LIMNOLOGICAL ASPECTS
OF
LOCH LOGAN,
AN URBAN IMPOUNDMENT

by

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CHAPTER 1

INTRODUCTION

1.1 GENERAL INTRODUCTION

Water is such a common substance that, like the air that is drawn into our lungs 30,000 times every day of our lives, we take it for granted. Vented upwards from deep within the earth through volcanoes, geysers and fumaroles during 5,000 million years of geological history, water has accumulated at the earth's surface to such an extent that it now covers 70 % of the surface to a mean depth of 3.8 kilometres. Forests and deserts have been created at will in response to past alterations in the distribution and abundance of water precipitated from the atmosphere as rain. On four occasions during the past 2 million years, vast expanses of water in the form of glacial ice overrode continents – depressing the land, sculpting valleys, depositing long mounds of gravel, damming lakes, recharting river courses and lowering the levels of the world's oceans by more than 100 meters (Vallentyne, 1974).

On land, water means food and survival when it comes at the right time: famine when it does not. In short, water is the chemical basis of all life – according to many, perhaps a universal requirement for the origin and persistence of life.

Water is unevenly distributed in nature. From **Table 1.1** one can see that inland water covers about 2 percent of the earth's surface (Wetzel, 1975) and that the proportion of water that is both fresh and liquid is very small indeed: just under 1 percent of the total. Of this, about a third ($4x10^6$ km^3) is surface water and the remainder ($1.1x10^6$ km^3) is groundwater (Davies & Day, 1998).

Table 1.1: Water in the Biosphere (Wetzel, 1975).

	Volume (thousands of km^3)	Percentage (%)	Renewal time (Retention time)
Oceans	1,370,000	97.61	37,000 years
Polar ice, Glaciers	29,000	2.08	16,000 years
Groundwater	4,000	0.29	300 years
Freshwater lakes	125	0.009	1-100 years
Saline lakes	104	0.008	10-1000 years
Soil & subsoil moisture	67	0.005	280 days
Rivers	1.2	0.00009	12-20 days
Atmospheric water (vapour)	14	0.0009	9 days

In addition to the uneven distribution of water over the earth's surface, the water of landmasses is not uniformly distributed over the major continents (**Table 1.2**). For example, the total groundwater runoff are the greatest in South America - nearly twice that per area of other continents – and with also the highest evaporation per area (Wetzel, 1975).

Among the continental landmasses, three hydrological regions have been recognised. The distribution of lakes is partly related to the distribution of lake basins and partly to that of water. Exhoreic regions, within which rivers originate and from which they flow to the sea, contain the major lake districts of the world and most of the lakes. Endorheic regions, within which rivers arise but never reach the sea, occur between subtropical deserts and the tropical and temperate humid regions. Arheic regions, within which no rivers arise, are desert areas that occur in the latitudes of the trade winds, and between which lies the zone of equatorial rains. With relatively small changes in climate endorheic regions, transitional in nature between the other two, can shift to exorheic of arheic characteristics (Wetzel, 1975).

Table 1.2: Water Resources and Annual Water Balance of the Continents of the World (Wetzel, 1975).

	Europe[a]	Asia	Africa	North America[b]	South America	Australia [c]	Total
Area (10^6 km^2)	9.8	45.0	30.3	20.7	17.8	8.7	132.3
Precipitation (km^3)	71,65	32,690	20,780	13,910	29,355	6,405	110,305
River runoff (km^3)							
Total	3,110	13,190	4,225	5,960	10,380	1,965	38,830
Underground	1,065	3,410	1,465	1,740	3,740	465	11,885
Surface	2,045	9,780	2,760	4,220	6,640	1,500	26,945
Total soil moistening	5,120	22,910	18,020	9,690	22,715	4,905	83,360
Evaporation	4,055	19,500	16,555	7,,950	18,975	4,440	71,475
Underground runoff (% of total)	34	26	35	32	36	24	31

a) Includes Ice land; b) Includes Central America, but not the Canadian archipelago; c) Includes New Zealand, Tasmania, and New Guinea.

1.1.1 WATER IN SOUTH AFRICA

South Africa ranges in climate from semi-arid to hyper-arid in the western part, with only a few relatively humid parts where the rainfall greatly exceeds 500 mm yr^{-1} in the eastern part. South Africa has an average rainfall of 452 mm yr^{-1}, but vast areas receive much less than this. Nowhere - except for a few tops in the Drakensberg and the south-western Cape - does rainfall exceed evaporation, while in many parts of the country evaporation outstrips precipitation by far. In the industrial heartland, Gauteng, evaporation is about twice as great as rainfall and in the lower Orange River valley it reaches a phenomenal value of more than ten times the rainfall. In simple terms, there is no such thing as a water surplus in South Africa. In fact, for most of Southern Africa, any rain that does reach the ground soon evaporates and re-enters the atmospheric phase of the water cycle (Davies & Day, 1998).

The distribution of rain in time and place is also marked. Firstly, rainfall is highly seasonal, being produced by weather systems in different regions at different times of the year. During the winter the prevailing north-westerly wind hit the western parts of the country first and drops much of its rain there, leaving the southern interior, including the Karoo, in a "rain shadow". In summer, rain normally falls in the north and the east while dry, high pressure air masses may persist for long periods in the south and west. Being dry, it cannot produce rain. Thus, South Africa experiences a wide range of climates (Davies & Day, 1998).

The water bodies can be divided into lentic (standing) and lotic (running) systems. Many South African rivers are intermitted of seasonal systems with no water flowing, or indeed with no water at all, for long periods. In the same vein, truly lentic systems traditionally comprise lakes and ponds, although in South Africa either type may dry up on occasion and therefore appear not to be there at all (Davies & Day, 1998).

Our country's population growth (which is estimated at 2.3 % per annum) is one of the highest in the world and will result in approximately 80 million people in South Africa by 2025. With all the above in mind, it is evident that the water supplies will not last long (**Figure 1.1**). At best, with the slowest estimated population growth and the smallest demand for water, supplies will no longer be able to meet the demand some time between 2020 (use of all surface water) and 2040 (use of surface and groundwater). At worst, with the highest possible population growth and water demand, supplies will be fully committed some time between 2003 and 2015. What is most disturbing is that even if we implement massive water-conservation measures, it will provide only a few years of grace before our general supplies will once again be inadequate (Davies & Day, 1998).

South Africa has many large reservoirs (dams) that store more than a million m^3 of water; a few even store close to 500 million m^3. However, this is not always enough. With the population growth around industrial areas more water is always needed, so inter-basin water transfer schemes are constructed to transfer water to these areas. There are several of these schemes in South Africa of which the Orange-Fish-Sunday and Lesotho Highlands schemes are some of the largest (Davies & Day, 1998). If there is no water left in South Africa, the transferring of water will have to come from neighbouring countries.

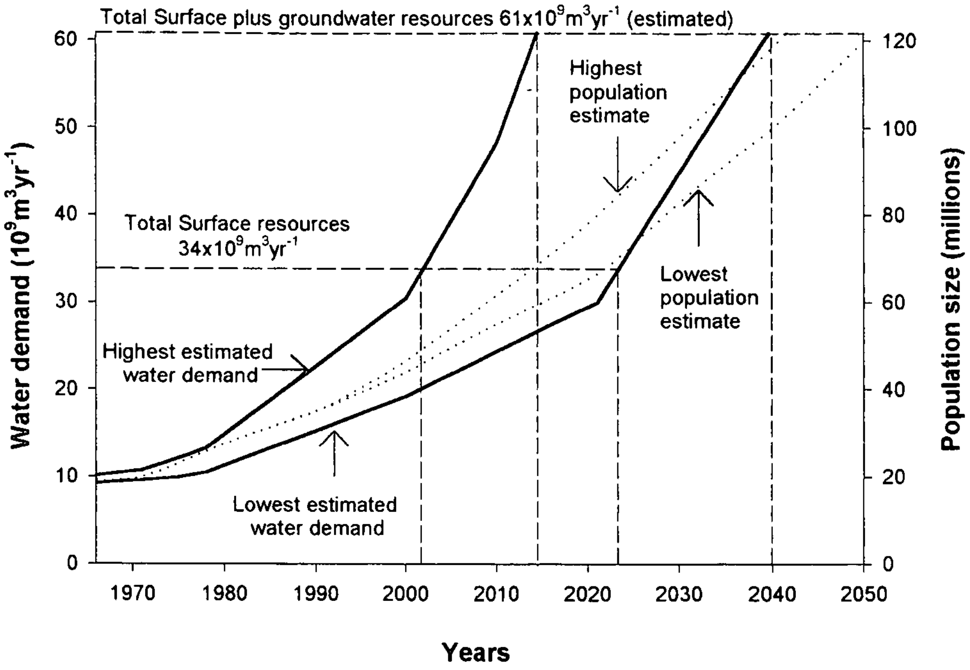


Figure 1.1: The relationship between demand for water and size of the human population of South Africa. The two dotted curves represent the fastest and slowest estimated rates of population growth. The two solid curves are the highest and lowest estimates of the amount of water needed to satisfy human requirements (redrawn from Davies & Day, 1998).

1.2 STRUCTURE OF LAKE ECOSYSTEMS

In a lake ecosystem there are four major elements that are part of the structure: morphometry, light and temperature zonation, chemical factors and biological zonation.

1.2.1 MORPHOMETRY

Although - beside their depth and surface area - it is not clear where ponds end and lakes begin, they share the same structural definitions (Horne & Goldman, 1994).

Surface water has two major regions. The littoral zone is the area from the shore to a depth where the mixed warm water still reaches the lakebed in summer. Beyond the littoral zone lies the open water, the pelagic or limnetic zone. This zone is characterised by the absence of lake bottom or shore (Horne & Goldman, 1994).

Important morphometric parameters most commonly used are:
Maximum length – the distance on the lake surface between the most distant points on the lakeshore.
Maximum width or breadth – the maximum distance on the lake surface at a right angle to the line of maximum length between the shores.
Area (A) – the area of the surface and each contour at depth z is best by planimetry.
Volume (V) – the volume of the basin is the integral of the areas of each stratum at successive depths from the surface to the point of maximum depth.
Maximum depth (z_m) – the greatest depth of the lake.
Mean depth ($\bar{z}$) – the volume divided by its surface area.
Relative depth (z_r)* – the maximum depth as a percentage of the mean diameter.
Shoreline (L) – the intersection of the land with permanent water is nearly constant in most natural lakes. The shoreline, however, can fluctuate widely in ephemeral lakes, and especially in reservoirs, in response to variations in precipitation and discharge.
Shoreline Development (D_L)* – the ratio of the length of the shoreline (L) to the length of the circumference of a circle of the area equal to that of the lake (Wetzel, 1975).

*($z_r = \frac{50 z_m \sqrt{\pi}}{\sqrt{A_0}}$; $D_L = \frac{L}{\sqrt{\pi A_0}}$) (Wetzel, 1975)

The areas of lakes are considered as the total area enclosed within the outline of the lake, including any islands. The number and size of the islands, if present, can have varying degrees of influence on the amount of water surface (Herdendorf, 1990). Thus, the more islands, the more the littoral zone is increased and the more the limnetic zone is reduced.

1.2.2 LIGHT AND TEMPERATURE

Light, temperature and wind mixing establish the second major element of lake structure. The penetration depth of sunlight divides the water mass in two. The upper, well-illuminated part is called the photic or euphotic zone; it extends from the surface downward to where light dims to about 1 percent of that at the surface. In this zone photosynthesis exceeds respiration during the day. The littoral and upper part of the limnetic zone is contained within the euphotic zone (Horne & Goldman, 1994).

The aphotic zone extends from below the photic and limnetic zone down to the bottom of the lake. In this area the light levels are too low for photosynthesis, so that oxygen is always consumed here by respiration. This region can also be called the profundal zone since it is usually deep. The layer between the euphotic and aphotic zones is called the dysphotic zone, and within these zones lies the compensation depth. In this area photosynthesis is equal to respiration (Horne & Goldman, 1994).

Temperature can also cause zonation and this usually occurs in the summer. It can only be noticed to its full extent in deep lakes. During thermal stratification three zones are present in temperate lakes (Horne & Goldman, 1994).

The upper warmer water is called the epilimnion, the middle part - where temperature changes with depth are the greatest - is the metalimnion (termocline) and the deepest portion is the hypolimnion. In summer the epilimnion can be completely or just partially mixed. During autumn and spring the water-column is turned over and well mixed from top to bottom. When a lake is destratifing and mixed from top to bottom it is termed holomictic (isothermally); if it is only partially mixed, it is called meromictic. If the lake is covered with ice in the winter, the ice prevents the wind from mixing the water (Horne & Goldman, 1994).

Furthermore there are different stratification patterns. Dimictic lakes mix twice a year, namely autumn and spring, and are covered with ice during the winter (Horne & Goldman, 1994). Monomictic lakes mix once a year. They fall into two categories: warm and cold monomictic. In warm monomictic lakes holomixis occur at temperatures above 4°C during winter (all monomictic lakes in South Africa, mostly large reservoirs, are of this type). In cold monomictic lakes, on the other hand, winter mixing occurs at temperatures below 4°C. These lakes are confined at high altitudes. Oligomictic lakes are either very small or very deep indeed. They are relatively warm at all depths and as such, even small temperature differences will produce very stable stratification. As a result, in the shallower systems that exhibit this characteristic, it is only during periods of unusually cold and windy conditions that overturn will occur (Davies & Day, 1998).

Polymictic lakes are shallow; they mix every few days or even daily all year round so that they may never exhibit thermal stratification at all. Amictic lakes, however, are covered with ice all year round and never mix (Horne & Goldman, 1994). Holomixis occurs when the whole lake mixes from bottom to top, whereas meromixis happens when only the top half mixes due to, for example, density differences between fresh and saline water. Depending on the type of mixing, the latter can be either ectogenic, crenogenic or biogenic meromixis (Wetzel, 1975).

The mixing of water by wind is done by seiches, turbulences and wave action. When wind blows over a lake and disturbs the flat surface, it starts a free oscillation movement in the water basin as it "seeks" to re-establish equilibrium after it has been displaced. By doing so it mixes the water (Davies & Day, 1998).

1.2.3 CHEMICAL FACTORS

The distribution of chemicals, especially nutrients, provides the third major element of lake structure. After the onset of thermal stratification, nutrients often become depleted in the epilimnion or euphotic zone, while at the same time remaining constant or even accumulating in the hypolimnion or aphotic zone. The depth, at which the rapid changes occur, is called the chemocline. In a few lakes this chemocline is a permanent feature. Usually chemical stratification is determined by thermal stratification. Thus, the permanent chemocline oven occurs in large, deep lakes with a permanent thermal stratification (Horne & Goldman, 1994).

In meromictic lakes the chemocline separates the monimolimnion (non-mixing bottom part) and the mixolimnion (upper mixing part) (Lampert & Sommer, 1997). The chemocline can have a steep salinity gradient (Wetzel, 1975).

1.2.4 BIOLOGICAL ZONATION

The fourth major element of lake structure is biological. Most organisms may be classified on the basis of their most common habitat. Plankton is the floating or weakly swimming organisms at the mercy of waves and currents. Zooplankton is the animal group and phytoplankton the algae. Other animals - such as fish - that inhabit the pelagic zone; are called the nekton. Neuston is the community of organisms that inhabits the surface water. A part of the neuston is the pleuston, which consists large floating assemblages blown about in the wind (Horne & Goldman, 1994).

In the littoral zone you found two groups; the attachment (epiphytic) algae and the epipelic algae. The former are attached to rocks or higher plants that provide a firm substrate; the latter are aquatic microphytes that dominate the sandy or muddy parts (Meybeck *et al.*, 1998). Organisms associated with the bottom of the lake are called benthic organisms or, collectively benthos. They are found submerged in the littorial, sublittorial and profundal zones. Organisms that move around on the bottom of a lake are called epibenthic organisms, while those that burrow beneath the mud surface are known as infauna. The epifloral habitat of submerged vegetation also provides a home for many types of algae and small invertebrates (Horne & Goldman, 1994).

1.3 CHEMICAL AND PHYSICAL CHARACTERISTICS

1.3.1 CHEMICAL CHARACTERISTICS

Age-old processes of rainfall, erosion and solution, evaporation and sedimentation regulate the chemical composition of natural waters (Horne & Goldman, 1994). The processes of solution, oxidation-reduction, the action of hydrogen ions and the formation of organic complexes control the organic weathering of soil and rock (Wetzel, 1975).

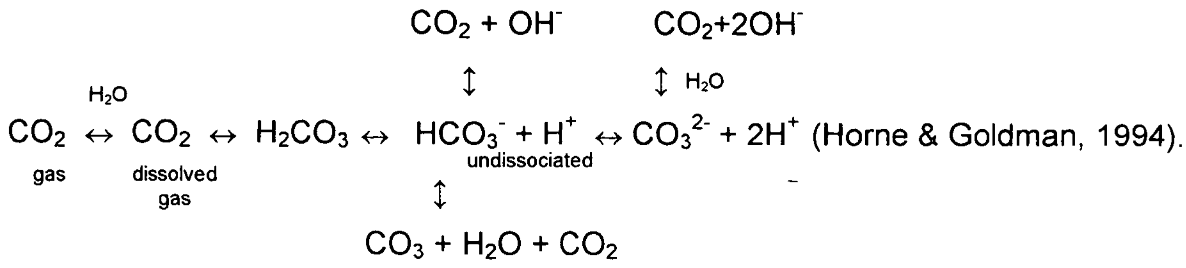
The total salinity of inland waters is usually dominated completely by four major cations, i.e. calcium (Ca^{2+}), magnesium (Mg^{2+}) sodium (Na^{+}), and potassium (K^{+}), and the major anions, i.e. carbonate (CO_3^{2-}), sulphates (SO_4^{2-}) and chlorides (Cl^{-}). The salinity is governed by contribution from the drainage basins rock runoff, atmospheric precipitation and the balance between evaporation and precipitation. A strong tendency exists for the proportions of major ions of surface water of the world towards Ca>Mg>Na>K and CO_3>SO_4>Cl (Wetzel, 1975).

The ionic composition of freshwater is dominated by diluted concentrations of alkaline earth components of bicarbonates, carbonates, sulphates, and chlorides. The concentration of four major cations, Ca^{2+}, Mg^{2+}, Na^{+}, K^{+} and four major anions, HCO_3^{-}, CO_3^{2-}, SO_4^{2-} and Cl^{-}, usually constitutes the total ionic salinity of water for all practical purposes (Wetzel, 1975).

The average salinity of surface water world-wide is about 120 mg/l (Wetzel, 1975). Water that contains less than 300 mg/l of salts (or total dissolved solids, TDS) can be considered to be fresh (the sea has about 35,000 mg/l salt). When almost saturated, water contains a salinity of about 350 g of salt per litre (Davies & Day, 1998).

To a large extent the pH of natural waters is governed by the interaction of H^{+} ions arising from the dissociation of H_2CO_3 (carbonic acid) and from OH^{-} ions, resulting from the hydrolysis of bicarbonate (Wetzel, 1975).

The pH is also strongly related to equilibrium processes of the carbon dioxide – bicarbonate system, including free carbon dioxide (CO_2), carbonic acid (H_2CO_3), bicarbonate ions (HCO_3^{-}) and carbonate ions (CO_3^{2-}). Atmospheric carbon dioxide is very soluble in water and when dissolved it is in equilibrium with carbonic acid (H_2CO_3) a weak acid (Brönmark & Hansson, 1998):



$$\underset{\text{gas}}{CO_2} \overset{H_2O}{\leftrightarrow} \underset{\text{dissolved gas}}{CO_2} \leftrightarrow H_2CO_3 \leftrightarrow \underset{\overset{\updownarrow}{CO_3 + H_2O + CO_2}}{\overset{\overset{CO_2 + OH^-}{\updownarrow}}{HCO_3^-}} + \underset{\text{undissociated}}{H^+} \leftrightarrow \overset{\overset{CO_2+2OH^-}{\updownarrow\ H_2O}}{CO_3^{2-}} + 2H^+$$

(Horne & Goldman, 1994).

At increasing pH levels, carbonic acid dissociates into a hydrogen ion (H^+) and a bicarbonate ion (HCO_3^-), which in turn dissociates into another H^+ ion and a carbonate ion (CO_3^{2-}). Photosynthesis and respiration are the major biological processes affecting pH by changing the amount of CO_2 in the water. The consumption of CO_2 During photosynthesis alters the equilibrium, causing an uptake of H^+ ions and a release of OH^- ions; thereby increasing the pH. If it is a calcium-rich region, photosynthesis may cause precipitation of calcium carbonate ($CaCO_3$). In soda lakes the sodium will replace the calcium and sodium carbonate (Na_2CO_3) will form (Brönmark & Hansson, 1998).

One main source of O_2 and carbon dioxide (CO_2) for all aquatic organisms is the atmosphere. Additional CO_2 is produced from organic carbon by respiration, especially in the sediments. Additional O_2 is released from the photolysis of water in the epilimnion during photosynthesis (Horne & Goldman, 1994). The depth distribution of free CO_2 is the mirror image of that of oxygen, because the photosynthetic reaction that produces oxygen in the epilimnion is reversed to favour respiration in the hypolimnion(Horne & Goldman, 1994).

In photosynthesis/respiration, $nCO_2 + H_2O \leftrightarrow (CH_2O)n + O_2$, n is usually 3, 6 or 12, e.g. pyruvate, glucose or sucrose. As a result of respiration and photosynthesis, the oxygen level in a waterbody will be the highest in the afternoon and the lowest in the morning before sunrise (Wetzel, 1975).

Not all elements are required in the same amount. Depending on the organism some elements can be more important than others (Horne & Goldman, 1994). The major elements found in water are Ca^{2+}, Mg^{2+}, Na^+ and K^+ cations, and SO_4^{2-}, Cl^-, CO_3^{2-} and HCO_3^- anions. Other than the major cations and anions, the iron (Fe^{2+}) ion can also be a major element, depending on the amount needed by the organism. If the organism is a diatom or Chrysophyte, silica (SiO_2) can also be regarded as a major element, but for all others it is a minor element (Horne & Goldman, 1994; Brönmark & Hansson, 1998).

The minor elements consist of nitrogen (N) as N_2 gas, nitrate (NO_3^-), nitrite (NO_2^-) and ammonia (NH_4^+) ions, phosphorus (P) as orthophosphate (PO_4^-), mono phosphates (HPO_4^{2-}) and dihydrogen phosphate ($H_2PO_4^-$), and silicon as amorphous silica (SiO_2) (Horne & Goldman, 1994; Brönmark & Hansson, 1998). The major and minor elements are also known as the macro nutrients (Salisbury & Ross, 1992).

The trace elements (micro nutrients) are those elements only needed in very small amounts like manganese (Mn^{2+}), zinc (Zn^{2+}), copper (Cu^{2+}), iron (Fe^{2+}), selenium (Se^{2+}) ions, etc. Some of these ions can be very toxic at high concentrations (Horne & Goldman, 1994).

Of the two nutrients, nitrogen (N) and phosphorus (P), nitrogen is the more abundant in nature (Davies & Day, 1998).

N and P are essential constituents of many biochemical processes. In water, N usually occurs in the form of gas, N_2 (Horne & Goldman, 1994), less abundant as NO_3^-, NO_2^- and NH_4^+ ions, and as a wide variety of nitrogen containing organic components. Nitrate (highly oxidised form of N_2) is seldom abundant in natural surface water, because it is incorporated into cells or is chemically reduced by microbes and converted into atmospheric nitrogen (denitrification). Nitrite is also intermediate in the interconversion of ammonia and nitrate, and is - even in low concentrations - toxic to aquatic organisms. Ammonia occurs in low concentrations in natural waters and is also a common pollutant associated - about five times more than P (Wetzel, 1975) - with sewage and industrial effluent (Davies & Day, 1998). Ammonia can occur either in the free unionised form (NH_3) or as ammonium ions (NH_4^+). In its unionised form, dependent on temperature and pH, ammonia is very toxic (Davies & Day, 1998). Some blue-green algae (Cyanobacteria) and bacteria are capable of nitrogen fixation, directly from atmospheric N_2 (Horne & Goldman, 1994).

Phosphorus is the most intensively studied element in limnology, especially its seasonal distribution in lakes (Wetzel, 1975). It is essential for all living organisms as it is an integral part of DNA (deoxyribonucleic acid) as well as RNA (ribonucleic acid).

In nature, inorganic phosphorus occurs almost entirely as the phosphate ion PO_4^{3-} (Horne & Goldman, 1994). Phosphorus gains access to water through sewage dumping in water, and agricultural runoff (Jones & Bowser, 1978). Beside external inflow of P, there is also internal loading from the sediments where PO_4^{3-} is released in the presence of Fe^{2+} ions and hydrogen sulphide (H_2S) during anoxic conditions. The Fe^{2+} binds with the PO_4^{3+} to precipitate as ferric phosphate ($Fe^{3+}PO_4$) (Horne & Goldman, 1994). This phosphorus zone in the lacustrine sediments may get as high as 0.75 % (by weight), but most values are less than 0.25 % (Jones & Bowser, 1978).

If too much nutrients enter a system so that it becomes over enriched and excessive algal blooms occur, the process is called eutrophication. This is usually a sign of degradation in water quality.

1.3.2 PHYSICAL CHARACTERISTICS

The load of suspended sediments primarily controls turbidity of water. Sediments consist of three primary components: (a) organic matter in various stages of decomposition; (b) particulate mineral matter, especially quartz; and (c) inorganic components of biogenic origin, e.g. diatom frustules and calcium carbonate (Wetzel, 1975).

With the decomposition of detritus by bacteria, using a number of different methods - methane fermentation, etc. - a variety of products form, for example methane (CH_4), CO_2 and H_2S. Therefore, O_2 is in great demand (Wetzel, 1975).

If there are high concentrations of nutrients, trace elements or biocides in the water, the sediments may remove them by adsorption. If their concentration is low, they tend to be released from the sediments into the water. Thus an increase in the amount of sediments may upset this equilibrium (Horne & Goldman, 1994). The exchange of phosphorus between the sediments and the overlaying water (internal loading) is a major component of the phosphorus cycle in natural waters (Wetzel, 1975).

Accumulation of sediments determines the life-span of a lake. The shallower the lake gets, the more its characteristics change (Davies & Day, 1998). Sedimentation cores contain information about the events that occurred in pre-cultural times in the lake and its catchment area, for example, historic development of metal pollution (Strumm & Baccini, 1978).

When the density of particulate matter suspended in the water (collectively termed seston) becomes great, a seston colour can be imparted to the water in spite of the relatively nonselective scattering properties of these materials. Different inorganic materials can change the water's colour (Wetzel, 1975). This can reduce the transparency of the water.

Suspended solids have an influence on the physical and chemical characteristics of an aquatic ecosystem. Firstly, large numbers of small particles suspended in water are visible as turbidity. The scattering of light is caused by any suspended matter, including clay, silts, finely dissolved organic and inorganic matter, plankton and other microscopic organisms, and as a result the water appear milky. The darkening or colouring of water is caused by absorption of light by dissolved substances such as humates (Davies & Day, 1998).

An Italian scientist, Secchi, invented a disk (secchi disk, black & white quadrants) (Horne & Goldman, 1994), that can be used to determine transparency as a function of the reflection of light from its surface (Wetzel, 1975). In many lakes the secchi depth is approximately one-third the depth of the photic zone (Horne & Goldman, 1994). When light penetration is reduced, less photosynthesis takes place (Brönmark & Hansson, 1998) and less plant material is produced. As a result of this, less food is available for organisms higher up the food chain (Davies & Day, 1998).

Temperature affects the solubility of oxygen nonlinearly; the concentration of oxygen soluble is inversely proportional to temperature (Horne & Goldman, 1994). Cold water increases it considerably, e.g. 0°C = 14.16 mg/l O_2, 10°C = 10.92 mg/l O_2, and 30°C = 7.53 mg/l O_2 (pure water, saturated air at 760mm Hg pressure) (Wetzel, 1975). Thus, at any given pressure, cold water contains a higher oxygen concentration when saturated than warm water (Horne & Goldman, 1994).

Together with light, temperature has an effect on photosynthesis: the colder the water the lower the photosynthetic rate (Wetzel, 1975). Temperature also influences other organisms' growth and reproduction, because with every 10°C increase in temperature all biochemical reactions double and vice versa (Davies & Day, 1998). This factor is known as the Q_{10} factor (Salisbury & Ross, 1992).

The colder the water, the denser it becomes and at 4°C water is the densest. However, if water is colder than this, it becomes less dense (Horne & Goldman, 1994; Davies & Day, 1998).

1.4 ORGANISMS IN LAKES

It is also appropriate to consider the plants, animals, fungi, and bacteria that give life and a great deal of interest to limnology. Algae are an important component of aquatic ecosystems, because through their density, abundance and diversity the health of their environment is reflected (see **Chapter 4**).

At the functional level organisms can be classified according to their sources of energy, sources of carbon and, in the case of some lower organisms such as sulphur bacteria, the molecule that serves as the electron donor. Classified on the basis of their energy sources are the phototrophs, which derive energy directly from sunlight in photosynthesis, and the chemotrophs, which utilise a chemical energy source. If the organism receives electrons from an organic compound it is an organotroph; if it derives them from inorganic matter it is a lithotroph (Horne & Goldman, 1994).

Organisms can also be classified on the basis of the type of carbon used for food. Autotrophic organisms (plants and chemolithotrophic bacteria) utilise inorganic carbon dioxide to produce organic matter, while heterotrophic organisms depend on deformed organic carbon such as glucose or pyruvate (Horne & Goldman, 1994). A large number of algae are heterotrophic, i.e. they can remain viable in bacterial-free cultures by means of the chemo-organotrophic uptake of dissolved organic compounds in the absence of light. However, their growth rates are lower (Wetzel, 1975). Higher in the food chain are hetrotrophic animals called herbivores, which utilise plant material for their growth, and higher-level hetrotrophs, carnivores, which feed on herbivores and, sometimes, each other. Omnivores eat both plants and animals and thus function at more than one trophic level (Horne & Goldman, 1994).

Aquatic organisms can be divided into several major groups: viruses, bacteria, fungi and fungilike organisms, algae and macrophytes, protozoans, rotifers, crustaceans, aquatic insects, worms and molluscs, and fish (Horne & Goldman, 1994; Brönmark & Hansson, 1998). Then there are also a few, like amphibians, reptiles, birds, and mammals, that require water during some stages of their life cycles (Horne & Goldman, 1994).

1.5 POLLUTING AGENTS

Water is an indispensable natural resource, fundamental to life, the environment, food production, hygiene and sanitation, industry and power generation (National State of the Environment Report - NSOER, 1999).

The world's available freshwater resources are already almost fully utilised and under stress. At the projected population growth and economic development rates, it is unlikely that the water resources in South Africa will be sustainable. Water will increasingly become the limiting resource in South Africa, and supply will become a major restriction of the future socio-economic development of the country, in terms of both the amount of water available and the quality of what is available (NSOER, 1999).

The scarcity of water is compounded by pollution of the surface and groundwater resources. Typical pollutants of South Africa's fresh water environment include industrial effluents, domestic and commercial sewage, mine drainage and agriculture and urban runoff and litter. As many of these sources are spread across the country, it is difficult to estimate the magnitude of the problem, though point sources can be identified and measured (NSOER, 1999).

Accordingly to NSOER (1999) three components have been chosen as indicators of water quality in South African surface waters: phosphorus and nitrogen, which are good indicators of the level of domestic pollution in surface waters, and total dissolved solids (TDS), which give an indication of the salinisation or salts content of the water.

Most of the major rivers in South Africa have an eutrophication (nutrient enrichment) problem (i.e. algal blooms, excessive macrophyte growth, unpleasant tastes and odours) because of nutrient enrichment from industrial, agricultural, etc. effluents. These all cause ecological changes to freshwater ecosystems (NSOER, 1999).

There are various responses at different levels in order to manage our water resources in a sustainable manner, including developing and adhering to international initiatives, setting relevant policy through legislation, implementing policy at an operational level (institutional arrangements, enforcement and monitoring) and implementing special programs to combat specific problems. The most important are the Water Services Act (Act 108 of 1997) and the National Water Act (Act 36 of 1998), which fall under authority of the Minister of Water Affairs and Forestry (NSOER, 1999).

Despite of all our policies and water acts, one of our biggest pollution problems is runoff water (urban and agricultural), also known as stormwater. Urban and suburban development, i.e. the creation of buildings and roads and the innumerable related activities, turns rain and snow into unwitting agents that damage our national waterways (National Resource Defence Council - NRDC, 1999).

In this runoff water hazardous materials can be found, like: sediments, toxic metal particles, fertiliser, oil, petroleum and grease, harmful bacteria and viruses, excess nutrients (**Table 1.3**) and pesticides (**Table 1.4**). Pollution released into the air - by transportation (**Table 1.5**), for example - fall on surfaces and are washed into water systems by rain (NRDC, 1999).

Table 1.3: Principal pollutants in stormwater (NRDC, 1999).

Category	Examples
Metals	zinc, cadmium, copper, chromium, arsenic, lead
Organic chemicals	pesticides, oil, petroleum, grease
Pathogens	viruses, bacteria, protozoa
Nutrients	nitrogen, phosphorus
Biochemical oxygen demand (BOD)	grass clippings, fallen leaves, hydrocarbons, human and animal waste
Sediment	sand, soil, silt
Salts	sodium chloride, calcium chloride

Table 1.4: Six pesticides found frequently in stormwater samples (NRDC, 1999).

Pesticide name	Human health and/or Environmental effects
2,4-D	Associated with lymphoma in humans; testicular toxicant in animals.
Chlorpyrifos	Moderately toxic to humans; neurotoxicant; can be highly toxic to birds, aquatic organisms and wildlife.
Diazinon	Moderately toxic to humans; neurotoxicant; can be highly toxic to birds, aquatic organisms and wildlife.
Dicamba	Neurotoxicant; reproductive toxicity in animals; association with lymphoma in some human studies.
MCPA (Methoxane)	Low toxicity to non-toxic in test animals, birds and fish; suspected gastrointestinal, liver and kidney toxicant.
MCCPP (Mecoprop)	Slightly to moderately toxic; some reproductive effects in dogs; suspected cardiovascular, blood, gastrointestical, liver and neurotoxicant.

Table 1.5: Source of heavy metals from transportation (NRDC, 1999).

Source	**Cd**	**Co**	**Cr**	**Cu**	**Fe**	**Mn**	**Ni**	**Pb**	**Zn**
Petroleum	∘			∘				∘	∘
Exhaust							∘	∘	
Motor oil & grease		∘			∘		∘	∘	∘
Antifreeze					∘				∘
Undercoating								∘	∘
Brake lining				∘	∘		∘	∘	∘
Rubber	∘			∘				∘	∘
Asphalt				∘			∘		∘
Concrete				∘			∘		∘
Diesel oil	∘								
Engine wear					∘	∘	∘	∘	∘

Consequences of this urban stormwater pollution are as follows: flooding and property damage; stream bank and bed erosion; siltation and sedimentation; increased water temperatures; harm to aquatic life, coastal shellfisheries and sport fishing; human illness; impacts on drinking water supplies and aesthetic losses (NRDC, 1999).

1.6 MOTIVATION AND OBJECTIVES FOR THIS STUDY

Impoundments in South African towns and cities are popular recreational attractions that add to quality of life experienced by urban residents. Impoundments are also being increasingly built as the focal point of urban commercial developments. Such impoundments, however, are receptacles for polluted runoff and discharges from upstream urban areas. The pollution frequently discharged to receiving streams can result in water quality problems in the impoundments. This does not only reduces their aesthetic value, but also undermines their role as centres for recreational activities. The importance of managing freshwater ecosystems is becoming more evident every year (Freeman *et al.*, 2000).

Loch Logan is an urban impoundment. Since it was built, it played an important role in the lives of the members of Bloemfontein's community. Its shores became popular picnic spots and canoeists practised their skills in the water. In 1997 urban developers started with the development of the Loch Logan Waterfront, which attracts more people to the area.

The addition of water to Loch Logan is through seepage water, rain and the stormwater that drain into Bloemspruit, which is largely condensed. The water that discharge through the canal into Loch Logan usually contains high concentrations of pollutants. Other pollution sources of Loch Logan are drain overflows in the catchment area, and rubbish and other waste materials dragged into the canal by street children and homeless people who live there. Urban runoff is a potential threat to the environment both as a source of harmful and toxic elements and as a source of plant nutrients, which promote eutrophication in waters. The high nutrient concentrations stimulate algal growth. These blooms (usually more than 60 µg/l Chl-*a*) cause foul odours and fish kills (due to oxygen deprivation), which discourage people form visiting Loch Logan. Baseline limnological monitoring is required to define the problem scientifically.

Objectives of the study:

1. to systematically describe the physical, chemical and biological features of Loch Logan,
2. to determine the seasonal and diel patterns of environmental parameters in Loch Logan,
3. to determine the trophic status of the waterbody and possible remedial actions,
4. to identify organisms causing algal blooms and the environmental variables responsible for their development and decline,
5. to investigate the composition and concentration of algae present at various times (seasonal) throughout the year,
6. to established whether the water quality of Loch Logan meets the desired water quality guidelines as prescribed by the Department of Water Affairs and Forestry (DWAF),
7. to put environmental variables measured in Loch Logan in perspective and to determine whether different systems have any characteristics in common,

using information from the literature and comparisons drawn between different aquatic ecosystems (especially urban impoundments), and

8. to contribute towards the scientific basis for the control of quality and resource management of urban impoundments.

The information collected from this study can serve as a basis for a management plan that could control the water quality and protect the biodiversity of the aquatic system.

CHAPTER 2

STUDY SITE

2.1 INTRODUCTION

An impoundment is a man-made or natural lake, pan or vlei, which has become surrounded by urban development (Freeman *et al.*, 2000).

Impoundments in South African towns and cities are not recent developments. For example, Zoo Lake in Johannesburg is a man-made impoundment that was built during the first decade of the 20th century. Additionally, natural water bodies, such as pans and vleis, have been gradually surrounded by urban development in certain cities (e.g. North End Lake, Port Elizabeth) and these have also become important aesthetic and recreational attractions for urban residents. More recently, commercial and recreational developments have increasingly been built with impoundments as their central feature, e.g. Bruma Lake and the Randburg Waterfront in Johannesburg (Freeman *et al.*, 2000).

Local authorities provide urban impoundments primarily for recreational purposes, stormwater control, and to improve the psychological well-being of city dwellers by relieving the pressure of modern urban life. In addition, urban water bodies are increasingly being developed because they enhance the value of real estate, houses, office blocks and commercial developments in their immediate vicinity (Wiechers *et al.*, 1996).

Water quality problems associated with urban impoundments are also not something new - Jan Smuts Dam in Brakpan has a documented record of such problems stretching back to 1940. Unfortunately many urban impoundments act as receptacles for upstream waste, resulting in silted up impoundments, the impoundments' enrichment with plant nutrients which often leads to the associated growth of undesirable algae and water plants (referred to as eutrophication), increased health risks due to bacteriological contamination; and aesthetic problems such as unsightly scum, floating debris and malodours (Freeman *et al.,* 2000).

Because of these problems, what should be a public asset can turn into a liability. Even more seriously, it can pose a health risk. Such water quality problems also tend to be accentuated in urban areas where man's activities are wide-ranging, densely concentrated and frequently culminate in the generation of numerous waste streams, which may enter watercourses (Freeman *et al.*, 2000).

2.2 LOCH LOGAN

Loch Logan was built in one of the canals of the Bloemspruit, in the Westdene area near the city centre of Bloemfontein. It is this canal that feeds Loch Logan with runoff water collected from the urban areas (~ 16 $\times 10^6$ m^2) it runs through. In 1997 a "waterfront" was developed on the banks of Loch Logan. The "waterfront" consists of coffee shops, restaurants, pubs, shops, a movie theatre and a gym on the eastern bank. The Bloemfontein Zoo's entrance is a couple of meters from the western bank with Kings Park Rose garden, where the monthly flee market is held, on the south-western bank. On the island there are braai areas and a performance stage (**Figure 2.1**).

Loch Logan's grid reference is 29° 06' 845" S and 26° 12' 505" E. It is located in a summer rainfall area, which receives between 500-700 mm per annum (Davies & Day, 1998), half of which is through thunderstorms (Koning, 1998). However, up to 57.9 mm, have fallen well into May (1999). In comparison with the average rainfall of South Africa, i.e. 452 mm per annum (Davies & Day, 1998), the Loch Logan area received an average of 558.9 mm/a over the past 10 years, and in 2000 and 2001 respectively 644.8 mm and 761.2 mm (SA Weather Bureau).

This rain is the runoff/storm water that is canalised to Loch Logan. The canal enters Loch Logan at its north-western side, opposite the impoundment wall. Eventually Bloemspruit flows into the Renosterspruit (about 12 km downstream from Loch Logan), which ends up in the Modder River near Glen Agricultural College above Glen Barrage.

Loch Logan has a volume of about 95,000 m^3, an area of approximately 42,115 m^2 - 4.2 ha, (the island excluded) - and a mean depth of 2.26 m (maximum depth = 3.8 m). In the middle of Loch Logan is an island that divides the water mass in almost two equal sized arms. The shoreline length of Loch Logan is 1,053 m, and that of the island is 559.8 m. The wall is situated on the south-eastern side of Loch Logan (**Figure 2.1**).

The sampling sites were (1) surface water and (2) bottom water at the south side of Loch Logan, just west of the wall, (3) at the canal inflow, and (4) in the canal at the bridge in Henry Street (**Figure 2.2**). **Photo's 1** to **4** show sampling sites and surroundings. The sites were selected due to easy access from the shore. Sampling site 1 was selected for an overall of Loch Logan's water, site 2 because of its depth of ± 3.5 m to show the difference in surface and bottom water quality, site 3 for the transition between the canal and Loch Logan, and site 4 to show the quality of the water that enters Loch Logan.

Wastes flush into Loch Logan with rainstorms (**Photo's 6** to **10**) and contribute to algal blooms when organic decomposed materials and inorganic nutrients released into the water (**Photo's 11** & **12**).

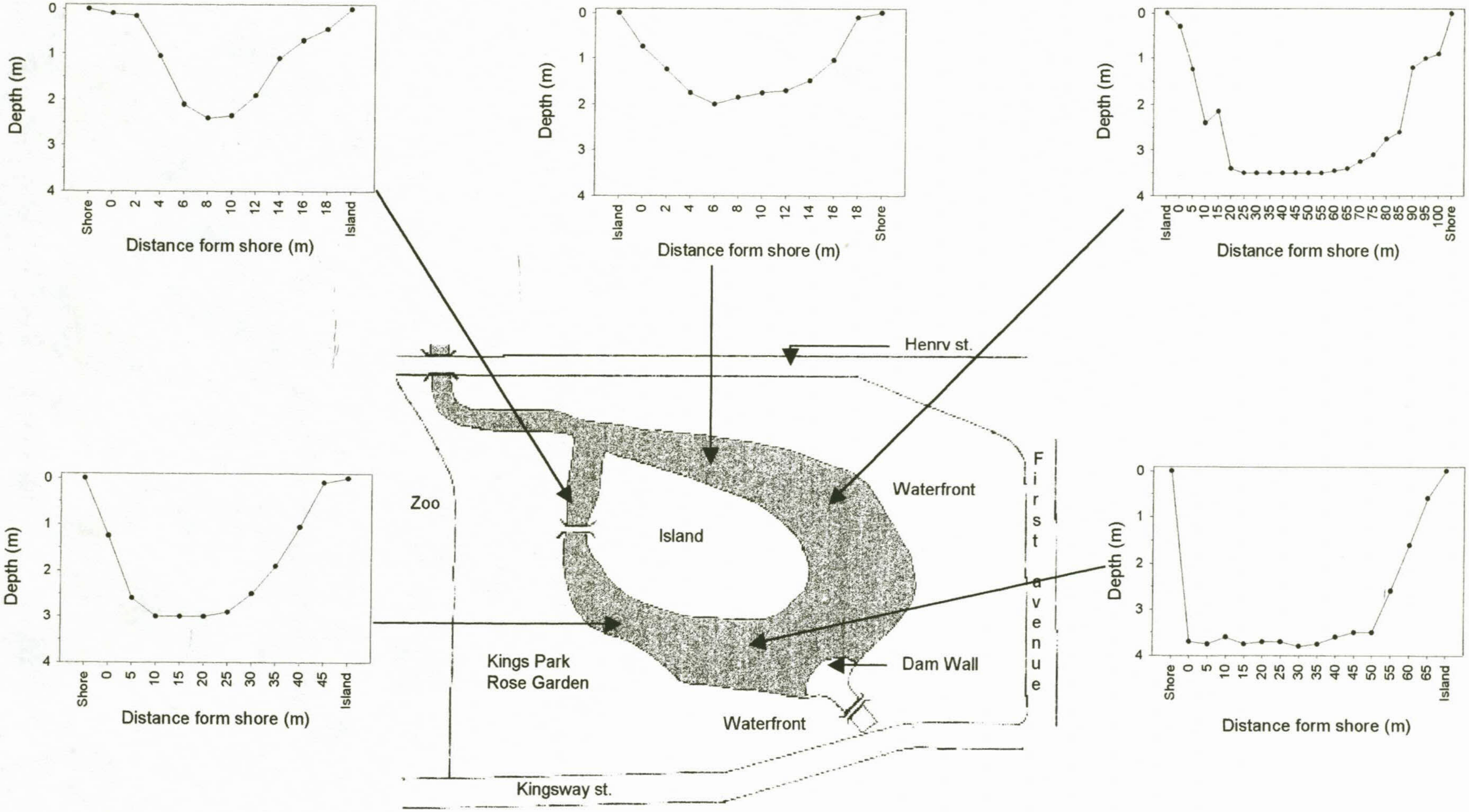


Figure 2.1: Loch Logan and cross section depth profiles at five points.

