THE POTENTIAL IMPACT OF AN INTER-BASIN WATER TRANSFER ON THE MODDER AND CALEDON RIVER SYSTEMS

by

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AN OVERVIEW OF INTER-BASIN TRANSFER SCHEMES AND THE SOUTH AFRICAN PERSPECTIVE

1.1) WATER OF THE WORLD

Water is a renewable resource and, unlike non-renewable resources such as oil or natural gas, there is no danger that the world is going to run out of water (Biswas, 1983). Current estimates indicate that the total volume of water on earth is 1.4 x 10^9 km^3, of which 97.3% is ocean water. However, only 2.7% of this volume is fresh water, of which 77.2% is stored in polar ice caps and glaciers, 22.4% as ground water and soil moisture, 0.35% in lakes and swamps, 0.04% in the atmosphere and less than 0.01% in streams. Thus, nearly 90% of all fresh water is stored in sources where it is not easily accessible (Golubev & Biswas, 1979).

The extent of water use for any one purpose differs from one country to another, and depends on a variety of factors like the state of economic development, including standard of living; importance and extent of a specific sector in the national economy, efficiency of water use and socio-cultural practices (Biswas, 1983). He also pointed out that the largest part of the global water budget is used for agricultural practices (on a global basis agriculture accounts for nearly 80% of total consumption).

Global warming also has an impact on the surface water resources of the earth that will increase over time. It is predicted that global warming will cause an increase in annual average river run-off, with the result that water availability will increase by 10-40% at high latitudes but decrease by 10-30% over some dry regions at mid-latitudes and in the dry tropics. Drought-affected areas will likely increase in extent (Falkenmark, 2007). Africa is confirmed as one of the continents most vulnerable because of multiple stresses and low adaptive capacity. The projections suggest increasing challenges in terms of increased water stress and adverse effects on food productions (Falkenmark, 2007). This is supported by a study that showed that less rain will fall annually in parts of Africa within 50 years due to global warming, with a decrease in water availability occurring across about 25% of the continent (Appel, 2006). If the rainfall decreases by 20%, Cape Town will be left with only 42% of its river water, and in northern African river water levels would drop below 50% (Appel, 2006).

1.2) A SOUTH AFRICAN PERSPECTIVE ON WATER RESOURCE

1.2.2) The South African climate

While considering the different needs and uses of water discussed above, it must be kept in mind that Africa has a strange geographical position. The Equator bisects the continent, which is squarely exposed to the glare of the sun like almost no other region. The wettest areas are situated around the equator. Vast quantities of moisture from sea and land evaporate and condense into clouds. As the rising air cools, it sheds most of its moisture as rain. Around the tropics of Cancer and Capricorn, the up draught heats up so that any remaining moisture is less likely to condense. In these regions are the arid parts of Africa - the Sahara, Namib and the Kalahari Deserts. The rainy season lasts eight or nine months closer to the Equator, grading down to a mere two or three months further away (Harrison, 1990). In southern and eastern Africa, the proximity of the oceans, and the mountainous terrain, complicate the picture. Here the major rainfall zones are not arranged in neat parallel belts, but vary with altitude. The mountainous areas receive the highest rainfall, the lowlands the least.

Perhaps the most important feature of Africa's rainfall is its unpredictability. In the humid regions, rainfall in any given year may be 15-20% more or less the norm. Moving into drier areas, the variation increases up to 30-40% or even more. Together with this, Africa's rains do not fall gently and evenly. They come predominantly as convective storms (Harrison, 1990).

Searching for a single cause of African droughts is probably futile. There are many different regimes of local and regional climate, resulting from different atmospheric processes and topographic features. There are also many different societies in the region, employing different patterns of land use that require varying quantities of water resources. Among short-term climatic fluctuations, droughts in arid and semi-arid regions can be seen as part of the normal climate. In such areas the statistical description of average annual rainfall is skewed because a small number of years with high rainfall are averaged out by a large number of low-rainfall years. It is necessary to look at other statistics, such as the median rainfall, the range (highest and lowest value) and the mode (most frequently occurring value) to describe the rainfall characteristics (Glantz, 1987). In terms of the above, it can clearly be seen that the dry climate in Africa dominates the water environment. This significantly increases the need for inter-basin water transfers.

Thus, the major African constraints according to Harrison (1990), are:

1) The climate, which include rainfall variability, rainfall in storms, alternation of dry and wet seasons and high temperatures.

2) Scarce water resources, which include seasonal variation of rivers, low surface water availability, high evaporation, few sources of shallow groundwater, flat topography and the supply of cheap irrigation water.

1.2.2) South African water resources

South Africa has a very low conversion of mean annual precipitation to mean annual runoff. The mean annual precipitation for the subcontinent is 497 mm (60% of the world average). Sixty-five percent of the country receives less than 500 mm of rainfall and 21% of the country receives less than 200 mm of rainfall per year, with the western part of the country being drier (DWAF, 1994).

Of the annual rainfall, only 8.6 % is converted to runoff. The remainder is lost through evaporation and groundwater sources (Davies *et al.*, 1992). In fact, South Africa has one of the lowest conversions of rainfall to runoff for any area of the world, with a total surface run-off of only 51-53 km^3 per annum (Koch *et al.,* 1990; 1994). Owing to the variability and the high evaporation losses from dams, only about 62% (33 km^3) of the average annual runoff can be used cost-effectively (DWAF, 1994).

In addition, the distribution of water across the sub-continent is spatially skewed. Rivers of the eastern escarpment yield 66 % of the total runoff, while 33 % of the land mass yields 1 %. Also, the bulk of South Africa's population is located in the Gauteng area, where evaporation exceeds precipitation and river flow can be extremely erratic (Davies *et al.,* 1992).

Despite effluent quality regulations, salinisation and eutrophication are two of the important problems threatening water supplies in South Africa. Salinisation is the process by which the concentration of total dissolved solids in inland waters, is increased. According to Koch *et al.,* (1990), salinisation is a common impact derived from urban, agricultural, mining and industrial activities and results from most activities involving the use of water. The situation is exacerbated further in South Africa where water is re-used because of the limited supplies.

The importance of salinisation as a measure of water quality lies in the fact that the usefulness of water, for most purposes, decreases with increased salt content. The cost of increased salinity in water supplied to industries varies according to the type of industry involved, but the cost of any increase in general is high. Du Plessis & Van Veelen (1991) calculated that an increase, from 300 to 500 mg/L in the salt content, could cost Rand Water (a water supply authority in South Africa) users R76 million per annum in terms of water purification. A further increase in TDS to 800 mg/L could cost an additional R63 million per annum (note that this was the projected costs in 1991). Since the water resources of South Africa are limited, and many rivers are non-perennial; salinisation is a problem that requires urgent attention.

Eutrophication on the other hand, is the enrichment of water with plant nutrients, particularly phosphorus and nitrogen, and the consequent excessive growth of phytoplankton and floating, or rooted macrophytes. Such blooms typically turn the water green, due to the presence of high concentrations of algae often accumulating as thick surface slicks and scums. Such blooms cause aesthetic, health and odour problems (Bowling & Baker, 1996). Eutrophication was only identified as a major problem in South Africa in 1975 when a survey was done on 98 impoundments to determine their trophic status (Toerien *et al.,* 1975). About half of the impoundments were found to be low in plant nutrients, eleven were heavily eutrophic and the rest were mesotrophic. Toerien & co-workers (1975) accordingly concluded that urban industrial development, which gives rise to nutrient-rich effluents, was the main cause of eutrophication. It is an important fact that the six most eutrophic impoundments in their study were also the impoundments that received large quantities of secondary purified sewage effluents. The importance of eutrophication lies in its consequences, namely: impaired water body aesthetics, increased water treatment costs, taste and odour problems, potential health risks and unwanted algal blooms (Hynes, 1970; Westlake, 1975; Du Plessis & van Veelen, 1991).

1.2.3) Southern African population numbers

The population of South Africa increased from 38 million in 1990 to 40.58 million in 1996. In 2001, the population of Southern Africa numbered just over 44.8 million, while the estimate in July 2005 was 47.4 million (39 people/km^2) (Statistics South Africa, 2005). The average population growth rate of South Africa is 1.06% (as determined in mid-2006).

The net implication of a rapid population growth is likely to be negative for Southern Africa in terms of water resources. The population of the country is expected to double from 1990 to 2018, which means that the population could be dangerously close to the 80 million people. The Population Development Programme regards 80 million people as the maximum sustainable carrying capacity of Southern Africa's water and other natural resources. The total population in the Free State province in the year 2001, was approximately 2.7 million people. However, HIV/AIDS can have a significant influence on the population growth in the country. In 2005, 320 000 people died of AIDS in the country, and the current HIV prevalence rate for adults (age 15 – 49 years) is 18.2 %. An estimated 5.5 million people in South Africa are HIV positive and there is approximately 1 000 infections daily (Knight, 2006).

In sub-Saharan Africa, where the welfare of more than 80% of the population is directly affected by rainfall because of agricultural activities, the impact of drought could be significant (Glantz, 1987). In addition, the effects of population growth on urban resources can be potentially devastating if badly managed. In South Africa, the government provides six (6) kilolitres of free drinking water to all households, which places even more strain on a resource that is already severely limited.

In view of the above factors, it is clear that water is a very scarce resource in Southern Africa, due to different reasons. These include erratic rainfall patterns, a low mean annual run-off rate, eutrophication, salinisation, fast-growing population numbers, and a high consumer-based water demand, including the supply of free water. Based on the above, it is clear that the demand of water in the country is currently higher than the supply.

South Africa is not the only country in the world that experiences these problems. In fact, different solutions for water scarcity are as old as man and some of these are discussed in the next section.

1.3) SOLUTIONS FOR THE PROBLEM?

As more water becomes necessary for agricultural, industrial and other purposes, certain regions of the world are continuously experiencing shortages in water supply. Accordingly, it has become necessary to explore alternative means to meet increasing water demands. These include desalination of sea water, weather modification (precipitation augmentation), iceberg towing, raw water shipment, flow regulation, more efficient irrigation, stricter law enforcement (conservation), making more efficient use of existing water supplies through reduction of waste, cleaning up polluted streams or recycling water in industry, and inter-basin water transfers (Sewell, 1974; Biswas, 1979; Micklin, 1985).

It seems as if most of these “alternative” water resources have limited potential over the foreseeable future. In the Middle East, more money has been spent on desalination plants than in any other region of the world. Most desalination plants use the so-called multiple stage flash (MSF) technology, but other methods such as electrodialysis (ED), vapour compression desalination (VCD) and particularly reverse osmosis (RO) have been investigated (Agnew & Anderson, 1992). Using new technologies, it is now possible to build and operate seawater desalination plants that charge consumers 0.45 – 0.52 €/m^3 (ZAR 5 - 7) over the 15-20 years design life of the facility (Albiac *et al*., 2003). Desalination is also a developed technology in the United States. Facilities already provided fresh water for municipal and industrial purposes at 650 locations in the 1980’s (Micklin, 1985).

In Spain, a commitment to desalination was proposed in 2004 (Downward & Taylor, 2007). Twenty-one desalination plants are planned for six provinces on the Spanish Mediterranean coast to supplement their water needs. However, desalinated water alone is unlikely to be sufficient to make up the water deficits and water users will have to accept a move to a full-price water recovery by 2010 under the European Union Water Framework Directive (of which Spain is a signatory). Anticipated water efficiencies resulting from higher water tariffs, increasing water re-use and water infrastructure improvements, in conjunction with increasing use of desalinated water, are expected to adequately address the province's current water needs (Downward & Taylor, 2007).

Weather modification through cloud seeding with precipitation-inducing agents such as silver iodide has potential for increasing precipitation, run-off and water supplies. Unfortunately, weather modification has the possibility for significant sociological, environmental and ecological impacts (Micklin, 1985). In South Africa, the South African Rainfall Enhancement Programme (SAREP) was conducted in the Limpopo province from December 1997 to the end of December 2000. This involved an operational cloud-seeding campaign based on hygroscopic flare-seeding technology (developed in South Africa) (Terblanche *et. al*., 2005). Based on the preliminary findings of this study, it was suggested that if 75 of the legitimate storms in the target area are seeded, a marked (10%) increase in rainfall over the area could be realised. Similarly, developments such as the use of icebergs (the question of controlled melting and the necessary links to the drinking water system have not been seriously addressed), and raw water shipment are discounted for the foreseeable future (Swinnerton & Sheriff, 1993).

Another alternative source of water seems to be the recycling of sewage water (Agnew & Anderson, 1992; Micklin, 1985). In South Africa, research was conducted as early as 1968 on the use of pond systems for the purification of sewage water (Meiring *et al.,* 1968). Another example can be taken from Windhoek, Namibia in 1958, when a small surface reservoir, the Goreangab Dam, (with a capacity of 3.6 Mm^3), was built downstream of Windhoek. Subsequently, a conventional treatment plant was constructed to treat the surface water from this reservoir to potable standards. During 1960, the city commissioned its new sewage purification plant on site adjacent to the Goreangab Dam, to deal with the city's domestic and industrial effluent. Until 1963, these two waste streams were treated together and in 1963, a series of anaerobic and aerobic oxidation ponds were added to the purification plant, and the bulk of the industrial effluent load were diverted to these ponds for treatment (du Pisanie, 2004).

In 1969, the above-mentioned conventional treatment plant was converted to treat not only the surface water from the Goreangab Dam, but also the final effluent from the nextdoor wastewater treatment plant in two separate treatment trains. This plant had an initial capacity of 4 300 m^3

per day. This reclaimed water was blended with water from the city's well field and was delivered as drinking water to the city's residents. At this initial stage, reclamation could account for up to 25% of the city's water consumption.

As the demand for water grew, the capacity of this plant had to be increased, and the city of Windhoek obtained loan finance from European Financial Institutions to construct a new 21 000 m^3/day reclamation plant, on a site adjacent to the old plant. Completed in 2002, this plant can now provide 35% of the daily potable requirements of the city. The city has also procured a private sector partner for a 20 year operation and maintenance contract for this plant. The design of the new plant was based on a multiple barrier system (du Pisanie, 2004).

The implementation of proper wastewater treatment systems is also suggested as the main solution for the effective use of water in Mexico city, where historical bad practices have created significant problems that needed urgent attention (Tortajada & Castelán, 2003) as well as in the north China (Wang & Jin, 2006).

The most probable approach that could maximise the water supplies of arid regions is improvements in water conservation methods (Micklin, 1985). Conservation includes the conservation of aquifers through licensed and regulated pumping, the elimination of wastage in the infrastructure and during water use, reducing and/or decreasing priority of inter-basin transfer to industries, reducing afforestation levels, swapping irrigated crops and increasing irrigation efficiency (Micklin, 1985; Nkomo & van der Zaag, 2004). For effective conservation, it is necessary to enact laws which maintain strict control over urban water uses, or implementing a variety of water-saving technical measures for urban water users. The most fundamental strategic measure is raising the price of water. However, in developing countries an increase in the price of water can have significant implications on lower income groups and is not always feasible. Along with this, changes in water laws are needed to facilitate shifts of water from lower to higher value uses (e.g. from irrigation to industrial). In South Africa, the Department of Water Affairs and Forestry is currently implementing a discharge charge system in which industries and other water users will be charged according to the quantity of pollutants discharged into a water resource. This is in accordance with the "polluter pays" principle, as is also contained in the National Environmental Management Act, 1998 (Act 107 of 1998).

Improved irrigation practices also have the potential to conserve significant quantities of water. Some promising measures are aimed at effectively using natural precipitation in agricultural practices. Farming techniques to retain moisture such as improved tillage, deep ploughing, surface mulching, as well as rain and snow melt storage in tanks/ponds, are considered to be particularly useful. Water that collects in closed basins in arid regions could be utilised by direct diversion to irrigated lands (Micklin, 1985).

Lastly, inter-basin water transfers can be considered as a method to increase water resources and is almost as old as man himself. Water transfers by canals can be traced back to at least 4000 years ago in Egypt (Lake Qarun), when the Romans built aqueducts in 200-300 BC up to 90 km in length delivering more than 100 litre per capita per day to Rome (Overman, 1968). In the former Soviet Union, large-scale diversion of water resources from the north to the south of Russia was proposed as early as 1871 (Micklin, 1988), while in Japan, inter-basin water transfer has been carried out since ancient times (Okamoto, 1983).

Several inter-basin water transfers were implemented throughout South Africa to augment the supply of freshwater in areas where insufficient resources occurred to such an extent that inter-basin transfers in South Africa are projected to be 4.82 x 10^9 m^3/year by 2017, involving 8.9 % of the total mean annual runoff (Davies *et al.,* 1992). In the following section, inter-basin water transfer will be defined and the design and implications/challenges of inter-basin water transfers throughout the world (with special emphasis on South Africa) will be discussed.

1.4) DEFINITION OF INTER-BASIN WATER TRANSFER

In simple terms, inter-basin water transfer is defined as: "*the artificial withdrawal of water by ditch, canal or pipeline from its source in one basin (or catchment) for use in another*". Micklin (1985) defines inter-basin water transfer as: "*the purposeful arrangement of natural hydrologic patterns via engineering works (dams, reservoirs, tunnels and pumping stations) to move water across drainage divides to satisfy human and other needs*". Biswas (1983) defines the term as: "*a large-scale artificial mass transfer of water from a water-surplus to a water-deficient region in order to further the economic development of the latter, mainly through agricultural and industrial development*". This could be achieved by diverting the course of a river, or by constructing a large canal which could carry a significant portion of available water. Both these alternatives have important economic, social and environmental impacts which need to be carefully analysed and evaluated (Micklin, 1985).

Long-distance water transfer, inter-regional water transfer, inter-river transfer, large-scale water transfer, inter-catchment water transfer and inter-basin water transfer are all terms which have been used to describe this transport of water from an area of surplus to one of deficiency. The common core of the terms used for transfers is the redistribution or the regulation of natural run-off over river basins. Transfers may be intermittent or pulsed, or may be seasonal or not. There are 15 possible forms of inter-basin water transfers, but if constant versus pulsed flows is added, and seasonal versus aseasonal deliveries, or abstractions, the sum reaches 60 different permutations (Davies *et al.,* 1992). From the above it is clear that the scale of inter-basin water transfers is not easy to define.

In order to establish guidelines for identifying water transfers that falls within the definition of inter-basin transfers, (in Canada specifically), Quinn (1981) used two criteria:

i) The diverted flow does not return to the stream of origin, or the parent stream, within 20 km of the withdrawal.

ii) The mean annual flow transferred should not be less than 0.5 m^3/s.

However, from a South African perspective, Davies *et al.*, (1992) argued against a distance and volume restriction for any definition, because 0.5 m^3/s amounts to a considerable volume annually, and this volume does not take the natural flow of a river into account (flow volumes between rivers could be considerable).

1.5) INTER-BASIN WATER TRANSFERS IN THE WORLD

Although there were many inter-basin transfers planned globally, a significant number was never implemented. Some of these are shown in Table 1.1 below, with a brief explanation why they were not implemented:

Table 1.1: Inter-basin water transfer schemes that have not yet been implemented and the reasons therefore

Country	Brief description of water transfer schemes	Reason for failure
Canada	**The GRAND canal**: a long dyke across James Bay, converting it into a freshwater reservoir from which some of its supply would be pumped uphill, first over the 300m divide into the Great Lakes Basin, then into other drier regions of the North American West (Quinn, 1988; Gamble 1988). **NAWAPA scheme**: would have diverted rivers in Alaska and British Columbia to serve the needs of the western and south-western parts of the United States,	The probability and success of water export depends on the situations in both countries and on the attitude of both Canadians and Americans, who are hardly keen on large scale water export or import. Furthermore, the United States of America has been seeking ways to support itself with its own water supply.

Country	Brief description of water transfer schemes	Reason for failure
	the Prairie provinces and the Midwest of the United States. This would have required the construction of 240 reservoirs, 112 irrigation systems and 17 navigation channels (Sewell *et al*., 1986).	
Egypt	**Jonglei Project** (Bailey & Cobb, 1984; Charnock, 1983): this would have constituted the first phase for increasing the Nile yield by diversion of 20 x 10^6 m^3/day from this river at Bor, through a 360 km canal to the mouth of the River Sobat. Sudan and Egypt would have shared the cost of the Jonglei Canal, and each would have taken half of the extra 4.75 billion m^3 of water. The canal should have improved agriculture and communications in Africa's largest country, improving living standards and enhance flood control.	Due to environmental concerns, opposition groups on the south were against the construction of the canal. When arrests were made, three people were killed in the riots (Collins, 1988). Despite the conflict, the construction proceeded, but when 250 km of the canal was completed, a large number of tiang perished in the ditch. Local people again started to protest against the canal and those opposed to the construction kidnapped some of the workers. An Australian bush pilot was accidentally killed in cross-fire. This caused the immediate termination of all work on the canal, and construction has not commenced since.
United States of America	See NAWAPA scheme (Canada)	
Russia	**Siberian Rivers Diversion** (Voropaev & Velikanov, 1985): The purpose of the planned Siberian rivers diversion project (to the Soviet Central Asia and Kazakhstan) was to supply water resources to the region's economy, primarily for agriculture. Water resources of Siberian rivers would have been used	Ecological concern was the main reason for this scheme not being implemented. However, at an international conference on "Transboundary water resources: protection and ecological stability strategy" held in 2003 at Akademgorodok the idea of

Country	Brief description of water transfer schemes	Reason for failure
	to irrigate 4.5 million ha, including 3 million ha in central Asia and Kozakhstan. About 25-27 x 10^9 m^3 of water would have been taken annually from the Ob River and from the lower reaches of the Irtish. The main intake would have been placed on the Ob downstream of the confluence.	diverting the Siberian rivers of Ob and Irtysh to Central Asia was touched upon once again. The ecological concerns raised during the first attempt at implementation however need to be addressed first.
Turkey	**Peace Pipeline Project** (Agnew & Anderson, 1992): surplus water would have been piped from the catchments of the Seyahn and Ceyhan Rivers in two pipelines. The western pipeline would link Turkey with Syria, the West Bank, Jordan and Saudi Arabia, and the eastern pipeline would go through the Gulf States to Oman. The Turkish weekly reported on May 2006 that the project is now (still) in the "conceptual debate phase". It is now envisaged that the pipeline will convey oil, gas, water and electricity.	Economic consideration was the main reason for failure.
Spain	The main goal of the Spanish National Hydrological Plan is the implementation of an inter-basin water transfer from the lower Ebro River to the north and south Mediterranean coast (Ibáñez & Prat, 2003).	• There will be an increase in salinity in the delta • There will be a decrease in the biological productivity, mostly due to the decrease of nutrient inputs. • The river will carry less sediment, which affects the geomorphology of the system.

Despite these numerous problems associated with inter-basin transfers, various schemes were successfully implemented or are currently being implemented globally. These are discussed below:

1.5.1) China

1.5.1(a) West, Middle and East Route

The spatial distribution of water and land resources in China is, as in most countries, very uneven. There is more water but less arable land in the south and *vice versa* in the north, therefore transferring water from south to north is considered to be the most important. In the early eighties, three major schemes were considered: West, Middle and East Route (Dakang & Changming, 1981; Changming *et al.,* 1985). These would have involved the river basins of both the Huang He and Chang Jiang rivers. This massive project (once completed) will divert up to 44.8 billion cubic meters of water through three canals to the north, approximately equal to the annual volume of the Yellow River in a normal year.

The first-phase eastern section of the project began at the end of 2002 with the long-term goal of providing water for east China's Jiangsu and Shandong provinces. Construction began one year later on the project's central section. The western section is scheduled to begin in 2010. A group of seven Chinese banks agreed on 29 March 2005 to grant a combined 48.8 billion yuan (5.9 billion US dollars) in loans to finance the remaining sections of this water diversion project. The remaining loans will come from the China Construction Bank, Bank of China, Agricultural Bank of China, Industrial and Commercial bank of China, Shanghai Pudong Development Bank and CITIC Industrial Bank.

1.5.1(b) The Grand Canal (Beijing-Hangzhou Grand Canal)

This diversion starts from Beijing in the north and terminates at Hangshou in the Zheijiang Province in the south (Changming *et al*., 1985). With a total length of 1 782 km it links five drainage systems: Hai He, Huang He, Huai He, Chang Jiang and Quiantang Jiang. The construction of this canal has a long history. About 2 400 years have elapsed since the initial excavation was started. It is the world's oldest and longest canal, far surpassing the next two grand canals of the world: Suez and Panama Canal. It is 1,795 km (1,114 miles) long with 24 locks and some 60 bridges. The building of the Grand Canal in China began in 486 B.C. during the Wu Dynasty. It was extended during the Qi Dynasty, and later by Emperor Yangdi of Sui Dynasty during six years of furious construction from 605-610 AD. In 604 AD, Emperor Yangdi of the Sui dynasty made a tour to Luoyang. In the second year, he moved the capital to Luoyang and ordered the canalization of the Grand Canal. This task lasted for six years and thousands of labourers were involved in it.

1.5.2) Japan

Because Japan is an island country, the water transfer distances are shorter and yearly volumes of water transferred are smaller than those of inter-basin water transfer projects in many other countries. The annual precipitation is abundant, but its monthly distribution is not uniform (Okamoto, 1983).

1.5.2(a) Inter-basin water transfer to Tokyo Metropolis

Tokyo, the capital of Japan, was a large city even as early as the sixteenth century. Due to an increase in the population, it became necessary to divert water from the Tama River in the middle of the seventeenth century. In the early twentieth century, water had to be supplied to a population of about 1.1 million in Tokyo, and 50 million m^3 were transferred annually. These temporary solutions could not satisfy the water demands of the Tokyo Metropolis. Consequently, it was decided to transfer water interregionally from the Tone River, which is the largest river in Japan and furthest away from Tokyo. Two reservoirs, Yagisawa and Shimokubo, were constructed to regulate the low flow. The demand for water for the Tokyo Metropolis continuously and rapidly increased and other reservoirs were also constructed at other tributaries of the Tone River (Okamoto, 1983).

1.5.2(b) Shin-Nippon Seitetsu Kabushiki Kaisha (Kitakyushu Area)

Shin-Nittestsu is the largest iron-manufacturing company in the world. When the company was started, it constructed a small reservoir of about 250 000 m^3 in a nearby small stream to start operations. However, after 10 years, they began to withdraw water from the remote Onga River. Thereafter, they constructed reservoirs of about 1.5-7 million m^3 to store water from the Onga River. The demand for water increased further and a larger storage reservoir was constructed upstream on the Onga River to increase the available water (Okamoto, 1983).

1.5.2(c) The Kagawa Irrigation Project

The northern region of Shikoku Island, where Kagawa Prefecture is located, is the driest district in Japan. An inter-basin water transfer project, which flows in the centre of Shikoku to the east, was planned but it was not realised for a long time because of opposition (Okamoto, 1983). However, the project was realised after the second World War, and a reservoir was constructed. At the Ikeda Barrage, some river flow was diverted and then transferred to Kagawa through a tunnel which passes through a mountain range.

1.5.3) North America

About 10% of Americans get their drinking water from the Great Lakes region and 40% of the nation's industry is located in the area (Carter & Hites, 1992). Sizeable inter-basin water transfers are not a recent phenomenon in the United States (Day *et al.*, 1982). Los Angeles and California began importing municipal water from the Owens valley, more than 400 km away, in 1913. Since then, a number of other transfers have been implemented, mainly for hydro-electric, municipal and irrigation purposes. By 1965 there were 146 inter-basin water transfer projects which transferred 26 million m^3 of water annually. The 1960's was the period of most interest in large-scale, long-distance water transfers (Micklin, 1985). Some of the water transfer schemes implemented in North America during this time are listed below:

1.5.3(a) The Ogoki Diversion (Day *et al.*, 1982; Micklin, 1985)

The Ogoki Diversion was completed in 1939 when the Kenogami River was dammed and the flow from 1 000 km^2 was transferred to Long Lake and Lake Superior. Four years later, the Ogoki River Dam diverted a flow of 32 800 m^2 in the Hudson Bay system to Lake Superior as well.

1.5.3(b) California State Water Project

This scheme was constructed to divert flow from northern California to the drier central and southern parts of the state. This was done through the seasonal regulation of the flow of the Sacramento River, along with transfer of water from the delta of the San Joaquin and Sacramento Rivers southwards to supply industry, municipalities and irrigation facilities. The major conveyance feature is the 715 km California Aqueduct that carries water down the San Joaquin Valley and then lifts it nearly a 1000m over and through the Tehachapi Mountains into southern California (Micklin, 1985).

1.6) INTER-BASIN WATER TRANSFERS IN SOUTHERN AFRICA

Moving closer to home, there are several inter-basin water transfer schemes that were implemented in southern Africa. Since southern Africa's water resources are in short supply, due to several factors, inter-basin water transfer schemes have been implemented throughout the region to augment the supply of freshwater. A total of 26 major inter-basin water transfers have been completed in southern Africa and they are listed in no particular order:

- Kunene - Cuvelai,
- Eastern National Water Carrier (Okavango - Swakop)
- Komati Scheme,
- Usuthu Scheme Maputo,
- Usuthu-Vaal Scheme,
- Grootdraai Emergency Augmentation (Orange - Limpopo basin),
- Vaal - Crocodile,
- Tugela - Vaal Scheme,
- Mooi - Umgeni Scheme,
- Umzimkulu - Umkomaas - Illovo Scheme, (Umzimkulu - Umkomaas basin),
- Amatole Scheme (Kei - Buffalo & Nahoon basin),
- Palmiet River Scheme,
- Riviersonderend - Berg River Project,
- Orange River Project (Orange - Great Fish basin),
- Orange - Riet,
- Caledon - Modder,
- Orange - Vaal,
- Lesotho Highlands Water Project (LHWP),
- Vaal - Gamagara Scheme,
- Springbok Water Scheme,
- Vioolsdrift - Noordoewer,
- Molatedi Dam - Gaborone,
- North - South Carrier,
- Turgwe - Chiredzi (Zambezi basin),

Driven by the need for the optimal utilisation of South Africa's scarce water resources, extensive tunnelling works will have to be undertaken during the coming decades to facilitate the transfer of water from wherever it may occur to where it can be applied to the overall benefit of the country. Basson & van Rooyen (1998) listed some planned or near completion water resource developments in southern Africa that will involve significant tunnelling works (Table 1.2).

Table 1.2: Planned water resource development in southern Africa that will involve significant tunnelling works

Location	Length (km)	Diameter (m)	Geology	Approx. timing
Tugela- Vaal (north)	4.1 and 5.3	3.0 – 3.2	Interbedded sandstone and siltstone	2012 - 2025
LHWP Phase II	27.3	4.5	Basalt	2003-2008

Location	Length (km)	Diameter (m)	Geology	Approx. timing
	44.7 37.4	4.03 3.89	Basalt Karoo sandstone and mudstone	
Orange-Vaal (alternative)	31 km total (0.55 km to 8 km sections)	6.1 – 6.7	Sandstone, mudstone and dolerite	2003-2008
Mzimvubu - Vaal	100km to 200 km total	2.5 – 6.1	Karoo sediments and dolerite	2015-2020

Some of the bigger inter-basin transfer schemes that have been built are discussed in more detail below:

1.6.1) Eastern National Water Carrier (ENWC) in Namibia

Forecasts of water shortages in the Okahanja/Windhoek area for the beginning of the 1980s led to the planning of the ENWC. The ENWC was to transport water by canal and pipeline from the Kavango River on the north-eastern border of Namibia, overland past Grootfontein to storage dams north of Windhoek. It was a four-phase project which would have ultimately transported water from the Kavango River south-westwards to Windhoek, a distance of some 750 km (Comrie-Greig, 1986).

The first two phases were completed by 1985, while phase III was constructed thereafter. Phase I, completed in 1978, involved the Von Bach Dam on the Swakop River, the Swakoppoort Dam 55 km below Von Bach, and a pump system to Windhoek, 53 km away. Phase II, comprising the earthfill Omatako Dam on the Omatako River, and a pump scheme, which transfers water from the Omakato River to the Von Bach Dam, was completed in 1983. Construction of phase III commenced in 1981 and comprised the 263 km long Grootfontein-Omatako Canal and the Karstland Borehole System. For 203 km, the canal is an open concrete-line structure designed to discharge between 2 to 3 m^3/s. Around 70 boreholes were sunk in the aquifer and electric pumps abstract water for transfer through pipes to the Grootfontein/Omatako Canal. The Karstland Borehole scheme yield between 15 to 20 x 10^6 m^3/year. Abstraction from this scheme could reach 35 x 10^6 m^3/year, although it is planned not to exceed groundwater recharge rates. Phase IV involved the link between the Kavango River and the Grootfontein/Omatoko Canal. Water is pumped out of the river near Rundu and transported by pipeline for about 250 km to the canal at Grootfontein (Comrie-Greig, 1986; Davies *et al.,* 1992). The draw-off rate from the Kavango River at Rundu is calculated at about

1-3% of the mean annual river flow, and hydrologists believe that this will have a minimal impact on the Okavango Swamps (Comrie-Greig, 1986).

1.6.2) Lesotho Highlands Water Project (LHWP)

The prime objective of the LHWP is to abstract water from rivers in the highlands of Lesotho, store it in reservoirs and transfer it, through gravity, to the water deficient Vaal region in South Africa for industrial and residential use. Before being transferred, the water is used to generate hydropower in Lesotho. South Africa payed for the full cost of the project except the hydropower component and is also paying US$ 1 million annually in royalties for the water delivered.

The treaty for the LHWP was signed on 24 October 1987. This project will ultimately transfer 2 200 x 10^6 m^3/year at its full capacity from the headwaters of the Orange River in Lesotho to the Ash River, a tributary of the Vaal River, the major tributary of the Orange River. The water will primarily be for industrial and domestic use in the Gauteng area. Phase I is divided into two. Phase IA comprised the Katse Dam on the Malibamatso River, the Senteline Dam on the Noqoe River and the Tlhaka Dam on the Hololo River. This phase has been completed. Tunnels connect Katse and Sentelina (48 km), and a tunnel (34 km) will also run beneath the Caledon and Little Caledon Rivers, from Tlhaka Dam to the Ash River. Phase IB entails the Mohale Dam on the Senqunyane River, and a 32 km tunnel connecting it to Katse. The Mohale Dam spilled for the first time on 13 February 2006.

Phase II will include the Mashai Dam on the Lower Malibamatso River and a tunnel connecting this to the Tlhaka Dam or, via a pump station, to Katse and then on to Sentelina via the existing tunnel. The final phase will comprise of the Tsoelike Dam and a tunnel and pump station to transfer water up to the Mashai (Davies *et al.*, 1992).

1.6.3) Orange-Fish Tunnel

The Orange-Fish tunnel is 82.45 km long and runs due south from an intake tower at Oviston on the Gariep Dam, to an outlet near the foot of the Teebus Kop. The tunnel is concrete-lined with a diameter of 5.33 m and runs under the Suurberg plateau for the whole of its length. With a gradient of 1 in 2000 the water runs under its own gravity from Oviston to Teebus. Here the waters of the Orange River emerge and are fed into the headwaters of the Great Brak River, which flows into the Grassridge dam, 40 km north of Cradock, then to the Great Fish River, through the Cookhouse Tunnel and finally to the Vogel River. The Teebus end of the Orange-

Fish Tunnel is equipped with huge pepperpot valves which control the flow of the water and would allow fish of about 30 cm and smaller to pass through (Cambray & Jubb, 1977).

1.6.4) Tugela-Vaal Scheme

This scheme lifts water 560 km over the Drakensberg Mountains in KwaZulu-Natal through several dams and pipelines. It delivers water to the Vaal River catchment for use in the Gauteng and Free State Provinces (Davies, 1989). The first phase became operational in November 1974 and yielded a net transfer of 130 x 10^6 m^3/year. Phase II, completed in 1982, contributes a further 217 x 10^6 m^3/year, with a total annual transfer of 347 x 10^6 m^3. This enabled the annual yield of the Vaal River to be increased by 800 x 10^6 m^3, where the Sterkfontein Dam is the main holding reservoir, from where the levels of the Bloemhof and Vaal Dams are controlled (Petitjean & Davies, 1988). Since water transfer over the Drakensberg would require the construction of reservoirs, channels and pumps, it opened the way to build a hydro-electric power station which could further exploit the potential of water resources being made available. Electricity is generated only during peak demand periods or emergencies by channelling water from the upper to the lower reservoir through reversible pump-turbine sets. By pumping water from the lower to the upper reservoirs during low-peak periods, this scheme helps to flatten the load demand curve of the national system by using the excess generating capacity available in these off-peak periods (ESKOM, 2005).

1.6.5) Mooi-Mgeni River transfer scheme

The existing water supply infrastructure of the Mgeni River system can only meet demands in the Durban/Pietermaritzburg regions up to the year 1999 at which point levels of assurance of supply become unacceptably low. Inter-basin water transfer is facilitated at present only through the Mearns emergency scheme commissioned in 1983 which, during drought conditions pumps water, when available, from the Mooi River to Midmar Dam with a maximum capacity of 3.2 m^3/s. To further augment the water supply in the Durban/Pietermaritzburg regions a scheme was constructed to transfer by tunnel (11 km with a nominal diameter of 3.5 m) from a dam (Mearns Dam) on the Mooi River, just downstream of its confluence with the Little Mooi River, discharging into a stream leading ultimately to Midmar Dam on the Mgeni River. The approximate average wet season flow is 6 m^3/s and the peak (short term) flow is 10 m^3/s. Because of the impending water shortages in the Mgeni River system, the existing Mearns emergency pumping scheme will have to be utilised continuously, subject to the availability of flow in the Mooi River (DWAF, 1996-7).

The Mooi-Mgeni transfer scheme was authorised to commence in March 2001. This authorisation was for:

- Increasing the height of the Mearns weir with 8 metres
- Raise the Midmar dam with 3.5 metres
- Provision of standby pumping capacity of exisiting Mearns station
- Servitudes of aqueduct on Mpofana, Lions and Mgeni Rivers.

This scheme was successfully constructed, and the new Midmar Dam was opened on 22 March 2004.

1.6.6) Water supply augmentation scheme to the Kwandebele Region of Mpumulanga (former Eastern Transvaal)

The former self-governing territory of KwaNdebele, as well as Moutse and part of Moretele 2, which now forms a region of the Mpumulanga (Eastern Transvaal) province, experienced very serious water supply shortages at the beginning of 1994. The region's population of 877 000 people required about 20.45 million m^3/annum of potable water. The existing KwaNdebele Regional Water Scheme was officially opened on 16 January 1999 and draws water from the Kameel River, the Rhenosterkop (Umkhombo) Dam on the Elands River, the Loskop Dam canals and the Bronkhorstspruit Dam. An additional 15 million m^3/annum (approximately 0.5 m^3/s) can be made available from Grootdraai Dam for transfer to the KwaNdebele region and additional water can also be allocated to the region from Bronkhorstspruit Dam.

Fifteen million m^3 per year is pumped through a dual 7 km long steel pipeline from Grootdraai dam into an existing canal. This canal then conveys water approximately 36 km to the forebay of the Grootfontein Pumping Station. Water is then pumped through another dual 7.5 km long steel pipeline to the Knoppiesfontein diversion tank where water is diverted to Bossiespruit Dam for SASOL and the balance goes on to the Trichardtsfontein Dam. From Trichardtsfontein Dam water is released down the Trichardtspruit, through the Syferfontein river diversion canals and down into the Rietfontein weir. The Rietfontein Pumping Station supplies water against a total head of 99 m to the western water storage reservoir at Matla Power Station. At Kendal, two new balancing reservoirs were constructed, from where a new gravity pipeline for 35 km conveys water directly to Bronkhorstspruit dam.

Water from Bronkhorstspruit Dam is released into the river below the dam and abstracted 14 km downstream. The Bronkhorstspruit purification works draws this water, as well as water originating in the Hondespruit catchment. After purification water is pumped 10 km to two storage reservoirs.

1.7) IMPORTANT CONSIDERATIONS OF INTER-BASIN WATER TRANSFERS

Any transfer of water within or between basins will have physical, chemical, hydrological and biological implications for both donor and recipient systems, as well as for their estuaries and local marine environment (Davies *et al.,* 1992). Inter- or intrabasin transfers of water can also affect water budgets both at their origins and at their destinations. Over the long term, these effects can alter the overall availability of water for domestic and industrial uses or can decrease the viability of existing water-supply systems (Barringer *et al*., 1994).

Inter-basin water transfer problems can be grouped under three headings: technological, socio-economical and environmental (meaning the natural environment). There is a strong interrelation not only within the main divisions but also between them (Golubev & Biswas, 1979). Naturally, as the size of the inter-basin water transfer projects increases, so does the complexity.

1.7.1) Problems associated with inter-basin water transfers

Apart from environmental consequences involving seepage losses, possible climatic changes and alterations of water quality, there are immense political and legal obstacles involving inter-basin water transfers. If many of the past and present experiences on long-distance water transfer are reviewed critically, the following major issues emerge (Biswas, 1983; Wishart & Davies, 2003):

a) Mass transfer of water is often justified by considering only the direct cost of transporting water.
b) Various other feasible alternatives to inter-basin water transfer are often not investigated, like more efficient use of available water, re-use of waste water, better management of watersheds, improved integration of surface and groundwater supplies and changing cropping patterns.
c) The agricultural sector is usually the major beneficiary of water transfer projects.
d) Opposition to large-scale mass transfer of water in developed countries, especially for interstate and international projects, is likely to increase as more and more water is required for various purposes.
e) The legal implications of interstate and international water transfers are complicated.
f) Since the mid-sixties, opposition to mass transfer of water has increased significantly on environmental and social grounds in developed countries.

g) The transfer of organisms between historically isolated catchments, pose a potential threat to the conservation of biodiversity by, *inter alia*, mixing genetically distinct populations and hence altering evolutionary processes and pathways.

Based on information obtained during or after the construction of the schemes discussed above, several additional problems and challenges were identified, mostly due to unforeseen ecological impacts that were never assessed before implementation (these are summarised in Table 1.3):

Table 1.3: Problems and challenges identified from inter-basin transfer schemes implemented across the world

Inter-basin water transfer scheme	Problems identified
West, Middle and East Route (China) (Herrmann, 1983; Yuexian & Jialian, 1983)	• The Eastern Route will follow the Grand Canal through or along various lakes near the eastern coast and consequently the groundwater level rose, as the levels in the lakes rose. • The rising of the groundwater table was accompanied by salinisation of the soils, especially in the northern dry part. • During the construction phase, soil erosion and disturbance of natural drainage occured. • Surface water was polluted and heavily silted. There was destruction of wildlife habitats, parks, recreation areas and historic sites. • Because of the high silt content and the low slope of the Eastern Route, the channel was badly silted. • The inflow of river water with a high content of nutrients and silt into the lakes had beneficial effects like oxidation of organic material and coliform reduction, but there were also detrimental effects like algal blooms, siltation, build-up of inorganic substances and lower re-aeration. • A decrease in dissolved oxygen in the hypolimnion occured. • Poisonous heavy metals were released and a reflux of phosphate from the sediments took place.

Inter-basin water transfer scheme	Problems identified
	• Other effects included the displacement of people, an increase in water-borne and water-based diseases and an increase in evapotranspiration which could have an effect on the microclimate of the region. • The migration of schistosomiasis was also identified as a concern.
Ogoki Diversion (North America) (Day *et al.*, 1982):	• Closure of the Waboose Dam created a 562 km^2 "mixed" river and lake type impoundment. • The closure also created a trophic upsurge due to the large proportional increase in water surface area and the relatively small quantity of peat bog inundation. The closure created an enlarged aquatic habitat. • Flow prevention exposed many fast flowing water habitats important for fish shelter and fish food production. Damming prevented downstream nutrient transfers and the drift of fish food that characterises river systems. • Water levels and renewal rates have declined in downstream main channel lakes. • This shift from "river" to "lake" brought about a decline in the high species diversity. • The diversion channel of the Ogoki River project increased the average flow in the Little Jackfish River. • Discharge fluctuations in the Little Jackfish River were greater. • The erosion in the Little Jackfish River increased, due to the increasing flow. • The pre-existing biological features of the lower river reaches have been eliminated. • Re-establishment occurred for a limited number of species capable of adapting to sustained scouring, turbidity, and flow fluctuation stresses. • The northern portion of Ombabika Bay experienced prolonged siltation and turbidity due to the inflow of the Little Jackfish River.

Inter-basin water transfer scheme	Problems identified
	• Heavily silted water decreased fish reproduction, food sources, reduced aquatic plant growth and overwhelmed benthic organisms. In Lake Nipigon the diversion caused higher long-term water levels. This led to intensive localised erosion of unconsolidated silt and sand deposits. • The diversion increased water levels with several centimetres in some of the Great Lakes. • Another major problem experienced from this transfer scheme was diversion-induced erosion. In reservoirs, diversion channels, and receiving water bodies erosion leads locally to increased turbidity, degraded water quality, impaired habitats for predator fish species and loss of property and cultural artefacts.
Siberian Rivers Diversion (Russia) (Vorapev & Velikanov, 1985)	• Drop in water levels. • At the place of diversion, the temperature of the water was lowered by 0.2-0.7°C and ice-forming occurred earlier. • Ice thickness increased. • The mixing zone of saline and fresh water moved southwards. • Inflow of dissolved silica was reduced. • Sea-surface temperature increased with more than 1 °C, this increased the fog and decreased the net incoming radiation. • Landslides and underwater erosion grew stronger at some places. • Modification of the ecosystem's biological productivity. • Possible loss of fodder from the Ob floodplains. • Since the flooded area will be reduced, a reduction in fish spawning and feeding areas occured. • The inflow of saline water to the Ob Bay, caused deterioration in hibernation and feeding conditions for most fish species, a reduction in spawning and

Inter-basin water transfer scheme	Problems identified
	feeding areas, a high death rate due to oxygen shortage, and retardation of biological processes. • A strip of territory 10-20 km wide along the Irtish became swamped. Grass meadows were replaced by agriculturally less valuable hydrophilic moss and shrub meadows. • A high content of soil moisture caused by higher underground water levels resulted in loss of arable lands. • Formation of swamps was accompanied by soil salinisation. • The canal partially intercepted ground and surface water resources which feed several lakes. • Rodents multiplied at places with disturbed soil and vegetation cover. • Appearance of wide spots of bare sands because of the construction of the canal.
Eastern National Water Carrier (Namibia) (Comrie-Greig, 1986; Davies *et al.,* 1992)	• The transfer of alien fish species from the Kavango River to the central drainage systems, such as the Swakop River. At present the Okavango system has no alien fish species, while the Swakop River has several. Grid-screens were constructed at the draw-off point on the Kavango River at Rundu and at other key points on the ENWC to prevent accidental transfer of fish or other organisms. • The possible transmission of schistomiasis. • Growth of algae within the open sections of the canal, and deterioration in water quality. • The effects of the open canal on migration rates and annual mortality of wild animals. The annual mortality of wild animals during the first year after completion of the canal was 17 500 animals. This included kudu, eland, gemsbok, ostrich, steenbok, duiker, caracal, wild cat and cheetah, but the main victims were snakes, warthogs, tortoises and other small animals.

Inter-basin water transfer scheme	Problems identified
	• Effects of groundwater withdrawal from the Karstveld Borehole scheme on the flora and fauna. • Evaporative water losses from the Omatako Canal and from reservoirs
Lesotho Highland Water Project (Davies *et. al*., 1992).	• Exploitation of the Upper Orange River in Lesotho will reduce the yield from the Orange River Project by more than 1 500 x 10^6 m^3/y. • On the other hand, when completed it will double the yield of the Vaal Basin and improve the water quality of the already greatly stressed Vaal River. • However, transfer of clear water to the turbid Vaal, may increase algal growth. • Apart from the possibility of biotic transfers, and water quality, temperature and hydrological consequences for both donor and recipient systems, the combined impacts on the Noque, Malibamatso, Ash, Caledon, Vaal and Orange Rivers will be considerable. • Phase III of the scheme will increase the flow of the Ash by an order of magnitude, with major implications on channel integrity and ecological functioning
Orange-Fish tunnel (South Africa) (Cambray & Jubb, 1977; Car, 1983).	• During July 1975 (just after the commissioning of the scheme) mass mortalities of fish occurred at the outlet and during August. • Some Orange River fishes were transferred to the Great Fish and Sundays River systems. • The pepperpot valve ports have a greater limiting effect on the size of fish passing through the tunnel and surviving than the grids at the Inlet Tower • Another problem that occurred is that *Simulium chutteri* appeared in large numbers in the Great Fish River, soon after receiving water from the Orange River. This is a pest species with major detrimental economic impacts on the agricultural practices along the river.
Mooi-Mgeni Transfer scheme	As part of the natural environment investigation, the

Inter-basin water transfer scheme	Problems identified
(DWAF, 1996-1997)	following were studied: Mearns Dam: inundation, barrier effects, changes in flow regime, changes in water quality and changes in conservation status. Transfer tunnel: soil disposal, construction activities at the portals and tunnelling effects on groundwater. Receiving stream: inundation of channel and floodplain areas, changes in flow regime, inter-basin species transfer and changes in water quality. • There were no impacts significant enough as to prevent the scheme. • However, serious impacts in the receiving stream and moderate impacts in the Mooi River, relating mainly to an increased and decreased flow regime respectively are inevitable.

These investigations confirmed that impacts associated with inter-basin transfers are related to the physical system, the biological system and the socio-economic system (Golubev & Biswas, 1983).

1.7.3) Factors to consider prior to implementation of an inter-basin water transfer scheme

In order to prevent these problems from occurring, it is necessary to thoroughly assess the potential impacts of a proposed scheme. The magnitude of the problems associated with inter-basin water transfers will differ from one project to another, but some of the major variables that should generally be considered are (Golubev & Biswas, 1979; Biswas, 1983):

1.7.2(a) Physical environment

- Water quantity: level, discharge, velocity, groundwater and losses.

The withdrawal of a significant volume of water could have the effect of discontinuing the river continuum, similar to the effect dams have. The River Continuum Concept (RCC) has been put forward in an attempt to understand river ecosystems (Vannote *et al.,* 1980). The RCC logically regards the entire lotic system as a continuous drainage basin gradient and states that, from the headwaters to the mouth of any river, there is a gradation of physical-chemical conditions that trigger a series of responses within riverine populations, which in turn result in a continuum of biotic adjustments and consistent patterns of loading, transport, utilisation and storage of water and matter, along its entire length. Headwaters tend towards detritus-based heterotrophy (primary production (P):respiration(R) < 1) with little primary production, relying on allochthonous inputs of organic material for energy. This is because of restricted light, a consequence of shading by riparian vegetation (Cummins, 1979). Downstream, in the mid-sized streams, the system becomes more autotrophic (P/R > 1), with increased production of autochthonous organic material, because of reduced riparian vegetation and relatively shallow, clear, water. Further downstream, in the large rivers, the system becomes heterotrophic again, due to light attenuation by depth and turbidity.

However, at the estuary, the velocity of the currents decrease and suspended solids flocculate. The invertebrate fauna of the upper reaches are dominated by shredders and collectors, which give way to grazers and collectors in the middle reaches and to collectors in the lower reaches. Species richness maximises in the middle reaches of the stream continuum (Stanford & Ward, 1979). The downstream communities, in the continuum, depend on the inefficient use of nutrients from upstream communities. Thus, lotic ecosystems receive a continuous supply of nutrients from upstream so one would not expect nutrients to be the primary limitation on algal and microbial biomass production (Elwood *et al.,* 1981). However, the RCC was rendered inapplicable on disturbed rivers by the Serial Discontinuity Concept (SDC) (Stanford & Ward, 1979). The SDC assumes the validity of the RCC and proposes that dams, or inter-basin water transfers, act as disruptions to the natural continuum of hydrological, physico-chemical and biotic changes in an impounded river. It is particularly the discharge from these impoundments and transfer schemes that appears to be detrimental to riverine biotas. Bickerton (1995) also confirmed that an inter-basin transfer causes a disruption of the river continuum downstream of the transfer in the following ways:

- Water quality: sediments, nutrients, turbidity, salinity, alkalinity, temperature effects and toxic chemicals.
- Land implications: erosion, sedimentation, salinity, alkalinity, waterlogging, changes in land use patterns, changes in mineral and nutrient contents of soils, earthquake inducement and any other hydrogeological factors.
- Atmosphere: temperature, evapotranspiration, changes in microclimate and macroclimate.

1.7.2 (b) Biological environment

- Aquatic factors: benthos, zooplankton, phytoplankton, fish, aquatic invertebrates, plants and disease vectors.

In addition to the invertebrate fauna, the kinds of fishes in a stream and variability of the fauna, can largely depend on schedules of stresses and disturbances (Meador & Matthews, 1992), life histories of organisms or their ability to withstand disturbance (Poff & Ward, 1990), and the size of the stream, habitat complexity or chemical composition of the water (Matthews *et al.*, 1996). Thus, the degree to which water transfer changes the physical features or flow schedules of a natural stream determines the extent of the effects on the fish fauna.

- Land-based factors: animals, vegetation, loss of habitat and enhancement of new habitats

1.7.2(c) Human activities

- Production: agriculture, aquaculture, hydropower, transportation (navigation), manufacturing, recreation and mining.
- Socio-cultural: social costs, including resettlement of people, infrastructural developments, anthropological effects and political implications.

1.7.2 (d) The availability of water (both in space and in time)

In order that reliable forecasts of water availability can be made, it is necessary to have adequate data over a reasonable period of time. Other problems associated with data on water quantity and quality (in addition to data scarcity), are misrepresentation of monitoring sites, unavailability of trained technicians and a lack of experienced professionals to analyse the data (Biswas, 1983).

1.7.2(e) The nature of demand functions

As the water requirements for various purposes continue to increase and available sources become more and more exploited and polluted, it is highly likely that emphasis will shift from a supply to a demand management perspective (Biswas, 1983).

1.7.2(f) The current efficiency of water use

As mentioned earlier, the agricultural sector (for example) is one of the most inefficient users of water, and improvements should be made. Most historic irrigation systems were inefficient in their water usage, but the situation has improved markedly in recent years. Unfortunately, instead of attempting to make irrigation systems more efficient and then maintaining them at such levels, engineers are instead investigating new sources of water for irrigation (Biswas, 1983). In addition, many households are using drinking water for gardening and industries not only use water, but also discharge pollutants into watercourses.

However, various methods are available to increase the efficiency of water use in the agricultural sector. Studies by Rockström *et al*. (2003) had shown that supplemental irrigation for smallhold farms can be achieved with water harvesting systems that collect local surface run-off in small storage structures. Even small volumes of stored water can significantly improve the household economy.

Another method that can increase water use efficiency is conservation tillage (Rockström *et al*., 2003). Conservation tillage covers a spectrum of non-inversion practices from zero tillage to reduced tillage and it aims to maximise soil infiltration and soil productivity while minimising water losses.

These methods are also supported by Studer *et al.* (1999), who stated that soil and water management recommendations to improve the agricultural productivity from erratic rainfall, can be grouped under three main headings: improved tillage practices, different water harvesting techniques and erosion control measures.

1.7.2 (g) <u>Analyses of interconnections</u>

Considering the long-term nature of river basin development processes and the long service time of the water transfer projects, the analysis of their interconnections can contribute to the evaluation of the impacts of water transfer solutions. The main concern is to consider the possible changes of the interactions during the river basin development process. Some of the interconnections between river basin development and water transfer projects that could be investigated, are:

a) The different demands for water.
b) Importance of the project.
c) Size and character of the project.
d) Complexity of the involved natural and socio-economic resources.
e) Resource demand.
f) Availability of the information needed.
g) Upstream and downstream interests.
h) Integration of the project with the basin wide system.

1.7.2(h) <u>Planning and execution</u>

According to Okamoto (1983) the following steps should be taken during the planning and execution of inter-basin water transfer projects:

1. Fully examine all the possible alternatives to the inter-basin water transfer.
2. Examine the advantages and disadvantages associated with the various routes available.
3. It is desirable that the plan, survey, design, construction and operation be made ingeniously and in stages so that the transfer effects can appear before the entire transfer is completed.
4. The effects will become larger when the project encompasses a larger area.
5. Care must be taken in planning the operation, since conflicts of interests are likely.
6. Carefully examine all potential negative effects.

