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EDITH COWAN UNIVERSITY
PERTH WESTERN AUSTRALIA

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Mine Voids Management Strategy (V): Water Quality Modelling of Collie Basin Pit Lakes.

By, Dr. Michael Müller
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Prepared for,

Department of Water (Western Australia)

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Frontispiece

Plate 1. Michelle Newport and Dr. Naresh Radhakrishnan surveying the bathymetry of the WO3 pit lake.

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Executive Summary

1. Pit lakes can form in open cut mining pits, which extend below the groundwater table. Once dewatering ceases, then groundwater, surface water and direct rainfall contribute to the formation of a pit lake.

2. Pit lakes are common in the Collie Basin in Western Australia (WA). They form a lake district consisting of 15 lakes, although two are currently being re-mined. As other mine operations in the Basin finish further pit lakes are anticipated, many of these potentially much larger than existing pit lakes (e.g., Muja). It is estimated that the total volume of water in Collie pit lakes exceeds 200 GL.

3. Collie pit lakes have different physico-chemical characteristics than natural lakes, such as a small catchment vs. relatively great depth, less nutrients, low pH but high metal concentrations.

4. The purpose of this document is to use modelling of representative pit lakes in the Collie Lake District, south-western Australia, to determine the effect of groundwater management and climate changes scenarios on pit lakes water quantity and quality.

5. This report was jointly funded by the Department of Water, Western Australia and the Australian Government under its $12.9 billion *Water for the Future* plan.

6. Modelling was made for a single example of each of the three lake types previously identified by conceptual modelling which showed differences between pit lakes due to higher pH and lower ORP in historic pit lakes, higher salinity in rehabilitated pit lakes and lower salinity and pH in un-rehabilitated pit lakes. That is, “historic”, “new rehabilitated” and “new un-rehabilitated” lake types respectively.

7. Model conclusions should not be regarded as accurate predictions due to inherent uncertainties, but they may still allow useful scenario testing and quantitative estimates under different environmental and operational scenarios.

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1 Background

1.1 Pit lake formation

Open cut mining operations have become common practice over the last few decades in Australia, as a method of extracting commercially useful ore found near the surface. Since backfilling is normally unfeasible practically or economically, an open pit after completion of extraction operations is left. This is called a mine void. After mine operations are discontinued and dewatering ceases, most of those that extend below the natural groundwater table, fill by inflow of groundwater, direct rainfall, and runoff from adjacent drainage basins and the void catchment. Natural filling may take many years to complete. To reduce oxidation of mining waste and wall rocks, to inhibit the activity of acidophilic sulphur-oxidizing bacteria, and to promote anoxic conditions at the lake bottoms which may minimize the formation of acids and dissolved metals, some pit lakes are rapidly filled with stream or river diversions. The water qualities in such pit lakes depend on the filling water and geological catchments and are highly variable. Although the water level may continue to fluctuate as it equilibrates or as climate and local groundwater levels alter, once containing water, the empty mine void has now become a pit lake.

During the first half of the twentieth century, most pit lakes formed as a result of coal mining were located in North America. With the introduction of high-powered steam shovels in 1911, the surface mining industry became a major source of coal in the United States (Gibb & Evans, 1978) and left hundreds of pit lakes. Since the implementation of the federal Surface Mining Control and Reclamation Act of 1977, the formation of coal pit lakes in the United States has virtually stopped. However, coal pit lakes are still allowed and are sometimes desirable, considering that backfilling is normally unfeasible practically or economically. Also the needs of communities and ecology may allow pit lakes. There are some pit lakes being constructed at coal-mining sites in Canada (Sumer *et al.*, 1995) to serve as fish and wildlife habitat and for recreational use.

New mining technologies have led to a large increase in open cut mining of gold, silver, uranium, and base metals (Miller *et al.*, 1996). Open cut mining is currently in use mainly in Australia, Bulgaria, Canada, Chile, Colombia, Indonesia, Kyrgyzstan, Mongolia, Namibia, Peru, Portugal, Russia, South Africa, United Kingdom, United States, and Zambia. The number of future open cut mines is likely to continue with current and predicted demands for minerals and energy, the global financial crisis notwithstanding. Except for those in the most arid areas, deep open cut mines are likely to develop pit lakes when mining operations end. Given the large number of pit lakes that will form worldwide and the large volume of water they will contain, the quality of the water in these lakes will be of profound importance, especially in areas with scarce water resources.

1.1 Pit lake characteristics

Pit lakes differ physically from natural lakes in having a markedly higher ratio of depth to surface area (Figure 1). This is described by percent relative depth, which is defined as the percentage of a lake's maximum depth compared to its width calculated from its surface area by assuming the lake is approximately circular. A typical natural lake has a relative depth of less than 2%, although some may exceed 5%. Pit lakes commonly have relative depths between 10 and 40% (Doyle & Davies, 1999). This causes pit lakes easily stratify with the consequential changes in chemical characteristics with depth. Total dissolved solids and electrolytic conductivity tend to increase with depth; values near the bottom are often several times those at the surface. The hypolimnion (lower stratum) of a stratified lake has the tendency to contain low dissolved oxygen concentrations, if enough oxygen demand (chemical and/or biological) is high enough. The existence of a sub-oxic or anoxic (no oxygen) layer in a pit lake can have significant effects on the lake's chemical and biological characteristics and thus on its potential for remediation.

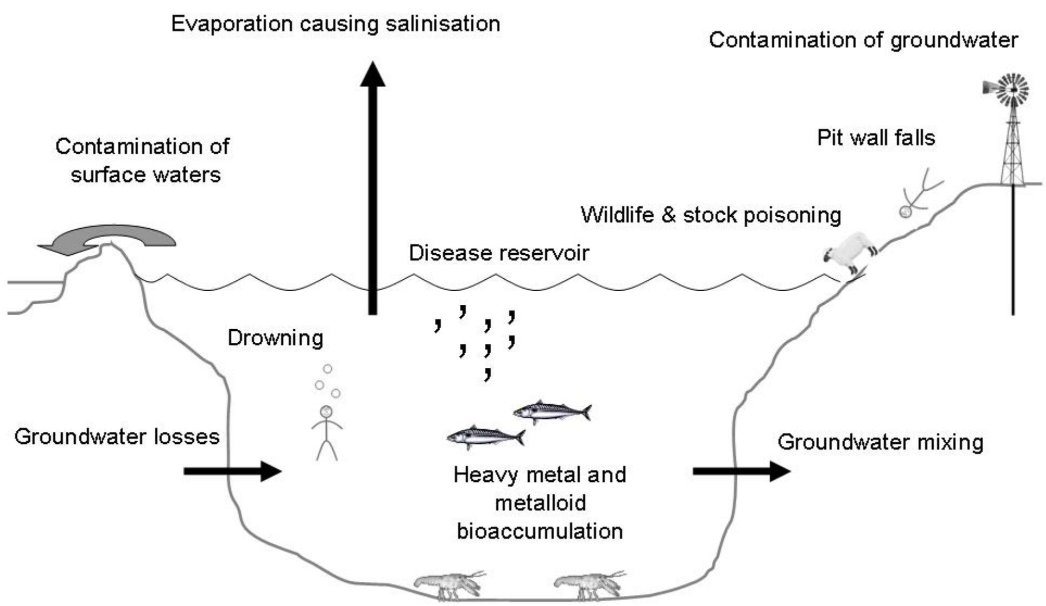

Figure 1. A conceptual model of the risks of pit lakes (after McCullough & Lund, 2006).

Where pit sides are battered for public access or to promote development of riparian (fringing vegetation) zones, deep pits will still have a bathymetry unlike natural lakes with steep sides below the battering. The size of mining pits in Australia ranges from relatively small urban borrow pits of about 100 m in diameter, to enormous open cut operations such as Mount Whaleback mine in the Central Pilbara, (WA) which will have final pit dimensions of 5.5 km by 2.2 km and a depth of 500 (Johnson & Wright, 2003). These new mining pit lakes have few natural counterparts in Australia, especially in depth. Furthermore, as the water level in the pit lake equilibrates, it is frequently deep within the walls of the open-cut, creating very little opportunity for natural slopes to the water surface; this also influences water mixing due to sheltering from winds (Huber *et al.*, 2008a).

As pit lakes typically have limited catchments, inflows of surface water tend to be small which may be useful in preventing worsening water quality from exposed geologies. However, where exposed geologies are not problematic, it may desirable for pit lake water quality to capture clean surface waters and small catchments may limit this. Pit lake water quality can be highly variable; particularly for acidity, salinity, hardness and metal concentrations which are primarily governed by the pit lake catchment hydrology and geochemistry (Miller *et al.*, 1996). For example, pit lake water quality may become acidic, through oxidation of reactive iron-bearing geologies as Acid Mine Drainage (AMD) (Klapper & Geller, 2002). Such acidic mine waters are often toxic to aquatic biota (Spry & Wiener, 1991; Doyle & Davies, 1999; Storer *et al.*, 2002; Stephens & Ingram, 2006). Pit lakes waters affected by salinity and acidity may also adversely influence nearby and regional groundwater resources and receiving environments, e.g., wetlands with contaminated plumes from flow-through pit lakes extending large distances down-gradient. The extent of such an impact may vary from insignificant in low hydraulic conductivity rocks and groundwater systems already saline, to considerable in high hydraulic conductivity rocks and naturally low-salinity groundwater environments (Commander *et al.*, 1994; Johnson & Wright, 2003). The majority of pit lake studies conducted in Australia have focussed on physical and chemical characteristics of water quality (Boland & Padovan, 2002; Jones *et al.*, 2008). These studies have demonstrated that pit lake water quality is influenced by many factors including climate, groundwater quality, depth, pit filling method and local mineralogy.

Many pit lakes contain high levels of acid, sulphate, and dissolved metals/metalloids. The chemical characteristics of a lake depend on the alkalinity of the local groundwater, the composition of the wall rocks, the chemistry of the surrounding vadose zone, and the quality and quantity of runoff from the surrounding land (Plumlee *et al.*, 1992; Davis *et al.*, 1993). Rock that is exposed to oxidizing conditions during dewatering can be a major source of acid, even though it lies below the water table before mining operations begin and after the lake fills (Miller *et al.*, 1996). The most common set of reactions producing acidity in mine lakes is the oxidation of sulphide and iron in pyrite (FeS_2) in the following two reactions (Castro *et al.*, 1999).

(1) $FeS_2 + 7/2\ O_{2(aq)} + H_2O \rightarrow Fe^{2+} + 2\ SO_4^{2-} + 2H^+$

(2) $Fe^{2+} + 1/4\ O_{2(aq)} + 5/2\ H_2O \rightarrow Fe(OH)_3 + 2H^+$

In natural systems pH is typically buffered by a carbonate buffer system (at pH of 6 to 8.5); however pit lakes of lower pH are often buffered by aluminium complexes (pH 4.5–5.5) or iron complexes (pH 2.0–4.0).

1.1 Australian pit lakes

Australia is among the top producers for many of the world's most important minerals (Mudd, 2007; Geoscience Australia, 2008). Major mining resources include diamonds, uranium, black coal, iron, gold, copper, lead, zinc, bauxite and mineral sands. Pit lakes occur in all states and territories in Australia However, most historic and contemporary mining activity is centred on the states of Western Australia (WA), Queensland and New South Wales (NSW) (Figure 2). Tasmania, Victoria, South Australia (SA) and Northern Territory (NT) are generally only important for certain minerals i.e., copper, gold, uranium, etc. (Mudd, 2007).

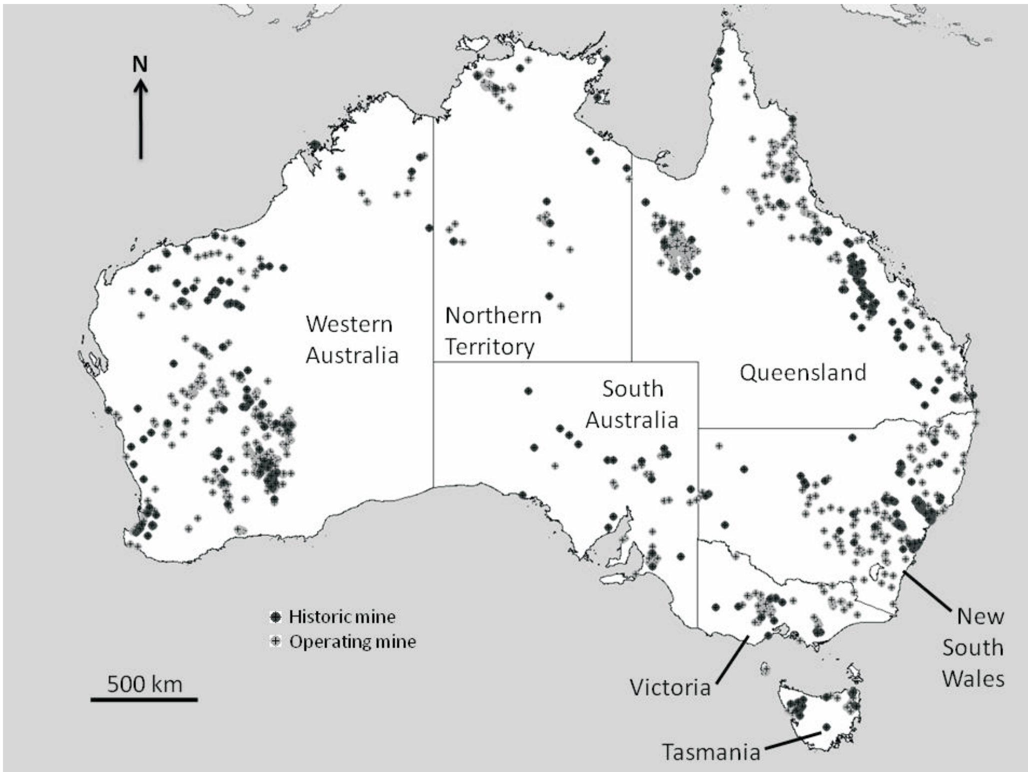

Figure 2. Distribution of historic and operating mines in Australia (after Kumar *et al.*, in press).

The mining areas also occur across a broad range of climatic regions (Figure 3). Approximately one-third of Australia is arid with rainfall less than 250 mm per year and another one third is semi-arid (250–500 mm per year). There are few areas where rainfall exceeds evaporation on an annual basis (Bell, 2001). Low rainfall and high evaporation rates exist in most parts of the country which may lead to net evaporation and the formation of hyper-saline pit lakes. Furthermore the groundwater in many parts of inland Australia is naturally brackish to hyper-saline. Low annual rainfall delays filling rates for new pit lakes facilitating oxidation of measures. A limited range of rivers and streams also limits opportunities for river rapid fill of pit lakes in many areas. However, surface discharge from pit lakes is also unlikely, which reduces a major source of environmental impact often seen in wetter climes. Contamination of regional groundwater in many arid areas can also often be a minimal risk as high evaporation rates ensure the pit lake remains a groundwater sink.

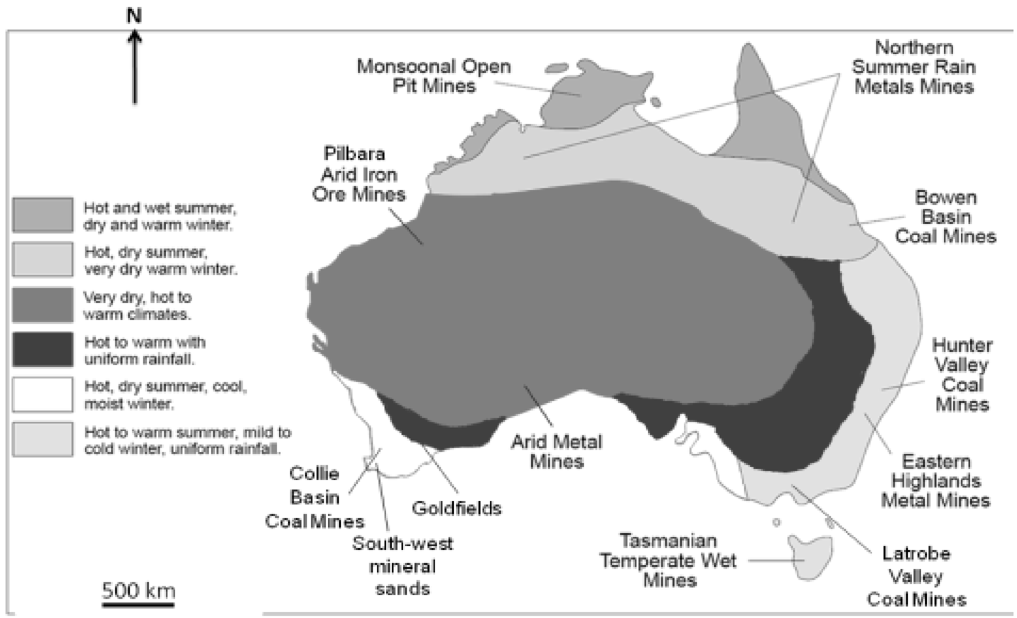

Figure 3. Australian pit lake classification after Mallet and Mark (1995), Johnson and Wright (2003) and (Kumar *et al.*, in press).

As one of the driest continents in the world and with the demand for water resources by industry and an increasing population, Australia may find pit lakes to be of significant potential use for both industry and surrounding communities (McCullough & Lund, 2006). It is not known how many pit lakes exist in Australia, since there is database for pit lakes at State or Commonwealth level. However, it was estimated in 2003 that there were 1,800 mine pits in Western Australia which potentially could form pit lakes (Johnson & Wright, 2003). Additionally, there are active or not-relinquished mining operations which add uncertainty to the number of pit lakes. Companies retain their leases over pit lakes with an option to over-mine as technology and economics alter the viability of their remaining resources.

A survey of mining operations in Australia found that 317 out of 517 mining operations contained potentially acid generating wastes (Harries, 1997). The same survey reported of the 176 mines that answered the questionnaire, 60 mines had water filled pits, but the pit lake water was similar to pre-mining groundwater. Nevertheless, seven sites had a total of 0.06 x 10^6 m^3 of acidic water at a pH of 2.5–3.5.

Australian pit lakes fall into four main categories in terms of their water quality. These are acidic (AMD affected), saline (can co-occur with AMD), neutral pH (but with some degree of contamination), and good water quality (but not necessarily comparable to natural regional water bodies) (Kumar *et al.*, in press).

1. **Acidic** – As examples, water quality of pit lakes of Collie (WA), Collinsville and Mt Morgan (both Queensland) are all degraded by AMD. Nevertheless, Collie pit lakes have low pH and toxic concentrations of Al primarily due to low buffering rather than high acidity inputs. Collinsville and Mt Morgan show similar classic AMD conditions of extremely low pH and very high metal concentrations. These latter pit lakes also show effects of ongoing salinisation.
2. **Saline** – In drier regions where net evaporation exceeds precipitation, and surface inflow to the pit is largely restricted to direct precipitation, can result in dramatic increases in salinity leading to brackish through to hyper-saline lakes. Such hyper-saline pit lakes of degraded value may also contaminate valuable regional groundwater resources in the future. For instance, in semi-arid regions such as the Collinsville region, high rates of evapo-concentration result in significant increases in pit lake salinity each year (McCullough *et al.*, 2008b).
3. **Neutral** – Mary Kathleen and Thalanga (Queensland), Ranger (Northern Territory) and Wedge Pit (WA) pit lakes have generally good water quality that is nevertheless contaminated by one or more metals; in these cases Cu, Zn, U and As respectively. Nevertheless, these pit lakes remain well suited to a variety of end-uses as individual contaminants can often be more readily remediated or treated than more complex pit lake chemistries. For example, As contaminated water is extracted from bores a few meters away from Wedge Pit, treated and used to supply potable water to Laverton.
4. **Good water quality** – Kemerton (WA) is a silica sand mining operation with few geological considerations or mining processes that result in contamination of pit lake waters, hence water quality is very good. However, there remain significant differences in lake shape and water quality compared to shallow naturally acidic wetlands nearby (McCullough & Lund, 2008).

1.1 Pit lake water quality over time

Water quality in pit lakes plays a dominant role in determining the range of end uses the lake can be used for (McCullough & Lund, 2006). The chosen end use will necessitate a certain water quality within the pit lake and remediation technologies will be needed in many cases to achieve the required end use water quality. Research is therefore required into water quality development in pit lakes by incorporating hydro-geological, limnological, biological and biogeochemical processes.

Current predictive models do not adequately account for sufficient of these processes for pit lakes to allow for useful predictions to be made (Jones, 1997). Instead, such models are likely to provide information for advancing current conceptual models and provide advice of pit lake response to different management scenarios (McCullough *et al.*, 2009). There are no Commonwealth or state guidelines for developing pit lakes as useful water resources. For instance, acidic and/or saline pit lakes influenced by AMD with acidic and metal contaminated water will need to be remediated using either chemical or biological methods (McCullough, 2007; McCullough *et al.*, 2008a; Neil *et al.*, 2009). Pit lakes contaminated with one or two metals but otherwise with good water quality can be used for a range of activities following chemical treatment such as selective precipitation. On the other hand, pit lakes with good water quality can be used immediately for uses such as aquaculture, water sports and recreation, etc. Even partial remediation of highly acidic and saline waters can allow this water to be used for activities such as dust suppression, potentially reducing demands on other higher quality water sources (McCullough & Lund, 2006). However, despite the potential and existing examples of possible beneficial end uses for pit lakes, there are many pit lakes across the Australian continent with no planned end uses (Farrell, 1998).

