A PRELIMINARY INVESTIGATION OF THE EFFECTS OF HYACINTH ON ALGAL GROWTH AND WATER QUALITY.

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A PRELIMINARY INVESTIGATION OF THE EFFECTS OF WATER HYACINTH ON ALGAL GROWTH AND WATER QUALITY

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SUMMARY

In Vernon Hooper and Hartbeespoort Dams it was observed that low concentrations of algae and better water quality were associated with the occurrence of Eichhornia crassipes (Mart.) Solms. In Vernon Hooper Dam in the first six months following the introduction of hyacinth control a reduction in the cost of water treatment of 61% was reported (Anon, 1984). Musil (1983) calculated that nutrient removal by a population of Eichhornia of the size that occurred in Vernon Hooper Dam (ca 20 ha) would not have been sufficient to account for the observed reduction in water treatment costs. Also, the area occupied by Eichhornia (10 to 20% of the surface area at full supply level) was too small for shading to account for the reduction in water treatment costs. These observations led to the suggestion that Eichhornia was producing a substance which inhibited algal growth (allelopathy), therefore an investigation of the effects of water hyacinth on algal growth and water quality was initiated. This study included a review of allelopathy between aquatic plants, an experimental investigation of the allelopathic potential of hyacinth, case studies of Vernon Hooper and Hartbeespoort Dams and an assessment of the prospects of using hyacinth to improve lake water quality.

The literature review of allelopathy with particular reference to aquatic macrophytes showed that, in common with

most other supposedly allelopathic interactions, the evidence is limited and often contradictory. The data confirmed the hypothesis that allelopathy occurs in aquatic ecosystems but its general significance could not be established. Rather little research has addressed allelopathic interactions involving water hyacinth and that which has (Sharma, 1985) suggests that algae inhibit the growth of Eichhornia.

The experimental investigation of the allelopathic potential of Eichhornia in Vernon Hooper Dam, undertaken between April 1984 and May 1985, provided no evidence of allelopathy between Eichhornia and algae. Inhibitors were found on only two occasions, apparently not associated with the presence of Eichhornia. Such infrequent occurrence of inhibition could hardly significantly reduce algal standing crops and thus treatment costs.

Contrary to earlier estimates (Musil, 1983), Musil and Breen (1985) calculated that in Vernon Hooper Dam a 20 ha population of Eichhornia could remove sufficient nutrients to affect algal growth. In addition to its effect on nutrient availability Eichhornia also tends to reduce the algal standing crop principally by shading and increasing sedimentation rates. However, the Vernon Hooper case study indicated that since the introduction of hyacinth control in September 1981, the algal standing crop increased, water quality deteriorated and chemical treatment costs increased.

Thus it seems unlikely that Eichhornia produces significant quantities of an algal inhibitor and although the population of Eichhornia on Vernon Hooper Dam probably improves water quality, its effect is not sufficient to limit the algal standing crop and maintain water quality within acceptable levels.

In Hartbeespoort Dam at the peak of the infestation, Eichhornia occupied ca 60% of the surface area. Compared with periods before and after, the period of high Eichhornia cover was associated with lower chlorophyll a concentrations and an altered phytoplankton species composition. These changes were not due to reduced nutrient loading rates and nutrient availability, and were probably due to the effect of Eichhornia on other chemical and physical conditions in the impoundment. In Hartbeespoort Dam the exponential increase in algal numbers following the submergence of large areas of Eichhornia killed by the application of herbicide, suggests that the major effect of Eichhornia on the phytoplankton was probably to act as a barrier to light penetration, resulting in light limitation of algal production. In common with the investigation of the allelopathic potential of Eichhornia and the Vernon Hooper Dam case study, the Hartbeespoort Dam case study suggested that allelopathy between Eichhornia and phytoplankton was of little or no significance. However a broader based approach to the investigation of the allelopathic potential of Eichhornia is needed before it is possible to make a conclusive statement regarding the significance of allelopathy between Eichhornia and phytoplankton.

The prospect of using Eichhornia to improve the water quality of lakes was assessed. Although allelopathy does not appear to play a role, Eichhornia tends to improve water quality primarily by removing nutrients, causing light limitation of algal production and increasing sedimentation rates. Use of complete Eichhornia cover in lakes does not appear to be an economically viable option as it would prevent recreational use of the waterbody and the reduced rate of oxygen replacement would tend to reduce water quality and increase water treatment costs. Partial cover of Eichhornia would permit the recreational use of the waterbody and allow gaseous exchange between the atmosphere and the water. In Vernon Hooper Dam, a severely eutrophic impoundment, the partial cover of Eichhornia is not sufficient to contain water quality problems within acceptable limits, however in waterbodies of lower trophic status a partial cover of Eichhornia could be sufficient to significantly improve water quality especially if the impoundment was relatively shallow.

Eichhornia could be most effectively used for nutrient removal and water quality improvement if grown on artificial ponds at point source nutrient inputs. Provided diffuse source nutrient inputs to downstream waterbodies were small to point source inputs. However further research into the potential of using Eichhornia to improve water quality in mesotrophic or meso-eutrophic lakes is required.

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C O N T E N T S

1. INTRODUCTION

During recent years it has been noted at both Vernon Hooper (Shongweni) and Hartbeespoort Dams that the occurrence of high concentrations of algae, normally associated with excessive eutrophication, was dramatically reduced when water hyacinth (*Eichhornia crassipes*) became dominant in these dams. Consequently, at Vernon Hooper Dam, the Durban City Council initiated an experimental programme for managing water quality through the controlled growth of hyacinth on about 10 to 20 percent of the dam's surface by restricting the hyacinth to one arm of the dam with a floating boom. Hyacinth is harvested at the rate at which it grows and is composted on site. This experiment proved so successful that it was decided to maintain a population of water hyacinth at Vernon Hooper Dam. Figures obtained from the City Engineer's Department in Durban showed that the introduction of hyacinth control initially achieved a reduction of 61% in chemical treatment costs (Anon., 1984).

Since algal production is light dependent, the development of a surface covering of floating *Eichhornia* plants could reduce algal growth. Because the population of *Eichhornia* on Vernon Hooper Dam occupies a small proportion of the total surface area of the impoundment this is certainly not the only and probably not the major factor which accounts for the apparent reduction in the algal standing crop. In addition, Musil

(1983) reported that in the Vernon Hooper Dam the area covered by water hyacinth was too small for nutrient uptake by Eichhornia to account for the reported reduction in the algal standing crop.

An alternative explanation is that hyacinth produces a substance which inhibits algal growth. The inhibition of germination, growth or metabolism of one plant because of the release of chemicals by a different plant is termed 'allelopathy'. The phenomenon is well documented, particularly amongst higher plants of terrestrial environments. It has also been reported in algae and other aquatic plants where its principal function appears to be to provide advantage by suppression of competitors. The dilution effect in aquatic environments tends to reduce the activity of algal toxins so that they are effective only at or near the site of production. For this reason the best documented examples of allelopathy among aquatic plants are the inhibition of algal epiphyton growth on leaves, stems and roots of aquatic plants. This may also apply to hyacinths since observations have shown that when growing well, hyacinths are largely devoid of epiphyton. There is less certainty about the effectiveness of the 'inhibitors' acting at sites removed from the plant, but it is not unlikely, as evidence suggests that the inhibitors in terrestrial environments are often waste products which can be active at low concentrations.

In view of the above, a preliminary study was initiated, aimed at establishing whether future research in this connection was justified.

The research programme comprised three main tasks, namely:

- A review of allelopathy with particular reference to aquatic macrophytes.
- An investigation of the allelopathic potential of hyacinths. This task involved laboratory experiments in which water was assayed for its allelopathic potential by algal bioassay techniques.
- An assessment of the evidence that water hyacinth reduced algal blooms in Vernon Hooper and Hartbeespoort Dams. With the Vernon Hooper and Hartbeespoort Dam case studies it was attempted to answer the following questions:

  - Is there any evidence that the algal standing crop in Vernon Hooper Dam has declined since the introduction of hyacinths?
  - If a reduction is evident, to what extent can it be attributed to altered nutrient loading, inflow of toxic material and nutrient uptake by hyacinth?
  - Is there any evidence that algal problems at Hartbeespoort Dam were reduced by the presence of hyacinth?

- What are the prospects of using Eichhornia to improve water quality in Vernon Hooper Dam and other impoundments?

The case studies were post event studies and thus were anlyses of data that were not collected for this particular project, making it difficult to arrive at a conclusive statement.

2. REVIEW OF ALLELOPATHY IN AQUATIC PLANTS

1. INTRODUCTION.

There has, over the past twenty years and more, been considerable speculation on the potential significance of the inhibition of germination, growth or metabolism of one plant or species by the release of chemicals from another. Interest in this phenomenon, termed allelopathy, stems from the intuitive feeling that it might explain a variety of frequently observed processes such as the success of certain weeds and the pattern of biological succession observed in aquatic and terrestrial environments.

Most attention has been directed at terrestrial plants where although the presence of allelopathic chemicals has frequently been demonstrated, they have seldom been shown to exert a significant effect under natural conditions. Thus for example, although Muller (1966) was able to demonstrate that potentially toxic substances from *Salvia leucophylla* were absorbed by the soil and could be retained there for months, and that they were able to enter seeds and seedlings, he was not able to detect them in sufficient quantity to produce inhibition as measured in the laboratory. In a number of studies of succession (Keever, 1950, Pinder, 1975 and Allen and Forman, 1976) it has been concluded that allelopathic interactions were rarely important. However, although allelopathy may be less common than once thought, there is convincing evidence that in some instances it plays a decisive role in determining vegetation structure e.g. *Conyza canadensis* (Keever, 1950) and *Adenostoma fasciculatum* (Christensen and Muller, 1975).

The role of allelopathy in aquatic plants has been less well researched despite the fact that there are just as many reasons to intuitively expect allelopathy to be an important strategy in avoiding and/or reducing competition.

2. ALLELOPATHY BETWEEN SPECIES OF PHYTOPLANKTON

Phytoplankton in natural water bodies show considerable temporal variation in species composition and abundance. The causes of this succcession, where one species gives way to another only to return as a dominant some time later, are undoubtedly varied and reflect both the physico-chemical environment and interaction between species. Rice (1974) has suggested that fluctuations in phytoplankton numbers during succession is controlled, at least in part, by allelopathic interactions.

Szczepański (1977) concluded that although there is considerable evidence of allelopathy among phytoplankton, there is a lack of confirmatory evidence that it is a significant phenomenon in natural waters. This is perhaps not surprising because of the very considerable dilution consequent upon release of a chemical from a cell into the surrounding water. To be effective, the chemical would require one or more of the following properties:

- be produced in large amounts, which is unlikely in small unicellular plants because the energy loss would be too great;
- be highly active at very low concentrations. This is not unlikely since many algae are known to release compounds which coordinate sexual reproduction;
- be highly specific, adversely affecting target species at very low concentrations;
- be persistent so that the chemical is not rapidly denatured and can therefore accumulate to sufficiently high concentrations.

It is only during the past few years that attempts have been made to correlate laboratory studies of allelopathy with observations in the field (Keating, 1977; 1978 and Blinn, Hurley and Brokaw, 1981). Keating reported that the sequence of phytoplankton species during development of algal blooms in eutrophic Linsley Pond was related to the effects of cell free filtrates from the dominant cyanobacteria. As might be expected the filtrates were observed to either have no effect (neutral effect) or an inhibitory effect (negative effect) on bloom forming algae that preceded them in the succession, implying that they gained competitive advantage by suppressing the growth of dominant algae. The effect of filtrate on bloom-forming species which succeeded them were either neutral or positive, implying that as the population developed it actually stimulated growth of some competitors which would ultimately replace them as the dominant species. In situ observations over a period of five years showed that diatom bloom populations varied inversely with the extent of development of the cyanobacteria populations which preceded them (Keating, 1978). Implication of allelopathy was confirmed by showing that cell-free filtrates from cultures of the dominant cyanobacteria inhibited growth of diatoms isolated from the lake water during blooms of cyanobacteria.

Allelopathy has also been implicated in the spatial zonation of algae in streams. Blinn, Hurley and Brokow (1981) found that <u>Thalassiosira pseudonana</u> replaced <u>Chaetoceros muelleri</u> at downstream sites and preliminary culture studies indicated that allelopathy may be operative.

Although Juttner & Faul (1984) did not attempt to relate the results of their laboratory studies to field observations, their results show that the interactions between algae are complex. They used five algal species to assay the growth inhibiting or stimulating properties of nutrient supplemented, fractionated and unfractionated water taken from a shallow eutrophic lake in SW-Germany between May and October 1982. During this period the chlorophyll concentrations were high (generally above 100 mg/m^3). There were marked differences in the response of the different algal species and in the growth inhibiting or stimulating activity of the fractionated and unfractionated lake water. Often, inhibiting as well as stimulating properties were observed in the same water sample. Thus, in many cases the growth effects observed with unfractionated lake water were the result of the sum of opposing individual effects.

These data imply that allelopathy is involved in the complex interactions which determine algal succession but stress also that there can be a wide range of responses to the same chemical : in some species growth may be retarded whilst in others it is stimulated. The ecological significance is therefore difficult to assess.

3. ALLELOPATHY BETWEEN AQUATIC MACROPHYTES AND MICROFLORA

Aquatic macrophytes are regarded as macroscopic algae and higher plants colonising aquatic environments. These plants provide a surface on which other organisms, particularly microscopic plants, can become attached and remain in the illuminated part of the water column. In addition, since plant cell membranes always allow some material to diffuse out, these sites also provide potential benefits for organisms colonizing submerged plants. However, as these surfaces, particularly leaves and photosynthetic organs become colonized and covered by these microscopic organisms (epiphyton), so the light reaching the photosynthetic apparatus of the host plant is progressively reduced, and growth of the plant may be depressed. Since it has frequently been observed that not all submerged plants or plant organs develop epiphyton (Howard - Williams, 1978; Anthoni *et al.*, 1980), it is tempting to speculate on the involvement of allelopathy in the control of epiphyton development, and possibly also in the growth of phytoplankton within the macrophyte community.

Conover and Sieburth (1963) demonstrated anti-bacterial activity in *Sargassum natans* and *S. fluitans* with maximum activity being observed in plants from communities in the centre of distribution of these species. They suggested that perhaps the dynamics of the whole community of plants, animals and bacteria might be controlled by the interplay of allelopathic chemicals. Extracts from another marine brown alga, *Fucus vesiculosus*, have been shown to inhibit the growth of unicellular marine algae (McLachlan and Craigie, 1964).

In freshwater environments the macroalgae have received most attention. Anthoni *et al.* (1980) demonstrated that crude extracts of *Chara globularis* depressed the rate of photosynthesis of the diatom *Nitzschia palea* and naturally occurring phytoplankton populations. The active components so far isolated are elemental sulphur and two organic sulphur - containing compounds. The two organic sulphur compounds have also been isolated from other related freshwater algae: Chara hispida, C. baltica, Nitella translucens and Tolypella nidifica (Wium-Andersen et al., 1982). Elemental sulphur, the most potent of the three substances extracted from C. globularis has recently also been found to occur in the submerged vascular macrophyte Ceratophyllum demersum (Wium-Andersen et al., 1983). Extracts from floating duckweed (Lemna minor), have been shown to have a bacteriostatic effect on Sphaerotilus natans, but not on other bacteria tested.

A number of studies have shown reduced phytoplankton development in areas supporting aquatic macrophytes (Hasler and Jones, 1949; Goulder, 1969; McVea and Boyd, 1975 and Brammer, 1979). Sharma (1985) has shown reduced growth of the acquatic macrophyte (Eichhornia crassipes) in a tank supporting phytoplankton. However, these data could not unequivocally be related to allelopathy since competition for nutrients, light and other factors could also explain these observations. At the present time therefore it is not possible to assess the ecological significance of allelopathy between aquatic macrophytes and microflora.

4. ALLELOPATHY BETWEEN AQUATIC MACROPHYTES

Although allelopathy is generally regarded as an interaction between two or more species, this is not always so since allelopathy may result in inhibition of a stage in the life cycle (e.g. germination) of a single species. In such a case, inhibition of germination ensures the presence of a viable seed-bank which can re-establish the population should it be disturbed. Aqueous extracts of Typha latifolia leaves and soil water emanating from Typha - dominated marshes have been shown to inhibit germination and reduce seedling growth of T. latifolia (McNaughton, 1968). More recently, Grace (1983) has shown that T. latifolia does not inhibit the germination of its seeds by the release of allelopathic substances, but that high concentrations of litter extract inhibit germination temporarily when water mould populations are allowed to develop. Decaying aerial organs of Typha latifolia and T. augustifolia, and other emergent aquatic macrophytes (Eleocharis palustris, Glycera aquatica, Schoenoplectus lacustris and Acorus calamus) have been reported to reduce growth and kill seedlings of the reed Phragmites communis (Szczpańska, 1971).

The common observation that aquatic macrophytes tend to occur in monotypic (monospecific) stands (Anthoni *et al*., 1980; Wium-Andersen *et al*., 1982; Kulshreshtha and Gopal, 1983) make it tempting to speculate that allelopathy between aquatic macrophytes may hold considerable significance for the structure and functioning of these communities. Although *Najas marina* usually occurs in mixed stands, *Myriophyllum spicatum* seldom occurs in these stands (Agami, 1982; cited in Agami & Waisel, 1985). It appears that the growth of both *N. marina* and *M. spicatum* is inhibited by the presence of the other species (Agami & Waisel, 1985). Kulshreshtha and Gopal (1983) showed that growth of shoots of *Ceratophyllum muricatum* was stimulated by extracts of its own shoots but was adversely affected by extracts of other rooted macrophytes (*Hydrilla verticillata* and *Hygrorhiza* sp.). Growth was affected very little or only slightly stimulated by extracts from three floating aquatic plants viz. *Eichhornia crassipes*, *Pistia stratiotes* and *Salvinia molesta*.

Some floating macrophytes are, however, known to release inhibitory metabolites into the water. Wolek (1974; cited in Szczpański, 1977) reported that high concentrations of metabolites released into the nutrient culture media from Lemna minor, L. gibba, Spirodela polyrrhiza and Wolffia arrhiza were inhibitory to other members of the family Lemnaceae. At low concentrations, however, the metabolites had a stimulatory effect. A more detailed analysis of the interaction between Spirodela polyrrhiza and Wolffia arrhiza confirmed that inhibitory metabolites were released into the culture media (Wolek, 1979).

These data show that in common with most other supposedly allelopathic interactions, the evidence is scattered and sometimes contradictory. Rather little research appears to have addressed allelopathic interactions involving the water hyacinth, Eichhornia crassipes.

5. NATURE OF ALLELOPATHIC COMPOUNDS

Very little research has been directed at the identification of compounds associated with allelopathy. Most chemical inhibitors are compounds that have been termed 'secondary' substances (Fraenkel, 1959; and Whittaker and Feeny, 1971) because they are of sporadic occurrence. It seems probable that research will show that allelopathy is based on the same or related compounds in a variety of organisms (Whittaker, 1970 and Whittaker and Feeny, 1971). Too little is currently known to permit generalization.

6. CONCLUSIONS

The ecological significance of chemical allelopathy (allelochemics, Barbour, 1973) is still not clear and much of the evidence is circumstantial, being based on correlation. To establish chemical allelopathy unequivocally the following sequence of steps are recommended:

1. negative association between the two species must be shown;
2. cause and effect should be established through experimentation;
3. where a chemical is involved its identity and the nature of its effect should be established;
4. the presence of the factor(s) should be demonstrated in the field, and should be shown to be operative;
5. the viability of the factor(s) in the field should be established.

There does not appear to be a single instance in the literature where all of these requirements have been met. The work of Wium-Andersen and his colleagues on Chara globularis is perhaps the most complete for aquatic plants. Their data confirm the hypothesis that allelopathy does take place in aquatic ecosystems but its general significance has not yet been established.

3. ALLELOPATHIC POTENTIAL OF EICHHORNIA IN VERNON HOOPER DAM

1. INTRODUCTION

Eichhornia crassipes first became a serious problem in Vernon Hooper Dam in 1974 when it covered the entire water surface (*ca* 80 ha) for the first time. During the winter of 1975 it was completely removed manually, only to be washed into the dam again as a result of heavy rains in December 1975. Continued manual removal could not contain the *Eichhornia*. By January 1981 *Eichhornia* again covered the entire surface (*ca* 80 ha) and by September 1981 it had been completely removed manually (Figure 5.3; Anon, 1984). As a result of tests by the Durban City Engineers on the nutrient uptake capacity of *Eichhornia*, the Durban City Engineer's Department decided to retain a population of *Eichhornia* in the impoundment. A floating boom was suspended across the Mlazi sector of the impoundment (Figure 3.1) in September 1981, confining the *Eichhornia* to an area of *ca* 20 ha at full supply level. The Umgeni Water Board's staff at the dam attempt to maintain *Eichhornia* cover at between 8 and 15 ha; however, rapid growth during summer usually results in the expansion of the population to 20 ha.

Three observations relating to the presence of *Eichhornia*

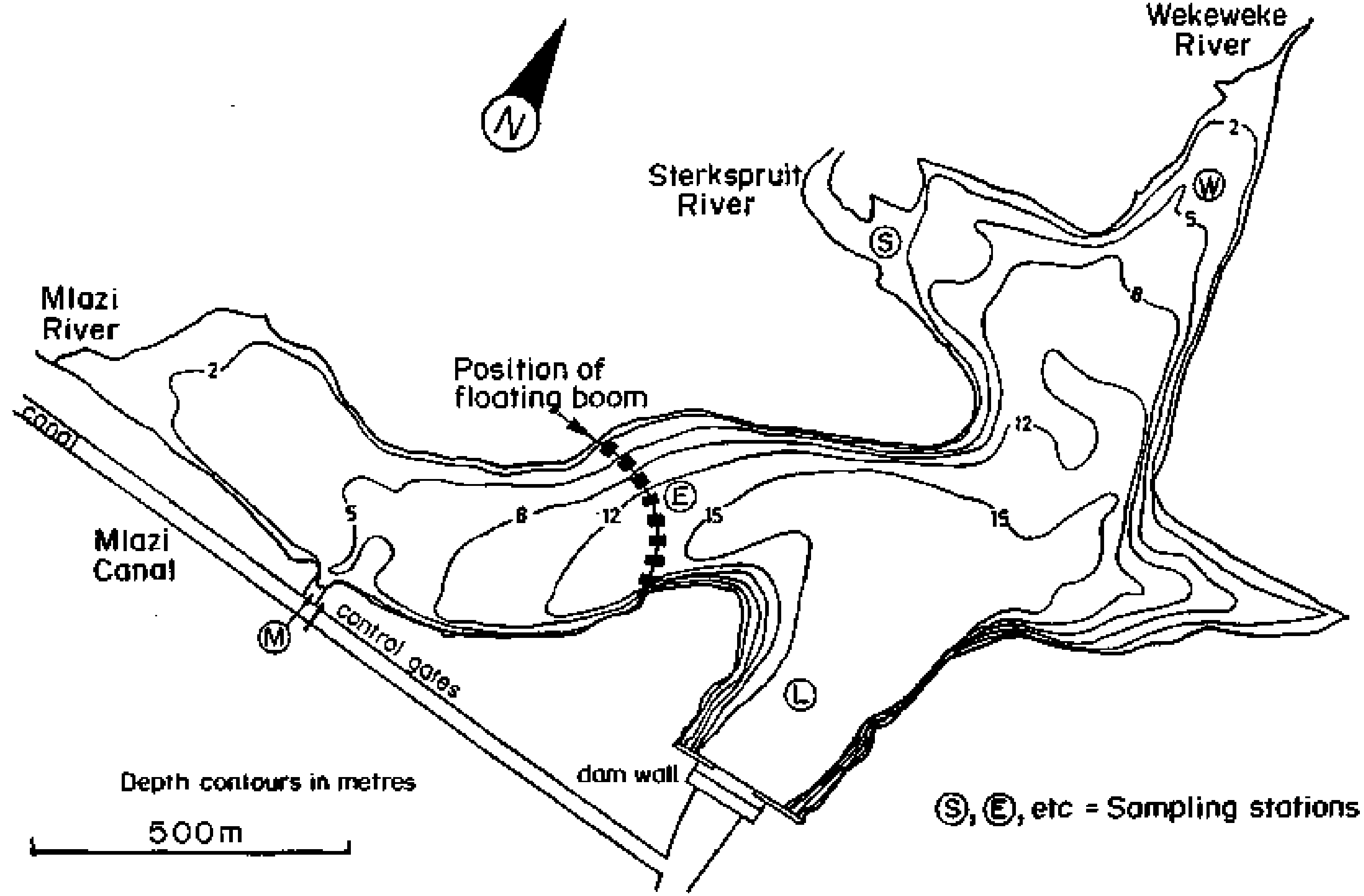

Figure 3.1: Map of Vernon Hooper Dam showing the positions of sampling stations and floating boom used to confine Eichhornia to the Mlazi sector of the impoundment. The five sampling stations are the Lake station (L), Eichhornia (E), Mlazi River (M), Sterkspruit river (S) and the Wekeweke river (W). All the water from the Mlazi river is usually diverted into the Mlazi canal and enters the impoundment at the Mlazi river station.

on Vernon Hooper Dam suggested that *Eichhornia* produced a substance which inhibited algal growth:

1. there was a large decrease in water treatment costs following the introduction of hyacinth control in September 1981. In the first six months following the introduction of hyacinth control water treatment costs were reduced by 61% and by 45% in the second six months;
2. Musil (1983) calculated that a 20 ha population of *Eichhornia* was too small for nutrient uptake by the plants to account for the improved water quality;
3. the area occupied by *Eichhornia* (between 10 and 25% of the surface area of the impoundment) was generally too small for shading to be of significance.

A number of bioassays were performed between March 1984 and May 1985 to test for the presence of inhibitors associated with the inflows or the *Eichhornia* in Vernon Hooper Dam.

2. METHODS

The *Selenastrum capricornutum* Printz algal assay was used to test for the presence of inhibitors in a) water from Vernon Hooper Dam, b) medium in which *Eichhornia* had grown and c) extracts of roots and decaying petioles of Eichhornia. The bioassay involves the measurement of the response of Selenastrum in batch cultures to the chemicals present in the sample.

a. Preparation of nutrient media

The Provisional Algal Assay Procedure medium (PAAP; National Eutrophication Research Program, 1971) referred to as complete PAAP, was used to grow the alga and Eichhornia in the laboratory prior to bioassay. PAAP medium without nitrogen and phosphorus (PAAP-NP) was prepared for resuspending centrifuged algal inocula and for the preparation of root extracts. Stock culture solutions of PAAP, PAAP without nitrogen (PAAP-N) and PAAP without phosphorus (PAAP-P) were prepared at ten times the concentration of complete PAAP for addition to some of the bioassay flasks. The PAAP-N was prepared by excluding the $NaNO_3$ stock solution and the PAAP-P was prepared by replacing the K_2HPO_4 stock solution with a KCl solution.

b. Preparation of algal inocula

Selenastrum capricornutum strain number [WR 100], was obtained from the National Institute for Water Research, CSIR. The alga was maintained in active growth phase by regular subculturing into fresh medium and cultures were maintained at 24 $\pm 1^{\circ}C$ in continuous light (90 $\mu E/m^2/s$).

Prior to use in a bioassay, one litre of a one-week old culture was centrifuged at ca 2 500 x g for five minutes. The supernatant was discarded and the algal pellet resuspended in 250 ml of PAAP-NP. One millilitre of this culture was used as the algal inoculum of each bioassay flask.

c. Lake water bioassay

In this bioassay surface samples, integrated over a depth of one metre, were collected from five stations in Vernon Hooper Dam. The positions of the five stations (Lake, *Eichhornia*, Wekeweke, Sterkspruit and Mlazi) are shown in Figure 3.1. Samples were transported to the laboratory within five hours of collection and if they could not be assayed immediately, they were stored overnight in the dark at 10°C. The procedure used is shown schematically in Figure 3.2. Initially, four treatments were used viz. filtered sample, filtered sample plus stock PAAP to give the final concentration of complete PAAP, filtered sample plus stock PAAP-N and filtered sample plus stock PAAP-P. In later bioassays the number of treatments was increased by adding different amounts of stock PAAP to some of the flasks. Also, a portion of the sample was first autoclaved, then filtered, divided into 50 ml aliquots and various stock solutions added. The range of treatments included in each bioassay are evident in the figures showing the results (Figures 3.3 to 3.9). After two weeks

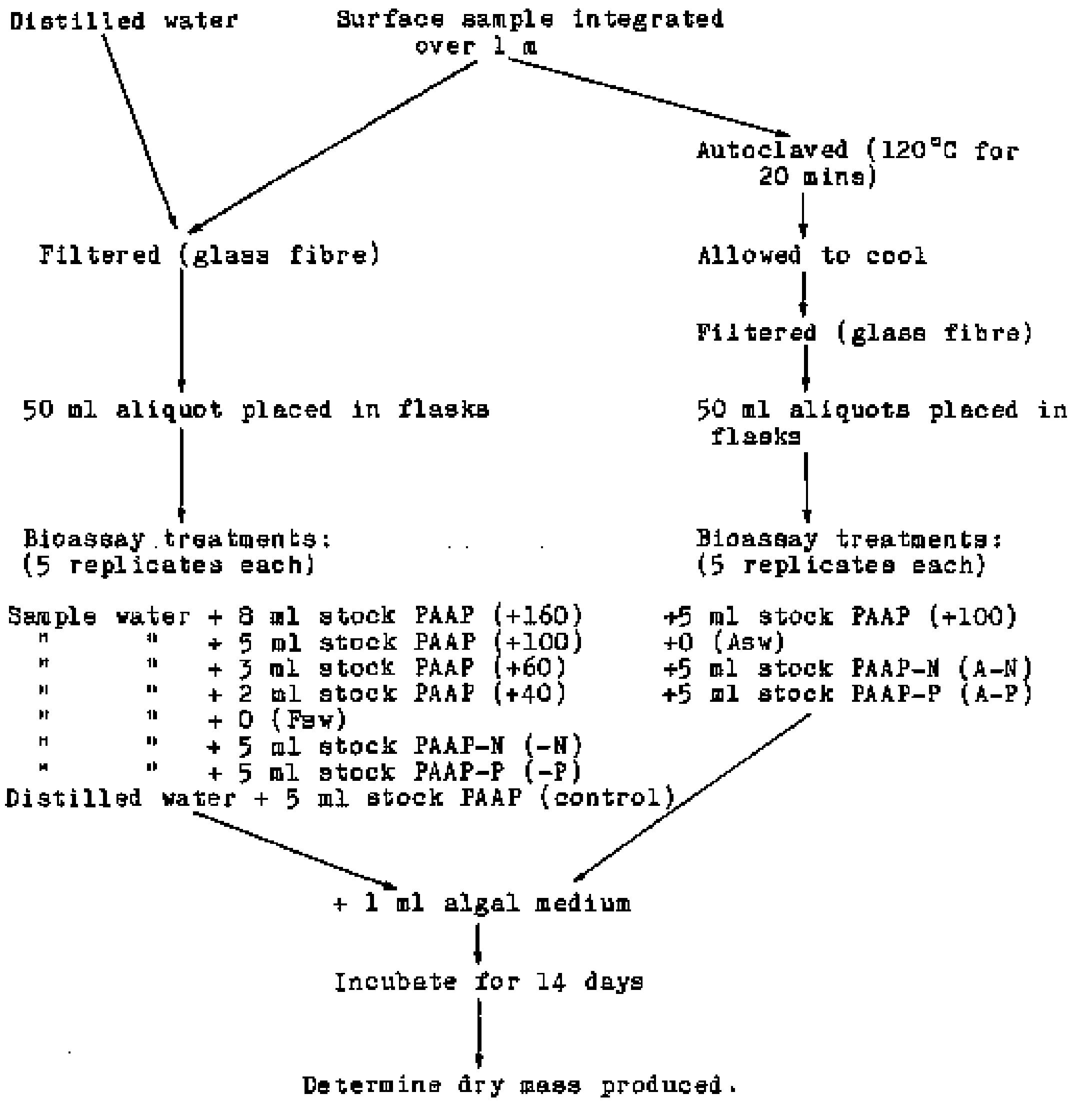

Figure 3.2: Flow diagram showing the procedure used for the lake water bioassays. A key to the abbreviations used is given in Appendix 1.

of growth in continuous light (90 $\mu E/m^2/s$) at 24 ± 1°C and with continuous aeration, the dry mass of Selenastrum produced was determined gravimetrically using prewashed, predried and preweighed Whatman glass fibre filters. The yield (as dry mass produced per litre) of each of the five replicates was determined and means and Least Significant Differences at the 95% level of confidence (L.S.D. 95%) were calculated. The increased number of treatments since August 1984 permitted the analysis of the relationship between yield and amount of PAAP medium supplied. With this approach it was thus possible to ensure that the harvest interval was appropriate and that the evidence of inhibited growth rate was not confounded with the effect of too short a growth period.

d. Assay of media in which Eichhornia had grown

Eichhornia plants were grown in 20 litres of complete PAAP for up to two weeks, after which the media were assayed for evidence of allelopathy. In all these bioassays, except the first, the 20 l of PAAP was prepared using distilled water (Table 3.1) in 25 l buckets which were painted black. The surface of the control treatments (no plants added) were covered with black plastic to reduce light penetration. In the experiments in which marginal form (small) plants were used the buckets were covered with black plastic and three to four plants, each bearing six to eight-leaves, were introduced through holes in the

Table 3.1: Variations in the method used for the bioassay of media in which *Eichhornia* had grown.

	March '84	April '84	June '84	Aug.'84	May '85
Form of plant used	Marginal	Marginal	Marginal	Marginal	Central
Water used to make medium	Tap water	Distilled	Distilled	Distilled	Distilled
No. of buckets:					
with *Eichhornia* (experimental)	6	6	6	6	4
without *Eichhornia* (control)	2	2	4	4	4
Weeks after which samples taken	1	1	1&2	1&2	2
Treatments used:*	S S+100	S S+100	S S+100 S+60 S+40	S S+100 S+50	S+100 S+75 S+50 S+25
No. of replicates per treatment	1(n=1)	1(n=1)	1(n=1)	1(n=1)	4(n=4)

*S = filtered sample + algal inoculum
S+100 = filtered sample + 5 ml stock PAAP to give the concentration of complete PAAP + algal inoculum.
S+75 = filtered sample + 3,75 ml stock PAAP to give a concentration of 75% of complete PAAP + algal inoculum.
S+60 = filtered sample + 3 ml stock PAAP to give a concentration of 60% of complete PAAP + algal inoculum.
S+50 = filtered sample + 2,5 ml stock PAAP to give a concentration of 50% of complete PAAP + algal inoculum.
S+40 = filtered sample + 2 ml stock PAAP to give a concentration of 40% of complete PAAP + algal inoculum.
S+25 = filtered sample + 1,25 ml stock PAAP to give a concentration of 25% of complete PAAP + algal inoculum.

plastic. When central form (large) plants were used (May 1985) only one plant could be placed in each bucket. In March, April, June and August 1984 Eichhornia plants were grown at 24 ± 1°C in continuous light (90 μE/m^2 /s). Because of their large size, the central form plants used in the May 1985 bioassay had to be grown in a greenhouse at Pietermaritzburg. In this greenhouse the May 1985 air temperatures ranged between 10 and 29°C and on an overcast day the light was reduced from ca 430 μE/m^2/s to ca 90 μE/m^2/s.

After a period of up to two weeks the plants were removed from the buckets, the culture solution was stirred well and filtered through washed glass fibre filters and assayed. In later bioassays a dilution series was included in which varying amounts of PAAP were added (Table 3.1). Means were calculated, but because of insufficient data (except in the May 1985 bioassay) confidence limits could not be attached to the means.

e. Root extract bioassay

Twenty central form Eichhornia plants were used to obtain 20g of fresh root material, one gram from each plant. The material was blended for five minutes in 500 ml of PAAP-NP at room temperature (ca 22° C) before filtration through glass fibre filters. In the first bioassay (April 1984) four serial ten-fold dilutions of this stock extract were prepared and used in the bioassay. Results of this bioassay indicated that more detailed examination of extracts of lower concentrations would be more useful. Thus, in subsequent bioassays (June, August and September 1984), the stock extract was diluted to 1% and then further to 0,75; 0,5; 0,25 and 0,1%. In the April and June 1984 bioassays the treatments included a control (no extract + PAAP) and for each dilution there was one flask to which stock PAAP was added to give the concentration of complete PAAP and one to which no PAAP was added. In addition to the treatments described above the August and September 1984 bioassays had three additional treatments:

2 ml of stock PAAP was added to 50 ml of extract to give a concentration of 40% of complete PAAP;
3 ml of stock PAAP was added to 50 ml of extract to give a concentration of 60% of complete PAAP; and
8 ml of stock PAAP was added to 50 ml of extract to give a concentration of 160% of complete PAAP.

