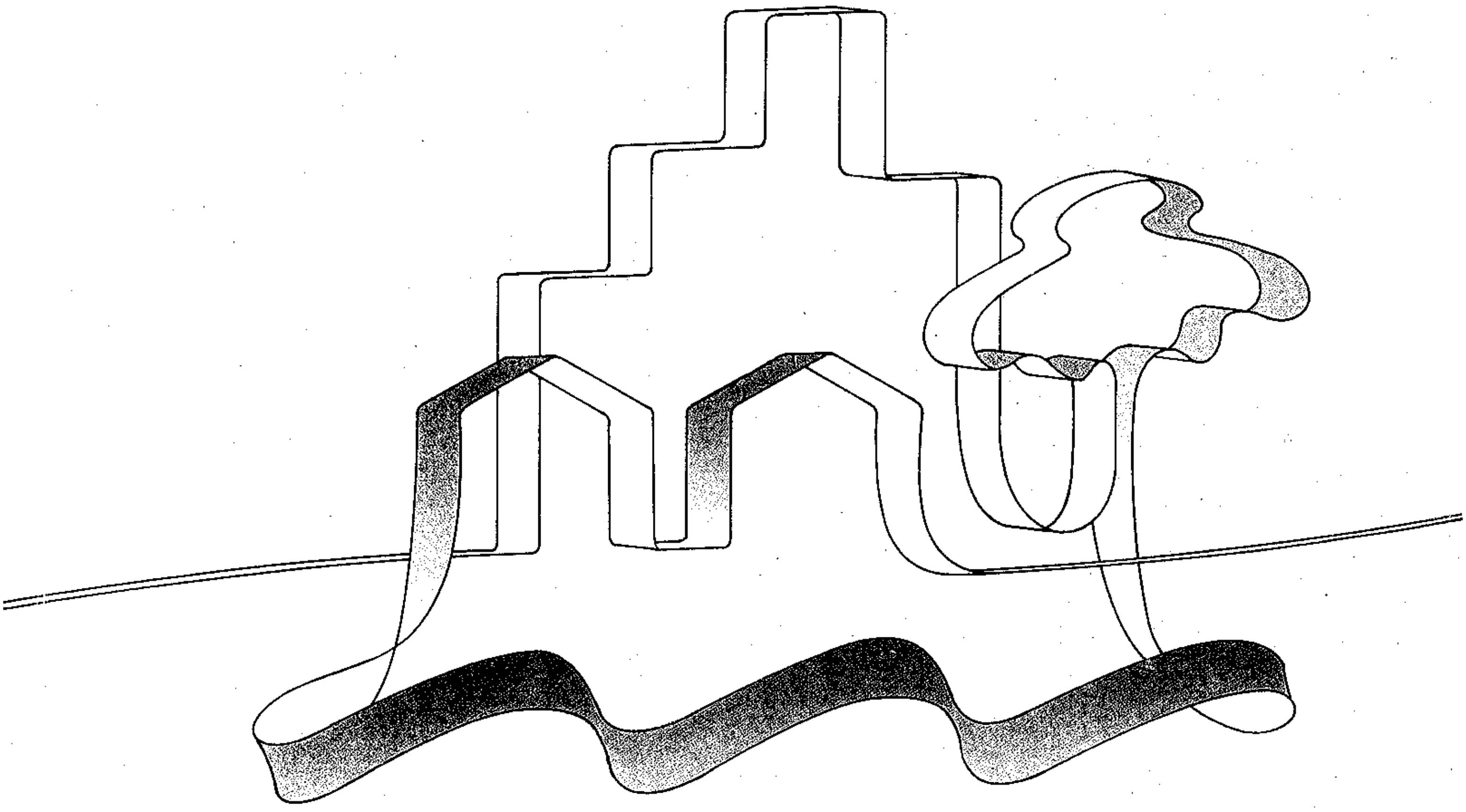
Urban Water Research Association of Australia

REVIEW OF ARTIFICIAL DESTRATIFICATION OF WATER STORAGES IN AUSTRALIA



Research Report No. 9

URBAN WATER RESEARCH ASSOCIATION OF AUSTRALIA

The Association was formed in 1986 following initiatives by the Australian Water Research Advisory Council and the Major Urban Water Authorities of Australia. The Association's primary role is to foster and promote a comprehensive, co-ordinated and cost-effective approach to urban water research within Australia, for both metropolitan and non-metropolitan areas.

The Association invites proposals for research work through its member authorities and allocates funding to approved projects on an annual basis. The actual research is undertaken by water authorities, research organisations, universities, consultants and government agencies.

The UWRAA Research Report series presents information resulting from research projects supported by the Association and is published as a record of the work undertaken and as a means of disseminating the research findings. The Association also encourages the presentation of findings by the researchers in professional journals and at conferences. The Association's reports are indexed on STREAMLINE, the national water data base.

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FOREWORD

This report is based on UWRA Research Project No. WS - 2 "Evaluation of Aeration/Destratification of Reservoirs". The report is the first of three scheduled reports on the project. The remaining two reports will deal with:

- development and validation of a numerical model of mixing processes in water bodies.
- detailed water quality studies of the implementation of aeration/destratification in a reservoir with water quality problems.

Organisational responsibility for the project reported herein was as follows:

Sponsoring Authority	:	Water Authority of Western Australia
Project Officer	:	Dr B Kavanagh Water Authority of Western Australia
Principal Researchers	:	Mr T McAuliffe Dr R Rosich Water Authority of Western Australia
Review Panel	:	Dr B Kavanagh Water Authority of Western Australia

The project was funded by the Urban Water Research Association of Australia, on the advice of the Association's Research Advisory Committee, and by the Water Authority of Western Australia.

SYNOPSIS

This report aims to review the current status of reservoir destratification in Australia, focusing on the effectiveness of artificial destratification in controlling biological and chemical aspects of water quality. The review presents information and comment from relevant literature as well as that obtained by discussion with persons responsible for Australian destratification applications.

Over fifty destratification applications have been examined in detail and are presented as case studies. The results of these applications are also examined in conjunction with discussions on the aquatic processes operating in stratified and destratified systems. The ways in which artificial destratification may act to alleviate water quality difficulties are discussed in the context of the Australian experience.

It is concluded that for the most part Australian destratification applications have been of limited success. The applications succeeded typically in attempts to alleviate problems associated with colour and turbidity and to a lesser degree problems associated with iron and manganese. It is in the area of algae control that destratification applications have been least successful, this report found that 68% of applications attempting to reduce and control algae levels failed to do so. The authors conclude that the relationships between mixing and algae responses are areas requiring the most urgent attention.

This report is the first part of a two-stage project and includes information explaining the principles of destratification as a water quality management tool. The second stage of this project, currently under way, involves field and experimental work on the effect of destratification on water quality in the Harding Reservoir in Western Australia. Both stages involve complementary hydrodynamic field and modelling studies of destratification. These latter studies, undertaken by the Centre for Water Research (The University of Western Australia) will be reported separately by the Urban Water Research Association of Australia.

THE STRUCTURE OF THIS REPORT

Discussion of the water quality concepts dealt within this report has been organised with reference to the problems encountered in stratified water bodies throughout Australia. Many readers will wish to focus on discussion relating to specific water quality problems relevant to their own experiences. Chapter 2 will serve as a means for these readers to key directly into such discussion and thus facilitate the use of this report as a water quality management tool.

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CHAPTER 1

INTRODUCTION

1.1 Stratification

Artificial destratification is the process of applying mixing to a water body to break down existing stratification or to prevent the onset of stratification. In Australian reservoirs this mixing has been provided in the main by artificial aeration but also by mechanical, propeller-like mixers.

The seasonal stratification of water bodies is a natural occurrence evident throughout Australia, most usually characterised by decreasing water temperature down through the water column. This stratification commonly forms as surface waters heat with the change of seasons and ensuing warmer weather. As cold water is more dense than warm water the presence of warm water overlying colder water establishes a degree of stability and resistance to mixing by wind. Stable layers form within the water column which will usually persist until a change of seasons forces a re-mixing.

It is common for temperate Australian water bodies to mix once per year; such systems are referred to as monomictic water bodies. Although many variations exist this simple rhythm usually involves three periods:

i) a period of stratification around and during the summer months;

ii) a relatively short period known as "overturn" where the stratification is broken down as the water body mixes in response to colder air temperatures and

iii) the remainder of the year, during the cooler months, when the water body remains mixed.

1.2 Water Quality Effects of Stratification

Stratification may lead to reduced water quality in a variety of ways, with the two main areas of effect being increased algae growth and/or an increase in the release of substances from the sediment.

Algae populations may increase with stratification usually as a result of either being able to trap more light by staying at or near the water surface or by having access to an increased amount of nutrients as a result of stratification or overturn. Particular types of algae may adversely affect water quality. This is of particular concern in water supply reservoirs as some algae when present in moderate to large numbers impart undesirable tastes and odours to the water. Furthermore, some blue-green algae are notorious for the release of toxic as well as odourous substances. Stratification may provide these algae with a competitive advantage over other types of algae.

The second general way in which stratification often reduces water quality is through enhancing the release of undesirable substances from the sediment. Such release is usually associated with a decrease in the concentration of dissolved oxygen in the bottom layer (hypolimnion). During mixed conditions oxygen is brought to the sediment by water that has previously been mixed and aerated at the surface. Stratification prevents this from occurring; mixed and aerated water is confined to the uppermost layer (epilimnion), while the hypolimnion has a limited amount of dissolved oxygen trapped in it at the time of stratification. Without light this oxygen is steadily depleted by the natural biological processes in the water and upper sediment.

Bacterial respiration at the sediment-water interface is a major consumer of this limited supply of oxygen and persists even after all the oxygen has been consumed. In the absence of dissolved oxygen, bacterial respiration continues in an anaerobic state causing compounds other than oxygen to be reduced. In the upper sediment some of the chemical bonds formed between elements such as iron, manganese and phosphorus are broken, releasing these elements to the overlying water. The release of such substances may lead to a variety of water quality difficulties, as discussed in Chapter 5.

Although the control of algae numbers and sediment release are the two main areas of concern in Australia, the complete list of problems associated with stratification is much broader. More specifically solutions through destratification have been sought to the problems of:

- unpleasant tastes and odours;
- high numbers of algae
- undesirable species of algae
- algae toxicity;
- high iron levels;
- high manganese levels;
- high colour and turbidity;
- highly variable water quality;
- low temperature releases;
- high zooplankton numbers;
- high numbers of aquatic insects and
- fish kills.

1.3 Destratification in Australia

The first attempt at artificial destratification in an Australian water body was in 1966 in Little Nerang Creek Dam in Queensland (Qld Department of Local Government and Gold Coast City Council). This "air gun" system was replaced by a form of variable offtake and then converted back to artificial destratification in 1983.

In 1974-75 experiments with destratification continued, using water jet and artificial aeration in a section of Eildon Reservoir in Victoria. The water column was successfully destratified in three hours, with the data obtained allowing the development of design criteria for future applications (Brown 1986, Burns 1977).

These latter experiments were carried out under the guidance of the Rural Water Commission of Victoria (formerly State Rivers and Water Supply Commission of Victoria) and it was in Victoria where the first full applications of destratification were initiated. The late 1970's saw destratification systems installed in Tarago Reservoir and Lake Eppalock. From this stage destratification applications have spread throughout Australia. To date, destratification has been tested or implemented in about 60 different systems in Australia, with varying degrees of success. The applications have grown in both number and type. Early forms were mostly limited to artificial aeration through air-diffuser bars, while more recently the desire for more efficient mixing has led to the implementation of mechanical mixers in some water bodies (eg. Myponga Reservoir, Lake Medlow).

1.4 The Overall Approach to this Report

This report aims to bring together experiences with the destratification in Australia and explain the principles behind the results obtained. The report does not deal with every attempt at destratification conducted in Australia to date. Rather the material presented represents information relating to those systems relevant to destratification in Australia today and for which information was able to be readily obtained. Further, the report aims to complement and build on an earlier review of destratification techniques in Australia, centring mainly on iron and manganese related problems (Brown 1986).

Information was obtained through written, telephone and personal approaches. The latter approach appeared vastly more effective, in soliciting the cooperation of those workers in this field throughout Australia and we highly commend it to anyone attempting a similar review.

The second stage of this project, currently underway, involves field and experimental work on the biological and chemical effects of various destratification strategies applied to the Harding Reservoir, Western Australia. This work is conducted in collaboration with the hydrodynamic studies of the Centre for Water Research, The University of Western Australia. This second stage is scheduled for completion in March 1990.

CHAPTER 2

AN OVERVIEW OF THE PROBLEMS ADDRESSED BY DESTRATIFICATION

2.1 Introduction

Destratification has been applied throughout Australia in an attempt to alleviate operational, aesthetic or consumer linked problems associated with a variety of water quality variables. These difficulties often arise as a result of a series of well understood and intermeshed aquatic processes. The following discussion links the problems associated with stratification to the relevant aquatic processes. It is intended that readers may use this Chapter to key into discussions, elsewhere in the report, pertinent to their particular interests.

2.2 Tastes and Odours

Unpleasant tastes and odours regularly give rise, throughout Australia, to reductions in the potability or aesthetic value of many water bodies. Tastes and odours are widely differing in nature and may be linked to a variety of causes.

2.2.1 High algae numbers

Algae numbers in a water body are linked closely to four groups of factors: light and temperature; mixing and water turbulence; predation; and nutrient supply. These factors interact to establish a particular level of algae (or "standing crop"). At any given time one of these factors may be present at a level such that it will act to control or limit algae numbers. A favourable change in this limiting factor may then give rise to a rapid increase in algae numbers and a resulting bloom.

Refer to: 4 Algae Growth
4.2 Nutrient Supply

4.3 Temperature Effects
4.4 Light and Algae Growth
4.5 Effects of Mixing
5 Releases from the Sediment
5.5 Phosphorus

2.2.2 Different types of algae

While the above four groups of factors ultimately determine algae numbers, more subtle variations of the same factors determine species composition. Although factors such as selective grazing by zooplankton are recognised as important, the dominant influences are believed to be the degree of mixing and nutrient ratios present in the water body.

Refer to: 4 Algae Growth
4.2.3 Nutrient ratios
4.5 Effects of Mixing

2.2.3 Reduced odours (such as hydrogen sulfide)

Odours produced by the release of highly reduced compounds such as rotten egg gas (hydrogen sulfide: H_2S) are particularly obnoxious and may be toxic. These odours are produced usually at the sediment-water interface or from rotting weed, as a result of highly reduced conditions (ie. after dissolved oxygen has been removed).

Refer to : 5 Releases from Sediment
5.3 Dissolved Oxygen and Redox Potentials
5.8 The Role of Organic Matter

2.2.4 High zooplankton numbers

High numbers of zooplankton may give rise to unpleasant tastes and odours, typically imparting a fishy odour to a water body. Zooplankton levels are controlled primarily by fish predation, temperature and the availability of their food source, (mainly phytoplankton).

Refer to : 7 Aquatic Animals
7.1 Zooplankton Numbers

2.3 High Levels of Iron and Manganese

Iron and manganese enter water bodies with inflow and runoff from catchment sources. Under normal oxidised conditions they enter as particulates and organic complexes of iron (III) and manganese (IV). In such forms they typically settle out to join iron and manganese already present in the submerged sediments. Under well oxidised conditions these elements stay bound to the sediment and pose few problems in the overlying water. This balance may, however, be upset in two basic ways. Firstly the iron and manganese entering a water body may settle too slowly and remain in the water column to pose supply or operational difficulties. More commonly, the balance may be disturbed through the development of reducing conditions at the sediment surface, which allow release of iron and manganese into the overlying water column.

Refer to : 5 Releases from the Sediment
5.4 Iron and Manganese
6 External Inputs
6.2 Iron

2.4 High Colour and Turbidity

Colour and turbidity in a water body may be derived from either external (allochthonous) or internal (autochthonous) sources. Although colour and turbidity in a water body may have a variety of sources, the areas in which destratification is relevant are somewhat narrower. For example, destratification cannot be expected to reduce the input of fine clay from an eroded catchment. This report shall consider only colour associated with the degradation of organic material and colour associated with iron and manganese. With regard to turbidity we shall consider the contribution of organic particles such as phytoplankton cells, and the flocculation or suspension effects of aeration.

Refer to : 5 Releases from the Sediment
5.6 Turbidity and colour
6 External Inputs
6.3 Turbidity

2.5 Variable Water Quality

Large fluctuations in water quality over a relatively short period of time may create significant operational difficulties. These variations may occur either on a seasonal basis (eg as a result of overturn), or on a shorter time scale, (hours to days) as a result of processes such as internal seiching. Other sources such as inflow and storm events are also responsible for difficulties. It is the first two processes though, that are most closely linked with stratification and where destratification offers potential for a lessening of water quality problems.

Refer to : 3 Stratification
3.2 Water Quality Variations.

2.6 Unfavourable Temperature and Dissolved Oxygen

Dissolved oxygen and temperature are most commonly used as indicators or measures of stratification. Although water quality problems associated with stratification are more commonly related to other water quality characteristics, both temperature and dissolved oxygen can also impact on water quality directly. For example, releases of poorly oxygenated or excessively high or low temperature water are rarely desirable due to adverse impacts on downstream flora and fauna. Destratification may be used to ensure these parameters are maintained at favourable levels.

Refer to : 8 Temperature and Dissolved Oxygen

2.7 Aquatic Insect Numbers

Large numbers of aquatic insects in a water supply may cause considerable problems. Such animals present in water at the point of consumption give rise to consumer complaint. The most notable example in Australia is that of phantom midge larvae in Myponga Reservoir. Changes in the water column structure by artificial aeration successfully reduced this problem.

Refer to : 7 Aquatic Animals
7.2 Aquatic Insects

2.8 Fish Numbers

Poor water quality, expressed as either low dissolved oxygen or the accumulation of reduced compounds, may lead to a reduction in fish numbers. In lakes where a decline in dissolved oxygen may directly or indirectly lead to fish kills or reduced fish growth, artificial destratification may play a desirable role.

Refer to : 7 Aquatic Animals
7.3 Fish Numbers.

CHAPTER 3

3.1 Development of Stratification

Stratification occurs commonly in many lakes, reservoirs and estuaries throughout Australia and is usually taken to describe the situation where the water column develops discernible layers, which impart a stability to the depth profile.

Typically two forms of stratification are considered, chemical and thermal (Symons *et al* 1970, Wetzel 1983). Chemical stratification is common in estuaries as intruding saline water flows in under fresher water. In lakes and reservoirs thermal stratification is more common and arises from seasonal temperature changes. During winter and early spring, lake water columns typically display a uniform temperature profile and are easily mixed by wind action (there are exceptions especially some very deep lakes). As atmospheric temperatures increase with advancing seasons so do surface water temperatures. This allows the development of a density gradient down through the water column which eventually exceeds the winds ability to mix the water body. This is the type of stratification which has been addressed most often by destratification techniques in Australia, and the type with which this report shall deal. Figure 3.1 presents a typical thermal stratification profile of a lake with the commonly accepted subdivision into three layers.

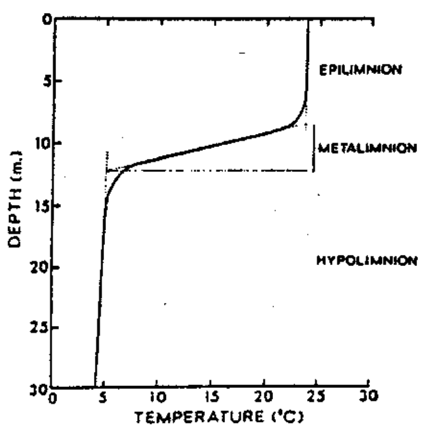

Figure 3.1 Typical thermal stratification of a lake into epilimnion, metalimnion and hypolimnion (Wetzel 1983).

The *epilimnion* is the uppermost layer of fairly uniform water. This water is usually warm, well mixed (by wind action) and fairly turbulent. This is the layer in which wind is still able to circulate water.

The next layer encountered is the *metalimnion* or *thermocline*. According to Wetzel (1983), this area is defined as "the water stratum of deep thermal gradient demarcated by the intersections of the nearby homiothermal epilimnion and the hypolimnion". In other words it is the region of most rapid temperature change lying between the almost uniform epilimnion and hypolimnion.

The *hypolimnion* is the bottom layer of water. Hutchinson (1957) defines this as the "deep, cold and relatively undisturbed region". This is the critical parcel of water that is trapped in contact with the sediment during stratification. It is in this layer of water that those substances released from the sediment are able to accumulate prior to overturn. Overturn is the period when stratification breaks down, leading to mixing throughout the entire water column.

3.2 Water Quality Variations

3.2.1 Overturn

As the progression of seasons continues after summer stratification, atmospheric temperatures fall. Surface waters may now cool and promote an overturn mixing of the water column. Wetzel (1983) describes the erosion of stratification by natural overturn as follows: "Surface waters cool and become more dense than underlying warmer epilimnetic water. As the surface water sinks, it is mixed by a combination of convection currents and wind - induced epilimnetic circulation. The penetration of surface waters into the metalimnion continues as the lake continues to cool. .... A progressive erosion of the metalimnion from above can be observed as the stratum of thermal discontinuity is reduced and the homiothermous epilimnion increases in thickness". In warm temperate and tropical climates (such as in most of Australia) the onset of inflow in winter also contributes to natural destratification and may be the dominant overturn process in some cases. This naturally occurring sequence of events has created difficulties at various times throughout Australia. At least three cases exist, where destratification has been initiated, in an attempt to stabilise water quality entering water treatment plants. Such plants typically face operational difficulties when large fluctuations in water quality occur. The redistribution of elements such as iron and manganese following accumulation and overturn can create such variations.

High seasonal variability in the levels of iron, manganese, sulphides, colour and turbidity have caused difficulties for the treatment plants associated with:

Chichester Dam in New South Wales	-	Case 43
Running Creek Reservoir in Victoria	-	Case 10
Woodford Creek Reservoir in New South Wales.	-	Case 28

In each case the variation was due to natural stratification and overturn, and in each case the use of artificial destratification was able to provide more stable conditions.

3.2.2 Internal seiching

The "internal seiche phenomenon" may create difficulties similar to those of overturn yet on a shorter time scale. The process of internal seiching in North Pine Dam allowed the layers of water established under stratified conditions to "tilt" in response to certain winds. This tilting can result in the deeper, poorer quality water being brought into draw-off levels for short periods during a 24 hour cycle. This may give rise to sudden variations in the levels of iron and manganese entering any associated treatment plant or water supply system. As reported by Brady and Madden (1978) this phenomenon created significant operational difficulties for the water treatment plant associated with North Pine Dam (Case 49).

At times the upper water layer (epilimnion) in North Pine Dam has also supported large crops of algae . This surface water is then made unavailable for withdrawal due to associated taste and odour problems. The coupling of these two problems, variable manganese levels and high algae crops, has often created great difficulties in providing high quality water from North Pine.

Artificial aeration trials were conducted at North Pine in an attempt to remove both problems, especially variation in manganese levels. These trials provided only limited success. The system still stratifies, yet operational difficulties have been eliminated through a different approach, namely the development of a manganese monitoring system in the treatment plant.

3.3 Recognising Stratification

Stratification is most readily perceived by the development of a varying temperature profile in the water column. Yet as discussed in Chapter 2 stratification may display a wide variety of water quality symptoms.

Temperature stratification itself creates few difficulties. As pointed out by Brown (1986) "Thermal stratification by itself has little effect on subsequent water use. The concern is that thermal stratification is usually closely associated with the development of chemical and biological stratification in the water body and it is these processes which contribute to water quality problems". Thus the recognition of stratification in forms other than temperature is important.

Stratification is best taken to have occurred whenever there is qualitatively significant partitioning of the vertical water column. It may even be appropriate to extend the definition of stratification beyond the vertical water column and into the horizontal plane of the reservoir. The objectives of the artificial aeration of Avon Dam in New South Wales, included not only circulation through the water column, but also the circulation of water from lower reaches, to the poorer quality upper reaches of the reservoir. This application of artificial aeration attempted to address variations in both the vertical and horizontal planes.

To prevent stratification the development of stability in the water column must be recognised early. The critical water quality characteristic reflecting stratification will vary from one water body to another. Thus for a water body experiencing algae difficulties due to microstratification rather than nutrient release (eg Little Para Reservoir), monitoring of temperature at close intervals of both depth and time may be required. For other systems, especially those concerned with the release of reduced substances from the sediments, dissolved oxygen is the critical parameter.

It is quite possible for stratification with respect to dissolved oxygen to form much more markedly than temperature stratification. This may even occur while circulation still continues. Provided the oxygen demand of the sediment and water column exceeds the amount of oxygen supplied from the upper layers, dissolved oxygen levels decrease at depth (Rosich and Cullen 1981).

Thus for many systems the monitoring of, and response to, dissolved oxygen levels is of much greater importance than temperature profiles. Figure 3.2 displays a temperature and dissolved oxygen profile measured during the development of stratification in the Harding Reservoir in Western Australia. Clearly dissolved oxygen stratification has developed much more strongly than temperature stratification. Over a depth of 19 m temperature falls a total of only 1.4 degrees, while dissolved oxygen saturation falls 71%. The operation of an aeration strategy on the basis of a temperature differential of even a few degrees would be entirely inappropriate in such a system.

To minimise the benefits of artificial mixing the development of strong and stable stratification must be prevented. Destratification of an already stratified water body is inefficient since more energy is required to remove the established stability. A further disadvantage is that this approach allows for increased opportunity for development of water quality problems. The destratification trials performed at North Pine Reservoir exemplify these two points. Destratification was easily achieved in this system if aeration was commenced early enough to prevent the formation of strong stratification. It was however, difficult to break down a well established stratification regime. The destratification of a well formed hypolimnion in North Pine may have contributed to "larger than usual" phytoplankton blooms. This could occur through providing nutrients, accumulated under reduced oxygen conditions, to the euphotic zone. Aeration to prevent the initial decline in dissolved oxygen may well eliminate such difficulties.

Stratification must be viewed in terms of those parameters most relevant to a given situation. It should be a flexible term and conceptualised as such. In this way stratification may be recognised and addressed in a manner appropriate to the particular water body in question.

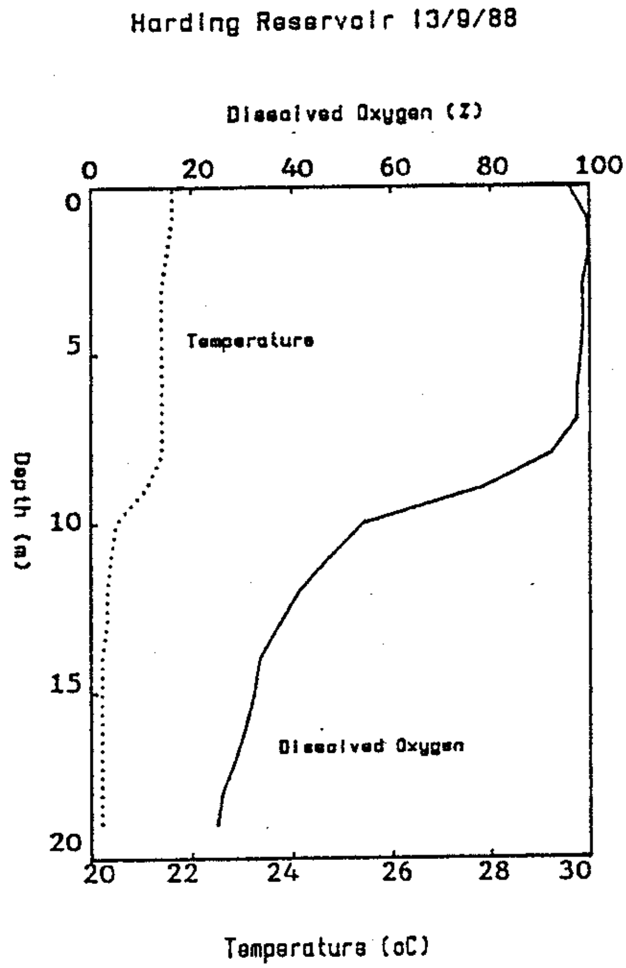

Figure 3.2 Temperature and dissolved oxygen profiles measured in Harding Reservoir during the early development of stratification.

3.4 Stratification/Destratification Models

3.4.1 Introduction

A variety of different approaches may be adopted in designing an appropriate artificial destratification system. The three systems presented in this report represent that which appears to be the most well-known and commonly used method (the Davis Method) and the two most recent developments in optimisation of destratification systems ("DEPLE" and "DYRESM"). We do not evaluate the performance of these mathematical interpretations, or necessarily recommend that which is most suitable. Instead, this section is included to stimulate thought and discussion on the merits of current designs and the usefulness of applying mathematical models to optimise efficiency. There can be little doubt that many destratification systems could benefit from a more quantitative evaluation of performance and design. As pointed out by Brown (1986) "Poor communication between the various professional groups involved and the pursuance of different objectives has led to a proliferation of designs and many have not been based on sound scientific or engineering principles. Poor design, or a lack of understanding of the basic principles involved is often the underlying reason for poor performance."

3.4.2 Davis method

The method outlined by Davis (1980) aims to provide an "empirical design procedure which can be used to identify the length of perforated pipe and compressor size needed for a particular reservoir". The end point of this empirical procedure is to begin mixing at the first signs of stratification and maintain complete mixing throughout the entire water column over the summer period.

The method aims to satisfy two main requirements:

1. "After the onset of thermal stratification in spring or early summer the device must be capable of mixing the greater part of the reservoir volume in a reasonably short time (say 5-10 days depending on reservoir volume) so that approximate isothermal conditions exist over the depth;

2. During operation of the device the oxygen demands within the water column should be met initially through mixing between the upper and lower water layers and in the longer-term through surface re-aeration".

There are basically 9 steps in designing the aeration system:

1. Obtain surface area and reservoir volume as a function of depth.

2. Choose a typical temperature profile from spring or early summer, ie. a temperature profile from when the aerator would first be turned on during the year.

This is one area where the model may fail to successfully simulate aerator performance over a period of years in the reservoir. As Davis (1980) states, "the choice of temperature profile ultimately decides the size of the compressor, length of perforated pipe and the time required to achieve destratification." Such critical dependence is unfortunate given the variable nature of Australian reservoirs, a "typical" profile may be repeated rarely over a number of years.

3. Calculate the stability of the reservoir. Here the model uses the definition of Symons and Robeck (1969) and calculates stability as

potential energy of a mixed system (PEM) -
potential energy of a stratified system (PES)

stability = PEM - PES(J)

4. Calculate the theoretical energy (E) required to destratify the reservoir.

E = stability and heat energy input from solar-radiation
energy from wind-induced mixing.

5. Calculate the free air delivery required at the compressor.

Davis (1980) bases this on a relationship obtained from field experiments.

$$\frac{\text{energy input by device to destratify reservoir}}{\text{total theoretical energy required to destratify reservoir.}} = 20$$

6. Calculate the length of perforated pipe required.

   This also is based on a relationship obtained from field experiments. It assumes a hole spacing of 0.3 m and a hole diameter of 0.88 mm.

7. Specify the air pressure requirements at the compressor and determine internal pipe diameter.

8. Check that the length of perforated pipe is sufficient to meet the specified air delivery.

9. Calculate the required anchor weights.

This method is well known and has been used in the design a number of Australian destratification systems. Brown (1986) suggests that this method appears to underestimate destratification time by about 11% although at the time data were only available for storages up to about 100 000 ML. Given this limitation the method may still provide a useful tool for the design of artificial aeration systems.

Brown (pers comm) indicates that : "There is no reasons to believe that systems designed in accordance with the Davis method would fail to perform in a predictable fashion when they were used in storages with volumes greater than 100 000 ML". The methods' main problems appear to lie in choosing an appropriate typical temperature profile.

3.4.3 DEPLE

As the name suggests "DEPLE" is a stratification and destratification model centred on oxygen depletion rates. This model was developed by The Electricity Commission of NSW (ELCOM) and runs on their VAX mainframe computer system.

The model aims to:

a) simulate the stratification of reservoirs by accurately predicting dissolved oxygen and temperature profiles;

b) simulate the effect of destratification measures by relating the increased circulation to these dissolved oxygen and temperature profiles until an acceptable water quality criteria has been met (ELCOM 1988).

The model receives input in the form of depth:area:volume characteristics and initial dissolved oxygen and temperature profiles. From these profiles the water column may be divided into epilimnion, metalimnion (thermocline) and hypolimnion. This information as well as wind speed, air temperature and humidity are supplied to the computer.

The model then utilises Lagrangian layered flow to evaluate the impact of water circulation on oxygen depletion or supply rates. The final input is then to specify the time period, or dissolved oxygen range, over which simulation is required. Due to the natural variation in climatic conditions the tendency is that the shorter the time period specified the more accurate the simulation will be. It is also necessary to identify any internal components that will affect dissolved oxygen levels, such as the input of sewage effluent.

From this information the model will produce "initial conditions and oxygen totals, final conditions and oxygen totals and the initial and final DO% saturation profiles at up to five stations in the reservoir" (ELCOM 1988). These data may then be used to derive oxygen depletion rates within the reservoir.

These oxygen depletion rates may in turn be used to simulate destratification using either of two methods: Lagrangian flows are identified by jet or Garton pumps or a table relates air-diffuser-induced Lagrangian flows. From this stage, profiles over a given time period or a dissolved oxygen range may be generated.

ELCOM have used this model extensively as a management tool in existing destratified reservoirs and as a predictive tool to estimate the need for an aerator prior to dam completion.

"DEPLE has been validated within 8% of measured dissolved oxygen concentrations at Lyell Reservoir, Wallerwang Reservoir, Liddell Freshwater Dam and Liddell Cooling Pond. It has been used to simulate the depletion and destratification of reservoirs at Healey Damsite, Plashett Damsite, Thompsons Creek Damsite and further studies at Lyell Reservoir (ELCOM 1988). Figure 4.3 shows dissolved oxygen contours simulated in Lyell Reservoir by "DEPLE".

3.4.4 DYRESM

Investigations are being carried out at the Centre for Water Research (The University of Western Australia) into the optimisation of effective artificial destratification systems. The Dynamic Reservoir Simulation Model, DYRESM (described by Imberger *et al* 1978), has been successfully modified to incorporate the dynamics of a bubble ploom's passage through a water column.