The potential use of pit lake water remains dependent on the pit lake water quantity and quality (Doupé & Lymbery, 2005). However, there is no central database of existing or future pit lakes currently available in Australia. There has also been very little research on pit lakes in general with a detailed literature review for this chapter producing little information. What published information that is available is typically in the form of *ad hoc* opportunistic studies across a diverse range of disciplines including environmental engineering, geology, chemistry and aquatic ecology. Although many State and Federal primary industry and environmental agencies do collate mining data, including sometimes those of pit lakes and their characteristics, these data are generally limited to current or only recently decommissioned pit lakes. Many Australian pit lakes are on un-relinquished mining leases. This situation makes the long-term acquisition of data required to study the evolution of the quantity and quality of pit lake water a very challenging exercise. Furthermore, it is suspected that many pit lakes are considered commercially sensitive and are therefore not generally available for sampling and data collection. Such lack of detailed data of pit lake water quantity and quality for many regions currently renders it impossible to assess the risk and opportunities presented by pit lakes to Australia. Moreover, there are no guidelines for ‘pit lakes’ at the level of Federal government to be followed. In the Federal government’s recent ‘Mine Rehabilitation Handbook’ guidelines (DITR, 2007) pit lakes are not mentioned.

1.1 Current study

Joint funded by the Department of Water, Western Australia and the Australian Government under its $12.9 billion Water for the Future plan, this project is focussed toward the management and use of pit lakes that have formed within the Collie Basin (the Collie Lakes District). The outcomes of this work are intended to support water resource planning and management in the Collie River catchment.

In late 2008, the Department of Water tendered a request for management of a research programme that would support and advise future water management in the Collie Basin in the south-west of Western Australia. A team lead by Edith Cowan University and comprising senior researchers from Mine Water and Environment Research Group (MiWER) and Centre for Ecosystem Management (CEM) at Edith Cowan University (ECU) and the School of Population Health, University of Western Australia (UWA) provided the successful tender for this research programme. This group of scientists have developed expertise in the area of environmental effects of mining over many years of specialist research and consultancy. Leading the mine water side of this research programme was Dr. C McCullough, Associate Professor

Mark Lund with Dr Lu Zhao of MiWER (Mine Water and Environment Research Group). Dr. Andrea Hinwood, Dr. Jane Heyworth and Mrs. Helen Tanner contributed considerable experience on human health issues and epidemiology to the health component of Task 2. All staff involved were successful researchers who have significant experience and a growing publication record in the mine water and environment and health area. The combined experience of the research team is unique within Australia.

The research programme activities were expected to run from March 2009 to May 2010. Altogether, 5 tasks were part of this research programme including:

1. Developing an inventory of pit lakes' data including history, storage, hydrology, water quality, water source and ecology and preparing a summary report that includes a preliminary assessment of end-use options for each pit lake and highlights gaps in existing data sets;
2. An assessment of the current effects of pit lakes on human health;
3. Development of a monitoring strategy for pit lakes and connected waters with special attention to those of the Collie Lakes District;
4. Production of a report outlining conceptual models of Environmental risk assessment, ecological limitations and health and grouping Collie pit lakes with regard to their geo-hydrology; and,
5. Geo-chemical modelling of water chemistry within pit lakes under different management scenarios to support management decisions.

This report fulfils Task 3 of this Collie Pit Lake research programme by developing a monitoring strategy for pit lakes, particularly designed toward the requirements of data collection from pit lakes within the Collie Basin. These data include hydrology, water quality, water source and ecology. Recommendations are also given as to how this data is analysed and reported. Knowledge gaps in existing monitoring strategy recommendations are indicated and recommendations are made into the continuous refinement of an ongoing monitoring programme for the 15 lakes in the Collie Lake District.

The purpose of this document is to recommend state-of-the-art monitoring design and sampling methodologies for environmental monitoring of pit lakes and their immediate catchments in the Collie Lake District, south-western Australia. This report gives an overview of regional and international environmental issues related to pit lakes, current national guidelines and best practice international operations and recommendations for monitoring pit lakes aquatic ecosystems. The purpose of monitoring selected indicators and their field sampling and analysis methods and techniques are described, and the practical temporal and spatial issues targeting episodic events are discussed in detail. Strategies for data analysis and reporting are also suggested for maximising data value and for enabling during further strategy development during long-term monitoring. Based on these general principles of monitoring, quality assurance, health and safety and budget recommendations are included as well.

The water quality and other environmental legacies of pit lakes following completion of mining operations is one of the most significant environmental issues facing the mining industry. The Collie region now has a Lake District of 15 pit lakes from historic (*ca*. 1960) and current open-cut mining activities. The current demand for water in the south-west of WA and its increasing scarcity means that Collie pit lakes represent a potentially valuable resource to both the environment and the community. Many of these lakes represent relatively good water quality that could be of risk to local and regional environments; and conversely of benefit to local communities if their environs are develop or managed to these ends (Zhao *et al.*, 2009). As a result, a monitoring strategy for these pit lakes is required in order to achieve more stringent demands on pit lake conditions at relinquishment made by state and federal regulation and the desired end uses of local communities (McCullough *et al.*, 2009).

Targeting the environmental issues specific for pit lakes, this report is divided in three main parts:

1) Introduction to cover a review of the status and environmental issues of pit lakes, its related guidelines and the purpose of a monitoring strategy for pit lakes;

2 The Collie Coal Basin

1.1 Background

The town of Collie (population over 10,000) is located on the north western rim of the Collie coal basin within the Collie River catchment. Collie lies nearly 160km south-southeast of Perth, and is the centre of coal mining industry in Western Australia (Figure 4). The major 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 tourism by the local business community and the Shire of Collie.

1.1 Geology

The Collie Basin covers an area of approximately 224 km^2, 27 km long by 13 km wide and elongating in a north-west to south-east direction. The basin consists of two lobe-shaped sub-basins, the Cardiff sub-basin (151 km^2) to the west and the Premier sub-basin (74 km^2) to the east, in part separated by a faulted basement high, known as the Stockton Ridge (Moncrieff, 1993).

The Collie coal basin is a small sedimentary basin occurring in the Collie River catchment (Figure 4; (CWAG, 1996)). The Basin contains up to 1400 m of Permian sedimentary rocks, covered by a thin layer of Cretaceous rocks. The base layer of pebbly mudstone is covered by layers of sandstone, shale and coal. There are up to 55 significant coal seams which are typically 1.5 to 5 m thick although the Hebe seam reaches 13 m thick glacial sediments and coal measures. There are an estimated 1,330 Mt of coal resource in the basin of which extractable reserves account for 480 Mt (Varma, 2002).

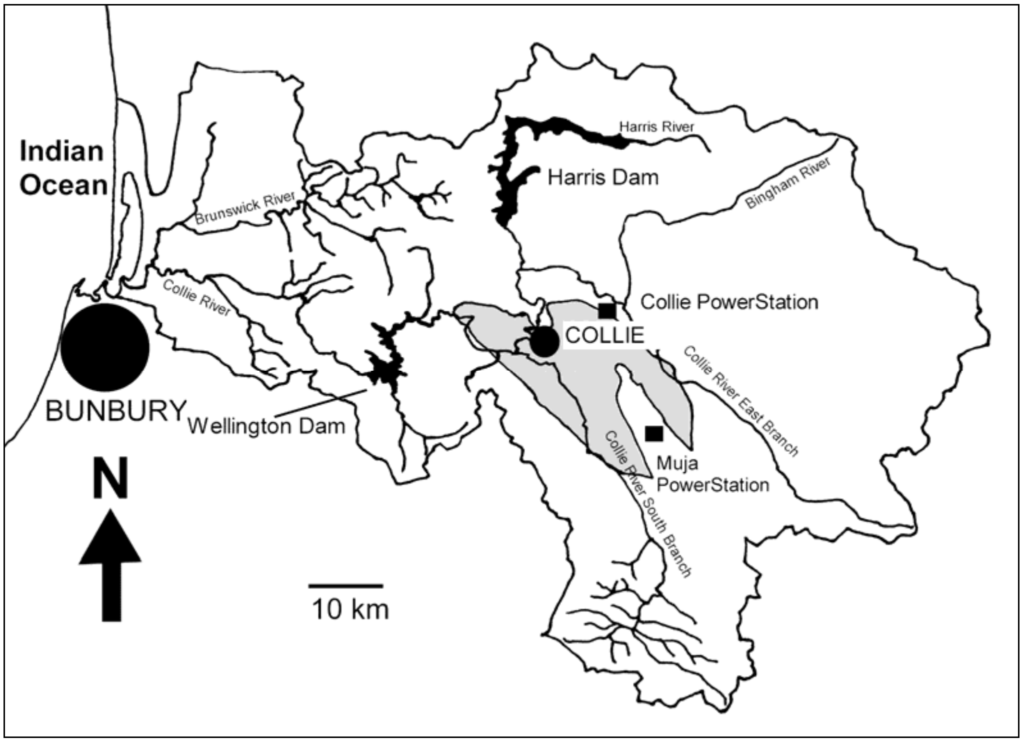

Figure 4. Location of the Collie Basin (after Neil *et al.* 2009).

1.1 Climate

Collie is located in the south-west of Western Australia. 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) (Commonwealth of Australia Bureau of Meteorology, 25/02/2009). Seventy-five percent of rainfall occurs in the five months from May to September (Figure 5). The 100 year mean annual rainfall for the Collie Basin is 939 mm, (Commonwealth of Australia Bureau of Meteorology, 25/02/2009) although this has decreased to an average of 690-840 mm over the past 20 years (Craven, 2003).

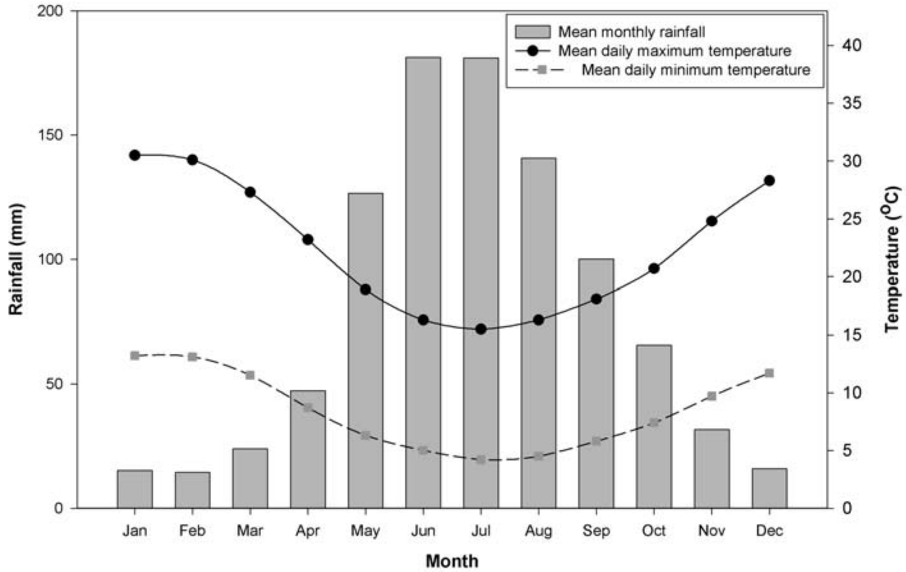

Figure 5. Mean temperature and rainfall climate of Collie (Commonwealth of Australia Bureau of Meteorology, 05/10/2005).

1.1 Groundwater

Groundwater resources of the Collie basin are fresh and discharge towards the Collie River, with seasonal fluctuations up to 1 m (Sappal *et al.*, 2000). The pH of groundwater is highly variable ranging from <4 to neutral (Varma, 2002).

Groundwater (in abstractable quantities) in the Collie basin is mainly contained within the sandstone of the Muja Coal Measures, Premier Coal Measures, Allanson Sandstone, Ewington Coal Measures and Westralia Sandstone of the Collie Group; within the sand and sandstone of the Nakina Formation; and in the surficial sediments (Varma, 2002). The hydrogeology of the Collie basin is complex, with multiple aquifers as a result of aquicludes and faulting (Varma, 2002).

1.1 Collie River

The Collie River is the main river system of the Collie basin, running almost 100 km westward to the Indian Ocean. It was once fresh but due to clearing of the upper catchment for agriculture, the salinity has risen to over 1000 mg L^{-1} (Mauger *et al.*, 2001). Total phosphorus levels were recorded at over 18 µg L^{-1} in July of 2004 (Salmon, UWA, unpublished data). The south branch of the river was diverted around the former WO5B (Lake Kepwari) mine pit during operations and has been used to fill the void when winter flows were sufficient.

Wellington dam was built on the Collie river, 35 km from the Collie townsite, in 1933 as a source for irrigation for the coastal plain (Mauger *et al.*, 2001). The dam was raised to its current capacity of 185 GL in 1960 and used for drinking water. Rising salinity in the river meant the dam was no longer suitable for drinking water and was replaced in this capacity by the Harris dam in 1989 (Mauger *et al.*, 2001).

1.1 Mining in Collie

Underground and open cut coal mining has taken place in the Collie basin since 1898. Until the mid 1990's coal mining was predominantly in the Cardiff sub-basin. In 1997 mining in the Cardiff sub-basin ceased and since then mining has taken place in the Premier sub-basin at the Muja, Ewington and Premier mines. The history of Collie coal mining is detailed in Stedman (1988). As a result of a dispute with the Government, six open cut pits were abandoned in 1950s and 1960s, which went on to form Stockton Lake, Ewington Lake, Blue Waters, Black Diamond (A & B) and Wallsend (used for landfill) (Figure 6).

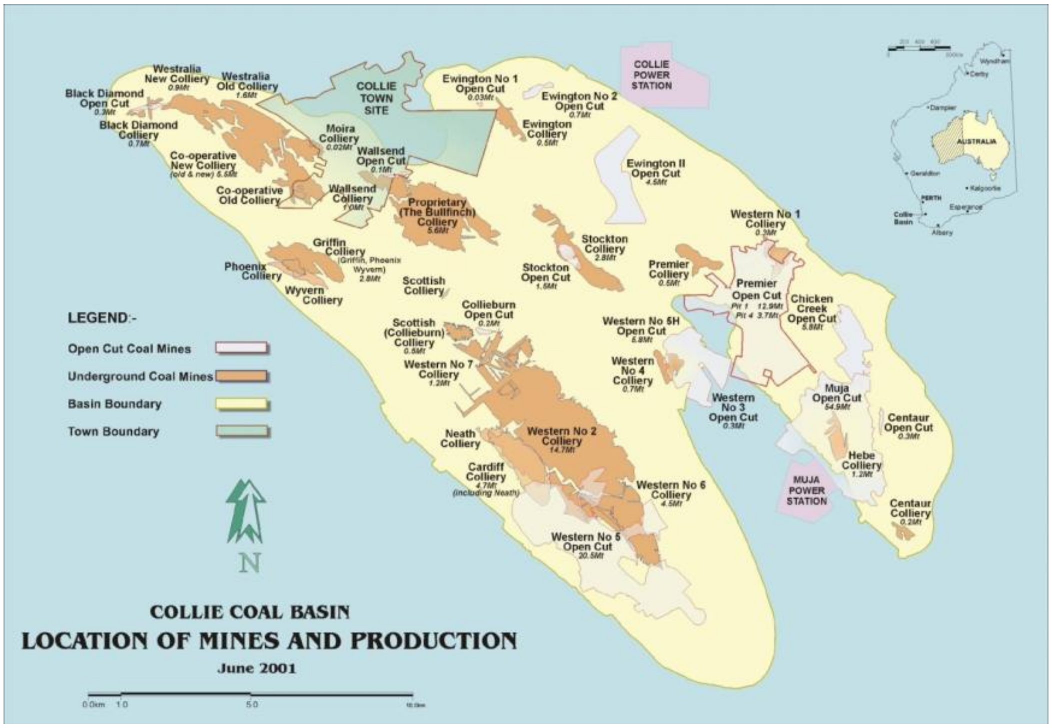

Figure 6. Historical mine workings in the Collie Basin (source unknown).

Currently two mining companies (Wesfarmers Premier Coal Pty Ltd and Griffin Coal Pty Ltd) have active mines in the Premier sub-basin. Wesfarmers Premier Coal Pty Ltd is currently rehabilitating or developing end uses for finished pits in the Cardiff sub-basin (Figure 7).

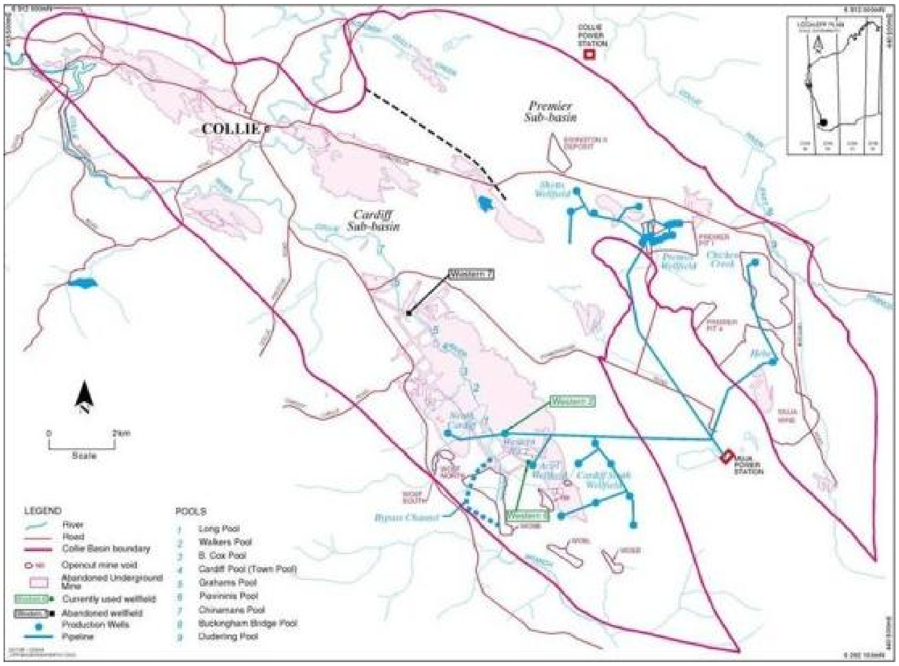

Figure 7. Current mining activities in the Collie Basin (source unknown).

1.1 Collie Pit Lakes

There are more than 15 mine lakes in Collie, with surface area between 1–10 ha, depth between 10–70 m, age between 1–50 years and pH 2.4–6.8 (Figure 8). Water quality of pit lakes of Collie is degraded by AMD, mainly in terms of low pH and elevated concentrations of selected metals.

Collie black coal has low sulphur concentrations (0.3–1%) {Le Blanc Smith, 1993 #164} and only produces low amounts of acidity through pyrite oxidation, ferrolysis and secondary mineralization. This low acidity is still sufficient to generate low pH in pit lakes due to low buffering capacity of surrounding geologies. These pit lakes also have very low nutrient concentrations of carbon, particularly in historic lakes where it may be at detection level of <1 mg L^{-1} {Zhao, 2009 #764}. The few ecological studies made on Collie pit lakes highlight nutrient limitation restricting algal productivity and hence lake foodwebs (Lund *et al.*, 2000; Lund *et al.*, 2006; Thomas & John, 2006; Salmon *et al.*, 2008).

Figure 8. Location of current Collie pit lakes (sourced from Google Earth). Note: Wellington Dam is a reservoir.

3 Pit Lake numerical modelling

Pit lakes are both potential water resources and potential environmental risks and, as such, raise significant environmental issues for the mining industry. Increased social expectation, such as legislation and regulation and desired end uses by local communities, are increasingly requiring higher standards of environmental assessment and management for pit lakes.

Given the long-term risks for contamination, mining companies and regulatory agencies rely on geochemical predictions of the future pit lake water quality, aiming to develop closure strategies that can provide sustainable solutions to protect the surrounding environment and groundwater resources. Model predictions should not be regarded as perfectly accurate due to inherent uncertainties in input datasets and in understandings of fundamental lake geo-chemical processes such as climatic interactions, wall-rock interactions and groundwater flow characteristics. However empirical models still allow us to integrate complex physical, chemical and biological processes that interact across multiple time and space scales. They also can provide excellent quantitative estimates of outcomes under different environmental and operational scenarios.

Many studies have been carried out in recent years to better understand the processes that influence the geochemistry of pit lake water (Miller *et al.*, 1996; Tempel *et al.*, 2000). Studies on long- and short-term water quality trends in pit lakes have identified important physical, geochemical, and biological processes that control their composition (Kalin *et al.*, 2001; Boland & Padovan, 2002).

There are several modelling strategies for mine lakes, hydrological modelling focusing on the input and discharge of ground water and surface water (Niccoli, 2009); physical limnology modelling for pit lake stratification and circulation (Hamblin et al., 1999; Castendyk & Webster-Brown, 2007); geochemical modelling on pit lake geochemistry and water quality (Eary, 1999); and biological modelling on ecological community structure (Kalin et al., 2001; Jin & Bethke, 2005). Numerical lake models integrate multiple processes influencing lake stratification and turnover. Nevertheless, a considerable amount of both pit lakes and often also relevant catchment data is needed to produce accurate results for modelling prediction in these disciplines.