Selenastrum was added and the cultures were aerated under continuous light (90 $\mu E/m^2/s$) at 24±1°C. After two weeks the dry algal mass produced was determined gravimetrically.

f. Decaying matter bioassay

Submerged, decaying petioles of central form *Eichhornia* plants were collected from Vernon Hooper Dam in May 1985. Five hundred grams (fresh weight) of this material was placed in 5 l of distilled water, mixed well with a glass rod and allowed to stand in the dark at 10°C for *ca* 15 hours. The liquid was strained through cheesecloth; some was autoclaved and cooled and both fresh and autoclaved extracts were filtered through glass fibre filters. The extract was then divided into 50 ml aliquots and seven treatments, each with five replicates, were applied:

A = autoclaved filtered extract + algal inoculum

A+100 = autoclaved filtered extract + 5 ml stock PAAP to give the concentration of complete PAAP + algal inoculum

F = filtered extract + algal inoculum

+160; +100; +60 and +40 = filtered extract + stock PAAP to give concentrations of 160%, 100% (complete); 60% and 40% of complete PAAP respectively + algal inoculum

d+160; d+100; d+60 and d+40 = as above but distilled water was substituted for the filtered extract.

The cultures were aerated and incubated at 24 ± 1° C in continuous light (90 $\mu E/m^2/s$) for two weeks, after which the dry algal mass produced was measured gravimetrically.

Means and Least Significant Differences (95%) were calculated. The slopes of plots of algal yield against amount of PAAP added for control and experimental treatments were compared with the expected slope (calculated from yield coefficients of Selenastrum and the amount of PAAP added).

3. RESULTS

a. Lake water bioassay

March 1984 (Figure 3.3): The addition of PAAP to Sterkspruit water resulted in a significant increase in the algal yield compared with the filtered Sterkspruit water whereas the increase in yield at the Eichhornia and Lake stations was not significant. However, in this bioassay a very concentrated PAAP stock solution was added to the filtered sample water and some precipitation later became evident in the stock PAAP. Use of this concentrated PAAP stock was discontinued. The low yield of the control and the presence of precipitates in the concentrated PAAP stock made the results unreliable and conclusions about the presence of inhibitors could not be drawn.

April 1984 (Figure 3.4): At all stations the algal yields of the filtered samples with complete PAAP were significantly ($p < 0,05$) higher than the filtered unspiked

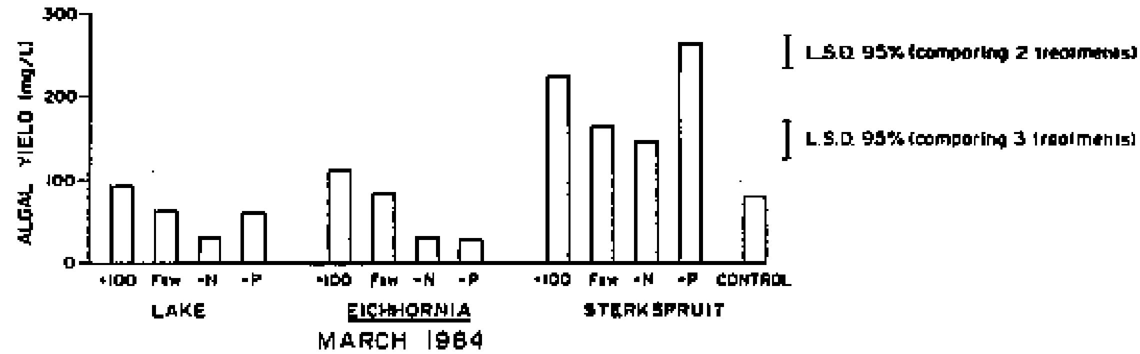

Figure 3.3: Results of the lake water bioassay of samples collected from Vernon Hooper Dam in March 1984. A key to the abbreviations used is given in Appendix 1.

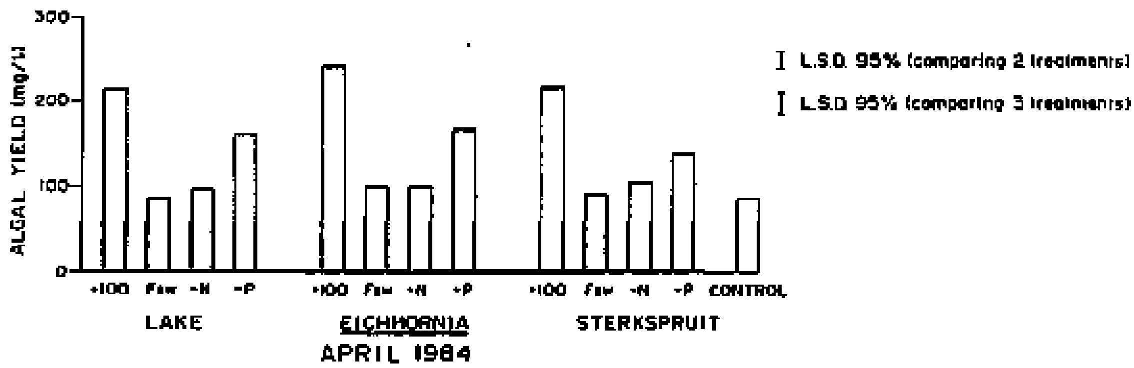

Figure 3.4: Results of the lake water bioassay of samples collected from Vernon Hooper Dam in April 1984. A key to the abbreviations used is given in Appendix 1.

sample yields. Also, at all stations the combined yields of the filtered sample water and the control were not significantly different from the yields of the filtered sample water with complete PAAP, indicating that inhibitors were not present.

June 1984 (Figure 3.5): The yields of the filtered sample water with complete PAAP treatments were significantly ($p<0,05$) higher than the filtered sample yields at all stations despite the magnitude of the L.S.D. The combined yields of the filtered sample water and the control were not significantly different from the yields of the filtered sample water with complete PAAP, indicating that inhibitors were not present.

August 1984 (Figure 3.6): The yields of the filtered sample water with complete PAAP treatments were significantly ($p<0,05$) higher than the filtered sample yields at all stations. The combined yields of the filtered sample water and the control were not significantly different from the yields of the filtered sample water with complete PAAP. Also, the mean slopes of the algal yield against amount of PAAP added (Table 3.[illegible] of the stations and control were not significantly ($p<0,05$) different, indicating that no inhibitors were present.

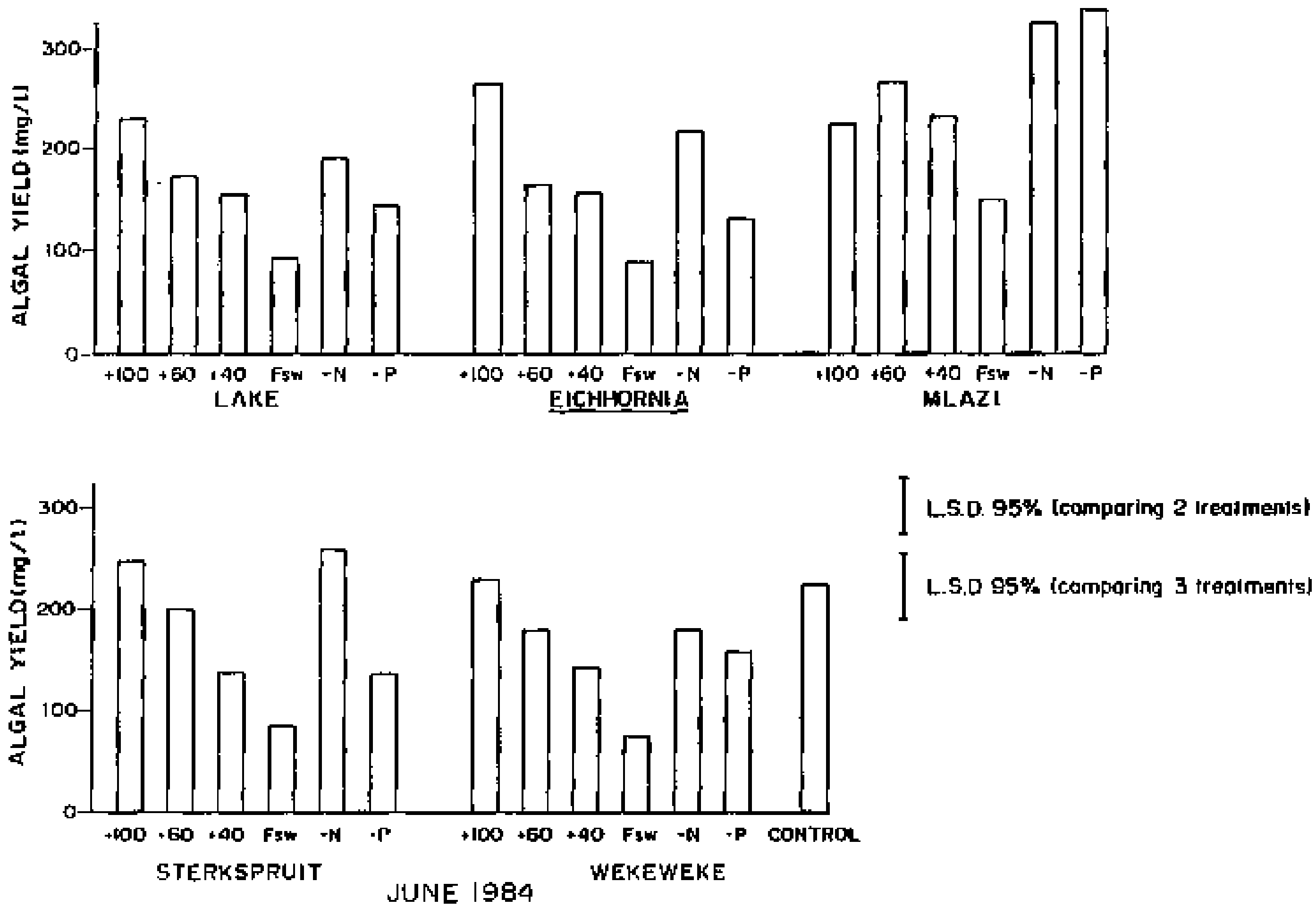

Figure 3.5: Results of the lake water bioassay of samples collected from Vernon Hooper Dam in June 1984. A key to the abbreviations used is given in Appendix 1.

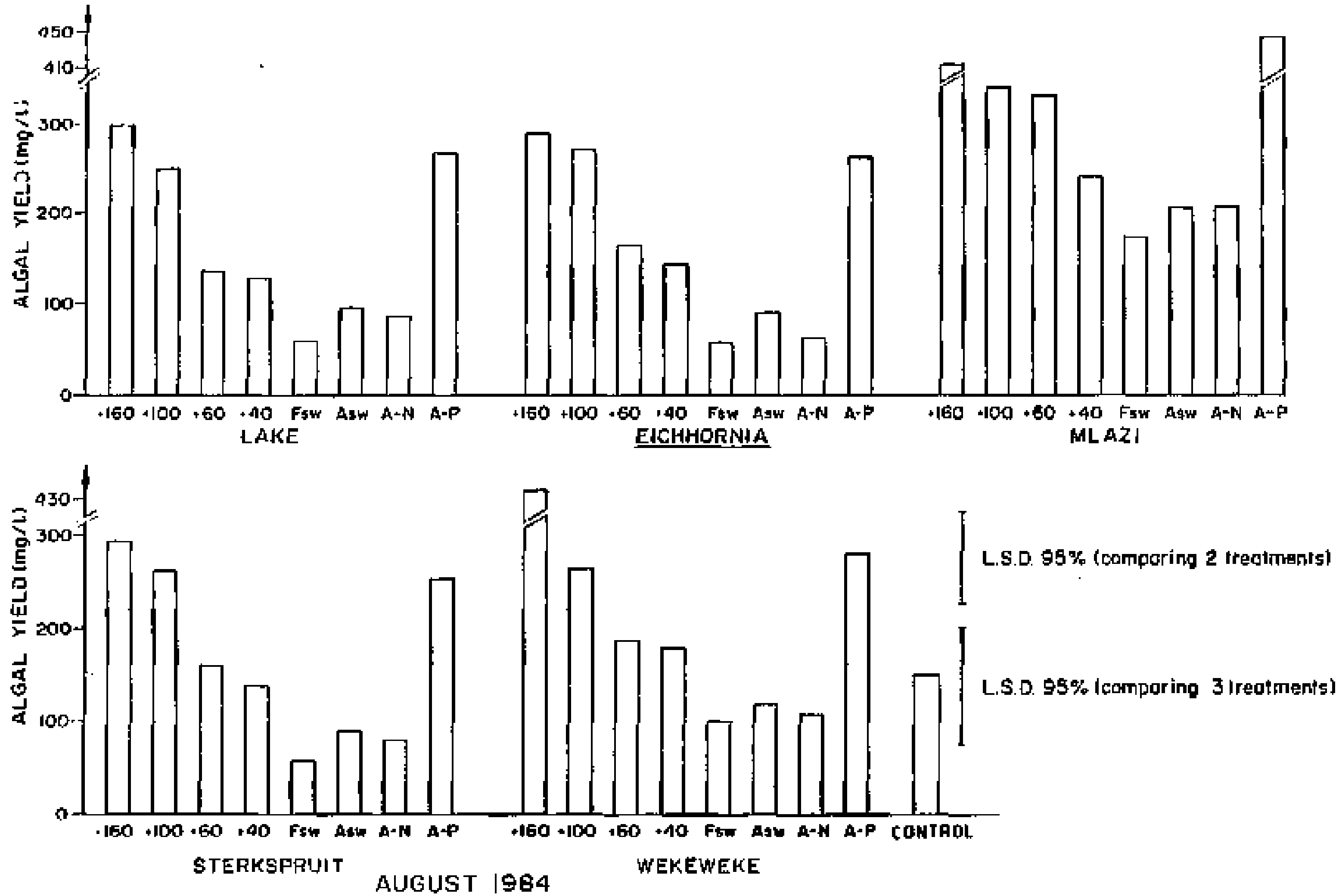

Figure 3.6: Results of the lake water bioassay of samples collected from Vernon Hooper Dam in August 1984. A key to the abbreviations used is given in Appendix 1.

Table 3.2: Mean slopes of algal yield against amount of PAAP added for the August 1984 bioassay.

Station	Mean slope (n=5)
Lake	1,58
Eichhornia	1,46
Mlazi	1,45
Sterkspruit	1,82
Wekeweke	1,87
Control	1,69

September 1984 (Figure 3.7): The yields of the filtered sample water with complete PAAP were significantly ($p<0,05$) higher than those of the filtered sample water treatments at all stations except the Lake station, indicating that inhibitors may have been present at the Lake station. The combined yields of the filtered sample water and the control of the Lake and Mlazi stations were significantly ($p<0,05$) lower than the yields of the filtered water samples with complete PAAP for these stations, suggesting the presence of inhibitors at the Lake and Mlazi stations. However, an analysis of variance

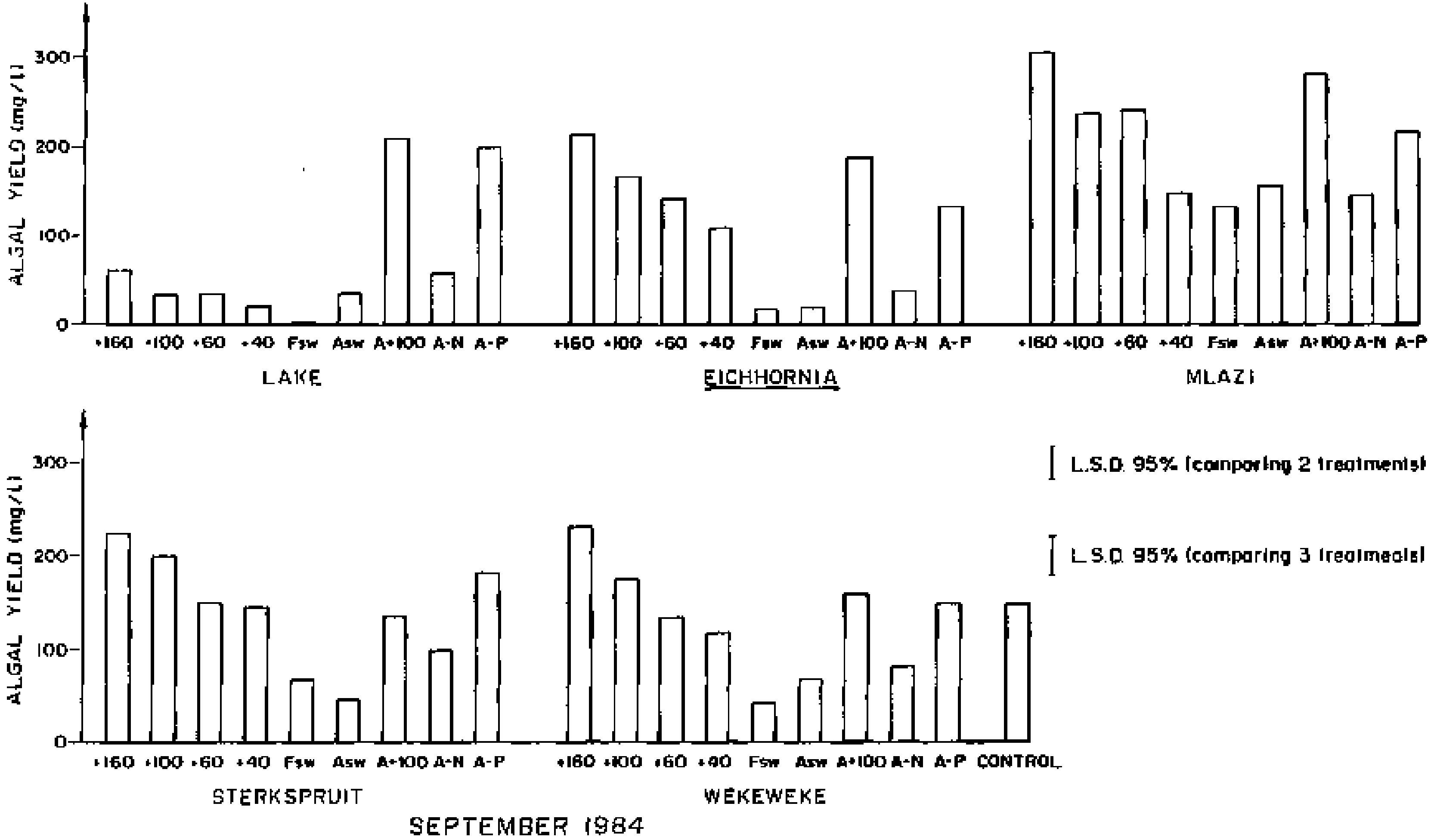

Figure 3.7: Results of the lake water bioassay of samples collected from Vernon Hooper Dam in September 1984. A key to the abbreviations used is given in Appendix 1.

of the slopes of algal yield against amount of PAAP added (Table 3.3) indicated that only the slope of the Lake station was significantly lower than the other slopes. This suggests that inhibitors were present in the Lake station only.

Table 3.3: Mean slopes of algal yield against amount of PAAP added for the September 1984 bioassay.

Station	Mean slope (n=5)
Lake	0,34
Eichhornia	1,14
Mlazi	1,10
Sterkspruit	1,00
Wekeweke	1,14

The yields of autoclaved filtered Lake water with complete PAAP and with PAAP-P were significantly ($p<0,05$) higher than the yields of all other Lake water treatments, suggesting that the putative inhibitor was heat-labile and when it was destroyed by autoclaving, the algal yield was determined by nitrogen availability.

December 1984 (Figure 3.8): The addition of complete PAAP to filtered water from all stations resulted in a significant ($p<0,05$) increase in the algal yield relative to the filtered sample treatments. At all stations except Wekeweke, the combined yields of filtered samples and the controls were not significantly different from the yields of the filtered samples plus complete PAAP. Also, the Wekeweke slope of the algal yield against the amount of PAAP added was significantly lower than the slopes of the other stations (Table 3.4).

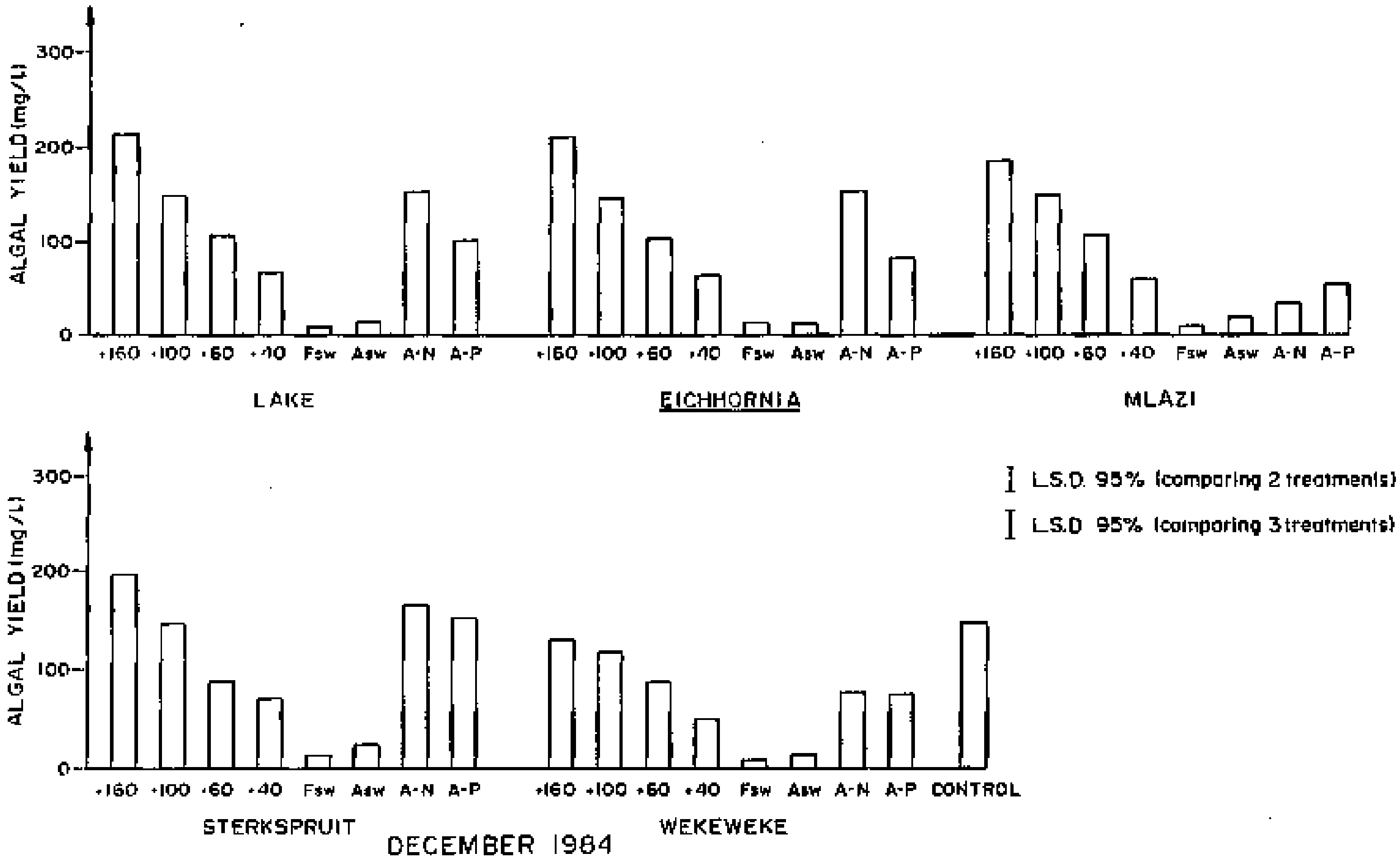

Figure 3.8: Results of the lake water bioassay of samples collected from Vernon Hooper Dam in December 1984. A key to the abbreviations used is given in Appendix 1.

Table 3.4: Mean slope of algal yield against amount of PAAP added for the December 1984 bioassay.

Station	Mean slope (n=5)
Lake	1,28
Eichhornia	1,23
Mlazi	1,10
Sterkspruit	1,16
Wekeweke	0,77

These data suggest strongly that an inhibitor was present at the Wekeweke station.

May 1985 (Figure 3.9): The slopes of the algal yield against amount of PAAP added of all the stations were not significantly different from the control; however, the measured slopes (Table 3.5) were considerably lower than the expected slope of 1,50 calculated from nitrogen and phosphorus yield coefficients of *S. capricornutum* and the amount of PAAP added. This suggests that there were inhibitors in all stations and in the control. The presence of inhibitors in the control indicated experimental error, and therefore, no reliable conclusions could be drawn from the data.

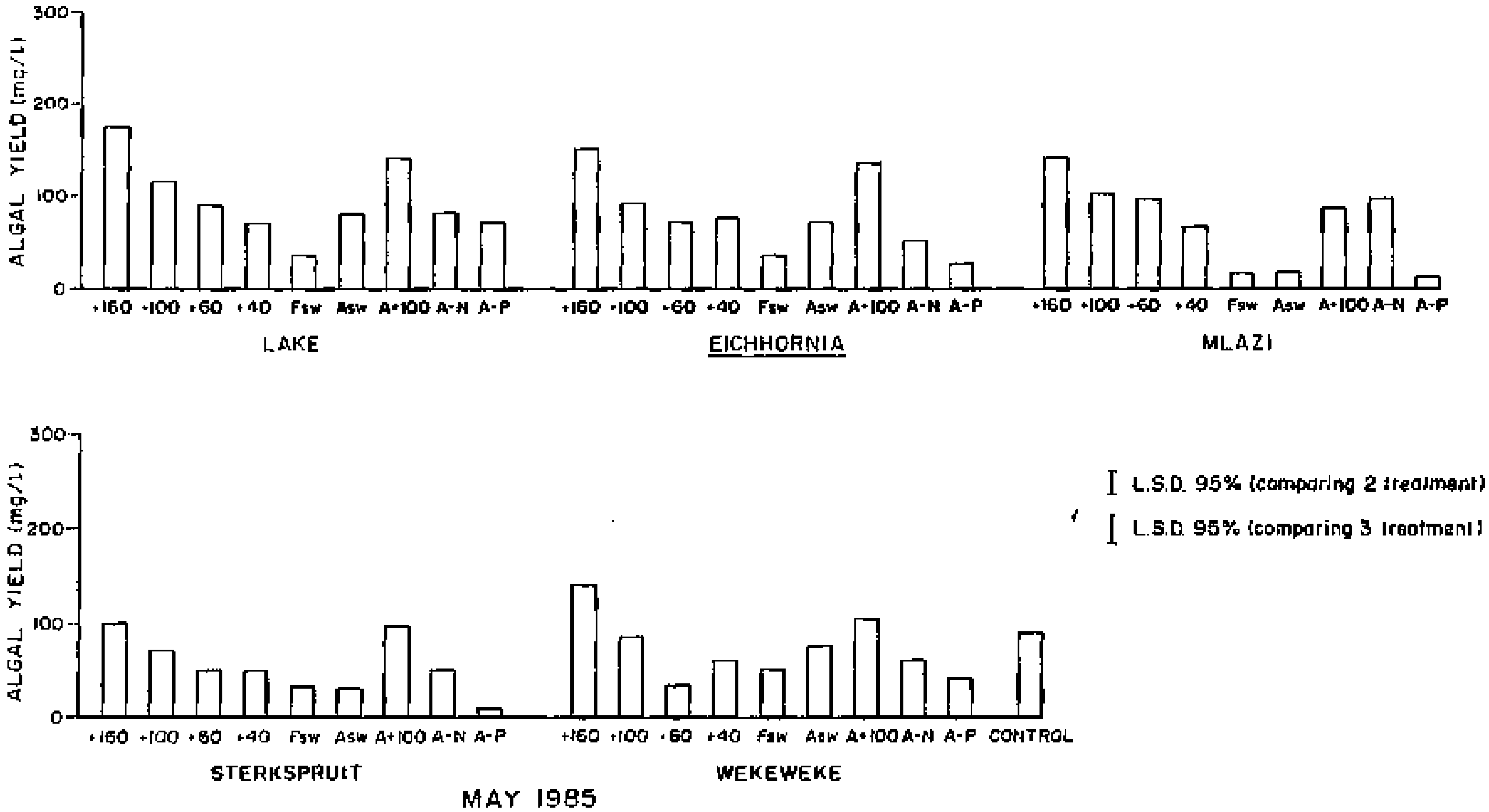

Figure 3.9: Results of the lake water bioassay of samples collected from Vernon Hooper Dam in May 1985. A key to the abbreviations used is given in Appendix 1.

Table 3.5: Mean slope of algal yield against amount of PAAP added for the May 1985 bioassay.

Station	Mean slope (n=5)
Lake	0,78
Eichhornia	0,67
Mlazi	0,73
Sterkspruit	0,41
Wekeweke	0,61

b. Assay of media in which *Eichhornia* had grown (Table 3.6)

March 1984: The algal yields of the media with *Eichhornia* plus complete PAAP (111 mg/l) and the control (139 mg/l) were much higher than the yield of the treatments without *Eichhornia*, and with and without complete PAAP. The addition of PAAP to the treatment without *Eichhornia* resulted in a very small increase (19 mg/l) in the algal yield relative to the increase in the treatments with *Eichhornia* (96 mg/l). This suggests that there were inhibitors in the media without *Eichhornia*, whereas, inhibitors were absent from the media with *Eichhornia*.

Table 3.6: Results of the bioassays of media in which *Eichhornia* had grown.

	Mean algal yield (mg/l)			
	Growth after Week 1		Growth after Week 2	
	With *Eichhornia*	Without *Eichhornia*	With *Eichhornia*	Without *Eichhornia*
MARCH 1984				
S	15	4		
S+100	111	23		
Distilled water + 100		139		
APRIL 1984				
S	105	315		
S+100	278	457		
JUNE 1984				
S	41	219	4	73
S+40	111	267	37	120
S+60	142	309	65	153
S+100	229	367	128	182
AUGUST 1984				
S	28	120	13	70
S+50	85	157	43	110
S+100	125	199	79	183
MAY 1985				
S+25			80	87
S+50			86	88
S+75			77	93
S+100			82	92

This suggests that the *Eichhornia* made the media (which had been prepared using tap water) more suitable for the growth of *Selenastrum*. One explanation might be that the plants released nutrients into the water, although this is probably not the only explanation as the yield of the media without *Eichhornia* to which PAAP was added was much lower than the yield of the media with *Eichhornia* to which PAAP was added. Inhibitors may have been present in the tap water and *Eichhornia* may have removed them, making the media suitable for the growth of *Selenastrum*.

April 1984: The algal yields of media without *Eichhornia* (controls) were much higher (*ca* 170 mg/l) than those of media with *Eichhornia* (experimental media). The addition of PAAP to both control and experimental media resulted in increases in the yields, indicating that inhibitors were not present.

June 1984: After both one and two weeks growth, the addition of PAAP to the media resulted in increased yields, the magnitude of the increase being proportional to the amount of PAAP added. After one week the difference between the control and experimental media was ca 150 mg/l but after two weeks it was smaller. From week 1 to week 2 the ability of both control and experimental media to support algal growth decreased although there was no evidence of inhibitors.

August 1984: After both one and two weeks of growth, the addition of PAAP to the media resulted in increased yields. The increases were larger when more PAAP was added. After one and two weeks the average difference between control and experimental media were similar but the yields after the first week were higher suggesting nutrient removal by Eichhornia. Again, no evidence of inhibitors could be found.

May 1985: The addition of increasing amounts of PAAP did not produce higher yields and the difference between control and experimental media was not significant. As shown in Section 3.1 the results of the bioassays performed in May 1985 were erroneous; therefore, no reliable conclusions could be drawn.

Although the medium with Eichhornia produced lower algal yields, because uptake by Eichhornia had reduced nutrient availability, there was no evidence of allelopathy.

c. Root extract bioassays (Table 3.7). In all cases except the stock extract (100%) of April 1984 the yield of the extract with no PAAP added was lower than that of the extract with complete PAAP. The algal yields from the stock extracts (100%) in April 1984 were probably not determined by nutrient availability; hence, the lack of response to nutrient addition. The yields of the controls of September 1984 were considerably lower than those predicted from the yield coefficients of Selenastrum and the amount of PAAP added, indicating experimental error.

These data are therefore not reliable and conclusions cannot be drawn regarding the effects of root extracts of Eichhornia on the growth of Selenastrum. In all other cases, except the 0,1% extract to which 160% of complete PAAP was added in August 1984, the yield of the flasks containing root extract was greater than that of the controls to which the same amount of PAAP had been added. Since almost all the treatment yields (i.e. those containing extract) were higher than the control yields, extracts of Eichhornia roots appear to stimulate the growth of Selenastrum, perhaps only as a result of the additional nutrients and vitamins present in the extracts.

Table 3.7: Results of root extract bioassays

Date	Treatment	Algal yield (mg/l)								
		Control	0,01	0,1	0,25	0,5	0,75	1	10	100%
April '84	+0		22	25				85	407	1799
	+100	120	143	168				238	565	1680
June '84	+0			38	34	69	101	192		
	+100	191		241	263	235	271	320		
August '84	+0	4		26	50	18	86	92		
	+40	72		106	132	122	174	154		
	+60	130		140	144	202	226	242		
	+100	128		228	182	312	300	320		
	+160	334		282	310	358	312	402		
September '84	+0	0		12	22	2	28	54		
	+40	20		4	6	32	36	64		
	+60	4		28	18	36	44	92		
	+100	28		46	64	62	54	100		
	+160	66		70	64	70	78	96		

d. Decaying matter bioassay

Figure 3.10 shows the mean algal yield against the concentration of PAAP added to the seven treatments. The mean algal yield of the autoclaved, filtered extract with complete PAAP was lower than the autoclaved filtered extract with no PAAP added. The mean algal yield of the filtered extracts increased with increasing amount of PAAP added (slope = 0,87). The mean algal yield of the control also increased with increasing amount of PAAP added, but the increase was more gradual (slope = 0,54) indicating depressed growth in the control (Figure 3.11). This indicates experimental error; therefore, no reliable conclusions can be drawn regarding the presence of inhibitors in the submerged decaying petioles of *Eichhornia*.

4. DISCUSSION

The hypothesis that *Eichhornia* in Vernon Hooper Dam may be releasing a substance which inhibited algal growth was developed largely as a result of three observations:

1. the reportedly large (61%) reduction in water treatment costs in the first six months following the introduction of hyacinth control in September 1981 and a reduction of 45% in the second six months (Anon., 1984);

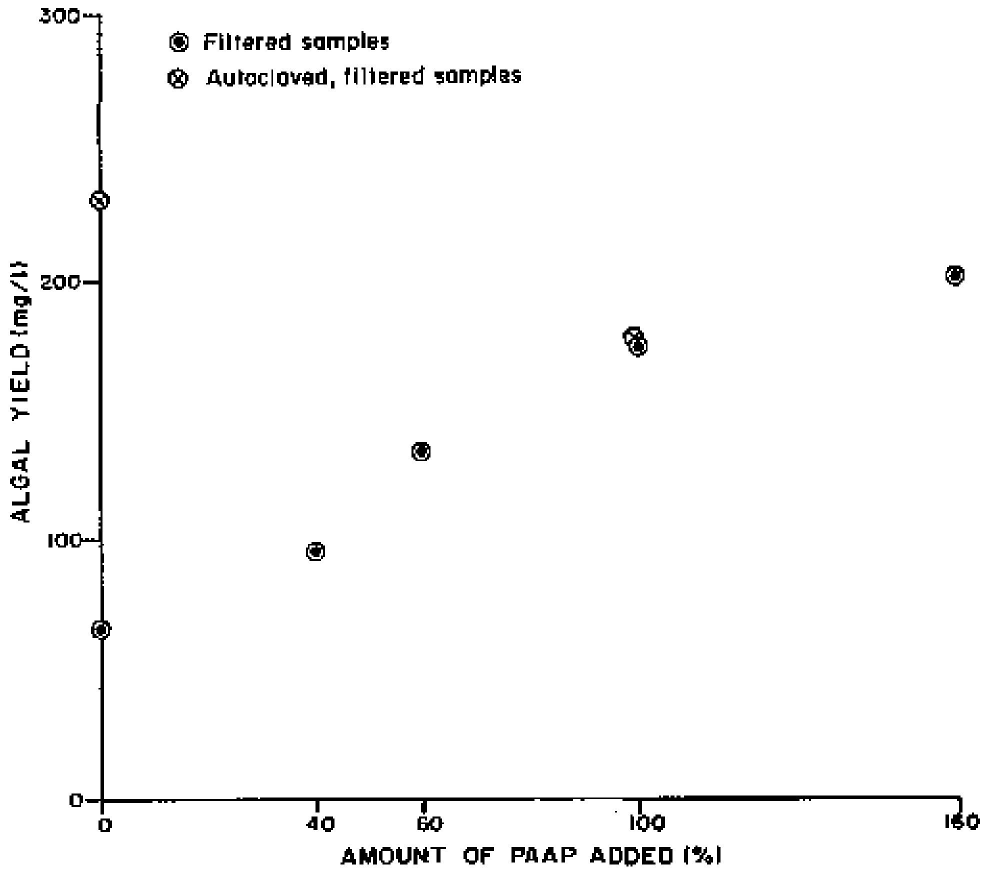

Figure 3.10: Mean algal yield (n=5) versus concentration of PAAP for the decaying matter bioassay.

