m).

GLOSSARY OF TEXT

Abiotic: Non-living material present in water.

Adsorb: Physical adherence to the surface of a molecule or particle.

Algae (singular, alga): Heterogeneous group of eukaryotic unicellular, colonial and multicellular organisms of simple structure, usually photosynthetic and traditionally included in the plant kingdom. They are aquatic or live in damp habitats on land and include unicellular organisms such as *Chlamydomonas* and diatoms, colonial forms such as *Volvox*, multicellular green, red and brown seaweeds and freshwater multicellular algae such as *Spirogyra*. Many are microscopic in size. No roots, stems and leaves are present. Reproductive organs are essentially one-celled, and the gametes are generally flagellated.

Algae: A heterogeneous group of eucaryotic unicellular, colonial and multicellular organisms of simple structure, usually photosynthetic, and traditionally included in the plant kingdom.

Algal Blooms: Dense growth of algae often linked to seasons.

Algologist: Person who investigates algae.

Amorphous: Not in a crystalline form.

Anaerobic: A condition involving the absence of free (elementary) oxygen in a medium such as water or sewage.

Anion: Negatively charged ion (e.g. Cl^-).

Aquatic: Living in or near water.

Assemblage: A group of algal species co-occurring at a specific time in a specific environment.

Assimilate: In autotrophic organisms, the uptake of nutrient elements and simple inorganic compounds such as CO_2, N_2, H_2O from the environment and their incorporation into complex organic compounds; in heterotrophs, the conversion of digested food material into complex biomolecules.

Autotrophic: Obtaining food as an autotroph

Bacillariophyceae (diatoms): A class of unicellular or colonial (rarely filamentous) algae, having a frustule which is highly ornamented, containing silica, and devided in two halves, called valves. Centric and pennate diatoms can be distinguished. A pennate diatom is elongate rather than circular in valve view. In the case of pennate diatoms the wall ornamentation is arranged along the sides of the longitudinal axis rather than about a central point (as in centric diatoms). The plastids contain the brown pigment isofucoxanthin as well as chlorophyll. The food reserves are in the form of oils and volutin, but never starch. Vegetative cells are non-flagellate, isogametes, and asexual reproduction takes place by cell division. Some members form auxospores or endospores, and others produce flagellated zooids which may be gametic in nature.

Biomass: Total mass, volume, pigment or energy equivalent of organisms in a given area or volume of water; the total mass of biological material of a group of individuals or organisms.

Bloom: Dense growth of planktonic algae giving a distinct color to the water body.

Buffering: The resistance to change in pH as a result of the presence in water of a weak acid and its salts; The bicarbonate and carbonate ions dissociate to establish an equilibrium:

$$HCO_3^- + H_2O -- H_2CO_3 + OH^-$$

$$CO_3^{2-} + H_2O -- HCO_3^- + OH^-$$

$$H_2CO_3 -- H_2O + CO_2$$

Photosynthesis and respiration are two major factors that influence the amounts of CO_2 in water. However, the equilibria of the reactions given above result in the buffering action of alkaline waters, which contain appreciable amounts of bicarbonate. Water tends to resist change in pH as long as these equilibria are operati0onal. An addition of hydrogen ions neutralises hydroxyl ions formed by the dissociation of HCO_3^- and CO_3^{2-}, but more hydroxyl ions are formed immediately by reaction of the carbonate with water. Consequently, the pH remains essentially unaltered, unless the supply of carbonate or bicarbonate ions is exhausted. Similarly, when hydroxyl ions are added they react with the bicarbonate ion.

Buoyant: Flourishing or able to float.

Carcinogenic: Causing cancer in humans or animals.

Catalyst: A substance which accelerates a chemical reaction, without itself being changed in the overall process. most biological catalysts are proteins.

Cation: Positively charged ion (e.g. Ca^{2+}).

Cell: The organised ultimate unit of structure and growth of a plant or animal. It is composed of a protoplast which, in plants, is generally surrounded by a cell wall.

Cellulose: A linear polysaccharide made up of glucose residues joined by ß, 1,4, 1,6 linkages, found in some invertebrates, algae, higher plants, fungi and bacteria, but lacking in most animals. The main constituent of plant cell-walls.