Water quality in a pit lake is determined by the varying proportions and chemistry of both groundwater and surface water sources flowing into the pit lake, and is combined with internal hydrochemical processes and limnological processes in the filling lake. There are many physical processes which control pit lake hydrodynamics. These include the shape, orientation of the lake, and climatic conditions at the site (Miller *et al.*, 1996; Huber *et al.*, 2008). Incoming solar and outgoing evaporation and radiation contributes to a heat balance control lake water temperature and therefore the density in freshwater systems, affect mixing and stratification, as will wind stress across the pit lake surface, transferring energy to depth. In turn, the hydrological pit lake balance is determined by lake depth and volume and relative inputs of water (e.g., precipitation, surface water and ground water influx) and water removal (e.g., evaporation, surface and ground water efflux, and abstraction) (Salmon *et al.*, 2008). Therefore, climate (temperature, precipitation, runoff, evaporation and wind), lake morphometry (depth, volume and catchment area), and the volume and chemistry of groundwater and surface water fluxes are required as background information. Changing chemical conditions over depth, such as the levels of dissolved oxygen, carbon dioxide, salinity, total dissolved solids (TDS), pH and redox potential, either due to gradients or thermo- haloclines, also need to be measured with frequency. Nutrient levels (N, P and C) along with other elements of water quality play an important role in determining the pit lake's biological ecological character with respect to productivity and community structure. Metal and metalloid concentrations are typically the main concern for water quality in pit lakes; whereas some metals are essential for ecological development (e.g., Ca, Mg and K), some metals are the control factors for acid produced oxidation and buffer of pH (i.e., Fe and Al) and toxic trace metals may be of concern for human health and environmental risk (e.g., heavy metals). Therefore, modelling pit lake water quality requires hydro-geological, limnological, and biogeochemical processes all to be considered.

In recent work of the Mine Water and Environment Research Group (MiWER), based at Edith Cowan University, Western Australia (Zhao *et al.*, 2009), a pit lake water quality database was compiled in a project joint funded by the Department of Water, Western Australia and the Australian Government under its $12.9 billion *Water for the Future* plan. This database was used in this current study to derive initial and boundary conditions for lake water quality modelling of representative pi lake types. Furthermore, modelled physical and chemical conditions were then validated to data from this database.

4 Modelling Software

4.1 Software history

The numerical modelling software used in this project has been developed since 2000. It aims to reflect the determining physical, chemical, and biological processes of acidic mining pit lakes. Figure 9 gives a schematic overview of important processes in acidic pit lakes that it attempts to model. Often pit lakes are fed by groundwater that may carry considerable amounts of acid mine drainage (AMD). Other sources of acidity are erosion materials from banks transported by rain water or mobilised by wind waves. Furthermore, waste rock and tailing often constitute larger component of the geology of the bottom of pit lakes. These materials can release substantial amounts of substance that can cause acidic reaction in the lake water. Circulation processes in the lake are driven by wind and influenced by temperature- and possibly concentration-caused stratification. Water quality in the lake is determined by chemical reactions and biological processes such as algal growth and nutrient cycles. All of these processes interact with each other, forming a complex feedback system.

In order to model this pit lake system, modelling knowledge from different scientific domains such as groundwater, lake circulation, hydrochemistry, and limnology needs to be combined. The modelling system MODGLUE couples the groundwater flow and transport model PCGEOFIM with the lake circulation water quality model CE-QUAL-W2 and the hydrochemical model PHREEQC. In addition, a simple erosion model provides AMD inputs from banks. Another feature of MODGLUE is the possibility to model the adding of substances to the lake. This allows one to model oxygen addition to the hypolimnion or chemical treatment of lake water to neutralize it. A mechanism called leaching allows modelling the release of substance from the lake bottom that can cause an increase of acuity in the lake water.

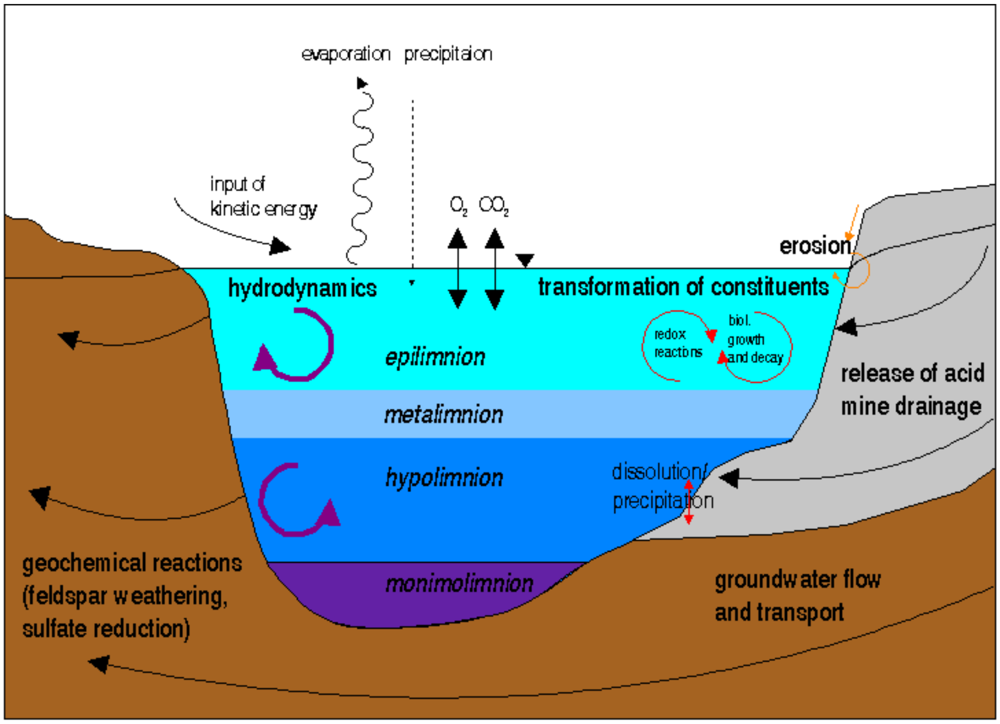

Figure 9. Important processes required for modelling in pit lakes.

4.2 Groundwater model: PCGEOFIM

PCGEOFIM (Sames et al., 2005)(Müller *et al.*, 2003) is a finite volume groundwater flow and transport model that was specifically designed for mining and post-mining areas more than 35 years ago. It provides some special features to account for mining-specific conditions. Subsurface parameters can be specified as time-dependent allowing for modelling of excavation of mine pits, filling with overburden, and creation of lakes, all in one model run. While working with a regular grid, multiple nested grid refinements that may overlap can be used to get higher resolution in regions of special interests. The PCGEOFIM model offers many ways to specify groundwater recharge. For example, keeping groundwater recharge constant over time, having it depend on groundwater levels, or getting it from a sophisticated coupling with a rainfall-runoff-soil-water-budget model.

PCGEOFIM provides a simple yet very useful mechanism to account for the interactions between lakes and groundwater. The lake is represented as a water level volume relationship, budgeting all in- and out-flows, such as groundwater and rivers. Precipitation and evaporation yields a new lake water volume and hence a new water level. This water level is used as the head for Cauchy boundary conditions that act jointly as "the lake". Rivers can also be represented by several Cauchy boundary conditions that act jointly while the discharge is calculate by Manning's formula. Many special boundary conditions, such as vertical and horizontal multi-level wells, defined outflow levels of lakes as well as sophisticate connections between rivers, lakes, and pipelines with control mechanisms can be used to provide a high level of representation of the real system.

4.3 Groundwater model coupling

MODGLUE can work without feedback to a groundwater model taking only specified inflows and outflows as input data. Therefore, MODGLUE can work with results input from other groundwater models (e.g., the Collie regional groundwater model under development) using this off-line approach. Furthermore, the online coupling with PCGEOFIM is designed as a loose coupling: only spatially distributed inflow and outflow fluxes are exchanged at every time step. These fluxes can be provided by a different groundwater model than PCEGOFIM, such as the SKM FEFLOW model. For instance the calculation results from MODFLOW LAK package or form FEFLOW via IFM can even be exchanged.

4.4 Lake model: CE-QUAL-W2

Deep lakes typically experience seasonal (or permanent) density stratification. Under Australian conditions, lakes stratify over the summer months, as surface waters become heated by solar radiation. Over the winter months, cooler air temperatures cool surface waters and winds drive convective overturn of the water body. Strong winds associated with episodic storm events (or cyclones) can input kinetic energy that can overturn this density stratification. Long-term evaporative losses can exacerbate saline conditions in the lake; salinity-driven long-term stratification may dominate over seasonal temperature-driven stratification. These stratification cycles impact on geochemical processes, including acidification of lake waters and in particular potential remediation processes such as sulfate reduction (Totsche *et al.*, 2006) and primary production (Beulker *et al.*, 2004).

Therefore, the ability to describe and predict long-term stratification cycles in the lake is critical to predict long-term water quality; numerical models of lake hydrodynamics allow the prediction of stratification dynamics. An effective hydrodynamic lake model must be able to capture all of the physical processes that govern the structure of the water column. CE-QUAL-W2 (Cole & Buchak, 1995) is a two-dimensional finite difference model for lakes and reservoirs that calculates flow, transport, and limnological water quality. CE-QUAL-W2 solves the Navier-Stokes-Equation with a large eddy diffusion approach, accounting for kinetic energy introduced by wind and density-driven flow determined by temperature and solution concentrations. It is widely used to model lake stratification and water quality of lakes, reservoirs, and rivers all over the world. PSU (2008) lists about 2,400 of its applications worldwide. Many water-quality determining processes, such as algal growth, nutrient cycle, and oxygen concentration can be modelled. However, CE-QUAL-W2 describes inorganic chemical processes insufficiently for modelling AMD effects. An example how a lake is represented with the finite difference grid in this modelling is shown in Figure 10.

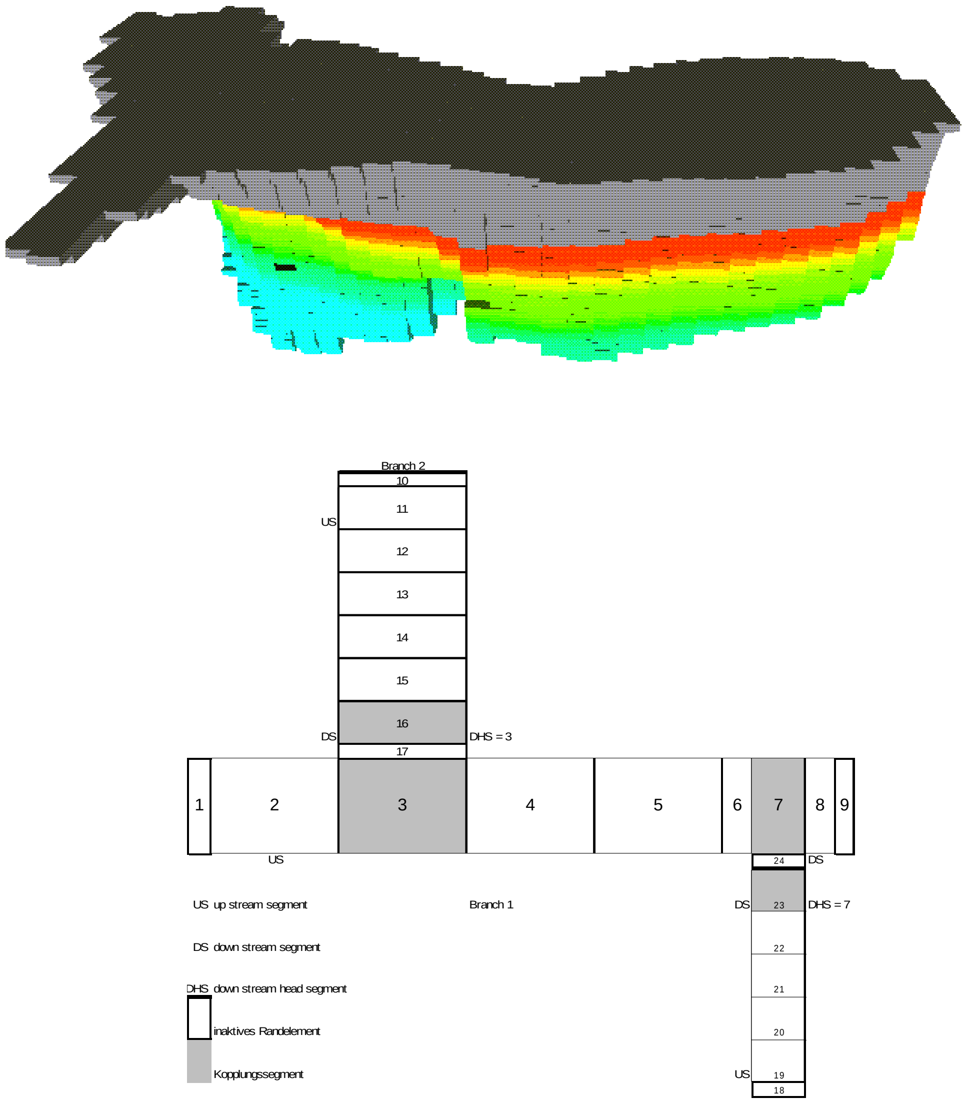

Figure 10. Example discretisation of a lake and its representation in the model grid.

4.5 Hydrochemical model: PHREEQC

PHREEQC (Parkhurst & Appelo, 1999) is probably the most widely used hydrochemical model that allows one to represent a wide range of chemical reactions. Nearly all inorganic and organic chemical reactions can be modelled with the help of its built-in programming language. The database of chemical species can be extended as needed for the representation of the hydro-geochemical system at hand. This wide capacity allows the use of this model across a wide variety of lake types.

4.6 Coupled model: MODGLUE

The development of MODGLUE (MODel for Prediction of Groundwater and Erosion influenced Lake Water Quality Using Existing Models) was begun in 2000 (Müller, 2004). The coupled model includes spatially distributed exchange of water flow and flux between groundwater and lake, limnological and chemical water quality changes as well as effects of eroded materials on water quality. The model is capable of representing all major influences on water quality and their interactions. It has been successfully applied to several lakes in Germany for prediction of water quality and evaluation of effects of lake treatments. The water quality of the pit lake Bärwalde has been predicted (Müller & Werner, 2004; Werner *et al.*, 2008). A new groundwater model capable of density-driven flow MODMST (Boy *et al.*, 2001) has been coupled to MODGLUE and used for water quality modelling of Lake Großkayna (Müller & Werner, 2003). Deep water aeration has been incorporated into the model with a new source-sink term that adds or subtracts substances at a given location with a provided schedule. Water quality predictions were conducted at other lakes in the internationally significant central German mining region including Lake Zwenkau, Lake Bockwitz, and Lake Hain (Müller, 2005). For this, the model was again modified to allow treatment of acidic lake water with alkaline substances. Table 1 gives an overview of some of the different pit lakes to date where MODGLUE has been applied, along with a short description of the task it modelled for.

Figure 11 shows the architecture of MODGLUE. The object-oriented programming is illustrated on the left. These existing models were wrapped with help of the Python programming language, applying the adapter pattern; this was then used as a component for MODGLUE, applying the façade pattern. This leads to a flexible setup, facilitating understanding and extension of the system.

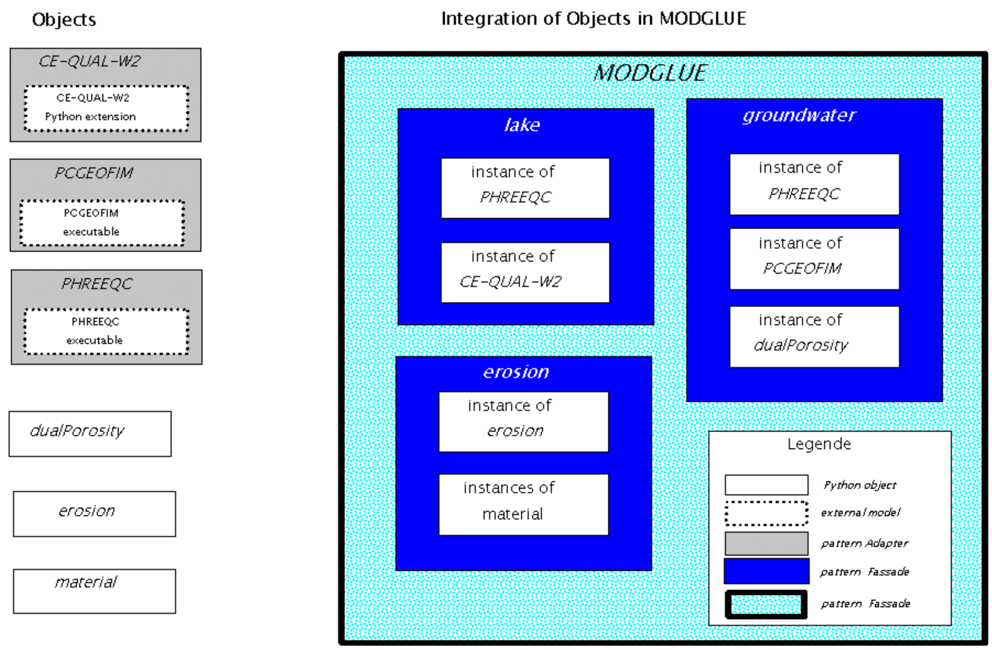

Figure 11. Components of MODGLUE and their integration in the model.

Table 1. Sites where geo-chemical modelling with MODGLUE has been applied (Germany).

Site	Task
Lake Cospuden	Hydrodynamic modelling
Lake Bärwalde	Water quality prediction
Lake Bockwitz	Water quality treatment evaluation
Lake-System Hain-Kahnsdorf	Hydraulic shortcut investigations
Lake Zwenkau	Water quality prediction
Lake Runstädt	Density driven groundwater in and outflow, long term water quality prediction

4.7 Iron and aluminium oxidation

Iron and aluminium react with oxygen and precipitate as hydroxides. Iron oxidation can be model as kinetic reaction depending on pH. As described in Salmon and Malstrom (2000), three different types of reactions: (1) homogenous abiotic, (2) heterogeneous abiotic (surface binding), and (3) microbial catalytic oxidations can distinguished for iron oxidation. In MODGLUE all three reaction types are represented. Homogeneous and heterogeneous abiotic reactions are model as independent form proton activity for pH < 2, reaction of first order for $2 \leq$ pH ≤ 5 and reaction o second order for $5 \leq$ pH ≤ 8 (Salmon & Malmström, 2000). Biological catalytic reactions are model based on Pesic (1989). Aluminium is model equilibrium reaction where either $Al(OH)_3$ or Gibbsite can be formed and precipitated. Precipitated materials are removed form the system.

4.8 Gas exchange with atmosphere

Oxygen and carbon dioxide are exchange with the atmosphere in upper most active model layer using a diffusion term that uses the saturation and the atmospheric partial pressure of each gas. Saturation pressures are determined as function of water temperature. For carbon dioxide the saturation pressure is also a function of lake water pH. Therefore, low inorganic carbon concentrations will result if the pH is low.

4.9 Redox reactions

Redox reactions can be model as equilibrium reactions or as kinetic reactions using PHREEQC*s BASIC routines. Routines for reduction of iron III, manganese III and sulphate as well as oxidation of sulfide, iron II, manganese II have been implemented and can be turned on or off.

4.10 Phosphate binding

Phosphorus has been identified a an important limiting nutrient in Collie pit lakes (Lund *et al.*, 2006; Lund & McCullough, 2009). Phosphate is binding and co-precipitating with iron hydroxide in a ratio of 1:10 (Klapper, 1992). A lower limit of phosphate concentration can be specified to prevent total depletion.

4.11 Photosynthesis

The representation of photosynthesis has been extended in MODGLUE to consider inorganic carbon a potential limiting factor in addition to phosphate, nitrogen and light. This allows for carbon limitation of photosynthesis for low pH waters that lose carbon through exchange with the atmosphere as described in section 4.8.

4.12 Interaction of chemical and biological processes

Chemical and biological reactions interacted together in MODGLUE in synergistic and antagonistic manners. For example, organic carbon created by photosynthesis decays and acted as electron donors for kinetic redox reactions. Conversely, photosynthesis was limited by a lack of inorganic carbon caused by chemically determined exchange with the atmosphere as described in section 4.11.

4.13 Representation of acidity and alkalinity sources

MODGLUE considered the following sources of constituents providing acidity or alkalinity:

- tributary inflows of river water at specified locations,
- outflows to rivers at specified locations,
- groundwater inflows and outflows with given spatial distribution,

- wind wave caused bank erosion,
- rainfall caused erosion of lake surrounding areas,
- release from sediments as function of time or a function of lake water quality (leaching).

Once the initial lake water and all sources were charge balanced, pH was calculated by the PHREEQC model component. Frequently, species concentrations obtained from measurements had charge balance errors. This was either due to inherent inaccuracies in chemical analysis or missing ions from the analysis or a combination of both. Therefore, all water compositions were charge balanced before they were used as input for the lake model. Usually, one species with high concentration were used from charge balancing. If the initial charge balance error was significant the concentration of the charge balancing species might change considerably. This was taken into account when interpreting results. Alternatively, a large initial charge balance error indicated that either some major ions missing from the analysis or that there was significant error in the determined concentrations.

Surface and groundwater concentrations were supplied as input data after applying the charge balancing procedure. Sinks due to surface and groundwater outflows were fluxes that consist of the outflow rate and the current lake water quality. The sources erosion and leaching were not directly specified but rather calculated within the model because they were dependent on the current state of the lake such as lake water quality, lake water level and time.

4.14 Erosion as constituent sources

Eroded material from lake catchments and littoral margins can bring considerable amounts of acidity into a lake. The effect of catchment erosion as a pit lake acidity source was modelled in two steps (1) determining of eroded mass and (2) elution of substances from this material.

Erosion caused by rainfall and runoff from lake catchments was calculated with a specified rate of mass per area and time. The active area was determined from the total lake area minus the area of the water filled portion of the lake. This allowed the effect of rising water table to be taken into account because the higher the water table the smaller the active area available for rain erosion.

Erosion caused by wind-driven wave action was calculated depending on the expansion of the lake. A water level rise causes an increase in lake surface area. For this area now covered with water, a mass of eroded catchment material is calculated from specified effective depth and the density of the eroded material.