Zero (0) on graphs indicate depth just off shore (~10 cm).

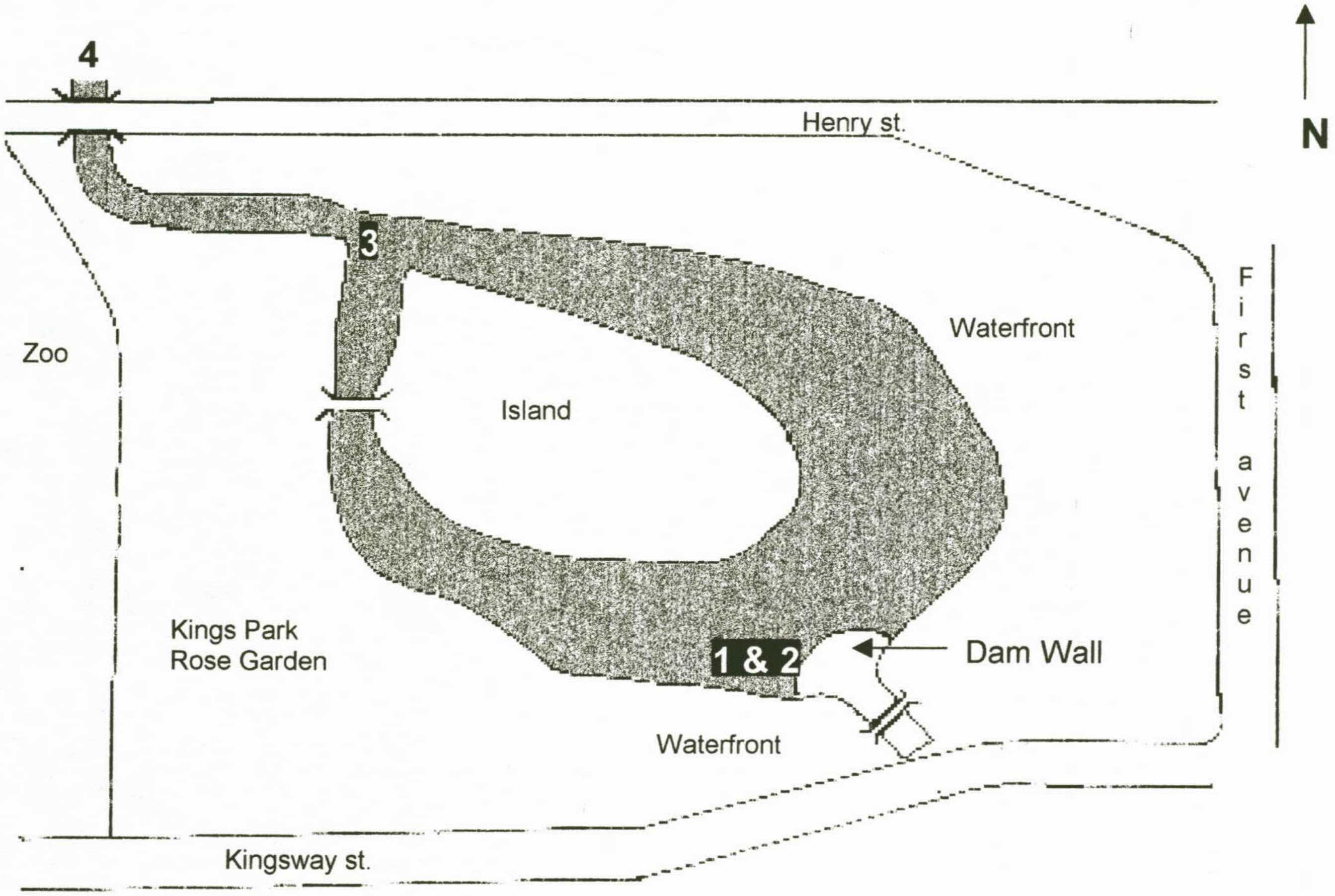


Figure 2.2: The four sampling points in Loch Logan: (1) Dam Wall (DW) Surface, (2) Dam Wall Bottom, (3) Inflow, and (4) Bridge (in the canal).

Photo 1: The Dam Wall (DW) sampling point.

Photo 2: The Inflow sampling point – next to the tree.

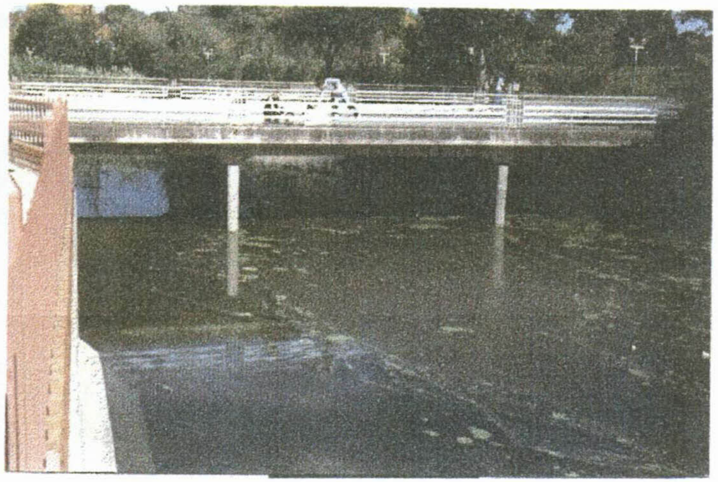


Photo 3: The Bridge sampling point, in canal.

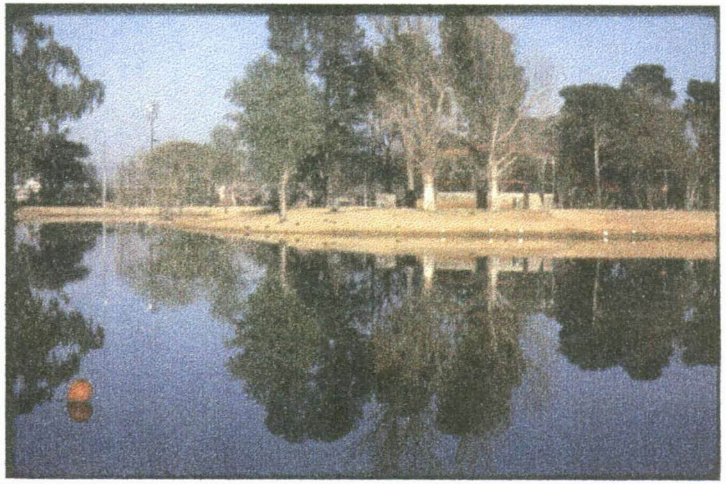


Photo 4: The island in the middle of Loch Logan.

Photo 5: A drain spill in the catchment area of Loch Logan.

Photo 6: Domestic waste in the canal above Loch Logan.

Photo 7: A waste pipe leaking into the canal.

Photo 8: Depicting the situation in the canal at Loch Logan.

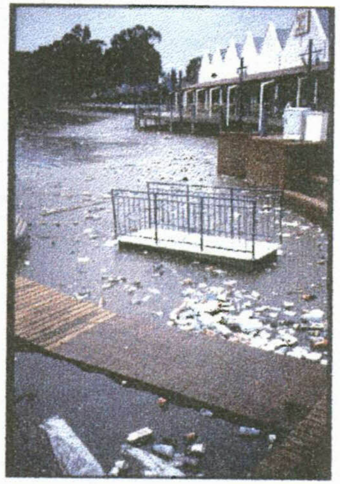


Photo 9: Waste at the north-eastern side of Loch Logan after rain.

Photo 10: Close-up of the waste in Loch Logan.

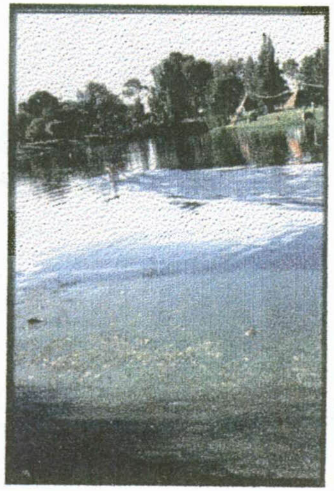


Photo 11: An algal bloom (cyanobacteria) in Loch Logan.

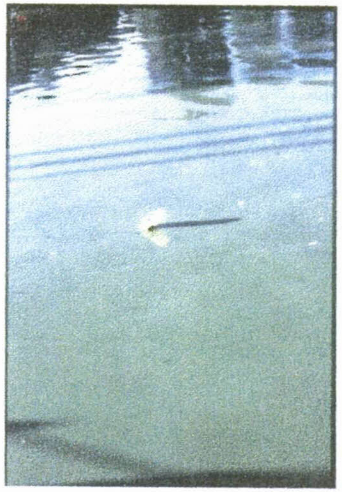


Photo 12: A close-up of the algal bloom in Loch Logan.

CHAPTER 3

PHYSICAL AND CHEMICAL CHARACTERISTICS OF LOCH LOGAN

3.1 INTRODUCTION

The physical attributes and chemical constituents of natural freshwaters differ from continent to continent - even from region to region - because they are regionally influenced by climate, geomorphology, geology, soils as well as by aquatic and terrestrial biota living in a particular area (Davies & Day, 1998).

Age-old processes of rainfall, erosion and solution, evaporation, and sedimentation (Horne & Goldman, 1994) regulate the chemical composition of natural waters. Climate affects water quality in a number of ways. For instance, temperature determines the rate and extent of various chemical interactions. Mean annual rainfall and seasonal differences in rainfall, determine the amount of water flowing in rivers or entering wetlands and lakes at different times of the year. Therefore, these factors also determine the degree of dilution of natural chemical constituents and of pollutants. Evaporation, on the other hand, concentrates substances in water (Davies & Day, 1998).

It has been well known even before 1960 that urban stormwater discharge contains high concentrations of a wide variety of potentially toxic chemicals. However, the chemical forms in which those contaminants exist and the duration of exposure that aquatic life would receive from such discharges are such that it would be expected to be indeed rare that contaminants in urban stormwater from residential and most commercial, industrial, and construction site areas would cause an impairment of the designated beneficial uses of water bodies (Lee & Jones-Lee, 1993).

In a report of the United States Environmental Protection Agency (US EPA), they indicated that roughly 30 to 40 percent of assessed rivers and estuaries were not supporting the uses for which they were designated (Lee & Jones-Lee, 1993). Therefore it can be assume that this applies to some lakes and dams as well.

Discharges of inorganic and organic compounds from domestic, agricultural and industrial effluents advance eutrophication of receiving waters. These wastewaters cause various pollution effects due to their high nitrogen (N) and phosphorus (P) concentrations, suspended solids, biological oxygen demand (BOD_5) and chemical oxygen demand (COD) (Rai & Jacobsen, 1990).

N and P are respectively the fourth and sixteenth most abundant elements in our solar system. In nature they are present in a N:P mass ratio estimated between 192 and 660. In contrast, the earth's crustal rocks are relatively poor in N, resulting in N:P that only varies between 0.01 and 0.8 by mass. The relative abundance of N and P found suspended or dissolved in lakes has been the subject of much discussion lately (Downing & McCauley, 1992) (**Table 3.1**).

Table 3.1: Demand and supply of selected essential elements in freshwater (Freedman, 1995).

Elements	**Concentration in plants (%) (demand)**	**Concentration in water (%) (supply)**	**Ratio of plants:water (approx.)**
Carbon	6.5	0.0012	5,000
Silicon	1.3	0.00065	2,000
Nitrogen	0.7	0.000023	30,000
Potassium	0.3	0.0023	1,300
Phosphorus	0.08	0.000001	80,000

Off all the nutrients such as C, N, P and micro-nutrients, P seems to be the most limiting factor in freshwater (Marzadori *et al.*, 1998). That is because the theoretically optimum N:P ratio for phytoplankton to grow, is 7:1 by mass and 16:1 by atom ratio (Horne & Goldman, 1994). However, if the N:P ratio is <10:1, N is probably the limiting factor. If it's >10:1, P is limiting (Barica, 1990)

This loading of nutrients (N and P) through pollution lead to eutrophication, which accelerates because of human activity. Pollution is an undesirable change in the physical, chemical or biological characteristics of air, water, soil, or food that can adversely affect the health, survival or activities of humans or other living organisms (Miller, 2002). Eutrophication - one of the biggest water quality problems world-wide - increases phytoplankton biomass, the major symptom of eutrophication, which leads to decreasing water clarity and oxygen levels in deeper parts. It also results in fish kills and taste and odour become a problem (Elizabeth *et al.,* 1992).

Because algae also use oxygen (O_2) for their metabolical processes (respiration), they deplete the O_2 concentration in the water during the night when they are not photosynthesising. Together with the decomposition microbes, a lot of O_2 is used and so the O_2 concentration in the water decreases. After total O_2 depletion, some microbes can use other electron acceptors such as NO_3-N, Fe^{3+}, Mn^{4+}, SO^{2-} and CO_2 for respiration (Moore *et al.,* 1992).

It is not just eutrophication that plays a part in O_2 depletion; temperature is also a factor. The higher the temperature, the less O_2 dissolve in the water. Temperature also plays a part in algal blooms. Furthermore, specific algae appear in specific seasons, for example: in Lake Okeechobee (Florida) cyanobacteria dominate in the summer, while diatoms dominate in the winter. In Lake Oglethope (Georgia) the winter algae consist of diatoms, green algae and flagellates, while the summer algae consists mostly of cyanobacteria, large dinoflagellates and euglenoides (Grover *et al.*, 1999).

Eutrophication also has an impact on the microbial loop, which is the carbon flow from dead material through microbes to plankton, invertebrates and fish. In highly eutrophic lakes, cyanobacteria mostly dominate. Most cyanobacteria are not prayed upon by zooplankton, invertebrates or fish, but are directly decomposed by microbes. This way the nutrient concentrations increase in the water and sediments, and more algal blooms can be decomposed (Nixdorf & Arndt, 1993).

Except for this internal loading, P is also released from the sediment. Because P has no gaseous form, everything ends up in the water of the lake (Kelly & Smith, 1996).

That is why eutrophic lakes show that much resistance to recovery even when the nutrient loading is been reduced (Janse *et al.,* 1995).

Aspects of the influence of pollution on physical and chemical characteristics of Loch Logan have been investigated – see **1.6 Motivation and objectives for this study**.

3.2 MATERIALS AND METHODS

Sampling took place once a week on Mondays (as far as possible) for a time period of 17 months, i.e., 24 January 2000 to 28 May 2001. *In situ* measurements were made and subsurface samples (1 litre) were taken – from shore, except the bridge sample was from the middle of the canal - and brought to the laboratory for chemical analysis. The analyses were done directly after arriving at the laboratory.

The water temperature (°C), concentration of dissolved oxygen (mg/l) and percentage of saturation were measured with an YSI Model 58 dissolved oxygen meter and were done *in situ.* The mean air temperatures for 2000-2001 were obtained from the SA Weather Bureau. Turbidity, a measurement of the concentration of suspended organic, inorganic and biological material in the water (clarity), was determined with an Aqua Lytic Turbidimeter AL 1000 and is expressed as Nephelometric Turbidity Unit (NTU).

A Secchi-disk was used to determine the light penetration depth of the water. Rainfall data for the Bloemfontein area for 2000-2001 were also obtained from the SA Weather Bureau.

To determine the pH and redox potential of the water the HANNA HI 9023 MICROCOMPUTER pH meter was used. These measurements were done *in situ*.

Conductivity (which serves as an indicator of the dissolved salts in the water and is expressed as mS/m) and total dissolved solids (TDS) (mg/l) were measured *in situ* using the HANNA HI 9811 pH-EC-TDS meter.

For ammonium-nitrogen (NH_4-N) (indication of eutrophication) determination, 50 ml unfiltered water was used with the Phenate Method as described in *Standard methods for the examination of water and wastewater* (1995). An intensely blue compound, indolphenol, is formed by the reaction of ammonia, hypochlorite and phenol catalysed by sodium nitroprusside. Absorbency* was read at 640nm.

Nitrate-nitrogen (NO_3-N) (indication of eutrophication) was determined by the use of 10ml GF/C filtered water together with the Brucine Method as described in *Analytical chemistry* (Jenkins & Medsker, 1963). The reactions of brucine and strychnine compounds with nitrate (dissolved by sulphuric acid) developed to a yellow brucine nitrate colour.
Absorbency* was read at 410nm.

Dissolved reactive *ortho*-phosphate (PO_4-P) (indication of eutrophication) was determined using 100ml GF/C filtered water together with the Stannous Chloride Method as described in *Standard methods* (1995). Ammonium molybdate reacts with stannous chloride, whereby molybdophosphoric acid is formed and reduced by stannous chloride to intensely coloured molybdenum blue. Absorbency* was read at 690nm.

Silica-silicon (SiO_2-Si) was determined by using 50ml GF/C filtered water with the Molybdosilicate Method as described in *Standard methods* (1995). Ammonium molybdate at pH 1.2 reacts with silica and a yellow colour is formed, of which the intensity is proportional to the concentration of “molybdate-reactive” silica. Absorbency* was read at 410nm.

*All absorbencies were determined with the Varian Cary 3 UV-Visible spectrophotometer, where after the unknown concentrations were determined by plotting it against a standard curve of known concentrations for each of the analyses.

3.3 RESULTS

3.3.1 TURBIDITY

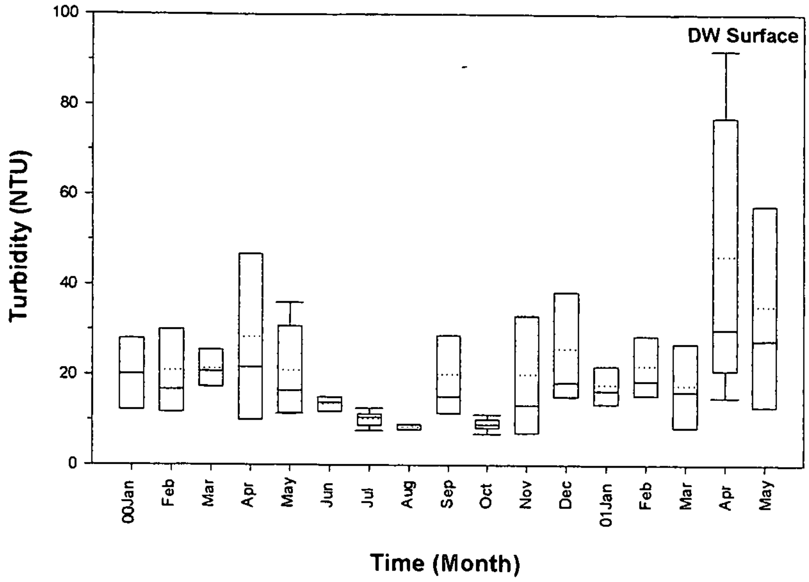
The turbidity in Loch Logan ranged between 0.04 and 92 NTUs for the surface water, with the average at 22.2 NTUs. The maximum turbidity for the bottom water was very high at 326 NTUs. The average turbidity in the bottom water was about three times higher than that of the surface water. There is not much difference in the turbidity of the surface sampling points (**Table 3.2**).

Table 3.2: Minimum, maximum and mean turbidity at the different sampling points in Loch Logan during the study period.

Sampling points	**Turbidity (NTU)**		
	Min	**Max**	**Mean**
Wall Surface	0.04	92.0	21.5
Wall Bottom	8.8	326.0	66.4
Inflow	6.4	80.0	23.32
Bridge	0.05	80.0	21.82

The turbidity increased during the rainy season (January 2000 to April 2000 and September 2000 to May 2001), but was relatively low during the winter period in 2000 (<20 NTUs, except for the Bottom, which was <30 NTUs). The turbidity in the bottom water (Wall Bottom sample) was always higher than that of the others (**Figure 3.1**).

The turbidity readings were the highest shortly after a rainstorm. With rain over the weekends the water was more turbid than when it rained earlier the week and readings were taken a few days later. There was an 87.0 % ($r^2 = 0.87$; $p < 0.001$) linear relationship between the Secchi depth and turbidity (**Figure 3.2**).



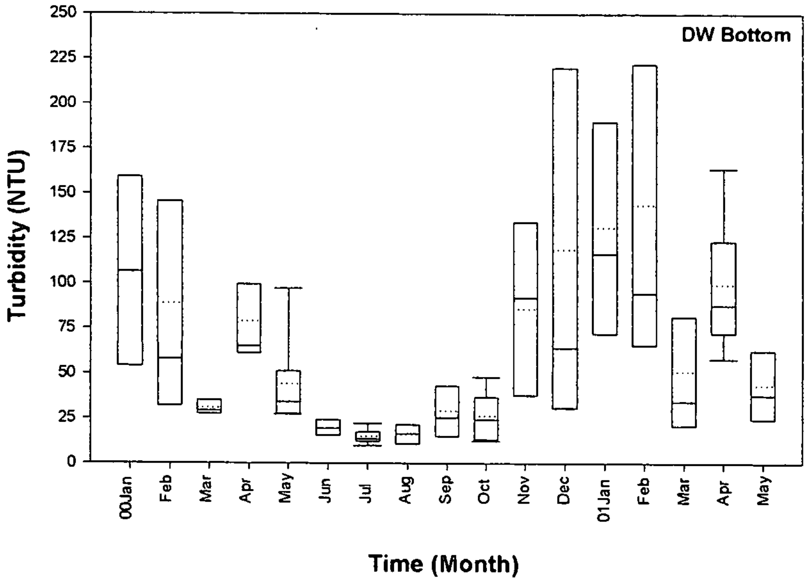
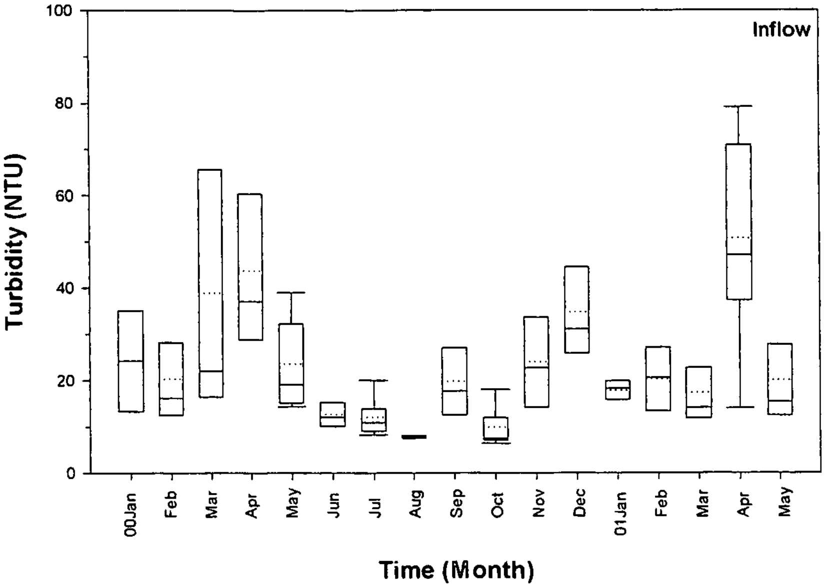


Figure 3.1a: Box plots of seasonal variation in turbidity during the study period at the Dam Wall (DW) Surface and Dam Wall Bottom (dotted line is the average and solid line is the median) in Loch Logan.



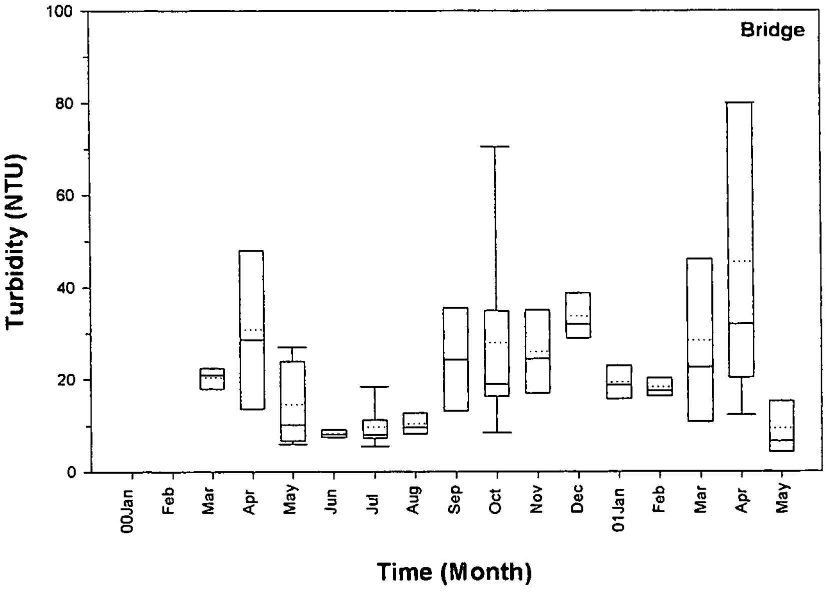


Figure 3.1b: Box plots of seasonal variation in turbidity during the study period at the Inflow and Bridge in Loch Logan.

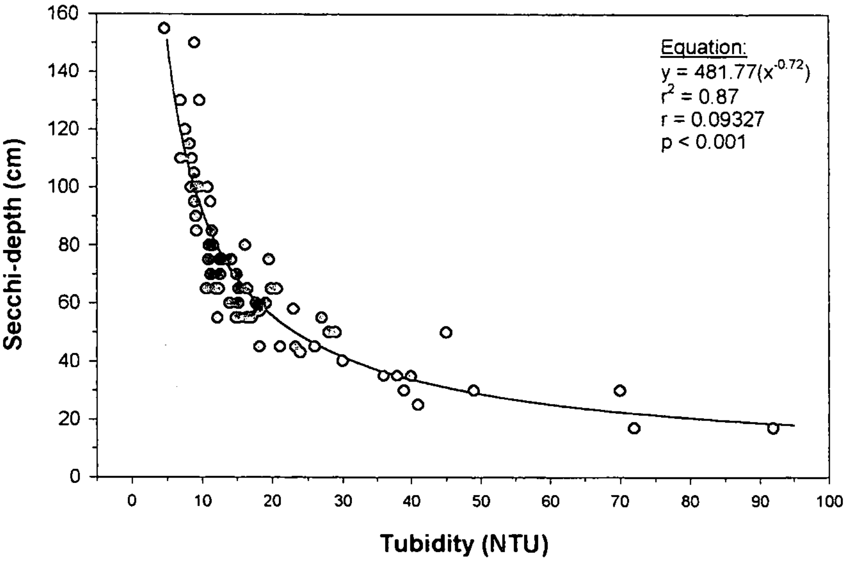


Figure 3.2: Relationship between Secchi depth and turbidity at the Wall Surface during the study period in Loch Logan.

3.3.2 CONDUCTIVITY

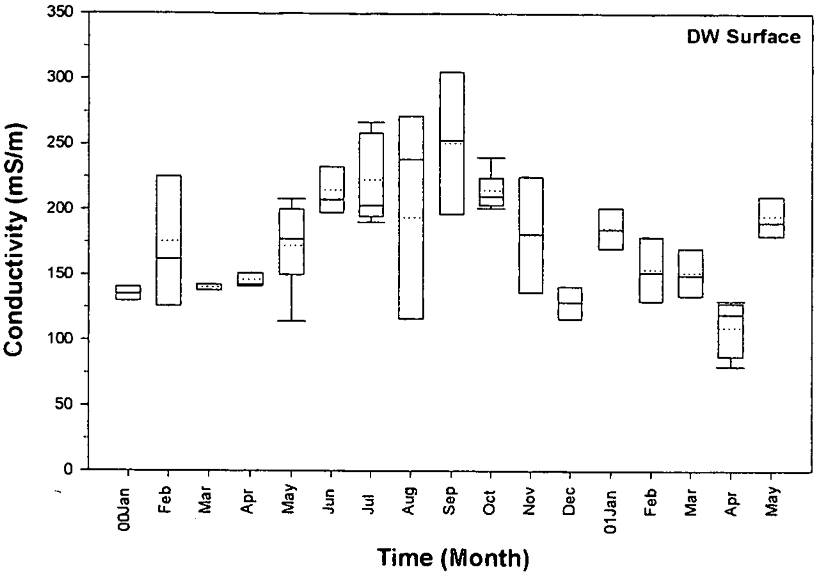
The conductivity in Loch Logan ranged from 8.0 to 56.5 mS/m (**Table 3.3**).

Table 3.3: Minimum, maximum and mean conductivity at the different sampling points in Loch Logan during the study period.

Sampling points	Conductivity (mS/m)		
	Min	**Max**	**Mean**
Wall Surface	8.0	30.9	17.5
Wall Bottom	9.0	56.5	17.6
Inflow	9.0	31.2	18.2
Bridge	12.0	36.0	21.8

The conductivity levels at the sampling points were more or less in the same range with a slightly higher average at the Bridge. The Wall Bottom displayed a very high conductivity reading during the first week of February 2000 following a chemical dosing of copper sulphate, after which it returned to almost the same level as the rest. The conductivity levels showed a clear seasonal patter, i.e. low during the summer with a clear build-up during the winter, followed by a decrease in levels after the rain in the spring and summer (**Figure 3.3**).

A statistically significant inverse correlation was found between turbidity and conductivity. Thirty-five point nine percent of the variation in conductivity was associated with the variation in turbidity ($r^2 = 0.359$; $p < 0.01$) (**Figure 3.4**). This indicates that with the increasing in salt concentration, the sedimentation of solids increased so that turbidity decreased and water became clearer.



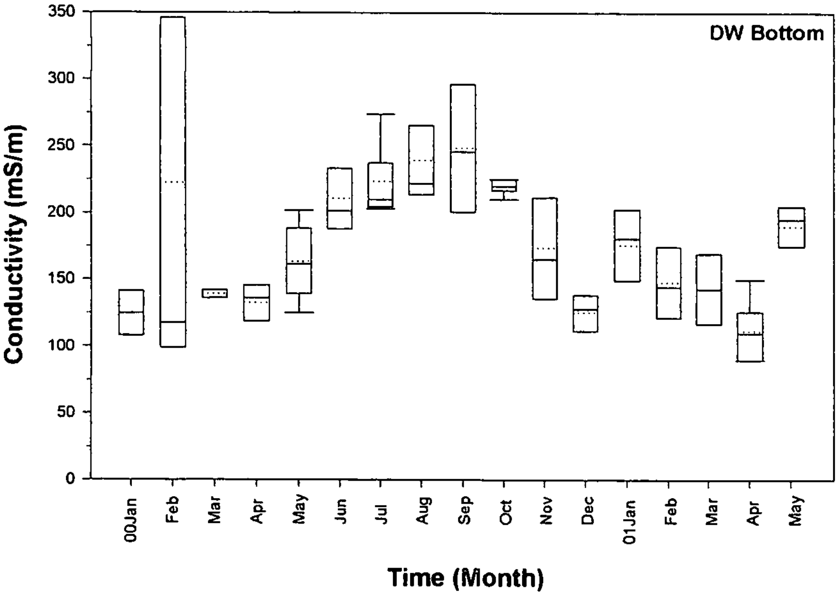
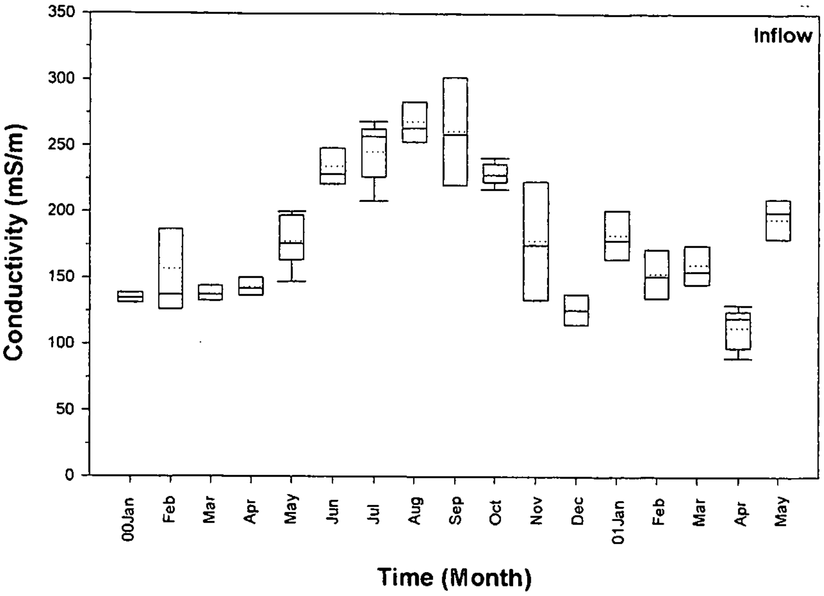


Figure 3.3a: Box plots of seasonal variation in conductivity during the study period at the Dam Wall (DW) Surface and Dam Wall Bottom in Loch Logan.



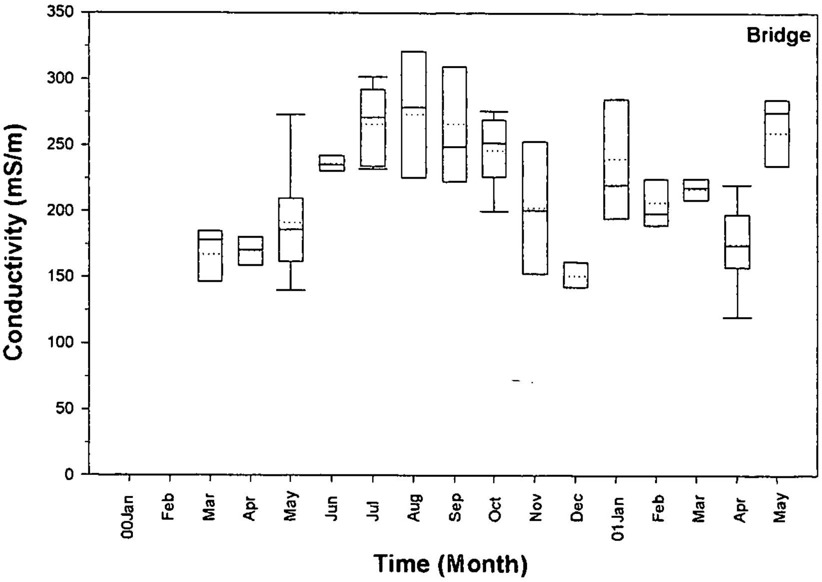


Figure 3.3b: Box plots of seasonal variation in conductivity during the study period at the Inflow and Bridge in Loch Logan.

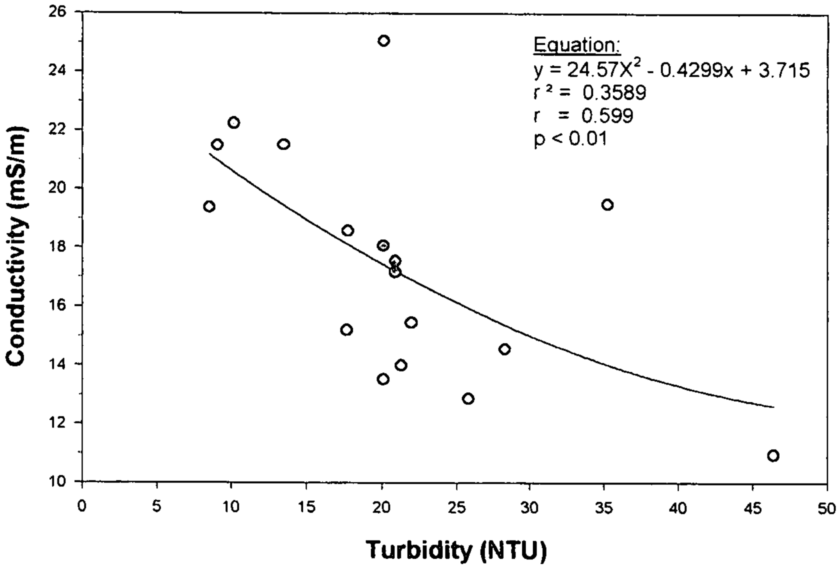


Figure 3.4: Relationship between conductivity and turbidity at the Wall Surface during the study period in Loch Logan.

3.3.3 NUTRIENTS: AMMONIUM- (NH_4-N) AND NITRATE-NITROGEN (NO_3-N) AS WELL AS REACTIVE *ORTHO*-PHOSPHATE (PO_4-P).

During the study period there was a large variation in NO_3-N as well as NH_4-N concentrations. The NH_4-N ranged between 1.0 and 1,032.6 µg/l and the NO_3-N from 1.0 to 1,375.1 µg/l (**Table 3.4**). The maximum of 1,032.6 µg/l NH_4-N was recorded in the Wall Bottom sample after an algal bloom in summer during December 1999/January 2000, and the lowest concentrations in the winter period except for the Bridge, where as the 1,375.1 µg/l NO_3-N was recorded during the first week of May 2000 (**Figures 3.5** and **3.6**). Most of the increases in the NO_3-N concentration, throughout all the sampling points, were associated with rainfall. This also applies to the NH_4-N concentration, especially in the Wall Bottom sample.

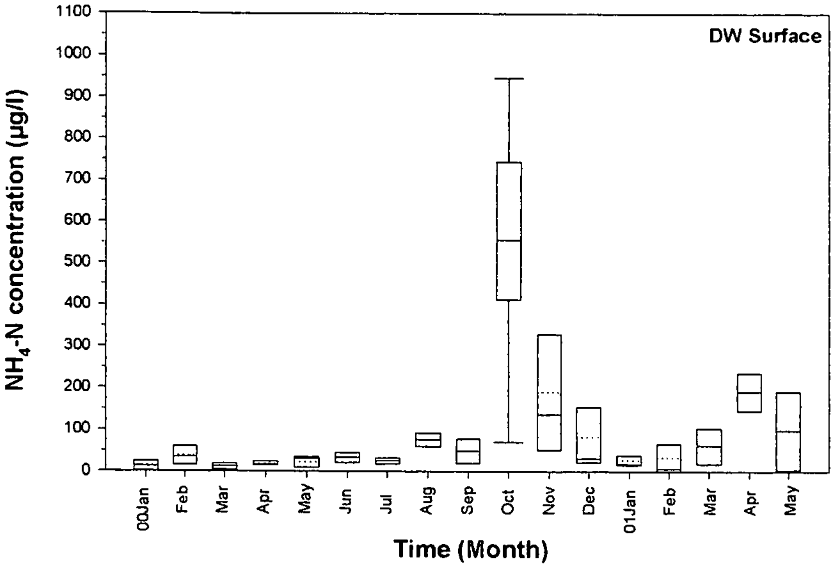
Table 3.4: Minimum, maximum and mean nitrogen at the different sampling points in Loch Logan during the study period.

Sampling points	**N**	**Nitrogen (µg/l)**		
		Min	**Max**	**Mean**
Wall Surface	NH_4-N	1.0	954.1	89.4
	NO_3-N	1.0	644.8	163.3
Wall Bottom	NH_4-N	14.9	1,032.6	336.8
	NO_3-N	1.0	629.8	172.3
Inflow	NH_4-N	1.0	718.8	91.32
	NO_3-N	1.0	658.9	174.8
Bridge	NH_4-N	1.0	671.9	208.3
	NO_3-N	1.0	1,375.1	294.4

The seasonal variation in the PO_4-P concentration followed the same pattern as the NO_3-N concentration, namely high concentrations during rainy periods. The PO_4-P ranged between 0.5 and 648.3 µg/l (**Table 3.5**), with the maximum (648.3 µg/l) recorded in the Wall Surface sample after it rained during the third week of July 2000 (**Figure 3.7**).

Table 3.5: Minimum, maximum and mean reactive *ortho*-phosphate at the different sampling points in Loch Logan during the study period.

Sampling points	**Reactive *Ortho*-phosphate (µg/l)**		
	Min	**Max**	**Mean**
Wall Surface	0.5	648.3	85.0
Wall Bottom	0.5	242.8	72.1
Inflow	0.5	328.7	62.6
Bridge	0.5	332.0	95.4



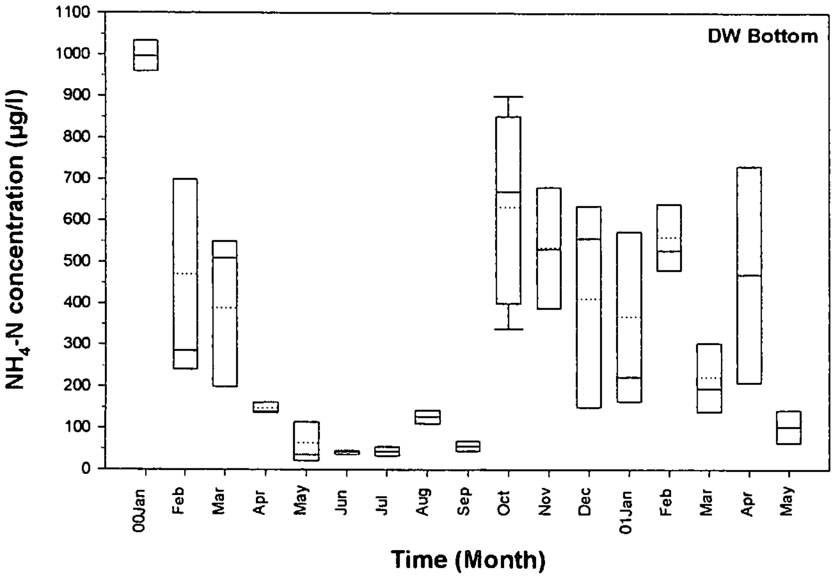
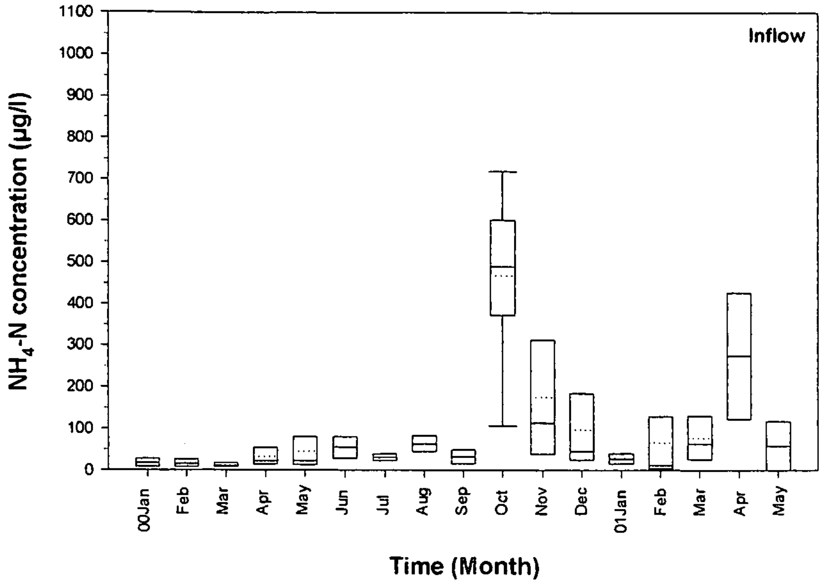


Figure 3.5a: Box plots of seasonal variation in ammonium (NH_4-N) during the study period at the Dam Wall (DW) Surface and Dam Wall Bottom in Loch Logan.