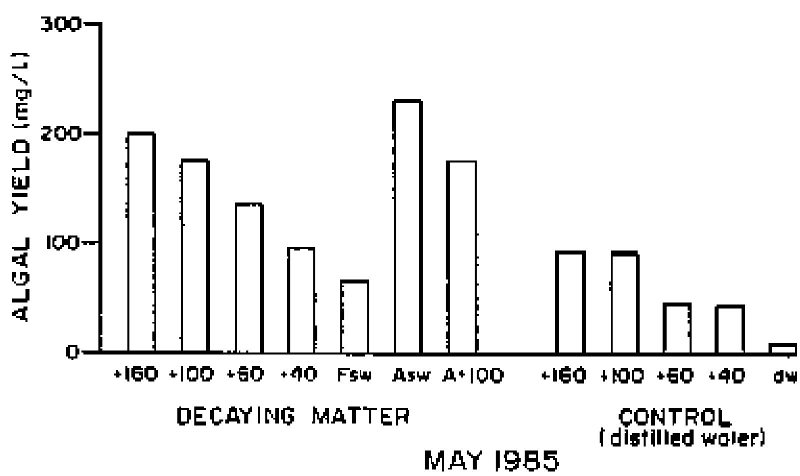

Figure 3.11: Results of the decaying matter bioassay. A key to the abbreviations used is given in Appendix 1.

2. the observation by Musil (1983) that a 20 ha *Eichhornia* population would have been too small to reduce algal growth rates by nutrient removal; and
3. the observation that *Eichhornia* was confined to a relatively small area of the lake surface, thus light limitation of algal growth rates was unlikely.

It has since been shown that the reported reduction in water treatment costs did not in fact occur (Chapter 4, Section 2.3). In addition algal-related water treatment problems (chlorophyll *a* concentrations and algal numbers) increased from 1975 to 1984/85. Also, Musil & Breen (1985) have since shown that a 20 ha population of *Eichhornia* would be sufficient to reduce algal growth rates in Vernon Hooper Dam. Although the *Eichhornia* population is not always as large as 20 ha, the combined effect of *Eichhornia* on nutrients, light and sedimentation rates would reduce the algal standing crop to levels lower than would occur without *Eichhornia*. Thus there are now no longer reasons to postulate the existence of an algal inhibitor produced by *Eichhornia*. Bioassays to test for the presence of inhibitors in water from Vernon Hooper Dam, in media in which *Eichhornia* had grown and in extracts of *Eichhornia* root material provided no conclusive evidence for the existence of an algal inhibitor produced by *Eichhornia*. The only apparently conclusive evidence of an inhibitor was found at the Lake station in September 1984 and at the

Wekeweke station in December 1984. Such infrequent manifestations of inhibition, if indeed they are correct, are unlikely to significantly reduce the algal growth and thus also water treatment costs. As this was a preliminary study of relatively short duration, there were many aspects of the interaction between *Eichhornia* and algae which could not be examined. Since inhibitors may be species specific, heat-labile, associated with particulate matter and/or only be produced when plants are under stress; a broader based experimental approach is necessary before allelopathy can be ruled out completely.

5. CONCLUSIONS

Although evidence of algal growth inhibition was found in Vernon Hooper Dam on two occasions, there was no evidence that inhibition was due to the presence of *Eichhornia*. Thus, it seems that any effect of *Eichhornia* on conditions in Vernon Hooper Dam was not due to allelopathy. Vernon Hooper Dam was probably a poor site to test for allelopathy because of the considerable inflow of organic and inorganic pollutants from the Sterkspruit which was not examined during this study (Archibald & Furness, pers. comm.).

4. VERNON HOOPER DAM CASE STUDY

1. INTRODUCTION

Vernon Hooper Dam is a small, monomictic impoundment situated 25 km west of Durban (Figures 3.1, 4.1a and b and Table 4.1). The dam was built in 1927 in a deep rocky gorge below the original confluence of the Mlazi, Sterkspruit and Wekeweke (Ugede) rivers. The mean annual rainfall is 830 mm (34 year average; Howes 1976), most rain occurring in the summer months. Winds are predominantly south-easterly and ambient temperatures vary between 13 and 30°C (Howes, 1976).

The Mlazi River drains 81% of the catchment (Archibald & Warwick, 1980). The predominant activity in this area is agriculture (dairy, stock and sugar cane farming) although there is a large KwaZulu township, Mpumulanga, approximately 10 km upstream of the dam. After conventional treatment, effluent (3,6 Ml/d in 1984; Forsyth, pers. comm.) from the Mpumulanga Domestic Sewage Works (completed in 1969) is irrigated over terraces of kikuyu grass prior to entering the river (Howes, 1976).

The flow from the Mlazi River can bypass the dam via a diversion canal and tunnel (Figure 3.1). The diversion is usually used during periods of high flow to reduce the input of suspended inorganic solids to the impoundment. Mean annual

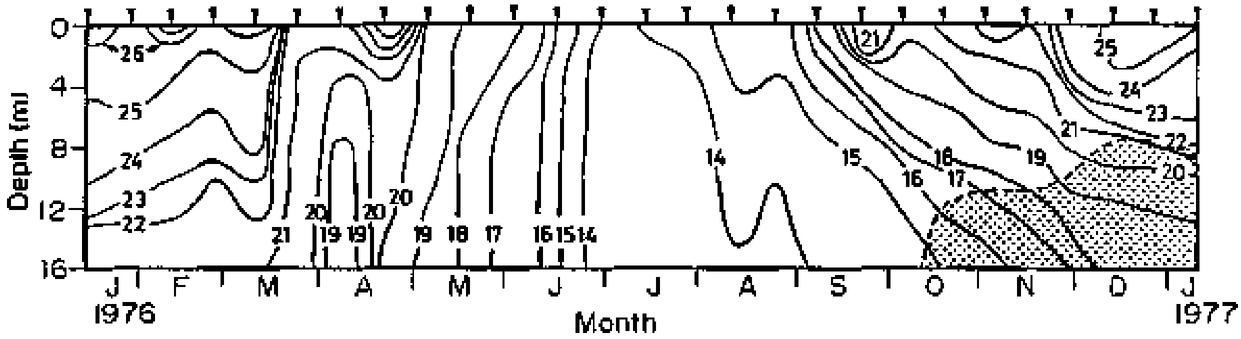

Figure 4.1a: Depth time isotherm plot (°C) in Vernon Hooper Dam during 1976 (shaded area indicates the extent of the anaerobic zone; after Archibald & Warwick, 1980).

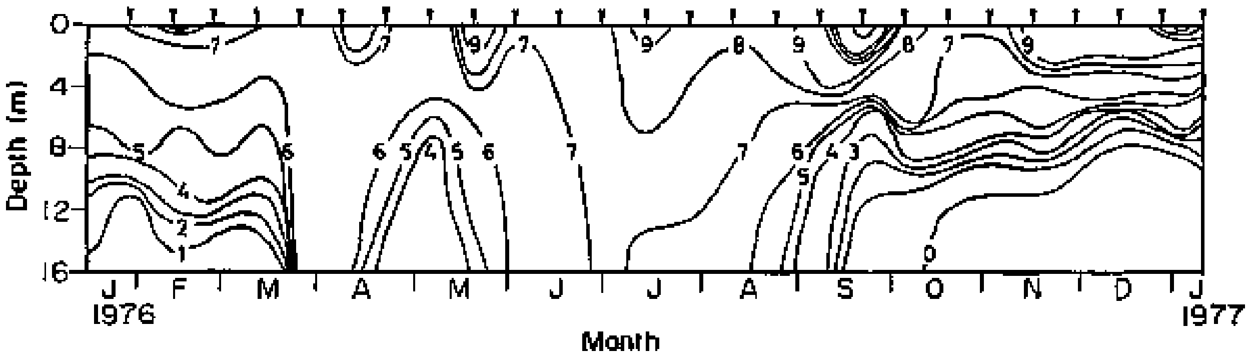

Figure 4.1b: Distribution of dissolved oxygen (mg/l) in Vernon Hooper Dam during 1976 (after Archibald & Warwick, 1980).

run off (MAR) into the dam is ca 40 x $10^6 m^3$/a and because of the small storage volume (7,16 x $10^6 m^3$ at full supply level) in relation to the MAR, the impoundment has a short hydraulic replacement period (73 days, Table 4.1).

Table 4.1: Characteristics of Vernon Hooper Dam (data from Archibald and Warwick, 1980 & Howes, 1976).

Location	20°52'S; 30°43'E
Altitude	300 m above sea level.
Catchment area	785 km^2
Surface area (at *F.S.L.)	0,81 km^2
Capacity (at F.S.L.)	7,16 x 10^6 m^3
Mean depth (at F.S.L.)	8,8 m
Maximum summer surface water temperature	27°C
Minimum winter surface water temperature	14°C
Water retention time	73 days (average between 1968 and 1984).

*F.S.L. = full supply level.

The Sterkspruit River drains 15% of the catchment in which land uses include an intensive battery system of poultry farming and the developing textile industries at Hammarsdale. This led to the construction of a wastewater treatment plant in 1963 (Archibald & Warwick, 1980). After treatment, the effluent from this plant (9Ml/d in 1984; Forsyth, pers. comm.) is discharged into the Sterkspruit River.

The Wekeweke River drains a small rural catchment in which sugar cane cultivation is the main agricultural activity (Archibald & Warwick, 1980).

Data supplied by the Umgeni Water Board were used to calculate the water and nutrient (N and P) inputs of the three rivers during the period 1975 to 1985. Unfortunately, the river stations (particularly the Wekeweke Station) are not situated in places suitable for accurate estimation of riverine nutrient loads. The Wekeweke River Station is ca 3,5 km upstream from the lake, close to the major source of nutrients in that catchment and therefore provides a better estimate of export from this catchment than load to the dam. However, based on the data available for the period 1975 to 1985, the Mlazi River contributed ca 65% of the inflowing water and total dissolved phosphorus (TDP) load and ca 55% of the total dissolved nitrogen (TDN) load to the dam. The Sterkspruit River contributed ca 30% of the inflowing water and TDP load and ca 40% of the TDN load and the Wekeweke River contributed ca 5% of the inflowing water, TDP and TDN loads. Clearly, the load received by the lake is determined principally by the Mlazi and Sterkspruit rivers (Figure 4.2a and b).

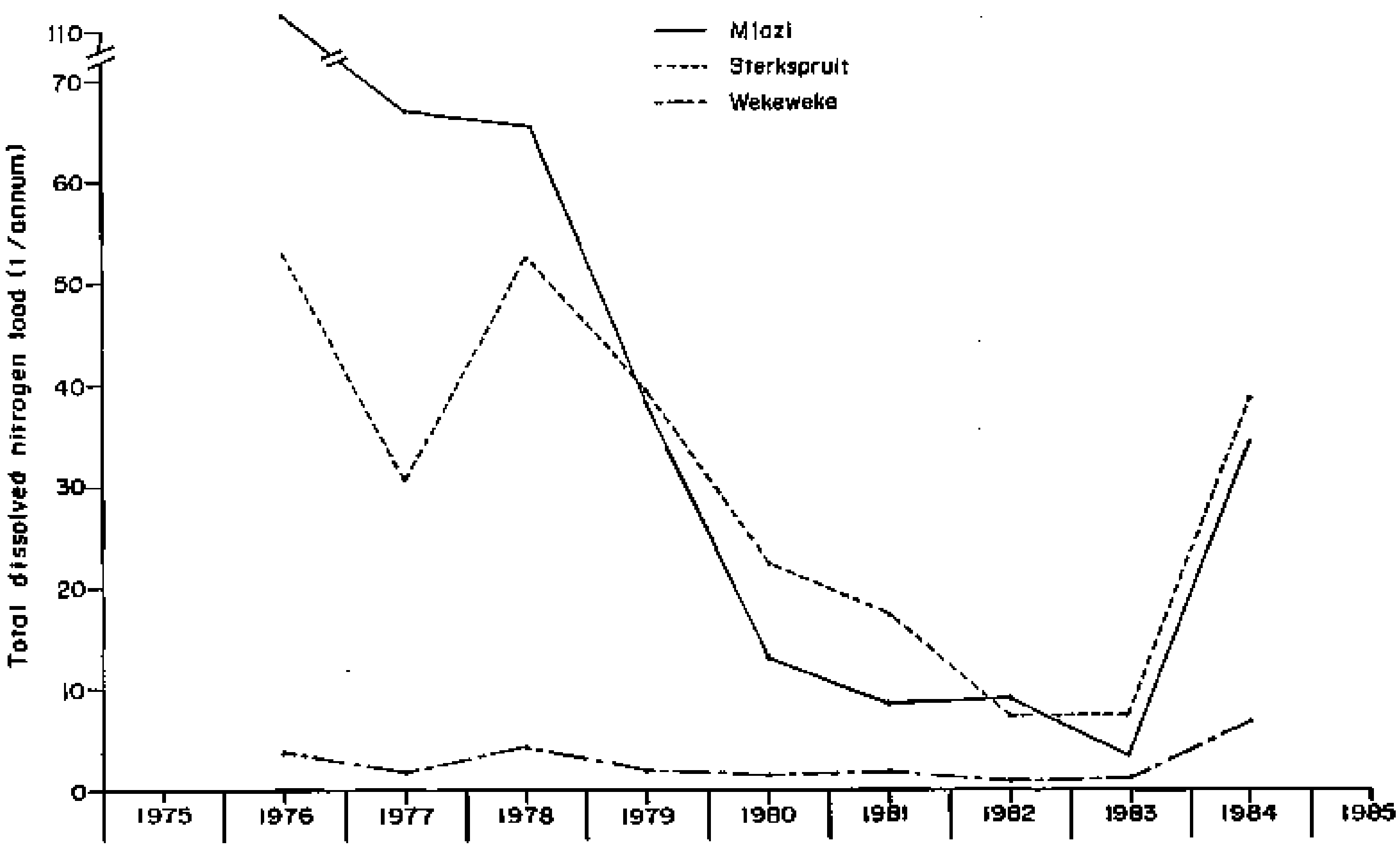

Figure 4.2a: Total dissolved nitrogen load (t/annum) from the three rivers flowing into Vernon Hooper Dam. Data from Umgeni Water Board.

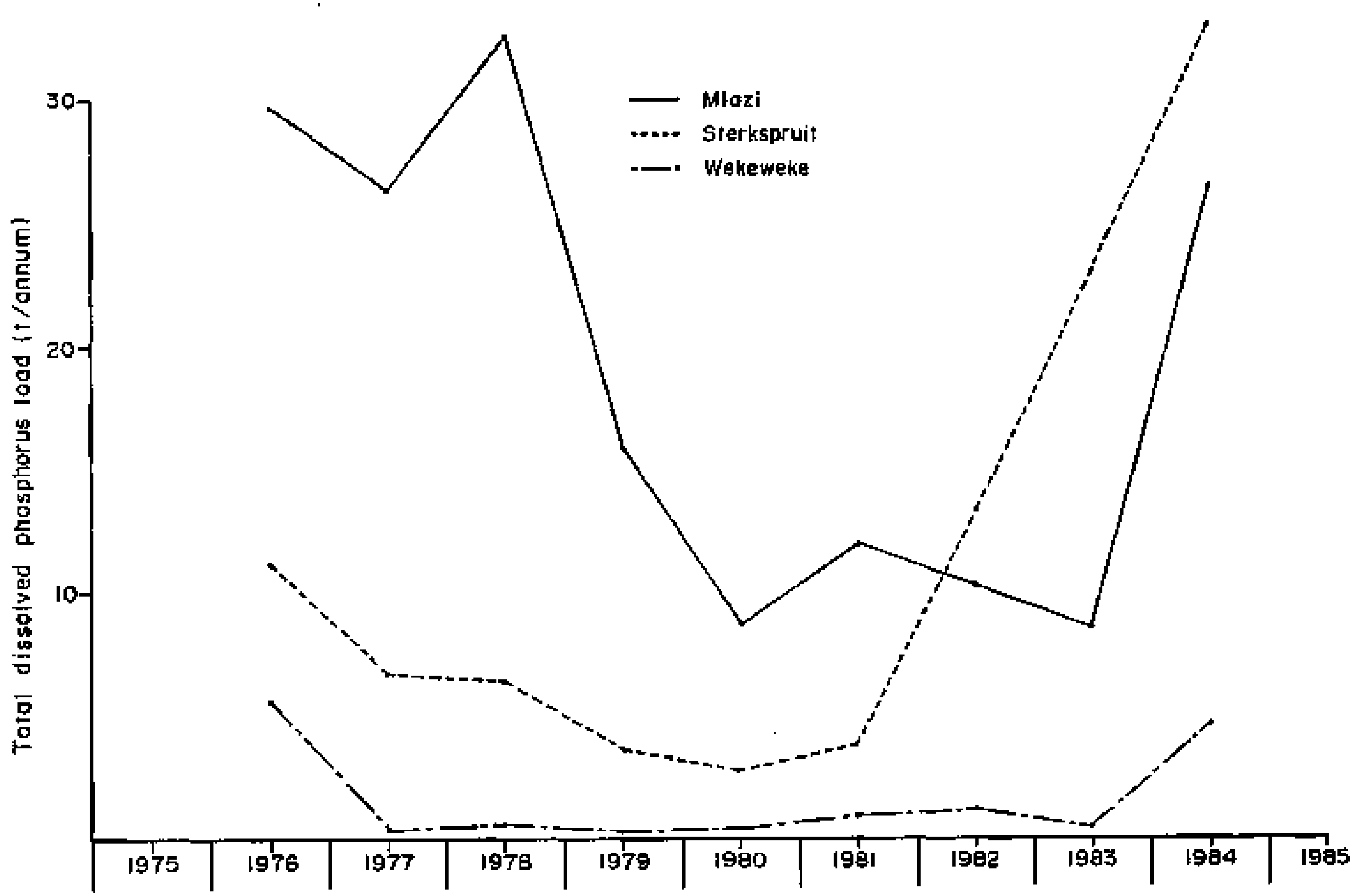

Figure 4.2b: Total dissolved phosphorus load (t/annum) from the three rivers flowing into Vernon Hooper Dam. Data from Umgeni Water Board.

Vernon Hooper Dam has been an important supply of potable water for the Queensburgh, Chatsworth and Umlazi areas of Durban though its quality has deteriorated in recent times (Howes, 1976). In 1976 Howes stated that its continued use as a water resource was seriously threatened by waste water disposal into the Mlazi and Sterkspruit Rivers.

The first real indication of the eutrophied state of Vernon Hooper Dam was the appearance of a prolific algal bloom during the severe drought of 1968 (Howes, 1976). On the basis of the algal growth potential of glass fibre filtered water collected during October 1973, Toerien et al. (1975) classified Vernon Hooper Dam as seriously eutrophied and Archibald & Warwick (1976) described Vernon Hooper Dam as the most eutrophic of the 25 major Natal impoundments studied.

The temporal variation in Eichhornia cover, phytoplankton populations and water quality were examined to determine whether the algal standing crop had decreased and water quality improved since the introduction of hyacinth. The effects of Eichhornia on the phytoplankton population were assessed in an attempt to establish the mechanism(s) involved. Unfortunately the data were discontinuous so the statistical significance of most the apparent trends could not be established.

2. TEMPORAL VARIATION IN EICHHORNIA COVER, PHYTOPLANKTON STANDING CROP AND SPECIES COMPOSITION AND WATER QUALITY

2.1 EICHHORNIA COVER (Figure 4.3)

Eichhornia was first noticed on Vernon Hooper Dam in the late 1960s/early 1970s (Upfold, pers. comm.). For many years it was not a problem but, in 1974 the in-lake phosphate levels reached 100 µg/l for the first time and during that year *Eichhornia* covered the entire water surface (*ca* 80 ha). Intensive manual harvesting during the winter of 1975 achieved complete removal by early spring of that year (Anon.,1984). However, the heavy rains in December 1975 washed a dense population of *Eichhornia* that had become established below the Mpumulanga sewage discharge point, down the Mlazi River and into the impoundment (Howes, 1976). After the 1975/76 rains, there was a considerable amount of *Eichhornia* in the Mlazi, Sterkspruit and Wekeweke sectors of the impoundment (Archibald, pers. comm.). Despite continued manual removal, the growth of *Eichhornia* was not contained (Anon., 1984). During 1979, *Eichhornia* spread extensively and by March 1980 covered *ca* 46 ha (Everett, 1980; cited in Musil & Breen, 1985). By January 1981 cover was again complete (*ca* 80 ha) and full scale mechanical removal was started. By June of that year, cover was reduced to *ca* 40 ha and by September complete removal had been achieved. As a result of pilot scale tests on the phosphorus and potassium

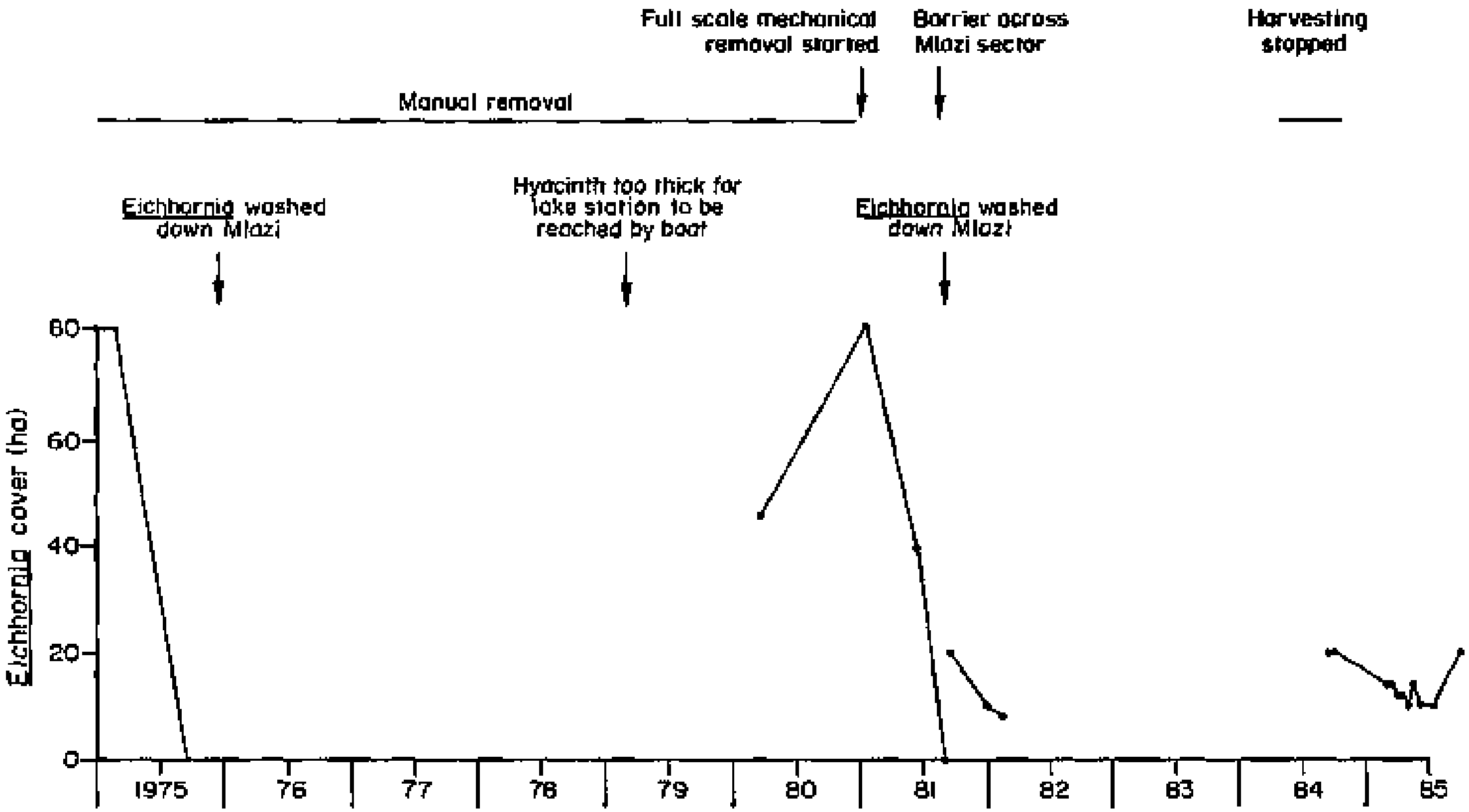

Figure 4.3: Temporal variation in visually estimated <u>Eichhornia</u> cover (ha) on Vernon Hooper Dam. Although <u>Eichhornia</u> was present throughout 1975 to 1980/81 the extent of the population was not recorded. At full supply level the surface area of Vernon Hooper Dam is approximately 80 ha.

uptake capacity of *Eichhornia*, the Durban City Engineer's Department decided to retain *Eichhornia* behind a floating boom enclosing *ca* 20 ha of the Mlazi sector of the impoundment. The boom was placed in position on 4 September 1981. In October 1981 *Eichhornia* was again washed down the Umlazi River and the entire water surface of the enclosure was covered within a short time. By harvesting approximately 100 tonnes per day *Eichhornia* cover was reduced to 10 ha by the beginning of 1982 and to 8 ha by mid-February 1982 (Anon., 1984). The staff at the Dam attempted to keep the *Eichhornia* cover at *ca* 8 ha (Spark, pers. comm.) but this was usually not possible during summer when growth rates are high. *Eichhornia* is not harvested for the four to five winter months of the year (between May and September) but harvesting is resumed in August or September once the cover has inceased to *ca* 15 ha. Unfortunately, the amounts of plant material harvested are not monitored. A mobile crane is used to remove the *Eichhornia* throughout the summer at the rate at which it grows (Spark, pers. comm.) and at the present time (1985) the standing crop varies between 8 and 20 ha during an annual cycle.

Based on a density of 11,7 t dry weight/ha (moderately crowded population; Musil & Breen, 1985) a moisture content of 95% (Scott *et al.* 1979) and an average nitrogen and phosphorus content of 1,6 and 0,43% respectively (Sharma, 1971); the fresh weight and nitrogen and phosphorus content of 8 and 20 ha populations were calculated. The standing crop would vary between 1 870 and 4 680 t fresh weight during an annual cycle. The nitrogen content of the populations was calculated to vary between 1 498 and 3 744 kg and the phosphorus content between 402 and 1 006 kg. However it is not possible to obtain an accurate estimate of nutrient removal and turnover as neither the size of the harvest nor the size of the hyacinth population is monitored.

2.2 PHYTOPLANKTON ABUNDANCE AND SPECIES COMPOSITION

i. Standing crop

Figure 4.4 shows the temporal variation in algal numbers in the surface water from Vernon Hooper Dam. Between 1975 and 1978, algal numbers were significantly lower ($p<0,05$) than in 1979 which were, in turn, significantly lower than the algal numbers reported for 1984. The data are, however, incomplete and biased because prior to 1984 algal numbers were not determined routinely (Table 4.2), but rather only when numbers were high or there appeared to be a sizeable difference between the surface and bottom samples (Peters, pers. comm.). Also, the average algal number determined for 1984 is an underestimate of total algal numbers since in eleven of the thirty one samples, the dominant alga (usually *Microcystis*) was too numerous to be counted and could, therefore, not be included. Despite these limitations, and

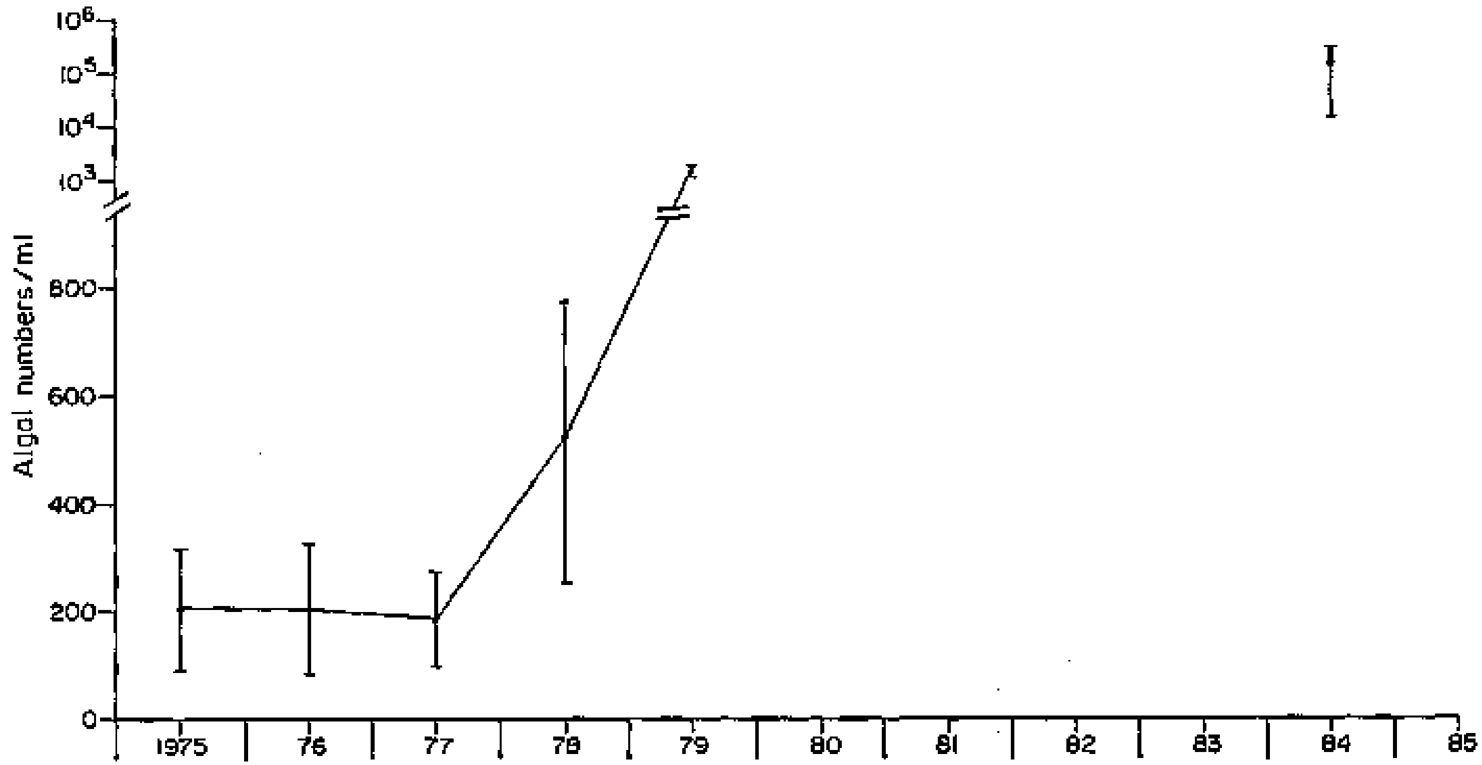

Figure 4.4: **Temporal variation in algal numbers (per ml) in the surface water of Vernon Hooper Dam.** Data from Umgeni Water Board. The frequency of dominance of the major algal groups is shown in Figure 4.7.

because of the magnitude of the increase in numbers, the results suggest a general trend of increasing algal numbers with time since 1977. Because of insufficient collection of data on algal numbers and _Eichhornia_ cover, the effect of _Eichhornia_ cover on algal numbers cannot be determined with any certainty; however, it appears that the presence of _Eichhornia_ has not reduced algal numbers to within acceptable levels.

Table 4.2: Number of surface water samples collected each month from Vernon Hooper Dam for determination of algal numbers.

	Total No.	Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Oct	Nov	Dec
1975	9	2	3	0	0	0	0	0	0	0	0	3	1
1976	20	2	2	1	1	1	4	3	2	1	2	0	1
1977	13	0	0	0	0	2	3	1	3	1	3	0	0
1978	21	0	0	0	0	1	3	3	3	4	3	3	1
1980-83	0	0	0	0	0	0	0	0	0	0	0	0	0
1984	31	2	4	0	0	2	4	1	5	5	4	1	2

An analysis of the chlorophyll _a_ concentrations for 1976 and 1984/85 support the view that algal populations in Vernon Hooper Dam increased since 1975. During 1976 and 1977

Archibald and Warwick measured chlorophyll a concentrations in the impoundment (Figure 4.5). The mean chlorophyll a concentration was 19,8 mg/m³ and the coefficient of variation 20%. Chlorophyll a concentrations measured between April 1984 and May 1985 are shown in Figure 4.6. The mean concentration (49,2 mg/m³) is significantly higher ($p<0,05$) though the coefficient of variation is similar (18%). These data suggest that chlorophyll a concentrations more than doubled between 1976/77 and 1984/85.

ii. Species composition

Archibald and Warwick (1980) noted that prior to 1976, particularly during the drier months following the industrial development of the catchment, extensive *Microcystis* blooms were responsible for major water treatment problems. During 1976, a particularly wet year, three diatom blooms occurred and no major bloom of cyanobacteria (blue-green algae) was recorded (Archibald & Warwick, 1980). The frequencies of dominance of a particular algal class (green algae, cyanobacteria and diatoms), as a percent of the number of algal counts per year, was determined for the period 1975 to 1984 (Figure 4.7). Since 1975 the frequency of dominance of cyanobacteria increased with time. Although the frequency of dominance and abundance of "nuisance" algae (the cyanobacteria) has apparently increased in the last ten years, extensive blooms of *Microcystis* also occurred prior to

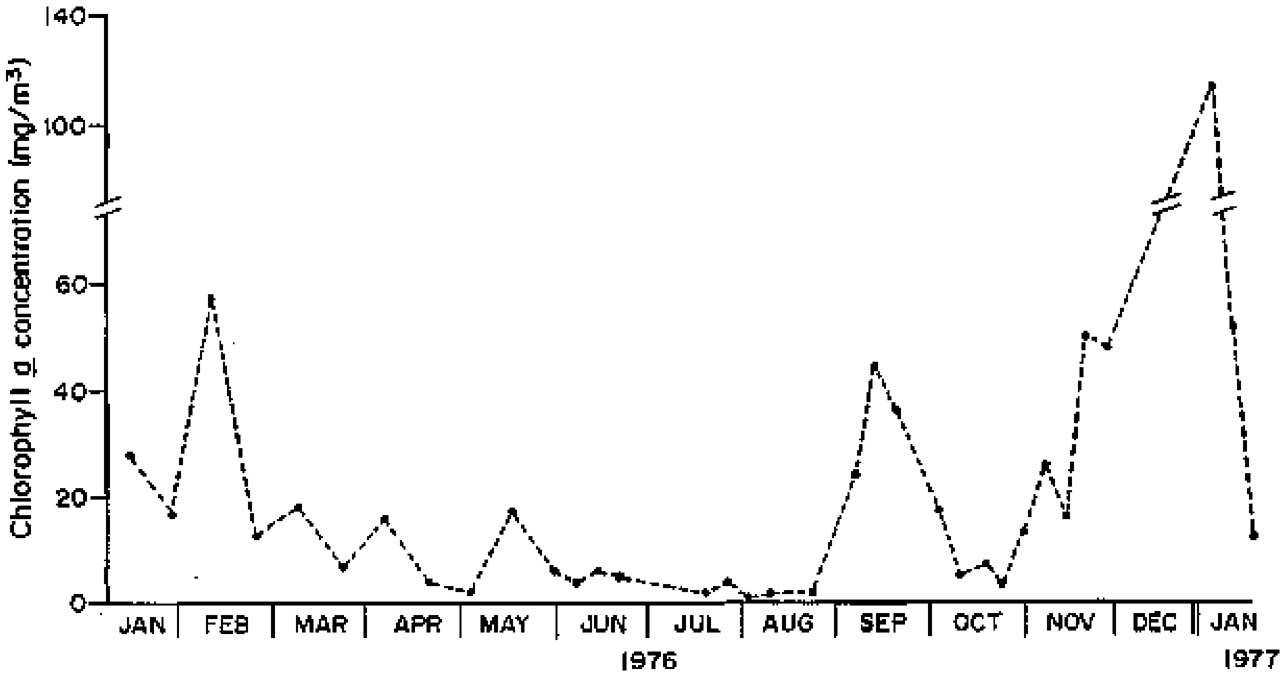

Figure 4.5: Temporal variation in chlorophyll a concentration in the surface water of Vernon Hooper Dam between January 1976 and January 1977. Data from Archibald (pers. comm.).