Chlorophyceae (green-algae): Algae unicellular, colonial, or filamentous; floating, swimming or attached and stationary; cells containing plastids (chloroplasts) in which chlorophyll is predominant, and in which there is usually a shiny, starch storing body, the pyrenoid; pigments are chlorophyll-*a*, chlorophyll-*b*, carotenes, xanthophylls, red carotenoids (haematochrome) sometimes present; food reserve is starch; nucleus definite (although often small and inconspicuous); cell wall, when present, composed of cellulose and pectose; swimming cells or motile reproductive elements furnished with 2 (usually), 4 or rarely as many as 8 flagella of equal length and attached in the anterior end; sexual reproduction by iso-, aniso-, and heterogametes.

Chlorophyll: Principal photosynthetic pigment of green plants and algae. It is located in the thylakoid membranes of chloroplasts where it absorbs light energy, absorbing mainly in red and violet-blue regions of the spectrum, chemically distinct forms having different absorption maximum.

Chrysophyceae (golden or golden-brown algae): Motile, unicellular condition most common; contain large amounts of carotenoid pigments which are predominant over chlorophyll-*a* and *c*; food reserve is composed of ß-linked glucan, chrysolaminarin and oil. Cell coverings include scales, loricas and close-fitting cell walls. The scales of members of the Mallomonadaceae are composed of silica; walls and loricas can be of a cellulosic nature with or without an impregnation of calcium carbonate and/or iron compounds. One of the most distinctive features of the Chrysophyceae is the internal formation of a cyst, or statospore, with a siliceous wall that constitutes a resting stage. Cell division occur in unicellular forms or fragmentation in multicellular forms; asexual reproduction can also be accomplished by production of zoospores or autospores. Sexual reproduction in a limited number of genera, most often isogamous.

Coccoid: Spherical or globose cell type and growth form or morphological type.

Colony: A group of individuals or cells of one species living together; in some instances it consists of a few individuals that can be attached to one another in a definite or regular pattern; cells developed together from a single original parent plant or reproductive cell. Each cell is theoretically capable of metabolic activities independent of the others.

Community: A well-defined assemblage of plants and/or animals within a specific area or body of water clearly distinguishable from other such assemblages.

Cryptophyceae (cryptophytes): Cells solitary (rarely colonial), mostly swimming, protozoanlike organisms with two, laterally inserted or subapical flagella unequal in length; chloroplasts few and large, brown, blue, or reddish, with pyrenoids commonly present; pigments are chlorophylls, carotenes, xanthophylls, phycocyanin and phycoerythrin (in some); food reserve in the form of solid starch or starchlike substances; cell covering firm periplast; a gullet commonly present in the anterior end; reproduction by longitudinal cell division; sexual reproduction unknown.

Cyanophyceae (blue-green algae): Plants unicellular, colonial, or in simple or branched (sometimes falsely branched) filaments; chloroplasts lacking, the pigments seemingly distributed throughout the entire protoplast (but contained in flat bodies known as thylakoids), often more dense in the peripheral region of the cell; pigments are chlorophyll-*a*, carotenes, xanthophylls (not necessarily present in all forms), phycoerythrin, phycocyanin; cell wall thin and four-layered with a gelatinous outer sheath; cell often with pseudovacuoles which refract light and obscure the true color of the cells which may be green, blue-green, gray-green, violet, tan, brown or purple; nucleus lacking but nuclear material occurring as a cluster of chromatic granules or fibrils in the midregion of the cell; motile cells and sexual reproduction absent; reproductive cell division by fission or by spores (endospores; akinetes); food storage questionably glycogen, possibly floridean starch; iodine test for starch negative.

Decomposition: Process of decaying and breakdown of organic material, including the dead remains of plants and animals by means of organisms (decomposers) that feed on dead plant and animal matter, thereby recycling the elements, organic and inorganic compounds to the environment.

Diatom: See Bacillariophyceae.

Differential equation: A relation between a unknown function and its derivatives.

DIN: See Dissolved inorganic nitrogen.

Dinophyceae (dinoflagellates): Cells solitary (rarely filamentous in a few marine genera); mostly swimming by two flagella of approximately equal length and mostly lateral in attachment, one flagellum wound about the cell in a transverse furrow, and one extended posteriorly from the point of attachment in a longitudinal furrow; cells mostly dorsiventrally flattened and differentiated, the longitudinal furrow extending along the ventral surface; cell wall (where present) firm and simple, or formed of regularly arranged, polygonal thecal plates (as in armored or thecate dinoflagellates); pigments in chloroplasts include chlorophyll-*a*, carotenes, xanthophylls, phycopyrrin, often giving the cells a reddish color; food reserve starch, starchlike substances and oil; a pigment-spot (possibly an eye-spot) commonly present; reproduction by longitudinal division, asexual zoospores, and by sexual fusion of gametes (known only in a few instances).