An UNESCO workshop held in April 1999, also identified five groups of criteria that can be used to evaluate inter-basin water transfer projects (Bruk, 2001):

- The receiving area must face water scarcity that cannot be avoided by other reasonable measures.
- Water resources of the area of origin must be adequate, and any loss must be compensated, using the word in a broad sense.
- Substantial environmental damage should not occur in either area.
- No substantial socio-cultural disruption should result in either area, including emotional and religious motivations.
- The benefits of the transfer should be equitably shared between the area of delivery and area of origin.

It is also important that, after an inter-basin water transfer is completed, some protection measures are planned and implemented to protect the water source. These should include (Jinghua & Yonghe, 1983):

a) Measures must be taken that the main canals are not used as sewage and industrial discharge channels.
b) Control over pollution sources must be strengthened.
c) Water source protection districts should be established.
d) River pollution monitoring should be improved and monitoring points be set up.
e) Laws should be drawn up providing rights to sue and demand compensation for losses from units that wilfully discharge waste water into the conveyance canals.

The destruction of wetlands below the Bennet Dam on the Peace River in British Columbia, the destruction of the livelihood of Indian Tribes by the construction of the James Bay project in northern Quebec, the eradication of the fisheries below the Aswan Dam in Egypt, the growth of large masses of water weeds behind dams constructed in Africa, and disastrous economic and other effects of diversions in the USSR (Sewell, 1974) are only some examples of negative results that occurred when inter-basin water transfer projects were implemented without thorough prior consideration and Environmental Impact Assessments (EIA's).

1.8) SOUTH AFRICAN WATER LAW

The water law of South Africa has been developed over the past 300 years, primarily in the interests of agricultural land owners, and more recently in the interests of industry and urban municipalities. First, the laws were based on the legal concepts brought by the Dutch and British colonialists that distinguish between private and public water. After that, the water laws were firstly codified in a number of Irrigation and Conservation Acts between 1906 and 1912, which divided public water into normal and surplus water. With increasing urbanisation and industrialisation a review of the laws became necessary which led to the promulgation of the Water Act of 1956 (Act 54 of 1956). The Act retained the principle of riparian rights but introduced some restrictions on the quantity of surplus water which could be stored or diverted without state sanction. Control by permit of the use of public water for industrial and urban purposes was introduced as well as the establishment of Water Boards (DWAF, 1994).

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

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

a) Land use and human activities influence and impact upon the hydrological cycle and need to be co-operatively managed.
b) Human use of water resources should not individually or cumulatively compromise the long-term sustainability of aquatic ecosystems.
c) Aquatic ecosystems may be sustained at different levels of ecological health, depending on human decisions, in order to achieve a balance between development and ecosystem health.
d) The ecological reserve in respect of international rivers should include sufficient water of sufficient quality for the full reach of the river.
e) Water quality management should ensure that water, of acceptable usable quality, continues to be available to the users thereof and the relevant aquatic ecosystems.

The above-mentioned principles stress that it is important to take a holistic, or all-embracing, view of water management (integrated environmental management), in which a comprehensive spectrum of demands is recognised and evaluated to assess their priority. Integrated environmental management (IEM) is designed to ensure that the environmental consequences of development proposals, such as inter-basin water transfers are understood and adequately considered in the planning process (DWAF & WRC, 1995). One example that illustrates the necessity for a holistic approach, can be found in Edwards County, Kansas, where the community viewed the water resource as a rural heritage and vehemently opposed a rural-to-urban interbasin water transfer (Solís, 2005).

In South Africa specifically, the National Water Act makes provision for the maintaining of an ecological reserve in all rivers. The ecological reserve is the quantity and timing of flow of the correct quality required to maintain the desired state of a water resource. In terms of the definition in the NWA, the ecological reserve is to protect aquatic ecosystems in order to secure ecologically sustainable development and use of the relevant water resource (National Water Act, 1998). It is, thus, possible to fail to meet the reserve as a result of over abstraction or as a result of over supplementation of water.

For example, the abstraction of water from the Caledon River will have an impact on the reserve downstream of the abstraction point and must be taken into account when determining the water users in the different catchments. On the other hand, the ecological reserve in the Modder River downstream of the outfall will be increased and this may provide additional opportunities for abstraction of water for irrigation purposes and providing households with potable water. If the ecological reserve is not taken into account with regard to abtraction or supplementation, invasive or pest species are likely to occur.

In addition to the above, the EIA Regulations of 3 July 2006, promulgated under the National Environmental Management Act, 1998 (Act 107 of 1998), list certain activities that require written authorisation from the competent environmental authority before it can be implemented. These activities include the construction of canals, abstraction of groundwater, diverting the flow of a river, as well as the construction of dams. The process for authorisation involves the following steps:

1. Application for authorisation to undertake the said activity.
2. The submission of a basic assessment report or scoping report.
3. The submission of a plan of study for environmental impact assessment.
4. The submission of an environmental impact report.
5. Consideration of application: deny/grant authorisation.

However, since the Caledon transfer scheme was constructed in 1995, it was not required by law to conduct an Environmental Impact Assessment

In addition, Section 28(1) of the National Environmental Management Act, 1998 places a duty of care on every South African citizen: "*Every person who causes, has caused or may cause significant pollution or degradation of the environment must take reasonable measures to prevent such pollution or degradation from occurring, continuing or recurring, or, in so far as such harm to the environment is authorised by law or cannot reasonably be avoided or stopped, to minimise and rectify such pollution or degradation of the environment*". In light of this section, the duty of care could be applied retrospectively, and should it be found that the inter-basin water transfer scheme, such as the Caledon scheme, causes serious damage to the environment, the applicant can be held liable for remediation costs.

In comparison to South African water law, one can consider the Tennessee Inter-basin Water Transfer Act, that requires public water providers whose rights are secured to acquire a permit for surface or groundwater withdrawals that are diverted outside their basin of origin and may adversely affect surface water flow (Feldman, 2001). This is also an innovative change in policy that will have future benefits.

1.9) CONCLUSIONS

Although it is noted that all rivers are naturally adapted to variation and many rivers require such variation as cues for development and different behaviour patterns, the significance of the impact depends on the degree of variation and this may differ from system to system (Davies *et al.,* 1992). Worldwide the tendency has been for the size and complexity of transfers to increase over time, and whereas small schemes cause only local problems, larger scale schemes are causing greater controversy because of economic, social and environmental impacts. Golubev & Biswas (1983) have pointed out that when water transfer schemes are first planned, very few of the relevant aspects such as environmental and social impacts are taken into account, and decisions are often made purely on economic grounds. Few predictive methods of assessing schemes exist, and where they do, they deal with each aspect separately, rather than adopting an integrated approach.

Because inter-basin water transfers affect many sectors of an economy, a social accounting matrix (SAM) is considered most appropriate for assessing impacts of inter-basin water transfers. The SAM models have been mainly applied in the literature to water management issues. Recently, SAM analysis has been applied to assess impacts of inter-basin transfers. A multi-regional-country level approach to the measurement of water benefits was used for the Komati and Thukela inter-basin transfers in South Africa (Matete & Hassan, 2005). However, these studies did not include ecological considerations of water transfer schemes (instream impacts) and as such, the total cost of the transfer scheme was not calculated. Hence, for sustainable development, in-stream impacts of inter-basin transfers have to be measured and compensated or mitigated against to ensure sustainable development, together with social and economical considerations (Matete & Hassan, 2005). This argument is also strongly supported by Snaddon *et al.* (1998), who assessed the total cost of three inter-basin water transfer schemes in southern Africa.

Micklin (1985) has suggested a number of alternatives to inter-basin transfers as a means of saving water or augmenting supplies. These include raising the price of water; wastewater treatment and re-use, and improved irrigation techniques as well as more innovative solutions such as desalination and weather modifications.

Some observers believed that interest in large-scale inter-basin transfers of water will decline in the wake of rapidly escalating costs, protectionist attitudes in upstream areas, and effective opposition by environmental lobbyists (Sewell *et al*., 1986). At the same time, however, there seems to be no shortage of proposals. Thus, it seems as if there is a continuing belief that inter-basin water transfers still represent the cheapest and most effective strategy available, particularly where problems are large in scale and where transfers from a technical view are relatively easy to accomplish.

To conclude, inter-basin water transfer schemes are enormously expensive, could result in significant environmental damage, and sometimes face opposition from areas of proposed export. However, if a defensible case can be made that other means of water augmentation are simply inadequate to provide a reasonable water supply, inter-basin water transfers may become a reality in solving most water supply problems (Micklin, 1985).

In view of this background, the inter-basin transfer scheme between the Modder and Caledon Rivers (*Novo* Transfer Scheme) was investigated and impacts identified and assessed. However, this study on the *Novo* Transfer Scheme focused on ecological considerations, and did not take economic and social challenges into account.

CHAPTER 2
THE *NOVO* TRANSFER SCHEME

2.1) THE STUDY AREA
2.1.1) The Modder River

The Modder River is a relatively small river which drains an area of 7 960 km^2 in the central region of the Free State Province, South Africa and has a mean annual run-off of 184 x 10^6 m^3 (Toerien *et al.*, 1983). The total catchment wherein the Modder River falls comprises an area of about 17 360 km^2 (Midgley *et al.*, 1994). The Modder River has a number of tributaries and these are, the Klein Modder River, Sepane- , Koring-, Kgabanyane-, Koranna-, Os-, Renoster-, Doring-, Riet- and Kaalspruit.

Water from the Modder River, is stored in the Rustfontein, Mockes, Mazelspoort and Krugersdrift Dams (Grobler & Toerien, 1986; Grobbelaar, 1992) (Table 2.1).

Table 2.1: Morphological and hydrological data of impoundments in the upper Modder River (Grobler & Toerien, 1986)

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

Rustfontein Dam is an important source of supply to the Bloemfontein area while the Mockes Dam is a balancing dam further downstream to regulate the flow through the Mazelspoort Weir where water is abstracted for urban consumption. Krugersdrift Dam is situated further downstream on the Modder River north of Bloemfontein and supplies water to the Modder River Government Water Scheme (DWAF & Ninham Shand, 1999). In the past, about 60% of the potable water supply of the city of Bloemfontein, was provided by the Modder River and the remainder was pumped from Welbedacht Dam (situated in the Caledon River with a storage capacity of approximately 115 million m^3) which is about 150 km south-east of Bloemfontein (Grobbelaar, 1992). At the moment, only about 25% of the potable water supply of Bloemfontein is provided by the Modder River.

Knellpoort Dam, which was completed in 1988, was built in the Rietspruit to augment the water supply to the Bloemfontein area by means of inter-basin water transfer. The Mean Annual Runoff (MAR) into the dam is 14.6 million m^3/year and the capacity of the dam is 36 million m^3 (thus about three times the MAR). The dead storage is 6 million m^3 to make provision for the silt that is transferred from the Caledon River. The evaporation rate from the impoundment is 55 000 m^3/day. Water from the Caledon River is supplied to the Knellpoort Dam by means of the Tienfontein pumping station and a canal with a silt trap, which stretches over a distance of about 2 km. This impoundment also counteracts the siltation problem at the Welbedacht Dam, where the high siltation rate has decreased the dam's gross storage capacity from the original 115 million m^3 to approximately 30 million m^3 in nine years. Thus, in order to prevent similar siltation problems, the Knellpoort Dam functions as an off-channel storage dam (DWAF, 1988).

When considering the Modder River catchment, it should be noted that the development process of river basins (from a water management perspective) can be divided into three consecutive periods: the natural (period I), the developing period (II) and the matured or fully developed period (III). During period I, there is no significant human interference in the river basin, the quantity and quality of water resources are in conformity with natural conditions and fluctuate with them. During period II, deliberate human interference is restricted to that of a local and regional character and grows to basin-wide dimensions. The natural run-off pattern gradually changes and becomes more regulated. Finally, period III is that of total regulation of the river basin.

Considering the fact that present estimates show that the Modder River is being over-exploited and this has necessitated transfers of water from the Caledon River to meet the needs of the user groups, it can be concluded that the Modder River is within Period III (matured/fully developed) of the development phase.

2.1.2) Climate of the area

The catchment area of the Modder River lies between 24° 40' E to 27° 0' E and between 28° 30' S to 29° 25' S (Toerien *et al.,* 1983), and is located in a summer rainfall area, which receives between 600-700 mm per annum, half of which is through thunderstorms (Grobbelaar, 1992). The Upper Modder catchment (upstream of Rustfontein Dam) receives the highest rainfall in the entire Lower Vaal River Subsystem. Rustfontein Dam has a mean annual precipitation of 568 mm/annum with a mean annual evaporation of 2500 mm/annum (Ninham Shand, 1995). Seventy-five percent of the rainfall occurs between October and March (Toerien *et al.,* 1983). The general climate in the study area is typical of the interior plateau of South Africa with warm wet summers and cold dry winters (DWAF & Ninham Shand, 1999).

The study area falls into the Highveld ecoregion (DWAF & Ninham Shand, 1999). In this ecoregion, plains generally characterise the area with low and moderate relief. The area also has significant lowlands having low and high relief, open hills with low relief and closed hills with moderate relief. The vegetation consists of a combination of grassland types, being moister towards the east and drier types towards the west and south. The median annual run-off per quaternary catchment varies from 10-250 mm. The coefficient of variation for annual simulated run-off per quaternary catchment varies from 40-160%. Mean annual run-off varies from 400-1200 mm, and altitude varies from 1250-1750 m above mean sea level. Rock types include sandstone, quartzite, mudstone, basalt and biotite granite. Soil texture types include sand-clay-loam, clay, sand-clay, sand-loam and loam-sand.

2.1.3) Geology and topography of the study area

The Modder River catchment lies within a geological area classified as the Molteno, Red Beds and Cave Sandstone Stages of the Stormberg Series and the Beaufort Series, both of the Karoo system (Grobler & Davies, 1981). Dolerite dykes occur in places. In the west, where the Modder River joins the Riet River, Dwyka tillite as well as various formations of the Ecca Group, which consists of interbedded sedimentary and volcanic material, is found (DWAF & Ninham Shand, 1999). In the east the topography is undulating, and it flattens towards the west. Most of the area is flat, with large areas being endoreic or non-draining areas (Grobler & Davies, 1981). The relief of the study area is very low, except for the extreme eastern part of the Caledon River in Lesotho. The gradient of the study area is therefore low (< 2%) (Seaman *et al.*, 2001). This is in accordance with Rooseboom (1992) who classifies both the Modder and Caledon Rivers as having a gradient between 0.5 and 1%. Approximately 90% of the Modder River can be categorised as lowland sand bed or lowland floodplain, which consists of a low gradient alluvial sand bed channel which is often confined, but has a fully developed meandering pattern within a distinct flood plain. It usually develops in unconfined reaches where there is increased silt content in bed or banks (Seaman *et al.*, 2001).

2.1.4) Soils of the study area

The soils along the river course are described as silty clays in the headwaters areas, with more sandy to silty clays being present downstream towards Rustfontein Dam. The soils are described as frequently dispersive and susceptible to erosion. The soils adjacent to the Modder River have high clay content and are not suitable for irrigation. Some distance from the river, sandy loams are present which are more suitable for irrigation (Ninham Shand, 1995).

2.1.5) Vegetation of the study area

The origin of the Modder River catchment, in the east, falls in the Moist Cool Highveld Grassland changing to Dry Sandy Highveld Grassland, then to Eastern Mixed Nama-Karoo and Kimberley Thorn Veld in the west. The Caledon River catchment falls in the Cool Moist Highveld Grassland (Seaman *et al.*, 2001).

The dominant grass species in the area is *Themeda triandra* (kangaroo grass), *Eragrostis chloromelas* (Boer lovegrass) and *Microchloa caffra* (pincushion grass). Along the river *Phragmites australis* (common reed), *Cyperus marginatus* (lodi grass), *Cyperus longus* (Galingale) and *Typha latifolia* (cattail) occur (Toerien *et al.*, 1983). The headwaters of the river is vegetated by grasses and sedges, with a few *Salix mucronata* (Cape willow) established downstream of the proposed outlet portal. If the headwater channel is vegetated it is with *Themeda triandra* and *Scirpus noduscus*. The riverine vegetation becomes woodier about 10 km upstream of Rustfontein dam. Vegetation occurring in this area includes *Salix mucronata, Protosparagus aspariodes, Populus nigra* (black poplar), *Populus casecens* (grey poplar) and sedges. *Rhus* and *Acacia* species also occur closer to Rustfontein Dam (Ninham Shand, 1995).

The riparian vegetation along the banks of the Modder and Caledon Rivers is modified, since both the rivers are impacted by anthropogenic influences (Seaman *et al.*, 2001). The riparian vegetation at the outflow of the *Novo* transfer into the Modder River and along the banks of the Caledon River above the point of abstraction was classified as Class E, indicating that the loss of natural habitat, biota and ecosystem functions are extensive (Seaman *et al.*, 2001).

2.1.6) Fish species

Identified fish species in the Modder River include: *Labeo umbratus* (moggel), *Cyprinus carpio* (Carp), *Barbus aeneus* (Smallmouth Yellowfish), *Barbus kimberleyensis* (Largemouth Yellowfish), *Barbus anoplus* (Chubbyhead Barb), *Barbus paludinosus* (Straightfin barb), *Clarias gariepinus* (Sharptooth Catfish) and *Austroglanis sclateri* (Rock catfish), *Tilapia sparmanii* (Banded tilapia), *Pseudocrenilabrus philander* (Southern Mouthbrooder) and *Gambusia affinis* (Mosquitofish) (Rossouw, 1973; Barkhuizen, 1994, 1995 & 1996). Table 3.2 lists the above-mentioned species conservation status as indicated on the IUCN list of endangered species (Red Data List).

Table 2.2: Conservation status of fish species on IUCN list (2006)

Species	Conservation status
Labeo umbratus	Not listed
Cyprinus carpio	Data deficient
Barbus aeneus	Not listed
Barbus kimberleyensis	Vulnerable
Barbus anoplus	Not listed
Barbus paludinosus	Least concern
Clarias gariepinus	Not listed
Austroglanis sclateri	Data deficient
Tilapia sparmanii	Not listed
Pseudocrenilabrus philander	Not listed
Gambusia affinis	Not listed

2.1.7) Terrestrial species

Rustfontein Dam is situated in a nature reserve and it provides habitat for a range of large animal species such as springbok, zebra, blesbok and black wildebeest. Many small mammals can also be expected in the Nature Reserve and on the surrounding farm lands. The rest of the catchment area is disturbed, but small mammals, invertebrates and reptiles are present. There are also domestic animals present, such as cattle, sheep and goats.

2.2) WATER TRANSFERS IN THE MODDER RIVER CATCHMENT AREA

Water demand in the Modder River catchment has exceeded supply in the past which necessitated the development of two transfer schemes:

1) From the Caledon to the Modder (Caledon-Bloemfontein pipeline), and
2) The Caledon to Modder also known as the *Novo* transfer scheme.

2.2.1) Caledon-Bloemfontein pipeline

The Caledon-Bloemfontein pipeline was commissioned in 1974 to supply potable water from the Welbedacht Dam on the Caledon River to Bloemfontein Municipality (now BloemWater), since the water obtained from the Mazelspoort Weir became insufficient (DWAF & Ninham Shand, 1999). As sediment deposition seriously affects the yield from Welbedacht Dam, Knellpoort Dam was commissioned to supplement the supply. Situated just downstream of Welbedacht Dam is the Welbedacht purification works with a capacity of 145 ML/day. This water is pumped after purification via a 6.5 km pressure pipeline and a 106 km gravity pipeline to Bloemfontein. The average capacity of the pipeline is 1.7 m^3/s and the maximum capacity 1.85 m^3/s.

2.2.2) The *Novo* Transfer Scheme

The *Novo* Transfer Scheme (the object of this study) was originally proposed to consist of the following components (Ninham Shand, 1995):

a) Tienfontein pump station on the banks of the Caledon River (Co-ordinates: 29º 46’ 00” S and 26 º 55’ 44” E). At Tienfontein pump station there is four vertical Sulzer pumps driven by 775 kW electrical motors. The pump capacity of each pump is 1000 L/s and the accumulated capacity at Tienfontein totals 3700 L/s.
b) A pipeline and canal from Tienfontein pump station to the Knellpoort Dam.
c) Knellpoort Dam on the Riet River (Co-ordinates: 29º 45’ 47” S and 26º 52’ 29” E).
d) The *Novo* pump station on the north eastern side of Knellpoort Dam to transfer water from the dam to the Modder River (Co-ordinates: approximately 29º 45’ 05” S and 26º 52’ 42” E).
e) The Novo pipeline that runs from the pumping station to the headwaters of the Modder River on Dewetskrom Farm, 19.8 km south east of Dewetsdorp (“the outfall”) (Co-ordinates: 29º 29’ 39” S and 26º 45’ 57” E). For most of the route the pipeline is parallel to a gravel road.
f) From the outfall portal the water flows down the Modder River to Rustfontein Dam (Co-ordinates: 29º 17’ 40” S and 26º 37’ 08” E), a distance of approximately 50 km. Water stored in Rustfontein Dam will be treated at the Rustfontein treatment works and pumped to Botshabelo, Bloemfontein or Thaba Nchu. Alternatively, water will be released from Rustfontein Dam to flow downstream into Mockes Dam from where it can be treated and pumped to Bloemfontein.

A map of the Novo transfer scheme is shown in Figure 2.1 below.

Knellpoort Dam was built in 1988 by the Department of Water Affairs and Forestry in the Rietspruit. The damwall was the first arched rollcrete wall in South Africa (45 m high). Arched dams are most commonly used in situations demanding high walls. Usually, their maximum depth is much greater than their mean depth, and as a result a considerable proportion of their volume is derived from the littoral areas which were created when the former riverine flood plain was permanently flooded (Thornton *et al.*, 1992). In the case of Knellpoort Dam, the majority of water was pumped from the Caledon River into the impoundment, since the Rietspruit (in which the impoundment is situated) is a very small river. Knellpoort Dam is an off-river storage dam in contrast to the more traditional form of in-stream impoundment, because the inflow and outflow are artificially created.

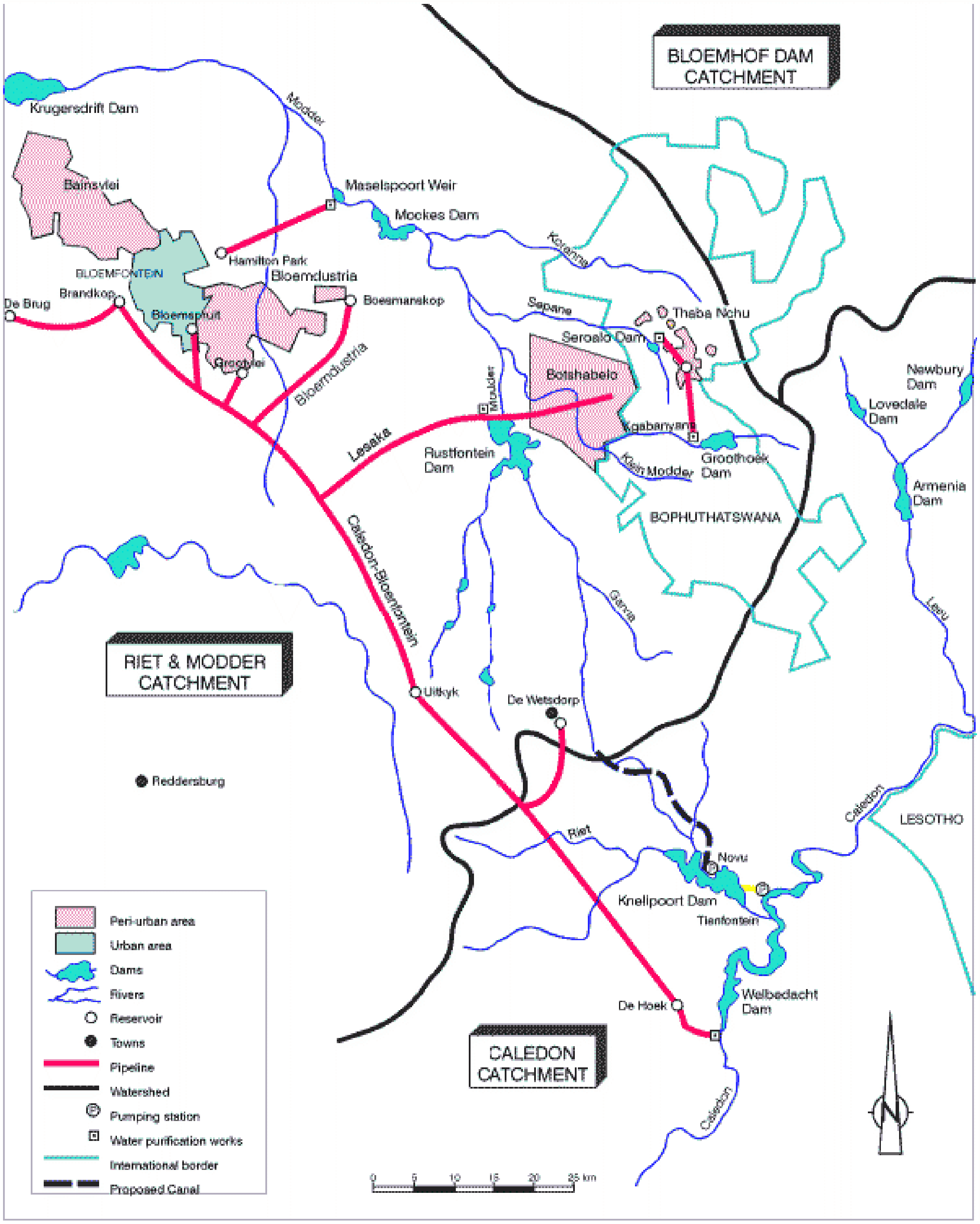


Figure 2.1: A map showing the Novo transfer scheme

At the *Novo* pump station, there are 2 pump sets, with a vertical and horizontal spilt casing pump per set. Vertical pumps lift the water from a tunnel *via* a 20 m deep shaft, while the horizontal spilt casing lifts water about 120 m to a break-pressure reservoir. The capacity of the *Novo* pump station is about 900 L/s. This is an unmanned pump station and is controlled by telemetry from the Welbedacht purification plant.

The canal from Knellpoort Dam to the Modder River runs from the proposed pumping station on the north eastern side of Knellpoort Dam to the headwaters of the Modder River on Dewetskrom Farm, 19.8 km south-east of Dewetsdorp. For most of the route, the pipeline is laid parallel to an existing gravel road. The pipeline is located approximately 10 m from the side of the road in the adjacent farmlands. The pipe is 1.2 m in diameter and made of concrete. The pipeline is buried at a depth of 0.3 m with the access portals no bigger than 2 x 3 m at approximately 500 m intervals. Farming activities are allowed above the pipeline. From the outflow portal the water flows down the Modder River to Rustfontein Dam, a distance of about 50 km. Water stored in Rustfontein Dam is treated at the Rustfontein treatment works and pumped for use to Botshabelo or Bloemfontein. Alternatively, water can be released from Rustfontein dam to flow downstream into Mockes Dam (DWAF & Ninham Shand, 1999).

The optimum operating levels at Knellpoort and Rustfontein Dams are 60% and 40% respectively. From the period between 1998 and 2003, water was released twice from Knellpoort to Welbedacht Dam. The pumping schedule is influenced by water demand and environmental requirements in the Modder River (Ninham Shand, 1995).

Currently, the *Novo* Transfer Scheme is capable of transferring 1.5 m^3/s from the Knellpoort Dam in the Caledon basin to the upper reaches of the Modder Basin upstream of Rustfontein Dam in order to meet the growing demands in the Bloemfontein area. This scheme can be upgrade to an ultimate capacity of 4 m^3/s (Ninham Shand, 1995).

According to Ninham Shand (1995), the *Novo* transfer scheme was expected to have the following impacts:

2.2.2 (a) <u>Expected negative impacts of the *Novo* Transfer Scheme</u>

i) Change in flow regime

The most significant impact of the *Novo* Transfer Scheme is the change in flow regime at the headwaters of the Modder River. The headwaters of the river are perennial, with flow occurring after rainstorms mainly between October and March. The magnitude and significance of the change in flow depends on the pumping programme. The relative change in the flow regime decreases the further one moves downstream from the outlet of the pumped water. In the headwater region the water transfer will significantly increase the mean annual run-off. However, in the downstream reaches next to Rustfontein dam the relative change in the mean annual run-off will be less. Specific impacts arising from the change in flow include disturbance to riverine biota, creation of a habitat for blackfly, soil erosion and change in channel morphology.

ii) Change in water quality.

iii) Change in conservation status.

O'Keeffe (1985) classifies the stretch of the Modder River that will be affected by transfer as class B. This means that slight but significant changes in the river could be expected such as mild pollution, water regulation, the presence of alien species, increased siltation and disturbed catchment vegetation.

iv) Damage to existing structures within the river.

v) Reduction in available water because of increased agricultural abstraction.

vi) Transfer of species from one river system to the other.

2.2.2(b) <u>Expected positive impacts of the *Novo* Transfer Scheme:</u>

i) Improved habitat for aquatic species.

ii) Provision of water to Bloemfontein and Botshabelo.

The pipeline will also have impacts on the river ecosystem, the negative impacts being disturbance to vegetation and agricultural land, construction impacts and the positive impacts being the provision of the water to Bloemfontein. However, it is believed that the impacts can be managed to within acceptable levels if mitigation measures are properly implemented.

Considering the above impacts, as well as the impacts from other water transfer projects in the rest of the world, some possible findings during this study could be:

a) The withdrawal of water could have the effect of discontinuing the River Continuum, as in the case of dams.
b) It could cause major changes in flow and subsequently impact on fish and invertebrates in the system. The diversion could promote riverine ecological instability, while channelisation can remove/disadvantage some flora and fauna.
c) Changes in turbidity levels.
d) Changes in nutrient concentrations.
e) Another major problem is diversion induced erosion (Day *et al.*, 1982). In reservoirs, diversion channels, can lead to erosion, which in turn can lead locally to increased turbidity, degraded water quality, impaired habitats for predator fish species and loss of property and cultural artefacts.

2.3) AIMS AND OBJECTIVES OF THIS STUDY

All schemes operating in South Africa up to 1988 were planned without comprehensive environmental/ecological impact assessment due primarily to the fact that it is only in more recent years that the damage caused by such projects has become apparent (Petitjean & Davies, 1988) and also because legislation did not require any studies to be done. Petitjean & Davies (1988) also state that all water projects carried out should be subject to compulsory environmental impact assessment, meaning the objective multidisciplinary assessment of all potential short- and long-term impacts of a proposed project, including an assessment of alternative strategies and alternative sites, before detailed planning can proceed.

Unfortunately not done before the construction of the inter-basin water transfer scheme, this study will concentrate on the following in order to determine the impacts on a multidisciplinary level:

- The ecological effects of the transfer on both the Modder and Caledon Rivers from a limnological point of view, thus excluding the human component and land-based investigations.
- The investigation of the physical and chemical changes of the water quality in both the Caledon and Modder Rivers. The water quality (physical and chemical) was monitored at the following points: Knellpoort and Rustfontein dams (in Knellpoort both the epilimnion and the hypolimnion, and in Rustfontein only the epilimnion); at the abstraction point at the Tienfontein pump station in the Caledon River; and in the Modder River, upstream of Rustfontein Dam (Figure 2.1). Variables determined during the study included: flow, water stage level, nutrients, Secchi depth, chlorophyll *a*, algal species, algal abundance, TDS, major cations, major anions, pH, suspended solids and turbidity. Since the inflow is at the headwaters of the Modder River, no point was sampled above the inflow.
- The determination of primary production in Knellpoort Dam.
- Water flows upstream of Rustfontein Dam, and erosion along the course of the river were investigated.
- Algal identification and enumeration were done. The presence of cyanobacteria (blue-green algae) in certain sections of the Modder River has been reported in the year preceding the study, and special consideration was given to conditions favouring these nuisance organisms.
- Macro-invertebrates were monitored at selected sites, to determine the composition of invertebrate communities, the health of the systems and the possibility of transfer of different species from one river to the other.

CHAPTER 3

THE ROLE OF TURBIDITY IN THE RIVERS AND RESERVOIRS OF THE *NOVO* INTER-BASIN TRANSFER SCHEME

3.1) INTRODUCTION

3.1.1) The significance of turbidity in the aquatic environment

Scattering of light in natural waters is primarily due to suspended particles, and the term turbidity has been applied to the concentration of suspended material as determined by any one of a number of different physical procedures. Scattering by any suspended particulate matter makes a major contribution to the vertical attenuation of light in natural waters. When a photon is scattered as a result of interaction with a particle in the water, it is diverged from its original path. As a result of repeated scattering, photons are made to follow a zigzag trajectory with an increase in the path length. They must traverse in passing through a given depth, with a consequent increase in the probability of capture by one of the absorbing components of the system. In this way scattering intensifies the attenuation of light with depth (Kirk, 1980).

Nephelometrically measured turbidity might be taken as a reasonable parameter by which the above-mentioned scattering properties of different waters can be compared, because there is an approximate constant relationship between nephelometric turbidity and the asymptotic backscattering coefficient for waters with different quantities of scattering material (Kirk, 1980). Another parameter, with which the scattering properties of water can be determined, is Secchi depth. There are at least three variables controlling Secchi depth (Lind, 1986):

1) Absorption and scattering of light by algae
2) Light scattering by non-algal particles
3) Non-particle light absorption such as dissolved humic acids

The size spectrum of particles dissolved and suspended in waters of lakes, rivers and oceans spans twelve orders of magnitude and as mentioned above, includes a variety of living organisms and non-living materials (Melack, 1985). Thomas & Meybeck (1992) defined the inorganic particle types in terms of size. They defined clay as particles with a diameter of < 4 µm, silt as particles with a diameter of 4 - 64 µm, sand as particles with a diameter of 64 µm - 2 mm and gravel as particles with a diameter of > 2 mm. Inorganic particles in freshwater may include detrital rock fragments, autogenic minerals and clays. These particles enter the pelagic region of lakes via riverine inputs, atmospheric deposition and sediment resuspension within the lake (Melack, 1985). They are formed by chemical precipitation, flocculation and decay of °plankton.

Increased levels of turbidity can reduce light penetration in both lakes and streams and are associated with decreased primary production, abundance of plant material, decreased abundance of other organisms (secondary production) and decreased production and fish abundance (Walling & Webb, 1992; Lloyd *et al.*, 1987). Grobbelaar (1984, 1985 and 1989) found that the presence of non-photosynthetic material has an influence on phytoplankton productivity, due to the rapid attenuation of vertical light penetration which results in a compresses photosynthetic profile and usually a shallow euphotic zone (Z_{eu}) in relation to the water column. In addition, an increase in turbidity impacts on feeding patterns of filter feeders, cayse physiological damage and limit habitat for certain invertebrate species (Wilber, 1983).

Turbidity also affects the usage of water for man. It is generally acknowledged that turbid water is less acceptable than clear water consumption, contact recreation, and perhaps aesthetic enjoyment (Lloyd *et al.*, 1987).

Turbidity also influences the chemical composition of natural waters, because the particles are generally charged, thus forming adsorption and desorption surfaces. In view of this, it is possible that phytoplankton can utilise PO_4 that is adsorbed onto suspended material. This possibility is further detailed and investigated in Chapter 5.

3.1.2) Sources of turbidity within a river

Rivers and river processes are considered to be one of the most important geomorphic systems on the earth's surface and discharge within a particular river system is influenced by a number of factors. Consequently, the multitude of different rivers and streams are classified according to discharge, length, basin area, sediment load and by various interrelationships such as those between discharge and solid load, solid load concentration, seasonal variations and basin relief. All these factors influence the flux of suspended matter through a river. To these factors should be added the type of rock or soil that is eroded and the changes brought about by human activity in a basin as well as in the river itself (Eisma, 1993).

Considering the above, there are two major natural sources of sediments to rivers: firstly, products of continental rock and soil erosion and secondly, autochthonous material which is formed within a water body (usually consists of algae and precipitated minerals). However, the autochthonous fraction is usually very small in rivers (Thomas & Meybeck, 1992) and consequently, the suspended sediment concentration is mainly governed by erosion.

Soil erosion occurs when the disaggregating forces of removal overcome the forces of internal resistance within the soil. If the intensity of precipitation is greater than the infiltration rate of the soil in a particular region, water would accumulate on the ground surface and eventually run over it as overland flow, acting as a force of removal. Overland flow appears to occur more commonly in arid and semi-arid regions, where the vegetation cover is sparse (Beaumont, 1975) or where deforestation or extensive agricultural activities have taken place (Meybeck *et al.*, 1992). Erosion of the land and transport of sediments are greater with overland flow, whereas more dissolved materials are transported by sub-surface flows (Allan, 1995). However, reservoirs can have a considerable influence on the suspended sediment concentrations in fluvial systems, by reducing the magnitude and frequency of run-off, increasing evaporation, restricting sediment transport and increasing downstream scour. Intra- and inter-basin transfers of water can also bring about major changes in natural fluvial systems.

3.1.3) Processes regulating quantity of suspended matter in fresh waters

In rivers and lakes, there are two processes that regulate the quantity of suspended matter at any given time, namely transportation and sedimentation/deposition. Both transportation and sedimentation are functions of current velocity, particle size and the water content of the particles. However, transportation is more important in rivers than in lakes/impoundments (Thomas & Meybeck, 1992).

3.1.3(a) <u>Transportation</u>

Under lotic conditions, there are two distinct sediment transport systems functioning: transport in suspension; and transport by traction along the bottom, often termed bedload. The particles being transported in suspension normally consist of finer materials, usually clays and colloids, and occasionally with a substantial proportion of silt, and are geochemically active. However, under extreme flow conditions, sand and gravel may become suspended. The particles transported by traction (bedload) consist of coarser materials, sands, gravels and larger particles, which move along the bottom, and are relatively geochemically inactive. Theoretically, the top layer of suspended matter should consist of clay, the middle layer of silt and the bottom layer of sand and gravel, but this distribution seldom occurs in nature (Thomas & Meybeck, 1992).

3.1.3(b) <u>Sedimentation</u>

Sedimentation is a process that occurs in both rivers and lakes and is dependent on two conditions. The particles must collide (dependent on the concentration) and cohere (dependent on the suspended solids present in water and the physico-chemical forces around the clay particles) (Terwindt, 1977). Suspended particles in natural water are usually negatively charged and repulse each other. Assuming the particles to behave like colloids, the suspended matter will flocculate at increasing salinity. This is because in fresh water, the particles are kept apart due to their negative charges but at higher salinities these charges become increasingly neutralised by the positively charged ions in solution. The positive ions form a cloud around each particle; thus an electric double layer is present, with the negative inner layer attached to the particle surface and the outer layer consisting of attracted positive ions (The potential between the outer and inner layer of the electrical double layer is known as the zeta (ξ) potential). Because of the increasing neutralisation of the negative inner layer at increasing salinity, the repulsive force diminishes, and when the particles come very near to each other, the attraction by Van Der Waals forces or inter-molecular forces can become so strong that the particles come together and flocculate (Eisma, 1993).

The degree of aggregation of particles has a definite effect on the rate of sedimentation. The median fall velocity of flocculated suspensions have a range of 0.1-0.5 nm/s or 0.4-1.8 m/h and the higher the sand content, the faster the flocculation (Terwindt, 1977). Further, in rivers, the rate of deposition depends also on the shear stress exerted by the fluid flow to the bed. Below a certain minimum value of the bed shear stress, particles in the flowing fluid can not remain in suspension but will all gradually sink to the bed.

3.1.4) Sediment yield and turbidity in Southern Africa

Sediment yield contributes a large part of the turbidity that is usually present in rivers. In Southern Africa, many rivers carry high concentrations of suspended material due to soil erosion which can be a result of the sparse vegetation (as is the case in the Caledon River), erratic rainfall and easily-weathered sedimentary rock (Palmer & O'Keeffe, 1990; Walmsley & Bruwer, 1980). The average sediment concentration in the Zaire River was found to be 34 mg/L, in the Niger River 210 mg/L and in the Zambezi River 90 mg/L (Eisma, 1993).

3.1.4(a) <u>Factors influencing erosion</u>

Soil erosion in Southern Africa is extensive and diverse. On the basis of climatic criteria, more than two-thirds of southern Africa is considered to be particularly susceptible to erosion by either wind or water. Areas in southern Africa with high concentrations of backland topography (e.g. Zimbabwe, Lesotho, Natal, Swaziland, Mpumulanga and Northeastern Cape) have a high erosion rate. High erosion rates are evident in a crescent area stretching from Bedford through Queenstown to Barkley East and from there to Smithfield in the Orange Free State (Le Roux, 1990). When anthropogenic influences such as the clearance of natural vegetation, agriculture, and other disturbances to the soil are considered as well, most regions of southern Africa can be described as being susceptible to soil erosion (Beckedahl *et al.*, 1988).

The factors that influence the extent and intensity of soil erosion in southern Africa are complex. Erosion is determined, firstly, by the nature of the processes operating and secondly, by the response of land systems to edaphic conditions, which include climate, geology, topography, soil characteristics and vegetation cover. Variations in the intensity and extent of soil erosion are further influenced by the nature of environmental change that has occurred within a particular area (Beckedahl *et al.*, 1988). Soil erosion is, therefore, dependent on external forces on one hand, and the physical, chemical and biological characteristics of the soil on the other.

The annual sediment yield for South Africa ranges from less than 50t/km^2 in the west to more than 250t/km^2 in the east and between 5 to 25 t/km^2 in the southern Cape Province (Le Roux, 1990). Rooseboom (1978) has estimated that in certain areas of South Africa, particularly where shales and mudstones of the Beaufort and Molteno series of the Karoo system predominate, annual sediment production can be as high as 100t/km^2. Most surface waters with Secchi disk transparencies lower than 0.5 m are to be found in the Eastern Cape Province, Free State and south-western Gauteng areas, where the annual sediment yield is higher than 200 t/km^2. Taking the study area into consideration, the sediment transport in the Modder River was found to be 304 t/km^2/y, in the Caledon River 940 t/km^2/y, and in Rustfontein Dam the sediment yield was found to be 283 t/km^2/y (Rooseboom, 1992).

Run-off, precipitation and evaporation all contribute in making the Caledon River (an important component of this study) a very turbid lotic system, especially during the rainy season. This creates the possibility that the transfer of water from the Caledon River, can have a significant impact on the turbidity patterns in Knellpoort Dam. Furthermore, transfer of water from the Knellpoort Dam could influence the turbidity and flow patterns in the Modder River. In this chapter, factors influencing flow and turbidity in the systems will be investigated and results of the influence of the turbid waters of the Caledon River on the rest of the *Novo* Transfer system are presented.

3.2) MATERIAL AND METHODS

The different sampling sites as shown in Chapter 2, were visited monthly from March 2000 to May 2001. *In situ* measurements were made during sampling visits and subsurface samples (2L) were taken and transported, on ice, to the laboratory for chemical and other analyses. The analyses were done within 48 hours and were stored, in the dark, at about 4°C to limit chemical and biological changes. During November 2000 and October 2001, 24 hour studies were carried out on Knellpoort Dam. During November 2000 water was transferred from the Caledon River to Knellpoort Dam, but during October 2001 no water was transferred (the transfer stopped in May 2001). Thus, two studies were done to compare conditions during and after transfer of turbid Caledon River water to Knellpoort Dam, at approximately the same time of year. At Knellpoort Dam, a monthly sample was taken from the shore at the dam wall of the impoundment, near the point of inflow. However, during the 24 hour studies, samples were taken at 2 different sites (Sites 1 and 2). The sampling points are shown in Figure 3.17(a) on page 70 in Chapter 3.

The rainfall data were obtained from the Department of Meteorology, Faculty of Natural and Agricultural Sciences, University of the Free State.

3.2.1) Turbidity, Secchi depth and total suspended solids (TSS)

The usefulness of turbidity values lies in its ease to measure and giving an indication for both light penetration and suspended sediment concentration (Lloyd, 1987). For waters where transparency is governed by suspended inorganic material, turbidity has considerable value as a measurement, because light attenuation is primarily caused by scattering, and turbidity is a measure of this optical property (Walmsley & Bruwer, 1980).

Particulate matter in suspension is defined as the material that is retained on a 0.4 - 0.5 μm pore size filter. Smaller material is considered to be “dissolved” but it may actually be colloidal or particulate (Eisma, 1993). Consequently, total suspended solids (TSS) were determined by weighing 0.45 μm Whatman GF/C filter papers (W_1), after washing with distilled water and drying them at 120 ^oC. A known volume of water was filtered, and the filter paper was again dried and weighed (W_2). The difference in W_2 and W_1 (done in duplicate and averaged) was taken as the weight of the TSS.

3.2.2) Light

Light profiles were taken in Knellpoort Dam during the transfer of Caledon River water (26/11/2000) as well as after the transfer (11/08/2001). Light measurements were made with a Biospherical Instrument Inc. light meter, fitted with a 4π sensor and was measured in mole quanta/m^2/s.

3.3) RESULTS

3.3.1) Seasonal and spatial changes in turbidity in the Modder and Caledon Rivers

The average turbidity at the different sampling sites in the *Novo* transfer Scheme is shown in Table 3.1.

Table 3.1: The average turbidity (surface water) at the different sampling sites in the *Novo* Transfer Scheme (n = 14)

Sampling site	Average Turbidity (NTU)	Min. and max. values	Standard deviation	Variance	Median
Caledon River	4 460	Min = 20 Max = 22 250	6 745	22 230	990
Knellpoort Dam	32	Min = 7 Max = 67	19	60	22
Modder River	165	Min = 14 Max = 460	144	446	135.5
Rustfontein Dam	32	Min = 17 Max = 56	11	39	31

These results as logarithmic values are also given in Figure 3.1.

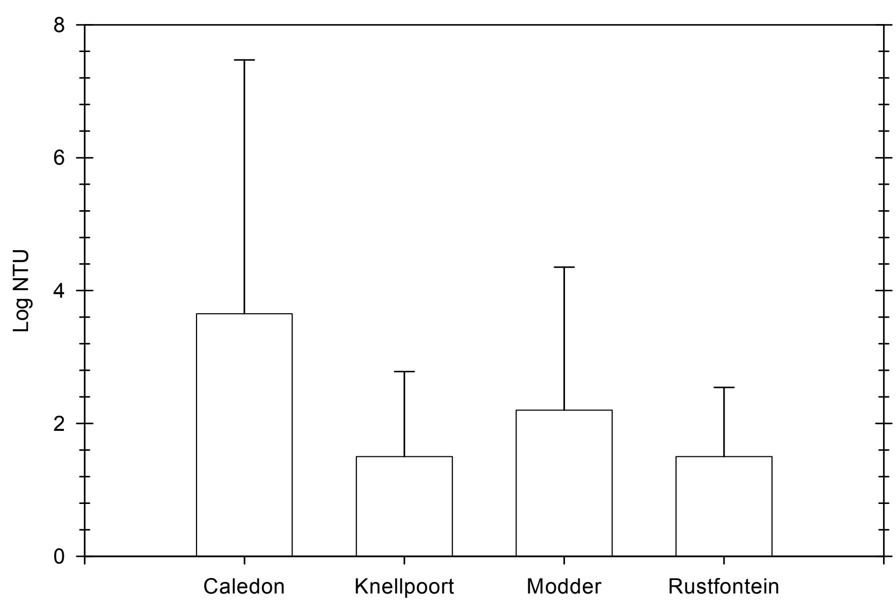
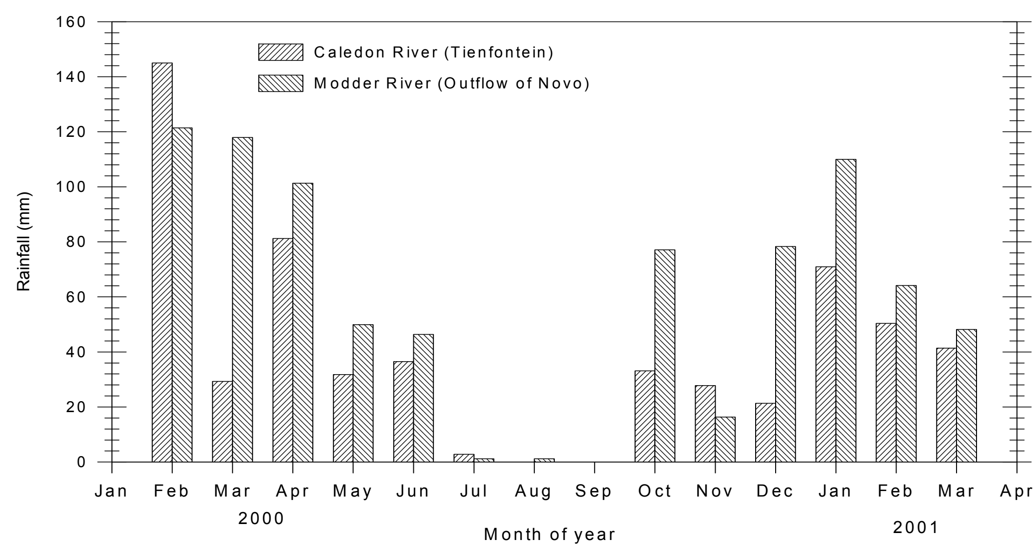


Figure 3.1: Bar chart showing the log values of average turbidity at the different sites in the transfer scheme. The error bars shown are the standard deviations.

Since there was such a large variance in the data of the Caledon River (average of 4 460 NTU and a standard deviation of 6 745), it is difficult to make conclusions from the averages, and the median values may be a better indication of the differences between the systems. However, it was confirmed during the study period that the turbidity in the Caledon River was consistently higher than that of the Modder River, except during the dry season, when flow was low. Although the variance in the Modder River was also significant (average 165 NTU, standard deviation 144) due to the seasonal fluctuations in turbidity, it was not as high as in the Caledon River.

The turbidity in the Modder River above Rustfontein Dam followed seasonal variations (Figure 3.3), with an increase after the rains in September to October (Figure 3.2) and a decrease during the dry season (winter) from June to September. The same pattern was observed for the Caledon River (Figures 3.2 and 3.4). As expected, the Secchi disk depth in the two rivers decreased when turbidity increased with higher rainfall (Figures 3.3 and 3.4). Where values are missing, it was due to either very high turbidities that influenced the readings or equipment that was not available to do the measurements.



<u>**Figure 3.2:** The seasonal variation in rainfall in the catchment area of the Caledon and Modder Rivers. No rainfall was recorded for the periond July to September (winter).</u>

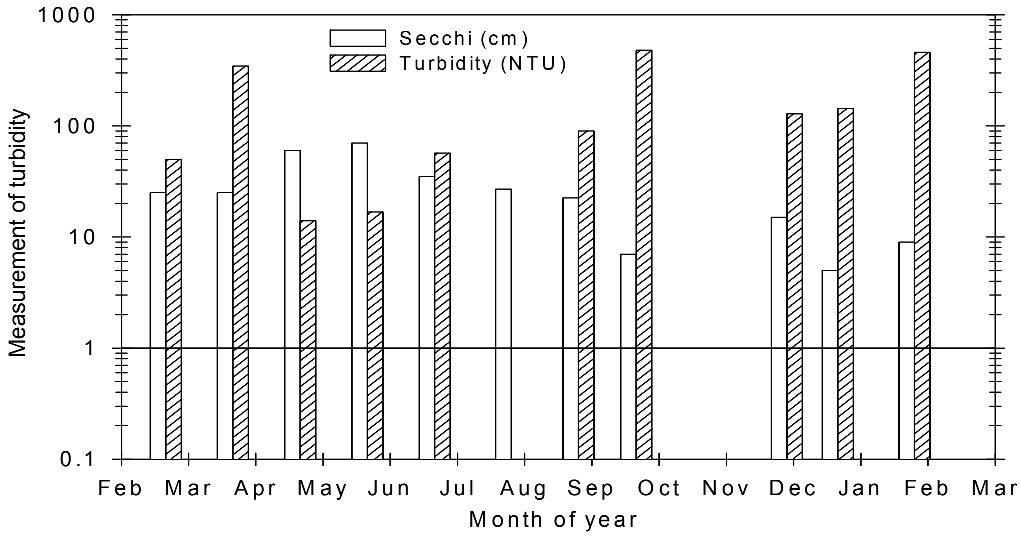


Figure 3.3: The seasonal variation in turbidity and Secchi disk depth in the Modder River above Rustfontein Dam.

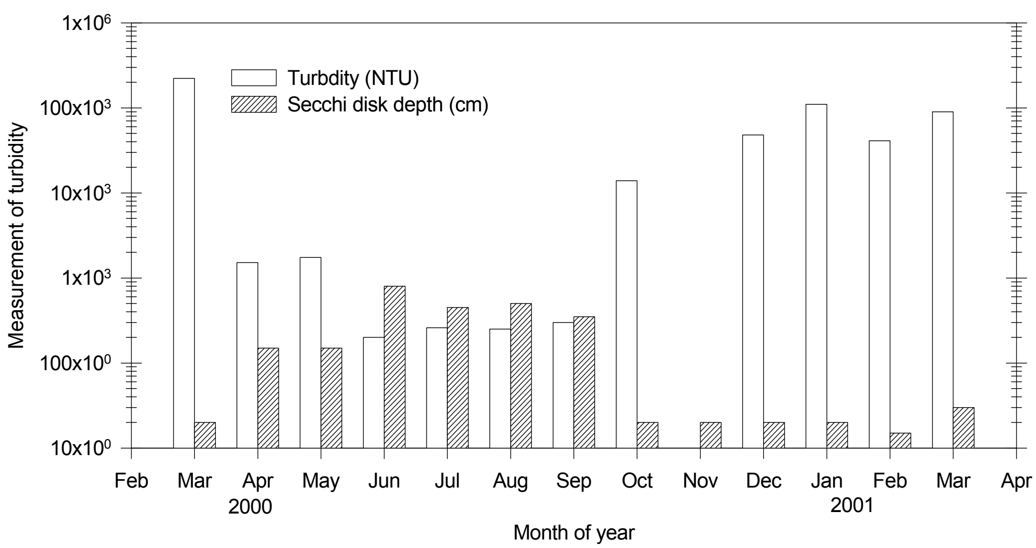


Figure 3.4: The seasonal variation in turbidity and Secchi disk depth in the Caledon River at the point of abstraction

A negative exponential relationship (r^2 = 0.704) was found between turbidity and Secchi depth in the Modder River (Figure 3.5). For the Caledon River, no significant relationship was found, although expected to be inverse. The lack of a significant relationship is most probably because the turbidity varied too much.

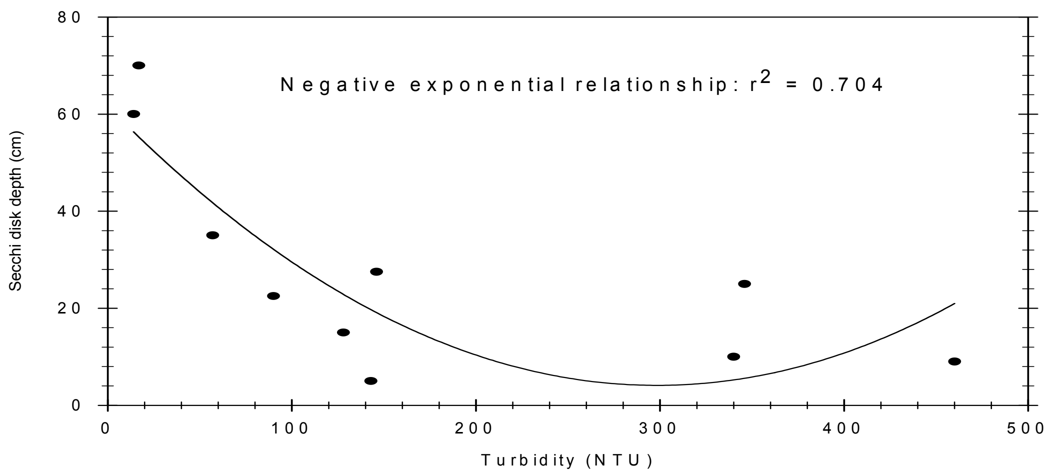


Figure 3.5: The negative exponential relationship between turbidity and Secchi depth in the Modder River.

