Final Report Project 3.3

Hydrological and geochemical processes and closure options for below water table open pit mines

August 2021

crctime.com.au

CRC TiME
Transformations in Mining Economies

Australian Government
Department of Industry, Science, Energy and Resources

AusIndustry
Cooperative Research Centres Program

PROJECT PARTNERS:

ISBN 978-1-922704-02-3

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Pit lake at Broad Arrow. Copyright Department of Mines, Industry Regulation and Safety, Western Australia

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1. Executive Summary

Post-mining landscapes will always be different to pre-mining landscapes. The challenge is to understand the impacts of mining activities, and how these impacts can be altered or ameliorated by management. Optimising pit closure outcomes for water quantity, quality and dependent-ecosystems and environments requires detailed knowledge of groundwater and surface water systems and their connectivity to potential pit lakes.

Dewatering of below – water table mine pits is necessary to create a dry mining environment. Mine dewatering lowers the regional water table surrounding the mine pit, and this zone of reduced water table can extend over tens of kilometres. After the cessation of mine dewatering, the area surrounding the mine that is subject to reduced groundwater levels will continue to increase, before eventually decreasing. Thus, it is possible that ecosystems that are not affected by drawdown during the mine's operational life might still be affected after mine closure. Accurate prediction of water table drawdown after mine closure requires detailed information on the aquifer system surrounding the mine, including geological structures that might act as barriers to groundwater flow. Such information is required beyond the region that is directly impacted by water table drawdown during mining operations.

When mine dewatering ceases, the water table near the mine pit will begin to rise. However, the water level may not stabilise for tens or hundreds of years and may stabilise at a level that is different from the original water table level. If the pit is not backfilled and not connected to a river network, then if evaporation from the pit lake exceeds precipitation and runoff into the pit, the final lake level will be below the original water table level. In this case, the regional water table will also remain below the original water table, and this may have long-term implications for groundwater-dependent ecosystems. If pits are connected to a river network, then rapid filling of the pit may occur, but the groundwater level may still take many years to recover.

The water quality of pit lakes will be largely determined by the rate and chemistry of groundwater flowing into the pit, runoff from the pit walls and lake catchment, the time required for the pit to fill, and the evaporation rate. Pit lake water quality may also be affected by the composition of waste rock and its proximity to the pit. For partially backfilled pits, in-pit storage of waste rock will also affect pit lake water quality. If evaporation exceeds precipitation and runoff into the pit, then the salinity of the pit lake will increase over time. Concentrations of other dissolved solutes will also increase over time. Potential uses of pit lakes will be determined by the water quality and by access and community safety issues. For fully backfilled pits, groundwater quality in the vicinity of the mine will be affected by the chemical composition of the backfill materials that are used (including any waste rock), and infiltration properties of the backfill materials.

Understanding pit lake evaporation rates, lake stratification cycles and how surface water and groundwater inflows to the pit changes over time, are essential for accurate prediction of the final pit lake water level, the time for the pit lake water level to stabilise, and the development of pit lake water quality. Changes in water flows over time due to landscaping and revegetation of pit surrounds need to be considered, together with changes over time in the chemical composition of waste rock leachates and runoff from waste rock and pit walls. The transient dynamics of pit lake hydrology are often overlooked. Almost all pit lakes will be terminal lakes immediately following closure, with long-term dynamics often take many years or even hundreds of years to develop. Even then, seasonal or inter-annual changes in rainfall and evaporation can result in lakes fluctuating between terminal, and flow-through and net groundwater recharge conditions.

Prediction of post-closure pit lake water levels are often based on numerical groundwater models, but few of these models are linked to validated numerical lake models that incorporate realistic estimates of pit lake evaporation. Pit lake models that are not coupled to groundwater models often make simplifying assumptions about groundwater inflow to the lakes and how this changes with time. The importance of such simplifying assumptions is rarely tested. How climate change impacts interactions between pit lakes and regional groundwater has rarely been explored. Many pit lake water quality models do not consider changes in groundwater inflows and other water balance components over time, or changes in the chemical composition of surface runoff.

Several innovative approaches are suggested for reducing the environmental footprint of mines and producing better environmental and economic post-closure outcomes and would benefit from further investigation. These include (i) the use of engineered barriers to limit groundwater connections between mines and adjacent ecosystems, (ii) managed aquifer recharge during mine operations and/or diversions of river water into pits post-closure to enhance water table recovery, (iii) modification of pit backfill, revegetation, and evaporation to achieve desired pit water levels, (iv) amendment of pit backfill materials to reduce oxygen levels and the development of acidic conditions in backfilled pits, and (v) the use of bioremediation approaches to improve pit lake water quality. None of these approaches will be suitable at all mine sites, and evaluation of their likely effectiveness in different environments is urgently required.

Several recommendations are made to improve our understanding of the interaction between pit lakes and the surface water and groundwater systems, and to better assess the effectiveness of different management approaches. Recommendations cover better understanding of hydrological processes, improved modelling approaches, evaluation of innovative approaches for improving water quantity and quality and data sharing.

1. Carry out generic modelling and modelling of existing mines to examine how groundwater and pit lakes interact post mine closure. This modelling should include predictions of rates of water level recovery, steady state pit lake water levels, and time for stable pit lake water levels to develop, and how this is affected by pit geometry and aquifer parameters.
2. Establish best practice models for prediction of evaporation rates from pit lakes.
3. Investigate how pit lake models can be linked with groundwater and surface water models. Such links are essential for accurately predicting how pit lake water levels and water chemistry change over time.
4. Establish a guidance document detailing advantages and limitations of different numerical models that can be used during mine closure planning, to predict the post-closure waterscape evolution.
5. Carry out generic modelling to examine the potential for managed aquifer recharge, rapid filling of pits with water and low permeability grout walls to improve mine closure options. Apply this modelling to demonstration case studies that can be validated with field data, exploring potential management options.
6. Carry out pit lake water balance and water quality modelling to assess potential beneficial uses of pit lakes in different environments. This should include assessment of the effectiveness of different geochemical interventions to improve pit lake water quality.
7. Long-term monitoring of pit lake dynamics and lake, groundwater and surface water quality and ecology should be included in closure plans. The value of using monitoring data from nearby existing pit lakes to evaluate model predictions for new mine pits should be assessed. More case studies of innovative closure options should be publicly documented and full-scale pilot studies should be established.

2. Introduction

2.1 The magnitude of the problem

Understanding the magnitude of mine pit lake issues in Australia is challenging, as there is no national inventory of mines across Australia (Unger et al., 2012), and datasets created based on state-based information are incomplete, due to differences in data collection in each jurisdiction (Campbell et al., 2017, Unger et al., 2012). However 2017 estimates by the Australia Institute (Campbell et al., 2017) suggest there are between 460 and 2944 mines in operation, depending on the definition used and whether individual mines within larger mining projects were counted. While the proportion of surface mines cannot be inferred from the national data, data from Queensland suggested about 80% of operational coal mines (total 52) were surface mines, and about 35% of operational non-coal mines (total 1148) were surface operations. Nationally, between 206 and 972 mining operation were listed as on care and maintenance, and around 60,000 mines had been abandoned across Australia. In contrast there were very few examples of mines that were fully closed and relinquished, with only 22 identified across Australia, with 18 of those in South Australia (Campbell et al., 2017). In Western Australia, it has been estimated that there are approximately 2000 mine pits, with more than half with the potential to become pit lakes, and 200 operations operating below the water table (DMP EPA, 2015). The proportion of mineral production from open cut mines has increased over time. For black and brown coal, for example, the proportion derived from open cut mines has increased from less than 30% prior to the 1940s to more than 70% since 2000 (Mudd, 2010). There is also a trend of more open cut mines operating below the water table.

Pit areas of individual mines vary widely but can be up to tens of square kilometres in size (Werner et al., 2020). Kennecott Copper Mine (Utah, USA) one of the largest mines in the world, is approximately 4 km wide and 1.2 km deep, and the Chuquicamata copper mine in Chile is approximately 4.5 km wide and 0.8 km deep. In Australia, the Fimiston Open Pit, near Kalgoorlie, is 3.5 km wide, 1.5 km wide and 550 m deep (Bungard et al., 2016), Newmont's Boddington gold mine is 4 km long and 1 km wide, and expected to ultimately be 700 m deep (Big Dog Hydrogeology, 2018), and Whaleback mine, Pilbara, is expected to have final dimension of 5.5 km x 2.2 km and a depth of 500 m (Kumar et al., 2009). In contrast, strip mining that is often used in coal mining, as found in the Bowen Basin and the Hunter Valley, typically results in many smaller shallower residual pits at each mine.

2.2 Regulation and management frameworks for mining and development of pit lakes

Regulation of mining activities in Australia is primarily undertaken at the state level, particularly related to granting mining exploration and mining title rights, and the environmental and OHS operations of mines. In all states, a mine closure plan must be submitted as part of the mining proposal application. Typically, statutory requirements require identification of legal obligations for rehabilitation and closure, baseline data and closure risk assessment, and proposed post-mining land uses agreed with key stakeholders. Proponents of new mining projects in Australia, are now required to provide detailed information on the impact of mining on groundwater systems both during and after mining, and develop a conceptual model for long-term changes in water quality in a mine pit lake, including predictive modelling if necessary (Johnson and Wright, 2003). In addition to state legislation, the Commonwealth Environment Protection and Biodiversity Conservation Act requires that any coal seam gas and large coal developments that have the potential to have a significant impact on water resources or other matters of national environmental significance (e.g., threatened plants and animals) are also referred to the Commonwealth for consideration.

Australian and international guidelines exist for mine site drainage water quality prediction or mine water management, and include:

- Best practice environmental management in mining, Environment Australia (EA, 2002);
- Mine site water management handbook, Mineral Council of Australia (MCA, 1997);
- Mining impacts on the fresh water environment: Technical and managerial guidelines for catchment scale management, "Environmental Regulation of Mine Waters in the European Union" programme, ERMITE (Younger and Wolkersdorfer, 2004);
- The Global acid rock drainage (GARD) guide (INAP, 2009);
- Prediction of water quality at surface coal mines, Acid Drainage Technology Initiative (ADTI, 2000);
- Managing acid and metalliferous drainage (DFAT, 2016);
- Engineering guidelines for the passive remediation of acidic and/or metalliferous mine drainage and similar wastewaters, "Passive in-situ remediation of acidic mine/industrial drainage" programme, PIRAMID (PIRAMID, 2003);
- Evaluating the potential impact of opencast coal minings on water quality: An assessment framework for Scotland (Younger and Sapsford, 2004); and
- Handbook of technologies for avoidance and remediation of acid mine drainage (Skousen et al., 1998).

However, these guidelines do not specifically consider processes affecting pit lake water quality, such as the potential for amelioration of metal concentrations or acidity production by processes in lake sediments.

In Western Australia, the primary concern at mine closure has been ensuring that the mine void is geologically stable and safe to the public. However, recently published guidance has considered the risk assessment of pit lakes to determine whether pit lake backfilling or other remediation strategies, are required (DMIRS, 2020). The focus is on potential issues with pit lakes, in line with an increase in below-water table mining operations in the Pilbara. A risk-based assessment of pit lakes is recommended, identifying pollution sources, pathways and receptors, with the final objectives being the same as for other landforms (safe, stable, non-polluting, capable of sustaining a post mining land use, meets agreed end land use). The WA guidance acknowledges that aspirational end uses (such as a regional lake with recreational or agricultural values) are not always possible, especially in the many arid environments of WA. Final management strategies for a pit lake that requires active remediation (e.g. ongoing water treatment or active pumping of fluids) are discouraged due to the ongoing financial liability. Several of the Pilbara iron ore mines require back filling as the pits are hydrogeologically connected to significant groundwater resources (Johnson and Wright, 2003). Similarly, in Queensland, a pit proposed to be situated wholly or partially in a floodplain must be rehabilitated to a safe and stable landform that is able to sustain post-mining land use (Department of Environment and Science, 2020).

2.3 Scope of review

This review examines issues related to water management in mine closure for open pit mines that extend below the natural water table and hence are dewatered to enable mining operations. As the focus is on issues that are specific to below-water-table open pit mines, most attention is on the impacts of dewatering on the regional groundwater system, the recovery of groundwater post-closure and the formation and evolution of pit lakes if mine pits are not backfilled. We do not specifically review revegetation or landscaping of pits, although these are significant issues for post-closure mine rehabilitation. They are important for water resources if they affect groundwater recharge and runoff into mine pits and are briefly discussed in this context. Restoration of surface water drainage, which may have been interrupted during mine operations is also not specifically discussed. While it is important for the post-closure water landscape, it is not specific to below – water table open pit mines. We also do not consider contamination issues related to infiltration through, or surface runoff from, waste rock dumps, except where these occur within the catchment of the pit, and hence influence pit lake water quality or where open pits are backfilled with waste rock materials. Contamination issues related to discharge of processing water are also not considered.

The rest of the report is divided into five chapters. Chapter 3 is a discussion of the extent of the groundwater system impacted by mine dewatering and how this area evolves post mine-closure. It also discusses potential impacts on groundwater-dependent ecosystems due to water table lowering and presents examples from Australia and overseas. Chapter 4 discusses the hydrology of open and backfilled pits post-mine closure and components of the pit water balance. Chapter 5 examines water quality issues both for backfilled pits and for pit lakes that develop where mine pits are not backfilled to above the natural water table level.

Chapter 6 discusses innovative and integrated strategies for minimising post-closure environmental impacts and the potential for delivering beneficial and economic use from pit lakes where these are part of the post-mining landscape. It discusses strategies for minimising the impacts of water table drawdown on groundwater-dependent ecosystems, both by minimising the spatial extent of drawdown during mine operations, and rapidly replenishing the groundwater system post-mine closure. Chapter 7 highlights knowledge gaps that currently constrain the design, management, and beneficial use of pit lakes in post-mining economies and proposes some future research areas to help reduce those constraints.

3. PIT DEWATERING AND GROUNDWATER DRAWDOWN AND RECOVERY

This chapter discusses how dewatering of mine pits affects the regional groundwater system, and the speed and extent of groundwater recovery following the cessation of pumping. It also discusses possible effects of water table lowering on groundwater-dependent ecosystems and presents Australian and international examples of impacts from existing mining operations.

3.1 Pit dewatering

Dewatering of below – water table mine pits is necessary to create a dry mining environment. For pits in very low permeability environments, where rates of groundwater inflow are low and unlikely to cause slope instability, mine dewatering is often accomplished using in-pit pumps. Water is collected in drains and channels within the pit and directed to low points or sumps from where it is pumped to a disposal point (Preene, 2015). In higher permeability environments, in-pit pumping will not be sufficient to remove inflowing groundwater, and dewatering bores are typically installed around the perimeter of the mine pit. Dewatering bores create a dry mine by lowering the regional water table, sometimes for distances of several kilometres around the pit. After mine closure, the groundwater will begin to return, but complete recovery of the groundwater may take many years or may not occur at all. This can have consequences for ecosystems surrounding the mine site that might be dependent on groundwater.

This chapter focusses on below – water table pits that are located in areas of sufficiently high permeability to require the use of dewatering bores. Volumes of water pumped to dewater these pits vary widely, and are mainly determined by pit size and depth, and aquifer permeability. At the upper end of the spectrum, in the Pilbara, dewatering associated with the Cloudbreak iron ore mine is predicted to be up to 100 GL/y (3000 L/s), and up to 65 GL/y (2060 L/s) for the Roy Hill mine (Roy Hill, 2019). During its first seven years of operation (1997-2004), dewatering rates for the Lihir gold mine, Papua New Guinea, averaged up to 700 L/s (Vogwill et al., 2009). The relatively high dewatering rate for this mine is partly due to its proximity to Louise Harbour, which maintains a high standing water level immediately adjacent to the mine. Average actual or predicted dewatering rates from other mines include 120 L/s for Newmont Boddington gold mine (Big Dog Hydrogeology, 2018), 150 L/s for McArthur River lead and zinc mine, NT, and 40 – 100 L/s for Hillside copper mine, South Australia[1]. Much lower rates of dewatering are required for Cape Preston iron ore mine, Pilbara (9 L/s; Aquaterra (2001)), the Shenhua Watermark coal mine, NSW (6 L/s; Zhao et al. (2017)) and Kookaburra Gully graphite mine (3.3 L/s; Parsons Brinckerhoff (2015)), mostly due to the low permeability of aquifer materials.

Dewatering water is commonly used for mine operations, particularly dust suppression and mineral processing, and can be used to supply staff accommodation and villages. In many cases, the mine site needs consume all dewater, and there is no need for alternative disposal. Where there is excess dewater, it is typically discharged to a surface water body. Dewater may also be used for environmental mitigation, for example to restore or

[1] https://energymining.sa.gov.au/minerals/mining/mines_and_quarries/hillside_project/chronology_of_rex_minerals_hillside_project_application_and_assessment

maintain surface flows, or for artificial recharge to maintain groundwater levels to protect dependent ecosystems (Preene, 2015). Artificial recharge is mostly used when there is no suitable surface water body for disposal or when the environmental impacts of surface water disposal are considered too great.[2] Another, less common form of dewater use is irrigation of agricultural developments[3]. In some areas discharge of mine water to the environment creates new ecosystems and the fate of these post-closure is largely unexplored. In some cases, these new ecosystems may have become culturally important.