The areas for both types of mass erosion were divided and sub areas with different properties. Based on the calculated masses form each subarea and the properties of pore water and cation exchange capacity a source term of substance was calculated. This is done depending from the current water quality suing PHREEQC.

4.15 Leaching as constituent source

The term leaching, as used here, describes the release of substances such as metal or nutrients from the lake sediments. This approach is based on observations and laboratory experiments carried out for Lake Zwenkau in Germany. Mining dumps of waste rock and tailings placed at the pit lake bottom often contain substantial amounts of substance that can cause acidic reactions if released into the lake water. Laboratory experiments with samples from different locations of Lake Zwenkau show release rates that depend on time and water quality of overlaying water. Typically, these demonstrate high initial rates decrease quickly with a long tail of lower rates.

MODGLUE calculated spatially distributed leaching of substance from sediments. The lake surface areas were divided into as many sections as needed. Each sections was then characterised by a specified leaching rate and a composition of substance that are leached. Each section also consisted of model cells that were explicitly specified and a leaching area that is also given as input. Depending on the water level, the specified cells were active or inactive. As soon as they were below the water table, they were activated and deactivated if they again became dry. Each cell had its own time counter so that the time dependent rate is calculated individually for each cell. This rate was multiplied by the active cell area resulting in the amount released substance. Each section also had an associated composition that specified what amount of which chemical species was added for each leached mole of substance.

This algorithm provided a dynamic source of substance that influenced the acidity of the lake water. The sections accounted for different areas of the lake such as waste dumps and natural subsurface materials such as varying bed geologies. Only cells that were under water leached substances and each cell had its individual age and therefore leaching rate. The leaching model has also been successfully applied to Lake Zwenkau where MODEGLUE was able to demonstrate leaching was the primary input source of acidity.

4.16 Treatment of lake water quality

The treatment of the lake water quality was also modelled by adding specified substances with given time-dependent rates at as many model cells as required. For instance this allowed us to introduce oxygen in the hypolimnion modelling the effect of deep lake aeration during water column mixing column and lake over turn or a storm event. Other future applications could include additions of alkaline chemical such as lime stone as calcite in the upper layer of a lake as would be conducted using a boat or land-based water quality remediation installation (Lund & McCullough, 2009).

5 Classification of Lakes

5.1 Motivation

Conceptual modelling of Collie Lake District pit lakes identified three major lake types (McCullough & Lund, 2010). These several lakes all showed similar water quality and environmental characteristics and therefore are expected to show similar developmental behaviour. These three basic lakes types included: Historic pit lakes (around 50 years old), New and Rehabilitated pit lakes (around 5–15 years old) and New and Un- rehabilitated pit lakes (around 5–10 years old, but with revegetation <5 years old) (Figure 12). Differences between pit lakes were predominantly due to higher pH and lower ORP in historic pit lakes, high salinity in rehabilitated pit lakes and lower salinity and pH in un-rehabilitated pit lakes. Modelling was made for a single example of each of the three lake types previously identified by conceptual modelling (Table 2). These lakes were each chosen for well-representing their respective lake types and also for having a good dataset to best enabling modelling accuracy.

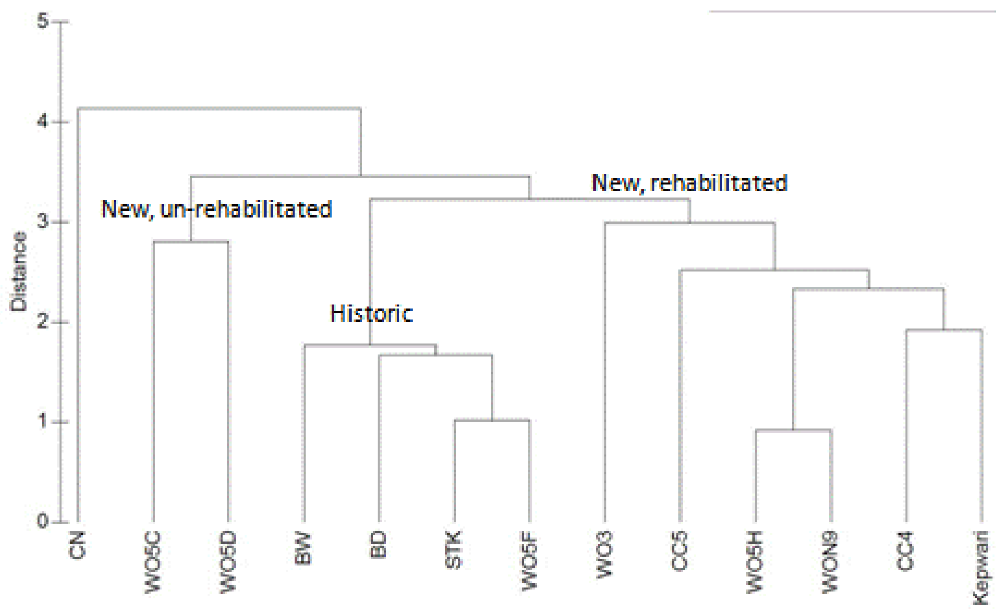

Figure 12. Cluster diagram of conceptual model of individual Collie pit lakes ((McCullough & Lund, 2010)).

Table 2. Lake types identified by conceptual modelling and their examples chosen for modelling.

Lake type	Representative lake modelled
New, rehabilitated	Lake Kepwari
Historic	Lake Stockton
New, rehabilitated	Lake WO5C

6 Model Establishment

6.1 Meteorological data

Metrological data are important for characterising the upper boundary of the lake. The Collie East weather station (ID 009994) is located close to the lakes under investigation (latitude: -33.36, longitude: 116.17). Hourly measurements were valuable starting from December 2002 until March 2010. In total about 65,400 dataset were used (Commonwealth of Australia Bureau of Meteorology, 25/03/2010). A few missing values (about 100) were surrogated by the values from the hour before. Some calculations, especially those for reproducing the rise of the watertable, started before 2002. For these years, starting with 1997, data from other years were used as presented in Table 3.

Table 3. Reassigment of meteorological data for missing years.

Original year of meteorological data	Reassigned year of meteorological data
2005	1997
2006	1998
2007	1999
2008	2000
2009	2001
2003	2002

Hourly values for air temperature, dew point temperature, wind speed, wind, direction, precipitation, and cloud cover were used as input data for the lake models. All data except that for the cloud cover could be taken form the Collie East station data. Cloud cover was calculated from precipitation data. Hours with precipitation were assigned a full cloud cover, whereas hours without precipitation were assigned no cloud cover.

6.2 Groundwater exchange

The water quality of the dominant water sources must be well described to allow the development of a robust lake water quality model. The water quality of surface inflows to any lake is strongly determined by their flow paths. This is especially so when those flow paths may place the water in contact with reactive minerals such as in the case of many of the Collie pit lakes where pyrite containing backfill may often constitute the immediate pit catchment and walls.

The hydrology of the region was described by Varma (2002) who considered that groundwater was a very important source of influx for the Collie lakes. A numerical groundwater model for the Collie Basin based on MODFLOW was build by Zhang *et al.* (2007). Recently, a FEFLOW-based groundwater flow has been developed and first results are becoming available (personal communication, Blair Thornburrow, SKM, New Zealand). The FELOW-based model is a successor of the MODFLOW-based model using much of the same input data but at the same time taking advantage of the greater flexibility of the FEM mesh of FEFLOW. This yields a better spatial representation of smaller features such as rivers without running the risk of producing narrow, elongate cells that tend to produce numerical instabilities as it is the case with MODFLOW.

Lakes are represented by either general head boundary or constant head boundary conditions in both groundwater models. This mean the lake water level is specified as input before the model run and the inflow to the lake is calculated with the groundwater model. Zhang *et al.* (2007) used seven iteration steps adjusting the level of Lake Kepwari (WO5B) to match measured and calculated lake water levels. This was achieved by a spreadsheet-based lake budget model that run alternating with groundwater model using the output of one model as the input for the other.

The groundwater inflows to Lake Kepwari that were calculated with the three different groundwater models are shown in Figure 13. Since results from the

FEFLOW-based model were only available starting from 2010, they were projected back to start in 1997, i.e., using the data from January 1st 2010 for January 1st 1997. Even with this shift in time of the data, the inflows differ rather little and show derivation only over longer time periods.

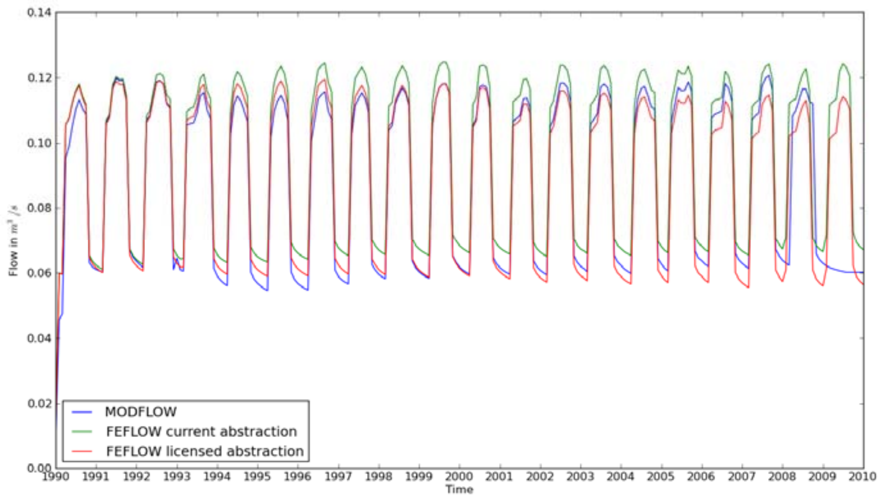

Figure 13. Groundwater inflow to Lake Kepwari as calculated with the MODFLOW-based model and the FEFFLOW-based model with current and licensed abstraction (FEFLOW inflows actually start in 2010 but were projected back to 1997).

6.3 Lake Kepwari

Lake Kepwari is an elongated lake with primary north-south extension with wider area in the north (Zhao *et al.*, 2009). The bathymetry of the lake can be seen in Figure 14. Especially the eastern side is characterised by steep slopes. The final lake is more than 75 m deep.

Based on this bathymetry, a discretisation of the lake body was performed to setup a two-dimensional model that can be used for hydrodynamic and water quality modelling. The result can be seen in Figure 15.

Vertically, the lake volume was divided in one-meter-layers starting form 110 m AHD to 200 m AHD. The upper layers will be dry for most scenario runs, but are included to allow for later scenarios with possible higher lake water levels.

Horizontally, the domain was divided into 13 active segments with a length of 200 m each. The segments start in the north and extend towards the south. Gray cells are inactive and will never be part of any simulation but are used internally because the model grid stored as an array in the simulation software.

Each cell has an individual width assigned to it. Therefore, multiplying layer height, segment length and cell width yields a cell volume. The sum of all cell volumes corresponds with the volume of the lake calculated with other software tools for spatial analysis.

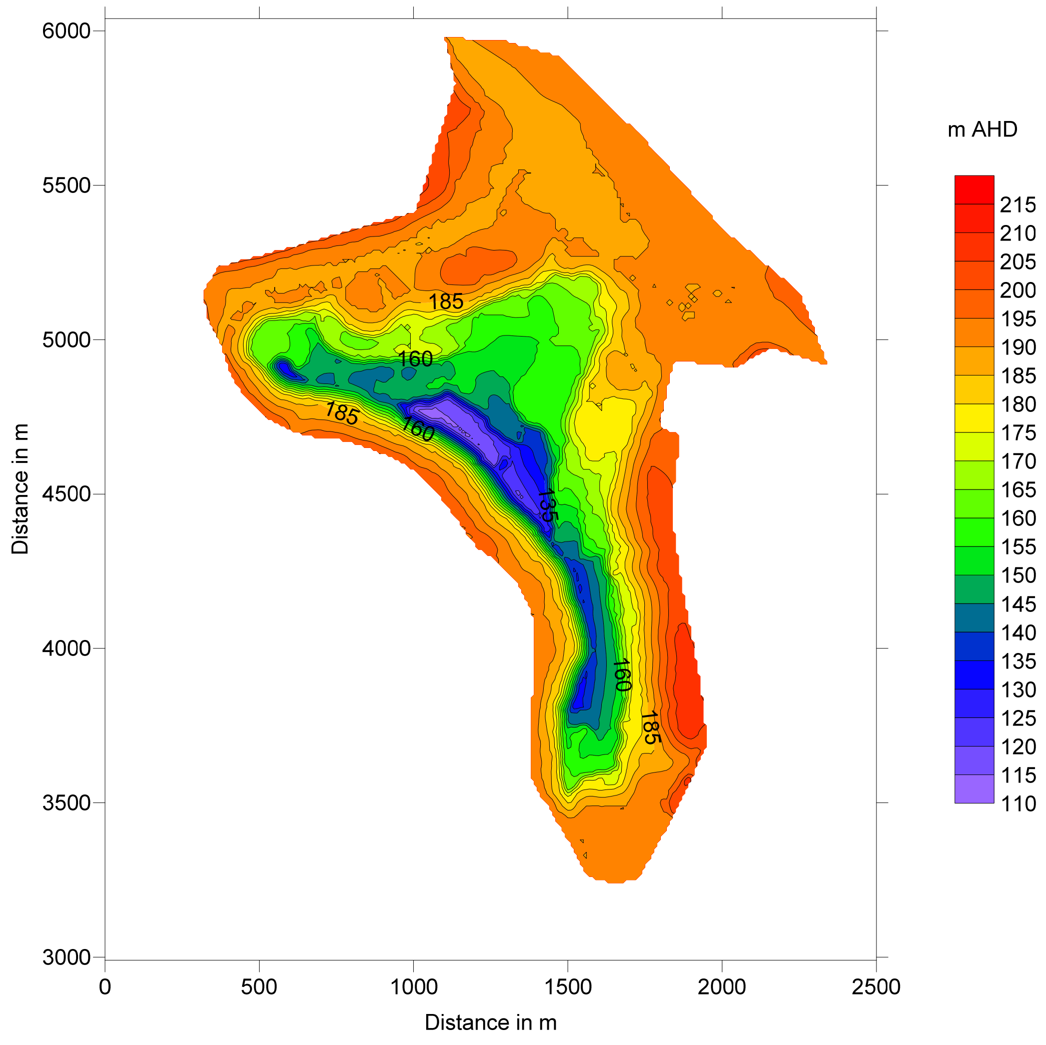

Figure 14. Bathymetry of Lake Kepwari.

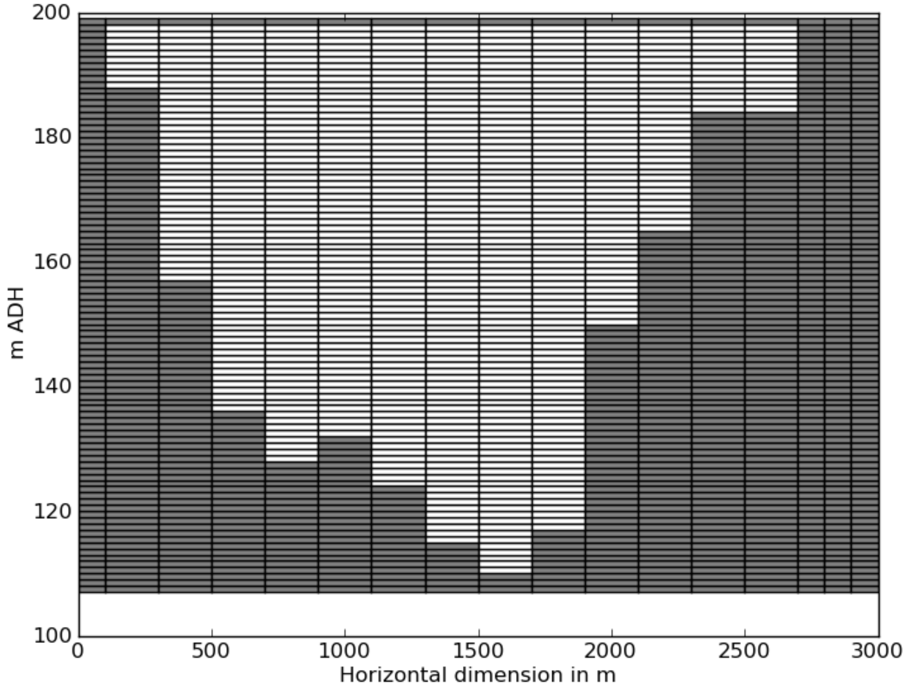

Figure 15. Discretisation for the Model of Lake Kepwari (north to south, white cells are active, gray cells inactive).

6.4 Lake Stockton

No historic pit morphology or more recent bathymetry data were available for Lake Stockton. Therefore, bathymetry was *ad hoc* mapped for this exercise by measurements of lake bottom height by boat and location of the lake shore were carried in April 2010. Emphasis on data collection was made over areas if the lake where depth change was greatest i.e., steepest regions such as the hole in the north of the lake and steep high walls in the west. A handheld GPS (Garmin etrex) was used to record surface coordinates as decimal degrees which were then converted to UTM units to match depth data (Garmin 12). Depth data was manually recorded as meters at each surface coordinate with sonar (Humminbird 110X). The locations of all the bathymetric measurements are shown in Figure 16.

Based on these data, a bathymetric map was derived as shown in Figure 17. Because no reliable measurements of lake water levels in m AHD are available for this un-monitored historic lake, the water table at the time of measurements was set to zero and all depths are shown as negative numbers. The bathymetry can be considered a rough approximation and should be qualified with measurements with a higher spatial resolution.

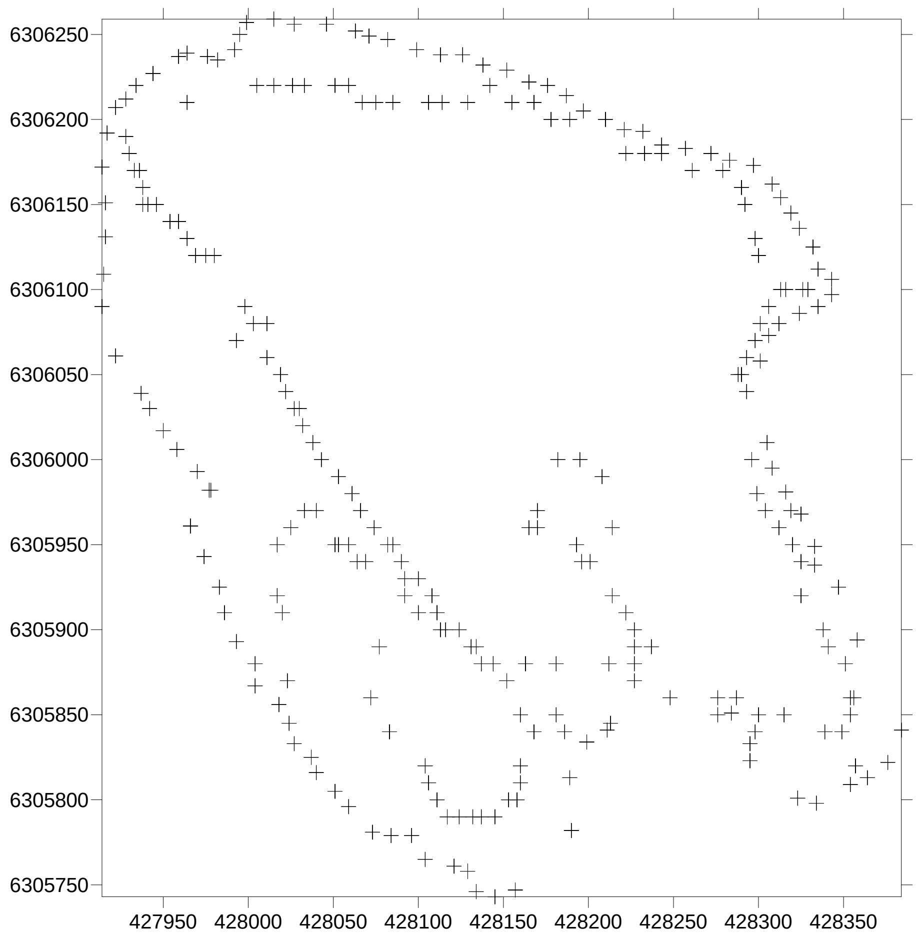

Figure 16. Measured lake bottom and lake shore heights. All values as UTMs, northings as 50 H 0 prefixes.

