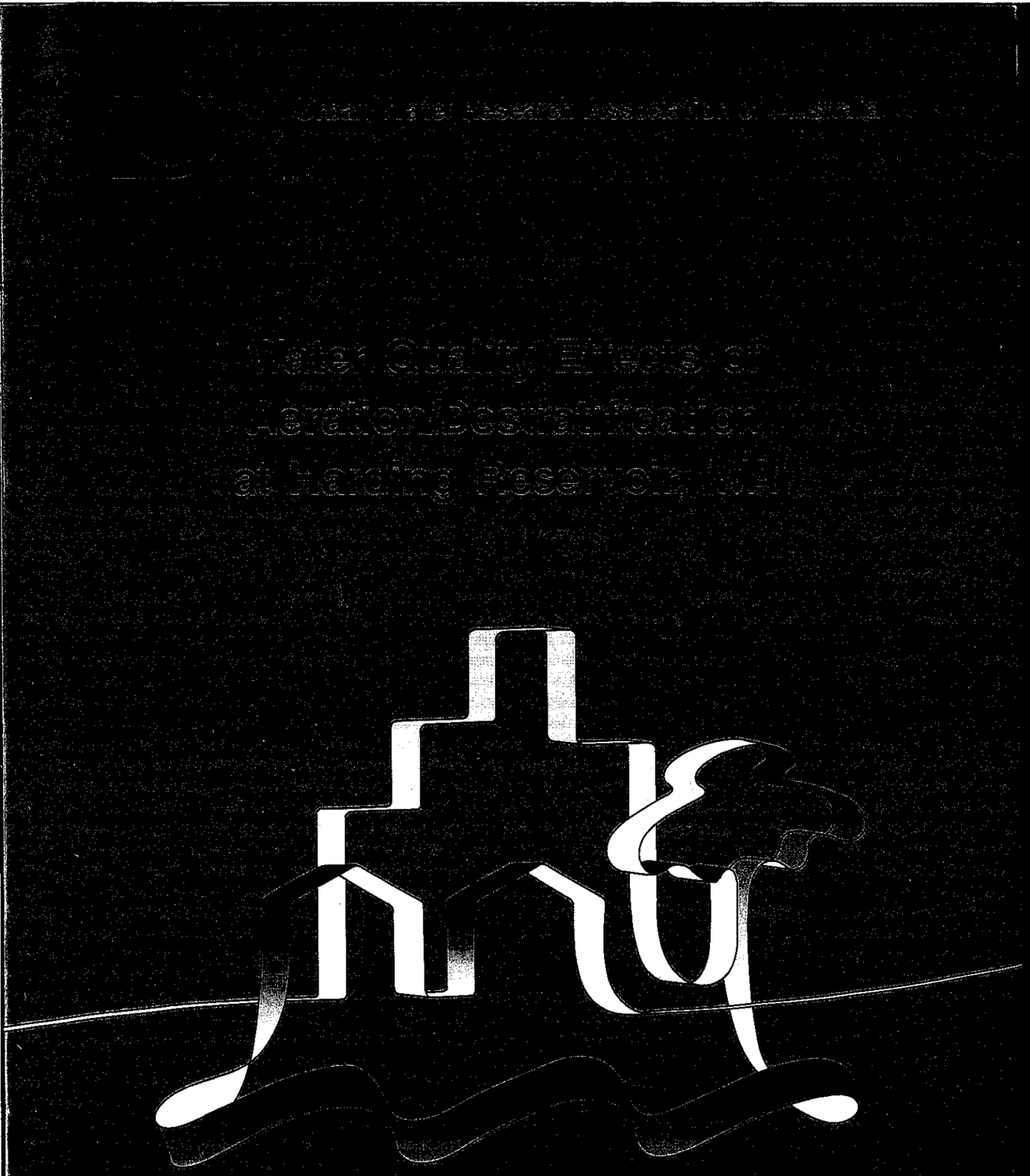
Water Quality Effects of Aeration/Destratification at Harding Reservoir, Utah



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Water Quality Effects of Aeration/Destratification at Harding Reservoir, WA

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FOREWORD

This report is based on UWRAA Research Projects No WS-2 and No WS-15: 'Evaluation of Aeration / Destratification of Reservoirs, Stages 1 and 2 (respectively)' and is the third and final report on these projects.

The first report, 'Review of Artificial Destratification of Water Storages in Australia' was prepared by the Water Authority of Western Australia (WAWA) and published in December 1989 as UWRAA Research Report No 9. The second report, 'Modelling and Design of Reservoir Aeration Destratification Systems' was prepared by the Centre for Water Research, University of Western Australia and published in April 1991 as UWRAA Research Report No 23.

Organisational responsibility for the portion of the projects reported herein was as follows:

Sponsoring Authority	:	Water Authority of Western Australia
Project Officers	:	Dr B Kavanagh Dr R S Rosich Water Authority of Western Australia
Research Agency	:	Water Authority of Western Australia
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This portion of the projects was funded by the Urban Water Research Association of Australia and the Water Authority of Western Australia.

SYNOPSIS

Harding Reservoir, located in the tropical region of Western Australia, exhibits natural stratification characteristics which are typical for water bodies in both tropical and warm temperate zones. Such stratification in Harding Reservoir has been associated with water quality complaints by consumers receiving supplies from the reservoir. Turbidity problems have also occurred from time to time due to various factors. To overcome these problems alternative groundwater sources are used during periods of high turbidity (and/or insufficient surface supply) together with artificial destratification by aeration of the reservoir during periods of reservoir use. Since introducing aeration/destratification techniques there have been no major outbreaks of taste and odour complaints from consumers of water supplied from Harding Reservoir.

A study was undertaken during the period January 1988 - June 1990 of the effects of artificial aeration/destratification on the quality of water in the reservoir and on the processes that control that quality. More specifically, the study aimed to clarify the biological, chemical and physical water quality effects of aeration/destratification and when to apply the technique; to increase the understanding of processes that determine water quality in water bodies; and to reduce the costs of aeration/destratification through improved design techniques and greater understanding of the processes involved. The findings of the study are described in the report.

ACKNOWLEDGMENTS

This study would not have been possible without the assistance and support of many people. These include the Manager (initially Jim Brown and latterly Graham Cargeeg) and the Regional Operations Engineer (Mike Tarca) from the North West Region and the Manager (Brian Kavanagh) from the Scientific Services Branch of the Water Authority of Western Australia.

Ross Sheridan, Phil Roberts, Barry Coles, Dorothy Colman and Tom Swan from the North West Region gave extensive support with advice, field work and the provision of data. The authors are indebted to many staff from the Scientific Services Branch including Kim Partridge for plankton identification and counting and Kevin Xanthis and the staff of the Analytical Section for other laboratory work. Brian Kavanagh, Mike Tarca, Phil Roberts and Dorothy Colman reviewed the manuscript.

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EXECUTIVE SUMMARY

1. INTRODUCTION

This report presents the results of Stage 2 of an aeration/destratification study of Harding Reservoir, Western Australia, funded by the Urban Water Research Association (UWRAA) and the Water Authority of WA. Stage 1 of the study was a joint project between the Scientific Services Branch of the Water Authority of WA and the Centre for Water Research, University of WA, and comprised:

- a comprehensive review of the status of reservoir destratification in Australia, evaluating fifty case studies in the technical literature (McAuliffe and Rosich, 1989,1990) and
- the development of a numerical model of bubble plumes, including preliminary field trials, to aid in the design of efficient aeration devices (Lewis *et al*, 1991).

Artificial destratification is the process of mixing the water column by aeration or mechanical means to disrupt or prevent the onset of stratification and so prevent the formation of oxygen-deficient bottom waters. The aim is to solve some common operational, aesthetic and consumer-linked problems, generally associated with the following:

- Colour and turbidity (often called "dirty water") which may result from high levels of iron and manganese released from the sediment during periods of stratification.
- Tastes and odours - due to either decomposition products (such as hydrogen sulfide) or arising from the growth of microorganisms such as algae.

Aeration/destratification has often been successful in controlling iron and manganese colour and turbidity problems. However, destratification has been least successful in controlling algae-related taste and odour problems, even where the amounts of nutrients released from the sediments were reduced.

1.1 Harding Reservoir

Harding Reservoir was constructed during 1983 and 1984 (Wark *et al*, 1986), brought into water supply service in 1985, and first reached full supply level in 1986.

Shortly after being brought into service, complaints were received from consumers regarding tastes and odours. On observing that these problems largely disappeared when the reservoir underwent natural destratification, it was decided to artificially destratify the waterbody by aeration. This action coincided with a significant reduction in the tastes and odours and associated complaints.

Other water quality problems experienced include 'dirty water' complaints and increased chlorine demand resulting in reduced microbial quality of the water delivered to consumers. These were generally linked to excessive primary production in the Reservoir (Rosich and Shier, 1988).

Periods of excessive turbidity in the water body have been associated with both major inflow events and overturn.

Early aerators were replaced on 18 February 1989 by the current one designed by the Centre for Water Research, University of WA. The design of this aerator was based on simulations using their computer model DYRESM (Dynamic Reservoir Modelling). The aerator consists of a pipe 400 m long suspended off the bottom, delivering air through 100 evenly spaced 1.58 mm diameter holes.

2. AIMS and OBJECTIVES

The broad aim of Stage 2 was:

- to study the application of aeration/destratification in a reservoir with water quality problems, so as to develop an understanding of the effect of this technique on water quality in the reservoir and identify the processes that control that quality.

Particular objectives of Stage 2 were:

1. to assess the relationships between water quality and climate, land-use and hydrology;
2. to assess the processes determining the impact on physical variables (temperature, dissolved oxygen, turbidity, colour, light penetration and pH) of stratification and destratification by aeration;
3. to assess the processes determining the impact on biological and chemical variables (phytoplankton, zooplankton, macrophytes, tastes and odours, nutrients, iron, manganese and major chemical components) of stratification and destratification by aeration and
4. to assess the processes determining the impact of stratification and destratification by aeration on water quality impacts of releases from sediment of phosphorus, iron and manganese.

3. THE APPROACH to the STUDY

The approach has been to study the processes that control the water quality in the Reservoir (Chapter 3).

Harding Reservoir is located in the tropics (Figure 3.1) and as such could be compared with other tropical water bodies. However, there are many water bodies outside the tropics that behave in similar ways to those within the tropics. In this Report, therefore, the term 'tropical' describes a water body based on the way it functions, rather than its geographic location:

- minimum temperatures always above 4^0C;
- a single unstratified period per year;
- destratification associated with the onset of the (usually brief) rainfall runoff period; and
- greater significance of bacterial respiration and other decomposer processes.

Thus, the term 'tropical' applies to water bodies in the warm temperate as well as the tropical zones of the world.

4. CLIMATE, LAND USE and HYDROLOGY of HARDING RESERVOIR

Harding Reservoir is located 23 kilometres south of Roebourne in the north of Western Australia (Figures 3.1 and 4.1). This is an arid tropical region characterised by high summer temperatures, high evaporation rates and unreliable rainfall (Chapter 4). Minimum temperatures range from an average of approximately 15°C in July to 26°C in January while maximum temperatures average around 26°C in July and 36°C in January.

The catchment area (1017 km^2) is characterised by mixed shrub and spinifex grassland and there is little human activity.

In view of the climatic conditions of the area, surface storages are often unable to provide a guaranteed water supply. Consequently, the water supply system serving the West Pilbara uses both the Harding

Reservoir surface storage and the nearby Millstream underground aquifer. When sufficient water of acceptable quality is available in Harding Reservoir it is preferentially used as a water supply.

The Reservoir was formed by the construction of a main embankment in the Harding River valley and a small, auxiliary embankment south in the reservoir flood zone (see Figures 4.1 and 4.4). At full supply level the Reservoir has a surface area is 14 km^2 and a capacity of 64 million m^3. Maximum depth is 24 m.

5. RESULTS

5.1 Stratification

Natural destratification events in Harding Reservoir may be derived from temperature changes associated with the onset of the cooler period or from intense rainfall events leading to massive inflows.

Persistent thermal stratification leads to a decrease in dissolved oxygen concentrations in the hypolimnion and the measurement of both the dissolved oxygen and temperature profiles give a better indication of stratified conditions than the measurement of temperature profiles alone.

The periods of strongest stratification (see Figure 6.1) occurred during March 1989 (late summer) and September October 1989 (spring).

Breakdown of stratification occurred in February and April 1989 because of inflow under cyclonic conditions; minimum temperatures were not reached until June and July (mid winter, Figure 6.2). Significant fluctuations in stratification were apparent from November 1989 onwards even with the aerator on (Figure 4.3).

Due to its relative shallowness Harding Reservoir can both stratify and destratify very quickly (in times as short as 24 h). Furthermore, the relatively warm water temperatures lead to rapid deoxygenation of the bottom waters during periods of stratification.

5.2 Turbidity

In water bodies such as Harding Reservoir turbidity may be caused by a number of factors. It may be the result of inflow of fine particulate matter from the catchment. This may either be of organic origin or inorganic clay colloidal material. Turbidity can also be due to processes within the water body, such as phytoplankton blooms. Another possible cause is the flocculation and precipitation of iron and manganese upon oxidation of deoxygenated waters containing the reduced metals. This may occur when overturn of stratified waters takes place and hypolimnetic water that has been in contact with the sediment surface (thus accumulating Fe^{2+} and Mn^{2+}) is circulated to the upper layers of the water body and becomes oxygenated.

On five occasions water at the Reservoir offtake exceeded the National Health and Medical Research Council (NHMRC) guideline value of 5 NTU (Figure 6.8), whereon supply was then taken from the Millstream aquifer. Note, however, that on none of the sampling occasions during this study did the turbidity at the surface of the main basin exceed this level.

5.3 Biological Components

Biological components measured as part of this study included regular phytoplankton and zooplankton counts and one macrophyte survey.

Harding Reservoir contains a very wide selection of phytoplankton species (Table 8.1). For most of the study period total phytoplankton cell numbers in the main basin were less than 10,000 cells mL^{-1}. The two

peaks that occurred in total cell numbers (Figure 8.1) were due primarily to peaks in blue-green algae numbers. Groups other than the blue-green algae contributed low amounts to total numbers in both absolute and relative terms.

One might expect such peaks to represent considerable increases in biomass. However, as Figure 8.3 shows, these peaks in cell numbers did not always have corresponding peaks in chlorophyll *a* concentrations (an indicator of phytoplankton biomass). The reason lies in the species composition of the cell count peaks. In the first peak (19 November 1988) 70 % of the cell numbers were of the small, colonial blue-green *Merismopedia*, while in the second peak the small colonial blue-green *Chroococcus* comprised 70 % of total cell numbers. These small cells typically measure around 2 um in diameter and while they contribute significantly to cell counts, they contribute little towards biomass.

There were some recordings of algae associated with taste and odour problems, notably *Anabaena*, *Microcystis*, *Peridinium* and *Melosira*. These were not present, however, in quantities sufficient to cause water quality problems during the study period.

Three genera of macrophytes (large aquatic plants) were identified in the Reservoir: *Spirogyra* sp, *Myriophyllum verrucosum* and *Najas marina*. Limited resources for sampling meant that exact estimates of distribution were not possible. In the case of *Myriophyllum* and *Najas* an estimate of the distribution (Table 8.2 and Figure 8.5) was made using boat and scuba surveys. In essence, though, macrophytes grow wherever the depth of the water is less than about 2-3 m.

Total zooplankton numbers varied considerably during the study period, from the lowest values of <600 individuals m^{-3} to more than 5,000 individuals m^{-3} (Figure 8.6), with the most abundant group being the rotifers (Figures 8.7 and 8.9). The other major groups present were the Calanoids, Cladocerans and Cyclopoids (Figure 8.8) with the Copepods and Cladocerans being the larger-sized zooplankton collected. All are common zooplankters, being widespread in Australian waters.

5.4 Releases from the Sediment

The sediments of Harding Reservoir contain sufficient levels of iron, manganese and phosphorus as to cause high concentrations in the overlying waters if conditions (primarily low dissolved oxygen concentrations) become favourable for their release from the sediments. Furthermore, they contain substantial concentrations of the sodium hydroxide extractable phosphorus fraction that is readily available to the biological system.

Based on the results of experimental sediment-water columns, once dissolved oxygen concentrations fall below about 40 % saturation there is likely to be an increased net release of iron, manganese and phosphorous from the sediments. All of these components are available at quantities sufficient to ensure that the sediments could act as a source of them for many years. The extent to which these components will affect the water quality of the surface waters is primarily dependant on the vertical mixing of the water column, including the degree and stability of stratification.

6. CONCLUSIONS

6.1 Overview

During the study period (and since) there were no major outbreaks of taste and odour complaints from consumers of water supplied from Harding Reservoir. Aeration/destratification was practised throughout this period whenever water was being supplied to consumers and thus there is an *a priori* case that the technique has been successful. However, during the study period water was not supplied to consumers

while the Reservoir was not being aerated/destratified and therefore the role of the aeration/destratification technique cannot be assessed with certainty.

It should be noted that the aeration/destratification technique was introduced into Harding Reservoir only a few years after construction was completed, in response to taste and odour problems. Those early problems may have been associated with the initial filling of such an artificial reservoir. Further, there are very few relevant data from that early period with which to compare the data obtained in this study.

Another major difficulty in assessing the role of aeration/destratification in Harding Reservoir is the relative instability of stratification under natural conditions. Before this study commenced it was thought that the Reservoir stratified for periods of months during the dry season. However, this view was based on insufficient measurements as is shown in the work by the Centre for Water Research, UWA (Lewis *et al*, 1991). Although the Reservoir can stratify strongly, the climatic conditions can cause rapid (days) change between stratified and destratified states. Main reasons for this instability are the local climatic and geographic conditions together with the relative shallowness (maximum depth 24 m) of the Reservoir.

As community concerns for taste and odour diminished the incidence of 'dirty water' complaints increased. A separate study of this issue (Henderson *et al*, 1990) showed that the main source of the particulates in 'dirty water' originated from the Reservoir. Thus, in the later stages of the aeration/destratification study attention was also focused on turbidity in the Reservoir.

6.2 Aeration

The aerator installed in Harding Reservoir during this study is located 4-6 m above the bottom to avoid the mass transport of nutrient rich bottom waters and bottom sediments to the upper waters . If these materials reach the upper waters algal production could be stimulated and/or turbid waters enter the supply.

The result has been the formation of a bottom layer (that is, below the level of the aerator) high in nutrients and particulates. On occasions, natural overturn (or simply deepening of the upper mixed layer to below the level of the aerator) has brought these waters to the surface. As a result the Reservoir was taken off supply due to high turbidity. On other occasions excessive turbidity in the surface layers, without overturn, decreased after the aerator was switched off.

The energy contributed by the aerator has on occasions been insufficient to maintain the waters above the aerator in a fully mixed state. As a result temperature differences of up to 4.5°C and differences in dissolved oxygen concentrations of up to 7.5 mg L^{-1} between the surface waters and those just above the aerator were observed.

6.3 Biological Components

Harding Reservoir exhibits a wide variety of phytoplankton and zooplankton species indicative of a biologically and chemically healthy system.

Levels of these groups (expressed as 3-8 chlorophyll *a* ug L^{-1} for phytoplankton) are in the range for eutrophic rather than hypereutrohic waters (10's to 100's for chlorophyll *a*). Although this biological production gives rise to a relatively high chlorine demand (3-4 mg L^{-1}) there were no incidences of major taste and odour complaints during the time of this study, nor have there been since.

The very few relevant data available for the Reservoir before the implementation of aeration/destratification means it is not possible to ascertain whether the levels of these biological components have changed and hence whether the apparent reduction in the taste and odour problem has been due to a change in these biological components.

Given the warm water temperatures (about 17-30°C), the ample available light and the availability of nutrients from the sediments the potential exists for excessive algal or zooplankton populations and resultant water quality problems.

6.4 Releases from the Sediments

The sediments of Harding Reservoir contain sufficient levels of iron, manganese and phosphorus as to cause high concentrations in the overlying waters if conditions (primarily low dissolved oxygen concentrations) become favourable for their release from the sediments.

Artificial destratification/aeration to provide high dissolved oxygen concentrations (> *ca* 40% saturation) throughout the water column and to the depth of the sediment surface would provide optimum conditions for the minimisation of iron, manganese and phosphorus release from Harding Reservoir sediments. The difficulty is to achieve this oxygen level without resuspending the bottom sediments.

7. RECOMMENDATIONS

It is recommended that:

1. Aeration/destratification continue to be used as a tool to assist in the management of water quality in reservoirs with problems similar to those of Harding.

2. In the case of water bodies where the aim is to control components whose concentrations increase to only slightly above acceptable levels, aeration/destratification should be commenced when the first signs of stratification appear.

3. To minimise the incidence of elevated turbidity and nutrients in the surface layers the elevation of the aerator should be regularly monitored to ensure that it stays sufficiently above the bottom of the reservoir. In a reservoir (such as Harding) formed in a river valley with a distinct channel in a flood plain, the aerator should be placed just above the flood plain of the old river valley. This will minimise the volume of water below the aerator where high concentrations of nutrients, iron and manganese can be found.

4. In any program to assess the impact of aeration/destratification on water quality, especially the biological compartment, relevant data must be available under natural conditions. Alternatively, the program should include studies of different aeration/destratification strategies using large plastic enclosures placed in the water body.

1. INTRODUCTION

1.1 General

This report presents the results of Stage 2 of the water quality component of a comprehensive aeration/destratification study undertaken by the Scientific Services Branch of the Water Authority of Western Australia between January 1988 and June 1990 of Harding Reservoir, Western Australia. The study was funded jointly by the Urban Water Research Association of Australia (UWRAA) and the Water Authority of Western Australia (WAWA). Stage 1 has been previously published as UWRAA Research Report No 9 (McAuliffe and Rosich, 1989), McAuliffe and Rosich (1990) and UWRAA Research Report 23 (Lewis *et al*, 1991).

Stage 2 concentrated on the physical, chemical and biological aspects of the water body with reference to the effects of an aeration program designed to maintain artificial destratification. The objectives for this phase of the Study are outlined in Chapter 2 of this Report.

Harding Reservoir (also known as Lake Poongkaliyarra) is a surface water storage of the West Pilbara Water Supply system, located in the tropical region of WA (Figure 3.1). Water taken from this Reservoir for supply is chlorinated and fluoridated, but does not receive any other treatment. Therefore, management of the Reservoir aims to maintain its water quality in accordance with the Authority's drinking water quality guidelines (which are based on those given in NHMRC, 1987). Naturally occurring phenomena such as stratification, turbidity and algae growth are undesirable elements, and one method of improving the quality is artificial destratification.

A brief introduction to major limnological processes is presented below, followed by a short account of the use of artificial destratification for water quality management. This provides a background for defining the objectives of this Study, as set out in Chapter 2.

1.2 Reservoir Limnology

1.2.1 Stratification

The formation of thermal, density dependent layers (stratification) in standing waterbodies is a major area of interest in freshwater limnology. Climatic patterns determine the extent and type of stratification. In Australia, many reservoirs are monomictic with a period of stratification during summer months, an overturn period when cooler inflowing or surface waters cause a sudden mixing of the water column and winter periods when the water column remains mixed.

Typical stratified water column profiles, including those in Harding Reservoir, can readily be divided into three layers:

- epilimnion,
- metalimnion and
- hypolimnion.

The epilimnion is the uppermost layer of fairly uniform water. This water is warmer than the lower layers and is usually well-mixed and relatively turbulent. It is in this region that wind action is able to readily circulate water. The metalimnion (or thermocline) is the region of most rapid density change. The hypolimnion is the bottom layer of water which is a colder and relatively undisturbed region. The latter is the critical parcel of water, trapped in contact with the sediment during stratification, in which substances released from the sediment are able to accumulate prior to overturn.

The implications of stratification are both chemical and physical. Without mixing of the whole water column, the water quality at depth becomes altered, mostly as a result of deoxygenation. Thus stratification can present problems for the maintenance of water quality fit for human consumption (see Section 1.4). Overturn, or at least a significant increase in vertical mixing, may result in a decrease in water quality to a level where utilisation of the water must be discontinued.

1.2.2 Biological and chemical water quality

The persistence of stratified conditions may lead to reduced water quality due to the release of various components from the sediments, increased algae growth due to nutrients released from the sediments then transported to the epilimnion, increased metal concentrations (especially iron and manganese), and increased levels of turbidity, colour and organic carbon.

Where nutrient and physical conditions allow reservoirs (whether stratified or not) can have problems with algae growth, toxicity resulting from algae growth, high zooplankton numbers and aquatic insects. The extent and type of macrophytes in a water body may play an important role in trapping nutrients and preventing phytoplankton blooms, but may interfere with the recreational use of the reservoir.

Phytoplankton in a water source may impart tastes and odours to the water while high zooplankton numbers may impart a "fishy odour" to the water. A list of species and the tastes/odours that they can cause is given in McAuliffe and Rosich (1989)

1.3 Artificial Destratification in Australia

1.3.1 Objectives of Destratification

Artificial destratification is the process of mixing the water column by aeration or mechanical means to disrupt or prevent the onset of stratification and prevent the formation of an anoxic hypolimnion. A detailed review of destratification in Australia is given in McAuliffe and Rosich (1989) and a summary given in McAuliffe and Rosich (1990).

Artificial destratification of water storages aims to improve water quality and hence solve some of the more common operational, aesthetic and consumer-linked problems. These are generally associated with the following:

- Tastes and odours - either due to algal growth or reduced compounds (such as hydrogen sulfide).
- Colour and turbidity (or "dirty water") which may result from high levels of clay turbidity or iron and manganese released from the sediment during periods of stratification.

By aerating the water column, mixing occurs in the water above the aerator. Among other things, this is thought to reduce phytoplankton growth by preventing accumulation of the algal cells at or near the water surface where they have access to high light conditions. Instead, they are mixed to lower levels where light for photosynthesis is restricted.

Aeration of the water in contact with the sediments (with or without destratification) aims to reduce release from the sediments of iron, manganese and phosphorus, these being components which can cause increases in turbidity, colour and algal growth.

1.3.2 Impact of artificial destratification on water quality

A review of artificial destratification in Australia (McAuliffe and Rosich 1989) found that in most cases (35 out of 50 systems studied) some degree of success was (subjectively) considered to have been achieved in maintaining high water quality. The aeration techniques employed succeeded in raising the levels of dissolved oxygen in the water column, which led to (some) control of colour/turbidity and iron/manganese related problems.

The success of aeration programs relies mainly on the maintenance of high dissolved oxygen concentrations at the sediment interface. This can be achieved by complete destratification where mixing of the water column includes the lowest layer of water. The achievement of oxygenation of the sediment interface depends largely on the design of the destratification installation and how effectively it is operated.

It is in the area of algae control that destratification applications have been least successful. It is often unclear why the application of aeration and/or destratification has not led to a reduction in algal problems, even where the amounts of nutrients released from the sediments have decreased.

1.4 Background to the Study

1.4.1 Harding Reservoir history

Harding Reservoir was constructed in 1983 and 1984 (Wark *et al*, 1986). It was brought into water supply service in 1985, and first reached full supply level in 1986.

Shortly after being brought into service, complaints were received from consumers regarding tastes and odours. These largely disappeared when the reservoir underwent natural destratification, so it was decided to use aeration to artificially destratify the waterbody. This had significant success in reducing the tastes and odours and associated complaints.

Other water quality problems experienced include 'dirty water' complaints and increased chlorine demand resulting in reduced microbial quality of the water delivered to consumers. This was generally linked to excessive primary production (Rosich and Shier, 1988).

Periods of excessive turbidity in the water body have been associated with both major inflow events as well as overturn.

1.4.2 Aerator history

Harding Reservoir was first artificially aerated in June 1986 with a prototype subsequently replaced in September 1988 by an aerator (aerator 2, see Figure 4.4) designed by Frank Burns and Associates of Victoria. This was subsequently replaced (18 February 1989) by aerator 3 which is in current use. This aerator was designed by the Centre for Water Research, University of WA on the basis of simulations using their computer model DYRESM (Dynamic Reservoir Modelling). The aerator consists of a 400 m long pipe suspended off the bottom, delivering air through 100 evenly spaced 1.58 mm diameter holes. Originally the aim was to suspend the pipe up to 6 m above the bottom, but recent checks have shown it to be 2-3 m above the bottom. Nevertheless, the aerator is at RL 44.4-45.4 m which places it above the general floor of the old flood plain of the River, that is at the base of most of the volume of the Reservoir, and about 15 m below the spillway. A logbook for aerator operation was established on 12 November 1986.

The aeration of the reservoir has been fairly continuous, with periods of non-operation caused mainly by inflow and overturn events which caused high turbidities, or compressor malfunction. The aerator was off for most of winter 1988 and 20/06/89 - 15/09/89.

1.4.3 Previous work

The study encompassed by this report is the second stage of the project carried out under the auspices of the Urban Water Research Association (UWRAA). The first stage was a joint project between the Scientific Services Branch of the Water Authority of WA and the Centre for Water Research, University of WA. This first stage comprised:

- a comprehensive review of the status of reservoir destratification in Australia, evaluating fifty case studies in the context of the current technical literature (McAuliffe and Rosich, 1989,1990) and
- the development of a numerical model of bubble plumes, including preliminary field trials (Lewis *et al*, 1991).

A three-weekly monitoring program has been undertaken since December 1987, with information being gathered on water quality and stratification in the reservoir. The water quality program was extended at the commencement of this project to provide additional data on the biological, chemical and physical aspects of the reservoir and its sediments.