DIP: See Dissolved inorganic phosphorus.

Discharge (Flow): Volume of water flowing downstream per unit time ($m^3 s^{-1}$).

Dissociation: Some of the molecules in water separate into hydrogen (H^+) and hydroxyl (OH^-) ions, a process called dissociation or ionisation.

Dissolved inorganic nitrogen (DIN): NH_4-N + NO_3-N + NO_2-N; usually in mg l^{-1} or $\mu g\ l^{-1}$.

Dissolved inorganic phosphorus (DIP): For the purpose of this study, equivalent to PO_4P; usually in mg l^{-1} or $\mu g\ l^{-1}$.

Dissolved oxygen: The amount of elementary oxygen present in water in a dissolved state. It is commonly reported in parts per million (by weight), or milligrams per liter, of oxygen in the water.

Diurnal: Refers to an event that occurs in a day (24-hour period) or recurs each day. Diel (=daily) is a more recent term (not yet in most dictionaries), that refers to events that recur at intervals of 24 hours or less with no connotation of either daytime or nighttime.

Diversity: A measure of the number and relative abundance of different species.

Dominant: Species most prevalent in numbers or biomass in a particular assemblage, or during a given period in a particular environment.

Ecology: The study of the mutual interrelationships between organisms and the environment.

Ecosystem: The community and the nonliving environment function together as an ecological system or an ecosystem.

Electrical conductivity: The ability of a sample of water to conduct an electrical current: because it depends on the number of ions in solution, also a measure of the total quantity of salts dissolved in a sample of water - the higher the conductivity, the greater the number of ions in solution. Conductivity is quoted as mS m^{-1} or, in non-SI terminology, as $\mu S\ cm^{-1}$, where S is a "Siemen", which is the reciprocal of an ohm (the unit of electrical resistance).

Enrichment: The addition of nutrients, usually inorganic nitrogen and phosphorus containing substances, to water which increase the availability of the nutrients used by aquatic organisms in their growth.

Enthalpy: The enthalpy of formation for an element or a substance is the energy necessary to create this element or substance.

Enzymes: Any of a large and diverse group of proteins which function as biological catalysts in virtually all biochemical reactions, essential in all cells, different enzymes being highly specific for a particular chemical reaction catalysed and substratum reacted on.

Epilimnetic: In the upper water layer, above thermocline in lakes, usually rich in oxygen.

Erosion: The removal of soil from the surface by water, wind, or gravity or organisms.

Estuary: The lower part of a river where freshwater and seawater meet.

Euglenophyceae: Cells solitary, swimming by 1, 2 or 3 flagella; a gullet present in the anterior end of the cell in many members, as is also a red eye-spot; chloroplasts few to many, variously shaped bodies (a number of genera colorless); pigments are chlorophyll-*a*, -*b*, carotenes, xanthophylls and often red carotenoids; pyrenoids usually present, either on or in the chloroplasts, or free in the cell; food reserve in the form of an insoluble starch, paramylum, which is negative to the iodine test, and fatty substances; nucleus large and centrally located; cell membrane in the form of pellicle bands, rigid or plastic with the cells metabolic, frequently striated; certain representatives, e.g. *Strombomonas* and *Trachelomonas*, surrounded by a lorica; sexual reproduction lacking; vegetative reproduction by longitudinal cell division and by encystment followed by multiplication of the cell.

Euphotic: Well-illuminated zone of surface waters allowing photosynthesis to take place; upper layer of the photic zone.

Eutrophic: Water bodies rich in nutrients, especially inorganic nitrogen and phosphorus substances, and hence having excessive plant growth.

Eutrophication: The process of enrichment of water bodies, rivers, etc. with nutrients, especially inorganic nitrogen and phosphorus substances, resulting in growth and therefore increased production of macrophytes and algae.

Filament: A linear series of cells set end-to-end, forming a thread, and held together by their cell walls or sheaths.

Flagellates: Group of unicellular eukaryotic microorganisms, including photosynthetic and non-photosynthetic, heterotrophic species, and classified in various ways as protozoans, protists or algae. Motile in adult stage, swimming by means of flagella, and include both free-living marine and freshwater species and some important human parasites.

Flocculation: Clumping of small particles in the disperse phase of a colloidal system through charge neutralisation; such as the clumping of clay particles and algal cells which can be brought about by the addition of Fe^{3+}.

Genus (plural, genera): Taxonomic group of closely related species, similar and related genera being grouped into families. Generic names are italicised in the scientific literature, e.g. *Scenedesmus* (green alga), *Stephanodiscus* (diatom) or *Euglena* (euglenophyte).

