# The Limnology of Hartbeespoort Dam

A report by the Limnology Division of the National Institute for Water Research, CSIR in collaboration with The Water Research Commission and Ecosystem Programmes, Foundation for Research Development, CSIR.

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# AKNOWLEDGEMENTS AND SUMMARY

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## 1.1 ACKNOWLEDGEMENTS

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The study would not have been possible without the dedicated and enthusiastic technical assistance from L.B. Bostock, S. Combrink, T.A. Fenn, L.M. Grimbeek, H.M. Herbst, M.J. Hills, R.F. Mitchell, A.M. Pais Madeira and S.D. van Blommestein. They contributed in a number of ways, including collection and analysis of field data and data capture.

All scientists contributed to the initial stages of model building, but later development of the model as a whole was largely carried out by P.J. Ashton, A.J. Twinch and especially K.L. Cochrane. During the course of the study a number of other people participated for varying periods of time.

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A technical committee made up of the scientists involved in the study plus the external advisers of the Limnology Division of the National Institute for Water Research (Professors CM. Breen and J.G. Field) and representatives of the Directorate of Water Affairs (Mr E. Braune, Mr S.de Wet) met at six-monthly intervals under the chairmanship of Dr H.N.S. Wiechers of the Water Research Commission to consider research progress and plan future action. In the first year of the study Professor B.H. Walker took part in the Technical Committee Meetings to advise on ecosystem modelling.

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Mrs P. Vos had the onerous task of typing this report and its many draft editions. The contributors gratefully acknowledge her skill and goodwill throughout this long task.

## 1.2 SUMMARY

Hartbeespoort Dam is a hypertrophic, warm, monomictic impoundment. With a mean depth of 9.6 m and a surface area of 20 $km^2$, the system demonstrates that hypertrophy is not confined to small shallow lakes as concluded by Barica (1981). The combination of high nutrient loading, high incident solar radiation, low wind speeds and warm water makes it the ideal environment for the prolific growth of the buoyant blue-green alga, *Miarooystis aeruginosa** Photosynthetic production of this organism is regulated mainly by the penetration of light through the water column. The *Miorooystis* population grows to such large levels that the population self-shades itself, i.e., the alga is usually the dominant factor regulating light penetration in the water. Unlike most hypertrophic lakes, wide spread anoxia resulting from the 'die-off* of the large *Miovoaystis* population does not occur in Hartbeespoort Dam. Rather, *Microcystis* tends to accumulate in large, floating mats termed hyperscums and then is redistributed throughout the dam under favourable wind conditions. However, the anaerobic hypolimnion, which may extend upward to within about 8 m of the lake surface, contains large concentrations of reduced compounds (e.g., NH^-N, $H_2S$) and it is the release and distribution of these throughout the water which may result in almost total lake anoxia at overturn.

The excessive nutrient loading to Hartbeespoort Dam has produced large bacterial populations concomitant with the large algal population, neither population being N or P growth rate limited. The bacteria are very small in size (0.1 um), whereas the colonies of the dominant *Mierocystis* tend to be too large for zooplankton grazing. Thus, although the rates of algal production for the dam are the highest measured in the world, the zooplankton population remains small in relation to the dam's primary producers.

Although the annual yield of fish from the dam was 348 kg ha $^{1}$ which in natural African lakes is exceeded only by three shallow (mean depth <3 m), eutrophic, tropical water bodies (Fryer & lies 1972), the fish population of Hartbeespoort Dam has been a victim of hypertrophy. The poor water quality, especially the high pH; the small zooplankton population and an algal population dominated by blue-green algae, have all but reduced the fish population to three species: *Clarices gariepinuej Cyprinus carpio* and *Oreoohromis mossambicus. C. carpio* and *0. mossambieus* are detritivores while *C. gariepinus* feeds mainly on invertebrates. The data produced by the study indicate that the main energy pathway in this hypertrophic lake is through a detrital cycle. Preliminary data indicated that the zoobenthos population was large, further emphasizing the importance of detritus in Hartbeespoort Dam. Thus the statement by Barica (1981), that hypertrophy produces ecosystem structural changes so that the main energy flow is through zoobenthic populations and detritus, appears to be generally applicable.

This finding is important for the future scientific research programme as detritus and major groups of detritus metabolizing organisms, the bacteria and benthic invertebrate fauna, have thus far not been studied sufficiently for understanding of the flow of phosphorus into and out of detritus.

The study has shown that ecosystem modelling has a very useful role in the management and integration of multi-disciplinary ecosystem research programmes. In particular, it aids communicating the rationale of specialist's research priorities to non-specialists and summarises existing knowledge of the functioning of the ecosystem under investigation.

At the interface between scientific research and management an important finding has been the large modifications (reductions) of the inflowing nutrient load between the point of river entry into the arm of the dam at weir A2M12 and the main basin of the dam. This should be considered in the modelling of nutrient load/impoundment response. This finding also stimulated the concept of pre-impoundment (see below) in the management of the dam. A related finding has been that there is a large discrepancy (approximately 50%) between the sum of the point source phosphorus loads discharged in the catchment (Grobler & Silberbauer 1984) and the river load measured at weir A2M12. Many factors, including inaccuracies arising out of measuring phosphate concentrations in surface water samples from the river at daily intervals, could contribute to this discrepancy.

From the direct management of Hartbeespoort Dam aspect, the discrepancy between point source loads and the river load at weir A2M12 bedevilled authoritative statements as to the impact of the 1 mg *Z* $^{1}$ orthophosphate phosphorus standard on the trophic status of the dam, for it would be unreasonably speculative to assume that the phosphate load on the dam would be reduced in direct proportion to the reduction at the point sources.

The study examined, in addition to the potential effect of the 1 mg *Z* $^{1}$ standard, a number of other management procedures studied and in certain instances utilized in other countries. Each of these procedures was assessed for its effectiveness, either singly or in combination, using published empirical models and the Hartbeespoort Dam Ecosystem model. The conclusions reached were:

(a) ]_jng *Z* _ P standard - it was estimated that, had implementation of this point source standard taken place in the hydrological year 1980/81, a reduction of 75% JLn the phosphorus load from point sources (i.e. from 539 t a $^{1}$ to 130 t a *) would have occurred. However, in 1980/81 only 283 tonnes P were measured at weir A2M12, just above the dam, indicating that about 50% of the point source load was lost in the river. If therefore the load at weir A2M12 were also to drop by 75% as a result of the 1 mg *Z* $^{1}$ P standard it would be 71 t a '. Greater precision in forecasting the likely alteration in the impoundment (A2M12) phosphorus load is not possible, due to the unknown extent of phosphorus losses in rivers. At a 75% reduction in the impoundment (A2M12 weir) phosphorus load, it is predicted that the dam will remain eutrophic (chlorophyll > 30 yg *Z* *) or near the lower limit of eutrophy. It is predicted that a reduction in the impoundment (A2M12) phosphorus load of about 85% will result in a condition bordering between eutrophy and mesotrophy (chlorophyll between 25 - 35 yg *Z* $^{1}$). This phosphate reduction can be brought about by an effluent phosphorus standard of 0,5 mg *Z* *, assuming no river phosphorus removal. It was concluded that the phytoplankton content of the dam would reach equilibrium with the new phosphorus load within three to six years.

(b) Pre-imp_oundment - a pre-impoundment of about 26 x. 10  m$^3$ (a tenth of the full supply volume of Hartbeespoort Dam) could cause an approximately 60% P load reduction in average years. This was regarded as a potentially promising eutrophication management option, if used to supplement point source nutrient removal.

(c) Aeration/destratification - was identified as an in-dam management technique which holds potential. Experimentation with this technique was attempted in the mid seventies, unfortunately with little success due to the short duration of the test and a variety of practical problems. The current investigation was a 'desk-study' with no in-dam experimentation being undertaken. A thorough literature search was made and using this information and the data from Hartbeespoort Dam, the possible biological and chemical consequences of aeration/destratification were assessed. It was predicted that the application of this technique may result in the change of the dominant algal species from a toxic blue-green to non-toxic green alga, as a result of water turbulence or a change in the nitrogen to phosphorus ratio. It was shown using the ecosystem model that should aeration/destratification result in dominance of the phytoplankton by a species palatable to phytoplankton grazing organisms, there would be a 50% decrease in the standing stock of phytoplankton and the zooplankton abundance would increase. Mixing of the cold bottom water with the warm surface waters may also result in a lowering of the water temperature with a concomitant decrease in water evaporation rate. However, aeration/destratification may also result in a number of undesirable consequences, for example the proliferation of filter clogging algae. Pilot or full-scale experimentation will be necessary to establish the true potential of this technique as a in-dam method for dealing with eutrophied waters.

(d) Protein harvesting -

(i) The fish community in Hartbeespoort Dam is being exploited by anglers close to its maximum sustainable yield (about 525 t a ). Consequently a commercial fishery would adversely affect recreational fishing and is therefore not recommended. It should be noted that while the fish yield (0,35 t ha a ) is large relative to most African lakes and dams (range reported 0,004 to 0,50 t ha a ), it is low relative to yields reported for fish ponds (1,0 to 2,0 t ha a ) receiving an external food source, the reasons being, *inter alia,* poor water quality, the small zooplankton population and an algal population dominated by unpalatable blue-green algae, as well as the fact that fish ponds are designed for high intensity fish production whereas dams are not.

(ii) It was estimated that approximately 3600 tons (wet mass) of *Microaystis* could be harvested per year. However, this particular alga is toxic to cattle and sheep, and possibly man, as well as being deficient in certain essential amino-acids and is therefore of little practical use. If the conditions in the dam could be changed to result in a more palatable algae, this will have definite advantages for its protein production potential. Unfortunately, the factors controlling algal type are not sufficiently well understood to make practical control in the dam a feasible undertaking.

(e) Bio-manipulation - a variety of bio-manipulation strategies to ameliorate the consequences of eutrophication were considered, but none appeared to hold much promise as viable eutrophication management techniques.

In summary, the only option for reducing the abundance of algae and improving water quality in Hartbeespoort Dam that could be recommended with a large degree of certainty was the reduction of the external phosphorus load. This could be achieved either through point source load reduction or pre-impoundment, but most likely through a combination of both.

In addition to the research needs listed above, this ecosystem study also high-lighted the need for continued research on the role of detritus and dissolved organic carbon in the functioning of hypertrophic impoundments, on physical limnology and the further refinement of the ecosystem model. From the management perspective important research and development needs are -

(a) The assessment of the response of Hartbeespoort Dam to the introduction of the effluent phosphate standard. This is crucial since such an assessment will validate the predictions made by the ecosystem model (and other, empirical, models), in order that such models can in future be used with confidence on other impoundments.

(b) The dynamics of phosphorus in water courses. The unaccounted losses of phosphates in river systems as well as in pre-impoundments are very significant and therefore need to be more thoroughly quantified. The removal mechanisms and the parameters which control them need to be identified and quantified. A thorough understanding of these processes may affect future decisions regarding stricter effluent standards and allowable impoundment phosphate loads.

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## 2. INTRODUCTION

As the population of South Africa expands and simultaneously living standards and expectations increase, ever greater demands are made on the limited water resources of the country. It has recently become apparent to planners that water resources can be limited not only by quantity but also by quality, due to impairment by mineralization and by the consequences of nutrient enrichment or eutrophication.

In the past, South African limnologists and water planners have tended to follow the philosophy of the northern hemisphere with regard to eutrophication. This philosophy is that eutrophication is undesirable and must be done away with, principally by limiting the input of phosphorus to the aquatic environment. It resulted in much research to prove that eutrophication could be eliminated by reducing the quantities of phosphorus reaching surface waters. Simultaneously, a related question was of paramount importance to water resource managers - what is the maximum amount of phosphorus that may be allowed to enter a standing water body without its becoming eutrophic? Research on this question has been carried out in many parts of the world, including South Africa, where work sponsored by the Water Research Commission was reported on by Walmsley & Butty (1980). Features common to the world-wide results are that the effects of a given annual load of phosphorus are modified by the mean depth and hydrology of the receiving body of water. However, none of the predictive relationships established have a statistically satisfactory predictive capability. For instance, the Walmsley & Butty model's confidence bands are such that at certain phosphorus loading rates the predicted resultant algal growth could be representative of conditions anywhere from oligotrophic to eutrophic. The same can be said of the confidence bands around the many relationships developed by the Organization for Economic Cooperation and Development (OECD 1982). Since the OECD data base was comprehensive, it is to be doubted that more studies adopting a similar 'black-box[1] approach will considerably improve the precision of the predictions.

The starting point for the studies presented in this report was that the imprecision surrounding 'black-box' models of eutrophication was in part due to a lack of understanding of the quantitative functioning of standing water ecosystems. An improved understanding would be useful in planning the active, informed management of a eutrophied water body, both to derive benefit from its enhanced productivity and to ameliorate its undesirable characteristics. Moreover, this greater knowledge of functioning should yield important leads to the refinement of empirical models of the relationships between nutrient load and trophic status.

Hypertrophic (excessively enriched) Hartbeespoort Dam was selected as the study site. This impoundment (20 km$^2$), which has posed unsolved management problems, enjoys intensive multiple use for irrigation, all forms of aquatic recreation and raw water supply, and is reasonably convenient to the laboratory. Toerien & Walmsley (1976) concluded that diffuse sources of phosphorus in the catchment of the dam are alone sufficient to maintain it in a eutrophic condition. It is a water body where in-lake biological, physical or chemical management is likely to be required. Hartbeespoort Dam represents the probable future fate of many other important impoundments in South Africa such as Vaal Dam, Bloemhof Dam, Loskop Dam and other dams in whose catchments there is intensive industrial development and urbanization. Hypertrophy is of national rather than regional importance.

The study presented here was structured around a simulation model of ecosystem functioning in terms of phosphorus cycling. A preliminary model was built before research and monitoring commenced, using the extensive but patchy existing data on the dam. In this initial phase the model helped clarify concepts and identify key research questions.

In the next phase of the study variations in the properties of the dam and those used to drive the model or to validate its output were monitored. At the same time many of the rates at which processes built into the model took place were measured. The preliminary model was frequently modified and updated as more knowledge became available and, where necessary, the research programme was modified. The model played a key role in research management and helped ensure that the diverse team research on different components of the ecosystem produced results which could be inter-related. Since nitrogen is also an important algal nutrient, it was included in the study but does not feature in the overall ecosystem model.

Once the aim of building a reasonable simulation model of the functioning of Hartbeespoort Dam had been achieved, attention was given to the following objectives of the study, using the model in a predictive mode:-

(a) evaluation of the impact of the 1 mg $Z^{1}$ orthophosphate standard for effluents discharged in the Hartbeespoort Dam catchment on the functioning of the ecosystem;

(b) assessment of the consequences of hypolimnetic aeration or destratification on the characteristics of the impoundment;

(c) feasibility of biological management of the impoundment to ameliorate undesirable properties;

(d) assessment of the protein production potential of the impound ment;

(e) evaluation of any other in-lake management options which might become apparent during the study.

This report is written for a wide readership, ranging from specialist scientists to decision takers and managers. This has been taken into account in the structuring and style of this document. It could well be that the chapters on methods and the limnology of the dam are of little interest to some readers concerned mainly in management. The chapters have been written to be read individually, if necessary.

# 3. THE STUDY AREA

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## 3.1 CONSTRUCTION HISTORY AND GEOLOGY

Hartbeespoort Dam (Fig. 3.1) was completed in 1925. Located 37 km west of Pretoria on the Crocodile River, immediately downstream of its confluence with the Magalies River, the dam is a variable radius, mass concrete structure. It has a trough spillway located on its western flank, which was fitted with radial crest gates in 1971 to increase the storage capacity of the impoundment.

Two outlet works were constructed, one on each flank serving canal systems supplying water to a 130 $km^2$ irrigation scheme. The outlet ports are located at 20 m below full supply level.

A summary of the morphological characteristics of the dam and the impoundment is given in Table 3.1.

The geological structure of the area consists of quartzites, conglomerates and shales of the Ventersdorp series and granites, dolomite belts, quartzites, shales and diabase of the Pretoria series (du Toit 1954). The dam wall is situated in an area of compact quartzites with a northerly dip of about 30 °.

Table 3.1. Morphological characteristics of Hartbeespoort Dam.

| | |
|---|---|
| **Hartbeespoort Dam Wall** | |
| crest level | 1167 m above mean sea level |
| high flood level | 1165 m |
| full supply level | |
| pre-1971 | 1160 m |
| post-1971 | 1162 m |
| crest width | 4.6 m |
| crest length | 140 m |
| outflow flood | 2322 $m^3s^{-1}$ |
| **Hartbeespoort Dam** | |
| maximum depth | 32.5 m |
| mean depth ($\bar{z}$) | 9.6 m |
| maximum breadth | 12 km |
| length | 5.6 km |
| full supply volume | |
| pre-1971 | 168 x $10^6 m^3$ |
| post-1971 | 195 x $10^6 m^3$ |
| full supply surface area | 20 $km^2$ |
| catchment area | 4144 $km^2$ |
| mean annual abstraction | 127 x $10^6 m^3$ |
| mean annual water residence time | 0.87 year |

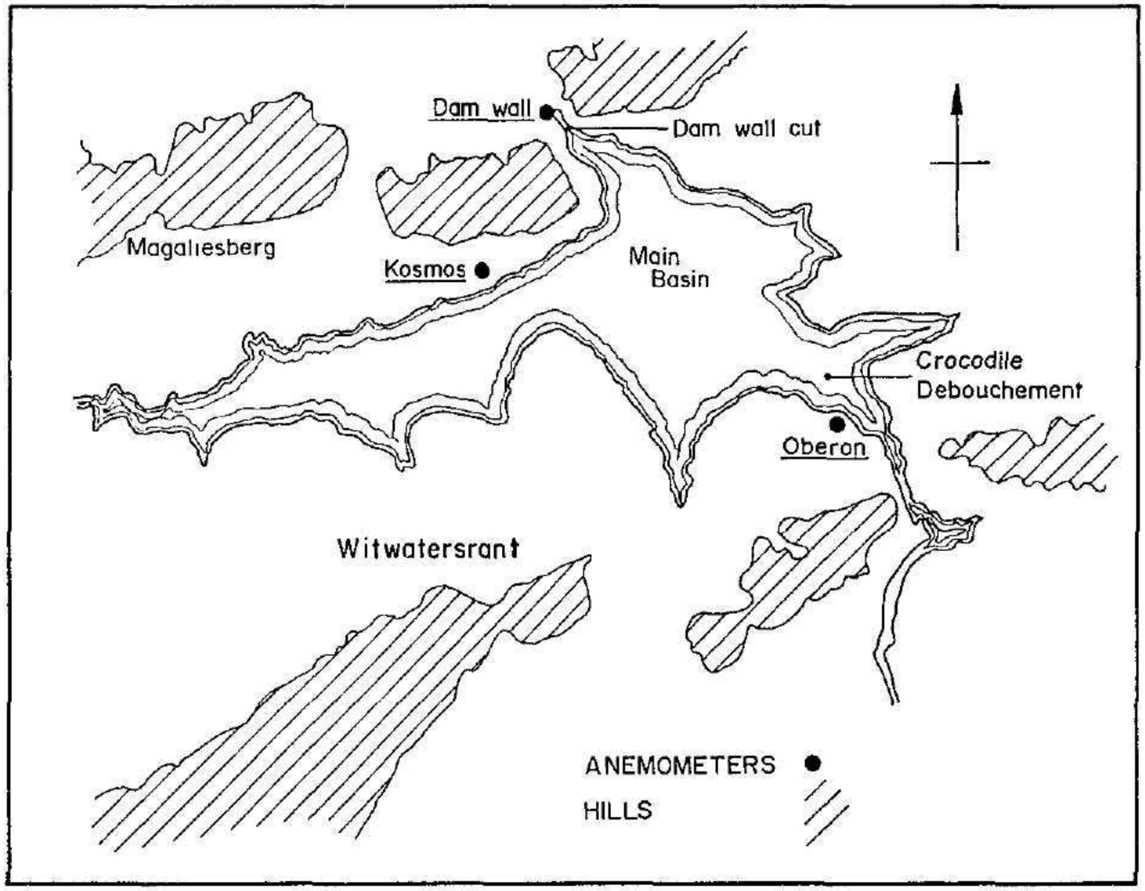

Figure 3.1. Hartbeespoort dam at full supply level, showing surrounding hills (land above 1280 mamsl) and principal study areas (traced from map 2527 DB/DD- government printer).

## 3.2 MORPHOMETRY

A morphometric map of Hartbeespoort Dam is given in Figure 3.2. The former stream beds are clearly visible despite some evidence of siltation.

## 3.3 LAND USAGE

Land usage in the Hartbeespoort Dam catchment can be divided into two principle categories; namely, rural and urban. Urban land use includes commercial, residential, and industrial areas associated with the northern suburbs of Johannesburg and other smaller towns on the Witwatersrand (Fig. 3.3). Twelve percent of the catchment is classed as urban, the bulk of this area being within the Crocodile River catchment area. The other 88% of the catchment is rural. Most of this area is undeveloped land used principally for grazing or natural reserves. The remainder is agriculturally developed.

## 3.4 WATER USAGE

Water from Hartbeespoort Dam is used for domestic consumption by the Magalies Water Board, for irrigation by the Hartbeespoort Government Water Scheme area, and for other uses by downstream consumers (released as compensation water). Water was also previously used for power generation. Of the approximately 130 x $10^6$ $m^3$ abstracted from the lake annually, 6% is used for domestic consumption, 12% is released as compensation water, and 82% is used for irrigation purposes. Table 3.2 (Section 3.6) gives the water balance of the lake. Domestic consumption was divided between Brits (94%), Kosmos (2%), Schoemansville (2%) and Meerhof (2%). However, abstraction is likely to increase as the area served by the Magalies Water Board expands. The lake is also a major centre for water sports, including fishing, boating and water ski-ing.

## 3.5 CLIMATE

Hartbeespoort Dam lies in the Transvaal Middelveld at an altitude of 1 162 m above mean sea level. Climatically, the lake is situated between the well-watered Drakensberg Mountains and the arid Kalahari Desert; the gradient of mean annual precipitation (MAP) being about 1 mm km $^1$ from east to west in this region. The MAP at the lake is approximately 700 mm per annum, being concentrated almost entirely during the summer months (October to March). Peak rainfall occurs during December (Fig. 3.4).

Monthly mean evapotranspiration potentials approximate closely the rainfall with a small surplus of rainfall (approximately 20 mm month *) in summer. Evapotranspiration potential exceeds rainfall at other times of the year. Evaporation, averaging 2 400 mm per annum, is more constant over the year than rainfall due to the concentration of cloud-cover in summer. This concentration also affects insolation, which commonly varies between 150 (winter) and 290 (summer) W m"$^2$ .

The mean annual air temperature is about 19.5 °C and ranges from < 2 °C in winter to > 32 °C during summer. Diel variations in temperature are approximately 10 to 15 °C year round.

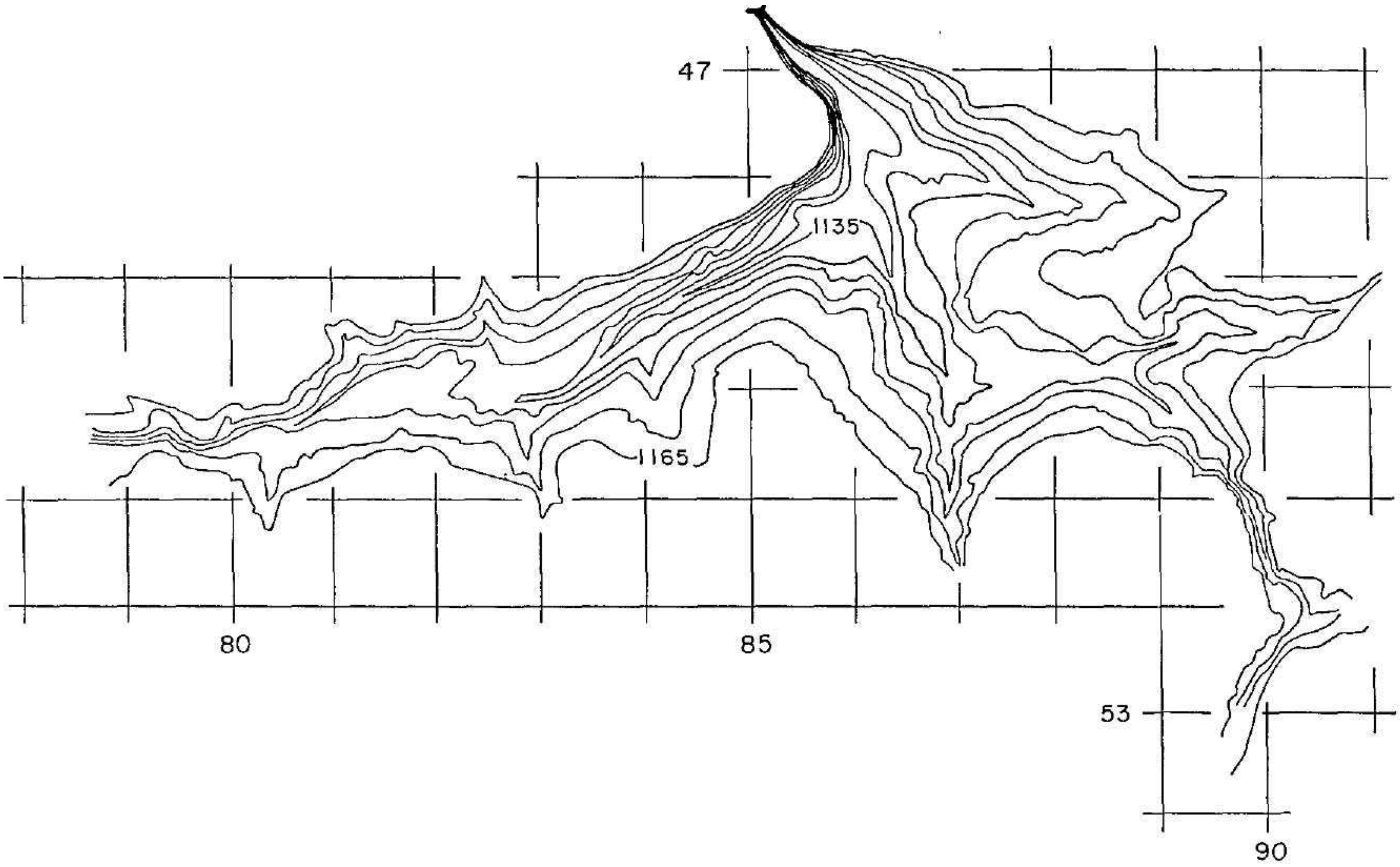

Figure 3.2. Morphometric map, showing contours at 5 m intervals and coordinates of national (1 km) grid (traced from sediment survey chart 75328/80 - Department of Water Affairs).

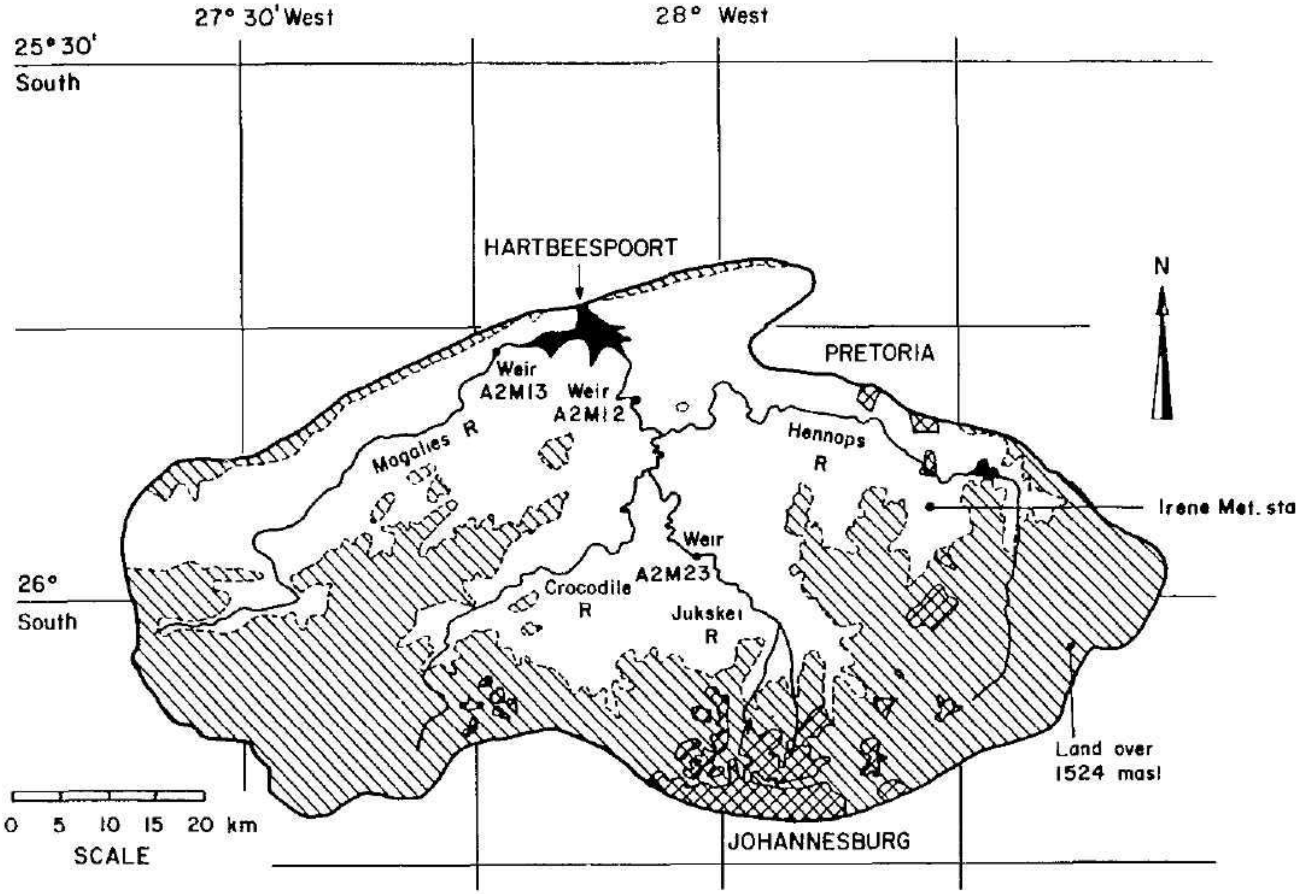

Figure 3.3 Hartbeespoort catchment showing main influent streams, flow gauging weirs, Irene meteorological station and geographical co-ordinates (DWA and Department of Transport) (Hely-Hutchinson, in preparation).

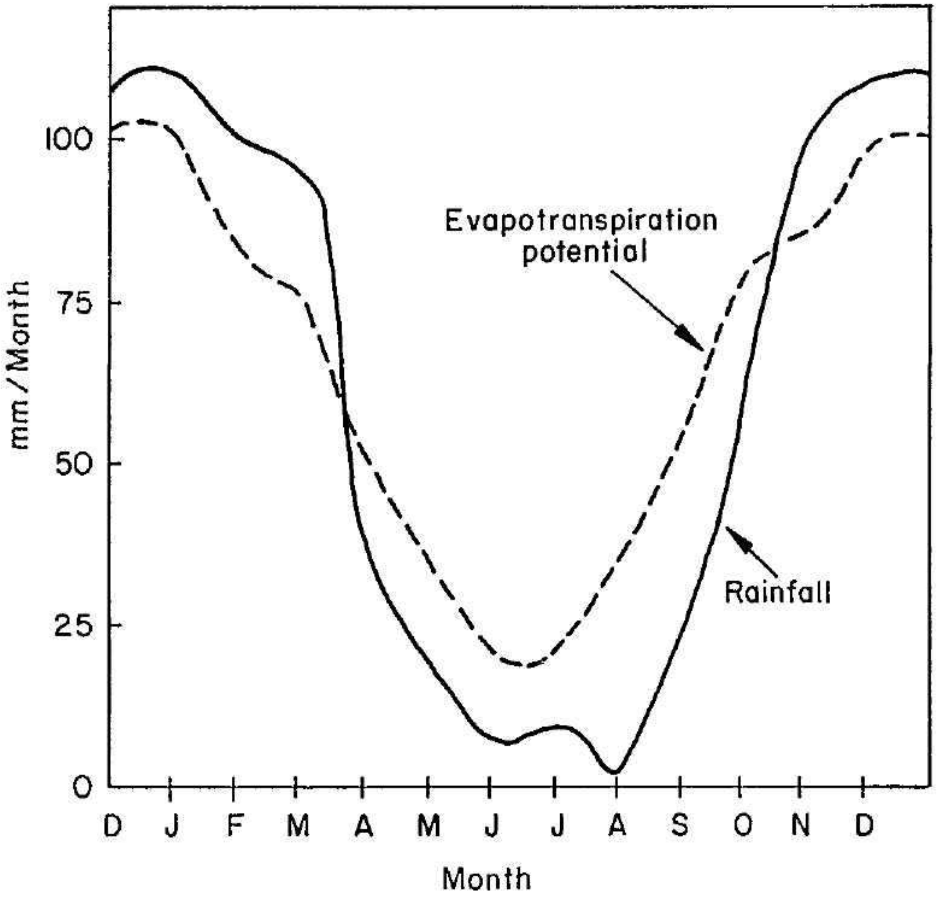

Figure 3.4. Mean monthly rainfall and evapotranspiration-potential for the area (after Schulze 1959).

The mean wind-speed on the lake surface is generally low. The mean monthly wind-speed varies seasonally between l m s in June and 2.5 m s $^{1}$ in October. The prevailing directions are ESE and NW. In the vicinity of the dam wall, winds tend to be stronger, often reversing direction during the day and gusting to over 10 m s . Although the area in the vicinity of the wall is small, it is significant that it is the deepest portion of the lake. It contains c. 10% of the volume at full-capacity and c. 20% at 40% capacity.

## 3.6 HYDROLOGY

River flow, like rainfall, is highly seasonal with the main flows occuring between October and March. The seasonality of the inflow has an effect on nutrient and sediments loads to the lake. The Crocodile River (Fig. 3.3) supplies over 90% of the inflow to the lake. In winter, flows in the Crocodile average 2.3 m$^3$ s *, whilst in summer flows can range from 2 to 20 m$^3$ s on a daily basis. In summer, base flow is normally less than 7 m$^3$ s . Spates commonly endure for between 1 and 20 days, and may rise and fall within a few hours or days. The mean annual runoff (MAR) is about 224 x 10$^6$ m$^3$ . Table 3.2 gives the average water budget for Hartbeespoort Dam, while Figure 5.6A (Section 5.1.4) shows the variation in dam volume over the last 20 years

Table 3.2. Long-term water budget for Hartbeespoort Dam (data for period 1964-1978; Directorate of Water Affairs, unpubl.).

| Inflows (x10$^6$ m$^3$) | |
|---|---|
| river inflow | 224.0 |
| precipitation | 9.5 |
| | 233.5 |
| | ====== |
| Outflows (x10$^6$ m$^3$) | |
| sluices | 101.9 |
| abstractions | |
| irrigation | 82.3 |
| compensation water | 16.9 |
| potable (since 1976) | 6.5 |
| seepage | 1.3 |
| evaporation | 18.9 |
| TOTAL | 227.8 |
| | ===== |

## 3.7 ECONOMICS OF HARTBEESPOORT DAM

In his economic survey of Hartbeespoort Dam and its environs, Hofmeyr (1978) identified three major types of development in the area (monetary values are based on a 1970 Rand):

(a) 138 km$^2$ of irrigable land are dependent on the Hartbeespoort Dam, the principal crops being tobacco, vegetables, and wheat, of which tobacco is the most important (20 - 30% of total RSA production). In 1970, crop production was valued at nearly 12 million Rand (R8m, R2.7m and R1.2m respectively). State income from sales of irrigation water during the same period netted R145 000.

(b) Recreational developments account for 1% of the improvements in the lake catchment and consist of accommodation; boating, sailing and angling facilities; and tourist amenities (such as zoological parks and game reserves). The commercial sector supporting this industry has invested R13.3m and generates a net annual profit of about R1.5m. Property developments at Schoemansville, Meerhof and Kosmos total R21.3m. Expenditure by recreational users amounted to R1.8m with some R13.3m invested in capital assets. The majority of the recreational expenditure was contributed by anglers.

(c) Domestic development has placed growing demands on water supply and sanitation facilities. Municipal wastewater treatment facilities treating effluents entering Hartbeespoort Dam have been estimated to cost from R2m (for conventional plants) to RIO.6m (for Advanced Wastewater Treatment [AWWT] plants) annually. At present only conventional plants are used, but upgrading of these plants is continuing as a result of the effluent phosphorus standards promulgated in 1980. Water treatment costs for potable water have been estimated to be about R136 000 p.a. (van Vuuren, de Wet & Cillie 1981).

Revenue and expenditure on goods and services relating to Hartbeespoort Dam are summarised in Table 3.3 for the period c. 1970.

Table 3.3. Income statement for use of Hartbeespoort Dam (c. 1970) (data from Hofmeyr 1978).

| Income (after tax) | R'000 | R'000 |
|---|---|---|
| AGRICULTURE | | |
| Sale of irrigation water | 145 | |
| Crop production | 12 000 | |
| Fish production | 65 | 12 210 |
| RECREATION | | |
| Service industry | 1 500 | |
| User expenditure | 1 800 | |
| Estimated annual capital investment* | 2 345 | 5 645 |
| TOTAL INCOME | | R17 855 |
| Expenditure | | |
| MUNICIPAL | | |
| Wastewater treatment | 2 000 | |
| Domestic water treatment | 136 | 2 136 |
| TOTAL EXPENDITURE | | R 2 136 |
| Net income | | R15 719 |

*calculated as depreciation (total capital investment ÷ 20 years).

Obviously, these figures are related to the rate of Usage of the lake's facilities and can be enhanced or reduced depending on the state of the lake. However, the net profit generated by lake-related industry would suggest that additional funds be expended to eliminate user-identified eutrophication-related problems (Table 3.4).

Table 3.4. User-identified problems (after Hofmeyr 1978).

| | |
|---|---|
| AGRICULTURAL USERS: | toxicity<br>reduction of nutrients by algae<br>non-potability |
| RIPARIAN OWNERS: | odours<br>excessive algal/macrophytic growth |
| ANGLERS/RECREATIONAL USERS: | algal mats<br>changing fish species<br>tastes/odours in fish catch<br>health hazards (bilharzia) |
| PROPERTY OWNERS: | aesthetic considerations<br>decreased property values |

# 4. METHODS

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## 4.1 LOCATION OF SAMPLING SITES

Figure 4.1 shows the locations of the sampling sites in and around Hartbeespoort Dam. Details of the parameters measured at each station are given in Table 4.1, which refers to the sections of this report giving details of the methods used.

Water samples were obtained at the surface and at five-meter intervals from the surface to bottom at the pelagic stations. Additional samples were obtained from 0.5 m 1, 2, 3, 4, 6, 8 and 13 m at station 1 (Fig. 4.1). Surface samples only were obtained from the fixed-depth shore stations (stations 2, 6, 7 and 8).

## 4.2 PHYSICAL LIMNOLOGICAL METHODS

### 4.2.1 General meteorological data

Meteorological data were supplied from three shore-based weather stations installed and operated by the Directorate of Water Affairs, Department of Environment Affairs. Wind speed, air temperature, humidity and evaporation measurements were made at two of these stations (Wl and W3 in Fig. 4.1) whilst only wind speed and direction were measured at station W2. Total incident solar radiation was measured with a Moll-Gorczynski solarimeter (Kipp and Zonen) at station W3.

Additional meteorological data were supplied by the Directorate of the Weather Bureau, Department of Transport Affairs, from meteorological stations at Brits (Hartbeespoort Agricultural Research Centre) and Pretoria.

### 4.2.2 General hydrological data

Hydrological data were supplied by the Directorate of Water Affairs from their gauging station network. Inflow volumes were supplied from weirs A2M12 (Crocodile River:daily) and A2M13 (Magalies River:weekly) (Fig. 4.1). Outflow volumes were measured daily at the dam wall (A2R01) and in the canals. Additional data on abstraction and seepage flows were provided by the Directorate in the form of monthly water balances.

### 4.2.3 Routine Physical Measurements

Temperature profiles at station 1 were determined using a Cole-Parmer model 8502-20 thermistor. At stations 3, 4 and 5 profiles were determined with a YSI model 57 thermistor. Temperature measurements were made at meter intervals from the surface. At the shore stations 2, 6, 7 and 8 temperature was measured with a standard laboratory mercury thermometer.

Water transparency was measured at stations 1, 3, 4 and 5 with a standard 25 cm black/white quadranted Secchi disc. At station 1, the attenuation of photosynthetically available radiation (PAR, 400-700 nm) was measured using a Lambda Instruments' quantum light meter, at

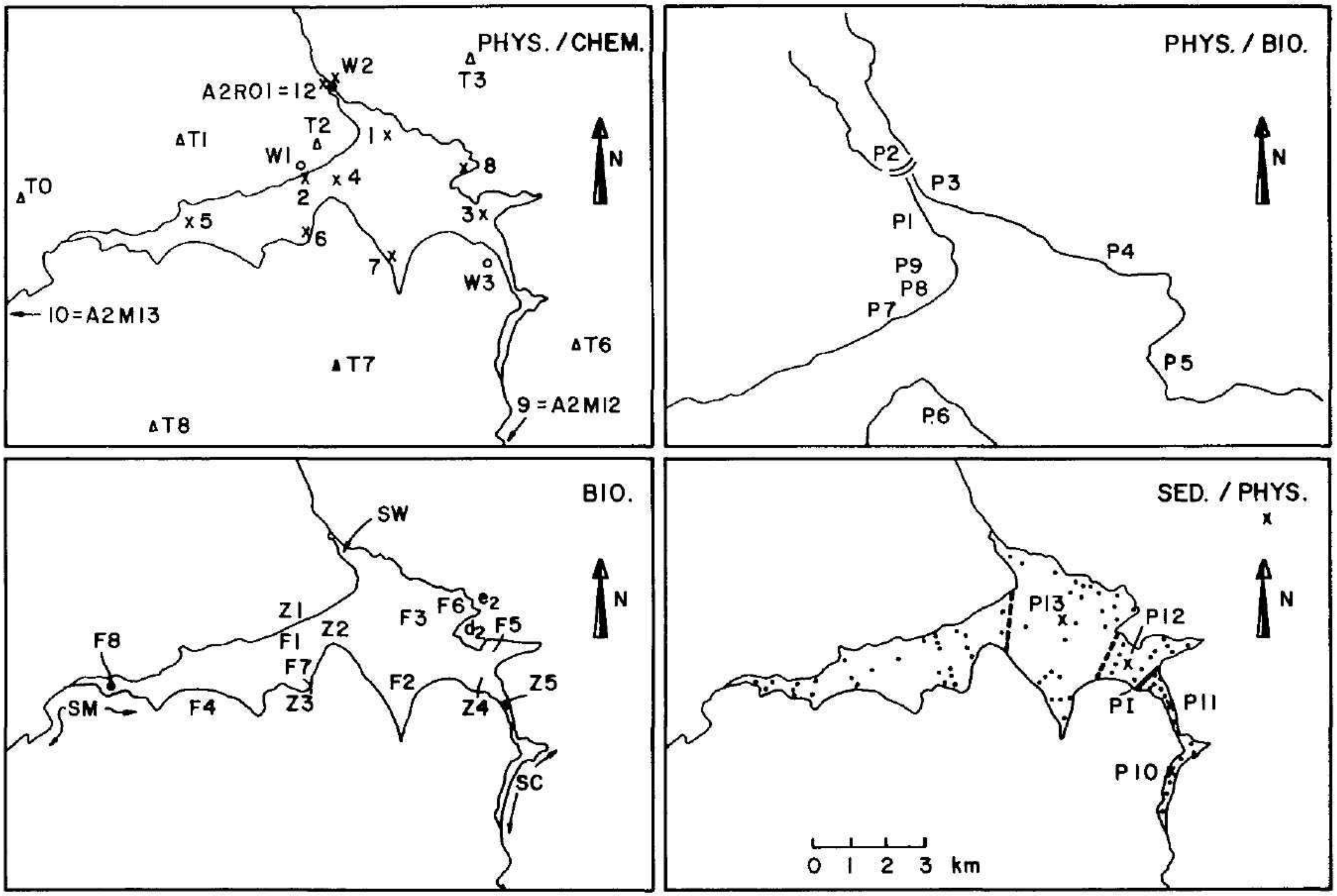


Figure 4.1. Physical (Phys.), chemical (Chem.), biological (Bio.)and sedimentological (Sed.) sampling points. See Table 4.1 for an index to stations and methodological descriptions.

Table 4.1. Sampling Notes on Fig. 4.1.

| Sta. No. | Description | refer to para. |
|---|---|---|
| TO - T8 | Trigonometric beacons used for position fixing | 4.2.4 |
| Wl - W3 | Weather stations Kosmos, Hartbeespoort Dam and Oberon respectively | 4.2.1 |
| 1 - 1 2 | Lake, shoreline and river sampling sites | |
| | - some routine physical measurements | 4.2.3 |
| | - full range routine chemical measurements | 4.3.1 |
| | - chlorophyll measurements | 4.4.2 |
| 9, 10-12 | Hydrological stations Crocodile R., Magalies R., canals and spillway respectively | |
| | - full range routine chemical measurements | 4. 3.1 |
| 1 | Routine and special physical measurements | 4.2.3 +<br>4.2.4 |
| | Routine and special chemical measurements | 4.3.1 +<br>4.3.2 |
| | Full range biological measurements | 4.4.1-5 |
| 2 | Routine chemical measurements | 4.3.1 |
| | P-fractionation and sediment chemistry | 4.3.2 +<br>4.3.3 |
| | Some biological measurements: bacteria, chlorophyll, zooplankton | 4.4.1<br>4.4.2<br>4.4.5 |
| | Temperature | 4.2.3 |
| 4 | Routine chemical measurements | 4.3.1 |
| | Special P-chemistry | 4.3.2 |
| | Temperature | 4.2.3 |
| 11 | Canal: chlorophyll | 4.4.2 |
| | Routine chemical measurements | 4.3.1 |
| | Hydrological measurements | 4.2.2 |
| SW, SC, SM | Scum formation areas | 4.4.2 +<br>4.3.2 |
| Fl - F8 | Fish population and feeding sampling sites | 4.4.7 +<br>4.4.8 |
| Zl - Z5 | Zoobenthos sampling sites | 4.4.6 |
| Dots (.) | Sediment sampling sites | 4.3.3 |
| PI - P13 | Survey points used for special physical measurements | 4.2.4 |

25 to 50 cm intervals, from the surface to the depth where 1% of the sub-surface value was recorded ($Z_{eu}$). Attenuation of the various components of PAR. was measured using glass colour filters, and the attenuation coefficients (e) were calculated using linear regression analysis (Robarts & Zohary 1984).

### 4.2.4 Specialised physical measurements

(a) Wind

Owing to the irregular terrain, winds measured at stations W1, W2 and W3 were insufficient to yield an accurate picture of wind conditions at the lake surface. Two anemometers (Theis model 4.3900.10 mechanical wind recorders) were mounted on a float and positioned wherever data on wind speed and direction at the lake surface were required.

(k) River flow

Measurements of river flow (>0.05 m s *) were made at points in the Crocodile River with an Ott type 10.152 current meter. Additional measurements were made using a drogue tracked tacheometrically with a sextant for a specified period of time.

(c) Lake currents

Lake currents were measured using drogues. The drogues were tracked using theodolites. Their positions were fixed at specified times by both vertical and horizontal angles from three positions (using various combinations of positions shown as P1 to P8 in Fig. 4.1). Wind-stresses on the staffs were taken into account in calculating current-speed and direction.

(d) Water mass identification

Turbidity (Hach Turbidimeter model 16800), conductivity (WTW model LF56) and temperature (Cole-Parmer model 8502-20) were measured in the Crocodile debouchement. Where turbidities in excess at 100 N.T.U. were detected, the suspension was diluted and the readings multiplied *pro rata.*

(e) Continuous temperature measurements

In order to evaluate seiche movements within the lake, a continuously recording thermistor chain (Anderaa model TR1) was deployed at station 1 during discrete periods in 1984.

(f) Hydrographical measurements

In addition to position fixing by theodolite, a marine hand-held magnetic bearing compass and a sextant were used. Surface currents were measured photogrammetrically in the vicinity of the dam wall (drift cards were photographed from the vantage point of station P3). Depth was measured with an echo-sounder (Kelvin Hughes).

(g) Computations

(I) Calculation of nutrient balance

The nutrient balances for Hartbeespoort Dam show the variation in quantities of five different nitrogen forms and three phosphorus forms within the lake. The balances are based on weekly data for lake level and nutrient concentration and compensate for the changing volume of water in each horizontal depth 'slice[1]. The total quantity of each nitrogen or phosphorus form in the lake was calculated as the integral:

$$T_1 = \int_{z_o}^{z_m} C_{lz} . A_z . dz \qquad [1]$$

where: T. = quantity of constituent 1 (tonnes) z

= surface of lake (depth = 0 m)

z = maximum depth of lake (m) m

$C_{12}$ = concentration of constituent 1 at depth z (kg $m^{-3}$)

A = area at depth z ($m^2$)

dz = depth interval (m)

(ii) Calculation of inflow and outflow loads_for ingut-output model.

This calculation utilizes chemical data for both the Crocodile (daily) and Magalies (weekly) rivers. Hydrological data were provided by the Department of Water Affairs (Section 4.2.2). The calculation of loads of each nutrient form used the formula recommended by Toerien & Walmsley (1978):

$$LC_n = \sum^{r} F.C_{ni} \qquad [2]$$

where: LC = load of constituent n (kg $a^{-1}$)

F. = total flow since previous sampling period ($m^3$)

C . = mean concentration of constituent n ni during sampling interval (kg $m^{3}$)

1,2 ... r = number of sampling periods during study

i = time index

### (iii) Calculation of predicted outflow loads and mass balance

The outflow of lake water from the impoundment at the dam wall consists of two portions: the release of compensation and irrigation water via the canals and drawdown via the radial sluices. Canal water is drawn off at a fixed depth in the impoundment, via a 3 m diameter port located 20 m below the full supply level. Water which is lost via the sluice gates is drawn from a depth of approximately 2 m . Therefore, outflow loads can be simulated by the summation of loads lost from the 2 m and 20 m depth intervals. A computer programme has therefore been designed to do this, taking into account the variation in lake depth and volume of each horizontal depth 'slice* on the date of sampling, and using data for outflow hydrology provided by the Department of Water Affairs.:

$$T_L = <(C1_2.V_S) + (C1_{20}.V_C)) \qquad [3]$$

where: $T_L$ = tonnage of constituent 1
$Cl_2$ = concentration of constituent 1 at a depth of 2 m (kg m $^{3}$)
$CI20$ = concentration of constituent 1 at a depth of 20 m (kg m~$^{3}$)
$V$ = volume of water lost through sluice gates (m )
$V_C$= volume of water lost via canals ($m^3$)

The summation of the predicted outflow load for each week can provide an estimate of the total predicted outflow for a particular period, e.g. one year.

Whole-lake nitrogen and phosphorus balances for Hartbeespoort Dam were also calculated using weekly data for lake levels and nutrient concentration, compensating for the changing volume of water in each depth 'slice'.

With the data on inflow and outflow nitrogen loads, plus the nitrogen content of the lake, the mass balance equation of Messer & Brezonik (1978) was used to calculate the quantities of nitrogen lost from the lake via sedimentation and denitrification.

$$N_{in} - N_{out} = N_{den} - \Delta N - N_{sed} \qquad [4]$$

$$N_{sed} = \frac{[N]sed}{[P]sed} \cdot (P_{in} - P_{out} - \Delta P) \qquad [5]$$

$$N_{den} = N_{in} - N_{out} - \Delta N - \frac{[N]sed}{[P]sed} \cdot (P_{in} - P_{out} - \Delta P) \qquad [6]$$

where: $N_{in}$ = inflow N load
$N_{out}$ = outflow N load
$N_{den}$ = quantity of N denitrified
$\Delta N$ = change in N content of lake
$N_{sed}$ = quantity of N lost to sediments
$[N]sed$ = total N concentration in sediments
$[P]sed$ = total P concentration in sediments
$P_{in}$ = inflow P load
$P_{out}$ = outflow P load
$\Delta P$ = change in P content of lake.

## 4.3 CHEMICAL LIMN0L0G1CAL METHODS

### 4.3.1 Routine chemical methods

Routine chemical analyses were carried out by the Analytical Services Division of the National Institute for Water Research using standard automated analytical techniques (NIWR 1974). These methods are summarised in Table 4.2. Samples were obtained with a 6 *£* opaque PVC Van Doom sampler and were processed within 12 h of collection. The sampling interval was 5 m from the surface to bottom except for oxygen (0, 0.5 m, 1, 2, 3, 4, 5, 6, 8, 10, 13 15, 20 25 and 30 m) and temperature (at metre intervals). Additional depth integrated samples were taken using a 5 m x 3.3 cm diameter hosepipe from the surface. Samples for total constituent analysis were homogenised using an Ultraturex tissue grinder or Branson model S125 sonifier.

Dissolved nutrient samples were filtered in the field through glass fibre filters (Sartorius SM134, Gelman A-E, or Millipore AP40) whilst samples for anion/cation analysis were filtered through Whatman No. 1 filters. Nutrient samples were preserved with 0.1 *m£* of a saturated mercuric chloride solution per 250 *xaZ* sample and were analysed within 48 h of collection. On return to the laboratory all samples were stored in 250 *m&* glass stoppered bottles at 4 °C .

### 4.3.2 Special chemical methods

#### (a) Phosphorus fractionation

Determination of the component fractions of soluble reactive phosphorus (SRP) followed the method of Downes & Paerl (1978) and used Sephadex G-25 coarse and G-10 gels in 40 cm columns (Pharmacia K26 and K16 columns respectively). The eluent passed directly through a Technicon Autoanalyzer I system and phosphorus fractions were determined using either a stannous chloride or ascorbic acid method (NIWR 1974). Surface, mid-depth and bottom samples were analysed from station 1 and surface samples from station 2.

### (b) Phosphorus turnover times

Phosphorus turnover times were determined using the $^{3}$ P method of Lean & Nalewajko (1979). Abiotic uptake of $^{3}$ P was separated from biotic uptake using 1% formalin AR (Tarapchak, Slavens & Maloney 1981).

### (c) Alkaline phosphatase activity

Alkaline phosphatase activity (APA) was determined spectrophotometrically using a p-nitrophenylphosphate substrate (Calbiochem-Behring SVR 869311; Fitzgerald & Nelson 1966).

### (d) Water samples from hyperscums

Water samples were obtained from *M-Ccroaysti-s* hyperscums (as defined by Zohary, in 1985) using pre-incubated dialysis bags containing distilled water. After a 24 h equilibration period, the contents of the dialysis bags were siphoned into 125 m£ glass stoppered bottles. Samples were analysed for dissolved oxygen, pH, nutrient concentrations, and $H_2S$.

## 4.3.3 Sediment chemistry

### (a) Phosphorus

Samples were obtained using a Van Veen grab and were homogenised on collection. Samples were stored undisturbed for 48 h at 6 °C before the overlying water was removed and the samples re-homogenised. After a further period of 12 h the remaining overlying water was siphoned off and filtered through 0,2 um filters (Millipore GSWP047) prior to analysis (this water was assumed to be analogous with sediment pore water as separated by centrifugation). Sub-samples of sediment were stored wet at 6 °C or dried at 60 °C . All analyses were based on wet-weight equivalents. Samples were analysed for water content (loss of mass on drying at 60 °*C)*, loss on ignition (at 500 °C), particle size distribution using the hydrometer method of Black (1965), organic carbon content using the Walkley-Black titration (Black 1965), pore water conductivity, pore water SRP, bioavailable sediment phosphorus using NTA extraction with a sediment: solution ratio of 0.1:50 (Golterman 1976), total phosphorus (TP) using potassium persulphate digestion (NIWR 1974), and 24 h equilibrium phosphorus (EP) level using a range of phosphorus concentrations from 0 to 1.0 mg 1 (EP was estimated as the concentration at which no uptake or release of phosphorus by the sediments occurred).

### (b) Nitrogen

Total Kjeldahl nitrogen content of sediment samples was measured after potassium persulphate digestion. Inorganic nitrogen fractions were measured in sediment pore^water extracted by centrifugation at 1 000 g for 10 min using the methods listed in Table 4.2.

- 20 -

Table 4.2. Analytical methods.

| Constituent | Method | Reference |
|---|---|---|
| pH, redox | Metrohm Herisau model E444 meter with a combined platinum electrode | |
| temperature | Cole Parmer 8502-20 thermistor or YSI model 57 thermistor | |
| (hydrogen sulphide) $H_2S$ | Chemetrics S10 kit | |
| total alkalinity | HC1 titration to pH 4.5 end point | APHA 1975 |
| dissolved oxygen | azide modification of Winkler titration | APHA 1975 |
| conductivity | potentiometrically | NIWR 1974 |
| calcium )<br>sodium )<br>magnesium )<br>potassium )<br>chloride )<br>iron )<br>manganese ) | atomic absorption spectophotometrically | NIWR 1974 |
| Kjeldahl N | persulphate digestion followed by $NH_4$ analysis | NIWR 1974 |
| ammonium ($NH_4$) | sodium hypochlorite method | NIWR 1974 |
| nitrate + nitrite ($N0_3$ + $N0_2$) | hydrazine reduction followed by $N0_2$ analysis | NIWR 1974 |
| nitrite ($N0_2$) | sulphanilamide method | NIWR 1974 |
| silica (Si) | molybdate method | NIWR 1974 |
| total P (TP) and total dissolved P (TDP) | persulphate digestion followed by SRP analysis | NIWR 1974 |
| soluble P (SRP) | ascorbic acid method | NIWR 1974 |
| dissolved organic carbon (DOC) | UV reduction method | NIWR 1974 |
| sulphate (S(\) | barium chloride method | NIWR 1974 |

#### (c) Experiments with intact sediment cores

A continuous-flow adaptor was used in experiments with intact sediment cores collected from exposed and submerged zones using a gravity corer (Twinch & Ashton, 1984).

#### (d) Enclosures

Circular galvanised steel enclosures (area =0.2 $m^2$) were used to isolate portions of sediment and overlying water for 24 h periods to eliminate the effect of mixing and sediment resuspension in shallow (0.25 m) water. Samples of lake water, and samples of water isolated from the sediment in polyethylene containers within the enclosures, were taken simultaneously for comparative purposes.

#### (e) Sedimentation

Sedimenting material was collected in sediment traps (sampling area =5.5 $cm^2$) suspended at various depths at stations 1, 3 and 4. Sedimented material was collected weekly and analysed for loss on ignition and total phosphorus content (as described in Section 4.3.3(a) above).

## 4.4 BIOLOGICAL LIMNOLOGICAL METHODS

### 4.4.1 Bacteriological analyses

#### (a) Bacterial numbers and activity

Water samples were collected simultaneously with samples for chemical analysis at station 1 between April 1981 and January 1983 and preserved in formalin for bacterial enumeration. Until July 1981, bacteria were counted using acridine orange and thereafter using DAPI (Robarts & Sephton 1981). Between December 1982 and April 1984, water samples were also used for heterotrophic uptake studies using a modification (Robarts & Sephton 1984) of the method of Hobbie & Crawford (1969).

#### (b) Nitrification

Nitrification rates were measured by serial chemical analysis of the different nitrogen fractions. Nitrification potential, assessed as the time taken to oxidize a known quantity of NHi+^N to $N0_2$-N (Cavari 1977), was measured at monthly intervals in both water samples and intact sediment cores.

#### (c) Denitrification

The 'acetylene block technique$^1$ (Yoshinari & Knowles 1976; Chan & Knowles 1979) was employed to measure denitrification rates in both water samples and intact sediment cores at monthly intervals. Extraction of dissolved gases was by multiple phase equilibration (Chan & Knowles 1979) and gas samples were analysed by gas chromatography.

### (d) Nitrogen fixation

Nitrogen fixation rates in both sediment and water samples were measured with the acetylene reduction technique (Ashton 1979, 1981), and gas samples were analysed gas chromatographically.

## A.4.2 Algological analyses

### (a) Chlorophyll

Chlorophyll a, collected on glass-fibre filters, was extracted in 90% boiling ethanol and was determined spectrophotometrically at 665 nm according to Nusch (1980).

### (b) Algal species composition

Algal species composition was determined from cell counts using the inverted microscope technique (Lund, Kipling & Le Cren 1958). Cell numbers were converted to volumes from linear dimensions using the formulae of Rott (1981).

### (c) Algal sedimentation

Sedimenting algae were collected in sediment traps as described in section 4.3.3(e) with the exception that traps contained 4% formalin for preservation of the algae. Algal cells were then enumerated as described in Section 4.4.2(b).

## 4.4.3 Primary production and respiration

Primary production was measured weekly using the $^{14}C$ light/dark bottle method as modified by Robarts (1984). To determine *in situ* respiration rates at weekly intervals between September 1982 and September 1983 the oxygen light/dark bottle technique Tschumi, Zbaren & Zbaren (1977) was used. Oxygen concentrations were determined spectrophotometrically (Mackereth, Heron & Tailing 1978) after fixation with Winkler reagents.

## 4.4.4 Assessment of algal toxicity

Scott, Barlow & Hauman (1981) have shown that *Microcystis* sp. toxicity in Hartbeespoort Dam is always linked with *Miovooystis aevuginosa* forma *aeruginosa,* which can be distinguished from the other common form, *Miovoaystis aevuginosa* forma *flos-aquae,* on the basis of colony form and cell size (Komarek 1958). The percentage composition of the two forms in lake samples was used as an index of potential toxicity.

## 4.4.5 Zooplankton

Zooplankton were sampled weekly at station 1 using a 60 ym mesh nylon net hauled vertically. Four samples were collected, of which one was preserved for enumeration of species composition and abundance (sub-sampled using the method of Allanson & Kerrich 1961) and three were used for dry biomass determination after removal of the algal component through floatation and differential centrifugation.

Zooplankton grazing rates were measured fortnightly at station 1 from January 1983 using an *in situ* method based on that of Haney (1971). Radio-labelled ($^{14}C$) *Chlorella* sp. was used as the food source. In addition, for each grazing experiment, data on species composition, abundance and total biomass were also collected. A diel survey measuring zooplankton grazing rates, abundance and biomass at six depths every four hours, was carried out during the 24 h period from 26 to 27 January 1984.

### 4.4.6 Zoobenthos population size and biomass

Zoobenthos samples were collected from five sites (Zl - Z5) using Van Veen grabs and perspex core tubes. Samples were poured through various mesh sized sieves and the retained animals were subsampled and counted using the method of Allanson & Kerrich (1961). Dry biomass was determined after oven drying at 60 °C for 24 h. Statistical analysis of 14 profundal and 15 littoral samples showed a contagious distribution of zoobenthos numbers and mass per unit surface area (Elliot 1977). Lake-wide densities and mass estimates with 90% confidence limits were determined by the method of Rendu (1978).

### 4.4.7 Fish population dynamics

Fish were sampled using a fleet of multifilament gill nets made up of 20 m panels of 35, 45, 57, 73, 93, and 120 mm stretch mesh nets, and two seine nets measuring 20 x 1.5 m of 13 mm stretch mesh and 100 x 3 m of 35 mm stretch mesh. A further «gill net panel with a stretch mesh size of 140 mm was added in June 1982. Monthly sampling was undertaken from January 1982 to December 1982; this was reduced to bimonthly sampling from December 1982 to April 1984. The fish caught, or a representative sub-sample, were examined for mass, standard and total length, sex and condition factor (assessed visually on a scale of 1 [inactive] to 7 [spent ]). Ovaries of active-ripe (4) or ripe (5) fish were collected for fecundity counts. The ages of fish were determined using scales *(0. mossambious)* , vertebrae *(C. aarpio)* and pectoral spines (C. *gariepinus),* Standing stock determinations were made for *0. mossambious* and C. *gariepinus* using the Petersen and Schnabel mark and recapture methods (Ricker 1975). The biomass of C. *oarpio* was determined by Leslie$^{1}$s method (Ricker 1975). Littoral poisoning with 5% WP rotenone applied in a concentration of 2 mg *2* in areas isolated with a seine net was undertaken at three stations during 1982 and 1984. Poisoned fish were collected over a 24 h period.

Angling data were provided by the Division of Nature Conservation, Transvaal Provincial Administration, and whole-lake angler counts were made by boat on five occasions (Cochrane 1985).

Adult total mortality rates were assessed from changes in catch per unit effort (CPUE) of gill net catches *(0. mossambious)* and angler catches (C. *oarpio)*. *C, gariepinus* rates were determined from length frequency data. Fishing mortality was computed from TPA angling returns. Young of the year mortality was calculated from catch curves (Ricker 1975) for *Oreoohromis* and from the Leslie matrix (Vaughan & Saila 1976) for the other two species.

### 4.4.8 Fish feeding

#### (a) Food sources of the major species

Having determined the optimal time for sampling over a 24 h period using Windell's index of fullness (Windell 1971), fish were obtained in the afternoon from littoral sites at intervals of 1 to 3 months. Fish < 10 cm total length were preserved whole in 4% formalin, whilst only the gut of the larger fish was preserved. Length, weight and sex of the fish were recorded, and the stomach contents were microscopically examined. The percentage abundance of 8 major food types was volumetrically determined, and the frequency of occurrence, as the number of times food items made up stomach contents in volumes in excess of 20% of total, was calculated to determine the food preferences of the fish species on a seasonal basis.

#### (b) Digestibility of food types in *O. mossambicus*

Fish confined in aquaria were fed a single food item (e.g. *Microcystis* sp. or *Daphnia* sp.). The protein, ash and phosphorus content of the food and of the faeces produced was determined and the assimilation efficiencies calculated using the methods of Edwards & Horn (1982) and Langner & Hendrix (1982).

#### (c) Urine phosphorus production in *O. mossambicus*

The change in SRP concentration of water in the aquaria (Section 5.4.18(e)) was determined as an estimate of urine phosphorus production (Langner & Hendrix 1982).

#### (d) Feeding, ingestion and defecation rates in *O. mossambicus*

Fish were collected at 2 h intervals over a 24 h period at station Fl (Fig. 4.1). Each sample size was 40 juveniles (6 - 10 cm total length). Half were preserved immediately in formalin and half were placed in aquaria filled with filtered lake water. These fish were held for 3 h before being preserved. All of the animals were weighed and measured. The gut plus contents was removed and dried to a constant weight at 80 °C for 60 h. As a control, the gut weight of a representative number of starved (for 1 week) fish was also determined, and this weight was subtracted from the combined gut and contents weight using a regression equation based on standard length. The weight of the gut contents was then standardised and rates of weight change in the gut contents, faecal production and ingestion were calculated according to Bowen (1976).

# 5. RESULTS AND DISCUSSION

## 5.1 PHYSICAL LIMNOLOGY 5.1.1

### Introduction to Lake Physics (a)

#### Definitions

Temperature, current, wind and light are the important physical factors in aquatic ecosystems. They are interlinked and play vital roles in the chemical and biotic functioning of lakes.

Radiation in lake-water is the residue of gains from direct sunlight, indirect radiation and long-wave radiation, and losses by reflection, upward scattering and long-wave emission (Wetzel 1975). The zone in which sufficient light exists for photosynthesis is known as the euphotic zone (Fig. 5.1). The rate at which light intensity declines with depth is defined by the extinction coefficient.

The effect of wind in mixing the irradiated surface water downward depends strongly on the surrounding terrain and land-use. Where the terrain is hilly and covered randomly with houses, trees and grass, winds become fragmented at ground-level. Large local variations arise in the stresses generated by these winds on the water. Thus surface- and return-currents may flow simultaneously in many different directions through a lake. They interact partly by entrainment (i.e., adjacent currents pulling each other) and shear (i.e., tearing of boundaries between adjacent currents). Because the energy transmitted by wind to water varies as the cube of wind-speed, local funnelling (in, for instance, a narrow channel) prejudices the authority of lake-wide wind-averages.

In winter, wind and gravity co-operate in mixing heavy, cooled surface-water down to the bottom of the lake. Thus temperature tends to be homogeneous over the whole volume. In summer, a lake is often divided into three superimposed horizontal density zones most easily separated by the associated temperature differences. (Fig. 5.1). The epilimnion contains the surface-water, which is heated by the sun and mixed by wind into the first few meters of the vertical profile. The metalimnion consists of one or more pycnoclines, i.e. steep persistent density-gradients which together form a barrier between the other two zones. Because of the non-linear relationship between temperature and density, the steepest temperature-gradients, or thermoclines, may be located appreciably below the pycnoclines in subtropical lakes (c.f. Denny 1972).

The hypolimnion is the lowest, coldest stratum, which often remains intact from spring through to autumn. While it is protected and undisturbed, it may lose oxygen and become anaerobic. It can be re-aerated by destratifying, or artificially mixing the strata. Even without destratification, nutrients are carried up and down through the zones by diffusion and convection. Heat is lost from the surface by evaporation and exchanged by conduction (sensible heat) and throughflow (advection).

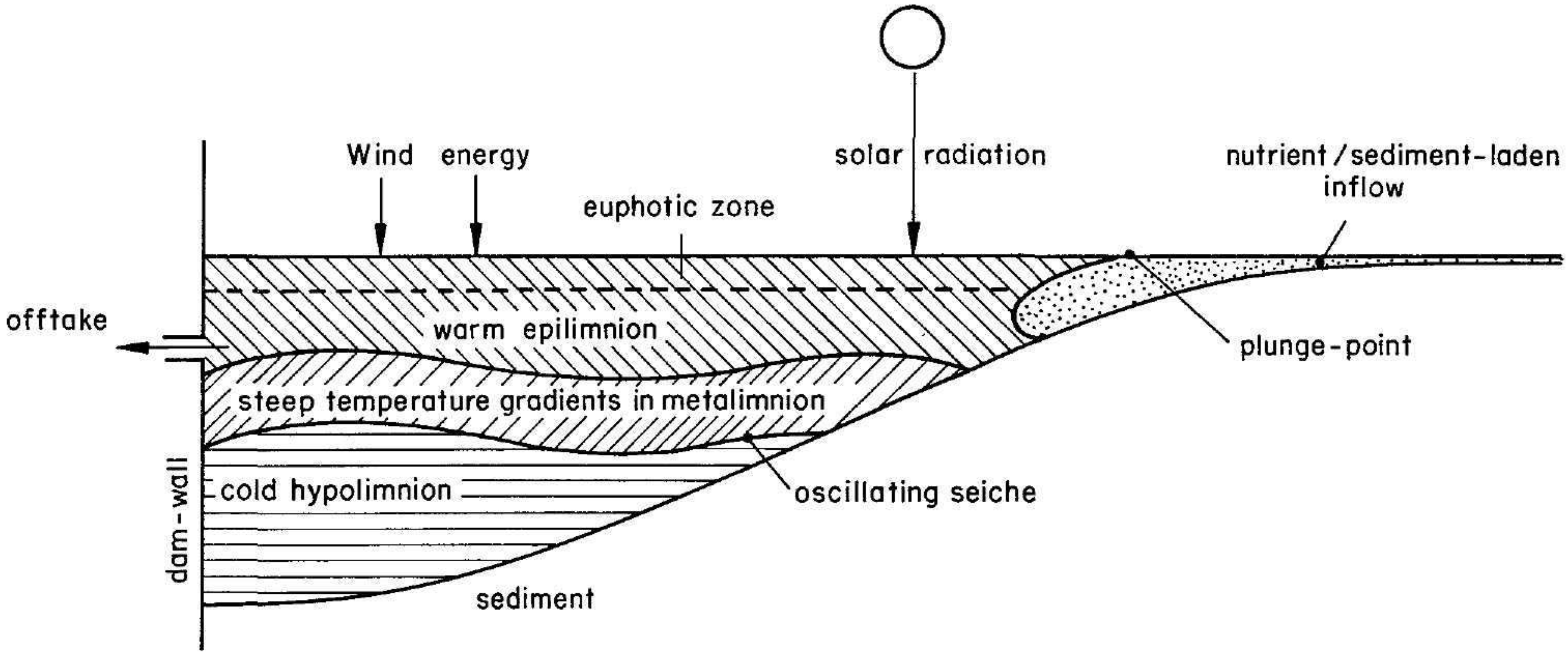


Figure 5.1. Conceptual depth/length profile of lake (Hely-Hutchinson, in preparation).

The thermal regime is often described by simple statistics. The work involved in raising temperatures from homogeneous winter values to the stratified levels is expressed as wind-work. Stability denotes the work needed to convert a stratified lake to constant temperature. Total heat is a measure of heat gained and lost over the seasons. The tropicality index, or lowest value of total heat divided by depth, is, as its name suggests, closely related to latitude. The likelihood of disturbing lateral constancy of stratification by inflow is defined by the internal Froude number, and the effect of wind on the epilimnion by the Wedderburn number. The event during which gravity and wind-stress first fully mix the lake-water during autumn is known as overturn.

In response to winds, surface-waves arise, the height and length of which depend on the fetch and depth of the relevant channel. In response to strong winds Langmuir cells are set up, in which the water near the surface rotates on axes parallel to the wind-direction. Where current-streams interfere, the energy may be converted to internal waves, that is oscillations of the thermal strata at what is known as the buoyancy frequency. In response to sufficiently violent changes in forcing functions (wind or current), travelling waves are set up which transmit disturbances across the lake, leading to oscillations whose frequencies depend on lake morphometry. These waves are known as seiches, which are sometimes large enough to raise hypolimnetic and metalimnetic water into the euphotic zone.

Lake ecosystems are products of hydrology. In semi-arid zones, such as the one under consideration, scattered storms produce inflow in the form of widely separated spates. The time distributions of currents, due to both influent streams and wind stress, strongly influence the erosion, transport and deposition of sediments. The period during which through-flowing water remains in a lake is known as residence time.

### (b) The functions of hydrodynamics

Hydrodynamics govern the chemical and biological properties of lakes by processes such as:

(i) mixing and separation of compounds by currents and thermal stratification;

(ii) aggregation, deposition and transport of compounds on sediments near influent streams and within the lake;

(iii) formation of density-currents by sediment transport;

(iv) isolation of the hypolimnion so that it becomes anoxic;

(v) modification of the biological environment by transport of nutrients;

(vi) transport of small, non-motile organisms, including maintaining certain phytoplankton in the euphotic zone;

(vii) provision of light energy for primary production.

The chemistry/biota affect the physics as follows:

(i) large accumulations of algae inhibit light penetration and may increase viscosity under exceptional circumstances;

(ii) dissolved solids modify densities.

### 5.1.2 Climate

#### (a) Solar radiation

Solar radiation falling on the catchment showed a distinct seasonal cycle. Lowest mean daily values of between 150 and 166 W m $^{2}$ were recorded in June of each of the three years (Fig. 5.2). Highest values for the years 1981/82 and 1982/83 (271 and 272 W m $^{2}$) were recorded in February and December, respectively. In summer 1983/84 solar radiation was higher than in previous years, with the summer maximum of 394 W m $^{2}$ recorded in January, this being indicative of the general summer increase. The increase was related to the reduced cloud and the associated reduced rainfall for this period.

#### (b) Temperature, rainfall and evaporation

Climatic variation in the catchment (Fig. 3.3) is illustrated in Figure 5.3. Temperature, rainfall and evaporation are roughly synchronous. Evaporation and maximum temperature tend to reduce under heavy rainfall» due to cloud cover. The evapotranspiration curve (Fig. 3.4) is also roughly synchronous with rainfall, yielding a relatively small surplus in summer which contributes to runoff.

#### (c) Wind

Wind speeds measured at Oberon (Fig. 3.1) were generally low and exhibited a typical seasonal cycle (Fig. 5.4). They were lowest between May and July each year (0.5 m s $^{1}$), and increased in August-September^ reaching peaks in October-November: 2.8 m s $^{1}$ in 1981, 2.4 m s $^{1}$ in 1982 and 2.2 m s $^{l}$ in 1983. Figure 5.5 shows the monthly distribution of hourly wind speeds blowing from 16 points of the compass during four selected months in 1981. In general, winds of lower velocity blew primarily from the E to SE and NW to W, while the strong winds blew from nearly all directions. In June 1981 wind speed did not exceed 2.8 m s $^{1}$ 91% of the time. This_ was typical of all winters. Wind speeds greater than 2.4 m s $^{1}$ are required for vertical mixing of the water column (Harris 1980; Scott, Myer, Stewart & Walther 1969). From these measurements, it is deduced that the weather conditions on the dam were predominantly calm. The influence on phytoplankton dynamics is explained in Section 5.4.

The relevence of the measurements is, however, prejudiced by the interaction of three factors. Firstly, the lake is dendritic, its shoreline development is 2.2 and no point lies more than 1.5 km from the shoreline. Secondly, it is surrounded by an irregular assortment of houses, trees and open veld (terrain

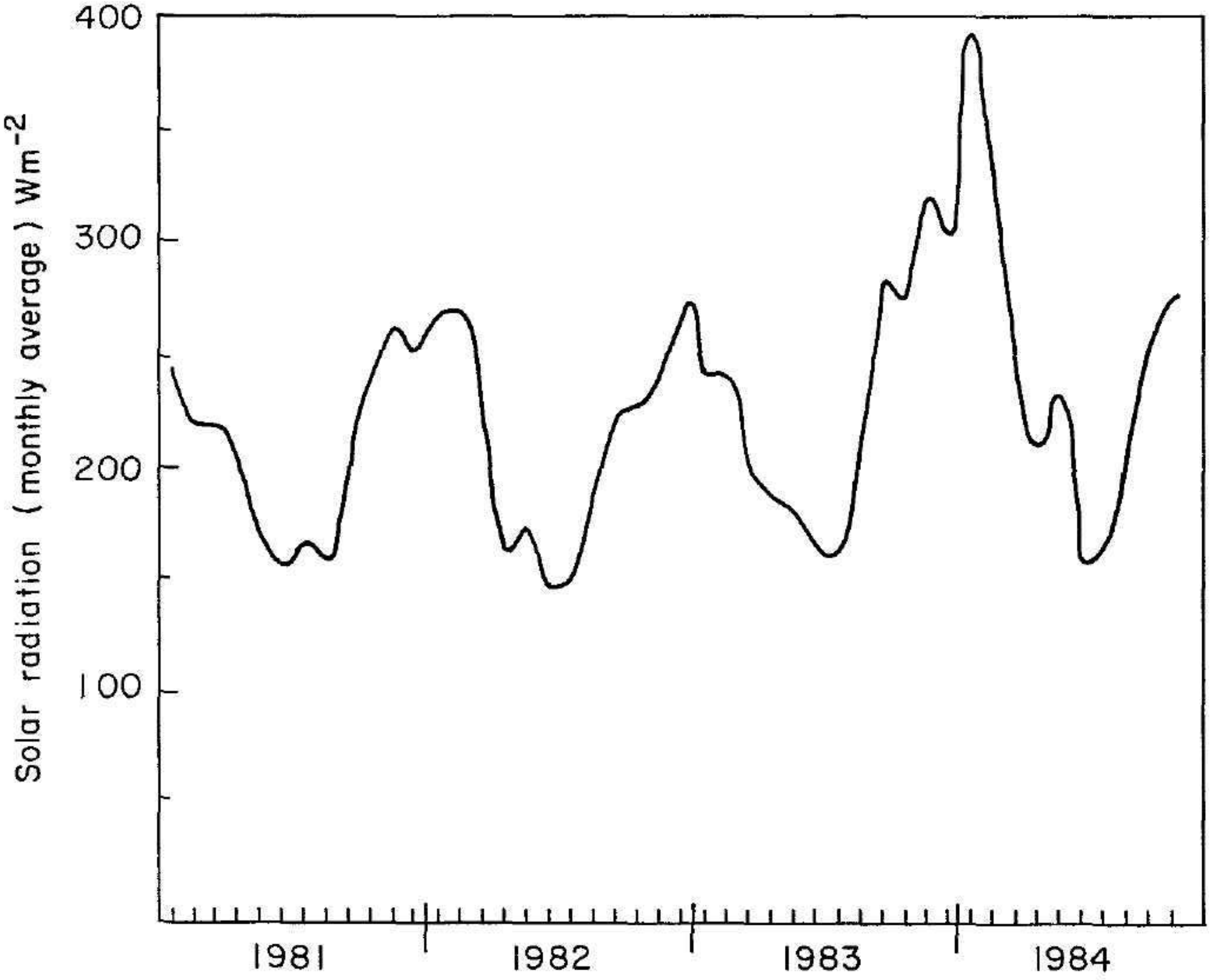


Figure 5.2. Monthly averaged solar radiation measured by the Department of Water Affairs at point W3 (Fig. 4.1).

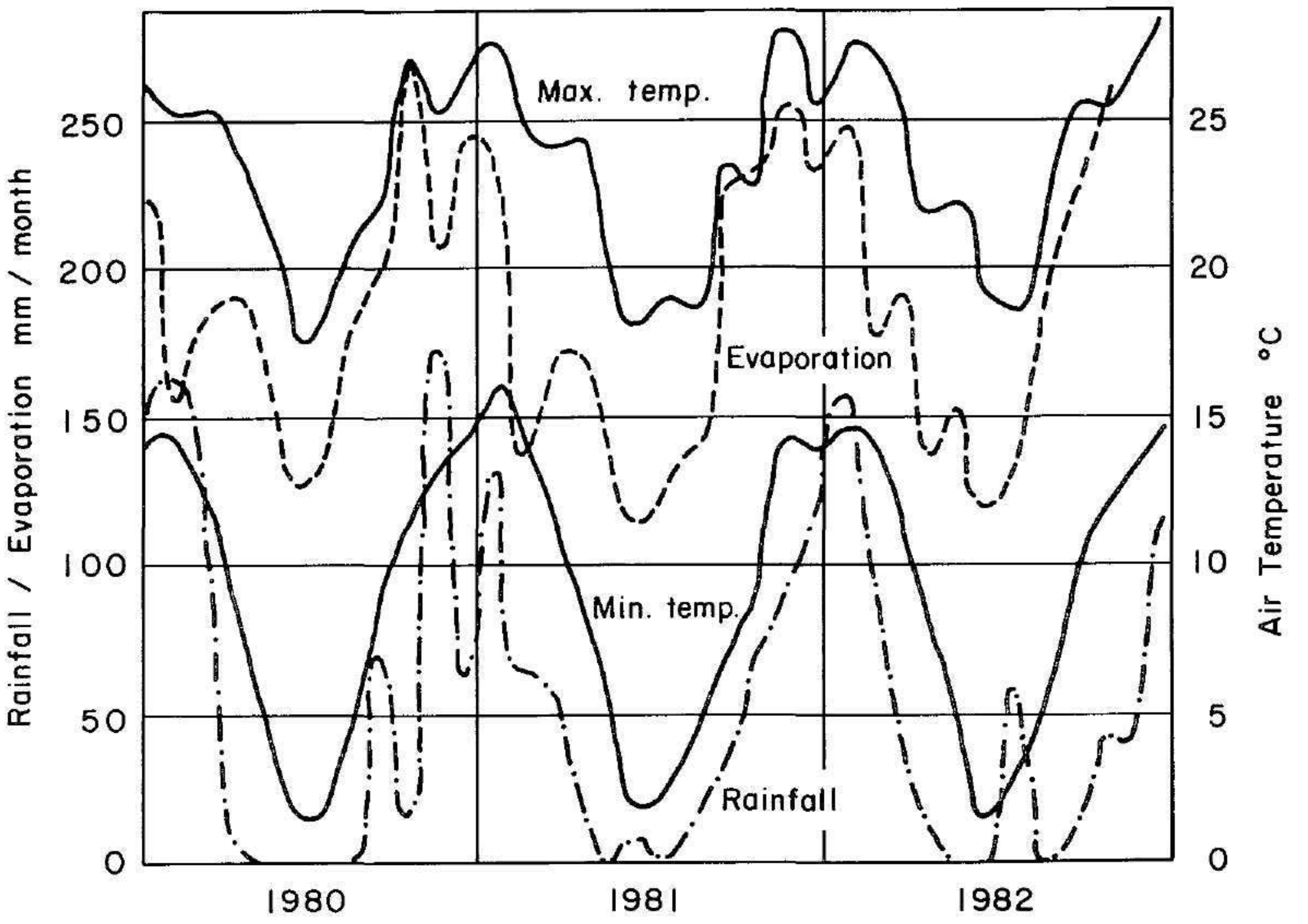


Figure 5.3. Monthly averaged climatic data for the catchment measured by the Department of transport at Irene (Fig. 3.3).

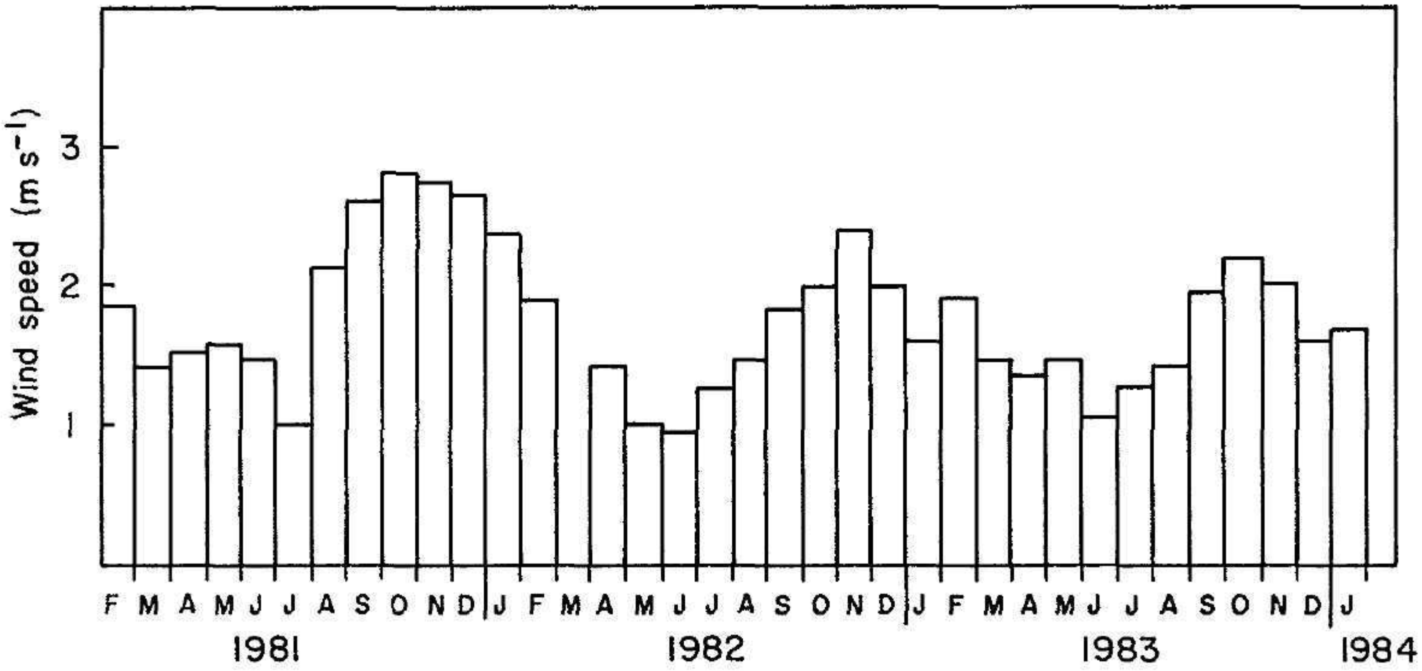


Figure 5.4. Mean monthly wind speed at point W3 (Fig. 4.1) (Zohary, in preparation).

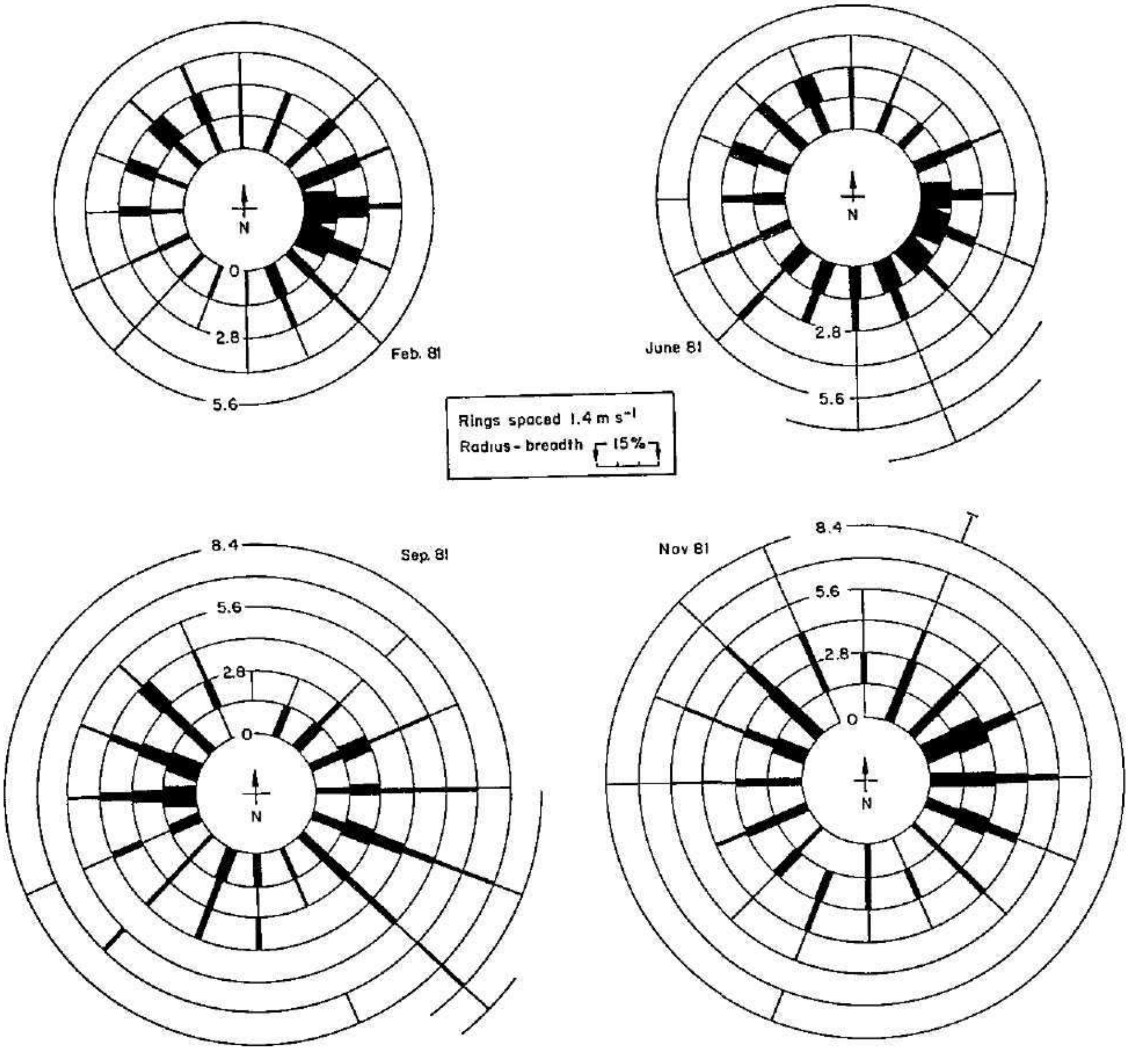


Figure 5.5. Representative monthly wind roses showing the distribution of wind speeds in 16 major directions, at point W3 (Fig. 4.1). Wind speed is indicated by the distance from the base of the circle. Radius breadth is proportional to the time spent at a specified speed and direction.

categories 1 to 3 - Newberry & Eaton 1974). Thirdly, the slope of the surrounding land varies from horizontal to vertical. Thus it is difficult to estimate wind speeds on the lake from on shore, even with the aid of a physical model (Waldeck, private communication, 1982).

