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PREDICTION OF WATER QUALITY IN FLOODED OPEN CUT BROWN COAL MINES IN VICTORIA

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Glossary

Within the body of this report there are several terms that may be new to readers not familiar with the science of freshwater systems. In order to familiarise the reader with some of these terms and the concepts associated with them, this short technical glossary has been prepared as a preface.

In the first instance, terms that are used to describe the various vertical strata that can become established in a lake are described. Such strata, or stratification of a lake, occurs as a result of density gradients. Most commonly, such density gradients result from thermal changes that arise in lakes resulting from seasonal solar heating and cooling cycles. When temperature is the cause of density strata becoming established in a lake, the lake is said to be ***thermally stratified***.

Thermal stratification arises from the relatively unique properties of water. Water has a maximum density at 4°C at 1 atmosphere pressure. The relationship between water density and temperature over the range of 0°C to 25°C is shown in Figure (a) of this glossary. From the data presented in this figure, it can be seen that as the temperature increases, the rate of change of density increases, so that at warmer temperatures, a smaller rise in water temperature is needed to induce a density gradient of any specified magnitude. This is an important factor when considering solar heating of lake surfaces during the summer months.

Furthermore, as a temperature gradient is established, the rate of change of the density increases even further. Figure (b) shows the relative thermal resistance (RTS) to mixing of a thermally stratified lake resulting from such density gradients. The greatest resistance to mixing occurs in the region of the water column where the water temperature is changing most rapidly with depth. This rapidly changing region of the vertical temperature profile is called the ***thermocline***.

Once a stable thermocline is established, wind driven mixing induces a shearing of the surface water above the region of the highest RTS from the water below the RTS. The surface waters then circulate independently of the lower waters. This well mixed surface water is referred to as the ***epilimnion***. The region immediately below, corresponding to the region of the thermocline is referred to as the ***metalimnion***. Below the metalimnion is the ***hypolimnion***. The hypolimnion is isolated from the atmosphere and re-oxygenation by the epilimnion, and in productive lakes with high biological activity can become depleted in oxygen. However, thermal stratification tends to be a seasonal phenomenon, and as the surface temperatures change with the season, stratification breaks down. Thermally stratified lakes usually having at least one annual cycle of complete mixing or ***turnover***.

Another form of density stratification that can occur in lakes results from salinity gradients. Stratification on the basis of salinity often results in very stable stratification that can last for many decades or indefinite periods. Such lakes that never, or rarely mix throughout their depth are termed ***meromictic*** lakes. The salinity gradient between the less dense surface water and denser bottom water is analogous to the thermocline and is referred to as the ***chemocline***. The denser water below the chemocline is referred to as the ***monimolimnion*** and the water above the chemocline the ***mixolimnion***. As with the hypolimnion in thermally stratified lakes the monimolimnion is isolated from the atmosphere and can become depleted in oxygen (**anoxic**).

The mixolimnion itself can become thermally stratified, and can then be separated in to strata as previously, with an epilimnion, a metalimnion and a hypolimnion overlying the monimolimnion. An example of such stratification is presented in Figure (c).

Three other terms are important in broad classification of lakes. These terms refer to the biological productivity status of the lakes.

Oligotrophic lakes

- A low nutrient supply in relation to the volume of water that they contain.
- As a general rule they are deep lakes with average depths greater than 15 metres and maximum depths greater than 25 metres.
- The waters are clear with plant growth occurring at various depths of the water column rather than just near the surface.
- Oligotrophic lakes have a high concentration of dissolved oxygen in the hypolimnion.
- The main points to remember is that oligotrophic lakes are deep, transparent and with a low density of plant life in the surface waters.

Eutrophic lakes

- Have a high nutrient supply in relation to the volume of the water they contain.
- As a result of this and other factors, they tend to have high densities of planktonic green and blue-green algae growing in the surface waters.
- In extreme cases they have the appearance of a thick green pea soup.
- Mats of rooted plants and filamentous algae may also carpet the bottom in shallow water areas, depending on the competition for light between the planktonic and benthic plants.
- As a general rule, naturally eutrophic lakes are shallow with average depths less than 10 metres and maximum depths less than 15 metres.
- Dissolved oxygen tends to be depleted in the bottom waters of eutrophic lakes during periods of restricted circulation as occur at times of thermal stratification.

Mesotrophic lakes

These represent an intermediate position between the extremes demonstrated by oligotrophic and eutrophic lakes.

The terms discussed above should give the reader sufficient background to better appreciate the findings of this study on the water quality of mine lakes being formed in worked out brown coal mines in Victoria.

More detailed information on lake systems can be obtained from limnology texts such as that by Wetzel, cited as Reference No.10 in the bibliography to the main report.

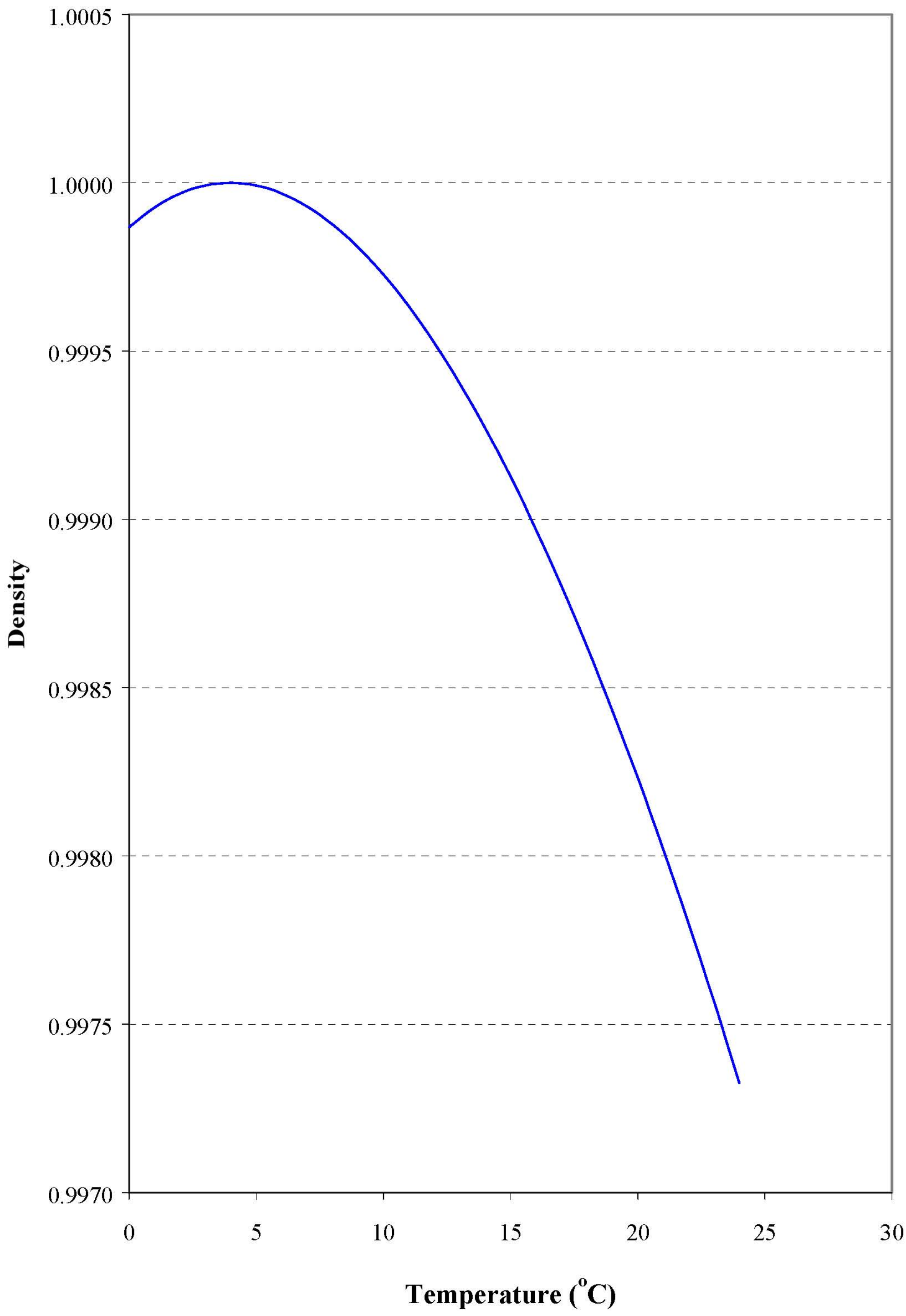

Figure (a) Change in density of water with temperature. Note that the rate of change in density increases with temperature. This means that at warmer temperatures only a small temperature difference is needed to induce a density gradient that at lower temperatures would require a larger temperature difference.

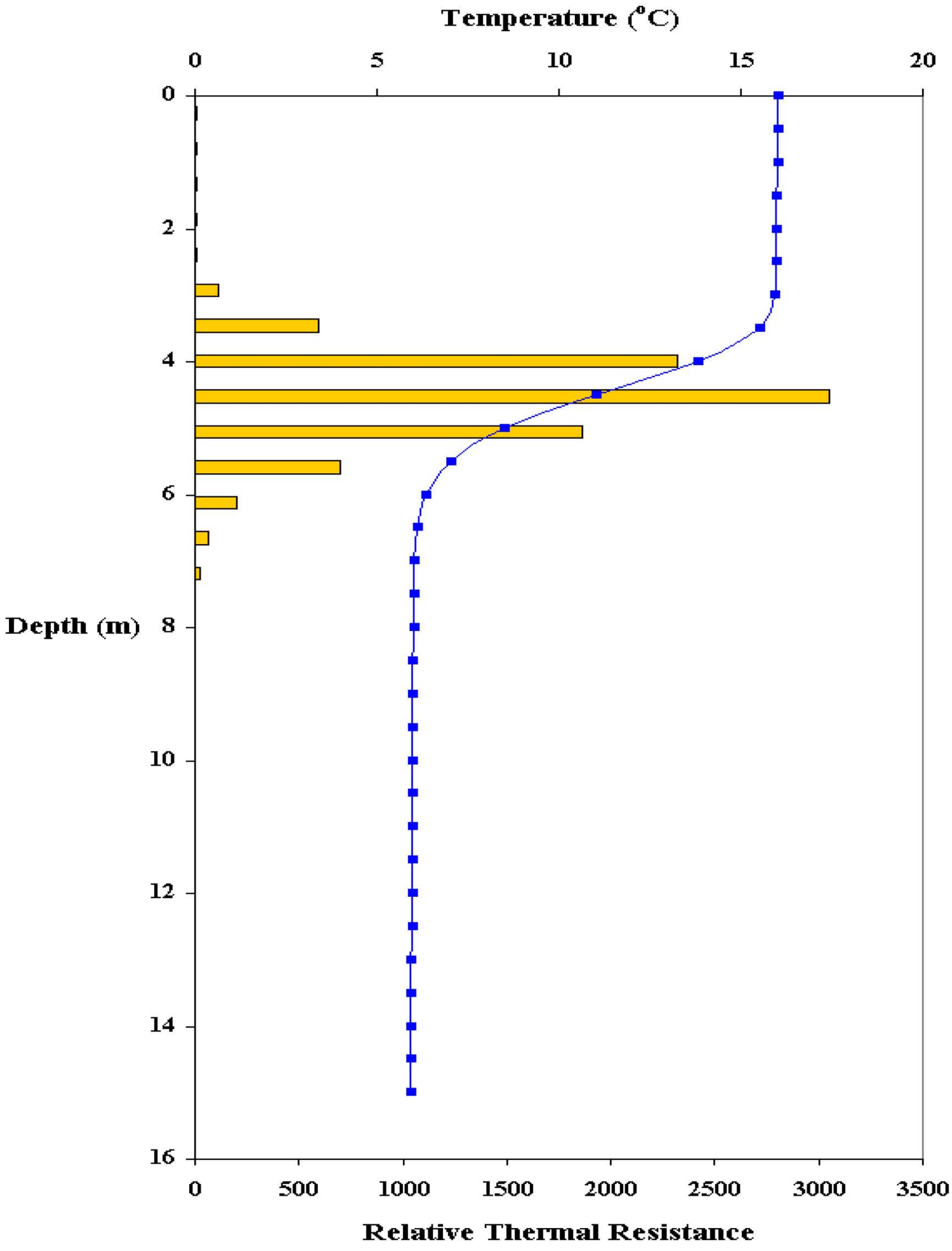

Figure (b) A simulated temperature profile showing the relative thermal resistance (RTS) to mixing throughout the whole depth of a water body as a result of temperature induced density gradients. It can be seen that at the thermocline the resistance to mixing is several orders of magnitude higher than at the surface or at the bottom, regions where no thermal gradients occur. The RTS in this instance has been calculated as the change in density per every half-metre depth relative to the change in temperature over the bottom half-metre depth.

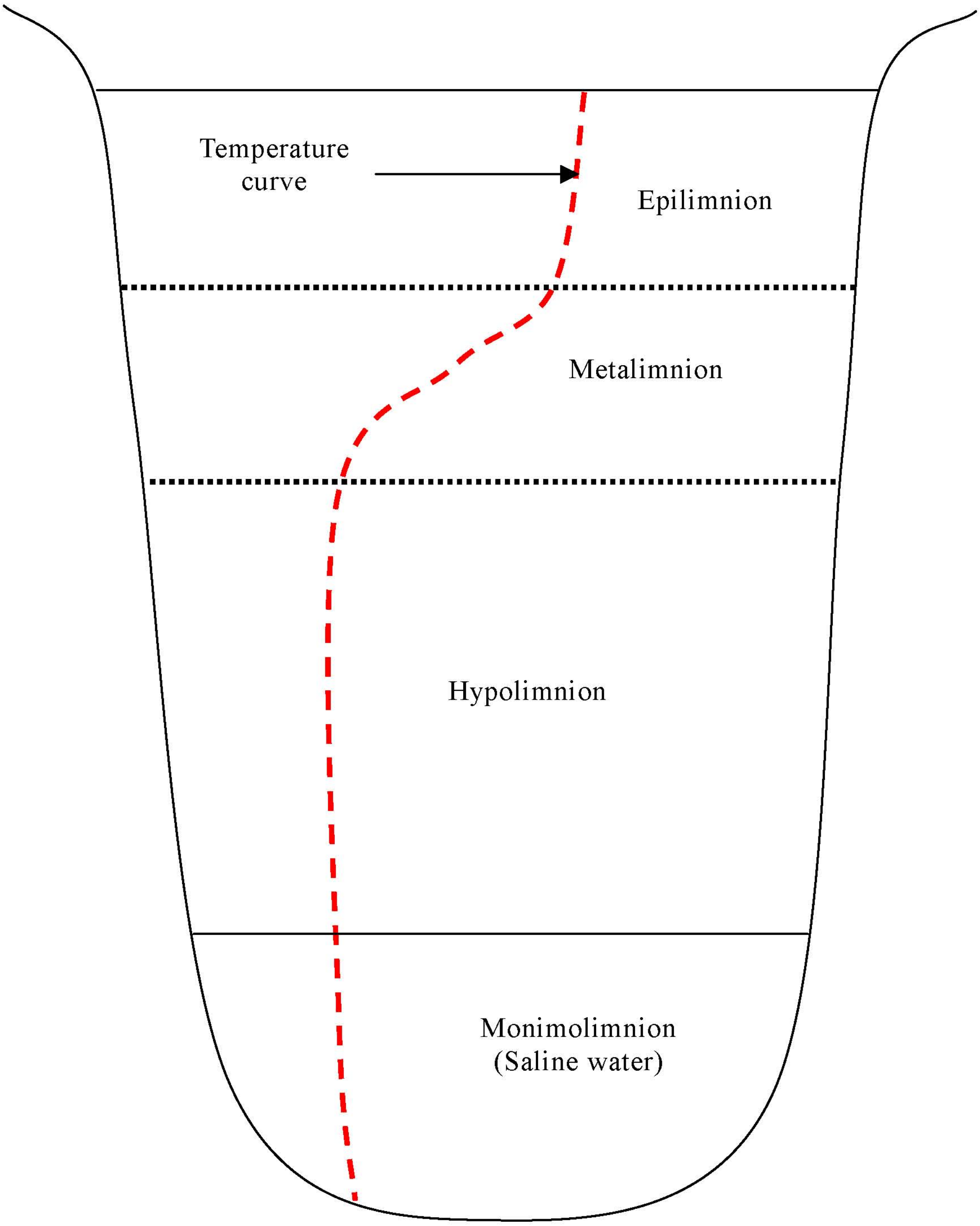

Figure (c) Schematic of a lake in section, showing the various vertical zones that may become established due to temperature and salinity induced density stratification within a lake.

Prediction of water quality in flooded open cut brown coal mines in Victoria

Summary

The issues facing the various open cut coal mine operators in evaluating the water quality of any future lakes formed in worked out mines are similar in all cases, even though the end water quality may differ from one mine to the next. These issues relate primarily to the availability of, and the quality of the water used for filling the worked out mines, and to the physical and biological processes that are involved in the cycling of chemicals in large, deep lakes.

The filling times for the different open cut mines varies with the availability of water from catchment streams. For a mine such as Yallourn, it is possible to fill the mine within a timeframe of 10 years or less without adverse impact on downstream water use requirements. For a mine such as Loy Yang, filling times are likely to be of the order of 150-175 years. Such a long filling time presents special problems that will require interim rehabilitation plans that will address issues associated with dust, fire hazard and batter stability. These issues are best addressed during the current mine development planning phase.

The overall conclusions of this study are that the lakes formed by flooding the worked out open cut mines in Victoria's Latrobe Valley are likely to be stratified primarily due to density gradients resulting from more saline water accumulating at depth, forming what are termed meromictic lakes. The water at the base of these mine lakes will become depleted of oxygen resulting in anaerobic conditions, with associated strongly reducing chemical environments prevailing. The implications for chemical cycling in the lakes of such stratification are discussed.

Nutrient levels in the source water, particularly from the Latrobe River catchment streams, are sufficiently elevated to initially suggest a possible eutrophication of the mine lakes. However, due to the long average water residence times resulting from small volume inflows of the catchment streams as compared to the large mine volumes, the nutrient depletion from the water column to the sediments of the lakes through biological cycling is likely to result in lakes of relatively high water clarity and low primary productivity. In other words, the mine lakes are likely to be of an oligotrophic status.

The issue of possible mine acidification is examined and it is concluded that within the main open cut mines in the Latrobe Valley, acidification is unlikely to be a major issue providing that good mining management practices for identification and handling of pyritic materials are followed.

1 Introduction

The open cut brown coal mining operations associated with electricity generation in Victoria will result in large pits that need rehabilitation after mining operations cease. Such pits will require to be filled with either a solid or liquid fill to a minimum depth such that the fill material will counter-act the ground water pressures from the surrounding aquifer systems, providing for batter stability.

During the operational life of the mines, such ground water pressures are overcome by pumping of the aquifers to reduce pressures acting on mine walls and floor and so maintain geotechnical stability of the mine. On closure of the mines, dewatering of aquifers for an infinite period is an impractical and uneconomic option. It is for this reason that the brown coal industry in Victoria, as part of its long-term mine development and mine rehabilitation planning, is examining the option of allowing worked out mine pits to fill with water to form mine lakes.

The flooding of pits created from worked out open cut brown coal mines in Victoria has the potential to create large lake systems that if properly managed could become major community assets. At the present time, there is little experience regarding the formation of mine lake systems in Victoria from worked out open cut brown coal mines. The only mine of any size that has been flooded in Victoria is the Yallourn North Extension Open Cut (YNEOC). The two mine lakes formed within this open cut are small compared to the lake systems that could be formed in the three major open cuts, Yallourn, Hazelwood and Loy Yang. However, the YNEOC mine lakes provide an insight into the final water quality of lakes that may be formed from filling of the other open cut mines in the Latrobe Valley.

2 Conceptual framework

The potential for creation of lake systems in worked out pits is only a feasible option where precipitation, surface drainage and ground water inflows exceed evaporative and ground water outflows. Where such inflows do not exceed the outflows, it is apparent that a lake system either cannot be formed, or if formed, the lake level will be at some depth below the mine crest level, at a point where the net inflows are exactly equal to the net outflows. This equilibrium lake surface level will oscillate around a mean annual level that is determined by the seasonal climatic conditions and seasonal water flow patterns.

With the brown coal mining operations in Victoria, the coal seams are very deep by world standards, in places being 90 metres or more in thickness. These coal seams are highly impermeable to water flow and act as barriers to flow or aquitards. Any inflow or outflow of water through these seams is associated with either cracks in the seams, which are uncommon, or during the operational life of the mines, with horizontal bores drilled into the mine walls to drain ground water from large vertical cracks in the seams behind the mine walls. In general, the potential for ground water inflow or outflow exchange with lakes formed within these open cuts is limited to saturated flows through the overlying overburden deposits and to inter-seam sandy deposits. The latter are found only at the Loy Yang mine to any significant extent.

Within Victoria, current State Government ground water policy precludes the formation of lakes with significant interconnection with ground water systems. As such, aquifer systems will need to be sealed prior to the formation of any lake. The main sources of water inputs to the lakes formed from the Latrobe Valley open cut mines will therefore be via precipitation, surface flows and saturated ground water flows from the local catchments to the mines. However, even though the State ground water policy requires sealing of all connection between aquifers and surface water after completion of mining, it is highly improbable that any sealing of existing aquifer pumping bores, or pressure seeps will be completely successful and durable in the long-term. It is therefore likely that some aquifer water will infiltrate into any mine lakes that are formed until such time that the pressure head of the lake level equals or exceeds the aquifer pressure. If the lake level pressure head exceeds any interconnected aquifer system pressure heads, then these interconnections will be a source of leakage of lake water to the aquifers.

During the first few years of the mine filling phase, while aquifer systems are still being pumped to maintain batter stability, such water can be used to help fill the lakes. After the lakes have filled, any deep aquifers that have a higher potentiometric head than that of the lake surface could be allowed to drain to the lakes from a specially established bore hole system. However, the cost of establishing and maintaining such a system of bores may not be economically attractive, or permitted under future regional ground water policies.

It is therefore likely that most of the direct water inflow to the mine lakes in the Latrobe Valley will be in the form of precipitation and evaporation over the area of the lake and the local catchment. This will result is some measure of local catchment surface and saturated ground water inflows. Additional inflows from catchments external to the local mine catchment such as by diverting existing rivers or pumping of water can be used to augment filling of the lakes. With the large pits in the Latrobe Valley, such cross catchment water transfers will be essential to fill the pits within a reasonable timeframe (decades rather than centuries), and in the case of Loy Yang, essential simply to fill the mine to top crest level.

With regard to the final water quality that can be expected within the mine lakes, two main processes will determine the chemistry. These are the chemistry of the source waters and their respective flows, and the hydrodynamic and biogenic cycling of the chemicals within the lakes themselves.

3 Water balance in mine lakes

3.1 Precipitation and evaporation

During the operational phase of an open cut mine there is a need to maintain batter stability and to provide benches for conveyor systems, road access and etc. For the Latrobe Valley brown coal mines these requirements result in an overall average slope of the mine walls of approximately 3:1 (horizontal:vertical). Such slopes in turn result in a larger mine crest area than mine floor area. Effectively this means that under the climatic conditions prevailing in the Latrobe Valley, that even in the event of no other sources of water inflow, the pits will fill to some equilibrium level. This equilibrium will be at a level when the evaporative losses are equal to the precipitation inflow from the mine area catchment above the lake level.

As an example of such a filling process, Figure 1 presents a generalised filling rate curve for a hypothetical open cut of a mine crest area of 21 km^2, mine perimeter of 19 km and a mine volume of 1.8×10^9 m^3. The precipitation and evaporation data for Loy Yang are used for this simulation.

Table 1 presents data on precipitation, lake evaporation rates, mine depths, and surface areas for the three main open cuts in the Latrobe Valley. Also presented in this table is an indication of the equilibrium water level that would likely be reached in each of the open cuts if the only mechanism of filling was through precipitation and evaporation.

From these data it is apparent that quite substantial water bodies will be formed in all of the worked out mines simply from natural precipitation over the mine area. However, large supplementary inflows will be required to allow for filling of the pits to crest height.

3.2 Ground water

In view of the fact that local catchment areas around the pits are small with relatively small unconfined ground water systems in the overburden layers, it is inevitable that significant additional sources of water will be required to supplement natural precipitation.

Existing major confined aquifer systems beneath the mines are currently being dewatered for reasons of mine stability. There is potential that some of this water could be used to aid with initial filling. For example current pumping rates at Loy Yang suggests that up to 200 L/s of such ground water could flow into the pits during initial filling. At Morwell the pumping volumes are around 800 L/s and at Yallourn East Field and Maryvale fields the pumping will be approximately 50 L/s[1]. If ground water regulations were to be amended to permit interconnection between the deep aquifers and the mine lakes via existing pumping bores, then these ground water resources could be utilised. However, as the lake levels rise, the difference between the pressure heads of the aquifers and the lake surfaces will decrease, with associated decreases in aquifer water inflows. Furthermore, in view of the fact that the piezometric heads of the main aquifer systems are below the crest level of the main mine pits, the interconnections between the aquifers and lake would become additional points of water loss from the lakes. For example, at Loy Yang, the various aquifer systems have piezometric heads of between 5 mAHD and 45 mAHD, with the lowest point on the mine crest being at approximately 55 mAHD. It is for the reason of not permitting contamination of ground water from mining sites that ground water and mining regulations require such bores to be sealed at the completion of mining operations. It is therefore unlikely that the deep aquifer systems will provide major sources of water for the mine lakes.