DYRESM is based on a Lagrangian scheme in which the water body is described as a series of horizontal layers of varying thickness. As explained by Patterson and Imberger (1988), "mixed layer deepening is modelled as convective overturn resulting from surface cooling, wind stirring at the surface, seiche induced shear at the pycnocline (the depth of most rapid change in density), and billowing at the pycnocline resulting from shear instability. Turbulent transport in the hypolimnion is modelled by a diffusion-like process, with an eddy diffusivity depending on the local density gradient and the rate of dissipation of turbulent kinetic energy. Inflow and outflow are confined to narrow regions adjacent to the insertion level and offtake, after, in the case of inflow, underflow and entrainment from the main body of the reservoir......"

"The model requires daily total values of river discharge, river temperature, offtake discharge, incident long and short wave radiation, and daily average values of air temperature, rainfall, humidity and windspeed." To simulate the effects of artificial aeration the model also requires the airflow rate for each plume, the number of plumes and the depth of each aerator.

Experiments conducted by Asaeda and Imberger (1988) indicated that the effectiveness of an aerator will be maximised "at the value of airflow ideal for the particular stratification". This maximum efficiency can be characterised by the dimensionless parameter NmH^3/Q where N is the Vaisala frequency of the stratification, H is the total depth, and Q represents airflow.

When this value is less than 3×10^4 a strong horizontal intrusion forms destratifying the area quickly. When the parameter lies between 3×10^4 and 3×10^5 several such intrusions form and maximum efficiency is obtained. When this parameter is greater than 3×10^5 the intrusions are weak and destratification is slow (Asaeda 1987).

Using this optimisation concept DYRESM has been modified with a view to determining design and operational guidelines for destratification systems.

Initial validation of this destratification model was carried out at Myponga Reservoir in South Australia. Figure 3.4 displays a comparison between temperature profile data measured in the reservoir and the results of the models' simulation of this period. As described by Patterson and Imberger (1988) these figures show "that the major features of the thermal structure have been reproduced. In particular, the timing of the onset of stratification, the heating events in mid November and early December, the rapid heating in late December and early January, the cooling at the end of February and again at the end of April, and the overturn in mid May are all predicted accurately by the model. The gradual deepening of the thermocline following the end of February indicated by the data is not reproduced by the model, which fixes the thermocline at the level of the aerator. This gradual deepening is possibly the result of convective overturn initiated by continued surface cooling, the model result, however, shows a period of heating during this period. This mismatch may be the result of the use of remote meteorological data. With this exception, the model has evidently accurately reproduced the dynamics of the reservoir."
This model has been used to design an aeration system for the Harding Reservoir in the north of Western Australia which aims to optimise aeration through

obtaining the greatest efficiency of energy input. Prior to installing the aerator simulations were run using field data with variations of port size, spacing and number as well as airflow rate. In this way the best performance of an aerator may be established before the aerator is introduced to the reservoir. Figure 3.5 gives an example of an efficiency simulation run on Harding Reservoir, evaluating the performance of differing designs of aeration in maintaining destratified conditions.

3.4.5 Horizontal flow and destratification

Both of the models mentioned above are one-dimensional simulations of reservoir dynamics. Artificial destratification, however, relies on mixing in both the vertical and horizontal planes. All of the destratification systems reviewed in this report (Appendix 1) have relied on one or two localised mixing sources. Momentum forces acting near these sources create uniformity of the water column in the immediate vicinity of the mixing sources. Mixing in the horizontal direction then results from horizontal currents driven by density differences between the localised uniform area and the far reaches of the lake.

The simulation model DYRESM has recently been modified to incorporate two dimensional mixing parameters.

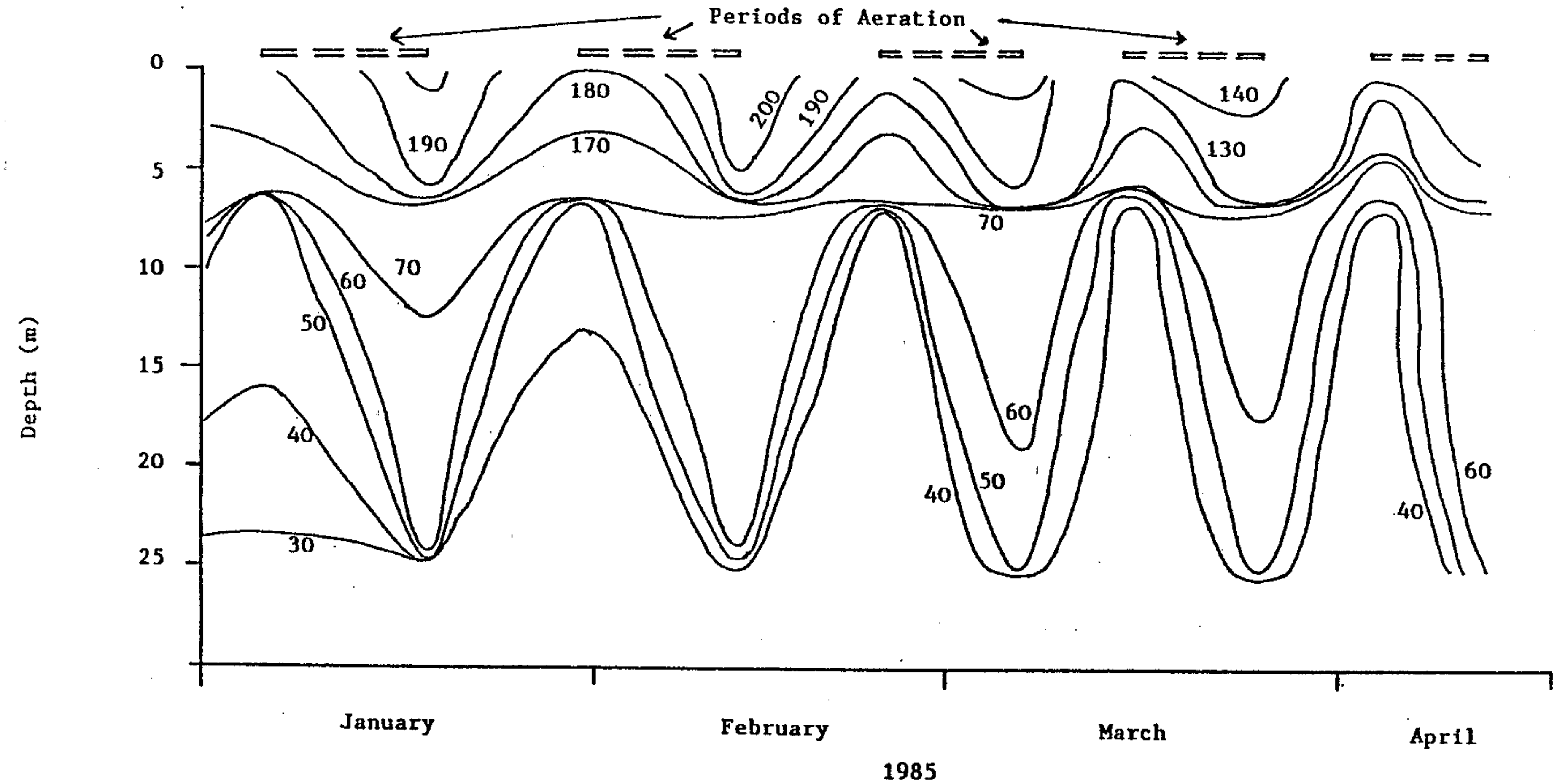

Figure 3.3 Dissolved oxygen (% saturation) contours derived from simulations using "DEPLE" in Lyell Reservoir (Elcom 1988)

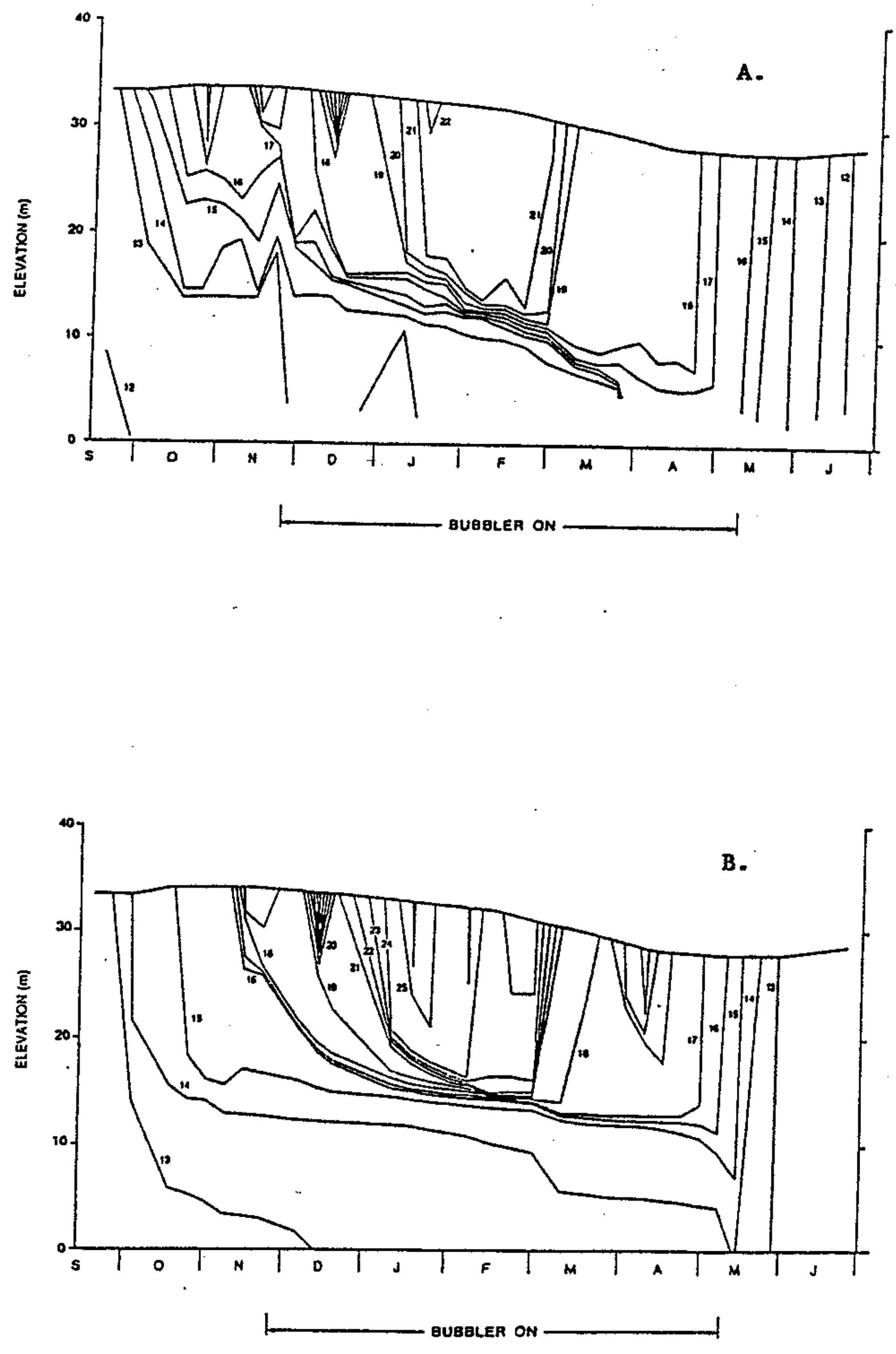

Figure 3.4 A comparison between temperature data measured in Myponga Reservoir during 1980/81. A. and a "DYRESM" produced simulation for the same period B. (Patterson and Imberger 1988).

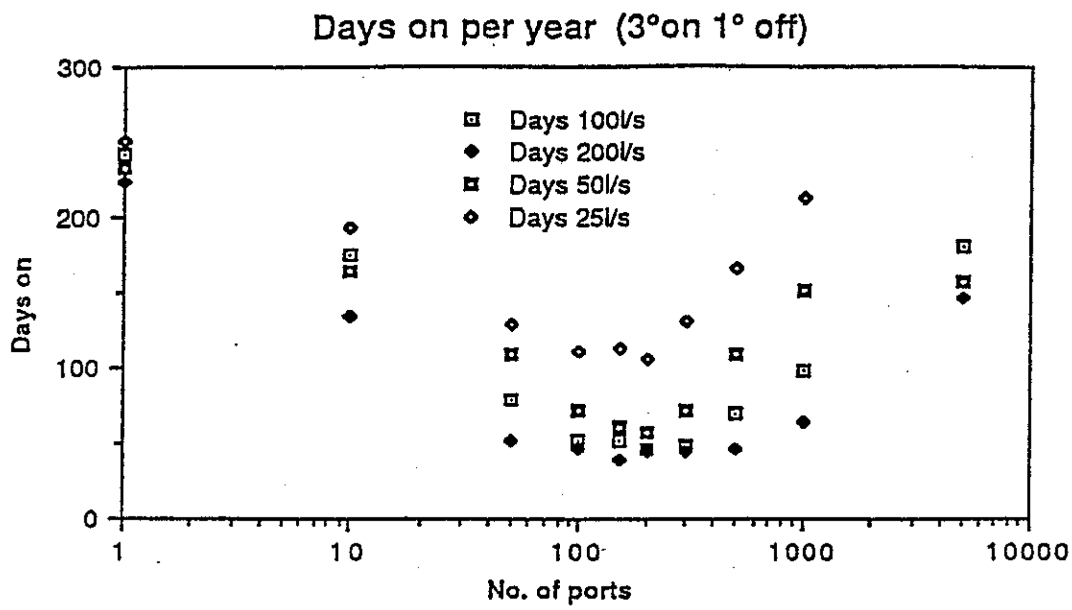

Figure 3.5 A "DYRESM" efficiency simulation run predicting the number of days a particular aeration system (with varying ports and airflow), would operate per year if installed in Harding Reservoir based on turning the aerator on in response to a 3 C temperature differential and off at a 1 C differential (Centre for Water Research 1988).

CHAPTER 4

ALGAE GROWTH

4.1 Algae Forms

The majority of algae problems addressed by destratification relate to phytoplankton populations. The term phytoplankton refers to free floating, essentially microscopic aquatic algae. Though some genera do possess motility it is limited, preventing them from swimming against currents. In the majority of healthy water bodies throughout the world, a number of planktonic algae species will coexist. Typically there is a balance allowing a variety of species to survive with one or two dominant species at any given time.

As discussed by Wetzel (1983) many attempts have been made to correlate phytoplankton type to the health or trophic status of a lake. Such comparisons are frequently modified and should only be used as a guide. They do however serve to illustrate some basic trends. Table 4.1 presents a general comparison of algae characteristics and lake trophic status.

Trophic Status	Water Characteristics	Dominant Algae	Other commonly occurring algae
Oligotrophic	Nutrient poor Weakly productive	Desmids Diatoms Chrysophyceae	A wide variety or mixed suite
Mesotrophic	Average productivity	Dinoflagellates eg *Peridinium* *Ceratium*	Still a Balanced variety
Eutrophic	Nutrient enriched Highly productive	Diatoms and Blue-Greens especially *Microcystis* *Anabaena*	Many other algae especially greens and blue-greens

Table 4.1 Typical phytoplankton associations in relation to Lake trophic status (Hutchinson 1957, Wetzel 1983).

The distinction between phytoplankton and larger filamentous algae forms (macroalgae) is somewhat arbitrary. Some filamentous algae (such as *Anabaena*) are accepted as belonging to phytoplankton, while others such as *Spirogyra* and *Cladophora* are not. The distinction is based on size and should not necessarily be taken to reflect an increased or decreased potential for water quality problems. Many macroalgae genera, such as *Cladophora* and *Spirogyra*, have been associated with the formation of tastes and odours. Filamentous algae can also cause operational difficulties in water treatment processes through filter clogging. Both rapid and slow sand filters may be clogged by filamentous algae.

As with phytoplankton, it is common for nutrient enrichment (especially in shallow waters) to manifest itself in increased macroalgae numbers. The distinction is best viewed as phytoplankton being readily found in the upper layers of a water column, while the larger macroalgae, because of their greater size, are confined to bottom or near-shore areas.

4.1.1 Blue-green algae (Cyanophyta)

Blue-green algae are similar to bacteria (sometimes termed cyanobacteria) in that they are procaryotic (lacking a membrane defined nucleus). They may occur in unicellular, filamentous (eg *Anabaena*, *Rivularia*) or colonial forms (eg *Microcystis*, *Merismopedia*).

A number of blue-green algae are notorious for causing water quality problems through the release of tasteful, odourous and sometimes toxic substances. It is noteworthy that these algae are tolerant of environmental stress and can withstand both high and low temperatures.

Several species (eg *Anabaena*) are also notable in their ability to utilize atmospheric nitrogen. This "nitrogen fixing" ability provides certain survival advantages to these algae. In water bodies poor in nitrogen, yet having sufficient levels of other nutrients such as phosphorus, these algae commonly dominate. Non-nitrogen fixers require dissolved nitrogen in other forms such as ammonium or nitrate and may fail to grow while blue-green algae flourish (Waite 1984, Wetzel 1983).

4.1.2 Green Algae (Chlorophyta)

Green algae represent a large and diverse section of phytoplankton and macroalgae. As with blue-green algae they are represented by unicellular (eg *Chlamydomonas*), filamentous (eg *Cladophora*) and colonial forms (eg *Volvox*).

Many planktonic species pose similar difficulties to blue-green algae through imparting tastes and odours to surrounding waters. For example, *Chlamydomonas* and *Dictyosphaerium* are known to impart a "grassy" taste to water supplies when present in moderate numbers (Bowles and Saunders 1986, Palmer 1962). Varieties of the larger filamentous forms may create similar difficulties. They also pose additional problems through their tendency to accumulate on shorelines, where they may die and rot, giving rise to highly disagreeable odours.

4.1.3 Diatoms (Bacillariophyceae)

The most distinguishing characteristic of diatoms are cell walls constructed primarily of silica. These "shells" form the basis for classification within diatoms. This shell does not readily decompose and therefore creates difficulties in water treatment operations due to filter clogging. Even though the diatoms may die when trapped by the filter, the silica based shell will remain and obstruct flow. Palmer (1962) attributed diatoms the dubious distinction of "the most important group of organisms that clog filters". *Asterionella, Fragilaria, Tabellaria, Synedra, Navicula, Cyclotella* and *Melosira* are most notable in this regard.

As with other groups of algae diatoms may also be responsible for taste and odour production. "*Asterionella* is considered one of the worst offenders among the diatoms, having a characteristic aromatic, geranium-like odour that changes to fishy when the alga is present in large numbers. *Tabellaria* produces a similar effect, while *Synedra* has an earthy to musty odour and *Stephanodiscus* is blamed for a vegetable to oily taste, with very little odour" (Palmer 1962).

4.1.4 Dinoflagellates (Dinophyceae in Pyrrhophyta)

Dinoflagellates are "unicellular flagellated algae, many of which are motile. Although a few species are naked or without a cell wall, most develop a conspicuous cell wall that is often sculptured and bears large spines and elaborate cell-wall processes" (Wetzel 1983).

Although most are quite susceptible to changes in parameters such as pH and temperature, the more troublesome species are not. For example *Ceratium*, *Glenodinium* and *Peridinium* are highly tolerant to pH and temperature extremes and exist in a wide variety of lakes throughout Australia (Wetzel 1983). *Ceratium*, *Glenodinium* and *Peridinium* (among others) are responsible for unpleasant tastes and odours in water bodies. (Bowles and Saunders 1986, Palmer 1962).

4.1.5 Golden-brown algae (Chrysophyceae)

The name comes from the often distinctive golden brown colour of these algae which is caused by the presence of certain carotenoid pigments. These algae are in the main unicellular, with few filamentous or colonial forms. The contribution these algae make to total phytoplankton numbers varies widely with prevailing environmental conditions. This group may also cause taste and odour difficulties, with a number imparting a fishy taste and odour to water when present in abundance.

The remaining groups of algae appear to play little or no role in generating water quality problems in Australian water supplies. A detailed list of taste and odour producing genera is contained in Section 4.6.

4.2 Nutrient Supply

Algae require both inorganic and organic nutrients to survive. For the majority of algae, phosphorus and nitrogen are the nutrients which most regularly limit growth in water bodies (Schindler *et al* 1977; Thomas 1973; Wetzel 1983). Diatoms are somewhat of an exception as they may be restricted by the supply of silica, although in most freshwater systems silica is usually in ample supply.

Wetzel (1983) suggests that if nitrogen is in excess, then an addition of phosphorus can theoretically generate "500 times its weight in living algae", likewise nitrogen may generate a 70 times increase. It is usually found that essential micronutrients such as potassium, sulphur, iron and zinc are available in ample quantity in freshwater systems.

4.2.1 Phosphorus

Weakly productive (oligotrophic) lakes are often limited by phosphorus, while nitrogen is available in excess. An increased phosphorus loading can thus lead to such lakes becoming more biologically productive with associated water quality problems.

Many examples of increased phosphorus supply leading to enhanced algae numbers exist throughout the world's lakes. A study conducted by Gilliom (1984), on phosphorus-limited lakes in the Puget Sound region of the USA, found a significant correlation between algae productivity (as measured by chlorophyll *a*) and water phosphorus levels. Thomas (1973) investigated the impact of phosphorus addition to the waters of Lake Zurich and concluded that an increased addition of phosphate to the lake would result in an "increase in bacterial content, increase in oxygen demand and an increase in growth of algae". Thomas concluded that to "avoid algae damage in lakes we need to reduce the supply of phosphate". Similar relationships have been identified in Australia. For example, the importance of phosphorus with regards algae growth has been identified in Canberra's urban lakes (Cullen and Rosich 1978, Rosich and Cullen 1981). The regular blooming of the blue-green alga *Nodularia* in the Peel-Harvey estuarine system of Western Australia has also been linked with phosphorus inputs to the water column (Hodgkin *et al* 1980).

Phosphorus may be derived from internal and external loading. Typically, external loading is subdivided into point and non-point source contributions. Sewage outfalls, piggery discharges, stock holding pens and industrial effluents are common examples of point sources of phosphorus. Non-point source pollution typically comes from runoff and groundwater input. Phosphorus run-off from an agricultural catchment has led to the eutrophication of the aforementioned Peel-Harvey system (Hodgkin *et al* 1980).

Internal inputs of phosphorus to reservoir waters come from sediment cycling processes. The sediments are an important buffering medium controlling the phosphorus concentration in the water column of many lakes. Eutrophication is sometimes considered "irreversible" because of continuing phosphorus release from the sediments. This loading has in the past led to situations where no decline in trophic status has been observed after the elimination of external nutrient inputs. For example, Ahlgren (1977) found that the sediments of Lake Norrviken, Sweden, were capable of releasing enough phosphorus to impede lake recovery, even after external loading from wastewater effluents had been completely eliminated for 10 years.

Such internal inputs of phosphorus are commonly stimulated by the development of an anoxic hypolimnion, resulting from stratification. This process is discussed further in Chapter 5, which deals with releases from the sediment.

4.2.2 Nitrogen

Nitrogen is accepted as less frequently being a limiting nutrient in freshwater systems. Nitrogen limited algae populations do still occur, as evidenced by an area of Lake Burley Griffin receiving an extremely high phosphorus load (Cullen and Rosich, 1978). In such a situation algae may respond to increased nitrogen inputs, with increased growth. It is useful, then, to have an understanding of the available sources of nitrogen to algae.

The earth's atmosphere is made up of approximately 78% molecular nitrogen (N_2) and acts as a sink for inorganic nitrogen. This nitrogen may be extracted from the atmosphere by "nitrogen fixing" blue-green algae, while inorganic nitrogen compounds (chiefly nitrate) may be taken from the water and transformed into nitrogen containing organic compounds by other algae. This organic nitrogen then passes back to the water and sediment through the excretions or death of organisms. Bacteria in turn convert these substances to inorganic nitrogen compounds, which may again pass to the water column or atmosphere. This cycling of nitrogen consists primarily of the processes of nitrogen fixation, assimilation, ammonification, nitrification, and nitrate reduction.

Nitrogen fixation is the conversion of molecular nitrogen (N_2) to ammonia. Many species of blue-green algae are able to fix nitrogen from the atmosphere. In nitrogen-poor situations this gives such algae a competitive advantage over other species.

Assimilation of nitrogen occurs as plants take up inorganic nitrogen from the water column or sediments. Apart from molecular nitrogen, four forms of nitrogen are readily available for assimilation; nitrate, nitrite, ammonium and urea (Anon 1983).

Ammonification is the process of microbial degradation of organic nitrogen to release inorganic nitrogen (ammonia). Thus the nitrogen bound in the tissues of dead plants and animals is returned to the water column as inorganic nitrogen, and hence made available for re-assimilation.

Nitrification is the process whereby ammonia or ammonium ions (from ammonification) are oxidised to nitrite and then to nitrate. Together with ammonification these steps make up the process of "mineralisation", which returns organic nitrogen to the water column as nitrate.

Nitrate reduction encompasses both the reduction of nitrate to molecular nitrogen, (denitrification returning nitrogen to the atmosphere) and reduction within the cell for assimilation.

These processes combine to ensure the provision of nitrogen for algae growth in a water body. The presence of the large pool of nitrogen in the atmosphere and the ability of certain algae to utilize this pool, play a significant role in establishing phosphorus limitation in many freshwater systems.

4.2.3 Nutrient ratios

Robin, South and Whittick (1987) (based on the work of many authors) suggest that the cellular nitrogen to phosphorus ratio (N:P) for "normal phytoplankton" populations ranges from 5:1 to 15:1. It is suggested that phytoplankton usually display nitrogen limitation up to an N:P ratio of around 30. Phosphorus limitation may occur at higher ratios. Carbon ratios should also be considered. The Redfield ratio of 106:16:1 describes the cellular carbon:nitrogen:phosphorus (C:N:P) ratio of phytoplankton growing in a nutrient rich environment. Cellular nutrient ratios greatly removed from this value suggests nutrient limitation, with a lower cell quota for the limiting nutrient (Vollenweider 1968).

Cellular nutrient ratios do not always reflect concentration ratios in the water body. As mentioned previously the ability of certain algae to fix atmospheric nitrogen may impinge on the effect of water body nutrient ratios. A relatively low N:P ratio would suggest nitrogen limitation, yet not in the case of nitrogen fixing, blue-green algae. These algae gain a competitive advantage in such a situation. Where low N:P ratios in a water body limit the growth of most algae, some blue-green algae are able to thrive by utilising dissolved phosphorus and atmospheric nitrogen.

4.3 Temperature Effects

Algae are able to survive successfully over a wide range of water temperatures. The extremes that will limit photosynthesis depend on the particular species in question. Certain diatoms display a minimum temperature for growth of around 5 C while for others this value is 15 C. Both green and blue-green algae typically require higher temperatures for optimum growth. For sustained photosynthesis, Wetzel (1983) allows the blue-green algae the distinction of being much more resistant to higher temperatures than the other groups of algae. Certain thermophilic (heat loving) algae have optimum growth rates at temperatures in excess of 45 C. This group is almost exclusively blue-green in character, and may photosynthesise at temperatures as high as 74 C (Wetzel 1983).

Recent studies by Robarts and Zohary (1987) indicate temperature may significantly influence the growth rate of bloom-forming, blue-green algae. The optimum temperature for the genera investigated (*Anabaena*, *Aphanizomenon*, *Microcystis* and *Oscillatoria*) was usually around 25 C. The effect of low temperatures varied among the groups. *Microcystis* was found to be restricted greatly at temperatures below 15 C, while *Oscillatoria* proved more tolerant and was able to grow at temperatures below 10 C. The critical minimum temperatures for *Anabaena* and *Aphanizomenon* were found to lie between the values obtained for the other two genera.

These authors also concluded that "direct temperature effects were secondary to indirect temperature effects (mixing and nutrients) in determining the dominance of bloom-forming cyanobacteria in lakes. Direct temperature effects probably act synergistically with other factors in this process" (Roberts and Zohary, 1987).

Given constant predation rates and a relatively constant nutrient supply, the interaction of temperature and light, as determined through mixing processes, may be expected to account for the vertical and even seasonal variation in the algae character of a water body.

4.4 Light and Algae Growth

The process of photosynthesis converts light energy into a form usable by algae. This process is essential to the growth of a cell, and hence to the algae population. The general form of the photosynthetic process is that carbon-dioxide, light energy and water is converted to molecular oxygen, a (CH_2O) compound, (upon which carbohydrates are based), and water. The equation shown below describes the photosynthetic production of glucose.

$$6CO_2 + 12H_2 + \text{light} \xrightarrow{\text{chlorophyll}} 6O_2 \; C_6H_{12}O_6 + 6H_2O$$

Given this relationship, it is obvious that light may greatly limit the growth of algae. Although light is a prerequisite for algae growth, it can also prove harmful. High light intensities can retard growth and eventually destroy the cell, this phenomenon is known as "photoinhibition".

More obvious is the failure of algae to grow at low light intensities. Under such conditions the amount of light received by the cell can be so low that the consumption of energy by respiration exceeds the production of energy by photosynthesis. This critical balance is reached at an intensity known as the "compensation point", a value which varies among species. Algae are able to survive for a time at intensities below this value, but prolonged exposure to light intensities below the compensation point will retard growth and eventually cause death.

This requirement gives rise to a distinct vertical distribution of algae through a water column. Typically photosynthetic rates, and hence algae growth, display a maximum just below the water surface (away from photoinhibition) and slowly decrease with depth due to decreased light penetration. Exceptions to this general pattern do occur. For example, at times deeper algae populations adapted to low light may exhibit photosynthetic rates equal to those in the epilimnion (Wetzel 1983).

This light dependence gives definition to a highly important layer of water, the "euphotic zone". This layer is described by Kirk (1983) as follows : "A useful, if approximate, rule of thumb in aquatic biology is that significant phytoplankton photosynthesis takes place only down to that depth at which the downwelling irradiance of PAR (photosynthetically active radiation) falls to 1% of that just below the surface" (Kirk 1983, Talling 1957).

Naturally the time spent by an algae cell in this zone, or in a particular depth in this zone is important. In the same way that absolute levels of irradiance can have a compensation effect, so may the movement of cells in and out of the euphotic zone. It follows then that the degree of mixing and turbulence experienced by algae may be crucial in determining growth.

4.5 Effects of Mixing

The degree of mixing a lake receives may influence both the number of algae present and the type of algae assuming dominance. In general algae numbers may be considered to be determined primarily by nutrient supply, (Section 5.2), or by light limitation effects. Mixing effects on these parameters also determine to a large extent which type of algae assumes dominance.

Both Kirk (1983) and Tilzer (1987) describe the advantage that the buoyancy regulation exhibited by some blue-green algae offers to trapping available light. Many bloom-forming, blue-green algae (such as *Microcystis* and *Anabaena*) contain gas vacuoles which regulate buoyancy and allow cells to migrate vertically through the water column. In productive lakes under stratified or weakly mixed conditions these vacuoles allow accumulation of cells near the surface, thus maximising light harvesting and limiting the amount of light reaching less mobile species.

A critical balance may be reached between the depth of the euphotic zone and the depth of mixing, which may limit total algae growth and impinge on the advantage enjoyed by some blue-green algae. If the depth of the upper mixed layer is increased, through a process such as artificial destratification, phytoplankton are mixed through a greater depth of water and as such may receive a lower average light intensity. In this way the total amount of photosynthesis (energy production) of the population may be reduced. The rate of respiration (energy consumption) of the phytoplankton population will suffer little change due to the depth of mixing. This balance gives rise to the notion of a critical depth, or compensation depth "beyond which respiratory carbon loss by the whole population exceeds photosynthetic carbon gain and so net phytoplankton growth cannot occur" (Kirk 1983). Thus it may not be essential (disregarding sediment release processes) to destratify the entire water column to control algae populations. Mixing in order to provide a sufficiently high ratio of mixed layer depth to well-lit depth may well be sufficient. Talling (1971) found ratios of between 3:1 and 4:1 to be typical of light-limiting conditions in Lake Windamere.

Spigel and Imberger (1987) cite examples where a shallow mixed layer favours blue-green algae. In experiments conducted by Humphries and Imberger (1982), *Microcystis aeruginosa* was able to out-compete non-buoyant species when conditions of calm or shallow-layer mixing prevailed : "The deeper the photic depth relative to mixed layer depth, the higher the depth-integrated growth rate of the population" (Humphries and Imberger 1982).

A buoyancy regulation mechanism may also provide a competitive advantage through nutrient trapping under stratified conditions. Ganf (1982) investigated the reasons causing a succession from green to blue-green algae in Mt. Bold Reservoir in South Australia. He found that "if there is a separation between optimal depth of light and of nutrients for growth, the success of a species may depend primarily upon its ability to overcome successfully this spatial separation". The non-buoyant green alga *Dictyosphaerium pulchellum* was able to grow in Mt. Bold Reservoir only until stratification had developed. From this point onwards dominance was assumed by the buoyancy, regulating blue-green algae *Anabaena spiroides* and *Microcystis aeruginosa*. Experiments indicated that with the development and continuation of stratification, nutrient depletion occurred in the euphotic zone. The potential for water from this zone to support algae growth progressively declined. Under experimental conditions significant growth was measured only in water samples from below 16-20 m depth, yet at this time both *Anabaena* and *Microcystis* were able to proliferate in the shallow (2.5 m) deep euphotic zone. It would seem that buoyancy regulation allowed these blue-green algae to move below the euphotic zone, trap nutrients from the deeper layers and return to the well lit euphotic zone (Ganf 1982).

Destratification of the entire water column, or even only part of it, may be used to deepen the mixed layer and remove this spatial separation. Thus, the competitive advantage experienced by blue-green algae under stratified conditions can be nullified. It is also possible that pH changes resulting from artificial aeration may favour a change from blue-green to green algae and diatoms. This possibility is discussed in Section 5.6.

4.5.1 Intermittent mixing

To date destratification programs in Australia have involved either continuous mixing, "feedback controlled" mixing in response to temperature or dissolved oxygen differentials, or as in Poowong Reservoir in Victoria, solar powered aeration. Intermittent mixing with a cycle period of a number of days appears not to have been attempted. Intermittent mixing with a cycle period of several days has been suggested for several Queensland applications, but not adopted "because of the higher level of operator involvement and monitoring required to ensure satisfactory performance" (Brown, pers. comm.). Such a mixing regime may offer advantages over continuous mixing for the control of algae.

