Government of **Western Australia**
Department of **Water**

Looking after all our water needs

Perth Shallow Groundwater Systems Investigation

Lake Gwelup

Hydrogeological record series

Report no. HG56
September 2012

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The western edge of Lake Gwelup, near a staff gauge. Photo taken: 3 September 2010.

Disclaimer

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Summary

Lake Gwelup is a groundwater-dependent flow-through wetland located in a swale of the Tamala Limestone on the Gnangara Mound, approximately 11 km north of Perth, Western Australia. Lake Gwelup is a conservation category and EPP (Environmental protection policy) wetland that supports rare vegetation and significant faunal habitats (Hill et al. 1996, Froend et al. 2004a, McHugh & Bourke 2007).

Lake Gwelup was identified as one of 39 prioritised sites in the management area review of shallow groundwater systems on Gnangara and Jandakot mounds. The lake is one of 28 sites examined in the Perth shallow groundwater system (SGS) investigation.

An interdependent relationship exists between water levels, flora and fauna. The preservation of Lake Gwelup as a key faunal habitat relies on the conservation of groundwater-dependent vegetation, which in turn relies on the maintenance of groundwater levels (DoW 2009b).

Lake Gwelup Reserve has a high social amenity given the accessibility of the lake in a highly urbanised area. The lake is ranked as a Category 2 wetland for in situ social values. This is the second highest ranking for in situ social values. These social values exist because:

- there is a large regional and local visitor catchment of the reserve
- the reserve is actively managed
- a significant portion of the reserve is used for public open space (Beckwith Environmental Planning Pty Ltd 2006).

It is important to link the management of ecological attributes with the maintenance of social values (DoW 2009b). The high social value of Lake Gwelup increases the importance of maintaining ecological values.

A drying climate, groundwater abstraction and surrounding land uses (such as historical horticultural activities and urbanisation) have had an adverse effect on the ecology of Lake Gwelup (Froend et al. 2004a). Risks to environmental values at Lake Gwelup are due to watertable decline and water quality degradation. The management area review recommended a hydrogeological investigation to assess the current interaction between groundwater and surface water at the lake and provide recommendations to assist its management.

The Perth SGS investigation included the installation of groundwater monitoring bores at the western and eastern edges of Lake Gwelup at varying depths in the Superficial aquifer. A 12-month groundwater monitoring program was undertaken between June 2009 and June 2010.

The Department of Environment and Conservation (DEC) conducted soil and groundwater acidity investigations approximately 400 m up-gradient of Lake Gwelup, within its groundwater catchment zone, between 2006 and 2010. The results of these investigations were used to complement the data from the SGS investigation.

Lake Gwelup has reverted to an ephemeral wetland, which it once was, rather than a permanently inundated wetland. The lake has dried each summer since 2001. The available water level records indicate the lake has dried on only three occasions in the preceding 80 years. This change is due to declining groundwater levels from groundwater abstraction and reduced rainfall. However, hydrographs of groundwater monitoring bores near Lake Gwelup (Appendix G) show groundwater levels have ceased declining at rates observed in the 1990s. This change appears to correlate with reduced groundwater abstraction from the Superficial and Mirrabooka aquifers (Figure 3).

When the lake is full, it interacts with groundwater in the upper half of the Superficial aquifer. This interaction ceases when the lake is dry. The lake acts as a groundwater flow-through system when full, with a strong vertical component to groundwater flow. Upward groundwater flow is evident at the eastern edge of the lake and downward groundwater flow is evident at the western edge of the lake. When the lake is dry, groundwater movement is dominated by horizontal flow in an east to west direction beneath the lake.

Groundwater quality at Lake Gwelup is affected by the volume and quality of groundwater flowing into the lake from the capture zone, inflows from surrounding drains and the oxidation of lake sediments, when the lake is dry.

The drying of the lake exacerbates the oxidation of lake sediments. This adversely affects the quality of groundwater flowing out of Lake Gwelup. The pH of groundwater near the watertable at the western edge of the lake was lower than other SGS bores at the lake and lower than the preferred range for freshwater ecosystems (pH 6.5–8.5). Groundwater in the lower half of the Superficial aquifer does not interact with the lake, therefore the oxidation of lake sediments is not likely to affect the quality of water at these depths.

Elevated nitrogen levels were recorded in the upper half of the Superficial aquifer beneath the western edge of the lake. This may be a result of internal eutrophication. This is the release of previously sorbed phosphorus and nitrogen from sediments as a result of drying, oxidation and the degradation of organic matter. This process was also inferred from investigations conducted at Lake Mariginiup (Searle et al. 2010). The sediments at the base of Lake Gwelup may have acted as a ‘sink’ for nutrient runoff during the horticultural dominated era between 1880 and 1970 and possibly from unsewered urban development, which was common until the 1970s. Elevated levels of nutrients were also recorded near the watertable up-gradient (east) of the lake. The increased disconnection between the lake and the watertable will lead to a diminishing influence of the lake on down-gradient water quality.

The Delawney Drain, Brushfield Wetland and groundwater inflow transports water with high concentrations of acidity, iron, sulfate and sulfate to chloride ratios (> 0.5). This is the main reason for lower surface water pH at the eastern edge of the lake near the drain discharge area. These conditions may affect bioavailable concentrations of heavy metals and affect ecological values and biological diversity (ANZECC & ARMCANZ 2000). The increased drying frequency and lower connection between surface water and groundwater may reflect an increased significance of surface water drainage inflow to the lake.

Sulfide (pyrite) oxidation is an acid generating process that has affected the quality of groundwater beneath Lake Gwelup and its capture zone. The management of acid sulfate soils is significant in maintaining environmental and social values at Lake Gwelup (Beckwith Environmental Planning Pty Ltd 2006). High sulfate concentrations, sulfate to chloride ratios (>0.5) and dissolved iron concentrations were recorded in groundwater at Lake Gwelup and its capture zone. Isolated areas of acidic groundwater and high concentrations of arsenic and aluminium were also recorded up-gradient of the lake. These conditions pose a risk to the long-term management of Lake Gwelup. They also reinforce the need for users of untreated groundwater from domestic bores to test the quality of groundwater to ensure it is suitable for use.

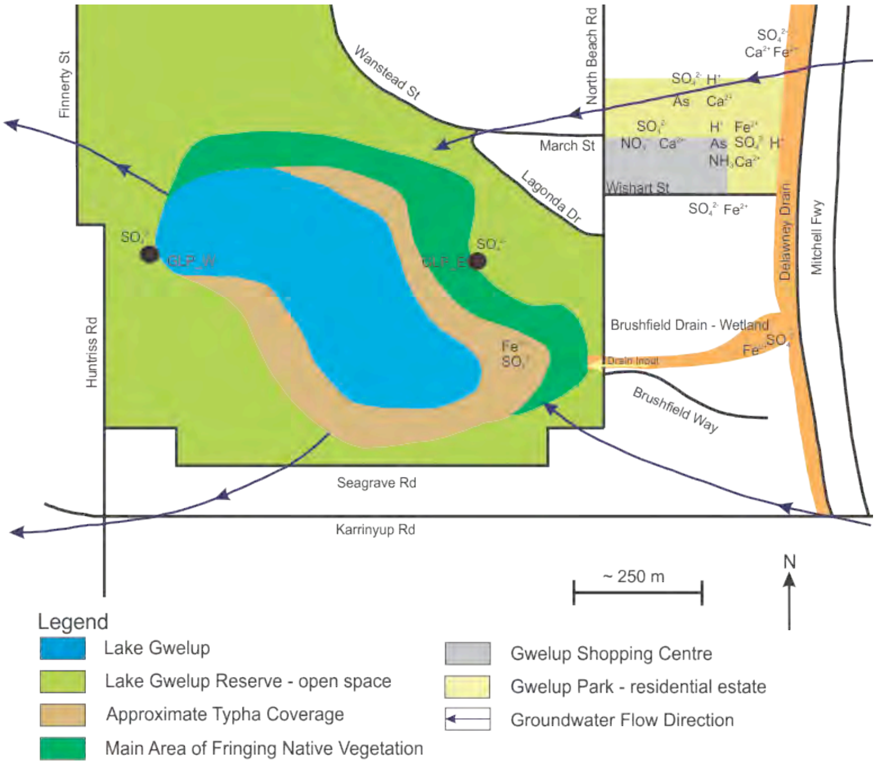
Sulfide oxidation may be triggered from dewatering and soil excavation associated with residential development or groundwater abstraction for private and public supply. Other Perth SGS investigations have found pyrite oxidation has affected the quality of groundwater and presents a risk to the ecological value of wetlands on the Gnangara Mound, including Lake Mariginiup, Egerton Seepage and the Lexia Wetlands (Searle et al. 2010, McHugh et al. 2011, Department of Water 2011).

High sulfate and iron concentrations were recorded to the base of the Superficial aquifer beneath the eastern edge of Lake Gwelup. This indicates a widespread groundwater quality issue in the Gwelup area, that has come from up-gradient of the lake. These conditions may be a legacy of up-gradient groundwater abstraction for public supply and historical horticultural activities.

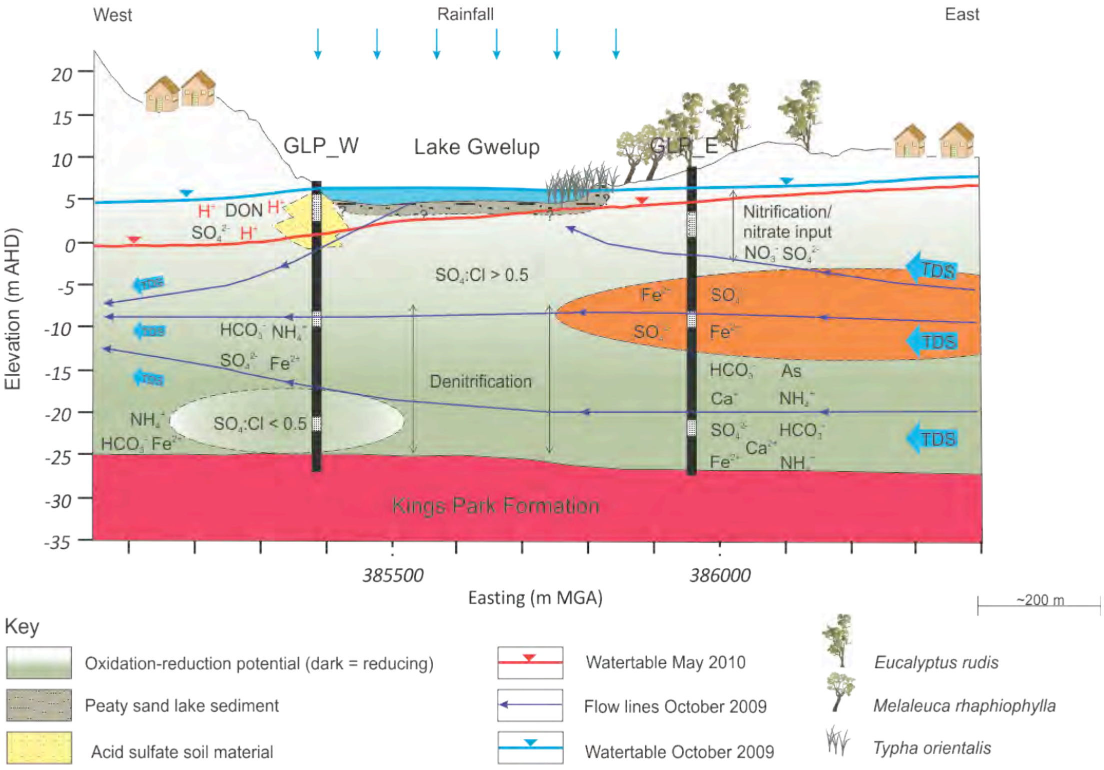
Water levels in Lake Gwelup need to be maintained to prevent the oxidation of lake sediments, to maintain sediment processes and to limit acidification and internal eutrophication processes. The quality of groundwater and drainage flowing into Lake Gwelup has a significant effect on the quality of water in the lake and on the maintenance of environmental and social values.

This study provides an improved conceptual understanding of the relationships between the hydrogeology and wetland ecosystem of Lake Gwelup (see following page). These conceptual models, discussed in detail in Section 6 (figures 53 and 54), and subsequent recommendations (Section 7), provide a basis for improving management strategies.

Recommendations for improved management and for further monitoring and investigation are detailed in Section 7 of this report. Management actions are concerned with improving the quantity and quality of surface water and groundwater inflow to Lake Gwelup. This includes a review of public water supply reserves and the environmental sensitivity classification of production bores in the Gwelup borefield. Further monitoring and investigation of the quality of groundwater migrating towards Lake Gwelup also needs to be conducted on a consistent basis. (see Section 7 for Recommendations in full).



Conceptual diagram of groundwater and surface water flow and quality near the watertable (groundwater flow when lake contains water, October 2009)



Conceptual hydrogeological diagram beneath Lake Gwelup

1 Introduction

1.1 Context and aims

Lake Gwelup is located 11 km north of Perth in the densely urbanised suburb of Gwelup. It supports rare wetland vegetation and has high social and recreational value. Lake Gwelup is located down-gradient of nearby production bores of the Gwelup bore field and is considered at significant risk from drawdown (McHugh & Bourke 2007). Acidic groundwater near the watertable, and high levels of arsenic and other metals such as aluminium have been recorded in the Gwelup area (Appleyard et al. 2006). These conditions may pose a risk to the ecological value of Lake Gwelup. A management area review (MAR) recommended a hydrogeological investigation to gain an understanding of:

- surface water–groundwater interaction at Lake Gwelup
- the quality of groundwater, surface water and drainage inputs to the lake.

Lake Gwelup is a 'flow-through wetland', and is a surface expression of groundwater of the Superficial aquifer of the Gnangara Mound on the Swan Coastal Plain. The Gnangara groundwater system (a term that refers to the Superficial aquifer and deeper Mirrabooka, Leederville and Yarragadee aquifers) is Perth's most significant water resource. It provides approximately half of Perth's potable water through the Integrated Water Supply Scheme (IWSS) (Department of Water 2009). The population of the Perth metropolitan area has almost doubled in the last 30 years, leading to a significant increase in water demand and increased stress on the Gnangara Mound.

Groundwater level decline on a regional scale throughout much of the Gnangara Mound is mainly due to a combination of groundwater abstraction and reduced rainfall since the 1970s (Yesertener 2008). Groundwater level decline may instigate the oxidation of naturally occurring iron sulfide minerals such as pyrite. This is an acid generating process that has affected the quality of groundwater near the watertable in Gwelup (Appleyard et al. 2006). Low pH (pH < 5) increases the risk of metal mobilisation, that are otherwise bound to sediment. Previous soil and groundwater investigations (Appleyard et al. 2004; Appleyard et al. 2006) have identified acidification in the City of Stirling. These processes have been linked to urban development. Dewatering and soil excavation for residential development projects are required for the installation of sewers and the construction of building footings. Previous groundwater investigations have recorded high concentrations of aluminium, arsenic and iron in conjunction with acid conditions in Gwelup. These conditions can pose a risk to human health as well as the environment (Appleyard et al. 2004).

Historical horticultural practices such as market gardening dominated the Gwelup landscape until the 1970s. These horticultural activities, due to significant fertiliser use, may have left a legacy of elevated nutrient levels in groundwater. Elevated nutrients can affect some native vegetation species if they are groundwater dependent, but may also result in elevated nutrient levels in wetlands interacting with the Superficial aquifer.

The installation of production bores in the Gnangara groundwater system for public water supply began in the 1970s. Groundwater abstraction from the Gwelup bore field began in 1974, located approximately 1 km up-hydraulic gradient (east) of Lake Gwelup. There has been a progressive shift in abstraction for public water supply from the Superficial aquifer to the Leederville and Yarragadee confined aquifers. This shift was made to limit the effect of groundwater abstraction on watertable decline and the quality of groundwater in the Superficial aquifer. Previous investigations have identified the degradation of groundwater quality in the lower half of the Superficial aquifer in production bores of the Gwelup bore field since the 1970s. The deterioration of groundwater quality is characterised by increasing concentrations of sulfate, calcium, iron and increasing sulfate to chloride ratios, probably due to pyrite oxidation (Appleyard et al. 2006).

1.2 Aims

This report aims to improve the understanding of the hydrogeology and hydrogeochemistry around Lake Gwelup. Specifically, the aims of the study are to:

- describe groundwater flow regimes around Lake Gwelup, showing seasonal trends between 2009 and 2010
- provide an understanding of the quality of groundwater beneath Lake Gwelup, extending to the base of the Superficial aquifer
- provide an assessment of the quality of groundwater flowing into and out of Lake Gwelup
- provide an assessment of the effects of drainage, climate and groundwater abstraction on water levels at Lake Gwelup
- determine the extent and severity of groundwater acidification up-hydraulic gradient of Lake Gwelup
- investigate the extent of arsenic contamination in shallow groundwater up-hydraulic gradient of Lake Gwelup
- develop a conceptual model of the relationships between wetland hydrogeology, chemistry and ecosystem function.

2 Background

2.1 Location

Lake Gwelup is located in the City of Stirling, 11 km north of Perth (Figure 1). It is 25 ha in area and comprises about a third of Lake Gwelup Reserve. The reserve is composed of native bushland (Bush Forever Site 212) and public open space (Beckwith Environmental Planning Pty Ltd 2006).

2.2 Climate

Temperature

The south-west of Western Australia has a Mediterranean-type climate (Csa climate; Kottek et al. 2006) with hot dry summers and cool wet winters. Rainfall only exceeds evaporation for about four months of the year. Since 1944, according to the Perth Airport weather station, 009021, the average summer maximum is 30.8 °C, and the average autumn, winter and spring maximum temperatures are 25.6 °C, 18.4 °C and 22.8 °C respectively (Bureau of Meteorology 2011).

Rainfall

Land-use changes, ocean temperatures, natural variability and circulation changes including the strengthening of the southern annular mode are likely contributing factors to the decline in annual rainfall in the south-west of Western Australia since the 1970s (van Ommen & Morgan 2010). However the strength and relative importance of each of these individual factors remains uncertain (Hope & Foster 2005; van Ommen & Morgan 2010). Rainfall data for the Perth Airport weather station shows a declining trend in rainfall over a 54-year monitoring period (Figure 2). The long-term average annual rainfall at the Perth Airport is 779 mm. Average annual rainfall between 1945 and 1975 was 836 mm, which declined by 14% to an average of 726 mm between 1976 and 2009.

Declining winter rainfall has resulted in a significant reduction in surface water inflow to dams used as a source of public water supply under the Integrated Water Supply Scheme (IWSS). Surface water inflow to dams has declined by more than two-thirds since the 1970s, as shown below (Water Corporation 2005).

Period	Surface water inflow to dams GL/yr
1911 to 1974	338
1975 to 2000	177
2001 to 2005	93
2006 to 2010	75

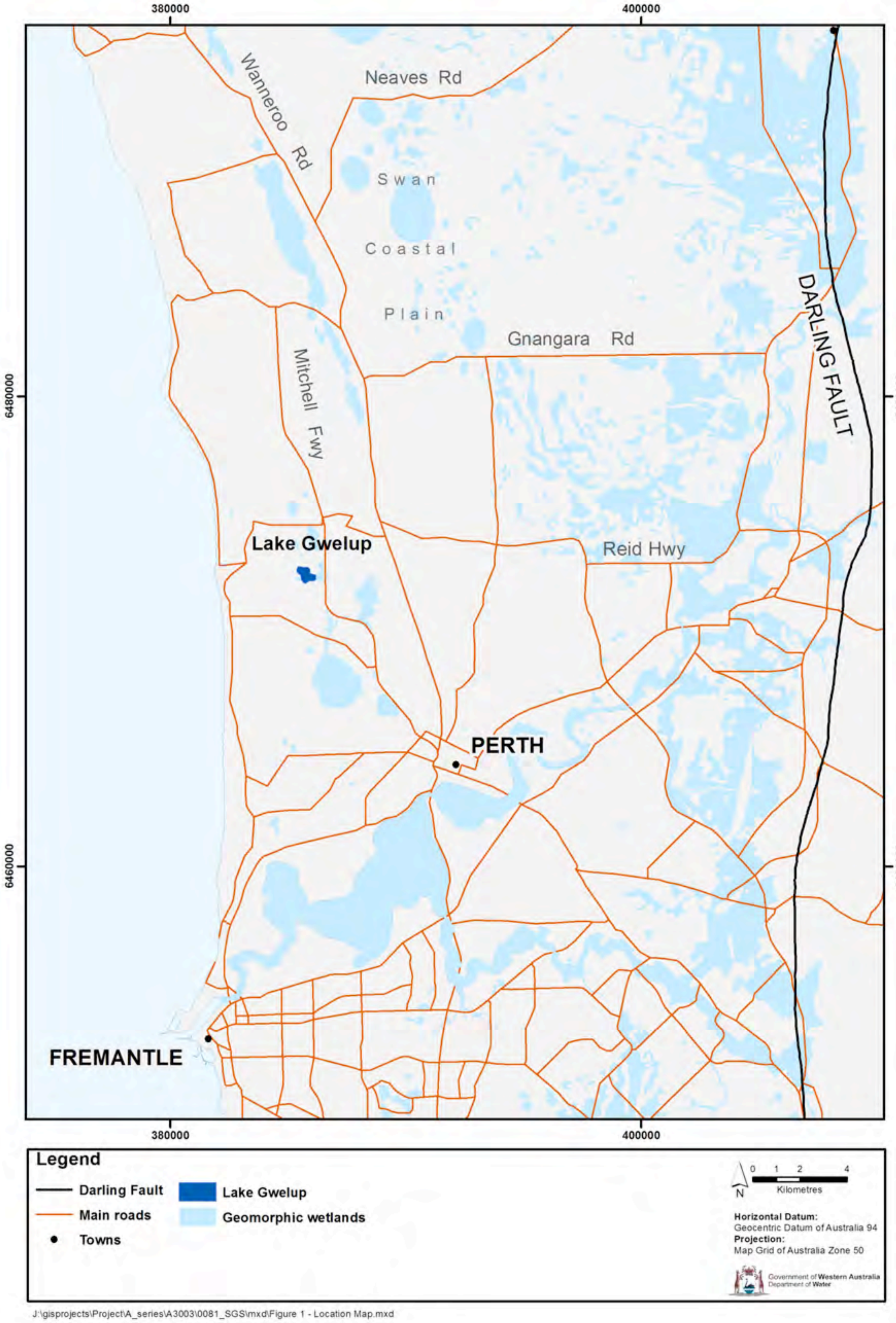


Figure 1 Regional location of Lake Gwelup

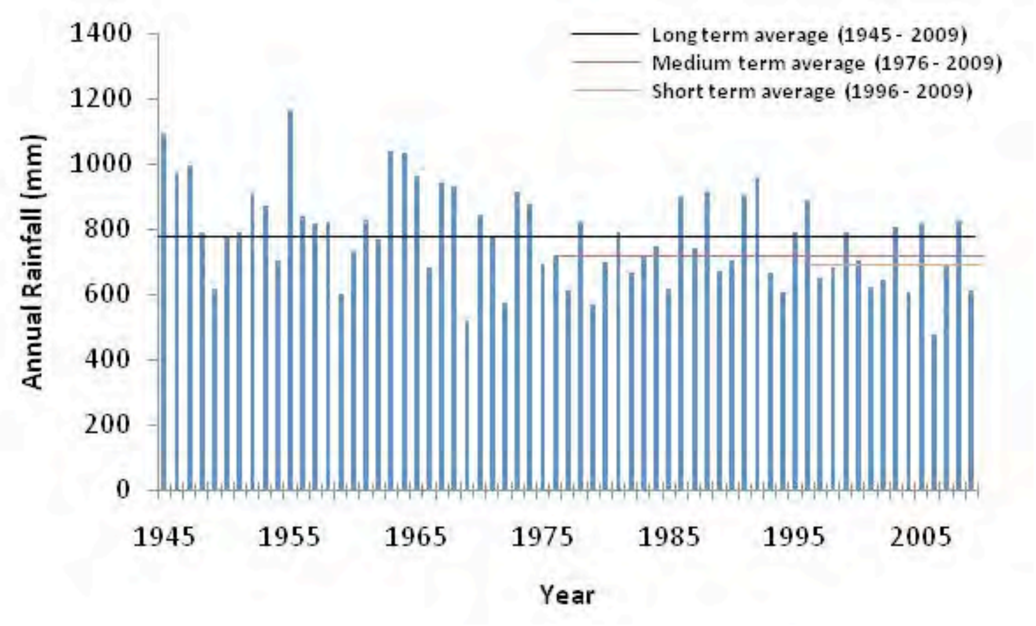


Figure 2 Annual rainfall from the Perth Airport weather station

Perth recorded its second lowest annual rainfall on record in 2010 (483.4 mm). This yielded 12.7 GL of stream inflow to IWSS reservoirs, which is the lowest on record (Water Corporation 2010). The decline in water stored in IWSS reservoirs has increased the need to develop alternative sources of water, and has led to increased abstraction from the Gnangara groundwater system.