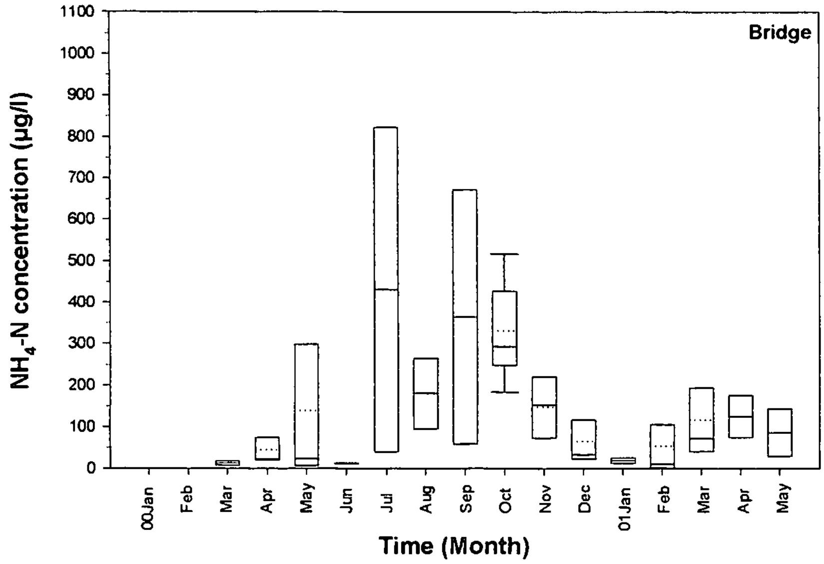
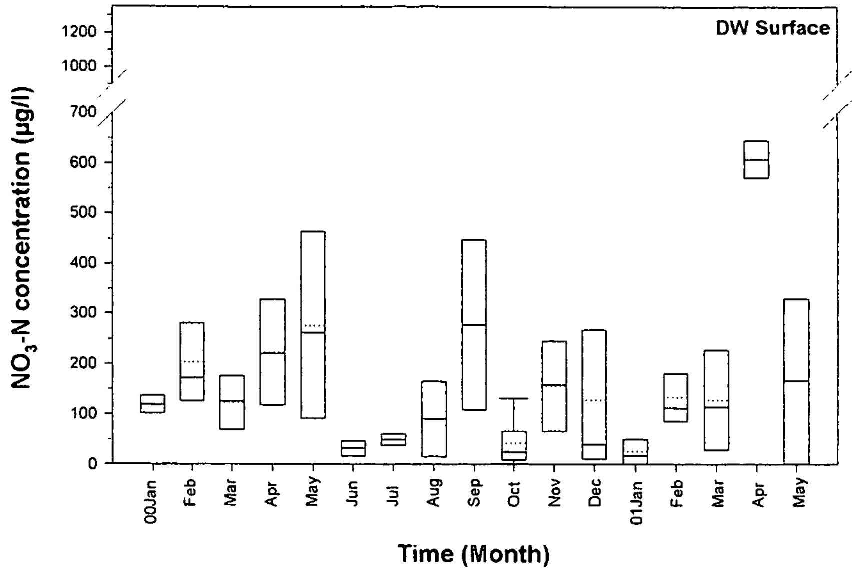


Figure 3.5b: Seasonal changes of ammonium (NH_4-N) in Loch Logan over the seventeen months study period at the Inflow and Bridge.



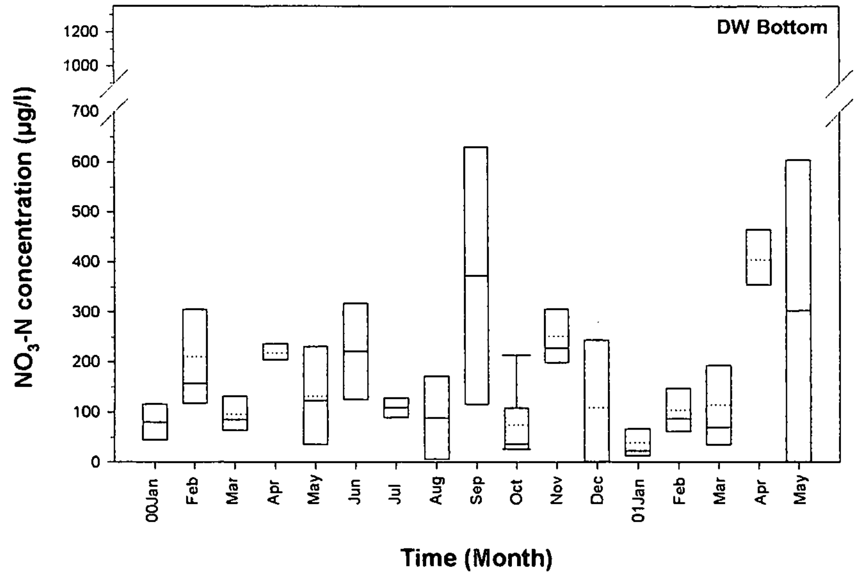
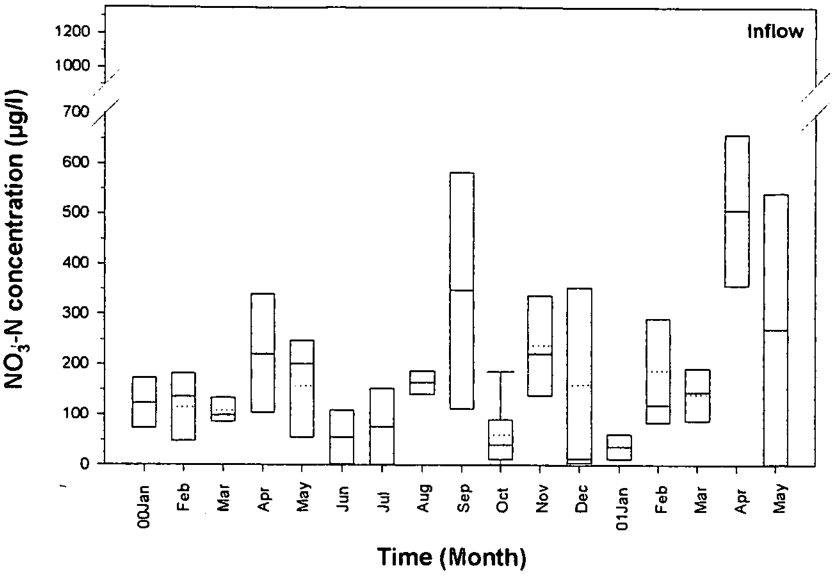


Figure 3.6a: Box plots of seasonal variation in nitrate (NO$_3$-N) during the study period at the Dam Wall (DW) Surface and Dam Wall Bottom in Loch Logan.



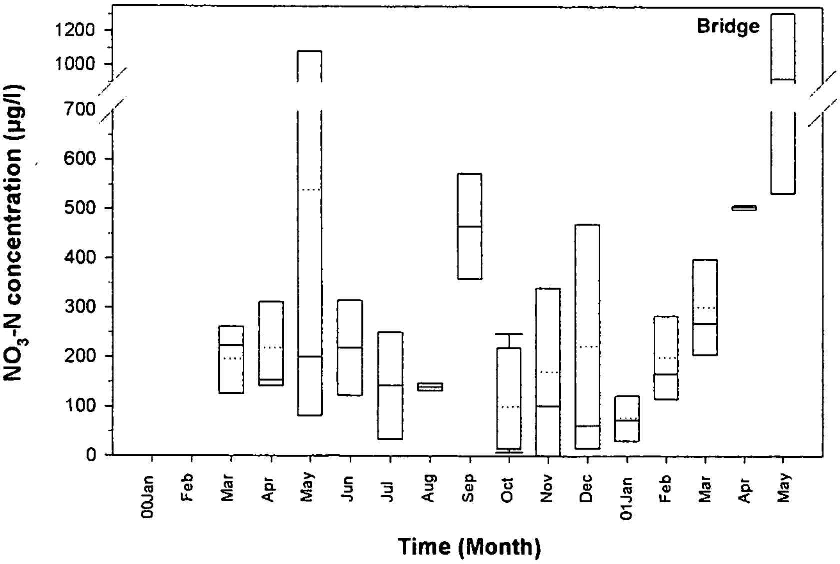
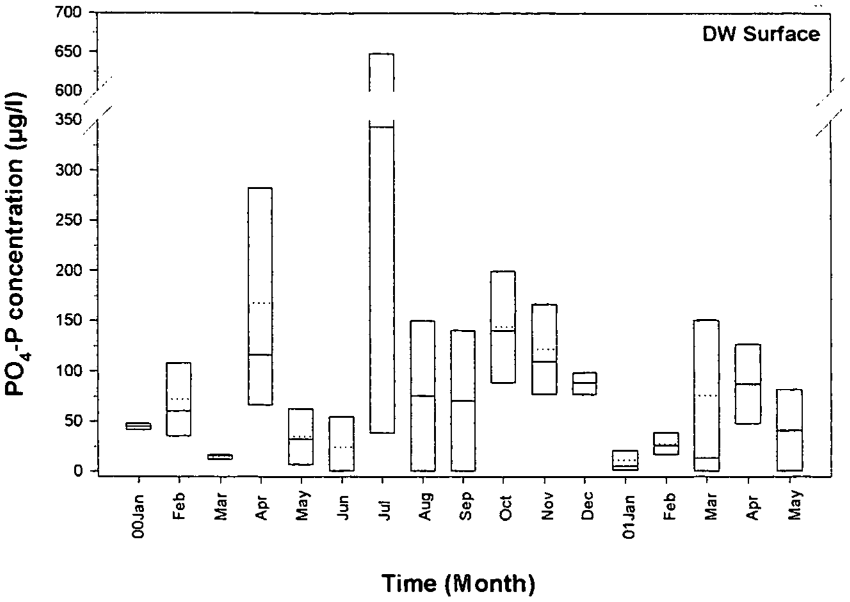


Figure 3.6b: Box plots of seasonal variation in nitrate (NO_3-N) during the study period at the Inflow and Bridge in Loch Logan.



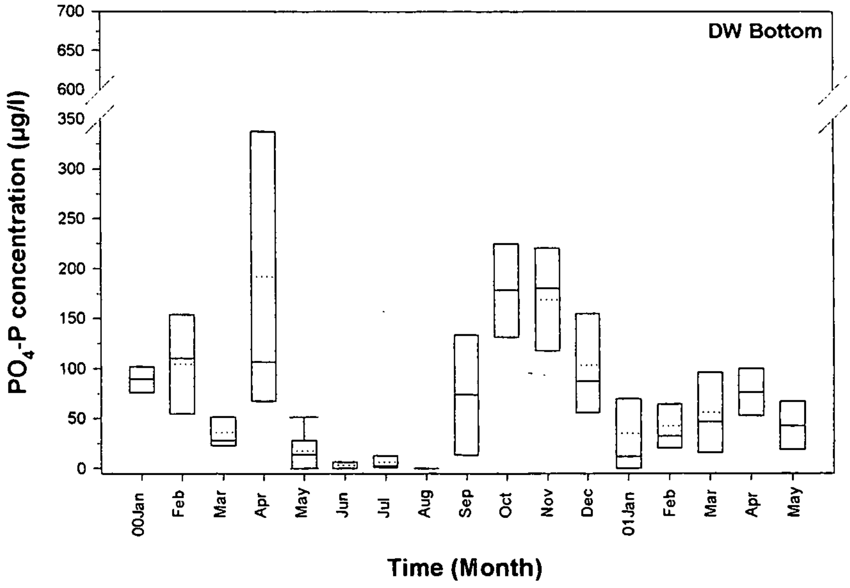
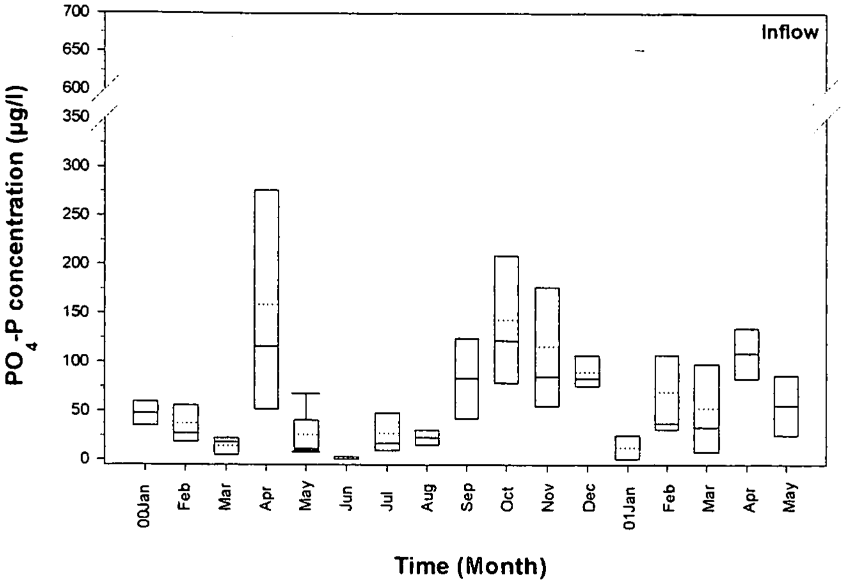


Figure 3.7a: Box plots of seasonal variation in reactive *ortho*-phosphate (PO$_4$-P) during the study period at the Dam Wall (DW) Surface and Dam Wall Bottom in Loch Logan.



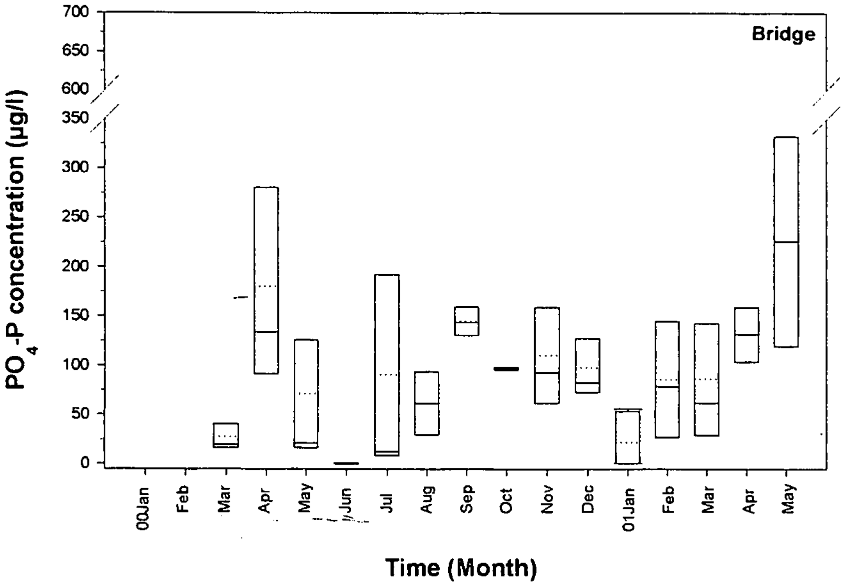


Figure 3.7b: Box plots of seasonal variation in reactive *ortho*-phosphates (PO_4-P) during the study period at the Inflow and Bridge in Loch Logan.

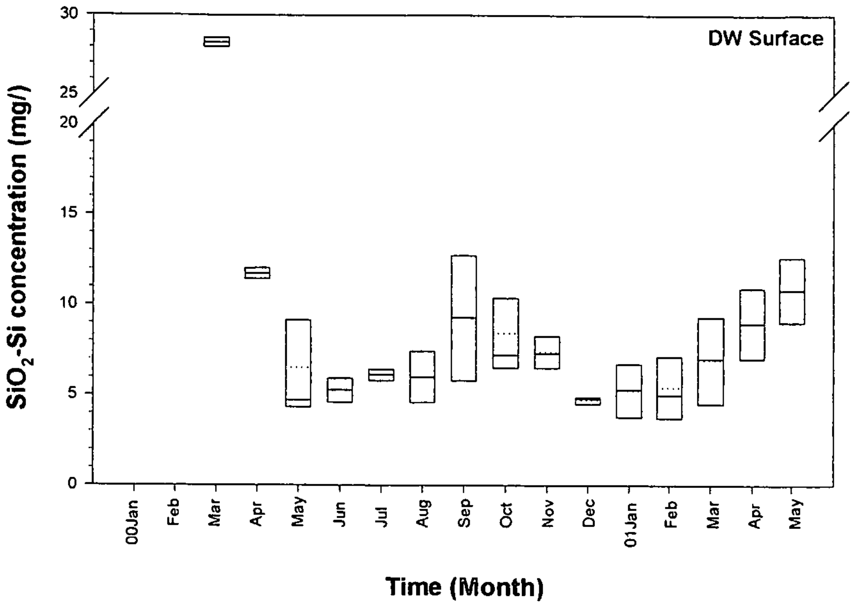
3.3.4 SILICA–SILICON (SiO_2-Si)

The SiO_2-Si concentration in Loch Logan ranged between 1.8 and 28.6 mg/l with an overall average of 8.3 mg/l (**Table 3.6**).

Table 3.6: Minimum, maximum and mean silica-silicon at the different sampling points in Loch Logan during the study period.

Sampling points	Silica-silicon (mg/l)		
	Min	**Max**	**Mean**
Wall Surface	2.5	28.6	8.8
Wall Bottom	3.5	18.5	8.1
Inflow	1.8	12.8	7.1
Bridge	3.9	18.5	9.2

A seasonal pattern was observed where the concentrations were higher during the late winter and lower during the summer. However, solubility of silica is higher in warm water and thus usual high concentrations during summer (Cole, 1993), but high diatom numbers can cause depletion. Except for March 2000 at the DW Surface, all the other readings were lower than 20 mg/l (**Figure 3.8**).



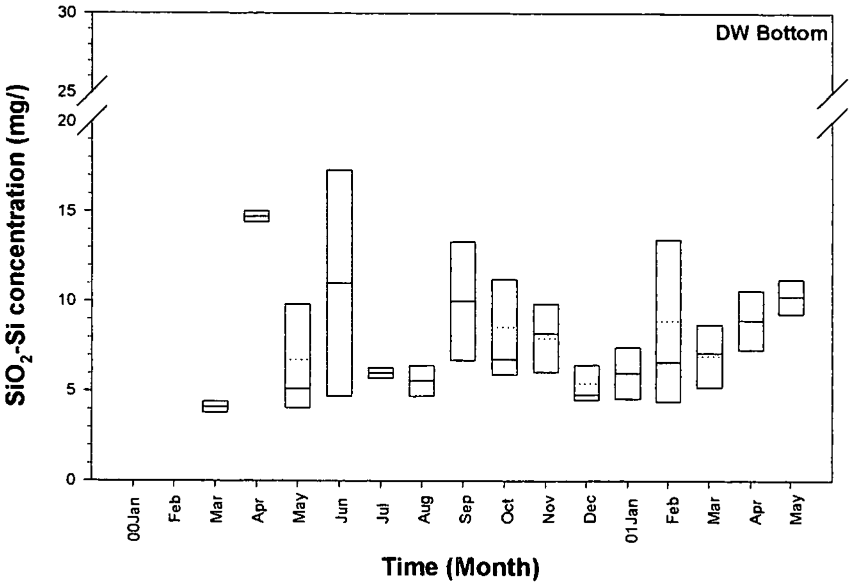


Figure 3.8a: Box plots of seasonal variation in silica (SiO_2-Si) during the study period at the Dam Wall (DW) Surface and Dam Wall Bottom in Loch Logan.

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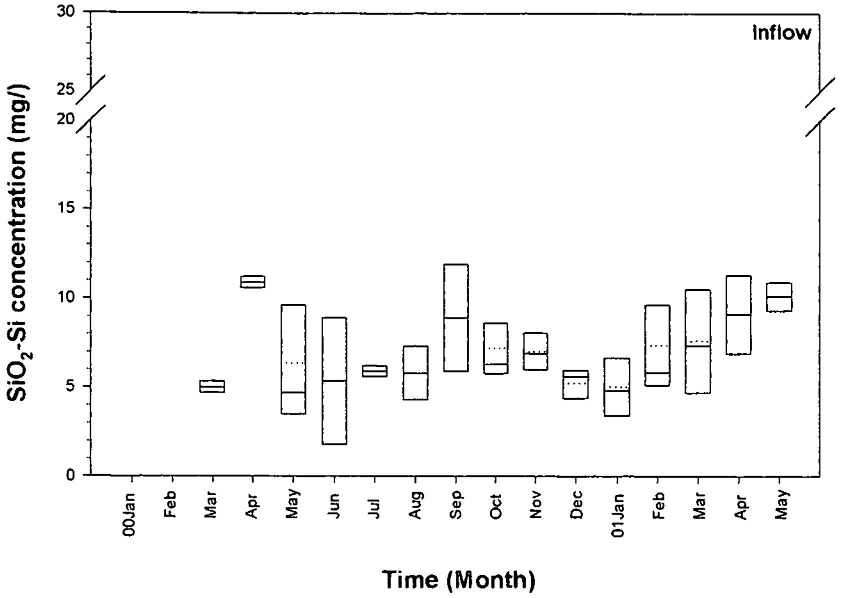


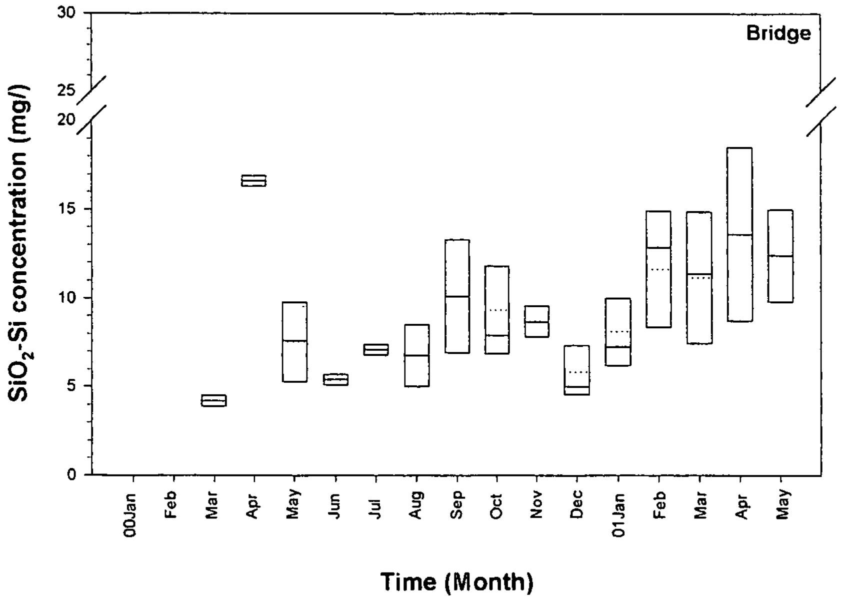


Figure 3.8b: Box plots of seasonal variation in silica (SiO_2-Si) during the study period at the Inflow and Bridge in Loch Logan.

3.3.5 OXYGEN, TEMPERATURE AND pH

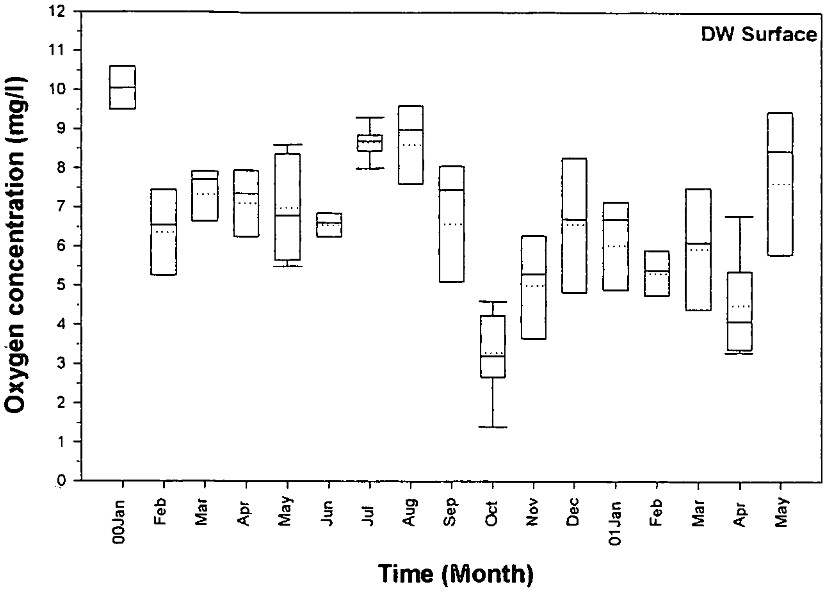
The dissolved oxygen concentration in Loch Logan varied between 0.1 and 10.6 mg/l (super saturated), with a mean percentage saturation of 72.8 %, except for the Wall Bottom sample (16.4 %) (**Table 3.7** and **Figure 3.9**).

Table 3.7: Minimum, maximum and mean dissolved oxygen concentration and percentage saturation at the different sampling points in Loch Logan during the study period.

Sampling points	**Oxygen**					
	Min		**Max**		**Mean**	
	mg/l	**%**	**mg/l**	**%**	**mg/l**	**%**
Wall Surface	1.4	15.4	10.6	127.0	6.6	72.7
Wall Bottom	0.1	1.2	7.7	67.6	2.0	16.5
Inflow	1.3	14.9	10.4	121.2	6.6	72.2
Bridge	0.9	9.9	9.2	119.9	6.5	73.4

The water temperature ranged between 8.0°C (winter) and 28.1°C (summer) (**Figure 3.10**), with a similar pattern to that of the mean air temperature. The air temperature ranged between -3.3°C (winter) and 32.5 °C (summer) (**Figure 3.11**).

The depth profiles of temperature and oxygen over a period of twelve months showed a clear pattern regarding the temperature and oxygen concentration. In the summer the oxygen concentration was very high (>6 mg/l) at the surface and very low (<0.5 mg/l) at the bottom, whereas in the winter there was not much of a difference between the surface oxygen concentration (>6.5 mg/l) and that of the bottom (>4.5 mg/l). There is a switch-over between the oxygen and temperature from summer to winter and again from winter to summer (**Figure 3.12**). During January 2000 the surface oxygen was exceptionally high (super saturated at 122.5 %; 10 mg/l) compared to that of February 2000 and January 2001 (74.9 %; 6 mg/l).



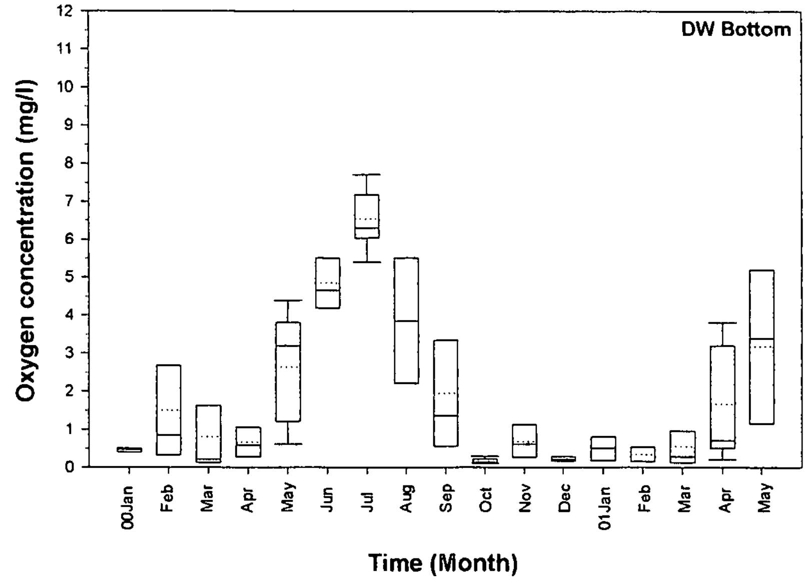
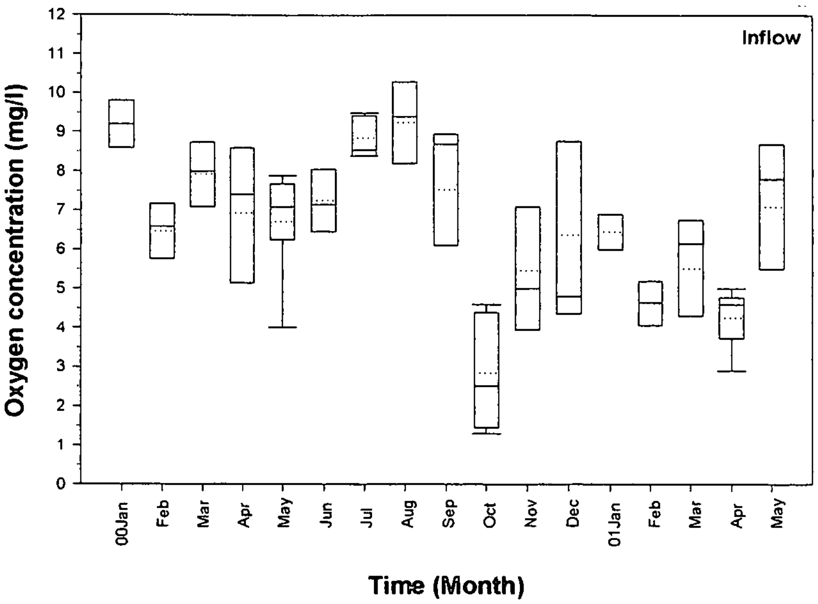


Figure 3.9a: Box plots of seasonal variation in dissolved oxygen (O_2) during the study period at the Dam Wall (DW) Surface and Dam Wall Bottom in Loch Logan.



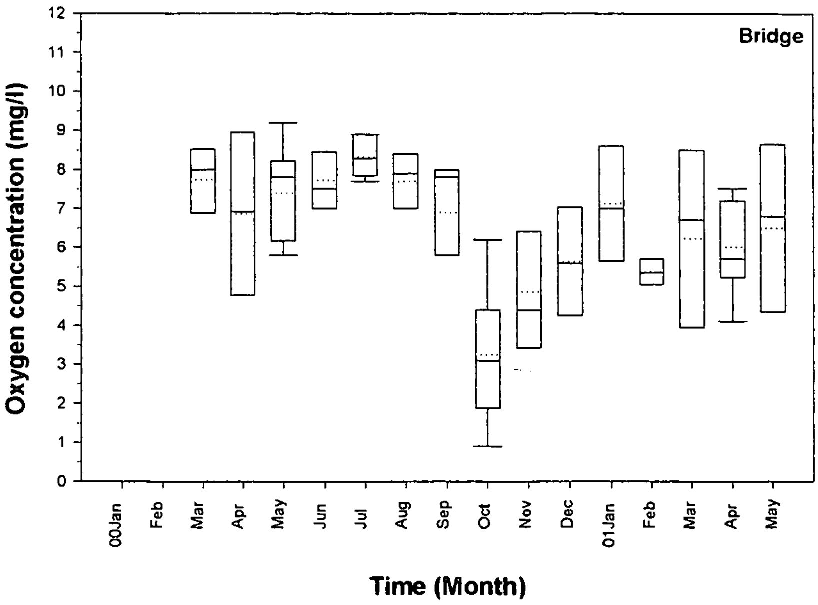
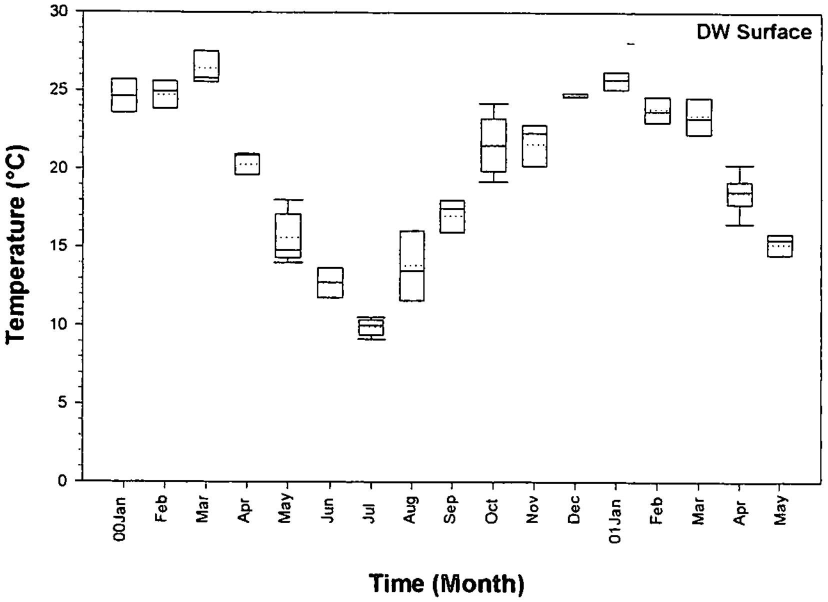


Figure 3.9b: Box plots of seasonal variation in dissolved oxygen (O_2) during the study period at the Inflow and Bridge in Loch Logan.



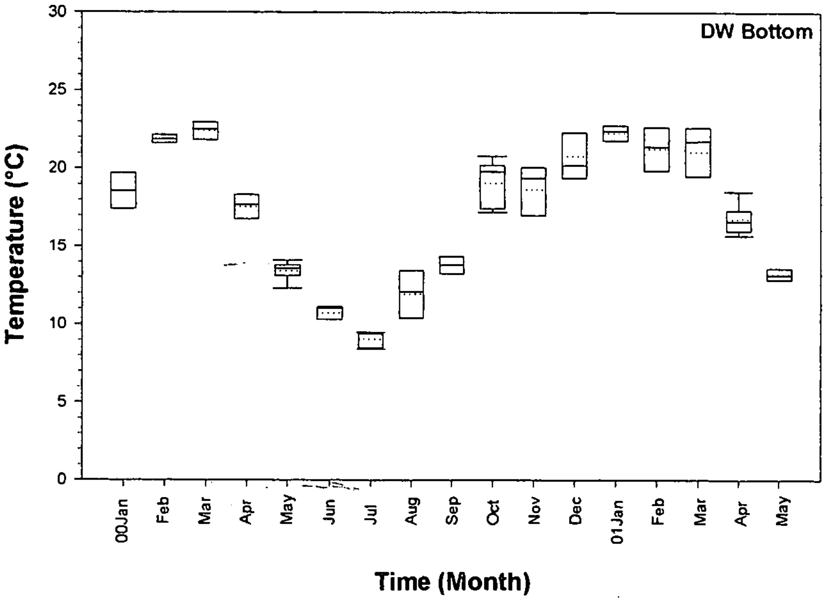
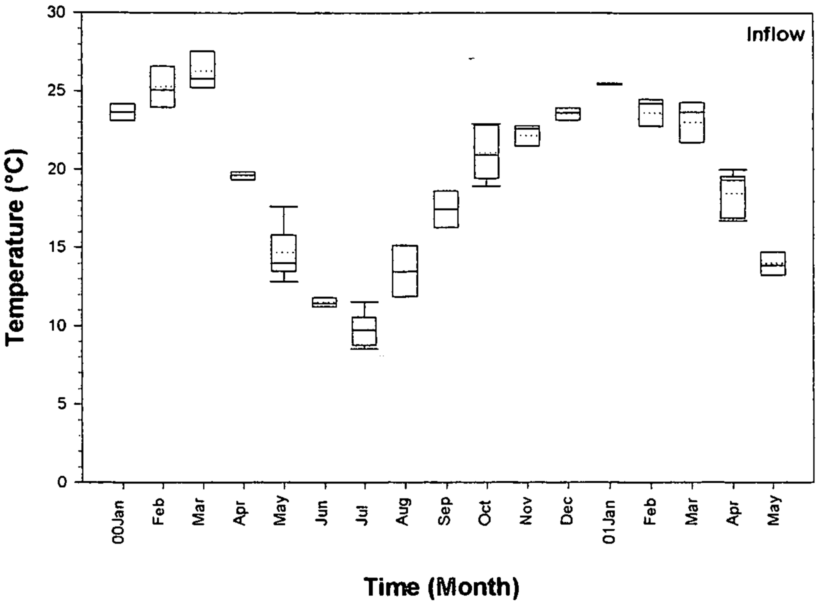


Figure 3.10a: Box plots of seasonal variation in water temperature during the study period at the Dam Wall (DW) Surface and Dam Wall Bottom in Loch Logan.



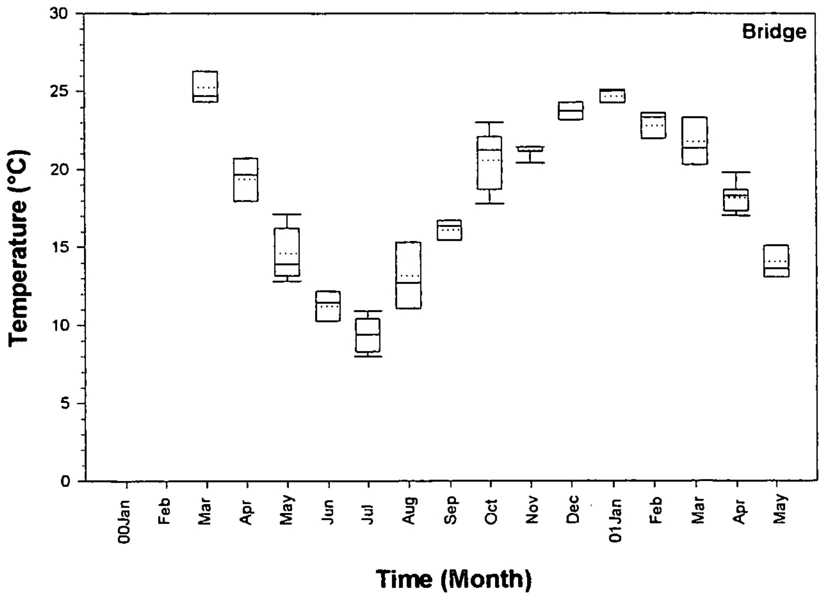


Figure 3.10b: Box plots of seasonal variation in water temperature during the study period at the Inflow and Bridge in Loch Logan.

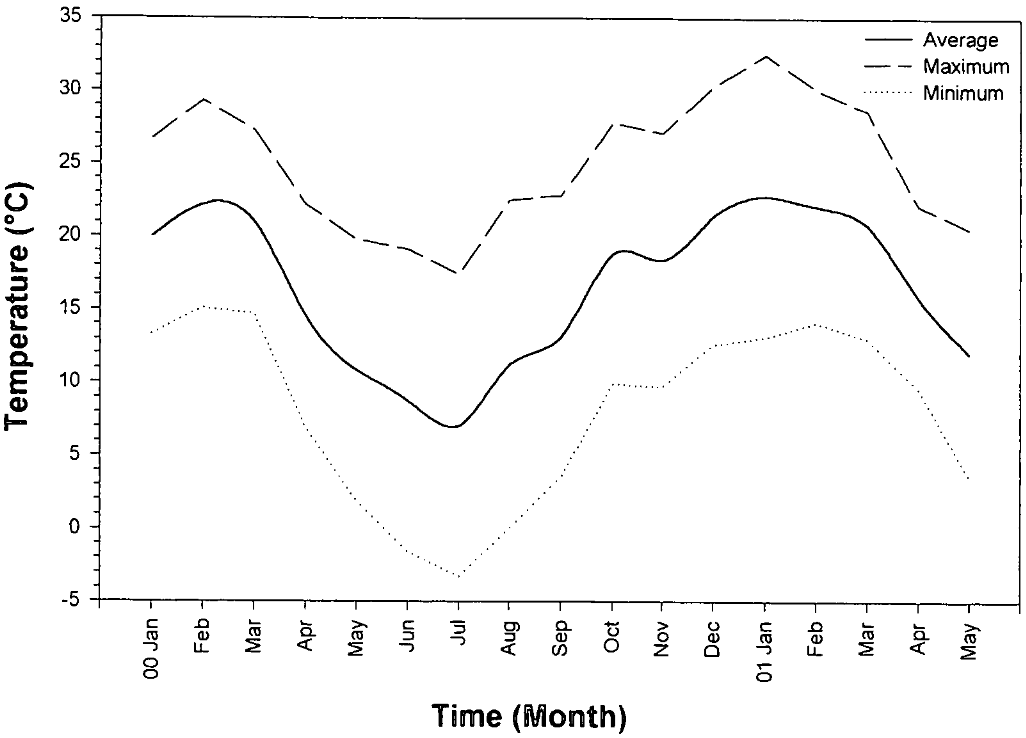


Figure 3.11: The minimum, maximum and average air temperatures of Loch Logan over the seventeen months study period.

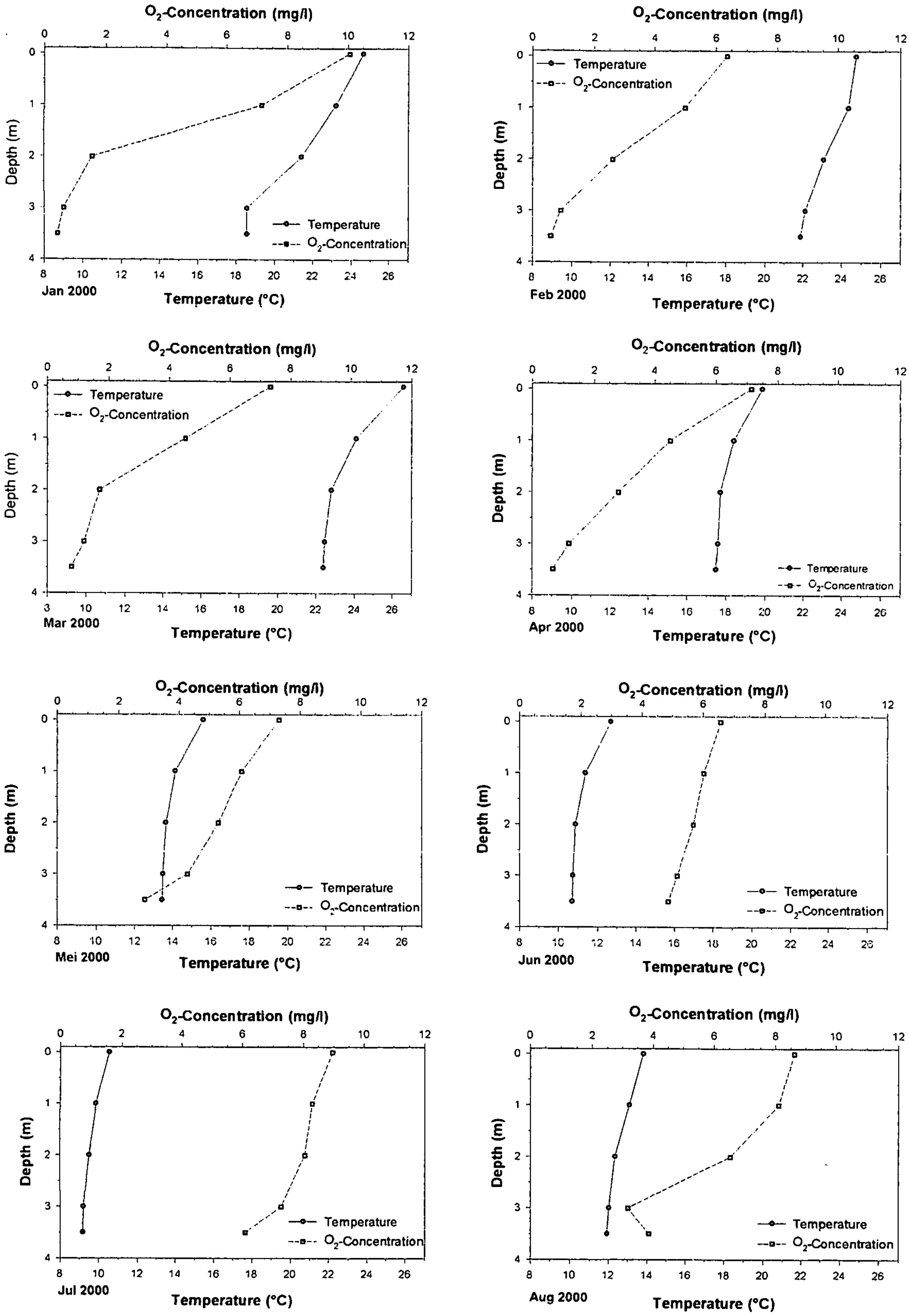
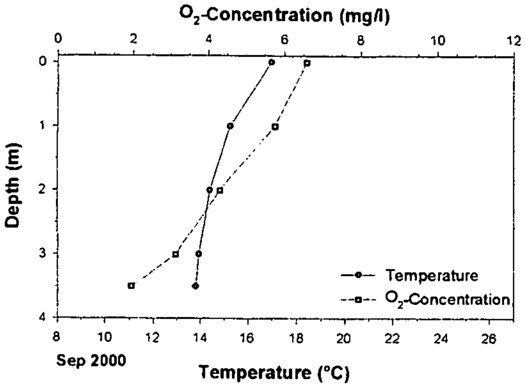
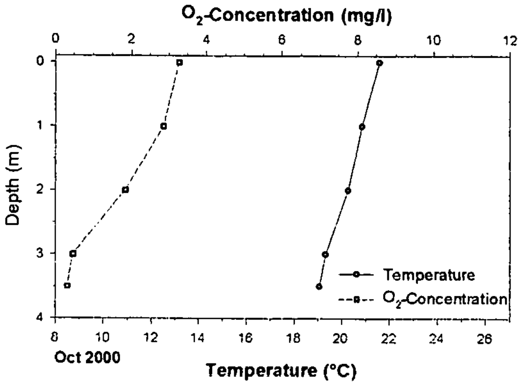
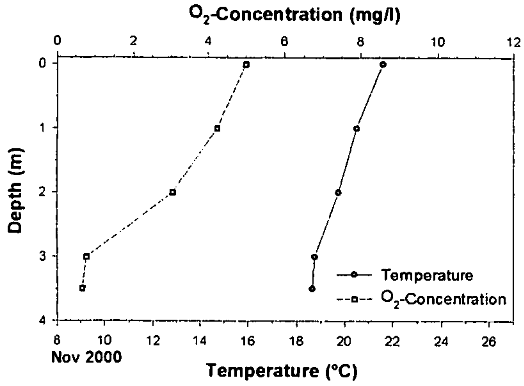
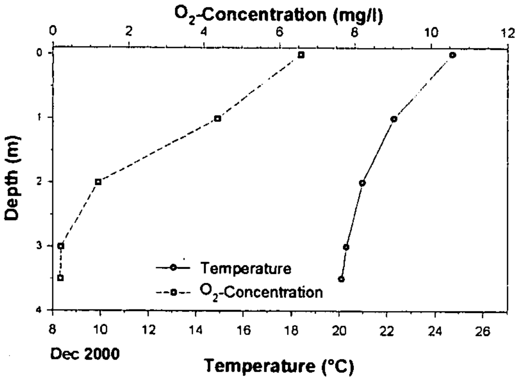


Figure 3.12a: The monthly changes in the dissolved oxygen and temperature depth profiles; January 2000 to August 2000.