3.2 Size and rate of expansion of drawdown cone during mine operations

The area over which groundwater levels are reduced because of groundwater pumping is referred to as the *drawdown cone*. This term is used because drawdown from an individual bore in a uniform aquifer will create a depression in the water table that has a conical shape, with greatest decrease in water level centred on the pumping bore. The term *drawdown* refers to the magnitude of the reduction in groundwater level. The size of the drawdown cone due to mine dewatering is important because this delineates the area that is potentially affected by mining activities. It is usually much greater than the area directly impacted by mining infrastructure or other surface disturbance (e.g., tree clearance). During dewatering, the drawdown cone will increase in size - initially rapidly and then more slowly over time. After cessation of pit dewatering, the area over which groundwater levels have been impacted will first increase before eventually decreasing. Ecosystems within the drawdown cone that are dependent on groundwater may be affected many years after mine closure. It is therefore important to understand the extent of the drawdown cone during mining and how it evolves, and whether and how it eventually recovers after mining ceases.

There are several analytical models that have been specifically developed to estimate the volumes of groundwater required to be pumped to dewater large mines (e.g., Marinelli and Niccoli, 2000). However, these dewatering models only allow calculation of groundwater inflows to the mine when groundwater levels stabilise, and they generally do not predict the rate of expansion of the drawdown cone over time or the subsequent recovery of groundwater levels after mine closure. The Theis equation is widely used in hydrogeology for estimating drawdown of the water table in response to pumping a bore and allows estimation of the evolution of the drawdown cone during pumping and its subsequent recovery after pumping ceases. Although not specifically developed for mine dewatering applications and not without its limitations, the Theis solution provides a reasonable estimate of the spatial extent of the drawdown and the magnitude of drawdown far from the pit in relatively homogeneous environments (Figure 1). [4]

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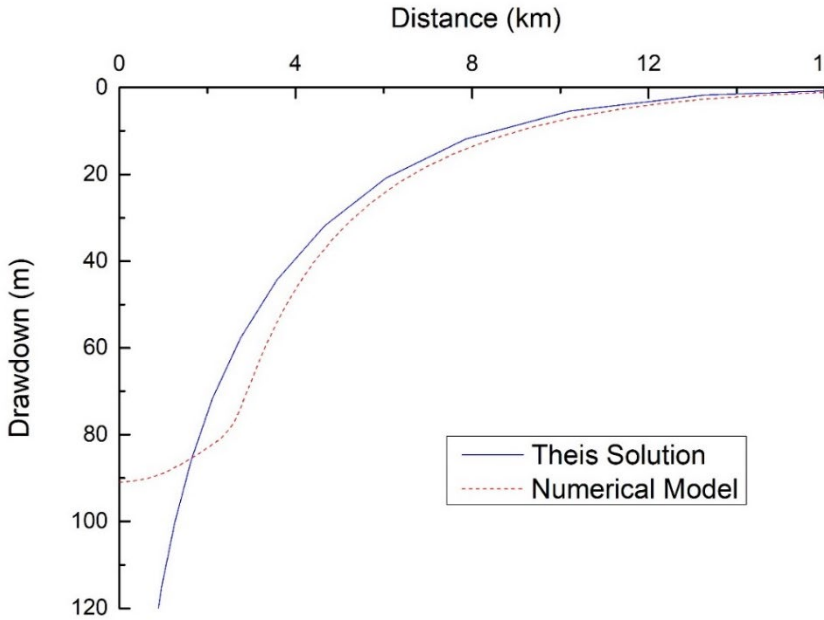
[2] Cloudbreak iron ore mine in the Pilbara is an example of a mine that reinjects excess groundwater abstracted from the dewatering operation (FMG, 2013, Youngs et al., 2012). This reduces any requirement to discharge surplus groundwater to the surface water environment, which could lead to negative ecological and/or cultural consequences, and has the additional benefit of limiting the spread of the drawdown cone. Reinjection enables groundwater levels to be preserved at Fortescue Marsh, a periodically flooded wetland of national significance, located south of the mine site (Windsor et al., 2012).

[3] https://www.epa.wa.gov.au/proposals/hamersley-agriculture-project-s46; Woodie Woodie trial shows mine water a risk for irrigators - ABC News; Pilbara Hinterland Agricultural Development Initiative - Issue 1 | Agriculture and Food

[4] The Jacob-Lohman Equation is another analytical solution that has been used to estimate flow rates to a mine pit (Hanna et al., 1995). However, while the Theis solution assumes a constant pumping rate with time, the Jacob-Lohman solution assumes that the pumping rate is set to deliver constant drawdown at the pit (which results in pumping rates that decrease with time). Both solutions assume a uniform aquifer that extends much further than the area impacted by pumping. They are



Most analytical models represent the aquifer as a homogeneous unit, defined by its transmissivity (T) and storativity (S). The transmissivity of an aquifer is the product of its hydraulic conductivity and thickness, and describes its flow characteristics. The storativity describes how the water volume changes as the water pressure (or water table elevation) changes, and is the key parameter describing the aquifer's water storage characteristics. The rate of expansion of the drawdown cone over time will depend on the rate of pumping and on aquifer transmissivity and storativity. For a given pumping rate, lower values of the ratio T/S result in increased drawdown near the pumping bore but reduced lateral spread of the drawdown cone (Figure 2). In practice, lower values of T/S will enable pit dewatering with lower pumping rates. The lateral spread of drawdown will be related to the drawdown required at the mine site (usually the depth of the mine below the water table), and the transmissivity and storativity of the aquifer surrounding the mine pit.

However, while the Theis solution provides a useful first approximation to the magnitude of drawdown and how this varies with aquifer conditions, numerical models are often preferred for estimating drawdown at specific sites as they can incorporate information on pit geometry and can accommodate irregular aquifer boundaries and spatial variations in aquifer transmissivity, storativity and thickness. Several different numerical models are available, including MODFLOW, SEEP/W, SEEP3D, and FEFLOW, and such models are widely used (e.g., Peksezer-Sayit et al., 2015). However, accurate predictions using these groundwater models require a good knowledge of

approximations that were developed to predict the impacts of pumping from irrigation and town water supply bores, and so make several assumptions which might not be appropriate for mine dewatering and recovery. In particular, they assume that transmissivity is constant, which will not be the case if dewatering reduces the thickness of the aquifer. Recovery calculations using the Theis equation also ignore the presence of the pit itself, and so are only appropriate if the pit is backfilled.

the hydraulic properties of the aquifers in the vicinity of the mines, geological structures in the area, and surface water features that may be recharge sources. This information is rarely available over the region potentially impacted by dewatering. As dewatering proceeds, the drawdown cone will expand, and it is not unusual for it to extend for several kilometres and sometimes tens of kilometres from the mine pit. Drawdown cones due to dewatering of open cut lignite mines in Greece extend for more than 4 – 5 km (Loupasakis et al., 2014), and water table drawdown at the completion of mining (after 21 years) at the Siviour graphite mine, Eyre Peninsula, South Australia is predicted to extend more than 5 km from the mine site (JBS&G, 2018). For the Hope Downs 1 North mine, in the Pilbara, drawdown exceeding 20 m is observed more than 4 km from the mine pit (Cook et al., 2017). Also in the Pilbara, total drawdown from multiple pits in the Roy Hill mine is expected to extend over an area of approximately of 35 km x 10 km at mine closure (Roy Hill, 2018), and at Cape Preston mine, drawdown of more than 0.5 m at completion of mining is expected to extend more than 10 km to the north and south of the mine pit, but less than 5 km to the east and west due to the alignment of geological structures (Aquaterra, 2001). However, the size and extent of drawdown is difficult to directly measure for many existing mines, due to an absence of monitoring bores in areas remote from the mine pit.

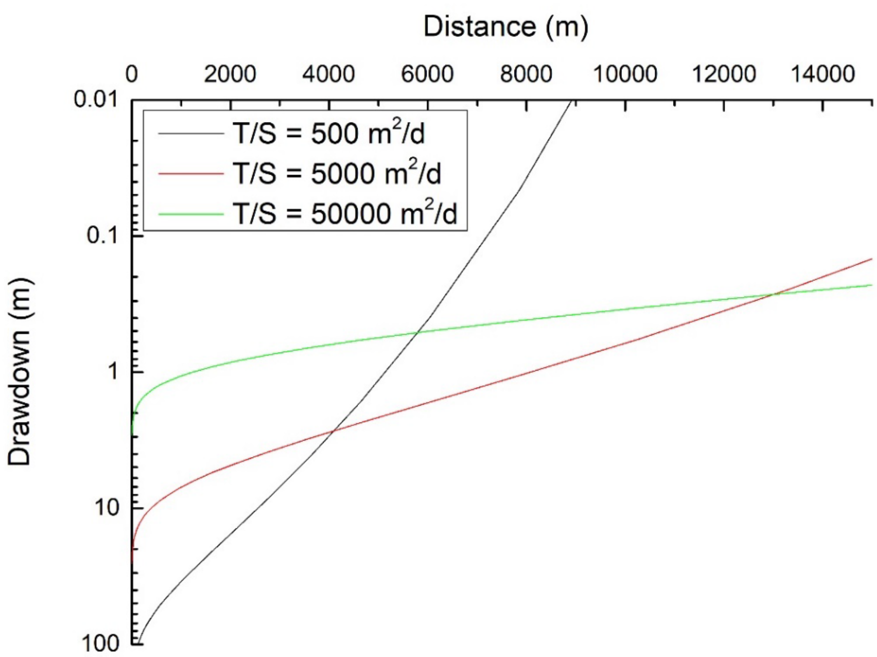


Figure 2: Drawdown versus distance from a pumping bore after 20 years of pumping, based on the Theis solution for a uniform aquifer with a pumping rate of 10 000 m^3/day. Drawdown for different values of the transmissivity to storativity ratio (T/S) are compared.

3.3 Rate of groundwater recovery post-closure

On cessation of pumping, the groundwater will immediately begin to recover in the vicinity of the mine pit, but, unless a barrier to flow is present, the drawdown cone will continue to expand and the water table will continue to fall in areas further from the pit. Figure 3 shows a numerical simulation of drawdown versus distance from a mine pit during groundwater level recovery for a hypothetic backfilled pit. At the completion of mining operations (t = 0), the drawdown cone extends approximately 30 km from the mine pit, with drawdown of more than 1 m extending for approximately 20 km. During recovery, the drawdown decreases near the pit, but the area affected continues to expand. After 50 years, the drawdown near the pit is reduced to less than a few percent of the drawdown at the completion of mining (approximately 3 m of drawdown remaining), but drawdown is still more than 1 m in an area extending more than 30 km radially from the pit, and more than 0.1 m over more than 50 km. Although in less permeable aquifers, the lateral extent of drawdown will be less (the simulation shown in Figure 3 used T/S = 10000 m^2/day, which is relatively high), the continued expansion of the drawdown cone after pumping ceases will occur irrespective of the aquifer hydraulic properties.

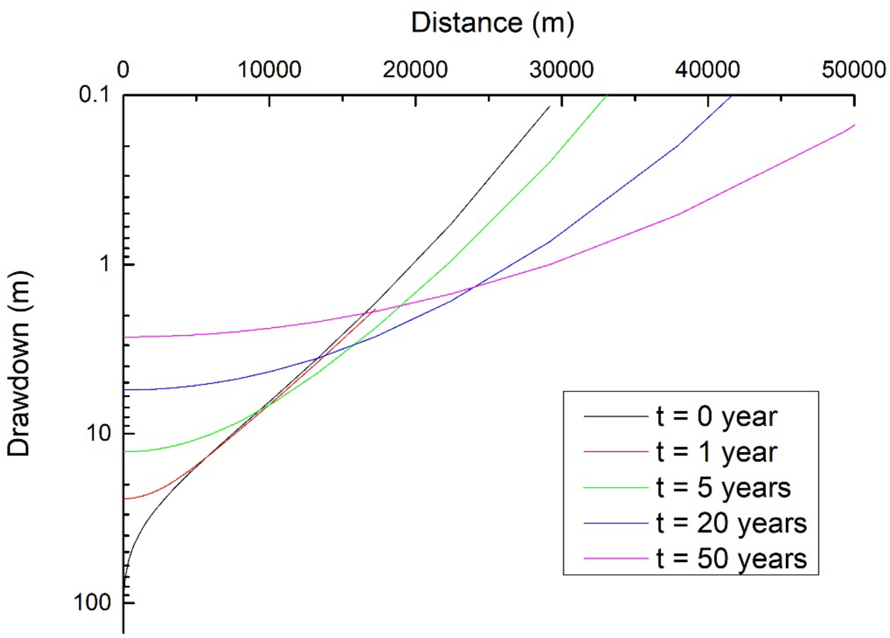


Figure 3: Drawdown versus distance during groundwater level recovery for a backfilled pit following 20 years of pumping at 100 000 m^3/day. Times reflect years since cessation of pumping. Results are based on the Theis solution for a uniform aquifer with T/S = 10000 m^2/day. During recovery, the drawdown decreases near the pit, but the area affected continues to expand.

Importantly, Figure 3 assumes that the aquifer is homogeneous and that it is sufficiently large in all directions that the drawdown cone does not reach the boundaries of the aquifer system. If the aquifer is constrained by less permeable sediments in one or more directions, then drawdown near the pit will be greater than the Theis solution predicts, and recovery will be slower. Figure 4 shows the effect of a linear barrier to groundwater flow, perhaps an impermeable fault or dyke, which is on one side of and 3000 m from a mine pit. Groundwater drawdown in shown at a location 2999 m from the pumping bore (adjacent to but on the mine side of the barrier). There is little drawdown at this distance during dewatering, due to the time for the effects of pumping to extend out 3000 m, but drawdown commences shortly afterwards with maximum drawdown occurring approximately 25 years after pumping has ceased. For this simulation, if a barrier is present, drawdown will be more than twice as large as if it is not present (Marshall et al., 2019).

Where mining developments occur in remote areas, there is often little characterisation of aquifer properties away from the immediate vicinity of the pit, and almost no characterisation beyond the limits of the drawdown

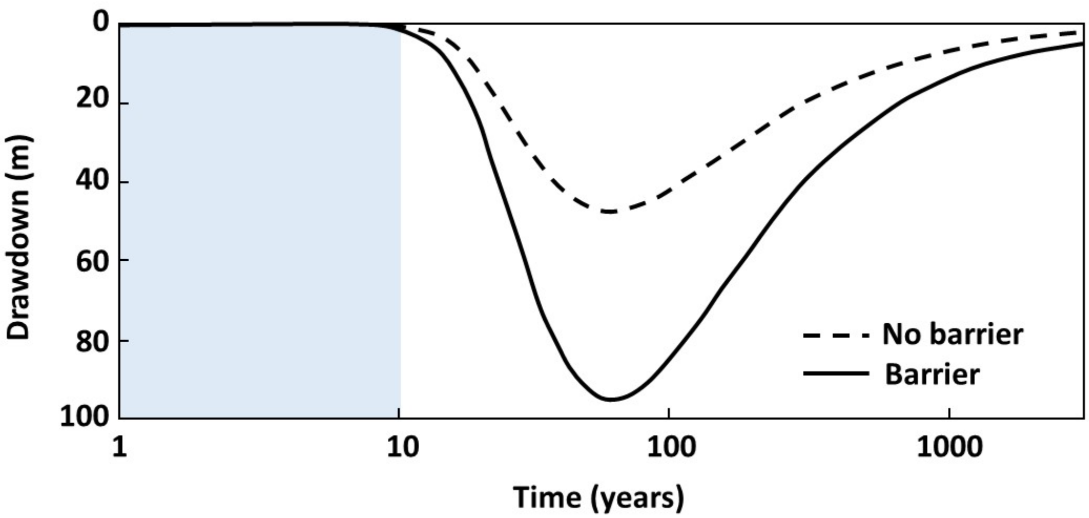


Figure 4: Simulation of drawdown and recovery for an aquifer with T/S = 120 m^2/day that is pumped at 10^5 m^3/day for 10 years. Groundwater drawdown is shown for a homogeneous aquifer, and for the case where an impermeable barrier is located 3000 m from the pumping bore. The plot shows drawdown at a location 2999 m from the pumping bore (1 m from the barrier). Maximum drawdown occurs after pumping has ceased, due to the time for the effects of pumping to extend to the observation point. The blue shaded area denotes the period of pumping. After Marshall et al. (2019).