This uncertainty was confirmed by correlations between wind measured with anemometers located in different areas and is discussed under section 5.1.7(c). Wind speeds measured on the dam wall could be 2-3 times those at Oberon. The main thermocline (Section 5.1.5(f)) may thus be established and deepened principally by winds blowing down the dam wall cut (Fig. 3.1).

### 5.1.3 Underwater light-climate

#### (a) Vertical extinction

The mean vertical extinction coefficients of the wavelength ranges passed by the filters used are shown in Table 5.1. Blue light (443 nm) was rapidly attenuated and the depth at which 1% of the sub-surface value remained ranged between 0.3 and 4.3 m (x = 2.3 m). Although there was no significant difference between the extinction coefficients for the blue-green, green and red wavebands, green light (550 nm) always penetrated the furthest (the attenuation coefficient of green light is referred to as e in the text). The level at__ which 1% of green light remained ranged between 0.5 and 9.5 m (x = 4.2 m).

#### (b) Euphotic zone

The euphotic zone depth (Z ) (the depth at which 1% of photosynthetically available radiation remained) ranged between 0.9 and 7.6 m (x = 4.3) in 1981/82 (Robarts 1984), between 0.6 and 6.4 m (x 3.7) in 1982/83 (Robarts & Zohary 1984) and between 0.45 and 8.4 m (x = 3.9) in 1983/84 (Fig. 5.58). These data indicate an unstable light regime for the phytoplankton population. During the three year study, light conditions in the lake were usually best in August-September and worst in January-February and coincided with the wax and wane of the algal population (see Section 5.4.7).

Table 5.1. Mean vertical extinction coefficients (e, In m ± 95% CL) obtained with optical filters (mid-points of wavelength ranges in parenthesis) used in Hartbeespoort Dam during 1981-84.

| Filter | e |
|---|---|
| Blue (443 nm) | 2.540 (± 0.292, n = 145) |
| Blue-green (520 nm) | 1.531 (± 0.206, n » 144) |
| Green (550 nm) | 1.397 (± 0.186, n = 152) |
| Red (670 nm) | 1.555 (± 0.212, n = 144) |

## 5-1-4 Hydrology and morphometry

The period 1980-1984 was characterised by a prolonged drought, the most severe in the Vaal catchment since records were first kept in 1906 (Alexander 1984). Rainfall, flow and evaporation in the Hartbeespoort Dam catchment during the relevant hydrological years (October-September) was as follows:

Table 5.2. Hydrological statistics from the Directorate of Water Affairs and Department of Transport.

| Hydrological year | rainfall (mm/year) | inflow ($m^3x10^6$) | outflow ($m^3x10^6$) | evaporation ($m^3x10^6$) |
|---|---|---|---|---|
| Mean | 695 | 168 | 153 | 18 |
| 1980-1981 | 696 | 234 | 212 | 26 |
| 1981-1982 | 543 | 144 | 186 | 25 |
| 1982-1983 | 520 | 99 | 166 | 14 |
| 1983-1984 | 676 | 121 | 99 | 14 |

[1] Mean annual rainfall' was averaged in time from 1916 to 1984, and flow and evaporation from 1923 to 1984. Rainfall was averaged in space from data taken at four stations, one close to the dam-wall and the others near the Jukskei, Hennops and Magalies headwaters.

Thus, for two years, rainfall was well below average. Inflow fell dramatically. Outflow was curtailed after three years to compensate. Evaporation decreased due to the reduced lake-area, in spite of reduced cloud cover. The extraction rate for irrigation above the lake was roughly constant at ca. 54 x 10 $m^3$ year. The drought has meant that the stored volume has been reduced to generally below 40% of its maximum value, the area to 10.5 x 10 $m^3$ as opposed to 20 (maximum) and the level by 7m from 1 162 to 1 155 m.a.s.l. (Fig. 3.2).

Four consequences of the reduced volume are: a smaller space for the biota to live; the conveyance of nutrients further into the lake; earlier overturn and a larger proportion of the lake-volume in the vicinity of the raft (station 1 - Fig. 4.1).

### Heat related statistics

### (a) Data

Simple physical, chemical and biological statistics are often taken as representative of whole lakes, and are cross-correlated. For Hartbeespoort Dam, the chosen physical statistics are diffusion, buoyancy-frequency, pycnoclines, stability, wind-work, total-heat, residence-time, Wedderburn number and thermocline. Measurements at station 1 (Fig. 4.1) were taken as representative of the lake, as it was situated at the deepest part and on the edge of the main basin.

Care was taken to simplify interactions between functions referred to the surface (for instance wind and radiation) and those referred to the bottom (for instance off-takes, some currents and resident chemicals) by choosing a hydrological year during which the level remained essentially constant. Thus, the summer of hydrological year 1981-82 was chosen as representative (Fig. 5.6) in deriving results discussed in Sections 5.1.5 (b, c, d, and g).

### (b) Vertical transport

Turbulent diffusion is one of the principal mechanisms of fluid transport (Robarts & Ward 1978). It tends to be slowest over the metalimnion (Fig. 5.1), the depth-range over which density gradients, and thus also buoyancy frequencies are highest, and diffusion coefficients lowest. In the total transport process, the breakage of pycnoclines, due either to shear stresses or motion across the littoral zone, would be in systematic series with turbulent diffusion in the hypolimnion.

Diffusion is intrinsically a passive process. In simple terms, given insulation at the boundaries of a medium and maintenance of fixed potentials (temperatures) at extremes, it leads to a constant gradient across the medium after infinite time has elapsed. Thus, in assessing values of diffusion from temperature profiles, the effects of forcing functions (external influences such as wind and radiation) should be eliminated (Jassby & Powell 1975). No direct method of eliminating the effects of forcing functions from the data set has yet been developed. Diffusion is, however, such a significant quantity that it it worthwhile to make the best possible estimate under practical conditions.

The profile for the summer of 1981-82 (Fig. 5.7) is smoothed, as shown in Figure 5.8, in order to eliminate computational instability due to seiches. In using the smoothed data, it was assumed that wind directly affected only the epilimnion, and that, because the euphotic zone ended generally above the pycnocline, temperature measurements in the metalimnion and hypolimnion could yield usable first-approximations to diffusivity in both these zones.

Diffusion coefficients for overlapping 5 m depth bands and monthly Intervals were averaged over the summer of 1981-82. The locus is displayed in Figure 5.9. The depth at which the minimum value occured provided an estimate of the mean position of the pycnocline. This corresponded closely to the oxycline shown in Figure 5.29, and may have been mainly responsible for its existence. The mean position of the pynocline also corresponded closely to the thermocline depth defined for an 8 km fetch in a study of continental, LEWG (Lake Ecosystem Working Group, University of Toronto) and Japanese Lakes (Shuter *et at** 1983).

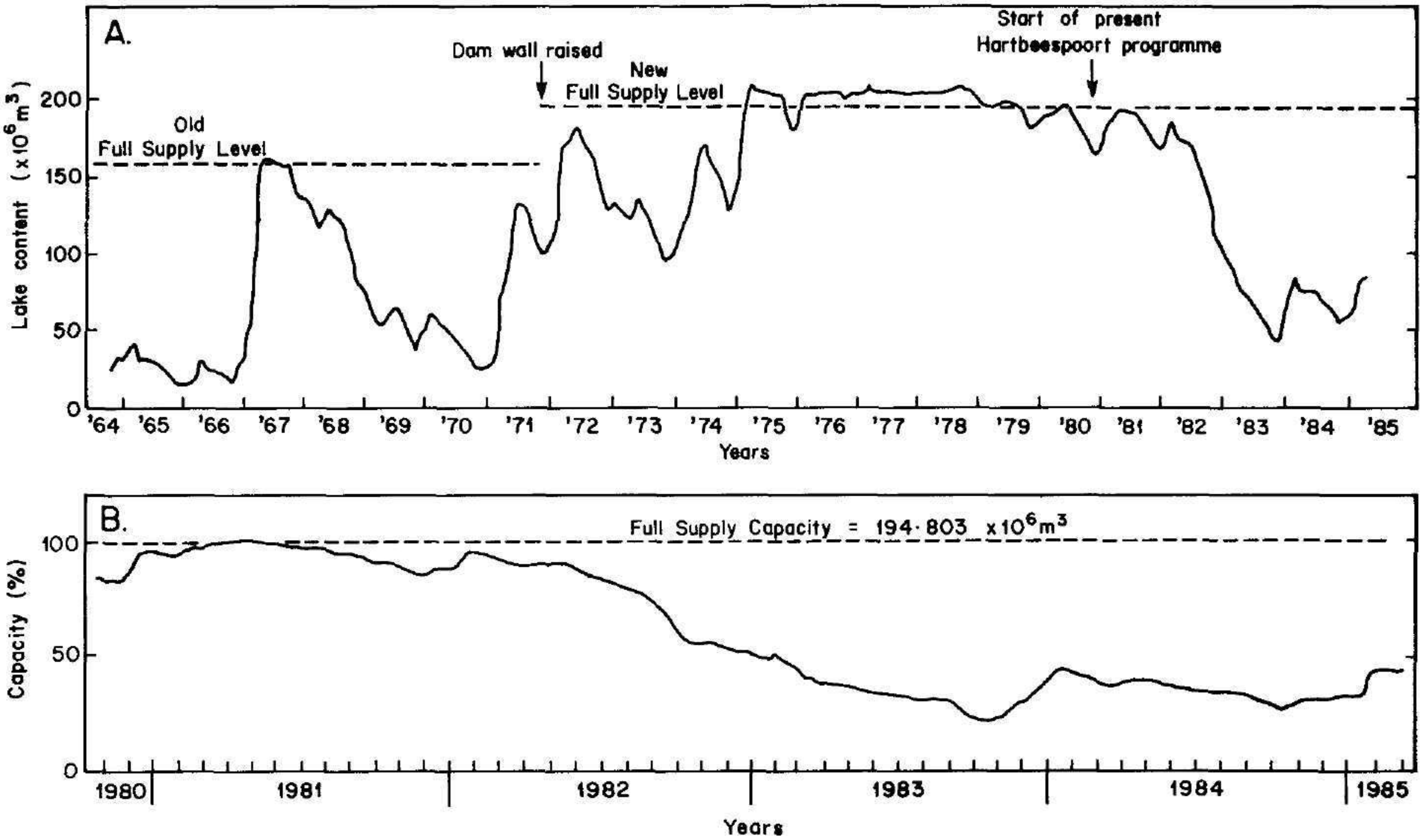


Figure 5.6 A. Variation in the volume of Hartbeespoort Dam, based on monthly values from October 1964 to March 1985. The raised dam wall in 1971 and initiation of the present Hartbeespoort programme are indicated.
B. Lake volume during the present study, showing the effects of the drought.

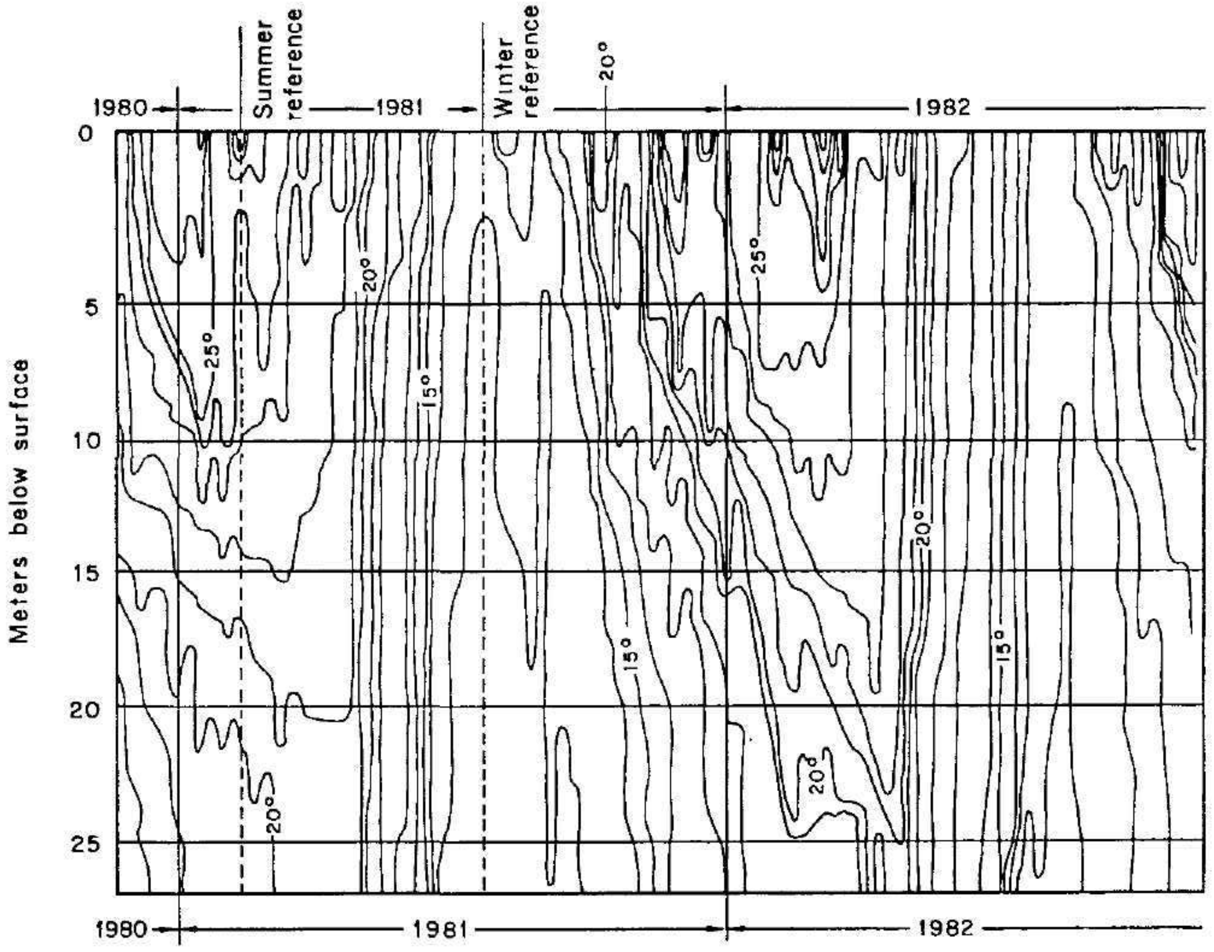


Figure 5.7. Isotherms in Hartbeespoort Dam measured during hydrological years 1980-1982

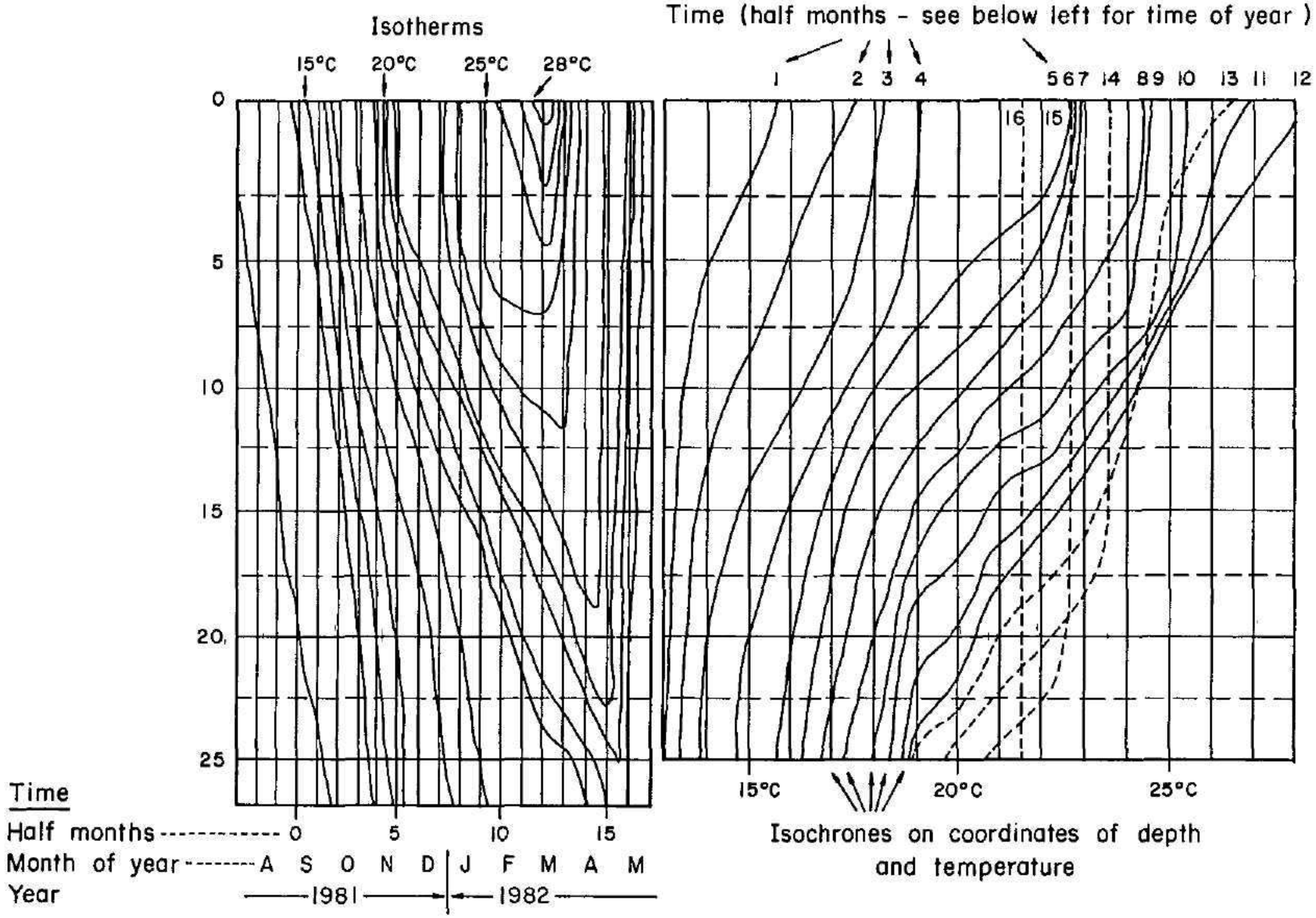


Figure 5.8. Smoothed isotherms and isochrones for the summer of 1981-1982. (Hely-Hutchinson, in preparation).

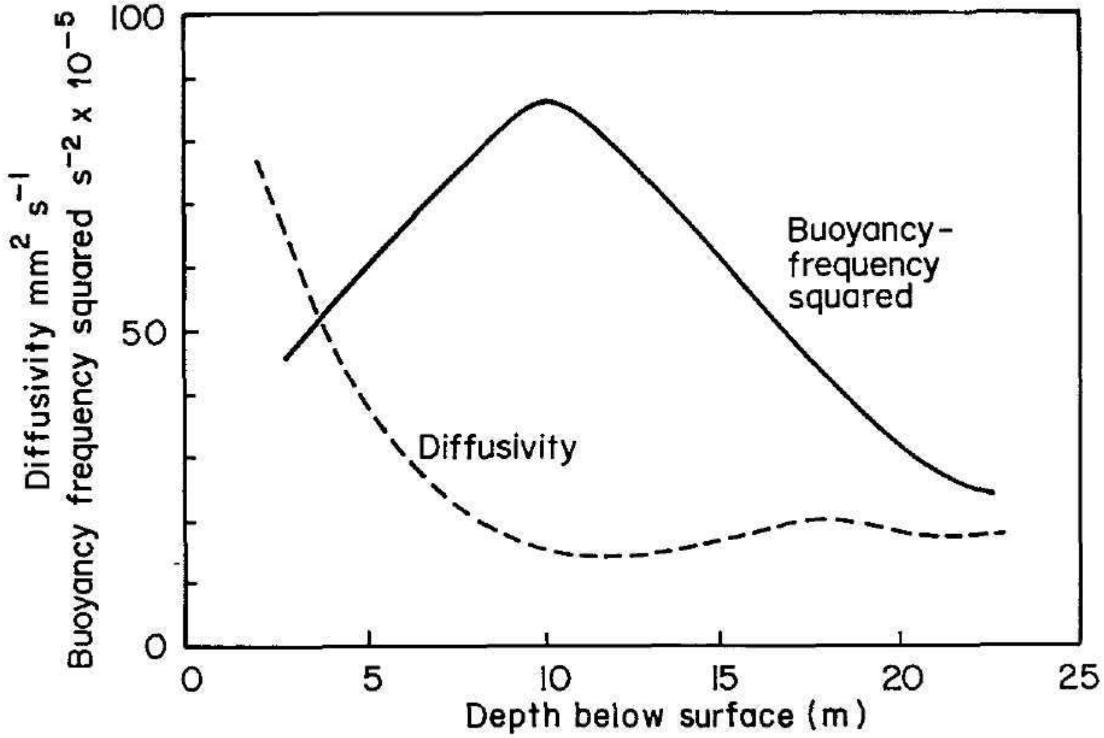


Figure 5.9. Mean diffusivity and buoyancy-frequency squared for the summer of 1981-1982, plotted against depth below surface, using data smoothed as in Figure 5.8. (Hely-Hutchinson, in preparation).

A comparison of values related to diffusivity in seven lakes (Table 5.3) shows that shallower, colder, high latitude lakes have broader ranges of buoyancy-frequency and summer temperature and lower ranges of diffusivity than do deeper, warmer, low-latitude lakes.

As will be explained in Section 5.1.7(c), it was estimated that seiches of mean amplitude 5 m occurred over 15% of the summer season. Taking into account the bottom area swept by the metalimnion, the vertical depths of the meta- and hypolimnion and the contact area between them, it was calculated that such seiches would lead to a 12% increase in mean diffusion over the summer season.

### (°) Buoyancy-frequency

The buoyancy, or Brunt-Vaissala frequency, serves three main functions. Firstly, it is used as an analog of the density gradient. Secondly, it defines the frequencies of internal waves, which are considered in detail in Section 5.1.7(d). Thirdly, it can be used for checking estimates of diffusivity.

Mean values for the summer of 1981-82 were calculated. The maximum value was an alternative estimate of the mean position of the pycnocline (Fig. 5.9). It was close to that of the minimum value of diffusivity (Section 5.1.5(b)).

### (d) Mean dissipation

In order to check the precision of estimates of diffusion and the buoyancy frequency, they are included in an estimate of dissipation, which is checked against two other related estimates.

In Figure 5.10, the mean product of diffusivity and the buoyancy frequency squared is shown as 0.0129 $mm^2$ s $^3$. The goodness of fit of the data to the curve (g.o.f. = 1 - SSq /SSq ) is 0.74. Thus the mean dissipation of the water mass is calculated to be 0.026 $mm^2$ s$\sim^3$ (Imberger 1982). Given a lake volume of 175 $Mm^3$, this means a power consumption of 4._5 kW. Alternatively, assuming a mean wind speed of 2.5 m s in summer (Section 5.1.2(c)) and a mean corresponding surface flow of 2% of this (Section 5.1.6(a)), the dissipation is 3.2 kW. Thirdly, given wind work of 1.344 k N m generated (at maximum rate) over four months (Section 5.1.5(e)), the dissipation was calculated to be 2.6 kW. The standard deviation of these three estimates is 28% of the mean, which gives an idea of their precision.

### (e) Stability, wind work and whole lake heat content

Weekly values for the Schmidt stability index (Symons 1969), Birgean wind work (Idso 1973) and whole lake heat content (Wetzel & Likens 1979) are shown in Figure 5.11.

In Hartbeespoort Dam, maximum stability values were recorded in mid-summer (January) when thermal stratification was greatest. Similarly, minimum stability values were recorded from after overturn (March - April) until the onset of stratification

Table 5.3. Values associated with diffusivity for seven lakes

| Lake | Reference | Latitude ° | Altitude m.a.s.l. | Zm m | Summer temp range °C | $N^2$ $-\log_{10} S^{-2}$ | Diffusivity $mms^2s^{-1}$ |
|---|---|---|---|---|---|---|---|
| Augher | Rippey 1983 | 54N | 0 | 5.5 | 8 - 18 | 2.5 - 4.7 | 0.5 - 10 |
| Grevelingen | de Bruijn *et al.* 1980 | 52N | 0 | 4.2 | 7 - 18 | 1 - 6 | 0.1 - 10 |
| Castle | Jassby & Powell 1975 | 41N | 1706 | 7.4 | 4 - 20 | 2.5 - 6 | 0.7 - 5.0 |
| Valencia | Lewis 1983 | 10N | 420 | 19.0 | 26 - 28 | - | 9 - 23 |
| Lanao | Lewis 1982 | 8N | 702 | 60.0 | 24 - 26 | - | 22 - 200 |
| McIlwaine | Robarts & Ward 1978 | 17S | 1368 | 9.4 | 22 - 24 | 3.5 - 4.3 | 16 - 50 |
| Hartbeespoort | This study 1985 | 26S | 1162 | 9.6 | 20 - 28 | 3 - 4 | 15 - 70 |

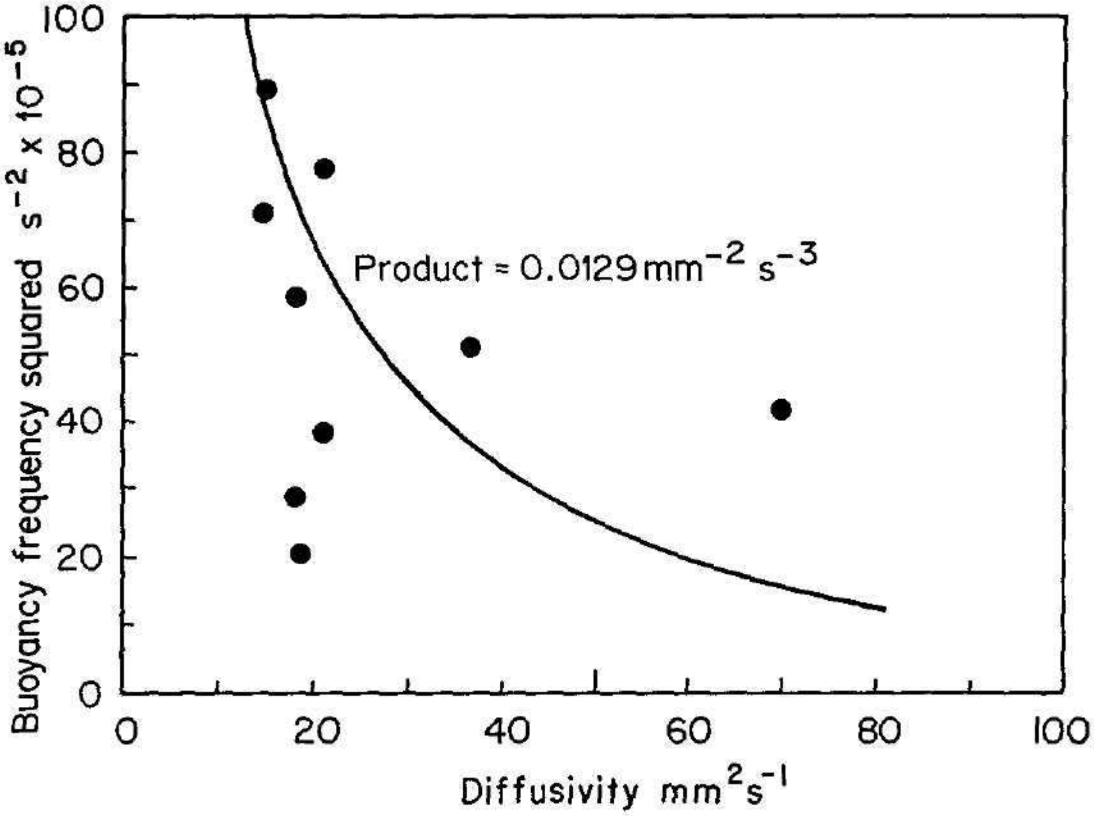


Figure 5.10. Buoyancy-frequency squared plotted against diffusivity (data as in Fig. 5.9) showing locus of mean product. (Hely-Hutchinson, in preparation).

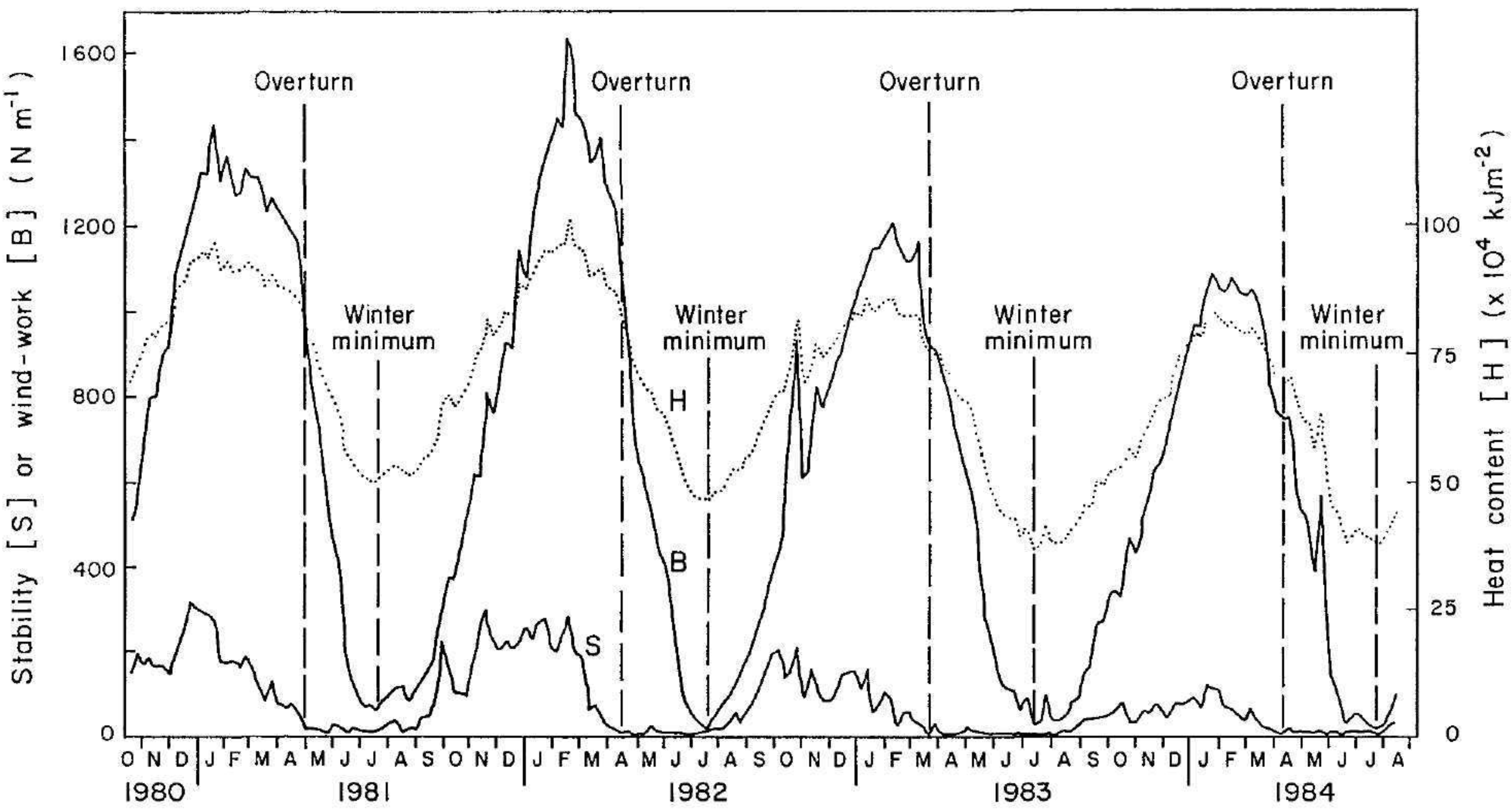


Figure 5.11. Variation in weekly values of Schmidt's stability index (S), Birgean wind-work (B) and whole lake heat content (H) at Hartbeespoort Dam. The dates of lake overturn and winter minimum heat content are also indicated. (Ashton, in preparation).

(August - September) during isothermal conditions. Some climatic events (e.g., storms, strong winds) decreased stability by increasing vertical mixing, while periods of calm, hot weather reduced vertical mixing and thus increased stability values.

The Birgean wind work statistic (B) indicates the amount of work that must be applied to the surface of the lake to distribute the summer heat income. Thus, the magnitude of B is dependent on the total heat content of the lake. This was clearly demonstrated by Hartbeespoort Dam, where B remained high at overturn, after which values for B and total heat content (H) declined rapidly as the lake continued to mix and lose sensible heat to the atmosphere. During each winter, minimum values for wind work and total heat were recorded. The lack of a single maximum point in summer was due to the local effects of climatic factors.

The values for lake stability (S) and B are influenced by the morphometry of the lake basin, local climatic factors and the latitude_of the lake. The highest values for S and B (307 and 1638 N m *, respectively) recorded at Hartbeespoort Dam were unexpected in view of the shallow mean depth (9.4 m), but did not persist for longer than one week.

The total heat budget of Hartbeespoort Dam, calculated as the difference between maximum (summer) and minimum (winter) values for heat content per unit area (Fig. 5.11), ranged between 0.45 (1983-84) and 0.55 G J $m^{2}$ (1981-82). These values are somewhat low, but are comparable to those of other lakes of similar size (Hutchinson 1957). With the progressive drop in lake volume during the study, successive summer and winter values for heat content per unit area were progressively lower and the total heat content of the lake also decreased. The tropicality index, at 5.2 $MJ.m^{3}$, fits into the series displayed, in order of latitude, by Allanson and Jackson (1983).

The progressive drop in lake level also gave rise to lower summer stability values than those recorded when the lake was full (1980-81) and reduced the quantity of wind work required to mix the lake. This progressive decrease in summer stability (resistance to mixing) was such that the low wind speeds normally recorded during February and March each year were sufficient to permit complete mixing (turnover) to occur on progressively earlier dates (Fig. 5.11). This was confirmed by examination of the chemical data which demonstrated the presence of aerobic conditions throughout the water column (see Fig. 5.29). Therefore, the stability index (S) provided a good indication as to when the lake was likely to turn over.

### (f) Thermocline

The depth locus of the main thermocline (Fig. 5.12) was calculated from the temperature profiles in terms of the points of inflection of cubic refinements (pooled g.o.f. was >0.96; for stratified periods only, >0.99; 24 readings during unstratified periods were rejected). The annual cycle could be explained by a cisoid of one year periodicity (pooled g.o.f. was >0.9; for stratified periods only, >0.98). In spite of variations in radiation (Section 5.1.2 (a)) and wind speed (Section 5.1.2(c))

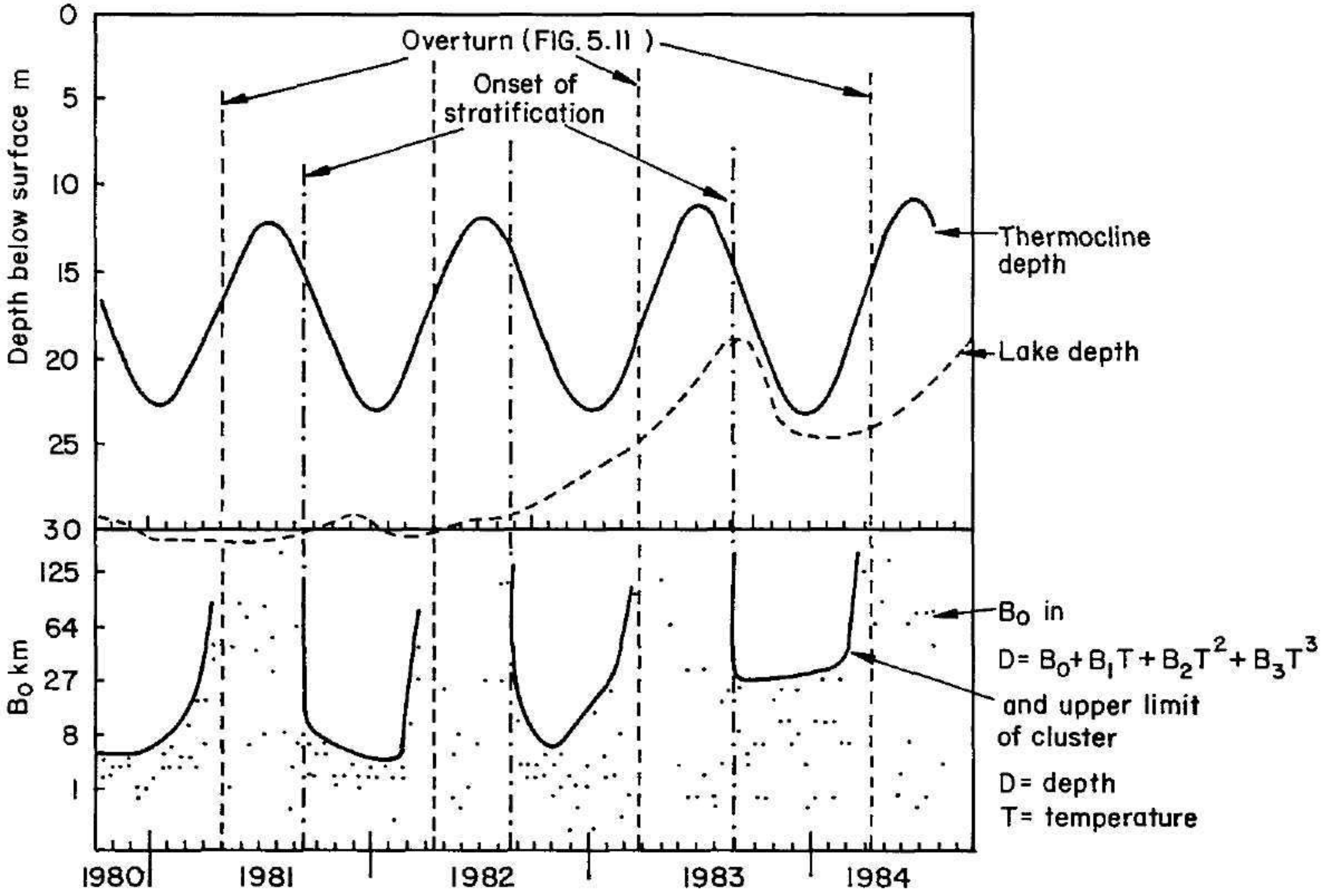


Figure 5.12. Depth below surface of main thermocline, illustrating annual constancy of cycle, regardless of changes in lake depth. Value of zero-order coefficient of cubic refinement, showing correlation with stratification. (Hely-Hutchinson, in preparation).

over 1980-1984, the amplitude of the cisoid remained sensibly constant at 10 m . The degree of stability (Section 5.1.5 (e)) correlated well with the depth of the lake below the thermocline, and overturn could be forecast by the onset of large changes in the parameters of the equation.

The difference between the mean depths of the pycnocline registered in Figure 5.9 (10 m) and the thermocline shown in Figure 5.12 (17.5 m) can be explained by closer examination of individual plots of temperature and density against depth. In most of them there existed, together with the main thermocline which deepened smoothly over the season, a sharply defined gradient at around 9 m depth. Although the temperature at that depth varied over the season, the gradient was virtually stationary, and thus predominated in the time averaging which produced the curves in Figure 5.9.

In theory, the main thermocline shown in Figure 5.12 could not have been deepened directly by wind mixing across another persistent thermocline. Deepening might, however, have been due to convection at the shore line. Breakage of thermoclines at shore-lines due to seiche action is discussed in Section 5.1.7(c). By the time the temperature of the rotated water had equalised at the raft, oxygen could have been removed from the down-driven epilimnetic water by decay and added to the up-driven hypolimnetic water by wind. A similar convective mechanism is postulated in Section 5.1.6.

Deepening due to inflow was discounted, as temperatures were generally lower than those of the lake. Outflow was a possible agent, as it was taken from the lower (colder) strata.

As measurements were taken at a fairly constant time of day, and the upper thermocline was not detected on each occasion, temporary, diurnal stratification could not be discounted. The effect would correspond to a breeze thermocline, which is differentiated from a larger, storm thermocline by Stauffer (1982). In its presence, daily averaged mixing would have been continuous and the oxycline would have represented merely the dynamic balance point between aeration due to wind and de-oxygenation due to biological oxygen demand. Correspondence with the mean breeze thermocline would thus be coincidental.

### (g) Wedderburn number

The dimensionless Wedderburn number quantifies the balance between forces due to applied wind and pressure reaction associated with thermocline tilt. It determines the applicability of one dimensional concepts and models, in which it is assumed that temperature strata are horizontal (Patterson *et at.* 1984).

Where the number is greater than ten, surface stirring is the dominant mixing process and the thermocline is essentially horizontal. As the number decreases toward 3, shear and tilt become more noticeable. Below 3, stirring at the upwind end produces an upwelling front which mixes the lake. At around 1, the therraocline surfaces at the upwind end. The Wedderburn number for Hartbeespoort Dam was highly variable, ranging from <<1 in winter to >100 in late summer (Fig. 5.13).

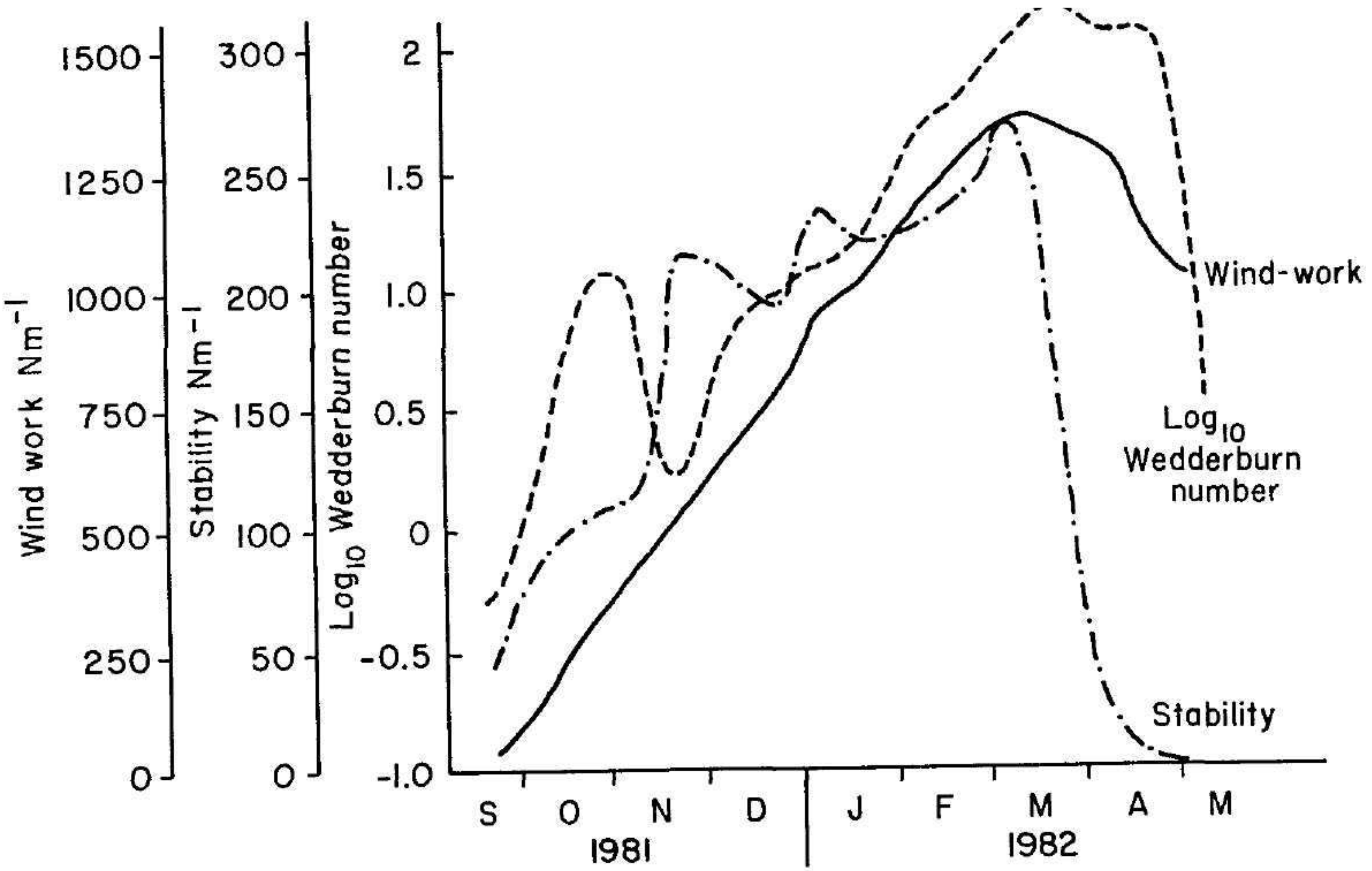


Figure 5.13. Wind-work, stability and Wedderburn number for the summer of 1981-1982, using data smoothed as in Figure 5.8. (Hely-Hutchinson, in preparation).

From Figure 5.13, it is deduced that temperature samples taken at the raft were usually representative of the lake mainly between December and April. Those taken in October and November should be viewed with care, in spite of the high degree of stability.

The specification of the Wedderburn number involved an assumption of an 8 km fetch in relation to wind-direction. In this context, it may be useful to regard Hartbeespoort as cruciform rather than dendritic. By horizontal resolution, it could probably be shown that the effective fetch is reasonably constant for all wind-directions. An 8 km fetch was assumed in two other calculations, namely pycnocline depth (Section 5.1.5(b)) and maximum wave-amplitude (Section 5.1.7(d)), in which the answers corresponded closely to observed values.

#### (h) Thermal budget

A thermal budget, calculated as in Hutchinson (1957) over the early summer of 1982-1983, was entered into Table 5.4 alongside those of other lakes.

Heat acquisition (Fig. 5.11) and incident short-wave radiation (Fig. 5.2) were measured directly. The latter was assumed to include contributions from the sun, sky and atmosphere. The generally-favoured 6% reflection/radiation ratio was used to calculate reflected radiation. Evaporation losses were derived from data_ supporting Table 5.2, assuming latent heat as 2.46 MJ kg $^{1}$. Sensible heat, which in many lakes is small, could not be assessed from either the Bowen ratio or Johnson expression, as the relevant local temperatures and vapour pressures were not known. Losses into vapour were ignored, as in all Che data entered directly into the Table. In yearly budgets for lakes Hula and Kinneret, such losses amounted to less than 1% of incident radiation. Transfer due to advection, which is neglected in many lakes, could not be assessed in Hartbeespoort owing to the absence of relevant temperatures.

The long-wave residue was inserted so as to balance the budget. The value corresponds to a difference of 11°C between air and water surface temperatures, a figure consistent with conclusions from a brief examination of the incomplete available data.

No obvious sub-classifications or general trends were found in Table 5.4. Thus the columns were characterised by means and standard deviations. As the Hartbeespoort entries were, in each case, within 1.2 standard deviation of the mean, and on average, within 0.6 standard deviations, they are regarded as consistent with the set. One of the largest proportional deviations (one S.D. in respect of the long-wave residue) indicates that advection and sensible heat could affect the balance.

### 5.1.6 Aperiodic, wind induced currents

In order to assess the significances of local variations from the whole lake averages, several specialised measurements were taken in discrete areas.

TABLE 5.4. Power budget during periods of rapid heat acquisition, in ten selected lakes, arranged in order of increasing latitude

| Lake | Reference | Latitude | Height above sea-level | Maximum depth | Mean Depth | Period | Short wave radiation | Reflection | Reflection radiation ratio |
|---|---|---|---|---|---|---|---|---|---|
| | | °S/N | m.a.s.l. | m | m | months | W m$^{-2}$ | W m$^{-2}$ | % |
| Titicaca | Carmouze *et al* (1983) | 16S | 3810 | 283 | - | November | 299 | -21[2] | 7 |
| Aswan | Omar & El Bakry (1980) | 22N | 175 | 90 | 25 | April | - | - | - |
| Hartbeesp. | This report (1985) | 26S | 1162 | 32 | 9.6 | Oct - Feb | 255 | -15 | 6 |
| Le Roux | Allanson & Jackson (1983) | 30S | 1200 | 69 | 29 | November | 224 | -16[2] | 7 |
| Kinneret | Hutchinson (1957) | 33N | -210 | 50 | 24 | May | 281 | -17[2] | 6 |
| Hula | Hutchinson (1957) | 33N | 67 | 4 | 1.7 | May | 302 | -18[2] | 6 |
| Mead | Hutchinson (1957) | 36N | 366 | 137 | 59 | May | 334 | -20[2] | 6 |
| de Grote Rus | de Bruijn (1982) | 52N | 0 | - | 5 | Mar & Apr | - | - | - |
| Petrusplaat | de Bruijn (1982) | 52N | 0 | - | 15 | Mar & Apr | - | - | - |
| Klammingen | Hutchinson (1957) | 59N | 0 | - | - | May - June | 209 | - | - |
| Mean | | 35.9 | 657 | 95 | 21 | - | 272 | 17.8 | 6.3 |
| Standard deviation (S.D.) | | 14.1 | 1211 | 93 | 18.2 | - | 45 | 2.3 | 0.5 |
| HBPD deviation/S.D. | | 0.7 | 0.4 | 0.7 | 0.6 | - | 0.4 | 1.2 | 0.6 |

ontinued)

| Reference | Long-wave residue | Radiation residue | Sensible heat | Evaporation loss | Advection transfer | Total acquisition | Acquisit withou advecti |
|---|---|---|---|---|---|---|---|
| | $W\ m^{-2}$ | $W\ m^{-2}$ | $W\ m^{-2}$ | $W\ m^{-2}$ | $W\ m^{-2}$ | $W\ m^{-2}$ | $W\ m^{-2}$ |
| Carmouze *et al* (1983) | -78 | 220 | -22 | -179 | - | - | 40 |
| Omar & El Bakry (1980) | - | 219 | +40[6] | -183 | -12 | 69 | 81 |
| This report | -61[1] | 179 | - | -131 | - | - | 40 |
| Allanson & Jackson (1983) | -101 | 122 | +7[7,6] | -46[7] | - | - | 101 |
| Hutchinson (1957) | -87 | 203 | -5 | -46 | - | - | 150 |
| Hutchinson (1957) | -87 | 242 | -5 | -194 | - | - | 12 |
| Hutchinson (1957) | -126 | 191 | +15 | -102 | 145[3] | 242 | 97 |
| de Bruijn (1982) | - | 62 | +3[6] | -28 | - | - | 37 |
| de Bruijn (1982) | - | 65 | +8[6] | -15 | - | - | 65 |
| Hutchinson (1957) | -54 | 155 | - | -53 | -1.3 | - | 102 |
| | -85 | 165.8 | 5.1 | -97.7 | - | - | 72.5 |
| iation (S.D.) | 24 | 63.9 | 18.0 | 69.4 | - | - | 41.4 |
| on/S.D. | 1.0 | 0.2 | - | 0.5 | - | - | 0.8 |

Value fixed to balance the power budget.
Values calculated from given ratio but not used in budget.
Difference between total acquisition and acquisition without advection.
Difference between total acquisition and advection-transfer.
Balance reckoned from radiation residue onwards.
From the texts, it is concluded that the negative values presented are intended to be read positive in connection with the budget.
The figures for sensible heat and evaporation-loss were transposed in the text. By comparison with other results, it is assumed that this was a misprint, and they were entered as shown.

Wind induced currents were relevant particularly to the locations and build-up of algal hyperscums (Section 5.A.8). Currents were measured on several occasions in three different areas: the dam wall cut, the main basin and the Crocodile debouchement (Fig. 3.1).

A sharp change in wind velocity occurred at around mid-day on 1983-08-04 (Fig. 5.14). The full response of the lake currents to the wind is shown in Figure 5.15. In order to obtain initial understanding, the long term trends and oscillatory components of the curves are separated out in Figures 5.16 and 5.18 and resolved orthogonally. An assumption was made that the modes were only loosely coupled over the measurement area. The following deductions were drawn in respect of steady state:

(a) Surface currents settled (in steady state) to around 2% of the driving wind speeds (Fig. 5.16), which agrees with the experien ces of both Langmuir and Olson (Hutchinson 1957).

(b) Steady state was reached approximately one hour after the change in wind velocity (Fig. 5.16).

(c) For currents flowing parallel to the dam wall cut, the vertical distribution (Fig. 5.16) agreed with that proposed by Hutchinson (1957) for a system closed at one end.

(d) Due presumably to the topography and morphometry, currents tended to enter and leave the cut with a corkscrewing motion, as shown in Figure 5.17. This convective effect is mentioned also Section 5.1.5(f).

### 5.1.7 Seiches

#### (a) Introduction

Seiches occur at interfaces between media. Surface seiches, i.e., those between water and air, are believed to be of too small an amplitude and too high a frequency to affect any significant limnological function in Hartbeespoort Dam.

The main limnological significance of internal seiches, which correlate with low Wedderburn numbers (Section 5.1.5(g)), lies in their tendency to break under shear and at boundaries. They can also confuse readings of temperature used to calculate whole lake statistics, and their effect thus needs to be taken into account when making such calculations. They they can also convey nutrients in and out of the euphotic zone.

Several seiches were detected in Hartbeespoort Dam, both by current and temperature measurements. They occurred in the main basin (on one occasion) and the dam wall cut (on several occasions). In the other sections of the lake, namely the Crocodile and Magalies debouchements, the exponential shape of the channels (c.f. Richardson 1953) and the presence of littoral vegetation (c.f. Madsen & Warncke 1983) possibly facilitated absorption rather than reflection of wave energy.

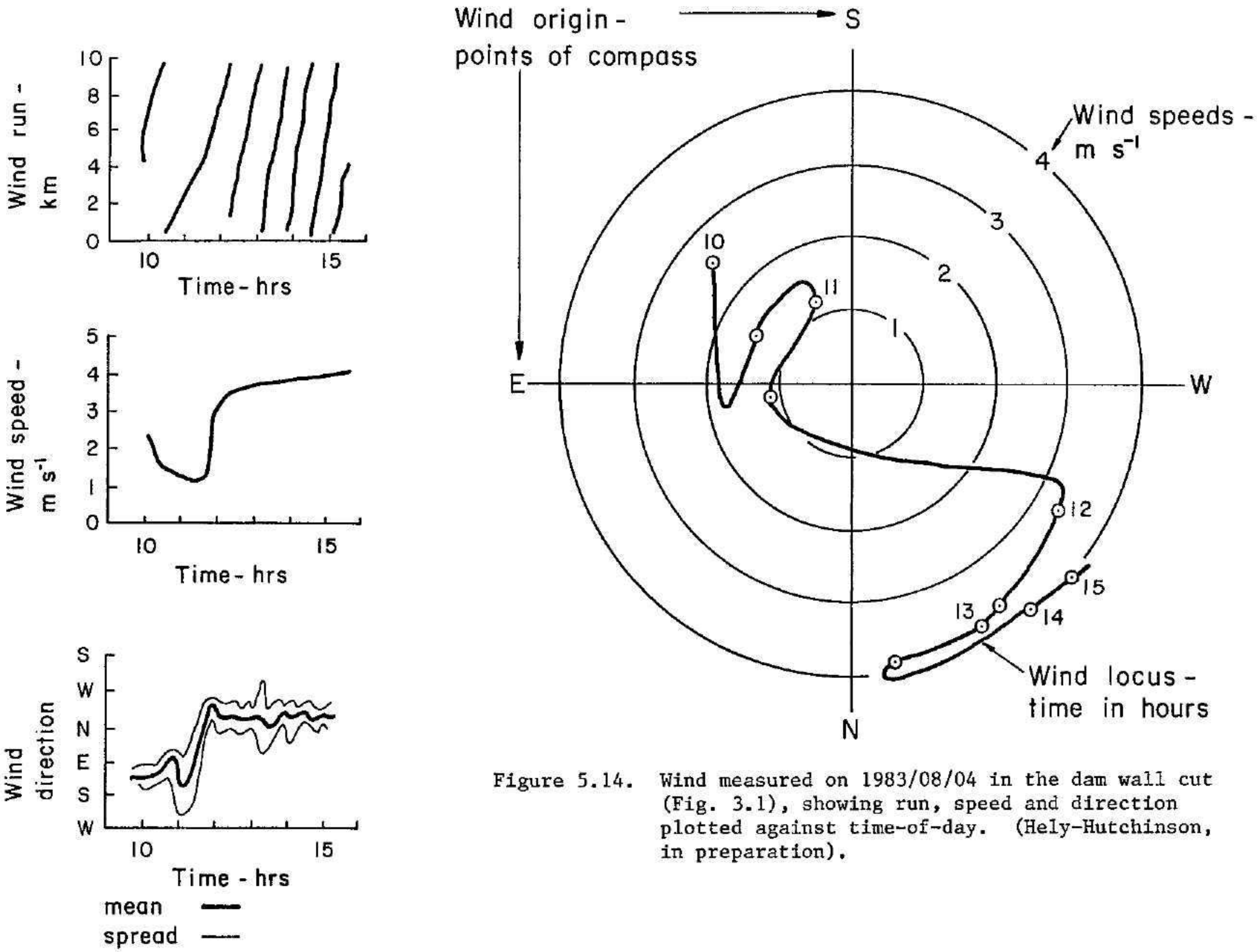


Figure 5.14. Wind measured on 1983/08/04 in the dam wall cut (Fig. 3.1), showing run, speed and direction plotted against time-of-day. (Hely-Hutchinson, in preparation).

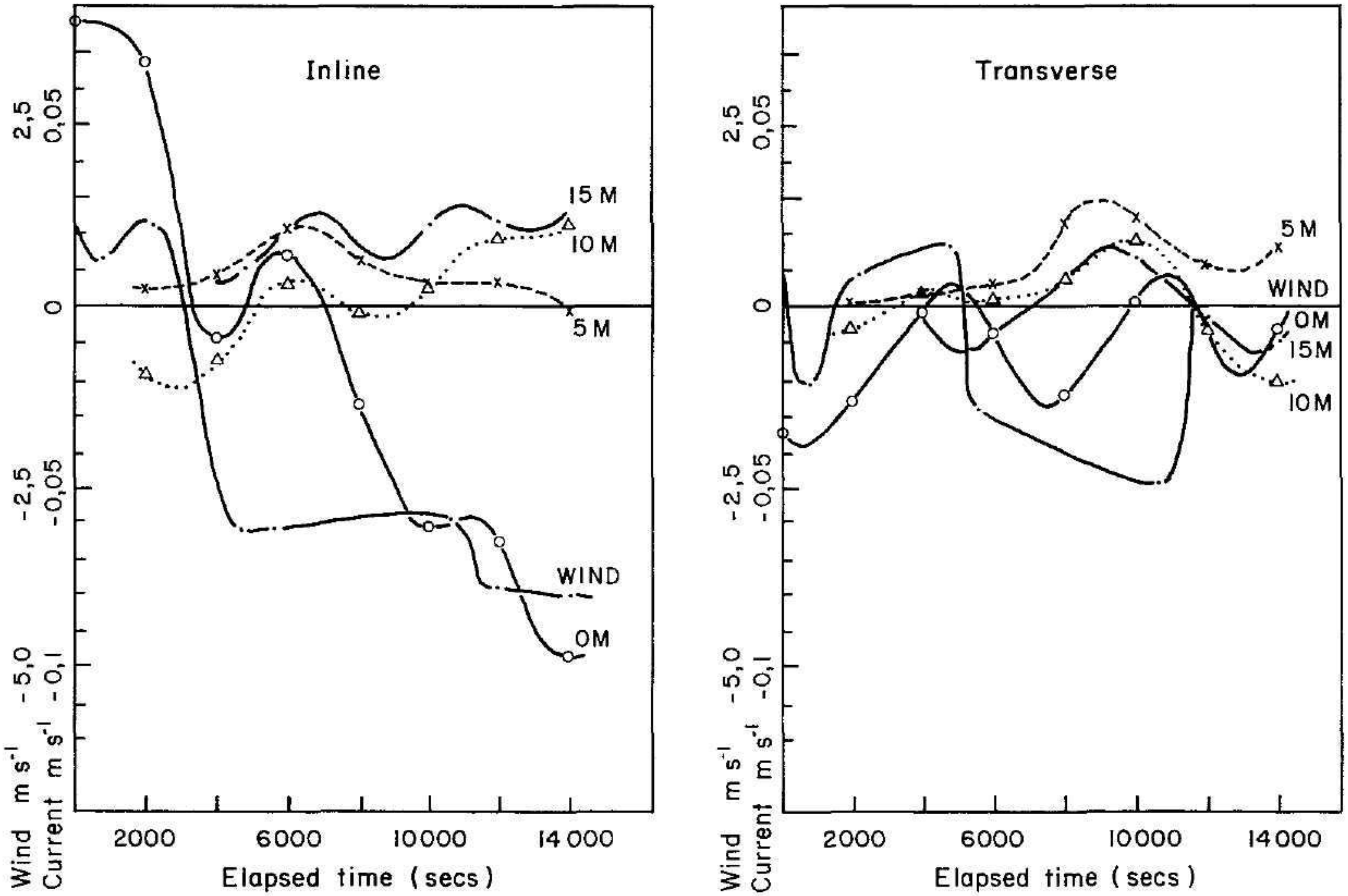


Figure 5.15. Currents measured on 1983/08/04 in the dam wall cut (Fig. 3.1) resolved in-line with and transverse to the cut (Hely-Hutchinson, in preparation).

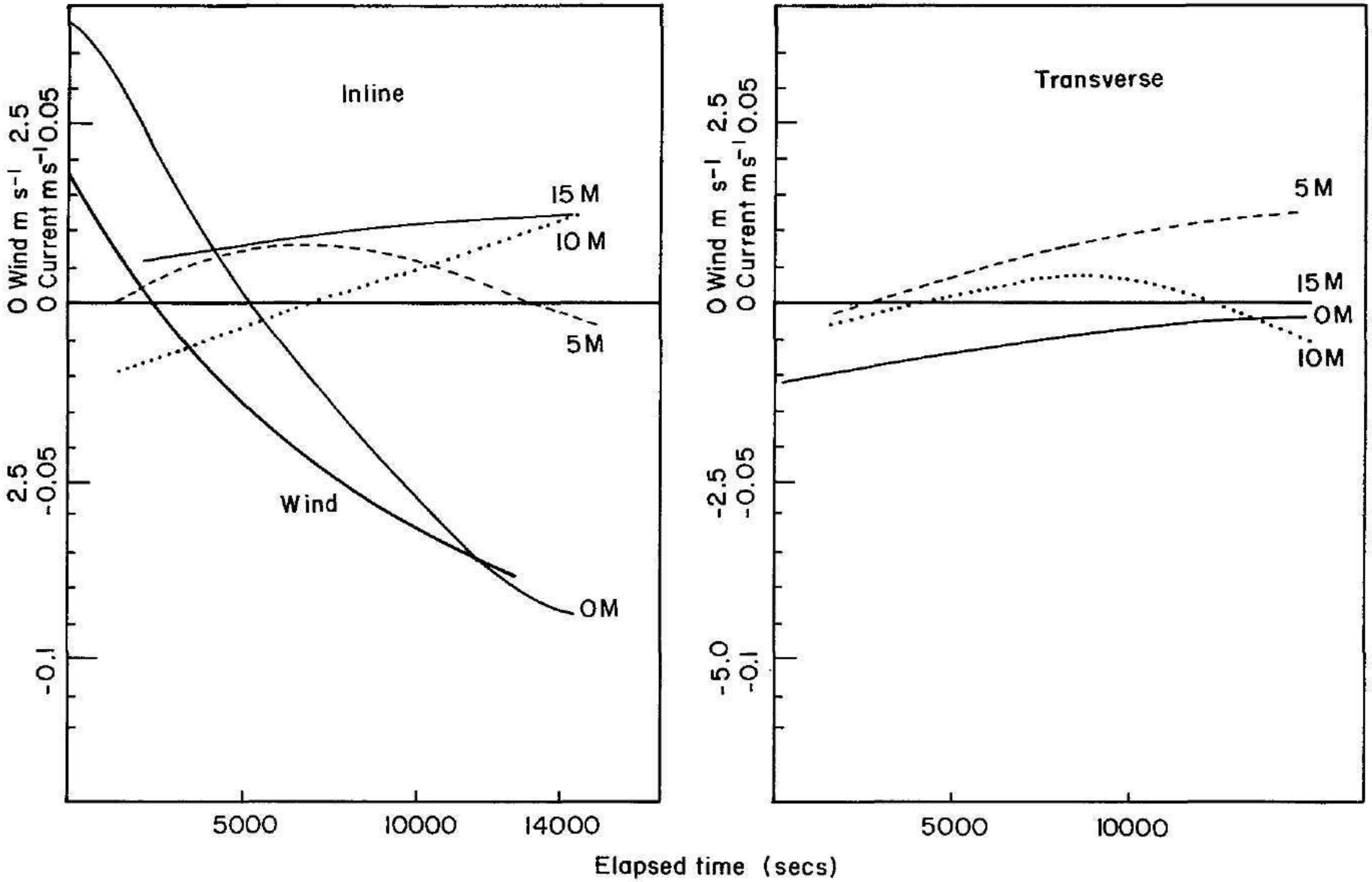


Figure 5.16. Long-term trends in currents in the dam wall cut (Fig. 3.1), resolved in-line with and transverse to the cut. (Hely- Hutchinson, in preparation).

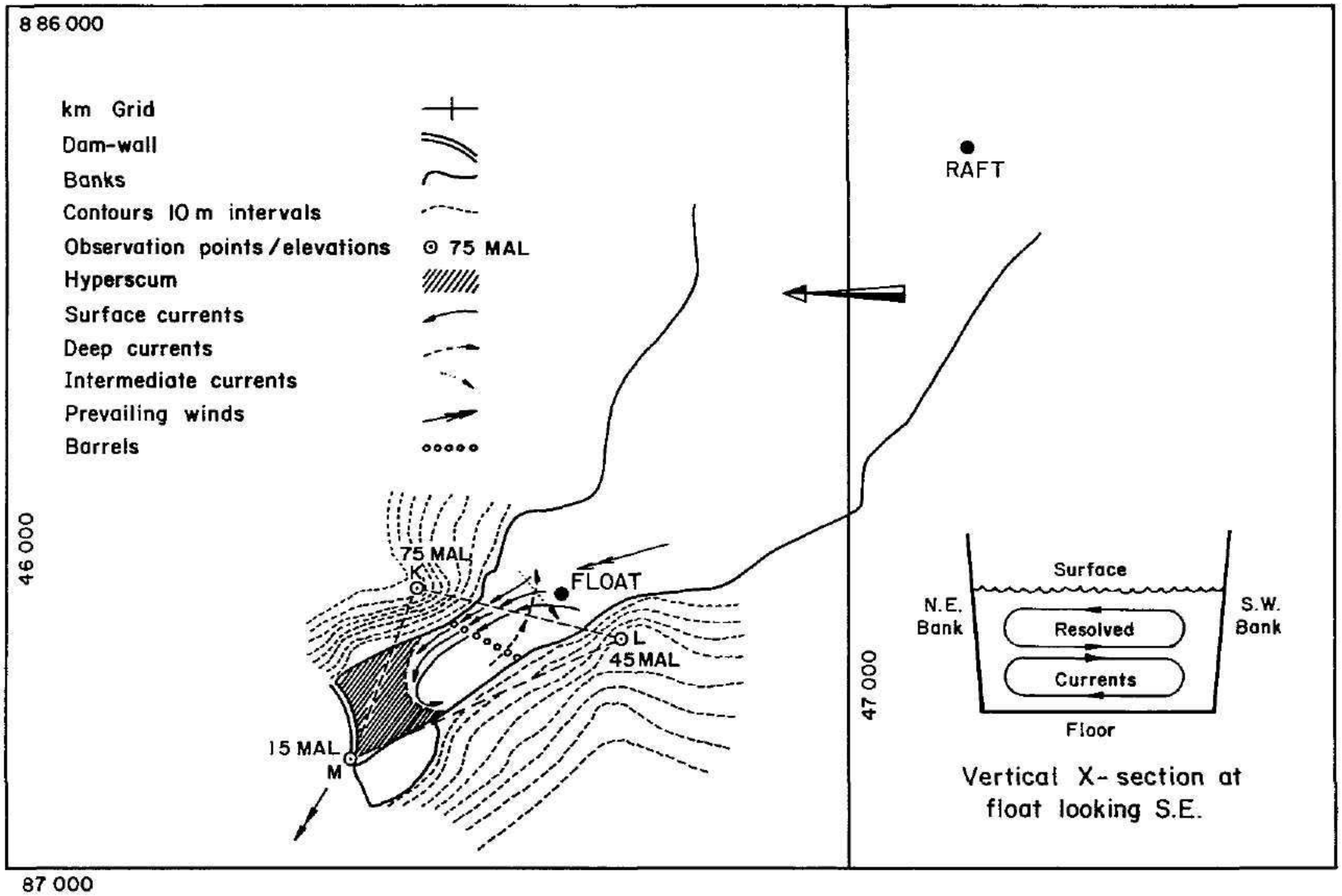


Figure 5.17. Prevailing currents in the dam wall cut (Fig. 3.1). Measured on 1983/08/11, showing gyre and return-current. (Hely-Hutchinson, in preparation).

(b) Dam wall cut (winter)

The main significance of seiche data taken near the dam wall has been to eliminate them when estimating long-term trends, and thus to avoid confusion in interpretation. The danger of this occurring is illustrated by the curves shown in Figure 5.18, in which the fluctuations are of the same order as the trends in Figure 5.16. The Wedderbum number was 0.5.

The observed seiches were associated with a thermocline created by diurnal heating of the water column probably fully mixed each night (c.f. Section 5.1.5(f)). The predominant periods lay between 2 000 and 8 000 seconds. These periods could be defined by the transverse dimensions of the cut (Fig. 5.17), that is, between 90 and 200 m, at a full depth of 20 m and a temperature differential due to diurnal heating of 0.4 °C, and assuming a single thermocline (Hutchinson 1957). There were some pronounced harmonic relationships, indicating that certain transverse resonant effects were coupled together.

The coupling mechanism from transverse to in-line motion may have been due to:

(i) coriolis, unlikely owing to the small dimensions involved;

(ii) wind, owing to the convolutions of the above surface channel;

(iii) current itself, owing to the convolutions of the below surface channel.

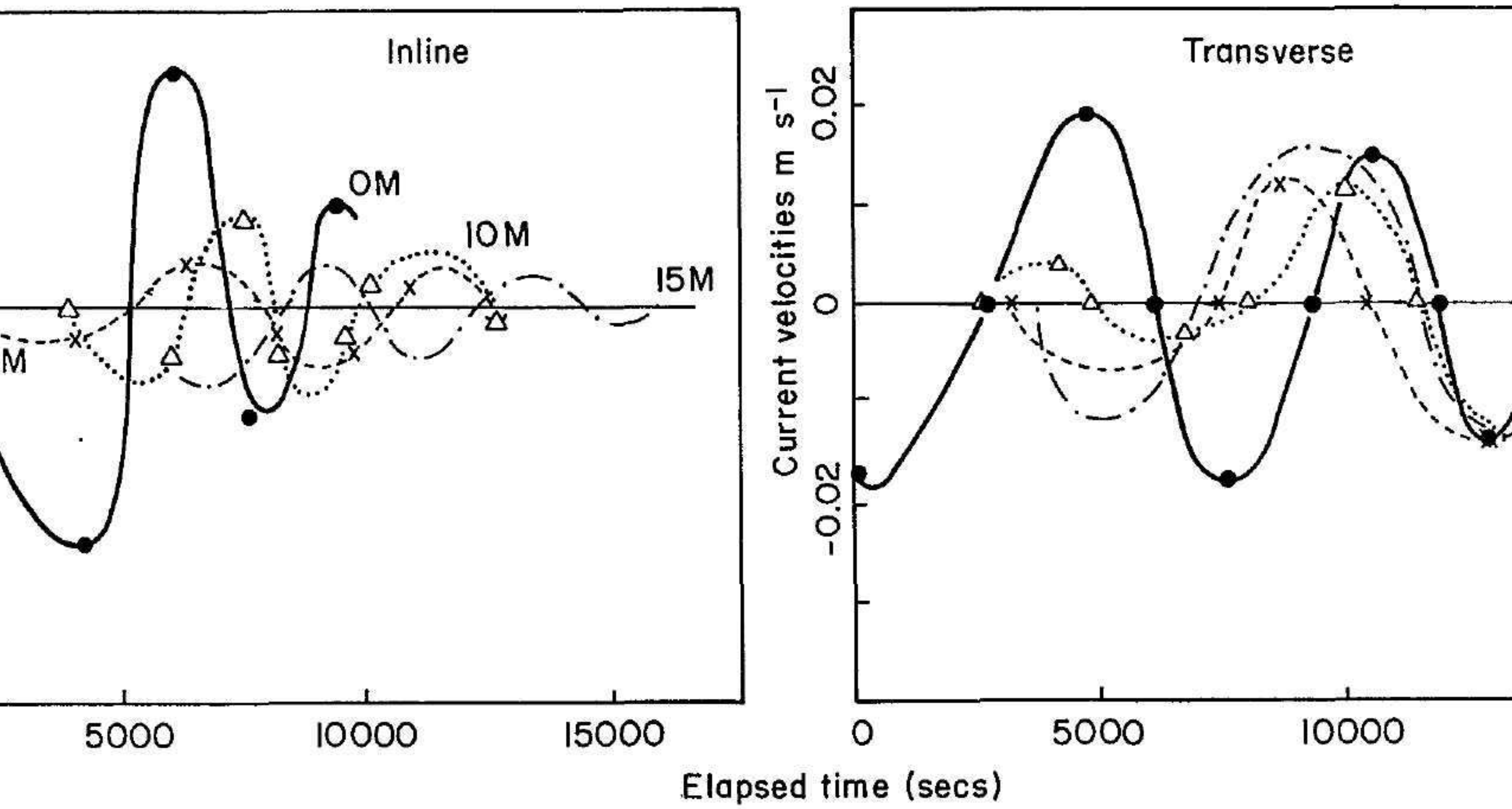
(c) Main lake (summer)

One main lake seiche was detected during the period (Fig. 5.19). It arose, on 1984-01-08, in response to a 17 m s spike of wind in the cut and lasted approximately 18 h, with a period of 12 h, which was consistent with a fetch of 4 km. This is the diameter of the main basin at reduced level, and is also the distance from the dam wall to the nearest bank down the channel. The Wedderburn number was 0.3. A rapid attenuation was seen, which might have been connected with the uneven width of the channel. It seems unlikely to have been attributable to the wind reversal at O6hOO, as the wind speed at that time was much lower.

A steady 8 m s$^{1}$ wind occurred about ten days later. The lake responded by mixing but not oscillating. The Wedderburn number was 1.5. This indicated that the onset of seiches occurred in response to windspeeds of between 8 and 17 m s . It would also be influenced strongly by the spectral distribution of the wind-time pattern. The creation of seiches by winds in the above velocity range was confirmed by Fischer and Smith (1983) in respect of another dendritic lake - Lake Mead, USA.

The likelihood of internal seiches in Hartbeespoort Dam has been assessed from four sources, namely:

(i) The frequency of high winds in the wind distributions shown in Figure 5.5;



.18. Oscillating components of current velocities measured on 1983/08/04 in the dam wall cut (Fig. 3.1), resolved in-line with and transverse to the cut (Hely-Hutchinson, in preparation).

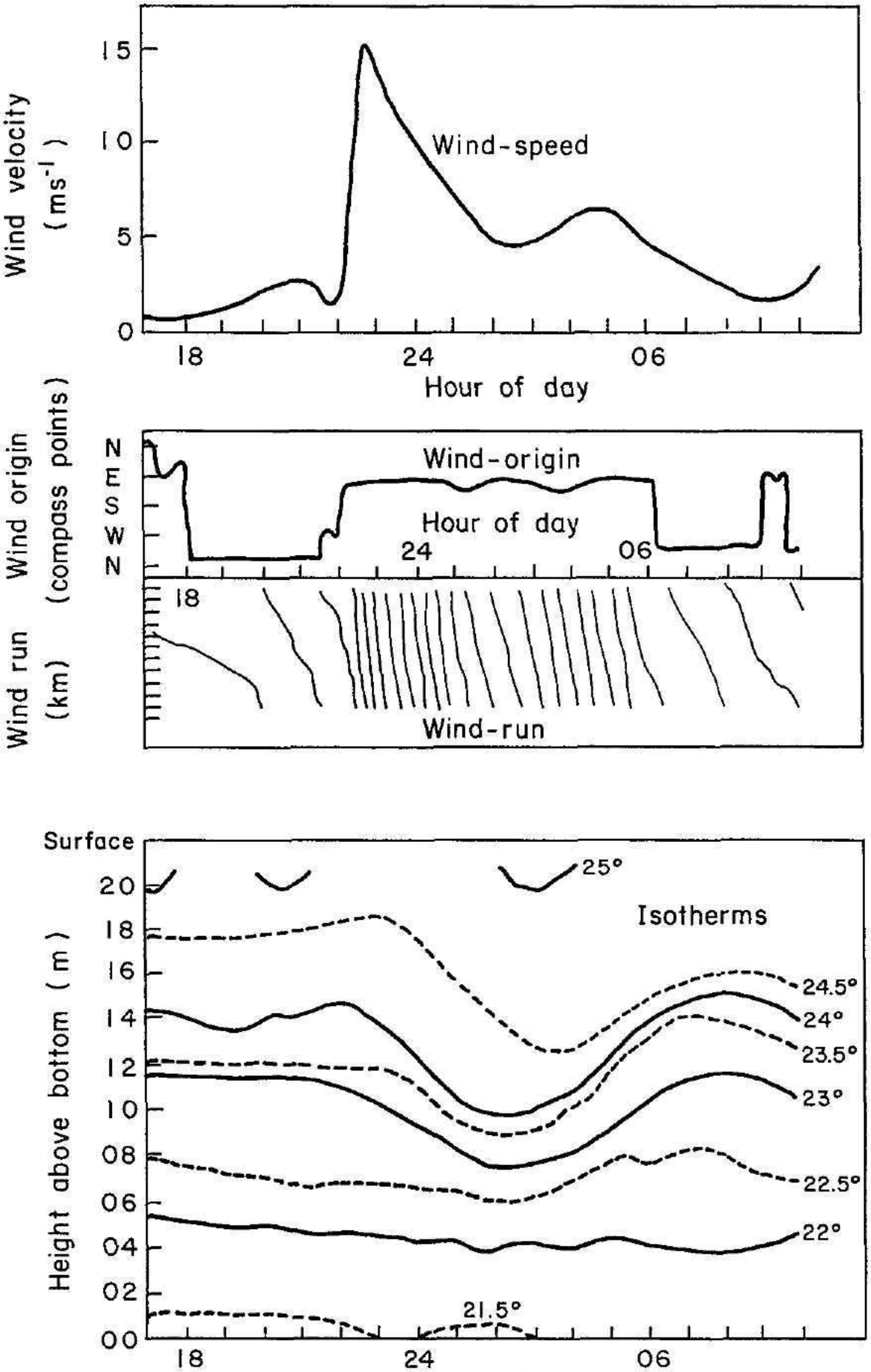


Figure 5.19. Seiche detected at point 1 (Fig. 4.1) on 1984/01/18-19, showing isotherms and associated wind-run, wind-origin and wind-speed. (Hely-Hutchinson, in preparation).

(ii) the ratio of wind velocities at Oberon and the dam-wall shown in Figure 5.20;

(iii) the frequency of large sudden variations in amplitude in the temperature profiles shown in Figure 5.7; and

(iv) the frequency of gusts of c 15 m s  amplitude measured at the dam wall during December 1983 and January 1984.

From this evidence, seiches of mean amplitude 5 m were estimated to occur over 15% of the summer season.

There appear to be more short-period variations in the thermal profile than in those of other lakes. If it is found that the measuring point lies at a thermal antinode, it may be advantageous to shift it to a quieter location.

(d) Other disturbances

The height of the highest surface waves observed during the period was approximately 1 m. This agrees with Stevenson's formulation (Hutchinson 1957) assuming a fetch of 8 km.

Langmuir cells were observed on several occasions in the dam wall cut in the presence of high winds. In their most spectacular manifestation, large nodules of algal scum were arranged in rows parallel to the wind-direction.

Internal waves (of higher frequency than seiches) were detected in temperature profiles. Although eddies at the buoyancy frequency (Fig. 5.9) could not be picked up individually at the 15 min sampling intervals, an idea of the amplitude could be gained from the variations. In the dam wall cut, in a temperature gradient of 0.2° m"$^1$, 0.2° variations were prevalent, indicating 1 m amplitude. In the Crocodile debouchement, where the gradient was 4° m"$^1$ during a flood, 0.4° variations were observed, indicating 0.1m amplitude. In neither case were these amplitudes comparable with those of the main basin seiches.

### 5.1.8 Inflow

Inflow currents were monitored at points shown in Figure 5.21, in order to help determine the introduction, deposition, uptake and flocculation of nutrients (c.f. Stumm & Morgan 1970, Chapra 1980, Hauenstein & Dracos 1984, Fischer & Smith 1983). The data gathered are regarded as a restricted set to be used later in conjunction with others for developing a comprehensive model.

Since knowledge was already available on the role of wind when little inflow current was present, emphasis was laid on the role of inflow in the absence of wind. As this situation almost never occurs in practice, the measurements were made under conditions of very high inflow, the effects of which would be more visible than those of wind at the time.

(a) Forcing function

The spate examined was, from the point of view of analysis, as productive a forcing-function as could be expected (Fig. 5.22),

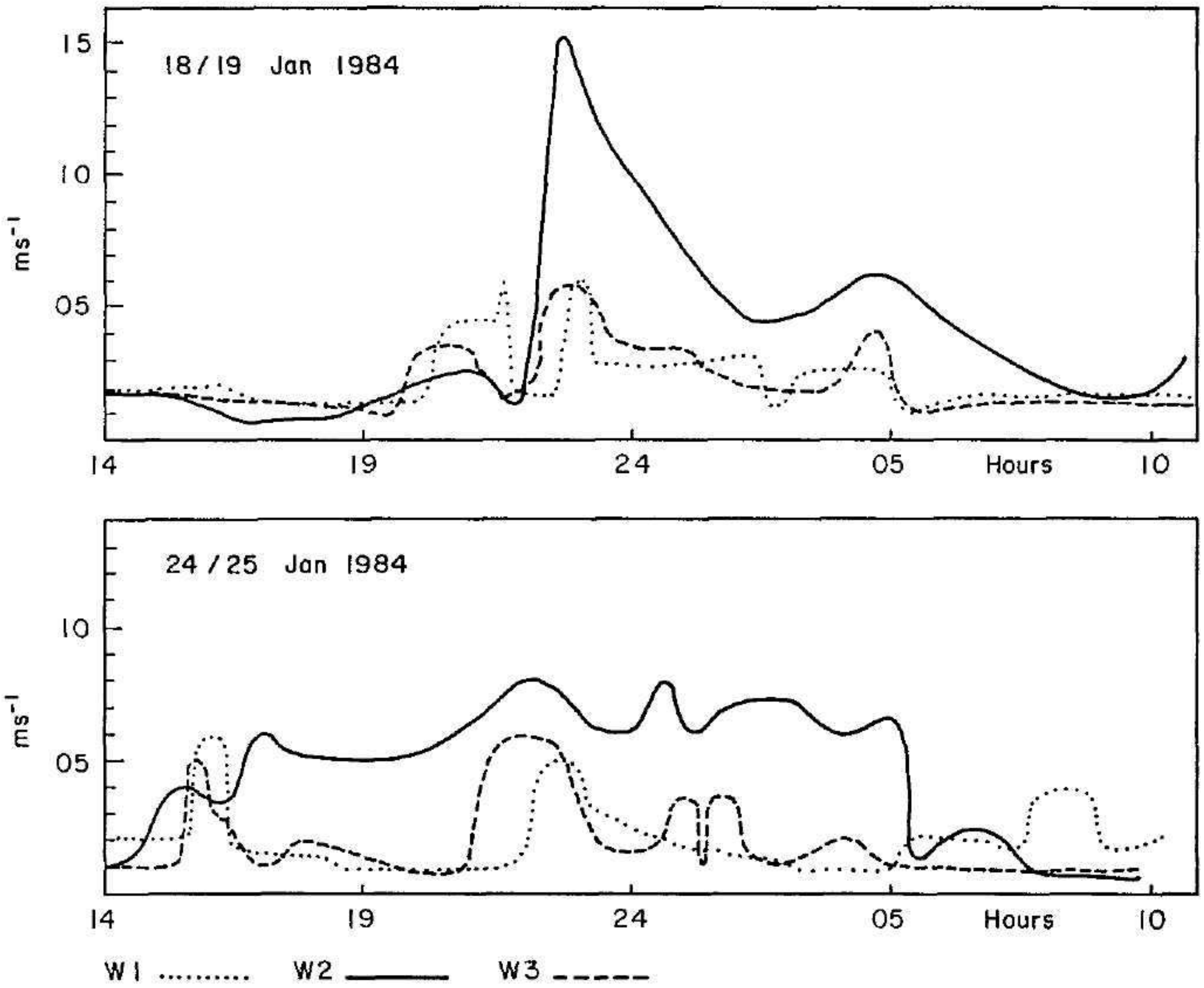


Figure 5.20. Comparison of simultaneous wind speed measurements at points W1-3 (Fig. 4.1) on two separate nights). (Hely-Hutchinson, in preparation).

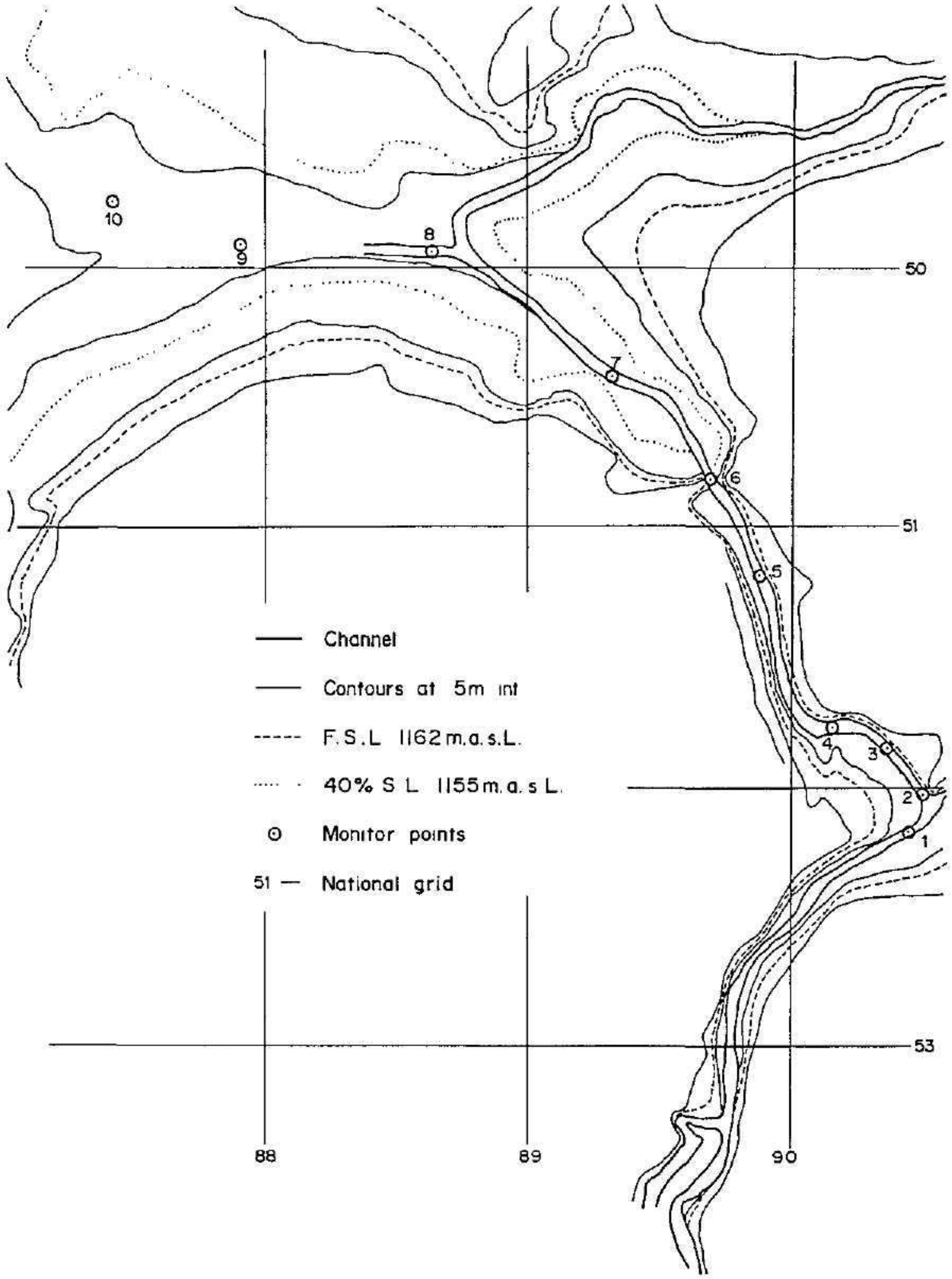


Figure 5.21. Crocodile debouchement (Fig 3.1) showing morphometry, old river channel, inflow sampling stations and 1 km grid lines. (Hely-Hutchinson, in preparation).

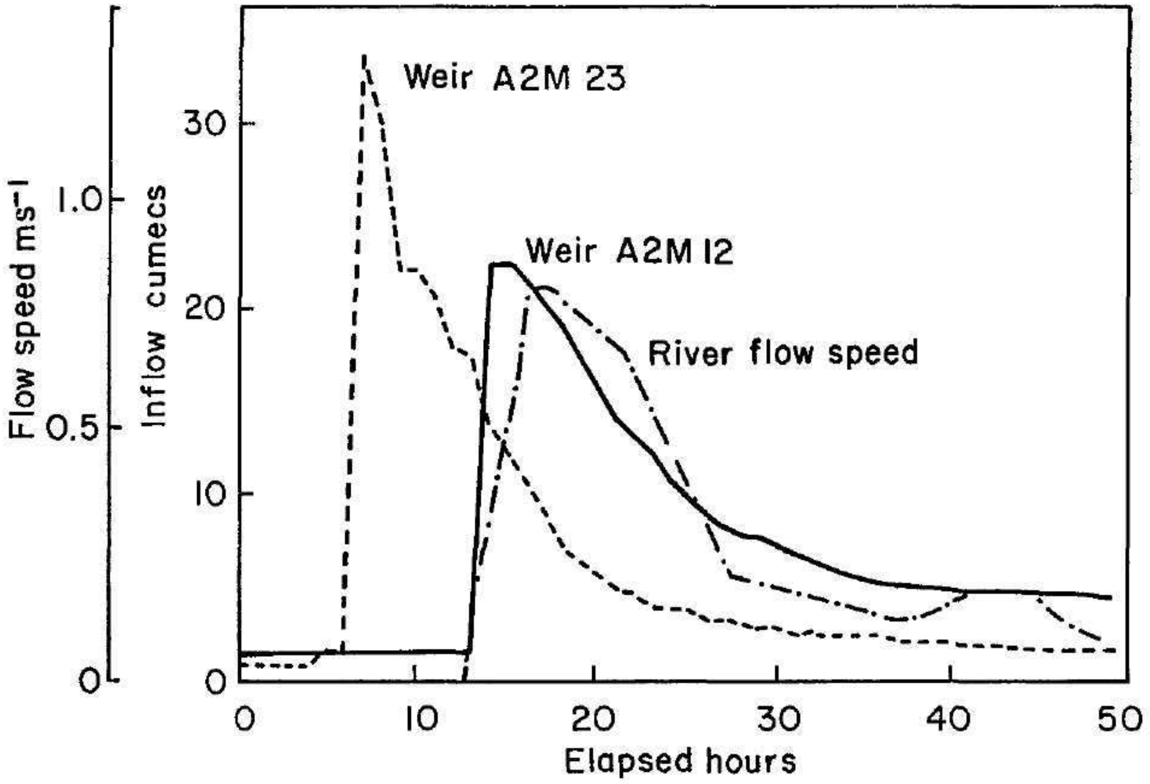


Figure 5.22. Inflow measured at weirs A2M12 and A2M23 (Fig. 3.3) and river flow-speed measured at point 5 (Fig. 5.21) (Hely-Hutchinson, in preparation), showing similar simple configurations of pulses.