Furthermore, and as to be expected, the TSS concentration in both rivers also increased after rains and decreased during the dry season (Figure 3.6).

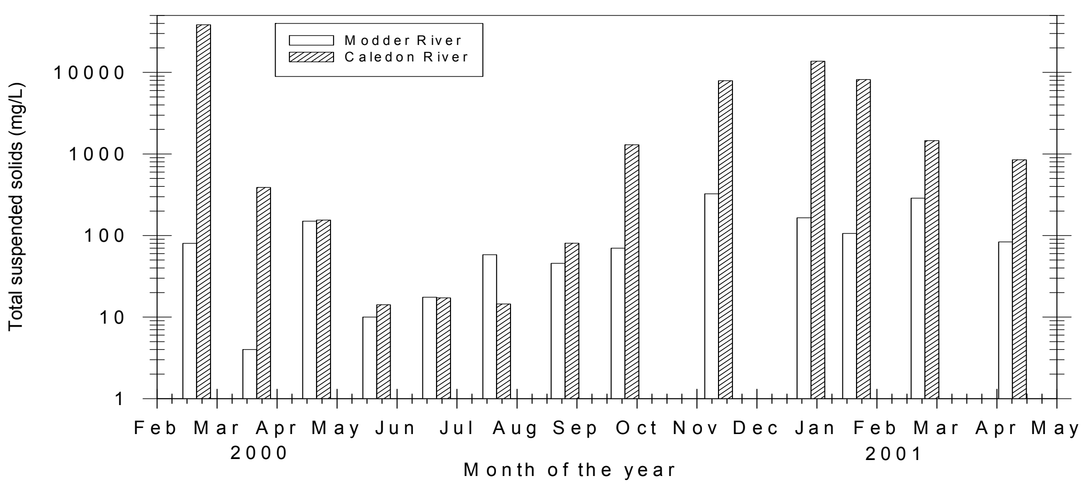


Figure 3.6: The seasonal variation in TSS concentration (mg/L) in the Caledon River at the point of abstraction and in the Modder River above Rustfontein Dam.

Seasonal variations in turbidity in the impoundments showed different patterns than those of the rivers and were not closely correlated to the rainfall patterns. In Knellpoort Dam, the turbidity was low during the study period (except in June at the bottom of the water column), but increased significantly (135% at the surface and 102% at the bottom of the impoundment) after water was transferred from the Caledon River (from November 2000) (Figure 3.7). During June

2000, the water sample taken at the bottom of Knellpoort Dam, contained organic detritus such as pieces of leaves and twigs.

Despite the transfer of water, the turbidity was low in Knellpoort Dam during February 2001. This can be ascribed to the fact that, since it was a calm day, mixing was minimal and increased turbidity was confined to a very small area of the dam at the inflow of transferred water from the Caledon River. An increase in chlorophyll *a* (from 4-14 µg/L in June 2000; and from 7-28 µg/L in February - March 2001) also contributed to the increase in turbidity in the impoundment during June 2000 and February - March 2001 (Figure 3.7).

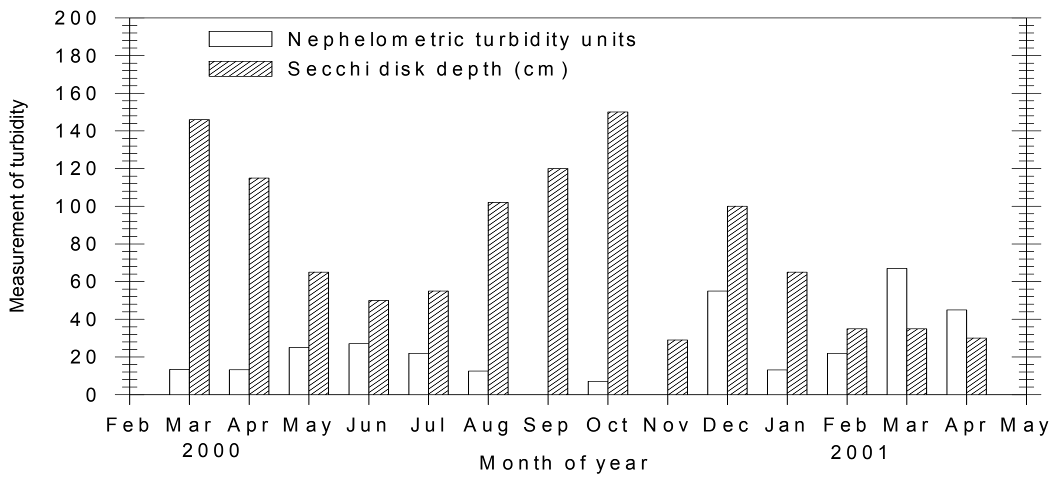


Figure 3.7: Seasonal variations turbidity and Secchi disk depth in Knellpoort Dam.

In Rustfontein Dam, turbidity varied little compared to Knellpoort Dam, and showed a decrease in turbidity from May to August, whereafter there was an increase with not much variation over time from September to March (Figure 3.8). The increases in May 2000 and March 2001 can also be ascribed to higher chlorophyll *a* values (from 5.73 - 25 µg/L in May; and from 6.5 - 18 µg/L in March). No significant increase in chlorophyll *a* was observed in the impoundment during October (results of the chlorophyll *a* determinations are shown in Section 6.3.1 of Chapter 6).

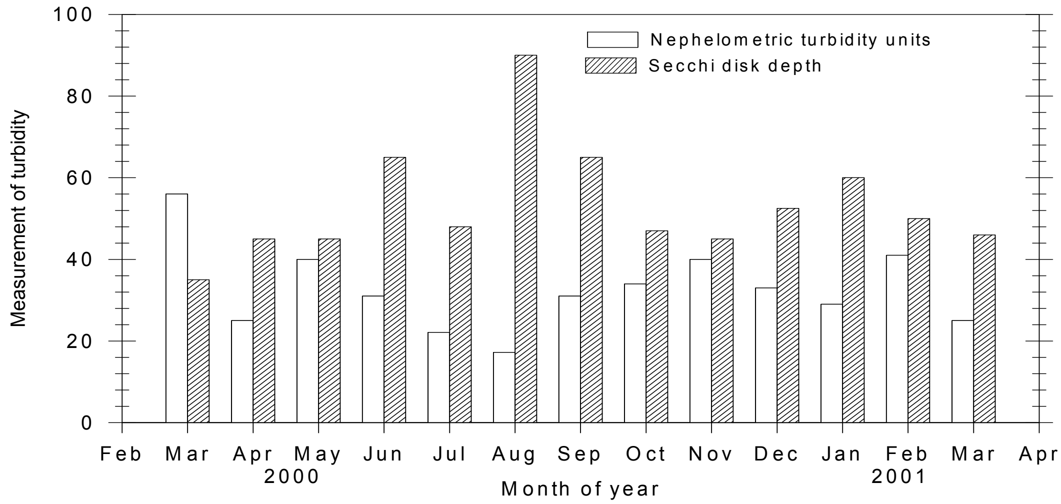


Figure 3.8: Seasonal variations in turbidity (NTU) and Secchi disk depth (cm) in Rustfontein Dam

Variations in turbidity of the Modder River were much larger than in the impoundments constructed in it (e.g. Rustfontein Dam). The minimum turbidity in the river was 14 NTU and the maximum about 460 NTU (a difference of 446 NTU), while in the impoundment the minimum was 17 NTU and the maximum 56 NTU (a difference of 39 NTU) (Figure 3.9). As also mentioned above, the average turbidity in the Caledon River (4461 NTU) was much higher than that of the Modder River (164 NTU), and was also consistently higher during the study period.

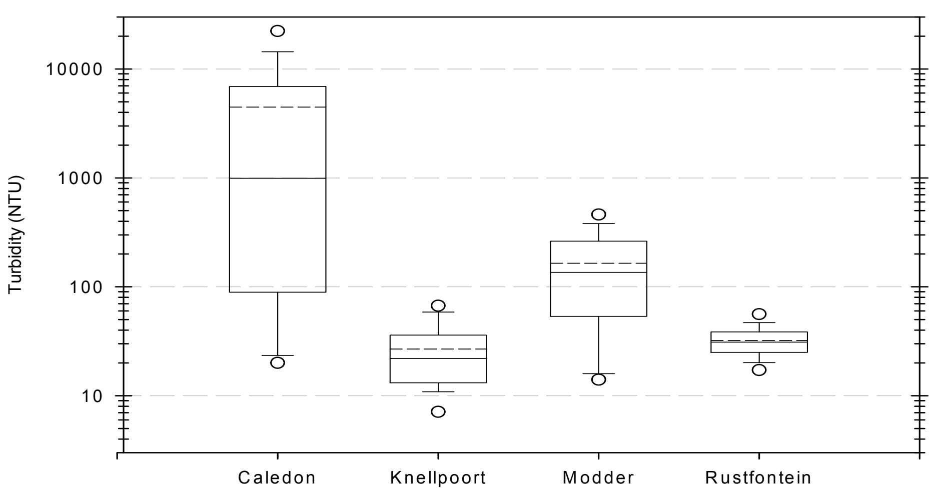


Figure 3.9: Box plot of spatial variation in turbidity in the Caledon and Modder Rivers and the impoundments. The horizontal lines in the boxes mark the median 10th, 25th, 50th 75th and 90th percentile points of the data. The box encompasses the 25th through to the 75th

percentiles. The 5th and 95th percentiles are shown as symbols below (o) and above the 10% and 90% caps, (T) respectively. The broken line represents the average value.

Calculation of the average Secchi depths in the different systems of the *Novo* Transfer Scheme, clearly shows that the Caledon River is more turbid than the Modder River, and that Rustfontein Dam is more turbid than Knellpoort Dam (Figure 3.10). The Secchi depth was also significantly less in the rivers than in the impoundments. In the Modder River, the average Secchi depth was 26 cm, in the Caledon River it was 19.5 cm, in Rustfontein Dam it was 53 cm and in Knellpoort Dam it was 74 cm (Figure 3.10). It is clear from the Secchi disk readings that the light penetration was much lower in the rivers than in the impoundments, due to the presence of higher suspended sediment concentrations in the rivers.

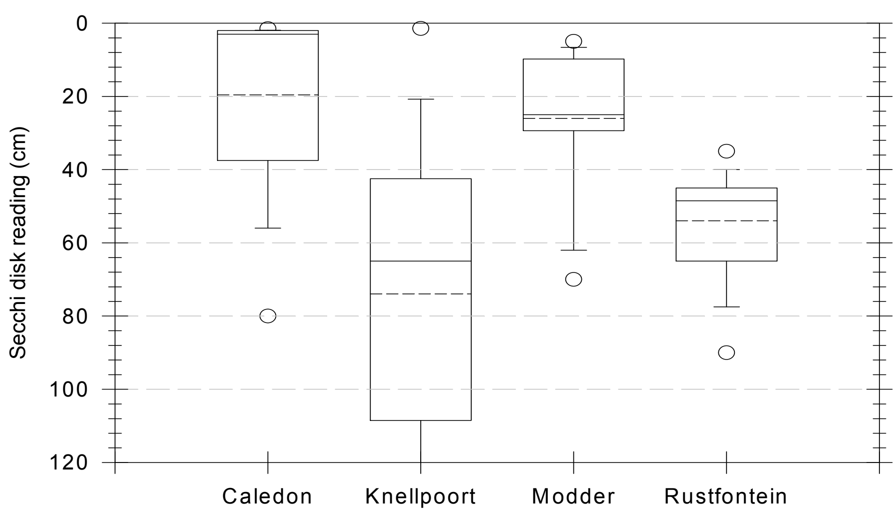


Figure 3.10: Box plot of spatial variation in Secchi depth in the Caledon and Modder Rivers and the impoundments. Please refer to Figure 3.9 for an explanation of the symbols and lines.

3.3.2) The correlation between turbidity and TSS

In view of the fact that increased turbidities are usually caused by increased suspended sediment loads (Lloyd *et al.*, 1987), the relationship between turbidity and TSS were determined from the data. A significant linear relationship was found for both the Modder (r^2 = 0.802) and the Caledon Rivers (r^2 = 0.874) (Figures 3.11 and 3.12).

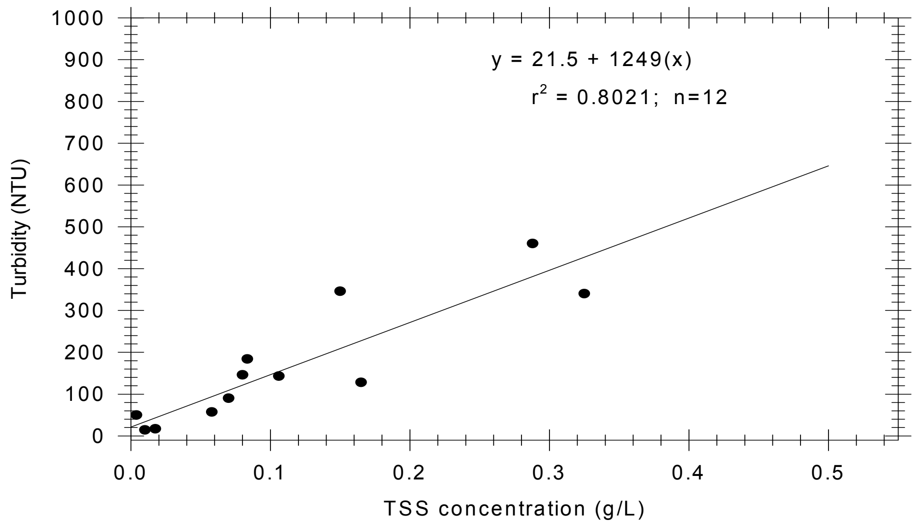


Figure 3.11: Relationship between turbidity and TSS concentration in the Modder River.

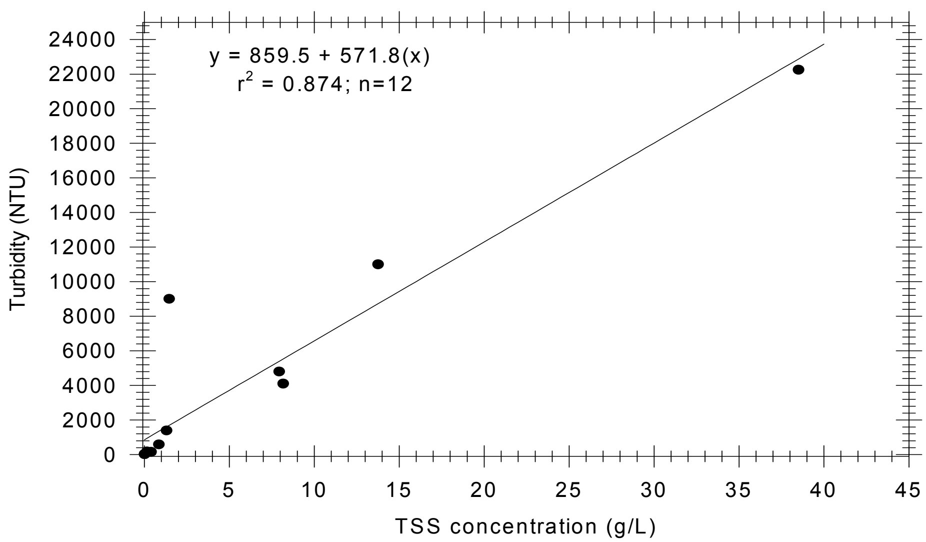


Figure 3.12: Relationship between turbidity and TSS concentration in the Caledon River

Important to notice are the differences in the slopes of the linear regression analyses between the data of the Modder and Caledon Rivers. The slope for the Modder River (1249) was more than double that of the Caledon River (571), indicating that there is a definite difference in the composition of suspended material between the two systems. Note also that the highest TSS in the Caledon River was almost 40 g/L.

3.3.3) Turbidity in Knellpoort Dam

Reservoirs are generally used for water storage along a river course, for off-river water storage, or for inter-basin water transfers. In accordance with this, Knellpoort Dam was built to augment the supply of potable water for Bloemfontein from Welbedacht Dam, the latter which became inadequate due to excessive loss of storage capacity due to siltation. This high siltation rate is due to the high sediment load carried by the Caledon River and it decreased the impoundment's gross storage capacity from the original 115 million m^3 to approximately 30 million m^3 in nine years (Ninham Shand, 1995).

In order to determine the impact of inflowing water from the Caledon River on the water turbidity of Knellpoort Dam, spatial measurements were made and these were plotted three-dimensionally. These measurements were done in November 2000, when water was being transferred and October 2001, after the transfer of water and are shown in Figures 3.13 and 3.14. The measurements were done during similar times of the year and period of 5 months lapsed from the time the transfer was stopped until the measurements were taken in October. Water is released into the dam at the surface, from the side about 120 metres from the dam wall. The transfer water had a turbidity of 3 800 NTU's and it was diluted to about 500 NTU's at the surface. This high turbidity continued for 20 m into the dam, whereafter it decreased steeply and reached values of 200 NTU and lower from 50 m from the inflow. At 100m from the inflow, the turbidity decreased to values of 10 NTU and lower. The turbidity also decreased sharply lateral to the inflow.

In the middle of the water column (Figure 3.13(b)), turbidity increased to 180 NTU at the inflow and stayed the same for about 80 m from the damwall. Thus, the decrease in turbidity in the middle layer of the water column was not as marked as at the surface. The turbidity laterally decreased from the inflow for about 20 m to both sides, and increased again after 40 m.

At the bottom of the impoundment (Figure 3.13(c)), the turbidity was 180 NTU at the inflow and remained constant for 100 m into the impoundment. There was a slight decrease along the main axis of the impoundment upstream from the wall. However, an increase (to about 320 NTU) away from the dam wall occurred.

From the results shown in Figures 3.13(a) – (c) it is clear that the major impact is at the surface, it then sinks to the bottom (Fig. 3.13(b) from where it then spread into the impoundment (Fig. 3.13(c)).

These three graphs show that the impact of the turbid inflow from the Caledon River is very localised. It is however important to note that the area of the impoundment that is affected, is influenced by the wind direction, the turbidity of the inflowing water, the volume that is abstracted, as well as the morphology of the impoundment. The results will probably vary significantly between a windless day and days with strong winds.

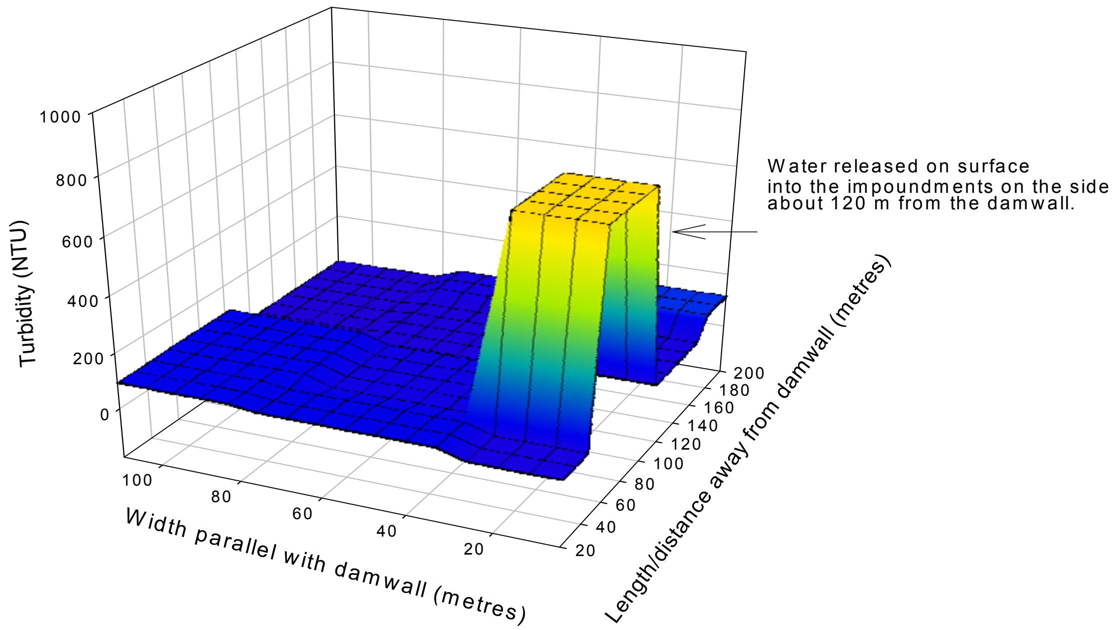


Figure 3.13(a): The distribution of turbidity at the surface of the water column of Knellpoort Dam during the transfer of water.

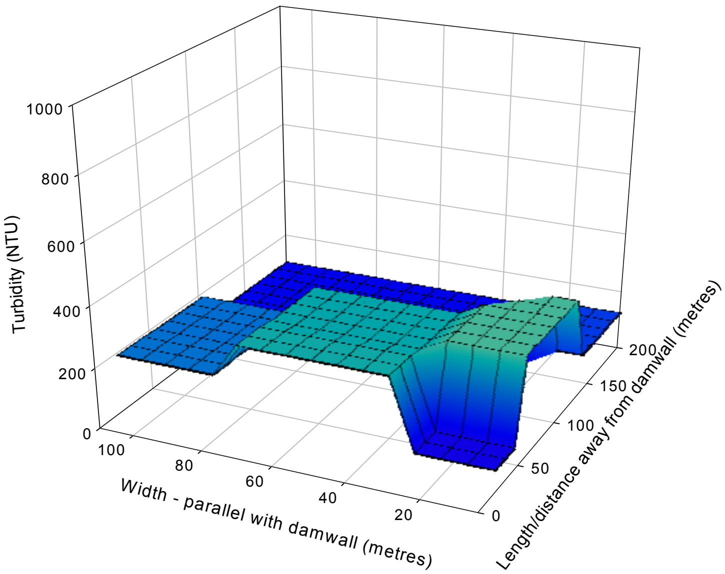


Figure 3.13(b): The distribution of turbidity at the middle of the water column of Knellpoort Dam during the transfer of water.

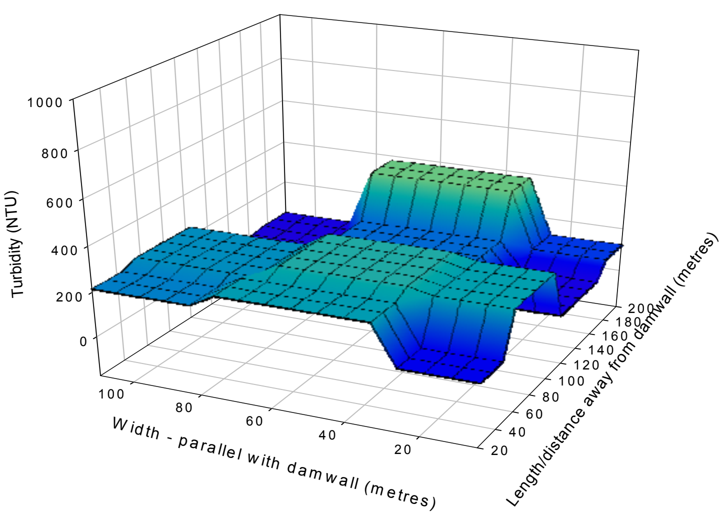


Figure 3.13(c): The distribution of turbidity at the bottom of the water column of Knellpoort Dam during the transfer of water.

Similar measurements were done in October 2001 (5 months after the transfer of water), and the results showed that turbidity varied between 20 and 24 NTU at the surface (Figure 3.14(a)) and no increases were observed along the axis of the impoundment. At the middle of the water column (Figure 3.14(b)), turbidity varied between 10 and 35 NTU, with a slight increase towards the middle of the study area. At the bottom of the water column (Figure 3.14(c), the turbidity was much higher than at the surface and middle reaches of the water column, with a maximum of 118 NTU. From these graphs it is clear that most of the suspended sediments have settled following the transfer of water (note that the Y-axis is about 6 times reduced compared to Figures. 3.13(a) – (c) and that the bottom water are more turbid than the surface waters. Re-suspension could be an important factor as shown by the increased turbidity at the bottom (Figure 3.14(c)), increasing away from the dam wall into the impoundment.

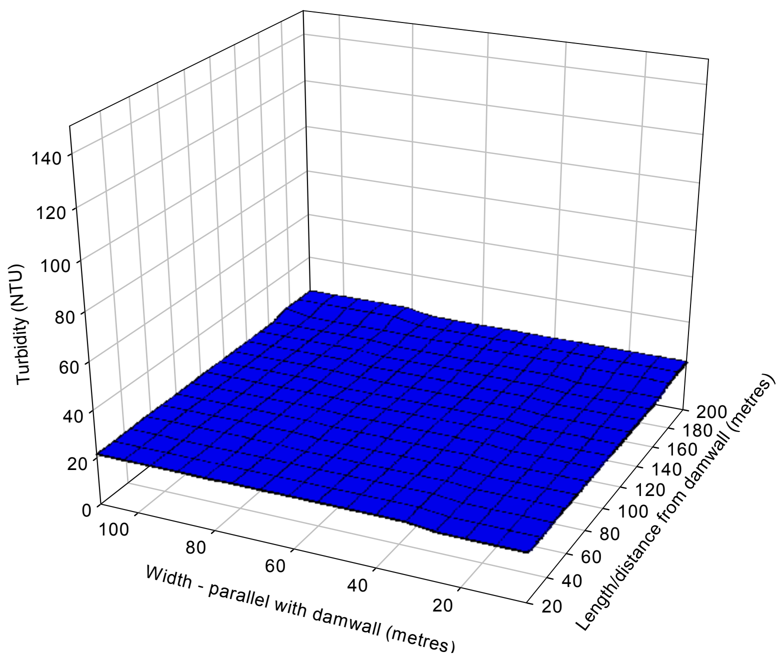


Figure 3.14(a): The distribution of turbidity at the surface of the water column of Knellpoort Dam in the absence of transfer.

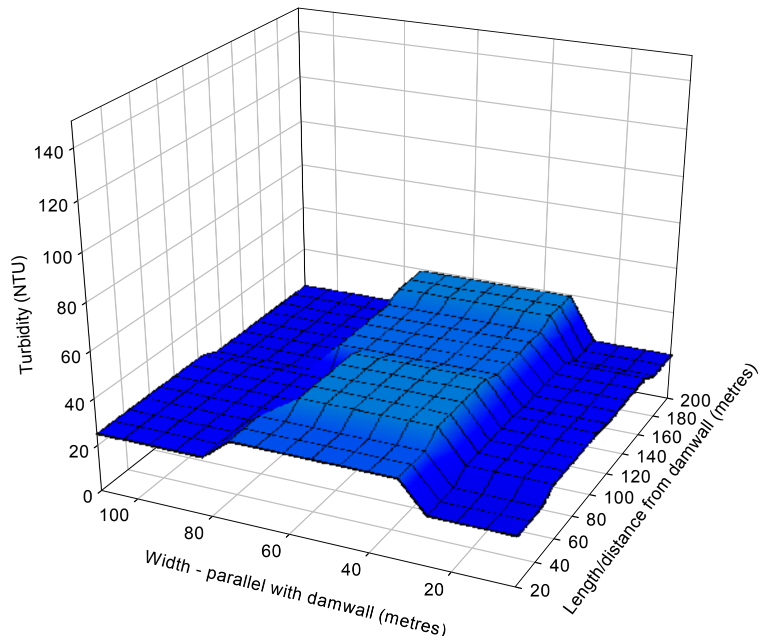


Figure 3.14(b): The distribution of turbidity at the middle of the water column of Knellpoort Dam in the absence of transfer

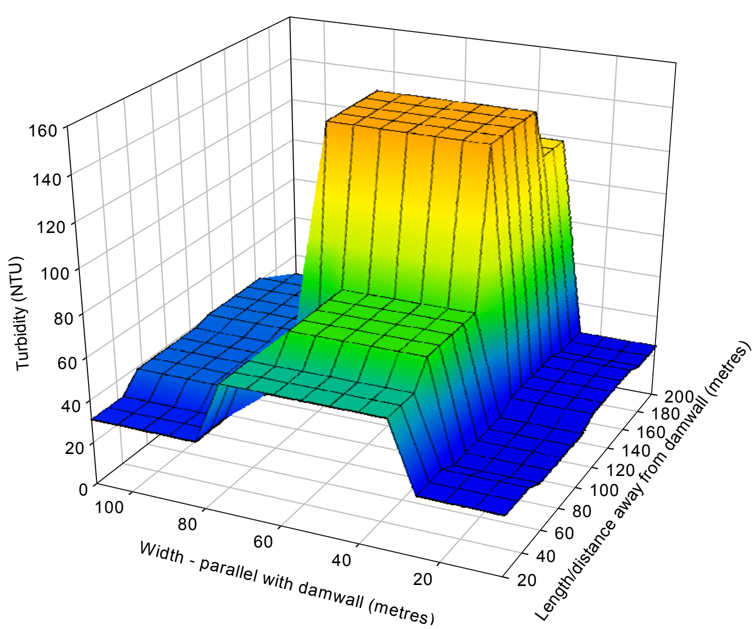


Figure 3.14(c): The distribution of turbidity at the bottom of the water column of Knellpoort Dam in the absence of water transfer

Sedimentation channels were built at Tienfontein pump station before the inlet of the transfer channel, to remove about 50% of the suspended silt (BloemWater, 2001). The efficiency of these channels were roughly determined by measuring the turbidity of the water before entering the sedimentation channels (4 300 NTU), water in the channels (4 200 NTU) and then of the water entering Knellpoort Dam (3 800 NTU). Thus, from the above it seems that about 88% of the silt from the Caledon River remains in the water that is transferred to Knellpoort Dam, while only 12% is removed by the sedimentation channels. The efficiency of the channels to remove silt is, therefore, much lower than expected.

The transfer of water from the Caledon River started in November 2000 and lasted until the beginning of May 2001. Figure 3.15 shows the variation in turbidity at the (monthly) sampling sites in Knellpoort Dam before and after the transfer. The average turbidity increased from 17 NTU to 40 NTU at the surface of the impoundment (an increase of 23 NTU or 135%). At the bottom of the impoundment, turbidity increased from 26 NTU to 53 NTU (an increase of 27 NTU or 102%).

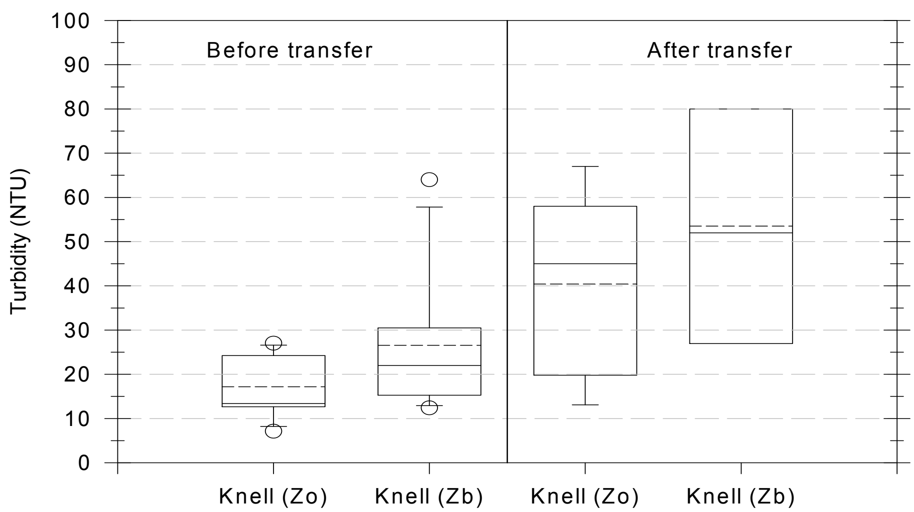


Figure 3.15: The average turbidity in Knellpoort Dam during the study period before and after the transfer of water from the Caledon River (Z_o = surface and Z_b = bottom of the water column). Please refer to Figure 3.9 for an explanation of the symbols and lines.

Accordingly, the TSS concentrations also increased in Knellpoort Dam after the transfer of water from the Caledon River (Figure 3.16). The TSS concentration increased at the surface from 7.5 mg/L to 30.3 mg/L (an increase of 22.8 mg/L or 304%) and at the bottom from 19 mg/L to 35.9 mg/L (an increase of 16.9 g/L or 89%). It is possible that resuspended detritus material could have influenced the measurements taken before the transfer of water occurred. Subtracting the detritus component, this gives a mean TSS concentration at the bottom of 13.7 mg/L, giving an increase of 22.2 mg/L or 162% after the transfer.

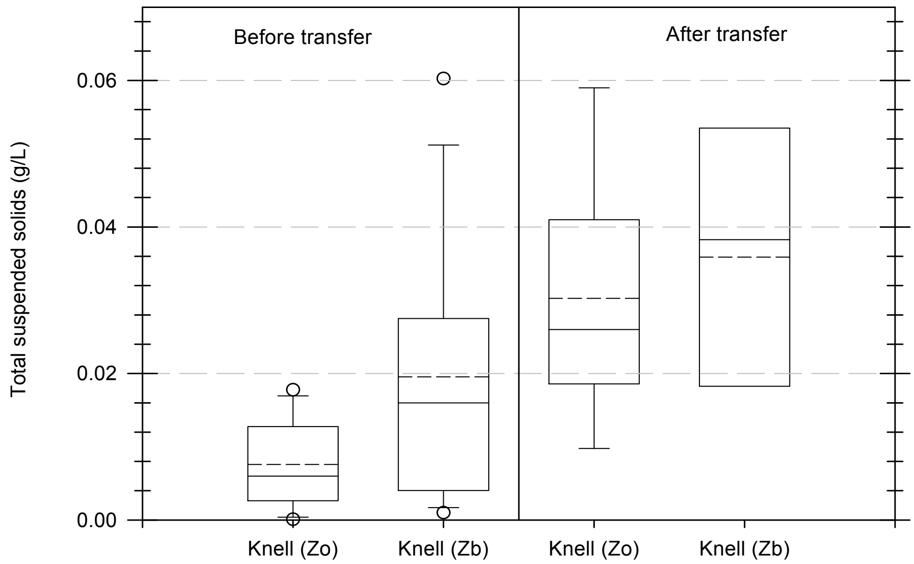


Figure 3.16: The TSS concentration in Knellpoort Dam before and after the transfer of water from the Caledon River. Please refer to Figure 3.9 for an explanation of the symbols and lines.

3.3.4) Underwater light climate

Light profiles that were taken in Knellpoort Dam during the transfer of water (November 2000), and five months after the transfer (October 2001), are shown in Figures 3.18 and 3.19. It is clear that light was attenuated more at the inflow (Site 1) than in the middle of the impoundment (Site 2) (the locations are shown in Figure 3.17) in November 2000 - during the transfer of water (Figure 3.18). The actual attenuation coefficients (k) were calculated from the following formula:

$$k = \frac{\ln Iz_d - \ln Iz_0}{(z_d - z_0)}$$

where Iz_d is the light intensity at a certain depth (n metres) and Iz_0 the light intensity at the surface.

At site 2 in Knellpoort Dam during the transfer of water, 1% of the surface irradiance was reached at a depth of 3.5 m, (k was calculated to be 1.96 m^{-1}), while at site 1 (where the turbid water flowed in), 1% of the surface irradiance was reached between a depth of 0.6 and 0.7m (k

= 7.46 m^{-1}). Thus, light attenuation was almost 4 times higher at sampling site 1, compared to site 2.

Figure 3.17: Satellite image of Knellpoort Dam, showing the approximate location of Sampling site 1 and 2, where measurements were done during both 24 hour studies (taken from Google Earth 2007).

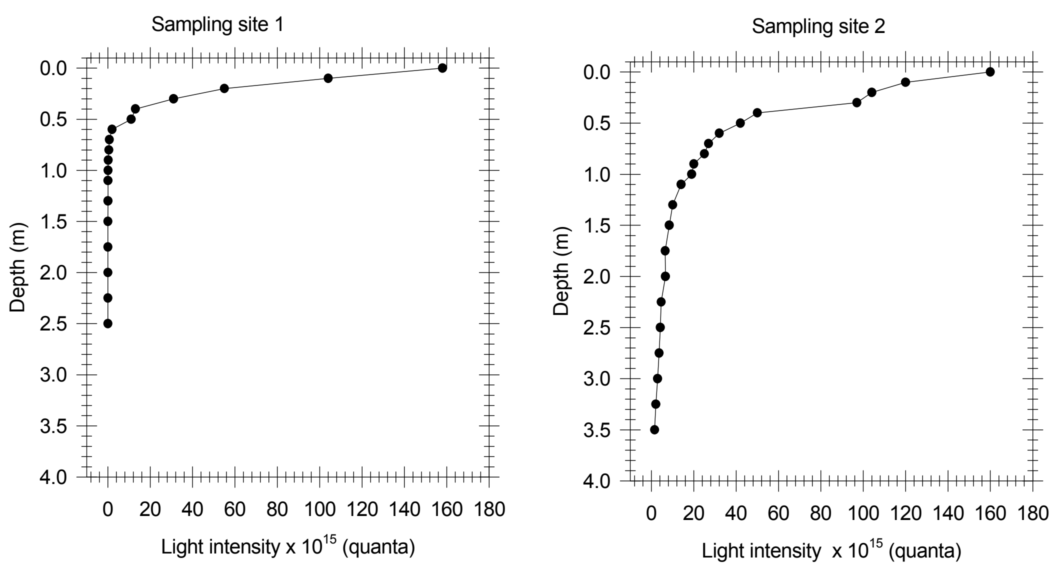


Figure 3.18: Light profiles in Knellpoort Dam in November 2000 when water was transferred. Site 1 was in the turbid area at the inflow of the transfer canal, Site 2 was in the middle of the dam.

In comparison, the light profiles taken in Knellpoort Dam during October 2001, after five months during which no water was transferred, show that there is almost no difference between the attenuation of light at Sites 1 and 2 (Figure 3.19), and the 1% irradiance was reached at a depth of 2.5 m at both sites. The attenuation coefficient (k) during October 2001 at Site 1 was 2.01, and at Site 2, it was 1.99. Considering the above, it can be concluded that the inflow of turbid water had a significant impact on the attenuation of light during transfer, with a marked increase in attenuation at Site 1 during this time.

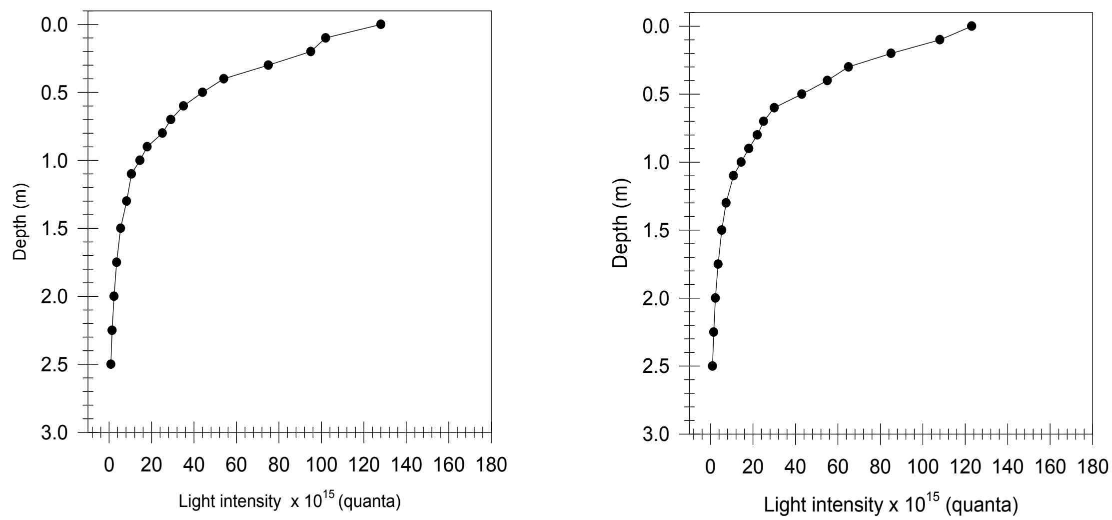


Figure 3.19: Light profiles in Knellpoort Dam as measured in October 2001, following a long period when no water was transferred. Site 1 was at the inflow of the transfer canal, Site 2 was in the middle of the dam. No water was transferred during these measurements.

3.4) DISCUSSION

3.4.1) Seasonal and spatial changes in turbidity in the rivers and impoundments

The flow regime of a river will determine its ability to erode and to transport sediment, and the route taken by water will influence the solute load. Further, the quantity of run-off in any given river catchment is a function of the catchment climate and topographic characteristics. Run-off is the difference between two larger variables, precipitation and evaporation, and is highly sensitive to changes in either variable (Conway & Hulme, 1993).

3.4.1(a) Supply of material to the Caledon River through erosion

For any river, the dominant control on suspended sediment concentration is the supply of material to it (Gustard, 1992). For the Caledon River specifically, soil erosion (whether natural or antrhopological) plays the main role in supplying material for suspended sediment transport.

This is due to the fact that two catchments in central and northern Lesotho form part of the Caledon River basin. Landforms in these catchments are dominated by undulating and rolling dissected plains on Red Bed formations, sandstone escarpments and scree slopes, basalt mountain slopes with or without inflexions and gently undulating pediplains on basalt and sandstones above the 1800 m contour (Qalabane, 1981). As early as the 1970's, the best estimates for Mean Annual run-off (MAR) in Lesotho, ranged from 100-200 mm/year with sediment yields varying from 200t/km^2/y to 3000t/km^2/y (with the highest values in the lowland areas). These high sediment yields are in accordance with Rooseboom (1992) who stated that some of the highest sediment yields in SA are to be found within the Caledon River catchment.

Land use has a significant influence in the erosion processes in Lesotho. The severity of erosion is aggravated by intensive cultivation of the spurs and pediment slopes in the lowlands and by overgrazing on the steep slopes in the mountains and on the scree slopes. An example of the effects of cultivation on erosion and sedimentation can be seen where large areas with fields and villages show higher sediment delivery to the catchment streams than smaller areas with limited croplands (Qalabane, 1981).

Although it must be noted that geological normal rates of soil erosion also operate within natural ecosystems that are in dynamic equilibrium (Beckedahl *et al.*, 1988), it can be accepted that overgrazing and wrong agricultural practices plays the most important role in accelerating the erosion process in Lesotho. In light of the above, a measure of erosion was also expected when water is released at the headwaters of the Modder River (near Dewetsdorp) via the pipeline. To mitigate this potential impact, BloemWater has constructed a channel at the upper reaches of the Modder River, thus increasing the width of the existing, natural river channel and improving the path in which the water will flow. The purpose of this channel is to prevent erosion of the river banks at the outlet and further downstream.

3.4.1(b) Seasonal turbidity patterns

During this study, the seasonal turbidity patterns (Figures 3.3 - 3.4) in the Modder and Caledon Rivers (where turbidity and TSS concentrations increased sharply after rainfall events) is a clear indication of the mobilisation of topsoil during stormwater run-off. This is in accordance with the

Vaal River where the most important factor that influences turbidity and thus the euphotic zone, is discharge (Roos & Pieterse, 1995a). This is also in accordance with Gustard (1992), Meybeck & co-workers (1992) who indicated that a large proportion of the suspended sediment transport occurs during storm events, when rainfall and stormwater run-off mobilise sediments. The seasonal and regional variations in water and sediment supply in many rivers result in a hysteresis loop in the relation between the suspended sediment concentration and discharge. This hysteresis indicates a decrease in the availability of erodable material during continuous discharge. In explanatory terms, during the dry period before a rainfall event, subaereal weathering can produce particulate material that is not transported. This is taken up at the beginning of the rainy season that produces run-off, and is quickly removed during high flow conditions. This was also illustrated in mid-December 1975, when a flood carried a large quantity of silt into the Lindleyspoort Dam (Walmsley, 1978). A marked increase in the turbidity of the bottom waters resulted. This indicated that the flood waters had entered the reservoir as an underflow with most of the allochthonous silt penetrating the bottom waters. Subsequent floods in March and May 1976 did not increase the overall turbidity of the dam, but brought about a clearing of the waters. This led to the assumption that earlier run-off carried away the most easily erodable material in the catchment, a fact which was substantiated by data gained from the inflowing waters (Walmsley, 1978; Eisma, 1993).

The theory of a hysteresis loop is, however, not supported by the seasonal turbidity values in the Modder and Caledon Rivers, since the turbidity values are high throughout the rainy season, and even show an increase later in the season. There is thus no evidence that the total suspended solids are only removed from the Caledon and Modder River catchments during the first rainfall event whereafter it becomes unavailable.

In rivers with strong seasonal flow (such as the Caledon and Modder Rivers) the material that was deposited during declining water levels, is transported again, together with eroded material, during rising water levels. In this way, ephemeral flows can transport large quantities of particulate material that had accumulated during the previous dry period. In semi-arid areas where there is no dense vegetation to protect the soils, this can result in high concentrations of suspended matter (Eisma, 1993). This is the case in the Modder and Caledon catchment areas.

These seasonal patterns of increasing turbidity (thus total suspended solids) after heavy rains and during periods of high discharge in the Caledon River becomes especially important when water is transferred *via* the *Novo* Transfer Scheme (Tienfontein and *Novo* pump stations), since the transfer of the highly turbid waters from the Caledon River has a marked impact on the water of the Knellpoort Dam. These impacts will be discussed in Section 3.4.4 of this chapter.

3.4.1(c) Spatial variation in turbidity

The limited variation between the minimum and maximum turbidity values (and overall lower values) in the impoundments compared to the rivers (Figure 3.9), supports the conclusion that impoundments act as sinks for suspended sediments and that flood and rainfall events do not influence impoundments in the same way as they do lotic systems. In addition, water movement in lakes or impoundments is mainly orbital/ oscillatory, due to the passage of waves, while wind induces major water circulation patterns. This involves low velocity currents that influence transport direction of wave-perturbated sediments (Horne & Goldman, 1994). Thus, linear flow is not an important factor regulating turbidity dynamics in lentic systems, but plays an important role in lotic systems (Thomas & Meybeck, 1992).

Our data in Chapter 6 indicates that the increases in turbidity in the impoundments were mainly due to algal growth. In Knellpoort Dam, however, transfer of water from the Caledon River was also found to cause a significant increase in turbidity (when the transfer scheme was operating). Since the seasonal patterns in productivity and turbidity were not influenced by any transfer of water in Rustfontein Dam, turbidity usually decreased in the periods prior to algal blooms, which created favourable conditions for algal growth. It must be considered that, when the transfer scheme is fully operational, transferred water from Knellpoort Dam *via* the Modder River, could influence turbidity patterns in Rustfontein Dam by decreasing the photic zone, influencing nutrient cycles and feeding patterns of fishes. The level of turbidity in the water that will enter Rustfontein Dam will depend on the flow dynamics of the river, as well as the sedimentation and transportation characteristics at any particular time.

During the study, it was observed on several occasions before the transfer took place, that the water samples taken at the bottom of Knellpoort Dam, contained organic detritus such as pieces of leaves, etc. This could be due to the disturbance of bottom sediments during sampling. It also explains the fact that difference between minimum and maximum values of both turbidity and TSS were less at the bottom of Knellpoort Dam than at the surface.

3.4.2) Secchi depth

According to two reports cited in Hutchinson (1957), the Secchi disk disappears at depths corresponding to 5-15% of solar radiation, which have been confirmed by many studies.

The average Secchi depth in the Caledon River was 7 cm shallower than in the Modder River (Figure 3.10). This can be ascribed to the fact that there is a hyperbolic relationship between Secchi disk depth and turbidity (the higher the turbidity, the lower the Secchi depth). Plots of Secchi disk transparency *vs*. both chlorophyll *a* concentration and turbidity of various surface waters showed that there was no relationship between Secchi disk transparency and chlorophyll *a*, whereas a hyperbolic relationship existed between Secchi disk transparency and turbidity (Lloyd *et al.*, 1987). An inverse relationship ($r^2 = 0.42$) was found between turbidity and Secchi depth in the Modder River (Figure 3.5). For the Caledon River, a negative exponential relationship (not linear) was found between turbidity and Secchi. This can be due to the fact that Secchi disk depth readings become more unreliable as the turbidity increased.

It is only in clear systems (with no or very little suspended inorganic materials), that Secchi disk readings are dependent on the phytoplankton present and can be used as an indicator of the algal biomass. It is thus apparent that the use of the Secchi disk to estimate algal biomass for the purposes of management and trophic classification is inappropriate for waters where even moderate quantities of non-algal turbidity are present (Lind, 1986), such as in the aquatic systems in the *Novo* Transfer Scheme. The relative significance of organic colour on transparency readings also depends on turbidity. Every 100 PtU of colour increases the value of 1/Secchi Disk (SD) by about 0.4. The implication of this is that the relative effect of organic colour on Secchi disk transparency is greatest in situations of low turbidity (Brezonik, 1978). Based on the above, it is clear that Secchi depths in this study were only an indication of the inorganic turbidity, as the river systems were turbid. The organic colour of the water was also not taken into account when measuring the Secchi depths. This was because suspended sediments dominated the colour of the water and no colouring of the water was present after filtration.

3.4.3) Relationship between turbidity and TSS

A linear relationship was found between turbidity and TSS in both the Caledon and Modder Rivers. This is to be expected and is in accordance with Lloyd *et al.* (1987) who stated that elevated turbidities are usually caused by suspended sediments.

The relationships between turbidity and suspended sediments are useful only for specific catchments due to particular sediment characteristics. Because of this, relationships between turbidity and suspended solids have limited value, for the following reasons (Lloyd, 1987):

a) The quantity of turbidity produced per unit of suspended sediments depends on particle size, shape (or angularity) and refractive index. Furthermore, these relationships will change with changes in hydrologic or hydraulic conditions as well as with geological composition.
b) Measurements of turbidity may include some settleable solids. It is sometimes difficult to distinguish between impacts by suspended material/settleable materials.

c) The relationship between turbidity and suspended solids concentration (SSC) may change along a downstream gradient from a sediment source. Larger particles, which generally produce less turbidity per unit concentration than smaller particles, gradually settle out, thus shifting the turbidity vs. SSC to a higher NTU per unit SSC in reaches downstream.

d) Depending on geomorphic, hydrologic and hydraulic factors, different streams are able to accommodate different levels of sediment input and may naturally support different biotic communities.

From Table 3.2 it can be seen that the largest percentage of particles in the Caledon River consist of fine and very fine sand, while a minimal percentage is coarse particles, silt or clay.

Table 3.2: The percentage fraction of different sizes of particles in the Caledon River.

% Clay	% Fine Silt	% Coarse Silt	% Coarse Sand	% Medium Sand	% Fine Sand	% Very Fine Sand
5	2	1	0.66	1.18	37.16	53.34

Unfortunately, the size fraction of the Modder River was not determined during this study. The difference in slopes between Figures 3.11 and 3.12 indicates that there is a significant difference between the sediment composition of these two rivers. This is probably due to the fact that the geological compositions of the catchment sediments differ. In addition, the main source of sediment to the Caledon River is erosion of the sandstones which explains the high percentages of fine and very fine sand.

3.4.4) Supply of suspended material to Knellpoort Dam through the transfer of water

3.4.4(a) The turbidity profile in Knellpoort Dam

Although the sediment input to lakes and reservoirs can be from river input, shoreline erosion, lake bed erosion, airborne input, autochthonous organic matter and autochthonous inorganic precipitates (Thomas & Meybeck, 1992), the sediment input in the case of Knellpoort Dam, is mainly due to the transfer of water. The transfer of water from the Caledon River to Knellpoort Dam causes the transfer of silt to a relatively non-turbid system and is, according to Walmsley (1978), a case of influx of highly turbid waters into an impoundment which decreases the depth of the photic zone.

In lentic waters, the different particle sizes are separated by hydraulic transport in a similar manner to that in rivers (Thomas & Meybeck, 1992), which implies that coarse sediment is deposited first at the river mouth. In the case of Knellpoort Dam, deposition will take place at the inflow channel from the Caledon River. However, when large quantities of sediments flow into an impoundment within a short time, a turbid bottom flow can develop that goes down the slope from the origin of the influx. It can continue over the bottom as a turbidity current, and concentrations can be in the order of 10^2 - 10^3 mg/L in such flows. This has been observed in reservoirs and large lakes (Eisma, 1993) and was also found in Knellpoort Dam, where the most turbid conditions were present at the bottom of the impoundment during the transfer as shown in the three-dimensional graphs (Figure 3.13 (a) – (c)).

The dispersal of sediments by a turbidity current is greatly influenced by the underwater topography and usually the turbid inflow is markedly separated from the clear water by a sharp front (Eisma, 1993). This was also confirmed during site visits at Knellpoort Dam on windless days, where the turbid water did not migrate far into the impoundment in the surface and middle water column, and a clear front could be observed, localised at the area of the inflow.

In shallow lakes located in equatorial regions, it was found that diurnal heating and nocturnal cooling episodes were sufficient to cause localised wind patterns, resulting in the mixing of the entire water column. In the case of very deep lakes, wind-mixing is insufficient to cause mixing of the entire water column, but often resuspends phytoplankton that settled out of the euphotic zone. Wind-induced mixing of all or part of the water column is important in Southern Hemisphere reservoirs such as Knellpoort Dam, due to their large surface areas and relatively shallow depth (maximum depth of 30 - 50 m). This is because the mixing of the water column could result in the release of bio-available nutrients from the sediment.

Apart from mixing, fine sediments in lakes are also transported in suspension by lake currents caused by winds. During extended periods of calm weather, suspended sediments will settle, even in shallow water (Thomas & Meybeck, 1992). The settling of very fine particles is influenced by the thermocline as the settling velocity is reduced where the viscosity of the water increases with depth (Eisma, 1993). An increase in wind leads to resuspension and the particles then continue to be transported. This intermittent transport occurs until the sediment is deposited in an area where the water movements are insufficient to resuspend or remobilise it (Thomas & Meybeck, 1992). Thus, the increase in turbidity in the water column, is only temporarily since the sediment particles will settle eventually. In view of the above, the direction and strength of the prevailing winds that blows during the transfer of water from the Caledon River, will be important in determining the distribution patterns of turbidity in the impoundment.

Comparing the three-dimensional distribution of turbidity during and after the transfer of water from the Caledon River, it is clear that water turbidity is significantly influenced by the transfer. This is however localised at the bottom of the impoundment, where the turbidity levels were much higher than at the surface and middle sections of the water column. As can be seen from Figure 3.14(a) – (c), suspended sediments settled out after transfer was stopped, causing the decrease in turbidity at the surface and middle of the water column, compared with the measurements done during November 2000 (during the transfer) (Figure 3.13(a) – (c)).

The increase in turbidity did not extend to the far side of Knellpoort Dam, where the *Novo* pump station is located, and is thus not expected to have an impact on the water to be transferred to the Modder River.

In comparison, a turbidity profile in Lindleyspoort Dam in November 1975 showed that turbidity stratification was associated with the thermal layering present and was brought about by the sedimentation of particles from the epilimnion. Such a profile during summer appears to be a common occurrence in reservoirs with silty waters (Walmsley, 1978). Although temporary thermal stratification was noticed in Knellpoort Dam, no stable turbidity stratification occurred and the highest turbidy was measured at the bottom of the impoundment. The highest increase in turbidity in Knellpoort Dam was observed at the surface, during the transfer of water, but it decreased considerably after 20 m from the source. At the bottom of the impoundment, the increase in turbidity was not as high, but it stayed the same throughout the width of the impoundment.

By implication, siltation could become a problem in Knellpoort Dam, if water is transferred from the Caledon River over a long period of time. The Welbedacht Dam, South Africa, situated in the Caledon River, already lost 80% of its storage capacity, due to siltation. Furthermore, it has been found that because of the high erosion rates, most of the reservoirs in Lesotho are filled with sediments within 10 years and all are filled within 30 years. On average, reservoir capacity losses in Lesotho are about 4-20% per year (incrementally) (Qalabane, 1981). The siltation of dams has major negative economic impacts as well as far-reaching social impacts in terms of available potable water supplies. Dian and Changxing (2001) also experienced this problem in the lower Yellow River in China where the overloaded flow of the river can quickly fill the reservoirs and increasingly raise the riverbed, attenuating the capacity of reservoirs to suppress floods and provide more water for the dry seasons and for river channels to convey floods. Also, the high sediment content pollutes the water and reduces the volume of usable water.