CSIRO has undertaken long-term predictive modelling of past and future winter rainfall over the south-west of Western Australia. The modelling suggests that over a 500-year period, with stable greenhouse gas conditions, natural variability can produce periods of rainfall decline such as that observed since the 1970s. However, incorporating the projected increased greenhouse gas trends, winter rainfall is predicted to further decline below levels currently considered to be within natural variations (IOCI 2005).

A recent study has indicated an inverse relationship between precipitation at Law Dome, eastern Antarctica and the south-west of Western Australia. The precipitation record at Law Dome was effectively a record of preserved snow accumulation, converted to a water equivalent using a density model (van Ommen & Morgan 2010). This record period for Law Dome extends over a 750-year period. The Law Dome precipitation series shows a positive anomaly (increased precipitation compared to long-term mean), which is statistically more significant (using a t-test) than the other 57 anomalies recorded in the 750-year period. The comparison between Law Dome and the south-west of Western Australia is considered valid due to the role of the meridional circulation south of Australia which produces cool, dry air to the south-west of Western Australia while at the same time producing warm, moist air to eastern Antarctica (van Ommen & Morgan 2010). The study concluded that increased precipitation since the 1970s at Law Dome is outside the range expected due to natural variability, and therefore supporting the theory of anthropogenic induced climate change. It is therefore inferred that the south-west of Western Australia may be experiencing its most extended period of dry weather (drought) in at least 750 years given the observed relationship with precipitation at Law Dome (van Ommen & Morgan 2010).

Extended periods of drought, in addition to changes in land and water use, contribute to the changing condition of wetlands. This effect has been observed since at least 1878, when Lake Gwelup was known as ‘Gwelup Swamp’, drying on a seasonal basis (Moore 2001). However, when the surrounding area was cleared, increased groundwater recharge and rising annual average rainfall between 1914 and 1920 caused groundwater levels to rise. This caused the swamp to become a more permanent water body and was thus renamed. The same pattern of urban development is also evident around Lake Claremont (once referred as Butlers Swamp) and Perry Lakes (previously known as Perry Swamps). Recent declining rainfall and reduced groundwater levels have caused some lakes on the Swan Coastal Plain to revert to an ephemeral condition (IOCI 2005). This, in conjunction with groundwater abstraction, appears to be the main reason why Lake Gwelup has reverted to an ephemeral wetland.

2.3 Land use

1880 to 1970 - horticulture

Oral histories compiled by the Gwelup Progress Association provide an interesting account of the historical horticultural activities that dominated the Gwelup area for about 90 years until the commencement of urbanisation in the 1970s.

The City of Stirling was identified as ‘swamp land that people could crop three times a year’ and was generally thought to be a productive horticultural precinct. The presence of peat in the City of Stirling was identified long before the onset of urbanisation and was crucial to the horticultural productivity of the area. Sand-dominated landscapes were considered ‘worthless’ as there was no irrigation in the area prior to the late 1930s and 1940s, making it difficult to produce a consistent crop on these nutrient poor sands. Upon the construction of irrigation bores, horticultural activities expanded to the ‘sanded land’, which was also much cheaper and easier to work than the ‘swamp land’ as horses and machinery could be used (Moore 2001).

Farming practices associated with horticultural activities in the Gwelup area comprised the burning of peat (Moore 2001) and the use of fertilisers. The burning of peat may have released significant amounts of stored acidity, perhaps adversely affecting the quality of groundwater prior to the onset of urbanisation. Fertiliser use during this horticultural era contributed to the presence of elevated nutrient concentrations in groundwater in combination with the commencement of unsewered urban development (Barber et al. 1993). Nitrate concentrations in the lower half of the Superficial aquifer, where production bores are screened, have declined since the late 1980s which has corresponded to an increasing trend in concentrations of ammonia, consistent with the process of denitrification (Appleyard et al. 2006).

1970 to current – urbanisation

Urban development began to intensify in the 1970s, increasing from 20% to 40% of the Gwelup area in a five-year period (Appleyard et al. 2006). Historical horticultural activities and the construction of additional unsewered urban development in the 1970s are likely to have been the dominant sources of nutrient related groundwater contamination. Progressive installation of reticulated sewage infrastructure in the 1980s led to the removal of most septic tanks and approximately 90% of the area covered is sewered residential development (Appleyard et al. 2006).

Urbanisation may result in higher rates of groundwater recharge (depending on urban density and location of stormwater discharge) through increased infiltration in soak wells and groundwater compensation basins. The high proportion of impermeable areas such as infrastructure, bitumen road reserves and concrete footpaths correspond with greater volumes of runoff. Land clearing activities also cause a reduction in water loss by transpiration (Appleyard et al. 2006). These factors are likely to have contributed to an increase in watertable elevation upon urbanisation in Gwelup. However, since groundwater abstraction began in conjunction with a decline in annual average rainfall, groundwater levels have declined. Based on results presented by Yesertener (2008) groundwater abstraction appears to be the main cause of groundwater level decline in the Gwelup area. Total groundwater level decline up to 3.75 m is inferred in a 6 km radius of the Gwelup bore field between 1979 and 2005, and approximately 3.0 m of this is attributed to groundwater abstraction (Yesertener 2008).

2.4 Water use

Groundwater abstraction

Groundwater is abstracted from the Gwelup groundwater area for public water supply, private licensed use and domestic bore use. Public water supply abstraction is from the Superficial, Mirrabooka, Leederville and Yarragadee aquifers. Water for public supply is provided under the Integrated Water Supply Scheme (IWSS). Private licensed and domestic bore use is from the Superficial aquifer.

The Leederville and Yarragadee aquifers are currently over-allocated in the Gwelup groundwater area (Table 1) (Department of Water 2009). There is no evidence to suggest this over-allocation has affected groundwater levels in the Superficial aquifer or water levels at Lake Gwelup (Figure 4). Licensed allocations from the Superficial and Mirrabooka aquifers are below allocation limits. Table 1 shows licensed entitlements for private use and public supply (IWSS) in the Gwelup groundwater area. It does not include private licensed abstraction outside the boundary of the area.

Table 1 Groundwater use in the Gwelup groundwater area, 2010–11 financial year. All units GL/year.

Aquifer	**Allocation limit**	**IWSS entitlement at 145 GL/yr baseline**	**Private licensed entitlements**	**IWSS abstraction**	**Total estimated abstraction**
Superficial aquifer	7.95	2.65	1.65	4.76	6.41
Mirrabooka aquifer	3.60	3.35	-	3.83	3.83
Leederville aquifer	5.30	7.00	-	7.49	7.49
Yarragadee aquifer	7.50	7.60	-	8.69	8.69

Public water supply bores on the Gnangara groundwater system

The volume of groundwater allocated for public supply has historically been based on a revised ‘variable groundwater abstraction rule’ (VGAR) (Department of Water 2009). The volume of water in dams directly relates to the amount of groundwater abstracted from production bores. The volume of water obtained from production bores is constrained by environmental sensitivity classifications to manage the effects of groundwater abstraction.

IWSS production bores are assigned an environmental sensitivity classification (ESC) of 1, 2 or 3. Production bores in the Gwelup bore field are all assigned an ESC of 3. This is considered the least sensitive classification and is associated with urban and coastal bore fields. ESC3 bores are located away from the recharge zone. The recharge zone is an area of the Gnangara Mound where groundwater levels are highly dependent on rainfall and less sensitive to drawdown (Department of Water 2010). Sensitivity classifications act as a guide for the magnitude of allowable increases in groundwater abstraction above the baseline quota of an individual bore. For example, abstraction from ESC3 bores can increase by 25% above the licensed quota of the bore, though any increase in abstraction from an individual bore must be within the total licensed allocation. ESC1 bores (e.g. the southern portion of the Pinjar bore field) are considered most sensitive to drawdown and no additional increase in abstraction from the licensed quota is permitted. ESC2 bores (e.g. the northern portion of the Pinjar bore field) correspond with a potential 10% increase in abstraction under certain circumstances. The low sensitivity classification of Superficial and Mirrabooka bores in the Gwelup bore field has resulted in abstraction from these bores exceeding their licensed quotas in most years. This low sensitivity classification may be due to previous assessments inferring a greater significance of drainage input recharging Lake Gwelup than regional groundwater levels and groundwater inflow to the lake.

The maximum annual groundwater allocation for the IWSS is 145 GL/yr. However, in exceptional circumstances this limit may be raised to 165 GL/yr. Exceptional circumstances rules may be triggered by low rainfall resulting in low water storage in IWSS reservoirs. This was the case in the financial year 2010–11 when 165 GL/yr was required, and in 2011–12 when 159 GL of groundwater was abstracted by production bores. The Water Corporation's baseline licensed allocation, from existing infrastructure, will be reduced to 120 GL/yr from 2012–13 (Department of Water 2009).

The Gwelup bore field

Groundwater for public supply is abstracted from the Gwelup bore field in the Gwelup groundwater area (GWA: Figure 5). The Gwelup bore field comprises bores in the unconfined Superficial aquifer, the semi-confined Mirrabooka aquifer and the deeper Leederville and Yarragadee aquifers. The cumulative volume of groundwater pumped from the Gwelup bore field has more than doubled since it was commissioned in 1974. This increase is mainly due to an increase in abstraction from the Leederville and Yarragadee aquifers, particularly since 2002 (Figure 4).

In the Gwelup area the Superficial and Leederville aquifers are separated by approximately 20 m of Kardinya Shale, acting as a confining layer. The Leederville and Yarragadee aquifers are also separated by approximately 100 m of South Perth Shale, also acting as a confining layer (Davidson 1995). This suggests abstraction from the Leederville and Yarragadee aquifers is unlikely to affect groundwater levels in the Superficial aquifer or lake levels.

Since 2002 groundwater abstraction from the Superficial and Mirrabooka aquifers has declined or eased and abstraction from the Leederville and Yarragadee aquifers has increased (figures 3 and 4). This period has coincided with groundwater levels near the watertable of the Superficial aquifer in the Gwelup area ceasing to decline at rates recorded in the 1990s, despite continued low rainfall (figures 3 and 4 and Appendix G). For example, since 2002 groundwater levels at GM17 (approximately 200 m south of Lake Gwelup; see figures 33 and 34) have stabilised and groundwater levels at GM13 (located approximately 500 m east of Lake Gwelup) have increased (figures 3 and 4). This highlights the influence of abstraction from the unconfined Superficial aquifer and semi-confined Mirrabooka aquifer on groundwater levels in the Gwelup area and groundwater flow towards Lake Gwelup.

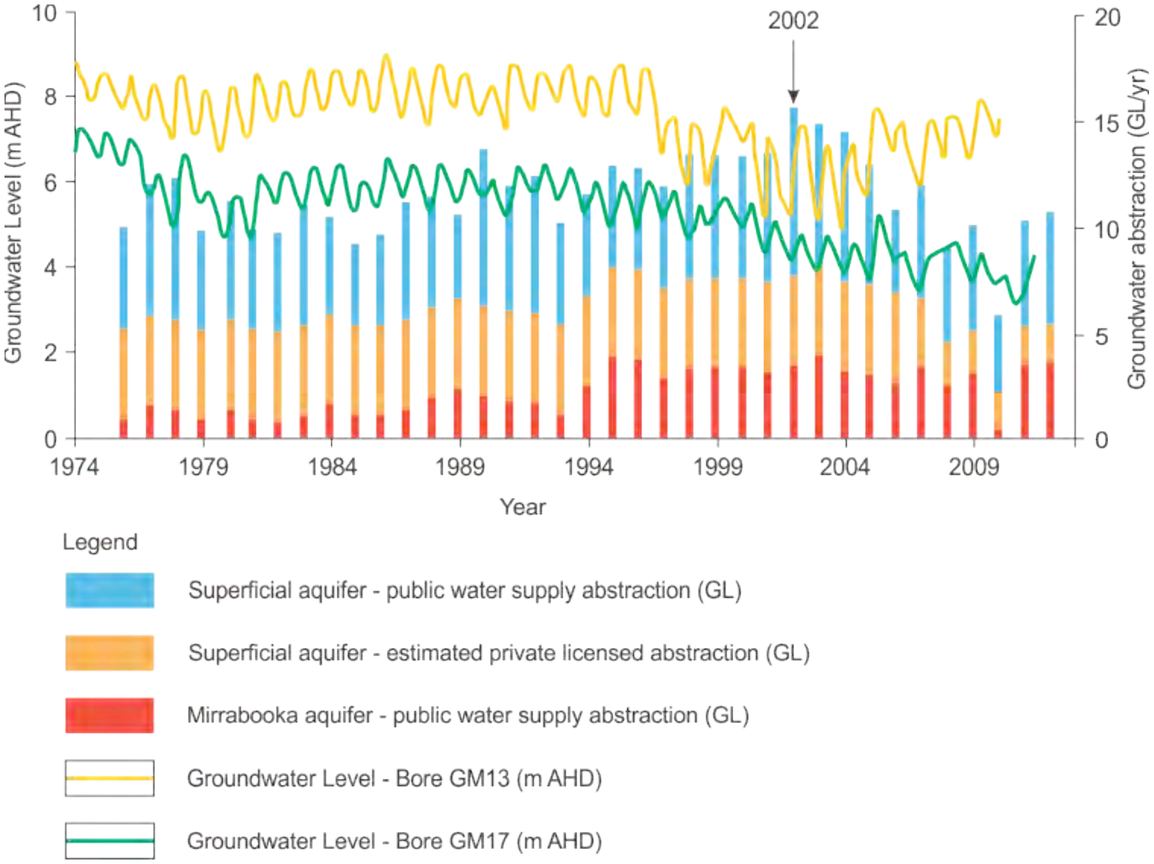


Figure 3 Annual groundwater abstraction from the Superficial and Mirrabooka aquifers in the Gwelup groundwater area. Private licensed abstraction is estimated based on licensed entitlements. Groundwater monitoring bores in the capture zone of Lake Gwelup are screened near the watertable of the Superficial aquifer.

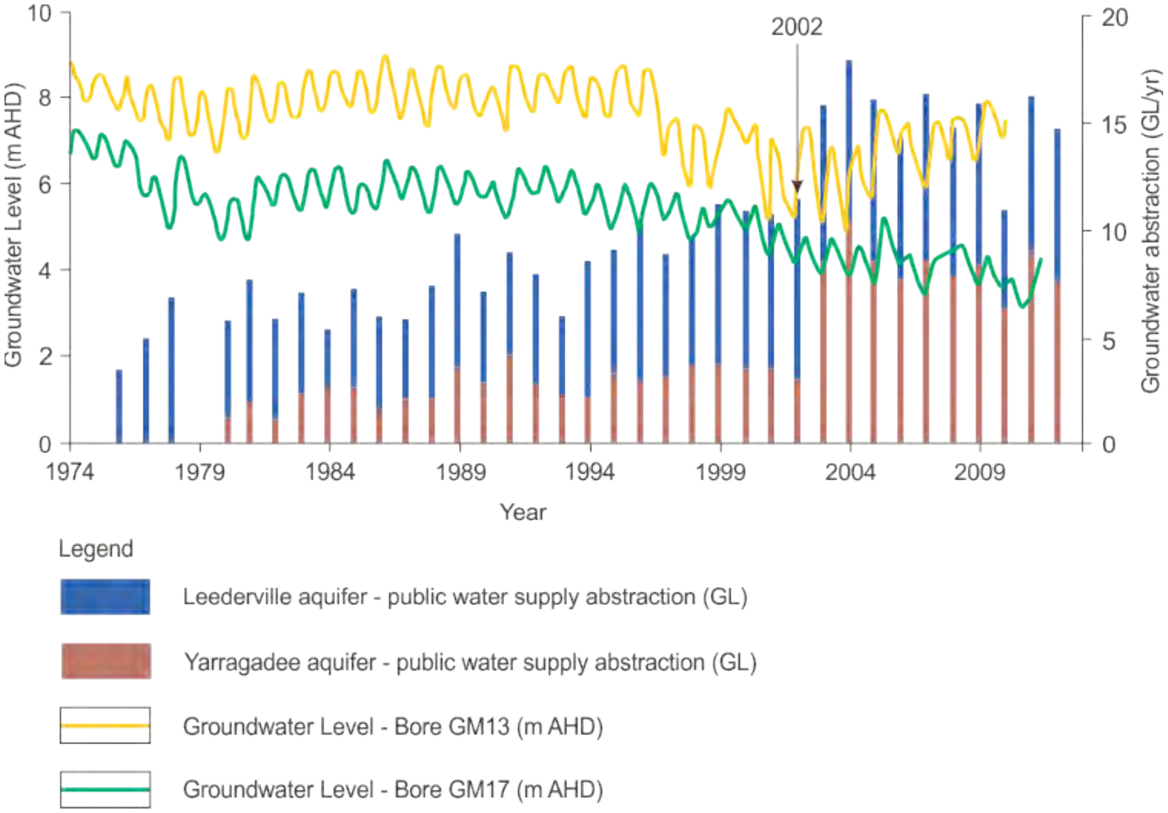


Figure 4 Annual groundwater abstraction in the Leederville and Yarragadee aquifers of the Gwelup bore field. Groundwater monitoring bores in the capture zone of Lake Gwelup, screened near the watertable of the Superficial aquifer.

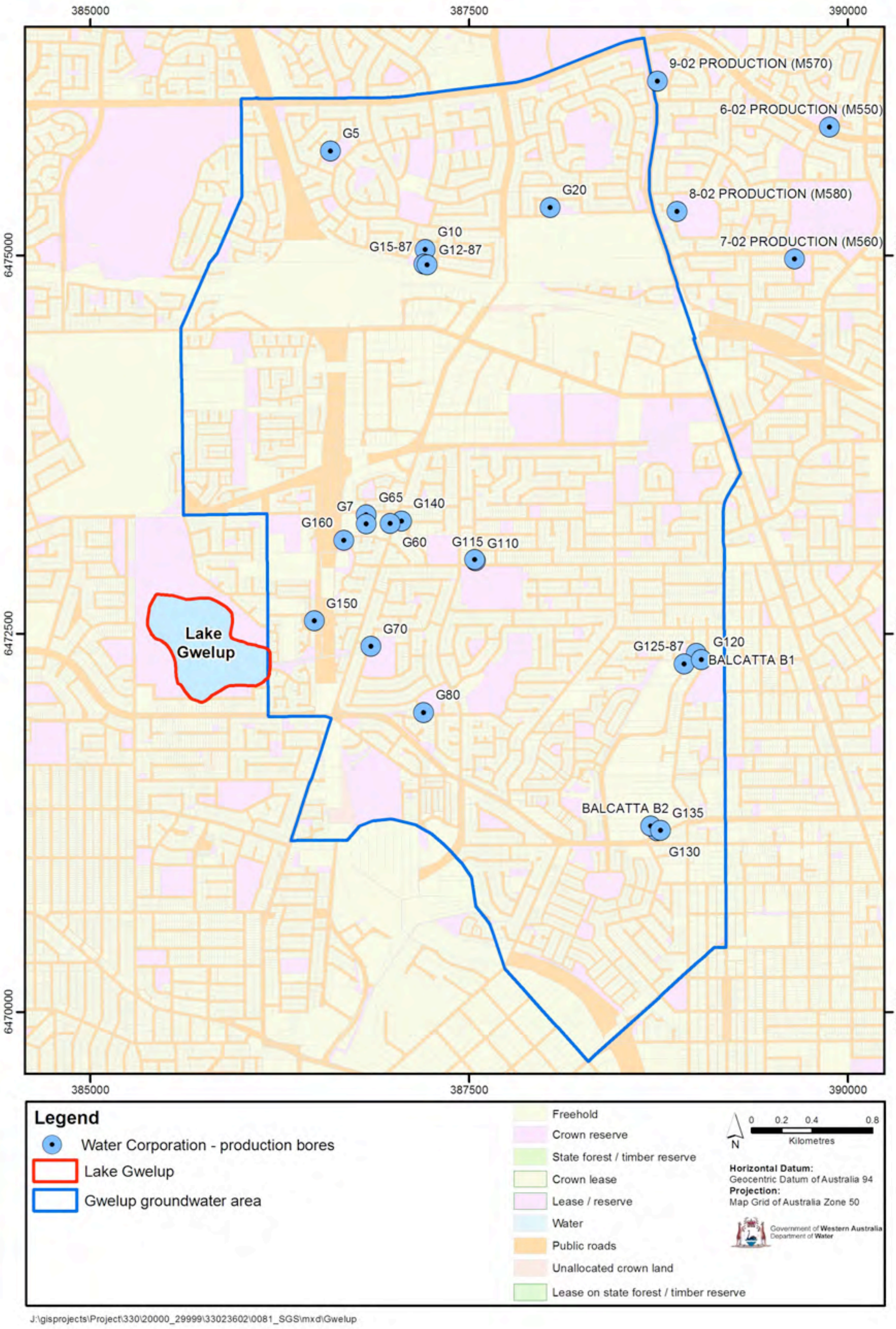


Figure 5 Groundwater abstraction from the Gwelup bore field (all aquifers)

Private licensed bores

Licensed groundwater abstraction is regulated by the Department of Water. Numerous groundwater licences for groundwater abstraction are provided up-gradient and down-gradient of Lake Gwelup (Figure 6). Private licensed entitlements (from the Superficial aquifer) in the Gwelup groundwater area totalled 1.65 GL in 2010–11.