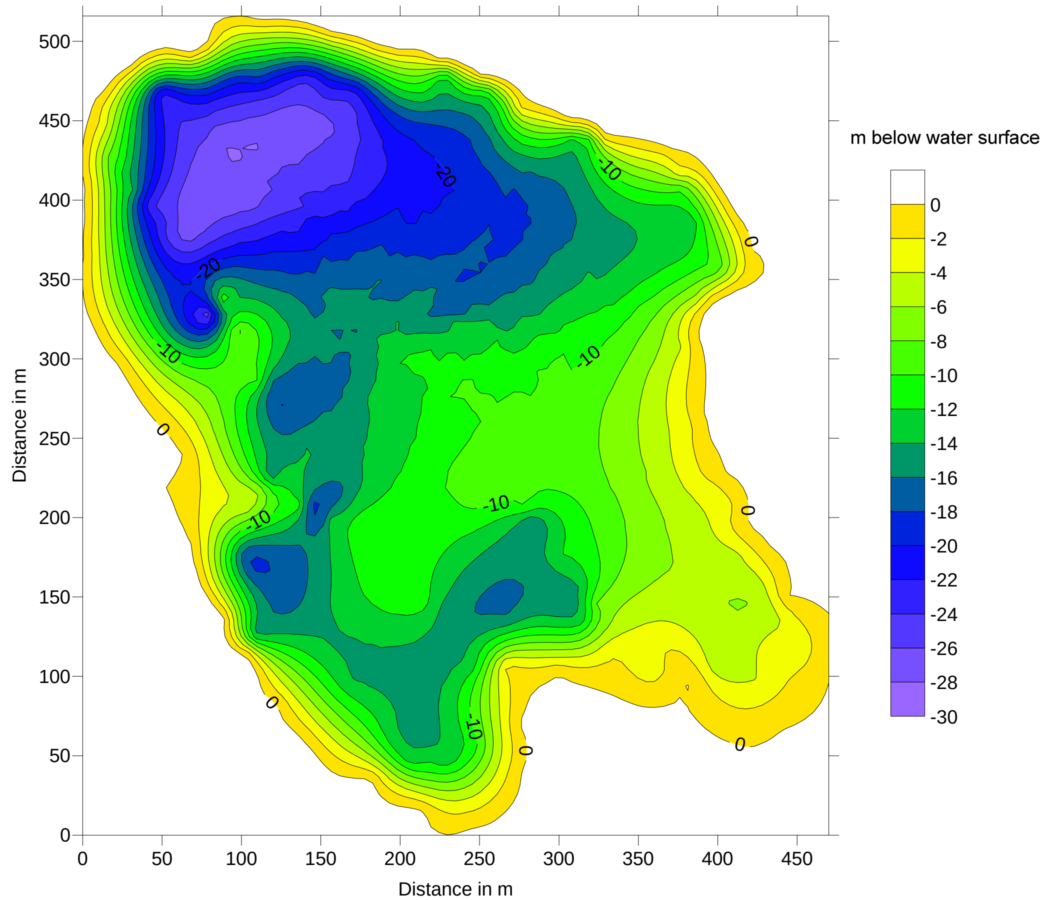

Figure 17. Bathymetry of Lake Stockton.

A model discretisation was derived from the bathymetry, which can be seen in Figure 18. Again, the layer thickness is one metre for all layers. A total of 35 active layers were used to represent the lake. Five additional layers above the water table were added to allow for scenarios with rising water table. Since no other information was available, these cells have the same spatial size as the cells at the water table.

Horizontally, 11 active segments with a length of 10 m each divide the lake. The segments start in the south and extend towards the north. This orientation is opposite to that of Lake Kepwari and was chosen to easy model runs. The deepest part of the model should ideally be located to the right in the model. Therefore, the orientation was chosen individually for each model based morphological conditions.

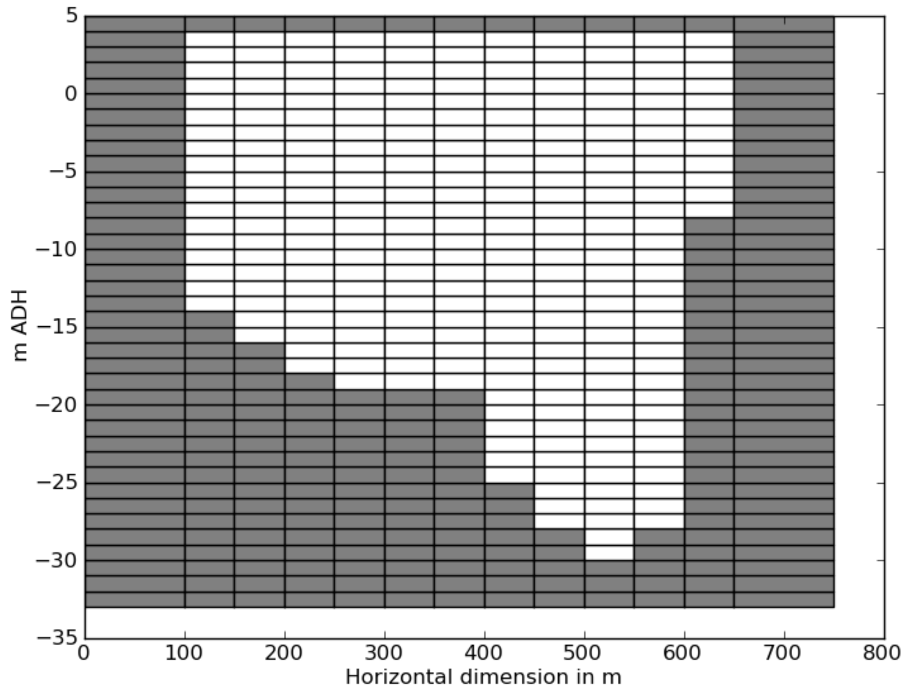

Figure 18. Discretisation for the model of Lake Stockton (south to north, white cells are active, gray cells inactive).

6.5 Lake WO5C

The same procedure as for the two other lakes was applied to Lake WO5C. The bathymetry is shown in Figure 19. As Lake Kepwari, this lake is elongate with steep banks. The discretisation is shown in Figure 20. Again, vertically a layer thickness of one metre was applied. In total, 52 layers were set active and possibly receive water during model runs.

Horizontally 14 active segments with a length of 100 m each were used. The orientation of the cells was chosen to be east to west, again placing the deepest segments to the right part of the model.

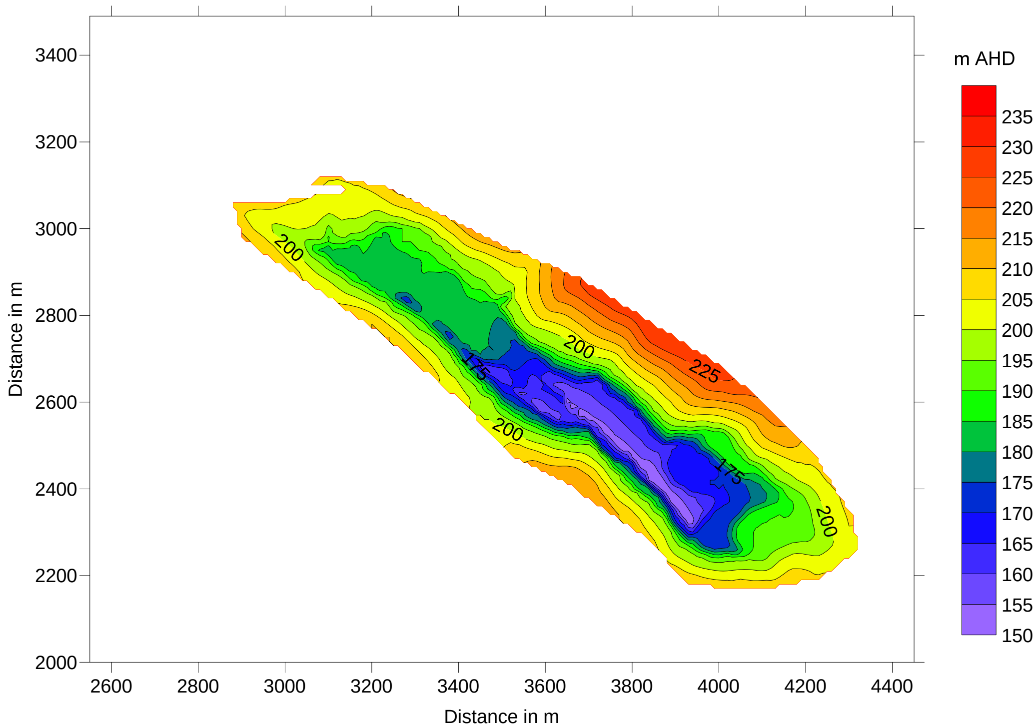

Figure 19. Bathymetry of Lake WO5C.

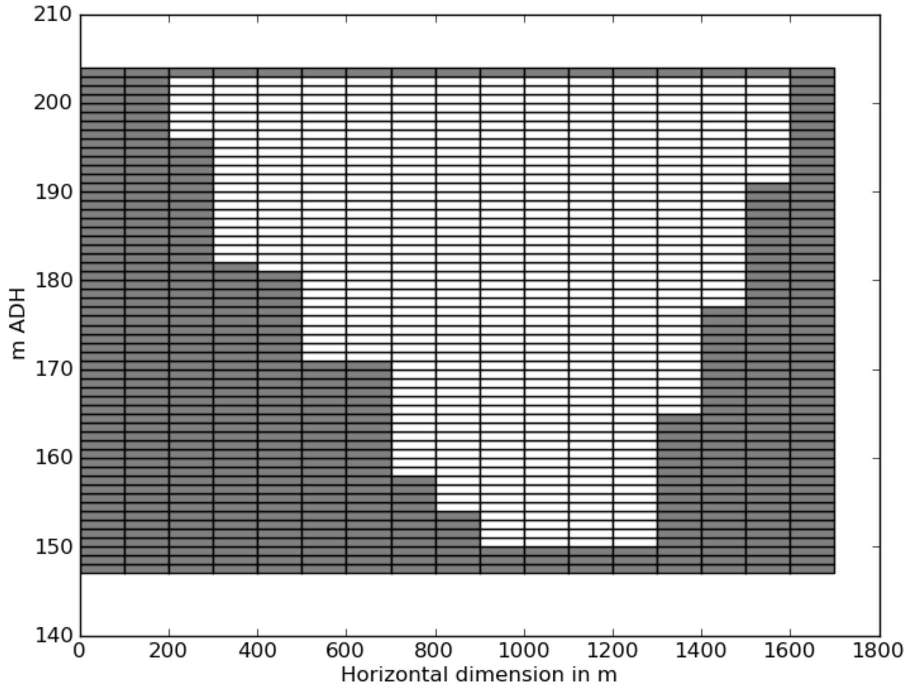

Figure 20. Discretisation for the model of Lake WO5C (east to west, white cells are active, gray cells inactive).

7 Results of Physical Modelling

7.1 Lake evaporation

Evaporation from the lake surface is an important sink of water from the lake and needs to be realistically represented in the model. The model for Lake Kepwari, as describe above, was used to calculate the evaporation of a period of 13 years (1997 – 2009). The calculation was done with the term algorithm of CE-QUAL-W2 (Cole and Buchak, 2005, pp. A17ff). The evaporation is calculated for each iteration step. The step length is variable and can be one hour to several seconds depending on numerical criteria. The evaporation depended on the air temperature, the dew point temperature, the wind speed and the surface water temperature, which was calculated using a heat budget model. Daily results were saved for all model cells at the surface and aggregated to monthly average values for the whole lake over the total calculation time.

The results were compared to measurement with class A pan evaporations for Western Australia (Luke *et al.*, 1987). The comparison is shown in Figure 21. Considering the different reference times and the deviations found in measurements, the agreement can be consider very good. The measured yearly, average value is 1,435 mm/annum whereas the calculated value is 1,414 mm/annum.

No calibration of evaporation has been done. All results were calculated with the input metrological data as described above and the internal parameters of CE-QUAL-W2. The good correspondence with the measured values suggests that projections into the future with metrological input data are possible. This allows for the investigation of the climate change the water budget of the lakes and derived from this on the development of its water quality.

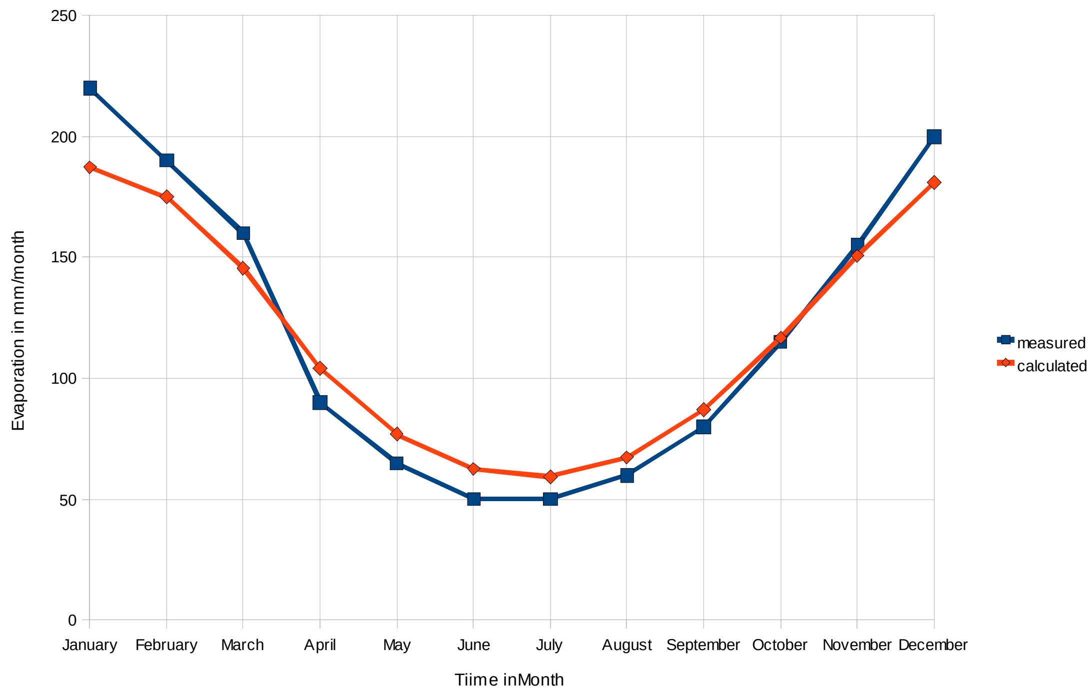

Figure 21. Comparison of measured and calculated values for long-term average monthly evaporation from Lake Kepwari.

7.2 Lake Kepwari

7.2.1 Inflow sources

The first step of physical modelling was to apply the different inflow sources to the lake. There is discontinuous inflow from River Collie into Lake Kepwari. Furthermore, there is groundwater inflow as calculated by the models based on MODFLOW and FEFFLOW as explained in section 6.2. For the FEFFLOW-based model two inflows scenario, one with current abstraction and one with the licensed abstraction from the groundwater are used. Because model output data for the FEFLOW-based model was available only from 2010 onwards, the values weer projected back to 1997 to make them comparable with the value from MODFLOW-based model and the measured water levels in the lake. As can be seen from Figure 13, the differences between the different groundwater models in terms of inflow to Lake Kepwari are small. All models overestimate the inflow and do not provide values for outflow. The following sections look at development of the calculated lake water level in comparison with measured values. A compensation flow is calculated to match both.

7.2.2 Inflow from MODFLOW-based groundwater model

The deviations between measured and calculated water levels for groundwater inflows calculated with the MODFLOW-based model are shown in Figure 22. There is a difference of up to 20 m between both levels. This was due to the fact that the groundwater model only calculates inflows to the lake but no outflows. The higher the lake level the larger the excepted outflows.

The compensation flow necessary to achieve a match of measured and calculated water levels was obtained form the model. The results of these calculations are shown Figure 23. There are a few peaks of high but short outflows that indicate lake water quickly flew into unsaturated aquifers. The correspondence of measured and calculate lake water levels is presented in Figure 24.

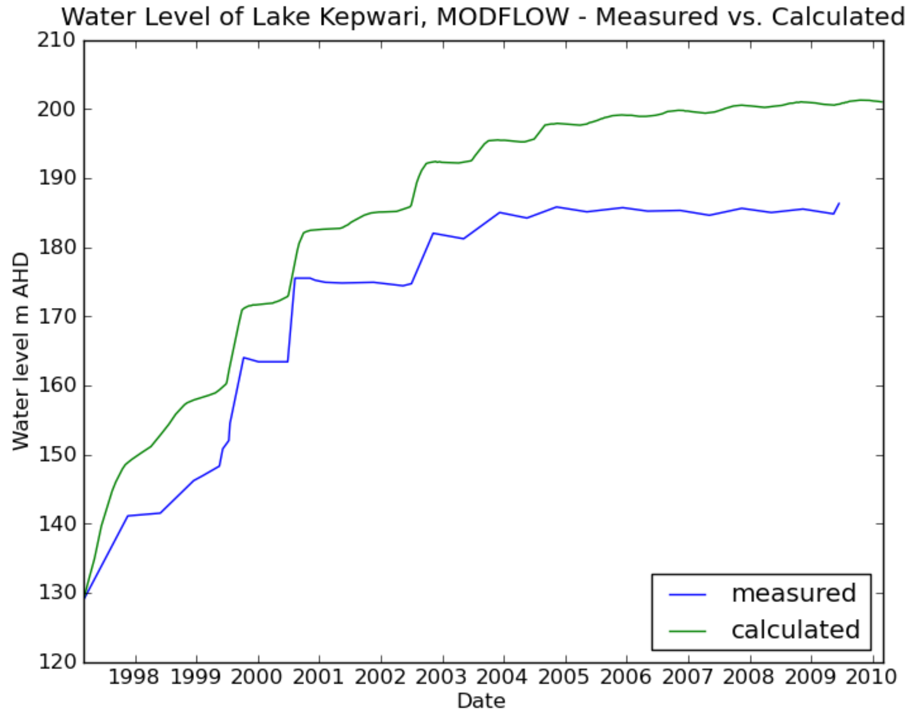

Figure 22. Comparison of measured and calculated water levels in Lake Kepwari over time using the inflows as calculated by the MODFLOW-based model, no compensation

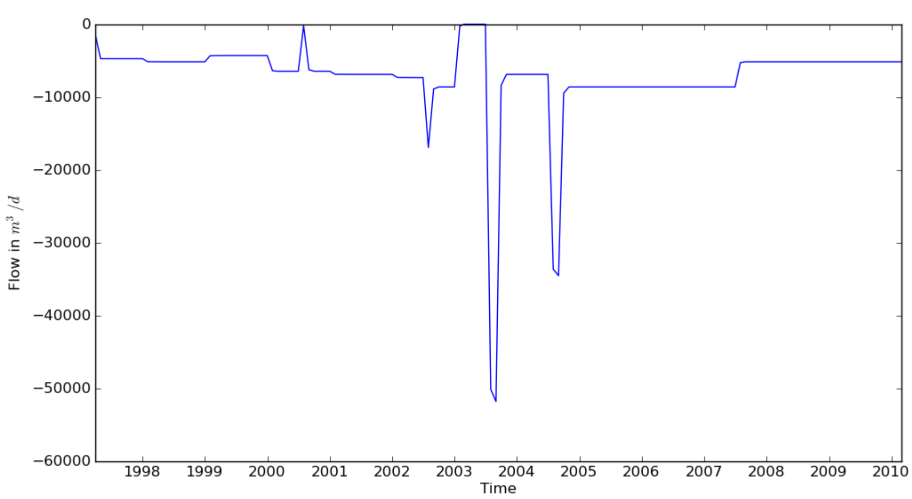

Figure 23. Compensation flow to achieve a match between measured and calculated lake water level of Lake Kepwari using the inflows as calculated by the MODFLOW based Model (negative values indicate outflows)

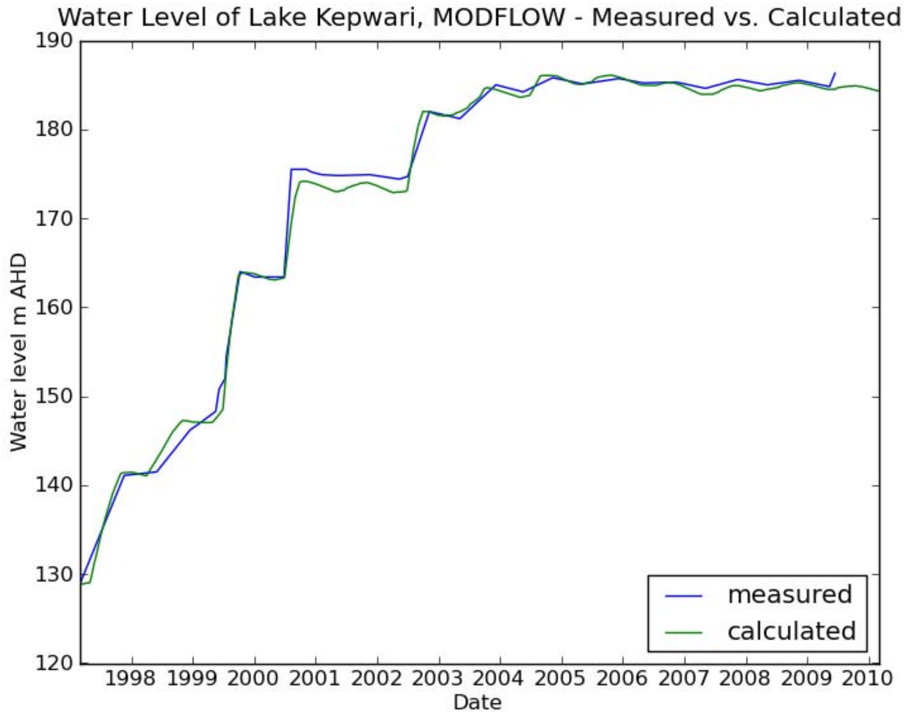

Figure 24. Comparison of measured and calculated water levels in Lake Kepwari over time using the inflows as calculated by the MODFLOW-based model, with compensation

7.2.3 Inflow from FEFLOW-base groundwater model with current abstraction

The compensation inflow calculated for the inflows from the MODFLOW-based model was used for the scenario with the inflows from the FEFFLOW-based model with the current abstraction. The calculated development of the water table compared to the measured values can be seen in Figure 25. The calculated water level from the year 2004 on was slightly higher then the measured vales. This was due to no further adjustment being applied to the compensation flow.