Another reason why the deep aquifers are unlikely to provide major sources of water is that as pumping stops and the piezometric pressures begin to return to their pre-mining state, the recharge rates of the aquifers diminishes. It has been estimated[2] that the total regional aquifer recharge is of the order of 150-200 L/s. This value is in general agreement with more recent estimates[3] that include the Balook and Boisdale aquifers with the Latrobe Valley Coal Measures aquifers. This is not a large amount and would not all be available to replenish lake water even if the piezometric pressures of the aquifers exceeded lake surface pressure heads.

3.3 Stream inflows

It is apparent from the discussion in Sections 3.1 and 3.2 that precipitation and ground water sources of water will be insufficient for filling of the lakes and that a very large portion of the inflows will need to be made up from existing surface drainage systems.

There are three main streams in the Latrobe Valley that can be used for filling the lakes. These are: -

- the Latrobe River, which flows along the northern boundary of the Yallourn Mine;
- the Morwell River, which flows along the western boundary of the Hazelwood Mine and further downstream is diverted around the eastern boundary of the Yallourn Mine just prior to its confluence with the Latrobe River; and
- Traralgon Creek, which flows past the western boundary of the Loy Yang Mine.

The average annual discharges, together with the equivalent daily discharge for each of these stream systems is shown in Table 2.

The Loy Yang Mine, which on completion will have the largest mine to be filled (Table 3), has the most limited access to surface water resources.

To examine the impact of river inflow volumes on filling times for each of the three main open cut mines in the Latrobe Valley, the data provided in Tables 1 and 3 were used in a simulation. The results of the simulations are presented in Figure 2. In all cases a ground water inflow of 50 L/s was assumed.

From the data in Figure 2 it can be seen that slight increases in inflow volumes when the volumes are small result in marked shortening in filling times. As the inflow volumes increase, there is a slowing in the rate of decrease in the duration of filling time. These data show that for Hazelwood and Loy Yang mines, trying to achieve river inflows in excess of around 200 ML/day yields little gain in shortening the filling time. For Loy Yang, data above 100 ML/day are not shown, as this rate of inflow exceeds the total average daily flow for Traralgon Creek (Table 2). Thus for Loy Yang, the shortest filling time that could be achieved is in the order of 55 years, although with a need for environmental flows within the creek, a time frame of around 150 years is more realistic for filling of this mine.

4 Salinity

The salinity of freshwaters is determined by a small number of chemical species. The major ionic species contributing to salinity in freshwaters are set out in Table 4.

Victorian freshwaters generally are dominated by sodium and chloride ions[4], whereas in many overseas countries, calcium and sulphate tend to be the dominant ions. The order of dominance of the major ionic species in Victoria varies with salinity, with the cationic order of dominance tending to be Na>Mg >Ca>K, and the anionic order of dominance being $Cl>HCO_3 + CO_3>SO_4$ in moderately saline lakes. In more saline lakes, these orders become Na>Mg>K>Ca and $Cl>SO_4>HCO_3 + CO_3$.

A scan of various bore hole water chemistry data for the three mines in the Latrobe Valley supports the general thesis presented by Williams[4]. Such data are presented in Table 5, together with data on these ionic species for the Latrobe River, the Morwell River and Traralgon Creek. Perhaps the major difference between the river waters and the various ground waters is that the river waters are less saline and the bicarbonate anion dominates the sulphate anion. More importantly however, for all of the waters, apart from the two very saline horizontal bores sampled at Loy Yang that are dominated by sodium chloride, the ratios of the various cations and anions fall within a relatively narrow band. Thus any mixture of the various sources of water will give a final water in which the cationic and anionic species contributing to salinity will be in a ratio rather similar to those presented in Table 5.

Knowing the approximate ratios of the major ionic species presents only part of the picture of the final salinity within the proposed lake systems. The filling rates and subsequent river inflows will have a large bearing on the actual salinity of the lakes in the worked out pits. If for example, we consider the simplest case, in which the water in the lakes is well mixed, then longer filling periods will result in a greater proportion of the water being evaporated off, resulting in a more saline water body. Similarly, if flow-through rates after filling are allowed to reduce to low levels, the final salinity of the water bodies will increase to an equilibrium level determined by the inflow concentration and evaporation rates across the lake surfaces. Figures 3 to 5 present data showing the required inflow rates for maintaining outflow salinities from the lakes at various total dissolved solids contents assuming complete mixing. It can be seen that for all of the mines, an ongoing inflow equivalent of approximately 30 ML/d will need to be maintained in order that the discharge from the lakes will comply with water quality requirements for the rivers downstream.

An additional issue arises with regard to salinity of the proposed lakes, and that is whether the lakes will remain well mixed, or will they stratify on the basis of salinity gradients. To investigate this, data collected from the two smaller mine lakes that have formed in the YNEOC, have been examined together with data from other parts of the globe, particularly from the brown coal regions in Germany. These results are discussed in the following sections.

5 Yallourn North Extension Open Cut

The YNEOC lake system is a small twin mine lake system formed from the natural filling of the mine. The two lakes are part of the one mine, but separated by an internal overburden dump within the mine. The northern lake has a surface area of approximately 250 hectares, while the southern lake has a surface area of approximately 7 hectares. Wet weather overflow from the northern lake flows to the southern lake via two surface drains incised through the separating overburden dump at each edge. For most of the year, the water levels in the lakes is below that of the interconnecting drains, but local drainage from this mid-field dump drains to the southern lake. Figure 6 presents a panoramic view and Figure 7 a plan view of the lake system.

The two lakes differ in several respects apart from area. The northern lake is approximately 15 m deep at it deepest point, while the southern lake is only 9 m deep. It also has an island of overburden capped coal material at one end.

The only known inflows are via local precipitation over the lake surface and local catchment, which has an area of approximately 4 km^2. Some of the catchment surface run-off from the north-western side of the open cut, an area containing rehabilitated external overburden dumps, is collected via constructed drains and channelled via the southern pond for ultimate discharge to the Latrobe River. Prior to filling of the southern lake in 1995, this channelled water discharged directly to the Latrobe River. There are no data on ground water inflows apart from the work of Deed[5] who sampled several springs in the area during 1975 and found them to be quite saline. One spring to the north of the open cut had a total dissolved solids (TDS) concentration of 5,690 mg/L, while a seep from the overburden face at the mine had a TDS of 1,850 mg/L. This overburden face TDS measured by Deed is surprisingly similar to the TDS of the water in the north western surface drains of the mid-field overburden dump that separates the southern and northern lakes at the present time. Data for the main salt species found in these lakes are presented in Tables 6 and 7.

Such high TDS levels are not uncommon where the ground water has been in intimate contact with clay materials. As an example of how readily the overburden clays can contribute to the salinity of ground water, Figure 8 presents data on two Loy Yang overburden clays, identified by their colour as a “grey” clay and the other as a “tan” clay. For each of the clays, 100 g of clay material was mixed with 200 g of demineralised water and allowed to stand in a covered beaker for the periods of time shown. These data show that water in contact with clay materials in the overburden can be expected to have salt concentrations of around 1000 mg/L or more. This is further evidenced from an examination of the TDS of the various seeps and standing waters in and around the existing open cut mines in the Latrobe Valley, where TDS levels of 1,500 mg/L are not uncommon. For example in the area of the proposed Maryvale extension of the Yallourn Open Cut, TDS levels of up to 3000 mg/L have been recorded from some of the overburden aquifer bores. In the Yallourn East Field at the coal overburden interface, the ground water has TDS levels of around 1300 mg/L[6].

5.1 Salinity stratification in the YNEOC lakes

The contribution of elevated ground water salinities to the final lake water salinity was evidenced in the data presented in Tables 6 and 7. What the tabulated data does not show is the vertical profiles of salinity in the YNEOC lakes. Figures 9 and 10 present examples of such profiles for the northern and southern ponds respectively.

From the data presented in these figures it is apparent that the salinity in both lakes is higher at depth than at the surface. There is a marked difference in the depth at which the very abrupt salinity gradient, the chemocline, is formed.

In the deeper northern pond the salinity does not change significantly until close to the bottom, around the 12 m depth, when it rapidly increases. This is in marked contrast to the smaller southern lake in which the salinity increases rapidly between 4 m and 6 m depth levels. There are two points to note from these data. Firstly both of the lakes are stratified on the basis of density gradients arising from salinity gradients, and secondly, that the southern pond has a much higher salinity.

Within the short time that the two lakes have been in existence, the northern lake filled about 1991 and the southern pond around 1995, the only possible causes of such salinity stratification are ground water interactions with surface waters.

The higher salt concentrations at the bottom of the lakes could have arisen as a result of ground water inflows either at depth or alternatively in the upper levels of the lakes and then flowing via gravity currents to the appropriate density depth. Unfortunately, a search for information on this aspect was unable to locate any documentation on ground water flows in this area, or anyone who has a former working association with the mine who could provide comment on possible ground water inflows. All that can be said with any confidence is that after abandonment of the mine, there did not appear to be any significant inflows from ground water, with the mine being considered a "dry" pit[7]. Such an observation is in accord with the low permeability of coal to water flow noted in Section 2. However, the fact that both of the lakes have salinities in excess of 700 mg/L TDS, indicates that ground water interactions are an important contributor to the water quality.

The higher salinity in the southern pond could have arisen from several causes. Firstly, the southern pond, during the operational life of the mine was used for ponding of fire services water. This water was used for spraying over the exposed coal seams to keep them moist to minimise fire risk. Any excess water from within the mine was then pumped back to the fire services pond. Such water would have leached salts from the coal, resulting in elevated salt levels being returned to the fire services pond. On filling of the lake, any such salts remaining in the fire services pond would contribute to the overall salt loading of the water.

Another possible contributing reason for the higher salt content in the southern lake is that it has a greater proportion of its surface comprised of overburden materials. As noted previously, despite its small size, this lake has an island of overburden capped coal built into it. The floor of the mine in the vicinity of the island has also been clay lined. Thus the southern lake provides a much smaller volume to clay surface ratio.

In support of the view that local water interaction with the clays is contributing to the higher overall salinity, several observations are presented. Firstly, it was already noted in Section 4 that Deed[5] had measured a TDS of 1850 mg/L in water from seeps in the overburden face of the YNEOC. Data in Tables 6 and 7 show that the drainage from the mid-field overburden dump has essentially the same TDS. Furthermore, if the data in Table 7 for the southern lake at the 8 metre depth are examined it will be seen that the salinity again is of the same magnitude. In Section 5, it was noted that wet weather overflows from the northern pond drain to the southern pond via open channels in the mid-field overburden dump, as does local drainage from this dump. It is therefore of particular interest to note that the “finger print” of several other species such as iron, silica and magnesium, is similar in both the overburden drainage and the bottom lake water. These data are very suggestive that inflows from the midfield overburden dump flow via gravity currents to the lower levels of the southern lake. Such inflows are probably indicative that additional salt has entered the bottom of the lake from the overburden capped island and the overburden sealing of the bottom of the southern lake that took place prior to filling.

Another point to note from the data in Table 7 is that the suspended solid load in the deeper water is approximately 3 times that at the surface. Such suspended solids will add to the overall density difference between the surface and bottom waters, contributing to the maintenance of the stable stratification at the relatively shallow depth of 4-5 m. There are several likely contributing reasons for the increased suspended solids at depth, but one of them again implicates overburden interactions. The idea for some interaction with clays has support from previous studies[8] into the higher turbidity and colour of the southern pond. These studies have suggested that the brown colour of the water may arise from colloidal clay materials being present in the water column.

5.2 Surface water inflows and stratification

It was noted in Section 5.1 that the southern lake in the YNEOC was stratified at a very shallow depth, and that this stratification was based upon salinity gradients. A contributing factor to the maintenance of such a salinity gradient is the surface inflow of water from the western catchment and overburden system that has been channelled to the southern lake since the lake has filled, and the wet weather overflows from the northern pond. Both these sources of water are less saline than the water in the southern lake.

Data in Figure 11 presents historical data on the TDS in the discharge from the western overburden drainage system. These data show that the TDS has dropped from an average level of around 1,300 mg/L during the early years when the mine was active, to around 500 mg/L in 1993, just prior to the diversion of this flow via the southern lake. This reduction in the salinity is attributed to the successful revegetation works that have taken place in the catchment and on the overburden dumps. There is no gauging data to estimate the inflows from this source, but it is highly probable that the surface inflow of this water, at a TDS of 500 mg/L, approximately 700-800 mg/L less than that of the surface waters of the lake, contributes to the maintenance of stratification at the relatively shallow depth. What is interesting to note, is that the difference in salinity between the top and bottom water in this lake is only around 450 mg/L, yet this results in a stable stratification of the lake. This lake provides an example of stratification that that is likely to occur when the larger open cut mines in the Latrobe Valley are filled, with less saline river water establishing a buoyant plume overlying the denser more saline waters below.

6 Impact of stratification on lake chemistry

Stratification of lakes has a significant impact upon chemical environment and thereby on chemical cycling in such lakes. This is due largely to the influence of stratification on biological activity and mixing patterns within such lakes. Before discussing some of the details it is useful to have an overall simplified picture of how lakes behave.

In many ways, a lake system can be viewed as a large biochemical reactor, filled with various inorganic and organic chemical species in both particulate and dissolved forms. The microscopic and macroscopic flora and fauna drive the chemical cycling. Such biologically driven chemical cycling in lakes arises via photosynthetically driven reduction reactions and subsequent release of such stored energy through various microbially mediated oxidations.

In large deep lakes such as will form in the pits of the open cut mines, cycling is aided by physical forces such as wind driven mixing and also very importantly, by the settling of particulate matter through gravity towards the bottom. Such particulate matter can be simple inorganic precipitates, clay particles, adsorbed inorganic and organic species on such particles, or dead cells of various planktonic plant and animal species. Microbial breakdown and utilisation of available energy sources occurs throughout the whole depth of lakes, so biological settling particulates are still part of the overall functional dynamics of lakes. However, over time, with such settling of particulate matter, some biologically important chemical species, such as phosphorus, sulphur, etc. are gradually lost to the bottom sediments.

Although there are chemical mechanisms for re-entrainment of some of the species such as phosphorus through redox driven reactions at the water-mud interface at the bottom where strongly reducing conditions can exist, overall there is net deposition of such species to the bottom sediments. In lakes such as in the YNEOC, where salinity stratification results in a permanent or long-term stable stratification not permitting the bottom most waters to mix with the overlying waters, such monimolimnia trap chemical species that have returned to solution from the bottom, and at the same time provide the conditions for burial of metal sulphides, which are generally less soluble under such conditions, in the bottom muds.

To demonstrate the influence of the interaction between biological activity and lake stratification on some of the broad physico-chemical environmental parameters it is useful to examine the data collected from the lakes in the YNEOC.

7 Oxygen levels, redox potentials, pH and temperature in the YNEOC lakes

It was previously shown in Figures 9 and 10 that there is a rapid change in salinity at depth in both the lakes resulting in salinity stratification. As a consequence of the isolation of the bottom water from that of the surface, biological activity in the bottom waters of the monimolimnion can result in oxygen depletion. This is evident in both of the YNEOC lakes as can be seen from Figures 12 and 13. The corresponding temperature curves are presented in Figures 14 and 15. These data show that in the southern lake relatively high levels of photosynthetic activity resulted in a supersaturation with oxygen at the surface of approximately 120 per cent. In the northern lake, which has a much higher water clarity, oxygen levels remained almost exactly at 100 per cent saturation throughout the whole of the mixolimnion. This would suggest that this lake is both well mixed in the mixolimnion and also that primary productivity is relatively low.

Although no instrumentation was available to measure redox potentials in the water column, in stratified lakes, below the chemocline or thermocline as the oxygen concentration drops, there is generally a reduction in redox potential associated with bacterial activity. In the case of the southern lake in the YNEOC, evidence of the strongly reducing conditions associated with the bottom was obtained from mud that adhered to the boat anchor. This was very black in colour and smelt strongly of hydrogen sulphide. The redox potential is therefore likely to be negative at the water bottom mud interface and close to zero just above the mud interface. This observation in itself indicates that the redox profiles in the YNEOC lakes are similar to that found in most other lakes, and closely follow the profiles of the oxygen depletion curves.

The temperature profiles show that in both lakes the surface water temperature was very similar at around 13.5-14^{o}C. In the northern lake the mixolimnion is well mixed throughout its depth as show with the oxygen profile. In the southern lake however, the profile is very different. Firstly the temperature of the mixolimnion below the surface is approximately 2^{o}C cooler than in the northern lake. This difference in temperature is attributed to the smaller thermal mass of the mixolimnion in the southern lake as compared to that in the northern lake, allowing more rapid seasonal cooling.

At the chemocline in the southern lake the temperature rises again to a metalimenetic maximum. This maximum is most probably the result of residual heat trapped in the monimolimnion from the previous summer's warmer temperatures. The dashed lines in Figure 15 show possible shapes of the temperature profile during summer. It should be noted that the monimolimnion temperature itself would have been slightly higher during summer than shown in Figure 15.

The biologically different processing that was occurring in the two lakes during May 1999 as evidenced by the oxygen curves is further shown from the pH profiles presented in Figures 16 and 17.

The pH curve for the southern lake is more easily explained.

The higher pH at the surface most likely results from the depletion of carbon dioxide as a result of the high primary productivity previously indicated by the supersaturated oxygen in the water at this level. Below the chemocline the lowering of the pH is suggestive of bacterial oxidation resulting in increased concentrations of carbon dioxide.

The pH curve for the northern lake shown in Figure 16 is extremely interesting. In view of the fact that the oxygen curve does not show any metalimenetic maxima corresponding to the pH rise, suggests that some form of photosynthetic chemotrophic bacterial reduction processes was occurring, resulting in a utilisation of carbon dioxide with the associated rise in pH. Below the chemocline, bacterial oxidation releases carbon dioxide, resulting in a slight decrease in pH again.

The complexity of the behaviour of these lakes on a seasonal pattern can be seen from data for the southern lake collected during late winter, in August 1999. These data are presented in Figures 18 to 20.

Figure 18 shows that the chemocline has remained at the same depth, but has sharpened markedly.

Figure 19 shows that the temperature in the mixolimnion has decreased by approximately 2°C since May and in the monimolimnion the temperature has decreased by approximately 0.75°C, with the temperature difference between the mixolimnion and monimolimnion increasing to approximately 2°C. It is at this time of year in late winter, when the mixolimnion is at its coldest relative to the monimolimnion that the density differences between the two layers would be at a minimum, and the opportunity for a turnover of the whole lake would be greatest. However, the salinity gradient on its own, disregarding the suspended solids differences between the monimolimnion and mixolimnion (Table 7) is equivalent to a thermal density gradient of approximately 4°C at these temperatures.

Observations on the Hazelwood Cooling Pondage, a large man made lake, showed that stable thermal stratification based on a gradient of 2.5°C was maintained through several days of strong wind[9]. Temperature profile measurements along a transect parallel to the wind direction showed that the thermal plume was sufficiently buoyant to resist mixing with the bottom water. All that occurred was that the thermal plume was laterally displaced to the leeward side of the pondage.

That such apparently small thermal differences can promote stable stratification is further demonstrated by the data in Figure 19 where the mixolimnion itself has become thermally stratified over approximately a 2°C range. Some evidence of the onset of such thermal stratification was present in the temperature profile taken in May. By August this thermal stratification has intensified and leads to most interesting behaviour in the biological processing within the lake.

Figure 20 shows the corresponding oxygen profile for the southern lake for August 1999. This profile shows the presence of two distinct metalimenetic maxima in oxygen concentration, one at the thermocline, the other at the chemocline. Oxygen saturation down the profile is close to 100 per cent at the maxima and drops to around 80 per cent at the minimum levels in both the mixolimnion and monimolimnion. Of particular interest is the fact that the water of the monimolimnion that was almost devoid of oxygen during the sampling in May, has been re-oxygenated. This could indicate that some mixing of the mixolimnion and monimolimnion has in fact occurred during the colder weather of winter or that photosynthetic activity at the chemocline is contributing to re-oxygenation.

Such metalimenetic oxygen maxima are quite common in stratified lakes. Wetzel[10] points out that such oxygen maxima are almost always produced by algae that develop more rapidly than they sink out into the lower depths. Furthermore, such algae are adapted to growing well under low temperature and light conditions at higher nutrient concentrations that prevail at these depths. *Ocillatoria*, a filamentous blue-green alga common to the Latrobe Valley, is often a contributor to such oxygen maxima.

Such metalimenetic oxygen maxima require stable stratification to occur, and these data provide additional evidence as to the stability of the salinity stratification in the southern lake.

This brief examination of the YNEOC lakes system tends to support the view of Hamilton-Taylor and Davison[11] who have pointed out that simple patterns of chemical behaviour in lake systems are rarely seen. This complexity arises as a result of the highly dynamic nature of lakes, where different processes can be operating on a variety of time scales such that non-steady state conditions can be regarded as the norm. These conditions are further complicated by the variety of scavenging processes such as detrital particles, phytoplankton and authigenic precipitates.

8 Nutrient status of future mine lakes in the Latrobe Valley

It was noted in Section 4 that it will be necessary to maintain an inflow of river water equivalent to an average of 30 ML/d to maintain salinities at levels appropriate for discharge to stream systems. With the Hazelwood and Yallourn mines considerably higher inflows could be sustained from the Morwell and Latrobe Rivers without significant impact on instream environmental flows. In view of the fact that most of the nutrient input will come from such inflows, it is useful to examine the likely nutrient loads that are likely to flow to the mine lakes.

Tables 8 and 9 present a summary of the historical water quality data available for these three streams from the WaterQ database. Using these data it is possible to calculate the area loading of nutrients to the future lakes for various river inflow conditions. Table 10 provides the results of one such calculation, assuming a river inflow equivalent to 100 ML/d to each of the lakes. This is a high inflow value, being roughly equivalent to total flow of Traralgon Creek (Table 2) and is much higher than the 30 ML/d flow indicated in Section 4 as an approximate minimum flow that would be required for maintaining relatively low salinity conditions in the lakes. It has been used as an extreme scenario to examine possible impacts of the nutrient levels in the streams on trophic status of the lakes. As comparative data, Table 11 presents permissible and danger levels for loadings these nutrients to lakes on an areal and average depth basis. From these data it can be seen that the nutrient loadings into all three proposed mine lakes in the Latrobe Valley, even at the high rates of river inflow used to generate the data in Table 10, fall into the permissible range of nutrient loadings. It is therefore highly likely that the open cut mine lakes in the Latrobe Valley will be relatively unproductive. The northern lake at the YNEOC is probably an indication of what the overall trophic status is likely to be.

9 Acid drainage - is it an issue in Victoria's brown coal fields?

9.1 Acid drainage

When discussing acid drainage two terms are often used interchangeably, acid mine drainage (AMD) and acid rock drainage (ARD). These two terms refer to the same process, that of oxidation of sulphide minerals to produce sulphuric acid. The difference in terms simply stems from whether the acid drainage arises within the mine from the oxidation of sulphides in the mine walls, or whether the acid drainage arises from mine spoil materials that have been excavated and dumped. In this report, since it deals with water quality in mine lakes, the term AMD, or simply acid drainage (AD) will be used when dealing with acid generation from pyrite oxidation, whether from mine walls or overburden dumps.

As noted, AD arises from the oxidation of metal sulphide deposits to produce sulphuric acid. The most common of the metal sulphides, predominantly due to the relative abundance of iron, are the several forms of iron sulphide, marcasite and pyrite. These forms differ only in crystal structure and will be referred to hereafter as iron pyrite, or simply pyrite. Other sulphides that are relatively common in some of the hard rock mining areas such as arsenopyrite and pyrrohotite, can be at levels of over 50 per cent (w/w basis) in some of Victorian gold mining regions[12]. These are not discussed in this report, mainly for the reason that the Latrobe Valley sulphide minerals have previously been identified as being predominantly in the form of marcasite and pyrite[13].

Formation of iron sulphide deposits arises under strongly reducing conditions. In eastern Australian coastal regions pyrite deposits are often associated with areas that have previously been inundated with marine waters and in which bacterial oxidation of organic matter was accomplished via reduction of sulphur[14]. At the Loy Yang mine for instance, there is a band of pyritic material at the top of the interseam that was laid down during marine intrusions during the mid-Tertiary[15]. This interseam is full of non-calcareous fossils of marine foraminifera and dinoflagellates.

In addition to the laying down of pyritic deposits during marine inundation, bacterial activity can also result in the incorporation of higher levels of sulphur within the organic fabric of the coal itself. Again at the Loy Yang Mine for example, there is evidence of such organic sulphur deposition where sulphur content of the coal can reach up to 2 per cent on a dry coal basis in a very narrow band of coal at the top of the M1B coal seam[15].