2. OBJECTIVES

2.1 Aims

The broad aim of Stage 2 of the study was:

- to study the application of aeration/destratification to a reservoir with water quality problems, so as to develop an understanding of the effect of this technique on water quality and the processes that control that quality.

This aim has been achieved in Harding Reservoir, in the north of Western Australia, but ONLY while aeration/destratification has been practised. This limitation was due to the need to supply drinking water from the Reservoir whenever possible and therefore we were unable to study its water quality during periods when it would be expected to stratify.

More specifically:

- clarify the biological, chemical and physical water quality effects of aeration/destratification and when to apply the technique;
- increase the understanding of processes that determine water quality in water bodies and
- reduce the costs of aeration/destratification through improved design techniques and greater understanding of the processes governing water quality.

2.2 Objectives

Particular objectives of Stage 2 were:

1. to assess the relationships between water quality and climate, land-use and hydrology;
2. to assess the processes determining the impact on physical variables (temperature, dissolved oxygen, turbidity, colour, light penetration and pH) of stratification and destratification by aeration;
3. to assess the processes determining the impact on biological and chemical variables (phytoplankton, zooplankton, macrophytes, tastes and odours, nutrients, iron, manganese and major chemical components) of stratification and destratification by aeration and
4. to assess the processes determining the impact of stratification and destratification by aeration on water quality impacts of releases from sediment of phosphorus, iron and manganese.

Chapter 3 discusses the conceptual model on which the work plan for the study was based. Then follows Chapter 4 which addresses objective 1., Chapter 6 objective 2., Chapters 7 & 8 objective 3. and Chapter 9 objective 4. Chapter 5 describes the sites chosen for the sampling program and the methods used in the study. Chapter 10 draws together the conclusions from the study and Chapter 11 the recommendations.

3. A CONCEPTUAL MODEL FOR WATER QUALITY IN TROPICAL RESERVOIRS

3.1 Objectives

Harding Reservoir is located in the tropics (Figure 3.1) and as such could be compared with other tropical water bodies. As part of work, a comparative review of the limnology of tropical and temperate reservoirs was undertaken and this chapter summarises the results (see also Rosich 1990).

However, there is perhaps only one simple definition of a tropical water body; namely "one that is located between the tropics", but this hides the diversity of types of such water bodies. At the same time, there are many water bodies outside of the tropics which behave in similar ways to those within the tropics.

This chapter seeks to highlight:

- that the processes occurring in all water bodies are similar,
- that the individuality of water bodies is due to differing relative importance of the various processes in each case and
- that the use of the process approach should be adopted in studying all water bodies.

While there are some trends in the differences between tropical and temperate waters, there are also patterns in the differences between water bodies located in the tropics.

Overall, it is argued that though there are some general differences in the limnology of water bodies located in the tropics, tropical limnology should not be treated separately. Rather, our understanding of the processes occurring in all water bodies should be improved so that we can understand the causes of the differences.

Furthermore, there are strong similarities between many water bodies in tropical and in warm, temperate regions (in, for example, Australia, southwestern United States, Africa and the Middle East) and in this report the term 'tropical waters' is used to encompass both regions.

Some suggestions for further study in tropical waters are also given.

3.2 Solar Irradiance

The annual maximum daily irradiance does not vary greatly with latitude, the range being of the order of 1.1-1.4 kJ cm^{-2} day^{-1} (Lewis, 1987). However, the reduced mean day length at higher latitudes results in a much lower total daily direct solar radiation, especially for latitudes above 45^0 (Table 3.1).

The actual total daily radiation reaching the surface of the earth is further modified by variations in absorption by the atmosphere, particularly cloud cover. Thus the greater extent of this cover around the equator results in the actual total daily radiation at the surface being at a maximum at latitudes near $20\text{-}30^0$ (Landsberg, 1961). Hence there are many regions beyond the tropics (in Australia, southwestern United States, North Africa, and the Middle East) which receive irradiance in excess of 460 kJ cm^{-2} yr^{-1}, whereas much of the Amazon region receives less than 280 kJ cm^{-2} yr^{-1}.

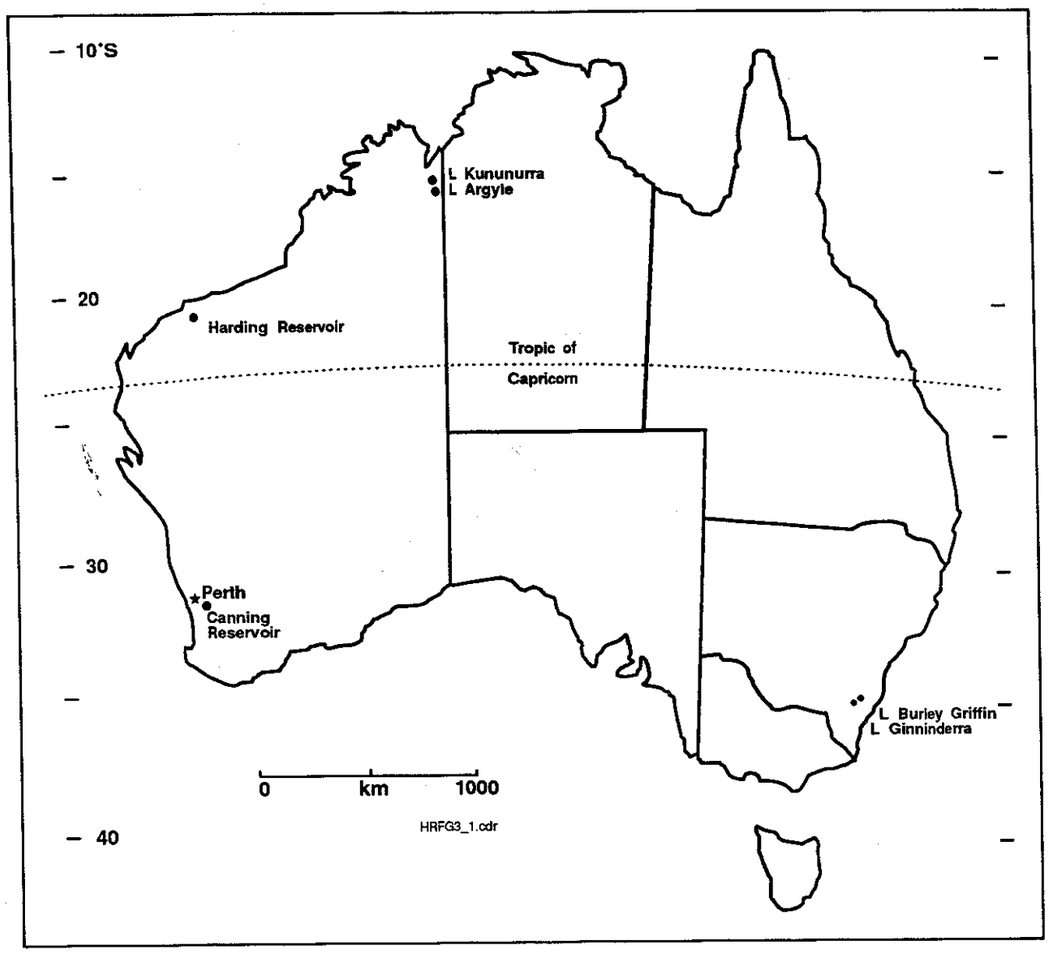


Figure 3.1 Some 'tropical' water bodies in Australia

Table 3.1 Daily direct solar radiation (kJ cm^{-2} day^{-1})

Latitude North	Jan	Apr	Jul	Oct
0	1.12	1.20	1.14	1.27
15	0.99	1.34	1.32	1.13
30	0.69	1.31	1.42	0.98
45	0.35	1.15	1.41	0.66
60	0.18	0.98	1.40	0.33
75	0	0.65	1.38	0.07
90	0	0.36	1.42	0

From: Hutchinson (1957)

3.3 Temperature, Stratification, Mixing and their Effect on Water Quality

The greater amounts of radiation reaching the surface of both warm temperate and tropical water bodies (hereinafter referred to as tropical water bodies) result in higher water temperatures. In addition, there is a greater potential for thermal stratification due to the greater rate of change in the density of water at higher temperatures (Table 3.2). Again, the water temperatures rarely fall to levels that inhibit physiological activity.

Table 3.2 Density of water

Temperature	Density	Change per ^{0}C
0	0.9998	
4	1.0000	+ 0.00005
10	0.9997	- 0.00005
15	0.9991	- 0.00014
20	0.9982	- 0.00018
25	0.9971	- 0.00022
30	0.9957	- 0.00028
35	0.9941	- 0.00032

From: Hutchinson (1957)

A major characteristic of tropical water bodies is that the breakdown of stable stratification is often forced more by water inflow rather than direct cooling of the stored water. This results, in part, from the greater amounts of daily radiation during the winter months (compared with those for cool temperate and arctic regions) and in part from the greater variation in water inflows.

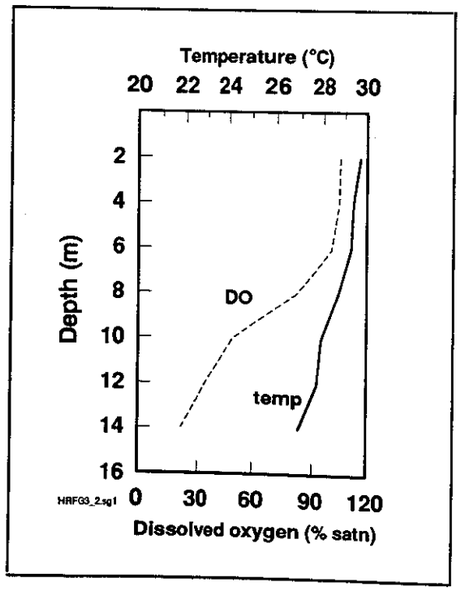


Figure 3.2 Dissolved oxygen and temperature profiles for Harding Reservoir, 7 January 1988 (aerator on).

Whether or not a tropical water body shows stable stratification seems to depend most on its depth, modified by the influence of the degree of wind mixing and altitude. A very lucid account of the interrelationships of radiation, latitude and elevation with stratification and mixing is given by Lewis (1987).

It is important to note, however, that stratification is merely the extreme situation of restricted vertical mixing. In the example of Harding Reservoir (Figure 3.1), even where artificial aeration (aerator at 60 L sec^{-1}, too low for complete mixing) gave a near uniform temperature profile, oxygen saturation decreased from near 100 % at the surface to below 50 % at the bottom (Figure 3.2 and Table 3.5).

This also illustrates the effect of the higher respiration rates that arise from warmer water temperatures (near to 30°C). This increased respiration is reflected in lower pH in the bottom water.

In addition, in unstratified water bodies materials (including nutrients) that constantly diffuse out of the sediments are transported to the surface and are available for primary productivity. However, there is almost no information available in the literature on the *rates* of vertical mixing in water bodies (whether stratified or not) and this is an area in need of urgent research.

In the example of Harding Reservoir, which receives inflow for only a brief period of the year, its eutrophic status (chlorophyll *a* 5-10 ug L^{-1}) is maintained by the recycling of nutrients within the water column and from the sediment.

3.4 Water Inflows, and Nutrient Loads and Modelling

A second major characteristic of tropical water bodies is that inflow is highly variable throughout the year, and in many cases is restricted to a brief period of only a few weeks. There are many reports in the literature which show that the great majority (say 90 %) of the load of materials, including nutrients, to these water bodies enters during only a small fraction (say < 10 %) of the time (Tables 3.3 and 3.4).

Table 3.3 Effect of rainfall on rural exports of phosphorus (kg P ha^{-1} yr^{-1})

Rainfall period	Total-P	Particulate-P
normal	0.006	0.003
drought	0.001	< 0.001
1-in-12 year flood	0.038	0.025
1-in-24 year flood	0.237	0.221

From: Cullen *et al* (1978)

This has two main consequences. Firstly, programs to measure input and output loads must be based on water flows (rather than time) and must monitor storm event runoff. Secondly, the question of the applicability of, for example, nutrient loading models must be considered. The data available suggest that they are applicable providing a correction is made if turbidity reduces the depth of light penetration compared with the depth of mixing (Rosich, 1983, Rosich *et al*, 1991).

Table 3.4 Annual phosphorus load (urban area) - effect of storm events

Model	Annual phosphorus load
weekly monitoring only	1,900 kg P
including storm model	4,200 kg P

From: Cullen *et al* (1978). The data here are only for 'normal' weather conditions.

3.5 Biogeochemical Water Quality Processes

A number of the characteristics discussed earlier can be illustrated by comparing several Australian water bodies located in the tropical and warm temperature zones at latitudes up to 35°S (Tables 3.5-3.7 and Figure 3.1).

The bottom temperatures for the two stratified reservoirs are close to the values predicted by Lewis (1987). However, the surface temperature for the three reservoirs at or below latitude 21°S are about 3°C higher while for the three other reservoirs the values are closer to those predicted.

Table 3.5 Example summer data - physical

Location	L Argyle	L Kununurra	Irrigation channels	Harding Reservoir	Canning Reservoir	L Burley Griffin	Lake Ginninderra
Latitude South	16	16	16	21	32	35	35
Max depth (m)	45	12	2	20	66	17	10
Mean depth (m)	8.4	8.7		4.5	15	4.7	3.5
Volume (10^6 m^3)	6,500	97		64	91	33	3.7
Area (10^6 m^2)	770	11		14	6.1	7	1.1
Stratified	yes	no ?		unstable	yes	no	no
Temperature - top (°C)	32	33	35	30	25	23	23
- bot	23	30		28 (aerated)	11.5	20	21
pH - top	8.7	8.5	9	8.5	6.5	6.6	6.8
- bottom	7.4	7.8		7.5 (aerated)	6.3	6.4	
Turbidity - top (NTU)	0.8	0.6	0.8	1.5	0.6	6	13
- bot	1.2	1.7	0.8	7	0.4	6	24
Colour - top (HU)	2	3	4	5	3		
- bottom	4				6		

The major ions in these water bodies (Table 3.6) show the increased proportions of sodium and chloride typical of mainland Australian waters. This is due to higher evaporation rates in the arid and semi-arid climates, the greater proportions of rainfall that are of oceanic origin, and the composition of the soils.

In Kununurra, WA, the large, deeper, primary reservoir Lake Argyle has a low standing crop of phytoplankton (chlorophyll *a* ~1 ug L^{-1}, Table 3.7) but a sufficient detrital fallout to render the hypolimnion anaerobic with a consequent elevation in iron and manganese levels. On the other hand, the secondary, shallower, irrigation reservoir Lake Kununurra does not show stable stratification, also has a low phytoplankton crop, but is dominated by extensive macrophytic growth, including rooted, emergent and attached plants.

Excessive macrophyte growth in the irrigation channels has shown the classical response to mechanical and chemical control strategies. An initial dominance by rooted, emergent vegetation (*Cumbungi*, which was controlled by mechanical and herbicide means) was replaced by first *Vallisneria*, then *Potamogeton*, and finally with a mat-forming filamentous alga. Continued acrolein treatment provides only short term relief from the choking effects of this submerged and filamentous growth.

Table 3.6 Example summer data - major ions
(mg L^{-1}, unless otherwise stated)

Location	L Argyle	L Kununurra	Irrigation channels	Harding Reservoir	Canning Reservoir	L Burley Griffin	Lake Ginninderra
Sodium	19	18	18	16	49	17	16
Potassium	2.8	2.3	2.3	2.9	1.3	2.7	3.7
Calcium	19	19	16	14	2.2	16	11
Magnesium	8	7	7	11	6.5	10	17
Chloride	13	12	12	16	86	19	26
Sulfate	6.4	6.6	6.6	5.6	15	26	12
Alkalinity (meq L^{-1})	1.96	1.92	1.81	1.87	0.18	1.59	1.86
Conductivity (mS m^{-1})	25	25	23	25	36		
Total filterable solids	205	200	190	200	160	180	190
Iron - top	<0.10	<0.10	<0.10	<0.10	<0.10		
- bottom	0.3			0.4			
Manganese - top	<0.04	<0.04	<0.04	<0.04	<0.04		
- bottom	0.5			0.1			
Silica (as SiO_2)	6.2	6.0	5.5	5.6	1.9		
Filterable organic carbon	2.4	2.8	3.4	4.4	2.6		

Harding Reservoir has a catchment without point sources of nutrients, with negligible agricultural activities (past or present), and was expected to not have eutrophication problems. However, the iron ore in the region contains significant levels of phosphorous and wash-off during the cyclonic runoff events is thought to provide the nutrient stimulus. The results presented in this Report indicate that the critical water quality processes in Harding Reservoir follow concepts applicable to a wide range of water bodies from all latitudes.

Many studies (for example, Burgis, 1978; Nilssen, 1984; Jorgensen, 1986; Lewis 1978; Ganf, 1982; Allanson and Hart, 1975; and Infante, 1978) have shown that the pattern of phytoplankton succession, phytoplankton-zooplankton relationships, and nutrient-biological growth relationships are essentially similar to those for cool temperate water bodies.

Of particular significance for tropical water bodies that do not show stable stratification is the role of sediments in supplying nutrients (especially phosphorous) to the water column. The results for Harding Reservoir (see Chapter 9) are comparable to those found for Lakes Burley Griffin and Ginninderra (Rosich and Cullen, 1981 and Rosich, 1982). The net flux of phosphorous into anaerobic waters overlying sediments has been documented many times, but the undirectional fluxes have been studied much less often. As shown in Table 3.8 these can be of enormous magnitude and are markedly affected by the oxygen status of the overlying waters. Nevertheless, the flux of phosphorous out of the sediments into an aerobic overlying water column can still be sufficient to maintain eutrophic conditions. Furthermore, 45-47 % of the total phosphorous content of the sediments of these lakes could be taken-up by algae.

Table 3.7 Example summer data - nutrients
(ug L^{-1}, except N:P and bacteria)

Location	L Argyle	L Kununurra	Irrigation channels	Harding Reservoir	Canning Reservoir	L Burley Griffin	Lake Ginninderra
Total P - top	6	6	8-18	10	2.5	27	16
- bottom	15	10		11	1.9	140	20
Filterable react P - top	5	4	4-7	2	1.9		
- bot	7	4		5	1.3		
Total N - top	230	220	120 to 380	280	120	650	780
- bot	290	200		400	180		
NO_3+NO_2-N - top	22	22	10-35	16	77	52	44
- bot	75	14		100	28		
NH_3-N - top	8	24	16-25	16	23		
- bot	140	14		22	10		
N:P - top	36	34	6-35	29	47	25	20
- bot	20	22		38	93		
Chlorophyll *a* - top	1.4	1.4	1.3 to 1.6	8.5	1.0	4.2	3.4
- bot	0.3	1.2			0.1		
Phaeophytins - top	<0.1	<0.1	<0.1 to 0.7	2.5	<0.1		
- bot	0.5	0.6			0.1		
Health-related bacteria	good	poor	poor	good	good		

The significant controlling factor in the release of sediment phosphorous appears to be the availability of sufficient readily-assimilable organic carbon as to allow high rates of bacterial metabolism maintain strongly reducing conditions within the sediment. This process can be turned to advantage in strategies to reduce the trophic status of lakes by adding an oxidant (nitrate) to stimulate bacterial respiration and hence rapidly reduce the level of the readily-assimilable carbon (Ripl, 1976). Very high losses of nitrate-nitrogen can occur at the sediment water interface (Table 3.9).

Table 3.8 Undirectional phosphorous fluxes at the sediment-water interface in column studies

Lake	Overlying water			
	Anaerobic		Aerobic	
	conc ug P L^{-1}	flux mg P m^{-2} hr^{-1}	conc	flux
Burley Griffin	970	3,400	68	230
Ginninderra	63	220	5	17

From: Rosich and Cullen (1981) and Rosich (1982). Fluxes measured with ^{32}P.

Table 3.9 Net flux at the sediment-water interface in columns of anaerobic water overlying sediments (mg m^{-2} day^{-1})

	Lake Burley Griffin	Lake Ginninderra
phosphorous gain[2]	3.2	0.1
NO_3-N loss[3]	1,010	940

Note:
1. From Rosich and Cullen (1981) and Rosich (1982).
2. Without added nitrate.
3. With added nitrate (480 mg N L^{-1}).

The data for total nitrogen given in Table 3.7 suggest that more emphasis should be placed on the denitrification process in natural waters. There appears to be a greater rate in waters located in the tropics resulting in differing total nitrogen levels: say about 250 ug N L^{-1} in the those waters compared with say 1000 for water located in the warm temperate zone and up to say 5000 for cool temperate waters.

3.6 Conclusions and Recommendations

There are many observable differences when comparing water bodies located in the tropic and warm temperate zones with those located in the cool temperate and arctic zones. This is especially true for the vertical mixing process, including the stability of stratification. These differences result from predictable responses to changes in the external factors that control the fundamental processes involved. At the same time, the biological processes seem to follow similar patterns in all waters although the relative rates of some of them may differ (especially denitrification which appears to be significantly greater in tropical waters).

Future studies of tropical waters must be based on gathering data on the processes involved, with the relative importance of the various processes being assessed for *each* body.

Because water inflows are often highly variable throughout the year sampling programs must be based on water flows (rather than time) and must monitor storm event runoff.

Water quality models must also take account of this variability.

Two areas requiring research are:

- the *rates* of vertical mixing (especially in waters that do not show stable stratification) and the water quality effects of these different rates and
- the rates and controls on the undirectional fluxes at the sediment-water interface.

The use of the term 'tropical' to describe a water body should be based on the way it functions, rather than its geographic location:

- minimum temperatures always above 4°C;
- a single unstratified period per year;
- destratification associated with the onset of the (usually brief) rainfall runoff period; and
- greater significance of bacterial respiration and other decomposer processes.

Thus the term 'tropical' would apply to water bodies in the warm temperate as well as the tropical zones of the world.

4. CLIMATE, LAND USE and HYDROLOGY of HARDING RESERVOIR

4.1 Climate

Harding Reservoir is located 23 kilometres south of Roebourne in the north of Western Australia (Figure 3.1 and 4.1). This is an arid tropical region characterised by high summer temperatures, high evaporation rates and unreliable rainfall.

The mean annual rainfall in the catchment typically varies from 300 mm to 350 mm. Although tropical, the catchment lies too far south to receive regular monsoonal rain. Instead, most rainfall events are associated with tropical cyclones. These typically arrive between December and April and may deliver some of the most intense rainfall in Australia. For example, nearby Whim Creek recorded a maximum daily rainfall of 740 mm in April 1898. Thirteen tropical cyclones have crossed the coast in this region since 1980.

The irregular nature of rainfall in the area means that the number of days of rain per year is low, typically between 25 and 35 days per year. The Harding Reservoir itself experiences an average rainfall of 306 mm falling over an average period of fourteen days.

Minimum temperatures range from an average of approximately 15°C in July to 26°C in January while maximum temperatures average around 26°C in July and 36°C in January.

The combination of high temperatures and low humidity lead to high levels of evaporation. The estimated mean annual reservoir evaporation is 2,700 mm, almost 9 times the average annual rainfall.

4.2 Land Use

The catchment area is characterised by mixed shrub and spinifex grassland. The area immediately around the reservoir consists largely of scree slopes made up of large dolerite boulders. These slopes have allowed very little soil development and as such vegetation is extremely sparse.

The catchment area was used as low density grazing by pastoral stations. Both sheep and cattle were grazed at densities in the order of 5-10 hectares per animal.

4.3 Hydrology

4.3.1 Conjunctive use

In view of the climatic conditions of the area, surface storages are often unable to provide a guaranteed water supply. Consequently, the water supply system serving the West Pilbara uses both the Harding Reservoir surface storage and the Millstream underground aquifer. When sufficient water of acceptable quality is available in Harding Reservoir it is preferentially used as a water supply. If either the water level or water quality fall substantially in the reservoir, water is pumped into the distribution system from the Millstream aquifer. High turbidity in the reservoir, that result from cyclonic rainfall (such as during cyclone Connie in December 1986 and Cyclone Orson in April 1989) or from overturn, is the major cause of extended periods of use of the aquifer.

Figure 4.2 displays the relative contribution made by both sources to the West Pilbara Water Supply since 1984.

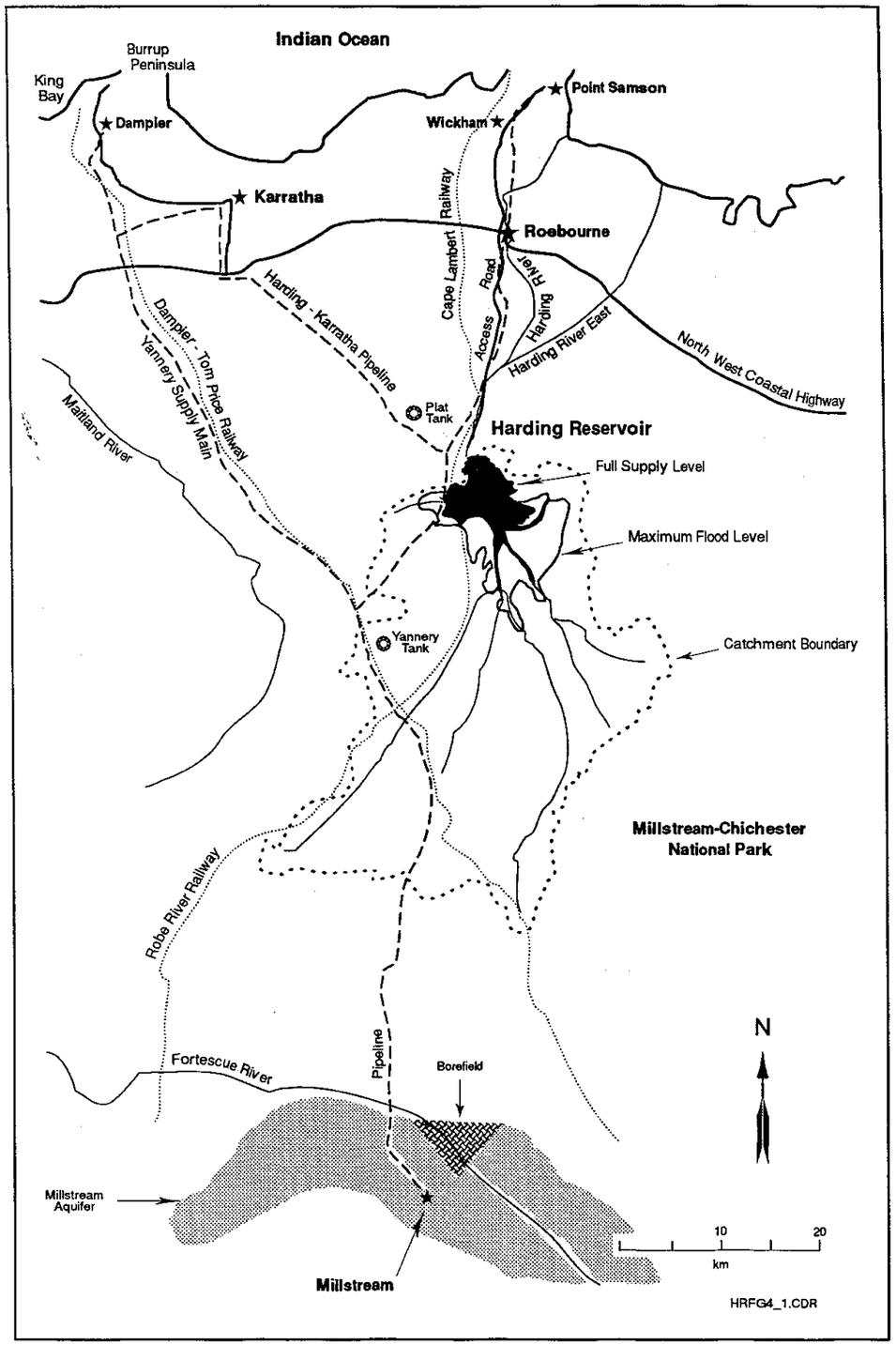


Figure 4.1 Harding Reservoir and West Pilbara Water Supply System

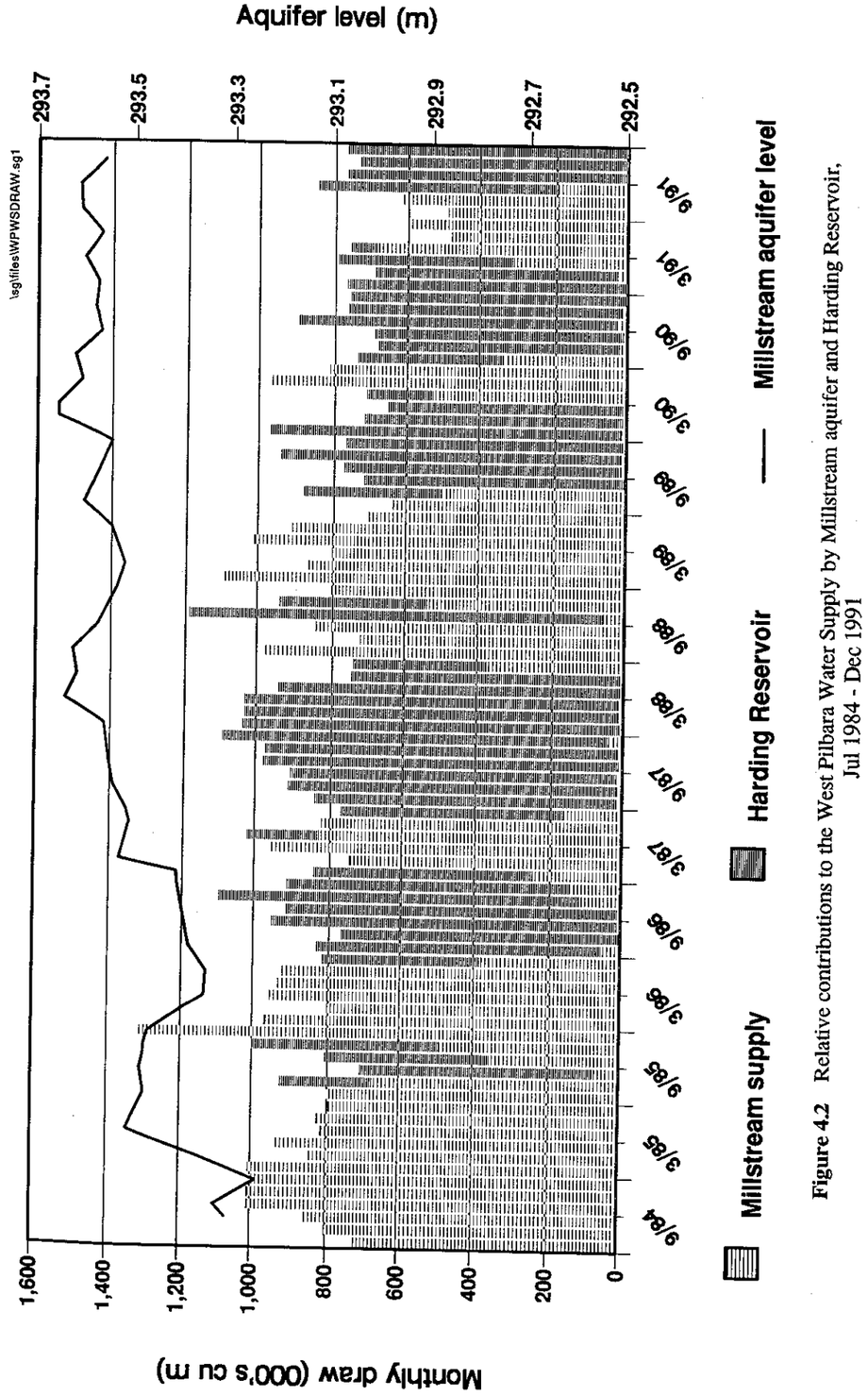


Figure 4.2 Relative contributions to the West Pilbara Water Supply by Millstream aquifer and Harding Reservoir, Jul 1984 - Dec 1991

4.3.2 Millstream aquifer

The Millstream aquifer is a 600 km^2 basin of calcrete. The water surface is at a depth of 20-25 m with a saturated zone of up to 25 m thick. The aquifer is estimated to have a storage volume in excess of 600 million m^3. Studies indicate that if required, an annual volume of 28 million m^3 could be safely drawn (at least with respect to maintaining the aquifer) from the system.