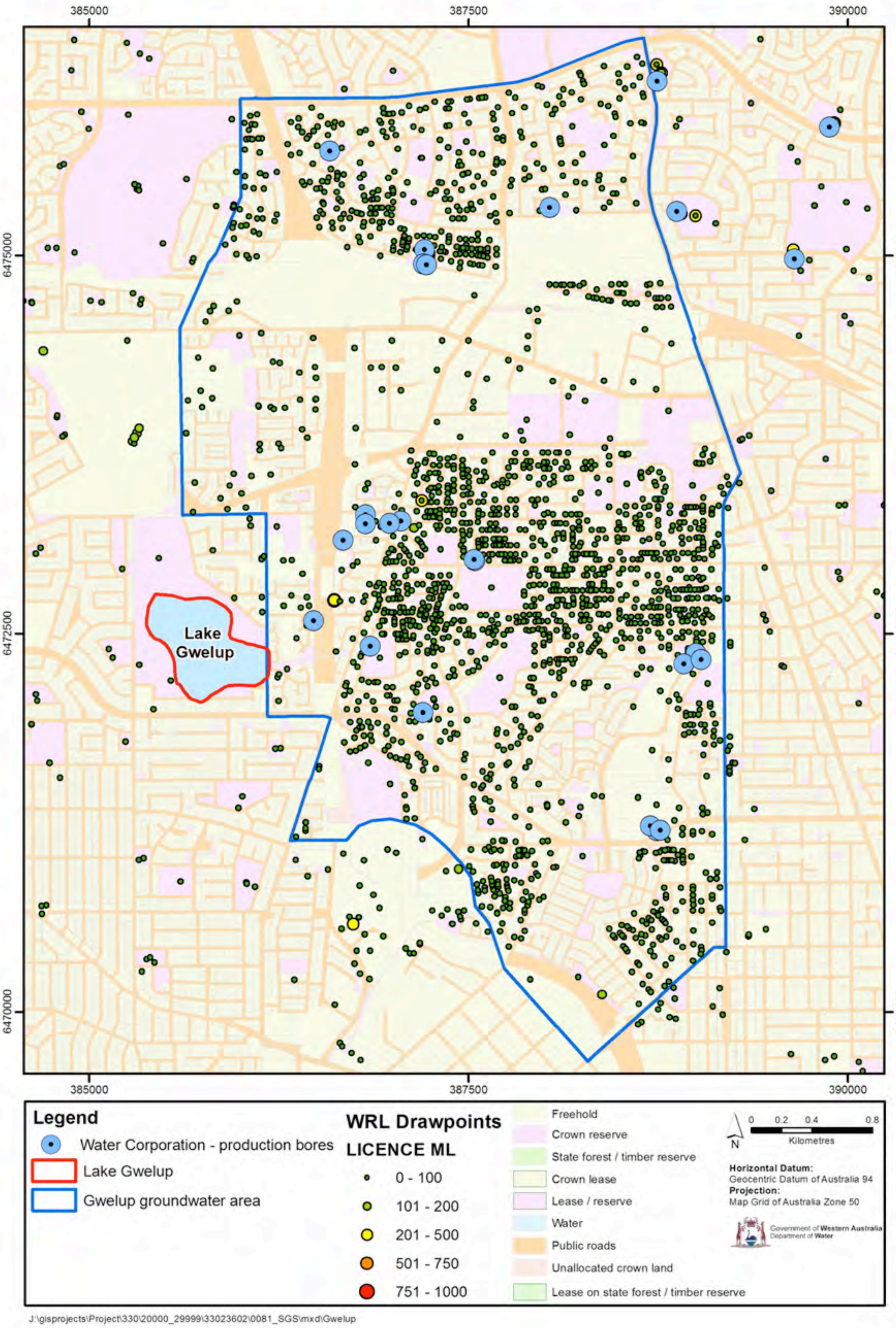


Figure 6 All licensed groundwater abstraction

Unlicensed domestic bores

Groundwater abstraction from domestic bores is difficult to accurately quantify because domestic bore use is not licensed in Western Australia. Approximately 72 500 homes are estimated to operate an unlicensed domestic groundwater bore within the Gnangara groundwater plan area, abstracting a cumulative total of about 58 GL/yr (18% of annual groundwater abstraction from the Gnangara Mound in 2009). Collectively this accounts for the third greatest use of groundwater behind abstraction for public water supply (42%) and abstraction for agriculture and horticultural use (22%) (Department of Water 2009). However, in Gwelup, it is estimated that 1.02 GL of groundwater is abstracted for unlicensed domestic garden bores (Department of Water 2009b).

Urban drainage

The network of drains and stormwater systems associated with increasing urbanisation has altered the natural regime of surface water entering Lake Gwelup. Water flows into Lake Gwelup via the Delawney Drain and the Brushfield Drain, which drains the Brushfield Wetland. When the lake is full, water can flow out of the lake to the north, via the Balcatta Branch Drain towards Lake Karrinyup (Figure 7).The Delawney Drain (Figure 8) and Brushfield Drain (Figure 9) are part of the Water Corporation’s main network of drains in the area. The Brushfield Drain receives water from four local roadway drains (from the north and south), the Delawney Drain and two below-ground drains (from the east and south) (Sawyer et al. 2008). It is likely that both receive groundwater discharge. Brushfield Drain discharges into the eastern edge of Lake Gwelup (Figure 10) and is a source of sulfate and iron enriched water.

In the 1920s when horticultural activities dominated the Gwelup landscape, the Brushfield Wetland was considered ‘clean and fit for drinking’ (Moore 2001). Given its visually pristine appearance people would do their washing in the Brushfield Wetland (Moore 2001). The Brushfield Wetland is now characterised by a striking orange colour indicative of suspended iron. The Brushfield Wetland has degraded due to local land and water use.

Surface water samples were collected from the Brushfield Drain and the Delawney Drain for water quality analysis between July 2007 and December 2007 (Sawyer et al. 2008). The pH of water near the intersection of Brushfield Wetland and Delawney Drain was above 5. Acidic pH between 3.0 and 4.0 was recorded at the mid-point of the Brushfield Drain and the discharge point into Lake Gwelup in November 2007 despite the use of calcite pellets in the Delawney Drain (Sawyer et al. 2008).

Figure 7 Drain flow around Lake Gwelup

Figure 8 Delawney Drain, 11 August 2010

Figure 9 Brushfield Wetland, 11 August 2010

Figure 10 Eastern edge of Lake Gwelup, 11 August 2010

2.5 Regional geology and geomorphology

Geology

The superficial formations of the Gnangara Mound are of Tertiary to Quaternary age, up to 110 m thick in the central region of the onshore Perth Basin. The Perth Basin comprises sediments of Permian to Cretaceous age up to 12 km thick (Davidson 1995). The superficial formations (Figure 11) are bound to the east by the Darling Fault which separates the Perth Basin from the rocks of the Yilgarn Craton and comprise four main geological units. The Safety Bay Sand (recent), Tamala Limestone (Late Pleistocene), Bassendean Sand (early to late Pleistocene) and the Guildford Formation (Pleistocene). These sediments are arranged parallel to the coast line and extend in an east to west direction in order of youngest to oldest (Davidson 1995). The Guildford Formation comprises a higher proportion of clay than the sand dominated dune systems of the Swan Coastal Plain (Davidson 1995). Lake Gwelup is underlain by sediments of Tamala Limestone.

The superficial formations beneath Lake Gwelup are underlain by the Kings Park Formation. The Kings Park Formation unconformably overlies the Jurassic and Cretaceous sediments of the Leederville Formation (Davidson 1995). The Kings Park Formation is of Palaeocene to Eocene age (Tertiary) and has a maximum onshore thickness of about 530 m around Claremont, west of Perth, approximately 15 km south of Lake Gwelup (Davidson 1995).The Leederville Formation consists of ‘grey, calcareous, glauconitic siltstone and shale of shallow marine to estuarine origin’ (Davidson 1995) and sub-crops the superficial formations where the Kings Park Formation is absent.

Geomorphology, soils and mineralogy

The surficial sediments of the Swan Coastal Plain are divided into a number of geomorphic units. Each geomorphic unit comprises unique soil characteristics in terms of soil particle size distribution, hydraulic conductivities and water retention characteristics that influence the volume and rate of recharge to the underlying Superficial aquifer.

The main geomorphological features of the Swan Coastal Plain in the Perth region are sand dominated dune systems. The coastal belt makes reference to the Quindalup and Spearwood dune systems. The Bassendean Dune System lies east of the Spearwood Dune System and is a highly leached and base cation depleted profile with negligible buffering capacity to resist changes in pH (Bawden 1991; Davidson 1995; Gozzard 2007). The Pinjarra Plain lies further east and comprises finer textured sediments. Lake Gwelup Reserve is located in the Spearwood Dune System (Figure 11).

The Spearwood Dune System consists of yellow to brown coloured sandy material dominated by quartz coated in iron oxide minerals such as goethite and hematite (Bawden 1991). The Spearwood Dune System is characterised by a core of aeolianite overlain by siliceous sand with weak podosol development (Salama et al. 2005).

Wetlands are generally located in swales and inter-dunal depressions of the Swan Coastal Plain (Davidson 1995). Lake Mariginiup, Lake Gnangara, Lake Jandabup and Lake Joondalup are all located in inter-dunal depressions between the Spearwood Dune System and the Bassendean Dune System. Lake Gwelup, however, is located within a swale of the Spearwood Dune System.

Acid sulfate soils

Acid sulfate soils (ASS) have been identified in Gwelup (Appleyard et al. 2004). Lakes and wetlands in the area were mapped as having a high to moderate likelihood of containing ASS, based on ASS disturbance risk mapping undertaken prior to 2006 (Degens 2006). The mismanagement of acid sulfate soils in the City of Stirling has caused groundwater acidification and released high concentrations of metals including aluminium, arsenic and iron (Appleyard et al. 2004).

Acid sulfate soils contain naturally occurring iron sulfide minerals such as pyrite, which are stable under anaerobic conditions however produce sulfuric acid when oxidised, according to the reaction below:

Oxidation of pyrite: $2FeS_2 + 15/2O_2 + 5H_2O \rightarrow 2FeOOH + 4SO_4^{2-} + 8H^+$

(Appelo & Postma 2005)

Potential acid sulfate soils (PASS) contain iron sulfide minerals that have not been oxidised to the extent that acidification processes have instigated the mobilisation of metals. A field pH test using the addition of suitably buffered (to pH 5.0–5.5) hydrogen peroxide solution (pH_{FOX}) can be used to test for PASS. A $pH_{FOX} \leq 3$ indicates the presence of PASS. Actual acid sulfate soils (AASS) are a product of oxidation processes that have caused field pH to decline to ≤ 4.0.

Soils containing naturally occurring sulfide minerals only become a management issue once oxidation pathways are introduced, such as from a lowering of the watertable or excavation of soil. The management of acid sulfate soils is dependent on the extent of the proposed soil and groundwater disturbing activities (DEC 2011).

Chemical analysis of soil samples by a laboratory is often necessary to quantify levels of sulfidic acidity (expressed in terms of %S or moles of H^+ per tonne). Laboratory analysis can be used to provide further confirmation of the source of acidification and is necessary to determine suitable liming rates for soil management.

Sand dominated sediments such as those of the Spearwood Dune System contain larger pore spaces which provide a better network of oxidation pathways than fine grained sediments. Sediments with larger pore spaces are subject to rapid oxygen ingress, and iron sulfide minerals will oxidise at a greater rate than in finer textured sediments with a lower specific yield and a higher water retention capacity. The rate of acidification is dependent on the amount of available buffering capacity.

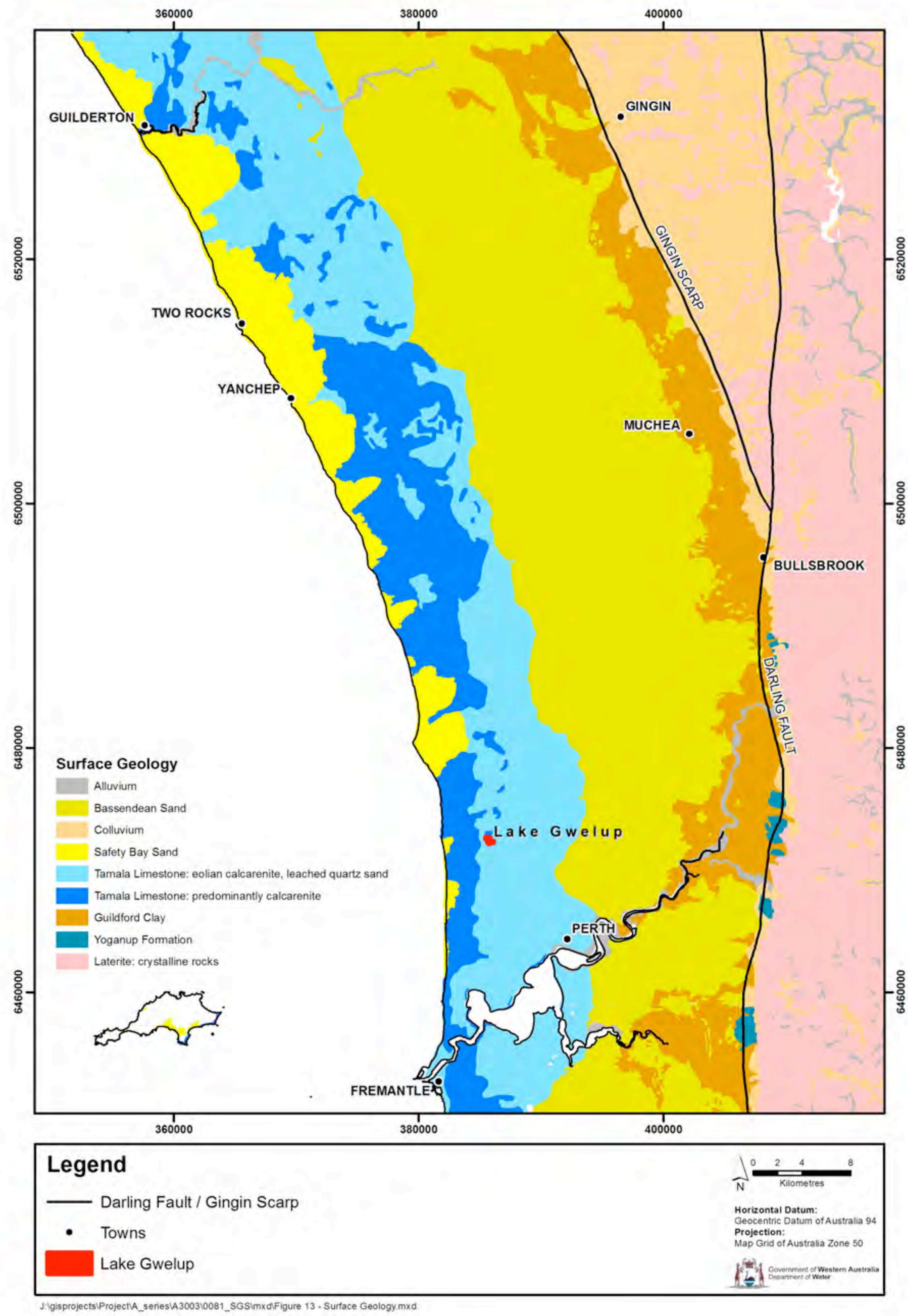


Figure 11 Regional surface geology, Swan Coastal Plain

Peat

Peat is brown or black and is a loose spongy material often containing a high organic matter content (Clarke et al. 1967). The Gwelup area is known to contain black, humic peats of moderately to well decomposed wetland vegetation (Angeloni 2003; Appleyard et al 2006). In Gwelup, strongly weathered peat has been observed above the watertable containing ferruginous and jarosite mottles and gypsum (Angeloni 2003; Appleyard et al. 2006). Other minerals have also been identified in peat in the Gwelup area including manganese oxides, hematite and schwertmannite (Angeloni 2003).

The presence of peat in Gwelup was viewed as an asset to the horticultural productivity of the area. However, a change in land use from agriculture to residential has led to significant volumes of peat being excavated and removed from the area due to its geotechnical unsuitability from which to base development. Dewatering activities, which lower the watertable, and the excavation of peat, to facilitate the installation of sewers and sub-surface infrastructure has caused groundwater acidification in Gwelup (Appleyard et al. 2004).

The acid generating potential of peat from the oxidation of iron sulfide minerals (such as pyrite) has been recognised since at least the 1930s (Teakle & Southern 1937). Liming was a common practice used by horticulturalists to neutralise acidification caused by the release of acidity stored in peat. Peats of the Swan Coastal Plain have been identified to contain up to 15% by weight of oxidisable sulfur (Appleyard et al. 2004).

Historical farming practices also included the burning of peat to generate ash for improving soil fertility and assisting in crop cultivation (Appleyard et al. 2006). It is likely that these practices exacerbated the release of acidity. Peat fires have been known to smoulder for months (Appleyard et al. 2006) thereby acting as a source of acidification for extended periods of time.

Arsenic has been recorded greater than 1000 mg/kg in Gwelup (Appleyard et al. 2006). Arsenic is not strictly confined to the peat layer in the soil profile. The vertical distribution of acidification and arsenic has been recorded to depths greater than 5.0 m below the peat layer, with up to 1% pyrite and arsenic at concentrations between 100 mg/kg and 400 mg/kg (Appleyard et al. 2006). Arsenic at concentrations greater than 100 mg/kg in residential areas may warrant further investigation to assess the level of risk to human health in case of exposure. Exposure may occur through the consumption of home grown vegetables (DEC 2010).

2.6 Hydrogeology

Superficial aquifer

The Superficial aquifer is an unconfined aquifer of Quaternary sediments with a saturated thickness up to 70 m and an average thickness of 45 m north of Perth (Davidson 1995). Regional groundwater flows towards the boundaries of the

Gnangara Mound from the crest (Figure 12). Lake Gwelup is located in the south-western portion of the Gnangara Mound and regional groundwater flows in a westerly direction towards the Indian Ocean. Lake Gwelup is a groundwater-dependent 'flow-through' wetland receiving groundwater from the east and discharging to the west.

Recharge of the Superficial aquifer is primarily from rainfall. The porous nature of sands associated with the Quindalup, Spearwood and Bassendean dune systems of the Swan Coastal Plain allows rainfall to recharge the Superficial aquifer. Recharge characteristics are similar in areas of Quindalup and Bassendean dune systems. However, a higher water retention capacity is associated with the Spearwood Dune System. This means the proportion of rainfall recharging the Superficial aquifer is slightly less (about 3 to 5% less) in sands of the Spearwood Dune System than in the Bassendean and Quindalup dune systems due to sediment texture (Xu et al. 2008).

In general, recharge of an unconfined aquifer is generally higher in residential areas than vegetated areas. Recharge in a residential area has been recorded to comprise approximately 50% of rainfall, compared with about 30% of rainfall in areas under pasture and about 20% of rainfall in areas of Banksia woodlands on the central Swan Coastal Plain (Xu et al. 2008). In Gwelup, groundwater levels have declined despite the high proportion of rainfall that is likely to be recharging the groundwater system. This is due to a combination of reduced rainfall and groundwater abstraction (Yesertener 2008).

Lakes and wetlands on the Swan Coastal Plain often become groundwater discharge areas, since they effectively act as evaporative basins (Xu et al. 2008). Most lakes on the Swan Coastal Plain are shallow and range in depth between 0.5 m and 3.0 m. Many lakes become dry due to declining water levels over the summer period (Davidson 1995). This is true for Lake Gwelup, which has dried each summer since 2001.

The hydrochemistry of an aquifer is strongly influenced by the composition of its sediments. The Tamala Limestone is rich in calcium carbonate and typically provides significant pH buffering capacity for sediment and groundwater (Davidson 1995). Groundwater in the Tamala Limestone is characterised as calcium bicarbonate type, as the composition of major ions is generally dominated by calcium and bicarbonate. Table 2 documents some groundwater quality information obtained in 1992 from the Gwelup bore field, screened in the Superficial aquifer (Tamala Limestone).

Previous assessments of groundwater quality are indicative of groundwater from the lower half of the Superficial aquifer, rather than near the watertable, because this is where production bores are screened (Table 2). In 1992, pH in the lower half of the Superficial aquifer in Gwelup ranged between 6.1 and 7.0. Salinity of the aquifer ranged between fresh and marginal according to total dissolved solids (TDS) and electrical conductivity (EC). Fresh groundwater contains less than 500 mg/L TDS and marginal groundwater contains 500 to 1000 mg/L TDS (Department of Water 2009).

The quality of groundwater in the lower half of the Superficial aquifer has changed since the 1970s. This change is characterised by increasing trends in calcium, sulfate and iron concentrations. This is probably due to pyrite oxidation (Appleyard et al.

2006). The most significant change has been recorded at G80, located approximately 1.5 km east of the southern boundary of Lake Gwelup (Appleyard et al. 2006).

Nitrate concentrations have declined since the late 1980s, accompanied by a subsequent rising trend in ammonia concentrations. This is consistent with the changes expected from a decrease in irrigated horticulture. The presence of nitrate in groundwater may have oxidised pyrite and organic matter causing changes in the quality of groundwater at these depths. Groundwater in the lower half of the aquifer is not acidic because it contains an adequate source of alkalinity. However, alkalinity concentrations are dynamic and if acid generating processes continue, levels of alkalinity may decline, rendering the aquifer susceptible to acidification.