It can be seen that the mechanisms for deposition of sulphur and more importantly pyrite in the Victorian coalfields is complex, and can be further complicated by dissolution and re-precipitation reactions over geological time. Understanding the basic redox chemistry of iron and sulphur in aqueous systems and possible deposition locations in the mining areas is important when considering AD issues.

In Victoria, the coals are generally of low sulphur content, most of which is organically bound. In the Latrobe Valley pyritic sulphur content of the coals is generally significantly lower than 0.1 per cent on a dry coal basis, with only the

YNEOC coals showing higher levels, up to 0.5 per cent. Anglesea coals, like the YNEOC coal has higher levels of sulphur (Table 12). Less data are available on overburden materials, but as noted, pyrite, if present in overburden in coal areas, it is likely to be concentrated in a band overlying the coal, or at the top of interseam sands deposited during past marine intrusions.

Within the Latrobe Valley coalfields, no extensive pyrite layers have been found. However, recently a narrow band of approximately 1-2 cm in depth was identified as being present in an area of current mining operations at the Yallourn East Field. In addition, some discrete nodules of pyrite up to 5 cm in diameter were identified in the metre or so of overburden above this pyrite layer[16]. These observations are similar to those of Murray[13] who reported that pyritic deposits in the Yallourn area occurred sporadically and infrequently as isolated deposits associated with wood, or as nodules, with the nodules being found on the coal surface. Such "hot spots" of pyrite have been encountered in other areas in the Latrobe Valley, but there are no known extensive pyrite deposits.

As a result of such intermittent pyrite deposits, some of the overburden dumps show evidence of pyrite oxidation. For example, the dumping of overburden from the Yallourn East Field from the area of the narrow pyrite band is resulting in the generation of a small volume of acidic water of a pH of around 3.5[16].

At Loy Yang there is also some minor acid seepage at the overburden dump. Recent column leaching work[17] using interseam material with a pyrite level of 2.6 per cent on a dry weight basis, taken from the interseam area noted previously, resulted in a drop in pH of the samples from a pH of 3.15 to approximately 2.0 over a period of 10 weeks. Such results suggest a potential for significant AD problems from the overburden dump where such materials are dumped and within the mine itself if any significant volumes of water leach through such material. However, the extent of the seepage in both cases is relatively small. On the overburden dump evidence of acid drainage tends to be in the form of a wet stain in one section of the dump resulting in very localised revegetation problems, without actually resulting in any run-off of acid water of particularly low pH. The pH of the run-off from the Loy Yang overburden dump system as a whole is in the order of pH 5-6. Field peroxide tests[18] on two Loy Yang overburden clays showed very little evidence of unoxidised pyrite. Field pHs of around 5.2 and 5.0 dropped to 4.5 and 4.2 respectively after 0.75 hours of peroxide treatment. There was no change in colour, vigorous effervescence or sulphurous odour evident.

Substantial exchangeable/soluble aluminium and hydrogen ions exist at these pHs[18] and it is likely that much of the acidity associated with the overburden dumps is as a result of various aluminium minerals present in the clays.

However, it is of interest to compare the seasonal change in the pH of the overburden dump run-off. Figure 21 presents such data for 1997-99 and it can be seen that the pH decreases during the wetter months of the year (Figure 22). Such cyclic changes in pH may be indicating low level oxidation of some pyrite within the dump, with oxidation occurring during the drier months and acid flushing during the wetter months. Such a seasonal pattern in stream pH occurs naturally in Marshy Creek at Anglesea[19]. This open cut mine is in an area previously identified by Deed[5] as being in an area of high pyrite deposition, and the seasonal changes in stream pH are believed to result from shallow pyrite deposits in the upper catchment[19].

In any area where pyrite is present, the overall acid-base and redox chemistry of the dump requires elucidation, as does the ultimate fate and chemistry of any acidic run-off in assessing any potential environmental risks. In the case of the Loy Yang overburden dump, the run-off joins the power station low quality water discharge, which is predominantly made up of cooling tower purge water. The final pH of this combined flow averages around pH 7.5 and no current environmental issues arise from the run-off from the overburden dump. Surprisingly, this small creek that drains the overburden dump has a macrobenthic fauna that is richer than Traralgon Creek into which the combined discharge flows[20].

9.2 Ground water acid-base and redox chemistry

Within the context of impacts on the final water quality of the mine lakes that may be formed at some time in the future, there are three main issues to be considered in relation to AMD. These are the sources of AD, the potential for in lake acidification of the water, and if acidification does occur the impact on water quality.

In Section 9.1 it was noted that AD is associated with pyrite deposits and that apart from some isolated areas, most of the Latrobe Valley coalfield areas are relatively free of such deposits. Extensive recent investigations specifically designed to identify potential AD problems in the Maryvale area of Yallourn concluded that the "chemistry of the water within the overburden dumps (Midfield and Township) and existing Morwell River diversion suggests that little sulphide material was deposited in these areas."[16]

An interesting aspect to the Maryvale study was the suggestion that a lot of the acidic ground water in the region could arise from non-pyritic oxidation of iron. This suggestion stemmed from the highly reducing ground water conditions, with Eh readings in the range of - 190 mV to 350 mV being recorded. The associated elevated carbonate levels suggested to the authors that carbon dioxide and possibly methane gas evolved from the underlying coal was responsible for the generally highly reduced soil conditions. In addition to the elevated bicarbonate levels the authors also found elevated levels of ferrous iron, which they suggested, through oxidation to ferric oxyhydroxides would generate acidity.

If indeed significant amounts of coal gases are being liberated as suggested by the authors, such evolution of gases to the soil would mitigate against oxidation of any pyritic deposits that may exist.

From the available data it is apparent that in the Latrobe Valley acid generation via oxidation of pyritic materials is unlikely to result in significant AD. Furthermore, the local catchments for the mines from which acidity could originate are small, and ground water flows that could deliver such acidity are also expected to be small. From the perspective of impacting on mine lake water chemistry it is worth examining some possible scenarios for acidification.

9.3 Water sources and lake water pH

When examining the likely water quality of the future brown coal mine lakes in Victoria, the likely pH is perhaps the most important parameter to predict. This is mainly because pH is a "master variable" in water chemistry. Together with the electron activity, measured as a Redox potential, it sets the environment within which the majority of the chemical reactions occur. The prediction of pH is particularly pertinent in relation to mine lake water quality as mine drainage water acidification is a well know feature of many mining operations.

Twenty five years ago Deed[5] carried out an examination of ground waters, overburden and coal types, for evidence of local pyrite deposits, associated with the various open cut mines in Victoria. His conclusions were that of the then currently operating mines, there were two that were likely to contain acid waters if filled as lakes. These were the Anglesea Mine in southern Victoria, and the YNEOC Mine. To date only the YNEOC has filled, and as shown by the data in Sections 5 t0 7 of this report, this mine has a pH that is in the range of 6 to 7.

There are a number of possible reasons why the predictions of acidification of the YNEOC did not eventuate. These reasons all relate to inflow water quality. Deed had carried out his estimates based upon the assumption that much of the inflow to the mine would come from ground waters and overburden drainage from which he had measured pHs of 3.7 to 4.8. Coincidentally, he did suggest that rehabilitation of the overburden and other exposed areas would likely retard pyrite oxidation rates and improve the chances of better water quality. Such rehabilitation has been carried out, apparently with considerable success. An example of the effectiveness of the rehabilitation works is presented in Figure 23, which shows the long-term changes in the pH of the water draining the catchment containing the major overburden dumps from the mine has increased from the low ranges measured by Deed during the mid 1970s to around 6.5 by the mid 1980s, remaining at those levels to the mid 1990s. More recent data were not available at the time of preparing this report, but the data in Figure 23 provide a clear indication that the catchment run-off has been the key determinant of lake pH. This is supported by information previously presented in Section 5.1 indicating that the YNEOC was a "dry" pit, that is, there was little ground water inflow during the operational life and after mine closure[7]. This is somewhat different to many of the now closed lignite pits in countries such as Germany, where lake water acidification is associated with the fact that acidic ground water inflows are the main source of water for filling the mines[21].

One of the mechanisms by which the rehabilitation works at YNEOC are contributing to preventing lake acidification, but which may not have been planned from the point of view of lake water quality, is that extensive lime application to the exposed areas took place to improve soil pHs for plant growth. Such liming has effectively increased the acid neutralising capacity of the soil, so that water percolating through the soil to any sources of pyrite oxidation is neutralising any acid being generated.

Evidence that such a mechanism may be operative is that during the course of this present study, when field soil and soil water pH tests were to be carried out, all free standing pooled water in drains and puddles in the YNEOC lakes' western and northern catchment areas had pHs in the range of 7.2 to 8.1. Such lime application as a preventative measure is likely to be less expensive and more effective than a liming application to treat AMD water. The reason for this is that lime treatment of AMD water often results in the precipitation of ferric oxyhydroxides on to the surface of the lime, decreasing the available surface area for further chemical neutralisation reactions.

The importance of acid neutralising capacity can not be over emphasised. It is a standard measure when assessing mine soil and rock for generating AD. For example, in the Yallourn Mine area, the clays appear to have a Net Acid Producing Potential (NAPP) of around -10 kg/tonne and the coals up to -20 kg/tonne[16]. Although such NAPP levels are in what is termed the "equivocal range", requiring further assessment, the clays are not highly acid producing as such.

However, as was pointed out in Section 9.1, there are some pyrite "hot spots" and it is most important that such areas be well managed to avoid significant problems with future lake water quality. As an example of the effects of AMD on lake water pH, a simulation using the chemical equilibrium package MINEQL+[22] was used to evaluate the inflow of highly acidic water from such a pyrite hot spot at Yallourn when mixed with Latrobe River water. The characteristics of the AMD and Latrobe River waters used in this simulation are shown in Table 13. The output of the simulation is shown in Figure 24. Also shown in Figure 24 are the simulations of mixing Traralgon Creek and Morwell River waters with such AD. Both these streams have much higher alkalinity than the Latrobe River water and this shows up with a higher capacity for neutralising the AD.

It will be seen from the data in Table 13 that the AMD water is very acidic, with a pH of 3.6. Likewise it has sulphur levels far in excess of other drainage waters for which data has been collected for in the Latrobe Valley. The simulation results presented in Figure 24 therefore present a very severe scenario for the Latrobe Valley region. More likely scenarios are like at the Loy Yang overburden dump drainage discussed in Section 9.1, where the combined flow from the power station low quality water system was effectively neutralising any AD.

As part of an evaluation of the likely impact of the Loy Yang overburden drainage after cessation of the power station operations, a simulation using a mixture of Traralgon Creek water and Loy Yang overburden drainage water at a ratio of 5 parts of creek water to one part overburden run-off water yielded a pH of 6.9. A pH of 5.4 for the overburden run-off was used in the simulation. This simulation presents what is more likely to be a truer picture of water quality inflows into the mine.

It was discussed previously in Section 3 that there will be a need to use river water to fill all of the mines. This in it self is very advantageous from the point of view of not only limiting the salinity of the mine lakes, but also in minimising the risk of acidification of the lake waters. The full extent of the benefits of having good quality water as a source can not be understated. An excellent example of this is presented by Klapper et al.[21] who cite the results of channelling Schwarze Elster River water to the northern basin of the Senftenberger See in a flow through system. The northern basin, like the southern basin of the Senftenberger See, had been stable at a pH of approximately 3 since filling. The two basins are only separated by an island and macrophyte stands. Within 18 months of the river diversion, the pH in the northern basin increased to over 7, while that in the southern basin remained at near pH 3. The comment by the authors was that the rapid changes in pH were similar to a chemical titration curve. The mechanism of increasing the pH was due to both simple dilution and to the acid neutralising capacity of the river water. The process was stepwise, with the iron and aluminium buffers of the acid water first being overcome, then moving to the carbonate buffering system in the pH range of 6 to 8.

Although the evidence presented in this section strongly indicates that lake acidification is not likely to be a significant problem in the Latrobe Valley, this does not mean that mine operators in the Latrobe Valley can be complacent. The data also show that it does not take a great deal of AD to have an adverse impact upon lake water pH and that any such lowering of pH can be very abrupt. Rather, these data should be used to emphasise the need for good characterisation of the mine areas for acid producing potential and to properly manage any pyrite deposits that are identified.

9.4 Within-lake processes and lake water pH

From the evidence presented in Section 9.3 it is apparent that the likelihood of acid lakes being formed after cessation of the open cut mining operations in the Latrobe Valley region through pyrite oxidation are low, provided good mining management practices are followed. However, one other source of potential acidification needs to be commented upon, even if only briefly, and that is the potential of acid production from within the lakes themselves.

The main non-pyritic source of hydrogen ions to the lake water will potentially arise through ion exchange processes between the lake water and the coal. The pH of Victorian brown coals in water is determined predominantly by the carboxylic acid groups in the coal and can vary significantly between seams[23]. This variation depends on what prior exchange processes have occurred, that is whether the functional groups are in the acid or metal salt form.

Such cation exchange equilibria are influenced by both the size of the cation and the charge on the cation, with the equilibrium constants in general, increasing with increasing ionic charge and decreasing with increasing ionic radius[24]. This means that in mine lakes, the replacement exchange order will be: -

$$Fe^{+++}, Ca^{++}, Fe^{++}, Mg^{++}, Na^{+}, K^{+}, H^{+}$$

When in the acid form, cation exchange results in the release of hydrogen ions, with an associated decrease in pH. Whereas when in the salt form there is an exchange of cations, with minimal impact upon pH. This process is evident from studies in to the cation exchange properties of Victorian brown coals, which showed minimal change in pH of YNEOC coal during experiments that was attributed to the exchangeable ion being sodium rather than hydrogen[25]. Similarly, earlier studies[13] on Yallourn coals found pH changes occurring during ion exchange extraction experiments that were attributed to state of the carboxylic acid groups on the coal.

Woskoboenko[26] presented data showing that the pH of Victorian brown coal slurries varied with both the solids content and state of oxidation of the coals, with the more highly oxidised coals having larger number of acid groups. The data showed that Yallourn and Loy Yang coal slurries with solids concentrations of around 15 per cent by weight, had pHs of around 4.1. The Morwell and YNEOC coals had pHs around 5. These data and observations on influence of the state of oxidation of the coal are in close agreement with the tests of Deed[5] who found that 5 per cent coal slurries in a mixture of sodium chloride and demineralised water of Yallourn, Morwell and Loy Yang coals had pHs ranging between 3.7 and 5.9. The lowest pHs being found with the Yallourn and Loy Yang coals, and the higher pHs with the Morwell coals.

These data show that on exposed coal seams, that would be more oxidised than the fresh coals tested by Woskoboenko, any ground water, itself elevated in salinity, seeping from overburden batters on to the coal, will result in acid pHs arising from ion exchange processes yielding hydrogen ions. It is not surprising then to find many of the waters in the Latrobe Valley coal fields flowing over exposed coal and from horizontal bores to be acidic with no evidence of sulphate generated acidity.

However, from the perspective of lake chemistry, this form of acidity will be an insignificant contributor to the overall water quality. This can be seen from the data of Woskoboenko[26] that also show that as the coal slurry solids loading approaches zero, the pH of the slurry very quickly approaches that of the original mixing water. As the mine lakes are not going to be coal slurries, the pH effect of the carboxylic acid groups in the coal on the mine walls will be minimal.

10 Internal ash, coal and overburden dumping

10.1 Overburden dumping

Up to this point the examination of the water quality of mine lakes formed within the operating open cut mines has focused upon the broader issues of water source quality, nutrient supply and salinity. However, it is also pertinent to examine the impact of current operational activities such as internal ash, coal and overburden dumping.

The contribution of overburden dumping to salinity of the lake systems has been discussed in connection to the density stratification of the YNEOC southern lake. It is therefore of operational interest to make some assessment of the potential contribution to salinity of any overburden materials that will be back filled into the mines. The other main issue to consider with the dumping of clay materials into the mines is that in some areas there are extensive dispersive clay deposits. Any such clays will add to the turbidity of the lakes if backfilled into the mines and left exposed or untreated.

Whether to backfill with overburden or not to backfill is a question that can only be addressed in conjunction with other operational requirements. Mine closure regulations require capping of bores and such capping is most readily carried out by providing a low permeability barrier of clay. Similarly if ash is dumped into the open cuts, providing a cap of clay material will almost certainly be required to minimise the leaching of various trace elements into the water column. Thus it is probably unavoidable that overburden dumping into the open cut mines will need to take place.

10.2 Ash dumping

Ash disposal within the open cut mines is perhaps one of the more important issues that needs to be addressed with regard to water quality impacts. The reason for this is that ashes contain elevated levels of a range of cations and oxyanions that are potentially toxic. Leachablility of the various elements depends on a number of factors that influence the form of the elements in the ash and also on the leaching environment.

In a review of leaching of major and trace elements from coal ash, Jones[27] pointed out that the leaching characteristics of the various major and trace elements follow three basic patterns. The first pattern referred to by Jones as Type 1, was characterised by rapid dissolution of the soluble fraction of the element on the surface of the ash. Most of the major cations and sulphur fall into this category. Type 2 leaching of elements was characterised by a relatively uniform increase in concentration with increasing solids liquid ratios. Several reasons were suggested for such a leaching pattern, buffering of a poorly soluble surface phase, or desorption from a major host phase or from the slow dissolution of the bulk glassy matrix of the ash. Type 3 leaching was characterised by a slow initial release followed by rising concentration with increasing liquid solids ratios.

With all of these leaching studies, the focus has been on trying to emulate leaching conditions that may arise in ash disposal dumps, resulting from either dry or wet ashing. Under these conditions, it has been presumed that due to the low concentration of nutrient and organic materials present in ash, biological activity will be low and the leaching will take place under oxic conditions.

Thus studies on the leaching of Latrobe Valley ashes such as those of Black[28], Mudd et al.[29] and Mudd et al.[30] have had a focus on estimating the levels ash pond leachate migrating to the environment under oxic conditions.

One of the main points that is apparent from all of the leaching studies is that the disposal of ash via wet ashing is very effective in removing most of the Type I soluble salts from the dumped ash. In the Latrobe Valley, the resulting ash water is disposed of via an integrated ash effluent disposal system to an ocean outfall, thereby reducing potential problems associated with ash disposal and leaching to ground water and surface freshwater ecosystems.

If ashes are to be dumped within open cut mines, ash from wet ashing systems will present a more benign waste to dispose of than would ashes from dry ashing systems. The potential for adverse impact on the water quality of the future lakes depends on a number of factors. These include the total volume of ash and the potential for leachate exchange with the lake water. From a water quality point of view it is desirable to avoid dissolution of large amounts of trace elements into the water column. It is therefore presumed that any ash dumps formed within the open cut mines will be clay sealed to prevent free exchange of overlying water with the ash and to minimise any leachate contamination of ground waters.

Within the open cut mines, the leaching environment will be determined by the source of infiltration and the direction of water flow. During the early years of mine filling, if the deep aquifers are not adequately de-watered, it is possible that seepage from underlying aquifers could occur through the base of the mine. Such seepage occurs at the Loy Yang Mine at the present time. Pressure from such seepage could potentially damage the integrity of any clay liners around the dumped ash, allowing for a freer exchange of water with the ash. Later in the life of the lakes, as the mines fill above the piezometric heads of the deep aquifers, the direction of any seepage would be in a downward direction.

It was indicated previously that the lakes that will cover any such ash dumps are likely to be meromictic having bottom waters that are more saline. Associated with such salinity stratification, biological activity results in depletion of oxygen, with the bottom waters being anoxic. We therefore have a very different leaching scenario to how standard column leaching tests are carried out. The risk of contamination of the overlying water or ground water system will in part be determined by how effective the clay liners will be in acting as aquitards. It can only be presumed that the rate of exchange of any soluble species within the ash dump will occur through very slow upward diffusion processes. However, in view of the fact that any water permeating through the clay liner will be anoxic, the leaching will take place in a strongly reducing environment. We can therefore expect that species such as arsenic, copper, lead, zinc, nickel and cadmium that are normally extracted in column leaching tests of ashes, will form insoluble metal sulphides and precipitate. Furthermore, many of these trace metals and various of the oxyanions, such as arsenate (As^{5+}) and arsenite (As^{3+}) have a high affinity for ferric hydroxide and will precipitate if released to the epilimnion. Such co-precipitation of various trace element species by the normal iron-manganese redox cycling in the lakes, together with biogenic precipitation of trace elements is likely to nullify any slow diffusion of trace elements to the water column from any sealed ash dumps placed at the base of the open cut mines. Dumping of ashes in encapsulated dumps at the base of the open cut mines, if well managed, is not likely to have a major influence on the water quality of the lakes to be formed.

10.3 Loose coal dumping

It was noted in Section 9.4 that the contribution of the coal in the mine walls to acidification of the mine lake water would be negligible. Likewise, loose coal dumped in the open cuts would not contribute to acidification in any significant way as the dilution factor will result in a mixture with a solids content close to zero. The main issues that would need to be considered with loose coal dumping is the larger reactive surface allowing leaching of salts and potential for entrainment of coal fines in the water column contributing to the apparent colour. Evidence of such apparent colour arising from entrainment of fine coal particles in the water column can be seen in existing fire services dams within the open cuts. Coal particles were also a major discharge water quality issue at Loy Yang during the early 1980s when conveyor wash down water found its way to the main drain and thence to Traralgon Creek. However, such entrained coal particles require turbulent flow to maintain their position in the water column. In large deep lakes such as the mine lakes to be formed in the open cuts, coal particles will settle out and not contribute to apparent colour.

True colour, arising from the dissolution of humics will not occur to any extent in non-alkaline waters and does not present itself as a water quality issue to be concerned about at the pHs anticipated in the mine lakes. The northern lake in the YNEOC discussed previously, provides an indication of the likely appearance of the mine lakes to be formed in the major open cuts.

With regard to the release of salts from the loose coal, the coal quality will largely determine how much salt is released. For example, it could be anticipated that Flynn coals with their higher sodium contents would yield more salt to the overlying lake water than would the coals currently being mined at Loy Yang. Any release of salts associated with the coal is rapid. For example, Bou-Raad et al.[31] using a water extraction technique showed that most of the anions and cations in black coals are leached out in the first 36 to 48 hours. The total amount being approximately 30-40 per cent of the total cations and anions present in the coal. Hodges et al.[32] showed that under acidic conditions with pHs of 2.5 it is possible to remove 80-90 per cent of the major cations from the coal.

As an example of the contribution to salinity that the salts from loose coal may represent, a test was carried out using Loy Yang weekly composite coal samples. Samples of 2, 4, 8 and 20 g were slurried up with 200 g of Melbourne tap water yielding water-coal mixtures of 100:1, 50:1, 25:1 and 10:1, and allowed to soak for several days. The results of this test are presented in Table 14. It can be seen from the data in Table 14 that loose coals contribute much less salt to the water than do clay samples, data for which are presented in Figure 8.

Dumping of loose coal into the open cut mines is therefore not likely to be of a major consequence from the point of view of leaching of chemical species or contributing to the colour of the water or to salinity.

11 Anglesea

The Anglesea Open Cut Mine presents a slightly different scenario to that of the Latrobe Valley brown coal mines. Although the basic principles related to lake water quality are the same, the geology of the region is different, and the quality of the available surface water for filling the mine uncertain.

Deed[5] identified the Anglesea mine as one of the mines that is likely to end up with an acidic lake if allowed to flood. This prediction was made on the basis of pyritic deposits in the area and the quality of the ground water. Very little work has been carried out since Deed's work to quantify the extent of the potential acidification at Anglesea. However, if we examine the historical pH records upstream from the mine in Marshy Creek, which is the main stream draining the area, it can be seen that for the years 1972 to 1994, the pH was generally acidic with a pH of approximately 3.5 (Figure 25). There was only a short period between mid 1979 and mid 1981 when some pHs reached 5.5 to 6.2. Since 1994 the pH in Marshy Creek has become cyclic, annually ranging between around 3.5 and 7.0 pH units (Figure 26). The higher pHs are associated with elevated colour (Figure 27), which in turn can be attributed to elevated iron levels (Figure 28). This seasonal pattern of low pHs also corresponds with higher winter flows in the creek[19].

The reasons for the changes to the water chemistry of Marshy Creek since 1994 are unclear and require site specific investigations to elucidate the details. In Section 9.1 in discussing the drainage from the Loy Yang overburden dump it was suggested that the seasonal pattern may be associated with shallow pyrite deposits that are oxidised during the drier months, with the acid being flushed out during the wet weather flows.

It should be noted that the recorded iron concentrations are exceptionally high, and give reason to for concern about possible iron toxicity.

Whatever the reason for the pH changes in Marshy Creek, it is apparent that the quality of the surface water available for filling the Anglesea mine is not as good a quality as the streams in the Latrobe Valley.

12 Concluding discussion

The data presented in this report show that it will be necessary to divert river systems in order to fill each of the open cut mines in the Latrobe Valley. Without such diversion works, none of the open cuts is likely to completely fill, with the result that over time, through evaporative losses, the lakes will become increasingly more saline.