The Millstream area is of considerable ecological importance. The permanent springs and pools support a habitat in stark contrast to the surrounding harsh environment. A major benefit of the conjunctive use strategy is the prolonged maintenance of this habitat.

4.3.3 Harding River and catchment

The Harding River catchment occupies 1071 km^2. The mean annual runoff is estimated at 39 mm. Given the low rainfall and high evaporation, this runoff is considered a relatively high figure, due primarily to the thin soil and rocky nature of the catchment.

A better appreciation of the arid nature of the catchment may be gained from the median annual runoff of 20 mm. Flow distribution in the catchment is highly skewed, yearly annual runoff varies from 0 to 274 mm.

Flows in the Harding River are most common in the first six months of the year following tropical cyclones and thunderstorms, but persist for only short periods. The combination of cyclonic rains, catchment shape and the rocky nature of the catchment, tend to generate "flash" flows, rising rapidly to a peak and then receding rapidly. Mean annual streamflow is 42 million m^3.

4.3.4 Harding Reservoir

At full supply level, the Harding Reservoir (Lake Poongkaliyarra) has a surface area of 14 km^2 and a capacity of 64 million m^3. Maximum depth is 24 m (Figure 4.4).

The full supply level (spillway height) for the Reservoir is at relative level (RL) 60.026 m AHD (Australian height datum).

During the study period presented in this report there were two major inflow events (Figure 4.3). The first resulted from cyclone Connie in January 1987 and the second, more prolonged inflow resulted from a combination of cyclone Orson in April 1989 and following unseasonal 'winter' rain.

4.3.5 Reservoir bathymetry

The Reservoir was formed by the construction of a main embankment in the Harding River valley and a small, auxiliary embankment south in the reservoir flood zone (see also Figures 4.1 and 4.4).

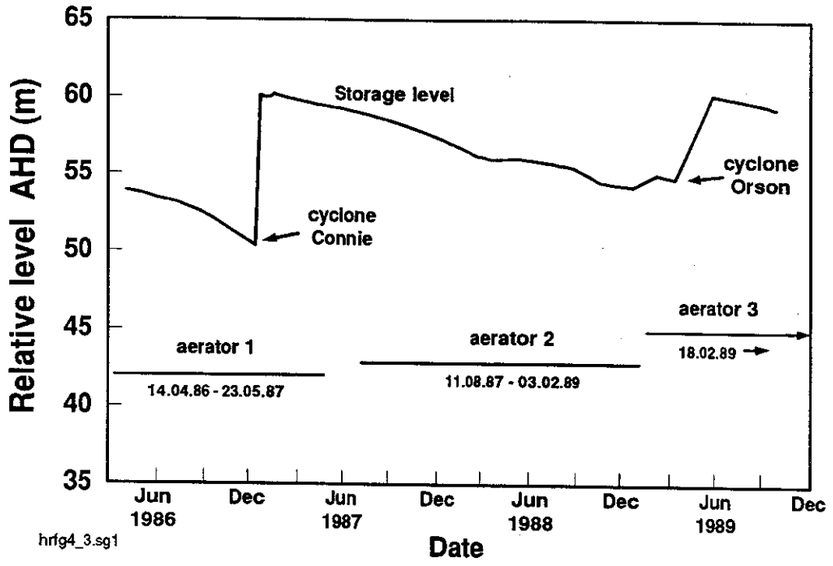


Figure 4.3 Storage and aerator levels for Harding Reservoir to the end of 1989

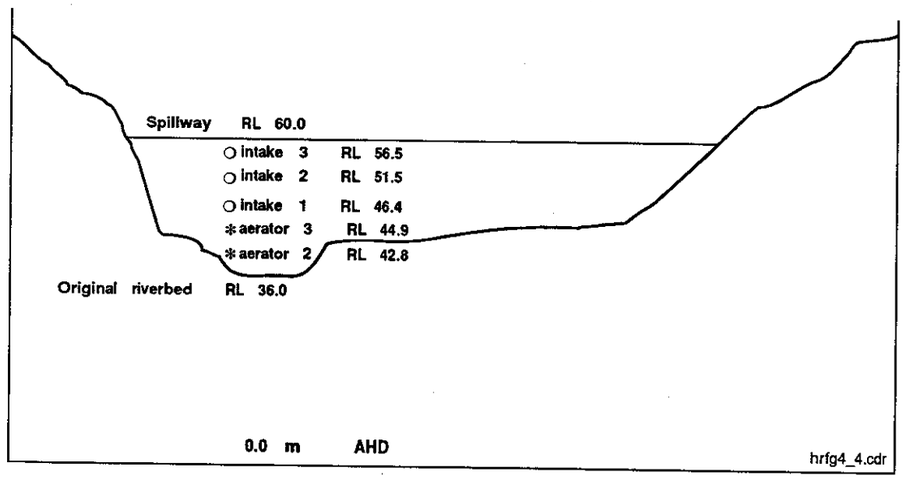


Figure 4.4 Harding Reservoir cross-section (schematic) at 100 m from the dam, showing the levels of aerators and intakes in operation during the study period

5. SITE DESCRIPTIONS and METHODS

5.1 Sampling Locations

Twelve sampling locations were used in this study. Table 5.1 lists their identification numbers, common names and maximum depths (at full supply level). Most locations were selected at regular distances from the dam wall, following the path of the old river bed. Exceptions are sites next to the offtake tower, adjacent to the aerator, and valve pit B (outlet from the reservoir to the distribution system, but upstream of chlorination).

Table 5.1 Harding Reservoir sampling stations

Station number	Common name	Depth at FSL (m)
Q7091076	0.1 km	20 m
Q7091084	0.5 km	22 m
Q7091077	0.6 km	17 m
Q7091086	0.7 km	22 m
Q7091078	1.5 km	17 m
Q7091079	2.5 km	15 m
Q7091080	3.5 km	18 m
Q7091081	4.5 km	11 m
Q7091082	5.5 km	7 m
Q7091085	offtake tower	16 m
Q7091087	adjacent to aerator	15 m (aerator 3) 17 m (aerator 2)
Q7091088	valve pit B	outlet of reservoir

In various parts of this report reference is also made to a location termed the "main basin". This represents an accumulation of data from the 0.5 km and 0.7 km sampling stations, which are approximately in the centre of the main reservoir basin.

The station names denoted by a kilometre distance from the dam wall were the sites most frequently sampled. As previously mentioned, these sites lie in the old river bed. Given this, it should be noted that these sites represent the maximum water depth at a given cross-sectional location in the reservoir. In most cases, the majority of the reservoir area surrounding these sites contains a water depth 4-6 m less than the sample site.

The distribution system was also sampled on some occasions.

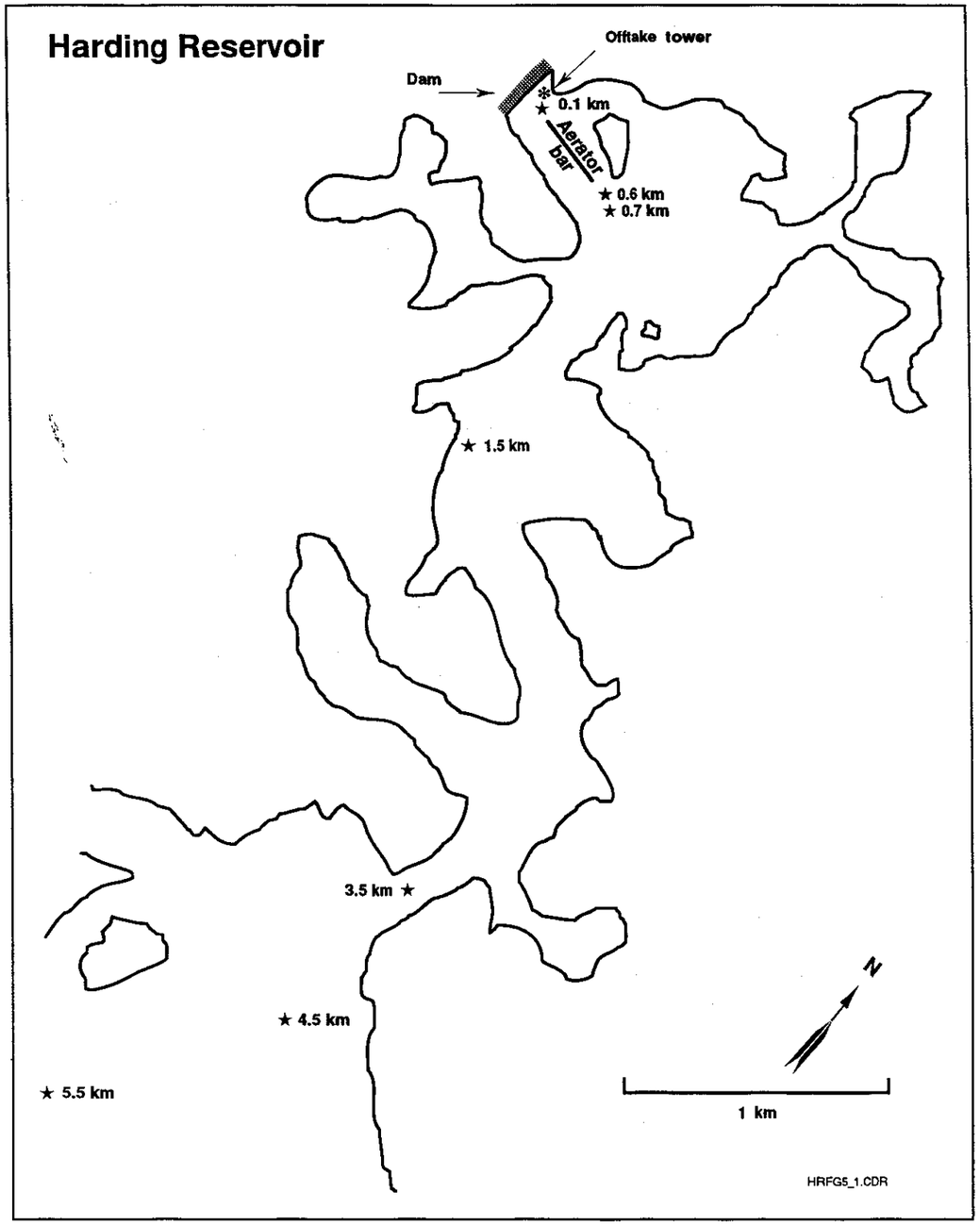


Figure 5.1 Harding Reservoir sampling locations

5.2 Sampling Frequency

Sampling frequency is always a compromise between conflicting demands:

- rapid changes in all variables during storm or overturn events;
- major changes in population characteristics of the biota (which can occur on time scales of hours or more) and
- the high costs of resources to sample frequently.

Regular water quality samples for this study were taken every 3 weeks from January 1988 to May 1990. This baseline program involved sampling for a comprehensive suite of biological, chemical and physical water quality variables (Section 5.3). Additional data were obtained from occasional diurnal sampling during 1988, sampling for sediments, sampling during aerator trials (September 1989), turbidity sampling following an overturn event (April 1990), and from samples taken during other special events.

5.3 Field Measurements, Sampling and Analysis Methods for Water Samples

The measurements and analyses made are outlined in Table 5.2

5.3.1 Sample collection and storage

All samples for analysis were collected in plastic or glass bottles; details are given in Table 5.3.

Samples for nutrient analyses were taken in new plastic bottles which had been soaked in 10 % HCl and rinsed several times with de-ionised water. Except for bacteria and pesticide sample bottles, all other bottles were soaked in 10 % Decon and rinsed several times with de-ionised water prior to use.

Most sample bottles were filled in the field after rinsing the collection vessel several times with the sample. Note, however, that bacteria, chlorophyll *a* and pesticide sample bottles were not rinsed but each was carefully filled without overflow. Bottles used for chlorophyll *a* sampling contained 1 mL of a calcium carbonate suspension to prevent any reduction in pH.

All samples collected in the field were immediately placed in ice-cooled insulated containers. Analyses normally began the day after sample collection. Samples were stored only for short periods (< 5 days) and always under refrigeration at approximately 4°C.

A Van Dorn type sampler was used for collecting water samples. Samples collected in this way were transferred to the appropriate bottles (see Table 5.3).

Samples were collected at different depths. At the deeper sites (up to 4.5 km from the dam) the number of samples collected varied depending on whether or not the reservoir was stratified.

Under stratified conditions four samples were taken:

- surface;
- above and below the thermocline and
- a bottom sample about 1 m above the bottom, taking care to avoid disturbing the sediments.

When the reservoir was destratified three samples were collected:

- surface;
- middle and
- bottom.

At the shallower 5.5 km site, only surface and bottom samples were collected.

Table 5.2 Field measurements and analyses made during the study

Variable	Details	Reference
turbidity	Lange laboratory turbidity meter	model LTP4
colour	spectrophotometer measurement at 400 nm	Varian DMS 100
temperature	calibrated alcohol-in-glass (0.50°C) and YSI Dissolved Oxygen Meter	
pH	Orion portable pH meter and Radiometer laboratory meter	model 250 model PHM 85
conductivity	TPS portable meter and Radiometer laboratory meter	model CDM 83
dissolved oxygen	YSI dissolved oxygen meter	model YSI-51
sodium potassium calcium magnesium iron manganese	atomic absorption spectroscopy	Varian Spectra AA40
alkalinity	acid titration with double pH endpoint	Radiometer autotitrator (modified)
chloride sulfate	ion-exchange with conductivity detector	Stainton *et al*
filterable reactive phosphorus (FRP)	colorimetric, molybdenum blue complex	Eisenreich, S J *et al* (1975)
total phosphorus (TP)	acid-persulphate digestion - molybdenum blue complex	Eisenreich, S J *et al* (1975)
total Kjeldahl nitrogen (TKN)	classical digestion method followed by automated distillation and acid titration with pH meter	APHA (1980) digestion Gerhardt distillation unit
ammonia-N	colorimetric, phenate hypochlorite auto-analyser method	Technicon manual
nitrite-N	cadmium reduction followed by sulfanilamide NED method for nitrite	Technicon manual
nitrite-N	as above, without the cadmium reduction	Technicon manual
pesticides	GLC methods	Chemistry Centre of WA
filterable organic carbon (FOC)	UV oxidation with conductivity detection	R S Rosich (*pers comm*)

Table 5.3 Sampling containers used

Size	Description and purpose
1 x 1 L	plastic bottle for TKN, TP
1 x 500 mL	plastic bottle for alkalinity, colour, turbidity and conductivity
1 x 500 mL	plastic bottle for chlorophyll *a*
1 x 100 mL	plastic bottle for NH_3-N, NO_3+NO_2-N and FRP. The sample was filtered in the field
1 x 100 mL	glass bottle for FOC, filtered in the field
1 x 100 mL	glass for bacteria
1 x 50 mL	plastic bottle for metals
1 x 2.5 L	glass bottle for pesticides
1 x 100 mL	plastic jar for zooplankton
1 x 500 g	screw-top plastic jar for sediment collection for analysis of chemical parameters
1 x 500 g	screw-top glass jar for pesticide analysis of sediment
1 x bag	plastic bag for pesticide analysis of vegetation

5.3.2 Field measurements

Dissolved oxygen was measured with a YSI Model 58 dissolved oxygen meter with the facility to record both mg L^{-1} and % saturation. Calibration of the meter was performed at the commencement of sampling. Field pH was recorded with an Orion Model 250 pH meter. Calibration of the pH meter was carried out, before each use, with two buffers that encompassed the range of measurement. The pH of the buffers used were 4.01, 7.00 and 9.18.

Surface temperature was also measured with an alcohol-in-glass thermometer. The pH was measured as described in Section 5.3.2.

Light penetration of the water column was estimated using a Li Cor model LI-188B photometer fitted with an underwater quantum sensor (LI-199SB, 400-700 nm). Vertical attenuation coefficients were calculated from data at three depths (0.1, 1.0 and 2.0 meters).

The euphotic zone was estimated as the depth at which the irradiance was 1 % of the level immediately below the surface.

A Secchi disc (200 mm diameter with alternating black and white quadrants) was also used to estimate the euphotic zone. This estimation was based on twice the Secchi depth.

5.4 Sampling and Analysis Methods for Vegetation and Sediments

5.4.1 Sediment collection

Sediment was collected using an Eckman grab. This sampled approximately the upper 7 cm of sediment. Sufficient replicate samples were taken to collect 10 L of sediment which was bulked and mixed in a 20 L

plastic bin. This bin was transferred to the laboratory and stored at 4°C. The sediment was re-mixed prior to sub-sampling for the establishment of experimental columns.

5.4.2 Composition of sediments

Samples were analysed for the following constituents:

- moisture content
- total P
- Phosphorous fractionation
- Kjeldahl N
- organic matter
- carbonate
- Fe, Mn, Ca, Mg, Zn, Pb and Cu.

Moisture content

A portion of the sediment was dried in an oven at 105°C overnight and then ground in a mortar and pestle before further analysis.

Digestion

Samples of approximately 0.3 g of dried sediment were digested in 75 mL digestion tubes with a 5:1 nitric acid and perchloric acid mixture. After digestion was complete (white fumes of perchloric acid) the tubes were cooled and then rinsed with 10 mL of 2M HCl into 100 mL volumetric flasks. The tubes were further rinsed with de-ionised water into the flasks and made up to volume.

Blanks and standards were taken through the complete procedures (for both phosphorus and metals) and good recoveries (95-105%) obtained for the standards.

Total Phosphorus

2 mL aliquots of the digested samples were transferred to a 75 mL digestion tube and heated to just dryness at 150°C to remove HCl. The residue was dissolved with 25 mL water and the orthophosphate analysed as the molybdenum blue complex (see Table 5.2).

Phosphorous Fractionation

Sediment samples were dried at 105°C for 24 hours and ground to a fine powder. Portions of these samples were then analysed following procedures outlined in Williams *et al* (1980). NaOH-extractable and HCl-extractable phosphorous fractions were analysed from 100 mg of the ground sediment The filterable reactive phosphorous (FRP, often referred to as orthophosphate) obtained from these extractions was determined as described in Table 5.2.

Kjeldahl nitrogen

Total Kjeldahl nitrogen was determined on the dry sediment (approximately 0.3 g) using the method as outlined in Table 5.2.

Organic matter

Dry sediment (about 1 g) was ignited at 550°C in a muffle furnace for one hour and the mass loss reported as organic matter.

Dean (1974) reported that this procedure gives an accurate measure of the organic carbon content of lake sediments and compares favourably with instrumented C-H-N analysers which measure the CO_2 evolved on combustion. In this study we used the conversion factor he determined:

$$Organic\ C = \frac{(organic\ matter)}{2.13 \pm 0.4}$$

Carbonate

The residue from the organic matter determination above was then ignited at 950 C for one hour and the additional mass loss gives an estimate of the amount of carbonate (as mg CO_2 g^{-1} of dry sediment) in the samples.

Note: to convert this ignition loss to equivalent $CaCO_3$ content of sediment the conversion factor is:

$$mg\ CaCO_3\ g^{-1} = (550 - 950°C\ ignition\ loss) \times 2.27$$

Metals

The nitric-perchloric acid digests were analysed for Fe, Mn, Ca, Mg and Zn by standard flame atomic absorption spectrophotometric technique (Table 5.2).

Cu and Pb were analysed using the graphite furnace atomic absorption spectrophotometric technique (Table 5.2).

5.4.3 Experimental columns

To construct the experimental sediment-water columns, 5 cm of homogenised sediment was packed into the bottom of 60 cm lengths of clear PVC tubing. The sediment was then covered with a 50 cm column of freshly sampled Harding Reservoir surface water, taking care to cause minimal disturbance of the sediment.

The end of each column was sealed with a rubber bung. The PVC used was approximately 9.3 cm internal diameter, giving an initial water column volume of 3.4 L and a sediment surface area of 68 cm^2. Columns were kept wrapped in black plastic to minimise algae growth and maintained in a temperature controlled room at 20°C ± 2°C.

Reconstituted, rather than intact, sediment-water columns was a compromise used in this study. While intact cores allow maintenance of the sediment microstructure and are a closer representation of the sediment-water interface *in situ*, they are difficult to collect and transport, especially from remote locations. Also, they do not eliminate the effects of lateral variation of the sediment surface.

The necessary components of reconstituted columns are much more easily collected. As well as this advantage, the homogenisation of the sediment helps ensure comparability between columns which have different conditions and controls.

Overall, reconstituted columns are most appropriate when a number of different conditions are being tested. For the regular monitoring of a sediment-water interface and associated release/uptake patterns, stronger arguments would exist for the use of intact cores.

Aeration was provided by aquarium air pumps delivering air through plastic tubing to an air stone held between 10 cm and 15 cm above the sediment surface.

Carbon additions were achieved using analytical grade sucrose.

Nitrate additions utilised calcium-nitrate tetrahydrate ($Ca(NO_3)_2.4H_20$).

5.4.4 Sampling procedures

Integrated water samples were removed from the columns using 4 evenly spaced rubber septa in the wall of the column tube. Using pre-washed syringes 50 mL of sample was removed from each column on sampling occasions. Water sampling was carried out after physical measurements [dissolved oxygen, pH and platinum electrode potential (PEP)]. These all involved a degree of water column mixing and aided collection of a representative, integrated water sample.

Dissolved oxygen (DO) was measured using a YSI model 58 meter. The probe of this unit was lowered ten times into the water column to promote mixing without resuspension, before the final DO concentration was recorded 5 cm above the sediment surface. Thus the DO concentration recorded was a mean value of that in the water column.

PEP was measured using the apparatus shown in Figure 5.2. The bright platinum electrode was inserted 5 mm into the sediment surface. The PEP was recorded when the shift is millivolt readings was less than 1.0 mV in ten seconds. This stability was usually reached in 3-5 minutes. Replicate readings which varied by more than 20 mV were repeated. Prior to each sampling occasion, the unit was calibrated in Zobells' solution (Zobell, 1946). Following replicate readings in each column, the electrode was washed and cleaned with an abrasive cloth to eliminate surface contamination of the platinum wire. These readings gave an indication of the redox potential of the sediment surface.

5.4.5 Macrophyte sampling

A combination of observation by boat and scuba survey of the reservoir was used to estimate the extent of area covered by macrophytes. This survey was conducted at the end of the expected maximum growth period, in April 1990.

To sample submerged aquatic vegetation scuba divers descended to the reservoir bottom, taking with them a 0.25 m^2 steel quadrat. As the divers descended the quadrat was lowered over plants occupying the area inside the quadrat. At the bottom the quadrat was checked to ensure only plants inside the area would be sampled.

Once the quadrat was in place the divers excavated all plant material and approximately the top 4-5 cm of sediment inside the quadrat. The plant material was then washed, bagged, labelled and stored on ice for transfer to the laboratory.

In the laboratory the material was sorted and weighed after drying at 70°C for 2 days.

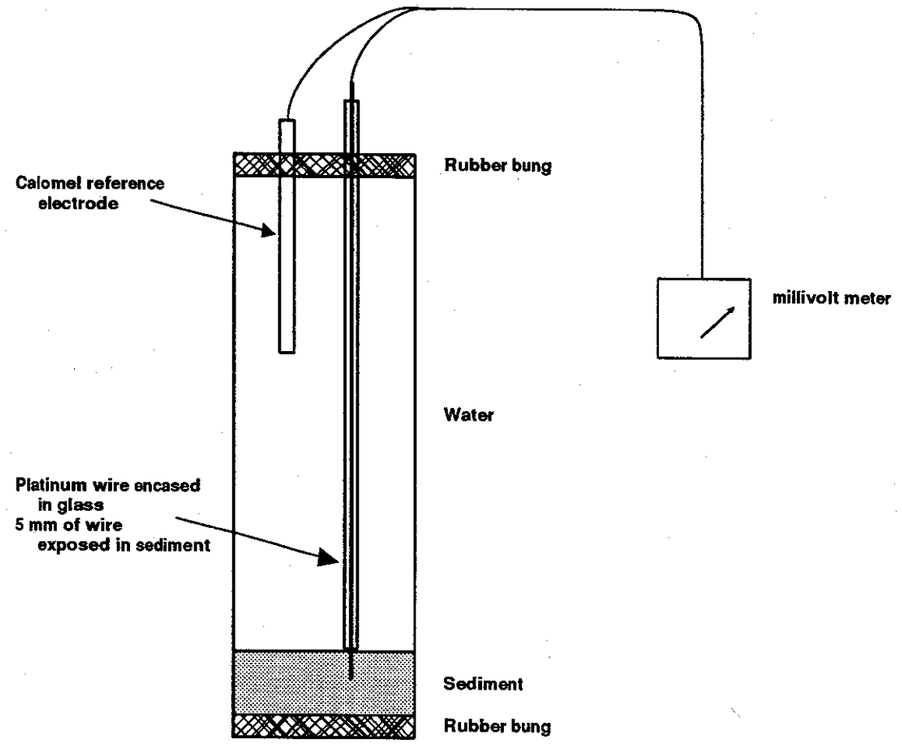


Figure 5.2 Measurement of platinum electrode potential (PEP) in experimental columns (not to scale)

5.5 Sampling and Analysis Methods for Zooplankton, Phytoplankton and Chlorophyll *a*

5.5.1 Chlorophyll *a* and phaeophytins

Chlorophyll *a* and phaeophytins were determined fluorometrically following the procedure outlined in APHA (1985), except that 10 % dimethyl sulfoxide (DMSO) in acetone was used for extracting the pigments. Fluorescence of the resulting extract was measured using a Turner Designs Series 10 fluorometer. Acidification of the extract converted chlorophyll *a* to phaeophytin causing a reduction in fluorescence which was used to determine the concentration of phaeophytins in the extract.

5.5.2 Phytoplankton

Phytoplankton counts were performed as outlined in APHA (1985). The process involved centrifugation of a 75 mL sub-sample and transfer of the concentrate to a haemocytometer, for identification and counting under a compound microscope, at 400x magnification. In the cases of colonial and filamentous genera, individual cells were counted.

5.5.3 Zooplankton

The zooplankton counted in the study comprise two major groups, rotifers and larger zooplankton (including *copepods, cyclopoids and cladocerans*).

The method of sampling entrapped both groups. A 340 mm diameter, 100 um mesh, cone-shaped, zooplankton net was lowered to 2 m above the sediment surface. The net was then drawn to the surface, trapping zooplankton in its path. Although some degree of net clogging and net avoidance may be expected, 100 % net efficiency has been assumed for calculations in this study. The concentrated zooplankton samples were then fixed in 70 % ethanol for storage prior to identification and counting.