Grazing: The process whereby an organisms (i.e. a protozoan) feeds on living plants (i.e. algae).

Heavy metals: Metals with an atomic mass > 40.08 (i.e. greater than that of Calcium).

Hydrolysis: The addition of the hydrogen and hydroxyl ions of water to a molecule, with its consequent splitting into two or more simpler molecules e.g. $C_{12}H_{22}O_{11} + H_2O = 2C_6H_{12}O_6$.

Impoundment: A reservoir upstream from a dam in a river used for collection and storage of water and for its controlled release as required for use.

Inorganic: Applies to material or molecules which do not contain carbon.

Inorganic: Material or molecules which do not contain carbon

Kjeldahl nitrogen: Organic N + NH_4-N.

Lentic: Living in a swamp, pond, lake or any other non-flowing (i.e. still) water.

Limiting nutrient: A nutrient element whose scarcity can limit plant growth (e.g. compounds of nitrogen and phosphorus); determines the rate of metabolic processes and forms part of metabolic products.

Lithosphere: The crust of the earth (Gr. *lithos*, stone, *sphaira*, globe).

Lorica: A rigid wall-like covering around a algal cell and separated by a space from the protoplast or cell wall. An opening or openings are present; when opening is at the anterior end, then a flagellum usually extend through it.

Lotic: Living in flowing (i.e. running) water.

Macrophyte: A large plant, observable with the naked eye; usually used in terms of vascular plants.

Major ions: Those ions (calcium, magnesium sodium, potassium, bicarbonate, carbonate, chloride and sulphate) that usually form the bulk of the total dissolved solids in inland waters.

Metabolic Pathway: Chain of enzyme-catalysed biochemical reactions in living cells.

Metabolism: Integrated network of biochemical reactions in living organisms, often referring to the biochemical changes occurring in the living organism or cell as a whole.

Metabolism: The integrated network of biochemical reactions in living organisms, often referring to the biochemical changes occurring in the living organism or cell as a whole.

Mineralisation: The breakdown of organic matter into its constituent inorganic components, carried out chiefly by decomposer microorganisms, and, for carbon, during respiration when carbon dioxide is returned to the environment.

Mixo-oligohaline: Term for the salinity range of 0,5 to 30 $^{o}/_{oo}$

Mixohaline: Term for the salinity range of 0,5 to 5 $^{o}/_{oo}$

Nitrate Reductase: A general name for enzymes that catalyse the conversion of nitrate to nitrite. The ability to reduce nitrate to ammonia is common to virtually all plants, fungi and bacteria. The first step, reduction of nitrate to nitrite is chemically difficult, and it involves a large and complex enzyme, nitrate reductase.
$NO_3^- + NADH + H^+ \text{---->} NO_2^- + NAD^+ + H_2O$

Nitrification: The oxidation of the ammonium ion to nitrite (e.g. by *Nitromonas*), and the oxidation of nitrite to nitrate (e.g. by *Nitrobacter*), carried out chiefly by a few groups of soil bacteria (nitrifiers), and also by a few species of fungi.

Nitrogen Fixation: The process whereby atmospheric elemental nitrogen is reduced to ammonia, and which is carried out in the living world only by some free-living bacteria and Cyanophyceae (blue-green algae) and by a few groups of bacteria in symbiotic association with plants (the *Rhizobium*-legume association and the actinomycete-non-legume associations). The reaction is catalysed by the enzyme complex nitrogenase. Biological nitrogen fixation is the chief process by which atmospheric nitrogen enters the biosphere and becomes available as a nutrient to other organisms. The incorporation of atmospheric nitrogen as a nitrogen source into the cells of certain blue-green algae and bacteria.

NTU: Nephelometric turbidity unit: A measure of the turbidity of water.

Nutrient: Any substance used or required by an organism as food.

Oligohalobien species: Species that occur in the salinity range of 5 to 15 $^o/_{oo}$

Oligotrophic: Providing inadequate nutrition; waters relatively low in nutrients, as some lakes and rivers whose waters are low in dissolved nutrients such as inorganic nitrogen an phosphorus and which cannot support much plant life.

Organic: Derived from, or showing the properties of, a living organism; substances containing carbon.

Organic: Molecules containing carbon; showing the properties of a living organism

pH: A measure of the acidity/alkalinity of a solution, the negative $\log_{10}$ of the hydrogen ion concentration. The pH of a neutral solution is 7, that of acid solutions less than 7 and of alkaline solutions greater than 7.