An increasing trend in arsenic concentrations was identified in groundwater near the watertable and in the lower half of the Superficial aquifer of the Gwelup groundwater area (Appleyard et al. 2006). Arsenic concentrations were generally higher near the watertable, where the mechanism for release of arsenic is due to the oxidation of arsenic containing pyrite. Concentrations of arsenic up to 7 mg/L have been measured in groundwater near the watertable (Appleyard et al. 2006). In the lower part of the aquifer, reducing conditions indicate a second mechanism responsible for the release of arsenic into groundwater. The release of arsenic may be due to the microbial reductive dissolution of arsenic containing oxyhydroxide minerals. This process does not appear to generate concentrations of arsenic as high as near the watertable, although elevated levels of arsenic have been identified between 5 µg/L and 15 µg/L in the lower half of the Superficial aquifer in the Gwelup groundwater area (Appleyard et al. 2006).

Table 2 Range in water quality for 11 production bores of the Gwelup bore field screened in the Superficial aquifer, data obtained in 1992 (Davidson 1995).

Parameter	**Range**			
	Low		**High**	
	Value	**Bore**	**Value**	**Bore**
pH	6.1	G130	7.0	G80
EC (mS/m)	50	G50	105	G80
TDS (mg/L)	281	G50	684	G80
Total alkalinity (mg/L)	67	G50	153	G80
Ca (mg/L)	27	G50	130	G80
Mg (mg/L)	7	G120	16	G80
Na (mg/L)	47	G50	66	G60
K (mg/L)	3	G10	8	G80
CO_3 (mg/L)	< 0.6	All	< 0.6	All
HCO_3 (mg/L)	81	G50	186	G80

Parameter	**Range**			
	Low		**High**	
	Value	**Bore**	**Value**	**Bore**
Cl (mg/L)	81	G50	117	G110
SO_4 (mg/L)	9	G10	225	G80
NO_3 (mg/L)	< 0.02	G80 and G10	6.77	G130
Dissolved Fe (mg/L)	1.18	G120	12.80	G80

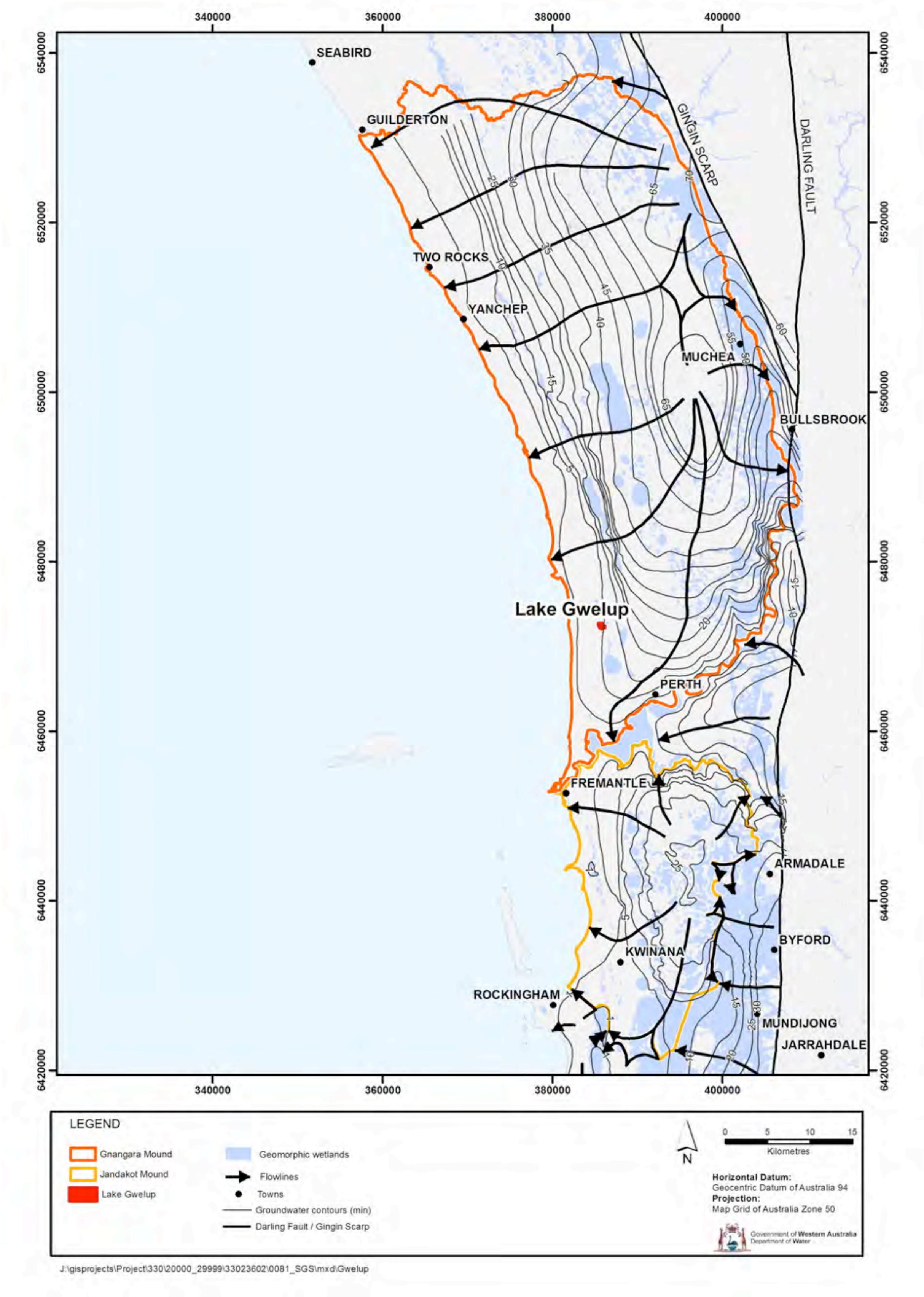


Figure 12 Regional groundwater flow in the Gnangara and Jandakot mounds

2.7 Ecology

Lake Gwelup is a groundwater-dependent wetland located in the Herdsman Vegetation Complex and is regarded as a key faunal habitat (Froend et al. 2004a). The ecological significance of Lake Gwelup has been recognised through a number of environmental conservation measures. The lake has been:

- recognised as 'regionally significant' by the Western Australian Water Resources Council in 1991
- recognised as a System 6 wetland by the Department of Conservation and Environment in 1983
- listed as a Bush Forever site by the Government of Western Australia in 2000 (No. 212)
- listed as a conservation category wetland
- listed as an EPP (Environmental Protection Policy) wetland (Froend et al. 2004a).

A drying climate, groundwater abstraction and surrounding land uses (such as historical horticultural activities and urbanisation) have affected the ecology of Lake Gwelup (Froend et al. 2004a).

Lake Gwelup Reserve is considered to have a high social amenity given the accessibility of the reserve in a highly urbanised area (DoW 2009b). There is a community group, Friends of Lake Gwelup, which assist the City of Stirling to manage the reserve (Beckwith Environmental Planning Pty Ltd 2006). Historical clearing of land has facilitated easy access to the lake and a large area is used as public open space. It is important to link the management of ecological attributes with the maintenance of social values (DoW 2009b). This signifies the additional importance of maintaining ecological values at Lake Gwelup given the high social amenity of the Lake Gwelup Reserve.

An interdependent relationship exists between water levels, flora and fauna. The preservation of Lake Gwelup Reserve as a key faunal habitat relies on the conservation of groundwater-dependent vegetation, which in turn relies on the maintenance of groundwater levels (Bamford & Bamford 2003).

The Lake Gwelup Reserve management plan (Ecoscape 2006) identified 80 native flora species in the reserve. *Jacksonia sericea* (common name: waldjumi), a low spreading shrub (up to 0.6 m high), was identified by the former Department of Conservation and Land Management (CALM) in the 1992 management plan as being highly dependent on groundwater and likely to become locally extinct if groundwater levels decline and the wetland dries. Monitoring of this species was recommended every 5 to 10 years (Ecoscape 2006). As Lake Gwelup has dried out each summer since 2001, monitoring of vegetation is considered necessary to more adequately characterise the changing ecological nature of the reserve over time.

The dominant vegetation species within Lake Gwelup Reserve have been noted as *Eucalyptus gomphocephala* (tuart), *Corymbia calophylla* (marri)*, Melaleuca* *rhaphiophylla* (swamp paperbark) and *Eucalyptus rudis* (flooded gum) (Froend et al. 2004b). The main area of the lake supports a 'herbland' of annual weeds and *Typha orientalis* (typha, bulrush) (Froend et al. 2004a). In 2004 an area of melaleucas was observed to have died at the western edge of the lake (Beckwith Environmental Planning Pty Ltd 2006).

Ecological water requirements (EWRs) can be inferred for a species at a wetland by subtracting the mean minimum water depth (m) of a species from the minimum elevation at which the species is present at the wetland (m AHD) (Froend et al. 2004b).

In March 2012 an assessment of vegetation between SGS monitoring locations was conducted by Department of Water staff. This assessment identified *Eucalyptus rudis* and *Melaleuca rhaphiophylla* as the dominant species east of the lake, with *Typha orientalis* dominating the edges of the lake. The western edge of the lake is mainly cleared for public open space (figures 53 and 54). At the eastern edge of the lake, the lowest elevation of *E. rudis* was recorded as approximately 7.5 m AHD and the lowest elevation of *M. rhaphiophylla* was recorded as approximately 6.0 m AHD. The mean minimum water depth for *M. rhaphiophylla* is 2.14 m and for *E. rudis* it is 3.26 m (Froend et al. 2004b). Therefore proposed EWRs at Lake Gwelup for dominant species may be:

- EWR for *E. rudis* at Lake Gwelup – 7.5 m AHD – 3.26 m = 4.24 m AHD
- EWR for *M. rhaphiophylla* at Lake Gwelup – 6.0 m AHD – 2.14 m = 3.86 m AHD.

These EWRs reflect the minimum water level required to maintain *E. rudis* and *M. rhaphiophylla* for species distribution around Lake Gwelup in 2012, based on methods described in Froend et al. (2004b).

T. orientalis is an introduced macrophyte species from the eastern states of Australia, which has been encroaching further into Lake Gwelup (Froend et al. 2004b). The spread of *T. orientalis* poses an ecological issue for the biodiversity and abundance of native vegetation because of its ability to exclude native competitors through its expansive growth form and its high reproductive capability, producing a large number of seeds with a high percentage of viability (Froend et al. 1993). The seasonal drying out of Lake Gwelup since 2001 has coincided with the continued encroachment of *T. orientalis*, and the decline in the condition of fringing *E. rudis* and *M. rhaphiophylla* (Froend et al. 2004b). These findings are consistent with other wetlands on the Swan Coastal Plain that have become seasonal wetlands as opposed to permanent wetlands (Froend et al. 1993).

The Lake Gwelup Reserve management plan (Ecoscape 2006) identified six native mammals, 40 reptiles and/or amphibians and 125 native bird species likely to inhabit the reserve (Ecoscape 2006). The maintenance of water levels and the preservation of moisture within the sediments at Lake Gwelup are important for the conservation of its waterbird and vertebrate populations, as well as the quality of water within the lake (Froend et al. 2004).

Water birds are likely to be very sensitive to drawdown of the watertable, due to their use of ponded areas in the lake and other birds that make use of the dense vegetation surrounding wetlands (Bamford & Bamford 2003). The Lake Gwelup Reserve management plan (Ecoscape 2006) recommended an increase in the buffer zone comprising areas of bushland surrounding the lake to approximately 100 m. This is in agreement with previous recommendations by the Water and Rivers Commission in 2000 (WRC 2000), and current Department of Environment and Conservation advice, that recommend buffer zones of at least 50 m surrounding wetlands in order to maintain ecological processes and food webs to accommodate birds and other fauna (DEC 2011).

2.8 Previous investigations on groundwater quality in the Gwelup area

Water quality investigations, Gwelup bore field, 1975 to 1992

CSIRO published a report discussing the quality of groundwater around the Gwelup bore field (located up-gradient of Lake Gwelup) between 1975 and 1992 (Barber et al. 1993).

High nitrate concentrations were recorded near the watertable in some areas of the Gwelup bore field. Concentrations of nitrate were greater than 10 mg/L to a depth of 7.0 m below the watertable (Barber et al. 1993). Elevated nitrate concentrations may be a legacy of previous horticultural activities or leachate from leaking septic systems from unsewered urban development.

Ammonia concentrations were low across the Gwelup bore field between 1980 and 1992, except for bores located near the unlined Hertha Road landfill (Figure 13). Ammonia was recorded at levels greater than 1 mg/L in production bores G50 and G60 (Barber et al. 1993). Ammonia was also recorded in a groundwater monitoring bore up to 190 mg/L near the Hertha Road landfill (Figure 13). This suggests the presence of a landfill leachate plume (Barber et al. 1993). These levels also significantly exceed the domestic non-potable groundwater use guideline for ammonia of 5 mg/L (DEC 2010). Based on regional groundwater contours, it is not anticipated that the landfill leachate plume will migrate towards Lake Gwelup. However, the extent of the landfill leachate plume should be delineated, given the widespread use of untreated groundwater in the area.

High sulfate concentrations were recorded at GM17, GM23, and GM30 between 1975 and 1992. In August 1983, sulfate concentrations were recorded up to 667 mg/L at GM23, with a corresponding low chloride to sulfate ratio of 0.2, which may indicate sulfide oxidation. A chloride to sulfate ratio less than 2.0 indicates an additional source of sulfate, which may be due to sulfide oxidation, seawater intrusion or fertiliser input (DEC 2011). Sulfate concentrations between 1970 and 1992 were generally below 100 mg/L with chloride to sulfate ratios above 2.0. Sulfate concentrations have increased since 1970. This coincides with a decline in horticulture and the commencement of groundwater abstraction for public supply and urbanisation (Barber et al. 1993). This suggests the oxidation of pyrite is a process that may have been taking place in the Gwelup bore field since the commencement of groundwater abstraction for public supply in 1974.

Dieldrin and DDT (dichlorodiphenyltrichloroethane) were detected in monitoring bores in the Gwelup bore field. DDT was detected at very low concentrations, marginally exceeding laboratory detection limits. Dieldrin however was identified in production bores G30, G40 and G50 in 1990. Dieldrin was recorded up to 0.19 µg/L, which exceeds the level at which pesticides can be reliably detected using validated analytical methods (0.01 µg/L). However 0.19 µg/L is below the health based drinking water guideline of 0.3 µg/L. This guideline is derived from assuming 10% of acceptable daily intake, which is conservatively inferred from consumption of 2 L of water per day, for a 70 kg adult (NHMRC 2004).

Screening of domestic groundwater bores by Department of Environment (DoE), 2002

Urban development in the City of Stirling has involved the excavation of peat, dewatering and the construction of artificial lakes in Spoonbill Reserve. These activities have not been managed adequately to prevent acidification and the release of metals including arsenic, aluminium and iron in groundwater near the watertable. Spoonbill Reserve is approximately 1.5 km east of Lake Gwelup and is acidic (pH < 3) (Figure 13). In 2002, the Department of Environment (DoE) conducted a screening pH assessment of over 800 domestic bores in the area. pH was used as an initial screening tool, and where pH was less than 5.5, samples were submitted to a laboratory for chemical analysis. Previous investigations in the Gwelup area suggest arsenic is confined to the framboidal shaped pyrite contained in peat material. Upon oxidation of arseno-pyrite, arsenic is released into the shallow groundwater system which also produces acidity (H+), sulfate and iron (Appleyard et al. 2006). A groundwater acidity plume was identified to extend approximately 800 m through Spoonbill Reserve. Contaminants such as arsenic and aluminium pose a risk to the quality of groundwater discharging into flow through wetlands by virtue of their extensive capture zones (Appleyard et al. 2006).

The pH assessment of domestic garden bores identified 50 bores with pH less than 5.5. Iron was recorded at concentrations up to 1300 mg/L, aluminium up to 290 mg/L, sulfate up to 4400 mg/L and arsenic up to 0.8 mg/L. High sulfate to chloride ratios (> 0.5) were recorded, indicating an additional source of sulfate, probably from pyrite oxidation. Almost half of the bores (24 bores) recorded arsenic at concentrations exceeding the national drinking water guideline of 0.007 mg/L (Appleyard et al. 2004).

Additional groundwater bores were installed as part of these investigations in February 2002. Groundwater quality sampling identified acidic groundwater (pH < 5) to 15.6 m below ground level, pH as low as 2.6 near the watertable, approximately 3 m below ground level. Arsenic was recorded up to 7.3 mg/L near the watertable, more than 1000 times the drinking water guideline (DEC 2010) (Appleyard et al. 2004).

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Figure 13 Location of the former Hertha Road landfill and Spoonbill Reserve in relation to Lake Gwelup

Soil and groundwater investigations, Gwelup, 2006

Groundwater investigations by Parsons Brinckerhoff (PB) in 2006 were conducted at Gwelup Park, a residential development, east of the Gwelup shopping centre. These investigations recorded an isolated area of groundwater acidification and high concentrations of arsenic near the watertable. Total acidity as $CaCO_3$ was recorded at 7600 mg/L, pH 2.4 and concentrations of arsenic up to 22 mg/L. A second round of groundwater sampling was undertaken in order to confirm results, where arsenic was identified at the same location at 12 mg/L. These results are inconsistent for the presence of arsenic in groundwater and this is at least in part due to poor groundwater sampling methodology. However, the results suggest the presence of very high concentrations of arsenic. DEC subsequently installed groundwater monitoring bores to more adequately define the extent of acidification and arsenic in groundwater. The results of these ongoing investigations are detailed in this report. The acidification and subsequent metal release in groundwater near the watertable was attributed to groundwater level decline exposing peat and to the excavation and removal of peat prior to construction activities. This is similar to the mechanism of arsenic release for the urban development around Spoonbill Reserve.

Regional assessment of the hydrogeochemistry of the Superficial aquifer in the Perth metropolitan area, 2003 to 2005

A regional hydrogeochemical assessment of the Superficial aquifer was conducted by the Department of Water. This was a one-off round of groundwater quality sampling in the Superficial aquifer in the Perth region, mainly conducted between 2003 and 2005. This study is limited because a one-off sampling event does not provide an adequate basis for comparison of groundwater quality over time. A further limitation is that most groundwater bores are constructed with long screens and samples probably represent mixed water from across the aquifer, rather than from a specific depth in the aquifer.

This study recorded an increase in salinity in the Gwelup groundwater area of up to 210 mg/L compared with previous investigations in 1992 (Davidson 1995). This was attributed to impacts of groundwater abstraction and the presence of less permeable lake sediments inhibiting infiltration of rainfall (Yesertener 2010). An increasing trend in the sulfate to chloride ratio was noted since 1972. In 1972, prior to the commencement of groundwater abstraction for public supply, the sulfate to chloride ratio was similar to seawater. In 1992 the sulfate to chloride ratio increased above 0.5 in a number of bores and ratios continued to increase over larger areas by 2005 (Yesertener 2010). This supports the findings of previous investigations that an additional source of sulfate has developed since the 1970s in the Gwelup groundwater area.

3 Investigation program

The following sections describe two investigations:

- work undertaken as part of the Perth SGS investigation by the Department of Water
- the Gwelup soil and groundwater acidity investigation undertaken by the Department of Environment and Conservation (DEC) between 2006 and 2010

3.1 Perth SGS investigation

Bore installation

Three groundwater bores were installed on the eastern (GLP_E) and western (GLP_W) edges of Lake Gwelup by Geo Sonic Drilling (formerly Great Southern Drilling) in March and April 2009 (figures 14 and 15). These investigations were recommended in the management area review (MAR) (McHugh & Bourke 2007). All six groundwater bores were constructed with 50 mm Class 12 PVC casing, installed with 3.0 m screens intersecting the watertable and 2.0 m screens intersecting the middle and base of the Superficial aquifer (Figure 14). Bore logs and coordinates in Appendix A.

Shallow groundwater bores (GLP_Wc and GLP_Ec) were installed prior to this investigation. Cores for acid sulfate soil analyses were collected adjacent to existing shallow groundwater bores using a Roto-sonic tractor-mounted drill rig to collect intact soil cores. The extraction of intact soil cores limits the amount of disturbance and oxidation, preventing changes in soil chemistry. The intermediate (GLP_Eb and GLP_Wb) and deep (GLP_Ea and GLP_Wa) bores were installed using a rotary air core drilling rig.

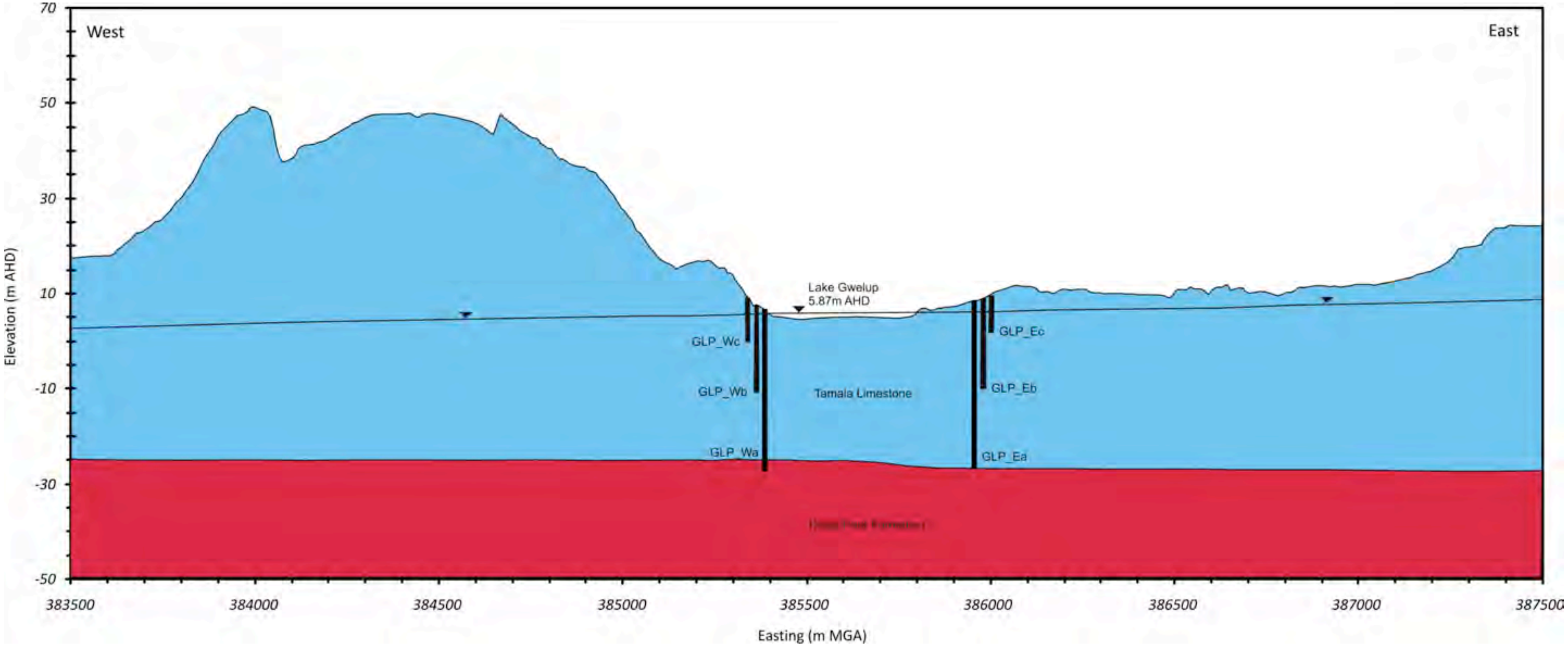

Figure 14 Cross-section – SGS bore locations

Groundwater monitoring

Groundwater quality monitoring was undertaken on eight occasions between June 2009 and June 2010 as part of the Perth SGS program. This high frequency of groundwater sampling provides a good understanding of seasonal variation in groundwater quality at Lake Gwelup for the monitoring period. The SGS monitoring program was complemented by monitoring undertaken by DEC in September 2010, which also included SGS bores either side of Lake Gwelup.