Knellpoort Dam has a gross storage capacity of 137 million m^3. If water is transferred permanently to the Knellpoort Dam in high flow periods with the transferred water having a total suspended load of on average 2000 mg/L, Knellpoort Dam will lose 50% of its capacity within approximately 20-25 years, which is cause for concern.

The seasonal fluctuations in turbidity in the Caledon River is especially important in the *Novo* Transfer Scheme, since the time of year will play an important role in the quantity of suspended solids that is transferred to Knellpoort Dam. To mitigate the possibility of capacity losses due to siltation, associated with the transfer of water, it is recommended that the pumping schedule at Tienfontein pump station be adapted so that water is transferred when turbidity values is not at its maximum. However, water can unfortunately only be abstracted when the flow in the Caledon River is 18 m^3/s or more (BloemWater, 2001), and this is usually during the rainy season when high turbidities occur. If possible, water should be abstracted at the end of the rainy season, when flow is still high, but when the turbidity is lower and in so doing prevent the transfer of very turbid water to the relatively clear impoundment.

Further possible measures to mitigate this impact, include the installation of filters or dosing the water with chemicals to cause the suspended sediments to flocculate within the sedimentation channels. The implementation of these measures will, however, cause an increase in the cost of water, and it might not be economically feasible. It is recommended that these options be thoroughly investigated for future purposes. These investigations should be done in conjunction with a cost/benefit analysis taking the possible loss of capacity of Knellpoort Dam due to turbid water transfer into account.

Some sedimentation management measures, proposed by Dian and Changxing (2001) include dam construction, storing silts on flood lands, and the use of the silt as fertilizer. However, none of these are considered to be suitable for the Caledon River at this stage.

3.4.4(b) The effect of an increase in turbidity on faunal components of a water body

An increase in turbidity can have serious consequences for a water body. Even low turbidities, near 10-25 NTU, and suspended sediment concentration near 35 mg/L or higher can have a negative impact on fish, such as loss or reduction of sight-feeding capabilities, reduced growth, increased stress, and interference with environmental cues necessary for orientation in migration (Lloyd, 1987). The European Inland Fisheries Advisory Commission (EIFAC) concluded that waters containing 0-25 mg/L chemically inert solids should not adversely affect freshwater fisheries, but that a suspended sediment concentration of 25-80 mg/L may lower the production of fish, while waters containing an SSC above 80 mg/L are unlikely to support good fisheries. A turbidity level of as low as 10 NTU can already cause declines in the feeding rate, food assimilation and reproductive potential of *Daphnia pulex*, which is an important food-source for fish (Lloyd, 1987).

In the Vanderkloof Dam, South Africa, high suspended loads influenced fish by causing a reduced growth rate, a decrease in size at first maturity and maximum size, and a movement inshore by large fishes to feed on the phytobenthos (Bruton, 1988). High turbidities in Lake Chilwa, Malawi, sharply reduced food availability in benthic offshore zones, and restrict fishes to pelagic and inshore food resources. The resuspension of sediments by wind action may also cause fish mortalities through deoxygenation of the water column. On the other hand, moderate turbidities appear to be beneficial to fish in estuaries by providing protection from predators in shallow, food-rich areas. Turbidity gradients may also provide a navigational aid to fish entering estuaries (Bruton, 1988).

To summarise, an increase in turbidity could have the following effects on fish:

1. Reduction in light penetration and of photosynthesis in micro- and macrophytes, resulting in reduced food availability and plant biomass,
2. Reduced visibility of pelagic food,
3. Reduced availability of benthic food due to smothering,
4. Clogging of gillrakers and gill filaments,
5. Reduced feeding efficiency, growth rates population size, egg and larval survival,
6. Reduced aerial predation risks.

Since Knellpoort Dam is being developed as a resort/recreational facility, an increase in turbidity can not only have a significant impact on fish feeding patterns if fish is introduced into the impoundment for recreational purposes, but also on the recreational value of the impoundment. The South African Water Quality Guidelines (DWAF, 1996b) state that waters that have a Secchi depth reading of less then one (1) metre, are unsuitable for swimming. However, if lack of clarity is the only consideration preventing the use of a water body for swimming, then it may be allowed, provided that all subsurface, potential hazards are removed and signs indicating water depth are clearly posted. Again, all these measures will have economic implications. In addition, the risk of disease transmission by organisms associated with particulate matter also increases, although this can not solely be determined on the basis of clarity measurements. It is important to note from the above that even a small increase in turbidity, can have significant consequences for a water body.

It is not only fish and the recreational value of an impoundment that could be influenced by a significant increase in turbidity. Invertebrates are also very susceptible to changes in turbidity. Some of the specific impacts (both positive and negative) that can occur are listed in Table 3.3.

Table 3.3: Effects of turbidity on aquatic invertebrates (Wilber, 1983)

Type of invertebrate	Effect of turbidity
Protozoa	Some species seem to be harmed by an increase in turbidity. Others such as amoebas are unaffected.
Porifera	High turbidity is harmful to sponges. Most require clean, clear water.
Coelenterata	Some forms tolerant against an increase in turbidity. The process of sedimentation has been proposed as the key adverse factor.
Ctenophora	Higher turbidity causes mechanical injury to various tissues.
Polychaeta	Low turbidity may stimulate certain biological processes. Higher turbidity may be harmful and even fatal.
Mollusca	Some species show a change of filtration rate as turbidity increases. Excessive turbidity is harmful.
Echinodermata	Crinoids seem to benefit from turbid water.
Chordata	A TSS concentration of up to 25 mg/L seems to be safe. Most of the adverse effects caused by turbidity are probably mechanical.

Other associated impacts of high sediment load include adsorption or release of P, N, heavy metals and other chemicals, which may affect the chemical properties of a waterbody. The high concentrations of the suspended sediments in the Caledon River, together with the fact that sediments can act as a source of nutrients, gave rise to the question if these sediments act as a source of bio-available nutrients for algal growth. Consequently, growth experiments were carried out with *Selenastrum capricornutum* to determine if any adsorbed nutrients were available from the sediments. The results are reported in Chapter 4.

3.4.5) Effect of an increase in turbidity on the underwater light climate

The zone of a water body where active photosynthesis takes place (the euphotic layer), is of great importance for the structure and functioning of water systems. In waters with substantial quantities of nutrients and where high standing crops of phytoplankton occur, the depth of the euphotic layer is diminished and primary productivity becomes light-limited as a result of self-shading by algal cells (Walmsley, 1978). In the case of waters with high suspended sediments, silt particles would be expected to produce a similar “shading” effect and, therefore, play a key role in governing productivity. However¸such waters are able to tolerate high nutrient loadings before experiencing algal blooms, because of the unfavourable underwater light climate and the adsorption of nutrients (Walmsley, 1978).

Mechanisms which affect the depth of the euphotic zone in impoundments, include: 1) sedimentation and the dilution of turbid reservoir waters by less turbid inflow waters, which increases the depth of the photic zone; 2) the influx of highly turbid waters which decreases the depth of the photic zone; and 3) the process of circulation which tends to increase and maintain the depth of the photic zone (Walmsley, 1978).

The transfer of water from the Caledon River (an example of the influx of highly turbid waters which decreases the depth of the photic zone), decreased the light penetration in Knellpoort Dam and consequently the euphotic depth. This could have a significant effect on the primary productivity of the receiving waters. This is supported by findings in lakes in Alaska, which showed a 1% decrease in light depth between turbidities of 2-10 NTU. These small increases in turbidity greatly reduced the productive volume of the studied lakes. A 5 NTU increase in turbidity of a naturally clear lake can reduce the productive euphotic volume by about 80%. While acknowledging the differences in lake systems between the southern and northern hemisphere, it is interesting that these observations of reduced light penetration with increasing turbidity were also corroborated by other studies in Alaska, where a highly significant relationship (r^2 = 0.96) was reported between light-extinction coefficient and turbidity. A negative relationship between turbidity and light penetration was also quantified for streams in the Birch Creek and Chatanika River drainages of interior Alaska. Based on these studies it was concluded that the quantity of light reaching a depth of 0.1 m, ranges between 75-79% of that available at the surface at turbidities of 0.5-10 NTU, from 60-69% at turbidities of 25-50 NTU, and from 0.3-5% at turbidities of 500-1000 NTU (Lloyd *et al.*, 1987).

Furthermore, studies in North Carolina, USA, showed that each unit (NTU) increase in turbidity caused a 0.06 unit (m^{-1}) increase in light-extinction coefficient (Reed *et al.*, 1983). Strong and similar relationships between turbidity and both 1% light depth and extinction coefficients were also observed in water bodies in South Africa (Lloyd *et al.*, 1987).

The differences in productivity in Knellpoort Dam during and after the transfer of water from the Caledon River due to the influence of turbidity on the light regime, are discussed in Chapter 6.

3.5) CONCLUSIONS

The Caledon and Modder Rivers showed large variations and seasonal patterns in terms of turbidity and TSS concentrations, with an expected increase during the rainy season and a decrease during the dry season. The impoundments in the *Novo* Transfer Scheme did not show the same large variations in turbidity and TSS than the rivers, but this is also to be expected since linear flow does not govern the turbidity dynamics in lentic ecosystems. Due to the different slopes in the relationship between TSS and turbidity in the two rivers, it is clear that there is a marked difference between the sediment composition of the Caledon and Modder Rivers. This clearly pointed to finer suspended materials being present in the Modder River, compared to the Caledon River.

The transfer of water from the turbid Caledon River caused an increase in turbidity in Knellpoort Dam. However, this turbidity decreased when transfer of water was terminated, and conditions returned to those prior to the transfer of the water. This increase in turbidity brought about certain effects, like limitation of underwater light penetration, and a consequent decrease of the euphotic depth. It can also influence other biota in the impoundment such as fish and invertebrates.

The turbidity did, however, not extend to the inflow of the dam where the *Novo* pump station is located and is thus not expected to increase turbidity in the Modder River. In fact, the possibility exists that the clearer water of Knellpoort Dam (at the *Novo* pump station) will decrease turbidity in the Modder River. A decrease of turbidity in the Modder River, together with high nutrient concentrations that could result from urban run-off, can cause extensive algal blooms, which do not exclude the development of nuisance algal blooms. However, the release of water through the pipeline into the Modder River, could result in erosion of the river banks (especially if the water is released in high volumes). This could result in an increase in turbidity. It is recommended that a sampling point be established in the Modder River, both upstream and downstream of the point of water release, in order to monitor changes in the system due to water transfer. Another suggestion to predict the erosion in the Modder River, was made by Hey (1986). He stated that it is possible, given the knowledge of the natural character of the channel, the change in the flow regime of the river when fully regulated and information on threshold discharges for bed material transport and bank erosion, to identify the reaches of channel where regulation is likely to cause erosion or deposition. To predict these changes it is necessary to identify whether or not the maximum regulated flow is in excess of bed material transport and bank erosion thresholds downstream from the proposed outfall. This method could be included in future studies on transfer systems. The transfer of water from the Caledon River, will alter the turbidity patterns in Knellpoort Dam during the transfer of water, and will possibly negatively influence seasonal algal growth. The possibility further exists that the sediments carry adsorbed nutrients with it, which could influence nutrient dynamics in Knellpoort Dam.

CHAPTER 4

THE CHEMICAL AND PHYSICAL CHARACTERISTICS OF THE SEDIMENT LOAD OF THE CALEDON RIVER AND ITS INFLUENCE ON BIO-AVAILABLE NUTRIENTS

4.1) INTRODUCTION

It has long been known that bottom sediments are important sinks and sources of nutrients. Golterman (1977) used sediments as a source of phosphate for algal growth in an experiment. It is also well known that soil particles adsorb nutrients, part of which becomes available for plant growth. Suspended solids contents of more than 700 mg/L in the river water have been reported for the Amazon and large quantities are also carried by the Congo, Mississippi, Niger, Benue, Niger-Benue, Nile, Orange and many other rivers. It is, therefore, possible that nutrients associated with suspended sediment particles could be important in waters carrying solid loads (Grobbelaar, 1983). Furthermore, it was proposed that suspended clay particles could increase the efficiency of dissoved organic carbon (DOC) uptake (Lind *et al*., 1997). Phosphates and metals are the most important compounds adsorbed onto the sediment particles and the particles size affects their sorption properties. Thus, suspended sediments play an important role in the transfer of nutrients and contaminants, in water-sediment interactions and as non-point source pollutants (Vaithiyanathan *et al.*, 1992). Keulder (1976) also confirmed that suspended minerals do certainly play a role in the mechanism of mineral nutrition of planktonic algae. In the case of zinc, the presence of the clay particles made the ions more available, acting as a nutrient source.

During rainfall events, phosphate levels in river systems may be elevated by run-off from the land, as well as by re-suspension and flushing of deposited material from the river bed into the water column. Adsorbed phosphate may be released from the sediments under conditions of high flow and under anoxic conditions from both sediments and water (DWAF, 1996a). In addition, the linear flow regime in rivers has a major influence on the mobility, availability and spatial distribution of phosphorus within a river. During periods of low discharge, stream bed sediments act as a sink for P (Horne & Goldman, 1994).

The presence of clay suspensoids in lakes and impoundments may have a profound effect on the community, not only by light attenuation, but also by restructuring the phosphate dynamics (Heath & Francko, 1988). Phytoplanktonic productivity in phosphorus-limited lake communities could be impacted heavily by siltation, and may become dependent on the sorption-desorption equilibria and the release kinetics of the phosphates (Heath & Francko, 1988). Taking the above into consideration, the suspended sediments in the Caledon River was investigated as a source of nutrients that could be transferred into Knellpoort Dam potentially resulting in high algal biomass.

4.2) MATERIAL AND METHODS

4.2.1) Analyses of suspended sediments

Suspended sediments from the Caledon River was collected from the settling canals at Tienfontein pump station, freeze-dried and thereafter analysed by Glen Agricultural College, Department of Soil Science for fraction size. This was done with a sieve analysis (particle size) of the dry fraction.

Samples from the same batch were sent to Central Analytical Laboratories (Pretoria) that analysed the suspended sediments for selected micro-nutrients. The micro-nutrients were determined through a digestive method.

4.2.2) Growth experiments

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

The algae for the growth experiments were washed three times with distilled water and then grown overnight in distilled water, to deplete the nutrients that were possibly stored in the algal cells due to luxury consumption. Thereafter, it was grown in GBG 11 medium differing in N and P concentrations for a period of two weeks. The N concentration was kept constant when the P concentration was varied, and *vice versa*. The different combinations of nutrient used are shown in Table 4.1. Samples were grown in triplicate and determinations were also done in triplicate. The concentrations of N and P in GBG11 growth medium are 109 mg/L and 49.5 mg/L respectively.

<u>**Table 4.1**: Variations of P and N concentrations in the GBG-11 growth medium as used in the algal assay experiments</u>

No.	[P] (mg/L)	[N] (mg/L)	N:P ratio	No.	[N] (mg/L)	[P] (mg/L)	N:P ratio
1	0.248	109	439	6	2.75	49.5	0.05
2	0.495	109	220	7	5.45	49.5	0.11
3	1.24	109	87.9	8	10.9	49.5	0.22
4	2.48	109	43.9	9	21.8	49.5	0.44
5	4.95	109	22	10	54.5	49.5	1.1

In growth experiments done by Droop (1974), the control of algal growth was shown to follow a threshold rather than multiplicative patterns, that is, non-limiting nutrients exert no control at all over the pattern of growth. The limiting nutrient was the one that showed the smallest cell quota, i.e. the subsistence quota ratio. Droop (1974) found that uptake of both limiting and non-limiting nutrients was controlled by internal as well as external substrate concentrations. Thus, there was a limit to luxury consumption of one nutrient when growth was limited by another. These observations made by Droop (1974) were taken into consideration when the experiments relating the algal biomass to nutrients (assumedly) adsorbed onto the sediments of the Caledon River were conducted.

In view of the above, N was kept constant at 109 mg/L when P was varied, and when N was varied, P was kept constant at 49.5 mg/L. It was, therefore, assumed that the nutrient that was kept constant, did not limit growth, and that the measured growth was related to the "limiting nutrient". The algae were grown in the laboratory for a period of 14 days under optimum growth conditions (in terms of light and temperature), whereafter chlorophyll *a* was used as an indicator of algal biomass.

The chlorophyll *a* concentration was measured using a method, as described by Sartory and Grobbelaar (1984) as described in Chapter 6. The chlorophyll *a* values obtained from the growth mediums with differing N and P concentrations were used to create regression graphs for available nutrient concentration *vs.* algal biomass.

In order to determine the adsorbed fractions of N and P, three different concentrations of suspended sediments (1 000, 2 000 and 5 000 mg/L) were added to GBG11 medium (one without N and one without P) and the chlorophyll *a* values were determined in triplicate for these media after 14 days of growth. Using the regression graphs obtained from the known concentrations of N and P, the quantity of adsorbed N or P of sediments from the Caledon River, was determined.

4.3) RESULTS

4.3.1) Particle size of the suspended sediments in the Caledon River

Considering the importance of particle size in nutrient dynamics, the fraction sizes of the suspended sediments in the Caledon River are shown in Table 3.2 in Chapter 3. Thomas & Meybeck (1992) defined the particle types in terms of their sizes: where clay has a diameter of < 4 μm, silt anything between 4 - 64 μm, sand as particles with a diameter of 64 μm – 2mm, and gravel anything larger than 2 mm. Table 3.2 shows that the different size fractions of the Caledon River suspended sediments are mostly made up of fine sand and very fine sand.

4.3.2) Chemical composition of the suspended sediments in the Caledon River

The chemical compounds found in the suspended solids of the Caledon River are given in Table 4.2.

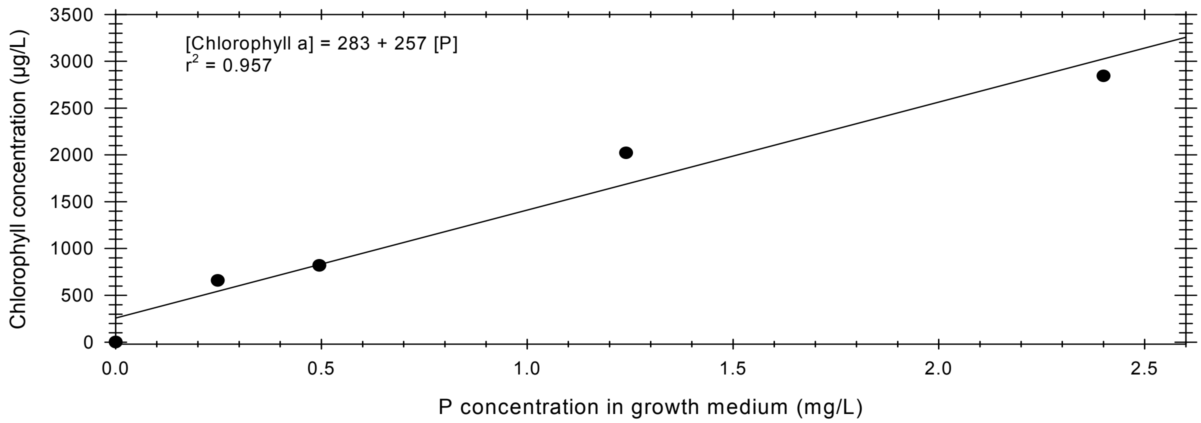
Table 4.2: The different constituents found in the suspended sediments of the Caledon River.

Constituent	Content (mg/kg)
Calcium (Ca)	374
Magnesium (Mg)	122
Potassium (K)	25
Sodium (Na)	21
Zinc (Zn)	0.09

Calcium and magnesium were dominant, while potassium and sodium were also present, although orders of magnitude lower than calcium and magnesium. Zinc was only present in trace concentrations.

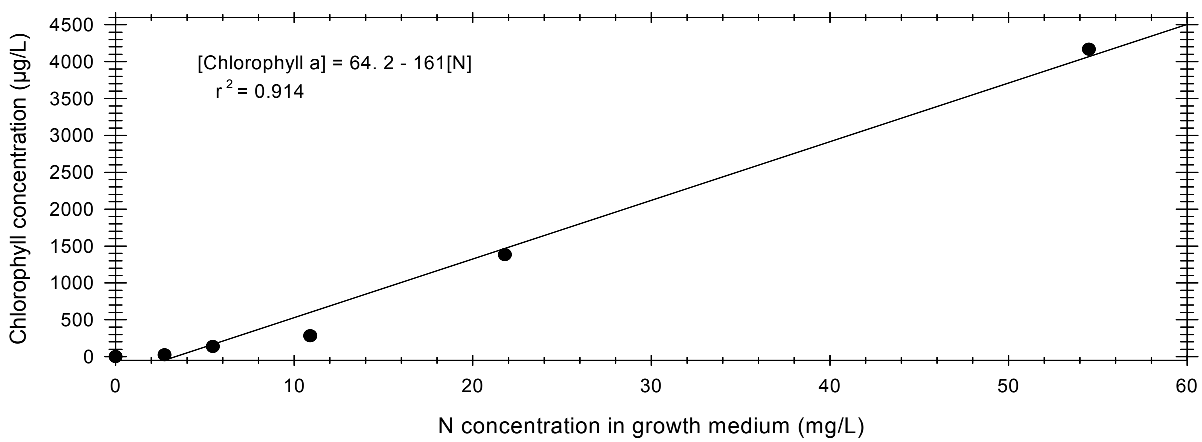
4.3.3) Bio-available nutrients in the suspended sediments of the Caledon River

The relationship between P concentration and chlorophyll *a* content of *Selenatrum capricornutum* is shown in Figure 4.1, where the line of best fit was found to be [Chlorophyll *a* (μg/L)] = 283 + (257)[P mg/L] with a correlation coefficient of $r^2 = 0.957$.



<u>**Figure 4.1**: The linear relationship between P concentration and chlorophyll *a* concentration (biomass).</u>

A similar linear relationship was found between N and chlorophyll *a*, as shown in Figure 4.2. The line of best fit was [Chlorophyll *a* (µg/L)] = 64.2 – 161.05 [N mg/L] with a correlation coefficient of $r^2 = 0.914$.



<u>**Figure 4.2**: The linear relationship between N concentration and chlorophyll *a* concentration (biomass).</u>

From the above relationships it is clear that the chlorophyll *a* content would be a good measure of, the bio-available N and P adsorbed on the sediments. The results of using the suspended sediments as N or P source are shown in Table 4.3.

Table 4.3: Bio-available N and P in Caledon River sediment

Quantity of sediment added (mg) to growth medium without P	[Chlorophyll *a*] (µg/L)	Bio-available P (mg/L)	Quantity of sediment added (mg) to growth medium without N	[Chlorophyll *a*] (µg/L)	Bio-available N (mg/L)
1000	275	0.015	1000	57.32	3.39
2000	504	0.213	2000	57.31	3.39
5000	420	0.14	5000	Below detection	Not measured

From the measurements between sediment concentration and bio-available P it can be seen that the highest yield in terms of sediment concentration was measured at a concentration of 2000 mg/L. This suggests that 1000 mg of sediment is equivalent to 106 µg of bio-available P. Unfortunately the relationship varied considerably and it can be argued that 5000 mg/L sediment was excessive and that this inhibited algal growth due to light limitation and complex formations between the various charged particles. It is known that N is highly mobile and that it does not adsorb readily. The results clearly support this, where minimal concentrations of N were associated with the sediments.

4.5) DISCUSSION

4.4.2) Particle size of the suspended sediments in the Caledon River

The suspended sediment load of a river represents the fine-grained material transported in suspension, with its weight supported by the upward component of fluid turbulence. Particles are commonly less than 0.2 mm in diameter and in most rivers the suspended load will be dominated by clay and silt-sized particles. However, this size differs between rivers, for example the Barwon River, Australia, where the majority of suspended sediments is clay-sized, while in the Middle Yellow River in China up to 60% of the suspended sediment may be composed of sand-sized particles (Gustard, 1992). In the Caledon River, the majority of the suspended load is represented by sand-sized particles (53% very fine sand and 37% fine sand).

An investigation of the particle size of transported material was carried out on the Buffalo River, Eastern Cape Province (SA) and Ciskei. Rivers in this region are generally turbid and carry heavy sediment loads (mainly clay), due both to the nature of the underlying geology and erosion due to poor catchment management. In the Buffalo River, particulate material was categorised into five size classes: 1) ultra fine (<80 μm); 2) very fine (80-250 μm); 3) fine (250-1000 μm); 4) coarse (1000-4000 μm); and 5) ultra coarse (>4000 μm). More than 95% of the dissolved and particulate material transported was ultra-fine (80 μm). It was concluded that too much emphasis has been placed on coarse material (>1mm) in theories of river ecosystem functioning. In the Caledon River most of the sediments are very fine sand (approximately 64 μm), that supports the theory that erosion is an important factor in South African where the suspended load consists mainly of finer particles (< 1mm).

The quantity of bio-available P is generally larger in fine grained sediment than in coarse grained sediments (de Jonge *et al.*, 1993). This is because the specific surface area of particulate matter is a key property which controls the adsorption capacity. Specific surface area is inversely proportional to particle size and decreases over three orders of magnitude from clay-sized particles to sand grains (Thomas & Meybeck, 1992; Webb & Walling, 1992). Therefore, the finest particles are generally the richest in trace elements and consequently, phosphorus sorption increases as particle size decreases. This was confirmed by Meyer (1979) who stated that the surface:volume ratio, acid-extractable Al^{2+}-content and organic matter contents of the sediments are all highly correlated with the P sorption index. Taking into account that the sediments of the Caledon River consist mostly of sand, it can be accepted that the quantity of bio-available P will be lower than that of rivers with silt as the major suspended load.

4.4.2) Chemical analyses of the suspended sediments of the Caledon River

Apart from the quantity of bio-available P, the particle size and mineralogy are also directly related, since individual minerals tend to form within characteristic size ranges. Sediments can thus also be described in terms of discrete compositional fractions, the overall characteristics of which are due to the variation in the properties of these fractions and the consequent changes in particle size (Thomas & Meybeck, 1992).

The analysis of the suspended sediments in the Caledon River showed that calcium was the most dominant mineral (374 mg/kg), with magnesium the second highest (122 mg/kg). The other two minerals that were measured, but in much lower quantities, were potassium (25 mg/kg) and sodium (21 mg/kg). Webb & Walling (1992) determined the typical sediment composition from rivers throughout the world as shown in Table 4.4.

<u>**Table 4.4**: The typical chemical composition of river sediments throughout the world according to Webb & Walling, 1992).</u>

Constituent	Concentration (mg/kg)
Al	90
Si	281
P	1.15
K	21
N	1.2
Mg	11
Ca	24.5
Na	7.1
SO_4	2.15
Cl	0.5
Inorganic carbon	10
Organic carbon	10
Organic nitrogen	1.2
Inorganic phosphate	0.7
Organic phosphate	0.45

From the above, it can be seen that the magnesium (Mg) and calcium (Ca) concentrations for the Caledon River were much higher compared to the world averages for rivers. Ca originates, almost entirely, from the weathering of sedimentary carbonate rocks, while Mg originates from the weathering of particularly Mg-silicate minerals and dolomite sodium which are generally found in association with chloride (Dörgeloh *et al.;* 1993; Allan, 1995). Thus the differences can be ascribed to differences in rock formations and the fact that most of the sediments in the Caledon River results from erosion. In addition, as was seen in Chapter 3, most of the suspended sediment in the Caledon River consists of sandy particles, which also influences the composition of the sediments. The possibility also exists that sewage and industrial effluent from Maseru (the capital of Lesotho) can influence the composition of the sediments being transported in the Caledon River.

4.4.3) Bio-available nutrients in the suspended sediments of the Caledon River

While phosphorus is generally considered to be the primary nutrient limiting algal growth in lakes, limitation of algal growth by nitrogen has been observed in freshwater. It is also commonly observed that the most pronounced phytoplankton responses to enrichment occur when both N and P are added together. Combined N and P enrichment enhanced algal growth much more frequently and more substantially than did addition of N or P singly (Elser *et al*., 1990).

Bonding of substances to sediment occurs as a result of two major groups of processes, namely physical and chemical sorption. The former primarily involves electrostatic attraction and ion exchange, whereas the latter includes exchange interactions with the OH-groups of clay minerals and metal hydroxides, and with the COO- and OH-groups of organic substances, chelation, interactions with Fe/Mn oxyhydrates and co-precipitation (Webb & Walling, 1992). With some sorption processes operating in natural waters, adsorption equilibria may be reached between suspended particulates and fine bed material and certain dissolved components. These equilibria are commonly regulated by temperature, pH and the nature of the clay material. Sorption can be either fast or slow. Fast sorption involves adsorption on an exchangeable ion surface, while slow sorption involves slow diffusion into the particle. The fast step of P sorption is dependent on the surface area and charge balance of the particles, whereas the slow step is very dependent on the composition (Webb & Walling, 1992).

In addition, the adsorption of phosphate can be viewed as the formation of a surface-limited solid solution where the effective surface is influenced by the P concentration. Temperature, pH and ionic strength of the water affect the equilibria and the kinetics of these reactions. Complicating the situation further are the effects of competitive reactions with other ions dissolved in the water or in the solid phase. Also, organic materials may coat the clays and greatly influence both the extent and rate of phosphate adsorption (Heath & Francko, 1988).

The quantity of bio-available P in sediments is usually 10-20% of the total sediment P. It is probable that this fraction consists of adsorbed and metal-associated P fractions (de Jonge *et al.*, 1993). In the Caledon River, the high concentrations of bio-available phosphate (106 mg/kg) could be ascribed to agricultural run-off from Lesotho and Free State farmlands. This is supported by a study in the Odra River (Thomas & Meybeck, 1992), where a maximum concentration of 9.6 mg/kg P was measured. This was ascribed to two reasons:

1) With the raised water level, large quantities of sediments were washed from fields and river banks downstream. In the pore water of muddy sediments high nutrient concentrations can be expected;
2) The flooding of agricultural areas and farms led to the remobilisation of fertilisers and nutrients from the soil and a considerable nutrient contribution was also given by inundated municipal wastewater treatment plants.

The bio-available N fraction from the sediments of the Caledon River was found to be 3.39 mg/L (3 390 mg/kg) for both 1 g and 2 g of added sediments. Since N is highly mobile (Grobbelaar, 1983) it is expected that only fractions will be associated with the sediment loads, in contrast to P which is highly immobile.

It was observed that a larger increase in sediment concentration (up to 5000 mg/L) did not necessarily result in an increase in algal biomass as expected. This was due to extreme turbidity caused by such high concentrations, where light becomes the primary limiting factor and not nutrients.

In Chapter 5 it is shown that the transfer of turbid water from the Caledon River caused an increase in the nutrient concentrations in Knellpoort Dam. Since the average turbidity in Knellpoort Dam is much lower than that of the Caledon River, the increase in bio-available nutrients, together with the favourable light conditions, can have a significant impact on algal growth in the impoundment. An increase in nutrient concentration in an impoundment due to sediment inflow, was also confirmed by Kawabata & Kagawa (1986), who found that by changing the water inlet site in a reservoir as well as the sediment composition had an effect on the nutrient concentration of the water, especially the TP concentration and consequent increase in the phytoplankton.

4.5) CONCLUSIONS

The largest fraction of sediment particles in the Caledon River, consists of very fine sand. This has implications in terms of the bioavialability of nutrients, as it was indicated by various researchers that the surface:volume ratio of a particle plays an important role in P sorption. In addition, the finer particles (< 1 mm) are important in the nutrient dynamics of South African rivers that are prone to erosion.

Due to the bio-availability of nutrients in the sediment load of the Caledon River (maximum bio-available N being 339 mg/kg and P being 106 mg/kg), it is possible that nutrients could increase in Knellpoort Dam due to the transfer of sediments. This could create more favourable conditions for algal growth and subsequently influence the trophic status of the impoundment. This increase in nutrients due to the transfer of water from the Caledon River was in fact observed and is discussed in Chapter 6.

In order to reduce the load of nutrients and other substances in the Caledon River basin it will be necessary to improve wastewater purification technologies both for municipal discharge and industries in the river basin. Furthermore, it will be useful to review and harmonise land uses and landscaping activities on riparian areas and floodplains in the whole river basin in order to decrease the possible nutrient loads due to erosion and farming activities. In addition, measures should be implemented to further reduce the sediment load into Knellpoort Dam by extending the sediment traps in the transfer process.

CHAPTER 5

THE POTENTIAL IMPACT OF THE *NOVO* TRANSFER SCHEME ON THE CHEMICAL QUALITY OF THE VARIOUS WATERS

5.1) INTRODUCTION

5.1.1) The characteristics of rivers and lakes

A lake's morphometry is a function of underwater contour lines, the shape of the lake, and its geologic origin. It is also different due to its structure; for example, deep, steep-sided lakes are quite different in almost all aspects from shallow ones. Once the lake basin is formed, physical, chemical, and biological factors interact to produce discernible structures within the water. These structures persist despite the continual movement of the water that is characteristic of all aquatic ecosystems. The relatively still waters of lakes have led to their definition as lentic environments, while rivers have waters known as lotic environments. The processes in lakes or impoundments are governed by light, temperature, chemical factors, biological zonation, the watershed and the atmosphere (Horne & Goldman, 1994).

Impoundments are water bodies formed or modified by human activity for specific purposes, such as Rustfontein and Knellpoort Dams. In this particular case, Rustfontein Dam was built to supply the city of Bloemfontein with potable water, while Knellpoort Dam was built as an off-river storage reservoir, to augment the water supply to Bloemfontein from Welbedacht Dam. As mentioned in Chapter 2, this was due to the fact that Welbedacht Dam lost a significant quantity of its storage capacity due to siltation.

Impoundments do show many of the same basic hydrodynamic, chemical and biological characteristics as natural lakes. Chemical changes and fluctuations which occur in impoundments are not unique compared with lakes, although their timing and intensity may be unusual. However, the operating regime for the purpose for which the impoundments were created, together with management interventions, may significantly alter their physico-chemical character and biological responses. This is because all impoundments are subject to water quality requirements in relation to a variety of human uses, since the variation in design and operation of control structures in impoundments can provide greater flexibility and potential for human intervention than in natural lakes (Thornton *et al.*, 1992).

Impoundments situated in densely populated or agricultural areas, have a tendency to be highly enriched with nutrients, although some have a more rapid flushing regime than natural lakes which partly masks the enrichment effects. There is, therefore, a potentially high dependency of productive capacity on management regimes, source water qualities, and internal chemical and biological process rates. In addition to these, inflow-outflow velocities and water body morphometry can also affect the within-lake characteristics of the impoundment (Thornton *et al*., 1992).

Rivers, on the other hand, are complex, longitudinal ecosystems and may be viewed as a series of reaches, or sectors, each receiving and discharging water, sediments, organic matter and nutrients (Petts, 1992). A principal ecological process in rivers, distinguishing them from other types of ecosystems, such as lakes and impoundments, is the unidirectional transport of materials, from the headwaters to the outflow. Rivers are also characterised by important storage characteristics. The dissolved output from a catchment to a river system is a function of the inputs, in-system storage and the phase transformations that take place during the residence time of the different elements in the catchment (Edwards, 1974). Input of nutrients into a river nearly always exceeds output (Golterman, 1975) and accumulation in sediments accounts for most of the difference. This accumulation should never be regarded as a loss because there are chemical interactions between sediments and overlying waters that release the nutrients. Because all rivers flow between multiple geographical boundaries, which introduce numerous factors such as altitude, climate, topography, geochemistry, hydrology and catchment land use, all in turn influencing the distribution of species, communities and habitats (Hynes, 1970; Beaumont, 1975; Petts, 1992; Jeffries & Mills, 1990; Davies *et al.,* 1993).

In summary, the character of lentic waters reflect the dominant features of light, heat and resulting stratification, while lotic habitats are best described by flow, erosion deposition and channel form (Jeffries & Mills, 1990).

5.1.2) The chemistry of freshwater (rivers and lakes/impoundments)

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

All natural surface waters contain dissolved and particulate organic and inorganic matter and the quantities can be high (Hynes, 1970). Rivers annually transport 3.9 billion tons of dissolved material to the oceans on a world-wide scale (Holeman, 1968). However, the physical attributes and chemical constituents of natural freshwaters differ from continent to continent, and even from region to region, as the result of differences in climate, geomorphology, geology and soils, aquatic and terrestrial biota (Dallas & Day, 1993; Walmsley & Davies, 1991).

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

According to Kilham (1990), the major mechanism controlling the evolution of specifically African waters during evaporative concentration, is the precipitation of $CaCO_3$. A substantial loss of carbon occurs during every evaporative concentration step and the subsequent precipitation of $CaCO_3$. The alkalinity of these waters is also affected by two additional processes: namely reverse weathering that decreases alkalinity, and the loss of sulphur to either sediments or the atmosphere that increases alkalinity. Thus, although atmospheric precipitation and rock dominance are potentially important mechanisms controlling world water chemistry, atmospheric precipitation plays a lesser role in Africa, and rock dominance is the major mechanism controlling the dilute African waters. It is possible that rock weathering masks the ionic composition of rain or that terrigenous dust and ash from fires, often determine the composition of rain. African rainwater is also rarely dominated by NaCl or any particular salt, because much of the rain falling in the interior of the continent originates from inland waters (Kilham, 1990). However, during the past decades, the importance of anthropological pollution has increased in determining the chemistry of freshwater, altering the natural composition proposed by Kilham (1990) and Gibbs (1970).

According to Golterman (1975), natural water components can be divided into five classes, namely dissolved inorganic ions and compounds, particulate inorganic compounds, dissolved organic compounds, particulate organic materials and dissolved gases. The dissolved inorganic constituents may be divided into major (macro) constituents, minor (micro) constituents, trace elements and gases (Table 5.1) (Hynes, 1970; Golterman, 1975; Horne & Goldman, 1994; Allan, 1995).

Table 5.1: The dissolved inorganic constituents in natural waters (Golterman, 1975)

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

*The constituents listed in brackets are sometimes regarded as major nutrients.
**Fe^{2+} can be regarded as major and minor nutrient depending on the requirements of different algae

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

The concentration of Na^+ is usually lower than both the concentrations of Ca^{2+} and Mg^{2+} (Golterman, 1975; Allan, 1995; Dörgeloh *et al.*, 1993) while K^+ is a common constituent of many minerals and is always present in fresh water (Hynes, 1970). K^+ originates from the weathering of silicate materials, particularly potassium feldspar and mica (Allan, 1995). Mg^{2+} originates from the weathering of rocks, particularly Mg-silicate minerals and dolomite sodium which are generally found in association with chloride. High concentrations of Ca^{2+} can offset the inhibitory effect of high Mg^{2+} concentrations (Bierhuizen & Prepas, 1985). Other inorganic ions or compounds which are present, usually occur at concentration orders of magnitude less than those of the major ions. Important minor nutrients include Mn^{2+}, Co^{2+}, Mo^{2+}, Cu^{2+}, and Zn^{2+} (Horne & Goldman, 1994).

The main sources of elements in rivers are chemical weathering, atmospheric inputs and the leaching of organic soils. In any region not yet affected by human activity, the variability in natural water quality depends on the combination of the occurrence of highly soluble/easily weathered minerals, the distance to the coastline, the precipitation/river run-off ratio and the occurrence of peat bogs, wetlands and marshes (Meybeck *et al.*, 1992).

Furthermore, water quality variability depends on the hydrological regime of a river, i.e. water discharge variability, the number of floods per year and their importance. During flood periods, water quality usually shows marked variation due to the different origins of the surface run-off, sub-surface run-off and groundwater discharge. Surface run-off is generally highly turbid and carries large quantities of total suspended materials, including particulate organic matter. Sub-surface run-off leaches dissolved organic carbon and nutrients from soils, whereas sub-surface waters provide most of the elements resulting from rock weathering (Meybeck *et al.*, 1992).

The reserve of nitrogen occurs in the atmosphere and biological processes are responsible for the movement and variation in ecosystems. Nitrogen (as NO_3) moves readily through most soils and ends up in aquatic ecosystems, while phosphorus, which occurs both as simple ionic orthophosphate (PO_4-P) and as bound phosphate, in soluble and particulate form, is less mobile (Hynes, 1970; Horne & Goldman, 1994 - see also the discussion of chapter 4). The concentration and rate of supply of nitrate is closely connected to land-use practices in the watershed. Nitrate also originates from protein decomposition, ammonia and agricultural practices (Hewitt, 1991). On the other hand, the natural occurrence of phosphorus in soil is known to be quite low and remains fairly stable because the majority occurs as insoluble inorganic phosphates (Hughes & Van Ginkel, 1994).

The reserve of phosphorus occurs in geological sediments and physical and chemical processes are responsible for the variations in nature. Phosphate in a water body originates from human urine, fertilisers, detergents (Hewitt, 1991) and re-suspension of sediments together with the mixing of pore waters within the water column (Hynes, 1970; Hernández-Ayón *et al.,* 1993). The minerals which bind to phosphorus and form insoluble bonds, are Ca^{2+}, Fe^{2+} and Mg^{2+} (Hughes & Van Ginkel, 1994). Phosphorus was found to typically be the nutrient that limits primary productivity in north temperate freshwater lakes and usually controls summer phytoplankton production (Lloyd *et al.*, 1987). Phosphorus is also more readily washed out of sandy, acid or waterlogged soils than it is from clay soils (Walling, 1980); and the quantity of phosphorus in drainage water is controlled by the nature of the soil as well as the quantity of phosphorus applied to the surface. Generally, only a small quantity of soluble soil phosphate is leached out by rain, and an even smaller quantity of the fixed phosphate in the soil is eroded away (Hughes & Van Ginkel, 1994). A quantifiable relationship exists between the quantity of nutrients reaching a water body and its trophic degree (measurable with some kind of trophic scale index) as was shown by Vollenweider (1969a; 1976). He also showed that there exists a clear relationship between the amount of P in a water body and the chlorophyll *a*. Further details on this relationship are given in Chapter 6.

Silicon is also an important nutrient in the solute budget and is of interest in the study of chemical weathering processes (Edwards, 1974). Most algae and animals only have a minor need for silica, but in the diatoms, silica forms the rigid algal cell wall or frustule which may account for half the cell's dry weight (Horne & Goldman, 1994). The form of silicon, when used as a structural component, is hydrated to form amorphous silica ($SiO_2.nH_2O$). This rigid material is highly perforated and surrounded on both sides by a thin cell membrane. Reactive silica is probably the only form available for diatom growth.

CO_2, O_2 and N_2 occur in significant quantities as dissolved gases in river water (Hynes, 1970; Golterman, 1975; Allan, 1995; Horne & Goldman, 1994). Biologically, N_2 gas is the least important of the three. Some cyanobacteria in the aquatic environment also have the ability to fix atmospheric nitrogen. Both CO_2 and O_2 occur in the atmosphere and dissolve in water, depending on the partial pressure, salinity and temperature. Photosynthesis and respiration are two important biological processes that alter the concentration of these two gasses (Hynes, 1970; Golterman, 1975).

From the above, it can be seen that different chemical compounds are present in varying concentrations in different water systems. With this in mind, the impact of the water transfer scheme on the chemical composition of the receiving system, was investigated.

5.2) MATERIAL AND METHODS

For this study, the ionic composition of the water was determined, as well as differences in conductivity, pH and dissolved oxygen concentrations. The nutrients that were considered in the two river systems and the impoundments, were total phosphates (TP), dissolved reactive orto-phosphates (PO_4-P), nitrate-nitrogen (NO_3-N), ammonium-nitrogen (NH_4-N) and silica-silicon (SiO_2-Si). Spatial differences were determined for all these parameters, as well as seasonal patterns and the impact of the transfer of water on these parameters.

In situ measurements of dissolved oxygen, pH, temperature, electrical conductivity and Secchi depth were done at monthly intervals and subsurface samples (2 litres) were taken (except at Knellpoort Dam, where a bottom sample was also collected), kept on ice and brought to the laboratory for chemical analyses. The analyses were done within 48 hours. Prior to analyses the samples were stored, in the dark, at 4°C to limit metabolic alterations.

The concentration of dissolved oxygen and percentage of saturation were measured *in situ* using a YSI Model 50B dissolved oxygen meter (5739 probe). The pH of the water was determined *in situ* with a HANNA HI 9073C Microcomputer pH meter. Electrical conductivity (which serves as an indication of the total dissolved salts in the water), was determined using a HANNA conductivity meter and expressed as mS/m.

The total dissolved solid concentrations (different ions) at Sannaspos in the Modder River and at Jammerdrif in the Caledon River were obtained from the Department of Water Affairs and Forestry, Pretoria. The different ion concentrations in Knellpoort and Rustfontein Dams were determined by the Institute for Groundwater Studies, University of the Free State.

The rainfall data were obtained from the Department of Meteorology, Faculty of Natural and Agricultural Sciences, University of the Free State.

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

Ammonium-nitrogen (NH_4-N) was measured using the phenate method as described in *Standard methods for the examination of water and waste water* (1995), using 25 ml Whatman GF/C filtered water. The method involved measuring the formation of an intensely blue compound, indophenol, which is produced by the reaction of ammonia, hypochlorite and phenol, catalysed by sodium nitroprusside. The developed colour was measured with a Hitachi spectrophotometer at 640 nm.

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

Silica-silicon (SiO_2-Si) (50 ml GF\C filtered water) was measured using a modified method as described in *Standard methods for the examination of water and waste water* (1995) - the silica yellow method. This involved a reaction of 1 ml HCl and 2 ml ammoniummolybdate to give a yellow colour at a pH of approximately 1.2, which was measured with a Hitachi spectrophotometer at 410 nm. The intensity of the yellow colour is proportional to the concentration of "molybdate-reactive" silica.

During November 2000 and October 2001, 24 hour studies were carried out at Knellpoort Dam. During November 2000 water was transferred from the Caledon River to Knellpoort Dam, but during October 2001 no water was transferred (the transfer stopped in May 2001). Thus, two studies were done to compare conditions during and after transfer of turbid Caledon River water into Knellpoort Dam, at approximately the same time of year.

5.3) RESULTS

5.3.1) pH and alkalinity

Figure 5.1 shows the variation in pH at the different sampling sites. The average pH at all the sites were within the same range and varied from a lowest value of 7.95 (Caledon River) to a highest value of 8.3 (the surface of Knellpoort Dam), indicating minimal variations in pH.

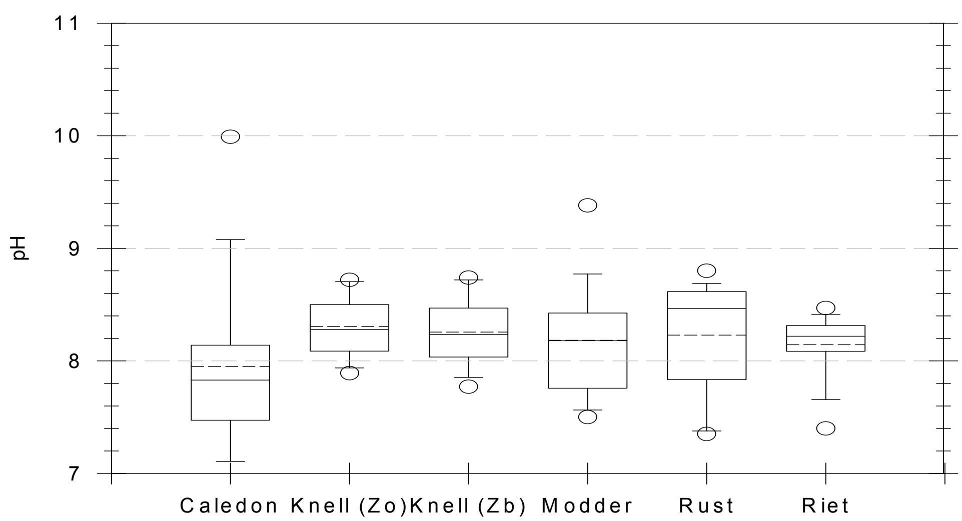


Figure 5.1: Box plot of pH values at the different sampling sites during the study period. Please refer to Figure 3.9 for an explanation for the interpretation of the symbols and lines.

The average alkalinity values for the different systems are given in Table 5.2.

Table 5.2: Average alkalinity values for the different systems in the Novo Transfer scheme

Sampling point	Alkalinity value (mg/L)
Caledon River	97
Knellpoort Dam	124
Rustfontein Dam	129
Modder River	167

5.3.2) Dissolved oxygen concentration

Typical saturation concentrations at 1400 m above sea level (Bloemfontein), and at TDS values below 3000 mg/L are: 10.98 mg/L at 5°C, 8.66 mg/L at 15°C and 7.8 mg/L at 20°C. An average value of 8.23 mg/L (between 15 and 20°C) was used to calibrate the meter to indicate dissolved oxygen saturation, which then corresponded to the average water temperature. Variations in dissolved oxygen concentrations at the different sampling sites are shown in Figure 5.2.

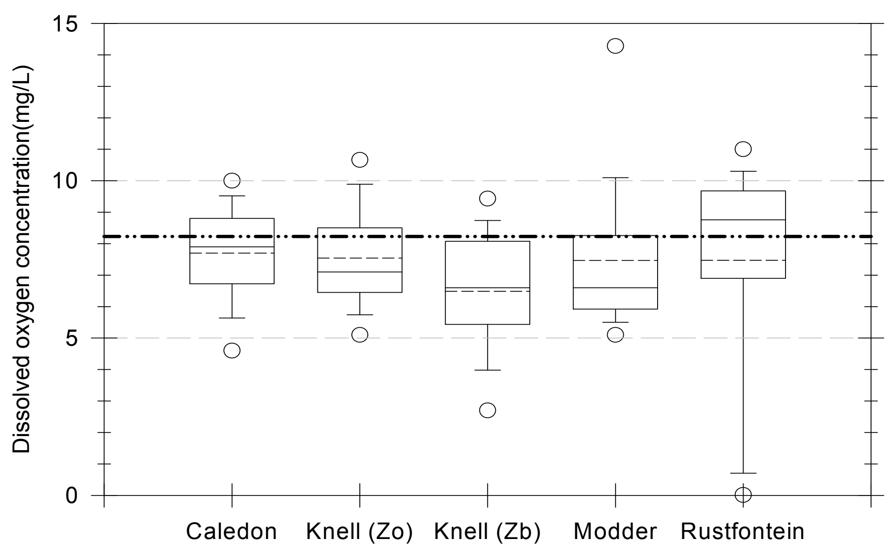


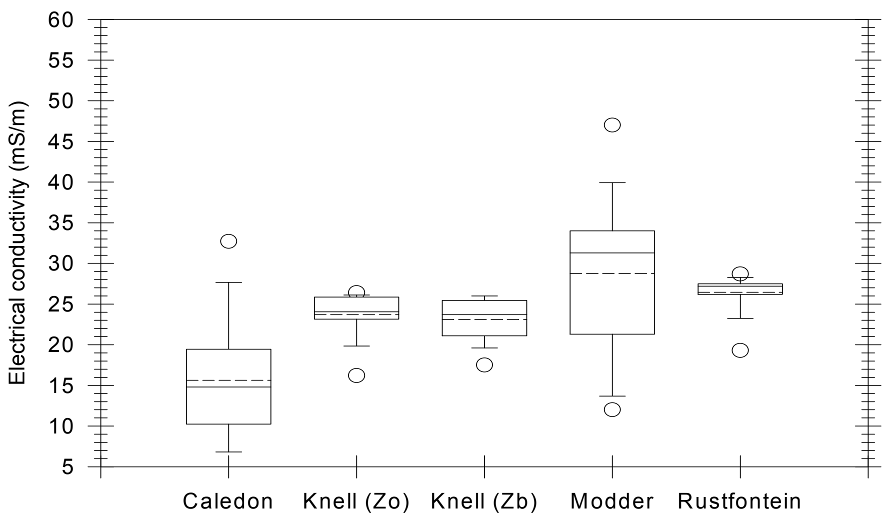
Figure 5.2: The variation in dissolved oxygen concentration (mg/L) at the different sampling sites. The solid and dotted horizontal line represents the saturation value.

As can be seen, the average dissolved oxygen concentrations at all the sites were generally lower than the saturation values, but all within the same range (Figure 5.2). The only sites that showed a significant lower concentration than the other sites, was the bottom waters of Knellpoort Dam and the Modder River.

5.3.3) Electrical conductivity

Electrical conductivity is a measure of the ability of water to conduct an electrical current. This ability is a result of the presence in water of ions such as carbonate, bicarbonate, chloride, sulphate, nitrate, sodium, potassium, calcium and magnesium, all of which carry an electrical charge. Many organic compounds dissolved in water do not dissociate into ions, and consequently, they do not affect conductivity (DWAF, 1996a).

Figure 5.3 shows the average conductivity in the different waters of the *Novo* Transfer Scheme. The conductivity was the highest in the Modder River (28.7 mS/m), with the Caledon River showing the lowest average conductivity (15.6 mS/m). Comparing the two impoundments, it can be seen that the average conductivity in Rustfontein Dam (26.4 mS/m) was slightly higher than that of Knellpoort Dam (23.3mS/m). The results also clearly show much higher variation in the rivers, compared to the dam sites, where the least variation occurred in Rustfontein Dam.



<u>**Figure 5.3**: The variation in conductivity in the different waters of the *Novo* Transfer Scheme.</u>

The variation in conductivity in the Modder and Caledon Rivers over the study period is shown in Figure 5.4. Conductivity followed exactly the same pattern in both the rivers, with an increase during the winter (May to August) and a decrease following the first rains of the rainy season in September.

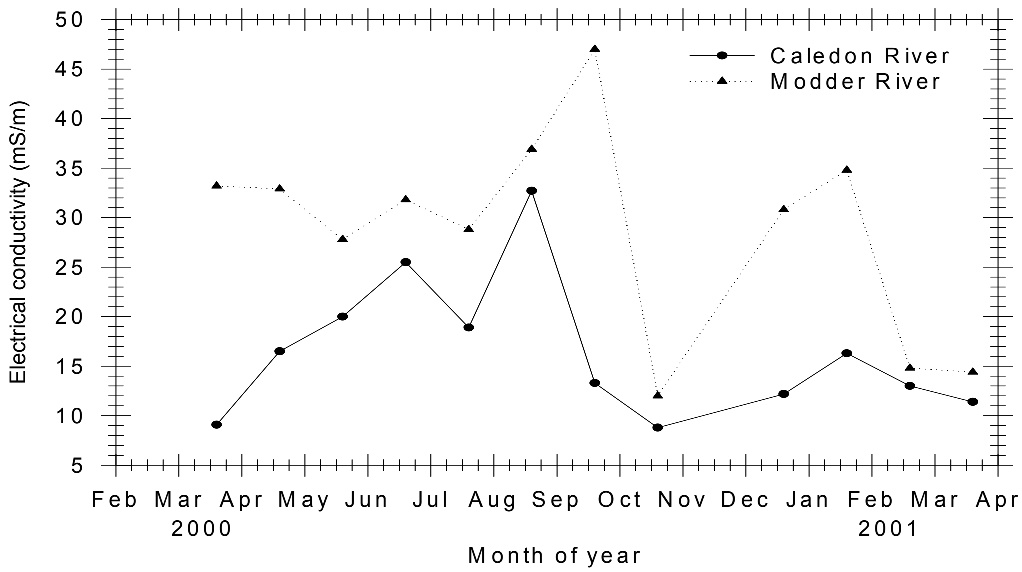


Figure 5.4: Seasonal variation of conductivity in the Caledon and Modder Rivers

As can be seen, the electrical conductivity was always higher in the Modder River than in the Caledon River (Figure 5.4). In fact, the average conductivity was 57% higher in the Modder River (28.7 mS/m or 187 mg/L TDS) for the study period compared to the Caledon River (16.4 mS/m or 106 mg/L TDS).