Both groups of zooplankton were counted and identified in the laboratory from 1-5 mL sub-samples of the fixed concentrate. For the larger zooplankton, sub-samples were transferred to a clear acrylic counting tray and identified with the aid of a dissecting microscope. Rotifers were identified and counted in a Sedgwick-Rafter cell using a compound microscope.

6. STRATIFICATION and PHYSICAL COMPONENTS

6.1 Introduction

Under natural conditions Harding Reservoir exhibits characteristics described in Chapter 3 as typical for tropical and warm temperate water bodies. These are:

- water temperatures always exceeding 4°C;
- a single generally unstratified period per year and
- natural destratification associated with the onset of a brief rainfall runoff period.

Natural destratification events in Harding Reservoir may be derived from temperature changes associated with the onset of the cooler period or from intense rainfall events leading to massive inflows. Both causes were observed during the study period. The unpredictable nature of climatic changes leads to the possibility of two separate stratification phases, during a spring-summer period, divided by cyclonic mixing. This was not observed during the study period and could be expected to occur only infrequently.

Persistent thermal stratification leads to a decrease in dissolved oxygen concentrations in the hypolimnion and thus the measurement of both the dissolved oxygen and temperature profiles gives a better indication of stratified conditions than the measurement of temperature profiles alone.

A simple indicator of stratification is given by the difference between surface and deeper water temperature and dissolved oxygen measurements. Data for the study period are summarised in Figure 6.1 and show that the periods of strongest stratification occurred during March 1989 (late summer) and September October 1989 (spring). Note that these deeper water data are for depths 9-15 m below the surface, that is **above** the aerator, rather than the bottom waters..

Breakdown of stratification occurred in February and April 1989 as a result of inflow under cyclonic conditions; minimum temperatures were not reached until June and July (mid winter, Figure 6.2). At the same time, note both the significant degree of and wide fluctuations in stratification from November 1989 onwards even with the aerator on (see also Figure 4.3).

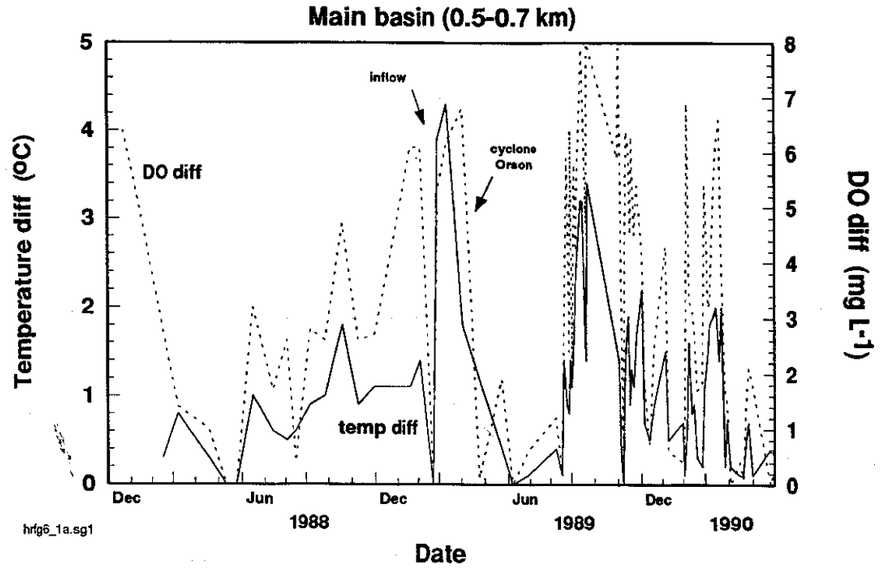
6.2 Results

6.2.1 Temperature

Surface water temperatures varied during the study period from a minimum of 17.8°C during July 1988 to a maximum of 31.1°C during March 1990. As expected, bottom temperatures were generally lower than surface temperatures. The overall patterns were similar for both the main basin and upstream locations (Figure 6.2). The greater temperature differentials in the main basin (mostly in spring and summer) reflect its greater depth relative to the upstream site. Note that the bottom waters data in Figure 6.2 are for the deepest waters at each site, and for the main basin are below the 9-15 m depths for the deeper-water data in Figure 6.1.

Under natural (non-aerated) conditions, Harding Reservoir could be expected to stratify during the summer period, and persist until March or April when major inflows (and surface cooling) would mix the water column. The column would then be expected to remain relatively uniform until around July-September when the first signs of persistent stratification would be expected to again appear. This sequence is typical of a monomictic stratification cycle, though the timing of events varies with regional climatic conditions.

Typical stratified water column temperature profiles from the Harding Reservoir are shown in Figure 6.3.



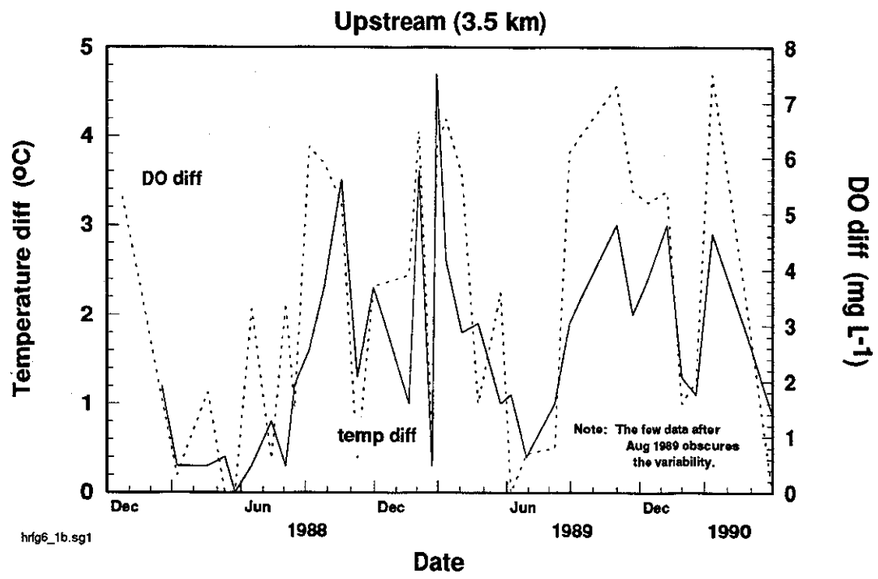
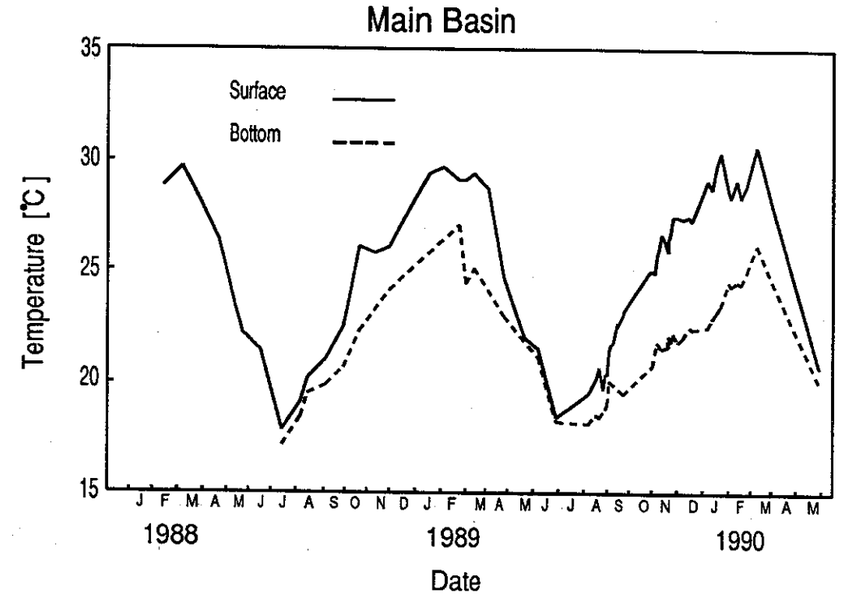


Figure 6.1 Stratification indicator for the main basin and upstream sites during the study period; the difference in temperature and dissolved oxygen concentrations between surface and deeper (at about 9-15 m below the surface, that is **above** the aerator rather than at the bottom) waters.



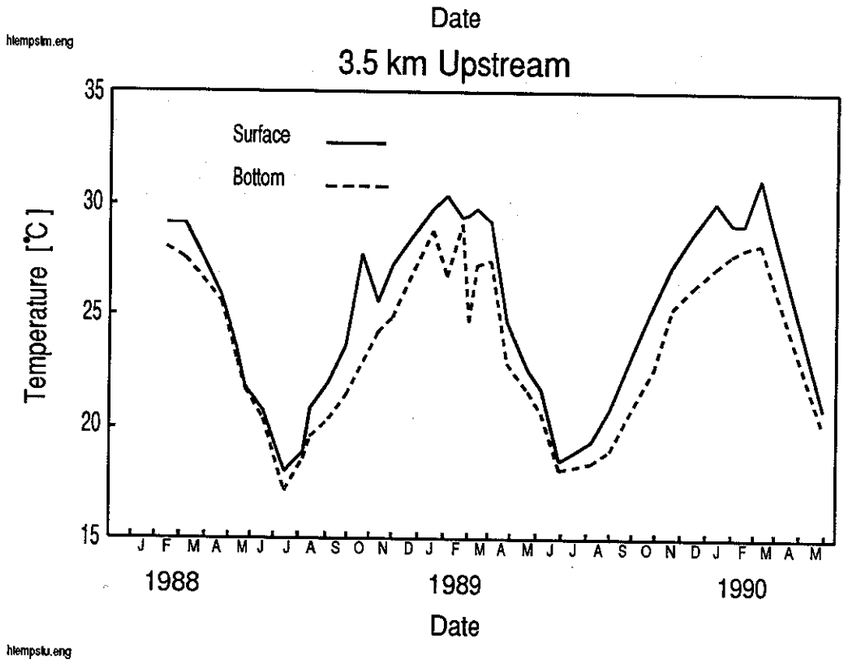


Figure 6.2 Surface and bottom temperatures in the main basin and upstream sites of Harding Reservoir, 1988-90. Note that the bottom waters data here are for the deepest waters at each site, and for the main basin are below the 9-15 m depths for the deeper-water data in Figure 6.1.

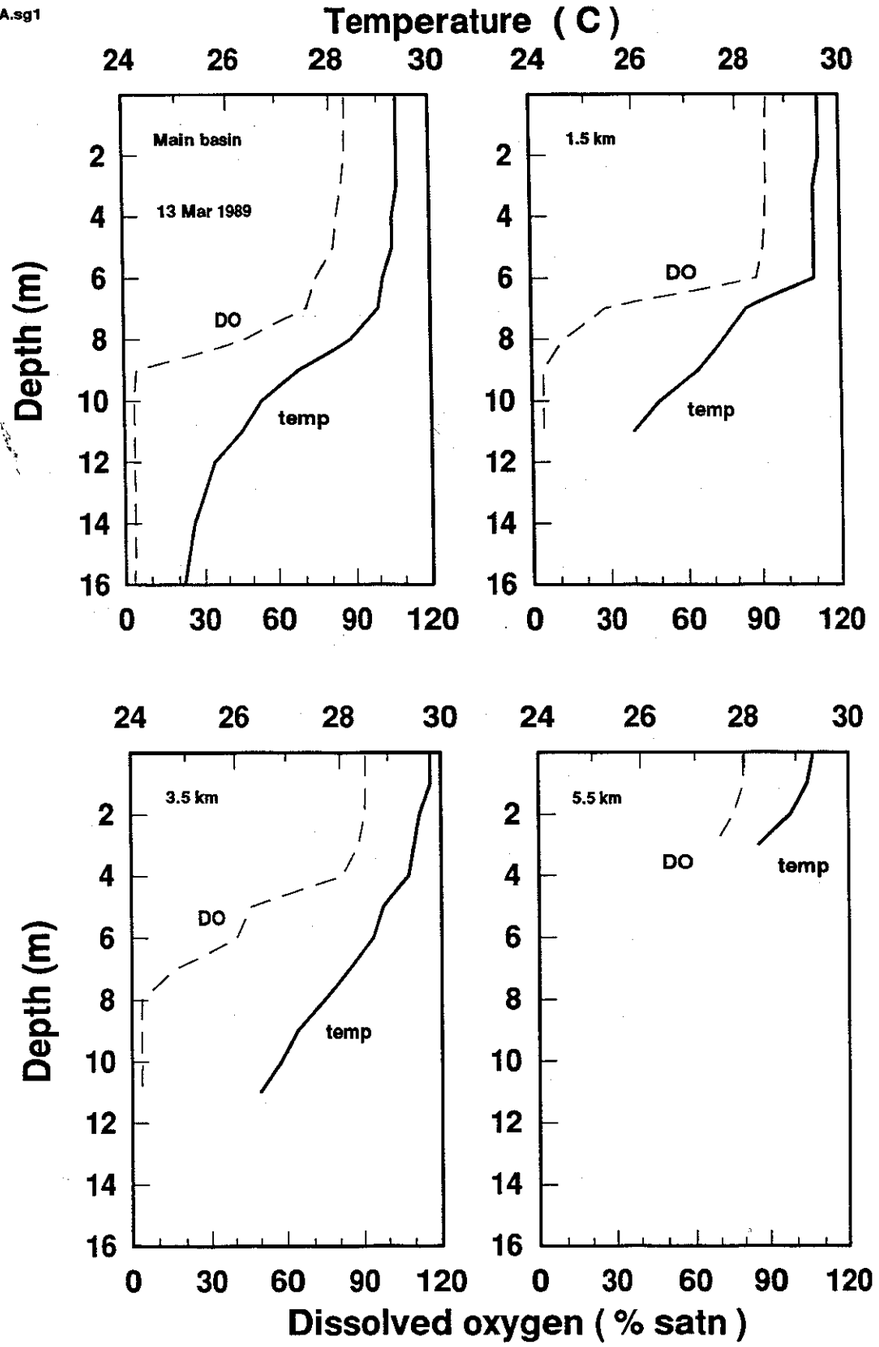


Figure 6.3 Vertical temperature and dissolved oxygen profiles for different sampling locations (0.7 km, 1.5 km, 3.5 km and 5.5 km) in Harding Reservoir during stratified conditions

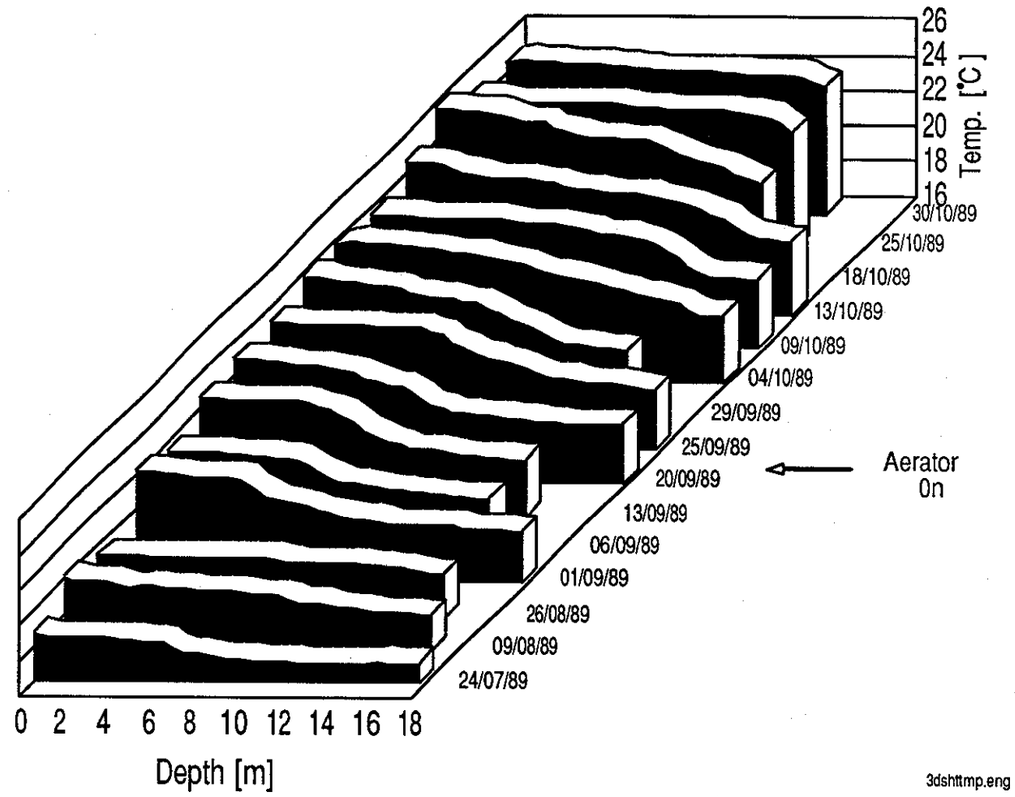


Figure 6.4 Temperature profiles showing the development of destratification as a result of aeration

As a result of aeration, the water column showed a lowering of the thermocline as surface waters become mixed with lower layers (Figure 6.4). The reduction in epilimnion temperature expected from this mixing process was more than compensated by warming due to solar radiation.

• 6.2.2 Dissolved oxygen

Variations in dissolved oxygen concentration during the study period were considerably more erratic than variations in temperature (Figure 6.5). These fluctuations are likely to be due to the effects of incomplete artificial aeration, as well as overturn. Stratification is shown by these Figures to be more persistent in the deeper main basin than the upstream area.

Dissolved oxygen concentrations in surface waters were usually maintained at or around 100 % saturated (8-10 mg L^{-1}). During stratified periods the water column typically displayed a rapid decline in dissolved oxygen concentrations between 6 and 10 m water depths (an oxycline). From this depth onwards, dissolved oxygen concentrations then usually declined more slowly to be less than 10 % saturation in the bottom water (Figure 6.6, 20-22 Sep 1989 data).

In the artificial destratification trial conducted during September and October 1989, mixing produced a deepening of the oxygenated layer towards the level of the aerator bar (Figure 6.6). Considering the guideline for the critical dissolved oxygen concentration (40 % saturation, see Section 9.3.2), prior to aeration this concentration persisted to only about 9.5 m. Five weeks after the commencement of aeration this concentration had reached about 14 m, that is nearly to the level of the aerator bar (about 14.5 m at the time).

The position of the aerator therefore determined the depth to which high oxygen concentrations were found. Water below the aerator (greater than about 14.5 m deep at the time) remained anoxic with less than 10 % dissolved oxygen saturation. This is further illustrated in Figure 6.7, which shows a deepening of the oxycline after commencement of aeration.

Under naturally occurring destratification, when cooler surface waters resulted in overturn of the water body, oxygenated water reached the sediment surface while the deoxygenated bottom waters mixed with the surface layers. When fully mixed in winter conditions, the dissolved oxygen concentrations were found to be close to saturation throughout the water column. The effects of this mixing are further shown by the chemical data (Chapter 7).

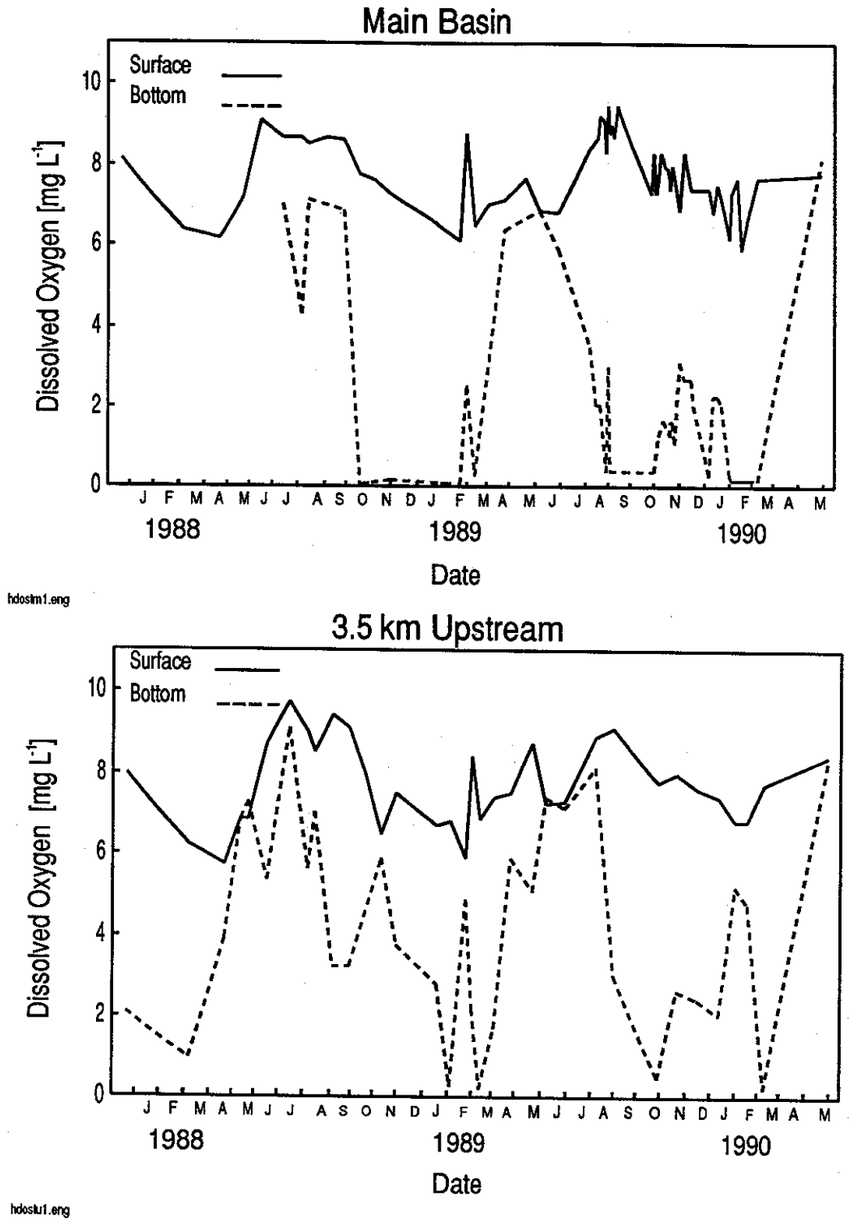


Figure 6.5 Surface and bottom dissolved oxygen concentrations for the main basin and upstream sites in Harding Reservoir, 1988-90

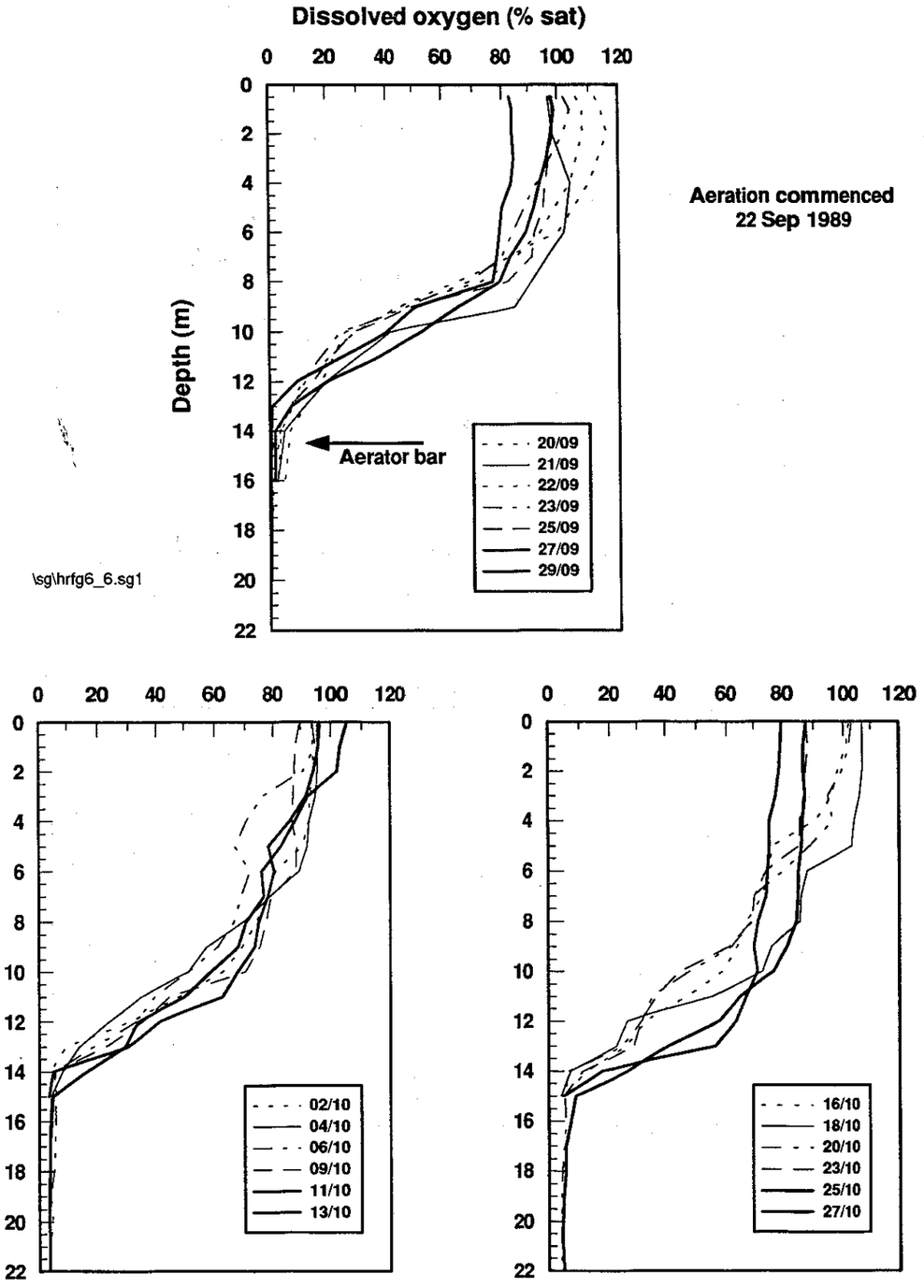


Figure 6.6 Sept-Oct 1989 aeration experiment - vertical dissolved oxygen profiles at the 0.7 km station

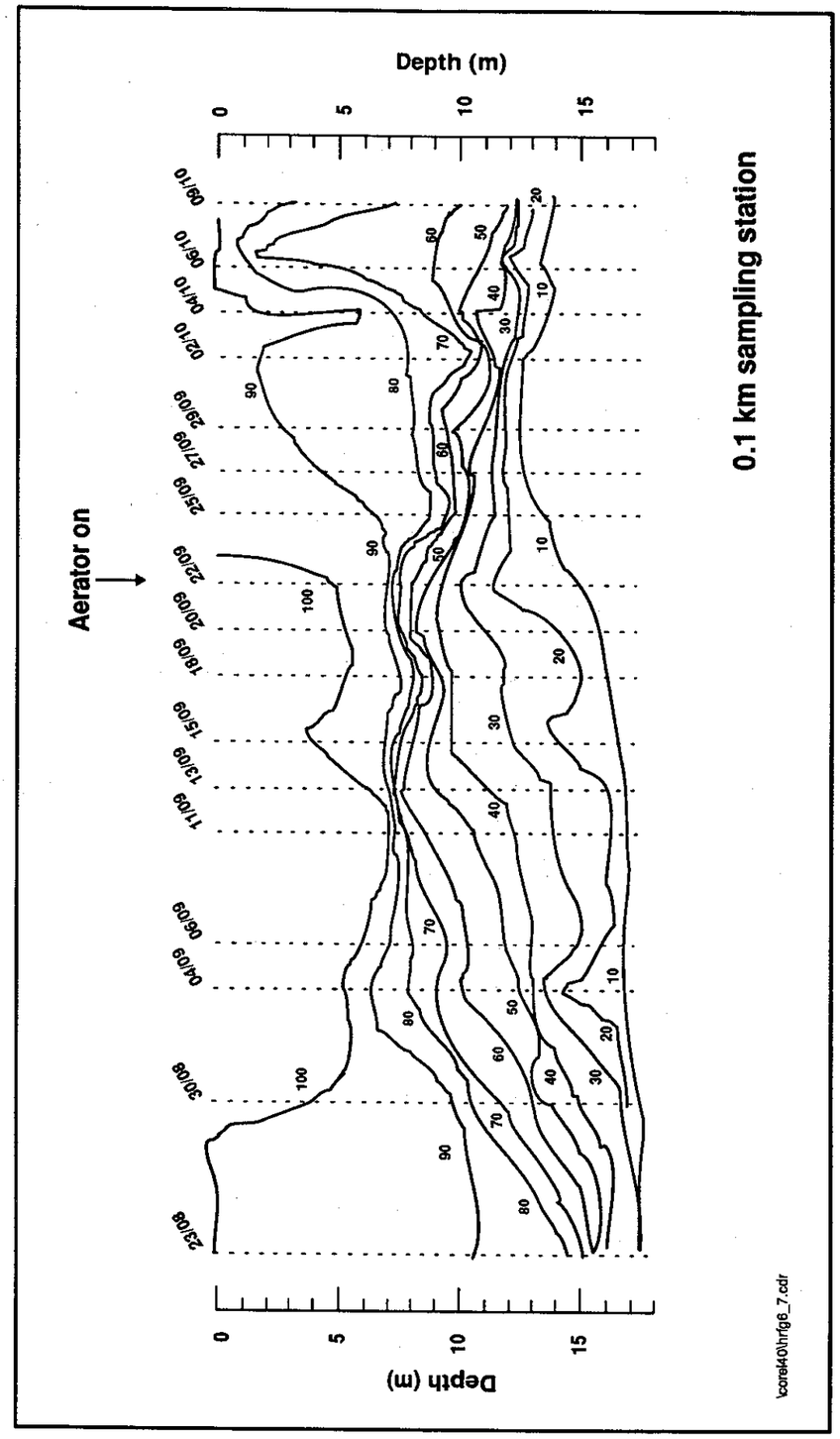


Figure 6.7 Dissolved oxygen (% sat) isopleths in Harding Reservoir before and after the commencement of aeration, 22 September 1989

6.2.3 Turbidity

Many areas of Australia commonly have turbid natural waters, because of the widespread occurrence of clays and highly weathered arid regions.