The Perth SGS investigation used the National Measurement Institute (NMI) laboratory for groundwater analysis, which included metals, nutrients, pesticides, herbicides and other inorganic parameters (Table 3). ALS Environmental (a Perth laboratory) was used for chemical groundwater analyses in September 2010 (Table 4).

Herbicides and pesticides were analysed on two occasions (June 2009 and January 2010). Metals, nutrients and other inorganic parameters were collected and analysed in each round of sampling. The shallow groundwater monitoring bore on the western edge of Lake Gwelup (GLP_Wc) could only be sampled on four occasions because the bore did not contain enough water to collect a sample between January 2010 and June 2010.

All groundwater monitoring was undertaken using a low flow bladder pump. Low flow sampling provides a more accurate assessment of in situ groundwater conditions than other practices such as hand bailing and high flow methods (Geotech 2009). Low flow sampling improves sample quality by reducing turbidity and minimising the loss of volatile compounds (Geotech 2009). Increased oxygen levels due to poor sampling techniques may change the chemical composition of groundwater and therefore provide misleading results. Methods that cause excessive disturbance of the sample such as hand bailing may cause the release of metals, change pH and ultimately fail to provide an accurate representation of groundwater quality.

Groundwater samples were collected once field readings (pH, EC, oxidation reduction potential and dissolved oxygen) of pump outflow stabilised. The Perth SGS investigation used a Quanta multi parameter probe to measure field readings and the DEC investigations used a Hydrolab MS.5 Minisonde to measure field readings.

Groundwater level monitoring for SGS bores was undertaken on a monthly basis and provides an understanding of water level fluctuations for the monitoring period. Seven additional bores are located around Lake Gwelup (figures 36 and 37) and these have been monitored for water levels by the Department of Water since the 1970s. These bores were monitored for water levels at a lower frequency but at times consistent with the SGS monitoring program.

The seven bores are shown in the table below and have been used to gain an understanding of sub-regional groundwater flow (see figures 36 and 37).

WIN* identification number	AWRC identification number
4373	61610032
4374	61610033
4375	61610034
4376	61610035
4377	61610036
4421	61610080
4432	61610091

**Department of Water's Water Information System*

Table 3 Groundwater laboratory parameters assessed in SGS investigations (June 2009 and June 2010)

Parameters analysed by the National Measurement Institute	
Total metals (not filtered)	Al, As, Cd, Cr, Fe, Hg, Mn, Ni, Se, Zn
Dissolved metals (filtered)	Al, B, Fe
Base cations	Ca, Mg, Na, K
Base anions	Cl, SO_4, CO_3, HCO
Nutrients	NOx, TKN, total N, NH_3-N/ NH_4-N, dissolved P, total P, PO_4-P
Herbicides and pesticides	Chlordane, Chlordane (Tech: a+g), DDD-p,p, DDE-P,P, DDT-P,P, Dieldrin
	OC's, OP's, Diuron, total Endosulfate, Endosulf-a, Diazinon, Endosulf-b
	Endrin, Ethion, HCH (BHC) a, b, d, HCH (BHC), Heptachlor
	Heptachlor epoxide, Heptachlor obenzene, Hexazinon, Bromophos-ethyl
Inorganics	pH, total acidity as $CaCO_3$, total alkalinity as $CaCO_3$, EC, DOC TSS, TDS, F, DON

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Figure 15 SGS bore locations at the eastern and western edges of Lake Gwelup

Acid sulfate soil testing

The Perth SGS investigation assessed soil acidity (pH_F) and potential acidity (pH_{FOX}) to six meters below ground level (~3 m below watertable) at the western (GLP_Wc) and eastern (GLP_Ec) edges of Lake Gwelup. These investigations provide an indication of the presence of acidity, the areas susceptibility to soil acidification and potential risks associated with watertable changes. Field testing was conducted at approximate 50 cm intervals in each of the two 6 m soil/sediment cores.

Samples were selected for laboratory analysis based on the results of field testing. Samples recording pH_F around 4.0 or pH_{FOX} around 3.0 were submitted for further testing to provide information regarding the amount of actual or potential acidity. Laboratory samples were submitted to NMI for chemical analysis. Laboratory analysis comprised the suite of assessments under the suspension peroxide oxidation combined acidity and sulfur (SPOCAS) method, the chromium reducible sulfur (S_{CR}) measurement and an assessment for a suite of metals Al, As, Cd, Cr, Fe, Mn, Ni, Se and Zn.

Surface water sampling

Three surface water samples were collected in September 2010 from the Brushfield Wetland, the eastern edge of Lake Gwelup and the western edge of Lake Gwelup near an existing staff gauge (Figure 16). These analytical results are tabled in Appendix C.

Field parameters (pH, EC, temperature, oxidation reduction potential and dissolved oxygen (DO)) were recorded using a Hydrolab Minisonde MS.5. Samples were collected for laboratory analysis: total and dissolved metals were collected in 60 mL plastic containers pre-preserved with nitric acid. Total metals were not filtered, dissolved metals were field filtered using a 0.45 micron syringe filter. Nutrients were collected in a 125 mL plastic container, pre-preserved with sulfuric acid and field filtered using a 0.45 micron syringe filter. Samples for inorganic analysis were collected in a 1 L laboratory supplied plastic container and were not filtered and not preserved,. All samples were placed in an Engel freezer until submitted to ALS laboratory for analysis according to Table 4.

Figure 16 Surface water sample locations at Lake Gwelup and Brushfield Wetland

Table 4 Laboratory parameters analysed for groundwater and surface water in September 2010

Parameters analysed by ALS Environmental	
Total metals (not filtered)	Al, As, Fe
Dissolved metals (filtered)	Al, As, B, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Se, Zn, Hg
Base cations	Ca, Mg, Na, K
Base anions	Cl, SO_4
Nutrients	Total N, TKN, NO, nitrite as N, NOx, total P
Inorganics	Total acidity as $CaCO_3$, total alkalinity as $CaCO_3$, EC, TDS, TSS

3.2 Gwelup soil and groundwater acidity investigation

Bore installation

This investigation, undertaken by DEC (2006 – 2010) aimed to delineate the lateral extent of acidification and arsenic near the watertable beneath a new residential development called Gwelup Park and to monitor the movement of a plume of acidic groundwater over time. The investigation area comprised the Gwelup shopping centre, Australia Post and the Gwelup Park residential development. This area is located approximately 400 m east (up-hydraulic gradient) of Lake Gwelup.

Fourteen groundwater bores were installed in 2006 (Figure 17). All bores were installed with a screen intersecting the watertable. Bores were generally installed with 50 mm Class 9 PVC casing, but GWA4 and GWA6 were installed with 20 mm Class 9 PVC casing. Bore construction details in Appendix B.

Eight additional groundwater bores were installed in November 2008 to refine the original delineation of the area of acidified groundwater with elevated arsenic concentrations (Figure 18). These were constructed with 50 mm Class 18 PVC casing with screens at the watertable. Two of these groundwater bores replaced existing groundwater bores – GWA9, which was continually dry, and GM13, which had been destroyed. Bore construction details in Appendix B.

Figure 17 Location of groundwater bores installed by DEC in 2006

Figure 18 Groundwater monitoring bores installed by DEC in November 2008

Groundwater monitoring

Groundwater monitoring was conducted by the Acid Sulfate Soils Section of DEC on five occasions between March 2006 and September 2010. Each groundwater monitoring event has varied with respect to groundwater monitoring equipment used, the groundwater bores incorporated into each investigation and the parameters assessed for each round of groundwater sampling. During this time, monitoring of some bores in the residential development ceased because they were demolished or buried due to activities associated with the continued development of the Gwelup Park residential estate. Details regarding these groundwater sampling events are in Appendix D.

The monitoring bores sampled between 31 August and 2 September 2010, are shown in Figure 19. Sampling was conducted using a low flow bladder pump. While pumping, field parameters (pH, EC, DO, Eh, Temperature) were recorded at five minute intervals for at least 20 minutes. Standing water levels were monitored during pumping to ensure minimal drawdown of the watertable. Laboratory samples were collected following the stabilisation of field parameters and submitted to ALS for analysis.

Laboratory supplied plastic containers were used for the collection of samples. Samples for inorganic analysis (total acidity, alkalinity, sulfate, chloride, Ca, Mg, K and Na) were not field filtered and collected in unpreserved 1 L plastic containers. Dissolved metals (Al, As, Cd, Cr, Fe, Mn, Ni, Se, Zn) and total metals (Al, Fe, As) were collected in pre-preserved 60 ml plastic containers with nitric acid. Dissolved metals and nutrients were field filtered with a 0.45 micron disposable filter. Nutrients were collected in a pre-preserved 125 mL plastic container preserved with sulfuric acid.

Acid sulfate soil testing

Eight intact soil cores were abstracted with a tractor mounted sonic drill rig in conjunction with the installation of groundwater monitoring bores between 18 and 20 November 2008 (Figure 18). Soil investigations were conducted to determine the risk of potential acidification of soil and impact to groundwater quality in the event of soil disturbance and/or lowering of the watertable.

Acid sulfate soil field testing was conducted in accordance with Department of Environment and Conservation guidelines (DEC 2011). No field pH testing was undertaken on the ‘yellow fill material’, which was observed in all cores between 1.0 m and 3.0 m below ground level. Fill material was present due to work associated with the construction of the residential development.

Laboratory analysis was conducted on 26 selected samples based on field pH results and different soil units in each core. Laboratory samples were collected in laboratory supplied acid sulfate soil sample bags and placed in an Engel freezer until delivery to the laboratory. Laboratory samples were submitted to ALS for chemical analysis. Laboratory analysis comprised the suite of assessments under the suspension peroxide oxidation combined acidity and sulfur (SPOCAS) method, an assessment of chromium reducible sulfur and metals (Ahern et al. 2004).

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Figure 19 DEC and Department of Water groundwater bores sampled in September 2010

The interpretation of field and laboratory results from acid sulfate soil investigations

Comparison of field and laboratory pH values for sediments

Understanding the difference between methods used to measure pH in the field and those used to measure pH in the laboratory assist in the interpretation of these results. pH measured in the field is generally more reliable than the laboratory pH assessment due to limitations regarding sample preparation methodology in a laboratory. These sample preparation techniques render the sample less representative of actual 'field' conditions.

Field pH testing was undertaken on a one part soil, five part deionised/distilled water mix.

Testing of pH in the laboratory followed the methodology detailed in Ahern et al. (2004). The sample preparation procedure involves grinding and crushing of the sample to pass through a 2 mm sieve. If carbonaceous material is present in the sample, the smaller particle size due to crushing and grinding of the sample reacts more effectively due to the higher surface area to volume ratio providing a false indication of the 'real' or 'field' buffering capacity of the sediment. Laboratory pH_{KCl} and pH_{OX} may record significantly higher pH values than their respective field samples. This is a particular issue when pH_{KCl} records higher results than field samples, because titratable actual acidity (TAA) is assumed to be zero when $pH_{KCl} \geq 6.5$. Therefore, TAA may not be analysed and presumed to be 0, where the field pH of a sample may show that some TAA is present, which is significant in poorly buffered soils.

Four sediment samples were submitted from GLP_Wc to the laboratory for analysis. The laboratory and field pH is compared in Table 5. The most significant change in pH is between pH_F and pH_{KCl} of the sample collected at 5.3 m below ground level. This significant difference is likely to be due to the presence of some carbonaceous material.

Table 5 Field and laboratory pH values for GLP_Wc in April 2009

Sample depth mbgl*	pH_F	pH_{FOX}	pH_{KCl}	pH_{OX}
2.0	6.53	5.26	6.0	5.4
5.0	7.21	2.21	6.1	3.2
5.3	3.24	2.08	5.7	2.6
5.8	5.23	2.15	5.8	3.0

** metres below ground level*

From investigations undertaken up-gradient of Lake Gwelup in November 2008, pH_{KCl} and pH_{OX} were generally higher than corresponding field pH (Table 6).

pH_{KCl} results did not indicate the presence of actual acid sulfate soil (AASS) in contrast to pH_F detecting the presence of AASS in nine of 26 samples submitted for laboratory analysis. pH_{OX} results recorded PASS in seven samples compared to all 26 pH_{FOX} samples recording PASS with $pH_{FOX} \leq 3.0$ or below. Interpretation of laboratory and field results can impact management objectives derived from field investigations and therefore an understanding of these processes is necessary.

The most significant difference in pH_F and pH_{KCl} was observed at MW2, 3.5 m below ground level, where pH_F recorded 4.0 and pH_{KCl} recorded 7.8. This was also reflected in pH_{FOX} which recorded 2.3 in contrast to pH_{OX} of 6.8. MB1 recorded an alkaline pH_{KCl} of 9.6, however the corresponding pH_F recorded a neutral pH of 7.0.

Table 6 Field and laboratory pH values in November 2008

Bore ID	Sample depth mbgl*	pH_F	pH_{FOX}	pH_{KCl}	pH_{OX}
MW1	4.50	7.4	2.7	6.9	3.3
	5.00	7.0	2.4	9.6	8.0
	6.25	6.5	2.6	9.2	7.6
	8.50	6.1	2.0	6.0	3.4
MW2	2.50	4.3	1.7	6.0	4.4
	3.50	4.0	2.3	7.8	6.8
	5.00	3.4	1.8	5.4	4.2
	6.00	3.8	2.8	5.7	4.8
	8.00	4.0	2.7	5.5	4.4
MW3	2.50	5.1	1.5	5.1	2.2
	3.25	4.2	1.3	5.3	2.2
	5.25	3.9	2.1	5.7	5.2
	8.00	6.5	2.1	5.8	4.4
MW4	3.50	7.7	1.2	6.6	2.6
	5.25	4.6	2.2	5.4	3.9
	6.00	2.9	2.5	5.5	4.3
	8.00	6.5	2.1	5.9	3.7
MW5	2.50	5.8	1.4	5.3	2.4
	3.25	3.1	1.3	5.4	2.5
	5.50	3.4	1.6	5.6	4.5
	7.25	3.5	2.1	5.8	3.9

Bore ID	Sample depth mbgl*	pH_F	pH_{FOX}	pH_{KCl}	pH_{OX}
MW6	3.50	5.8	1.3	5.3	2.7
	4.50	6.2	2.0	5.9	3.9
	7.50	6.2	1.6	5.4	2.5
GM13	7.75	6.7	2.4	6.0	4.6

** metres below ground level*

Chromium reducible sulfur (S_{CR}) measurement and the assessments involved in the suspension peroxide oxidation combined acidity and sulfur (SPOCAS) suite

The suite of assessments in the SPOCAS analysis and the S_{CR} measurement were analysed in accordance with Ahern et al (2004). SPOCAS comprises a suite of assessments where the peroxide oxidisable sulfur (S_{POS}) assesses inorganic and organic acidity, providing a good management tool with respect to the acid generating potential of a soil. S_{POS} is calculated in the following way, where S_P represents the peroxide sulfur assessment and S_{KCl} represents potassium chloride extractable sulfur:

$S_{POS} = S_P - S_{KCl}$

S_P should not be considered in isolation when assessing the acid generating potential of a soil because it includes an assessment of sulfate salts such as gypsum which have no acid generating potential. S_{KCl} provides a measure of adsorbed and soluble sulfate (such as gypsum) and is therefore used to calculate S_{POS}, a more reliable assessment of the acid generating potential of a soil on which to base calculation of liming rates (Ahern et al. 2004).

The chromium reducible sulfur (S_{CR}) assessment is a measurement of the inorganic sulfide content only and does not account for acidity that may be derived from the oxidation of organic matter. S_{CR} measures inorganic sulfur compounds comprising iron disulfides such as pyrite, elemental sulfur, thiosulfate tetrathionate, polythionates and acid volatile sulfur (Ahern et al. 2004). S_{CR} and SPOCAS may be applied differently and can be used independently to gain an understanding of the source of acidity present. In sediments known to contain organic matter, the SPOCAS method should be used to gauge the overall acid generating potential of the soil. The S_{CR} method can be used to gain an understanding of the proportion of acidity related to the oxidation of inorganic sulfidic material. Soil management should not be isolated to inorganic sulfidic acidity but should include all potential sources of acidification whether sourced from organic matter, metal hydrolysis or other means.

3.3 Data accuracy and precision

Field duplicates, rinseate samples and calibration checks were used to confirm results of soil and groundwater testing. All soil and groundwater samples were collected and transported under suitable Department of Water or Department of Environment and Conservation ‘chain of custody’ documentation.

Field decontamination procedures for groundwater monitoring undertaken by DEC included a three stage rinsing process for each element of the low flow pump. This involved rinsing equipment in a 20 L container of tap water, followed by a 20 L container of tap water mixed with Decon 90 solution and a final rinse in a 20 L container of distilled water. Each 20 L drum used in the decontamination process was discarded and renewed every second day. Bladders were discarded after three uses, and new tubing was used in the collection of samples from each bore. The collection of rinseate samples further confirmed the suitability of field decontamination procedures in the field for DEC and Department of Water investigations.

The recommended laboratory holding time for the assessment of pH in water is only six hours and this reflects the rate at which groundwater chemistry can change. It also means that laboratory measured pH may not accurately reflect the pH measured in the field (Figure 20). There is not a strong correlation between field pH and laboratory pH in groundwater for monitoring undertaken throughout the SGS program ($R^2 = 0.55$) (Figure 20). This suggests that active geochemical processes are changing the pH of samples once removed from their environment. The average difference between laboratory and field pH was 0.29, the greatest difference was 0.74. Field pH is used in the interpretation of results.

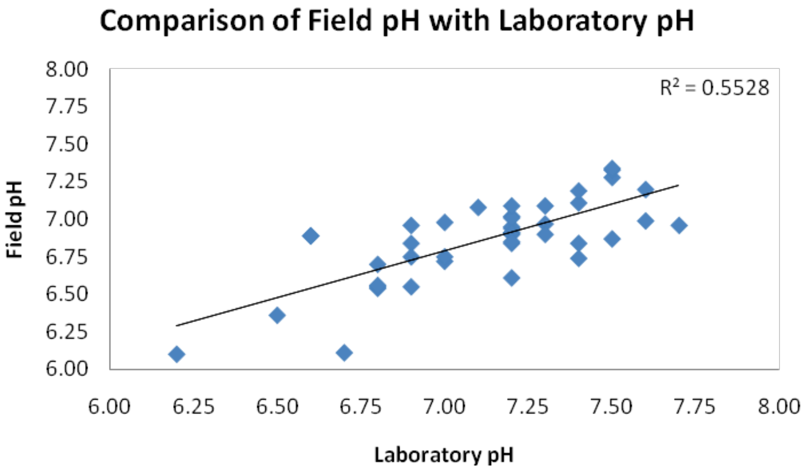


Figure 20 Comparison of field pH and laboratory pH, SGS investigations June 2009 – June 2010

Laboratory quality assurance and quality control (QA/QC) information for all investigations comprised laboratory duplicates, laboratory blanks, relative percentage differences (RPD’s) and matrix spike percentages. Laboratory blanks did not detect the presence of any contaminants of concern, relative percentage differences were recorded at less than 40% and matrix spike percentages were consistently between 75% and 120%.

An electrical balance or ionic balance can be used to assess the accuracy of laboratory data as the solution should be electrically neutral (Hounslow 1995). The sum of cations should equal the sum of anions in meq/L. To calculate meq/L:

$$\text{meq/L} = \frac{\text{mg/L x valency}}{\text{formula weight}}$$

The anion–cation balance is expressed in terms of a percentage and is expressed from the following equation:

$$\text{electrical balance (\%)} = \frac{\text{sum of cations} - \text{sum of anions}}{\text{sum of cations} + \text{sum of anions}} \text{ x } 100$$

The anion–cation balance should generally be within 5% to suggest satisfactory analysis has been achieved for each sample. If the electrical balance is greater than 5%, this may suggest poor analytical practices, the presence of other constituents that were not calculated in the balance, or that the water was highly acidic (Hounslow 1995).

Groundwater monitoring throughout SGS investigations generally recorded a cation–anion balance less than 5%, with eight results above 5% and three results (GLP_Ea 24/7/2009, GLP_Ea 11/9/2009 and GLP_Wa 22/9/2009) above 10% (the raw data is given in Appendix E). The results above 10% were all recorded for bores screened at the base of the Superficial aquifer which recorded very high bicarbonate concentrations.

Field and laboratory QA/QC procedures undertaken for Department of Water and DEC soil and groundwater investigations suggest the data is reliable from which informed decisions and reliable conclusions can be drawn.

4 Geology

4.1 Superficial formations

Drilling carried out for this study indicate that Lake Gwelup is underlain by quartz sands derived from Tamala Limestone. Tables 7 and 8 provide a brief lithological description of deep bores constructed on the western and eastern margins of the lake to depths of 38 m and 34 m respectively. A geological cross-section is shown in Figure 21. Drilling indicated the Tamala Limestone in this area is underlain by sediments of the Kings Park Formation.

Table 7 Lithological description of GLP_Ea, eastern edge of Lake Gwelup

Depth mbgl*	Age	Geological formation	Lithological description
0–34	Quaternary	Tamala Limestone	Silty sand, sand, calcarenite and sandy limestone
34–38	Tertiary	Kings Park Formation	Shale, clayey sand and silty clay

** metres below ground level*

Table 8 Lithological description of GLP_Wa, western edge of Lake Gwelup

Depth mbgl*	Age	Geological formation	Lithological description
0–31	Quaternary	Tamala Limestone	Sand, limestone and calcarenite
31–34	Tertiary	Kings Park Formation	Clay, calcarenite, sand and mudstone

** metres below ground level*

At the eastern edge of Lake Gwelup, Tamala Limestone extends to about 34 m below ground level or –26 m AHD. It comprised the following:

- Tamala Sand between 0 m and 14 mbgl
- medium grained calcarenite between 14 m and 30 mbgl
- sandy limestone material extending to 34 mbgl.