The 1 M $m^3$ pulse was sensibly isolated from others before and after. Although it contained only about 0.7% of the average annual recharge, and was thus smaller than many, the leading edge was exceptionally sharp (less than one hour) and the tail was a nearly perfect exponential decay with a time constant of approximately 10 h . The Laplace transform was thus $1/(p + 1/36\ 000)$, which yielded a spectral function fairly flat from infinite period down to 10 h, and a 6 dB per octave decrement at shorter periods (Fig. 5.23). In qualitative terms, this meant, for instance, that a toe of influent water stretched far into the lake which did not happen when the same quantity of water was released as base flow over a longer period. When supplementary data have been gathered, this effect will be expressed numerically, along with others, to form a component of the above model.

(b) Spate

Four input-processes were demonstrated during the excercise:

(i) The spate took approximately 10 h to cover the 80 km from its source (Johannesburg) to the lake (Fig. 3.3). Thus it travelled at approximately $2.5\ m\ s^{1}$. The sharpness of the leading edge was presumably due firstly to the paved surfaces on which the rain fell, and secondly, to the fact that the storm itself travelled roughly downstream with the current. This effect confirms that reported in Foroud *et at.* (1984).

(ii) On entering the channel, the low conductivity run-off pushed the high conductivity resident base water ahead. Thus the run-off arrived some ten hours after the front of the spate, when the current had decayed to approximately 0.5 of its peak value of 22 $m^3\ s^{-1}$ (Fig. 5.22).

(iii) The conductivity response preceded, by some 5 hours, the temperature response, in respect of both rise and fall (Fig. 5.24).

(iv) Turbidity double-peaked (Fig. 5.25), firstly 4 hours behind the front and secondly, a much broader peak, when the run-off arrived.

From this it was deduced that the main sediment load was eroded from the ground onto which the rain fell, and a second, smaller load was resuspended from the channel. In view of the gap between the loads it is deduced that the spate at no stage rose above the river-banks.

The onset of the second turbidity peak coincided with the start of the drop in conductivity, which was possibly influenced as much by the aggregation of solutes as by the change from basewater to runoff.

(c) Lake response

Radiation, wind and inflow influenced the debouchement as follows:

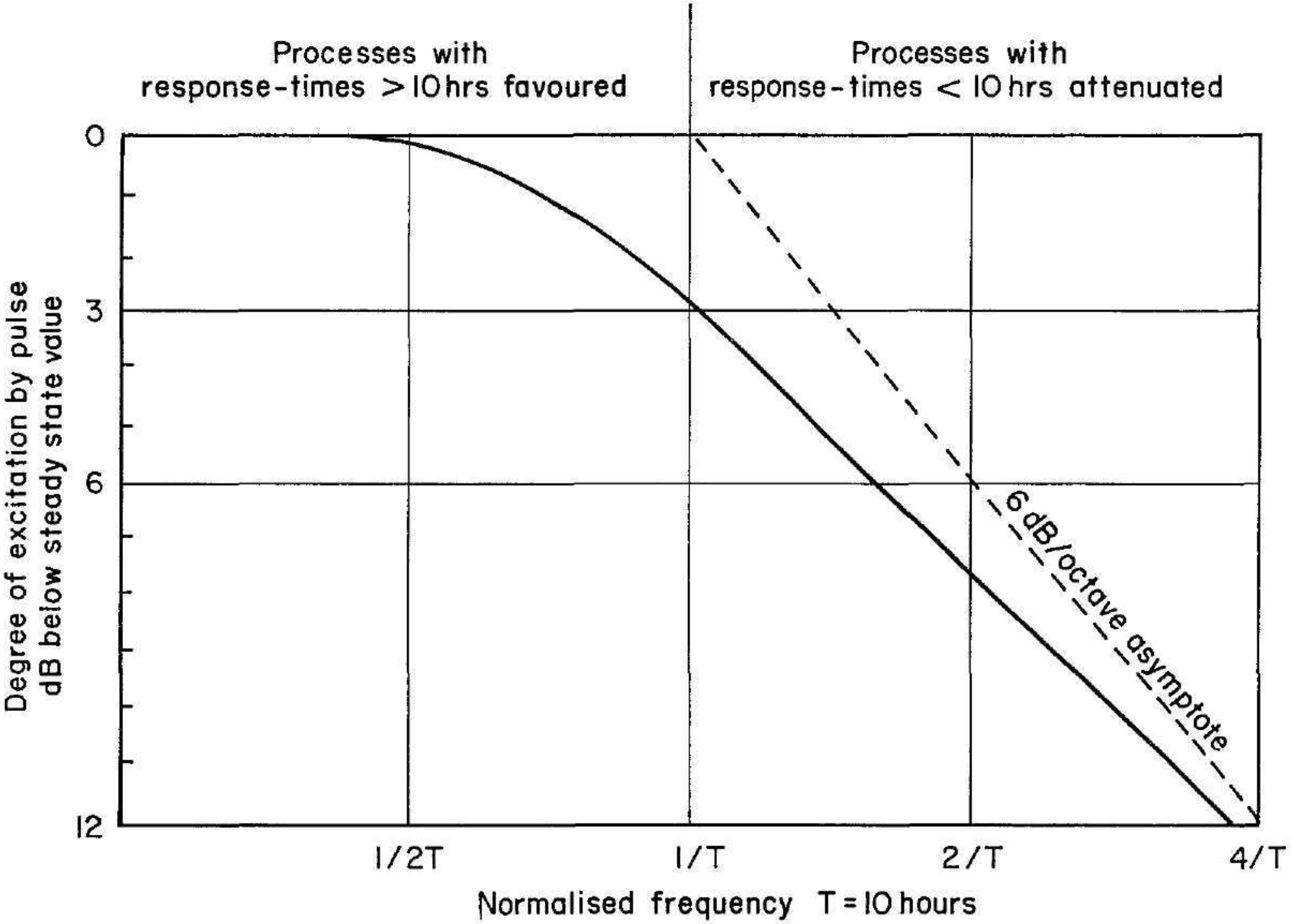


Figure 5.23. Spectral distribution of inflow-pulse at weir A2M12 (Fig. 5.22) (Hely-Hutchinson, in preparation).

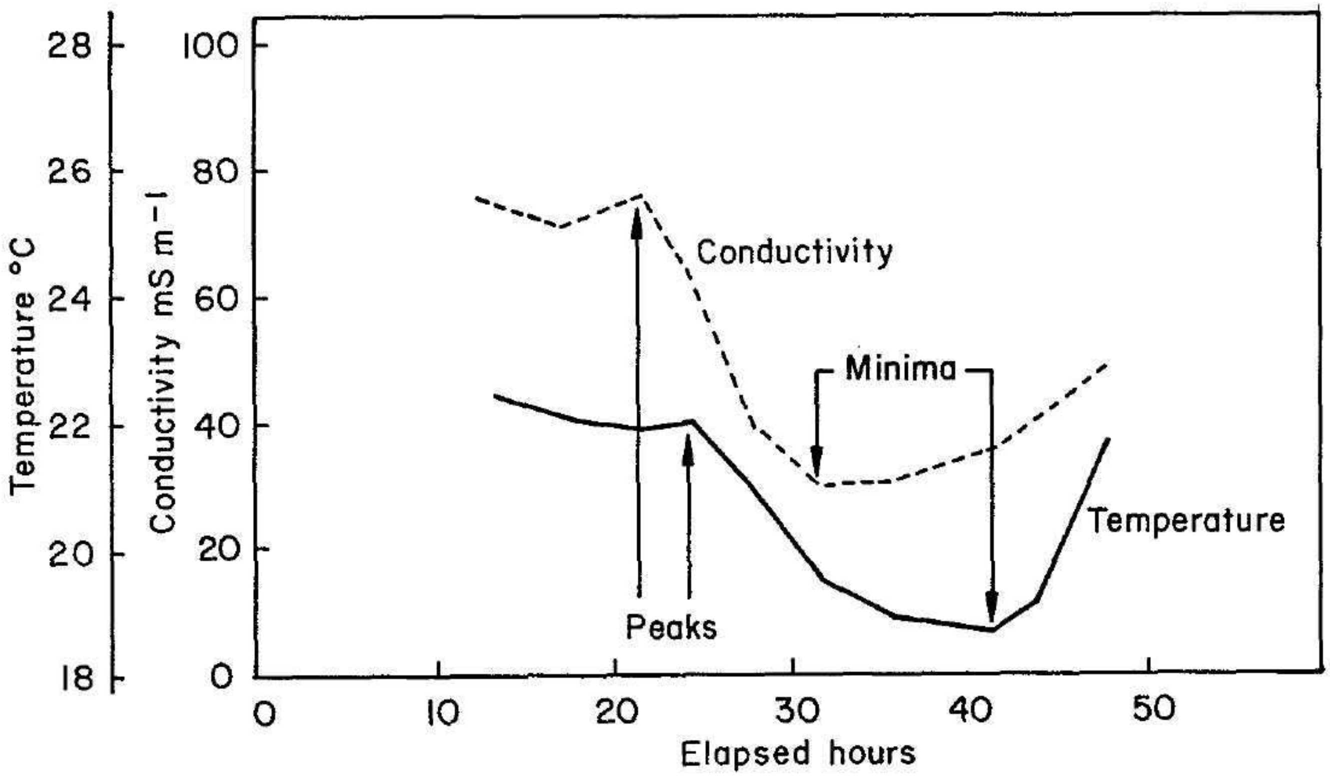


Figure 5.24. Conductivity and temperature measured at point 3 (Fig. 5.21), showing similar patterns but different reponse-times. (Hely-Hutchinson, in preparation).

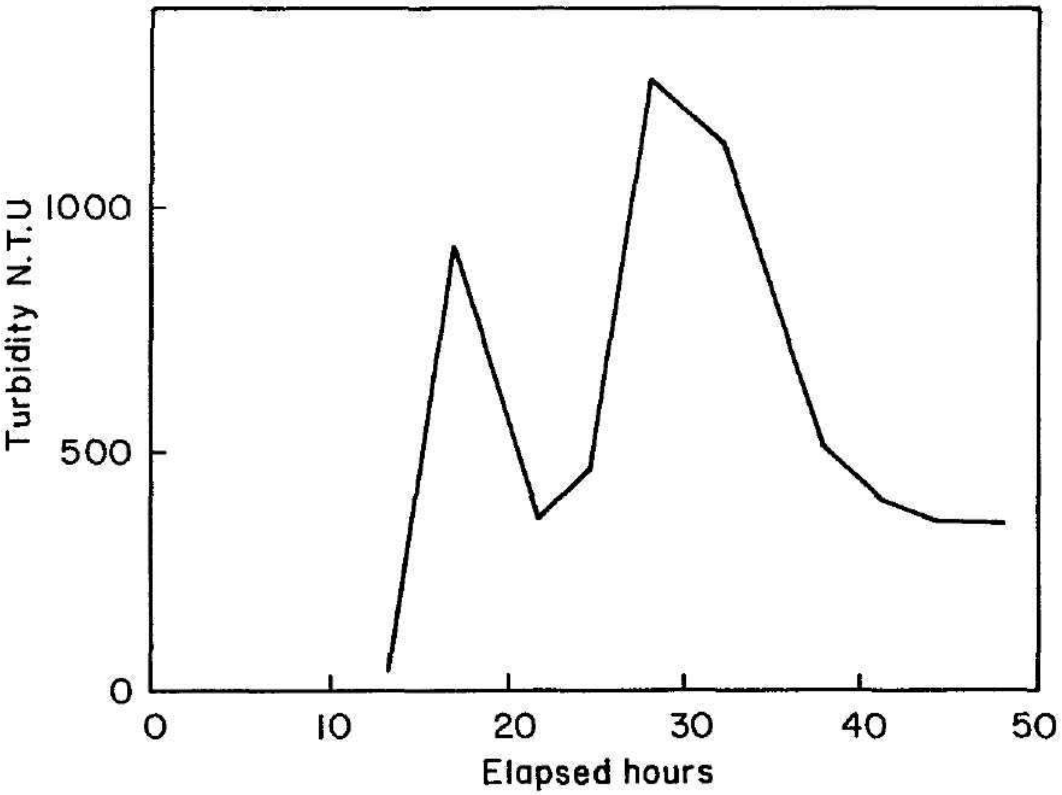


Figure 5.25. Turbidity measured at point 5 (Fig. 5.21), showing twin peaks. (Hely-Hutchinson, in preparation).

(i) During and after the spate, the denser influent water flowed under the resident lake water in the debouchement. The following three processes contributed to the density-differential:

Initially, the conductivity of the spate was higher than that of the lake proper, indicating a higher concentration of dissolved solids (Fig. 5.26).

At two stages, high concentrations of suspended matter arrived.

Inflow temperature (Fig. 5.27) at all times remained lower, and at some times much lower than that of the lake proper.

(ii) A 4 m s $^{1}$ wind blew over the lake during the storm, that is, before the spate arrived, and effectively mixed the water in the debouchement, probably right down to the sediment (Fig_. 5.27). At the end of the first day, a second (3 ms *) wind began to mix the cold bottom water with the surface layer, which had been heated by diurnal solar radiation (Fig. 5.27). This second wind was not, however, strong enough to influence the spate water near the sediment (Fig. 5.27).

(iii) The conductivity difference appeared to progress along the lake bottom for a shorter distance than the temperature difference, again suggesting aggregation. The turbidity peaks decreased fairly evenly along the bottom, more or less in-line with the temperature difference (Fig. 5.28).

### 5.1.9 Summary of physical limnology results

The thermal budget of Hartbeespoort Dam was consistent with examples taken from nine lakes In the tropical and temperature zones. Budget components such as solar radiation were seasonal and highly variable from year to year. The depth of penetration of solar radiation varied widely and was limited largely by the density of algae growing in the irradiated (euphotic) zone as demonstrated In Section 5.4.9.

Rainfall and evapotranspiration potential were roughly synchronous over the seasons. Excess moisture (that is - runoff) was the residue of variations In these large quantities, and varied widely over daily to decadal time-scales. Two main consequences were that careful control was kept on outlets to customers and a significant proportion of natural inflow arrived In the form of isolated spates.

During one such spate, which was monitored, the warmer base-flow water was pushed ahead of the colder runoff. Two turbidity peaks occurred; the first at the front of the spate and the second at the arrival of the runoff. A drop in conductivity coincided with the arrival of the second turbidity peak and preceded the drop in temperature. Conductivity was possibly as strongly influenced by the aggregation of solutes onto particles as by the changeover from base-water to run-off. The colder, denser, turbid runoff flowed along the floor of the lake for at least 36 hours after the current peak had passed. Thus spates probably exerted a crucial influence on both eutrophication (Section 5.2) and siltation (Section 5.3).

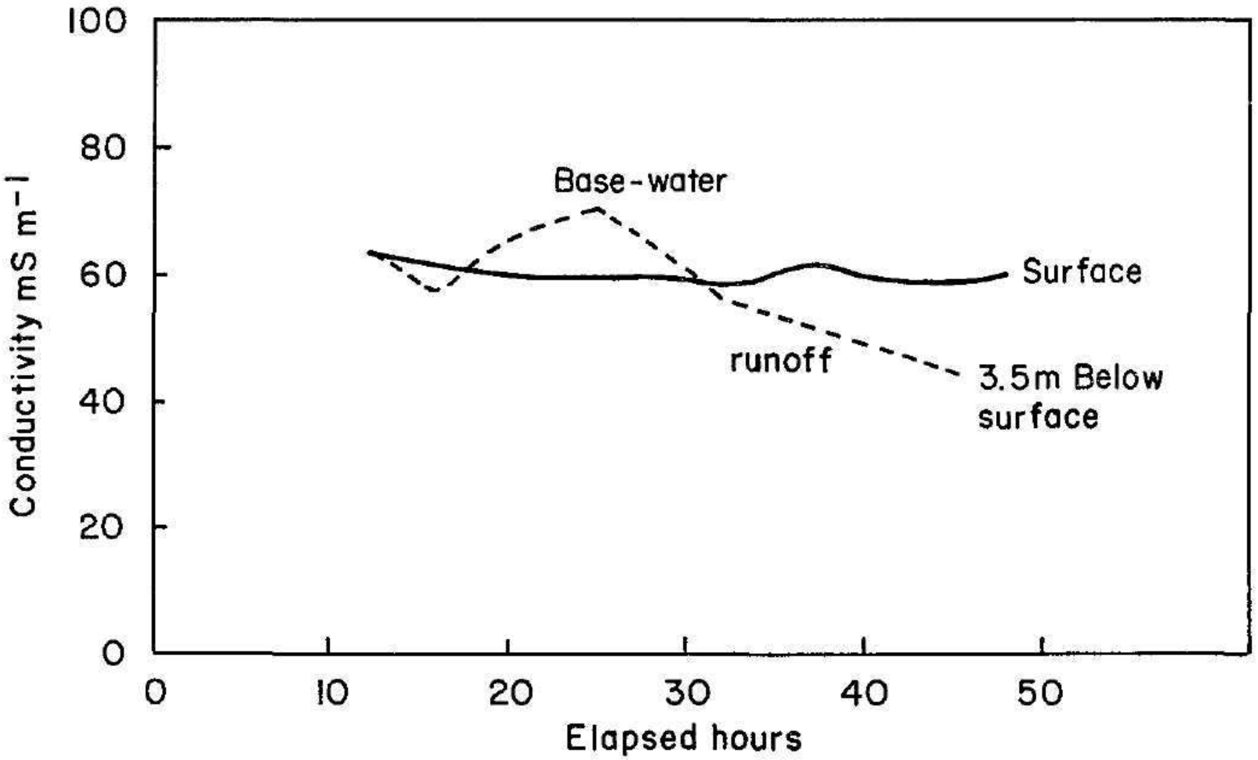


Figure 5.26. Conductivity measured at point 8 (Fig. 5.21), showing arrival of basewater and then runoff along the bottom. (Hely-Hutchinson, in preparation).

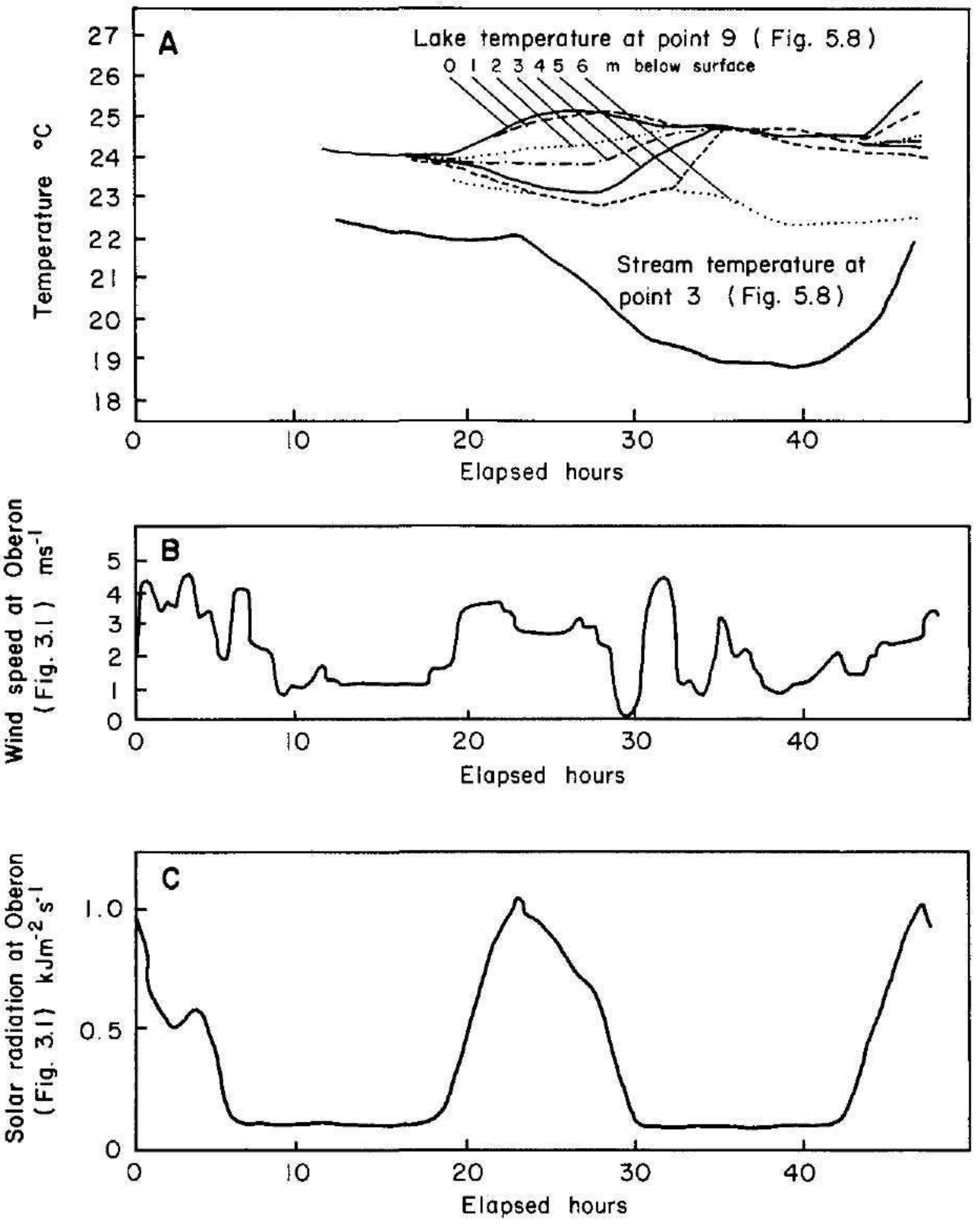


Figure 5.27. Statification and mixing at point 8 (Fig. 5.21) showing correlations between lake and river temperatures (A), wind speeds (B) and solar radiation (C). (Hely-Hutchinson, in preparation).

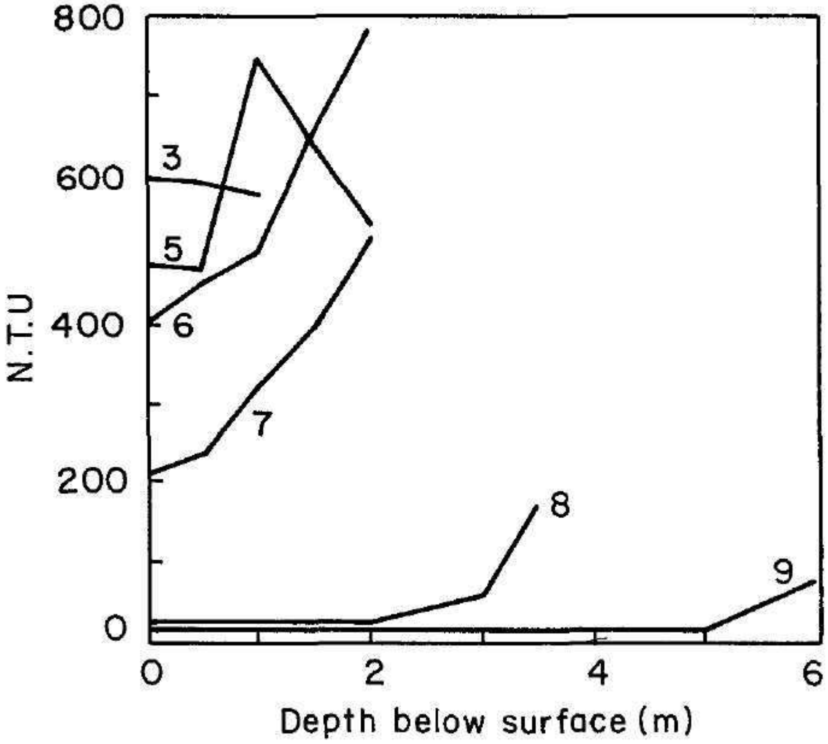


Figure 5.28. Time-averages of turbidity, measured at various depths at points 3 to 9 (Fig. 5.21), showing progressive decrease along channel (Hely-Hutchinson, in preparation).

Wind speeds over the main basin tended to be low (on average 1.7 m s"$^{1}$), which enabled *Miavoeystis* (the dominant alga, Section 5.4.6) to maintain its position within the euphotic zone and to accumulate (Section 5.4.14). Accumulations were blown into sheltered inlets, where they thickened vertically to form consolidated scums (Section 5.4.8).

Due to the terrain, wind patterns in such inlets could be very different from those in the main basin. Differences were most noticeable in the dam-wall cut, which was in-line with the NNW-SSE slope of the terrain between Johannesburg and Brits. The ambient wind, which tended to reverse direction twice per day in typical mountain fashion, was concentrated and accelerated by the surrouding hills.

Like other lakes in southern Africa, Hartbeespoort Dam was found to be warm, monomictic; that is - stratified and mixed once per year. Temperatures, which varied from 12°C in winter to between 18 (bottom) and 26 (surface) in summer, influenced the distribution of species (Section 5.4.14). The study period covered one of the most severe droughts in record. This resulted in a reduced epilimnion, a very much smaller hypolimnion and nutrients being carried further than normally into the main basin during the period. There was no indication that the biomass changed dramatically over the period.

The oxycline was found to be close to a persistent breeze-thermocline, and may have been controlled mainly by the limit to daily mixing. The level was well above that of the classically defined thermocline, the depth of which varied cyclically, independent of lake content. A period at the end of 1983, when the thermocline lay along the lake bottom (Fig. 5.12), corresponded to a brief collapse of the oxycline (Figs 5.29, 5.53 and 5.54). Times of overturn (that is - sudden, large increases in diffusion in Autumn) depended closely on lake content and occurred in March (when low) and April (when full).

Diffusivities in the hypolimnion and metalimnion, averaged over a summer period, were found to be consistent with those of four other low-latitude, high-altitude lakes. There was strong evidence that convection occurred at the lake shorelines, together with thermocline breakage due to seiche action. These effects would increase effective diffusivity.

Total heat, when expressed as a tropicality index, was found to follow trends for the southern African region. Total heat and wind work, which were the results of external influences (wind and radiation) , did not alter greatly over the drought period. Stability reduced considerably in tune with the reduced volume. Overturn could be confidently forecasted in terms of either stability trends or the typifying equations of the main thermocline.

The thermal profile was equalised over the whole lake by gravity forces within a day or so. Wind in the dam wall cut was violent enough to disturb this steady state by reducing the Wedderburn number sufficiently to set seiches in motion. During seiche action quantities of water were transferred back and forth across the lake. The largest seiches could raise anoxic hypolimnetic water temporarily into the euphotic zone and submerge algae below the zone for several hours at a time. Where readings of limnological parameters were taken at thermal antinodes, seiches could render them only of local and instantaneous significance. By comparison with temperature profiles in other lakes, those taken at the main measurement point seemed to suffer from unusual short-term variability. If this is found to be due to seiche-action, the point may be shifted to a quieter location.

## 5.2 CHEMICAL LIMNOLOGY

### 5.2.1 Introduction to lake chemistry

The composition and abundance of the chemical substances and gases dissolved in water has a profound influence on the biota. Elements of particular importance are oxygen, carbon, nitrogen, phosphorus and silica. An absence of oxygen precludes the survival of virtually all animals and many other organisms. Bound carbon, nitrogen, and phosphorus are essential for the growth of primary producers such as the phytoplankton, among which the diatoms also require silica for growth.

The atmosphere and photosynthesis by aquatic plants and algae are the sources of dissolved oxygen. Carbon dioxide arises mainly from processes of aerobic decomposition of organic matter and respiration in the water. Sources of other inorganic dissolved matter in a lake are principally the inflowing water, the bottom of the lake and, to a lesser extent, the atmosphere. Losses of dissolved inorganic substances occur with the water released from the lake, by biological uptake, chemical precipitation and adsorption onto sediment particles.

Summer stratification (Section 5.1.5(f)) results in important changes in the chemical composition of the epi- and hypolimnion, particularly in hypertrophic lakes. The epilimnion isolates the hypolinmion from the atmosphere and provides a rain of organic matter into the hypolimnion. Bacterial decomposition of this organic matter rapidly depletes the oxygen in the hypolimnion. Certain compounds containing iron, manganese and phosphorus, which are insoluble under aerobic conditions, become soluble under anaerobic conditions. Anaerobic hypolimnetic waters therefore usually contain elevated concentrations of phosphates, iron and manganese. Other processes which take place under anaerobic conditions include the denitrification of nitrates to nitrogen gas and the reduction of sulphates to hydrogen sulphide.

An important feature of hypertrophic impoundments is their low nitrogen: phosphorus ratio. This is brought about by the input of effluents with low N:P ratios, elevated levels of denitrification in such waters and also through losses of ammonia by volatization at the high pH values caused by intense photosynthesis. Dominance of the phytoplankton by blue-green algae is characteristic of water bodies in which the nitrogen : phosphorus ratio is low.

In this section on Chemical Limnology emphasis is placed on those chemicals of biological importance. The variation in their concentration levels over the three year period of the study is reported. These data are compared with chemical measurements made by Hutchinson, Pickford & Schuurman (1932) when the lake was oligotrophic and with later conditions described by Allanson & Gieskes (1961) and by Scott *et at.* (1977; 1980).

### 5.2.2 The oxygen regime and its relationship to physical characteristics

Hartbeespoort Dam is a warm monomictic lake (Robarts $_e$*t at'* 1982, Section 5.1). Both thermal and oxygen stratification develop in the lake during spring (September to November) and the lake remains stratified throughout summer. Stratification breaks down in late summer-early winter (March to May). A distinct, classically-defined thennocline has been observed, although the oxycline usually provides a more clearly defined index of stratification. The position of the oxy-thermocline varies during the season (Figs. 5.7 and 5.29) although it averages between 10 and 15 m for most of the season. During hydrological year (HY) 1983 and part of HY 1982 when draw down began having an effect on lake volume, wind-mixing had a greater effect on the oxy-thermocline causing greater fluctuations of the hypolimnetic surface. Scott *et at.* (1977) observed a similar instability in the oxy-thermocline during 1973-74 when the lake level was similarly low and their data shows possible polymicism (their Fig. 19, which shows oxygenation of the hypolimnion during January 1974). Scott *et aV*'s. Figure 10 shows also that the thermal regime was affected during this period. Increased nutrient concentrations coinciding with this period also act to substantiate this interpretation. Recent data (October to December 1983) also suggest that at low levels Hartbeespoort Dam can show signs of polymicism.

Oxygen saturations range from nil to over 100% with values of 150 - 200% not uncommon in surface waters during summer.

### 5.2.3 Inorganic chemistry

Using conductivity (Eg) as an index of the inorganic ion status of lake waters, Hartbeespoort Dam is fairly typical of most southern African man-made lakes (cf. Thornton & Nduku 1982). Conductivity ranged from 50 - 65 mS m"$^1$ during 1980 - 83 (Fig. 5.30) and has increased gradually since the study of Scott *et al.* (1977; Appendices 5.1 to 5.4 and Table 5.5). The increase in conductivity values in recent years may be a function of the lack of significant inflow rather than a real increase in mineralization in the lake catchment. However, it is difficult to accurately assess this trend until such time as more normal rainfall is experienced. At present, Hartbeespoort Dam is on the boundary of Tailing and Tailing's (1965) Class I soft water African lakes. The lake is alkaline (mean pH = 9.2) with an alkalinity of between 130 and 200 mg *Z*~$^l$ $CaC0_3$ .

Most of the inorganic ion concentrations remain relatively constant throughout the year. This is in agreement with the findings of Scott *et at** (1977) who noted slight changes in ion concentrations as a result of storm events and changing hydrological regime. Iron is an exception to this general rule undergoing a definite seasonal cycle (Fig. 5.31). Iron maxima occur during spring in the surface waters (September to November) whilst hypolimnetic maxima occur during winter. Cation dominance is in the order Mn<Fe<K<Mg<Ca/Na. Calcium/sodium appear to be co-dominant in Hartbeespoort Dam. Anion dominance is $Cl<SOi_t<HCO_3/CO_3$ . There has been no change in the order of dominance of the major ions in the lake since the study of Scott *et at** (1977) although slight increases in some ionic concentrations have been observed (Table 5.5). The increase in chloride concentrations, which has been widely publicised in the popular press due to alleged damage to tobacco crops, is such that Hartbeespoort Dam falls

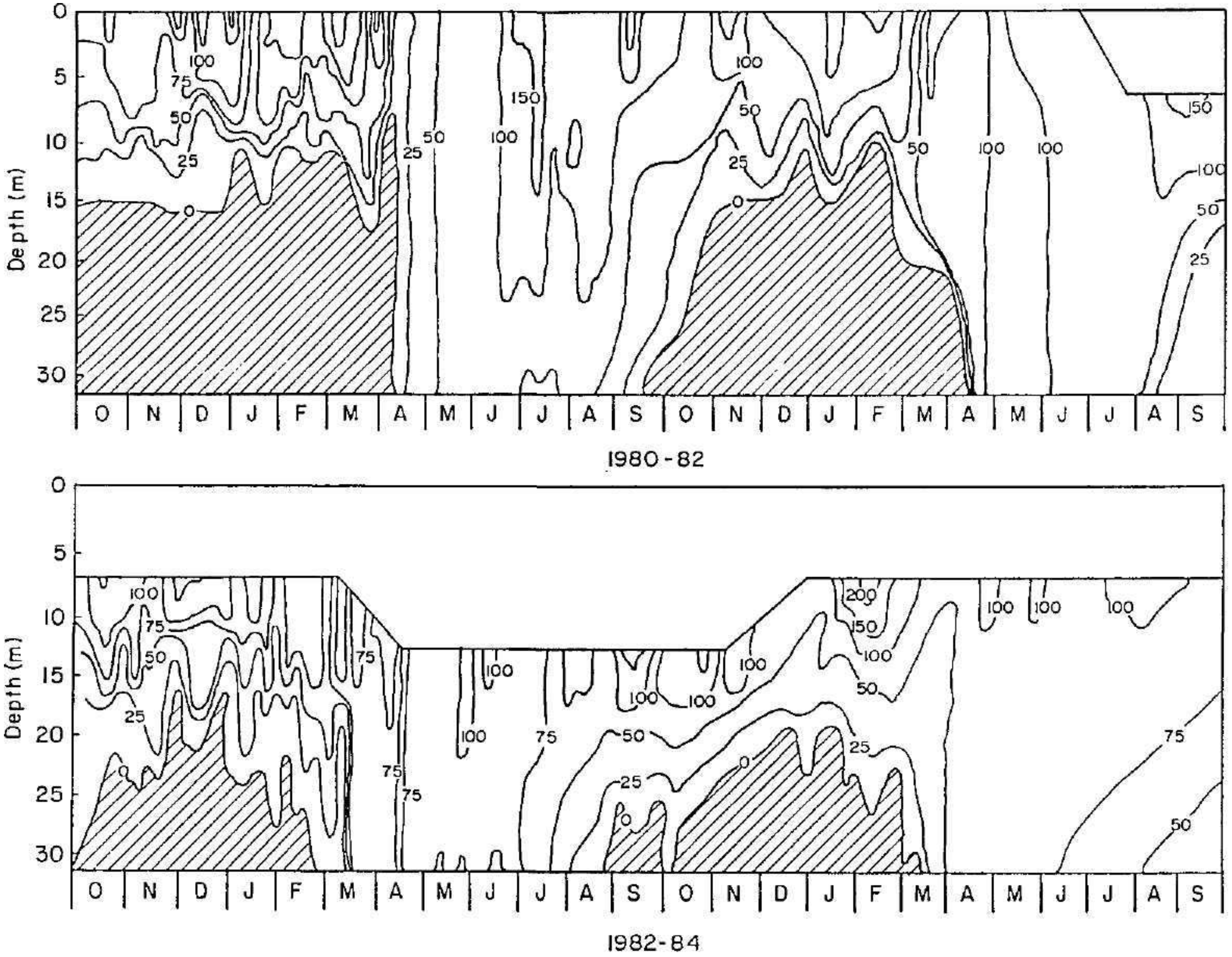


Figure 5.29. Dissolved oxygen isopleths showing changes in the percentage saturation in Hartbeespoort Dam during the period October 1980 to September 1984. Hatched area indicates anaerobic zone.

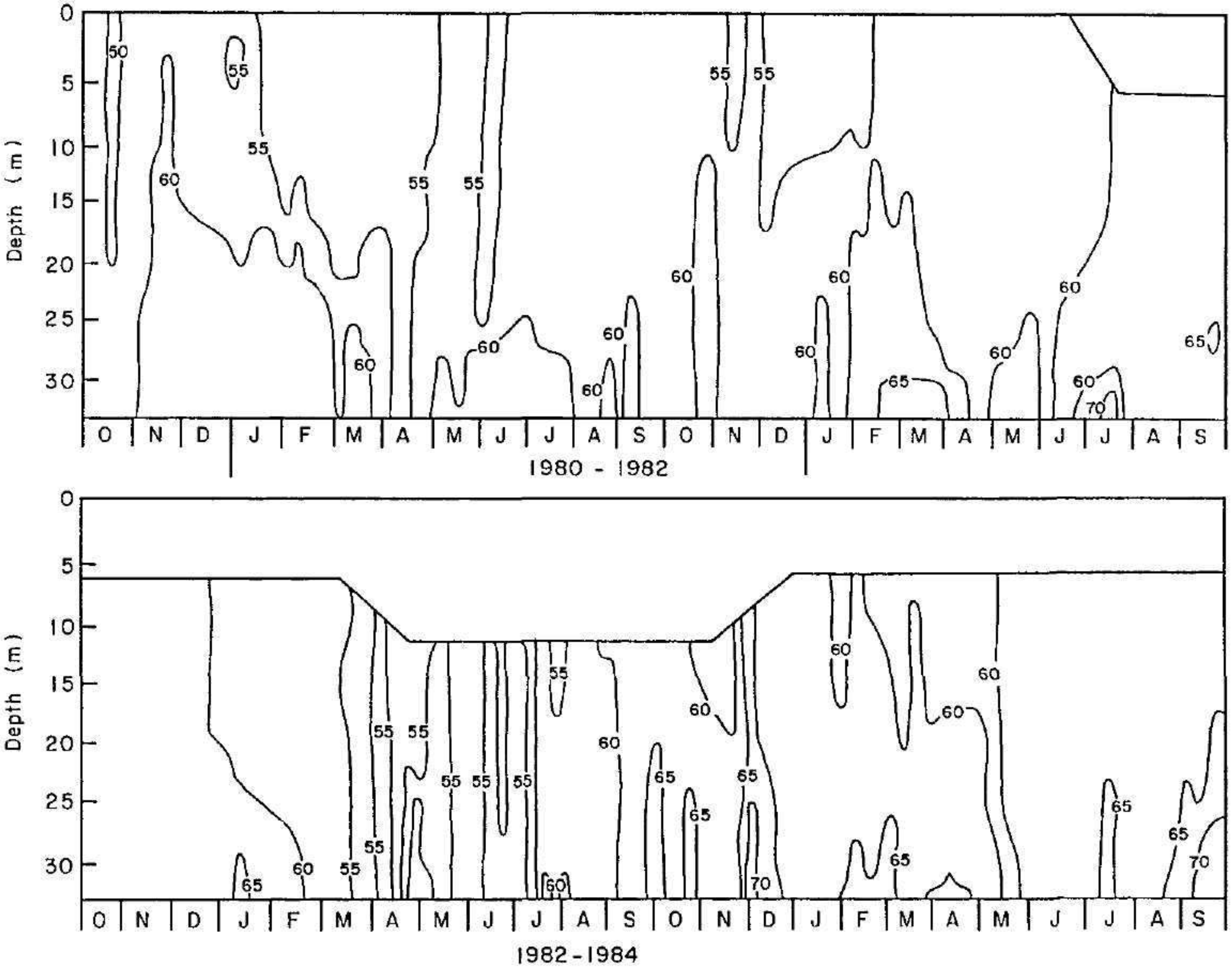


Figure 5.30. Conductivity isopleths showing changes in conductivity (mS $m^{-1}$) in Hartbeespoort Dam between October 1980 and September 1984.

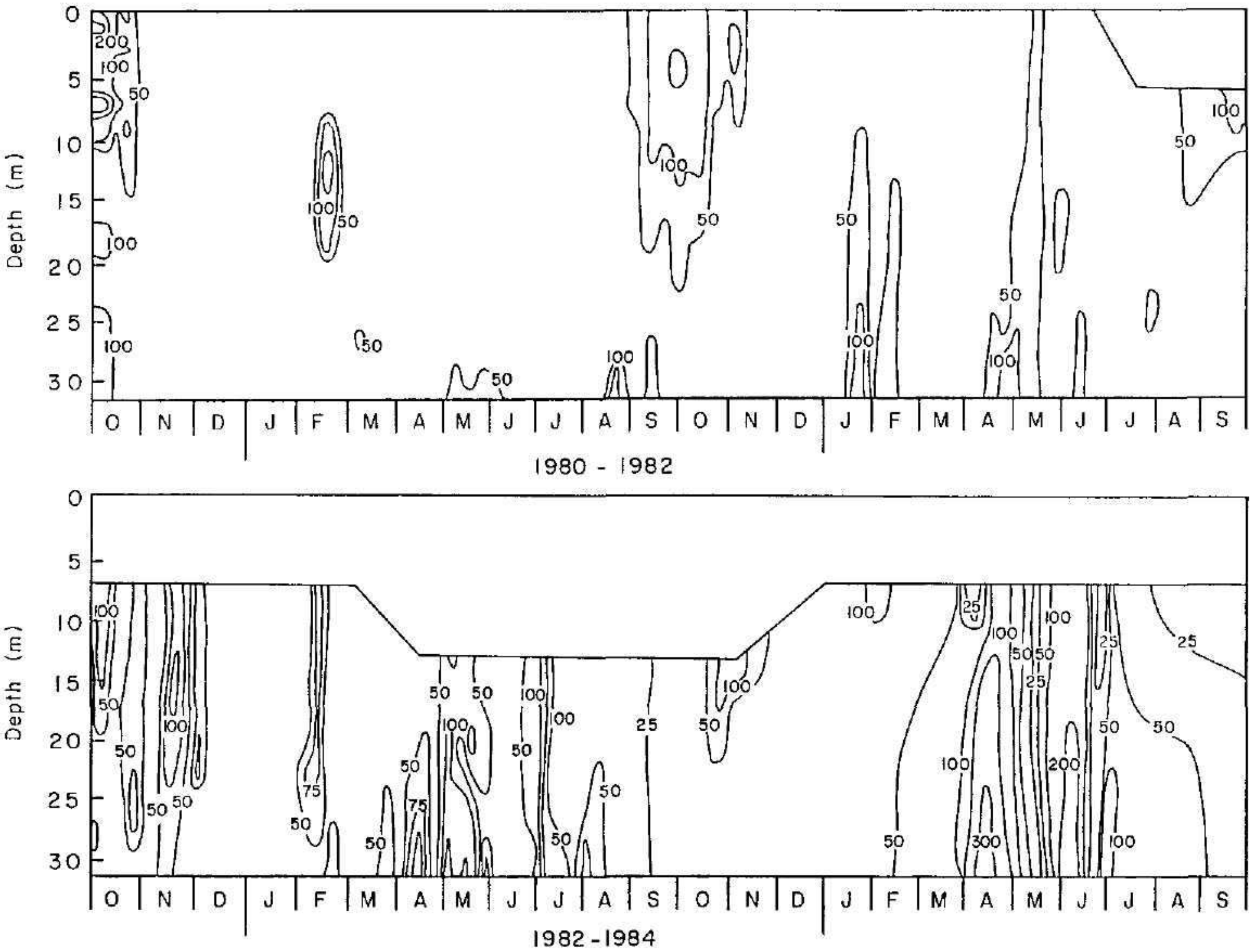


Figure 5.31. Distribution of dissolved iron (μg $\ell^{-1}$) in Hartbeespoort Dam during the period October 1980 to September 1984.

Table 5.5. Comparison of mean (range in brackets) surface water chemistry of Hartbeespoort Dam, 1928-1984. Concentrations in mg/ℓ .

| Parameter | 1928 | 1958 | 1974 | 1981 | 1982 | 1983 | 1984 |
|---|---|---|---|---|---|---|---|
| Nitrate-N | 0.68 | 2.27 | 2.09 | 0.93 | 1.10 | 0.94 | 1.46 |
| SRP | 0.005 | 0.05 | 0.64 | 0.30 | 0.39 | 0.42 | 0.52 |
| pH | 7.2 | (7.8-9.1) | (8.5-10) | 9.3 | 9.45 | 9.2 | 9.2 |
| Silica | – | 5.3 | 3.9 | 5.4 | 5.2 | 4.7 | 4.4 |
| Magnesium | 4 | – | 19 | 22 | 23 | 21 | 21 |
| Chloride | 7 | – | 42 | 43 | 48 | 65 | 58 |
| Sulphate | 10 | – | 64 | – | 100 | 113 | 114 |
| Conductivity (mS $m^{-1}$) | – | – | 50 | 55 | 59 | 59 | 67 |
| Alkalinity ($CaCO_3$) | 127 | – | 130 | 142 | 160 | 135 | 138 |
| Max. SDT[e] (m) | – | 3.8 | 3.0 | 3.0 | 2.9 | 5.9 | 3.1 |
| Reference | a | b | c | d | d | d | d |

a) after Hutchinson *et al.* (1932);
b) after Allanson & Gieskes (1961);
c) after Scott *et al.* (1977);
d) present study;
e) secchi disc transparency.

at the extreme of natural, non-saline waters (Wetzel 1975) but is not excessive. Nonetheless, any further increase should be monitored and possibly controlled should such increases be anthropologically generated and not an artefact of evaporative concentration (see below).

There have been few apparent changes in the chemical environment of the lake as a result of low rainfall and extensive drawdown. In general, it is difficult to distinguish between temporary effects such as (perhaps) the increase in conductivity values and longer term effects such as (perhaps) increasing chloride concentrations, although the latter decreased during early 1984 in response to the dilution factor resulting from rainfall (Table 5.5). Thus, in terms of the chemical environment of the lake, it is not possible to discuss the effect of the low rainfall period with any certainty except insofar as anion/cation concentrations have increased since 1980-81. These changes have been noted in the previous sections.

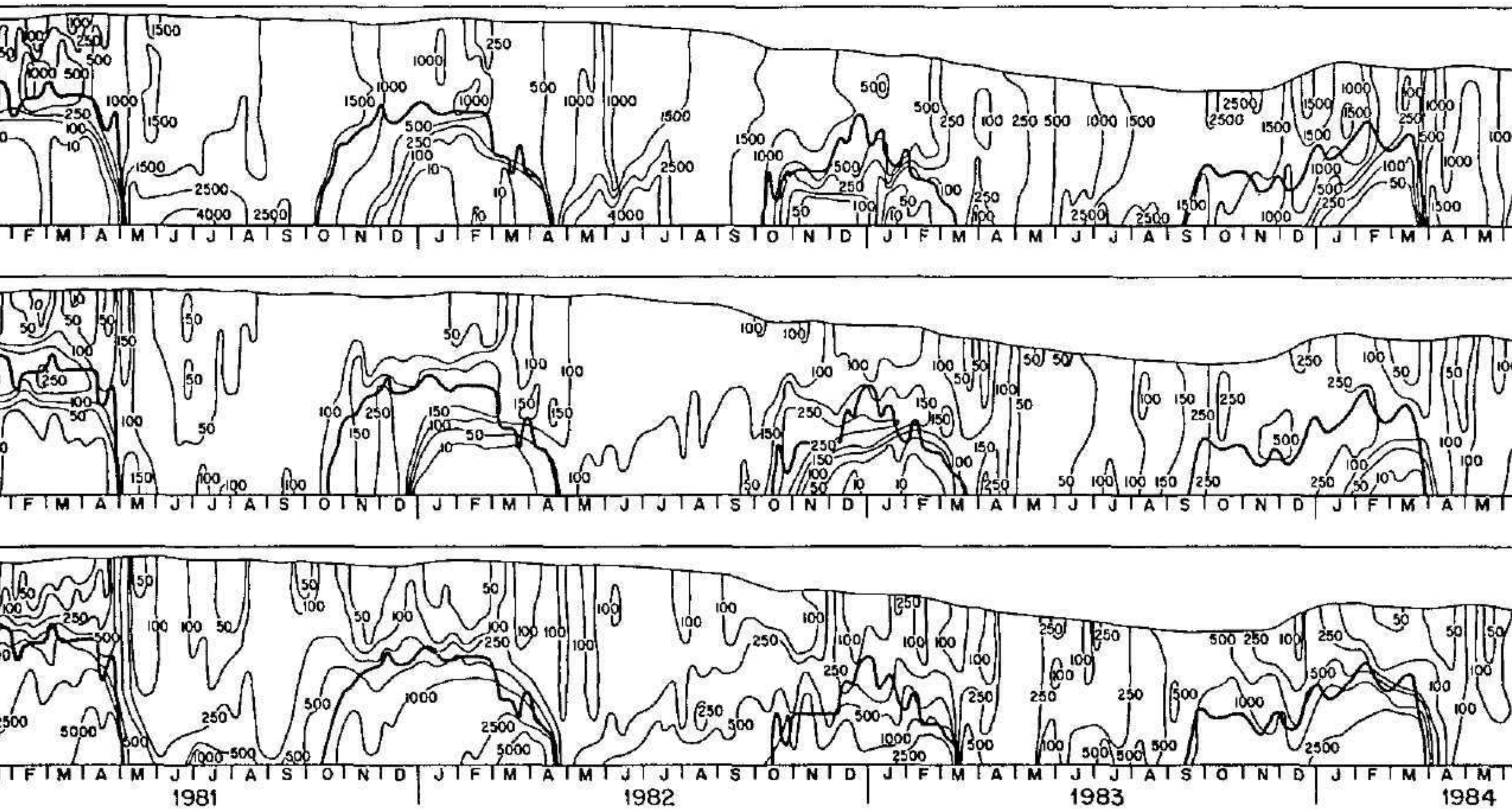
### 5.2.4 Nitrogen

In common with other eutrophic and hypertrophic lakes (Barica 1981), nitrogen concentrations in Hartbeespoort Dam are high, reflecting the degree to which the lake is enriched. Treated domestic and industrial effluents are the major source (> 90%) of the lake's nitrogen content (Toerien & Walmsley 1978, Scott *et at.* 1980). During recent years the total nitrogen concentration in the lake has steadily increased due to progressive urbanization and development within the catchment (Table 5.5). As a consequence, Hartbeespoort Dam has been likened to an oxidation pond (Cholnoky 1958, Allanson & Gieskes 1961) and is now regarded as one of the most hypertrophic lakes on record (Robarts 1984). Implementation of the proposed 1 mg *£* * phosphate standard by 1986 will, depending on the type of process employed, concomitantly reduce the nitrogen load to the impoundment and thus reduce the present rate of enrichment.

#### (a) In-lake nitrogen concentrations

In Hartbeespoort Dam the inorganic nitrogen forms $NO_3$-N, $NO_2$-N and $NH_4$-N make up between 50 and 70% of the total nitrogen content. The remainder is composed mainly of soluble forms of organic nitrogen (20 - 35%) and, to a lesser degree, live (phytoplankton, zooplankton) and dead (detritus) particulate organic nitrogen.

Depth-time isopleth plots of inorganic nitrogen concentrations ($NO_3$-N, $NO_2$-N and NH4-N) in Hartbeespoort Dam are shown in Figures 5.32A, B and C respectively, for the study period 80-10-20 to 84-07-31. In each diagram the zero mg $O_2$ *Z* $^{l}$ isopleth line demarcates the extent of the anaerobic hypolimnion. Short-term fluctuations brought about by physical factors such as wind-mixing and changing lake levels modified the distinct seasonal concentration patterns of each of the three nitrogen forms. However, despite this variability, the basic seasonal concentration patterns were repeated each year (Fig. 5.32) and are typical of highly enriched lakes (Hall *et at.* 1978, Jones *et al.* 1980, Bostrom 1981).



2. Depth versus time isopleth plots of A) $NO_3$-N, B) $NO_2$-N and C) $NH_4$-N concentrations in Hartbeespoort
1980-10-20 to 1984-08-27; after Ashton (1985). (All concentrations µg $\ell^{-1}$; dark line indicates ana
zone.)

The concentration patterns of $N0_3$-N (Fig. 5.32A) and $N0_2$-N (Fig. 5.32B) during the 46 month study showed several close similarities. Both compounds were characterized by steep concentration gradients during summer stratification and the lowest concentrations of both $N0_2$-N and $N0_3$-N were recorded in the anaerobic hypolimnion.

During summer $N0_2$-N concentrations increased sharply at or just above the oxycline and then decreased in the epilimnion. The highest summer concentrations of NO3-N were recorded above the oxycline in the epilimnion. After overturn, $N0_3$-N concentrations throughout the water column increased dramatically.

During winter, high NO2-N and $N0_3$-N concentrations were recorded immediately above the sediments. With the onset of summer stratification each year, $N0_2$-N and $NO_3$-N concentrations in the hypolimnion decreased rapidly; similar patterns have been recorded by Jones *et at.* (1980) and Bostrom (1981).

Conversely, ammonia ($NH_4$-N) concentrations were highest in the anaerobic hypolimnion during late summer, decreasing sharply above the oxycline to a minimum (< 250 ug -2 $^1$) in the aerobic epilimnion (Fig. 5.32C). The destruction of this steep concentration gradient during and after overturn has been described in detail by Robarts *et at.* 0 982). During winter isothermal conditions, NH4-N concentrations in the water column were generally low. The accumulation of $NH_4$-N in the bottom waters indicated that either NH4-N had diffused from the sediments or was lost as a result of the ammonification of sedimented detritus, or that both processes occurred simultaneously (Bostrom 1981). With the onset of summer stratification during September - October each year, dissolved oxygen concentrations in the hypolimnion were rapidly depleted. A steep NH^-N concentration gradient re-developed^ with low (< 250 yg *£* *) epilimnetic and higher (> 1 000 yg *S.* *) hypolimnetic concentrations. The extent of this summer NH4-N concentration gradient decreased from that recorded during the 1980-81 summer to that found during the 1983-84 summer.

Examination of the variations in $N0_3$-N concentrations in profundal waters 0.5 m above the sediment surface provide an insight into the nitrogen cycling processes that take place in this part of the water column (Fig. 5.33). During summer stratification, $N0_3$-N concentrations are usually less than 10 yg £ $^1$ whilst $NH_4$-N concentrations increase progressively from the onset of stratification to just before overturn due to the ammonification of sedimented organic material. Prior to complete overturn, NHi^N concentrations in the relatively small hypolimnion plummet to near zero as a result of dilution with epilimnetic water and nitrification while $N0_3$-N concentrations rise rapidly due to nitrification. From overturn until early August (the isothermal period) the well-mixed lake cools and loses sensible heat to the atmosphere (Section 5.1). During this period NH^-N concentrations remain low while $N0_3$-N concentrations show a steady if somewhat erratic increase as the NH4-N that diffuses out of the sediments is nitrified to $N0_3$-N (Cavari & Phelps 1977). When the lake gains heat during August and September, thermal stratification gradually develops and

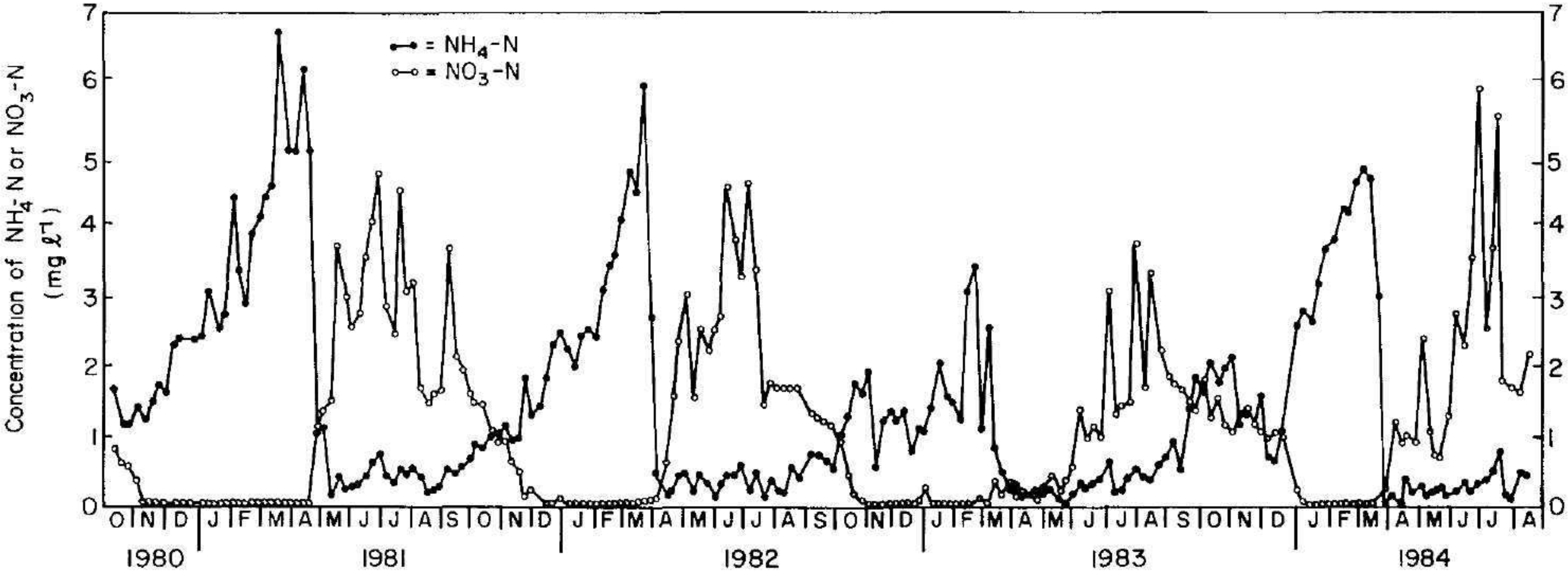


Figure 5.33. Concentrations of $NH_4$-N and $No_3$-N in the profundal zone (0.5 m above sediments) of Hartbeespoort Dam from 1980-10-20 to 1984-08-27. After Ashton (1985).

profundal oxygen concentrations decrease though a true anaerobic hypolimnion does not yet develop. In this period, NHif-N concentrations increase as nitrification slows and $N0_3$-N concentrations decrease as denitrification rates increase, When a distinct anaerobic hypolimnion develops, these trends are accentuated and the remaining $N0_3$-N is rapidly denitrified while NHi+-N concentrations continue to rise. Seasonal variations in the rates of nitrification and denitrification in Hartbeespoort Dam are described in Section 5.4.4.

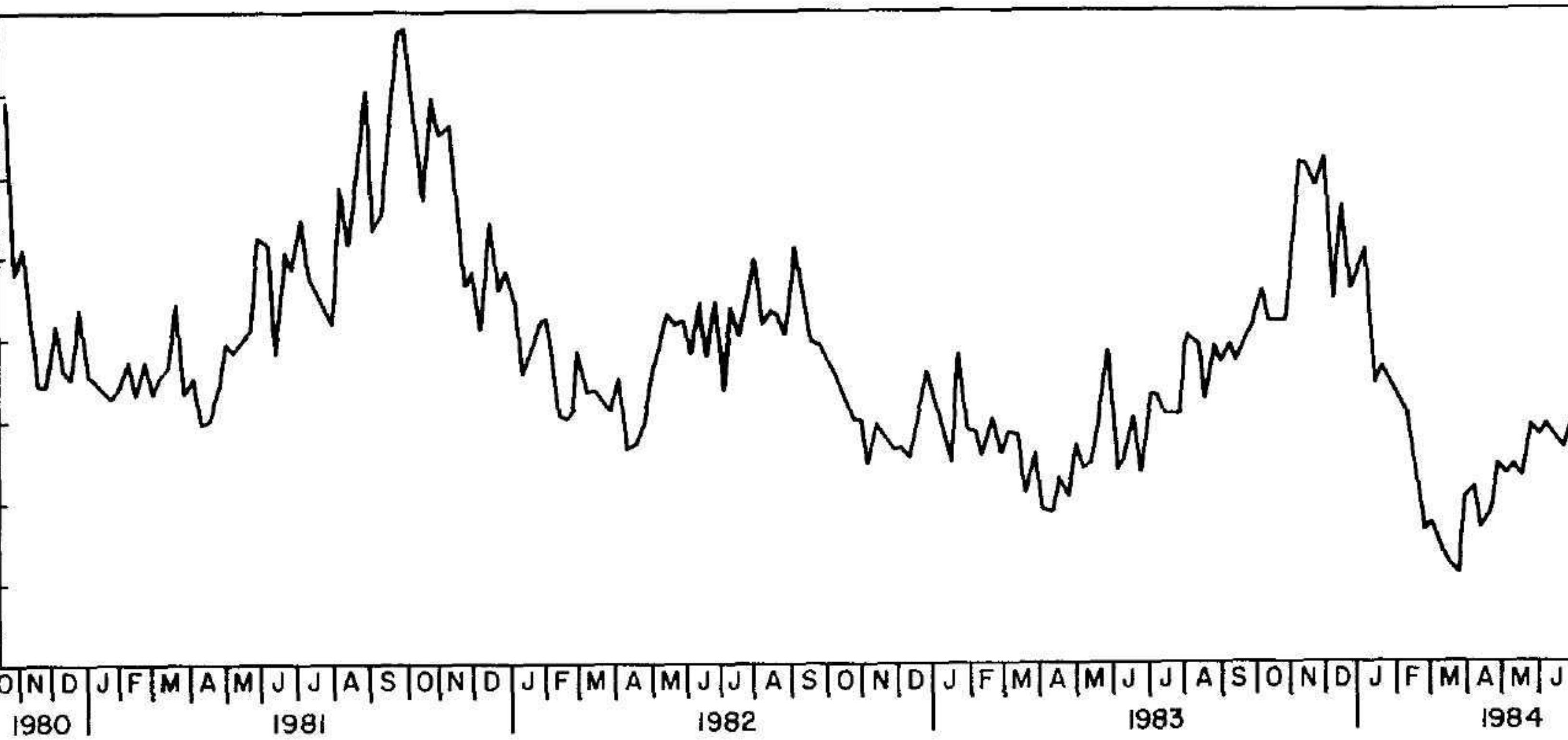
During this study, falling lake levels caused a decrease in the size of the anaerobic hypolimnion in successive summers from 1980-81 to 1983-84 and caused a progressive decrease in profundal NH4-N concentrations (Fig. 5.33). This trend was halted during January and February 1984 when a rise in lake level permitted deepening of the anaerobic hypolimnion and caused a sudden rise in NH^-N concentrations (Fig. 5.33).

### (b) N:P ratios

Weekly values of the ratio of total dissolved nitrogen to total dissolved phosphorus (the 'N:P ratio' on a mass basis) in Hartbeespoort Dam are shown in Figure 5.34. The N:P ratio tended to follow a distinct seasonal cycle in the lake, with minimum values recorded at or just before overturn and maximum values at the onset of stratification each year. Despite seasonal variations in the N:P ratio, the ratio remained below 10.0, a characteristic of eutrophic and hypertrophic lakes (Smith 1983). Indeed, the lowest value of 1.04 recorded on 21/3/84 is similar to values recorded in the highly enriched Rietvlei Dam (Ashton 1981). The highest N:P ratio found in Hartbeespoort Dam (7.83 on 1/10/81) is still below the level (10.0) considered to characterize highly eutrophied lakes (Smith 1983). The N:P ratios measured during successive annual overturns in Hartbeespoort Dam progressively decreased (Fig. 5.34). This may be related to increased rates of denitrification and sedimentation associated with the declining lake level. The present range of low N:P ratios found in the lake is ideal for the growth of blue-green algae (Smith 1983). However, should the N:P ratio continue to decrease, the lake will provide conditions suitable for the growth of nitrogen-fixing blue-green algae (Ashton 1979, 1981).

### (c) Inflow/outflow loads and the nitrogen balance

The first rough estimates of the nitrogen loads reaching Hartbeespoort Dam were made by Osborne & Halliday (1976). Subsequent, more accurate estimates have been made by Scott *et at.* (1977, 1980), Toerien & Walmsley (1978) and Thornton & Walmsley (1982). These authors demonstrated that the largest proportion (> 90%) of both the hydrological and nutrient loads is delivered to the lake via the Crocodile River. During this study inputs of nitrogen from the atmosphere and minor streams were assumed to be insignificant. Since the lake and its catchment are located in the summer rainfall zone, river flows are highly seasonal, with major inflows entering the lake during the summer months. The nitrogen loads reaching the lake show similar variability, and the highest loads enter the lake during summer.



gure 5.34. Weekly values for the whole-lake N:P ratio, (total dissolved nitrogen:total dissolved phosphorus), in Hartbeespoort Dam from 1980-10-20 to 1984-08-27. After Ashton (1985).

The quantity and composition of the total nitrogen load entering Hartbeespoort Dam during the first three and a half years of this study are shown in Table 5.6. Oxidized forms of nitrogen ($NO_2$-N and $NO_3$-N) comprised, on average, 84.7% of the total nitrogen load to the lake while NH^-N accounted for 2.9% and organic nitrogen 12.4% . The progressive decrease in the total nitrogen load to the lake reflects the progressively worsening drought that characterized the study period. Due to the progressive decrease in the nitrogen loads entering the lake, areal loading rates also declined though this was partly offset by the receding lake level.

Monthly values for the major components of the nitrogen balance in Hartbeespoort Dam are shown in Figure 5.35. During the first 20 months of the study, the lake level remained virtually constant at full supply capacity. During this period the combined nitrogen content of the lake fluctuated between 305 and 480 tonnes in response to the variable inflow and outflow loads as well as to losses by sedimentation and denitrification. The highly variable rates of nitrogen loss by net sedimentation (Fig. 5.35) were due to the interactions between gross sedimentation and resuspension which are driven by climatic and hydrodynamic forces. Since net sedimentation of nitrogen is calculated from an average N:P ratio in the sediments, variations in this quantity reflect variations in the net sedimentation of phosphorus (Fig. 5.37B).

Between May 1982 and November 1983 the lake level fell 12 m (a 70% drop in volume) due to a combination of increased draw-offs of irrigation water and reduced river inflows. In the same period the lake nitrogen content fell from 350 to 60 tonnes, a decrease of 83% . The major causes of this loss were net sedimentation and denitrification (Fig. 5.35). Reduced denitrification during the remainder of 1983, combined with increased inflow loads, caused a rise in lake nitrogen content. Typically, denitrification losses were greatest during the spring, summer and autumn months each year. This feature supports the observations of Messer & Brezonik (1978, 1983) who observed si-

**Table 5.6.** Areal loading rate and composition of nitrogen loads entering Hartbeespoort Dam.

| Hydrological year | Areal Loading rate ($g\ m^{-2}a^{-1}$) | Total load (tonnes) | Load composition (%) | | |
|---|---|---|---|---|---|
| | | | $NH_4$-N | ORGANIC N | $NO_2$-N+$NO_3$-N |
| 1980-81 | 88.0 | 1709 | 3.5 | 15.7 | 80.0 |
| 1981-82 | 79.6 | 1448 | 2.1 | 16.8 | 81.1 |
| 1982-83 | 76.0 | 771 | 2.5 | 8.8 | 88.7 |
| 1983-84* | 58.5* | 482* | 3.3 | 8.4 | 88.3 |

*Based on data from first 6 months.

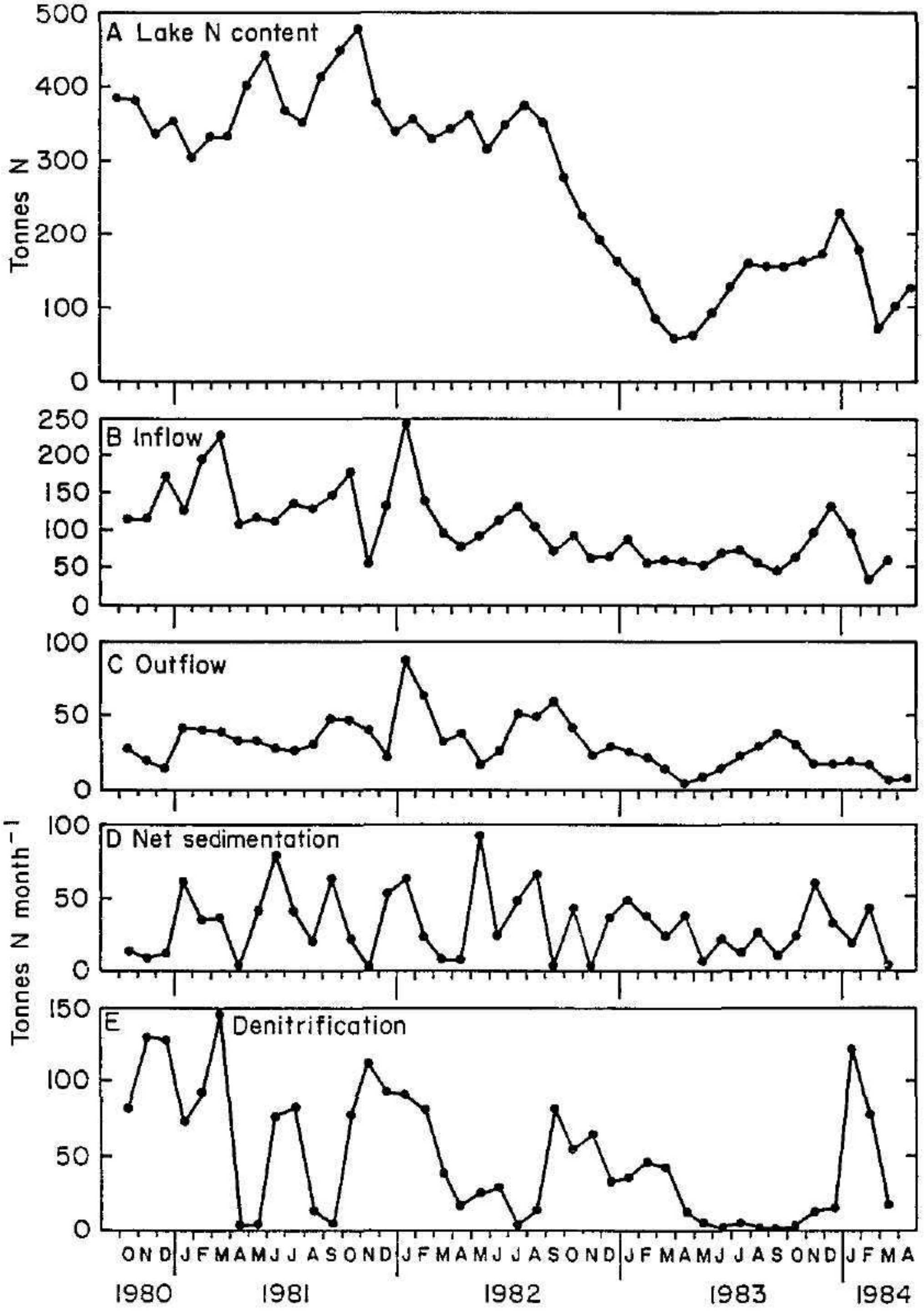


Figure 5.35. Monthly values for the major components of the Hartbeespoort Dam nitrogen balance, from October 1880 to April 1984.

milar trends in Lake Okeechobee, Florida. The data shown in Figure 5.35 have been recalculated on an annual basis and are shown in Table 5.7.

From the data presented in Table 5.7 it is evident that sedimentation and denitrification accounted for the major proportion of the nitrogen lost from the lake, again supporting the observations of Messer & Brezonik (1978). Each year, denitrification losses from the lake were equivalent to between 39 and 49% of the total annual nitrogen inflow load via the Crocodile and Maealies Rivers.

Table 5.7. Summarized annual nitrogen budget for Hartbeespoort Dam. (All values are in metric tonnes of N). Values in parentheses indicate percentage of inflow load.

| Hydrological year | Initial N content of lake | ΔN | Inflow | Outflow | Sediments | Denitrification |
|---|---|---|---|---|---|---|
| 1980-81 | 388 | +62 | 1709 | 387<br>(22.6%) | 419<br>(24.5%) | 841<br>(49.2%) |
| 1981-82 | 450 | -173 | 1448 | 540<br>(37.3%) | 418<br>(28.9%) | 663<br>(45.8%) |
| 1982-83 | 277 | -118 | 771 | 282<br>(36.6%) | 304<br>(39.4%) | 303<br>(39.3%) |

### (d) Nitrogen in Hartbeespoort Dam sediments

Chemical analysis of 48 sediment samples collected from different areas in Hartbeespoort Dam showed extremely high variability in nitrogen content (x = 0.36%; cv = 122%), with the highest values recorded in sediments from the Crocodile River arm of the lake. A more detailed description of Hartbeespoort Dam sediments is given in Section 5.3,1. Because of extremely high variability in the nitrogen and phosphorus contents of the sediments, the sediment N:P ratio varied from 1,02 to 10.69 with a mean of 2.04 and no spatial separation of different sediment types could be made.

Vertical profiles of different inorganic nitrogen forms in sediment cores collected from aerobic (marginal) and anaerobic (profundal) zones in the lake also showed great variability. Representative profiles of N0$_3$-N and NH$_4$-N in the two sediment types are shown in Figure 5.36. The 'marginal$^1$ core is characterized by a shallow (s 1 cm) aerobic zone with high N0$_3$-N concentrations overlying a deeper anaerobic zone where N0$_3$-N concentrations are virtually undetectable. In the aerobic zone, NH^-N concentrations are low, increasing sharply in the anaerobic zone. The 'profundal$^1$ core was characterized by an almost complete lack of N0$_3$-N throughout its depth and very high NH^-N concentrations (Fig. 5.36). Further cores collected from the profundal zone after overturn showed the development of a shallow (ca. 3 mm) aerobic microzone at the surface^ This microzone contained high concentrations (> 2 500 yg £ $^1$) of both N0$_3$-N and NH$_4$~N and supported high rates of nitrification and denitrification (Section 5.4.4).

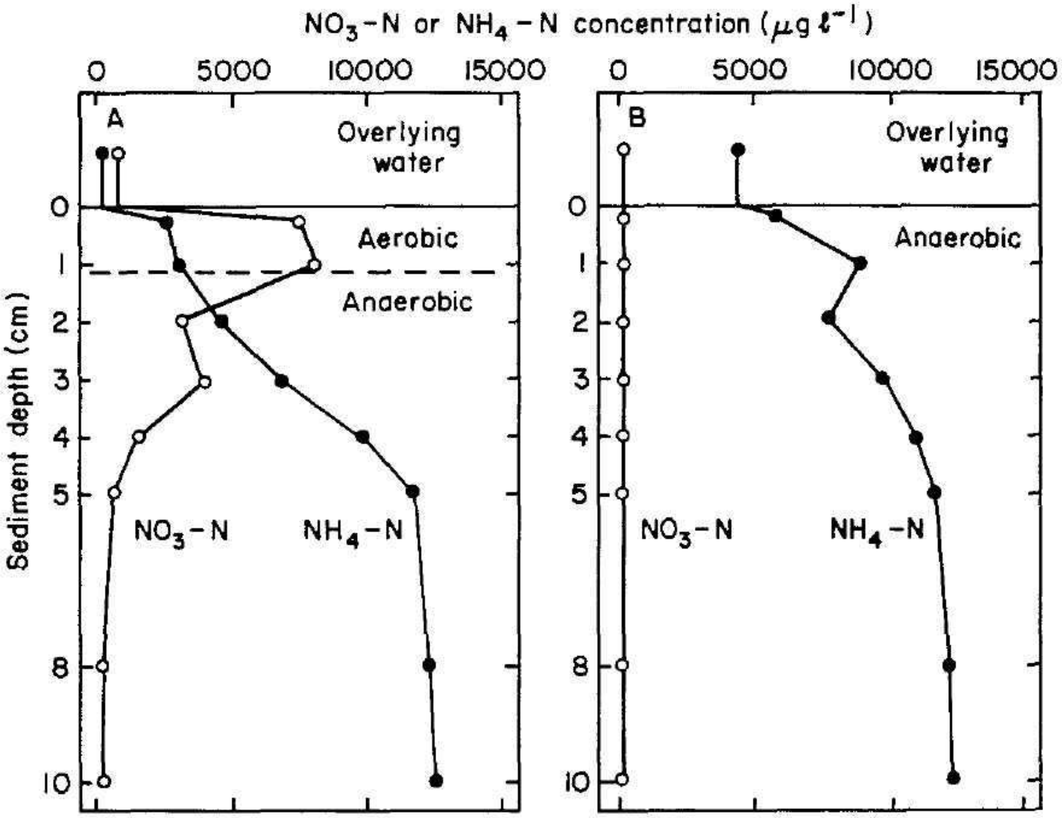


Figure 5.36. Comparison of vertical pore-water $NO_3$-N and $NH_4$-N concentration profiles in A) Marginal and B) Profundal sediments from Hartbeespoort Dam. (Samples collected on 28 January 1984). After Ahston (1985).

### 5.2.5 Phosphorus loads and the phosphorus balance

Estimates of phosphorus loads were first made for Hartbeespoort Dam in 1975 by Toerien & Walmsley (1976) and Osborn & Halliday (1976). Estimates of the 1974 phosphorus loads to Hartbeespoort Dam are given in Table 5.8, together with more recent estimates made by Scott *et at.* (1980) and other authors (c.f. Thornton & Wrlmsley 1982 and this study). All of these studies have found that the major source of phosphorus to Hartbeespoort Dam is the Crocodile River. Although there has been some hydrologically-induced variability, the mean phosphorus load to the lake was 276 ± 130 s.d. tonnes P (n = 7)

Monthly values for the components of the phosphorus balance are shown in Figure 5.37. During the first 20 months of the study when the lake level remained virtually constant at full supply capacity, the phosphorus content of the lake fluctuated between 69 and 115 tonnes in response to variable inflow and outflow loads. Between May 1982 and November 1983 the 12 m drop in lake level (a 70% drop in volume) was accompanied by a drop in phosphorus content from 115 to 31 tonnes, a decrease of 73%.

Seasonal cycles were pronounced in the lake (Fig. 5.37A), but were moderated in the inflows due to the higher inflow loads of phosphorus which was associated with increased river flows in summer and the high proportion of sewage-derived phosphorus in the Crocodile-Jukskei River which maintained high phosphorus loads during winter (Fig. 5.37B).

Losses of phosphorus via the outflow (Fig. 5.37(C)) were dependent on outflow hydrology; high outflow loads always being associated with increased draw-offs of irrigation water during both spring and winter planting periods. The net sedimentation losses shown in Figure 5.37D were highly variable and no seasonal pattern was evi-

**Table 5.8.** Comparison of phosphorus loads to Hartbeespoort Dam. Loads in metric tonnes.

| | Year | Total-P | Reference |
|---|---|---|---|
| Inflow | 1973 | 260 | Osborn & Halliday (1976) |
| | 1974 | 336 | Osborn & Halliday (1976) |
| | 1975 | 350 | Toerien & Walmsley (1976) |
| | 1977 | 418 | Scott *et al.* (1980) |
| | 1981 | 283 | This study |
| | 1982 | 323 | This study |
| | 1983 | 206 | This study |
| Outflow | 1981 | 80 | This study |
| | 1982 | 107 | This study |
| | 1983 | 86 | This study |

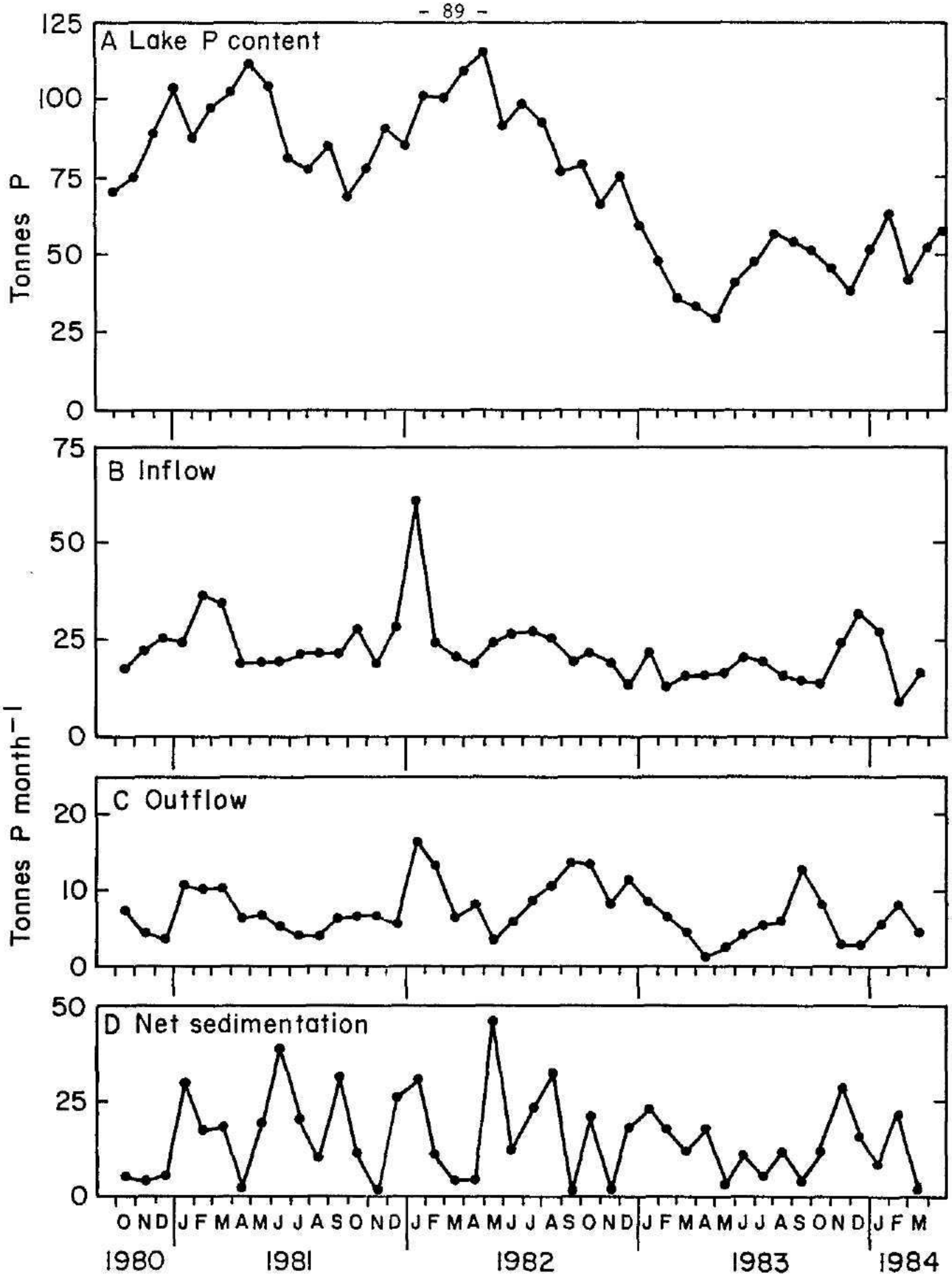


Figure 5.37. Monthly values for the major components of the Hartbeespoort Dam phosphorus balance, from October 1980 to April 1984.

dent. This was brought about by interactions between gross sedimentation and resuspension which were, in turn, driven by hydrodynamic and climatic forces.

The range of total phosphorus concentrations in the lake was 0.4 mg ^ $^{l}$ to over 2.0 mg *Z*~$^{x}$ (Fig. 5.38), which is greater than the 0.3 to 1.3 mg *I* $^{1}$ reported by Scott *et at.* (1977). Annual maximum phosphorus concentrations occurred in the hypolimnion in summer, whilst minimum concentrations occurred in the surface waters at various times of the year. The distribution of other phosphorus fractions in Hartbeespoort Dam surface water for 1982-83 is shown in Figure 5.39. SRP formed about 60% of the TP concentration in the lake. Of this, LMWP (low molecular weight phosphorus eluting at approximately 2 x void volume) accounted for between 70 and 100% of the SRP concentration (over 100% recovery was due to the analytical error which was estimated to be about 10%). HWMP (high molecular weight phosphorus eluting at void volume) was rarely observed, but was detectable on five occasions during winter circulation. Maximum concentrations of HMWP were observed just prior to the onset of stratification and amounted to < 10% of the TP concentration, or between 5 and 15% of the SRP concentration. Unreactive (SUP) and particulate (PP) phosphorus fractions accounted for 16% and 26% of the TP concentration on average, respectively. No XP (a low molecular weight compound defined by Lean 1973) was distinguished, presumably due to its absence in the system. If an XP fraction, molybdate reactive, was present in Hartbeespoort Dam, it is probable that it was included in the LMWP fraction with which it would elute due to the short columns employed in this study (Clarke, NCRL, pers. comm.).

The high concentration of SRP in Hartbeespoort Dam relative to the TP concentration was characteristic of a eutrophic impoundment (Peters 1979, OECD 1982). Downes & Paerl (1978) and White & Payne (1980) suggest that the dominance of orthophosphate-LMWP in this fraction is common. In New Zealand lakes, orthophosphate LMWP accounts for between 63 and 100% of the SRP concentration, although the percentage of LMWP was found to decrease with increasing trophic status (White & Payne 1980). This observation led them to conclude also that detectable quantities of HMWP were indicative of eutrophic systems, as this fraction increased in concentration as LMWP decreased in their lakes. HMWP formed between 2 and 76% of the SRP concentration in the New Zealand lakes which were located predominantly in agricultural catchments. HMWP concentrations found in Hartbeespoort Dam fall at the lower end of this scale (5-15%) which suggest that the findings of White & Payne (1980) might be related to catchment land use. The major source of P to Hartbeespoort Dam was treated municipal wastewater discharged into the Crocodile River, which has apparently led to lower HMWP concentrations being observed in the lake. This aspect would require further investigation to clarify the reason for the differences between Hartbeespoort Dam and the New Zealand lakes.

White & Payne (1980) also showed increased HMWP concentrations at the end of summer (their data for Lake Okaro show an increase in HMWP from 0 to 25% of the SRP concentration between March and May 1978). This seasonal distribution is similar to that observed in Hartbeespoort Dam where HMWP was measured following overturn in 1982 (0 - 58% of the SRP concentration). The occurrence of HMWP in Hart-

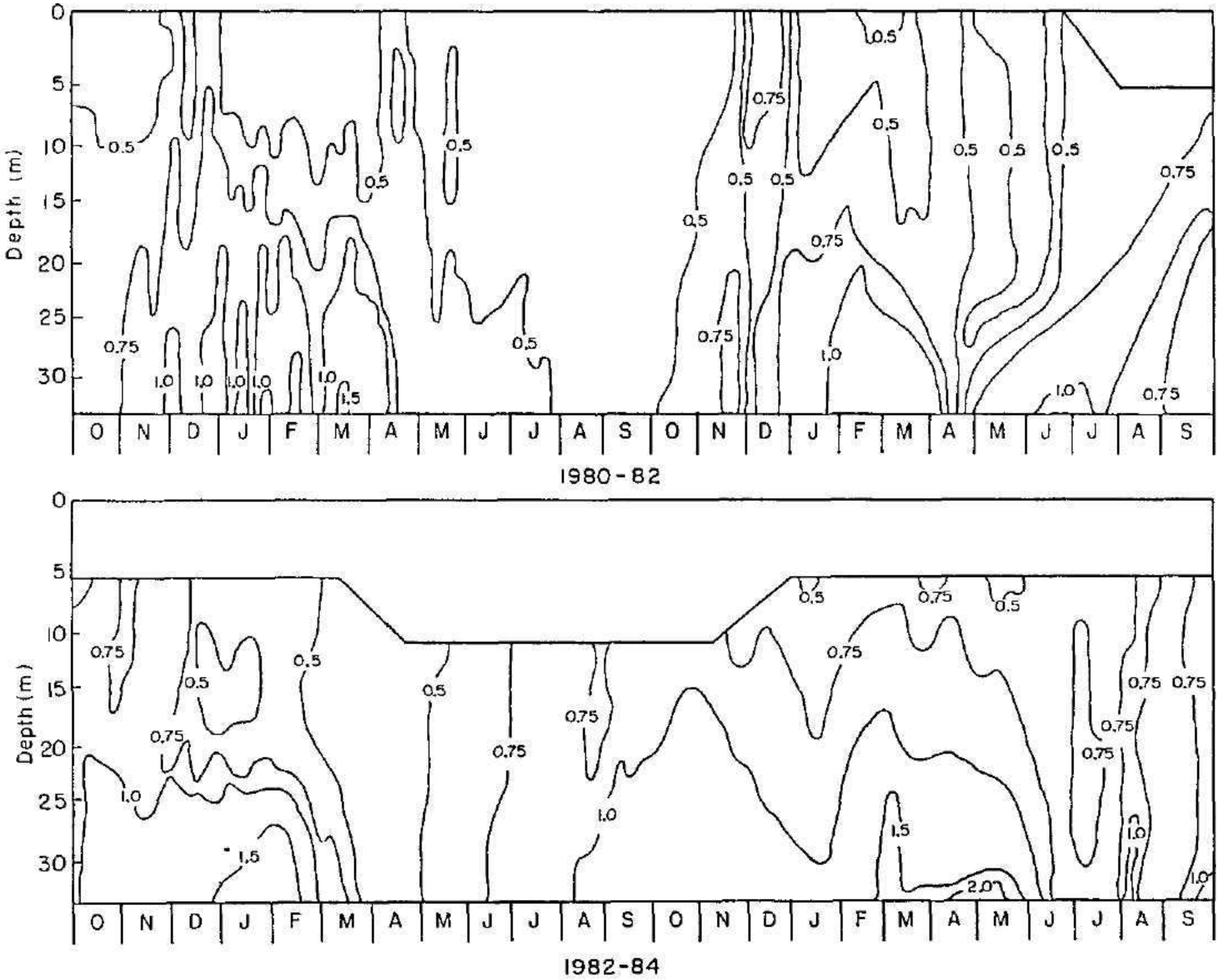


Figure 5.38. Total phosphorus isopleths showing changes in total phosphorus concentration (mg $^{-1}$) in Hartbeespoort Dam between October 1980 and September 1984.