The time frame for filling the mines varies considerably, with Yallourn Mine, which has the smallest volume and being best serviced by availability of surface water resources for filling purposes, being able to be filled within a decade without any major impacts upon the flow of the Latrobe River. If a particularly wet year occurs within the filling period, utilising flood flows could considerably shorten the required filling time.

At the other extreme, the Loy Yang Mine, will have the largest pit, yet has the most limited access to surface water resources for filling. It is likely that a time frame of 150 years will be required to fill this mine. Closure plans will therefore require to incorporate on going maintenance programs for a very considerable period. From the view point of minimising such on going costs, it is imperative that consideration to alternative rehabilitation options for this mine to cover the interim filling period be developed.

Although the time frame for filling a mine such as Loy Yang is long, it should be kept in that the final rehabilitation option is for even longer time spans. After rehabilitation the mines are to be returned to the community essentially for an infinite period of time. It is therefore most important that any rehabilitation options that are implemented are self sustaining, requiring minimal on-site maintenance. Any options that require the establishment of infrastructure for ongoing dewatering of the mines or ground water, or other requirements that will involve significant recurring capital or maintenance expenditures beyond the initial filling period, are unlikely to be viable or acceptable to the community.

For the Anglesea Mine, more site specific work is required to obtain a better understanding of the quality of the available water resources before any firm conclusions regarding the filling and final water quality of a future mine lake in this open cut can be reached.

One of the difficulties that was experienced in this study was the lack of detailed information on likely ground water inflows through the surrounding overburden to the mines after closure. From discussions with various personnel associated with the development of the mines, the general view appears to be that only small amounts of water will enter as ground water from the local catchments. From studies carried out in Germany, rehabilitation of surrounding areas with vegetation can increase the rates of evapotranspiration to an extent that local ground water inflows are insignificant[33]. Management of ground water will be a major issue in ensuring good water quality in future lakes and this is an area that requires more attention.

One of the findings of this study was the possibility of overburden salt leaching resulting in salt stratification of the lakes. Current proposals are for all of the mines to practice backfilling with overburden. Already, large parts of the Yallourn Mine have been backfilled. Such backfilling, although a desirable practice for a number of other reasons may have the unplanned side effect of contributing to the establishment of meromictic lakes. The establishment of meromictic lakes in itself is not undesirable. However, one potential problem with large meromictic lakes connected to a river system is associated with possible build up of high levels of hydrogen sulphide that could result in a sudden release of this gas if the lake were to eventually turn over. Such a release could potentially have catastrophic effects on the river downstream.

One of the major issues concerning the mining companies associated with extraction of brown coal in Victoria is the potential for acidification of the water. Although acid drainage issues cannot be disregarded, the limited extent of pyritic deposits associated with the Latrobe Valley mines, and the need to use good quality water from catchment streams for filling purposes, will result in lakes with near neutral pHs.

Similarly, in view of the fact that there is generally a net phosphorus loss to the sediments in lake systems, and that the average residence time for water will be lengthy due to the volume of the lakes, the lakes are very likely to be of low productivity with resultant high clarity. The northern lake at the YNEOC is probably a good example of the likely final water quality that can be expected in such lake systems in the Latrobe Valley. The main difference will be that with the proposed lakes, each having a river inflow and an outflow, will ensure that the salinity of the surface waters of the lakes will be less than that of the YNEOC lakes and be suitable for discharge to the catchment streams.

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14 References

1 Woodward-Clyde (1999) Proposed Maryvale Coal Field Development: Environmental Effects Statement, Main Report. AGC Woodward-Clyde Pty Ltd., p10- 28.

2 Brumley J. C., C. M. Barton, G. R. Holdgate and M. R. Reid (1981) *Regional Groundwater Investigations in the Latrobe Valley: 1976-1981*. State Electricity Commission of Victoria and Victorian Department of Minerals and Energy, p37.

3 Walker G. and F Mollica (1990) Review of the Groundwater Resources in the South East Region. Department of Water Resources Victoria, Report No.54, Water Resource Management Report Series.

4 Williams W. D. (1967) The Chemical Characteristics of Lentic Surface Waters in Australia: A Review. In Australian Inland Waters and their Fauna: Eleven Studies. Edited by A. H. Weatherley, Australian National University Press.

5 Deed R (1975) Expected Quality of Water in Flooded Latrobe Valley Open Cuts. State Electricity Commission of Victoria Planning and Investigations Department, Scientific Division, Report No. 314.

6 Sinclair Knight Mertz and Douglas Partners (1999) Maryvale Project Morwell River Diversion: Functional Design, Geotechnical Investigation and Design. Environmental Effects Statement, Maryvale Coal Field Development, AGC-Woodward-Clyde, Supplementary Reports, Volume 2, p39.

7 Kevin Brown - Yallourn Mine Planning Engineer (pers. comm.)

8 Envirogen (1996) Yallourn North Extension Open Cut – colour survey. Report to Yallourn Mine, April 1996, Envirogen Report No. 9603242.

9 Harasymiw B. and B. Wright (1998) Eelhole Creek temperature regulation by control of Hazelwood cooling Pond storage levels. HRL Technology Pty. Ltd., Report No. HLC/97/622.

10 Wetzel R. G. (1983) Limnology. Saunders College Publishing. 2nd edition, p164.

11 Hamilton-Taylor J., and W. Davison (1995) Redox driven cycling of trace elements in lakes. In Physics and Chemistry of Lakes, eds A. Lerman, D.M. Imboden and J. R. Gat, Springer-Verlag, Berlin. Ch. 8, pp217-263.

12 EPA Victoria (1999) Acid sulphate soil and rock. Information Bulletin, Publication No.655.

13 Murray, J. B. (1968) The state of combination of the inorganic ash-forming constituents of Yallourn brown coal. State Electricity Commission of Victoria

Planning and Investigations Department, Scientific Division. Miscellaneous Report No.MR-145.

14 Morse, J. W., F. J. Millero, J. C. Cornwell and D. Rickard (1987) The chemistry of the hydrogen sulphide and iron sulphide systems in natural waters. Earth Sci. Rev. 24:1-42.

15 Holdgate. G (pers. comm.)

16 Earth Systems (1999) Acid Drainage Risk Assessment and preliminary baseline study, Maryvale Project for Yallourn Energy. Report by Earth Systems in association with Sinclair Knight Mertz, January 1999.

17 McIlwaine, L. (1998) Prediction of sulphide oxidation rate of waste rock material and consequences for acid rock drainage generation at Loy Yang Coal Mine. Report to Loy Yang Mine.

18 Ahern, C. R., Y. Stone and B. Blunden (1998*) Acid Sulphate Soils Assessment Guidelines*. Published by the NSW Acid Sulphate Soils Management Advisory Committee, Wollongbar, NSW, Australia.

19 Maryann Thorp – Anglesea Power Station (pers. comm.)

20 Harasymiw, B. (1994) Traralgon Creek Biological Survey 1993. State Electricity Commission of Victoria, Research and Development Department, Laboratory Report No. LO/93/748.

21 Klapper, H., K. Friese, B. Scharf, M. Schimmele and M. Schultze (1998) Ways of Controlling Acid by Ecotechnology. In Acid Mining Lakes: Acid Mine Drainage, Limnology and Reclamation. Eds. Walter Geller, Helmut Klapper and Wim Salomons. Springer. Ch.22.

22 MINEQL+© (1998) A chemical equilibrium modelling system. Version 4 for Windows. W D Schecher and D C McAvoy. Environmental Research Software, Hallowell, ME..

23 Brockway, D. J. and R. S. Higgins (1991) Brown coal sampling, analysis and composition. In The Science of Victorian Brown Coal: Structure, Properties and Consequences for Utilisation. Ed. R. A. Durie, Butterworth-Heinman Ltd, Lond., Ch.5.

24 Hillier, L .A. and R. H. Herber (1960) Principles of Chemistry. McGraw-Hill Book Company Inc., p477.

25 Lafferty, C. J., M. D. Hobday and G. J. Perry (1988) Victorian brown coal as a heavy metal scavenger. In Australian Coal Science Conference 1988, 16-18 May 1988 (University of Adelaide), pp. A3:1.1- A3:1.8.

26 Woskoboenko, F., W. O. Stacy and D. Raisbeck (1991) Physical structure and properties of brown coal. In The Science of Victorian Brown Coal: Structure, Properties and Consequences for Utilisation. Ed. R. A. Durie, Butterworth-Heinman Ltd, Lond., Ch.4.

27 Jones, D. R. (1995) The leaching of major and trace elements from coal ash. In Environmental aspects of trace elements in coal. Edited by D. J. Swaine and F. Goodarzi. Kluwer Academic Publishers. Ch.12, pp221-262.

28 Black, C. (1990) *Yallourn W ash leaching*. State Electricity Commission of Victoria, Research and Development Department Report No. SC/90/174.

29 Mudd, G. M., J. Kodikara and T. McKinley (1996) *Assessing the environmental impact of coal ash disposal*. AIE 7th Australian Coal Science Conference, Monash University, Gippsland, 2-4 December 1996, pp363-370.

30 Mudd, G. M., J. Kodikara and T. McKinley (1998) Coal ash leachability: detailed field studies. AIE 8th Australian Coal Science Conference, University of New South Wales, 7-9 Dec 1998, pp357-362.

31 Bou-Raad, M., M. Hobday, and C Rix (1996) *Quantitative analysis of sulphur, nitrogen, chlorine and other non-metallic elements in coal*. AIE 7th Australian Coal Science Conference, Monash University, Gippsland, 2-4 December 1996, pp463-470.

32 Hodges, S., R. Saripalli, A. Campisi (1994) *Acid washing of brown coals*. AIE 6th Australian Coal Science Conference, Newcastle, pp .

33 Klapper, H. and M. Schultze (1995) *Geogenically Acidified Mining Lakes – Living Conditions and Possibilities for Restoration*. Int. Revue ges. Hydrobiol. 80:639-653.

Table 1 Annual evaporation and precipitation at each of the three main open cut mines in the Latrobe Valley of Victoria. Also shown are the mine surface areas and predicted equilibrium lake surface levels in the absence of additional sources of inflow.

	Yallourn Mine	**Hazelwood Mine**	**Loy Yang Mine**
Annual precipitation (mm)	902	711	793
Annual evaporation (mm)[(a)]	980	1015	942
Net precipitation deficit (mm)	78	304	149
Mine surface area (km^2)	19	14.7	20.2
Mine lake surface below lowest point of crest (m)	29[(b)]	84	59
Approximate maximum depth of mine (m)	90	130	150

[(a)] *The lake surface evaporation rates are estimated by multiplying Pan A evaporation rates for each mine by a factor of 0.75. This factor is a general approximation. In reality it will vary from site to site and during the course of filling within each mine as the wind sheltering coefficients of the mine walls change. However, from the point of view of this study, the value used is of sufficient accuracy to provide indicative lake levels.*

[(b)] *This is the depth to which Yallourn Mine would fill in the absence of large shallow areas created by internal overburden dumps. Any large areas of relatively shallow depth will impact upon the evaporative surface area of the lake, allowing for higher water levels to be attained.*

Table 2 Average annual and equivalent average daily discharges for the Latrobe River, the Morwell River and Traralgon Creek.

	Average annual discharge (GL)	**Average daily flow (ML)**
Latrobe River	660	1808
Morwell River	135	427
Traralgon Creek	35	96

Table 3 Mine characteristics as currently planned for the completed open cut mines in the Latrobe Valley. Average batter slopes of 3:1 (horizontal:vertical) are assumed. All data courtesy of mine planning engineers from the respective mines and GeoEng Pty Ltd.

	Yallourn Mine	Hazelwood Mine	Loy Yang Mine
Mine area at crest (km^2)	19	14.7	20.2
Mine volume (m^3)	7.32×10^8	1.48×10^9	2.05×10^9
Mine perimeter (km)	21	18.5	18.8
Maximum depth (m)	90	130	150

Table 4 Major cations and anions in freshwater systems contributing to salinity.

Cations	Anions
Calcium (Ca^{2+})	Bicarbonate (HCO_3^-)
Magnesium (Mg^{2+})	Carbonate (CO_3^{2-})
Sodium (Na^+)	Sulphate (SO_4^{2-})
Potassium (K^+)	Chloride (Cl^-)

Table 5 Distribution of major ions contributing to salinity in various bores and surface water in the Latrobe Valley. Values for Loy Yang seams are based on data provided by Geo-Eng Pty Ltd and are the average values from the various bores drilled into the seams identified. Other mine data collected during this study. River water data comes from the SECV WaterQ database.

	Average values						
Loy Yang (Seam ID)	Na^+	Mg^{2+}	Ca^{2+}	K^+	Cl^-	$HCO_3^-+CO_3^{2-}$	SO_4^{2-}
Coal	53	12	5.8	-	153.5	-	50.0
M2B	83.0	12.9	4.3	1.6	154.5	4.8	24.6
M2C	64.4	8.6	4.8	2.0	107.7	8.5	38.1
M2CISS	93.5	13.8	3.8	2.1	169.5	5.4	27.1
Tr	30.0	2.9	2.8	0.9	50.5	9.2	5.1
Horiz. bore LY3546	670	140	37	32	1515	2.2	24
Horiz. bore LY3663	610	88	35	13	1325	0	45
Hazelwood							
Horiz. bore M3656	86	10	7.8	7.1	160	4	26
Horiz. bore M3662	97	11	6.9	8.1	180	2.8	44
Base of western OB batters	125	23	15	12	217	6	79
Yallourn							
East Field Overburden drainage	154	43	19	5	365	6	12
Major surface waters							
Latrobe River	22.1	4.7	4.6	3.0	42.8	17.2	8.6
Morwell River	33.4	8.9	9.2	3.0	69.0	36.1	8.7
Traralgon Creek	27.4	6.7	10.1	2.4	43.8	41.9	5.1

Table 6 Distribution of major species contributing to salinity in the YNEOC lakes during sampling in May 1999. Surface samples only.

	Southern Lake	Overburden Run-off (north western drain)	Northern Lake
pH	6.3	6.4	6.8
Conductivity mS/m	165	273	108
Suspended Solids mg/L	7.1	25	1.3
Total Dissolved Solids mg/L	1150	1855	810
Chloride mg/L Cl	475	920	250
Sulphate mg/L SO_4	190	130	195
Bicarbonate mg/L $CaCO_3$	2.7	3.8	2.7
Sodium mg/L Na	215	390	125
Calcium mg/L Ca	61	89	49
Magnesium mg/L Mg	47	63	30
Potassium mg/L K	6	4.6	6.2
Iron mg/L Fe	<0.1	<0.1	<0.1
Manganese mg/L Mn	0.1	0.3	<0.1

Table 7 Distribution of major ionic species contributing to salinity in the YNEOC lakes as found in August of 1999. From the data it appears that the overburden run-off is contributing to the higher salinity of the southern lake and to the salinity stratification that exists in this lake. See text for discussion

	Southern Lake		OB run-off channel	Northern Lake	
Depth	**(surface)**	**(8 metres)**		**(surface)**	**(12 metres)**
PH	7.1	5.9	5.8	6.3	6.1
Conductivity mS/m	184	200	299	116	117
Suspended Solids mg/L	15	42	8.9	2.7	4.4
Total Dissolved Solids mg/L	1235	1380*	1835	740	720
Chloride mg/L Cl	475	485	820	255	260
Sulphate mg/L SO4	190	260	125	185	180
Bicarbonate mg/L $CaCO_3$	4.2	3.5	3.7	2.8	2.7
Sodium mg/L Na	270	275	440	160	160
Calcium mg/L Ca	61	76	80	51	50
Magnesium mg/L Mg	53	61	70	35	36
Potassium mg/L K	6.5	6.7	4.8	6.7	6.7
Silica mg/L SiO_2	16	20	17	4	4.1
Manganese mg/L Mn	0.1	0.3	0.3	<0.1	<0.1
Iron mg/L Fe	4.5	4.5	5.6	<0.1	<0.1

**These data suggest that there was contamination of the depth sample with water from the surface. This is possible as the sampling apparatus was not completely watertight. More accurate depth-salinity readings are presented in Figures 9, 10 and 18, data for which were obtained using a calibrated submersible data logger.*

Table 8 Summary of major chemical species of the three major streams that would be required as sources of water for maintaining water flow through conditions in the mine lakes in the Latrobe Valley.

		Latrobe River at Brown Coal Mine Bridge	Morwell River at Amiets	Traralgon Creek at Jones Lane
pH		7.2	7.3	7.2
Alkalinity (Bicarbonate)	mg/L $CaCO_3$	17.2	36.1	41.9
Colour	Pt/Co	50.9	44.0	46.3
Conductivity	mS/m	16.2	31.8	24.9
TDS	mg/L	117.7	180.8	141.3
Suspended Solids	mg/L	24.8	18.8	21.9
Turbidity	NTU	22.4	19.8	20.8
Oxygen	mg/L	8.4	8.0	7.3
BOD	mg/L	1.7	1.2	1.2
COD	mg/L	16.9	15.8	15.3
TOC	mg/L	9.52	9.5	6.3
E. coli	orgs/100 ml	238.0	558.4	534.7
Ammonia-N	mg/L	0.128	0.129	0.133
Kjeldahl-N	mg/L	1.315	0.452	0.295
Nitrate-N	mg/L	0.205	0.289	0.222
Nitrite-N	mg/L	0.006	0.004	0.003
Orthophosphate-P	mg/L	0.043	0.023	0.033
Total phosphorus-P	mg/L	0.097	0.068	0.072
Sulphate	mg/L	8.6	8.7	5.1
Chloride	mg/L	42.8	69.0	43.8
Silica	mg/L SiO_2	8.7	14.4	15.6
Aluminium	mg/L	1.4	1.6	2.0
Sodium	mg/L	22.1	33.4	27.4
Potassium	mg/L	3.0	3.0	2.4
Calcium	mg/L	4.6	9.2	10.1
Magnesium	mg/L	4.7	8.9	6.7

Table 9 Summary of trace chemical species of the three major streams that would be required as sources of water for maintaining water flow through conditions in the mine lakes in the Latrobe Valley.

		Latrobe River at Brown Coal Mine Bridge	**Morwell River at Amiets**	**Traralgon Creek at Jones Lane**
Mercury (total)	µg/L	0.1	0.1	0.1
Selenium (total)	µg/L	0.4	0.2	0.1
Iron (total)	µg/L	1533.3	1477.9	1540.8
Manganese (total)	µg/L	96.0	167.0	92.7
Zinc (total)	µg/L	46.8	31.6	25.5
Copper (total)	µg/L	14.5	5.1	4.6
Cadmium (total)	µg/L	0.4	0.7	1.0
Nickel (total)	µg/L	3.2	3.3	2.9
Lead (total)	µg/L	3.4	2.0	3.4
Cobalt (total)	µg/L	1.6	2.7	0.9
Chromium (total)	µg/L	2.8	2.0	2.4
Boron (total)	µg/L	45.7	59.4	41.2
Selenium (dissolved)	µg/L	0.3	0.1	0.1
Iron (dissolved)	µg/L	319.7	368.3	478.7
Manganese (dissolved)	µg/L	39.3	85.4	73.0
Zinc (dissolved)	µg/L	21.2	21.9	16.4
Copper (dissolved)	µg/L	7.5	1.9	2.2
Cadmium (dissolved)	µg/L	0.2	0.2	0.2
Nickel (dissolved)	µg/L	7.7	2.7	2.6
Lead (dissolved)	µg/L	6.8	1.4	1.8
Cobalt (dissolved)	µg/L	10.7	1.4	0.6
Chromium (dissolved)	µg/L	1.6	1.1	2.0

Table 10 Areal loading of nitrogen and phosphorus in the three mine lakes to be formed in the Latrobe Valley assuming that the Yallourn mine will use Latrobe River water, Hazelwood mine will used Morwell River water and the Loy Yang mine will use Traralgon Creek. Flows equivalent to 100 ML/d, which is equivalent to the total average daily flow of Traralgon Creek, have been used in this simulation. Values in g m^{-2} $year^{-1}$.

Lake	N	P
Yallourn	3.18	0.19
Morwell	2.17	0.17
Loy Yang	1.18	0.13

Table 11 Provisional permissible loading levels of total nitrogen and total phosphorus in g m^{-2} $year^{-1}$. (Data from Wetzel 1983).

	Permissible loading		Dangerous loading	
Mean depth (m)	**N**	**P**	**N**	**P**
5	1.00	0.07	2.00	0.13
10	1.50	0.10	3.00	0.20
50	4.00	0.25	8.00	0.50
100	6.00	0.40	12.00	0.80
150	7.50	0.50	15.00	1.00
200	9.00	0.60	18.00	1.20

Table 12 Typical iron and sulphur content of some Victorian brown coals.

Location	Component (% dry coal basis)			
	Total iron	**Non-pyritic iron**	**Total sulphur**	**Pyritic sulphur (calculated)**
Anglesea Open Cut[(a)]	0.25	0.12	3.3	0.13
Yallourn Open Cut[(a)]	0.09 – 0.26	0.09 – 0.26	0.14 – 0.59	0.0
Yallourn Open Cut[(b)] **(East Field)**	0.5 - 0.6	0.5 -0.6	0.2- 0.3	0.0
YNEOC[(a)]	0.33 – 1.02	0.24 - 0.60	0.17 – 0.63	0.02 – 0.53
Hazelwood Open Cut[(a)]	0.04 – 0.37	0.04 – 0.26	0.16 –0.50	0.0 –0.14
Loy Yang Open Cut[(b)]	0.05 – 0.12	0.05 – 0.12	0.2 – 0.4	0.0 – 0.1

(a) Data from Deed (1975)[5]

(b) Data from HRL coal chemistry database. Analyses carried out on weekly composite samples of coal.

Table 13 Chemical characteristics of the acidic mine drainage water containing high levels of sulphate and Latrobe River water, used in the simulation of Yallourn Lake pH.

Parameter	Units	Latrobe River at Brown Coal Mine Bridge	Yallourn MW08 seepage[a]
pH		7.2	3.65
Alkalinity (Bicarbonate)	mg/L CaCO3	17.2	0
Conductivity	mS/m	16	2610
TDS	mg/L	118	1749
Sodium	mg/L	22.1	215
Calcium	mg/L	4.6	95
Magnesium	mg/L	4.7	169
Potassium	mg/L	3.0	2
Silica	mg/L SiO_2	8.7	10
Aluminium	mg/L	1.4	3.32
Sulphate	mg/L	8.6	985
Chloride	mg/L	42.8	261
Iron (dissolved)	µg/L	319.7	1.53
Manganese (dissolved)	µg/L	39.3	20.8
Cadmium (dissolved)	µg/L	0.2	0.1
Chromium (dissolved)	µg/L	1.6	1
Cobalt (dissolved)	µg/L	10.7	
Copper (dissolved)	µg/L	7.5	1
Lead (dissolved)	µg/L	6.8	1
Nickel (dissolved)	µg/L	7.7	151
Zinc (dissolved)	µg/L	21.2	698

[a] *Data from Reference 16.*

Table 14 Total dissolved solids and pH of slurries of 2, 4, 8 and 20 g samples of Loy Yang weekly composite coal samples mixed with 200 g of Melbourne tap water.