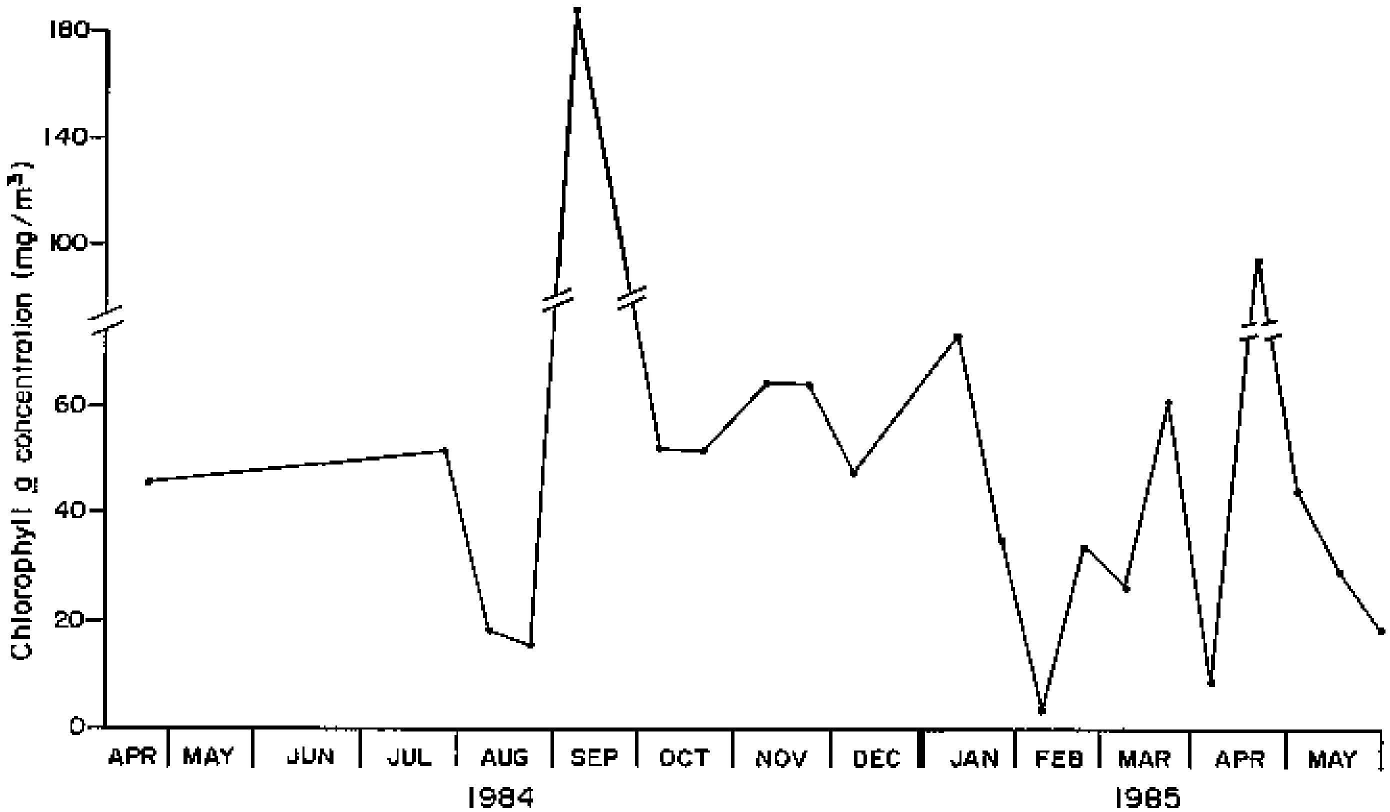

Figure 4.6: Temporal variation in chlorophyll a concentration in the surface water of Vernon Hooper Dam between April 1984 and May 1985.

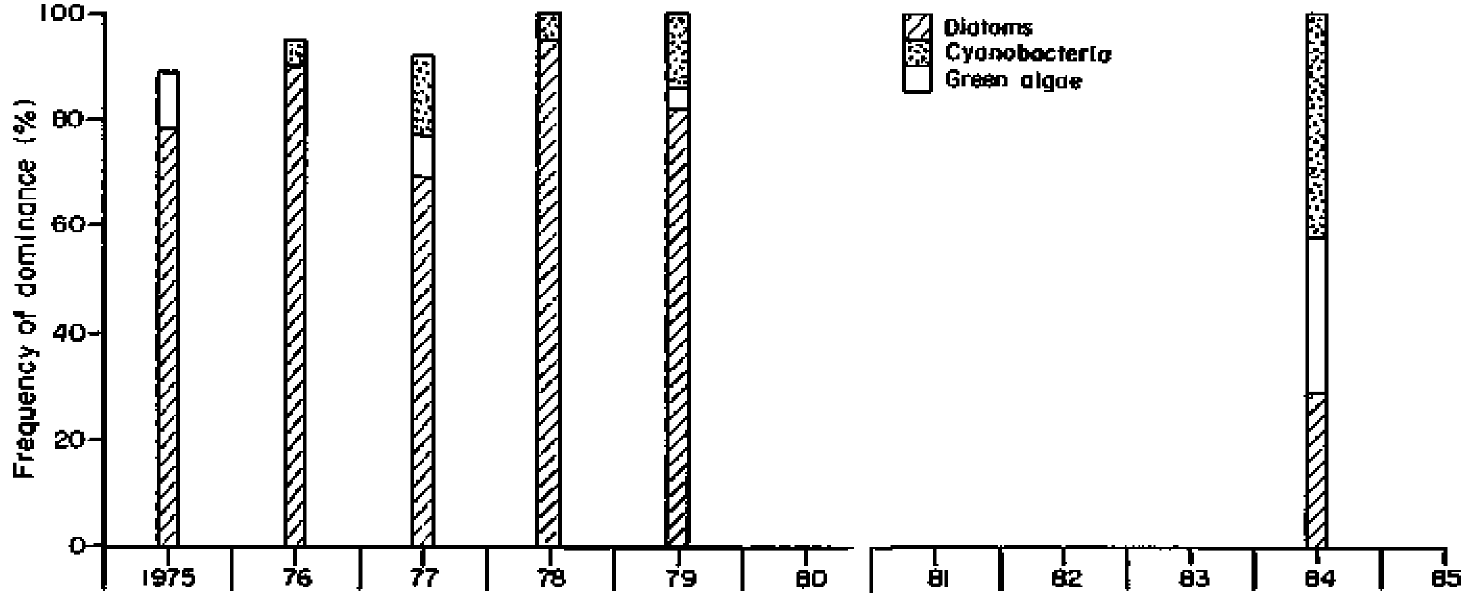

Figure 4.7: Temporal variations in the algal class dominance (%) in the surface water of Vernon Hooper Dam. Data from Umgeni Water Board. The algal species composition was not determined for the period 1980 to 1983.

1976 and caused major water treatment problems (Archibald & Warwick, 1980).

2.3. WATER QUALITY

Figures from the Durban City Engineer's Department show that in the year following water hyacinth control, brought into effect in September 1981, there was a reduction in water treatment costs despite poor rainfall and increasing nutrient loading. The reductions were reported to be 61% in the first six months (December 1981 to May 1982) and 45% in the second six months (Anon., 1984). However, the change in water treatment costs following hyacinth control was compared with and expressed as a percent of the cost of treatment of "manganese" water. "Manganese" water occurred following oxygen depletion, either as a result of stratification or full hyacinth cover reducing the rate of oxygen exchange between the atmosphere and the water. In 1981 it cost 4,4 c/m^3 to treat "manganese" water; whereas, treatment of "normal" water cost 1,3 c/m^3 and algae laden water cost 2,2 c/m^3 (Anon., 1984). Thus, although the figures for the first and second six month periods following hyacinth control (1,7 and 2,4 c/m^3 respectively) indicated an improvement in water quality compared with the period when hyacinth cover was complete, it was felt that a more detailed examination of the available data was required as these data gave no indication of the frequency of occurrence of each of the three different types of water. To this end the temporal variation in the quality of water from the dam outlet and three depths of a lake station was investigated.

i. Outlet water quality

From data collected by the Durban City Enginer's Department and the Umgeni Water Board, summer (October to March) and winter means of the permanganate value, conductivity, *E. coli* Type 1 numbers and manganese concentration of the outflow and and annual mean water treatment costs were calculated.

a) Permanganate value (PV)

The PV gives an indication of the organic content of a sample since the PV is the amount of oxygen required to chemically oxidize (by addition of potassium permanganate) the generally unstable organic compounds to stable, relatively inert end products. Other measures of the organic content of water are the biochemical oxygen demand (BOD) and the chemical oxygen demand (COD); however these were not determined.

Temporal variations in the summer and winter means of the permanganate value of the outlet water from Vernon Hooper Dam between 1975 and 1985 are shown in Figure 4.8a. Between 1975 and 1978 PVs varied seasonally with summer values (*ca* 3,5 mg/l) higher than winter values (*ca* 2,5 mg/l). Between 1978 and 1982 there was very little seasonal variation and PV

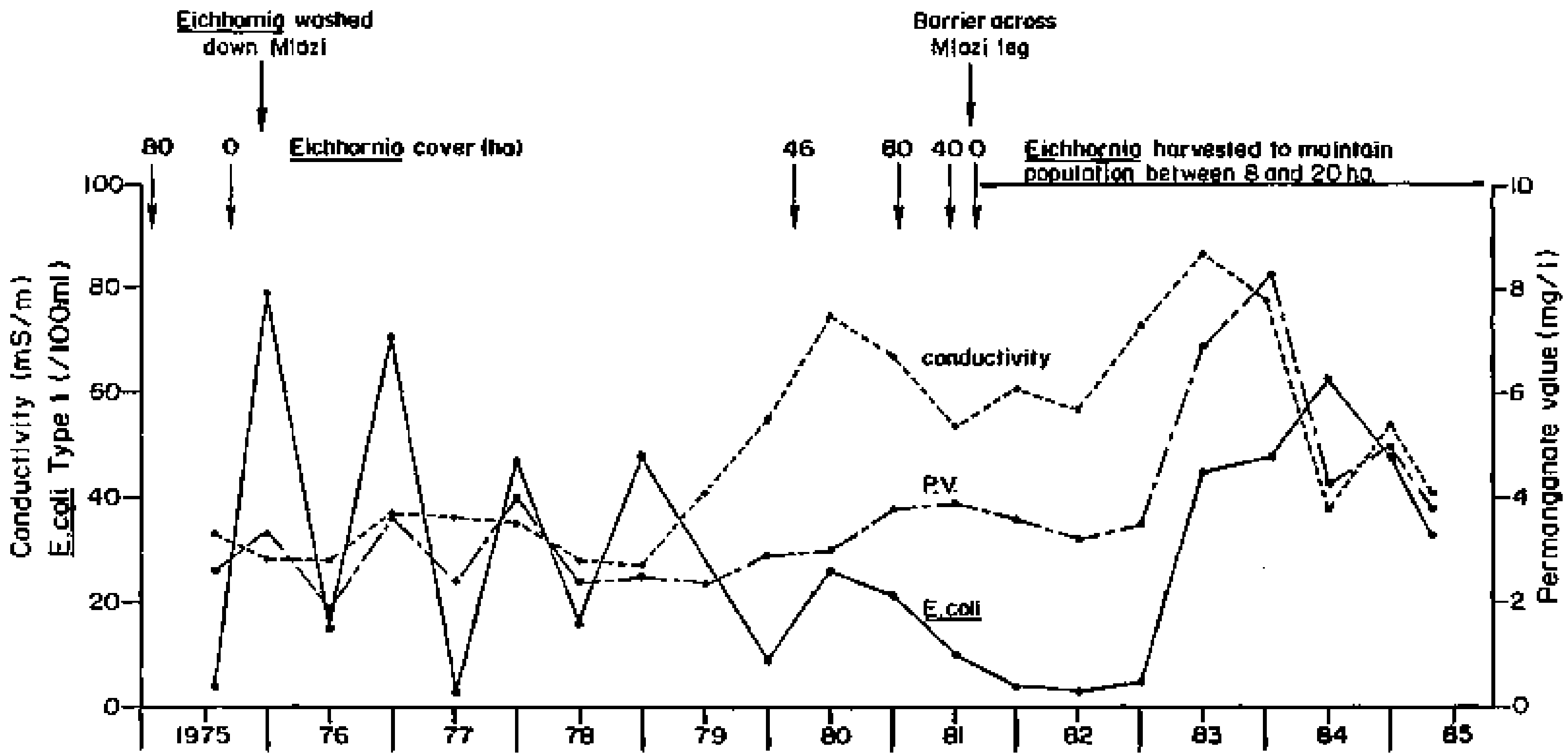

Figure 4.8a: Temporal variation in summer (October to March) and winter means of permanganate value (mg/l), conductivity (mS/m) and *E. coli* Type 1 numbers (per 100 ml) of the outlet of Vernon Hooper Dam. Data from Umgeni Water Board.

appears to have increased gradually from *ca* 2,5 mg/l in 1978 to *ca* 3,5 mg/l in 1982. Since 1982, PVs have generally been higher than between 1975 and 1982. The mean PV for winter 1983 (6,9 mg/l) and summer 1983/4 (8,3 mg/l) were two to three times higher than previous means. They were also higher in summer (8,3 and 5,4 mg/l in 1983/4 and 1984/85 respectively) than the corresponding values for the winters of 1983, 1984 and 1985 (6,9; 4,3 and 3,8 mg/l respectively). There is thus some evidence of the reappearance of seasonal variation.

When the variation in PV is compared with the incidence of water hyacinth it appears that the introduction of hyacinth control (in September 1981) did not significantly affect water quality as evidenced by PVs since winter 1981. At the present time the only explanation that can be offered for the marked increase during 1983 is an increase in the input of organics from the Sterkspruit river. There seems no justification in relating it to hyacinth cover.

b) Conductivity

The conductivity of a solution depends on the quantity and type of dissolved salts present and for dilute solutions it is approximately proportional to the total dissolved ionizable solids content (Tebbutt, 1977). The maximum conductivity "recommended" by the South African Bureau of Standards (SABS) for water for domestic supplies is 70 mS/m.

Between 1975 and 1978 the conductivity of the outlet of Vernon Hooper Dam was fairly constant (*ca* 30 mS/m; Figure 4.8a) but after the winter of 1979, the mean conductivity of the outlet water increased dramatically. But in the recent past (1984/85) levels were again low, though slightly higher than 1978/79. Notwithstanding the marked increase in conductivity between 1979 and 1984, the mean conductivity values of the raw lake water exceeded the SABS acceptable limit for domestic supplies on only four occasions. Currently it is well below this limit.

Inspection of the data before and after initiation of hyacinth control (September 1981) leads to the conclusion that hyacinth played an insignificant role in determining the pattern of change in conductivity relative to the role played by other factors, such as dry periods as a result of drought.

c) *E. coli* Type 1

Escherichia coli, a normal bacterial inhabitant of the mammalian intestine, is used as an indicator of faecal contamination (Tebbutt, 1977, and Cabelli, 1978). Of the data available for Vernon Hooper Dam, the presence of *E. coli* Type 1 is the best indicator of human excretal pollution. Water containing *E. coli* Type 1 was therefore considered potentially dangerous in that other pathogenic organisms might also be present (Tebbutt, 1977).

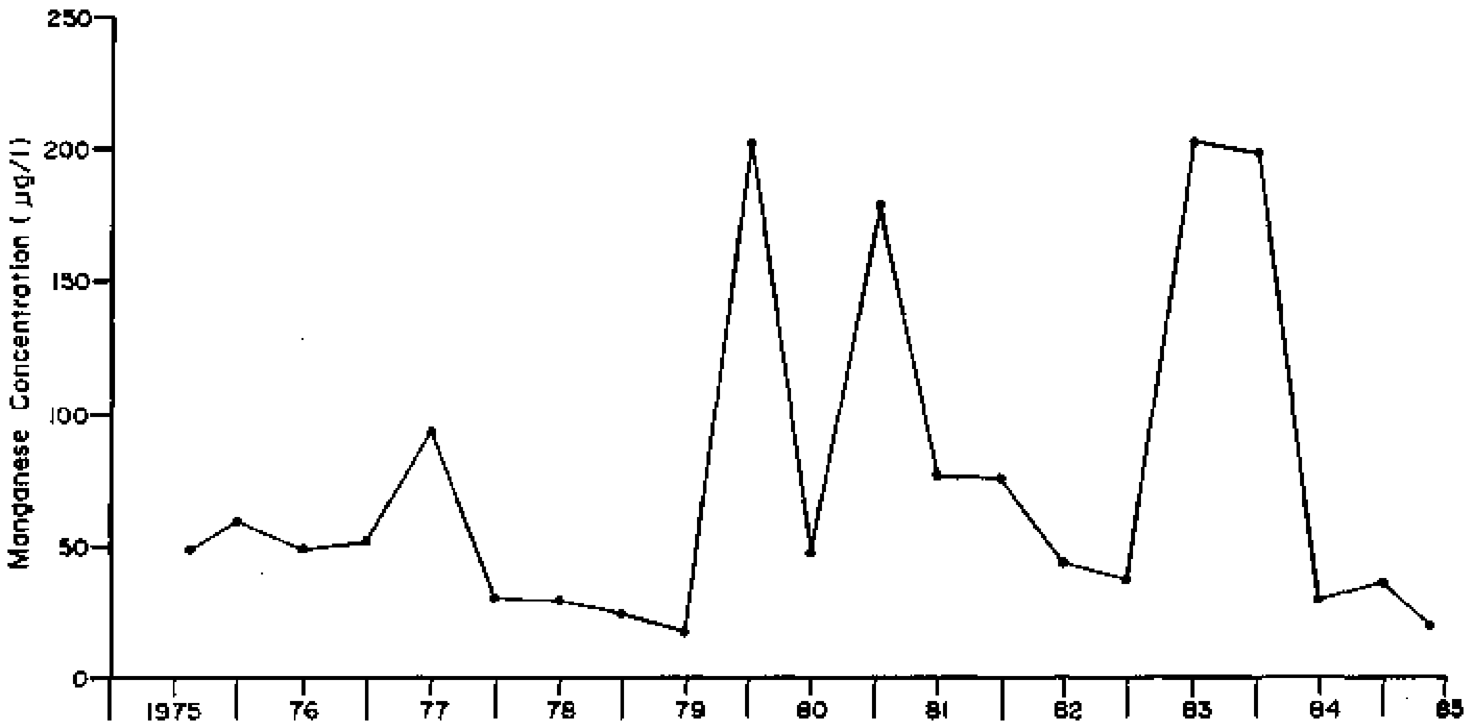

Figure 4. 8b : Temporal variation in the summer and winter means of the manganese concentration (μg/l) of the outlet of Vernon Hooper Dam. Data from Umgeni Water Board.

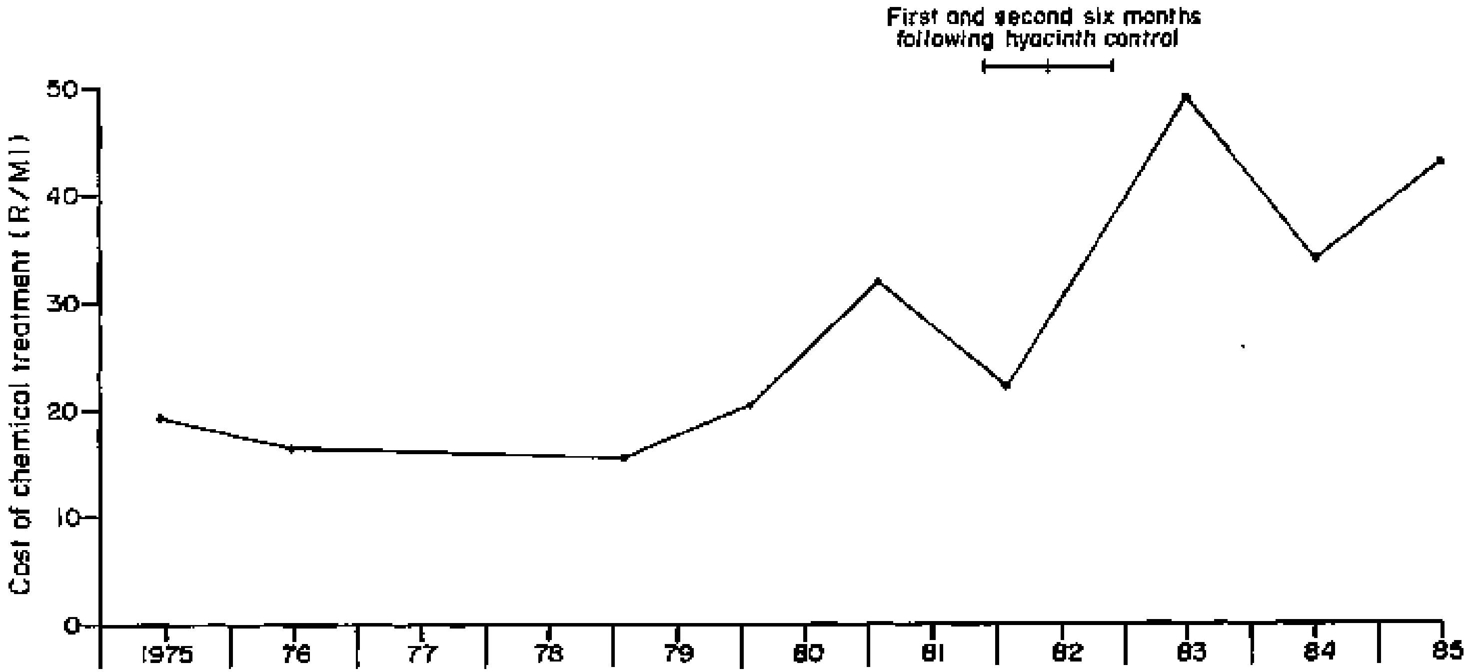

Figure 4.9: Cost of chemical treatment per unit volume of water treated (R/Ml), at Vernon Hooper Dam between 1975 and 1985. The quantity of chemicals used and their cost in 1984 was used to determine the annual chemical treatment cost per unit volume of water treated. Data from Umgeni Water Board.

The temporal variation in summer and winter means of *E. coli* Type 1 numbers in the outlet water from Vernon Hooper Dam between 1975 and 1985 is shown in Figure 4.8a. Between 1975 and winter 1979, *E. coli* Type 1 numbers exhibited seasonal variation with summer values (*ca* 60/100 ml) higher than winter values (*ca* 10/100 ml) because of higher run off in summer. Between 1979/80 and 1982/83 *E. coli* numbers were generally low (*ca* 10/100 ml) and no distinct seasonal trends were evident.

Since the beginning of 1983 *E. coli* Type 1 numbers in the outlet water have increased but remain below the peaks for 1975/76 and 1976/77. *E.coli* Type 1 numbers in the outlet vary considerably, but unlike the permanganate value and conductivity, they do not show a deterioration in water quality with time. The high values for conductivity, PV and *E. coli* recorded during 1982/84 are particularly striking and imply a distinctly different lake behaviour, but one which cannot be directly linked to the presence of water hyacinth, perhaps because the effect of *Eichhornia* on the whole lake was reduced because it was confined to the Mlazi leg only.

d) Manganese concentration

The concentration of managanese in the water indicates the extent to which reducing conditions have developed. These conditions occur frequently in eutrophic lakes, especially when mixing of the water column is not complete. Between winter 1975 and 1979 manganese concentrations in the outlet (ca 0,05 mg/l; Figure 4.8b) were generally lower and less variable than since 1979. The very high manganese concentrations during summer 1979/80 and 1980/81 can be attributed to complete hyacinth cover. Following hyacinth control in September 1981 the mean manganese concentration of the outlet decreased. During winter 1983 and summer 1983/84 manganese concentrations were as high as during 1979/80 and 1980/81, although the hyacinth cover was relatively low (20 ha and less). During 1984/85 the manganese concentration was not dissimilar from that during the period 1975 to 1979. However, between 1979 and 1984 there were periods of very high manganese concentrations (ca 0,2 mg/l). None of these fluctuations can be clearly related to the presence of an area of controlled hyacinth thus it must be concluded that the hyacinth control strategy has played a minor role in determining the manganese concentration of the outlet.

e) Cost of water treatment

Perhaps the best index of water quality is the cost per unit volume of treating the water to achieve a specific water quality. From the volume of water treated, the quantity of chemicals used and the unit price of these chemicals during 1984 the annual cost of chemically treating water at the Shongweni Treatment Works was calculated (Figure 4.9). Since the unit cost of water treatment between 1976 to 1985 was calculated by using the cost of the chemicals at one particular point in time (1984) the data are independent of inflation. An additional factor which complicates the interpretation of these data is the use of different approaches to the treatment of water during the study period. Although the water supplied from the Shongweni Treatment Works requires further treatment before distribution to the consumer, it was felt that the unit cost of chemically treating the water at Shongweni Treatment Works would be an adequate index for assessing changes in water quality.

The unit cost of chemical treatment at Shongweni Treatment Works (Figure 4.9) was fairly constant until 1980. Since then treatment costs per unit volume have been more variable and higher and recently (after 1983) they have been approximately double the costs between 1975 and 1979. Thus, it seems that since 1980 the water quality of the outlet has deteriorated quite significantly. It now costs twice as much to treat the water as it did in 1979. Also, the cost of controlling the hyacinth must be added to the chemical treatment costs.

Another feature of the results is that the improvement in PV, conductivity , E. coli and manganese concentration during 1984/85 is not evident in the treatment costs.

Introduction of hyacinth control did not appear to significantly alter the increasing trend in water treatment costs. The hyacinth control programme introduced in September 1981, entailed maintaining the *Eichhornia* cover at between 8 and 20 ha but it is likely that between 1974 (when *Eichhornia* cover first increased to *ca* 80 ha) and September 1981, the average *Eichhornia* cover was greater than 20 ha. It is, therefore, desirable to extend the assessment of trends in water quality to the period before 1975 i.e. when the hyacinth population was building up.

Howes (1976) gave the average summer and winter water quality of the outlet water, for various periods, up to 1975. From data supplied by the Umgeni Water Board, five year averages were calculated for the period 1975 to 1985 and, with the data from Howes (1976) are presented in Table 4.3.

There is no clear trend of deteriorating summer water quality with time, the mean summer water quality for the period 1964 to 1969 being very similar to that between 1980 and 1985. However, winter water quality does appear to have deteriorated with time. The lack of a consistent trend in the water quality of the outlet is due to a number of factors. The first, and probably the dominant factor, is that there have been numerous changes in the catchment which would have influenced the quality of the water.

Table 4.4 : History of changes in Vernon Hooper Dam and its catchment since 1963 (from Howes 1976 unless indicated otherwise).

1963 -	establishment of the Hammarsdale Sewage Works (effluent discharged into the Sterkspruit River).
1966 -	introduction of chlorination of Hammarsdale effluent to ensure a pathogen-free discharge.
1968 -	severe drought and a prolific algal bloom (Microcystis) appeared for the first time.
July 1968 to February 1969 -	water quality degenerated from a PV of 1,8 mg/l to 5,6 mg/l and from a conductivity of 34,5 mS/m to 56,0 mS/m.
1969 -	hydrogen sulphide present in the outlet; chlorine demand high; high alum dose necessary; regular use of copper sulphate as an algicide by liquid dispersion from a boat.
-	Mpumalanga Sewage Works completed (effluent discharged into the Mlazi River).
1970 -	modifications to the Hammarsdale Sewage Works completed.
1970 to 1975 -	improvement in water quality of the Sterkspruit River compared with 1964 to 1969.
1975 -	water withdrawn from upper take-off valve to avoid manganese-rich water.
-	two algal blooms (both Melosira) occurred (in November and December).
1976 -	Extensive diatom blooms occurred in February, September and October (Archibald & Warwick, 1980).

Table 4.3: Average values of the summer and winter water quality variables of the outlet water from Vernon Hooper Dam. Data prior to 1975 from Howes (1976). The 1975-80 and 1980-85 averages were calculated from the Umgeni Water Board data.

	PV(mg/l)		Conductivity (mS/m)		E.coli Type 1/ 100 ml	
	Summer	Winter	Summer	Winter	Summer	Winter
Up to 1963	4,4	1,5	14,5	16,3	711	9
1964-69	4,5	2,0	53,2	38,2	22	3
1970-75	2,7	3,2	26,1	31,4	46	45
1975-80	3,3	2,5	36,4	40,0	51	16
1980-85	4,8	4,4	66,6	55,5	25	31

Some of these changes are listed in Table 4.4. Secondly, the changes affect the water quality variables differently. Thirdly, by 1968 the impoundment was already eutrophic, as evidenced by the appearance of a prolific algal bloom in that year (Howes, 1976); and fourthly, five year averages are insensitive measures of water quality variation. Although recent (1980 to 1985) summer water quality has been as poor as previously (1964 to 1969), winter water quality has deteriorated, possibly due to increasing effluent discharges from Hammarsdale.

On the basis of summer PV and conductivity, water quality between 1970 and 1980 was better than the five year periods before 1970 and after 1980. The influence of Eichhornia on the water quality of the outlet is however, not clear and cannot be separated from other effects.

The analysis of trends in water quality from the outlet is further complicated by the fact that water can be drawn from various depths. It was considered useful, therefore, to assess the quality of water from three depths (surface, 6 m below the surface and at the bottom) of a central lake station.

ii) Lake Water Quality

Water was collected approximately once a week from a lake station situated ca 300 m from the wall of Vernon Hooper Dam. Surface water was collected and using a Van Doorn depth sampler, water was collected from 6 m below the surface (middle) and from the bottom. Annual means of the following measurements were considered:

a. Permanganate value (PV)

b. Conductivity

c. Total dissolved phosphorus concentration

d. Total dissolved nitrogen concentration

e. pH

f. Manganese concentration

A consideration of the temporal variation of each of these parameters between 1975 and 1985, is followed by a discussion of water quality.

a) Permanganate value

Throughout the period 1975 to 1985 the annual mean PVs of the surface and middle water were very similar (Figure 4.10a). The PV of the bottom water for any particular year was always higher than that of the surface and middle water, particularly since 1980. Higher PVs in the bottom water result from sedimentation and decomposition of organic material produced in the surface layers of the lake. Between 1975 and 1980 the mean PVs of water from all depths were generally lower than the PVs of the respective depth between 1980 and 1985, the difference between 1975 to 1980 and 1980 to 1985 being most marked in the bottom water.

On the basis of PVs of the lake, particularly of the bottom samples, it appears that water quality has deteriorated since 1980. However, a deterioration in the water quality of the outlet (based on PV) was evident only later (during winter 1983 and summer 1983/84). The delay in the deterioration in outlet water quality (PV) possibly reflects the ameliorating effect of the variable depth draw off. However, in winter 1983 and summer 1983/4 water levels (Figure 4.13a) would have been too low to permit the use of upper off-take valve.

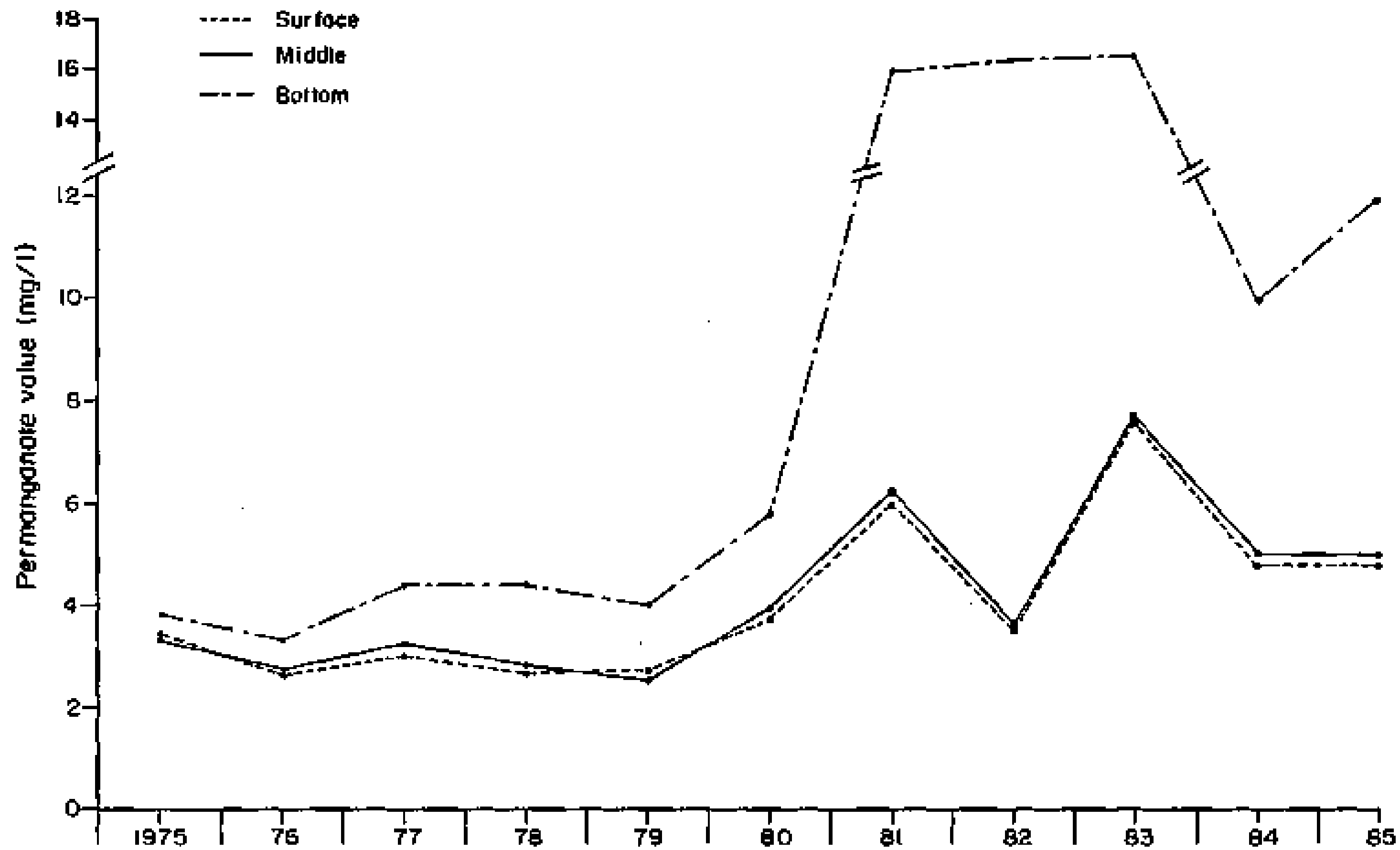

Figure 4.10a. Temporal variation (annual means) in permanganate values (mg/l) of three depths (surface, 6 m below the surface (middle) and bottom) of a lake station in Vernon Hooper Dam. Data from Umgeni Water Board.

b) Conductivity

Each year the differences between the annual mean conductivities of water from different depths were not significant ($p<0,05$). However, the variation between years, especially after 1979, was large (Figure 4.10b). Prior to 1980 the annual mean conductivities were generally low ($\leqslant 40$ mS/m) and not very variable (range: 25 - 40 mS/m) but, since 1980, annual mean conductivity has been higher (up to ca 105 mS/m in 1980 and ca 85 mS/m in 1983) and more variable (range : 42 - 109 mS/m). Values for 1984 and 1985 were only slightly higher than those before 1980.