The Kings Park Formation comprises fine grained shale and was identified from 34 m to the end of the investigated core at 38 mbgl. Detailed logs are provided in Appendix A.

At the western edge of Lake Gwelup, Tamala Limestone extends to about 31 mbgl, or –24 m AHD. It comprised the following:

- Tamala Sand between 0 m and 8 mbgl

- a fully saturated limestone cavern (no sample retrieval) between 13 m and 14 mbgl. The lateral extent of the cavern is not known.

The Kings Park Formation, acting as a confining layer, extended from 31 mbgl to the maximum extent of drilling at 34 mbgl and comprised medium to course grained clay. Detailed logs are provided in Appendix A.

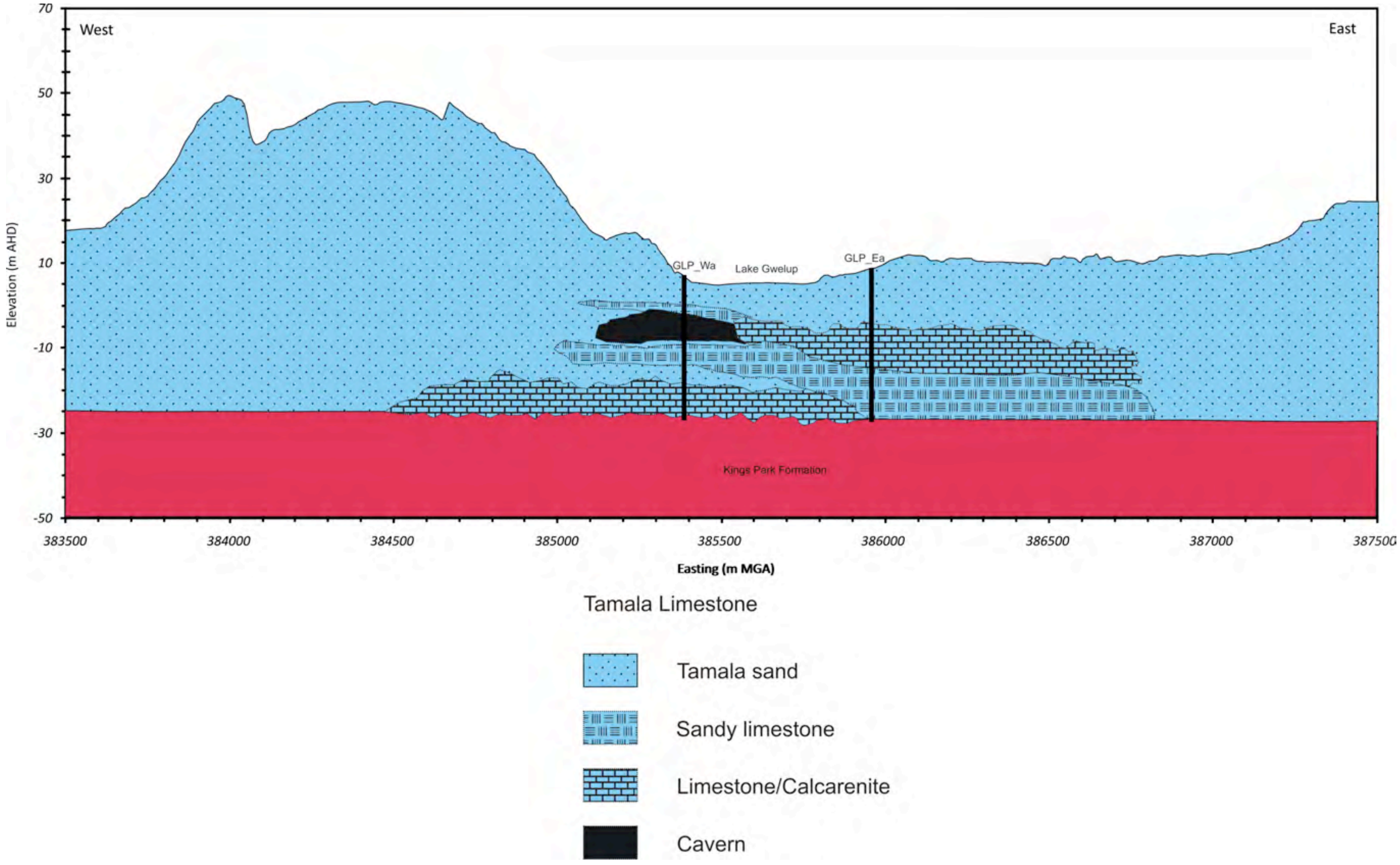


Figure 21 Lake Gwelup – geology cross-section

4.2 Acid sulfate soils

Results of field testing

The results of acid sulfate soil field testing at the eastern and western edges of Lake Gwelup for the Perth SGS investigation are presented in Figure 22.

At GLP_Ec pH_F ranged between 6.89 and 8.19, pH_{FOX} ranged between 5.15 and 5.76 (Figure 22). At GLP_Wc pH_F ranged between 3.24 and 7.21, pH_{FOX} ranged between 1.75 and 5.45 (Figure 22).

PASS was recorded between 5.0 m and 6.0 mbgl and AASS was recorded between 5.3 m and 5.5 mbgl at GLP_Wc. A significant drop in pH_F from 5.0 mbgl (pH 7.2) to 5.3 mbgl (pH 3.2) was recorded, indicating oxidation of pyrite. A corresponding decrease in pH_{FOX} at the same depth also indicates some yet to be oxidised PASS material remains.

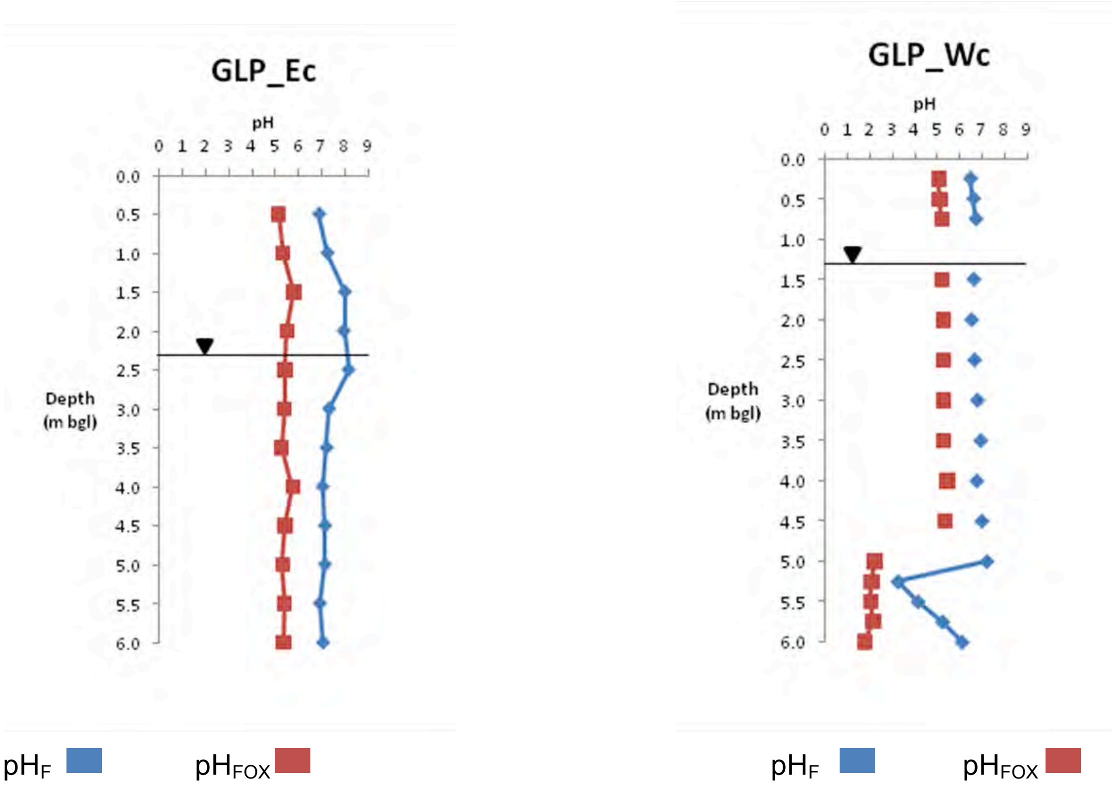


Figure 22 Field pH testing at GLP_Ec and GLP_Wc, groundwater levels: September 2009

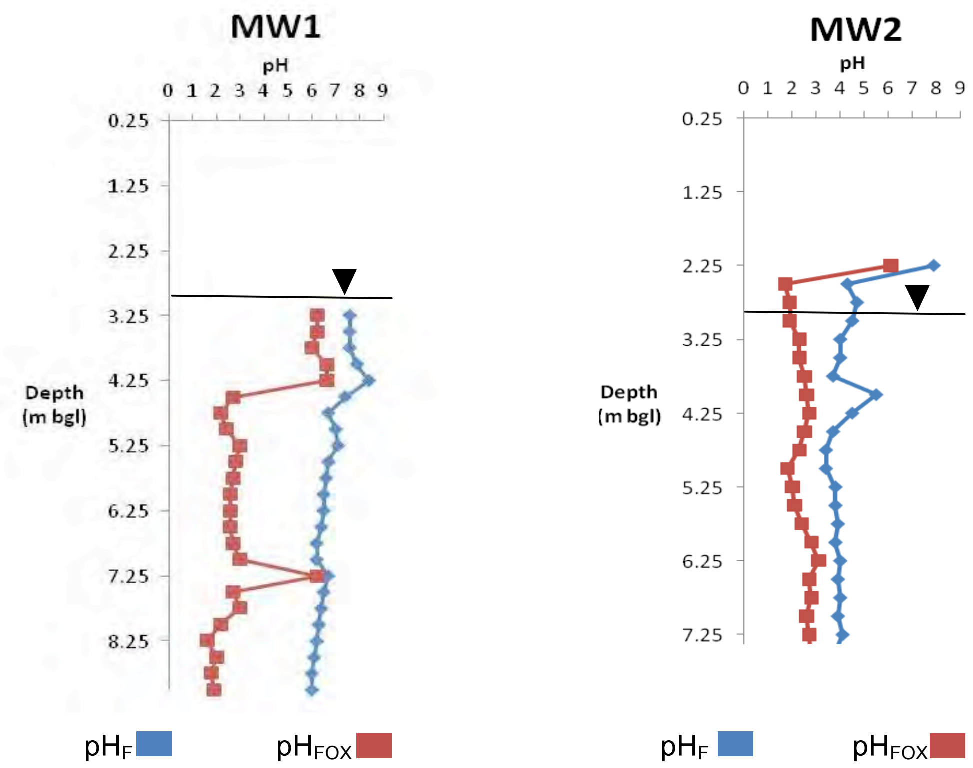
The results of ASS field testing undertaken up-hydraulic gradient of Lake Gwelup (MW1 to MW6, GWA9 and GM13) are presented in figures 23 and 24.

PASS was identified at MW1 (Figure 23). pH_F at MW1 ranged between 6.0 and 8.4, pH_{FOX} ranged between 6.6 and 1.2. PASS was identified between 4.5 m and 9.0 mbgl, intersected by a thin layer of sediment recording pH_{FOX} 6.2 at 7.25 mbgl.

AASS and PASS was identified at MW2 (Figure 23). pH_F ranged between 3.4 and 7.9, pH_{FOX} ranged between 1.7 and 6.1. AASS was identified between 3.25 m and 4.00 mbgl and between 4.25 m and 8.00 mbgl. PASS was identified between 4.0 m and 4.25 mbgl.

AASS and PASS was identified at MW3 (Figure 23). pH_F ranged between 3.9 and 6.9, pH_{FOX} ranged between 1.3 and 3.6. AASS was identified between 5.00 m and 5.75 mbgl and PASS was identified throughout the remainder of the investigated profile.

AASS and PASS was identified at MW4 (Figure 23). pH_F ranged between 2.9 and 8.4, pH_{FOX} ranged between 1.2 and 6.7. AASS was identified at 6.0 mbgl and PASS was identified from 3.25 mbgl to the end of the investigated profile.



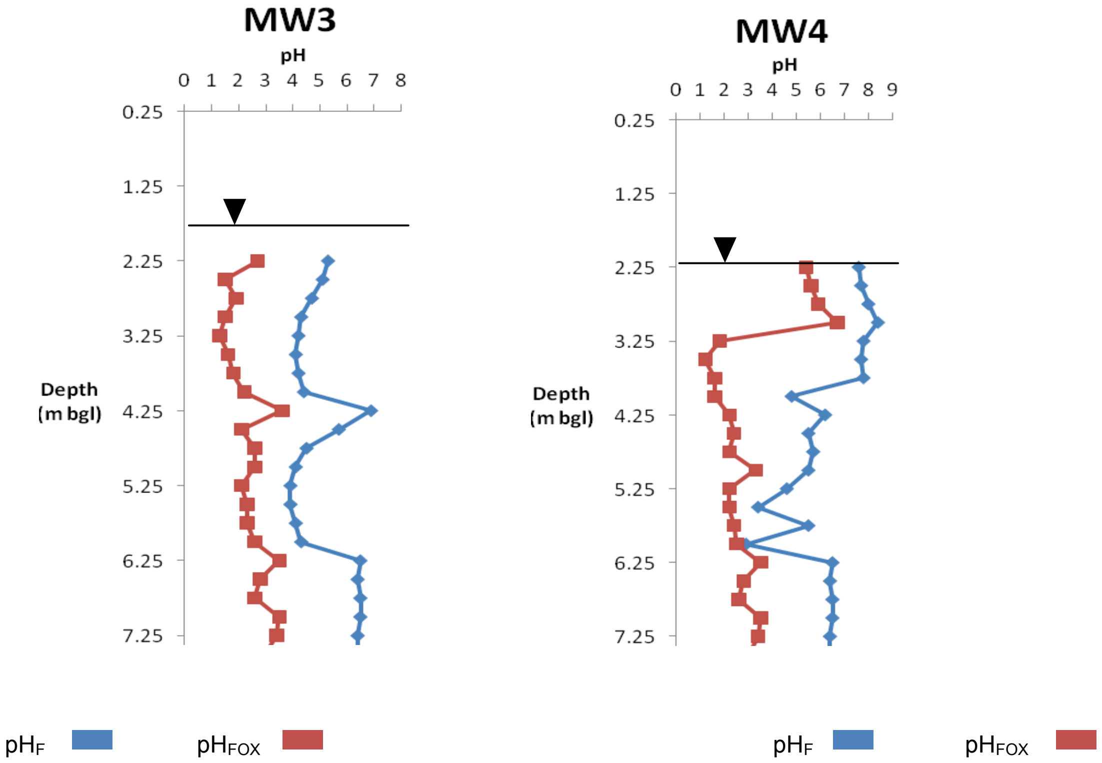


Figure 23 Field pH testing at MW1 to MW4, groundwater levels: November 2008

AASS and PASS was identified at MW5 (Figure 24). pH_F ranged between 3.1 and 5.7, pH_{FOX} ranged between 1.3 and 3.8. AASS was identified between 2.75 m and 8.0 mbgl and PASS was identified between 1.75 m and 2.75 mbgl.

PASS was identified at MW6 (Figure 24). pH_F ranged between 5.8 and 6.7, pH_{FOX} ranged between 1.3 and 4.9. PASS was identified between 3.25 m and 8.0 mbgl.

No AASS or PASS was identified at GWA9 (Figure 24). pH_F ranged between 6.7 and 8.4, pH_{FOX} ranged between 3.4 and 7.2.

PASS was identified at GM13 (Figure 24). pH_F ranged between 6.6 and 8.4, pH_{FOX} ranged between 2.4 and 6.9. PASS was identified at 7.75 mbgl.

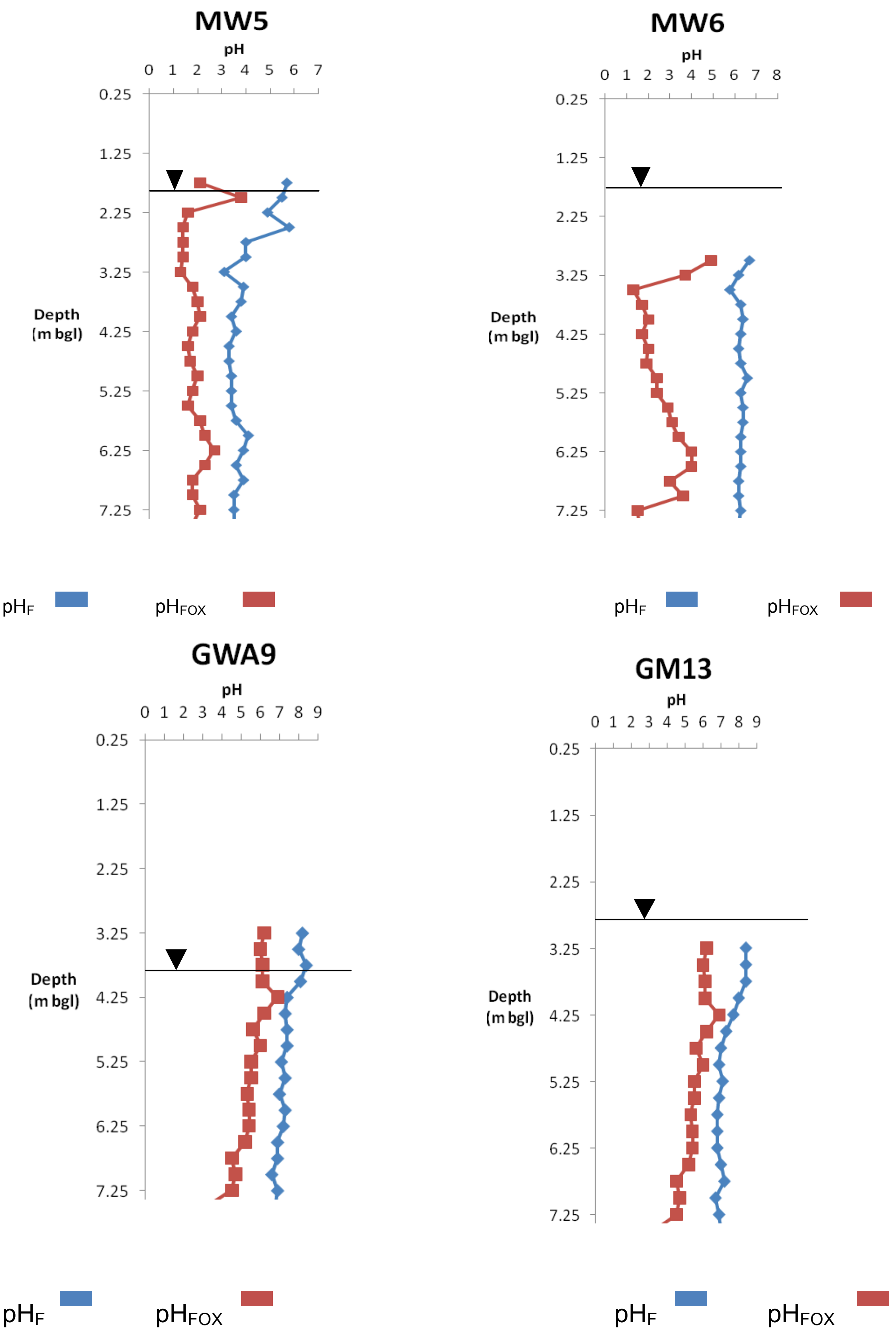


Figure 24 Field pH testing MW5, MW6, GWA9, GM13, groundwater levels: November 2008

Results of laboratory analysis

Four sediment samples were selected from GLP_Wc for laboratory analysis. comprising SPOCAS, S_{CR} and metals (tables 9 and 10). No samples were submitted for further analysis from GLP_Ec as field pH testing did not indicate the presence of any acid generating potential.

Table 9 Acid sulfate soil laboratory analysis, GLP_Wc

Sample depth mbgl	S_{CR} mol H+/t	S_{CR} %S	TAA mol H+/t	TPA mol H+/t	TSA mol H+/t	S_{POS} %S
2.0	< 2	<0.01	< 1	< 1	< 1	< 0.01
5.0	12	0.02	< 1	30	29	0.06
5.3	181	0.29	1	200	200	0.34
5.8	25	0.04	< 1	54	53	0.09

Note: Shading indicates results above action criteria (0.03% S or 18 mol H+/t) (DEC 2010)

*Table 10 Metals analysis for sediment samples at GLP_Wc (mg/kg). Reference to Ecological Investigation Levels in **bold italics** (DEC 2010)*

Sample depth (m bgl)	Al	As	Cd	Cr	Fe	Mn	Ni	Se	Zn
	-	***20***	***3***	***400***	***-***	***500***	***60***	***-***	***200***
2.0	3920	< 0.5	< 0.5	11.0	440	2.3	0.61	< 0.5	< 0.5
5.0	3270	0.76	< 0.5	12.0	1030	4.6	0.73	< 0.5	< 0.5
5.3	6030	4.90	< 0.5	19.0	5950	21.0	1.60	< 0.5	2.0
5.8	2330	1.10	< 0.5	6.1	1330	4.1	0.65	< 0.5	< 0.5

The main results from laboratory analysis for metals from selected samples at GLP_Wc are:

- Aluminium concentrations in sediments around Lake Gwelup ranged between 3270 mg/kg and 6030 mg/kg with higher aluminium concentrations identified in acidic conditions (pH 3.2 at 5.3 mbgl).
- Iron concentrations are elevated throughout GLP_Wc, with higher concentrations at low pH. Iron concentrations ranged between 440 mg/kg (2.0 m) and 5950 mg/kg (5.3 m).
- Arsenic, chromium, manganese, nickel and zinc were identified in soil at GLP_Wc. However concentrations are low and possibly reflective of natural ‘background’ conditions.

Results of acid sulfate soil laboratory analysis for investigations conducted up-gradient of Lake Gwelup detailed in Table 11. Results of metals analysis in Table 12.

Laboratory results confirm the presence of ASS at MW1 to MW6.