Time	**Total dissolved solids (mg/L)**				**pH**			
Weight of coal	2 g	4 g	8 g	20 g	2 g	4 g	8 g	20 g
Water:coal ratio	100:1	50:1	25:1	10:1	100:1	50:1	25:1	10:1
24 (hours)	109	123	175	310	5.84	5.40	5.08	4.80
72 (hours)	107	136	193	322	5.45	5.22	5.02	4.73

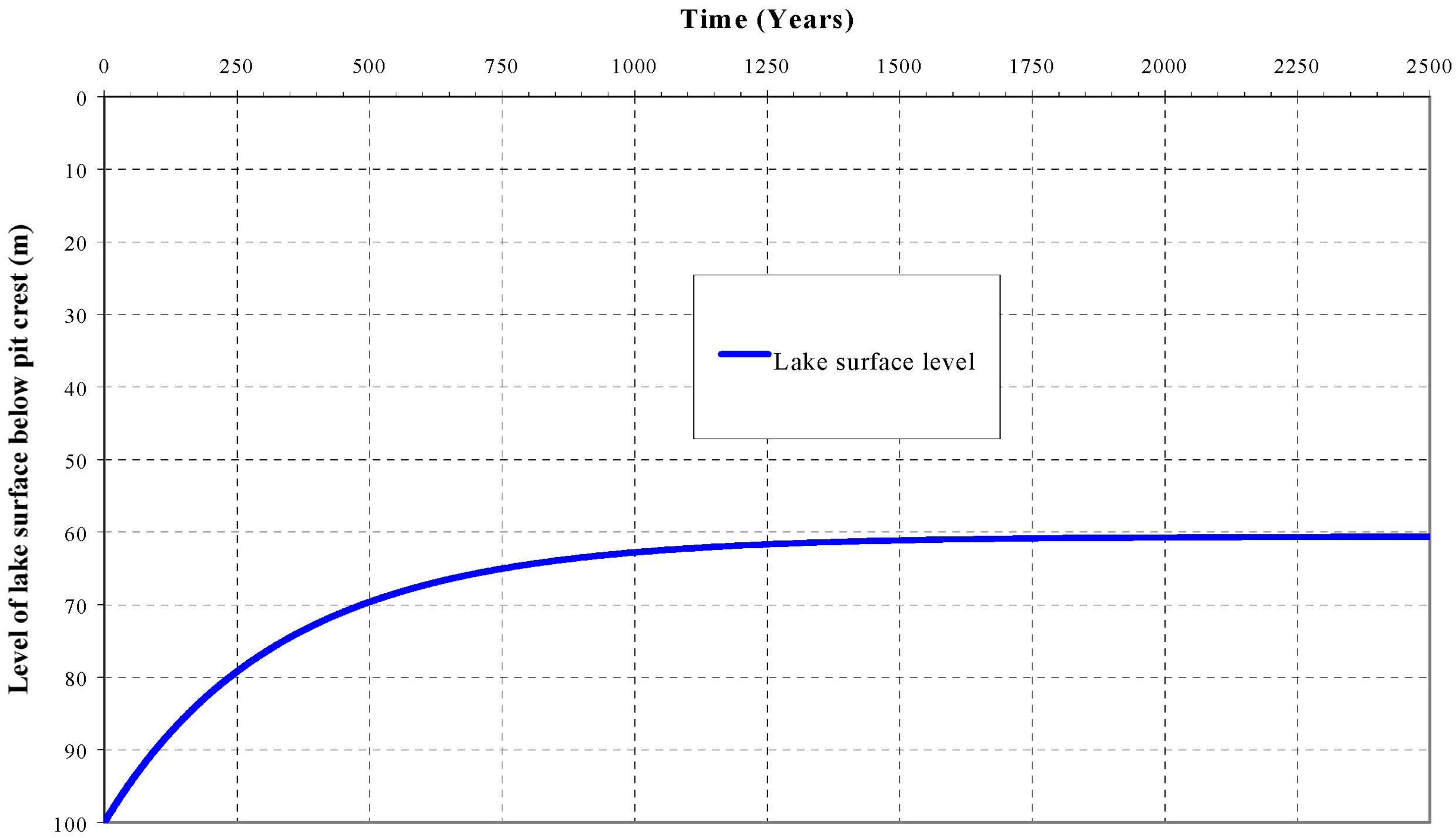

Figure 1 Filling rate curve for a hypothetical mine lake in the Latrobe Valley of Victoria with average batter slopes of 3:1, a surface area of 21 km^2 and depth of 100 m. Average annual precipitation and evaporation rates of 793 mm and 942 mm are used. No other inflows or outflows are assumed.

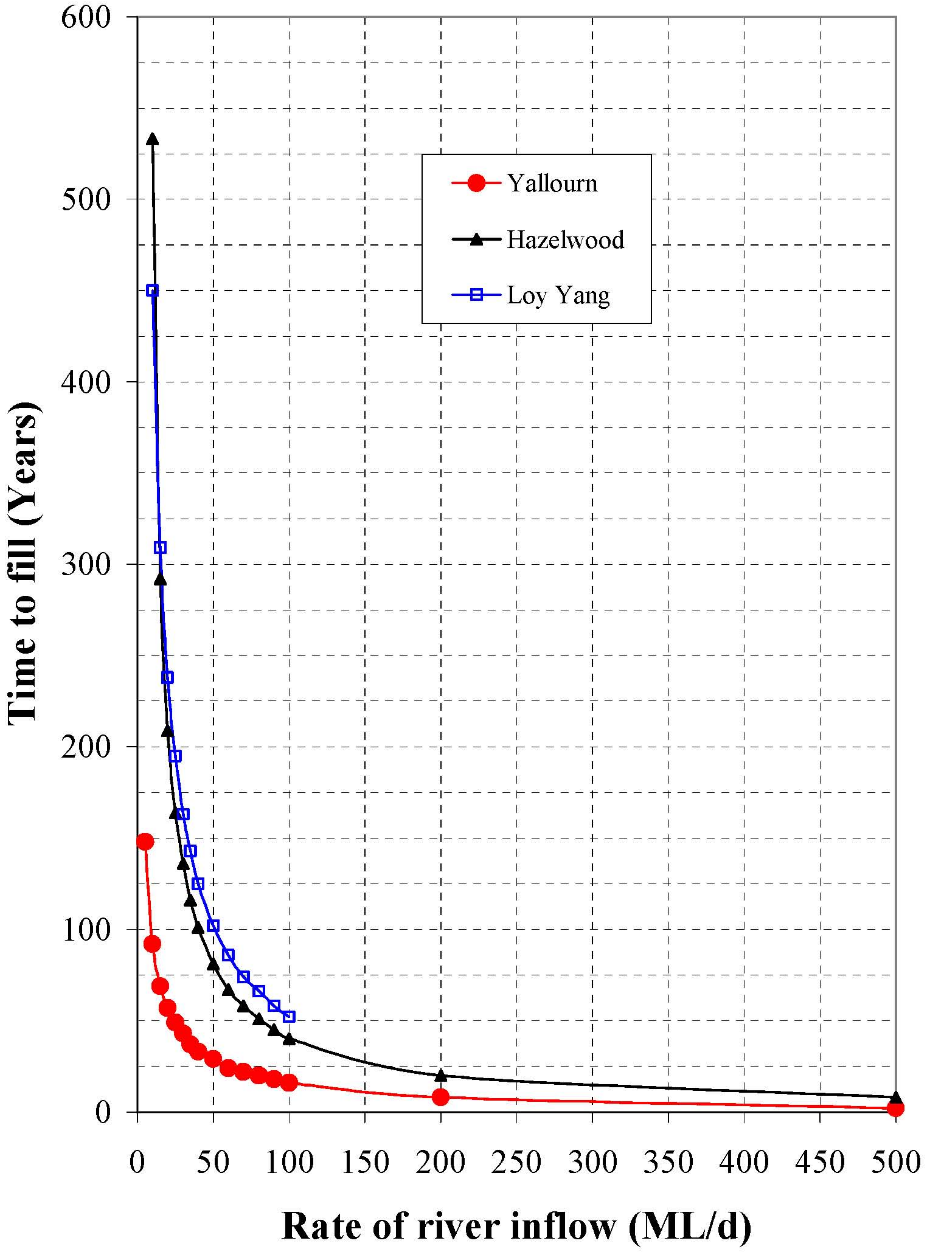

Figure 2 Filling times for the three open cut mines under different river inflow scenarios. Ground water inflows of 50 L/s are assumed in all cases. At the lowest rate of inflow of 5 ML/d, Hazelwood would not fill and Loy Yang would take nearly 1000 years to fill.

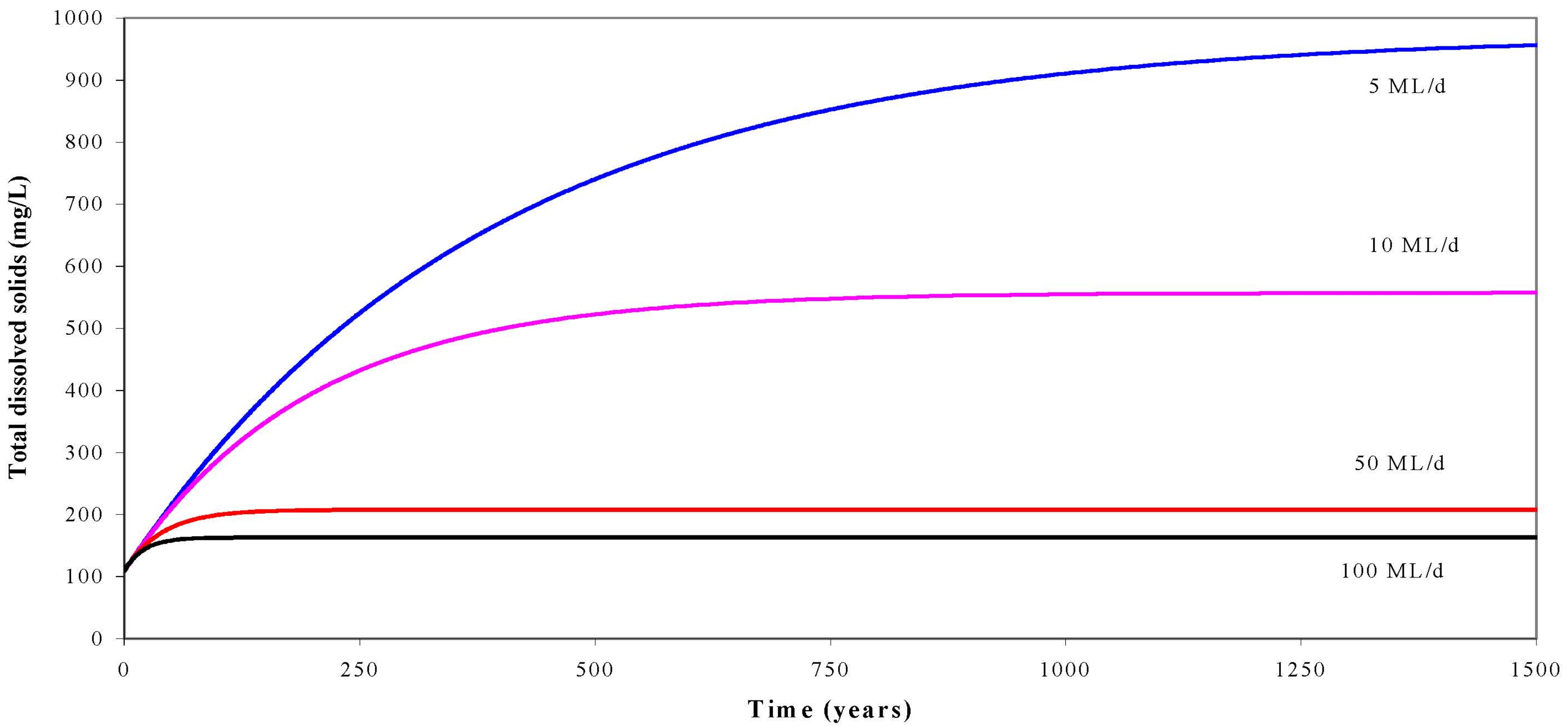

Figure 3 Examples of potential TDS levels in the Yallourn Lake assuming different rates of river flushing after filling. Filling rate was assumed at 100 ML/d of river inflow. Ground water inflows of 50 L/s at 1000 mg/L total dissolved solids are assumed.

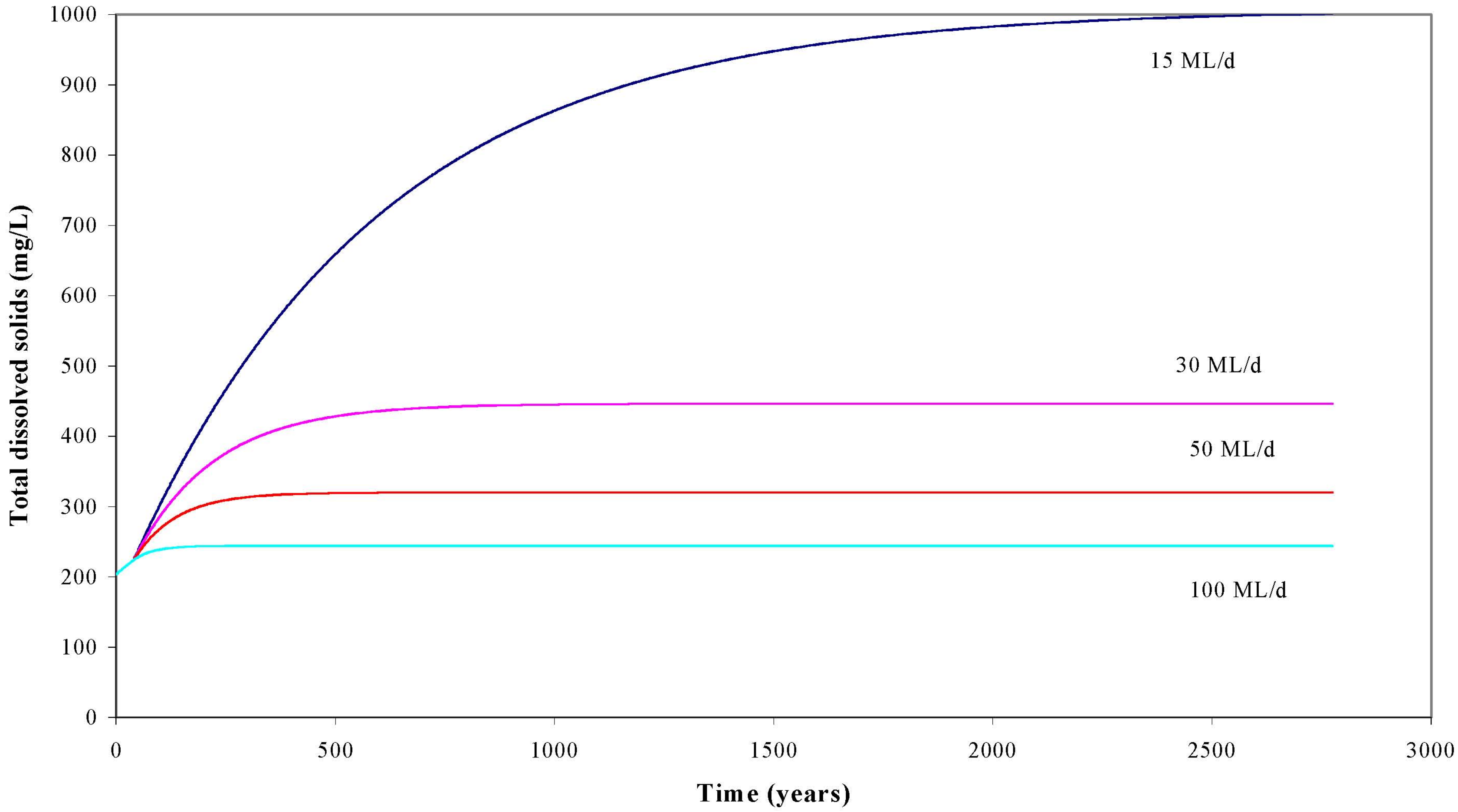

Figure 4 Examples of potential TDS levels in the Hazelwood Lake assuming different rates of river flushing after filling. A filling rate of 100 ML/d of river inflow was used. Ground water inflows of 50 L/s at 1000 mg/L of total dissolved solids are assumed.

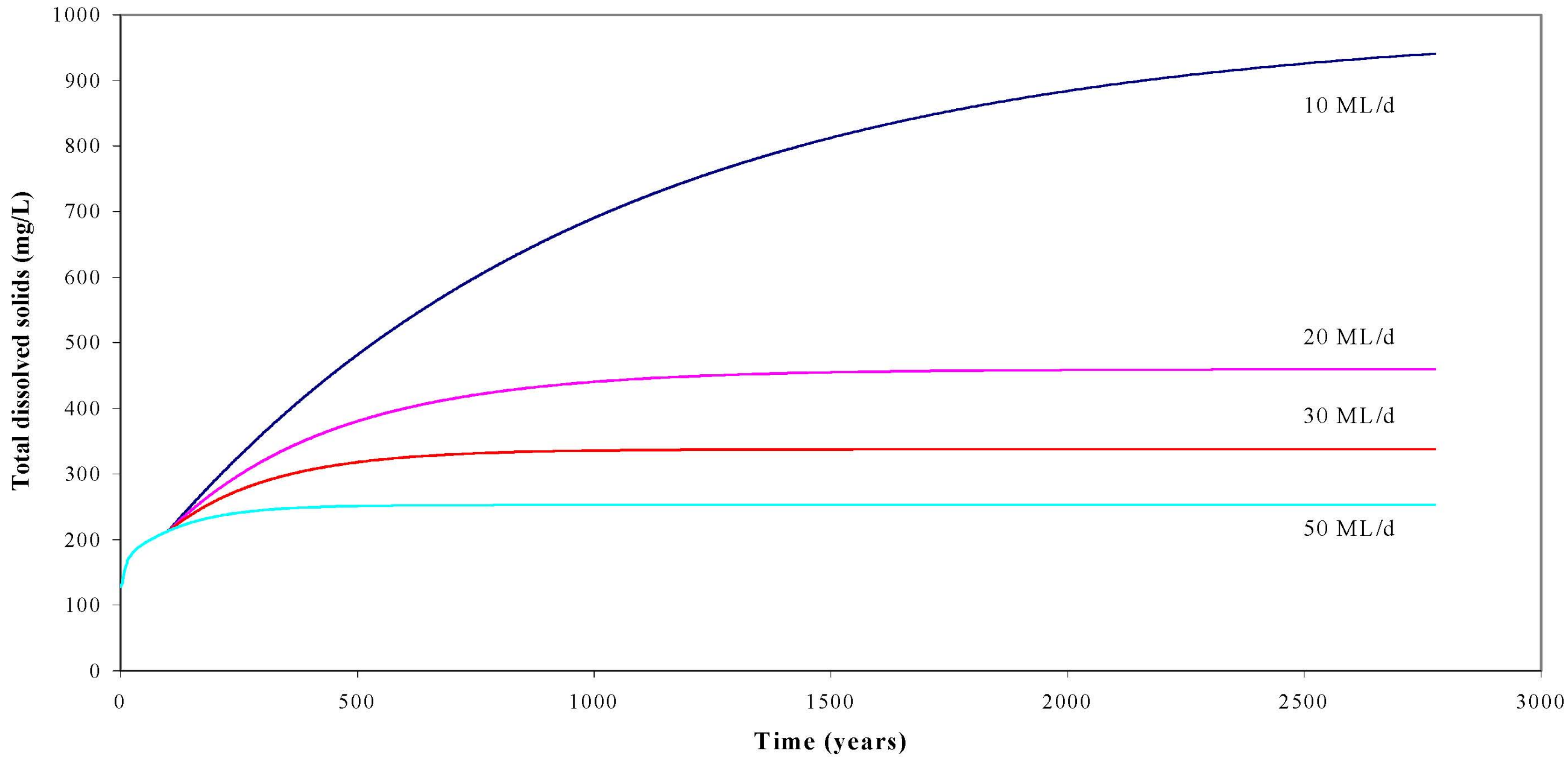

Figure 5 Examples of potential TDS levels in the Loy Yang Lake assuming different rates of river flushing after filling. Filling rate was assumed at 50 ML/d of river inflow. Ground water inflows of 50 L/s at 1000 mg/L of total dissolved solids are assumed.

Figure 6 A view of the YNEOC lake system. The northern lake is in the foreground and the southern lake is in the background.

Figure 7 An aerial view of the YNEOC lake system circa 1995 prior to rehabilitation works of the overburden dumps and surrounding areas aving been completed. The large mid-field overburden dump and the small overburden capped island in the sourthern lake are clearly visible.

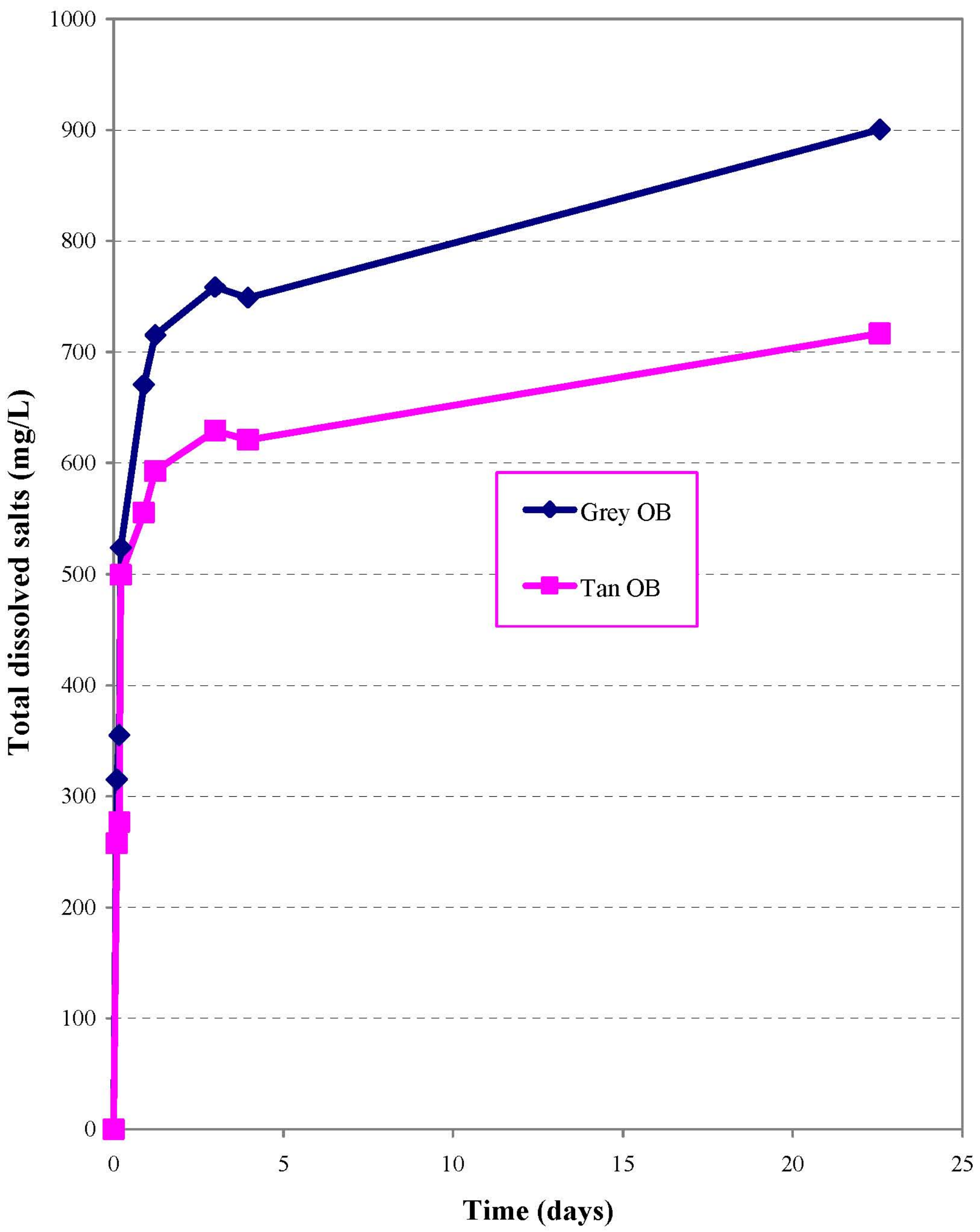

Figure 8 Change in total dissolved solids concentration of two Loy Yang clay water mixtures with time.

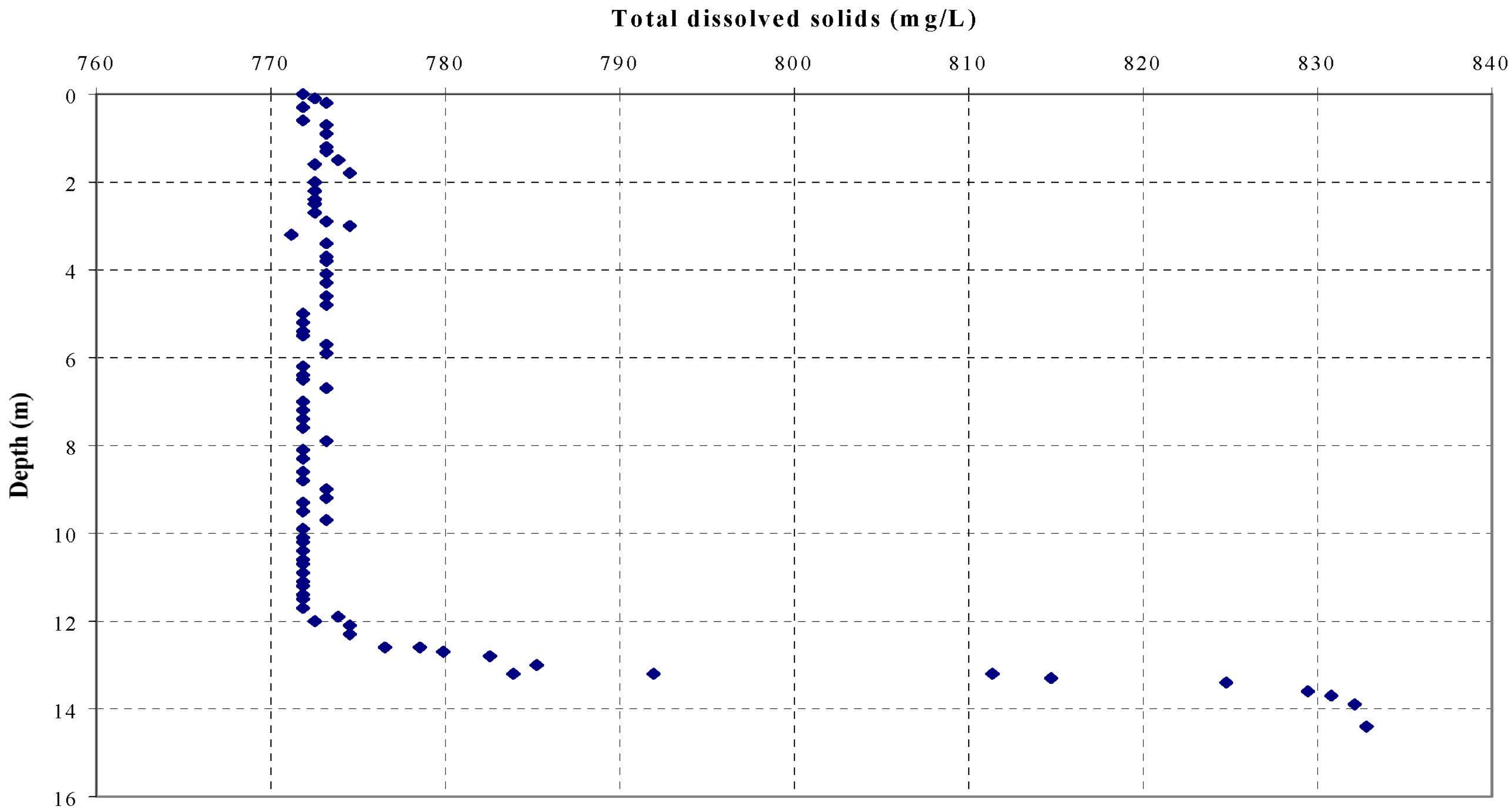

Figure 9 Total dissolved solids concentrations as calculated from the conductivity measurements taken in the northern YNEOC lake during May 1999. The data show the existence of a very sharp chemocline between 12 m and 14 m depths. The difference in salinities between the mixolimnion and the monimolimnion is only of the order of 60 mg/L of dissolved salts.