Basically the argument for intermittent mixing lies in the concept that certain types of algae (eg blue-green algae) do best under stagnant conditions while other (green algae and diatoms) prefer more turbulent water. Mixing the water column for a number of days may thus act to control those types of algae not favoured by mixing (ie. blue-green algae will be kept in check). A period of calm following mixing may act against other types which were favoured by the mixing. The aim of this approach would be to keep to a minimum the growth of all planktonic algae types (Ganf 1982).

Obviously the length of the mixing cycle is likely to be critical. The period of negative conditions for one type of algae must not be such so as to allow other algae to flourish. It is also likely to be crucial that the stagnant period is not so long as to allow depletion of oxygen at the sediment-water interface or the development of stratification to the extent that it can no longer be broken down.

Intermittent mixing has been successfully trialed overseas. Lake Fischkaltersee in West Germany has been aerated primarily to reduce numbers of the filamentous blue-green alga *Oscillatoria*. Prior to the use of an intermittent strategy, continuous aeration initially produced the desired objective of almost completely eliminating *Oscillatoria* but also resulted in an almost two-fold increase in the green algae and diatom populations. This outcome was still viewed as positive, as the prime aim was to control the blue-green alga.

Following the trial of intermittent mixing the system was once again mixed on a continuous basis. The benefits were, however, short-lived and during the third and fourth year of continuous mixing *Oscillatoria* once again became dominant. "The density of these cyanobacteria even exceeded the normal levels under conditions prior to therapy measures" (Steinberg and Zimmermann 1988). As a result of this unfavourable outcome intermittent destratification was fully implemented. Once again the blue-green algae or "cyanobacteria" were controlled (Figure 4.1).

Steinberg and Zimmermann concluded that blue-green algae were unable to respond quickly enough to cope with the rapidly changing environment associated with intermittent mixing. In addition intermittent mixing may have the advantages of being less expensive than continuous mixing, may allow loss of cells through settling during the stagnant period and be suitable to shallower lakes, where light limitation from permanent mixing is more difficult to achieve.

Such an approach may well combine some of the advantages of both destratification and hypolimnetic aeration. Hypolimnetic aeration has not been dealt with in this report as no such systems have been installed in Australia. One of the main advantages of hypolimnetic aeration is that it allows settling of cells from the euphotic zone, as also suggested above for intermittent mixing. A discussion of the principles of hypolimnetic aeration may be found in Fast (1981).

Intermittent mixing may well offer advantages over continuous mixing regimes,. however, the period over which it was examined in the abovementioned example was relatively short. Given further evaluation this method may prove a viable alternative to conventional continuous destratification.

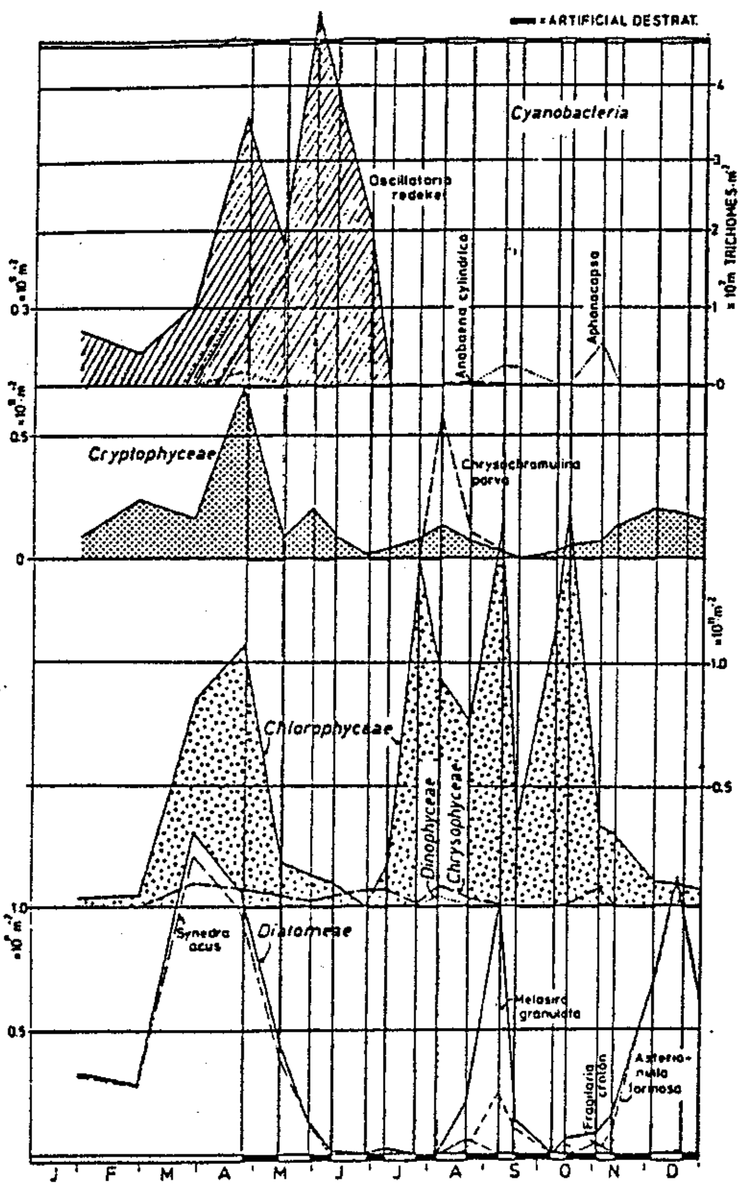

Figure 4.1 Phytoplankton succession during intermittent mixing in Lake Fischkaltersee (from Steinberg and Zimmermann 1988).

4.6 Taste and Odour Generation

Unwelcome tastes and odours in water bodies may be derived from a variety of sources. These include anaerobic decomposition, the growth of algae and fungi, industrial and agricultural input and contamination from recreational usage. The sources able to be addressed by destratification are limited to biological excretions and anaerobic decomposition.

The role destratification may play in eliminating odours derived from anaerobic decomposition lies in the supply of oxygen. The formation of the odourous hydrogen sulphide gas (H_2S, rotten egg gas) occurs after the removal of dissolved oxygen from the hypolimnion. Destratification is capable of providing a continuous supply of dissolved oxygen and eliminating the formation of reduced odours. Further discussion of this point is made in Chapter 5.

Destratification may influence both the level and type of biological growth in a water body as discussed earlier. In this way the accumulation of taste and odour producing compounds may be prevented. Although fungi are known to produce similar effects, most information seems to be recorded on the production of tastes and odours from algae. Table 4.2 lists some algae known to generate taste and odour difficulties.

Genus of Algae		Taste/Odour	
		moderate nos.	abundant nos.
Cyanophyta	(Blue-green Algae)		
	Anabaena	grassy, musty	septic
	Anabaenopsis		grassy
	Aphanizomenon	grassy	septic
	Coelosphaerium	grassy	grassy
	Cylindrospermum	grassy	septic
	Gloeotrichia		grassy
	Microcystis	grassy	septic
	Nostoc	musty	septic
	Oscillatoria	grassy	musty, septic
	Rivularia	grassy	musty

Chlorophyta (Green Algae)		
Actinastrum		grassy, musty
Chara	garlic	garlic, spoiled
Chlamydomonas	musty, grassy	fishy, septic
Chlorella		musty
Cladophora		septic
Closterium		grassy
Coelastrum		grassy
Cosmarium		grassy
Crucigenia		grassy
Dichotomosiphon		
Dictyosphaerium	grassy	fishy
Eudorina		fishy
Gloeocystis		septic
Gonium		fishy
Hydrodictyon		septic
Mougeotia		
Nitella	grassy	grassy, septic
Oocystis		green
Palmella		
Pandorina		fishy
Pediastrum		grassy
Scenedesmus		grassy
Spirogyra		grassy
Staurastrum		grassy
Tribonema		fishy
Ulothrix		grassy
Volvox	fishy	fishy
Zygnema		
Bacillariophyceae (Diatoms)		
Asterionella	geranium, spicy	fishy
Atteya		
Cyclotella	geranium	fishy
Cymbella		
Diatoma		aromatic
Fragilaria	geranium	musty
Melosira	geranium	musty
Meridion		spicy
Navicula		
Nitzschia		
Pleurosigma		fishy
Rhizosolenia		
Stephanodiscus	geranium	fishy
Synedra	grassy	musty
Tabellaria	geranium	fishy
Chrysophyceae		
Chysosphaerella		fishy
Dinobryon	violet	fishy
Mallomonas	violet	fishy
Synura	cucumber, spicy	fishy
Uroglenopsis	cucumber	fishy

Pyrrhophyta			
	Ceratium	fishy	septic
	Cryptomonas	violet	violet
	Glenodinium		fishy
	Peridinium	cucumber	fishy
Euglenophyta			
	Euglena		fishy

Table 4.2 Algae identified as causing unpleasant tastes and odours in water storages (Bowles and Saunders 1986, Palmer 1962).

Substances known to be produced by algae and aquatic fungi that cause taste and odour difficulties include: geosmin, methylisoborneol, isopropylmethoxypyrazine, lactones and dimethylsulfides (Genber 1983, Juttner 1983).

Table 4.3 lists those flavour causing substances identified from water samples from the Harding Reservoir in Western Australia.

Compound	Taste/Odour	Threshold Taste Concentration	Observed Concentration	Mode of Formation
Methyliso-borneol	Earthy/musty	30	6-25	Biological Activity
Geosmin	Earthy/musty	10	3-10	Biological Activity
Dimethyl-trisulphide (DMTS)	Swampy	5 - 10	0-55	Biological Activity (?)
Dimethyl-tetrasulphide (DMTeS)	Swampy	5 to 10 (?)	0-45	Biological Activity (?)

Note 1. Threshold taste concentration (TTC): the minimum concentration at which the taste of a compound can reliably be detected in water.

Table 4.3. Flavour producing compounds identified from Harding Reservoir water (from Kavanagh 1986).

4.7 Destratification Applications

Artificial destratification has been used in a number of cases throughout Australia to address, at least in part, problems associated with algae growth. (Table 4.4).

State	Water Body
New South Wales	Carcoar Dam Chaffey Dam Chichester Dam Glenbawn Dam Glennies Creek Reservoir Lake Medlow Lyell Reservoir Malpas Reservoir Mardi Dam Prospect Reservoir Split Rock Dam Suma Park Reservoir Windamere Dam
Queensland	Hinze Dam Lake Manchester North Pine Dam Solomon Dam
South Australia	Barossa Reservoir Happy Valley Reservoir Kangaroo Creek Reservoir Little Para Reservoir Myponga Reservoir Warren Reservoir
Victoria	Bullarto Reservoir Wombat Reservoir
Western Australia	Harding Reservoir

Table 4.4 Australian cases where artificial destratification has been used in an attempt to address problems associated with algae growth.

The degree of success reported in controlling algae growth through the use of destratification varies widely. The Australian experiences range from extraordinary success at Suma Park Reservoir to the ongoing difficulties experienced in Little Para and Myponga Reservoirs.

Prior to artificial destratification Suma Park Reservoir regularly had taste and odour difficulties believed to be associated with algae growth. High levels of algae, especially blue-green algae, were believed to be responsible. Following the successful destratification of this system algae numbers have been reduced, and taste and odour problems eliminated. Whilst it is not possible to attribute the success of destratification to any particular factor, both the depth of mixing and oxygen penetration to depth have increased following destratification (see Case 44). It is possible then that algae growth was reduced by either mixed layer deepening (as described earlier in section 4.4) or through reduced nutrient release from the sediments, in response to oxygenation Section 5.4. Additional pre- and post-aeration data would be required to resolve this issue.

In the case of both Little Para (Case 6) and Myponga Reservoirs (Case 7) destratification has, to date, not been totally successful in reducing algae numbers. As the case studies for these systems show, stratification has still developed to some degree, allowing excessive algae growth on occasions. Current efforts to increase the degree of mixing in both systems may yield better results.

CHAPTER 5.

RELEASES FROM THE SEDIMENT

5.1 Introduction

Iron and manganese have long been notorious for causing water quality problems in surface storages. When present in sufficient concentrations these compounds may lead to dirty water complaints, undesirable tastes, discolouration of the water and precipitates or slimes in reticulation and distribution systems. The National Health and Medical Research Council (1987) guidelines for the upper limits of iron and manganese concentrations in drinking water are 0.3 mg L^{-1} and 0.1 mg L^{-1} respectively. More stringent conditions have been set in the European Economic Community Directives on Water Quality with guide levels of 0.05 and 0.02 mg L^{-1} respectively, and maximum levels of 0.2 and 0.05 mg L^{-1} respectively.

High phosphorus and nitrogen levels are also rarely desirable in lakes and reservoirs. These elements are essential nutrients for the growth of algae and increased phosphorus concentrations usually result in larger populations of algae (Ahlgren 1977; Cullen and Rosich 1978; Gilliom 1984; Golterman 1977; Hodgkin *et al* 1980; Rosich 1983; Thomas 1973; Vollenweider 1969; Welzel 1983). High levels of algae may then lead to difficulties such as impaired filtration, high chlorine demand, dirty water complaints, poor taste and odour and even toxicity (Banens and Fisher 1987; Falconer *et al* 1982; Kavanagh 1986; Palmer 1962).

In lake systems sediments are the main sink for many components, including iron, manganese and phosphorus. Sediments also play a significant role in determining the concentrations of these components in the overlying waters. There are substantial fluxes, in both directions, of these materials across the sediment-water interface. The rates of unidirectional fluxes depend on a number of factors, most notably dissolved oxygen level, organic matter level, oxidation-reduction (redox) potential and pH.

5.2 Unidirectional and Net Fluxes

Much of the literature obscures, or overlooks, the distinction between unidirectional and net fluxes. A simple illustration is in human growth. The mass of most humans varies little over their adult life, let us say 5 kg. If, say, 0.5 kg of materials are consumed each day then over 40 years the total input would be 7,300 kg. Clearly, then, there must be an ouput of 7,295 kg over that 40 years.

Thus:

unidirectional fluxes		
	input	0.5 kg day^{-1}
	output	0.49966 kg day^{-1}
net flux		0.00034 kg day^{-1}

The most important point here is that the net flux is the very small difference of two much larger numbers and consequently very small changes in the unidirectional fluxes can cause a large change in the net flux. For example, if the input in the example above was to increase by only 0.00034 kg day^{-1} (0.07%) the net flux would increase by the same amount, 0.00034 kg day^{-1}, but which is a 100% increase (ie an additional 5 kg) over 40 years.

Another important point is that all of these fluxes can vary significantly over short timescales and give rise to a sudden increase/decrease in the mass of the person.

The significance of these fluxes, and their variation, in lake systems is discussed in later Sections.

5.3 Dissolved Oxygen and Redox Potentials

The main impact of changing dissolved oxygen levels is reflected in changing redox potentials.

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

$$C_6 H_{12} O_6 \text{ (glucose)} + 6H_2O \rightleftharpoons 6O_2 + 24H^+ + 24e^-$$

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

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

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 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 iron and manganese compounds that have been reduced are also more complex than shown in the following equations.

Aerobic respiration:
$C_6H_{12}O_6$ (glucose) + $6O_2 \rightleftharpoons 6CO_2 + 6H_2O$

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

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. Table 5.1 shows the order in which compounds are reduced in response to falling redox potential.

Table 5.1 Redox potential for the reduction of selected components of natural waters. Values are for pH = 7.0 and 25 C (from Syers *et al* 1973).

Electron Acceptor	pE^o
O_2	13.72
NO_3^-	12.65
MnO_2	8.5
Fe^{3+}	.67
SO_4^{2-}	-3.75

pE^o is a measure of redox potential (high values represent oxidizing conditions, low values reducing conditions). In practice, however, measurement of redox potential presents many difficulties and, as such, the above values are best treated as relative rather than absolute figures. Nevertheless, they are useful for predicting a sequence of events. Thus oxygen will be consumed before reduction of nitrate, manganese compounds, iron compounds and so on. This order can be interrupted by the supply of an acceptor higher in the sequence. This is the reason destratification can interrupt the reduction of iron and manganese. By supplying enough oxygen, redox levels will not fall to levels sufficient to allow the reduction of iron and manganese. Applications of nitrate to the sediment surface have been used in a similar fashion to prevent falling redox potentials (Ripl 1976).

5.4 Iron and Manganese

In aerobic natural waters iron and manganese are present only as the oxidized forms iron(III) and managanese(IV). In these forms their solubility is extremely low (< 0.001 mg L^{-1}). Hence the elements are present mainly as hydrated iron(III) and manganese(IV) oxides.

Under anaerobic conditions the solubilities are greatly increased and can reach levels of 10 mg L^{-1} and more. Solubilities under these conditions are thought to be controlled by mineral carbonates (Nriagu and Dell, 1974).

In catchment runoff and aerobic lakewaters iron and manganese are present at low concentrations (up to about 0.3 mg L^{-1}) and mainly in colloidal and particulate forms. Consequently these components settle to the sediments. On the other hand, in sediment pore-waters iron and mangenese are very much higher (up to 10 mg L^{-1}) or more (Glass and Poldoski 1975, Syers *et al* 1983).

Thus there is a concentration gradient across the sediment-water interface and therefore a continual, large flux of iron and manganese out of the sediments. As the iron and manganese enter aerobic water, oxidation takes place, followed by precipitation out of solution and then re-settlement to the sediments. Thus the net flux (on timescales of a year or more) is to the sediment.

However, in many water bodies there may be temperature induced density stratification into upper (epilimnion) and lower (hypolimnion) layers. Further, the hypolimnion in such water bodies is isolated from atmospheric oxygen and continuing respiration can lead to anaerobic conditions. Thus there will be a net flux of iron and manganese out of the sediments until their concentrations in the hypolimnion approach those in the sediment pore-waters. In these circumstances the boundary between low and high iron and manganese concentrations occurs at the interface between the aerobic epilimnion and the anaerobic hypolimnion, rather than at the sediment-water interface. Figures 5.1 and 5.2 illustrate the relationship between dissolved oxygen levels and iron and manganese concentrations in water overlying sediments.

In lakes used for water supply purposes such physical and chemical stratification can lead to problems through :

(a) limiting the volume of water of acceptable quality and

(b) at overturn (whether by natural or artifical means) by the injection of high levels of iron and manganese into the surface layer - until oxidation, precipitation and re-settlement occurs.

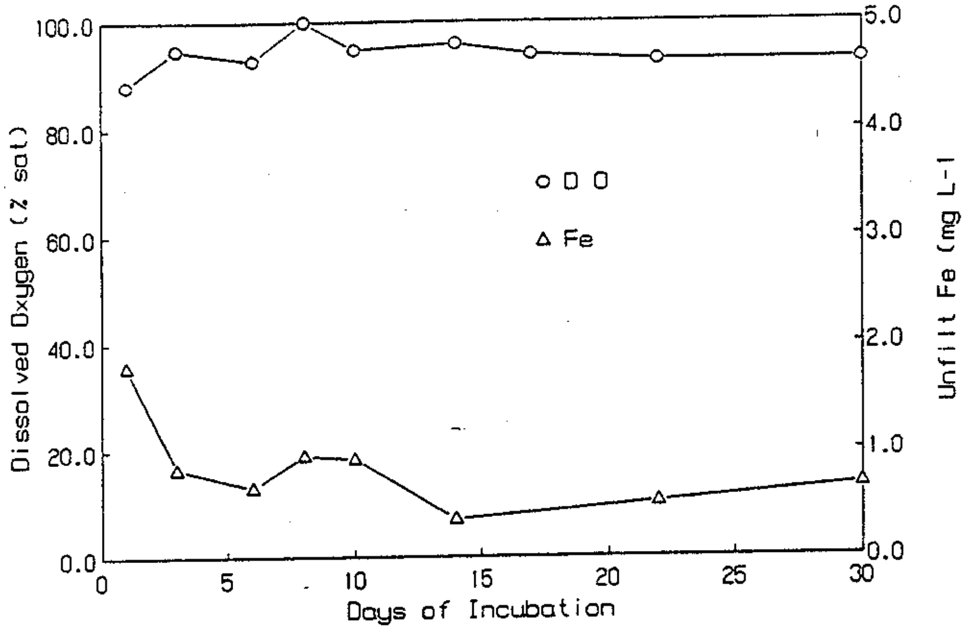

Figure 5.1a Iron and dissolved oxygen concentrations in an aerated experimental sediment-water core from the Harding Reservoir, WA.

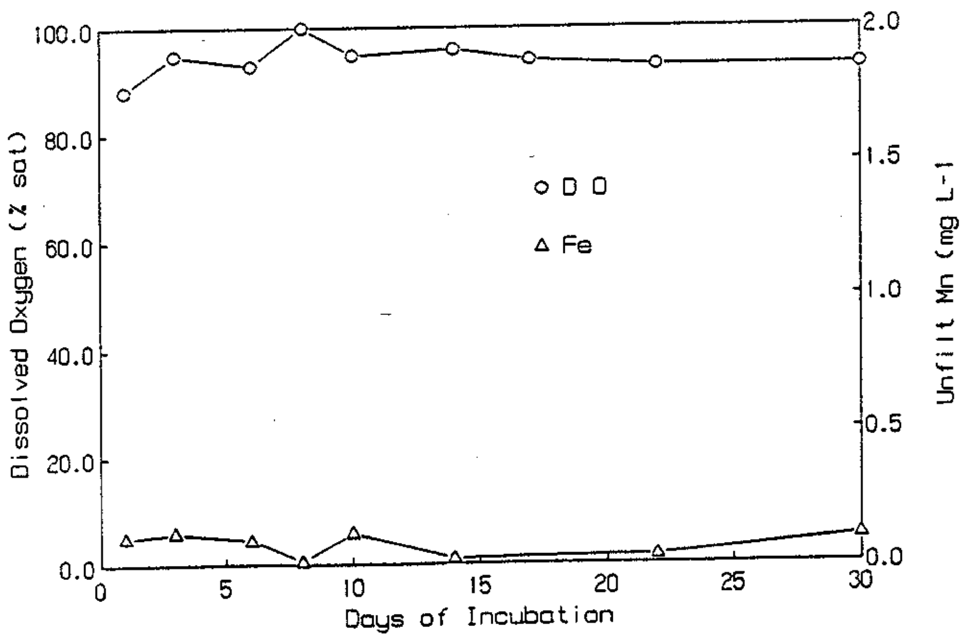

Figure 5.1b Manganese and dissolved oxygen concentrations in an aerated experimental sediment-water core from the Harding Reservoir WA.

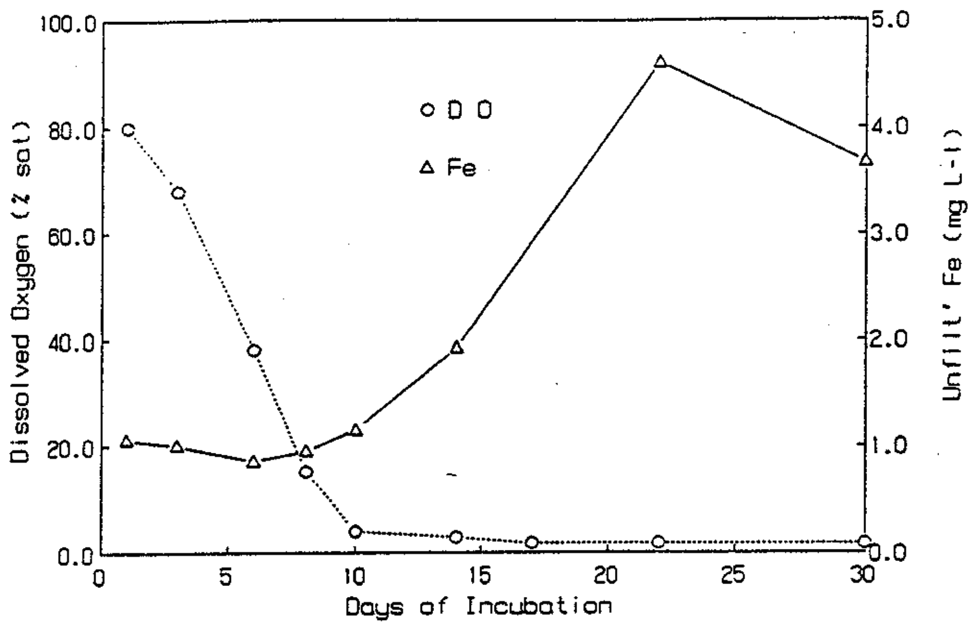

Figure 5.2a Iron and dissolved oxygen concentrations in an non-aerated experimental sediment-water core from the Harding Reservoir.

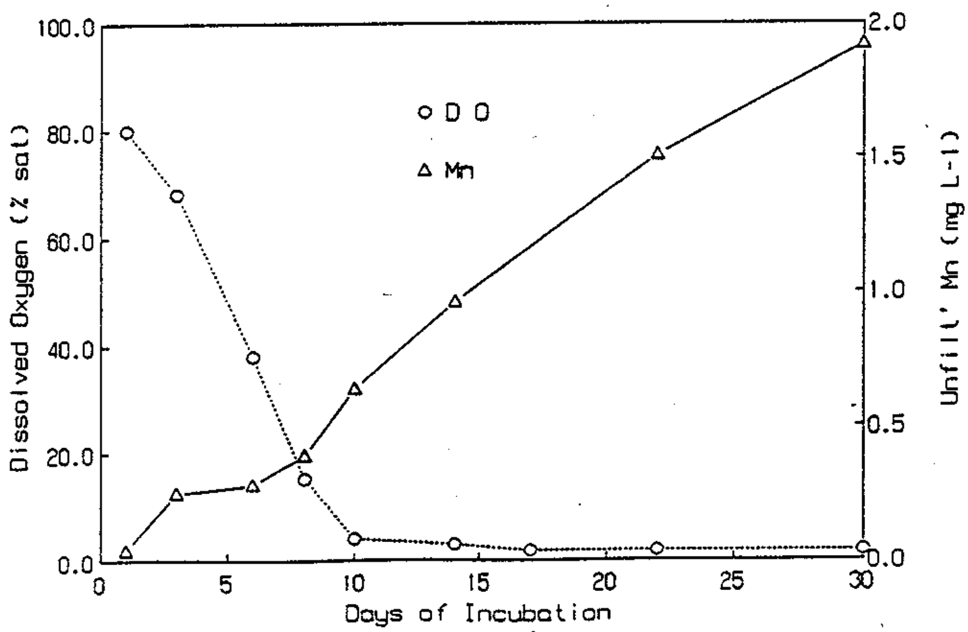

Figure 5.2b Manganese and dissolved oxygen concentrations in an non-aerated experimental sediment-water core from the Harding Reservoir.

5.5 Phosphorus

Phosphorus occurs in natural systems essentially only as various forms of orthophosphate, that is in forms where each phosphorus atom is bonded to four oxygen atoms. Simple phosphates, such as trisodium phosphate (a heavy duty detergent), are very soluble in water, with concentrations being measured in %. At the same time other salts such as $Ca_3(PO_4)_2$, $AlPO_4$ and $FePO_4$, are much less soluble, while more complex materials such as apatites, $Ca_x(OH)_y(PO_4)_z$, are even less soluble.

Nevertheless, even the apatites allow the capacity for a substantially greater concentration in solution of phosphates (about 30 ug P L^{-1}) than is typical for unpolluted waters (say < 10 ug P L^{-1}).

The orthophosphate group adsorbs very strongly to hydrated metal oxides, especially those of iron(III); thus iron often provides the main control on phosphorus concentrations in aerobic waters in the presence of iron(III) oxides. As redox potentials fall iron(III) may be reduced to iron(II) allowing previously insoluble phosphorus into pore waters. In anaerobic sediment pore-waters, concentrations of phosphorus can reach 10 mg P L^{-1} (see for example, Glass and Podolski, 1975). In iron-containing sediments the major control is probably the iron(II) mineral vivianite, $Fe_3(PO_4)_2.8H_2O$ which is considerably more soluble than iron(III) compounds (Nriagu and Dell, 1974),

Thus, as in the case of iron and manganese, there is a concentration gradient across the sediment-water interface and therefore a continuing flux of phosphate out of the sediments.

5.5.1 Phosphorus exchange with sediments

If the overlying water is aerobic the returning hydrated iron(III) oxides (Section 5.4) will re-adsorb the phosphates and return them to the sediment.
If, however, there is an overlying anaerobic hypolimnion there will be a net flux out of the sediments until the sediment output flux is balanced by the returning flux of phosphate adsorbed to hydrated iron(III) oxides precipitating out of the aerobic epilimnion. These two situations are illustrated in Figure

5.3. It is for these reasons that the reduction of iron(III) is often cited as being of major importance in the net release of phosphorus under anaerobic conditions (Bostrom *et al* 1982; Fillos 1977; Li *et al* 1972; Lijklema 1977; Rosich 1982; Syers *et al* 1973).

Table 5.2 gives a comparison of some unidrectional and net fluxes for phosphorus while Figure 5.4 shows the net release of phosphorus under conditions of decreasing dissolved oxygen.

Table 5.2 Comparison of some unidirectional and net fluxes at the sediment-water interface (Rosich and Cullen, 1981).

	mg P m^{-2} day^{-1}	
	Unidirectional	Net Release
Lake Burley Griffin (ACT)		
aerobic (1)	230	
anaerobic (1)	8,100	3.7
Lake Ginninderra (ACT)		
aerobic (1)	17	
anaerobic (1)	220	0.01

Note: (1) Refers to the oxygen content of the overlying water.

Many factors control the tendency of a given sediment to release phosphorus. The degree of phosphorus saturation, the particular types of phosphorus contained by the sediment and the sensitivity of the sediment to environmental changes are all crucial. Basically the mechanisms by which sedimentary phosphorus is released are diffusion, turbulent resuspension, bioturbation and gas convection. Falling redox potential within the sediment usually establishes a concentration gradient which allows for diffusion of phosphorus from the sediment. Diffusion and redox-controlled dissolution are usually considered to be the major mechanisms by which phosphorus release occurs given that microbial respiration is the key process which provides the necessary reduction in redox potential (Bates and Neafus 1980; Bostrom *et al* 1982; Mortimer 1941a; Syers *et al* 1973).

Problems for water supply can then occur through phosphorus-induced excessive algae growth in the epilimnion if:

(a) phosphorus diffuses at a sufficient rate out of the hypolimnion into the euphotic epilimnion (Rosich and Cullen, 1980; Rosich 1988) or

(b) overturn (whether by natural or artificial means) leads to the injection of high levels of phosphorus into the euphotic epilimnion.

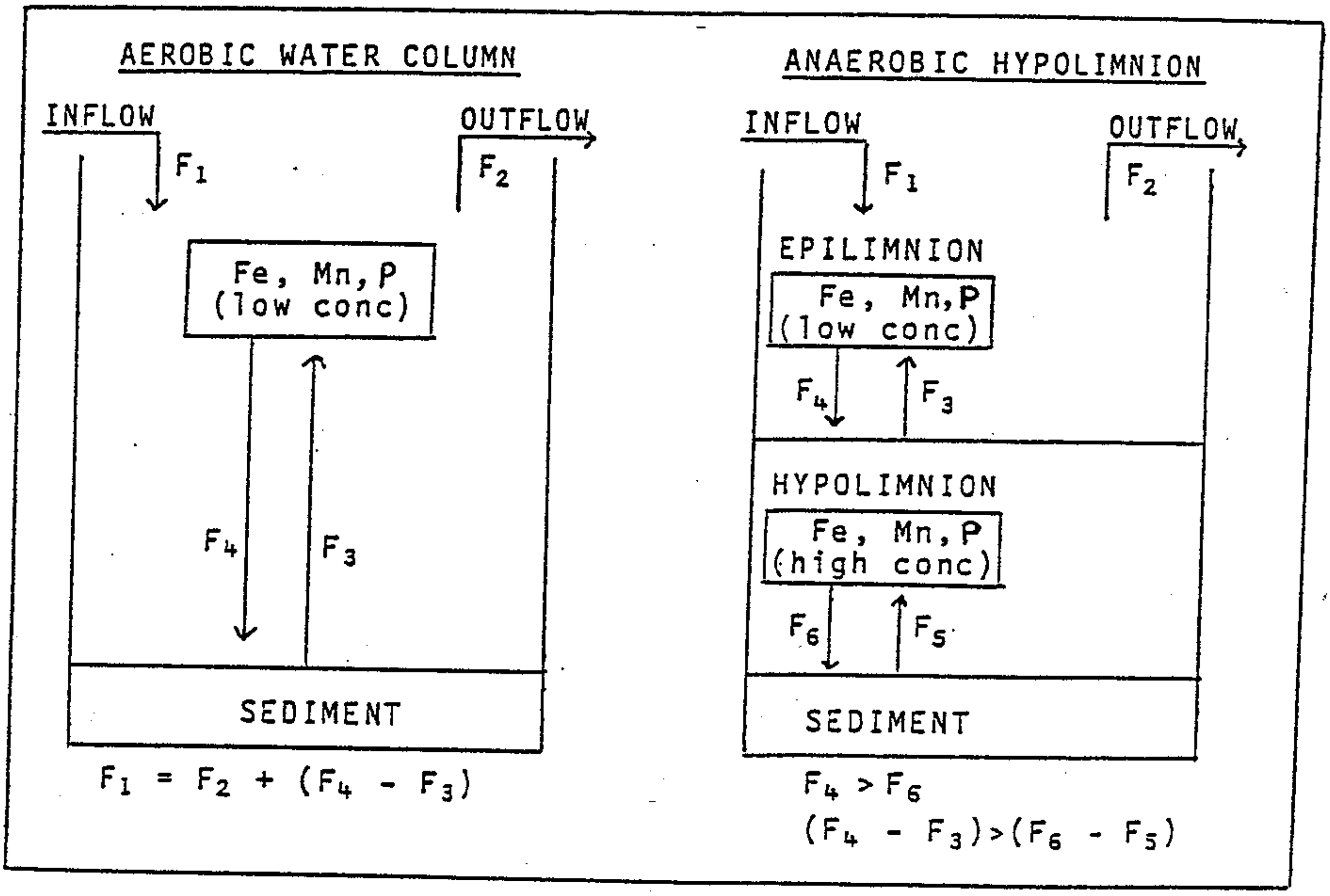

Figure 5.3 Schematic representation of major fluxes of iron and manganese in water bodies with and without an aerobic hypolimnion.