While the rate of groundwater recovery will be mostly controlled by the aquifer hydraulic properties, the final, steady state groundwater level will be controlled by management of the pit and surrounds. Most important is the decision on whether to backfill the pit to above the water table. Figure 5 compares groundwater level recovery in the centre of a hypothetical mine pit that is backfilled versus a pit which is not backfilled (in the latter case, the groundwater level is equivalent to the pit lake water level). The groundwater level never fully recovers in a pit which is not backfilled if evaporation from the open pit exceeds rainfall and surface inflows. Even if the evaporation rate were negligible, the recovery would be slower if the pit were not backfilled, because the entire pit volume would need to be filled with water, compared to only 20-30% of the original pit volume that would need to be refilled if the pit were backfilled, due to the volume occupied by solid particles. In the case of a pit that is not backfilled, the final groundwater levels will largely depend on evaporation rates from the open pit. In the case of Kookaburra Gully graphite mine, it is estimated that the final water level in the pit will be 10 m below the original groundwater level due to the effects of pit lake evaporation (Parsons Brinckerhoff, 2015). Following closure of the Siviour graphite mine, steady-state groundwater levels are predicted to remain below original levels by approximately 6 m at the western pit, and by approximately 1 m at the eastern pit (JBS&G, 2018). There is also potential for landscaping and revegetation of pit surrounds (and of the pit itself if the pit is backfilled) to change surface water flows and infiltration and groundwater recharge rates. This has the potential to impact on groundwater level recovery but has not been systematically examined.

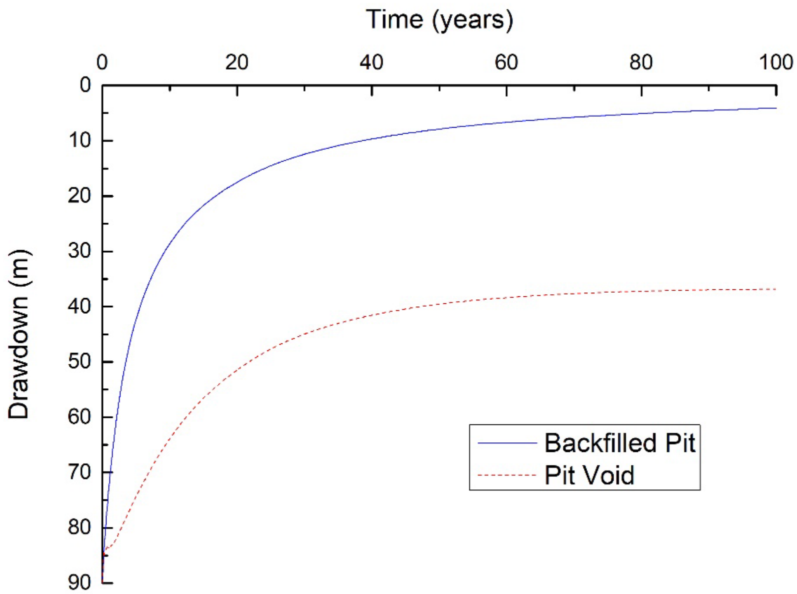


Figure 5: Numerical simulation of water level recovery in a mine pit which is backfilled versus a similar pit which is not backfilled, plotted versus time since pumping ceases. For the pit which is not backfilled, an evaporation rate that is 3 m/y in excess of rainfall is assumed. As with Figure 1, the numerical model uses 24 dewatering bores located around a pit that is approximately 2 km x 1 km in size, pumping a combined rate of 117,000 m^3/day for 20 years prior to cessation of pumping. The aquifer thickness prior to pumping is approximately 240 m, hydraulic conductivity is 1.5 m/day and storativity is 0.1. These values reflect a T/S ratio of approximately 3300 m^2/day. After Bozan (2021).

Model predictions for times for stabilisation of groundwater levels post-closure for existing pits range from several years to several hundred years. In the Pilbara, deep pits and large dewatering volumes generally result in long timeframes. At the Cloudbreak mine, areas of drawdown are predicted to continue for at least 40 years, post-closure (FMG, 2011), while time frames of approximately 100 and 200 years have been estimated for Roy Hill mine (Roy Hill, 2019) and the North Star mine (Worley Parsons, 2013), respectively. Timeframes of several tens of years are most commonly reported in the broader literature, with estimates of more than 50 years for Muja coal mine pit, Collie, WA (Kumar et al., 2009), more than 80 years for Caldag nickel mine, Turkey (Peksezer-Sayit et al., 2015), and 100 – 200 years for Kookaburra Gully graphite mine, Eyre Peninsula. In the latter case, the dewatering rate is very low (approximately 3.3 L/s), but the low permeability of aquifer materials results in a very slow groundwater recovery.

At the other end of the spectrum, faster recoveries are expected when mines are near surface water bodies that can act as a direct water source for the pit, or that drive hydraulic gradients and therefore groundwater inflows into the pit. At the Lihir goldmine, Papua New Guinea, groundwater is expected to recover rapidly on mine closure due to the proximity of Louise Harbour, which maintains a high groundwater level close to the open pit. At the Siviour graphite mine, Eyre Peninsula, it is predicted that following closure, the pits will reach steady state within two years due to the relatively high aquifer transmissivity and infiltration from the Driver River (JBS&G, 2018).

3.4 Environmental impacts of water table drawdown

Environmental impacts of water table drawdown, during mining and post closure, mostly relate to groundwater-dependent ecosystems (GDEs). In some cases, mine water management involves pumping water directly to these ecosystems to prevent possible negative impacts of water table drawdown. The main issue for mine closure is that environmental impacts on GDEs surrounding mine pits continue beyond the cessation of dewatering activities and in fact the area affected can increase (Figure 3). Supplementation of their water needs may therefore need to continue for a very long time if GDEs are to be protected. GDEs that occur beyond the extent of the drawdown cone during mine life (and hence are not impacted during mine life) may be affected post-closure and so these ecosystems may also need to be supplemented post-closure.

GDEs may include springs, rivers and creeks that receive groundwater inflow, terrestrial vegetation that accesses groundwater for transpiration, and stygofauna (Hatton and Evans, 1998). Plant species that rely heavily on groundwater mostly occur in areas with shallow water tables (e.g., Melaleuca spp.), although some species can access deeper water tables (to 10 m or more) when shallow soil water supplies are limited. Declines in the water table will therefore reduce availability of groundwater to terrestrial vegetation with potential adverse impacts on vegetation health. There are numerous examples of impacts of drawdown on terrestrial vegetation. A few of these are from the mining industry. In coal mining areas of northern Shaanxi Province of China, for example, observed vegetation decline was directly linked to water table decline due to mine activities (Li et al., 2013). In the Pilbara region, impacts on phreatophytic vegetation due to drawdown associated with dewatering are expected at a number of mine sites (e.g., Maunsell, 2006), and dewatering of the Hope Downs 1 mine and the subsequent decline in groundwater levels has been associated with a reduction in evapotranspiration from nearby plant communities (Pfautsch et al., 2015).

Flow derived from groundwater often sustains rivers and streams during periods of low rainfall. For streams that receive groundwater seepage, declines in the water table will reduce surface water availability. Wetlands, swamps and marshes can also be groundwater dependent. Reductions in stream flow due to dewatering of open pit mines have been measured in South Africa (Dennis et al., 2020), Spain (Rubio and Fernandez, 2010) and China (Hao et al., 2009). In the Yushenfu mining area, western China, the number of springs decreased from 2580 to 376 between 1994 and 2015, coincident with a decrease in the water table of up to 15 m (Fan et al., 2018). In the Pilbara, some of the surplus dewatering water from the Hope Downs 1 mine is discharged to to Weeli Wolli Creek to maintain a high water table, and hence ensure that flow at Weeli Wolli Spring is maintained. On the Eyre Peninsula, dewatering associated with the Siviour graphite mine is expected to result in depletion of the adjacent Driver River, causing complete drying of the stream for part of its length (JBS&G, 2018), although the discharge of mine water will be used to supplement downstream flows. Dewatering at the Shenhua Watermark coal mine in NSW is expected to increase leakage from Mooki River into the underlying aquifer, however the leakage is estimated to be less than 0.02% of mean river flow (Zhao et al., 2017). Drawdown associated with Fortescue's Solomon mine in the Pilbara, has impacted the extent of permanent pools along Kangeenarina Creek, which are believed to be groundwater dependent. Dewatering caused the surface water area of these pools to decrease from about 10.4 ha in 2004 to 4.9 ha in 2013. Pool supplementation commenced using reinjection bores in 2013 and direct surface water supplementation in 2014 (FMG, 2015). Supplementation is planned to continue until the groundwater has recovered to the extent that these pools are self-sustaining (FMG, 2020). Dewatering of the McArthur mine, Northern Territory is predicted to impact groundwater-fed waterholes on culturally significant sections of the McArthur River (KCB, 2017).

Groundwater-dwelling animals (stygofauna) have been recognised from many different environments, and often have a high degree of local endemism. Water table decline can potentially affect stygofaunal populations, particularly if these populations are restricted to shallow aquifer zones that become dewatered (Hose et al., 2015). Mining approvals may be conditional on protection of these endemic stygofaunal communities as they can trigger clauses in environment protection acts related to species extinction.[5] This can be an issue for mine closure because of the temporarily increased extent of the drawdown cone. Although we are not aware of any documented examples of impacts on stygofauna communities during mining operations or closure, this might be due to insufficient monitoring.

The major challenges associated with reducing drawdown impacts on GDEs are the same, wherever groundwater extraction occurs:

(i) Identifying and mapping GDEs. Existing GDE maps are usually incomplete, can be unreliable, and small ecosystems are often overlooked in large-scale maps. This is particularly problematic with terrestrial GDEs, as our knowledge of rooting depths of vegetation is incomplete.

(ii) Predicting impact of groundwater level decline on GDE health is extremely challenging and requires additional research. Increased groundwater and vegetation monitoring in areas of mining developments would improve the knowledge base in this area.

[5] Mining activity at Orebody 23, in the Pilbara was halted in 2001 following discovery of a stygofaunal community that was believed to be locally endemic. Mining was subsequently allowed to continue subject to regular stygofaunal monitoring. Follow-up work established that most species were more widespread than previously thought, and had distributions that extended beyond the zone of mining impact (Finston et al., 2004). In 2016, WA EPA advised against constructing the Yeelirrie uranium mine because of risks to stygofauna, but the mine was approved subject to conditions in 2017.

4. POST-CLOSURE HYDROLOGY OF MINE PITS

This chapter discusses the hydrology of pit lakes, and how they are connected to groundwater and surface water. It discusses pit lake water balances, changes in groundwater inflow to pit lakes over time and approaches for coupled modelling of pit lakes and the surrounding groundwater system.

4.1 Pit management strategies at mine closure

Williams (2009) discusses some of the advantages and disadvantages of backfilling mine pits, for mine operations and for post-closure pit conditions. Most backfilling strategies relate to storage of waste material, typically waste rock and/or tailings. Pits make be fully backfilled with waste material, partially filled, or not filled at all, and each of these options has a different impact on pit hydrology as well as the evolution of pit lake water quality. Predicting how backfilling impacts pit water quality and quantity creates opportunities for interaction between mine operational decisions and improved closure strategies. In some cases, backfilling of pits may facilitate beneficial post-closure land use. [6]

Pits that are fully backfilled to ground level may not form lakes, though they may still become wetlands or damplands, depending on the surface hydrology and the steady-state depth to groundwater post-closure. Partial pit backfill options might include backfilling to just above the post-closure water table to inhibit the formation of a pit lake, or preferential placement of spoil in the final years of the mine life to reduce the volume of the eventual pit lake (BMA, 2009). Johnson and Wright (2003) suggested that very large pits (e.g., Fimiston Open Pit, WA) are best left unfilled, particularly if they are not near important aquifers, local streams, rivers or other water sources. Some mine pits may be too large to be backfilled.

Where deep pit lakes are expected to form post-closure, the pit may be used as a repository of sulfidic material during mine operations, so the material becomes covered with lake water post-closure and formation of acidity is reduced (Johnson and Wright, 2003). This option for containment of potential acid producing material, relies on permanent lake stratification to keep the sulfidic material separated from oxygenated lake water (see Section 5.1.1).

McCullough et al. (2013) presented two case studies from Western Australia where fully backfilling a pit was expected to result in poorer regional closure outcomes than partial backfilling or retaining a terminal pit lake. In both cases (Nifty Copper and Tallering Peak), they predicted that fully backfilling the pit would lead to groundwater throughflow (Figure 8), with possible activation of potentially acid generating material in the pit. In the partially backfilled scenario, they predicted that the lake water level reached equilibrium and the lake would become a terminal sink and chemical release to surrounding groundwater would be minimised. However, potential impacts on GDEs of lower regional water levels may not have been considered in these cases.

In contrast, Bickford and Breckenridge (2013) outlined that the preferred closure strategy of the Corani silver-lead-zinc mine, Peru, was backfilling the pit with potentially acid generating waste rock. They predicted that if a

[6] For example, Oaky Creek coal mine, Bowen Basin; https://www.glencore.com.au/media-and-insights/news/oaky-creek-rehabilitation-earns-government-approval)

pit lake were allowed to form, acid water would be produced at a rate of 20 L/s, however full backfilling of the pit would reduce this volume to around 2.5 L/s due to the low groundwater flow rate through the backfilled pit.

4.2 Post-closure water balance of pit lakes

The pit lake water balance specifies the volumes of water added to and removed from the pit lake over time through various pathways (Figure 6). It can be written as

$$\Delta V = (V_P + V_{Gi} + V_{Si}) - (V_E + V_{Go} + V_{So})$$

The water balance, and changes in the water balance over time, can be used to predict long term water levels in the lake, the time it takes to reach equilibrium, explore interactions between pit lake water and surrounding water systems (groundwater and surface), and the likelihood of pit overflow during prolonged or extreme wet periods. Changes in each of the water balance terms over time also needs to be considered. As discussed in Section 3.3, groundwater inflows into the lake will usually increase over time as the regional groundwater recovers, and so estimation of changes in groundwater inflow over time are particularly important. Evaporation may increase over time as pit lakes fill, as the water surface area increases and the extent of wind sheltering decreases (Section 5.1.1). All water balance terms can change over time due to climatic fluctuations and climate change.

The importance of accurately estimating the different terms in the water balance will depend upon their magnitude. In many mining areas in Australia, evaporation is the major loss term of the water balance, and its accurate estimation is usually important, and is separately discussed below (see Section 5.1.1). Groundwater inflows and outflows are typically estimated using several different approaches of varying complexity and accuracy (Section 3.3.2).

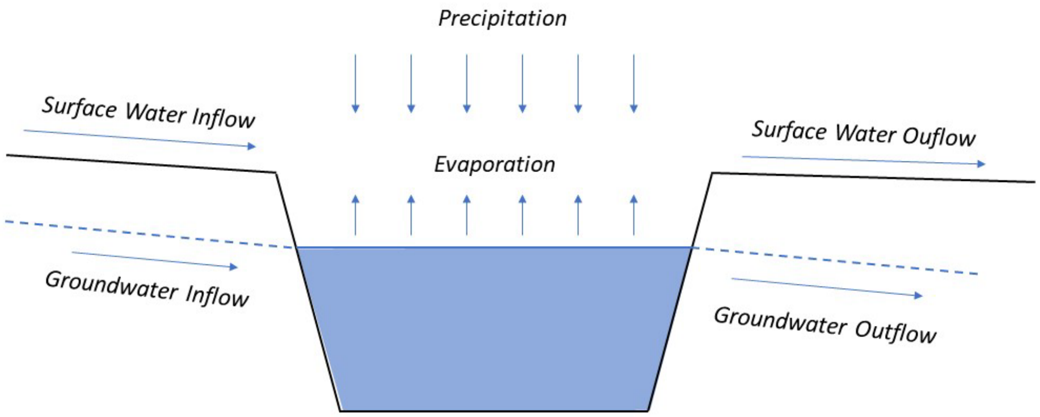


Figure 6: Conceptual pit water balance.

While groundwater may dominate inflows for pit lakes in arid areas, pit lakes in wetter climates may be dominated by surface flows. For example, many Queensland coal-mine pits are dominated by surface water, with relatively minor inflows of groundwater. A topography, hydrology, and flooding analysis for the Moranbah region mines has shown the potential for an integrated system connecting pit lakes with local surface water drainage (Cote et al., 2020). In NSW, inputs of groundwater are usually greater with complexity associated with surface water – groundwater interactions as captured by many reports prepared by the Independent Expert Scientific Committee on Coal Seam Gas and Large Coal Mining Development.[7]

A water balance for an existing pit lake, Lake Stockton in the Collie district of WA, is depicted Figure 7 (Carlino and McCullough, 2019). In this case, the main geological material surrounding the lake has low permeability and so precipitation and surface runoff are the inputs, with evaporation the main output. Nevertheless, this is classified as a throughflow lake according to Figure 8, as it has both groundwater inflow and outflow. Lake Stockton is supplemented with dewatering discharge from a neighbouring mine, and the volume of groundwater inflow to the pit lake is expected to increase when this supplementation ends (Carlino and McCullough, 2019).

