MEMORANDUM

TO	Emily Heffernan	**DATE**	30 November 2015
CC			
FROM	Dr Clint McCullough	**PROJECT No.**	1542819-002-M-Rev0

HAZELWOOD MINE FIRE INQUIRY – TERMS OF REFERENCE 8 AND 9

Dear Emily,

This memorandum details responses to the questions raised in your letter dated 18 November 2015 (Annexure A).

I am an environmental consultant with especial expertise in mine water management at mine closure. I also work as an academic and research scientist through adjunct positions at both University of Western Australia and Edith Cowan University. I am a recognised international expert in mine pit lakes with strategic mine project planning experience around the world. I have over 20 years' experience with mining rehabilitation across Australasia, Asia and the Americas. I have published over 90 peer-reviewed papers and book chapters, with a number of highly-cited leading mine closure practice review and opinion papers for pit lake issues.

A copy of my CV in brevity is attached as Annexure B.

This technical memorandum has been prepared in response to two letters received from King & Wood Mallesons received on 17 October 2015 and 18 November 2015 (Annexure A), for the purposes of the Hazelwood Fire Mine Enquiry. This memorandum provides advice in response to the final report of Jacobs Australia Pty Limited (Jacobs) dated 16 November 2015 (the Jacobs Report) which was provided by Jacobs to the Hazelwood Mine Fire Board of Inquiry, for the purposes of Terms of Reference 8 and 9 (TOR 8 and 9). Further information relevant to the Inquiry has been forwarded to me by King & Wood Mallesons as below (Table 1).

Given the very short period of time in which to prepare an expert statement upon the Jacobs Report and its appendices, this technical memorandum is necessarily brief and from a high level perspective.

Table 1: Documents received from King & Wood Mallesons.

Date received	Document
5 November 2015	■ GHD (2015). Hazelwood Groundwater Modelling Report. ■ GHD (2015). Geotechnical Stability, (draft findings only) Part 1 and Part 2.
17 November 2015	■ James Faithful's Witness Statement together with Annexures.
22 November 2015	■ Witness Statement of Ronald Mether, Energy Australia Yallourn, without annexures.
27 November 2015	■ HRL (1998). Factors affecting water quality in a flooded brown coal mine. ■ PanTek Solutions (2003). Water quality in the Hazelwood Mine lake.

In this memorandum I refer to various examples of pit lakes, I use the term 'pit lake' commensurately with established mine water literature definitions, namely that "*Pit lakes form after the closure of open pit mines that extend below the pre-mining water table and reflect permanent modifications to hydrologic systems resulting from mining*" (Castendyk & Eary, 2009). Pit lake characteristics all vary amongst and even within regions in terms of depth and backfill extent, volume and depth.

1.0 QUESTION 1

Is the approved model for the final rehabilitation of Hazelwood Mine detailed in the Work Plan Variation (of a partial pit lake within a lowered landform) (the Approved Final Rehabilitation Model), a feasible and appropriate model for final rehabilitation from the perspective of:

(a) achieving a safe and stable final landform; and;

(b) returning the Mine site to a condition which will enable future beneficial land use and which will complement the surrounding environment?

1.1 Achieving a safe and stable final landform

I believe that the closure concept of a partial pit lake within a lowered landform is a reasonable closure strategy for the Hazelwood Mine.

The fire risk in Latrobe Valley coal arises from either external ignition sources or spontaneous combustion when coal is exposed to air drying and oxygen. Coal measures above the water line are proposed to be covered by overburden in the approved work plan. Submergence of coal measures below the final proposed pit lake water level will prevent self-oxidation and external ignition.

1.2 Returning the mine site to a condition enabling complementary beneficial land use

A large lake formed upon closure and rehabilitation of the Hazelwood Mine void is incongruous within the regional context of the Latrobe Valley. However, recognition must be given to two key points, namely that;

1) a mine pit lake is an inevitability; and,

2) the Hazelwood Mine void rehabilitation plan must be considered in context of other regional mine closures.

These points are discussed in turn below.

1.2.1 Pit lake inevitability

Rehabilitating the Hazelwood Mine void as a terrestrial landform with no pit lake is an impracticable and wholly unreasonable closure option that should considered no further. As demonstrated by the very high coal:waste strip ratios of the resource there is simply insufficient overburden or other wastes to be backfilled. This is typical of both coal and iron ore mines and is the reason why these mines regularly lead to large pit lakes forming (McCullough *et al.*, 2013b).

1.2.2 Regional mine closures

The socio-environmental context in which Hazelwood Mine will close should not be considered in terms of the contemporary socio-environmental context. Rather, it should be considered in terms of a future date where adjacent mines of Loy Yang and Yallourn will also close to form large pit lakes. As such, although the physical geography of the region is not currently defined by the presence of large lakes, there will be three (or more) pit lakes following planned mine closures of all three Latrobe Valley coal projects in 2030–2040. Following these cumulative landscape alterations, the post-mining regions will become transformed into a pit lake districts *sensu stricto* McCullough and Van Etten (2011).

For example, the following table demonstrates examples of pit lake districts internationally (Table 2).

Table 2: Examples of pit lake districts internationally (after McCullough and Van Etten (2011)).

Lake District	Country	Number of lakes in District	Reference
Athabascan Oil Sands region	Canada	0 current (26 proposed)	(Charette & Wylynko, 2011)
Borská Nížina lowlands	Slovakia	11 current	(Otahel'ová & Oťahel', 2006)
Central German and Lusatian districts; Rhenish district	Germany	370 current; 205 current	(Schultze *et al.*, 2013a)
Collie Lake District	Australia	13 current (more proposed)	(Kumar *et al.*, 2013)
Iberian	Spain	22 current	(Sánchez-Espanã *et al.*, 2008)
Łęknica	Poland	>100	(Żurek, 2012)

2.0 QUESTION 2

Given the progression of the mining operations at the Hazelwood Mine, and the extent of progressive rehabilitation and in-pit dumping of materials to date, are there any reasonably practicable final rehabilitation models for the Hazelwood Mine, other than the Approved Final Rehabilitation Model?

2.1 Other practicable final rehabilitation models

A number of international, national and state guidelines are available to provide direction for mine closure planning of mine voids that will results in pit lakes. These voids are typically relatively deep below regional groundwater levels and/or have a high coal:waste strip ratio. Sometimes pit lakes are the preferred closure outcome as backfilling can lead to significant environmental risk from degrading regional ground water quality (McCullough *et al.*, 2013b).

Albertan guidelines for the Canadian Oil sand Region of Athabasca recommend that pit lakes be *reclaimed* to maximise end use opportunities, seeking new end use values rather than attempting to fruitlessly attempt to recreate the previous terrestrial landform and associated land use values (CEMA, 2012).

Commonwealth guidelines for pit lake closure also advise that pit lake closure landforms may substantially modify future land use options and mine closure risks at a landscape scale (DITR, 2015). However, these guidelines also indicate that pit lakes are frequently the only closure strategy as a result of no backfill undertaken due to:

a) backfilling unfeasible in terms of mine planning such as waste scheduling limiting waste or waste placement availability;

b) unreasonable backfill costs;

c) insufficient backfill materials available.

Whilst most states have only very generic guidance statements toward rehabilitating mining disturbances during mine closure, Western Australia has featured guidelines for mine closure involving pit lakes since 2011 (DMP/EPA, 2011). The most recent *Guidelines for preparing mine closure plans* recommend pit lakes as final landforms following an assessment where long term risk is shown to be acceptable, where end use benefits are identified, or where resource sterilisation from backfill may occur (DMP/EPA, 2015).

It should be noted that in-pit dumping of a significant volume of overburden waste and fly ash has already occurred in the Hazelwood Mine pit void.

It is my opinion that a partial backfill closure strategy is clearly the most appropriate closure option for the Hazelwood Mine pit with no other practicable closure options requiring further consideration. This is based on:

1) the high cost of backfill;

2) insufficient backfill material; and,

3) partial backfill activities underway;

3.0 QUESTION 3

Has a final rehabilitation model similar to the Approved Final Rehabilitation Model been implemented elsewhere:

(a) in Australia, or;

(b) internationally?

3.1 Australian

Australian examples of mine closure commensurate with the approved work plan of the Hazelwood Mine for the Final Rehabilitation Model with partial backfill include the following.

3.1.1 Upper Hunter Valley Lake District

Coal mines of the Upper Hunter Valley District are planned to be many years away from closure. Nevertheless, regionally operators have collectively begun closure planning for voids as pit lakes under the auspices of the New South Wales Minerals Council and ahead of regulatory requirements. Pit lakes of this region will feature a range of areas, depth and shapes. Currently, most pits are still operationally utilised and pit lake planning is only at a very high conceptual planning stage that has not extended to planning of individual operations. Consequently, efforts have not yet been made by operators to reshape pit walls, cover exposed coal measures, revegetate surrounding landforms, etc.

A review of national and international grey (e.g. reports) and peer-reviewed (e.g. conference proceedings, books and industry and academic journal papers) literature was undertaking to provide examples of leading industry practice mine pit void beneficial end uses examples suitable for both dry and wet pits (lakes) (Figure 1) of the Upper Hunter Valley coal mining region. These final pit lakes will take a variety of forms, depending upon the context and final end use. Some will be completely backfilled, some partially and some not backfilled at all. Final mine closure landforms for representative mining projects have been summarised across the regional landscape to provide a snap-shot of current regional closure planning. Pit lake water quality for representative pit voids of the region was also summarised and extrapolated for future water quality expectations and issues. Recommendations of potential end uses options for the region were then made based upon their suitability in this mining industry and socio-environmental context. Study presentation, media and engagement with industry members, regulators and community stakeholders was undertaken. An open workshop undertook a Strengths, Weaknesses, Opportunities and Threats (SWOT) analysis to broadly define regional possible sustainable end uses with key stakeholders from regulators, industry and community.

Figure 1. Glencore Ravensworth West void pit lake, Upper Hunter Valley, NSW currently being used for water transfer and storage prior to conceptual closure planning and rehabilitation works.

3.1.2 Lake Kepwari (brown coal)

Lake Kepwari is located in the Collie Coal Basin, south-western Australia. The recreation and nature conservation values of the south-west are highly regarded with promotion for wildlife and recreation-based tourism by local business associations and government. The basin now has 13 pit lakes of a range of age, size and water quality, yet all are acidic due to acid and metalliferous drainage (AMD). Further, much larger pit lakes are planned from ongoing mining in the region (Lund *et al.*, 2012).

Mining of the Lake Kepwari pit began with diversion of the seasonal Collie River South Branch (CRSB) around the western lake margin and ceased in 1997. Reactive overburden dumps and exposed coal seams were then covered with waste rock, battered and revegetated with native plants.

To further reduce wall exposure and resulting acid production, the lake was rapid-filled by a predominantly saline first-flush diversion from the CRSB over winters from 2002–2008, omitting a 2001 low-flow year. Filling commenced under a licence requiring that all river pools downstream of the void were filled before water was diverted into the void.

The lake flagged as a water-based community recreation resource for water skiing and swimming (Evans & Ashton, 2000; Evans *et al.*, 2003). However, although CRSB inflow initially raised water pH to above pH 5 and lake water met recreation guidelines during filling, a failure to identify and manage ongoing acidity inputs adequately meant that water quality subsequently declined to below pH 4 (Salmon *et al.*, 2008). Although the relatively good water quality of the pit lake still lent itself to a range of potential end uses, low pH and elevated metal concentrations degraded water quality (Lund & McCullough, 2009). Consequently, proposed end uses have never been realised and, as closure criteria have not been met, the lease remained unrelinquished.

During August 2011, heavy rainfall led to the CRSB flowing into the lake and decanting downstream again. During this time, although the decant was uncontrolled, CRSB water quality end use values were not significantly impaired (McCullough *et al.*, 2013a) and lake water quality was significantly improved (McCullough *et al.*, 2012; McCullough, 2015).

An engineered lake river flow-through trialled flow-through from 2012–2014 (McCullough & Harkin, 2015) with the lake becoming neutral by early 2014 and now becoming fresher over time. Concomitant decreases in metal concentrations and increases in organic carbon and phosphorus concentrations have also occurred with an increase in aquatic biotic diversity and abundance as a result. During flow-through there has been no degradation of downstream values defined by water quality guidelines for aesthetics and recreation, livestock drinking or aquatic ecosystem protection with decreases in nutrient concentrations in the eutrophic river.

Figure 2. Collie River inflow to Lake Kepwari during the 2013 winter season.

Please note that this example has been erroneously described in the Jacob's report as open to the public use as of 2014.

3.1.3 Ngalang Boodja Mine Lake Aquaculture Project (brown coal)

Lake WO3 is a small acidic pit lake in the middle of Collie Coal Basin and is the oldest of the Cardiff sub-Basin lakes. It is maximum 8 m depth, with an area of 4.7 ha and volume of only 300 ML and the water is blue coloured with pH down to 4.2 and elevated high aluminium and nitrate nutrients concentrations (McCullough et al., 2010) (Figure 3a). It has waste materials around the rehabilitated edges which have been graded into the lake to provide lower angle slopes to the water line.

A five-year project was conducted in Collie, Western Australia by a consortium of organisations including the mining company on which the farm was built, Premier Coal, the WA aquaculture peak industry body, ACWA, a local fish processor and the local Indigenous corporation representing the Collie Noongar community, Ngalang Boodja Council Aboriginal Corporation (NBCAC) (Ngalang Boodja Enterprises Pty Ltd, 2013). The effect of the project on the lake water quality (both positive and negative) and end use values was also assessed as part of a research trial funded by the Australian Coal Association Research Program (ACARP) (Kumar *et al.*, 2011a; Kumar *et al.*, 2011b).

The aim of the project was to create an asset for the Collie Indigenous community that would act as both a source of income for community projects as well as a facility for training. The major objectives of the project were to:

- Assess the commercial viability of mine lake aquaculture by undertaking a 2.5 ha pilot project
- Build the capacity of the NBCAC to self-govern the commercial entity within five years
- Develop long-term "value to community" solutions for existing mine voids
- Build the capacity of local people to participate in emerging aquaculture industry development clusters in the region.

The project had its genesis in a series of ACARP-funded projects conducted in the late 1990s and early 2000s that were aimed at identifying the major source of the acidity in mine lakes in Collie, Western Australia and conducting pilot projects to assess different treatment technologies. Six farm ponds were used to demonstrate the biological feasibility of using the treated water to grow a local freshwater crustacean, marron, and a freshwater table fish, silver perch. Following the success of this project the water treatment system was up-scaled and a commercial-sized marron farm comprising 22 marron ponds was constructed on rehabilitated land leased to NBCAC. The first harvest was successfully completed in 2011 and the second harvest in 2012.

NBCAC considered that the project provided a cost effective and proven means of transforming a liability into an asset, while at the same time offering training, education and job opportunities to Indigenous people in an area of high unemployment and socio-economic deprivation. NBCAC considered that the project also created a sustainable enterprise for the community.

Research indicated that, although no significant benefits to lake water quality were forthcoming because of the nutrient sink capacity from the AMD in the pit lake (Lund *et al.*, 2010), no significant water quality impacts were observed either with generally good water quality in pit lakes experiencing aquaculture (Yokum *et al.*, 1997; McNaughton & Lee, 2010).

Figure 3: The Ngalang Boodja Mine Lake Aquaculture Project (a) aerial view (Nearmap) and (b) side view at Yancoal Premier Coal Ltd in Collie, Western Australia.

3.1.4 Batesford Quarry Lake urban development (limestone)

Rarely do opportunities arise allowing closure of a mine site to be planned as a key beneficial land use asset within a new urban growth area; particularly a long way in advance of cessation of mining.

Strategic planning has been initiated by quarry owners ten years before mining ceases at the Batesford Quarry (Gerner & McCullough, 2014). In liaison with the planning agencies, the planned closure of a large 100 year old limestone quarry pit will form a lake on completion of mining, with a well-considered beneficial end use in an urban setting in mind. The quarry pit and adjoining land at Batesford South abut the edge of the City of Greater Geelong, Victoria, Australia. Geelong is experiencing growth pressures that necessitate development of a new growth area from about 2025.

A multidisciplinary expert team assessed the constraints, opportunities and feasibility of rehabilitating the quarry and pit lake and developing the surrounding land as Geelong's next urban growth area. Proposed work involved placing and grading waste materials around the edges to provide stable pit lake edges.

The concept is exciting and demonstrates that a former mine site could become a valued home to about 40 000–45 000 people. About 30% of the land will also be developed as an ecological riverine corridor including the pit lake that will become a regionally iconic lake as the centrepiece of an urban development.

Figure 4: Artist's impression of the Batesford South Pit Lake at completion of quarrying and once groundwater equilibrium and filling is complete.

3.2 Internationally

International examples are generally better developed than Australian examples. In a large part because they often happen closer to human habitation centres than many large open cut Australian mining operations. Both of these reasons may place them as better examples to Hazelwood closure opportunities than the aforementioned domestic examples.

International examples similar to Hazelwood Mine approved Work plan conceptual designs for closure are described below. More case studies are listed in Annexure D.

3.2.1 Rother Valley Country Park, United Kingdom (coal)

Rother Valley Park was planned in the early 1970s, but mining did not commence until the mid-1970's (McCullough *et al.*, 2009). Sheffield City Council began to determine the feasibility of the park in 1972 after the National Coal Boards announced that they intended to establish open cut coal mining in the region. The proposed park covered the local council areas of Rotherham, Sheffield, North and East Derbyshire and the South Yorkshire county councils.

A Joint Committee of the five county councils oversaw the funding for development and running of the park. The Joint Committee throughout this period established a community consultation program. In 1977 they produced a *Development Options Report* that allowed for significant community input into the final design of the park. From the outset it was envisaged that the site would be rehabilitated in such a manner that it would become a recreational facility that would attract tourists to the region. By 1978 the Final Report was published and used as the basis for the final rehabilitation design.

The restoration program achieved three main objectives:

1) Create four river-filled main lakes with adjacent open land to provide a wide variety of recreational pursuits both water and land based (Figure 5);

2) Provide several different habitats for the many different plants and animals that inhabit or migrate through Rother Valley;

3) Create an efficient flood control system to protect areas of housing and industry downstream.

Figure 5: Rotherham County Park, after rehabilitation (Rotherham Metropolitan Borough Council, 2004).

Although all five councils initially offered rehabilitation funding the Rotherham Metropolitan Borough Council continues to be the sole entity responsible for the funding and maintenance of the park. Rother Valley incorporates many of the features of world's best practice rehabilitation such as early planning, community involvement, strong local council involvement, a long time frame for the development, and commitment from all of the parties involved in the closure process. Furthermore, Rother Valley Park has now developed strong commercial outcomes through tourism-based activities.

3.2.1.1 Lusatian Region – Germany (brown coal)

Germany provides a broad range of examples of pit lakes with many end uses and a range of approaches to filling and neutralizing acidic water bodies for optimal uses (McCullough & Schultze, 2015). The Lusatian region, in the former socialist German Democratic Republic was the centre of East Germany's energy production until the privatization of mines and consequent closure of most pits in 1990. Lignite mining had provided the brown coal needed to supply the region's energy needs. Mine dewatering activities also increased water levels in the River Spree and Schwarze Elster, supporting downstream economic activities and securing the water supply of Berlin and other population centres (Lienhoop & Messner, 2012).

The rapid collapse and closure of lignite mining resulted in negative socio-economic effects for the region, but a plan to convert the area to a rehabilitated Lake District for tourism and recreation is learning from similar regions in other parts of Germany and capitalising on the transformed landscape for beneficial purposes.

The ultimate vision is planned to consist of 18 lakes, with nine core lakes, all connected by canals and providing a suite of recreational facilities such as boating, windsurfing, sailing and water skiing. Tourist accommodation is starting to be developed beside the filling lakes and there are co-existing plans to incorporate nature conservation areas with recreation and tourism. Each lake has been slowly filling over the past decade and many are starting to reach their expected completion dates. Lienhoop and Messner (2012)) provide the following estimations regarding the volume and estimated completion dates (Table 3).

Table 3: Lusatian lignite region planned mine pit lakes (Schultze *et al.*, 2010).

Lake	Surface Area (ha)	Volume (million m^3)	Estimated Completion Date (as of Dec 2009)
Ilse See	771	153	2015
Sedlitzer See	1330	206	2015
Partwitzer See	1120	130	2012
Geierswalder See	620	92	2012
Neuwieser See	632	56	2015
Blunoer Sudsee	350	64	2015
Sabrodter See	136	27	2015
Bergener See	133	3	2015
Spreetaler See	314	97	2015

The Lusatian Region is one of the driest in Germany, which dramatically influences options for re-filling pit lakes on a broad scale. Historic filling practices were based on groundwater, but for this region a carefully managed water transfer system is in place which diverts water from three river systems, reservoirs, pit lakes, operating mines and water treatment plants. The system is operated using a computer-based model and assesses volumes and water quality to assist with *neutralization* (Schultze et al., 2013b). The rehabilitation of the decommissioned lignite mining area, approximately 1000 km^2, is under the responsibility of the Lusatian and Central-German Mining Administration Company (LMBV), which is a state-owned entity (Lienhoop & Messner, 2012).

(Schultze *et al.*, 2013a) describes how the mine closure of these pit lakes was managed (Figure 6). The methods used to fill them have included:

- natural groundwater recharge;
- filling with mine water from operating mines (dewatering);

- rapid fill and/or diversion of river water.

The results of these efforts vary greatly and require regular monitoring as neutralized lakes can become acidic afterwards in certain circumstances. The river flow through has produced some of the most successful outcomes. In Lusatia, neutralization efforts are geared towards lakes where there is an identified demand for other uses and priority is given to lakes that will discharge into river systems (Schultze *et al.*, 2011b).

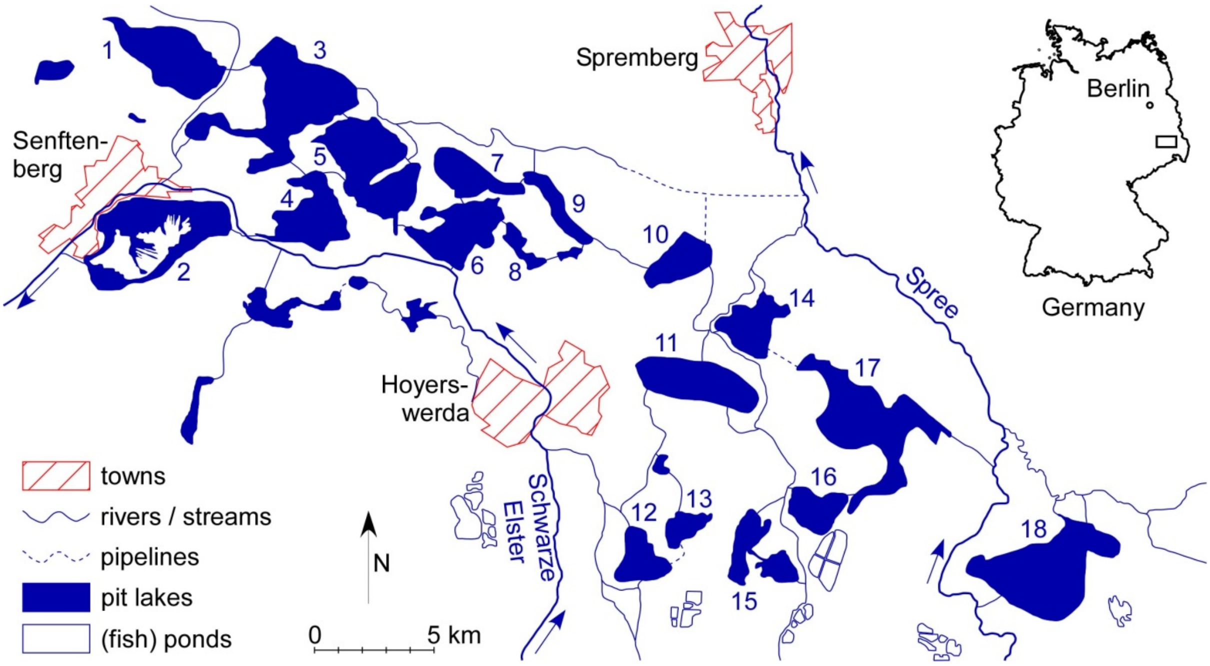

Figure 6: Lausitzer Seenland", the central part of the Lusatian lignite mining district. 1 Lake Großraeschen, 2 Lake Senftenberg, 3 Lake Sedlitz, 4 Lake Geierswald, 5 Lake Partwitz, 6 Lake Neuwiese, 7 Lake Bluno, 8 Lake Bergen, 9 Lake Sabrodt, 10 Lake Spreetal, 11 Lake Scheibe, 12 Lake Knappensee, 13 Lake Graureihersee, 14 Lake Bernsteinsee, 15 Lake Lohsa I, 16 Lake Dreiweibern, 17 Lake Lohsa II, 18 Lake Baerwalde. Only selected towns are shown for orientation (after Schultze et al., 2011b).

Lienhoop and Messner (2012) completed an economic study which determines the non-market value of creating a lake district. The study assesses stakeholder support for recreational and amenity based uses of the lakes (Figure 7, Figure 8). Their approach quantifies the economics of providing such a facility in comparison to more traditional water users, particularly economic based demand for ongoing water allocations (such as aquaculture or other industries). The authors argue a simple market-based assessment underestimates the benefits of pit lakes and that instead all benefits (particularly recreational) should be considered when looking at costs and benefits of restoration.

Figure 7: Still filling pit lakes Bergheide and Ilse and associated tourism infrastructure.

Figure 8: Acidic Lusatian coal pit lakes Sedlitz and Geierswald showing tourism walking tracks and water recreation opportunities.

4.0 QUESTION 4

What further work is required at the Hazelwood Mine in order to successfully implement to the Approved Final Rehabilitation Model? Are you able to identify short, medium and long term work priorities in this regard, having regard to the planned Mine closure date (2033)?

4.1 Further work required at the Hazelwood Mine

I recommend that the following further studies be undertake in order to provide for practicable rehabilitation of the Hazelwood Mine void in a timely manner with reasonable cost and with outcomes of significantly reduced risk and improved opportunity.

4.1.1 Conceptual Mine Closure Plan. Priority: short term

A priority for closure planning should be the development of a high-level conceptual mine closure plan (CMCP) to consolidate current knowledge and understanding, identified knowledge gaps, propose studies against milestone for completion and demonstrate consideration and feasibility of the final conceptual design for closure.

This document then becomes a communication tool for regulators and the community, as well as internally helping to direct funding for closure and operational mine planning including opportunities for progressive rehabilitation.

4.1.2 Water balance with climate change. Priority: short term

Connection of the pit lake to surrounding groundwater sources can play a large role in the water quality and hydrological cycle/budget of the pit lake; if a pit lake water surface is above the water table, water will flow out of the pit to the groundwater and thus provide a pathway to transport potential contaminants to a larger area (Niccoli, 2009).

A detailed probabilistic water balance should be undertaken that includes assessment of climate change under scenarios spanning most recent predictions. A sensitivity analysis should be then made to determine potential weaknesses in the water balance model and consequently areas for knowledge improvement such as research.

4.1.3 Weather station installation. Priority: short term

Site specific climate data should be collected from the void itself at the predicted equilibrium lake surface level. The monitoring data generated will be used to inform long-term water balance and water quality modelling (Huber *et al.*, 2008).

4.1.4 Fly ash geochemistry. Priority: medium term

Although mine pits have been used as sites for disposal of ashes of coal combustion in power plants internationally, there are advantages and disadvantages accompanying subaqueous disposal of waste. The geochemistry of all backfill materials must be understood (Puhalovich & Coghill, 2011). In general, the stability of the conditions inside the deposited waste and at its interface with its aqueous environment is a main prerequisite for successful long term storage of waste below a water cover. Risks such as long term change of conditions inside and around the waste deposits and the pit lakes, as groundwater contamination or as toxication of aquatic life have to be evaluated carefully and site specifically. However, there are no scientifically reasonable arguments for a general preclusion of the subaqueous disposal of waste in pit lakes (Schultze *et al.*, 2011a).

Although considered “relatively benign” (Annexure 9, Work Plan Variation 2009) the geochemistry of fly ash does not appear to have been studied under the subaqueous environment and pit lake water quality solvent conditions expected in the final pit lake.

For fly ash above the final water level, it is also not clear how stable this material will be to erosion; especially wave action. There does not appear to have been revegetation trials undertaken so it is also unknown how successfully the material can be revegetated.

4.1.5 Final landform vision. Priority: short term

Although there is a process for key stakeholder engagement ongoing since 1999 (Annexure 9, 8.1) it is not clear if stakeholder input had been sought for closure planning and the conceptual final landform design. A final closure vision must be defined for the mine closure that has involved documented engagement from community and regulator engagement inc. shire, Traditional Owners, and other key interested parties such as Non-Government Organisation (NGOs). A risk/opportunity assessment such as a SWOT analysis is typically undertaken against different practicable and reasonable closure options (Swanson, 2011). The results of this are then communicated to the community through media such as radio, newsletter, local newspapers and the company website.

4.1.6 Wildlife habitat. Priority: medium term

Identifying wildlife habitat in the conceptual plan means that studies must be undertaken to support the viability of developing these proposed end use values and also to underpin their development. For example; what species would be sought/are desirable? What habitat do they require?

4.1.7 Geotechnical stability Priority: medium term

Overburden and coal batter angles are stated at 3H:1V and 2.5–3.0H:1V respectively. However, these gradients appear to be unjustified or validated. For example, what material characteristic studies are they based upon? Often waste is also strategically dumped around pit walls to stabilise and achieve stable for vegetation and safe. Geotechnical studies incorporating ongoing assessment of landform materials mechanical and geochemical weathering properties should form a key component of future closure planning studies.

4.1.8 Eutrophication and algal bloom risk. Priority: medium term

High levels of nutrients may present in groundwater naturally and through the intense agricultural activities of the pit lake catchment. Risk assessment, such as water quality and associated algal biomass modelling,, should be undertaken to understand the risk of algal blooms and to determine management strategies (if required).

4.1.9 Wave action and erosion

Hazelwood have proposed to assess the potential for wave action to erode batters and batter profiles. Wave erosion studies should form part of a separate assessment to determine the effect of long-term wave action on proposed wildlife habitat such as riparian vegetation, habitat heterogeneity, vegetation littoral habitat and woody debris. It should also examine stability of safe access for swimmer/boaters.

4.1.10 Riparian vegetation selection and trials. Priority: long term

Current revegetation strategies do not consider riparian species (Annexure 9, Appendix A).

The salt-tolerant species already recommended for urban run-off ponds rehabilitated into wetlands may be appropriate, however species selection suitable for the closure objectives should be made and their revegetation success trialled.

4.1.11 Contaminated Sites. Priority: short term

If not already present, a database should be established to track all fuels and other chemical stored on site. Investigations should also be undertaken to identify and remediate as required any contaminated sites that can be addressed in a progressive manner.