Since the conductivity in the Caledon River was lower than in Knellpoort Dam, the possibility exists that the transferred water could affect the composition of the Knellpoort Dam water. Figure 5.5 shows the average variation in conductivity in Knellpoort Dam, before and after the transfer of water. There was no statistical significant decrease in conductivity levels in Knellpoort Dam after the transfer of water. At the surface, the average conductivity decreased from 23.8 mS/m, to 23.5 mS/m. At the bottom of the dam, the average conductivity showed almost no variation from 23 mS/m to 23.2 mS/m (Figure 5.5). Noticeable, however, is the fact that the variance was much less following the transfer, compared to before the transfer.

During the study period, the average conductivity in Knellpoort Dam was 23.4 mS/m and in Rustfontein Dam it was 26.46 mS/m. No seasonal patterns were evident in the two impoundments (Figure 5.6). The results clearly show the stability of water in impoundments compared to the lotic systems investigated.

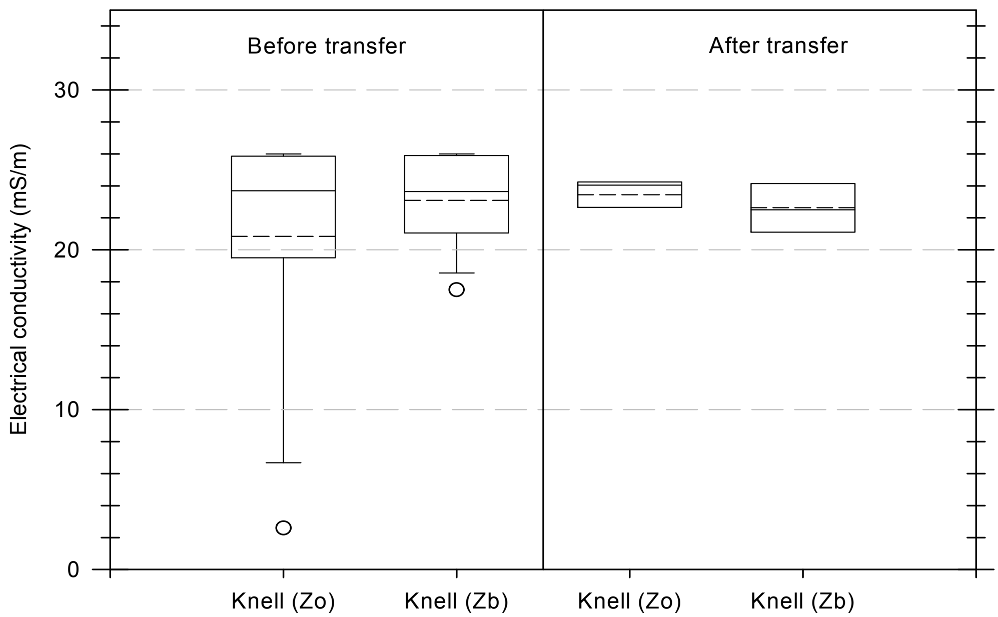


Figure 5.5: The variation in conductivity in Knellpoort Dam before and after the transfer of water from the Caledon River.

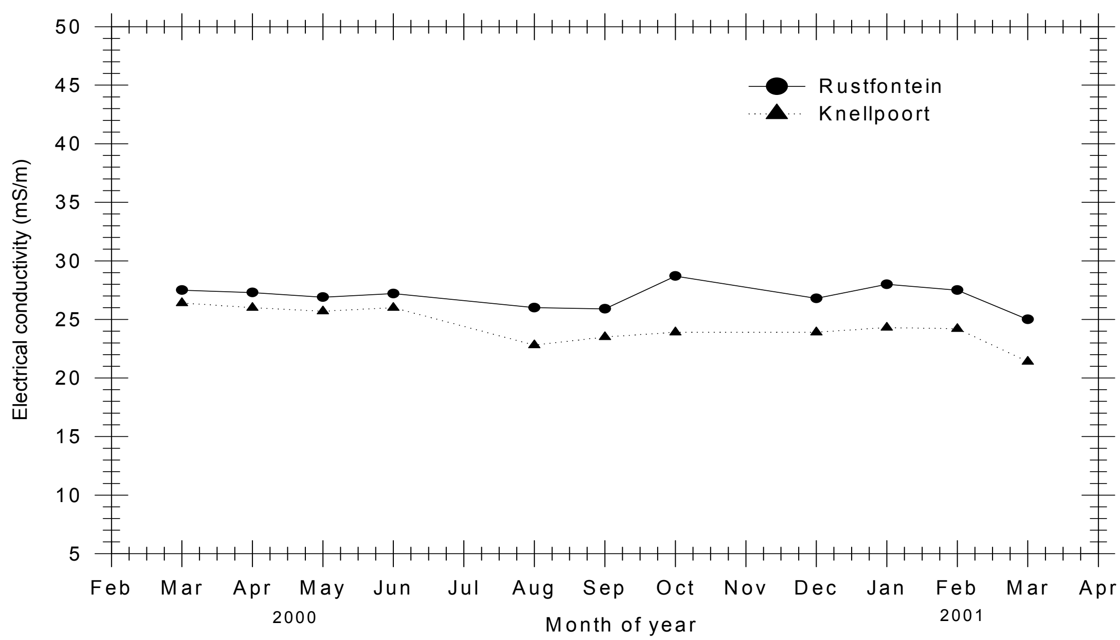


Figure 5.6: The seasonal variation in conductivity in Knellpoort and Rustfontein Dams.

The following general relationship is commonly used as an approximation for determining the total dissolved salt (TDS) concentration from electrical conductivity (EC) for South African inland waters (DWAF, 1996a):
TDS (mg/L) = EC (mS/m) at 25°C x 6.5.

Using this equation, the average TDS values at the different sites were calculated (Table 5.3). Waters with a pH lower or higher than 5.5 -7.5 typically require a conversion factor greater than 6.5. Thus, the values in the table are only an approximation of the real TDS concentrations.

Table 5.3: The average TDS concentration (mg/L) at the different sampling sites derived from the EC values (TDS (mg/L) = EC (mS/m) at 25°C x 6.5)

Sampling site	TDS concentration (mg/L)
Caledon River	108.8
Knellpoort Dam	153.9
Modder River	186.9
Rustfontein Dam	171.8

5.3.4) Ionic composition of the rivers and impoundments

In both the rivers and the impoundments of the *Novo* Transfer Scheme, the anion CO_3^{2-} dominated the water chemistry. The percentage dominance for CO_3^{2-} in the Caledon River was 53%, while it was 43,5% in the Modder River and 37% in Rustfontein Dam. The dominance of the other ions, excluding CO_3^{2-}, is shown in Figures 5.7 and 5.8. The dominance was calculated in terms of meq/L.

In the Modder River the ionic composition was, in order of dominance (Figure 5.7): $Na^+ > Ca^{2+} > Mg^{2+} > K^+ : CO_3^{2-} > Cl^- > SO_4^{2-} > F^-$. In the Caledon River, the ionic composition was in order of dominance (Figure 5.8): $Ca^{2+} > Mg^{2+} > Na^+ > K^+ : CO_3^{2-} > SO_4^{2-} > Cl^- > F^-$. These differences are significant, specifically the dominance of NaCl in the Modder River compared to $CaSO_4$ in the Caledon River.

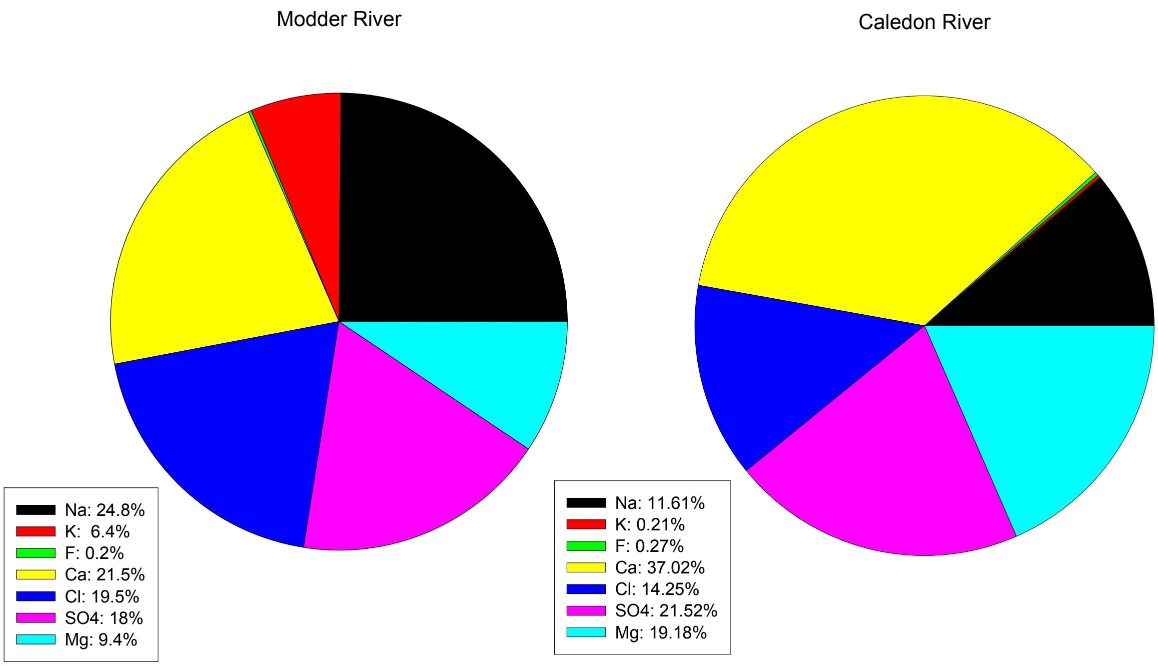


Figure 5.7: The average ionic percentage composition in the waters of the Modder (left) and Caledon (right) Rivers , excluding CO_3^{2-}.

In Rustfontein Dam, the order of dominance was similar to that of the Modder River which supplies it with water and agrees with the results of Seaman *et al.* (2001), namely $Na^+ > Ca^{2+} > Mg^{2+}$: $CO_3^{2-} > SO4^{2-} > Cl^- > K^+$ (Figure 5.8).

In Knellpoort Dam, the ionic composition had a dominance order of: $Ca^{2+} > Na^+ > Mg^{2+} > K^+$: $CO_3^{2-} > SO_4^{2-} > Cl^- > F^-$ (Figure 5.8). The resemblance to the ionic dominance order of the Caledon River is evident.

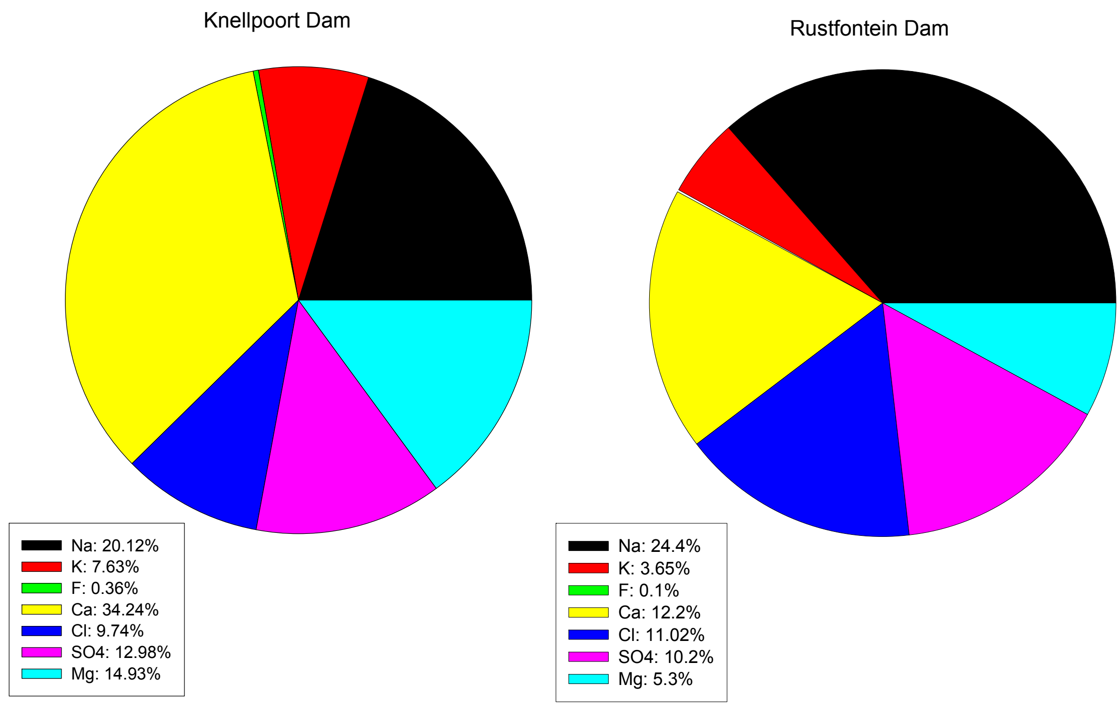


Figure 5.8: The average ionic percentage composition in the waters of Knellpoort Dam (left) and the Rustfontein Dam (right), excluding CO_3^{2-}.

5.3.5) Spatial variation in TP and PO_4-P

The TP and PO_4-P concentrations during the study period at the different sampling sites, are shown in Table 5.4. This clearly shows that the Modder River system contained significantly more P than the Caledon River. The small fraction of PO_4-P compared to the TP in the Caledon River is evident and indicates a very large fraction of organic material, possibly associated with pollution/eutrophication. It is also important to note that the percentage PO_4-P/TP increased from the lotic to the lentic state of the systems and that there was a significant decrease in both PO_4-P and TP following the impounding of the waters.

Table 5.4: The average TP and PO_4-P concentrations at the different sampling sites as measured during this study

Sampling site	PO_4-P (µg/L)	Standard deviation (SD)	TP (µg/L)	SD	% PO_4-P/TP
Caledon River	26.5	26	570	638	5
Knellpoort Dam	15.3	18	80	9	19
Modder River	56	68	291	182	20
Rustfontein Dam	27	27	95	53	28

The variation in the data (standard deviation) was the most significant for the Caledon River, while there was also significant variation for the PO_4-P for all the other sampling sites.

In the Caledon River, a linear relationship was found between TP and total suspended solids (Figure 5.9) supporting the fact that phosphate can be adsorbed onto the silt particles. This was also illustrated by Grobbelaar, (1983), who showed that bio-available N and P is absorbed unto suspended solids in the waters of the Amazon River. However, none of the other nutrients in the Caledon River showed any relationship with the TSS concentration, therefore the possibility exists that organic material is responsible for the relationship between TP and TSS.

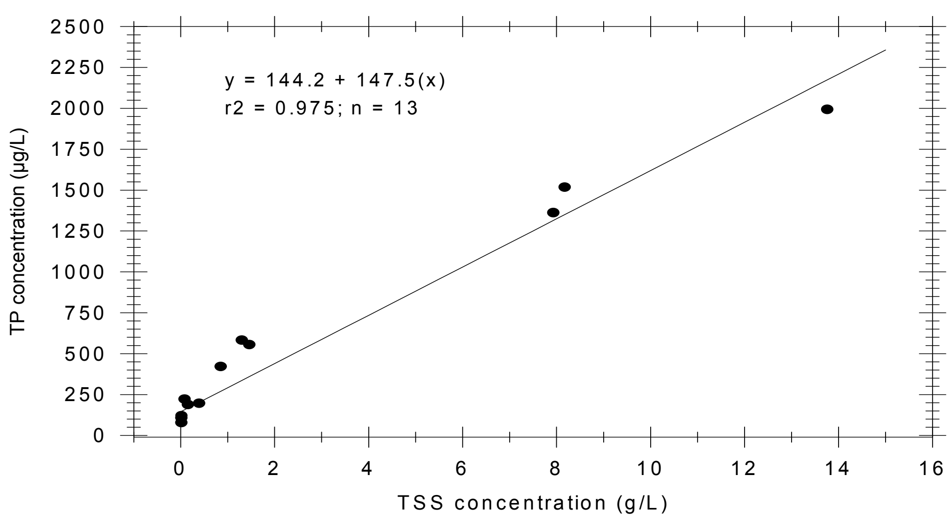


Figure 5.9: The linear relationship between TP and TSS concentration in the Caledon River

The consequence of this is that the transfer of turbid water from the Caledon River to the Knellpoort Dam can also cause enrichment of the water with phosphate, which together with other favourable conditions, such as high light availability, can lead to the development of algal blooms in Knellpoort Dam. To investigate this possibility, the average TP and PO_4-P concentrations in Knellpoort Dam before and after the transfer of water from the Caledon River were compared (Figure 5.10).

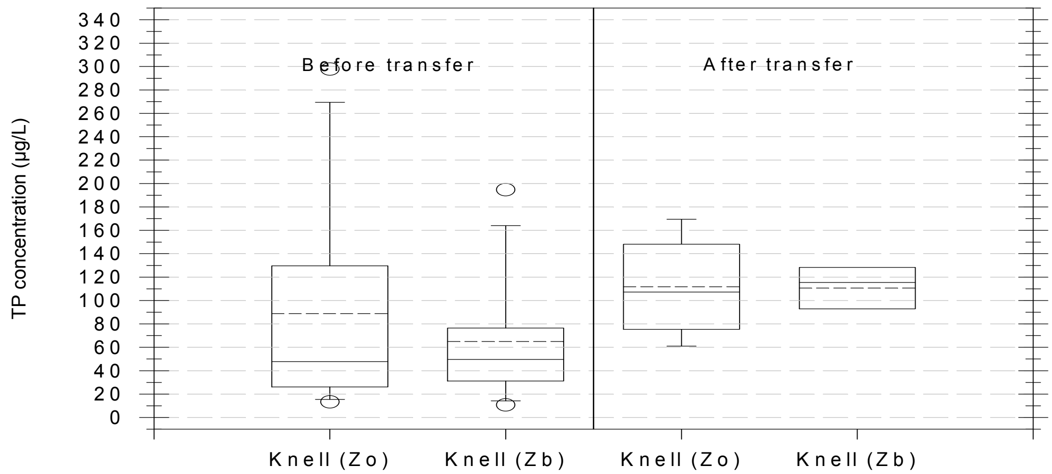


Figure 5.10: The variation in TP concentration (µg/L) in Knellpoort Dam before and after the transfer of water from the Caledon River

Figure 5.10 shows an increase in TP concentration at the surface and the bottom of Knellpoort Dam after the transfer of water. This can be expected, since the TP concentration is seven times higher in the Caledon River than in the impoundment. At the surface, the average TP concentration increased from 88.7 µg/L to 111.7 µg/L (an increase of 23 µg/L or 26%), while at the bottom the TP concentration increased from 64.97 µg/L to 110.59 µg/L (an increase of 45.62 µg/L or 70%). It was calculated that approximately seventy-four percent (74%) of the increase of TP at the surface was ascribed to the PO_4-P fraction.

The increase at the bottom of the dam was 44% higher than at the surface. The greater increase at the bottom could be ascribed to fact that a turbidity current develops in the impoundment which transports the sediment-rich water of the Caledon River to the bottom of the impoundment, carrying adsorbed phosphates with it. The presence of a turbidity current was confirmed in Chapter 3, when a three-dimensional turbidity profile of the inlet area was determined. On average, the total increase of TP concentration in Knellpoort Dam was 34.3 µg/L or 44%.

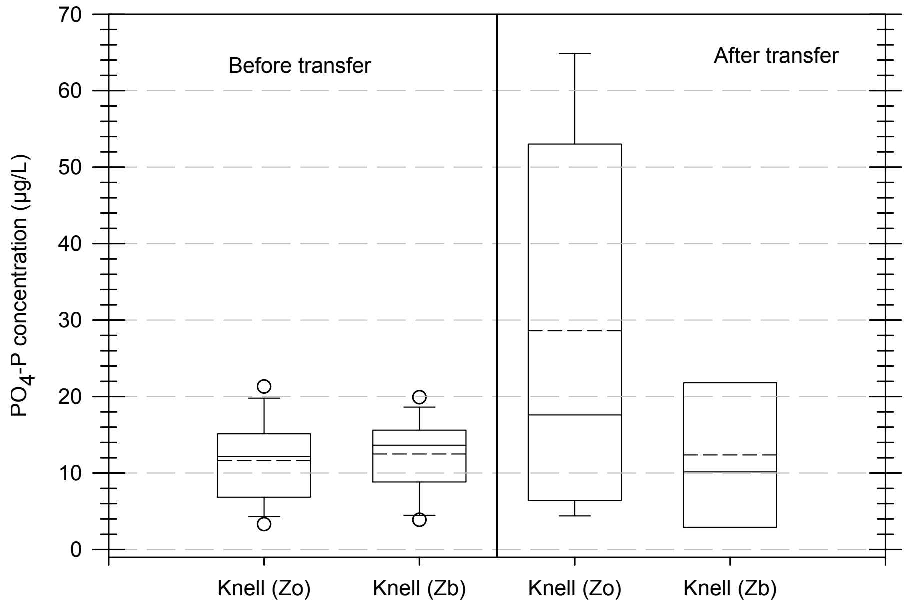


Figure 5.11: The variation in PO_4-P concentration in Knellpoort Dam before and after the transfer of water from the Caledon River.

There was a significant increase in the PO_4-P concentration at the surface of Knellpoort Dam after the transfer of water (Figure 5.11). At the surface of the water column, PO_4-P concentration increased from an average of 11.6 µg/L to 28.6 µg/L (an increase of 17 µg/L or 146%), and at the bottom there was no difference.

Shown in Table 5.5 are the average summer P concentrations (December to February), and from this it is clear that the Modder system contained more phosphate than the Caledon system.

Table 5.5: The average inorganic summer P concentration in the *Novo* Transfer Scheme

Sampling site	Average summer P-concentration (µg/L)
Caledon River	35
Knellpoort Dam	26
Modder River	88
Rustfontein Dam	53

5.3.6) Spatial variation in NO_3-N and NH_4-N

The average NO_3-N concentration in the Caledon River (322 µg/L) was 3.5 times higher than in the Modder River (94 µg/L), 5 times higher than in Knellpoort Dam (66 µg/L) and 7.5 times higher than in Rustfontein Dam (42.8 µg/L) (Table 5.6). The results also show a significant decrease in NO_3-N once the water is impounded and an increase in NH_4-N in Knellpoort Dam compared to the concentrations in the river. Both the NO_3-N and NH_4-N concentrations were lower in Rustfontein Dam than in Knellpoort Dam.

Table 5.6: The average NO_3-N and NH_4-N concentrations at the different sampling sites of the *Novo* Transfer Scheme

Sampling site	[NO_3-N] (µg/L)	SD	[NH_4-N] (µg/L)	SD
Caledon River	322.1	226	12.15	54
Knellpoort Dam	66.5	63.5	49.97	117
Modder River	93.9	142	12.7	12
Rustfontein Dam	42.8	39	9.58	7.5

The same as the P concentration, the average summer (December to February) inorganic N concentration (NO_3-N + NH_4-N) is important in determining whether algal blooms could develop or not. These were the highest in the Caledon River and were much reduced once the water was impounded (Table 5.6). The reduction was more marked for the Caledon/Knellpoort (3.2 times) compared to the Modder/Rustfontein (1.5 times) impoundment.

Shown in Table 5.7 are possible trophic classifications based on the summer N concentrations, and from this it is clear that the Caledon system is considerably more nutrient rich in terms of nitrogen than compared to the Modder system.

Table 5.7: The average inorganic summer N concentration at the different sampling sites of the *Novo* Transfer Scheme

Sampling site	Average summer N concentration (µg/L) (NO_3-N + NH_4-N)	Classification- DWAF (1996a)	Classification - Wetzel (1983)
Caledon River	649	Mesotrophic	Meso-eutrophic
Knellpoort Dam	200	Oligotrophic	Oligo-mesotrophic

Sampling site	Average summer N concentration (µg/L) (NO_3-N + NH_4-N)	Classification- DWAF (1996a)	Classification - Wetzel (1983)
Modder River	88	Oligotrophic	Ultra-oligotrophic
Rustfontein Dam	57.4	Oligotrophic	Ultra-oligotrophic

As for TP it is to be expected that the transfer of Caledon water to the Knellpoort Dam will lead to enrichment in terms of nitrogen of this impoundment. This was indeed the case, and is shown in Figure 5.12.

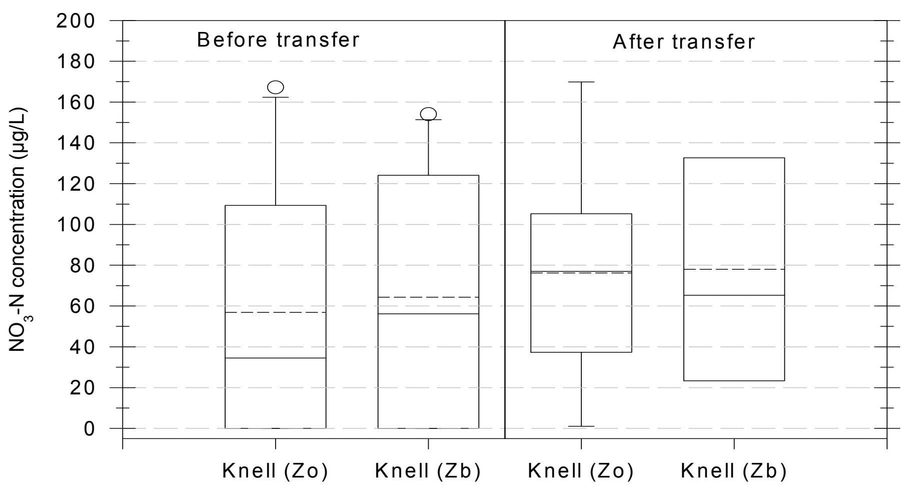


Figure 5.12: The variation in NO_3-N concentration in Knellpoort Dam before and after the transfer of water from the Caledon River.

At the surface of Knellpoort Dam, the NO_3-N concentration increased after transfer of water from 56.8 µg/L to 76.2 µg/L (an increase of 19.4 µg/L or 34%). At the bottom, the NO_3-N concentration increased from 64.3 µg/L to 78 µg/L (an increase of 13.7 µg/L or 21%). On average, the total increase in NO_3-N in Knellpoort Dam was 33.1 µg/L or 27.5% (Figure 5.12).

As in the case of TP and PO_4-P, the transfer of water from the Caledon River, increased the levels of NO_3-N in Knellpoort Dam, but not that of NH_4-N, since these concentrations were very low throughout the study period. Taking both NO_3-N and NH_4-N into consideration, the average total N concentration is within the same range in Knellpoort Dam than in the Modder River (116.5 µg/L and 106.6 µg/L respectively).

Calculating the N:P ratios for the whole study period (using NO_3-N and PO_4-P), the average for the Caledon River was 12.15, for Knellpoort Dam it was 5.1, and for both the Modder River and Rustfontein Dam it was 1.6. The implication of these results is discussed below.

5.3.7) Seasonal variation in TP, PO_4-P and NO_3-N

The TP, PO_4-P and NO_3-N concentrations in the Caledon and Modder Rivers were (on average) higher during the summer rainy season compared to the drier winter months (Figures 5.13 and 5.14). For the Modder River, the NO_3-N concentrations were below the detection limit for five sampling times during the study period (as can be seen on the graph). The maximum NO_3-N concentration in the Modder River was 439 µg/L, while in the Caledon River, it was 652 µg/L.

The PO_4-P concentrations for both systems did not show a clear seasonal trend. However, the lowest concentration was measured in February 2001 and in March 2001 it was undetectable. The minimum concentration was 0 µg/L for both systems and maxima were 101 µg/L and 193 µg/L for the Caledon and Modder Rivers respectively. There is thus a large variation in the data obtained during the study period.

The TP concentrations in the Caledon River showed much greater variations than in the Modder River, and reached a maximum of 1992 µg/L (with a minimum of 71.9 µg/L). The TP concentrations in the Modder River reached a maximum of 602 µg/L (with a minimum of 50.6 µg/L). The maximum concentrations in both systems were reached during the rainy season (summer) from December to February 2001.

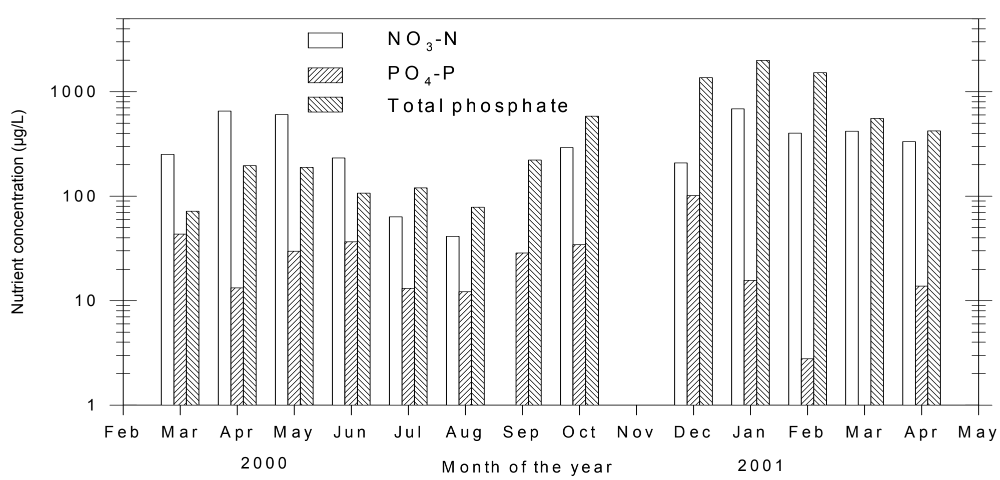


Figure 5.13: The seasonal variation of TP, PO_4-P and NO_3-N concentrations in the Caledon River.

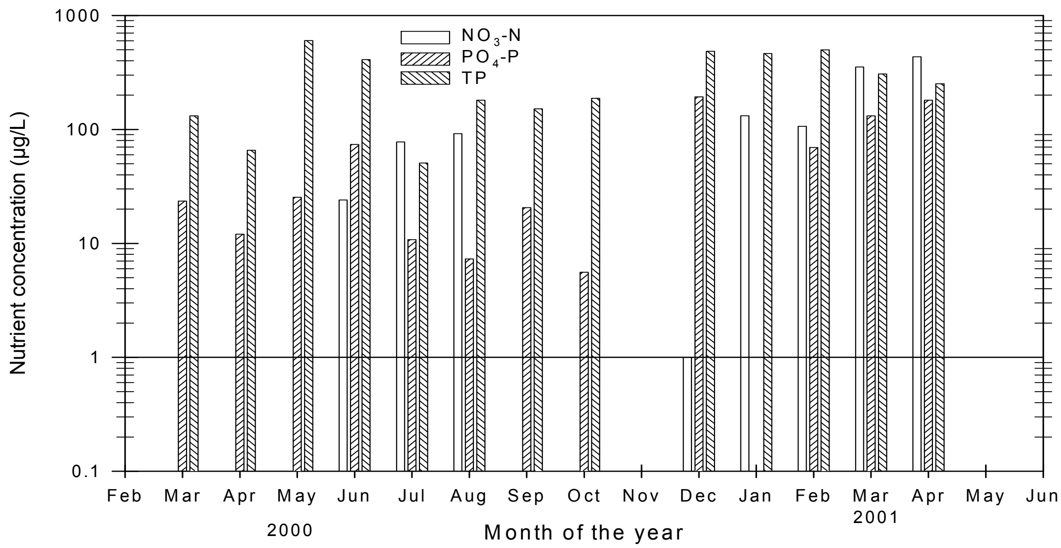


Figure 5.14: The seasonal variation in TP, PO_4-P and NO_3-N concentrations in the Modder River.

5.3.8) Spatial variation in SiO_2-Si

The average SiO_2-Si concentration in the Caledon River was 12.8 mg/L, in Knellpoort Dam 3.45 mg/L, in the Modder River 7.87 mg/L and in Rustfontein Dam it was 1.96 mg/L. The world average concentration for dissolved silica in large rivers is about 13 mg/L and it ranges from less than 0.5 to 60 mg/L in lakes (Horne & Goldman, 1994). The transfer of SiO_2-Si due to the silica-rich water of the Caledon River to Knellpoort Dam, was determined (Figure 5.15).

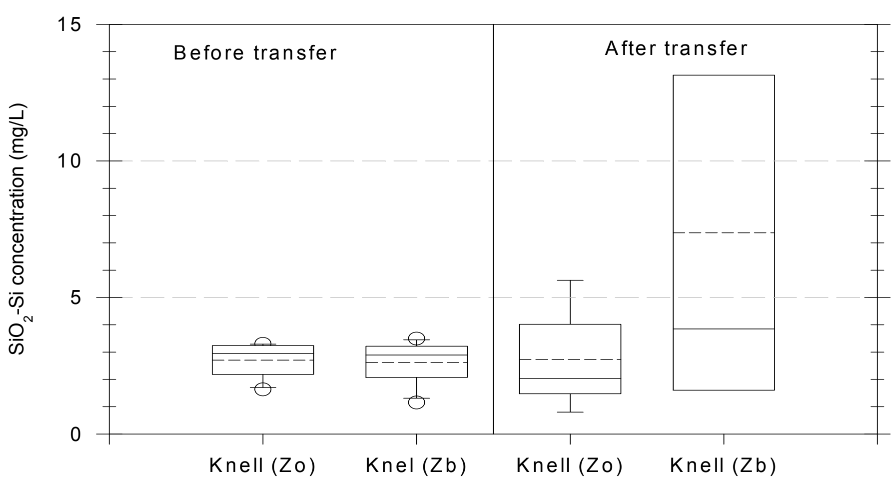


Figure 5.15: The variation in SiO_2-Si concentration in Knellpoort Dam before and after transfer of water from the Caledon River.

The transfer of water from the Caledon River caused an increase in the SiO_2-Si concentration at the surface of the water column from 2.7 mg/L to 2.72 mg/L (an increase of 0.02 mg/L or 0.7%), while a marked increase was observed at the bottom of the dam from 2.6 mg/L to 7.4 mg/L (an increase of 4.8 mg/L or 184%) (Figure 5.15). On average, the increase in SiO_2-Si in Knellpoort dam was 2.41 mg/L or 92%.

5.3.9) Diurnal variation of nutrients in Knellpoort Dam during and after transfer of water from the Caledon River

Two (2) 24-hour studies were carried out in Knellpoort Dam during the study period. One during November 2000, during the transfer of water, and one during October 2001, five months after transfer was stopped. The month of October was chosen in order to ensure that the climatic conditions between the two studies would be similar. Figures 5.16 - 5.23 show the diurnal variations in the nutrient concentrations at Sampling Site 1 and 2 during these periods. Site 1 was at the inflow of the transfer canal near the wall of the impoundment, and Site 2 was upstream from the wall (in the direction of the *Novo* pump station), more or less in the middle of the impoundment (approximately at the same position where the light profile measurements were done and as shown in Figure 3.17(a)).

During November 2000, the TP concentration at sampling site 1 showed a large increase at the bottom of the impoundment at 12h00, which was ascribed to the transfer of turbid water with a high TP content, which sank to the bottom. Thereafter, the concentration decreased during the night, probably as a result of mixing. The volume of water transferred through the channel was also decreased during the night, which could have added to the observed decrease.

The possibility that mixing occurred could be inferred from the increase observed at the bottom at 18h00 following the decrease at the surface (Figure 5.16). At sampling site 2, little variations in TP concentration was observed and the concentrations were much lower compared to Site 1 (Figure 5.16), indicating that the inflow had a localised effect on the receiving waters. In addition, at Sampling Site 2 the fluctuations at the bottom of the impoundment appeared to correlate with those in the surface water column.

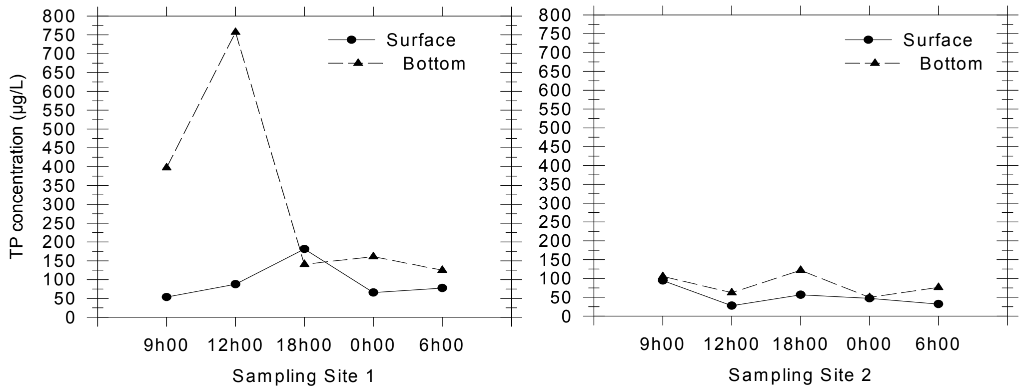


Figure 5.16: The diurnal variation in TP concentration in Knellpoort Dam during the transfer of water (November 2000).

During October 2001 (the second 24-hour study after the transfer of water was stopped), the TP concentrations were lower and in the same range at both the sampling sites than that at Site 2 during November 2000. Significant variations in concentrations occurred during the 24-hour study period, e.g. an increase at the bottom in TP at 0h00 (Figure 5.17 - note that the Y-scale is only a quarter of that of Figure 5.16). A midnight peak is seen at the bottom near the dam wall as well as in the open waters. Over a period of 24 hours it is difficult to explain the variance in nutrient concentrations at the surface and bottom of the water column, as it can be due to a variety of factors such as biological and chemical processes, as well as wind mixing and transport. Therefore, a long-term study on the potential impact on diurnal processes is recommended during periods of transfer, in order to get a clearer picture of the processes involved in nutrient dynamics in Knellpoort Dam.

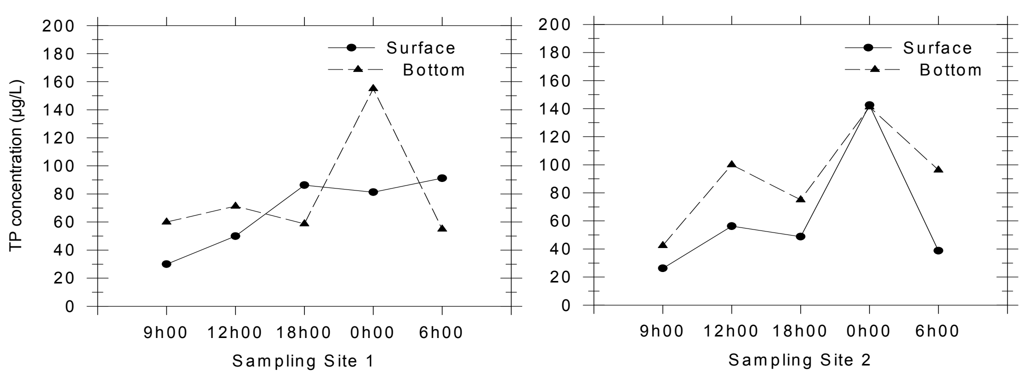


Figure 5.17: The diurnal variation in TP concentration in Knellpoort Dam after the transfer of water (October 2001).

During the transfer of water, the PO_4-P concentration followed the same pattern as TP at the bottom and surface of the water column at Site 1 (Figure 5.18). However, a significant increase of the PO_4-P concentration at the bottom of the impoundment at Site 1, coincided with a significant decrease in the PO_4-P concentration at the surface. This is possibly because of mixing in the late afternoon when it is usually most windy. It is also clear the PO_4-P contributed significantly to the TP content of the water. At Site 2, no large variations were observed (Figure 5.18) and the PO_4-P concentrations were generally low.

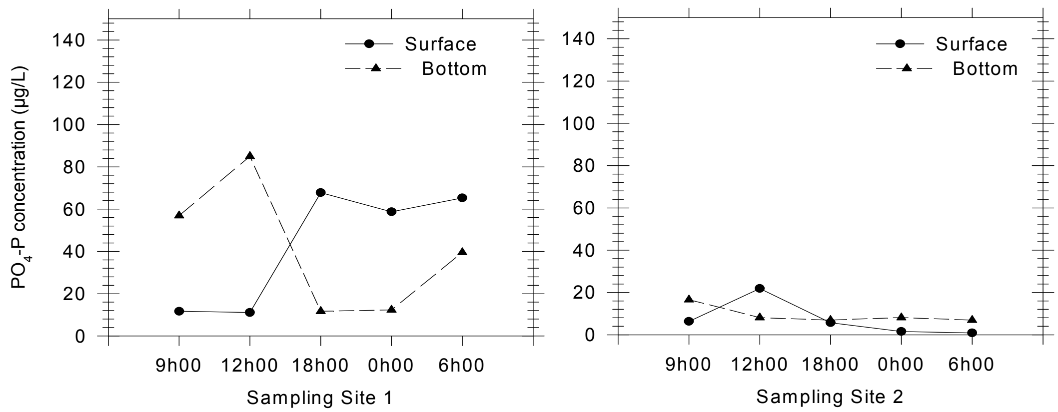


Figure 5.18: The diurnal variation in the PO_4-P concentration in Knellpoort Dam during the transfer of water (November 2000).

During October 2001, the PO_4-P concentration followed the same diurnal patterns as the TP concentrations, and they were generally low (Figure 5.19). There appears to be a definite midday decline with a late afternoon early evening peak. Again this is ascribed to the late afternoon winds which are common for this area.

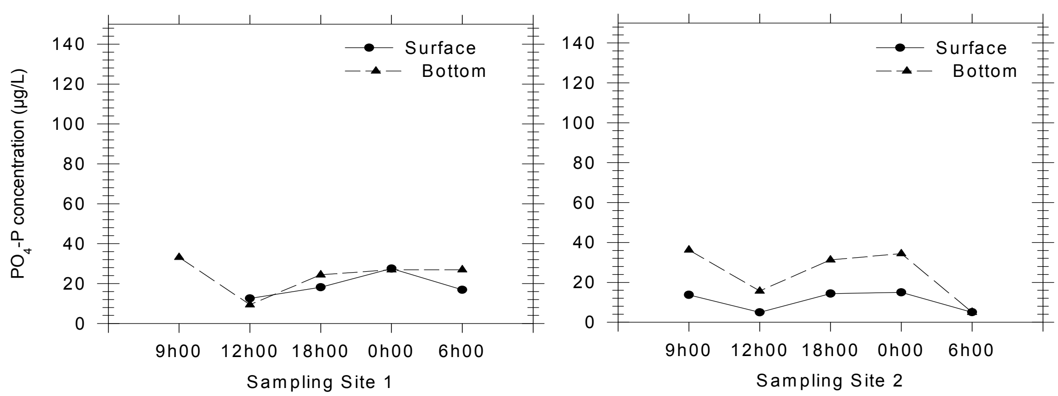


Figure 5.19: The diurnal variation in PO_4-P concentration in Knellpoort Dam after the transfer of water was stopped (October 2001).

The NH_4-N concentrations were very low during both the 24 hour studies (< 5 µg/L) and never increased above this concentration. Since the detection limit of the phenate method is 5 µg/L, it is concluded that any variations in the NH_4-N contents were insignificant.

The NO_3-N concentrations, on the other hand, were relatively high during both 24-hour studies that can be indicative of a low organic load or high nitrification. However, during this study the rate of nitrification was not determined. During November 2000 the NO_3-N concentration followed the same patterns as TP at the bottom of Site 1. There was a significant increase at the bottom of the impoundment during the day, but the concentration decreased as the transfer stopped. At both sites it appears as if there is a decrease during the day and an increase during the night. There was little difference between the surface samples at site 1 compared to those at site 2 (Figure 5.20).

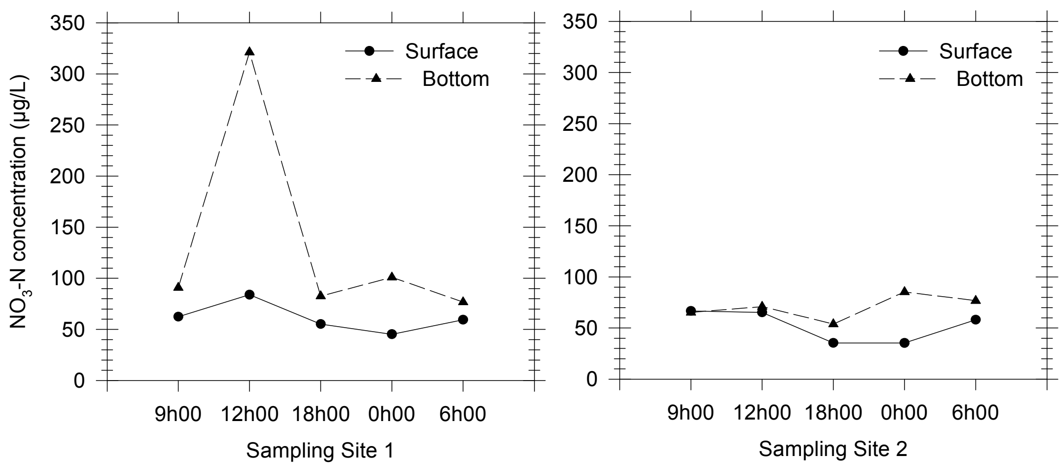


Figure 5.20: The diurnal variation in NO_3-N concentration in Knellpoort Dam during the transfer of water (November 2000).

In October 2001 the average NO_3-N concentration was higher than during November 2000. At Site 1, the NO_3-N increased during the night at the bottom (Figure 5.21). A possible explanation for this increase might be biological processes such as nitrification which generally occurs in the upper 5 cm of mud where nitrate concentrations in the interstitial water are higher than those in the overlying water. In addition, as NO_3-N decreased at the bottom, it increased at the surface.

At Site 2, NO_3-N increased at the bottom at 12h00, and decrease thereafter, while it increased at the surface during the night (Figure 5.21). Mixing obviously also influences the distribution of NO_3-N where the inverse relations between bottom and surface waters can clearly be seen.

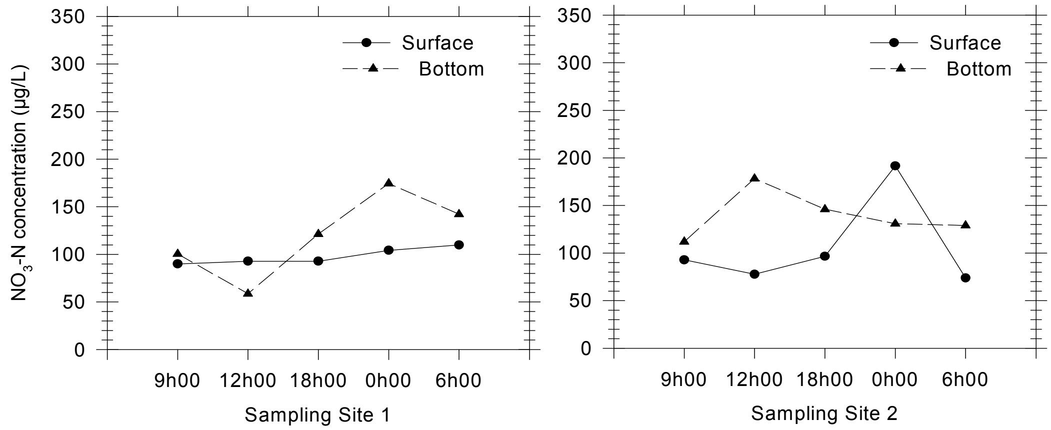


Figure 5.21: The diurnal variation in the NO_3-N concentration in Knellpoort Dam after the transfer of water (October 2001).

The SiO_2-Si concentration at Site 1 during November 2000 was relatively low despite the inflow of water from the Caledon River. As the concentration decreased at the surface, it increased at the bottom of the water column. At Site 2 the SiO_2-Si concentration did not show large variations, except for a slight increase during the night which decreased again in the early morning (Figure 5.22). The inverse relationship between the SiO_2-Si at the dam wall could be due to the transferred water sinking to the bottom, while in the open water the concentrations followed the same trend. In addition, wind mixing in the late afternoon could explain the increase at the bottom.

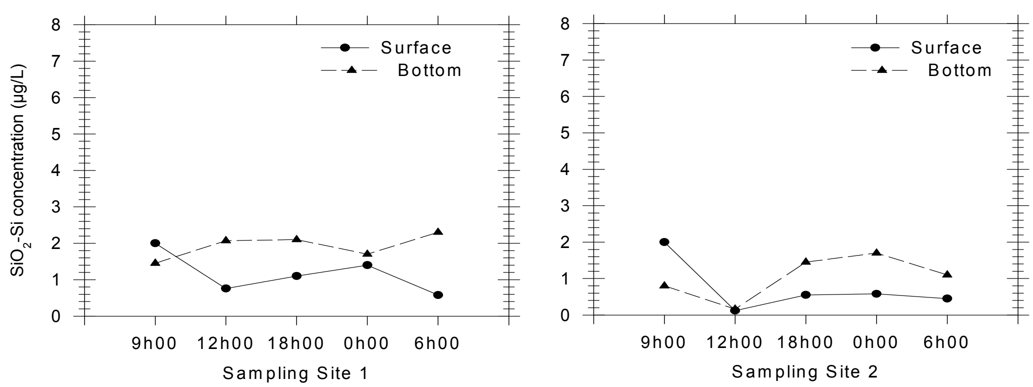
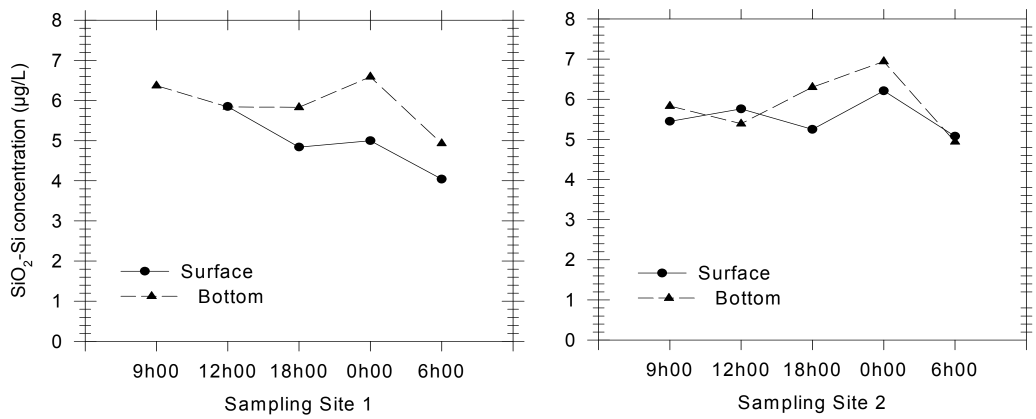


Figure 5.22: The diurnal variation in the SiO_2-Si concentration in Knellpoort Dam during the transfer of water (November 2000).

During October 2001, the SiO_2-Si concentration was much higher than during November 2000. This is probably due to enrichment from water of the Caledon River, which took place during the transfer. The concentrations followed the same trend at Sites 1 and 2 at the bottom and the surface. There was a slight increase in SiO_2-Si during the night, which decreased in the early morning and was similar to the trend at Site 2 during November 2000 (Figure 5.23). As previously noted, concentrations at the bottom were generally higher than at the surface.



<u>**Figure 5.23**: The diurnal variation in the SiO_2-Si concentration in Knellpoort Dam after the transfer of water (October 2001).</u>

The average concentrations of all the nutrients (except TP and PO_4-P) were higher during the second 24 hour study compared to the first. Concentration through evaporation could be a factor causing the variability and the decrease in TP and PO_4-P could be due to algal uptake, complexing processes, precipitation and sedimentation.

5.4) DISCUSSION

5.4.1) pH and alkalinity

The pH of natural waters is affected by factors such as temperature, the concentration of inorganic and organic ions, and biological activity. For surface water, pH values can range between 4-11 (DWAF, 1996a). Most surface waters have a pH of 6-9, naturally acid waters have a pH of 5-6, leachate from mines can have a pH as low as 2, and very eutrophic waters can have a pH as high as 11 (Horne & Goldman, 1994). pH in freshwater also varies according to the rate and quantity of primary productivity in a water body (Vollenweider, 1969b), because as CO_2 is used in photosynthesis, photosynthesis and hydroxide (OH^-) is released into the water. The relative proportions of the major ions could also influence the pH of natural waters and are determined by geological, atmospheric influences (Gibbs 1970; Hynes, 1970; Golterman, 1975), and also by anthropological influences.

The importance of pH lies in the fact that it affects the availability and toxicity of constituents such as trace metals, non-metallic ions such as ammonium, and essential elements such as selenium. Metals most likely to have increased detrimental environmental effects as a result of lowered pH, are Ag, Al, Ca, Co, Cu Hg, Mn, Ni, Pb and Zn (DWAF, 1996a). Since the adsorptive properties of large molecules (such as polyphenolics) and of particulate matter in water depend on their surface charges, altering the pH can also alter the degree to which nutrients such as PO_4^{3-}, trace metals and biocides adsorb to these materials. This is of particular significance where lowered pH levels can lead to the release of toxic metals from sediments. Thus, if significant changes in pH were to occur in the turbid waters of the Caledon River, or in Knellpoort Dam, it could have an important impact on metal and ion interactions within themselves and in the Modder River when water is transferred.

The average pH at most of the sampling sites was higher than the range between 6 - 8 as reported to be the expected average for natural ecosystems, by the Department of Water Affairs and Forestry. However, grassland rivers in Africa, such as the Modder and Caledon Rivers tend to be neutral to slightly alkaline (Awachie, 1981). The pH of both the Modder and Caledon Rivers as well as that of the other sampling sites, compares well with the average pH of the Vaal River, South Africa, which is 8.1, and the Orange River, with an average pH of 8.2 (Roos & Pieterse, 1995a). The variation in pH in the Caledon River during the study period was small, which is to be expected, since larger rivers require a great quantity of water to alter their pattern of discharge and characteristics (Hynes, 1970). In addition, the small variations in pH during the study period indicated a well buffered system. Thus, the transfer of water from the Caledon River, via Knellpoort Dam, to the Modder River, is not expected to have a significant effect on the pH of any of the receiving waters. A large variation in pH in Knellpoort Dam is also not expected, since there are little anthropogenic influences that can alter the pH, neither were significant algal blooms observed in the impoundment during the study period.

The buffering capacity of water involves carbon dioxide gas that dissolves in water to form soluble carbon dioxide. This reacts with water to form undissociated carbonic acid (H_2CO_3), which dissociates and equilibrates as bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}) (Horne & Goldman, 1994):

$$\underset{\text{Gas}}{CO_2} \Leftrightarrow \underset{\text{Dissolved gas}}{CO_2} \Leftrightarrow \underset{\text{Undissociated carbonic acid}}{H_2CO_3} \Leftrightarrow \underset{\text{Bicarbonate}}{HCO_3^- + H^+} \Leftrightarrow \underset{\text{Carbonate}}{CO_3^{2-} + 2H^+}$$

The series of reversible chemical changes is very important in determing the buffer capacity and thus controlling pH in natural waters. Plants use CO_2 for photosynthesis, and the rate-limiting step for photosynthetic carbon uptake is the dehydration of carbonic acid. As CO_2 is used in photosynthesis or dissolves from the air, the pH should change, since carbonic acid is either removed or added. However, the pH shift is reduced (buffered) by the large quantities of carbonate and bicarbonate present. This inorganic carbon equilibrium is the major pH buffering system for lakes and streams, and usually keeps the pH within a range of 6 to 9. At the average pH range, bicarbonate (HCO_3^-) is most abundant. Carbon dioxide is most abundant at low pH, while CO_3^{2-} dominates in high pH conditions. About 35% of the total inorganic carbon is present as bicarbonate at pH 6, but rises to 95% at pH 8. The precise levels of each component phase will vary with the temperature and ionic composition of the water.

The rate limiting step for photosynthetis namely, carbon uptake (the dehydration of carbonic acid) does not usually limit the overall photosynthetic carbon yield, since wind mixing normally replaces CO_2 as it is taken up. During short periods of intensive photosynthesis, however, this source may be insufficient to meet the maximum demands of the plants for CO_2. Furthermore, the photosynthetic uptake of equilibrium CO_2 continually pulls the reaction toward $CaCO_3$ until photosynthesis is stopped by darkness, lack of CO_2 or other limiting nutrients (Horne & Goldman, 1994).