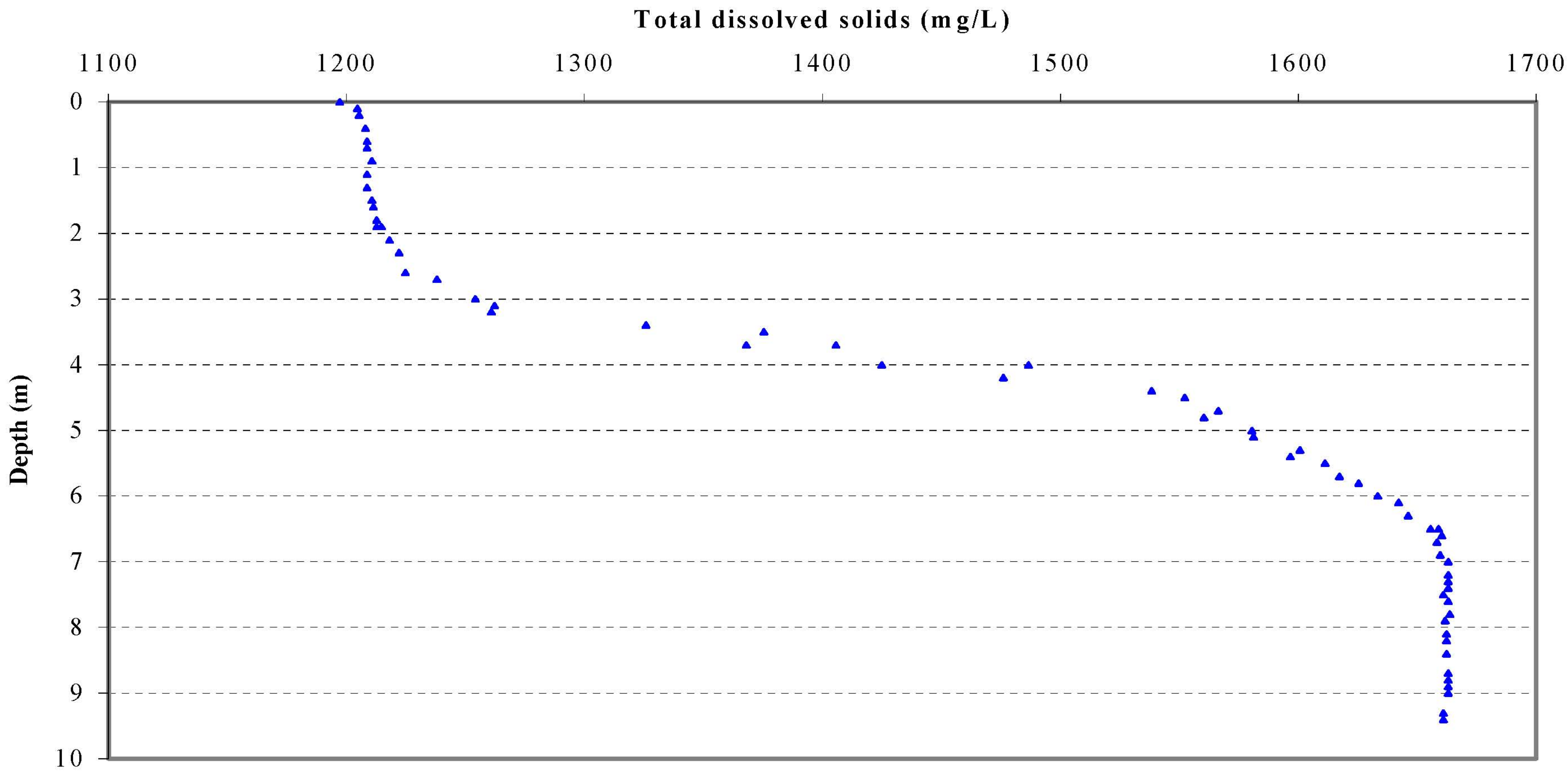

Figure 10 Total dissolved solids concentrations as calculated from the conductivity measurements taken in the southern YNEOC lake during May 1999. The data show the existence of a chemocline between 3 m and 6 m depths. The difference in salinities between the mixolimnion and the monimolimnion is of the order of 450 mg/L of dissolved salts.

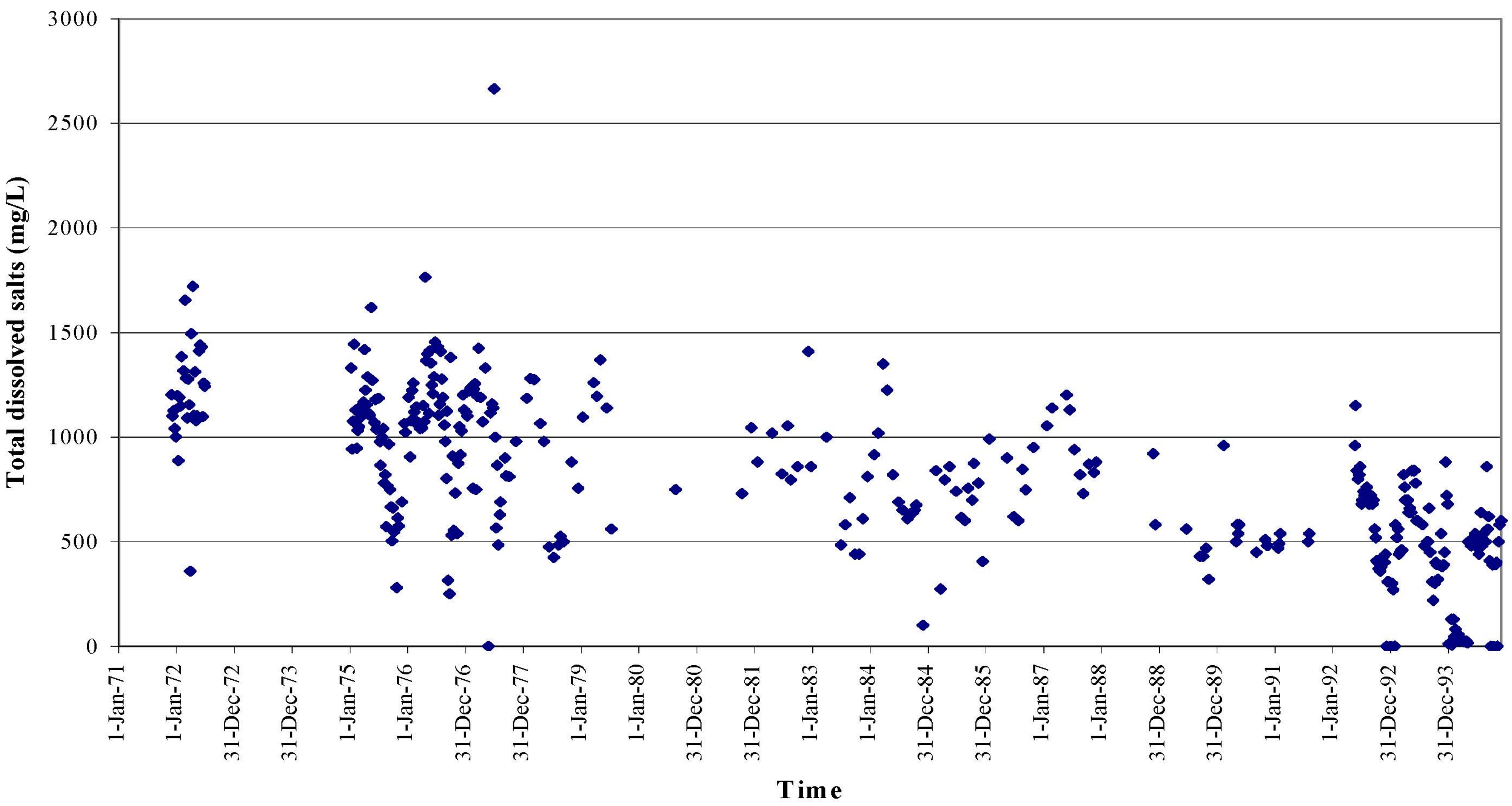

Figure 11 Historical data on the TDS levels in the eastern overburden dump catchment drainage at the YNEOC. The reduction in the salinity over time is attributed to the rehabilitation works that have taken place.

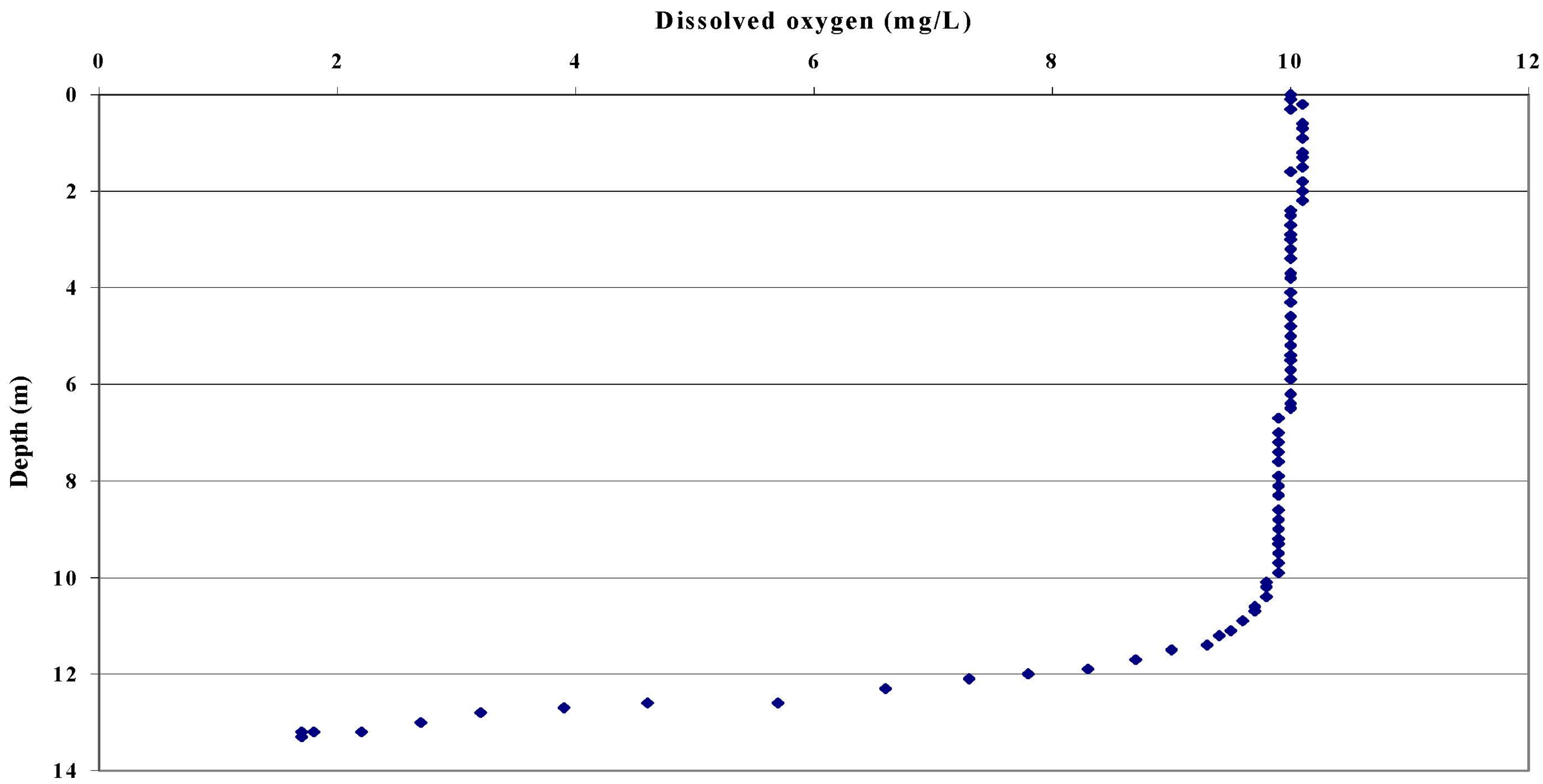

Figure 12 Vertical profile of oxygen concentration in the northern lake in the YNEOC during early May 1999. The profile shows a well-oxygenated upper mixolimnion and a very sharp decrease in oxygen concentrations through the chemocline, dropping less than 2 mg/L in the monimolimnion or approximately 15 per cent saturation. In the mixolimnion the oxygen saturation is almost 100 per cent.

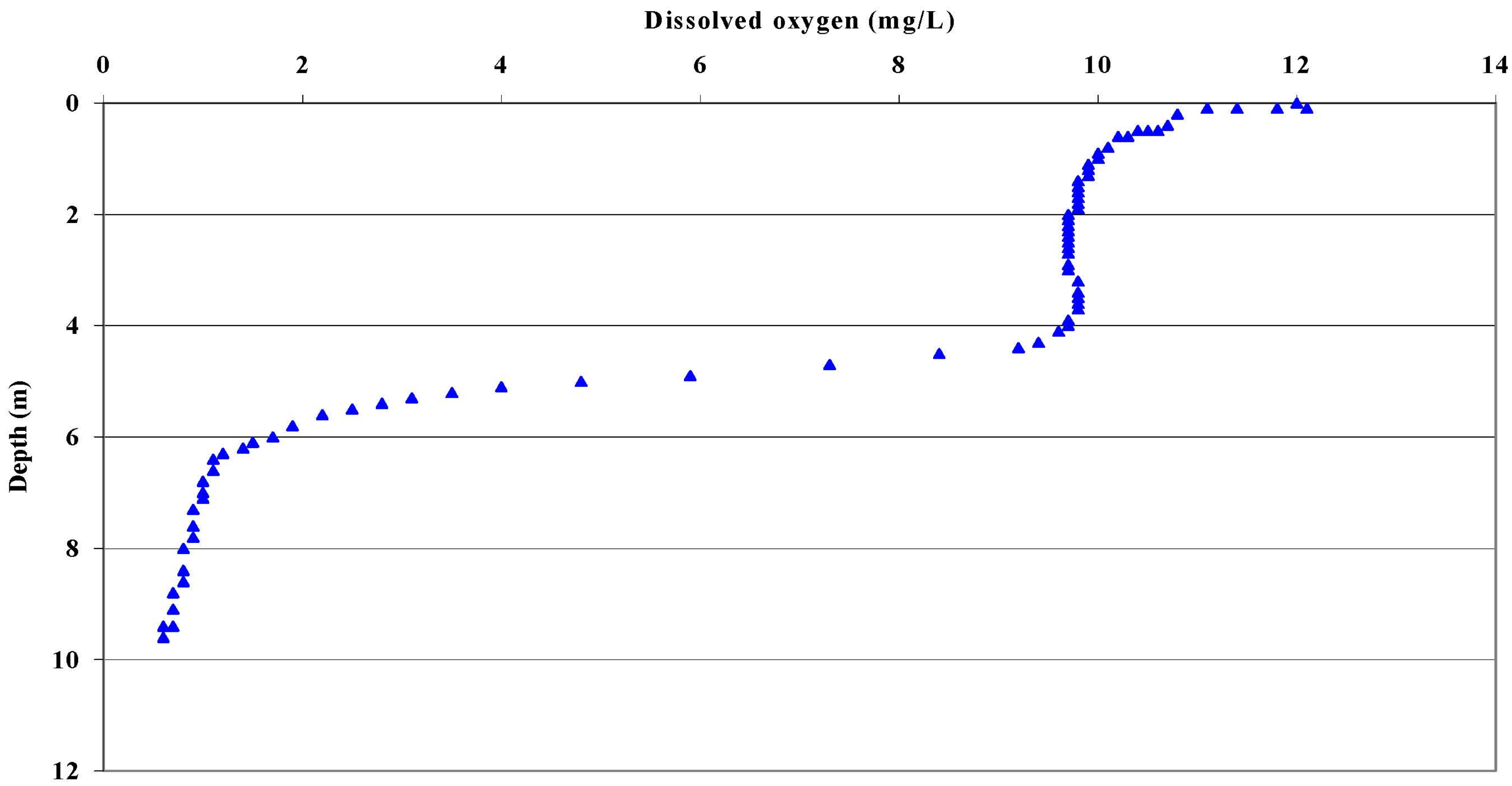

Figure 13 Vertical profile of oxygen concentration in the southern lake in the YNEOC during early May 1999. The profile shows a well-oxygenated upper mixolimnion and a very sharp decrease in oxygen concentrations through the chemocline, dropping approximately 0.6 mg/L in the monimolimnion. This represents approximately 6 per cent saturation. At the surface the oxygen level is supersaturated to approximately 120 per cent saturation as a result of photosynthetic activity.

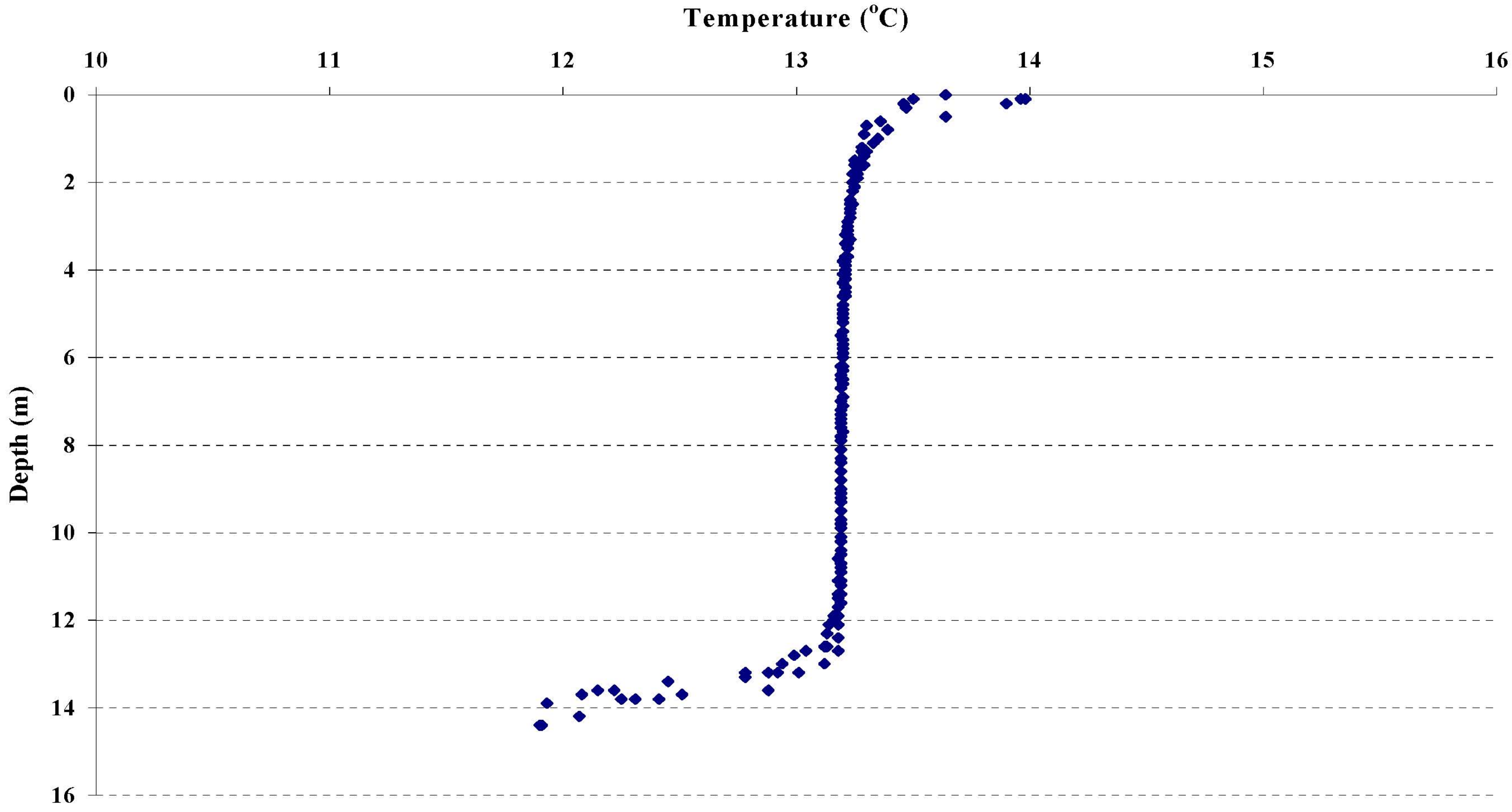

Figure 14 Vertical temperature profile in the northern lake in the YNEOC during early May 1999. As with the oxygen profile, the data shows a well-mixed mixolimnion with a slight surface warming. At the chemocline the temperature decrease sharply.

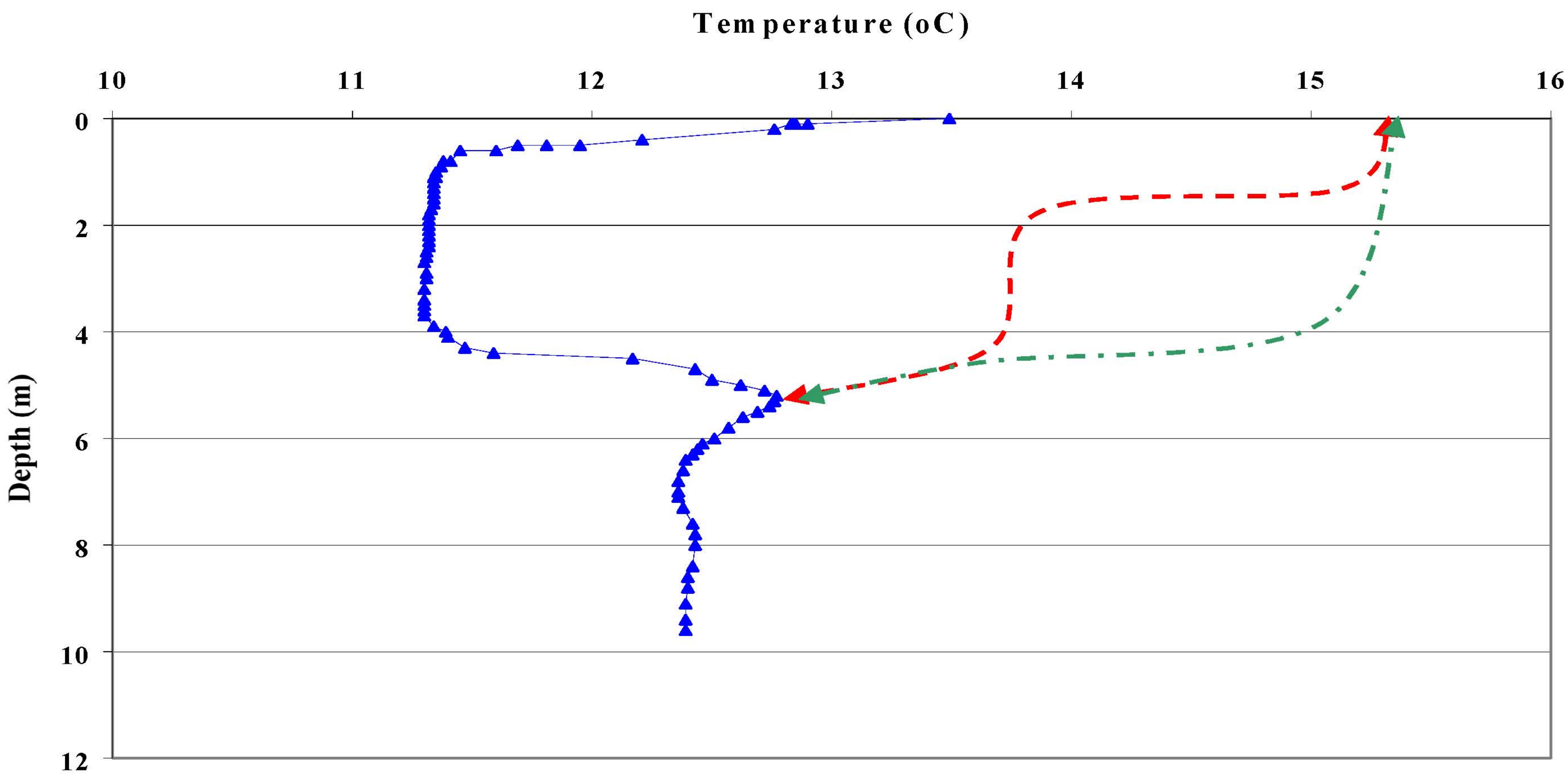

Figure 15 Vertical temperature profile in the southern lake in the YNEOC during early May 1999. The temperature in the mixolimnion has dropped approximately 2^{o}C below that of the mixolimnion in the northern lake, and is also cooler than the water in the monimolimnion. The warmer top layer of the monimolimnion shows the evidence of earlier warmer temperatures that prevailed in the mixolimnion during summer. The broken lines suggest two possible shapes of the summer temperature profile.