4.1.12 Closure Objectives and Developing Closure Criteria. Priority: short term

Following determination of a final land use vision from engagement with stakeholders that is agreed to by regulators, preliminary closure objectives and criteria should be explicitly defined. These definitions will help guide closure planning and also provide confidence from regulators that the closure vision is well understood and directed.

4.1.13 Closure risk workshop. Priority: short term

A closure risk workshop should be undertaken to advise mine closure planning of key risks as soon as practicable. This provides more time to then management any identified high risk closure hazards as well as preventing key planning milestones being overtaken that may prevent some strategies and activities then being able to be undertaken e.g., inappropriate waste placements (DITR, in press).

4.1.14 Socio-economic analysis of end uses. Priority: long term

The final land use vision should be validated by a regional socio-economic analysis cumulatively incorporating the social and economic effects of closure of the Hazelwood Mine with the new landscape opportunities proposed in the approved closure work plan (Lienhoop & Messner, 2012).

4.1.15 Flow-through closure. Priority: medium term

Consideration should be given to opportunities for flow-through of the Morwell River at closure as part of more holistic a closure strategy to provide improved pit lake water quality, improved river water quality, and improved regional connectivity of aquatic and riparian habitat (McCullough & Schultze, 2015).

4.1.16 Long-term pit lake water quality prediction. Priority: short term

Mine lake water quality post-closure will most critically of all parameters determine the beneficial uses of the pit (McCullough & Lund, 2006). PanTek Solutions (2003) has modelled pit water balance and pit water quality post-closure. Pit water quality will be affected by:

- Groundwater inflows and seepage
- Direct rainfall and catchment runoff.
- Leachate from the ash pond.
- Leachate from the coal and overburden.

A detailed water quality model should be established to predict long term geochemistry and knowledge gaps and need for further study. This model should incorporate hydrogeological modelling and key hydrodynamic lake processes such as stratification and mixing (Castendyk *et al.*, 2014).

4.1.17 Potential impacts on the hydrology and water quality of the Morwell River. Priority: medium term

With a significant freeboard remaining in the pit void upon equilibrium, it appears unlikely that decant from the pit lake to the river will occur.

However, the presence of the Hazelwood Mine void may affect the hydrology of the Morwell River (Annexure 15). The post-closure pit lake may influence the river hydrology through;

- groundwater gains or losses;
- water quality changes through groundwater seepage.

A study should be undertaken to determine the likelihood and effect of groundwater interactions between the pit lake and the Morwell River. This assessment should also investigate potential water quality changes of the river in the context of current and probable future river water quality and current and probable future river values.

MEMORANDUM

LIMITATIONS

Your attention is drawn to the document ""Important Information Relating to this Report", which is included as Annexure E to this technical memorandum. This document is intended to assist you in ensuring that your expectations of this memorandum are realistic, and that you understand the inherent limitations of a memorandum of this nature. If you are uncertain as to whether this memorandum is appropriate for any particular purpose please discuss this issue with us.

GOLDER ASSOCIATES PTY LTD

Dr Clint McCullough
Associate, Principal Environmental Scientist

Ed Clerk
Principal, Principal EHS Consultant

Annexures

A – Letters received from King & Wood Mallesons dated 17 October and 18 November 2015.

B – Curriculum vitae – Dr. Clint McCullough.

C – Key papers cited.

D – Further relevant examples of rehabilitated coal mine lakes.

E – Statement of Limitations.

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ANNEXURE A

Letter received from King & Wood Mallesons

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By courier

Dear Sir,

Hazelwood Mine Fire Inquiry

We act for Hazelwood Power Corporation, Hazelwood Power Partnership, International Power (Australia) Pty Ltd and GDF SUEZ Australian Energy and its related entities (together, **GDFSAE**) in relation to the Board of Inquiry appointed to inquire into and report on certain matters relating to the Hazelwood Coal Mine Fire in 2014 specified in the Terms of Reference dated 26 May 2015 (**TOR**).

GDF SUEZ Australian Energy is the owner and operator of the Hazelwood Coal Mine and Power Station.

This is the second Board of Inquiry process in relation to the Hazelwood Coal Mine Fire. The first Inquiry was completed in 2014, and a report was published by that Board.

The TOR for the present Inquiry are enclosed.

Our client wishes to retain you as an expert for the purposes of the process which the Board wishes to follow in relation to paragraphs 8 and 9 of the TOR. Paragraphs 8 and 9 of the TOR concern short, medium and long term options for the rehabilitation of the Latrobe Valley mines.

The process the Board is following in relation to TOR 8, 9 and 10 is set out in the attached letter from the Board dated 8 October 2015.

As relevantly detailed in that letter, it is envisaged that:

1. a draft report prepared by the expert appointed by the Board (Jacobs) in relation to TOR 8 and 9 has been circulated to all parties who have been granted leave to appear (namely our client, the State of Victoria, Energy Australia Yallourn Pty Ltd, AGL Loy Yang Pty Ltd and Environment Victoria Inc) today;

KWM

KING&WOOD MALLESONS

2. there will be a "technical experts review conference" with Jacobs to discuss the Jacobs draft report on 27 and 28 October 2015 at which the parties are invited to have 1-2 representatives attend (including any expert);
3. Jacobs will finalise its report by 13 November 2015;
4. any expert reports obtained by the parties (including our client) in relation to TOR 8 and 9 (and 10) are to be filed by 23 November 2015;
5. hearings in relation to TOR 8 and 9 (and 10) will be conducted on 8-11 December 2015 in Morwell.

As is apparent from this summary, our client wishes to retain you to attend the technical experts review conference on 27 / 28 October 2015, prepare a report in relation to the Jacobs report, and attend the hearings on 8 - 11 December 2015. In relation to the hearings in December, it is unlikely that you will be required to attend all four days – however the precise days on which attendance will be required are not known at this stage.

Please find enclosed the following background materials for your assistance:

Hazelwood Mine Fire Inquiry

1. The TOR;
2. Practice Direction for Terms of Reference 8, 9, 10 (Mine Rehabilitation);
3. Letter from Principal Legal Adviser, Hazelwood Mine Fire Inquiry, dated 8 October 2015 concerning the Board's intended process in relation to TOR 8 – 10;
4. Summary of Community Consultations on Rehabilitation Terms of Reference;
5. Summary of public submissions to the Board on matters the subject of TOR 8 and 9;
6. Victorian Government submission on Rehabilitation Terms of Reference;
7. AGL (Loy Yang) submission on Rehabilitation Terms of Reference;
8. Energy Australia (Yallourn) submission on Rehabilitation Terms of Reference;
9. Environment Victoria submission on Rehabilitation Terms of Reference;
10. GDF SUEZ Australian Energy submission on Rehabilitation Terms of Reference;
11. Statement of James Faithful of GDFSAE filed in the first Hazelwood Mine Fire Inquiry; and
12. 2014 Hazelwood Mine Fire Inquiry Report – Chapter 3.

Licence materials

13. Mining Licence 5004 together with Approved Work Plan – September 1996;
14. Instrument of Amalgamation of additional mining licences into MIN5004 – July 1996;

KWM

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15. Instrument of Variation and Addition of Licence Conditions – January 2015;

16. Approved Work Plan Variation dated May 2009;

17. EPA Licence 46436;

18. Ground Control Management Plan.

Jacobs Report

19. Draft report of Jacobs dated 12 October 2015 entitled "*Review of Future Rehabilitation Options for Loy Yang, Hazelwood and Yallourn Coal Mines in the Latrobe Valley*" together with its Appendices;

20. *Explanatory Note for Jacobs report into Future Options for the Rehabilitating [sic] the Hazelwood, Yallourn and Loy Yang Mines in Latrobe Valley*;

21. Jacobs Memorandum dated 13 October 2015 entitled "Basis of Cost Estimates for Risk Issues"; and

22. PowerPoint presentation delivered by Jacobs at 'webinar' on Thursday 15 October 2015.

Thanks for your assistance.

Yours faithfully,

King & Wood Mallesons

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18 November 2015

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By email

Dear Sir,

Hazelwood Mine Fire Inquiry

We refer to our previous correspondence, and to the Final Report of Jacobs Australia Pty Limited (**Jacobs**) dated 16 November 2015 (**the Jacobs Report**) which has been provided by Jacobs to the Hazelwood Mine Fire Board of Inquiry for the purposes of Terms of Reference 8 and 9 (**TOR 8 and 9**).

Expert report

We write to request that you prepare a report for the purposes of the Hazelwood Mine Fire Inquiry in which you set out your expert opinion on the following matters:

1 Is the approved model for the final rehabilitation of Hazelwood Mine detailed in the Work Plan Variation (of a partial pit lake within a lowered landform) (**the Approved Final Rehabilitation Model**), a feasible and appropriate model for final rehabilitation from the perspective of:

(a) achieving a safe and stable final landform; and

(b) returning the Mine site to a condition which will enable future beneficial land use and which will compliment the surrounding environment?

2. Given the progression of the mining operations at the Hazelwood Mine, and the extent of progressive rehabilitation and in-pit dumping of materials to date, are there any reasonably practicable final rehabilitation models for the Hazelwood Mine, other than the Approved Final Rehabilitation Model?

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3. Has a final rehabilitation model similar to the Approved Final Rehabilitation Model been implemented elsewhere:

   (a) in Australia, or

   (b) Internationally?

4. What further work is required at the Hazelwood Mine in order to successfully implement to the Approved Final Rehabilitation Model? Are you able to identify short, medium and long term work priorities in this regard, having regard to the planned Mine closure date (2033)?

In preparing your expert report in relation to the matters set out above, please:

1. Set out the matters, facts and circumstances on which you rely, and any relevant assumptions which you have made, for the purposes of your opinion;

2. Enclose as annexures a copy of our letter dated 17 October 2015 and subsequent correspondence (including this letter): Attachment A; and

3. Attach a current CV: Attachment B.

Timing of report

The Board has determined that any expert reports obtained by the parties (including our client) in relation to TOR 8 and 9 are to be filed by 27 November 2015.

On this basis, we request that you finalise your report by this date if at all possible.

Hearings

As noted in our letter dated 17 October 2015, it is likely that you will be required to give evidence at the hearings in relation to TOR 8 and 9 which have been set down for 8 - 11 December 2015 in Traralgon. It is unlikely that you will be required to attend for more than one day. However at this stage we do not know the dates on which your attendance will be necessary. We will keep you informed in this regard.

Thank you for your assistance.

Yours faithfully,

King & Wood Mallesons

ANNEXURE B

Curriculum vitae – Dr. Clint McCullough

Education

PhD Aquatic Ecotoxicology Charles Darwin University, Darwin, 2006

MSc Freshwater Ecology, Waikato University of Auckland, Hamilton, 1998

BSc Zoology, James Cook University of North Queensland, Townsville, 1994

Affiliations

Australasian Institute of Metals and Metallurgy (AusIMM) CP(Env)

International Mine Water Association (IMWA)

University of Western Australia (UWA)

Edith Cowan University(ECU)

Australian Society for Limnology (ASL)

Relevant Experience

Mine Closure Planning

Mine Closure Costing

Mine Pit Lakes

Mine Water Impacts

Environmental Monitoring and Risk Assessment

Acid Mine Drainage (AMD)

Mining Rehabilitation

Golder Associates Pty Ltd - Perth

Associate, Principal Environmental Scientist

Dr Clint McCullough is an Associate and Principal Environmental Scientist with over 20 years research and consultancy experience in environmental management issues. Clint also holds senior adjunct lecturing and research positions at the University of Western Australia and Edith Cowan University.

Clint has international experience with ecological and geochemical applications and key expertise in mining and the environment, especially mine closure and mining impacts on waters. He is a recognised leading international expert on mine pit lake sustainability, closure planning and rehabilitation, with project experience across Australasia, Asia, and the Americas. Clint also makes regular presentations to government, international conferences and professional bodies on mine closure helping develop closure regulatory guidance for pit lakes of the Commonwealth of Australia, Western Australia and South Australia, and the Canadian Oil sands.

Clint has authored over 90 published peer-reviewed papers and book chapters, and recently published the book “Mine Pit Lakes: Closure and Management” through the Australian Centre for Geomechanics.

Relevant project experience includes:

- **Mine Pit Lakes.** International closure planning advice for pit lakes in settings from the oil sands of Canada to the South American Andes and US and Australian deserts. Beneficial end use planning with international literature review and precedent advice for development proponents. Acid and metalliferous drainage (AMD) mitigation and bioremediation of water quality; both internal and for pit lake discharges.
- **Mine Closure Planning**. Conceptual and detailed (definitive) closure plan development for mines in Australasia, Asia, North and South America. Corporate mine closure standard development. Closure planning to internal, state/national regulations or international leading practice. Stakeholder consultation and engaged closure strategy and end use development. Final landform and site layout options assessment and planning. Final land use, closure objectives and closure criteria development. Liability and closure cost estimates for internal accounting and regulatory reporting requirements.
- **Mine Water Quality**. Water quality assessment and site-specific/site relevant trigger values to meet Australasian guidelines. Interpretation of water quality results and ecotoxicological assessments e.g., for new ANZECC guidelines for magnesium and uranium. Client assistance meeting discharge criteria by development of water quality criteria accounting for reference conditions, regional biota tolerances, previous disturbances and cumulative impacts.
- **Environmental Impact Assessment and Management Planning**. Environmental survey and monitoring of water bodies for water quality and biota: diatoms, phytoplankton zooplankton, macroinvertebrates, fishes, amphibians and birds. Management planning to mitigate environmental impacts from mining and mine waters; operational or following closure.

Clint can be contacted on +61 8 9213 8255 or cmccullough@golder.com.au

ANNEXURE C

Key papers cited

Mine Water and the Environment (2006) 25: 220–226 © IMWA Springer-Verlag 2006

Technical Communication

GDFS.0001.003.0032

Opportunities for Sustainable Mining Pit Lakes in Australia

Clint D. McCullough and Mark A. Lund

Centre of Excellence for Sustainable Mine Lakes and Centre for Ecosystem Management, Edith Cowan Univ, 100 Joondalup Dr, Joondalup, WA 6027, Australia; corresponding author's e-mail: c.mccullough@ecu.edu.au

Abstract. Due to operational and regulatory practicalities, pit lakes will continue to be common legacies of mine lease relinquishments. Unplanned or inappropriate management of these geographical features can lead to both short- and long-term liability to mining companies, local communities, and the nearby environment during mining operations or after lease relinquishment. However, the potential for pit lakes to provide benefit to companies, communities, and the environment is frequently unrecognised and yet may be a vital contribution to the sustainability of the open-cut mining industry. Sustainable pit lake management aims to minimise short and long term pit lake liabilities and maximise short and long term pit lake opportunities. Improved remediation technologies are offering more avenues for pit lakes resource exploitation than ever before, at the same time mining companies, local communities, and regulatory authorities are becoming more aware of the benefit these resources can offer.

Key words: AMD; Australia; end use; mine waters; pit lake; sustainable mining

Introduction

Being a finite abstraction, "sustainable mining" is something of an oxymoron for what is inherently unsustainable activity (Mudd 2005). Nevertheless, in an era of increasing recognition of environmental and social damage from an ever-growing scale of mining coupled with increasing corporate social conscience for these activities, the mining industry usually works to reduce operational risk and retain its social licence to mine the community resource through a variety of strategies. Many of these strategies are focused around the concept of sustainability, including creating sustainable livelihoods (employment, community development, and infrastructure), optimising resource use, and final closing of mining operations in a manner that minimises social and environmental harm and yet retains future options for the lease (BHP Billiton Plc 2005; Rio Tinto Plc 2005). Although understandings do vary (Mudd 2005), sustainable mining commonly incorporates "the evaluation and management of the uncertainties and risks associated with earth resource development" (Meech 1999). This sustainability definition also fits well with the understanding of most government authorities concerned with the regulation of environmental and social impacts of mining (Mudd 2004). As a result of this regulatory focus, sustainability of mining leases is often solely concerned with minimising the immediate and long term risks to all stakeholders concerned (e.g. the social and ecological environment surrounding the mine).

One potential legacy of open-cut mining is the mining pit(s) left after rehabilitation operations are completed. Some of these pits are constructed either in part, or in whole, below the surrounding natural ground water levels. As a consequence, once dewatering operations stop and surface and ground waters equilibrate, these pits may form pit lakes (Castro and Moore 1997). The pit lake water may be contaminated with elevated concentrations of heavy metals and/or acid mine drainage (AMD) (Banks et al. 1997). The pit lake may act as a ground water "sink" under low rainfall/high evaporation climes, increasing in salinity whilst lowering surrounding ground water levels (Commander et al. 1994). Alternatively, in higher rainfall environments, clean surrounding ground water can be contaminated as it passes through the pit lake.

Regulators generally prefer complete backfill of pits with waste rock, tailings and/or operation wastes to reduce ground water interaction and acid production (Johnson and Wright 2003). However, although backfilling may be considered a simple solution to the formation of pit lakes, it is often not cost effective or even desirable. In fact, complete backfills are rare due to the high expense involved and potential contamination issues associated with the fill material. Nonetheless, pit lakes may quantitatively contribute more to mine water pollution than tailings and waste rock dump leachates arising from the same mining lease (Younger 2002).

There are an estimated 1,800 open-cut pits in Western Australia alone, ranging from one or two hectares in area and a few meters deep to the increasingly large modern pits of several square kilometres in area and hundreds of meters deep (Johnson and Wright 2003).

These pit lakes have few natural counterparts in the Australian landscape where natural lakes tend to be more shallow and seasonal in nature. Therefore, mining pit lakes represent a novel addition to the aquatic resources of the region. The nearest ecological counterparts of these new lakes are reservoirs, but the cross-sectional profiles of reservoirs are different and by their nature have reasonable turnovers of the water in them (through high capture and exploitation rates). Especially in dry continents such as Australia, and also as an aspect of water-related issues of the mining industry internationally, pit lakes may be seen as a model of the challenges of management of the wider water resource crisis facing the global community today (Brown 2003).

Environmental and Social Risks of Pit Lakes

Like most developed countries and states, performance bonds are held by government pending appropriate rehabilitation and final relinquishment of mining leases. For example, currently the Western Australian mining industry has around $350 million worth of unconditional performance bonds held against it by the State Government on grounds of environmental performance (Western Australia Chamber of Minerals and Energy 2004). However, in many states, these performance bonds occur in a regulatory environment with no specific water quality guidelines for managing pit lake risks [e.g. Nguyen (2006)], even though mine waters have been internationally identified as the greatest off-site risk of mining to local communities and their environment (Younger 2002), and that water use at many of Australia's remote, arid locales can represent a large proportion of the local supply (Brown 2003).

There are clearer standards for managing the vast masses of waste rock that may arise in the excavation of an open-cut pit and which may be the single largest cause of ecological impacts in a pit lake (Mudd 2005). However, at present, there are only regulatory guidelines available for natural lakes, which may be overvalued, or otherwise inappropriate analogues to pit lakes. Consequently, regulation of pit lake water quality in much of Australia is made on case-by-case assessments and pit lake water quality is regulated according to either specific end-use requirements, or safety of the surrounding environment (Evans et al. 2005).

During active mining operations, pit water management is typically well understood and regulated (Johnson 2003). However, following mine closure, the management and relinquishment requirements for developing pit lakes are far less well understood by either mining companies or their regulatory bodies (Pilkey 2003). As a result, pit lakes tend to have limited biological activity and chemical interactions dominate (Castendyk and Webster-Brown 2006). Our experiences researching pit lakes in Australia suggest that even after 50 years, ecological processes are often still very restricted (Lund et al. 2006; McCullough and Lund 2006).

Pit lakes present many significant health and safety issues for both the mining company and adjacent human and wildlife communities for many years following cessation of mining operations (Doupé and Lymbery 2005). For example, pit walls can be unstable and can become more so during lake filling. Pit lakes tend to have a low surface area to depth ratio compared to natural lakes and have steep sides. This steepness may produce risks for lake users such as local communities, live stock, and wild life where there is a risk of falls from the pit "high walls", or for swimmers where there is a risk of drowning with the limited shallow margin (Figure 1).

Wildlife drinking from pit lakes may ingest contaminated water, which could cause severe trauma or and/or eventually death. Acidic pit lake water may also remove natural oils from the feathers of waterfowl leading to their deaths through drowning or exposure (Woodbury 1998). Bioaccumulation of elevated heavy metals may be of concern to wildlife and communities utilising pit lake fisheries (Evans et al. 2000).

Although the impacts of mining has not yet been specifically examined, land use change by humans has long been recognised as a causative factor in the outbreak of mosquito-borne diseases (Norris 2004). By increasing their breeding habitat, pit lakes may also harbour waterborne diseases and their vectors,

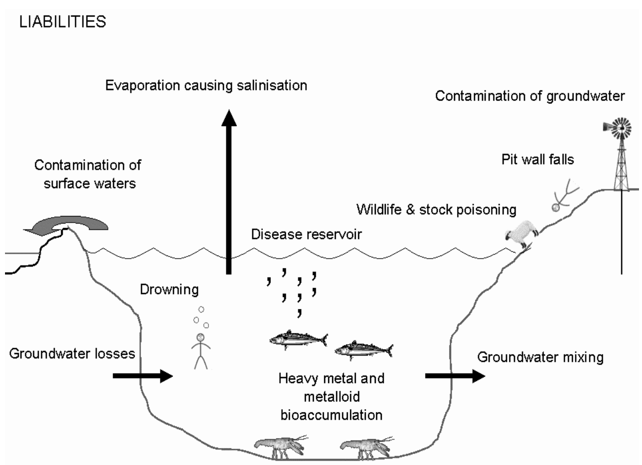

Figure 1. Potential liabilities of pit lakes to communities and the environment

as mosquitoes (Pilbara Iron Ore Environmental Committee 1999). For example, mosquitoes of the *Culex* genus, some of which are capable of transmitting Ross River fever and Australian encephalitis, have been found in abandoned pit lakes of the Collie region (Lund et al. 2000). As pit lakes become remediated and increase in nutrient status over time, they may become even more attractive to laying adult mosquitoes (Leisnham et al. 2005, 2006) and may consequently form more significant sources of disease-carrying mosquito vectors in historical mining regions (Johnson and Wright 2003).

Some pit lakes e.g., the western desert's Pilbara, Goldfields, etc. suffer problems associated with hypersalinity. In areas with high evaporation and/or low precipitation and low ground water flow rates, hypersalinity may be caused by saline ground water being drawn into these lakes through evaporation (Johnson and Wright 2003). Saline ground water intrusion may result in long term increases in pit lake salinity and lowered ground water levels (Commander et al. 1994). In addition to direct loss of habitat through reduced ground water levels, deteriorated ground water quality may contaminate underground (stygofauna) biotic communities (Environmental Protection Authority 2005), or overflow into surface water environments utilised by local communities and endemic biota (Kuipers 2002; Younger 2002). Nevertheless, ground water quality deterioration by mining has been considered inconsequential in some areas that are deemed only useable for mining purposes (Taylor et al. 2004).

Conversely, in higher rainfall areas such as Collie in Australia's southwest, or in the high rainfall area of the "Top End" of the Northern Territory, lake inflow exceeds evaporation, resulting in a flow of water out of the lake into the ground water. Contaminants in these pit lake, such as heavy metals and low pH, may be transported into the ground water and discharge downstream (Mudd 2002b). Contamination of through-flow ground waters may have profound consequences for natural and human communities in arid regions of Australia that are almost entirely dependent on ground water (Mudd 2002a).

During active mining operations, pit water management is typically well understood and regulated (Johnson 2003). However, following mine closure, the management and relinquishment requirements for developing pit lakes are far less well understood by either mining companies or their regulatory bodies. Remediation by fast-filling through river diversion may fill pit lakes in a timescale of only years (Schultze et al. 2002, 2003), while ground water-filled final pit lake levels, and chemical and biological conditions may take centuries to reach equilibrium levels (Johnson and Wright 2003).

Predictive geochemical modelling of pit lake water chemistry can be a powerful tool for the preparation and ongoing management of final hydrology and water quality of these lakes (Castendyk and Webster-Brown 2006). However, the majority of predictive models in current use and development are adapted from research into natural lakes or reservoirs. Although these systems may share some of the same physical and chemical complexes of pit lakes, as aforementioned, pit systems differ in many fundamental ways which may lead to either inaccuracies or simply lack of confidence in prediction and consequent acceptance of modelling conclusions (Wright 2000).

Furthermore, although the primary use of predictive water quality models is to satisfy regulatory agencies, water quality is only one of the issues needing consideration. Health and safety issues, such as final pit lake water heights and interactions with surrounding water bodies, flood risks, and possible disease vectors, may remain undefined.

Nevertheless, although pit lakes may present risks to the environment and local communities through both their structural safety and water quality issues, pit lakes typically remain the cheapest, and often, most practical option for relinquishment of many open-cut leases and will continue to be formed across Australian and the world as mining companies cease their operations.

Sustainable Pit Lakes

Probably the most commonly quoted definition of sustainability for human societal activities as a whole, and one that is widely accepted by mining companies and regulatory authorities, is that "sustainable development is development that meets the needs of the present without compromising the needs of future generations to meet their own needs" (World Commission on Environment and Development 1987). However, in the mining industry, sustainability post mining is typically only defined as leaving no ongoing environmental and social impact from mining operations [e.g. O'Reilly (2003)]. Consequently, a better definition of mining sustainability that also encompasses benefits that mining legacies such as pit lakes may offer may be "minimising long term risks of pit lakes, whilst maximising both short and long term benefits, for all stakeholders". Pit lakes can provide economic, health, welfare, safety or aesthetic benefits to the community (Doupé and Lymbery 2005; Johnson and Wright 2003) (Figure 2).

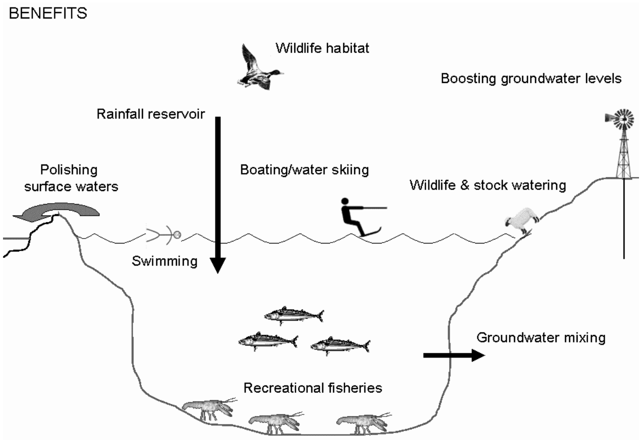

Figure 2. Potential benefits of pit lakes to communities and the environment

In contrast to the risks and liability that pit lakes may represent to companies, adjacent communities, and the environment, pit lakes may also represent significant benefits, frequently untapped in the pursuit of lease viability and profitability (Table 1). As an example, Australia has the lowest rainfall of any continent and water has been recognised as a limiting and highly valuable resource (Smith 1998). Some of these pit lake opportunities, such as recreational swimming, are organic developments of their communities and, whilst unrecognised and unregulated by local authorities, are already well-established in many arid mining regions with reasonable pit lake water quality (pers. obs.).

Other pit lake opportunities will require specific and direct support from mining companies and regulatory authorities, and the willingness and acceptance of local communities, who will often be a direct beneficiary of such opportunities, e.g. aquaculture and irrigation, direct contributions to local business ventures, employment, and associated income (Doupe and Lymbery 2005).

A widely accepted definition of health takes the potential benefits of a pit lake legacy even further: "health is a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity" (World Health Organization 1946). Like any significant geographical feature, pit lakes represent focal points to remote Australian mining communities, in an otherwise featureless landscape. These focal points also serve to engender the psychological benefit of a sense of place in addition to the more tangible end use benefits (Kozlowski and Hill 2000).

Water Quality Issues

The substantial cost of finding, developing and accessing water sources has meant that the mining industry has become adept at optimising water consumption through recycling and development of technologies that minimise water use (Western Australia Chamber of Minerals and Energy 2004).

Nevertheless, many domestic mining operations are located in arid areas across Australia and are still restricted by the availability of water resources. Pit lakes represent a huge potential source of water for mining companies and their communities and the local environment. Although pit lake opportunities may be desirable, a fundamental constraint is frequently the existing or future pit lake water quality

Table 1. Examples of Australian pit lake end uses beneficial to mining companies, their local communities, and the natural environment, and an Australian example of that end use (end use categories, after Doupe et al. 2005).

Beneficial end use type	Example of end use opportunity taken	Example location (primary resource mined) and reference
Aquaculture	Assorted in-fish and marron	Granny Smith Mine, Goldfields (gold) Wesfarmers, south-west Western Australia (coal) (Otchere et al. 2004; Syddell 2004)
Industry water	Reduced salinity water for haul road dust suppression	Collinsville Coal Project, North Queensland (coal) (McCullough et al. 2006)
Irrigation	Mango horticulture	Enterprise Pit, Northern Territory (gold) (Pine Creek Community Government Council 2003)
Mitigation wildlife conservation	Constructed wetlands for waterfowl	Capel Lakes, south-west Western Australia (mineral sands) (Doyle and Davies 1999)
Potable water source	Remote mining town supply	Wedge pit, Goldfields (gold) (Australian Labor Party, 5 Feb, 2004)
Recreation and tourism	Boating, water skiing, bathing	Historic and new Collie pit lakes (coal) (Lund 2001; Chapman 2002; Western Australian Tourism Commission 2003)
Research and education	Formal and informal education opportunities	Most pit lakes have this capacity
Sacrificial	Saline river first-flush storage	Chicken Creek, southwest Western Australia (coal) (Bills 2006)

(Doupé and Lymbery 2005; Johnson and Wright 2003). The science of many remediation strategies is well-established and there is a broad range of remediation technologies to select from. Remediation, sometimes only required in the first years following pit flooding (Younger 2000), is also increasingly available for other water quality issues as international interest in pit lake legacies continues to grow and people recognize that there are untapped potential pit lake benefits (Klapper and Geller 2002).

Conclusions

Pit lakes will continue to contribute to the legacy of the mining industry across the globe. However, knowledge of pit lake science and the interaction and utility of these features for adjacent communities is often inadequate for much of Australia's differing climatic, geological and social regions. As a result, pit lake currency and prediction of some of these regions are particularly poorly understood, especially in the dominant mining areas of the Australian semi-arid and arid interior.

It follows that a pit lake management view that only considers minimisation of liability may miss significant opportunities for maximising the benefits that these water sources can offer; both now, during mine operation and in the future after the mine lease has been relinquished. Although beneficial end uses for pit lakes have potential for environmental impacts and an actual or perceived impact upon human health and safety (Doupé & Lymbery 2005), end use opportunities and benefits extend beyond the mining company to the local community and also to the environment (Noronha 2004).