Turbidity of water is a measure of the particulate matter in suspension which causes a reduction in light penetration due to scattering. Turbidity is a (twofold) problem for potable water supplies. Firstly, visible turbidity raises concern among consumers and secondly as turbidity increases the efficiency of bacterial disinfection by chlorine decreases.

In water bodies such as Harding Reservoir turbidity may be caused by a number of factors. It may be the result of inflow of fine particulate matter from the catchment. This may either be of organic origin or inorganic clay colloidal material. Turbidity can also be due to processes within the water body, such as phytoplankton blooms. Another possible cause is the flocculation of soluble iron and manganese upon oxidation of deoxygenated waters containing the reduced metals. This may occur when overturn of stratified waters takes place and hypolimnetic water that has been in contact with the sediment surface (thus accumulating Fe^{2+} and Mn^{2+}) is circulated to the surface and becomes oxygenated.

The persistence of turbidity is affected by numerous factors, depending on the nature of the particulate matter. Clay turbidity coagulates and settles in water with increased salinity, high divalent and trivalent cation concentrations and low pH. Flocculation and settling of clays when mixed with long chain organic molecules also occurs.

On none of the **sampling occasions** during this study did the turbidity at the surface of the main basin exceed the NHMRC guideline value of 5 NTU (Figure 6.8). However, the 3.5 km upstream site did peak at 40 NTU in December 1988, and at all times was generally more turbid than the main basin (Figure 6.9).

Nevertheless, water at the Reservoir **offtake** exceeded 5 NTU on five occasions during the study period whence supply was taken from the Millstream aquifer.

Increased turbidity was evident during the cyclone seasons from December to April (for example, after cyclone Orson in April 1989), associated with influxes of cooler, turbid water from the catchment area after heavy rainfall.

During periods of thermal stratification, turbidity was higher at the bottom of the reservoir than at the surface. This is likely to be associated with several factors:

- the low dissolved oxygen concentrations giving rise to reducing conditions and hence increases in iron and manganese in the water column and
- decreased pH decreasing the rate of coagulation of colloidal and fine particulate material.

This increase in turbidity with depth is evident in Figures 6.8 and 6.9. It is further illustrated in Figure 6.10 which shows the varying turbidity at depth through the water column.

An event of increased turbidity occurred in April 1990 (that is, after the main monitoring period covered by this report) when turbidity peaked at 9 NTU at the offtake tower. As a consequence the aerator was switched off following which the turbidity decreased. A similar major increase occurred during early July 1992 when turbidity at the offtake tower exceeded 10 NTU.

The April 1990 turbidity event coincided with a deepening of the upper mixed layer (due to convective cooling) to below the level of aerator (Figure 6.11). This overturn would have resulted in the mixing of turbid and iron and manganese rich hypolimnetic water with oxygenated surface water. It is therefore considered that this rise in turbidity was associated with oxidation and coagulation of soluble iron and

manganese into particulate matter. Increased concentrations of unfiltered iron and manganese were recorded in the surface waters associated with this turbidity event.

6.2.4 Colour

Colour was generally higher in the bottom waters than at the surface, especially at greater depth in the main basin (Figure 6.8). While iron and manganese concentrations that may contribute to the colour were also higher at all times in the bottom waters, the peaks in concentration did not necessarily coincide with peaks in colour (see Chapter 7). Seasonal peaks in colour, coinciding with peaks in turbidity, may be the result of inflow of dissolved organic matter in runoff from cyclonic rainfall.

Surface water colour was generally below the NHMRC guideline value. In bottom waters this value was often exceeded, relating mainly to periods of thermal stratification during the summer months (Figure 6.1).

6.2.5 Light

Light extinction coefficients were recorded from September 1989 to July 1990 (Figure 6.12), but measured less frequently at the upstream site than at the main basin site. Nevertheless, the results show that there was generally higher light extinction in the upstream area (Figure 6.12).

There was a reduction in light penetration in the main basin (that is, an increase in light extinction coefficient) associated with the rise in turbidity noted in Section 6.2.3. This increase in turbidity persisted for three months during which time the reservoir was mostly destratified (Figure 6.1). Note, however, that the turbidity event that occurred in the main basin did not appear to occur at the upstream site.

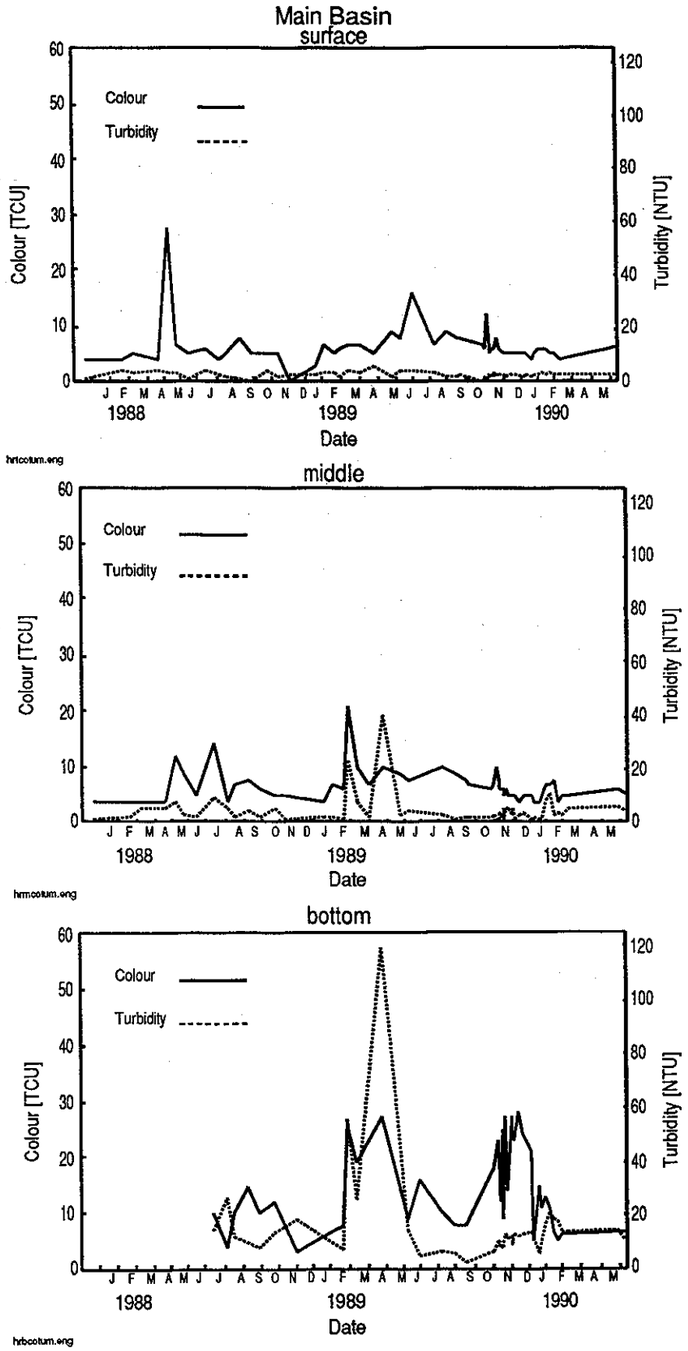


Figure 6.8 Colour and turbidity levels in the main basin of Harding Reservoir, 1988-90, at the surface, middle and bottom of the water column

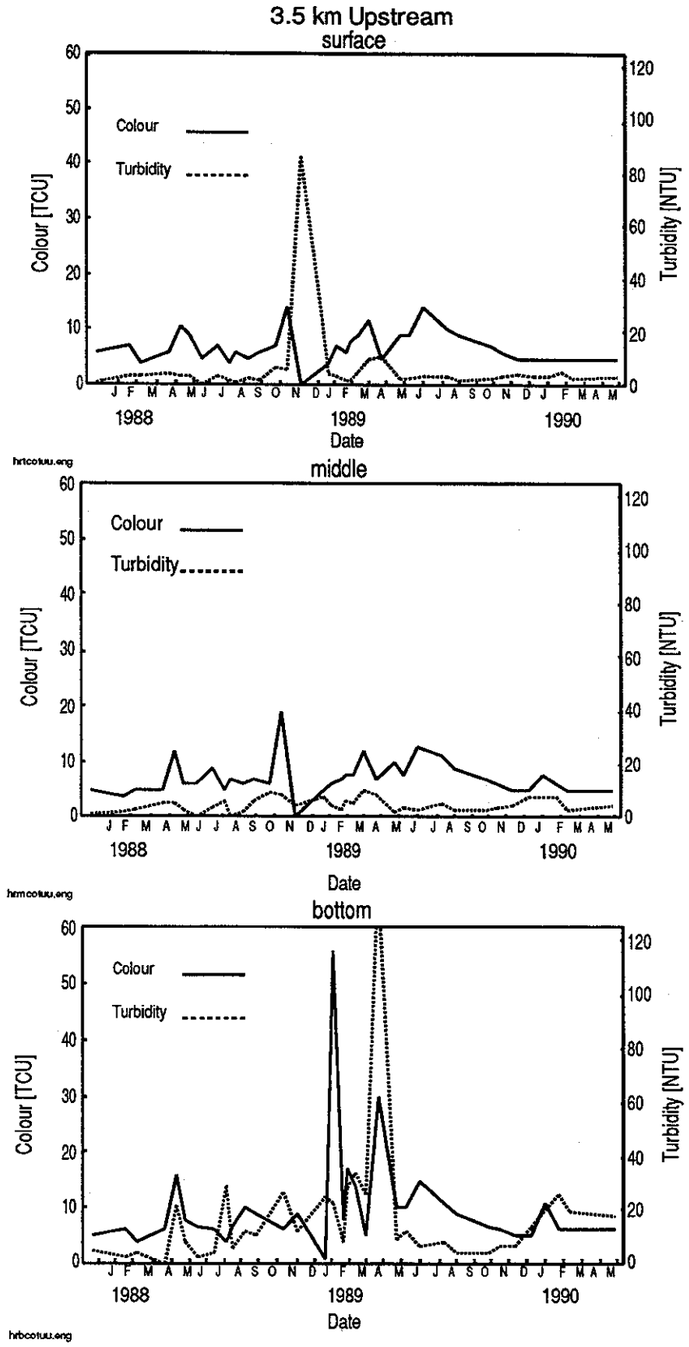


Figure 6.9 Colour and turbidity levels at the upstream site of Harding Reservoir, 1988-90, at the surface, middle and bottom of the water column

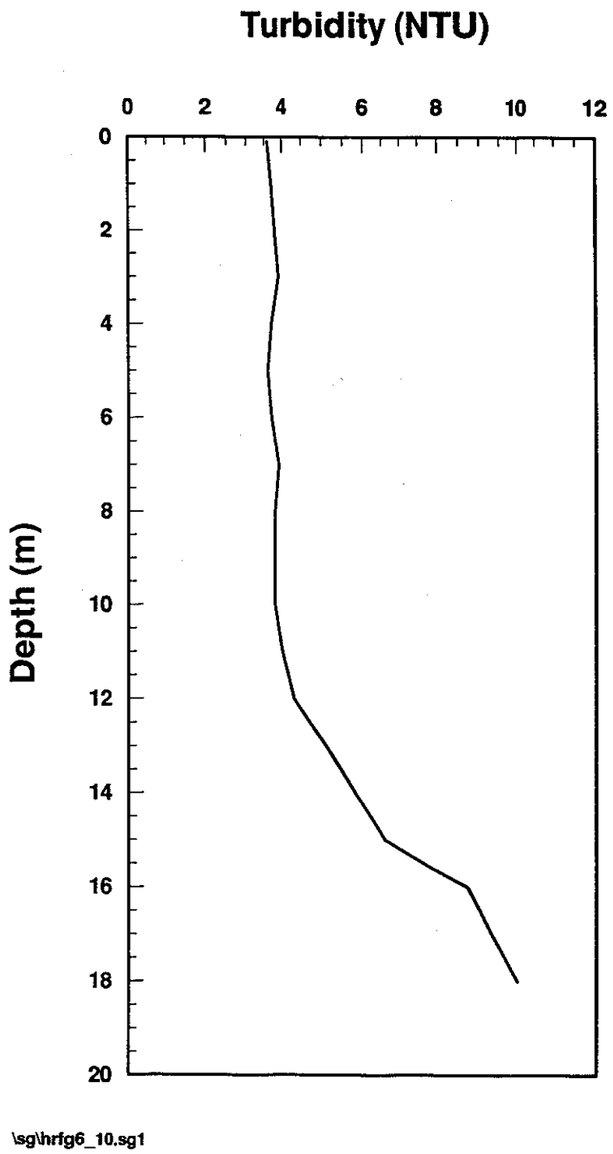


Figure 6.10 Typical vertical turbidity profile for Harding Reservoir during stratified conditions

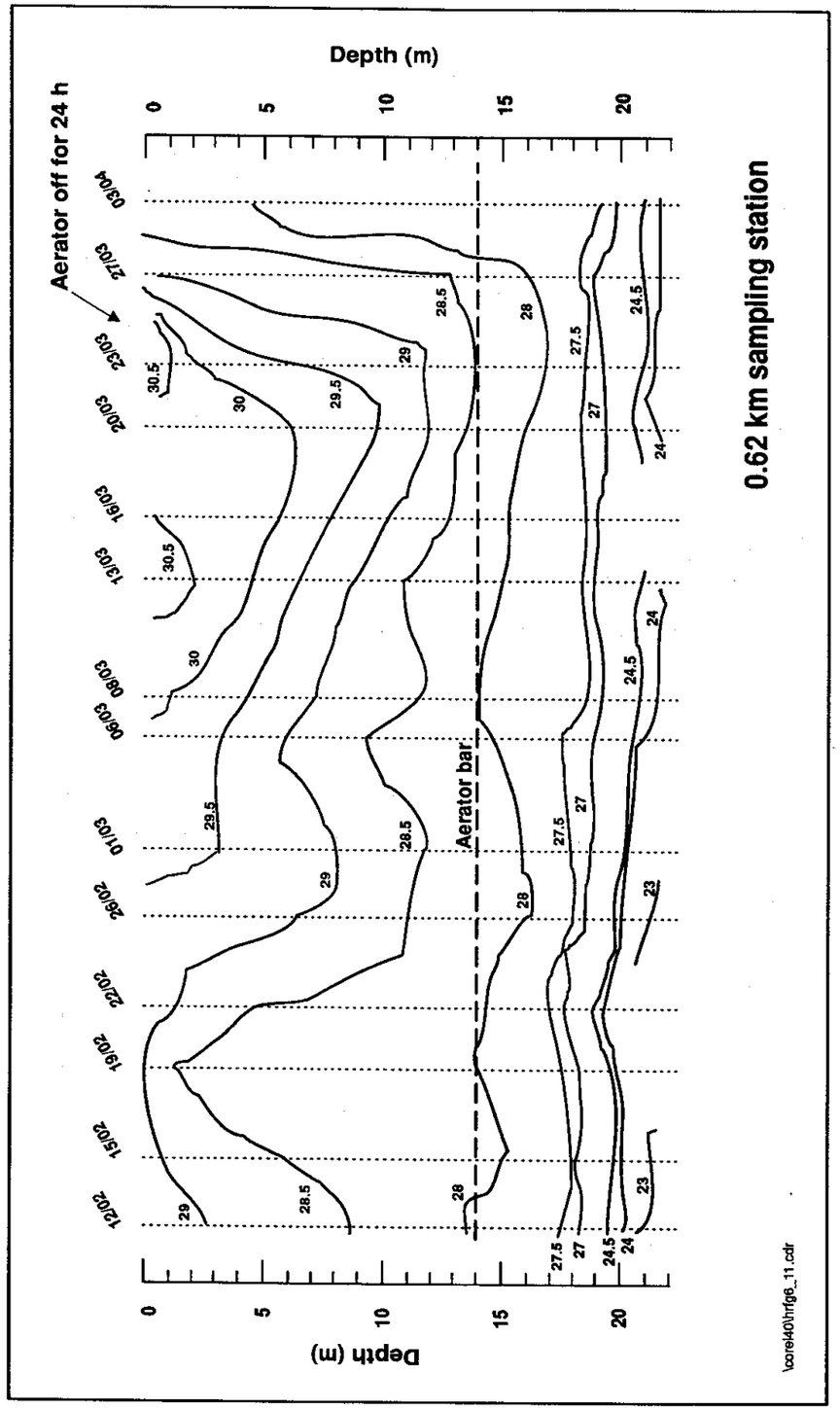


Figure 6.11 Temperature isopleths in Harding Reservoir over the period of the increased surface turbidity event, 10 April 1990

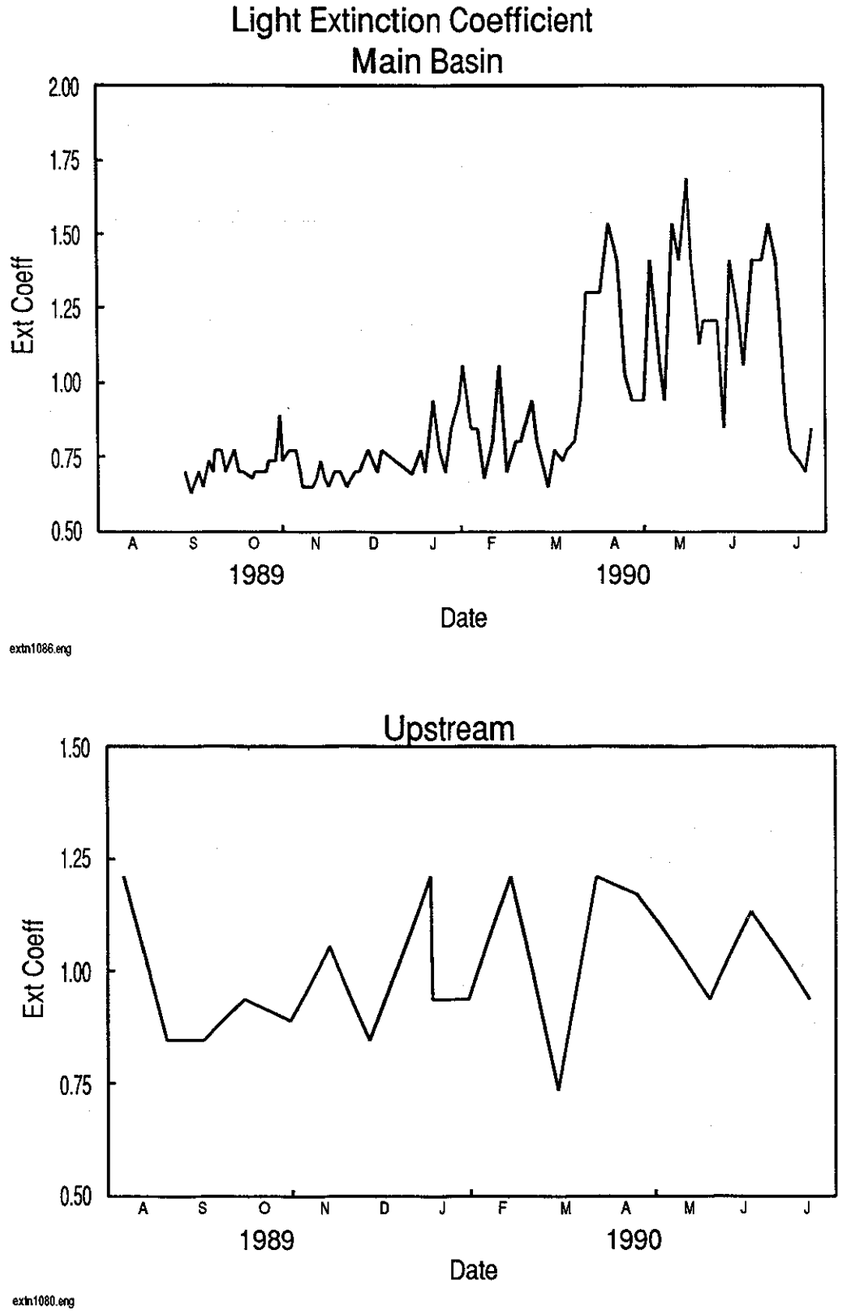
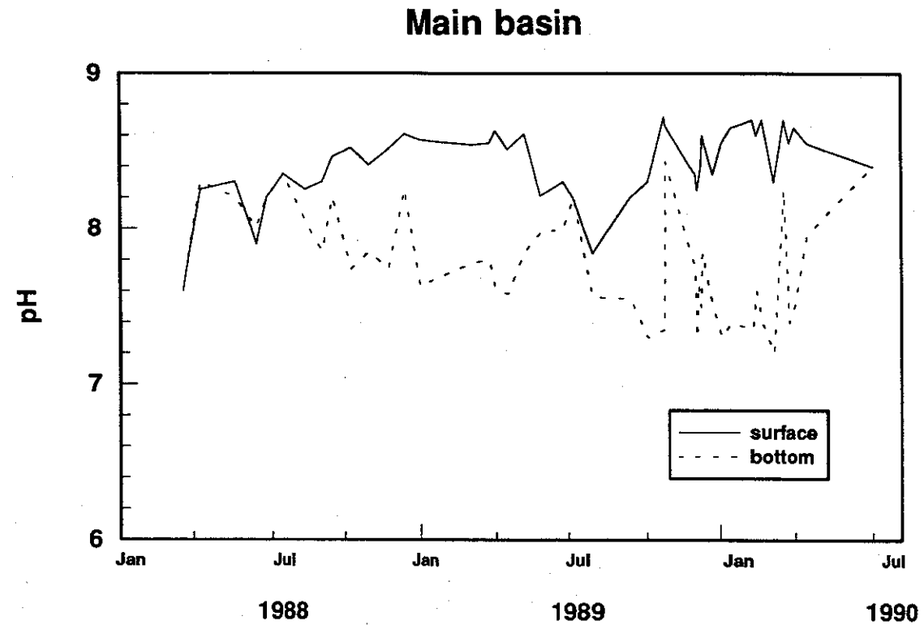


Figure 6.12 Light extinction coefficients measured during a one-year period for the main basin and upstream (3.5 km) sites



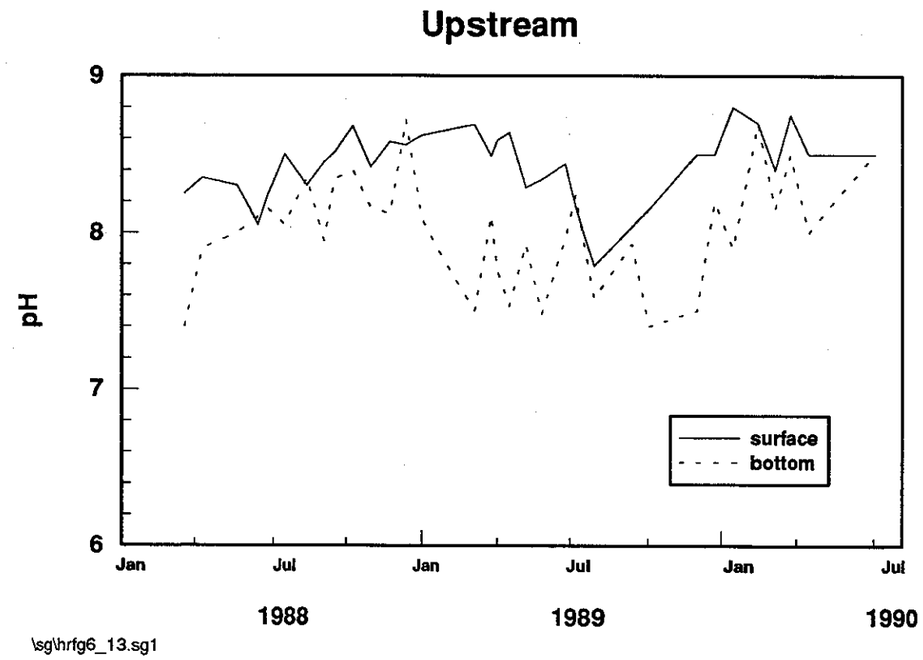


Figure 6.13 pH values in the surface and bottom waters of the main basin and upstream sites of Harding Reservoir, March 1988 - July 1990

6.2.6 pH

The pH of the water column during the sampling period is shown in Figure 6.13. Surface waters generally had higher pH values than middle and bottom waters. The difference between surface and bottom pH was highest during periods of thermal stratification (see Figure 6.1). This is likely to be due to respiration in the bottom waters increasing the concentration of carbon dioxide and thus lowering the pH, while photosynthetic activity in the surface waters consuming carbon dioxide and thus increasing the pH (see Figure 6.14). The differences between the carbon dioxide concentrations in the surface and bottom waters was less at the shallower upstream site (Figure 6.14). This is probably due to both the greater loss of carbon dioxide by diffusion from the shallower bottom waters and greater mixing of the surface and bottom waters, again because of the shallower depth.

Overall, the pH in Harding Reservoir (generally around 8) is high within the range for natural waters, reflecting moderate alkalinity levels and lower solubility of carbon dioxide at the relatively high water temperatures.

6.3 Discussion

The physical variables measured in Harding Reservoir play a pivotal role in the biological and chemical characteristics of the water body. Artificial aeration of the main basin resulted in an increase in dissolved oxygen concentrations (but not necessarily to 100 % saturation) throughout the water column above the aerator bar; but did not prevent thermal stratification from occurring, nor anaerobic conditions developing below the aerator bar.

The presence of a layer of deoxygenated water at the sediment interface (that is, below the aerator) resulted in the undesirable release from the sediments of phosphorus and reduced iron and manganese. This situation allows overturn events such as that which occurred in April 1990 to cause water quality problems in the surface waters. The lowering of the aerator bar to the bottom of the old river bed, a depth of 24 m below full supply level, could be beneficial in the future as a means of minimising the volume of the hypolimnion. Whilst it is likely that this strategy would disturb the sediments and cause an increase in turbidity, various other aerators in Australian reservoirs operate in this fashion with no persistent increase in turbidity observed. In the case of some storages in the Blue Mountains in New South Wales, the aerator effectively becomes buried during the winter 'off season'. When turned on for the first time after winter, these aerators cause a plume of sediment rich water to rise to the surface but the particulate matter resettles from the water column in a short time.

The colour and turbidity of Harding Reservoir water is an important issue, as it has been found that 'dirty water' problems in the distribution network originates from the Reservoir (Henderson *et al*, 1990).

Increased colour and turbidity due to run-off (from cyclonic rainfall) into the Reservoir is difficult to control. Whilst various options have been reviewed (Rosich and Shier, 1988), the strategy used during this study to deal with the incidences of high turbidity was to discontinue supply from Harding Reservoir and rely on the Millstream aquifer until the turbidity settled.

Increased turbidity in the hypolimnion of the reservoir when stratified is thought to be mainly due to particulate iron and manganese. Reduced iron and manganese is released from the anaerobic sediments (see Chapter 9). These are oxidised to the insoluble iron(III) and manganese(IV) forms in the hypolimnetic waters when the dissolved oxygen concentrations exceed about 2.5 mg L^{-1}. Alternatively, they are oxidised in the oxygenated epilimnetic waters, after advection or diffusion into these waters, and then settle back to the lower waters.

Turbidity is the result of both organic and inorganic particulate matter suspended in solution. A small particle will remain in suspension if the Brownian motion of the particle is sufficient to overcome gravity.

The nature of the interactions of the particles with one another is a major factor in determining particle aggregation and thus sedimentation or dispersal.

Turbidity in Harding reservoir is likely to be mainly due to clay colloidal material, predominantly smectites (Henderson *et al*, 1990). One of the fundamental properties of clays is the electrical charge (predominantly negative) on their surfaces. Smectites and other expanding-lattice clays have large surface areas and large electrical potentials (White, 1987). Because of the small size of colloidal particles the surface to volume ratio is high, and hence surface chemistry plays a part in the formation of an overall net charge on the clay particles. If the net force between two particles is attractive then aggregation will occur, but if the net force is repulsive then the particles will remain dispersed.

The clay fraction also contains accessory minerals (hydrated iron and aluminium oxides) and amorphous material which may form coatings on the clay minerals. The charge on these minerals is highly pH dependent with reversible adsorption of potential-determining H^+ ions at their surfaces.

It is possible, therefore, that the lower pH's in the hypolimnion (Figure 6.13) are responsible for increasing the electrical potential on the colloidal material, increasing the repulsive interparticulate forces and maintaining dispersal of the particles responsible for the turbidity of the water.

If this is the case, then artificial aeration of the hypolimnion **without destratification** would result in increased pH's (through the loss of carbon dioxide), hence reduced surface potentials and consequently lower turbidities in these waters and decrease the detrimental effects of overturn events. Indeed if full destratification was achieved, then 'overturn' events as such would never occur.