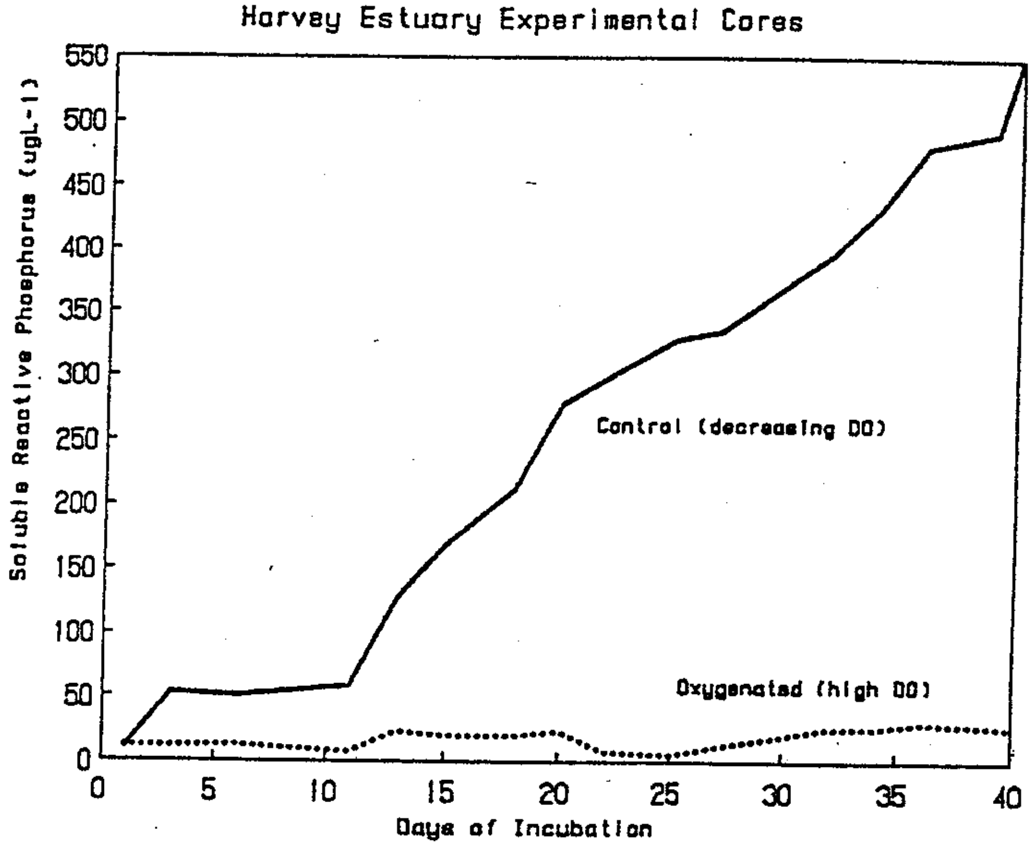

Figure 5.4 A comparison of sediment phosphorus release from Harvey Estuary sediment-water cores; under control (allowed-to-go anaerobic) and aerated conditions.

5.5.2 Biological availability of phosphorus forms

Some of the confusion that has in the past surrounded this topic arises from the application of different timescales. Some early studies allowed only a few days for algae to grow utilizing sediment as a nutrient source, while other studies allowing several weeks found much greater availability. Overall there appears to be a general consensus that the hydroxide-soluble (non-apatite) fraction is that portion most readily available to algae.

For example Cullen *et al* (1978) and Rosich and Cullen (1981) found that with an incubation time of several weeks, about 90% of hydroxide-soluble and about 30% of acid-soluble phosphorus was available to algae. Since the majority of the phosphorus was present in hydroxide-soluble forms, overall more than half of the total phosphorus of Lake Burley Griffin in Canberra was available to algae over the test period. The authors thus calculated that sufficient phosphorus was available to maintain eutrophic conditions for about 30 years even if all internal sources of phosphorus could be eliminated.

Williams *et al* (1980) found that the alga *Scenedesmus quadricauda* was able to utilize between 8% and 50% of total phosphorus present in sediments of the Great Lakes, and that there was a direct relationship between phosphorus utilization by this alga and the hydroxide-soluble fraction of the sediment. They concluded that both the acid-soluble and organic phosphorus fractions were unavailable to the test alga, qualifying this conclusion by pointing out that these latter two forms of phosphorus may not be entirely "unavailable" but that utilization may take considerable time.

These conclusions are consistent with the earlier findings of Golterman (1977). Through experiments using both artificial ponds and laboratory cultures of *Scenedesmus* sp. Golterman concluded that not all forms of sedimentary phosphorus were equally available for algae growth. Phosphate bound into the clay lattice was not "available" while iron bound (hydroxide-soluble) phosphate was entirely "available". He also concluded that crystal size determined the availability of acid-soluble phosphorus. The term "available" is perhaps inappropriate. Phosphorus bound to the sediment is definitely "unavailable" to algae and must be released or mobilized by physical and chemical processes, and so the term "potentially available" is more suitable.

5.6 Turbidity and Colour

Colour may be derived from either catchment or internal sources. Colour may be produced in the water body by algae, or may come from the breakdown of organic compounds in the sediment or catchment soils. Work conducted by Bowles *et al* (1979) identified a correlation between water colour and associated vegetation type. A positive correlation was especially noted with some rough-barked eucalypt species (Bowles *et al* 1979).

Oades (1982) suggests that most of the yellow-brown coloured organics in natural waters will be fulvic acids. Other colour may be associated with water rich in iron and manganese and this yellow to brown colour is suggested by Oades (1982) as being "probably due to metal-organic complexes which protect ferrous and manganous ions from ready oxidation which would normally cause precipitation of hydroxides at near neutral pH values". It is in the latter area that destratification is relevant. By the processes previously outlined destratification may supply dissolved oxygen, maintain high redox levels and prevent the release of iron, manganese and associated colour.

It should also be noted that colour can have some benefits. Although it certainly reduces the aesthetic value of water, it is not harmful to health. The advantage it may offer is through creating light limitation. Colour present in a water body will restrict light penetration and may act to reduce the level of algae growth in a system. Such a situation has been reported in Tarago Reservoir in Victoria (Bowles *et al* 1979). Algae growth is the result of a number of variables, and light is only one, but in certain instances light penetration may be the most limiting of these variables.

The light scattering suspended material in water that gives rise to turbidity may be organic, inorganic, or associations of organo-mineral complexes (Oades 1982). The areas in which destratification may affect turbidity are :

i) Changes to the amount of phytoplankton able to contribute to turbidity.

The effects destratification may have on phytoplankton populations have been discussed at length in Chapter 4. Suffice to say here that destratification can significantly affect the level of phytoplankton in a water body.

ii) A change in the ionic charge of substances in solution.

The oxidation of reduced substances in a water column may promote the coagulation of negatively charge suspended colloids. In this way oxygenation may promote the removal of turbidity as the negatively charged, turbidity-causing colloids coagulate and settle with positively charged oxidized compounds, such as iron hydroxides.

iii) A change in the degree of disturbance of the sediment surface.

It is possible that the disturbance caused by aeration at the sediment-surface could cause increased turbidity. During the collection of information for this review such disturbance was noted as occurring in certain systems. In each case such turbidity persisted for only a short period of time. In some of the Blue-Mountains storages a plume of dirty water is often brought to the surface when the aerator is first turned on. This is believed to be due to sediment disturbance. This turbidity quickly dissipates and is not apparent during the normal operation of the aerators (Sydney Water Board, 1988).

5.7 pH

Earlier in this Chapter pH has been mentioned as a variable affecting various sediment release processes (eg both nutrient release and iron and manganese release). However, in most natural waters pH changes are best viewed as a result of other processes, rather than a primary causal agent.

Repetitive respiration by the biological community can lead to a decrease in pH through the addition of carbon dioxide. Similarly the consumption of CO_2 through photosynthesis may raise pH, through the production of hydroxy/ions. As a consequence pH may vary diurnally as the balance switches from day-time photosynthesis to night-time respiration, or vary with depth in the same fashion. Large algae blooms are able to generate an associated high pH due to the high rates of photosynthesis.

It is possible that artificial aeration may lower pH through increasing the supply of CO_2. Lorenzen (1977) suggests that this may then bring about changes in algae populations "Blue-green algae may have a competitive advantage at high pH. It is therefore possible that lowering the pH as a result of mixing could favour non blue-green algae species over blue-green species". In this way pH changes due to mixing may play a role in assisting the often expected change from blue-green algae to green algae and diatoms discussed in Chapter 4.

5.8 The Role of Organic Matter

The decline in redox potentials described in preceding sections is mediated by the supply of "readily-utilizable carbon". This is that fraction of organic matter which may be rapidly broken down by microbial activity to impart a substantial oxygen demand on the water. In other words, an increase in the amount of readily-utilizable carbon will increase respiratory activity which will in turn lead to increased oxygen demand. Figure 5.5 shows the effect an increased carbon supply (sucrose) had on oxygen demand in experimental sediment-water cores from the Harding Reservoir, WA. An increased supply of readily-utilizable carbon greatly increased oxygen demand in the 0.5 m water column overlying sediment in these cores. If oxygen is in limited supply (for example due to stratification) then redox potentials will fall and may lead to net releases of phosphorus, iron and manganese.

When investigating Canberra's urban lakes Rosich and Cullen (1981) found that the amount of this readily-utilizable fraction of organic matter in the sediment limited the capacity of bacterial activity to reduce iron(III) and hence reduced the flux of phosphate across the sediment-water interface . They found that the fluxes for Lake Burley Griffin were considerably higher than for Lake Ginninderra (see Table 5.2). Both lakes had similar quantities of sedimentary total phosphorus, and total organic matter. The reduction of iron(III) to iron(II) through bacterial activity was proposed to be controlled by the amount of readily-utilizable organic matter, and that this fraction was only a few percent of the total organic matter present.

An increase in readily-utilizable carbon may occur as a result of increased primary productivity. As phytoplankton die carbon settles through the lower water levels onto the sediments. This generates an increased oxygen demand through microbial respiration and hence may lead to increased nutrient release. This in turn may increase phytoplankton growth, and so on.

A certain level of oxygen (or other suitable oxidant) must be made available to the surficial sediments to allow the degradation of organic matter while minimizing the fluxes of undesirable materials. Artificial destratification is one method to provide an increased supply of oxygen. Others include aeration of the hypolimnion (without destratification) and oxidation of the readily-utilizable carbon through the addition of nitrate.

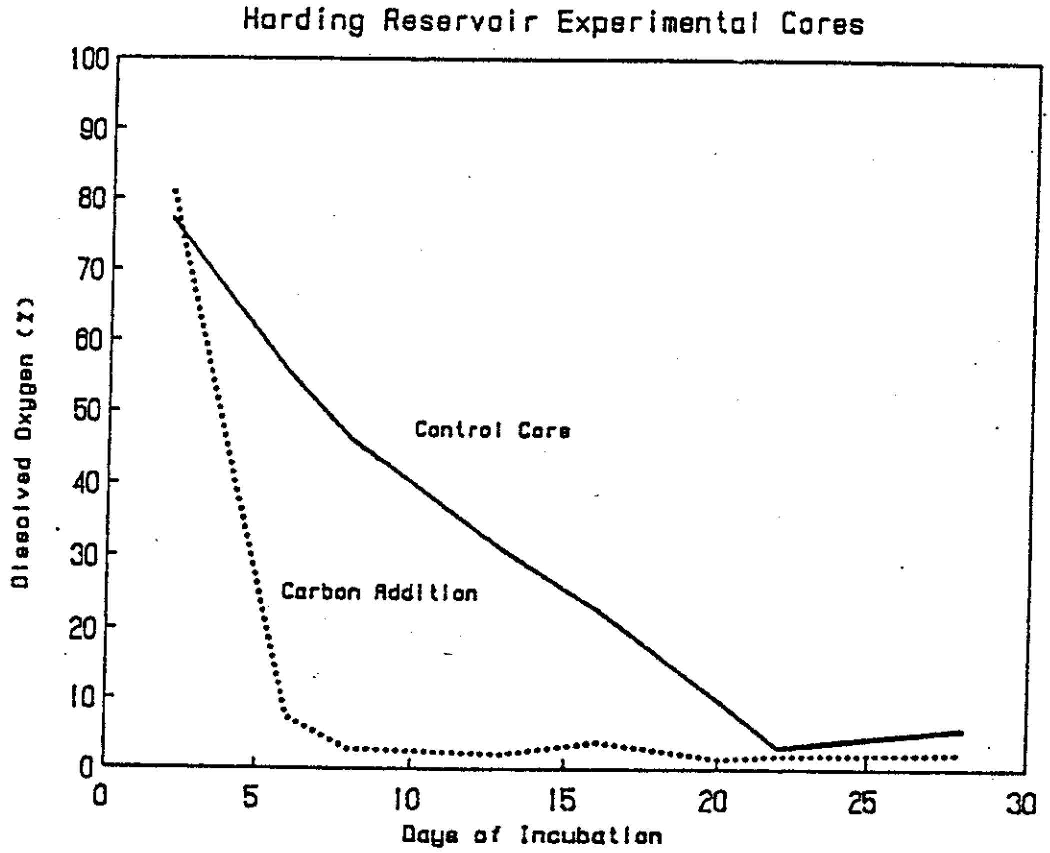

Figure 5.5 A comparison of oxygen depletion rates in experimental sediment-water cores under; unaltered (control) conditions and after the addition of a carbon source (sucrose).

5.9 Destratification

The purpose of artificial destratification is often primarily to maintain aerobic conditions throughout as much of the water column as possible, preferably down to the sediment surface. This is done by mixing the water column so as to make atmospheric oxygen available to the whole water column. The mixed layer is now likely to be deeper than the euphotic zone (the layer in which light is sufficient to maintain net primary production). Under some conditions this may reduce the net production of algae because while photosynthesis will be reduced, respiration will be unchanged (Kirk 1983, Talling 1971).

Destratification through artificial aeration has been used in a variety of instances throughout Australia to maintain high redox potentials at the sediment water-interface. In this way the release of undesirable substances into the overlying water may be averted.

The following is a list of those systems for whom case studies are attached in Appendix 1, and where destratification has sought, at least in part, to reduce releases from the sediment.

Table 5.3 Destratification applications in Australia

New South Wales	Avon Dam	Chichester Dam
	Cordeaux Dam	Deep Creeks
	Glennies Creek Dam	Greaves Creek Reservoir
	Lake Medlow	Lower Cascade Reservoir
	Lyell Reservoir	Manganese Creek Reservoir
	Mardi Dam	Porters Creek Dam
	Rocky Creek Dam	Woodford Creek Reservoir.
Queensland	Callide Dam	Hinze Dam
	Lake Manchester	Lake Morris
	Little Nerang Creek Dam	North Pine Dam
	Solomon Dam.	
South Australia	Barossa Reservoir	
Victoria	Bullarto Reservoir	Dartmouth Reservoir
	Lake Eppalock	Running Creek Reservoir
	Sugar Loaf Reservoir	Tarago Reservoir
	Thomson Dam	Wombat Reservoir.

The experiences of Dam #2 (Figure 5.6) in the Cordeaux system and those of Bullarto Reservoir (Table 5.4) exemplify the results achievable through successful destratification. Destratification of Cordeaux Dam #2 significantly improved water quality with typically colour, turbidity and iron levels being reduced. Figure 5.7 displays the changes in iron and dissolved oxygen profiles due to destratification.

	Pre-Aeration (Dec 1985)		During Aeration (Jan 1989)	
Parameter	Surface	7m	Surface	7m
Dissolved Oxygen (%)	94	3	90	84
Iron total (mg L^{-1})	0.63	1.42	0.42	0.43
Iron soluble (mg L^{-1})	0.55	0.82	0.37	0.38
Turbidity (FTU)	4	26	2	1
True Colour (Pf-Co)	80	140	22	16

Table 5.1 A comparison of selected water quality parameters before and during artificial aeration in Bullarto Reservoir (Daylesford Water Board 1989, Burns 1989).

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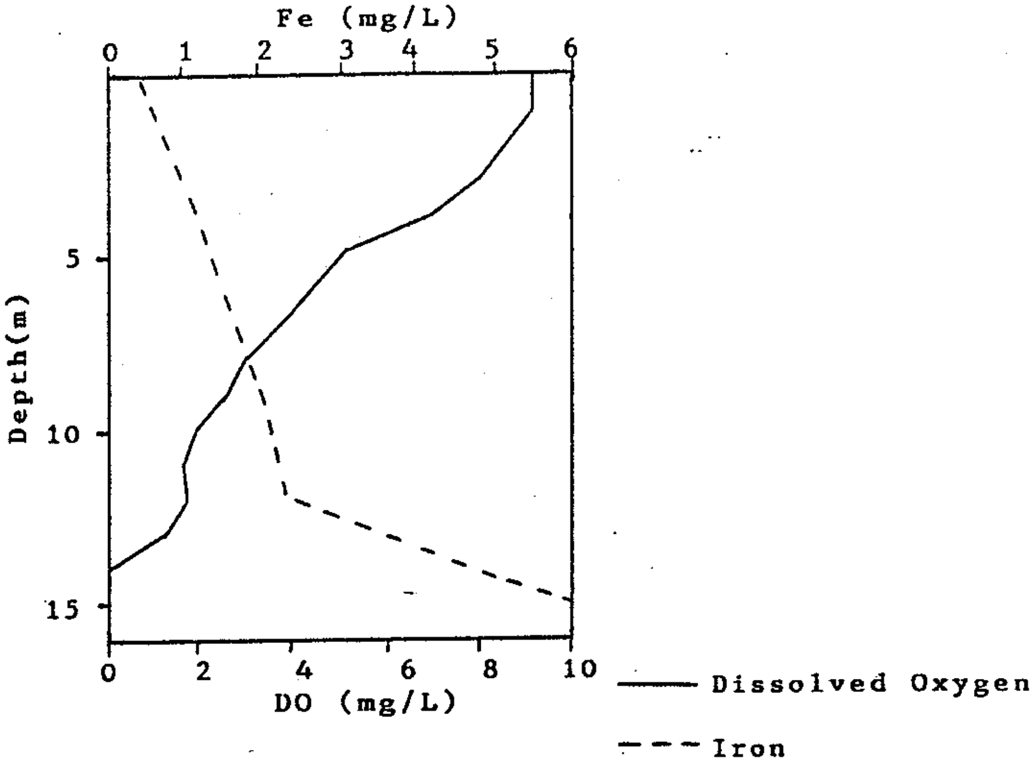

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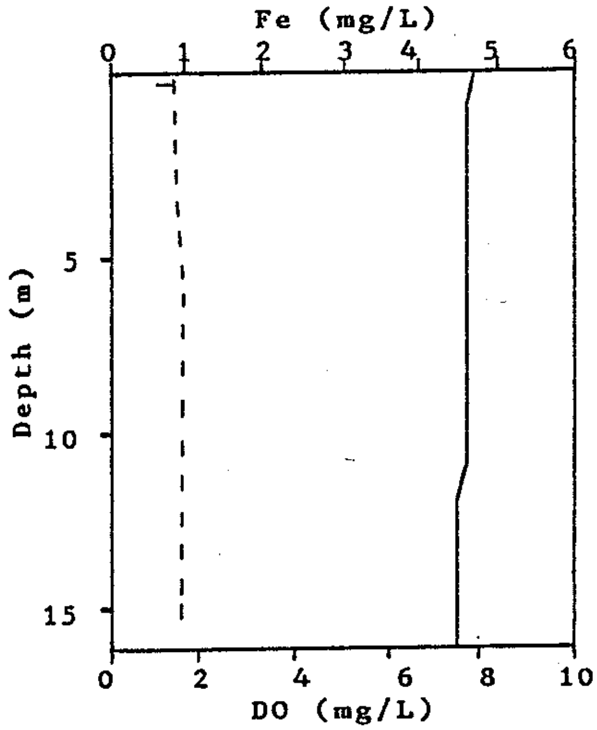

Figure 5.6 Typical DO and Iron profiles from Upper Cordeaux Dam #2 when stratified and destratified (S. W. B. 1988).

CHAPTER 6.

EXTERNAL INPUTS

6.1 Introduction

Problem causing substances, such as iron, manganese and phosphorus may find their way into a water column from either external (catchment) or internal (sediment) sources. For the main part destratification programs have been complemented to control internal loading. Destratification may, however, also often unexpectedly affect compounds entering a water body from external sources. There are instances throughout Australia where artificial aeration has significantly affected the fate of iron and turbidity entering from runoff and riverine sources.

6.2 Iron

Information from the Sydney Water Board (1988) suggests that artificial aeration in Lake Medlow, Lower Cascade and Greaves Creek Reservoirs may have adversely affected the settling of incoming iron containing materials. Prior to the artificial aeration of these reservoirs iron entering the systems from surface and stream runoff settled out to the sediments. Following the introduction of artificial aeration the settling of incoming iron was occasionally impeded, "total iron levels are often higher during summer aeration periods than at other times, in surface waters. Aeration seems to be preventing the settling of ferric hydroxides or particulates with iron attached." (Sydney Water Board, 1988).

It is possible that increased water turbulence promoted by artificial aeration is significant enough to maintain colloids in suspension when prior to aeration the quiescent water column allowed coagulation and settling. It may also be possible that aeration has affected the degree of coagulation of incoming iron compunds. The chemical nature of the water column may have been changed sufficiently such that the coagulation of iron does not occur with enough magnitude to promote settling. Perhaps the nature or number of binding sites has been altered such that coagulation is limited to a smaller mean particle size which is more easily kept in solution.

It is also possible that the increase in energy input to the system has lessened the particle size through the action of sheer forces. In theory the rate of coagulation of compounds will reach a maximum value at an optimum level of energy input. Deviation away from this (eg through increased energy input) could result in a reduced rate of coagulation.

Regardless of the reason for this phenomenon aeration continues to be used in the systems referred to above. The Sydney Water Board (1988) has concluded "This unexpected consequence is nevertheless outweighed by the benefits of aeration in controlling the iron (II), manganese (III) and hydrogen sulphide problems". The Sydney Water Board continues its investigations into this phenomenon and currently treats the problem with alum dosing.

6.3 Turbidity

Similar difficulties to those mentioned above have been experienced in relation to turbidity. As with iron, aeration may disturb the settling of turbidity entering a system with runoff.

Chichester Dam on the central coast of New South Wales is artificially aerated in an attempt to control seasonal fluctuations in water quality, in particular high turbidity, high iron and manganese levels and algae blooms. Aeration has proved to be quite successful in this system, but it has been noted that it tends to maintain high turbidity levels resulting from storm events.

The Hunter District Water Board (1988) reports that high turbidities are not uncommon in Chichester following significant rainfall in the catchment and associated inflow. To avoid aeration keeping inflowing particles in solution, aeration is temporarily suspended. In this way incoming turbidity is allowed to settle. A fine balance is required with such a strategy, as Chichester Dam may rapidly begin to re-stratify in response to the cessation of aeration. When it is deemed appropriate the aerators are turned back on and slowly lowered from the surface, causing minimal disturbance to the settled particles. (Hunter District Water Board, 1988).

CHAPTER 7

AQUATIC ANIMALS

7.1 Zooplankton Numbers

Happy Valley Reservoir in South Australia appears to be the only example in Australia where one major aim of a destratification program has been control over zooplankton numbers. Both high phytoplankton and zooplankton levels have caused considerable taste and odour difficulties in this system.

"High zooplankton numbers occurring each summer represent the most significant biological water quality problem in Happy Valley Reservoir" (Burch, 1987). For example, the State Water Laboratory metropolitan report of 1985/86 mentions that, "odours of intensity greater than 2 in Happy Valley and at the B/C point were caused by the green algae *Ankistrodesmus*, *Dictyosphaerium* and *Oocystis* on 4 occasions (vegetable, grassy) and probably zooplankton on 5 occasions (fishy)". (Engineering and Water Supply, 1988).

Burch (1987) links zooplankton growth closely with temperature. "In Happy Valley Reservoir water temperature restricted major zooplankton population growth events or blooms to summer stratification conditions with temperatures greater than 16 C." Given this correlation one would expect that a reduction in water temperature due to destratification could restrict zooplankton numbers.

Similar conclusions have been drawn from overseas work. Lackey (1973a) lists a change in the annual water temperature cycle as a probable cause for an apparent decline in zooplankton abundance in Parvin Lake, Colorado, due to destratification. Unfortunately the anticipated decline in zooplankton numbers has not eventuated in Happy Valley Rservoir. Although artificial aeration continues during summer, the lake still stratifies to some extent each year. Zooplankton numbers are still high and copper sulphate dosing is periodically required to control zooplankton levels (indirectly through control of phytoplankton).

7.2 Aquatic Insects

Perhaps one of the most successful applications of artificial aeration in Australia occurred somewhat by accident in Myponga Reservoir in South Australia. Aeration of this reservoir was initially commenced in 1980 in the form of an "air curtain" around the offtake tower. This strategy was aimed at forming a barrier to prevent *Chaoborus* (phantom midge) larvae (Figure 7.1) entering Adelaide's reticulation system.

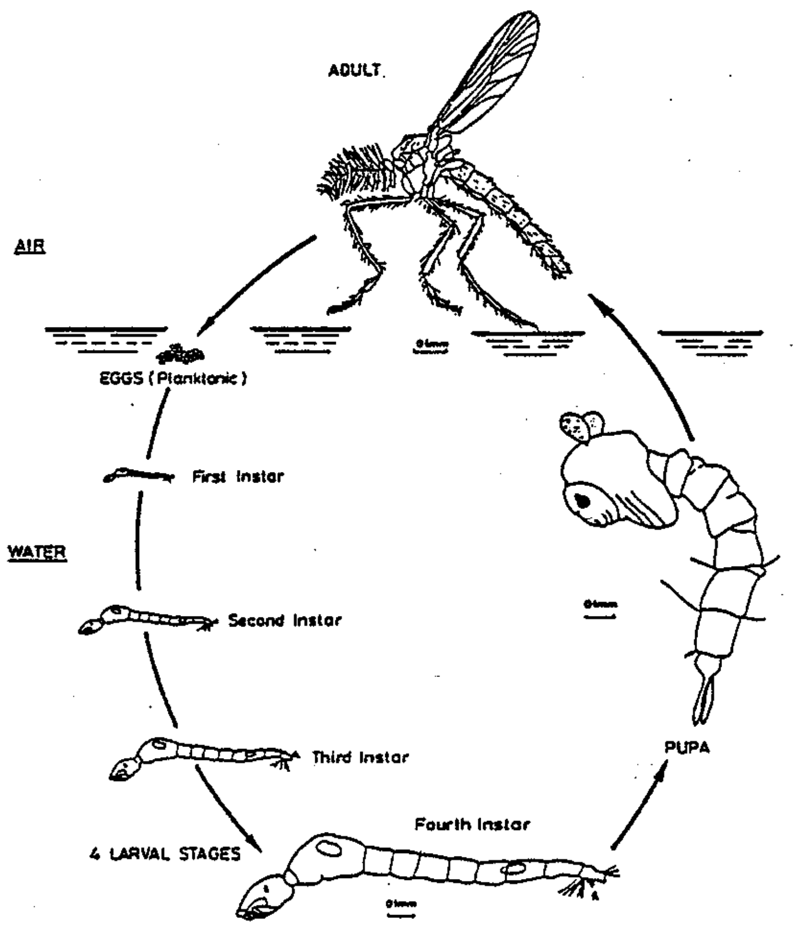

Figure 7.1 The life cycle of *Chaoborus* sp. (phantom midge) in Myponga Reservoir (Suter 1983).

This air curtain strategy did indeed prove effective, but not through the method intended. Instead of acting as a barrier the aeration prevented phantom midge intrusions through a deepening of the epilimnion. The effect is explained by Suter (1983): "The air curtain did not keep the chaoborid larvae away from the outlet tower, but its main effect was to increase the depth of the epilimnion, ensuring the offtake values were always in well oxygenated water. Because the chaoborids have a diurnal vertical migration behaviour, occupying the cooler, oxygen poor waters of the thermocline and hypolimnion during the day and the warmer aerated epilimnion during the night, the high concentrations of the midge larvae were physically separated from the offtake values during the period of maximum water flows ...... The aerator also seemed to have an effect in reducing the population size of phantom midge in the Myponga Reservoir. This combined with the increased depth of the epilimnion controlled the phantom midge problem experienced in 1980."

7.3 Fish Numbers

The review by Summerfelt (1979) details examples of artificial aeration benefiting fish populations in some North American lakes. The author points out that stratification may adversely affect fish numbers through oxygen depletion, temperature restrictions and the accumulation of toxic reduced compounds. Summerfelt proposed a standard of 4 mg L^{-1} dissolved oxygen as being a minimum level required to allow the "continued existence of a varied fish fauna... Also, reduced oxygen levels increase the toxicity of un-ionized ammonia (NH_3) which commonly is present in the anoxic environment" (Summerfelt 1979).

Fish normally avoid an oxygen depleted hypolimnion and restrict themselves to the oxygen rich upper layers of an oxygen stratified water body. It is logical then to expect successful destratification to extend the depth distribution of fish in a previously oxygen stratified lake (Miller and Fast 1979, Summerfelt 1979). Summerfelt (1979) points out that low levels of oxygen also restrict fish growth rates and that destratification may increase both the depth distribution and growth rates of some fishes. Laverty and Nielsen (1970) also report exceptional progresses in growth rates of fish in response to destratification in Lafayette Reservoir in California.

Artificial aeration to increase fish numbers has been successfully applied in Australia, at Lake Bullen Merri. Prior to aeration, natural stratification and the oxygen demand created by decaying *Nodularia* alage resulted in severe oxygen stratification. This reduced the area of the lake available for fish production. The artificial aeration of Bullen Merri has successfully restored a desirable oxygen regime and increased fish populations (Conservation, Forests and Lands, 1989).

CHAPTER 8.

TEMPERATURE AND DISSOLVED OXYGEN

Temperature and dissolved oxygen values are most commonly used as indications of the degree of stratification present in a water body. Normally temperature measurements offer a convenient, reasonably reliable and easily measured indicator of stratification, while dissolved oxygen levels, though more difficult to measure accurately, usually offer a more closely linked indicator of deteriorating water quality at depth.

The release of water low in dissolved oxygen or extreme in temperature is, however, rarely desirable and examples exist where destratification has been implemented to prevent such releases. Effective destratification will create a uniform water column where previously high variations existed. Examples of the change in temperature and dissolved oxygen (DO) profiles that may be obtained are given in Figures 8.1 and 8.2.

A primary reason for conducting destratification trials in Dartmouth Reservoir, Victoria, was to gauge the impact of destratification on the temperature of water released into the Mitta Mitta River. Previously, cold water releases had created difficulties down stream through the "burning-off" of crops receiving irrigation water from this system. Summer discharge water from Dartmouth Reservoir may be as low as 10-15 C colder than the normal temperature of the Mitta Mitta River. Concern was also expressed relating to the impact of this cold water on the biota of the river.

It was hoped that artificial aeration of Dartmouth Reservoir might break down the year-long thermal stratification of this system. Water temperature in summer prior to aeration was typically 22-25 C at the surface and 7-8 C at depth. The target temperature for discharge water receiving artificial aeration was set at 18 C (Croome and Welsh 1988, Welsh 1984).

To date the artificial aeration systems employed in Dartmouth Reservoir have not been effective. Aeration has failed to destroy the thermocline except in the immediate vicinity of the aerator. During the most recent aeration trial

CHAPTER 9.

CONCLUSIONS

9.1 The Performance of Artificial Destratification in Australia

The purpose of the analysis presented in this section is not to focus on destratification performance in individual cases, but rather to assess the degree of success the technique has achieved in addressing water quality problems throughout Australia. This assessment is based on the case studies presented in this report.

Performance has been judged as either successful, of limited success, a failure or uncertain, depending on the degree to which destratification solved the water quality problems for which it was implemented. In the case of chemical water quality variables, "successful" instances have been judged as those where the variables in question (eg iron concentrations) have been reduced to levels set as acceptable by the National Health and Medical Research Council (1987). Cases displaying marked reductions but not to these guideline levels have been categorized as of "limited success". For biological water quality problems, successful applications have been seen to be those resulting in the elimination of the problem, usually due to a major reduction in the numbers of the problem-causing organism(s). The results of the assessments are given in Tables 9.1 and 9.2.

Naturally such an evaluation is subjective, but should still serve to give insight into the success of destratification as a water quality management tool in Australia. It must also be remembered that this review has not had access to all water quality data for each system, nor information on all systems destratified throughout Australia. Both considerations may reduce the accuracy of such an assessment.

Specific water quality objectives often were not set for the destratification applications and therefore it was not possible to assess their success against case-specific objectives. Furthermore, it was often not possible to assess the adequacy of the system design or the way it was operated. Consequently we could not be sure that lack of success in improving water quality was due to the destratification technique itself or deficiencies in its implementation.

It must also be emphasised that the criterion for "successful" we used is very stringent, particularly where the objective of the destratification was for an improvement in water quality but not necessarily to comply with the National Health and Medical Research Council (1987) guideline.

The majority of cases evaluated in this review (70%) have had at least limited success in maintaining high water quality through the implementation of an artificial destratification program (Table 9.1). However only 24% of applications have been seen to more completely succeed in maintaining high water quality.

Table 9.1 Degree of effectiveness of artificial destratification in maintaining high water quality in the systems listed in Appendix 1.

Degree of Success	Number of Cases
Success	12 (24%)
Limited Success	23 (46%)
Failure	14 (28%)
Uncertain	1

Evaluation of the degree of success achieved in response to particular water quality difficulties provides a better insight into why such a high proportion of cases have enjoyed only limited success. As may be seen from Table 9.2, artificial destratification has been most successful in raising dissolved oxygen levels, reducing colour and turbidity; and in reducing levels of iron, manganese and sulphides in the water column. This degree of success has not, however, carried over into the field of algae control.

Table 9.2. The degree of success achieved over specific water quality problems in systems listed in Appendix 1.