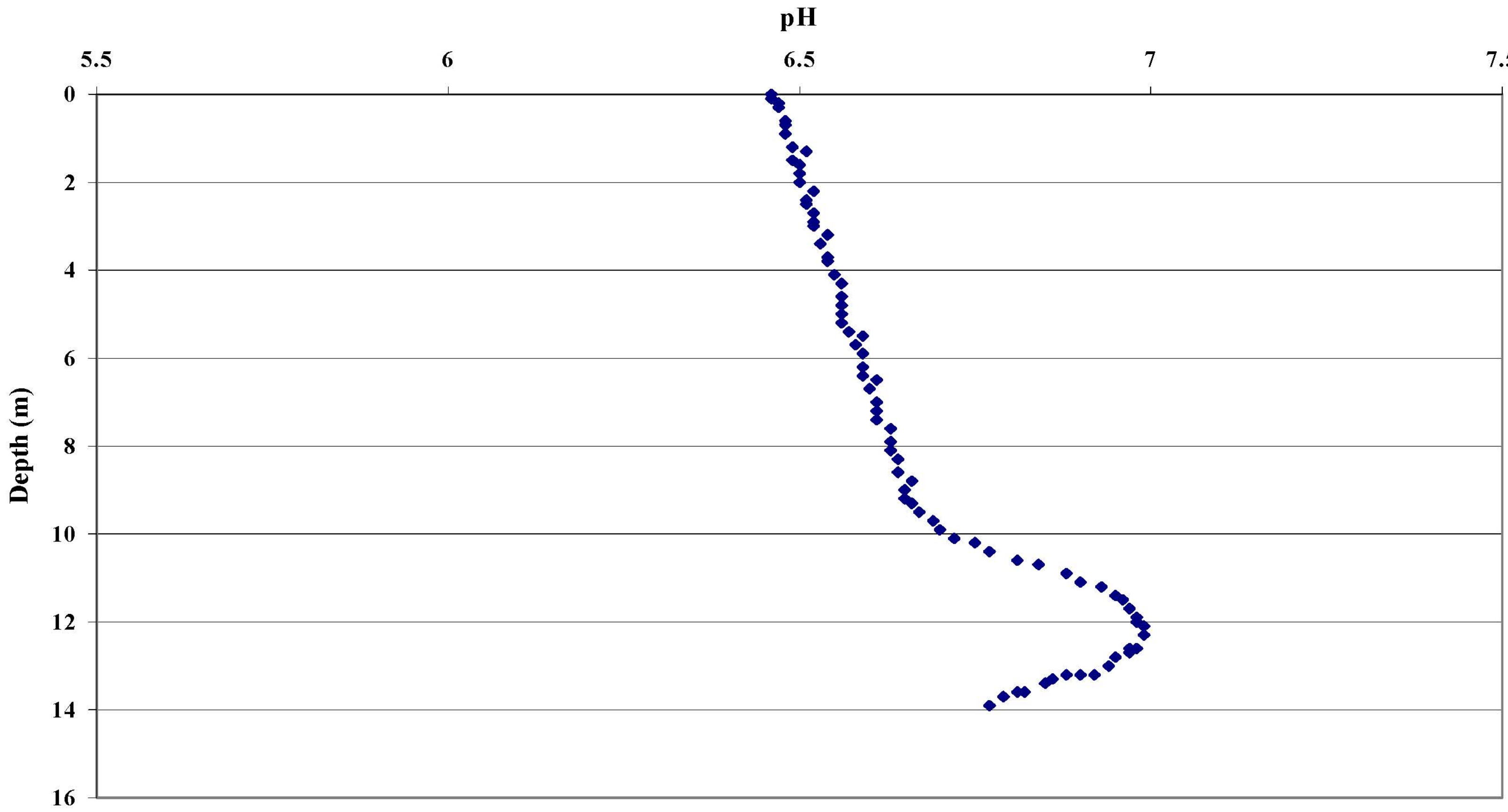

Figure 16 Vertical pH profile during May 1999 in the northern lake of the YNEOC. This pH profile when taken in conjunction with the oxygen profile is suggestive of some form of photosynthetic chemotrophic bacterial reduction process. Below the chemocline the lowering of the pH is suggestive of bacterial oxidation resulting in increased concentrations of carbon dioxide

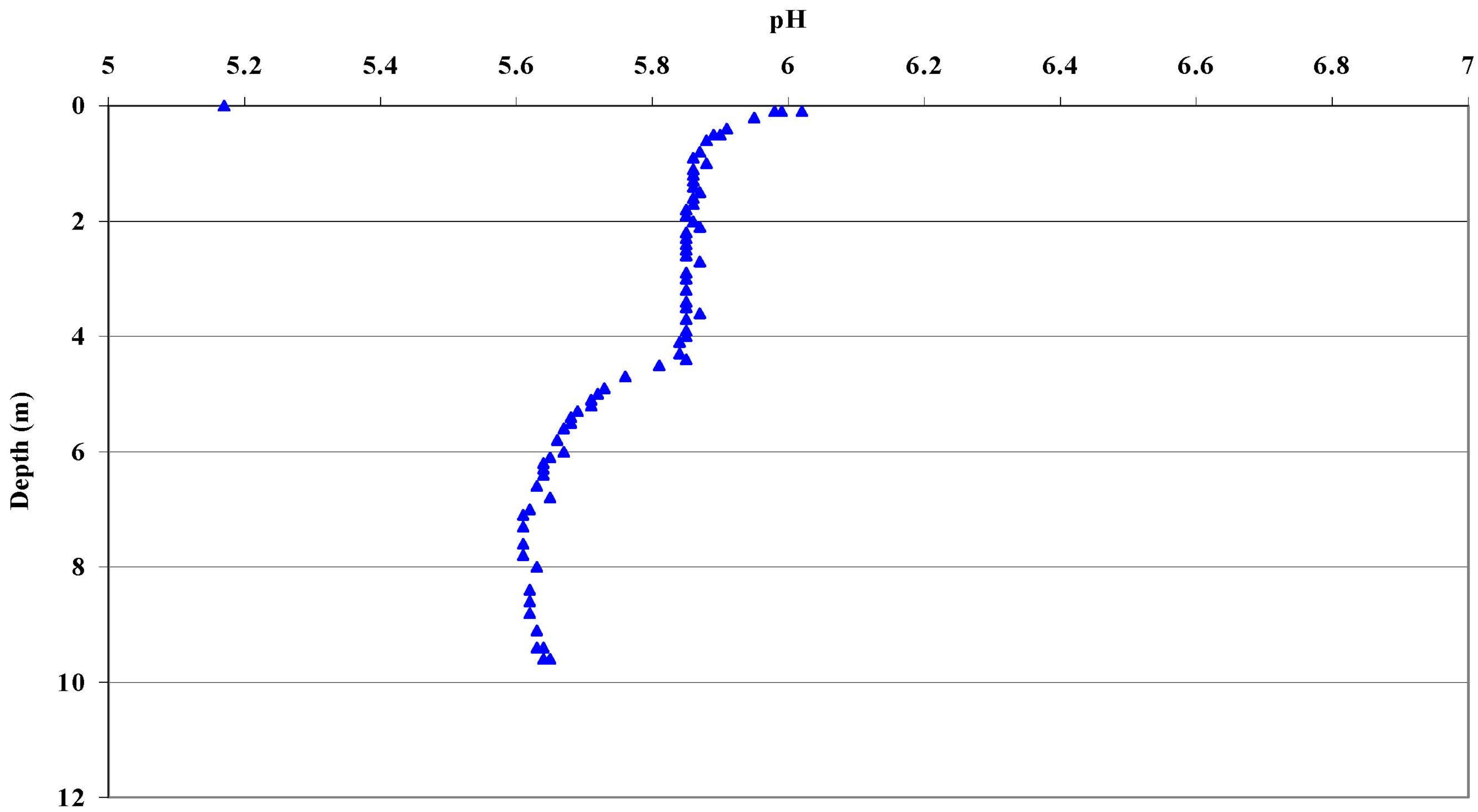

Figure 17 Vertical pH profile during May 1999 in the southern lake of the YNEOC. The higher pH at the surface most likely results from the depletion of carbon dioxide due to the high primary productivity previously shown by the supersaturated oxygen in the water. Below the chemocline the lowering of the pH is suggestive of bacterial oxidation resulting in increased concentrations of carbon dioxide.

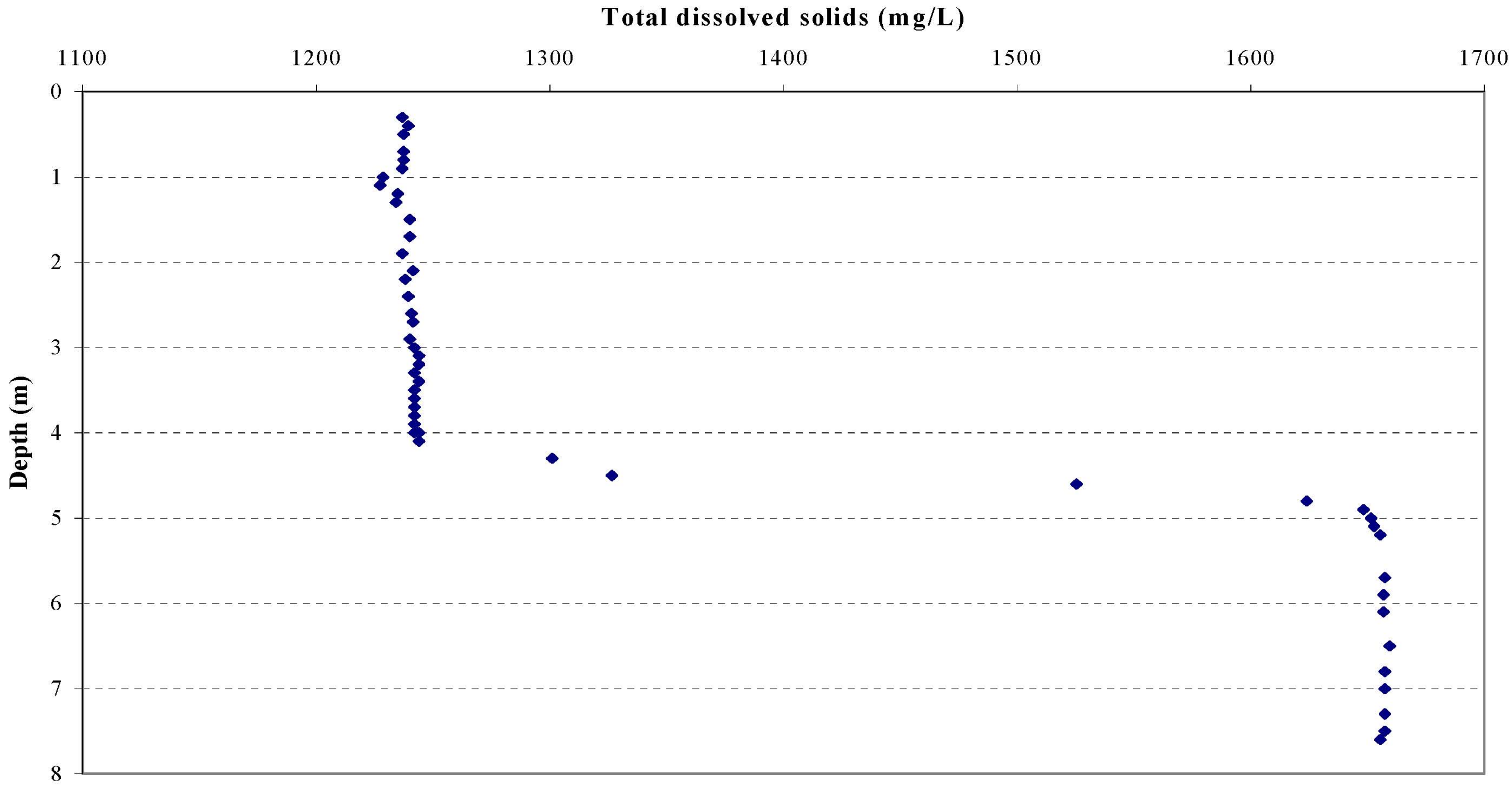

Figure 18 Total dissolved solids concentrations as calculated from the conductivity measurements taken in the southern YNEOC lake during August 1999. The difference in salinities between the mixolimnion and the monimolimnion is of the order of 450 mg/L of dissolved salts as previously observed during May 1999.

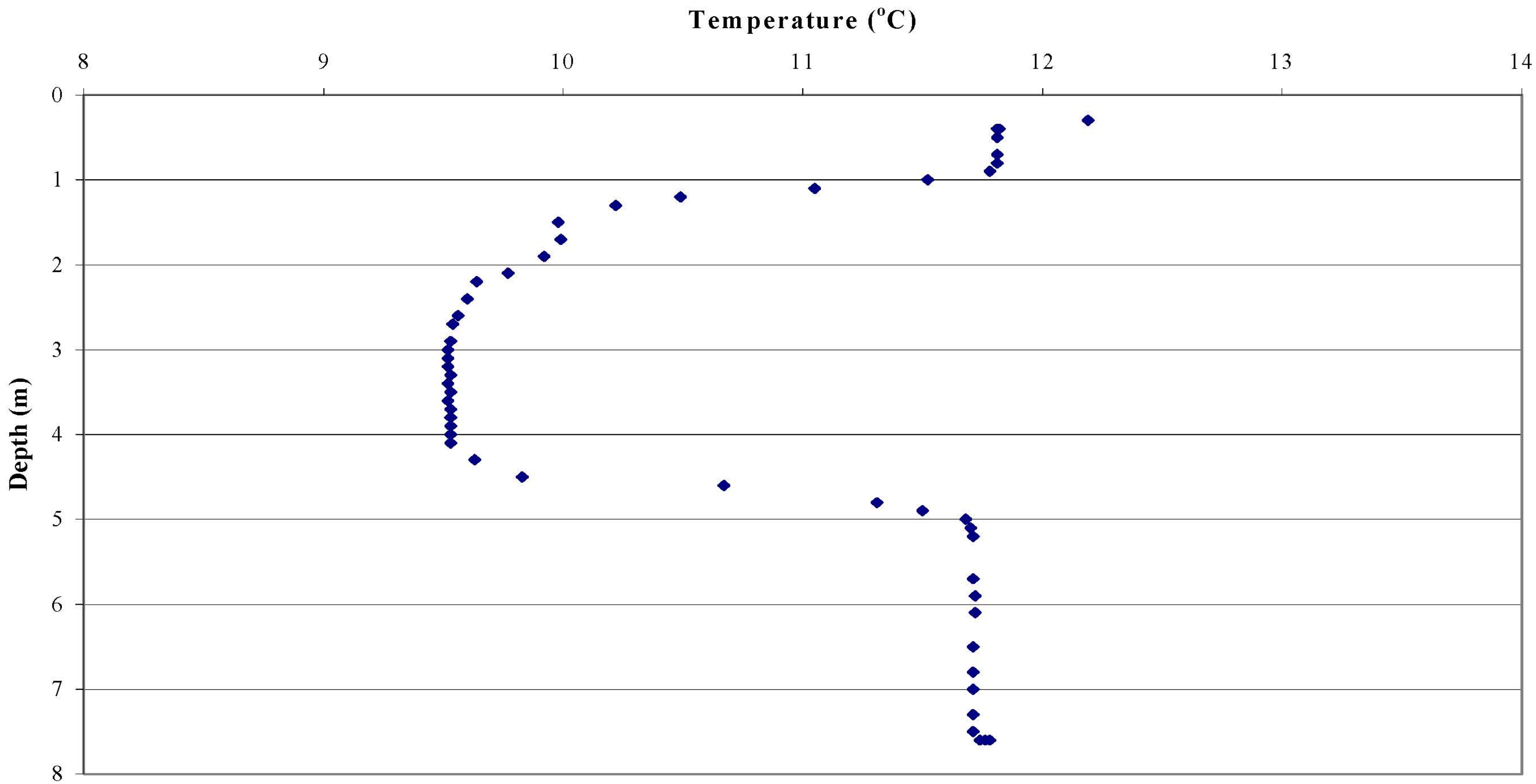

Figure 19 Vertical temperature profile in the southern lake in the YNEOC during early August 1999. The temperature in the mixolimnion has dropped approximately 2°C from that during May (Figure 15). The temperature of the monimolimnion has decreased by only about 0.75°C since May. The warmer top layer of the monimolimnion has disappeared, leaving no evidence of earlier warmer temperatures that prevailed in the mixolimnion during summer. The thermal stratification in the mixolimnion appears to have intensified since May.

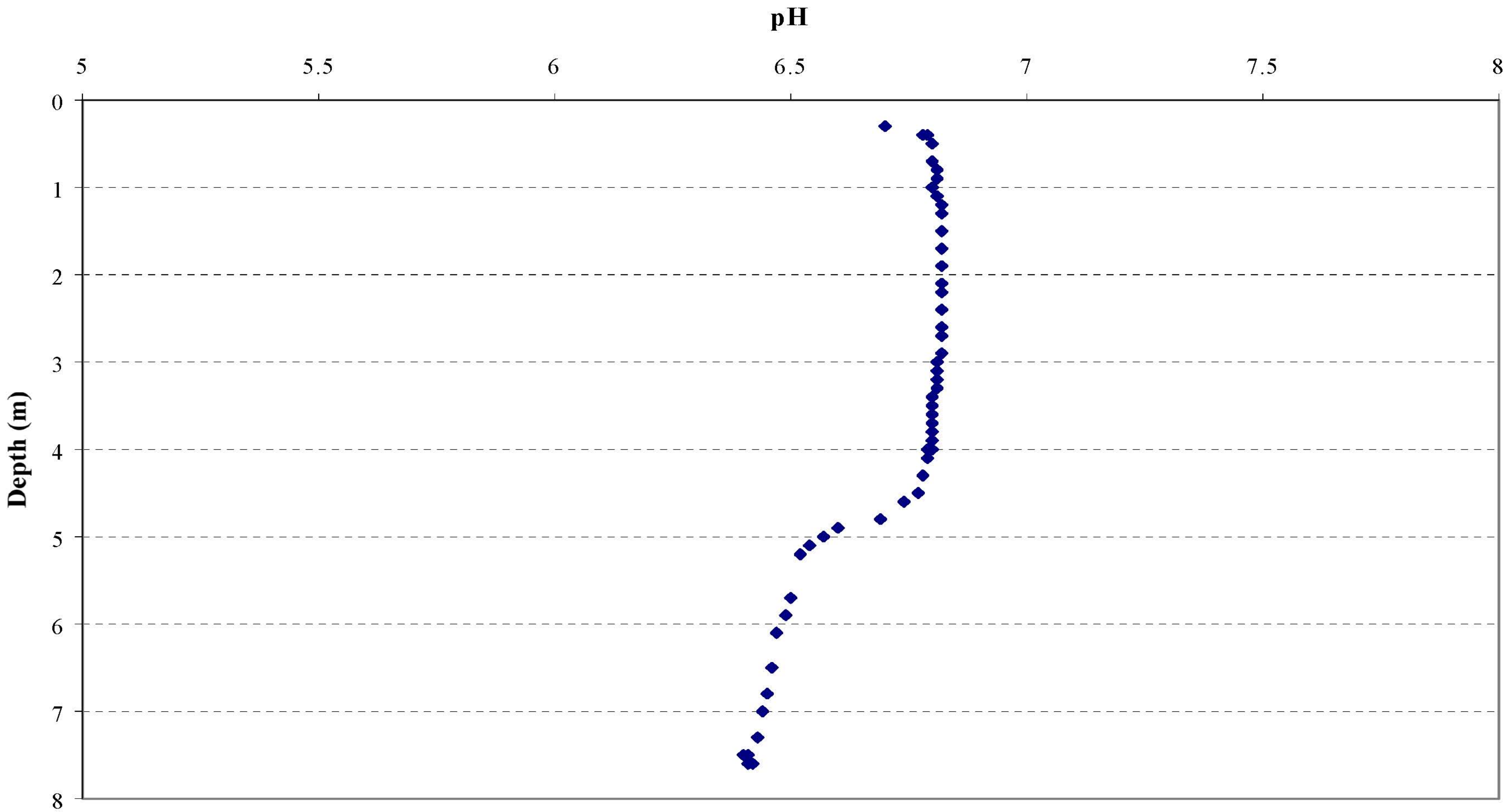

Figure 20 Vertical pH profile taken in August 1999 in the southern lake of the YNEOC. The higher pH at the surface most likely results from the depletion of carbon dioxide as a result of the high primary productivity previously shown by the supersaturated oxygen in the water. Below the chemocline the lowering of the pH is suggestive of bacterial oxidation resulting in increased concentrations of carbon dioxide.

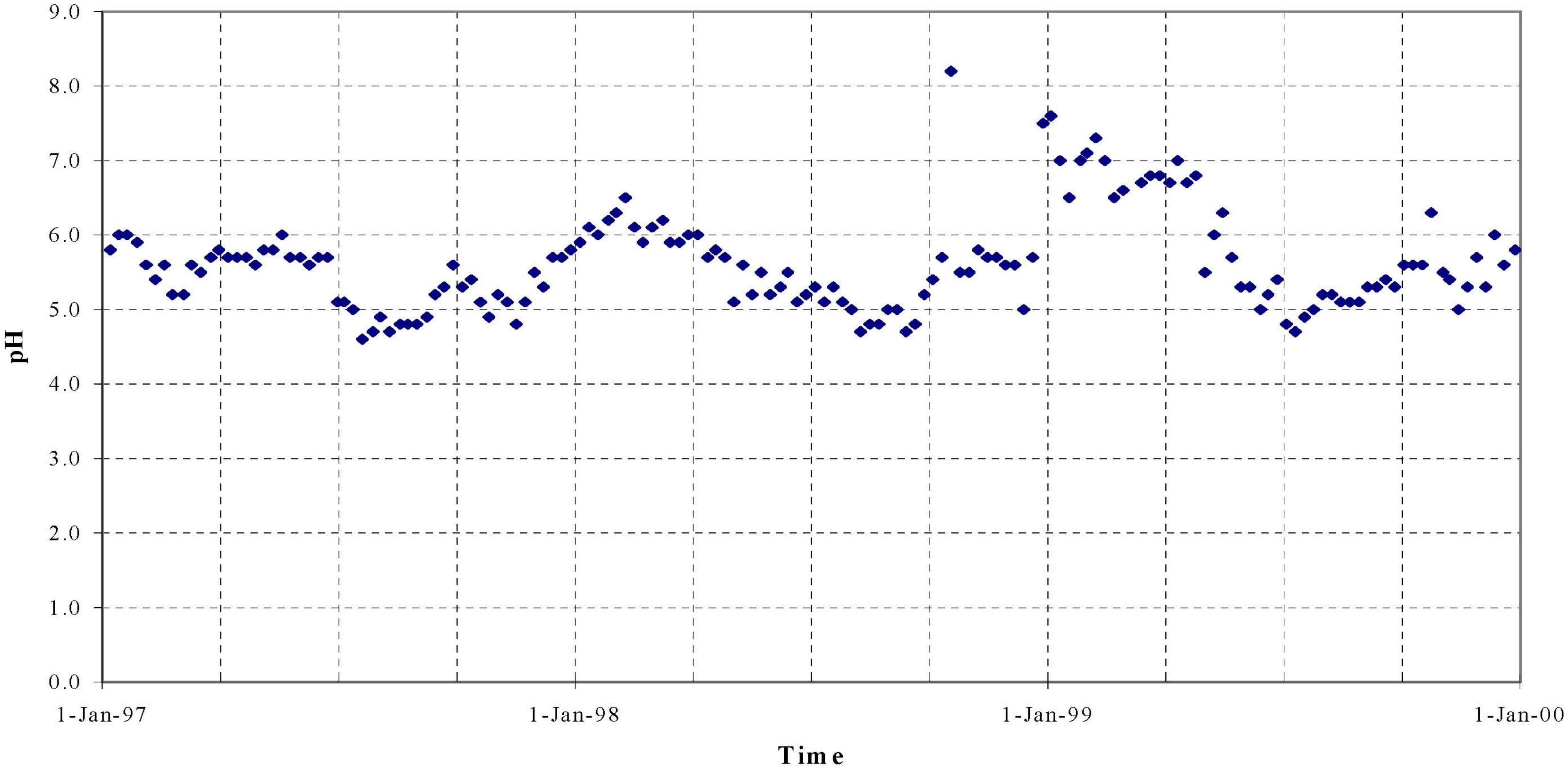

Figure 21 Loy Yang overburden dump drainage pH during for the years 1997-99. There is little evidence of a serious acid drainage from the dump. The seasonal pattern of lower pHs during winter-spring coincides with the wetter months of the year. This could be an indication of some low level oxidation occurring during the drier months with acid flushing during the wetter months.