The term alkalinity refers to the buffering capacity of the carbonate system in water and is used interchangeably with acid neutralising capacity (ANC), which refers to the capacity to neutralise strong acids. Alkalinity in water is due to any dissolved species that can accept and neutralise protons. Based on the alkalinity values indicated in Table 6.2, it can be seen that the Modder River has a higher buffering capacity than the Caledon River. In addition, the two impoundments have buffering capacities within the same range. Systems with higher buffering capacities usually remain at pH's of between 6 and 9, and this equilibrium is not easily shifted.

5.4.2) Dissolved oxygen

The maintenance of adequate dissolved oxygen concentrations is critical for the survival and functioning of the aquatic biota since it is required for the respiration of all aerobic organisms, especially the animals. Dissolved oxygen is usually not a limiting factor in running water, except in situations where the depletion of oxygen is caused by intensive biological activity (Awachie, 1981; Allan, 1995), which is not the case in either of the rivers in the *Novo* Transfer Scheme. Recovery from oxygen deficiency also occurs more rapidly in shallow and turbulent reaches where there is maximum contact between water and air (Walling & Webb, 1992). High water temperatures combined with low dissolved oxygen concentrations can cause stress effects on aquatic organisms. Under such conditions increased toxicity of zinc, lead, copper, cyanide, sulphide and ammonia have been observed (DWAF, 1996a). Again, no adverse effects due to oxygen stress are expected due to the transfer of water, since all the oxygen concentrations in the systems were within the same range (near saturation level). The lower oxygen concentrations at the bottom waters of Knellpoort Dam and the Modder River, are probably due to respiration that occurs in this region.

The redox potential is the electrical voltage that exists after connecting two electrodes, one made of hydrogen (H) and the other made of the material under consideration (a heavy metal). At neutral pH and 25°C, most common metals and nutrients are thermodynamically stable in their most oxidised form. When oxygen in the water reaches low levels and the environment becomes anoxic, a series of substances undergo chemical reduction, each at a specific redox potential. The effect is strongest in a thin layer, (the oxidised or reduced microzone), at the sediment-water interface. Manganese and iron are easily removed from anoxic sediments due to chemical reactions, while zinc, cobalt, molybdenum and copper are less mobile and dependent on living organisms for cycling (Horne & Goldman, 1994). We did not measure the redox potential of the sediments, but our data indicates nutrient release and resuspension from the sediments. There is therefore internal loading taking place in the Knellpoort Dam and the transfer of sediments rich in nutrients from the Caledon River, enhances this process.

5.4.3) Electrical conductivity

The higher conductivity during the dry season, in both the Caledon and Modder Rivers, is in accordance with Awachie (1981), who states that in African rivers the conductivity is normally higher during the dry season. This does not necessarily mean that the total quantity of salts is less during flood periods, but that it rather is a reflection of more diluted conditions. Alternatively, ion concentrations might not change significantly with fluctuations in discharge. This is expected when the water chemistry reaches equilibrium with the soil through which it percolates, or when concentrations approach saturation values. Contrary to these two common patterns, however, some ions have been found to increase in concentration with rising discharge (Allan, 1995; Golterman, 1975). This relationship was also found by Roos & Pieterse (1995b) who found that salinity in the Vaal River, South Africa, displayed seasonal changes that were strongly influence by turbid conditions following rainfalls, and that 56% of the variation in conductivity in this river was associated with variation in turbidity. They also stated that most rivers exhibit decreasing conductivity with increasing flow.

The target water quality guideline range for conductivity, as proposed by the South African Water Quality Guidelines (DWAF, 1996b) for recreational use, is below 70 mS/m. No health, aesthetic or treatment effects are associated at these levels. Both the rivers and the impoundments have conductivity levels far below this guideline concentration. However, no range is proposed by DWAF for conductivity values in natural aquatic ecosystems, except that the TDS (total dissolved salts) concentration should not be changed by more than 15% from the unimpacted condition of the system to the impacted situation. Since both the rivers are already impacted by human and other influences, there is no baseline against which to measure changes in TDS. The transfer of water also did not result in a more than 15% increase in the average TDS value of Knellpoort Dam.

The average conductivity values of typical unpolluted rivers in general are approximately 35mS/m, which are higher than those found in both the Caledon and Modder Rivers (Webb & Walling, 1992). The conductivity of the Vaal River, South Africa, was, on average, 76 mS/m (Roos & Pieterse, 1995b) and that of the Orange River between 18-30 mS/m for the period 1977-1997 (DWAF, 1997). Since all the average conductivity values of the different components (both rivers and impoundments) of the *Novo* Transfer Scheme were below the value of typical unpolluted water (35mS/m) (Figure 5.6) for the study period (even after the transfer), no adverse effects such as an increase of salinity levels of Knellpoort Dam, is expected because of the transfer of water from the Caledon River. In fact, it is more likely that the water transferred from the Knellpoort Dam (23.7 mS/m), will dilute the more saline waters of the Modder River (28.7 mS/m), once this section of the transfer scheme is put into operation.

The high conductivity values measured in the Modder River could be due to agricultural run-off from adjacent farms. During 1996-1997, the average conductivity in the Modder River was found to be 36 mS/m (Koning, 1998), and this was ascribed to the influence of the cities, Bloemfontein and Botshabelo. The reason for the lower conductivity value during 2000-2001, could be that the sampling point used during this period was upstream from Rustfontein Dam, while the sites during 1996-1997 were downstream of Rustfontein Dam. Since the sampling point for this study was upstream from both Botshabelo and Bloemfontein, urban discharge is ruled out as a possible contributor to conductivity. The Caledon River, on the other hand, could be influenced by sewage run-off from Maseru, or industrial activities taking place in this area.

The different salinities of the Modder and Caledon Rivers could also be attributed to the geological formations underlying the rivers and not solely to agricultural runoff from the Modder River catchment. The mountain mass of Lesotho within the catchment area of the Caledon River consists of basalt, approximately 1 400 m thick and belongs to the Stormberg Series. This overlays a layer of Cave Sandstone, the interface varying in altitude between 1 800 and 2 100 m above sea level (Grobbelaar & Stegmann, 1976). The Modder River catchment, on the other hand, lies within a geological area classified as the Molteno, Red Beds and Cave Sandstone

Stages of the Stormberg Series and the Beaufort Series, both of the Karoo system, with dolerite dykes occurring in places (Grobler & Davies, 1981). Weathering in the catchment of the Modder River is definitely more pronounced than for the Caledon and this could contribute to the higher salinities of the former.

Freshwater species are routinely found in waters with less than 1000 mg/L, whereas only a few freshwater species are found in waters with more than 10 000 mg/L TDS (Bierhuizen & Prepas, 1985). The TDS concentrations in fresh water are generally greater than 65 mg/L in water in contact with precambrian shield areas and in the range of 200 - 1100 mg/L in water in contact with palaeozoic and mesozoic sedimentary rock formations (DWAF, 1996a). Since the average conductivity in the Modder River was higher than at the other sampling sites (the reasons for this were discussed earlier), the TDS values will consequently be higher. Therefore, the transfer of water from the Knellpoort Dam, may dilute the water of the Modder River, decreasing the TDS concentration.

Discharge does not affect conductivity levels in impoundments in the same way as in rivers. Impoundments store water from peak floods and this water is usually low in conductivity. Thus, conductivity levels below an impoundment will usually be lower than up-stream. In the Great Fish River, South Africa, conductivity dropped from 21.0 mS/m to 13.0 mS/m after an impoundment was constructed in it (Palmer & O'Keeffe, 1990).

Talling & Talling (1965) classified lakes according to their conductivity: Class I corresponds to waters with a conductivity of less than 6 mS/m and includes many lakes supplied directly by surface run-off or rivers of low salinity. Most of the small lakes of volcanic origin and the "river-lakes" are in this class. Class II includes lake waters of conductivity between 6 and 60 mS/m and Class III covers very saline lakes in which the accumulation of salts within closed basins has gone further, often forming solid deposits (especially Na_2CO_3, $NaHCO_3.2H_2O$) around the lakes.

The relationship between turbidity and TDS will be the same as turbidity and conductivity which is discussed in detail in Chapter 3.

5.4.4) Ionic composition of the *Novo* Transfer Scheme

The ionic composition of river waters over the world varies significantly and depends on a number of factors. Table 5.8 shows the average values of the various constituents in 60 major rivers worldwide.

Table 5.8: The average concentrations of major constituents in rivers of the world (Meybeck *et al.*, 1992)

Constituent	Concentration (mg/L)
SiO_2	2.4 – 20
Mg	0.85 - 12.1
Na	8 - 25.3
K	0.5 – 4
Cl	0.6 – 25
SO_4	2.2 – 58
HCO_3	10 – 170
pH	6.2 - 8.2
TSS	10 – 1700
NH_4	0.005 - 0.04
NO_3	0.05 - 0.2
PO_4	0.002 - 0.025

In addition to the above, calcium (Ca) is one of the major constituents of freshwater. According to Meybeck (1996), it ranges between 2 - 50 mg/L with an average of 27.5 mg/L. The above can be compared to the average composition of African river waters (Allan, 1995) as shown in Table 5.9. From this it is clear that the river waters of Africa fall within the ranges of the major rivers in the world.

Table 5.9: Average composition of African river waters (mg/L)

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

The chemical composition of African lake water was described by Talling & Talling (1965). They state that in the numerous pans of South Africa, the bicarbonate ion generally forms less than half of the total anions, and chloride is often present in comparable amounts. The pans also show an extensive range of alkalinity. However, according to their classification, Knellpoort and Rustfontein Dams fall in the highly variable but large group of "river-lakes", formed by local obstructions in the rivers which consequently determine their major chemical characteristics.

The importance of the different ions lies in their roles played in various metabolic processes. Ca^{2+} is essential for metabolic processes in all living organisms and as a structural and skeletal material in many. Mg^{2+} is needed by all cells for phosphate transfer and act as a transition metal in the reactive centre of chlorophyll. Na^{+} is important for all metabolic processes, K^{+} is an enzyme activator and Cl^{-} plays a role at the photolysis of water, ATP formation and phosphorylation reactions, to name a few (Horne & Goldman, 1994).

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

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

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

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

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

According to the classification of Dallas & Day (1993), the Modder and Caledon Rivers are Category 2 rivers, where Ca^{2+}, Mg^{2+} and Na^{+} are the dominant cations. However, from 1997-2000, the dominance of Ca^{2+} has slightly decreased, while both Na^{+} and Cl^{-} have slightly increased in dominance (Table 5.10). Although these changes can not be viewed as statistically significant at this stage, it may be cause for concern if the trend continues in future.

Table 5.10: The difference in the dominant ions present in the Modder River over the two study periods.

Ion	**1996 – 1997** (Koning, 1998)	**2000 - 2001**
Na^{+}	23.73	24.85
Ca^{2+}	23.81	21.57
Cl^{-}	19	19.5

During the 2000 - 2001 study period, Na^{+} and Ca^{2+} (meq/L) were co-dominant in the Modder River, but Cl^{-} and SO_4^{2-} dominated over Mg^{2+}. This was also the case in Rustfontein Dam. In the Caledon River, Ca^{2+} was dominant, while Cl^{-}, SO_4^{2-} and Mg^{2+} all dominated over Na^{+}. In Knellpoort Dam, the dominant ions were Ca^{2+} and Na^{+} (as in the Modder River, although the percentage dominance of Na^{+} was much lower in the impoundment), but Mg^{2+} and SO_4^{2-} dominated over Cl^{-}, which were different from both the rivers.

In African lakes, Cl^{-} is generally the anion present at the highest concentration, while SO_4^{2-} is present at low concentrations (Talling & Talling, 1965). In Rustfontein Dam, Cl^{-} was present at the highest concentration in accordance with Talling & Talling (1965), however, in Knellpoort Dam, SO_4^{2-} dominated.

Of the cations, Na^+ is generally dominant in African lakes, while K^+ is apparently always present in smaller quantities than sodium (Talling & Talling, 1965). In both the impoundments, Na^+ was dominant. The variation of Ca^+ and Mg^{2+}, however, follows variable patterns in different lakes, with a rise in concentration with increasing salinity (Talling & Talling, 1965).

The difference in dominant ions in the waters of the Caledon River, Knellpoort Dam and Modder River could have a significant impact if large volumes of water are transferred over long periods. If water is transferred from the Caledon River, it could lessen the dominant role of Na^+ in Knellpoort Dam, simultaneously increasing the dominance of Cl^-, while the transfer of water from Knellpoort Dam to the Modder River, could decrease the dominance of both Na^+ and Cl^- in this river, shifting its ionic composition more to its present status as a Category 2 river. This is confirmed by Thornton *et al.* (1992) who stated that the transfer of water between basins can result in an impoundment having a very different ionic composition to that which would occur naturally in the receiving basin. Some impacts on the biota due to these changes can be a change to species more common in marine habitats, but this will only occur in case of high salinity. However, the major consequence of a change in the ionic composition, is the changes in alkalinity. Such changes will result in changes in the plankton composition. Talling & Talling (1965) stated that in most alkaline lakes, Entomostraca disappears, while Rotifera and blue-green algae then occur more frequently. If an impoundment changes from Class I to Class II, the algal composition will change from one dominated by *Melosira* and desmids, to one dominated by *Nitzschia* species.

Another important aspect is that different ionic and metal compositions can influence the different species of P present in a water body and consequently the bio-availability of phosphorus. A study conducted by de Jonge *et al.* (1993) found that the fraction of adsorbed SRP (soluble reactive phosphorus) decreased rapidly with depth. The metal associated P was highest near the sediment surface and did not decrease dramatically with depth, while the calcium-associated phosphorus was rather constant with depth. The redox-sensitive and mainly iron-associated phosphorus strongly decreased with depth. It was also found that the calcium-associated fraction of P is the largest while the organic fraction was the second most important. However, in the Knellpoort Dam, the soluble reactive phosphorus fraction of the total phosphorus concentration did not vary significantly between the surface and the bottom of the impoundment.

5.4.5) Spatial variation in TP and PO4-P

Contemporary views conceptualise P dynamics as a set of processes by which available P is taken up by plankton, utilised in the food chain, and eventually released either directly or by detrital mineralisation, rendering unavailable forms of P available. This P cycle is considered to be fundamentally the same for all P-limited communities. Although communities are viewed as differing quantitatively by the rates at which P moves through the various forms in this scheme, they are not viewed as differing in the fundamental structure of the P cycle itself (Heath & Francko, 1988).

Although phosphorus is not needed for growth in such large quantities as carbon, oxygen, hydrogen, or nitrogen, it is perhaps the most common growth-limiting element in freshwater (Horne & Goldman, 1994; Schindler *et al*., 1977). In South Africa, phosphorus concentrations between 10-50 µg/L are commonly found, while concentrations as low as 1 µg/L of soluble inorganic P may be found in “pristine” waters and as high as 200 mg/L of TP in some enclosed saline water bodies (DWAF, 1996a). P is extremely reactive under oxidising conditions, and interacts with many cations (such as Al, Fe and Ca) to form relatively insoluble compounds that precipitate. Availability is also reduced by adsorption of P onto inorganic colloids, organic compounds such as humic and particulate materials (DWAF, 1996a).

The average TP concentration in the Caledon River was seven times higher than in Knellpoort Dam, six times higher than in Rustfontein Dam and two times higher than in the Modder River. The TP concentration was constantly higher in the Caledon River than in the Modder River, and much higher during the rainy season. The much higher TP concentration in the Caledon River could be ascribed to phosphate-ions that are adsorbed to the suspended particles. These high concentrations of TP are, however, not necessarily biologically available. In addition, the Caledon River system is probably light-limited due to the high turbidity concentrations and this restricts algal growth, despite the high nutrient concentrations.

Although high, the N and P concentrations in both the Caledon and Modder Rivers were lower than that found in other Southern African rivers, for example, in the Marimba River, Zimbabwe (Nhapi & Tirivarombo, 2004). This river is highly impacted by sewage inflow from the city of Harare, and shows concentrations of approximately 13.5 mg/L N and 2.6 mg/L P.

Considering the PO_4-P concentrations, the average in the Caledon River was two times higher than in Knellpoort Dam. In the Modder River, the average PO_4-P concentration was higher than in both the Caledon River and Knellpoort Dam, although the TP concentration was lower. In view of this, it is possible that there are more bio-available P in the Modder River than in the other rivers and impoundments.

In the Vaal River, a statistically significant positive correlation between PO_4-P and TP was found (r^2 = 0.46, p > 0.001) (Roos & Pieterse, 1995a). Five percent of the TP concentration in the Caledon River, 19% of the TP concentration in Knellpoort Dam, 20% of the TP in the Modder River and 28% of the TP concentration in Rustfontein Dam, was in the form of PO_4-P. Thus, a large portion of the TP concentration in the Modder River system, was in the form of PO_4-P, while this was not the case in the Caledon River systems. The ratio of PO_4-P to TP is usually low in oligotrophic waters, whereas at high TP concentrations, the dissolved inorganic phosphorus pool reaches almost 100% of the TP (Harris, 1986).

Roodeplaat Dam (De Wet, 1986) and Hartbeespoort Dam (Robarts, 1984), both eutrophic systems, showed that the PO_4-P fraction was approximately 90% and 66% of the TP fraction respectively. Furthermore, it was found in waters where the PO_4-P constitutes a small fraction of TP, blue-green algae tended to develop (Talling & Talling, 1965). From the PO_4-P:TP ratios, shown in Table 6.3, the Caledon River can be viewed as oligotrophic, while the Modder River, Knellpoort Dam and Rustfontein Dam can be viewed as mesotrophic.

In comparison to the above, the South African Water Quality Guidelines classify the trophic status of freshwaters in South Africa according to the average inorganic P-concentration during summer (Table 5.11).

Table 5.11: Trophic status of South African freshwaters according to the average summer P concentration (DWAF, 1996a)

Average summer P concentration (µg/L)	Trophic status
< 5	Oligotrophic, usually moderate levels of species diversity, low productivity, no nuisance growth of aquatic plants or blue-green algal blooms.
5-25	Mesotrophic, usually high levels of species diversity, productive, nuisance growth of aquatic plants and non-toxic blooms of blue-green algae.
25-250	Eutrophic, usually low levels of species diversity, highly productive, nuisance growth of aquatic plants and toxic blooms of blue-green algae.
>250	Hypertrophic, very low levels of species diversity,

Average summer P concentration (µg/L)	Trophic status
	very highly productive, nuisance growth of aquatic plants and blooms of blue-green algae, often toxic

Based on the above, the four systems studied were classified according to their trophic status (Table 5.12).

Table 5.12: Trophic status of the different systems based on the average summer P concentration based on the South African Water Quality Phosphate Guidelines

Sampling site	Average summer P-concentration (µg/L)
Caledon River	35 (Eutrophic)
Knellpoort Dam	26 (Mesotrophic/Eutrophic)
Modder River	88 (Eutrophic)
Rustfontein Dam	53 (Eutrophic)

This clearly suggests that all the systems in the *Novo* Transfer Scheme can be classified as eutrophic, except Knellpoort Dam, which falls on the border between mesotrophic and eutrophic. The average summer P concentration in Knellpoort Dam was determined after the transfer. This probably resulted in a higher average P content and could have influenced the trophic status.

However, the average guideline concentrations for phosphate in oligotrophic waters in Greece are 0.02 µg/L, in mesotrophic waters 0.09 µg/L and in eutrophic waters it is 0.34 µg/L, which are orders of magnitude lower than the South African Guidelines (Ignatiades *et al.*, 1992). This indicates how different systems react differently to eutrophication.

Although Nurnberg & Peters (1984) illustrated with a refined bioassay technique that under most conditions, soluble reactive phosphate (SRP) accurately estimated P availability, algae are capable of utilising other forms of soluble P than PO_4-P. Many investigations showed that algal phosphatase that are bound to certain algal membranes, increase in activity when phosphorus becomes deficient. Consequently, particulate P can provide 28 to 41% of algal available P, in addition to dissolved PO_4-P (Newman & Reddy, 1993).

The possible influence of an increase in turbidity on the rate of phosphate uptake, the immediate fate of phosphate and the potential rate of release of phosphate from dissolved phosphorus compounds was predicted by Heath & Francko (1988). They investigated these processes in the epilimnia of two impoundments and one natural lake in north-central Oklohoma during summer and late autumn. They found that in the lake dominated by clay suspensoids, the fate of phosphate differed from that of the others, with much of the P adsorbing to suspended silts and clays rather than being taken up by the biota. The study further suggests that clay suspenoids may represent another important factor that determines the fundamental structure of the P cycle by influencing the availability of P. This could become the fate of phosphorus in Knellpoort Dam if turbidity continues to increase with the transfer of water, where the P will adsorb onto the clay particles rendering them biologically unavailable (this is also discussed in Chapter 4). Sediment resuspension can also be a significant cause of P release. Newer evidence suggests that bacterial degradation of organic matter under anaerobic conditions may also be a significant source of phosphorus release from sediments (Evans *et al*., 1997).

A study by Teodoru and Wehrli (2005) on retention of sediments and nutrients in the Iron Gate I reservoir on the Danube River, showed that the sediment accumulation corresponded to 5% TN, 12% TP and 50% TSS of the incoming loading. A mass balance revealed that more N and P are leaving the impoundment than entering via the inflow. This shows that the impoundment was temporarily acting as a nutrient source, and not as a nutrient sink, as most other impoundments. When acting as a sink, there are two possible mechanisms. The first is that a graded surface permits a greater area of the sediment to be in contact with the hypolimnion and for the sediment oxygen demand to deplete a greater hypolimnetic volume, with an associated fall in redox causing an increment in dissolved phosphorus to the hypolimnion. The alternative mechanism is based on the association of phosphorus “injection” to the epilimnion during wind-mixed epiosdes, when the enhanced shear stress on shallow sediments resuspends fine deposits and the waters into full circulation (Reynolds, 1996).

The Target Water Quality Range (DWAF, 1996a) for P states that the trophic status of a freshwater body should not increase above the present level. Knellpoort Dam was classified as mesotrophic (PO_4-P = 12 μg/L) before the transfer and as mesotrophic/eutrophic (PO_4-P = 25 μg/L) after the transfer. Thus, the transfer of the water from the Caledon River did cause a change in the trophic classification of the impoundment, and can continue to do so, especially if large volumes of water are transferred over extended periods of time.

Unfortunately, no significant transfer of water took place between the Knellpoort Dam and the Modder River during the study period. The only transfer during this study was during the last week in April 2001, to complete the commissioning of the transfer scheme. Water samples were taken from the Modder River at the end of this time and a decrease was observed in the

TP concentration, from 306 µg/L before the transfer to 251 µg/L after the transfer (a decrease of 55 µg/L or 18%). Although some measure of dilution is to be expected when the water of the Knellpoort Dam is transferred to the Modder River, care must be taken not to view this decrease (measured only once) as the norm, since TP concentrations can vary significantly in a river, both spatially and over time. Furthermore, flow in the Modder River can sometimes increase up to 19 m^3/s in the rainy season (Koning, 1998), which will imply that water from the *Novo* pumping station will sometimes contribute only 5% of the total flow. Under such conditions the transfer of water will have little effect on TP concentrations.

The fact that the water of Knellpoort Dam consistently had much lower TP and PO_4-P concentrations than the Modder River, suggests that (together with low flow conditions in the Modder River and extended periods of transferring water), the transfer of water from the *Novo* pump station at Knellpoort Dam could dilute and thus decrease these concentrations in the Modder River. On the other hand, transferring water (phosphate-rich) from the Caledon River for extended periods could increase the TP and PO_4-P concentrations in Knellpoort Dam.

Consideration must be given to the fact that any assessment of the influence of inorganic phosphorus concentrations should be coupled to an evaluation of the ratio of inorganic nitrogen to inorganic phosphorus and *vice versa*. Calculating the N:P ratios for the whole study period (using NO_3-N and PO_4-P), the average for the Caledon River was 12.15, for Knellpoort Dam it was 5.1, and for both the Modder River and Rustfontein Dam it was 1.6. Before the transfer of water from the Caledon River, the N:P ratio in Knellpoort Dam was 5.02 and after the transfer 3.05. From this it is clear that the transfer of water had a major effect on the N:P ratio of Knellpoort Dam.

A study by Sakamoto (1966) noted that the chlorophyll *a* yield in Japanese lakes followed a logarithmic function with both TP and TN. He concluded that over the range 10<TN:TP<17 by weight, chlorophyll *a* yield was very nearly balanced with respect to both TP and TN, but that chlorophyll *a* was dependent only on TN when TN:TP<10, and only on TP when the TN:TP>17.

Based on the above calculated ratios by Sakamoto (1966), the chlorophyll *a* yield of the Caledon River is considered to be dependent on both TN and TP, and for the Modder River and Rustfontein Dam, the chlorophyll *a* yield is dependent on TN (thus the system is nitrogen limiting). For Knellpoort Dam the system was also found to be nitrogen limited, and even more so after the water was transferred. However, as will be discussed in Chapter 6, turbidity also plays an important role with regard to algal growth in these turbid systems, by limiting the available light.

Vollenweider (1976) stated that a quantifiable relationship exists between the quantity of nutrients reaching a water body and its trophic degree measurable with some kind of trophic scale index. Thus, chlorophyll *a* could be predicted from the loading characteristics of lakes, by relating phosphorus concentration to loading. Oligotrophic lakes would be found below the 10mg/m^3 critical loading P concentration, and eutrophic lakes would be found above the 20 mg/m^3 critical loading concentrations.

However, variability in the TN:TP ratios may result in varied TP:chlorophyll *a* relationships. This is because when the N:P ratio drops and nitrogen becomes more limiting, the concentration of dissolved phosphorus (PO_4-P) should increase and represent a greater proportion of the TP (Smith, 1982). As a result, TP would no longer represent the mean particulate phosphorus concentration and the chlorophyll *a*:TP relationship would become variable (Smith, 1982) (also see Chapter 6).

5.4.6) Spatial variation in NO_3-N and NH_4-N

Since nitrogen is often in short supply for plant growth on land, it can also be a growth-limiting nutrient in water. In some aquatic ecosystems nitrogen is the element that limits plant growth the most. This tends to occur most frequently in lakes at the eutrophic or oligotrophic ends of the trophic spectrum (Horne & Goldman, 1994). In South Africa, inorganic N in natural aerobic surface waters is usually below a concentration of 500 µg/L, but may increase to above 500 µg/L in highly enriched water. The processes of ammonification, nitrification, denitrification and the active uptake of nitrate by algae and higher plants, are all regulated by water temperature, pH, phosphorus concentration, oxygen availability and pH (DWAF, 1996a). In some impoundments, denitrification can result in a substantial reduction in the inorganic NO_3-N concentrations compared with the concentrations of the source waters. This is usually associated with a greater instability in the water column due to enhanced turbulence in an impoundment, its relative shallowness, long water retention time and reduced dissolved oxygen content of the water (Thornton *et al*., 1992).

The average NO_3-N concentration of 94 µg/L in the Modder River above Rustfontein Dam, is lower than the 100 µg/L NO_3-N as determined for unpolluted world rivers (Webb & Walling, 1992). During a study conducted during 1996 - 1997 (Koning, 1998), the average NO_3-N concentration in the Modder River was 230 µg/L. This average concentration was determined at five sampling sites downstream from Rustfontein Dam, whereas the samples taken during this study was taken above Rustfontein Dam. Our results suggest that Botshabelo (a big city and informal settlement) and the city of Bloemfontein itself, probably pollute the Modder River downstream from Rustfontein Dam. This downstream pollution was also illustrated by the lower average conductivity found in the Modder River during 2000 – 2001 compared to the measurements taken from 1996 – 1997. The NO_3-N concentration in the Caledon River was much higher than 100 µg/L (being 322 µg/L). The potential sources could be agricultural run-off from Lesotho, the geological formations underlying the river, as well as sewage and industrial pollution from Maseru, the capital of Lesotho.

In lakes, a concentration of less than 100 µg/L NO_3-N may limit growth, while levels above 400 µg/L would not (Horne & Goldman, 1994). However, the South African Water Quality Guidelines considers inorganic N concentrations below 500 µg/L to be sufficiently low to limit eutrophication and reduce the likelihood of nuisance growths of cyanobacteria (blue-green algae) and other plants. The guidelines also classify freshwater bodies according to the average summer N concentrations (Table 5.13):

Table 5.13: Classification of trophic status of freshwater bodies according to inorganic N concentration (DWAF, 1996a)

Average summer N concentration (µg/L)	Trophic status
<500	Oligotrophic
500 – 2500	Mesotrophic
2500 - 10000	Eutrophic
> 10000	Hypertrophic

Comparing these values with NO_3-N concentrations throughout the world, and those given by Wetzel (1983) (Table 5.14), the concentrations proposed by the DWAF are considered to be unrealistically high.

Table 5.14: Classification of trophic status of freshwater bodies according to inorganic N concentration (Wetzel, 1983)

Inorganic N concentration (µg/L)	Trophic status
<200	Ultra-oligotrophic
200-400	Oligo-mesotrophic
300-650	Meso-eutrophic
500-1500	Eutrophic
>1500	Hypereutrophic

According to Wetzel (1983), the Modder River and Rustfontein Dam are both ultra-oligtotrophic in terms of average inorganic nitrogen concentrations, while Knellpoort Dam can be classified as oligo-mesotrophic and the Caledon River as meso-eutrophic. The classification by Wetzel (1983) also gives a much narrower concentration range than DWAF (1996a) to determine the trophic status, and is considered to be a more accurate indication of the trophic status of freshwaters.

Rustfontein Dam, Knellpoort Dam and the Modder River have average summer NO_3-N concentrations of below 100 µg/L, lower than the maximum suggested by Horne & Goldman (1994) (being 100 µg/L) and the DWAF (1996a) (being 500 µg/L) to limit algal growth. When the NH_4-N concentrations are added, the average total inorganic N in Knellpoort Dam rises above the critical concentration suggested by Horne & Goldman (1994) to limit algal growth. In the Caledon River, the total inorganic N increases above 500 µg/L, the concentration suggested by DWAF (1996a) that would be the transition between oligotrophic and mesotrophic.

In Knellpoort Dam, the high average NH_4-N concentration in summer can possibly be ascribed to ammonification at the bottom of the dam resulting in NH_4-N concentrations to increase as high as 500 µg/L. Omitting these values, the annual average NH_4-N concentration in Knellpoort Dam, was only 3.45 µg/L. The average values of the total dissolved nitrogen (NO_3-N and NH_4-N) used to determine the trophic status by Ignatiades & co-workers (1992) were 1.68 µg/L in eutrophic waters, 1.17 µg/L in mesotrophic waters and 0.7 µg/L in oligotrophic waters. Based on the NH_4-N concentration in Knellpoort Dam and the values proposed by Ignatiades & co-workers (1992), the system can then be considered to be eutrophic.

As discussed earlier, NH_4-N concentrations increased in the impoundment during the summer, due to the decomposition of organic material, but were mostly low during the rest of the study period. Considering the transfer of water between the two systems, the lower NO_3-N concentration in Knellpoort Dam (66.5 µg/L) could decrease the NO_3-N concentration in the Modder River (94 µg/L) during the transfer of large volumes of water for extended periods, while, for most of the time, NH_4-N is not expected to play a significant role because of the low concentrations present.

5.4.7) Seasonal variations in N and P

The seasonal variation in nutrient concentrations is in accordance with Roos & Pieterse (1995a), who found that in the Vaal River, most of the annual load of soluble nutrients, such as phosphorus and nitrogen, are transported during periods of high discharge following rainfall. This phenomenon was also illustrated in other river systems, such as the Humber River, England (House *et al.*, 1997), the Barwon-Darling River, Australia (Bowling & Baker, 1996), the Swan River, Australia (Thompson & Hosja, 1996), the River Spercheios, Greece (Kormas,

1999) and the Upper Orange River, South Africa (Keulder, 1979). Two mechanisms may be involved in the high concentration of PO_4-P after rains, namely percolation and erosion (Golterman, 1975). Percolation is a source of substantial quantities of P, probably due to wash-out from fertilised agricultural soils. Erosion occurs where the soil is geologically unstable, or plant cover is sparse or absent, as in the case of Lesotho, through which the Caledon River flows. Since erosion represents the removal of solid materials, such as clay particles, the phosphate adsorbed to the particles is carried off as bounded (Horne & Goldman, 1994). The phosphate transported during erosion, is usually much lower than transported through percolation, but this is probably not the case in the Caledon River, where erosion plays a very important role.

The importance of the seasonal variations in nutrient concentrations and the potential role it can play in the quality of transferred waters between the two systems, formed this basis of this investigation. Thus, if water is being transferred during periods of high flow in the Caledon River (which is usually the case), higher concentrations of nutrients will be transferred to Knellpoort Dam. Consequently, water will be transferred to the Modder River during periods of low flow in the river (when there is a need for potable water during drier periods), causing a large dilution in the river due to the generally nutrient-poor waters of Knellpoort Dam.

5.4.8) Spatial variations in SiO_2-Si

Compared to the levels of other nutrients, such as nitrogen and phosphorus, the SiO_2-Si concentrations are little influenced by human impacts. A shortage of silica relative to the other nutrients has major consequences in anthropogenic lake eutrophication. Over-enrichment with N and P nutrients alters the biogeochemical cycle of silica allowing for a dissolved silicate limitation to occur on a more frequent basis. These changes in dissolved silicate availability cause the replacement of diatoms by other algae that do not require silica for growth, such as blue-green algae (Conley, 2000; Friedl *et al*., 2004).

The reason for the high concentrations present in the Knellpoort Dam, is the high sediment loads, which sink to the bottom where it accumulates. Diatoms were the dominant algae in Knellpoort Dam for a significant part of the study period, thus an increase in SiO_2-Si due to the transfer of water could play an important role in providing favourable conditions for the growth of these algae. One species that were often present, *Alaucosira granulata*, is an algae that could cause filter- and screen-clogging in water purification works (Carter-Lund & Lund, 1995). Since Knellpoort Dam was built to augment the supply of potable water for Bloemfontein, the presence of this algal species could pose a problem when the water has to be purified.

5.5) CONCLUSIONS

The transfer of water from the Caledon River, via Knellpoort Dam, to the Modder River caused an increase in the concentrations of TP, PO_4-P, NO_3-N and SiO_2-Si in Knellpoort Dam, due to the nutrient-rich waters of the Caledon River. Since the effect of the inflowing water seems to be localised to the area at the inflow channel, the water quality at the *Novo* pump station (on the opposite side of the impoundment) was little influenced. However, if transfer takes place over a long period of time, it can have a significant impact on the nutrient concentrations of the impoundment. Similarly, if the ionic composition changes over a long period of water transfer, the algal communities in the different systems can change.

Diurnal patterns of nutrient concentrations in Knellpoort Dam will also be influenced by the transfer of water, These changes will, however, be limited to the period of transfer, and is not expected to have long-term effects on the nutrient cycles and patterns, except for increasing the average concentrations. In addition, a decrease in the nutrient concentrations in the Modder River is expected when water is transferred from the *Novo* pump station, because of the nutrient-poor waters of Knellpoort Dam at the abstraction point to the Modder River.

The increase or decrease in nutrient concentrations in the impoundments and rivers of the *Novo* Transfer system will be dependent on the volumes of water being transferred, the nutrient concentrations of the transferred water (nutrient load), the time of year (which season) of transfer and flow conditions in the rivers. Together with light availability and favourable temperatures, phosphorus, nitrogen and silica are important nutrients required for algal growth. Because of this, the increase or decrease in the concentrations of these nutrients through the transfer of water, could have an important impact on algal growth and composition in these systems. Furthermore, a low N:P ratio (that can develop as the nutrient composition of Knellpoort Dam changes) could give rise to the development of blue-green algae. This is further discussed in Chapter 6.

Generally the nutrient status of the two river systems is oligotrophic to mesotrophic. There are indications that pollution of the Caledon River occurs. However, it is unlikely that this will have an effect on the trophic status of the Modder River. More importantly for the Modder River is development in its catchment, the expansion of the urban areas and increased water demands.

CHAPTER 6

THE PHYTOPLANKTON IN THE VARIOUS WATERS OF THE *NOVO* TRANSFER SCHEME

6.1) INTRODUCTION

6.1.1) General introduction

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

The physiological and biochemical properties of algae are dependent on external conditions (Kozitskaya, 1990) and through their adaptive strategies, they can survive in dynamic environments (Köhler, 1994). This is illustrated by the fact that diatoms even occur in fast-flowing rivers. Several of these variables that can have an influence in the *Novo* Transfer system, are discussed in the following sections.

6.1.2.) Factors influencing algal growth

6.1.2(a) <u>Flow</u>

One of the major problems that algae experience in rivers, opposed to the relatively stable environment in lentic waters, is the persistent and unidirectional passage of water (Hynes, 1970; Golterman, 1975). Variability in flow may lead to rapid movement or suspension of material with an increase in turbidity and increased attenuation of light. Consequently, periods of relatively stable environmental conditions (necessary for the achievement of an equilibrated steady-state), are much shorter than in lakes, due to shifts in river morphometry, velocity and in trophic interactions along a river (Vannote *et al*., 1980). Descy (1993) and Dokulil (1994) also stated that flood episodes are major disturbances in otherwise continuously mixed environments (rivers), affecting the composition and biomass of the plankton. It was found that some algae require turbulent conditions for optimal growth (Hynes, 1970), thus illustrating their evolutionary adaptive strategies. Algal development can also be influenced by stream invertebrate grazers as they can efficiently crop algae growing on stones and rocks (Iversen *et al.,* 1991).

In lakes, water movement transports phytoplankton from a high-light, low-nutrient, low predator environment near the surface to the dark, nutrient-laden deeper waters, which contain numerous predators (Horne & Goldman, 1994). The plankton dynamics in reservoirs are particularly influenced by the effects of greater flow-through and turbulence, the nutrient loading the reservoir normally receive, the plankton populations of the inflows relative to those in the reservoir itself, as well as the management controls applied (Thornton *et al.,* 1992).

6.1.2(b) Light

Different algal genera have different light requirements. For instance, many diatoms appear to be fairly indifferent to light, while many Chlorophycaceae require fairly high intensities (Hynes, 1970). Underwater light attenuation in turbid systems is largely a function of suspended particle concentration and size (Dokulil, 1994). Consequently, Secchi depth, euphotic zone and the extinction coefficient can all be related to the quantity of suspended solids.

The Caledon River is a very turbid system and the influence of the turbidity on the water of the Knellpoort Dam, was already discussed in Chapter 3. Thus, in the *Novo* Transfer Scheme, turbidity is expected to play an important role in influencing algal growth and succession. In turbid waters, the spectral quality of the available light is different from that reaching the surface. In very turbid waters, the ratio of the euphotic zone: aphotic zone is very small and the production profile is compressed and limited to the surface waters of water column. The closer the ratio of the euphotic zone: aphotic zone is to 1, the more favourable the light availability.

Grobbelaar (1990) found that in turbid systems the ratio of euphotic to aphotic depth is usually small and mixing may exceed the compensation depth. A model was developed for predicting algal growth in natural systems. In order to compensate for various light regimes to which the phytoplankton would be subjected, the losses due to dark respiration were taken into account as well as the changes in efficiency in light utilisation. The model resulted in different production profiles being generated for the same surface conditions, but with different mixing depths, where the phytoplankton become more efficient as the light regime deteriorates. It was also found that the "critical mixing depth" is approximately 2.5 times greater that previously accepted, being 20 times the euphotic depth. Grobbelaar (1992) also found that the ratio of euphotic to mixing depth is the most important factor affecting overall productivity in two reservoirs in the Modder River (Mockes and Krugersdrift Dams), and that nutrients are of secondary importance only. Based on these studies, it must be taken into account that the mixing depth in the systems could also play a significant role in phytoplankton productivity.

6.1.2(c) Nutrients

Marine and freshwater phytoplankton have similar growth requirements for the major nutrients: C, N, and P, although sources and ratios of these nutrients may differ between lentic and lotic systems, and the same physiological processes are involved in their use by algae, and therefore the same theoretical considerations underlie their ecological analyses (Tett *et al.,* 1985).

Research has shown that more than one nutrient resource can limit the growth rates of phytoplankton. Liebig's law of the minimum states that only a single growth factor can be limiting at any given time. In other words, the nutrient whose cell quota is lowest relative to need is the one that controls growth at that particular time (Droop, 1974). Since different organisms in a population could also be limited by different limiting resources, it could be necessary to identify the limiting factors for the various populations of a community (Melack, 1995).

Important concepts to note in order to understand the interactions between algal growth and nutrient dynamics are the *optimum nutrient ratio* and the *nutrient loading* concept. The *optimum nutrient ratio* is the ratio at which a transition from one nutrient limitation to another occurs (both could thus be limiting), or where the resource requirement is such that the resource is not in short supply relative to another (Rhee & Gotham, 1980). The *nutrient loading* concept states that a quantifiable relationship exists between the quantity of nutrients reaching a water body and its trophic degree (measurable with some kind of trophic scale index) (Vollenweider, 1976). For example, chlorophyll *a* could be predicted from the loading characteristics of lakes, by relating the phosphorus concentration to loading, using the following equation (Vollenweider, 1976):

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

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

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

(L_p/q_s) = average inflow concentration

z = mean depth

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

In rivers and lakes, factors other than nutrient concentrations can influence algal physiology, and determine the relative growth rates and structure of the communities. Although P, N and C are the key nutrients for most algae, the availability of a fourth nutrient may limit the growth of a particular species e.g. the availability of silica can limit the growth of diatoms (Kilham, 1971; Grobbelaar & House, 1995). Other important aspects complicating nutrient and algal interactions include the interactive effects between nutrients and trace components (e.g. vitamins or trace metals), as well as the different chemical forms of N and P (Grobbelaar & House, 1995). Essential trace elements can, under certain circumstances, be limiting for growth and development of algae with special requirements (Rósen, 1981). Additional factors that may influence algal growth are: alkalinity, conductivity, pH, the substratum, scouring and grazing by animals (Hynes, 1970). For example, Rósen (1981) stated that when an algae exhibits an affinity for high levels of total phosphorus (> 20 µg/L), there are indications that these algae adapt to high levels of nitrogen (> 600 - 1 000 µg/L), pH values lower than 7, conductivity values higher than 100 mS/cm and a low transparency (less than 2m). In general, high conductivity values as well as high nutrient levels indicate that most organic and inorganic nutrients are present in excessive quantities. Algal growth is thus a function of a combination of different factors.

In this study, the seasonal and spatial variations in algal growth, together with the different factors that influence it in the *Novo* Transfer system were determined. In addition, the effect of transferring water from the Caledon River to Knellpoort Dam, and hence to the Modder River on algal growth, was taken into consideration.

The potential bioavailability of nutrients adsorbed onto the sediment particles of the turbid waters of the Caledon River, was also briefly investigated, and was reported on in Chapter 4.

6.2) MATERIAL AND METHODS

6.2.1) Algal identification

Sub-surface samples of 2 litres were taken of which 100 ml samples were fixed with 2% formaldehyde for algal identification. Applying pressure to cells can damage the gas vacuoles of the cyanobacteria, which causes the cyanobacteria to settle out at the bottom of the container for countings. Hitting a rubber stopper with a rubber hammer on a container filled with sample water, generated this pressure. After this, 6 ml of the water were extracted and poured into Utermöhl counting chambers, and covered with a cover-glass slip. The samples were sometimes diluted, depending on the turbidity of the sampled water. A settling time of two days (48 hours) was allowed for each sample, before enumeration was done.

A Zeiss inverted light microscope was used to identify the dominant species and 20 blocks of known dimensions were counted. These results were then multiplied by a factor to obtain the total counts per millilitre. Algal family dominance was determined as a percentage of the total algal community.

6.2.2) Chlorophyll *a* determination

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

Chlorophyll *a* in extract (µg//L) = $(A_{665} - A_{665}a) \times 28.66$

where,

A665 = absorbance of ethanol extract at 665 nm before it was acidified minus absorbance at 750 nm

A665a = absorbance of the acidified ethanol extract at 665 nm minus the absorbance at 750 nm.

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

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

6.2.3) Primary productivity

Primary production was measured *in situ* on two occasions during the study period (November 2000 and October 2001) in Knellpoort Dam. The main purpose was to determine possible influences of the inflow of turbid water into Knellpoort Dam. The first measurements were during the transfer of water, and the second 5 months thereafter.

Since the water column was mixed an integrated water sample was used for the measurements.

Two ampoules, each containing 10µCi as $NaH^{14}CO_3$ were added to a plastic container that contained 1.5 litres of the integrated water sample. 100 ml Round bottom flasks (DURAN-Schott), with a standard ground socket and plastic stoppers were used as incubation flasks**.**

One darkened and two translucent (light) bottles (in horizontal positions per pre-selected depth) were suspended on an aluminium rod which floats at a point near to the dam wall of the impoundment (where turbidity was high) and in the middle of the impoundment. The dark bottles were prepared by painting the bottles black and then covering them with aluminium foil. Click-on metal clamps on an aluminium disk were used as a fixture for the incubation bottles. The aluminium disks were arranged to include incubation at the surface (0 cm), 12.5 cm, 25 cm, 50 cm, 100 cm and 150 cm. The incubation depths during October 2001 were at the surface (0 cm), 25 cm, 50 cm, 100 cm, 150 cm and 200 cm. The difference in incubation depths was because the photic depth was much deeper due to the lower turbidity in the impoundment during October 2001.

Incubation times of 4 hours were used, between 10h00 and 14h00. After incubation the bottles were stored in a darkened wooden box (to prevent exposure to light) and were immediately processed, using the acidification and bubbling method (Schindler *et al.,* 1972).

After the incubation period, 10 ml was extracted from each incubation bottle and added to a scintillation vial. Then 0.5 ml of a 0.4 N HCI solution was added and the scintillation vials were put into a bubbling-apparatus for 5 minutes to strip the inorganic $^{14}CO_2$ from the water samples. After bubbling, 10 ml of a liquid scintillation solution, Insta Gel (a xylene-based cocktail manufactured by Packard) was added to each sample as well as 0.5 ml of a carbon dioxide absorber "Carbosorb". The samples were then stored in the dark before measuring the dpm's in a Beckmann Liquid Scintillation Counter.

The total cumulative uptake of ^{14}C during incubation was determined and normalised to incubation time to give an hourly rate, by using the equation according to Vollenweider (1969a):

$$P_z(mgC\,m^{-3}\,h^{-1}) = \frac{^{14}C_{ass} \times {}^{12}C_{avail}}{^{14}C_{added}} \times K_1 \times K_2 \times K_3 \times K_4$$

$$P_z = \frac{^{14}C_{ass} \text{ X } {}^{12}C_{avail}}{^{14}C_{added}} \text{ x } K_1 \text{ x } K_2 \text{ x } K_3 \text{ x } K_4$$

Where:

P_z = hourly rate (time integrated average) of total carbon assimilated (photosynthesis) at depth z

$^{14}C_{ass}$ = average disintegrations per minute (DPM) of aliquots from 2 light bottles minus DPM in dark bottle.

$^{14}C_{added}$ = total^{14}C (DPM) available for assimilation

$^{12}C_{avail}$ = total DIC available for photosynthesis (mg/L). Calculated from total alkalinity, pH and temperature values.

K_1 = a conversion factor for the aliquots

K_2 = isotopic factor (= 1.06); the isotope discrimination factor corrects for the fact that ^{14}C, being heavier than ^{12}C, is taken up more slowly

K_3 = a dimension factor to convert mg/L to mg/m3 (x1000)

K_4 = a time factor. If K4 is not used in the formula, then P_z will be equal to mg m^{-3} h^{-1}

(where x = incubation time).

The maximum light saturated chlorophyll-specific photosynthetic rate (or assimilation number) (P^B_{max}, in mgC.mgChlorophyll-*a*$^{-1}$.h^{-1}), was determined by dividing the maximum rate of photosynthesis at any depth in the profile (P_{max}, in mgC.m^{-3}.h^{-1}) with the average chlorophyll *a* concentration in the euphotic zone (Harris, 1980).

6.3) RESULTS

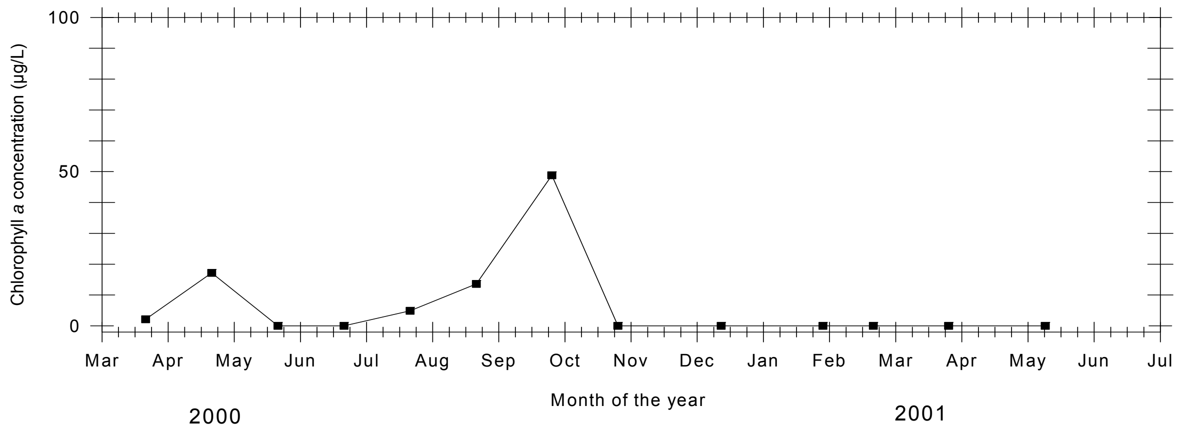
6.3.1) Phytoplankton seasonality and spatial distribution

Figures 6.1 (a) – (d) show the seasonal variation in the chlorophyll *a* concentrations at the sampling sites in the *Novo* Transfer Scheme. There was an increase in chlorophyll *a* in the Caledon River during April and September. During April the algal population was dominated by a *Cyclotella* sp. and during the September (spring) increase it was dominated by *Stephanodiscus hantzhii* and various Chlorophyte species.

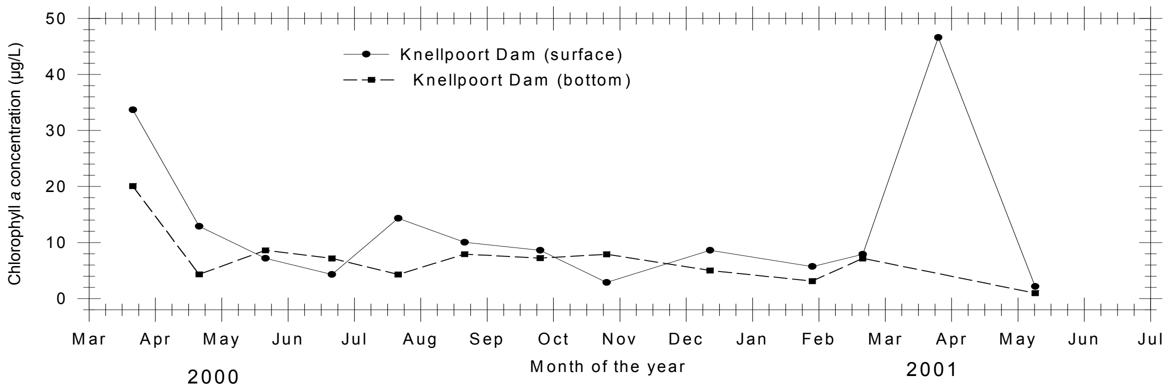
In Knellpoort Dam there was an increase in chlorophyll *a* during March 2000 and March 2001. The algal population was dominated by a *Chlorella* sp. during 2000 and by *Anabaena circinales* (900 cells/ml) during 2001. In Rustfontein Dam, there was a marked increase during September, dominated by *Microcystis aeruginosa* and an *Oocystis* species.

In the Modder River, chlorophyll *a* increased during May and again during January. The chlorophyll *a* increase in May was caused by a bloom of *Microcystis aeruginosa* (up to 36 000 cells/ml), while the increase during January was caused by *Anabaena circinales* (805 cells/ml) and *Microcystis aeruginosa* (146 cells/ml). Thus, in the Modder River, the increase in chlorophyll *a* during the study period was mainly due to cyanobacteria.

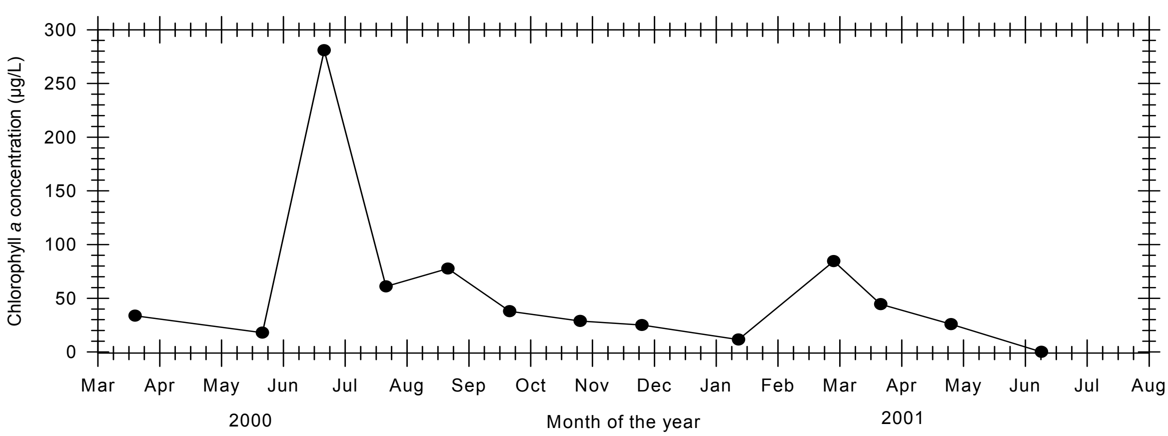
6.1(a) Caledon River



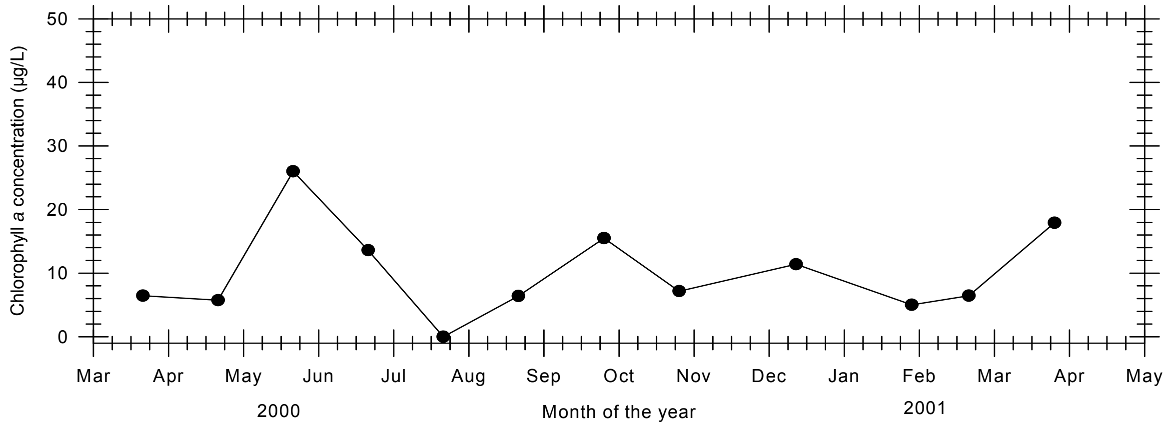
6.1(b) Knellpoort Dam



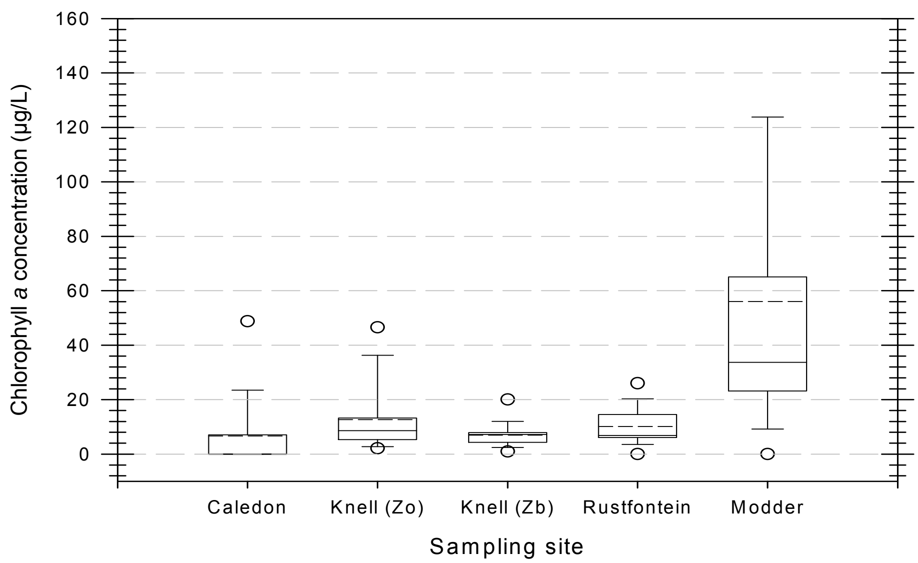
6.1(c) Modder River



6.1(d) Rustfontein Dam



<u>Figure 6.1</u><u>: Seasonal variation in chlorophyll *a* in the *Novo* Transfer scheme.</u>



<u>Figure 6.2</u><u>: A box plot summary of the spatial variations in chlorophyll *a* at the sampling sites during the study period.</u>

The average chlorophyll *a* concentrations were low in the Caledon River, Knellpoort and Rustfontein Dams throughout the study period, compared to the significantly higher concentrations in the Modder River (Figure 6.2). The average chlorophyll *a* concentration in the Caledon River was 6.7 µg/L, in Knellpoort Dam (surface of the water column) it was 12.7 µg/L, in the Modder River 56 µg/L and in Rustfontein Dam it was 10 µg/L.