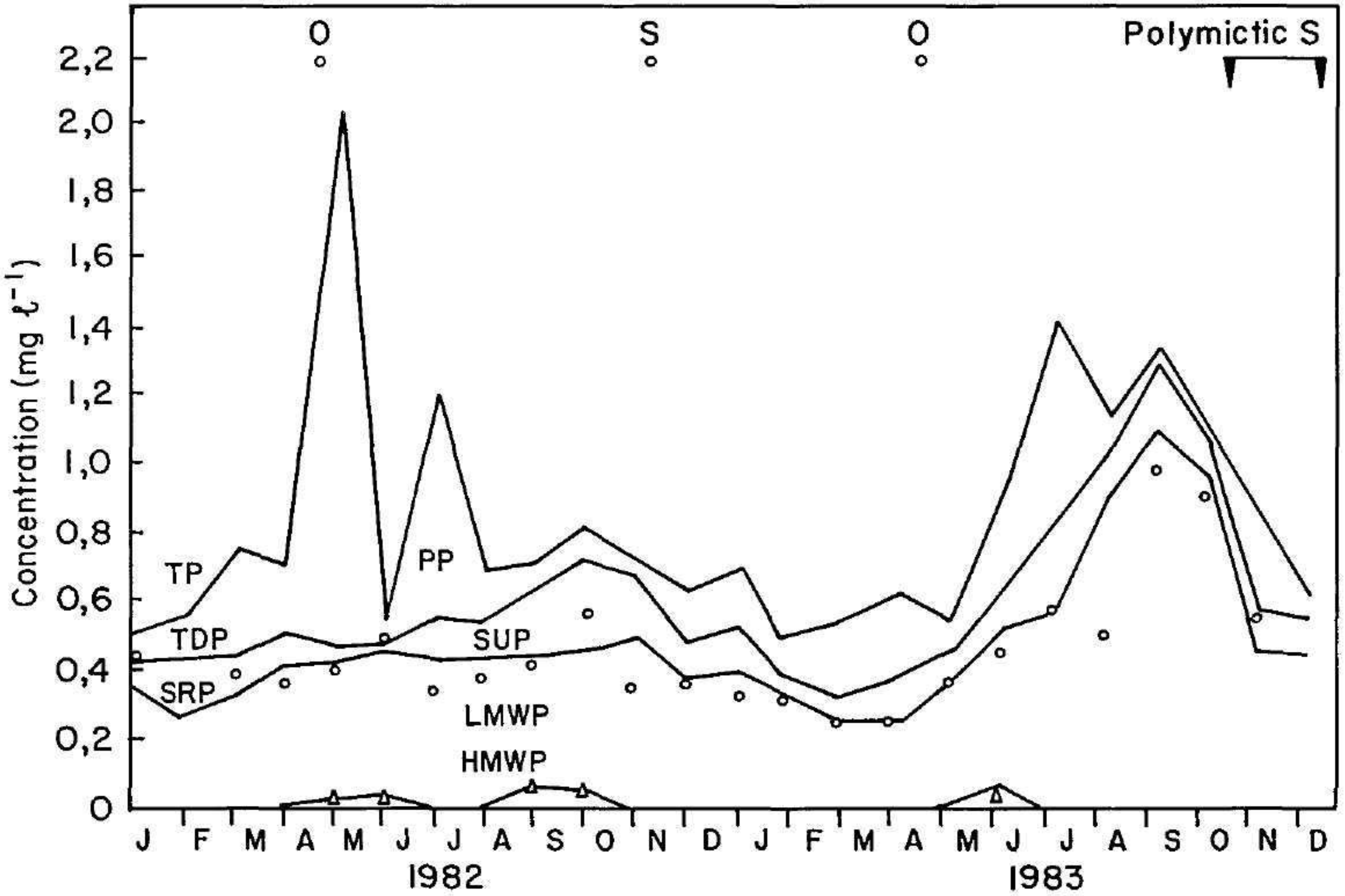


Figure 5.39. Distribution of phosphorus in surface waters of Hartbeespoort Dam between January 1982 and December 1983. *From top to bottom, lines show variations in total phosphorus concentration* (TP, mg $\ell^{-1}$), total dissolved phosphorus concentration (TDP, mg $\ell^{-1}$), and soluble reactive phosphorus concentration (SRP, mg $\ell^{-1}$). The difference between TP and TDP represents particulate phosphorus concentrations (PP), between TDP and SRP soluble, unreactive phosphorus concentrations (SUP, mg $\ell^{-1}$). Open circles show the distribution of low molecular weight phosphorus concentrations (LMWP, mg $\ell^{-1}$) and triangles the distribution of high molecular weight phosphorus (HMWP, >5000 MW units, mg $\ell^{-1}$). Points of stratification are indicated by 'S' and points of overturn by 'O'.

beespoort Dam may be related to the increase in bacterial numbers. In May-June 1982, following overturn, bacterial populations doubled (from <10 x $10^6$ to >21 x $10^6$ *m2~*$^{l}$*)*, and similarly_prior to stratification, bacterial numbers exceeded 37 x $10^6$ m£ $^1$ (Fig. 5.50). A similar peak was observed in June 1983 (Fig. 5.51). The simultaneous occurrence of HMWP and aquatic bacteria has not been reported in the literature.

## 5.3 SEDIMENTS

Sediments have been shown to play a central role in phosphorus cycling in lakes (Syers *et at,* 1973, Bostrom *et at.* 1982), particularly in relation to eutrophication management through nutrient load reduction. Lakes retain large amounts of phosphorus in the sediments and during eutrophication this build up is accelerated. Some of the sediment phosphorus becomes irreversably bound by geochemical transformations, but some remains loosely bound and can be transported into the overlying water under suitable conditions, thereby becoming available to organisms in the water. This internal phosphate loading can delay, or even prevent, trophic recovery in eutrophic lakes in which rehabilitation measures are implemented. Quantification of sediment/water phosphate flux, and identification of the factors that control the rates and directions of flux, are therefore important requirements for effective eutrophication control. This section addresses the question of sediment/water phosphate exchange with a view to incorporating it into the Hartbeespoort Dam ecosystem model.

### 5-3.1 Characterisation and distribution

Physico-chemical characteristics of bottom sediments from Hartbeespoort Dam, compared with those from other South African impoundments (Grobler & Davies 1981) where possible, are summarised in Table 5.9. Variability between samples was generally high for all parameters measured suggesting a heterogenous distribution of sediments in the dam. The importance of overlying water depth in controlling sediment distribution and determining sediment characteristics is evident in its significant ($p = 0.05$) negative correlation with equilibrium phosphate, pore water SRP and pH and its positive correlation with total phosphorus (TP), biological available phosphorus (BAP), organic carbon and loss on ignition. These relationships indicate that the finer grained deep water sediments, resulting from 'sediment focussing$^1$ (Evans & Rigler 1983), are more acid and contain more total phosphorus (TP) and organic carbon (OC), but less pore water SRP, than those in marginal zones.

In a dendritic impoundment with two river inflows contrasting in chemical and hydrological characteristics, and entering remote 'river-like$^1$ arms, site dependent differences in sediment characteristics would be expected (Bostrom *et at.* 1982). When sediment data from Hartbeespoort Dam was categorised into Main Basin, Magalies River Arm and Crocodile River Arm, as shown in Figure 4.1 (Section 4.3.3), with the exception of silt content which was highest in the Crocodile River Arm, no site specific trends in the non-phosphorus parameters were evident. However, central tendencies for all phosphorus fractions measured in the Crocodile River Arm were significantly higher (P - 0.05) than those at the other two

Table 5.9. Descriptive statistics for sediment characteristics in Hartbeespoort Dam and other South African impoundments. Means and ranges presented (others from Grobler and Davies 1981).

| Units | Parameter | Hartbeespoort Dam | Others |
|---|---|---|---|
| mg g$^{-1}$ | Total P | 0.99(0.18-2.73) | 0.38(0.15-0.98) |
| mg g$^{-1}$ | Bioavailable P | 0.58(0.11-1.88) | 0.04(0.0 -0.17) |
| mg ℓ$^{-1}$ | Pore water SRP | 0.68(0.0 -2.7) | - |
| mg ℓ$^{-1}$ | Equilibrium P | 0.24(0.17-0.95) | - |
| % | Organic Carbon | 2.9 (0.1 -5.8) | 1.5(0.6-4.0) |
|  | pH | 7.5 (6.9 -8.4) | 6.7(4.9-8.3) |
|  | Particle Size |  |  |
| % | >300μm (sand) | 45 (6-93) | - |
| % | 50-300μm (Silt) | 46 (5-84) | - |
| % | <50μm (Clay) | 9 (0-46) | - |

- data not available

sites (Fig. 5.40). Since the main river inflow enters via the Crocodile River Arm, this suggests that a large proportion of the incoming phosphorus load is lost to sediments nearest the inflow, probably via silt deposition, and that the characteristics of sedimenting material varies with site in the dam (Section 5.3.3). On the basis of this sediment survey it is not possible to classify Hartbeespoort Dam according to a single sediment type. Although the phosphorus fractions in Hartbeespoort Dam sediments are at least twice as high as comparable mean values for other South African impoundments, and generally higher than most recorded values (Boström *et at** 1982) due to its hypertrophic condition, factors such as morphometry and loading pattern interact to create a wide range of sediment types in the dam (Table 5,10).

### 5.3.2 Sediment/water phosphate exchange

(a) The relationship between phosphate concentration in the overlying water and sediment/water phosphate exchange

The ability of Hartbeespoort Dam sediments to take up or release phosphate, depending on concentrations in the overlying (or surrounding) water was demonstrated using sediment subsamples in suspension and intact sediment/water systems with continuous flow adaptors (Fig. 5.41). Over the range of phosphate concentrations used in these experiments both methods yielded linear relationships between phosphate concentration in the water and the amount of phosphate taken up or released from sediments. The existence of a sediment/water phosphate equilibrium, facilitating phosphate release when concentrations in the water drop below equilibrium and phosphate uptake by sediments when the equilibrium level in the water is exceeded, is clear from these results. However, under natural conditions the estimates using intact sediment cores are more applicable because they provide more realistic simulations of areal exchange between bottom sediments and the overlying water.

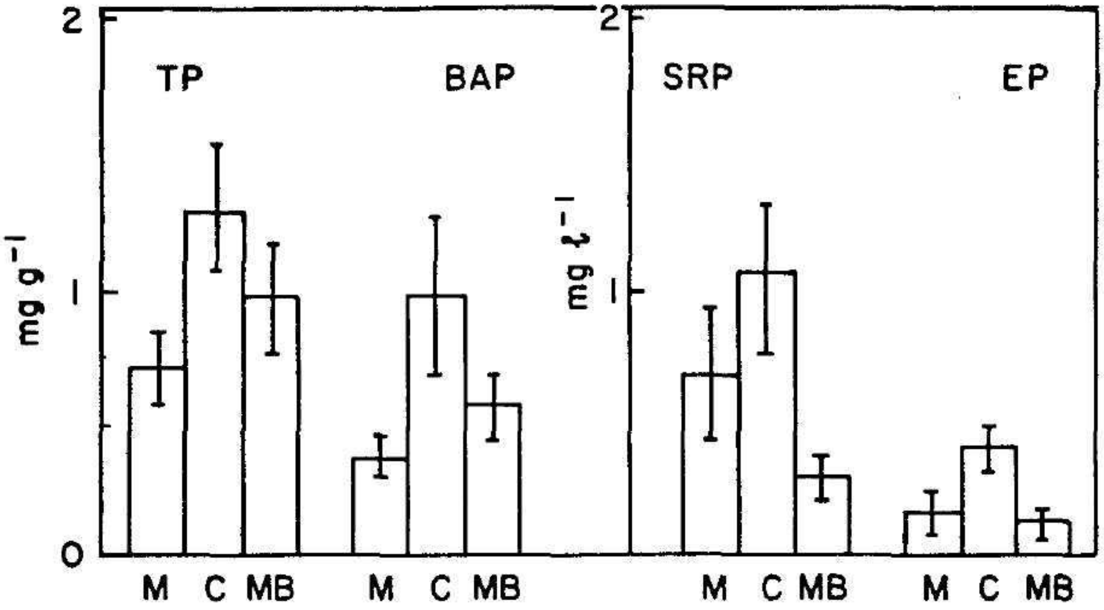


Figure 5.40. A comparison of mean values for various phosphorus fractions in sediments at different sites in the dam. TP = total phosphorus; BAP = biologically available phosphorus; SRP = soluble reactive phosphorus in the pore water; EP = equilibrium phosphorus; M = Magalies Arm; C = Crocodile Arm; MB = Main Basin. Confidence bands (P = 0.05) indicated as bars.

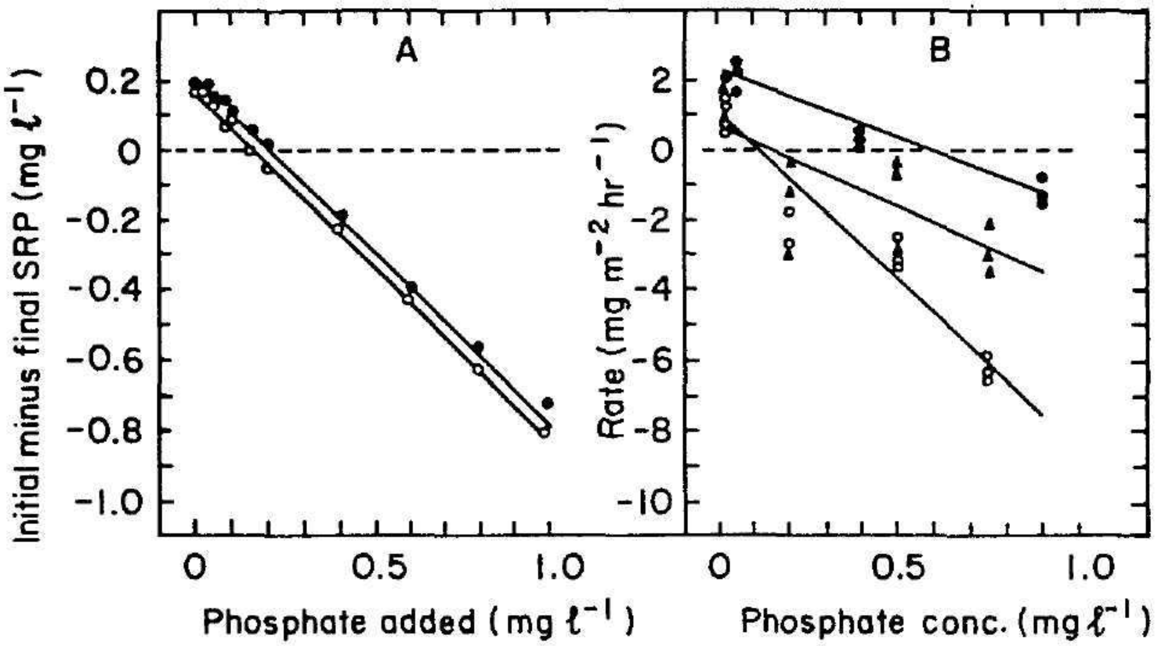


Figure 5.41. Phosphate uptake/release by sediments under aerobic conditions treated with varying concentrations of phosphate A) Using sediment subsamples B) Using continuous-flow systems with intact cores. Symbols represent different sites in the Crocodile River arm (Twinch 1984).

Table 5.10. Mean values for regression constants derived from relationships similar to those show in Figure 5.41 at various sites in Hartbeespoort Dam. The phosphate uptake release characteristics are described by simple linear equations:

(y = mx + c) where:
y = phosphate flux rates (mg m $^{2}$ h $\bar{*}$)
m = slope
c = y intercept (maximum release rate).

The equilibrium concentration SRP is calculated as the concentration at which flux rate = 0 using mean slope and intercept. Rate (0.45) = predicted flux rate at 0.45 mg *Z* in overlying water (mg m $^{2}$ h ) (negative values indicate uptake; CV = coefficient of variation in %).

| Site | Slope ($\bar{m}$) | y intercept ($\bar{c}$) | $SRP_o$ | Rate (0.45) |
|---|---|---|---|---|
| <10 m depth | | | | |
| Magalies | -0.0081 | 5.83 | 0.719 | +2.19 |
| CV | 56 | 45 | - | |
| (n = 5) | | | | |
| Crocodile | -0.0064 | 1.33 | 0.208 | -1.55 |
| CV | 63 | 44 | - | |
| (n = 4) | | | | |
| Main basin | -0.0140 | 7.08 | 0.505 | +0.78 |
| CV | 78 | 39 | - | |
| (n = 3) | | | | |
| >10 m depth | | | | |
| Magalies | -0.0038 | 4.12 | 1.085 | +2.41 |
| CV | 42 | 37 | - | |
| (n = 5) | | | | |
| Crocodile | -0.0048 | 4.57 | 0.952 | +2.41 |
| CV | 46 | 37 | - | |
| (n = 5) | | | | |
| Main basin | -0.00326 | 4.77 | 1.463 | +3.30 |
| CV | 35 | 15 | - | |
| (n = 5) | | | | |

The linear relationships (Fig. 5.41) provide a convenient means of predicting the direction and rate of phosphate flux between sediments and water at varying phosphate concentrations in the water on an areal basis. Diffusion models, commonly used for modelling sediment/water phosphate flux under non-mixed conditions (Bostrom *et at* 1982) are not applicable to turbulent conditions which predominate in shallow water bodies (Hakanson & Jansson 1983). Under such conditions turbulence is the dominant factor controlling the rate of phosphate flux and, since it varies with wind and water movement in lakes, it cannot be accurately simulated in experimental systems. During these experiments water residence time in the continuous-flow systems was about 15 min and there was insufficient turbulence to cause sediment resuspension. Under natural conditions in shallow waters turbulence ranges between virtually zero at times of calm to turbulent resuspension during storm events.

Conditions in the continuous-flow systems were assumed to represent an 'intermediate' condition.

The phosphate exchange characteristics measured at various shallow and deep sites in Hartbeespoort Dam during 1982-84 using the approach shown in Figure 5.41 are summarised in Table 5.10. The phosphate exchange characteristics for shallow water sediments generally were more variable than those from deep water zones. In shallow water areas equilibrium phosphate levels ranged from 0.72 mg *Z* $^{1}$ in the Magalies arm to 0.21 mg *Z* $^{1}$ in the Crocodile Arm. These values are probably of greatest significance in assessing whether the sediments are acting as phosphate sources or sinks at the ambient phosphate concentrations in the dam (about 0.45 mg *Z* $^{1}$). The predicted rates and directions of flux at this concentration are also shown in Table 5.10 (note that positive values indicate phosphate release). These data show that when concentrations in the overlying water are less than the equilibrium concentration (Magalies Arm and Main Basin) phosphate release would be induced and when the reverse gradient occurs (Crocodile Arm) phosphate uptake will occur. Assuming that the relative surface areas of these sediments are equal a mean rate of flux from shallow water sediments (< 10m) of +1.42 mg m $^{2}$h * can be calculated. In the deep zones (> 10m) the sediments exhibited a markedly higher release potential with equilibrium phosphate concentrations ranging from 0.95 mg *Z* $^{1}$ to 1.46 mg *Z* $^{1}$ and release rates at ambient phosphate concentrations in the dam (0.45 mg *Z* $^{l}$) ranged from 2.4 mg m $^{2}$h $^{1}$ to 3.3 mg m $^{2}$h *. In the deep zones no site dependent differences were evident. On the basis of data in Figure 5.42 it is clear that these release rates cannot be maintained indefinitely.

Variability in the phosphate exchange characteristics at different sites (particularly in shallow zones) was high with coefficients of variation frequently exceeding 50%. Furthermore, due to practical limitations in the procedure used, the number of measurements made was inadequate to obtain a reliable indication of the relative surface areas of sediment that are representative of the exchange characteristics shown in Table 5.10. For modelling purposes this posed a problem that was solved by assuming that the ambient phosphate concentrations in the dam must reflect the average phosphate equilibrium concentration for sediments in the dam. Thus the mean slopes and Y intercepts shown in Table 5.10 were used to describe sediment/water phosphate exchange in the dam assuming equilibrium concentrations equal to ambient phosphate concentrations in the dam.

Since the internal loading from sediments is a potentially important factor controlling the response of eutrophic lakes to rehabilitation measures, long-term phosphate release experiments were undertaken on aerobic sediment cores from three sites in the dam (Fig. 5.42). The rate of phosphate release decreased gradually with time approaching a slow, but persistent, base rate after about three weeks. The absence of a phosphorus input to the sediments via sedimentation makes it difficult to assess the applicability of these experiments to natural _conditions. Phosphorus sedimentation rates of up to 300 mg m $^{2}$d (mean 137 mg m $^{2}$d ), some of which was bioavailable, were measured in sediment traps in the dam (Section 5.3.3). Even under a reduced phosphate loading regime these inputs would tend to perpetuate the more rapid release rates measured at the initial stages of the experiment.

The declining release rates as the experiments progressed may indicate that the easily exchangeable phosphate in the surface sediments was gradually removed and that subsequent release rates were limited by slow phosphate diffusion from deeper sediments. An alternative, and perhaps more likely, explanation is that slow geochemical binding of exchangeable phosphate occurs in the deposited sediments resulting in slow but virtually permanent immobilisation of short-term exchangeable phosphate. These experiments indicate clearly that the instantaneous exchange rates measured in short-term experiments cannot be maintained indefinitely. Quantification of the limits of phosphate release, in response to reduced phosphate inputs, remains an urgent requirement. In the absence of quantitative information it is assumed for modelling purposes that sediment P is irreversibly bound at a rate of 10% p.a.

### (b) The influence of dehydration and rewetting on sediment/water phosphate exchange

The influence of dehydration on sediment/water phosphate exchange was examined by monitoring sediment phosphate exchange characteristics along a transect through the draw-down zone at two sites (Fig. 5.43) and in laboratory simulation experiments (Fig. 5.44).

The laboratory simulation experiment was based on a series of cores from approximately 10 cm above the water line in the draw-down zone in the Crocodile River Arm (these are referred to as initial exposed cores). In addition another series of cores from about 10 cm below the water line were sampled as controls (these are referred to as initial submerged cores). The initial exposed cores are estimated to have been exposed for approximately 48 h. Phosphate exchange characteristics of the initial exposed and initial submerged cores were measured immediately. Thereafter some of the initial exposed cores were stored in the laboratory under a 10 cm layer of distilled water and others were stored dry. At weekly intervals phosphate exchange characteristics of these wet and dry cores were monitored (Fig. 5.44).

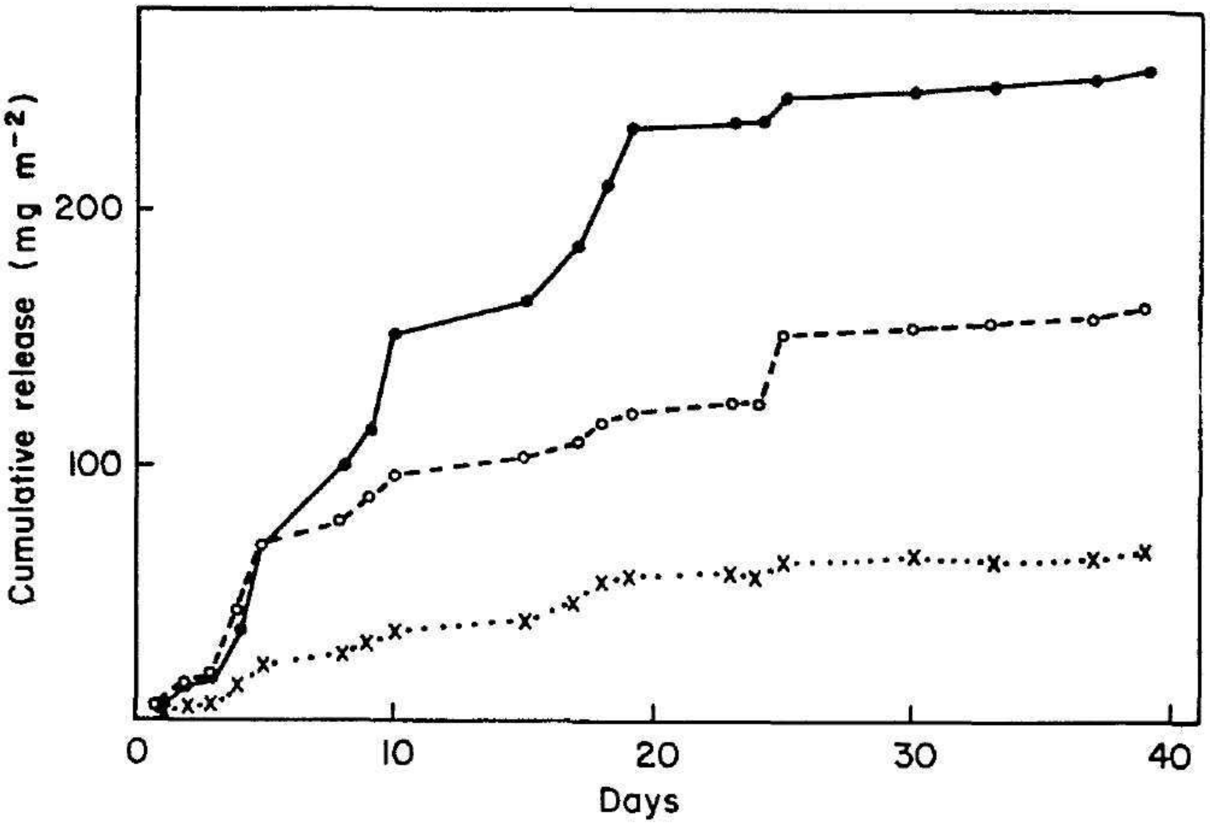


Figure 5.42. Cumulative phosphate release from intact sediment cores from 3 sites. Solid line = Magalies Arm shallow site; broken line = Crocodile Arm deep site; dotted line = Magalies Arm deep site.

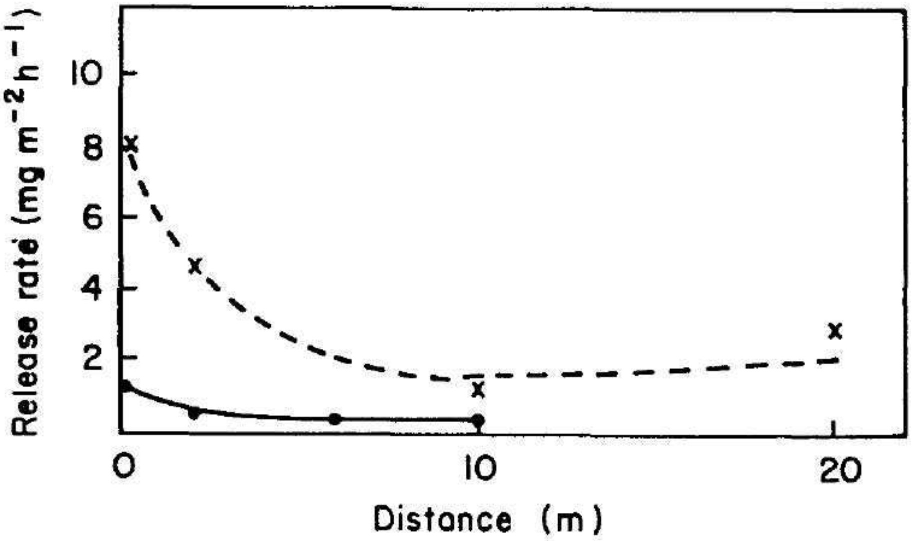


Figure 5.43. Phosphate release from intact cores collected below the waterline and along transects within the draw-down zone in the Magalies Arm (X) and the Crocodile Arm (●). Distance is from water's edge.

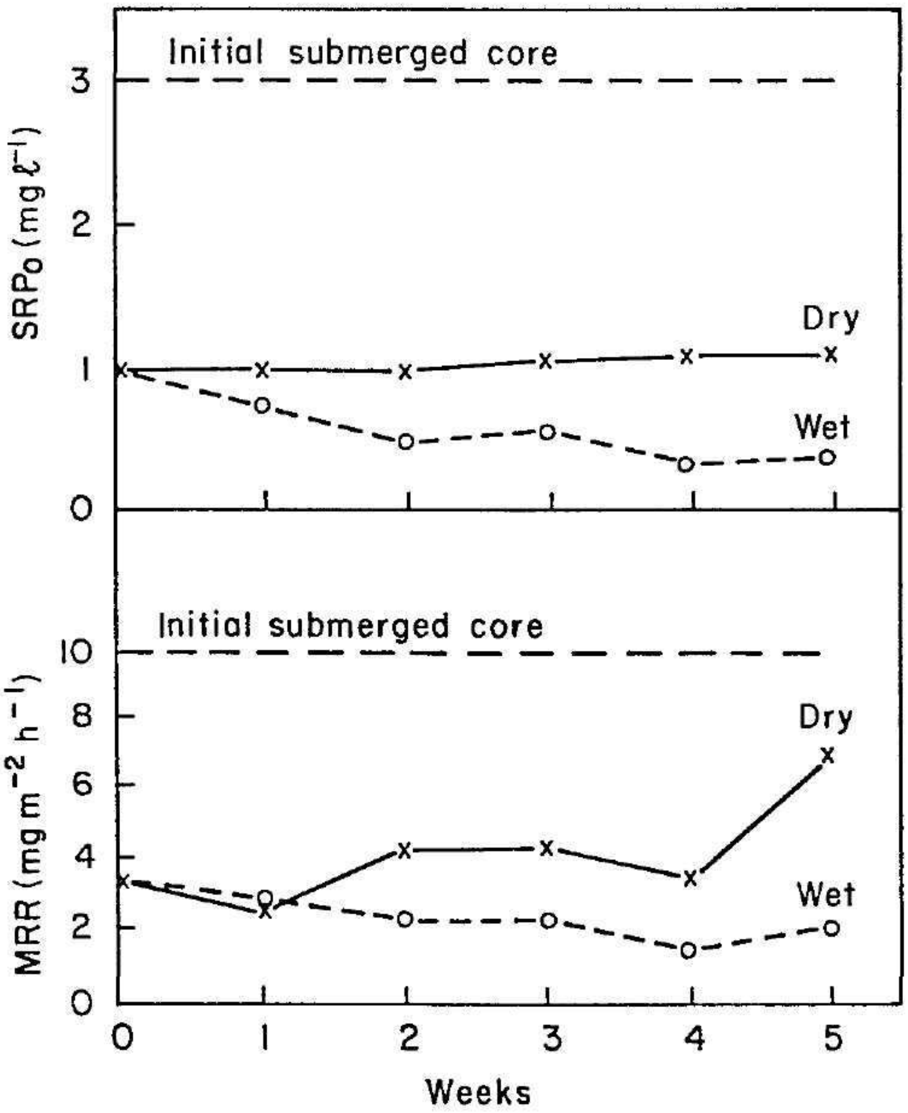


Figure 5.44. Changes in phosphate exchange characteristics of sediments collected above the waterline in the Crocodile Arm, compared with sediments samples below the waterline, during five weeks storage. The exposed cores were stored dry (X) and wet (o). MRR = maximum release rate; $SRP_o$ = equilibrium phosphate concentration.

A comparison of phosphate exchange in the initial exposed and initial submerged cores shows that even a short period of exposure results in a big change in exchange characteristics. Both EP and maximum release rate (MRR) in the initial exposed cores were less than half those in the initial submerged cores. Prolonged exposure of the initial exposed cores resulted in little further change in phosphate exchange characteristics (Fig. 5.44). The initial exposed cores that were stored under water showed progressive decreases in both EP and MRR over the 5 week experiment. EP dropped by about 65% and MRR by about 40%.

The decreasing phosphate release potential of dehydrated sediments was also evident in experiments along transects in the draw-down zone (Fig. 5.43). Although the release potential from sediments in the Magalies River Arm was far greater than in the Crocodile River Arm both sites showed rapidly declining release potential with increasing distance from the water line, and, assuming progressively increasing dehydration with distance from the water line, supporting the conclusions of the laboratory simulations. These experiments suggest that the periodic exposure of marginal sediments to the atmosphere during draw-down, which can effect 50% of the sediment surface area in the dam, results in a marked increase in phosphate uptake potential on re-flooding. Prolonged storage of re-flooded sediments under laboratory conditions resulted in a continuing declining trend in phosphate release potential for five weeks suggesting continuous geochemical binding of sediment phosphorus. Under these conditions sedimentation is not taken into account. Under natural conditions the rapid deposition of sediments (Section 5.3) would result in rapid changes in the characteristics of surface sediments and for this reason the laboratory experiments must be interpreted with caution. Sedimentation probably results in reflooded sediments rapidly attaining phosphate characteristics more typical of permanently flooded sediments.

### (c) The influence of temperature, pH and oxygen on sediment/water phosphate exchange

Examples of the influence of varying temperature, within the surface sediments, on SRP release from intact sediment cores in continuous-flow experiments are shown in Figure 5.45. In these experiments temperature was recorded continuously via a thermocouple positioned 1 cm below the sediment surface. At temperatures between 4 and 25 °C no change in SRP concentration in outflows from continuous-flow sediment/water systems was evident at two sites in the dam. Within the temperature range in the dam (9-25 °C) it therefore seems that temperature does not have a marked influence on sediment/water phosphate exchange, which contrasts with findings in some other waters (Bostrom *et at,* 1982). Similar experiments in which pH levels in the overlying water were adjusted showed that at two sites (deep main basin and shallow Crocodile River arm) pH values between 4 and 10 had no influence on phosphate release (Fig. 5.46) whilst at two shallow sites in the Magalies River arm release rates were more than doubled at high pH. This indicates that pH increases in some marginal areas, due to algal or macrophyte photosynthesis, could induce significant

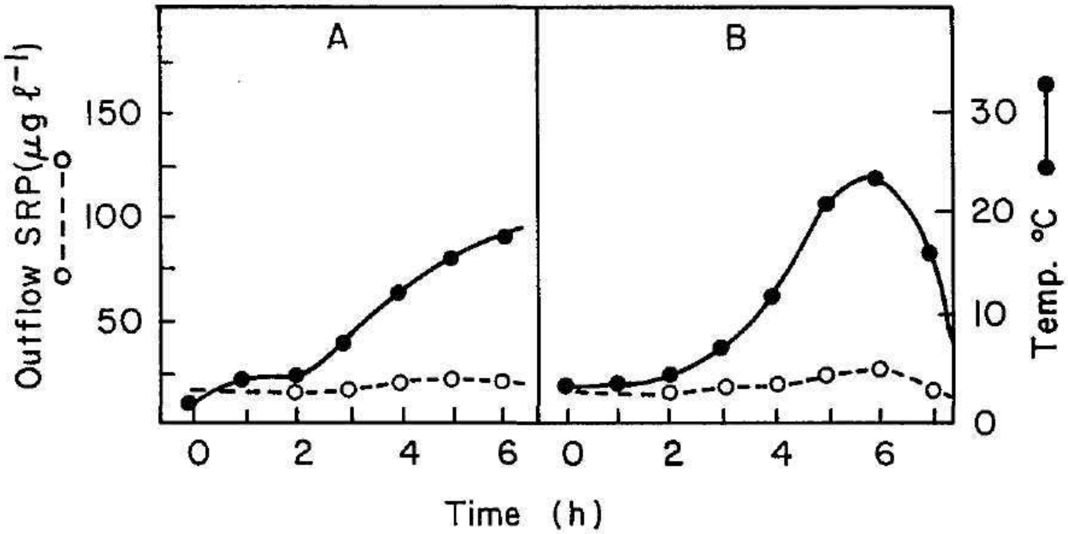


Figure 5.45. The influence of temperature fluctuations (1 cm beneath the sediment surface) on soluble reactive phosphorus concentrations in the outflows from continuous-flow sediment water systems. A) Deep station in the Main Basin. B) Shallow site in the Crocodile Arm.

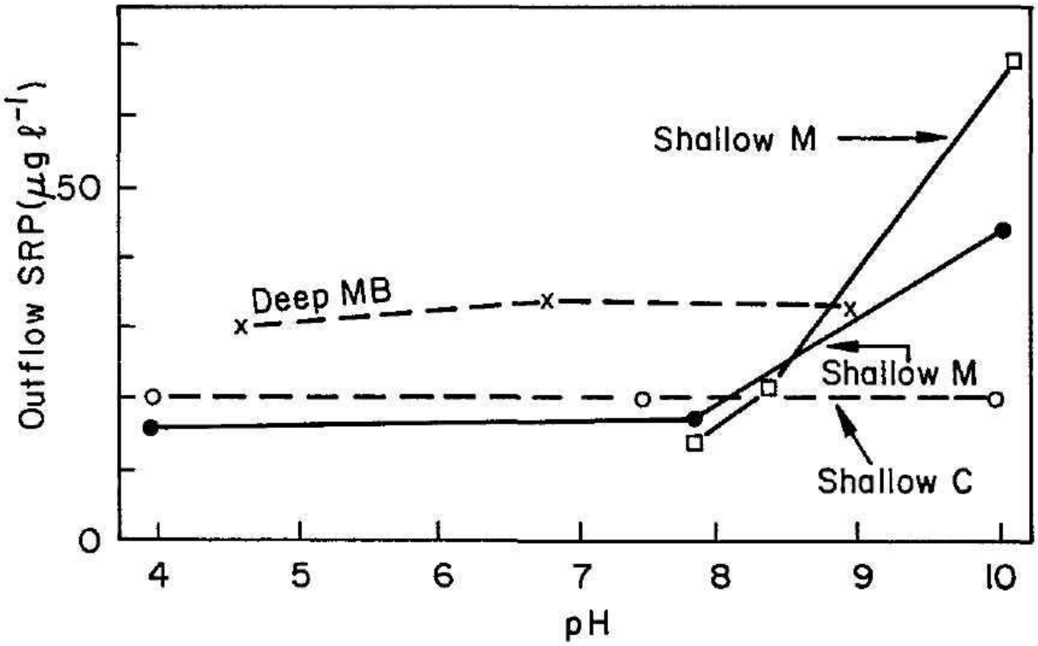


Figure 5.46. The influence of pH fluctuations on soluble reactive phosphorus concentrations in the outflows from continuous-flow sediment/water systems from shallow and deep sites in the Main Basin (MB), Crocodile Arm (C) and Magalies Arm (M).

significant releases of phosphate from sediments, as has been shown for other lakes (Jacoby *et at.* 1982). The data available are insufficient to quantify the relative surface areas of sediment that may exhibit this pH dependency. For modelling purposes the influences of temperature and pH on sediment/water phosphate exchange were considered to be of minor importance since values measured in the lake were always within the experimental ranges used.

As expected the sediments from the deep areas showed the typical response to anaerobic conditions with release rates increasing five fold when inflows to sediment/water systems were purged of oxygen with nitrogen gas and sodium sulphite solution. The consequences of this in the hypolimnion are clear in the phosphate build up that characterises stratified periods in the dam (Section 5.2.3). During stratified periods the rate of phosphorus movement train hypolimnetic sediments to the epilimnion is not dependent on sediment/water phosphate gradients but on mechanisms controlling vertical transport across the thermocline. Increased phosphate release from anaerobic sediments is included in the ecosystem model.

### (d) Seasonal and abiotic influences on sediment/water phosphate exchange

Three deep stations were monitored regularly to assess temporal fluctuations in sediment/water phosphate exchange. These results are summarised in Figure 5.47. Mean values for maximum phosphate release rate and equilibrium phosphate concentration from the three stations showed no marked seasonal fluctuation, suggesting that biotic processes, that would be expected to show some seasonal response, were not dominating the exchange. This is borne out by the data in Table 5.11 which show that addition of azide (as a biological inactivator) generally had little influence on the exchange characteristics. On the basis

Table 5.11. Linear regression constants and related parameters describing the phosphate exchange characteristics of sediments from Hartbeespoort Dam prior to and following addition of azide. Definitions as for Table 5.7.

| Date | Site | Azide | y Intercept | Slope | $SRP_o$ |
|---|---|---|---|---|---|
| 82-10 | Mag | no | 3.496 | -0.00378 | 0.925 |
| | | yes | 3.358 | -0.00477 | 0.704 |
| 83-04 | Mag | no | 1.450 | -0.00716 | 0.202 |
| | | yes | 1.446 | -0.00537 | 0.269 |
| 83-04 | Mag | no | 1.508 | -0.00275 | 0.548 |
| | | yes | 1.279 | -0.00163 | 0.784 |
| 83-05 | Croc | no | 5.261 | -0.00754 | 0.697 |
| | | yes | 8.697 | -0.01028 | 0.846 |
| 83-07 | Croc | no | 4.986 | -0.00382 | 1.305 |
| | | yes | 3.436 | -0.00293 | 1.173 |
| | Mag | no | 1.301 | -0.00169 | 0.770 |
| | | yes | 0.913 | -0.00111 | 0.823 |

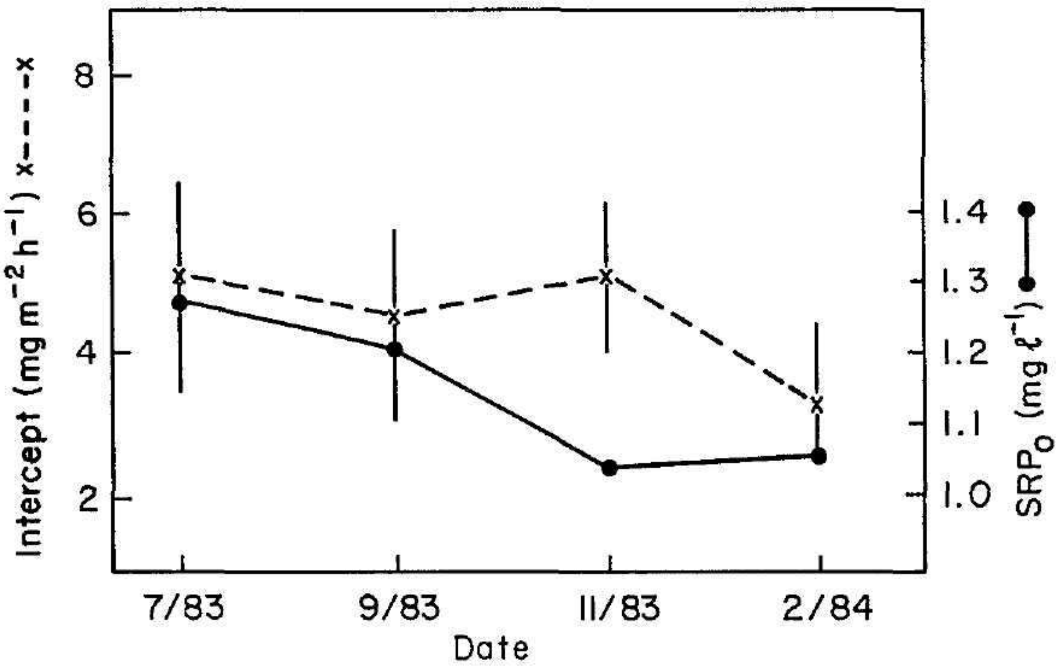


Figure 5.47. Temporal changes in sediment/water phosphate exchange characteristics in the Main Basin. Intercept = maximum release rate; SRP$_o$ = equilibrium phosphate concentration. Vertical bars indicate range.

of this no attempt was made to differentiate between biotic and abiotic processes in modelling sediment/water phosphate exchange.

### 5.3.3 Sedimentation

The rates of sedimentation of various components at different deep water sites in the dam are summarised in Table 5.12. Despite frequent unavoidable discontinuities in the data some clear patterns are evident. The mean deposition rates of dry material, organic material (estimated as loss on ignition) and phosphorus was highest in the Crocodile River Arm (CRA). In the CRA the rates of sedimentation tended to increase with increasing river flow while those at the other sites further from the river showed no discernable relationship with flow rate (Fig. 5.48). In Hartbeespoort Dam, therefore, the highest sedimentation rates were recorded in the fairly shallow area adjacent to the main river inflow.

A comparison of phosphorus and organic content of the sedimenting material shows that the CRA is generally characterised by sedimenting material lower in both of these components, compared with the main basin, particularly at times of high flow in the river. This is attributed to a larger proportion of inorganic silt in the area closest to the river inflow.

Table 5.12. Characteristics of sedimenting material collected in sediment traps at sites in Hartbeespoort Dam. LOI = loss on ignition, DRY = dry mass sedimentation ORG = organic matter sedimentation based on LOI, PHOS = phosphorus sedimentation PCON = phosphorus content of sedimenting material (per unit dry mass). MB = Main Basin CRA = Crocodile River Arm, MAG = Magalies River Arm.

| Site | | n | LOI % | DRY g m$^{-2}$d$^{-1}$ | ORG g m$^{-2}$d$^{-1}$ | PHOS g m$^{-2}$d$^{-1}$ | PCON mg g$^{-1}$ |
|---|---|---|---|---|---|---|---|
| All data | mean | 58 | 24.3 | 73.4 | 13.6 | 137.4 | 3.1 |
| | min | | 10.9 | 5.7 | 1.6 | 12.7 | 0.2 |
| | max | | 41.4 | 494.5 | 75.6 | 414.9 | 7.3 |
| MB Bottom | mean | 15 | 27.2 | 42.2 | 10.3 | 131.4 | 3.6 |
| | min | | 15.5 | 5.7 | 1.6 | 12.7 | 0.2 |
| | max | | 41.4 | 106.8 | 28.8 | 199.4 | 7.3 |
| MB middle | mean | 14 | 30.1 | 20.8 | 5.9 | 74.7 | 4.1 |
| | min | | 19.5 | 8.4 | 2.6 | 36.6 | 2.0 |
| | max | | 40.6 | 38.9 | 10.8 | 118.7 | 7.1 |
| CRA Bottom | mean | 21 | 17.5 | 131.1 | 18.8 | 183.5 | 2.1 |
| | min | | 10.9 | 13.7 | 3.0 | 41.5 | 0.8 |
| | max | | 24.7 | 494.5 | 54.3 | 414.9 | 3.0 |
| MAG bottom | mean | 8 | 23.5 | 85.6 | 18.8 | 142.1 | 2.8 |
| | min | | 20.8 | 12.6 | 3.0 | 46.1 | 0.9 |
| | max | | 27.2 | 363.3 | 75.6 | 318.6 | 4.4 |

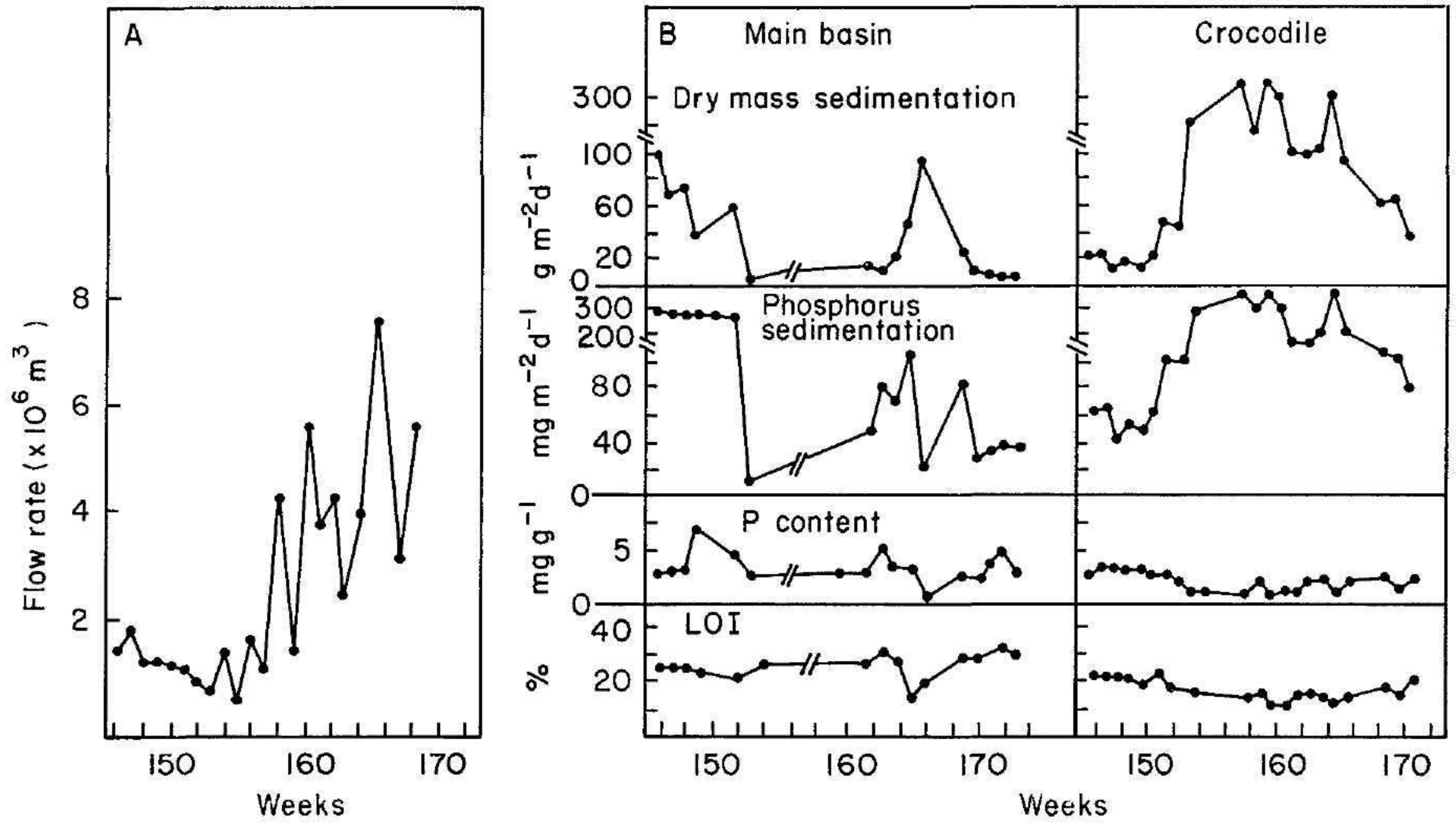


Figure 5.48. A) Crocodile River flow rates. B) Fluctuations in rates of sedimentation and characteristics of the sedimenting material in the Main Basin and Crocodile Arm. Week numbers from start of the study. LOI = mass loss on ignition.

These results, based on sedimenting material trapped 1 m above the sediment surface, demonstrate that there is a large gross flux of phosphorus onto the sediments via sedimentation in the dam. If the mean deposition rate measured during this study is applied to all areas of the dam exceeding 5 m water depth at FSL (about 60% of the surface area) an annual flux of 660 tonnes of phosphorus to the sediments can be calculated. This estimate exceeds the total annual phosphorus load to the impoundment by almost three and indicates that internal processes (resuspension and recycling) are major contributors to gross phosphorus sedimentation in Hartbeespoort Dam or that the method overestimates sedimentation rates. Based on mass balance calculations net phosphorus retention in the dam is closer to 60% of the total load (Section 5.2.5).

### 5.3.4 Vertical stratification in sediments

Single cores from deep zones in the Main Basin and Crocodile River Arm showed complex vertical profiles (Fig. 5.49). (The depth to which the gravity corer used penetrated the sediments was largely dependent on the sediment texture. However, at neither site do the cores represent the entire sediment depth. Gravity corers are not suitable for the sampling of deep sediment cores.) Water content never dropped below 55% and in the Crocodile River Arm the profile showed little change in water content with depth. However, at the Main Basin the profile was more variable, and water content tended to increase below 40 cm. The organic content of the stratified sediments (reflected by mass loss on ignition) was distinctly higher in the top 5 cm in the Main Basin (25-55%), below this levels dropped sharply to between 5 and 20%. In the Crocodile River Arm high organic content was evident in the top 1 cm only (25%). Below this, levels fluctuated between 3 and 12%, generally remaining lower than corresponding levels in the Main Basin.

Total and biologically available (NTA extraction) phosphorus showed similar patterns of stratification. In the Main Basin both peaked at 5 cm depth and showed an inconsistent decreasing trend to a depth of about 30 cm, below which concentrations again increased. In the Crocodile River Arm both phosphorus fractions were lower than corresponding levels in the Main Basin, particularly in the top 5 cm. Generally, about 60 - 70% of the total phosphorus was extractable with NTA indicating that it is mobile and can be taken up by organisms under suitable conditions.

Detailed analysis of these vertical profiles cannot be attempted here. They do, however, show that the sedimenting material has varied considerably over the years. No clear trends indicative of accelerating eutrophication are evident in these cores. The differences in sediment characteristics between the Main Basin and Crocodile River Arm probably reflect the influence of the silt carried in by the Crocodile River and are in agreement with the data discussed in Section 5.3.3.

On the basis of the bioavailable phosphorus profiles it is clear that a large pool of potentially mobile phosphorus has accumulated in the sediments and, as shown in Section 5.3.2, this can be transported into the overlying water under suitable conditions.

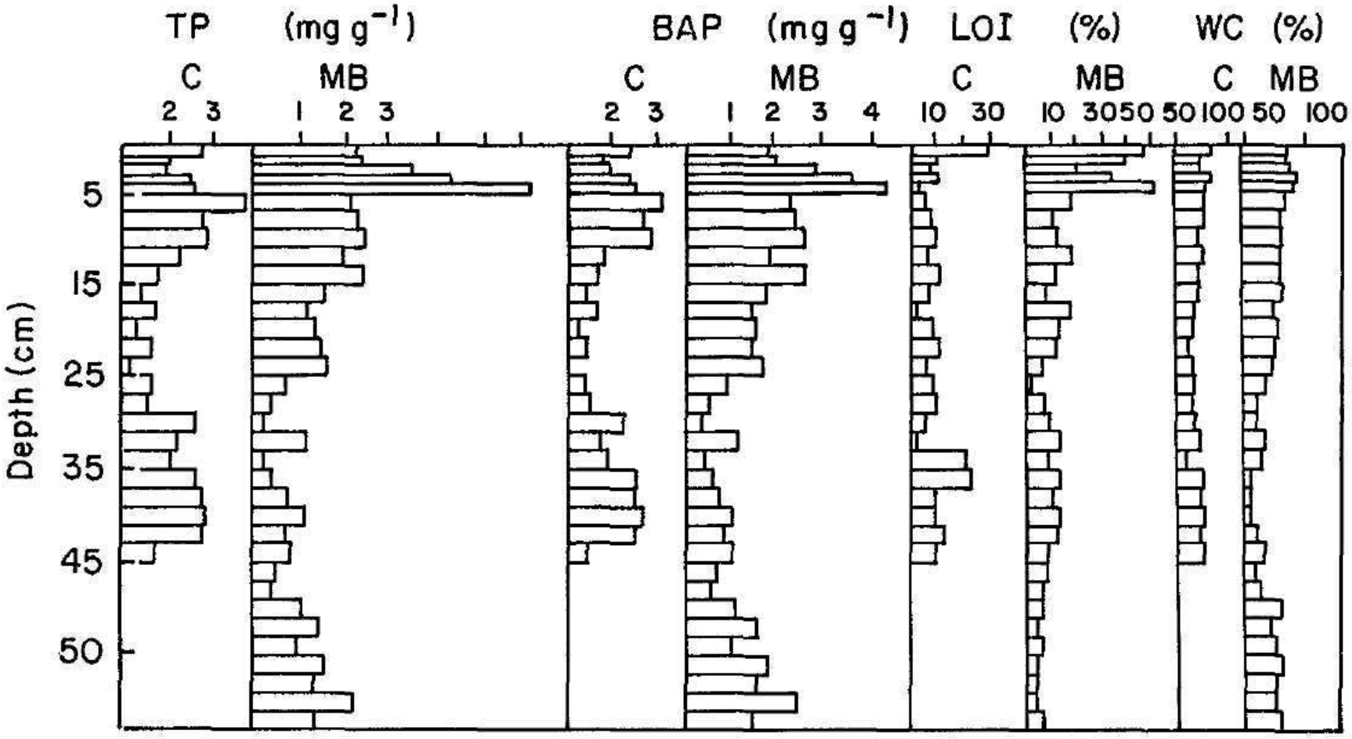


Figure 5.49. Vertical profiles of total phosphorus (TP) biologically available phosphorus (BAP), mass loss an ignition (LOI) and water content (WC) in sediments from deep sites in the Crocodile Arm (C) and Main Basin (MB).

## 5.4 BIOLOGICAL LIMNOLOGY

### 5.4.1 Introduction to lake biology

An understanding of the biology of lakes is obtained by determining the role of living organisms in ecosystem functioning. Life requires energy which is captured from light by photosynthesising organisms, mainly plants. Major groups of plants in lakes are the macrophytes (Leaved plants), which may be emergent or submerged, and free-floating or rooted to the bottom, and the algae, which may be planktonic (the phytoplankton), attached to the bottom (the benthic algae) or to other plants (the epiphyton). The role of plants is that of primary producers, utilizing sunlight energy and dissolved chemical substances to produce new biomass. They form the major link between lake chemistry and biology.

Aquatic animal life is divided into animals which live on the bottom (the zoobenthos), small animals which live in the open water (the zooplankton) and large animals such as fish. Functionally animals are consumers and may be herbivores (plant-eating), detritivores (detritus-eating), carnivores (meat-eating) or omnivores (mixed diet).

Detritus is non-living particulate organic matter derived from dead plants or animals and from animal faeces. Detritus and dissolved organic matter are the energy base for the third major functional group of living organisms, the decomposers, which are bacteria and other micro-organisms. They mineralize or break down detritus and so provide the second important link between the biotic and abiotic components of the ecosystem. Many detritivores derive their nutrition more from the decomposers in the detritus than directly from the detritus itself.

An even simpler functional classification of life forms is the division into autotrophs (meaning self-feeding) and heterotrophs (feeding on organic substances). All organisms deriving energy only from sunlight are autotrophs, whereas all other organisms which utilize previously bound energy are heterotrophs.

Interrelationships between functional groups are conceptually simple. For instance, phytoplankton is consumed by zooplankton, which is eaten by fish. When the fish dies it forms detritus and dissolved organic carbon, which are decomposed to carbon dioxide, nitrate, orthophosphate and small amounts of other elements. Together with sunlight these substances are the necessities for further phytoplankton growth. In reality, however, the trophic (feeding) relationships between the functional groups are very much less direct and energy flows through many pathways as it is utilized. Due to the numerous pathways which organic matter and energy may follow from the producers to the decomposers, ecologists recognise food webs, rather than simple pathways leading directly from producers to herbivores to carnivores.

The importance of the various functional groups changes with the chemical nature of lakes. In oligotrophic (nutrient poor) lakes the open water food web is dominated by the phytoplankton-zooplankton - fish chain. In eutrophic (nutrient rich) systems the phytoplankton becomes dominated by algal species of low palatability to zooplankton, which markedly decreases the importance of the direct link from phytoplankton to zooplankton. It would appear that the dominant pathways between phytoplankton and fish become phytoplankton - detritus - fish or even phytoplankton - fish.

In studies such as that reported here, where an important objective is to quantify the functioning of an ecosystem, most of the biota are treated as functional groups rather than as separate species. This statement is, however, untrue of the approach adopted to the fish in Hartbeespoort Dam, where the dominant species are treated individually. Research is undertaken to measure the size of the functional groups, to identify the major pathways of exchange between functional groups, the rates at which such exchange takes place and the major factors governing these rates. It is assumed that exchanges between minor functional groups do not materially effect exchanges between major functional groups.

This section of the report is a presentation of the research findings on these aspects of the biological functioning of Hartbeespoort Dam.

### 5.4.2 Bacteria - distribution, seasonal cycles and controlling factors

Figure 5.50 shows the seasonal cycles of the upper (8 m) water temperature and mean number of bacteria for 1981 to 1983. The population showed two annual peaks, a winter peak following overturn and a summer peak following stratification. A similar pattern was observed in deeper waters. To relate this seasonal cycle to various physical-chemical changes the data were divided into three categories: epilimnion (0 to 8 m); aerobic hypolimnion (water >15 m, with $0_2$ > 2 mg *2* $^{1}$*)* and anaerobic hypolimnion (water >15 m).

The range in bacterial numbers was epilimnion, 2.3 - 44.3 x $10^6$ bacteria m£ *; aerobic hypolimnion, 1.4 - 22.7 x $10^6$ m£ $^l$*;* and anaerobic hypolimnion, 3.2 - 31.2 x $10^6$ *m2* $^l$ (Robarts & Sephton 1984). Table 5.13 shows the results of a Spearman Rank Correlation analysis of the data, Epilimnetic population fluctuations were most closely related to changes in DOC and temperature. In the aerobic hypolimnion population size was inversely related to $0_2$. In the anaerobic hypolimnion it was inversely related to NH^-N. These correlations were significant but not strong. Weak correlations resulted because different physiological types were not enumerated (Robarts & Sephton 1984). Jones (1971) also found that eplimnetic bacterial populations in eutrophic lakes were influenced by temperature.

Aerobic zone bacteria were predominantly cocci (66.0 - 87.4%, x = 78.1%) with diameters usually between 0.1 and 0.2 um . Anaerobic bacteria were also mainly cocci (51.6 - 72.4%, x = 63.*570* but greater morphological variation was observed.

### 5.4.3 Bacterial heterotrophic activity

The data analysis of bacterial counts with various physical, chemical and biological variables indicated that little information on the bacteria of Hartbeespoort Dam was to be gained with this approach. While bacteria represent only a small portion of a lake's biomass an understanding of their function is essential since they

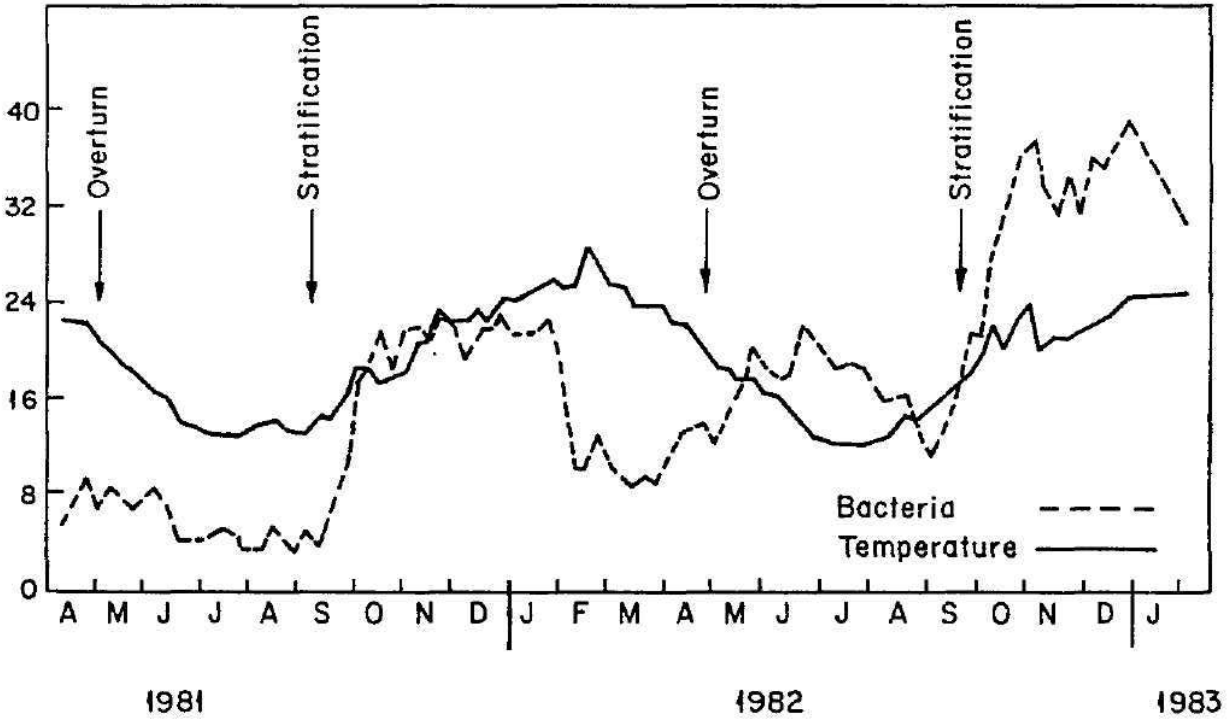


Figure 5.50. Seasonal variation of mean bacterial numbers (X $10^6$ $m\ell^{-1}$) and temperature (°C) for the upper 8 m of Hartbeespoort Dam (from Robarts & Sephton 1984).

Table 5.13. Spearman Rank correlation analysis of bacterial counts and environmental parameters for Hartbeespoort Dam. r = correlation coefficient; n = number of observations; p = probability.

EPILIMNION

| | Temp | Chl *a* | D.O.C. | $NO_3$-N | $NH_4$-N | Part N | % $O_2$ sat | pH | Tot. P | $PO_4$-P |
|---|---|---|---|---|---|---|---|---|---|---|
| r | 0.43 | 0.09 | 0.48 | -0.38 | -0.07 | 0.04 | 0.04 | 0.23 | 0.14 | 0.14 |
| n | 714 | 706 | 566 | 721 | 567 | 721 | 688 | 703 | 566 | 567 |
| p | 0.001 | 0.010 | 0.001 | 0.001 | 0.05 | 0.001 | 0.146 | 0.001 | 0.001 | 0.001 |

HYPOLIMNION-AEROBIC

| | Temp | Chl *a* | D.O.C. | $NO_3$-N | $NH_4$-N | Part N | % $O_2$ sat | pH | Tot. P | $PO_4$-P |
|---|---|---|---|---|---|---|---|---|---|---|
| r | 0.21 | -0.06 | 0.26 | -0.21 | -0.02 | -0.13 | -0.67 | -0.22 | 0.42 | 0.53 |
| n | 140 | 154 | 153 | 155 | 155 | 155 | 149 | 152 | 155 | 155 |
| p | 0.007 | 0.239 | 0.001 | 0.004 | 0.419 | 0.058 | 0.001 | 0.004 | 0.001 | 0.001 |

HYPOLIMNION-ANAEROBIC

| | Temp | Chl *a* | D.O.C. | $NO_3$-N | $NH_4$-N | Part N | % $O_2$ sat | pH | Tot. P | $PO_4$-P |
|---|---|---|---|---|---|---|---|---|---|---|
| r | -0.22 | -0.17 | -0.08 | 0.31 | -0.47 | -0.11 | | 0.23 | -0.16 | -0.09 |
| n | 89 | 93 | 97 | 97 | 96 | 97 | | 91 | 96 | 96 |
| p | 0.021 | 0.050 | 0.209 | 0.002 | 0.001 | 0.149 | | 0.013 | 0.061 | 0.188 |

are the first link joining the biotic and abiotic components. Consequently, studies were started in late 1982 on bacterial heterotrophic utilization of dissolved organic carbon (e.g., glucose). The study focussed on heterotrophic bacteria as photosynthetic bacteria were not present since the anaerobic zone essential for their growth, was always out of the region to which light penetrated (Figs 5.29 and 5.58).

The seasonal cycles of bacterial numbers for the period in which heterotrophic uptake experiments were done are shown in Figure 5.51. These data differ from those of past years in two ways (cf. Fig. 5.50). First, previous years have had a major peak in summer followed by a smaller bacterial peak in winter. Second, bacterial numbers in summer 1983/1984 never attained the high numbers of past years.

Figure 5.52 presents the annual cycle of V , the maximum velocity of substrate uptake. V ranged between Cr \unmeasurable) in March 1983 to a high of 3.9 pg^*"$^{1}$ h"$^{1}$ in January 1983. V was usually greatest at the surface while little difference was noted between the rates measured at 5 and 10 m . Basically, the rates showed a seasonality with the highest rates in summer and the lowest in winter. V$_{max}$ was low in early summer (November to January) 1983/84 but increases sharply in mid-January.

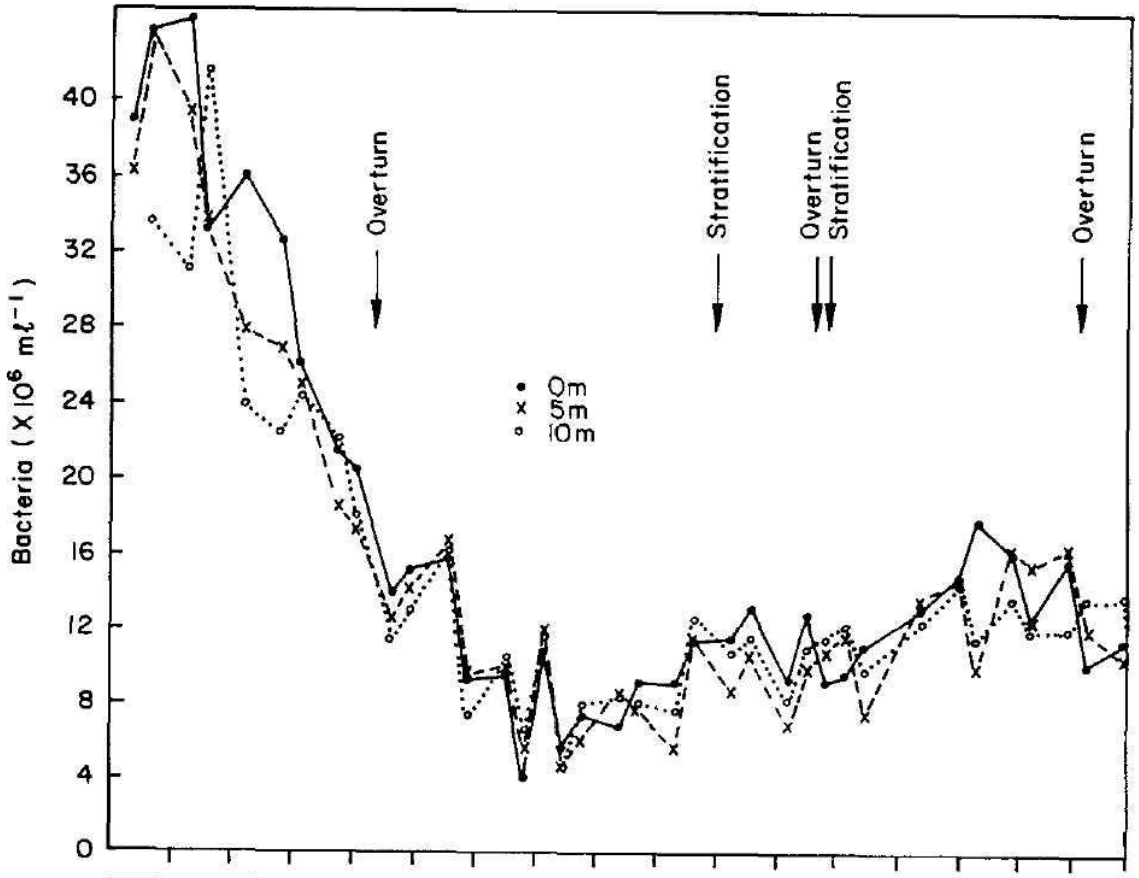
Table 5.14 presents the results of a regression analysis between V and some physical, chemical and biological parameters. V was significantly related to bacterial numbers, percentage saturation of oxygen and water temperature although the relationships were not particularly strong. A strong relationship was found between V and primary production.

(K + S ), the theoretical maximum natural substrate concentration, was lowest in summer and highest in winter (Fig. 5.53) and reached 242 ug C *Si*~$^{l}$ in June 1983. (K + S ) was often highest at the surface which was the region of maximum algal production (Fig. 5.62). No significant relationship between primary production and (K + S ) was found possibly because this parameter is also influenced by allochthonous organic matter. The high (K 4- S ) values in winter are probably the result of the lower bacterial numbers and the inverse relationship between substrate turnover time (T ) and temperature (see below) which allows organic compounds to accumulate near the surface, the region of highest algal production.

Figure 5.54 shows that T - the time required for the bacterial population to remove all the available substrate, was largest (342 h) at the surface in September 1983. A general seasonal trend of long turnover times in winter and short turnover times in summer can be seen. In summer 1982/83 turnover time was shortest at the surface but from March onwards, the reverse was usually true. No explanation is presently available for this.

Table 5.14 shows that turnover time was inversely related to bacterial numbers, water temperature and primary production.

The difference between the numbers of bacteria and heterotrophic activity for 1982/84 may be indirectly related to the drought. The



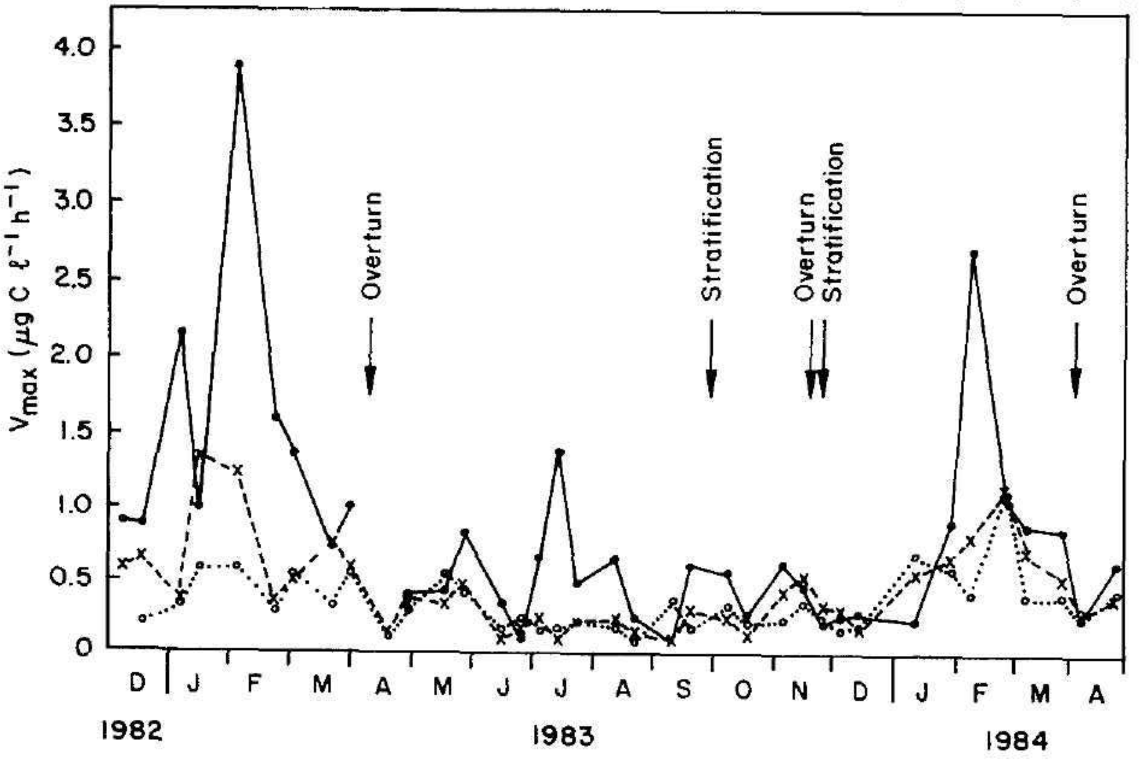
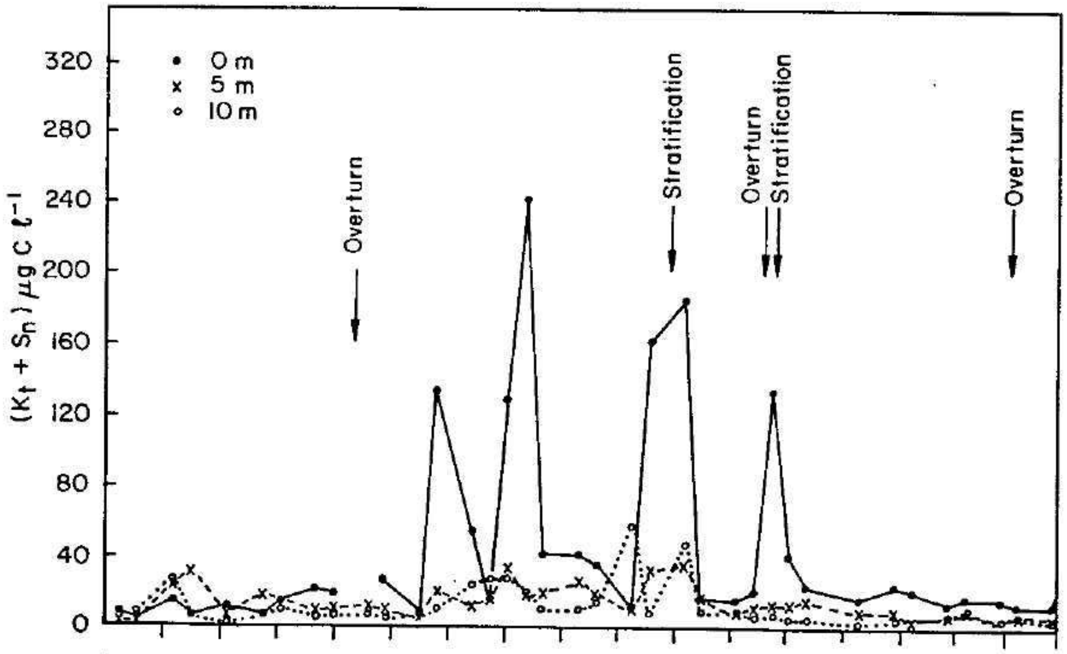


Figure 5.51. (upper) Seasonal fluctuations of bacterial numbers at 3 depths in Hartbeespoort Dam for the water samples for which heterotrophic utilization of glucose was measured.

Figure 5.52. (lower) Seasonal fluctuations of $V_{max}$, the maximum rate of glucose uptake, at 3 depths in the epilimnion of Hartbeespoort Dam.



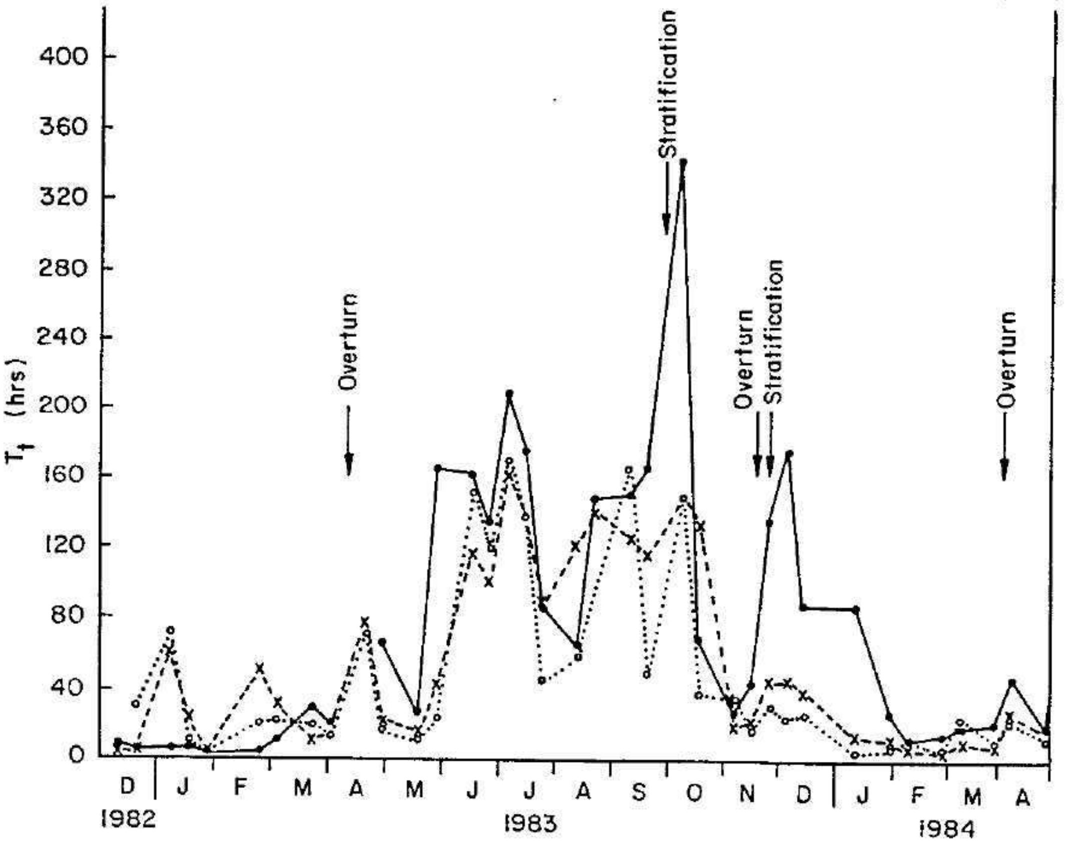


Figure 5.53. (upper) Seasonal fluctuations of $(K_t + S_n)$, the theoretical glucose concentration, at three dephts in Hartbeespoort Dam.

Figure 5.54. (lower) Seasonal cycles of $T_t$, the time required for the bacterial population to remove all the glucose present in a water sample, at three depths in Hartbeespoort Dam.

Table 5.14. Best-fit regression analysis of heterotrophic uptake parameters and various physical, chemical and biological parameters for Hartbeespoort Dam during 1982 - 1984. (Robarts, in prep.) r = correlation coefficient; n = number of observations; p = probability.

| | | Bacteria ($\times 10^6$ mℓ) | % Sat. of $O_2$ | Temperature (°C) | Primary Production (mg C $m^{-3}$ $h^{-1}$) |
|---|---|---|---|---|---|
| $V_{max}$ | n | 110 | 109 | 110 | 29* |
| | r | 0.582 | 0.566 | 0.600 | 0.817 |
| | p | <0.001 | <0.001 | <0.001 | <0.001 |
| $T_t$ | n | 110 | - | 110 | 29* |
| | r | -0.621 | - | -0.711 | -0.711 |
| | p | <0.001 | - | < 0.001 | <0.001 |

* for surface samples only

lower water levels have reduced water column stability (Section 5.1.8). The stability of the water column increased in January 1984 although it never became as stable as in previous years. This coincided with an increase in bacterial numbers, $V_{TH3.X}$ and a decrease in both glucose and phosphorus turnover times (Figs 5.52 & 5.56). It is not yet clear why these changes appear to be related to water column stability. It might be expected that this could be related to the association found between stability and primary production (Robarts 1984). Table 5.14 supports this.

The present study of Hartbeespoort Dam has shown that generally a large and active bacterial population was present. The heterotrophic activity expressed as V of Hartbeespoort Dam bacteria approached the highest published rates, (K + S ) fell within the range reported by others while surface T values tended towards lower recorded values (cf. Hoppe 1978). Unfortunately, similar data are not available for other African lakes. In the analysis to date quantitative relationships between the aquatic bacteria and the phytoplankton have been established. This might have been expected but the demonstration of these relationships is not always easy to do and few published quantitative results of this nature are available.

### 5.4.4 Nitrogen cycling

Nitrogen occurs in the biosphere in a variety of forms ranging in oxidation state from +5 to -3. Inorganic nitrogen is present primarily as highly oxidized nitrate and nitrite, as reduced ammonia and as molecular nitrogen. A variety of intermediate gaseous oxides of nitrogen, important in atmospheric chemistry, seldom occur in significant quantities in natural waters (Brezonik 1972). Naturally occurring organic nitrogen consists primarily of amino and amide (proteinaceous) nitrogen, along with some heterocyclic compounds such as purines and pyrimidines. Nitrogen compounds are present as cellular constituents, non-living particulate matter, soluble organic compounds and inorganic ions in solution. All these forms

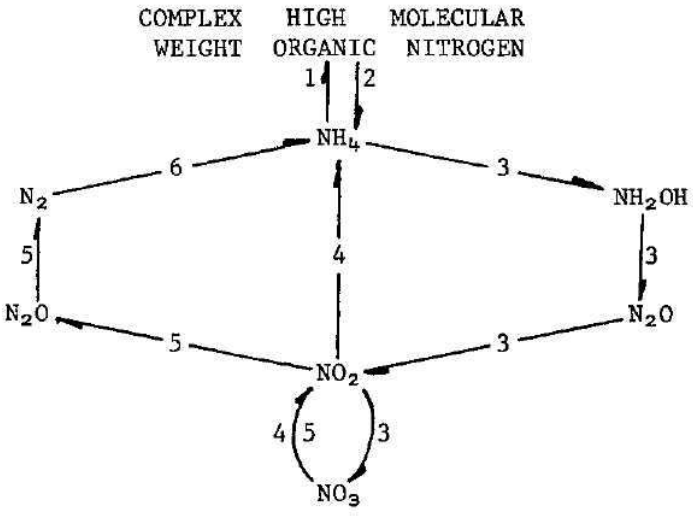


Figure 5.55. Simplified inorganic nitrogen cycle
(1 = ammonia assimilation; 2 = ammonification;
3 - nitrification; 4 = nitrate assimilation;
5 = denitrification; 6 = nitrogen fixation).

are interrelated by a series of reactions known collectively as the 'nitrogen cycle[1], which demonstrates the flow of nitrogen from inorganic forms in soil, air and water into living systems and then back again into inorganic forms. Figure 5.55 presents *a* simplified reaction sequence of the interconversions between organic nitrogen and the main inorganic forms in the aquatic environment and it is evident that these reactions are biologically mediated (Painter 1970).

Nitrogen has long been considered one of the principal nutrients limiting primary production rates in aquatic ecosystems, the availability of nitrogen profoundly influencing and, in some cases, controlling algal biomass (Hutchinson 1957). However, information such as rates of inorganic nitrogen turnover, relative nutritional importance of the various nitrogen forms and effects of concentration on rates of assimilation are scarce (Liao & Lean 1978).

The available evidence indicates that nitrogen transformations mediated by microorganisms can exert a marked effect on the biology of a lake (Keeney, Herbert & Holding 1971). Of these transformations, the oxygen demand created by nitrification (Burns & Ross 1972) and removal of inorganic nitrogen via denitrification (particularly in the absence of dissolved oxygen) are often of considerable importance (Hall, Collins, Jones & Horsley 1978). In the present study, major emphasis has been placed on the processes of nitrogen fixation, nitrification and denitrification.

### (a) Nitrogen fixation

Nitrogen fixation is the process whereby atmospheric dinitrogen gas ($N_2$) is enzymatically converted to NH^ in the absence of oxygen and is followed by subsequent amination reactions that regulate the incorporation of NH^ into proteins. Normally, nitrogen fixation is carried out by specialized groups of aerobic bacteria and heterocystous blue-green algae that are equipped with specific physiological mechanisms preventing deactivation of the nitrogenase enzyme complex by oxygen. However, certain less-common groups of anaerobic methanogenic bacteria are also capable of fixing atmospheric nitrogen. Normally, nitrogen fixation only occurs in lakes that are nitrogen limited, having low concentrations of inorganic nitrogen (Home & Viner 1971) and can provide a significant proportion of the lake's nitrogen budget (Ashton 1981).

In Hartbeespoort Dam, nitrogen fixation could not be detected in any of the sediment or water samples examined. This is somewhat unexpected in view of the low N:P ratios found in the lake and earlier observations, based on algal bioassays, that the lake is severely nitrogen-limited (Scott *et at** 1977, 1980). However, the relatively high concentrations of inorganic nitrogen found in Hartbeespoort Dam promote the growth of non-nitrogen-fixing blue-green algae, such as *Microoystis* (Home & Viner 1971), which can competitively exclude nitrogen-fixing blue-green algae (Section 5.4.7). The high concentrations of NH4-N found in the lake would also inhibit bacterial nitrogen fixation (Fainter 1970, Brezonik 1972). Thus, in contrast to the case of nearby Rietvlei Dam where nitrogen fixation can provide up to 40% of the annual nitrogen load (Ashton 1981), planktonic nitrogen fixation appears to be unimportant in Hartbeespoort Dam.

Extensive marginal communities of submerged macrophytes and their associated epiphytic flora can develop during periods when lake levels are stable. Much of this epiphytic flora is composed of heterocystous blue-green algae (Ashton - unpublished data) and Rogers (1981) has shown that similar communities in a Pongola flood plain pan are capable of fixing atmospheric nitrogen. However, fluctuating lake levels during the present study prevented the development of submerged macrophyte communities and eliminated the possibility of epiphytic nitrogen fixation. During this period, no nitrogen fixation was detectable in the weakly-developed periphytic communities.

### (b) Nitrification

Nitrification is the process whereby ammonia is oxidized first to nitrite and then to nitrate by a select group of aerobic autotrophic bacteria which obtain their energy by nitrogen oxidation and their cellular carbon by reduction of carbon dioxide. It has also been reported that a variety of heterotrophic bacteria, actinomycetes and fungi are capable of nitrification, generally at much slower rates (Brezonik 1972, Bostrom 1981). However, the role of these organisms in aquatic nitrification is not well understood. The significance of nitrification in the nitrogen cycle lies in the conversion of labile ammonia (ammonium ion), which may be lost from solution by sorption onto sediments and by volatilization at high pH, to a more stable form (nitrate). On the other hand, denitrification reactions reduce nitrate to molecular nitrogen, and nitrification is thus important in producing the reactants for this nitrogen sink.

Laboratory experiments on Hartbeespoort Dam water and sediment samples indicated that the two nitrification steps, NH^ to $N0_2$ and $N0_2$ to $N0_3$, occurred in two distinct phases, confirming the results obtained by Cavari & Phelps (1977). The far higher rates of nitrification in surface sediment samples reflect the greater numbers of nitrifying organisms present in surface sediments compared to lake water (Cavari 1977). The rates of nitrification activity in Hartbeespoort Dam water and sediment samples are shown in Table 5.15. Though these values are potential rates of activity, based on the oxidation of known NH^-N spikes, they provide a basis for comparison with other studies. Field measurements using unspiked surface water samples yielded nitrification rates of 2.11 to 2.46 yg *8.* $^1$ h $^1$.

These are equal to approximately half the potential rates of activity measured in spiked surface water samples from Hartbeespoort Dam and 'slightly higher than *in situ* rates measured by Christofi *et at.* (1981) in eutrophic Blelham Tarn. Nitrification potentials measured in surface samples of Hartbeespoort Dam sediments and at the ammonia-oxygen chemocline in summer were up to 10 times higher (Table 5.15); these rates were approximately five times higher than laboratory rates measured by Henriksen (1980) in surficial sediments from shallow Danish fjords.

The Hartbeespoort Dam results support other observations that nitrification in lakes is usually highest at sediment - water interfaces under aerobic conditions (Serruya 1975; Isirimah, Keeney & Dettmann 1976; Stewart, Sinada, Christofi & Daft 1977; Bostrom 1981) and confirm that the process can be significant in

Table 5.15. Nitrification potentials in Hartbeespoort Dam water and sediment sampes. (All values converted from times taken to oxidize a known spike of NH^ to $N0_2$). (Each value is a mean of four replicates).

| Month | Depth at which sample collected | | | | |
|---|---|---|---|---|---|
| | Surface sediments | 1 m above sediments | Below water surface 10 m | 5 m | 1 m |
| | ($\mu g\ g^{-1}\ h^{-1}$) | | ($\mu g\ \ell^{-1}\ h^{-1}$) | | |
| Jun. 83 | 21 | 16 | 13 | 7 | 5 |
| Jul. 83 | 21 | 17 | 11 | 6 | 4 |
| Aug. 83 | 21 | 16 | 14 | 7 | 4 |
| Sep. 83 | 12 | 9 | 18 | 5 | 3 |
| Oct. 83 | 5 | 6 | 21 | 5 | 5 |
| Nov. 83 | 2 | 4 | 22 | 4 | 4 |
| Dec. 83 | 1 | 1 | 22 | 5 | 3 |
| Jan. 84 | 1 | 1 | 21 | 7 | 5 |
| Mar. 84 | 1 | 1 | 16 | 6 | 5 |
| Apr. 84 | 28 | 23 | 10 | 7 | 5 |

the water column during summer stratification (Burns & Ross 1972; Hall *et at.* 1978; Takahashi, Yoshioka & Saijo 1982). The development of zones of nitrification at the ammonium-oxygen chemoclines (Table 5.15), located approximately 10m below the water surface, is of particular interest when the seasonal patterns of deoxygenation and sediment NHt^-N release recorded in Hartbeespoort Dam and other eutrophic If Ices subject to stratification are considered (Christofi, Preston & Stewart 1981). The observations by Cavari (1977) and Bostrb'm (1981) that nitrification rates in profundal waters were greatest at overturn were confirmed in this study (Table 5.15).