Conductivity values indicate that the water quality of the lake has deteriorated since 1980, although apparently improving in 1984 and 1985. The temporal variation in the conductivity of the lake station is very similar to that of the outlet because the difference between the different depths of the lake station is small and not significant.

c) Total dissolved phosphorus (TDP) concentration

The annual mean TDP concentrations of the three depths show similar trends, even though the concentration in the bottom water was consistently higher than in middle water, which was higher than in the surface water (Figure 4.10c). The lower TDP concentrations in the surface water relative to the bottom water probably reflect the high algal standing crop which removes phosphorus from the water. The higher

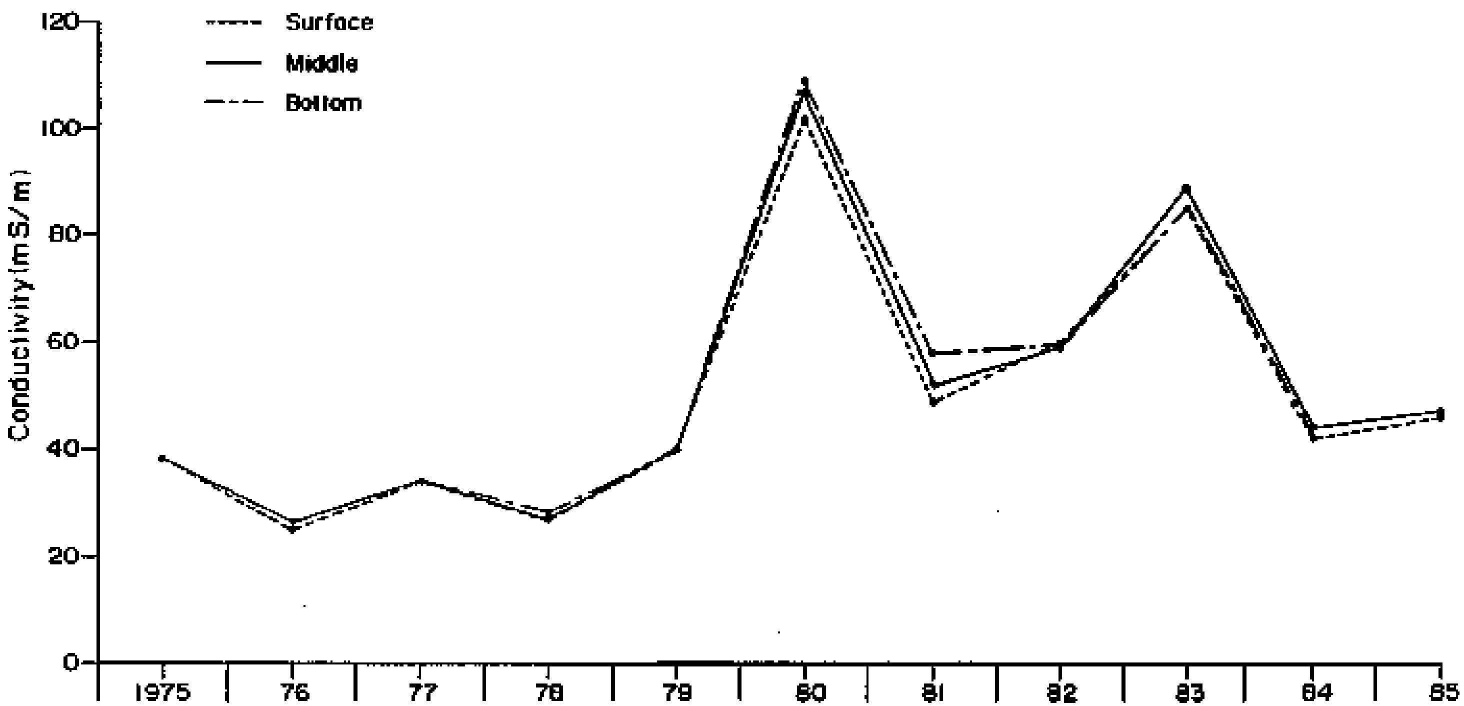

Figure 4.10b. Temporal variation (annual means) in conductivities (mS/m) of three depths (surface, 6 m below the surface (middle) and bottom) of a lake station in Vernon Hooper Dam. Data from Umgeni Water Board.

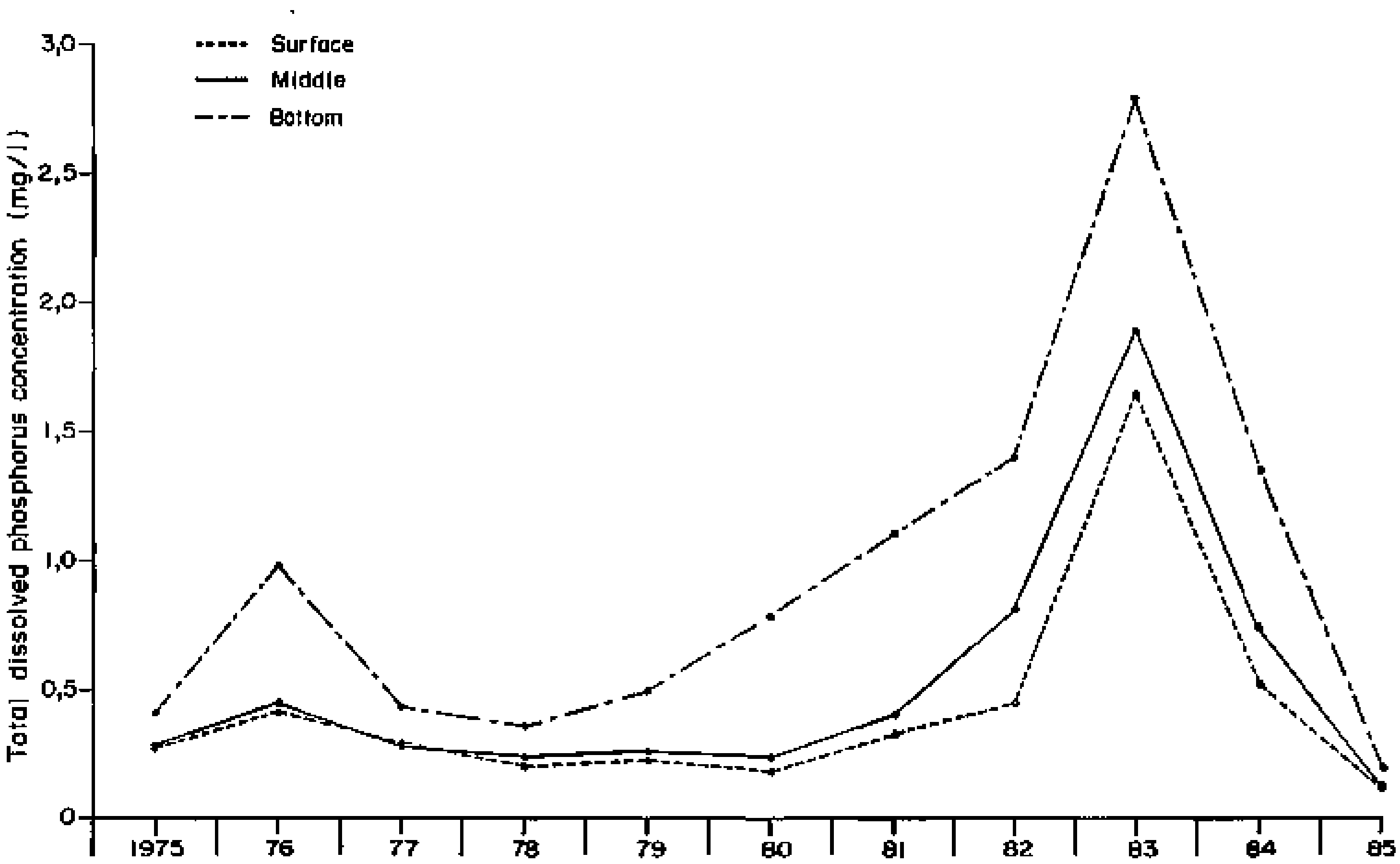

Figure 4.10c. Temporal variation (annual means) in total dissolved phosphorus concentrations (mg/l) of three depths (surface, 6 m below the surface (middle) and bottom) of a lake station in Vernon Hooper Dam. Data from Umgeni Water Board.

concentrations in the bottom waters result from the processes of sedimentation and decomposition which release phosphorus, and the greater release of phosphorus compounds from the sediments under anaerobic conditions.

For the period 1975 to 1985 the TDP concentrations of surface and bottom water of Vernon Hooper Dam, measured by the Umgeni Water Board, were considerably higher than those recorded in several South African impoundments (Table 4.5), indicating that between 1975 and 1985 Vernon Hooper Dam was highly eutrophic. However, for 1976 the TDP concentrations were considerably higher than those recorded by Archibald & Warwick (1980), possibly because the technique used by the Umgeni Water Board was not as sensitive as that of Archibald and Warwick (Archibald, pers. comm.).

Table 4.5: Mean total dissolved phosphorus concentrations (mg/l) of surface and bottom water from lake stations in some South African impoundments (data from Walmsley & Butty, 1980).

	Total dissolved P concentration (mg/l)		
Trophic Classification	Surface	Bottom	Study period
<u>Oligotrophic</u>			
Nagle	0,004	0,007	Nov 1977-Nov 1978
Albert Falls	0,003	0,004	Nov 1977-Nov 1978
<u>Oligo-mesotrophic</u>			
Midmar	0,007	0,008	Nov 1977-Nov 1978
Henley	0,008	0,019	Nov 1977-Nov 1978
<u>Mesotrophic</u>			
Rust der Winter	0,032	0,055	Aug 1977-July 1978
Bronkhorstspruit	0,054	0,089	Aug 1977-Dec 1976
<u>Eutrophic</u>			
Loskop	0,040	0,047	Aug 1977-July 1978
Vernon Hooper	0,045	0,047	Jan 1976-Dec 1976
*Vernon Hooper	0,410	0,980	Jan 1976-Jan 1977
*Vernon Hooper	0,475	1,075	Jan 1976-Dec 1984
<u>Highly eutrophic</u>			
Bospoort	0,063	0,123	Aug 1977-July 1978
Rietvlei	0,650	0,930	Jan 1976-Jan 1977

* Data from Umgeni Board Water Board.

Relative to the 1980 to 1985 period, the TDP concentration of surface and middle waters between 1975 and 1980 were generally low (ca 0,250 mg/l) and not very variable. There were two peaks in TDP concentration, a fairly small peak occurring in 1976 and a larger longer-lasting peak, during the period 1982 to 1984. The peaks were most pronounced in the bottom waters. In 1985 the TDP concentrations of the three stations were the lowest for the whole study period.

On the basis of total dissolved phosphorus concentrations, it is evident that by 1975 Vernon Hooper Dam was already severely eütrophic. The higher TDP concentrations in the bottom water during 1981 to 1984, relative to the period 1975 to 1980 indicate increasing eutrophication and deteriorating water quality. However, in 1985 TDP concentrations in the three lake stations declined, indicating an improvement in water quality to levels observed between 1975 and 1979. There does not appear to be any clear relationship between TDP concentration and water hyacinth possibly because other factors such as drought were more important in determing the TDP concentration.

d) Total dissolved nitrogen (TDN) concentration

The TDN concentration of water from the three depths is variable, the bottom concentrations generally being highest and the surface lowest (Figure 4.10d). This probably reflects the utilization of nitrogen in the surface water by algae, and

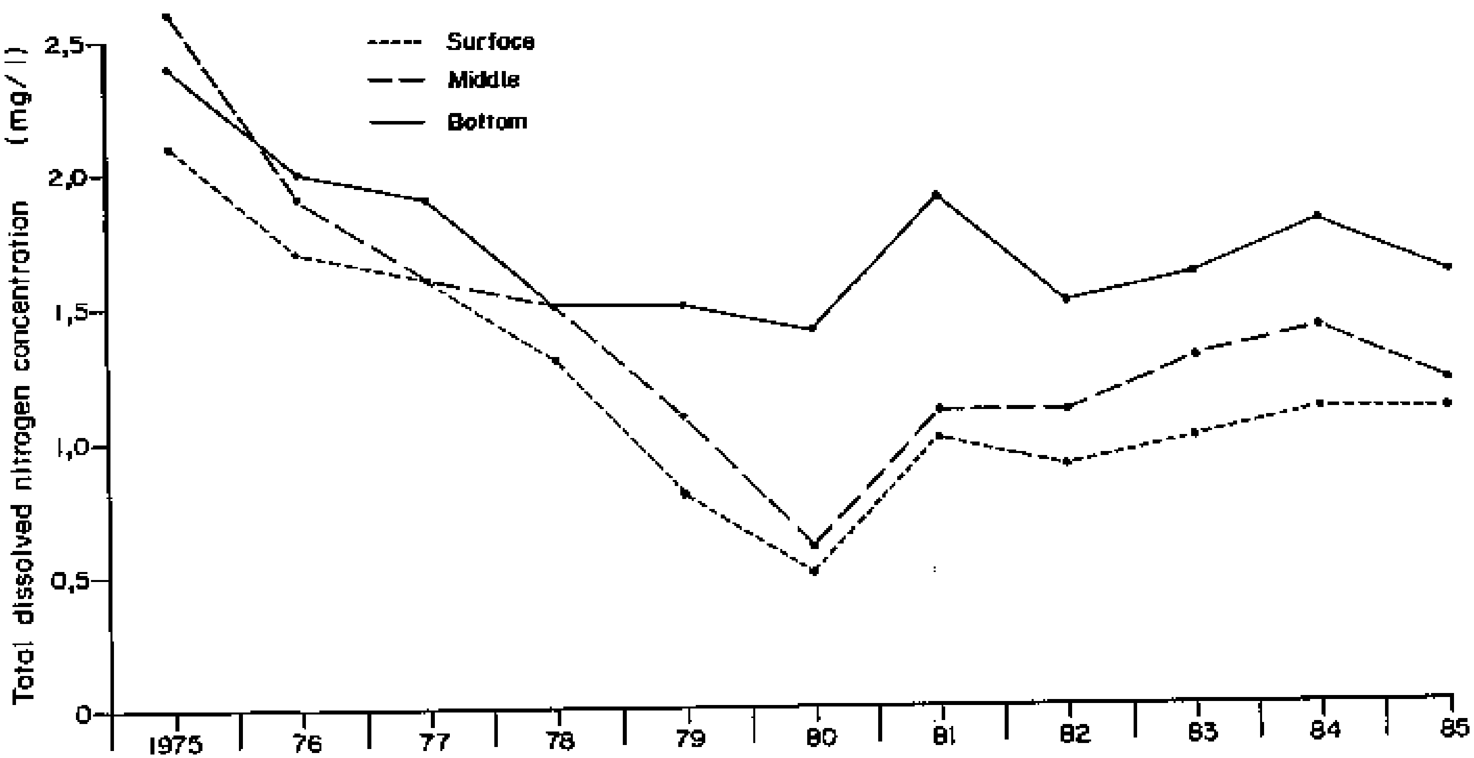

Figure 4.10d. Temporal variation (annual means) in total dissolved nitrogen concentrations (mg/l) of three depths (surface, 6 m below the surface (middle) and bottom) of a lake station in Vernon Hooper Dam. Data from Umgeni Water Board.

sedimentation and decomposition of dead organic matter into the deeper parts of the lake. Periods of low nitrate concentrations since 1979 (Figure 4.10e), suggest that since 1979 nitrogen fixation may have been an important source of nitrogen.

The trend in annual mean TDN concentration of the three lake stations is different from that of the previous variables. Between 1975 and 1985 TDN concentrations were highest in 1975 (ca 2,4 mg/l) and then showed a general trend of decreasing surface and middle TDN concentrations until 1980.

After 1980 TDN concentrations showed an increasing trend reaching ca 1,4 mg/l by 1985. Total dissolved nitrogen is, therefore, the only variable which shows a trend apparently related to the introduction of hyacinth control in September 1981. Hyacinth cover (although not quantified) increased markedly between 1976 and 1980 (Figure 4.3), thereafter it was reduced to occupy a very small area (8 to 20 ha). Thus, the decreasing TDN concentrations were associated with a build up of hyacinth and the increasing concentrations with the decline of hyacinth. This suggests further that harvesting since 1981 has not been effective in removing nitrogen to a degree which regulates the TDN concentration in the lake.

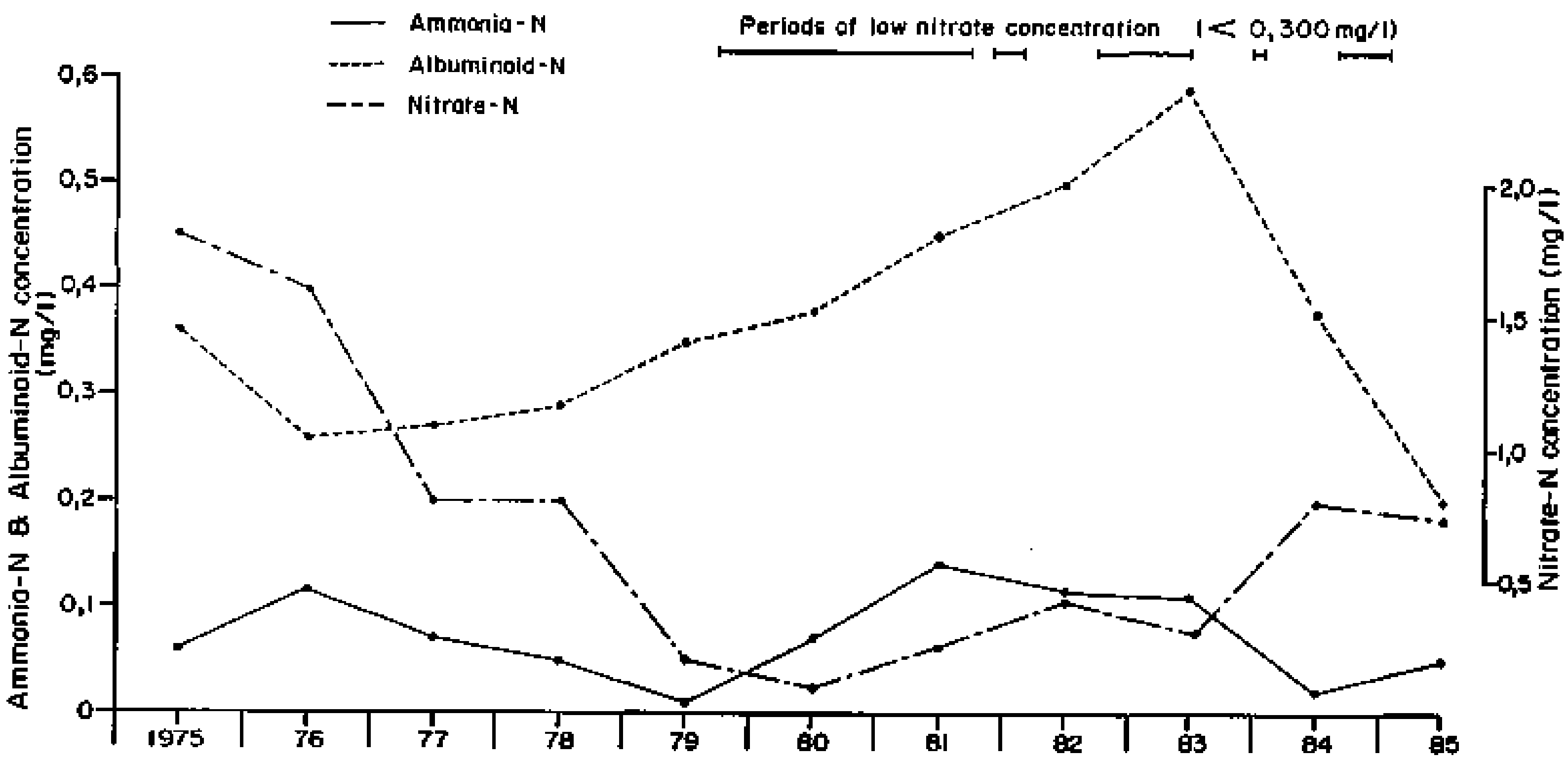

Figure 4.10e. Temporal variation (annual means) in ammonia, nitrate and albuminoid nitrogen concentrations (mg/l) of the surface water of a lake station in Vernon Hooper Dam. Data from Umgeni Water Board.

In 1976 the algal growth rate in Vernon Hooper Dam were frequently phosphorus limited (Archibald & Warwick, 1980). Nitrogen requirements are ca 7 times greater than phosphorus requirements (Goldman & Horne, 1983) and between 1975 and 1985 the TDN:TDP ratio was always less than 6,5 (Figure 4.10f). This suggests that, generally, primary production in Vernon Hooper Dam was not limited by phosphorus availability. The lower N:P ratios since 1980 indicate that phosphorus limitation would be unlikely at the present time.

e) pH

Each year the mean pH of the surface water was higher than that of middle water which in turn was higher than that of bottom water (Figure 4.10g). This is expected since high pH values are associated with high rates of photosynthesis. Between 1975 and 1985 the annual mean pH (7,68) was very variable (range 7,00 - 8,27) and no general trend of pH with time was apparent. This is probably because Vernon Hooper Dam was already eutrophic in 1975 and supported dense algal growths (Howes, 1976 and Archibald & Warwick, 1980).

f) Manganese Concentration

The manganese concentration in lake water is usually a reasonable index of the water quality and trophic state of an impoundment since it is a measure of the extent to which reducing conditions develop in the bottom waters. High

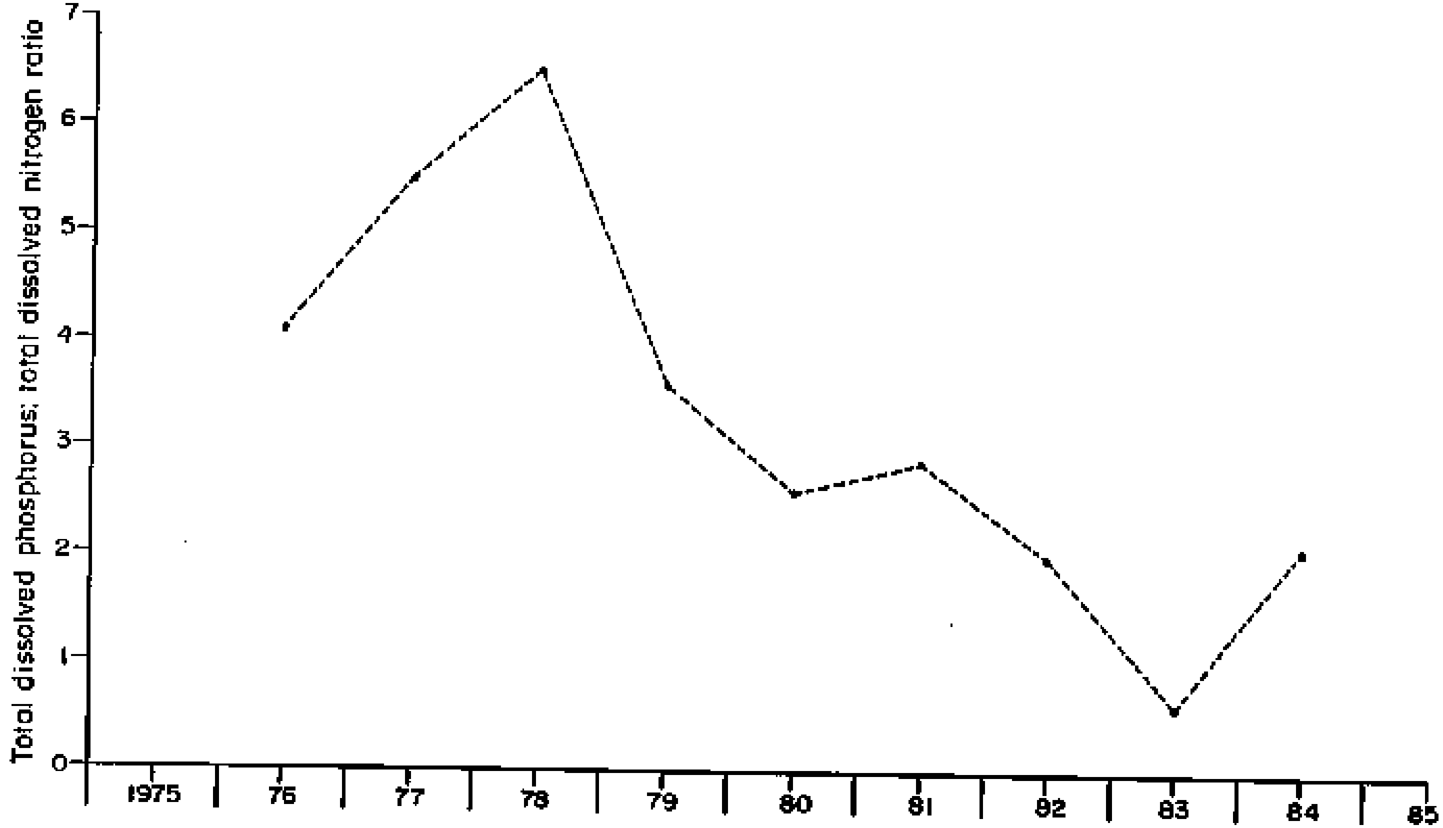

Figure 4.10f. Temporal variation (annual means) in total dissolved nitrogen: total dissolved phosphorus ratio of the surface water of a lake station in Vernon Hooper Dam. Data from the Umgeni Water Board.

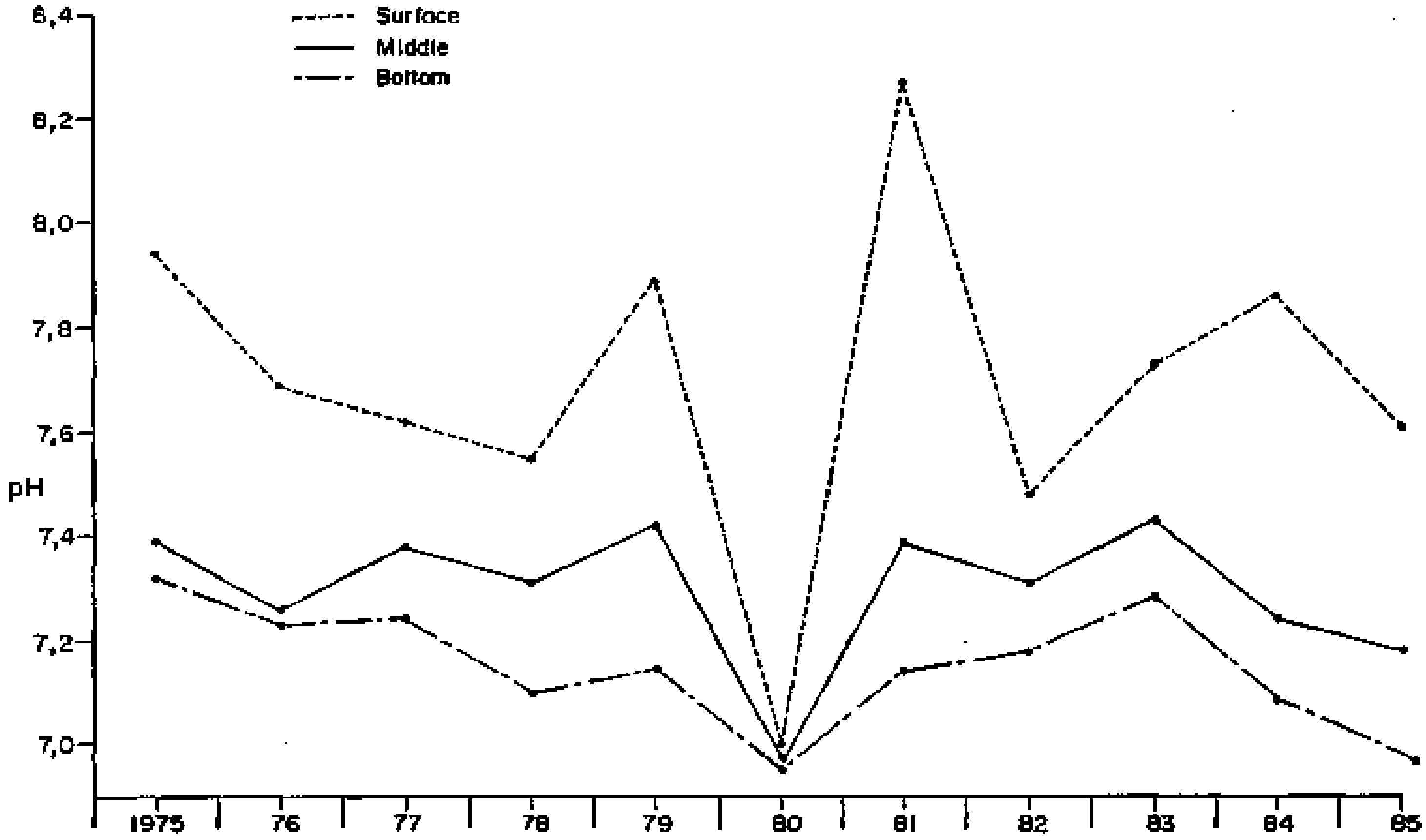

Figure 4.10g. Temporal variation (annual means) in pH of three depths (surface, 6 m below the surface (middle) and bottom) of a lake station in Vernon Hooper Dam. Between March 1980 and September 1981 pH was measured only five times, thus the 1980 and 1981 mean pH values are not true annual means. Data from Umgeni Water Board.

manganese concentrations are likely to occur first at the sediment/water interface, but as the lake becomes increasingly anoxic, high levels will also occur closer to the water surface. Generally, high levels are restricted to the hypolimnion in stratified systems such as Vernon Hooper Dam.

Each year the mean manganese concentration of the bottom water was considerably higher than that of the middle water which was higher than that of the surface water (Figure 4.10h). Between 1975 and 1980 the mean manganese concentrations at all depths were generally lower than at corresponding depths since 1980, the difference between the two periods being most marked in the bottom water. The manganese concentrations of surface and middle water during 1984/85 were similar to those of the 1975 to 1980 period; however the bottom water does not show a similar decrease during 1984/85. Although the lake was already eutrophic in 1975, the data imply substantially worse conditions since 1980. The build up of hyacinth up to 1980 seems to have had little discernable effect on manganese concentrations. Despite restriction of hyacinth to 20 ha and less since September 1981, manganese concentrations increased. Other factors, such as inflow volume, water level and flushing rate probably played an important role in determining manganese concentrations in the lake.

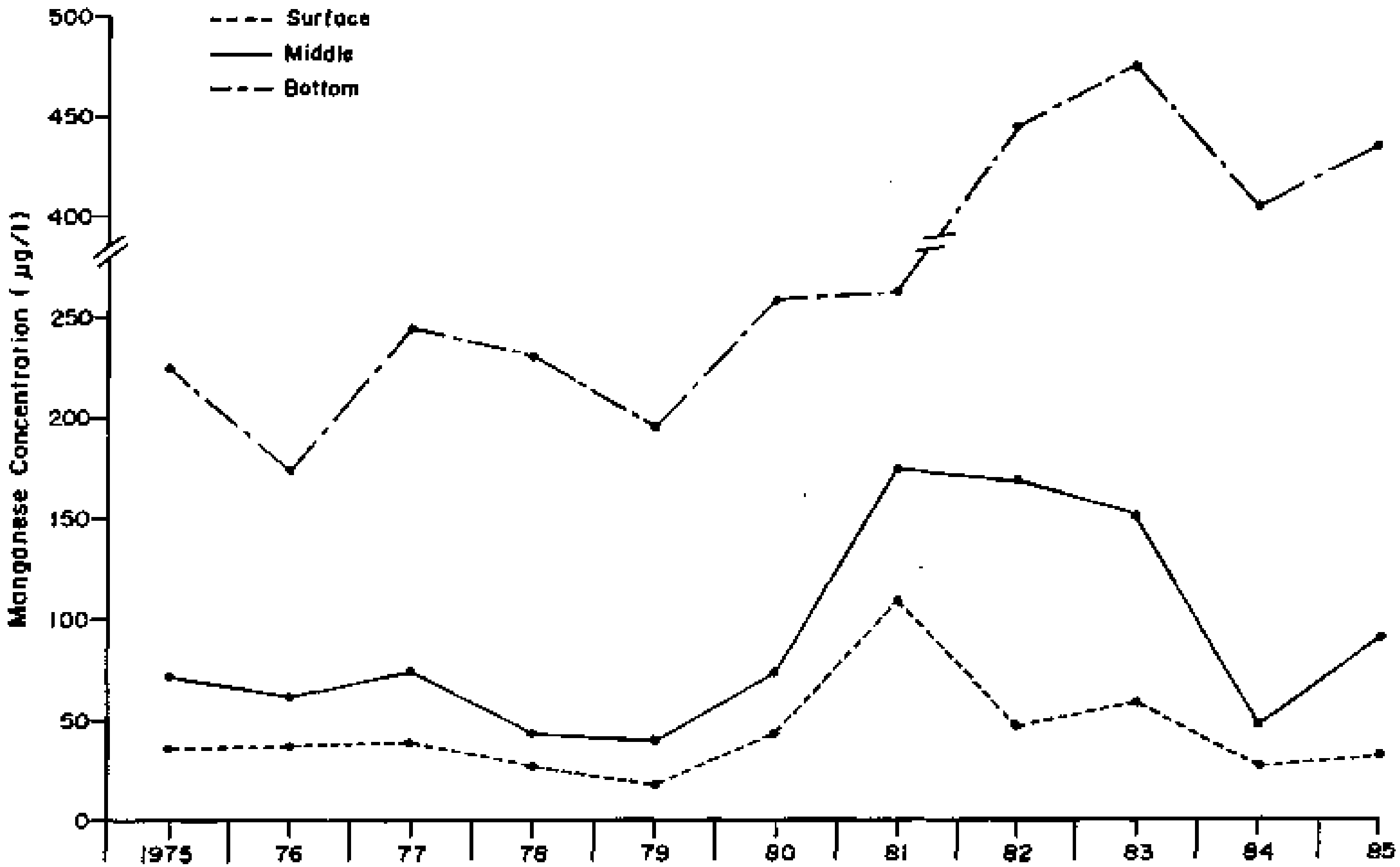

Figure 4. 10h : Temporal variation (annual means) in manganese concentrations (µ g/l) of three depths (surface, 6 m below the surface (middle) and bottom) of a lake station in Vernon Hooper Dam. Data from Umgeni Water Board.

iii. Conclusions

The temporal variation in several indicators of lake water quality suggested that since the 1975 to 1980 period, water quality in Vernon Hooper Dam deteriorated. The periods of poor water quality (often concomitant with drought and consequent reduced flushing rates) seem to have been more severe in terms of water treatment. The periods of good water quality have not shown any marked deterioration. Since Vernon Hooper Dam is already severely eutrophic and receives industrial effluents from Hammarsdale, factors other than the presence of the hyacinth are probably of overriding significance in determining the water quality. If hyacinth were not present, the water quality would be worse;however it is not possible to quantify this effect. The direct and indirect effects of water hyacinth on the phytoplankton population are assesed in the next section.

3. INTERACTION BETWEEN HYACINTH COVER AND PHYTOPLANKTON

3.1 INTRODUCTION

Chlorophyll a concentrations and algal numbers indicate an increase in the phytoplankton standing crop in Vernon Hooper Dam since 1976. However, it is difficult to establish the magnitude of the increase because whilst chlorophyll concentration increased by ca 2,5 times since 1976 (Figures 4.5 and 4.6), algal numbers showed a substantially greater increase, from ca 200 cells/ml in 1976 to ca 170 000 cells/ml in 1984 (Section 2.2.i). These discrepancies result from differences in methodology, sampling bias and species differences in the chlorophyll content per cell.

The phytoplankton species composition apparently changed from populations frequently dominated by diatoms to more frequent dominance by cyanobacteria (Section 2.2. ii).

Notwithstanding the low levels of accuracy attributable to these data, an increase in the phytoplankton standing crop and a change in the phytoplankton species composition, is evidence that suggests a significant deterioration in water quality. The importance of *Eichhornia* in determining algal standing crops and species composition in Vernon Hooper Dam is considered below.

3.2 PHYTOPLANKTON STANDING CROP

Nutrient loads

Total dissolved nitrogen and phosphorus loads to Vernon Hooper Dam between 1975 and 1984 are shown in Figures 4.11 and 4.12 respectively. The TDN load decreased from ca 168 t/a in 1976 to ca 30 t/a in 1980, after which loads followed a slight decreasing trend until 1984 when the load more than doubled, reaching ca 75 t/a. The TDP load decreased from ca 46,5 t/a in 1976 to ca 10,5 t/a in 1980. Since 1980 TDP loads have increased substantially to ca 32,0 t/a in 1983 and later to their highest level, ca 84 t/a in 1984. The changing N and P loads resulted in the ratio of N:P loads being less than one since 1982, making nitrogen limitation of algal productivity more likely than phosphorus limitation.

Algal response to the nutrient load will be influenced by the hydrology of the impoundment because this determines the replacement period and flushing rate. The storage and inflow volumes (Figures 4.13 a and b) are generally high in summer and low in winter. Between 1979 and 1983 inflow volumes were particularly low (ca 10 x $10^6 m^3$/six months), resulting in lower storage volumes during this period, the storage volume reaching its lowest (ca 1,4 x $10^6 m^3$) in winter 1983. Between 1976 and 1979 and in 1984 the replacement period (Figure 4.13c) was short (less than 0,16 years), but between 1980 and 1983 the replacement period was approximately double this value.