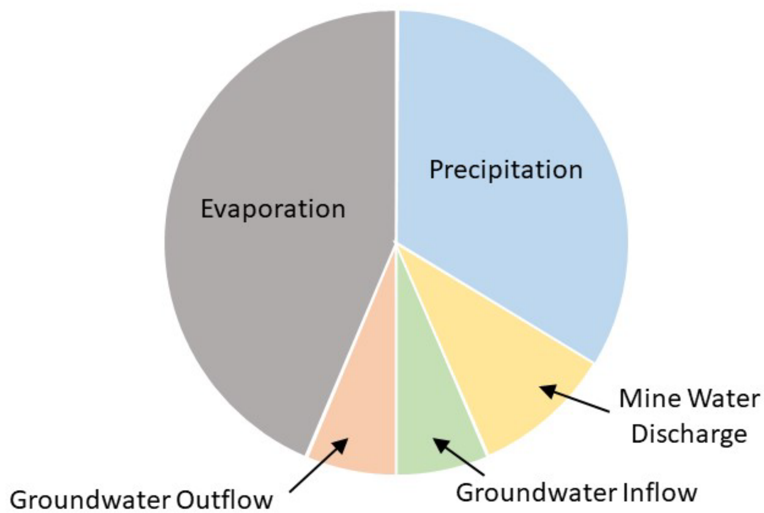


Figure 7: Water balance of Lake Stockton, a former open cut coal mine in the Collie Basin, Western Australia. The Stockton mine was abandoned in 1957 and became a lake when groundwater recovered. Since 1996, the lake was also supplemented with dewatering discharge from a neighbouring mine. Lake Stockton is a throughflow lake, according to the classification of Figure 8, as it has both groundwater inflow and outflow. After Carlino and McCullough (2019).

4.3 Pit lake connections with groundwater and surface water systems

As described above, pit lakes interact with the regional hydrological system. In Australia, pit lake connections to groundwater have received most attention, and several publications include a conceptual model of pit lake – groundwater connections (Johnson and Wright, 2003, McCullough et al., 2013). When the final pit lake water

[7] https://iesc.environment.gov.au/system/files/iesc-advice-russell-vale-2020-120.pdf; https://iesc.environment.gov.au/system/files/iesc-advice-narrabri-2020-119.pdf

level is below the original water table position, the pit behaves as a terminal sink for groundwater (Case 1; Figure 8). Such *terminal pit lakes* are more likely to form in arid climates and for lakes with large surface areas, where the evaporative loss will be large (Castendyk et al., 2015). These lakes have the advantage of keeping mine-impacted water on the mine site, but dissolved solids and trace metal concentrations will increase over time through evapo-concentration. If dilution by rainfall and surface water inflow are not sufficient to prevent the pit lake becoming highly saline, then off-site discharge of highly saline pit lake water into the groundwater can occur via density-dependent flow, even though the lake water level is lower than the regional groundwater level (Simmons and Narayan, 1997).

If a lake has both groundwater inflow and outflow, then it is classified as a *groundwater through-flow lake* (Case 2). For a pit lake without river outflow, this will usually occur when groundwater inflow exceeds evaporation. Flow-through lakes are more common on sloping topography in humid climates and for lakes with small surface areas. While the flow-through lakes may not experience the same degree of evapo-concentration as terminal pit lakes, mine impacted water may discharge into the adjacent groundwater system.

Groundwater recharge lakes occur when there is groundwater outflow but no groundwater inflow (Case 3). This is most likely to arise when precipitation and/or surface runoff dominant the water inflows, and exceed evaporation from the pit lake surface. These lakes usually occur in areas of high rainfall, or where surface water is diverted into the pit once mining has ceased. While the lake discharges to the surrounding groundwater, the high volumes of surface flow typically dilute the lake water, and chemical fluxes from lake to groundwater may be small. In the case of back-filled pits (Case 4), groundwater throughflow is expected to occur and again can result in contamination of the groundwater downgradient. In this case, the impact on groundwater will depend largely on the nature of the backfill material (e.g., the degree of acid producing material). Backfilled pits may also act as recharge sources if the backfill material is much more permeable than the surrounding aquifer material and so permits high infiltration rates.

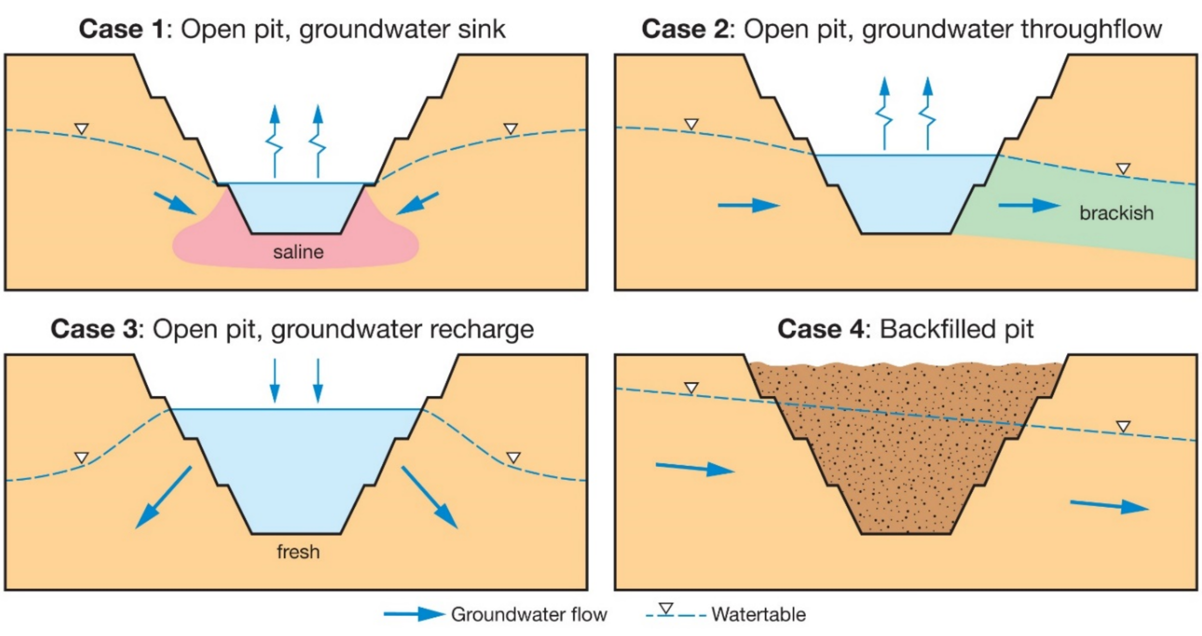


Figure 8: Hydrogeological environments for below – water table mine pits. Modified from Johnson and Wright (2003).

While this simple classification of pit lakes is conceptually useful, it simplifies the transient dynamics of pit lake hydrology. Almost all pit lakes will be terminal lakes immediately following closure, as groundwater will flow into the pit from all directions. The above classification is therefore most applicable to the long-term dynamics, which might take many years or even hundreds of years to develop (see Section 2.3). Even then, seasonal or inter-annual changes in rainfall and evaporation can result in lakes fluctuating between terminal, and flow-through and net groundwater recharge conditions.

Equivalent classifications can be used for surface flows (Figure 9). In high rainfall areas, pit lakes can be dominated by surface water inflows, and can be specifically engineered so that they are connected to the surface water drainage system (i.e. as a *surface water through-flow system*). This connection to the river network can be either 'controlled' (using inlet and outlet structures) or 'uncontrolled' (where the river flow is not regulated), and river outflow can be either more or less than river inflow depending on the lake water balance (Lund et al., 2018). The use of surface water through-flow systems to manage systems of relatively shallow pit lakes from lignite mining is widespread in Germany (Oldham et al., 2019). The approach minimizes evapo-concentration of lake waters and concentrations of chemicals discharging to the adjacent groundwater. Some pit lakes may receive river inflow, but if precipitation plus total surface water inflow is less than evaporation ($V_E > V_P + V_{Ri} + V_{SR}$), then there will not usually be a permanent surface water outflow. However, during very high rainfall periods, rainfall and surface water inflow may exceed evaporation, and so outflow may still be possible (i.e. they become *event-based* surface water through flow lakes). If pit lake water quality is poor, the overflow of pit lakes into the surface drainage network during high rainfall events can cause downstream contamination, and some mine pits require perpetual pumping to keep pit lake water levels low and prevent them overflowing into surface water systems (Maest et al., 2020). Pit lakes with river outflow but no river inflow are also possible in cases where precipitation and surface runoff into the pit exceed pit lake evaporation (Figure 9).

Lund et al. (2018) note that in arid regions, river flows can be highly variable, and that the connection of seasonally-flowing and intermittent rivers to pit lakes poses greater challenges than for permanent rivers. These challenges includes the potential loss of environmentally important peak flows and baseflows to reaches downstream of the pit lakes.

Most of the Collie pit lakes (WA) are groundwater flow-through lakes, although in winter Ewington Lake oscillates between a recharge lake in winter (due to lake precipitation and surface water inflows) and a terminal groundwater lake in summer when evaporation rates are higher (Lund et al., 2012). -Some of the pit lakes in the Collie district, including Lake Kepwari, discharge to natural waterways (and ultimately to the Collie River), and are river through-flow lakes as well as groundwater throughflow lakes. For Lake Kepwari, mean river inflows and outflows exceed mean groundwater inflows and outflows, but while groundwater inflows and outflows vary little from year-to-year, river inflows are highly variable and do not occur in all years. River inflows and outflows are generally similar in timing and magnitude, but during low rainfall periods when lake levels are low (for example, in 2015), river outflows can be delayed by several months as the lake first fills before outflow occurs (Figure 10). Changes in water chemistry can also occur between inflows and outflows due to chemical processes in the lake, as well as precipitation and evaporation. Changes in water flows and water quality between river inflow and outflow can impact the downstream catchment.

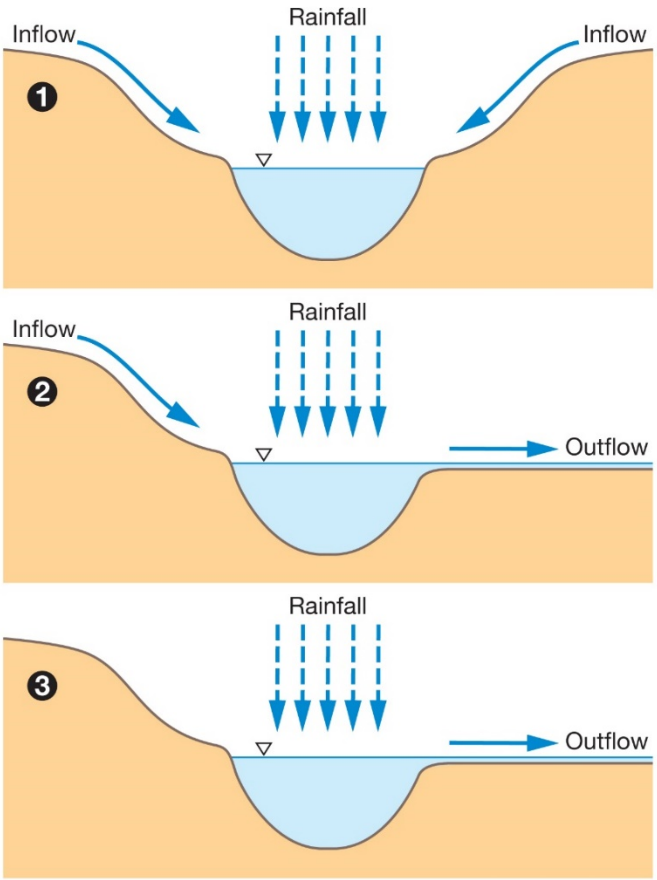


Figure 9: Conceptual classification of pit lake – river interaction. 1) Inflow only. 2) Surface water throughflow. 3) Outflow only.

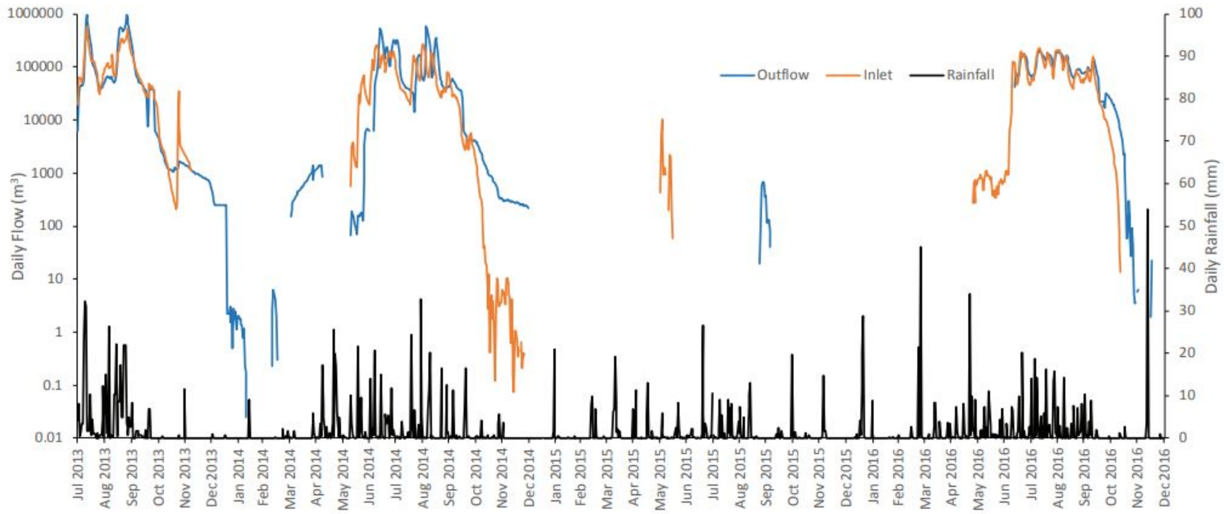


Figure 10: Inflows and outflows of Collie River through Lake Kepwari between 2013 and 2016. From Lund et al. (2018). Reproduced with permission of the author (30 September 2021).

4.4 Evaporation rates from pit lakes

The evaporation rate is an important component of the pit lake water balance, as the balance between evaporation and groundwater and surface water inflows will influence the pit lake water level and hence groundwater levels surrounding the mine. Evaporation will also affect the rate at which salinity and metal concentrations increase, particularly in arid to semi-arid regions (Shevenell, 2000).

The lake surface water temperature effects evaporation from the lake surface and lake surface temperature is significantly impacted by wind sheltering and lake stratification dynamics. Therefore, while a water balance calculation could be conducted independently of stratification modelling, the integration of water balance and stratification modelling can permit an accurate feedback between stratification, water temperature, evaporation and the lake water balance (see Section 4.1.1).

There have been numerous attempts to determine pit lake evaporation based on measured pan evaporation rates, with mixed accuracy. Some estimates of lake evaporation have been taken from evaporation measured in a Class-A evaporation pan on land, by using a designated pan coefficient (often 0.7), but results are often inaccurate. For example, (Shevenell, 2000) found that evaporation in mine pit lakes in Nevada was much lower than that seen in natural terminal pit lakes, because the pit lakes were at much higher elevations and had much lower surface to depth ratios.

Considerable progress has been made by some mining companies by the installation of evaporation pans and weather stations that float on the pit lake, which represents pit lake conditions better than ground-level weather stations and evaporation pans. This permits evaluation of the evaporation models. McJannet et al. (2017) applied the floating pan system to a Pilbara mine and assessed a range of evaporation models. They concluded that a site-specific aerodynamic model that relied on an equilibrium temperature assumption provided reliable estimates of evaporation from the studied pit lake (bias in annual average evaporation = 4%) compared to a pan coefficient approach using a regional Class-A pan (bias = 12%). They also assessed a generalised version of the aerodynamic model using regional weather data including a sheltering factor (bias = 6%). The importance of better understanding the pit lake aerodynamics was noted. When appropriate wind speed scaling was applied, the evaporation rate predicted by the CSIRO aerodynamic model for a coal mine pit lake in Norwich Park Mine, Central Queensland was only 5% less than in-pit measured evaporation (McJannet et al., 2019).

The widely used lake stratification model DYRESM (Imerito, 2007) calculates evaporation based on bathymetry and meteorological data (Sivapalan, 2005). It employs bulk aerodynamic formulae for calculation of the flux of latent heat from the water surface to the atmosphere, which have been shown to competently capture the surface fluxes of latent heat, as well as momentum and sensible heat from a variety of water bodies across a broad range of climatic conditions (Schladow and Hamilton, 1997). Helfer et al. (2011) demonstrated excellent agreement between average evaporative losses predicted by DYRESM, an independent water balance calculation, measured pan evaporation, and Penman-Monteith estimates.