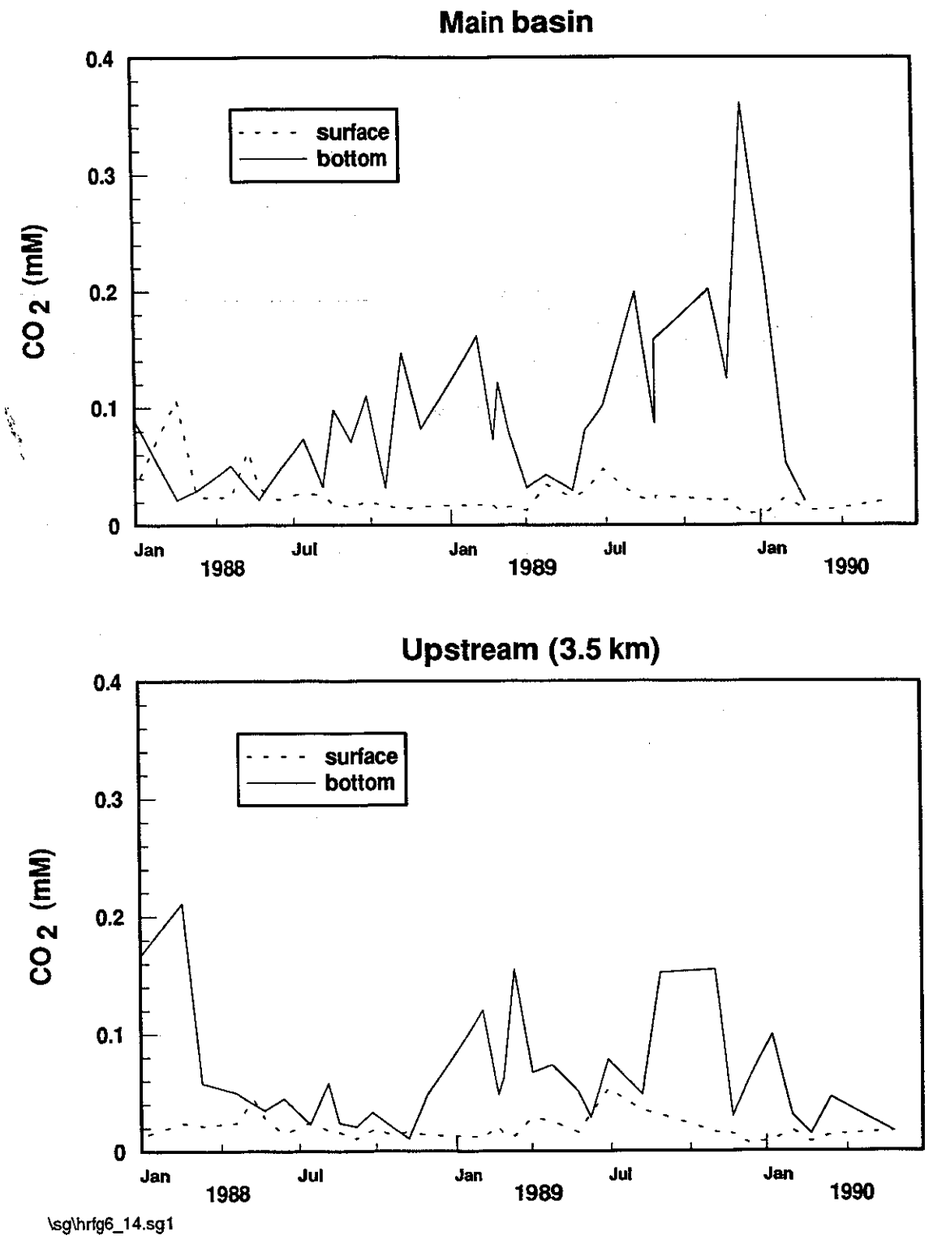


Figure 6.14 Carbon dioxide concentrations (millimole per litre) in the surface and bottom waters of the main basin and upstream sites of Harding Reservoir, Jan 1988 - Jun 1990

7. MAJOR COMPONENTS

Major components were measured as part of this study and the major variations seem to be related to evaporative concentration during the long dry spells, as expected. Only a brief outline of the data is presented.

7.1 Total Filterable Solids

Data for the study period are shown in Figures 7.1 and 2.

As discussed in Chapter 4 Harding Reservoir receives only infrequent inflow and has an estimated mean annual evaporation almost nine times the mean annual rainfall. Consequently the total filterable solids (by summation, TFSS) increases by evaporative concentration in between inflow events (see also Figure 4.3).

Note that there are wider and more rapid fluctuations in the bottom waters in response to inflow events (February and April-July 1989). This suggests that cooler, inflow waters enter the reservoir below (warmer) upper layers before complete mixing is achieved. In 1989 complete mixing appeared to occur during August.

7.2 Sodium, Calcium, Magnesium and Potassium

The first three of these (Na, Ca and Mg) generally follow the trend for TFSS, except that calcium is a greater proportion of the total after inflow events. On the other hand, potassium appears to show less variation over the same period - a possible explanation could be that the concentration of potassium is controlled by equilibrium adsorption onto the particulates in the water.

7.3 Carbonates, Chloride, Sulfate and Silica

At the pH values found in Harding Reservoir (maximum about 8.7) more than 95 % of the carbonates (shown in total in Figure 7.2 as alkalinity) are present as bicarbonate.

Both alkalinity and chloride closely follow changes in the TFSS. On the other hand, sulfate follows the trend less well and this, as suggested above for potassium, may also be due to control by equilibrium adsorption (see also Rosich *et al*, 1979).

Silica is usually found in concentrations close to the equilibrium solubility (Stumm and Morgan,1981) and here shows the opposite trend to TFSS. This presumably is a result of decreased solubility as the ionic strength increases.

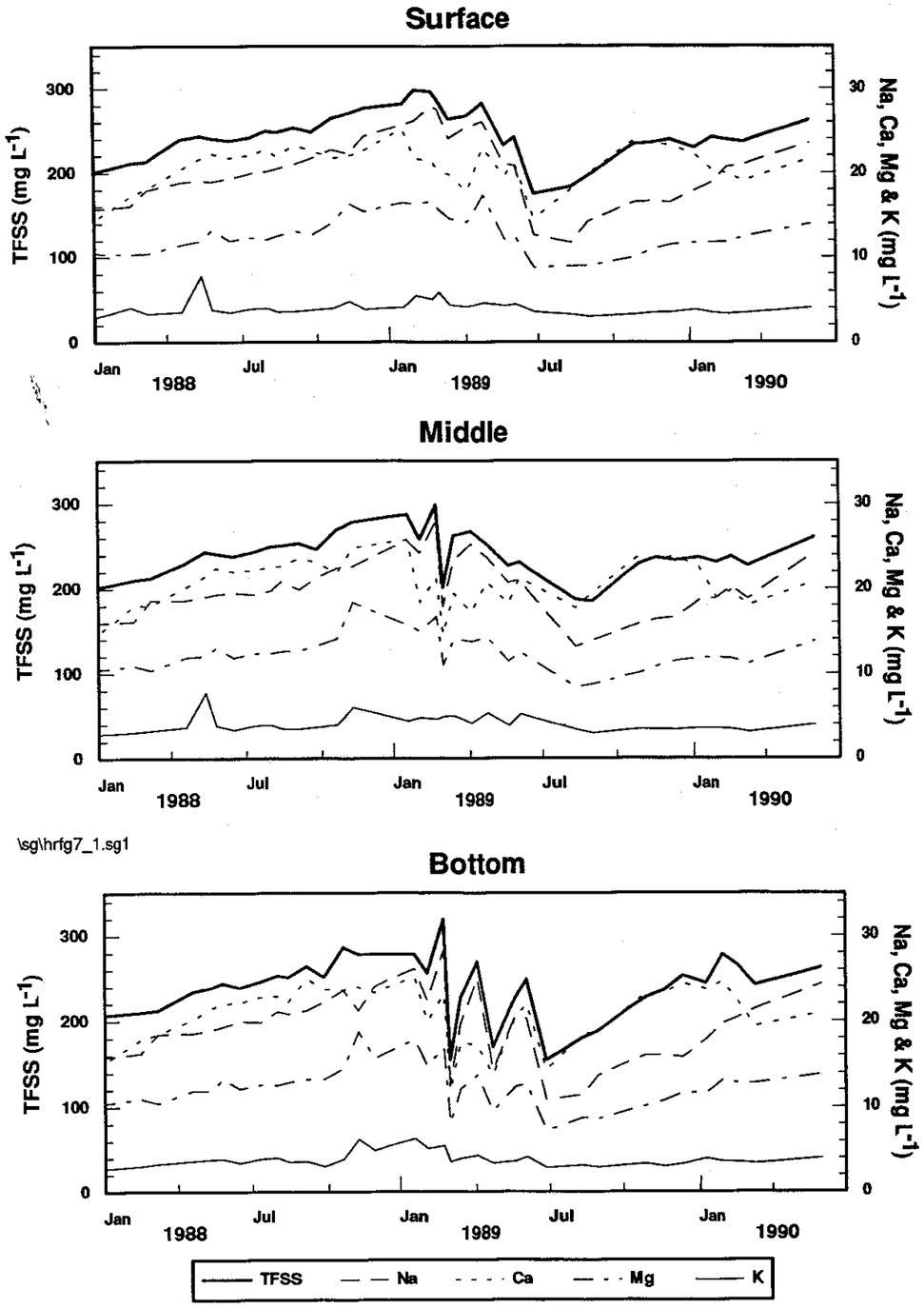


Figure 7.1 Total filterable solids by summation (TFSS) and major cations (Na^+, K^+, Ca^{2+} and Mg^{2+}) during the study period

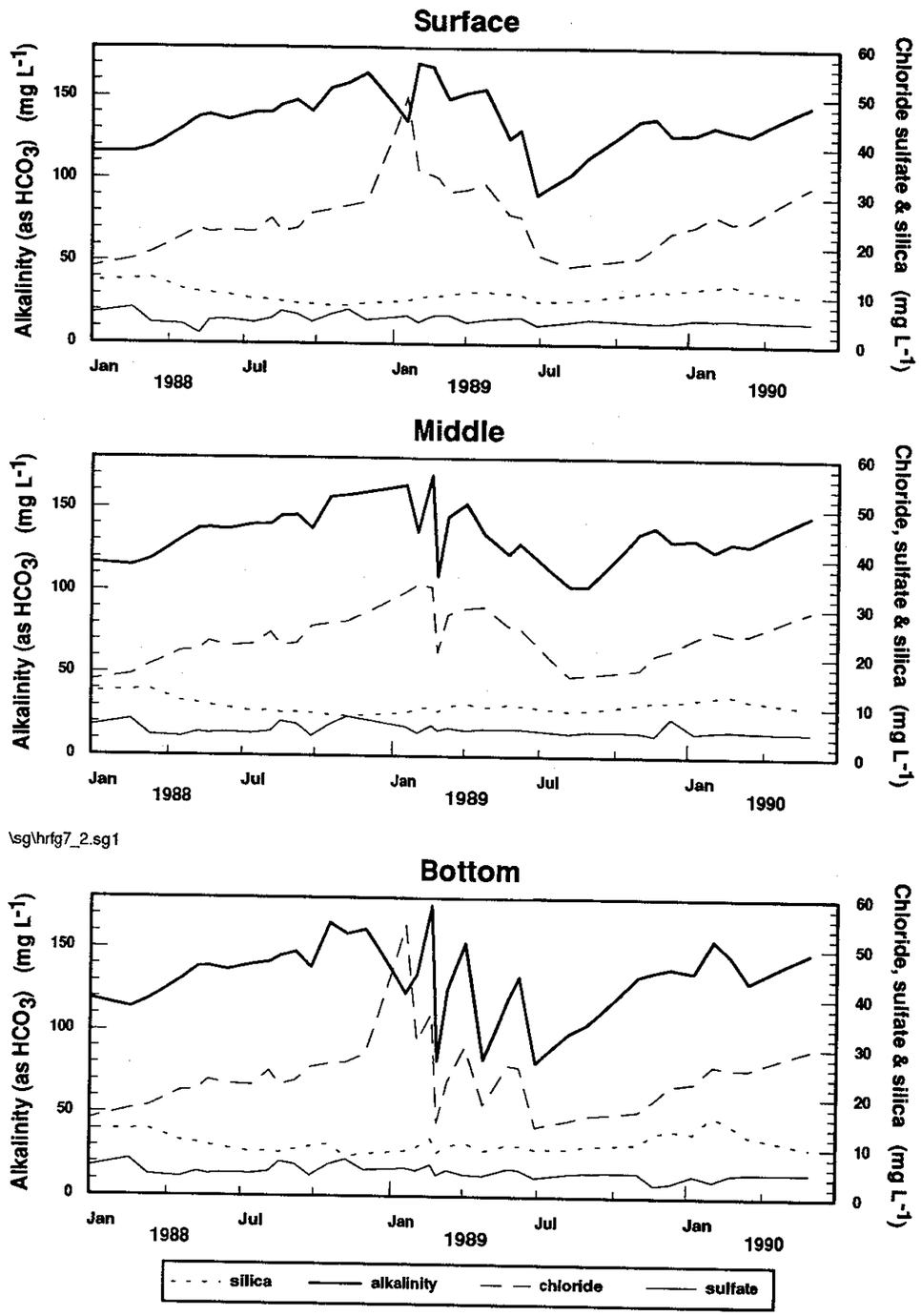


Figure 7.2 Silica and major anions (Cl^-, SO_4^{2-} and carbonates shown as alkalinity) during the study period. At the pH values found in Harding Reservoir (maximum about 8.7) more than 95 % of the carbonates are present as bicarbonate.

8. BIOLOGICAL COMPONENTS

8.1 Introduction

Biological components measured as part of this study included regular phytoplankton and zooplankton counts as well as one macrophyte survey.

Phytoplankton were counted and identified at three-weekly intervals for samples from the top 1 m of the water column at the 0.5 km and 3.5 km sampling locations. This was augmented by more intensive sampling (twice per week) at certain periods when taste and odour complaints were received from the supply. On occasions sampling also included assessment of lateral and vertical variations.

A survey of submerged aquatic vegetation was carried out in April 1990. The aim was to identify the extent of macrophyte growth in the reservoir at the end of the summer growth period. This is intended to provide baseline data on macrophyte growth, species composition, and species density against which the results of future studies could be compared.

8.2 Results

8.2.1 Phytoplankton

Harding Reservoir contains a very wide selection of phytoplankton species (see Table 8.1).

For the majority of the study period, total phytoplankton cell numbers in the main basin were less than 10,000 cells mL^{-1}. Notable deviations from this trend occurred during October to December 1988, when cell concentrations progressively reached 28,500, 45,600 and 10,400 cells mL^{-1}; and at the end of the study period in April 1990 when cell concentration rose from 6,000 to 52,000 cells mL^{-1}. Following this latter peak, concentrations again fell to less than 10,000 cells mL^{-1} within three weeks (the next sampling occasion, Figure 8.1).

The two peaks that occurred in total cell numbers were due primarily to peaks in blue-green algae numbers (Figure 8.1). As shown in Figure 8.1, groups other than the blue-green algae contributed relatively low amounts to total numbers in both absolute and relative terms.

One might expect such peaks to represent considerable increases in biomass. However, as Figure 8.3 shows, these peaks in cell numbers did not always have corresponding peaks in chlorophyll *a* concentrations (an indicator of phytoplankton biomass). The reason lies in the species composition of the cell count peaks. In the first peak (19 November 1988) 70 % of the cell numbers were of the small, colonial blue-green *Merismopedia*, while in the second peak the small colonial blue-green *Chroococcus* comprised 70 % of total cell numbers. These cells typically measure around 2 um in diameter and while they contribute significantly to cell counts, they contribute little towards biomass.

Similar peaks in surface phytoplankton cell numbers occurred at much the same time at the 3.5 km sampling location (Figures 8.1 and 8.2). Chlorophyll *a* values at the 3.5 km site (Figure 8.4) varied a little more than at the main basin site. Once again total cell numbers consisted mainly of blue-green algae (Figure 8.2). As was the case in the main basin area *Merismopedia* and *Chroococcus* made up of more than 70 % of the total cells on both occasions of peak cell numbers.

Chlorophyll *a* values were highest in the surface waters, as would be expected due to the greater availability of light for photosynthetic activity. Phaeophytin values were lower in bottom waters at both sites, probably due to the dead algae cells that fall through the water column. Phaeophytins are the

breakdown products of chlorophylls and thus increases in their levels usually result from the die-off of phytoplankton cells.

There were some recordings of alga associated with taste/odour problems, notably *Anabaena*, *Microcystis*, *Peridinium* and *Melosira*. These were not present, however, in quantities sufficient to cause water quality problems during the study.

Table 8.1 Typical suite of phytoplankton found in the main basin of Harding Reservoir

Phytoplankton	cells mL^{-1}
Chlorophyta	
Ankistrodesmus	10
Chlamydomonas	20
Coelastrum	90
Kirchneriella	90
Scenedesmus	60
Tetraedon	30
Unidentified uni-cell	100
Chrysophyta	
Melosira	110
Navicula	15
Cyanophyta	
Chroococcus	20
Coelosphaerium	90
Merismopedia	120
Unidentified uni-cell	640
Pyrrhophyta	
Peridinium	30

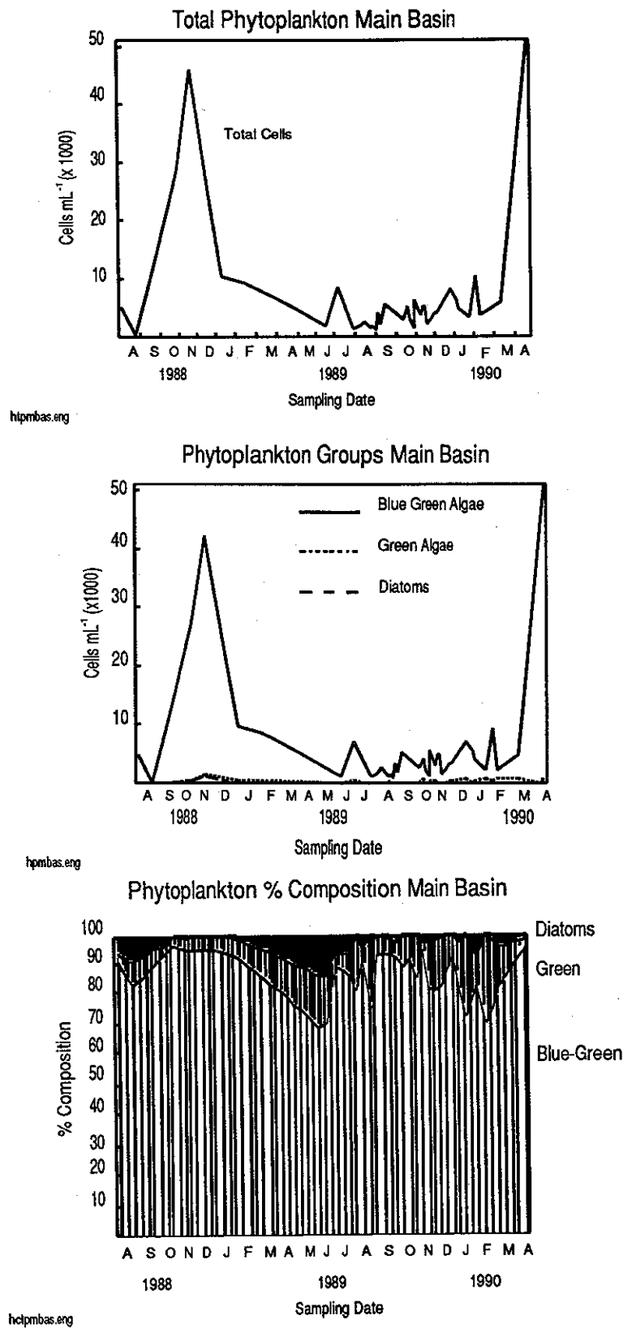
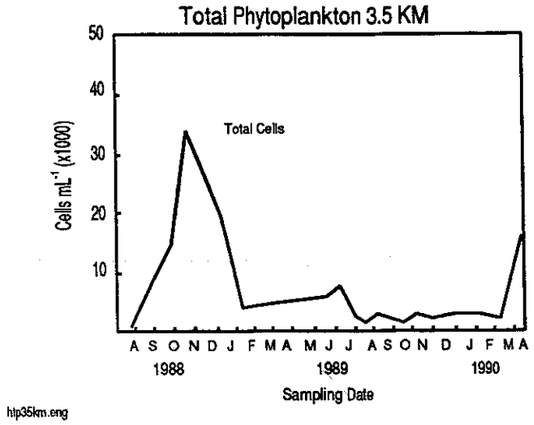


Figure 8.1 Phytoplankton counts in samples from the main basin: total cell numbers; cell numbers of major groups; and percent contribution of major groups to total numbers



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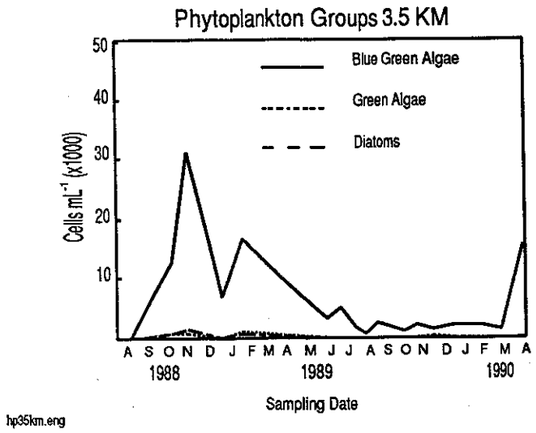

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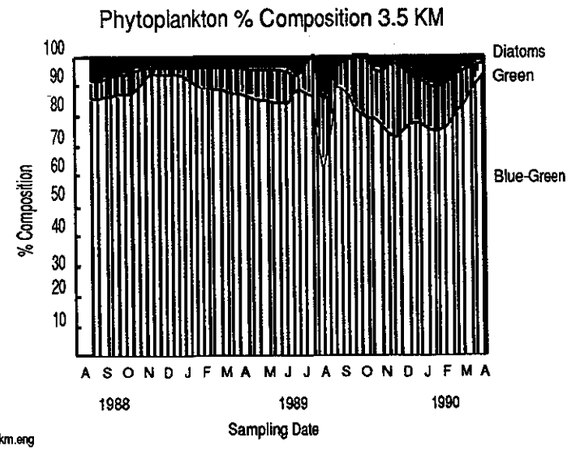

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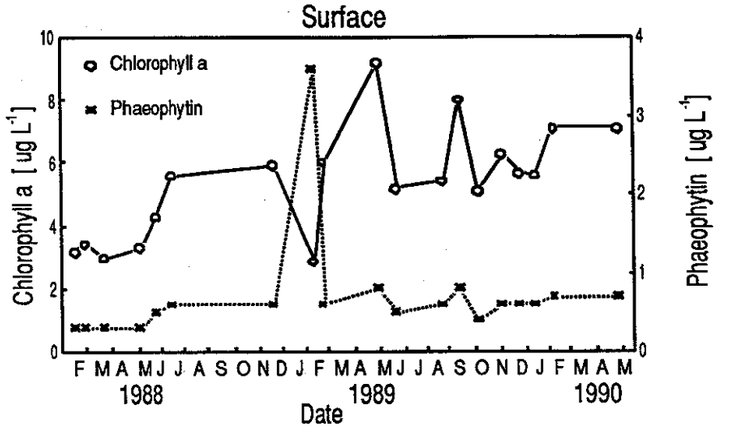
Figure 8.2 Phytoplankton counts in samples from the upstream (3.5 km) site: total cell numbers; cell numbers of major groups; and percent contribution of major groups to total numbers



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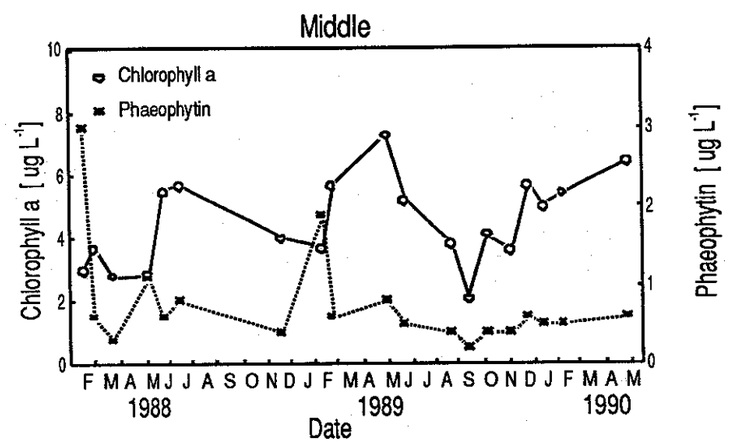

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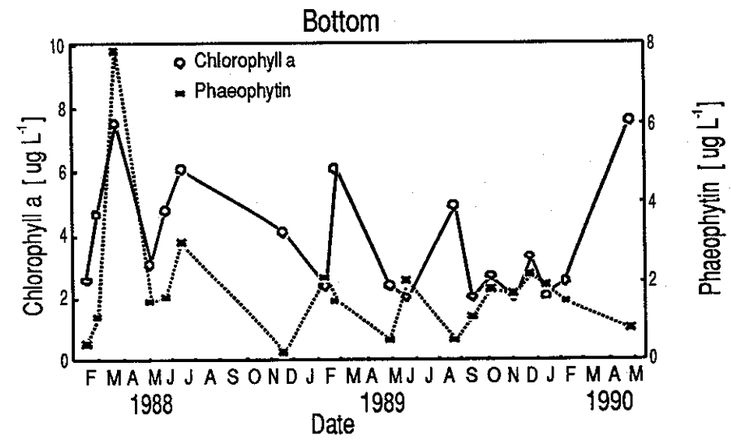

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Figure 8.3 Chlorophyll *a* and phaeophytin values for samples from the main basin at the surface, middle and bottom of the water column

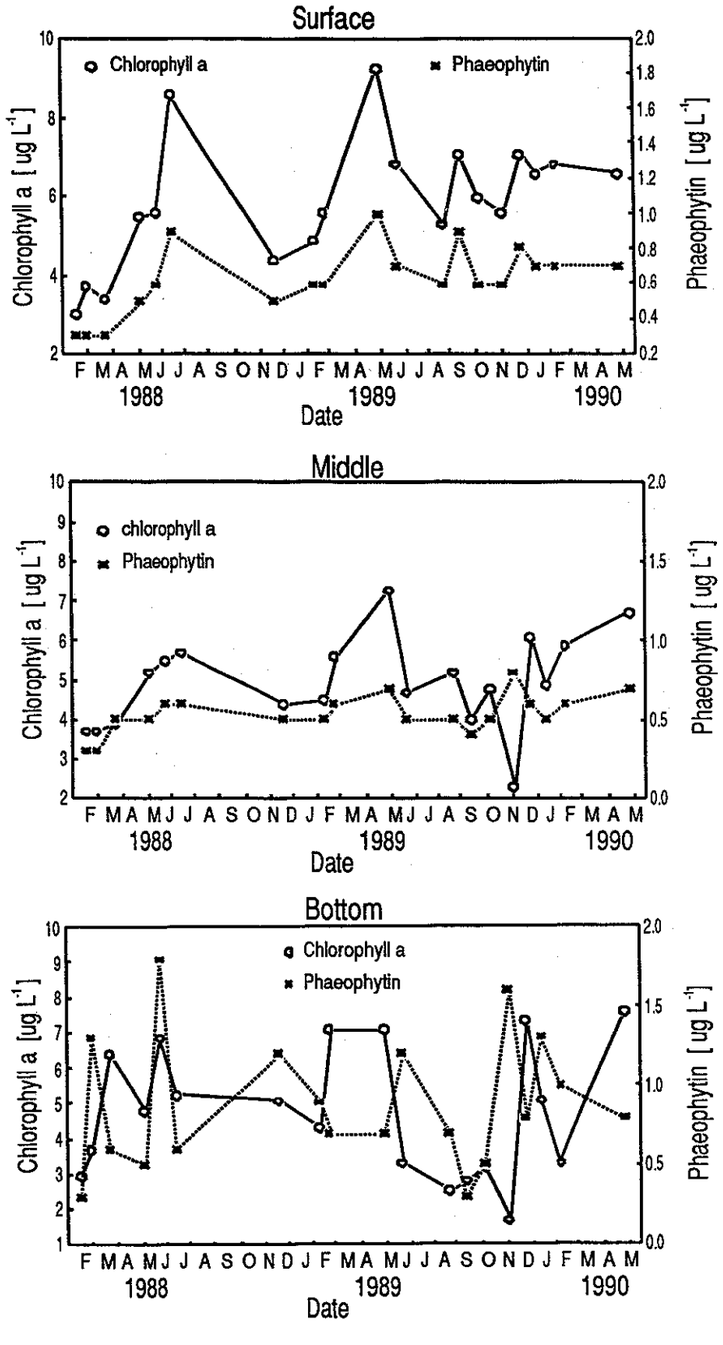


Figure 8.4 Chlorophyll *a* and phaeophytin values for samples from the upstream site (3.5 km) at the surface, middle and bottom of the water column

8.2.2 Macrophytes

Three genera of macrophytes were identified in the Reservoir: *Spirogyra* sp, *Myriophyllum verrucosum* and *Najas marina*. Limited resources for sampling meant that exact estimates of distribution were not possible. In the case of *Myriophyllum* and *Najas* an estimate of the distribution was made using boat and scuba surveys.