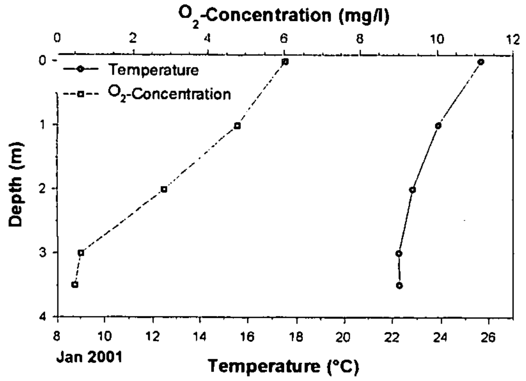
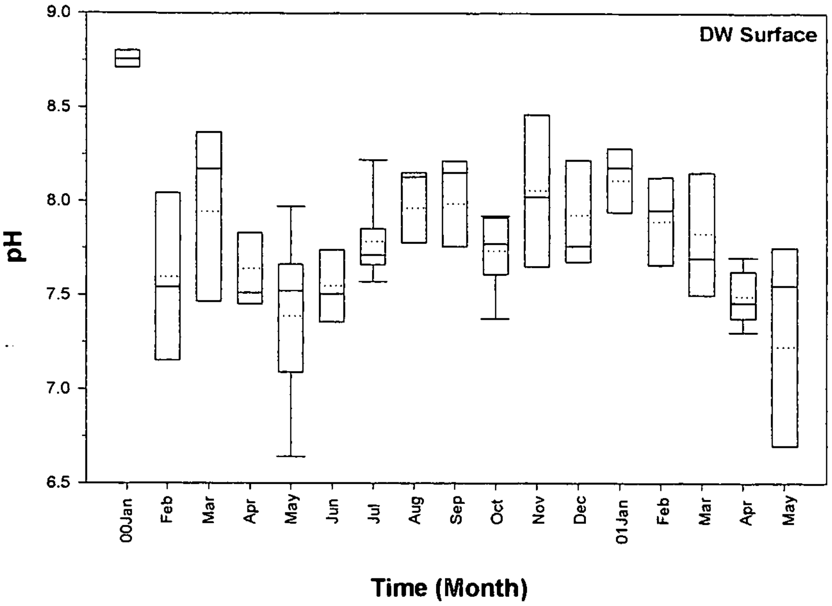


Figure 3.12b: The monthly changes in the dissolved oxygen and temperature depth profiles; September 2000 to January 2001.

The pH in Loch Logan (mean = 7.66) indicates alkaline water. The highest pH measured was 8.8 and the lowest 6 (**Figure 3.13**). A pH less than 7.0 occurred mostly in the bottom samples at the dam wall.

There was a positive correlation between the Wall Surface's oxygen concentration and its pH (r^2 = 0.167; $p < 0.001$) (**Figure 3.14**). It is important to note that these measurements (dissolved oxygen, temperature and pH) were made at a given time of day, therefore it can vary according to the time of day.



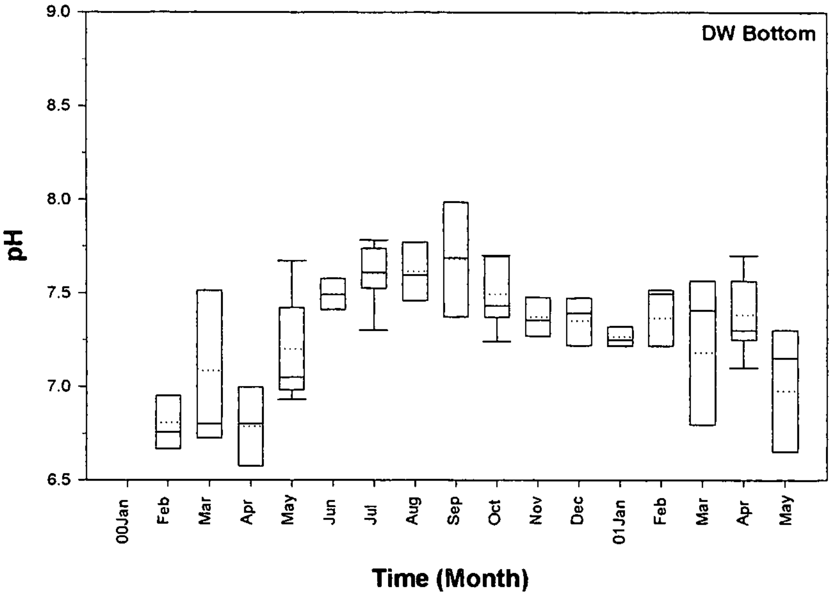
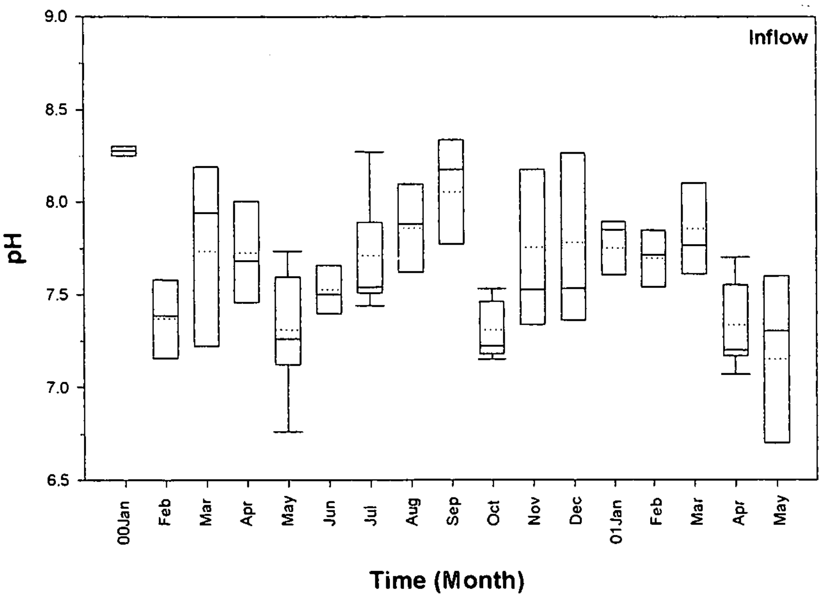


Figure 3.13a: Box plots of seasonal variation in pH during the study period at the Dam Wall (DW) Surface and Dam Wall Bottom in Loch Logan.



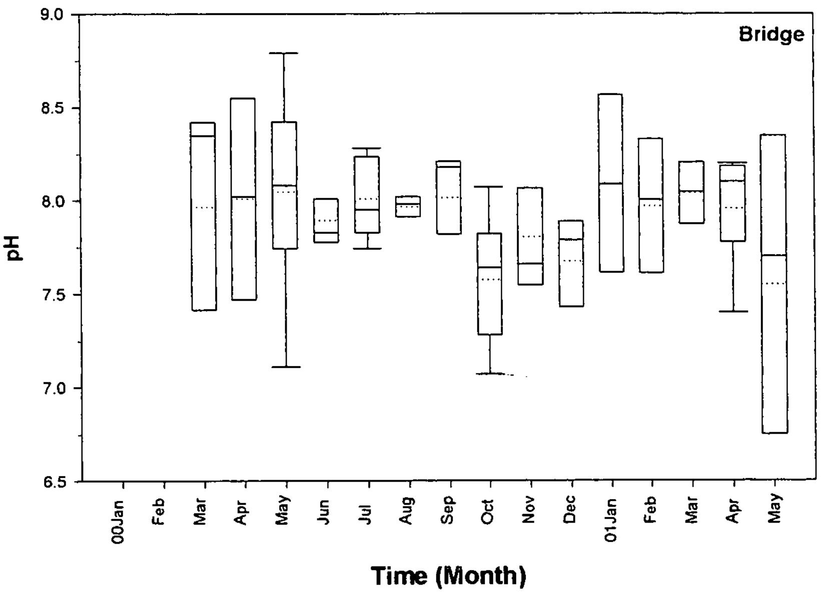


Figure 3.13b: Box plots of seasonal variation in pH during the study period at the Inflow and Bridge in Loch Logan.

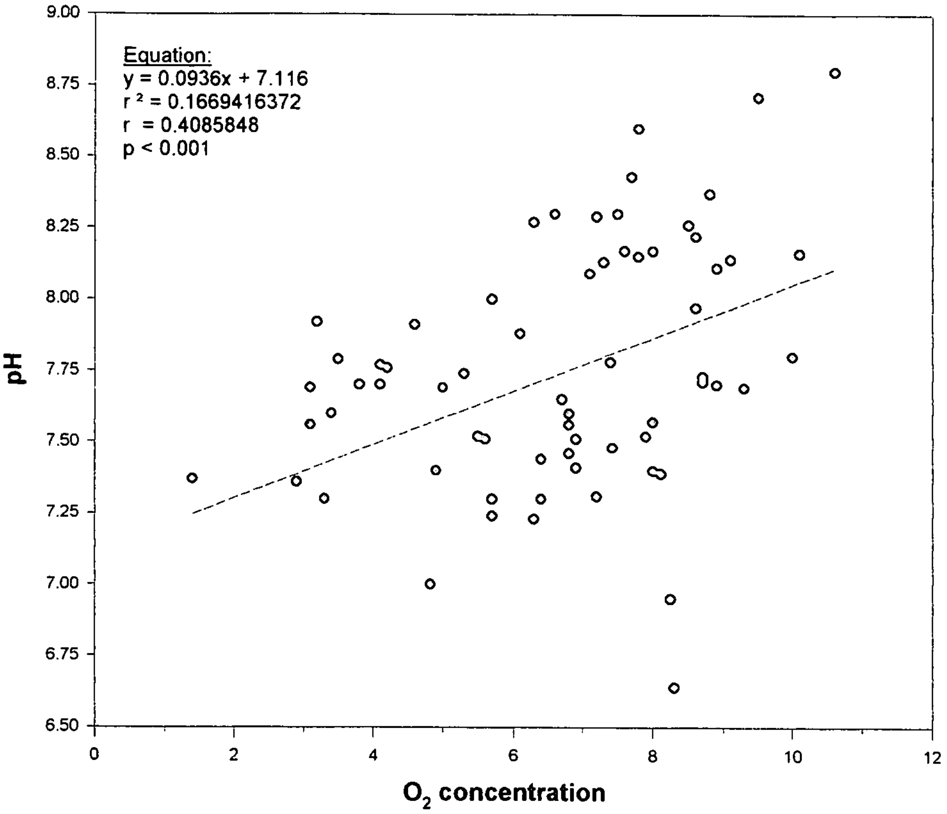


Figure 3.14: Relationship between pH and dissolved oxygen at the Wall Surface during the study period in Loch Logan.

3.4 DISCUSSION

3.4.1 TURBIDITY

Turbidity is an optical property of water where suspended and some dissolved materials such as clay, silt, finely divided organic and inorganic matter, and plankton cause light to be scattered and absorbed rather than transmitted in straight lines (Lloyd, 1987). Water and pigment can also attenuate sunlight passing the surface of lakes and reservoirs (Lind *et al.*, 1992). Whether any disturbance or not, water-column mixing as such, or in combination with inorganic turbidity, causes concomitant changes in the light field over time. Consequently, vertical mixing and/or suspended solids significantly affect phytoplankton's photosynthesis and productivity in aquatic environments (Dokulil, 1994) (see **Chapter 4**).

Turbid systems in the southern hemisphere commonly comprise rivers, impoundments and shallow lakes (Dokulil, 1994) and half of South Africa's 60 large impoundments, before 1981, had a mean Secchi depth of less than 70 cm (Grobler & Davies, 1981). The average surface turbidity in Loch Logan was 22.2 NTUs, with an average Secchi depth of 63 cm (15-160 cm). The Secchi depth and turbidity of Loch Logan are comparable to several South African impoundments, e.g. North End Lake (1991-94) = 14 NTUs, Wemmer Pan (1991-94) = 22 NTUs and Johannesburg Zoo Lake (1998-94) = 18.3 NTUs (Wiechers *et al.* 1996). However, accordingly to the South African Water Quality Guidelines for Recreational Use (Department of Water Affairs and Forestry - DWAF, 1996a), water with a Secchi depth less than 100 cm (1m) is unsuitable for swimming or any full-contact recreation.

There was a significant (87.0 %) exponential relationship between the turbidity and Secchi depth of Loch Logan. When the turbidity increased, the Secchi depth decreased. The seasonal variation in turbidity is associated with the rainfall pattern in the area, i.e. summer rainfall. In summer the turbidity increased (aside from increases in phytoplankton density) when sediment and other particles were washed into Loch Logan. The time lapse between thunderstorms and the actual measurement was important. This is because measurements were taken on Mondays, and when it rained during the previous week the sediment and other particles that contributed to the turbidity had time to settle to the bottom of the water-column. The water was then less turbid than when it rained over the weekend. During the winter, a low-raining season, turbidity was low; this could be due to vertical mixing of algae and suspended solids in the water-column as a result of polymixis (multiple mixing) (Dokulil, 1994; Horne & Goldman, 1994).

The higher turbidity at the bottom was possibly due to sedimentation and accumulation of suspended solids and detritus.

Because the water in Loch Logan is not that turbid, the euphotic zone was relatively deep (~1.4 m) and resulted in favourable underwater climate that enhance the photosynthesis rate of the algae (see **Chapter 5**).

3.4.2 CONDUCTIVITY

Salinisation is a growing problem in freshwater ecosystems in many parts of the world (Prinsloo & Pieterse, 1994), especially in South Africa (NSOER, 1999). Generally, salinity increases with time in densely populated areas, and is an indicator of the extent to which water is used and reused. It is also an indicator of eutrophication of the extent to which nutrient-rich effluents have contaminated the water supplies (Du Plessis & Van Veelen, 1991).

The total dissolved salts (TDS) concentration is directly proportional to the electrical conductivity (EC) of water. The TDS (mg/l) of most natural waters can be estimated by multiplying EC (mS/m) by a constant. This factor generally falls in the range of 5.5-7.5 and for South African waters this factor is approximately 6.5 (DWAF, 1996b). This is affected by change in the ion concentration of a solution and to a lesser degree, a change in ionic composition and temperature (Allan, 1995).

The average conductivity of Loch Logan was 18.8 mS/m (125 mg/l TDS), which is well within the target range proposed by Quality of Domestic supplies (WRC, 1998). It states that no health, aesthetic or treatment effects are associated within the range (EC = 0-70 mS/m, TDS = <450 mg/l). The average conductivity of Camps Drift Canal, Pietermaritzburg (1990-94) - 19 mS/m; 129 mg/l TDS - coincides with that of Loch Logan. The average conductivity and total dissolved salts of some of South Africa's urban impoundments fall outside this range with an EC average of 97 mS/m and a TDS average of 814.3 mg/l (Wiechers *et al.*, 1996).

The conductivity was the highest in the canal at the bridge and decreased towards the dam wall. This indicated a dilution of the salts - probably through seepage water - as it entered Loch Logan. It has also been noted that there was a seasonal variation in Loch Logan's conductivity: during the summer, the conductivity was lower than in the winter. This is because in the summer the rainwater diluted the salt concentration and in the winter the conductivity increased through evapoconcentration (DWAF, 1996b). The latter was also found by Wiechers *et al.* (1996) in the Hennops Dam, Centurion (1993-94) where the conductivity was higher in the winter (58 mS/m) than in the summer (52 mS/m).

The exceptionally high conductivity measured at the bottom during February 2000 (**Figure 3.3**) could be explained by the fact that copper sulphate was added to the water to control an algal bloom at the end of January 2000 (see **Chapter 4**).

Conductivity was inversely related to turbidity in Loch Logan. Thirty-five point nine percent of the variation in conductivity could be ascribed to the variation in turbidity. The turbidity of a system does not only depend on rainfall but also on other factors, i.e. flow and vegetation cover. Roos and Pieterse (1995) also illustrated a similar relationship in the Vaal River, South Africa. The more rain that fell, the lower the conductivity and the higher the turbidity and *vice versa*. High salt concentrations increased the sedimentation of solids in the water, which leads to clearer water.

3.4.3 NUTRIENTS: AMMONIUM- (NH_4-N) AND NITRATE-NITROGEN (NO_3-N) AS WELL AS REACTIVE *ORTHO*-PHOSPHATE (PO_4-P)

The average NO_3-N and NH_4-N concentrations of Loch Logan were 201.2 µg/l and 181.5 µg/l, respectively. This is in fact lower than the concentrations of other urban impoundments (mean NH_4-N = 2519 µg/l, mean NO_3-N = 2162 µg/l) that have been studied in South Africa (Wiechers *et al.,* 1996). However, Loch Logan's nitrogen concentrations coincided with that of North End Lake (1991-94) (NH_4-N = 290 µg/l; NO_3-N = 210 µg/l) (Wiechers *et al.,* 1996).

The average NO_3-N concentrations of the Gariep Dam (380 µg/l) and Van Der Kloof Dam (370 µg/l) for 1998-99 were twice that of Loch Logan (Venter, 2000). The average sum of the ammonium and nitrogen concentration (NO_3-N + NH_4-N) were sometimes higher than 1000 µg/l, and according to the South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996b) this is in the range of mesotrophic systems. Oligotrophic water has an average summer inorganic nitrogen concentration of less than 500 µg/l.

The nitrogen in Loch Logan did evidently not only come from the stormwater. Nitrate was probably produced by the nitrification of ammonia excreted by fish and by ammonium mineralised from waste food and fish faeces (Axler *et al.,* 1996). The nitrification of ammonium to nitrate may reduce the ammonium concentrations, but the nitrate is also readily available to algae.

The mean PO_4-P concentration in Loch Logan was 78.8 µg/l and although it is considered as eutrophic conditions by the South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996b), it is still not as high as that of some of South Africa's other urban impoundments (mean PO_4-P = 447 µg/l) (Wiechers *et al.,* 1996). The impoundment with the average PO_4-P concentrations closest to that of Loch Logan's is Zoo Lake (1989-94) (130 µg/l) (Wiechers *et al.,* 1996). The PO_4-P concentration of Loch Logan also reached the hypertrophic level (>250 µg/l) during periods that followed a heavy rainstorm. The high nutrient concentrations emphasise that the runoff water that entered Loch Logan was in fact well enriched. It was also noted that in the winter the nutrient concentrations (NO_3-N, NH_4-N and PO_4-P) in Loch Logan were, on average, higher than in the summer. This was also reported in the Hennops Dam, 1993-1994 (Wiechers *et al.,* 1996), which could be the result of lower nutrient uptake by microbes due to lower metabolic rates. The average PO_4-P concentrations of the Gariep Dam (27 µg/l) and Van Der Kloof Dam (34 µg/l) were about a third of that of Loch Logan (Venter, 2000).

During May to August the PO_4-P concentration in the bottom water was very low, due to the addition of 200-litre Sudfloc - commercial flocculent - to Loch Logan's water, for experimental purposes in order to bring about P inactivation (see **6.2.4**). The flocculent also reduced the PO_4-P of the Inflow and Wall Surface sampling points for May and June.

Nutrients that are stored in the sediment can be re-introduced into the water-column. This occurs under anoxic conditions and with phosphorous release, iron and sulphate both play a role (Charboneau, 1999) (see **3.4.5.1**). The release of phosphorus from sediments is a significant feature of lakes with anoxic hypolimnia (Rippey *et al.,* 1997); this internal loading can sustain eutrophic conditions and probably occurs in every eutrophic lake (Cooke *et al.,* 1993a).

The microbial loop also plays part in the system. Soluble organic matter, known as extracellular products of photosynthesis (ECPP), continually leaks from phytoplankton cells. Small fragments lost during feeding and organic matter from fish and zooplankton excretion together with ECPP form a pool of "soluble detritus" analogous to the particulate detritus of rivers and terrestrial ecosystems. Soluble detritus is used by small hetrotrophs (bacteria, microflagellates and ciliates) and returns to the main food web when larger grazers, such as the cladoceran *Daphnia*, rotifers, "herbivorous" copepods, and even very small fish consume it. In this way organic matter remains part of the planktonic habitat. It becomes important in summer when low inorganic nutrient levels reduce the productivity of larger phytoplankton (Horne & Goldman, 1994).

In oxygenated sediments phosphorus is immobilised by sorption with Fe^{3+} to form $FePO_4$ and this process is reversed under reducing conditions when Fe^{3+} changes to Fe^{2+} and PO_4^{3-} and hydrogen sulphide (H_2S) are released. Fe^{2+} and S^{2-} bind to precipitate as ferrous sulphide (Caraco *et al.*, 1993; Horne & Goldman, 1994; Kelley & Smith, 1996). Clear H_2S odours from the bottom samples indicated that these processes were also present in Loch Logan. The higher levels of H_2S in the water might have elevated the pH (Stephen *et al.*, 1997).

3.4.4 SILICA-SILICON (SiO_2-Si)

The average of Loch Logan's silica concentration was 8.3 mg/l, which corresponded with that of Camps Drift Canal (1990-94) (7.16 mg/l) and Nigel Dam (1991) (13 mg/l) (Wiechers *et al.*, 1996). Lakes world-wide has a silica concentration that ranges form 0.5 mg/l to 60 mg/l (Horne & Goldman, 1994). Although silica is not an indicator of eutrophication or pollution, it did fluctuate with algal (diatom) growth. There is also a seasonal pattern to be seen in the silicon concentrations (**Figure 3.8**). During the winter months it increased and during the summer months it decreased again, following the diatoms numbers, which were high in the summer and low in the winter (see **Chapter 4**). It is known that the silica is incorporated into the cell walls of the diatoms.

3.4.5 OXYGEN, TEMPERATURE AND pH

3.4.5.1 Oxygen

The mean percentage saturation of dissolved oxygen and concentration of dissolved oxygen in Loch Logan's surface water was 72.8 % and 6.57 mg/l, and for the bottom water only 16.5 % and 2.0 mg/l respectively. Low oxygen concentrations can result in fish kills in a system with a large fish population when the oxygen starts to deplete (Horne & Goldman, 1994; Davies & Day, 1998). The minimum oxygen concentration for intermediate fish species, e.g. carp (found in Loch Logan), is 0.8 to 2.3 mg/l, whereas an oxygen concentration lower than 0.5 mg/l is lethal to carp (DWAF, 1996c). Fish kills did occur in the summer when the bottom oxygen concentrations were very low (~0.5 mg/l). The oxygen concentration showed a seasonal variation where it was higher in the winter than in the summer. Beside the fact that oxygen is more soluble in colder water than in warmer water (see **3.4.5.3**), the metabolic processes of aquatic organisms are slower in winter than in summer and therefore need less oxygen during the winter for biochemical reactions (Horne & Goldman, 1994). However, the Chl-*a* concentration was also high in the winter and, through photosynthesis, it contributes to the oxygen concentration (see **Chapter 5**).

Oxygen is also a terminal electron acceptor in systems that have a redox potential above 500 mV, whereas in systems with a redox potential below 300 mV, SO_4^{2-} or CO_2 would be the most likely electron acceptor (Moore *et al.*, 1992). Loch Logan had an average redox potential of -52.9 mV, which means that oxygen was probably not the only electron acceptor. NO_3-N, Fe^{3+}, Mn^{4+}, SO_4^{2-} and CO_2 are also used by microbial respiration (Moore *et al.,* 1992), which could have been the electron acceptor used in Loch Logan.

With the bottom oxygen concentration so low, oxygen depletion may have occurred from time to time. Severe oxygen depletion may result in an accumulation of ammonium, methane, hydrogen sulphide, ferrous and manganous ions and increased rates of phosphorus being released from the sediment (Axler *et al.,* 1996). The release of ammonium and phosphorus from the sediment influences the concentrations already present in the water (Hamilton & Schladow, 1997), probably explaining the high ammonium concentrations in the bottom water of Loch Logan (see **3.4.3**).

3.4.5.2 Temperature

A clearly seasonal pattern as well as an obvious linear relationship was found between the air and water temperature in Loch Logan. The greatest source of heat in water is solar irradiance by direct absorbency, therefore the water was warmer in the summer than in the winter (Wetzel & Likens, 1991). The water temperature never dropped lower than the minimum air temperature (-3.3°C) or rose higher than the maximum air temperature (32.5°C), thus water provided a stable environment in terms of temperature (see **Chapter 5**).

3.4.5.3 Oxygen vs. Temperature

From the oxygen/temperature depth profiles of the water-column at the dam wall, a slight layering can be seen in the summer. This divided the water-column in an upper epilimnion and a lower hypolimnion, but with no clear thermocline in between. Because of Loch Logan's shallow depth, it is presumably polymictic and this layering occurred once a day (see **Chapter 5**) and indicated mixing that started during the afternoon and continued until the next morning.

In the winter when the water temperature was lower, there was no sign of this temporary layering. This could have been the reason why the bottom oxygen concentration was higher in the winter than in the summer. The oxygen from the surface had been mixed through the water-column to the bottom. During winter, the oxygen concentration was also on average higher (approximately 1.05 mg/l) at the surface than during the summer. The saturation concentration of dissolved oxygen ($\underline{C}^*$) is determined from the surface water layer temperature, T_S: $\underline{C}^* = \exp[7071 - 1.31\log(T_S + 45.93)]$ (Hamilton & Schladow, 1997). Horne and Goldman (1994) found that oxygen dissolved easier in colder water than in warmer water. The transfer velocity is further dependent on wind velocity and wave action (Hamilton & Schladow, 1997). The distribution of oxygen could also be helped by seiches, wind mixing (Davies & Day, 1998). A slight change in temperature in the hypolimnion also influences the release of phosphorus from the sediment (Gonsiorczyk *et al.*, 1997); in **Figure 3.7** it can be seen that with the change in seasons, there is an increase in bottom PO_4-P concentration (see **3.4.5.1** as well).

3.4.5.4 pH

The pH levels - slightly alkaline - of the surface sampling points in Loch Logan were very similar. The bottom sample's pH, however, was always about 0.5 pH units lower than that of the surface. This could be explained by the lower oxygen concentration at the bottom, together with higher CO_2 levels from decomposition (see **Chapter 1**), which lowered the pH. According to South African Water Quality Guidelines for Recreational Use (DWAF, 1996a) no health or aesthetic effects are expected in this pH range. Sixteen point seven percent ($p<0.001$) of the variation in pH (surface water) coincided with the variation in oxygen concentration. In the bottom water there were various metabolic processes going on that evidently influenced the pH.

Loch Logan's pH of 7.66 is comparable with South Africa's impoundment's average pH of 7.9. Alexander Dam had the lowest pH average (6.4) and Zeekoeivlei the highest average (8.8) (Wiechers *et al.,* 1996).

3.5 APPLICATIONS OF LAKE CONCEPTS TO LOCH LOGAN

Rivers, lakes, ponds and impoundments are all aquatic systems, but with their own definition and classifications to distinguish them from one another. One of the main differences between lentic and lotic systems is flow (Horne & Goldman, 1994)..

With the inflow of a river into a lake, three transitional zones can be found. At the river inflow there is the riverine zone. It is relatively narrow and well mixed, therefore it is also quite turbid with low light penetration. Next is the transition zone with subsequent increasing light penetration because of the settling of sediments to the bottom with slower flowing water. With the slower flow and increasing light, primary production of phytoplankton starts to pick up. Last is the lacustrine zone which has the characteristics of a lake: sedimentation is low and light penetration high. Nutrients that flowed from the river accumulate here, especially in the hypolimnion if present (Thornton *et al.*, 1990).

Similar zones can be identified in Loch Logan. The canal that feeds Loch Logan represents the riverine zone. The water is "fast" flowing and well mixed, but because the canal is made out of cement it is not that turbid all the time. The transitional zone in Loch Logan is the inflow area. It is still shallow (1-2m), so that the inflowing water mixes the water and sediment is suspended in the water-column. Finally, the rest of Loch Logan is classified as the lacustrine zone. The water is less turbid than the rest of the zones and conductivity is lower because of dilution and settling. There is also nutrient accumulation in this zone that enhances primary production. More algal growth is therefore expected here than in the canal or transition zone.

Loch Logan shows slight layering (epi- and hypolimnion) in the summer. The penetration of light also brings forth another type of zonation, namely the photic zones (see **Chapters 1** & **5**). Loch Logan has a euphotic zone of 1.4 m.

3.6 CONCLUSIONS

The physico-chemical characteristics of Loch Logan show distinctive seasonal trends. Climate and rainfall mostly influence the physical and chemical characteristics. Turbidity and conductivity variations, for instance, are affected by rainfall, which occurs mostly in the summer. The temperature of the water influences the dissolved oxygen concentration and percentage of saturation. The oxygen and the temperature, in their turn, affect nutrient release from the sediment. Loch Logan is a small system, thus highly sensitive for inflow. For example, with a catchment area of ~16 x 10^6 m^2 there is approximately 4 x 10^5 m^3 runoff water after a 25 mm thunderstorm. If only 30 % (120,000 m^3) of this runoff water ends up in Loch Logan (95,000 m^3), it can theoretically replace its total volume. The short residence time results in extreme oscillations to occur in physical and chemical parameters.

In terms of nutrient concentrations in the water of Loch Logan, it can be classified as eutrophic. The nutrient concentrations slightly differ, with the bridge sample always the highest of the surface water samples. As the nutrients entered the lake part of Loch Logan the concentration is reduced because of dilution and absorption in the sediments, as well as by algae.

Loch Logan has a lot of nutrients captured in the sediments. During 1997-1998 the Bloemfontein Transitional Local Council conducted a study and they found that the average nutrients were very high (PO_4-P = 1,479 µg/l; NH_4-N – 3,421 µg/l), the conductivity was over 50 mS/m and the pH had an average of 8.1 (Van Veelen *et al.*, 1998). However, at the same time they were conducting the study, the waterfront was being developed and the nutrients could have come from the building sites. The only place where all those nutrients could have gone through the years are into the sediments, where it can be released under certain conditions.

The high nutrient levels of the water, combined with the nutrients in the sediment, can have unsuitable long-term effects on Loch Logan, like mass algal blooms, fish kills and odour problems. To prevent this from happening, various management and restoration steps must be taken. Some of the steps may include aeration of the hypolimnion water, skimming of the surface during algal blooms, pumping in fresh, clean water to dilute the system and the dredging of the sediment to get rid of the imbedded nutrients. See **Chapter 6** for a discussion on possible management options.

CHAPTER 4

PHYTOPLANKTON OF LOCH LOGAN

4.1 INTRODUCTION

Phytoplankton is the "grass" (primary producers) of lakes and oceans and most species have world-wide distribution. They are small, free-floating algae ranging in size from the single-celled picoplankton, which are less than 5 µm in diameter, to colonial forms the size of peas (Horne & Goldman, 1994) and filamentous algae form even larger. Phytoplankton is largely restricted to lentic waters and large rivers of reduced current velocity (Wetzel, 1975).

'Phyto' means 'plant' and 'plankton' has its roots in the Greek word for 'wandering'. Hence, phytoplankton means 'wandering plant', which well describes both its nomadic way of living within a lake or pond and its wide global distribution (Brönmark & Hansson, 1998). The algae that are common in lakes can be classified into seven divisions (Horne & Goldman, 1994):

Bacillariophyta (golden-brown, diatoms): Usually microscopic, filamentous or unicellular; e.g. *Aulacoseira, Nitzschia, Navicula* spp.

Chlorophyta (grass-green): Micro- or macroscopic, filamentous, colonial or unicellular, some are flagellated; e.g. *Oocystis* sp.

Dinophyta (red-brown, dinoflagellates): Microscopic, unicellular or small chains, are all flagellated, cellulose cell wall (when present); e.g. *Peridinium, Ceratium* spp.

Cyanophyta (blue-green, cyanobacteria): Micro- or macroscopic, usually filamentous, but also unicellular or colonial, some can float and glide; e.g. *Anabaena, Oscillatoria, Chroococcus, Microcystis* spp.

Chrysophyta (yellow or brown-green): Microscopic, unicellular or colonial, some are flagellated; e.g. *Mallomonas* sp.

Cryptophyta (various colours, cryptomonads): Microscopic, unicellular, flagellated; e.g. *Rhodomonas* sp.

Euglenophyta (various colours, euglenoids): Microscopic, unicellular, flagellated; e.g. *Euglena* sp.

Oligotrophic lakes are dominated by small edible algae, mesotrophic systems have similar average levels of both size fractions, and in eutrophic systems, inedible algae show increasing dominance, which implies that differential grazing losses play an important role in modifying the responses of phytoplankton communities to enrichment (Watson *et al.,* 1997). The changes in trophic status, accompanied by growth of algae and other aquatic plants in rivers, lakes and reservoirs, are the norms used to asses the effects of inorganic nutrients on aquatic ecosystems (**Table 4.1**).

Table 4.1: Typical symptoms associated with different trophic status of an aquatic ecosystem (modified from DWAF, 1996b).

Trophic status	**Effects**
Oligotrophic	Usually moderate levels of species diversity; usually low productivity systems with rapid nutrient cycling; no nuisance growth of aquatic plants or blue-green algae.
Mesotrophic	Usually high levels of species diversity; usually productive systems; nuisance growth of aquatic plants and blooms of blue-green algae; algal blooms seldom toxic.
Eutrophic	Usually low levels of species of diversity; usually high productive systems; with nuisance growth of aquatic plants and blooms of blue-green algae; algal blooms may include species that are toxic to man, livestock and wildlife.
Hypertrophic	Usually very low levels of species diversity; usually very high productive systems; nuisance growth of aquatic plants and blooms of blue-green algae, often including species that are toxic to man, livestock and wildlife.

4.2 FACTORS INFLUENCING ALGAL GROWTH IN LOCH LOGAN

The physiological and biochemical properties of algae are dependent on external conditions and through their adaptive strategies, they survive in a variable environment.

4.2.1 LIGHT

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

Loch Logan is a fairly turbid system (see **3.3.1**) and that alter the quality of light penetration. High turbid systems often have a high potential trophic status due to high nutrient availability but carrying capacity is not reached because of strong light limitation (Dokulil, 1994).

Only a fraction of the light penetrating the water is used for photosynthesis, and the fraction absorbed by algae increases with biomass but decreases with rising inorganic turbidity. Light is attenuated rapidly in turbid waters and a small portion of the energy flux is used during photosynthesis. In addition, blue light is attenuated more rapidly than red light. Water movement can result in marked variations in the interception of light by phytoplankton and the depth of the mixing layer determines the time spend in the dark, with the possibility of exceeding the 'critical mixing depth' (where the photosynthesis rate of algae is equal to that of respiration). The frequency of light fluctuations appears to be much higher in turbid than in clear waters (Grobbelaar, 1985).

Fractional light absorption has been found to increase with biomass. Although the fraction of light, which is intercepted by phytoplankton, decreases with rising non-algal turbidity it was found that the specific activity and light utilisation index increased under such conditions and it was concluded that the phytoplankton was shade adapted. The smaller the euphotic:aphotic zone ratio, the lower will the overall productivity be (Grobbelaar, 1989).

4.2.2 TEMPERATURE

Temperature is a major factor controlling the rate of photosynthesis in all plants. In addition, photosynthetic algae occur in the hottest and coldest environments in which autotrophic plants can be found (Davison, 1991). Regarding warm temperate lakes (winter temperatures above 10°C and summer temperatures of 28-30°C) studies showed broad differences between phytoplankton communities in lakes with different chemical compositions (Grover *et al.,* 1999).

Temperature plays a role in photosynthesis: it affects enzymes and physical processes such as diffusion and cellular pH. The lower the temperature, the slower enzyme reactions and *vice versa*. The factor by which a reaction increases when a 10°C increase in temperature occurs, is called the Q_{10}. If the reaction doubles, Q_{10} equals 2 (Salisbury & Ross, 1992). The low temperature limitation of electron transport or carbon fixation reduces the ability of the plant to use light. This results in excess light energy that may cause photoinhibition due to damage to the photosynthesis system II (PSII) apparatus. In the green uni-alga *Dunaliella*, growth at low temperatures induces a reduction in energy transfer to PSII; this is presumably a mechanism that avoids low temperature induced photoinhibition. PSII is also believed to be the most thermolaible aspect of the photosynthetic apparatus, causing the reduction in photosynthesis at temperatures above the temperature optima. In red algae, high temperature inhibitions of photosynthesis are associated with a disruption of energy transfer between phycobilisome and PSII, similar to the disruption of energy transfer that occurs in chilling-sensitive cyanobacteria (Davison, 1991).

Algae can be genetically adapted for photosynthesis at different temperatures. Low-temperature environment algae achieve higher photosynthetic rates at low temperatures; they need lower optimum temperatures for photosynthesis. In the case of high-temperature environment algae the opposite is true (Davison, 1991).

4.2.3 NUTRIENTS

Most aquatic systems are resource-limited, where P and N are often the primary limiting nutrients. In principal, the potential of a nutrient to be limiting, can be estimated from the Redfield ratio (106C:16N:1P) for the key nutrients. To ensure survival, a competitor must be able to maintain net population growth at resource levels less than required by other species. Algae are particular adapted to scavenge their environments for resources, be it through structural changes, storage or increased resource utilisation efficiency. Internal adjustments by algae involve biochemical and physiological adaptations, whilst they can also excrete substances to enhance nutrient availability. Algae excrete extracellular phosphatases almost immediately upon the onset of P limited conditions. Algae can also excrete other compounds and change the pH of their surroundings, which in turn can render absorbed P available (Grobbelaar & House, 1995).

In addition, algae can store resources like P in excess of their immediate needs. This excess or "luxury" uptake is clearly distinct from the Michaelis-Menten nutrient uptake kinetics, which is based on external resource concentrations. Later it has been concluded that the growth rate of phytoplankton is more closely related to the cellular nutrient content than to external concentrations. It is therefore necessary to establish a relationship between the cell quota of a nutrient and the growth rate of an alga. In general form it is:

$$\mu = \mu_{\max}(1-\frac{k_q}{Q})$$

where μ = specific growth rate, μ_{max} = maximum specific growth rate, k_q = the minimum cell quota for the limiting resource or the subsistence quota and Q = cell species and nutrients such as P, N (NO_3, NH_4 and area), Si, Vitamin B_{12} and Fe. In situations of decreasing nutrient concentrations, the growth rate of an organism with a constant cell quota would decrease accordingly. When an organism can decrease its cellular requirements for a nutrient, it will be able to offset much of the decrease in resource availability, thus minimising the effects of nutrient depletion on the growth rate. In terms of the steady-state nutrient assimilation, the above equation can be rewritten as:

$$\mu = \frac{\mu_{\max}[S]}{K_s+[S]}$$

where [S] = steady state substance concentration and K_s = the half-saturation constant for steady-state nutrient uptake (Horne & Goldman, 1994; Grobbelaar & House, 1995).

If an alga can adapt its quota flexibility, it might out-compete competitors in a nutrient-limited environment. High quota flexibility is common for resources found at low concentrations (P and N), whilst low quota flexibility is found for resources that are present at high concentrations. If an alga can adapt its quota flexibility by lowering its half-saturation constant, this means a higher initial slope of the nutrient-growth curve. A higher initial slope implies a high growth rate at low resource concentrations, and thus that a particular organism has a competitive advantage (Grobbelaar & House, 1995).

When an over enrichment of nutrients (mostly P) occurs in an aquatic system, it usually leads to eutrophication of the system with massive algal blooms. The blooms usually consist of certain genera of filamentous, nitrogen-fixing blue-greens (cyanobacteria) (Harding & Paxton, 2001). Sometimes blooms exist of *Microcystis* spp. Loch Logan is a textbook case, as is several of South Africa's other impoundments, e.g. Hartbeespoort Dam, where the blooms mainly consist of *Microcystis aeruginosa* (Harding & Paxton, 2001).

4.3 MATERIALS AND METHODS

4.3.1 DETERMINATION OF CHLOROPHYLL-*a* CONCENTRATION

The chlorophyll-*a* concentration of Loch Logan was measured once a week, from January 2000 to May 2001. The four sampling points were: (1) surface water and (2) bottom water at the south side of the wall, (3) at the canal inflow, and (4) in the canal at the bridge in Henry street (**Figure 2.2**).

The use of chlorophyll-*a* as an index of trophic status is based on the fact that it is normally the most abundant and important pigment in phytoplankton cells. Thus, measurements provide a convenient estimation of the algal biomass (Walmsley, 1984). Chlorophyll-*a* in Loch Logan was measured using a modified method described by Sartory and Grobbelaar (1984), and involved filtering a known volume of water, where after the filter paper was boiled for 5 minutes in 10 ml 95 % ethanol at 78°C. The absorbency was measured at 665nm and 750nm. After adding 100 µl of 0.3 N HCl, the absorbency was measured again after 2 minutes at 665nm and 750 nm with a Varian Cary 3 UV-Visible spectrophotometer.

The following formula was used:

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

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

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

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

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

4.3.2 ALGAL SPECIES COMPOSITION

Only the algae from the Bridge and Wall Surface were counted. The dominant algal species were identified with an inverted Zeiss Light Microscope after fixation with formaldehyde (final concentration of 2 %) and placed in a sedimentation chamber for at least 24 hours. The number of a specific algal species was determined in a known volume of water, counting the individuals (cells, filament and colonies) occurring in 20 blocks of known dimensions. The result was multiplied by a constant to obtain the total counts. Algal species were determined as a percentage of the total community.