Nitrification significantly modifies the aquatic environment by increasing the oxygen demand because two moles of oxygen are required to oxidize one mole of ammonia (Brezonik 1977, Cavari 1977). However, even though the presence of oxygen is a necessary prerequisite for nitrification, very low levels of oxygen are sufficient since nitrification has been shown to occur down to 0.3 mg $0_2$ £ (Gunderson, Carlucci & Bostrb'm 1966). In anaerobic Hartbeespoort Dam samples, nitrification rates dropped to zero but rose rapidly when oxygen concentrations were increased above 1 mg $0_2$ *2* $^{l}$. Hall *et at.* (1978) showed that the areal hypolimnetic oxygen deficit due to nitrification in Grasmere amounted to some 100 mg $0_2$ m $^2$ d representing between 8 and 17% of the total lake oxygen consumption. This value was much higher than the 1.25% reported by Burns & Ross (1972) for Lake Erie. These observations that nitrification significantly increased the oxygen demand generated in hypolimnetic waters were substantiated by field observations and measurements in Hartbeespoort Dam. During the 1981 overturn, for example, oxygen concentrations dropped below 1 mg *2* throughout the water column and $N0_3$-N concentrations rose from less than 50 yg *Z* * to over 1 500 yg *Z* in ten days (Robarts *et at.* 1982). Similar, though less marked, patterns were recorded at each successive overturn (Fig. 5.32).

In laboratory experiments conducted between 15 and 30 °C, nitrification was found to have a Qio value of 1.83 (range = 1.80 to 1.86, n = 18). This value compared well with the figure of 1.93 obtained by Klapwijk & Snodgrass (1982). Thus, nitrification is weakly temperature-dependant, increasing rapidly with increasing temperature over the range of temperatures normally found in Hartbeespoort Dam, with highest rates in summer.

Bacteria of the genera *Nitrosomonas* (which convert NHi* into $N0_2$) and *Nitrobacter* (which convert $N0_2$ into $N0_3$), are the principal but not the only agents of nitrification (Alexander 1961, Fainter 1970). These bacteria are obligate chemoautotrophs and, interestingly, seem to be inhibited by high substrate (i.e. NH4) concentrations (Vincent, Downes & Vincent 1981). This was substantiated in laboratory experiments using Hartbeespoort Dam samples - nitrification rates increased with increasing NH^-N concentrations up to 2 mg *2* $^{l}$ and then levelled off. At NH4-N concentrations above 10 mg *Z* *, nitrification rates decreased to a minimum. Thus, the hypolimnetic NHi+-N concentrations found in Hartbeespoort Dam at overturn (up to 7 mg *Z* ) are sufficient to sustain maximal nitrification rates. Similarly, the high NHi+-N concentrations found at or below the oxycline in summer (up to 1.8 mg *Z* ) can also sustain high nitrification rates.

Experiments to determine whether the pH range found in the profundal zone of Hartbeespoort Dam (7.0 to 8.5) affected nitrification rates were inconclusive. Core samples of Hartbeespoort Dam sediments and overlying water appeared to be able to rapidly change the adjusted pH to its original value.

### ($^c$) Denitrification

Denitrification consists of the following sequence of transformations , involving four separate and identifiable enzyme systems:

$$NO_3 \rightarrow NO_2 \rightarrow NO \rightarrow N_2O \rightarrow N_2$$

In denitrification, nitrate serves as a terminal electron acceptor by facultative aerobic and anaerobic bacteria, usually in the absence of oxygen, and nitrite is formed as the first intermediate in the process (Delwiche & Bryan 1976). Since the principal end product ($N_2$) is a nitrogen form not utilizable by most organisms, this reaction generally acts as a nitrogen sink (Brezonik 1972, Andersen 1977, Cavari & Phelps 1977, Vincent *et al.* 1981).

Lake sediments have long been recognized as a major site for the process (Nedwell 1975, Brezonik 1977, Knowles 1979, Jones *et al.* 1980, Jones & Simon 1981). The importance of sediments and profundal zone water as sites for denitrification was confirmed in Hartbeespoort Dam. Significant rates of denitrification were measured in intact sediment cores and water samples collected from 1 m above the sediment surface during winter (Table 5.16). No denitrification was detected in water samples collected during winter at depths of 1, 5 and 10m below the lake surface where dissolved oxygen concentrations in the water column were in excess of 6.0 mg -? . A comparison of littoral (water depth = 5 m) and profundal (water depth = 18 m) denitrification rates revealed that denitrification rates in the profundal zone were approximately 100% higher than those in the littoral zone (Table 5.16).

Table 5.16. Denitrification rates ($N_2O$ production rates) in water samples and intact sediment cores from Hartbeespoort Dam. (Each value is a mean of four replicates).

| Zone | Depth below water surface | | | 1 m above sediments | intact sediment core ($\mu$mol $m^{-2}$ $h^{-1}$) |
|---|---|---|---|---|---|
| | 1 m | 5 m | 10 m ($\mu$mol $\ell^{-1}$ $h^{-1}$) | | |
| Littoral | - | - | ND | 0.02 | 102 |
| Profundal | ND | ND | ND | 0.04 | 200 |

ND = not detectable; - = depth not sampled (too shallow)

The seasonal variation in denitrification rates measured in intact sediment cores and water samples is shown in Table 5.17. From these results it is evident that the low denitrification rates measured in profundal water samples during winter increased markedly at the onset of stratification. After this increase, denitrification rates dropped rapidly to near zero, probably as a result of depletion of substrate ($NO_3$-N). At the same time, water samples collected 10 m below the lake surface showed increased denitrification activity. Therefore, it can be concluded that as stratification develops, denitrification at or just above the sediment surface decreases due to lack of substrate and the zone of maximum denitrification activity follows the oxycline upwards, using the supply of NO3-N produced during nitrification. At overturn, when oxygenated water is mixed down to the sediment surface, nitrification of accumulated NH^-N would provide further substrate for denitrification activity at the sediment surface, giving rise to the increased rates of denitrification (Table 5.17) during March and April.

The presence of measurable denitrification activity in aerobic water samples corroborates the observations of Cavari & Phelps (1977). However, this has several implications since the role of oxygen has been surrounded by some degree of controversy' (Brezonik 1977, Delwiche & Bryan 1976) and therefore its potential controlling effect has been considered in some detail (Cavari & Phelps 1977, Nakajima, Hayamizu & Nishimura 1984a, 1984b). Tiren *et at.* (1976) pointed out that denitrification could occur in sediments even though the overlying water contained 6 to 8 rag $0_2$ £ . The possible occurrence of anoxic microsites in apparently aerobic sediments has also been considered by Brezonik (1977) and Nakajima *et at,* (1984a, 1984b). Oxygen concentration, in turn, exerts a marked effect on the redox potential (Eh) of sediments and the inter-relationship of Eh and denitrifying activity has attracted considerable attention (e.g. Jones *et at,* 1980). Kessel (1978) concluded that denitrification occurred below the oxidized surface sediments where the Eh was not greater than +100 mV. This agreed well with the findings of Johnston, Holding & McCluskie (1974) who observed an Eh shift from +340 mV to +100 mV as oxygen was used up and a poising of the sediment Eh at this lower level until the nitrate was consumed. Graetz, Keeney & Aspiras (1973) reported that nitrate stabilized sediment Eh at between +200 mV and +100 mV.

The highest denitrification rates measured in water samples and intact sediment cores from Hartbeespoort Dam (Table 5.17) are comparable to, but often exceed, the highest values recorded in the literature. In water samples, Goering & Dugdale (1966) recorded 1.1 Mmol *2 d* beneath the ice cover of an Arctic lake while Brezonik & Lee (1968) measured 0.6 - 1.6 ymol *2* d $^1$ in the hypolimnion of a _Florida lake and Larsen (1977) measured 0.2 to 5.3 umol *&* $^1$ d $^1$ in the hypolimnetic waters of five Danish lakes. Messer & Brezonik (1983) reported denitrification rates of 1.4 to 3.6 mg N m $^2$ d for sediment samples from Lake Okeechobee, Florida. However, measured denitrification rates in lacustrine sediments are somewhat more variable than the results of Messer & Brezonik (1983) would infer. Vollenweider (1968) found 2.6 to 56.5 mg N m~$^2$ d"$^1$ for six

Table 5.17. Denitrification rates ($N_2O$ production rates) in water samples and intact sediment cores from Hartbeespoort Dam. (All samples from profundal zone of lake). Each value is a mean of four replicates).

| Month | Depth below water surface 1 m | 5 m<br>(µmol ℓ$^{-1}$ h$^{-1}$) | 10 m | 1 m above sediments (µmol ℓ$^{-1}$ h$^{-1}$) | intact sediment core (µmol m$^{-2}$ h$^{-1}$) |
|---|---|---|---|---|---|
| Jun. 83 | ND | ND | ND | 0.04 | 200 |
| Jul. 83 | ND | ND | ND | 0.04 | 195 |
| Aug. 83 | ND | ND | 0.01 | 0.05 | 220 |
| Sep. 83 | ND | ND | 0.04 | 0.19 | 295 |
| Oct. 83 | ND | ND | 0.09 | 0.48 | 145 |
| Nov. 83 | ND | ND | 0.11 | 0.08 | 32 |
| Dec. 84 | ND | ND | 0.15 | 0.02 | 22 |
| Jan. 84 | ND | ND | 0.16 | 0.02 | 20 |
| Mar. 84 | ND | ND | 0.07 | 0.59 | 160 |
| Apr. 84 | ND | ND | 0.03 | 0.92 | 315 |

ND = no denitrification activity detectable.

Swiss lakes while Andersen (1977) measured maximum rates of 100 to 500 mg N m $^{2}$ d in six unstratified Danish lakes. A study of 29 Danish lakes by Jensen & Dahl-Madsen (1978) showed great variability, with values ranging from 0.5 to 300 mg N m $^{2}$ d . Jones & Simon (1981) calculated that denitrification rates in profundal and littoral sediments were similar, though often higher in littoral sediments due to the rapid provision of substrate ($NO_3$) by parallel nitrification processes. In the above studies, losses of nitrogen from the different systems involved accounted for between 0.1 and 81% of the total annual nitrogen loading to the system.

The presence of high denitrification rates in Hartbeespoort Dam during the winter months (Table 5.17) implies that the process is not significantly affected by lower water temperatures. This hypothesis was checked in the laboratory by measuring rates of denitrification activity at temperatures between 15 and 25 °C . It was found that the Qio of denitrification activity in the laboratory was 1.37, a value which corresponds very closely to the value of 1.35 measured by Cavari & Phelps (1977). This temperature range covers most of the temperature experienced in the field and therefore provides an acceptable assessment of the effects of temperature on the process.

It was evident from this study that an adequate supply of substrate is a major feature regulating the rate of denitrification in Hartbeespoort Dam. Maximum rates of activity_were recorded at a substrate ($NO_3$-N) concentration of 3.0 mg ^ $^{1}_{>}$ a value occasionally attained in profundal waters during the winter months (Fig. 5.32A). Similarly, high denitrification rates were recorded at the ammonia-oxygen chemocline during summer stratification. Both of these zones are characterized by high nitrification rates and thus nitrification must be the major mechanism involved in providing substrate for denitrification leading to the subsequent loss of nitrogen from the impoundment. The dissolved organic carbon (DOC) concentrations found in Hartbeespoort Dam (4-12 mg £ $^{1}$; Robarts, unpublished data) would probably not be limiting to these processes.

Nitrification can influence the composition of algal populations (Liao & Lean 1978) by elimination of algal species which are unable to compete with bacterial nitrogen uptake and those species which preferentially utilize NH[+-N (Cavari 1977). When high nitrification rates supply sufficient N0$_3$-N to sustain high rates of denitrification, considerable quantities of nitrogen may be lost from the impoundment thus lowering the N:P ratio. This would provide a competitive advantage to blue-green algae such as *Microcystis* and *Anabaena* (Smith 1983).

### 5.4.5 Phosphorus cycling between the water and the phytoplankton

Lean & Nalewajko (1979) have proposed a radio-bioassay procedure using $^{32}$P to measure biotic phosphorus demand in lakes. This technique has been successfully applied in temperate lakes (Vincent 1981) and has been used in conjunction with standard chemical analytical techniques (Downes & Paerl 1978) and measurements of alkaline phosphatase activity (APA) to define the nutritional status of algae relative to P.

#### (a) Phosphorus turnover times

$^{32}$P uptake kinetics in Hartbeespoort Dam conformed to Lean's Type I, log-linear curve (Lean & Nalewajko 1979), and were log-linear over the 4 h incubation period throughout the study. Initial experiments indicated that the phosphorus uptake curves remained log-linear over a period of 54 h .

Type I, log-linear uptake curves are indicative of eutrophic systems (Lean & Nalewajko 1979). In studies of Heart Lake (Canada) Lean and Nalewajko found that such curves were often associated with nearly equivalent phosphorus uptake and release rates. Healey (1982) gives evidence that cyanobacterial uptake of phosphorus becomes passive under such conditions; e.g., phosphorus is simply moved through the cell from the water back into the water. Measured phosphorus release rates in Hartbeespoort Dam showed nearly 100% release of $^{32}$P into the medium over a short period (<1 h). Subsequently, re-uptake over longer term (2-3 h) was observed which resulted in a 90% release of $^{32}$P on average. These findings support the findings of Lean & Nalewajko (1979) who found that Type I curves were associated with long turnover times. Hartbeespoort Dam phosphorus turnover times (Ptt) were long ranging from 83.2 ±0.2 (S.D.) d in August 1983 (the end of winter) to 0.4 ± 0.0 d in June 1983. No pattern of seasonal variation in pH was evident, nor could these occassional very low values be reationally explained. Long turnover times suggest (a) high ambient P concentrations, (b) low algal biomass, (c) algal phosphorus sufficiency, and/or (d) other conditions not conducive to algal growth (Rigler 1964, 1966, Peters 1975, Lean & Nalewajko 1979, White *et at.* 1982). Low algal biomass can be dismissed as a potential reason for long Ptt in Hartbeespoort Dam since minimum chlorophyll levels measured during the study period were in excess of 10 mg m $^{3}$ (Fig. 5.56) and averaged 140 mg m $^{3}$ . Phosphorus concentrations, on the other hand,

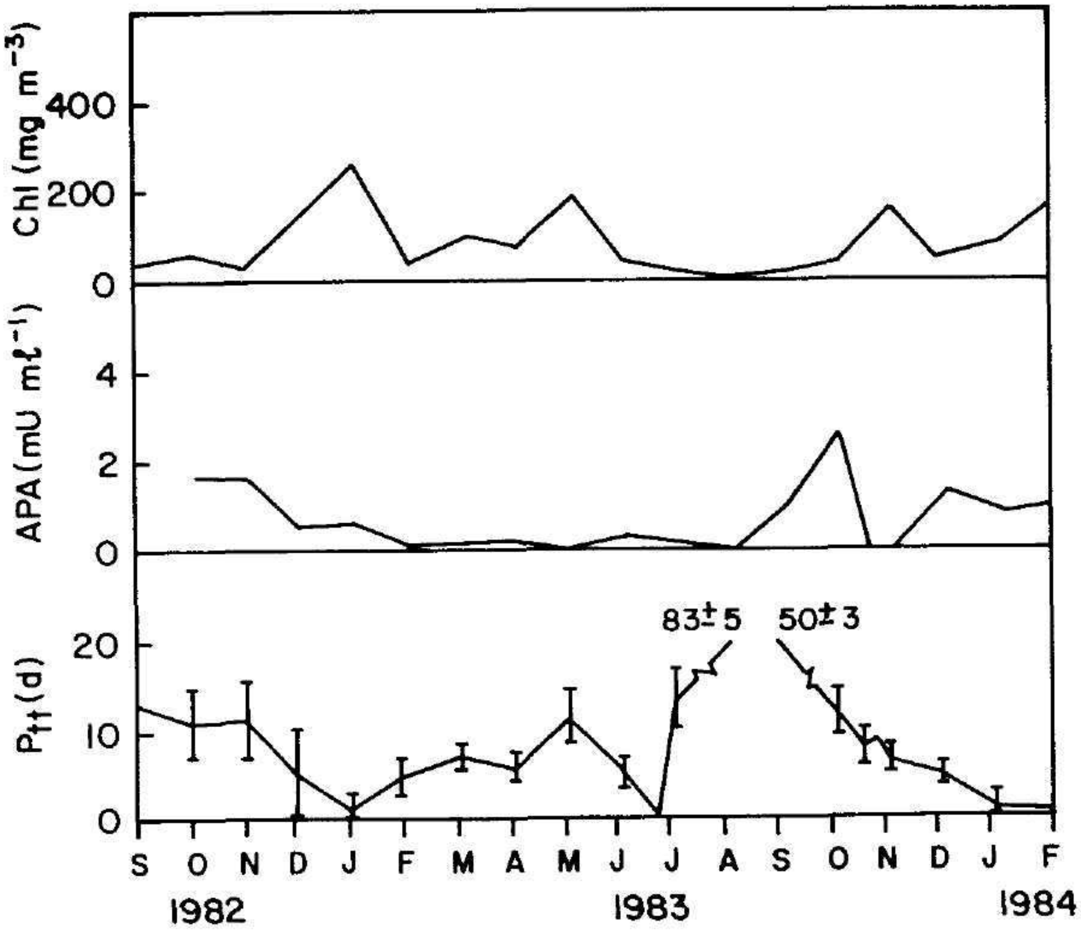


Figure 5.56. Phosphorus turnover times ($P_{tt}$, days) in relation to alkaline phosphatase activity (APA, mU mℓ$^{-1}$) and chlorophyll α (Chl, mg m$^{-3}$) in surface waters of Hartbeespoort Dam from September 1982 to February 1984. Vertical bars on $P_{tt}$ show standard deviation of measurements about the mean.

were extremely high during the study period, ranging from 0.29 to 2.05 mg *£ **, with more than 50% of the total phosphorus concentration being in the biologically available LMWP form (Section 5.2.5). This indicated that the algal population was P-sufficient, a factor which is discussed more fully below. With regard to other factors, Robarts (1984) has suggested that the underwater light climate is probably the most important factor influencing algal growth in Hartbeespoort Dam. However, Lean & Nalewajko (1979) and White *et at.* (1982) have failed to find any significant effect of light on Ptt using near surface samples.

Variations of_two orders of magnitude in light intensity (from 22.5 uE m $^2$ s $^l$ to between 550 and 2 200 uE m~$^2$ s $^l$*)* had little effect on turnover times at the $P<0.01$ level in Hartbeespoort Dam. Variations in filter pore diameter from 0.22 to 0.45um also had an insignificant effect on Ptt at the $P < 0.01$ level. This suggests a minimal bacterial uptake component since only the 0.22 ym filters would retain most bacteria (Robarts & Sephton 1984). Abiotic uptake was consistently an important component (<50%), of the total measured uptake. $^{32}$P release from abiotic particles was, however, lower than from similarly labelled biotic particles. Thus, of the factors influencing phosphorus turnover times in Hartbeespoort Dam, ambient P concentrations and algal P-sufficiency would appear to be the foremost reasons for the long turnover times recorded from the impoundment.

### (b) Phosphorus demand

Several methods are currently used to assess the demand of algal populations for phosphorus in aquatic systems (Vincent 1981); namely, $^{32}$P uptake kinetics, cellular phosphorus status, ATP response and APA (alkaline phosphatase activity). Of these methods $^{32}$P uptake kinetics and APA measurements were employed directly on samples from Hartbeespoort Dam. Cellular phosphorus accumulation, as an indirect measure of cellular phosphorus status, was estimated as particulate nitrogen (PN) to particulate phosphorus (PP) ratios after the method of White *et at.* (1982).

$^{32}$P turnover times have been discussed above (in Section (a)) and are extremely slow, indicative of an eutrophic system. APA was minimal and often at the limits of detection of the method employed (Fig. 5.56) . Maximum APA was measured during spring (October 1982) prior to the onset of stratification. APA decreased with decreasing Ptt until December when it increased slightly. The maximum value of APA, 48.4 mU *mH* *, was obtained on one occasion, but APA rarely exceeded 2.5 mU *m£* * throughout the study. Low values of APA are indicative of an enriched or P-sufficient system (Vincent 1981, Healey 1982). Cellular phosphorus status as estimated by PN:PP ratios varied from 0.3 to 2.4 during the study. These ratios suggest a deficiency of nitrogen which is consistent with bioassay evidence (Scott *et at.* 1977). At no time, however, did ambient nutrient concentrations reach a level where nitrogen limitation might seriously result. The phosphate deficiency index (PDI, as defined by Lean & Pick 1981 and Lean & White 1983) was variable but well within the range of P-sufficiency, All of the indices used suggest that phosphorus in the impoundment is present in sufficient quantity to meet the requirements of the algal population throughout the year.

### 5.4.6 Phytoplankton seasonal cycles

The seasonal changes of the phytoplankton community composition in the main basin of Hartbeespoort Dam during 1981-1984 are shown in Figure 5.57. Based on cell volume, as well as cell numbers, the blue-green alga (Cyanobacterium) *Mioroaystis aeruginosa* Ktitz. emend. Elenkin was the dominant phytoplankton species in the lake for most of the year. *M. aeruginosa* reached its lowest concentrations in late winter and spring each year when it represented less than 25% of the total phytoplankton volume. The filamentous diatom *Melosira granulata* appeared for short periods in spring (1981), or in winter (1982) and occasionally dominated the phytoplankton populations for a few weeks. *Oooystis* sp. was the dominant alga during spring (September and October) each year along with other chlorophytes such as species of *Ankistrodesmus, Coelastrum, Cosmarium, Pediastrum* and *Scenedesmus.* By early summer *Microaystis aeruginosa* comprised more than 50% of the phytoplankton volume. The *Microcystis* population continued to increase so that by December it usually represented more than 90% of the total algal biomass. In 1981 and 1982 *Micvoeystis* dominated (>95% by volume) the phytoplankton through overturn into May (early winter). For 6 to 7 months the phytoplankton population of the lake was virtually unialgal. During June and July the *Mieroeystis aeruginosa* population slowly declined until *Melosira granulata* or chlorophytes replaced it as the dominant species in late August. In 1983 the successional episode extended longer, into January-February 1984, and cryptophytes were more abundant than in previous years. A late reappearance of *Melosira* in January was also noted and *Miovocystis* dominance was established only in February.

### 5.4.7 Phytoplankton biomass

Hartbeespoort Dam is characterized by dense algal populations that are maintained virtually throughout the year, due to warm water temperatures (Fig. 5.3), high insolation (Fig. 5.2) and abundant nutrients (Figs. 5.32, 5.38).

A diagram of the depth versus time distribution of chlorophyll *a* concentration (Fig. 5.58) at station 1 (Fig. 4.1) shows the temporal changes that occurred in the algal population of the main basin of this hypertrophic lake. Surface (top 11 cm) chlorophyll *a* concentrations ranged over 3 orders of magnitude from minima lower than 5 mg m$^{3}$ (usually following overturn) to summer maxima exceeding 1 000 mg m$^{3}$. Usually the bulk of the chlorophyll *a* was contained in the upper 4-5 m of the water column, except immediately after overturn or during windy periods, when the vertical distribution of chlorophyll was homogeneous. The distribution of the algal populations varied markedly with time and space, depending primarily on the wind regime.

Figure 5.59 shows the temporal changes in mean 0-5 m chlorophyll concentrations (hose-pipe samples) at three pelagic stations (1, 4 and 5, Fig. 4.1). Horizontal variability in chlorophyll concentrations was high when *Uieroeystis* dominated the phytoplankton. This species has a buoyancy mechanism (Reynolds & Walsby 1975) that

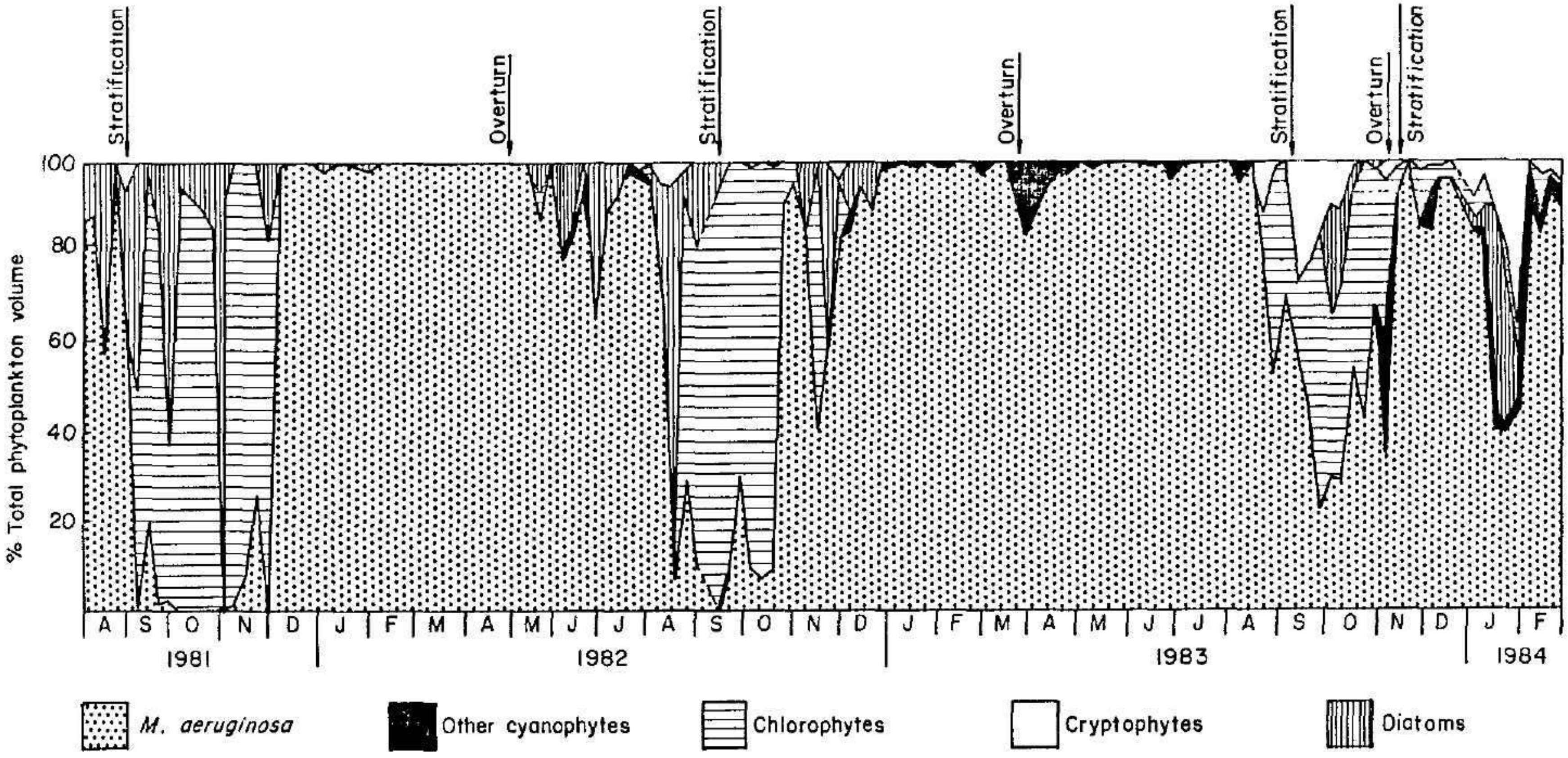


Figure 5.57. Seasonal variations in the composition of the phytoplankton community in the upper 8 m of the main basin of Hartbeespoort Dam, 1981-1984 (Zohary, in preparation).

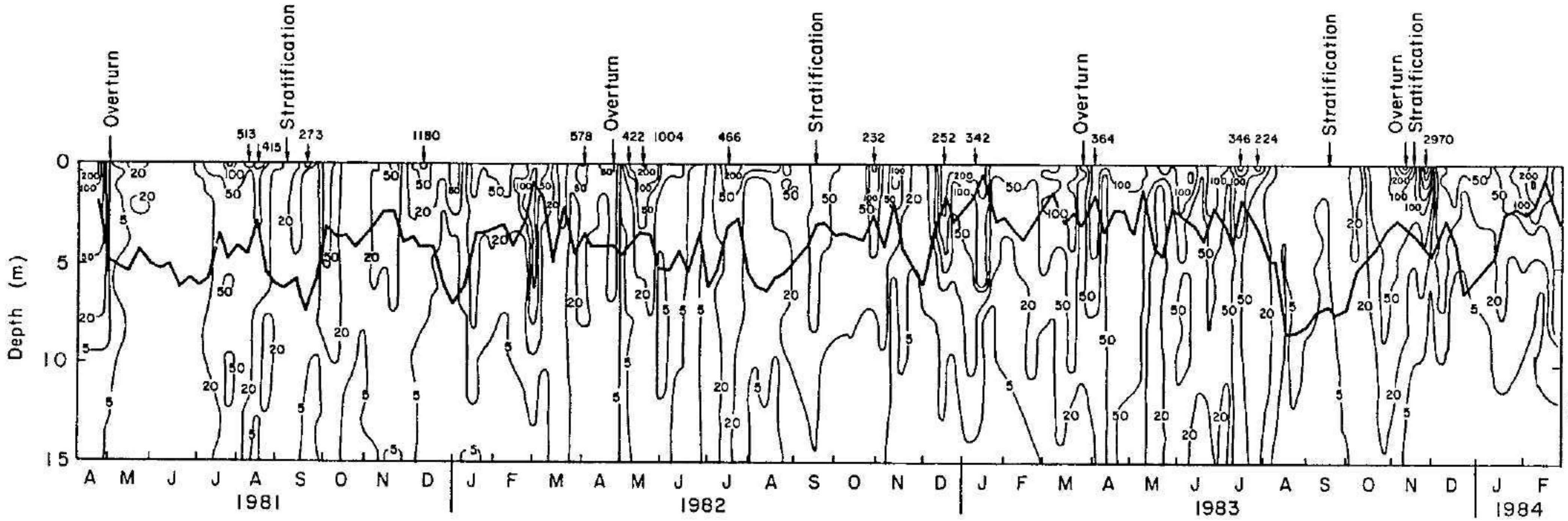


Figure 5.58. Depth-time distribution of chlorophyll α concentrations (mg $m^{-3}$) at the main basin of Hartbeespoort Dam, 1981-1984. The thick line indicates the depth of the euphotic zone.

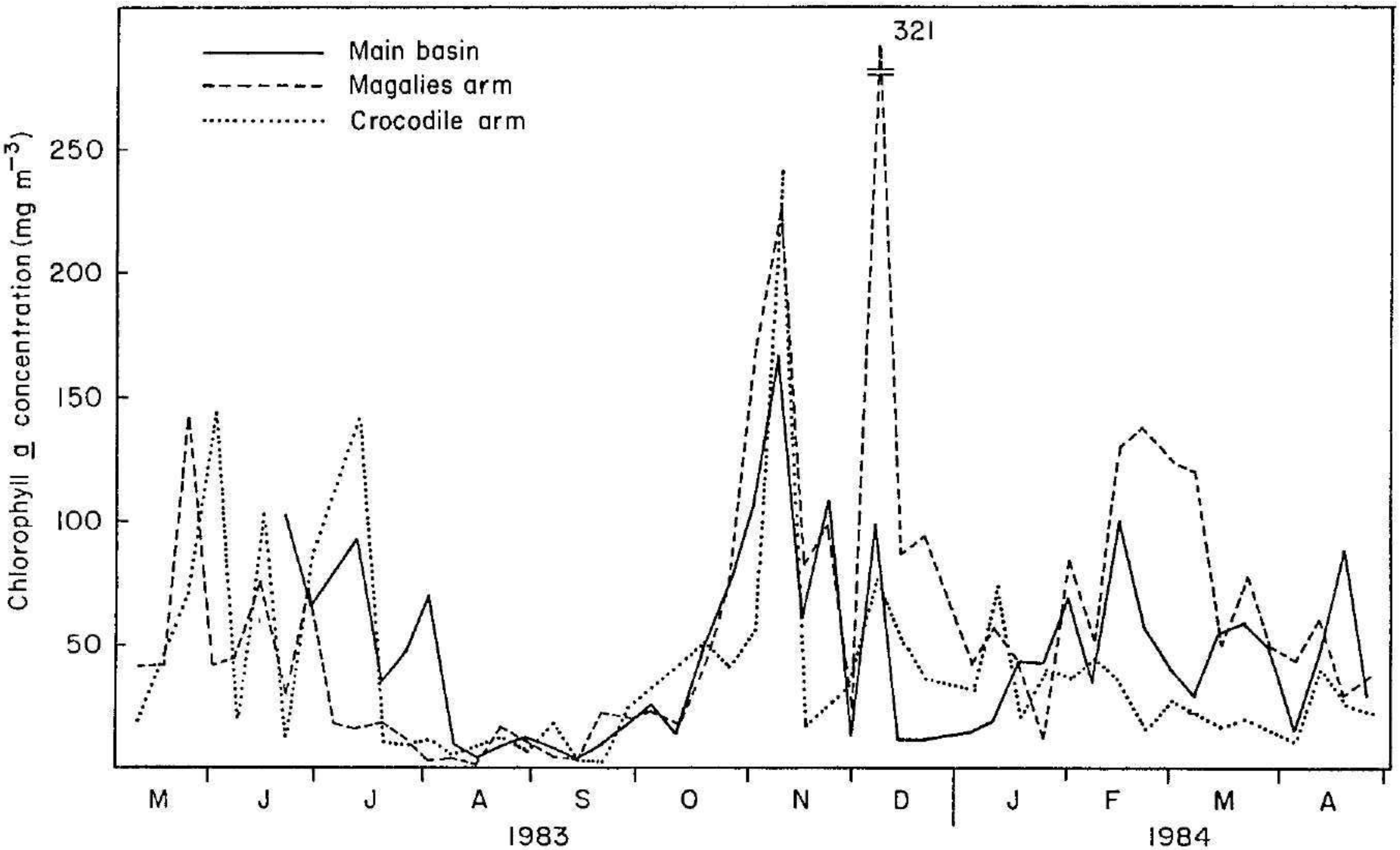


Figure 5.59. Temporal changes of mean chlorophyll concentration in the upper 5 m of the water column at 3 pelagic stations.

allows it to float to the surface during calm weather. *Miovocystis* drifted with surface currents, accumulating temporarily in different parts of the lake. As a result, week to week variations in chlorophyll concentrations and *Microoystis* biomass were sometimes more pronounced than the seasonal changes (e.g., December 1983, station 4 in Fig. 5.59, see also Fig. 5.67). The horizontal distribution of chlorophyll was more homogeneous in spring, when *Mi-ovocystis* declined and non-buoyant algal species were dominant (Fig. 5.57) while wind speeds increased (Fig. 5.5).

### 5.4.8 *Micvooystis* hyperscums

During prolonged periods of calm weather, buoyant *Micvooystis* colonies drifted and accumulated at wind-protected lee shores, forming thick, crusted, surface scums in which chlorophyll *a* concentrations usually exceeded 100 000 mg m $^{3}$ . These scums often exceeded a hectare in area, measured several decimeters in thickness and remained in the same site for between a few days and several months (Zohary 1985). These accumulations are termed hyperscums to distinguish them from the more temporary, thin, surface films of blue-green algae that are known as water blooms or scums.

The largest hyperscums formed in Hartbeespoort Dam in winter, when wind speeds were lowest (Section 5.1.2). When water levels changed, hyperscums formed at different wind-protected locations at different times. At full supply level a several km long hyperscum formed in the Crocodile River inlet in May-July 1982. At the low water level of winter 1983 a hyperscum formed against the dam wall. This existed for 3^ months and contained up to two tonnes of chlorophyll ji and up to 70% of the total phytoplankton biomass (Fig. 5.60).

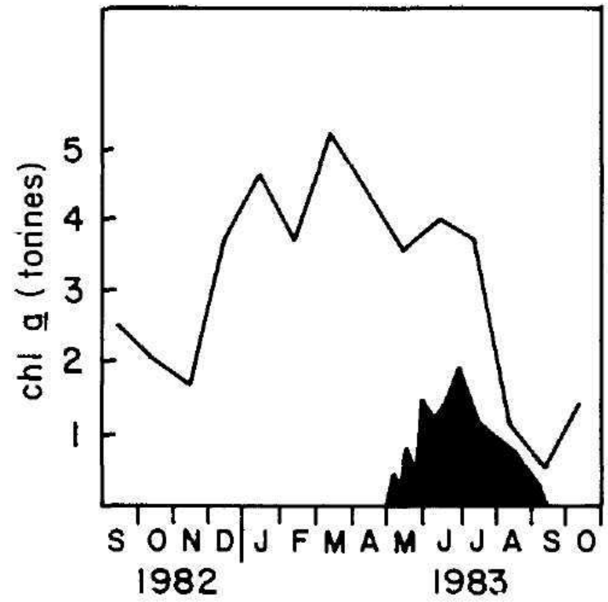


Figure 5.60. Total chlorophyll *a* in Hartbeespoort Dam (area below line) and the proportion of the total that was contained in a *Microcystis* hyperscum (shaded area).

Due to the extremely high algal density the hyperscum environment differed markedly from the open water environment. In Table 5.18 ranges of various environmental parameters measured over the same period of time in a hyperscum and in the main basin are compared.

The hyperscum environment was continuously aphotic and anaerobic with a fluctuating temperature regime and low pH values. With time conditions became increasingly reduced, sulfide and ammonia accumulated while nitrate and nitrite were depleted. Radiocarbon uptake experiments have shown that *M-icrocystis* remained viable in the hyperscums for at least two months, retaining its phytosynthetic capacity (Zohary 1985). Therefore, when hyperscums broke with increased wind speeds the lake was inoculated with substantial amounts of *Miorooystis**

Table 5.18. Comparison of ranges of several physical and chemical parameters from a hyperscum and from surface water at the main basin of Hartbeespoort Dam, for the period 1 June to 11 September 1983.

| | Hyperscum | Main basin surface water |
|---|---|---|
| Surface temperature, °C | 4.2 - 23.4 | 12.5 - 16.7 |
| Euphotic zone depth, m | 0 | 1.8 - 8.4 |
| % Oxygen saturation | 0 | 65.5 - 125.3 |
| $H_2S$, mg $\ell^{-1}$ | 1 - 10 | 0 |
| pH | 6.2 - 6.8 | 8.8 - 9.6 |
| $NH_4$-N, µg $\ell^{-1}$ | 120 - 273 000 | 70 - 700 |
| $NO_2$+$NO_3$-N, µg $\ell^{-1}$ | 20 - 580 | 590 - 2 700 |
| SRP, µg $\ell^{-1}$ | 300 - 56 000 | 360 - 850 |
| chl *a*, µg $\ell^{-1}$ | 144 400 - 302 000 | 4.2 - 346 |

### 5.4.9 Algal biomass and underwater light attenuation

Robarts & Zohary (1984) have demonstrated that chlorophyll *a* (algal biomass) affected the visible, or photosynthetically available radiation, region of the light spectrum in Hartbeespoort Dam, The presence of even a relatively small concentration of chlorophyll *a* (14.7 mg $m^{3}$) significantly increased the absorption of light in Hartbeespoort Dam water indicating that chlorophyll *a* was a major factor determining underwater light attenuation in the lake. This can also be seen in the depth-time diagram of chlorophyll *a* in which the changes in the euphotic zone depth ($z_{eu}$) are also indicated (Fig. 5.58).

The regression equation for the relationship between e (the attenuation of green light) and the mean chlorophyll *a* concentration of the euphotic zone (B) of Hartbeespoort Dam was

$$e_{mm} = 0.65 + 0.012 B \ (r = 0.86, N = 148, P = 0.00001).$$

The relationship between chlorophyll *a* concentration and attenuation accounted for 73% of the variance suggesting other factors were also influential.

The slope of the above regression line, e (the incremental increase in the attenuation coefficient per unit chlorophyll, 0.012 $m^2$ mg *), is similar to that reported by others (Table 5.19). The intercept (e , $m^2$ mg *) represents the proportion of light extinction due to factors other than chlorophyll a. £ was low compared with values reported from lakes of known inorganic turbidity such as Lake George, Uganda where e was 2.55 to 2.83 (Ganf 1974a). Both the slope and intercept are least squares estimates and should be used with caution (Dubinsky 1980). The theoretical proportion of light attenuation due to chlorophyll *a* concentration for each date was estimated as (e . -e )/e . . The attenuation of light due to chlorophyll *a* ranged from non-detectable up to 92% in January 1983 (X = 41.1%).

**Table 5.19.** Comparison of *in situ* derived values of $\varepsilon_s$ ($m^2\ mg^{-1}$) from various lakes with the $\varepsilon_s$ value obtained for Hartbeespoort Dam (South Africa) phytoplankton (after Robarts & Zohary 1984).

| Dominant alga | $\varepsilon_s$ | Reference |
|---|---|---|
| *Stephanodiscus hantzchii* | 0.007 | Jewson 1977 |
| nannoplankton | 0.0086 | Bindloss 1974 |
| *Stephanodiscus hantzchii* | 0.009 | Jones 1977 |
| *Oscillatoria agardhii* | 0.010 - 0.011 | Jewson 1977 |
| *Microcystis aeruginosa* | 0.012 | Robarts & Zohary 1984, Hartbeespoort Dam |
| *Microcystis aeruginosa* | 0.012 - 0.016 | Ganf 1974a |
| *Anabaena circinalis* | 0.013 | Reynolds 1975 |
| *Microcystis aeruginosa* | 0.021 | Robarts 1979 |
| *Oscillatoria agardhii* | 0.027 | Jones 1977 |

In theory, when chlorophyll *a* is the dominant (>50%) factor regulating light attenuation, the size of the organism or colony containing it, should have a measurable effect on the depth of the euphotic zone. The larger the particle for a constant chlorophyll *a* concentration the lesser should be the light attenuation and hence the deeper the euphotic zone depth (Kirk 1975). In Hartbeespoort Dam, changes in size of chlorophyll-containing particles were primarily due to changes in colony size of *Micvoaystis aeruginosa.* It was not possible using field data to measure the effect of colony size on z at a constant chlorophyll concentration. Instead, using the l?^2/83 data, 11 dates were selected by Robarts & Zohary (1984) on which *Miarocystis aeruginosa* dominated the phytoplankton (> 90% by volume) and chlorophyll *a* concentration was the major (50 - 80%) factor controlling light attenuation. When chlorophyll accounted for less than 50% of the attenuation, the effect of factors such as gelbstoff and changes in chlorophyll *a* concentration could not be separated from those of changing colony size. Similarly, when chlorophyll accounted for more than 80% of light attenuation, ambient chlorophyll concentrations exceeded 185 mg m $^3$ and the method of analysis was not sensitive enough to separate effects of colony size from the effect of high chlorophyll concentrations on light attenuation (Robarts & Zohary 1984). Colony size was expressed as pg chl. *a* per *Micvocy&tis aeruginosa,* colony. The hypothesis of Robarts & Zohary (1984) was that when chlorophyll *a* concentration was the dominant factor controlling light attenuation $z_{fiU}$ would decrease as the amount of chlorophyll *a* per colony decreased".

The relationship between z and *Mi-cvocyst-is aerug-Cnosa* colony size is shown in Figure 5.61. The significant correlation calculated supported the hypothesis. However, the data points covered a wide range of chlorophyll concentrations (41 - 185 mg m $^{3}$) and the relationship may have been due to this variation. The data were plotted as z versus chlorophyll *a* for these 11 data points (Fig. 5.61). wo significant (P <0.90) relationship could be established. This indicated that the relationship between colony size and z in Figure 5.61 was not due to variations in chlorophyll concentrations. It was concluded by Robarts & Zohary (1984) that colony size did have a significant effect on light attenuation and z in Hartbeespoort Dam.

A laboratory experiment was also carried out by Robarts & Zohary (1984) to examine the effect of water turbulence on colony size and the effect of colony size on light attenuation when chlorophyll *a* accounted for greater than 95% of attenuation. They found that large colonies of *Microcystis* subjected to a water current of 4-5 cm s $^{1}$ rapidly fractured. The rate of colony fracture decreased logarithmically and after 1.5 h of turbulent mixing only small changes occurred. During the 8 h experiment chlorophyll *a* concentration changed slightly from 1 590 to 1 560 mg m $^{3}$ . As colony size decreased with time the depth of the euphotic zone decreased and e increased from 0.007 to 0.014 m$^{2}$ mg $^{1}$ . The data demonstrated that at a high chlorophyll *a* concentration, changes in *Micro-cystis* colony size affected light attenuation.

### 5.4.10 Primary production and its relationship to underwater light, temperature, algal biomass, nitrogen and phosphorus

Representative depth-profiles of primary production for Hartbeespoort Dam are shown in Figure 5.62. A , the maximum rate of photosynthesis in the depth profile, rangea Between 12.4 mg C m $^{3}$h * in April 1981 and 5916 mg C m $^{3}$h $^{1}$ in December 1981 (Robarts 1984). These were the lowest and highest values recorded during the three year period (Table 5.20). Calculation of volumetric production per unit chlorophyll *a* showed that >70% of the depth-profiles showed surface inhibition. Paerl, Tucker & Bland (1983) have shown that *Microcystis* can increase its carotenoid content thus protecting the cells from high light intensities as found at the surface of lakes. Their primary production data from North Carolina did not show surface Inhibition but their surface irradiance was only 1 300 uE m $^{2}$ s $^{1}$ in summer. In constrast, surface irradiance at Hartbeespoort Dam in summer was about 2 500 uE m~$^{2}$ s"$^{1}$ (Fig. 5.2). For Hartbeespoort Dam, the combination of surface inhibition and the attenuation of light in the water column, resulted in production being restricted to the upper 4 m of the water column (Fig. 5.62).

A representative diagram of the seasonal variation of the integral, hourly primary production (ZA, mg C m~$^{2}$h $^{1}$) for Hartbeespoort Dam (Fig. 5.63) shows a general trend to higher values in summer and lower values in winter. *ZA* ranged between 46.3 and 3380.7 mg

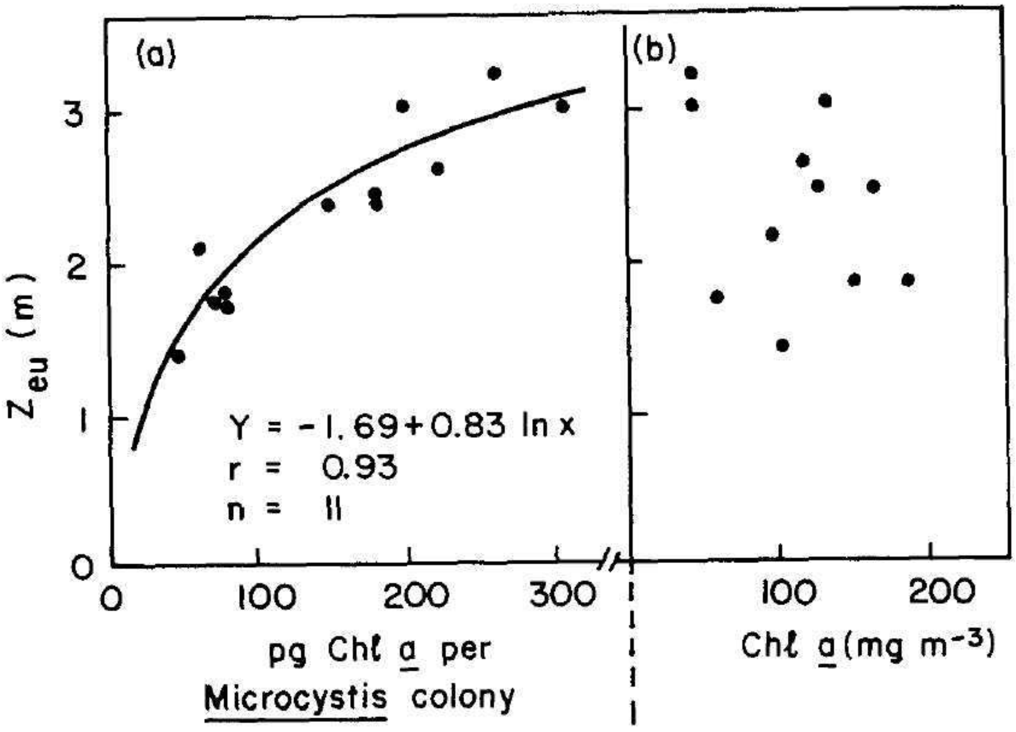


Figure 5.61(a). The relationship between euphotic zone depth ($Z_{eu}$) and mean *Microcystis* colony size (expressed as the chlorophyll α content per colony) in Hartbeespoort Dam.

(b) The relationship between $Z_{eu}$ and ambient chlorophyll α concentration for the points in Fig. 5.61(a) (modified from Robarts & Zohary 1984).

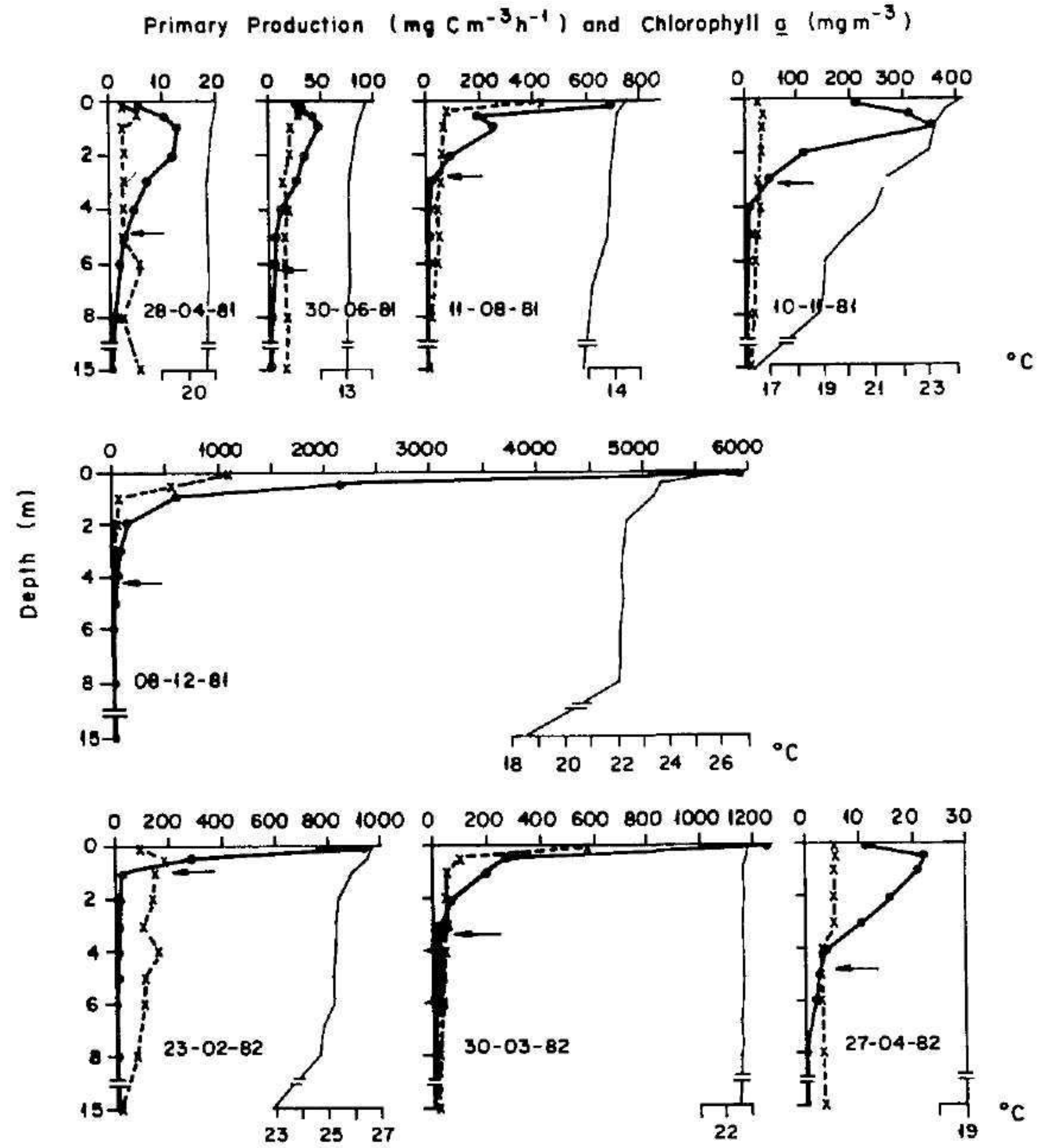


Figure 5.62 Representative depth-profiles of primary production (•) and chlorophyll concentration (x) in Hartbeespoort Dam (after Robarts 1984). The arrows indicate the depth of 1% of surface radiation. Thin lines are water temperature.

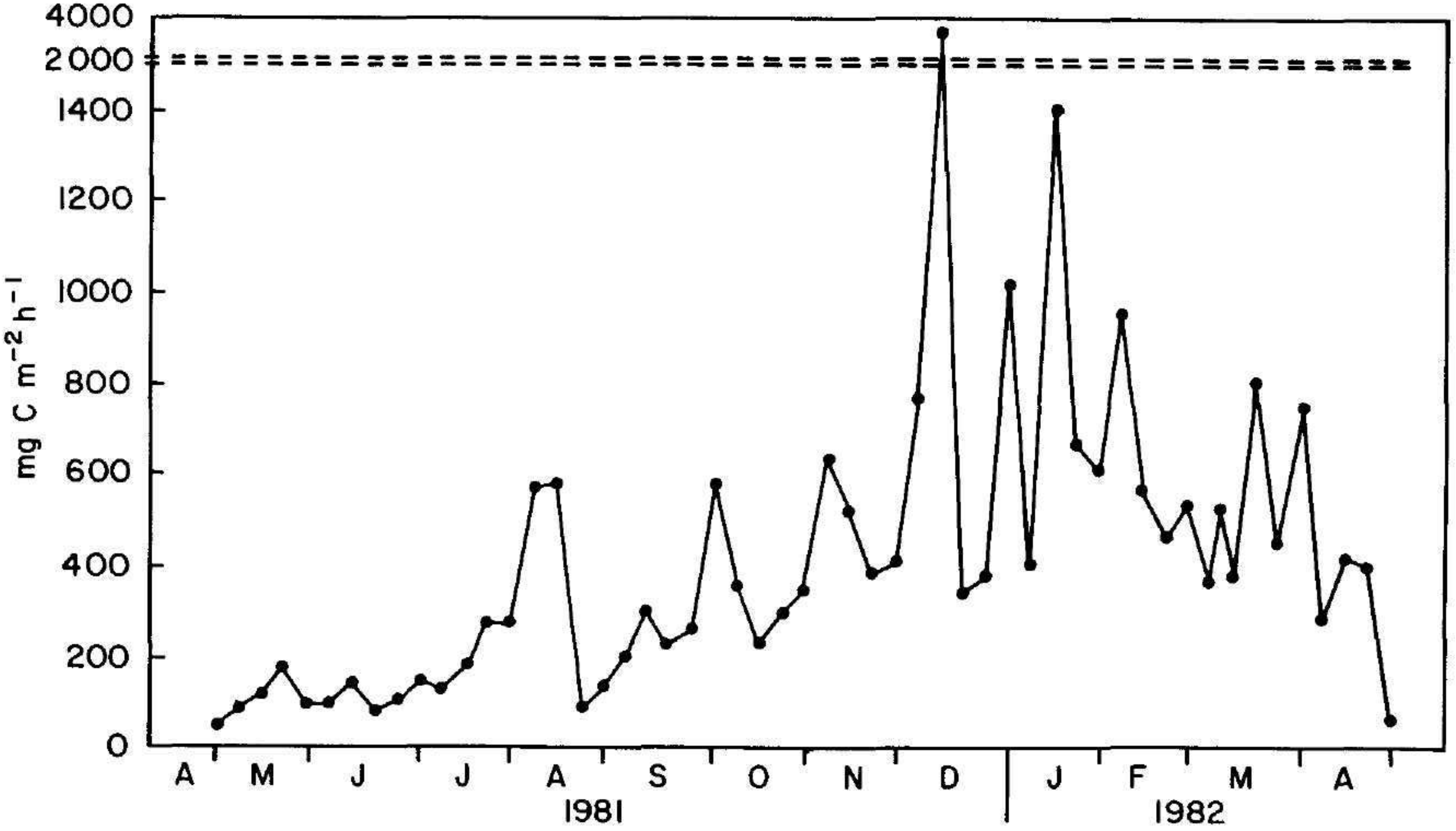


Figure 5.63. Representative seasonal cycle of integral, hourly primary production of Hartbeespoort Dam (modified from Robarts 1984).

C m"$^2$!*"$^1$ (Robarts 1984; Robarts & Zohary 1984) between 1981 and 1984 (Table 5.20). Statistical analyses of the means for these data indicated no significant difference between years. Hartbeespoort Dam is characterized by high £A values and on this basis appears to be the most productive freshwater (Table 5.21) or saline lake (cf. Hammer 1981) studied.

These data confirm the importance of light attenuation in the water to primary production. As algal biomass increased A also increased resulting in an increasing proportion of the phytoplankton becoming self-shaded. The result was that at very high chlorophyll *a* levels ZA approached A as light attenuation restricted the euphotic zone to approximately the area occupied by A . A s noted earlier, changes in *Miorooystis* colony size had a moderating influence on increasing light attenuation with increasing biomass but this effect was limited as the data from this study have demonstrated. The daily rates of areal photosynthesis (Z£A) in Hartbeespoort Dam ranged between 0.4 and 30.9 g C m $^z$d~$^1$ in 1981/84 (Robarts 1984, Table 5.20). A significant exponential relationship (r - 0.71, p = <0.001, n = 53) was found between *ttA* and the weekly solar radiation which accounted for 51% of the seasonal variation in 1981/82 (Robarts 1984). The importance of solar radiation to daily primary production decreased in each successive year. The correlation (r = 0.53, p = <0.001, n = 148) for the three years was significant but accounted for only about 28% of the variance in EZA. This was probably due to the stronger influence of changes in algal biomass and light attenuation.

**Table 5.20.** A comparison of Hartbeespoort Dam primary production data for the 3 year study period.

| | 1981/82 | 1982/83 | 1983/84 |
|---|---|---|---|
| $_{max}$ (mg C m$^{-3}$ h$^{-1}$) | 12.4 - 5916 | 13.8 - 1859 | 14.6 - 4087 |
| A (mg C m$^{-2}$ h$^{-1}$) | 46.9 - 3381 | 46.3 - 2292 | 57.6 - 3010 |
| ΣA (g C m$^{-2}$ d$^{-1}$) | 0.4 - 30.9 | 0.4 - 17.5 | 0.5 - 25.1 |
| nnual (kg C m$^{-2}$ y$^{-1}$) | 1.47 | 1.83 | 2.14 |
| hole lake (t C y$^{-}$ ) | 28 769 | 30 014 | 18 687 |

Using the daily primary production estimates the annual production for Hartbeespoort Dam varied from 1.47 kg C m"$^2$ for 1981/82, to 2.14 kg C m $^2$ for 1983/84 (Table 5.20). Annual whole lake production was similar in 1981/82 and 1982/83 but showed a marked decline in 1983/84 (Table 5.20) with the reduction in lake volume associated with the drought.

Temperature can be an important limiting factor to aquatic primary production with its effects on several physiological parameters. The photosynthetic saturation parameter, I, (the level of irradiance at which a cell's photosynthetic mechanism becomes saturated), varied widely in Hartbeespoort Dam ranging between 26.3 and 741.3 yE m~$^2$s~$^I$ . Raps *et at.* (1983) concluded that the I, for *Micro-oystis aeruginosa_* can exceed 565 yE m $^2$s $^1$ and recorded a value of up to I 400 uE m $^2$s $^1$ . Log I, was significantly related to water temperature (r = 0.49, p = <0.001, n = 139) as has been demonstrated by Tailing (1957), Robarts (1984) and others.

Table 5.21. A comparison of Hartbeespoort Dam primary production data with that recorded for other warm, freshwater lakes. Oxygen production data were converted using the relationship: mg C = 0.375 (mg $O_2$).

| Lake | $A_{max}$ (mg C $m^{-3}h^{-1}$) | $\Sigma A$ (mg C $m^{-2}h^{-1}$) | $\Sigma\Sigma A$ (g C $m^{-2}d^{-1}$) | Reference |
|---|---|---|---|---|
| Broa (Brazil) | - | - | 0.10 - 0.40 | Tundisi (1980) |
| Castanho (Brazil) | - | - | 0.50 - 1.50 | Schmidt (1973) |
| Chad (Chad) | 66 - 336 | 61 - 318 | 0.70 - 2.69 | Lemoalle (1973) |
| Crescent is. Crater (Kenya) | 19 - 68 | 105 - 293 | 1.13 - 3.15 | Melack (1979) |
| George (Uganda) | - | 375 - 750 | 1.95 - 5.80 | Ganf (1975) |
| Hartbeespoort (South Africa) | 12 - 5916 | 46 - 3381 | 0.40 - 30.90 | Robarts (1984)<br>Robarts & Zohary (1984) |
| D. Helvecio (Brazil) | - | - | 0.21 - 0.45 | Tundisi *et al* (1981) |
| Kinneret (Israel) | - | - | 0.56 - 8.05 | Berman (1976) |
| Lanao (Philippines) | - | - | 0.40 - 5.00 | Lewis (1974) |
| P.K. le Roux (South Africa) | - | 6*-33* | - | Hart, Allanson & Selkirk (1983) |
| McIlwaine (Zimbabwe) | 155 - 653 | 248 - 653 | 1.64 - 6.03 | Robarts (1979) |
| Midmar (South Africa) | 0.7 - 38.4 | 0.9 - 53.4 | 0.01 - 0.42 | Akhurst (1983) |
| Naivasha (Kenya) | 56 - 90 | 128 - 214 | 1.39 - 2.33 | Melack (1979) |
| Oloiden (Kenya) | 98 - 281 | 146 - 420 | 1.58 - 4.54 | Melack (1979) |
| Sibaya (South Africa) | 5 - 26 | - | 0.23 - 1.85 | Allanson (1979) |
| Swartvlei ** (South Africa) | 5 - 13 | 13 - 37 | - | Robarts (1976) |
| Winam Gulf (Kenya) | 86 - 240 | 150 - 341 | 1.61 - 3.68 | Melack (1979) |
| Wuras (South Africa) | 45 - 420 | 19 - 192 | 0.31 - 1.69 | Stegmann (1982) |

* range of means for 1977 - 83

** estuarine - lake ecosystem

Photosynthetic capacity, A /B, has great physiological and ecological significance as it is linked to the growth rates of phytoplankton assemblages (Harris, Haffner & Piccinin 1980). Many workers have found a general trend for increasing A /B with water temperature (see Robarts 1984). A /B from April l9§? to April 1984 ranged from 1.5 to 14.3 mg C (mg$^{m}$fifil *a)* $^{1}$h~$^{1}$ and was significantly (r - 0.74, P = <0.001, n = 149) related to water temperature (Fig. 5.64).

Falkowski (1981) noted that the variations in A /B are primarily due to the effects of nutrients, temperature, cell size and light history. Eppley (1972) has demonstrated that the algal carbon/chlorophyll *a* ratio is an important factor. The A /B values for Hartbeespoort Dam were within the range found by others but lower than the upper theoretical maximum of 25 mg C (mg Chi *a)* $^{l}$h calculated by Falkowski (1981). For Hamilton Harbor (Canada) phytoplankton, Harris & Piccinin (1977) predicted an A /B of 13 mg $0_2$. (mg Chi *a)~$^{l}$h~$^{l}$* (4.9 mg C[mg Chi *a]* $^{l}$*h~$^{l}$)* at *ffl*°C* . From the equation in_ Figure 5.64 A /B at 20 °C would be 5.6 mg C (mg Chi *a)* *h $^{1}$ . The similarity of these predictions would seem to support a universal relationship between photosynthetic capacity and temperature. However, such a conclusion may not be warranted because, as Li (1980) has noted, our understanding of the influence of temperature on phytoplankton growth in natural systems is inadequate.

Generally, there is a poor relationship between *Zh* and B (chlorophyll *a,* mg m $^{3}$) because, as Bindloss (1974) noted, the algal population density influences both the horizontal (A ) and vertical (e ) components of the photosynthesis depth-profile by self-shading. However, for Hartbeespoort Dam there was a significant power relationship between *ZA* and B (r = 0.78, P = <0.001, n = 147) and also a significant linear relationship between £A and £B (chlorophyll a, mg m~$^{2}$) (r = 0.74, P = <0.001, n = 147). These relationships indicated other factors were influential on 2A as already demonstrated.

Smith's (1979) study of primary production in North American lakes examined the relationships between production and nitrogen and phosphorus concentrations. The mean daily rate of photosynthesis at optimal depth, A , was correlated with mean total P ($r^2$ - 0.91). In contrast, integral rates of photosynthesis were linked less tightly to nutrient concentration because of their simultaneous dependence on transparency. The relationships between A and nutrients for Hartbeespoort Dam are given in Table 5.22.

The data in Table 5.22 show that there was no significant correlation between any photosynthetic parameter and nitrogen and phosphorus.

**Table 5.22.** **Correlation between some photosynthetic measurements and total phosphorus (TP), total nitrogen (TN) and $NH_4$-N for Hartbeespoort Dam during 1981/82.**

| | TP | | | TN | | | $NH_4$-N | | |
|---|---|---|---|---|---|---|---|---|---|
| | n | r | p | n | r | p | n | r | p |
| $\Sigma\Sigma A$ | 42 | 0.09 | >>0.1 | 45 | 0.06 | >>0.1 | 37 | 0.23 | >>0.1 |
| $\bar{A}_{opt}$ | 42 | 0.06 | >>0.1 | 45 | 0.11 | >>0.1 | 37 | 0.16 | >>0.1 |
| v | 42 | 0.04 | >>0,1 | 45 | 0.23 | >0.1 | 37 | 0.29 | >0.1 |

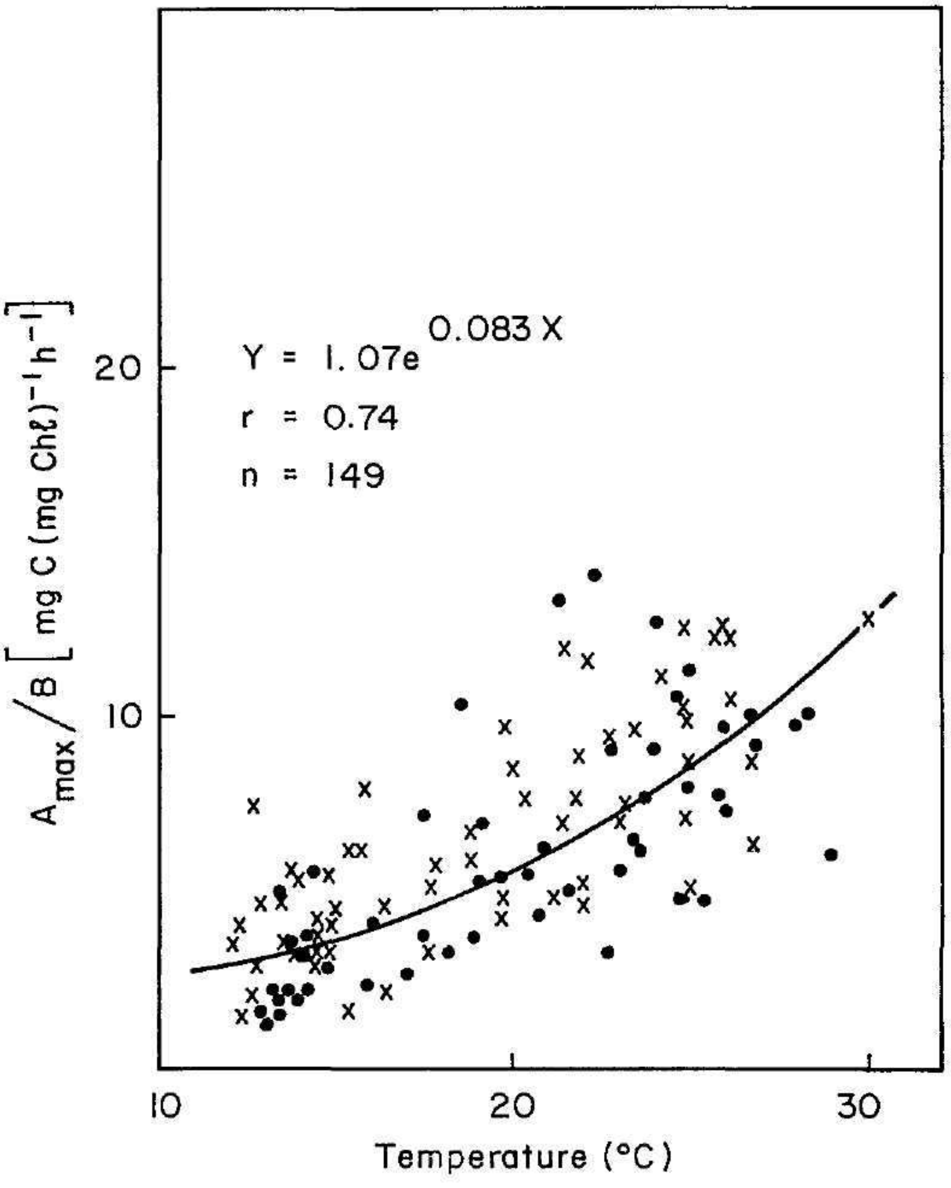


Figure 5.64. The relationship between photosynthetic capacity ($A_{max}/B$) and water temperature for Hartbeespoort Dam phytoplankton. Chlorophyll *a* was extracted using acetone in 1981/82 and ethanol in the following years; 1981/82 data (●), 1982/84 data (x).

NH^-N was used in the analysis because *Miorooystis aeruginosa* has been shown to be able to use only this form of nitrogen (Kappers 1980) although this conclusion may have been due to her experimental procedure. The lack of correlation between primary production and the major growth nutrients in a hypertrophic system such as Hartbeespoort Dam was not surprising. The data show that nitrogen and phosphorus occur in concentrations in excess of algal requirements (see also Section 5.4.5).

In summary, the factors permitting and controlling primary production in Hartbeespoort Dam could be subjectively separated into two categories. In Category 1, nutrients (N + P) , which were present in abundance, permitted large standing crops of *Microcystis aeruginosa* to develop. Wind patterns determined the dramatic spatial and temporal changes in algal standing crop. In Category 2 were the factors which affected the process rates. The buoyancy mechanism and the changes in colony size of *Microoystis* usually enabled the alga to remain in the euphotic zone. With increasing phytoplankton vertical stratification, A was increasingly important in integral production. The saturation parameter I, and photosynthetic capacity were temperature dependent. Physical factors were therefore dominant in regulating phytoplankton production in Hartbeespoort Dam.

Robarts (1984) has shown that water column stability was significantly related to primary production because stability increased with increasing epilimnetic temperature and increasing stability was indicative of less wind-induced mixing. This meant that the phytoplankton were less likely to be removed from the central basin of the lake and that the buoyancy mechanism of *Mieroeyst'is* would be effective in keeping the population in the euphotic zone.

### 5.4.11 Respiration

Respiration by the planktonic (bacteria, phytoplankton, zooplankton) community (= community respiration) is an important loss to algal gross primary production. Several workers (e.g. Dring & Jewson 1982; Williams, Heinmann, Marra & Purdie 1983) have recently demonstrated that the $^{14}$C uptake method for measuring primary production, gave estimates of gross production when short-term incubations (2 - 4 h) were employed. As this was the method used to measure primary productivity in Hartbeespoort Dam quantification of respiratory losses was essential for estimating net production and for simulating phosphorus flow from the primary to the secondary producers.

The changes with time and depth of community respiration rates (R , mg $0_2$ m $^3$ h ) in the upper 10 m of Hartbeespoort Dam during 1982/83 are shown in Figure 5.65A. R averaged 56 mg $0_2$ m $^3$ h , and reached a maximum of 810 mg $0_2$ m $^3$ h on 11 January 1983, at 1 m, the depth of maximum chlorophyll concentration (435 mg m $^3$). Community respiration rates were not detectable during late winter and spring, when chlorophyll concentrations and water temperatures were lowest and when the water column was undersaturated with oxygen (see also Tschumi, Zbaren & Zbaren 1977).

Community respiration rates in the upper 6 m were largely dependent on algal biomass, expressed as chlorophyll (r = 0.86, n - 141, P = 0.001). At 8 and 10 m, where chlorophyll concentration was

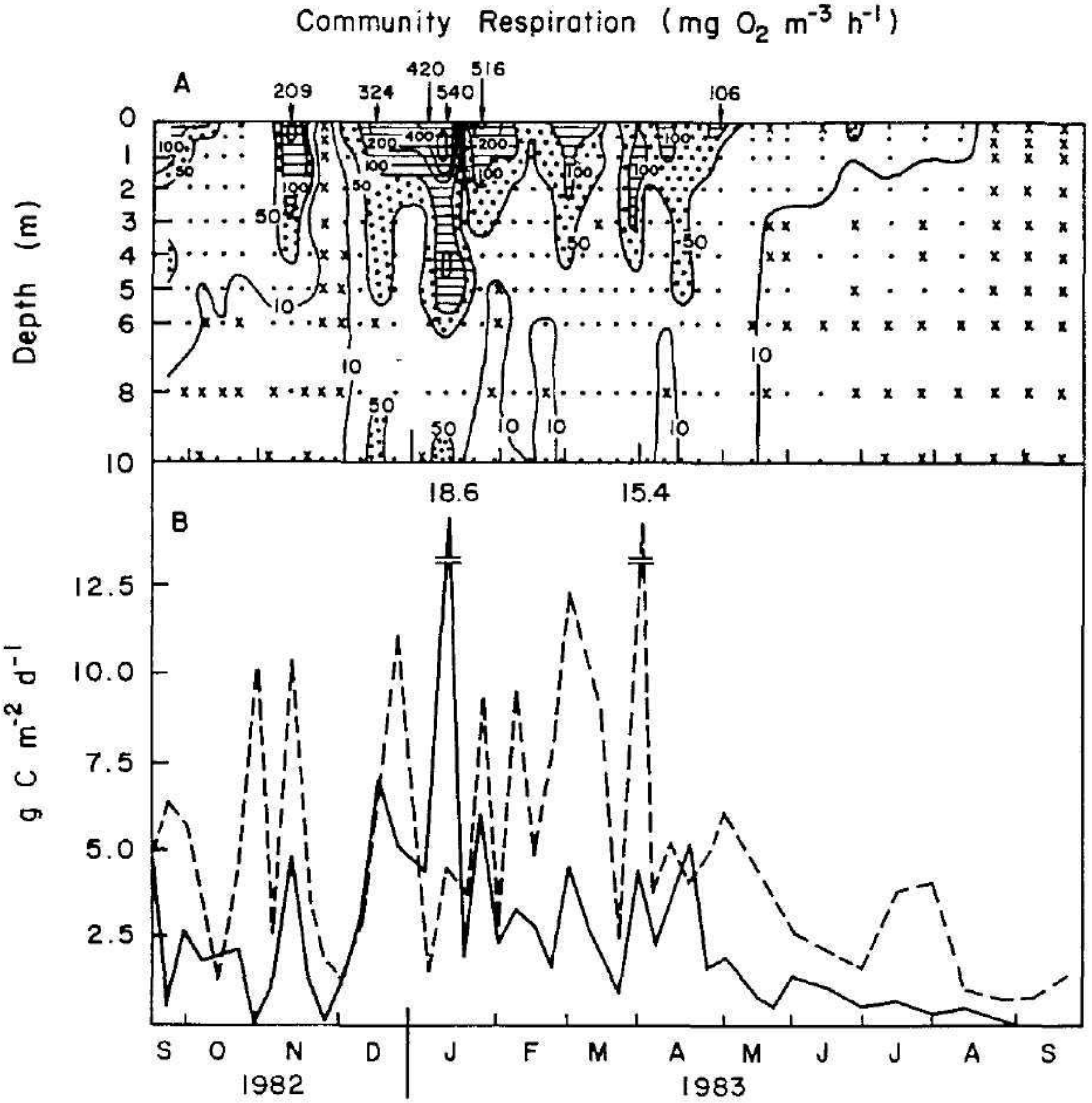


Figure 5.65. Community respiration rates (R$_C$ora^ *-$^n$ Hartbeespoort Dam, 1982/83.
A. changes with time and depth of $R_{com}$ in the upper 10 m stratum.
Areas where $R_{com}$ exceeds 50 mg $0_2$ m~3 h~* are shaded for convenience. Crosses represent respiration values that were calculated to be negative.
B. Seasonal changes in daily areal rates of community respiration, $^{2R}$com ($^{s\circ}$li$^d$ line) and gross production rates (broken line).
j$R_{cora}$ values were converted from oxygen to carbon units assuming RQ= 1.00. (7ohary, Pais Madeira & Breen, in prep.)

usually less than 20 tng m $^{3}$ (Fig. 5.58), the relationship between R and chlorophyll was not significant, indicating that planktonic organisms other than algae were the major oxygen consumers at these deeper depths.

Plankton respiration rates are commonly expressed as specific rates per unit algal biomass (K , mg $0_2$ (mg Chi) * h $^{1}$), assuming that phytoplankton are the major oxygen consumers. In the upper_6 m of the water column R ranged from 0.1 to 2.9 mg $0_2$ (mg Chi) * h *. This range is in agreement with other reported values (Table 5.23). Since both maximum and minimum values were measured when *M, aerugi--nosa* was dominant, this range can be regarded as specific for the natural populations of this species in Hartbeespoort Dam.

The seasonal variations in daily areal rates of community respiration and gross primary production are shown_ in Figure 5.65B. Measured hourly respiration rates (mg $0_2$ m $^{3}$ h *) were integrated over the upper 10 m of the water column by planimetry and multiplied ky. 24_h and by 0.375 to give areal daily rates in carbon units (mg C m $^{2}$ h $^{1}$), permitting their comparison with daily gross production rates (ZZA, Section 5.4.10). On average, respiration processes in the upper 10 m of the water column consumed approximately 50% of the oxygen produced by photosynthesis. Respiration exceeded production significantly only on two occasions in January 1983 (Fig. 5.65B). No significant decline in oxygen saturation was detected on those occasions. Net production (gross production minus respiration losses) of 50% of gross was high when compared with estimates of net production in other highly eutrophic systems, e.g. 20% of gross for net phytoplankton production in Lake George (Ganf 1972), 15% of gross in Lough Neagh (Jewson 1976) and 40% of gross in Kinnego Bay (Jones 1977). The high net production rates in Hartbeespoort Dam allowed maintenance of the large phytoplankton standing stocks.

According to Barica (1981) unstable oxygen regimes are typical of hypertrophic lakes, where periods of oxygen supersaturation due to dense algal blooms may be followed by periods of oxygen depletion or even complete anoxia when the blooms crash. Rapid bacterial decomposition of the dead algal cells exert a huge oxygen demand and, depending on the degree of deoxygenation, severe fish kills may follow. Hartbeespoort Dam though hypertrophic and characterised by dense blue-green algal blooms and large and active bacterial populations (Sections 5.4.2 and 5.4.7) does not undergo such catastrophies. During a full annual cycle the biological oxygen demand in the aerobic epilimnion rarely exceeded photosynthetic oxygen production and on the average consumed only 50% of gross production. This does not imply low respiration rates. Rates of community respiration in Hartbeespoort Dam were higher than values reported from other Jrreshwater bodies (Table 5.23). The maximal rate, 809 mg $0_2$ m $^{3}$ h $^{l}_{t}$ is slightly higher than rates measured in Lake George, Uganda, 4-fold higher than the maximal community respiration rate reported from the highly eutrophic Kinnego Bay and is up to 20 fold higher than maximal rates reported from other fresh water systems (Table 5.23). Maximal water column respiration in Hartbeespoort Dam (2R = 2 060 mg $0_2$ m $^{2}$ h $^{1}$) was nearly double that measured during a ni>loom crash in a hypertrophic Canadian pothole lake ( R = 1 080 mg $0_2$ m"$^{2}$ h"$^{1}$) (Barica 1975). But in Hartbeespoort Dam the enhanced rates were due to the dense, active algal populations (chlorophyll content of the water column was 1 810 mg Chi m $^{2}$), with no evidence for concomitant or subsequent bloom collapse

Table 5.23. Reported maximal values of community respiration ($R_{com}$) and ranges of specific respiration rates ($R_{est}$) measured in field studies of Hartbeespoort Dam (Zohary, Pais Madeira & Breen, in prep.)

| Lake | $R_{com}$ mg $O_2$ $m^{-3}$ $h^{-1}$ | $\Sigma R_{com}$ mg $O_2$ $m^{-2}$ $h^{-1}$ | $R_{est}$ mg $O_2$ (mg Chl)$^{-1}$ $h^{-1}$ | Reference |
|---|---|---|---|---|
| Lough Neagh, N. Ireland | 76 | - | 0.1-2.9 | Gibson 1975 |
| Lough Neagh, N. Ireland | 45 | - | 0.3-1.1 | Jewson 1976 |
| Kinnego Bay, N. Ireland | 200 | - | 0.2-2.7 | Jones 1977 |
| Loch Leven Scotland | - | - | 0.1-3.9 | Bindloss 1974 |
| Lake Horkkajärvi, Finland | $53^a$ | - | - | Salonen *et al.* 1983 |
| Lake Nimetön Finland | $36^a$ | - | - | Salonen *et al.* 1983 |
| Little Lake Conway Florida | - | 346 | - | Fontaine and Ewel 1981 |
| Lake George, Uganda | 544 | - | 0.2-4.5 | Ganf 1972, 1974b<br>Ganf and Horne 1975 |
| Hartbeespoort Dam, South Africa | 810 | 2060 | $0.1-2.9^b$ | This study |
| Canadian Pothole lake | - | $1080^c$ | - | Barica 1975 |

a. Converted from carbon to oxygen units assuming RQ = 1.00
b. Range for 0-6 m depth only
c. Measured during a bloom collapse

and algal death. Yet oxygen demand that does not originate from the respiratory requirements of the epilimnetic plankton may cause complete anoxia in Hartbeespoort Dam. During the 1981 overturn the entire lake became virtually anaerobic due to the dispersion of hypolimnetic oxygen demand throughout the water column (Robarts *et al.* 1982).

## 5.4.12 Algal sedimentation

Losses of viable algal cells from the euphotic zone by sinking can have a substantial impact upon the dynamics of phytoplankton populations (Reynolds & Wiseman 1982).

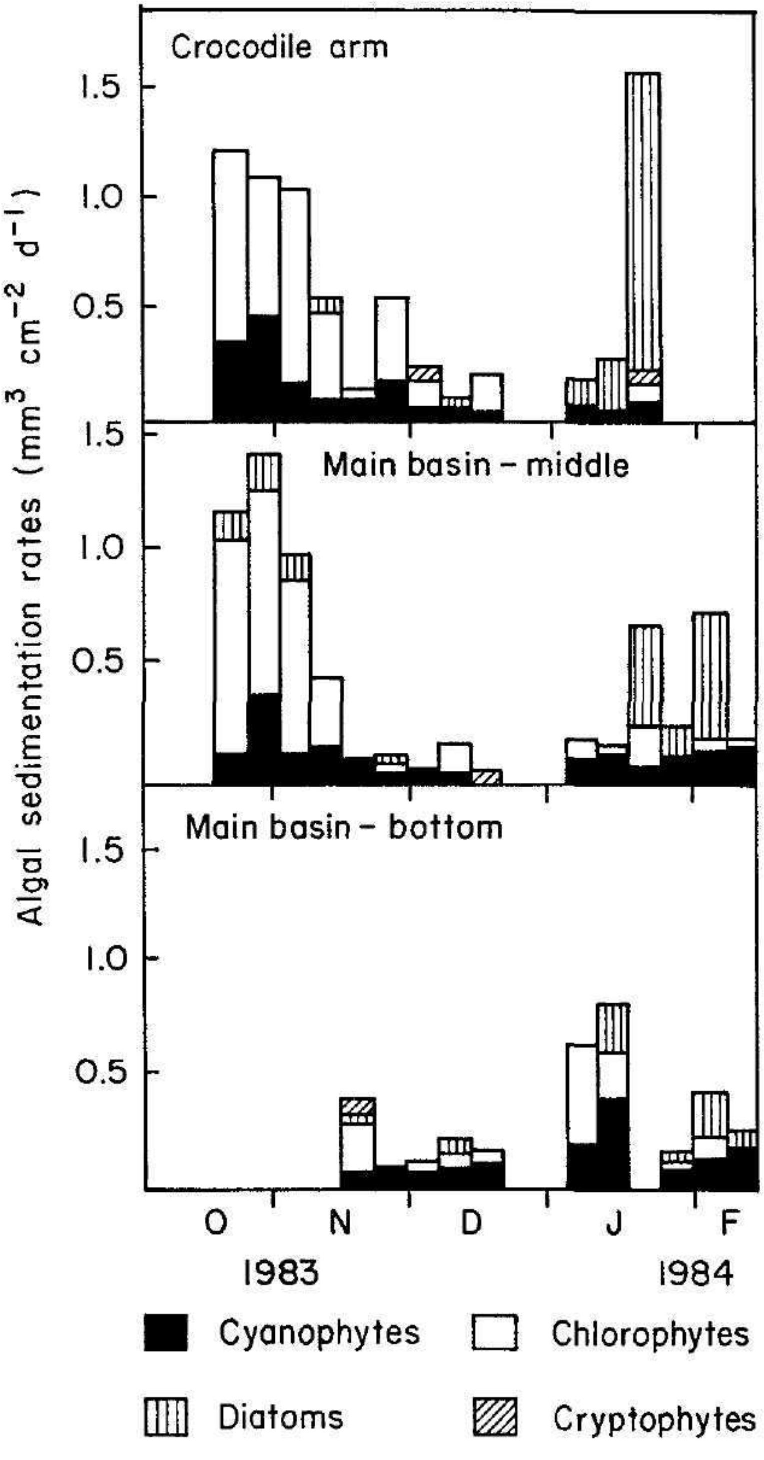


Figure 5.66. Temporal changes in algal sedimentation rates at the Crocodile arm (Station 5, Fig. 4.1) and at the main basin (station 1, Fig. 4.1; 'middle'-traps 8 m above bottom mud; 'bottom'-traps 1 m above bottom mud). The width of the histogrammes corresponds with the duration of exposure of the sediment traps and their heights - to the volume of algal material sedimented. Shadings indicate the proportion of the total sedimented volume contributed by each of the different algal classes. Periods without histogrammes indicate missing data.

Temporal changes in algal sedimentation rates at the main basin and at the Crocodile Station of Hartbeespoort Dam (Station 3, Fig. 4.1) between October 1983 and February 1984 are shown in Figure 5.66. Algal sedimentation rates ranged from 0.04 to 1.59 $mm^3$ cm $^2$ d *. The higher sedimentation rates were usually due to sinking of chlorophytes (green algae) and diatoms (Fig. 5.66).

Mean algal sedimentation rate from all available data was 0.45 $mm^3$ cm $^a$ d $^l$. This value corresponds with the deposition of a 0.17 cm thick layer of viable algal cells per annum. This estimate may, however, be low as higher sedimentation rates are expected in winter and autumn when *Miarocystis* migrates to the sediments (Reynolds *et at.* 1981).

Sinking losses generally followed the temporal pattern of phytoplankton abundance in_the water column (Figs. 5.57, 5.66). Sedimentation rates ($mm^3$ cm $^2$ d $^l$) at the raft station were linearly related (r = 0.80, n = 15, P = 0.01) to the chlorophyll content (mg m $^2$) of the top 15 m of the overlying water column.

In general algae were trapped only during or immediately after the periods of their seasonal abundance, implying that phases of resuspension of previously sedimented algae were not observed. Highest sinking losses followed the population maxima of each species, except for cryptophytes that were rarely trapped. Similarly, Reynolds & Wiseman (1982) reported that cryptophytes were rarely caught in their sediment traps. It is likely that this group is heavily grazed and/or decomposes faster than other groups while still in the water column.

Sinking losses of the larger algal species (e.g. *Pediastnon)* and heavier diatoms constituted a major loss process to their populations, but were less important or negligible in the population dynamics of small heavily grazed species *(Ooaystis*, Cryptophytes). Highest sinking losses of *Mioroeystis* (up to 2 x $10^7$ cells cm $^2$ d $^J$) were encountered during the period of decline of the planktonic population which also coincided with the time of hyperscum break up. Lowest sinking losses (down to 6 x $10^5$ cells cm $^2$ d *) occurred during the phase of stable water column populations.

The high sinking losses of chlorophytes and diatoms in comparison to *M, aeruginosa* most probably contributed to the dominance of the latter.

### 5.4.13 *Mievocystis* toxicity

Previous research, using mouse tests, has demonstrated that toxicity of *M, aeruginosa* in Hartbeespoort Dam is always associated with the morphological colony type *M. aeruginosa* forma *aeruginosa* and not with the morphological type *M. aeruginosa* forma *flos-aquae* (Scott *et at.* 1981).

Figure 5.67 gives the percentage composition of *M. aeruginosa* colony type and the total *Microaystis* volume at the raft station from November 1982 to June 1984. The total *Miorooystis* volume showed large fluctuations with no apparent seasonal trend and ranged from 0.7 $mm^3$ $IT^1$ in November 1982 to 268 $mm^3$ $S\sim^l$ in November 1983. *Miovocystis* was never totally absent from the sampling station over the study period.