There has been growing recognition in the last decade of the need to plan for mine closure; increasingly even before mining operations begin (Brown 2003). Although mining companies and their local communities will be clearly oriented primarily towards mining operations as the major industry in the area, an overly narrow view of mining being the only successful use for the lease land may fail to recognise that pit lakes may be a *boon* to the post-mining community. Further, communities benefiting from pit lakes are also more likely to support lease relinquishment than those that are left with a neutral situation, or even worse a liability remaining from their local pit lakes. Water quality may initially, or eventually, restrict many of end use opportunities. However, current and emerging technologies may enable remediation of these mine waters to standards whereupon they can then be used for many of these beneficial end uses. Nevertheless, in order for pit lakes to be a viable relinquishment option for a company, community and the environment, a management strategy for the development and final form of the pit lake must be considered well before rehabilitation operations have begun (Evans and Ashton 2000; Evans et al. 2003). Consequently, pit lakes need to be planned for (Evans and Ashton 2000; Evans et al. 2003), not only to minimise risks, but also to maximise these opportunities for benefit.

In conclusion, for best sustainable management of lease resources for companies, communities and the environment, pit lake management should be more than simply parochial meeting of regulatory criteria to lease relinquishment. Assessing current and potential end uses for pit lakes is an important, yet little-recognised way, in which significant benefits to all three of these stakeholder groups can be made over an indefinite long-period of time, and in a mutually beneficial fashion.

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Ecological Restoration of Novel Lake Districts: New Approaches for New Landscapes

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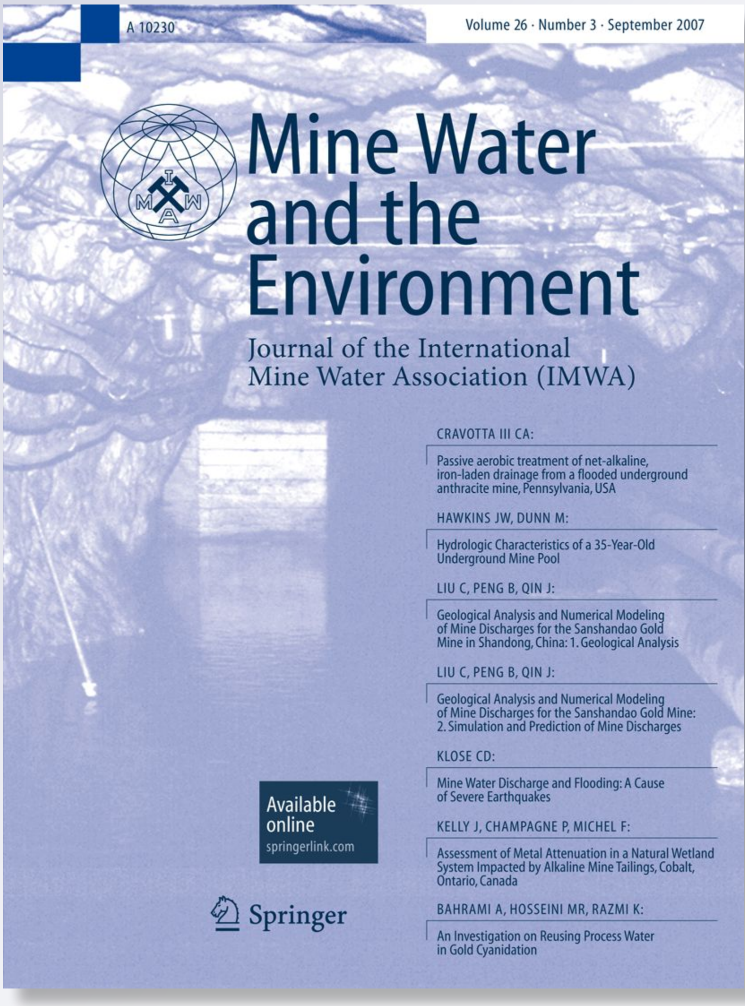

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Ecological Restoration of Novel Lake Districts: New Approaches for New Landscapes

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Abstract Mine void pit lakes often contain water of poor quality with potential for environmental harm that may dwarf other mine closure environmental issues in terms of severity, scope, and longevity. This is particularly so when many pit lakes occur close together and thus form a new "lake district" landscape. Pit lakes that can be developed into healthy lake or wetland ecosystems as a beneficial end use provide opportunities for the mining industry to fulfil commitments to sustainability. Clearly articulated restoration goals and a strategic closure plan are necessary to ensure pit lake restoration toward a new, yet regionally-relevant, aquatic ecosystem, which can achieve sustainability as an out-of-kind environmental offset. Such an approach must also consider obstacles to development of a self-sustaining aquatic ecosystem, such as water quality and ecological requirements. We recommend integration of pit lakes into their catchments as a landscape restoration planning exercise with clearly-identified roles and objectives for each new lake habitat and its surrounds.

Keywords Australia · Germany · Mining · Pit lake · Rehabilitation · Restoration

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Introduction

With increased frequency and growing scale, open-cut/cast mining has left a legacy of many thousands of mine pit voids worldwide (Castendyk and Eary 2009; Klapper and Geller 2002). Where backfilled pits is not an economic or feasible option and the pit extends into the water table, then pit lakes ranging from very deep (e.g. hard rock mining pits >250 m deep) to shallow (e.g. dredge ponds <10 m) may form (Castro and Moore 2000).

At the same time, there is a growing demand on natural water resources. Many regions have seen reduced regional recharge and decreases in water quality through pollution, leading to damage or complete loss of aquatic habitats (Pyke 2004). This demand has been simultaneous with increased mining and may sometimes be a direct result of this activity. These pressures continue to contribute to an international loss of aquatic habitat types ranging from seasonal wetlands to entire lake systems.

Rehabilitation of post-mining terrestrial landforms to provide restored ecosystems has now become a well-researched (and generally successful) practice that borrows from both disciplines of ecology and engineering. Indeed, post-mining rehabilitated ecosystems are a significant landscape feature in many regions with mining history. However, this landscape restoration typically ceases at the edge of open-cut/cast pits, except where backfill and/or landscaping directly incorporate the pit back into the surrounding terrestrial ecosystem. Generally, pit lakes are left unconsidered in this process, as 'elephants in the mine closure room'. Geochemical weathering processes, such as acid and metalliferous drainage (AMD), may then lead to poor water quality, resulting in lake waters toxic to aquatic life (McCullough 2008). Such quality impaired pit lakes typically have few environmental values and may even detrimentally affect regional water bodies through contamination of surface and groundwater sources (McCullough and Lund 2006). Such pit lakes are often a social and environmental liability to the surrounding region (Doupé and Lymbery 2005), yet they are frequently underestimated in terms of their scope and the magnitude of their environmental impacts. Indeed, of all mine closure legacies, pit lakes frequently have the most severe environmental impacts, given that these impacts can continue even after the mine is closed and the greater catchment is rehabilitated (Younger 2002).

Notwithstanding the significant environment and community problems that can be caused by pit lake landscape features, a number of pit 'lake districts' have formed over the past few decades, or are currently being formed for closure and return to state governments over the next few decades (Table 1). Through improvements in scale of mineral extraction technology, these more recent pit lakes are generally deeper and of greater volume than older lakes. Although it is often assumed that pit lakes will follow an evolution from young to mature lakes, resulting in lakes with a well-developed ecosystem (Kalin and Geller 1998), there are many examples of pit lakes formed soon after open cut mining became commonplace that have not improved in environmental quality or in biological measures (such as biodiversity and ecological function) many decades after forming (McCullough et al. 2009b). Instead, many pit lakes present continued risks to surrounding natural ecosystems, and it is likely that many of these new habitats will continue to display degraded ecosystems for many hundreds of years (Castendyk 2011).

However, pit lakes also represent significant opportunities. There are many potential benefits, most of which are untapped during mine closure planning by mining companies and regulators concentrating on terrestrial restoration outcomes. If appropriate restoration can be achieved, these large pit lake water bodies represent potentially valuable environmental and social resources (McCullough and Lund 2006; McCullough et al. 2009a), particularly in the face of global aquatic ecosystem losses (Sklenička and Kašparová 2008). Such post-mining use of an industry legacy would help advance expectations of best-practice mining environmental sustainability when pit lakes are final landforms.

This paper explores options for ecologically restoration that are rarely applied to pit lakes. We identify both opportunities and constraints within a contemporary mine closure and restoration context, and recommend regional planning strategies to best realise a restored pit lake ecosystem that will have significant environmental value and that will be successfully integrated into its broader ecological landscape.

Historical and Current Practice

Traditionally, pit lakes and even the pit void structure itself have rarely been considered in mine rehabilitation and closure plans, aside from geotechnical health and safety aspects. The latter have generally been achieved through simple structures such as earthen bunds and fences, e.g. DMP/EPA (2011). Some engineering technologies even take advantage of this isolationist perception to use pit voids as reservoirs for tailings storage or as sacrificial sumps for AMD or erosion products from over-burden and other disturbed landforms (Puhalovich and Coghill 2011; McCullough and Lund 2006).

Some good examples of pit lake rehabilitation have occurred in Germany and Central Europe where community pressure and interest in beneficial end uses have encouraged development of pit lakes (Schultze et al. 2011; Żurek in press). However, there are very few examples of restored pit lakes where pit lakes and their immediate surrounds have been rehabilitated to specifically restore *ecosystem* values (regional or otherwise). Where restoration has been achieved, it has sometimes been incidental (Charles 1998) and/or some ecosystem properties and processes developing naturally after many decades or even following filling. As a result, there are no demonstrative examples of pit lake restoration success for most regions and mining types. This has been because rehabilitation of pit lakes, let alone restoration of a sustainable ecosystem therein, has not often been a focus for mine closure planning. Indeed, the closest examples to restoration of an

Table 1 International examples of pit lake districts

Lake district	Country	No. of lakes in district	References
Athabascan oil sands region	Canada	0 current (26 proposed)	Westcott and Watson (2007)
Borská Nížina lowlands	Slovakia	11 current	Otahel'ová and Otahel' (2006)
Central German and Lusatian Districts; Rhenish District	Germany	370 current; 205 current	Schultze et al. (2011)
Collie Lake District	Australia	13 current (more proposed)	Kumar et al. (in press)
Iberian	Spain	22 current	Sánchez-Espanã et al. (2008)
Łęknica	Poland	>100	Żurek (in press)

ecosystem of regional value typically appear when aesthetics for recreation are foremost goals for rehabilitation. For example, woodland or forest may then be created to develop this recreational end use, which indirectly benefits the regional environment, e.g. Żurek (in press).

The use of water quality as the most common, and often sole, criteria chosen by regulators may be because most countries have well developed water quality guidelines that lend themselves to this application (Jones and McCullough 2011). Consequently, in some instances, pit lakes have been relinquished to the state with restoration requirements, or at least consideration, to state or national water quality guidelines. For example, stock water drinking guidelines tend to be applied by regulators and as rehabilitation goals if the regional economy is predominantly agricultural, and environmental guidelines used if this is an explicit state or federal requirement (e.g. Axler et al. 1998) or if there is risk of discharge to other regional water bodies. Given that environmental water quality guidelines typically surpass guidelines for other water body uses (e.g. industrial, agricultural use), relinquishing a pit lake with environmental water quality standards may allow for many other end uses as well. However, other, equally-important, ecological variables are generally not considered (Lund and McCullough 2011; McCullough et al. 2009a).

Pit Lakes As Out-of-Kind Environmental Offsets

Rehabilitation practices mitigating environmental impact during operations and then rehabilitating remaining disturbed or reformed terrestrial habitat will undoubtedly reduce overall environmental impact and biodiversity losses from the post-mining landscape. However, it is difficult to see how the goal of achieving no net biodiversity loss in their operations, or even a "net positive impact" (NPI) on biodiversity, often proposed by many 'blue chip' mining companies, e.g. Rio Tinto (2008), can be achieved when a significant proportion of the mine footprint becomes inundated at completion of mining and consequent cessation of dewatering. Instead, developing aquatic ecosystems in and around a pit lake may help to achieve this biodiversity through a form of out-of-kind offset (Fig. 1). Such offsets are now well-established in regulatory policy in many countries, including USA, Australia and South Africa (McKenney and Kiesecker 2010) where concerns about mining and, in particular, mine water issues such as pit lakes, are growing (Newmont Golden Ridge Ltd. 2009). In this closure planning model, all mining activities ranging from the direct impacts of mining through to access corridors and other peripheral disturbances away from the mine footprint result in net loss of terrestrial ecosystems. Many contemporary restoration strategies may redress this

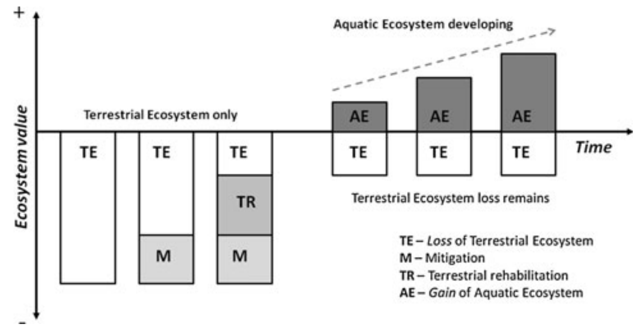

Fig. 1 Hierarchy of increasing biodiversity achievements through standard-practice terrestrial rehabilitation and then inclusion of pit lake aquatic ecosystem in post mining landscape restoration efforts (after NSW EPA 2002)

loss through mitigation of potential impacts and rehabilitation of impacted sites. Still, excavation of a vast open pit that floods to form a lake will result in irreversible net loss of terrestrial ecosystems. Recognition of the value of a developing aquatic ecosystem in the developing pit lake, and deliberate and targeted restoration of this ecosystem toward a regionally relevant aquatic ecosystem of value may be a suitable offset which redresses a net terrestrial ecosystem loss. Overall, there may even be a net ecosystem value (e.g. biodiversity) or gain following mine closure when the pit lake ecosystem is included in mine site rehabilitation accounting. As with terrestrial ecosystems, this gain is likely to significantly develop further ecological value over time as the lake fills and a burgeoning aquatic ecosystem develops.

Restoration theory and practice guidelines are generally well developed for terrestrial ecosystems and typically seek to restore the disturbed landscape towards a regional 'analogue system'. This analogue system may represent either the pre-mining ecosystem type that was lost or alternatively a local reference ecosystem. For example, in the case of a forest lost due to mining, a reasonable analogue ecosystem in the first instance would be the exact pre-mined forest type, but if this was not possible, then a regionally-appropriate forest could be selected. However, there is often a gross dichotomy between mine closure criteria for terrestrial and aquatic communities on rehabilitated mining leases. This difference of expectations for ecological goals at mine closure extends even to the edge of the pit lake, where riparian vegetation is seldom either representative of the region or self-sustaining (e.g. Fig. 2). Some of this difference may be because the former habitat was not a lake and the former terrestrial habitat has been covered by significant volumes of lake water.

However, wherever the degree of environmental modification is severe, such as typically follows mining, it is common that post-mining landscapes are modified to such an extent that restoration to a pre-mining landscape is

Fig. 2 A typical 'bathtub' ring effect showing failure of a functional riparian vegetation community to develop, WO3 lake (50 years old), Collie Lake District, Australia

virtually impossible. Similarly, terrestrial goals for the areas now occupied by the lakes need to be abandoned and alternative restoration goals must then be sought. Pit lakes and their terrestrial surrounds are often seen as classic examples of novel ecosystems, with combinations of species and environmental conditions not previously found (Hobbs et al. 2009). However, this need not lead to a complete abandonment of restoring ecological values; as a first goal, areas above water that will form lake riparian and catchment could, and should be, clearly identified and restored to integrate pit lakes into the broader regional landscape. Obtaining at least some properties and/or values of regional reference aquatic ecosystems may even be a preferred goal, especially where amphibious ecosystems are regionally rare (Brewer and Menzel 2009). The process of determining and defining appropriate goals and end point criteria for completion, as well as monitoring to ensure restoration is on the right trajectory to meet these goals (Society for Ecological Restoration International 2004), are therefore integral components of ecological restoration relevant to developing pit lake ecosystems.

Environmental Restoration Goals for Pit Lakes

Ecological sustainability is paramount to the regional value of these new lakes and their collective lake district. As with all restoration goals, although significant management intervention may be required during periods of physical and ecological development, the objective of management should be to restore an independently self-sustaining ecosystem for both terrestrial and aquatic habitats that is integrated into the new landscape. The first step in development of a pit lake ecosystem of environmental value is to identify an "identifiable desired state" (c.f. Grant 2006) as a restoration goal. Desired environmental values may come from a number of different, and often complementary, end points. They may, for example, include the pit lake and its catchment providing habitat for charismatic (typically waterfowl and mammalian) species (Santoul et al. 2004). Simultaneously, the pit lakes may provide seasonal habitat for migratory bird species. Although it is unlikely that the inherently artificial nature of the pit lake landscape will provide the often specific and narrow habitat and food requirements that many rare species require (Kumar et al. in press), some endangered species may still be able to use pit lake districts as a long-term refuge if the catchment-scale landscape approximates that of a natural lake district (sensu Brewer and Menzel 2009). Importantly, for the pit lake and its catchment to contribute value to the regional environment, there should be a restoration target for aquatic, amphibious, littoral, riparian (lake edge shallows and immediate terrestrial margin), and terrestrial ecosystems, that will have ecological value and are regionally representative (Van Etten 2011). It should be noted that, in order to contribute to regional biodiversity, species that are found in the lake should not be those that are already common elsewhere. A similar caveat may hold for the genetic diversity within species that occupy the new lake ecosystems and their catchments through artificial translocation or natural migrations. Lake district ecosystems dominated by limited gene pool or demes will likely be less genetically diverse and resilient than natural lake districts that have developed over many thousands of years (Shwartz and May 2008).

Compromised Ecosystems

Some pit lakes and their catchments may be so disturbed, such as through extensive and inappropriate (e.g. steep and eroding) terrestrial catchment and lake morphologies, or through ongoing chemical processes such as AMD, that they present long-term legacies of compromised ecosystems.

Such lakes will present no environmental value, even with pro-active restoration strategies in place. Other pit lakes may be deemed 'unmanageable 'or 'un-fixable' due, not to limited treatment or remediation knowledge, but rather a lack of financial or community will. Such lakes should then be regarded as impaired ecosystems. An outcome for these lakes has been proposed as 'novel' ecosystems that may contribute to scientific understanding through 'natural experiments', c.f. Hobbs et al. (2006). Indeed, there have even been proposals to maintain especially acidic lakes such as these as valuable extreme and unique ecosystems warranting protection under legislation (Nixdorf et al. 2005). It is unclear, however, how a regional ecology could ever benefit from the presence of such potential risk in the landscape. A more preferable stance may be that restoration endeavours for pit lakes and their districts need to look past traditional measures of restoration success such as approximation of regional physico-chemical quality and biotic diversity and assemblages and instead focus on fundamental ecological processes that potentially could be restored (Hobbs et al. 2009). Such basic processes include development of nutrient cycling, functional feeding groups, and/or trophic structures that might satisfactorily compare to those of regional reference aquatic systems.

Similarly, measuring pit lake values in terms of recovery or development of ecosystem structure (sensu Bell 2001) and services, such as habitat complexity, forms of carbon storage (e.g. through net respiration: production ratios) and other measures, may help identify contributions of significant ecological values to a region's natural landscape, even for highly impaired pit lake ecosystems. Whether the lake district is natural or anthropogenic in origin may be entirely academic to the provision of these ecosystem services. Indeed, such new constructs containing common or even alien species may present greater opportunity for ecosystem services than their natural counterparts in the landscape (Ewel and Putz 2004; Lugo 1992).

Restoring Ecosystem Values to Impaired Pit Lakes

Some impaired pit lake systems may naturally develop ecosystems of environmental value over time though natural, albeit slow, remediative processes such as succession driven along a restorative trajectory, e.g. water quality remediation by primary production and sulphate reduction (King et al. 1974). However, these processes may occur at too slow a rate or may be inhibited by negative feedback loops that preserve degraded states, c.f. Suding and Hobbs (2009), (Fig. 3). The ecological successions of pit lakes in these instances will need to be mediated by management interventions. Due to lack of long-term studies, it is largely unknown to what degree pit lakes, often described as examples of primary succession (Kalin et al. 2001), follow classic succession models, which generally presume gradual, predictable recoveries.

Prior to mining, a landscape dominated by terrestrial ecosystems has ecological values that are definable by measures such as biodiversity, presence of rare species, productivity, and other ecosystem services (Fig. 3). During mining, the ecosystems may face a substantial decrease in their terrestrial ecosystem value due to vegetation clearance and topsoil removal, excavation of overburden, and actual ore extraction, forming an open mining pit. In addition, road building, vehicular disturbance (dust and noise), and loss of habitat connectivity around the pit void will extend this phase of decreased ecosystem value.

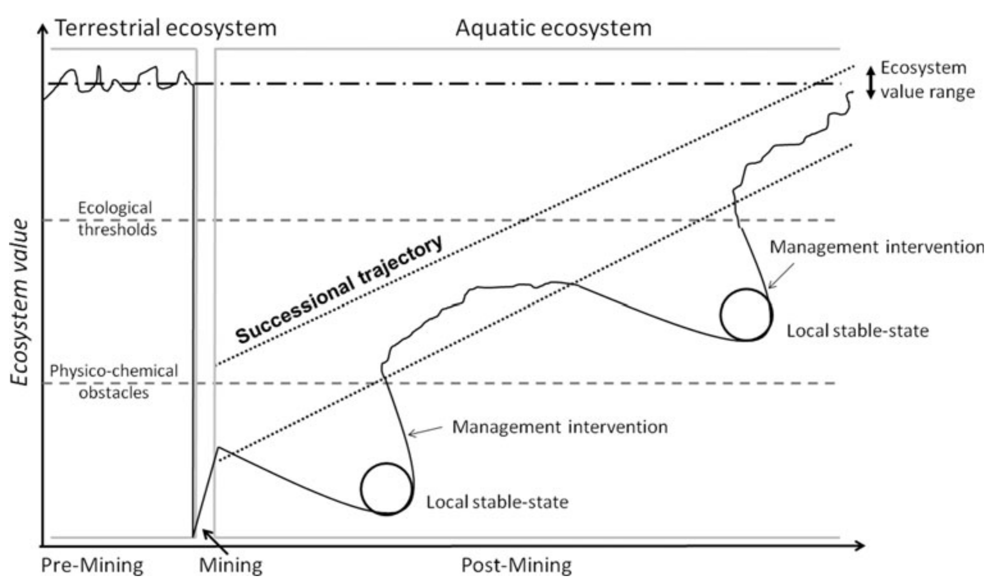

Fig. 3 Successional development of a pit lake ecosystem from low ecological value immediately following mining to attainment of prior ecological value, albeit now dominated by aquatic ecosystems. Local-stables states demonstrate fundamental ecological thresholds management restoration activities must overcome to realise a self-sustaining aquatic ecosystem of value (after Grant 2006)

Formation of the pit void and then flooding when dewatering ceases will mean significant loss of terrestrial habitat. Following rehabilitation of remaining terrestrial habitats, some terrestrial ecosystem of ecological value will be regained. However, the pit area will have been submerged and converted to an aquatic ecosystem habitat; terrestrial habitat thus lost cannot be rehabilitated and realised as terrestrial habitat ever again.

Through natural ecological succession processes, this evolving lake system may develop increased ecosystem values over time as some primary production begins both within and on lake banks and as fauna and flora colonise (Fig. 3). However, fundamental physico-chemical conditions may limit ecological development of the lake below a successional threshold, even at this early stage, such as through AMD toxicity or other water quality issues. The ecological consequences of AMD often include low species' diversity caused by pH stress and exposure to high concentrations of contaminants (Lee and Kim 2007; Nixdorf et al. 2001), low trophic states, low nutrient concentrations and low rates of primary production. Water quality is a master threshold factor for almost all pit lake ecological processes and especially for lower level species.

For example, pit lakes frequently display chemically-driven alternative stable states of poor water quality (sensu Sim et al. 2009). Abiotic processes may be the dominant determinant for that particular lake, e.g. ongoing and irreversible increases of salinity in regions of low net precipitation. Local stable states of poor water quality may also be due to biotic remediation processes that are present but weaker than their opposite and concurrent abiotic processes, for example catchment and internal formation of acidity occurring at greater rates than external and internal microbial driven-alkalinity generation processes. This state of aquatic ecosystem development may be very stable. For example, development of a basic self-sustaining food chain with phytoplankton algae in the lake is initially challenged by water toxicity and nutrient concentrations. In this example, management intervention to improve water quality, such as by active or passive remediation of AMD or similar issues, may be required before ecosystem development can continue to a high level of ecological complexity (Fig. 3). Thus, management intervention would be necessary to elevate the ecological succession path above a water quality threshold so that the pit lake ecosystem may continue to develop and achieve greater ecological value (Klapper and Geller 2002).

A pit lake ecosystem with high rates of primary production may contribute to the ecological value of a pit lake in many ways. Algal primary producers play an important role in natural lakes, providing the dominant allochthonous energy sources that are the basis of lake-ecosystem food webs (Bott 1996). Primary producers such as algae can facilitate sulphate production by providing a carbon source for sulphate reducing bacteria (SRB), which increase alkalinity and pH in AMD-impaired lakes (Lund and McCullough 2009). Algae can chelate metals directly, reducing toxicity and may absorb phosphorus and fix inorganic carbon into organic forms, reducing microbial carbon limitation and accelerating development of a natural food chain (Nixdorf and Kapfer 1998).

Conversely, AMD may lead to low pH and high acidity, increased metal and/or other contaminant concentrations, and a paucity of the macro-nutrients carbon and phosphorus, thereby limiting primary production rates and primary producer biomass. These limitations may then cascade as bottom-up controls on higher trophic levels and reduced abundance of taxa (McCullough et al. 2009b). Pit lake restoration efforts in this instance might target the biotic processes needing assistance or the water quality issues that limit ecological succession, such as low nutrient levels (e.g. phosphorus, carbon) (Tittel and Kamjunke 2004).

Adequate and appropriate revegetation within catchments is also important in developing functional lake riparian vegetation which, in turn, may play a key role in many pit lake ecological processes. Even with good pit lake water quality, many pit lakes fail to attain bank vegetation of any description, even after many years (Fig. 2). Riparian vegetation is important to integrate pit lakes into their greater catchments and to form connected and functioning landscapes. There may also be interactions between terrestrial and aquatic ecosystem components remediating physico-chemical water quality issues, in addition to providing ecological habitat. Individual ecological components must be clearly identified and considered in the context of the overall ecosystem as well as the pit lake ecosystem development. The contribution of organic carbon from riparian and catchment vegetation was recognised many years ago as a primary causative factor in water quality improvements in AMD pit lakes (Campbell and Lind 1969). Riparian vegetation may also contribute to bank stabilisation, facilitating further littoral and riparian establishment. The development of sustainable pit lake communities for finfish and large crustacea require an environmental suite that is more holistic than just water quality, including habitat such as fallen logs and bank overhangs, as well as food resources (McCullough et al. 2009b; Van Etten 2011).

Conclusions and Recommendations

Aquatic habitats are increasingly diminished in their frequency and quality through both local and global anthropogenic activities. Concurrently, the growing activities of open-cut mining are contributing potential lake aquatic habitats to post-mining landscapes. These pit lakes environments often display impoverished ecologies of little value to a regional environment and may even present an environmental risk to nearby natural water bodies due to poor water quality and/or other ecological factors. There is often little or no planning for a functioning pit lake of targeted ecological value. Pit lakes are often overlooked in rehabilitation efforts because aquatic habitats were not present previously in many of these disturbed mining locales. Nonetheless, pit lakes represent significant landscape restoration opportunities for replacement (or offset) of lost terrestrial habitat values with the alternative habitat values of an aquatic landscape or entire lake districts.

Fundamental restoration theory directs mine closure planners of post-mining landscapes that will contain pit lakes to first identify end use values. These may be environmentally specific endpoint values or endpoints that still allow alternative uses, such as recreation or aquaculture/agriculture.

How do we 'restore' pit lakes as ecosystems then? Achieving a desirable pit lake ecosystem involves more than just attaining good water quality. Water quality guidelines are only the beginning. Recognition of limiting factors to development of a self-sustaining ecology of regional values is essential. It must also be recognised that there the scope for management manipulation of the lake will be much more limited after filling; therefore, any obstacles to ecosystem development should be identified and remedied as much as possible prior to filling, starting with water quality. Establishing environmental values at higher levels than simply improved water quality must incorporate ecological approaches, a goal which is frequently ignored by restoration managers and regulators (Lund and McCullough 2011; McCullough et al. 2009b). Such an approach may assist in clearly articulating targets for the long term sustainability of pit lake districts. Such ecological versus physical/chemical-driven approaches should also recognise that mine water-affected landscapes, such as pit lakes, are more than just a geochemical environment, with consequent (though often simple) requirements for fundamental limnological and ecological processes that also need to be addressed if restoration to a representative functional ecosystem is to be successful.

Although it is likely that the broad environmental requirements for food and habitat will be very similar to those in natural systems, pit lake biota and their ecological requirements remain rarely studied and poorly understood. As such, there remains a pressing need for catchment-scale rehabilitation attempts of pit lakes to move towards development of aquatic ecosystems as a best practice. These restoration attempts are likely to initially fall short of attaining satisfactory ecosystem values due to a lack of knowledge of general pit lake ecological processes and intrinsic site-specific considerations. However, monitoring and ad hoc investigation studies of combined physico-chemical and ecological characteristics of these early attempts will provide fertile insight for future restoration attempts.

In conclusion, we hope that this paper serves to develop the field of mining closure planning by considering pit lake ecosystems as desirable and valid restoration goals. Considering mine water legacies in the context of their catchments, and vice versa, will also lead to realisation of more holistic environmental benefits to post-mining landscapes. We trust that the transdisciplinary perspective offered will translate into improved community and regulatory involvement in mine closure planning, and will encourage the mining industry to seek out additional opportunities to effectively achieve environmental sustainability targets when presented with new landscape challenges.

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TECHNICAL ARTICLE

Mine Closure of Pit Lakes as Terminal Sinks: Best Available Practice When Options are Limited?

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Abstract In an arid climate, pit lake evaporation rates can exceed influx rates, causing the lake to function as a hydraulic terminal sink, with water levels in the pit remaining below surrounding groundwater levels. We present case studies from Western Australia for two mines nearing closure. At the first site, modelling indicates that waste dump covers for the potentially acid forming (PAF) material would not be successful over the long term (1,000 years or more). The second site is a case study where PAF management is limited by the current waste rock dump location and suitable cover materials. Pit lake water balance modelling using Goldsim software indicated that both pit lakes would function as hydraulic terminal sinks if not backfilled above long-term equilibrium water levels. Poor water quality will likely develop as evapo-concentration increases contaminant concentrations, providing a potential threat to local wildlife. Even so, the best current opportunity to limit the risk of contaminant migration and protect regional groundwater environments may be to limit backfill and intentionally produce a terminal sink pit lake.