4.4 RESULTS

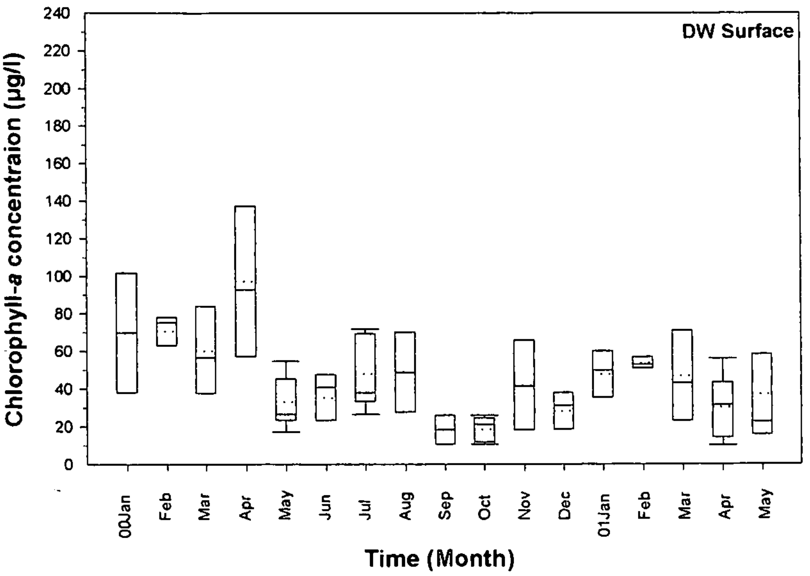
4.4.1 CHLOROPHYLL-*a*

The chlorophyll-*a* (Chl-*a*) concentration of Loch Logan ranged between 2.0 and 487.2 µg/l (**Table 4.2**). A seasonal variation was noticeable; this was also affected by rainfall. During March/April 2000 an exceptional algal bloom occurred at the Bridge, which lead to a bloom in the Inflow during April 2000. This bloom at the Bridge occurred at the same time that the PO_4-P was extremely high at that point (see **Chapter 3**). The largest fluctuation in chlorophyll-*a* occurred at the Inflow (**Figure 4.1**).

Table 4.2: Minimum, maximum and mean chlorophyll-*a* of the sampling points.

Sampling points	**Chlorophyll-*a* (µg/l)**		
	Min	**Max**	**Mean**
Wall Surface	7.9	149.0	46.3
Wall Bottom	2.0	209.2	48.9
Inflow	2.9	225.5	65.3
Bridge (canal)	2.9	487.2	66.7

The South African Quality Guidelines For Recreation Use (DWAF, 1996a) state that chlorophyll-*a* concentrations above 30 µg/l, for free-floating algae, cause severe nuisance algal blooms. Thus chlorophyll-*a* concentrations higher than 60 µg/l were considered as algal blooms. The algal blooms in Loch Logan occurred when nutrient concentrations (NO_3-N and PO_4-P) were high and turbidity was low. However, some of the blooms coincided with higher turbidity. Turbidity, together with high temperatures, are favourable for Cyanobacteria. Of all the Chl-*a* data collected, half of it was above 40 µg/l and those above 60 µg/l were 30-40 % of the total.



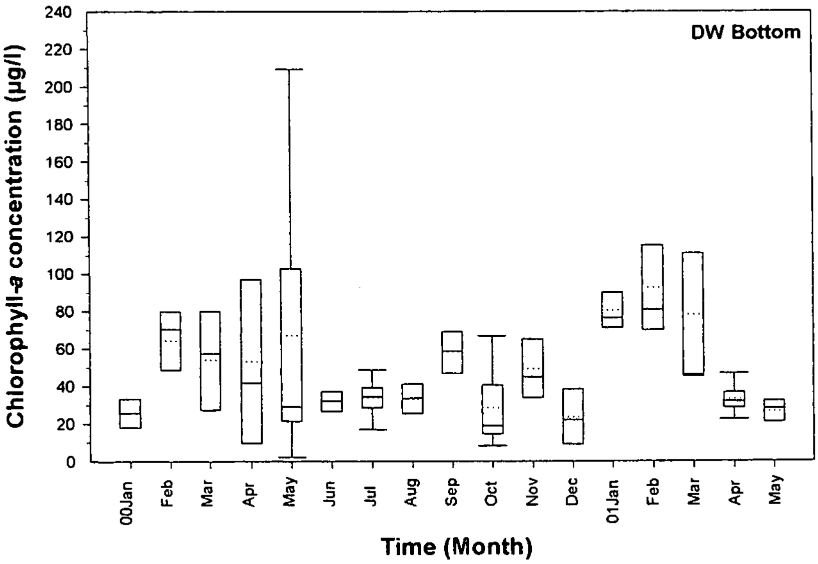
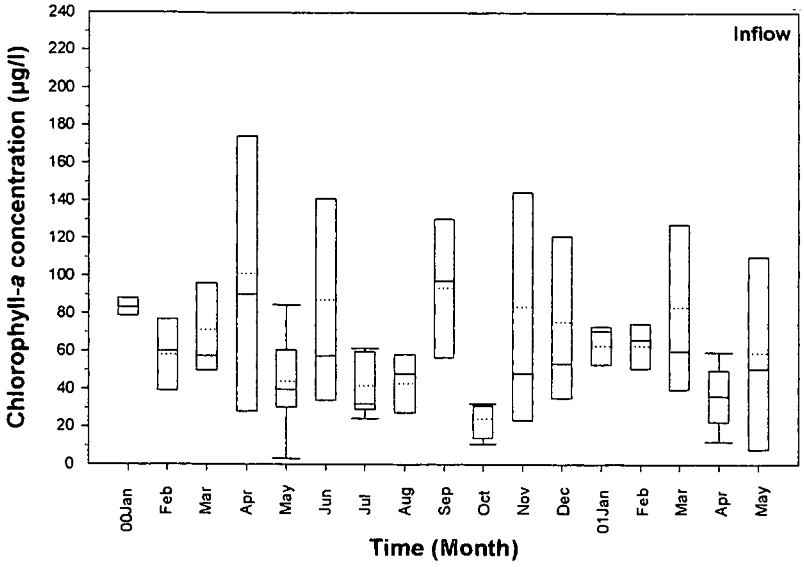


Figure 4.1a: Box plots of seasonal variation in Chlorophyll-*a* during the study period at the Dam Wall (DW) Surface and Dam Wall Bottom (dotted line is the average and solid line is the median) in Loch Logan.



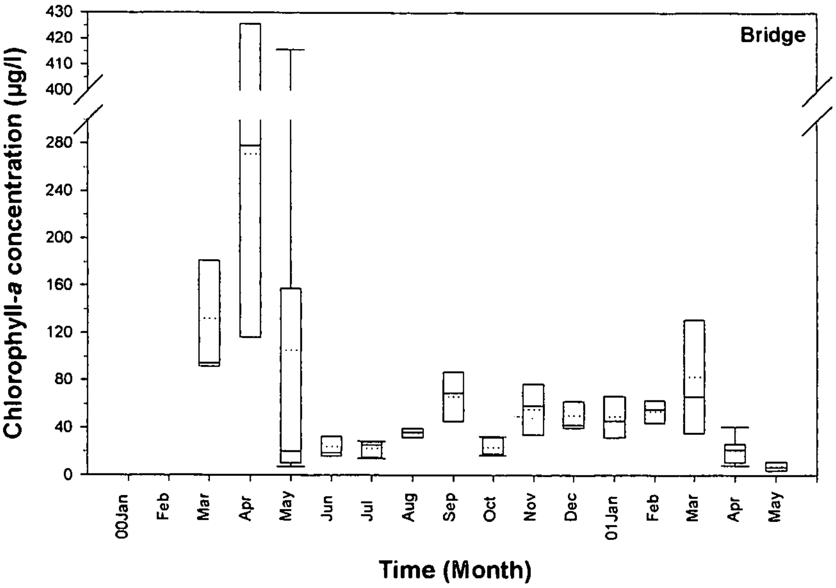


Figure 4.1b: Box plots of seasonal variation in Chlorophyll-*a* during the study period at the Inflow and Bridge in Loch Logan.

4.4.2 ALGAL SPECIES COMPOSITION

The total algal species identified in Loch Logan at the Wall Surface and Bridge consisted of 54 and 46 species respectively. These species can be divided into s divisions, namely Cyanophyta, Bacillariophyta, Chlorophyta, Cryptophyta, Euglenophyta and Chrysophyta. The average amount of algae at the Wall Surface was 15,893 cells/ml and at the Bridge it was 11,360 cells/ml (**Table 4.3**). The species listed in **Table 4.4** are those that were more than 1 % of the biomass.

Table 4.3: The mean amount of algal cells per ml in the Wall Surface and Bridge samples.

Year	Month	Mean cells/ml	
		Dam Wall Surface	Bridge
2000	Jan	70565	-
	Feb	19474	-
	Mar	23626	-
	Apr	9249	14506
	May	3867	3790
	Jun	7966	4226
	Jul	14832	11760
	Aug	14734	15393
	Sep	22382	18521
	Oct	9135	9558
	Nov	13232	20142
	Dec	21193	19040
2001	Jan	25812	21931
	Feb	13694	9923
	Mar	28546	14517
	Apr	6648	3653
	May	5883	2008

(- = not determined)

Table 4.4: The dominant species (%) of Loch Logan during total study period.

Division Species	Mean percentage of algae (%)	
	Wall Surface	Bridge
Cyanophyta	**2.95**	**8.12**
Microcystis sp. (single cells)	1.07	6.35
Bacillariophyta	**8.37**	**11.28**
Pennate diatoms (*Navicula & Nitzschia* spp.)	8.11	10.72
Chlorophyta	**84.45**	**75.58**
Actinastrum sp.	3.62	4.73
Carteria sp.	--	1.36
Chlamydomonas sp.	3.59	4.01
Chlorella sp.	53.12	37.27
Chlorococcum sp.	8.73	10.91
Oocystis sp. (colony)	1.65	1.69
Oocystis sp. (single cells)	6.35	7.11
Scenedesmus sp.	--	1.46
Monoraphidium sp.	1.34	1.57
Cryptophyta[a]	**0.01**	**0.0**
Euglenophyta	**4.13**	**4.6**
Euglena sp.	--	1.08
Phacus sp.	1.49	2.58
Trachelomonas sp.	1.62	--
Chrysophyta[b]	**0.09**	**0.12**

([a] = *Chroomonas acuta* and *Cryptococcus* sp. each occurred only once in the Wall Surface sample; [b] = *Mallomonas* sp. occurred a few times in the Bridge and Wall Surface sample, whereas *Synura* sp. only occurred once in the Wall Surface sample).

Figures 4.2 and **4.3** show the dominant divisions at the two sampling points and their seasonal variation. The algal species composition within a division (on the average) remained the same for the study period. For example, when the Euglenophyta decreased from winter 2000 to spring 2000 the numbers of species in the division stayed the same, but a decrease in cell numbers occurred.

The Bacillariophyta numbers increased during the winter months when competition from other divisions were low, and decreased when warmer conditions favour the others. Cyanophyta proliferated under the warmer conditions - their numbers increased during the summer months and in the summer 00/01 there was a bloom of *Microcystis* and *Anabaena* spp. In April 2000 there was an one-time bloom at the Bridge that consisted mostly of *Chroococcus*. However, they never occurred after that again, thus forms less than 1 % of the algal composition based on total study period. Cryptophyta seldom occurred, but Chlorophyta dominated in the water throughout the study period. Chrysophyta only occurred in the autumn and winter of 2001. The Euglenophyta numbers increased during the autumn and winter months; during the winter of 2001 there was a bloom of *Euglena*, *Lepocinclis, Phacus* and *Trachelomonas* spp.

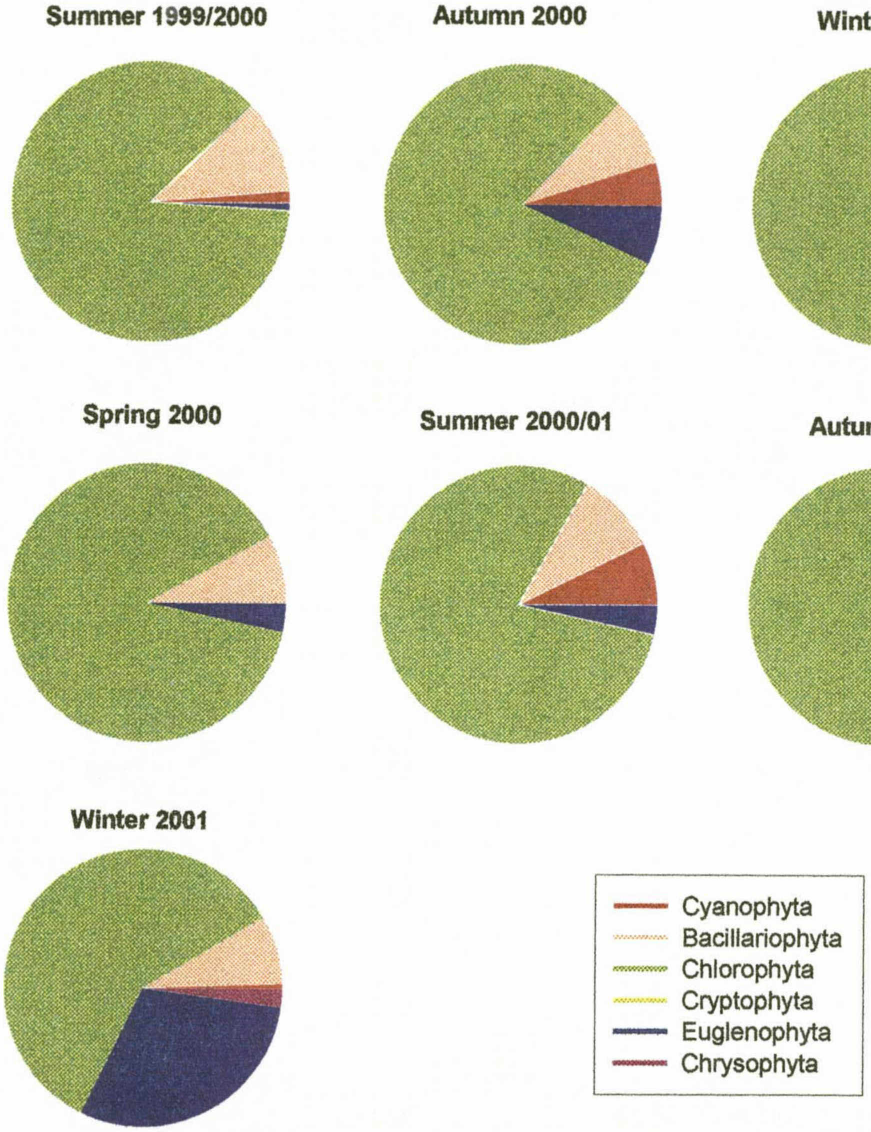


Figure 4.2: Pie charts of seasonal variation in the algal composition at the Dam Wall Surface from Summer 1999/2000 to Winter 2001 in Loch Logan.

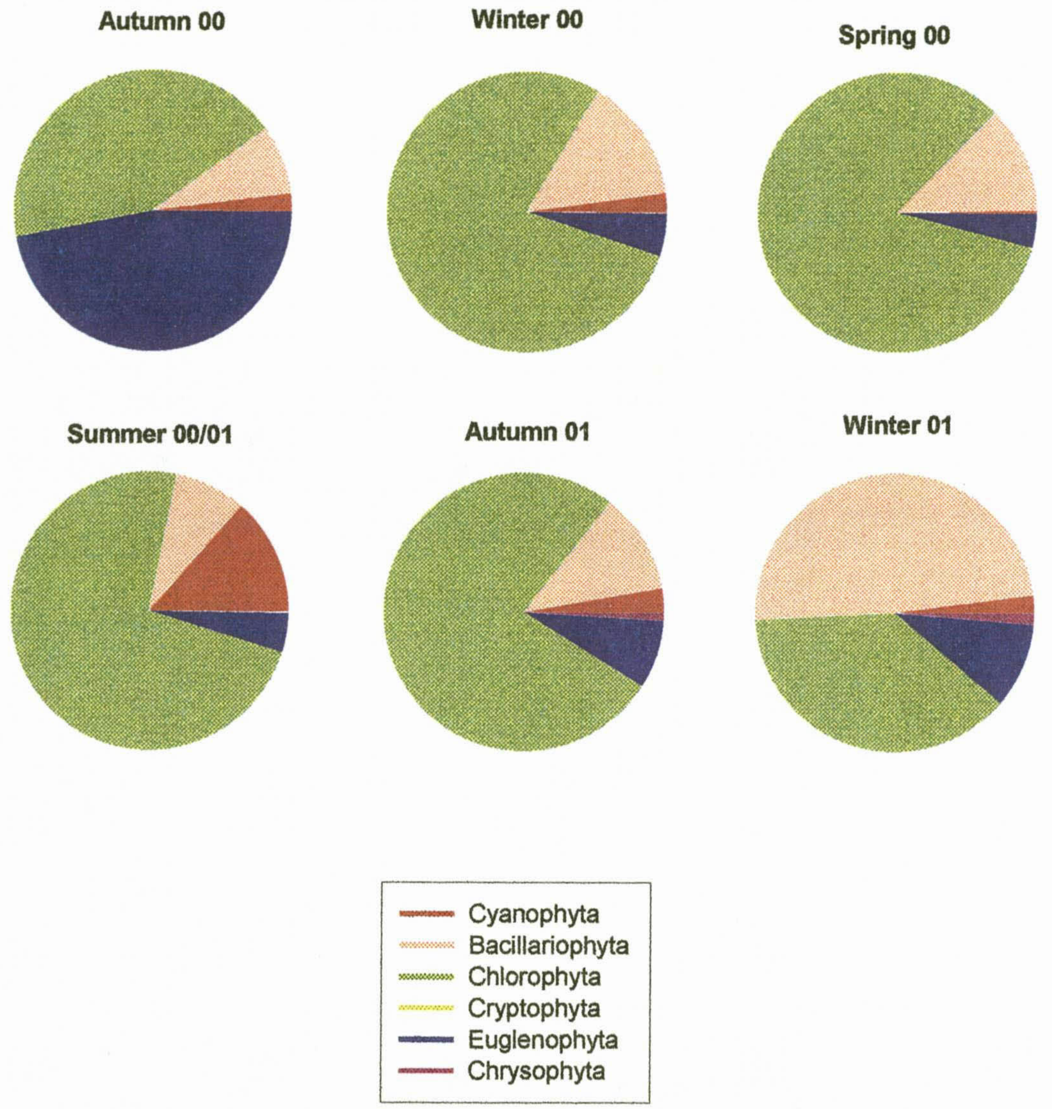


Figure 4.3: Pie charts of seasonal variation in the algal composition at the Bridge (in the canal) from Autumn 2000 to Winter 2001 in Loch Logan.

4.5 DISCUSSION

4.5.1 CHLOROPHYLL-*a*

The overall average chlorophyll-*a* concentration of Loch Logan was 56.4 µg/l. This indicates eutrophic conditions (the Wall Bottom sample excluded). According to the Water Quality Guidelines for Recreational Use (DWAF, 1996a), Chl-*a* >30 µg/l can have health and aesthetics effects, however, there is no effect on health or aesthetics with non-contact recreation, as practice in Loch Logan. A seasonal pattern occurred in all the samples: high chlorophyll in the summer and low in the winter. Algae and Chl-*a* are mainly influenced by temperature (see **4.2.2**). Other variables influencing algae include nutrients and rainfall.

There were several algal blooms that occurred in Loch Logan. Problems associated with blooms were mainly bad odours and visual impression (see **Photo's 11** & **12**). At the Dam Wall Surface and the Bridge most of the algal bloom occurrences were in the warmer months, but at the Inflow the Chl-*a* concentration was constant (=40 µg/l) throughout the year (**Figure 4.1**). The blooms at the Wall Surface and the Bridge correlated with the occurrence of high nutrients (PO_4-P and NO_3-N) (**Figures 3.6** & **3.7**) at those sampling points. When it rained, the algae were washed out of the system and nutrients into the system, and it took the algae a week to 10 days to respond to the high nutrient concentrations. Thus, it sometimes happened that the algal blooms did not peak in the same month the nutrients peaked, but peaked during the next week.

In the canal there is not much sediment that can serve as a nutrient trap, especially for phosphorus. So, most of the nutrients end up at the Inflow and are available for algae to incorporate in their cells. In Loch Logan there was a higher concentration of average Chl-*a* at the Inflow than at the Wall surface. The light penetration depth, top to bottom, also favoured high algal growth.

The relatively high Chl-*a* concentration that occurred in the Wall Bottom sample can be ascribed to various factors because there is no light at the bottom to sustain algal growth. The high Chl-*a* concentration could result from algae that move vertical by means of gasvacules in order to escape the high light intensity of the surface waters (the samples were taken at noon). Furthermore, dense carbohydrates (from photosynthesis) act as ballast during the day and also cause sinking (Horne & Goldman, 1994). It could also be the result of dead algae - it did occur synchronised with the algal blooms at the surface - that sank to the bottom and had not yet lost their Chl-*a*, or it could be dead leaf material like grass clippings or other fine organic matter that contains chlorophyll-*a*.

4.5.2 ALGAL CONTROL

There were two big algal blooms in Loch Logan in January and April 2000. In January 2000 50 kg copper sulphate ($CuSO_4 \cdot 5H_2O$) (19.907kg Cu), which is widely used as an algicide to control algal blooms (Ludyanskiy *et al.,* 1990), was added to Loch Logan. The theoretical Cu concentration after it was added to Loch Logan was calculated as 133.95 µg/l, which is well below 500 - 1000 µg/l for drinking water (WRC, 1998). However, the Cu concentration in Loch Logan was determined to be less than 80 µg/l. The difference between the theoretical and actual concentration could partially be ascribed to absorption of Cu in the sediment. The lethal concentration of Cu for *Selenastrum* and *Scenedesmus* is above 90 µg/l and for *Chlorella* (dominant algae in Loch Logan, see **4.5.3**) above 95 µg/l (Wong, 1989). The $CuSO_4$ was immediately effective by suppressing photosynthesis and by destructing organic matter (Ludyanskiy *et al.,* 1990), and the Chl-*a* decreased within two days. In April 2000 the $CuSO_4$ (3 kg) was only added at the Bridge sampling point, but had an effect within a weeks time. The algicidal action of $CuSO_4$ increases with rising temperatures (Ludyanskiy *et al.,* 1990), thus the quick reaction.

4.5.3 ALGAL SPECIES COMPOSITION

The structure of the phytoplankton association and the abundance of species vary along trophic and seasonal gradients. Interactions between these gradients distinguish among the functional characteristics of species to influence the composition of phytoplankton assemblages (Huszar *et al.,* 1998).

Phosphorus is generally the primary factor limiting phytoplankton growth and production in most lakes (Watson & McCauley, 1988) (**Figure 4.4**). The more the phosphorus concentration increase, the more the algal biomass concentration increase. The Cyanophyta's biomass increases the fastest with increasing phosphorus against that of Chrysophyta whose biomass stays constant. Other factors that also have an influence on phytoplankton growth are pH, temperature, light and rainfall. The average dissolved inorganic nitrogen (DIN):dissolved inorganic phosphorus (DIP) ratio of Loch Logan was 28.5:1, which indicated a phosphorus limiting system. However, most of the weekly measured DIN:DIP ratios were lower than 10:1, the median ratio was 5:1, which suggested that Loch Logan is most of the time nitrogen limiting, from there the occurrence cyanobacterial blooms.

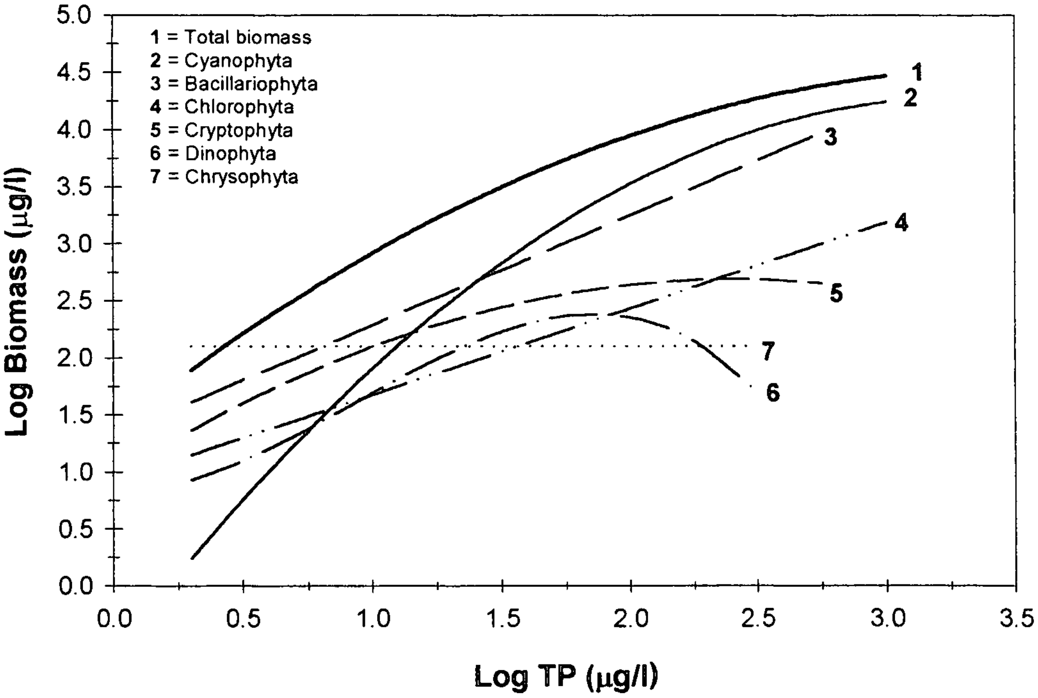


Figure 4.4: The change in total and taxonomic group average summer biomass with total phosphorus (TP) as predicted by a polynomial least-squares regression analysis. Data from 91 north temperate lakes (Redrawn from Watson *et al.*, 1997).

Some of the species found to be dominant in Loch Logan pose problems to water purification, while others act as indicators of water pollution. These characteristics of the dominant species in Loch Logan are listed in **Table 4.5**. There is not a noteworthy difference in dominant species between the Wall Surface and the Bridge sampling points. *Trachelomonas* sp. was the only species that occurred as dominant at the Wall Surface but not at the Bridge; whereas three species, *Carteria*, *Scenedesmus* and *Euglena*, occurred as dominant at the Bridge but not at the Wall Surface.

Table 4.5: Some characteristics of and problems caused by the dominant species found in Loch Logan (*Knutsom & Sterk, 1996; ** Standard Methods, 1995; ***Canter-Lund & Lund, 1995; ****Rosén, 1981).

SPECIES	CHARACTERISTICS
Microcystis sp.	-Floating in open and still waters* -Common in fertile eutrophic waters*
Navicula sp.	-Clean-water algae** -Filter- and screen-clogging algae* -Commonly found in and on sediments as well as in surface water*
Nitzschia sp.	-Fresh-water pollution algae*** -Second largest genus of freshwater algae***
Actinastrum sp.	-Surface water algae**
Carteria sp.	-Freshwater pollution algae**
Chlamydomonas sp.	-Freshwater pollution algae** -Wastewater-treatment-pond algae**
Chlorella sp.	-Filter- and screen-clogging algae** -Freshwater pollution algae** -Commonest genus living with freshwater invertebrates***
Chlorococcum sp.	-Widespread in aquatic habitats and soil***
Oocystis sp.	-Surface-water algae* -Mesotrophic-water algae****
Scenedesmus sp.	-One of commonest freshwater algae*** -Eutrophic-water algae****

(**Table 4.5** continues)

SPECIES	CHARACTERISTICS
Monoraphidium sp.	-Surface-water algae* -Oligotrophic-water algae****
Euglena sp.	-Commonest in water rich in organic matter*** -Freshwater pollution** -Occur in water with low dissolved oxygen***
Phacus sp.	-Commonest in water rich in organic matter*** -Freshwater pollution*** -Occur in water with low dissolved oxygen***
Trachelomonas sp.	-Commonest in water rich in organic matter*** -Filter- and screen-clogging algae** -Occur in water with low dissolved oxygen*** -Eutrophic-water algae****

Most of the dominant algae that occurred in Loch Logan were algae typical of eutrophic waters, as well as according to the algal genera pollution index score: *Euglena* – 5, *Chlamydomonas* and *Scenedesmus* – 4, *Nitzschia* and *Navicula* – 3, and *Phacus* - 2 (Leupold, 1988). From **Figure 4.4** it is clear that, with the increase in total phosphorus the algal biomass also increases, and this brings about a shift in species composition. In Loch Logan it has been found that after there was an increase in PO_4-P, an increase in algal numbers occurred, with rapid growth in some species numbers that shift the species composition for a short period. In the Lobo Reservoirs (Brazil), one eutrophic and the other one more oligotrophic, the following algae were found: eutrophic reservoir – *Melosira* (*Aulacoseira*) *italica*, various *Navicula* spp. and *Nitzschia* spp.; oligotrophic reservoir - *Melosira* (*Aulacoseira*) sp., *Ankistrodesmus* sp., *Cosmarium* sp., *Staurastrum* sp., *Oocystis* sp., *Pediastrum* sp., *Tetraedron* sp. and *Scenedesmus* sp. (Henry *et al.,* 1984). The algal species that were present in the oligotrophic Lobo Reservoir, were also present in Loch Logan, but in very small numbers.

Other lakes that have algal species comparable to that of Loch Logan are Norwegian lakes. There were *Oscillatoria* spp. and *Anabaena* spp. in the summer and autumn, and during the spring the following diatoms - *Stephanodiscus*, *Synedra* and *Nitzschia* spp. (Løvstad, 1986).

The algal divisions that dominated most during the algal blooms were Cyanophyta (*Microcystis* sp.), Chlorophyta (*Chlorella* sp.) and Bacillariophyta (*Nitzschia* sp.).

There was also a clear seasonal variation in the algal composition. The Chlorophyta (green algae) existed in large numbers throughout the study period. Algae like *Chlorella*, *Chlamydomonas*, *Chlorococcum* and *Oocystis* were present throughout the whole year, whereas algae like *Tetraedron*, *Tetrastrum* and *Staurastrum* were only present in the warmer months. Conditions that may favour green algae are high light intensity and nutrients as well as intermediate pH and temperatures (Watson *et al.,* 1997).

The Cyanophyta rarely occurred in the winter. It was found that cyanobacteria were even present in low numbers in oligotrophic waters, but it was further found that they preferred the following conditions: pH values around 8.5, water surface temperatures above 25°C, conductivity below 80 mS/m, Secchi depth between 25-45 cm (Bowling & Baker, 1996), low light intensity (cyanobacteria have in general lower light life saturation constants than green algae because of the formation of accessory pigments and increased levels of chlorophyll; Cromar & Fallowfield, 1997), and a stratified water-column (Zhang & Prepas, 1996). These conditions coincide with Loch Logan's summer conditions (**Chapter 3**). Cyanobacteria also prefer waters with high phosphorus relative to nitrogen. Some species can fix their own nitrogen from the atmosphere (Steinberg & Hartmann, 1988), to outcompete the other algae, especially green algae because cyanobacteria have a slower growth rate (Cromar & Fallowfield, 1997). However, several cyanobacterial blooms did occur in Loch Logan, but it was overall dominated by green algae.

Most cyanobacteria can produce toxins that can be a threat to living organisms, especially people, since water-treatment processes only partially filter out cyanobacteria and dilute their toxins. Two of the most toxic cyanobacteria species are *Anabaena flos-aquae* and *Microcystis aeruginosa*. *Anabaena* and other cyanobacteria produce the following neurotoxins: anatoxin-a, anatoxin-a(s) (only *Anabaena*), saxitoxin, and neosaxitoxin (**Figure 4.5**). *Microcystis* also produces hepatotoxins. Those hepatotoxins consisting of seven amino acids are called microcystins and those consisting of five amino acids are called nodularins (Carmichael, 1994) (**Figure 4.6**).

In the early 1980's studies were conducted in the Hartbeespoort Dam (eutrophic to hypertrophic), where they found that the cyanobacterial species *Microcystis aeruginosa* comprised more than 99 % of the phytoplankton biomass, and was present for at least 10 months of the year. Toxicity tests on *M. aeruginosa* confirmed that it was toxic. Over the years many organisms, e.g. fish and birds, died as a result of these blooms (Harding & Paxton, 2001).

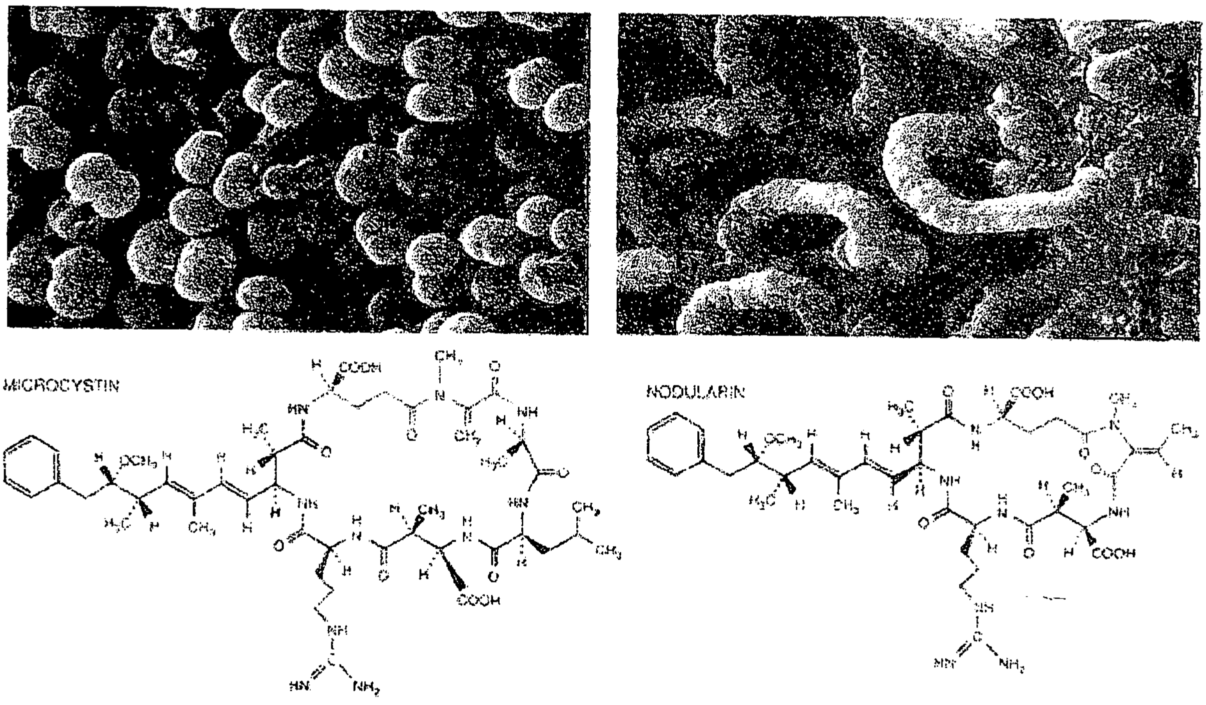


Figure 4.5: Cyanobacterial species *Microcystis aeruginosa* (left micrograph) and *Nodularia spumigena* (right micrograph), shown enlarged some 2,500 and 1,250 times respectively, are among the many forms that synthesise toxins destructive to liver cells known as hepatocytes. Microcystin (seven amino acids) and nodularin (five amino acids) are two of the toxins (Carmichael, 1994).

Although Bacillariophyta (diatoms) prefer water temperatures below 15°C (Zhang & Prepas, 1996), they (mostly pennate diatoms) were present throughout the year with an average of 9.83 % of the total algae. Other conditions that favour diatoms are well-mixed water-columns and relatively high silica and phosphorus concentrations (Zhang & Prepas, 1996). Even in the Bacillariophyta division there are certain prefences, for example *Stephanodiscus astraea* thrives in eutrophic waters with a silica concentration below 1 mg/l, in contrast, while *Melosira* (*Aulacoseira*) *granulate* is constantly found in water with a high silica content. *Stephanodiscus astraea* and *Nitzschia* sp. are very dominant in African lakes (Kilham, 1971).

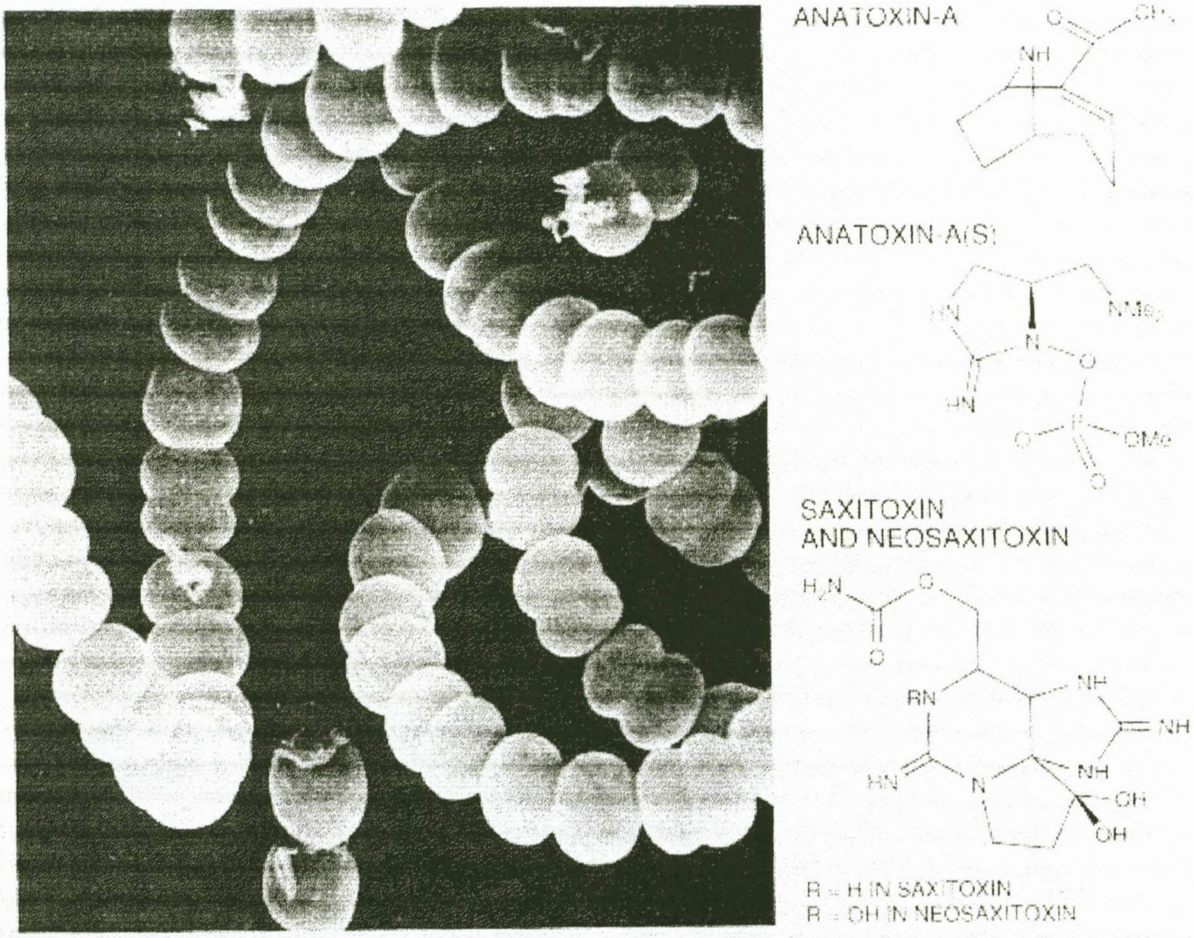


Figure 4.6: *Anabaena flos-aquae* (micrograph) magnified some 2,500 times, is a major producer of neurotoxins, poisons that interfere with the functioning of the nervous system. Anatoxin-a, anatoxin-a(s), saxitoxin, and neosaxitoxin are some of the toxins produced (Carmichael, 1994).

The Euglenophyta were present throughout the study period and are generally associated with high amounts of dissolved organic matter (Brönmark & Hansson, 1998). They were more dominant in the winter than in the other seasons, because they have the ability to migrate accordingly to light response (Horne & Goldman, 1994) to outcompete the others for light for photosynthesis in the winter.

Chrysophyta (yellow/brown/green) are usually found in oligotrophic waters (Watson *et al.*, 1997). However, the species *Mallomonas* was spotted in about 20 out of 140 samples and *Synura* in about 3 samples. Some species of Chrysophyta also use silica (Hecky & Kilham, 1988). There was only one occurrence of Cryptophyta, *Cryptococcus* sp., in Loch Logan; they rarely occur in highly eutrophic lakes (Watson *et al.*, 1997).

4.6 CONCLUSIONS

Algal growth is a function of temperature, light, nutrients and flow. The algal blooms that occurred in Loch Logan were a result of, amongst other factors high nutrient concentrations during spring and summer. The Chl-*a* concentrations during these blooms typically reached hypertrophic levels (>100 µg/l); the Chl-*a* was averagely 59.4 µg/l and caused aesthetic problems. Environmental health problems could also be caused. Although there is no full-contact recreation at Loch Logan, it is still risky: a canoe or boat may tip over during one of the blooms. Other problems caused by eutrophication are depletion of oxygen during decomposition. Fish kills may result from low oxygen levels as well as bad odours, death of other aquatic organisms and the decreasing of water quality in rivers or lakes lower in the system.

High suspended solids contribute primarily to the high turbidity in Loch Logan when it rained, but after the sediments and other particles settled out of the water-column, the turbidity was controlled by high Chl-*a* concentrations. The algal concentration had an average of 18,285 cells/ml at the Wall Surface and 12,070 cells/ml at the Bridge. This means about 395 cells/ml/µg/l Chl-*a* at the Wall Surface and 195 cells/ml/µg/l Chl-*a* at the Bridge.

Loch Logan is a very dynamic system with large variations in its limnological aspects. When it rains, the incoming water adds nutrients to the system and it also flushes the system of algae. After such a flush the system restarts its ecological cycles of nutrient availability for the algae to take up and increase in numbers.

The cycle normally started with the spring rain. High concentrations of organic and inorganic matter were flushed into Loch Logan. The decomposition of organic matter depleted the oxygen so that it caused fish kills. Through microbial decomposition of the organic matter, inorganic N and P were released into the system where algae incorporated it into their cells. This stimulated an algae bloom during the following 10-14 days and the oxygen levels, produced during photosynthesis, increased again.

Although the algal blooms were controlled by the addition of an algicide ($CuSO_4$), one should be looking for other methods to control and even to prevent these blooms (see **Chapter 6**).

CHAPTER 5

DIEL VARIATIONS IN LOCH LOGAN

5.1 INTRODUCTION

In the same way that physical and chemical characteristics differ between continents and regions as a result of influences from climate, geomorphology, aquatic biota that live in a particular area, etc., there are diel changes/fluctuations in an aquatic system (Davies & Day, 1998).

During diel (24h-day/night) changes the nutrient cycles play a role in the changes of nutrient concentrations, for example the switch from ammonium (NH_4-N) to nitrate (NO_3-N) (nitrification) to atmospheric nitrogen (N_2) (denitrification) and *vice versa* (nitrate assimilation), and from *ortho*-phosphate (PO_4-P) to soluble organic phosphorus and back (phosphatase induction) (Wetzel, 1975; Horne & Goldman, 1994). These changes are further influenced by other parameters, like temperature, oxygen, pH, etc., the same way as founded in seasonal variations (see **Chapter 2**).

Although appreciable quantities of organic matter synthesised by terrestrial plants within the drainage basin can be transported to freshwater ecosystems in either dissolved or particulate forms (allochtonous primary production), much of the organic matter of lakes is produced within the lake by phytoplankton, littoral macrophytic vegetation and by sessile algae (autochthonous primary productivity) (Wetzel & Likens, 1991).

The complex biochemical reactions of photosynthesis can be summarised by the general redox reaction:

$$6CO_2 + 12H_2O \underset{\text{Pigment receptor}}{\overset{}{\underset{\text{Light}}{\leftrightarrow}}} C_6H_{12}O_6 + 6O_2 + 6H_2O.$$

Techniques for measuring rates of photosynthesis are based on the stoichiometry of this reaction, e.g. rates of oxygen production, rates of utilisation of CO_2 in water or changes in the concentration of organic matter. Variations in the metabolic state of the phytoplankton can be large; measurements of primary productivity may reflect the rates of certain species rather accurately, but for other species rates are estimated poorly (Wetzel & Likens, 1991).