Table 11 ASS results up-gradient of Lake Gwelup 18 - 20 November 2008

Bore ID	**Sample depth mbgl**	**S_{CR} mol H+/t**	**S_{CR} %S**	**TAA mol H+/t**	**TPA mol H+/t**	**TSA mol H+/t**	**S_{POS} %S**
MW1	4.50	12	0.02	< 2	39	39	0.13
	5.00	< 2	< 0.02	< 2	< 2	< 2	0.02
	6.25	< 2	< 0.02	< 2	< 2	< 2	< 0.02
	8.50	18	0.03	< 2	64	62	0.12
MW2	2.50	< 2	< 0.02	4	< 2	< 2	< 0.02
	3.50	< 2	< 0.02	< 2	< 2	< 2	< 0.02
	5.00	< 2	< 0.02	9	24	16	0.02
	5.00 (D)	< 2	< 0.02	10	27	17	0.04
	6.00	< 2	< 0.02	4	< 2	< 2	< 0.02
	8.00	< 2	< 0.02	7	15	7	< 0.02
MW3	2.50	106	0.17	10	287	277	0.50
	3.25	75	0.12	6	224	218	0.43
	5.25	< 2	< 0.02	< 2	< 2	< 2	< 0.02
	8.00	< 2	< 0.02	2	10	8	0.03
MW4	3.50	62	0.10	< 2	58	58	0.12
	5.25	< 2	< 0.02	15	26	11	< 0.02
	6.00	< 2	< 0.02	6	10	4	< 0.02
	8.00	37	0.06	< 2	26	25	0.06
MW5	2.50	62	0.10	5	118	112	0.18
	3.25	44	0.07	2	87	85	0.14
	5.50	12	0.02	6	10	4	< 0.02
	7.25	12	0.02	5	22	17	0.03
MW6	3.50	206	0.33	138	1370	1230	0.39
	4.50	< 2	< 0.02	< 2	4	2	0.03
	7.50	94	0.15	6	175	169	0.29
GM13	7.75	37	0.05	< 2	< 2	< 2	< 0.02

Note: Shading indicates results above action criteria (0.03% S or 18 mol H+/t) (DEC 2011)

Table 12 Metals analysis in sediment up-gradient of Lake Gwelup 18-20 November 2008 (mg/kg).

Bore ID	Sample Depth	Al -	As 20	Cd 3	Cr 400	Fe -	Mn 500	Ni 60	Se -	V 50	Zn 200
MW1	4.50	900	7	< 1	3	2 470	< 5	< 2	< 5	< 5	< 5
	5.00	290	< 5	< 1	< 2	910	< 5	< 2	< 5	< 5	< 5
	6.25	400	< 5	< 1	< 2	440	< 5	< 2	< 5	< 5	< 5
	8.50	1310	12	< 1	7	1 920	< 5	< 2	< 5	< 5	< 5
MW2	2.50	550	< 5	< 1	3	3 760	< 5	< 2	< 5	< 5	< 5
	3.50	250	11	< 1	< 3	390	< 5	< 2	< 5	< 5	< 5
	5.00	1780	< 5	< 1	15	1 260	< 5	< 2	< 5	7	< 5
	5.00 (D)	1910	11	< 1	17	1 580	< 5	< 2	< 5	7	< 5
	6.00	970	< 5	< 1	6	390	< 5	< 2	< 5	< 5	< 5
	8.00	2420	< 5	< 1	26	1 500	< 5	< 2	< 5	11	< 5
MW3	2.50	740	72	< 1	3	27 500	< 5	< 2	< 5	< 5	< 5
	3.25	730	64	< 1	4	11 200	< 5	< 2	< 5	9	< 5
	5.25	380	< 5	< 1	2	280	< 5	< 2	< 5	< 5	< 5
	8.00	2180	5	< 1	17	3 030	< 5	< 2	< 5	18	< 5
MW4	3.50	200	18	< 1	< 2	2 260	< 5	< 2	< 5	< 5	< 5
	5.25	1850	< 5	< 1	11	2 050	< 5	< 2	< 5	< 5	< 5
	6.00	1250	< 5	< 1	7	1 220	< 5	< 2	< 5	< 5	< 5
	8.00	2510	9	< 1	20	4 270	< 5	3	< 5	39	< 5
MW5	2.50	70	9	< 1	< 2	1 570	< 5	< 2	< 5	< 5	< 5
	3.25	100	20	< 1	< 2	2 350	< 5	< 2	< 5	< 5	< 5
	5.50	1810	< 5	< 1	18	920	< 5	< 2	< 5	10	< 5
	7.25	1860	< 5	< 1	15	920	< 5	< 2	< 5	13	< 5
MW6	3.50	620	196	< 1	10	134 000	1240	< 2	< 5	9	< 5
	4.50	240	14	< 1	6	4 700	< 5	< 2	< 5	22	< 5
	7.50	1480	30	< 1	18	3 880	< 5	< 2	< 5	7	< 5

Bore ID	Sample Depth	Al -	As 20	Cd 3	Cr 400	Fe -	Mn 500	Ni 60	Se -	V 50	Zn 200
GM13	7.75	1720	< 5	< 1	10	260	< 5	< 2	< 5	6	< 5

** From Department of Environment and Conservation (2010)*

Note: Shading indicates results above ecological investigation levels (DEC 2010)

Sediment acidity

Acid sulfate soil field testing conducted either side of Lake Gwelup revealed two contrasting profiles with respect to their acid generating potential. Potential for acidification from the oxidation of pyrite was identified at GLP_Wc from 5.0 m below ground level. However investigations did not detect the potential for acidity at GLP_Ec.

Acidification due to the decomposition of organic matter and production of carbon dioxide is not likely to lower pH of soil to below 4.6 (Appelo & Postma 2005). Organic acidity may be the main cause of pH decline to between 4.5 and 5.0.

The oxidation of pyrite appears to have caused a significant pH decline between 5.3 m and 5.5 m below ground level at GLP_Wc. This may be due to the fluctuation in watertable levels shown during SGS monitoring, which cannot be accurately defined as the bore (GLP_Wc) was dry between December 2010 and June 2011. A high seasonal amplitude in groundwater levels was recorded at GLP_Wc. Groundwater levels at GLP_Wc fluctuated between 1.29 mbgl (September 2009) to below the extent of the screened interval at 4.0 mbgl. There is a more significant groundwater level decline at the western edge of the lake than the eastern edge of the lake (Figure 26). The rate of groundwater level decline appears to be magnified when the lake dries.

The rate and extent to which the oxidation of pyrite causes pH to decline depends on the carbonate content of the sediment. It is likely that sufficient carbonaceous material is present at 5 m bgl at GLP_Wc to prevent pH decline despite sulfidic acid generating potential at this depth (S_{POS} 0.06% S). The portion of inorganic sulfidic acidity (S_{CR}) at this depth was 0.02% S. Higher levels of sulfidic acidity (S_{POS} 0.34% S; S_{CR} 0.29% S) is present at 5.3 mbgl which appears to have sufficiently depleted the available buffering capacity lowering pH to 3.2.

The results of sediment investigations show that where acidity is present, identified by low pH, very little titratable actual acidity (TAA) or retained acidity was recorded (Table 9 and Table 11). Titratable peroxide acidity (TPA) is high at more acidic locations. However, neither TPA nor TAA quantitatively recover retained acidity from iron oxyhydroxysulfate minerals such as jarosite and other iron and aluminium sulfate minerals (Ahern et al. 2004). TAA is measured and calculated from titrations undertaken in a laboratory and indicates the presence of soluble acidity. Acidic sediments may therefore contain insoluble compounds such as jarosite, schwertmannite and other iron and aluminium sulfate minerals that are stable at low pH (< 4.5). If so, these compounds may be mobilised if geochemical conditions change, such as from significant fluctuations in watertable levels (Ahern et al. 2004).

Investigations did not detect the presence of peat either side of Lake Gwelup, however peat was recorded up-gradient of the lake. Peat has previously been excavated and removed from the newly constructed residential development Gwelup Park. Small layers of peat were found to remain at MW3 and MW6. Organic rich peaty sand was identified between 2.25 m and 2.50 m below ground level at MW3, and lighter more organic rich black peat material was found between 3.25 m and 4.00 m below ground level at MW6. The laboratory results indicate peat to be a significant source of stored and potential acidity with S_{POS} in MW3 up to 0.50% S (S_{CR} 0.17% S) and MW6 recording 0.39% S (S_{CR} 0.33% S) with very high concentrations of TPA at 287 moles H^+ per tonne and 1,370 moles H^+ per tonne respectively. The national action criteria for management of acidic soil is 0.03% S or 18 moles H^+ per tonne (DEC 2011).

Metals in sediments

Acidification can cause the mobilisation of metals, such as aluminium, arsenic and iron. Metals analysis recorded elevated concentrations of aluminium, iron and arsenic in sandy sediment and peat. While this provides a good indication of sediment quality and potential management issues, metal levels in sediment does not often correlate well with potential environmental problems upon acidification because metals in water are generally orders of magnitude less than levels in sediment.

Aluminium concentrations were higher in sediment collected from the edges of Lake Gwelup than in sediment up-gradient of the lake. This may indicate higher clay content in sediment near the lake. The highest aluminium concentration was identified in acidic conditions at 5.3 m below ground level at GLP_Wc, recording 6030 mg/kg. Acid conditions instigate the release of previously sorbed aluminium.

Iron was recorded at very high concentrations, up to 134 000 mg/kg, in the peat material up-gradient of Lake Gwelup. Higher concentrations of iron were also identified in areas subject to acidification. GLP_Wc at 5.0 mbgl, pH 3.24, recorded iron at 5950 mg/kg in comparison with 5.0 m below ground level, pH 7.21 and iron at 1030 mg/kg.

Elevated concentrations of arsenic were identified in peat up-hydraulic gradient of Lake Gwelup, while much lower concentrations were recorded in sand and silty sand material. Arsenic was recorded at 196 mg/kg in peat at MW6. This exceeds the screening health risk criteria of 100 mg/kg for residential sites (DEC 2010). Acid generating processes such as the oxidation of pyrite may lead to the release of arsenic into the groundwater system.

5 Hydrogeology

5.1 Lake water levels

EWP 465 is a water level gauge located at the western edge of Lake Gwelup (Figure 25) and has been used to monitor water levels at the lake since 1921. Water levels at Lake Gwelup were recorded up to 8.1 m AHD in 1968 (Figure 27). This was the maximum controlled water level as lake levels were regulated by the elevation of the outflow drain at the western edge of the lake (Arnold 1990).

Figure 25 Staff gauge EWP 465, western edge of Lake Gwelup, 3 September 2010

Lake Gwelup was recorded to have dried on three occasions between 1921 and 2001, however since 2001 the lake has dried each year (figures 26 and 27). The bathymetry of the lake basin at EWP 465 is 5.0 m AHD and this has historically been used as the reference point to determine whether the lake has dried. The staff gauge (EWP 465) is not located at the lowest point in the lake and therefore it is likely that in some years when the lake has been recorded as dry, there was still some surface water ponding. Previous investigations suggest the bathymetry of the lake basin to be below 4.6 m AHD. The lowest water levels at the lake were recorded in 1980 at 4.49 m AHD (Arnold 1990).

Figure 26 Lake Gwelup when dry, view to the west, 21 March 2012

The seasonal drying of lakes in the Perth metropolitan area has increased in the last 30 years consistent with declining rainfall and the increased use of groundwater. Other groundwater-dependent flow-through lakes on the Gnangara Mound have dried for greater portions of the year since the 1970s. This includes Lake Gnangara, Lake Jandabup and Lake Mariginiup. Lake Mariginiup was not recorded to have dried until 1978. However, the lake dried on average for approximately two months each year between 1978 and 1997. Since 1998 the lake has dried for about six months of the year (Searle et al. 2010). The drying of both Lake Mariginiup and Lake Gnangara has coincided with lake acidification which highlights the importance of groundwater management in maintaining water levels at environmentally sensitive areas.

To maintain sediment processes, sediments must remain saturated throughout summer each year. If sediment processes are to be maintained, the watertable must not decline to depths greater than the extent to which capillary forces can maintain sediment saturation. As an approximate guide, the watertable should not decline more than 0.5 m below the lake bed surface (Froend et al. 2004b).

There are no Ministerial conditions relating to the maintenance of water levels in Lake Gwelup. This is despite the declining trend in groundwater levels, lake levels and the location of the Gwelup bore field in the lake's capture zone.

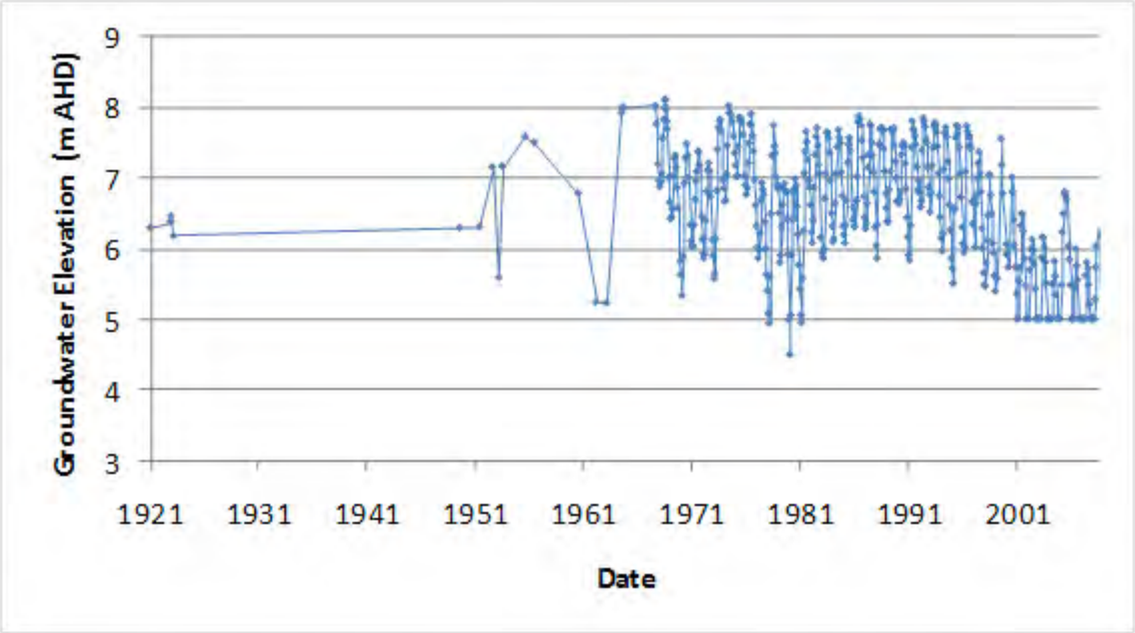


Figure 27 Lake Gwelup surface water levels, EWP 465, 6162504, sediment elevation at EWP 465 is 5.0 m AHD

5.2 Groundwater levels

Hydrographs for seven Department of Water groundwater monitoring bores near Lake Gwelup (GM14, GM13, GM30, 8525, GM17, GCM, GM8) show a general decline in groundwater levels since the commencement of monitoring in the 1970s (hydrographs in Appendix G, bore locations illustrated in figures 33 and 34). This has coincided with the commencement of groundwater abstraction from the Gwelup bore field, a declining trend in annual rainfall and increased residential development. This assessment of declining groundwater levels is supported by Yesertener (2008). Groundwater levels in the Superficial aquifer have declined by between 1.0 m and 5.0 m when compared with groundwater levels initially recorded between 30 and 40 years ago. Between 2005 and 2010, the data shows groundwater levels ceased declining at rates observed in the 1990s despite continued low rainfall (Appendix G). This emphasises the connection between groundwater abstraction from the Superficial and Mirrabooka aquifers and groundwater level decline around Lake Gwelup. Groundwater abstraction has ceased consistently increasing from the Superficial and Mirrabooka aquifers since the early 2000s (Figure 4). Groundwater level information around Lake Gwelup (Appendix G) also indicates that increased rates of abstraction from the Leederville and Yarragadee aquifers since 2000 do not appear to have influenced groundwater levels at the watertable.

Yesertener (2008) used cumulative deviation from the mean (CDFM) methods to suggest groundwater level decline is mainly due to a combination of groundwater abstraction and reduced rainfall. Most significant groundwater level decline is centred on the Gwelup bore field up-gradient of Lake Gwelup and in its capture zone. Yesertener (2008) documented a 3.75 m decline in groundwater levels in a 6.0 km radius of the Gwelup bore field. Of this total decline 3.0 m was attributed to groundwater abstraction for the period 1979 - 2005 (Yesertener 2008). This suggests the effects of groundwater abstraction are more significant on groundwater level decline in Gwelup than reduced rainfall.

The proportional effects of reduced rainfall and groundwater abstraction should be considered in terms of location in the regional Gnangara Mound flow field (Figure 12). The influence of reduced rainfall is greater at the crest of the mound, near Lake Pinjar and the Yeal Nature Reserve. Reduced rainfall is predicted to have caused groundwater level decline up to 4.0 m near the crest of the mound (Yesertener 2008). The proportional influence of lower rainfall on declining groundwater levels decreases with distance from the crest of the mound. Gwelup is located away from the crest of the mound and the influence of groundwater abstraction appears higher than reduced rainfall on lowering the watertable in this area.

Three Department of Water monitoring bores up-gradient of Lake Gwelup (GM13, GM14, GM30) indicate a gradual increasing trend in groundwater levels since 2005, while groundwater levels at the other four monitoring bores (8525, GM17, GCM, GM8) have remained stable between 2005 and 2010. The stabilisation of groundwater levels since 2005 is likely to be due to reduced abstraction from the Superficial and Mirrabooka aquifers. This suggests a high influence of abstraction from the Superficial aquifer on lake levels and groundwater levels and indicates further reduction will assist in recovery of lake levels and the watertable. It also illustrates the significance of the previously described environmental sensitivity classification (ESC). Production bores of the Gwelup bore field have an ESC of 3, that permits increased abstraction in certain circumstances.

Seasonal groundwater level fluctuation around Lake Gwelup appears to be between 1.0 m and 1.5 m, illustrated by the differences in groundwater elevations in the seasonal watertable contour diagrams (figures 33 and 34). However, groundwater levels near the watertable at the western edge of Lake Gwelup (GLP_Wc) fluctuated by more than 2.7 m over the monitoring period. This is likely due to the presence of limestone caverns to the west of the lake.

The magnitude of groundwater level decline at GLP_Wc increases as the lake dries (Figure 28). This significant groundwater level decline contrasts with the gradual decline and recovery exhibited by GLP_Ec. The precise extent of groundwater level fluctuation could not be assessed at GLP_Wc because the bore was dry in February, March and May 2010. The elevation at the base of the screened interval at GLP_Wc is 3.1 m AHD, and this indicates the bore to be dry (Figure 28).

As the lake dried, groundwater levels at the western edge of the lake declined at a greater rate and to a greater extent than the eastern edge of the lake (Figure 28). Groundwater levels at GLP_Wc declined by at least 2.0 m in two months, between December 2009 and February 2010. This suggests that as Lake Gwelup dries, the loss of surface water–groundwater connection changes groundwater flow regimes.

The drying of the lake appears to be exacerbated by the presence of caverns (as shown in the geological cross-section; Figure 21) beneath the western edge of the lake. A similar situation is inferred at the down-gradient edge of Lake Nowergup (located about 37 km north of Perth) which has been artificially supplemented with water from the Leederville aquifer since 1989 (Searle et al. 2011). Groundwater levels at the western down-gradient edge of Lake Nowergup also record greater seasonal amplitudes than bores on the up-gradient side of the lake. These amplitudes were recorded up to 2.7 m at the down-gradient edge of the lake in 2002. The likely presence of solution channels (caverns) and the permeability of Tamala Limestone was considered the probable cause for the steep hydraulic gradient at the western edge of Lake Nowergup (Searle et al. 2011).

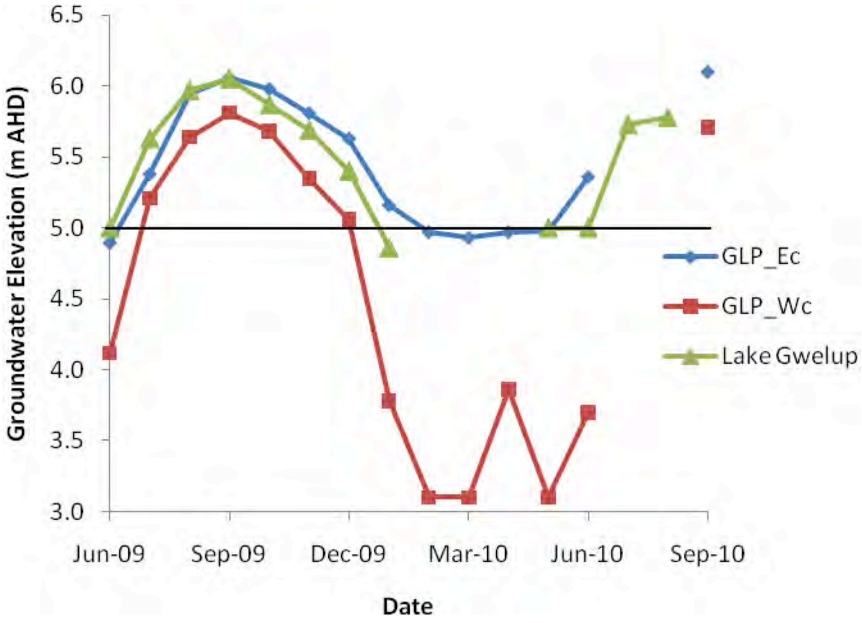


Figure 28 Groundwater levels at the watertable either side of Lake Gwelup, June 2009 to September 2010. The lake bed level at EWP 465 is 5.0 m AHD (black line). Note: GLP_Wc was dry in February, March and May 2010 and these points represent the base of the screened interval of the monitoring bore

A high level of connectivity is shown by similar groundwater elevations shown throughout most of the vertical extent of the Superficial aquifer (figures 29 and 30). However different flow regimes are observed at GLP_Wc (figure 30), strongly suggesting heterogeneity in the upper half of the Superficial aquifer. This is suspected to be linked to the presence of caverns at the western edge of the lake, the extent of which are unknown.

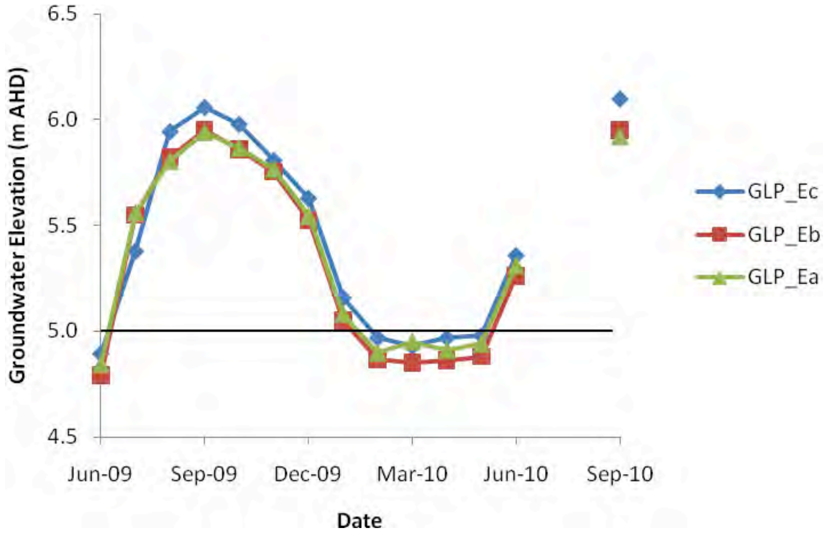


Figure 29 Groundwater levels at the eastern edge of Lake Gwelup between June 2009 and September 2010. Black line represents lake bed at EWP 465 – 5.0 m AHD.