Fringing topographical features (e.g., embankments around most mine lakes) significantly modify the wind field acting over the surface of the pit lake. Thus it is likely that parts of the lake water surface will be protected from the full effects of wind action on mixing and latent and sensible heat fluxes (Imberger and Parker, 1985). Each of these three processes impact on surface water temperatures and therefore on evaporative water losses; in turn the evaporative losses impact on the steady-state water balance and lake levels. A wind sheltering algorithm can be developed using pit topography and local wind speed and direction data. As the pit fills with water, the degree of sheltering changes. For lakes with high levels of evaporative losses, steady state water levels may remain well below ground surface; under these conditions the effective wind field on the lake surface may only be 10% of the wind field experienced 10 m above ground level. Such a high degree of sheltering would have impacts on evaporation, and thus feeds back into the water balance and the steady-state lake water levels (Oldham, 2014). However, we are not aware of any studies that have specifically examined changes in evaporation over time as pit lakes fill post-mine closure.

Newman et al. (2020) used stable isotopic analysis of water (δ^2H and $\delta^{18}O$ of H^2O) and sulfate ($\delta^{34}S$ and $\delta^{18}O$ of SO_4) to estimate evaporation losses from surface water bodies in a mine pit lake (Nevada, USA). While this method is theoretically promising, the results were inconsistent, and the authors recommend that this method is only suitable for lakes with well-defined input concentrations and groundwater inflow.

4.5 Rates of groundwater water inflow to pit lakes over time

As discussed in Chapter 2, groundwater recovery following cessation of dewatering can take many decades. While groundwater models that are used to calculate the recovery of groundwater levels post mine closure enable calculation of changes in groundwater inflow to pits over time, often pit lake models either assume constant groundwater inflow or simplify the dynamics of the groundwater system. For example, (Hancock et al., 2005) assumed that as the pit lake filled, groundwater inflow to the pit decreased in direct proportion to the pit lake water level. Numerical groundwater models such as MODFLOW and FEFLOW will provide more accurate estimates of groundwater inflows and outflows to and from pit lakes and have been used as part of mine closure plans. However, we are not aware of any examples where pit lake inflows are explicitly presented, even though they will be calculated by many of the numerical models. For this reason, there remains a poor understanding of how groundwater inflows to pits change over time following mine closure.

According to the Theis equation, the flow of groundwater back towards the mine pit will initially be very rapid but will decrease over time (Figure 11). However, as discussed in Section 2.2, the Theis equation does not consider the geometry of the mine pit, and so only provides an approximation of the groundwater flux. As discussed by Eary and Castendyk (2009), the pit walls are not necessarily vertical and inclined walls provide larger groundwater seepage faces and therefore larger groundwater inflows to the pit. Changes in groundwater inflow with time will depend on final pit geometry and therefore may be different to that indicated in Figure 11. Eary and Castendyk (2009) noted that the Berkeley pit lake in Butte, Montana, one of the longest-studied pits in the world, filled at a constant rate for the first 15 years owing to this affect. In summary, pit geometry will affect how groundwater inflows change over time, and groundwater inflows will decrease as the regional groundwater recovers. However, the relationship between pit geometry and the evolution of groundwater inflows has not been specifically examined.

Changes in the groundwater inflow rate over time determine chemical fluxes into the lake and can be important for the evolution of lake water quality. In non-mining lake research, the different impacts of land use and climate change on regional groundwater levels and the subsequent change in chemical fluxes into standing water bodies, is an area of intense focus at present. To our knowledge, similar investigations have not been undertaken for pit lake water quality prediction.

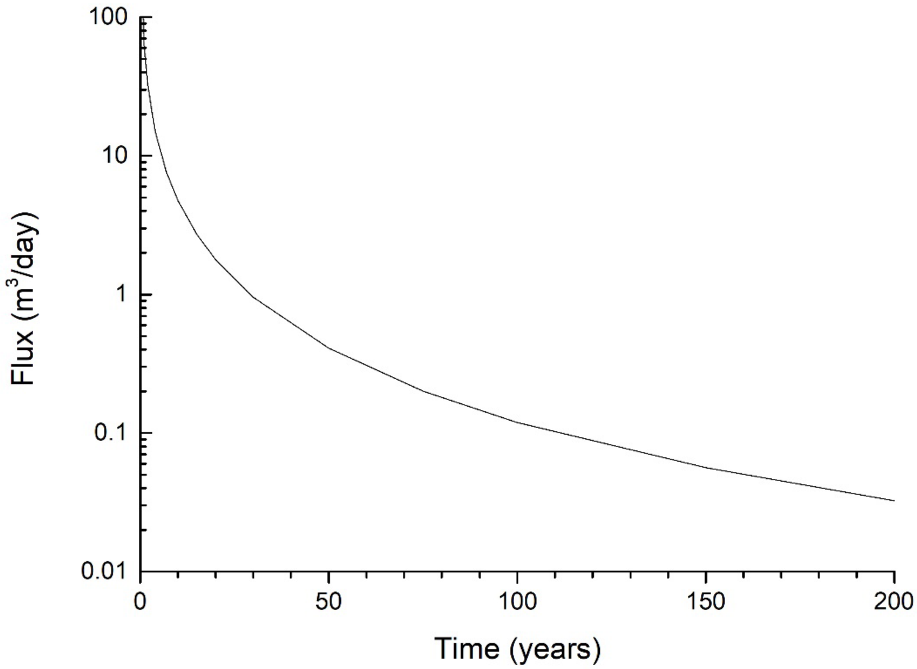


Figure 11: Groundwater flow towards a backfilled mine pit over time since cessation of mine dewatering. This simulation uses the Theis solution and assumes a circular mine pit with a diameter of 200 m. Other model parameters are the same as those given in Figure 3, above.

4.6 Modelling groundwater inflows to pit lakes

Accurate quantification of the mine water balance requires modelling of the pit lake and its connections to the groundwater and surface water. A literature review on modelling of groundwater flows into artificial lakes shows that the current understanding is not sophisticated, with no established methodology for modelling interactions and feedbacks between lakes and groundwater systems. Such lake – groundwater interactions are likely to impact the biogeochemistry of the lake, and its eventual water quality.

A recent study used the lake package in MODFLOW to simulate such a scenario, and found that the calibrated model was sensitive not only to changes in parameters but also to variations of the lake inflow and outflow (El-Zehairy et al., 2018). The authors conclude that: (i) complex dynamics of artificial lake interactions with groundwater requires appropriate and reliable data sets to carry out reliable simulations; (ii) it is important to have at least daily time series of climatic driving forces (e.g., precipitation, and potential evapotranspiration), lake inflows, lake outflows and state variables, including at least lake stages and spatially distributed hydraulic heads data; and (iii) according to their sensitivity analysis, the most important parameters are lake bed leakance and hydraulic conductivity of the underlying aquifer(s). The authors further argued that the high sensitivity of artificial lake models to inflow and regulated outflow points out the critical importance of reliable estimates of these data when modelling interactions of artificial lakes with groundwater. Only one study specific to pit lakes was found, which used MODFLOW and its lake package to investigate the effects of slope collapse on the surface water-groundwater interaction – a topic outside the scope of this report.

Three-dimensional lake models, such as DYRESM and GLM, can account for spatial and temporal variability of inflowing groundwater, and when coupled with biogeochemical models (e.g. CAEDYM and AED), can predict the impact of these inflows on pit lake water quality. These coupled models can also be used to track the impact of lake sediment biogeochemical processes on reducing groundwater inflows over time (typically by reducing sediment permeability due to mineral precipitation in the sediments). These lake models require groundwater flows as input data, and these can be provided via field measurements (e.g., Oldham et al., 2019) or via groundwater models. However, changes in groundwater over time are best obtained from groundwater models. Note that lake models have highlighted that the location of groundwater inflows relative to temperature stratification depths determines the impact of inflowing groundwater on lake water quality. Groundwater models rarely provide such detailed spatial prediction of groundwater inflows. The difference in spatial and temporal scales used in groundwater models and lake models makes full coupling challenging, as briefly described below.

Xu et al. (2021) applied HydroGeoSphere (HGS), a coupled surface water – groundwater model, to model groundwater-lake interactions at a large spatial scale. However, there are several challenges in applying these models to pit lakes. A primary challenge associated is the spatial scale of the domain. Xu et al. (2021) employed a mesh spatial resolution of 2 to 10 km., and this coarse resolution limits the detail that can be resolved in the topography, the surface water network, and the lake bathymetry. Nevertheless, in their study, modelled surface water behaviour showed a good match with observed conditions with little intervention beyond subsurface parameter tuning. However, incorporation of a high level of local detail may be important for pit lake modelling. The temporal scale can also be a challenge, with high resolution of climate data often desirable for modelling surface water systems, but very long simulation periods (hundreds of years or more) required to properly simulate groundwater flow systems. (In the study of Xu et al., year-to-year variability in hydrologic conditions was not addressed.) These processes collectively require very long model run times. While coupled models such as HGS can simulate lake water levels, they do not simulate in-lake processes as well as the dedicated lake models.

When we consider variable time and spatial scales the problem gets even more complex. For example, modelling these interactions at regional level raises the problem of which type of model grid to use (structured vs. unstructured) and model discretization (high discretization is needed for the complex interaction problems at the interface aquifer-pit but unpractical at large scales for computing reasons). Recent examples (Herron et al., 2018, Janardhanaran et al., 2016) of regional modelling of groundwater-surface water interactions in the mining context in Eastern Australia did not use fully coupled models because of i) high data requirements for parameterisation, ii) operational constraints, iii) general numerical instability of such models and long runtimes that would severely limit a probabilistic uncertainty analysis. This raises important practical questions that are worth further research. A relatively old but important guide has been provided by Hunt et al. (2003).

5. WATER QUALITY CONSIDERATIONS

This chapter discusses water quality in pit lakes and downstream environments. It reviews chemical and physical processes in pit lakes, and predictive water quality modelling. Potential uses of pit lakes, based on water quality criteria, are also discussed.

5.1 Chemical and physical processes in pit lakes

The evolution of water quality in pit lakes, including variables such as redox potential and pH, is initially determined by the quality of surface and sub-surface inflows and how the inflow waters interact with pit mineralogy. Key determinants include the local geology, and the composition of waste rock and proximity of waste rock deposits to the mine pit. However, as the volume of inflows relative to the lake volume decreases (as would be expected as the pit fills), the physical, chemical and biological processes in the lake itself begin to impact, and may eventually dominate, lake water quality. The physical processes of the lake, such as stratification and mixing, control the transport of many chemical species to different regions of the lake. Primary production and the metabolism of organic matter are mediated by lake biota and microbes. The buffering of pH (i.e. capacity of the system to resist pH change) is controlled by geochemical processes, including the dissolution of inorganic carbon in circum-neutral waters, or the precipitation and dissolution of mineral phases under acidic or alkaline conditions.

There are many feedbacks between these different processes. The impacts on water quality of all of these lake processes must be balanced against the impact of inflowing waters; for example the geochemical characteristics of inflows may counteract alkalinity generation within the lake. Alternatively, lake processes may ameliorate poor water quality of inflowing water.

The relative contributions of different processes will be context-specific and strongly impacted by, for example, mine pit morphology, mineral deposits, and local meteorology. In all cases, water balance models are an essential prerequisite for prediction of changes in water quality (Section 3, also Oldham (2014)). Some generic conceptual pit lake models showing the major components of stratification and geochemical cycles are presented in Figure 12, and some of the key pit lake processes are discussed below.

5.1.1 Temperature and stratification

Pit lakes undergo annual water temperature cycles due to heat exchange between water and the atmosphere, as well as well as temperature exchanges between the different vertical layers in the lake and the sediments. Temperature can dominate water density, and temperature gradients (*thermoclines*) in lakes can result in a water column with two or more horizontal layers, termed *density stratification*. In deep lakes, density stratification is typically either permanent (termed a *meromictic lake*) or disrupted by weather conditions once (*monomictic*) or twice (*dimictic*) a year (Boehrer and Schultze, 2008). In Western Australia, many pit lakes greater than 10-20 m deep stratify during the summer period when a thermocline develops between the upper warmer water and cooler lower water (DMIRS, 2020). The thermocline is typically disrupted seasonally by convective cooling or winter storms (i.e. the lakes are monomictic).

Intrusion of high salinity groundwater and/or long term evapo-concentration of lake waters can also produce extremely salty water and will play a role in lake density stratification (Castendyk et al., 2015). These hypersaline waters may accumulate on the bottom of the pit lake creating permanent stratification (i.e. a meromictic lake). Rapid surface water filling of the Berkely pit (Montana, USA) created a freshwater lens over the top of higher salinity bottom water and the lake became permanently stratified (Gammons and Duaime, 2006).

Each stratification layer may have distinct chemical composition; in pit lakes the chemical composition will likely be a function of that layer's exposure with reactive wall materials. This density stratification constrains the mixing of surface and deeper waters, can minimize replenishment of deep waters with oxygen, and can therefore control redox states, geochemical cycles, and acidification processes (Figure 12). Alternatively, periods of mixing (disruption of stratification) can homogenize the water column, mixing dissolved oxygen from the lake surface throughout the water column and transporting deep, carbon dioxide-rich water to the lake surface. Maintenance or disruption of stratification has long been used to manage lake water quality around the world, however as far as we are aware, this approach has not been incorporated into long term closure strategies.

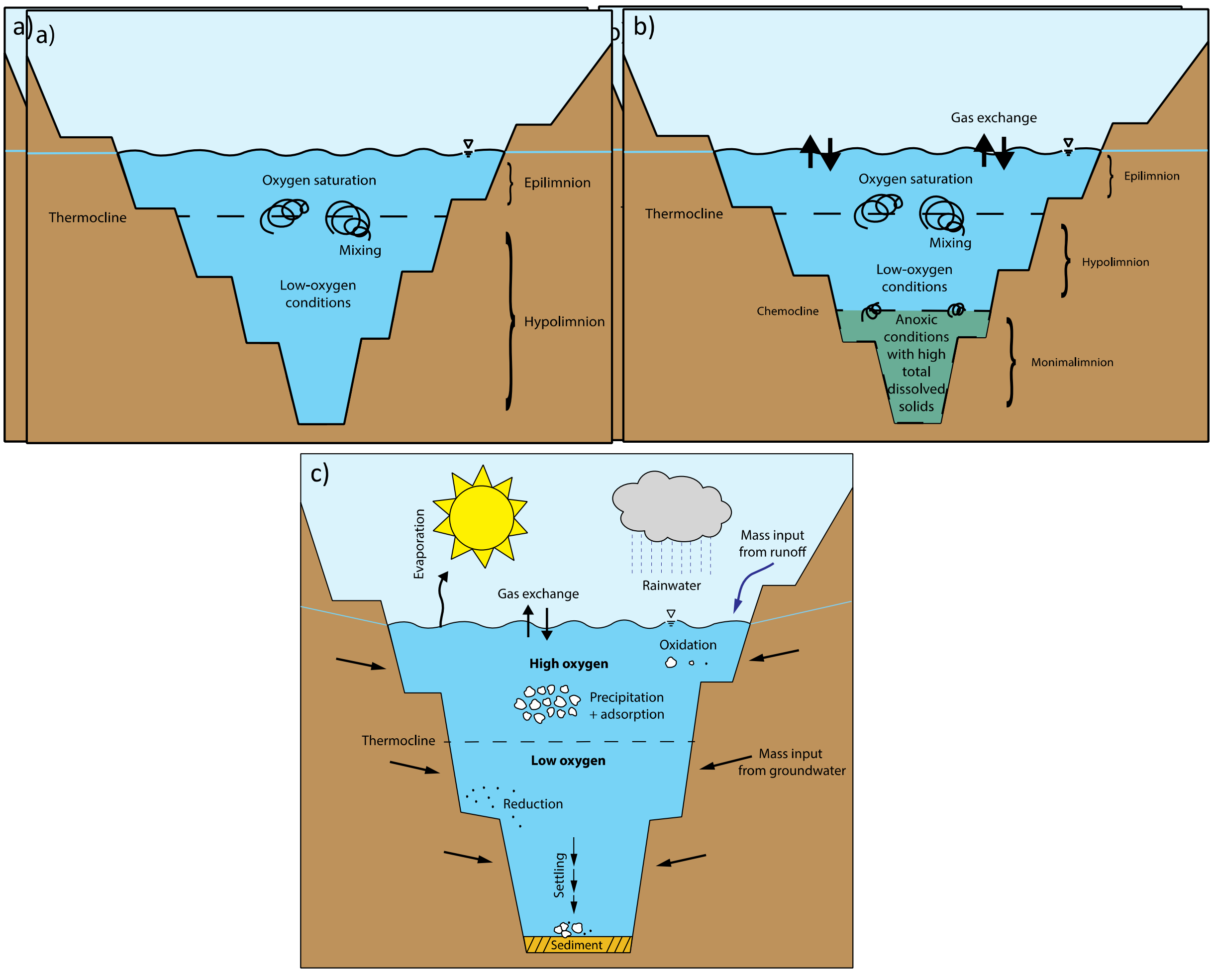


Figure 12: Conceptual models of lake processes in a pit lake a) two-layer temperature stratification b) three-layer temperature stratification with permanent monimolimnion, and c) geochemical processes under stratification. Figure from Oldham (2014), modified from Wielinga (2009).