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Introduction

Due to operational and regulatory practicalities, pit lakes will continue to be common legacies of many mine lease relinquishments. Weathering of potentially acid forming (PAF) waste materials in pit lake catchments, such as pit wall rock, waste rock dumps, and tailings storage facilities, may produce acid and metalliferous drainage (AMD) that reports to nearby rivers and lakes (Younger 2002). Although material geochemical characterisation and placement/storage strategies are often available to mitigate or contain AMD production, many currently operating or planned mines do not have these considerations in place for a variety of historical and contemporary socio-economic and regulatory reasons (Hilson and Haselip 2004; Botha 2012).

AMD-degraded water quality in pit lakes may reduce regional environmental values and may present risks to surrounding communities and environmental values (McCullough and Lund 2006; Hinwood et al. 2012). Mine closure guidelines and standards increasingly require chemical safety and long-term low risk to surrounding ecosystems for closure practices to be acceptable (ANZMEC/MCA 2000; ICMM 2008; DMP/EPA 2011). Unplanned or inappropriate management of pit lakes can lead to both short- and long-term liability to mining companies, local communities, the government, and the nearby environment during mining operations or after lease relinquishment (McCullough and Van Etten 2011).

As a consequence, most developed jurisdictions are consistent in their requirement for mining companies to plan and/or rehabilitate to minimise or prevent any potential deleterious effects of the pit lake water body on regional ground and surface water resources (Jones and McCullough 2011). The focus of most general or ad hoc pit lake regulation is to protect human and ecological communities from adverse effects of the pit lake. For example, in Australasia, closure guidelines are generally oriented to aquatic ecosystem protection, based on ANZECC/ARMCANZ (2000) criteria. Such guidelines generally emphasize either a demonstration of null-negative effects of the lake or require management to achieve the required level for compliance (Kuipers 2002). However, AMD treatment may be very costly and difficult to achieve in remote mining regions (Kumar et al. 2011). As a result, sustainable pit lake management aims to minimise short- and long-term pit lake liabilities and maximise short- and long-term pit lake opportunities (McCullough et al. 2009).

In an arid climate, pit lake evaporation rates can exceed water influx rates, causing the pit lake to function as a hydraulic 'terminal sink'. Mean water levels in these pit lake can remain below surrounding groundwater levels. This paper describes how a terminal lake approach was applied to meet regulatory concerns for mine closure planning to achieve better environmental outcomes for two mines in different highly active mining regions of Western Australia. Our study used simple but robust pit lake water balance modelling, incorporating both hydrogeological and meteorological variables to determine equilibrium pit lake heights relative to local groundwater levels. The resulting models indicated that the case study pit lakes would likely remain as mean terminal sinks long (at least hundreds of years) after closure.

Study Sites

There are many examples of successful dumping of mine waste under wet covers or at the bottom of pit lakes (Schultze et al. 2011). We present two case studies from semi-arid and arid Western Australia that are relevant to other arid regions with active mines, e.g. southwest US and South Africa. Both open-cut mining operations are remotely located hundreds to thousands of miles from population centres and regional services. Both are currently developing detailed mine closure plans and face difficulties with PAF materials management in above-ground waste landforms where potential cover materials in the regional environments primarily consist of highly dispersive clays and sand. Geochemical testing indicates both pit lake catchments are likely to develop AMD-degraded water quality over time (unpublished data).

We assumed that AMD runoff would be allowed to flow into the pit after closure, even though a safety bund (known in the US as a berm) might be constructed around the perimeter of the pit (DMP 2010). We assessed three post-closure scenarios for each of the open pits: pit not back-filled and a pit lake forming, pit partially backfilled to below pre-mining groundwater levels with pit lake forming; and pit fully backfilled.

Nifty Copper Operation, Aditya Birla

Nifty Copper Operation (Nifty) is located in the Pilbara region of Western Australia, approximately 1,200 km north-northeast of Perth (Carver 2004) (Fig. 1). The Nifty copper deposit is the most significant ore deposit in the Neoproterozoic Paterson region of Western Australia (Huston et al. 2005). The Pilbara has an arid climate with two distinct rainfall patterns: in summer, rainfall occurs from either tropical cyclones or thunderstorms, while winter rainfall is typically from low pressure trough systems. Average annual rainfall in this highly active mining region is low, ranging from 200 to 420 mm/year, while evaporation averages around 4,000 mm/year (Kumar et al. 2012). Monthly evaporation significantly exceeds rainfall throughout the year and seasonally ranges from around 150 to 200 mm per month from May to August (the dry season), up to 450 mm in December and January (the wet season; BOM 2012).

On a regional scale, the Nifty Copper Operation lies within the Paterson orogeny of the Paleoproterozic to Neoproterozoic era. The Nifty deposit itself is a structurally-controlled, chalcopyrite-quartz-dolomite placement of carbonaceous and dolomitic shale (Anderson et al. 2001). The Nifty mine pit lies in a syncline within shales of the

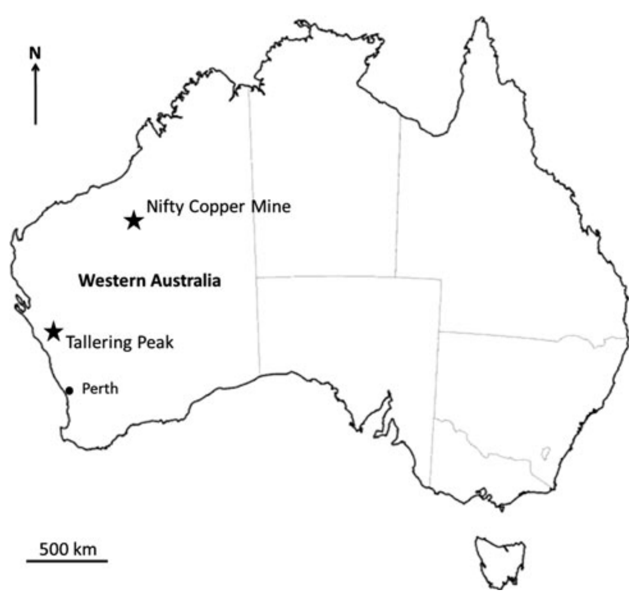


Fig. 1 Case study locations

upper Broadhurst Formation, which forms part of the Yeneena Supergroup. The folded shale of the Broadhurst Formation hosts the main copper ore body at Nifty Copper Operation and is strongly PAF. The mine stratigraphy consists of four units, the Foot Wall beds, the Nifty member, the Pyrite Marker bed and the Hanging Wall beds (Anderson et al. 2001).

Tallering Peak Iron Ore Mine, Mount Gibson Mining

Tallering Peak iron ore mine (Tallering Peak), which is owned and operated by Mount Gibson Mining (MGM), is located in the semi-arid midwest mining region of Western Australia (Kumar et al. 2012), approximately 300 km north of Perth (Fig. 1). Tallering Peak commenced production in 2004 and is predicted to continue operations until late 2013. Final landforms consist of mine pits T3, T4, T5, and T6A, and associated waste rock dumps. After closure, the partially backfilled mine void at T5, which is the largest pit, is expected to fill, mostly through groundwater inflow.

Arid Climate Conceptual Modelling of Pit Lake Water Balance and Water Quality

Climate is the most important factor of the hydrologic processes associated with a pit lake (Castendyk 2009). Changes in climate (e.g. temperature, rainfall, wind, precipitation amount, and distribution) affect individual hydrologic components differently. In general, surface hydrologic processes (e.g. direct precipitation, evaporation, and surface water runoff, including occasional stream or river inflows) are defined by regional climate to form a simple water balance budget for the pit lake (Fig. 2). Groundwater inflows are generated from precipitation recharge and tend to buffer short-term climatic changes, but long-term climatic changes will be reflected in groundwater inflows. Modelling of such groundwater and climate processes is often used to predict final water balances in pit lakes (Vandenberg 2011).

Post-closure pit lakes in an arid environment are typically classified as either 'through-flow' (Fig. 3a) lakes or terminal sinks (Johnson and Wright 2003). Terminal sinks may occur in arid climates where the evaporation potential is higher than average rainfall runoff (Niccoli 2009). During groundwater level rebound at the end of mining and pit void filling, the pit lake water level rises to a level where inflows (direct rainfall, catchment and pit wall runoff, and groundwater inflow) are in equilibrium with evaporation losses. Hence, pit lake water level does not rise to levels higher than adjacent groundwater levels and water does not seep into the groundwater system. The water quality of terminal sink lakes is expected to show increased acidity, metals, and salt concentrations over time as solutes introduced through groundwater inflow and pit wall runoff are concentrated by evaporation (Fig. 3b) (Miller et al. 1996).

Following groundwater rebound and dissolution of the cone of depression the pit lake begins to fill with water and groundwater influx into the pit initially increases as the influx area increases. Later, discharge slows as the change in head decreases (Gammons et al. 2009). As a result, total inflow into the pit lakes is expected to gradually decrease as the open pits fill, while total outflow is expected to increase due to increased evaporation from the greater lake area. At some stage, total inflow approximates total outflow and the water level in an open pit will reach equilibrium, albeit responding dynamically to changes in seasonal precipitation and evaporation rates. Water level fluctuations may occur, e.g. due to occasional cyclones.

If the steady-state pit lake elevation stabilizes below the surrounding pre-mining groundwater level, the pit lake becomes a terminal sink, with no water released into the environment through seepage into the groundwater system. However, if the final pit lake elevation reaches the

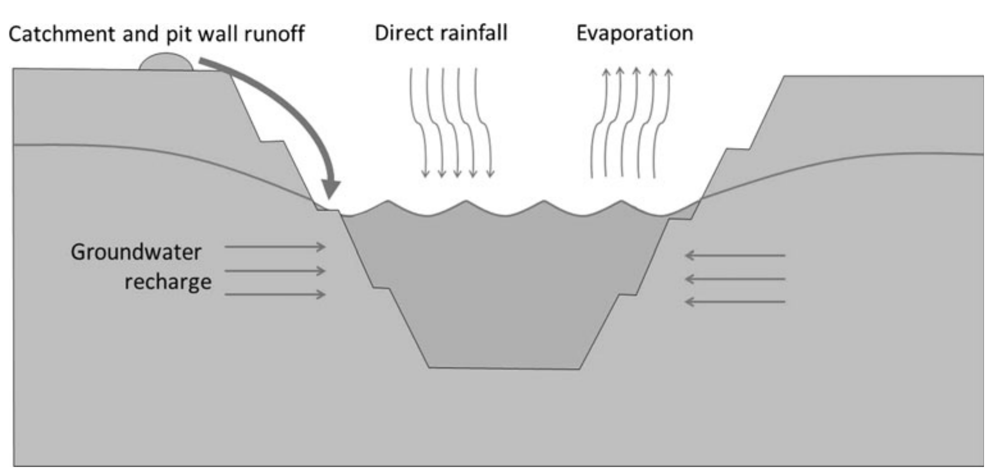


Fig. 2 Conceptual pit lake key water balance processes

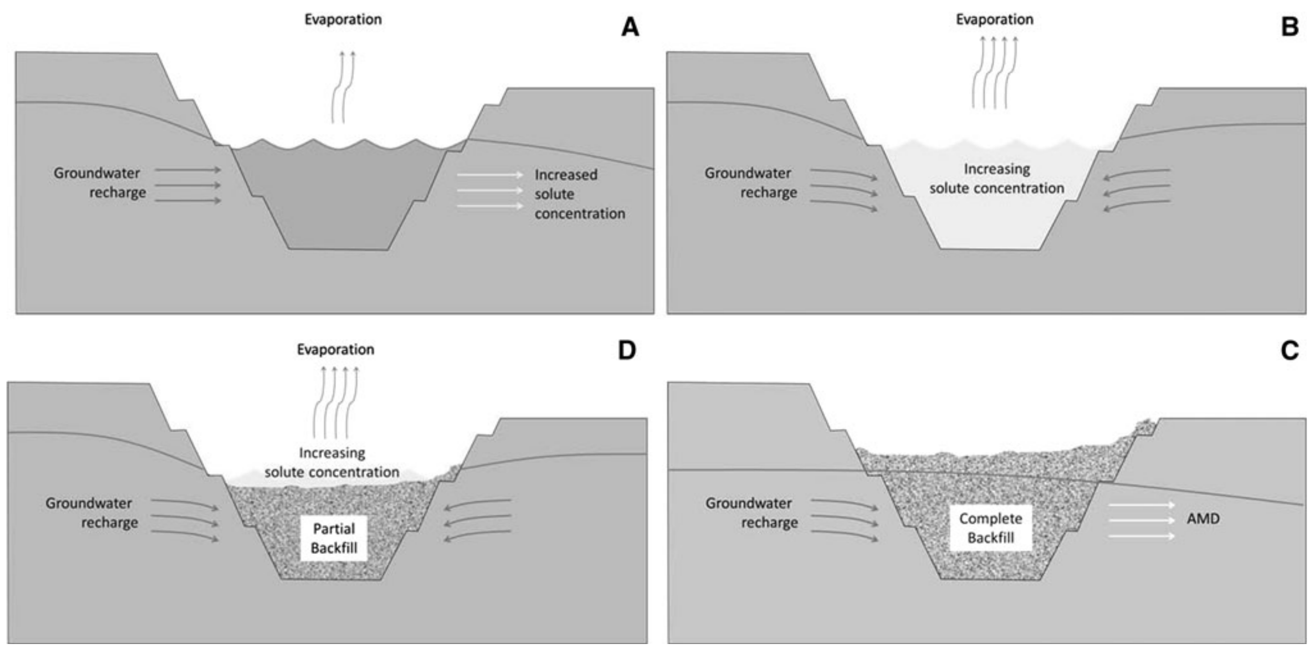


Fig. 3 Conceptual equilibrium hydrogeological regimes for an arid region pit lake. *Clockwise* from *top left* through-flow system, terminal sink, completely backfill through-flow system, partially backfill terminal sink

surrounding pre-mining groundwater level, the pit lake becomes a through-flow system with water being released to the environment through groundwater seepage, potentially spreading AMD plumes to environmental receptors.

Complete backfill is often recommended to avoid many issues associated with poor pit lake water quality developing from weathering of PAF material in the pit void and pit walls (Puhalovich and Coghill 2011) (Fig. 3c). If backfill volumes and distributions are small enough to permit accumulation of water above the backfill (Fig. 3d), then this use of the pit void will remove the mine waste from the typically higher rates of weathering and transport encountered when placed above ground. However, the pit backfill volumes and/or placement may also cause pit lake surface area reductions as waste is typically placed in the pit by tipping over the high wall. This change in surface area can thus alter the pit lake hydrological balance by decreasing net evaporation, which can change the pit lake from a terminal sink lake to a through-flow type. If the water quality in the pit lake is poor, this contaminated water may be released into the regional groundwater system.

Empirical Modelling

A water balance model for each of the closure scenarios was modelled using GoldSim software (Goldsim 2011). GoldSim is a Monte Carlo simulation software package for dynamically modelling complex systems. Monte Carlo simulations are a class of computational algorithms that rely on repeated random sampling of those components of the model with inherent uncertainty in their estimation when undertaking the simulations. Monte Carlo methods are especially useful for simulating systems with many coupled degrees of freedom, such as fluids. Pit lake hydrological inflows were defined as direct rainfall, runoff (catchment and pit wall), and groundwater inflow. Outflows were defined as evaporation from the lake surface, groundwater seepage (if any), and overflow (if any).

A GoldSim variation of a multi-state Markov chain model first developed by Srikanthan and McMahon (1985) was used to generate stochastic rainfall data from rainfall data from each mine site, with data gaps amended by correlation with publically available data from the nearest Bureau of Meteorology (BOM) weather station. In basic terms, the model generated a synthetic sequence of daily precipitation based on the probability of rainfall in one ‘state’ (states are essentially ranges in daily rainfall and are subject to user-defined limitations in terms of both their quantity and internal boundaries) being succeeded (on the following day) by rainfall in the same or another state. These probabilities were collated in a transition probability matrix (TPM). Seasonality was modelled by using 12 separate TPMs, one for each calendar month. The inputs required by GoldSim to generate stochastic data are the TPMs for each calendar month, the number of states and their boundaries, and distribution parameters derived for those days that exceed the adopted upper range of rainfall used to define the monthly TPMs.

The model was calibrated using the daily, monthly, and annual statistics of the observed data (which was the

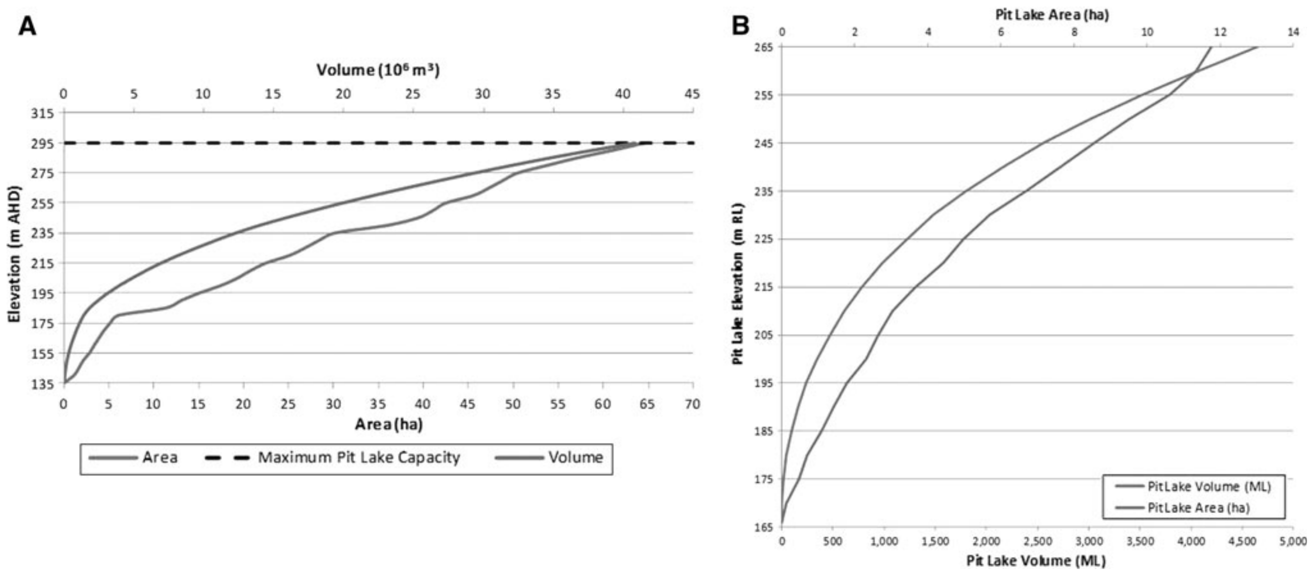


Fig. 4 Elevation-volume-area curve for (**a**) current Nifty Copper Operation and (**b**) current Tallering Peak open pits

primary input). This was achieved by iteratively (and manually) varying the limits of each state as well as the number of states to be considered in certain months. Following the calibration, 100 stochastic rainfall sequences were generated every 200 years in length to simulate the pit closure scenario models.

The volumes of rainfall onto the pit lakes were proportional to their associated surface areas (Fig. 4a, b). As the non-backfilled open pit filled with water, the lake surface increased and therefore, the volume of rainfall entering the lake increased. An elevation-volume-area curve developed for each pit design at closure was used to estimate the area of the pit lake as the water level rises. The curve was modified for the partially backfilled scenario based on the level of backfill, with 100 % of the direct rainfall first filling up the backfill void, based on the assumption that there was no pit outlet and that there was no evaporation during rainfall events. When the water level reached the backfilled elevation, the direct rainfall volume was based on the pit lake area above the backfill. In the case of the backfilled pit scenario, 50 % of the direct rainfall on the pit surface was expected to infiltrate directly into the footprint of the backfilled pit to fill backfill voids (Williams 2012).

Based upon site visits, pit wall rainfall runoff was assumed to mostly collect on mine benches or to evaporate before reaching the pit lake. However, for high rainfall periods, the runoff may then overflow the benches and flow into the pit lake; thus, a greater proportion of the runoff reaches the lake as rainfall increases. Also, if rainfall occurred on the previous day, ponds of water may still be present on the mine benches, coupled with higher antecedent moisture conditions; additional rainfall is therefore likely to overflow into the pit lake. Pit wall runoff coefficients were therefore applied to the rainfall to estimate runoff from the pit walls, based on the following empirical relationship:

$$\begin{aligned}\text{Runoff}_{\text{Pitwall}} &= \text{Rainfall} \times \text{Runoff coefficient}_{\text{Pitwall}} \\ &\quad \times (\text{Area}_{\text{Pitwall}} - \text{Area}_{\text{Pitlake}}) \qquad (1)\end{aligned}$$

The runoff coefficient was varied depending on the amount of rainfall and whether rainfall occurred on the previous day, based on regional site experience (Table 1). As the open pit fills with water, a portion of the pit walls are covered by water and the area of exposed pit wall is reduced, reducing the volume of pit wall runoff. In the partially backfilled scenario, the portion of the pit walls exposed stays constant until the pit lake water level rises above the backfilled level. In the fully backfilled scenario, the pit wall was not exposed and therefore there was no pit wall runoff.

The catchment area around each pit was estimated and it was assumed that up to 60 % of the runoff would occur during high rainfall events during cyclonic activity (BOM 2012) and that no runoff takes place during rainfall events less than 5 mm/day. Catchment runoff coefficients were applied to the rainfall to estimate runoff from catchments adjacent to the pit, based on empirical relationship (2). Runoff coefficients were assumed from calibrated values used in previous project experiences in the region which were within the range given published studies from Western Australia (Williams 2012). The catchment runoff coefficient for Nifty Copper Operation was assumed to vary depending on the amount of rainfall as shown in (Table 1).

Table 1 Tallering Peak pit wall runoff coefficients for pit void walls and catchment (after Williams 2012)

Daily rainfall (mm)	Previous day rainfall (mm)	Walls	Catchment
<5	N/A	0	0
<40	<20	0.30	0.15
<40	>20	0.65	0.40
≥40	N/A	0.65	0.40

Runoff coefficient is the rainfall fraction incident on the surface that does not infiltrate; *N/A* not applicable

$$\mathrm{Runoff}_{\mathrm{catchment}} = \mathrm{Rainfall} \times \mathrm{Runoff\ coefficient}_{\mathrm{Catchment}} \times \mathrm{Area}_{\mathrm{Catchment}} \quad (2)$$

Nifty Copper Operation groundwater inflows (Q) were estimated using the Dupuit Equation for horizontal flow conditions as the main aquifer through the pit is an unconfined channel aquifer that the lake excises:

$$Q = K\left(\frac{h_1^2 - h_2^2}{L}\right) \quad (3)$$

where K is the average hydraulic conductivity of the rock mass, h_1 is the pre-mining groundwater elevation, h_2 is the pit lake elevation (which increases as the pit fills up with water), and L is the horizontal flow length from pre-mining water level to the pit lake surface (h_1 to h_2) (Fetter 1994).

Tallering Peak Iron Ore groundwater inflows were estimated using the Dupuit-Forchheimer equation for radial flow conditions for an unconfined aquifer:

$$Q = \pi K \frac{h_0^2 - h_w^2}{\ln \frac{R}{r_w}} \quad (4)$$

where K is the average hydraulic conductivity of the rock mass, h_0 is the pre-mining groundwater level, h_w is the pit lake level (which increases as the pit fills up with water), r_w is the pit diameter at the base, and R is the radius of Influence that can be expressed using the Cooper-Jacob equation (Cooper and Jacob 1946) as:

$$R = 1.5\sqrt{\frac{Kbt}{S_y}} \quad (5)$$

where b is the thickness of the aquifer, t is the time since the start of mining operations, and S_y is the specific yield of the aquifer. The equation indicates that groundwater inflows will decrease as the pits fill up with water and the radius of influence increases with time.

A partially backfilled option for the T5 pit was assessed on a proposed volume of backfilled PAF material. Based on the slope of the pit wall (32°), we assumed that the backfilled material would be disposed of in the bottom of the pit and not by end dumping from the edge of the pit.

MGM supplied Golder with the open pit shells for T5 at the end of mining, from which we created the elevation-volume-area relationship (Fig. 4b). Rainfall from the last 30 years was assumed to be representative of the current rainfall conditions on-site and was used to generate a stochastic rainfall distribution. The runoff coefficient for the pit wall was assumed to be 80 % from calibrated values used in previous project experiences in the region. The evaporation data applied in the model were obtained from the SILO Data Drill (http://www.nrm.qld.gov.au/silo). The Data Drill accesses grids of data interpolated from point observations by the Bureau of Meteorology. Interpolations are calculated by splining and kriging techniques. The data in the Data Drill were all estimated as there are no original meteorological station data available in the calculated grid fields. A monthly “Class A” lake to pan coefficient (BOM 2012) was used to estimate evaporation from the pit lake surface (Hoy 1977; Hoy and Stephens 1979).

Evaporation loss was not considered in the fully backfilled pit scenario; however, when the water level exceeded the backfilled elevation in the partially backfilled scenario, evaporation was simulated in the models. A reduction in evaporation rates was assumed as the depth of the lake surface below the adjacent ground level increased to reflect the influence of reduced wind across the lake surface.

Groundwater seepage from the pit lake into the groundwater system will occur when the water level within the pit reaches a level greater than the surrounding groundwater level. Thus, groundwater seepage was estimated using Eq. (4) for radial flow conditions and an unconfined aquifer when h_w was greater than h_0.

Results

Nifty Copper Operation

The open pit scenario with no backfill was identified by modelling as an evaporative sink (Fig. 5a). Modelling of the partially backfilled scenario showed that the equilibrium pit lake water level would be more than 10 m above the elevation of the backfill, and was identified as a terminal sink due to the equilibrium pit lake water level being lower than the surrounding groundwater level (Fig. 5b). The fully backfilled scenario indicated that the pit would become a through-flow system with water contained in the backfilled pit seeping into the groundwater system (Fig. 5c). If the PAF material already contained in the pit leached chemicals harmful to the environment, this closure option may present a significant risk at mine closure.

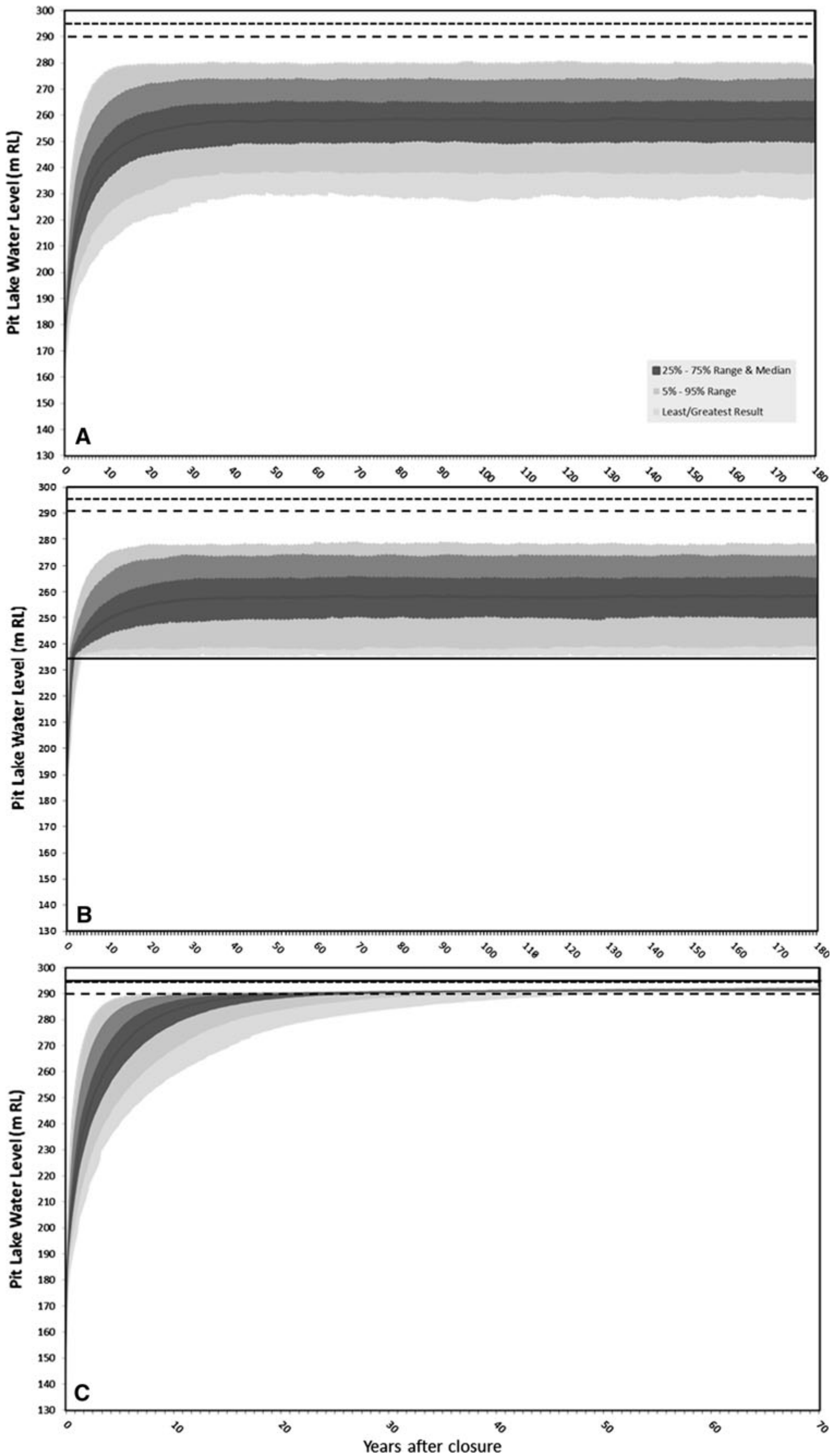


Fig. 5 Nifty Copper Operation predicted pit lake levels for: **a** No backfill; **b** Most reactive waste partially backfilled, and: **c** Complete backfill. *Dotted black line* indicates pit lake overflow level, *dashed line* baseline groundwater level, *solid line* backfill level

A partially backfilled option model was developed based on the proposed volume of backfilled material provided by the mining company at the time. The model results indicate that the pit lake water level would stabilise after about 40 years to a median level of 259 m AHD (Australian hydraulic datum level that corresponds approximately to mean sea level). The results also show that there is a 95 % probability that the lake level will not exceed 275 m AHD and a 5 % probability that it will not exceed 239 m AHD. The latter would cover the deposited waste material with 4 m of water. A pit lake level of 275 m AHD is equivalent to 20 m of freeboard and an additional pit lake capacity (e.g. for buffering volume during heavy rain events) of approximately 11.5 million m^3.

The hydrogeological system is expected to remain a sink, with equilibrium groundwater levels below the pre-mining groundwater level of 290 m AHD. Furthermore, pit lake levels are expected to stay below the static groundwater levels of 285 m AHD in the adjacent Nifty Palaeochannel, indicating that there is little risk of pit lake water flowing into the palaeochannel system.