6.3.2) Algal dominance and seasonal succession

In the Caledon and Modder Rivers, and in Knellpoort and Rustfontein Dams, the phytoplankton community was represented by Euglenophyceae, Cyanophyceae, Bacillariophyceae, Chlorophyceae and Cryptophyceae. No Dinophyceae were present in any of the system during the study period. The specific dominant algal species were also determined and is detailed in Tables 6.3 – 6.6.

These phytoplankton families showed patterns of seasonal succession in both the rivers and the impoundments of the *Novo* Transfer scheme (Figures 6.3 - 6.6). Please note that the plots are not plotted incrementally, and that each percentage was plotted separately. Where there is no surface area plot, no algal species were found in the sample from that specific month.

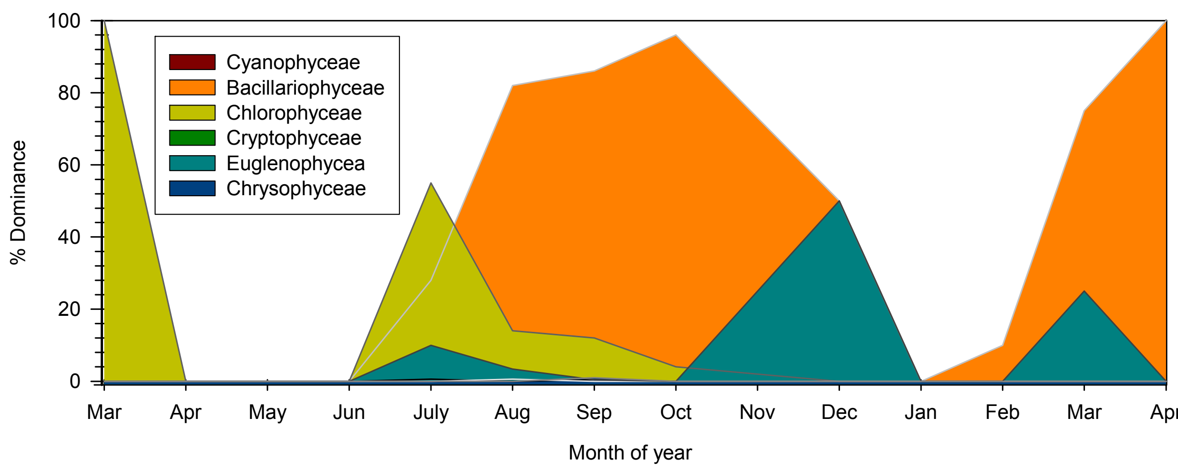


Figure 6.3: Algal dominance and succession in the Caledon River.

In the Caledon River (Figure 6.3), the family that dominated the algal population during winter, spring and summer, was the Bacillariophyceae. At the start of the winter, 50% of the population was dominated by Chlorophyceae. During the end of spring and the start of summer, Euglenophyceae showed a dominance of 45% and 30% respectively. The Cyanophyceae never showed any dominance during the study period. The order of dominance was (starting autumn 2000): Chlorophyceae - Bacillariophyceae -Euglenophyceae - Bacillariophyceae - Euglenophyceae.

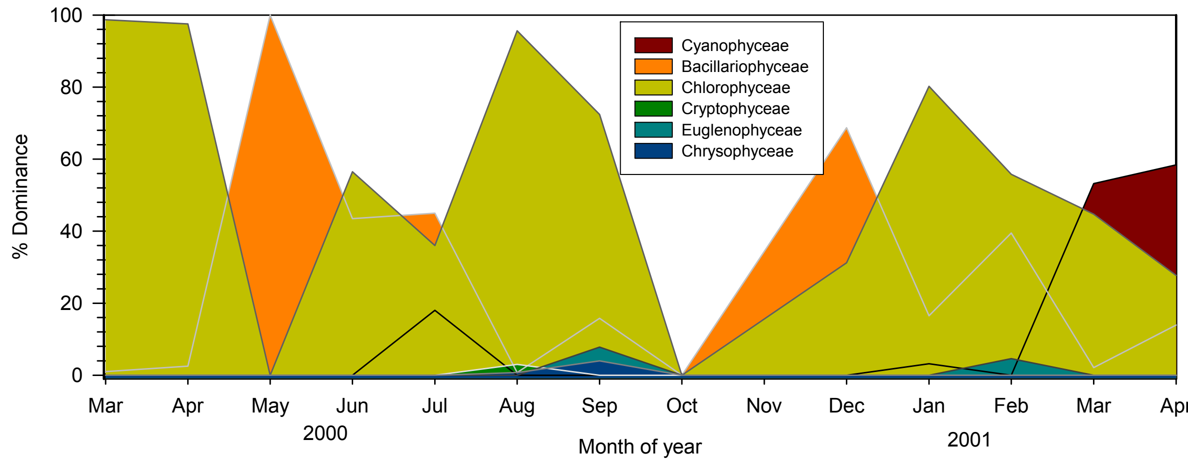


Figure 6.4: Algal dominance and succession in Knellpoort Dam (surface sample).

In Knellpoort Dam (Figure 6.4), the Chlorophyceae dominated the algal population (about 100% dominance) throughout autumn, the end of winter and the onset of spring. As soon as the dominance of Chlorophyceae decreased, Bacillariophyceae started to dominate (onset of winter and end of spring). During the summer of 2001, the Cyanophyceae family showed a dominance of 50%, but limited numbers (if any) were present during the earlier part of the study. The Euglenophyceae never showed any dominance. The order of dominance was (starting autumn 2000): Chlorophyceae - Bacillariophyceae - Chlorophyceae - Bacillariophyceae - Chlorophyceae - Cyanophyceae. Thus, it seems as if Cyanophyceae replaced Bacillariophyceae as the dominant family during 2001.

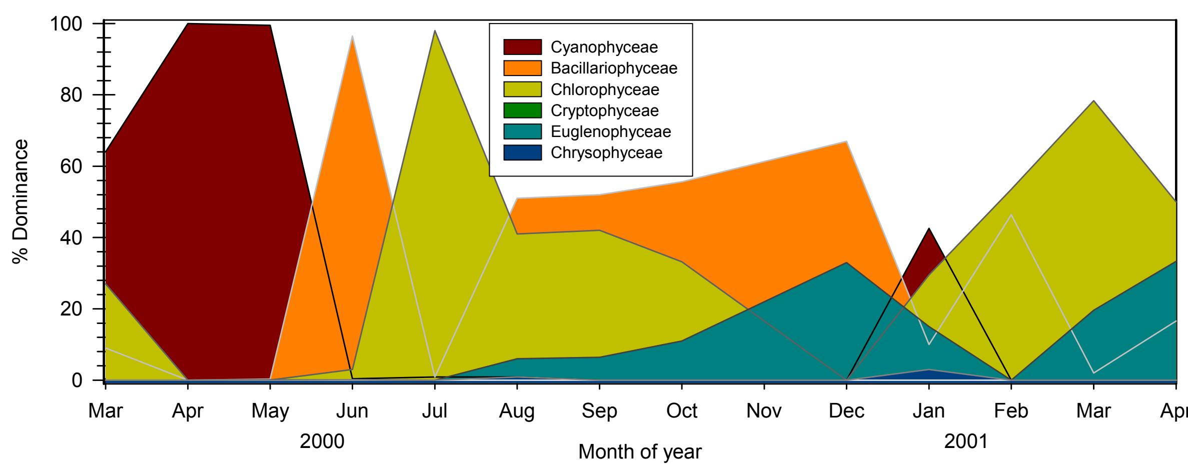


Figure 6.5: Algal dominance and succession in the Modder River.

In the Modder River (Figure 6.5), the Bacillariophyceae dominated (sometimes as high as 100%) during the end of autumn, the end of winter and during both the onset and end of summer. As in the case of Knellpoort Dam, the Chlorophyceae started to dominate the population when the Bacillariophyceae dominance decreased (at the start of winter and mid-summer). Cyanophyceae dominated during autumn and summer, while Euglenophyceae showed a slight dominance during spring and start of summer together with the Bacillariophyceae. The order of dominance was (starting autumn 2000): Cyanophyceae - Bacillariophyceae - Chlorophyceae - Bacillariophyceae - Euglenophyceae - Bacillariophyceae - Cyanophyceae - Chlorophyceae and Bacillariophyceae - Euglenophyceae.

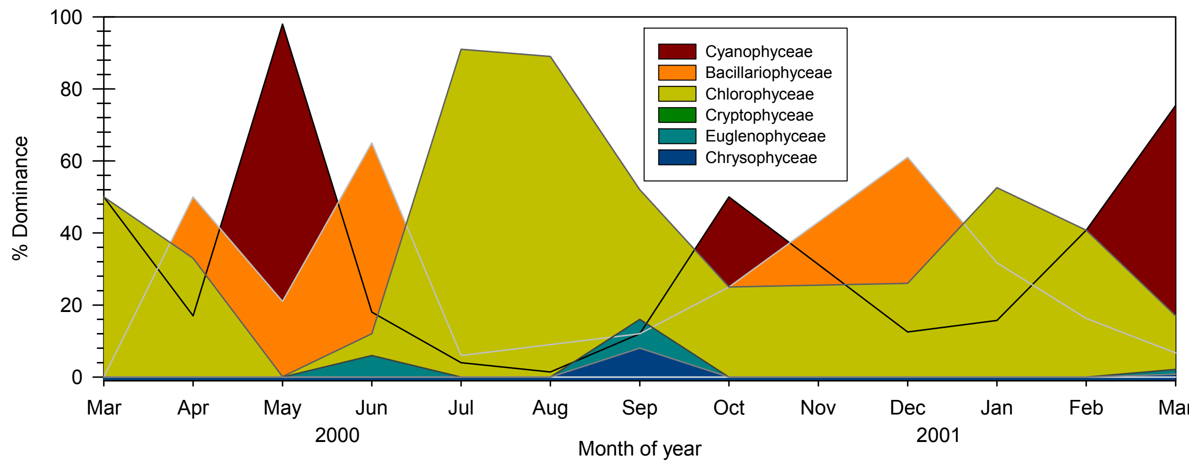


Figure 6.6: Algal dominance and succession in Rustfontein Dam.

In Rustfontein Dam (Figure 6.6) the Bacillariophyceae dominated the algal population during the end of autumn and end of spring. The Cyanophyceae dominated during the end of autumn, spring and summer. The Chlorophyceae dominated during winter and summer. The Euglenophyceae never showed any dominance. The order of dominance was (starting autumn 2000): Bacillariophyceae - Cyanophyceae - Bacillariophyceae - Chlorophyceae - Cyanophyceae - Bacillariophyceae - Chlorophyceae - Cyanophyceae.

Chrysophytes occurred only during September in the Caledon River, Knellpoort Dam and Rustfontein Dam. However, in the Modder River, Chrysophytes also occurred during June and December, but this algal family never showed any dominance.

Considering the algal families present in the different waters of the *Novo* Transfer Scheme, the specific factors that caused a family to dominate, were determined. The domination of specific algal genera sometimes occurred over a period of more than one month. For simplification, only the month with the highest percentage dominance during the period of dominance (which was higher than 30%), was used to illustrate genera dominance (Tables 6.1 - 6.4).

Table 6.1: Seasonal variation in specific algal dominance in the Caledon River

% Dominance	Month	Season	Specific genera
Bacillariophyceae			
96	October	Spring	*Alaucosira* Various pennate diatoms
100	February	Summer	Various pennate diatoms
Chlorophyceae			
100	March	Autumn	*Coelastrum*
55	July	Winter	*Chlorella,* *Chlamydomonas,* *Oocystis,* *Ankistrodesmus*
Euglenophyceae			
50	December	Summer	*Lepocinclis*

Table 6.2: Seasonal variation in specific algal dominance in the Modder River

% Dominance	Month	Season	Specific genera
Bacillariophyceae			
96.5	June	Winter	*Stephanodiscus hantzhii* *Nitzschia sp.*
67	December	Summer	*Alaucosira* Various pennate diatoms
56	October	Spring	*Stephanodiscus* *Cyclotella*
Chlorophyceae			
98	July	Winter	*Chlamydomonas* *Chlorella*
42	September	Spring	*Chlamydomonas* *Chlorella* *Chlorococcum, Scenedesmus*
78.4	March	Autumn	*Chlamydomonas*
			Chlorella

% Dominance	Month	Season	Specific genera
Cyanophyceae			
99.5	May	Winter	*Microcystis*
42.6	January	Summer	*Microcystis*
Euglenophyceae			
33	December	Summer	*Lepocinclis*
33	April	Autumn	*Ttrachelomonas*

Table 6.3: The seasonal variation of specific algal dominance in Knellpoort Dam

% Dominance	Month	Season	Specific genera
Bacillariophyceae			
100	May	Winter	*Alaucosira* *Stephanodiscus* Various pennate diatoms
45	July	Winter	*Alaucosira* *Nitszchia* Various pennate diatoms *Stephanodiscus hantzhii*
68.7	December	Summer	*Alaucosira* Various pennate diatoms
39.5	February	Summer	*Cyclotella*
Chlorophyceae			
98.7	March	Autumn	*Chlorella* *Carteria*
56.5	June	Winter	*Chlamydomonas* *Chlorella* *Chlorococcum*
95.6	August	Winter	*Carteria* *Chlorella*
80.2	January	Summer	*Chlorella* *Staurastrum* *Oocystis* *Sphaerocystis* *Chlamydomonas*

% Dominance	Month	Season	Specific genera
Cyanophyceae			
58.4	March-April	Autumn	*Anabaena circinales*

Table 6.4: The seasonal variation in specific algal dominance in Rustfontein Dam

% Dominance	Month	Season	Specific genera
Bacillariophyceae			
50	April	Autumn	*Melosira* *Cyclotella*
65	June	Winter	*Cyclotella* *Melosira* *Stephanodiscus*
61	December	Summer	*Melosira* *Stephanodiscus* *Cyclotella* Pennate diatoms
Chlorophyceae			
33	April	Autumn	*Ulothrix*
91	July	Winter	*Chlorococcum* *Chlorella* *Oocystis*
52.6	January	Summer	*Oocystis* *Coelastrum* *Chlamydomomonas*
Cyanophyceae			
98	May	Winter	*Microcystis*
50	October	Spring	*Apatococcus*
75.4	March	Autumn	*Microcystis* *Anabaena*

6.3.3) Primary productivity

6.3.3(a) Primary productivity vs. light intensity

The availability of light has a major impact on the dynamics and structure of most aquatic and terrestrial communities, together with advective and dispersive processes that control the transport of planktonic organisms. Consequently, light sometimes becomes the limiting factor, instead of nutrients (Huisman *et al.,* 1999). Because of this (and with reference to Chapter 3 that illustrated a significant increase in turbidity in Knellpoort Dam with a transfer of water) it was expected that the transfer of turbid water will have an influence on the phytoplankton productivity in Knellpoort Dam.

Figure 6.7 shows the primary productivity profiles in Knellpoort Dam on 22 November 2000 (during the transfer).

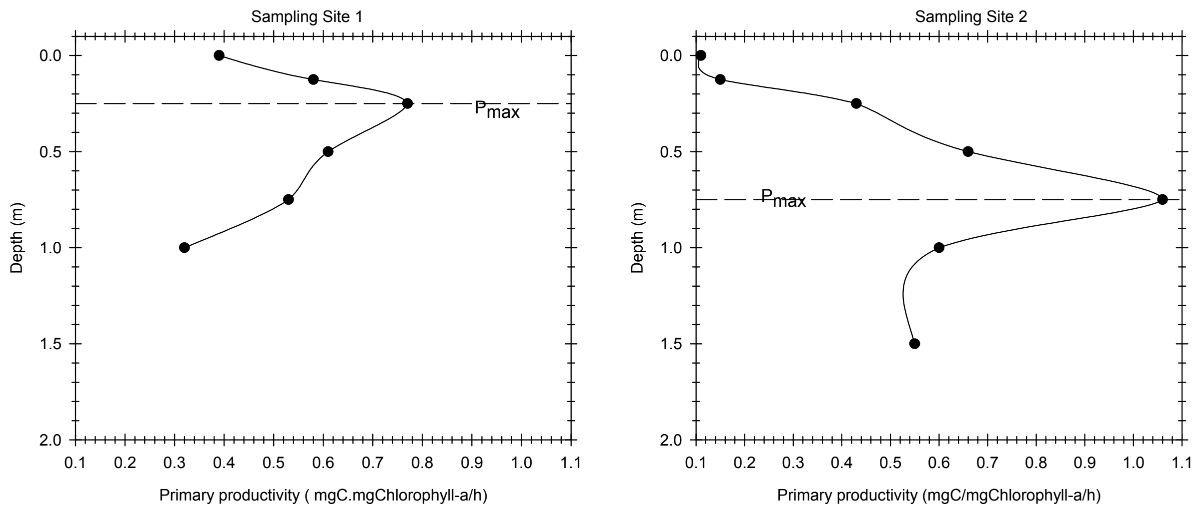


Figure 6.7: The productivity profiles at Sampling Sites 1 and 2 in Knellpoort Dam during the transfer of water (November 2000).

At sampling site 1, during the transfer (turbid conditions), the light saturated rate of photosynthesis (P_{max}) was calculated as 9.94 mgC .m^{-3}.h^{-1}, and the photosynthetic capacity (P^B_{max}) was calculated to be 0.77 mgC.mgchl-a^{-1}.h^{-1}, while at site 2 the light saturated rate was 13.65 mg C.m^{-3}.h^{-1} and the photosynthetic capacity was 1.05 mgC.mgchl-a^{-1}.h^{-1}. The integral rate of photosynthesis for sampling site 1 was 7.3 mgC.m^{-2}.h^{-1} and at site 2 it was 11.6 mgC.m^{-2}.h^{-1} (37% higher). Furthermore, at sampling site 1, the surface inhibition was calculated to be 49%, while at sampling site 2 it was 89.7%. The increase in surface inhibition can be ascribed to the lower turbidity at sampling site 2, and can also clearly be seen in the differences between the two productivity profiles.

The primary productivity profiles showed marked differences between the two sampling sites. The profile was compressed for sampling site 1 (the turbid station), where P_{max} (maximum productivity) occurred at 0.25 m, while P_{max} at the clearer station occurred at 0.75 m. The euphotic zone was measured with the Secchi disk at about 0.65 m for Site 1 and 3.5 m for Site 2.

After the transfer of water was stopped, the P^{B}_{max} at site 1 was 1.31 mg C.mgchl-*a*$^{-1}$.h^{-1} and at site 2, it was 1.46 mg C.m^{-3}.h^{-1}. The light saturated rate of photosynthesis (P_{max}) for sampling site 1 was 7.33 mgC.m^{-3}.h^{-1} at sampling site 2 it was 6.8 mgC.m^{-3}.h^{-1}. Apart from the similar productivity values, the productivity profiles at the two sampling sites were also similar and P_{max} at both sites was measured at approximately 0.5 metres, with no compressed profile observed (Figure 6.8). The euphotic zone measured with the Secchi disk, was approximately 2.45 metres at both the sites.

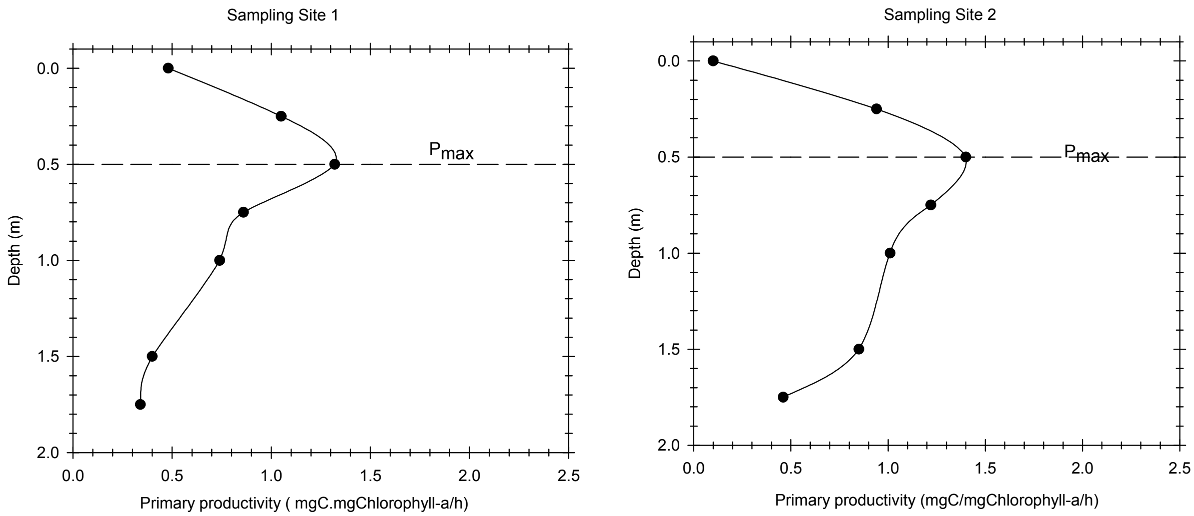


Figure 6.8: The primary productivity profiles at sampling sites 1 and 2 in Knellpoort Dam after the transfer of water (October 2001).

The higher productivity values during November 2000 than October 2001 at sampling site 2 could be due to the higher temperatures as well as the higher concentration in nutrients, which created more favourable conditions for algal growth.

Based on the above, it can be deducted that the transfer of turbid water compresses the light profile in Knellpoort Dam, but that the effects are localised to the area of the impoundment receiving the inflowing water and also limited to the period during transfer, whereafter conditions in the impoundment returned to the same state as before the transfer.

6.3.3(b) The relationship between primary productivity (algal growth) and nutrient concentration

In Chapter 5, the increase in nutrients in Knellpoort Dam, due to the transfer of water from the Caledon River was shown. There was an increase of 44% in the TP concentration, an increase of 100% in the PO_4-P concentration, an increase of 27.5% in NO_3-N and an increase of 92% in the SiO_2-Si concentrations.

The TN:TP ratio in both Knellpoort and Rustfontein Dams were 1.6, in the Caledon River it was 12.5 and in the Modder River 5.1. From this it was inferred that the water was N-limited in all the systems, except the Caledon River, where algal growth was dependent on both N and P concentrations (Sakamoto, 1966). He stated that over the range 10<TN:TP<17 by weight the chlorophyll *a* yield was nearly balanced with respect to both TP and TN, but that chlorophyll *a* was dependent only on TN when TN:TP<10, and only on TP , when the TN:TP>17 (see below for a further discussion).

After investigating the relationship between nutrient concentrations and chlorophyll *a* in the *Novo* Transfer Scheme it was found that there was no statistically significant relationship between TP and chlorophyll *a* in Knellpoort or Rustfontein Dams, although 37% of the chlorophyll *a* in Knellpoort and 67% of the chlorophyll *a* in Rustfontein Dam were correlated with the TP concentration. It was also found that 46% of the chlorophyll *a* concentration in Knellpoort Dam correlated with N and 55% in Rustfontein Dam, but these were statistically insignificant. In the Modder River, a linear relationship was found between TP and chlorophyll *a* ($\log_{10}$[chlorophyll *a*] = 0.67689 ($\log_{10}$TP]). This relationship is shown in Figure 6.9.

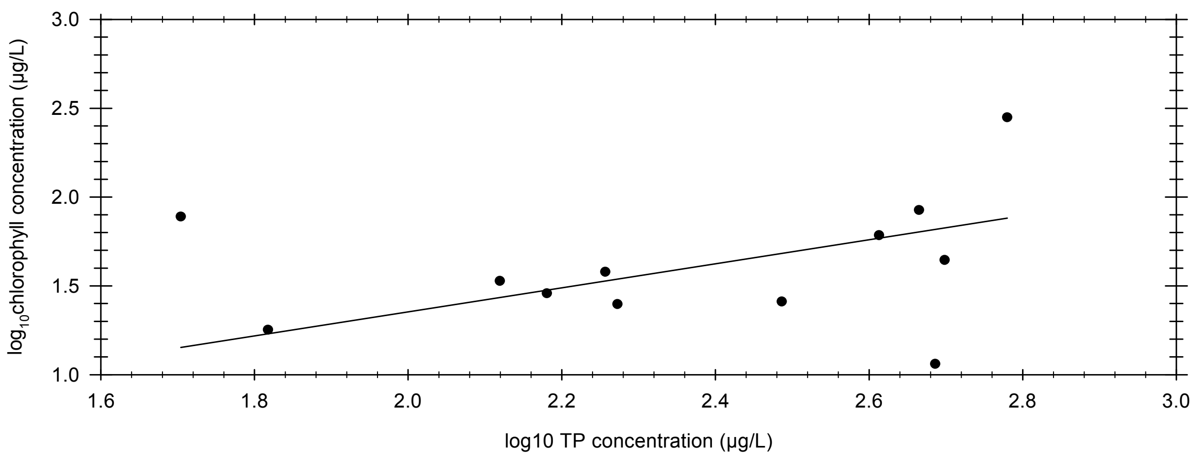


Figure 6.9: Relationship between TP concentration and chlorophyll *a* in the Modder River.

6.4) DISCUSSION

6.4.1) Phytoplankton seasonality and spatial variation

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

Typically, the annual range of total phytoplankton biomass in the southern Hemisphere spans at least one and often three orders of magnitude. Seasonal temperature differences are usually too low to account for this variability at low-latitudes where most southern Hemisphere lakes and reservoirs are located. Instead, the erratic and highly seasonal distribution of rainfall and river inflows, combined with seasonal wind patterns are the cause of the high variability found in these generally shallow water bodies. Consequently, the increase in chlorophyll *a* during April (autumn) in the Modder River can be ascribed to favourable light conditions as well as a decrease in flow rates at the end of the rainy season. During September (spring), flow was also low before the start of the rainy season, thus also stimulating algal growth in the Modder River.

Cyanophyceae occurred frequently at the sampling site in the Modder River. Steinberg & Hartmann (1988) stated that above a threshold of 10 µg/L TP, the development of cyanobacteria can be described by physical factors, such as water column stability. When the turbulence of the water column is rather low, cyanobacteria can build up to dense populations. Cyanobacterial blooms also occur more readily at elevated water temperatures, high pH values and low N:P ratios. The N:P ratio in the Modder River was low during the study period (1.6), and the pH was high during May and January when the blooms occurred ranged between 8.36 and 8.6. Together with sufficient light, it created favourable conditions for cyanobaterial growth. During a study conducted from 1996-1997, no cyanobacterial blooms were observed in the Modder River, but the possibility was predicted if turbulence in the river decreased (Koning, 1998). The flow in the river at the sampling site used during 2000-2001, was less than at the sampling sites used during 1996-1997. Thus, it seems as if nuisance blooms can develop in the Modder River. This was proven by the occurrence of Cyanophyceae in both the Modder River and Rustfontein Dam during this study (2000 - 2001).

The single most important factor that can explain the higher average chlorophyll *a* concentrations in the Modder River, are the higher nutrient concentrations, especially PO_4-P (Table 6.5).

Table 6.5: The different nutrient concentrations at the sampling sites (as taken from Chapter 5).

Sampling site	NH_4-N (µg/L)	NO_3-N (µg/L)	PO_4-P (µg/L)
Caledon River	17.46	322	26.5
Knellpoort Dam	43.28	66.5	13.01
Modder River	12.7	93.9	57.97
Rustfontein dam	9.58	42.8	26.66

Together with high nutrient concentrations, light availability in the Modder River was high in comparison to the Caledon River. The average flow in the Modder River was much lower than in the Caledon River and during low flow conditions, the time the algae spend at a specific site in the river is much longer. It is also possible that, during low flow conditions, the underwater light climate is favourable because the mixing depth becomes less than the euphotic depth and light reaches the river bed.

In Rustfontein Dam, light availability was also higher than in the Caledon River, but the nutrient concentrations were much lower than in the Modder River. This explains the lower chlorophyll *a* concentrations, while in the Caledon River, high flow and turbidity, and thus turbulence and/or light limitation, limited algal growth, despite the fact that nutrient concentrations were high.

In Knellpoort Dam, the average PO_4-P concentration was low, possibly limiting algal growth. However, the transfer of water from the Caledon River to Knellpoort Dam, showed an enrichment of the impoundment (Chapter 5). The cyanobacteria *Anabaena circinales* occurred during March (900 cells/ml) and April (183 cells/ml) 2001 (after transfer), whereas no or very low numbers of Cyanophyceae was observed during the earlier part of the study. Thus, it is possible that the transfer of nutrients into the dam, together with favourable underwater light climate (which was already present), created conditions that favoured the growth of *Anabaena circinales* above other algal species. This is in accordance with a study in the USA, where the cyanobacteria *Anabaena macrospora* and *Oscillatoria limnetica,* increased from <5% to almost 100% in Peter Lake within the first year of enrichment. Cyanobacteria also increased in West Long Lake, from 10% to almost 100% within the first year following enrichment. However, this is not the rule, and a study by Cottingham *et al.* (1998) found no increase in the cyanobacteria, even when the N:P ratio was high due to enrichment.

Although cyanobacteria have slow growth rates, they become dominant because of their low preference by grazers and because of their buoyancy. Buoyancy, due to gas vacuoles, is finely regulated, permitting cyanobacteria to exploit water column variability and to maximise opportunities for light and resource harvesting (Recknagel, 1997). Usually, *Microcystis* sp. increases in cell numbers during cyanobacterial blooms, while *Anabaena circinales* decreases. *Microcystis* can tolerate nitrate deficiency at concentrations of phosphorus sufficiency while *Oscillatoria* depends on an adequate supply of nitrate. In addition, *Microcystis* seems to be less affected by changes in nutrient supply but more affected by changes in light, transparency and temperature. In contrast, *Oscillatoria* seems to be not only sensitive to changes in nutrient conditions but could be heavily grazed (Recknagel, 1997). However, there are many exceptions in the distribution of cyanobacterial species and most of them may occur even in nutrient-poor water (Rósen, 1981).

During blooms, cyanobacteria adversely affect water quality, not only because of degradation products, but many are toxic. It can cause gastro-enteritis and skin irritations (Bowling & Baker, 1996). Table 6.6 lists the potential health effects that cyanobacteria can have according to DWAF (1996b).

Table 6.6: The health effects of cyanobacteria present in potable water

Cyanobacterial cell numbers/ml	**Health effects**
0 – 5	No health effects
50 - 14 000	Possible chronic effects associated with the long-term ingestion of cyaobacteria at this concentration. Possible acute neurotoxic effects associated with the ingestion of particularly *Anabaena* sp. Taste and odour problems likely.
14 000 – 42 000	Possible acute hepatotoxic effects associated with the ingestion of *Microcystis* sp. in children (10 kg) drinking 1litre water a day.
> 42 000	Significant risk of acute and chronic effects associated with the ingestion of the algae

Together with health effects when ingested, there are also risks associated when cyanobacteria are present in water used for recreational purposes (Table 6.7). Since Knellpoort Dam is a popular destination for fishing and other water sports, cyanobacterial blooms can have serious consequences if they become more prevalent.

Table 6.7: The effects of the presence of cyanobacteria in water used for recreational purposes

Algae range (blue-green units*)	**Effects**
Target Water Quality Range 0 – 6	No health effects expected. No cyanobacteria bloom expected to occur.
> 6	Cyanobacteria present in significant numbers and scum formation likely. Recreational users should increase their vigilance for algal scums and avoid all contact with scums. Health effects likely with accidental ingestion of the scums and skin irritation likely with contact with the scums.

* Blue-green units refer to the number of algal colonies/filaments counted in a two-minute scan

of 0.5 ml of water at x200 magnification.

Another observation that could be important was the occurrence of very high numbers of *Aulacoseira granulata* in Knellpoort Dam at a depth of 2-2.5m during November 2000. High numbers of *A. granulata* can have adverse effects on water purification, since it is a known filter- and screen-clogging alga (Carter-Lund & Lund, 1995). *A. granulata* usually blooms in spring and is kept in the euphotic zone by turbulence in the water column. It disappears with the onset of thermal stratification, but re-appears for short periods in the summer, whenever there is mixing of the water column sufficient to maintain them in the euphotic zone. This was in fact the case during November 2000, when strong winds were prevalent.

Considering the rest of the *Novo* transfer scheme (from Knellpoort Dam to the Modder River), the transfer of water can have important effects on algal growth. As stated earlier, the chlorophyll *a* concentration was much higher in the Modder River than at the other sampling sites. Since the nutrient concentrations in Knellpoort Dam is much lower than in the Modder River, it can possibly decrease the nutrient concentrations in the river during transfer, which could limit algal growth in the upper reaches of the Modder River. However, pollution emanating from Bloemfontein and Botshabelo plays an important role in the nutrient loading of the Modder River downstream from Rustfontein Dam (Koning, 1998). It is concluded that the *Novo* Transfer Scheme will not have a significant influence on nutrient concentrations in the lower part of the Modder River. Furthermore, the applicability of nutrient limitation in flowing water has long been questioned, based on the reasoning that currents transport a fresh supply of nutrients to the individual cells. Thus, no matter how low the concentration of nutrients are in any water, the cell's environment is "physiologically enriched" by the flow of water (Hynes, 1970; Allan, 1995). The transfer of water from the relatively clear Knellpoort Dam can also improve water transparency of the Modder River. This could result in more frequent algal blooms, including cyanobacterial blooms. On the other hand, the pumping of water from Knellpoort Dam by the *Novo* pump station, will increase turbulence and flow in the upper reaches of the Modder River, which could inhibit cyanobaterial blooms (Steinberg & Hartmann, 1988).

Species composition could influence the rate of primary productivity in turbid systems, since different algal species compete differently for available light. In a study by Huisman *et al.* (1999) it was found that *Chlorella* had the lowest light requirement, followed *Microcystis, Aphanizomenon* and *Scenedesmus*. Algae can adapt to turbid conditions by capturing variable light by increasing their chlorophyll content per cell. This was illustrated by Santamaria-del-Angel *et al.* (1996) who tested phytoplankton from the Colorado River Delta to see whether, under conditions of high turbity, they were capable of higher rates of photosynthesis when exposed to intermittent compared to constant light. They used different illumination types, the first being constant cool white light at an intensity of 66.3 µmol Quanta $m^{-2}.s^{-1}$. The second type of lamp was cool xenon emitted from a strobe with a frequency of 600 flashes per minute and an incubation irradiance of 66.17 µmol Quanta $m^{-2}.s^{-1}$. In samples from high turbid locations, the authors found a higher concentration of chlorophyll and a higher concentration of chlorophyll per cell. These algae had higher assimilation rates and rates per cell when exposed to the stroboscopic lamp than the samples from the clear water station in the lake. Samples taken from the clear water station had higher assimilation rates and assimilation rates per cell when exposed to constant light compared to the algae from turbid locations. This increase in the concentration of chlorophyll per cell is a result of the ability of the phytoplankton to adapt to low light conditions by increasing the size of the photosynthesising unit. Such adaptation can take place in a short time, of minutes to hours. From this it is fair to conclude that an increase in turbidity of Knellpoort Dam may result in algae that can adapt to low light or intermittent light to become dominant.

6.4.2) Phytoplankton composition

6.4.2(a) Species dominance and seasonal succession

Seasonal climatic cycles induce corresponding fluctuations in phytoplankton abundance and productivity at all latitudes, the magnitude of these fluctuations tending to increase with diminishing light intensities. Diatoms display early successional episodes and these are followed by chlorophytes, and finally cyanobacteria. Extreme habitat modifications (such as hypertrophy and/or salinity) tend to lead to dominance of the habitat by single taxon, often represented by a single species (Cottingham *et al*., 1998).

The presence of the five dominant algal groups in the *Novo* Transfer Scheme was similar to the Vaal River, South Africa, where the phytoplankton community was represented by six algal groups: Dinophyceae, Euglenophyceae, Cyanophyceae, Bacillariophyceae, Chlorophyceae and Cryptophyceae (Pieterse, 1987). According to Pieterse (1987) the succession in the Vaal River was *Trachelomonas* (March), *Cyclotell*a (April and May), *Ankistrodesmus* (June and July), *Stephanodiscus* (August), *Micratinium* (September) and *Chlamydomonas* (October). The warmer months had the highest diversity and the lowest were found during late winter, early spring (August and September).

There was greater variation in family domination in the Modder River compared to the other three systems of the *Novo* Transfer Scheme, which illustrates the more eutrophic conditions in this river. This is supported by a study conducted by Cottingham *et al.* (1998), who found that more than half of the species were turning over from one year to the next. This also suggests that the onset of enrichment promotes major changes in the presence and absence of particular members of the phytoplankton community.

Species composition in a phytoplankton community changes according to the succession stage. It has been observed that the appearance and succession of a certain phytoplankton species was closely related to the available nutrients (Kawabata & Kagawa, 1986; Tremel, 1996, Cottingham *et al*, 1998). This was also observed in Knellpoort Dam where cyanobacteria appeared after the transfer of nutrient-rich water from the Caledon River. However, diverse assemblages of phytoplankton are usually observed in natural waters, especially in oligotrophic regions, implying that phytoplankton can successfully co-exist while competing for a few limiting nutrients (Siegel, 1998). In fact, a scaling analysis demonstrated that rates of competition should increase with cellular abundance and phytoplankton size. For a typical eutrophic system (relatively large cell abundances and cell sizes), resource competition among algae appear to be likely. However, in oligotrophic waters (low cell abundances and small cells), rates of competitive displacement should be greatly reduced (Siegel, 1998).

In rivers, diatoms are the most abundant and species-rich primary producers, occurring in all habitats from the source to the mouth (Jüttner *et al.*, 1996) and they are seldom absent from plankton communities in all types of waters (except under very oligotrophic conditions) (Rósen, 1981). Numerically, diatoms were dominant in the Vaal River, South Africa, in April and August (autumn and spring) while green algae were dominant during the other months. Diatoms were also dominant more than once in the *Novo* Transfer Scheme. In fact, in the Caledon River, the Bacillariophyceae dominated twice during the study period and was replaced by either Chlorophyceae or Euglenophyceae. In Knellpoort Dam, the Bacillariophyceae also dominated twice and was replaced by Chlorophyceae. In the Modder River, the Bacillariophyceae dominated four times and was replaced by Cyanophyceae, Chlorophyceae and Euglenophyceae (in that order), while in Rustfontein Dam the Bacillariophyceae dominated three times and was replaced by Cyanophyceae and/or Chlorophyceae.

The phytoplankton succession patterns in both the impoundments (where Bacillariophyceae was replaced by Chlorophyceae) are in accordance with other studies that showed that increased diatom numbers are closely correlated with increased mixing depth as a result of wind-mixing (spring and autumn). These are closely followed by an increase in green algae which are particularly evident at the onset of thermal stratification. Large summer blooms of nitrogen-fixing cyanobacteria (*Anabaena* sp.) follow the green algae and these are in turn replaced by blooms of the non-nitrogen fixing cyanobacteria (*Microcystis*).

The fact that the phytoplankton population in the Caledon River consisted mainly of diatom species, agrees with results obtained elsewhere, e.g. the Gironde estuary, which also has a combination of high turbidity and high river run-off. The population in the estuary was characterized by the diatoms *Skeletonema costatum* and several *Thalassiosira* spp. (Muylaert & Sabbe, 1999).

Stephanodiscus hantzschii is another typical diatom species which is often very common in eutrophic rivers and *C. meneghiniana* is often found in the freshwater tidal and oligohaline zones of turbid estuaries and can also nourish heterotrophically. Both of these diatoms occurred in the Caledon River. Reynolds *et al.* (1994) suggested that in turbid systems the major primary productivity occurs in shallow side basins and that species in these systems are adapted to strong variations in the light climate.

Another implication of the dominance of diatoms in the Caledon and Modder Rivers is that silica as nutrient is always available. The SiO_2-Si concentration in the Caledon River was 12.8 mg/L (Section 6.3.8), higher that the world average (9 mg/L) and the average for African rivers (11.24 mg/L) (Golterman, 1975; Allan, 1995) while the average SiO_2-Si concentration in the Modder River was 7.87 mg/L. In the Orange River, South Africa, the average SiO_2-Si value was 8.09 mg/L (DWAF, 1997).

Different diatoms, however, require different concentrations of silica, e.g. *Alaucosira granulate* was consistently found in water with high silica concentrations, while *Stephanodiscus astraca* occurred in waters with silica concentrations of less than 1 mg/L (much lower than the silica concentrations in the Modder and Caledon River) (Kilham, 1971). *Alaucosira granulate* occurred in both the Modder and Caledon Rivers, but *Stephanodiscus* sp. never dominated in the Caledon River, where silica concentrations were high.

6.4.2(b) <u>Algae as indicator species in the *Novo* Transfer Scheme</u>

Phytoplankton species within distinct taxonomic divisions often share several physiological and morphological efficiencies. Thus similar characteristics between algal species can identify a specific group as “indicators” of water quality.

Rósen (1981) used the total P value to characterise the total nutrient supply and to predict indicator species. The basis for his classification of indicator species was established on their occurrence in lakes with different phytoplankton volumes but above all their relations to certain chemical and physical parameters. According to him, the cyanobacteria and green plankton algae comprise species (except the well-known eutrophic ones), indicating either nutrient poor waters, clear lakes or high or low contents of humic matters, while species of Chrysophyceans often dominate in nutrient poor lakes. In extreme oligotrophic lakes the planktonic diatoms are not very common and occur most frequently in lakes with high salinities. As concluded in Chapter 6, the waters of the transfer scheme are eutrophic.

A few typical algal indicators of eutrophy are *Tetraedron minimum*, *Scenedesmus* spp., *Crucigenia tetrapedia* and *Pediastrun boryanum*. Other species like *Cryptomonas oata*, *Cyclotella comta* and *Peridinium* cf *cinctum* are designated as being indifferent (Rósen, 1981; Tremel, 1996). Of the diatom species, *Alaucosira granulate* usually occurs in eutrophic conditions, while *Cyclotella* sp. is associated with oligotrophic conditions (Rósen, 1981). *Alaucosira granulate* occurred at all the sampling sites, which were classified as eutrophic, however, *Cyclotella* sp. occurred in the Modder River and in Rustfontein Dam, and is therefore not an indicator of oligotrophic water as suggested by Rósen (1981).

Rósen (1981) further stated that the family Chlorophyceae are seldom noted in great numbers but are very common in moderately and extremely eutrophic lakes (TP = 27 µg/L and conductivity = 116 µS/cm). The Chlorophyceae comprised a large part of the algal community in all the waters of the *Novo* Transfer Scheme. Of the species mentioned in the tables above, *Oocystis* is listed by Rósen (1981) as having an oligotrophic affinity (TP < 11 µg/L) and a high tolerance towards low pH values. *Oocystis* occurred in both the Caledon River and Rustfontein Dams, which had an average TP concentration of 570 µg/L and 94 µg/L respectively, also contrary to the view of Rósen that this is an indicator of oligotrophic waters. *Staurastrum* is listed as an intermediate species and occurs quite frequently in nutrient-poor waters, but has been noted in more nutrient-rich systems, which explains its presence in Knellpoort Dam. Of the Euglenophyceae, *Trachelomonas* is listed as having a preference for waters which are very nutrient-rich, and in the *Novo* Transfer Scheme, it occurs in the Modder River, which had an average TP concentration of 291 µg/L.

In nutrient-rich lakes, cyanobacteria such as *Microcystis*, *Anabaena* and *Oscillatoria* tend to increase their cell numbers under stable calm conditions in summer to the extent that they from surface scums. This was the case in both the Modder River (when the flow was low) and Rustfontein Dam. Of concern, though, is the fact that *Anabeana circinales* appeared in Knellpoort Dam after the transfer of water. The potential problems this may cause in the use of the water body were discussed above in Section 6.4.1.

6.4.3) Primary productivity and light

Siltation can be a considerable problem in reservoirs and lakes, especially in artificial impoundments heavily impacted by run-off from agricultural watersheds. Allochtonous input of clay suspensoids and resuspension of clays within these basins can reduce light penetration, thereby directly influencing phytoplankton productivity (Heath & Francko, 1988).

Because total productivity is directly proportional to light available at any depth, a reduction in light penetration caused by turbidity above clear-water conditions would cause a corresponding decrease in phytoplankton productivity (Heath & Francko, 1988). Consequently, in turbid systems primary production is limited by light rather than by nutrient availability (Grobler & Toerien, 1986; Lloyd *et al.*, 1987). Turbidity also influences the critical mixing depth, results in fluctuating light intensities and affects the passive or active movement of algae in the water (Dokulil, 1994). This could be detrimental if it limits the food supply for fishes and macro-invertebrates. Suspended sediments can smother benthic organisms and interfere with fish feeding patterns. As discussed in Chapter 5, suspensoids in a turbid lake may, furthermore, affect productivity by influencing the availability of P either by acting as a source of P, or by sorbing dissolved P, rendering it unavailable for planktonic growth (Heath & Francko, 1988). On the other hand, an increase in turbidity could be beneficial if it limits the growth of toxic cyanobacteria.

At sampling site 1, during the transfer (turbid conditions), the light saturated rate of photosynthesis was 9.94 $mgC/m^3/h$ while at sampling site 2 (clear conditions) during this time, it was 13.65 mg $C/m^3/h$, an increase of 37% in the photosynthetic rate. This agrees with findings of Lloyd *et al.* (1987) who stated that a turbidity of only 5 NTU can decrease the primary productivity of shallow clear-water streams by about 3 – 13%, while an increase of 25 NTU may decrease primary productivity by 13 -50% .

In comparison, primary production as determined by Grobbelaar (1992) in two impoundments in the Modder River, Mockes Dam and Krugersdrift Dam, showed that the light saturated rate of photosynthesis (P_{max}) in Mockes Dam varied from 11.63 $mgC/m^3/h$ to 425 $mgC/m^3/h$, while in Krugersdrift Dam it varied from 5.14 $mgC/m^3/h$ to 488.26 $mgC/m^3/h$. The average P_B^{max} in Krugersdrift Dam was almost twice that of Mockes Dam for the period before the flooding that occurred during the study period, but after the floods P_B^{max} was more than 1.5 times higher in Mockes than in Krugersdrift Dam. Before the floods Krugersdrift Dam was the clearer of the two impoundments, whereas after the floods Mockes Dam was a little clearer than Krugersdrift Dam. From this, he concluded that turbidity influences overall productivity, and that nutrients influence productivity only when a more favourable underwater light regime prevails, confirming the differences observed in Knellpoort Dam (in the turbid and clearer zones). The primary productivity values in Knellpoort Dam was, however, much lower than the maximum found by Grobbelaar (1992), but was in the same range than the minimum values. The highest P_{max} value obtained for Spioenkop, a turbid impoundment in the Thukela River, was 280 $mg.m^{-3}.h^{-1}$

(Hart, 1999), much higher than in Knellpoort Dam.

There was a decline in water column phytoplankton production at the eastern region of Lake Chapala, a turbid, shallow lake in Mexico, during the dry season, due to a decrease in the photic depth caused by increased sediment resuspension accompanying the declining water level (Lind *et al.*, 1997). The degree of resuspension was a function of wind velocity and the nature of the sediments. These clays can be unusually small with a mean diameter of 0.5 µm. In many lakes, higher inorganic turbidity accompanies periods of rainfall due to run-off of clays from the watershed. For Lake Chapala, this pattern is the opposite. Despite transient periods of extremely high turbidity at the onset of the rainy season near the inflow, the rainy season brings about the greatest transparency over most of the lake (Lind *et al.*, 1992). In the case of Knellpoort dam, an increase in turbidity will only occur when water is transferred from the Caledon River, because there is no large inflow into the dam from the small Rietspruit, which only flows during periods of high rainfall.

Phytoplankton primary productivity essentially follows the same pattern in turbid waters as in clear waters, except that productivity profiles are compressed, due to the rapid attenuation of light (Grobbelaar, 1985). This can clearly be seen at sampling site 1 during November (Figure 6.7) where P_{max} occurred at a depth of 0.25 m, while P_{max} at sampling site 2 was at a depth of 0.75 m. In Lake Chapala, where nutrient concentrations were high and TN:TP was 1.47, the production rates at the depth of optimum irradiance (P_{opt}) were very high. These rates, which once exceeded 5 g C. $m^{-3}.d^{-1}$, are comparable or even higher than in most eutrophic lakes. However, because of the high turbidity, such production rates are restricted to a layer of only a few cm and total water column production is low (Lind *et al.*, 1997), although the productivity per unit area is the same than in clearer systems..

From the productivity profiles in Knellpoort Dam it was seen that, despite the transfer of water from the Caledon River, productivity was similar between sites 1 and 2. Thus, the transfer of water did not affect productivity in the dam far from the point of water intake.

Considering the light penetration, the attenuation coefficient at sampling site 1 during the transfer was calculated as 7.47 m^{-1}, while at sampling site 2 during this time it was 1.96 m^{-1}. This clearly illustrated the effect of the inflow of turbid water on the light penetration in Knellpoort Dam. The attenuation coefficient at sampling site 1 during October 2001, five months after the transfer was stopped, was 2.01 m^{-1}, and at sampling site 2 it was 1.99 m^{-1}. Thus, when no water was transferred, the attenuation coefficients were the same for both sampling sites (and similar to site 2 during the transfer), illustrating that light penetration is only influenced during the period that water is transferred, and that conditions return to that prior to the transfer. This is probably due to sedimentation of the particles over a period of time, if no transfer of water takes place.

The critical mixing depth is also important in determining overall productivity, as it influences the time phytoplankton spend in the light and dark, if the mixing depth exceeds the photic depth. This was illustrated in a study conducted in shallow, turbid ponds in Kenya where light limitation did not exert an influence on the accumulation of algal biomass, even though the ponds were among the most turbid ever recorded (Sarnelle *et al.*, 1998). This re-enforces the importance of determining the ratio of mixed depth/euphotic depth in assessing light limitation. A similar conclusion about the importance of mixing depth to algal productivity was reached by Grobbelaar (1985) in his comparison of shallow and deep reservoirs with high organic turbidity. Thus, the unfavourably low photic zone compared to the mixing depth can be considered important in turbid systems if the critical mixing depth is exceeded (Grobbelaar, 1989, Fichez *et al.*, 1992). This is further illustrated in two South African reservoirs (Wuras Dam and Gariep Dam) where the light attenuation was similar but the $Z_{mix} : Z_{eu}$ ratios were not. In Gariep Dam the critical mixing depth is always exceeded while in Wuras Dam it rarely does. Talling (1971) also suggested that many natural phytoplankton populations will not be capable of positive net photosynthesis when $Z_{mix}: Z_{eu}$ exceeds 5. Thus, the $Z_{mix} : Z_{eu}$ ratio in Knellpoort Dam (6.67 when water is transferred) can account for the relatively low productivity in Knellpoort Dam during the transfer of water. In addition to $Z_{mix} : Z_{eu}$ another factor is the rate of phytoplankton circulation through the euphotic and aphotic zones (Lind *et al.*, 1992), as well as nutrient concentrations, which probably limited productivity during the period where no water was transferred.

Lind & co-workers (1997) also determined the photosynthetic efficiencies in the turbid Lake Chapala and they found them to be comparable to those of clear water lakes. They were also similar to those found by Grobbelaar (1989) for turbid Wuras Dam, South Africa but are about an order of magnitude greater than for Gariep Dam, South Africa.

6.4.5) Relationship between primary productivity and nutrients

An increase in nutrient concentrations was observed in Knellpoort Dam after the transfer of water from the Caledon River (Chapter 5). This can have serious implications for algal growth. Cottingham & co-workers (1998) found that increased nutrient inputs increased total phytoplankton biomass, primary productivity, chlorophytes, cryptomonads and species turnover rates in three enriched lakes in America, while cyanobacteria increased in two of the three lakes.

However, nutrient addition also led to declines in previously dominant dinoflagellates and chrysophytes, and in species diversity. At the species level, there were large changes in community composition from year to year in both enriched and reference lakes, suggesting that the phytoplankton community composition is highly dynamic even in the absence of enrichment. Overall, changes in total biomass, productivity and species diversity were consistent among the enriched lakes, while changes in species diversity differed due to variation in the physical, chemical and biotic environment of each lake. This was also the case in Knellpoort Dam, where the nutrient concentrations were increased during the transfer of water from the Caledon River. Cyanobacteria occurred in the impoundment after the transfer of water, whereas no cynaobacteria were present in the period prior to the transfer.

The dependence of phytoplankton biomass on P is remarkably general in lakes. Since Sakamoto (1966) and Vollenweider (1976) first demonstrated the strong empirical links between P, N, and chlorophyll *a* and its subsequent generalization by Dillon & Rigler (1974), over 60 chlorophyll *a* :P relationships have been published. P is expected to be a relatively poor predictor of phytoplankton biomass in N-limited lakes, and conversely, N is a poor indicator of phytoplankton abundance in P-limited lakes.

However, determining the relationship between chlorophyll *a* and N, the impoundments in the *Novo* Transfer scheme did not show the same linear relationships between this nutrient and chlorophyll *a* as were found in other studies. The reasons for this are probably because chlorophyll *a* concentrations were very low throughout the study period and never increased significantly in these impoundments, although the nutrients showed some variations. Another factor could be that TN was taken as the sum of NO_3-N and NH_4-N; or the limited number of data points. Because chlorophyll *a* values were often undetectable in the Caledon River due to the very high turbidity values, no relationship could be found in this river either. Turbidity thus also plays an important role in determining the relationship between chlorophyll *a*, N and P.

Furthermore, no statistically significant relationship was found between TP and chlorophyll *a* in the two impoundments, or in the Caledon River. However, the linear relationship found between TP and chlorophyll *a* in the Modder River is in agreement with a linear relationship found by Koning (1998).

The fact that chlorophyll *a* in the Modder River showed a linear relationship with TP, while it did not in the impoundments, can be explained by the fact that the N:P ratio in the river was 5.1, while for both impoundments it was 1.6. Thus N is more limiting in the impoundments than in the Modder River, resulting in the poor correlation between TP and chlorophyll *a* (see also Prairie *et al.*, 1989). The same linear relationship between TP and chlorophyll *a* will be expected for the Caledon River, with a TP:TN ratio of 12.5, but as explained earlier, turbidity is the most important limiting factor for algal growth in this river and dominates any relationship between chlorophyll *a* and nutrients.

From the above, it can be inferred that TP can not be used as in the indicator in the impoundments of the *Novo* Transfer scheme to predict algal blooms, since these systems are N-limiting. It can neither be used in the Caledon River, since turbidity influences this relationship in the river. However, it seems as if it can be used as an indicator in the Modder River, since two studies have shown a linear relationship between TP and chlorophyll *a*. It is important, however, to realise that annual variations can change this relationship.