Some studies of pit lake stratification have focused on ratios of lake depth to area, with the assumption that wind mixing is promoted when the ratio of depth to area is low. Pit lakes are typically steeper and deeper than natural lakes, with wind fetch reduced across their surface by the presence of overburden and waste rock dumps (Huber et al., 2008, Jewell, 2009). Pit walls can also shade the surface of the lake from solar radiation. The combination of a relatively deep pit lake and wind sheltering could impede whole-lake mixing, as has been found in mountainous lakes and reservoirs (Jewell, 2009).

5.2 Evolution of pit lake water quality with time

Hydrochemical evolution of pit lakes may take centuries, and there has been little long-term monitoring of pit lake water quality. As indicated above, lake water quality will depend on complex interactions between an array of limnological processes, including oxygen status of the lake, pH, hydrogeological flows, water quality of any inflows and wall rock composition.

For many pit lakes the principal source of solutes is from groundwater inflows (particularly saline and hypersaline pit lakes) and catchment area inflows from storm events. A minor source is solutes present in rainfall. For some pit lakes, a significant source of solutes is reactive minerals, particularly sulfide minerals, present in the rocks exposed in the pit walls and floor, and waste rock material that is retained in the pit void. These minerals react with oxygen and water to form sulfuric acid, or sulfate salts if there are sufficient acid-neutralising minerals in pit wall rocks. The most important process for metal ion release is dissolution of metals under a combination of acidic and oxidising conditions. Increased dissolution of minerals can also be observed in the presence of highly saline conditions, as well as anoxic conditions (notably ferrous and manganese ions). Wall rock run-off may result from from ephemeral wash-off by rain, of dissolved efflorescent salts and oxidation products present on exposed rock surfaces. Wall rock leaching can also occur when groundwater or pit lake water is in direct contact with wall rocks over a longer timescale.

Castendyk et al. (2015) outlined the trend typically observed in solute run-off or leaching from wall rocks and their impact on pit lake water quality: 1) rapid concentration increases at the pit bottom due to release of soluble components from weathered wall rock and wastes rock, 2) dilution of pit lake water as groundwater enters the pit, which typically has lower metal concentrations than found at the pit bottom; 3) in terminal pit lakes, a gradual increase in all constituents over time due to evapo-concentration.

Another source of potential source of pit lake solutes is mine waste (waste rock or process tailings) stored in a containment facility, i.e., a waste rock dump (WRD) or tailings storage facility (TSF). A well-constructed WRD will contain PAF materials in a low oxygen/low seepage environment, with only NAF material and soil forming the crests and slopes of the landform. Solutes can be mobilised by infiltrating rainwater and seepage quality will be variable, and is typically assessed using standardised laboratory testing of waste materials and ongoing groundwater quality monitoring program. TSF seepage usually presents a greater environmental risk, particularly if the facility is unlined, as the gangue minerals often include reactive sulfide and metalliferous minerals, and tailings are deposited in a saturated state. It is common practice to minimise potential environmental impacts from these storages by placing them within the “cone of depression” resulting from mine dewatering, thereby ensuring fluids and solids flow towards the mine pit, rather than to nearby receptors (such as livestock watering bores).

Mine water acidification is usually linked to mining of reduced sulphide containing ores. Such ores are common in the coal mining industry. Open cut metalliferous mines are typically located in the oxidised weathering zone, where carbonate alteration can neutralise acid water. However, as pits deepen, there is greater potential for exposing primary or sulfidic ores. Several Pilbara iron mines encounter Mount McRae shale, a highly reactive pyritic black shale (Johnson and Wright, 2001). Lakes from former gold, metal-sulfide, or coal mines usually contain elevated concentrations of heavy metals (e.g., Al, Cd, Cu, Fe, Mn, Ni, Pb, Zn, U) and/or metalloids (e.g., As, Sb, Se, Te) that can threaten the environment. Alternatively, lakes formed from mining of iron ore, gravel, or industrial minerals (e.g., talc, asbestos) may have quite low concentrations of dissolved metals. The mobility of most metals and metalloids is strongly pH-dependent.

Redox conditions in the water column can also play a critical role in determining which trace metals may end up in solution when groundwater interacts with ore bearing rocks. As described above, redox is affected by the mixing of oxygenated waters to depth, typically controlled by stratification. In a rare study, the evolution of pH and water quality in Lake Kepwari, a former coal mine in Collie, Western Australia, has been described by Salmon et al. (2008) in the decade post-filling, as well as how the water quality responded to seasonal disruption of stratification.

In arid environments, evaporation is likely to be one of the factors affecting pit lake water quality, particularly if pits are not connected to a surface water drainage system and so do not receive significant diluting surface water flow. For example, Mount Goldsworthy pit lake (Pilbara) increased in total dissolved solids (TDS) from 1400 mg/L in 1982 (Johnson and Wright, 2003) , to 4200 mg/L in 1992, and 5600-5900 to 5500 mg/L in 2005 (Sivapalan, 2005). Frequently mine pit lake risk assessments assume that pit lakes rapidly undergo evapoconcentration (i.e. within 5 years), such that pit lake water becomes too saline to be palatable for either livestock or migratory birds. However, a recent study of mine pit lake water quality found that the TDS of pit lakes fed with fresh (TDS < 5000 mg/L) groundwater typically only exceeded livestock drinking water guidelines in pit lakes that were more than 20 years old (Linge et al., 2021).

While contamination from elevated toxic elements has long been studied, an emerging area of mine water quality is consideration of contamination issues from materials and chemicals used on site. In particular, the use of per- and poly- fluoro alkyl substances (PFAS) has become a major issue for groundwater contamination at airports and defence bases in Australia and word wide (Banwell et al., 2021). PFAS do not readily break down and can leach through soils to contaminate groundwater and surface waters PFAS chemicals have been long used in mining operations as part of both processing and fire fighting activities, with sources of PFAS on a mining site likely to be diffuse and difficult to identify. Pit lake water quality can be impacted through groundwater, use of contaminated backfill, or storage of mine impacted water. Recently the Western Australian Department of Water and Environmental Regulation has declared the historic Mount Whaleback mine in the Pilbara to be a contaminated site, in part based on the presence of PFAS in soil and groundwater at locations associated with fire training activities and wastewater treatment[8]. It is likely that many other mining operations will have similar contamination issues with PFAS.

5.3 Predictive modelling of pit lake water quality

Numerical models of lake processes allow us to test our understanding of what drives the overall water quality in pit lakes and to quantify the relative contribution of inflows versus lake processes. Most importantly, numerical models also allow us to investigate the evolution of lake processes over longer time and length scales than we have experimental data. Thus, numerical modelling should be considered an experimental

[8] https://www.wa.gov.au/service/environment/environment-information-services/pfas-investigations-western-australia

methodology; scenarios can be simulated under different environmental conditions, and we can conduct sensitivity analyses to better understand the system under investigation (Salmon et al., 2017).

As outlined above, capturing the physical limnology of the pit lake is critical to understand seasonal mixing that can influence water quality. While some studies have predicted physical limnology based on geometry (e.g. pit depth and aspect ratio), a consistent relationship between geometry and physical limnology that applies to a wide range of pit lakes has not been found (Castendyk et al., 2015). Numerical stratification models, such as DYRESM provide more detailed predictions of future pit lake physical processes and can be implemented on specific sites with relatively easy to gather information (e.g., local meteorology, inflows and outflows, pit bathymetry (see Oldham (2014) for a complete listing).

In contrast, due to the complexity of lake process interactions, long term predictions of pit lake water quality require more sophisticated coupling of water quality models that incorporate chemical mass balance, chemical equilibria and kinetics, and lake ecology (e.g. CAEDYM), with lake stratification models (e.g. DYRESM, Salmon et al. (2017)). The water quality models require significantly more input data (Oldham, 2014) but are essential to accurately predict future lake conditions. Numerical models have been used to investigate the response of pit lake water quality to remediation measures, such as nutrient or organic carbon addition to stimulate alkalinity generation from sediments (Wendt-Potthoff et al., 2002) or to remove metals from the water column (Crusius et al., 2003). A comparison of numerical simulation results with carefully conducted field and laboratory observations allow our conceptual models to be rigorously tested under controlled scenarios.

Stand-alone geochemical models also exist that start with development of a conceptual model that describe chemical mass balances of the pit lake, and will typical include both wall-rock runoff (release of solutes from the pit wall due to rain) and wall-rock leaching (solute release to groundwater inflow), although they are not always differentiated in models (Castendyk et al., 2015). Approaches to quantify these processes are use of laboratory and field tests, leaching and batch tests and modelling mineral dissolution kinetics and oxidation modelling. Geochemical models such as PHREEQC can be validated by comparing with experimental leaching results. It is important to note however, that these models rarely capture lake processes or stratification dynamics, however PHREECQ functionality was implemented into a coupled hydrodynamics – water quality - ecology model by Salmon et al. (2017); comparison with pit lake validation data from Lake Kepwari, in Collie Western Australia, was promising.

Projections of pit lake water salinity are sometimes over-estimated because they are mainly based on evaporative enrichment but neglecting salt decay rates from spoil piles. For example, salinity export was found to be over-estimated by between 2.5 and 12.5 tonnes for a 100-year period in a Queensland confidential case study (Edraki et al., 2019, Edraki et al., 2021). A classification scheme, bench-top funnel leaching tests, medium scale leaching procedures, and a numerical model were developed to help the mine sites assess the potential release of salts from spoil piles. These results are affected by the length and timescales used in the tracer tests and may not be directly transferable to real spoil dumps; to address that, larger-scale tracer tests are needed. The authors suggest the next step would be to validate models through field monitoring of spoil piles to better understand the spatial and temporal distribution of moisture (and water tables) inside the piles.

5.4 Downstream impacts on surface water and groundwater quality

In climatic zones where evaporation markedly exceeds precipitation, mine pit lakes may act as groundwater sinks post-closure and any outward movement of mine-influenced water from the pit remains minimal. However, in wetter climates, pits may become through-flow lakes that could adversely affect groundwater and surface water quality downgradient of the pit. Adverse effects may thereby occur downstream of pit lakes or backfilled pits.

In the case of a pit lake, discharge of high salinity water into surrounding aquifers may occur due to evaporative enrichment of lake waters. Leaching of pit walls may impact pit lake waters and downstream drainage, depending on the rock types hosting the deposit (Plumlee et al., 1997). Similarly, groundwater throughflow through backfilled pits may be impacted depending largely on the nature of the filling material. For instance, an abundance of sphalerite in many metal deposits, can lead to zinc being the dominant trace metal in drainage waters from such deposits, while arsenic-rich deposits have arsenic-rich waters, and so on. Filling with acid-generating waste material, i.e. waste containing residual sulfide minerals such as pyrite (FeS_2), galena (PbS), sphalerite (ZnS) or arsenopyrite (FeAsS), may result in leakage of elevated TDS to groundwater characterised by low pH and elevated metal and metalloid concentrations. If, however, backfill material comprises silicate minerals, the effect on water chemistry is primarily an addition of cations (e.g. Na^+ from the dissolution of Na-feldspars; Ca^{2+} released from plagioclase, hornblendes and pyroxenes) and silica and, as most silicate weathering reactions consume acid, also an increase in pH. Thus, the geology of the mined deposit and/or the backfill will govern the alterations to downstream water chemistry to a large degree. A comprehensive review of the geological controls on the composition of drainage waters can be found in Plumlee et al. (1997) and Nordstrom (2011).

A plume emanating from a backfilled pit will move downstream with the advective velocity of groundwater. With time and distance from the pit, the contaminant plume disperses due to subsurface heterogeneities and diffusion, thereby lowering concentrations by spreading the contaminant mass over an ever-larger area. The nature of the contamination may induce further reactions within the aquifer altering the chemistry of the groundwater. For instance, the mobility within the aquifer of trace metals such as lead, copper, zinc, cadmium, and nickel, and metalloids such as arsenic, emanating from a pit, may be controlled to a large extent by sorption reactions. Binding to variably charged mineral surfaces such as iron oxides and organic matter may control trace metal fate in groundwater. At circum-neutral pH, these sorption reactions lead to trace metals remaining largely immobile. However, under acidic conditions, trace metals are mobilised and result in elevated metal and metalloid concentrations in groundwaters.

Besides sorption reactions, leachate from a backfilled pit will likely induce redox reactions which exert an important control on groundwater chemistry. For instance, the addition of dissolved organic matter that may leach from the backfill may induce reductive dissolution of minerals and subsequent release of trace metals. Equally, if the flooded backfill material provides additional oxidants, such as oxygen, to the groundwater throughflow, precipitation of metal oxides may occur. Low pH discharge from the pit may be neutralized within the aquifer. If the geology comprises carbonate minerals, the pH of the groundwater is likely to be buffered because of the fast dissolution kinetics of carbonates. Typically, calcite and aragonite are the most reactive carbonate minerals, and therefore the most effective lithology at consuming acid. Other carbonates such as rhodochrosite, dolomite, and magnesite are less reactive and therefore less effective at buffering acidic waters.

There are numerous examples of surface water and groundwater contamination from discharge from pit lakes and backfilled pits. Arsenic contamination of groundwater and surface water occurred downstream of the Harvard pit lake, California (USA), a former gold mine that was closed in 1994 (Eary and Castendyk, 2009). The Berkeley pit lake, Montana (USA), has been a source of highly acidic discharge following mine closure. As pit lake levels rise during groundwater recovery post-mining, groundwater contamination has become a serious concern at the site (Kuznetsova and Ivanov, 2019).

Where pit lakes are connected to the surface water drainage system, downstream surface water impacts can occur. In some cases, positive impacts to the surface water system can result. Lake Kepwari, in Collie Western Australia, was breached by a diverted eutrophic river after closure. The inflowing pulse of river water was fresher and higher in alkalinity than the pit lake, temporarily improving mine pit lake surface water quality and isolating colder acidic pit lake waters by stratification (Salmon et al., 2017). The diversion also reduced nutrients in the river, during passage through the pit lake (McCullough et al., 2012).

5.5 Potential uses of pit lakes, based on water quality

In a review of pit lake end uses, McCullough et al. (2020) found that end-use values typically fall within three types: wildlife, recreation or primary production. Most common end uses were wildlife habitat, fisheries, recreational, storage of water, waste storage and treatment. Almost all end uses required circum-neutral pH and low concentrations of contaminants. Appropriate assessment of long term geotechnical stability of the pit walls and any associated land forms is also important (de Bruyn et al., 2019), particularly for end uses where ongoing access to the pit lake is required.

The beneficial reuse of a water resource is governed by its water quality. Water quality parameters such as salinity, turbidity, dissolved oxygen concentration, and nutrient content may be relevant when considering the end-use of the lake. Most aquatic life forms have narrow ranges of pH tolerance. The application of appropriate water quality criteria will be determined by the proposed use. Example water quality guidelines are in Table 1.

Table 1: Examples of water quality guidelines that can be used to determine water value

Guidelines	Comment	Reference
Australian Drinking Water Guideline (ADWG)	Human health limits and aesthetic quality (colour, odour) limits	(NHMRC and NRMMC, 2011)
Livestock	Livestock drinking water quality limits related to cattle	(ANZECC and ARMCANZ, 2000)
Irrigations	Long term trigger values (up to 100 years) for irrigation of crops and pastures.	(ANZECC and ARMCANZ, 2000)
Aquaculture	Values for aquacultures and human consumption of aquatic foods	(ANZECC and ARMCANZ, 2000)
Ground or surface water in suspected or known contaminated sites	Generic assessment criteria developed by WA Department of Health. Value is generally 10 times the corresponding ADWG health value, or equal to the ADWG aesthetic value.	(DoH, 2014)
Recreational water quality and aesthetics	Due to a relatively low exposure compared to drinking water, is recommended to consider a value of 10 times the corresponding ADWG health value	(NHMRC, 2008)
Environmental protection	Trigger values for protection of freshwater saltwater aquatic ecosystems. For those parameters that are not considered toxicants, the default value guideline for physical and chemical stressors is presented instead.	(ANZECC and ARMCANZ, 2018)

In the recent study of mine pit lakes in Western Australia, a screening risk assessment was carried out by comparing concentration data from a surface water sample with water quality guidelines (Linge et al., 2021). Typically, each guideline covered 20-30 parameters but none were completely comprehensive compared to the analyte list. Generally, the parameters that most frequently exceeded guidelines were major elements (e.g. Cl, SO_4, Na, TDS), rather than toxic metals. Increasing salinity may preclude most end use options, regardless of the trace elements present. Other factors that influence the range of potential beneficial end uses of pit lakes are their proximity to communities and local infrastructure, economic diversity of local towns, i.e. are they primarily reliant on mining, or there are other industries that lessen the economic impact of closure. Maller and Mark (2001) (described in Hunt (2013)) developed a set of decision pathways to determine options for open void closure focused on climate, site geology and social settings . Hunt (2013) concluded that evaluation of beneficial end-uses of mine pits must be embedded in a wider mine closure framework with specific legislation required to guide the process, with early planning and stakeholder engagement.