This model showed two main consequences to long-term AMD management at mine closure if the pit was backfilled above the surrounding groundwater level:

1. Reduction in evaporative losses would likely lead to a through-flow scenario. As the proposed material was predominantly PAF, it is therefore likely that water quality would be impacted by AMD as it flows through the pit waste backfill. Due to the through-flow nature of the backfilled pit, the water would then be released to the environment as seepage from the lake to groundwater (Fig. 6), leading to an increased risk of negative effects on local and possibly regional groundwaters, and any dependent ecosystems.
2. Waste landforms without effective cover systems to reduce infiltration may generate and transport AMD if the partially backfilled pit lake did not function as a terminal sink. In this scenario, AMD leachate from waste rock dumps containing PAF would enter the vadose zone (the area of unsaturated ground above the groundwater level), but would not be transported in the local groundwater plume toward the pit lake, since it would not be acting as a terminal sink. Instead the AMD plume would be transported by the regional groundwater system and potential surface water receptors, such as groundwater-dependant ecosystems of seasonal lakes, creeks, and wetlands.

Tallering Peak

In the no-backfill scenario, model results indicated that the open pit would fill gradually and eventually reach equilibrium seven years after closure (Fig. 7a). The equilibrium water level would then be around 231 m RL (project area relative level); lower than the pre-mining groundwater level (estimated at 238 m RL).

The partially backfilled option was based on the proposed volume of backfilled material provided by MGM. In this scenario, the model results indicate that the open pit would gradually fill with water and eventually reach equilibrium five years after closure (Fig. 7b) at around 236 m RL, i.e. below the pre-mining groundwater level. The final pit lake would be above the backfill level, covering the PAF material. Oxidation rates of the PAF material might then be significantly reduced because of the much lower oxygen diffusion rates through water. A final terminal sink would also entrain AMD contaminated waters away from sensitive environmental receptors such as a nearby ephemeral creek that flows into the Greenough River.

In the fully backfilled scenario, the model results indicate that the backfilled material voids would fill with water

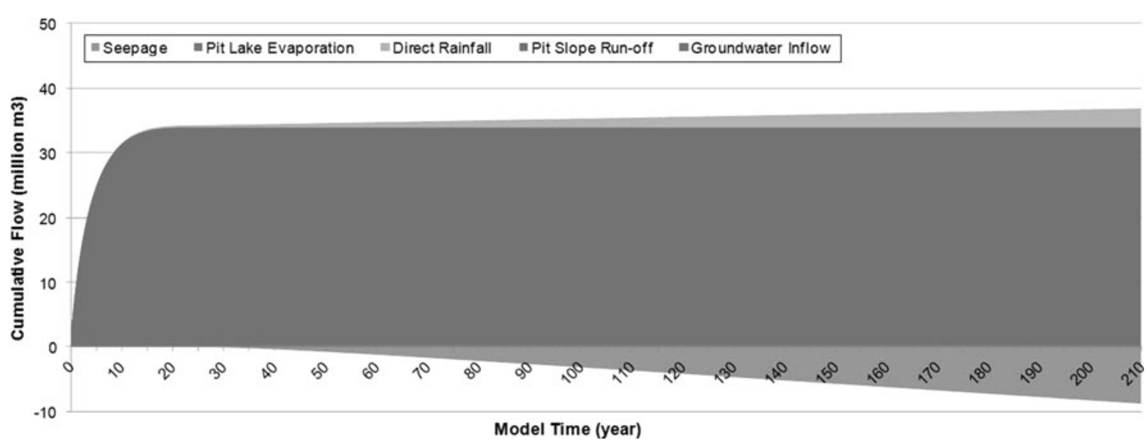
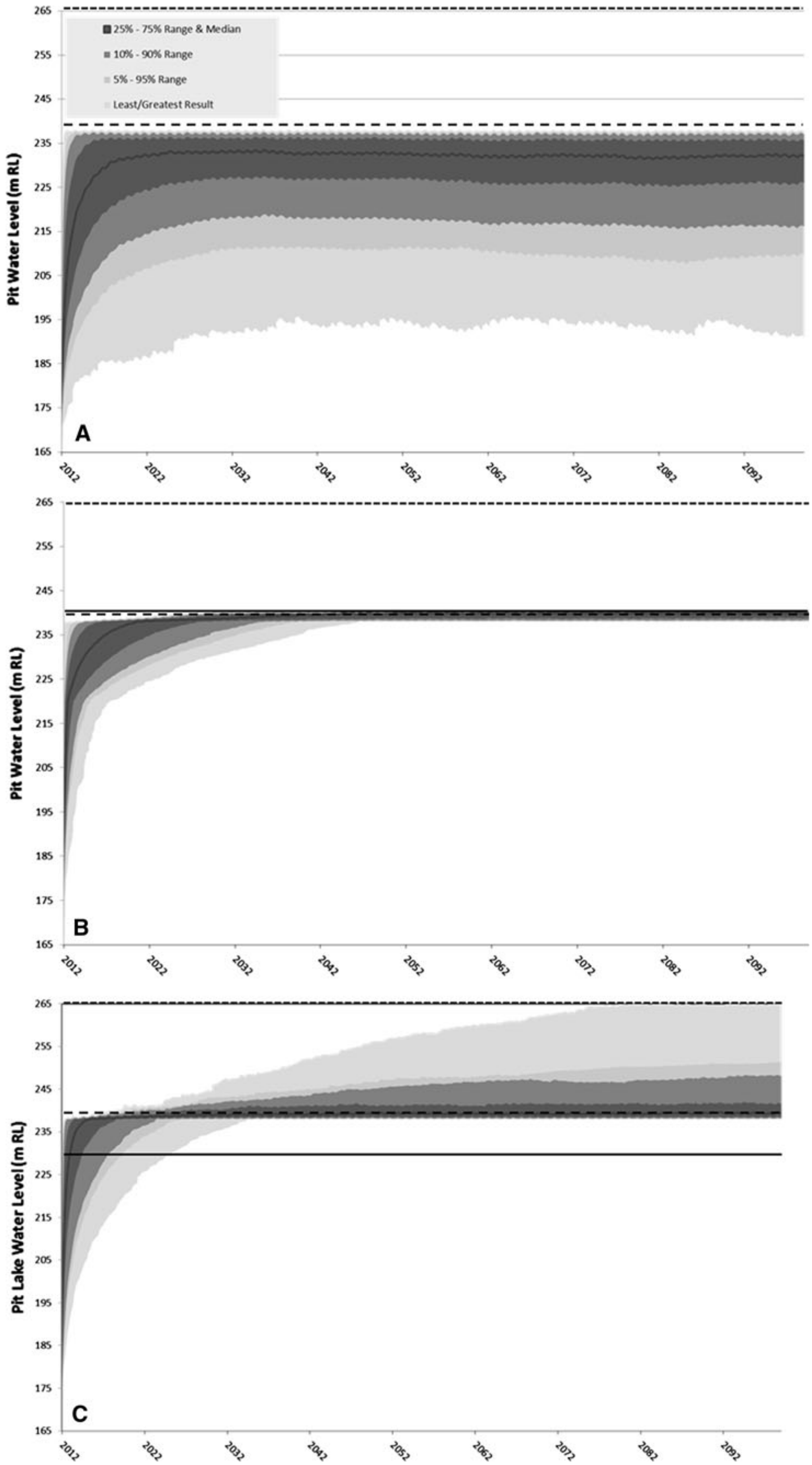


Fig. 6 Predicted cumulative flow for Nifty Copper Operation

Fig. 7 Tallering Peak T5 predicted pit lake levels for **a** no backfill, **b** most reactive waste partially backfilled and **c** complete backfill. *Dotted black line* indicates pit lake overflow level, *dashed line* baseline groundwater level, *solid line* backfill level



(Fig. 7c). The water level within the backfilled pit would reach equilibrium three years after closure at around 238 m RL, about the same as the groundwater level in the area.

While a terminal sink is unlikely to introduce leachable compounds into the local groundwater system, a through-flow system toward a seasonal creek line in the southwest will most likely affect the groundwater system. Based on our analyses, the open pit with no backfill and the partially backfilled scenarios were identified as likely terminal sinks. In contrast, the fully backfilled scenario was predicted to be a through-flow system and likely to introduce AMD into the groundwater system. Furthermore, there was a 5 % probability that after 35 years, the fully backfilled pit water level would rise high enough to decant to nearby surface waters.

Discussion

Mine closure is increasingly recognised as a whole-landscape development exercise that must take into account all closure landform elements and how they will interact over time (Younger and Wolkersdorfer 2004; McCullough and Van Etten 2011). The catchment provides an ideal scale at which to holistically consider performance and interaction of these closure elements; often, the catchment will be artificially constrained such that the lowest part where water reports in the open pit becomes a pit lake.

Both of these case studies indicate that a partial backfilled pit and the formation of a terminal sink pit lake may pose less environmental risk than a completely backfilled pit where contaminants could be transported by seepage to the groundwater (MCA 1997).

Nevertheless, the water quality of terminal sink lakes is expected to deteriorate over time due to evaporation (Eary 1998), particularly in highly alkaline or acidic lakes (Eary 1999). Since such lakes are essentially abiotic, with little or no attenuation of contaminants occurring, this poor water quality is unlikely to be resolved naturally, even over long time scales (McCullough 2008). Poor water quality in such lakes may pose a threat to local wildlife and migratory waterfowl and will have limited options for post-mining use. Although not desirable in itself, this water quality deterioration indicates that the pit lake is functioning as a terminal sink and protecting the greater undisturbed regional environment off the project footprint from seepage of AMD-contaminated water resulting from exposed pit wall or in-pit disposal of waste rock.

In the long term, increasing solute concentrations in the terminal sink pit lake would increase water density. This concentration change may cause density-driven flow into the surrounding groundwater under certain hydrogeological conditions (Gvirtzman 2006) and should be investigated as part of a complete risk assessment process for development of a definitive phase mine closure plan strategy.

Stability of physical and chemical conditions inside the deposited waste and at its interface with the lake environment is the main prerequisite for successful long term storage of waste in a pit lake (Schultze et al. 2011). As such, climate change should also be a key consideration in the development of pit lakes used as terminal sinks for mine closure. For example, an increasingly wet climate may lead terminal sink pit lakes to become through-flow through seepage or even decant, i.e. overflow. Similarly, even though mean net precipitation may not change, an increase in intense rainfall events such as cyclone frequencies may still lead to similar mobilisation of degraded pit lake waters. Such inappropriate application of an terminal sink conceptual model to sites that fail, even infrequently, to behave as terminal sinks may present risk to downstream water resources (Bredehoft 2005). Consequently, although pit lakes as terminal sinks may greatly reduce risk of off-site water quality problems, conditions such as decanting (through high/seasonal rainfall events or filling to higher level than expected) (Commander et al. 1994), density- driven seepage caused by increased salinity, or the pit lake rising to heights above surrounding groundwater levels during high/seasonal rainfall events, should be explicitly considered as part of the conceptual model driving a closure plan incorporating terminal sinks as key design elements. Further important considerations will be the potential for resource sterilisation through any backfill activity and health and safety considerations for both human and wildlife populations associated with retaining an open pit as a final landform. The latter may require a formal environmental risk assessment of the effects of a terminal pit lake in the closure landscape.

Conclusions

Although often prescriptively proposed as best practice by a number of regulatory and sustainability organisations, fully or partially backfilled pit may sometimes lead to poorer regional closure outcomes than retaining a pit lake of some form, especially in arid and semi-arid regions. This demonstrates the need to consider mine closure planning on a case-by-case basis as well as for closure strategies to be founded on good empirical evidence, with water balance and geochemical modelling results frequently being key considerations. Furthermore, a good knowledge of pre-mining conditions and groundwater system will almost always be mandatory to develop a reliable water balance model and predictive simulations of any pit closure scenarios.

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Consequences and opportunities from river breach and decant of an acidic mine pit lake

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ABSTRACT

Mining typically diverts natural water courses during operations which are then not reinstated at closure allowing an isolated pit lake to form in the open cut void. Pit lake water quality may then degrade over time.

A heavy rainfall event led a diverted river to breach a large acidic, coal mine pit lake allowing assessment of river flow-through for both lake and downstream river environments.

Lake and river water quality samples were interpreted and compared to end use value guidelines. Fresher, more alkaline and nutrient-richer river water interacted with saline and acidic pit lake water improving upper lake water quality and enabling beneficial end use opportunities.

There was no significant risk of toxicity to downstream river livestock drinking water during the period. However, water quality at all sites sampled (including a reference site) exceeded pH and Zn ecosystem protection guidelines, and some recreational and aesthetic guidelines.

River flow-through is being trialled as the most sustainable long-term closure option for this lake. Flow-through may also represent the best closure scenarios for mine pit lakes with similar socio-environmental contexts.

1. Introduction

Due to operational and regulatory practicalities, pit lakes are common mine closure legacies for many open-cut mines. Pit lakes may form in open cut mine voids extending below the water table when they subsequently fill with ground and surface waters upon cessation of dewatering operations. Where natural water courses have been diverted during operations, these original water courses are then typically not reinstated in order to minimise risk to downstream values.

Pit lake water quality is often degraded by acid and metalliferous drainage (AMD) leading to acidic water with elevated metal concentrations (Castro and Moore, 2000; McCullough, 2008). Salinity may also increase through time due to evapo-concentration processes (Niccoli, 2009; McCullough et al., 2013b). This impaired water quality may leave a significant liability at mine closure and may also reduce end use values and opportunities (McCullough and Lund, 2006). Closure guidelines increasingly require of post-mining land uses of equivalent capacity to pre-mining conditions (Jones and McCullough, 2011; Jones, 2012). Good mine closure minimises long term post-mining landform liabilities and maximises benefits to stakeholders and the environment. However, the risks presented by pit lakes are regularly neglected in mine closure planning (McCullough and Lund, 2006; McCullough et al., 2009; Vandenberg et al., 2015).

One strategy that is increasingly used to maintain or improve pit lake water quality from both salinisation and acidification is to direct river flow (often back to its original course) through pit lakes (McCullough and Schultze, 2015). River flow-through strategies can increase the range of end uses available at closure and minimise long term lake liability issues for example, so that beneficial end uses dependent upon the water quality such as irrigation, recreation and wildlife habitat can be achieved (McCullough and Lund, 2006; McCullough and Van Etten, 2011).

A number of pit lakes have now used a river flow-through as part of closure including many pit lakes of the Central German Mining District (Klemm et al., 2005; Schultze et al., 2011b); British Columbia, Canada (Pelletier et al., 2009); Tennessee, USA (South Pit Lake, Wyatt et al., 2006); Northern Territory, Australia (Enterprise Lake, Fawcett and Sinclair, 1996), and Waihi, New Zealand (Golden Cross, Ingle, 2002; Castendyk and Webster-Brown, 2006). Flow through is also proposed in closure planning for pit lakes in the

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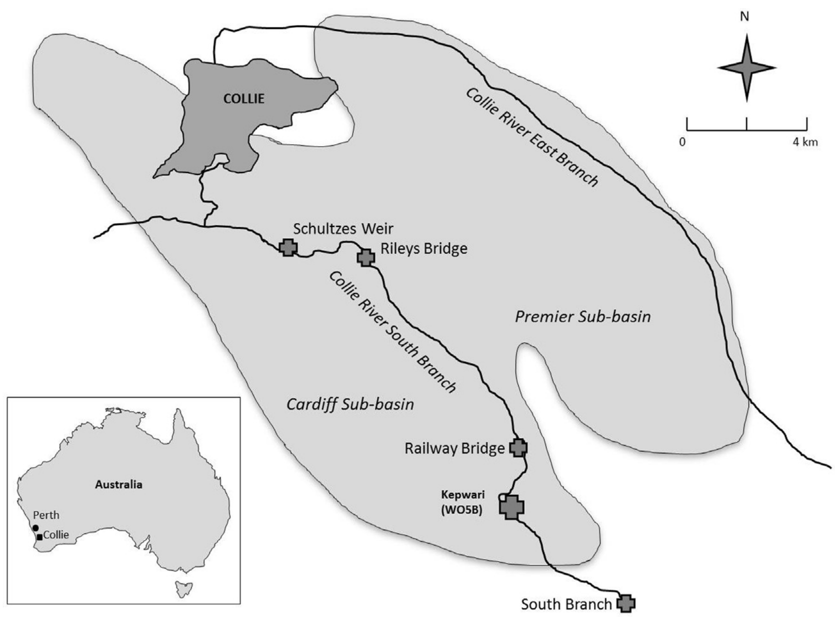

Fig. 1. Location of Lake Kepwari (large cross) in the Collie Coal Basin in south-western Australia. Water quality sampling sites above (South Branch) within (Lake Kepwari) and below (Railway and Rileys Bridges, and Schultzes Weir).

Athabasca oil sands region (CEMA, 2012), Western Australia (CEMA, 2012) and the Rhenish Mining District in Germany (Schultze et al., 2011a).

A pit lake may have positive effects on degraded rivers, such as nutrient removal, reduced suspended sediment loads and flood mitigation. Flow-through may also resolve many acidic pit lake water quality issues, such as low pH and elevated metal/metalloid concentrations. Connecting a pit lake to a river also increases the effective size of the pit lake catchment, allowing for greater inputs into the lake of nutrients, organic matter, plant propagules and biota (Lund et al., 2013).

Nonetheless, flow-through may create new liabilities associated with contamination of the river downstream with AMD from the pit lake. Risks include increased acidity, metal/metalloid, nitrate and ammonia concentrations, sediment retention, altered flow regimes, C and P nutrient and the lake acting as a barrier migrating biota. Many of these impacts are similar to those encountered after the installation of dams and weirs (McCartney et al., 2001).

This study describes river flow-through and decant of a pit lake during a flood event as an opportunity to evaluate river flow-through of an acid pit lake on pit lake and downstream river water quality. The key objectives for the current study were to determine if the AMD discharge reduced river end use values; and to ascertain how the inflow of river water altered water quality and limnology of the pit lake; in particular, if river inflow mitigated pit lake AMD. Together, the study sought to determine if river flow-through might form a long-term closure strategy for an acid pit lake.

2. Methods

2.1. Study area

The south west of Western Australia is regarded as highly biodiverse, with eight of the ten native freshwater fish found in the south-west endemic (Morgan et al., 1998). At least five species of native freshwater fish with limited distributions are also specifically found in the study area located in the Collie region itself (Whiting et al., 2000). However, the biodiversity hotspot tag comes at a price, as these areas are listed for having the most endemic species and being the most threatened areas in the world (Myers et al., 2000; Malcolm et al., 2006) especially the aquatic ecosystems found there (Horwitz et al., 2008).

Collie is situated in an area of Mediterranean climate, with hot, dry summers (range 12–29 °C) and cool, wet winters (range 4–15 °C). Seventy-five percent of the rainfall occurs during the five months from May to September. The 100 year mean annual rainfall for the Collie Basin is 939 mm, although this has decreased to an average of 690–840 mm over the past 20 years (Lund et al., 2012). The mining of the Lake Kepwari void in south-west Australia (WO5B, Fig. 1) began with diversion of the seasonal Collie River South Branch (CRSB) away from the pit site around the western margin and ceased in 1997. During rehabilitation, reactive overburden dumps and exposed coal seams were covered with waste rock, battered and topsoil replaced, and revegetated with native plants. To reduce wall exposure and acid production, the pit void was rapid-filled by a brackish first-flush diversion from the CRSB over three winters from 2003 to 2005 (Salmon et al., 2008). The river diversion pathway was maintained around the lake.

Although river water initially raised water pH to above pH 5, lake pH subsequently declined to below pH 4 by 2011 and displayed elevated solute concentrations as a result of acidity inputs, most likely though in-catchment and in-lake acidity generation, and acidic groundwater inflow (Müller et al., 2011).

The volume of the lake is now around 32×10^6 m^3, with a maximum depth of 65 m and surface area of 103 ha. Although Lake Kepwari was proposed as a recreation resource for primary water contact activities such as swimming and water skiing (Evans and Ashton, 2000), low pH and high metal concentrations currently restrict authorised use of the lake for these purposes (Neil et al., 2009).

2.2. River breach and decant

On the 24th August 2011, a rainfall of 85.6 mm in Collie over 48 h (BOM, 2012) led to rapidly rising high flows in the CRSB of a 1:8 year magnitude (DOW, 2013) (Fig. 2). The water level in the CRSB rose, overtopping and then eroding the engineered northern dyke wall that separated the CRSB diversion from Lake Kepwari. As a result, water levels in Lake Kepwari rose from 185.1 m by 1.7 m

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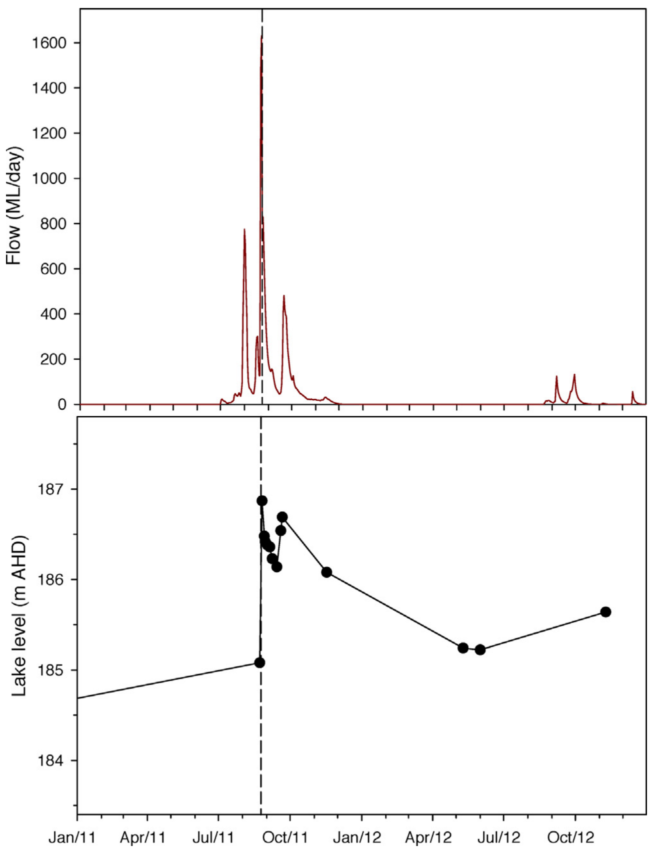

Fig. 2. CRSB daily flow rate at Schultzes Weir monitoring station (top) and Lake Kepwari water level in Australian height datum (AHD) for 2011–2012. Breach indicated by dashed vertical line.

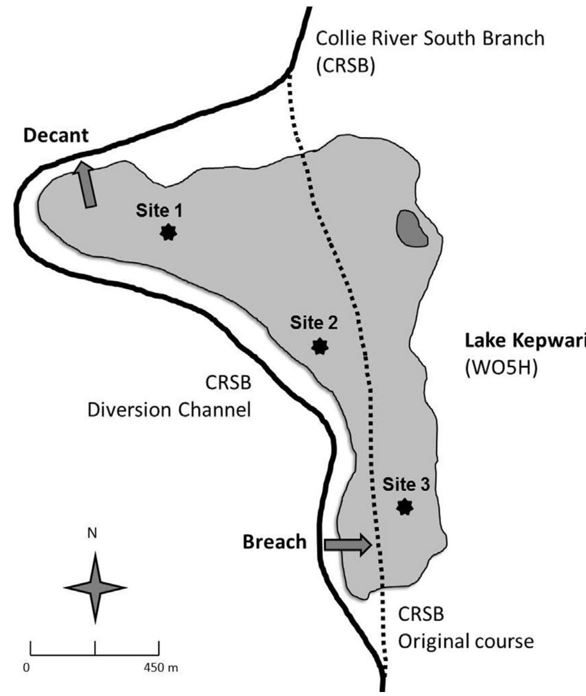

Fig. 3. Lake Kepwari showing lake sampling locations. Dotted line is original and solid line is diversion CRSB paths.

as indicated on the lake edge gauge board to 186.8 m (AHD) (Fig. 2) adding 3 ha to the surface area of 103 ha (McCullough et al., 2012) and increasing lake volume by around 6%. Lake water then decanted through a previously designed outlet before overtopping this and then decanting back through the breach as CRSB levels dropped. Over the following summer lake water levels fell back again before slightly rising the next winter.

There is now a permanent connection to the river and lake at the breach point and through a newly designed lake outflow (McCullough and Harkin, 2015). However, remediation works and low rainfall meant that no flow-through the lake occurred in 2012.

2.3. River water quality assessment

CRSB water quality was monitored above and below Lake Kepwari from 24 August 2011 (the day after breach) until the CRSB ceased flow on 16 October 2012. CRSB flow was not sufficient to discharge into Lake Kepwari in 2012 and only flowed in winter as per normal seasonal hydrology.

Sample sites included a reference point above the lake (South Branch) with pastoralism the only catchment influence, a site in the lake (site 1) near the lake decant and three sites on the CRSB to a distance around 10 km below the lake. At each CRSB monitoring site, samples were taken below the surface from a well-mixed section.

A multi-parameter metre (Hanna Instruments) was used to sample water quality at each CRSB site for pH and dissolved oxygen (DO, % saturation). At each site, a water sample was taken, filtered (0.5 μm GF/C Pall Metrigard USA) and acidified with reagent grade nitric acid (1% v/v to achieve a pH <2). After storage at 4 °C, analysis was by Inductively Coupled Plasma Mass Spectrophotometry (ICP-MS, Varian) for Al, Fe, Mn and Zn concentrations (as per APHA, 1998).

CRSB water quality results were compared to Australasian (ANZECC/ARMCANZ, 2000) livestock drinking, recreation and aesthetic and the 80% aquatic ecosystem protection guidelines (as the CRSB is mesotrophic-eutrophic and highly disturbed (Wetland Research & Management, 2009). pH and EC compared against guidelines for south-western Australian lowland river water quality (ANZECC/ARMCANZ, 2000).

2.4. Lake water quality assessment

Pre breach (2010–2011), lake water column physico-chemistry profiles were taken quarterly to account for seasonal variation with a Hydrolab Datasonde 4a multiparameter probe (Austin, USA) at the deepest point near the lake centre (Site 2, Fig. 3). Following the breach in August 2011, profiles were made at Sites 1–3. At each site, on each occasion, profiles of temperature, pH, specific conductance (EC) and oxidation-reduction potential (ORP, platinum electrode) and chlorophyll a after the recommendations of Gammons and Tucci (2011). TDS was estimated from EC by calculation through the equation TDS = EC (μS/cm) × 0.60.

Using a Teflon trace metal Kemmerer bottle (Wildco, USA), a water sample was collected on every sampling event from immediately below the lake surface waters as well as from ca.0.50 m above the sediment (Gammons and Tucci, 2011). Upon collection, half of the water sample was filtered through 0.5 μm GF/C (PAL Metrigard, USA). A filtered aliquot was frozen (−20 °C) and then later analysed for ammonia/ammonium (NH_3–N), nitrate–nitrite (NO_x–N), total filterable nitrogen (TFN), filterable reactive phosphate (FRPP), SO_4^{2-}, Cl^- and dissolved organic carbon. The remaining filtered samples were acidified with reagent grade nitric acid (1% v/v to ensure a pH of <2) immediately and stored at 4 °C until analysed for Al, As, Ca, Cd, Cr, Co, Cu, Fe, Hg, Mg, Mn, Na, Ni, Pb, Se, U and Zn by Inductively Coupled Plasma Optical Emission Spectrophotometry (ICP-OES). The unfiltered aliquot was persulfate digested and

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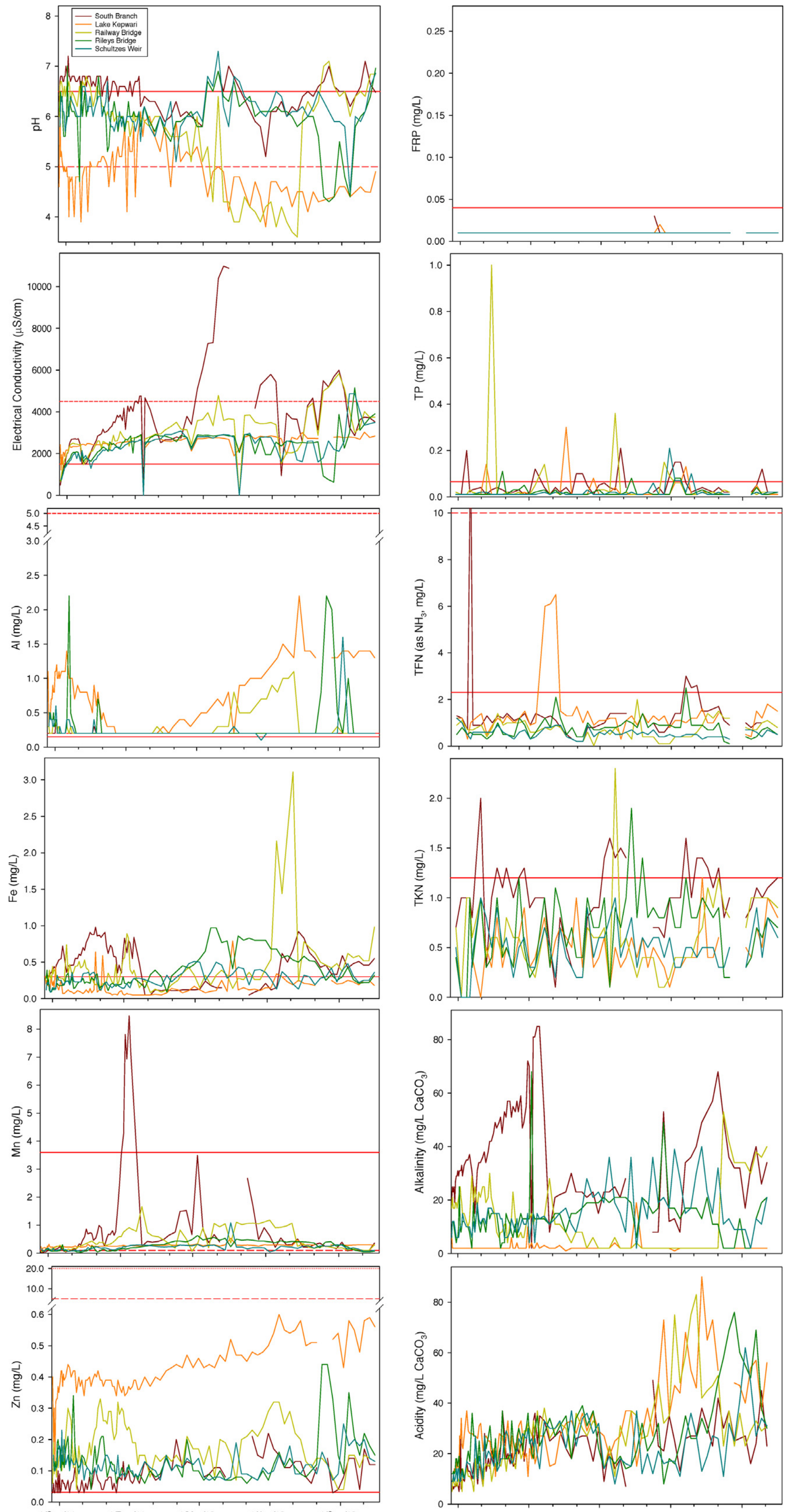

Fig. 4. Water quality in Lake Kepwari and CRSB between 23 August and 24 October 2011 for selected parameters. Note: line gaps show missing data and red lines indicate guideline levels: solid lines Ecosystem Protection, dashed lines Primary Recreation and dotted line Stock Drinking Guidelines (ANZECC/ARMCANZ, 2000). The legend lists sampling locations in succession, beginning at the most upstream reference site (South Branch), through to Lake Kepwari (site 3) and ending at the most downstream site below Lake Kepwari (Schultzes Weir) (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.).

then analysed for total P (TP) and total N (TKN). All water quality analyses followed standard methods (APHA, 1998).