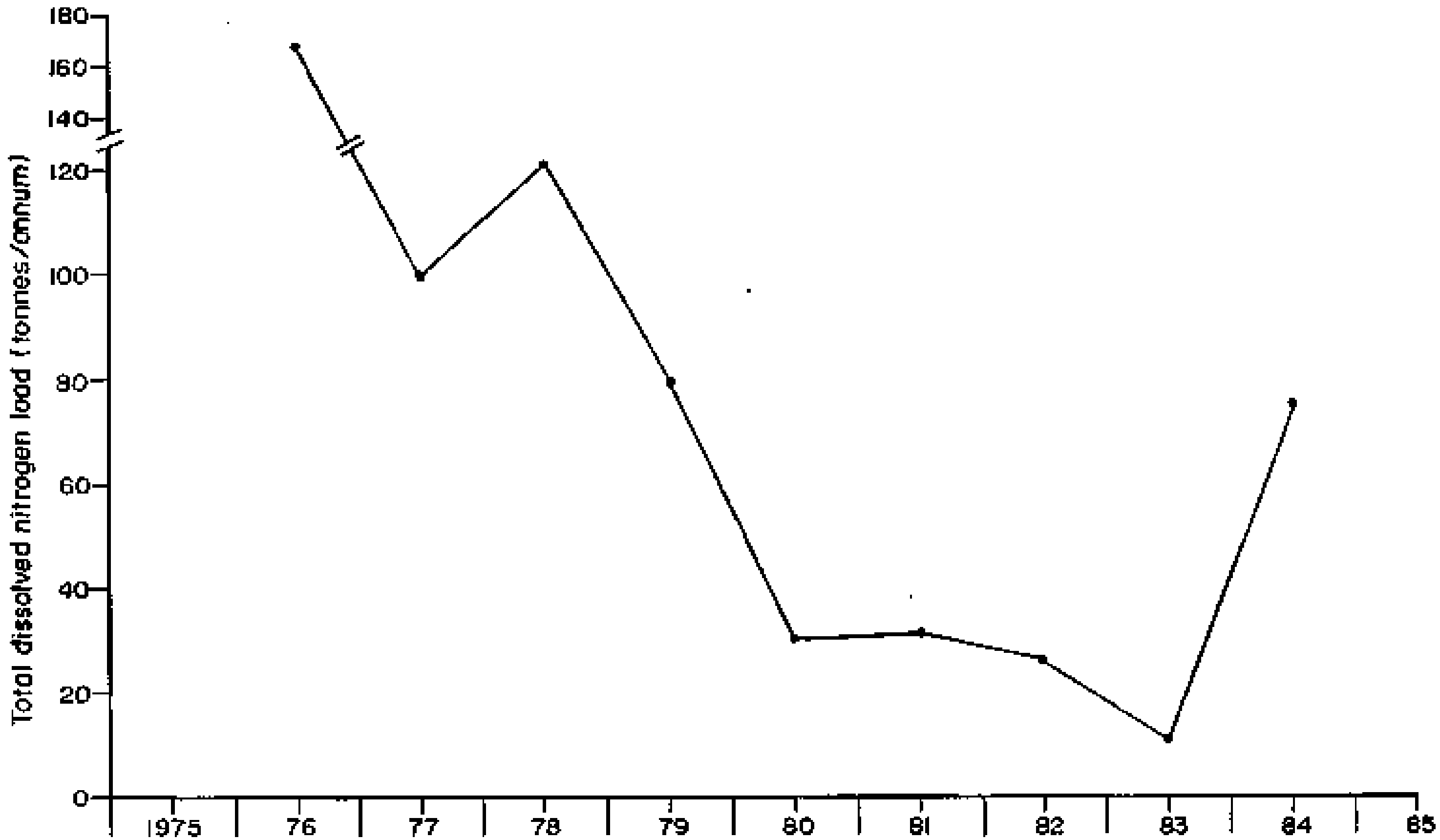

Figure 4.11: Total dissolved nitrogen load (tonnes/annum) to Vernon Hooper Dam from 1976 to 1984. Data from Umgeni Water Board.

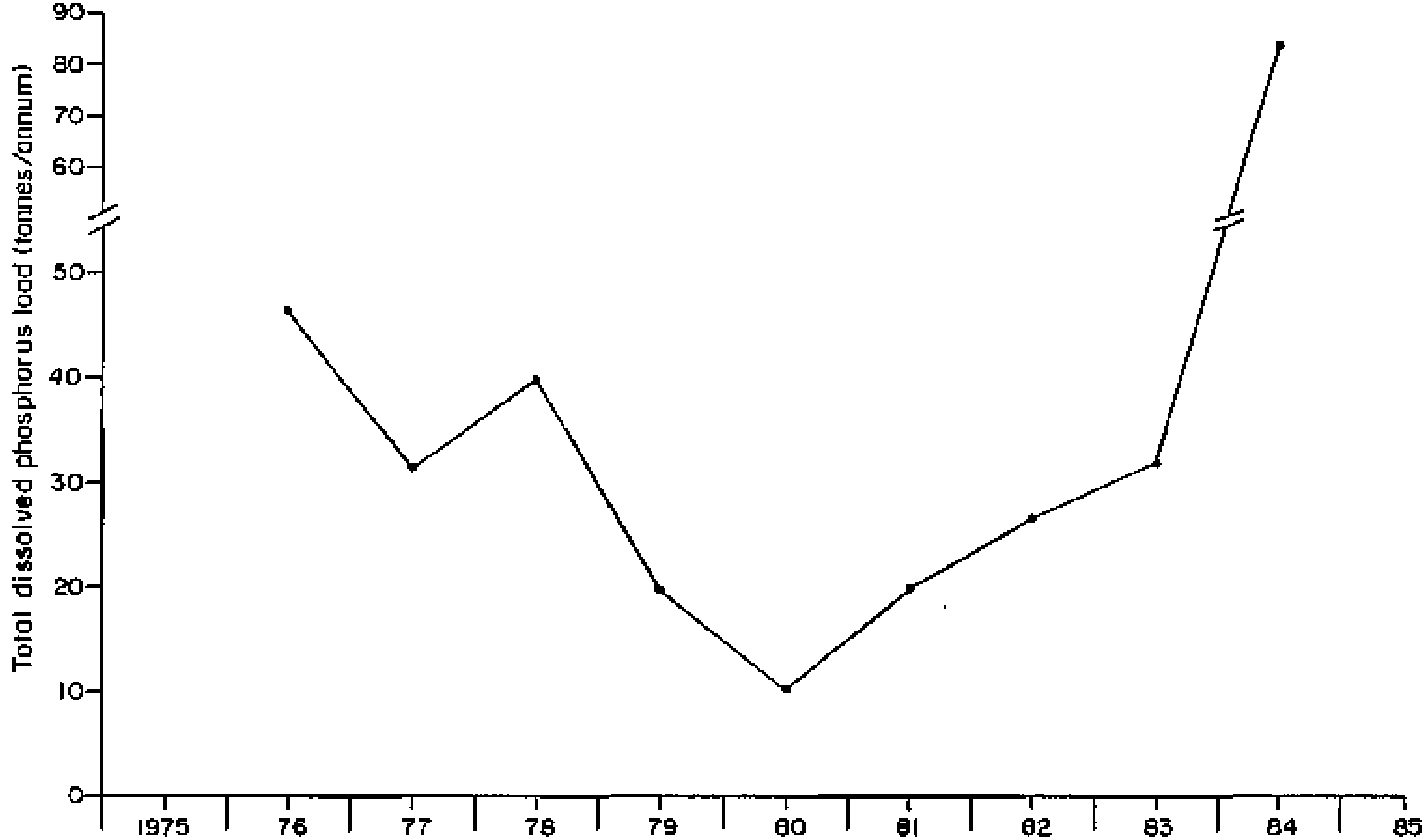

Figure 4.12: Total dissolved phosphorus load (tonnes/annum) to Vernon Hooper Dam from 1976 to 1984. Data from Umgeni Water Board.

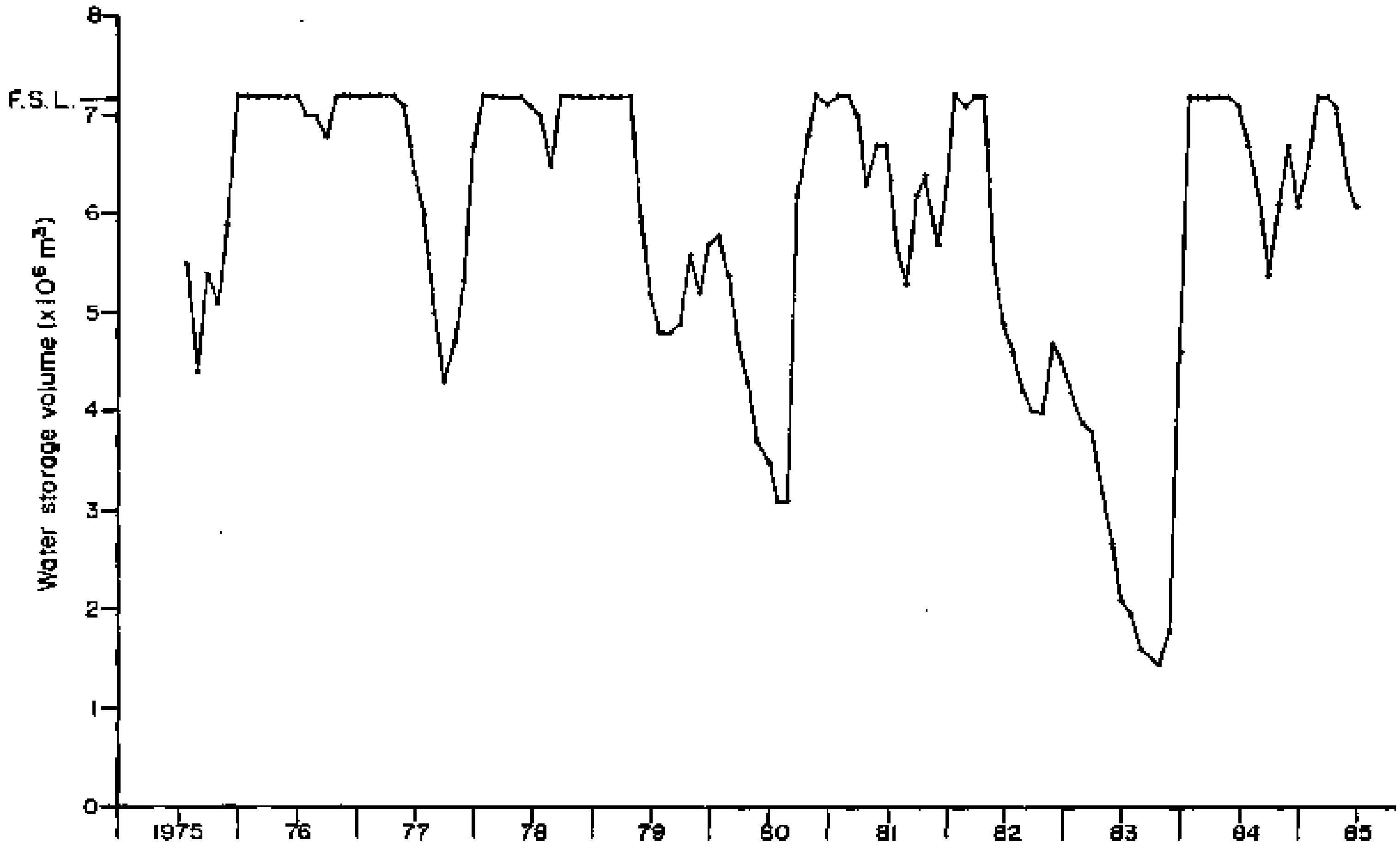

Figure 4.13a. Water storage (x $10^6 m^3$) in Vernon Hooper Dam from 1975 to 1985. Data from Umgeni Water Board. F.S.L. is the full supply level.

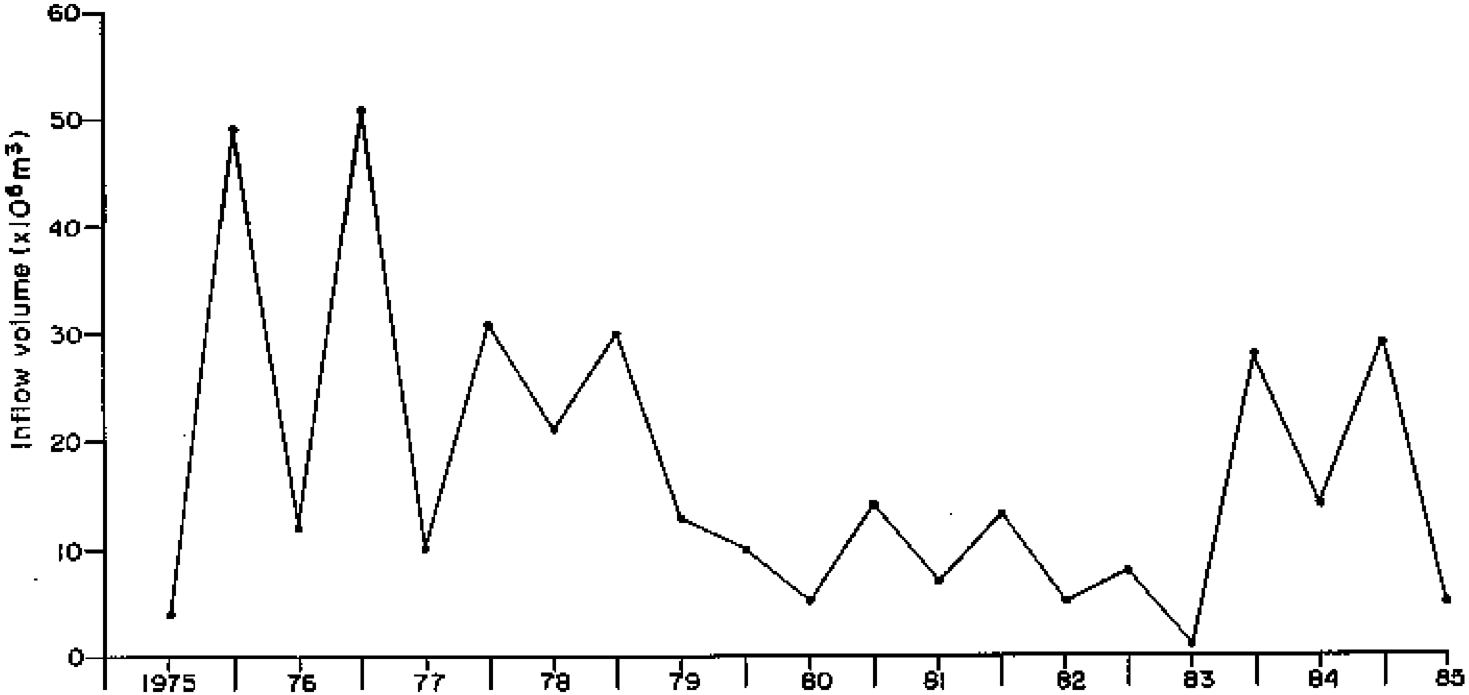

Figure 4.13b: Summer (October to March) and winter inflow volumes (x $10^6 m^3$) to Vernon Hooper Dam from 1975 to 1985. Data from Umgeni Water Board.

Figure 4.13c. Mean annual residence time (per annum) of Vernon Hooper Dam from 1976 to 1984. Data from Umgeni Water Board.

A number of equations have been developed to predict chlorophyll a concentrations from total phosphorus loads and hydrological data (e.g. Vollenweider, 1975; Lee, Rast & Jones, 1978 and OECD, 1982). Since phosphorus is not presently the growth rate limiting nutrient, it has had limited value for prediction of algal chlorophyll a concentrations in recent years at Vernon Hooper Dam. Nitrogen loads are generally not used to predict algal standing crops because of the potential input by fixation of atmospheric nitrogen. Consequently, currently available models use phosphorus loading to predict change in algal standing crop even though it is generally accepted that phosphorus may not be the nutrient limiting algal productivity. The equation described by Lee, Rast & Jones (1978) was derived from data from a large number of temperate Northern Hemisphere lakes. The OECD modified this equation to reduce an over-prediction error.

The OECD (1982) equation is:

$$\text{Chl } \underline{a} = 0{,}37\ [(L(P)/q_s)/(1 + \sqrt{\tau_w})]^{0{,}79}$$

where Chl a = mean annual euphotic zone chlorophyll a concentration (mg/m^3)

$L(P)$ = annual areal phosphorus load ($mgP/m^2/a$)

q_s = hydraulic loading (m/a)

τ_w = hydraulic residence time (a)

This equation was used to predict the chlorophyll a concentration and thus the potential effect of phosphorus loads and hydrological conditions of Vernon Hooper Dam on the phytoplankton populations between 1976 and 1984 (Figure 4.14).

Predicted chlorophyll a concentrations between 1976 and 1980 were fairly constant and although low (ca 45 mg/m^3) relative to those subsequent to 1980, they were high relative to chlorophyll a concentrations normally encountered, indicating that by 1976 Vernon Hooper Dam was already eutrophic. Since 1980 predicted chlorophyll increased to a maximum of ca 175 mg/m^3 in 1983 and then decreased to 112 mg/m^3 in 1984. The predicted chlorophyll a concentration for 1984 was ca 2,7 times higher than that for 1976, indicating a potential increase in the algal standing crop of ca 2,7 times since 1976 or a change in species composition to species with higher chlorophyll a contents per unit mass. The observed increase in chorophyll a concentrations between 1976 and 1984/5 was of similar magnitude (ca 2,5 times); but on both occasions observed values were less than 50% of predicted values. This poor level of prediction could imply that phosphorus has not been the growth rate limiting nutrient since 1976, a view which is supported by the observed low N:P ratios during this period (Figure 4.10f). That the equation used had not been developed taking southern African data into account probably contributed to the low level of prediction obtained, as could poor sampling strategy.

It seems that in recent years the algal growth rate in Vernon Hooper Dam has been limited by nitrogen, thus the use of phosphorus loads to predict algal standing crops is not justified. In an attempt to take nitrogen limitation into account, predicted chlorophyll a concentrations were corrected by the N:P ratio of the inflow based on an N:P requirement of 7:1 by weight (Goldman & Horne, 1983). Corrected chlorophyll a concentrations (Figure 4.14) between 1976 and 1984 were fairly constant and low (ca 15 mg/m^3). In 1976 the observed (19,8 mg/m^3) and modified predicted (21,2 mg/m^3) concentrations were similar; however, in 1984/5 the observed concentration (49,2 mg/m^3) was ca 3,4 times higher than the predicted concentration for 1984 (14,4 mg/m^3). The elevated chlorophyll a concentrations and the low nitrate concentrations, especially since 1980 (Figure 4.10e) indicate access to additional nitrogen, and a large input of nitrogen through nitrogen fixation is implicated. It is evident therefore that neither phosphorus nor nitrogen river inputs can be used to accurately predict algal standing crop in Vernon Hooper Dam.

The deterioration in water quality after the 1975-80 period probably reflects both increased nutrient inputs reaching the impoundment (from river loading and possibly atmospheric nitrogen fixation) and a reduced effect of Eichhornia, as the introduction of hyacinth control in September 1981 reduced hyacinth cover from ca 80 ha to less than 20 ha.

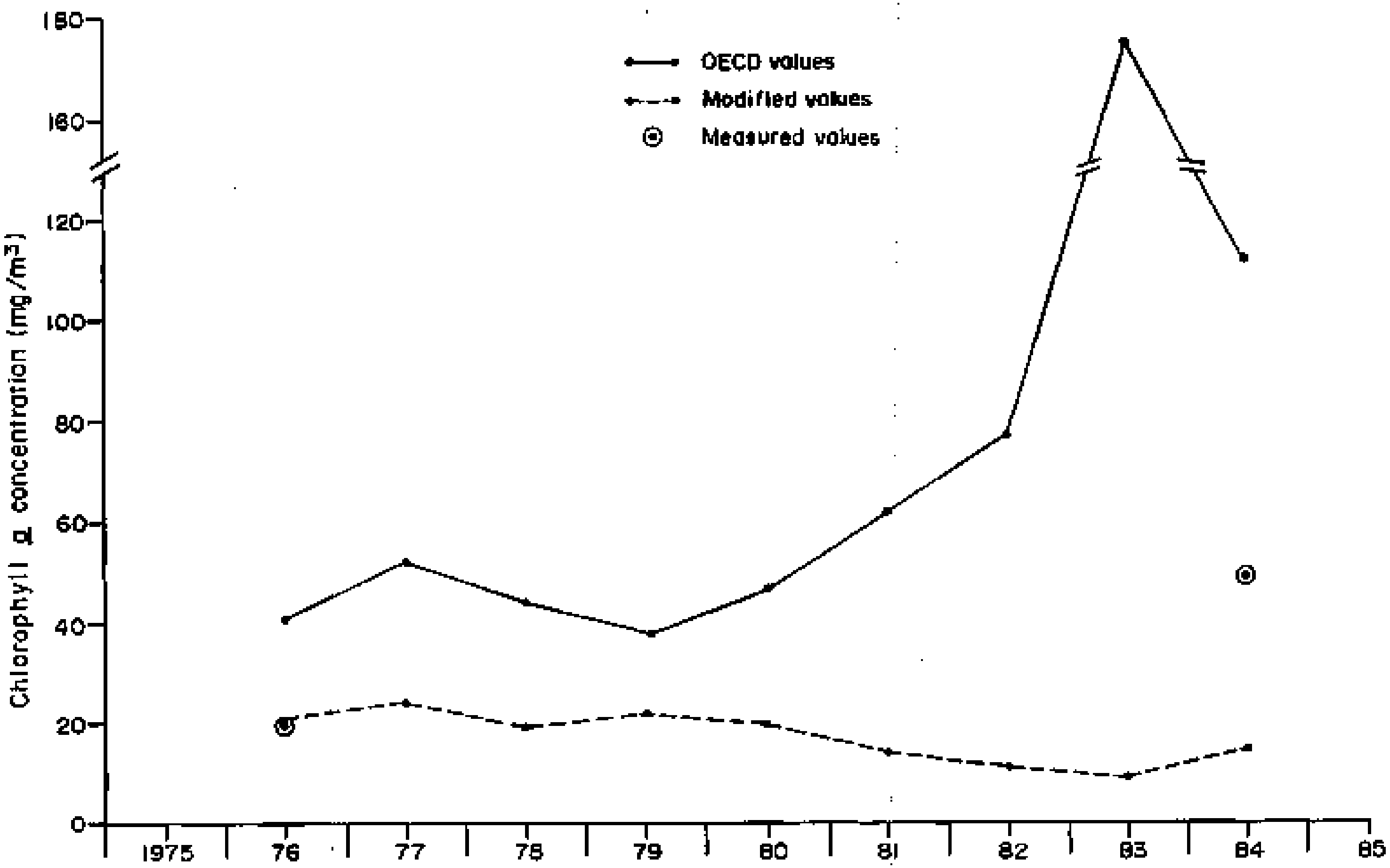

Figure 4.14: Measured, predicted (OECD) and modified predicted chlorophyll a concentrations (mg/m^3) in Vernon Hooper Dam between 1976 and 1984. Chlorophyll data for 1976 from Archibald (pers. comm.).

Nutrient removal by *Eichhornia*

In 1974 *Eichhornia* covered *ca* 80 ha of Vernon Hooper Dam (i.e. the entire surface area). By September 1975 it had been completely removed and, although not monitored, it increased markedly between 1976 and 1980 following its reintroduction to the lake. Since 1981 populations have been maintained at between *ca* 8 and 20 ha (Figure 4.3) by confining the hyacinth to one sector of the impoundment and harvesting it periodically.

Musil & Breen (1985) used a growth kinetics model developed for *Eichhornia* to identify the limiting nutrient and predict population sizes, yields, growth rates and frequencies and amounts of harvest necessary under varying conditions of nutrient loading and climate. They found that during the period December 1981 to August 1983, when inflow and nutrient loads were low, a 20 ha population of hyacinth would have been sufficient to remove those fractions of the phosphorus inflow loads that are readily available for plant growth. However, during this period the impoundment was often not full; therefore, less than 20 ha of surface would have been available for hyacinth growth in the Mlazi arm. Since the Mlazi arm is relatively shallow (Figure 3.1) a decrease in the lake volume would result in a larger decrease in the surface area of the Mlazi arm than in the rest of the impoundment. Also, estimates of the amounts and frequencies of harvest required to remove the predicted yields of a 20 ha population indicate that during

Table 4.6: Nitrogen and phosphorus half saturation coefficients (Toerien, 1974).

Organism	Growth rate limiting nutrient	Half saturation coefficient (µg/l)	Reference
*Chlorella pyrenoidosa**	N	1 400 – 3 000	Shelef *et al*.(1968).
*Chlorella pyrenoidosa***	N	700 – 14 000	Shelef *et al*.(1968).
Mixed algae	N	450	Shelef *et al*. (1968)
Selenastrum gracile	N	150	Middlebrooks *et al*. (1971).
Eichhornia crassipes	N	399,8 - 1505,6 (mean 976)	Musil (1982)
*Chlorella pyrenoidosa**	P	55	Zabat *et al*.(1970)
*Chlorella pyrenoidosa***	P	21-29	Zabat *et al*. (1970)
Selenastrum gracile	P	10	Middlebrooks *et al*. (1971).
Selenastrum capricornutum	P	3,7-5,7	Toerien *et al*. (1971)
Eichhornia crassipes	P	41,1 - 161,8 (mean 94,1)	Musil (1982)

* High temperature strain
** Emersion strain.

cover increases the surface boundary layer thereby reducing wind-induced mixing and increasing sedimentation rates. However, the dominant physical effect of Eichhornia cover is the barrier it provides to the penetration of light. Moderately crowded populations reduce the light to less than 1% of the surface irradiance (Ashton, pers. comm.). Below a mat of Eichhornia, algal photosynthesis is greatly reduced. In addition, the decreased energy input would tend to lower surface water temperatures. However, in Vernon Hooper Dam in 1984/5 there was no significant difference ($p < 0,05$) in the surface water temperatures of the inflow to the Eichhornia (Mlazi), the Eichhornia station and the lake station. The effect of Eichhornia on light and temperature since its control in September 1981 is likely to have been small because the plants are maintained at ca 10% of the surface area of the impoundment increasing to 25% for short periods. Prior to September 1981, especially during maximum cover, its effect on light and temperature would have been more marked. Unfortunately it is not possible to differentiate between the effects of nutrient removal and of shading, since the periods of maximum nutrient removal by Eichhornia coincide with those of maximum shading.

Toxins

The observations, following the introduction of hyacinth control, that the standing crop of Eichhornia was small and the reported reduction in chemical treatment costs was large, led to the suggestion that *Eichhornia* was producing a substance which inhibited algal growth. Results of algal bioassays during 1984/85 (Chapter 3), indicated that inhibitors did not play a significant role in determining algal standing crop and water quality in Vernon Hooper Dam. Since the reduction in water treatment costs did not occur and the algal standing crop apparently did not decrease, it is unlikely that an inhibitor was present.

3.3. PHYTOPLANKTON SPECIES COMPOSITION

In addition to removing nutrients and influencing light and temperature, *Eichhornia* also lowers oxygen concentrations, pH and alkalinity and increases dissolved carbon dioxide concentrations (Ultsch, 1973; McVea & Boyd, 1975 and Gopal *et al*., 1984). These changes and others, such as altered nutrient loads, contribute to changes in the algal species composition of an impoundment. In Hartbeespoort Dam diatoms dominated the phytoplankton population when *Eichhornia* cover was high and cyanobacteria dominated when *Eichhornia* cover was low (Chapter 5). In Vernon Hooper Dam the period of *Eichhornia* build up (between 1976 and 1980) coincided with a period when diatoms dominated the phytoplankton populations *ca* 80% of the time and cyanobacteria only *ca* 10% of the time. Unfortunately there are no measurements of the algal species composition between 1980 and 1983; however, in 1984, a period of relatively low *Eichhornia* cover, cyanobacteria dominated the phytoplankton more frequently (*ca* 42% of the time) and diatoms dominated less frequently (*ca* 29% of the time). This suggests that the reduction in the hyacinth standing crop could have resulted in a higher incidence of cyanobacteria, but the evidence is inconclusive.

3.4 CONCLUSIONS

Although hyacinth cover influenced algal standing crop and species composition when cover was high, recently (since September 1981) the effect of *Eichhornia* on phytoplankton populations and other conditions in Vernon Hooper Dam has decreased because *Eichhornia* cover has been reduced to 20 ha or less.

4. GENERAL CONCLUSIONS

Water quality has deteriorated and chemical usage has increased since the introduction of hyacinth control to Vernon Hooper Dam in September 1981. Increased nutrient inputs to the system and detrimental hydrological conditions were reflected by the deterioration in water quality after the 1975-80 period. Furthermore, because hyacinth cover was restricted to 20 ha or less within the Umlazi arm, the effect of the population on the overall water quality of the impoundment was minimised. When *Eichhornia* covered the entire surface of Vernon Hooper Dam it would tend to reduce the algal standing crop by nutrient removal, shading, higher sedimentation rates and changes in other factors which could result in a change in the species composition of the algae. Under the present conditions of eutrophy and low *Eichhornia* cover, the presence of *Eichhornia* decreases algal standing crops, however, it is not sufficient to maintain the algal standing crop within acceptable limits. Also, prior to the introduction of hyacinth control, chemical treatment costs were much less than present water treatment costs, particularly since the cost of controlling water hyacinth must be added to the cost of chemical treatment. It is unlikely that *Eichhornia* produces an algal growth inhibitor. Algal standing crop has increased markedly over the past few years, principally as a result of elevated nutrient loads from the catchment and from within the impoundment.

5. HARTBEESPOORT DAM CASE STUDY

1. INTRODUCTION

The purpose of this chapter is to study the effects of Eichhornia on the algal populations of Hartbeespoort Dam. The recent report on the limnology of Hartbeespoort Dam (NIWR, 1985)and the report on the chemical control of water hyacinth on Hartbeespoort Dam (Scott *et al.*, 1979) were particularly useful in this regard. Hartbeespoort Dam is a warm, monomictic impoundment (Scott *et al.*, 1977; 1980; Robarts *et al.*, 1982; Robarts, 1984 and NIWR, 1985), overturn occurring in April (Scott *et al.*, 1977; 1980 and Robarts, 1984). It is situated 40 km north of Johannesburg, South Africa. The dam is fed by two main river systems: the Crocodile and Magalies Rivers (Toerien & Walmsley, 1978). The Crocodile River, draining the urban and industrial areas of Johannesburg, contributes approximately 90% of the inflowing water (Toerien & Walmsley, 1978 and Scott *et al.*, 1980) and over 95% of the nutrient load (Scott *et al.*, 1980 and Ashton, 1985). The Magalies River drains a predominantly agricultural area (Toerien & Walmsley, 1978). During 1971/72 the wall was raised by 2,44 m (Olivier, 1977) and in one week (end January 1975) following the heavy rains during the summer of 1974/75, the enlarged lake filled for the first time (Scott *et al.*, 1980 and Ashton, pers. comm.). Some characteristics of the impoundment are given in Table 5.1.

Table 5.1: Characteristics of Hartbeespoort Dam (after Scott *et al*., & NIWR, 1985).

Location	25°43'S; 27°51'E
Altitude	1 170 m above sea level.
Catchment area	4 140 km^2
Surface area (at *F.S.L.)	20 km^2
Capacity (at F.S.L.)	193 x 10^6 m^3
Mean depth (at F.S.L.)	9,6 m
Maximum depth (at F.S.L.)	32,5 m
Maximum summer temperature	26 - 30°C
Minimum winter temperature	9 - 11°C
Temperature stratification	Present from Sept.-April.
Anaerobic hypolimnion	Present from Oct.-April.
Water retention time	Variable, usually 6 months to 1 year.

*F.S.L. = Full supply level.

The mean annual rainfall in the catchment is approximately 700 mm, being concentrated almost entirely during the summer months (October to March; Scott *et al*., 1977; 1980 and NIWR, 1985). Irradiation commonly varies between 150 and 290 W/m^2 (winter and summer respectively; NIWR, 1985). The mean annual air temperature is about 19,5°C (NIWR, 1985). The mean wind-speed on the lake surface is generally low and varies seasonally between 3,6 km/hr in June and 9,0 km/hr in October (NIWR, 1985).

Hartbeespoort Dam was originally built for irrigation purposes (Steÿn et al.,1975; Scott et al., 1980 and Robarts et al., 1982). Of the approximately 130 x 10^6 m^3 abstracted from the lake annually, 82% is used for irrigation, 6% for domestic consumption and 12% is released for other uses by downstream consumers (NIWR, 1985). The lake is also a major centre for water sports, including fishing, boating and water ski-ing (Scott et al., 1977 and NIWR, 1985).

Hutchinson et al. (1932) reported on a visit to Hartbeespoort Dam during April 1928 and on the basis of a low plankton content described it as oligotrophic. Thirty years later, in 1958 Cholnoky likened the dam to an oxidation pond because of its eutrophic nature (cited in Steÿn et al., 1975; Scott et al., 1977; 1980 and Robarts, et al., 1982). In 1961, Allanson & Gieskes described the impoundment as very eutrophic (cited in Robarts et al., 1982). As a result of studies conducted between April 1972 and April 1975, Steÿn et al. (1975) and Scott et al. (1977) described Hartbeespoort Dam as highly eutrophic. It has since been described as hypertrophic (Scott *et al*., 1980; Robarts *et al*., 1982; Robarts, 1984 and NIWR, 1985) and characterised by dense algal populations that are maintained throughout the year (NIWR, 1985) due to an abundant supply of nutrients (Robarts, 1984 and NIWR, 1985), warm temperatures and high insolation (NIWR, 1985).

2. TEMPORAL VARIATION IN *EICHHORNIA* COVER, PHYTOPLANKTON STANDING CROP AND SPECIES COMPOSITION AND NUTRIENT LOADING RATE

a. *Eichhornia* Cover

According to the records kept by the Department of Water Affairs, *Eichhornia* was first noticed on Hartbeespoort Dam in 1959 (Scott *et al*., 1979). For a number of years the plants were controlled by mechanical means (Anon., 1971). The filling of the enlarged lake in 1975 and the resultant inundation of large partly vegetated areas that were virtually inaccessible by boat or vehicle, provided an ideal environment for the rapid proliferation of the *Eichhornia* plants (Scott *et al*., 1979). In April 1975, *Eichhornia* covered approximately 5% of the surface area of the impoundment mostly in the marginal vegetation and boat houses (Ashton, pers. comm.). In 1976 *Eichhornia* growth assumed troublesome proportions (Scott *et* al., 1979) and by October 1977 it occupied between 50 and 60% of the surface area of the impoundment (Scott et al., 1979 and Ashton, pers. comm.). Scott et al. (1979) estimated that by mid-October 1977 the fresh mass of Eichhornia present in the impoundment was 300 000 tonnes.

Since mechanical control was inadequate and biological control organisms were insufficiently tested for use in South Africa, it was decided that chemical control would be the most effective method of controlling Eichhornia on Hartbeespoort Dam (Scott et al., 1979). Two herbicides were used. A herbicide with the active ingredient terbutryn (a triazine derivative), was applied aerially on the following dates:

18 October 1977 (1250 ha sprayed),
3 December 1977 (320 ha sprayed),
30 January 1978 (240 ha sprayed) and
end of March, 1978 (not monitored scientifically).

Fresh growth of Eichhornia necessitated the resprayings in December 1977 and in 1978. Between 1 November 1977 and 31 March 1978, diquat was used to treat approximately 100 ha of marginal Eichhornia that were inaccessible to aircraft (Scott et al., 1979). Terbutryn is partly a systemic herbicide, affecting the photosynthetic processes of plants. It was relatively slow acting and caused the death and submergence of

Eichhornia in four to five weeks (Scott *et al.*, 1979), whilst diquat was rapid-acting, killing *Eichhornia* plants within two weeks (Ashton, pers. comm.).

b. Phytoplankton Standing Crop and Species Composition

The temporal variation in phytoplankton chlorophyll *a* concentrations is shown in Figures 5.1a and b and 5.2a and b. Unfortunately different sampling methods were employed by the the various workers. Sampling between 1973 and 1976 (Figure 5.1a) was monthly or bi-monthly, whereas between 1980 and 1984 sampling was weekly and therefore results were more variable (Figure 5.2a). To reduce the variability of the 1980 to 1984 data, moving averages of four week periods were calculated (Figure 5.2b). A further problem is related to the technique used to collect the water and the distribution pattern of the dominant alga (*Microcystis aeruginosa*). This alga tends to accumulate in the upper 2 to 4 m of the water and exhibits lateral patchiness. The use of 7 or 8 m hosepipes between 1973 and 1978 would have resulted in dilution of the upper water with the algal-poor lower water (Ashton, and Scott, pers. comm.). Also during this period the diameter of the hosepipes used were relatively small, causing an underestimate of the phytoplankton chlorophyll *a* concentration (Ashton, 1985).