Water Quality Problems	Number of Cases		
	Success	Limited Success	Failure
Unpleasant tastes & odours	3 (25%)	1	8 (67%)
High chlorine demand	1	-	1
High algae levels	4 (16%)	4 (16%)	17 (68%)
Undesirable algae species	4 (27%)	4 (27%)	7 (46%)
High colour and turbidity	4	6	2
High iron or manganese or sulphides	12 (38%)	10 (31%)	10 (31%)
Variable water quality	7	1	1
Low DO levels/release	23 (74%)	-	8
Low temperature	-	-	1
High zooplankton/insect numbers	1	-	1
Fish kills	1	-	1
Dirty water complaints	-	-	1

9.2 Where Artificial Destratification has Failed

In many cases artificial destratification has successfully reduced the release of iron, manganese, sulphides and other reduced ions from the sediment. In such cases the destratification program has maintained high levels of dissolved oxygen at the sediment interface, ensuring high redox potentials and minimising the reduction of iron, manganese and sulphate compounds. In cases where destratification has failed to lower the release of iron, manganese etc from the sediment, the failure can usually be traced back to an inability to maintain high dissolved oxygen levels throughout the entire water column.

In a number of cases water bodies have experienced both sediment release and algae difficulties. In some of these cases even though control over sediment release appears to have been achieved, algae levels have remained high. For example, artificial aeration has had "limited success" in addressing the water quality difficulties historically experienced in Little Bass Reservoir in Victoria (Case 18). Prior to artificial aeration surface water during summer was poor in quality due to algae blooms, while bottom water was unusable due to the accumulation of reduced substances such as iron and manganese.

Artificial aeration now successfully maintains high quality bottom water (ie low in Fe, Mn), yet surface waters still often experience high algae levels. Other similar examples exist.

In the cases where control over algae levels have not been achieved the cause is difficult to identify. Where destratification has reduced iron and manganese release (and presumably nutrient release) yet not controlled algae levels, it seems likely that some other factor is, in the main, controlling algae numbers. The responses of algae to mixing are less well understood than the responses of sediment to dissolved oxygen levels and thus require more detailed investigation.

9.3 Areas Requiring Further Attention

As shown by Table 9.2 it is in the field of algae control that destratification has had least success and it is in this area that the role of destratification as a management tool appears to be least well understood. The degree of mixing experienced by the water column and the bearing this has on the growth of an individual algae cell needs to be well understood before one may accurately predict the response of the algae community to artificial destratification.

Where algae blooms occur in response to increased light rather than increased nutrients, mixing may be used to remove the algae cells away from the upper, well-lit zone and into the darker zones for periods long enough to retard their growth. To achieve this we need a better understanding of the relationships between:

- i) energy input and the movement of water,
- ii) the movement of water and the movement of algae cells,
- iii) the movement of cells and the amount of light they receive, and
- iv) the amount of light received and the effect on growth.

A better understanding of these principles, as well as a better understanding of the sediments' response to mixing in a particular water body, will go a long way to improving the ability to predict the effectiveness of artificial destratification.

It is also imperative that a broad point of view be taken when attempting to reduce water quality difficulties. The process responsible for a given difficulty will rarely be independent of other processes operating within the water body. Consideration must be given to all the interelationships and processes that combine to determine water quality in each particular system.

GLOSSARY OF TERMS

algae:	Simple, mostly aquatic organisms containing photosynthetic pigments. There are marine and freshwater types of both single and multicellular forms.
biomass:	The total living organic matter present in a specified situation eg the biomass of the surface waters.
blue-green algae:	Unicellular or filamentous algae without a well defined nucleus and usually blue-green in colour. Some species produce toxic substances.
chlorine demand:	A measure of the amount of chlorine consumed by reaction with variety of organic and inorganic components when chlorine is added.
chlorophylls :	The green pigments of plants that absorb light energy (necessary in the photosynthesis process).
cladocerans:	form of zooplankton.
copepods:	form of zooplankton.
cyanobacteria:	Blue-green algae.
cyanophyta:	Blue-green algae.

diatoms: Planktonic algae whose outer shells are composed of silica.

epilimnion: The uppermost and usually turbulent layer of a stratified water column, displaying relatively uniform temperatureand dissolved oxygen levels.

euphotic zone: That depth of water to which the light available allows photosynthese to equal or exceed respiration. Usually taken as that depth of water to which the downwelling irradiance of photosynthetically active radiation falls to 1% of that just below the surface. A general measure of the depth providing light for algae growth.

eutrophic water body: An enriched water body displaying high levels of nutrients and high levels of biological growth.

green algae: (Chlorophyta) A large group of algae containing a chlorophyll mix similar to higher plants. There are marine and freshwater types.

homiothermal: A water layer without temperature variation.

hypolimnion: The deep, relatively undisturbed and cold bottom layer of a stratified water column.

limnology: The study of inland waters.

mesotrophic water body: A water body displaying moderately enriched characteristics, between oligotrophic and eutrophic.

metalimnion: The layer between the hypolimnion and epilimnion in a stratified water column, usually the layer of the thermocline (the layer of maximum rate of change of temperature).

monomictic lake: A lake that mixes once per year.

oligotrophic water body : Waters of low trophic level, that is poor in nutrients and exhibiting low levels of biological growth.

overturn: The period when stratification breaks down and the water body mixes to develop more uniform vertically conditions

oxycline: The layer of water displaying the most rapid vertical change in oxygen concentration.

oxidation: The loss of electrons from an atom, often by the addition of oxygen to a compound.

phaeophytin: A breakdown product of chlorophyll, used as an indicator of the levels of recently dead algae cells.

photosynthesis: A process occurring in plants, algae and some bacteria. The synthesis of carbohydrates from light, carbon dioxide and water. Involves the production of oxygen and the consumption of carbon dioxide. Assisted by chlorphyll.

photo-oxidation: An increase in dissolved oxygen levels from oxygen produced directly by photosynthetic processes. For example photo-oxidation of an amount of water containing phytoplankton.

phytoplankton: The plant component of plankton.

plankton: Free-floating, microscopic plants and animals found in water bodies, both marine and freshwater.

pycnocline: As with "thermocline" but referring specifically to a vertical change in density rather than temperature.

respiration: A metabolic process of cells resulting in the release of energy in a form which the organism may use. These are both aerobic (oxygen consuming) and anaerobic forms.

rotifers: A phylum/class of microscopic zooplankton.

Secchi depth: A measure of light penetration through a water column using optical sighting of a black and white disk.

selective withdrawal : Removal of water from differing depths in the water column to supply water of the best quality.

standing crop: Biomass present in the water body at a given time.

stratification: The presence of discernible layers in a water body, usually due to different densities of water.

thermocline: The layer of water displaying the most rapid vertical change in temperature (metalimnion).

trophic status: A measure of the degree of enrichment of a water body. For example, oligotrophic, mesotrophic, eutrophic.

zooplankton: The animal portion of the plankton.

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APPENDIX 1

CASE STUDIES

APPENDIX 1

CASE STUDIES

Case	Overall Performance	Taste Odour	Chlorine Demand	High Algae Numbers	Undesirable Algae	Colour & Turbidity	Fe,Mn & Sulphides	Variable Water Qual	Low DO	Low Temp	Zooplankton	Fish Kills	Dirty Water
1	o	+	-										-
2	-			-			-						
3	o			-					+		-		
4	+								+				
5	o			-	o				+				
6	-	-		-	-								
7	o	-		-	-						+		
8	-	-		-					-				
9	-			-	-		-						
10	o					+	+	+	+				
11	o						o						
12	+						+		+				
13	-						-		-	-			
14	-						-		-				
15	+			+			+		+				
16	o			-		o	o		+				
17	+	+		+		+	+		+				
18	o	-		-			+		+				
19	o	-		-			+		+				
20	o			-	+								
21	+			+					+			+	
22	+					+	+		+				
23a	u					u	u						
23b	o					o	o						
24	o						o						
25	o					+	o		+				
26	o						o		+				
27	o			o					+				
28	+							+					
29	o					o			+				
30	-						-		-				
31	o	-		-		o	o	+					
32	-						-	o					
33	+					o	+	+	+				
34	o			o	o								
35	-			-	-								
36	-	-		-	-								
37													
38													
39	-			-	-		-		-				
40	+				o		+	+	+				
41													
42	-	-		-	-								
43	+		+			o	+	+	+				
44	+	+		+	+				+				
45a	o	o		o	+	-	-		-				
45b	o				+	-	o		-				
46	-						-		-				
47	-						-	-					
48													
49	o			-			o	+					
50	+						+		+				
51	o			o	o		+		+				
52	o						o		+				

+ = Success o = Limited Success - = Failure u = Uncertain

Table A1 Summary of case performance as evaluated by this review. Further evaluation is contained in Chapter 2

Table A2 A summary of the principle role of the water bodies during the period of aeration, following as case studies.

Principle Role	No. of Cases
Water supply storage :	41
Indirect supply :	1
Hydro-electricity :	3
Irrigation/recreation :	5
Fishery :	1
Aerator trial :	1

CASE 1

HARDING RESERVOIR

State:	Western Australia
Controlling Authority:	Water Authority of Western Australia
Location:	23 km South of Roebourne in North-Western Australia - latitude 20°45'S
Capacity at F.S.L.:	64×10^6 m^3
Surface Area at F.S.L.:	14 km^2
Maximum Depth:	20 m
Mean Depth:	4.6 m
Catchment Area:	1071 km^2
Nature of Catchment:	Native vegetation (sparse)
Water Quality Problems:	Taste and odour High chlorine demand Dirty water complaints
Management Practices:	Destratification
Destratification History:	

Effects of Artificial Destratification:

The Harding Dam was built to augment the capacity of the Pilbara water supply in North-Western Australia, which provides water to the Dampier, Karratha, Wickham and Roebourne areas. Shortly after water was utilized from the Harding

Reservoir a consumer complained of unpleasant tastes and odours. Results such as those shown in Table C1.1 have established a link between those tastes and odours and primary productivity within the water body.

Table C1.1 Flavour producing compounds identified from Harding Reservoir water (from Kavanagh 1986).

Compound	Taste/odour	Threshold taste concentration Range mg L^{-1}	Observed Concentration Range mg L^{-1}	Mode of Formation
Methyliso-borneol (MIB)	earthy/musty	30	6-25	biological activity
Geosmin	earthy/musty	10	3-10	biological activity
Dimethyl - trisulphide DMTS)	swampy	5 to 10	0-55	biological activity (?)
Dimethyl-tetrasulphide DMTeS)	swampy	5 to 10 (?)	0-45	biological activity (?)

Note 1. Threshold taste concentration (TTC): the minimum concentration at which the taste of a compound can reliably be detected in water.

Early investigations into this problem found that the difficulties "largely disappeared" during periods of natural destratification. On this basis artificial aeration to establish year-long destratification was recommended.

The sequence of events has been summarized by Rosich and Shier (1989).

"An outbreak of consumer complaints occurred in mid-late November 1985. They related to musty or earthy taste in the supply. It was claimed that it also made tea and coffee unpalatable. The source was shut off on November 28, 1985 because of taste problems. This generally unsatisfactory situation continued throughout the period of December 1985 to May 1986. The source was brought back on stream June 11, 1986.

During the period that the source was not in use because of unacceptable taste, further testing was carried-out to identify the nature of the agents causing taste and odour.

It was established during this period that the water body was eutrophic; stratification and destratification was occurring; the thermocline between 8 - 12 metres depth was reflected in lowered pH, higher iron and manganese concentrations in the hypolimnion; and taste thresholds as high as 30 were recorded; geosmin and 2-methylisoborneol were indicated as the cause of earthy-musty tastes and odours with algae being implicated. The lowering in dissolved oxygen concentrations at lower levels of the stratified body of water was considered undesirable. Low taste levels associated with uniform temperature profiles were noted."

To date, aeration continues in this system on a "trial basis". Aeration appears to have virtually eliminated the taste and odour difficulties, holding taste to a threshold of 2 - 3. However, excessive chlorine demand and dirty water complaints are now of major concern.

Further investigations are currently being carried out by the Water Authority of Western Australia and the Centre for Water Research, University of Western Australia, utilizing laboratory and field investigation in conjunction with mathematical modeling (DYRESM), to optimize the artificial aeration of Harding Reservoir and minimise primary productivity.

References:

Kavanagh (1986)

Rosich & Shier 1989.

CASE 2

BAROSSA RESERVOIR

State:	South Australia
Controlling Authority:	Engineering and Water Supply Department
Location:	37 km North-East of Adelaide - latitude $34^{o}15'$
Capacity at F.S.L.:	4.5×10^6 m^3
Surface Area at F.S.L.:	62 ha
Maximum Depth:	28 m
Mean Depth:	7.3 m
Catchment Area:	235 km^2
Nature of Catchment:	Isolated from a rural and agricultural catchment Primarily supplied from South Para Reservoir.
Water Quality Problems:	Primarily iron and manganese deposition Occasionally algae - phytoplankton
Management Practices:	Artificial aeration (once only) Water treatment plant Occasional copper sulphate dosings
Destratification History:	Once only destratification event.

Effects of Artificial Destratification:

Artificial aeration of this system took the form of a once off mixing during the 1984/85 summer to alleviate water quality problems associated with iron and manganese deposition. Aeration was not commenced until January, once stratification had already developed. As a result of this late start Brown (1986) reports there was a "very slow erosion of the hypolimnion with subsequent nutrient injection into the epilimnion ..... The storage did not become isothermal until April, about one month earlier than was usually the case". Surprisingly this nutrient injection had no apparent effect on phytoplankton populations within the system. Having said this it should be noted that Barossa has had an occasional algae problem. The last time copper sulphate dosing was in the summer of 1982/83.

In recent years Barossa has posed few water quality problems, hence the non-pursuance of artificial aeration. Algae levels have remained relatively low, while iron and manganese problems are controlled at the water filtration plant. (Engineering and Water Supply 1988).

References:

Brown (1986)

Engineering and Water Supply (1988)

CASE 3

HAPPY VALLEY RESERVOIR

State:	South Australia
Controlling Authority:	Engineering and Water Supply Department
Location:	16 km South of Adelaide - latitude 35°S
Capacity at F.S.L.:	12.7×10^6 m^3
Surface Area at F.S.L.:	188 ha
Maximum Depth:	22.5 m
Mean Depth:	6.8 m
Catchment Area:	451 km^2
Nature of Catchment:	Isolated from catchment. Water supplied from Mt.Bold Reservoir.
Water Quality Problems:	Taste and odour - zooplankton - algae/phytoplankton Low dissolved oxygen levels
Management Practices:	Destratification Copper sulphate dosing Water treatment plant being constructed
Destratification History:	1981 - Current 24hrs/day in summer period

Effects of Artificial Destratification:

Aeration was introduced into this system in 1981 to raise dissolved oxygen levels at the bottom outlet to the southern suburbs of Adelaide. Other than the need to supply oxygenated water, the main areas of concern within Happy Valley have been phytoplankton and more acutely zooplankton numbers. Both have caused odour problems within this system.

For example, the State Water Laboratory metropolitan report of 1985/86 mentions that, "odours of intensity greater than 2 in Happy Valley and at the

B/C point were caused by the green algae *Ankistrodesmus*, *Dictyosphaerium* and *Oocystis* on 4 occasions (vegetable, grassy) and probably zooplankton on 5 occasions (fishy)". Treatment did however remove these odours "In the Happy Valley distribution odours of intensity greater than 2 were chlorinous except for hydrocarbon odours on one occasion (Engineering and Water Supply 1988)".

Phytoplankton levels did change significantly with aeration. Initial high volume aeration destratified the system, created turbulent conditions and reduced numbers of the then dominant *Dictyosphaerium*. A subsequent reduction in air supply after one week allowed restratification. This low volume mixing brought bottom nutrients into the euphotic zone assisting an "exceptional biomass development of *Cyclotella*" and following zooplankton problems of that summer (Burch 1987).

As a result of initial attempts it was decided that the only possible useful course for aeration of Happy Valley is if mixing is of a level sufficient to maintain an oxidized sediment water interface. It was hoped that this would then reduce the transfer of productivity up through trophic levels within this system. Burch (1987) reported that prior to aeration the "concentrations of all nutrients were highest adjacent to the sediment and decreased progressively towards the surface." Thus it was deemed desirable to reduce the internal cycling of nutrients released from the sediment under low oxygen conditions.

Artificial aeration of Happy Valley continues each summer period. Stratification has however developed to some extent each year and zooplankton numbers still cause concern. Copper sulphate treatment to indirectly control zooplankton numbers (through phytoplankton control) was required on 6 occasions in the summers of both 1984/85 and 1985/86 and on 3 occasions in 1986/87.

Regardless of the artificial aeration currently implemented, zooplankton (mainly *Calamoecia* and *Ceriodaphnia)* seem to thrive on the green algae of this system, which in turn exist fruitfully in the waters of Happy Valley Reservoir.

References:

Brown (1986)

Burch (1987)

Engineering and Water Supply (1988)

CASE 4

HINDMARSH RESERVOIR

State:	South Australia
Controlling Authority:	Engineering and Water Supply Department
Location:	60 km South of Adelaide - latitude 35°30'S
Capacity at F.S.L.:	0.5×10^6 m^3
Surface Area at F.S.L.:	10.5 ha
Maximum Depth:	13.5 m
Mean Depth:	4.3 m
Catchment Area:	65 km^2
Nature of Catchment:	Primarily natural bushland. Some water supplied offstream from the Hindmarsh River.
Water Quality Problems:	Taste and odour - low dissolved oxygen
Management Practices:	Destratification Copper sulphate dosing
Destratification History:	Initially aerated in 1982 Recommenced 1988/89

Effects of Artificial Destratification:

During the 1980/81 summer a high number of taste and odour complaints were linked to low dissolved oxygen levels (rather than algae levels) in the reservoir. Measurements of dissolved oxygen in the outlet pipe recorded values less than 1 mg L^{-1} dissolved oxygen. As an immediate solution water supply was changed to Myponga Reservoir, while as a more long-term measure an artificial aeration system was installed. This system proved effective in raising dissolved oxygen levels above 5 mg L^{-1}. Even though aeration raised dissolved oxygen levels, Hindmarsh was not used as a water supply for the 3 years prior to the 1988/89 summer. This was due to poor water quality associated mainly with high colour. As other initiatives (including sediment dredging and a selective filling strategy) have improved water quality the system is expected to be once again used for water supply.

To avoid a repeat of the complaints received in 1980/81 the aeration system will also be recommissioned.

References:

Brown (1986)

Engineering and Water Supply (1988)

CASE 5

KANGAROO CREEK RESERVOIR

State:	South Australia
Controlling Authority:	Engineering and Water Supply Department
Location:	18 km South of Adelaide - latitude 34°15'S
Capacity at F.S.L.:	24.4×10^6 m^3
Surface Area at F.S.L.:	121 ha
Maximum Depth:	50 m
Mean Depth:	20 m
Catchment Area:	289 km^2
Nature of Catchment:	Mainly low density agriculture
Water Quality Problems:	Taste and odour - algae/phytoplankton blooms Low dissolved oxygen
Management Practices:	Destratification Copper sulphate dosing
Destratification History:	Two trial periods 1977/78, 1978/79

Effects of Artificial Destratification:

Kangaroo Creek Reservoir has at times experienced problems associated with both deoxygenated water and high algae levels. Annual stratification from September to May has, in the past, led to deoxygenated water entering Adelaide's reticulation system. Algae levels have required copper sulphate dosing 15 times between 1971 and 1978. Eighty five percent of dosings were to treat the green alga *Ceratium*, while the blue-green algae *Microcystis* and *Anabaena* have led to most consumer complaints.

Aeration was installed on a trial basis to evaluate its impacts on these problems. The aerator was run for 19 days during the 1977/78 summer, (trial 1) and for 47 days during the 1978/79 summer, (trial 2).

Aeration had marked effects on temperature and dissolved oxygen in the reservoir, causing isothermal conditions to the level of the aerators. During the second trial period, monitoring of the water entering the reticulation system revealed dissolved oxygen levels below 6 mg L^{-1} on only two occasions. In the previous non aerated year dissolved oxygen levels had been less than 6 mg L^{-1} for 15 weeks over the same period.

The effect of aeration on phytoplankton in this system is somewhat unclear. Immediately after trial 1 there was a reduction in algae biomass, however an input of *Ceratium* from another reservoir led to a bloom requiring copper sulphate dosing. The role played by aeration in this process is unclear. The second trial displayed no increase in phytoplankton levels due to aeration. There was a change in species composition, with dominance transferring from *Cryptomonas* to *Cyclotella* and *Closterium* (Croome 1979).

Blue-green algae have not been a problem in Kangaroo Creek Reservoir since 1983/84. The system remains green and diatom dominated without aeration. The reservoir still stratifies, with relatively high algae levels. During June 1986 high *Ceratium* levels contributed to a chlorophyll *a* level of just over 25 ug L^{-1}.

Kangaroo Creek is a storage reservoir which does not supply water directly to the distribution system. The Department of Engineering and Water Supply's current policy is not to treat storage reservoirs for algae blooms. Water from Kangaroo Creek Reservoir is transferred down the Torrens River, diverted into Hope Valley Reservoir, and then to a filtration plant prior to entering the distribution system. Destratification and copper sulphate dosing have ceased in this system since 1980 and 1981 respectively (Engineering and Water Supply 1988).

References:

Brown (1986)

Croome (1979)

Engineering and Water Supply (1988)

CASE 6

LITTLE PARA RESERVOIR

State:	South Australia
Controlling Authority:	Engineering and Water Supply Department
Location:	23 km North-East of Adelaide - latitude 34°30'S
Capacity at F.S.L.:	21.4×10^6 m^3
Surface Area at F.S.L.:	150 ha
Maximum Depth:	49.5 m
Mean Depth:	14.2 m
Catchment Area:	83 km^2
Nature of Catchment:	Principally cleared land
Water Quality Problems:	Taste and odour - algae/phytoplankton blooms
Management Practices:	Destratification Copper sulphate dosing Water Treatment Plant
Destratification History:	1978 - Current Operated 24hrs/day in summer period

Effects of Artificial Destratification:

When first implemented, the aeration system in Little Para destratified the storage down to the level of the aerators. This resulted in an improvement in colour and turbidity as well as more uniform temperature and dissolved oxygen levels. Nutrient, iron and manganese levels were also more evenly distributed throughout the water column. It is possible that this redistribution of nutrients contributed to a change in the phytoplankton community after aeration, which culminated in a copper sulphate dosing for *Microcystis* in January 1979. Water was supplied to Adelaide from this storage for the first time five days later (Croome 1979).

Aeration has continued each summer in the Little Para system. Typically this water body is dominated mainly by green algae and diatoms, however, "microstratification" (surface heating) has disturbed this balance in each summer period from 1983 to 1987. During the summer of 1985/86 aeration was used from October through May and maintained dissolved oxygen at levels greater than 7 mg L^{-1}. Microstratification as represented by temperature profiles displayed in Table C6.1 did however result in *Anabaena* blooms requiring dosing with copper sulphate twice during November 1985. Apart from these blooms chlorophyll *a* levels remained relatively low, with almost 90% of samples less than 10 ug L^{-1}.

Table C6.1. Temperature profiles displaying microstratification in Little Para Reservoir during the summer period 1985/86 (Engineering and Water Supply 1988).

Depth (m)	22/10/85	29/10/85	12/11/85	22/11/85	17/12/85
Surf	14.8	16.3	20.0	18.0	18.3
1	14.7	16.3	19.5	18.0	18.3
2	14.7	16.3	18.6	18.0	18.3
3	14.7	16.3	17.9	18.0	18.3
4	14.7	16.3	17.8	18.0	18.3
5	14.7	16.0	17.7	17.9	18.3
6	14.7	15.8	17.6	17.7	18.3
7	14.7	15.8	17.5	17.6	18.3
8	14.8	15.7	17.5	17.5	18.3
9	14.7	15.6	17.3	17.5	18.3
10	14.7	15.5	17.3	17.5	18.3
15	14.5	15.3	17.2	17.4	18.3
20	14.3	15.0	16.8	17.3	18.3
25	14.3	14.5	16.2	17.3	18.3

Microstratification again led to a blue-green algae bloom in Little Para in March 1988. It appears that although the changes in temperature were only slight and did not reduce oxygen levels at depth, the stability this layering represents may have favoured *Microcystis* . Temperature profiles taken during this period are displayed in table C6.2.

Table C6.2 Temperature profiles displaying microstratification during the period of a *Microcystis* bloom in March 1988. (Engineering and Water Supply 1988).

Depth (m)	16/02/88	23/02/88	4/03/88	8/02/88	11/03/88
Surf	21.6	21.3	20.5	21.9	20.2
1 m	21.6	21.3	20.5	21.9	20.2
2 m	21.0	21.3	20.6	21.8	20.2
3 m	20.7	21.2	20.6	21.5	20.2
4 m	20.5	21.0	20.6	21.4	20.1
5 m	20.5	20.9	20.6	21.3	20.1
6 m	20.4	20.9	20.7	-	20.1
7 m	20.3	20.8	20.7	-	20.1
8 m	20.3	20.7	20.7	-	20.1
9 m	20.3	20.7	20.7	-	20.1
10 m	20.3	20.7	20.7	21.3	20.1
15 m	20.3	20.7	20.7	21.2	20.0
20 m	20.3	20.5	20.7	21.2	20.0
25 m	20.3	20.5	20.5	21.2	20.0

Apart from the taste and odour problems often associated with *Microcystis* blooms, there is also a possibility of the genus producing toxins. Such was the case with the bloom in Little Para when surface scums were analysed. This toxicity was apparently confined to the lake and no toxins were detected in the intake water. In response to this bloom the lake was dosed with copper sulphate and activated carbon was introduced at the filtration plant to eliminate odours. The aerator was turned off during dosing. (Engineering and Water Supply 1988).

The Engineering and Water Supply Department has concluded that the current aeration facility in Little Para Reservoir remains inadequate for effective control of blue-green algae.

References:

Brown (1986)

Engineering and Water Supply (1988)

CASE 7

MYPONGA RESERVOIR

State:	South Australia
Controlling Authority:	Engineering and Water Supply Department
Location:	54 km South of Adelaide - latitude 35°30'S
Capacity at F.S.L.:	26.8×10^6 m^3
Surface Area at F.S.L.:	280 ha
Maximum Depth:	40 m
Mean Depth:	9.6 m
Catchment Area:	124 km^2
Nature of Catchment:	Agricultural and bushland, small urban area
Water Quality Problems:	Taste and odour - Algae/phytoplankton blooms Phantom midge larvae
Management Practices:	Destratification Copper sulphate dosing Water treatment plant
Destratification History::	1980 - 1988/89 Artificial aeration 1988/89 - Mechanical mixing Mixers operate 24hrs/day in the summer period.

Effects of Artificial Destratification:

This system was initially aerated in 1980 in the form of an "air curtain" around the offtake tower to prevent *Chaoborus* (phantom midge) larvae entering the reticulation system. Downstream consumers in the southern suburbs of Adelaide typically complained of "worms" in their water, when phantom midge larvae were discharged from Myponga.

The air curtain management strategy proved effective in containing these intrusions, not through acting as a barrier, but rather through deepening the epilimnion. As explained by Suter (1983) "The air curtain did not keep the chaoborid larvae away from the outlet tower, but its main effect was to increase the depth of the epilimnion, ensuring the offtake valves were always in well oxygenated water. Because the chaoborids have a diurnal vertical migration behaviour, occupying the cooler, oxygen poor waters of the thermocline and hypolimnion during the day and the warmer aerated epilimnion during the night, the high concentrations of the midge larvae were physically separated from the offtake values during the period of maximum water flows .... The aerator also seemed to have an effect in reducing the population size of phantom midge in the Myponga Reservoir. This combined with the increased depth of the epilimnion controlled the phantom midge problem experienced in 1980."

Myponga Reservoir has also commonly experienced up to three blue-green algae blooms per year, typically around December, January and April. During the summer of 1983/84 the reservoir was successfully aerated in terms of the phantom midge problem, but not so with regards phytoplankton. Inefficient aeration brought nutrients from depth into the well lit epilimnion throughout the stratification period, October 1983 - April 1984.

By November 1983 *Microcystis* was the dominant phytoplankton and required copper sulphate dosing in December and January. Immediately post treatment Myponga returned to a green algae dominated system however *Ceratium* and *Anabaena* required dosing by March 1984. Odour problems were associated with each of these blooms, hence the need for dosing.

Until the summer of 1988/89 Myponga had been artificially aerated in response to the onset of stratification. Regardless of this, similar problems to those experienced in 1983/84 have occurred each summer. Odour problems during the 1985/86 summer were believed to be associated with *Synura*, *Microcystis*, *Anabaena* and the green algae *Ankistrodesmus*, *Oocystis* and *Scenedesmus*. *Anabaena* levels required dosing twice. Similar problems occurred during the 1987/88 summer with *Anabaena* requiring copper sulphate dosing on 4 occasions. Aerator failure for prolonged periods also caused pronounced stratification at depths greater than 10m. As the level of the thermocline rose, complaints of *Chaoborus* larvae in the distribution system once again occurred.

In an effort to improve mixing efficiency and control both phantom midge and algae levels the aeration system at Myponga has been replaced by three "flyght" mixers mounted on tracks attached to the dam wall (Figure C7.1). The mixers may be set at variable depths and angles relative to the water surface. It is believed the algae blooms experienced during the summer period in Myponga can be reduced with efficient and thorough mixing.

During well mixed periods Myponga typically displays relatively high nutrient concentrations that are not usually reflected in high biomass (Table C7.1). Myponga appears to be a light-limited system where native turbidity and colour levels severely restrict the depth to which algae may grow.

Table C7.1. Examples of nutrient and biomass levels from Myponga Reservoir.

Date	P04	TP	Chl *a*	Total Cells/mL
03/08/87			0.51	18
10/08/87			0.46	38
17/08/87	55	110	0.67	143
24/08/87			0.39	67
31/08/87			0.56	50
21/09/87	60	87	1.44	327
28/09/87			2.50	50

It is believed that through mixing will maintain low levels of algae through removal of cells from the euphotic zone for periods of time sufficient to inhibit growth. Further discussion of this principle is contained in Chapter 5 "Algae Growth". The effectiveness of the mixers in Myponga is currently under investigation. Initial indications are that they will provide effective mixing to approximately 2-4m below the level at which they are set.

To date information is still being gathered to understand the most effective mixing strategy for this system. During December 1988 the mixers were set at 25m providing good oxygen concentrations to 27m depth, however, as was a problem with artificial aeration, surface heating allowed "microstratification" to develop, leading to rising *Anabaena* levels. *Anabaena* rose from 300 cells/mL to 1800 cells/mL in approximately 4 days. Although this only represented a chlorophyll *a* level of 6 ug L^{-1}, copper sulphate dosing was required due to the potential consumer problems associated with *Anabaena*.

It is hoped that as a better understanding of these mixers as an operational tool is gained, such instances may be eliminated. The mixers allow considerable versatility and a short response time, within half a day of recognising a surface heating problem one mixer should be able to be raised to disturb any developing stability in the shallow 2m euphotic zone.

References:

Brown (1986)
Engineering and Water Supply (1988)
Suter (1983)

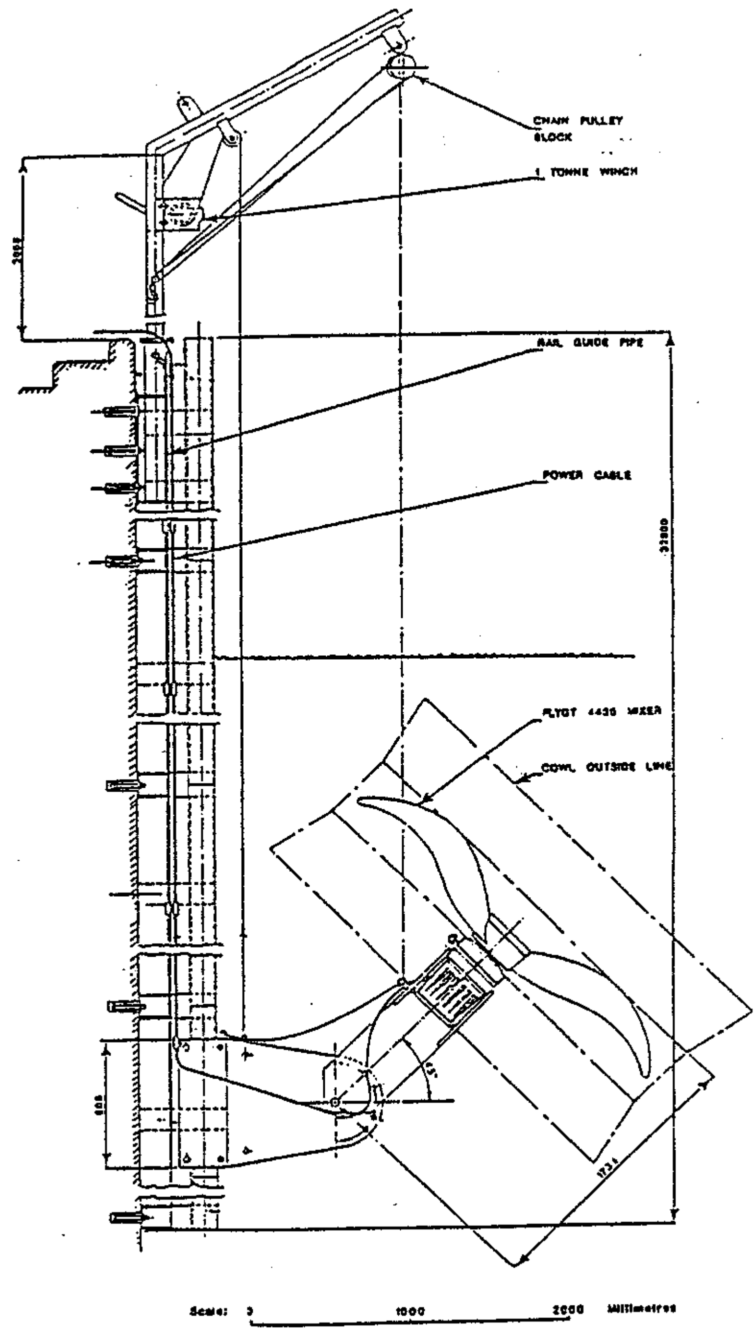

Figure C7.1 A representation of the artificial mixing system operated in Myponga Reservoir during the 1988/89 summer period (Engineering and Water Supply 1988).