Lake water quality results were compared to ANZECC/ARMCANZ (2000) livestock drinking water guidelines, recreational water quality and aesthetic guidelines, and 80% aquatic ecosystem protection guidelines (due to the lake being a constructed water body). Where 80% ecosystem protection guidelines were not available, water quality results were compared against guidelines for default south-western Australian lake and reservoir water quality (ANZECC/ARMCANZ, 2000).

2.5. Data analyses

Univariate analysis were made by developing time-series plots for CRSB and Lake Kepwari water quality data in the SigmaPlot graphing package (Systat Software Inc., 2011). Principal component analysis (PCA) ordination was undertaken in Primer multivariate statistics software (PRIMER-E Ltd, 2006). Vectors were then plotted over the centroid of each lake water quality sample (halfway between surface and bottom water quality points) to separate water chemistry into major phases of change.

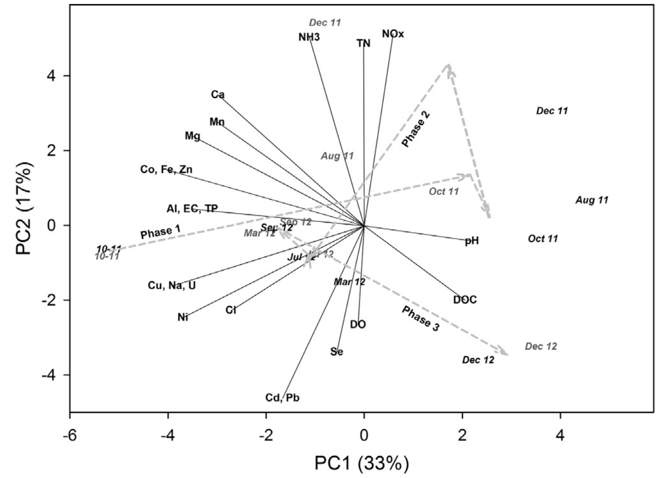

Fig. 5. PCA analysis of mean surface and depth water quality of Lake Kepwari from 2010 (mean June, September and December 2010 and March 2011 water quality), to December 2012. Black font is bottom and grey font is surface water samples. Only eigenvectors >[0.14] shown. Light grey vectors indicate movement of mean water quality sample centroids over time.

3. Results

3.1. River water chemistry

The breach and subsequent decant of lake water from Lake Kepwari did not present any significantly increased risk to stock drinking water in the CRSB below the breach point (Fig. 4). Water quality at all sites sampled exceeded ANZECC/ARMCANZ (2000) regional trigger values for pH, DO; and 80% ecosystem protection guidelines for Fe and Zn, and recreational water quality and aesthetic guidelines for pH, DO, Fe, and Mn.

Lake Kepwari (Site 3) exceeded pH 5.0 guideline levels for primary contact recreation at all times except for two sampling events shortly after breach. pH values at all other river sites were also below the lower ecosystem protection guideline of 6.5 for most of the sampling period apart from the South Branch upstream reference site which rose above this guideline around two weeks after breach. Dissolved oxygen (DO) was below recreation and ecosystem protection guidelines in all sites for much of the monitoring period, although DO was generally high in Lake Kepwari.

Aluminium concentrations were below recreation and stock drinking water guidelines at all sites and times. Despite this Al concentrations were significantly higher in Lake Kepwari than the other sites, with elevated concentrations downstream of the lake in apparent response as well. Unusually high Al concentration at Rileys Bridge in mid-September 2011 and mid-August 2012 indicates other anthropogenic sources of Al into the river below the Lake Kepwari, most likely from other regional mine water discharges e.g., from mine dewatering.

Fe concentrations exceeded recreational and aesthetic water quality and interim ecosystem protection guidelines of 0.3 mg/L at most sites during the sampling period. A spike in Fe concentration occurred at the South Branch in late June 2012 when no flow-through was occurring. Concentrations were highest above the lake at the reference site indicating high background levels and suggesting that the pit lake even reduced Fe at downstream sites.

All sites exceeded recreational and aesthetic guidelines for Mn, with highest concentrations in South Branch indicating Mn concentrations were already elevated in the CRSB. Indeed, Mn concentrations only exceeded 80% ecosystem protection guideline of 3.6 mg/L at a peak of 8.47 mg/L in mid-December 2011.

Zinc concentrations exceeded 80% ecosystem protection guidelines of 0.031 mg/L at all sites. Zn concentrations were highest in Lake Kepwari at around 0.4 mg/L, this doubled South Branch concentrations of around 0.08 mg/L to around 0.15 mg/L downstream of the lake. Another spike in Zn concentration at Rileys Bridge again indicated another mine water source of Zn above this site. However, Zn concentrations did not exceed recreational and aesthetic guideline of 5 mg/L or livestock drinking guidelines of 20 mg/L.

FRP was below aquatic ecosystem protection guideline concentrations at all times and almost always below limits of detection (LOD). TP regularly exceeded aquatic ecosystem protection guideline concentrations for all sites, but infrequently for Lake Kepwari and more frequently for both Southern Branch upstream and Railway Bridge (immediately downstream). TFN did not exceed any guidelines except for over a two-week period in Lake Kepwari and in the upstream South Branch and Rileys Bridge sites at the end of flow in October. TKN concentrations only exceeded guidelines for aquatic ecosystem protection for South Branch upstream and for Railway Bridge and Riley's Bridge sites downstream of Lake Kepwari.

3.2. Lake water quality and limnology

The overall trend for water chemistry pre-to post breach 16 months later was lake water quality improvement increases in pH and DOC and decreases in Al, Ca and N concentrations. Monitoring before and after river flow-through identified three major trajectories of water chemistry change (Fig. 5).

Phase 1: **Pre-breach (March 2010–December 2011).** Surface and bottom lake water chemistries that were near identical become markedly different following flow-through.

2010–2011 differed from post-breach October 2011 lake water quality predominantly due to decreased concentrations of TP, most metals, EC and increased pH. Total N and N fraction concentrations then increased as DO and DOC concentrations increased over the next few months, presumably due to decomposition of an organic load brought in by the river.

Phase 2: **Post-breach (December 2011–September 2012).** Surface and bottom lake water chemistry markedly different.

Following cessation of CRSB flow into the lake, from December 2011 to final turn-over in mid-winter July 2012, pH and concentrations of DOC, and total N and N fractions then all decreased as EC and concentrations of most metals and TP all increased over summer 2011–2012.

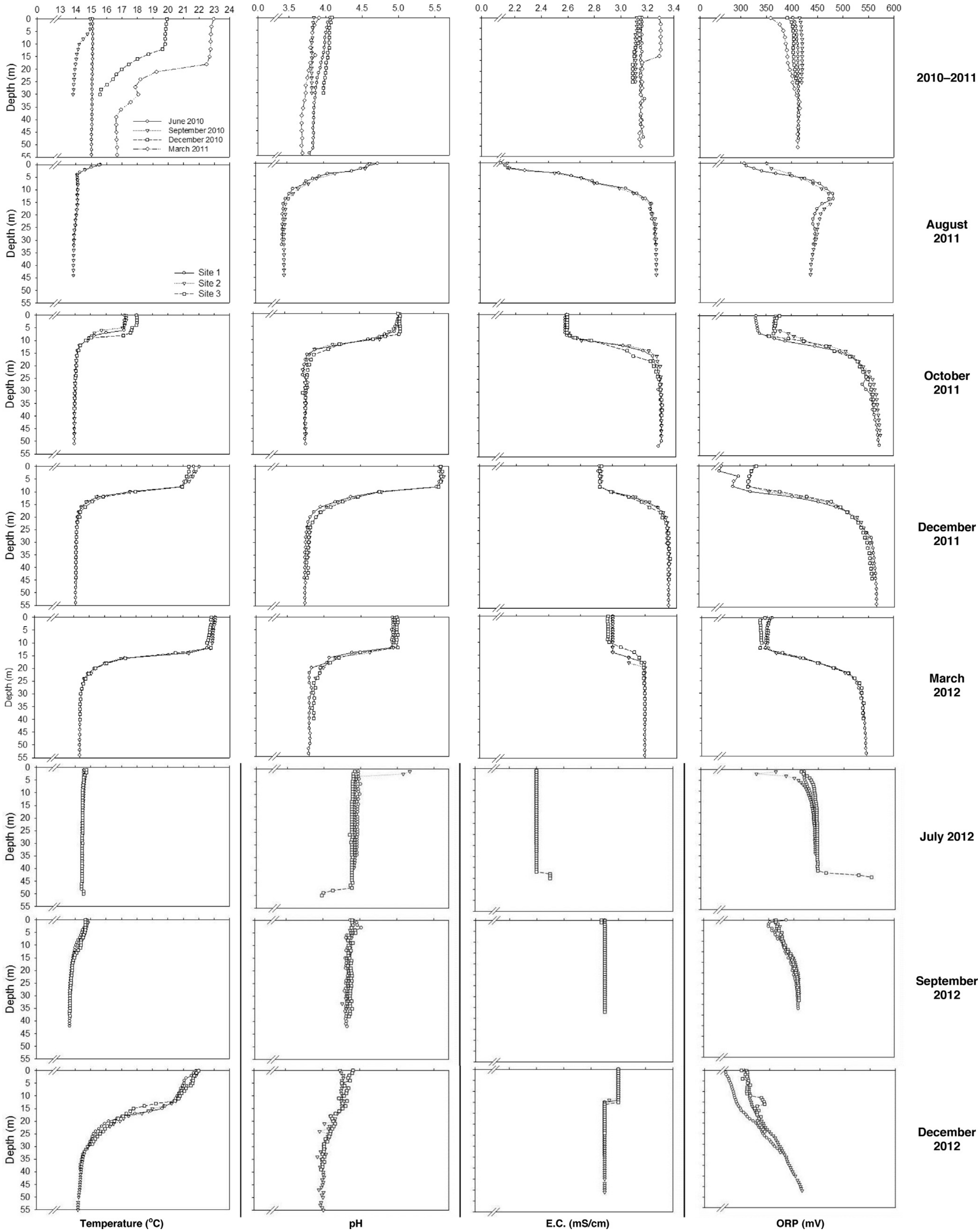

Fig. 6. Water column physico-chemistry profiles across depth in Lake Kepwari before and after the 23 August 2011 CRSB breach and flow-through.

Phase 3: **Recovery (September 2012–December 2012)**. Surface and bottom lake water chemistries very similar again.

In the final three months, pH and DOC increased again, EC and some metals and N concentrations all decreased.

Lake water column physico-chemistry profiles for 2010–2011 (pre-breach) and 2011–2012 (post breach) are shown in Fig. 6.

Prior to breach during June 2010 to March 2011, Lake Kepwari showed complete mixing during winter months with thermal stratification presenting as a 15–18 m deep epilimnion during summer months. Lake Kepwari surface water temperature evenly increased in the following seven months following the breach from 16 °C to 23 °C, likely as a result of fine weather and solar warming. Complete overturn then occurred in winter June, however, thermal stratification did not re-establish until summer in December likely as a result of atypically cooler temperatures in early summer 2012 (BOM, 2012).

CRSB flow is seasonal (dry in summer) with salinity from upper catchment secondary salinisation varying from fresh, to brackish at the start and end of winter flows before dilution by rain (Wetland Research & Management, 2009). However, the flood event primarily brought fresh water into the lake resulting as a marked halocline. This halocline became more sharply defined and deepened to 10–15 m chemocline depth over the following months, although the difference across the temporary chemocline waters decreased over time as bottom water salinity increased from 2.1 to 2.5 mS/cm. Presumably this deepening was due to evapo-concentration of surface waters and increasing salinities in the river inputs and also possibly due to partial mixing between surface and bottom waters as the temporary chemocline deepened.

Significant lake improvements in pH were observed following lake breach. During June 2010 to March 2011, Lake Kepwari pH ranged from 3.7 to 4.1 and exceeded recreational water quality guidelines of pH 5.0–9.0 (when weakly buffered) (NHMRC, 2008) and 90% ecosystem protection guidelines of pH 6.5–8.5 (ANZECC/ARMCANZ, 2000). pH was consistent between epilimnion and hypolimnion during this time. Following the breach, bottom water pH increased, ranging from 4.6 to 5.6, peaking in December 2011. However, by December 2012 with no flow-through that year, lake pH had returned to pre-breach levels of pH 4.

From June 2010 to December 2012, ORP ranged from 250 to 570 mV, peaking in December 2011 (bottom waters) and lowest immediately following breach and in December 2012 (surface waters). Following the breach, surface water ORP showed a decrease to 300–350 mV but an increase to 450–480 mV in the bottom waters. There was a further increase in the hypolimnion ORP in October to around 570 mV.

It is unclear what caused this ORP increase, however, it was also seen, albeit to a reduced extent in bottom waters where ORP increased from mean 330 mV to mean 350 mV. Different to the other profiles, ORP varied between sites with Site 1 (closest to the breach) displaying ORP values around 50 mV lower than the other surface water sites.

Higher ORP was likely due to inputs of acid groundwater when the lake became an evaporative sink during this dry season (McCullough et al., 2013b). Low ORP following the winter rains was likely due to lower acidity surface runoff water accumulating and even slightly surcharging Lake Kepwari during the wet season (Müller et al., 2011). This may indicate low dissolved metal concentrations and higher pH in the incoming river waters prior to mixing with higher ORP lake waters.

pH exceeded guideline levels for recreation and ecosystem protection for all bottom water samples at all sampling times (Table 1). However, although bottom water samples were always below ecosystem protection guidelines, three surface water samples from October 2011 to March 2012 were above the recreational guideline of 5.0.

For a short period following summer, dissolved oxygen (DO) in bottom waters were below ecosystem protection guidelines of 80% saturation, however, surface water always meet these requirements. Chlorophyll a was below ecosystem protection guidelines at all. At only a few mg/L dissolved organic carbon (DOC) was always low concentrations; although it increased in surface waters slightly following river breach and then decreased again. Because of the high salinity resulting from original fill waters remaining, electrical conductivity (EC) always exceeded environmental guidelines of 1.5 mS/cm for surface and bottom waters. Sulfate concentrations did not exceed recreational guidelines of 400 mg/L in any samples; whilst Cl exceeded recreational guidelines of 300 mg/L in all samples.

Total nitrogen and its fractions of NH_3 and NO*x* exceeded ecosystem protection guidelines before and after breach. Concentrations of many metals were below aquatic ecosystem protection guidelines. Exceptions were Al, Cd, Cu, Hg, Ni, Pb, U and Zn; notably the lake water's saline matrix reduced detection limits often to above guideline values. Na concentrations exceeded recreational guidelines in all pre-breach and 2012 samples.

Stock water quality guidelines were exceeded in the lake for NO*x* at all times, Pb after March 2012 and Se in December 2012. U exceeded stock drinking guidelines before but not after breach. Hg may have exceeded stock watering guidelines on more than one occasion prior to and following breach. However, laboratory limits of detection were too high to determine whether concentrations were less than the guidelines.

All sites and times exceeded TDS, Cl, NH_3, Al, Mn, Ni guidelines for recreation. Fe only exceed recreation guidelines prior to breach and Na exceeded guidelines prior to breach during no flow in 2012 before December. Hg may have exceeded recreation guidelines on more than one occasion prior to and following breach.

4. Discussion

Lake TKN concentrations may have reduced in response to algal assimilation during an observed high algal biomass period in late December 2011 whilst TP may have maintained more original river concentrations due to reduced sorption of P in incoming River water as a result of decreased surface water concentrations of Al and Fe (Beulker et al., 2003; Lessmann et al., 2003) following complete flushing of lake surface waters by March 2012. The generally high concentration of nitrogen in Lake Kepwari was likely a result of ANFO (ammonium nitrate, fuel oil) blasting residue (Frandsen et al., 2009) and fertiliser from river rapid-fill and groundwater contribution to its filling.

Low chlorophyll a concentrations are typically low algal biomass in acidic pit lakes (Tittel and Kamjunke, 2004). Similarly, low DOC concentration are likely due to carbon mineralisation (Gennings et al., 2001) and (to a lesser extent) metabolisation (Znachor and Nedoma, 2009).

DO decreasing down through the CRSB likely due to biochemical oxygen demand by decaying organic matter from flood-mobilised allochthonous vegetation (Vannote et al., 1980). Elevated EC over summer months commensurate with declining lake water level indicate that groundwater inflows may have been unable to maintain lake water level during this time as had been found elsewhere (Santofimia and López-Pamo, 2013) either due to naturally declining groundwater level over summer or low rates of hydraulic conductance into the lake.

Al, Cu, Fe and Ni are naturally elevated in groundwater surrounding Lake Kepwari, regularly exceeding water quality guideline levels in the region's groundwater as does NO_3 and NH_3 (GHD, 2010). Calcium loss from freshwater systems has been described as a global phenomenon resulting from anthropogenic

Table 1
Lake Kepwari 2010–2012 mean water quality monitoring data against Australasian (ANZECC/ARMCANZ, 2000) water quality guidelines for 80% aquatic ecosystem protection and south-west lakes and reservoirs default conditions. Font style indicates exceedance for one or more guideline. All units are mg/L unless otherwise indicated.

	Livestock Drinking (Low Risk)	Recreational Water Quality & Aesthetics	Aquatic Ecosystem Protection	2010–2011		Aug 2011		Oct 2011		Dec 2011		Mar 2012		Jul 2012		Sep 2012		Dec 2012	
				Top	Bottom	Top	Bottom	Top	Bottom	Top	Bottom	Top	Bottom	Top	Bottom	Top	Bottom	Top	Bottom
pH (pH units)[Ω]	ID	*5.0–9.0*	**6.5–8.0**	***4.0***	***3.8***	***4.7***	***3.4***	**5.0**	***3.8***	**5.6**	***3.8***	**5.0**	***3.9***	***4.5***	***4.5***	***4.4***	***4.3***	***4.3***	***4.0***
DO (%)	ID	*>80*	**>90**	105	107	93	93	106	96	91	***76***	**80**	***72***	96	98	100	101	100	98
EC (mS/cm)	4.5	*1.5*	**0.30-1.50**	***3.13***	***3.08***	***2.1***	***3.1***	***2.5***	***3.1***	***2.7***	***3.2***	***2.8***	***3.0***	***2.9***	***2.9***	***2.9***	***2.9***	***3.0***	***2.9***
TDS	(Poultry) 2000	*1000*	ID	*1878*	*1848*	*1287*	*1848*	*1512*	*1866*	*1636*	*1892*	*1680*	*1800*	*1740*	*1740*	*1740*	*1740*	*1800*	*1740*
Chla	ID	ID	**0.003-0.005**[a]	**<0.0001**	**<0.0001**	**0.0008**	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**0.0002**	**0.0003**	**<0.0001**	**0.0004**	**<0.0001**	**0.0002**
DOC (0.5)	ID	ID	ID	1.2	1.6	2.9	0.30	3.20	1.90	3.5	0.53	3.7	2.9	1.5	1.4	1.1	1.1	2.5	1.4
SO_4^{2-} (0.5)	1000	*400*	ID	182	171	83	110	86	112	92	127	87	100	91	86	55	53	114	117
Cl^- (0.5)	ID	*400*	ID	*831*	*785*	*693*	*800*	*693*	*813*	*693*	*793*	*781*	*835*	*822*	*813*	*819*	*802*	*835*	*821*
NO_3 as N (0.002)	0.03	*1*	**17**[a]	0.83	0.78	0.66	0.92	0.69	0.83	*3.2*	*3.7*	0.59	0.73	0.58	0.55	0.85	0.87	0.74	0.71
NH_3 as N (0.003)	ID	*0.01*	**0.01**[a]	***0.04***	***0.04***	***0.03***	***0.05***	***0.02***	***0.05***	***0.07***	***0.15***	***0.04***	***0.05***	***0.06***	***0.05***	***0.07***	***0.07***	***0.03***	***0.02***
TN[g] (0.05)	ID	ID	**0.35**[a]	**1.53**	**1.36**	**1.15**	**1.1**	**0.76**	**0.87**	**3.4**	**3.9**	**0.99**	**1.04**	**0.83**	**0.69**	**0.93**	**0.96**	**0.91**	**0.89**
FRP (0.002)	ID	ID	**0.005**[a]	0.005	0.004	<0.002	<0.002	0.003	0.003	0.004	<0.002	**0.006**	0.006	<0.002	<0.002	0.003	<0.002	0.003	<0.002
TP[*] (0.02)	ID	ID	**0.01**[a]	**0.025**	**0.025**	<0.01	<0.01	0.01	0.01	**0.02**	**0.03**	**<0.02**	**<0.02**	**<0.02**	**<0.02**	**<0.02**	**<0.02**	**<0.02**	**<0.02**
Al[b] (0.1)	5	*0.2*	**0.15**[f] (pH >6.5)	***1.75***	***1.66***	***0.63***	***1.4***	0.03	0.04	*0.14*	***1.8***	***0.39***	***1.2***	***1.3***	***1.3***	***1.3***	***1.3***	***0.80***	***0.67***
As (0.01)	5 (as As V)	*0.05*	**0.36**	<0.01	<0.01	<0.01	<0.01	0.007	**1.3**	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.02	0.02	<0.01	<0.01
Ca (0.05)	ID	ID	ID	35	35	20	28	25	29	24	31	24	25	28	29	30	30	11	9
Cd[*] (0.002)	0.01	*0.005*	**0.0008**	**<0.002**	**<0.002**	**0.0004**	**0.0008**	**<0.002**	**<0.002**	**0.0004**	**0.0006**	**0.003**	**<0.002**	**<0.002**	**<0.002**	**<0.002**	**<0.002**	**<0.002**	**<0.002**
Cr (0.005)	1 (as Cr VI)	*0.05*	**0.04**[c]	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Co (0.01)	1	ID	ID	0.09	0.09	0.04	0.07	0.04	0.05	0.034	0.057	0.05	0.07	0.06	0.06	0.06	0.06	0.02	0.01
Cu[*] (0.02)	0.4	*1*	**0.0025**	**0.054**	**0.060**	**0.003**	**0.003**	**0.005**	**0.004**	**0.004**	**0.010**	**<0.02**	**<0.02**	**<0.02**	**<0.02**	**<0.02**	**<0.02**	**0.012**	**0.007**
Fe (0.05)	Not sufficiently toxic	*0.3*	**0.3**	***0.46***	***0.50***	0.15	***0.43***	0.08	0.23	0.03	0.24	0.08	0.2	0.18	0.17	0.17	0.16	0.07	0.05
Hg (0.05)	0.002	*0.001*	**0.0054**	***<0.05***	***<0.05***	***<0.05***	***<0.05***	***<0.05***	***<0.05***	***<0.05***	***<0.05***	***<0.05***	***<0.05***	***<0.05***	***<0.05***	***0.11***	***0.09***	***<0.05***	***<0.05***
Mg (0.1)	2000[d]	ID	**2.5**[h]	**104**	**103**	**52**	**71**	**57**	**77**	**62**	**79**	**97**	**102**	**75**	**76**	**84**	**84**	**33**	**27**
Mn (0.01)	Not sufficiently toxic	*0.1*	**3.6**	*0.31*	*0.30*	*0.21*	*0.31*	*0.27*	*0.26*	*0.23*	*0.27*	*0.30*	*0.31*	*0.28*	*0.28*	*0.31*	*0.31*	*0.11*	0.08
Na (0.5)	ID	*300*	ID	**451**	**451**	143	200	142	99	172	146	**311**	**327**	**354**	**359**	**375**	**374**	172	146
Ni (0.02)	1	*0.1*	**0.017**	**0.10**	***0.12***	**0.04**	**0.07**	**0.04**	**0.04**	**0.03**	**0.06**	**0.08**	**0.10**	***0.11***	**0.07**	**0.07**	**0.06**	**0.07**	**0.07**
Pb[*] (0.1)	0.1	*0.05*	**0.0094**	***<0.1***	***<0.1***	0.003	**0.012**	0.005	0.008	0.002	**0.01**	***0.13***	***<0.10***	***<0.10***	***<0.10***	***<0.10***	***<0.10***	***0.12***	***0.13***
Se[*] (0.002)	0.02	*0.01*	**0.011**	***0.32***	***0.32***	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	***0.920***	***0.117***
U[e] (0.001)	0.2	ID	**0.006**	**0.21**	**0.22**	<0.001	<0.001	<0.001	<0.001	**<0.01**	**<0.01**	**0.079**	**0.065**	**0.096**	**0.104**	**0.131**	**0.156**	**0.023**	**<0.01**
Zn[*] (0.05)	20	*5*	**0.031**	**0.72**	**0.73**	**0.345**	**0.46**	**0.39**	**0.50**	**0.31**	**0.52**	**0.469**	**0.602**	**0.435**	**0.489**	**0.583**	**0.598**	**0.297**	**0.137**

Notes:
ID Guideline value indeterminate (ANZECC/ARMCANZ, 2000).
[a] NO_3 default trigger value for s-w Australia freshwater lakes and reservoirs.
[b] May be tolerated if not provided as a food additive and natural levels in the diet are low.
[c] Figure may not protect key test species from acute (and chronic) toxicity. 'A' indicates that trigger value > acute toxicity value; note that trigger value should be <1/3 of acute figure (ANZECC/ARMCANZ, 2000).
[d] Drinking water containing magnesium at concentrations up to 2000 mg/L has been found to have no adverse effects on cattle (ANZECC/ARMCANZ, 2000).
[e] 95% protection. Hogan et al. (2005).
[f] May also be toxic at low concentrations at pH values less than 6.5. From Neil et al. (2009).
[g] Represents turbid lakes only; clear lakes typically have much lower values (ANZECC/ARMCANZ, 2000).
[h] Low calcium concentration waters and 99% protection only. Van Dam et al. (2010).
[*] Elevated detection limit due to high TDS concentrations in samples.
[Ω] Low acidity buffered waters.

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activities (Jeziorski et al., 2008). Although metal concentrations were reduced overall river flow-through, there also appeared to be significant loss of calcium from the lake. However, the increased risk of metal toxicity at lower calcium concentrations (Markich and Jeffree, 1994) is likely offset by lower metal concentrations overall (Neil et al., 2009).

Elevated ORP at sites closest to the breach likely reflects river water input diluting and neutralising acid lake water (Schultze et al., 2002). It is likely that metal precipitation and dissolved organic carbon chelation, and neutralisation with higher pH river water would have decreased acidity and increased pH during flow-through (Schultze et al., 2002). Given the relatively high volumes of water in-flowing into the Lake following the breach, significant dilution may have also contributed to surface water pH increase. The further pH peak in December may be attributed in a small part to nitrate assimilation of the lake's mixotrophic algae (Kumar et al., 2011) producing alkalinity (Davison et al., 1995). However, the low chlorophyll concentrations in the lake at this time indicate this contribution to water quality remediation would not be significant. Increased algal biomass would largely be a response to elevated DOC and P concentrations following by CRSB inflow (Beulker et al., 2003; Tittel and Kamjunke, 2004). Indeed, phytoplankton appear to be more restricted by these macronutrient concentrations than by mineral acidity (Neil et al., 2009; McCullough and Horwitz, 2010).

Mining lease conditions often specify the use of ecosystem protection guidelines for regulating discharge water quality limits (Jones and McCullough, 2011). Although upstream Zn and Al concentrations were already elevated in the Collie River, downstream sites showed Al, Mn and Zn concentrations likely elevated by the lake Kepwari breach and decant. Although there are no 80% ecosystem protection guidelines for Al where pH <6.5, Al concentrations at the majority of sites were potentially toxic, due to increased monovalent Al concentrations at these lower pH values (Neil et al., 2009). However, the regular exceedance of water quality guidelines also by the reference South Branch site indicates that Collie River background concentrations were already elevated for many of these water quality parameters. Although there are no known mine influences above this point, catchment activities such as through forest clearance and farming have degraded water quality through salinisation (Tingey and Sparks, 2006) and eutrophication (Wetland Research & Management, 2009).

Surface and ground waters of catchments comprising mining resources also often show elevated solute concentrations in baseline conditions due to their unique geologies. Consequently, the geology of the Collie Coal Basin is likely to have influenced water quality complicating the interpretation of reference conditions for the CRSB (Castendyk, 2009; McCullough and Pearce, 2014). From a management perspective, the Collie River catchment water quality is likely regionally unique and locally-derived water quality guidelines should therefore be developed for the site based upon reference data (McCullough et al., 2013a).

There appears to be little risk of P remobilisation and eutrophication as Fe and Al concentrations remain elevated. However, a decrease of these metals over time and increasing pH may increase water column concentrations of P (Kleeberg and Grüneberg, 2005).

The current closure plan to excise Lake Kepwari from its broader catchment may represent significant residual risk remaining even following careful closure planning as a result of the poor lake water quality and large volume. An alternative flow-through may present a more sustainable closure alternative by reducing risk to environmental and social end uses of the lake and surrounding groundwater and water bodies such as the CRSB downstream. Fresher and moderately alkaline lake surface water (essentially CRSB water) from riverine flow-through is more likely to enable the lake to meet regulatory requirements for recreational use. Further, there is potential for the water quality of CRSB water, particularly for AMD removal of excess nutrients such as P (Kleeberg and Grüneberg, 2005), to be enhanced by passage through the pit lake as has been observed with metals removal in pit lake flow-through situations (Klemm et al., 2005).

5. Conclusions

Meromixis has been used as a strategy to constrain AMD to lower pit lake depths away from environmental receptors which typically inhabit shallower lake levels e.g., Fisher and Lawrence (2006). However, meromixis persistence is not likely given Lake Kepwari decreased in salinity from 3.2 mS/cm prior to breach to 2.9 mS/cm by December 2012, and with the likelihood of saline water influx in seasonal river flow freshette. Stratification may, however, temporarily reform over winter and spring for some years once river flows become fresh. Use of a diversion weir in the breach point or river could further be useful in enhancing a halocline or preventing it through select hydrograph diversions, if desired.