6.5) CONCLUSIONS

Different factors influence phytoplankton composition and productivity in aquatic systems. Of these, flow, light and nutrients are the most important. In the different waters of the *Novo* Transfer Scheme, there are large variations in the algal composition. In the Caledon River, the Bacillariophyceae (specifically *Cyclotella* and *Stephanodiscus* spp.) dominated the phytoplankton throughout the study period, and because it is known that they could grow heterotrophically, especially under low light, this might explain the dominance. Turbulence in the river, together with high turbidity, probably limited the development of cyanobacterial blooms.

In Knellpoort Dam, the algal community was dominated by Bacillariophyceae and Chlorophyceae, and Cyanophyceae numbers increased only after the transfer of nutrient-rich water from the Caledon River. Since the study commenced, Cyanophyceae was not found to be present in this impoundment until after the transfer of water, thus, the transfer of water from the Caledon River, caused a shift in species dominance (to Cyanophyceae). This could have serious implications for the uses of the impoundment, which include water purification for drinking water and recreation, as cyanobacterial blooms can be harmful to humans and animals.

Furthermore, since light is an important factor limiting algal growth in the Caledon River, it can become important in Knellpoort Dam when water is being transferred from the Caledon River. Primary productivity measurements made during and after the transfer of water, showed that increased turbidity compressed the productivity profile at the inflow of transferred water, and decreased the light saturated rate of photosynthesis, compared to a sampling site in the middle of the impoundment away from the inflow of turbid water. After the transfer was stopped, the productivity showed the same pattern at the inflow as in the middle of the impoundment. However, the possibility still exists that a larger area of the impoundment will be influenced over long periods of water transfer. The direction and force of the wind, as well as the volume of water being transferred, plays an important role in the size of the area that will be affected.

In the Modder River, the algal community was dominated by Bacillariophyceae, Chlorophyceae and Cyanophyceae, and it was the same in Rustfontein Dam. Higher concentrations of nutrients and high light availability are probably responsible for the high occurrence of cyanobacteria in the Modder River and Rustfontein Dam, in addition to more stable conditions (less mixing).

The transfer of water from Knellpoort Dam to the Modder River could dilute the nutrient concentration in the river. It could also increase light availability and stream flow. Higher light availability and higher nutrient concentrations could favour algal growth, while higher stream flow could limit the development of blue-green algal blooms. However, it is difficult to predict the specific effects on the phytoplankton of the Modder River, since no significant transfer of water occurred via the *Novo* pump station during the study period.

Chlorophyll *a* showed no statistically significant correlations with TN in any of the systems, although all the systems (except the Caledon River) can be viewed as nitrogen limiting based on the N:P ratio.

In addition, no significant correlations between chlorophyll *a* and TP were found in any of the systems, except for the Modder River, which had a N:P ratio of 5.1. The high turbidity of the Caledon River was probably the most important factor influencing the correlations between algal growth and nutrients. It can be concluded that neither TN nor TP concentrations can be used in the *Novo* Transfer system to predict algal blooms, since algal growth is also influenced by other important factors.

CHAPTER 7
INVERTEBRATES IN THE MODDER AND CALEDON RIVERS

7.2) INTRODUCTION

Running waters present unique patterns of distribution of biological diversity among taxonomic groups and among regions. Aquatic systems are no different from terrestrial systems, and thus knowledge of the number of species in streams and rivers of a region is more complete for vertebrates than for invertebrates. Consequently, the knowledge is also more complete within temperate zones than in the tropics. However, although invertebrate diversity is incompletely catalogued, it generally exceeds vertebrate diversity at any location. Allan & Flecker (1993) for example found the diversity of insects in tropical streams to exceed that found in temperate streams. Macro-invertebrates can have an important influence on nutrient cycles, primary productivity, decomposition, and translocation of materials. Interactions among macroinvertebrates and their food resources vary among functional groups, but they constitute an important source of food for numerous fish (Wallace & Webster, 1996). On the other hand, various factors can influence the distribution of invertebrates in lotic waters and may affect the populations in the Modder and Caledon Rivers.

7.1.1) Invertebrate drift

Invertebrate drift is defined as the transport of organisms in running water through currents. Elliot (1970) and Car (1983) both concluded that invertebrate drift in rivers is subject to a recurrent temporal pattern of activity within a 24-hour period called diel periodicity. Car (1983) further concluded that fluctuations in drift density are determined by various factors, including the density of invertebrates in the benthos, the stage in their life history, their activity and behaviour and the current velocity to which they are exposed. In addition, he stated that drift can be constant (occurring at any time), behavioural (with a diel pattern with greater nocturnal activity) or catastrophic (associated with physical disturbance of the benthos).

Stream communities are resilient and drifting animals allow for rapid community recovery from disturbances caused by interruptions in flow due to water transfer. This is because effects of disturbances in streams are short-lived because denuded patches are colonised rapidly by mobile community members (Townsend, 1989).

7.1.2) Sediment stability

The major subsurface habitat in freshwater includes the sediments of running waters (rivers and streams) and lentic bodies (ponds and lakes). Subsurface habitats can be viewed as habitats along a physical continuum of high flow/large particle environment against low-flow/fine sediments. Sediment size and water flow rates play important roles in determining the biological diversity and abundance of bottom-dwelling (benthic) organisms in freshwater, even for those biota that spend some of their life cycle in the water column or on land (Palmer *et al*., 1997). A study conducted by Iversen *et al*. (1991) showed that large differences in invertebrate density and diversity occurred between samples collected near the banks and midstream. These patterns were attributed to invertebrates either avoiding unstable sand or preferring areas with high algal biomass, confirming that sediment instability was one of the main factors regulating invertebrate distribution. Increasing the slope, stream velocity and sediment transport in lowland streams, can alter substrata variation and stability in such a way that it significantly affects algal biomass development and distribution, as well as the distribution density and structure of the invertebrate communities.

7.1.3) Availability of oxygen

The most important ecological processes in freshwater sediments are decomposition of organic matter, the uptake and transfer of materials, and production by aquaphytes and certain bacteria. All three of these processes are influenced directly or indirectly by sediment biota and by the availability of oxygen, which in turn has a huge influence on the sediment biota. The maintenance of aerobic conditions in surface sediments is the most important factor affecting diversity, being high when it is aerobic and vice versa (Palmer *et al.*, 1997).

7.1.4) Light/turbidity

The underwater light climate plays an important role in invertebrate dynamics. Lloyd *et al.* (1987) observed that seasonal zooplankton densities in glacially turbid lakes were as little as 5% of those in clear-water lakes. Furthermore, zooplankton density decreased with decreasing euphotic depths. Even low turbidity reduced the reproductive potential of *Daphnia* spp. below that in clear waters (Lloyd *et al*., 1987). In accordance with these findings, McCabe & O'Brien (1983) observed that a turbidity of 10 NTU resulted in significant declines in the feeding rate and food assimilation capability of *Daphnia pulex*. Some of the impacts (both positive and negative) on invertebrates that can occur due to turbidity are listed in Table 7.1.

Table 7.1: Effects of turbidity on aquatic invertebrates (Wilber, 1983)

Type of invertebrate	Effect of turbidity
Protozoa	Some species seem to be harmed by an increase in turbidity. Others such as amoebas are unaffected.
Porifera	High turbidity is harmful to sponges. Most require clean, clear

Type of invertebrate	Effect of turbidity
	water.
Coelenterata	Some forms tolerant against an increase in turbidity. The process of sedimentation has been proposed as the key adverse factor.
Ctenophora	Higher turbidity causes mechanical injury to various tissues.
Polychaeta	Low turbidity may stimulate certain biological processes. Higher turbidity may be harmful and even fatal.
Mollusca	Some species show a change of filtration rate as turbidity increases. Excessive turbidity is harmful.
Echinodermata	Crinoids seem to benefit from turbid water.
Chordata	A TSS concentration of up to 25 mg/l seems to be safe. Most of the adverse effects caused by turbidity is probably mechanical.

7.1.5) Diel periodicity

Invertebrate taxa collectively exhibit four types of diel patterns: i) similar downstream patterns, which are generally different from upstream patterns, ii) similar benthic patterns, different from drift, iii) a periodic pattern, and iv) independent patterns for each type of directional movement (Bergey & Ward, 1989). A study conducted on diel patterns in Buckhorn Creek (a third order river in the Rocky Mountains, USA), showed a peak in invertebrate numbers during the night; with sunset, sunrise and daytime following in decreasing order.

7.2) MATERIALS AND METHODS

Since the main concern with regard to invertebrates in the Novo Transfer scheme, was the possible transfer of invertebrate species, the invertebrates were sampled in order to determine the species present at the different sampling sites. In addition, the sites were scored once according to the SASS5 method as developed by Chutter (1994; 1998). The current upgrade is Version 5, which is specifically designed to comply with international accreditation protocols (Dickens & Graham, 2002). This was only done to obtain a better understanding of the ecological conditions at the different sampling sites, and was not a major focus of this study. Among the taxonomic groups living in running waters, invertebrates are good candidates to monitor ecological changes caused by human impacts, because they have a relatively short lifespan and exhibit a multitude of strategies to adapt to environmental gradients (Gayraud *et al*., 2003).

The following habitats were sampled where possible:

Stones in current: The sampling net was placed close to, and downstream of the stones to be kicked, in a position where the current will carry the dislodged organisms into the net. The stones were kicked over and against each other to dislodge the animals at different places, for approximately two minutes.

Stones out of current: For this and all remaining biotopes, the sampling net was moved to catch the biota as they were dislodged. Approximately one square metre of these stones was sampled by kicking the stones in the same manner as described for stones-in-current, whilst continuously sweeping the net in the disturbed area.

Sand, gravel and mud: In all three cases the substrate was stirred by shuffling or scraping the feet for approximately half a minute, while continuously sweeping the net over the disturbed area to catch the dislodged organisms.

Aquatic vegetation: The net was pushed against and amongst the vegetation under the water in an area of approximately one square metre.

Before and after disturbing the site, approximately one minute of hand-picking for specimens that may have been missed by the sampling procedure was carried out. Once the sample was collected, it was washed down to the bottom of the net, and tipped into a tray with water. Before scoring began, leaves, twigs and other loose debris were removed from the tray. Organisms were identified to family level.

Not all the habitats were present at the different sampling sites. The Caledon River in particular, does not contain stony habitats, and has only sandy sediments and vegetation to sample. This was taken into account when determining the SASS5 scores, and habitats were scored with the HABS1 score sheets. HABS1 has a range of scores from 10-100, in steps of five, depending on the combination of the biotopes present.

The fish species were not sampled.

7.3) RESULTS

7.3.1) Invertebrate community composition at the different sampling sites

The invertebrate families that were found at the different sampling sites in the Novo Transfer Scheme are given in Table 7.2.

Table 7.2: The different invertebrate families present at the different sampling sites (including all habitats) of the *Novo* Transfer scheme

Sampling site	Invertebrate families present
Caledon River	• Notonectidae • Dytiscidae • Baetidae • Gomphidae • Ceratopogonidae
Knellpoort Dam	• Muscidae • Baetidae • Corixidae • Chironomidae • Coenagriidae
Rietspruit (does not flow during dry periods)	• Baetidae • Gomphidae • Muscidae • Coenagriidae
Modder River	• Notonectidae • Corixidae • Belastomatidae • Lestidae • Coenagriidae • Nepidae • Shrimps • Hydrachnellea • Gomphidae • Baetidae • Ceratopogonidae
Rustfontein Dam	• Baetidae • Corixidae • Notonectidae • Shrimps • Belastomatidae

The Modder River showed the largest abundance of macro-invertebrates during the study period, while the Rietspruit had the lowest abundance. The taxa that occurred at all the sampling sites included: Hemiptera, Ephemeroptera, Diptera and Odonata. Coleoptera

(Dytiscidae) only occurred in the Caledon River, while Crustacea and Hydracarina only occurred in the Modder River. Baetidae was the only invertebrate family that occurred at all the sampling sites.

7.3.2) Ecological condition of the river sites in terms of SASS5

In addition, the sites in the rivers were scored in terms of SASS5 to determine their ecological integrity (Table 7.3). Only the rivers were scored, as the SASS5 method was specifically developed for flowing water.

Table 7.3: SASS5 scores at the different sampling sites

Sampling site	Habitats sampled	Habitat score	SASS5 score	No. of taxa	ASPT (score/taxon)
Caledon River	Marginal vegetation Sand	50	32	5	6.4
Rietspruit	Stones in current Aquatic vegetation Mud/sand	80	17	3	5.6
Modder River	Stones in current Aquatic vegetation Mud/sand	80	61	8	7.6

Based on the above, Table 7.4 was used to determine the ecological condition of the sites, taken from the SASS4 User Manual that was published in 1995 (Thirion *et. al*., 1995).

Table 7.4: Categories used to classify habitat, SASS5 and ASPT values

Habitat	SASS5	ASPT	Condition
> 100	> 140	>7	Excellent
80-100	100-140	5-7	Good
60-80	60-100	3-5	Fair
40-60	30-60	2-3	Poor
< 40	< 30	< 2	Very poor

The sampling sites in the different rivers can thus be classified as:

Caledon River = Poor. Although the ASPT is between 5-7, both the habitat and SASS5 scores were low.

Rietspruit = Fair. Although the SASS5 score at this site was very low, it was mainly due to low flow conditions during most of the study period. The habitat score and the ASPT both placed this site in the “Good” category.
Modder River = Good. The habitat and ASPT scores placed this site the “Excellent” category, but the SASS5 value was in the “Fair” category.

7.4) DISCUSSION

7.4.1) Invertebrate community composition at the different sampling sites

The occurrence of Baetidae at all the sampling sites is to be expected, since they are the most common mayflies that occur in large low-lying rivers and prefer gently flowing water, according to the SASS5 guideline. Chutter (1970) also considers Baetidae to be stones/vegetation-in-current inhabitants.

The sampling site in the Modder River provided suitable habitats for the development of invertebrate communities, including stones-in-current, aquatic vegetation and mud/sand. Samples taken during the study period at this site also showed high concentrations of chlorophyll *a*, indicating the presence of a food source. In addition, the site has low flow characteristics, high dissolved oxygen concentrations, and low turbidity, all contributing to create favourable conditions for invertebrates. The state of this reach of the Modder River is similar to that of the Umzimvubu River, South Africa, where faunal diversity was low, but many taxa found were sensitive to water quality change, indicating good water quality (Madikizela & Dye, 2003). This is also in accordance with Sheldon (2005), who stated that in large dryland rivers, site specific invertebrate richness tends to be lower than in equivalent sized permanent rivers, and is often correlated with the duration of flow, water permanence, suitability of habitats and proximity to permanent refugia.

Since the Modder River showed the highest abundance of invertebrate families, the transfer of species into this sysStem, due to the *Novo* Transfer scheme, is not considered to be of concern. However, seven (7) of the invertebrate families occurring in the Modder River prefer slow-flowing or standing waters (Notonectidae, Corixidae, Belastomatidae, Lestidae, Hydrachnellea, Gomphidae and Baetidae). This implies that if the flow of the Modder River increases due to transfer of water, the species composition could be altered significantly. This concern is substantiated by several studies that are summarised below.

In the River Glen, Linlcolnshire, England comparison of the frequency of individual taxa in summers when flows were below or above the flow sustained by the inter-basin transfer (0.107

m/s) showed a relative high number of taxa, including Sailidae, Molannidae, *Planorbis* spp. and Haliplidae, to be more frequent in low-flow summers, whereas other taxa, including Rhyacophilidae and Leptoplebidae were more frequent in high-flow summers. This shows that macroinvertebrates from a small river are responsive to flows, even when these responses are measured using a relatively small sample data set. Further, the change in flows brought about by a relatively small inter-basin water transfer can be of a magnitude which is expected to have a measurable impact on the fauna (Bickerton, 1995).

A study by O'Keeffe & De Moor (1988) that compared data on the hydrology, chemistry and benthic invertebrates of the Great Fish River prior to the opening of the Orange/Fish River interbasin water transfer scheme in 1977, with similar data for post transfer conditions, found that the invertebrate communities of riffles have changed substantially, and only 33 percent of taxa identified were common to both the pre- and post-transfer surveys. In particular, the dominant chironomid, hydropsychid and simuliid species have changed substantially, but no evidence was found that overall invertebrate population richness changed. The most noticeable change in the fauna has been the replacement of the pre-transfer dominant simuliids (*Simulium adersi* and *S. nigritarse*) by *S. chutteri*, a blood-feeding pest of livestock, which caused considerable problems to farmers after the transfer. These changes in invertebrate species can be attributed to the more permanent flow and increased area of erosion habitats. In addition, this study conducted on the Orange/Fish River transfer found that out of a total of 66 taxa, only 22 were common to both the pre- and post-transfer collections. The Baetids *Ephemeroptera*, *Cheumatopsyche afra* and *Simulium chutteri* were much more abundant in the post-transfer samples. The dominant chironomid species in pre-transfer samples, *Orthocladius* sp. was not found in any of the post-transfer samples.

Golladay & Hax (1995) also studied the effect of an engineered flow disturbance on meiofauna in an intermittent north Texas prairie stream, where water is released into the headwaters of a natural stream channel, as part of an inter-basin water transfer. It was found that before diversion, the meiofaunal density was stable or increasing on substrata at all sites. However, following water diversion, total meiofaunal diversity was reduced to 1-2% of prediversion levels and all meiofaunal taxa were affected.

Lastly, Snaddon & Davies (1998), also observed a decrease in taxon richness of the invertebrate communities below the transfer outlet, compared to the river above transfer, for the inter-basin transfer in the Theewaterskloof. Sensitive families such as the heptageniid Ephemeroptera and leptocerid Trichoptera were not recorded below the outlet during transfer months.

During this study, the Caledon River did not show a high abundance of invertebrate families. The high flow conditions, absence of suitable habitats and high turbidity of this river are considered the main factors that limited the abundance of invertebrates at this sampling site. In addition, the majority of the substrate in the Caledon River is unstable sand (as shown in Chapter 4), due to extensive erosion in the upper parts of the catchment and high flows, thus also contributing to the low invertebrate diversity and population numbers.

It is not expected that the extraction of water from the Caledon River, will have any significant effect on the invertebrate community, as it is usually done during periods of high flow, when species abundance will be very low.

7.4.2) Ecological condition of the river sites in terms of SASS5

The classification of the river sites as stated above, were not entirely in accordance with the findings in Chapter 5, where the Caledon River was classified as mesotrophic – eutrophic in terms of both the average summer N and P concentrations (DWAF, 1996a), but as ”Poor” in terms of SASS5. The Modder River, on the other hand, was classified as oligotrophic in terms of the average summer N concentration and as eutrophic in terms of the average summer P concentration, while it was classified as “Good” in terms of SASS5. It is, however, not possible to assume correlations between SASS5 and chemical analyses, since one important limitation of the SASS5 method is that it does not identify the nature of the chemical changes.

7.5) CONCLUSIONS

Thorough long-term surveys of rivers before transfer schemes are built, are necessary in order to assess the effects of the scheme on the ecology following the transfer of water. No survey in terms of SASS5 was done before the Novo Transfer scheme was constructed. As water was not transferred to the Modder River during the study period, the effects on the invertebrate community could not be determined, and possible impacts are predicted on observations made from other studies.

It is not expected that there will be a transfer of undesirable invertebrate species from the Caledon River. Species diversity at the transfer site is generally low, and the high flow conditions during transfer periods will limit the species diversity even more. The main concern in terms of invertebrate diversity is the potential impact of increased flow in the Modder River during periods of transfer. It is considered important that a system be put in place to conduct SASS5 (at least quarterly) on both the Caledon and Modder Rivers, in order to establish whether there is a significant impact resulting from the transfer scheme. The average flow conditions in the Modder River (under normal conditions) should be determined and the flow rate of the transfer must be adjusted so as not to increase the normal flow by orders of magnitude. In fact, the environmental impact assessment done by Ninham Shand for the *Novo* Transfer Scheme recommended the implementation of a water quality monitoring programme upstream of Rustfontein Dam to determine baseline conditions in the Modder River, together with surveys to determine aquatic invertebrate species in the Modder River upstream of Rustfontein Dam and in Rustfontein and Knellpoort Dams (Ninham Shand, 1995).

Another aspect in terms of flow resulting from the transfer could be the alteration of the flow regime in the Modder River to a less variable perennial flow. As was also indicated by O'Keeffe & De Moor (1988), it is recommended that in order to minimise the potential ecological effects of changing the flow patterns in the Modder River, the flow should be stopped in winter during periods of transfer (or for one or two months during some years), to simulate the effects of the periodic droughts in the natural regime.

CHAPTER 8
FINAL CONCLUSIONS

As stated in Chapter 2, this study concentrated on specific aspects in order to determine the ecological impacts of the *Novo* transfer scheme from a limnological point of view. These included:

- The physical and chemical changes of the water quality in both the Caledon and Modder Rivers, as well as Knellpoort Dam and Rustfontein Dam. Variables determined during the study included:
    - nutrients;
    - chlorophyll *a*;
    - total dissolved solids (TDS)/ conductivity;
    - major cations;
    - major anions;
    - pH;
    - oxygen;
    - suspended solids/ turbidity.
- Bio-availability of nutrients adsorbed onto suspended particles carried into Knellpoort Dam by the transfer of water from the Caledon River.
- Primary production in Knellpoort Dam during and after the transfer of water from the Caledon River.
- Algal identification and enumeration, with a specific focus on the development of cyanobacteria (blue-green algae) blooms and conditions favouring these algae.
- Monitoring of macro-invertebrates at selected sites, to determine the composition of invertebrate communities, the ecological health of the different systems and the possibility of transfer of different species from one river to the other.

The main findings and results after investigating each of the above aspects are highlighted below.

8.1) Physical and chemical changes of the water quality of the different systems

8.1(a) <u>Nutrients</u>

Although high, the average N and P concentrations in both the Caledon and Modder Rivers were lower than that found in other Southern African rivers, for example, in the Marimba River, Zimbabwe (Nhapi & Tirivarombo, 2004). This river is highly impacted by sewage inflow from the city of Harare, and shows concentrations of approximately 13.5 mg/L N and 2.6 mg/L P.

The results show that the TP concentration was constantly higher in the Caledon River than in the Modder River, and much higher during the rainy season. The much higher TP concentration in the Caledon River could be ascribed to phosphate-ions that are adsorbed to the suspended particles. These high concentrations of TP are, however, not necessarily biologically available. In addition, the Caledon River system is probably light-limited due to the high turbidity concentrations and this restricts algal growth, despite the high nutrient concentrations.

Considering the PO_4-P concentrations, the average in the Caledon River was two times higher than in Knellpoort Dam. In the Modder River, the average PO_4-P concentration was higher than in both the Caledon River and Knellpoort Dam, although the TP concentration was lower.

Five percent of the TP concentration in the Caledon River, 19% of the TP concentration in Knellpoort Dam, 20% of the TP in the Modder River and 28% of the TP concentration in Rustfontein Dam, was in the form of PO_4-P. Thus, a large portion of the TP concentration in the Modder River system, was in the form of PO_4-P (which is biologically available), while this was not the case in the Caledon River systems. The ratio of PO_4-P to TP is usually low in oligotrophic waters, whereas at high TP concentrations, the dissolved inorganic phosphorus pool reaches almost 100% of the TP (Harris, 1986).

The average NO_3-N concentration of 94 µg/L in the Modder River above Rustfontein Dam, is lower than the 100 µg/L NO_3-N as determined for unpolluted world rivers (Webb & Walling, 1992). During a study conducted during 1996 - 1997 (Koning, 1998), the average NO_3-N concentration in the Modder River was 230 µg/L. However, the average concentration for the period 1996-1997 was determined at five sampling sites downstream from Rustfontein Dam, whereas the samples taken during this study was taken above Rustfontein Dam. Our results suggest that Botshabelo (a big city and informal settlement) and the city of Bloemfontein itself, probably pollute the Modder River downstream from Rustfontein Dam. This downstream pollution was also illustrated by the lower average conductivity found in the Modder River during 2000 – 2001 compared to the measurements taken from 1996 – 1997. On the other hand, in the Caledon River the NO_3-N concentration was much higher than 100 µg/L (being 322 µg/L). The potential sources could be agricultural run-off from Lesotho, the geological formations underlying the river, as well as sewage and industrial pollution from Maseru, the capital of Lesotho.

Based on the average summer P concentration (DWAF, 1996a), all the systems in the *Novo* Transfer Scheme can be classified as eutrophic, except Knellpoort Dam, which falls on the border between mesotrophic and eutrophic. The Target Water Quality Range (DWAF, 1996a) for P states that the trophic status of a freshwater body should not increase above the present level. Knellpoort Dam was classified as mesotrophic (PO_4-P = 12 μg/L) before the transfer and as mesotrophic/eutrophic (PO_4-P = 25 μg/L) after the transfer. Thus, the transfer of the water from the Caledon River did cause a change in the trophic classification of the impoundment, and can continue to do so, especially if large volumes of water are transferred over extended periods of time.

According to the inorganic N concentration, the Modder River and Rustfontein Dam are both ultra-oligtotrophic in terms of average inorganic nitrogen concentrations, while Knellpoort Dam can be classified as oligo-mesotrophic and the Caledon River as meso-eutrophic (Wetzel, 1983).

According to the calculated TP:TN ratios by Sakamoto (1966), the chlorophyll *a* yield of the Caledon River is considered to be dependent on both TN and TP, and for the Modder River and Rustfontein Dam, the chlorophyll *a* yield is dependent on TN (thus the systems are nitrogen limited). For Knellpoort Dam the system was also found to be nitrogen limited, and even more so after the water was transferred.

The seasonal variation in nutrient concentrations in the different systems of the *Novo* transfer scheme is in accordance with studies by Roos & Pieterse (1995a), who found that in the Vaal River, most of the annual load of soluble nutrients, such as phosphorus and nitrogen, are transported during periods of high discharge following rainfall. This phenomenon was also illustrated in other river systems, such as the Humber River, England (House *et al.*, 1997), the Barwon-Darling River, Australia (Bowling & Baker, 1996), the Swan River, Australia (Thompson & Hosja, 1996), the River Spercheios, Greece (Kormas, 1999) and the Upper Orange River, South Africa (Keulder, 1979).

8.1(b) <u>Chlorophyll *a*</u>

The average chlorophyll *a* concentrations were low in the Caledon River, Knellpoort and Rustfontein Dams throughout the study period, compared to the significantly higher concentrations in the Modder River. The average chlorophyll *a* concentration in the Caledon River was 6.7 μg/L, in Knellpoort Dam (surface of the water column) it was 12.7 μg/L, in the Modder River 56 μg/L and in Rustfontein Dam it was 10 μg/L.

There was an increase in chlorophyll *a* in the Caledon River during April and September 2000. During April the algal population was dominated by a *Cyclotella* sp. and during the September (spring) increase it was dominated by *Stephanodiscus hantzhii* and various Chlorophyte species.

In Knellpoort Dam there was an increase in chlorophyll *a* during March 2000 and March 2001. The algal population was dominated by a *Chlorella* sp. during 2000 and by *Anabaena circinales* (900 cells/ml*)* during 2001. In Rustfontein Dam, there was a marked increase in chlorophyll *a* during September 2000, and the algal population was dominated by *Microcystis aeruginosa* and an *Oocystis* species*.*

In the Modder River, chlorophyll *a* increased during May 2000 and again during January 2001. The chlorophyll *a* increase in May was caused by a bloom of *Microcystis aeruginosa* (up to 36 000 cells/ml), while the increase during January was caused by *Anabaena circinales* (805 cells/ml*)* and *Microcystis aeruginosa* (146 cells/ml). Thus, in the Modder River, the increase in chlorophyll *a* during the study period was mainly due to cyanobacteria. Steinberg & Hartmann (1988) stated that above a threshold of 10 µg/L TP, the development of cyanobacteria can be described by physical factors, such as water column stability. When the turbulence of the water column is rather low, cyanobacteria can build up to dense populations. Cyanobacterial blooms also occur more readily at elevated water temperatures, high pH values and low N:P ratios. The N:P ratio in the Modder River was low during the study period (1.6), and the pH was high during May and January when the blooms occurred ranged between 8.36 and 8.6. Together with sufficient light, it created favourable conditions for cyanobacterial growth, as indicated by Steinberg & Hartmann (1988). During a study conducted from 1996-1997, no cyanobacterial blooms were observed in the Modder River, but the possibility was predicted if turbulence in the river decreased (Koning, 1998).

8.1(c) <u>Total dissolved solids/ conductivity</u>

The target water quality guideline range for conductivity, as proposed by the South African Water Quality Guidelines (DWAF, 1996b) for recreational use, is below 70 mS/m. No health, aesthetic or treatment effects are associated at these levels. Both the rivers and the impoundments have conductivity levels far below this guideline concentration. In addition, the average conductivity values of typical unpolluted rivers in general are approximately 35mS/m, which are higher than those found in both the Caledon and Modder Rivers (Webb & Walling, 1992). The conductivity of the Vaal River, South Africa, was, on average, 76 mS/m (Roos & Pieterse, 1995b) and that of the Orange River between 18-30 mS/m for the period 1977-1997 (DWAF, 1997). Since all the average conductivity values of the different components (both rivers and impoundments) of the *Novo* Transfer Scheme were below the value of typical unpolluted water (35mS/m) for the study period (even after the transfer), no adverse effects such as an increase of salinity levels of Knellpoort Dam, is expected because of the transfer of water from the Caledon River. In fact, it is more likely that the water transferred from the Knellpoort Dam (23.7 mS/m), will dilute the more saline waters of the Modder River (28.7 mS/m), once this section of the transfer scheme is put into operation.

The higher conductivity during the dry season in both the Caledon and Modder Rivers, is in accordance with Awachie (1981), who states that in African rivers the conductivity is normally higher during the dry season.

8.1(d) Major cations and anions

According to the classification of Dallas & Day (1993), the Modder and Caledon Rivers are Category 2 rivers, where Ca^{2+}, Mg^{2+} and Na^{+} are the dominant cations. During the 2000 - 2001 study period, Na^{+} and Ca^{2+} (meq/L) were co-dominant in the Modder River, but Cl^{-} and SO_4^{2-} dominated over Mg^{2+}. This was also the case in Rustfontein Dam. In the Caledon River, Ca^{2+} was dominant, while Cl^{-}, SO_4^{2-} and Mg^{2+} all dominated over Na^{+}. In Knellpoort Dam, the dominant ions were Ca^{2+} and Na^{+} (as in the Modder River, although the percentage dominance of Na^{+} was much lower in the impoundment), but Mg^{2+} and SO_4^{2-} dominated over Cl^{-}, which were different from both the rivers.

It is possible that when water is transferred from the Caledon River, it could lessen the dominant role of Na^{+} in Knellpoort Dam, simultaneously increasing the dominance of Cl^{-}, while the transfer of water from Knellpoort Dam to the Modder River, could decrease the dominance of both Na^{+} and Cl^{-} in this river, shifting its ionic composition more to its present status as a Category 2 river. This is confirmed by Thornton *et al.* (1992) who stated that the transfer of water between basins can result in an impoundment having a very different ionic composition to that which would occur naturally in the receiving basin. Some impacts on the biota due to these changes can be a change to species more common in marine habitats, but this will only occur in case of high salinity. However, the major consequence of a change in the ionic composition is the change in alkalinity (Thornton *et al*., 1992). Such changes will result in changes in the plankton composition. Another important aspect is that different ionic and metal compositions can influence the different species of P present in a water body and consequently the bio-availability of phosphorus (de Jonge *et al*., 1993).

8.1(e) pH

The average pH at most of the sampling sites was higher than the range between 6 - 8 as reported to be the expected average for natural ecosystems, by the Department of Water Affairs and Forestry. However, grassland rivers in Africa, such as the Modder and Caledon Rivers tend to be neutral to slightly alkaline (Awachie, 1981). The pH of both the Modder and Caledon Rivers as well as that of the other sampling sites, compares well with the average pH of the Vaal River, South Africa, which is 8.1, and the Orange River, which has an average pH of 8.2 (Roos & Pieterse, 1995a).

8.1(f) <u>Oxygen</u>

No adverse effects due to oxygen stress are expected due to the transfer of water, since all the oxygen concentrations in the systems were within the same range (near saturation level). The lower oxygen concentrations at the bottom waters of Knellpoort Dam and the Modder River are probably due to respiration that occurs in this region.

8.1(g) <u>Suspended solids/ turbidity</u>

The Caledon and Modder Rivers showed large variations and seasonal patterns in terms of turbidity and TSS concentrations, with an anticipated increase during the rainy season and a decrease during the dry season. This is in accordance with the Vaal River where the most important factor that influences turbidity and thus the euphotic zone, is discharge (Roos & Pieterse, 1995a). This is also in accordance with Gustard (1992) as well as Meybeck & co-workers (1992) who indicated that a large proportion of the suspended sediment transport occurs during storm events, when rainfall and stormwater run-off mobilise sediments.

The impoundments in the *Novo* Transfer Scheme did not show the same large variations in turbidity and TSS than the rivers, but this is to be expected since linear flow does not govern the turbidity dynamics in lentic ecosystems. Due to the different slopes in the relationship between TSS and turbidity in the two rivers, it is clear that there is a marked difference between the sediment composition of the Caledon and Modder Rivers. This pointed to finer suspended materials being present in the Modder River, compared to the Caledon River.

The transfer of water from the turbid Caledon River caused an increase in turbidity in Knellpoort Dam. However, this turbidity decreased when transfer of water was terminated, and conditions returned to those prior to the transfer of the water. This increase in turbidity brought about certain effects, such as a reduction in light penetration that was associated with decreased primary production, which is in accordance with studies by Walling & Webb (1992) and Lloyd *et al.*, (1987). Grobbelaar (1984, 1985 and 1989) also found that the presence of non-photosynthetic material has an influence on phytoplankton productivity, due to the rapid attenuation of vertical light penetration which results in a compresses photosynthetic profile and usually a shallow euphotic zone (Z_{eu}) in relation to the water column. Turbidity can also affect the usage of water from Knellpoort Dam for man. It is generally acknowledged that turbid water is less acceptable than clear water consumption, contact recreation, and perhaps aesthetic enjoyment (Lloyd *et al.*, 1987).

The turbidity current observed during the transfer of water did not extend to the inflow of Knellpoort Dam where the *Novo* pump station is located and is thus not expected to increase turbidity in the Modder River. In fact, the possibility exists that the clearer water of Knellpoort Dam (at the *Novo* pump station) will decrease turbidity in the Modder River. A decrease of turbidity in the Modder River, together with high nutrient concentrations that could result from urban run-off, can cause extensive algal blooms, which do not exclude the development of nuisance algal blooms. However, the release of water through the pipeline into the Modder River could result in erosion of the river banks (especially if the water is released in high volumes) which could then result in an increase in turbidity.

8.2) Bio-availability of nutrients adsorbed onto suspended particles

The possibility that the sediments from the Caledon River carry adsorbed nutrients with it, which could influence nutrient dynamics in Knellpoort Dam was identified during this study. Subsequently, this possibility was investigated and it was found that in the Caledon River, the majority of the suspended load is represented by sand-sized particles (53% very fine sand and 37% fine sand) - very fine sand particles are approximately 64 μm in size. The quantity of bio-available P is generally larger in fine grained sediment than in coarse grained sediments (de Jonge *et al.*, 1993). This is because the specific surface area of particulate matter is a key property which controls the adsorption capacity.

The bio-available N fraction from the sediments of the Caledon River was found to be 3.39 mg/L (3 390 mg/kg) for both 1 g and 2 g of added sediments. Since N is highly mobile (Grobbelaar, 1983) it is expected that only fractions will be associated with the sediment loads, in contrast to P which is highly immobile. The bio-available P fraction in the Caledon River was found to be 106 mg/kg. This high concentration could be ascribed to agricultural run-off from Lesotho and Free State farmlands.

It was also observed that a larger increase in sediment concentration (up to 5000 mg/L) did not necessarily result in an increase in algal biomass due to more available nutrients as expected. This was due to extreme turbidity caused by such high concentrations, where light becomes the primary limiting factor and not nutrients.

8.3) Primary productivity in Knellpoort Dam

The primary productivity in Knellpoort Dam was determined at two sampling sites: a site in close proximity to the inflow of transferred water from the Caledon River (Sampling site 1) and a site in the middle of the impoundment (Sampling site 2). Two (2) primary productivity determinations were done, one during the transfer of water and the other five (5) months after the transfer of water has stopped.

At sampling site 1 during the transfer (turbid conditions), the light saturated rate of photosynthesis (P_{max}) was calculated as 9.94 mgC.m^{-3}.h^{-1}, and the photosynthetic capacity (P^{B}_{max}) was calculated to be 0.77 mgC.mgchl-a^{-1}.h^{-1}, while at site 2 the light saturated rate was 13.65 mg C.m^{-3}.h^{-1} and the photosynthetic capacity was 1.05 mgC.mgchl-a^{-1}.h^{-1}. This agrees with findings of Lloyd *et al.* (1987) who stated that a turbidity of only 5 NTU can decrease the primary productivity of shallow clear-water streams by about 3 – 13%, while an increase of 25 NTU may decrease primary productivity by 13 -50% .

The integral rate of photosynthesis for sampling site 1 was 7.3 mgC.m^{-2}.h^{-1} and at site 2 it was 11.6 mgC.m^{-2}.h^{-1} (37% higher). Furthermore, at sampling site 1, the surface inhibition was calculated to be 49%, while at sampling site 2 it was 89.7%. The increase in surface inhibition can be ascribed to the lower turbidity at sampling site 2, and can also clearly be seen in the differences between the two productivity profiles.

The primary productivity profiles showed marked differences between the two sampling sites. The profile was compressed for sampling site 1 (the turbid station), where P_{max} (maximum productivity) occurred at 0.25 m, while P_{max} at the clearer station occurred at 0.75 m. The euphotic zone was measured with the Secchi disk at about 0.65 m for Site 1 and 3.5 m for Site 2. This illustrates that phytoplankton primary productivity essentially follows the same pattern in turbid waters as in clear waters, except that productivity profiles are compressed, due to the rapid attenuation of light (Grobbelaar, 1985). In Lake Chapala, where nutrient concentrations were high and TN:TP was 1.47, the production rates at the depth of optimum irradiance (P_{opt}) were very high. These rates, which once exceeded 5 gC.m^{-3}.d^{-1}, are comparable or even higher than in most eutrophic lakes. However, because of the high turbidity, such production rates are restricted to a layer of only a few cm and total water column production is low (Lind *et al.*, 1997), although the productivity per unit area is the same than in clearer systems.

After the transfer of water was stopped, the P^{B}_{max} at site 1 was 1.31 mg C.mgchl-a^{-1}.h^{-1} and at site 2, it was 1.46 mg C.m^{-3}.h^{-1}. The light saturated rate of photosynthesis (P_{max}) for sampling site 1 was 7.33 mgC.m^{-3}.h^{-1} at sampling site 2 it was 6.8 mgC.m^{-3}.h^{-1}. Apart from the similar productivity values, the productivity profiles at the two sampling sites were also similar and P_{max} at both sites was measured at approximately 0.5 metres, with no compressed profile observed. The euphotic zone measured with the Secchi disk during this time, was approximately 2.45 metres at both the sites.

In comparison, primary production as determined by Grobbelaar (1992) in two impoundments in the Modder River, Mockes Dam and Krugersdrift Dam, showed that the light saturated rate of photosynthesis (P_{max}) in Mockes Dam varied from 11.63 mgC/m^3/h to 425 mgC/m^3/h, while in Krugersdrift Dam it varied from 5.14 mgC/m^3/h to 488.26 mgC/m^3/h. The average P_B^{max} in Krugersdrift Dam was almost twice that of Mockes Dam for the period before the flooding that occurred during the study period, but after the floods P_B^{max} was more than 1.5 times higher in Mockes than in Krugersdrift Dam. Before the floods Krugersdrift Dam was the clearer of the two impoundments, whereas after the floods Mockes Dam was a little clearer than Krugersdrift Dam. From this, he concluded that turbidity influences overall productivity, and that nutrients influence productivity only when a more favourable underwater light regime prevails, confirming the differences observed in Knellpoort Dam (in the turbid and clearer zones). The primary productivity values in Knellpoort Dam was, however, much lower than the maximum found by Grobbelaar (1992), but was in the same range than the minimum values. The highest P_{max} value obtained for Spioenkop, a turbid impoundment in the Thukela River, was 280 mg.m^{-3}.h^{-1} (Hart, 1999), much higher than in Knellpoort Dam.

Considering the light penetration, the attenuation coefficient at sampling site 1 during the transfer was calculated as 7.47 m^{-1}, while at sampling site 2 during this time it was 1.96 m^{-1}. This clearly illustrated the effect of the inflow of turbid water on the light penetration in Knellpoort Dam. The attenuation coefficient at sampling site 1 during October 2001, five months after the transfer was stopped, was 2.01 m^{-1}, and at sampling site 2 it was 1.99 m^{-1}. Thus, when no water was transferred, the attenuation coefficients were the same for both sampling sites (and similar to site 2 during the transfer), illustrating that light penetration is only influenced during the period that water is transferred, and that conditions return to that prior to the transfer. This is probably due to sedimentation of the particles over a period of time, when no transfer of water takes place.

8.4) Algal identification and enumeration

In the Caledon and Modder Rivers, and in Knellpoort and Rustfontein Dams, the phytoplankton community was represented by Euglenophyceae, Cyanophyceae, Bacillariophyceae, Chlorophyceae and Cryptophyceae. No Dinophyceae were present in any of the systems during the study period. The specific dominant algal species were also determined and is detailed in Tables 6.3 – 6.6.

The presence of the five (5) dominant algal groups in the *Novo* Transfer Scheme was similar to the Vaal River, South Africa, where the phytoplankton community was represented by six (6) algal groups: Dinophyceae, Euglenophyceae, Cyanophyceae, Bacillariophyceae, Chlorophyceae and Cryptophyceae (Pieterse, 1987).

There was greater variation in family domination in the Modder River compared to the other three systems of the *Novo* Transfer Scheme, which illustrates the more eutrophic conditions in this river. This is supported by a study conducted by Cottingham *et al.* (1998), who found that more than half of the species were turning over from one year to the next. This also suggests that the onset of enrichment promotes major changes in the presence and absence of particular members of the phytoplankton community.

Species composition in a phytoplankton community changes according to the succession stage. It has been observed that the appearance and succession of a certain phytoplankton species was closely related to the available nutrients (Kawabata & Kagawa, 1986; Tremel, 1996, Cottingham *et al*, 1998). This was also observed in Knellpoort Dam where cyanobacteria appeared after the transfer of nutrient-rich water from the Caledon River. However, diverse assemblages of phytoplankton are usually observed in natural waters, especially in oligotrophic regions, implying that phytoplankton can successfully co-exist while competing for a few limiting nutrients (Siegel, 1998).

8.5) Monitoring of macro-invertebrates

According to SASS5, the sampling sites in the different rivers were classified as:
Caledon River = Poor.
Rietspruit = Fair.
Modder River = Good.

The classification of the river sites as stated above, were not entirely in accordance with the findings in Chapter 5, where the Caledon River was classified as mesotrophic – eutrophic, and the Modder River as eutrophic in terms of both the average summer P concentrations (DWAF, 1996a). It is, however, not possible to assume correlations between SASS5 and chemical analyses, since one important limitation of the SASS5 method is that it does not identify the nature of the chemical changes.

It is not expected that there will be a transfer of undesirable invertebrate species from the Caledon River. Species diversity at the transfer site is generally low, and the high flow conditions during transfer periods will limit the species diversity even more. The main concern in terms of invertebrate diversity is the potential impact of increased flow in the Modder River during periods of transfer.

8.6) Other findings

In addition to the above results, some possible findings during this study were anticipated and stated as such in the introduction (Chapter 2). These included:

f) The withdrawal of water could have the effect of discontinuing the River Continuum, as in the case of dams.
g) It could cause major changes in flow and subsequently impact on fish and invertebrates in the system. The diversion could promote riverine ecological instability, while channelisation can remove/disadvantage some flora and fauna.
h) Changes in turbidity levels.
i) Changes in nutrient concentrations.
j) Another major problem is diversion induced erosion (Day *et al.*, 1982). In reservoirs, diversion channels, can lead to erosion, which in turn can lead locally to increased turbidity, degraded water quality, impaired habitats for predator fish species and loss of property and cultural artefacts.

Accordingly, the following was confirmed during the study:

a) Since the withdrawal of water from the Caledon River will only take place during the high flow season, it should not have a significant impact on this River Continuum. In addition, the high turbidity limits algal growth as well as invertebrate abundance.
b) It is anticipated that the release of a significant volumes of water from Knellpoort into the Modder River will result in a change in the flow of the river, with a subsequent change in the algal and invertebrate communities.
c) The transfer will result in a change in the turbidity in Knellpoort Dam, if water is transferred over a long period of time. However, since the abstraction point to the Modder River (from Knellpoort Dam) is on the opposite side of the impoundment, it will not influence turbidity levels in the Modder River.
d) Similar to the turbidity levels, the possibility certainly exists for a change in the nutrient concentration in Knellpoort Dam, which is currently low in nutrients, due to the inflow of more nutrient-rich water from the Caledon River. As stated above, the abstraction from

Knellpoort Dam into the Modder River is done from the other end of the impoundment, and this might dilute the nutrient-rich waters of the Modder River.

e) If large volumes of water are transferred to the Modder River, the possibility for erosion exists. It is therefore necessary to put measures in place to prevent this from occurring.

Based on the above, it is clear that the problems experienced by inter-basin transfer schemes across the world, also applies to the Novo transfer scheme. Unfortunately, this study was done after the scheme was already constructed, and therefore, mitigation measures will have to be reactive, instead of proactive (which is not the ideal situation).

Based on the results and findings, and the lessons learnt from similar systems across the world, the following recommendations should be implemented in order to minimise the impact the Novo transfer scheme has/may have on the different systems:

- Abstraction from the Caledon River must be done at the end of the rainy season, when turbidity levels are lower, but stream flow is still high enough.
- The sediment channels into the Knellpoort Dam are not very effective and alternative measures should be investigated to remove the sediment. These can include screens, or "snake channels" (channels that meander with slow-flowing water) that will remove more of the sediment. The sediment can also be removed at the Tienfontein pump station itself.
- The nutrient and turbidity status of the Knellpoort Dam must be carefully monitored to ensure that the impoundment does not become eutrophic or too turbid. The impoundment is used for the storing of "expensive water", and thus must maintain a certain level of ecological health. In addition, Knellpoort Dam is a popular recreational facility and pollution thereof will result in a loss of this income as well as the loss of a natural asset.
- A sampling point should be established in the Modder River, both upstream and downstream of the point of water release, in order to monitor changes in the system due to water transfer.
- When discharging the water from Knellpoort Dam into the Modder River, cognisance must be taken of the natural seasonal flow of the river, and care must be taken not to exceed the normal flow with too big a margin. Erosion measures must be put in place at the point of discharge and carefully monitored.

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SUMMARY

The Modder River is a relatively small river which drains an area of 7 960 km^2 in the central region of the Free State Province, South Africa and has a mean annual run-off of 184 x 10^6 m^3. Water demand in the Modder River catchment has exceeded supply in the past which necessitated the development of two transfer schemes:

1) From the Caledon to the Modder (Caledon-Bloemfontein pipeline), and
2) The Caledon to Modder also known as the *Novo* transfer scheme.

This study focussed on the ecological effects of the *Novo* transfer scheme on both the Modder and Caledon Rivers from a limnological point of view, thus excluding the human component and land-based investigations.

The transfer of water from the turbid Caledon River caused an increase in turbidity in Knellpoort Dam. However, this turbidity decreased when transfer of water was terminated, and conditions returned to those prior to the transfer of the water. This increase in turbidity brought about certain effects, like limitation of underwater light penetration, and a consequent decrease of the euphotic depth. The turbidity did, however, not extend to the inflow of the dam where the *Novo* pump station is located and is thus not expected to increase turbidity in the Modder River.

The largest fraction of sediment particles in the Caledon River, consists of very fine sand. This has implications in terms of the bioavialability of nutrients and consequently, the maximum bio-availability of nutrients in the sediment load of the Caledon River was found to 339 mg/kg N and 106 mg/kg P.

The transfer of water (and nutrient-rich sediments) from the Caledon River, via Knellpoort Dam, to the Modder River caused an increase in the concentrations of TP, PO_4-P, NO_3-N and SiO_2-Si in Knellpoort Dam. Since the effect of the inflowing water seems to be localised to the area at the inflow channel, the water quality at the *Novo* pump station (on the opposite side of the impoundment) was little influenced. However, if transfer takes place over a long period of time, it can have a significant impact on the nutrient concentrations of the impoundment. Similarly, if the ionic composition changes over a long period of water transfer, the algal communities in the different systems can change.

In the different waters of the *Novo* Transfer Scheme, there are large variations in the algal composition. In the Caledon River, the Bacillariophyceae (specifically *Cyclotella* and *Stephanodiscus* spp.) dominated the phytoplankton throughout the study period. Turbulence in the river, together with high turbidity, probably limited the development of cyanobacterial blooms. In Knellpoort Dam, the algal community was dominated by Bacillariophyceae and

Chlorophyceae, and Cyanophyceae numbers increased only after the transfer of nutrient-rich water from the Caledon River. This could have serious implications for the uses of the impoundment, which include water purification for drinking water and recreation, as cyanobacterial blooms can be harmful to humans and animals. In the Modder River, the algal community was dominated by Bacillariophyceae, Chlorophyceae and Cyanophyceae, and it was the same in Rustfontein Dam, an impoundment in this river used for domestic water supply and recreation.

As water was not transferred to the Modder River during the study period, the effects on the invertebrate community in this river could not be determined, and possible impacts are predicted on observations made from other studies. From these, it is not expected that there will be a transfer of undesirable invertebrate species from the Caledon River. Species diversity at the transfer site is generally low, and the high flow conditions during transfer periods will limit the species diversity even more. The main concern in terms of invertebrate diversity is the potential impact of increased flow in the Modder River during periods of transfer.

Based on this study, the following recommendations are made:

- Abstraction from the Caledon River must be done at the end of the rainy season, when turbidity levels are lower, but stream flow is still high enough.
- The sediment channels into the Knellpoort Dam are not very effective and alternative measures should be investigated to remove the sediment.
- The nutrient and turbidity status of the Knellpoort Dam must be carefully monitored to ensure that the impoundment does not become eutrophic or too turbid.
- When discharging the water from Knellpoort Dam into the Modder River, cognisance must be taken of the natural seasonal flow of the river, and care must be taken not to exceed the normal flow with too big a margin. Erosion measures must be put in place at the point of discharge and carefully monitored.

Key words: *Novo water transfer scheme, inter-basin transfer, turbidity, primary productivity, nutrients, invertebrates*

OPSOMMING

Die Modderrivier (met ‘n afvloei van 184 x 10^6 m^3 per jaar) is `n redelike klein rivier wat `n area van 7 960 km^2 in die sentrale deel van die Vrystaat, Suid-Afrika dreineer. In die verlede, het die aanvraag na water die voorsiening oorskrei, wat die volgende oordragskemas tot gevolg gehad het:

1) Van die Caledon na die Modderrivier (Caledon-Bloemfontein pyplyn), en
2) Van die Caledon na die Modderrivier (ook bekend as die *Novo* oordragskema).

Hierdie studie het gefokus op die ekologiese impak van die Novo oordragskema op beide die Caledon- en Modderrivier vanuit `n limnologiese oogpunt, en sluit dus die menslike en landelike aspekte uit.

Die oordrag van water vanuit die troebel Caledonrivier het ‘n toename in troebelheid in Knellpoort dam veroorsaak. Hierdie troebelheid het egter afgeneem sodra die oordrag van water opgehou het, en toestande het teruggekeer na dié van voor die oordrag. Die toename in troebelheid het sekere impakte tot gevolg gehad, soos die beperking van ligindringing met `n gevolglike afname in die eufotiese diepte. Die toename in troebelheid het egter nie gestrek tot by die invloei van die dam waar die *Novo* pompstasie geleë is nie, en daar word dus nie verwag dat die troebelheid in die Modderrivier beïnvloed sal word nie.

Die grootste gedeelte van die sedimentpartikels in die Caledonrivier bestaan uit baie fyn sand. Dit kan die bio-beskikbaarheid van voedingstowwe beïnvloed en gevolglik is die maksimum bio-skikbaarheid van N en P bepaal. Die resultate toon dat die maksimum bio-beskikbaarheid vanaf die sedimente 339 mg/kg N en 106 mg/kg P is .

Die oordrag van water (en voedingstofryke sedimente) van die Caledonrivier, via Knellpoort Dam, het `n toename veroorsaak in die konsentrasies van TP, PO_4-P, NO_3-N and SiO_2-Si in Knellpoort dam. Aangesien die effek van die invloei gelokaliseerd was in die area van invloei, is die waterkwaliteit by die *Novo* pompstasie (aan die oorkant van die dam) nie beïnvloed nie. As wateroordrag egter vir lang periodes aanhou, kan dit `n merkbare impak op die voedingstofkonsentrasies in Knellpoort dam hê. Indien die ioonsamestelling van die water verander, kan dit ook veranderinge veroorsaak in die alggemeenskap samestelling.

Daar is groot variasie in die algsamestelling van die verskillende stelsels van die *Novo* oordragskema. In die Caledonrivier, het die Bacillariophyceae (spesifiek *Cyclotella* en *Stephanodiscus* spp.) die fitoplanktongemeenskap gedomineer. Turbulensie in die rivier, tesame met hoë troebelheid, het waarskynlik die ontwikkeling van blou-groen alge beperk. In Knellpoort dam is die fitoplantongemeenskap gedomineer deur Bacillariophyceae en

Chlorophyceae, en Cyanophyceae het net toegeneem na die oordrag van voedingstofryke water vanaf die Caledonrivier. Dit mag ernstige gevolge inhou vir die menslike gebruik van die dam, wat drinkwater en ontspanning insluit, aangesien blou-groen algopbloeie skadelik kan wees vir mense en diere. In die Modderrivier, is die alggemeenskap gedomineer deur Bacillariophyceae, Chlorophyceae en Cyanophyceae, met dieselfde situasie in Rustfonteindam, wat gebou is in die rivier vir drinkwatervoorsiening en ontspanningsdoeleindes.

Aangesien water nie oorgedra is na die Modderrivier gedurende die studieperiode nie, is die impak op die invertebraat gemeenskap nie bepaal nie, en moontlike impakte is voorspel op grond van die resultate van ander studies. Vanuit hierdie resultate, word daar nie verwag dat ongewenste spesies oorgedra sal word vanaf die Caledonrivier na die Modderrivier nie. Spesie diversiteit by die oordragpunt is laag, en die hoë vloeistoestande gedurende oordragtye sal die spesie diversiteit nog verder verlaag. Die grootste verwagte impak op invertebraat diversiteit is die verhoogde vloei in die Modderrivier indien water daarna oorgedra word.

Gebaseer op die studie, word die volgende aanbevelings gemaak:

- Onttrekking vanuit die Caledonrivier moet gedoen word tydens die einde van die reënseisoen, wanneer troebelheid laer is, maar stroomvloei nog hoog genoeg.
- Die sedimentkanale wat in Knellpoortdam invloei is nie effektief nie en alternatiewe metodes moet ondersoek word om die sediment te verwyder.
- Die voedingstof- en troebelheidstatus van die Knellpoortdam moet noukeurig gemonitor word om te verseker dat die dam nie eutrofies of te troebel raak nie.
- As water vrygelaat word vanuit Knellpoortdam na die Modderrivier, moet die natuurlike vloei van die rivier in ag geneem word, en die normale vloei moet nie te veel oorskrei word nie. Maatreëls moet getref word om erosie te beperk by die punt van uitvloei en moet ook gemonitor word.

Sleutelwoorde: *Novo water oordragskema, inter-opvanggebied oordrag, troebelheid, primêre produktiwiteit, voedingstowwe, invertebrate*