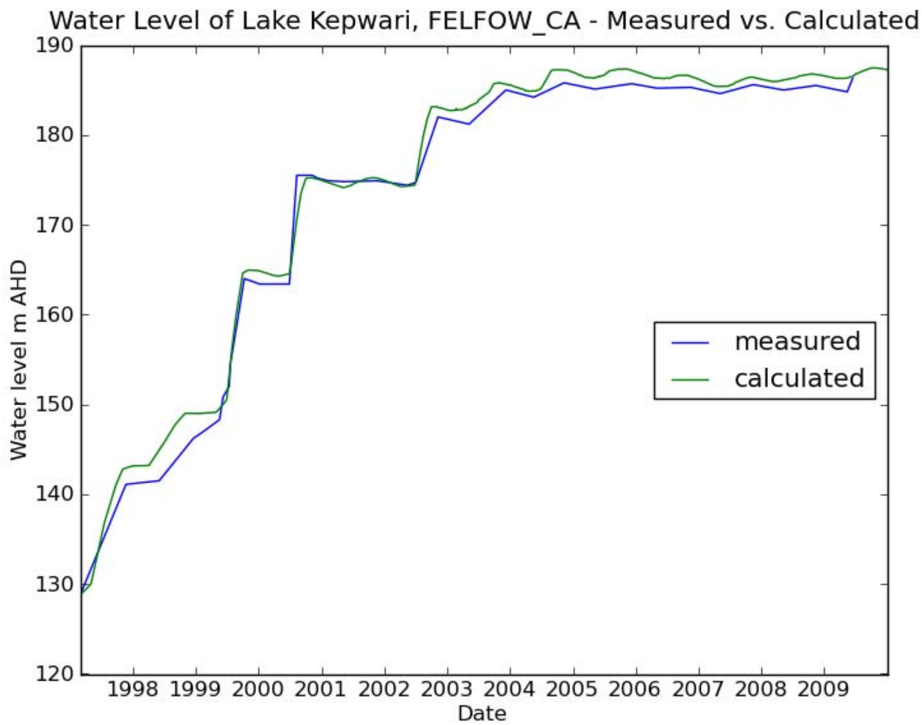

Figure 25. Comparison of measured and calculated water levels in Lake Kepwari over time using the inflows as calculated by the FEFFLOW-based model with current abstraction from groundwater, with compensation.

7.2.4 Inflow from FEFLOW-based groundwater model with licensed abstraction

The calculation was repeated with the inflows to Lake Kepwari derived from the FEFLOW-based model with the licensed abstraction. The lake levels obtained from the model and the measured values are depicted in Figure 26. Again, there was good agreement between calculate and measured values. Starting in mid 2006, there is some slight deviation of the calculated values showing lower water level than calculated.

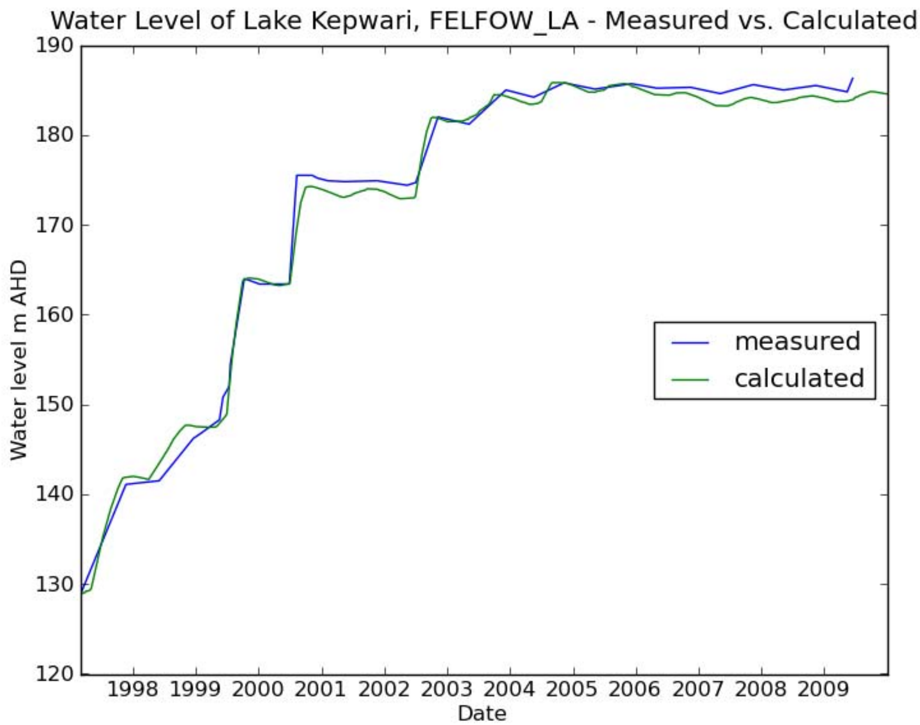

Figure 26. Comparison of measured and calculated water levels in Lake Kepwari over time using the inflows as calculated by the FEFFLOW-based model with licensed abstraction from groundwater, with compensation.

7.2.5 Selection of groundwater scenario for further calculations

The differences between the three groundwater inflows from the MODFLOW-based model and the two scenarios of the FEFFLOW-based model are small. The inflows from the MODFLOW-based model will be used for all further calculations as they are approximately in the middle of the other two scenarios.

7.2.6 Determination of surface runoff

The surface runoff was determined based on the work of Varma (2002). The groundwater inflows and outflows from Varma (2002) are shown in Figure 27. These numbers and the sum of ground inflow and the compensation flow are in the same range. Therefore, the approach to calculate surface water runoff as a function of lake water level was adopted.

The relationship between lake water level and surface runoff as determined by Varma (2002) in Figure 28 was nearly linear with some flattening of the curve at lower an higher water levels. Based on these findings, a water level dependent surface runoff was derived, were runoff was only applied during the period from June to October as can be seen in Figure 29. This surface runoff was subtracted from the compensation (Figure 23). This flow was combined with the groundwater inflow to create an effective groundwater inflow and outflow. As a result of these calculations two distinct inflows were used as model input: (1) surface runoff, (2) groundwater inflow each with its own water quality characteristics. This means that in model are two different fluxes into the lake. Groundwater outflow constituted a flux from the lake calculated from outflow rate and water composition for each time step.

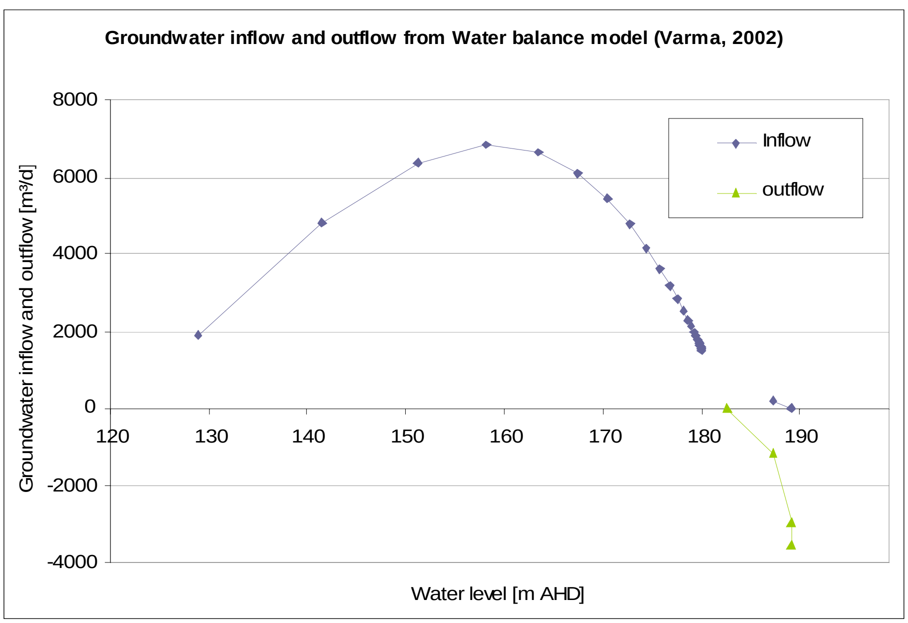

Figure 27. Groundwater inflow for Lake Kepwari (from Varma, 2002).

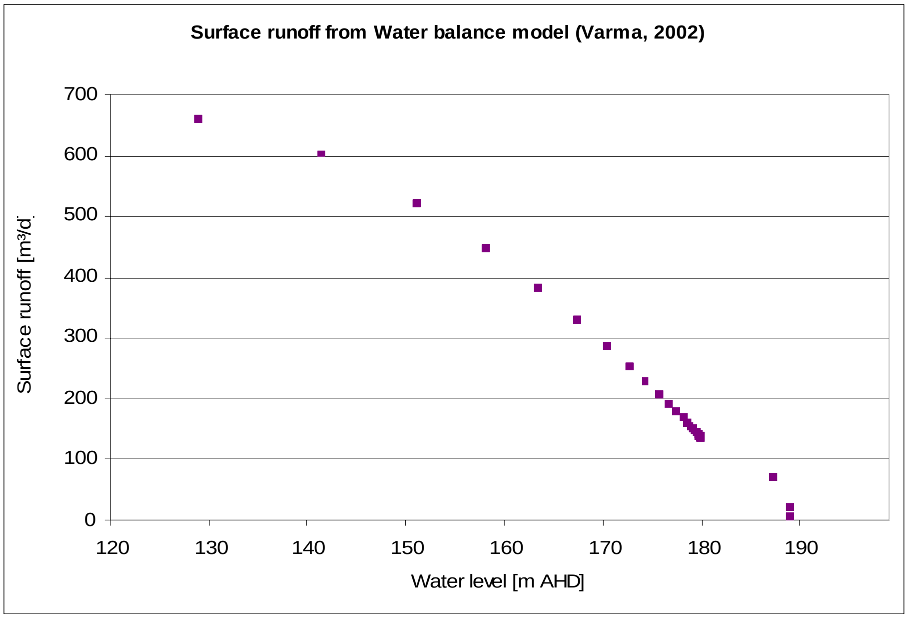

Figure 28. Surface runoff for Lake Kepwari (from Varma, 2002).

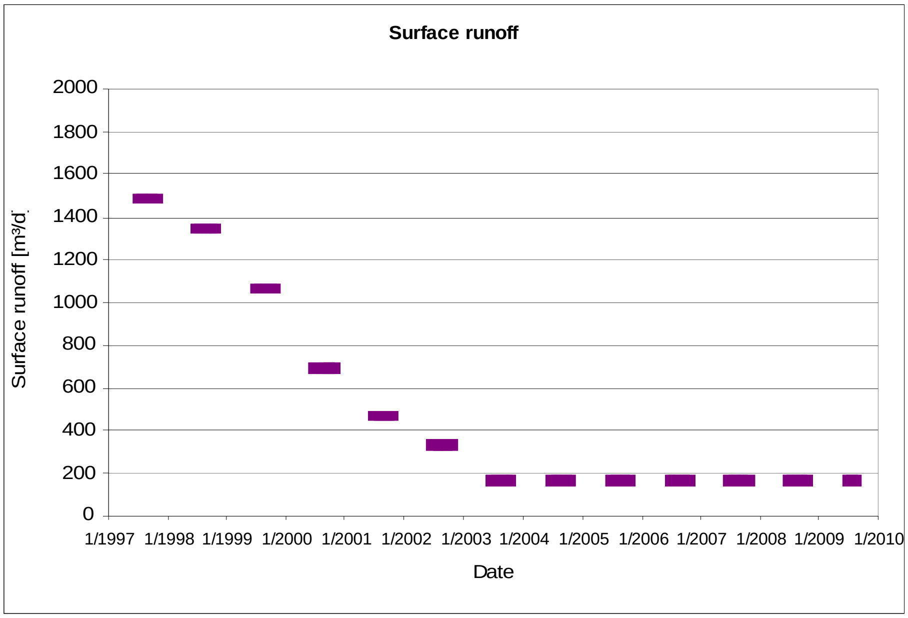

Figure 29. Modified surface runoff for Lake Kepwari with runoff only from June to October.

7.2.7 Calculation of temperature distribution

Data of temperature distributions over time and the depth f the lake were reported by Salmon *et al*. (2008) (Figure 30).

The temperature distribution over the same time period and at the deepest point of the lake was extracted from the modelling results. The graph in Figure 31 shows these results with the same colour scale as the measured values. The distributions are similar. The very deep epilimnion at the end of the summer stagnation period could nearly be reproduced with the MODGLUE model.

All 13 model years were calculated with one model run. The modelling software typically produced stable results for modelling periods of decades or even century allowing for predictive modelling of temperature distribution.

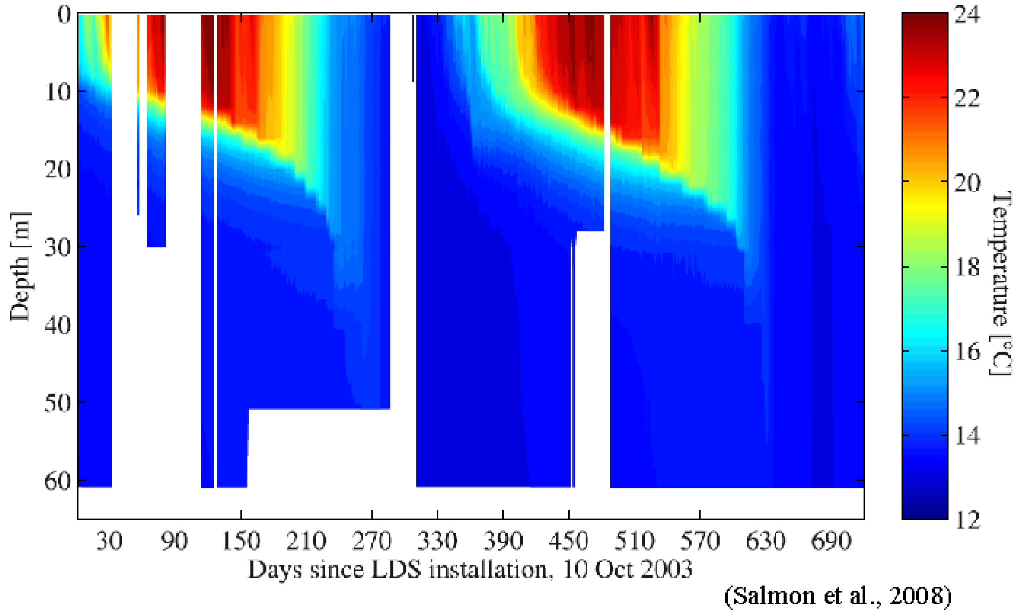

Figure 30. Measured temperature distribution in Lake Kepwari (from Salmon et al. 2008).

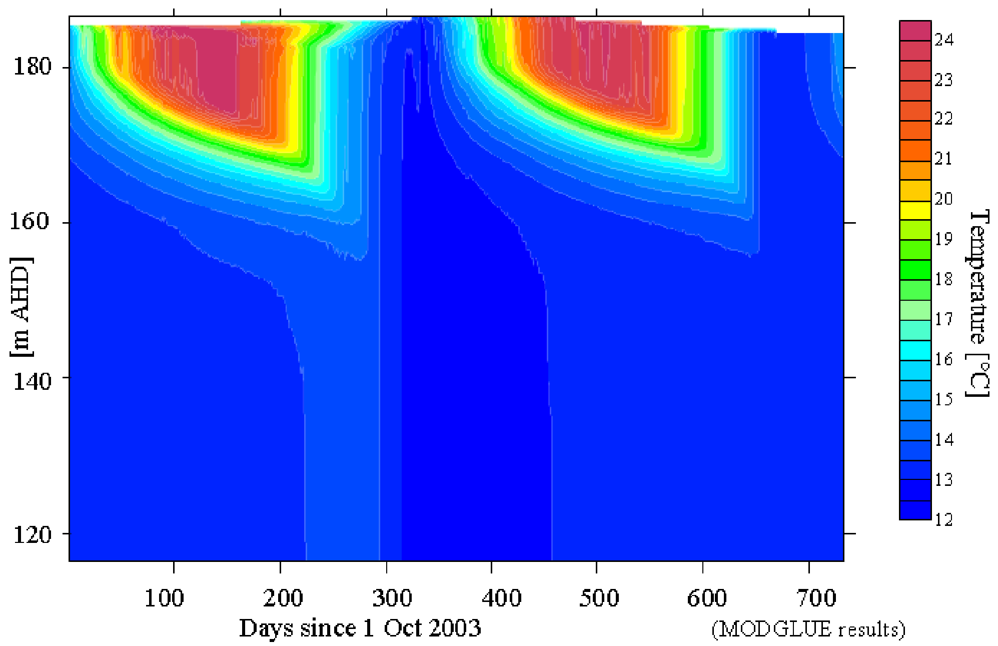

Figure 31. Calculated temperature distribution in Lake Kepwari.

7.3 Lake Stockton

7.3.1 Inflow from MODFLOW-base groundwater model

There was no data available for the inflows to Lake Stockton for the MODFLOW-based groundwater model.

7.3.2 Inflow from FEFLOW-based groundwater model with current abstraction

The lake inflows calculated with the FEFFLOW-based groundwater model where applied. Again, the data was available from 2010 onwards. In order to be consistent with the approach for Lake Kepwari, data were projected back to 1997, using data from January 1st 2010ß as January 1st 1997. The results of this calculation are shown in Figure 32. There was a very clear trend of a falling water level indicating the need for more inflow to the lake to maintain its current water level. Since no observation data for the lake water level were available, the current water table is shown as reference. The assumption was that the water table is approximately stable over time. Seasonal variations were not shown but are likely present as the model suggests.

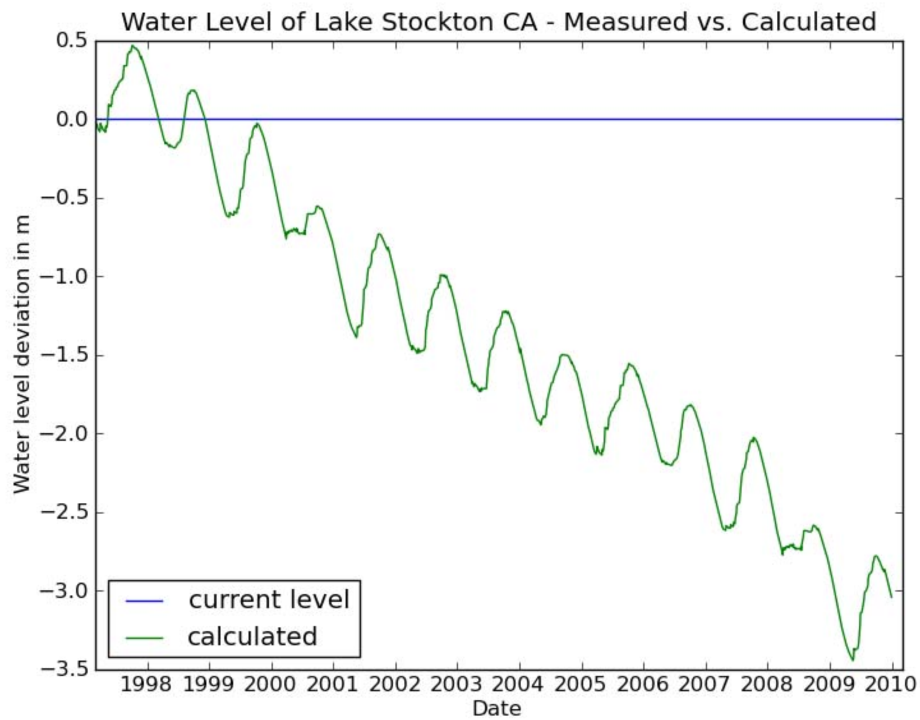

Figure 32. Water level in Lake Stockton over time using the inflows as calculated by the FEFFLOW-based model with current abstraction from groundwater without compensation (there are no measurements of the lake water level, so the current level was used as zero marker).

A compensation inflow rate was then determined. A constant value of 0.0012 m^3/s (or 103.68 m^3/d or about 0.1 ML/d) produced a water table that varies around the current water table as can be seen from Figure 33.

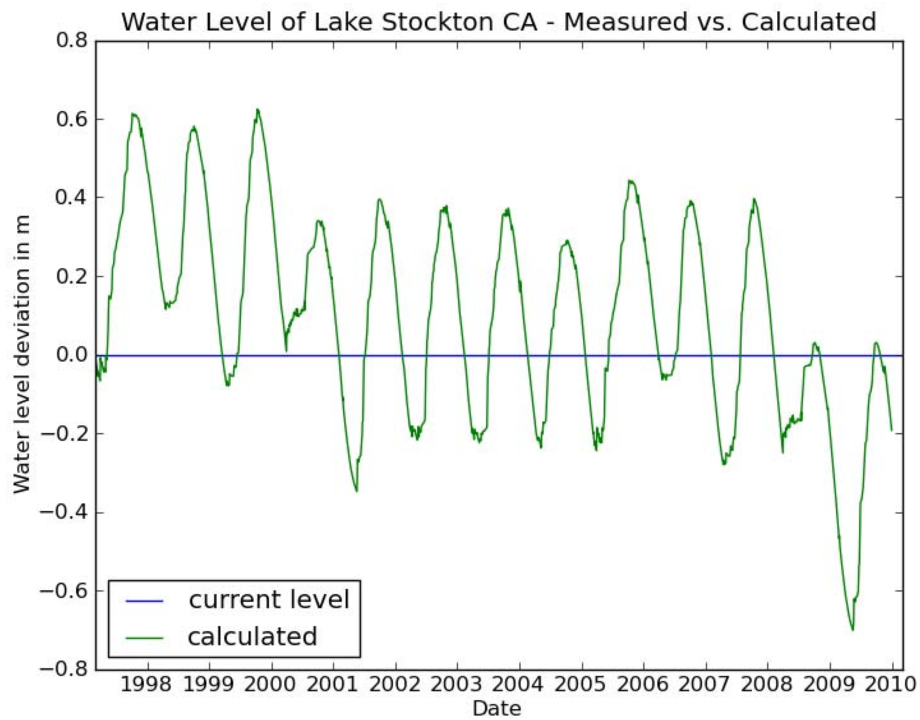

Figure 33. Water level in Lake Stockton over time using the inflows as calculated by the FEFFLOW-based model with current abstraction from groundwater with compensation (there are no measurements of the lake water level, so the current level was used as zero marker).