Using seasonal rivers presents challenges in maintenance of desired haloclines during no flow periods and as in the case of the CRSB where salinity (due to secondary salinisation) varies in the river across the flow season. We expect lake water quality will improve and be maintained at a similar quality to the CRSB through successive flow-through events should environmental conditions persist. Catchment discharge studies are now underway to examine both with what frequency such flow-through will occur under historic rainfall patterns as well as predicted climate change impacts (Hobday and Lough, 2011).

The possibility for mine lake contaminants to be transported to lotic environmental receptors and the potential bioaccumulation of elements such as heavy metals and metalloids should also be considered in flow-through of pit lakes (Miller et al., 2013). River flow-through remediated lake water pH above recreational guideline pH values in the upper 10 m waters in which people would swim. Lake water quality studies for elevated metal and metalloid concentrations against swimming guidelines for chronic health outcomes are also still required (Hinwood et al., 2012).

Although the initial decant of acidic waters to the CRSB below the lake afforded little dilution by the river, flow-through, via a controlled CRSB diversion to Lake Kepwari is being trialled to mitigate or even completely avoid this by only decanting a controlled flow to allow for dilution of lake AMD water by the CRSB. This management is discussed in detail in McCullough and Harkin (2015).

Like all mine closure planning practice, determination of final closure strategies should be made on a risk-based approach with explicit consideration of the complex system being managed though an adaptive management framework (Allen et al., 2011). A flow-through closure strategy for pit lakes should always first seek to maintain any existing river values such as aquatic ecosystems, swimming and livestock watering. Only then should it seek to improve lake water quality to achieve values such as recreational or wildlife end uses.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecoleng.2015.10.001.

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Engineered flow-through closure of an acid pit lake: a case study

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Abstract

The Lake Kepwari flow-through trial has monitored both lake and river ecology and water quality for two years flow-through, plus one year post-breach monitoring (2012). The flow-through trial involved both the development of a management plan and application of the appropriate licensing framework through the Department of Water state regulatory agency via an operating strategy document. The aim of this document was to establish the suitability of flow-through as a closure strategy.

The key conclusion of the three-year flow-through trial is that that social and environmental risk and benefit to the Collie River South Branch (CRSB) and Lake Kepwari are best managed using flow-through as a closure strategy.

Lake water quality has been significantly improved through nutrient addition and acidity neutralisation. Improved water quality, particularly in terms of elevated pH and reduced metal/metalloid concentrations, has now significantly reduced the socio-environmental risk of closure of this landform.

This water quality improvement has also led to improvements in aquatic biota biodiversity and abundance, and changed biota assemblages to those of a more functional lake more typical of a freshwater ecosystem. This improvement also better enables the lake to meet the identified end use values of recreation/aesthetics and livestock drinking than would the no flow-through strategy. River flow-through in particular remediates pH above recreational guideline pH values in the upper 10 m epilimnion waters in which people would swim.

The objectives of the flow-through trial were first to maintain key CRSB values for aquatic ecosystems, swimming and livestock watering, and then to provide for lake water quality improvements to meet recreational criteria and to improve wildlife values. However, pit lake risk reduction and lake end use value improvement do not appear to have occurred at the expense of CRSB water quality and aquatic ecology, which have been maintained. Flow-through appears to have significantly increased both total volume in the lower CRSB and flow peak events, and may further assist CRSB social and environmental values. Additionally, the pit lake appears to be fulfilling a theoretically predicted function of reducing CRSB nutrient loads, which are elevated as a result of farming land use practices within the shared catchment.

In line with adaptive management principles, water quality assessment of Lake Kepwari and the CRSB (inlet and outlet) is continuing until the lake is relinquished back to the state. Owing to the low apparent risk, monitoring is now undertaken through a less intensive operational monitoring strategy rather than the previous more intensive trial program.

1 Introduction

Mine pit lakes may form at mine closure when voids formed through mining extractions have extended below groundwater (Castro and Moore, 2000). Internationally and within Australia (Kumar et al., 2013), acid and metalliferous drainage (AMD) is a particularly common problem for coal pit lake water quality (Vandenberg et al., 2015). Even if it is not acidic, pit lake water quality may become degraded gradually through dissolution of contaminants and evapoconcentration (McCullough, Marchand et al., 2013; Castendyk, Balistrieri et al., 2014). There are few long-term strategies to remediate AMD impacted pit lake water quality (McCullough, 2008).

Pit lake waters may discharge into surface and groundwater or present direct risks to wildlife, stock and human end users. As a result, contaminated pit lake waters can present significant risk to both surrounding and regional communities and natural environments (McCullough and Lund, 2006).

We propose that riverine lake flow-through may often be a valid mine closure strategy in order to mitigate pit lake water contamination in pit lakes with poor water quality. Chemical and biological processes such as dilution, absorption, flocculation and sedimentation reduce solute loads from river and lake.

1.1 Site overview

The southwest part of Western Australia is regarded as highly biodiverse for aquatic ecology (Horwitz et al., 2001). The town of Collie is approximately 160 km south southeast of Perth, and is the centre of the coal mining industry in Western Australia (Figure 1). The majority land uses in the catchment are coal mining, timber production, power generation and agriculture. Approximately 79% of the catchment is state forest. The recreation and nature conservation values of the forest areas are highly regarded, along with the recreational opportunities provided by the Wellington Reservoir and other surface waters, including some pit lakes. These values have led to increased promotion of the area for wildlife and recreation-based tourism by the local business associations and shire.

Underground and open cut coal mining has taken place in the Collie Basin since 1898 (Stedman, 1988) (Figure 7). More than 100 years of coal mining in Collie have resulted in the formation of at least 13 pit lakes that range from <1–10 ha in surface area, <10–70 m in depth, 1–50 years in age and 2.4–5.5 pH (Lund et al., 2012). These lakes all differ in the extent of rehabilitation and water quality, although all are acidic owing to AMD (McCullough et al., 2010).

Lake Kepwari (WO5B) is situated in the Cardiff sub-basin. Mining began in pit WO5B with diversion of the Collie River South Branch (CRSB) around the western lake margin and ceased in 1997. The lake was planned as a regional contact aquatic recreation resource (Evans and Ashton, 2000; Evans et al., 2003). Prior to rapid filling with water, reactive overburden dumps and exposed coal seams were covered with waste rock, battered and revegetated with endemic flora by direct seeding (Figure 2). To further reduce wall exposure and rates of resulting acid production, the lake was rapid filled with a seasonal diversion from the CRSB over the winters from 1999 to 2008, omitting 2001, a low-flow year. The volume of Lake Kepwari is around 32 Gl, with a maximum depth of 65 m and surface area of 1.03 km² (McCullough et al., 2010).

Although CRSB water initially raised water pH to above pH 5 (Salmon et al., 2008), pH then fell below pH 4 and displayed elevated concentrations for some metals and metalloids as a result of acidic groundwater inflow (McCullough et al., 2011). Although the reasonably good water quality of the pit lake potentially lends itself to a range of end uses, low pH and high Al concentrations remain a challenge to remediate (Lund and McCullough, 2009; Neil et al., 2009).

1.2 Background

During the third week of August 2011, 85.6 mm of rainfall in Collie within 48 h led to high flows in the CRSB. The water level in the CRSB rose, which led it to overtop and cause failure of the dyke along the south branch wall that separates it from Lake Kepwari (PMS, 2012). Approximately 30 m of wall failed, with another section of wall also damaged. The water level in Lake Kepwari increased in height by 1.7 m over a surface area of 106 ha with a flood return period of only around a seven-year average recurrence interval (ARI) event. Around 2 GL of CRSB water then flowed through Lake Kepwari, decanting through both culverts in the northeast and northwest sides of the Lake (Figure 3).

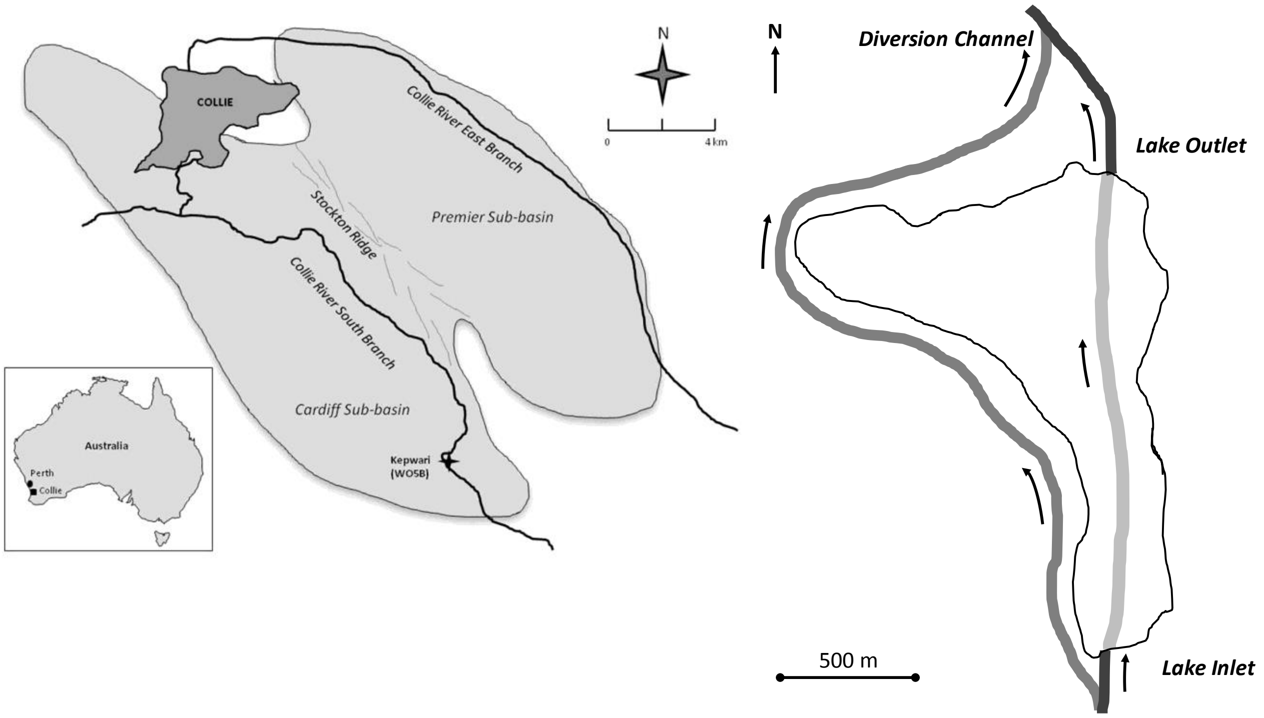

Figure 1 **Location of Lake Kepwari in Collie, Western Australia**

Figure 2 **Lake Kepwari foreshore showing establishing native riparian vegetation**

Figure 3 **Aerial view of lake breach point in October 2011 (a) and breach view from lake in March 2012 (b)**

An investigation found slightly degraded CRSB water quality against a background of water quality significantly degraded by deforestation and agriculture causing secondary salinisation and eutrophication (McCullough and Pearce, 2014). CRSB key end uses of stock watering and contact recreation were not considered significantly impaired by this altered water quality (McCullough, Ballot et al., 2013).

Significant lake water quality improvements were observed following lake breach as halocline development reinforced by thermal stratification as the season warmed maintained a less acidic and fresher/warmer layer over more acidic/colder waters representative of original lake water quality. The river flow-through further remediated pH above recreational guideline pH values in the upper 10 m mixolimnion waters in which people would swim (McCullough et al., 2012).

Following broad stakeholder engagement, including community presentations, a flow-through trial that met leading industry practice was proposed. The Lake Kepwari closure strategy sought first to maintain key CRSB values for aquatic ecosystems, swimming and livestock watering. It then pursued lake water quality improvements to meet recreational criteria and to improve wildlife values. Trial development included a monitoring and reporting plan agreed upon by regulators. Regular quarterly assessment was a key component of this plan and was planned for a minimum of three years for the biota and water chemistry of both Lake Kepwari and CRSB.

During 2012, CRSB flow was not sufficient to inflow into the lake or to cause decant. During this time, water quality in Lake Kepwari deteriorated, and aquatic biodiversity and abundances declined. Prior to CRSB flow-through into Lake Kepwari in 2013, the water column chemistry was very similar at surface and bottom sampling locations. Following the 2013 flow-through, the surface and bottom water quality improved by becoming elevated in pH, nutrients (C, N and P) and reduced metal concentrations.

The effects of the 2011 breach on lake (McCullough et al., 2012) and CRSB (McCullough, Ballot et al., 2013) water quality have been discussed previously. This paper discusses the monitoring results of the 2012–2014 planned Lake Kepwari flow-through trial and provides advice and recommendations to proponents considering flow-through as a mine closure strategy for acidic pit lakes.

2 Approach

The aim of this study was to determine if flow-though was the most appropriate (lowest risk, high benefit) closure strategy for the Lake Kepwari pit lake. Specifically, this study sought to understand how the inflow of circumneutral (and alkaline) CRSB water improved lake water quality, and how passage of CRSB through Lake Kepwari affected CRSB hydrology. Information on how lake flow-through influenced lake and CRSB aquatic biota and how CRSB water and sediment quality was affected is presented elsewhere.

2.1 CRSB flow

Daily rainfall was sourced using the Bureau of Meteorology's Collie East weather station (BOM, 2015). No data smoothing or detrending was used. CRSB flow volumes were measured through three flow gauging stations using bubble flow meters located at the lake inlet, lake outlet and at the outlet of the diversion channel (Figure 1).

2.2 Lake water quality

Four seasonal lake water quality samples were taken quarterly as part of monitoring following breach at three sites across the river's path flowing through the lake from March 2012 to December 2014. All water quality analyses followed standard methods (APHA, 1998). Environmental response data for diatoms, phytoplankton, zooplankton and benthic macroinvertebrate assemblages were collected simultaneously during site visits (discussed elsewhere).

A Hydrolab Datasonde 4a (Austin, USA) multiparameter meter was used to make water column profiles at every metre for oxidation-reduction potential (ORP), dissolved oxygen (DO), temperature, pH and electrical conductivity/total dissolved solids (TDS).

A water sample was also collected for nutrients and metals from the lake surface waters as well as another from approximately 0.30 m above the lake benthos with a Teflon trace metal Kemmerer bottle (Wildco, USA). Upon collection, half of the water sample was filtered through 0.5 µm glass fibre filter paper (PAL Metrigard, USA) and the other half was left unfiltered. Filtered samples were analysed for ammonia (NH_3-N), nitrate-nitrite (NOx-N) and filterable reactive phosphate (FRP), SO_4^{2-}, Cl^- and dissolved organic carbon. Water sample aliquots for nutrients were stored in clean, high-density polyethylene bottles and frozen at −20°C until analysis. Unfiltered samples were persulphate digested and then analysed for total P and Kjeldahl N. Remaining filtered samples were acidified with reagent grade nitric acid (1%) immediately and stored at 4°C until analysed for selected elements by inductively coupled plasma optical emission spectrophotometry and inductively coupled plasma mass spectrophotometry (ICP-MS).

Water quality results were compared to Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand (ANZECC/ARMCANZ) (2000) livestock drinking water guidelines, 80% ecosystem protection guidelines and recreational water quality and aesthetic guidelines.

3 Results

3.1 CRSB flow

In 2013, two heavy rainfall events, one in the second week of August and one in the third week of September, provided heavy flows through the diversion channel, bypassing the lake flow-through (Figure 5, top). Not unsurprisingly, these dates also coincided with the highest flows into the lake (Figure 5, middle). However, these flood events also coincided with the highest flows out of the lake exceeding inflow volumes. Although there was a heavy rainfall episode mid-May, the CRSB did not start flowing into the diversion in 2013 until late July, when inflow gates were opened to divert CRSB flow into Lake Kepwari (Figure 4). River flow monitoring began immediately.

The inflow gates were fully closed on 22 August, when pH differences between upstream and railway bridge sites became greater than 20%. Inflow gates were then progressively opened until completely open on 16 September 2013. During high flow periods from 16 September until late September 2013, over-topping occurred at the previous breach site.

The total discharge from Lake Kepwari at the lake outlet is a combination of CRSB diversion inflow volumes and catchment contributions from the lake itself (Figure 5, bottom). As expected, flow-through of Lake Kepwari appeared to slightly reduce mid-season lake outlet baseflow by smoothing flood peaks. Nevertheless, high rainfall “storm” events still presented high flows at the lake outlet. In both base and flood flow cases, it is unclear without either modelling or baseline data to know unequivocally how baseflow and floods would present at these sites with CRSB flow still maintained through the diversion channel only.

In 2014, there were heavy rainfall episode in early June and mid-September (Figure 5, top). Although the early rain event appeared to be largely absorbed by the dry post-summer catchment, sustained winter rainfall led to a lake water level rise of only around 0.5 m in early July. The second heavy rainfall event lead to a similar magnitude lake water level rise with a quicker response when the catchment was already saturated at the end of winter. Lake water level fluctuation during 2014 was less than 1 m, and this presented similarly at Cardiff Pool, where water level fluctuation was also less than 1 m.

The early June rainfall began in the CRSB flowing at upstream PML45 in early June 2014 (Figure 5, middle). Flow then ceased in early November 2014, showing 153 days flow in total. Flow at the lake outlet commenced in late May 2014 and ceased in late November 2014 for approximately 193 days in total, sustaining downstream flow a further 40 days longer than flow into the lake. Diversion channel flow commenced around three weeks later than lake inflow in late June 2014 and then ceased in mid-November 2014 after approximately 176 days. Inflow and outflow gates were not closed in response to low pH trigger events at any time during 2014 flow.

Flow rates to, and particularly from, the lake were notably bimodal, with peaks in late July and early September once flow was well established. Showing both the increased catchment afforded by inclusion of the lake as well as the very high runoff coefficient of this catchment dominated by Lake Kepwari, flow from the lake of up to 7 m^3/s was around twice as high as flow into the lake at both times.

Cumulative flow volumes had inflection points coinciding with these peak flow periods (Figure 5, bottom). At 14 Gl, total flow volumes out of the lake exceeded inflow volumes at around 40% greater than the input flows of 4 Gl. Less than 3% of lake discharge occurred through the diversion channel.

The total discharge from Lake Kepwari at the lake outlet (PML46) is a combination of CRSB diversion inflow (PML45) volumes as well as catchment contributions from the lake itself (Table 1). It is not clear how flow-through of Lake Kepwari influences the downstream hydrograph at PML46 relative to the previous flow at this point attributed to the diversion and associated catchment. For example, either modelling or baseline data is required to know unequivocally how baseflow and floods would present at PML46 with CRSB flow maintained through the diversion channel only. However, owing to the small diversion catchment and lower runoff coefficient relative to the lake's surface, it is likely that increased flow-through has slightly extended flow initiation and cessation timing with sightly increased mid-season baseflow and increased flood peaks.

Table 1 Total 2013 and 2014 flow volumes (GL) during 2013 Lake Kepwari flow-through trial

Site	2013	2014
PML 45	11.682	10.231
PML 46	17.299	14.382
Difference	5.617	4.151
PML 51	2.695	0.486

Figure 4 CRSB flow-through into Lake Kepwari in July 2013

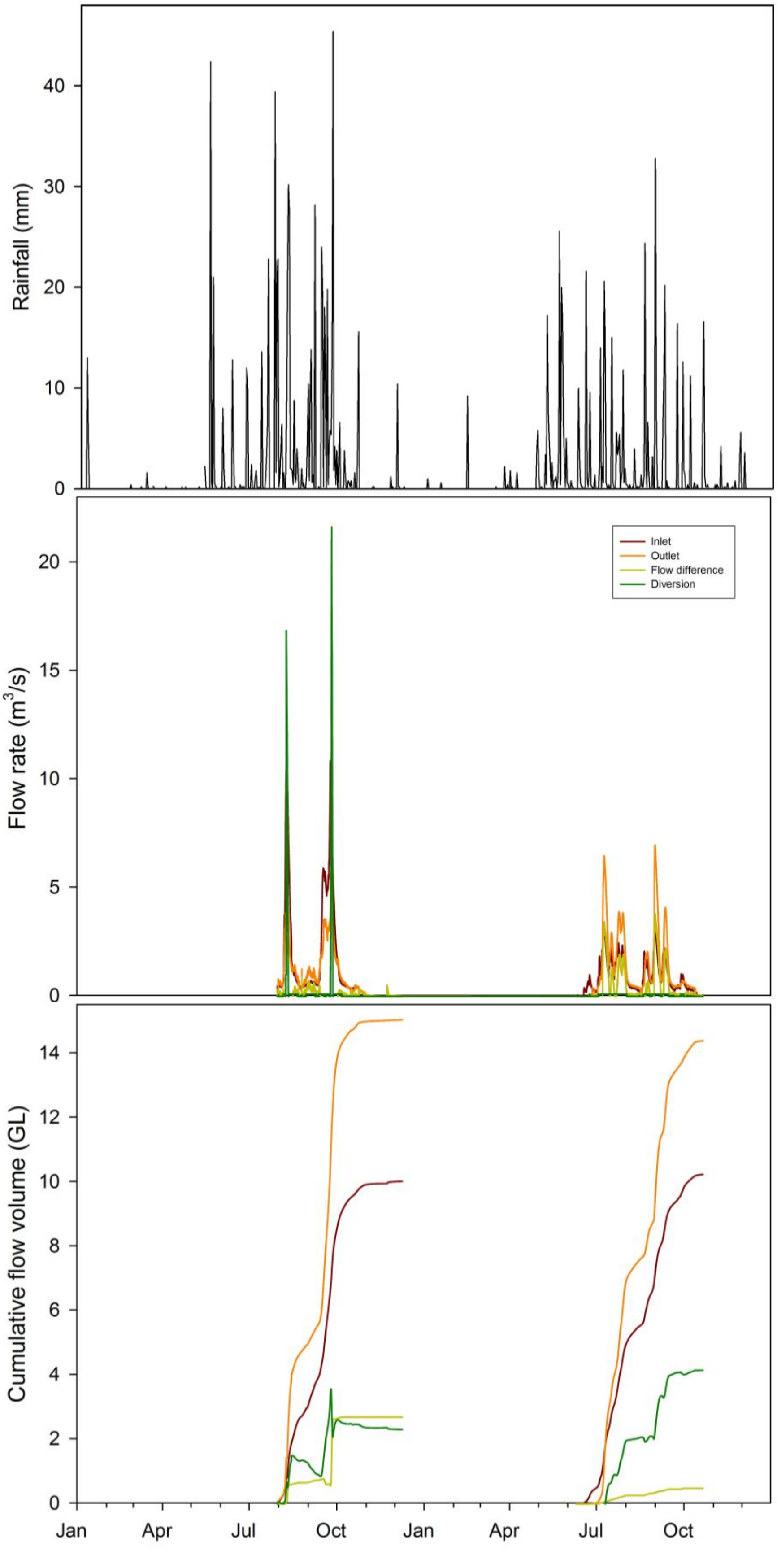

Figure 5 Mean daily rainfall for Collie East, estimated daily (top), daily flow (middle) and cumulative (bottom) CRSB flow past gauging stations immediately above and below Lake Kepwari and in the lower diversion channel

3.2 Lake water quality

Lake Kepwari water quality monitoring data from first filling events indicated improvement of pH when diversion occurred (Figure 6, Figure 7, top). River diversion ceased when the lake was almost full, and pH significantly declined; it continued to do so for a further three years when the lake was completely isolated from the CRSB by the diversion channel embankment. When this embankment failed and fresher circumneutral waters flowed through the lake, surface water quality markedly improved. When the trial began flow-through of CRSB flows in 2013 and 2014, significant further water quality improvements were made, including to deeper lake waters, which represent the bulk of lake volume. Lake pH in 2014 was higher than any historic levels and, importantly, on a reversed and upward trend to that prior to riverine inflow.

During filling, lake water was brackish, reflecting the first flow diversion of the seasonal CRSB waters (Figure 7, bottom). Following flow-through, lake salinity has consistently declined. Although current lake salinity (as electrical conductivity, EC) is elevated, this is only slight and is not expected to impair lake values significantly.

Figure 6 Monitoring Lake Kepwari water quality

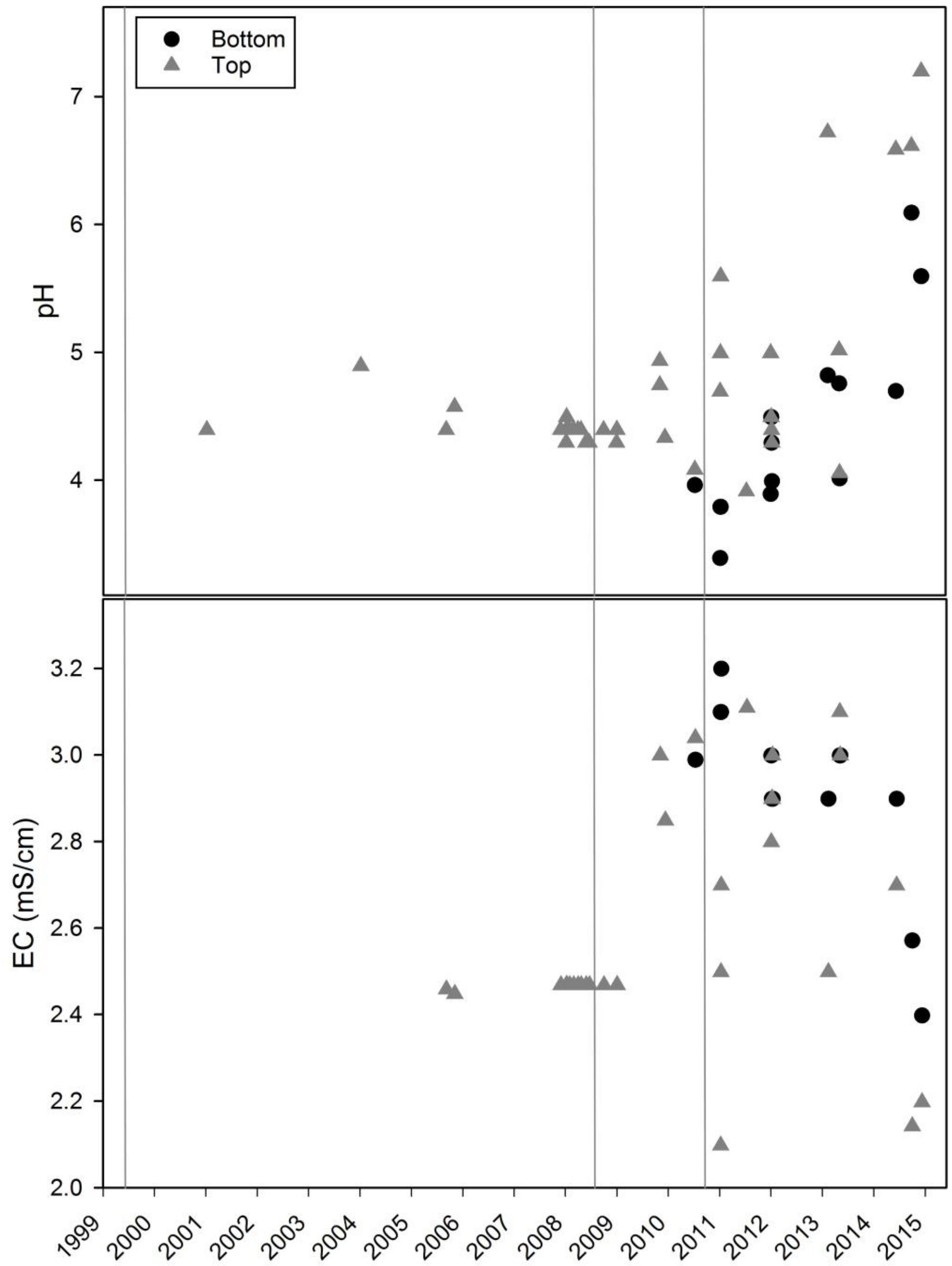

Figure 7 Time series model of Lake Kepwari pH (top) and EC (bottom) historically, during and after flow-through began (after McCullough et al., 2010; McCullough et al., 2012). Dashed regression line indicates surface water pH trend over time

4 Summary

The Lake Kepwari flow-through trial monitored both lake and river ecology and water quality from two years flow-through, plus one year post-breach monitoring (2012). The initial breach event led to improved water quality localised to thermally and/or chemically stratified surface waters, which were then compromised by mixing with more acidic lower water column waters. However, with sustained flow-through, the water column now appears to be well mixed such that, although highest water quality still occurs primarily in surface waters, improved water quality is now found though the lake water column (Figure 8).

Lake water quality in 2014 was significantly improved through nutrient addition and acidity neutralisation above and beyond the flushing afforded by successful flow-through in 2013 after the no flow-through conditions in 2012. Improved water quality, particularly with regard to elevated pH and reduced metal/metalloid concentrations has now significantly reduced the socio-environmental risk of closure of this landform.

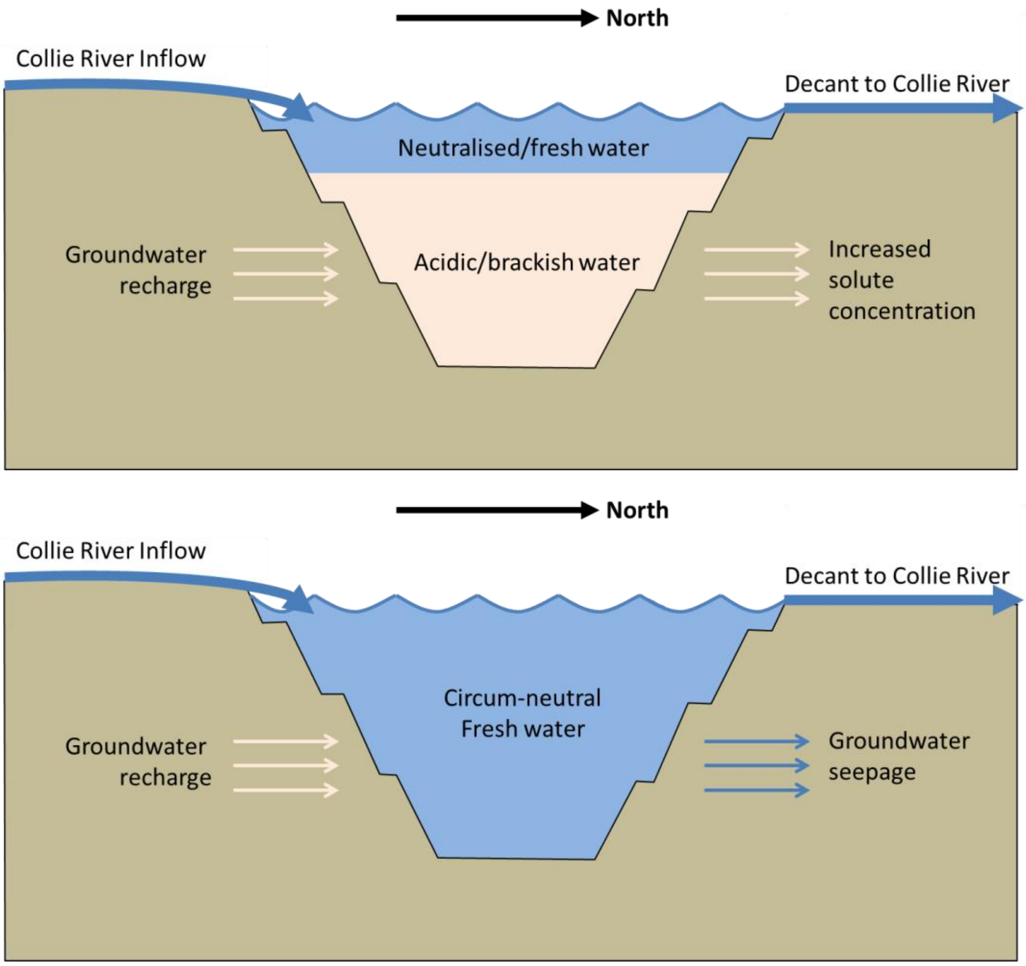

Figure 8 Initial Lake Kepwari saline stratification following breach during flow-through (top) and established fresher water following repeated flow-through during trial (bottom, after McCullough et al., 2012)

This water quality improvement has also led to improvements in aquatic biota biodiversity and abundance, and has changed biota assemblages to those of a more functional lake more typical of a freshwater ecosystem. This improvement also better enables the lake to meet the identified end use values of recreation/aesthetics and livestock drinking than would the no flow-through strategy. River flow-through in particular remediated pH above recreational guideline pH values in the upper 10 m epilimnion waters in which people would swim (Hinwood et al., 2012).