The distribution and density of macrophytes are given in Table 8.2 and Figure 8.5.

Table 8.2 Macrophyte distribution and density, April 1990

Myriophyllum verrucosum	
Distribution:	Limited to areas with less than 3 m water depth and providing a sediment rather than rock substrate. Sharp demarcations were obvious, based on these criteria.
Density:	Average 450 g dry mass m^{-2} Range 155-780 g dry mass m^{-2}
Najas marina	
Distribution:	Limited to the shallow flood plain area south of the 3.5 km station
Density:	Average 270 g dry mass m^{-2} Range 230-320 g dry mass m^{-2}
***Spirogyra* sp**	
Distribution:	Limited to growth attached to rocks around the edges of the Reservoir.

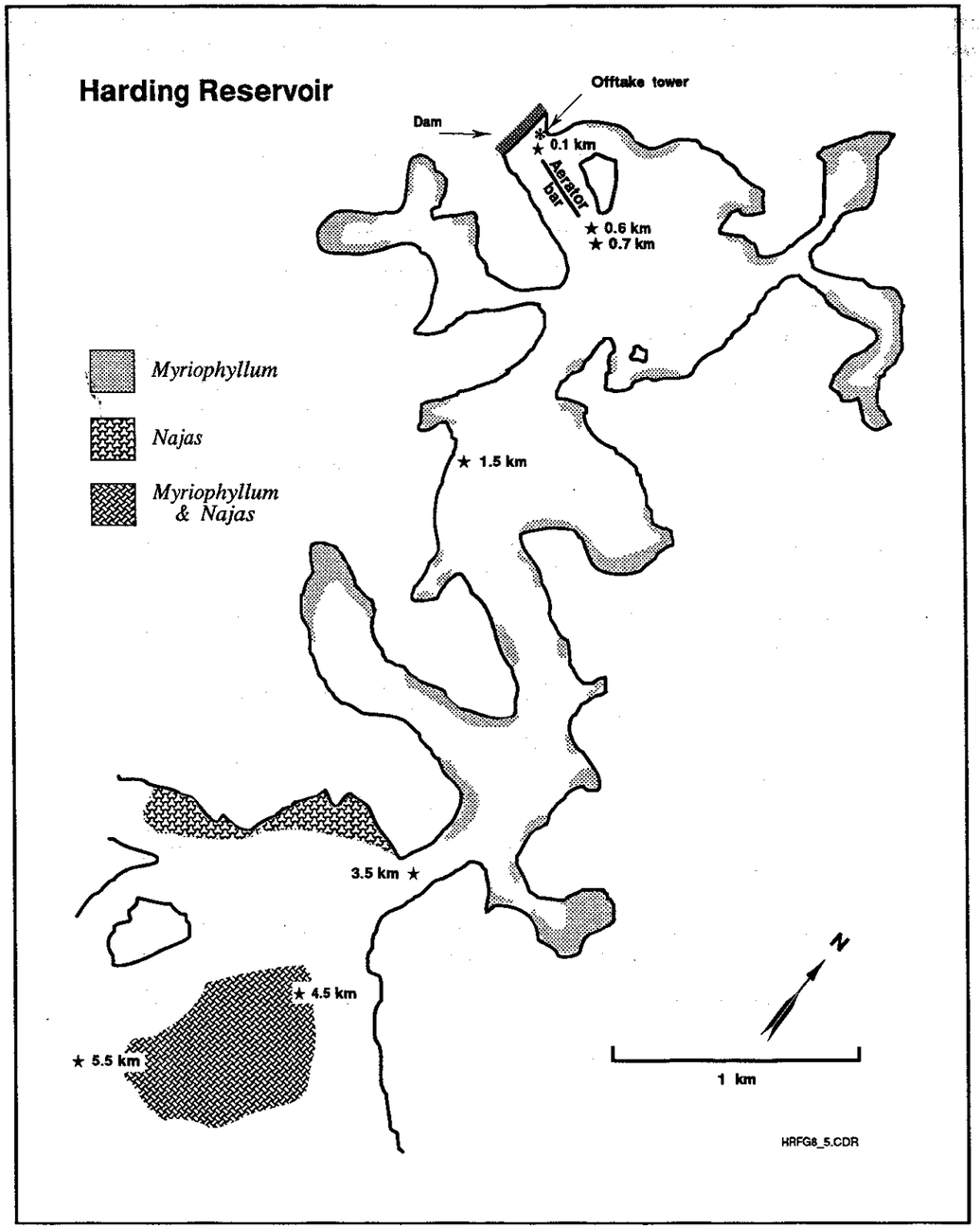


Figure 8.5 Map of Harding Reservoir showing the estimated macrophyte distribution

8.2.3 Zooplankton

Total zooplankton numbers varied considerably during the study period, from the lowest values of <600 individuals m^{-3} to more than 5,000 individuals m^{-3} (Figure 8.6).

The most abundant zooplankton group in the Reservoir was the rotifers and are discussed separately (see also Figures 8.7 and 8.9).

The other major groups present were the Calanoids, Cladocerans and Cyclopoids (Figure 8.8). The Copepods and Cladocerans were the larger-sized zooplankton collected.

The three most common rotifers present were:-

- *Keratella tropica* (Apstein)
- *Brohiomus felcatus* (Zacharias)
- *Filinia apoliensis* (Zacharias)

These are all common zooplankters, widespread in Australia. They are present in billabongs of New South Wales and Victoria where it has been found that species dominance may change as frequently as every three days. The three-weekly sampling may, therefore, not indicate changes in species dominance.

There were no apparent seasonal trends in the abundance or species composition of the zooplankton. Spatial variation in zooplankton abundance and composition was noticeable (Figures 8.10 and 8.11), but major groups generally dominated throughout the reservoir for any one sampling occasion.

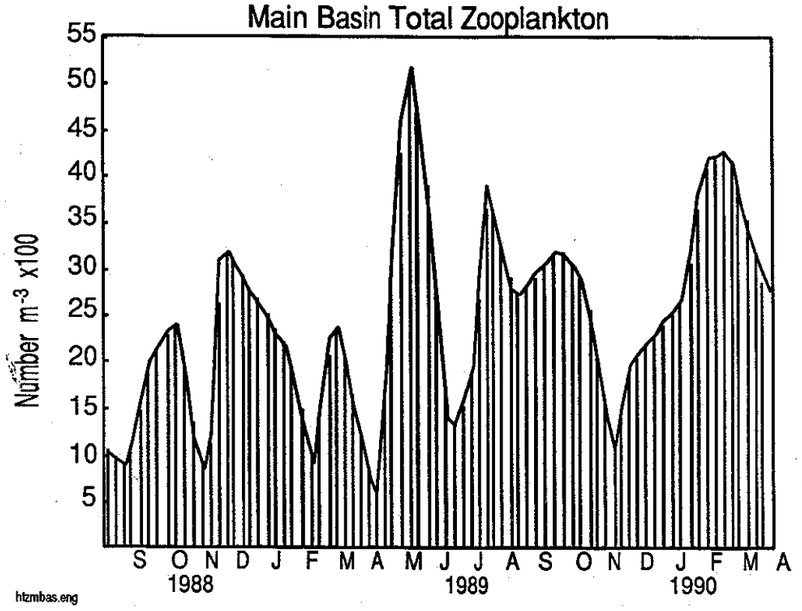


Figure 8.6 Zooplankton counts for the main basin (excluding rotifers), 1988-90

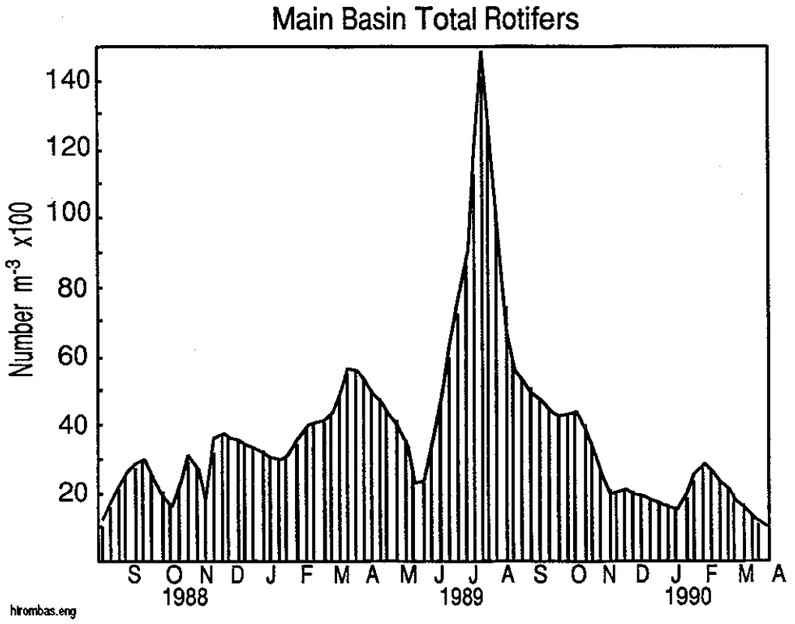
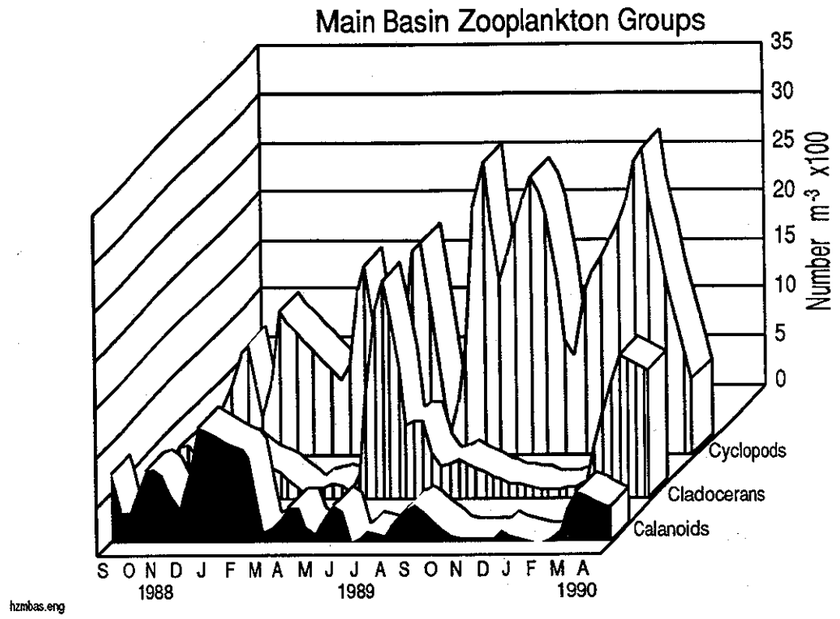


Figure 8.7 Total rotifer counts for the main basin, 1988-90



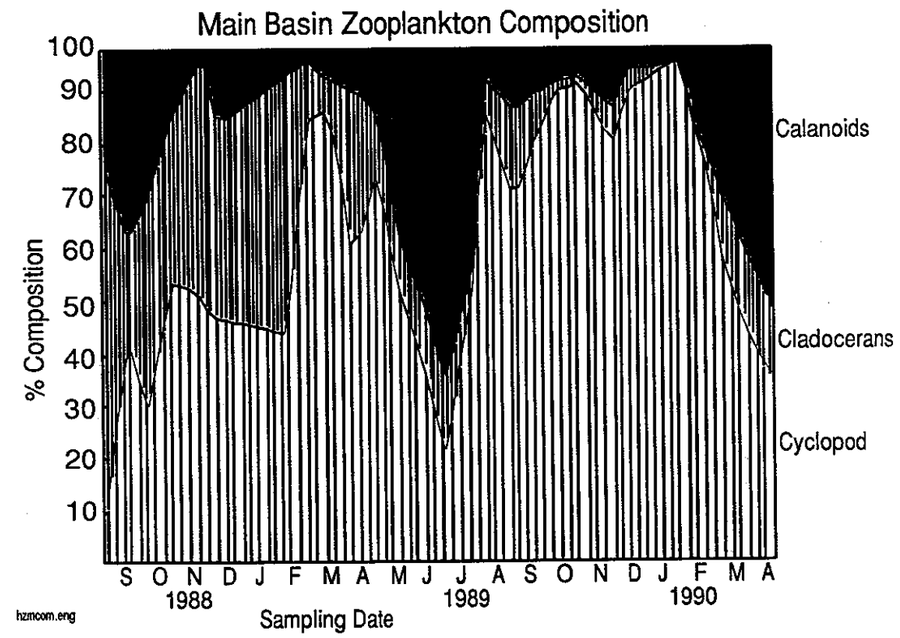
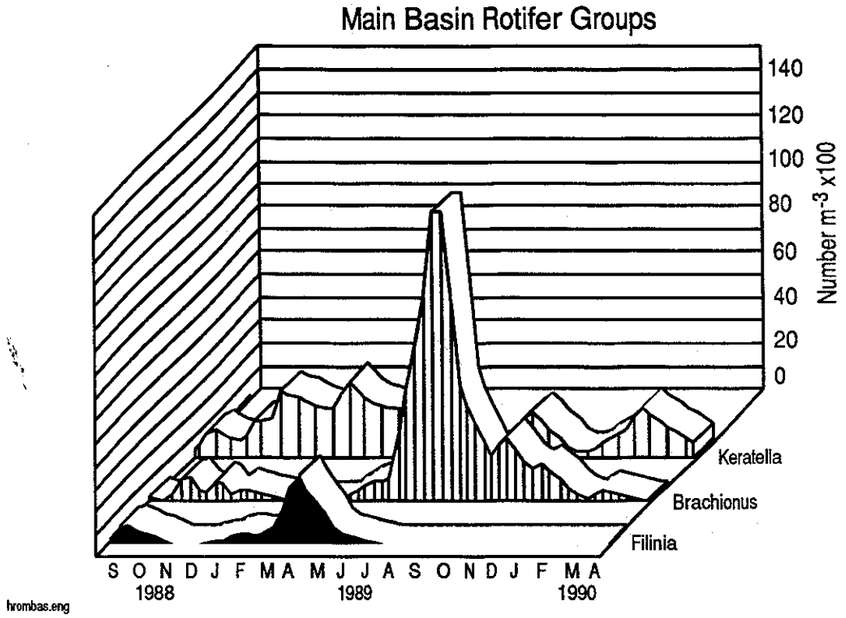


Figure 8.8 Major zooplankton groups (excluding the rotifers) and % composition of these groups in the main basin, 1988-90



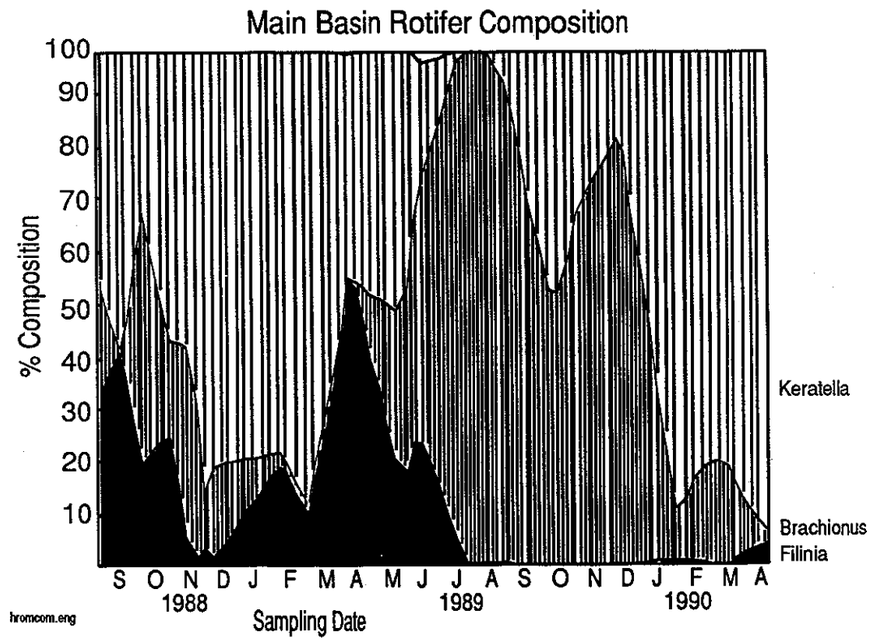


Figure 8.9 Major rotifer groups and % composition of these groups in the main basin, 1988-90

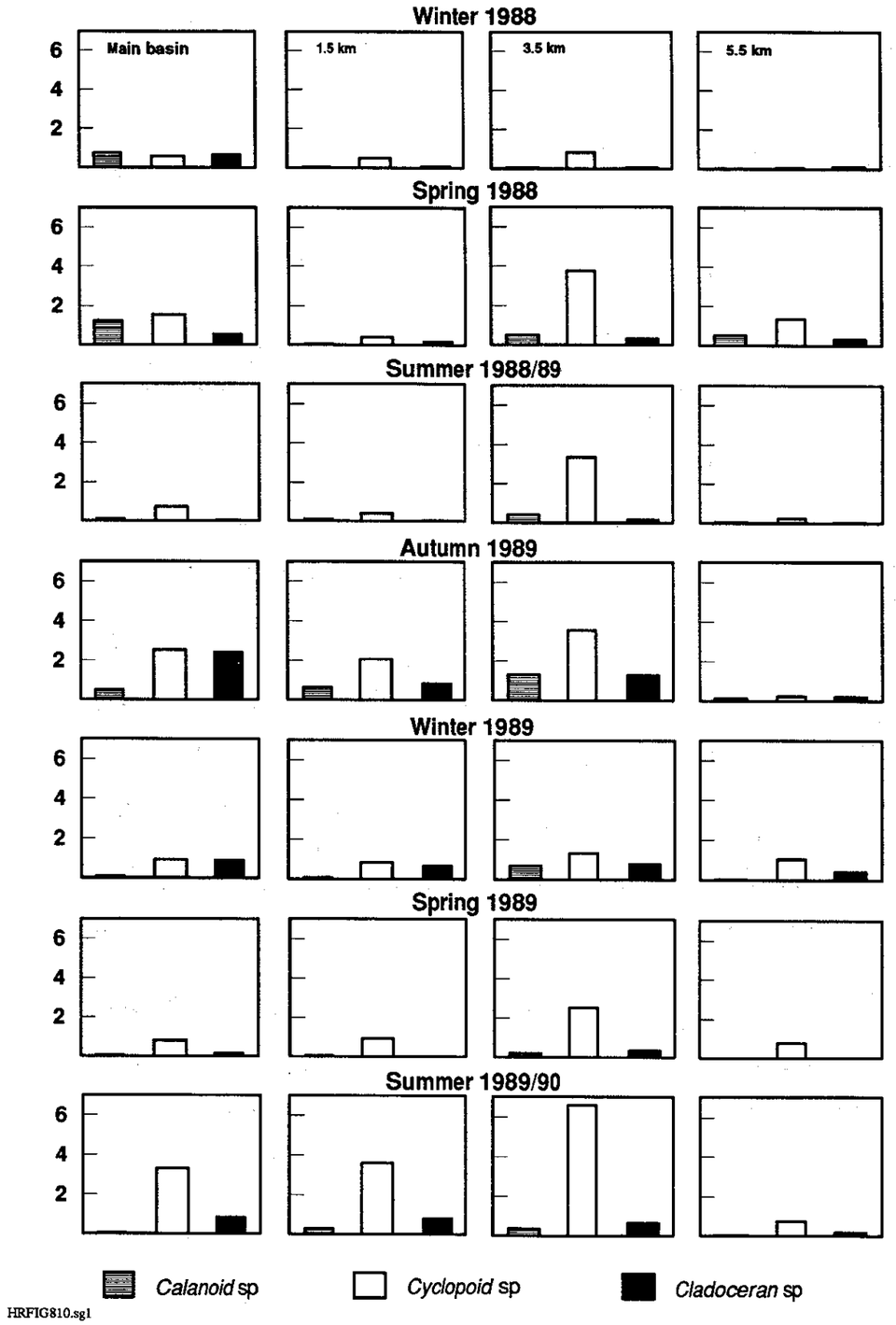


Figure 8.10 Distribution of major zooplankton groups in Harding Reservoir, 1988-90 (000's m^{-3}).

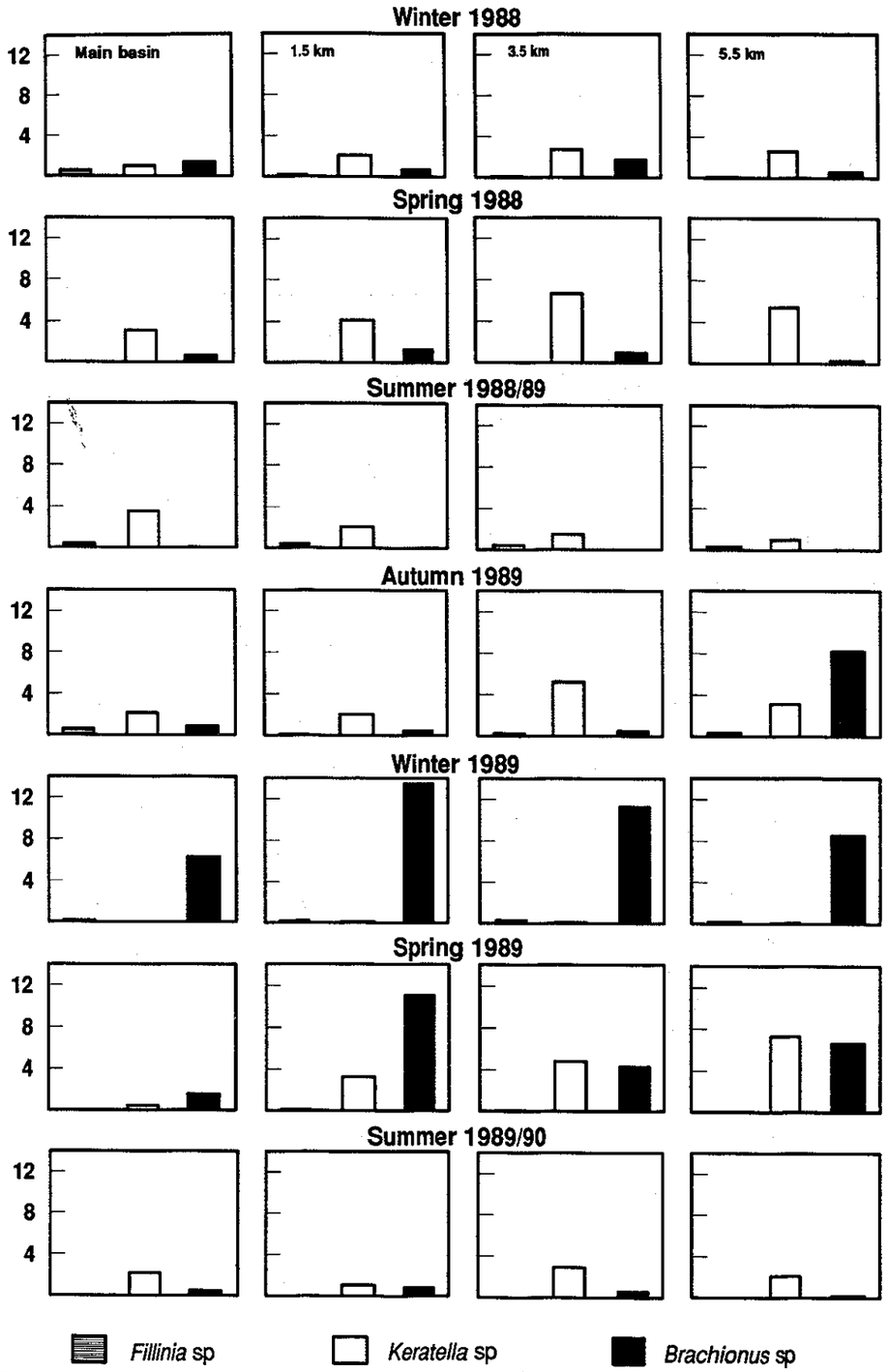


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Figure 8.11 Distribution of major rotifer groups in Harding Reservoir, 1988-90 (000's m^{-3}).

8.3 Discussion

A wide variety of phytoplankton species present in a natural water body is indicative of a biologically and chemically healthy system. It is when a system becomes nutrient-enriched (leading to high total numbers of phytoplankton and, usually, dominance by one or two species) that problems arise. Many species when present in large numbers impart disagreeable tastes and odours to a water supply. Certain species may also produce toxins giving rise to health risks from the water supply. Also, phytoplankton are often responsible for the clogging of filters in water treatment plants.

Artificial destratification may act to limit phytoplankton numbers through restricting either light or nutrient supply. The effects of mixing on phytoplankton numbers is discussed in more detail in McAuliffe and Rosich (1989). Suffice to say here that destratification may cause changes in both total phytoplankton numbers and species composition.

As with phytoplankton, zooplankton numbers may directly cause water quality difficulties through imparting disagreeable tastes and odours and through the clogging of filters. For example, zooplankton populations have caused taste and odour difficulties in Happy Valley Reservoir in South Australia since the early 1980's (Burch, 1987). The reservoir is artificially aerated and is occasionally dosed with copper sulfate to control zooplankton through control of their primary food source, phytoplankton.

Zooplankton may also affect water quality through their link with phytoplankton. Zooplankton are an important component of the overall aquatic food chain, and are often considered to be the main source of mortality of phytoplankton populations.

The impacts of zooplankton grazing on phytoplankton populations have been recognised for a considerable time. For example, Harvey *et al* (1935) established that grazing by zooplankton was responsible for the timing and magnitude of spring phytoplankton blooms in the English Channel near Plymouth.

Given the tropical location of Harding Reservoir and the resultant warm water temperatures (ranging from about 17°C to about 30°C) and relatively high nutrient concentrations, the Reservoir would be expected to be a productive system. As such, it is always possible that algae blooms or excessive zooplankton numbers may occur. At the same time, respiration and activity by predators, and thus mortality, would also be high.

In Harding Reservoir the phytoplankton biomass (as measured by chlorophyll *a*, 3-8 ug L^{-1}) is in the range (3-8 ug L^{-1}) quoted by Vollenweider (1968, 1976) for eutrophic water bodies. This eutrophic state is reflected in a sufficiently high production of organic matter as to result in a moderately high chlorine demand (about 3-4 ug L^{-1}) in potable water supply and occasional noticeable taste and odour. Major taste and odour problems, widespread oxygen depletion and fish kills require hypereutrophic conditions (10's to 100's ug chlorophyll *a* L^{-1}). Water bodies with an oligotrophic status, for example Canning Reservoir (Figure 3.1, 0.3-1.0 ug chlorophyll *a* L^{-1}), exhibit none of the above problems.

The program of artificial aeration was instituted to alleviate the taste and odour problems experienced in the pre-aeration phase of the reservoir's history. It would seem, therefore, that while artificial aeration has been least successful in algae control in many other systems (McAuliffe and Rosich, 1989) it appears to have proved effective in meeting the original management aim in the Harding Reservoir, namely the reduction of tastes and odours.

However, in this study there have been too many other variations to allow us to state that aeration has controlled the algae populations and thereby controlled the taste and odour problem.

9. RELEASES FROM THE SEDIMENT

9.1 Introduction

This chapter describes a series of experiments, utilising reconstituted sediment-water columns, aimed at characterising the release of various elements from superficial sediments of Harding Reservoir.

In lake systems sediments are the main sink for many components including iron, manganese and phosphorous. However, adsorption and release processes allow sediments to act as not only a sink for these elements but also as a source. There are substantial fluxes, in both directions, of such materials across the sediment-water interface. In this way sediments play a significant role in determining the concentrations of these components in overlying waters. Examples exist where lake systems have failed to respond favourably 10 years after the elimination of external loadings of phosphorous, due to internal loading from the sediments (Ahlgren, 1977 and Golterman, 1975, 1977).

The magnitude of unidirectional fluxes of elements such as iron, manganese and phosphorous from a sediment depend on a number of factors, most notably dissolved oxygen concentration, the amount and type of organic matter present, oxidation-reduction (redox) potential and pH. It was outside the scope of this study to estimate unidirectional fluxes (which would require the use of radiotracers), however, net release characteristics were examined.

9.2 Sediment Characterisation

The general composition of sediments samples taken at the two main locations are given in Table 9.1

The concentrations of chemical components, other than inorganic phosphorous fractionation, are typical of concentrations found in sediments of Western Australian water storages (Rosich *et al*, 1991, and Rosich, 1990).

The inorganic phosphorous fractionation results provide important information. The NaOH-extractable inorganic phosphorous fraction is generally accepted as a representation of phosphorous loosely adsorbed to surface sites of hydrated oxides of such as iron, manganese and aluminium and to surface sites of clays. Part of this fraction, especially iron and manganese adsorbed phosphorous, is redox-sensitive. This means that a fall in redox potentials at the sediment surface, sufficient to allow a reduction of iron(III) to iron(II), will promote the release of this phosphorous into interstitial waters within the sediment and ultimately into the water column.

The HCL-extractable phosphorous fraction represents the portion considered to be bound within the crystal lattices of calcium compounds. This fraction, sometimes referred to as apatite-phosphorous, is relatively redox insensitive. Note that the bonding of phosphorous in both fractions is affected by pH and temperature.