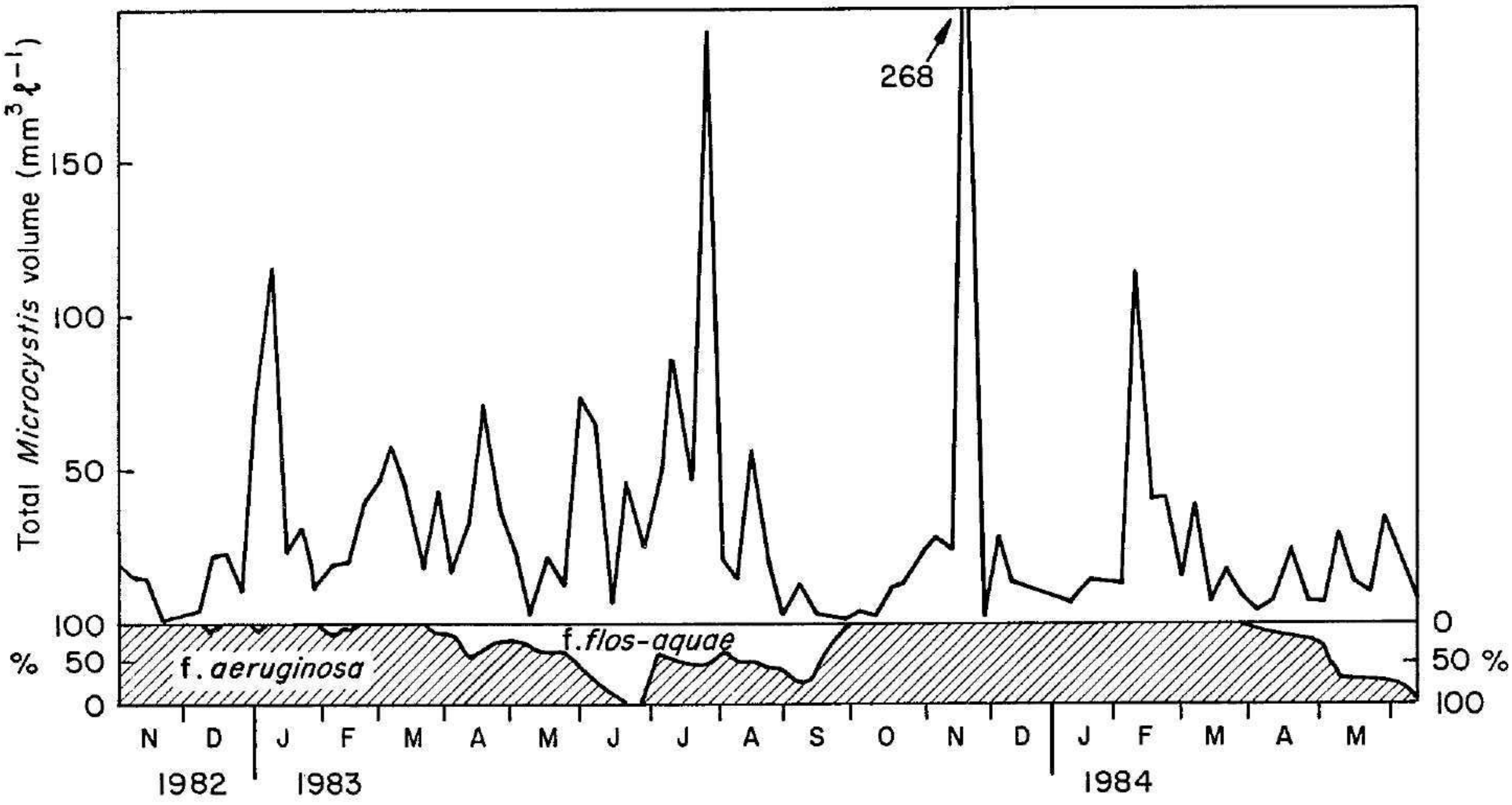


Figure 5.67. Absolute amounts of *Microcystis aeruginosa* at Station 1, Hartbeespoort Dam from November 1982 to June 1984. The bottom part of the figure illustrates the relative abundance of the two forms of *M. aeruginosa* viz. forma *aeruginosa* (= *M. toxica*) and forma *flos-aquae*.

The lower portion of Figure 5.67 shows that *M. aeruginosa* forma *aeruginosa>* dominated the *Mierocystis* component of the phytoplankton. The forma *aeruginosa* has been judged by Komarek (1958) to be the same species as *M. toxica* Stephens which was originally described from Vaal Dam. For 11 of the 20 months of observation this was the only form of *Mierocystis* present in the lake. The non-toxic forma *flos-aquae* only formed a significant proportion of the *Mierocystis* population during the colder months of the year. For short periods in June 1983 and June 1984 forma *flos-aquae* was the only form of *M. aeruginosa* present at the sampling station.

*Mierocystis* toxins were invariably present in all regularly collected samples supplied to the National Chemical Research Laboratory for chemical structure work over the study period. Direct confirmation of toxicity with mouse tests was not carried out on a routine basis during the present study but from current and earlier observations it can be accepted that toxic *Mierocystis* will be present in Hartbeespoort Dam for at least 80% of the time in any one year.

*Miorooystis* is recognized as an important eutrophication problem and is receiving attention in other NIWR projects.

### 5.4.14 Algal competition and succession

Paerl & Ustach (1982) suggested that the buoyancy mechanism of blue-green algae imparts an ecological advantage by ensuring access to $C0_2$ during periods of high photosynthetic activity and high water pH. The formation of large surface populations, they noted, would also shade underlying phytoplankton restricting their photosynthetic rates and competitive potential. If *Microoystis aeruginosa* can utilize $HC0_3$ the photosynthetic production of this alga in Hartbeespoort Dam would not be limited by inorganic carbon since sufficient quantities exist (Robarts 1984).

In a study of *Mierocystis aeruginosa* dynamics in an Australian impoundment Humphries & Imberger (1982) showed that *Mieroaystis aeruginosa* grew successfully because even slight positive buoyancy of cells led to a substantial growth advantage over negatively buoyant species in the water body. This growth advantage stems from two consequences of positive buoyancy. The first is that the population increases toward the surface. The second is that the algae could maximize the photic depth to mixed layer depth (epilimnion) quotient (z /z ). Over a diurnal cycle a number of shallow mixed layers woula be formed within the epilimnion as the surface water heated and cooled. The depth to which the epilimnion extended would be the maximum depth to which the water had been recently mixed. The algal buoyancy mechanism would ensure that the cells remained in the shallower diurnally mixed layers, except during wind activity strong enough to destroy these shallower layers. In this way z in the quotient would be reduced and the time the algae spent out or the euphotic zone would be reduced. In contrast, Humphries & Imberger (1982) noted that the vertical distribution of negatively buoyant cells is more even with depth within the epilimnion. The mixed layer for these cells is then the most recent, and deepest, depth to which the water was mixed. In comparison to the z experienced by positively buoyant cells, z for negatively buoyant cells would be large thereby reducing the quotient z /z and increasing the time the cells spent in the aphotic zone. u

Reynolds & Walsby (1975) noted that the growth of bloom-forming species, such as *Mierocystis aeruginosa,* seemed to be confined to lakes in which average values of z /z were within the range 0.5 to 3.5. The depth of the Hartbeespoort Dam epilimnion was taken as that depth at which the oxygen concentration was zero and was shallowest in December 1982 when it reached a depth of 12 m. Throughout the study the epilimnion was always greater than the euphotic zone. The quotient, z /z , ranged between 0.04 and 0.65 (x = 0.19). On the basis of Reynolds & Walsby's (1975) observations *Microeystis aeruginosa* should not have been the dominant alga in Hartbeespoort Dam. But, as Humphries & Imberger (1982) have noted, this quotient can be altered by algae with buoyancy mechanisms. As the bulk of the algal population in Hartbeespoort Dam was usually contained in the upper 4 m (Fig. 5.58) the quotient calculated using z did not reflect the actual proportion of photic to aphotic zones experienced by *Microeystis aeruginosa.* As *Microeystis aeruginosa* accumulated at the surface the concomitant increase in colony size would moderate light attenuation thereby reducing population self-shading. The double ecological advantage of buoyancy mechanism and large colony size makes *Microoystis aevuginosa* a formidable and successful competitor in lakes where N & P limitation of growth rate is non-existent, or rare, and where wind-induced mixing is low.

Storms do occur at Hartbeespoort Dam and cause deepening of the mixed layer and reduction in *Mierocystis aevuginosa* colony size. During periods of increased mixing large organism size is a disadvantage since larger particles for a given chlorophyll concentration will capture less light than an equivalent volume of smaller particles (Kirk 1976). Smaller colony size results in a decrease in the euphotic zone depth (Fig. 5.61). This disadvantage may be offset by the increased rate of circulation through the mixed layer and by several possible physiological adaptations of the algae (Perry, Talbot & Alberte 1981). Storms are generally of short duration and *Microeystis aeruginosa* appears to be able to remain dominant.

However, the succession in Hartbeespoort Dam of *Microeystis aeruginosa* to *Melosira granulata* followed by *Oocystis* with increased wind activity seems to indicate that *Microeystis aeruginosa* cannot compete in a fluctuating light environment for long periods. Walsh & Legendre (1982) and Perry, Talbot & Alberte (1981) have shown that *Oocystis* and diatoms, respectively, are adapted to living in the daily, widely fluctuating light environments such as would occur in the epilimnion of a lake.

In conclusion, *Microeystis aeruginosa* was the dominant primary producer in Hartbeespoort Dam because it was suitably adapted to living in an environment in which N & P were in excess of algal requirements and where a major factor limiting phytoplankton growth rate was light. The buoyancy mechanism of *Mierocystis aeruginosa* and the changes in its colony size allowed the alga to exploit and moderate the underwater light climate to its advantage, thus effectively excluding other species for most of the year in this hypertrophic lake subject to low daily wind speeds. Van Rijn & Shilo (1983) concluded from their studies of *Osaillatoria* that when several factors are operative in regulating the buoyancy mechanism of blue-green algae, the presence or absence of light is dominant over the effect of $C0_2$ or nutrient availability.

### 5.4.15 Zooplankton

#### (a) Seasonal abundance, composition and production

Zooplankton may be broadly divided into two trophic levels, herbivores (usually filter-feeders) and carnivores (raptorial), both of which are important in the diet of zooplanktivorous fish. The zooplankton community of Hartbeespoort Dam is composed primarily of cladocerans, copepods, rotifers and midge larvae. Of these groups the herbivorous cladocerans frequently contribute most to total zooplankton standing stock. Rotifers, although often numerically abundant due to their small size do not significantly influence total standing stock. The two copepod species present are the herbivorous calanoid copepod *Thewnodiaptomus syngenes*, and the carnivorous cyclopoid copepod *Thermocyclops oblongatus* which is present throughout each year. Another zooplankton carnivore is the phantom midge larva *Chaobovus* sp. Highest numbers of this predator occur during the summer when, due to their ability to temporarily inhabit anoxic water, *Chaobovus* larvae are usually found in deep water or within the hypolimnion, and thus avoid high fish predation pressure.

Mean annual whole lake and volumetric estimates of zooplankton standing stocks and production are shown in Table 5.24. Secondary production by zooplankton was calculated using the mean annual standing stock values from Table 5.24 and a production/ biomass turnover coefficient estimated from Waters (1977). In modelling zooplankton standing stock, conversion from dry weight to tonnes of phosphorus was carried out using a coefficient of 0.015 (Zison, Mills, Deimer & Chen 1978). For conversion from dry to fresh weight a coefficient of 6 was used (Waters 1977). Both total annual primary (Section 5.4.10) and secondary production per cubic metre have increased over the three year study period. However, whole lake secondary production and biomass were lowest in 1983 due to the low lake volume during the prevailing drought conditions.

The herbivorous cladocerans *Daphnia pulex* and *D. longi-spina** due to their large size (adults = 2.5 mm) and frequently high num-

**Table 5.24.** **Zooplankton biomass and production estimates for Hartbeespoort Dam. (All values in dry weight).**

| | 1981 | 1982 | 1983 |
|---|---|---|---|
| Mean annual standing stock (g $m^{-3}$) | 0.181 | 0.273 | 0.405 |
| Whole lake mean annual standing stock (tonnes) | 31.7 | 38.9 | 24.9 |
| Secondary production (g $m^{-3}$ $y^{-1}$) | 4.5 | 6.8 | 10.1 |
| Whole lake secondary production (tonnes $y^{-1}$) | 837.2 | 1023.0 | 677.8 |

bers, contribute most to zooplankton standing stocks and high grazing rates from autumn through to mid-summer. The *Daphnia* population has a seasonal pattern of high densities in spring followed by a rapid decline from January to April when *Ceviodaphnia retieulata* becomes the most abundant organism (Fig. 5.68). This trend, which was also reported in Hartbeespoort Dam by Seaman (1977), occurred in 1981, 1982 and 1984. Atypically, during the drought year of 1983, the *Daphnia* population declined in January but recovered in February before declining again when *Ceriodaphnia* re-appeared in extremely high numbers during March, During the periods of co-existence of *Daphnia* and *Ceriodaphnia* in February, July, August, November and December 1983 a marked spatial separation was evident between their population maxima (Fig. 5.69), and daytime maximum numbers of *Ceviodaphnia* were recorded deep in the water column, or near the bottom when the lake was unstratified, below the population maxima of *Daphnia.* This vertical separation of these cladocerans, also noted in Hartbeespoort Dam by Connell (1978), may indicate that these herbivores are not competing for the same food resource.

Species succession is evident within the genus *Daphnia.* Separation of *D. putex* and *D. longispina* is not easily carried out routinely. However, periodic close examination of adult *Daphnia* showed that after its absence from January to April of 1981 and 1982, the *Daphnia* population which reappeared consisted of *D. longispina. D. pulex* appeared approximately two months later when both species co-existed, after which *D. pulex* became more abundant and *D. longispina* numbers declined. However, following the persistance of *D. pulex* through the summer of 1982/83, *D. longispina* did not occur during autumn 1983.

The spring increase in *Daphnia* numbers and the brief occurrence of *Bosmina longivostvis* (Fig. 5.68) are both associated with a period, from September to November, of low phytoplankton volume but high non-blue-green algal species diversity and abundance (Section 5.4.6). Increasing water temperature in spring and more abundant edible cryptophytes and green algal food results in a high zooplankton biomass, and consequently an unusual negative association exists between zooplankton biomass and chlorophyll *a* concentration in Hartbeespoort Dam (1981 r - 0.53; 1982 no correlation; 1983 r = 0.65). Highest phytoplankton volumes occur when *Micvocystis* dominates (frequently over 90 % of phytoplankton volume, Section 5.4.6). Formation of large colonies of this blue-green alga reduces zooplankton grazing pressure.

The competitive interaction between *Daphnia* and *Bosmina* has been studied by De Mott (1982), De Mott & Kerfoot (1982) and Goulden, Henry & Tessier (1982). *Bosmina* was shown to feed more efficiently on the small flagellate alga *Chlamydomonas* than *Daphnia* which in turn was better able to feed on bacteria than *Bosmina.* These differences in feeding efficiency on various foods may explain the co-existance of these cladocerans during the brief periods of non-blue-green algal abundance. The reason for the failure of *Bosmina* to co-exist with *Daphnia* during spring 1983 is unclear. Factors other than direct competition for the same food resource must play a role.

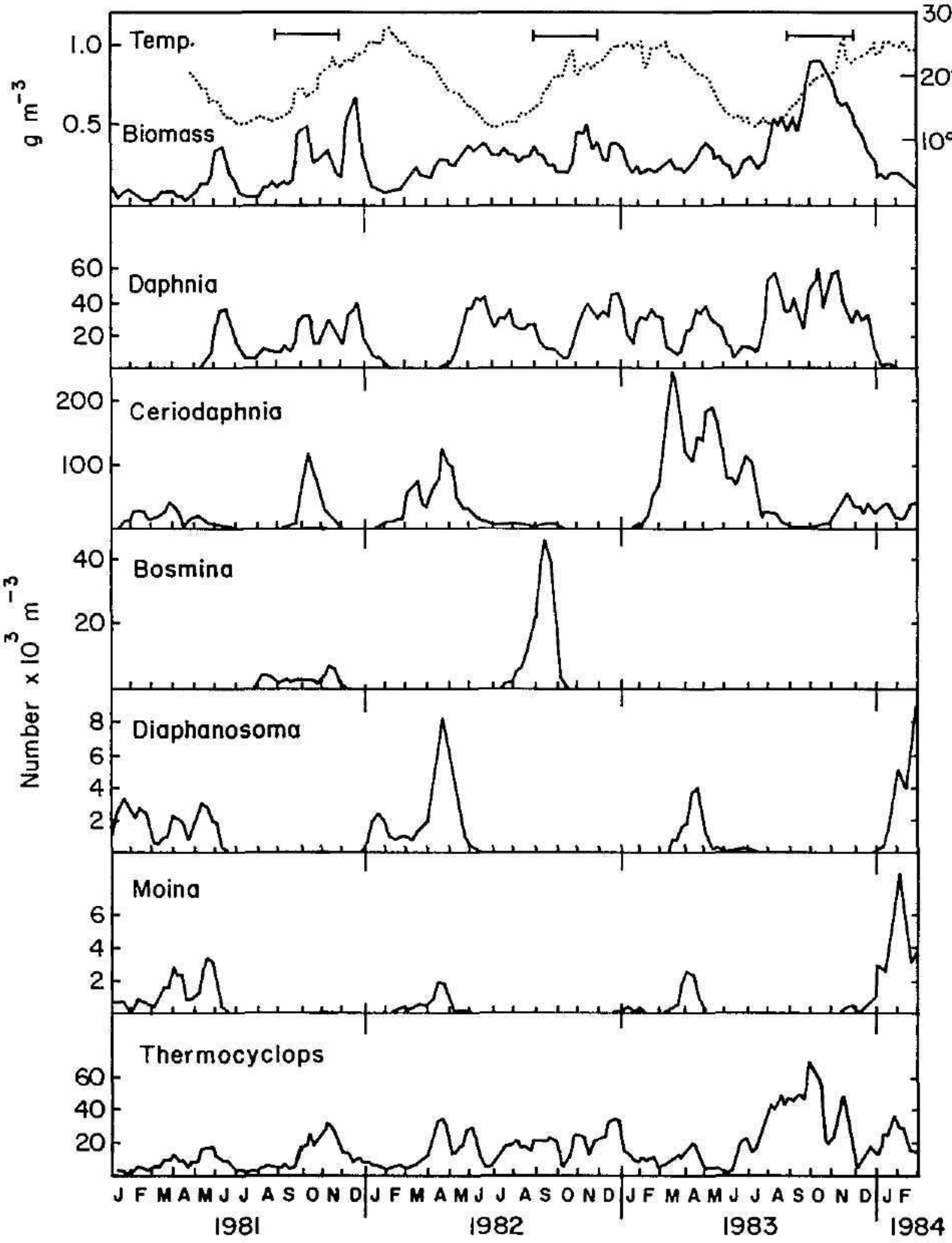


Figure 5.68. Total biomass, temperature and numbers of the major zooplankton genera. Horizontal bars indicate periods of abundance of non-blue-green algae.

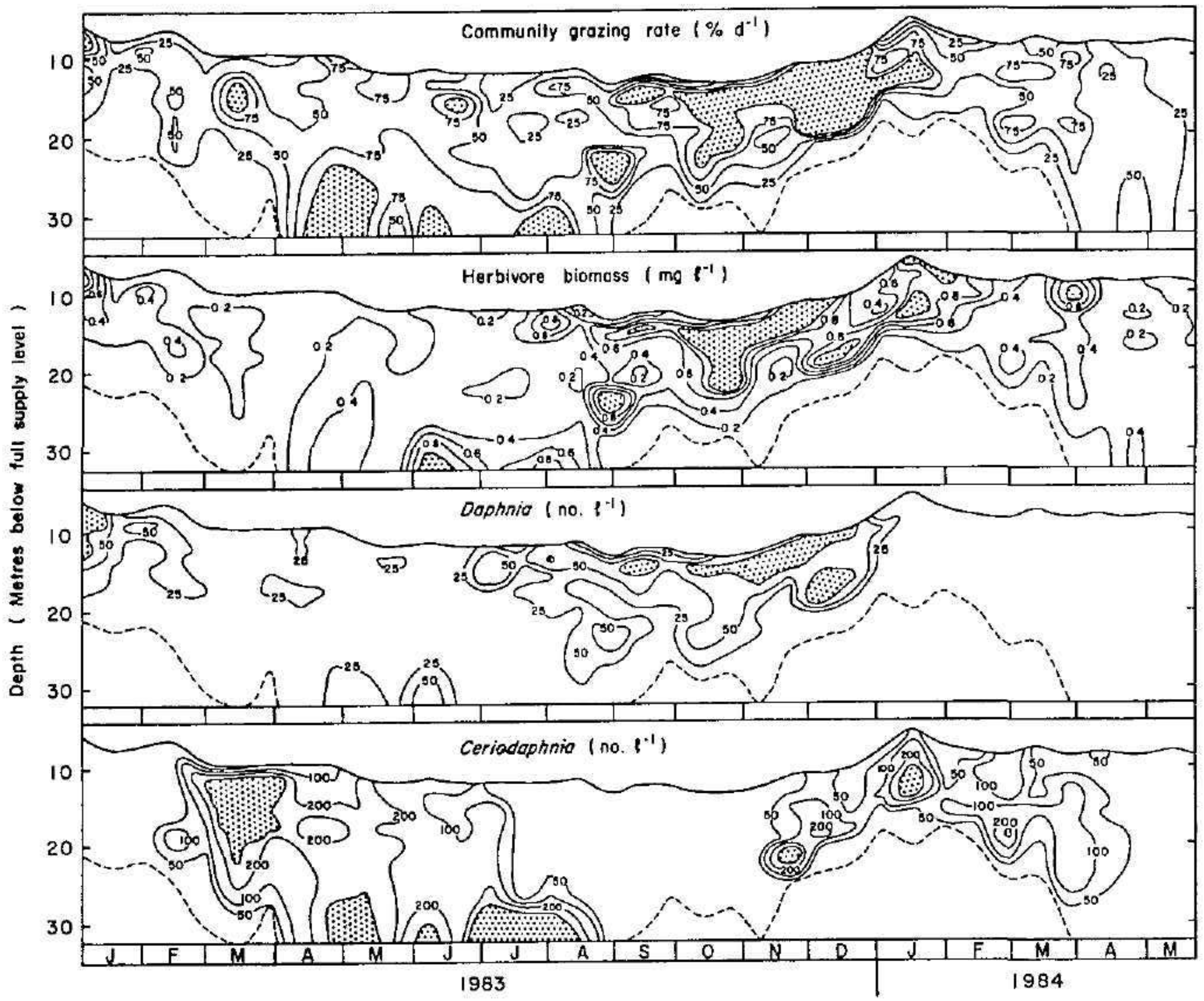


Figure 5.69. Isopleths of zooplankton community grazing rate, herbivore biomass, *Daphnia* and *Ceriodaphnia* density throughout the water column at station 1. (Stippled area = grazing rate > 100% d$^{-1}$; biomass > 1.0 mg ℓ$^{-1}$; *Daphnia* > 100 ℓ$^{-1}$; *Ceriodaphnia* >300 ℓ$^{-1}$. Oxycline shown by broken line).

The spring increase in zooplankton biomass, primarily due to rapid growth of *Daphnia,* provides the fish fry of Hartbeespoort Dam with an important source of food at the time of rapid growth. Zooplankton, particularly large *Daphnia,* are a major component of the gut content of *Oreochromis mossambious* fry of 1-4 g from December to February (Section 5.4.18).

This period of high *Oreochromis* predation pressure corresponds with a fall from high *Daphnia* numbers in December to low numbers in January and February. The carp *Cyprinus oavpio* also feeds partly on zooplankton when *daphnia* are abundant, usually from September to November. The barbel *Clarias gaviepinus* is an important zooplanktivore. Zooplankton form a major component of the diet of large *Clarices* (>500 g) from July to November, during the period of *Daphnia* dominance. Therefore a combination of high fish predation pressure and a change in phytoplankton species composition, results in the sharp decline of *Daphnia* and total zooplankton biomass in January.

### 0>) Diel vertical migration and grazing rates

The phenomenon of upward migration of zooplankton at night and downward migration during the day has been widely reported in the literature. Evening migration of zooplankton into surface waters richer in phytoplankton, and their return to deeper water during the day to avoid visual detection by predatory fish, is regarded as a primary advantage of diel vertical migration to zooplankton. Connell (1978) reported that this pattern of diel migration was reversed in Hartbeespoort Dam, and downward night-time migratory behaviour was marked in *Daphnia* and *Thermooyolops* populations. However, a diel study carried out in late January 1984 showed that the zooplankton community present exhibited an upward evening migratory pattern. Species composition during this study differed from that found by Connell (1978). The most abundant herbivorous crustaean present in January 1984 was *Ceviodaphnia.* This cladoceran clearly exhibited upward migration during the night, whereas maximum numbers of *Ceriodaphnia* during the day occurred below the euphotic depth. Highest grazing rates during the diel study closely followed the pattern of vertical distribution of maximum *Cerodaphnia* numbers. Comparison between day and night mean profile grazing rates *(%* d *) showed no difference in diel grazing rates (P >0.j.O). Calculations of mean profile specific grazing rates (% d 1 mg *), to eliminate the effect of patchy distribution of zooplankton, also showed no significant difference between day and night biomass specific grazing activity (P >0.20).

### (c) Zooplankton feeding studies

Filter-feeding zooplankton are often non-selective with regard to the type of food ingested. Some zooplankton species may ingest organic and inorganic particles that fall within an acceptable size range, whereas other organisms may also select particles on the basis of palatability (Burns 1968, 1969; Boyd 1976; Lehman 1976; Fernandez 1979). Therefore, suspended particles other than just algal cells are ingested by many zooplankton species. Small algal colonies, individual algal cells, free living bacteria, suspended organic detritus supporting attached bacteria, and even suspended inorganic particles to which organic matter may be adsorbed (Arruda, Marzolf & Faulk 1983), are all components of the diet of filter-feeding zooplankton. Consequently, the terra 'herbivore' is applied loosely to filter-feeding zooplankton to include both algae and other suspended particulates ingested. The green alga *Chlorella* was chosen as the radiolabelled food for feeding experiments on the zooplankton of Hartbeespoort Dam because its small spherical shape (5-12 um diameter), unicellular form, lack of flagella or sheath, and natural occurrence in the phytoplankton, readily allowed ingestion of this palatable alga by herbivorous zooplankton.

The zooplankton community grazing rate (CGR) calculated as *%* d$^{1}$ represents the maximum potential percentage of the surrounding lake water that may be filtered by the zooplankton present. High grazing rates are not recorded throughout the water column and not all food particles encountered may be gathered by the zooplankton. In addition to rejection of some particles, preferential filtration from natural mixtures of food, patchy distribution of phytoplankton and low grazing efficiency of zooplankton on large colonial algae, particularly *M'icroeystiSj* prevent the removal of all food resources during periods of high grazing activity. Therefore, the CGR measured using *ChloreZla* represent 'maximum' grazing rates unlimited by variations in food type and palatability.

Periods and depths of high CGR corresponded with high herbivore biomass shown in Figure 5.69. Zooplankton avoided the region of low dissolved oxygen at the interface between the epilimnion and anoxic hypolimnion. In contrast, the study of Haney (1973) in stratified Heart Lake, Ontario, showed that the zooplankton grazing rate was occasionally highest at the epilimnion-hypolimnion boundary. High numbers of bacteria can occur at this interface which provide the zooplankton with an important food source. In Hartbeespoort Dam, however, the density of epilimnetic bacteria exceeded that of hypolimnetic bacteria (Robarts & Sephton 1984, Section 5.4.2).

A positive linear correlation existed between CGR and herbivore biomass ($r = 0.77$, $n = 247$). The highest grazing rate measured (581.5 % d$^{1}$ at 2 m on 22.11.83) occurred when a dense aggregation of *Daphni-a* was encountered. All other high grazing rates above 200 % d * were associated with high numbers of this large herbivore. The pattern of distribution of high densities of both major herbivores, *Daphnia* and *Ceviodccphna*, corresponded to high CGR (Fig. 5.69). It is also evident that grazing activity was dominated by each of these organisms in turn. *Daphnia* influenced grazing activity from January to mid-February 1983, thereafter *Ceviodaphnia* dominated from mid-February to July, followed again by *Daphnia* from July until December when high numbers of *Ceviodaphnia* returned. Co-dominance occurred briefly at the overlap of these periods. Repetition of this seasonal pattern did not occur during the first quarter of 1984 when temporary co-dominance of the herbivorous cladocerans *Moina* and *Viaphanosoma* occurred with *Ceriodaphnia.*

Seasonal variation in CGR was evident when integrated mean grazing rates were calculated (Fig. 5.70). Integration of grazing rates over the sampled vertical profile, by weighting of rates corresponding to vertical compartmentalization of the lake, resulted in mean grazing rates which incorporated the vertical variations and structure of the zooplankton and their grazing activity. From mid-January to June 1983 integrated CGR varied from 19.8 to 78.5 % d $^{1}$, and was low during periods of both highest temperatures (January-February) and lowest temperatures (July-August). Throughout the spring and early summer of 1983, when *Daphnia* dominated the zooplankton community, integrated CGR$^{1}$s rose steadily reaching a maximum of 260 % d $^{1}$ at 23.1 °C in early December. With the decline in density of *Daphnia* in late December, the integrated CGR dropped sharply. Due to the presence of *Moina* and *Diccphanosoma* the integrated biomass and CGR from January to March 1984 was higher than that of the corresponding period in 1983. Integrated zooplankton standing stocks and CGR's measured in Lake Le Roux in 1982 by Hart *et al.* (1983) were generally high in spring and summer and lowest in late June. Hart showed that zooplankton biomass was positively correlated with water temperature (number of days above 20 °C) and secchi depth. This was not the case in Hartbeespoort Dam where highest grazing rates occurred during the spring period of green algal abundance (lowest chlorophyll *a* concentration) and *Daphnia* dominance.

Changes in the integrated specific grazing rate reveal a different seasonal pattern of grazing activity (Fig. 5.70c). High integrated specific grazing activity (SGA) occurred in summer and autumn, and low SGA was recorded in winter and spring. Any direct effect of temperature on SGA was, however, masked by changes between periods of *Daphnia - Ceriodaphnia* dominance. High SGA occurred when zooplankton biomass was low during summer and the dominant herbivore was the very small cladoceran *Ceriodaphnia.* The metabolic energy demand of this high temperature - small body size population is probably extremely high and consequently SGA is correspondingly high (Lampert 1977). With the fall of mean water temperatures in autumn, reduction of SGA did not occur until July when both SGA and temperature were low. However, due to the appearance of a growing *Daphnia* population during that month, the reduction in SGA cannot be regarded as under the direct influence of temperature alone. Periods of *Daphnia* dominance are characterized by low SGA. The rapid rise of grazing rates in spring was due to a steady increase in the number and biomass of *Daphnia* and not due to an increase in SGA. Only when *Ceriodaphnia* again occurred in December 1983 did SGA rise during the brief summer period of *Daphnia/Ceriodaphnia* co-dominance. However, the period of high SGA from February to June 1983, which was associated with *Ceriodaphnia* dominance, did not occur in 1984. Lower densities of *Ceriodaphnia* than during the previous summer, and the presence of *Moina* and *Diaphanosoma*$_{3}$ resulted in lower SGA typical of periods of *Daphnia* dominance. The seasonal pattern of SGA in Hartbeespoort Dam differs from that recorded in Lake Le Roux by Hart *et at.* (1983). High SGA was recorded from April to November 1982, in Lake Le Roux, with low SGA occuring in October and November 1981, January 1982 and February 1983.

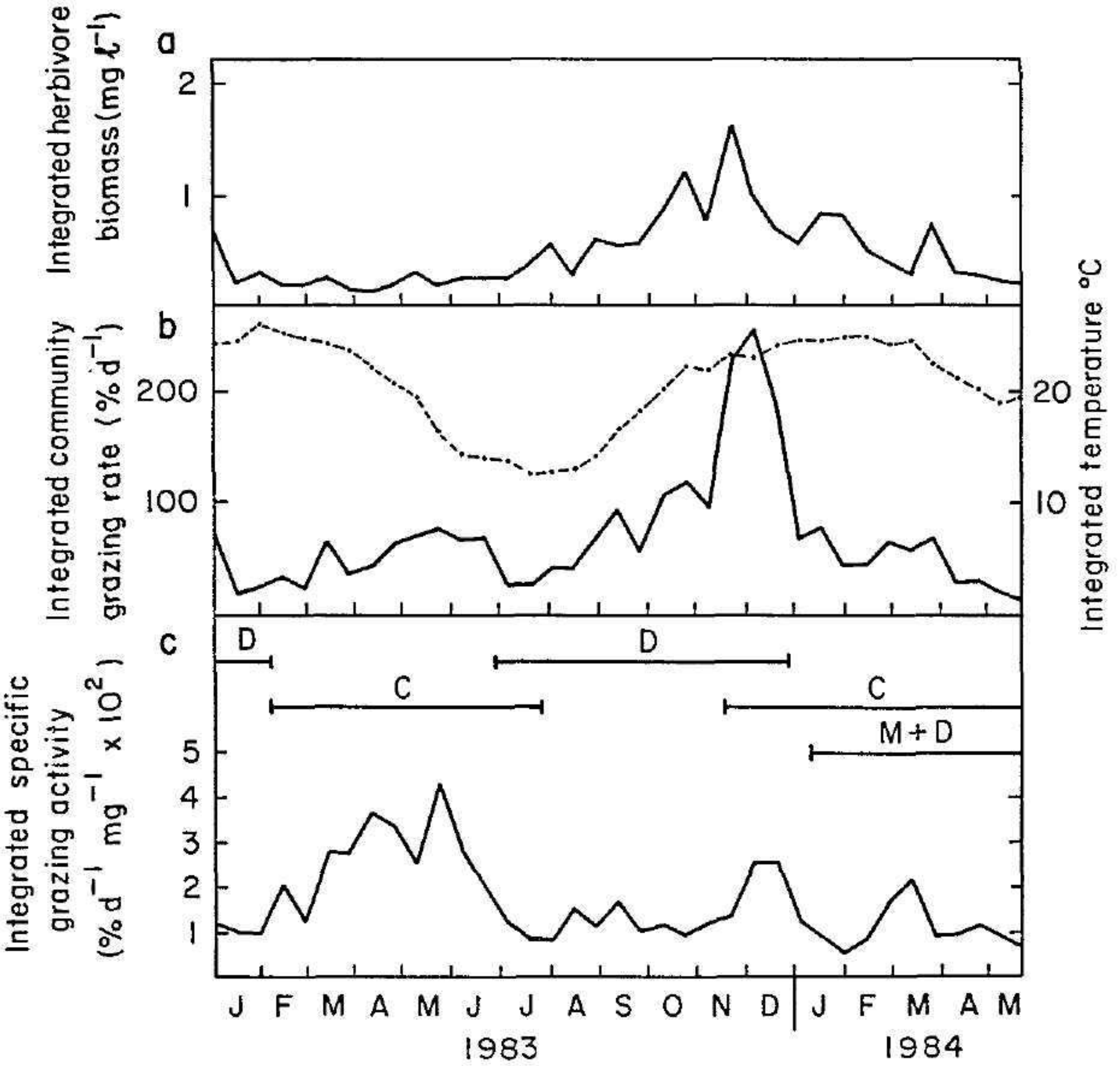


Figure 5.70. Seasonal variation of: (a) integrated herbivore biomass, (b) integrated grazing rate (solid line) and temperature (broken line), (c) integrated specific grazing activity, over total aerobic water column. Periods of dominance and co-dominance of *Daphnia* (D), *Ceriodaphnia* (C), *Moina* and *Diaphanosoma* (M+D) are shown.

The influence of dominant herbivore species on SGA was further demonstrated by the relationship shown in Figure 5.71. Low herbivore biomass, characteristic of periods of *Ceriodaphnia* dominance, corresponded to low CGR but high SGA. Conversely, periods of high herbivore biomass, when *Daphnia* dominated, had high CGR but low SGA.

Simultaneous experiments carried out *in situ* showed no significant difference between the CGR on naturally occurring *Oocystis* or on *Chlorella.* Occasional measurements of the filtration rates of the major cladoceran species were carried out between August 1983 and May 1984. Results of two series of *in situ* experiments, when *Daphnia* was sufficiently numerous to determine its filtration rate on radiolabelled *Chlorella,* showed that the high CGR when *Daphnia* was dominant could be attributed to the high filtration rate of individual large *Daphnia* (Table 5.25). Filtration rates of small *Daphnia* (<1.5 mm), *Moina* and *Diaphanosoma*, which are of a comparable size, were similar. Filtration rates recorded for *Ceriodaphnia* were variable. The filtration rates of *Ceriodaphnia* measured at temperatures above 21 °C were consistantly lower than the rate recorded at 18.7 °C . Due to absence of *Ceriodaphnia* during *in situ* experiments at 14.3 °C no conclusions can be made on the effect of temperature on the filtration rate of *Ceriodaphnia.*

*In situ* experiments conducted using radiolabelled colonies of the blue-green alga *Microcystis* showed that ingestion of small colonies by zooplankton did occur. Feeding rates on *Microcystis* colonies (colony diameters 5-20, 20-40, 40-60, 60-100 and 100-150 ym) compared to rates recorded simultaneously using *Chlovella* (Fig. 5.72) showed that feeding rates of *Ceriodaphnia* and *Diaphanosoma* on *Microcystis* colonies of 5-20 urn were lower than on *Chlovella* cells of a similar size (5-12 pm). Selection of *Microcystis* by *Ceriodaphnia* was limited to small colonies, and a rapid decline in feeding efficiency occurred with increasing colony size. *Diaphanosoma* did not feed efficiently on *Microcystis* colonies of greater than 100 ym diameter but was less limited by increasing colony size than *Ceriodaphnia.* Feeding efficiencies of *Thermodiaptomus syngenes* on *Miorooystis* colonies indicated that this large calanoid copepod was readily able to ingest *Microcystis* colonies of below 60 urn diameter. However, due to the low numbers of *Thermodiaptomus* present, reliable estimates of its size selection and feeding efficien-

**Table 5.25.** **Filtration rates of major cladoceran species in Hartbeespoort (µℓ individual $^{-1}$ h$^{-1}$) using labelled *Chlorella*.**

| Number of experiments | 5 | 3 | 3 | 5 | 2 | 3 |
|---|---|---|---|---|---|---|
| Temperature | 14.3 | 18.7 | 21.4 | 21.5 | 24.4 | 24.4 |
| *Daphnia* > 1.5 mm | 1121 | - | - | 983 | - | - |
| *Daphnia* < 1.5 mm | 186 | - | - | 164 | - | - |
| *Ceriodaphnia* | - | 297 | 111 | 110 | 130 | 121 |
| *Diaphanosoma* | - | 161 | 183 | - | 182 | 166 |
| *Moina* | - | 179 | 234 | - | 178 | 239 |

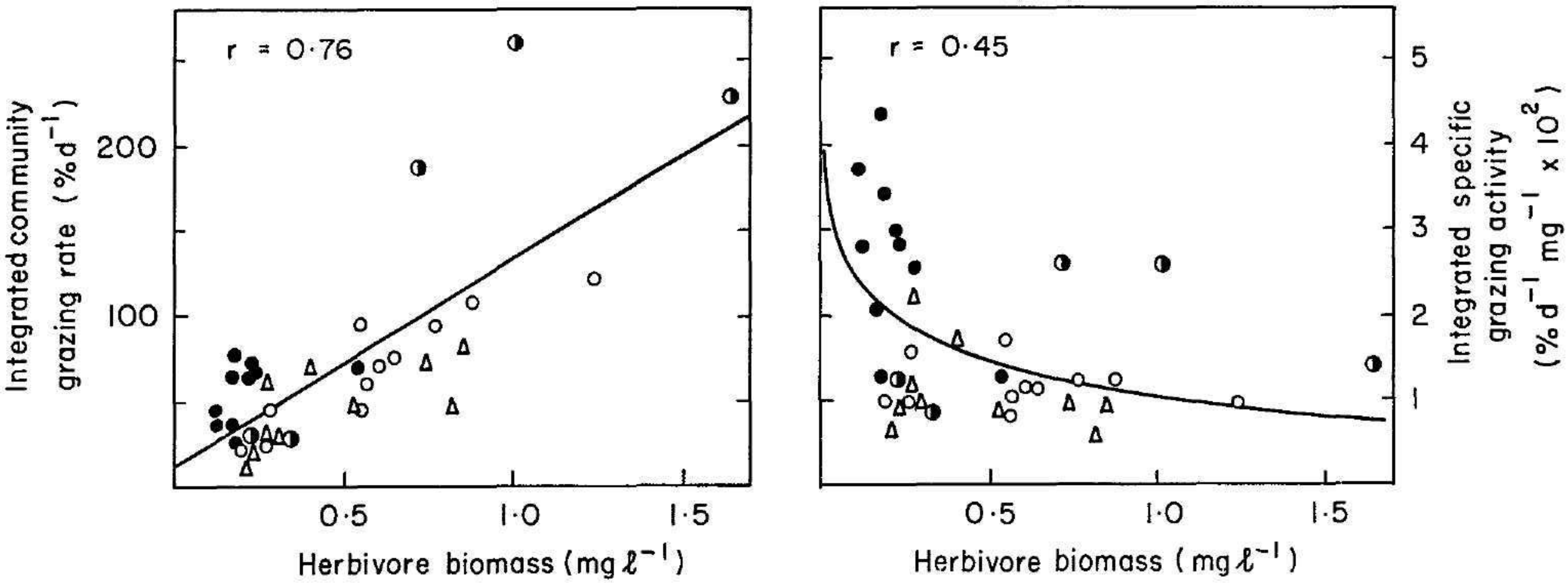


Figure 5.71. Relationships between zooplankton feeding activity and herbivore biomass.

| | |
|---|---|
| O *Daphnia* phase | ● *Ceriodaphnia* phase |
| ◑ *Daphnia* - *Ceriodaphnia* co-dominance phase | Δ *Moina* - *Diaphanosoma* co-dominance phase. |

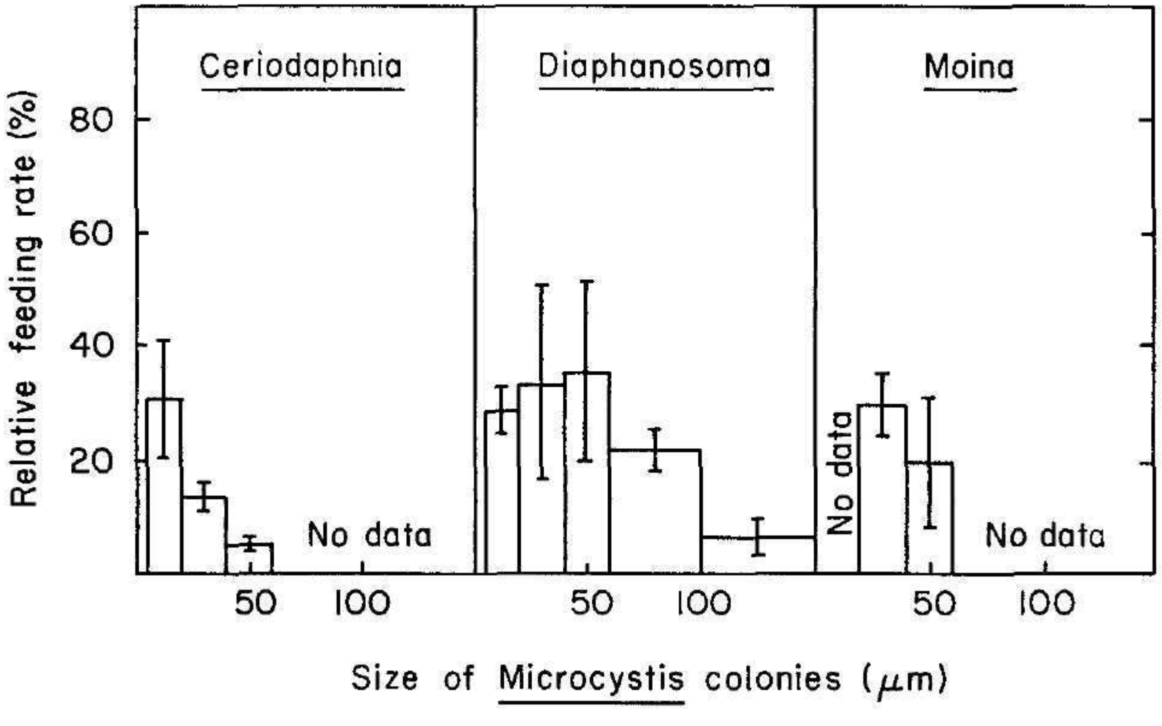


Figure 5.72. Feeding rates of three cladoceran species on increasing size classes of *Microcystis* colonies in relation to their feeding rates on *Chlorella* (100%). Standard error shown as vertical bars.

cies were not possible. No data on the feeding efficiency of *Daphnia* on *Miarocystis* colonies are available due to the absence of this herbivore during the experimental period. De Bernardi, Giussani and Lasso Pedretti (1981) showed that *Mioroaystis* could be efficiently ingested and utilized by three species of *Daphnia* if the algal colonies were small (1-5 cells). Thompson, Ferguson and Reynolds (1982) classified *Mioroeystis* colonies >50 ym as partly edible, and colonies <50 um as inedible based on the analysis of crustacean gut contents (Ferguson, Thompson and Reynolds 1982) and information from the literature.

From the relationship between SGA and biomass shown in Figure 5.71, zooplankton grazing rates may be estimated during periods of both low biomass *(Ceriodaphnia* phase) and high biomass *(Daphnia»* or *Moina* and *Diaphanosoma* phases). Data on zooplankton biomass, grazing rate and feeding efficiency on natural green and blue-green algae are of particular value in modelling the utilization of food, and thus nutrients, by the zooplankton community of Hartbeespoort Dam.

### 5.4.16 Zoobenthos

#### (a) Introduction

No formal project on the role of zoobenthos in the Hartbeespoort Dam ecosystem was included in the present study. However, zoobenthos, particularly chironomid midge larvae, were abundant in the guts of several fish species (Section 5.4.18). Detritivorous invertebrates may also be important in recycling energy from sedimented detritus to other compartments of the lake ecosystem. The abundance and distribution of this group of animals in the dam were therefore superficially surveyed.

A series of replicate samples were collected from three littoral (0.5-1 m water depth) and two profundal (15 m water depth) sampling stations (Fig. 4.1). As the numerical distribution and biomass of the fauna at all stations was contagious, log transformations were carried out to normalize data and maximum likelihood estimators for the mean of a log-normal population with 90 % confidence intervals were calculated according to Rendu (1978).

#### (b) Estimates of benthic faunal numbers and biomass

Zoobenthos biomass was higher in littoral regions than in profundal and both faunal numbers and biomass of animals was higher in the Crocodile River arm than in the Magalies River arm (Tables 5.26 and 5.27). The benthic faunal samples collected indicated a significant difference between faunal densities in the two river arms. Lake faunal densities were calculated by considering the Crocodile and Magalies River arms as separate entities. When the study was carried out (September 1982) the lake contained 78 % of its full supply capacity.

It was estimated that the extended Crocodile River arm contributed 135 ha of the lake's littoral region (0-5 m below water surface) and the Magalies arm 473 ha . Similarly, the profun-

Table 5.26. Benthic invertebrate density (numbers $m^{-2}$), biomass ($g\ m^{-2}$) and 90 % confidence intervals at the two profundal stations in Hartbeespoort Dam, based on five replicate samples at each sampling point.

| | Station Z2 (Magalies) | | Station (Z4 (Crocodile) | |
|---|---|---|---|---|
| | Mean | 90 % Confidence intervals | Mean | 90 % Confidence intervals |
| Large Chironomidae | 1838 | 1310 - 3836 | 1457 | 772 - 7095 |
| Large Oligochaeta | 1107 | 731 - 2494 | 414 | 325 - 684 |
| *Chaoborus* sp. | 213 | 136 - 592 | 231 | 153 - 515 |
| Dry biomass of above | 0.90 | 0.69 - 1.57 | 0.33 | 0.20 - 1.01 |
| Small Chironomidae | 2327 | 799 - 83956 | 282 | 131 - 2145 |
| Small Oligochaeta | 9537 | 4311 - 71718 | 2139 | 672 - 176180 |
| Ostracoda | 3955 | 2757 - 9484 | 30441 | 16858 - 127122 |
| *Physa* sp. shells | - | - | 1900 | 750 - 29584 |

Table 5.27. Benthic invertebrate density (numbers $m^{-2}$), biomass (g $m^{-2}$) and 90 % confidence intervals at the three littoral stations in Hartbeespoort Dam based on five replicate samples at each sampling point.

| | Station I Z1 (Magalies) | | Station Z2 (Magalies) | | Station Z5 (Crocodile) | |
|---|---|---|---|---|---|---|
| | Mean | 90 % confidence intervals | Mean | 90 % confidence intervals | Mean | 90 % confidence intervals |
| Large Chironomidae | 50554 | 28952 - 199638 | 21494 | 8914 - 264726 | 96339 | 68680 - 201059 |
| Large Oligochaeta | 7649 | 2904 - 146501 | 3456 | 1313 - 61862 | 4540 | 3493 - 7900 |
| *Austrocaenis* sp. | 50 | 40 - 83 | 505 | 293 - 1798 | 2126 | 1404 - 5383 |
| Dry biomass of above | 5.81 | 3.83 - 14.69 | 5.60 | 2.71 - 36.80 | 15.21 | 9.65 - 35.11 |
| Small Chironomidae | 6991 | 4618 - 17702 | 14610 | 9039 - 44110 | 147350 | 81602 - 615334 |
| Small Oligochaeta | 21505 | 15101 - 46817 | Outside range of tables | | 19028 | 12568 - 48179 |
| Ostracoda | 23464 | 14517 - 70839 | 97236 | 64224 - 246201 | 20097 | 12014 - 67425 |
| Nematoda | 722 | 248 - 26043 | 22910 | 6999 - 2835155 | 9085 | 2888 - 652848 |
| *Physa* sp. shells | 507 | 299 - 1623 | 1304 | 584 - 11462 | 160 | 106 - 405 |

dal zone extended by the Crocodile arm measured 211 ha and that of the Magalies and remaining main basin measured 843 ha .

Table 5.28 summarizes the components of the total zoobenthos biomass estimate obtained for the lake in September 1982, which was approximately 50 tonnes dry weight which may be compared to the 25 to 39 tonnes of zooplankton (Table 5.24). Assuming that dry weight is 15 % of wet weight (Morgan 1980) this converts to 334 tonnes of zoobenthos. It should be noted that this biomass estimate is on the conservative side. It did not include Nematoda, Ostracoda, smaller Chironomidae and Oligochaeta which in sorting were retained by a smaller mesh net. Although volumetrically small these animals formed a large numerical proportion of the zoobenthos (Tables 5.26 and 5.27).

Table 5.28. Dry biomass estimates of zoobenthos in Hartbeespoort Dam, September 1982.

| | Area (ha) | Biomass (kg ha$^{-}$ ) | Lake Biomass (kg) |
|---|---|---|---|
| Magalies and main basin littoral | 427.7 | 57.0 | 24 379 |
| Magalies and main basin profundal | 843.1 | 3.3 | 2 782 |
| Crocodile arm littoral | 135.0 | 152.1 | 20 534 |
| Crocodile arm profundal | 210.9 | 9.0 | 1 898 |
| TOTAL FOR LAKE | 1616.7 | | 49 592 |

(c) Calculation of production of zoobenthos

Although no estimate of the mean annual biomass of zoobenthos was available it was considered worthwhile to see how productive the estimated population of zoobenthos could be, and its potential contribution to the total biomass of secondary producers in Hartbeespoort Dam. Waters (1977) states that a good estimate of the production of zooplankton, zoobenthos and fish could be obtained by multiplying the mean annual standing stock of these three groups of secondary producers respectively by 15-20, 6-8 and 0.5-1. Hence an estimate of the zoobenthos production, assuming that 334 tonnes represents the mean annual standing stock, would be from 2004 to 2672 tonnes. This crude estimate illustrates the potential importance of zoobenthos production in the Hartbeespoort Dam ecosystem.

(d) Discussion

Estimates of the biomass of zoobenthos of Hartbeespoort Dam (3.1 g $m^{2}$) are at the upper limit (1-3 g $m^{2}$) of the mean annual biomass for tropical lakes (Morgan 1980). The Chironomidae were exceptionally abundant at some stations (Table 5.27) but most individuals were in the small size range. If predation and natural mortality remained low then the biomass of zoobenthos could become considerably higher. Predation on Chironomidae by fish in October/November 1982 became intense when juvenile *0, mossconbicus*, the only species besides *C. flaviventris* to breed that year (Section 5.4.17), were found to feed almost exclusively on chironomid larvae during a certain stage of their development (Section 5.4.18).

It was not possible to monitor population changes in this group as sampling for zoobenthos was carried out only in September 1982. However, based on the September 1982 data with a preponderance of small chironomids it is possible to develop a successional hypothesis. Low predation on benthic chironomids in winter and early spring could allow the development of a very large population of these animals in early summer. This large chironomid population then provides an abundant energy-rich supply of food for fish. With the onset of warmer weather adult *0. mossombicus* start breeding prolifically and the growing fry and juveniles then reduce the population of these chironomids through predation. With the onset of colder temperatures the fish stop breeding and as larger *0. mossamb-icus* change their diet (Section 5.4.18) the predation pressure on zoobenthos is considerably reduced. Chironomidae now re-establish a large population, which provides a potentially abundant food supply for the following summer's brood of fish fry.

The importance of zoobenthos in Hartbeespoort Dam could not be accurately assessed from this pilot study. However, their high biomass and numerical abundance indicated that zoobenthos is important and warrant detailed study.

### 5.4.17 Fish population dynamics

#### (a) Species composition

To date 12 species of fish have been recorded in Hartbeespoort Dam but of these *Oreoohromis mossambious> Cyprinus carpio* and *Clarias gariepinus* made up over 90 % of the icthyomass caught using four different types of sampling gear (Table 5.29).

Table 5.29. Species composition fish of catches in Hartbeespoort Dam (from Cochrane 1984).

| Method | Gill nets | Seine net (13 mm stretch) | Seine net (35 mm stretch) | Angling returns |
|---|---|---|---|---|
| Period of sampling | Nov. 1981- April 1984 | Nov. 1981- Jan. 1982 | Nov. 1981- Jan. 1983 | March 1982- Jan. 1983 |
| Species | | % composition of catch by mass | | |
| *O. mossambicus* | 66 | 86 | 50 | 15 |
| *C. carpio* | 1 | 4 | 6 | 74 |
| *C. gariepinus* | 26 | 3 | 43 | 10 |
| Others | 7 | 7 | 1 | 1 |
| Total mass (kg) | 1201 | 142 | 16 247 | 59 710 |

Other commonly encountered species were *Chetia flaviventris* and *Barbus marequensis*. The remaining species were caught only occassionally and in low numbers.

The number and biomass of adults of the three dominant species are shown in Table 5.30. The increased biomass of *0. mossambious* in October 1983 compared to October 1982 was a result of good recruitment to the adult stock from the fry spawned in summer 1981/82 following a mild winter mortality (Cochrane 1984). The apparent increase in *C, gariepinus* numbers and biomass was a result of sampling bias. The higher figure in 1983 was based on larger samples and a higher number of tag recaptures and was thus more reliable. The *0, mossambiaus* population could be expected to show wide fluctuations in standing stock, as a result of variable winter mortality, even under relatively stable environmental conditions. Conversely the *C. oavpio* and *C. gariepinus* populations are not as sensitive to small fluctuations in temperature or other known environmental parameters and thus their populations should be relatively stable. The total biomass of the three species, approximating total ichthyomass, can be estimated as:

| | |
|---|---|
| 1982 | 1160 tonnes |
| 1983 | 1400 tonnes |

The biomass of 0+ fish, as determined by the 3 poisoning replicates in 1982 and 1984, is shown in Table 5.31. The dominance of the three major species was shown in the samples. *Chetia flaviventris* was also present in large numbers but, as this is a littoral species, littoral poisoning overestimated its abundance in relation to the other three species.

The higher biomass, of the four species in 1984 compared to 1983 (Table 5.31) was due to sampling earlier in the year in 1984, before water temperatures started to drop and fish moved into deeper water. *C, oavpio* and *C. gariepinus* also showed greater spawning intensity and larval/fry survival in 1984 due to the greater abundance of aquatic vegetation.

Cochrane (1984) and Cochrane & Robarts (in prep.) have discussed the reasons for the current species composition in Hartbeespoort Dam. There has been a decline in predatory, riverine species, such as *Barbus marequensis* and *Barbus mattozi,* due to

Table 5.30. Numbers and biomass of the dominant fish species in Hartbeespoort Dam.

| Species | Date of estimate | Age group | Number | 95 % confidence limits | | Biomass (t) |
|---|---|---|---|---|---|---|
| *O. mossambicus* | Oct 82 | 2++ | 307 635 | 210 487 - | 470 501 | 289 |
| | Oct 83 | 2++ | 861 803 | 607 199 - | 1 272 226 | 522 |
| *C. gariepinus* | Oct 82 | Adult | 120 667 | 41 136 - | 603 33 | 200 |
| | Oct 83 | Adult | 175 276 | 80 279 - | 478 026 | 292 |
| *C. carpio* | March 82 | Adult | 243 000 | 217 247 - | 281 278 | 580 |

2++ = Fish 2 years old or older
Adult = Fish 1 year old or older

Table 5.31. Mean biomass and percentage of total of all fish species in 3 samples collected by littoral poisoning in Hartbeespoort Dam in March 1982 and January 1984.

| Species | 1982 Mass (kg ha$^{-1}$) | 2SE | % | 1984 Mass (kg ha$^{-}$ ) | 2SE | % |
|---|---|---|---|---|---|---|
| *O. mossambicus* | 17.2 | 13.2 | 56 | 225 | 390.2 | 84.2 |
| *C. carpio* | 1.6 | - | 5 | 6.3 | 4.7 | 2.4 |
| *C. gariepinus* | - | - | - | 16.2 | 13.0 | 6.1 |
| *C. flaviventris* | 11.7 | 11.7 | 38 | 17.8 | - | 6.7 |

declining water quality, particularly increasing pH and incidence of anaerobiosis. Hypertrophic conditions have led to an increase in the importance of detritus as a food source to fish which, coupled with decreasing water quality, has favoured the detritivorous/benthivorous *C. oarpio* and the omnivorous C. *gariepinus. 0. mossombious*, a phytoplanktivore and detritivore, is under stress, at least partially as a result of the dominance of the phytoplankton by the nutritionally inferior *M. aeruginosa* (Cochrane 1984).

(b) Distribution

There were significant differences in the catch per unit effort (CPUE) of the major species with the large seine net between the southern and northern (including north-eastern) shores. In October 1982 catches of *0. mossambicus* and *C. gariepinus* were 11 and 3 times higher, respectively, on the southern than the northern shores. In May 1982 catches of *C. oarpio* were approximately 6 times higher on the southern than the northern shores, with the exception of the bay enclosed by zone $d_2e_2$ where catches were high (Fig. 4.1). The full results are shown in Table 5.32.

Table 5.32. Comparison of CPUE in major shore line areas of Hartbeespoort Dam in May and October, 1982. Zones are shown in Fig. 4.1.

| Species | Month | Zone 1 | CPUE kg | Zone 2 | CPUE kg |
|---|---|---|---|---|---|
| *O. mossambicus* | May | S.shore + $d_2e_2$ | 50.92 | N.Magal + NE-$d_2e_2$ | 7.54 |
| | Oct | S.shore | 220.4 | N.shore | 18.4 |
| *C. gariepinus* | May | S.shore + NE | 35.11 | N.Magal. | 3.35 |
| | Oct | S.shore | 144.8 | N. shore | 50.0 |
| *C. carpio* | May | S.shore + $d_2e_2$ | 17.21 | N. shore + -$d_2e_2$ | 3.01 |
| | Oct | Total | 5.7 | | |

The results of a statistical investigation into factors regulating the distribution of fish are shown in Table 5.33. Catches of *C. carpio* were not large enough for this analysis.

A five variable equation, based on 11 data points, accounted for 67 % of the variation observed in the CPUE while a three variable equation, based on 24 data points, accounted for 49 % of the variability (Table 5.33).

Table 5.33. Relationships between combinations of environmental variables and CPUE in October 1982 for *O. mossambicus* and *C. gariepinus* ir Hartbeespoort Dam. (* = adjusted multiple; Galpin 1981).

| Variable(s) | n | $r^2$ | Probability |
|---|---|---|---|
| 1/Gradient | 24 | 0.43 | $< 0.05$ |
| 1/Gradient, NW fetch | 24 | 0.49* | $< 0.01$ |
| 1/Gradient, NW fetch, ESE fetch, % Clay | 11 | 0.67* | $< 0.03$ |

Greatest production of food organisms for fish will occur in shallow, sheltered areas (Hakanson & Jansson 1983) and thus greatest fish densities could be expected in these areas as well. Thus the gradient was the major factor in determining fish distribution in Hartbeespoort Dam. While fetch should have been be important, the correlation between CPUE and ESE fetch was negative, as would be anticipated, but that between NW fetch was positive. ESE and NW are the two major wind directions on Hartbeespoort Dam (Section 5.1.2) and the contradiction in correlation coefficients suggests their importance in the equations was not causative. As well as influencing actual abundance, gradient would influence the efficiency of the seine net used, efficiency declining with higher gradient, but this bias was minor compared to the real differences in abundance.

### (c) Production and yield

#### (i) Growth

The growth curves of the three species are shown in Table 5.34. The asymptotic lengths of male and female

Table 5.34. Von Bertalanffy growth curves for the three dominant fish species in Hartbeespoort Dam.

| Species | Sex | Curve |
|---|---|---|
| *O. mossambicus* | Male | $lt = 36.41\ (1-e^{-0.49(t-0.38)})$ |
| | Female | $lt = 30.91\ (1-e^{-0.58(t-0.43)})$ |
| *C. carpio* | Male | $lt = 57.56\ (1-e^{-0.56(t+0.15)})$ |
| | Female | $lt = 59.49\ (1-e^{-0.67(t-0.14)})$ |
| *C. gariepinus* | Male | $lt = 97.32\ (1-e^{-0.21(t+0.22)})$ |
| | Female | $lt = 79.00\ (1-e^{-0.23(t+0.60)})$ |

lt = length at age t years.

*C. carpio* were considerably less than the approximately 70 cm observed in the dam and thus these curves underestimate the growth rates of the species. Calculated asymptotic lengths agree closely with those observed in the other two species. It is apparent that *C. aappio* had the greatest growth rate in the first year and would have been available to anglers at an age of approximately six months (Table 5.35).

The growth rate of *0. mossambious* fry was highest in 1981/82 (Fig. 5.73) when density was low and there was emergent and submerged aquatic vegetation in the littoral zone. The marginal vegetation would have increased epiphytic production and hence food availability for fry (Hakanson an Jansson 1983). Growth rate was lowest in 1982/83 when density was intermediate but there was a complete absence of aquatic vegetation due to falling water level. It was not possible to make similar comparisons for the other two species.

**Table 5.35.** **Length: mass relationships of Hartbeespoort Dam fish and some important growth parameters: SL = standard length, M = mass, $L^{\infty}$ and $M^{\infty}$ = asymptotic SL and M respectively.**

| Species | M : SL relationship | Sex | $L^{\infty}$ (cm) | $M^{\infty}$ (g) | Mass at 12 mths (g) |
|---|---|---|---|---|---|
| *O. mossambicus* | $M = 0.031\ SL^{3.02}$ | M | 36.41 | 1 608 | 28 |
| | | F | 30.91 | 981 | 21 |
| *C. carpio* | $M = 0.037\ SL^{2.94}$ | M | 57.56 | 5 533 | 613 |
| | | F | 59.49 | 6 096 | 538 |
| *C. gariepinus* | $M = 0.015\ SL^{2.96}$ | M | 97.32 | 11 510 | 139 |
| | | F | 79.00 | 6 210 | 190 |

### (ii) Fecundity

*C. carpio* had the highest fecundity per unit mass, followed by *C. gariep-inus* and then, at a much lower level, the mouth brooding *0. mossambicus* (Table 5.36). *C. carpio* and *C. gariepinus* normally spawned when there was an inflow of water and the lake level rose, flooding marginal vegetation. *C. oavpio* may spawn more than once a year but the eggs spawned in any one year are distinguishable by size in mature ovaries (Gromov 1979). *C. gariepinus* is generally considered to be a single spawner (Clay 1979). Thus counts of mature eggs in ripe ovaries will provide estimates of the number of eggs spawned per year per fish. 0. *mossambious* is known to be a multiple spawner. The number of spawnings per year in Hartbeespoort Dam was calculated from the percentage of mouth-brooding females caught in summer and the mean time period of mouth-brooding. The figures obtained were (2SE in brackets):

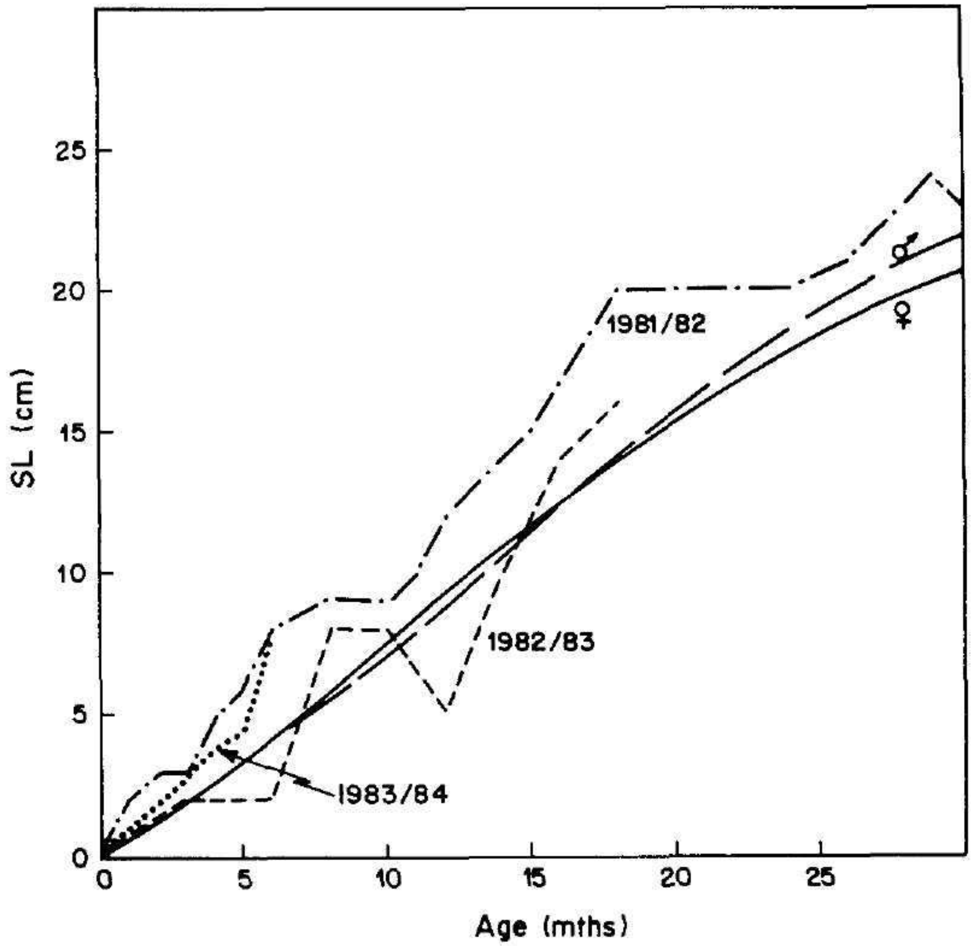


Figure 5.73. Growth of *O. mossambicus* spawned in the summers of 1981/82, 1982/83 and 1983/84 as shown by length frequency modes, and computed growth curves (Table 5.31). ♂ = male, ♀ = female growth curve calculated from scale rings. SL = standard length (rom Cochrane 1985).

Table 5.36. **Best-fit fecundity to size relationships for major fish species in Hartbeespoort Dam.**

| Species | Period of sampling | n | $r^2$ | Significance | Equation |
|---|---|---|---|---|---|
| *O mossambicus* | 10.82-10.83 | 53 | 0.76 | 0.99 | $F = 40.45\ M^{0.63}$ |
| *C carpio* | 3.83- 9.83 | 30 | 0.64 | 0.99 | $F = 0.45\ SL^{3.58}$ |
| *C gariepinus* | 9.82-11.82 | 16 | 0.61 | 0.99 | $F = 0.17\ SL^{3.30}$ |

% mouth-brooding (2SE) = 28.7 (24.05)
Time of brooding per spawning (d) = 23 (3.3)

This resulted in a mean of 2.7 spawnings per fish between October and March.

The age at first maturity was found to be one year for *C. carpio* and two years for *C. gariepinus.* In the summer of 1982/83, 1+ (one year old) *0. mosscanbicus* were found to be breeding when they exceeded 100 g mass. However, in 1983/84, when the fish density was considerably higher, due to low winter mortality in 1981/82 and low water levels, 2+ fish were not breeding and the age at first maturity was assumed to be three years. The virtual absence of 3+ fish in the population at this time prevented confirmation of this.

(iii) Mortality

*C. carpio* was the most heavily exploited fish species, followed by 0. *mossambious* and then *C. gariepinus* (Table 5.37). The very small losses due to natural mortality in *C. carpio* suggested that the species was being fully or over-exploited during the period of study. Using the suggestion of Gulland (1970) that at maximum sustainable yield (MSY) , natural mortality and fishing mortality should be approximately equal, *C. carpio* was overexploited, *C. gariepinus* was slightly over-exploited and *0. mossambicus* was underexploited. This is discussed further under the next section (Yield), and under Management (Section 7.4).

In addition to the mortality rates shown in Table 5.37 *0. mossambicus* suffered young of the year mortality, caused primarily by cold, in winter (Cochrane 1984). The extent of this mortality varied each year. Examination of Figure 5.74 and the study of the scale ring frequencies of *0. mosscanbicus* showed the following categories of winter mortality for the past seven years:

Table 5.37. Mortality rates of the major fish species in Hartbeespoort Dam. All rates given as annual expectation of death.

| Species | Mortality | Period studied | rate |
|---|---|---|---|
| *O. mossambicus* | Total | Dec 81 - Dec 83 | 0.62 |
| | Natural | ** | 0.46 |
| | Fishing | Oct 82 - Oct 83 | 0.16 |
| | 0+ | (Jan-Mar) 82,83,84 | $1-(3.6x10^{-6})$ |
| *C. carpio* | Total | Mar 82 - April 83 | 0.85 |
| | Natural | ** | 0.06 |
| | Fishing | Mar 82 - Feb 83 | 0.79 |
| | 0+ | Mar 82 - Mar 83 | $1-(7.1x10^{-6})$ |
| *C. gariepinus* | Total | Oct 82 - Oct 83 | 0.21* |
| | Natural | ** | 0.09 |
| | Fishing | Oct 82 - Sept 83 | 0.12* |
| | 0+ | Oct 82 - Oct 83 | $1-(2.6x10^{-6})$ |

* = mean value, mortality increases exponentially with size
** = not applicable, natural mortality derived from total and fishing mortality.

| Date of Spawning | Winter Mortality |
|---|---|
| 1975/76 | 3 |
| 1976/77 | 1 |
| 1977/78 | 3 |
| 1978/79 | 4 |
| 1979/80 | 4 |
| 1980/81 | 3 |
| 1981/82 | 2 |
| 1982/83 | 1 |

1 = 0-24 % mortality, 2 = 25-74 %, 3 = 75 - 99 %, 4 = 100%.

These approximations show a mean mortality of approximately 50 % per annum. However, it was apparent that standing stock can vary considerably (Fig. 5.74 and Table 5.30) which would have to be taken into account in any management strategy.

(iv) Yield

The mean catch per angler day of the three species and the total effort are shown in Figure 5.75. The effort declined during 1983, probably as a result of the drought and falling water levels. Figure 5.75 demonstrates the importance of *C. cavipio* in angler catches and the seasonal nature of *0. mossambicus* catches, peaking in summer and reaching 0 in mid-winter. *Clarias* catches tended to increase during the study period but this was due to the

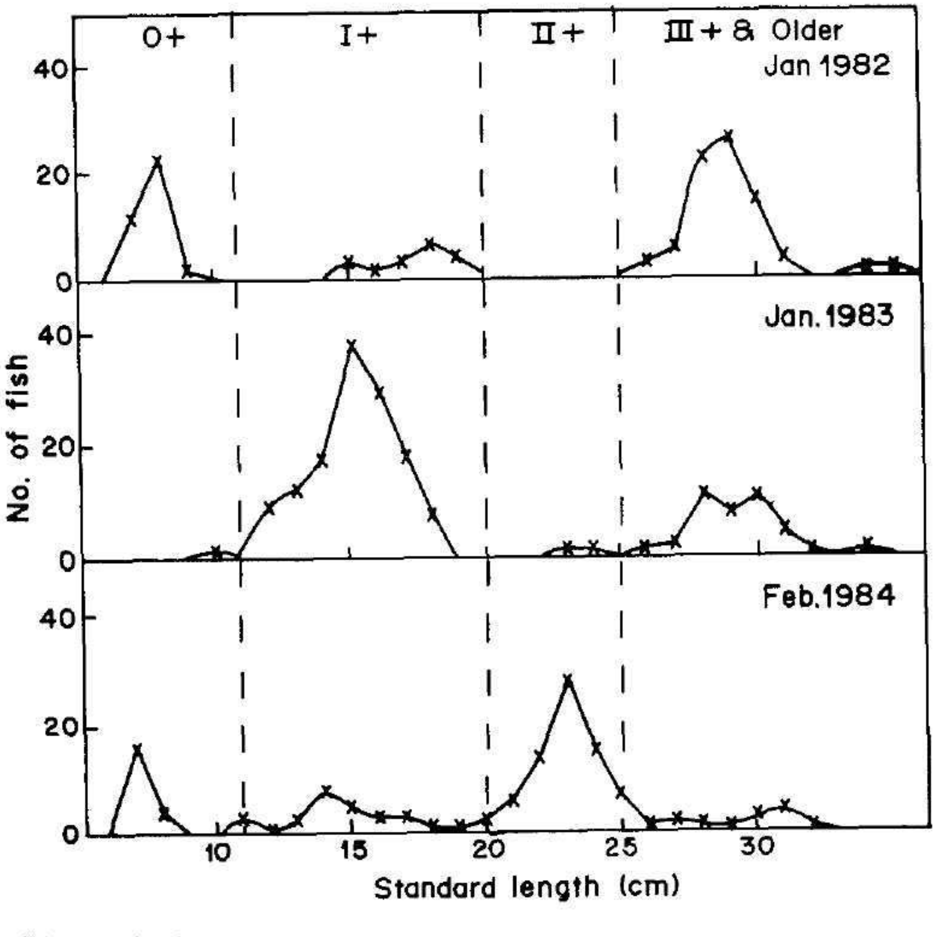


Figure 5.74. Standard length frequencies of *O. mossambicus* caught in gill nets in January, 1982 & 1983 and February 1984. Roman numerals indicate age class distributions. Catches in each case are the total caught in 420 m of gill nets (from Cochrane 1984).

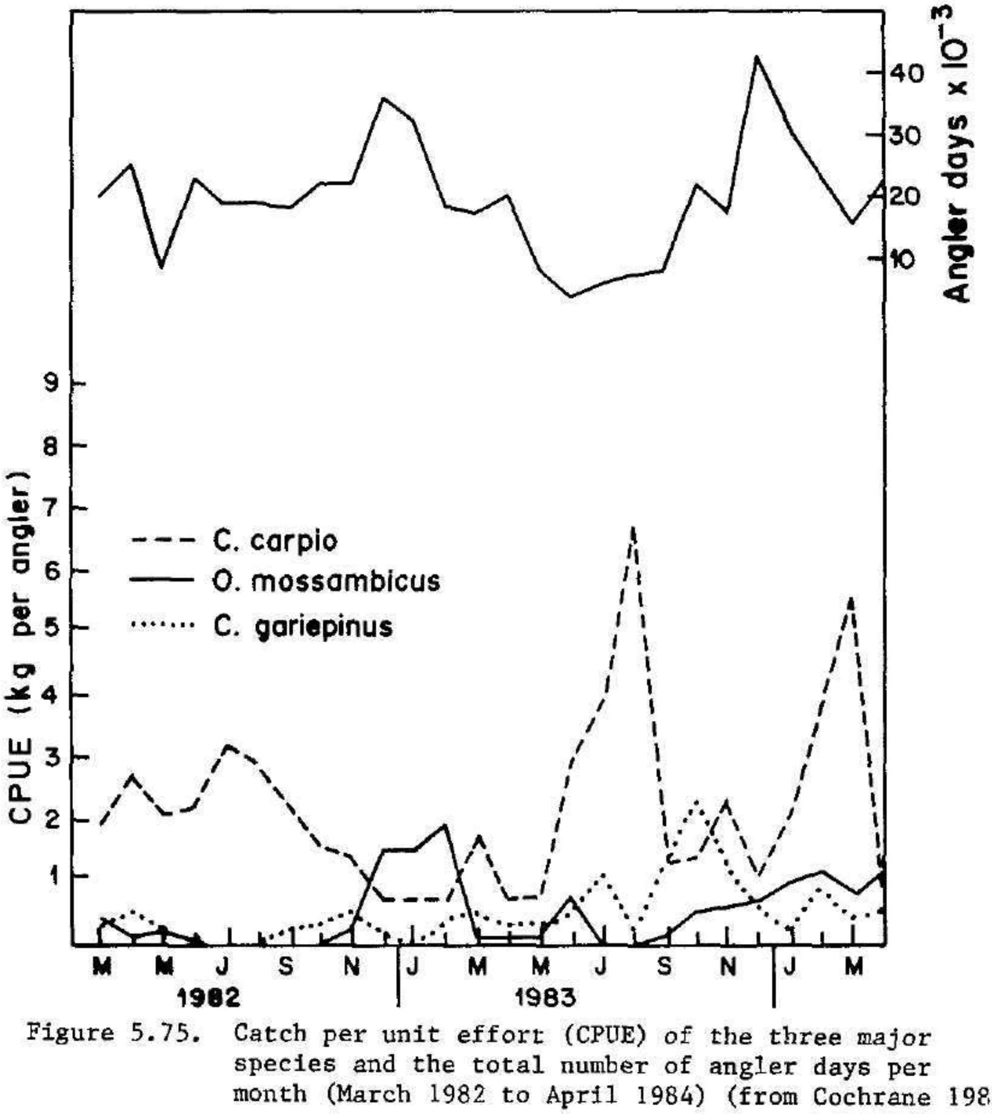


Figure 5.75. Catch per unit effort (CPUE) of the three major species and the total number of angler days per month (March 1982 to April 1984) (from Cochrane 1985).

The results of angler catches are summarized in Table 5.38.
The mean annual estimates are based on the 20 months data.

Table 5.38. Annual angler catch statistics for Hartbeespoort Dam (March 1982 - April 1984).

| | Total (26 mths) | Mean Annual catch (t) |
|---|---|---|
| No. of anglers | | |
| Catch (t) | | 233 723 |
| *O. mossambicus* | 311 | 44 |
| *C. carpio* | 970 | 449 |
| *C. gariepinus* | 220 | 102 |
| Total | | 695 |

Total yield of all three species is likely to be high because of the low water levels which would increase fish catchability. The mean annual catch of 695 t represents 769 kg ha at the mean volume during this period (45.2 %) or 348 kg ha $^{1}$ at full supply.

(d) Synthesis of results

The biological results described above were used to construct single species models for the three dominant species to determine sustainable yield (Cochrane 1985). Model output is described and discussed in Chapter 6.

### 5.4.18 Feeding biology of fish

(a) Introduction

Population studies of the fish fauna in Hartbeespoort Dam (Section 5.4.17) revealed that of the twelve species of fish encountered in the lake *0, mossambieus, Cypvinus oavpio*$_{3}$ *Clavias gariepinus* and *Chetia flaviventvis* were numerically and in total biomass the most abundant. For this reason studies on feeding biology have concentrated mostly on these species although data on the food types utilized by *Bavbus mattozi*$_{3}$ *Barbus marequensis* and *Engraul-ioypris bvevianali-s* are also included.

*Oreoohromis mossombicus* has been described as a phytoplanktivore (Caulton 1979) a microphagous omnivore (Balarin 1979) and a detritivore (Bowen 1980). Because of its reliance on these food sources and its breeding behaviour, this species is preadapted for an existence in a lacustrine environment. Therefore, a study of the food sources, feeding behaviour and anatomical adaptations to a changing diet in *0. mossombicus* were carried out in greater detail than for the other fish species.

(b) The diet of the fish species in Hartbeespoort Dam

Figure 5.76 shows the feeding preference of seven of the more frequently encountered fish species in the lake. The two

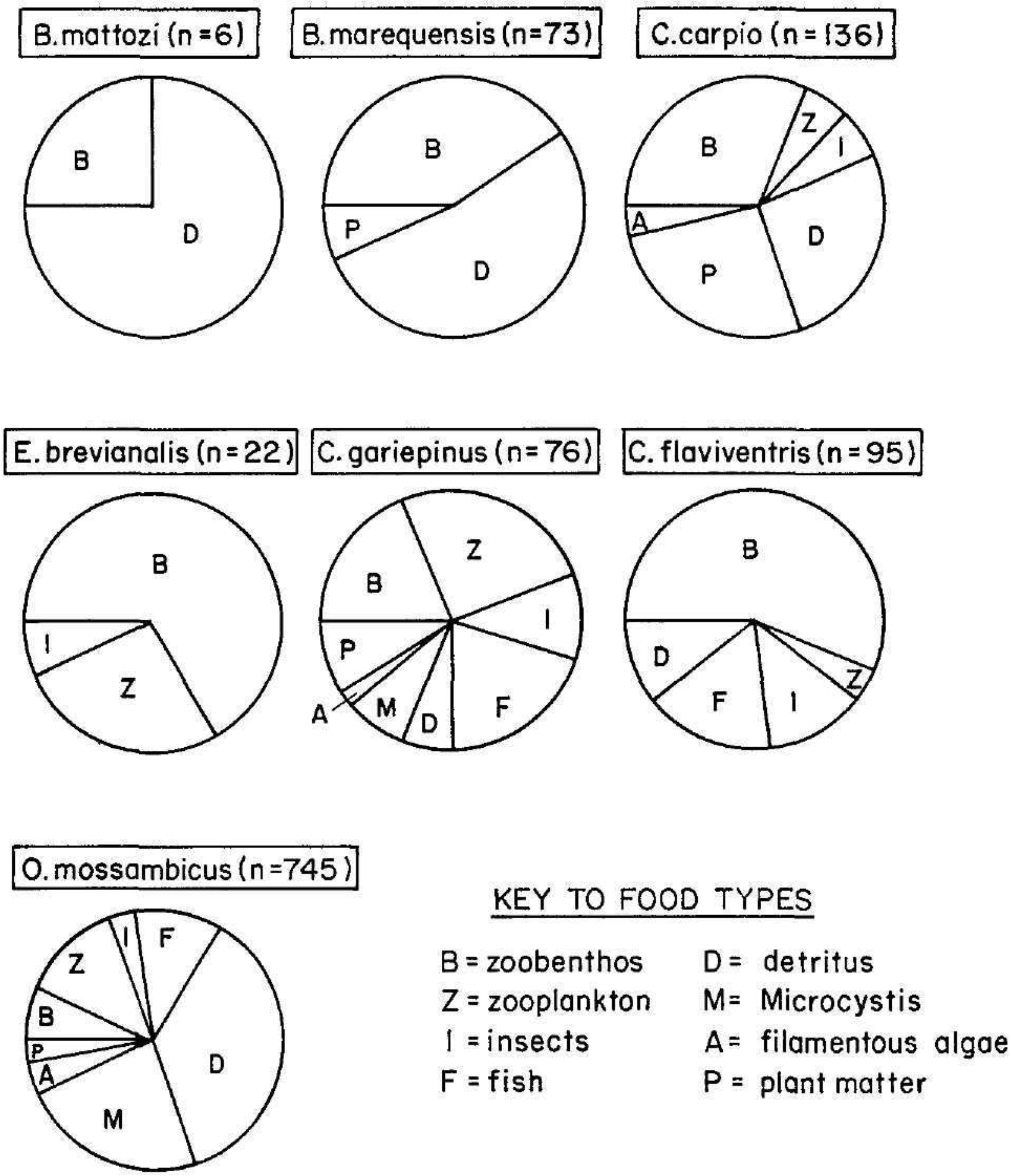


Figure 5.76. The abundance of identifiable food types occurring in volumes exceeding 20% of total gut volume of the common fish species in Hartbeespoort Dam.

indigenous *Barbus* species and the exotic carp, *Cyprinus carpio*, are mainly benthic grubbers and zoobenthos and detritus form a major portion of their diet. Donnelly (1982) found that *Barbus mattozi* from a small impoundment in Zimbabwe had a mainly carnivorous diet with an increasing preference for fish in larger individuals. In the present study carp fed on aquatic raacrophytes when they were present, and a high incidence of *Polygonum* leaves and seeds were mainly responsible for the abundance of plant matter in Figure 5.76.

*Eugraulicypris brevianalis* showed a preference for zoobenthos and zooplankton. Although apparently a specialized feeder the restricted habitat (inflowing regions of the rivers) and low numbers of this species made its overall role as a user of these food resources negligible compared to the other fish species.

The barbel *Clarias gariepinus* and the canary kurper *Chetia flaviventris* had predominanty carnivorous diets with zooplankton, zoobenthos, fish and insects forming the major food items ingested by these species. These two species were the most important secondary consumers (feeding on food types other than plant matter), with fish and insects being utilized equally frequently by both, but with a strong preference being shown for zooplankton by *C. gariepinus* and for zoobenthos by *C. flaviventris.*

Examination of the food ingested by the blue kurper *Oreochromis mossambicus* showed that it fed mostly on detritus and *Miorocystis* sp. , these food items appearing in excess of 20 % of total gut contents in more than half the fish stomachs examined (Fig. 5.76).

### (c) The feeding habits of the three major fish species in Hartbeespoort Dam

#### (i) *Cyprinus carpio*

Detritus and zooplankton were found in the majority of stomachs examined. The important food items (>20 % of gut content volume) in the various size groups of fish during the four seasons are shown in Figure 5.77. The smaller fish (0-10 g) fed predominantly on zoobenthos, insects and zooplankton with larger fish feeding progressively more on plant matter and detritus in summer and autumn. Fish over 500 g fed mostly on detritus and benthos in spring, on plant matter in summer and on detritus and plant matter in autumn and winter. Zoobenthos formed an important dietary component in all size groups of fish and insects were frequently a major portion of carp gut contents in summer.

#### (ii) *Clarias gariepinus*

The main food items of all sizes of fish included fish, insects and zoobenthos (Fig. 5.78). Small barbel (<10 g) fed most frequently on zoobenthos (mostly Chironomidae) and insects, larger fish (10 - 100 g) had large quantities of fish in their stomachs whereas fish of over 500 g relied in winter and spring most frequently on zooplankton

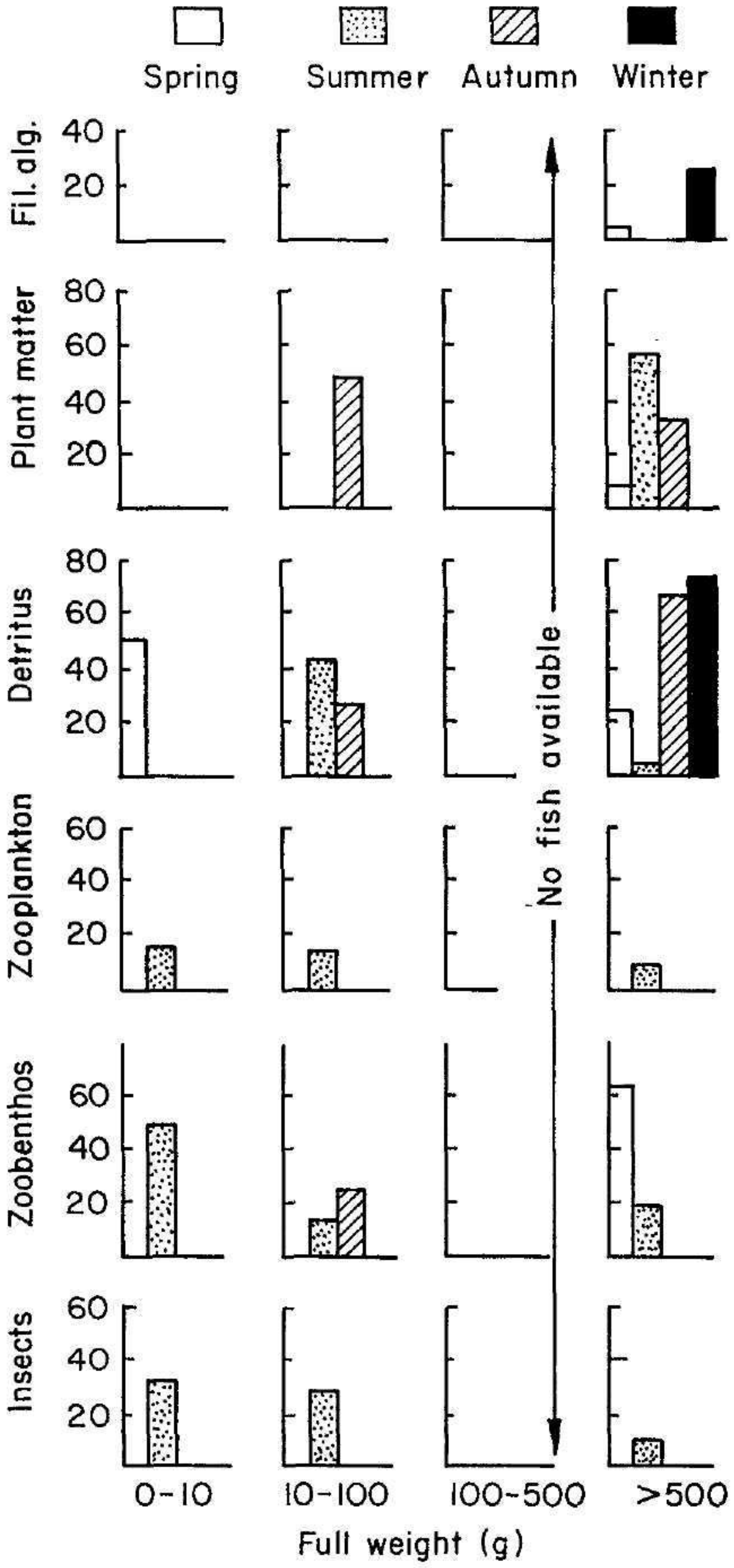


Figure 5.77. The percentage abundance of food items occurring in excess of 20% of gut contents volume in *C. carpio* for different mass classes.

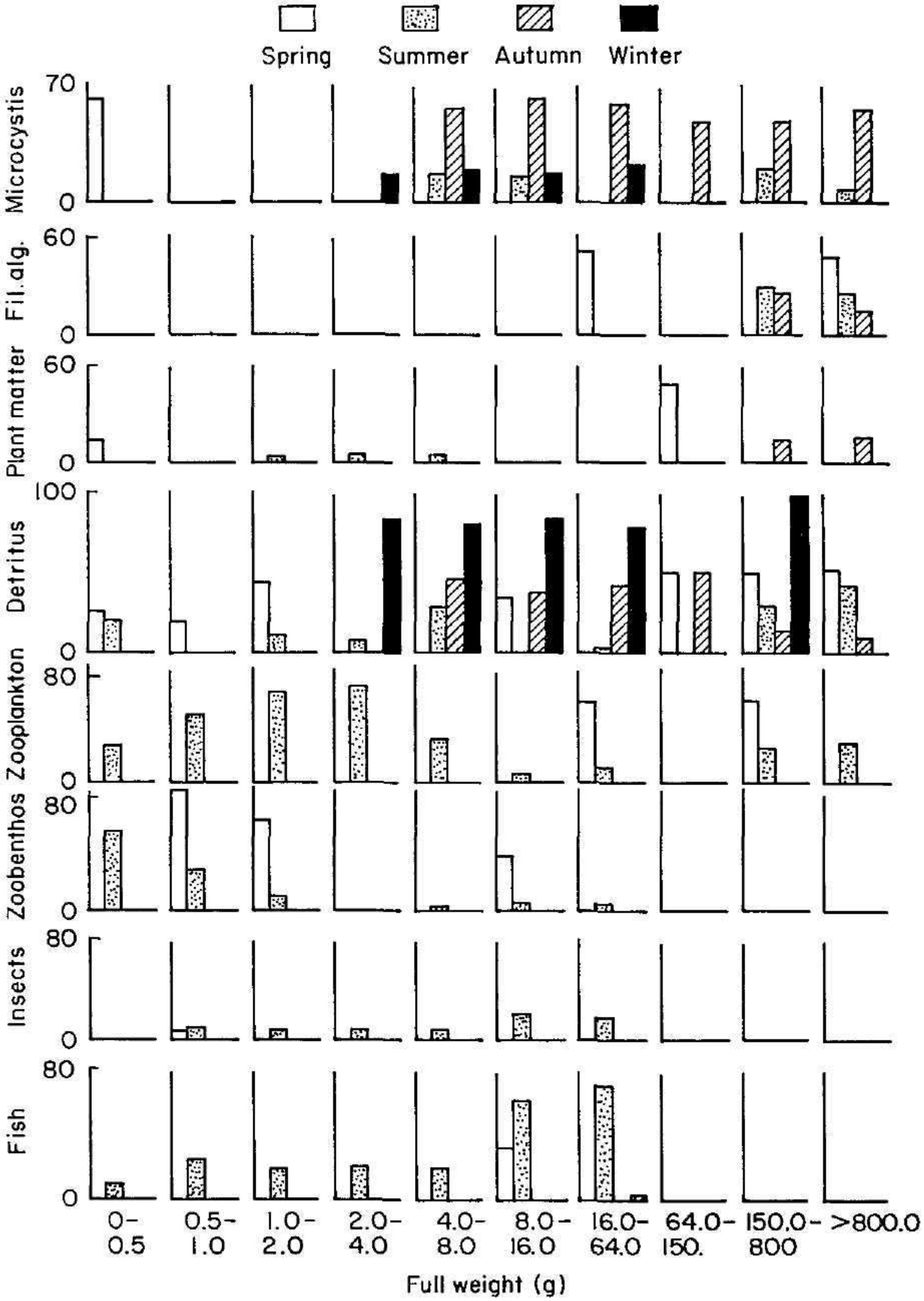


Figure 5.78. The percentage abundance of food items occurring in excess of 20% of the volume of gut contents in *O. mossambicus* for different mass classes.