CASE 8

WARREN RESERVOIR

State:	South Australia
Controlling Authority:	Engineering and Water Supply Department
Location:	40 km North-East of Adelaide - latitude 34°30'S
Capacity at F.S.L.:	4.77×10^6 m^3
Surface Area at F.S.L.:	105 ha
Maximum Depth:	17.4 m
Mean Depth:	6.45 m
Catchment Area:	119 km^2
Nature of Catchment:	Principally bushland, some agriculture
Water Quality Problems:	Low dissolved oxygen levels Taste and odour - Algae/phytoplankton blooms
Management Practices:	Destratification Copper sulphate dosing Water treatment plant

Destratification History::

Compressor:	250 cfm run initially for 8 hours/day then operated continuously.
Pipework:	2 x 6m 25mm UPVC.
Diffusers:	nil, holes drilled in pipes.
Commenced:	January 1988 after strong thermal stratification had developed. Continued until May.

Effects of Artificial Destratification:

Water from Warren Reservoir is chloraminated prior to entering the discharge system, therefore high dissolved oxygen levels are desirable. Prior to aeration, dissolved oxygen levels fell with the development of stratification. Initially aeration elevated dissolved oxygen levels to above 5 mg L^{-1} throughout the water column, this happy situation was however short-lived. During the summer period of 1988/89 strong stratification developed bringing with it low dissolved oxygen and high algae levels. It appears as if current aeration is insufficient to maintain acceptable water quality in Warren Reservoir.

References:

Engineering and Water Supply (1988)

CASE 9

AROONA RESERVOIR

State:	South Australia
Controlling Authority:	Electricity Trust of South Australia
Location:	Leigh Creek
Capacity at F.S.L.:	
Surface Area at F.S.L.:	85 ha
Maximum Depth:	
Mean Depth:	
Catchment Area:	Small catchment area via two streams
Nature of Catchment:	Desert country, has been heavily grazed
Water Quality Problems:	High iron and manganese Algae blooms
Management Practices:	Destratification Copper sulphate dosing
Destratification History::	December 1988 -

Effects of Artificial Destratification:

Due to repeated dam overflows, it is difficult to evaluate the initial impact of destratification. Based on the experience of early December when the reservoir failed to destratify the Electricity Trust believe it may be necessary to redesign the aeration equipment.

Algae blooms historically experienced in this system have included *Microcystis*, *Volvox* and *Botryococcus*.

References:

- Electricity Trust (1989)

CASE 10

RUNNING CREEK RESERVOIR

State:	Victoria
Controlling Authority:	Melbourne and Metropolitan Board of Works
Location:	40 km North of Melbourne - Latitude $37^{o}45'$S
Capacity at F.S.L.:	0.3×10^6 m^3
Surface Area at F.S.L.:	5 ha
Maximum Depth:	14 m
Mean Depth:	5 m
Catchment Area:	192 km^2
Nature of Catchment:	Mixture of forest and agriculture.
Water Quality Problems:	High and variable colour and turbidity High and variable temperature High and variable Fe/Mn levels.
Management Practices:	Destratification Water treatment plant.
Destratification History::	1985/86 - 1988/89 Decommissioned January 1989.

Effects of Artificial Destratification:

Running Creek reservoir was first aerated in the summer of 1985/86 in an attempt to reduce both the variability and absolute levels of iron, manganese, colour and turbidity. Prior to the implementation of artificial destratification the associated water treatment plant had great difficulty in coping with large variations in these parameters. Apart from a more stable water quality other improvements due to destratification are:

- a reduction in Fe and Mn levels;
- lower turbidity and colour:
- higher levels of dissolved oxygen.

The aerator has been operated in an automatic mode each summer, switching on when the temperature difference between the water surface and aerator level exceeds 2°C. This strategy has resulted in better overall water quality being provided to about 90% of downstream consumers.

Despite these improvements water quality in the system was still poor, requiring expensive treatment. Given this and the small size of the reservoir it was decided to decommission the supply in January 1989. As a result both the aeration facility and the treatment plant are no longer in use.

References:

Brown (1986)

Melbourne and Metro Board of Works (1988)

CASE 11

SUGARLOAF RESERVOIR (FORMERLY WINNEKE)

State:	Victoria
Controlling Authority:	Melbourne and Metropolitan Board of Works
Location:	35 km North East of Melbourne - Latitude 37°45'S
Capacity at F.S.L.:	95.0×10^6 m^3
Surface Area at F.S.L.:	405 ha
Maximum Depth:	68 m
Mean Depth:	19.2 m
Catchment Area:	Pumped from Yarra River & Maroondah Reservoir
Nature of Catchment:	Mixture of forest and agriculture.
Water Quality Problems:	Iron, manganese and sulphide levels
Management Practices:	Destratification Water treatment plant.
Destratification History::	Feb - May 1984 Dec - May 1985

Effects of Artificial Destratification:

Prior to aeration, manganese levels posed considerable difficulties in the treatment plant associated with this reservoir. The characteristics of the source water greatly influence the water quality in this reservoir, hence data interpretation must proceed cautiously. It does however appear that artificial aeration improved water quality in this system. During the operation of the aerator manganese levels were lowered, ranging from 0.02 to 0.1 mg L^{-1}.

The role played by aeration is somewhat unclear as low levels of manganese have been maintained in the epilimnion since 1985 without aeration. Utilizing some changes to the water treatment plant and a selective withdrawal strategy manganese levels have generally been kept below 0.02 mg L-1 in the treatment water. The aerator has not been used since 1985.

References:

Brown (1986)

Melbourne and Metro Board of Works (1988)

CASE 12

THOMSON DAM

State:	Victoria
Controlling Authority:	Melbourne and Metropolitan Board of Works
Location:	175 km East of Melbourne - Latitude 37°45'S
Capacity at F.S.L.:	146.2×10^6 m^3
Surface Area at F.S.L.:	600 ha
Maximum Depth:	80 m
Mean Depth:	24.4 m
Catchment Area:	486 km^2
Nature of Catchment:	Primarily forest
Water Quality Problems:	Iron, manganese and sulphide levels Low dissolved oxygen
Management Practices:	Destratification
Destratification History::	1984 - 1985 In use prior to selective withdrawal

Effects of Artificial Destratification:

Thomson Dam began filling in 1983 and in 1988 was about 60% full. The dam was constructed with only a bottom outlet therefore there was considerable concern for ongoing water quality. With filling, stratification developed and resulted in oxygen depletion at depth. This gave rise to elevated concentrations of iron, manganese, ammonia and sulphide.

Aeration has led to lower levels of these parameters than would otherwise be present during the stratified period. "Water quality of both the reservoir and outflows was generally improved following use of the artificial mixing system" (Melbourne and Metro Board of Works 1988).

Information up to mid 1984 details the following:

- Turbidity levels have decreased throughout the entire water column due to:
  (a) a decrease in the amount of surface material washed in, and
  (b) promotion of flocculation processes by the aeration system.

- Sulphides are no longer present.

- A reduction in manganese levels to 0.05-0.07 mg L^{-1}. Without the aerator they could be expected to be >0.5 mg L^{-1} in the hypolimnion, similar effects have been noted with total iron.

However, results such as turbidity changes should be interpreted cautiously. Changes may have occurred due to:

(a) aerator effects;
(b) natural effects of storage and reduced storm input.

Since the commissioning of the multi-level offtake towers (1985) the aerator has been abandoned. The selective withdrawal facility has enabled good quality water to be released throughout the year.

References:

Brown (1986)
Melbourne and Metro Board of Works (1988)

CASE 13

DARTMOUTH RESERVOIR

State:	Victoria
Controlling Authority:	Rural Water Commission of Victoria
Location:	280 km North-east of Melbourne - latitude 36°40'S
Capacity at F.S.L.:	4000 x 106 m3
Surface Area at F.S.L.:	-
Maximum Depth:	164 m
Mean Depth:	-
Catchment Area:	3600 km^2
Nature of Catchment:	Primarily bushland and forest
Water Quality Problems:	Cold water releases Low dissolved oxygen High levels of sulphides
Management Practices:	Destratification
Destratification History::	December 1980 Trial 1 Dec - Feb 1982/83 Trial 2 Dec - Jan 1987/88 Trial 3

Effects of Artificial Destratification:

These trials were conducted to gauge the impact of destratification on temperature and water quality both within the reservoir and in that water released into the Mitta Mitta River. In the past, concern had been expressed due to adverse environmental effects of cold water discharges from Dartmouth. These included "burning off" of crops and the release of distasteful hydrogen-sulphide odours. Summer discharge water from below the thermocline in Dartmouth may be anoxic, high in sulphide, iron and manganese, as well as being 10-15 C colder than the normal temperature of the Mitta Mitta River.

It was hoped that the aeration system would break down the year long thermal stratification of Dartmouth. Water temperature in summer prior to aeration was typically 22-25 C at the surface and 7-8 C at depth, with aeration the target for discharge water was 18 C.

Trial 1

The aerator "failed to destratify the reservoir even partially. The failure was due to an air line blockage leading to an air flow less than design capacity to the aerators, only 95-115 $Lsec^{-1}$ of the design capacity of 450 $Lsec^{-1}$ was supplied "(Croome and Welsh 1988).

Trial 2

"This second destratification attempt has also been regarded as unsuccessful..... In retrospect, it appears that mixing occurred over a large area at this time. Nonetheless, the reservoir was stratified for most of the period over which the destratification equipment was used" (Croome and Welsh 1988). This failure to destratify was probably due to restricted air flow coupled with the generation of momentum currents from withdrawal.

Trial 3

Operational Parameters for this aeration system are displayed in Table C13.1. This system failed to destroy the thermocline except in the immediate vicinity (40m) of the aerator, indeed thermal stratification intensified over the period of the trial. As a result of this poor mixing there was also no apparent rise in released water temperature. Water released into the Mitta Mitta River measured around 11 C, well short of the target of 18 C. Dissolved oxygen levels were low at depth and a hydrogen-sulphide smell was noticeable at times immediately below the dam. Dissolved oxygen levels quickly rose as water travelled down the river.

As a result of the repeated failure of these trials the possible role destratification may play in Dartmouth is being fully re-evaluated. The report of Croome and Welsh 1988 recommends that the target release temperature of 18 C is likely to be unrealistic, and propose to utilize mathematical modelling techniques (see DYRESM, Chapter 4), to help evaluate the role of destratification.

References:

Brown (1986)

Croome and Welsh (1988)

Welsh (1984)

Rural Water Commission (1988)

Table C13.1 Operational parameters of aeration system used in trial 3 at Dartmouth Reservoir (Croome and Welsh 1988).

	8.12.87	11.12.87
Aeration length	3 x 12 m	3 x 12 m
No. of outlets	48	48
Depth below FSL	95 m	95 m
Av Op hr d^{-1}	20	20
Air pressure kPa	980	1070
Air flow $l\ s^{-1}$	350	500

CASE 14

LAKE EPPALOCK

State:	Victoria
Controlling Authority:	Rural Water Commission of Victoria
Location:	25 km East of Bendigo - latitude 37°S
Capacity at F.S.L.:	311×10^6 m^3
Surface Area at F.S.L.:	3200 ha
Maximum Depth:	37.5 m
Mean Depth:	9.6 m
Catchment Area:	2120 km^2
Nature of Catchment:	Primarily Agriculture with considerable erosion.
Water Quality Problems:	Fe/Mn deposition in pipelines.
Management Practices:	Destratification trials.
Destratification History::	2 trials 1977 and 1980/81

Effects of Artificial Destratification:

Artificial destratification was evaluated in an attempt to reduce the levels of iron and manganese entering the Bendigo pipeline. The efficiency of water moving through such a pipeline may be greatly reduced, due to increased friction from bacterial deposition of iron and manganese slimes. Increased levels of iron and manganese in the reservoir during the summer stratification period exacerbated this problem. .

The first trial in 1977 achieved stratification down to the level of the aerators, improved dissolved oxygen levels and reduced iron and manganese concentrations. After this encouraging result a second trial was initiated for the summer of 1980/81. During this trial the lake developed anoxic conditions at depth, with only 20% saturation above the level of the aerators. Perhaps the two main reasons for the apparent failure of this second trial are summarised in conclusions drawn by Welsh (1981).

"Artificial aeration in order to prevent stratification should be attempted earlier than November, when weather conditions are less likely to prevent the destratification:, and

"The use of thermal stability charts and measurement for the operational management of destratification should be discouraged".

Since this time destratification has not been taken up as a management option in Lake Eppalock.

References:

Brown (1986)

Rural Water Commission (1988)

Welsh (1981)

CASE 15

TARAGO RESERVOIR

State:	Victoria
Controlling Authority:	Rural Water Commission of Victoria
Location:	90 km South-East of Melbourne - latitude 38^oS
Capacity at F.S.L.:	$37.6 \times 10^6 m^3$
Surface Area at F.S.L.:	360 ha
Maximum Depth:	23 m
Mean Depth:	10.5 m
Catchment Area:	-
Nature of Catchment:	Primarily eucalypt forest
Water Quality Problems:	Trial Aeration only
Management Practices:	Destratification trials.
Destratification History::	2 trials 1977 and 1980/81

Effects of Artificial Destratification:

Aeration trials were conducted at Tarago Reservoir during 1976 and 1977 to:

1. "Establish the practicability and economics of destratification as a reservoir management technique ....";

2. "... to investigate the effects of artificial destratification on all aspects of water quality ...". (Bowles *et al* 1979).

Destratification in Tarago Reservoir was effective both in breaking down existing temperature stratification in summer, and in preventing the formation of temperature stratification in spring. Destratification also led to an increase in dissolved oxygen levels down to the level of the aerators, however mixing was not sufficient to prevent stratification of dissolved oxygen. High oxygen levels were restored to the water column only after natural overturn.

A reduction in iron and manganese at depth was noted as a result of destratification. This reduction should have two beneficial effects.

"Calculations of iron loading indicate that about 86 t (8.62×10^4 kg) of iron would enter the pipeline per annum, of this, 88% is due to the withdrawal of hypolimnion water. If destratified water was used throughout the summer period, the total loading would decrease to about 18 t (1.83×10^4 kg) per annum. Thus destratifying Tarago Reservoir would perform two functions, firstly it would improve the quality of the potable water supply and, secondly, by preventing the introduction of a high iron supply, it would reduce deposition in the pipeline". (Bowles *et al* 1979).

Destratification also increased the depth of light penetration into the water column. This potentially poses problems of increased primary productivity and algae levels. Increased light penetration means the depth of water in which algae are able to grow, given sufficient nutrients, is also increased.

Even though light penetration increased in this system no detrimental effect is probable. Mathematical interpretations suggest that light was the main factor controlling growth in this system prior to destratification. Even though light penetrations increased so did the depth of mixing. This accentuated the "light limitation" as algae was mixed into the deeper dark layers where it cannot survive. This principle is discussed in detail in Chapter 5, "Algae Growth".

References:

Bowles *et al* (1979)
Brown (1986).

CASE 16

BULLARTO RESERVOIR

State:	Victoria
Controlling Authority:	Daylesford Water Board
Location:	100 km North-west of Melbourne - latitude 36°30'S
Capacity at F.S.L.:	0.2×10^6 m^3
Surface Area at F.S.L.:	-
Maximum Depth:	6 m
Mean Depth:	-
Catchment Area:	-
Nature of Catchment:	State Forest
Water Quality Problems:	Taste and Odour - algae blooms High Fe/Mn, colour and turbidity
Management Practices:	Destratification
Destratification History::	1985 - Current 6 hrs/cycle, 2 cycles/day

Effects of Artificial Destratification:

Bullarto and Wombat Reservoirs have had similar water quality problems. Prior to the implementation of artificial aeration good quality water was not readily available during the summer period. Summer stratification led to high levels of taste and odour producing algae in surface waters, and disagreeably high levels of iron, manganese, colour and turbidity in the deeper waters. As with Wombat Reservoir, destratification has had considerable success in combating some of these problems. Table C16.1 compares selected water quality parameters before the implementation of aeration with more recent readings taken during January 1989.

Table 16.1 A comparison of selected water quality parameters before and during artificial aeration in Bullarto Reservoir. (Daylesford Water Board 1989, Burns 1989).

PRE-AERATION (December 1985)		
	Depth	
	Surface	7.00 m
Dissolved Oxygen (%)	94.00	3.00
Iron total (mg L^{-1})	0.63	1.42
Iron soluble (mg L^{-1})	0.55	0.82
Turbidity (FTU)	4.00	26.00
True colour (Pt-Co)	80.00	140.00

DURING AERATION (January 1989)		
	Depth	
	Surface	5.00 m
Dissolved Oxygen (%)	90.00	84.00
Iron total (mg L^{-1})	0.42	0.43
Iron soluble (mg L^{-1})	0.37	0.38
Turbidity (FTU)	2.00	1.00
True colour (Pt-Co)	22.00	16.00

This beneficial response to destratification has not been as marked with regards to algae. Taste and odour producing algae are still found during the summer period in Bullarto Reservoir. Although a complete elimination of such algae would be desirable, good quality water is still available. Destratification has provided the option of removing water from deeper in the water column and avoiding algae.

References:

Burns (1989)

Daylesford Water Board (1989)

CASE 17

WOMBAT RESERVOIR

State:	Victoria
Controlling Authority:	Daylesford Water Board
Location:	100 km North-west of Melbourne - latitude 36°30'S
Capacity at F.S.L.:	0.6×10^6 m^3
Surface Area at F.S.L.:	-
Maximum Depth:	14.5 m
Mean Depth:	-
Catchment Area:	-
Nature of Catchment:	State Forest
Water Quality Problems:	Taste and Odour - algae blooms High Fe/Mn, colour and turbidity
Management Practices:	Destratification
Destratification History::	1985 - Current 6 hrs/cycle, 2 cycles/day

Effects of Artificial Destratification:

Wombat and Bullarto Reservoirs have had similar water quality problems. Prior to the implementation of artificial destratification good quality water was not readily available during the summer period. Summer stratification led to high levels of taste and odour producing algae in surface waters, and disagreeably high levels of iron, manganese, colour and turbidity in the deeper waters.

Since the system has been artificially aerated good quality water has been readily available throughout the summer period (Table C17.1).

Table C17.1 Selected water quality parameters at 10m depth in Wombat Reservoir during January 1989, (Daylesford Water Board 1989, Burns 1989).

Variable	Value at 10m depth
Dissolved oxygen %	88
Iron total (mg L^{-1})	0.18
soluble (mg L^{-1})	0.12
Turbidity (FTU)	3
True Colour (Pt-Co)	17

A comparison between these figures and those recorded prior to artificial aeration (Table 17.2) indicate significant improvements have occurred.

Table C17.2. Selected water quality parameters prior to artificial aeration in December 1985. (Daylesford Water Board 1989, Burns 1989).

	Depth (m)		
Variable	0	6	14
Dissolved oxygen (%)	100.0	33.0	2.0
Iron total (mg L^{-1})	0.27	0.28	1.38
Iron soluble (mg L^{-1})	0.18	0.16	0.47
Turbidity (FTU)	10.0	9.0	12.0
True colour (Pt-Co))	95.0	75.0	42.0

This comparison indicates destratification has been quite successful in alleviating the water quality difficulties present in this system.

References:

Burns (1989)

Daylesford Water Board (1989).

CASE 18

LITTLE BASS RESERVOIR

State:	Victoria
Controlling Authority:	Korumburra Water Board
Location:	100 km East of Melbourne - latitude 38°S
Capacity at F.S.L.:	0.24×10^6 m^3
Surface Area at F.S.L.:	-
Maximum Depth:	10 m
Mean Depth:	-
Catchment Area:	-
Nature of Catchment:	Agricultural, eg Dairy farms
Water Quality Problems:	Algae - Phytoplankton blooms High Fe/Mn levels
Management Practices:	Destratification Copper sulphate dosing in the past
Destratification History:	1980/81 - Current

Effects of Artificial Destratification:

Burns (1988) points out that the problems presented by this system prior to artificial destratification were, "unpleasant tastes and odours, high levels of soluble iron and manganese, algae blooms and poor bacteriological quality". To some extent this reservoir has been used as an experimental system to evaluate different aeration mechanisms. Originally the reservoir was aerated via a "manually controlled 50 L/s diesel compressor delivering air to a 12 m long aerator on the bed of the reservoir" (Burns 1988). Since this time, the system has been modified to incorporate an automatically controlled electric compressor and then a compressor powered by solar energy. Although little information is available on the performance of this system, information from the Korumburra Water Board indicates all three systems have proved useful.

To date aeration has had limited success in this water body. Prior to aeration surface water was poor in quality due to algae blooms and bottom water unusable due to the accumulation of reduced substances such as iron and manganese. Information from the Korumburra Water Board indicates aeration has been successful in addressing only one of these problems. The penetration of oxygen to the sediment-water interface during the previously stratified period has prevented the deterioration of bottom waters. The surface waters are still however often poor in quality due to high algae levels. The algae species causing difficulties in Little Bass Reservoir are predominantly green algae (including *Synura*, *Volvox* and *Spirogyra*). As discussed in the main body of this report the advantages of destratification in controlling algae levels are most often noticed in connection with blue-green species. The principles outlined in Chapter 5 "Algae Growth", may shed light on why algae still cause problems in this system.

Regardless of these algae problems, the artificial aeration of this reservoir has improved the quality of water available for supply. When algae levels do not permit the offtake of surface water, it is drawn from deeper levels which are now rich in oxygen and low in unwanted elements.

References:

Burns (1988)
Korumburra Water Board (1989)
Welsh (1984)
Rural Water Commission (1988)

CASE 19

KORUMBURRA NO 1 RESERVOIR

State:	Victoria
Controlling Authority:	Korumburra Water Board
Location:	115 km East of Melbourne - latitude 38°S
Capacity at F.S.L.:	-
Surface Area at F.S.L.:	-
Maximum Depth:	5 m
Mean Depth:	-
Catchment Area:	-
Nature of Catchment:	Agricultural, eg Dairy farms
Water Quality Problems:	Algae - Phytoplankton blooms High Fe/Mn levels
Management Practices:	Destratification Water Treatment Plant Copper sulphate dosing in the past

Destratification History:

Effects of Artificial Destratification:

Korumburra Reservoir has been destratified to alleviate problems similar to those of Little Bass Reservoir. The results of destratification have also been similar to those of Little Bass. Destratification has not removed the algae problems of this system, but has allowed previously unusable water to be taken from the bottom of this system.

References:

Korumburra Water Board (1989)

CASE 20

LANCE CREEK RESERVOIR

State:	Victoria
Controlling Authority:	Wonthaggi - Inverloch Water Board
Location:	100 km South-east of Melbourne - latitude 38°S
Capacity at F.S.L.:	1.6×10^6 m^3
Surface Area at F.S.L.:	80 ha
Maximum Depth:	10 m
Mean Depth:	5 m
Catchment Area:	20 km^2
Nature of Catchment:	Intensive agriculture
Water Quality Problems:	Taste and Odour
	Undesirable algae species in high levels
Management Practices:	Destratification
	Copper sulphate dosing

Destratification History:

Effects of Artificial Destratification:

Lance Creek Reservoir supplies water to the townships of Wonthaggi, Cape Patterson and Inverloch, all of which have experienced taste and odour problems associated with algae levels in the reservoir.

Table C20.1 displays the succession and levels of blue-green algae in Lance Creek Reservoir during 1983/84. As mentioned by Bowles and Saunders (1986) the presence of high numbers of such algae is cause for concern due to the possible toxic consequences of certain species especially *Microcystis* and *Anabaena*.

Table C20.1 The Seasonal occurrence of blue-green algae in Lance Creek Reservoir (after Bowles and Saunders 1986).

Genus	Time present	Maximum m L^{-1}	Number of days present
Anabaena spp. >100 cells m L^{-1}	Oct 1982 - Feb 1983	32 000 Mid Dec	117
	Nov - Dec 1983	2 400 Late Nov	
	Jan - Feb 1984	4 200 Early Feb	149
	Mar - May 1984	32 000 Late May	
Aphanizomenon sp. >100 filaments m L^{-1}	Nov - Jan 1982/1983	8 500 Mid Dec	51
	May 1984	3 000 Mid May	21
Oscillatoria limnetica >100 filaments m L^{-1}	Feb - May 1983	90 000 Mid Mar	90
	Dec 1983	5 600 Mid Dec	
	Jan - Feb 1984	16 000 Late Feb	64
	Mar - Apr 1984	9 000 Late Mar	
Microcystis sp. >10 colonies m L^{-1}	Dec 1982 - Mar 1983	100 Mid Jan	70

Little information is available on the exact effect of aeration on the algae population of Lance Creek. Information from the Wonthaggi - Inverloch Water Board does suggest that aeration causes a shift from blue-green algae to less concerning green algae. Excessively high numbers of green algae are then dosed with copper sulphate if required. The principles that may lead to such a species shift are outlined in Chapter 5 "Algae Growth".

References:

Bowles and Saunders (1986)

Wonthaggi - Inverloch Water Board (1989)

CASE 21

LAKE BULLEN MERRI

State:	Victoria
Controlling Authority:	Department of Conservation Forests and Lands
Location:	200 km West of Melbourne - latitude 36°30'S
Capacity at F.S.L.:	-
Surface Area at F.S.L.:	488 ha
Maximum Depth:	63.5 m
Mean Depth:	39.0 m
Catchment Area:	1000 ha
Nature of Catchment:	Cleared agricultural land
Water Quality Problems:	Low DO limiting fish production Phytoplankton blooms
Management Practices:	Destratification
Destratification History:	1982 - 1986, Manual diesel pump 1986 - Current, Automatic electric unit

Effects of Artificial Destratification:

The artificial aeration of Lake Bullen Merri has been to maintain the fishery value of this system. Prior to aeration, natural stratification and the oxygen demand created by decaying Nodularia resulted in severe oxygen stratification. This reduced the area of the lake available for fish production.

According to Conservation, Forests and Lands (1989), destratification has resulted in three main benefits in this system:

"1. Control of temperature and DO limits which provides for suitable fish habitat.

2. Production of zooplankton increase which enables supporting fish populations to be maintained...

3. Control of unsightly algae blooms. The presence of aeration and subsequent destratification has not been proven to completely remove algae blooms, however it certainly has proved to reduce algae problems in Bullen Merri."

The aeration system was initially operated by a manually run, diesel compressor. In June 1986, this was converted to an automatic electrical system. "Use of a manual control of aeration period did work, provided adequate length of aeration was allowed for. This was up to 12 hours on some occasions. However, the need to run the air compressor was often not realized until the lake had begun to stratify. Then in some cases aeration was carried out for lengths of time far in excess of the time required for destratification. In summary: the use of manual aeration was inefficient and unreliable in ensuring fish habitat was maintained in Bullen Merri. The use of automatic aeration has enabled a much more economical and efficient way of providing suitable fish habitat" (Conservation Forests and Lands 1989).

References:

Conservation Forests and Lands (1989)

CASE 22

AVON DAM

State:	New South Wales
Controlling Authority:	Sydney Water Board
Location:	25 km West of Wollongong - latitude 34°21'S
Capacity at F.S.L.:	214.4×10^6 m^3
Surface Area at F.S.L.:	1055 ha
Maximum Depth:	66 m
Mean Depth:	21.1 m
Catchment Area:	140 km^2
Nature of Catchment:	Primarily bushland
Water Quality Problems:	High colour, turbidity and iron levels
Management Practices:	Destratification
Destratification History:	1985 - 1986 Current 2 x 300 L sec^{-1} 12 hrs/day during summer period

Effects of Artificial Destratification:

Aerators were installed at 2 locations of 20m depth in Avon Lake, in an attempt at full water column destratification. Prior to the introduction of artificial aeration natural stratification developed over the spring/summer period. This typically led to values of less than 1 mg L^{-1} dissolved oxygen, 9-12m below the water surface. As a result colour, turbidity and iron levels were high at depth in the water column.

There were 3 main aims to this aeration strategy:

1) "To destratify the whole lake to a depth of 20m";
2) "To prevent dissolved oxygen depletion near the offtake";
3) "To circulate higher quality water from the downstream part of the lake to the upper reaches" (Sydney Water Board (1988).

Aeration during the summers of 1986/87 and 1987/88 has had varying degrees of success in meeting these aims.

1. Water column mixing

The two aerators did significantly disturb water column stratification. Previously a well defined thermocline was obvious, during aeration this definition did not develop. Dissolved oxygen was maintained at higher values down to the level of the aerators, while also being favourably affected through a decline in the rate of oxygen depletion. The above describes the extent to which vertical mixing occurred, complete destratification to the depth of the aerators was not achieved.

2. Prevention of DO depletion near the offtake

As mentioned in point 1, dissolved oxygen levels were elevated. "By preventing oxygen depletion the consequential elevation of iron levels was minimised. During the aeration period water could have been drawn off at any depth without jeopardising water quality in the reticulation system." (Sydney Water Board (1988). Table C22.1 presents typical levels of selected water quality variables at depth, before and during aeration

Table C22.1 Comparison of colour, turbidity and iron at site DAV7 in Avon Lake before and after artificial aeration (Sydney Water Board (1988).

	Depth m	Colour C.U	Turbidity NTU	Iron mg L^{-1}
Pre aeration (26.2.85)	12	35	3.2	1.4
(12.3.85)	12	105	5.4	3.6
During aeration (mean)	15	10	1.2	0.23
(maximum)	15	17	2.4	0.66

3. Circulate higher quality water to the upper reaches

Temperature and dissolved oxygen profiles indicate water circulation was occurring. The presence of a second aerator, in the upper reaches, does however mean that localised effects masked the observance of any horizontal circulation.

Aeration was successful in elevating dissolved oxygen levels at depth in the reservoir. This then carried through to reduced iron, colour and turbidity levels.

Aeration has been conducted during the summer of 1988/89 using only one aerator. Results were unavailable at the time of completion of this review.

The improvement in water quality in this system has led to it being preferred over Cordeaux Reservoir as the water supply for the south coastal region around Wollongong.

References:

Brown (1986)

Sydney Water Board (1988)

CASE 23

CORDEAUX DAM (UPPER)

State:	New South Wales	
Controlling Authority:	Sydney Water Board	
Location:	15 km North west of Wollongong - latitude 34°21'S	
	Main Lake	**Dam #2**
Capacity at F.S.L.:	93.6×10^6 m^3	1.18×10^6 m^3
Surface Area at F.S.L.:	780 ha	-
Maximum Depth:	52 m	11 m
Mean Depth:	12 m	-
Catchment Area:	90 km^2	8.2 km^2
Nature of Catchment:	Primarily bushland, some orchards	
Water Quality Problems:	High Fe levels High colour and turbidity	
Management Practices:	Destratification	
Destratification History:	1985/1986 1986/87 5 - 10 hours per day during summer period	

Effects of Artificial Destratification:

Cordeaux Dam provides water for the south coastal area around Wollongong, and is divided into two areas, the main lake and a second smaller dam.

Main Lake

Prior to artificial aeration discharge water quality was often poor. Mean discharge quality was characterised by colour 24, turbidity 6.3 NTU and iron levels of 1.9 mg L^{-1}.

During the first aeration period of 1985/86 an improvement was noted in these parameters. During the aeration period mean discharge values were colour 11, turbidity 2.4 NTU and iron levels of 0.22 mg L-1. The extent to which aeration caused these improvements is uncertain, it was noted that these conditions were generally better than in the previous year.

During the second aeration period in the summer of 1986/87, "there was no significant improvement in water quality after aeration commenced, in fact iron levels at 6 m tended to be slightly higher" (Sydney Water Board 1988).

Although improvements were noted with the initial aeration it is likely that the comparison has been biased through heavy rains prior to the summer of 1984/85. When compared to a long term data set it appears as if aeration has had no significant effect on water quality. Iron levels have not changed greatly since the late 1900's.

Dam #2

As with dam #1 this area typically displayed high levels of colour, turbidity iron and manganese. Typical profiles are displayed in Table 1.

Table C23.1 Typical Water Quality profiles of selected parameter from Upper Cordeaux Dam #2 (Sydney Water Board 1988)

Depth m	Colour C.U	Turbidity NTU	Iron mg L^{-1}
0	30	1.9	0.44
3	40	4.6	1.03
6	55	7.2	1.54
9	70	9.2	2.07
12	90	11.2	2.40
15	120	19.5	5.94

Following heavy rains this dam may spill into dam #1 contaminating water quality in the main reservoir.

Destratification of this storage occurred after 6 weeks of aeration and resulted in an immediate improvement in water quality.

"Colour, turbidity and iron levels were reduced to 30 to 40 C.U. 4 to 5 NTU and iron 1 to 1.2 mg L^{-1} throughout the profile. These levels were maintained until mid March" S.W.B (1988).

Further improvements were noticed after March. Figure C23.1 displays the change in certain water quality parameters as a result of destratification in this storage.

Cordeaux Dam probably has little future as a water supply storage and may soon be decommissioned, no aeration took place during the summer of 1988/89.

Avon Dam is the preferred water supply for the South Coastal region.

References

Brown (1986).

Sydney Water Board (1988).