The measurement of the various forms of phosphorus in a sediment allows some degree of assessment of the amount of phosphorous that may be released from that sediment. If a sediment has a high amount of NaOH-extractable phosphorous it may be expected to release phosphorous in response to falling redox potentials, such as occurs when dissolved oxygen concentrations fall very low. On the other hand, if the phosphorus in a sediment is primarily HCl-extractable it may be expected to display little variation in release patterns despite varying dissolved oxygen concentrations.

Table 9.1 Sediment characteristics at 0.5 km and 3.5 km sampling locations (dry basis except moisture content)

Site	Unit	0.5 km	3.5 km
water content	%	73.5	69.6
wet:dry ratio		3.77	3.35
organic matter	%	11.7	11.7
total P	ug g^{-1}	895	830
NaOH- P	ug g^{-1}	190	275
HCl-P	ug g^{-1}	560	490
org P	ug g^{-1}	131	156
TKN	ug g^{-1}	2,870	2,760
Fe	mg g^{-1}	68.9	67.2
Mn	mg g^{-1}	2.5	1.3
Ca	mg g^{-1}	8.1	11.7
Mg	mg g^{-1}	29.9	28.3
Zn	ug g^{-1}	76	74
Pb	ug g^{-1}	15	14
Cu	ug g^{-1}	74	76

Table 9.2 compares phosphorous fractionation results from Harding Reservoir (0.5 km and 3.5 km stations), with results presented for marine and estuarine sediments by Lukatelich and McComb (1985). The ocean sampling point is typical of oligotrophic marine systems, while both the Peel Inlet and Harvey Estuary are highly eutrophic systems. The ocean sediment releases very little phosphorous under aerobic or anaerobic conditions, while both estuarine sediments display high net release rates, especially under anaerobic conditions. The explanation of this different behaviour put forward by Lukatelich and McComb (1985) rests largely with the difference in the inorganic phosphorous fractions.

Table 9.2 Phosphorous fractionation results of the Harding Reservoir compared with marine and estuarine results from Lukatelich and McComb (1985)

	Harding Reservoir		Peel Inlet	Harvey Estuary	Ocean
	0.5 km	3.5 km			
total P	900	830	235	430	380
NaOH extractable P	189 (21%)	275 (33%)	59 (25%)	93 (22%)	23 (6%)
HCl extractable P	560 (62%)	490 (59%)	113 (48%)	220 (51%)	320 (84%)
organic P	131 (15%)	156 (19%)	65 (28%)	123 (29%)	33 (9%)

Although, like the oceanic sediment, both Harding Reservoir sediments have the majority of their total phosphorous bound in the relatively inert HCL-extractable fraction, they still hold high absolute amounts of Na-OH extractable P. Given this, it is reasonable to expect that the Harding Reservoir sediments may release large net quantities of phosphorous in response to falling redox potentials. At the same time, the greater water depth in the Reservoir (compared with the Peel-Harvey system where major problems occur due to release of nutrients from the sediments) ameliorates the impact of any sediment release.

9.3 Sediment-Water Release Experiments

9.3.1 Phosphorous, iron and manganese release

Just as net sedimentary phosphorous release may increase in response to falling dissolved oxygen levels, so may net sedimentary iron and manganese release.

Microbial degradation of organic matter occurs continuously in the water column and upper layers of the sediments of most water bodies. As this organic matter is decomposed electrons (e^-) are produced, as shown in the following example:

$$C_6H_{12}O_6 + 6H_2O \rightleftharpoons 6CO_2 + 24H^+ + 24e^-$$

Under aerobic conditions dissolved oxygen serves as an electron acceptor for this process.

$$O_2 + 2H_2O + 4e^- \rightleftharpoons 4OH^-$$

In this manner dissolved oxygen levels decline as decomposition continues and may eventually fall to zero. When this occurs microbial decomposition does not cease, rather other compounds take the place of dissolved oxygen as electron acceptors. Under such anaerobic conditions other reducible materials (such as nitrate, iron (III) and manganese (IV) oxides, organic matter and sulfate) may act as electron acceptors forming nitrogen gas, iron (II) and manganese (II) ions, methane, and hydrogen sulfide respectively.

In this way respiration continues in the absence of oxygen (anaerobic respiration). As these new electron accepting compounds are reduced, previously insoluble elements (such as iron, manganese and phosphorous) may be released into solution.

In the following examples, glucose has been used to represent the organic matter being decomposed. In reality, the microbial degradation process would involve more complex organic compounds arising from biological material. The naturally occurring iron and manganese compounds that have been reduced are also more complex than shown in the following equations.

Aerobic respiration:

$$C_6H_{12}O_6 + 6O_2 \rightleftharpoons 6CO_2 + 6H_2O$$

Anaerobic respiration:

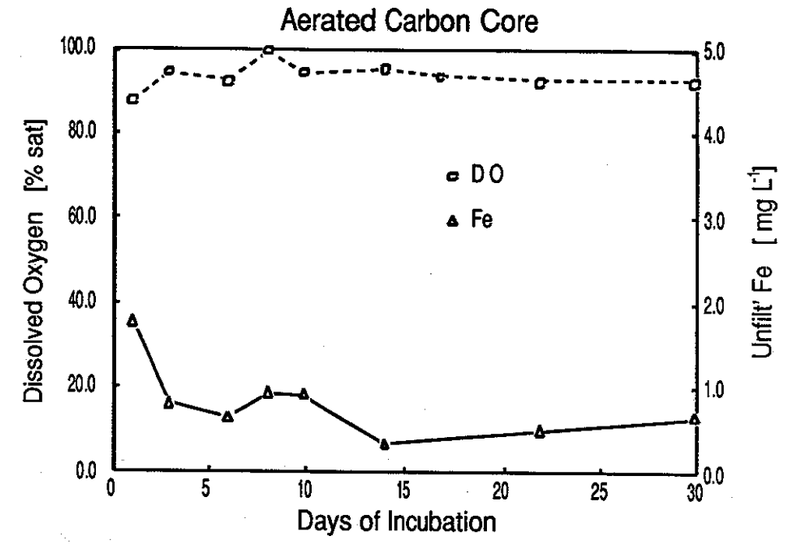
$$C_6H_{12}O_6 + 24Fe(OH)_3 + 48H^+ \rightleftharpoons 6CO_2 + 24Fe^{2+} + 66H_2O$$
$$C_6H_{12}O_6 + 12Mn(OH)_4 + 24H^+ \rightleftharpoons 6CO_2 + 12Mn^{2+} + 42H_2O$$
$$5C_6H_{12}O_6 + 24NO_3^- + 24H^+ \rightleftharpoons 30CO_2 + 12N_2\uparrow + 42H_2O$$
$$C_6H_{12}O_6 + 3SO_4^{2-} + 6H^+ \rightleftharpoons 6CO_2 + 3H_2S + 6H_2O$$
$$C_6H_{12}O_6 \rightleftharpoons 3CO_2 + 3CH_4$$

The anaerobic reactions only occur in the absence of oxygen (that is at low redox potentials) since at higher redox potentials oxygen is the preferred electron acceptor.

Figures 9.1, 9.2 and 9.3 compare iron, manganese and total phosphorous concentrations respectively, in experimental columns under (a) aerated and (b) non-aerated conditions (when dissolved oxygen concentrations depleted) and with sucrose added as a carbon source.

Sucrose was added to allow comparison at extreme dissolved oxygen concentrations. Without this additional carbon source dissolved oxygen did not fall below 50 % saturation. The effect of carbon addition will be discussed more fully in subsequent sections.

It is readily apparent that under reduced oxygen concentrations the net release of iron and manganese into the water column was markedly increased, while for phosphorus the effect was much less. Note, though, that the concentration of phosphorus in both columns is sufficient to maintain eutrophic conditions.



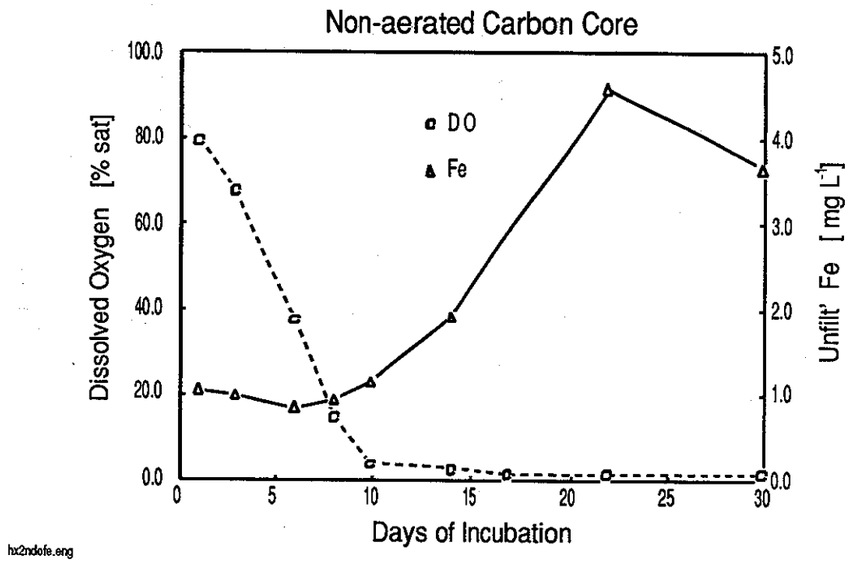
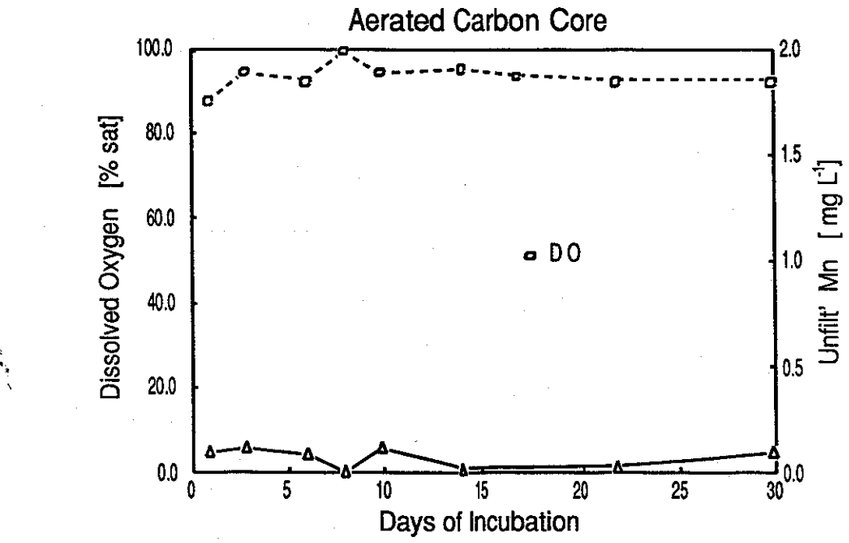


Figure 9.1 Iron concentrations in experimental columns (containing added carbon) under aerated and non-aerated conditions, and in relation to dissolved oxygen concentrations.



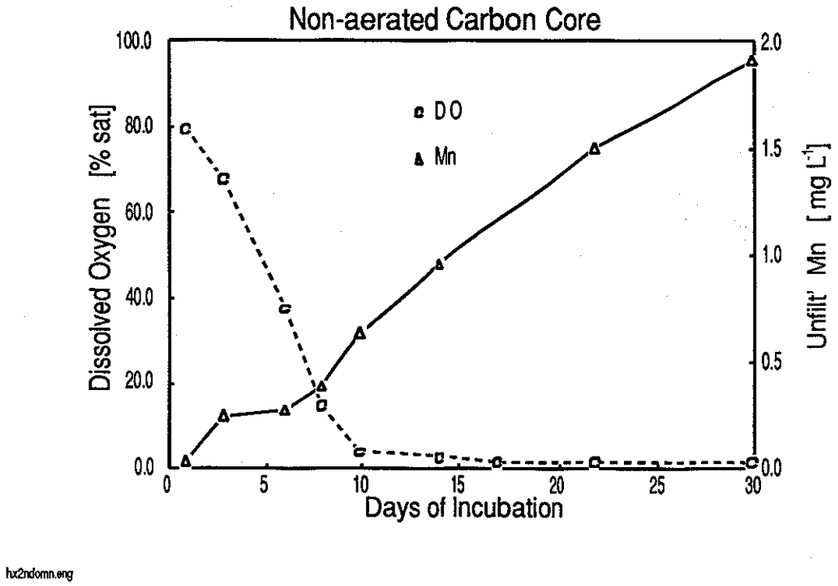
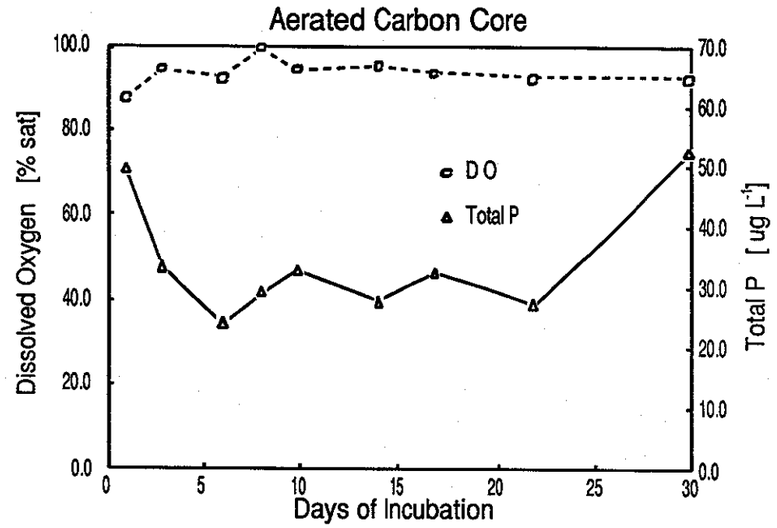


Figure 9.2 Manganese concentrations in experimental columns (containing added carbon) under aerated and non-aerated conditions, and in relation to dissolved oxygen concentrations.



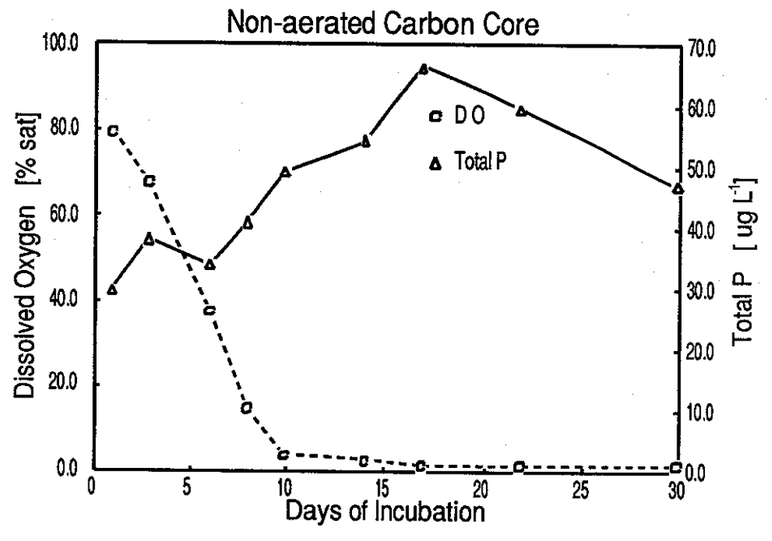


Figure 9.3 Phosphorus concentrations in experimental columns (containing added carbon) under aerated and non-aerated conditions, and in relation to dissolved oxygen concentrations.

9.3.2 Critical dissolved oxygen concentration

Various authors have attempted to establish the critical dissolved oxygen concentration below which the release of phosphorous, iron and manganese is enhanced. Ishikawa and Nishimura (1989), when modelling phosphorous release patterns, determined a critical dissolved oxygen concentration of around 1 mg L^{-1}. Frevert (1980) found no net phosphorous release from Lake Constance above 1.2 mg L^{-1} dissolved oxygen, while Fillos and Biswas (1976) found positive phosphorous release in Lake Mohegan with dissolved oxygen concentrations up to 1 mg L^{-1}. Mortimer (1971) lists a larger range for the critical dissolved oxygen concentration for the Great Lakes region. He considered that at dissolved oxygen concentrations above 1-2 mg L^{-1} these sediments would exert only a "measurable but quantitatively unimportant influence on the chemistry of the overlying waters".

Although it was not the intention of this study to establish exact critical dissolved oxygen concentrations, the results of some experiments do allow a degree of speculation. Columns in which dissolved oxygen did not fall below 50 % saturation displayed net iron, manganese and phosphorous release patterns similar to those of aerated (100 % dissolved oxygen saturation) columns (Figure 9.4).

Columns experiencing dissolved oxygen depletion greater than 50 % displayed enhanced net iron, manganese and phosphorous release. Based on the above observations and the results given earlier in Figures 9.1, 9.2 and 9.3 the critical dissolved oxygen concentration for enhanced net iron, manganese and phosphorous release in these columns was between 40 % saturation (3.7 mg L^{-1}) and 20 % saturation (1.8 mg L^{-1}).

The relevance of these values to the field situation requires some qualification. The dissolved oxygen concentrations measured in the experimental columns were mean values. Measurements were taken after the water column was mixed and as such may not be an accurate reflection of the concentration at the sediment surface itself. Holdren and Armstrong (1980) point to micro-anaerobic zones at the sediment surface resulting in over estimation of what was thought to be "aerobic release".

It is quite possible that the non-aerated experimental columns used in this study could develop dissolved oxygen stratification between the mixing events associated with sampling. In such a case the measured mean value of dissolved oxygen for the whole water column would be higher than that experienced at the sediment surface.

9.3.3 Carbon levels

Dissolved oxygen concentrations have been shown to be critical determinants of the amount of net release of phosphorous, iron and manganese. In turn, these oxygen concentrations themselves depend on the nature and levels of carbon in the sediment.

As previously stated the dissolved oxygen concentration of a hypolimnion may fall as microbial degradation of organic matter proceeds. It follows that this process is dependent on the level of organic matter (often expressed as carbon content). The higher the carbon level the greater the rate and duration of oxygen depletion (Bostrom and Pettersson, (1982); Holdren and Armstrong, 1980; McAuliffe and Rosich, 1989).

Figure 9.5 compares dissolved oxygen depletion in (a) a control column and (b) a column receiving a readily reducible carbon addition, in the form of an addition of 0.4 g of sucrose.

By the processes described earlier the carbon addition has resulted in reduced concentrations of dissolved oxygen in the water columns. This may ultimately lead to increased net release of phosphorous iron and manganese as described in the preceding section.

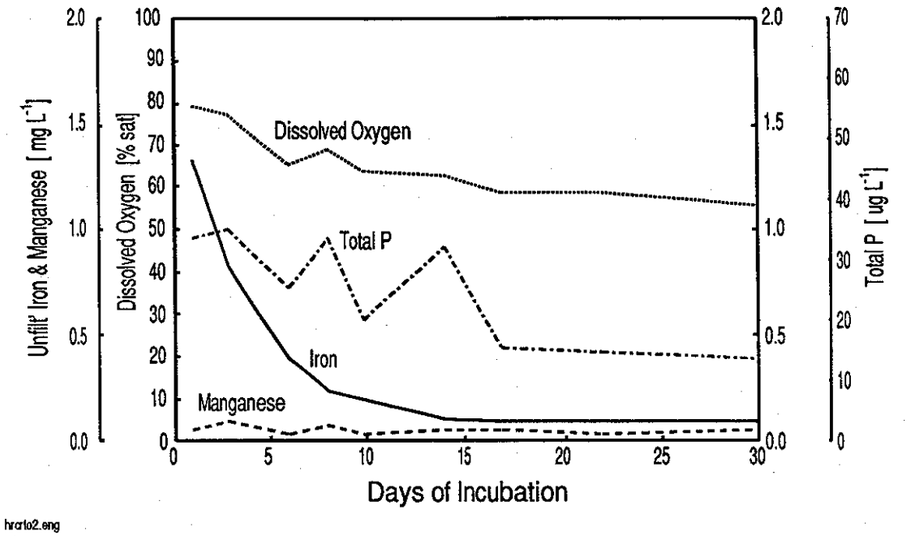


Figure 9.4 Iron, manganese and phosphorous concentrations in experimental columns in which dissolved oxygen concentrations were maintained above 50 % saturation.

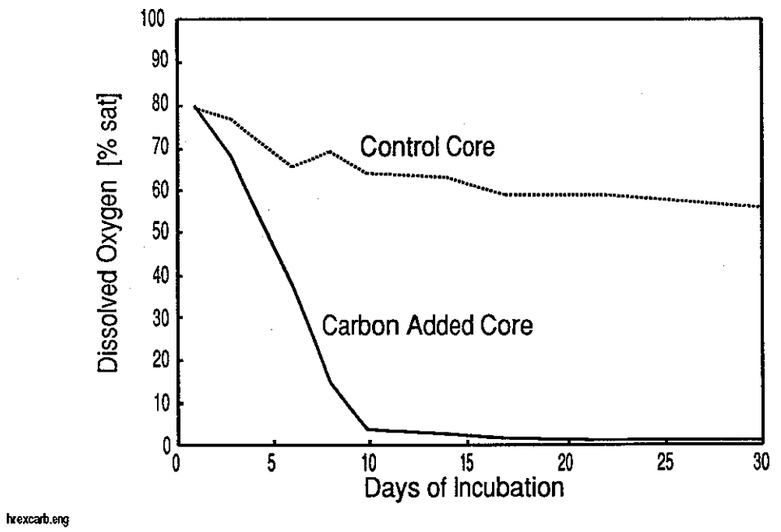


Figure 9.5 Dissolved oxygen depletion in a) a control column and b) a column receiving a readily reducible carbon addition

9.4 Summary and Conclusions

The sediments of Harding Reservoir contain sufficient levels of iron, manganese and phosphorus as to cause high concentrations in the overlying waters if conditions (primarily low dissolved oxygen concentrations) become favourable for their release from the sediments. Furthermore, they contain substantial concentrations of the sodium hydroxide extractable phosphorus fraction which is readily available to the biological system.

In the case of Harding Reservoir, as with many other water bodies, stratification allows a layer of water to be effectively trapped adjacent to the sediment surface. Due to restricted access of this hypolimnion water to the air-water interface, microbial degradation of organic matter may result in oxygen depletion of this hypolimnion. Excluding the effects of overturn of the water column, the rate and duration of this oxygen depletion will depend on the level of readily-assimilable carbon.

Based on the results of experimental sediment-water columns, once dissolved oxygen concentrations fall below about 40 % saturation there is likely to be an increased net release of iron, manganese and phosphorous from the sediments. All of these components are available at quantities sufficient to ensure that the sediment would be able to act as a source of them for many years. The extent to which these components will affect the water quality of the surface waters is primarily dependant on the vertical mixing of the water column, including the degree and stability of stratification.

Artificial destratification/aeration to provide high dissolved oxygen concentrations (> *ca* 40% saturation) throughout the water column and to the depth of the sediment surface would provide optimum conditions for the minimisation of iron, manganese and phosphorus release from Harding Reservoir sediments. The difficulty is to achieve this oxygen level without resuspending the bottom sediments.

10. CONCLUSIONS

10.1 Overview

During the study period (and since) there were no major outbreaks of taste and odour complaints from consumers of water supplied from Harding Reservoir. Aeration/destratification was practised throughout this period whenever water was being supplied to consumers and thus there is an *a priori* case that the technique has been successful. However, during the study period water was not supplied to consumers while the Reservoir was not being aerated/destratified and therefore the role of the aeration/destratification technique cannot be assessed with certainty.

It should be noted that the aeration/destratification technique was introduced into Harding Reservoir only a few years after construction was completed, in response to taste and odour problems, and those early problems may have been associated with the initial filling of such an artificial reservoir. Further, there are very few relevant data from that early period with which to compare the data obtained in this study.

Another major difficulty in assessing the role of aeration/destratification in Harding Reservoir is in the relative instability of stratification under natural conditions. Prior to this study commencing it was thought that the Reservoir stratified for periods of months during the dry season. However, this is an artefact of infrequent measurements as shown in the work by the Centre for Water Research, UWA (Lewis *et al*, 1991). Even though the Reservoir can stratify strongly, the climatic conditions can cause rapid (days) change between stratified and destratified states. Main reasons for this instability are the local climatic and geographic conditions together with the relative shallowness (maximum depth 24 m) of the Reservoir.

As community concerns for taste and odour diminished the incidence of 'dirty water' complaints increased. A separate study of this issue (Henderson *et al*, 1990) demonstrated that the main source of the particulates in 'dirty water' originated from the Reservoir. Thus, in the later stages of the aeration/destratification study attention was also focussed on turbidity in the Reservoir.

10.2 A Conceptual Model for Harding Reservoir

Harding Reservoir exhibits characteristics described in Chapter 3 as typical for tropical and warm temperate water bodies:

- minimum temperatures always above 4°C,
- a single stratified period per year (under natural conditions) and
- natural destratification associated with the onset of the (usually brief) rainfall runoff period.

Given that significant inflow only occurs during a very limited part of the year (and in the case of Harding Reservoir not even every year) nutrient release from the sediments and recycling within the water column play major roles in maintaining the biological populations in the water column.

10.3 Climate, Land Use and Hydrology

The Reservoir is located in an arid tropical region characterised by high summer temperatures, high evaporation rates and unreliable rainfall. Most rainfall events are associated with tropical cyclones, which since 1980 have occurred on an average of a little over one per year.

The majority of the catchment is characterised by mixed scrub and spinifex. There has been little impact on the catchment by man with, in the past, only low density grazing by sheep and cattle.

10.4 Stratification and Physical Components

The aerator installed in Harding Reservoir during this study is located 4-6 m above the bottom so as to avoid the mass transport of nutrient rich bottom waters and bottom sediments to the upper waters where algal production could be stimulated and/or turbid waters enter the supply.

The result has been the formation of a bottom layer (that is, below the level of the aerator) high in nutrients and particulates. On occasions, natural overturn (or simply deepening of the upper mixed layer to below the level of the aerator) has brought these waters to the surface resulting in the Reservoir having to be taken off supply due to high turbidity. On other occasions, excessive turbidity in the surface layers (that is, at the offtake level) without overturn has decreased after the aerator was switched off.

At the same time, the energy contributed by the aerator has been insufficient to maintain the waters above the aerator in a fully mixed state resulting in temperature differences of up to 4.5°C and differences in dissolved oxygen concentrations of up to 7.5 mg L^{-1} between the surface waters and those just above the aerator.

10.5 Major Components (Na^+, K^+, Ca^{2+}, Mg^{2+}, SO_4^{2-}, Cl^- and carbonates)

As expected, these are not significantly affected by aeration/destratification. Rather, they respond to the cycle of inflow and evaporation - decreasing in concentration after inflow and then approximately doubling due to evaporative concentration during the subsequent dry season.

10.6 Biological Components

Harding Reservoir exhibits a wide variety of phytoplankton and zooplankton species indicative of a biologically and chemically healthy system.

Levels of these groups (expressed as 3-8 chlorophyll *a* ug L^{-1} for phytoplankton) are in the range for eutrophic rather than hypereutrohic (10's to 100's for chlorophyll *a*) waters and thus although this biological production gives rise to a relatively high chlorine demand (3-4 mg L^{-1}) there were no incidences of major taste and odour complaints during the time of this study, nor have there been since.

However, the very few relevant data available for the Reservoir prior to the implementation of aeration/destratification means it is not possible to ascertain whether or not the levels of these biological components have changed and hence whether or not the apparent reduction in the taste and odour problem has been due to a change in these biological components.

Given the warm water temperatures (in the range of about 17-30°C), the ample available light and the availability of nutrients from the sediments (see the next section) the potential exists for excessive algal or zooplankton populations and resultant water quality problems.

10.7 Releases from the Sediments

The sediments of Harding Reservoir contain sufficient levels of iron, manganese and phosphorus as to cause high concentrations in the overlying waters if conditions (primarily low dissolved oxygen concentrations) become favourable for their release from the sediments.

Based on the results of experimental sediment-water columns, once dissolved oxygen concentrations fall below about 40 % saturation there will be increased net release of iron, manganese and phosphorous from the sediments. All of these components are available in the sediment at quantities sufficient to ensure the sediment would be able to act as a source of them for many years. The extent to which these components

will affect the water quality of the surface waters is primarily dependant on the vertical mixing of the water column, including the degree and stability of stratification.

Artificial destratification/aeration to provide high dissolved oxygen concentrations (> *ca* 40% saturation) throughout the water column and to the depth of the sediment surface would provide optimum conditions for the minimisation of iron, manganese and phosphorus release from Harding Reservoir sediments. The difficulty is to achieve this oxygen level without resuspending the bottom sediments.

11. RECOMMENDATIONS

It is recommended that:

1. Aeration/destratification continue to be used as a tool to assist in the management of water quality in reservoirs with problems similar to those of Harding.

2. In the case of water bodies where the aim is to control components whose concentrations increase to only slightly above acceptable levels, aeration/destratification should be commenced when the first signs of stratification appear.

3. To minimise the incidence of elevated turbidity and nutrients in the surface layers the elevation of the aerator should be regularly monitored to ensure that it stays sufficiently above the bottom of the reservoir. In a reservoir (such as Harding) formed in a river valley with a distinct channel in a flood plain, the aerator should be placed just above the flood plain of the old river valley. This will minimise the volume of water below the aerator where high concentrations of nutrients, iron and manganese can be found.

4. In any program to assess the impact of aeration/destratification on water quality, especially the biological compartment, relevant data must be available under natural conditions. Alternatively, the program should include studies of different aeration/destratification strategies using large plastic enclosures placed in the water body.

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