Light and available nutrients are the most important factors governing phytoplankton growth and primary production (Wetsteyn & Kromkamp, 1994). For many reservoirs and some lakes, phytoplankton production is limited by light attenuation by particles and not by some element like nitrogen or phosphorus. The relationship of the light climate to algal productivity of biomass has been described for numerous lakes in which phytoplankton are the principal determiners of the light climate (Lind *et al.,* 1992). Temperature also influences the primary productivity rate (see **4.2**).

5.2 MATERIALS AND METHODS

Two 24-hour diel studies were conducted in Loch Logan during 2000. The first was during the winter, from 13/07/2000 (09h00) to 14/07/2000 (12h00), and the second during the spring, from 09/10/2000 (09h00) to 10/10/2000 (09h00). These studies are illustrated in the work program in **Table 5.1**. Sampling points were the same as those described in **Chapters 3** and **4**.

The following physical and chemical parameters were determined according to the methods in **Chapters 3** and **4**: temperature, turbidity, oxygen, pH, conductivity, Secchi depth, NH_4-N, NO_3-N, PO_4^{3-}-P and Chl-*a*. The determination of the algal species composition were done as described in **Chapter 4**.

Protocol for Primary Production (*in situ*) was done as follows: 1 ml $H^{14}CO_3$, 20 µCi, was added to 2.5 L dam water, after which 100 ml was poured into each of two transparent round bottom flasks and 100 ml into one dark round bottom flask. Flasks were corked and lowered in the water-column at seven depths (0, 0.125, 0.25, 0.5, 0.75, 1.0 and 1.5 m). The incubation period was 4 hours (10h00 to 14h00). After incubation, the flasks were removed, placed in a dark box and immediately taken to the laboratory for processing. In the laboratory 5 ml of the incubated water in each flask was put in a scintillation flask with 0.5 ml 0.1N HCl. Air was bubbled through the flasks for 5 minutes to fully mix the acid with the water. 10 ml Scintillation fluid was added to each flask and the ^{14}C was counted with a Beckman LS 6500 scintillation counter. Calculations were done using a custom computer program, where the dark flask's values were subtracted from the ^{14}C up-take rates in the transparent flasks (Roos & Pieterse, 1992).

Table 5.1: Work program for 24-hour diel studies.

Day	Time	Analysis / activity
13/07 & 09/10/2000	09h00	**2** & prepare for Primary Production (PP)
	10h00	PP install
	12h00	**1** & PP depth profiles
	14h00	PP take out - to lab for processing
	15h00	**1**
	18h00	**2**
	21h00	**1**
14/07 & 10/10/2000	00h00	**1**
	03h00	**1** (only 14/07)
	06h00	**2**
	09h00	**1**
	12h00	**1** (only 14/07)

1 = Temperature, oxygen, depth profiles, pH, redox potential, conductivity, turbidity and Secchi depth.

2 = **1** + Chl-*a*, PO_4-P, NO_3-N and NH_4-N.

The chemical analyses were done in the laboratory directly after sampling.

5.3 RESULTS AND DISCUSSION

5.3.1 TURBIDITY

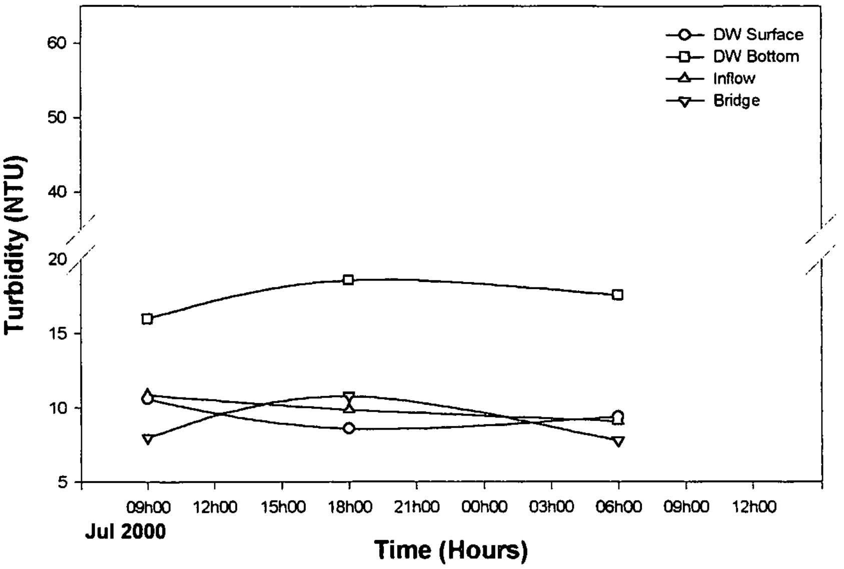
The turbidity of the surface water ranged from 7.8 to 10.9 NTUs (Jul) and from 5.8 to 15.2 NTUs (Oct) (**Table 5.2**).

Table 5.2: Minimum and maximum turbidity during the 24-hour studies.

Sampling points	Date	Turbidity (NTU)	
		Min	Max
Wall Surface	Jul 2000	8.6	10.6
	Oct 2000	5.8	9.5
Wall Bottom	Jul 2000	16.0	18.6
	Oct 2000	13.7	60.0
Inflow	Jul 2000	9.1	10.9
	Oct 2000	7.4	12.8
Bridge	Jul 2000	7.8	10.8
	Oct 2000	8.0	15.2

During July, the turbidity shows no clear diel pattern (only three samples were taken at each point); in October there was a bit of fluctuation at the bottom. The reason for this fluctuation was unknown. High turbidity levels were recorded in the bottom water sample of October (**Figure 5.1**).

The turbidity of the surface water did not change much over the 24 hours of the study periods, but the high turbidity of the Wall Bottom samples indicated a high organic load. This load could be from organic matter flushed in by rain from the canal (more than 90 mm spring rain had already fallen). High microbial action in decomposing organic matter resulted in low oxygen concentrations (see **5.3.4.1**).



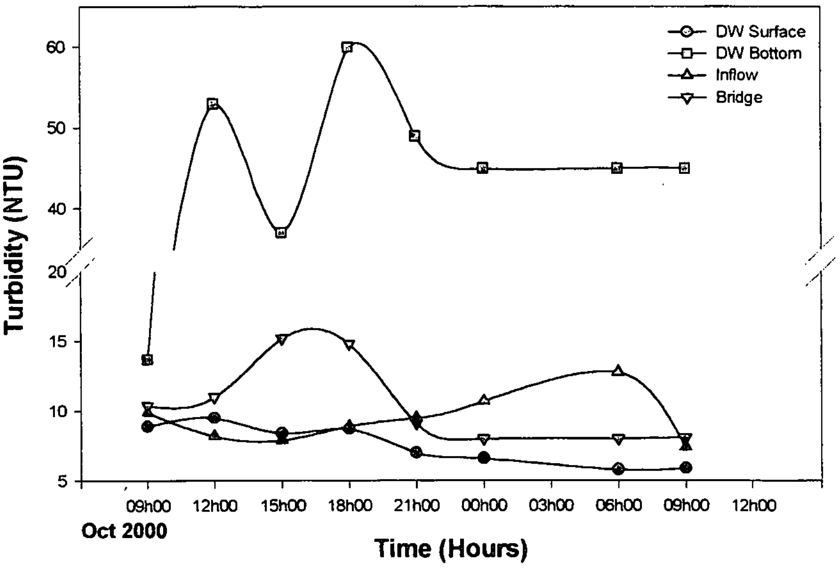


Figure 5.1: The diel changes of turbidity in Loch Logan for the July and October 2000 24-hour studies.

The Secchi disk was used as an alternative method to measure light penetration in water (Lampert & Sommer, 1997). By its basic function of reflecting light from the surface of the disc (Wetzel & Likens, 1991), a mean of 73.75 cm was recorded during July and 105.42 cm during October in Loch Logan. The lower the light intensity became in Loch Logan at the end of a day, the shallower the Secchi depth was due to low light penetration into the water. Therefore, the Secchi disk-depth was not regarded as a good indication of light penetration.

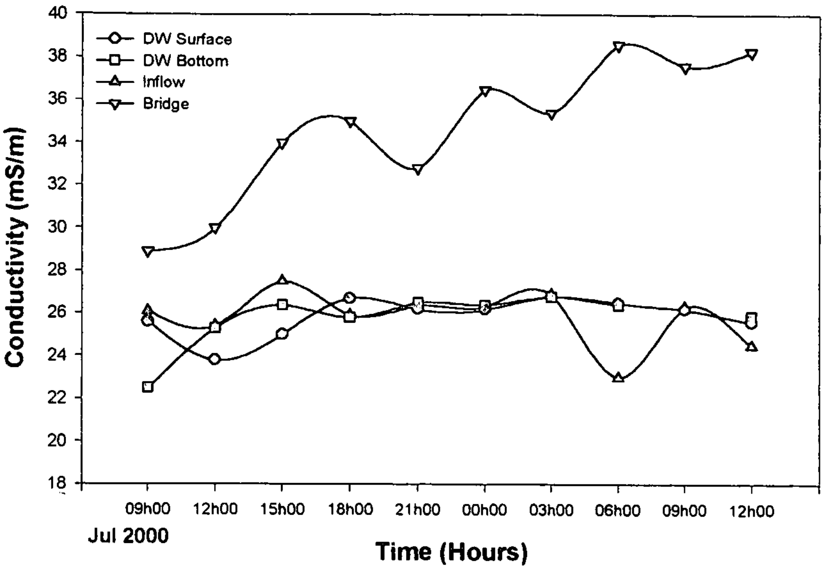
5.3.2 CONDUCTIVITY

The conductivity of Loch Logan ranged between 22.5 and 38.6 mS/m (Jul) and from 19.5 to 29.8 mS/m (Oct) (**Table 5.3**).

Table 5.3: Minimum and maximum conductivity during the 24-hour studies.

Sampling points	**Date**	**Conductivity (mS/m)**	
		Min	**Max**
Wall Surface	Jul 2000	23.8	26.8
	Oct 2000	19.5	22.7
Wall Bottom	Jul 2000	22.5	26.8
	Oct 2000	20.0	23.6
Inflow	Jul 2000	23.0	27.5
	Oct 2000	21.0	25.1
Bridge	Jul 2000	28.9	38.6
	Oct 2000	24.1	29.8

In July there was not a large variation in the conductivity over the 24 hours, except at the Bridge. The Bridge sample showed a clear increase in conductivity: an indication of saline seepage water. A slight increase was observed in October at all the sampling points during the 24-hour study period (**Figure 5.2**).



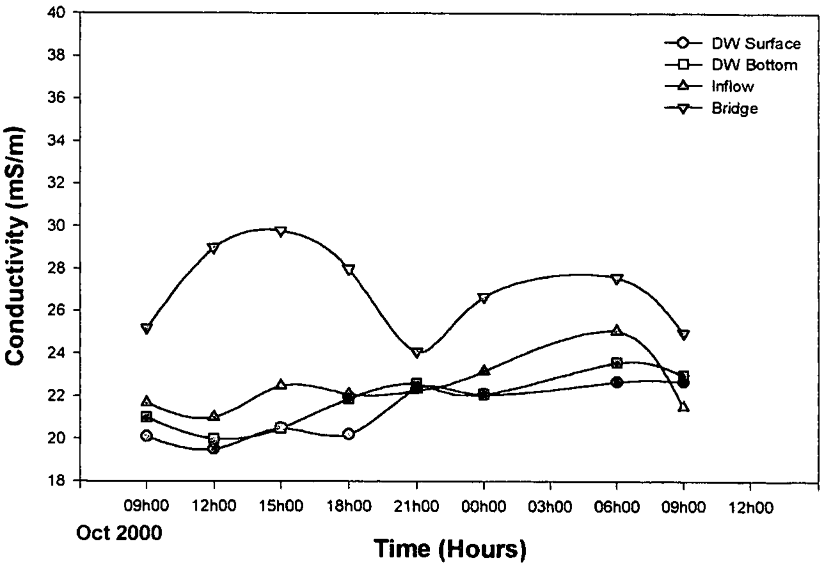


Figure 5.2: The diel changes of conductivity in Loch Logan for the July and October 2000 24-hour studies.

5.3.3 NUTRIENTS: AMMONIUM- (NH_4-N) AND NITRATE-NITROGEN (NO_3-N) AS WELL AS REACTIVE *ORTHO*-PHOSPHATE (PO_4-P)

The ammonium (NH_4-N) concentrations were relatively low during the winter (17.3 to 231.2 µg/l) in comparison with the high concentrations during spring (145.0 to 2,146.5 µg/l). The ammonium concentration during October was significantly higher in the bottom water (mean = 1,351.2 µg/l). However, the opposite was observed for the nitrate (NO_3-N) concentrations: in the winter it ranged from 1.0 to 304.5 µg/l and in the spring from 1.0 to 25.2 µg/l (**Table 5.4**).

Table 5.4: Minimum and maximum nitrogen during the 24-hour studies.

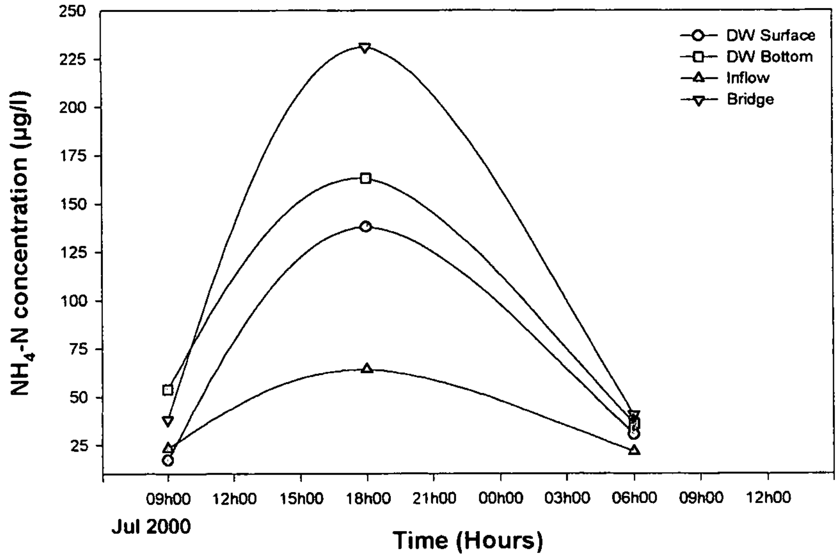
Sampling points	N	Date	Nitrogen (µg/l)	
			Min	**Max**
Wall Surface	NH_4-N	Jul 2000	17.3	138.0
		Oct 2000	527.4	945.1
	NO_3-N	Jul 2000	1.0	92.4
		Oct 2000	11.7	24.9
Wall Bottom	NH_4-N	Jul 2000	35.9	163.0
		Oct 2000	421.1	2,146.5
	NO_3-N	Jul 2000	1.0	128.3
		Oct 2000	1.0	25.2
Inflow	NH_4-N	Jul 2000	21.5	64.0
		Oct 2000	410.4	744.5
	NO_3-N	Jul 2000	1.0	1.0
		Oct 2000	3.0	20.9
Bridge	NH_4-N	Jul 2000	35.1	231.2
		Oct 2000	145.0	338.4
	NO_3-N	Jul 2000	0.7	304.5
		Oct 2000	1.0	15.3

In July the NH_4-N reached its peak during the late afternoon and the same goes for October, except for the Wall Surface and Bridge (**Figure 5.3**). The NO_3-N concentration, in July, was the lowest during the night, and in October it was almost the same except for the Bridge (**Figure 5.4**). In the Vaal River near Venterskroon, diel studies showed an increase of NH_4-N by day and a decrease by night, as well as a decrease in NO_3-N around 12h00 (Pieterse & Roos, 1992). Similar results were recorded in Loch Logan.

There are several reactions that could influence the concentrations of ammonium and nitrate in the water. Increases in ammonium are influenced by, the inflow of polluted water and ammonification where organic detritus, amino acids (-(NH_2)), are converted to NH_4^+, the excretions of fish and zooplankton (Lampert & Sommer, 1997); and under anoxic conditions NH_4^+ is released from the sediment and NO_3^- is converted back to NH_4^+ through nitrate assimilation. Decreasing ammonium concentration is caused by uptake from phytoplankton and through nitrification ($NH_4^+ \rightarrow NO_3^-$) (Horne & Goldman, 1994). In July there was a higher Chl-*a* concentration, which could be the reason for the low ammonium concentration during the day and the high concentrations in the evenings. There was none or less uptake of ammonium during the night.

The increase in nitrate concentrations is also influenced by inflow of polluted water and nitrification. The reduction is influenced by nitrate uptake of phytoplankton, denitrification ($NO_3^- \rightarrow NO_2^- \rightarrow N_2$) and nitrate assimilation (Wetzel, 1975; Lampert & Sommer, 1997).

The temperature and oxygen concentration probably also played a role in the release of ammonium, especially from the sediment at the Wall Bottom. The warmer the water gets through the day, the less oxygen is present in the water and the more ammonium is released from the sediment. If ammonium reaches a high enough concentration it can inhibit nitrate reduction and nitrate accumulates (Horne & Goldman, 1994). This releases from the sediment mostly influence the bottom water that overlays the sediment, but in July - when the water-column is well mixed - it can also influence the surface water. At the Bridge, where the water is not very deep, there was a similar reaction as at the bottom because of organic detritus accumulation, but by October the first rains (more than 90 mm since September) had already fallen and the debris were washed into Loch Logan and settled on the bottom. Most of this debris consisted of organic material. During decomposition the ammonium concentrations increase and the oxygen concentration decrease (see **5.3.4.1**).



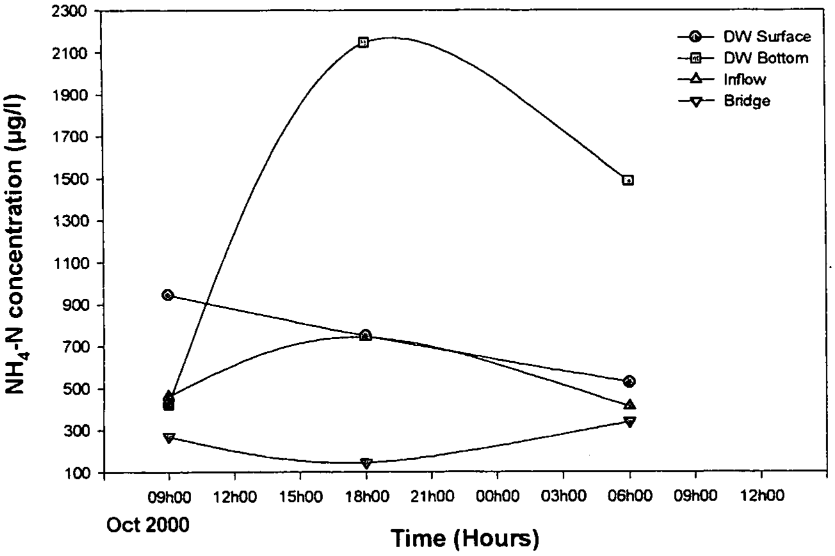
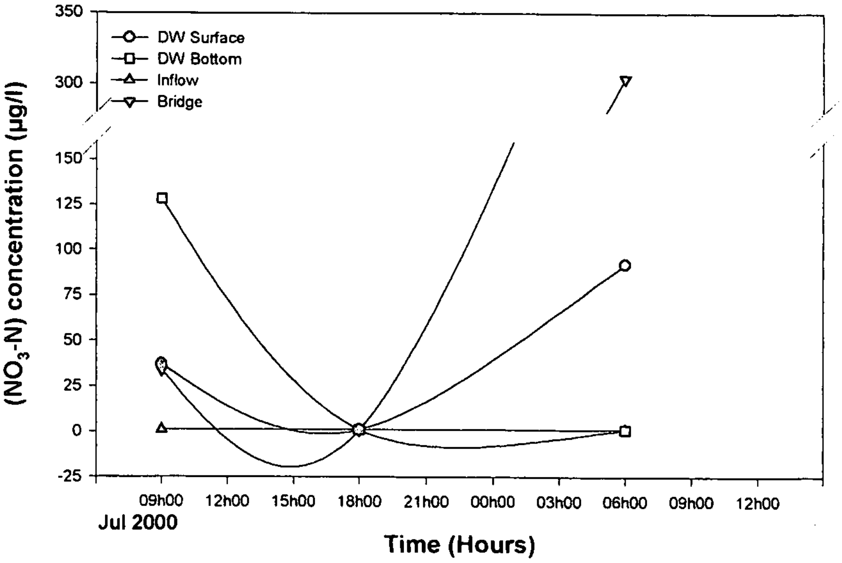


Figure 5.3: The diel changes of ammonium (NH_4-N) in Loch Logan for the July and October 2000 24-hour studies.



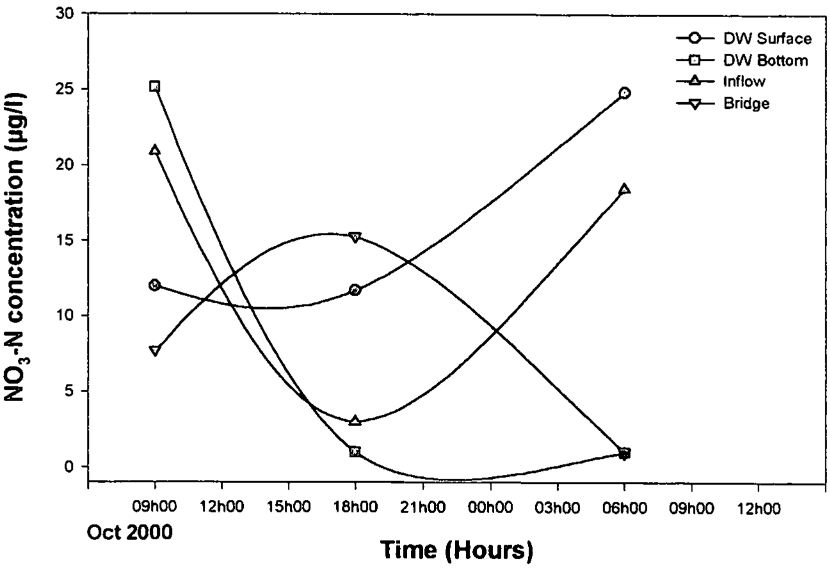


Figure 5.4: The diel changes of nitrate (NO$_3$-N) in Loch Logan for the July and October 2000 24-hour studies.

The PO_4-P concentrations ranged between 0.5 and 39.0 µg/l (Jul) and from 124.0 to 410.9 µg/l (Oct) (**Table 5.5**). In July the PO_4-P reached its lowest concentration during the night, but in October only the Bridge reached its lowest concentration during the evening (**Figure 5.5**).

Table 5.5: Minimum and maximum reactive *ortho*-phosphate during the 24-hour studies.

Sampling points	**Date**	**Reactive *Ortho*-Phosphate (µg/l)**	
		Min	**Max**
Wall Surface	Jul 2000	0.5	39.0
	Oct 2000	191.9	216.7
Wall Bottom	Jul 2000	0.5	0.5
	Oct 2000	238.9	410.9
Inflow	Jul 2000	0.5	16.8
	Oct 2000	162.3	271.0
Bridge	Jul 2000	0.5	12.7
	Oct 2000	124.0	172.7

Factors that influence the increase of PO_4 are: inflow of sewage, fish and zooplankton excretion, release from bacterioplankton and phytoplankton, decay of organic material, phosphatase of soluble organic phosphorus as well as release from sediment in anoxic conditions. The decrease in PO_4 is influenced by uptake from bacterioplankton and phytoplankton, reversed phosphatase as well as binding with iron (Fe) under oxic conditions to settle in the sediment (Horne & Goldman, 1994).

In October the oxygen level dropped very low in the bottom waters and this increased the phosphate released from the sediment, whereas in July this internal loading did not have an influence on the phosphate concentration. In October the phosphate concentration was about an order larger than that of July. The very low PO_4-P concentration during July could be ascribed to flocculants that were added to the water during a previous month (see **3.4.3**). The Inflow sampling point shows a similar pattern as the Wall Bottom in October; this could also be due to the low oxygen concentrations as phosphorus was released from the bottom.

A similar pattern to that of the PO4-P in October was found in the Vaal River near Venterskroon by Pieterse and Roos (1992).

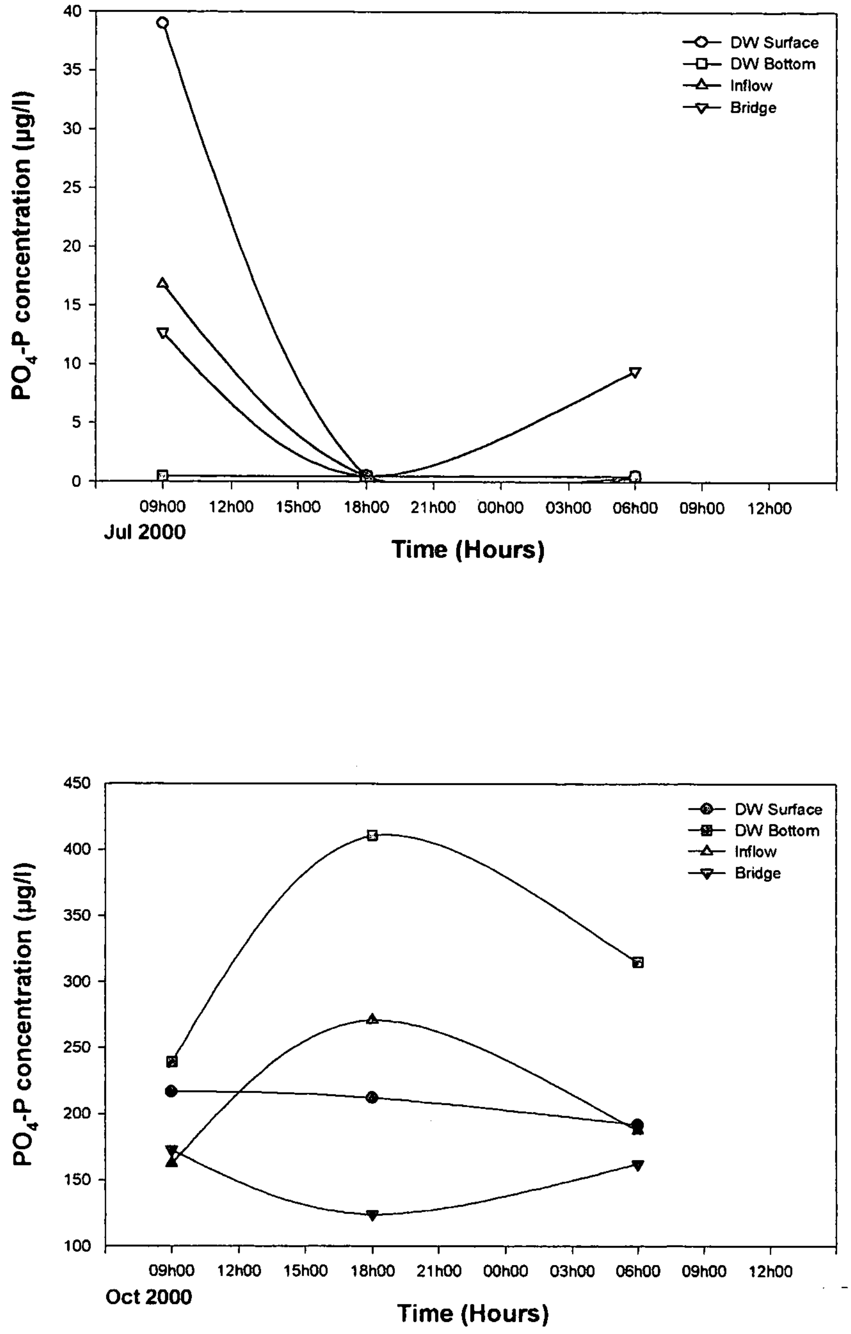


Figure 5.5: The diel changes of reactive *ortho*-phosphate (PO_4-P) in Loch Logan for the July and October 2000 24-hour studies.

5.3.4 OXYGEN, TEMPERATURE AND pH

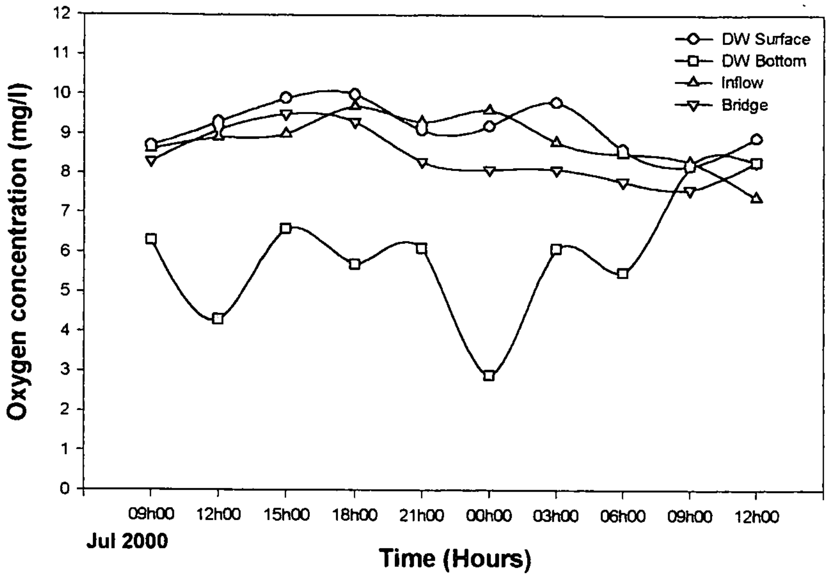
5.3.4.1 Oxygen

In Loch Logan the surface water oxygen concentration ranged during the 24-hour diel studies between 7.36 mg/l (66.0 %) and 10.0 mg/l (91.9 %) in July and between 0.3 mg/l (3.8 %) and 3.1 mg/l (35.9 %) in October (**Table 5.6** and **Figure 5.6**).

Table 5.6: Minimum and maximum dissolved oxygen and percentage saturation during the 24-hour studies.

Sampling points	Date	Oxygen			
		Min		Max	
		mg/l	%	mg/l	%
Wall Surface	Jul 2000	8.2	74.0	10.0	91.9
	Oct 2000	1.4	15.4	3.1	35.9
Wall Bottom	Jul 2000	2.9	26.1	8.25	73.2
	Oct 2000	0.2	2.5	0.5	6.0
Inflow	Jul 2000	7.36	66.0	9.7	88.6
	Oct 2000	0.6	6.0	2.4	27.8
Bridge	Jul 2000	7.6	67.0	9.5	87.2
	Oct 2000	0.3	3.8	2.8	30.0

Besides temperature, photosynthesis and respiration were the two major factors that influenced the oxygen concentration in the water. During the day the phytoplankton photosynthesised and produced oxygen, this oxygen production is, however, much higher than the oxygen consumption through respiration. During the night there was no photosynthesis and the oxygen concentration declined because of respiration. In eutrophic waters the oxygen concentration is at its highest during the afternoon and at its lowest in the morning before sunrise (Horne & Goldman, 1994).



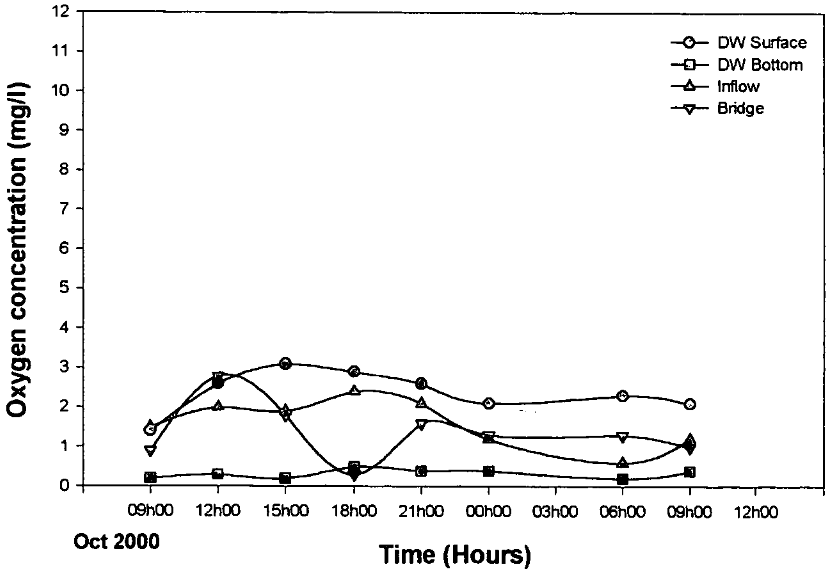


Figure 5.6: The diel changes of dissolved oxygen (O_2) in Loch Logan for the July and October 2000 24-hour studies.

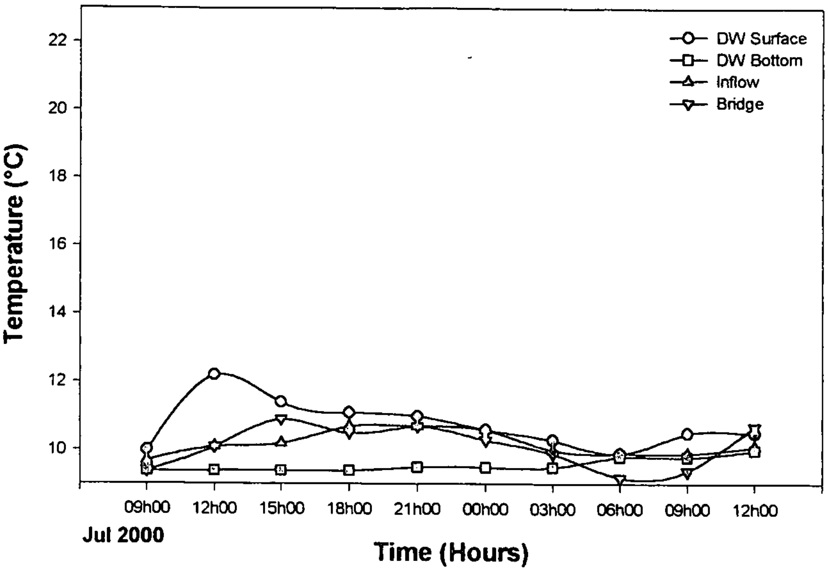
The average oxygen concentration in Loch Logan was much higher in July than in October (**Figure 5.6**). In July the water temperature was lower and consequently more oxygen was dissolved in the water. With the lower temperatures the organisms' metabolic processes were slower too and they consumed less oxygen through respiration. The Chl-*a* concentration was high in the winter diel study compared to the October diel study. The oxygen concentration was low in October because of high decomposition, high temperatures and low Chl-*a* concentrations (see **5.3.5**).

The Dam Wall Surface diel oxygen concentration during October, in Loch Logan, followed the same pattern as in the Vaal River and at Marksdrift (Orange River) (September 1998; Venter, 2000). However, Loch Logan's oxygen concentration (1.4-3.1 mg/l) was significantly lower than that of the Vaal River (8.25-10.7 mg/l) and that at Marksdrift (8.38-9.04 mg/l; Venter, 2000). Oxygen concentrations in rivers are usually higher than that in dams or impoundments, due to continues mixing of the water.

5.3.4.2 Temperature

The surface temperature ranged between 9.2 and 12.2°C (Jul) and between 17.0 and 22.0°C (Oct) during the 24-hour studies. There were distinctive day/night temperature variations in October, but less in July (**Figure 5.7**).

Water temperature is mainly influenced by atmospheric temperature. The average minimum atmospheric temperature of July was –3.3°C and the maximum was 17.4°C; the average minimum for October was 9.9°C and the maximum was 27.8°C (SA Weather Bureau). The fluctuation in water temperatures was less extreme and never ranged outside that of the atmospheric temperatures.



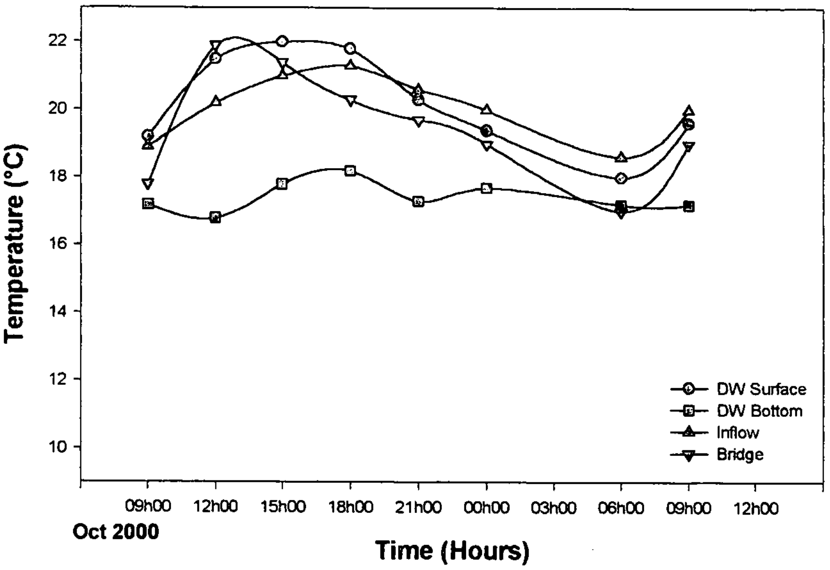


Figure 5.7: The diel changes of water temperature (°C) in Loch Logan for the July and October 2000 24-hour studies.

5.3.4.3 Oxygen and temperature depth profiles

The depth profiles of temperature and oxygen during the 24-hour diel periods illustrated clear day/night patterns (**Figures 5.8** & **5.9**). After a 24-hour period, the temperature and oxygen levels were approximately where they were the previous day.

During July the surface water temperature started to warm during the morning (13/07/2000) and increased from 10.0 to 12.2°C (12h00). As the surface water cooled down in the afternoon, the heat was distributed to lower layers and eventually displayed a uniform temperature from surface to bottom early the next morning. Throughout the twenty-four hours the temperature in the bottom water remained almost constant at about 9.5°C (**Figure 5.8**).

A similar pattern was noticed in the oxygen concentration profile. As the sun rose (13/07/2000) the oxygen concentration increased (max at 18h00, 10.0 mg/l) due to an increase in photosynthesis (**Figure 5.8: 12h00-18h00**). During the night the oxygen concentration decreased (respiration) with the lowest concentration in the early morning. Unlike the temperature there was a variation in the oxygen concentration at the bottom, i.e., during the night the oxygen concentration decreased.

The depth profiles for October's twenty-four hour study (**Figure 5.9**) were totally different from that of July. The temperature was higher in October than in July and therefore the oxygen solubility was lower. The Chl-*a* concentration was lower in October and thus photosynthesis and respiration was lower as well. In the bottom water there was neither in the oxygen concentration nor in the temperature a variation, which indicated that the water-column did not mix as well during October. The oxygen concentration in the bottom was very low and reached almost anoxic conditions, which could result in nutrient release from the sediment as seen in **5.3.3**.

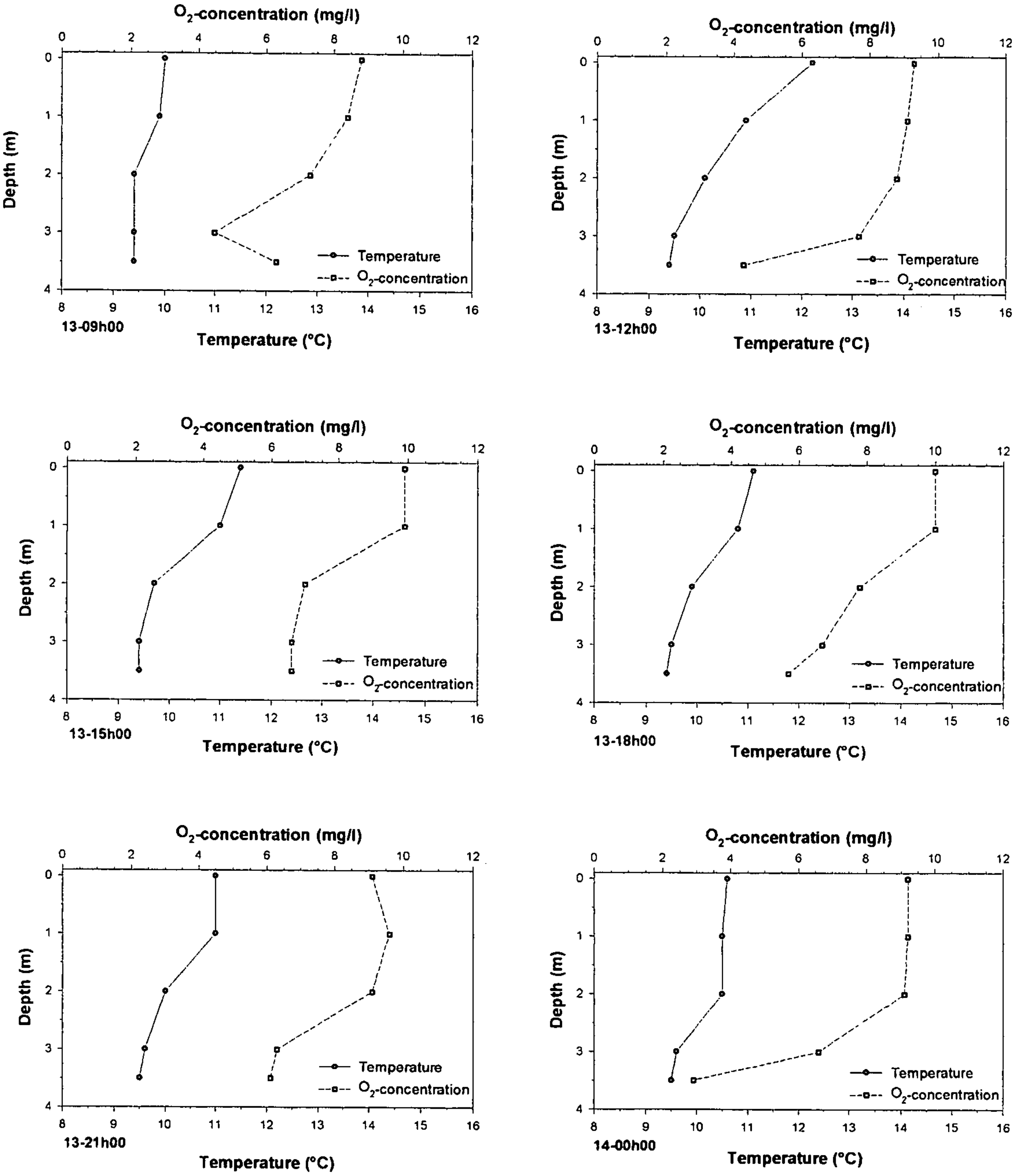


Figure 5.8a: The diel dissolved oxygen and water temperature depth profiles in Loch Logan for the July 2000 24-hour study; 09h00 to 00h00.