Phosphatase: A general name given to enzymes that catalyse the hydrolysis and synthesis of organic phosphate esters and also the transfer of phosphate groups from one compound to another.

Photosynthesis: The process whereby plants convert CO_2 and water into large, energy-rich, organic compounds by absorbing light energy.

Phytoplankton: Free-floating or weakly swimming microscopic aquatic plant life (includes different algal species).

Plankton: The usually small marine or freshwater free-floating plants (phytoplankton) and animals (zooplankton) drifting near the surface of the surrounding water.

Pollution: The presence of any foreign substance that impairs the usefulness of water.

Population: A group of individuals of the same species living in a certain area.

Primary production: Fixation of inorganic carbon into organic matter by autotrophs, which are therefore called primary producers.

Quartz: Silicon dioxide in a crystalline form.

Raw Water: Water which is available as a supply for use, but which has not yet been treated or purified.

Re-aeration: Contact between air and water permitting the absorption of oxygen into the water from the air.

Retention time: The duration of time that elapses before the reservoir or river volume can be (theoretically) displaced by the inflow volume.

Salinity (Saltiness): The mass of dissolved inorganic salts in a unit volume of water.

Secondary salinisation: Salinity caused by human disturbances

Sedimentation: A process used in water and sewage treatment in which the rate of flow of the water is reduced or stopped, permitting the settling out by gravitation of suspended particles. Sedimentation processes promote gravity settling of solid particles to the bottom of the water column where accumulated solids are removed. Sedimentation occurs naturally in rivers and lakes.

Sewage: The spent water supply after it has received the various household, industrial, and other wastes of a community.

Solar zenith angle: Angle between the sun's direction and perpendicular to the earth surface.

Species: The smallest unit of classification commonly used. A taxonomic unit having two names in binomial nomenclature (e.g. *Scenedesmus opoliensis*), the generic name and specific epithet (italicised in the scientific literature). Similar and related species are grouped into genera. Species can be subdivided into subspecies, geographic races, varieties (e.g. *Scenedemus opoliensis* var. *mononensis*), for cultivated plants named cultivars, and for domesticated animals, breeds or strains. Species in asexually reproducting organisms, such as certain algae, are to a large extent based on morphological, genetic and biochemical characteristics, habitat and host range.

Succession: Geological, ecological, or seasonal sequence of species; the sequence of different communities developing over time in the same area, leading to a dynamic steady state or climax community (used especially of plant or microbial communities); the occurrence of different species over time in a given area.

Taxon (plural, taxa): General term that can be applied to any defined unit (e.g. species, genus, family) in the classification of living or fossilised organisms.

TDS: Total dissolved salts - salinity status (mg l^{-1})

Temperate lake: Lake situated in area with moderate climate.

Temporal changes: Changes over time.

Terrestrial: Living or found on land, as opposed to in rivers, lakes or oceans or in the atmosphere.

Thermal-stratification: Water column arranged in layers because of a difference in temperature.

Time-lag: The time difference between peaks in nutrients and peaks in algal biomass.

Total dissolved salts (TDS): Total amount of material dissolved in water. The most common dissolved substances are usually the cations Na^+, K^+, Ca^{2+} and Mg^{2+} and the anions HCO_3^- (bicarbonate), CO_3^{2-} (carbonate), Cl^- and SO_4^{2-}; usually in mg l^{-1}.

Total nitrogen (TN): Kjeldahl nitrogen + NO_3-N; usually in mg l^{-1}.

Trihalomethanes: The presence of three halogen atoms (any of F, Cl, Br, I or At) in a methane molecule; organic contaminants that are produced in the disinfection treatment process of water.

Trophic: Pertaining to, or connected with, nutrition and feeding.

Turbidity: An expression of the optical property of water that causes light to be scattered and absorbed. In clear water bodies turbidity is primarily determined by organic suspensoids such as algal cells. In contrast, it is the concentration of predominantly inorganic particles, derived from soils in the catchment, that dominates in turbid waters (e.g. African and Australian waters). Commonly measured in nephelometric turbidity units (NTU).

Unicellular: Consisting of one cell.

Variety: A subdivision of a species, owing to its uniformity, either to genetic isolation in nature, or to artificial propagation in cultivation. Used in different senses by different specialists.

Water quality: Those characteristics (determined by the combined effects of its physical attributes and its chemical constituents) of a supply of water which are important in determining its purity and usefulness to man.

Zooplankton: Free-floating or weekly swimming aquatic animal life.