(*Daphnia* sp. mostly). Bruton (1979) explained why small *C. gariepinus* did not feed as readily on zooplankton as larger individuals. He postulated that the absolute volume of the buccal cavity in larger fish made it energetically more economical for larger fish to filter out zooplankton than smaller fish. Smaller fish also spend most of their time in the littoral zone and zooplankton are not as abundant there as in the pelagic areas. This would make encounters by smaller fish with zooplankton less frequent than would be the case with larger fish in the pelagic zone. This was also borne out by the abundance of zoobenthos in the gut of smaller fish (Fig. 5.78). An examination of the seasonal abundance of zooplankton (Section 5.4.15) revealed that zooplankton were scarce when *C. gaviepinus* were below 100 g in weight (December to February). It may be that the animal protein consumed by *C. gariepinus* reflected the relative abundance of the various faunal groups in the lake at the time of collection of fish.

### (iii) *Oreochromis mossambious*

Small fish, present in large numbers only in spring and summer, fed mostly on the zoobenthos, zooplankton and, as they became larger, to a greater extent on fish (Fig. 5.79). *Miorooystis* and detritus in summer and autumn increased in importance as fish mass increased from 4 g, and detritus was the most abundant food component of large fish in winter (Fig. 5.79).

Figure 5.80 summarizes the change in diet of *0. mossambious* % An initial preference for zoobenthos in fish up to 1 g is followed by a greater preference for zooplankton in fish up to 4 g . These small fish also had a large percentage of stomachs filled with fish scales. Fish up to 64 g in weight showed some cannibalism, feeding on smaller fish, but the major food sources of fish over 4 g were detritus and *Miovooystis*.

Bowen (1976) noted that the ratio of gut length to total body length in fish gave a good indication of their main dietary requirements. Carnivores had a ratio of gut length to total length of 0.67, omnivores 1.62 and herbivores 4.21. The gut length/total length ratio of 121 formalin preserved *0. mossambious* in Bowen's (1976) study was 3.93. Unfortunately he did not include the size range of fish used in this study. In the Hartbeespoort Dam study a selection of 65 *0, mossambiaus* of various sizes gave a ratio which increased with total length of the fish from 0.58 for fish under 2 cm to 11.02 for fish over 35 cm (Fig. 5.81) suggesting increased herbivory in older fish.

The dietary changes which occurred in *0. mossambious* with an increase in size are therefore also reflected by internal anatomical and physiological changes.

## (d) The digestibility of *Microcystis*

As the alga *Miorooystis* formed a major portion of gut contents of *0. mossambious* and as several workers have conflicting viewpoints regarding the digestibility of this alga (Moriarty &

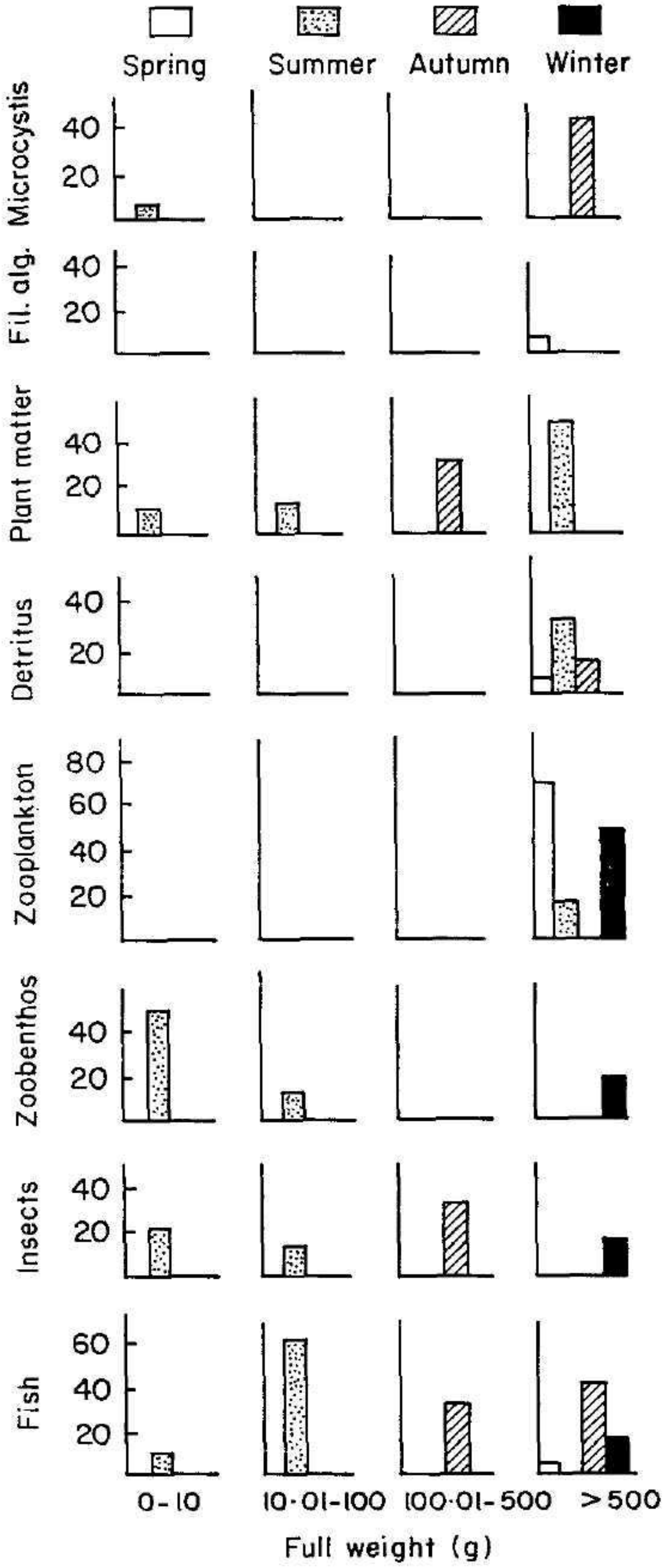


Figure 5.79. The percentage abundance of food items occurring in excess of 20% of the volume of gut contents in *C. gariepinus* for different mass classes.

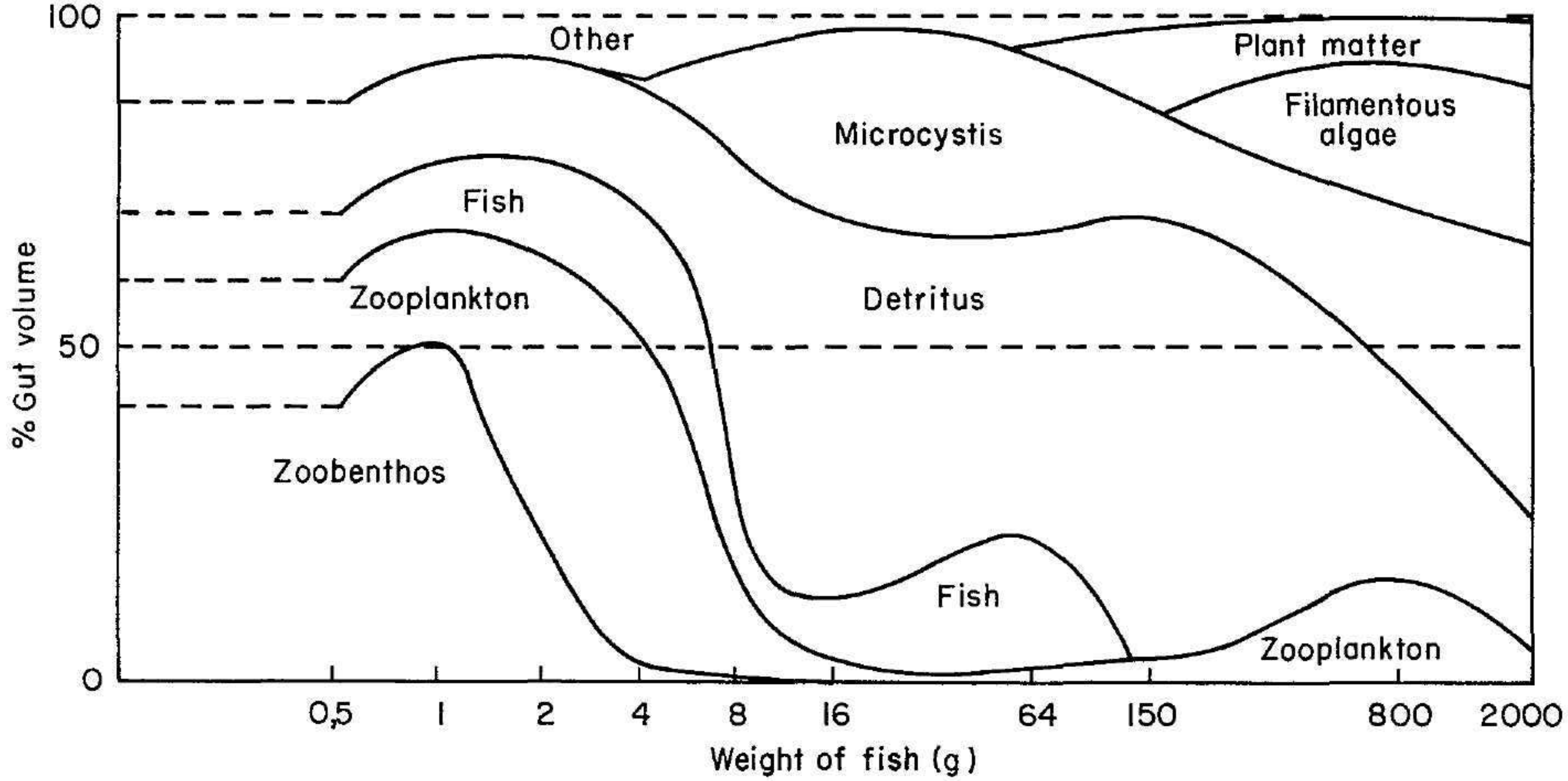


Figure 5.80. Diagrammatic representation of the major food items, as a percentage of gut volume, with increasing size in *O. mossambicus*.

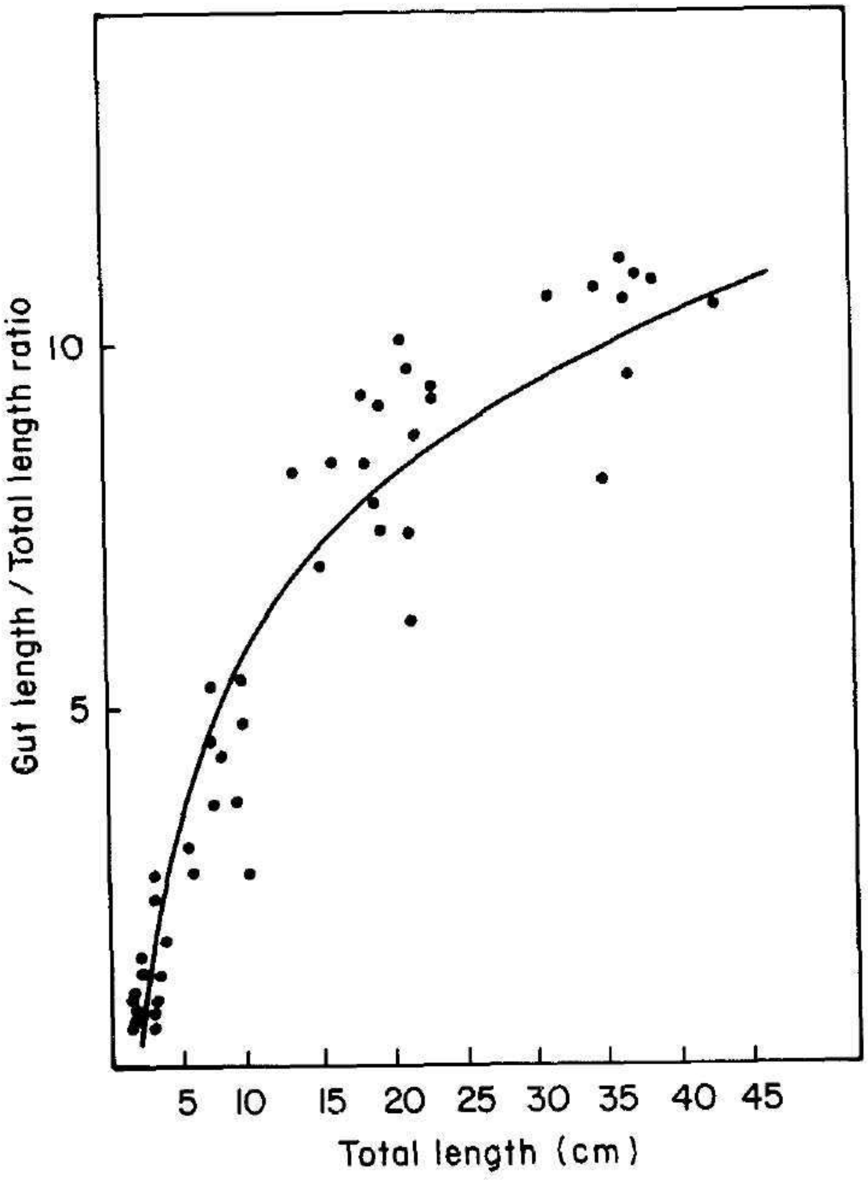


Figure 5.81. Regression curve representing ratio of gut length to total length of fish (cm), with increasing fish total length.
$y = -2.03 + 3.37 \log_e x$ ($r = 0.91$; $n = 65$).

Moriarty 1973b) it was considered important to examine the assimilation of *Mieroeystis.* Laboratory experiments revealed that dry *Mioroeystis* contained 56 % protein and 0.62 % phosphorus. Fish fed on a diet of pure *Mierocystis* assimilated from 32-97 % of the protein (x = 65 %, n = 19), and from 65-84 % of the phosphorus (x = 76 %, n = 4). The total assimilation efficiency for *Mierocystis* by *0. mossambieus* was 20-93 %, x = 55 % (n = 22).

Light and transmission electron microscopic examination of *Mierocystis* from fish faeces revealed that although there were still apparently some undamaged cells, many *Miovooystis* cells had retained their shapes but had lost almost all their cell contents. It was apparent that the cell walls of these empty cells withstood the drainage of the cell contents, retaining the original shape of the cell. There is evidently some disruption of the cell wall, which allows digestive enzymes into the cell, and is followed by a breakdown of the contents which then leach out.

It can be concluded that *Mierocystis* is utilized by *0. mossambieus* .

### (e) Feeding periodicity and rates of defecation and excretion in *0. mossambieus* in Hartbeespoort Dam

An initial survey carried out in December 1984 and data presented in Figure 5.82 revealed that, like most species of 'Tilapia', *0. mossambious* restricted its feeding activity to the daylight hours. To arrive at the data presented in Figure 5.82 the weight of gut contents of all fish were converted to weights representative of the standardized median weight of fish collected over the 24 h survey (0.85 g dry weight or approximately 4.75 g wet weight which represents 7.0 cm total length). For each group of fish the means and 95 % confidence intervals of weights of gut contents for each two hour period were calculated. These data were then used to calculate the rate of change in gut content weights (AG), the faecal production rate (FPR) and the rate of food ingestion (IR).

Where: $$\Delta G_{(t)} = \frac{G_{(t+2)} - G_{(t)}}{2}$$

and, $$FPR_{(t)} = \frac{G_{(t)} - GH_{(t)}}{3}$$

and, $$IR_{(t)} = \Delta G_{(t)} + FPR_{(t)}$$

Where: $G_{(t)}$ and $G_{(t+2)}$ are mean weights of gut contents at selected time (t) and two hours later

$GH_{(t)}$ mean weight of gut contents of fish caught at time (t) but held without food for 3 h

(t) all rates are expressed per hour or at that time (t)

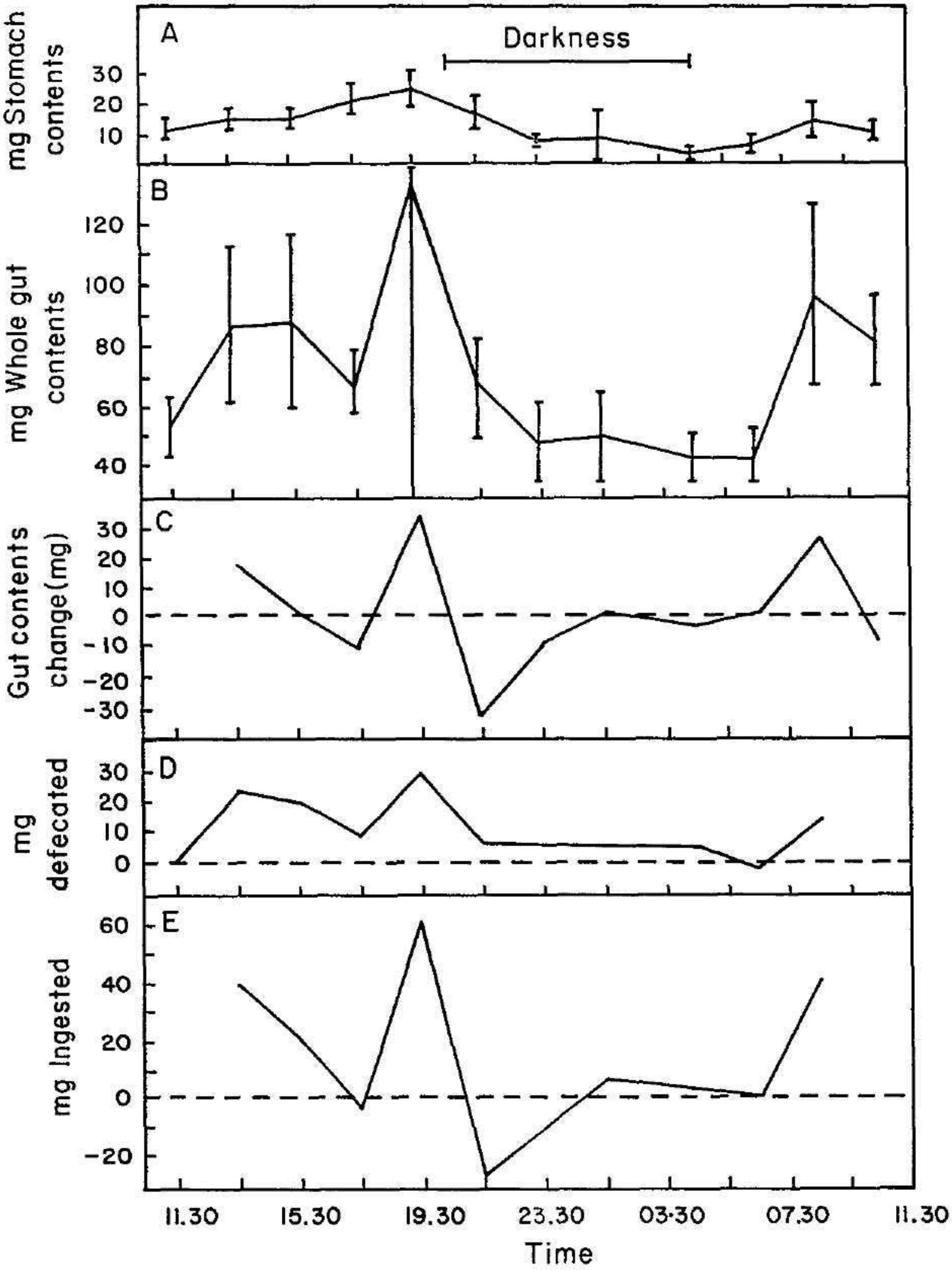


Figure 5.82. The diurnal feeding activity of *O. mossambicus* 9 - 10 Feb. 1984. (a) Mean and 95% confidence intervals for standardized weights of stomach contents of fish. (b) Same for total gut contents. (c) Change in gut content mass. (d) Mean weights of faeces deposited (e) Mean weights of food ingested. (c) (d) & (e) expressed as mg $h^{-1}$. Bar = 95% confidence limits.

The highest feeding activity during a 24 h period, based on stomach fullness and ingestion rates (Fig. 5.82), was between 06h30 and 08h30 in the morning and between 17h30 and 19h30 in the evening. Feeding remained high throughout the daylight hours. The rate of faecal production was also higher during the daylight hours with the highest rate occuring between 17h30 and 19h30 (Fig. 5.82).

The calculated ingestion rates (IR) are the sum of two variables AG and FPR which were calculated from highly variable data (Fig. 5.82). This would indicate that there would frequently be no statistical differences between many of the estimated ingestion rates. The general trends in feeding rates seen in this study as well as those found by Bowen (1976) would however, indicate that fish feed at varying rates over a 24 h period. Bowen (1976) found that higher feeding activity occurs during daylight and also when calm windless conditions prevailed.

The 24 h ingestion rate of *0. mossambioue* juveniles was estimated as the mean hourly ingestion rate multiplied by 24. This gave an ingestion estimate of 453 mg food (g fish) d (both fish and food expressed as dry weights). This estimate was higher than values calculated from data produced by other workers (Moriarty & Moriarty 1973a, Bowen 1976, Caulton 1982) which ranged from 65-272 mg (g fish)"$^1$ d"$^1$. Caulton (1978) found that an increase in water temperature from 18 to 30 °C led to an increase in the ingestion rate from 540 to 1 800 mg of dry plant matter in 50 g *Tilapia rendalli.* Food low in nutritional value is ingested at a slower rate than food of an intermediate nutritional value (Taghon & Jumars 1984). The ingestion rate however, was found to decrease once the nutritional value of food became very high (Jobling 1981). An abundance of food of high nutritional value leads to an increased ingestion rate but a decrease in the assimilation efficiency of the food (Pandian & Raghuraman 1972). It would thus appear that either the food matter ingested was of intermediate nutritional value, or present in abundance. Both these factors would have led to a high rate of ingestion.

An estimate of daily faecal production was determined from the mean hourly ingestion rates and this came to 314 mg dry matter per gram of fish per day. Excretion rates of urine were calculated as ue P expressed as SRP per e of fish (Table 5.39).

**Table 5.39.** **Calculated excretion rates of SRP in urine by known weights of *O. mossambicus* from Hartbeespoort Dam for 1983.**

| Date | No fish | Total wt(g) | Temp °C | Rate of Urine produced $\mu g\ g^{-1} d^{-1}$ |
|---|---|---|---|---|
| 5-12 Aug. | 10 | 181.63 | 20-25 | 15.12 |
| 29 Aug. - 2 Sept. | 10 | 191.20 | 22-25 | 16.08 |
| 11-21 Nov. | 10 | 239.54 | 27 | 45.12 |

# 6. ECOSYSTEM INTERACTIONS AND MODEL STRUCTURE

## 6.1 INTRODUCTION

The results presented in the previous sections range over lake physics, lake chemistry and lake biology. These results have been inter-related by building a model of the functioning of Hartbeespoort Dam. The process modelled is the flow of phosphorus through the ecosystem. The model attempts to simulate the quantitative flow of phosphorus between various components of the ecosystem and eventually out of the system. The inflow of phosphorus to the system is one of the input variables which drive the model. There are 15 other input variables which are mostly properties of river hydrology, impoundment volume and climate.

The model runs by taking values of the input variables and using them to calculate new values of state variables (the output of the model). State variables are mainly concentrations of inorganic phosphorus and abundance of various biological functional groups. The calculations within the model are based on the rate relationships measured during the research programme.

The model thus represents a summary of everything that has been studied and quantified about the functioning of the ecosystem. In order to test the representativeness of the model its output is compared with data measured in the dam. The quality of the model as a simulator of events taking place in the dam is a measure of the level of quantitative understanding of phosphorus flow achieved.

The structure of the model reflects the important assumptions made at the commencement of the study. During the study it was decided that too little was known about the importance of the zoobenthos to warrant its inclusion. The development of weed beds in Hartbeespoort Dam is limited and there were no records of extensive growths of benthic algae, so these components were omitted. It was assumed that fringing emergent aquatic vegetation was unimportant in the overall functioning of the ecosystem and that piscivorous birds could be ignored.

Theoretically the model should be transferable to other impoundments in southern Africa. However, at this stage the data requirements are large. It is recommended that no attempts to transfer the model be made until the model structure has been examined by sensitivity analysis.

## 6.2 STRUCTURE OF THE HARTBEESPOORT DAM ECOSYSTEM MODEL

### Introduction

The assumptions built into the model which are not discussed in the text are shown in Appendix 6.1. The model is run by a FORTRAN V programme and a list of the terms and major rates used are given in Appendices 6.2 and 6.3.

The model runs on daily iterations but receives monthly input of hydrological and climatic parameters. Loading figures are divided by 30 to approximate daily loading rates and hydrological and climatic state variables are assumed to remain constant throughout the month.

The model is divided into thirteen sub-compartments (Fig. 6.1). The currency used in the model is phosphorus. Phosphorus exchanges between compartments are computed daily. At the end of all the exchange computations, the total daily change in each sub-compartment is calculated and added to the sub-compartment state variable. Output of the model consists of the major state variables at the end of each thirty day iteration.

The input data for model validation consists of mean monthly values for the three years from September 1980 to August 1983. During this period conditions were unusual due to a severe drought and lake volume fell from a peak of 100% to a minimum of less than 25% at the end of the three year period. For this reason the long-term predictions have been undertaken using only the 1980/81 data when conditions were more stable. These data are then read in repeatedly over each year for the number of years simulated. The implications of changes in the hydrological regime to model output are discussed in Section 6.3.

### Calibration

The version of the model presented in this report was completed in August 1984 and there has not been time to accurately calibrate it. During the construction of the model, which has occurred more or less continuously since the first version was produced in 1981, rough calibrations were undertaken for each compartment with each change in that compartment. However, a general calibration of the model has not yet been performed and is planned for 1985/86.

The precipitation of particulate phosphorus from epilimnion to hypolimnion, the resuspension of particulate phosphorus from the sediments and the solubilisation of particulate phosphorus all occur at rates derived by calibration using the 1980/81 observed data. The calibration was designed to maintain the concentration of dissolved phosphorus in the hypo- and epilimnions within the range observed during the year.

In the phytoplankton subcompartment, death rate was the only rate process for which no field measurements were available. This rate was calibrated roughly (set at either a high or low fixed value, depending on water temperature) to produce an output in which the phytoplankton do not disappear over a 50 year model run. The hyperscum sub-compartment was loosely calibrated on wind speeds to simulate seasonal occurrence. Hyperscum building and breaking rates are assumptions and it is probable that, while the time of occurrence of hyperscums in the model exceeds their actual occurrence, the model underestimates the mass of phytoplankton incorporated into hyperscums.

Zooplankton in the model were calibrated on the basis of mortality rate and algal assimilability to simulate the seasonal trend observed in 1980/81. Calibration to obtain the observed zooplankton masses during this period has not yet been undertaken.

The fish subcompartment was also calibrated on mortality rates to simulate the biomass measured in 1982 and the seasonal trends caused by recruitment, growth and mortality.

FROM

| TO | Dissolved P Inflow | Particulate P Inflow | Hypolimnetic Sediments | Epilimnetic Sediments | Hypolimnetic Particulate P | Epilimnetic Particulate P | Hypolimnetic Dissolved P | Epilimnetic Dissolved P | Phytoplankton | Phytoplankton Scum | Zooplankton | Fish | Outflow |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Dissolved P Inflow | | | | | | | | | | | | | |
| Particulate P Inflow | | | | | | | | | | | | | |
| Hypolimnetic Sediments | | | | ○ | ● | | ○ | | ○ | | ○ | ○ | |
| Epilimnetic Sediments | | | ○ | | | ● | | ○ | ○ | ● | ○ | ○ | |
| Hypolimnetic Particulate P | | | | | | ○ | | | | | | | |
| Epilimnetic Particulate P | | ● | | ● | ○ | | | | ○ | ● | ○ | ○ | |
| Hypolimnetic Dissolved P | | | ○ | | ● | | | ○ | | | | | |
| Epilimnetic Dissolved P | ○ | | | ○ | | ● | ○ | | ● | ● | ● | ● | |
| Phytoplankton | | | | | | | | ○ | | ○ | | | |
| Phytoplankton Scum | | | | | | | | | ○ | | | | |
| Zooplankton | | | | | | ○ | | | ○ | | | | |
| Fish | | | | ○ | | ○ | | | ○ | | ○ | | |
| Outflow | | | | | ○ | ○ | | ○ | ○ | ○ | ○ | | |

Figure 6.1. A connectivity matrix of the major sub-compartments in the Hartbeespoort Dam ecosystem model. Open circles indicate time-varying coefficients and closed circles indicate constant coefficients. System connectivity = 30.1%.

The thermal regime (Section 5.1.5)

In the model the lake is stratified for part of the year. An oxycline forms the barrier between an anaerobic hypolimnion and an aerobic epilimnion, in which oxygen is not considered to be limiting. Living phytoplankton, zooplankton and fish are not found in the hypolimnion.

Lake depth, and height of the oxycline from the bottom at the deepest point, are input variables and are assumed to remain constant throughout each month. From these parameters, using polynomials fitted to the hypsographic data, the total dam volume and hypolimnetic volume are calculated. The epilimnetic volume is then obtained by subtraction. A fourth volume, the volume of the epilimnion directly above the hypolimnion, PELVOL, is also calculated. The areas of hypolimnetic and total sediments are calculated from power relationships fitted to measured data and the area of epilimnetic sediments derived by difference.

At the end of each month, with change in oxycline depth, particulate and dissolved phosphorus are transferred between epilimnion and hypolimnion in proportion to the changes in volume. Similarly, sedimentary phosphorus is transferred between epilimnetic and hypolimnetic sediments in proportion to changes in area.

In the model processes occurring in the hypolimnion are temperature independent and only epilimnetic and surface temperatures are utilised.

Water-chemistry (Sections 5.2.1 and 5.2.5)

The interactions between dissolved and particulate phosphorus and the other sub-compartments are discussed under the appropriate sub-compartments .

There is an initial sedimentation of the particulate phosphorus in the inflow which is split between hypolimnion and epilimnion in direct proportion to their volumes. Sedimentation of particulate P in the lake also occurs and is similarly split between the two layers. There is a daily loss of 50% of the particulate P in the epilimnion directly above the hypolimnion (PELVOL) to the hypolimnion.

Chemical and bacterial decomposition of particulate P in the hypo- and epilimnion occur separately and at rates dependent on the particulate P concentration. Conversion of dissolved P to particulate P occurs through the biotic compartments and is dealt with there.

Sediments (Section 5.3)

The sediment sub-model is divided into two compartments; one relating to the exchange of particulate P and the other to soluble phosphate exchange. Particulate inputs are via sedimentation from the phytoplankton, zooplankton, fish and suspended particulates and particulate outputs are via resuspension and fish grazing.

When the lake is stratified the sediments are divided into epilimnetic (beneath the epilimnion) and hypolimnetic (beneath the hypolimnion) .

The soluble phosphate exchange between bottom sediments and the overlying water has been modelled on the basis of laboratory determined relationships between the phosphate concentration in the overlying water and the rate and direction of phosphate flux across the interface. No attempt is made to distinguish between biotic and abiotic processes. The pool of mobile sediment phosphorus, which has the potential to be released into the water, is assumed to be reduced by a factor of 0.027% per day to simulate geochemical immobilisation. This rate has not been adequately validated but is incorporated for improved simulations. Due to variability in the uptake/release relationships, the equilibrium phosphate concentration (i.e., the overlying water concentration at which neither uptake nor release will occur) is assumed to extend over a range of 20% of the previous years annual mean value for epilimnetic dissolved phosphorus. Hypolimnetic sediments are assumed to have a range of 80-200% of the annual mean to allow for greater dissolved phosphorus build up in hypolimnetic sediments.

Phytoplankton sub-model (Sections 5.4.6 to 5.4.14)

The phytoplankton populations are vertically stratified within the epilimnion so that their euphotic zone concentration is higher than their mean epilimnetic concentration. Within the euphotic zone, their spatial distribution is assumed to be homogeneous.

Chlorophyll *<x* concentration regulates the underwater light climate (Robarts & Zohary 1984). In return phytoplankton growth rate is regulated by the underwater light climate and temperature. Gross primary production rates are predicted from chlorophyll *a* concentration, temperature and light attenuation, using equations derived from *in situ* radiocarbon uptake experiments. Respiration losses are then subtracted and algal growth rates are computed from net production. As the algal standing stock increases, the euphotic depth decreases and algal growth becomes light-limited as a result of self-shading. Phytoplankton growth rate can be further modified by the epilimnetic dissolved phosphorus concentration.

Losses from the phytoplankton population are due to the following processes: grazing by zooplankton and fish, natural mortality (photo-oxidation at water surface, lysis and decomposition), sedimentation, outflow losses, and scum formation. Grazing losses are discussed in the respective sub-models. Natural mortality is determined as a function of temperature and standing stocks. Dead phytoplankton contribute both to the dissolved phosphorus compartment through mineralization and to particulate phosphorus by decomposition. Sedimentation losses from the phytoplankton are calculated from chlorophyll concentrations. Outflow losses are directly proportional to the volume of water leaving the lake and mean lake chlorophyll concentration.

Hyperscum builds when the mean monthly wind speed is lower than 2 m sec , and its rate of accumulation is inversely proportional to wind speed. Losses from the hyperscum are due to sedimentation, decomposition and photo-oxidation at the surface. When wind speed is greater than 2 m sec , the hyperscum breaks and 50% of its biomass returns to the phytoplankton standing stock. The rest is assumed to be non-viable colonies which contribute to particulate phosphorus.

### Zooplankton sub-model (Section 5.4.15)

Phosphorus input into the zooplankton is governed by the rate of ingestion of phytoplankton and suspended particulates. Losses of phosphorus are by egestion, excretion, ecdysis, natural mortality, chaoborid fly-off and outflow from the lake. Regular *in situ* experiments determined the rate at which a known biomass of zooplankton feed, and so biomass specific grazing rates during periods of dominance by various herbivorous cladocerans throughout the year are calculated. These community grazing rates are modified by food availability (proportion of food present as phytoplankton or suspended particulates). Utilization of phytoplankton is further modified by seasonal changes in the feeding efficiency of zooplankton on phytoplankton, based on monthly changes in the proportions of green or blue-green algae present (Robarts & Zohary 1984).

Loss of phosphorus by ecdysis is dependent on biomass, and losses by excretion and egestion depend on biomass and rate of nutrient uptake. Losses by mortality and chaoborid fly-off are dependent on biomass and temperature, both losses being highest at high temperatures when generation times are shortest and adult chaoborids emerge. Fish predation of zooplankton is controlled in the fish sub-model by fish population growth, and zooplankton lost from the lake is dependent on biomass and outflow hydrology.

### Structure of the fish sub-model (Sections 5.4.17 and 5.4.18)

The fish sub-model is driven by the population dynamics of the three species included: *Oreoehromis mossambious*$_a$ *Cyprinus aarpio* and *Clarias gaviepinus.* Because of the much slower turn-over time in the fish species, compared to other components of the model, and the ability of fish to adjust their diet according to food availability, it has not been possible to link fish production to production of other biotic components. The impact of environmental conditions, apart from phosphorus, on fish cannot be assessed using this model. Fish are included in the model to assess and simulate the effect of the fish community on the rest of the system. However, the population size is linked to phosphorus concentrations according to the relationship described by Hanson and Leggett (1982). The mean annual fish standing stock will therefore remain constant under constant phosphorus concentrations.

Actual mortality rates measured in the field were subject to errors and these errors became magnified in the conversion of annual to daily rates. It was necessary to adjust these rates to achieve population stability from year to year under constant phosphorus concentrations.

Each species is divided into 0 +■ fish (young of the year) and adults. Growth in adults is determined by calculating daily growth increments from the calculated relationships for each species between initial length and growth increment. In 0 + fish growth in mass is assumed to be linear. Growth in *0. mossambious* is temperature dependent.

Spawning of *0. mossambicus* occurs in"December and January while the other two species spawn in November. The sex ratio is 1:1 for all species. Recruitment to the 0 + class occurs on a daily basis over the spawning period and is calculated from the product of the mean individual fecundity and the number of females. Each *0. mossamb'Ccus* female spawns three times a year while the other two species spawn once. In *C. oarpio* and *C. gariepinus* a proportion of the eggs are assumed to die instantly while, in the mouth-breeding *0. mossamb-ieus*, this does not occur. Natural mortality rates in the first year of life are considerably higher than in subsequent years. The natural mortality rate of *C. carpio,* the fastest growing species, drops in its first year when the fish reach a mass of 150 g and when the fish also becomes subject to fishing mortality.

Ingestion is determined from gross population growth of each species and assimilation efficiency is taken to be a constant 0.4, on a daily basis. *0. mossambicus* feeds on phytoplankton, zooplankton, particulate and sedimentary phosphorus, the relative proportions of each component varying according to seasonal abundance of the different size classes, which utilize different food sources. *C. carpi-o* feeds exclusively on sedimentary phosphorus and *C. gaviepvnus* on sedimentary phosphorus and zooplankton in equal proportions. There is no seasonal variation in food types utilized by the latter two species. Sedimentary phosphorus in the model include zoobenthos, plant material and detritus. Algae includes *Miorooystis* and filamentous algae.

Excretion rate is a constant 0.03 of body weight for all species and 0.00004 of urine is phosphorus.

## 6.3 MODEL VALIDATION

Rigorous sensitivity analyses and validation of model output are planned for Phase 2 of the Hartbeespoort Dam Ecosystem study. The validation presented in this section is a preliminary comparison between predicted and observed parameters.

### Soluble phosphorus

Three complete years of limnological and hydrological data covering the period September 1980 to August 1983 were available for model validation. The model was run over three years, using the hydrological and climatic input data measured over this period, and the predictions of selected state variables were compared to the actual values measured in the lake. Over the study period lake conditions altered drastically as a result of the drought. This is reflected in Figure 5.6 which shows a drop in dam volume from a peak of 100% to less than 25%.

The model output of epilimnetic phosphorus concentration compares favourably with the observed general trend (Fig. 6.2). The predicted concentration shows an increase over the three years due to the increased contribution of effluent to total river flow over the drought period. The model output did not predict accurately the month to month fluctuations observed in the lake. This weakness is due to incomplete knowledge of sediment:water interactions and the mechanism governing determination of equilibrium conditions.

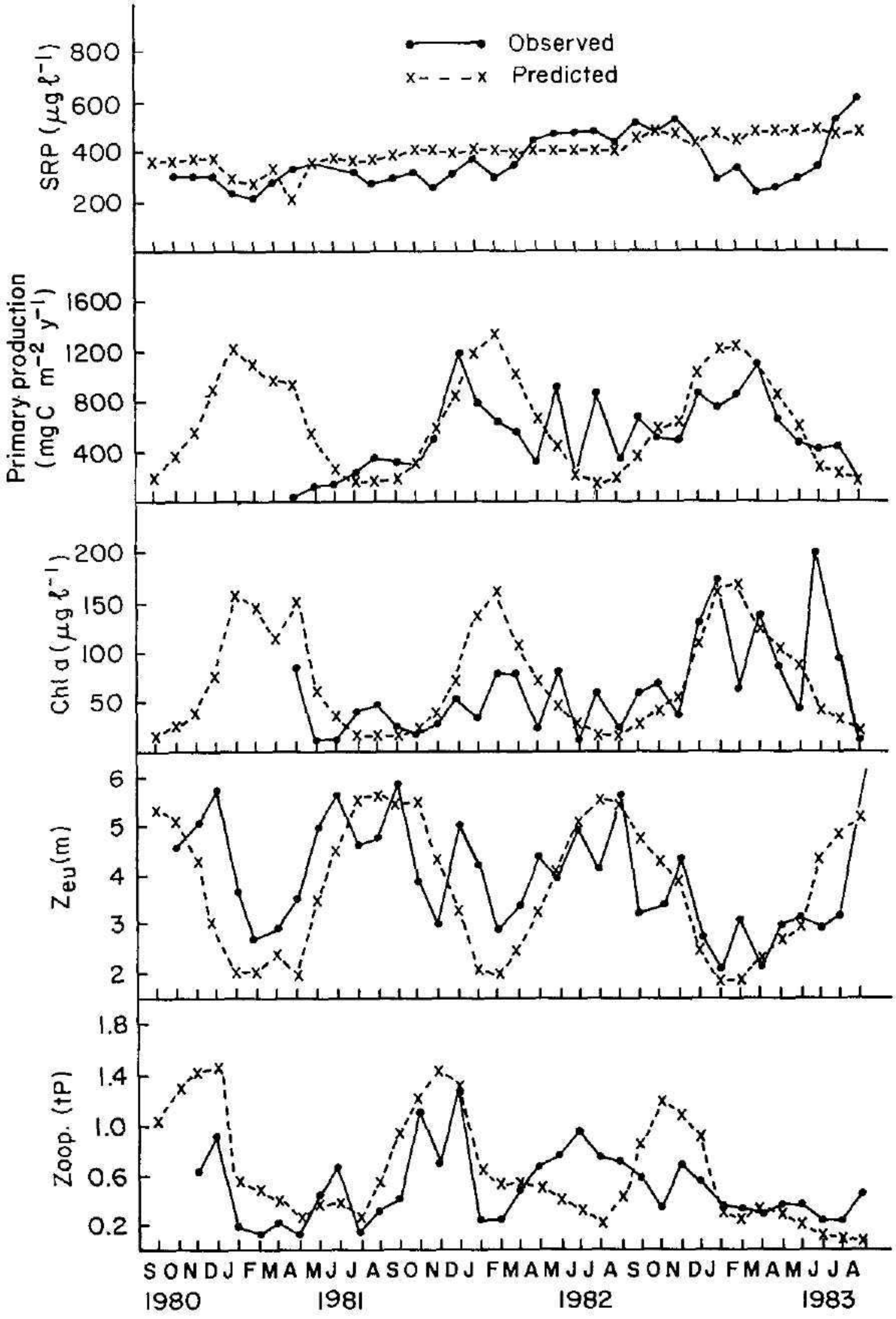


Figure 6.2. Output from the model compared to observed values in the lake (September 1980 to August 1983).
SRP = mean epilimnetic soluble reactive phosphorus concentration; Chl α = mean euphotic zone chlorophyll α concentration; $Z_{eu}$ = mean euphotic zone depth; Zoop = lake total zooplankton as tonnes phosphorus.

## Phytoplankton

Measured and simulated values of phytoplankton primary production, of the euphotic zone depth and of chlorophyll concentration within this zone are shown in Figure 6.2. In "general, a good agreement was obtained between model output and the measured values. The range of the values predicted for each of the parameters, as well as their seasonal cycles, compared closely with observed values. Measured values were more erratic than model predictions. This is to be expected as model output values are representative lake-mean values. In contrast, measured values were monthly means of weekly measurements from the main station only. In Section 5.4.7 we have demonstrated that week to week variations in chlorophyll concentrations at one sampling site can be greater than the seasonal changes due to horizontal migration of buoyant *Mvcvocystis* with surface currents. Both primary production rates and the underwater light climate in Hartbeespoort Dam were highly dependent on the fluctuating chlorophyll concentrations (Section 5.4.19). It is suggested that model output produces 'lake mean' values of chlorophyll concentration that are more representative than actual measurements at any one station.

## Zooplankton

The observed pattern of seasonal fluctuations in zooplankton standing stock from November 1980 to March 1982 is well reflected by the model output. Except during May and June 1981, model output overestimates zooplankton biomass (Fig. 6.2).

As stated in Section 5.4.15 zooplankton biomass and chlorophyll concentration were negatively correlated in the lake during 1981 and 1983, but no correlation was observed in 1982. This lack of association between zooplankton and its phytoplankton food resource, observed primarily during the winter of 1982, plus the poor agreement between observed and predicted primary production and chlorophyll *a* concentration over the same period, are the cause of the model[1] s failure to predict the high levels of zooplankton biomass present in the lake from April to September 1982. In 1983, when zooplankton biomass and chlorophyll *a* were again negatively correlated, predicted zooplankton biomass again approached observed values.

## Fish

As fish are calibrated to remain constant under constant mean annual phosphorus concentrations, validation of the fish sub-compartment was not undertaken.

In conclusion, the model provides realistic simulations of measured trends in the major sub-compartments (Fig. 6.2). The overall tendency is for the model to under-predict total phosphorus and chlorophyll (Table 7.1). This bias should be significantly reduced after the planned calibration of the full model.

# 7. MANAGEMENT

## 7.1 INTRODUCTION

One of the objectives of managing Hartbeespoort Dam is to reduce the size of the phytoplankton standing crop and to modify the phytoplankton species composition so that blue-green algae are no longer dominant. Achievement of this objective would mean that aesthetically undesirable hyperscums and the worrying occurrence of toxic *Microcystis* in a raw-water resource would no longer occur. Furthermore, water treatment costs would be lowered and taste and odour problems should diminish.

Management of the biological properties of a reservoir may be undertaken by modification of the chemistry or the physical properties of the water body or by intervention in its biological processes through introducing new species or harvesting selected components of the biota. In this section the consequences of phosphorus load reduction, pre-impoundment, artificial mixing, including aeration and destratification and biological manipulations are considered, using both the ecosystem model and other information. Cost estimates for these management options are beyond the scope of this report.

In this chapter it is necessary to review the projections of the ecosystem model against general limnological knowledge for two important reasons. The first is that the model is not built to predict species successions and replacements which may take place with changes in ecological conditions. The second reason is that care in interpreting model output is necessary when driving variables are set at levels far beyond the range present in the data on which the model was built (Section 6.3). In this respect, it is important that the data base on which the model was built and its extension, against which state variable values were assessed, does not include a high rainfall year or a low phosphorus input.

Among the problems identified by users of the dam as given in Table 3.4 (Section 3.7) is the health hazard of bilharzia. The reality of this problem requires confirmation, since annual medical examination of the Hartbeespoort Dam limnological research team (up to sixteen people) has failed to reveal a single case of bilharzia after four years. The remaining user identified problems can all be related back to the abundance of phytoplankton and the dominant phytoplankton species, the management of both of which is considered in this section.

Mention should be made of areas of concern regarding the quality of Hartbeespoort Dam water which are not addressed in this study, but which may be associated with eutrophication. The first is the chloride content of the water, which is not affected to any marked degree by biological or chemical processes in the dam. Chloride concentrations always increase after domestic and certain industrial uses of water. The second is the dissolved organic carbon content of the water and in particular the abundance of trihalogenated carbon precursors. Since the 1984 announcement that Hartbeespoort Dam is to be regarded as the raw water source for the new Magalies Water Board, these properties of the water have assumed considerable importance in relation to water purification. The whole organic carbon cycle did not receive attention during the studies reported here, so that management recommendations on these aspects cannot be made. Further potential management problems are related to the probable presence of mucopolysaccharides in the water (Hayes and Greene 1984). These problems are addressed in recommendations for further research (Section 8).

## 7.2 PHOSPHORUS LOAD MODIFICATION

### 7.2.1 Introduction and assumptions

Phosphorus load modification is the most common method of eutrophication management employed (Uttormark & Hutchins 1980). This is largely due to the more controllable nature of phosphorus when compared with nitrogen and other driving forces (Toerien 1975). As a result of changes in the water pollution control legislation in South Africa (RSA 1980), wastewater treatment works and other effluent producers in the Hartbeespoort Dam catchment must conform to a 1 mg *9* dissolved orthophosphate-phosphorus standard by August 1985. The effects of this standard are discussed in paragraph 7.2.4 below. In addition to evaluating the potential effects of this single standard, this section examines the effects of a range of other phosphorus load reductions using the NIWR ecosystem model (DEFMOD) and a selection of the more commonly used empirical (statistical) models. The conditions examined range from complete elimination of point-source phosphorus discharges to the current point-source discharge load.

A number of assumptions have been made to ensure consistent handling of the data used in the various models to generate predictions. These are:

(a) the 1980-81 data set was used as the baseline. Hydro logical conditions during this period were 'normal' when assessed against long-term environmental trends. The impact of hydrological variability is discussed in Section 7.2.4.

(b) load is as measured at weirs A2M12 and A2M13 on the two major rivers which flow into Hartbeespoort Dam (Fig. 3.3).

This discussion deals only with the limnological effects of various treatment methods resulting from changes in P-load and does not consider the engineering or economic implications of various forms of P-load modification.

### 7.2.2 The models used and comparison of their outputs

Many models are available to predict the trophic status of impoundments from their nutrient loading and hydraulic characteristics. In this section the output of the Hartbeespoort Dam ecosystem model, DEFMOD (Chapter 6) is compared with outputs of the OECD suite of models (OECD 1982), of the Walmsley-Butty models (Walmsley & Butty 1980) of the Reckhow general model (Reckhow 1979; subsequently referred to as the Reckhow model) and of the Grobler-Silberbauer model (Grobler & Silberbauer 1984).

The Organisation for Economic Cooperation and Development (OECD) eutrophication models are described in detail by the OECD (1982). The phosphorus and chlorophyll relationships have been validated using a southern African data set by Thornton & Walmsley (1982) and Walmsley & Thornton (1984). Jones & Lee (1984) also demonstrated the applicability of the USA-OECD chlorophyll model using a South African data set. It should be pointed out that the OECD (1982) models relate <u>annual</u> mean chlorophyll concentration to annual phosphorus loads, but the USA OECD model relates summer mean chlorophyll concentration to annual phosphorus loads. The OECD (1982) recognised that the Vollenweider (1968) model relating P-concentration to flushing over-predicted in-lake P-concentrations. Thornton & Walmsley (1982) also suggested that the Vollenweider model over-predicted in-lake P-concentrations in southern African lakes. The OECD amended the original Vollenweider model to reduce this prediction error. This correction is applied to the model used in this section. The models used in this report are those derived from the combined OECD data base (OECD, 1982), which state

$$[P] = 1.55\ [L_{Tp}/q_s\ (1 + T_w{}^{\circ\text{-}5})]^{0\text{-}82} \qquad [1]$$

$$[CHLjE = 0.37\ [L_{Tp}/q_s\ (1 + T_w{}^{\circ\text{-}5})]^{0\text{-}79} \qquad f_2]$$

where [p] = annual mean in-lake_total phosphorus concentration (mg m $^3$)

[CHL]E = annual mean euphotic zone chlorophyll concentration (mg m $^3$)

L - areal total phosphorus loading rate $^{TP}$ (g m~$^a$ y"$^1$)

q = areal water loading rate ( m y )

$T_W$ = water residence time (y)

The Walmsley-Butty models relating chlorophyll concentration to areal phosphorus loading were derived using a totally South African data set consisting of 21 man-made lakes (Walmsley & Butty 1980). The models state:

$$[CHL]5 = 0.84\ L_{Tp} + 2.97 \qquad [3]$$

$$[CHL]5 = 1.62\ L_{Qp} + 3.80$$

where [CHL]5 = annual mean chlorophyll concentration (mg m $^3$) in the surface waters sampled with a 5 m hose pipe.

wp = areal total phosphorus loading rate (g m $^2$ y )

= areal orthophosphate loading rate (g m $^2$ y"$^1$) .

Reckhow (1979) proposed a general phosphorus-loading relationship for lakes, which states:

$$[P] = L / 11.6 + 1.2\ q \qquad [4]$$

where: [P] = annual mean in-lake phosphorus concentration (mg m $^3$)

L = areal phosphorus loading rate (g m $^2$ y *) . q

= areal water loading rate (my$^1$) .

Grobler & Silberbauer (1984) developed a modification of the basic OECD model in which an impoundment specific sedimentation rate, s, was substituted for $T_w$ °«$^5$ . The Grobler-Silberbauer model states:-

$$[P] = W/(Q + s.V) \qquad [5]$$

where: |"P] = annual mean in-lake total phosphorus concentration (mg m $^3$)

W = total annual total phosphorus load (kg)

Q = total annual water inflow ($10^6$ m$^3$)

V = Annual mean volume of dam (10° m$^3$)

s = phosphorus sedimentation coefficient.

Grobler & Silberbauer (1984) calculated s to be 2.9 for Hartbeespoort Dam, but subsequently Grobler (personal communication) has corrected this value to 3.5, the value used here. Grobler and Silberbauer used their predicted phosphorus concentrations with Rast, Jones & Lee's (1983) model for summer mean chlorophyll concentration to predict mean annual chlorophyll concentration. The formula is

$$[CHL] = 0.45\ [P]^{\circ * 79} \qquad [6]$$

where [CHL] = mean annual chlorophyll, unspecified as to depth.

This prediction of chlorophyll assumes that there is no difference between mean annual chlorophyll and mean summer chlorophyll in South African impoundments.

Measured values of the annual mean in-lake phosphorus [PI and chlorophyll [CHL] concentrations are compared with values predicted by the described models in Table 7.1. The Reckhow model over-estimated [P], while the other models underestimated [P]. There was no substantial difference between the mean deviations of [P] predictions of DEFMOD, the OECD model and the Grobler-Silberbauer model, whose mean deviations from the observed were -26, -27 and -25% respectively. From Table 7.1 it may be seen that the range of the individual percentage deviations for DEFMOD was considerably lower than for the remaining models. The mean percentage deviation of the Reckhow model predictions was +32%. It is concluded that these data do not provide grounds for selecting any of these four models as providing better predictions of [P] than the others.

Table 7.1. Observed characteristics of Hartbeespoort Dam and predicted values of the phosphorus and chlorophyll content arising from various models (see text). Deviations of predicted values from observed are shown as percentages of the observed in parenthesis after the predicted values.

| | 1980/81 | 1981/82 | 1982/83 |
|---|---|---|---|
| Observed Values | | | |
| V $10^6 m^3$ | 184.2 | 170.4 | 85.0 |
| Q $10^6 m^3$ | 245.6 | 157.8 | 98.8 |
| W t | 283 | 323 | 206 |
| $L_{TP}$ g $m^{-2}$ $y^{-1}$ | 14.5 | 17.8 | 20.0 |
| $L_{OP}$ g $m^{-2}$ $y^{-1}$ | 12.6 | 15.2 | 17.3 |
| $q_s$ m $y^{-1}$ | 12.67 | 8.70 | 9.53 |
| $T_w$ y | 0.75 | 1.08 | 0.86 |
| [P] mg $m^{-3}$ | 493 | 543 | 634 |
| [CHL]5 mg $m^{-3}$* | 31 | 35 | 59 |
| 95% C.L.** of mean | 25-38 | 26-44 | 40-78 |
| [CHL]E mg $m^{-3}$ | 40 | 44 | 94 |
| 95% C.L.** of mean | 30-50 | 34-61 | 60-128 |
| Predicted Values | | | |
| DEFMOD | | | |
| [P] mg $m^{-3}$ | 347(-30) | 412(-24) | 481(-24) |
| [CHL]5 mg $m^{-3}$ | 29(-7) | 24(-31) | 32(-45) |
| OECD | | | |
| [P] mg $m^{-3}$ | 299(-39) | 448(-17) | 479(-24) |
| [CHL]E mg $m^{-3}$ | 59(+47) | 87(+97) | 93(-1) |
| WALMSLEY-BUTTY | | | |
| [CHL]5 mg $m^{-3}$ from $L_{TP}$ | 15(-52) | 18(-49) | 20(-66) |
| [CHL]5 mg $m^{-3}$ from $L_{OP}$ | 24(-23) | 28(-20) | 32(-46) |
| RECKHOW | | | |
| [P] mg $m^{-3}$ | 541(+10) | 807(+49) | 868(+37) |
| GROBLER-SILBERBAUER | | | |
| [P] mg $m^{-3}$ | 318(-35) | 428(-22) | 520(-18) |
| [CHL] | 43 | 54 | 63 |
| Deviation from [CHL]5 | (+39) | (+54) | (+7) |
| Deviation from [CHL]E | (+7) | (+23) | (-33) |

V, Q, W, etc. are defined in the text.

*Observed [CHL]5, a volume weighted mean calculated from chlorophyll concentrations measured at discrete depths of 0, 0.5, 1, 2, 3, 4, and 5 metres.

**C.L. is confidence limits.

The Walmsley-Butty model which predicted [CHL] from L was better than the Walmsley and Butty model which used L . The latter model need not be considered further, DEFMOD and the Walmsley-Butty L_ model underestimated [CHL]5 by almost identical mean deviations (DEFMOD -28% and Walmsley-Butty -30%). The deviations of predicted chlorophyll values arising from the OECD models and the Grobler-Silberbauer models were particularly variable, but each included predictions in a single year which were highly accurate. The mean deviation of the OECD model prediction was 48% and for the Grobler-Silberbauer model was 33 or 22% depending on which observed chlorophyll value it is compared with. Judged in this fashion the Grobler-Silberbauer [CHL]E predictions were superior to the other predictions.

Another comparison between observed and predicted chlorophyll values may be based on the confidence limits of the mean observed chlorophyll values. In this comparison the Grobler-Silberbauer [CHL]E prediction was the only chlorophyll prediction which always fell within the 95% confidence limits of the observed mean. All other predictions, including the Grobler-Silberbauer [CHL]5* were within the 95% confidence limits of the observed means in only 1 of the 3 years. This suggests that the chlorophyll predicted by the Grobler-Silberbauer model is [CHL]E and it is treated as such in Tables 7.2 and 7.3.

### 7.2.3 The effect of the 1 mg *2* * phosphorus standard on the total phosphorus load on Hartbeespoort Dam

Total phosphorus load estimates for Hartbeespoort Dam in 1980-81 differ markedly depending on their method of calculation. Grobler & Silberbauer (1984), using measured point-source loads and diffuse source loads calculated from export coefficients, estimated an annual load of 539 tonnes (27.0 g $m^{-2}$ $y^{-1}$ ) in 1981. (This will be referred to as catchment load in the rest of the text.) From measurements of flow and total phosphorus concentration made daily at weir A2M12 on the Crocodile River and made weekly at weir A2M13 on the Magalies River, a load of 283 tonnes (14.5 g $m^{-2}$ $y^{-1}$) was estimated. (This will be referred to as weir impoundment load in the rest of the text.) This discrepancy suggests that in-stream losses of about 47.5% of the catchment load estimates occurred, or that one or both of the phosphorus load estimates is incorrect. Discrepancies between loads calculated from point source loads plus diffuse source loads and loads measured from flows and concentrations at river weirs can be due to several facts that are at present mostly unquantified. Firstly all three load estimates (diffuse source, point source and river weir) are subject to error in flow and concentration measurement. The error has two components - methodological imprecision and unrepresentativeness of samples, both of which apply to flows and concentrations. River systems are known to retain phosphorus, particularly at low flows (Hill 1982). In the catchment of Hartbeespoort Dam, Ashton 0 981) has shown that Rietvlei Dam retains up to 67% of the annual incoming total phosphorus load. There is therefore clear evidence that processes, which modify phosphorus loads, take place in rivers and impoundments. In comparing the two estimates of the total phosphorus load on Hartbeespoort Dam it is not possible to state which is the more accurate.

This discrepancy between catchment and weir impoundment_ load estimates makes it difficult to assess what impact the 1 mg *2* phosphate standard will have on the actual load to Hartbeespoort Dam. Grobler and Silberbauer (1984) estimated that by 1990 the 1 mg *2*~$^{l}$ standard will reduce catchment loads by approximately 75%, from 539 tonnes in 1980-81 to 130 tonnes in 1990. Assuming no in-stream losses 130 tonnes will only represent a 46% decrease in the 1980-81 load measured at the weirs. However, if the existing proportion of apparent in-stream phosphorus loss (47.5%) between the point source and the weir occurs following the implementation of the 1 mg *2* standard, load reductions of up to 75% of weir estimates can be expected.

In view of these uncertainties it was necessary to assess the possible changes to the phosphorus load to the dam after the introduction of the 1 mg *2* orthophosphate standard at the extremes of the minimum likely change and the maximum likely change. The maximum likely change assumes that after the phosphorus standard becomes effective the point source load will be reduced in the river in direct proportion to the reduction which took place before the phosphorus standard. This change would be a reduction of approximately 75% of the present annual weir/impoundment load of 283 tonnes to about 71 tonnes. The minimum likely change after introduction of the standard can be arrived at by assuming that all the point and diffuse source phosphorus reaches the dam. In this case the present annual weir/impoundment load of 283 tonnes drops to 130 tonnes, which for present purposes is taken as equivalent to a 50% reduction of the present phosphorus load.

### 7.2.4 The impact of the 1 mg *2* phosphate standard on the phosphorus and chlorophyll content of Hartbeespoort Dam

Predictions of the annual mean concentrations of total phosphorus and epilimnetic chlorophyll using the models compared in 7.2.2 are shown in Table 7.2. Total phosphorus loads of 283 tonnes (the 1980/81 load at weir A2M12 + A2M13), 142 tonnes (50% of the 1980/81 load) and 71 tonnes (25% of the 1980/81 load) are used in the predictions following the least and greatest estimates of post-standard loads developed in Section 7.2.3. The percentage of time that severe nuisance conditions of over 30 ug *2* chlorophyll would be expected, shown in Table 7.2, has been arrived at from Walmsley's (1984) relationship between mean annual chlorophyll and this percentage, which is

$$F = 1.19 \text{ (mean chlorophyll)} - 5.36$$

where F = the frequency of occurrence of severe nuisance conditions, expressed as a percentage of the year.

Examination of recorded weekly chlorophyll concentrations in Hartbeespoort Dam substantiated this relationship. There are methods for predicting maximum chlorophyll concentrations from mean annual concentrations. These are cited by Grobler & Silberbauer (1984). Testing these methods against the three available annual means and maxima for Hartbeespoort Dam showed that they gave very inaccurate estimates of maximum chlorophyll. Maximum chlorophyll predictions have therefore not been presented here.

Being a dynamic model, DEFM0D reflects the fact that the Hartbeespoort Dam ecosystem is not in equilibrium with the external load of phosphorus, which varies from year to year with a long-term upward trend.

When DEFMOD is allowed to iterate at the 1980/81 phosphorus load, [P] rises from 347 ug £ to reach an asymptote of 649 yg *2* , as discussed below (Fig. 7.1). DEFMOD predictions of [P] and fCHLl in Table 7.1 are instantaneous values, while those given in Table 7.2 are asymptote values.

The rise in the soluble reactive phosphorus concentration with time shown in Figure 7.1 for loads in excess of 0.25 of the present load is due to the gradual saturation of the phosphate binding capacity of the sediments of Hartbeespoort Dam. As the sediments lose their capacity to absorb phosphorus so the dissolved phosphorus content of the water increases. In this scenario the phosphorus binding capacity of sediment which is continually being carried into the dam is ignored. Grounds for ignoring this new sediment are that the phosphorus binding capacity of that major part of the sediment load which is carried down the Crocodile River is satisfied while it is in the river bed, accounting in part for the observed phosphorus loss in the river (Section 7.2.3).

Values of [P] and [CHL] predicted using the various models (Table 7.2) reflect the substantial difference between DEFMOD, which does not assume a phosphorus-limited algal growth and a general steady state, and the other models which do make these assumptions. Thus a 50% reduction in the external phosphorus load results in a

Table 7.2. Hartbeespoort Dam predicted in-lake annual mean total phosphorus [P], chlorophyll concentrations [CHL] and frequencies, as percentages of the year in which I" CHL] will exceed[F] 30 ug/£ $^{1}$ using various models and total phosphorus loads of 283 (A), 142(B) and 71(C) tonnes per annum. Hydrological conditions assumed to be similar to those in 1980/81.

| Model | [P] mg m$^{-3}$ | [CHL] mg m$^{-3}$ | F% |
|---|---|---|---|
| (A) 283 tonnes | | | |
| DEFMOD | 649 | 27 | 27 |
| OECD | 299 | 59 | 64 |
| WALMSLEY-BUTTY* | - | 24 | 23 |
| RECKHOW | 541 | - | - |
| GROBLER-SILBERBAUER | 318 | - | 46 |
| (B) 142 tonnes | | | |
| DEFMOD | 618 | 27 | 27 |
| OECD | 170 | 34 | 35 |
| WALMSLEY-BUTTY* | - | 14 | 11 |
| RECKHOW | 270 | - | - |
| GROBLER-SILBERBAUER | 159 | 25 | 24 |
| (C) 71 tonnes | | | |
| DEFMOD | 103 | 23 | 22 |
| OECD | 96 | 20 | 18 |
| WALMSLEY-BUTTY* | - | 9 | 5 |
| RECKHOW | 135 | - | - |
| GROBLER-SILBERBAUER | 80 | 14 | 11 |

*Predicted from orthophosphate loads of 246 (A), 123 (B) and 61.5 (C) tonnes

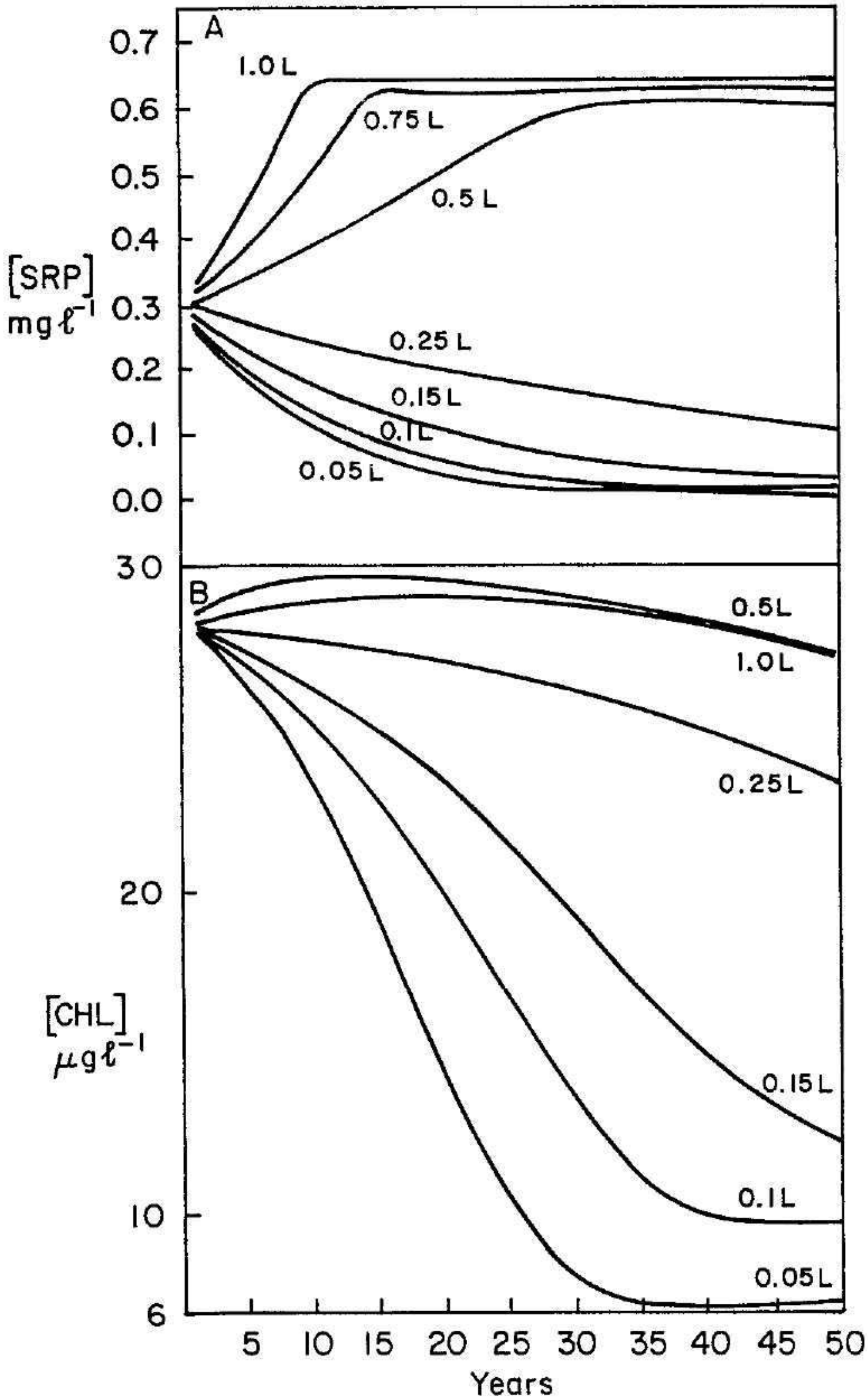


Figure 7.1. Predicted response to phosphorus load reduction, (A) of epilimnetic soluble reactive phosphorus concentrations (SRP) and (B) chlorophyll concentrations (Chl α) over 50 years. Numbers in Figure indicate proportion of 1980/81 load measured at the weir.

minimal reduction in [P] and [CHL] predicted by DEFMOD, but in substantial reductions in the concentrations predicted by the other models. At a load of 142 tonnes (50% present load) differences in predictions of [CHL] and F due to the various models are less than they are at higher loads. However, when the phosphorus load is reduced by 75% of the present load to 71 tonnes y there is again greater variability in predicted [CHL] and F.

In Table 7.2 DEFMOD stands out from the other models in that it predicts a lesser reduction in [CHL] and F in response to the 1 mg *2* $^{1}$ orthophosphate standard than do the other models. In the unlikely event of the phosphorus load on the dam dropping by only 50% (to 142 tonnes y $^{1}$) , Hartbeespoort Dam will remain highly eutrophic with three of the four predictions giving severe nuisance conditions more than 24% of the time. Interpretation of the consequence of the optimistic load forecast of 71 tonnes y must depend on the reader's preference in models. The OECD model and DEFMOD suggest that conditions will still be highly eutrophic with severe nuisance conditions about 20% of the time, while the Walmsley-Butty and Grobler-Silberbauer models suggest a considerable improvement with severe nuisance conditions only 5 or 11 % of the time. Since hyperscum formation is due not only *to* phytoplankton abundance but also to meteorological conditions, it is not possible to predict the extent and duration of hyperscums under the conditions of reduced phosphorus loading. However, if the phytoplankton abundance declines to levels predicted by the Walmsley-Butty or Grobler-Silberbauer models then it is reasonable to expect an observable decline in hyperscums.

The models can be used to estimate the level to which the total phosphorus load should be reduced in order to make a substantial impact on the trophic status of Hartbeespoort Dam. For this purpose the hydrological record for the 1980/81 year has been used. Predicted concentrations of [P] and [CHL] for a range of total phosphorus loads are shown in Table 7.3. [CHL] values predicted by DEFMOD and the OECD model for total phosphorus loads of less than 71 tonnes are similar, and greater than the Walmsley-Butty predictions. Following the OECD criteria, 8 mg m $^{3}$ [CHL] represents the boundary between eutrophic and mesotrophic conditions. To reduce the trophic status of Hartbeespoort Dam to this level, which would represent a dramatic reduction in the present phytoplankton abundance, would require that the total phosphorus load be reduced to about 15 to 40 t y . At low total phosphorus loads, the OECD and Reckhow models predict similar [P] concentrations, DEFMOD predicts much lower values and the Grobler-Silberbauer model is intermediate. It does seem likely that DEFMOD over-estimates [P] when the total phosphorus load is high and under-estimates [P] when the total phosphorus load is low.

Of all the models dealt with here only DEFMOD has the potential to reveal how long ecosystem properties will take to respond to a changed nutrient load (Fig. 7.1). Since Figure 7.1 was prepared by iterating the 1980/81 hydrological data, little reliability can be placed on the number of years required to reach asymptote values. In the natural and highly variable hydrological regime, abnormally high inflows which would flush out the dam would be expected. These would tend to reduce the recovery time. Sonzogni *et at* (1976) have shown that under steady state conditions the reduction of in-lake phospho-

Table 7.3. Predicted annual mean total phosphate [P], chlorophyll [CHL] and frequencies, as percentages of the year in which [CHL] will exceed 30 yg *i* $^{1}$ (F) using various total phosphate annual loads. Calculations made using 1980/81 hydrological regimes and fractions of the 1980/81 total phosphate load (L).

| | | | | | | |
|---|---|---|---|---|---|---|
| Total phosphorus load t $y^{-1}$ (L) | 283 | 142 | 71 | 42 | 28 | 14 |
| Fraction of L | 1.0 | 0.5 | 0.25 | 0.15 | 0.1 | 0.05 |
| **Predictions of [P] mg $m^{-3}$** | | | | | | |
| DEFMOD | 659 | 618 | 103 | 31 | 15 | 10 |
| OECD | 299 | 170 | 96 | 63 | 45 | 26 |
| Reckhow | 541 | 270 | 135 | 81 | 54 | 27 |
| Grobler-Silberbauer | 318 | 159 | 80 | 47 | 31 | 16 |
| **Predictions of [CHL] mg $m^{3}$** | | | | | | |
| DEFMOD | 27 | 27 | 23 | 12 | 10 | 7 |
| OECD | 59 | 34 | 20 | 14 | 10 | 6 |
| Walmsley-Butty* | 24 | 14 | 9 | 7 | 6 | 5 |
| Grobler-Silberbauer | 43 | 25 | 14 | 9 | 7 | 4 |
| **Predictions of (F) % of year** | | | | | | |
| DEFMOD | 27 | 27 | 22 | 9 | 7 | 3 |
| OECD | 65 | 35 | 18 | 11 | 7 | 2 |
| Walmsley-Butty | 23 | 11 | 5 | 3 | 2 | 1 |
| Grobler-Silberbauer | 46 | 24 | 11 | 5 | 3 | 0 |

*from orthophosphate load, where 1.0L = 246 tonnes y

rus concentrations followed reduced phosphorus load in direct proportion to the reduction in load. Thus a 50% reduction in load would be expected to result in a 50% reduction in lake concentration. The time taken to reach 95% of the expected change is estimated to be three phosphorus residence times, which in the case of Hartbeespoort Dam is equal to about a year. This estimate should be treated with some circumspection as Hartbeespoort Dam is not in a steady state, has a strong seasonality in inflow and a large internal phosphorus load. The response of lakes to an reduced nutrient load is influenced by the physical properties of the lake and the period of time over which it has been eutrophied. Thus the phytoplankton of Haley's pond (Bailey, Scott, Courtemanch & Dennis 1979) responded immediately to a reduction in external phosphorus load, but it had been eutrophied for only four years. At the other extreme Shagawa Lake (Larsen, van Sickle, Malueg & Smith 1979), in which the internal phosphorus load is important, showed no response to reduced nutrient load after six years. In 1979 Rast, Jones & Lee (1979) could find only ten water bodies for which there was adequate chlorophyll data before and after nutrient load reduction. Except for the two cases cited above the chlorophyll loads of the lakes took between two and four years to decline by about 50%. More recent comparisons such as those of Welch, Rock, Howe & Perkins (1980), Effler, Field, Meyer & Sze (1981) and Thornton & Nduku (1982) report response times of the phytoplankton of between two and a half and five years.

Because DEFMOD takes account of the internal phosphorus load in Hartbeespoort Dam it predicts a 35 year recovery time when the model is iterated using the 1980/81 hydrological data. Compared to the literature reported information presented above, this appears to be too long which probably reflects error in the assumed time of one year for equilibration between sedimentary and water-column dissolved phosphorus. While the duration of the recovery period would obviously be greatly influenced by the general hydrological regime over a period of years (a dry or wet phase of the 10 year cycle) a subjective guess is that it could take between 3 and 6 years to reach a new equilibrium level.

### 7.2.5 P-management through pre-impoundment

A simple, effective means of reducing phosphorus loading rates to polluted water bodies is the construction of 'pre-impoundments[1] upstream of the main inflows. The aim is to trap a proportion of the phosphorus load before it enters the main water body. This approach utilises the well-documented ability of water bodies to retain large proportions of the incoming phosphorus load (Kirchner & Dillon 1975) and retention efficiencies as high as 97% have been reported (Fiala & Vasata 1982; Stepanek 1980).

The Crocodile River Arm of Hartbeespoort Dam was used in a series of simulations to assess the potential role of a pre-impoundment in reducing phosphate loading to the impoundment. It must be stressed that the feasibility of this site from an economic or engineering viewpoint has not been considered here. The site was chosen because its morphometric features have been characterised and modelled (Twinch 1984) and because it is regarded as a suitable site at which general principles can be demonstrated.

Three pre-impoundment sizes were considered during the simulations. The smallest assumed a dam wall across the junction of the Crocodile River Arm and the Main Basin (Fig. 3.1) at a height sufficient to maintain existing full supply levels. The others assumed progressive 5 m increases in the height of the dam to a maximum of FSL + 10 m. The largest pre-impoundment had a volume of 26 x 10 m$^3$, a mean depth of 8.2 m and a surface area of 322 ha, based on the existing morphometry of the Crocodile River Arm. This corresponds to a pre-impoundment with a volume of about one tenth of the full supply volume of Hartbeespoort Dam.

Two approaches were used to estimate phosphate retention in the hypothetical pre-impoundments. Both made use of data from Section 5.3. for the period October 1980 to October 1983. Mean monthly values for Crocodile River flow rate and SRP concentration were derived from daily values and mean monthly water level in the impoundment was derived from weekly values.

The first approach was to use the Vollenweider/OECD nutrient budget model (without correction factor for in-lake concentration predictions) to predict phosphate concentrations in the pre-impoundments on a monthly basis -

$$P = L/qs\ (1 + T_w^{\circ *5})$$

where: P = predicted phosphate concentration (ing £ *))

L = areal phosphate loading rate (g m $^2$ m )

qs = areal water loading rate (m month )

$T_w$ = water residence time (months).

Assuming that inflow equals outflow, the phosphate load leaving the pre-impoundments was calculated as the product of predicted phosphate concentration and flow rate and the proportion retained was calculated by difference. This model is known to overestimate in-lake concentrations (OECD 1982) so the predicted retentions of phosphate in the pre-impoundments can safely be assumed to be conservative.

The second approach employed a more mechanistic model (RIVMOD) designed to predict phosphate losses in the Crocodile River Arm due to direct uptake by bottom sediments. Details of this model are given in Twinch (1984). Essentially the model uses experimentally determined sediment/water phosphate exchange characteristics (Section 5.3.3) to quantify phosphate flux between sediments and overlying water. Both the rate and direction of flux are dependent on phosphate concentrations in the water. Variable hydrological conditions in the Crocodile River and water depth in the impoundment, which interact to determine water residence time in the Crocodile River Arm, are taken into account in the model and pre-impoundment can be simulated by fixing water level to the required height.

The results of the simulations are summarised in Table 7.4. Although there were discrepancies between predictions using the two models, in general they were remarkably similar. With no pre-impoundment the average annual retention of phosphate in the Crocodile River Arm was 17-28% during the simulation period.

Pre-impoundment at FSL increased this to 31-37% and with increasing pre-impoundment size retention increased to a maximum of 60-63% at FSL + 10 m .

Both models used in the predictions are dependent on water residence time (T ) and this dependence makes it possible to estimate phosphate retention (Pret) in the pre-impoundments from water residence time using the following equations:

$$\text{Pret (Voll/OECD)} = (100 \times Tw^{\circ\prime 5})/(1 + T\ ^{\circ\prime 5})) \quad (r^2 = 0{,}99)$$
$$\text{Pret (RIVMOD)} = (100 \times Tw)/(] + Tw)\ ^{W} \quad (r^2 = 0{,}93)$$

Since the simulation was characterized by atypically low river flows, these equations were used to predict annual phosphate retention in the simulated pre-impoundments at mean, maximum and minimum annual flows in the Crocodile River calculated for the period 1972-82 (Table 7.5). These data indicate that over the full range of hydrological conditions in the Crocodile River, between 1972 and 1983,

TABLE 7.4. Measured loading (L, tonnes $y^{-1}$) to Hartbeespoort Dam during th three year simulation period and the predicted loading ($P_L$ following construction of various pre-impoundments. OECD = predictions using the Vollenweider/OECD nutrient budget mode and RIVMOD = predictions using the model of Twinch (1984). Annual flows also shown. % = percentage of measured loa retained.

| Pre-imp. volume | Year | L | OECD $P_L$ | OECD % | RIVMOD $P_L$ | RIVMOD % | FLOW $10^6 \times m^3$ |
|---|---|---|---|---|---|---|---|
| Variable | 1 | 245 | 164 | 33 | 196 | 20 | 234 |
| (no pre- | 2 | 287 | 190 | 34 | 217 | 24 | 149 |
| impoundment) | 3 | 182 | 150 | 18 | 170 | 7 | 101 |
| | x | | | 28 | | 17 | |
| 4.4 $m^3 x 10^6$ | 1 | 245 | 165 | 33 | 196 | 20 | 234 |
| (at FSL) | 2 | 287 | 180 | 37 | 197 | 31 | 149 |
| | 3 | 182 | 105 | 42 | 108 | 41 | 101 |
| | x | | | 37 | | 31 | |
| 12.8 $m^3 10^6$ | 1 | 245 | 134 | 45 | 155 | 37 | 234 |
| (FSL + 5 m) | 2 | 287 | 142 | 50 | 137 | 52 | 149 |
| | 3 | 182 | 81 | 56 | 65 | 64 | 101 |
| | x | | | 50 | | 51 | |
| 26.3 $m^3 x 10^6$ | 1 | 245 | 112 | 54 | 127 | 48 | 234 |
| (FSL + 10 m) | 2 | 287 | 117 | 59 | 100 | 65 | 149 |
| | 3 | 182 | 65 | 65 | 45 | 75 | 101 |
| | x | | | 60 | | 63 | |

Table 7.5. Predicted phosphate retention (%) in pre-impoundments of three sizes based on water retention times calculated from mean, min and maximum annual flows in the Crocodile River between 1972 and 1983. OECD = predictions using the Vollenweider/OECD nutrient budget model; RIVMOD = prediction using the model of Twinch (1984).

| Pre-impoundment | River flow ($10^6 \times m^3 y^{-1}$) Mean (194) | Min (37) | Max (404) |
|---|---|---|---|
| 4.4 $m^3 \times 10^6$ | | | |
| OECD | 34 | 43 | 27 |
| RIVMOD | 21 | 36 | 12 |
| 12.8 $m^3 \times 10^6$ | | | |
| OECD | 47 | 56 | 38 |
| RIVMOD | 44 | 63 | 28 |
| 26.3 $m^3 \times 10^6$ | | | |
| OECD | 56 | 65 | 47 |
| RIVMOD | 62 | 77 | 44 |

significant phosphate load reductions to Hartbeespoort Dam could be achieved by constructing a pre-impoundment and that the average retention efficiency could be as high as 56-62% in the largest of the pre-impoundments used. Even if FSL was maintained, 21-34% retention could be expected during years of average river flow. The magnitude of these conservative estimates is such that pre-impoundment must be regarded as a feasible supplementary eutrophication management option. Pre-impoundment alone will not overcome the existing eutrophication problems in Hartbeespoort Dam since reductions in excess of 75% of the current phosphorus loading will probably be required (Section 7.2).

By the year 2000 projected changes in the drainage basin will almost double the flow in the Crocodile River. Thus, pre-impoundment water residence time will decrease progressively in future with a concomitant reduction in phosphorus retention efficiency. A more careful analysis of the implications of this on the use of pre-impoundment as a management option is in progress.

### 7.2.6 Discussion of load modification

Two approaches to phosphorus load modification have been presented. The first is the legislated point-source control strategy involving a 1 mg *Z* $^{1}$ effluent phosphate standard. Numerous cases of effective lake rehabilitation through point source control have been recorded; notably Lake Washington (Edmondson & Lehman 1981), Shagawa Lake (Larsen *et at.* 1979), Lake Sammamish (Welch *et at.* 1980), the North American Great Lakes (Chapra, Wicke & Heidtke 1983), Wahnbach Reservoir and other German lakes (Benndorf, Uhlmann & Putz 1981), various Swedish lakes (Ryding 1981), and numerous Dutch lakes (Klapwijk 1981). Point source control is a lake management strategy that is easily administered and policed, and relatively easily implemented through the upgrading of sewage works. It is thus a popular and effective means of reducing eutrophication in lakes throughout the world. In southern Africa, point source control has been shown to be successful in controlling eutrophication in Lake Mcllwaine, Zimbabwe, a lake morphometrically similar to Hartbeespoort Dam (Thornton 1982). However, as Robarts (1985) points out, Lake Mcllwaine was never as highly enriched as Hartbeespoort Dam.

In drainage basins where diffuse source loading is important in maintaining eutrophic conditions in receiving waters, phosphorus load management is restricted to careful control of land use practices and optimization of the natural ability of aquatic and wetland ecosystems to retain some phosphorus in through-flowing water. Simulations of a pre-impoundment above Hartbeespoort Dam suggest that the existing annual phosphorus loading, measured at the weir, could be reduced by about 60% during average hydrological years by the construction of a pre-impoundment of 26 $m^3 \times 10^6$ (about 12% of the volume of Hartbeespoort Dam at FSL). The expected impoundment (Weir A2M12) load reduction based on the implementation of the 1 mg *Z* $^{1}$ standard is between 50% and 75% (Section 7.2.4), thus the load reductions obtained by pre-impoundment and by implementing the 1 mg *Z* could be similar. Since load reductions in excess of 80% will be required to prevent severe nuisance conditions in Hartbeespoort Dam (Grobler and Silberbauer 1984) neither pre-impoundment, nor nutrient removal, alone, will achieve the required level of load reduction.

On the basis of available information it seems that pre-impoundment (either singly or in series) could be an essential supplementary eutrophication management option for Hartbeespoort Dam if long-term reductions in nuisance algal blooms are to be attained through reduction in phosphorus loads.

## 7.3 HYPOLIMNETIC AERATION AND DESTRATIFICATIQN

### 7.3.1 The consequences of stratification in enriched lakes

During summer stratification, the epilimnion is separated from the hypolimnion by a thermal gradient, so that mixing of the water in the two layers is inhibited (Section 5.1.5). Exchange of dissolved gases and chemical substances between the two layers is retarded. In hypertrophic lakes, such as Hartbeespoort Dam, decay of the rain of organic matter from the highly productive epilimnion rapidly depletes the oxygen reserves of the hypolimnion. Within a month of the onset of stratification the hypolimnion becomes anaerobic. Under the reducing conditions ammonia is the dominant form of bound nitrogen. Anaerobic organisms also reduce sulphates to hydrogen sulphide and nitrates to nitrogen gas. Aerobically precipitated iron, manganese and orthophosphates are resolubilized under anaerobic conditions. With the lack of vertical mixing into the epilimnion the concentrations of these substances in the hypolimnion rise. Most of these chemical changes seriously detract from the usefulness of hypolimnetic water as a domestic and industrial raw water supply.

Biological consequences of stratification extend beyond the changes indicated above. Phytoplankton species, such as diatoms, which rely on physical mixing of the water column to keep them in suspension, disappear under stratified conditions because they sink to the bottom. The phytoplankton comes to be dominated by motile species or by species with buoyancy mechanisms, particularly blue-green algae such as *Micvocystis*. Zooplankton and fish no longer have access to the deep water due to the lack of oxygen. Shapiro *et at.* (1982) suggested that this restriction results in a feed-back mechanism whose consequences are an increase in phytoplankton abundance and a dominance by large colonial phytoplankton species such as *Micvocystis* . This hypothesis is based on several facts. Most zooplankton-eating fish rely on sight to detect their food and prey more heavily on large zooplankton than on small. Only large zooplankton species graze large or colonial phytoplankton. Zooplankton have a diel vertical migration and spend the day-light hours at greater depths where darkness hides them from fish. When the hypolimnion is anaerobic, zooplankton vertical migration is restricted and predation pressure on large zooplankton species increases to the point where they disappear in the stratified period. Small zooplankters maintain grazing pressure on small phytoplankters, so that the phytoplankton comes to be dominated by large or colonial species. In the absence of grazing pressure the abundance of large or colonial species increases to the point when it is limited by light or nutrients, and the abundance of the phytoplankton increases. Shapiro *et at.* (1982) also suggest that the high pH of epilimnetic eutrophic water inhibits cyanophages, which means that eutrophic conditions favour blue-green dominance in yet another way.

### 7.3.2 Hypolimnetic aeration and destratification as management tools

Aeration of the deep layer of enriched lakes to improve water quality and biological conditions may be carried out in two ways. In hypolimnetic aeration, hypolimnetic water is aerated without mixing it with epilimnetic water. In destratification the water column is vertically mixed so that the hypolimnion disappears and oxygen is distributed through the whole column.

Hypolimnetic aeration has been applied in cases where the lower temperature of the deep water is required, either as a summer refuge for cold water fish species or as a source of cold raw water (a requirement in some countries such as West Germany). Pastorok, Lorenzen & Ginn (1981) point out that, compared with destratification, hypolimnetic aeration shows greater promise for controlling internal phosphorus loads. This is because the preservation of the low hypolimnetic temperature precludes the accelerated decay of organic matter and consequent phosphorus release that takes place in the warmer hypolimnetic conditions resulting from destratification. This property of hypolimnetic aeration is only of importance where the internal recycling of phosphorus is a significant factor in the nutrition of the phytoplankton. Pastorok *&t at.* reported that the few studies of phytoplankton in relation to hypolimnetic aeration indicate minimal impact on chlorophyll concentrations, algal abundance, species composition or primary production.

There are, however, recorded changes in the zooplankton and fish which might, given time, change the phytoplankton. Experimental evidence is lacking. Since there is no specific need to maintain a cool hypolimnion in Hartbeespoort Dam and since the internal nutrient load under present conditions is unimportant in maintaining the present phytoplankton abundance, hypolimnetic aeration is not considered further in relation to Hartbeespoort Dam.

Destratification and hypolimnetic aeration can lead to desirable chemical changes. Hydrogen sulphide, iron and manganese concentrations decline to acceptable levels. Although destratification increases the heat content of the whole water body, surface water temperatures are lowered, which could result in a lowering of evaporation and the saving of significant quantities of water (D.F. Toerien, pers. comm.). Interest in destratification has been further stimulated by the large changes in the abundance and composition of the phytoplankton which frequently occur in destratified lakes. The causes of these changes are not clearly understood (see reviews of Toerien, de Bruin & Pieterse (1982), Pastorok, *et at.* (1981) and Shapiro *et at,* (1982)). Many destratification experiments have been inadequately monitored and the rate of induced circulation has seldom been recorded. Results therefore appear to be contradictory, with reports of phytoplankton abundance increasing, decreasing or unaffected and species composition altered or unaltered. Perspectives of what constitutes an eutrophic lake vary in different parts of the world and all too often descriptions of destratification experiments do not give adequate attention to key conditions in the lake prior to destratification. This hinders the understanding of the probable causes of phytoplankton changes or the lack of them from comparisons between lakes. It seems likely, however, that phytoplankton increase was sometimes due to nutrient transport from the hypolimnion to the epilimnion in lakes where epilimnetic phytoplankton growth was nutrient limited. In some cases induced circulation sufficient to destratify was insufficient to overcome the buoyancy or motility of the phytoplankton, which remained in the euphotic zone and increased due to the enhanced nutrient supply. Phytoplankton decreases and species changes have been ascribed to a variety of factors such as lower pH, increased availability of carbon dioxide, lower euphotic zone temperatures, an altered light regime for the individual cell or colony, a change in the dominant form of bound nitrogen and changes in the food web.