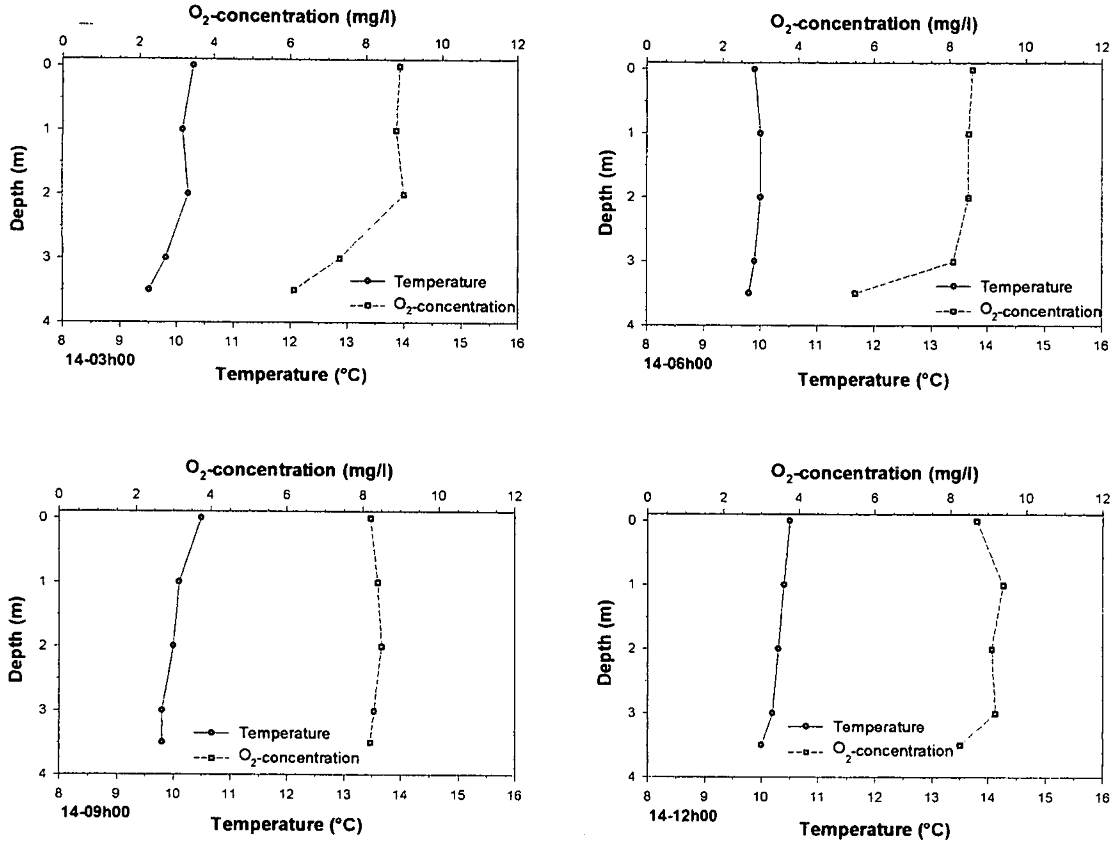


Figure 5.8b: The diel dissolved oxygen and water temperature depth profiles in Loch Logan for the July 2000 24-hour study; 03h00 to 12h00.

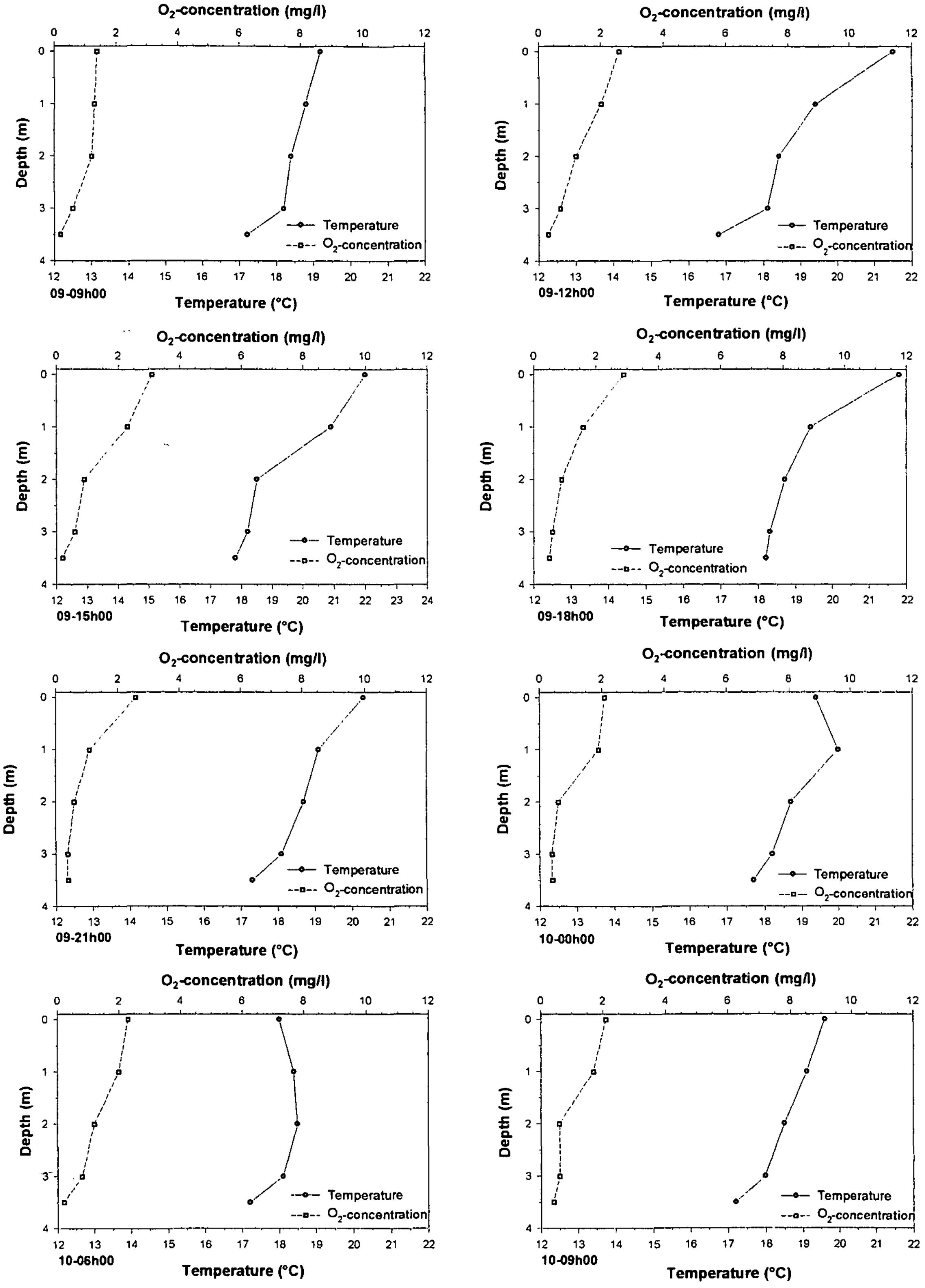


Figure 5.9: The diel oxygen and temperature depth profiles in Loch Logan for the October 2000 24-hour study.

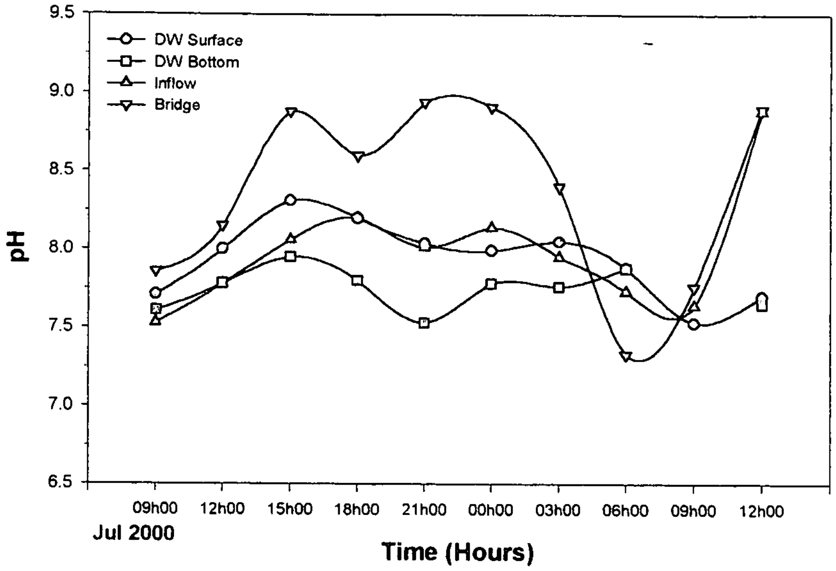
5.3.4.4 pH

The pH over the 24-hour study periods ranged between 7.53 and 8.94 (Jul) and between 6.91 and 7.46 (Oct); slightly alkaline. The Wall Bottom water showed the lowest pH (**Figure 5.10**).

The fluctuation in the surface water's pH coincided with the fluctuations in the oxygen concentration. During the day the algae photosynthesise by taking up carbon dioxide and releasing oxygen, so the pH increase. During the night respiration is more important, oxygen uptake and carbon dioxide release lowers the pH. As the oxygen level rose, so did the pH level, which peaked in the afternoon. In the early mornings it reached a minimum that coincided with the period when the oxygen level was low and the carbon dioxide (CO_2) concentration was higher. In October, however, the pH did not follow the total oxygen pattern. This could be ascribed to the very low oxygen concentration, thus the system was probably dominated by a high carbon dioxide concentration. The carbon dioxide (CO_2) concentrations were not measured, but because the pH is influenced by the CO_2 levels in the water (Horne & Goldman, 1994), it can be assumed that during the pH decrease the CO_2 levels increased as a result of aquatic organisms' respiration.

The Dam Wall Surface diel pH (7.1-7.46) during October 2000, in Loch Logan, did not follow the same pattern as in the Vaal River (8.01-8.7) nor at Marksdrift (7.97-8.14) (Venter, 2000). In Loch Logan the pH decreased during the night and increased during the day, where as in the Vaal River and Marksdrift the pH increased during the night and day.

In July there was a positive regression of 82 % between the Wall Surface oxygen concentration and pH ($r^2 = 0.82$; $p < 0.001$), but a negative regression of 6.3 % between the Wall Bottom oxygen concentration and pH ($r^2 = 0.063$; $p > 0.1$). In October both the Wall Surface ($r^2 = 0.408$; $p < 0.1$) and Wall Bottom ($r^2 = 0.182$; $p > 0.1$) had negative regressions, of 40.8 % and 18.2 % respectively, between the oxygen concentrations and pH (**Figure 5.11**).



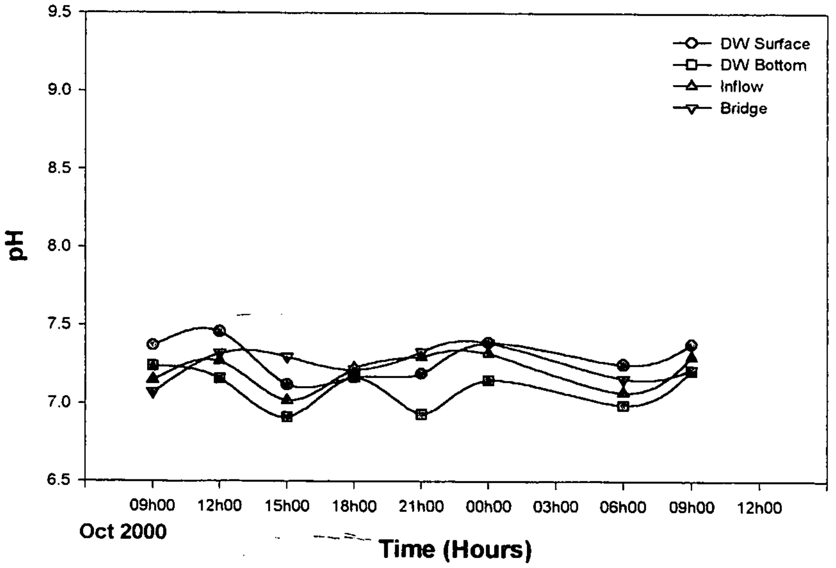


Figure 5.10: The diel changes of pH in Loch Logan for the July and October 2000 24-hour studies.

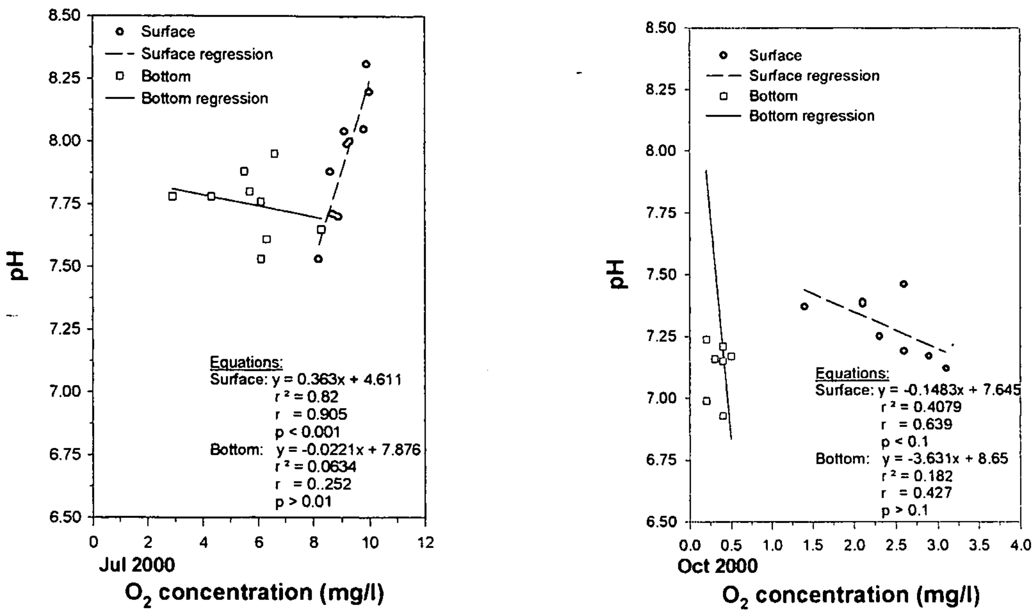


Figure 5.11: Relationship between pH and dissolved oxygen at the Wall Surface and the Wall Bottom in Loch Logan for the July and October 2000 24-hour studies.

During July at the Wall Surface there was and a linear regression, 82 %, between the pH and oxygen concentration, whereas in October the regression, 40.8 %, was inverse linear. The latter could be a result of rising CO_2 concentrations in the water through respiration, for in July the high Chl-*a* concentrations saturated the water with oxygen. In both July and October there was an inverse linear regression, 6.3 and 18.2 % respectively, between the pH and oxygen concentration levels, which was an indication of high CO_2 concentrations from microbial respiration or other processes, especially during October.

5.3.5 CHLOROPHYLL-*a*

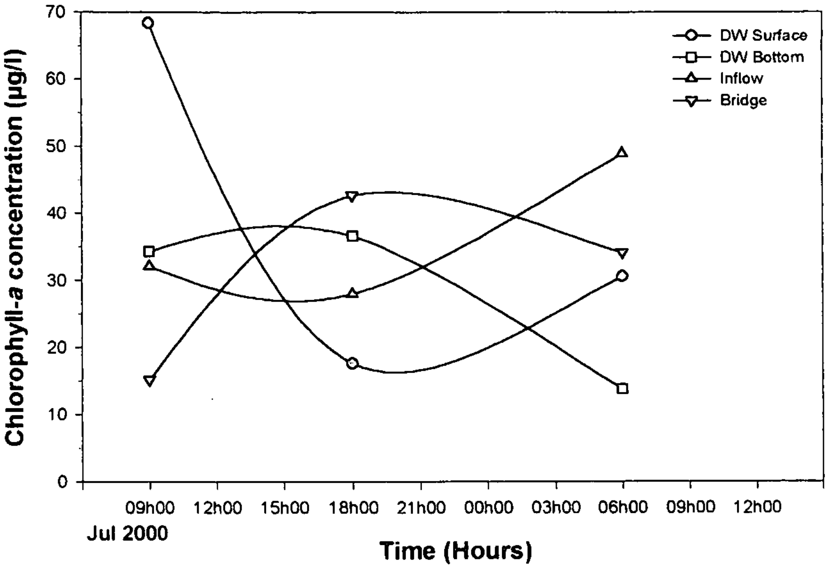
The chlorophyll-*a* concentrations in the surface waters ranged from 17.6 to 68.4 µg/l (Jul) and from 5.4 to 42.5 µg/l (Oct) (**Table 5.7** and **Figure 5.12**). The algal assemblage at the Wall Surface and Bridge is shown in **Table 5.8**.

Table 5.7: Minimum and maximum chlorophyll-*a* during the 24-hour studies.

Sampling points	**Date**	**Chlorophyll-*a* (µg/l)**	
		Min	**Max**
Wall Surface	Jul 2000	17.6	68.4
	Oct 2000	5.4	18.6
Wall Bottom	Jul 2000	13.8	36.6
	Oct 2000	18.6	71.1
Inflow	Jul 2000	28.0	48.9
	Oct 2000	10.9	12.1
Bridge	Jul 2000	32.1	42.7
	Oct 2000	16.8	42.5

The chlorophyll-*a* concentration at a given point can be influenced by several factors. Flagellated algae are able to migrate vertically accordingly to the light intensity in the water; the higher the light intensity the deeper they will migrate in the water-column to avoid photo-inhibition. For small algae viscous forces are important to keep them from sinking too fast, others use shapes (Wetzel, 1975; Horne & Goldman, 1994).

In July there was no clear pattern between the Chl-*a* concentrations at the difference sampling points. This could be due to the wind that blew on the 13/07/2000 and mixed the water-column. In October all the Chl-*a* concentrations reached a peak at 18h00, which indicated an increase during daytime and a decrease during the night. There were more flagellated algae per ml in October than in July (**Table 5.8**).



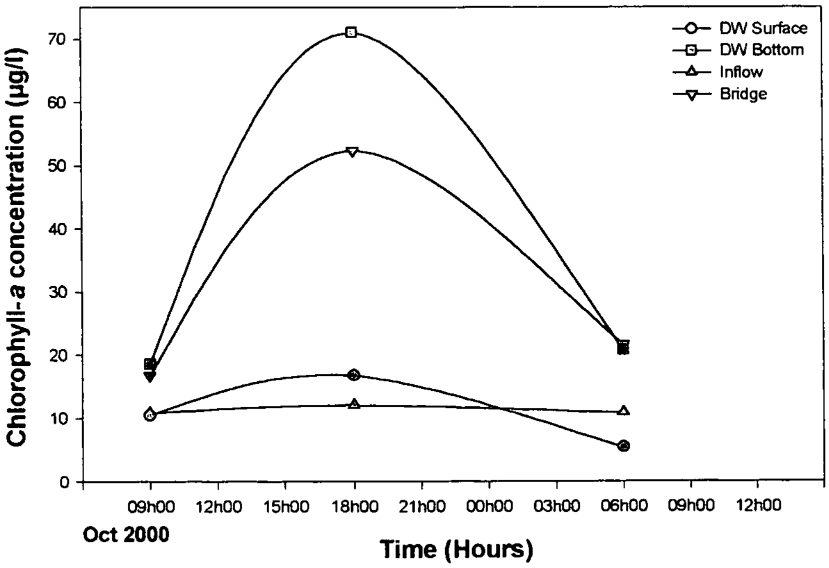


Figure 5.12: The diel changes of chlorophyll-*a* in Loch Logan for the July and October 2000 24-hour studies.

The high concentration of Chl-*a* in the bottom water during the late afternoon, may be the result of the photosynthetic production of dense carbohydrates during the day, which acts as ballast to cause sinking. The ballast is used up overnight and the algae float to the surface in the morning to resume the cycle (Horne & Goldman, 1994).

Table 5.8: Species present at Wall Surface and Bridge.

SAMPLES	**13/07/2000**				**09/10/2000**			
	Wall Surface		**Bridge**		**Wall Surface**		**Bridge**	
DIVISIONS **SPECIES**	**Cells/ ml**	**%**	**Cells/ ml**	**%**	**Cells/ ml**	**%**	**Cells/ ml**	**%**
Cyanopyhta	**147**	**1.3**	**147**	**1.0**	--	--	**74**	**0.6**
Microcystis sp. (single cells)	--	--	--	--	--	--	74	0.6
Bacillariophyta	**2,636**	**22.8**	**2307**	**16.0**	**330**	**3.3**	**447**	**4.2**
Aulacoseira sp.	--	--	--	--	--	--	37	0.6
Navicula sp.	--	--	74	0.5	--	--	--	--
Nitzschia sp.	2,636	22.8	2,233	15.5	330	3.3	440	3.9
Chlorophyta	**8,791**	**75.9**	**11,937**	**83.1**	**9,778**	**96.3**	**10,804**	**94.2**
Carteria sp.	--	--	257	1.8	--	--	--	--
Chlamydomonas sp.	440	3.8	513	3.6	220	2.2	--	--
Chlorella sp.	6,114	52.8	9,115	63.4	8,127	80.0	9,591	83.7
Chlorococcum sp.	1,209	10.5	1,245	8.7	184	1.8	257	2.2
Crucigenia sp.	--	--	--	--	--	--	37	0.3
Gleocystis sp.	37	0.3	--	--	--	--	--	--
Monoraphidium sp.	403	3.5	330	2.3	--	--	74	0.6
Oocystis sp. (colony)	74	0.6	37	0.3	184	1.8	184	1.6
Oocystis sp. (single cells)	440	3.8	403	2.8	842	8.3	440	3.9
Planktosphaeria sp.	37	0.3	--	--	--	--	--	--
Scenedesmus sp.	--	--	37	0.3	--	--	--	--
Sphaerocystis sp.	--	--	--	--	37	0.4	37	0.6
Spirogyra sp.	--	--	--	--	184	1.8	184	1.6
Tetrastrum sp.	37	0.3	--	--	--	--	--	--
Euglenophyta	--	--	--	--	**37**	**0.4**	**110**	**1.0**
Phacus sp.	--	--	--	--	37	0.4	110	1.0
TOTAL	**11,574**	**100**	**14,391**	**100**	**10,145**	**100**	**11,465**	**100**

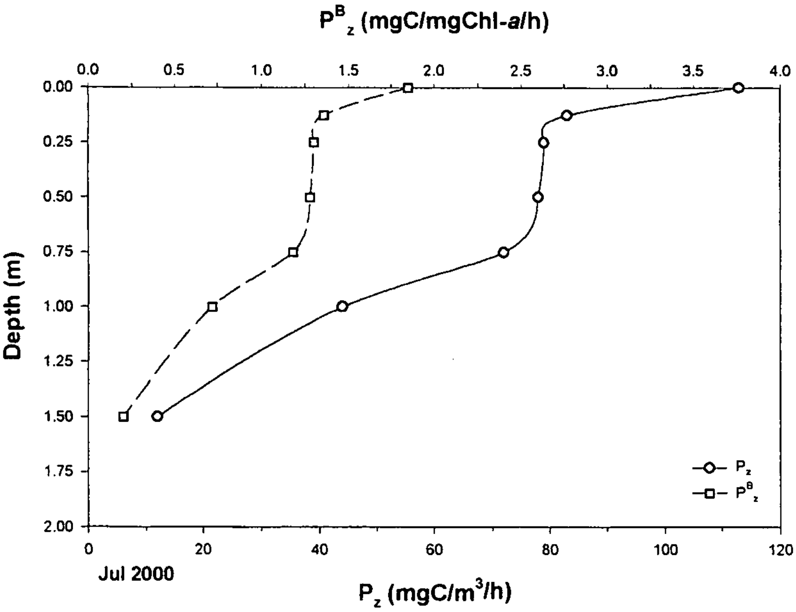
5.3.6 PRIMARY PRODUCTIVITY

The photosynthesis depth profiles differed between July and October: the photosynthesis per volume (P_z) for July was higher than that of October. However, the photosynthesis per unit Chl-*a* ($P^B{}_z$) for October was higher than that of July. In July the highest productivity was at the surface with a steep decline towards 0.125 m, followed by an almost vertical decline down to 0.75 m and a further steep decline towards 1.5 m. In October the productivity increased (54.86 % photoinhibition) - from the surface to 0.125 m, followed by a vertical decrease to 0.5 m and a further steep decrease down to 1.5 m (**Table 5.9** and **Figure 5.13**).

The same species present at the Wall Surface samples in **Table 5.8** were present with the primary production. Other related primary productivity variables are shown in **Table 5.10**. The C^{14}/ml for July was more than twice that of October, and all added up to a thirty two times higher photosynthetic rate per day for July over October.

Table 5.9: Primary Productivity for 13 July and 9 October 2000.

Depth (m)	**Pz (mgC/m^3/h)**		**$P^B{}_z$ (mgC/mgChl-*a*/h)**	
	Jul	**Oct**	**Jul**	**Oct**
0	113	17	1.85	1.62
0.125	83	38	1.36	3.6
0.25	79	36	1.3	3.43
0.5	78	37	1.28	3.48
0.75	72	25	1.18	2.41
1.0	44	17	0.72	1.64
1.5	12	4	0.2	0.35



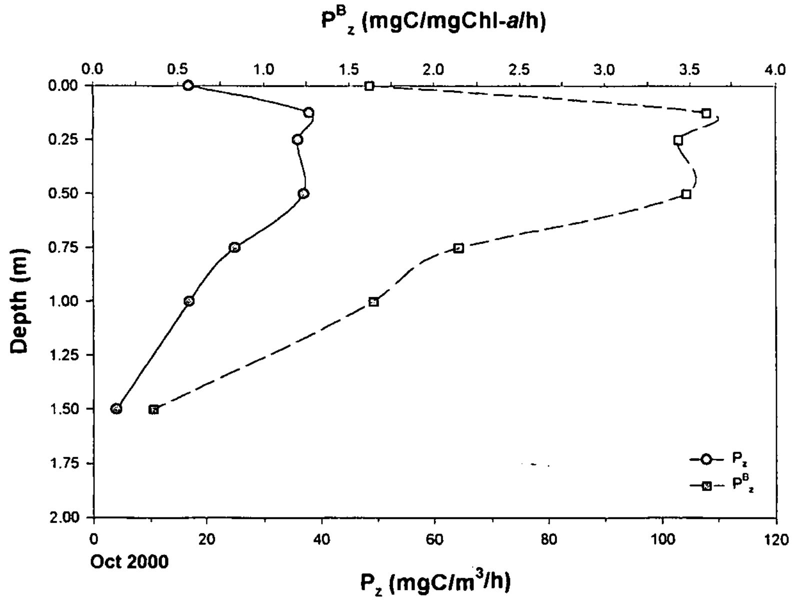


Figure 5.13: The photosynthesis depth profiles (P^B_z & P$_z$) in Loch Logan for July and October 2000.

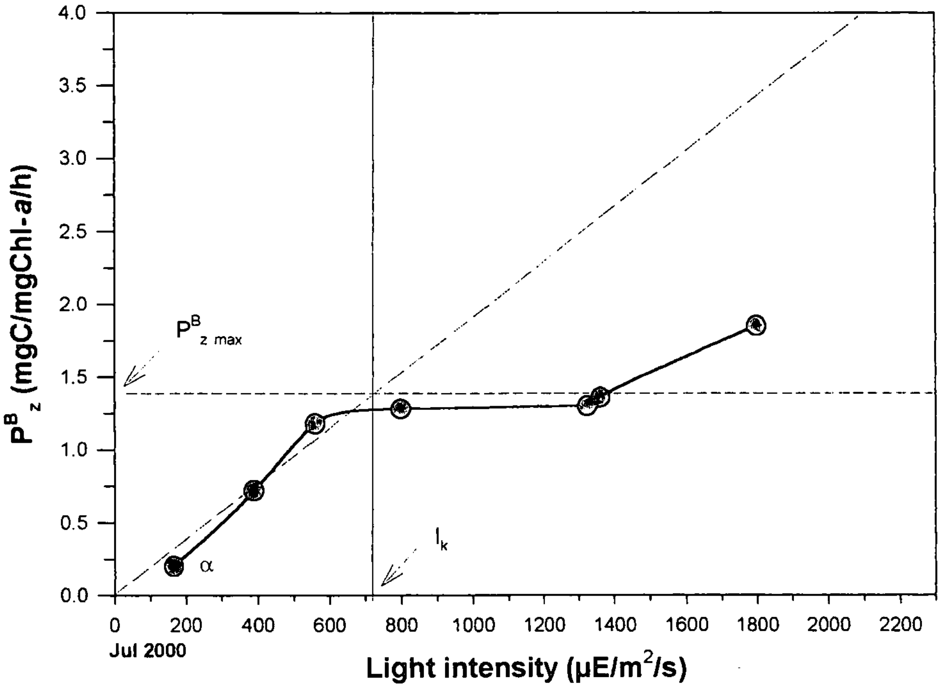
Table 5.10: Other Primary Productivity variables.

Variable	**Date**	
	13/07/2000	**09/10/2000**
Chlorophyll-*a* (µg/l)	60.9	10.5
Dissolved inorganic carbon (DIC) (mg/l)	25.6	17.4
C^{14}/ml	31,028.7	14,402.4
Euphotic zone depth (Z_{eu}) (m)	1.4	1.4
Light saturated rate of photosynthesis (P_z) (mgC/m^3/h)	112.5	37.79
Surface photo-inhibition (%)	0.0	54.86
Surface photo-inhibition (PI) (%/Em^2)	0.0	6.93
Assimilation number = Photosynthesis capacity ($P^B{}_z$) (mgC/mgChl-*a*/h)	1.85	3.6
Daily rate of areal photosynthesis (ΣPd) (mgC/m^2/d)	86,467.6	2,693.7
The initial slope of the light-saturation curve (α^B) (mgC/mgChl-*a*/Em^2)	0.723	1.099

On the photosynthesis/light intensity (P/I) curve graph the July data were as follows: I_k (onset on light saturation) = 710 µE/m^2/s, $P^B{}_{z\ max}$ = 1.42 mgC/mgChl-*a*/h and α^B = 0.723 (37.5°). The October data were: I_k = 910 µE/m^2/s, $P^B{}_{z\ max}$ = 3.68 mgC/mgChl-*a*/h and α^B = 1.099 (58.5°) (**Figure 5.14** and **Table 5.10**).

The light intensity profile for July ranged from 1,800 (0.0 m) to 165 µE/m^2/s (1.5 m) and that of October from 2,200 (0.0 m) to 180 µE/m^2/s (1.5 m) (**Figure 5.15**).

P_z measured the photosynthesis rates in terms of mgCarbon/m^3/hour whereas $P^B{}_z$ measured it as mgCarbon/mgChl-*a*/hour. In July the Chl-*a* concentration (60.9 µg/l) was much higher than that in October (10.5 µg/l) although the algal composition was very much the same. Therefore P_z was much higher in July that in October when carbon incorporation was measured per volume. However, the $P^B{}_z$, which was measured by the amount of carbon that was incorporated by the number of chlorophyll-*a* present, was higher in October than in July because of lower Chl-*a* concentrations. The photosynthesis rate at 1.5 m, just below the euphotic depth (Z_{eu} = 1.4 m), was nearly the same. The light-saturation curve (α^B) for October was higher (1.5 times) than that of July (**Table 5.10**), which confirms that the algae had a higher productivity efficiency per Chl-*a* during October.



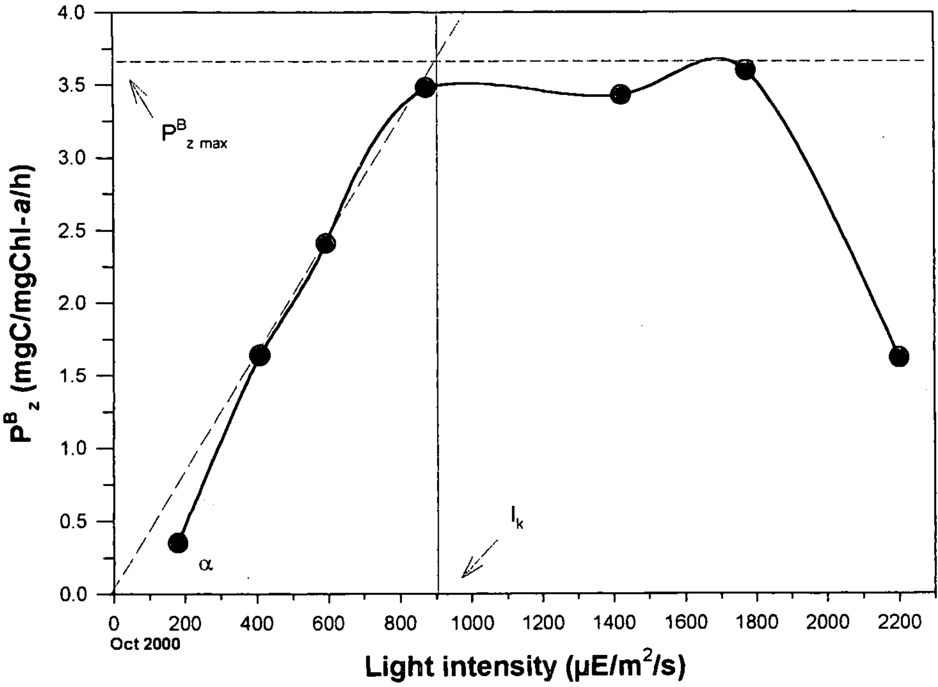


Figure 5.14: The P/I curves in Loch Logan for July and October 2000.

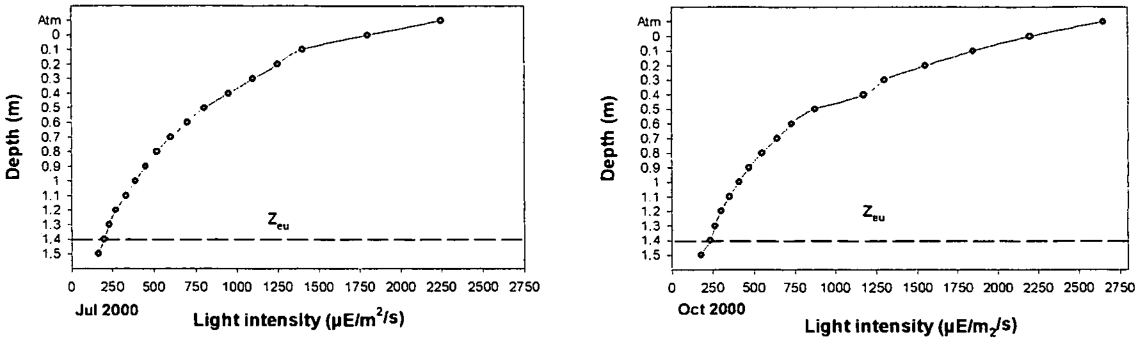


Figure 5.15: The primary productivity light intensity depth profiles in Loch Logan for July and October 2000.

The photosynthesis *versus* light intensity curves showed the differences between the P_Z and the P^B_Z of July and October. Photo-inhibition (54.86 %) occurred in October when the light intensity was 2,200 µE/m^2/s at the surface. The high percentage photo-inhibition encountered could be overestimated because of the light and dark bottle technique - normal water movement might move cells rapidly into and out of the euphotic zone, thereby minimising the magnitude of photo-inhibition (Grobbelaar, 1985). In July there was no photoinhibition. The light intensity was lower (about 250 E/m^2/s) and the higher Chl-*a* concentration could have resulted in an over shade by algae.

The low temperature in July inhibited photosynthesis and primary production of the algae (Davison, 1991) (see **4.2.2**), and therefore indirectly the production of oxygen. Roos and Pieterse (1992) conducted a primary productivity study in an impounded part in the Vaal River, near Venterskroon during late September. With a Chl-*a* of approximately 45 µg/l and moderate temperature, the P_Z reached a maximum of about 280 mgC/m^3/h (Roos & Pieterse, 1992).

The primary productivity that was measured in the Gariep Dam (February 1999) was at 14.6 mgC/mgChl-*a*/h (Venter, 2000) more than 10 times higher than that of Loch Logan (3.68 mgC/mgChl-*a*/h) in October 2000. That of the Vaal River (April 1999) was approximately 20 times higher (76 mgC/mgChl-*a*/h; Venter, 2000) than that of Loch Logan. The photosynthesis depth profiles of Gariep Dam and Loch Logan looked similar with the P^B_{zmax} at 0.125 m, where the P^B_{zmax} of the Vaal River was at 0.5 m (Venter, 2000).

Except for the higher light intensity at the surface in October, the rest of the profiles for July and October were nearly the same (**Figure 5.15**) with the Z_{eu} at 1.4 m. Light intensity decreased exponentially with depth. The loss of light is expressed mathematically by the extinction coefficient (ε_λ) of the solution, the fraction of light absorbed per meter of water. The formula is as follows:

$I_z = I_0 e^{-\varepsilon\lambda z}$

where I_0 = intensity penetrating the surface, z = path length, and ε_λ = extinction coefficient for the wavelength in question (Horne & Goldman, 1994). The extinction coefficients for Loch Logan were as follows: July - 0.1 m = 2.513, 0.75 m = 1.557, 1.5 m = 1.593, and October - 0.1 m = 1.733, 0.75 m = 1.744, and 1.5 m = 1.669. Near the surface July's ε_λ was higher than October's, but deeper in the water-column October's was higher.

The aphotic zone to euphotic zone ratio of turbid waters can be very high and if the mixing depth exceeds the euphotic depth, phytoplankton cells may spend considerable periods of time in the dark (non-photosynthetic conditions) (Grobbelaar, 1985) and losses due to respiration become more pronounced (Grobbelaar, 1992). Loch Logan's average euphotic:aphotic zone ratio for was 0.67:1 (1.4 m:2.1 m). The smaller the ration, the lower the productivity (Grobbelaar, 1989).

5.4 CONCLUSIONS

Not all the physico-chemical characteristics of Loch Logan showed distinctive daily trends. The high nitrogen and phosphorus concentrations, together with the Chl-*a* concentrations, clearly showed Loch Logan to be eutrophic. The only parameters that did show a diel pattern were temperature, oxygen and pH. However, for clearer results regarding the diel patterns in Loch Logan measurements should be taken more often, i.e. hourly.

It is clear that temperature had a major controlling factor on productivity during the winter, and thereby preventing major algal blooms from occurring in the winter.

CHAPTER 6

LAKE RESTORATION AND MANAGEMENT OPTIONS

6.1 INTRODUCTION

The availability of clean, freshwater and not the availability of fossil fuels (which can be supplemented or replaced with alternative power sources), is the basis of the existence of human societies and economics. Freshwater is essential for many forms of life, and is required by humans for drinking, agriculture and most industrial processes. Its role in recreational activities is well known (Cooke *et al.,* 1993b).

During the last few decades, the word “eutrophication” had been used more and more to denote the artificial and undesirable addition of plant nutrients, mainly phosphorus and nitrogen, to waterbodies. In some situations, this view can be misleading, because what is an undesirable addition to one waterbody may be harmless, or even beneficial, to another waterbody. Nevertheless, nutrient status is most commonly known as the trophic state (oligo-, meso-, eu- or hypertrophic) of a waterbody, which is manifested by an intense proliferation of algae and higher aquatic plants, and their accumulation in the waterbody in excessive quantities. These accumulations can result in detrimental changes in water quality and in the biological populations of a waterbody, which can interfere significantly with human use of a water resource (Ryding & Rast, 1989).

Eutrophication, in the original sense, represented the natural ageing process of a lake (Ryding & Rast, 1989), but man pollutes water in several ways (introduction of C, N, P and micronutrients) that can accelerate (cultural) eutrophication (Marzadori *et al.,* 1998). Organic and inorganic chemicals may be discharged from point sources (e.g. sewers of domestic and industrial origin) or non-point sources (e.g. pesticides and fertiliser runoff from agricultural land). Precipitation transfers pollution from the atmosphere to water bodies (De Haas *et al.,* 1989).

The importance of eutrophication lays in its effects, namely impaired aesthetics of water treatment costs, taste and odour problems and potential health risks. It has been determined that - as in other countries - phosphorus is the primary limiting nutrient which control algal growth in the majority of South African reservoirs. In a few cases, however, nitrogen was found to be limiting (Du Plessis & Van Veelen, 1991).

Loch Logan shows typical eutrophication characteristics with its high nutrients, low oxygen concentrations (in summer), and algal blooms. In Loch Logan the limiting nutrient through most of the year is nitrogen, however, but phosphorus tend to be limiting for short periods.

Eutrophic lakes show resistance to recovery, which means that the system is relatively insensitive to changes and nutrient input (Janse *et al.,* 1995), but while there is a solid scientific understanding and basis for managing the eutrophication of freshwater, improvements are still necessary (Rippey *et al.,* 1997).

The first and most obvious step towards the protection and restoration of a lake or reservoir is to limit, divert or treat excessive nutrients, organic, and silt loads. Unlike rivers, which should respond quickly to pollution abatement unless toxic materials have been added to their sediment, lakes and reservoirs trap and recycle nutrients, silt, toxins and organic matter. Thus, the elimination or curtailment of loading - an expensive and necessary step - may be insufficient to produce immediate and long lasting improvements due to the presence of extensive shallow areas or to the internal recycling of toxins and nutrients. Both of these lake problems allow the continued production of algae and macrophytes (Cooke *et al.,* 1993b).

Restoration is any active attempt to return an ecosystem to an earlier condition following degradation resulting from any kind of disturbance. Restoration involves repair of ecological damage, a return of species and processes to their former states, and is holistic in its approach to returning the lake and its watershed to an approximation of predisturbance condition (Cooke *et al.,* 1993b).

Management involves and attempts to remedy, improve or change conditions, usually of some specific lake component, often with human uses in mind. Management is not necessarily involved in returning the lake or reservoir to some earlier, predisturbance condition but instead is directed towards amelioration of one or more specific problems (Cooke *et al.,* 1993b). A protocol for the management of urban impoundments is shown in **Figure 6.1**.

The history of this environmental technology (restoration and management) is comparatively short, although there has been a long period of awareness of pollution. In 1947 Hasler was among the first to recognise that restoration of eutrophic lakes would be difficult. He stated: "The problem is especially serious because there is no way at present for reversing the process of eutrophy. Unless some method can be discovered for oxidising a large proportion of the addition deposits created by fertilizers, the lake's service to the man is limited to a relatively short period" (Cooke *et al.,* 1993b).

Through the years many management and restoration options have been developed. They can be classified into two groups, namely catchment and in-lake options. Catchment options consist of management of urban or agricultural runoff, biological treatment/wetlands and chemical treatment, nutrient diversion and advanced treatment of wastewater*, dilution and flushing*, education and legislation (Ryding & Rast, 1989; Harding & Quick, 1992). The in-lake options are as follows: sediment removal*, sediment treatment through oxidation* and phosphorus inactivation*, biomanipulation, algicides, barley straw bales, hypolimnetic aeration* and withdrawal* as well as artificial circulation* (Harding & Quick, 1992; Cooke *et al.,* 1993b; Eiseltová, 1994).

*Options discussed in the chapter, reduction in phosphorus can be regarded as a method to prevent algal blooms, whereas the other methods are treating the blooms.

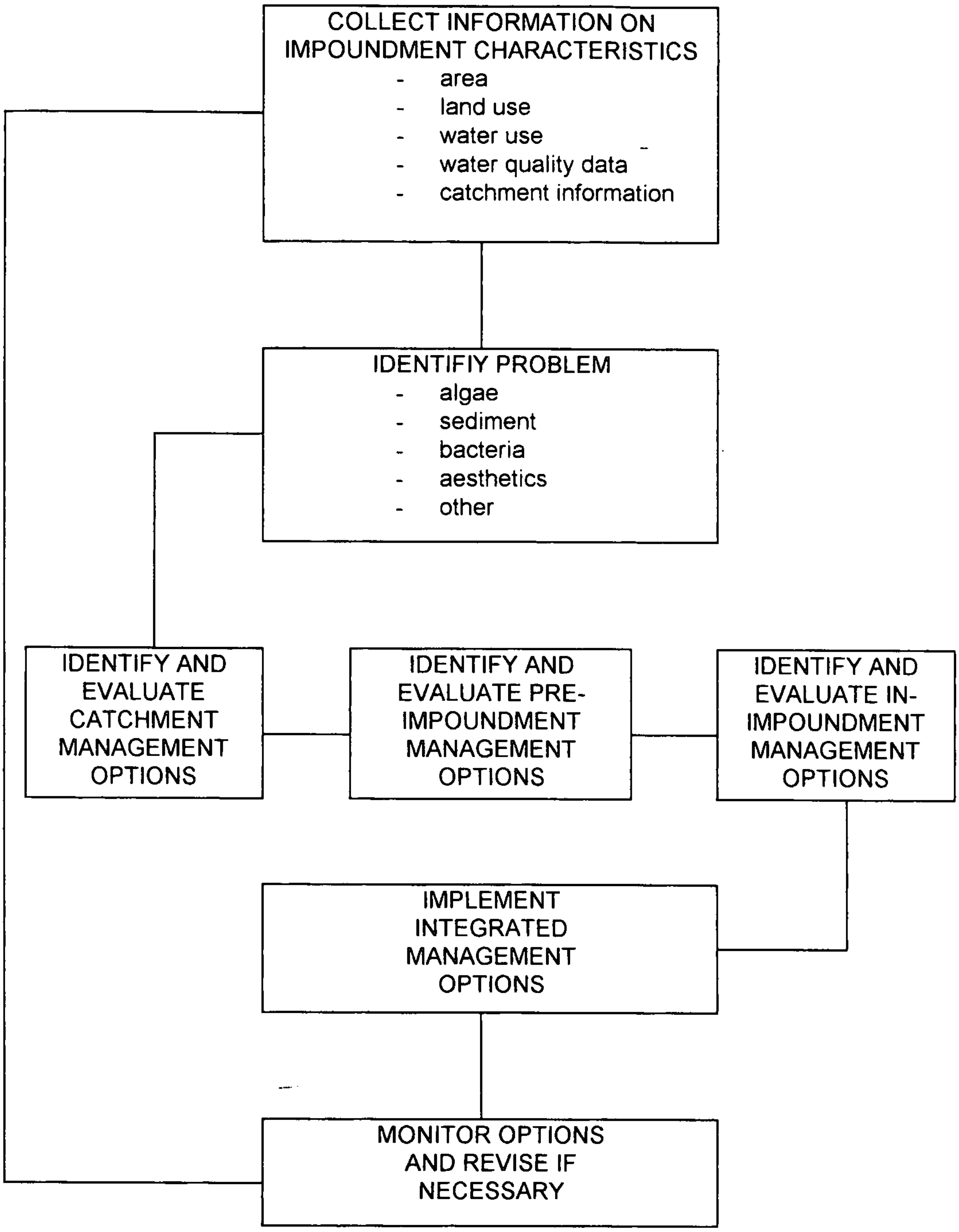


Figure 6.1: The management of urban impoundments (Wiechers *et al.*, 1996).