Pit lake risk reduction and lake end use value improvement have not occurred at the expense of CRSB water quality and aquatic ecology, which have been maintained. There is also a possibility that greater flow volume may further assist CRSB social and environmental values. Additionally, the pit lake appears to be fulfilling a theoretically predicted function of reducing CRSB nutrient loads (von Sperling and Grandchamp, 2008; McNaughton and Lee, 2010) elevated as a result of farming land use practices within the shared catchment (Wetland Research and Management, 2009).

5 Conclusion

The Lake Kepwari flow-through trial has demonstrated that social and environmental risk and benefit to the CRSB and Lake Kepwari are best addressed by flow-through as a closure strategy. In line with adaptive management principles, continuing water quality assessment of Lake Kepwari and the CRSB (lake inlet and lake outlet) will be undertaken until the lake is relinquished back to the state. Owing to the low apparent risk, continued monitoring will be made through a revised operating strategy that recognises the learnings from the first phase of the trial and that is a less intensive operational monitoring strategy rather than the current more intensive trial program.

Existing riverine system values must be maintained first and foremost. We further suggest that decant river water quality may, in some circumstances, be improved, notably so in examples of meso-eutrophic river waters flowing through slightly acidic pit lakes.

Flow-through closure proposals for pit lakes must be scientifically justifiable and follow a risk assessment approach (McCullough and Schultze, 2015). This approach should consider both null-positions of no flow-through, as well as variations on flow-through hydrology, such as complete or partial flow-through, or flow-through only for certain seasons and/or flow events. In addition to explicitly considering an assessing risk, benefit and opportunity should be considered, as pit lakes offer an unusually high utility for a region relative to other closure landforms (McCullough and Van Etten, 2011; Gerner and McCullough, 2014). Owing to the high uncertainty, the biotic and physico-chemical attributes of both the upper and lower river and lake should be well monitored (Castendyk, Eary et al., 2014b). Monitoring should directly feed into an adaptive management framework approved by key stakeholders (McCullough et al., 2009).

Acknowledgements

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Key issues in Mine Closure Planning Related to Pit Lakes

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ABSTRACT

Pit lakes form when surface mines close and open pits fill with water, either through groundwater recharge, surface water diversion or active pumping. Historically, the success in closing mines with pit lakes has varied tremendously: there are well known examples of legacy sites requiring perpetual treatment, whereas some other pit lakes have achieved various beneficial end uses. Although access to case studies is often limited, mining companies contemplating new open pit mines have a number of examples in both success and failure from which to draw "lessons learned" that can be used in future mine closure planning.

This paper discusses key issues that should be addressed in the mine planning process to increase the likelihood of successful mine closure. Examples of issues and potential management strategies to address them are given. The key issues examined in this paper include: determining potential risks and beneficial end use opportunities, developing closure objectives and criteria, which may include various water quality, riparian and littoral targets; anticipating and meeting stakeholder and regulator expectations; subaqueous disposal of liquid and solid mine waste; predicting and managing water balances; identifying contaminants of concern; historical reliability of model predictions; mitigating acid mine drainage; the importance of understanding long-term vertical mixing regimes; and health and safety issues.

Keywords: mine pit lakes; sustainability, AMD, closure, planning

INTRODUCTION

Pit lakes form when surface mines close and open pits fill with water, either through passive groundwater recharge, surface water diversion or active pumping. They often display poor water quality through Acid Mine Drainage/Acid and Metalliferous Drainage (AMD). Historically, the success in closing mines with pit lakes has varied tremendously: there are well known examples of legacy sites requiring perpetual treatment, whereas some other pit lakes have achieved various beneficial end uses. Although access to case studies is often limited, mining companies contemplating new open pit mines have a number of examples in both success and failure from which to draw "lessons learned" that can be used in future mine closure planning (Castendyk 2011).

This paper discusses key issues that should be addressed in the mine planning process to increase the likelihood of successful mine closure. Examples of issues and potential management strategies to address them are given with reference to previous experiences in North America, Australia and Asia.

KEY ISSUES

Determining Closure Objectives and Developing Closure Criteria

Discharge criteria applied to pit lakes are site-specific and dependent on the responsible regulatory agency. In most jurisdictions, there are no set guidelines for pit lake discharge. If pit lake water concentrations are below applicable generic water quality guidelines, then water quality would be deemed acceptable, but this will rarely be the case. More likely, site-specific objectives will need to be developed by the proponent of each pit lake. Site-specific objectives can be derived based on effects thresholds, technological limits, background concentrations, or combinations thereof.

Pit lakes are generally expected to be managed as closed-circuit waterbodies until they achieve water quality that will not cause adverse effects to aquatic life, at which time they can be reconnected to the receiving environment. If water quality in the pit lakes is not adequate by the time the lakes fill, active treatment may be required, as well as water diversions around the pit lake.

There are three nested "layers" that can be used to define and gauge success in pit lake closure:

1. **End use** – will the pit lake and associated watershed meet land use requirements for post-closure mine sites that are set regionally and nationally?
2. **Objectives** – will the pit lake meet functional targets that are achievable, desirable to stakeholders and acceptable to regulators?
3. **Criteria** – will the pit lake meet prescriptive criteria, such as site-specific water quality and toxicological thresholds?

There are several sources of information that can be used to define success, such as:

- Corporate sustainability goals and targets (MMSD 2002);
- Commitments made by the mining company in environmental impact assessments (EIAs) and other applications, which include commitments made by previous property owners;
- Numerical predictions that have been generated in EIAs and that have been used in ecological risk assessments;

- Stakeholder expectations;
- Regulatory requirements (Jones and McCullough 2011);
- Analogue lake studies (Van Etten et al. 2014);
- Observed water quality from existing pit lakes in similar geologic deposits (Johnson and Castendyk 2012)
- Leading, international mining-industry practice; and
- Prescriptive, site-specific objectives that are based on biological thresholds and ecological risk assessments.

The importance of developing closure criteria for pit lakes early in the planning process cannot be overstated, because all mine closure design and mitigation should be directed toward meeting these criteria.

Anticipating and Meeting Stakeholder and Regulator Expectations

As with the other components of mine operation and closure, all stakeholders should be identified early and consulted for their input on end of mine life quality and objectives, including objectives for pit lakes (Swanson 2011). Early engagement of stakeholders can lead to constructive input into the planning of pit lakes, reduced costs, fewer delays, and overall public/stakeholder/regulator acceptance.

Design for pit lakes is typically done by involving engineers and scientists, but not stakeholders (Swanson 2011). It is recommended to consult stakeholders on visions for pit lakes and potential beneficial end uses of pit lakes (McCullough and Lund 2006). Participation by communities in developing mine remediation targets leads to better decisions, and in some cases to lower overall costs for mine remediation (NOAMI 2003). This is because the major stakeholders were involved from the beginning in decisions that could affect their enjoyment/use of the landscape. Information presented to communities on pit lake predictions can be complex, and thus information should be presented in an easy-to-understand format in order to engage the stakeholders in constructive discussions (NOAMI 2003).

Predicting and Managing Water Balances

The time to refill pit lakes is site-specific and must be determined on a case-by-case basis. In cases with high rates of evaporation or highly permeable aquifers, the pit lakes can refill in a few years. In arid regions, some pit lakes will never refill passively, and are termed “terminal” pit lakes (McCullough et al. 2013) because they act as a groundwater sink. While not ideal, such lakes may be used as mitigation to prevent contaminated groundwater from migrating away from a mine site. In terminal lakes, evaporation is the only route through which water leaves a pit lake, so it can be expected (and readily predicted with mass balance models) that concentrations of solutes will increase over time (Castendyk and Eary 2009; Geller et al. 2013a). The ultimate concentrations may be controlled by solubility, which can be predicted using geochemical software.

In the sub-Arctic region of Canada, where net evaporation is low, it is expected that pit lakes will refill passively, but it is preferable to accelerate the filling process to reduce the closure management period. This option should be evaluated as part of the closure planning process, in consideration of regional surface hydrology and availability of water to be used for filling.

Connection of the pit lake to surrounding groundwater sources can play a large role in the water quality and hydrological cycle/budget of the pit lake; if a pit lake water surface is above the water table, water will flow out of the pit to the groundwater and thus provide a pathway to transport potential contaminants to a larger area (Castendyk and Eary 2009).

Understanding Long-term Vertical Mixing Regimes

Compared to natural lakes, pit lakes are more prone to become meromictic (lower layers non-mixing) because they generally have smaller surface areas, larger depths and higher salinities. Vertical mixing in lakes is primarily driven by wind currents across the lake surface, and the smaller fetch of pit lakes provides less opportunity to translate wind energy into water currents that are necessary for lake turn-over.

In pit lakes, as in natural lakes, the frequency and depth of vertical mixing will affect many other variables. These parameters must be defined in advance of developing geochemical predictions of water quality so that accurate volumes for epilimnion, hypolimnion, and monimolimnion layers can be accurately represented and mixed at appropriate intervals. Vertical mixing transports oxygen to the lower portion of the lake, which in turn affects biological and chemical reactions. For example, oxidation state influences the mobilization of metals and cycling of nutrients. Of particular importance is the potential effect of oxidation state on sulfide minerals; under oxidizing conditions, sulfide minerals will react to form sulfuric acid and dissolved metals, whereas under reducing conditions, sulfide minerals will precipitate – a process that has been used to mitigate AMD in meromictic pit lakes (Pelletier et al. 2009). Given the influence of vertical mixing on these processes, the anticipated mixing behavior of a pit lake should be evaluated and understood as early as possible in the mine planning process.

There are a variety of guidelines that describe lake geometries that will affect lake mixing. The most common is the relative depth, defined as the maximum depth as a percentage of mean diameter. Natural lakes usually have relative depths of less than 2%, whereas pit lakes typically have relative depths of 10 to 40% (Doyle and Runnels 1997). While measures such as relative depth provide useful descriptors of pit geometries, they are not predictive measures because they do not account for other important variables, such as water density and wind speed. The most reliable method for predicting lake mixing is through the use of numerical models (such as CE-QUAL-W2 or DYRESM) that mechanistically account for these variables.

Identifying Contaminants of Concern

There are a wide range of contaminants of concern (COCs) in pit lakes. The most common COCs in hardrock pit lakes are low pH and elevated element concentrations caused by acidic mine drainage (AMD). AMD is a phenomenon that occurs when sulfur-bearing waste rock, tailings or other materials are weathered during mining and mine closure practices. Weathering of sulfide minerals can lead to release of acid and elevated concentrations of contaminants in runoff, groundwater or pit lake water. These acidic waters often carry a high load of elements that are more soluble at low pH. AMD is commonly associated with coal and hard rock mines

The COCs at a given mine are often, but not always, related to an obvious source such as the ore body or extraction chemicals. For example, the Berkeley Pit Lake in Montana, which is perhaps the most famous “worst-case” example of a pit lake, is a former copper mine pit that now contains levels of copper, zinc, and iron that exceed water quality guidelines by orders of magnitude (Gammons and Duaime 2006). Similar contamination has been observed at copper mines in California (Levy et al. 1997) and Sweden (Ramstedt et al. 2003).

Long-term water quality in a pit lake can be influenced by hydrochemical processes such as geoenvironmental characteristics, water balance, mineral solubility, and sediment biogeochemical processes (Geller et al. 2013a). Constituents that most often exceed guidelines are copper, cadmium, lead, mercury, nickel and zinc, followed by arsenic, sulfate, and cyanide (Kuipers et al. 2006). Blasting residues such as ammonia and nitrate are also often elevated in mine waters, and may persist into closure (Banks et al. 1997). In sub-Arctic Canadian mines, salinity and major ions are typical COCs (Environment Canada 2012) because of saline groundwater that must be dewatered for mining. The saline groundwater may be disposed of in pit lakes, or saline groundwater may flow passively into pit lakes at closure when dewatering ceases. In oil sands pit lakes, the COCs are primarily organic constituents such as naphthenic acids, phenolics and polycyclic aromatic hydrocarbons originating from process waters and tailings (CEMA 2012).

Less obvious COCs may be present as well. For example, at the proposed Gahcho Kué Diamond Mine (De Beers 2012), geochemical testing of pilot plant tailings identified phosphorus as a COC, which led to changes in the closure plan to mitigate runoff from mine wastes and to avoid eutrophication of closure waterbodies. Total suspended solids can be expected to be elevated during the early years of lake development, before vegetation becomes established in the littoral zone, but this should be a temporary phenomenon in a properly designed pit lake.

In summary, while there may be obvious COCs at a given mine, a full suite of metals, major ions, nutrients and organics should be evaluated to determine site-specific COCs prior to mine development.

Mitigating Acid and Metalliferous Drainage (AMD)

Poor water quality degraded by AMD is the single biggest environmental risk and cause of beneficial end use loss for pit lakes (McCullough 2008). Mine drainage may be acidic, neutral or even alkaline as constituents such as metals and metalloids may be in elevated concentrations in all. Once begun, the process of AMD is very difficult to stop. Hence, the emphasis on AMD management should always be first on preventing weathering of potentially acid generating (PAG) materials by exposure to water and oxygen (Castendyk and Webster-Brown 2007). This process begins by long-term geochemical characterization of all materials that may contact pit lake water or water sources including above ground sources, such as waste rock dumps and tailings impoundments, and below ground sources, such as backfill and fractured geologies.

Disposal of PAG materials above the water table is usually best suited to arid climates where AMD production will be limited by water availability. However, a strategy often considered to reduce pit lake AMD issues is subaqueous disposal of PAG occurring in tailings, waste risk and pit shell exposures (Dowling et al. 2004). However, subaqueous disposal of waste should not be thought of as a singular solution to PAG management. Rather it is merely one consideration of a broader closure strategy that, when used appropriately and in certain circumstances, may reduce AMD production and long-term environmental and social liability.

Where AMD has not been prevented, a number of active and passive treatments are available, although all of these treatments should be considered requiring ongoing attention and maintenance (Gammons et al. 2009; Geller et al. 2013b; Younger and Wolkersdorfer 2004). Active treatments may be simple limestone or lime putty additions to treat acidity, although the ongoing cost, particularly in remote areas once mine infrastructure is closed should not be under-estimated. The economic liability to the remaining responsible jurisdiction is likely to exceed the economic benefit from mining with a few generations of treatment, which is why active treatment is only typically sought when there is a risk of off-site contamination exposure to social or environmental receptors.

Passive treatments may range from strategic catchment-scale diversions of inflows to attenuate and dilute pit lake waters (McCullough and Schultze 2015) to initial or ongoing treatment with biologically active materials such as nutrients and organic matter (Kumar et al. 2011).

Subaqueous Disposal of Liquid and Solid Mine Waste

The option to dispose of mine waste in pit lakes is often attractive to mining companies because it is more cost effective than other treatment or disposal technologies. Disposal of mine waste in pit lakes is an accepted practice in some industries and regions (Davé 2009; Dowling et al. 2004; Schultze et al. 2011). However, it is controversial and considered unproven until demonstrated at the field scale in the oil sands industry (OSTC 2012). If successful, several other companies in the region will likely apply water-capped tailings technology with a potential savings of billions of dollars for the industry as a whole compared to other disposal technologies. Deep pit disposal of fine tailings has also been approved for the diamond mining industry in Northern Canada (De Beers 2012).

If subaqueous disposal of tailings are contemplated, the following issues should be evaluated to reduce risks to closure water quality:

- **Tailings resuspension** – a hydrodynamic analysis should be completed to understand the potential for resuspension of fine particles, and the formation of buoyant plumes;
- **Metal leaching and AMD** – geochemical testing should be completed to predict the potential for acid generation and metal leaching, and to understand which oxidation state would minimize these effects on water quality; and
- **Sediment toxicity** – standard bioassays should be conducted to predict the toxicity to benthic organisms.

Health and Safety Issues

The most significant acute health and safety risks for persons in and around pit lakes relate to falls and drowning. Pit lake highwalls may often be unstable, particularly following rebounding groundwater pore pressures and decades of wave action. Unstable walls frequently result in slips that may endanger nearby structures and persons near the highwall (McCullough and Lund 2006). Where communities reside nearby, pit lakes may present risks for recreational swimmers where there is a risk of drowning with the steep lake margin typically of pit lake edges or by falls from high walls into water or submerged obstacles that have not been regraded (Ross and McCullough 2011).

Chronic health risks are not well understood, but there is potential for health issues for recreational users in AMD contaminated pit lake water; even in remote areas where pit lakes may be used as recreational opportunities. Low pH and elevated contaminant concentrations may lead to skin and eye damage and irritation, particularly for regular exposures in vulnerable groups such as children and the elderly (Hinwood et al. 2012).

There are also human health risks where end uses include fisheries; either planned or unplanned. Aquatic ecosystem foodchains have been found to accumulate contaminants such as selenium, mercury and cadmium. These metals bioconcentrate in keystone predator sportsfish and crustacea (McCullough et al. 2009b; Miller et al. 2013).

Historical Reliability of Model Predictions

The reliability and accuracy of mine water predictions was examined by Kuipers et al. (2006) in a comparison of water quality predictions made in environmental impact statements to operational water quality observed at hardrock mines. The mines that were examined included major mines across the Western USA, but the issues they identified are applicable to mines worldwide. They found that in the majority of cases, water quality predictions did not perform well, and impacts were often underestimated. They identified three main causes for the discrepancies:

- **Inadequate hydrologic characterization** – inaccuracies arose from overestimating dilution potential, poor characterization of the hydrologic regime and poor flood forecasting.
- **Inadequate geochemical characterization** – inaccuracies arose from inadequate sampling of geologic materials, lack of proper geochemical testing of materials such as metal leaching and AMD potential and improper application of test results to models.
- **Mitigation failure** – in many cases, mitigation was assumed to reduce concentrations, but the mitigation was either not effective or not implemented.

Although poor water quality prediction performance has been found at hardrock mines, present and future pit lake modelling efforts should be able to improve upon this record. Success in predicting water quality will be reliant on following leading modelling practices that were not adhered to in many of the case studies in Kuipers et al. (2006). Guidance for predicting pit lake water quality is provided in a companion document by Maest et al. (2005) as well as by Vandenberg et al. (2011).

In particular, a post-audit of water quality predictions is essential (Dunbar 2013) for identifying excursions from predictions early in the mine life and applying adaptive management strategies as soon as possible. Post-audits of modelling predictions should be available to stakeholders, reviewed by regulators, and ideally, disseminated to the wider modelling community so that they can learn from the strengths and weaknesses of past experiences and continually improve their methods.

CONCLUSIONS

Pit lakes are highly variable systems with a wide range of outcomes observed worldwide in terms of chemical characteristics and suitability for aquatic habitat. While there are examples of very unsuccessful pit lakes, these serve as "lessons learned" that can be followed to increase the likelihood of success in constructing future pit lakes (Castendyk 2011). The most important lessons learned are to develop a conceptual model of the pit lake and understand its processes as early as possible; engage stakeholders early in the process; begin environmental monitoring at the exploration stage and conduct a post-audit of predictions to guide adaptive management (Castendyk 2011; Gammons et al. 2009).

The key issues described above should be considered in each of the planning, designing, commissioning, and abandonment stages of a pit lake. The outcome of a decision made or an assessment completed during a previous stage of development may be found to be incorrect or no longer valid as environmental data or stakeholder or regulator requirements evolve. Or, the pit lake and its inflows may be altered by changing mine plans or mine closure plans in response to fluctuating commodity prices. Consequently, mining companies should anticipate an iterative process whereby assumptions and decisions are refined to reduce uncertainty related to the issues above. This may involve reconsidering options and revisiting strategies discounted earlier under different circumstances such as understanding of the physico-chemical context and of regulatory and other social constraints and expectations. This iterative process of pit lake closure planning refinement should form an explicit part of mine closure planning for the broader site (McCullough et al. 2009a).

Guidance manuals (e.g., CEMA 2012; McCullough 2011) and compilations of pit lake experiences and research (Castendyk and Eary 2009; Gammons et al. 2009; Geller et al. 2013a) have been developed in the past five years, and these should be consulted throughout the planning, design, and construction process for additional details.

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ANNEXURE D

Further relevant pit lake develop examples of rehabilitated coal mine lakes

Post-Mining End Use	Location	Type of Mining Operation	Description	Reference
Recreation and Tourism	New Federal States, Germany	Lignite	Numerous lignite surface mining and refinement plants in the territory of the former German Democratic Republic closed and being remediated for recreational use.	Krüger *et al.* (2002)
	TXU Mining Tatum Mine, Beckville, Texas, USA	Lignite	Reclaimed pit lake using a pond-in-series design to create five wetland areas. Native grasses and forbs planted and more than 40 acres of hardwood species established. Fish stocking also made for 7 local sports-fishing species.	http://www.mii.org/ReclStories/Tatum/Tatum.html
	Stone Mining Company, Pikeville, Kentucky, USA	Coal	Coal slurry impoundment converted into a recreational lake approximately 21 acres in area. Lake stocked with bluegill, channel catfish and largemouth bass.	http://www.mii.org/stonecoal/stonecoal.html
	Lake Kepwari, Collie, Western Australia	Coal	Final open void shaped and contoured to create a recreational lake with central island, Lake Kepwari. Government funding available to provide boat launching facilities, ablution blocks, parking bays and BBQ facilities. Development currently limited by acidic water, however, new river flow through strategy may resolve this. Currently used by Defence Force personnel and commercial divers for deep diving training. 97.8 ha area and around 65 m deep.	http://www1.premiercoal.com.au/environ_mine.asp Ashton and Evans (2005)
	Stockton Lake, Collie, Western Australia	Coal	Historic lake used for camping by 'grey nomad' caravans. Also used extensively for water skiing and deep SCUBA diving due to very clear (albeit acidic) water and underground adits for cave diving. Also picnics and marron (crayfish) fishing. 15.4 ha area and 25 m deep.	McCullough *et al.* (2010) Buzzacott and Paine (2012) Hinwood *et al.* (2012)
	Black Diamond Lake, Collie, Western Australia	Coal	Historic lake used for camping and BBQs predominantly by local residents. Also picnics and marron (crayfish) fishing. 4.6 ha area and 8 m deep.	McCullough *et al.* (2010) Buzzacott and Paine (2012) Hinwood *et al.* (2012)
	East Pit Lake, AB, Canada	Coal	Collaborative effort by Transalta and the Stony Plain Fish & Game Association. Developed to sustain a sport fishery. Provincial awards for 'best project to ensure the future of healthy fish or game' and 'best fishery project'. 19 ha area and 8 m deep.	https://circle.ubc.ca/bitstream/handle/2429/12596/1991+-+Brocke,+Chymko+-+The+Whitewood+Mine+Lake.pdf?sequence=1

Post-Mining End Use	Location	Type of Mining Operation	Description	Reference
	Black Nugget Pond, AB, Canada	Coal	Successful fish stocking program, despite elevated major ions. 7.3 ha area and 5.4 m deep.	Mudroch *et al.* (2002)
	Pleasure Island Pond, AB, Canada	Coal	Good recreational fisheries on a “put and take” basis. Stocking discontinued in 1979 due a to a public access dispute with private land owner.	Mudroch *et al.* (2002)
	Mine Lake 116, Lusatia, Germany	Lignite	Mountain bike and walking trails. Wildlife habitat and lookouts and sculptures. Two hours from Berlin in area of otherwise economic decline.	Schultze (2012)
	Lake Geierswald, Lusatia, Germany	Lignite	Geierswald See is a former open-pit lignite coal mine flooded in the early 2000s in a development project initiated by state government. Beach-based recreation including café’s, BBQ, boat hire and sailing. Two hours from Berlin in area of otherwise economic decline.	http://www.gettyimages.co.uk/detail/news-photo/small-pier-juts-into-artificial-geierswalder-see-lake-on-news-photo/178119519 http://www.dw.de/germanys-great-lakes-take-shape/a-1632138
Wildlife Conservation	Arch Coal, Mingo Logan Mine, Southern West Virginia, USA	Coal	Over 200 acres of ponds and wetlands created on previously mined land. Attracts water fowl, aquatic species and other wildlife	http://www.mii.org/ArchWetland/ArchWetland.html
	Alford Field Mine, Petersburg, Indiana, USA	Coal	Two separate pits were mined and reclaimed with several peninsulas and coves, as well as some islands. Numerous ponds and small wetlands included in rehabilitated area to maximize the area’s potential for use as fish and wildlife habitat.	http://www.mii.org/Lakewoods/Lakewoods.html
	Oxford Mining, Muskingum County, Ohio, USA	Coal	Large pit impoundment and two large wetland impoundments constructed as part of the reclamation plan. Developed year-round and part-year storage areas, heavily vegetated pasture and areas planted with native trees to enhance specific wildlife habitat.	http://www.mii.org/oxfordcoal/oxfordcoal.html
	Universal Mine Slurry Wetland Area, Universal, Illinois, USA	Coal	Over 80 acres wetland created from a coal wash slurry deposit. Includes 20 acres of permanently impounded water and surrounding wildlife habitat.	http://www.osmre.gov/news/092205.htm

Post-Mining End Use	Location	Type of Mining Operation	Description	Reference
	Lovett and Silkstone Lakes, AB, Canada	Coal	Naturally colonized by local fish; growth rates exceed those in nearby natural lake; high density and diversity of benthic invertebrates; staging area for migratory birds; supports terrestrial wildlife. 6.4 and 6.0 ha area and 14.8 m and 18.0 m deep, respectively.	http://www.sherritt.com/getattachment/c51c1317-1e5a-45b9-9d41-7af863910357/Sherritt-s-Coal-Business-Receives-Environment-Awar
	Lac Des Roches	Coal	Pit lake used to avoid backfill costs. Flow though from Jarvis Creek with littoral zone developed to improve productivity at 5% total lake area. A number of measures to improve biodiversity including artificial habitat for periphyton cover and fish habitat, macrophytes translocation and construction of catchment habitat for bighorn sheep. 16.2 ha area and 70.0 m deep.	Mudroch *et al.* (2002)
Aquaculture	Arch Coal Mingo Logan Mine, Thacker Fork, West Virginia, USA	Coal	Cold water from mine used for fish breeding in an arctic char hatchery. Dissolved gases extracted and liquid oxygen added before water is pumped to incubators and trays in the hatchery.	http://www.archcoal.com/environment/aftermining.asp
	Lake WO3, Premier Coal Mine site, Collie, Western Australia	Coal	Acid mine lake water treated by a fluidised limestone chip treatment system and then gravity fed into ponds used for marron (freshwater crayfish) aquaculture by local indigenous business group.	Ngalang Boodja Enterprises Pty Ltd (2013) McCullough *et al.* (2010)
	Lake WO5H, Premier Coal Mine site, Collie, Western Australia	Coal	Acid mine lake water treated by a fluidised limestone chip treatment system and then gravity fed into six polyculture ponds used for silver perch and freshwater crayfish aquaculture research. 43.5 ha area and 81 m deep.	Whisson and Evans (2003); Stephens and Ingram (2006) McCullough *et al.* (2010)
	Ridgeway North, NC, USA	Coal	Extensive shoreline contouring for erosion control and lake shore development. Creation of wetlands. Support fish population.	http://www.clu-in.org/download/issues/mining/Hard_Rock/Tuesday_April_3/Case_Studies/03_Peacey.pdf

Post-Mining End Use	Location	Type of Mining Operation	Description	Reference
Agriculture	Vickery Mine, Gunnedah, NSW	Coal	Shallow void rehabilitated to grazing land currently leased to several leaseholders. Stocking rates at approximately 0.5 head:2 ha.	(NSWMC, 2014)
	Freedom mine, Bismarck North Dakota, USA	Lignite	Mined land reclaimed to croplands for wheat, grasslands for livestock grazing and hay production, and the creation of wetlands for wildlife. Irrigation water sourced from surface water collected in sediment ponds	http://www.mii.org/coteau/coteau.html
	Belle Ayr coal mine, North East Wyoming, USA	Coal	Land has been reclaimed for the pre-mine use of livestock grazing and wildlife habitat. Success from planting of trees and shrubs enhanced through use of trickle irrigation and water harvesting.	http://www.fs.fed.us/rm/sd/land_disturbed_by_mining.pdf
	Oxbow mine, North Western Louisiana, USA	Lignite	Permanent ponds incorporated into the post-mining topography to increase landowner value and provide water for cattle and wildlife	http://www.mii.org/Oxbow/Oxbow.html
Water Storage/ Potential Future Uses	Westside Mine, Newcastle, New South Wales	Coal	Proposed for water storage and placement of coal reject material with long-term plan to assess for developments such as landfill, recreation or aquaculture.	http://www.glencorecoal.com.au/EN/Operations/Documents/Final%20Void%20Management%20Plan.pdf
Industrial Water	Garrick East pit, Collinsville Queensland	Coal	Remediated (reduced salinity) pit lake water for haul road dust suppression to relieve pressure on competitive regional water resources	(McCullough *et al.*, 2008)
Chemical Extraction	Piast/Czeczott, Poland	Coal	Different treatment and disposal schemes are described and compared from a technical-economical point of view. Recommends drying of sodium chloride (NaCl) and sale in Poland and/or on the export market as preferred option.	Ericsson and Hallmans (1994)

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ANNEXURE E

Statement of Limitations

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