A recent confidential assessment of post mining land uses in the Bowen Basin, considered long-term community priorities, planning constraints and opportunities, and collaborative opportunities for beneficial use of regional post-closure mine assets. The region's strengths include a vast network of water supply infrastructure with well-established water planning processes for water trading. Residual pit lakes from open cut mining could play a positive role and be integrated into the water supply network, an option that would improve current plans for rehabilitation of residual pit as non-use management areas. The NSW Government is considering the use of final pits in the Hunter Valley as water reservoirs.[9]

9 https://www.smh.com.au/politics/nsw/multiple-sydney-harbours-plan-to-drought-proof-nsw-with-lakes-20191217-p53knb.html

Table 2: Case studies of uses of pits

Beneficial End Use	Examples
Energy	Kidston Pumped Storage Project: A feasibility study into the construction of a pumped storage hydroelectric power plant at the disused Kidston Gold Mine in North Queensland, similar to the Wivenhoe Pump Storage scheme and Snowy's Tumut 3 scheme. Intermittent generation can therefore be stored and dispatched to the grid during periods of high demand. https://arena.gov.au/projects/kidston-pumped-storage-project/
Potable or industrial water source	Wedge Pit Lake: Used as a municipal potable water supply for Laverton (McCullough et al., 2020) Mine pit lake water frequently used for dust suppression
Aquaculture	Highland Valley Copper (Canada) and in the Alta Floresta region (Brazil) (Otchere et al., 2004)
Recreation and Tourism	Lake Kepwari https://www.dbca.wa.gov.au/news/lake-kepwari-transformed-into-major-tourism-drawcard
Wildlife Conservation	Beenup Titanium Minerals Project – rehabilitation of a mineral sands mine to permanent wetlands which is now host to rare flora species, with potential for environmental education, research and eco-tourism https://www.bhp.com/sustainability/case-studies/2018/08/closure-and-water-management-at-beenup/
Irrigation	Assessed for the Pilbara region, pilot using dewatering from Woodie Woodie Manganese Mine (DPIRD, 2017)

6. INTEGRATED AND INNOVATIVE MANAGEMENT STRATEGIES

This chapter discusses some innovative management strategies that could be implemented at mine sites to improve post-closure water quantity and water quality outcomes. Some of these strategies have already been trialled at select mine sites, while others have yet to be tested.

6.1 Introduction

While adverse environmental impacts of mining on water quality and water quantity can be significant if not carefully managed, there are strategies for reducing undesirable changes in water quality and quantity. Strategies that aim to improve water-related environmental outcomes of pit closure include:

(i) Strategies that aim to reduce the extent of the drawdown cone during mining and/or improve the rate of recovery of the water table post closure. This can have benefits in terms of water availability for GDEs and improved water quality.

(ii) Strategies that aim to reduce adverse water quality impacts on the surrounding environment (e.g., groundwater, rivers) through water treatment, mixing/dilution or promotion of in situ chemical processes.

(iii) Strategies that aim to promote improved water quality in pit lakes, for example selective placement of waste material in the pit, addition of organic carbon (e.g., municipal waste) or alkalinity generating materials into the pit, removal of reactive material from the pit side walls, landscaping of the pit and its strategic filling to optimise lake stratification regimes.

These are dealt with in Sections 5.2 and 5.3 respectively, below. Some of these strategies are currently implemented during the operational phases of mining, but also have potential to be used to improve closure outcomes. Others have been implemented for mine closure, but there are significant opportunities for improvement. In all cases, post-closure water management can be improved if planning commences during the operational phase of mining, and detailed modelling of impacts of the water quantity and quality of the mine pit and the surrounding surface water and groundwater system is needed. This will often require collection of data beyond that which would usually occur in support of mine water management operations. It is also important to note that strategies aimed at reducing the extent of water table drawdown post mine-closure might not necessarily lead to the best water quality outcomes, and strategies aimed at optimising water quality outcomes do not necessarily lead to the best water quantity (e.g., water table) outcomes. The relationship between these two, sometimes competing strategies, is discussed in Section 5.4. Resolving these conflicts requires consideration of the interaction between the different components of the hydrological landscape.

6.2 Strategies to reduce water table drawdown post mine closure

Strategies to reduce water table drawdown post mine closure fall into two categories: those that aim to reduce the spatial extent of water table drawdown that occurs due to mine dewatering, and those that seek to speed up the recovery of water levels post-closure.

6.2.1 Engineered barriers

A small number of mines have applied engineering techniques for reducing the extent of the drawdown cone resulting from mine dewatering by installing low-permeability barriers. Such techniques are inevitably required where pits border lakes or occur close to the sea or ocean, but they have also been applied to limit the spread of the drawdown cone either to protect GDEs or to prevent dewatering of geological deposits that may potentially become contaminant sources. These approaches can also have operation benefits, by reducing volumes of water that must be pumped. The most used low permeability barriers (LPBs) are soil-bentonite or soil-cement-bentonite slurry walls. Timms et al. (2013) summarise the key properties of LPB that have been installed at 12 mine sites in Australia and overseas. The barrier walls at these sites ranged between 0.8 and 3 m in width, and were up to 4000 m in length. Depths varied between 6 and 50 m, and the increased cost of deeper barriers might limit the application of this technology at some sites. To be effective, LPBs need to penetrate below the base of permeable aquifer zones. The Carrington open pit coal mine, NSW, constructed a LPB to reduce seepage from the Hunter River through alluvial aquifers overlying coal seams. The barrier was 15 m deep and 1200 m long and cost $3 million dollars to construct. In contrast a barrier wall between a lake and open pits in arctic climatic conditions northern Canada cost $1.3 billion to construct, although the high cost in this case was partly due to the difficult climatic conditions that required construction within a short 10-week period (Timms et al., 2013). In some cases, modifying the bentonite content of such barriers may allow cost reductions without impacting effectiveness(Timms and Holley, 2016). Further work is also needed on the long-term management and/or removal of these barriers after mine rehabilitation is complete (Timms and Holley, 2016). In the Pilbara, Fortescue Metals is planning a grout curtain to minimise drawdown to the west of their Solomon mine, to prevent the risk of development of acidic groundwater if geological units are exposed to air.[10]

In principle, low permeability barriers will not only reduce the extent of the drawdown cone, but also reduce the volume of water that needs to be pumped to dewater the pit during mine operations. While their cost and efficacy require further analysis, they potentially reduce water management costs during mining operations and reduce environmental impacts post-closure.

6.2.2 Managed aquifer recharge

Dewatering of open pit mines commonly result in large volumes of groundwater being abstracted from in-pit and ex-pit borefields to enable dry mining conditions. Some of this supply is utilised for operational requirements, such as dust suppression and ore processing. However, often dewatering volumes exceed operational water requirements and disposal of excess water is required, and over the last decade, managed aquifer recharge (MAR) has started to play a role in water surplus management options (Rubio and Fernandez, 2010, Windsor et al., 2012, Youngs et al., 2012). However, managed aquifer recharge may potentially also play a role in facilitating mine closure. Managed (i.e. enhanced) aquifer recharge may facilitate the accelerated return of groundwater to pre-mining or near pre-mining levels outside of the pit area and also accelerate pit filling. The former is of particular importance if the recovery of groundwater levels is linked to the rehabilitation of groundwater dependant ecosystems, e.g., the resumption of natural spring flows. These potential advantages are noted in Ministerial Statement 584 (Schedule 2) for the Hope Downs Iron Ore Mine, Pilbara, which states

[10]https://www.epa.wa.gov.au/sites/default/files/PER_documentation/solomon-iron-ore-project-sustaining-production-per_part%202.pdf

"Decommissioning strategies may involve enhancing aquifer recovery through artificial recharge of flood flows in some sub-catchments" and that closure planning should "include an investigation of the practicality of Aquifer Storage and Recovery" (Minister for the Environment and Heritage, 2002). Groundwater modelling has indicated that after closure, it could take 60 years for the water levels at the Hope 1 North Pit to return to pre-mining levels and for resumption of natural spring flow in the vicinity of the mine. Artificial recharge may potentially shorten these timescales and allow the cessation of spring flow supplementation earlier following mine closure (Johnson and Wright, 2003).

Successful implementation of MAR in mine closure, however, requires careful management of injection bore locations and volumes, to ensure that water re-circulates back into the dewatered orebody only after cessation of mining. The degree of hydraulic connectivity between the orebody and the MAR target aquifer will determine timescales of water movement and this, in turn, will dictate an optimal distance between the pit and potential injection bores (Smith, 2014, Youngs et al., 2012). Further research is needed on the potential role of MAR in mine closure, and the conditions under which it may be an effective strategy.

6.2.3 Design consideration for evaporation rate and backfill materials

When a pit is not fully backfilled and not connected to a river system, the evaporation rate from the pit lake is a major control on water table recovery after mine closure. High evaporation rates will lead to lower final water levels, both within the pit lake and the surrounding areas. Few studies have specifically examined the effect of evaporation on pit lake water levels. One exception is the Mine Closure Plan for the Jack Hills mine, located in the mid-west of WA. Here, groundwater and pit lake modelling was used to estimate final equilibrium pit levels under average rainfall and evaporation conditions, high rainfall and low evaporation (95th percentile of monthly values for rainfall and 5th percentile for evaporation) and low rainfall and high evaporation (5th percentile of monthly values for rainfall and 95th percentile for evaporation) scenarios. It was found that the equilibrium water level in the pit would be 442-450 m AHD for average conditions (assuming typical runoff coefficients), 481-489 m AHD for high rainfall - low evaporation and that the pit would be dry for low rainfall – high evaporation conditions. (The pit base was located at 412 m AHD.) The time to reach the new equilibrium state was 50 years for average conditions and 138 years for high rainfall low evaporation conditions.

Oldham (2014) presented a sensitivity analysis using the DYRESM-CAEDYM numerical model. The model was implemented at an (unnamed) mine site in the Pilbara, to explore the impact of wind sheltering on evaporation rates and therefore on future pit lake water levels. When 100% wind speed was used, DYRESM-CAEDYM predicted a water level about 15 m lower than predicted by groundwater modelling. When wind was reduced to 10% (i.e. assuming a 90% sheltering), the model demonstrated that sheltering not only reduced the wind field over the lake, but also increased stratification due to lower wind mixing, and increased surface water heating. Increased evaporation due to higher surface water temperatures countered the decreased evaporation due to lower winds. Overall, under 90% sheltering, the lake water level was 3 m higher after 30 years. This study shows the importance of accurately modelling the feedback of wind sheltering into lake stratification dynamics. It also should be noted that long term stable stratification is generally desired to isolate poorer quality bottom waters.

While we are not aware of any studies that have considered manipulation of evaporation rates to create desirable water level conditions, either within the pit lake or regionally, such a strategy might be possible. Pit landscaping and/or partial pit backfill options might consider designs to reduce wind speed across the pit lake, and hence reduce evaporation rates. As shown by Oldham (2014), a detailed exploration of the impact of lower wind speeds on lake physics is required to ensure that the expected result would eventuate. Other innovative strategies might include the use of shade balls, which are small plastic spheres floated on top of a pit lake, originally designed to prevent birds from landing on toxic tailing ponds.[11] Shade balls would both reduce wind speed and provide thermal insulation to lake surface waters, and should decrease evaporation, leading to increased groundwater recovery.

Figure 13: Shade balls on a small reservoir. Image from https://energyvulture.com/2016/02/14/shade-balls-roll-their-way-into-the-spotlight/

[11] https://en.wikipedia.org/wiki/Shade_balls

6.2.4 Temporary diversion of river water into pits

There are several examples of empty pits being flooded with surface water at the completion of mining to rapidly raise the lake water level. There are four main reasons for rapidly filling pit lakes: to reduce slope erosion and the risk of landslides; to reduce the risk of acidification from exposure of sediments and waste materials to oxygen; to create permanent density stratification in the lake and isolate hypersaline poor-quality waters in the pit bottom; and to rapidly return the groundwater system to its final post-closure state (Schultze et al., 2011). If the waters used to flood the pit are high quality, then rapidly filling the pit can also create an initial pit lake water quality benefit. On the other hand, initial flooding of the pit results in the pit water level being higher than the level of the surrounding groundwater, which will cause pit water to enter the groundwater system. If the water quality in the pit lake is poor, then there is potential to contaminate the groundwater system (Niccoli, 2009). Also, while this approach has several potential benefits for pit lake water quality, these need to be carefully balanced against the potential adverse environmental impacts on river systems. However, the data required to carry out such assessments are not always available.

Rapid filling of pit lakes has been practiced in Germany for more than 50 years (Schultze et al., 2011). Pusch and Hoffmann (2000) describe a case where rapid filling of open lignite mine pits with water from a nearby river was used in Germany. Dewatering of the pits had resulted in more than 2 m of drawdown over an area of 2500 km^2, and the time for natural refilling was estimated to be at least 30 years. Water from the river was used to increase the rate of recovery but resulted in a reduction in river flow with severe changes in river ecology over 230 km of river length and reduction in fish populations along the affected section of river.

In the German climate, the occurrence of floods is not sufficiently predictable for them to be considered as a reliable means for rapidly filling pits and use of floodwaters can also result is erosion risks. For this reason, diversion rates in the order of a few kL per second are considered a better option, and permanent diversion of rivers into pits (with mechanisms to reduce flows during periods of low river flow) is common. In some areas, the use of mine dewater from active pits are used to rapidly fill pits that are being closed (Schultze et al., 2011). For example, the pit lake Altdöberner See is part of the 'Lausitzer Seenland' (Lusatian Lake District) and will be the largest lake in Brandenburg, Germany. The lake was created by filling the excavation pit of the mine 'Greifenhain' with water. Flooding of the mine pit started in 1998 and ended in 2007. Water collected from the dewatering of a separate mine pit was treated through the addition of lime and oxygenation and used to flood the mine void. In 2007, the lake volume was 80 GL. The lake level continues to rise by around 1.3 m/y through groundwater inflow and direct rainfall, with the final water level expected to be reached between 2021 and 2025 (Schultze et al., 2011). This example has proven to be a remarkable success at many social, environmental and economical levels (Lienhoop and Messner, 2009, Lintz et al., 2012, Schultze et al., 2011)[12].

In one of the more spectacular examples of rapid lake filling, in 1996 the Island Copper pit in British Columbia, Canada, was filled over 6 weeks via a sidewall cut through to the ocean. Around 300 GL of seawater was discharged into the pit, creating a lake that was over 200 m deep (Stevens et al., 2005). Since initial filling with ocean water, a surface layer of freshwater has accumulated, creating permanent stratification that has isolated poor quality deep waters. There is ongoing investigation into lake hydrodynamics that may, over time, erode the pycnocline and eventually expose the poor quality water (Stevens et al., 2005).

[12] https://www.nationalgeographic.co.uk/travel/2019/03/germany-lifting-black-clouds-lusatia

In environments with pronounced wet and dry seasons, the use of high river flows during the wet season may potentially allow rapid pit filling with less environmental consequences for the river systems. In northern Australia, Enterprise Pit Lake (NT) was rapidly filled by artificial diversion of Pine Creek. The design included a floodway cut from Pine Creek, so that pit lake is essentially an off stream storage that is recharged each wet season from Pine Creek high flows (Boland and Padovan, 2002). In this case, the rapid filling was designed to prevent oxidation of sulphide material in the pit, but it will also have contributed to a rapid groundwater recovery surrounding the mine site.

For the Corani sliver-lead-zinc mine in Peru, (Bickford and Breckenridge, 2013) discuss the use of “surface water diverted from various parts of the project site” to rapidly recharge groundwater after backfill of the pit. Modelling suggested that it would be necessary to divert 19 ML/day to completely saturate the backfilled pit within a single year, although a slightly longer timescale might be more feasible with available surface water resources. Rapid filling of the mine pit is desirable to minimise acid-generation from waste materials, and the very low hydraulic conductivity of the bedrock material (10^{-2} – 10^{-3} m/day) allows this to occur relatively rapidly.