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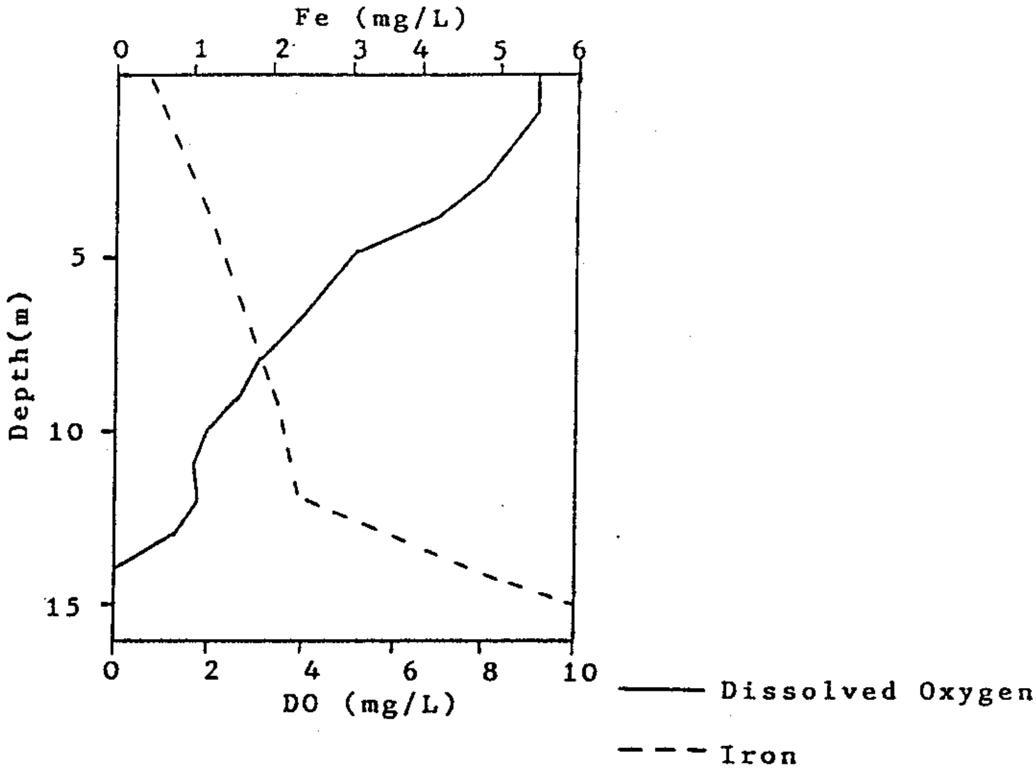

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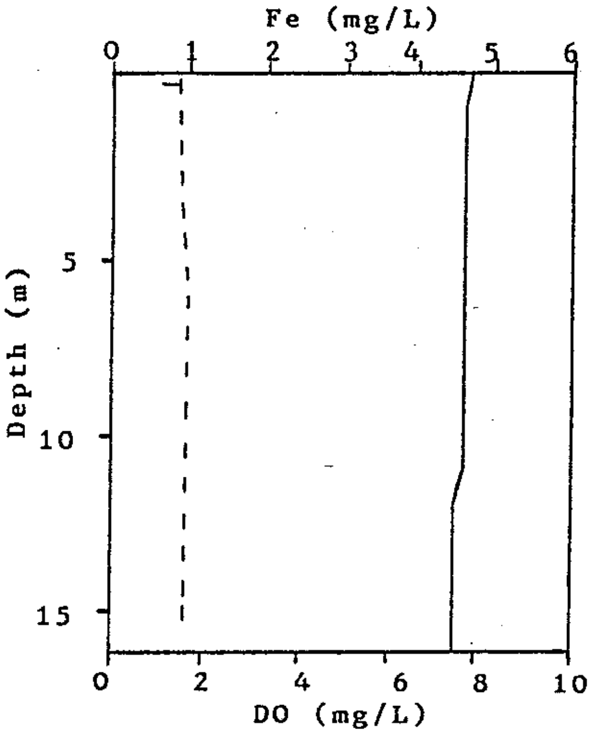

Figure C23.1 Typical DO and iron profiles from Upper Cordeaux Dam #2 when stratified and destratified, (S.W.B. 1988).

CASE 24

GREAVES CREEK RESERVOIR

State:	New South Wales
Controlling Authority:	Sydney Water Board
Location:	Near Blackheath in the Blue Mountains 110 km West of Sydney - latitude 33°30'S
Capacity at F.S.L.:	$0.31 \times 10^6 \text{ m}^3$
Surface Area at F.S.L.:	5 ha
Maximum Depth:	11.7 m
Mean Depth:	6.2 m
Catchment Area:	-
Nature of Catchment:	Primarily native forest
Water Quality Problems:	High iron, manganese and sulphide
Management Practices:	Destratification Alum. Treatment in situ

Effects of Artificial Destratification:

Greaves Creek Reservoir supplies water to the Mount Victoria, Blackheath area in the Blue Mountains of New South Wales. The system is located below Lake Medlow and has behaved in much the same manner. As with Lake Medlow total iron values at the surface are often high during aeration presumably due to aeration preventing settling, however the benefits of aeration outweigh this disadvantage.

References:

Brown (1986

Sydney Water Board (1988)

CASE 25

LAKE MEDLOW

State:	New South Wales
Controlling Authority:	Sydney Water Board
Location:	Near Blackheath in the Blue Mountains 110 km West of Sydney - latitude 33°30'S
Capacity at F.S.L.:	$0.3 \times 10^6\ m^3$
Surface Area at F.S.L.:	6 ha
Maximum Depth:	16.7 m
Mean Depth:	5 m
Catchment Area:	-
Nature of Catchment:	Bushland, some development including urban. 25% *pinus radiata* forest
Water Quality Problems:	High iron, manganese and sulphides Occasionally algae - phytoplankton High turbidity
Management Practices:	Destratification Alum. Dosing in situ
Destratification History:	1980/81 - Current 1988/89 changed to a mechanical mixer

Effects of Artificial Destratification:

As with Greaves Creek, Lower Cascade, and Woodford Creek, Lake Medlow is a Blue Mountains storage which has suffered from high iron levels.

Stratification has in the past resulted in the accumulation of high levels of iron and increased turbidity at depth. Artificial aeration for approximately 16 hours per day, in response to weekly dissolved oxygen measurements falling below 5 mg L^{-1}, has reduced this problem. Table C25.1 presents selected water quality profiles prior to and during aeration periods.

Table C25.1 Comparison of pre aeration and during aeration profiles of selected water quality parameters from Lake Medlow. (compiled from Smalls & Petrie (1983) and Sydney Water Board (1988).

	Depth	Temp	DO	Turbidity	Total
Pre aeration	(m)	(OC)	(mg L^{-1})	(NTU)	(mg L^{-1})
April 1980	0	14.0	8.1	2.1	1.01
	5	13.6	0.6	5.2	3.95
	10	8.8	0.0	17.1	11.20
January 1981	4	19.2	3.4	4.4	2.6
	8	18.6	1.8	6.6	4.2
	12	17.8	0.4	12.5	8.8
During aeration					
April 1982	0	16.9	7.7	2.5	0.88
	5	15.9	7.2	2.4	0.91
	10	15.9	6.7	1.8	0.91
	16	15.3	6.9	1.9	1.00
January 1981	0	22.0	7.7	1.3	0.66
	5	20.5	8.3	1.4	0.66
	10	20.0	6.5	1.4	0.74
	15	19.5	6.1	1.8	0.98

Clearly artificial aeration appears to have been successful in preventing high levels of iron and turbidity from accumulating in the Lower waters of Lake Medlow.

There has been one disadvantage of artificial aeration noted in this system. Surface runoff and stream input also introduce iron into this water body. In the past it was assumed that this iron "settled out" and was bound in the sediments of Lake Medlow, mixing seems to have changed this.

"Total iron levels are often higher during summer aeration periods than at other times, in surface waters. Aeration seems to be preventing the settling of ferric hydroxides or particulates with iron attached. This unexpected consequence is nevertheless outweighed by the benefits of aeration in controlling the reduced iron, manganese and hydrogen sulphide problems" (Sydney Water Board 1988).

At the moment this difficulty is kept in check through alum. dosing.

Investigation into this phenomenon is continuing. As is the evolution of destratification in Lake Medlow. During the summer of 1988/89 the artificial aeration system was removed and replaced by a mechanical propeller mixing system. The mixer is situated 1 m from the bottom of the lake, on a vertical axis and has a 1m radius span. Initial observations indicate the system may provide more efficient mixing than the aerator.

References:

Brown (1986)

Sydney Water Board (1988)

CASE 26

LOWER CASCADE RESERVOIR

State:	New South Wales
Controlling Authority:	Sydney Water Board
Location:	Near Katoomba in the Blue Mountains
	103 km West of Sydney - latitude 35°S
Capacity at F.S.L.:	0.34×10^6 m^3
Surface Area at F.S.L.:	3.4 ha
Maximum Depth:	19 m
Mean Depth:	10 m
Catchment Area:	-
Nature of Catchment:	Primarily native forest
Water Quality Problems:	High iron, manganese and sulphides
Management Practices:	Destratification
Destratification History:	1980/81 - Current

Effects of Artificial Destratification:

Lower Cascade Reservoir supplies water through a bottom offtake. Stratification in this system has led to high concentrations of iron being released. As part of the management strategy outlined in Smalls and Petrie (1983) aeration is initiated such that a volume weighted mean dissolved oxygen concentration of 5 mg L^{-} is maintained in the storage. Aeration has continued each summer since 1980/81 and is still initiated in response to falling oxygen levels.

Destratification has successfully reduced the levels of iron being discharged from this system.

Lower Cascade experiences the same phenomenon that is sometimes noted in Lake Medlow, ie aeration appearing to hold total iron in solution. Once again the benefits of destratification outweigh this disadvantage.

References:

Brown (1986)
Smalls and Petrie (1983)
Sydney Water Board (1988)

CASE 27

PROSPECT RESERVOIR

State:	New South Wales
Controlling Authority:	Sydney Water Board
Location:	30 km West of Sydney - latitude 33°40'S
Capacity at F.S.L.:	50.2×10^6 m^3
Surface Area at F.S.L.:	512 ha
Maximum Depth:	24 m
Mean Depth:	9.8 m
Catchment Area:	9.7 km^2
Nature of Catchment:	Receives from Warragamba and Upper Nepean Rivers
Water Quality Problems:	Algae - phytoplankton levels
Management Practices:	Selective Water Exchange Destratification Copper Sulphate Dosing
Destratification History:	Once only in 1976

Effects of Artificial Destratification:

Algae blooms were occurring as early as 1968 in Prospect Reservoir, when bloom levels of Asterionella and Anabaena were recorded. These blooms were controlled by copper sulphate dosing.

The progression and eventual death of any such algae bloom leads to a significant "detrital fallout". When these dead cells fall to the hypolimnion and sediment they provide a source of organic carbon. This then promotes increased respiratory activity and therefore a greater oxygen demand. The fall in dissolved oxygen level may then give rise to increased release from the sediment surface of a variety of reduced substances, including nutrients (P and N). This then may lead to future algae problems, and dirty water. This progression is discussed in detail in Chapter 6, "Releases from the Sediment"

This sequence of events was averted in Prospect Reservoir by the injection of cold, oxygen rich water from Lake Burrangorang, described by Bowen (1981), as a process of "hypolimnetic aeration by replacement". The implementation of this strategy in Prospect Reservoir has effectively raised the thermocline above the bottom of the euphotic zone. This now gives rise to algae blooms below the thermocline. This means that photosynthesis is occurring in the hypolimnion. Photosynthesis evolves oxygen, so these blooms supply oxygen to the hypolimnion, a process common in shallow waters, known as photo-oxidation. Water may then be selectively drawn from the upper layers, above the bloom.

The events of 1975/76 caused this strategy to fail. During 1975 phosphate applications in the catchment were high. This raised the phosphorus level in the reservoir by about four fold, resulting in an increase in algae "standing crop", leading to a decline in oxygen levels, via the process described earlier. The normal management strategy broke down as no suitable water for injection was available from Lake Burrangorang. A temporary aeration system was installed.

The artificial aerator created isothermal conditions and successfully restored dissolved oxygen levels to 6 mg L^{-1} at depth. This mixing did however appear to encourage growth of the diatom Melosira, but not to levels sufficient to create operational difficulties or cause consumer complaints.

This is the only time aeration has been employed in Prospect Reservoir, instead, the more economical selective withdrawal strategy has continued to be utilised.

References:

Bowen (1981)
Brown (1986)
Petrie and Smalls (1981)
Sydney Water Board (1988)
Van-Winterberg *et al* (1985.

CASE 28

WOODFORD CREEK RESERVOIR

State:	New South Wales
Controlling Authority:	Sydney Water Board
Location:	Near Hazelbrook in the Blue Mountains 86 km West of Sydney - latitude 35°S
Capacity at F.S.L.:	0.85×10^6 m^3
Surface Area at F.S.L.:	20.3 ha
Maximum Depth:	11.5 m
Mean Depth:	4.2 m
Catchment Area:	
Nature of Catchment:	Significant urgan catchment from townships of Hazelbrook and Woodford, includes possible septic tank seepage.
Water Quality Problems:	Unstable Fe, Mn and H_2S levels entering Water treatment plant
Management Practices:	Destratification Water treatment plant
Destratification History:	1980/81 - current

Effects of Artificial Destratification:

Aeration was commenced in Woodford Creek Reservoir to reduce levels of iron, manganese and hydrogen sulphide. Oxygen demand within Woodford Creek Reservoir maybe of a level sufficient to virtually strip dissolved oxygen from the water column in as little as 4-6 days. As the water is fully treated consumer complaints are rare, however variable levels of reduced compounds entering the treatment plant has caused operational difficulties.

Aeration has been successful in stabilizing the levels coming into the treatment plant.

References:

Brown (1986)

Sydney Water Board (1988)

CASE 29

DEEP CREEK RESERVOIR

State:	New South Wales
Controlling Authority:	NSW Department of Public Works
Location:	Near Eurobodalla 270 km South of Sydney - latitude $36^{o}S$
Capacity at F.S.L.:	4.5×10^6 m^3
Surface Area at F.S.L.:	53 ha
Maximum Depth:	20 m
Mean Depth:	8.5 m
Catchment Area:	
Nature of Catchment:	-
Water Quality Problems:	High colour and turbidity Low dissolved oxygen
Management Practices:	Destratification
Destratification History:	

Effects of Artificial Destratification:

As with Porters Creek Reservoir, prior to destratification reduced substances accumulated in the hypolimnion during the summer stratification period. After autumn turnover this resulted in poor quality water being released to consumers.

Destratification was implemented to accentuate the removal of colour and minimise the likelihood of consumer complaints. Destratification appears to have had a considerable favourable impact on colour in this system, with colour being reduced from levels as high as 80-90 H.U. down to a more acceptable 20-30 H.U.

It is possible the major benefit may have been obtained during the initial filling period. Given this, it is believed destratification may still be reducing colour and certainly raises oxygen levels, as a result destratification continues each summer.

References:

Brown (1986)

NSW Public Works (1988)

CASE 30

MANGROVE CREEK

State:	New South Wales
Controlling Authority:	NSW Department of Public Works/Gosford City Council
Location:	50 km North-west of Gosford - latitude 34°15'S
Capacity at F.S.L.:	190×10^6 m^3
Surface Area at F.S.L.:	700 ha
Maximum Depth:	-
Mean Depth:	24.3 m
Catchment Area:	100 km^2
Nature of Catchment:	Native Bushland
Water Quality Problems:	Taste and odour Low dissolved oxygen High colour and turbidity High iron, manganese and sulphide
Management Practices:	Destratification Water treatment plant
Destratification History:	

Effects of Artificial Destratification:

Mangrove Creek is a relatively new dam that has been filling for 6 years. The dam supplies water to Gosford, via Mooney Dam. Significant dissolved oxygen stratification, often with only minor temperature stratification, has at times led to poor discharge water quality.

To avoid this situation and to avert the costs associated with construction of a multi-level offtake facility, aeration was initiated. Testing of the destratification system has been carried out over the 1987/88 summer. Results at this stage are inconclusive, but it does not appear as if the current aeration facility will provide full lake destratification. The current system provides 300 L sec^{-1} of air through a 200 m long aeration bar.

References:

Brown (1986)

Gosford Council (1988)

NSW Public Works (1988)

CASE 31

MARDI DAM

State:	New South Wales
Controlling Authority:	Public Works Department NSW/Wyong Shire Council
Location:	Wyong on NSW central coast - latitude 33°15'S
Capacity at F.S.L.:	7.6×10^6 m^3
Surface Area at F.S.L.:	66 ha
Maximum Depth:	23 m
Mean Depth:	11 m
Catchment Area:	Off creek storage. Wyong River & Ourimbah Creek.
Nature of Catchment:	Agricultural, light and intensive
Water Quality Problems:	Taste and odour - algae/phytoplankton blooms Variable water quality High iron and manganese
Management Practices:	Destratification Water treatment plant Copper sulphate dosing
Destratification History:	1985 - Current

Effects of Artificial Destratification:

Mardi Dam has experienced consumer complaints associated with taste and odour and operational difficulties resulting from fluctuations in water quality entering the treatment plant. Natural summer stratification has caused elevated levels of turbidity, colour, iron and manganese to accumulate in the hypolimnion. At the time of winter overturn these high concentrations are distributed through the water column, creating operational difficulties for the associated treatment plant.

An aeration system was first installed in Mardi Dam during 1985 in an attempt to maintain uniform water quality throughout the year and avoid the peaks associated with winter overturn. It was also hoped that destratification would eliminate algae blooms requiring copper sulphate treatment.

Destratification has been successful in providing uniform water quality, but less so in removing algae.

Table C31.1 shows typical levels of some nuisance water quality parameters before and after destratification commenced. These figures indicate destratification has met the uniform water quality objective. Seasonal variation is now minimal.

Table C31.1 Typical levels of selected water quality parameters before and after the initiation of the destratification program in Mardi Dam.

Pre-destratification	Summer	Winter
Colour (apparent) (HU)	40.0	55.0
Turbidity (NTU)	2.5	3.5
Iron (mg L^{-1})	0.6	1.3
Manganese (mg L^{-1})	0.095	0.20
After destratification began		
Colour (apparent) (HU)	40.0	
Turbidity (NTU)	3.0	
Iron (mg L^{-1})	0.6	
Manganese (mg L^{-1})	0.085	

Destratification has not had the desired effect on algae levels, copper sulphate dosing is still required in response to algae blooms.

References:

Brown (1986)

Wyong Shire (1988)

CASE 32

PORTERS CREEK

State:	New South Wales
Controlling Authority:	NSW Public Works Department
Location:	In Shoalhaven City Council, 150 km South of Sydney - latitude - 35°30'S
Capacity at F.S.L.:	2.5×10^6 m^3
Surface Area at F.S.L.:	-
Maximum Depth:	17 m
Mean Depth:	-
Catchment Area:	-
Nature of Catchment:	-
Water Quality Problems:	Variable water quality Low dissolved oxygen Numerous consumer complaints
Management Practices:	Destratification
Destratification History:	1984/85 - Current

Effects of Artificial Destratification:

Prior to artificial aeration, natural summer stratification gave rise to low dissolved oxygen and high iron levels in the hypolimnion. These high iron levels were then released to the entire water column at the time of autumn turnover.

This situation has led to high numbers of consumer complaints from "brown staining" effects of distribution water. Destratification was initiated in the summer of 1984/85 and has continued in an attempt to eliminate this problem.

Artificial aeration has reduced the turnover effect. Iron levels do, however, still reach high maximums during summer, (1.1 mg L^{-1} during 87/88). The number of consumer complaints remains virtually unchanged. Little data is as yet available for interpretation from this system and other factors, such as change of source water during the summer period, may mask the effects of aeration for some time.

References:

NSW Public Works (1988)

CASE 33

ROCKY CREEK DAM

State:	New South Wales
Controlling Authority:	Public Works Department NSW
Location:	Near Lismore, Northern NSW - latitude 28°30'S
Capacity at F.S.L.:	13.6×10^6 m^3
Surface Area at F.S.L.:	200 ha
Maximum Depth:	19 m
Mean Depth:	6.8 m
Catchment Area:	-
Nature of Catchment:	-
Water Quality Problems:	Low dissolved oxygen High iron, manganese and sulphides Variable water quality
Management Practices:	Destratification Water treatment plant
Destratification History:	1981 - Current

Effects of Artificial Destratification:

Stratification in Rocky Creek Dam prior to artificial aeration led to low levels of dissolved oxygen and the accumulation of iron and manganese in the hypolimnion. Winter overturn then brought this iron and manganese into the upper water column and out into the reticulation system. This then gave rise to consumer complaints.

Although aeration began in June 1983 it was initially inadequate and failed to maintain high DO at depth in the water column. Mackenzie *et al* (1984) report that it was not until the autumn of 1983, following enlargement of the aeration facility that "effective destratification" was achieved. As displayed by Figures C33.1 and C33.2 this resulted in isothermal temperature and raised DO conditions.

"The results of water analysis show that the seasonal turnover and associated poor quality water has been eliminated. Further, when an adequate supply of air is continuously maintained, sufficient to destratify the storage, the water is of a quality that meets the desirable current criteria set by the Health Department" Mackenzie *et al* (1984).

Having investigated the artificial aeration of Rocky Creek Dam these authors were able to conclude:

"Seasonal thermal stratification and turnover is prevented by aeration".

"The poor quality water associated with turnover is eliminated".

"Proper destratification will provide water with a total iron concentration of around 0.3 mg L^{-1} and a colour in the order of 20 HU......"

"Irrespective of the type of treatment, aeration is necessary and should be provided as a permanent solution". Mackenzie *et al* (1984).

References:

Brown (1986)

Mackenzie *et al* (1984)

NSW Public Works (1988).

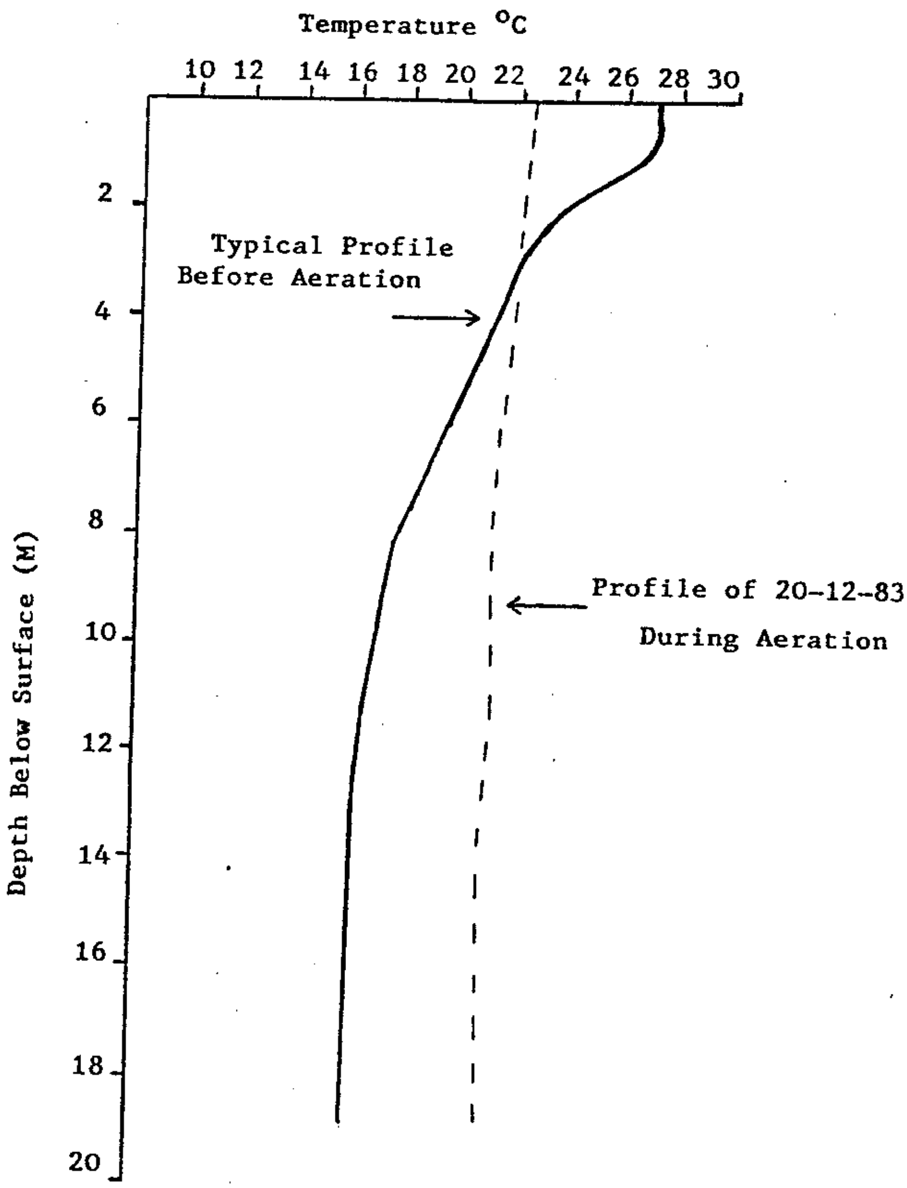

Figure C33.1 Rocky Creek Dam temperature for the month of December, before and during successful aeration (from Mackenzie *et al* (1984)

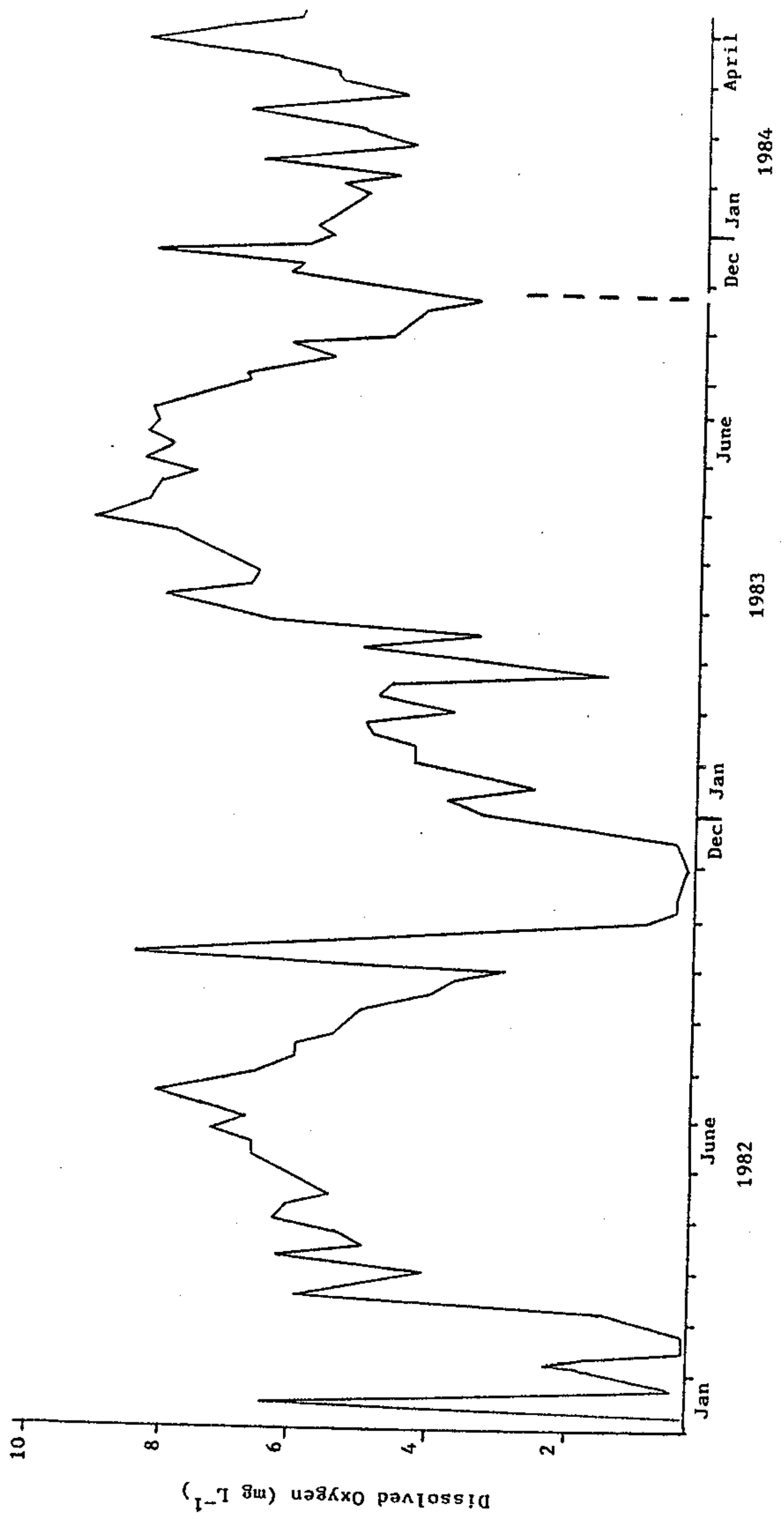

Figure C33.2 Dissolved oxygen levels at 15 m depth in Rocky Creek Dam after aeration was initiated (from Mackenzie *et al* (1984)).

CASE 34

CARCOAR DAM

State:	New South Wales
Controlling Authority:	Public Works Department NSW
Location:	50 km South-west of Bathurst - latitude 33°30'S
Capacity at F.S.L.:	36.4×10^6 m^3
Surface Area at F.S.L.:	385 ha
Maximum Depth:	41 m
Mean Depth:	9.5 m
Catchment Area:	228 km^2
Nature of Catchment:	Principally farming. Also effluent input from an abbattoir.
Water Quality Problems:	Algae - phytoplankton blooms
Management Practices:	Destratification
Destratification History:	

Effects of Artificial Destratification:

Carcoar Dam is a recreational water body and does not serve as a water supply. Blue-green algae blooms have been experienced in this system with sufficient intensity to periodically close the dam. Artificial aeration was implemented in an attempt to reduce the number and intensity of these blooms.

Initially aeration took the form of a 40 m pipe with air diffusers supplying up to 120 L sec^{-1}. This system proved inadequate and failed to either successfully destratify Carcoar or reduce algae blooms. Greater success has been achieved since the diffusers have been removed and the length of aeration bar has been doubled.

Since these alterations the system is maintained in a destratified state. Although algae blooms still occur they are greatly reduced in frequency, and considered to be "kept in check".

References:

Brown (1986)

NSW Water Resources (1988).

CASE 35

CHAFFEY DAM

State:	New South Wales
Controlling Authority:	NSW Water Resources Commission
Location:	35 km South-east of Tamworth - latitude 31°S
Capacity at F.S.L.:	62×10^6 m^3
Surface Area at F.S.L.:	542 ha
Maximum Depth:	30 m
Mean Depth:	11.4 m
Catchment Area:	420 km^2
Nature of Catchment:	Agriculture
Water Quality Problems:	Algae - phytoplankton blooms
Management Practices:	Destratification
Destratification History:	

Effects of Artificial Destratification:

Chaffey Dam "has displayed blue-green algae blooms 12 months of the year, with such algae being found all year in the upper reaches of the dam." Up until now the diffuser bars have proved ineffective. The water column is able to display isothermal conditions yet significant dissolved oxygen stratification. The intended management strategy based on the experiences of Carcoar Dam is to lengthen the aerator bar and remove the diffusers.

References:

Brown (1986)

NSW Water Resources (1988).

CASE 36

GLENBAWN DAM

State:	New South Wales
Controlling Authority:	NSW Water Resources Commission
Location:	15 km East of Scone - latitude 32°S
Capacity at F.S.L.:	720×10^6 m^3
Surface Area at F.S.L.:	2600 ha
Maximum Depth:	85 m
Mean Depth:	27.7 m
Catchment Area:	1295 km^2
Nature of Catchment:	Agriculture
Water Quality Problems:	Algae - phytoplankton blooms - taste, odour and toxicity
Management Practices:	Destratification
Destratification History:	1987/1988 - Current

Effects of Artificial Destratification:

Glenbawn Dam, like Carcoar and Windamere suffers from intense algae blooms. Algae blooms in Glenbawn are more concerning than in Windamere as the system indirectly acts as a water supply. Glenbawn discharges water into the Hunter River which supplies water to the Scone, Aberdeen and Muswellbrook area.

The aeration system installed in Glenbawn is still undergoing changes and to date has proved ineffective in controlling algae blooms. Two significant blue-green algae blooms occurred in this system in 1988, one in May and one in October. The bloom during October resulted in *Microcystis* being discharged to the river and creating taste and odour problems for downstream consumers. fortunately treatment plants were able to initiate activated carbon filtration. Although costly this strategy minimised the impact of this algae discharge on the consumer.

References:

Brown (1986)

NSW Water Resources (1988)

CASE 37

GLENNIES CREEK DAM

State:	New South Wales
Controlling Authority:	NSW Water Resources Commission
Location:	Near Singleton - latitude $32^{o}30'S$
Capacity at F.S.L.:	283×10^{6} m^{3}
Surface Area at F.S.L.:	1540 ha
Maximum Depth:	56 m
Mean Depth:	18.4 m
Catchment Area:	233 km2
Nature of Catchment:	Small Agricultural, Native bush and forest
Water Quality Problems:	Algae - phytoplankton blooms High iron and manganese levels
Management Practices:	Destratification
Destratification History:	

Effects of Artificial Destratification:

Glennies Creek Dam supplies water to the Singleton area and was initially aerated via 42 meters of aerator bar providing air at approximately 135 L sec^{-1} through bubble diffusers. This system proved inadequate. Stratification still formed during the summer period, resulting in low DO the release of iron, manganese and nutrients from sediments, and subsequent algae blooms.

Based on the experiences of Carcoar Dam it was decided to modify the aerator system. Glennies Creek is now aerated with one 200 m bar running continuously during summer at 300 L sec^{-1}, and providing air through holes drilled in the pipe rather than diffusers. This aerator is able to generate isothermal conditions, while the water column still remains stratified with regards dissolved oxygen.

As the problems of nutrient, iron and manganese release are associated with low DO, and not temperature change, they still remain as problems in Glennies Creek. The dam still experiences *Anabaena* and *Microcystis* blooms. Full results are not yet available but it appears as if the absence of a bloom during the 1987/88 summer may have been due to successful destratification for the first time.

References:

Brown (1986)

NSW Water Resources (1988)

CASE 38

SPLIT ROCK DAM

State:	New South Wales
Controlling Authority:	NSW Water Resources Commission
Location:	60 km North of Tamworth - latitude 32°15'S
Capacity at F.S.L.:	398×10^6 m^3
Surface Area at F.S.L.:	2150 ha
Maximum Depth:	52 m
Mean Depth:	18.5 m
Catchment Area:	1650 km^2
Nature of Catchment:	Agricultural
Water Quality Problems:	Algae - phytoplankton blooms
Management Practices:	Destratification being considered
Destratification History:	

Effects of Artificial Destratification:

Split Rock Dam experiences large blue-green algae blooms throughout the year. These blooms necessitate the closure of this storage throughout the summer period. There has still been no decision made as to whether or not this system will be aerated. It is likely that the future will see aeration of Split Rock along similar lines to Carcoar Dam.