7.3.3 Inflow from FEFLOW model with licensed abstraction

The same procedure as for the current abstraction was used for the inflows calculated with the FEFLOW-based groundwater model with licensed abstraction. As can be seen in Figure 34, there was a sharper drop in the water table than for the current abstraction.

A compensation inflow rate was then been determined. A constant value of 0.0015 m^3/s (or 129.6 m^3/d or about 0.13 ML/d) produced a water table that varies around the current water table as can be seen from Figure 35.

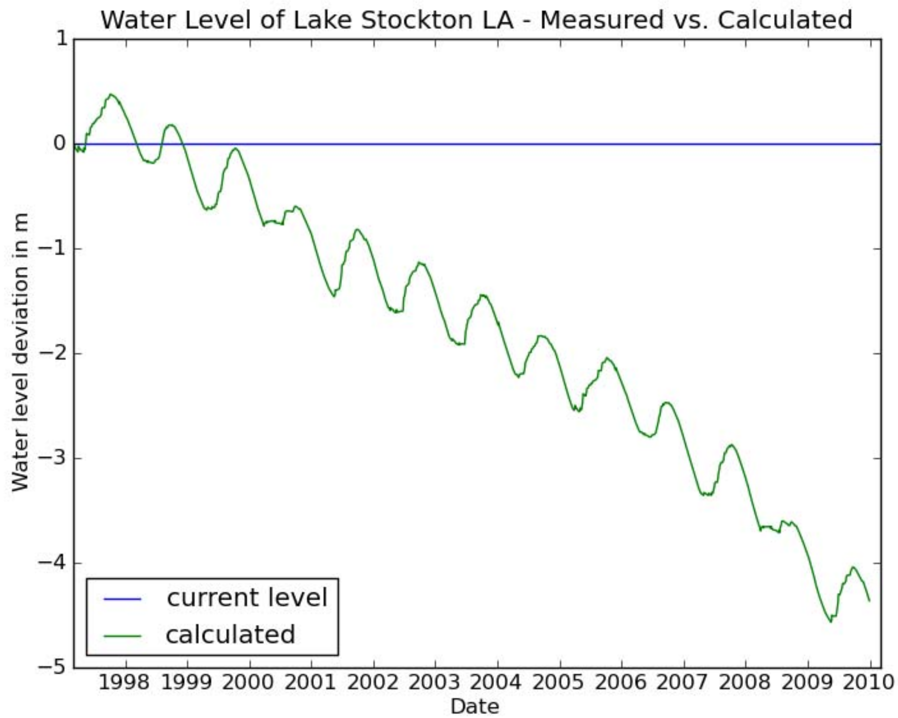

Figure 34. Water level in Lake Stockton over time using the inflows as calculated by the FEFFLOW-based model with licensed abstraction from groundwater without compensation (there are no measurements of the lake water level, so the current level was used as zero marker).

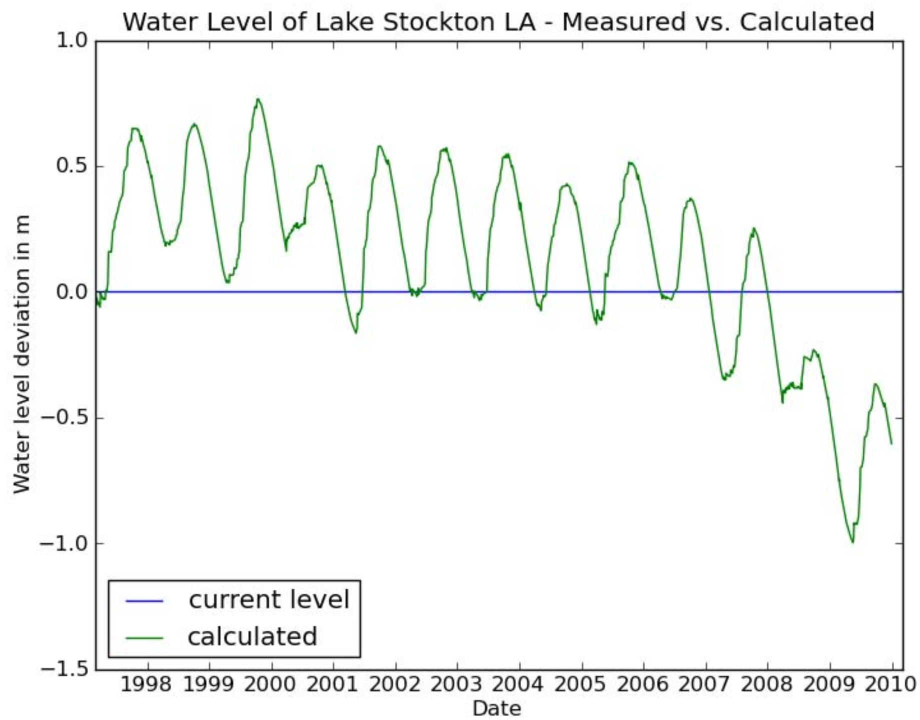

Figure 35. Water level in Lake Stockton over time using the inflows as calculated by the FEFFLOW-based model with licensed abstraction from groundwater with compensation (there are no measurements of the lake water level, so the current level was used as zero marker).

7.3.4 Calculation of temperature distribution

The distribution of the water temperature was calculated for a period of 12 years. The results are compared to measured values in Figure 36. The seasonal formation of epilimnion and hypolimnion in the lake could be modelled over the whole duration of 12 years. Because frequent measured values were available for 1997 and 1998 only, Figure 37 shows the detail for this period. A good agreement between modelled and measured data could archived for the distribution over time and depths as well as for the temperature values themselves.

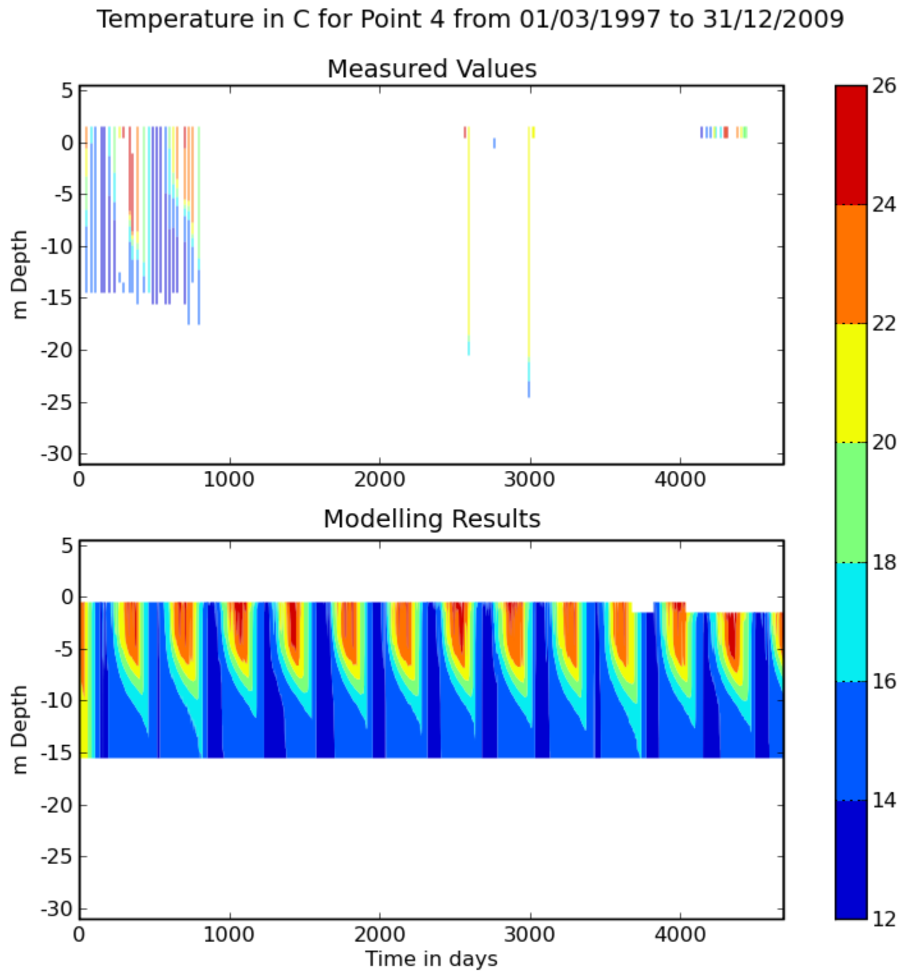

Figure 36. Modelled and measure temperatures from 1997 to 2009 for Lake Stockton.

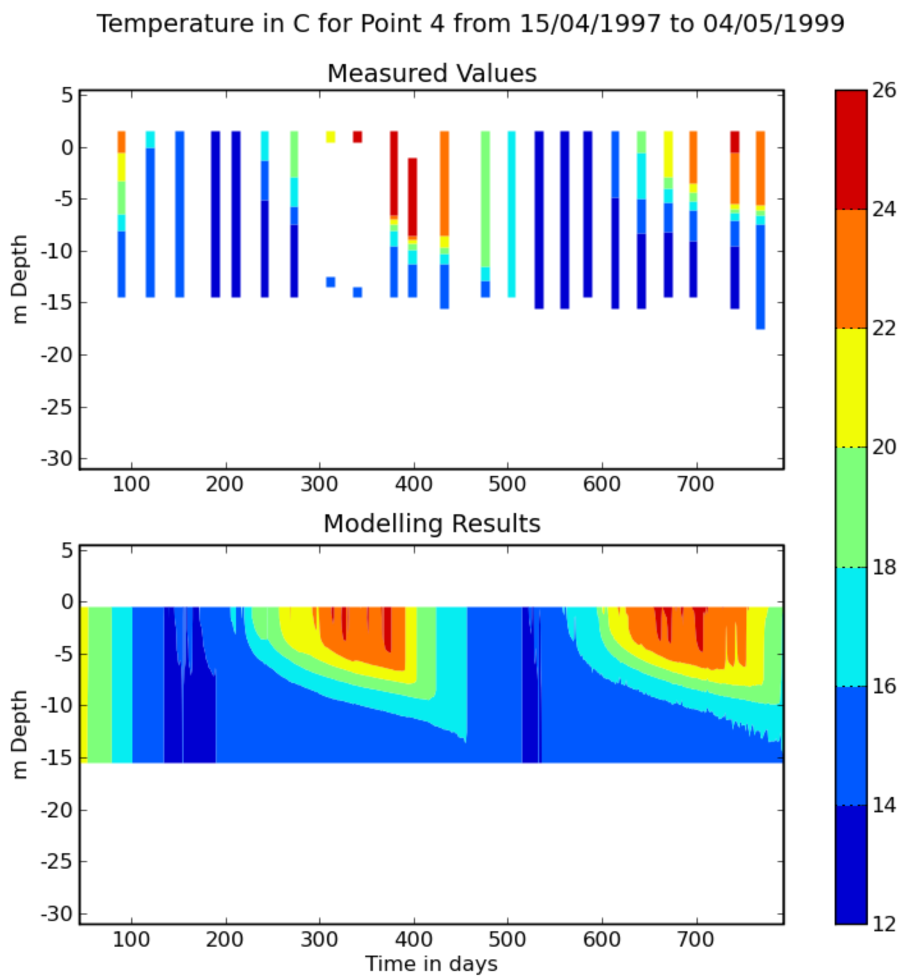

Figure 37. Modelled and measure temperatures from 1997 to 1998 forLake Stockton.

7.4 Lake WO5C

No ground inflow data was available for Lake WO5C. A calculation without inflows was performed. The resulting water levels are shown in Figure 38. This graph shows clearly that inflows are needed to reproduce the measured water levels over time. A compensation inflow rate was also determined. The values are depicted in Figure 39. Besides seasonal variations, there was a clear trend of declining inflow rates. Using this compensation flow, a reasonable agreement between measured and calculated water table levels could be achieved as shown in Figure 40.

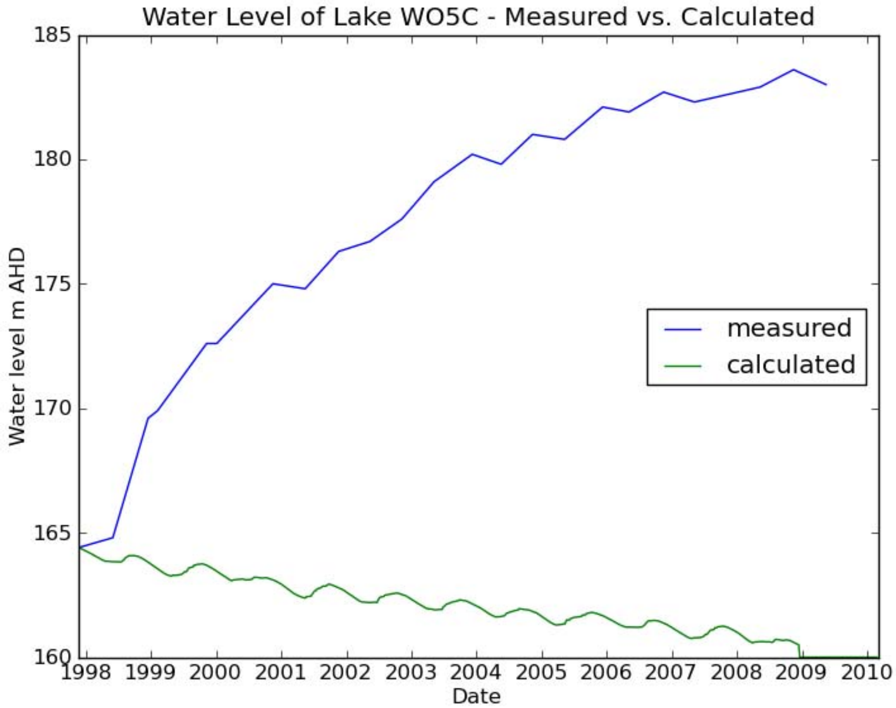

Figure 38. Comparison of measured and calculated water levels in Lake WO5C over time, without compensation.

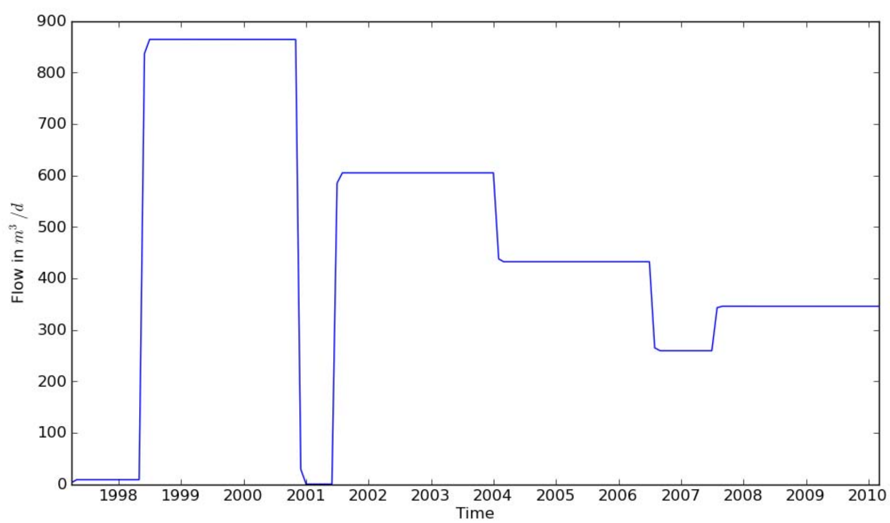

Figure 39. Compensation flow to achieve a match between measured and calculated lake water level of Lake WO5C

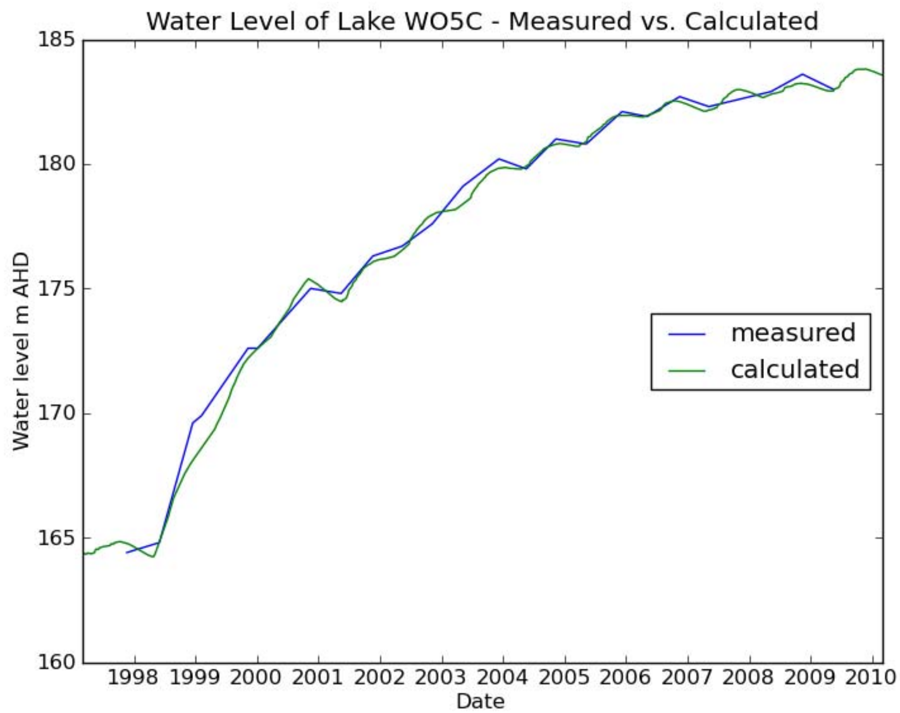

Figure 40. Comparison of measured and calculated water levels in Lake WO5C over time, with compensation.

8 Water Quality Modelling

8.1 Sources of acidity

The water quality of the pit lakes in the Collin Basin is largely determined by the pH value of the lake water. Even after partial filling with river water and a temporally increases in pH there seem to be sources of acidity that decrease the pH after awhile. For instance this type of behaviour could be observed for Lake Kepwari.

As discussed in section 4.15, leaching from mining sediments might be a cause of long-term input of acidity to the lakes. This source was used for the further investigations.

8.2 Lake Kepwari

8.2.1 Concentrations of initial lake water

The model runs for the water quality simulations start in June 1999. This date was chosen to satisfy the requirement to start with begin of filling with Collie water in 1999. The first analysis of the lake water quality available is from June 2001 (Zhao *et al.*, 2009). The initial water quality shown in Table 4 is therefore an approximation of a possible water quality. To preserve the measured pH, the chloride concentration has been adjusted in pre-processing with PHREEQC to 537 mg/L.

Table 4. Composition of intial lake water for Lake Kepwari.

Species	Concentration (mg/L)
pH	4.2
TDS	1,200
SO_4^{2-}	120
Na	255
Cl	550
Ca	20
Mg	59
K	5.3
Al	4
Fe	0.8
Mn^{2+}	0.25
NH_4^+	0.5
NO_3^-	1.5
Si	3.5
TP	0.01

8.2.2 Concentration for inflowing water from Collie River

The concentrations of the species of the inflowing water from Collie River are kept constant throughout the whole modelling period. The concentrations are shown in Table 5. To preserve the measured pH, the chloride concentration has been adjusted in pre-processing with PHREEQC to 833 mg/L.

Table 5. Composition of filling water from Collie River. Average concentrations for 2004 from Salmon *et al.* (2008).

Species	Concentration (mg/L)
pH	6.7
TDS	1,500
SO_4^{2-}	60.5
Na	392
Cl	879
Ca	30.7
Mg	83.1
K	3.6
Al	0.06
Fe	0.31
Mn^{2+}	0.08
NH_4^+	0.03
NO_3^-	0.81
Si	3.01
TP	0.02

8.2.3 Concentration for inflowing groundwater

The composition of the inflowing groundwater was estimated from the general flow conditions as shown in (Figure 41) and the location of bores (Figure 42). The assignment of bores to the inflow from the aquifers that are connected with the lake is not unique allowing for different interpretations. Therefore, four different groundwater compositions as shown in Table 6 were used. Each composition was used for all groundwater inflows of one scenario. Hence, our different groundwater inflow scenarios were constructed.

Table 6. Chemical composition of inflowing groundwater (mg/L) for Lake Kepwari (after Varma, 2002, based upon one sample from July 1998).