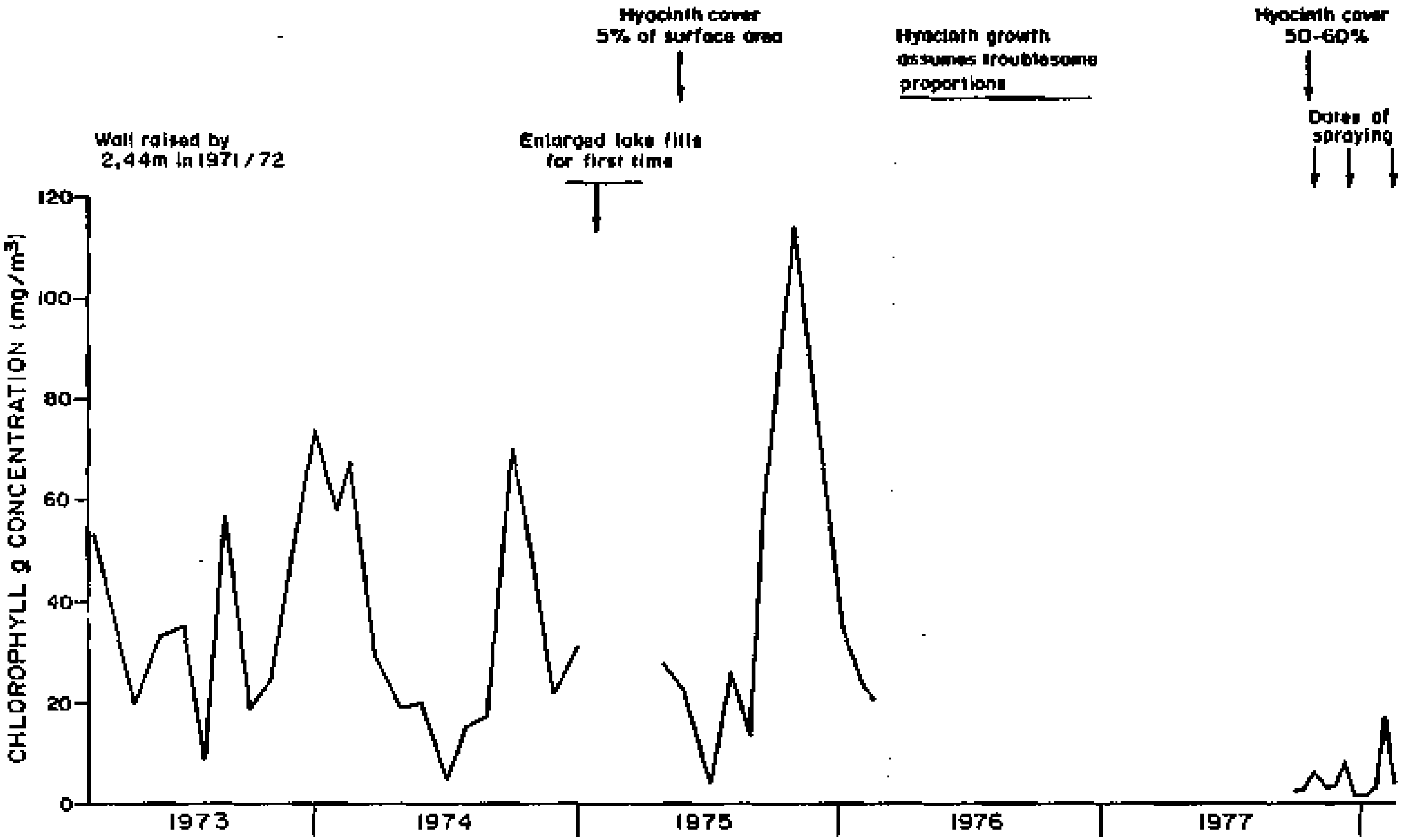

Figure 5.1a: Chlorophyll a concentration (mg/m^3) in Hartbeespoort Dam water sampled with a 7 or 8 m hosepipe. The 1973-1975 data (8 m hosepipe; Scott, pers. comm.) are means of six lake stations. The 1975-1976 data (7 m hosepipe; Ashton, pers. comm.) are means of two lake stations and the 1977-1978 data (7 m hosepipe; Scott et al., 1979) are means of three lake stations.

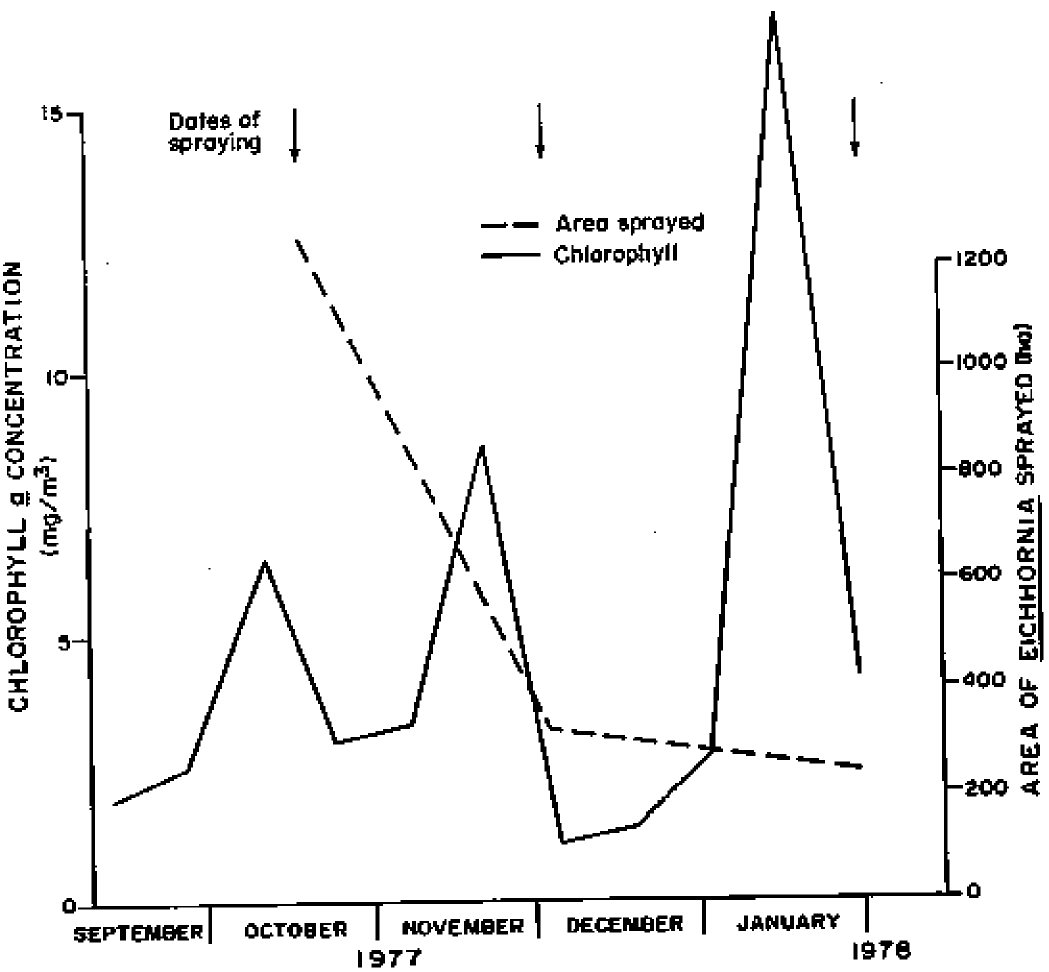

Figure 5.1b: Mean chlorophyll *a* concentrations (mg/m^3, three stations) and areas of *Eichhornia* sprayed in Hartbeespoort Dam between September 1977 and January 1978. Chlorophyll *a* concentrations are the mean of three lake stations. Data from Scott *et al*. (1979).

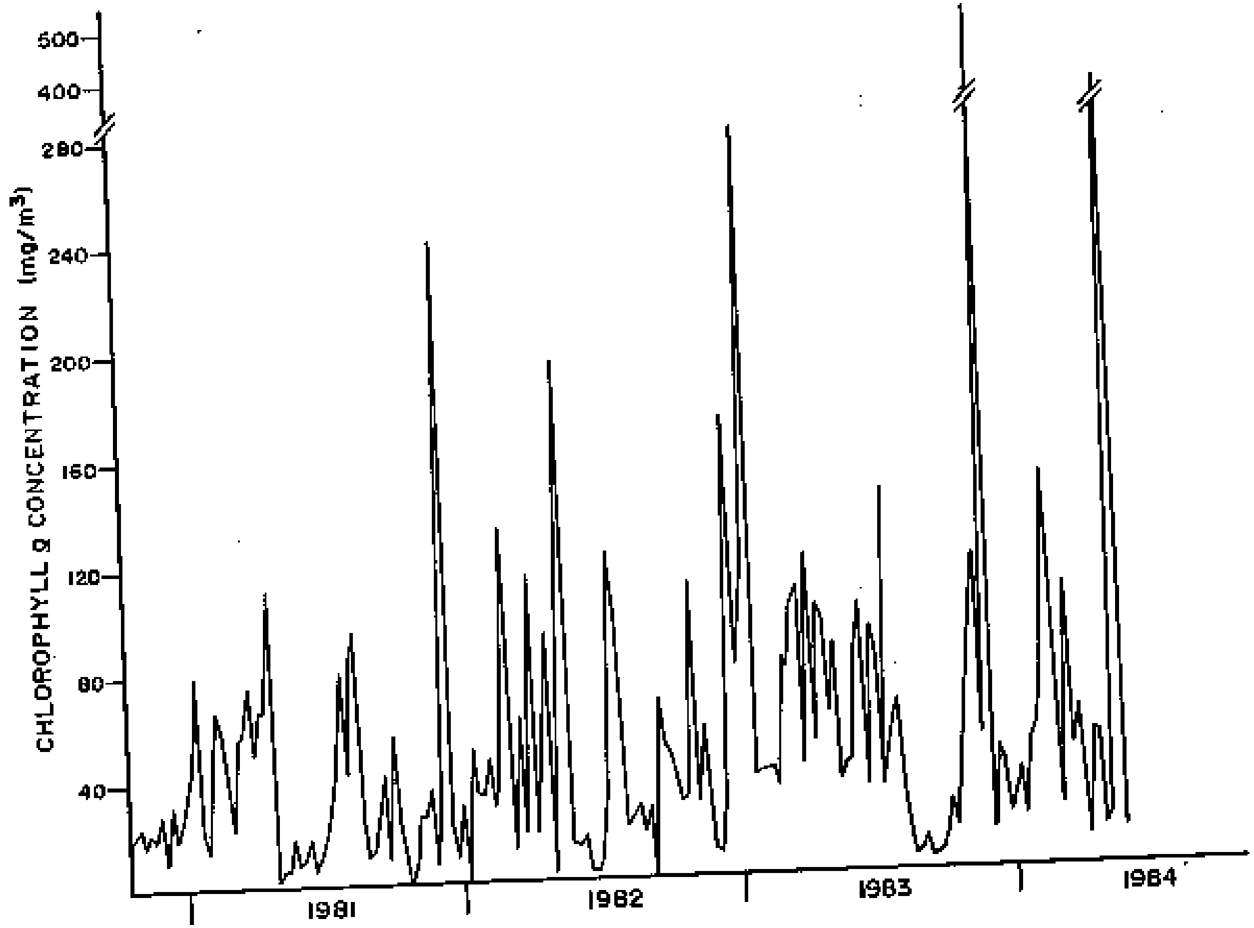

Figure 5.2a. Chlorophyll $\underline{a}$ concentration (mg/m^3) in Hartbeespoort Dam water sampled over six metres for 1980 to 1984. Each point is the value from one lake station. Data from Ashton & Zohary (pers. comm.).

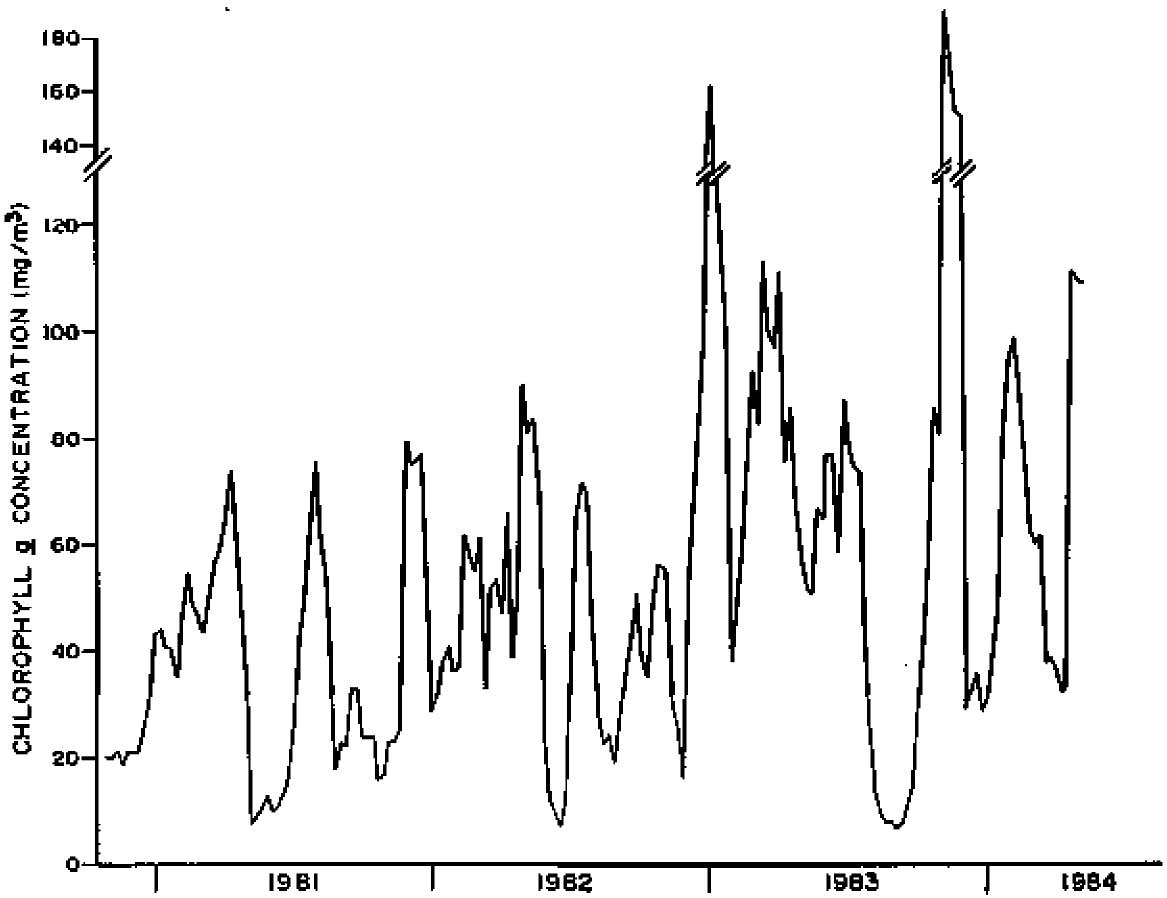

Figure 5.2b. Moving average of chlorophyll a concentration (mg/m^3) in Hartbeespoort Dam water sampled over six metres. Each point is the mean of samples from one lake station and averaged over four weeks e.g. the average concentration for weeks 1 to 4 was plotted at week 2, the average for weeks 2 to 5 was plotted at week 3, etc.

Comparing Figures 5.1a and 5.2b, it is apparent that the 1973 to 1976 and 1981 and 1982 periods exhibited the same range of phytoplankton chlorophyll a concentrations whereas, those recorded during 1977/78, when Eichhornia cover was greatest, were generally lower than those of all the other periods. However, the consecutive low values prior to spraying could have occurred in any subsequent year (Figure 5.2a), and recently, between April and October 1985 chlorophyll a concentrations were low (between 15 and 20 mg/m^3 Zohary, pers. comm.). The reason for the continued low concentrations after spraying is not clear. Terbutyryn is known to influence the growth of phytoplankton(Barnsley, 1974 and Robson et al., 1974; cited in Scott et al., 1979). It may also have increased the sedimentation rate of the phytoplankton. However, since only the Eichhornia was sprayed,and most of the herbicide on the dead sprayed Eichhornia would have been inactive by the time the plants sunk, the amount of terbutryn reaching the water was small and because of the dilution effect, the concentration in the water was low. Scott et al. (1979) stated that between September 1977 and January 1978, total algal counts were generally low. The evidence available to Scott et al. (1979) suggested that the presence of Eichhornia plants, rather than the spraying, was responsible for the reduction in algal numbers. Therefore, it seems reasonable to conclude that the effect of terbutryn on phytoplankton chlorophyll a concentrations was negligible.

Scott *et al.* (1979) observed that total algal counts increased dramatically in mid-January 1978 but decreased sharply in late January 1978 because of heavy rains and floods. Chlorophyll *a* concentrations exhibit similar trends (Figure 5.1b). By early December 1977 (approximately six weeks after the first spraying) the *Eichhornia* plants sprayed and killed by the first herbicide application would have sunk (*ca* 800 ha of plants had sunk). At this time live *Eichhornia* covered slightly more than 16% of the surface area of the impoundment (sprayed area was 320 ha; Scott *et al.*, 1979). From 5 December 1977 to 16 January 1978, there was an exponential increase in the phytoplankton chlorophyll *a* concentrations of the three lake stations ($n=12$; $p < 0{,}001$). Thus, it seems that during the period of high *Eichhornia* cover, the low phytoplankton chlorophyll *a* concentrations were due to the presence of *Eichhornia*. An explanation for the exponential increase in chlorophyll *a* concentrations is that the decaying *Eichhornia* released nutrients. However, total nitrogen and phosphorus concentrations in Hartbeespoort Dam water increased only slightly from November 1977 to January 1978 (Section 3a, Table 5.4). An alternative explanation is that the increase was due to increased light penetration following the submergence of *ca* 800 ha of dead *Eichhornia* plants by early December 1977.

Before *Eichhornia* populations were large (between September 1972 and February 1975) and again after they had been reduced (1981 to 1984) the scum forming blue-green alga, *Microcystis aeruginosa*, dominated the phytoplankton (Toerien *et al*., 1976; Scott *et al*., 1980; Robarts, 1984 and NIWR, 1985). During these periods diatoms (especially *Melosira granulata*) occasionally became dominant during winter and early spring (Scott *et al*., 1980; Robarts, 1984 and NIWR, 1985). Between 1981 and 1984, *Melosira granulata* var. *angustissima* appeared for short periods and in late winter and early spring occasionally dominated the phytoplankton population for a few weeks (NIWR, 1985). However, according to Scott *et al*. (1980) and Ashton (pers. comm.) *Microcystis* populations did not appear in their usual numbers during 1976 and 1977 when *Eichhornia* cover was increasing.

Between September 1977 and January 1978 the phytoplankton population was dominated by diatom species, notably *Cyclotella meneghiniana* and *Melosira granulata* var. *angustissima*; *Microcystis aeruginosa* occurring on three occasions, though only in very low numbers (Scott *et al*., 1979). Thus, not only were chlorophyll *a* concentrations lower, but the species composition of the phytoplankton was also different when compared with periods before and after the period of extensive *Eichhornia* cover. However, in some marginal parts of the lake where *Eichhornia* plants were kept out by booms, etc. heavy concentrations of *Microcystis* were found (Ashton, pers. comm.).

c. Nutrient Loading Rates

In aquatic ecosystems, phosphorus and nitrogen are the two nutrients which most frequently limit the growth rates of the primary producers. Since the reduction in algal standing crop during the period of high *Eichhornia* cover could have been due to reduced nutrient input, nitrogen and phosphorus loading rates have been examined.

Total phosphorus and nitrogen loads are shown in Figures 5.3 to 5.6. The loading rates from 1973 to January 1975 are based on total dissolved phosphorus and nitrogen concentrations and are therefore under-estimates of the total nutrient loading rate.

Where known, the total phosphorus and nitrogen loading before and after the period of extensive *Eichhornia* cover was not noticeably different from that during extensive *Eichhornia* cover. Therefore, it can be postulated that the loads would not have changed significantly during the period of *Eichhornia* build up and the apparent decline in chlorophyll *a* concentrations. Although the lower chlorophyll *a* concentrations were probably not due to decreased nutrient loading, they may have been caused by reduced nutrient availability.

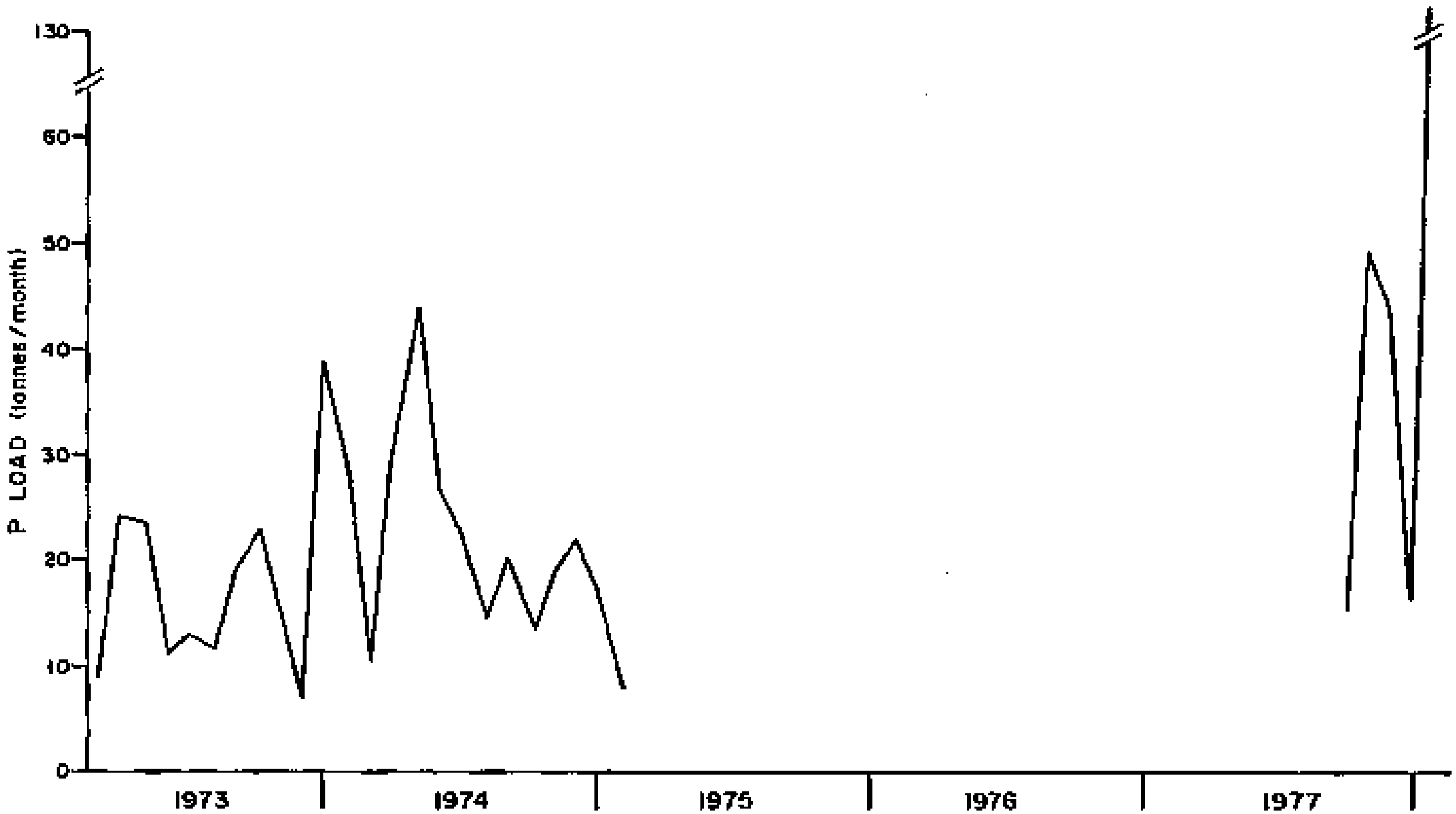

Figure 5.3: Phosphorus load (tonnes/month) to Hartbeespoort Dam. Data for 1973 to 1975 is total dissolved phosphorus (glass fibre filtered) from Toerien & Walmsley (1978). Data for 1977 to 1978 is total phosphorus from Scott et al. (1979).

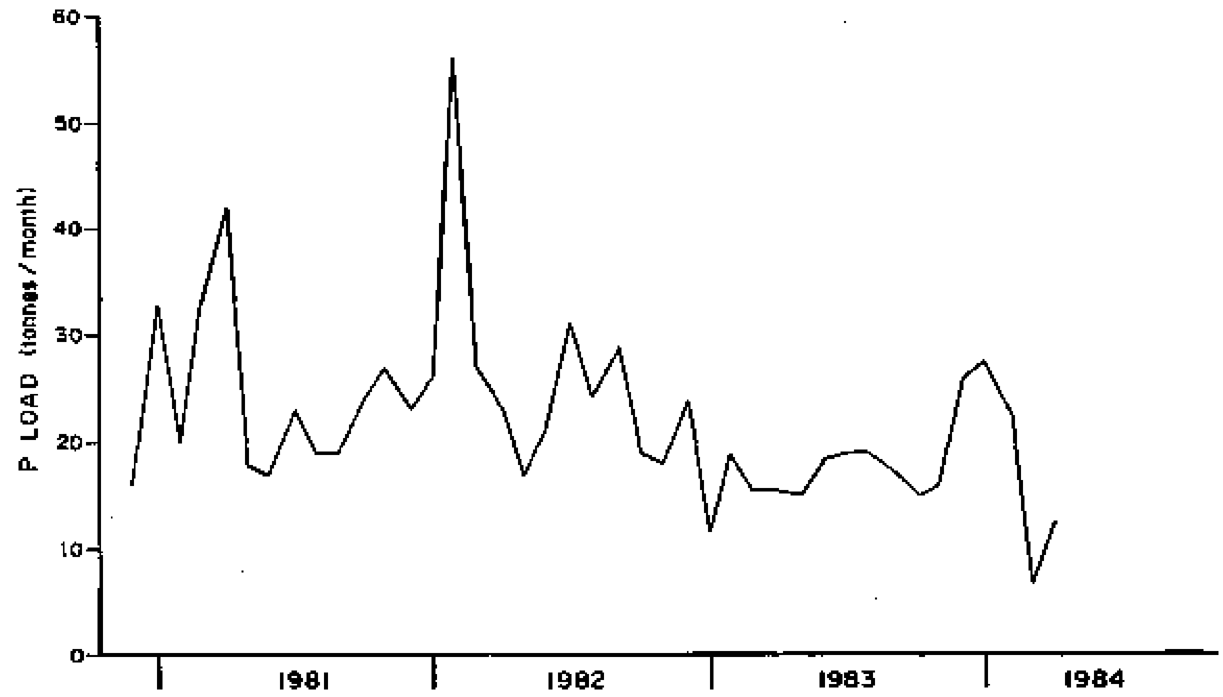

Figure 5.4: Total phosphorus load (tonnes/month) to Hartbeespoort Dam for 1980 to 1984. Data from Ashton (pers. comm.).

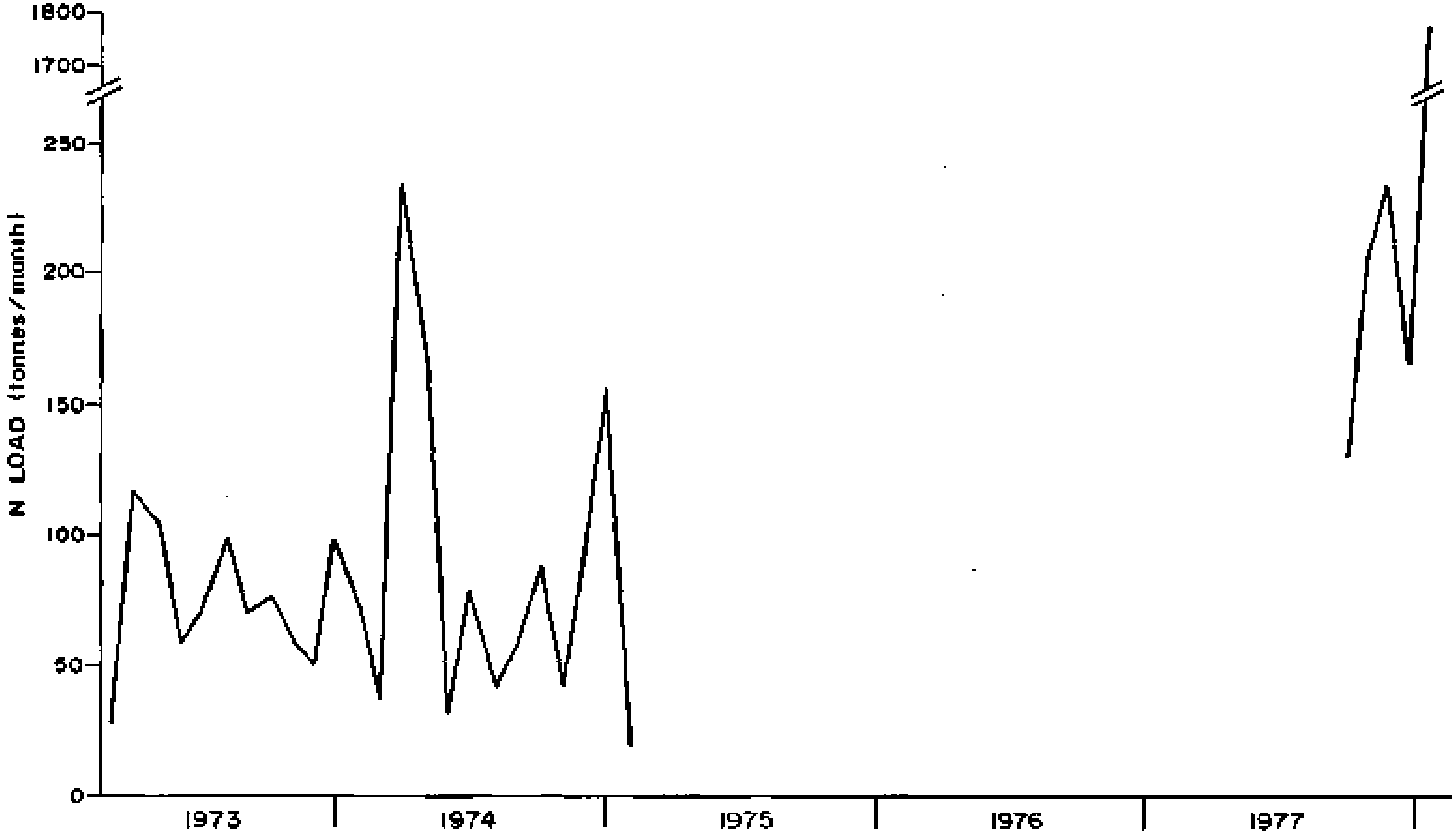

Figure 4.5: Nitrogen load (tonnes/month) to Hartbeespoort Dam. Data for 1973 to 1975 is total dissolved nitrogen (glass fibre filtered) from Toerien & Walmsley (1978). Data for 1977 to 1978 is total phosphorus from Scott <u>et</u> <u>al</u>. (1979).

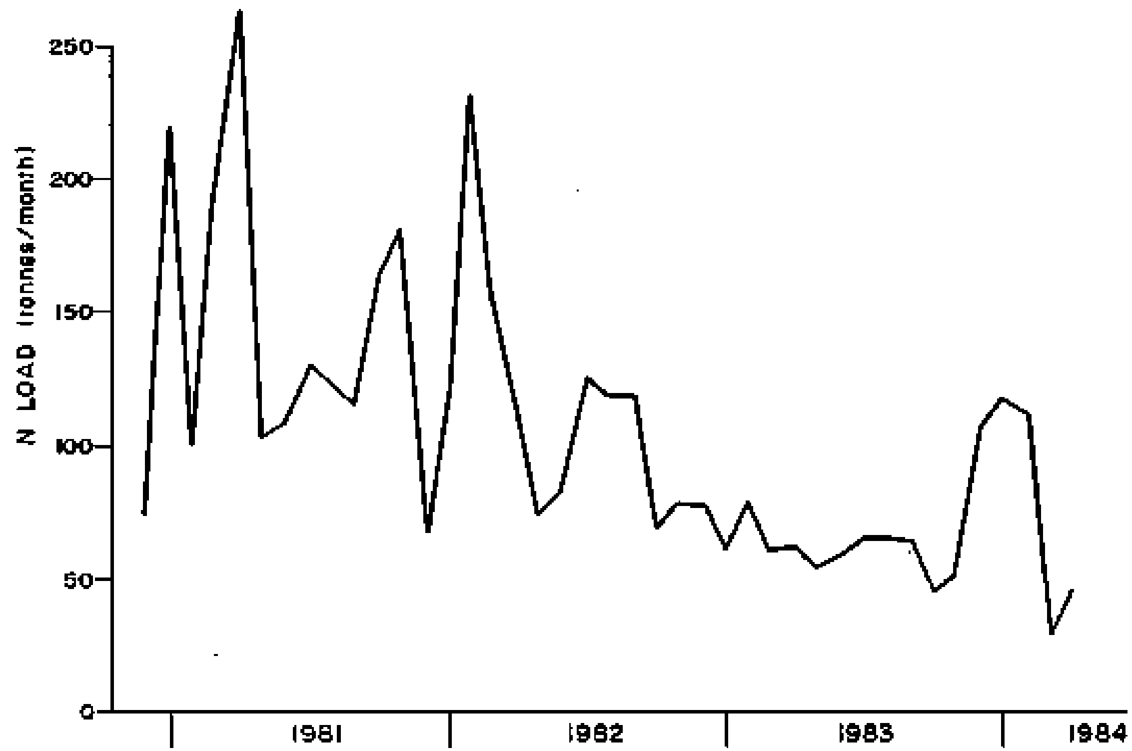

Figure 5.6: Total nitrogen load (tonnes/month) to Hartbeespoort Dam for 1980 to 1984. Data from Ashton (pers. comm.).

During 1973, 1974 and 1982 to 1984 the lake volume was more variable and generally lower than during 1975, 1976, 1977 and 1981 (Figures 5.7 and 5.8). Since the effect of the nutrient load is influenced by the volume of the receiving water body and the volume of the inflow, it is necessary to take these parameters into account when comparing nutrient loads. One way of doing this is to use the OECD (1982) equation to predict chlorophyll a concentrations (See Chapter 4; Section 3.2). This equation was used to predict chlorophyll a concentrations in Hartbeespoort Dam and the annual mean observed chlorophyll a concentrations were calculated (Table 5.2).

The predicted chlorophyll a concentrations are all higher than observed concentrations (Table 5.2), probably due in part to the difficulty in obtaining a true mean chlorophyll a concentration because *Microcystis* accumulates at the surface in patches (Ashton and Scott. pers. comm.). Also, factors other than the phosphorus load, such as light (Robarts, 1984) may have determined the algal standing crop. The predicted chlorophyll a concentration for 1977 (the period when *Eichhornia* cover was greatest) is within the range of the predicted chlorophyll a concentration for the other years. However, the observed chlorophyll a concentration for 1977 (based on data for four months) was approximately an order of magnitude lower than the chlorophyll a concentrations observed during the other years.

winter harvesting ca 100 t of fresh hyacinth daily would be sufficient, but during summer two to three times as much fresh plant material (ca 161 to 288 t) would have to be harvested. However, following the introduction of hyacinth control in September 1981, the population has often been maintained at less than 20 ha so it would not be large enough to remove the nutrients brought in by the river. In addition, growth virtually ceases in winter and harvesting is stopped. Thus during these months the hyacinth cannot significantly decrease nutrient availability in the impoundment.

Thus it seems that since the introduction of hyacinth control the size of the population and the amount of hyacinth harvested, although not measured, would be insufficient to limit algal growth rates through nutrient removal, especially during period of elevated inflows and nutrient loads. However, during the period of hyacinth build-up prior to the introduction of hyacinth control, the considerably larger populations could have been effective in reducing nutrient concentrations to levels limiting to algal growth rates.