In the Latrobe Valley, plans were developed to hasten groundwater recovery by pumping groundwater from Latrobe Valley aquifers and diverting surface flow from the Latrobe River system into coal mine pits. In the case of the Hazelwood mine, one of the options considered involved filling of the lake with 25 GL of surface water and 17 GL of groundwater per annum for 16 years commencing in 2021. Even with this initial filling, complete recovery of the groundwater is predicted to take more than 100 years (Gresswell et al., 2019). However, questions have been raised about the impacts that the scheme would have on the surface and groundwater systems, with water loss to evaporation from the pits of around 5 GL/y for Hazelwood mine, but 15 GL/y for the three mines in the area (Loy Yang, Hazlewood and Yallourn mines), potentially higher under a future drier climate. There are completing demands on water resources in the area, including irrigation and baseflow to rivers. The scheme has not yet been approved and remains controversial.

Szcepinski (2001) compared rates of groundwater recovery without intervention, and with supplementation of water from outside the cone of depression, for a large lignite mine in Poland. A groundwater model predicted that natural recovery (no supplementation) would take about 60 years, but the recovery time was reduced to 18 years with supplementation of 4 m^3/s from another (unspecified) water source.

Where open pits occur in areas of high aquifer permeability, filling the pit cannot be separated from recovery of the groundwater. If the drawdown cone extends for several kilometres, then the volume of water required to fill the pit and the drawdown cone will be very large – much greater than the volume of the pit itself. Thus, while the potential to rapidly fill mine pits using river floodwater may be effective in areas of low aquifer permeability, further research is needed on its efficacy in areas of high aquifer permeability. Detailed balancing of pit lake water quality benefits and possible adverse impacts on river systems is also required.

6.3 Strategies to improve water quality outcomes

For through flow pit lakes, a long-term concern is the down-gradient movement of poor-quality plumes in the groundwater. Contaminant migration in groundwater can potentially impact rivers and creeks, and other groundwater-dependent ecosystems. Lake processes can theoretically be manipulated to isolate and/or improve water quality of lake water, and the potential for such remediation has been explored internationally.

6.3.1 Reducing groundwater quality impacts of backfilled pits

Where the volume of ore is small compared with the volume of the extracted material, mine pit closure strategies may involve sequential backfilling or postmining backfilling of open pits. Backfill material is commonly mine "waste", i.e. spoil, tailings, overburden. Once groundwater levels recover, groundwater throughflow will usually occur and can result in contamination of the groundwater downgradient. The latter is dependant mainly on the degree of acid producing material in the backfill. If reactive materials are present and exposed, acidic discharge with elevated metal and metalloid concentrations may occur to the groundwater system as a result of the oxidation of sulfides contained within the backfill. Prevention and mitigation of contaminated leachate from backfilled pits has received considerable attention and several case studies on the geochemistry of waste rock, their acid generating potential and possible mitigation options can be found in the scientific literature (e.g., Chapman et al., 1998, Villain et al., 2010, Younger et al., 2002).

To reduce oxidation of reactive mining waste, pits may be backfilled only to below the pre-mining water table. Upon inundation, oxygen supply to the backfill is reduced substantially as the rate of diffusion of oxygen through water is approximately 10 000 times slower than through air. Thus, a water cover provides an effective measure to reduce the rate of sulfide oxidation. Any backfill material placed above the post-mining water table should consist of inert waste rock (DFAT, 2016).

Where a water cover cannot be maintained throughout the year, other covers have been proposed. For example, low permeability barrier layers that limit the oxygen supply to potentially acid-generating waste have been used where the groundwater surface is very low in the waste deposit. Covers that are not based on the limitation of oxygen diffusion rates, but on consumption of oxygen by organic material, or inhibition of oxidation reaction rates have also been developed. Reviews of different cover types can be found in DFAT (2016) and Johnson and Hallberg (2005).

Alkaline amendment of backfill material with, e.g., lime or carbonates is a further option to mitigate water quality impacts of backfilled pits. The control of acid mine drainage by adding neutralizing capacity has been demonstrated, e.g., at Brukunga Mine, South Australia, where pyrite and pyrrhotite were mined by open pit methods between 1955 and 1972. Here, long term remediation of the mine is based upon mine pits being backfilled with compacted, saturated co-disposed waste rock, tailings and crushed limestone (4-6% by dry mass) (Cox et al., 2006).

Amendment of backfill material with organic material to induce strongly reducing conditions is aimed at removing dissolved oxygen and enhance microbial iron and sulfate reduction. McCullough and Lund (2011) demonstrated the feasibility of using sewage and organic garden waste, as these are often available at mine sites or from nearby service towns, to successfully neutralise acidic pit lake water from tropical, North Queensland, Australia.

All the above methods are regarded passive control mechanisms aimed at prevention and mitigation of acidic mine discharge at the source. This is generally considered the preferable option. Where it is not feasible it may become necessary to collect and treat mine water.

6.3.2 Improving pit lake water quality

There has been considerable research over the last two decades on whether pit lakes can be manipulated, whether passively or actively, to develop improved water quality. This work has for the most part focussed on the effectiveness of organic carbon additions to improve alkalinity and increase lake water pH (e.g., Axler et al., 1996, Klapper and Geller, 2001, Lessmann et al., 2003, Wielinga, 2009). Most of the European studies concluded that over the longer term, passive remediation was less effective, and active remediation was too costly.

A number of bioremediation strategies have been trialled in Australia to improve pit lake water quality. A trial of bioremediation of pH in acidic pit lakes in the Pilbara used spoilt hay bales to promote the growth of sulphate reducing bacteria (SRB), with some indication of efficacy on a limited (7 month) trial (Green et al., 2017). Sewerage sludge was more successful at promoting SRB, in part due to its ability to neutralise acid mine drainage (Kumar et al., 2013).

In Australia, Lake Kepwari, a former coal mine in the Collie region, WA, was filled with water from the Collie River over three seasons and remains a surface flow-through system. Salmon et al. showed that seasonal discharge into the pit of Collie River water with high organic carbon loading was not able to sustainably raise the pH above 4, due to ongoing buffering by dissolved aluminium complexes (Salmon et al., 2008, Salmon et al., 2017). Continued seasonal inputs of riverine water are expected to benefit pit lake water quality. Manipulation of surface water inflows and outflows to pit lakes can also be used as a tool for improving water quality, and there is potential to optimize such engineering designs.

Mine pit lakes with high TDS water could be treated with conventional water treatment technologies such as reverse osmosis, although barriers to their use include high energy requirements, and appropriate disposal of the brine. Thiruvenkatachari et al. (2016) evaluated the application of an integrated forward osmosis (FO) and reverse osmosis (RO) system with three different coal mine waters, containing various concentrations of sulphates and silica that are generally associated with scaling and fouling of membrane systems. The combination of FO with RO provided a better performance than FO-only or RO-only systems. This technology may offer mine sites a readily controllable onsite water treatment method that can be tailored to each site's water management requirements. Because the brine is used to keep a high salt concentration in the draw solution for the RO system, this integrated treatment technology produces small volumes of brine (less than 1%).

Other examples where pit backfilling has been used for novel approaches include the Ranger Uranium Mine (NT), where management of large volumes of process water is being addressed by using a brine concentrator to produce a distillate of suitable quality to discharge to the environment, while the brine is stored in the underfill of a closed mine pit (ERA, 2019).

6.4 Optimising water quality and water quantity benefits

Strategies aimed at optimising water quality outcomes do not necessarily lead to the best water quantity (e.g., water table) outcomes. For example, previous researchers have advocated maintaining pit lakes as terminal groundwater sinks to minimise offsite movement of contaminants in the groundwater systems (e.g., McCullough et al., 2012). However, this requires that the water table is maintained below its natural level (either through lake evaporation, or direct pumping), which will result in a lower regional water table and potential impacts on groundwater-dependent ecosystems. It is important to balance the need for water table recovery with the desire to maintain good pit lake water quality and minimise the potential for offsite impacts, either through the groundwater or surface water systems. The balance will depend upon the nature and proximity of groundwater-dependent ecosystems to the mine site, and the threat that is posed to these by lower water table levels and groundwater and surface water contamination. This requires models that consider water table drawdown and pit lake physical processes, as well as processes impacting pit lake, surface water, and regional groundwater water quality. Optimising these models to provide the best futures requires a detailed understanding of impacts on all components of the water cycle, but also balancing water outcomes other environmental, social and economic considerations. While our models are not yet sufficiently developed for this purpose, this must remain the ultimate goal.

7. CONCLUSIONS AND RECOMMENDATIONS

This report provides a review of water quantity and quality issues associated with the closure of below water table open pit mines. It includes an overview of simple modelling of groundwater recovery, and a brief review of more complex numerical modelling of pit lake processes. To our knowledge it is the first review that integrates waterscape evolution after mine closure, encompassing regional groundwater recovery, the stabilisation of pit lake water levels, and the development of pit lake water quality through time. Some of the key conclusions of this work are:

1. The area surrounding the mine pit that is subject to reduced groundwater levels will continue to increase after cessation of mine dewatering, before eventually decreasing. Thus, it is possible that ecosystems that are not affected by drawdown during the mine's operational life might still be affected after mine closure. Identification of GDEs and prediction of likely impacts due to water table drawdown remains a challenge for water managers. These issues are not unique to mine sites.
2. Accurate prediction of water table drawdown after mine closure requires detailed information on the aquifer system surrounding the mine, including geological structures that might act as barriers to groundwater flow. Such information is required beyond the region that is directly impacted by water table drawdown during mine operations.
3. The water quality of pit lakes will be largely determined by the rate and chemistry of groundwater flowing into the pit, runoff from the pit walls and lake catchment, the time required for the pit to fill and the evaporation rate. Pit lake water quality may also be affected by the composition of waste rock and its proximity to the pit. For partially backfilled pits, in-pit storage of waste rock will affect pit lake water quality. If evaporation exceeds precipitation and runoff into the pit, then concentrations of dissolved solutes in lake water will increase over time. For fully backfilled pits, groundwater quality in the vicinity of the mine will be affected by the chemical composition of the backfill materials that are used (including any waste rock), and infiltration properties of the backfill materials.
4. Understanding pit lake evaporation rates, lake stratification cycles and how groundwater and surface water inflows to the pit changes over time, are essential for accurate prediction of the final pit lake water level, the time for the pit lake water level to stabilise, and the development of pit lake water quality. There are numerous examples of the use of numerical groundwater models for estimating of groundwater recovery rates, but few that are linked to validated numerical lake models, incorporating estimates of pit lake evaporation. Pit lake models that are not coupled to groundwater models, often make simplifying assumptions about groundwater inflow to the lakes and how this changes with time. The importance of such simplifying assumptions is rarely tested. How climate change impacts interactions between pit lakes, regional groundwater and surface water, and water-dependent ecosystems has rarely been explored in the published literature.
5. Rapidly filling mine pits with water can have advantages, immediately for pit lake water quality, and then by increased groundwater recharge and availability to groundwater-dependent ecosystems. Rapid pit filling by diversion of surface water into pits can be effective where pits occur in areas with low aquifer transmissivity. Managed aquifer recharge may have a role in rapid recovery of groundwater levels in areas with higher aquifer transmissivity.
6. Contamination of mine sites, and of mine pit water, with mining operation chemicals such as PFAS has not yet been comprehensively investigated. Assessments of water quality for post-mining use of pit lakes will require a more comprehensive assessment of persistent chemicals.

Knowledge gaps and recommendations

Post mining landscapes will always be different to pre-mining landscapes. The challenge is to understand the impacts of mining-related activities, and how these impacts can be altered or ameliorated by management. Finding the optimal outcome requires a better understanding of the system, and how different components of the system interact, than we currently possess. In the following we list several knowledge gaps, and recommendations to improve our understanding. Many of the knowledge gaps are not specific to mine hydrology and mine closure planning. The knowledge gaps and recommendations are grouped into four areas: hydrological processes, modelling, exploring new solutions and data sharing.

Hydrological Processes

We do not have a good understanding of how groundwater systems interact with pit lakes. Studies of pit lakes usually make simplified assumptions on their connection with groundwater, and groundwater studies usually simplify processes in pit lakes. We do not have a good understanding of how pit lake geometry affects groundwater flows to pit lakes over time after mine closure, how the pit lake evaporation rate changes as the pit fills. We do not have good understanding of short and long-term changes in surface hydrology in post-mining landscapes, and how this affects pit water balances.

Recommendation One:

Carry out generic modelling and modelling of existing mines to examine how groundwater, surface water and pit lakes interact post mine closure. This modelling should include predictions of rates of water level recovery, steady state pit lake water levels, and time for stable pit lake water levels to develop. The analysis needs to consider how the pit geometry and aquifer parameters affect these flows. It also needs to consider how the evaporation rate changes as the pit lake fills. Such models could also examine how post-mining land use and climate interacts with pit water balances via surface flows, groundwater flows and water re-use. It should include sensitivity analysis to all model parameters.

Recommendation Two:

Establish best practice models for prediction of evaporation rates from pit lakes.

Modelling

There is an opportunity to improve the water balance and water quality modelling that is performed for pit closure. Currently, groundwater, surface water and pit lake models are rarely linked, and unrealistic assumptions are sometimes made to simplify the modelling task. Model areas may not always incorporate the entire area potentially affected by drawdown during mine operations and mine closure. Today, best practice modelling includes both sensitivity analysis and uncertainty analysis. While there are numerous examples of optimisation of mine dewatering solutions (Peksezer-Sayit et al., 2015), but we are not aware of any optimisation of water management for mine closure.

Recommendation Three:

Investigate how pit lake models can be linked with groundwater and surface water models. While it will always be necessary to make simplifying assumptions about some components of the water balance, when such assumptions are made they should be explicitly documented, and the effects of the assumptions should be explored as part of sensitivity analysis. In particular, links between pit lake models, groundwater models and surface water models are essential to accurately predict how pit lake water levels and water chemistry change over time. Models that predict mine closures outcomes at particular sites should include realistic sensitivity and uncertainty analysis.

Recommendation Four:

Establish a guidance document detailing advantages and limitations of different numerical models that can be used during mine closure planning, to predict the post-closure waterscape evolution. The document should emphasise the extent to which different models link surface water, groundwater, and pit lake water balance and water quality, with life-of-mine water management and mine closure planning. A key outcome would be a decision-tree tool to guide the selection of models.

Exploring New Solutions

Several innovative solutions have been proposed for improving post-closure outcomes for pit lakes and backfilled pits. These include managed aquifer recharge and rapid initial filling of pits with water, low permeability grout walls, engineering design to reduce evaporation rates and/or promote infiltration through backfilled pits, and geochemical interventions to improve water quality and prevent lake acidification. Some of these potential solutions have been applied at individual mine sites, but we do not have a general understanding of the conditions under which these methods will be most effective.

Recommendation Five:

Carry out generic modelling to examine the potential for managed aquifer recharge, rapid filling of pits with water and low permeability grout walls to improve mine closure options. Examine how pit lake evaporation and the hydraulic properties of and infiltration rates through pit backfill materials impacts on post-closure water tables, and hence the potential for engineering design to impact these variables. Such generic modelling could then be applied to demonstration case studies that can be validated with field data, exploring potential management options. Demonstrating the applicability of generic models to individual mine sites will demonstrate their utility for mine closure planning to consider and assess different options more easily.

Recommendation Six:

Carry out pit lake water balance and water quality modelling to assess potential beneficial uses of pit lakes in different environments. This should include assessment of the effectiveness of different geochemical interventions to improve pit lake water quality, particularly focussing on pH and salinity amelioration. Given the likely requirement for on-going management, study sites should be identified in localities where an ongoing social or economic benefit from the pit lake can be identified.

Monitoring and Data Sharing

Long-term monitoring of pit lakes (water levels and water quality) is essential for improving our understanding of lake dynamics and chemical processes. Comparison of model predictions with monitoring data, from both the modelled pit lake and nearby existing pit lakes, is also key for evaluation of models. Access to information is critical for planning the best future outcomes. As much as possible, monitoring data should be publicly available.

Individual mines are part of a large hydrological system, and it is critical that planning for mine closure includes information on all impacts that will affect the hydrological system and downgradient terrestrial and aquatic ecosystems. Mining companies are investing in technical studies to identify potential impacts from final pits but typically on a mine-by-mine basis. Assessment of the environmental, social and economic benefits that could be achieved at the regional scale is required, and this requires sharing of information.

Recommendation Seven:

Long-term monitoring of pit lake dynamics and lake, groundwater and surface water quality and ecology should be included in closure plans. The value of using monitoring data from nearby existing pit lakes to evaluate model predictions for new mine pits should be assessed. More case studies of innovative closure options should be publicly documented and full-scale pilot studies should be established.

Acknowledgements

The authors are indebted to the project steering group and other industry representatives for constructive advice, access to reports and other data, and reviews of early versions of this report. This group included Shawan Dogramaci (Rio Tinto), Kirsty Beckett (FMG), Robert Hughes and Ursula Salmon (DWER, WA Government), Matt Goode (Golder), Gabor Bekesi (DEM, SA Government), Jasmine Gale (South32) and Craig Lockhard (BHP).

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