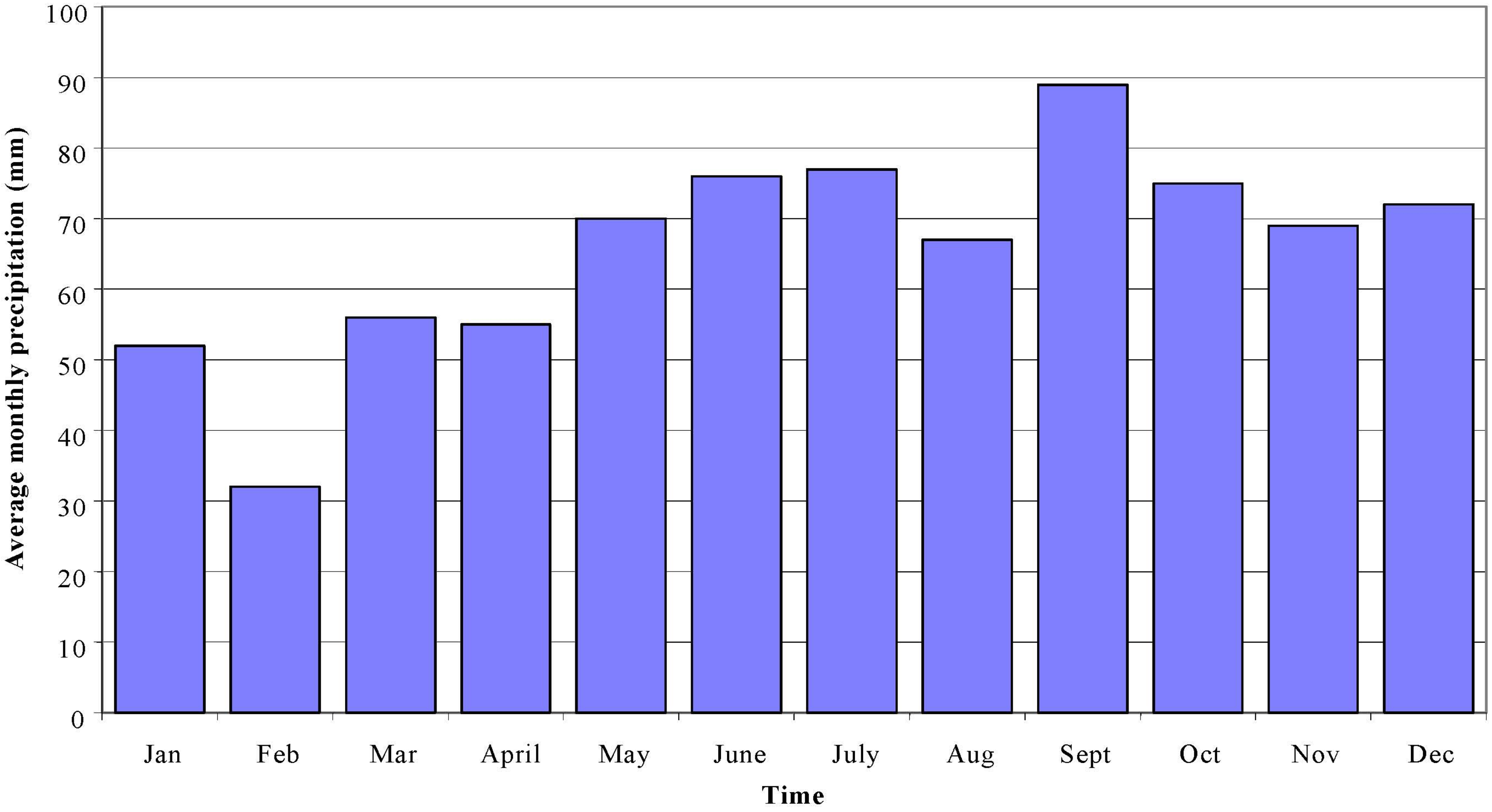

Figure 22 Average monthly rainfall at Loy Yang. These data, together with the overburden run-off pH data suggest that the run-off pH is lowest at times of highest rainfall.

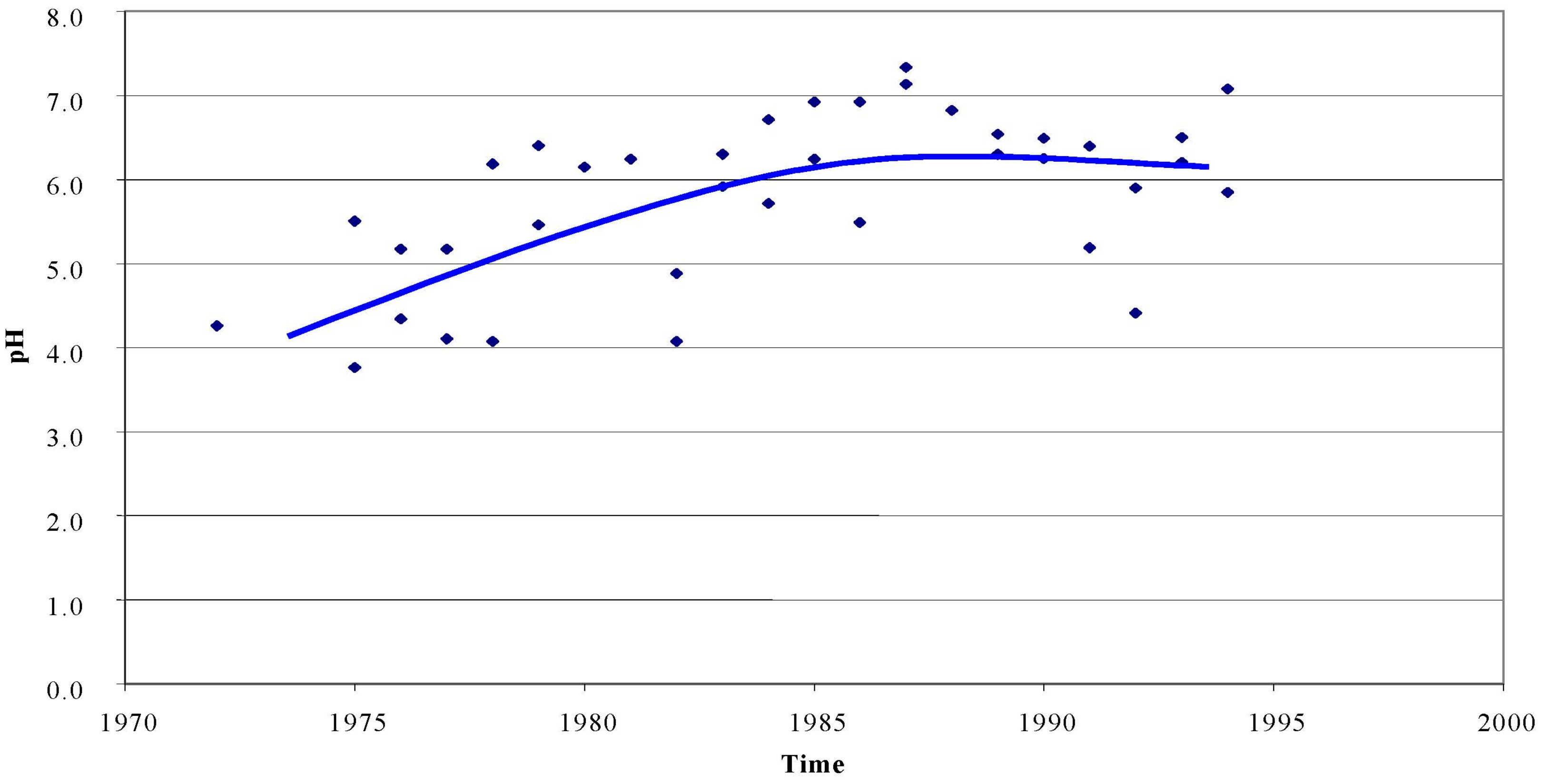

Figure 23 Hydrogen ion concentrations averaged over 6-monthly blocks for the period 1971 to 1995. The effect of the rehabilitation works in the overburden areas is clearly visible, with the pH trend in the run-off from the catchment containing the overburden dumps rising from around 4 in the early 1970s to around 6.5 by the mid-1980s.

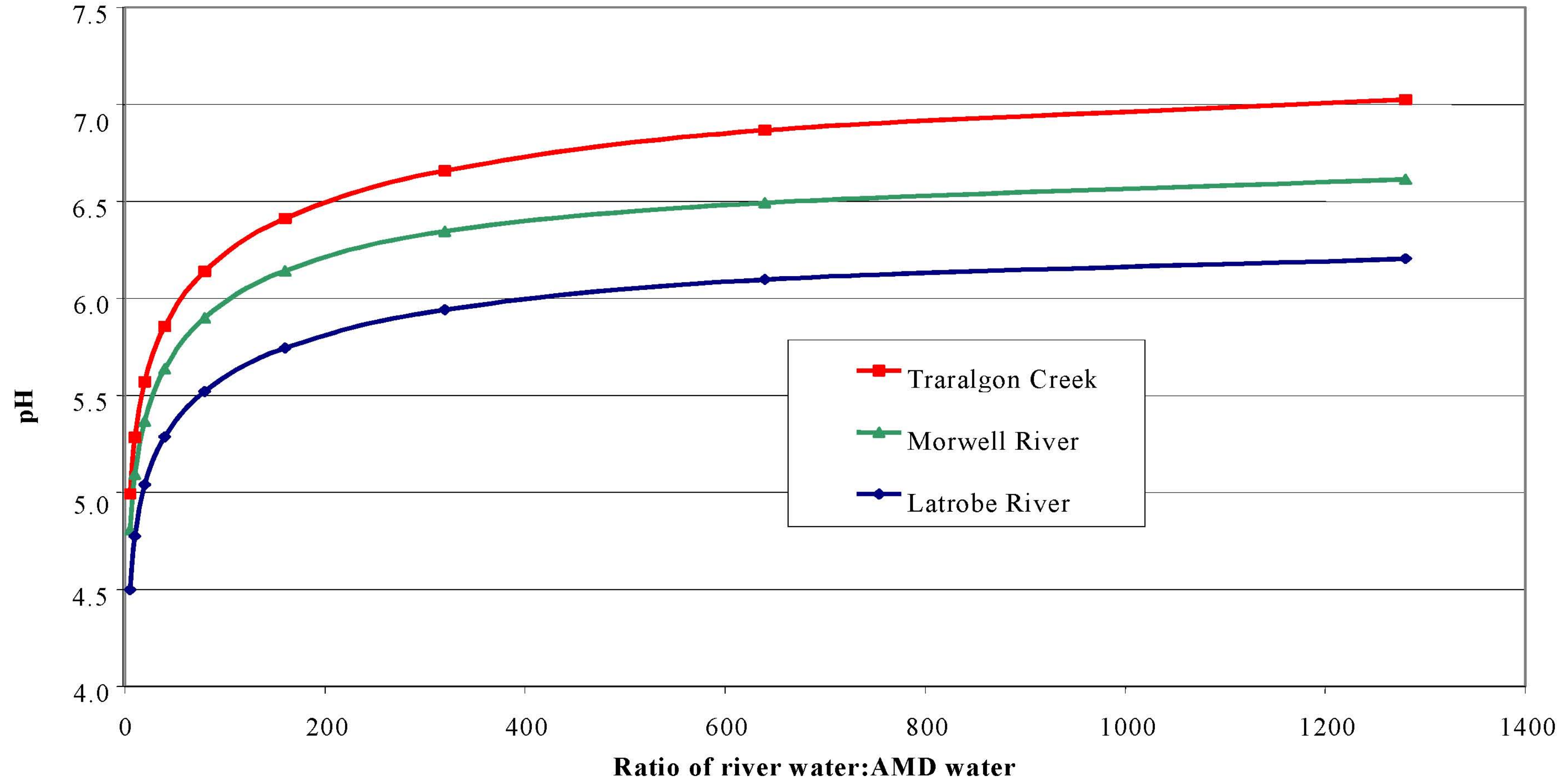

Figure 24 Simulation showing the final pH of lake supply water assuming various extents of mixing of acid mine drainage with water from the Latrobe River, the Morwell River and Traralgon Creek. It can be seen that the smallest stream, Traralgon Creek has the highest buffering capacity. The largest stream, the Latrobe River, has the poorest buffering capacity. The characteristics of the acid water used in this simulation are presented in Table 13.

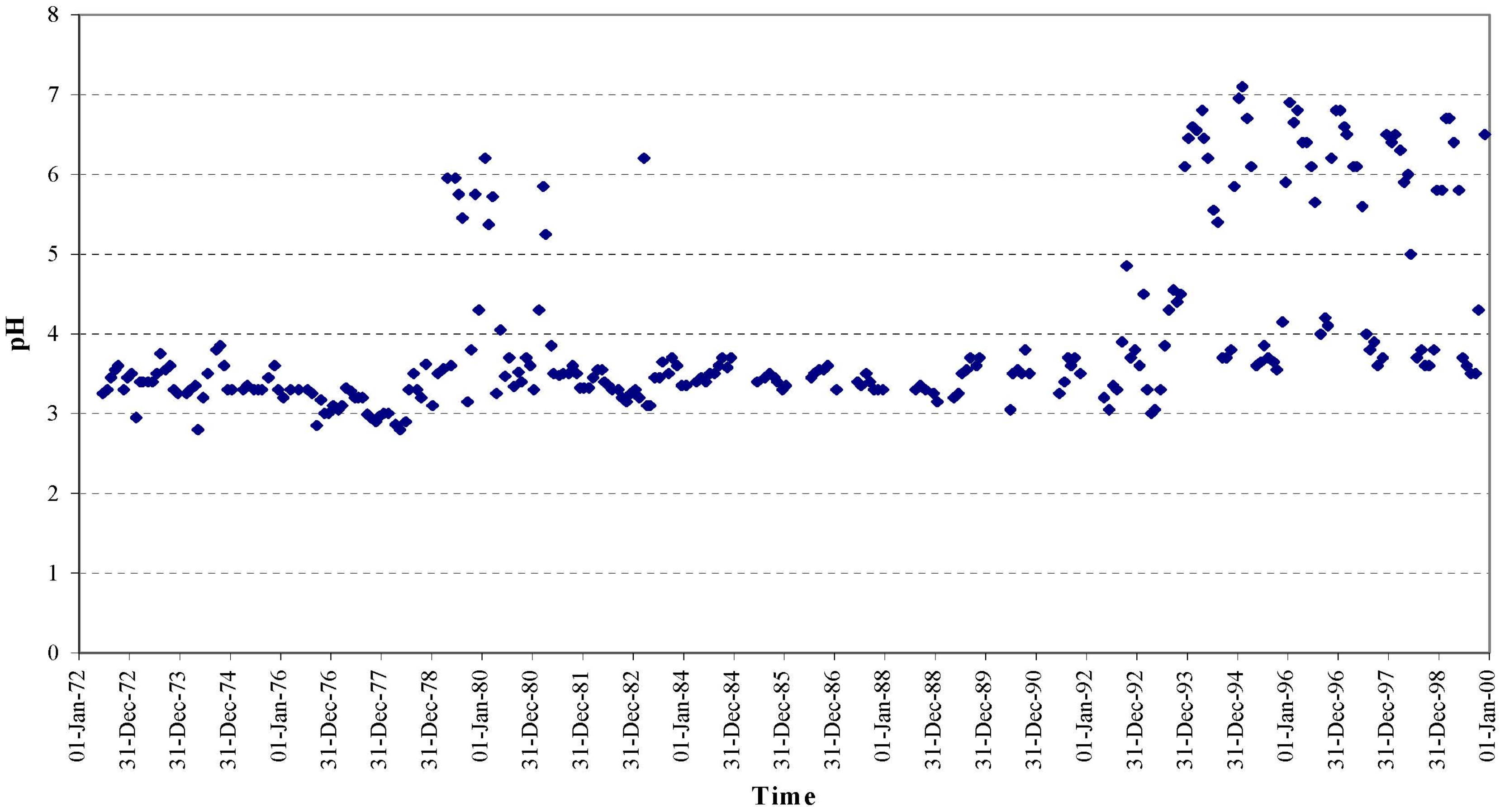

Figure 25 Historical data showing pHs in Marshy Creek upstream from the Anglesea Open Cut during the period 1972 to 1999. See text for discussion.

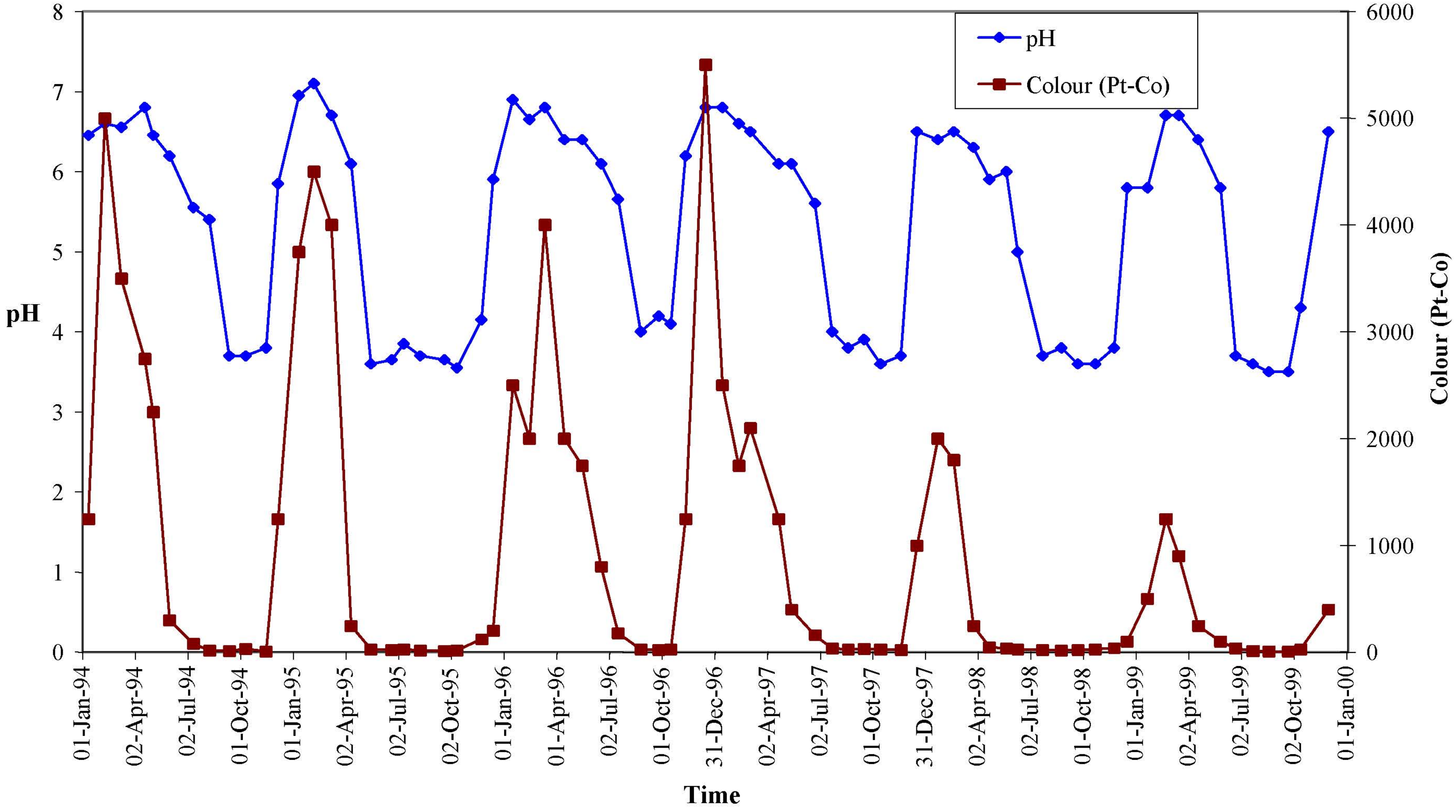

Figure 26 Historical data for Marshy Creek at Anglesea showing the seasonal relationship between pH and colour for the period 1994 to 1999. The elevated colour and neutral pHs arise at times of low seasonal flow in the creek.

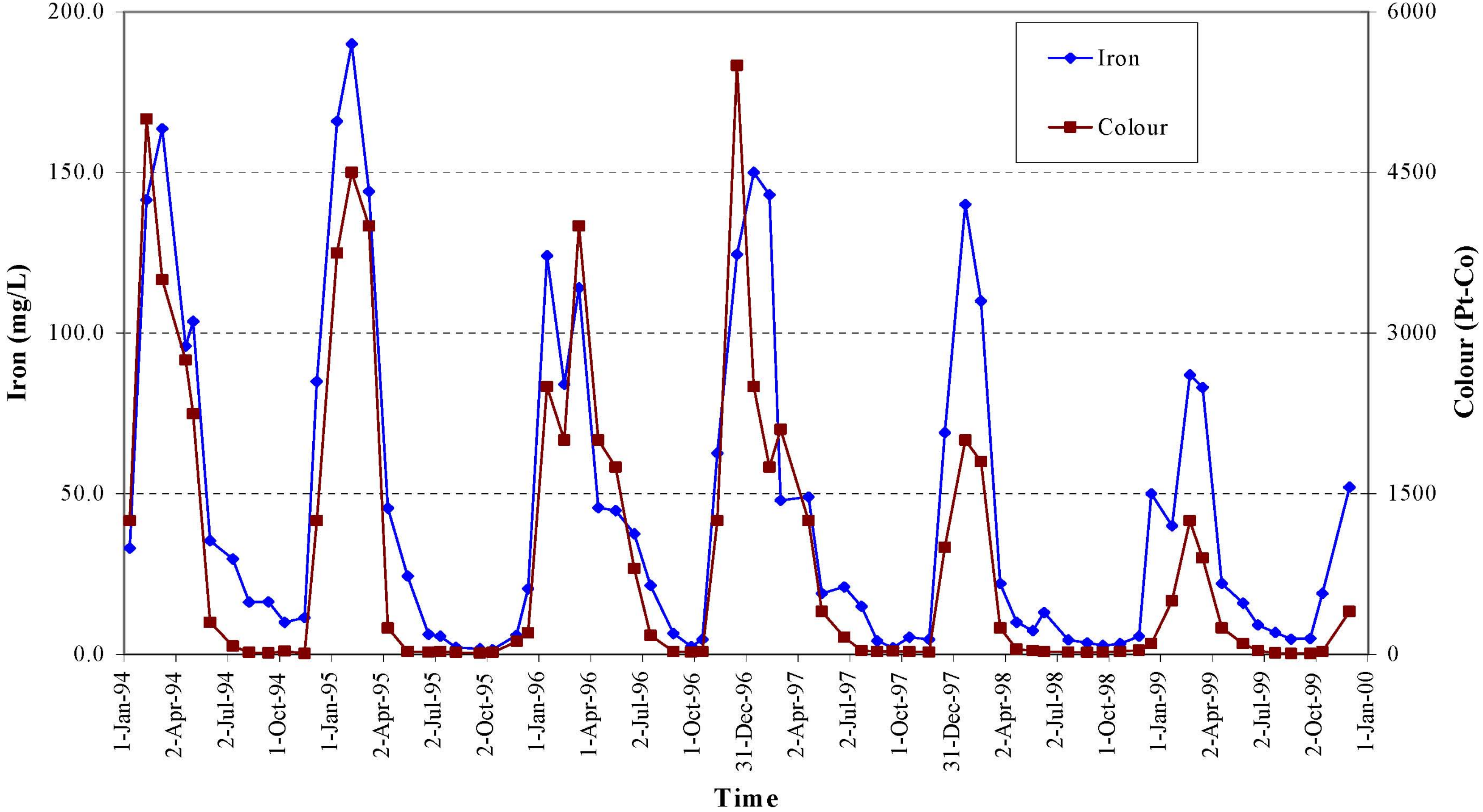

Figure 27 Historical data for Marshy Creek at Anglesea showing the seasonal pattern of total iron and colour in the creek between 1994 and 1999. The elevated colour and iron level occur at times of low seasonal flow in the creek.