6.2 METHODS, CASE STUDIES AND COSTS

6.2.1 DIVERSION AND ADVANCED TREATMENT (P REMOVAL) OF WASTEWATER

The first step in restoring or improving the quality of eutrophic or hypereutrophic lakes/reservoirs is to remove or treat direct inputs of wastewater, stormwater or both. Such sources usually contain relatively high concentrations of P and N. Unless such external inputs (loading) are reduced, any long-term benefits from in-lake treatments to reduce nutrients will usually not realise (Cooke *et al.,* 1993b).

Diversion and advanced wastewater treatment (AWT) are the two techniques most commonly used to reduce external loading. Diversion of treated sewage or industrial wastewater involves the construction of interceptor lines to convey the wastewater away from the degraded water body to waters that have greater assimilative capacity (e.g. where light limits; Cooke *et al.*, 1993b).

AWT (tertiary treatment) reduces the P concentration in wastewater effluents that continue to enter the lake. Alum (aluminium sulphate), lime (calcium carbonate) or iron (ferric chloride) removes P. Efficiencies for P removal with AWT can range from 90-99 %. Treatment of river inflows has resulted in residual concentrations of only a few µg/l (Cooke *et al.,* 1993b).

6.2.1.1 Case studies

6.2.1.1.1 Lake Washington, USA

The diversion of secondary treated domestic wastewater from Lake Washington from 1964 to 1967 by the municipality of Metropolitan Seattle, and its subsequent fast recovery, is undoubtedly the most celebrated case in the world. The reason for this is its rapid and complete recovery at a time when there existed considerable doubt about the prospects for restoring lakes once they had become eutrophic (Edmondson & Lehman, 1981).

TP declined from a mean annual 64 µg/l prior to diversion to equilibrium concentration of about 21 µg/l by 1972, 5 years after diversion was completed.

Chl-*a* decreased from its prediversion summer mean of 36 µg/l in direct proportion to the decrease in TP. The Chl-*a* reached a level of 7 µg/l by 1969 and by 1975 a 7-year mean of 6 µg/l. The Secchi depth increased from 1 m to 3.1 m during this time.

The lakes statistics in the late 1970's was as follows: TP - about 17 µg/l, Chl-*a* - about 3 µg/l, Secchi depth - nearly 7 m. The reason for this fast recovery was due to the depth of the lake (64 m max; 37 m mean), fast renewal rate (0.4/yr), its oxic hypolimnion and its relatively short history of enrichment (Cooke *et al.,* 1993b).

6.2.1.1.2 Lake Zürich, Switzerland

Over a 10-year period beginning in about 1965, P removal was installed in all treatment plants within the immediate drainage to Lake Zürich. Conventional wastewater treatment plant construction was initiated in 1955. About 54 % of the P load entering the lake prior to 1955 has been removed by AWT. The inflow concentration of TP decreased from 80 µg/l in 1976 to 47 µg/l in 1984. The lake has responded positively as its TP gradually declined over a 13-year period (1974 to 1986) from 93 to 54 µg/l (Sas *et al.*, 1989).

A substantial decrease in hypolimnetic oxygen deficit also occurred, with the greatest change observed in the 10 years following conventional wastewater treatment in 1955 (Shantz, 1977). Oxygen deficit averaged 27 % less during 1971 to 1976 than during 1953 to 1959.

The Secchi depth of Lake Zürich has increased about 50 % as an annual mean between 1966 and 1975, compared with 1953 to 1965 (Shantz, 1977). Secchi depth rarely exceeded 6 m in the years before treatment, but since the installation of AWT it commonly reaches 10 m. The greatest improvement in Secchi depth has occurred during winter and autumn. The lake is now clearer during those times than it was before the turn of the century (Sas *et al.*, 1989; Cooke *et al.,* 1993b).

6.2.1.2 Costs

The costs involved for diversion of effluents vary greatly from site to site in relation to the transport distance from the lake to the receiving water body or treatment facility. The costs for AWT per capita would tend to be more equal from site to site, because costs should depend on volume of wastewater, chemicals for treatment, and sludge disposal. There should not be new distance transport costs unless the wastewater was not previously treated and a whole new plant and sewer system is required (**Table 6.1**; Cooke *et al.*, 1993b).

Table 6.1: Estimated costs for diversion and AWT to restore five lakes (values in parenthesis adjusted to 1990 US dollars) (Cooke *et al.*, 1993b).

Lake	Treatment	Year	Construction			Operation (per year)	
			$ x 10^6	$ x 10^3/ha	$ per capita	$ x 10^3	$ per capita
Washington	Diversion	1967	94.9 (366)	41.6	171 (660)	2138	4
Sammamish	Diversion	1968	4.5 (16.6)	8.3	370 (1362)	146	12
Norrviken	Diversion	1969	44.5 (148)	550.0	106 (352)	6736	16
Shagawa	AWT	1973	1.9 (5.5)	6.0	380 (1102)	389	77
Zürich	AWT	1975	36.0 (86)	13.2	252 (605)	1500	115

6.2.1.3 Possible application in Loch Logan

Diversion of stormwater from Loch Logan would probably be the most effective and permanent solution to its eutrophication problem. That is, building an extension of the canal around Loch Logan so that it rejoins the canal below the dam wall to divert the inflow *via* the canal. This way the nutrient enriched stormwater will not enter Loch Logan. A borehole next to Loch Logan can be used to replace water loss due to evaporation.

Advanced treatment of the water is not practical at all. There is no building space for a plant because the upstream canal is almost totally underground, situated beneath buildings. However, the costs of such a project would probably be several millions of rands. Another problem is that the water flows in large pulses during the rainy season.

6.2.2 DILUTION AND FLUSHING

It has been said: "the solution for pollution is dilution". The technique of diluting and flushing can achieve improved quality in eutrophic lakes in two ways: (1) the concentration of limiting nutrients can be reduced (dilution); and (2) the water exchange rate is increased (flushing). Both processes can reduce the inflow concentration of the limiting nutrient, resulting in a decreased lake concentration, which determines the maximum biomass. By increasing the water input, the flushing rate is increased which, in turn, increases the loss rate of algae from the lake. Dilution can be effective even when the increase in flushing rate is insufficient to cause a significant loss of algae. On the other hand, increases in flushing rate can affect a significant loss without achieving a reduction in the limiting nutrient concentration. Other effects of dilution are also possible, such as increased vertical mixing and a decrease in the concentration of algal excretory products. The latter may influence the kinds and abundance of algae (Cooke *et al.,* 1993b).

Dilution is usually feasible only where large quantities of low-nutrient water are available for transport of the affected lake. Treatment effectiveness is greater the lower the concentration of the limiting nutrient in the dilution water relative to that in the lake and its natural flow. In some instances, improvements can be achieved by adding water with a moderate to high nutrient content but these results are less certain than with low-nutrient water (Cooke *et al.,* 1993b).

6.2.2.1 Case studies

6.2.2.1.1 Green Lake, USA

Green Lake (Seattle metropolitan area) represents a typical case where dilution was a practical restoration alternative.

Dilution was proposed as the primary treatment for Green Lake in 1960 (Sylvester & Anderson, 1964) and was instituted in 1962. The dilution of Green Lake represents a process with a relatively lower rate of flushing.

A marked improvement in Chl-*a*, TP and Secchi depth was noted during the first few years after dilution was initiated. Only one predilution year's data existed for comparison with three years of postdilution monitoring, which began in 1965. Water visibility during the summer increased nearly four fold to an average depth of 4 m (because the mean depth is 3.8 m, most of the lake bottom was visible), and Chl-*a* decreased more than 90 % from 45 to 3 µg/l. The summer average for TP deceased from 65 to 20 µg/l. A substantial decrease in the fraction of cyanobacteria was observed, particularly during spring and early summer.

Chl-*a* and TP averaged 35 and 55 µg/l, respectively, during the summer of 1981; the lake quality had degraded markedly during the late 1970's, primary as a result of declining dilution water inputs. No water was added in 1982. Dilution water was subsequently added in modest amounts on a regular basis to avoid deterioration in the lake quality.

Resent work on Green Lake has shown that many of its problems are due to internal loading of P, a fact not taken in account in the earlier work. Internal load is high in spite of nearly all of its area being unstratified and apparently aerobic during summer (Perkins, 1983). Internal loading accounted for 21 % of the total annual loading of P during 1981. During the three summer months, however, when Chl-*a* averaged 38 µg/l (with a maximum of 60 µg/l) the internal source accounted for 88 % of the total. The average areal gross release rate was determined to be 40 mg/m^2/d.

Dilution remained the one principal technique of choice to improve and maintain Green Lake's quality with the goal being a mean summer TP concentration of 28 µg/l (Cooke *et al.*, 1993b).

6.2.2.2 Costs

For Green Lake water from Lake Washington would have cost \$10x10^6 (15x10^6 m^3/yr) for 20 years and for city water \$17.7x10^6 (10.4x10^6 m^3/yr) for 20 years (1990 dollars).

If the lake in question is situated in an urban setting and domestic water is available, then improvement may be possible for less than \$100,000 for construction, water costs, and the first year of maintenance and operation. If the lake is near a free-flowing river and a diversion of a portion of the river flow through the lake in summer is feasible, then the costs would involved facilities, pumps and pipes, operation and the prevention of side effects (Cooke *et al.*, 1993b).

6.2.2.3 Possible application in Loch Logan

There are two boreholes next to Loch Logan. This low-nutrient water can be used to dilute the water in Loch Logan. The dilution would probably reduce phosphorus and nitrogen and lower the Chl-*a*. However, P releases from the sediment could be substantial because of the sudden difference between concentrations in the water-column and the sediment, as found by Pekkarinen (1990) in some studies.

The costs would be relatively low if the water from the boreholes is used. If municipal water is used, the costs would still be relatively low.

6.2.3 HYPOLIMNETIC WITHDRAWAL

Employing hypolimnetic withdrawal requires that the depth at which water leaves the lake be changed from the surface to near the maximum depth so that nutrient-rich water is discharged from the lake. Coincidentally, the hypolimnion detention time is shortened, the chance for anaerobic conditions to develop is decreased and the availability of nutrients to the epilimnion, through entrainment, is reduced. This technique is applicable to stratified lakes and small reservoirs to which anaerobic hypolimnia is restricted to the habitat for fish and promote the release of P, toxic metals, ammonia, and hydrogen sulphide from sediment (Cooke *et al.,* 1993b).

6.2.3.1 Case studies

6.2.3.1.1 Austrian Lakes

Special extracting tubes (Olszeuski tubes) were installed in Piburger See, Reither See and Hechtsee. All three lakes are relatively small. In Piburger See water was drawn from a depth of 23 m (lake's max depth is 24.6 m). Although the oxygen level increased, there was no recognisable oligotrophication (1970). After 7 years there was a slightly decline in epilimnetic TP (5 µg/l) (Pechlaner, 1979).

In contrast to Piburger See, Reither See showed a marked improvement in quality following installation of a tube at near maximum depth (5.2 m) in 1972. TP decreased from an annual mean of 38 µg/l in 1974 to 21 µg/l in 1977. The water's Secchi depth nearly doubled over the 4-year period following installation.

A tube was placed (25 m deep) in Hechtsee in 1973. The depth of the placement was not near the maximum depth (56.5 m), because of meromixis and odours from the monimolimnion (dense, lower stratified layer) were quite strong. A tube was installed above the monimolimnion. During the first four years following installation, P output exceeded input, which was the actual decrease in lake TP content. TP above 25 m declined by 70-80 % from 1973 to 1977 while, as expected, TP below 25 m changed little and actually showed some increase (Cooke *et al.,* 1993b).

6.2.3.1.2 USA Lakes

Withdrawal systems were installed in the shallower of two basins of Lake Wonoscpomuc in 1980 at 15 m. The lake quality had improved substantially following initiation of withdrawal. Hypolimnetic TP decreased from 400 µg/l to less than 100-50 µg/l over 5 years following the start of withdrawal. Epilimnetic TP decreased from 24-30 µg/l to 10-14 µg/l over the same period (Nurnberg, 1987).

Two systems were installed in Lake Waramaug in 1983 at 8.3 m. No significant changes in TP occurred during the first 3 years.

In Lake Ballinger the device was installed in 1982. The hypolimnion became oxygenated by 1985 and TP decreased from 450-900 µ/l to 100-150 µg/l (Cooke *et al.*, 1993b).

6.2.3.2 Costs

Installation costs (1990 dollars) for the US lakes systems were as follows: Lake Ballinger (41 ha, 3.4 m^3/min flow) - $304,000, and Lake Waramaug (278 ha, 6.3m^3/min) - $45,000 (Cooke *et al.*, 1993b).

6.2.3.3 Adverse Effects

Discharge of hypolimnetic water that may contain high concentrations of P, ammonium, hydrogen sulphide, and no oxygen may cause a water quality problem downstream (Cooke *et al.*, 1993b).

6.2.3.4 Possible application in Loch Logan

Hypolimnetic withdrawal can be applied in the spring and summer when the oxygen is low and the NH_4-N is high in the bottom water. This will reduce internal loading of nutrients. The sluice gate can also be opened during floods to flush out polluted sediments. The costs would be relatively low.

6.2.4 PHOSPHORUS INACTIVATION

The use of chemical precipitation to remove phosphorus from water has a long history. The first chemical precipitation treatment of lakes was performed in Sweden during the late 1960's and early 1970's. The first lake treatment in the USA was Horseshoe Lake in Wisconsin in 1970. Aluminium, iron and calcium salts have been used for centuries to treat drinking water. These same salts are the primary chemicals to precipitate phosphorus (Charboneau, 1999).

Nuisance algal blooms can be reduced or eliminated if nutrient concentrations are lowered to growth-limiting levels by diversion or pre-treatment of loading, by dilution, or a combination of these methods (Cooke *et al.,* 1993b).

P inactivation is a lake restoration/improvement technique. Its purpose is to lower the lake's P content by removing P from the water-column (P precipitation) and by retarding P release from lake sediment (P inactivation). Aluminium salt, aluminium sulphate (alum), sodium aluminate, or iron and calcium salts can be added (Cooke *et al.,* 1993b).

The pH of the water determines which aluminium hydrolysis products will dominate, and their solubilities. At pH 6-8, insoluble polymeric $Al(OH)_3$ dominates, whereas at pH 4-6 various soluble intermediate forms occur and at a pH less than 4 hydrated and soluble Al^{3+} dominates. When alum ($Al(SO_4)_3 \cdot 14\ H_2O$) is added to soft waters, acid neutralising capacity (ANC) is quickly exhausted and soluble aluminium species dominate. At higher pH levels (>8.0), the amphoteric nature of aluminium hydroxide forms the aluminate ion. High pH will also lead to the liberation of P sorbed to the floc. In lakes with low or moderate ANC (0.6-1.0 meq/l), alum addition produces a decline in pH at a low or moderate alum dose, leading to the appearance of and increase in the concentration of toxic, soluble aluminium forms, including $Al(OH)_2^+$ and Al^{3+}. This limits the amount of aluminium sulphate, which can be added to soft water lakes. This problem has been addressed by adding buffers, usually sodium aluminate, to the lake or to the slurry as it is applied. Though more costly than aluminium sulphate, sodium aluminate ($Na_2Al_2O_4 \cdot N\ H_2O$), a high alkalinity compound, has the additional benefit of having aluminium content. This approach will maintain a pH in the 6-8 ranges and will provide excellent P sorption (Cooke *et al.,* 1993a).

The properties of $Al(OH)_3$ important to P inactivation treatments are its low or zero toxicity to lake biota under normal treatment conditions (pH>6.0), its ability to adsorb large amounts of inorganic P, and the essentially irreversible binding of P to the flux (except at a high pH). Low or zero dissolved oxygen conditions do not solubilise the floc and allow P release. $Al(OH)_3$ is not effective in removing or sorbing dissolved organic P compounds (Cooke *et al.,* 1993a).

Phosphorus also forms precipitates and complexes with iron and calcium. Inorganic iron exists in the oxidised ferric (Fe^{3+}) or the reduced ferrous (Fe^{2+}) forms, depending upon pH and oxidation-reduction potential. $Fe(OH)_3$ strongly sorbs P and forms part of an oxidised ‘microzone’ over the sediment surface, providing high primary P removal and retention in sediments is by solid $FeOOH$-PO_4 complexes, and is maximal at pH 5-7. In reduced state (Fe^{2+}), iron becomes soluble and P is released; a change that occurs rapidly so that even brief periods of thermal stratification and dissolved oxygen consumption lead to P release. It appears to occur on a diurnal basis in the littoral areas of some lakes. P is also released from iron complexes during periods of high pH levels (Cooke *et al.,* 1993a).

Calcium carbonate (calcite) and calcium hydroxide can be added to a lake or it can be produced during periods of photosynthesis. Calcite sorbs P, especially when pH exceeds 9.0, and result in significant P removal. At high levels of pH, Ca^{3+} and P, hydroxyapatite forms. Unlike $Fe(OH)_3$, it has its lowest solubility at pH>9.5, and P sorbs to it. The solubility of calcite and hydroxyapatite increases as the CO_2 concentration increases and pH falls, as would occur in a hypolimnion or dark littoral zone with intense respiration, leading to P release. As with iron, effective P removal and inactivation are possible with calcium but conditions for continued P sorption can be lost without an additional management step (Cooke *et al.*, 1993a).

6.2.4.1 Case studies

6.2.4.1.1 Shallow lakes

The release of P from sediment is significant in shallow lakes because P concentration in the photic zone is directly affected and ideal. Microbial activities are enhanced by the higher temperatures of sediments in these lakes and may lead to sediment anoxia with iron reduction and P release. Some of these shallow lakes - like some deeper, dimictic lakes - have groundwater inflows in shallower areas and these flows force additional releases of P from the sediment.

Initially, P inactivation in shallow, continuously mixed lakes was considered to be ineffective because the first three attempts failed. However, Lake Långsjön, Sweden and Lake Lyngby Sø, Denmark failed because external loading was believed to have redistributed the floc to the lake's centre (Cooke *et al.,* 1993b).

P inactivation should be effective in mixed lakes because the treatment affects the entire water-column. Of the six shallow lakes in Washington State, USA, that received alum treatment since the three failures, five were successful, as predicted, and one was a failure.

In Wapato Lake, areal coverage of the macrophyte *Ceratophyllum* increased dramatically, pH increased to 10.1 and TP doubled in concentration between years. This failure may be due to the increased pH, leading to P release via ligand exchange and to the contribution of P to the water-column by *Ceratophyllum*.

Long Lake, a shallow (z_{max} = 3.7 m; z = 2.0 m) eutrophic soft water lake dominated by *Elodea densa* and high internal P loading, was treated with alum in 1980. In 1985, TP abruptly returned to pre-treatment levels and then declined again in 1986 to 1987. By 1990, TP had slowly returned to pre-treatment levels and a second application was carried out in October 1991, using Na_2CO_3 buffered alum. The increase in TP in

1985 was associated with a decline in abundance of macrophytes. This allowed wind mixing to stir sediment bound or $Al(OH)_3$ bound P into the water-column. In 1986, *E. densa* biomass returned to removal levels and P concentrations again fell. Thus the alum treatment of Long Lake was very effective in controlling internal P release for 4 years and partially effective for 9 years. Associated with lower TP in Long Lake was lowered algal abundance and increased Secchi depth. This long-term evaluation has demonstrated the effectiveness of P inactivation in shallow lakes and strongly supports the conclusion that the process should become an important management tool for such lakes (Cooke *et al.*, 1993b).

6.2.4.2 Costs

Dramatic reductions in P inactivation costs have occurred since Horseshoe Lake (USA) was treated in 1970.

The costs to treat Three Mile Pond[a] (266 ha treated with 20.0 gAl/m^3) in 1988 with the high-speed system were $640/ha. In comparison the costs to treat Cochnewagon Lake[a] (1986, 133 ha treated with 18.0 gAl/m^3) with the modified harvester were $844/ha and the costs to treat Medical Lake[b] (1977, 227 ha treated with 12.2 gAl/m^3) with the old large system were $1101/ha. In most cases, including all of those with the original large design, costs per hectare were twice that of the Three Mile Pond treatment (Cooke *et al.*, 1993b).

[a] Aluminium sulphate and Sodium aluminate.

[b] Aluminium sulphate.

6.2.4.3 Possible application in Loch Logan

Phosphorus inactivation was tested in Loch Logan during May 2000. Two hundred litre Sudfloc (commercial flocculent) was added from a boat and mixed at the back by the propeller. When the Sudfloc is added to a lake with a neutral pH of 6 to 8, a milky solution is produced that quickly forms an insoluble colloidal aluminium hydroxide floc that binds inorganic phosphorus. The floc increases in size and precipitates out of the water-column resulting in more visibly clear water within hours. The floc rests on top of the lake's sediment and continues to sorb and retain phosphorus, even under reducing conditions. This floc blanket over the sediment prevents phosphorus from being released and added to the internal lake load. The precipitation process also removes organic and inorganic particulates that clear the water and reduces available nutrients to algae (Charboneau, 1999).

The phosphate concentration decreased for a month or two in the surface as well as in the bottom water after the application. After the first spring rains had fallen, the phosphate concentration increased again in both the surface and bottom water. The cost of the experimental treatment was about R2,000. Because of the very short residence time in Loch Logan during the summer, the chemicals flush out and re-applications could be as frequent as every 2 to 3 weeks. These multiple applications will result in high costs. The pH should also be monitored because it had reached levels as low as ± 6 in the past.

The applications of aluminium compounds to shallow polymictic lakes have negative and positive effects. The positive effect is that it mixes through the whole lake to bind with P, the negative effect is that because of Loch Logan's size, it has a short retention time and during a rainstorm new phosphates flush in.

6.2.5 SEDIMENT OXIDATION

A restoration technique to oxidise the top 15 to 20 cm of anaerobic lake sediment was developed by Ripl in 1976. The objective is to reduce internal P loading in lakes which have anaerobic sediment and high interstitial P concentrations, and in which iron redox reactions control P interchange between sediment and overlaying water. By oxidising the organic matter through increased denitrification, greater binding of interstitial P with ferric hydroxide complexes should occur, resulting in lower release rates of P. Also, sulphate reduction is prevented, thus decreasing the complexation of iron with sulphur.

To deplete the organic matter in sediments, and thereby restoring an oxidised state, a solution of $Ca(NO_3)_2$ is injected into the sediment to stimulate denitrification. Nitrate is preferred as the electron acceptor because, being a liquid solution, it penetrates into the sediment more rapidly and to a greater degree than added oxygen. Ferric chloride ($FeCl_3$) is added initially to remove hydrogen sulphate (H_2S) and form ferric hydrogen ($Fe(OH)_3$), which binds interstitial P. Lime ($Ca(OH)_2$) is added next to raise the pH to optimum levels and to encourage microbial denitrification. Because the redox potential for nitrate reduction is higher than that of iron reduction, the latter would inhibit and P would remain complexed with ferric compounds (Cooke *et al.*, 1993b).

6.2.5.1 Case studies

6.2.5.1.1 Lake Lillesjön, Sweden

Following the treatment of Lake Lillesjön (Sweden) sediments in 1975, interstitial P content in the top 20 cm dropped by 70 to 85 % of the 1974 to 1975 levels. The decreased levels persisted through at least 1977. Despite the high loading of NO_3 to the sediments, nitrogen was lost through evolution of N_2 gas, and ammonium actually decreased. Denitrification of the added NO_3 was complete after 1.5 months. The oxygen demand of the sediment also decreased by about 30 %. Recycling of P and N to the water overlying treated sediment was reduced to between 10 % and 20 % of the rate observed prior to restoration. Oxygen demand of sediment and sediment P release has continued to remain below 10 years after treatment (Ripl, 1986).

6.2.5.1.2 White Lough, Ireland

P concentration in stratified White Lough, Ireland, decreased following a calcium nitrate treatment of its sediment. P release from sediment was delayed (about 1 month) and the maximum hypolimnetic concentration reached was 30 % less following treatment. Because lake results conformed to those predicted from laboratory experiments, sufficient control of lake P would have been expected had the close been the optimum of 30 to 60 gN/m^2 instead of 24 gN/m^2. The release pattern for iron was nearly identical to that of P, but manganese (Mn) and dissolved oxygen were unaffected (Foy, 1986).

6.2.5.2 Costs

The total cost of the Lake Lillesjön (Sweden) treatment was about \$130,000 (1990 dollars), of which 44 % was for development of the device and preliminary lake investigations. The equipment part of this, of course, would not be necessary in subsequent lake projects. The chemicals applied to the 1.2 ha lake area represented only 6 % of the total cost, while 28 % of the total went for equipment installation. The latter would not vary greatly with the area treated. The remaining was for equipment and labour.

The cost of an ideal dose (40 gN/m^3) of calcium nitrate to White Lough (Ireland) was compared with that of an earlier (1980) treatment of the lake with iron/alum. The total costs (1990) for calcium nitrate and iron/alum were \$23,970 and \$6,400, respectively. Also, the cost per treated hectare was \$5,200 and \$1,390, respectively (Cooke *et al.*, 1993b).

6.2.5.3 Possible application in Loch Logan

For sediment oxidation special equipment is needed to "inject" the chemical solutions into the sediment, which makes this option more expensive. However, it is a reasonable alternative to alum treatment to inactivate sediment P and the effects are more permanent.

However, in shallow lakes like Loch Logan where pH and temperature may have a greater influence on P release during summer, sediment oxidation may not be as efficient to decrease the P loading (Cooke *et al.*, 1993b).

6.2.6 HYPOLIMNETIC AERATION

Depletion of oxygen in the hypolimnia of eutrophic lakes is one of the first signs of eutrophication. If enrichment becomes sufficient to exhaust all or a substantial portion of the hypolimnetic oxygen reserve before autumn destratification occurs, anoxia will result. Anoxia can produce several undesirable changes in lake quality including accelerated internal recycling of nutrients, solubilisation of metals that are undesirable in water supplies and illumination of fisheries, especially regarding coldwater species.

Hypolimnetic aeration, first developed in Austria, is a lake management technique designed to counteract hypolimnetic anoxia and its associated problems. The specific objectives of hypolimnetic aeration may be as many as three-fold. The first and usually most attainable is to raise the oxygen content of the hypolimnion without destratifying the water-column or warming the hypolimnion. The second, which is largely dependent on the first, is to provide an increased habitat and food supply for coldwater fish species. Thirdly, if sediment-to-water exchange of P is controlled by iron redox, the internal loading of P should decline by establishing aerobic conditions at the sediment water interface. Other constituents that reach high and possible undesirable concentrations under anaerobic conditions, such as NH_4^+, Mn and Fe, should also be diminished by hypolimnetic aeration (Cook *et al.*, 1993b).

In most cases where hypolimnetic aeration was used, the oxygen concentration in the hypolimnion increased. Hypolimnetic P decreased 55 % in Lake Wesslinger (Germany) following aeration. A 30 % decrease in P was also observed in the hypolimnion of Lake Waccabuc (USA) following aeration (Cooke *et al.*, 1993b).

6.2.6.1 Costs

The largest lake so far to be treated by hypolimnetic aeration is Tegeler See, which is 420 ha. Fifteen Limnox units were installed in 1980 to aerate 32×10^6 m^3 of hypolimnion, delivering 4.5 tons of O_2 per day. Total initial cost of the aeration system was $2,732,000 (1990 dollars). That is about $340/kg$O_2$, or $6,500/ha for 6 months of operation. Cost would also decrease in course of time.

Aqua Technique has reported the installation and operation costs for seven partial air-lift projects. At a power rate of \$0.09/kw-h, the average operation cost/kgO_2/d was 0.072 ± \$0.026 (or 0.8 kwh/kgO_2), and the average installed cost/kgO_2/d was 457 ± \$280 (1990 dollars). When the latter value is divided by 180d, that cost becomes \$2.50/kgO_2, which is probably the most useful cost indicator for hypolimnetic aeration since success depends on satisfying and oxygen demand (Cooke *et al.*, 1993b).

6.2.6.2 Possible application in Loch Logan

Hypolimnetic aeration is a crisis management option when organic load is high and the oxygen concentration is low. It is not as permanent as other techniques such as P inactivation. However, the aeration can disturb and mix the sediment through the water-column and it is therefore advisable that the aeration pipes are suspended ~0.5 m above the sediment.

6.2.7 ARTIFICIAL CIRCULATION

Artificial circulation is also referred to as destratification. Circulation has been achieved by pumps, jets and bubbled air. Complete circulation is usually the objective and in the majority of cases examined, either stratification was prevented or destratification resulted. Unlike hypolimnetic aeration, the temperature of the whole lake is raised with complete circulation; the greatest increase in temperature occurs at depths that were previously part of the hypolimnion.

The principal improvements in water quality caused by complete circulation are aeration and chemical oxidation in the entire water-column. Complete circulation may reduce the internal loading of P if the principal mechanism was due to iron solubility in the profundal sediments. Complete circulation may also reduce algal biomass by increasing the mixed depth, thereby reducing available light, and by subjecting mixed algal cells to rapid changes in hydrostatic pressure. In some cases, phytoplankton biomass and P content either showed no change at all or was increased following circulation.

Circulation would probably be best used alone as a management technique and not combined with other methods designed to reduce P content. This is because the benefits for algal control would be best achieved in non-nutrient-limited situations, and because increased circulation may encourage internal loading of P and counteract other efforts to lower the P (Cooke *et al.,* 1993b).

Artificial mixing could prevent the growth of *Microcystis* because this organism benefits from a stable, stratified water-column. Gas vesicles provide the *Microcystis* colonies with buoyancy, which enables them to concentrate their biomass in the upper mixed layer. In this way, *Microcystis* increases its total daily light dose during periods of low wind speed, while non-buoyant phytoplankton suffers from increased sedimentation losses in a stable lake. In an artificially deep-mixed lake, *Microcystis* loses its advantage of buoyancy and will receive a much lower light dose (Visser *et al.,* 1996).

6.2.7.1 Costs

Cost information for artificial circulation on a project basis is relatively scarce. An annual cost of $46,000 (1990 dollars) for two air compressors producing an airflow rate of 1200 CFM at standard conditions has been cited. This price includes pipes and air diffusers. At the recommended rate of 9.2 $m^3/min \cdot km^2$, this represents a cost of $390/ha (for first year of operation), with is modest in comparison with other restoration techniques.

Other cited initial and annual costs of 13 projects in Florida, USA, ranged from $290 to $3,460/ha and $86 to $1,641/ha, respectively. Mean values for initial and annual cost were $718 and $320/ha (1990 dollars), respectively (Cooke *et al.,* 1993b).

6.2.7.2 Possible application in Loch Logan

Artificial circulation was successfully implemented in deep lakes, but not in shallow lakes. In the latter the chances are good that the circulated water in Loch Logan may disturb and distribute sediment through the water-column and this could worsen the nutrient problem in the water-column. Therefore, this is not recommended as an option in Loch Logan.

6.2.8 SEDIMENT REMOVAL

When properly conducted, sediment removal is an effective lake management technique. Objectives of sediment removal are deepening, nutrient control, removal of toxic substances and rooted macrophyte control.

There are two major variations on removal of sediment from freshwater lakes and reservoirs. The first one, lake drawdown followed by bulldozer and scraper excavation, has limited application. It has been used most successfully in small reservoirs. The obvious limitation of this technique is that water must be drained or pumped from the basin. A second drawback is that the basin must be allowed to dewater sufficiently before earth-moving equipment can operate. In spite of these problems, plus the added concern or truck traffic to transport the removed sediment, this approach has been used with some success, for example at Steinmetz Lake (USA).

The second and by far most common sediment removal technique involves the use of dredges. Many types of dredges are in use today. Dredges can be classified in three categories: mechanical, hydraulic, and “special purpose” dredges (Cooke *et al.,* 1993b).

6.2.8.1 Case studies

6.2.8.1.1 Lake Trummen, Sweden

After years of receiving sewage discharge, wastewater, domestic waste discharge, etc., the conditions became so bad in the early 1960's that citizens of Växjö considered filling the basin (Björk, 1974). Limnological investigations conducted revealed major internal nutrient recycling from the extremely rich upper 1 m of sediment.

Half-a-meter of sediment was dredged from the main lake basin in 1970, and another half-a-meter was removed in 1971. The total volume of sediment removed was about 30×10^5 m^3. This reduced the P content in the surface layer of sediment from approximately 2.4 mg/l to 0.1 mg/l.

TP reduction in the surface water of Lake Trummen after dredging was from 600 µg/l to a range of 70 to 100 µg/l (Cooke *et al.,* 1993b).

6.2.8.2 Costs

The most far-reaching plan they had for Lake Tuusulanjärvi (Finland) (area = 5.95 km^2) was the removal of sediment from the heavily eutrophicated northern end over an area of 3.5 hectares (350,000 m^2), one meter deep, in the beginning of the 1990's. The cost of removing a sediment layer of one meter was expected to be about two million marks. However, the water quality of a whole lake is not expected to be improved by sediment removal of small areas (Pekkarinen, 1990).

The variables that affect the costs of sediment removal include the type of equipment, the volume of the material to be removed, availability of a disposal site, the distance to the disposal area and the ultimate use of the removed material. Sediment removal costs of 11 USA lakes are provided in **Table 6.2**.

6.2.8.3 Possible application in Loch Logan

Dredging can work for Loch Logan. After the old sediment is removed, the newly exposed sediment will then act as a sink for nutrients by extracting it from the water-column. A few facts should be taken into consideration before dredging. Firstly, Loch Logan has to be drained, which can result in bad odours rising from the sediments. Secondly, it will take time and noise pollution will be a problem. Thirdly, there will be several disposal traffic around the Waterfront, which all could be quite inconvenient for those who make use of the premises. Fourthly, it is quite expensive to remove even the smallest amount of sediment.

An alternative option is to use vacuum sucking to extract the sediment.

Table 6.2: Sediment removal costs of 11 USA lakes (Cooke *et al.*, 1993b).

Lake	Treatment Type	Area (ha)	Sediment removed (m^3)	Treatment Cost ($/ha)
Hall Moon Lake	Dredging (~30 % of basin)	53.4	25 x 10^3	3,205
Lilly Lake	Dredging (100 % of basin)	35.6	680 x 10^3	6,876
Commonwealth Lake	Dredging (100 % of basin)	2.6	19 x 10^3	8,653
Steinmetz	Draining and Bulldozing (75 % of basin)	1.2	2 x 10^3	10,849
Carnegie Lake	Dredging (75 % of basin)	110	765 x 10^3	18,081
Lenox Lake	Dredging (100 % of basin)	13.4	76 x 10^3	19,992
Nutting Lake	Dredging (56 % of basin)	31.6	275 x 10^3	28,288
Sunshine Springs	Dredging (100 % of basin)	0.4	5.1 x 10^3	40,248
Krause Springs	Dredging (100 % of basin)	0.3	4.9 x 10^3	47,631
Collins Park Lake	Dredging (100 % of basin)	24.3	52 x 10^3	58,767
59th Street Pond	Draining and Bulldozing (100 % of basin)	1.8	13 x 10^3	150,907[a]

([a] = all sediment from 59th Street Pond had to be trucked out of New York City, thus escalating the costs).

6.3 CONCLUSIONS

The most applicable solution for the reduction of P in Loch Logan seems to be a combination of various management options. For example, hypolimnetic withdrawal and aeration during the spring and summer when organic loads and nutrients concentrations are high and oxygen concentrations are low, and phosphorus inactivation or sediment oxidation in the winter when there is less rain to wash the chemicals out of the system.

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SUMMARY

At present many water resources in South Africa are polluted by industrial effluents, domestic and commercial sewage, agricultural runoff and litter. Declining water quality, associated with human activities, has created environmental and public health problems.

Loch Logan is an urban impoundment near the city centre of Bloemfontein, Free State, South Africa, which is fed through Bloemspruit, a canal, which collects stormwater runoff. The catchment area is ~16 x 10^6 m^2 and after a rainstorm ±30 % of the rain ends as runoff water in Loch Logan.

The study was conducted between January 2000 and May 2001 to determine among others the seasonal and diel variation patterns of limnological characteristics, i.e. physical, chemical and biological parameters, to contribute to the limnological knowledge for proper management and conservation of the system.

The physico-chemical characteristics of Loch Logan showed a distinctive seasonal trend, which was primarily influenced by climate and rainfall. Discharge has been proven to be an important variable influencing the water quality of Loch Logan. After a rainstorm the nutrient concentrations increased in Loch Logan, as well as the turbidity, but the conductivity decreased. All the data collected showed that Loch Logan is an eutrophic system with an average reactive *ortho*-phosphate (PO_4-P) concentration of 78.7 µg/l and a maximum of 648.3 µg/l. The average and maximum nitrogen concentrations were also high: ammonium nitrogen (NH_4-N) 181.5 µg/l and 1,032.6 µg/l, respectively, and nitrate nitrogen (NO_3-N) 201.2 µg/l and 1,375.1 µg/l, respectively.

The pollution sources, organic and inorganic, are mainly stormwater runoff, garden and domestic trash and also human sewage. During the decomposition of these pollutants the oxygen (O_2) concentration decreased drastically, especially in the bottom water layer where the oxygen concentration was usually low. The average oxygen concentration in the surface water was high at 6.6 mg/l (72.8 %) compared to the average of 2.0 mg/l (16.5 %) in the bottom layer.

The annual average chlorophyll-*a* concentration in Loch Logan (average 56.8 μg/l and maximum 487.2 μg/) was typically that of a eutrophic system (15-75 μg/l). Eutrophication is therefore seen as the single most important threat to the water quality of the Loch Logan aquatic system. A higher average in Chl-*a* concentration was found during the summer than during the winter. Some of the dominant algal species (*Chlamydomonas, Chlorella, Euglena, Scenedesmus*) are indicators of polluted water and the algae also shown seasonal trends. During blooms the algal composition mainly consists of cyanobacteria, namely *Microcystis* sp. (dominating) and *Anabaena* sp.

In the diel variations there was no distinctive patterns in the physico-chemical characteristics, except for temperature, oxygen and pH. The photosynthesis rate per chlorophyll-*a* was higher in October (3.68 mgC/mgChl-*a*/h) than in July (1.24 mgC/mgChl-*a*/h), which indicated that temperature was one of the controlling factors of photosynthesis.

Various management and restoration options for Loch Logan were discussed, for example diversion of stormwater, diluting, flushing and phosphorus inactivation. The solution for the pollution is possibly a combination of the options mentioned above.

Key words: urban impoundment, eutrophication, physical and chemical parameters, nutrients, seasonal cycles, diel changes, algae, blooms and management.

OPSOMMING

Huidiglik is baie van Suid-Afrika se waterbronne besoedel deur industriële uitvloeisels, huishoudelike- en besigheidsriool, landbou afloop en rommel. Verlaging in waterkwaliteit wat met menslike aktiwiteite geassosieer word, veroorsaak omgewings en publieke gesondheidsprobleme.

Loch Logan is 'n stedelike dam naby die middestad van Bloemfontein, Vrystaat, Suid-Afrika, wat gevoed word deur Bloemspruit, 'n kanaal, wat stormwater afloop versamel. Die opvangsgebied is ~16 x 10^6 m^2 en na 'n reënstorm eindig ±30 % van die reën in Loch Logan as afloopwater.

Die studie is vanaf Januarie 2000 tot Mei 2001 uitgevoer en onder andere is die seisonale en daaglikse veranderingspatrone van limnologiese eienskappe, byvoorbeeld fisiese-, chemies- en biologieseparameters, bepaal sodat bydraes tot die limnologiese kennis gemaak kan word vir bestuur en bewaring van die stelsel.

Die fisies-chemiese eienskappe van Loch Logan wys duidelike seisonale neigings wat hoofsaaklik deur die klimaat en reënval beïnvloed is. Dit is bewys dat afloopwater 'n belangrike invloed op Loch Logan se waterkwaliteit het. Na 'n reënstorm het die voedingstofkonsentrasies in Loch Logan verhoog asook die troebelheid, maar die geleiding het verlaag. Al die data wat versamel is, dui daarop dat Loch Logan 'n eutrofiese stelsel is met 'n gemiddelde reaktiewe *orto*-fosfaat (PO_4-P) konsentrasie van 78.7 µg/l en 'n maksimum van 648.3 µg/l. Die gemiddelde en maksimum stikstof konsentraties was ook hoog: ammonium stikstof (NH_4-N) was onderskeidelik 181.5 µg/l en 1,032.6 µg/l, en nitraat stikstof (NO_3-N) was onderskeidelik 201.2 µg/l en 1,375.1 µg/l.

Die besoedelingsbronne, organies en anorganies, is hoofsaaklik stormwater afloop, tuin- en huishoudelike afval asook menslike riool. Tydens die afbreek van die besoedelingstowwe het die suurstof (O_2) konsentrasie drasties gedaal, veral in die bodemlaag waar die suurstof konsentrasie meestal baie laag was. Die gemiddelde suurstof konsentrasie in die oppervlak water was 6.6 mg/l (72.8 %) in vergelyking met die gemiddeld van 2.0 mg/l (16.5 %) in die bodemlaag.

Die jaarlikse gemiddelde chlorofil-*a* konsentrasie in Loch Logan (gemiddeld 56.8 µg/l en maksimum 487.2 µg/l) was tipies dié van 'n eutrofiese stelsel (15-75 µg/l). Eutrofikasie word dus gesien as die enkel mees belangrikste bedreiging vir die waterkwaliteit van Loch Logan se akwatiese stelsel. 'n Hoër gemiddeld in Chl-*a* konsentrasie is gedurende die somer gevind as gedurende die winter. Van die dominante algspesies (*Chlamydomonas, Chlorella, Euglena, Scenedesmus*) is aanduiders van besoedelde water en die alge toon ook 'n seisonale neiging. Die algsamestelling tydens die opbloeie het meestal uit die sianobakterieë, naamlik *Microcystis* sp. (dominant) en *Anabaena* sp. bestaan.

Tydens die daaglikse veranderinge is daar geen duidelike patrone van die fisieschemiese eienskappe waargeneem nie, behalwe vir temperatuur, suurstof en pH. Die fotosintese tempo per chlorofil-*a* was hoër tydens Oktober (3.68 mgC/mChl-*a*/h) as tydens Julie (1.24 mgC/mChl-*a*/h), wat 'n aanduiding is dat temperatuur een van die beheerende faktore van fotosintese was.

Verskeie bestuur en restorasie keuses vir Loch Logan is bespreek, byvoorbeeld omleiding van stormwater, verdunning, spoeling en fosfaat inaktivering. Die oplossing vir die bekamping van besoedeling is waarskynlik 'n kombinasie van bogenoemde aksies.

Sleutelwoorde: stedelike damme, eutrofikasie, fisiese en chemiese parameters, voedingstowwe, seisoenale siklusse, daaglikse veranderinge, alge, opbloei en beheer.