Species	CRM-79	CRM-85	CRM-86	CRM-89
pH	4.1	5.6	4.5	5.6
TDS	1,760	420	560	640
SO_4^{2-}	72	75	27	59
Na	470	110	170	180
Cl	860	130	310	290
Ca	5	2	4	5
Mg	63	3	20	21
K	10	2	2	18
Fe	0.6	0.1	0.05	3.9
Mn^{2+}	0.26	0.18	0.09	0.32
NO_3^-	0.5	0.5	0.5	0.5
P	0	0.05	0.17	0.04
HCO_3^-	0.5	19	0.5	17

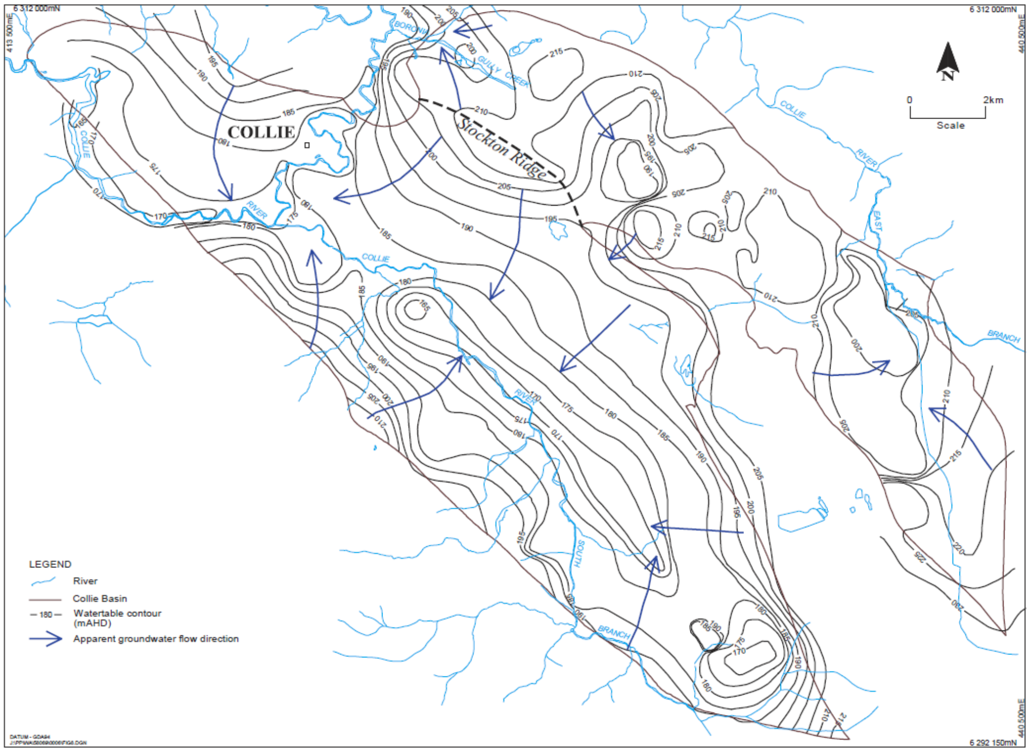

Figure 41. Water table contours from Varma (2002).

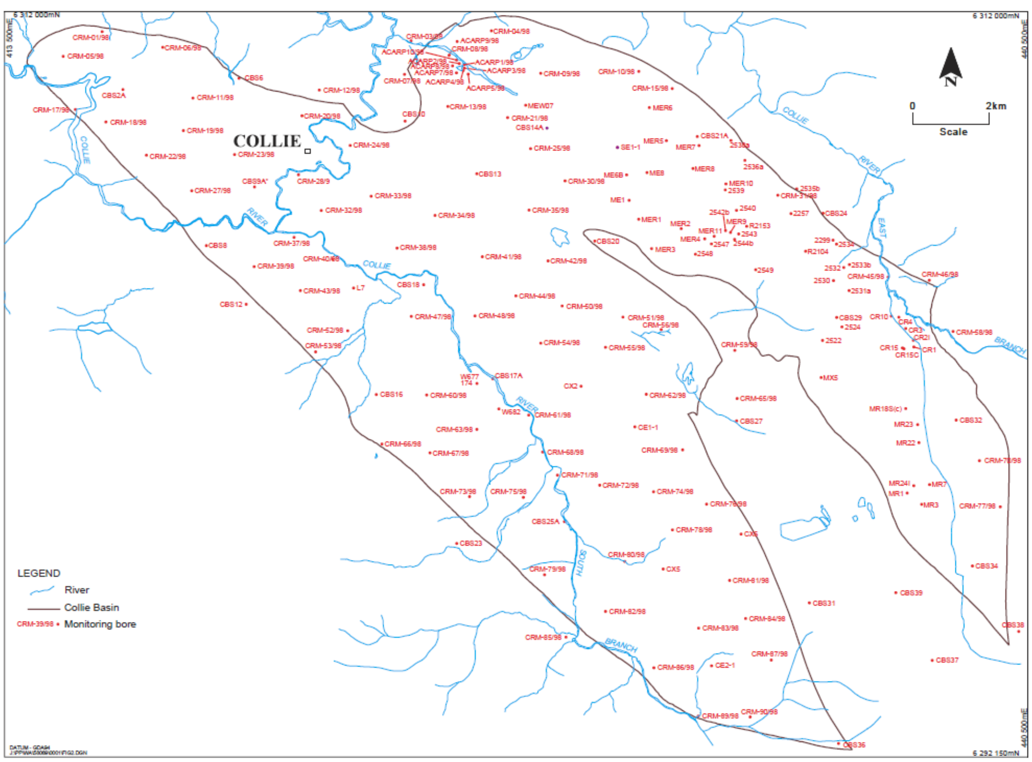

Figure 42. Bore locations from Varma (2002).

8.2.4 Concentration for inflowing water from surface runoff

There is one measurement of the composition of runoff to the lake taken in July 2004 (Salmon *et al.*, 2008). For comparison, two other compositions that were measured at Lake Zwenkau in Germany (LMBV, 2009) are shown in Table 7.

Table 7. Comparison for chemical omposition (mg/L) of pit lake catchment runoff waters. Kepwari measurement based upon one measurement in July 2004 during a rainfall event from *Salmon e. al. (2008)*; low and strong acidic based on measurement at Lake Zwenkau (LMBV, 2009).

	Kepwari	Low acidic	Strong acidic
pH	3.6	5.5	4
SO_4^{2-}	35	300 (charge)	2,500 (charge)
Na	5.4	2.3	18.3
Cl	0	35.45	35.45
Ca	2.55	100.2	700.2
Mg	3.85	24.3	100.3
K	0.96	0.39	4.5
Al	1.55	0.001	10
Fe	0.04		
Fe^{2+}		0.001	350
Fe^{3+}		0.56	40
Mn^{2+}	0.09	0.055	0.055
NH_4^+	0.14	0.017	0.017
NO_3^-	0.02	0.014	0.014
Si	0.27		
TP	0.26		
FRP		0.031	0.031
C(4)		25	300

8.2.5 Concentration for inflowing water from surface runoff

Leaching, as described in section 4.15, can be a significant source of acidity to a lake. Unfortunately, no information about sediment/dump composition that can be used for the calculation of a leaching rate is available. The area that would be active for leaching has been determined using the available digital elevation data and the distribution of the dumps. The area as it increases with the rising water level is shown in Figure 43. Proof-of-concept model runs were performed with leaching rates and compositions measured at Lake Zwenkau (LMBV, 2009).

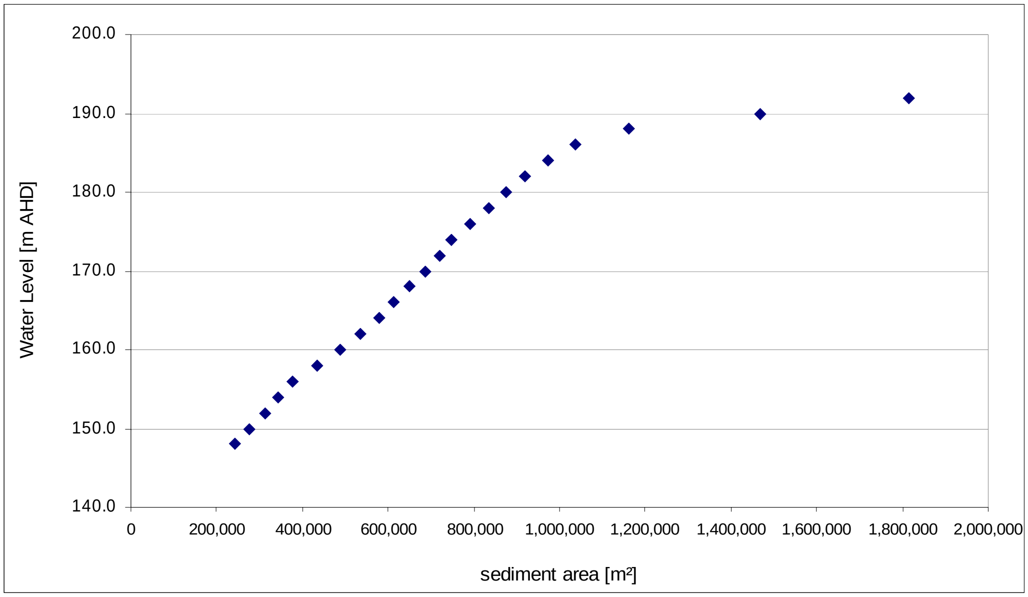

Figure 43. Calculated sediment surface that contributes to leaching as a function of lake water level

Table 8. Composition of leachate as measured for Lake Zwenaku (LMBV, 2009). The given amounts cause 1 mole acidity (KB8.2) per gram substance.

Species	Acidity (g/mol)
Al	2.45
Fe	10.15
Ca	3.64
Mg	4.42
Na	0.02
SO_4^{2-}	69.83

The leaching rates vary over time from 0.4 mol/m^2/d to 0.006 mol/m^2/d with largest rates in the beginning, a strong decrease during the first days, and an asymptotical approach to the lowest values.

8.2.6 Modelling results

The measured pH values of Lake Kepwari are shown in the upper part of Figure 44. There is a clear trend of the measured pH value that increases over time until about 2000 modelling days. This increase is due to the flooding with water from Collie River up to the year 2005. After flooding was ended a significant drop in pH could be observed. This trend could be reproduced with the model as shown in the lower part of the model for a scenario with leaching based on data from Lake Zwenkau. A quantification of the contribution of can be given after the evaluation of the results of the model scenarios described above. Measurements of the sources of acidity are essential for a better quantification.

The distribution over time for measured and calculated concentrations of total dissolved solids is shown in Figure 45. The general trend could also be reproduced.

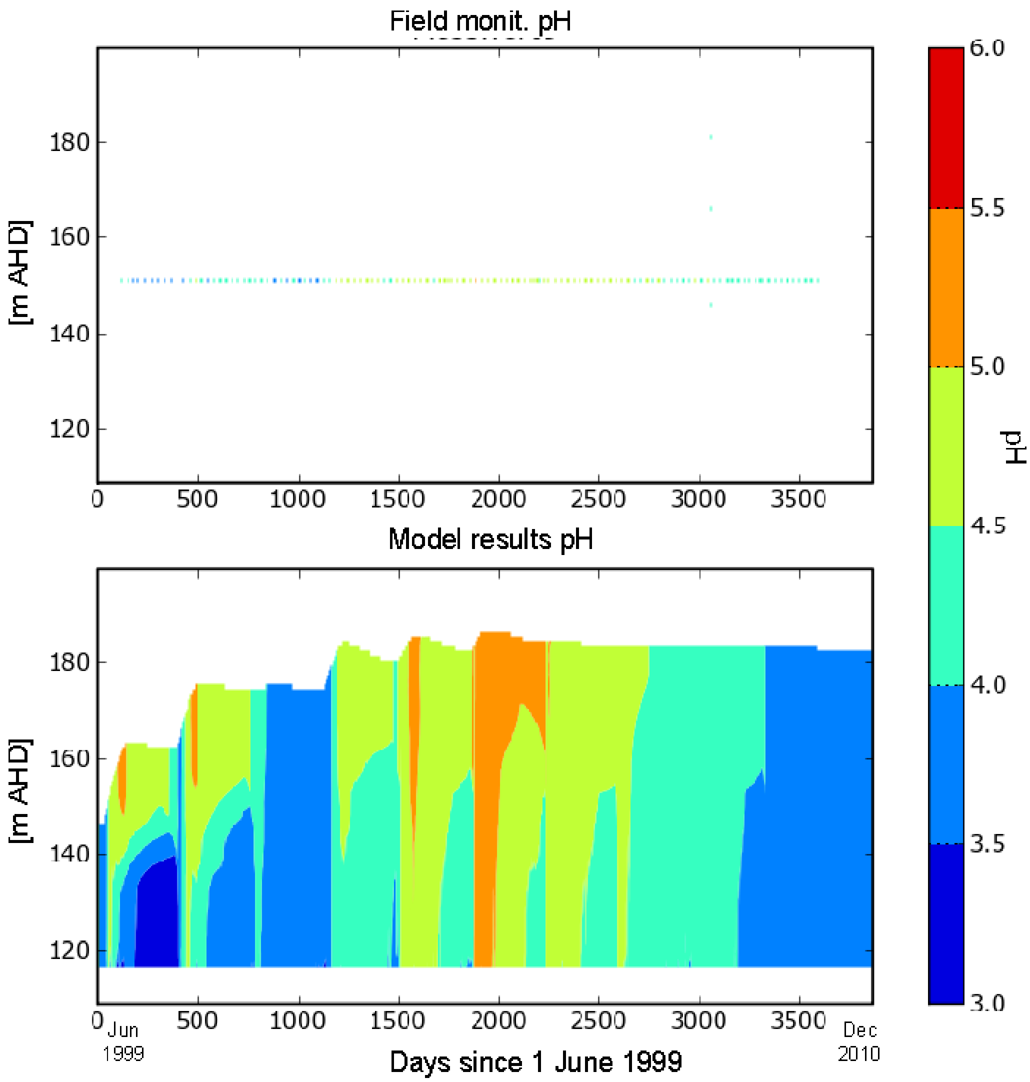

Figure 44. Measured and calculated pH-Distribution for Lake Kepwari.

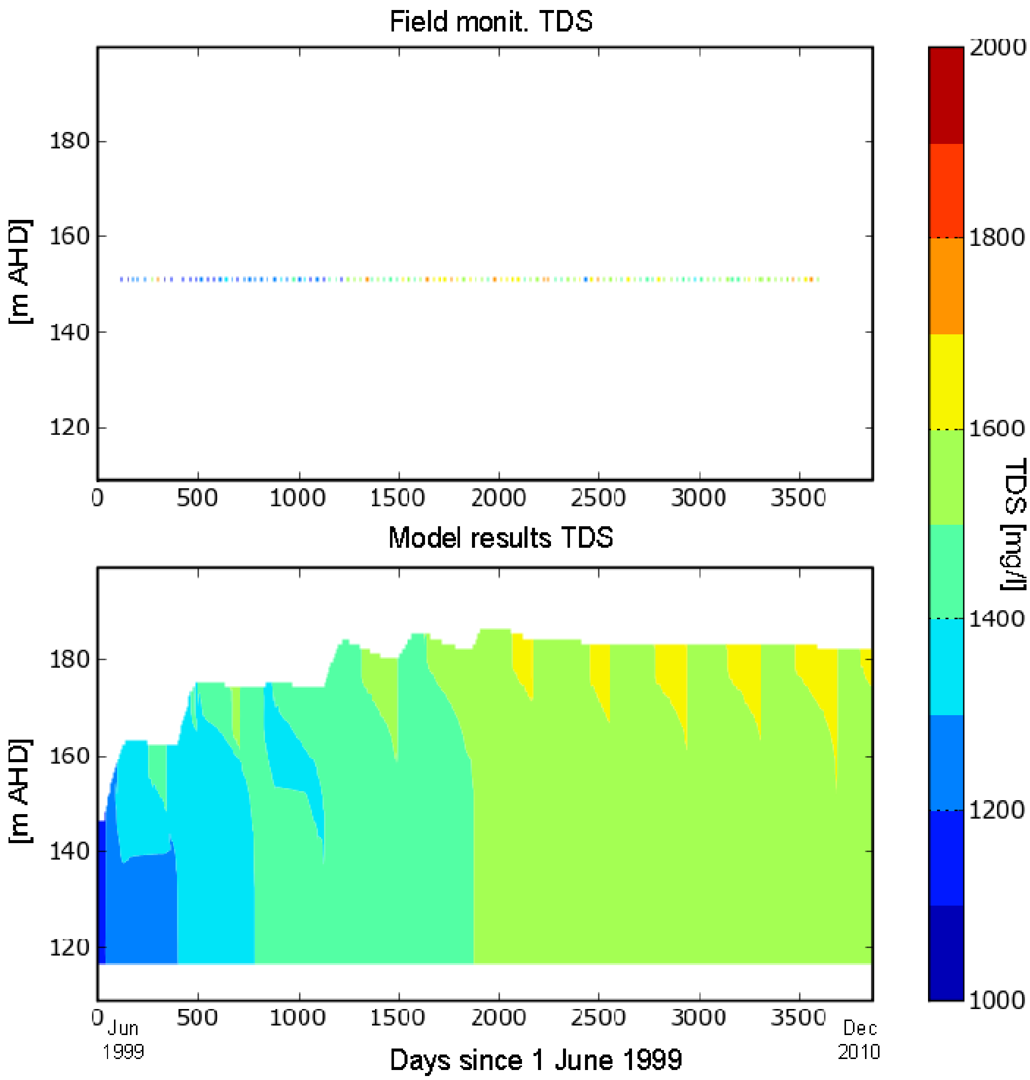

Figure 45. Measured and calculated TDS-Distribution for Lake Kepwari.

8.2.7 Lake Stockton

Results to come

8.3 Lake WO5C

Results to come

9 Conclusions

Testing of a single scenario for all three lake types was complicated due to the physical, chemical and hydrological characteristics of the different pit lakes. The new-formed pit lakes have formed only recently after recent cessation of mining operations and are in the process of filling up either slowly by ground water or rapidly by surface water.

Detailed surface hydrological modelling has not been completed for the representative pit lake surface catchments. The effects of varying catchment size on surface water runoff volume and quality remain unquantified. However, the modelled pit lake water balance in all models is dominated by groundwater influx.

The influence of the climate change was considered in the groundwater model based on FEFLOW with a 10% reduction in groundwater recharge. The changes in groundwater inflow to the lakes are small compared to the differences caused by applying current and licensed abstraction of groundwater. However, the lakes were modelled as constant head boundary conditions specifying the lake stage before the calculation. As can be seen from the model calculation performed in this study, this leads to more groundwater inflow into the lakes as can be derived from the observed water table development.

Lake evaporation rates for Lake Kepwari were reproduced with the model based on the application of physical laws. Both, yearly values and the seasonal variations for long term averages could be computed in good correspondence with measured values. This suggests that the model can be used to model the influence of climatic change on the water budget o the lake. Furthermore, the development of lake water levels could be computed with MODGLUE and used for the groundwater model as input for the boundary conditions representing the lakes. In turn, the resulting groundwater inflows could be used by the lake models as data inputs. This feedback loop would allow for more accurate groundwater inflow calculations. The groundwater model should also allow outflow from the lake into the groundwater.

The differences in groundwater inflow due to different abstraction are relatively small and do significantly influenced the development of the water quality. On the other hand, water quality data for the groundwater inflows are based on few measurements both in spatial as well as the time distribution and therefore provide a much wider range of possible groundwater inflow fluxes.

The effect of decreasing pH values after a temporary increase after addition of neutral water cannot be explained with the groundwater qualities found. Two causes, the erosion from the catchment and leaching form the sub-aquatic sediments, might be responsible for this re-acidification. The leaching was applied with tentative parameters obtained from the Lake Zwenkau case study. The model results are promising and show the observed effect of falling pH values caused by a release of acid producing materials from the sediments.

10 Recommendations

- Pit lake modelling for Collie is clearly limited by available data both from the pit lakes and particularly from groundwater model results. The groundwater inflow as calculated by the groundwater models deviated considerable from inflows adjusted in this study. This is due to the fact that the groundwater models are large scale models. The lakes are small features and not the main concern of the groundwater modelling studies.
    - An incorporation of the boundary conditions in the groundwater model that are based on the results of this study could improve the results considerably. This would mean having a feedback between groundwater inflows and outflows and the lake water budget as calculated with MODGLUE. This would allow for inventions of climate change and lake evaporation in addition to groundwater recharge. Furthermore, a groundwater model that is capable of output lake inflows and outflows separate for each aquifer would ease the assignment of groundwater quality to inflows.
- Bathymetry data are very important to calculate an accurate volume of the lakes and to get reliable representation of spatial distribution of shallow and deep areas of a lake.
    - Bathymetry data for Lake Stockton was obtained during this study but can be considered estimates and needs further refinement to improve model representation of the real lake bathymetry. Bathymetry data should also be collected for other pit lakes (particularly historic lakes) so that these are able to be modelled.
- The relative contributions to the pit lake acidity budgets from surface and groundwaters across the different pit lake types are still not well understood.
    - NAG/NAP testing and characterisation of pit lake soils and sub-soils may help better explain these sources of continuing incoming acidity

as well as assist in predictions of future acidity budgets and water qualities.

- An additional source of acidity was identified as a result of the modelling. Model runs with leaching from sub-aquatic sediments based on measured rates and compositions of leached materials from the Lake Zwenkau case study produced good agreement of measured and calculated pH values, especially the decease of pH after a temporarily increase could be reproduced.

    - To make these models more reliable, sediment samplings from different locations of the Collie pit lake bottoms are strongly recommended. The sampled sediments need to be analysed for physical and geochemical properties such as active porosity, composition of pore water and cation exchange capacity of the materials. If possible, samples should be taken as columns to get information about vertical distribution of these properties. This can provide a basis for estimating the total amount of acidity in the sediments in addition to the rate of its release and composition over time. In the same manner as the subaquatic sediments, bank material should be analysed to identify zones for which erosion rates and their impact on lake water quality can be modelled.

- Remediation of lakes by alkaline material addition has been both attempted and reported in previous studies (Lund & McCullough, 2009). The effects of lake treatment with different alkaline substances can be represented in the model. Required model inputs for this task are amount and schedule of addition as well as chemical composition of the added substance. The substance efficiency in the chemical reactions is less than 100% precent. There can be many reasons for this such as partial solution of solids, reaction with other species and resulting precipitation or sedimentation before the chemical reaction is complete.

    - Remediation strategies should be investigated further in laboratory experiments and considered by future modelling studies.

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