Nutrient concentrations

Total dissolved nutrient concentrations reflect the balance between nutrient inputs, losses and uptake. The total dissolved nitrogen and phosphorus concentrations in the surface water of Vernon Hooper Dam between 1975 and 1985 are shown in Figures 4.10 c and d. The TDN concentration, but not the TDP concentration, appears as if it could have been influenced by hyacinth cover. The decrease in TDN concentration between 1976 and 1980 was associated with the build-up of hyacinth cover and the subsequent increase in TDN concentrations with the reduction in hyacinth cover to between 8 and 20 ha. However, the trend in nitrogen concentrations is also reflected in the nitrogen loading data (Figure 4.11). The reduction may therefore have little to do with the standing crop of *Eichhornia*, being determined more by the external loading rate. Also, since *ca* September 1979 there have been periods of low nitrate concentrations in the surface water (Figure 4.10e) during which nitrogen fixation could have provided a source of nitrogen. The TDN:TDP ratios (Figure 4.10f), nutrient concentrations and the half saturation constants of a number of algae and *Eichhornia* (Table 4.6) indicate that in Vernon Hooper Dam nitrogen is most likely the growth rate limiting nutrient. However, the extent to which the reduction in surface nitrogen concentrations between 1975 and 1979 was due to reduced nitrogen loading or uptake by *Eichhornia* cannot be determined because *Eichhornia* growth and harvesting was not adequately monitored. Hyacinth harvesting was insufficient to reduce nutrient concentrations to levels which maintain chlorophyll *a* concentrations within acceptable limits.

Light and temperature

Light is essential for algal growth and temperature affects the rate of reaction and thus the growth of algae. *Eichhornia*

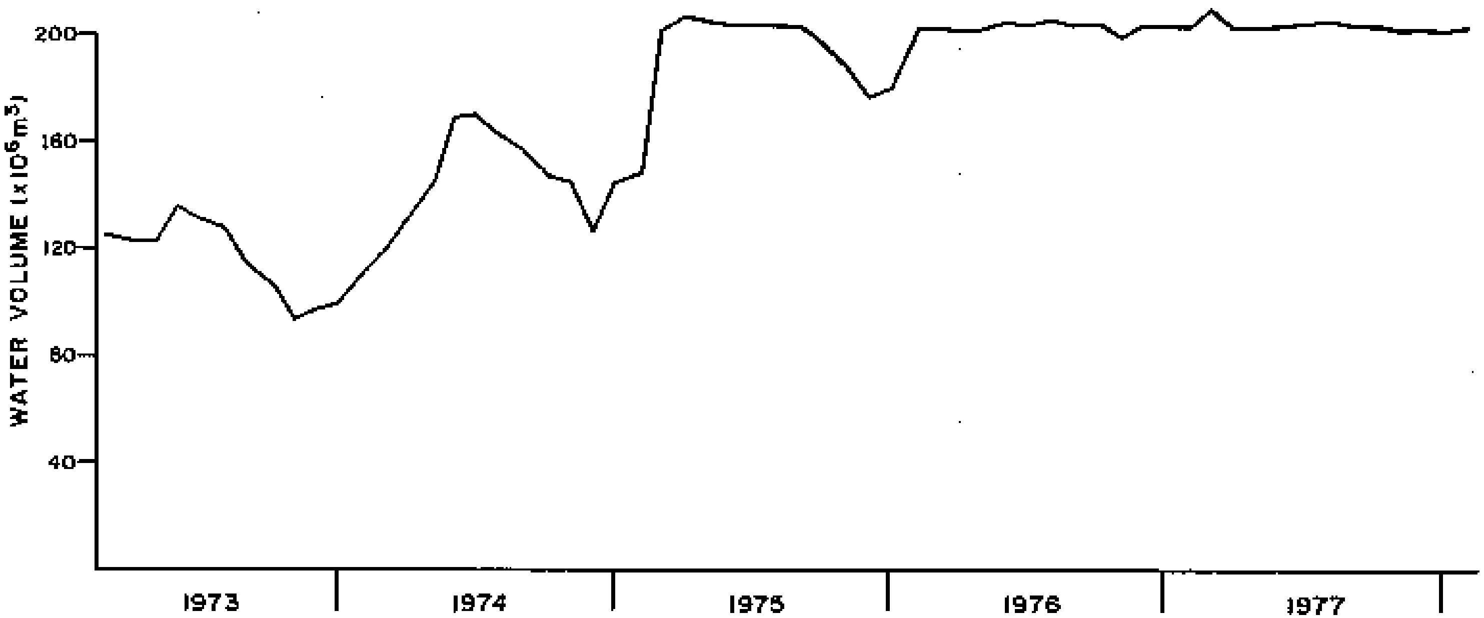

Figure 5.7: Water storage (x $10^6 m^3$) in Hartbeespoort Dam from 1973 to 1978. Data from Department of Environment Affairs records.

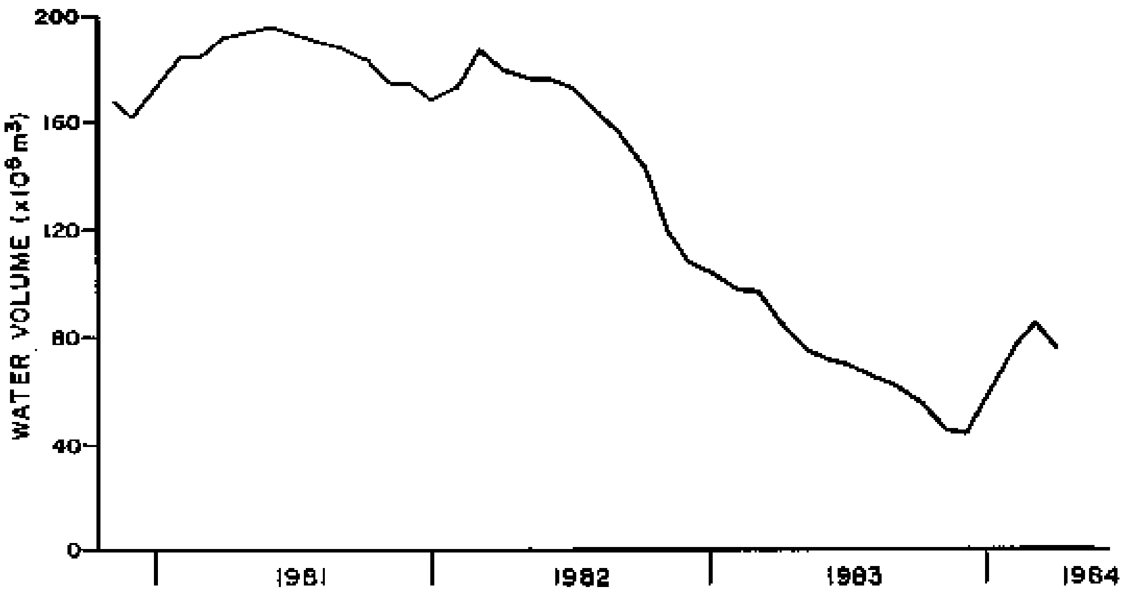

Figure 5.8: Water storage (x $10^6 m^3$) in Hartbeespoort Dam from 1980 to 1984. Data from Department of Environment Affairs records.

Table 5.2: Observed predicted annual mean chlorophyll a concentrations in Hartbeespoort Dam.

Year (Jan-Dec)	Mean annual chlorophyll a concentration (mg/m^3). Observed	Predicted
1973*	28,9	89,2**
1974	34,8	75,8**
1977***	4,7	75,9
1981	39,0	62,0
1982	46,8	92,0
1983	69,2	94,1

* based on data from February to December 1973

** based on total dissolved phosphorus loads

*** based on data for four months at the height of the *Eichhornia* infestation (viz. September to December, 1977)

This substantiates the earlier conclusion that the lower chlorophyll a concentrations that occurred when Eichhornia cover was greatest, were not due to reduced nutrient loading rates. They could, however, have been due to nutrient limitation of phytoplankton growth rates as a result of nutrient uptake by Eichhornia. This possibility is examined in Section 3a.

3. EFFECTS OF EICHHORNIA COVER ON THE PHYTOPLANKTON STANDING CROP AND SPECIES COMPOSITION OF HARTBEESPOORT DAM.

Eichhornia cover influences a number of conditions in a waterbody that determinine both the biomass and species composition of the phytoplankton population. Some of these conditions and their effects are shown in Figure 5.9.

a. Nutrient Concentrations

The likelihood of nutrient uptake by Eichhornia resulting in concentrations that would limit phytoplankton growth rates, would have been greatest when Eichhornia production was greatest. This, presumably, would have been just before the herbicide spraying programme was initiated. Accordingly, nutrient removal by Eichhornia in the month prior to the first spraying, was estimated.

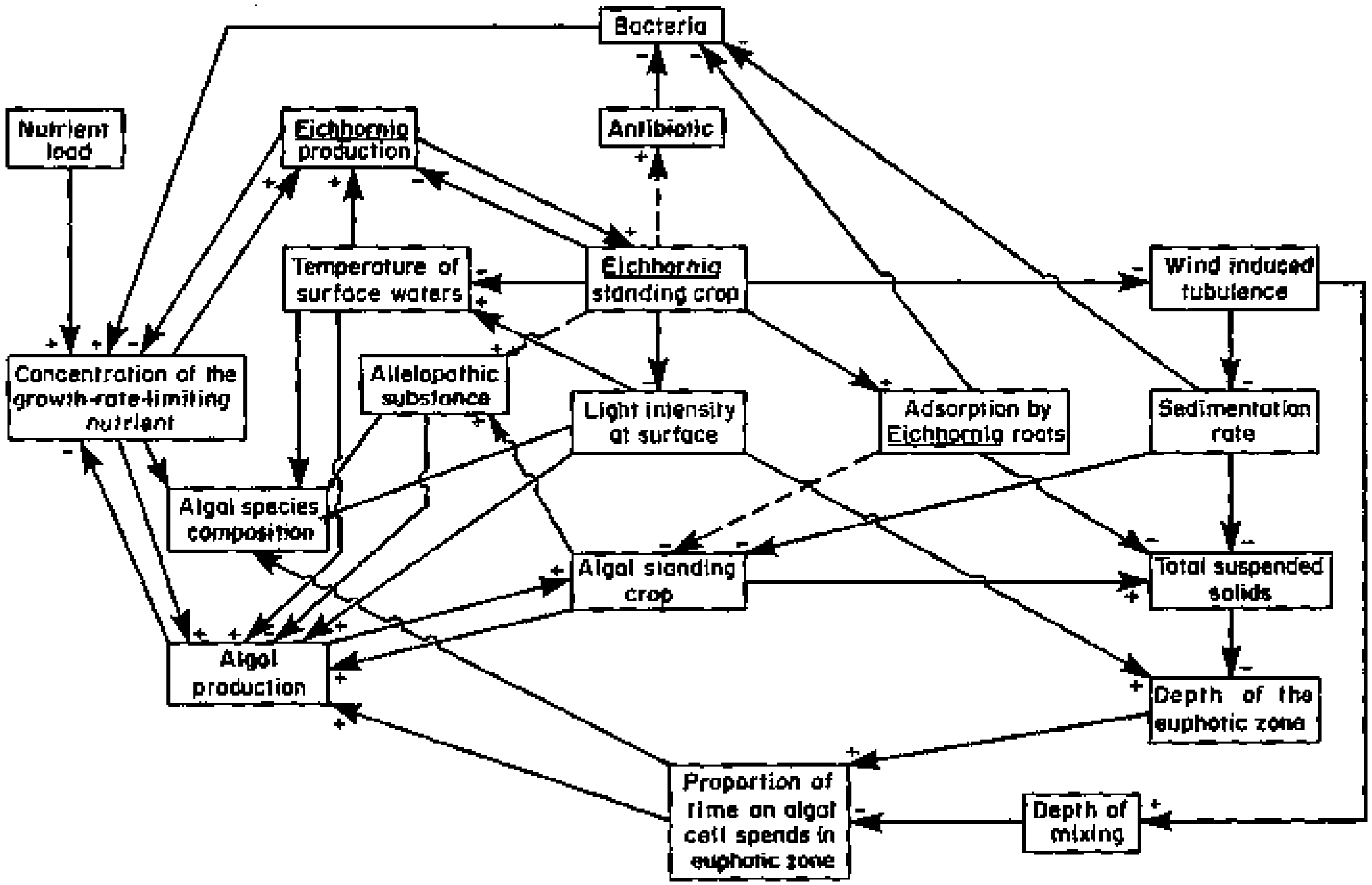

Figure 5.9: Conceptual diagram showing possilbe (---) and known (——) effects of Eichhornia cover on some conditions in a waterbody.

Scott *et al.* (1979) and Musil (1982) measured the growth rate of marginal plants grown in enclosures in Hartbeespoort Dam during October 1977. Based on the mean growth rate, the standing crop prior to spraying (estimated by Scott *et al.*, 1977), and the nutrient content of plants, the amounts of phosphorus and nitrogen taken up by *Eichhornia* between 18 September and 18 October 1977 (date of first spraying) were estimated (Table 5.3). Although the use of these data overestimates the amounts of nutrients removed, they give some measure of the nutrient removal potential of *Eichhornia*. The total phosphorus and nitrogen inputs and contents of the impoundment are also shown in Table 5.3.

During October 1977 the phosphorus content of *Eichhornia* was three to four times that of the impoundment; whereas, the nitrogen content of the impoundment was greater than that of the *Eichhornia*. Since *Eichhornia* production was estimated (by back extrapolation from the estimated standing crop and growth rate) for the period 18 September to 18 October 1977, the average river loading for September and October 1977 would best reflect the nutrient inputs for the period during which production was estimated. The average input of phosphorus was approximately the same as that estimated to be taken up by *Eichhornia*. The uptake of nitrogen by *Eichhornia* was greater than the nitrogen input; however, the phosphorus and nitrogen content of the impoundment did not change much from September to October. The lack of change in nitrogen content indicates

Table 5.3: Fresh mass and nutrient content of *Eichhornia* and total phosphorus and nitrogen inputs and contents of Hartbeespoort Dam during September and October 1977.

	Eichhornia^{xxx}			River Inputs^{++}			Impoundment Water^{++}	
	18 Sept.	18 Oct.	Uptake	Sept.	Oct.	Ave.	Sept.	Oct. (before spraying)
Fresh mass (t)^{xx}	143 000	300 000	157 000	-	-	-	-	-
Phosphorus (t)	28^{x}- 34^{+}	59^{x} -71^{+}	32^{x} -37^{+}	24	49	37	17	21
Nitrogen (t)	198^{+}-222^{x}	419^{+}-468^{x}	219^{+}-245^{x}	207	208	207	583	610

x based on data from Musil (1982) for marginal plants

\+ based on data from Musil (1982) for central plants

++ based on data from Scott *et al*. (1979)

xx weighted averages based on data for 12 - 30 September 1977.

xxx based on the estimate of standing crop by Scott *et al*. (1979) on 18 October 1977 and back extrapolation from growth rates.

that *Eichhornia* growth and nutrient uptake was overestimated and/or there are other sources of nitrogen. Thus, the total inflow of phosphorus would not have been taken up by *Eichhornia*, sedimentation probably accounting for the lack of change in the phosphorus content of the impoundment. During September and October the uptake of nutrients by *Eichhornia* was not sufficient to alter the total phosphorus and nitrogen content of the impoundment.

Phosphorus and nitrogen concentrations in the lake water (Table 5.4) appear not to have decreased from September to October, also indicating that nutrient removal from the water was less than nutrient inputs for this period. The possibility that nutrient concentrations were sufficiently low to limit the growth rate of the phytoplankton has been examined.

Half saturation coefficients (the concentration of the growth rate limiting nutrient that will result in the organism growing at half its maximum growth rate) can be used to predict whether the growth rate of an organism will be limited by a particular nutrient concentration if no other variables limit the growth of that organism. Toerien (1974) recorded the half saturation coefficients of a number of algae. These, and the coefficients for *Eichhornia* are given in Table 4.6.

Table 5.4: Mean monthly total phosphorus and nitrogen concentrations in Hartbeespoort Dam from September 1977 to January 1978. Data from Scott et al. (1979).

	Concentrations in lake water (mg/m^3)	
Month	Total phosphorus	Total nitrogen
September	90	3 010
October: before spraying*	110	3 000
after spraying**	140	3 370
November	190	3 970
December	220	2 400
January	230	3 870

* 17 October 1977 value

** Average of values determined on 24 and 31 October 1977

The half saturation coefficients indicate that during September, phosphorus concentrations had the potential to limit the growth rates of Eichhornia and the high temperature strain of Chlorella pyrenoidosa. Nitrogen concentrations in September had the potential to limit the growth rate of both strains of Chlorella pyrenoidosa. For the other algae for which half saturation coefficients are given, neither nitrogen nor phosphorus concentrations would have been growth rate limiting.

The half saturation coefficients indicate that during September, phosphorus concentrations had the potential to limit the growth rates of Eichhornia and the high temperature strain of Chlorella pyrenoidosa. Nitrogen concentrations in September had the potential to limit the growth rate of both strains of Chlorella pyrenoidosa. For the other algae for which half saturation coefficients are given, neither nitrogen or phosphorus concentrations would have been growth rate limiting. Hence, it seems possible that between September 1977 and January 1978, the growth rates of phytoplankton in Hartbeespoort Dam were not limited by nutrient concentrations.

b. Light and Temperature

Mats of Eichhornia would reduce wind-induced mixing of the surface waters (Scott et al., 1979); thereby increasing sedimentation rates, decreasing the suspended solids content of the water and decreasing the depth of mixing. The suspended solids content of the water would be further reduced by the adsorption of particulate material and organic debris around Eichhornia roots (Scott et al., 1979 and Gopal et al., 1984). In Hartbeespoort Dam during the period of high Eichhornia cover, the light extinction coefficients were approximately half the values recorded by Scott et al. (1977). Light extinction coefficients during 1981 to 1984 (NIWR, 1985) were also higher than those recorded during high Eichhornia cover. During high Eichhornia cover, the low attenuation of light in temperatures occurred in Hartbeespoort Dam, leading Scott et al. (1979) to state that the effect of Eichhornia on water temperature was possibly the cause of the more gradual development of thermal stratification in 1977 compared with 1973. The decrease in water temperatures was probably not sufficient to affect algal productivity.

c. Other Effects

i. Bacteria

Between September 1977 and January 1978, bacterial counts in Hartbeespoort Dam were generally low, with occasional fluctuations (Scott et al., 1979 and Figure 5.11). Bacterial counts at the river inflows without Eichhornia were significantly higher (p=0,01) than those from the main lake basin. This, according to Scott et al. (1979), indicated that bacterial growth in the lake itself was inhibited by the presence of Eichhornia. Part of the reduction in bacterial numbers could have been due to the concentration of these organisms in the rhizosphere of Eichhornia plants and the increased rates of sedimentation. During the period of high Eichhornia cover, nutrient concentrations were not reduced appreciably, thus it seems that the effect of reduced bacterial numbers on the phytoplankton was probably negligible.

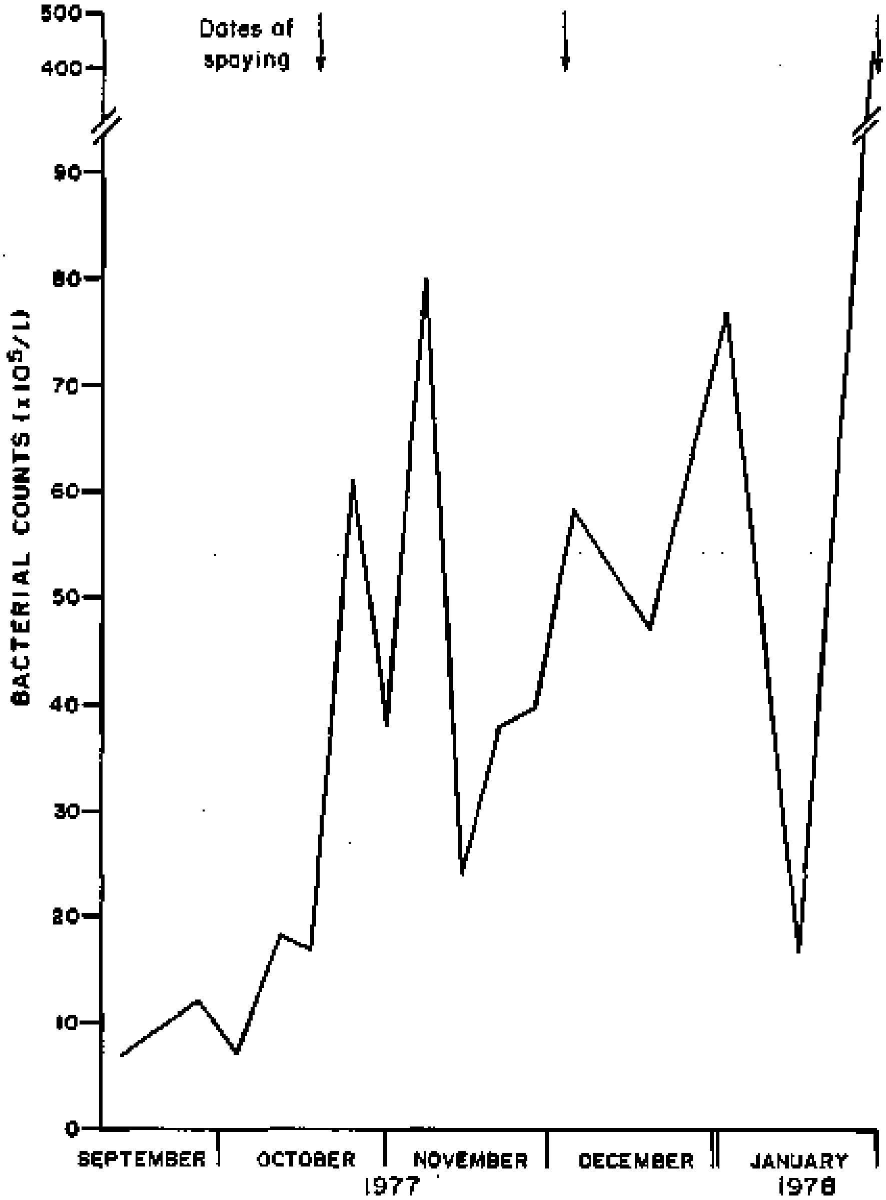

Figure 5.11: Bacterial counts (x 10^5/l) in Hartbeespoort Dam (mean of three lake stations) between September, 1977 and January, 1978. Data from Scott *et al.*, (1979).

clear areas adjacent to plant cover, associated with the decreased suspended solids content of the water, would increase the amount of light available for algal production (Figure 5.10). In addition, the reduced wind-induced mixing in areas covered by Eichhornia would decrease the depth of mixing and thus increase the proportion of time an algal cell would spend in favourable light conditions. This would also tend to increase algal production; however, the increased sedimentation rates would tend to decrease the algal standing crop.

However, mats of Eichhornia provide a physical barrier against the penetration of light (Scott et al., 1979 and Gopal et al., 1984). At the density recorded on Hartbeespoort Dam (25 kg fresh mass/m^2 ; Scott et al., 1979) the Eichhornia, where present, completely prevented the penetration of light. The presence of this physical barrier to light penetration would negate any improvement in underwater light climate caused by the effect of Eichhornia on wind induced mixing. However, Eichhornia covered only ca 50% of the lake surface a any one time and mats of Eichhornia were blown back and forth daily, resulting in continually changing light conditions and preventing any acclimatisation of the algae to the light conditions in the impoundment.

Under large mats of Eichhornia, temperatures are lower (Ultsch, 1973 and Gopal et al., 1984) as a result of reduced light penetration (Scott et al., 1979 & 1984). Lower water

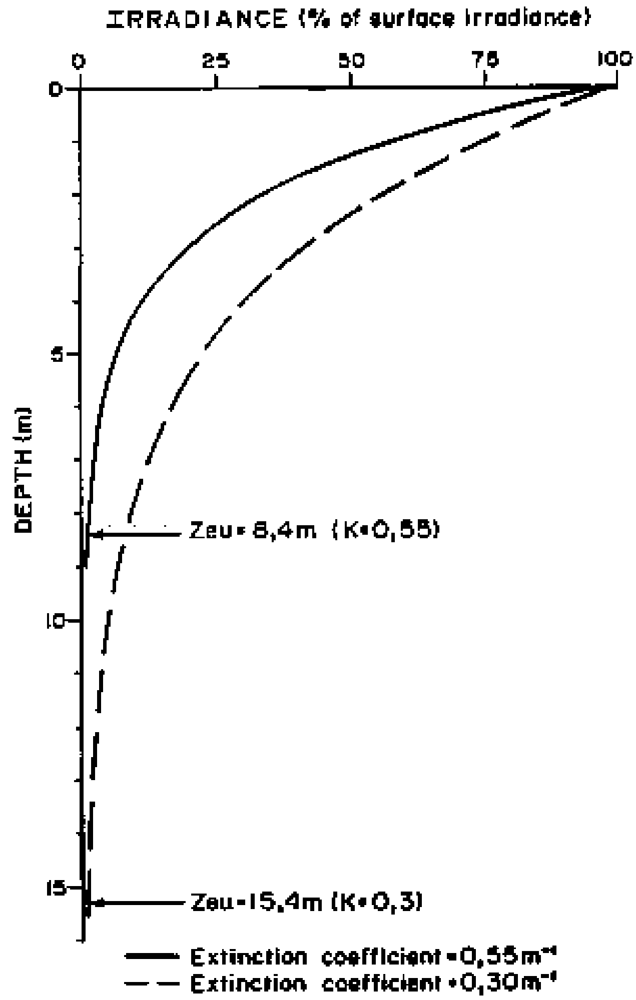

Figure 5.10: Effect of Eichhornia cover on the underwater light climate of Hartbeespoort Dam. The extinction coefficient (K) of 0,55 m^{-1} is the median of 50 values given by Scott et al. (1977) for Hartbeespoort Dam between May 1974 and March, 1975. The extinction coefficient of 0,3 m^{-1} is based on data from Scott et al. (1979).

ii. Physical conditions

In addition to the above effects, *Eichhornia* also lowers oxygen concentrations, pH and alkalinity and increases dissolved carbon dioxide concentrations (Ultsch, 1973; McVea & Boyd, 1975 and Gopal *et al*., 1984).

In Hartbeespoort Dam during high *Eichhornia* cover, oxygen concentrations were generally low, rarel reaching 100% saturation. This contrasted markedly with the high values recorded for the corresponding period in 1973/74 (Scott *et al*., 1979). Scott *et al*. (1979) stated that the low oxygen values in 1977 were due, in part, to the very low numbers of algae and partly to the consumption of oxygen by *Eichhornia* roots and decaying organic debris. The variation in the alkalinity results of Scott *et al*. (1979) does not appear to be due to *Eichhornia* cover. During high *Eichhornia* cover, the pH values were more than one pH unit lower than those found during 1973/74 (Scott *et al*., 1979). The increased sedimentation rates would have favoured species which sediment out slowly such as *Cyclotella meneghiniana* var. *angustissima* and *Melosira granulata* (Ashton, pers. comm.). *Eichhornia* resulted in changes in several physical and chemical conditions all of which would have contributed to the change in the phytoplankton species composition from a population dominated by cyanobacteria (*Microcystis aerunginosa*) to one dominated by diatoms (*Cyclotella meneghiniana* var. *angustissima* and *Melosira granulata*).

iii. Allelopathy

The reduction in phytoplankton standing crop could have been due to the release of algal inhibitors by Eichhornia (allelopathy). The data do not permit the separation of this effect from that of other changes due to the presence of Eichhornia. However, the effect of Eichhornia on factors other than allelopathy was sufficient to account for the observed reduction in the phytoplankton standing crop, thus it seems unlikely that allelopathy played a role.

4. CONCLUSIONS

In Hartbeespoort Dam the period of high *Eichhornia* cover was associated with lower chlorophyll *a* concentrations and an altered phytoplankton species composition compared with periods before and after. These changes were apparently not due to altered nutrient loading rates and nutrient limitation. The change in the phytoplankton populations was probably due to the effect of *Eichhornia* on other physical and chemical conditions in the impoundment. Following the submergence of large areas of dead *Eichhornia* by early December 1977 the total nitrogen and phosphorus concentrations increased only slightly, but there was an exponential increase in chlorophyll *a* concentrations. This indicates that the major effect of *Eichhornia* cover on the phytoplankton population was due to its effect on the underwater light climate as a result of it being a physical barrier to light penetration.

6. PROSPECTS OF USING EICHHORNIA TO IMPROVE WATER QUALITY

In aquatic ecosystems the growth rate limiting nutrient is usually either nitrogen or phosphorus and an increase in the rate of supply of the limiting nutrient will often cause eutrophication. Eutrophication results in a number of water quality problems, most of which result from the occurrence of undesirable quantities of phytoplankton in the water (Jones & Lee, 1982 and Toerien, 1977). Limiting water fertility, preferably by controlling phosphate, is generally regarded to be the most desirable eutrophication control strategy (Toerien, 1977). Nutrients enter impoundments from both point (discrete) and non-point (diffuse) sources. Although point source nutrient inputs can be reduced by advanced chemical treatment systems; costs tend to be prohibitive (Kadlec & Tilton, 1979) and in some catchments the contribution from non-point sources has been so high that removal of phosphate from point sources would have negligible effects on the trophic status, and hence water quality, of impoundments (Pretorius, 1983). Thus, the in-lake use of aquatic macrophytes to remove nutrients and improve water quality has attracted considerable attention. The warm climate of South Africa is well suited to the use of the free floating aquatic macrophyte, *Eichhornia crassipes.*

Eichhornia apparently does not inhibit phytoplankton growth by allelopathy, however, it does affect a number of factors which influence water quality. Not only does *Eichhornia* remove nutrients, it also acts as a physical barrier, almost completely preventing light penetration and reducing wind induced mixing. Thus *Eichhornia* reduces the rate of algal production, increases algal losses by increasing the rate of sedimentation (Figure 5.9) and reduces the rate of gaseous exchange between the atmosphere and the water.

If the *Eichhornia* was permitted to increase to completely cover the impoundment, the recreational value of the impoundment would decrease as full *Eichhornia* cover precludes boating and fishing and decreases the aesthetic appeal of the impoundment. However, algal-related water problems would decrease sharply because of shading and nutrient removal. However, full *Eichhornia* cover results in the development of reducing conditions, resulting in the occurrence of "manganese" water. "Manganese" water is more expensive to treat than "algal laden" water. In addition to the high cost of treating "manganese" water, the cost of controlling the *Eichhornia* and the loss in recreational value of the lake must be considered, indicating that the use of full *Eichhornia* cover to improve water quality does not appear to be an economically viable prospect.

One could, however, use a partial cover of *Eichhornia* to improve water quality. This has the advantages of permitting boating and fishing on the uncovered area of the impoundment and the population can be maintained at a size which will not adversely affect oxygen exchange and thus water quality. In order to prevent the uncontrolled increase in the *Eichhornia* population during favourable conditions, it is necessary to confine the *Eichhornia* to one section of the impoundment.

In Vernon Hooper Dam the *Eichhornia* is confined behind a floating boom and at full supply level it occupies up to 25% of the surface area of the impoundment. This population does not adversely affect oxygen exchange in Vernon Hooper Dam. Although the *Eichhornia* in Vernon Hooper Dam reduces the algal standing crop and production by nutrient removal and shading, the reduction is not sufficient to maintain water quality problems within acceptable limits. This is probably because Vernon Hooper Dam is already severely eutrophic, the nutrient content of the impoundment is high so the proportion of the total nutrient content of the impoundment removed by *Eichhornia* growth and harvesting is small.

In Vernon Hooper Dam a larger population of *Eichhornia* than the present 20 ha would be required to reduce algal related water quality problems to within acceptable levels. How quickly an improvement in water quality would occur cannot be assessed, but it is unlikely to be rapid as once a lake has become eutrophic, it can remain so for a very long time, even if the source of external nutrients is cut off (Hynes, 1971). The effectiveness of partial Eichhornia cover in improving water quality is likely to be greater in lakes that are not as eutrophic as Vernon Hooper Dam.

The nutrient concentrations in effluent from point sources is usually reduced before reaching standing waterbodies because of both dilution and the natural self purifying property of rivers. Thus nutrient concentrations are lower at the input to waterbodies than at the point sources upstream of the waterbodies. Since the growth rate of Eichhornia is reduced by low nutrient concentrations, higher growth rates and hence greater nutrient removal would be achieved at point sources.

Thus it seems that the in-lake use of Eichhornia for water quality improvement in Vernon Hooper Dam is not efficient and is unlikely to result in the required reduction in nutrient availability in other systems as eutrophic as Vernon Hooper Dam. An alternative approach, nutrient removal from point sources involves growing Eichhornia in artificial ponds at the point sources.

This approach has several advantages:

- nutrient concentrations are generally high;
- environmental parameters (such as temperature, humidity and flow through the root system of Eichhornia) are more easily manipulated to increase the growth rates of Eichornia;
- the possibility of spreading Eichhornia to lower reaches of the catchment is reduced; and
- because the pollution load to the river will be reduced, the quality of water entering the lake will be improved considerably.

However, a disadvantage of this approach is that there are usually several point sources within one catchment and diffuse source inputs may be large. The effect of the point source use of Eichhornia in the Vernon Hooper Dam catchment cannot be accurately assessed, however, as with the in-lake use of Eichhornia, the effect is likely to be small at least initially, because of the severely eutrophic state of Vernon Hooper Dam.

Eichhornia could be most efficiently used for nutrient removal and water quality improvement if grown in artificial ponds at point sources in catchments where the nutrient input from diffuse sources is small relative to the point source input.

The in-lake use of Eichhornia to control or improve lake water quality is likely to be most effective where the receiving waterbody is of relatively low trophic status.

It must be noted that *Eichhornia crassipes* is a proclaimed noxious weed throughout South Africa. *Eichhornia* has the potential for very rapid growth, particularly in eutrophic conditions, and may cover vast areas of standing and slowly-running water. *Eichhornia* prevents boating and fishing, disrupts water flow in irrigation channels and blocks sluices. It also creates ideal conditions for malaria mosquitoes and bilharzia snails to breed and may increase the loss of water from the surface of a lake by its transpiration (Davies & Day, 1986). Because of these problems extreme caution is necessary if *Eichhornia* is to be used for water quality improvement. It is recommended that use of *Eichhornia* be restricted to catchments in which *Eichhornia* already occurs, thus reducing the risk of spreading it to catchments presently free of *Eichhornia*.

7. RESEARCH NEEDS

It was the stated intention of this preliminary study to assess from available information, the evidence that *Eichhornia* was implicated in improving the quality of water stored in the Vernon Hooper and Hartbeespoort dams. Unfortunately, discontinuous monitoring and the use of widely divergent techniques complicated interpretation to the extent that inferences could not be made with confidence.

Although the evidence was supportive of the view that *Eichhornia* played an important role in the regulation of water quality, the mechanisms involved could not be established but they are likely to be different in different waterbodies. The case studies were of severely eutrophic impoundments in which the super-abundance of nutrients probably reduced the impact of *Eichhornia* on nutrient availability. It is therefore suggested that the potential of using *Eichhornia* to improve water quality under mesotrophic conditions warrants investigation.

The successful use of *Eichhornia* in shallow ponds for management of water quality may indicate a potential for use in small, shallow, mesotrophic impoundments.

Data from the preliminary experiments on the allelopathic potential of Eichhornia must be interpreted with caution. The fact that convincing evidence of allelopathy with the test organism was not found, should not be taken to imply that allelopathy does not occur. Inhibitors may be species specific, in which case they probably would escape detection in a bioassay yet play a significant role in determining phytoplankton succession in the waterbody. A broader based approach is indicated.

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APPENDIX 1

Abbreviations used in Figures 3.2 to 3.9 and 3.11:

+ 160	=	filtered sample water + 8 ml stock PAAP to give a concentration of 160%, complete PAAP being referred to as 100% + algal inoculum.
+ 100	=	filtered sample water + 5 ml stock PAAP to give the concentration of complete PAAP + algal inoculum.
+ 60	=	filtered sample water + 3 ml stock PAAP to give a concentration of 60% of complete PAAP + algal inoculum.
+ 40	=	filtered sample water + 2 ml stock PAAP to give a concentration of 40% of complete PAAP + algal inoculum.
FSW	=	filtered sample water + algal inoculum.
-N	=	filtered sample water + 5 ml stock PAAP-N + algal inoculum
-P	=	filtered sample water + 5 ml stock PAAP-P + algal inoculum.
ASW	=	autoclaved filtered sample water + algal inoculum.
A+100	=	autoclaved filtered sample water + 5 ml stock PAAP + algal inoculum.
A-N	=	autoclaved filtered sample water + 5 ml stock PAAP-N + algal inoculum.

A-P = autoclaved filtered sample water + 5 ml stock PAAP-P + algal inoculum.

Control = distilled water + 5 ml stock PAAP + algal inoculum.