Pastorok *et al.* (1981) concluded that a shift of phytoplankton composition from blue-green algal dominance to green algal dominance is usually associated with a substantial decline in pH to a final pH value less than 7.5. Where there was no change in blue-green dominance or even an increase in phytoplankton abundance there was either no pH change or the pH was less than 7.5 before mixing. However, it should be borne in mind that Pastorok *et al.* also state that a shift away from buoyant blue-green algae only takes place when the induced circulation is sufficient to keep them mixed throughout the water column (Section 5.A.14). This was substantiated by information obtained from the Biesbosch Reservoirs near Rotterdam in the Netherlands. These three water bodies are built in old polders and are artificially destratified to control phytoplankton. Reservoir de Gijster is the only one in which nuisance blooms of buoyant blue-green algae *(Anabaena, Miorocystis)* occur (van Breemen, personal communication). This reservoir has a broad shelf at 7 i below full supply level but the other reservoirs do not. The blue-green algae proliferate in this shallower water above the shelf where they are not mixed down into the aphotic zone.

Artificial circulation has been seldom induced before the onset of stratification. This lack of experience is unfortunate for two reasons. Firstly the power required to maintain homoiothermy throughout the season is less than that required for more rapid destratification. Secondly, the biological results of destratification are not necessarily the same as those which would come about by preventing its development. Nicholls *et al.* (1980) and Ellis & Tait (1981) artificially circulated lakes for two years. In both studies a blue-green algal phytoplankton in the stratified lakes changed initially to a mixture of diatoms, green algae and cryptomonads. *Daphnia* populations increased sharply and grazed this more palatable assemblage heavily. In the second year *Ceratiwn hirudineVta* (a motile dinoflagellate), which is too large to be grazed by *Daphnia*, dominated the phytoplankton and reached very great densities. In spite of the circulation the *Cevativm* population in one lake collapsed, and in so doing, deoxygenated the entire water column so that all the zooplankton and fish in the lake died. The phytoplankton returned to predominately green algae. Here the study ended, but it would be compatible with earlier findings of the study to hypothesize that *Cevatiwn* would again eventually dominate and that the population would again collapse. This sequence of events is undesirable and not an improvement over a blue-green algae dominated phytoplankton.

It is unfortunate that these two Canadian studies are the only information available on long term mixing of eutrophic lakes. Their findings need to be confirmed. Pastorok *et al.* observed that Nicholls *et al.'s* study did not continue long enough for the fish population to reach an equilibrium with the new zooplankton composition. Had sufficient aeration to prevent the *Cevattvxn* collapse been applied, in subsequent years the *Ceratiim* population might not have become as large as it did in the second year.

### 7.3.3 Destratification in the management of Hartbeespoort Dam

From March 1975 to March 1976, the Departement of Water Affairs pumped 14 $m^3$ of air per minute through six 'aerators' placed 70 m apart at ± 30 m depth in a line extending about 500 m away from the wall of Hartbeespoort Dam. Observations of the physical, chemical and biological conditions during aeration were made by the NIWR. The design of the aerators was such that destratification was not induced, although oxygen was recorded at greater depths in the summer near the aerators than in the main basin. Surface water temperatures were about 1°C lower in the boil above the aerators than in the main basin. Differences in pH and water chemistry were not significant. Phytoplankton was monitored only near the aerators. *Miaroaystis aeruginosa* disappeared in mid-winter but was present and frequently dominant at other times. Compared to observations made earlier than the aeration year, there was an increase in the diversity of the phytoplankton, which included two species which had not been recorded since 1928, when Hartbeespoort Dam was oligotrophic.

The Departement of Water Affairs terminated the aeration for unspecified reasons, though it is known that, using a diesel-engined compressor, running expenses were high, there was a noise problem and the bottom water remained anaerobic in summer. Evaluated against a far more detailed data-base on the phytoplankton, which is now known to include up to thirty genera, the phytoplankton of the year of aeration is not as peculiar as it at one time seemed to be.

Several of the biological changes observed from year to year in Hartbeespoort Dam have a direct bearing on the probable consequences of artificial mixing. The diatom *Melosira granulata* is abundant in the phytoplankton only in spring as the lake warms and winds are increasing (Section 5.4.6), and hence natural mixing is taking place.

*Daphnia* (a large zooplankter) appears in late winter and builds up to large numbers in the spring when *Micvocystis* does not dominate the phytoplankton (Sections 5.4.15 and 5.4.6). In early summer fish fry, which prey heavily on zooplankton, appear in large numbers, *Miarooystis* increases sharply and the *Daphnia* population declines rapidly to disappear by mid-summer (Sections 5.4.6, 5.4.15, 5.4.17, 5.4.18). Causal relationships between these changes have not been proved, but the changes are consistent with Shapiro *et at.*'s (1982) previously mentioned hypothesis. In'Hartbeespoort Dam stratification leads to an enhanced loss of nitrogen through denitrification of nitrate in the anaerobic hypolimnion (Section 5.4.4,). This lowers the N:P ratio which would tend to favour the blue-green algae above the greens.

The morphometry of Hartbeespoort Dam would have an important bearing on the type of mixing necessary to disadvantage *Microoystis* by mixing it down into the aphotic zone. The dam is dendritic and shallow. Should artificial mixing be undertaken solely to destratify the water body (i.e. eliminate the summer thermocline) the greater part of the surface of the dam would not be disturbed and a large *Miorooyst-is* population would be maintained. This is because up-and-down-welling would be localized to the vicinity of the aerators. In order to reduce the *Microcystis* population mixing would have to take place over most of the dam surface.

In every year of the study the natural autumnal destratification (ie. the overturn) has been followed by the build-up of very large surface blooms of *M-icvocyst'Ls* and the formation of hyper scums (Section 5.4.8). Winter mixing of the water column is weak and *Miovocystis* buoyancy is sufficient to maintain this species in the upper layers of the water column. This suggests that artificial mixing might have to be extended to August when wind speeds usually increase.

The outcome of artificial mixing of Hartbeespoort Dam may be examined using the ecosystem model, but the model has several limitations which prevent its use for conclusive predictions of the consequences of aeration-destratification. These are:

1. With the exception of phosphorus the model does not consider the movement or behaviour of chemical elements. This weakness is particularly significant with respect to nitrogen and hence to N:P ratios.

2. The rate of decomposition of particulate P and uptake and release of dissolved P by the sediments are assumed to be independent of temperature.

3. Phytoplankton and zooplankton are treated as single communities and species changes cannot be predicted. The fish species composition is fixed and will not respond to environmental change other than in phosphorus concentration.

Hence many of the likely consequences of aeration-destratification, such as the influence of changes in hypolimnetic temperature on internal loading, changes in rates of decomposition in the hypolimnion, changes in N:P ratios or algal species dominance cannot be predicted by the model. However, the likely impact of these factors may be inferred from the literature and results of the individual projects.

Aeration/destratification was simulated initially by setting oxycline height above the bottom sediment to zero and thus preventing the formation of a hypolimnion. This resulted in a dilution of the phytoplankton concentration throughout the whole lake volume. A subsequent increase in growth following improved light penetration increased phytoplankton mass by 6% (Table 7.6). Zooplankton increased by 14% due to the inclusion of the previously unavailable hypolimnetic suspended particulates (HYPART) in their detrital food resource (EPPART).

A further simulation, carried out in addition to removal of the hypolimnion following aeration/destratification, was improvement in the pallatability of phytoplankton to zooplankton. This simulates the behaviour of the ecosystem were the phytoplankton species to shift

Table 7.6. Simulation of events possibly following continuous aeration and/or destratification. Mean annual tonnes P at equilibrium after 5 years.

| Event simulated | Model output (tonnes P) | |
|---|---|---|
| | Phytoplankton | Zooplankton |
| Unmodified model | 2.50 | 0.73 |
| (a) Removal of anaerobic zone | 2.64 | 0.83 |
| Change in standing stock | +6% | +14% |
| (b) (a) plus increased utilization of phytoplanton by zooplankton | 1.28 | 1.29 |
| Change in standing stock | -49% | +77% |

from blue-green to green algae as a result of destratification. The increase in the ability of zooplankton to utilize phytoplankton led to a sharp decrease in phytoplankton standing stock (49%) due to zooplankton grazers and a sharp increase in the zooplankton standing stock followed (77%). The dynamics of fish in the model were not sensitive to such changes.

### 7.3.4 Discussion of destratification

The model's usefulness in this consideration of destratification is that it gives a first estimate of the likely change in the phytoplankton abundance were the palatibility of the phytoplankton to change as a result of destratification. It also shows that destratification without a change in the phytoplankton species composition would not achieve the objective of reducing the abundance of phytoplankton.

The literature reviewed (Section 7.3.2) showed that two circumstances were necessary to disadvantage the blue-green algae. These were that the pH should drop to below 7.5 and that the induced circulation should be sufficient to overcome the buoyancy of *Miovocystis.*

Reynolds (1973) found that the maximum buoyancy of unicellular (not colonial) *M. aeruginosa* was about 60 p s $^{1}$, which provides a measure of the minimum vertical currents that it would be necessary to induce. *Miorooystis* colonies have a greater buoyancy. The likely outcome of artificial mixing for the pH is a question that cannot be answered solely from physical and chemical considerations of the carbonate/bicarbonate system as the phytoplankton would also be competing for carbon dioxide (Pearl *et at.* 1983), and modifying the dynamic equilibrium.

Davis (1980) has set out guidelines for the design of circulation systems. He warns that, in lakes such as Hartbeespoort in which the surface-waters heat rapidly, an undersized system may not ensure the desired mixing intensity (isothermy) and may leave a refuge for blue-green on the surface. Thus the system needs to be designed with a variable mixing capability. A watch must, however, be kept on sediment resuspension by excessive currents.

Using Davis's energy summation, the power needed to overcome stability has been calculated in respect of Hartbeespoort from Figure 5.11. Mixing due to wind and throughflow has, as suggested by Davis, been ignored, on the basis that, if sufficient power can be distributed in their absence by a reasonably inexpensive system, wind and throughflow will act as variable bonuses. Due to surface water cooling, a term must be added to compensate for reduced back-radiation (Section 5.1.5(h)).

Assuming *5%* efficiency, the compressor power required to keep Hartbeespoort Dam fully mixed throughout the summer _is estimated to be 35 kW, and the necessary air-delivery 0.136 $m^3$ s $^1$, feeding along a 70 m airline. Drilled into the 50 mm i.d. pipe would be 210 - 0.8 mm holes spaced every 0.3 m. The appropriate compressor would cost about R30 000 (1985 prices).

A short (70 m) line might however, not exert effective control over a dendritic lake with 2.2 shore line development. Several smaller, cheaper compressors (say 4 at R10 00 delivering 0.04 $m^3$s * each) and longer lines might do the job more effectively.

Considering further aspects of the effects of destratification, species successions even within the palatable category of phytoplankton should not be neglected. We would expect *Melosiva granulata* to increase in abundance as an immediate consequence of increased mixing (Section 5.4.6), but the likely permanance and intensity of this change is unknown, in part because the trophic relationships of the ecosystem would also change. Pastorok *&t at.* (1981) point out that there have been too few experiments in which destratification has been closely monitored over periods of years to draw conclusions as to its permanent consequences. They also stress the need for studies of the effect of variation in the timing of initiation of destratification in the annual cycle.

In view of these considerations it should be obvious that it is premature to say more than that destratification holds promise as a useful management technique. This promise should be confirmed by experimentation with variation in rates of circulation and in timing of initiation. Such studies could now be made along the lines suggested by Davis (1980) since so much information on the normal functioning of Hartbeespoort Dam is available.

## 7.4 BIOLOGICAL MANIPULATION AND PROTEIN HARVESTING

Although it is widely recognized that the only long-term solution to the problems of eutrophication is reduction of nutrient inputs (Smith & Shapiro 1980), manipulation of the biological components of an ecosystem can reduce the symptoms of eutrophication (Shapiro 1979b). The most obvious symptoms of eutrophication are an increase in algal abundance and a switch from dominance by green algae to dominance by blue-green algae. Ultimately nuisance algal scums develop. Most research into biomanipulation has been aimed at eliminating these undesirable symptoms.

One strategy has been to manipulate the ratio of total nitrogen (TN) to total phosphorus (TP) concentrations, either by removing P or adding N. Smith (1983) has clearly demonstrated that lakes having epilimnetic TN:TP ratios greater than 29:1 by weight typically exhibit low proportions of blue-green algae regardless of the ambient concentrations of either nutrient. In addition, Schindler (1977) and Flett *et at.* (1980) also reported shifts in species composition in several Canadian lakes resulting from changes in N:P loading ratios. Similarly, blue-green algae declined in Onondaga Lake, New York, after phosphate-containing detergents were banned. The TN:TP ratios in Onondaga increased from 3.2 before the ban (1968/69) to 20.3 after the ban in 197A (Sze 1980). However, in highly eutrophic lakes reduced P loads may be followed by increased rates of P release from the sediments and subsequently little change in the epilimnetic TN:TP ratios. For example, in Lake Trumen, Sweden, P loads were drastically reduced by sewage diversion. However, the proportion of blue-green algae decreased only 12 years later after the nutrient-rich sediments were dredged and the TN:TP ratio increased (Smith 1983). Consequences of the implementation of the 1 tag I standard for N:P ratios in Hartbeespoort Dam will depend on the effluent treatment used to achieve the standard. Assuming that physical-chemical phosphate removal would have no effect on the nitrogen load and that modified activated sludge processes would remove 40% of the current nitrogen load, N:P ratios might rise from their present 3.5:1 to 19:1 (no N load reduction) or to 13:1 (40% N load reduction). These projected ratios do not allow for denitrification losses within the dam which might halve the quantity of nitrogen (Section 5.2.4) giving final ratios of 9.5:1 or 6.5:1.

Due to technical difficulties, algal harvesting as a means of phosphorus removal from lakes is not a common management measure. However, in Hartbeespoort Dam up to 50% of the phytoplankton standing stocks may accumulate into hyperscums during winter (Zohary, in prep.) which would facilitate removal. While harvested *Microoystis* may provide a rich source of protein, it is toxic most of the year (Section 5.4.13) and probably could not be used as animal feed.

The impact of zooplankton grazing on phytoplankton standing stocks has been examined by many researchers. Zooplankton grazing has been estimated to reduce algal biomass by up to 40% in Lake Norrviken, equivalent to a reduction in total phosphorus of 170 Ug *£* (Shapiro 1979b). In Lake Washington, Edmondson and Litt (1982) and Shapiro (1979a) recorded decreasing algal density and increasing water transparency with increasing *Daphnia* populations. Conversely, elimination of herbivorous zooplankton has led to large algal blooms (Shapiro 1979b). McNaught & Scavia (1976) and Shapiro (1979a) have suggested that zooplankton grazing may also determine phytoplankton species composition. In experimental enclosures Schoenberg & Carlson (1984) have demonstrated the ability of introduced large zooplankton species to reduce the proportion of blue-green algae present, but this has not been attempted on a larger scale.

It has been suggested that aquatic communities are controlled mainly by predation from the top of the trophic pyrimid (McNaught & Scavia 1976; Wurtsbaugh, Li & Li 1981). It is also possible for herbivorous fish to reduce the algal population. Leventer (1979) stated that the phytoplanktivorous silver carp, *Hypophthalmiathys molitrix,* can reduce algal biomass by 25% . This species can feed on *Microcystis* (Leventer 1979). Conversely, Nakashima & Leggett (1980, 1982) suggested that fish may influence eutrophication by the retention of phosphorus in fish tissue or indirectly by reduction of herbivorous zooplankton populations and a shift to smaller zooplankton species.

This reduces zooplankton grazing pressure on the algae and results in increased algal blooms (Anderson, Berggren, Cronberg & Gelin 1978; Shapiro 1979 a, b). Removal of fish has frequently led to an increase in large zooplankton species and a decrease in algal abundance (Anderson *et at.* 1978, Shapiro 1979a). It has also been suggested that fish contribute to internal loading of phosphorus through mechanical stirring up of sediments and by digestive activities (Lamarra 1975). In view of the high dissolved phosphorus concentrations in Hartbeespoort Dam the impact of fish on internal loading will have a negligible impact on nutrient dynamics.

In order to test the applicability of the above observations to Hartbeespoort Dam, the following biomanipulation strategies are examined in this section, making use of the Ecosystem Model:

(a) Harvesting of phytoplankton scums as a means of reducing algal standing stock and phosphorus

(b) Harvesting phytoplankton and zooplankton from the water column. The practical problems associated with this option are recognized.

(c) Increasing direct grazing on phytoplankton by fish by increasing the population of *0. mossambious* or by replacing *0. mossambicus* with a hypothetical obligate phytoplanktivore.

(d) Reducing the population of selected fish species to reduce grazing on zooplankton, thus increasing zooplankton grazing on phytoplankton.

(e) Harvesting of fish to make optimal use of the high fish production and the heavy recreational demand.

### 7.4.1 Model structure in relation to biomanipulation

In using the model to test the impact of biomanipulation it is important to take note of the factors regulating net growth of the biotic compartments in the model. These were discussed under Section 6.2 and can be summarised as:

(a) Phytoplankton - light, temperature, phosphorus concentration. Scum formation is regulated by wind.

(b) Zooplankton - phytoplankton abundance and assimilability, particulate phosphorus abundance, zooplankton density, temperature.

(c) Fish - dissolved and particulate phosphorus concentration.

Phytoplankton growth is regulated by temperature and depth of light penetration. Light penetration is in turn regulated by phytoplankton concentration. Therefore, phytoplankton growth is regulated by phytoplankton concentration-mediated feedback. Phosphorus levels limit growth rate only when the phosphorus concentration is low. The phosphorus half-saturation constant for phytoplankton growth is 20 yg P *Z*~$^{l}$ (Holm & Armstrong 1981, Nicklisch & Kohl 1983). Algal species succession is not incorporated into the phytoplankton submodel because the model is structured on rates measured for the whole algal community.

Most phosphorus that accumulates in scums is lost from the free-floating phytoplankton compartment. High losses occur daily from scums to sediment phosphorus, suspended particulate phosphorus and dissolved phosphorus compartments of the ecosystem model. These loss rates may be overestimated in the present model structure. Reinoculation of the lake with scum algae during scum-breaking returns very little algae to the phytoplankton compartment.

Zooplankton respond to changes in the abundance of phytoplankton and suspended particulate detritus. Their grazing rate is regulated by both fluctuations in zooplankton biomass and by seasonal changes in their ability to utilize the available phytoplankton. This latter regulatory function simulates the seasonal changes in the proportions of blue-green and green algae present in the lake with regard to zooplankton grazing pressure. The role of temperature in the zooplankton sub-model is primarily through its regulation of natural mortality.

In the model fish will not respond to changes in phytoplankton and zooplankton abundance. The logarithmic relationship between fish and total phosphorus (Hanson & Leggett 1982) as used in the model is questionable but was the only relationship available. It is likely that a decrease in phytoplankton abundance, particularly if accompanied by a shift from dominance of blue-green algae to more palatable green algal species, will result in a change in fish species composition and abundance. This will not be shown in the model output. It is also likely that reduction in phosphorus concentration would influence fish production only when nutrient limitation affects the phytoplankton. Thus the immediate response of fish to small changes in existing phosphorus concentrations, as predicted by the model, may be erroneous.

### 7.4.2 Biomanipulation and protein harvesting under current phosphorus loads

#### (a) Removal of scums

The model was modified to allow daily removal of all algal scums formed and therefore simulated scum harvesting at a theoretical maximum rate. This continual removal of algal biomass resulted in a total loss of approximately 24 tonnes P per annum. However, scum removal led to a minimal change in the phytoplankton (1% reduction of the mean annual standing stock). In the model, 35% of algae in the scum are lost daily (to sediment P, particulate P and dissolved P), and only a negligible proportion of the scum algae returns to the phytoplankton during the period of scum-breaking. Consequently, removal of scums has little effect on phytoplankton standing stock. However, it is possible that losses from scums are overestimated in the model, and that the contribution of breaking scums to the phytoplankton standing stock are underestimated.

#### (b) Removal of phytoplankton and zooplankton

Removal of phytoplankton other than that which is contained in scums would, in practice, be a very difficult management strategy to implement. However, this management option has been simulated by the model. Removal of phytoplankton (PHYTO) would also result in the simultaneous removal of a similar proportion of zooplankton (ZOOP), therefore, joint PHYTO-ZOOP removal was examined (Table 7.7). Removal of 1 t P PHYTO and 0.3 t P ZOOP per month resulted in little change in standing stocks (Table 7.7). Removal of phytoplankton (primarily *Miorocystis)* under present loading rates had little effect due to the resultant improvement in light penetration which allowed a compensatory increase in algal growth rate. In addition the reduction of zooplankton grazing pressure also aided recovery of algal standing stock. With the doubling of PHYTO and ZOOP removal to 2 t P and 0.6 t P respectively, zooplankton standing stock was depressed due to the rate of removal exceeding the potential rate of recovery.

Table 7.7. Simulated removal of phytoplankton (PHYTO) and zooplankton (ZOOP) (equal proportions of standing stocks). Mean annual tonnes P at equilibrium after 5 years.

| Predicted PHYTO | | Predicted ZOOP | |
|---|---|---|---|
| Unmodified model output | 2.50 | Unmodified model output | 0.73 |
| Removal of 1 t P as PHYTO month$^{-1}$ | 2.50 | Removal of 0.3 t P as ZOOP month$^{-1}$ | 0.68 |
| Change in standing stock | 0% | Change in standing stock | -7 % |
| Removal of 2 t P as PHYTO month$^{-1}$ | 2.21 | Removal of 0.6 t P as ZOOP month$^{-1}$ | 0.48 |
| Change in standing stock | -12% | Change in standing stock | -34% |

The resultant low zooplankton grazing pressure and further improvement of light penetration allowed faster algal growth and thus, despite the drastic removal measures, the phytoplankton standing stock was maintained at only 12% below the unmodified level.

(c) Fish harvesting

The fish of Hartbeespoort Dam provide an important recreational resource for the PWV area. Cadieux (1980) estimated that between 1975 and 1977 an average of 263 000 angler days per annum were spent at the dam. In this study the number of angler days per annum for 1982 and 1983 was estimated at 201 000. Hofmeyr (1978) calculated that an expenditure of R772 000 was generated by angling on Hartbeespoort Dam in 1976. This figure would have escalated considerably since then as a result of inflation. However, conditions in Hartbeespoort Dam are not ideal for angling and several popular angling species, including black bass, *Mioropterus 8almoid.es*$_9$ are seriously affected by the high pH and large fluctuations in dissolved oxygen concentration resulting from poor water quality (Cochrane 1984). As the 1 mg *£* standard is not predicted to reduce phytoplankton standing stock below the frequent nuisance category (20 Mg *Z* ) it is unlikely to improve water quality and thus benefit the quality of fish production.

The mean annual yield of fish taken from the dam by anglers between March 1982 and April 1984 was estimated to be 695 t (348 kg ha *) made up of:

| | | |
|---|---|---|
| 449 t | *C. carpio* | (65%) |
| 144 t | *0. mossambicus* | (21%) |
| 102 t | *C. gariepinus* | (15%) |

These yields are above the long-term sustainable yield as the low water level over much of the study period increased the susceptibility of fish to capture. Use of the equation of Beverton & Holt (Ricker 1975) indicates that *C. carpio* and C. *gariepinus* are over-exploited while *0. mossambicus* is under-exploited. Mechanistic production models for each of the three major species were constructed using observed growth, fecundity, breeding and mortality data (Cochrane, 1985). The maximum sustainable yields indicated by these models were:

| | |
|---|---|
| *C. carpio* | 375 t |
| *0. mossambicus* | 120 t |
| *C. gariepinus* | 30 t |
| TOTAL | 525 t |

Cochrane & Robarts (in prep.) estimated total fish yield from two primary productionryield relationships at 337 kg ha $^{1}$ (674 t) and 264 kg ha $^{l}$ (528 t) respectively. These figures and the results from the mechanistic models also indicate that the community is being exploited close to its sustainable yield.

Cochrane (1984) and Cochrane & Robarts (in prep.) showed that the *C. cavpio* and *C. gariepinus* populations are favoured by the current hypertrophic conditions. Improved water quality resulting in a decline in pH, decreased fluctuations in epilimnetic oxygen concentrations and a switch from *Microcystis* dominance to a palatable green algal species may decrease total yield but would enhance the fish species diversity and increase the value of the lake as a recreational fishing resort.

### (d) Manipulation of fish populations

The manipulation of the fish populations to reduce phytoplankton (using the ecosystem model) standing stocks was unsuccessful (Table 7.8). The strategies used were aimed at:

(a) increasing direct grazing on phytoplankton by fish

(b) decreasing grazing by fish on zooplankton thus increasing the zooplankton population and hence zooplankton grazing on phytoplankton.

Using the model, the former option was investigated by increasing the *Oreochromis* population without altering its diet and simulating the introduction of a hervibore, by altering the diet of *0. mossambicus* to phytoplankton alone. Predictably, the greatest reduction in phytoplankton was achieved by increasing the biomass of the purely phytoplanktivorous species to 2.0 t P. This resulted in a reduction of phytoplankton by only 5%.

Table 7.8. Selected model output for different fish populations. The starting fish population (FISHP) consists of 6.0 tonnes of P made up of 1.5 tonnes of *0. mossambicus,* "3.0 tonnes of *C. oarpio* and 1.5 tonnes of *C. gariepinus*- All masses shown as tonnes of phosphorus PHYTO = Phytoplankton, ZOOP = zooplankton and FISHP = fish standing stock.

| | PHYTO | ZOOP | FISHP |
|---|---|---|---|
| Normal | 2.50 | 0.73 | 6.50 |
| Increase *O. mossambicus* to 2.0 t | 2.48 | 0.72 | 6.94 |
| change in standing stock | -1 % | -1 % | +7 % |
| Replace *O. mossambicus* with obligate herbivore | 2.44 | 0.73 | 6.52 |
| change | -2 % | 0 % | 0 % |
| Increase total herbivore to 2.0 t P | 2.36 | 0.72 | 7.35 |
| change | -6 % | -1 % | +13 % |
| Decrease *C. carpio* and *C. gariepinus* to 50 % existing mass | 2.49 | 0.73 | 3.68 |
| change | 0 % | 0 % | -43 % |
| Decrease *C. carpio* and *C. gariepinus* to 10 % existing mass | 2.49 | 0.73 | 1.41 |
| change | 0 % | 0 % | -78 % |
| Decrease FISHP to 50 % existing mass | 2.50 | 0.73 | 3.25 |
| change | 0 % | 0 % | -50 % |
| Decrease FISHP to 10% existing mass | 2.51 | 0.74 | 0.65 |
| change | 0 % | +1 % | -90 % |

Decreased fish predation pressure on zooplankton was achieved by reducing the total fish population by 90% and by reducing the populations of *C. cavpio* and *C. gariepinus*» which feed most heavily on invertebrates, by 90%. In both cases the impact on phytoplankton and zooplankton standing stock was negligible suggesting that zooplankton are not limited by fish predation.

### 7.4.3 Biological manipulation following 75% P-load reduction

Following the implementation of the 1 mg P *Z* * standard, loading will, at best, be reduced by approximately 75% at the weir. Results of a series of modifications to the model to simulate biomanipulation strategies in conjunction with the effects of 75% load reduction are shown in Table 7.9. Phosphorus limitation to phytoplankton growth is slight with 75% load reduction and there is little change to phytoplankton and zooplankton standing stocks. Consequently, the additional strategy of phytoplankton and zooplankton removal does not result in a significantly greater reduction of standing stocks than that achieved by their removal under normal loading conditions

(Table 7.7). It is unlikely that the dominant phytoplankton will change from *Microcystis* to algal species more easily utilized by zooplankton. However, if such a change can be induced following 75% load reduction, the resulting standing stocks predicted are similar to predictions based on the same shift in dominant phytoplankton species following aeration/destratification with no load reduction (Table 7.6).

The impact of manipulation of the fish population on phytoplankton and zooplankton following 75% load reduction was negligible (Table 7.9). The replacement of *0. mossambicus* with an obligate herbivorous species and increasing the biomass to 2 t P reduced phytoplankton by a further 6%. Decreased grazing on zooplankton, achieved by reducing the total fish population by 90%, did not affect the zooplankton population and reduced phytoplankton standing stock by only a further 2% . The reasons for this lack of impact on phytoplankton are:

the high production rate of phytoplankton, even under predicted reduced loads,
the fact that zooplankton do not appear to be regulated by fish predation and
the low grazing rate by zooplankton on *Microeystis.*

Table 7.9. Results of simulated biomanipulations following an estimated 75% load reduction after implementation of the 1 mg P £ $^{1}$ standard. (Mean annual tonnes P after 5 years).

PHYTO = phytoplankton, ZOOP = zooplankton.

| | Event simulated | Model output | |
|---|---|---|---|
| | | PHYTO | ZOOP |
| a) | No modification | 2.50 | 0.73 |
| b) | 75% load reduction | 2.39 | 0.70 |
| | change in standing stock from (a) | -4% | -4% |
| c) | (b) plus removal of 2 t P PHYTO and 0.6 t P ZOOP | 2.09 | 0.45 |
| | change in standing stock from (b) | -13% | -36% |
| d) | (b) plus increase in utilization of PHYTO by ZOOP | 1.26 | 1.29 |
| | change in standing stock from (b) | -47% | +84% |
| e) | (d) plus 90 % reduction in fish biomass | 1.24 | 1.29 |
| | change in standing stocks from (d) | -2% | 0% |
| f) | (d) and increase in herbivorous fish biomass (2 t P) | 1.19 | 1.22 |
| | change in standing stocks from (d) | -6% | -5% |

The second explanation may be a feature of the model, rather than a reflection of the status in the ecosystem itself.

### 7.4.4 Conclusions and recommendations

#### (a) Protein harvesting and biomanipulation under current phosphorus loading

The removal of scums has a negligible effect on phytoplankton standing stock. The predicted maximum amount of phosphorus that may be removed from the lake per annum by scum harvesting is 24 tonnes P, assuming P to be about 1% of dry algal mass (Reynolds *et al* 1981) and dry mass to be about 6% of fresh mass (Ashton & Zohary, unpublished data) 24 tonnes P represents approximately 4 000 tonnes fresh algal mass based on the hypothetical daily removal of all scums. Practical problems arising from implementation of a scum harvesting program are the probable high cost and the limited use to which harvested algal protein can be put, due to the periodic toxicity of *Miorocystis.* Thus, as a management option, scum harvesting is only beneficial through elimination of a periodic eyesore and associated foul odours, and will contribute to the impoundment's recreational potential. Similarly, removal of both phytoplankton and zooplankton from the lake is also a management strategy that would be difficult to implement and would reduce phytoplankton standing stock by a maximum of 12%.

Surplus fish production is being heavily utilised by recreational anglers and the establishment of a commercial fishery would adversely affect the recreational fishing. In view of the socio-economic importance of the angling on Hartbeespoort Dam, it is recommended that the fishery be reserved purely for recreational use.

Stocking of game fish such as black bass is pointless with the existing poor (high pH and variable oxygen concentration) water quality. The fishery would benefit most from improvements in water quality which would increase the abundance and improve the condition of the large *Barbus* species and black bass.

The steep shoreline gradients caused a decrease in fish biomass by decreasing macrophyte and benthic production (Section 5.4.16). The overall fish production of the lake could be increased by creating artificial reefs or bays. The former would increase growth of epiphytes which could be utilized by fish and provide shelter for fry. The latter would enhance the potential for natural macrophyte populations with the same advantages as with artificial reefs. Clearly a reduction in water level fluctuations would permit macrophytes to become better established in the dam. Such drastic and expensive management action could be subsidised by charging more for angling at the dam. However, high charges could only be justified if other facilities, including water quality were considerably improved.

During the period of study the absence of spawning by *C. eavpio* and *C. gaviepinus* in 1982/83 had a negative impact on fish production. Such occurrences are likely to be rare but could be overcome by stocking when required. This may be of value for *C. aarpio* which is heavily utilised by anglers.

(b) Protein harvesting and biomanipulation following load reduction

It was predicted that the 1 mg *Z* phosphorus standard would not affect the mean annual chlorophyll concentration in the dam (Section 7.2.4). Hence the implementation of the standard should not significantly affect the potential yield of protein from either primary or secondary producers.

At the estimated best possible load reduction with implementation of the 1 mg *2* $^{l}$ standard (75% at the weir) phosphorus concentrations will still be high enough to permit high phytoplankton growth and *Miovocystis* dominance. The model predictions, listed below, suggest that most biomanipulation strategies will be of little value:

(i) Removal of scums will have no benefit other than the temporary elimination of the scums and their associated disadvantages such as odour and interference with angling or boating;

(ii) Removal of phytoplankton and zooplankton from the water column would not result in significant reductions in phytoplankton standing stock;

(iii) Attempting to increase zooplankton and fish grazing on *Microcystis,* by manipulation of the fish population would have no noticeable effect on phytoplankton standing stock;

(iv) Removal of the maximum yield of fish from the lake would represent a phosphorus loss of only 2-3 tonnes per annum which would not affect lake trophic status; and

(v) An increase in availability of phytoplankton to zooplankton, resulting from a switch from *Mierooystis* to a palatable green algal species, may result in a decrease in phytoplankton standing stock of almost 50% .

The model predictions are subject to error and the percentage reductions predicted cannot be assumed to be exact. Nevertheless, results and observations on primary production, zooplankton, fish feeding and fish population dynamics obtained in the individual studies, have led to similar conclusions, independently of the model. Therefore, the combined predictions of separate project results and the ecosystem model are that the only potentially effective biological management strategy which warrants further study is an algal species change. Such a change could possibly be brought about by aeration/destratification, a change in the N:P ratio or a decrease in pH. However, the mechanisms regulating such changes and the actual responses are not clearly understood (Section 7.4) and pilot studies are required.

A further possibility is that after reduction of the external load of phosphorus, the internal load, released from the sediments, will assume a dominant role in algal nutrition. Should this prove to be the case, in-lake phosphorus management through chemical precipitation might prove useful. The need for such lake management could only be assessed from an evaluation of post phosphorus load reduction conditions.

# 8. FUTURE RESEARCH REQUIREMENTS

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Arising from this study we identify the following research requirements.

## 1. ECOSYSTEM MODEL

At present the Hartbeespoort Dam ecosystem model, DEFMOD, is comparable with existing empirical models in predicting future phosphorus and chlorophyll conditions in Hartbeespoort Dam, and it can be anticipated that the precision of the model output will be improved by calibration. DEFMOD is based on theoretical principles, as opposed to simple, statistical relationships, and thus the factors influencing model output and system behaviour can be identified and management strategies evaluated accordingly. Similarly, the comprehensive structure of the model permits greater versatility in testing the roles and future status of different system components than simple two or three variable empirical models. It is therefore important to continue the development of DEFMOD.

The first priority is a structured sensitivity analysis which will permit identification of the components and rates regulating model output and hence will enable the structure of DEFMOD to be rationalised and possibly simplified. Once this has been undertaken the model could be calibrated to improve precision.

The data and rates incorporated into DEFMOD have been derived largely during a severe drought. The validation or improvement of these model parameters under higher inflow and full supply conditions is necessary. Thus DEFMOD should be regularly updated and continue to be used in the ordering of research priorities.

The study on Hartbeespoort Dam was intended to increase knowledge of nutrient cycling in Hartbeespoort Dam and in other southern African impoundments, particularly eutrophic and hypertrophic systems. Therefore, in order to gain maximum benefit from the study, the applicability of DEFMOD to other impoundments, using existing data, should be investigated. This will assist in improving the structure of DEFMOD as well as improving and consolidating knowledge of nutrient cycling in other lakes.

## 2. THE DETRITUS AND DISSOLVED ORGANIC CARBON CYCLE

It has been concluded that there is an important gap in the first phase studies in that too little is known of the cycling of phosphorus through the decomposition phases and the intermediate decomposition products - dissolved organic carbon and detritus. Major contributors to detritus cycling, the microbes and the zoobenthos, should receive research attention aimed at quantifying their role in the phosphorus cycle. This understanding could be used to improve DEFMOD.

Recent consideration of Hartbeespoort Dam as a future large source of potable water has revealed a dearth of knowledge on the dissolved organic carbon content and dynamics in the dam. This needs quantification for the design of water treatment works.

## 3. PHYSICAL LIMNOLOGY

The study has so far clearly shown that it is the physical properties of heat, wind and light that govern the chemical and biological functioning of Hartbeespoort Dam. More information is needed on the extent and nodes of seiche movements. The vertical amplitude of internal seiches in the gorge leading to the dam wall should be considered in the siting, design and operation of intake towers for water treatment works. Identification of the position of the nodes of seiches associated with various wind directions is essential to the accurate measurement of thermocline position and hence of epilimnion volume. This information would allow considerably improved measurement of the transport of nutrients between the hypo- and epilimnion, the importance of which in ecosystem functioning will increase as external phosphorus loads decrease after the 1 mg *2* phosphorus standard. A better knowledge of physical limnology would allow for the quantification of the vertical transport of nutrients in terms of wind speed, direction and duration.

## 4. THE DYNAMICS OF NUTRIENTS IN WATER COURSES

The study has shown that there is an almost complete lack of quantification of the dynamics of nitrogen and phosphorus in water courses and identification of the major factors controlling their dynamics. This information is essential to the rational management of the quality of water resources and the rational definition of effluent quality standards. Further quantitative data are required to improve predictions regarding the effect of pre-impoundments on phosphorus transport.

It is possible that the bio-availability of phosphorus released from nutrient removal activated sludge wastewater treatment plants will change after implementation of the 1 mg */2* orthophosphate standard. This would alter phosphorus dynamics in both the river and the dam and therefore the bio-availability of the phosphorus arriving at the dam should be monitored before and after the phosphorus standard is introduced.

## 5. MANAGEMENT RELATED STUDIES

Management actions which might lead to a change in the dominant phytoplankton group, *Miovocystis aevuginosa*$_3$ have been identified as those most likely to result in a large (50%) reduction in phytoplankton abundance through food chain events (Section 7.4.4). Aeration and destratification through the induced changes in water column stability, N:P ratios and pH might bring about the desired phytoplankton change and should be studied on a pilot scale (isolation columns). Such studies should include the effect of variation in the intensity of the induced mixing on phytoplankton species composition. Zooplankton response to phytoplankton change should be monitored and the experiments should be conducted under various stocking rates of fish. Water temperatures and wind speeds should be recorded to allow for calculation of water saved by reduced evaporation, whose monetary value could contribute to the expenses of operating an aeration system. Finally as the load of phosphorus to Hartbeespoort Dam declines the isolation columns could be used for experiments of the chemical inactivation of dissolved phosphorus, if necessary.

## 6. SURVEILLANCE OF THE CONSEQUENCES OF THE 1 mg l"$^1$ ORTHOPHOSPHATE STANDARD

The real impact of the reduced phosphate load due to the 1 mg *2* phosphate standard can only be assessed in impoundments whose status before the imposition of the standard was well-known. The biology and chemistry of Hartbeespoort Dam is known better than that of any other South African impoundment. Provided surveillance of Hartbeespoort Dam is maintained, it will provide the best concrete evidence of the impact of the standard. Requirements of the monitoring programme include measurement of the load of phosphorus entering and leaving the dam and of the water quality and phytoplankton and zooplankton species composition and abundance at a single sampling point (the raft station) in the main basin of the dam. Sampling intervals (continuous flow measurements, daily water samples from the Crocodile River, and weekly samples from the other sampling sites) should be as they have been in the first phase of the study.

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APPENDIX 5.1

STATION: HARTBEESPOORT DAM RAFT (EPLIMNETIC HOSEPIPE)

| parameter | units | year | OCTOBER | NOVEMBER | DECEMBER | JANUARY | FEBRUARY | MARCH | APRIL | MAY | JUNE | JULY | AUGUST | SEPTEMBER |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| pH | | 1980-81 | 9,4 | 9,4 | 9,4 | 9,6 | 10,0 | 9,6 | 9,4 | 8,9 | 9,0 | 9,2 | 9,1 | 9,0 |
| | | 1981-82 | 8,9 | 9,4 | 9,5 | 9,7 | 9,7 | 9,4 | | | | | | |
| | | 1982-83 | | 8,8 | 9,1 | 9,4 | 9,3 | 9,6 | 9,5 | 9,5 | 9,3 | 8,8 | 8,9 | 9,2 |
| | | 1983-84 | 9,3 | 9,5 | 8,8 | 9,3 | 9,7 | 9,5 | 9,2 | 9,3 | 9,0 | 9,2 | 8,3 | 9,1 |
| Conductivity | mS/m | 1980-81 | 51,9 | 57,8 | 58,3 | 55,7 | 53,7 | 52,2 | 52,1 | 54,8 | 55,8 | 56,3 | 57,8 | 58,1 |
| | | 1981-82 | 58,4 | 56,4 | 54,1 | 54,2 | 56,9 | 59,1 | 58,2 | 58,3 | 60,0 | 61,7 | 63,4 | 64,0 |
| | | 1982-83 | 64,8 | 63,4 | 63,2 | 59,5 | 58,4 | 56,4 | 55,4 | 56,1 | 55,2 | 56,6 | 58,9 | 60,3 |
| | | 1983-84 | 61,8 | 60,8 | 63,4 | 59,7 | 60,1 | 59,9 | 62,0 | 63,3 | 64,1 | 64,9 | 75,7 | 75,6 |
| Calcium | mg/1 | 1980-81 | 42 | 43 | 42 | 40 | 37 | 39 | 40 | 39 | 42 | 42 | 43 | 44 |
| | | 1981-82 | 44 | 40 | 39 | 39 | 40 | 43 | 43 | 44 | 48 | 46 | 47 | 46 |
| | | 1982-83 | 49 | 48 | 49 | 45 | 48 | 46 | 44 | 46 | 45 | 47 | 50 | 48 |
| | | 1983-84 | 49 | 54 | 48 | 45 | 46 | 50 | 50 | 55 | 64 | 56 | 55 | 47 |
| Magnesium | mg/1 | 1980-81 | 24 | 24 | 23 | 23 | 21 | 21 | 21 | 21 | 20 | 22 | 22 | 23 |
| | | 1981-82 | 24 | 24 | 24 | 23 | 22 | 22 | 21 | 22 | 23 | 23 | 22 | 23 |
| | | 1982-83 | 24 | 24 | 23 | 21 | 20 | 19 | 20 | 19 | 20 | 20 | 21 | 22 |
| | | 1983-84 | 24 | 21 | 20 | 18 | 19 | 19 | 19 | 20 | 19 | 21 | 22 | 23 |
| Kjeldahl-N | µg/1 | 1980-81 | 1143 | 756 | 797 | 855 | 971 | 775 | 963 | 692 | 698 | 402 | 710 | 757 |
| | | 1981-82 | 623 | 681 | 1064 | 1151 | 995 | 1395 | 835 | 993 | 735 | 1102 | 906 | 980 |
| | | 1982-83 | 812 | 610 | 560 | 694 | 613 | 657 | 673 | 902 | 782 | 826 | 1278 | 1333 |
| | | 1983-84 | 1275 | 787 | 692 | 754 | 560 | 402 | 432 | 453 | 329 | 699 | 971 | 1250 |
| Ammonium-N | µg/1 | 1980-81 | 69 | 55 | 107 | 58 | 40 | 53 | 242 | 109 | 108 | 57 | 115 | 136 |
| | | 1981-82 | 156 | 45 | 89 | 59 | 62 | 109 | 123 | 76 | 136 | 90 | 101 | 52 |
| | | 1982-83 | 92 | 94 | 66 | 70 | 95 | 94 | 108 | 167 | 134 | 230 | 552 | 475 |
| | | 1983-84 | 560 | 560 | 133 | 92 | 50 | 59 | 57 | 83 | 64 | 71 | 55 | 239 |
| Nitrate+Nitrite | µg/1 | 1980-81 | 1390 | 882 | 827 | 313 | 180 | 287 | 509 | 1498 | 1415 | 1405 | 1461 | 1700 |
| | | 1981-82 | 1768 | 1515 | 990 | 918 | 390 | 542 | 711 | 1004 | 1130 | 1511 | 1730 | 1829 |
| | | 1982-83 | 1484 | 1276 | 992 | 697 | 654 | 129 | 235 | 340 | 701 | 1517 | 2004 | 2338 |
| | | 1983-84 | 2507 | 1571 | 1259 | 1596 | 631 | 288 | 971 | 1005 | 1293 | 1531 | 1895 | 2496 |
| Nitrite-N | µg/1 | 1980-81 | 76 | 48 | 104 | 42 | 27 | 45 | 58 | 89 | 47 | 45 | 50 | 55 |
| | | 1981-82 | 71 | 82 | 72 | 57 | 40 | 100 | 111 | 70 | 85 | 92 | 87 | 82 |
| | | 1982-83 | 63 | 110 | 94 | 86 | 105 | 39 | 85 | 44 | 51 | 85 | 114 | 211 |
| | | 1983-84 | 265 | 317 | 282 | 228 | 71 | 71 | 92 | 96 | 105 | 64 | 77 | 87 |
| Silica | mg/1 | 1980-81 | 5,0 | 5,5 | 4,7 | 5,0 | 5,5 | 5,9 | 6,3 | 5,9 | 5,5 | 5,3 | 5,3 | 5,0 |
| | | 1981-82 | 5,4 | 4,2 | 4,9 | 5,2 | 5,5 | 5,6 | 5,8 | 5,8 | 5,6 | 5,5 | 4,4 | 4,2 |
| | | 1982-83 | 4,6 | 4,8 | 4,6 | 4,7 | 4,8 | 4,7 | 5,0 | 5,5 | 5,0 | 4,6 | 4,5 | 4,0 |
| | | 1983-84 | 3,2 | 4,5 | 4,7 | 5,1 | 4,9 | 4,3 | 4,9 | 5,1 | 5,0 | 4,0 | 2,2 | 1,3 |
| Total-P | µg/1 | 1980-81 | 462 | 413 | 496 | 492 | 397 | 406 | 522 | 491 | 452 | 484 | 453 | 387 |
| | | 1981-82 | 389 | 415 | 400 | 473 | 485 | 592 | 545 | 542 | 749 | 668 | 639 | 735 |
| | | 1982-83 | 787 | 764 | 572 | 496 | 457 | 417 | 514 | 397 | 562 | 780 | 870 | 958 |
| | | 1983-84 | 934 | 550 | 473 | 619 | 598 | 562 | 705 | 717 | 759 | 780 | 824 | 863 |

APPENDIX 5.1 (cont.)

STATION: HARTBEESPOORT DAM RAFT (continued)

| parameter | units | year | OCTOBER | NOVEMBER | DECEMBER | JANUARY | FEBRUARY | MARCH | APRIL | MAY | JUNE | JULY | AUGUST | SEPTEMBER |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| SRP | µg/l | 1980-81 | 308 | 314 | 309 | 232 | 215 | 263 | 334 | 345 | 378 | 310 | 278 | 299 |
| | | 1981-82 | 310 | 258 | 312 | 376 | 288 | 342 | 439 | 463 | 470 | 462 | 434 | 503 |
| | | 1982-83 | 461 | 512 | 451 | 280 | 325 | 227 | 245 | 296 | 335 | 520 | 618 | 812 |
| | | 1983-84 | 774 | 437 | 407 | 474 | 450 | 467 | 604 | 580 | 632 | 643 | 640 | 749 |
| **Chloride** | mg/l | 1980-81 | 48 | 45 | 49 | 46 | 40 | 39 | 42 | 39 | 40 | 40 | 42 | 43 |
| | | 1981-82 | 43 | 44 | 45 | 44 | 41 | 44 | 44 | 52 | 53 | 52 | 56 | 56 |
| | | 1982-83 | 64 | 63 | 67 | 65 | 65 | 63 | 62 | 62 | 64 | 68 | 67 | 64 |
| | | 1983-84 | 63 | 59 | 63 | 48 | 48 | 47 | 52 | 53 | 52 | 52 | 53 | 49 |
| Alkalinity | mg$CaCO_3$/l | 1980-81 | 144 | 141 | 150 | 146 | 131 | 137 | 135 | 137 | 141 | 139 | 150 | 153 |
| | | 1981-82 | 151 | 151 | 200 | 192 | 152 | | | | | | 146 | 142 |
| | | 1982-83 | 142 | 142 | 141 | 133 | 130 | 127 | 124 | 131 | 135 | 134 | 138 | 140 |
| | | 1983-84 | 147 | 141 | 123 | 126 | 133 | 131 | 129 | 131 | 139 | 145 | 150 | 155 |
| Iron | µg/l | 1980-81 | 170 | 46 | < 25 | < 25 | < 25 | < 25 | < 25 | < 25 | < 25 | < 25 | < 25 | 96 |
| | | 1981-82 | 134 | 53 | < 25 | < 25 | 27 | 29 | 71 | 320 | 46 | 53 | 62 | 165 |
| | | 1982-83 | 112 | 71 | 32 | 57 | 72 | 51 | 64 | 70 | 43 | 65 | 26 | 33 |
| | | 1983-84 | 46 | 50 | 85 | 50 | 42 | 45 | 96 | 90 | 200 | 74 | 86 | 89 |
| DOC | mg/l | 1980-81 | 5,1 | 6,6 | 6,3 | 6,0 | 5,6 | 5,7 | 5,4 | 5,4 | 5,5 | 5,5 | 6,5 | 6,2 |
| | | 1981-82 | 5,9 | 7,8 | 8,0 | 7,4 | 7,0 | 7,7 | 6,6 | 7,9 | | | | |
| | | 1982-83 | | | | | | | | | | | | |
| | | 1983-84 | | | | | | | | | | | | |

APPENDIX 5.2

## STATION: MAGALIES RIVER (A2M13)

| parameter | units | year | OCTOBER | NOVEMBER | DECEMBER | JANUARY | FEBRUARY | MARCH | APRIL | MAY | JUNE | JULY | AUGUST | SEPTEMBER |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| pH | | 1981-82 | | | | | | | | | | | | |
| | | 1982-83 | | 7,8 | 8,2 | 7,9 | 8,3 | 8,0 | 8,2 | 8,1 | 8,3 | 8,0 | 8,3 | 8,6 |
| | | 1983-84 | 8,3 | 8,1 | 7,7 | 8,1 | 8,1 | 8,0 | 8,0 | 8,1 | 8,0 | 8,0 | 8,2 | 8,3 |
| Conductivity | mS/m | 1981-82 | | | | | | | 35,5 | 36,6 | 38,1 | 37,1 | 38,8 | 40,3 |
| | | 1982-83 | 42,0 | 40,6 | 39,2 | 31,6 | 43,0 | 41,0 | 37,8 | 40,5 | 40,6 | 39,1 | 44,9 | 43,4 |
| | | 1983-84 | 45,3 | 36,5 | 35,1 | 36,9 | 46,6 | 42,8 | 47,1 | 45,1 | 45,6 | 45,6 | 51,3 | 52,7 |
| Calcium | mg/l | 1981-82 | | | | | | | | 35 | 34 | 32 | 33 | 34 |
| | | 1982-83 | 37 | 36 | 37 | 28 | 40 | 41 | 38 | 41 | 40 | 40 | 44 | 42 |
| | | 1983-84 | 43 | 38 | 29 | 33 | 40 | 40 | 47 | 48 | 52 | 47 | 43 | 37 |
| Magnesium | mg/l | 1981-82 | | | | | | | | 25 | 26 | 25 | 27 | 29 |
| | | 1982-83 | 28 | 28 | 27 | 20 | 24 | 31 | 26 | 29 | 26 | 27 | 26 | 30 |
| | | 1983-84 | 30 | 23 | 20 | 20 | 25 | 26 | 29 | 30 | 29 | 30 | 32 | 33 |
| Kjeldahl-N | µg/l | 1981-82 | | | | | | | 153 | 278 | 224 | 160 | 256 | 198 |
| | | 1982-83 | 177 | 252 | 163 | 150 | 455 | 152 | 168 | 213 | 253 | 178 | 396 | 323 |
| | | 1983-84 | 400 | 477 | 384 | 339 | 247 | 188 | 161 | 111 | 250 | 263 | 280 | 299 |
| Ammonium-N | µg/l | 1981-82 | | | | | | | 38 | 35 | 27 | 44 | 29 | 36 |
| | | 1982-83 | 62 | 100 | 38 | 65 | 176 | 43 | 55 | 69 | 102 | 101 | 108 | 145 |
| | | 1983-84 | 95 | 162 | 63 | 88 | 52 | 64 | 77 | 64 | 72 | 64 | 42 | 36 |
| Nitrate+Nitrite | µg/l | 1981-82 | | | | | | | 751 | 855 | 719 | 820 | 541 | 533 |
| | | 1982-83 | 447 | 1290 | 321 | 955 | 622 | 296 | 653 | 503 | 730 | 735 | 1910 | 67 |
| | | 1983-84 | 1080 | 668 | 634 | 649 | 889 | 395 | 596 | 794 | 850 | 669 | 467 | 447 |
| Nitrite-N | µg/l | 1981-82 | | | | | | | 16 | 12 | 13 | 14 | 9 | 16 |
| | | 1982-83 | 27 | 54 | 14 | 51 | 35 | 10 | 11 | 7 | 15 | 14 | 144 | 38 |
| | | 1983-84 | 12 | 58 | 22 | 35 | 19 | 12 | 20 | 15 | 17 | 16 | 13 | 11 |
| Silica | mg/l | 1981-82 | | | | | | | 11,6 | 11,9 | 11,2 | 10,0 | 11,4 | 11,7 |
| | | 1982-83 | 12,8 | 12,1 | 12,1 | 13,2 | 11,8 | 14,1 | 12,0 | 13,3 | 10,5 | 11,6 | 10,8 | 12,9 |
| | | 1983-84 | 13,5 | 12,6 | 10,0 | 11,5 | 11,4 | 11,7 | 13,3 | 12,6 | 12,6 | 12,4 | 11,9 | 13,8 |
| Total-P | µg/l | 1981-82 | | | | | | | 113 | 178 | 140 | 120 | 80 | 88 |
| | | 1982-83 | 113 | 436 | 70 | 103 | 328 | 110 | 168 | 158 | 220 | 175 | 492 | 282 |
| | | 1983-84 | 626 | 274 | 280 | 239 | 260 | 152 | 163 | 171 | 179 | 152 | 146 | 174 |
| SRP | µg/l | 1981-82 | | | | | | | 40 | 55 | 18 | 45 | 32 | 51 |
| | | 1982-83 | 79 | 341 | 29 | 85 | 224 | 60 | 92 | 94 | 143 | 75 | 383 | 201 |
| | | 1983-84 | 467 | 100 | 66 | 92 | 186 | 73 | 83 | 46 | 109 | 87 | 77 | 110 |
| Chloride | mg/l | 1981-82 | | | | | | | 7 | 6 | 7 | 6 | 7 | 7 |
| | | 1982-83 | 6 | 12 | 9 | 7 | 17 | 8 | 7 | 10 | 13 | 9 | 18 | 17 |
| | | 1983-84 | 21 | 11 | 12 | 7 | 18 | 9 | 15 | 9 | 10 | 14 | 13 | 12 |
| Alkalinity | mgCaCO$_3$/l | 1981-82 | | | | | | | 190 | 190 | 193 | 174 | 200 | 108 |
| | | 1982-83 | 216 | 188 | 199 | 155 | 185 | 233 | 201 | 220 | 205 | 212 | 197 | 214 |
| | | 1983-84 | 215 | 188 | 150 | 166 | 190 | 195 | 219 | 214 | 214 | 224 | 236 | 229 |
| Iron | µg/l | 1981-82 | | | | | | | | 280 | 181 | 125 | 100 | 177 |
| | | 1982-83 | 148 | 413 | 117 | 4880 | 540 | 82 | 183 | 41 | 30 | 49 | 29 | 74 |
| | | 1983-84 | 86 | 2017 | 1007 | 960 | 49 | 71 | 47 | 71 | 93 | 64 | 47 | 63 |

APPENDIX 5.3

STATION: CROCODILE RIVER (A2M12)

| parameter | units | year | OCTOBER | NOVEMBER | DECEMBER | JANUARY | FEBRUARY | MARCH | APRIL | MAY | JUNE | JULY | AUGUST | SEPTEMBER |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| pE | | 1981-82 | | | | | | | | | | | | 7,9 |
| | | 1982-83 | | 7,5 | 7,6 | 7,5 | 7,6 | 7,6 | 7,8 | 7,8 | 8,0 | 7,6 | 7,8 | 7,7 |
| | | 1983-84 | 7,6 | 7,6 | 7,3 | 7,4 | 7,7 | 7,7 | 7,3 | 7,4 | 7,6 | 7,5 | 7,6 | 7,6 |
| Conductivity | mS/m | 1981-82 | | | | | | | 72,3 | 75,6 | 78,9 | 80,0 | 79,8 | 73,4 |
| | | 1982-83 | 72,2 | 66,9 | 57,8 | 56,1 | 59,2 | 61,8 | 66,5 | 59,9 | 67,9 | 61,3 | 65,1 | 68,0 |
| | | 1983-84 | 57,0 | 58,4 | 62,2 | 66,2 | 77,2 | 68,5 | 66,4 | 73,6 | 76,1 | 74,3 | 86,9 | 84,1 |
| Calcium | mg/l | 1981-82 | | | | | | | 53 | 55 | 59 | 59 | 55 | 52 |
| | | 1982-83 | 52 | 50 | 45 | 42 | 46 | 47 | 45 | 51 | 47 | 54 | 54 | 49 |
| | | 1983-84 | 45 | 50 | 45 | 46 | 57 | 46 | 53 | 62 | 75 | 64 | 66 | 41 |
| Magnesium | mg/l | 1981-82 | | | | | | | 23 | 25 | 26 | 27 | 25 | 24 |
| | | 1982-83 | 22 | 22 | 18 | 17 | 19 | 20 | 22 | 20 | 22 | 24 | 23 | 24 |
| | | 1983-84 | 21 | 18 | 17 | 19 | 23 | 18 | 19 | 17 | 23 | 22 | 23 | 23 |
| Kjeldahl-N | µg/l | 1981-82 | | | | | | | 1154 | 1401 | 1513 | 1858 | 1638 | 1196 |
| | | 1982-83 | 1171 | 621 | 650 | 598 | 831 | 719 | 928 | 1210 | 1348 | 1297 | 1096 | 1233 |
| | | 1983-84 | 1207 | 709 | 602 | 482 | 511 | 633 | 821 | 839 | 1020 | 1302 | 1580 | 1564 |
| Ammonium-N | µg/l | 1981-82 | | | | | | | 103 | 371 | 256 | 352 | 143 | 81 |
| | | 1982-83 | 567 | 81 | 132 | 158 | 139 | 111 | 106 | 66 | 252 | 133 | 152 | 243 |
| | | 1983-84 | 356 | 220 | 157 | 72 | 91 | 134 | 112 | 90 | 144 | 180 | 78 | 178 |
| Nitrate+Nitrite | µg/l | 1981-82 | | | | | | | 8477 | 10560 | 11910 | 11515 | 10535 | 9076 |
| | | 1982-83 | 8961 | 7748 | 5497 | 5962 | 6646 | 7527 | 9065 | 9100 | 7780 | 10083 | 7580 | 8688 |
| | | 1983-84 | 5612 | 5457 | 5349 | 4902 | 5018 | 6580 | 9850 | 10587 | 11700 | 13045 | 10110 | 8413 |
| Nitrite-N | µg/l | 1981-82 | | | | | | | 63 | 87 | 164 | 178 | 151 | 71 |
| | | 1982-83 | 152 | 62 | 66 | 155 | 45 | 44 | 31 | 26 | 174 | 100 | 137 | 76 |
| | | 1983-84 | 111 | 224 | 111 | 48 | 62 | 90 | 252 | 126 | 159 | 190 | 51 | 83 |
| Silica | mg/l | 1981-82 | | | | | | | 7,4 | 4,3 | 4,0 | 4,3 | 4,9 | 5,3 |
| | | 1982-83 | 6,9 | 6,8 | 5,7 | 6,4 | 6,2 | 5,9 | 5,9 | 4,1 | 3,5 | 2,1 | 2,8 | 3,4 |
| | | 1983-84 | 6,5 | 6,4 | 6,4 | 6,7 | 3,8 | 3,6 | 7,7 | 5,9 | 4,6 | 6,1 | 3,9 | 6,3 |
| Total-P | µg/l | 1981-82 | | | | | | | 2182 | 3201 | 3212 | 2773 | 2989 | 2953 |
| | | 1982-83 | 2425 | 2579 | 1209 | 1694 | 1663 | 2159 | 2828 | 3263 | 2734 | 3233 | 2360 | 3300 |
| | | 1983-84 | 2476 | 1513 | 1369 | 1577 | 1388 | 1818 | 2682 | 3167 | 3185 | 4330 | 2884 | 2486 |
| SRP | µg/l | 1981-82 | | | | | | | 2029 | 2851 | 3006 | 2628 | 2686 | 2699 |
| | | 1982-83 | 2242 | 2322 | 1149 | 1505 | 1464 | 1724 | 2353 | 2862 | 2250 | 2783 | 1960 | 2775 |
| | | 1983-84 | 2082 | 1267 | 1174 | 1294 | 1159 | 1717 | 2195 | 2864 | 2977 | 3431 | 2567 | 2327 |
| Chloride | mg/l | 1981-82 | | | | | | | 62 | 65 | 69 | 71 | 74 | 75 |
| | | 1982-83 | 70 | 72 | 50 | 53 | 65 | 73 | 69 | 76 | 70 | 78 | 78 | 93 |
| | | 1983-84 | 80 | 78 | 57 | 59 | 80 | 77 | 67 | 93 | 87 | 88 | 73 | 63 |
| Alkalinity | mgCaCO$_3$/l | 1981-82 | | | | | | | 121 | 124 | 114 | 116 | 114 | 123 |
| | | 1982-83 | 113 | 115 | 105 | 100 | 101 | 104 | 110 | 122 | 126 | 120 | 121 | 107 |
| | | 1983-84 | 111 | 99 | 99 | 98 | 107 | 97 | 86 | 100 | 112 | 105 | 109 | 126 |
| Iron | µg/l | 1981-82 | | | | | | | | 190 | 88 | 79 | 68 | 83 |
| | | 1982-83 | 116 | 173 | 288 | 259 | 513 | 73 | 119 | 30 | 43 | 45 | < 25 | 28 |
| | | 1983-84 | 204 | 157 | 306 | 72 | < 25 | 72 | 93 | 87 | 83 | 66 | 62 | 41 |

APPENDIX 5.4

STATION: CANALS (A2R01)

| parameter | units | year | OCTOBER | NOVEMBER | DECEMBER | JANUARY | FEBRUARY | MARCH | APRIL | MAY | JUNE | JULY | AUGUST | SEPTEMBER |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| pH. | | 1981-82 | | | | | | | | | | | | 8,0 |
| | | 1982-83 | 7,3 | 8,2 | 8,6 | 12,1 | 8,7 | 8,9 | 9,1 | | | 9,1 | 8,7 | 8,9 |
| | | 1983-84 | 8,5 | 8,5 | 8,6 | 8,4 | 8,6 | 9,0 | 8,6 | 8,6 | 8,8 | 8,8 | 8,8 | 8,4 |
| Conductivity | mS/m | 1981-82 | | | | | | | 58,4 | 59,0 | 60,5 | 61,8 | 62,1 | 62,9 |
| | | 1982-83 | 63,7 | 61,5 | 62,7 | 61,3 | 57,3 | 55,8 | 54,7 | | | 55,4 | 56,9 | 60,0 |
| | | 1983-84 | 63,1 | 65,3 | 65,6 | 62,6 | 61,4 | 61,1 | 62,5 | 64,4 | 67,4 | 64,4 | 74,7 | 77,5 |
| Calcium | mg/l | 1981-82 | | | | | | | 46 | 46 | 47 | 47 | 49 | 47 |
| | | 1982-83 | 50 | 49 | 50 | 49 | 48 | 47 | 43 | | | 46 | 47 | 48 |
| | | 1983-84 | 48 | 55 | 50 | 45 | 46 | 47 | 50 | 56 | 64 | 55 | 54 | 39 |
| Magnesium | mg/l | 1981-82 | | | | | | | 22 | 22 | 23 | 23 | 23 | 24 |
| | | 1982-83 | 23 | 24 | 23 | 23 | 21 | 20 | 20 | | | 20 | 20 | 23 |
| | | 1983-84 | 24 | 22 | 20 | 19 | 21 | 19 | 19 | 21 | 22 | 21 | 23 | 24 |
| Kjeldahl-N | µg/l | 1981-82 | | | | | | | 983 | 665 | 599 | 745 | 893 | 927 |
| | | 1982-83 | 1235 | 1468 | 977 | 1157 | 979 | 701 | 781 | 905 | 728 | 1122 | 1192 | 1562 |
| | | 1983-84 | 1396 | 1202 | 1209 | 1336 | 1062 | 760 | 419 | 443 | 550 | 759 | 1049 | 1143 |
| Ammonium-N | µg/l | 1981-82 | | | | | | | 330 | 315 | 74 | 99 | 133 | 290 |
| | | 1982-83 | 662 | 878 | 462 | 635 | 368 | 242 | 251 | 241 | 143 | 330 | 528 | 650 |
| | | 1983-84 | 730 | 580 | 609 | 920 | 763 | 418 | 112 | 100 | 122 | 165 | 210 | 400 |
| Nitrate+Nitrite | µg/l | 1981-82 | | | | | | | 725 | 1313 | 1599 | 2135 | 2084 | 1763 |
| | | 1982-83 | 2453 | 605 | 662 | 614 | 722 | 402 | 319 | 487 | 932 | 1516 | 1920 | 2312 |
| | | 1983-84 | 352 | 501 | 489 | 1430 | 1523 | 469 | 1135 | 1193 | 1836 | 2200 | 2218 | 2596 |
| Nitrite-N | µg/l | 1981-82 | | | | | | | 178 | 314 | 105 | 69 | 125 | 263 |
| | | 1982-83 | 261 | 282 | 234 | 170 | 200 | 98 | 103 | | | 98 | 199 | 332 |
| | | 1983-84 | 352 | 501 | 489 | 372 | 247 | 115 | 138 | 97 | 92 | 70 | 78 | 97 |
| Silica | mg/l | 1981-82 | | | | | | | 5,0 | 5,6 | 5,2 | 5,0 | 4,9 | 4,6 |
| | | 1982-83 | 5,3 | 5,2 | 4,8 | 4,9 | 5,3 | 4,9 | 5,2 | | | 4,8 | 4,7 | 4,3 |
| | | 1983-84 | 3,6 | 4,3 | 4,6 | 5,4 | 5,5 | 4,6 | 4,9 | 5,1 | 4,7 | 4,3 | 2,5 | 1,7 |
| Total-P | µg/l | 1981-82 | | | | | | | 779 | 521 | 552 | 557 | 675 | 678 |
| | | 1982-83 | 1275 | 998 | 858 | 800 | 605 | 474 | 456 | 407 | 1020 | 767 | 876 | 1027 |
| | | 1983-84 | 1078 | 764 | 823 | 877 | 779 | 679 | 675 | 660 | 814 | 807 | 812 | 902 |
| SRP | µg/l | 1981-82 | | | | | | | 577 | 460 | 499 | 498 | 634 | 638 |
| | | 1982-83 | 1127 | 940 | 735 | 647 | 461 | 376 | 306 | 345 | 406 | 544 | 656 | 850 |
| | | 1983-84 | 852 | 678 | 655 | 757 | 702 | 570 | 579 | 555 | 704 | 620 | 666 | 841 |
| Chloride | mg/l | 1981-82 | | | | | | | 49 | 49 | 49 | 48 | 48 | 50 |
| | | 1982-83 | 54 | 47 | 52 | 51 | 49 | 53 | 54 | | | 58 | 56 | 62 |
| | | 1983-84 | 64 | 63 | 61 | 51 | 48 | 51 | 56 | 58 | 66 | 53 | 56 | 49 |
| Alkalinity | mgCaCO$_3$/l | 1981-82 | | | | | | | 144 | 141 | 144 | 147 | 145 | 147 |
| | | 1982-83 | 144 | 165 | 154 | 153 | 147 | 133 | 134 | | | 140 | 140 | 140 |
| | | 1983-84 | 150 | 150 | 140 | 133 | 134 | 132 | 121 | 127 | 138 | 146 | 146 | 153 |
| Iron | µg/l | 1981-82 | | | | | | | | 174 | 41 | 31 | < 25 | 38 |
| | | 1982-83 | 67 | 131 | 38 | 37 | 85 | 34 | 37 | | | 39 | 27 | < 25 |
| | | 1983-84 | < 25 | < 25 | < 25 | < 25 | < 25 | 26 | 41 | 26 | 29 | 26 | 46 | 58 |

## APPENDIX 6.1

# HARTBEESPOORT DAM ECOSYSTEM MODEL ASSUMPTIONS

Assumptions, as listed below, are defined as rates or constants for which accurate, applicable values could not be obtained during the study or from the literature. Approximate values used were based either on the best possible literature values or informed estimates. Assumptions specifically referred to in the text are not included in this table.

1. Within each layer the distribution of P is homogeneous for each subcompartment, except phytoplankton and fish.

2. Benthic invertebrates and macrophytes are insignificant in nutrient cycling in the system.

3. Depth of the oxycline remains constant within the month, changing only at the end of thirty day iterations.

4. Reactions in the hypolimnion are not temperature dependent.

5. Factors such as temperature and pH have a negligible influence on sediment/water phosphate exchange.

6. Mean phosphate levels in the epilimnion reflect the sediment/water phosphate equilibrium concentration.

7. The sediment/water phosphate exchange characteristics for the whole dam are adequately described by the mean values generated from a limited number of experiments in selected areas.

8. Luxury uptake or internal storage of phosphorus and release of excretory products by the phytoplankton do not contribute significantly to phosphorus cycling in the lake.

9. Zooplankton are treated as a single unit, not divided by taxon or food type consumed.

10. Abundance of fish is related to total phosphorus concentration according to the relationship of Hanson and Leggett (1982).

11. Species composition does not change with changes in trophic status.

APPENDIX 6.2

## GLOSSARY OF TERMS

---

| Name | Definition | Units |
|---|---|---|
| AFIS(1)N(2,3) | No. of adult fish sp. 1, 2 and 3 | |
| AINCR(l) 2,3) | Growth increment of adult fish sp. 1, 2 & 3 | cm d"$^1$ |
| ALEN(l) 2,3) | Mean standard length of adult fish sp. 1, 2 & 3 | cm |
| ALGALC | Phytoplankton biomass in euphotic zone | mgC m $^2$ |
| AMASS(l) 2,3) | Mean mass of adult fish sp. 1, 2 & 3 | g |
| AMAX | Max. volumetric rate of pri. production | mgC m$^3$ h $^1$ |
| ARCHING | Daily change in area of hypolimnetic sediments | ha |
| AREA | Lake surface area | ha |
| ASIMD | Assim. efficiency of ZOOP on detritus (EPPART) | |
| ASIMP | Assim. efficiency of ZOOP on phytoplankton (PHYTO) | |
| AVEP | Mean annual epilimnetic soluble phosphorus concentration | mg £ |
| BADECO | Bacterial decomposition of particulate phosphorus | |
| BHARV 2 | Harvest of 0+ C. carpio | t d"$^1$ |
| CHLCNC | Mean lake chlorophyll concentration | mg m $^3$ |
| CHLZEU | Mean chlorophyll concentration in euphotic zone | mg m $^3$ |
| CHNG | Daily change in volume of hypolimnion | m$^3$ x 10 |
| CODE1,2 | Date codings | |
| DAMVOL | Volume of lake | m$^3$ x 10$^6$ |
| DAYS | Number of days in month | |
| DEATH | Phytoplankton mortality | tP d"$^1$ |
| DEATHR | Phytoplankton death rate | d"$^1$ |
| DEP | Depth of lake | m |
| DEPDIS | Change in epilimnetic dissolved phosphorus | tP d |
| DEPPRP | Change in epilimnetic particulate phosphorus | tP d |
| DFBIO(l) (2,3) | Mass of fish sp 1, 2 and 3 before removal of daily mortality | t Biomass |

| Name | Definition | Units |
|---|---|---|
| DHYDIS | Change in hypolimnetic dissolved phosphorus | tP dy * |
| DHYPRP | Change in hypolimnetic particulate phosphorus | tP d ^ |
| DHYSED | Change in hypolimnetic sedimentary phosphorus | tP d $^{1}$ |
| DIS | Monthly inflow of soluble P | tP |
| DISP | Dissolved P content of the lake | tP |
| DISPI | Daily inflow load of soluble P | tP |
| DISPO | Daily outflow of soluble P | tP |
| DPARTP | Daily change in PARTP | tP |
| DPCHNG | Amount of dissolved P transferred between hypolimnion and epilimnion | tP d $^{1}$ |
| DPHYTO | Daily change in PHYTO | tP |
| DSCUM | Daily change in SCUM | tP |
| DSHSED | Daily change in epilimnetic sedimentary P | tP |
| DZOOP | Daily change in ZOOP | tP |
| ECDYS | Loss of ZOOP through ecdysis | tP d"$^{1}$ |
| EMIN | Attenuation coefficient of PAR | In m $^{1}$ |
| EP | Total monthly evaporation | $m^3$ x $10^6$ |
| EPBDEC | Bacterial decomposition of EPPART | tP d"$^{1}$ |
| EPCONC | Concentration of epilimnetic dissolved P | mg *Z* * |
| EPDISP | Total epilimnetic dissolved P | tP |
| EPDPO | Canal outflow of EPDISP | tP d"$^{1}$ |
| EPEQIL | Sediment water equilibrium phosphorus concentration in epilimnion | ug *Z* |
| EPIDEP | Depth (thickness) of epilimnion | m |
| EPIVOL | Volume of epilimnion | $m^3$ x $10^6$ |
| EPPART | Total epilimnetic particulate phosphorus | tP |
| EPPPO | Canal outflow of EPPART | tP d~$^{l}$ |
| EPPROP | Area of epilimnetic sediments as a proportion of total sediment area | |
| EPSUSL | Conversion of EPPART to EPDISP | tP d"$^{1}$ |
| EVAP | Daily evaporation | $m^3$ x $10^6$ |
| EXRAT | Excretion rate | tP d™$^{1}$ |
| FAB | Food assimilation of fish | tP |
| FBIO(l) 2,3) | Biomass of fish species 1, 2 and 3 | tP |
| FFIS(1)N(2,3) | No. of fry (0+) of fish sp. 1, 2 and 3 | Nos. |

| Name | Definition | Units |
|---|---|---|
| FIDECO | PARTP consumed by fish | tP d"$^{1}$ |
| FISECO | SEDP consumed by fish | tP d"$^{1}$ |
| FISEXP | PARTP egested by fish | tP d"$^{1}$ |
| FTSHP | Total fish phosphorus content | tP |
| FISURP | DISP produced by fish urine | tP d"$^{1}$ |
| FLAG | Term to permit partitioning of P between epi- and hypolimnion on first iteration | |
| FLOIN | Daily inflow load of total P | tP d"$^{1}$ |
| FLOINT | Monthly inflow load of total P | tP |
| FLOUT | Daily outflow load of total P | tP d"$^{1}$ |
| FLOUTT | Monthly outflow load of total P | tP |
| FLYOFF | Loss of ZOOP via Chaoborid flyoff | tP d"$^{1}$ |
| FMASS(l) 2,3) | Mass of fry of fish species 1, 2 and 3 | g |
| FOOD | Total food consumed by fish | tP |
| FPHYTO | Fraction of food present as phytoplankton for zooplankton | - |
| FRAT | Proportional daily change in TOTP | |
| Fl | Amount of PHYTO consumed by fish | tP d"$^{1}$ |
| F2 | Amount of ZOOP consumed by fish | tP d"$^{1}$ |
| F3 | Amount of SEDP and EPPART consumed by fish | tP d"$^{1}$ |
| GR | Biomass specific grazing rate of zooplankton | |
| GRAD | Gradient of line in sediment uptake/release rate: dissolved P concentration relationship | |
| GROWTH | Gross growth of *0. mossambicus* pop. | tP d $^{1}$ |
| GR0W2 | Gross growth of *C. oafpio* pop. | tP d"$^{1}$ |
| GR0W3 | Gross growth of *C. gariepinus* pop. | tP d ' |
| HARV(l) (2,3) | Mass of fish species 1, 2 and 3 harvested | tP d"$^{1}$ |
| HC0DS,HCOSL | Y intercept in sediment uptake/release rate: dissolved phosphorus concentration relationship for hypolimnion | |
| HYAREA | Area of hypolimnetic sediments | ha |
| HYBDEC | Bacterial decomposition of HYPART | tP d"$^{1}$ |
| HYCONC | Concentration of hypolimnetic dissolved phosphorus | mg *Z* * |

| Name | Definition | Units |
|---|---|---|
| HYDISP | Total hypolimnetic dissolved phosphorus | tP |
| HYDFO | Canal outflow of HYDISP | tP d"$^{1}$ |
| HYEQIL | Sediment: water equilibrium phosphorus concentration in hypolimnion | Pg £ * |
| HYPART | Total hypolimnetic particulate phosphorus | tP |
| HYPHSD | Sedimentation of phytoplankton into hypolimnion | tP d $^{1}$ |
| HYPPO | Canal outflow of HYPART | tP d"$^{1}$ |
| HYPRDS | Uptake of dissolved P by sediments | tP d"$^{1}$ |
| HYPROP | Area of hypolimnion sediments as a proportion of total sediment area | |
| HYPRSD | Initial sedimentation of particulate P to hypolimnetic sediments | tP d $^{1}$ |
| HYPSED | Total hypolimnetic sedimentary P | tP |
| HYPVOL | Volume of hypolimnion | m$^{3}$ x 10$^{6}$ |
| HYSDET | Sedimentation of particulate P to hypolimnetic sediments | tP d |
| HYSDSL | Release of soluble phosphorus from the sediments | tP d $^{1}$ |
| HYSUSL | Conversion of HYPART to HYDISP | tP d"$^{1}$ |
| HYZOEX | Particulate P loss from zooplankton to hypolimnetic sediments | tP d |
| MAXEEQ | Maximum epilimnetic dissolved P concentration | mg *Z* * |
| MAXHEQ | Maximum hypolimnetic dissolved P concentration | mg *Z* $^{1}$ |
| MINEEQ | Minimum epilimnetic dissolved P concentration | mg *Z* $^{1}$ |
| MINHEQ | Minimum hypolimnetic dissolved P concentration | mg *Z* |
| OMALG | Prop, of *0. mossambious* pop feeding on algae | |
| OMDET | Prop, of *0. mossambicus* pop. feeding on detritus | - |
| OMZOOP | Prop, of *0. mossambious* pop. eating ZOOP | |
| OUTPRP | Daily prop, of lake volume in outflow | |
| OXY | Height of oxycline above maximum depth | m |
| PAR | Photosynthetically available radiation | pE m $^{2}$ s |
| PARTPI | Daily inflow of particulate P | tP d $^{1}$ |
| PARTPO | Daily outflow to particulate P | tP d $^{!}$ |
| PCONF | Max. fish biomass sustainable at current TOTP concentration | t |

| Name | Definition | Units |
|---|---|---|
| PELVOL | Volume of epilimnion directly above hypolimnion | m$^3$ x 10 |
| PEMORT | Mortality of *0. mossambious* occurring in epilimnion directly above hypolimnion | tP d |
| PHAREA | Previous day's hypolimnetic area | ha |
| PHGRMX | Maximum PHYTO growth rate from PRIPRO | d"$^1$ |
| PHYCNC | Mean lake phytoplankton-P cone. | mgP *Z*~$^l$ |
| PHYGRO | Phytoplankton growth | tP d"$^1$ |
| PHYKS | Half-sat, const, for phytoplankton | mg P *$*~$^l$ |
| PHYMU | Phytoplankton growth rate mod. by CONC | d"$^1$ |
| PHYOUT | Phytoplankton lost via outflow | tP d"$^1$ |
| PHYSDR | Phytoplankton sedimentation rate | mm$^3$ cm $^2$ d $^1$ |
| PHYSED | Phytoplankton sedimentation | tP d"$^1$ |
| PHYTO | Phytoplankton standing stock . | tP |
| PHYZEU | Phytoplankton biomass in euphotic zone | tP |
| PPCONF | Max. fish biomass sustainable at previous day's TOTP concentration | t |
| PPCHNG | Amount of particulate P transferred between epilimnion and hypolimnion | tP d |
| PRESED | Precipitation of particulate P to sediments | tP d $^1$ |
| PRINET | Net primary production rate | mgC m $^2$ hr |
| PRIPRO | Gross primary production rate | mgC m $^2$ hr |
| PSAREA | Previous day's epilimnetic area | ha |
| RAIN | Daily rainfall | m$^3$ x 10$^6$ |
| RCOM | Phytoplankton community respiration rate | mg $0_2$ m$^3$ hr"$^1$ |
| RECR(l) 2,3) | Recruitment of fish species 1, 2 and 3 | |
| RESPI | Areal respiration rate | mgC m $^2$ hr |
| RESUS | Resuspension of EPPART from sediments | tP d"$^1$ |
| RIV | Monthly outflow via rivers | m$^3$ x 10 |
| RIVCAN | Daily outflow via river and canal | m$^3$ x 10 |
| RIVIN | Daily inflow via rivers | m$^3$ x 10 |
| RIVOUT | Daily outflow via river | m$^3$ x 10$^6$ |
| RN | Monthly inflow via rivers | m$^3$ x 10 |

| Name | Definition | Units |
|---|---|---|
| RVCN | Monthly outflow via river and canal | m$^3$ x 10 |
| SCMBLD | PHYTO loss to scums | tP d"$^1$ |
| SCMBRK | Breakage loss from scum | tP d"$^1$ |
| SCMDIS | SCUM loss to DISP | tP d"$^1$ |
| SCMOUT | SCUM loss via outflow | tP d"$^1$ |
| SCMPRT | SCUM loss to EPPART | tP d"$^1$ |
| SCMSED | SCUM loss to sediments | tP d"$^1$ |
| SCUM | Standing stock of phytoplankton scums | tP |
| SEDCNG | Amount of sedimentary phosphorus transferred between hypolimnion and epilimnion | tP d |
| SEDET | Sedimentation of PARTP | tP d"$^1$ |
| SEDP | Quantity of sediments in lake | tP |
| SHAREA | Area of epilimnetic sediments | ha |
| SHCODS | Y intercept in sediment uptake/release rate: dissolved P concentration relationship for epilimnetic sediments | |
| SHCOSL | See above | |
| SHDSOL | Release of dissolved P by sediments | tP d"$^1$ |
| SHPHSD | Sedimentation of phytoplankton in epilimnion | tP d |
| SHPRDS | Y intercept in sediment uptake/release rate: dissolved P concentration relationship for epilimnetic sediments | |
| SHPRSD | Initial sedimentation of particulate P to epilimnetic sediments | tP d |
| SHRMOR | Mortality of fish occurring in epilimnion above epilimnetic sediments | tP d |
| SHRSED | Total sedimentary P in epilimnetic sediments | tP |
| SHRVOL | Volume of epilimnion | m$^3$ x 10$^6$ |
| SHSDET | Sedimentation of particulate P to epilimnetic sediments | tP d |
| SHZOEX | Particulate P loss from zooplankton to epilimnetic sediments | tP d |
| SPILL | Daily water loss via spillway | m$^3$ x 10 |
| SPL | Monthly water loss via spillway | m$^3$ x 10 |
| SPLPRP | Proportion of spillway water to DAMVOL | |
| STOUTT | Total outflow of phosphorus | tP mth"$^1$ |
| STRIVO | Canal outflow of phosphorus | tP d"$^1$ |

| Name | Definition | Units |
|---|---|---|
| STROUT | Daily outflow of phosphorus | tP d$^{T1}$ |
| STRQ | Retention coefficient | |
| STSPDP | Spillway outflow of dissolved P | tP d"$^{1}$ |
| STSPPP | Spillway outflow of particulate P | tP d"$^{1}$ |
| SUSOL | Solubilization of PARTP | tP d"$^{1}$ |
| TEMP | Mean monthly whole lake temp | °C |
| TEMPFA | Factor for regulating *0. mossambicus* growth by temp | - |
| TIN | Total monthly inflow of water | m$^{3}$ x 10$^{6}$ |
| TOT | Total monthly inflow load of P | tP |
| TOTEP | Annual total of daily EPCONC values | mg *iT*$^{l}$ |
| TOTHAR | Total fishing mortality | tP d"$^{1}$ |
| TOTHYP | Annual total of daily HYCONC values | mg *Z*~$^{l}$ |
| TOTIN | Total daily inflow of water | m$^{3}$ x 10$^{6}$ |
| TOTMOR | Total natural mortality of fish | tP d"$^{1}$ |
| TOTOUT | Total daily water loss from lake | m$^{3}$ x 10$^{6}$ |
| TOTPI | Total daily inflow load of P | tP d"$^{1}$ |
| TOTSED | HYPSED + SHRSED | tP |
| TOUT | Total monthly outflow water loss | m$^{3}$ x 10$^{6}$ |
| TSTRQ | Sum of daily retention coefficients for the month | |
| TSURF | Mean monthly water temp, at depth of lm | °C |
| VOLZEU | Volume of water in euphotic zone | m$^{3}$ |
| WIND | Mean monthly wind speed | m sec |
| YR(1) (2) | Coding for years | |
| ZEU | Depth of euphotic zone | m |
| ZFOOD | Total food source of zooplankton | tP |
| ZOGRD | Detritus ingested by zooplankton | tP d"$^{1}$ |
| ZOGRP | Phytoplankton ingested by zooplankton | tP d $^{1}$ |
| ZOODET | ZOOP lost by natural mortality | tP d"$^{1}$ |
| ZOOEG | Egestion by zooplankton | tP d"$^{1}$ |
| ZOOEXP | Total particulate P loss from zooplankton | tP d"$^{1}$ |
| ZOOP | Zooplankton standing stock | tP |
| ZOOPIN | P incorporated into ZOOP from food | tP d"$^{1}$ |
| ZOOURP | Dissolved P excreted by ZOOP | tP d"$^{1}$ |
| ZOOUT | ZOOP lost via dam outflow | tP d"$^{1}$ |

APPENDIX 6.3

## RATES AND EQUATIONS USED IN THE HARTBEESPOORT DAM ECOSYSTEM MODEL

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Unless otherwise stated, equations are derived from this study

### 1. Thermal regime

$$DAMVOL = -0.2420072 + 0.72251404\ DEP + 0.000000010276194\ DEP^6$$
$$-0.13785097\ DEP^2 + 0.0000036190398\ DEP^5 - 0.000291\ 28524\ DEP^4$$

HYPVOL = as for DAMVOL but with OXY the independent variable

1 1 1Q705 '

AREA ■ ^ms~5 DAMV0L)

HYAREA - as for AREA but with HYAREA the independent variable.

### 2. Water chemistry

(a) EPSUSL = ((0.00213333 * EPPART) + 0.068) x EPPART
HYSUSL - as above but HYPART is the independent variable

(b) SHPRSD = ((-0.002 * TIN) + 0.08) + EPPART
HYPRSD = as above but TIN and HYPART are the independent variables

### 3. Sediment - water interactions

HYSDSL = ((- GRAD * HYCONC) + HCOSL) * HYAREA
where GRAD = 0.000019444
HCOSL = CRAD * HYEQUIL

HYPRDS, SHDSOL and SHPRDS are calculated in the same manner, SHDSOL and SHPRDS utilizing GRAD, and the y-intercept and area calculated for epilimnetic sediments.

## 4. Phytoplankton

CHLZEU = 2.51 * CHLCNC

EMIN = 0.012 * CHLZEU + 0.635

ZEU = 4.605/EMIN (4.605 = $\ell$n 100%)

AMAX = 0.023 * CHLZEU * TSURF $^{1.83}$

PRIPRO = 2.12 * (AMAX/EMIN) + 35.95

RCOM = 1.22 CHLZEU - 8.57

PHGRMX = $\frac{1}{T} \ln\left(\frac{\text{ALGALC + (PRINET * 8)}}{\text{ALGALC}}\right)$

(from Peterson 1978, where T = 1 day)

PHYMU = PHGRMX * $\left(\frac{\text{EPCONC}}{\text{PHYKS + EPCONC}}\right)$

CHL = C/37.5 (Peterson 1978)

CHL = P * 0.6

Fresh mass = C * 5 (Ahlgren 1980)

C = $O_2$ * 0.375

## 5. Zooplankton sub-model

GR = 74.91 * (ZOOP ** (-0.90)) * ZOOP/100

ECDYS = 0.05 * ZOOP (Bottrell 1975)

ZOOURP = (0.05 * ZOOPIN) + (0.05 * ZOOP) (Peters and Rigler 1973)

## 6. Fish sub-model

### Length to mass conversion

AMASS1 = 0.031 ALEN1$^{3.02}$

AMASS2 = 0.160 ALEN2$^{2.53}$

AMASS3 = 0.015 ALEN3$^{2.95}$

### SL:TL conversion

ALEN2 = (TL - 0.64)/1.09 TL = total length

### Daily length increment for adults

$$\mathrm{AINCR1} = \frac{13.65 - (0.46 * \mathrm{ALEN1})}{360}$$

$$\mathrm{AINCR2} = \frac{32.33 - (0.51 * \mathrm{ALEN2})}{360}$$

$$\mathrm{AINCR3} = \frac{19.5 - (0.24 * \mathrm{ALEN3})}{360}$$

### Daily adult mortality rates

(a) Natural

Species 1 = 0.0004

Species 2 = 0.0004

Species 3 = 0.0001

(b) Fishing

Species 1 = 0.0005

Species 2 = 0.00215

Species 2 = 0.001 (0 + fish greater than 150 g)

Species 3 = 0.0001

### Daily 0 + natural mortality rates

Species 1 = $1 - e^{-0.52}$ (Nov. & Dec)

Species 1 = $(1 - e^{-0.85})/30$ (Jan. - Aug.)

Species 2 = $(1 - e^{-2.20})/30$ up to 150 g mass

Species 2 = 0.0001 greater than 150 g mass

Species 3 = $(1 - e^{-0.80})/30$

### Fecundity

Species 1 = 6 000 eggs per summer

Species 2 = 200 000 * 0.00004

Species 3 = 200 000 * 0.00012

Daily mass increment of 0 + fish

Species 1 = 0.2 g
Species 2 = 4,2 g
Species 3 = 0.29 g

Temperature factor for regulating growth rate in *0* mossambicus*

9 07ft*} TEMPFA = ~~0.0086~~
~~x (TKHP$^{2}$ ' 0™ )~~ ^ „ ^^ ^^
0.0086 x (17.75$^{2-0783}$) 13ı8
1978)

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