References:

Brown (1986)

NSW Water Water Resources (1988)

CASE 39

WINDAMERE DAM

State:	New South Wales
Controlling Authority:	NSW Water Resources Commission
Location:	75 km North of Bathurst - latitude 32°30'S
Capacity at F.S.L.:	368×10^6 m^3
Surface Area at F.S.L.:	2030 ha
Maximum Depth:	53 m
Mean Depth:	18.1 m
Catchment Area:	1070 km^2
Nature of Catchment:	Agricultural
Water Quality Problems:	Algae - phytoplankton blooms High iron and manganese levels Low dissolved oxygen
Management Practices:	Destratification
Destratification History:	

Effects of Artificial Destratification:

Windamere Dam is still filling and at the time of the completion of this report was 85% full. It is an irrigation dam that feeds into Burrandong Dam via a lengthy river stretch. As with Glennies Creek, aeration was implemented primarily in response to algae blooms and high iron and manganese levels. Aeration in Windamere Dam has to date proved ineffective. Stratification still forms during summer resulting in nutrient, iron and manganese release from the sediment and subsequent *Anabaena* and *Microcystis* blooms at the surface. Unlike Glennies Creek the situation is not so critical. Windamere is not a water supply and has a variable offtake facility. Consequently during periods of algae blooms water is discharged from the bottom layers. Although this water is reduced and high in iron and manganese it is quickly oxidised as it is released to the river.

The aeration facility at Windamere Dam is deemed to be inadequate. It is the same as the initial system at Glennies Creek and relies on 42 m of diffused air bars. This system is currently under review. To date the problems associated with cold water releases from Dartmouth Reservoir have not occurred with Windamere. Unlike Dartmouth the discharge volumes from Windamere are small (about 200 ML per day), hence the impact of cold water discharge has not been a concern.

An example of the algae levels still able to be generated by Windamere is displayed in Table C39.1. Even though a diverse cross section is represented, the bloom of *Microcystis* dwarfs the other genera.

Table 39.1 Cells counts taken from station 1 in Windamere Dam, November 2, 1988. (Information from NSW Water Resources Commission 1988).

Genera	Cells m L^{-1}
Blue-green algae	
Microcystis	88,000
Anabaena	1,578
Green algae	
Dictyosphaerium	14,065
Chodatella	3,517
Oocystis	1,353
Golenkinia	1,308
Closterium	609
Sphaerocystis	451
Kirchneriella	406
Scenedesmus	226
Ankistrodesmus	203
Actinastrum	136
Staurastrum	113
Diatoms	
Melosira	68
Synedra	68
Navicula	23
Pyrrhophyta	
Cryptomonas	158

References:

Brown (1986)

NSW Water Resources (1988)

CASE 40

LYELL RESERVOIR

State:	New South Wales
Controlling Authority:	NSW Electricity Commission
Location:	110 km North-west of Sydney - latitude 33°30'S
Capacity at F.S.L.:	26.1×10^6 m^3
Surface Area at F.S.L.:	195 ha
Maximum Depth:	42 m
Mean Depth:	13.4 m
Catchment Area:	380 km^2
Nature of Catchment:	60% cleared farmland, 30% natural bushland
Water Quality Problems:	Low dissolved oxygen Odours - Hydrogen sulphide - algae/phytoplankton Variable Water Quality
Management Practices:	Destratification Water treatment to eliminate concentration effects of cooling tower

Effects of Artificial Destratification:

Prior to the installation of an artificial destratification system at Lyell Reservoir, anoxic conditions at times developed within 5 m below the water surface. This lead to 4 main problems.

1) Those problems experienced at the associated Wallerawang power station, as a result of having to use anoxic water.

2) Disagreeable odours due to "rotten egg" gas. Such instances may result from the release of hydrogen-sulphide (H_2S) formed in the absence of dissolved oxygen. These odours led to considerable complaint from construction workers at the Lyell Reservoir site.

3) High algae levels. Prior to aeration the stipulated limit of 50 *Microcystis* cells per mL was regularly exceeded. There was concern that such a situation could lead to a seeding of Lake Burrangorang (Sydney's major water supply).

4) High variations in water quality entering the cooling water treatment plant.

To combat these problems an artificial aeration system was installed in Lyell Reservoir. This involves a permanently placed, full automatic, feedback controlled, pneumatic destratification system. Intermittent aeration is based on utilising a thermistor chain and a submerged dissolved oxygen probe. The system is triggered by either a 5 C temperature differential or a DO level falling to 3 mg L^{-1}, the aerator then turns off once DO level has risen to 5 mg L^{-1}.

This system has been effective in combatting each of the 4 problems identified above. Dissolved oxygen is maintained at desirable levels and hydrogen sulphide is no longer detectable. The *Microcystis* limit has been exceeded on one occasion, however this was associated with a large injection of nutrients from a sewerage treatment plant, this was only a problem for a period of approximately 1 month. It is also considered that the destratification has led to considerable increases in efficiency of the treatment plant through stabilizing incoming water quality.

References:

Brown (1986)

Elcom (1988)

CASE 41

THOMSONS CREEK RESERVOIR

State: New South Wales

Controlling Body: NSW Electricity Commission

Location: 120 km North-west of Sydney - latitude 33°30'S

Development approval is currently being sought for this reservoir. Based on the Electricity Commission's mathematical simulations a destratification system will be installed as the dam is constructed. Eutrophication simulations using "DEPLE" (see Chapter 4 "Destratification Models") predict dissolved oxygen depletion well before the turnover period.

References:

Elcom (1988)

CASE 42

MALPAS RESERVOIR

State:	New South Wales
Controlling Authority:	Armidale City Council
Location:	32 km North east of Armidale - latitude 30°30'S
Capacity at F.S.L.:	12.98 x 10^6 m^3
Surface Area at F.S.L.:	1.8 km^2
Maximum Depth:	18 m
Mean Depth:	-
Catchment Area:	210 km^2
Nature of Catchment:	All agricultural
Water Quality Problems:	Algae - phytoplankton blooms - taste, odour and toxicity
Management Practices:	Destratification
Destratification History:	1988/89

Effects of Artificial Destratification:

Malpas Reservoir supplies water to the NSW city of Armidale. Blooms of *Microcystis* have been a cause of considerable concern in this reservoir during the summer period. *Microcystis* has created difficulties associated with unwelcome tastes and odours and also consumer health. Evidence of liver damage in consumers supplied from the reservoir has been attributed to toxic blooms of *Microcystis* in the source water (Falconer *et al* 1982).

Destratification has been instigated in Malpas primarily in response to *Microcystis aeruginosa*, but it is also hoped it will provide other benefits. Banens and Fisher (1987) list the expected benefits of the destratification of Malpas as follows:

. The production of near isothermal conditions

. A near oxygen saturated water column

. No longer elevated levels of Fe, Mn, P,N

. Avoidance of reduced compounds such as hydrogen sulphide gas

. A reduction in blue-green algae blooms

. A change from a blue-green dominated to a green and diatom dominated system.

. "A significant reduction in the frequency of *Microcystis* blooms".

Aeration is still being conducted on a trial basis with little success achieved through intermittent operation. Stratification has developed and *Microcystis* blooms have been present during the 1988/89 summer. It is hoped that an increase in operating time from 6 hr day^{-1} to 24 hr day^{-1}, based on the 1987/88 trials, may be more successful.

References:

Banens and Fisher (1987)

Armidale City Council (1989).

CASE 43

CHICHESTER DAM

State:	New South Wales
Controlling Authority:	Hunter District Water Board
Location:	Newcastle, central coast of NSW - latitude 33°°
Capacity at F.S.L.:	20.3×10^6 m^3
Surface Area at F.S.L.:	180 ha
Maximum Depth:	37 m
Mean Depth:	25.1 m
Catchment Area:	197 km^2
Nature of Catchment:	-
Water Quality Problems:	Variable Water Quality Low DO High turbidity, iron and manganese Algae - phytoplankton blooms High chlorine demand
Management Practices:	Destratification Water Treatment Plant
Destratification History:	1983 - Current Aerated when DO <70%

Effects of Artificial Destratification:

Chichester Dam has experienced poor water quality associated with:

Seasonal fluctuation in water quality;
High turbidity
High iron and manganese levels
Algae blooms.

Three different aeration systems have been installed in Chichester in an attempt to alleviate these problems. The current system, installed in March 1988, utilises two aerating bars each 66 m in length and two 122 l sec^{-1} compressors. The bars are 50 mm polythene pipe suspended 1 m above the sediment surface. Artificial aeration has favourably affected all of the aforementioned parameters, with the exception of turbidity.

Artificial aeration has stabilised the previously high seasonal variation in water quality. This has increased the efficiency of the associated Dungog water treatment plant.

The aeration system maintains a destratified water column. Mixing achieves fairly uniform temperatures and dissolved oxygen levels. If aeration is shut down (usually due to high turbidity) during the summer period, stratification may quickly reform. Table C43.1 displays the response of DO and temperature to the cessation of aeration after the 15/12/88 sampling occasion.

Table 43.1 Response of temperature and dissolved oxygen to the cessation of aeration after 15.12.88 in Chichester Dam.

Depth .	8.12.88		15.12.88		20.12.88	
	Temp C	DO(mg L^{-1})	Temp	DO	Temp	DO
Surface	21	6.3	22	7.1	25	9.1
2	21	6.4	22	7.2	23	9.4
4	21	6.4	22	7.3	22	6.2
6	21	6.4	22	7.4	21	5.9
8	21	6.4	22	7.3	21	6.5
10	21	6.4	22	7.3	21	6.5
12	21	6.4	22	7.3	21	6.6
15	21	5.6	22	7.3	21	6.2
20	21	6.1	22	7.2	21	5.5
25	21	6.0	22	7.3	21	5.2
Bottom	21	6.1	22	7.3	21	5.0

High turbidities experienced in this system usually occur in response to heavy rainfall in the catchment. As a result of this, aeration is discontinued temporarily. The aerators are raised, turned back on and then gradually over a period of days lowered back down the water column. Turbidity has also increased at times due to aeration. "Small increases in turbidity sometimes occur after turning the aeration system on, but this soon settles out again" (Hunter District Water Board 1988).

It has been the case with Chichester that after period of stratification and a subsequent fall in dissolved oxygen, iron and manganese levels may rise. Operation of the aeration system then reduces the levels once again.

Aeration has also been used as a tool for minimising primary productivity. "If the algae counts have been high or rising, operating the aeration system has brought them back down" (Hunter District Water Board (1988). The Hunter District Water Board estimates that destratification has reduced chlorine demand from 25 mg L^{-1} to 1.8 mg L^{-1}.

References:

Hunter District Water Board (1988)

CASE 44

SUMA PARK RESERVOIR

State:	New South Wales
Controlling Authority:	Orange City Council/Public Works Department
Location:	Orange, in country NSW - latitude 33°S
Capacity at F.S.L.:	18.2×10^6 m^3
Surface Area at F.S.L.:	166 ha
Maximum Depth:	27 m
Mean Depth:	10.9 m
Catchment Area:	-
Nature of Catchment:	-
Water Quality Problems:	Taste and Odour Algae - phytoplankton blooms Low dissolved oxygen
Management Practices:	Destratification Copper Sulphate dosing prior to aeration
Destratification History:	1979/80 - Current

Effects of Artificial Destratification:

Prior to artificial aeration water from Suma Park Reservoir often presented a taste and odour problem. It was proposed that these tastes and odours were being generated by high algae levels, which were associated with summer stratification. The initial management strategy was to treat developing algae blooms with copper sulphate, this typically required 5-6 tonnes per year. Even though Suma Park had a multi-level offtake, summer conditions made the supply of high quality water a difficult achievement. Temperature stratification could be as great as 20 C to 7 C over 27 m, with dissolved oxygen levels falling to zero. Water could not be drawn from depth due to poor water quality associated with low DO, nor could it be taken from the upper layers due to high algae levels.

Destratification was initiated and proved successful in solving both problems. Figure C44.1 displays surface algae levels before and during aeration. The normally high levels of algae have not reappeared since artificial aeration. It should however be noted that the peaks displayed in Figure C44.1 are higher than normally experienced before aeration. This was during a period of trial with the aerator and is explained by Judell (1981) "Experience between November 1978 and April 1979 has indicated that intermittent aeration following stratification of the reservoir led to increased concentrations of nitrates and phosphates in the epilimnion and blooms of blue-green algae". Effective aeration since this time has prevented the return of algae blooms in Suma Park.

Figure C44.2 presents dissolved oxygen profiles down to 20 m, one day before, and 20 days after the commencement of aeration. Aeration clearly eroded oxygen stratification and increased dissolved oxygen levels at depth in the water column.

References:

Brown (1986)

Judell (1981)

Orange City Council (1988).

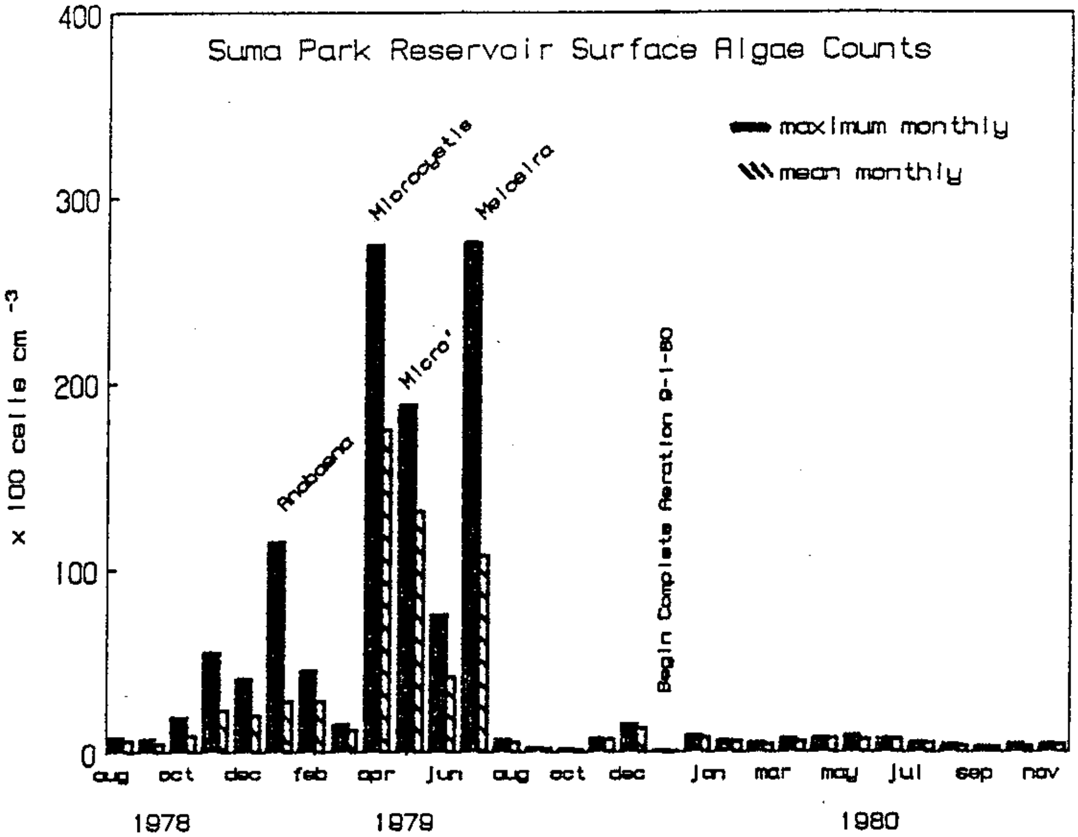

Figure C44.1 Maximum and mean monthly surface algae counts from Suma Park Reservoir.

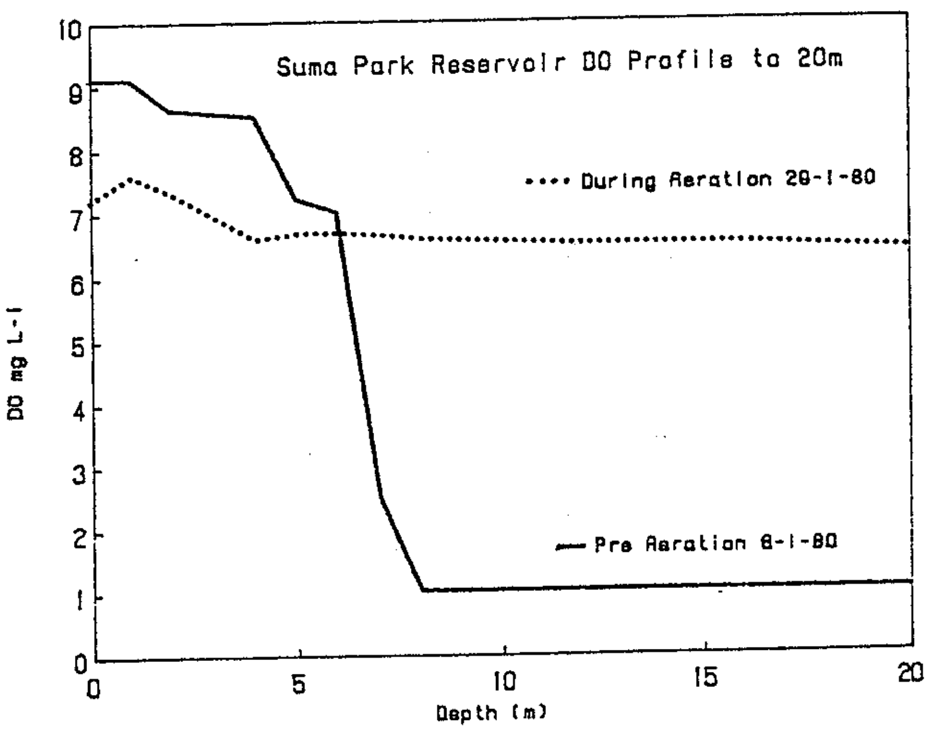

Figure C44.2 Dissolved oxygen profiles before and during aeration in Suma Park Reservoir.

CASE 45

HINZE DAM

State:	Queensland
Controlling Authority:	Gold Coast City Coundil
Location:	20 kms West of Gold Coast -latitude 28°S
Capacity at F.S.L.:	41.8×10^6 m^3
Surface Area at F.S.L.:	600 ha
Maximum Depth:	26.5 m
Mean Depth:	7 m
Catchment Area:	207 km^2
Nature of Catchment:	
Water Quality Problems:	Algae - phytoplankton blooms - strong odours High iron and manganese levels
Management Practices:	Destratification Scouring
Destratification History:	1981 - Current

Effects of Artificial Destratification:

Water is drawn from two locations in Hinze Dam. The "Lower Offtake" is situated near the dam wall, while the "Upper offtake" is located in the more shallow upper reaches of the lake.

Hinze Dam has suffered water quality difficulties associated with strong odours resulting from nuisance algae blooms and high levels of iron and manganese associated with low levels of dissolved oxygen.

Aeration has been applied in the vicinity of both offtake towers in an attempt to eliminate these difficulties.

Lower Offtake

Aeration has proved inadequate to date in this region. Brown (1986) reports that during the summer of 1985/86 aeration resulted in very slow mixing. Aeration was commenced after the onset of stratification and failed to achieve isothermal conditions until after 74 days of aeration. Oxygen levels did not begin to rise in the lower water column until after 101 days of aeration. Changes in iron and manganese levels were noted but probably reflected a slight redistribution in the water column, rather than a more beneficial reduction in release from the sediment. Nutrients may also have been redistributed to the lit zone resulting in heightened algae numbers.

Stratification still results in unfavourable water quality conditions in this region as evidenced by Table C45.1.

Table C45.1 Selected water quality parameters from lower intake site November 2 1988 (Gold Coast 1988).

Depth m	Temperature C	DO mg L^{-1}	Colour TCU	Total Fe mg L^{-1}	Total Mn mg L^{-1}
Surf	27.7	8.4	10	0.32	0.02
3	23.5	6.5	10	0.31	0.02
6	19.3	4.3	10	0.38	0.03
9	17.3	4.0	20	0.68	0.05
12	16.6	0.5	35	0.99	0.08
15	16.4	0.4	45	1.45	0.23
18	16.4	0.4	50	3.55	0.41

At this point in time *Anabaena* was the dominant alga in the system, displaying 6840 cells/ML at the surface. This gave rise to a "strong odour" around the offtake tower. The aeration system was turned on in response to this poor water quality. By December 8 little change was obvious in water chemistry but the *Anabaena* bloom had been replaced by much lower (360 cells/ML) levels of the more desirable green alga *Synedra*.

Upper Offtake

Brown (1986) reports that the effects of aeration are quickly noticed in the vicinity of the offtake tower in the upper reaches of Hinze Dam. Operation of the aerator can produce an isothermal water column within 24 hours. Iron levels have been reduced from 3.4 mg L^{-1} to 0.5 mg L^{-1} with manganese levels falling from 1.2 mg L^{-1} to 0.2 mg L^{-1}.

As with the lower offtake region, poor water quality is still evident during summer (Table C45.2).

Table C45.2 Selected water quality parameters from upper intake site October 27 1988 (Gold Coast 1988).

Depth m	Temperature C	DO mg L^{-1}	Colour TCU	Total Fe mg L^{-1}	Total Mn mg L^{-1}
Surf	27.9	5.8	30	0.51	0.08
2	26.9	5.2	30	0.52	0.10
4	26.8	4.8	15	0.59	0.10
6	20.0	0.5	15	1.00	0.25
8	18.2	0.4	15	2.59	0.60
10	17.8	0.4	60	7.30	1.19
18	16.4	0.4	50	3.55	0.41

Similar algae responses have been noted to those at the lower intake, with desirable shifts from blue-green algae to green algae.

Favourable results obtained from Little Nerang Dam have led to the lower outlet value in the main offtake tower being converted to a scour value. This should help combat the effects of sedimentation.

References:

Brown (1986)

Gold Coast (1988)

CASE 46

LITTLE NERANG CREEK DAM

State:	Queensland
Controlling Authority:	Gold Coast City Council
Location:	30 kms South-west of Gold Coast - latitude 28°S
Capacity at F.S.L.:	8.4×10^6 m^3
Surface Area at F.S.L.:	50 ha
Maximum Depth:	39 m
Mean Depth:	16.8 m
Catchment Area:	31 km^2
Nature of Catchment:	
Water Quality Problems:	Low dissolved oxygen High iron and manganese levels
Management Practices:	Destratification Scouring
Destratification History:	Original experiments 1966 - 1968 1983 - Current

Effects of Artificial Destratification:

Aeration is conducted in Little Nerang Creek Dam in an attempt to reduce iron and manganese concentrations in the water column, and to avoid the algae difficulties experienced in Hinze Dam.

Aeration has proved somewhat inadequate, having limited success in dealing with these problems.

1983/84
System destratified within two weeks. Iron and manganese levels uniform through the water column to the depth of the aerators (0.2 - 0.4 mg Fe L^{-1} and 0.01 - 0.02 mg Mn L^{-1}) (Brown 1986).

1985/86

Destratification achieved very slowly, over 47 days. Low oxygen levels and minimal impact on iron and manganese levels (Brown 1986).

1986/87

For the 1985/86 period the aerator bar was shortened from 38 m to 5.3 m length. Decreases in the effectiveness of the aerator are consistent with an under-designed system (Brown pers. comm).

The system stratified despite the efforts of the aerators. Oxygen levels fell at depth, resulting in increased concentrations of iron and manganese down through the water column. Concentrations of these elements were not as high as recorded prior to the implementation of an aeration strategy. Pre-aeration values were typically as high as iron 5 mg L^{-1}, and manganese 0.7 mg L^{-1}. Table C46.1 displays profiles for February 1987.

Table C46.1 Selected water quality parameters from Little Nerang Creek Dam February 17, 1987 (Gold Coast 1988).

Depth m	Temp C	DO mg L^{-1}	Total Fe mg L^{-1}	Total Mn mg L^{-1}
Surf	26.4	6.7	0.32	0.02
3	26.3	6.5	0.34	0.03
6	26.0	4.5	0.44	0.06
9	25.8	3.3	0.46	0.07
12	25.8	2.8	0.61	0.10
15	25.5	1.5	0.97	0.18
18	25.2	0.6	0.86	0.20

1987/88

As was the case in 1986/87, stratification developed despite aeration. It appears as if the aerators were even less effective during this summer, as unfavourable parameters rose to higher levels (Table C46.2).

Table C46.2 Selected water quality parameters from Little Nerang Creek Dam January, 1988. (Gold Coast 1988).

Depth m	Temp C	DO mg L^{-1}	Total Fe mg L^{-1}	Total Mn mg L^{-1}
Surf	25.5	2.9	0.95	0.10
3	23.1	1.8	1.16	0.12
6	22.9	2.2	1.13	0.13
9	22.7	2.0	1.29	0.13
12	22.7	2.0	1.30	0.14
15	21.9	0.7	1.37	0.14
18	21.6	0.3	1.34	0.13
21	21.0	0.2	0.85	0.14
24	20.5	0.2	1.28	0.14
27	20.5	0.1	1.28	0.13

The surprisingly low dissolved oxygen and high Fe and Mn levels at the surface indicate some mixing was occurring. Such values would not normally be obtained. Even though the aerator was creating a degree of mixing it was obviously not enough to supply oxygen from the surface to combat the summer oxygen demand. It appears as if this then led to reducing conditions developing at the sediment-water interface and an increase in water column iron and manganese levels (see Chapter 6 "Releases from the Sediment).

To date Little Nerang Creek Dam has been spared the blue-green algae problems of Hinze Dam.

Positive results in reducing iron and manganese levels appear to have been obtained through scouring. Little Nerang has the capacity to use a 40 inch scour and remove accumulated fine sediment. Further results are required, but this strategy may be effective in removing sources of oxygen demand and/or iron and manganese deposits.

References:

Brown (1986)

Gold Coast (1988)

CASE 47

ENOGGERA DAM

State:	Queensland
Controlling Authority:	Brisbane City Council
Location:	10 kms West of Brisbane - latitude 27°30'S
Capacity at F.S.L.:	4.5×10^6 m^3
Surface Area at F.S.L.:	75 ha
Maximum Depth:	14 m
Mean Depth:	6 m
Catchment Area:	33 km^2
Nature of Catchment:	
Water Quality Problems:	Variable Water Quality (Persistent colour, Fe and Mn after natural overturn)
Management Practices:	Destratification trial Water Treatment plant
Destratification History:	Trial only, during winter 1980

Effects of Artificial Destratification:

The once off artificial aeration of Enoggera Dam was to investigate the effects aeration would have on persistent colour, iron and manganese. Aeration took place after the natural overturn during July and August 1980 and had no measurable effect on the above parameters.

It is possible that aeration may have created a shift in algae dominance favouring green algae. Prior to aeration the flagellate *Synura* was the dominant alga in the water column. During aeration green algae assumed dominance and persisted for 2-3 months. Changes in algae species with mixing are discussed in Chapter 5 "Algae Growth". More data would be required to determine if mixing selected for green algae in this case.

High and persistent colour is no longer a problem in Enoggera as the water is treated, and colour solved by alum dosing.

References:

Brown (1986)

Brisbane City Council (1988)

CASE 48

LAKE MANCHESTER

State:	Queensland
Controlling Authority:	Brisbane City Council
Location:	20 kms South-west of Brisbane - latitude 27°30'S
Capacity at F.S.L.:	26×10^6 m^3
Surface Area at F.S.L.:	280 ha
Maximum Depth:	31 m
Mean Depth:	9.3 m
Catchment Area:	74 km^2
Nature of Catchment:	
Water Quality Problems:	Trial to evaluate aeration effects on Fe/Mn
Management Practices:	Destratification trial Water Treatment plant
Destratification History:	One trial April - May 1978

Effects of Artificial Destratification:

The once only aeration of Lake Manchester was an experimental investigation into the effects of aeration on iron and manganese levels. Natural turnover appears to occur at around April-May in this system, and as such the results of an aeration trial conducted at this time must be considered carefully.

It is possible that artificial aeration had the following effects. The creation of isothermal conditions, the elevation of manganese levels in surface waters and the creation of uniform iron levels throughout the water column.

References: Brisbane City Council (1988)
Brown (1986)
Solly (1981)

CASE 49

NORTH PINE DAM

State:	Queensland
Controlling Authority:	Brisbane City Council
Location:	25 kms North-west of Brisbane - latitude 27°30'S
Capacity at F.S.L.:	203×10^6 m^3
Surface Area at F.S.L.:	2000 ha
Maximum Depth:	27 m
Mean Depth:	10 m
Catchment Area:	347 km^2
Nature of Catchment:	Primarily forest and grazing land
Water Quality Problems:	Variable water quality Highly variable manganese level Algae - Phytoplankton blooms
Management Practices:	Destratification trial Water Treatment plant Copper sulphate dosing
Destratification History:	3 trial aerations

Effects of Artificial Destratification:

In the past the "internal seiche phenomenon" has made the supply of high quality water difficult for North Pine Dam. The effects of this phenomenon are dealt with in detail by Brady and Madden (1978) and are only briefly described here.

Seasonal stratification in this water body, as with many others, leads to a fall in dissolved oxygen levels down through the water column and an accumulation of reduced substances in the hypolimnion. For example the hypolimnion often accumulates high levels of iron, manganese, and phosphate.

The process of internal searching in North Pine Dam allows the layers of water established under stratified conditions to "tilt" in response to certain winds. This tilting can result in the lower, reduced water being brought into draw-off levels for short periods during a 24 hour cycle. This gives rise to variable levels of manganese entering the associated water treatment plant, which may pose considerable operational difficulties.

At times the upper water layer (epilimnion) has also supported large crops of algae in this system. This water is then made unavailable for withdrawal due to associated taste and odour problems. The coupling of these two problems, variable manganese levels and high algae crops, has often created great difficulties in providing high quality water from North Pine.

Artificial aeration trials were conducted at North Pine in an attempt to remove both problems, especially variation in manganese levels. The main observations arising from these trials were that:

- Destratification redistributed iron and manganese throughout the water column;
- Destratification was easily achieved if aeration commenced early enough to prevent the formation of strong stratification. It was however difficult to break down a well established stratification regime;
- Destratification of a well formed hypolimnion may have contributed to the formation of "larger than usual" phytoplankton blooms. This could occur through providing nutrients from the hypolimnion into the lit zone.

The aeration trials conducted at North Pine provided an interesting insight into the possible effects of destratification. Destratification has played no further role in this water body. The development of a manganese monitoring system has eliminated the operational difficulties associated with manganese variability while algae blooms can be dosed if required.

References:

Brady and Madden (1978)
Brisbane City Council (1988)
Brown (1986)
Solly (1981)

CASE 50

CALLIDE DAM

State:	Queensland
Controlling Authority:	Queensland Water Resources Commission
Location:	175 kms North-west of Brisbane - latitude 24°30'S
Capacity at F.S.L.:	57.6×10^6 m^3
Surface Area at F.S.L.:	771 ha
Maximum Depth:	27.8 m
Mean Depth:	7.5 m
Catchment Area:	518 km^2
Nature of Catchment:	
Water Quality Problems:	High levels of iron, manganese, sulphides and ammonia Low dissolved oxygen levels
Management Practices:	Destratification
Destratification History:	1985 -

Effects of Artificial Destratification:

When operational problems, (blocked diffuser holes) were removed, the aeration system in Callide Dam began to operate well.

Ross and Evans (1986) report that "after an initial experimental period, no real problems were experienced with aerator operation".

The aerator improved water quality in the dam, although unseasonal climatic conditions may have been a contributing factor.

A well oxygenated layer of good quality water was maintained to the depth of the aerator (14m) throughout most of the summer. This layer contained lower concentrations of ammonia, and total and soluble iron and manganese than during the 1981/82 study.

The quantity of poor quality water beneath the aerator was small in relation to the total stored volume and was unlikely to cause problems at overturn.

Circulation induced by the aerator warmed the bottom waters of the storage by 3-5 C above temperatures measured in earlier studies.

Variations in the depth of the thermocline occurred only slowly when the aerator was erating continuously. The depth of oxygenated good quality water did, however, decrease rapidly (by approximately 3 m per week) when the aerator was switched off.

Overall this aeration system appears to perform well.

References:

Brown (1986)

Ross & Evans (1986)

CASE 51

SOLOMON DAM

State:	Queensland
Controlling Authority:	Queensland Water Resources Commission
Location:	Palm Island, 60 km North of Townsville - latitude $18^o30'S$
Capacity at F.S.L.:	0.5×10^6 m^3
Surface Area at F.S.L.:	6.9 ha
Maximum Depth:	13.4 m
Mean Depth:	7.1 m
Catchment Area:	-
Nature of Catchment:	
Water Quality Problems:	High levels of iron and manganese Algae - phytoplankton blooms
Management Practices:	Destratification
Destratification History:	1983 - Current

Effects of Artificial Destratification:

Artificial aeration was introduced in Solomon Dam to reduce high levels of iron and manganese as well as hopefully keep troublesome levels of algae in check.

Brown (1986) and Hawkins and Griffiths (1983) report considerable success with this strategy. "Prior to aeration, maximum concentrations of 24.3 mg L^{-1} of iron and 4.1 mg L^{-1} of manganese were recorded in the hypolimnion during mid summer (Brown 1986). Aeration virtually eliminates iron and manganese from the water column, while dissolved oxygen is around 60-80% saturation.

Blue-green algae may still be found in the system, but not in the high levels previously recorded. It is possible that aeration has increased the length of time that diatom numbers are high during the year. High diatom numbers can cause considerable blocking of flow and filters.

References:

Brown (1986)

Hawkins and Griffiths (1983)

CASE 52

LAKE MORRIS

State:	Queensland
Controlling Authority:	Cairns - Mulgrave Water Supply Board
Location:	20 km North-west of Cairns latitude 16015'S
Capacity at F.S.L.:	45.5 x 106 m3
Surface Area at F.S.L.:	332 ha
Maximum Depth:	30 m
Mean Depth:	13.7 m
Catchment Area:	44 km2
Nature of Catchment:	
Water Quality Problems:	High iron and manganese levels Low dissolved oxygen
Management Practices:	Destratification Water treatment plant
Destratification History:	Trial 1978/79 Permanent 1980 - current

Effects of Artificial Destratification:

Brown (1986) reports that operation of the permanent aerator has had significant positive impacts on iron and manganese levels. Continuous aeration during the 1980/81 summer period produced a fully mixed water column and reduced iron, 12.0 mg L^{-1} to 1.8 mg L^{-1}, and manganese, 1.3 mg L^{-1} to 0.25 mg L^{-1}, levels. Since this time aeration has continued with similar positive effects, though it should be noted that in many water supplies iron levels at the lower end of this scale would still be considered undesirable.

Iron and manganese remaining in solution after aeration are reduced to very low levels by direct filtration at the Cairns Water Treatment plant. Local authorities consider the current strategy to be most successful (Brown pers comm.).

References:

Brown (1986)

Brown and Jory (1985)

Brown *et al* (1982)