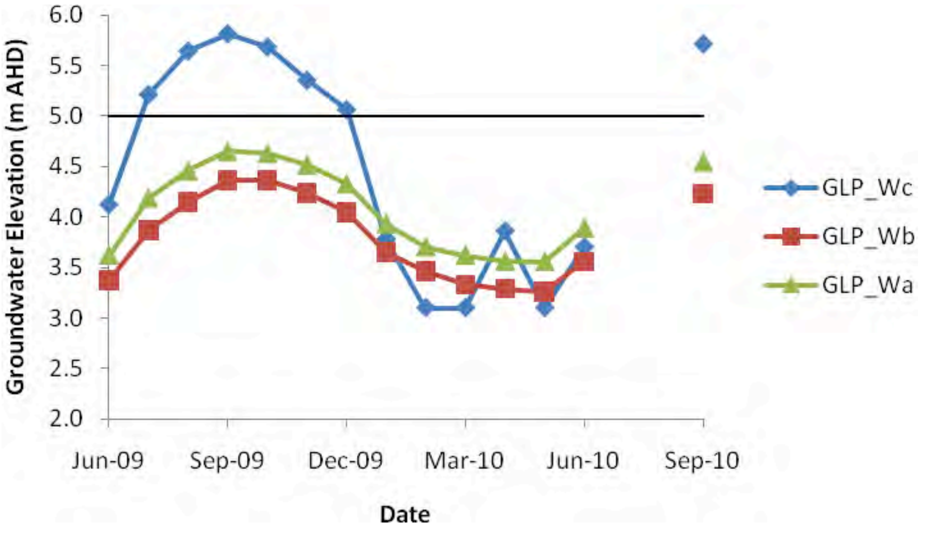


Figure 30 Groundwater levels at the western edge of Lake Gwelup between June 2009 and September 2010. Note: GLP_Wc was dry in February, March and May 2010 – these plots represent the base of the screened interval. Black line represents lake bed at EWP 465 – 5.0 m AHD

5.3 Groundwater flow

Groundwater elevations in Tamala Limestone around Lake Gwelup ranged between 3.08 m AHD (GM8, 21 April 2010) to 10.57 m AHD (GM14, 27 October 2009). Groundwater flows in an east to west direction. Watertable contour diagrams were developed using groundwater elevation data from surrounding Department of Water monitoring bores and SGS bores from this investigation (figures 33 and 34).

Lake Gwelup is a surface expression of the watertable and functions as a flow-through lake when the lake is full, such as in October 2009 (figures 31 and 33). This is based on an upward hydraulic gradient on the up-gradient eastern side of Lake Gwelup, as groundwater discharges into the lake. A downward hydraulic gradient is present on the down-gradient western edge of Lake Gwelup, recharging the Superficial aquifer as water discharges from the lake (Townley et al. 1993). Figure 31 (October 2009) illustrates the predominantly horizontal nature of groundwater flow in the lower half of the Superficial aquifer, indicating groundwater connection with Lake Gwelup is confined to approximately half the vertical extent of the aquifer.

When the lake is dry, the groundwater flow regime changes (such as in May 2010), and there is no interaction between groundwater and surface water (figures 32 and 34). When the lake is dry, groundwater flows horizontally with minimal vertical flow. Throughout the 20th century, when the lake was permanently inundated for extended periods of time, it is likely there was constant interaction between surface water and groundwater.

As the staff gauge (EWP 465) is not located at the lowest point in the lake, some minor local scale groundwater-surface water interaction may be evident if some ponding remains in the lake. For example, this may have been evident in January and/or June 2010 when GLP_Wc measured water elevations less than 5.0 m AHD and GLP_Ec recorded groundwater elevations at 5.16 m AHD and 5.36 m AHD respectively.

5.4 Lake Gwelup capture zone

The geometry of Lake Gwelup and the underlying Superficial aquifer can provide an indication of the geometry of the capture zone (Townley et al. 1993). The capture zone of Lake Gwelup is located up-gradient to the east and a release zone down-gradient to the west. The quality of groundwater in the capture zone should be considered in the design of management objectives for Lake Gwelup.

A theoretical approach can be used to provide an indication of the vertical extent of a capture zone according to Townley et al. (1993). Assuming isotropic conditions, the vertical extent of the capture zone is calculated by the length of the lake (2a) in the direction of average groundwater flow relative to the thickness of the aquifer (B) (Townley et al. 1993). The length of Lake Gwelup (2a), based on the centre line through the lake in an east to west direction is about 400 m. The width of the lake is therefore regarded as the distance in the north-south direction, and is about 500 m.

The Superficial aquifer extends to between 32 m and 35 m beneath Lake Gwelup. According to Townley et al. (1993) given Lake Gwelup is more than 10 times longer than the depth of the underlying aquifer, under isotropic conditions the lake will draw from the total depth of the aquifer.

The data suggests the vertical extent of the capture zone may extend to the middle of the Superficial aquifer, however does not extend to the base of the aquifer. This indicates that while methods described by Townley et al. (1993) may provide a reasonable estimation in certain circumstances, it should be understood that some degree of anisotropy appears to be present. This is also evident from the conceptual geological representation of Tamala Limestone beneath Lake Gwelup (Figure 21).

Under isotropic conditions, the width of the capture zone is estimated to be twice the width of the lake (Townley et al. 1993). Based on the previous interpretation of width, (north–south direction), the capture zone is 1 km wide. However a more precise assessment of the capture zone will need to consider groundwater heads over a wider area and anisotropic characteristics that may include the resistance of sediments at the base of the water body as well as varying rates and volumes of groundwater flowing in and out of the lake.

Given the uncertainties in anisotropic characteristics, the extent of the capture zone of Lake Gwelup cannot be accurately estimated using methods described by Townley et al. (1993). However, it is very likely that groundwater monitored 400 m up-gradient of Lake Gwelup is within the lake’s capture zone.

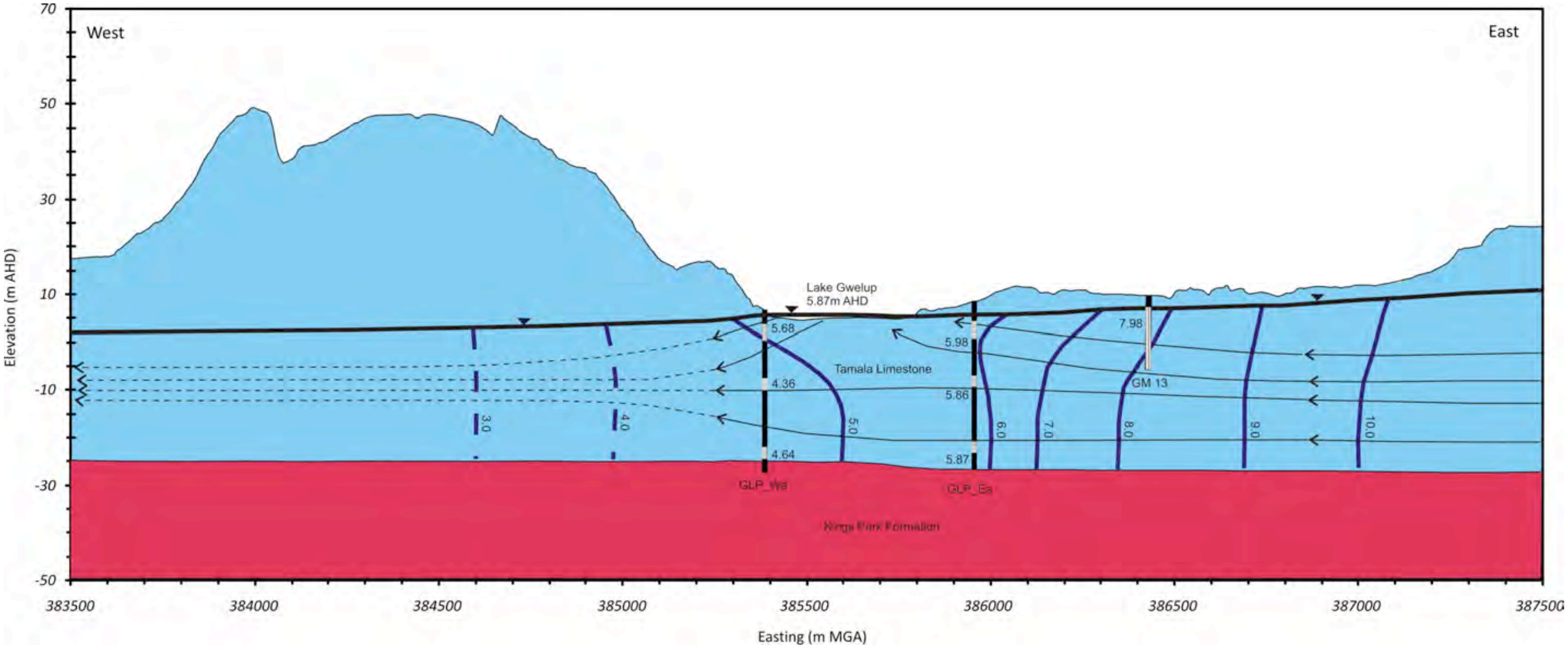


Figure 31 *Cross-section beneath Lake Gwelup. Groundwater flow paths when the lake contains water at 5.87 m AHD, equivalent to 0.87 m of ponded water at EWP 465 (October 2009)*

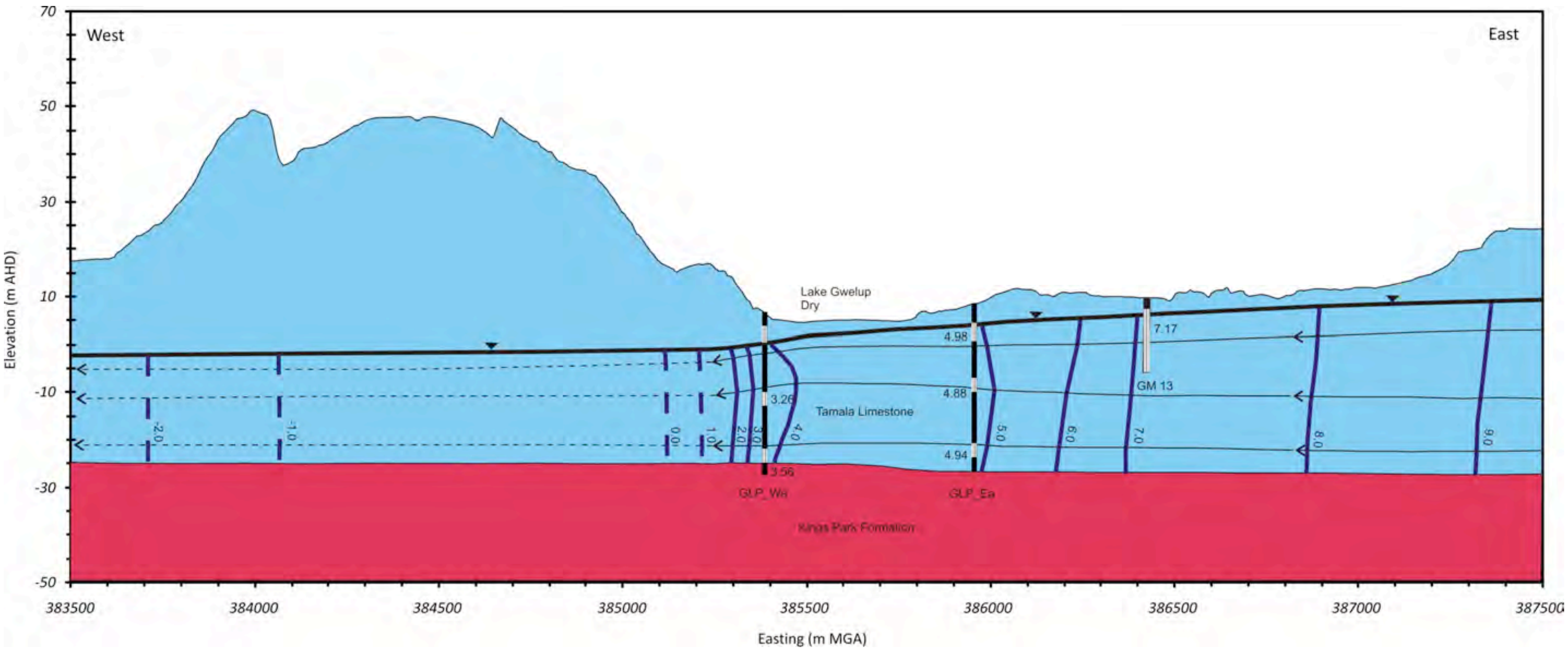


Figure 32 Cross-section beneath Lake Gwelup. Groundwater flow paths when the lake is dry (May 2010)

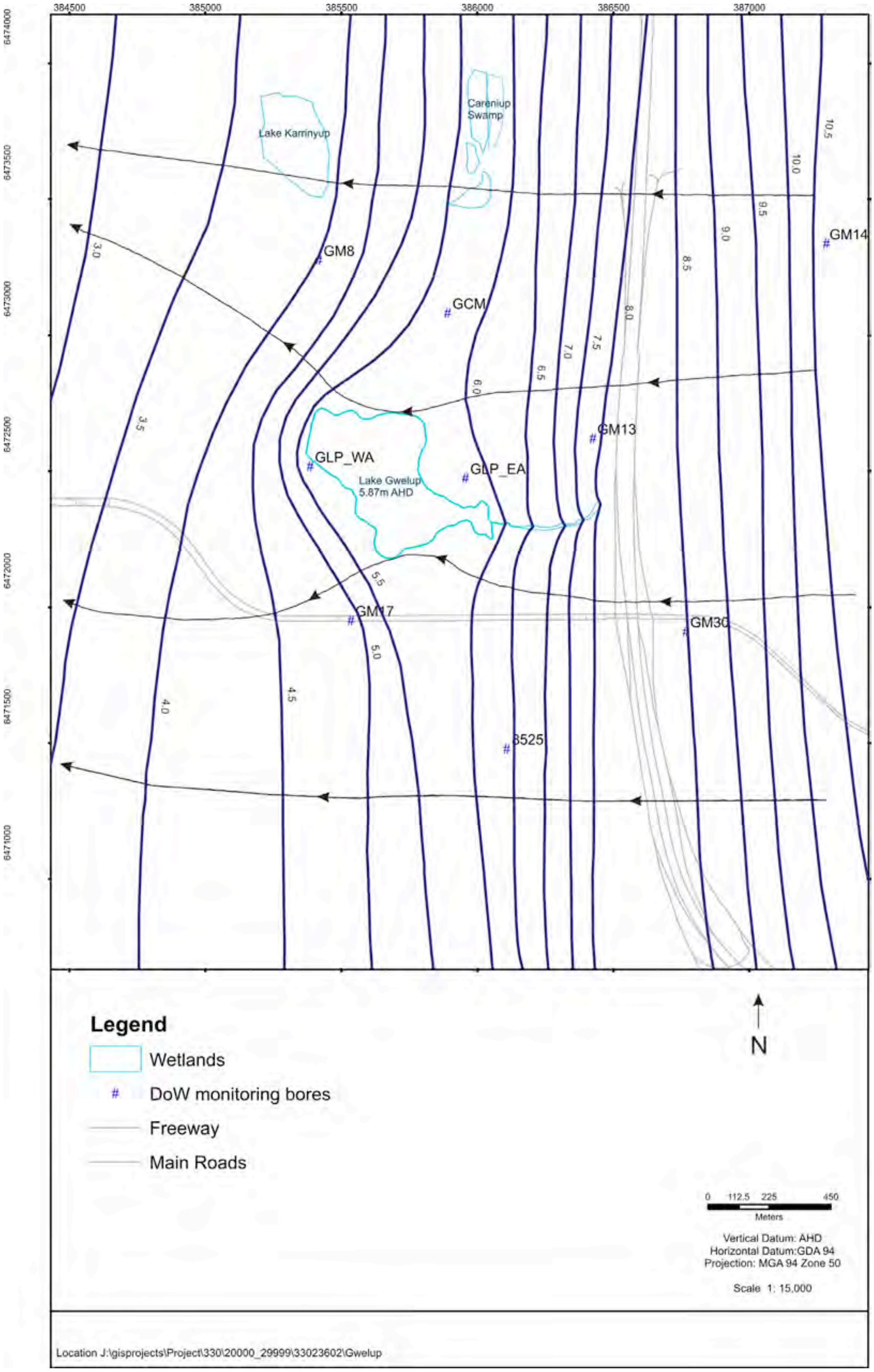


Figure 33 Watertable contours showing approximate capture and release zones (October 2009)

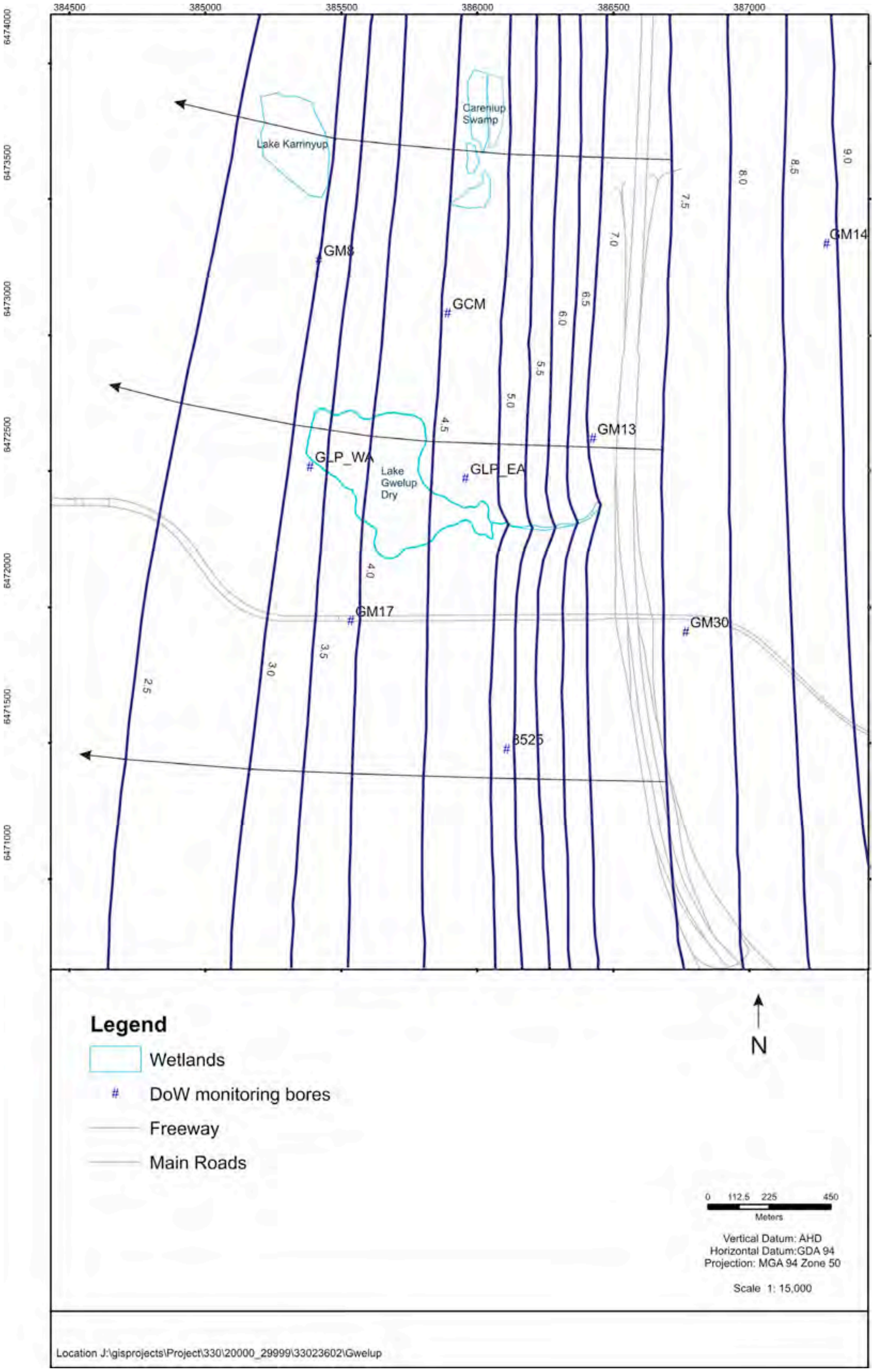


Figure 34 Watertable contours when the lake is dry (May 2010)

5.5 Hydrogeochemistry

Major ions

Concentrations of major ions (calcium, sodium, magnesium, potassium, sulfate, chloride and alkalinity) in the Perth SGS investigation were used to create a ‘Piper diagram’ (Figure 35). The chemical composition of groundwater generally reflects the nature of the geology and the diamond part of a Piper diagram can be used to characterise different water types (Hounslow 1995). Groundwater beneath Lake Gwelup is of calcium bicarbonate type reflecting chemical characteristics derived from Tamala Limestone. Groundwater 400 m up-gradient of Lake Gwelup is also characterised by water of calcium bicarbonate type comprising a similar composition of base cations dominated by calcium, followed by sodium, magnesium and potassium.

The Piper diagram shows higher sodium concentrations in groundwater flowing into the lake, than in groundwater flowing out of the lake. This is particularly evident for groundwater in the top half of the Superficial aquifer. Lower concentrations of sodium in groundwater flowing out of Lake Gwelup may indicate dilution with rain water. Lower levels of base cations flowing out of the lake may reflect acid generating processes from sulfide oxidation exacerbated by the drying of lake sediments. A mixing of water type, for example with sodium chloride type water of Bassendean Sand, is not considered likely given Lake Gwelup is not located close to a geological boundary or inter-dunal depression.

The Piper diagram shows sulfate and chloride concentrations increase with depth in the Superficial aquifer beneath Lake Gwelup. The highest sulfate concentrations were recorded in intermediate bores. It is interesting to note that groundwater at the base of the aquifer, beneath the western edge of the lake (GLP_Wa), is not dominated by sulfate and is different to the other bores investigated in this study.

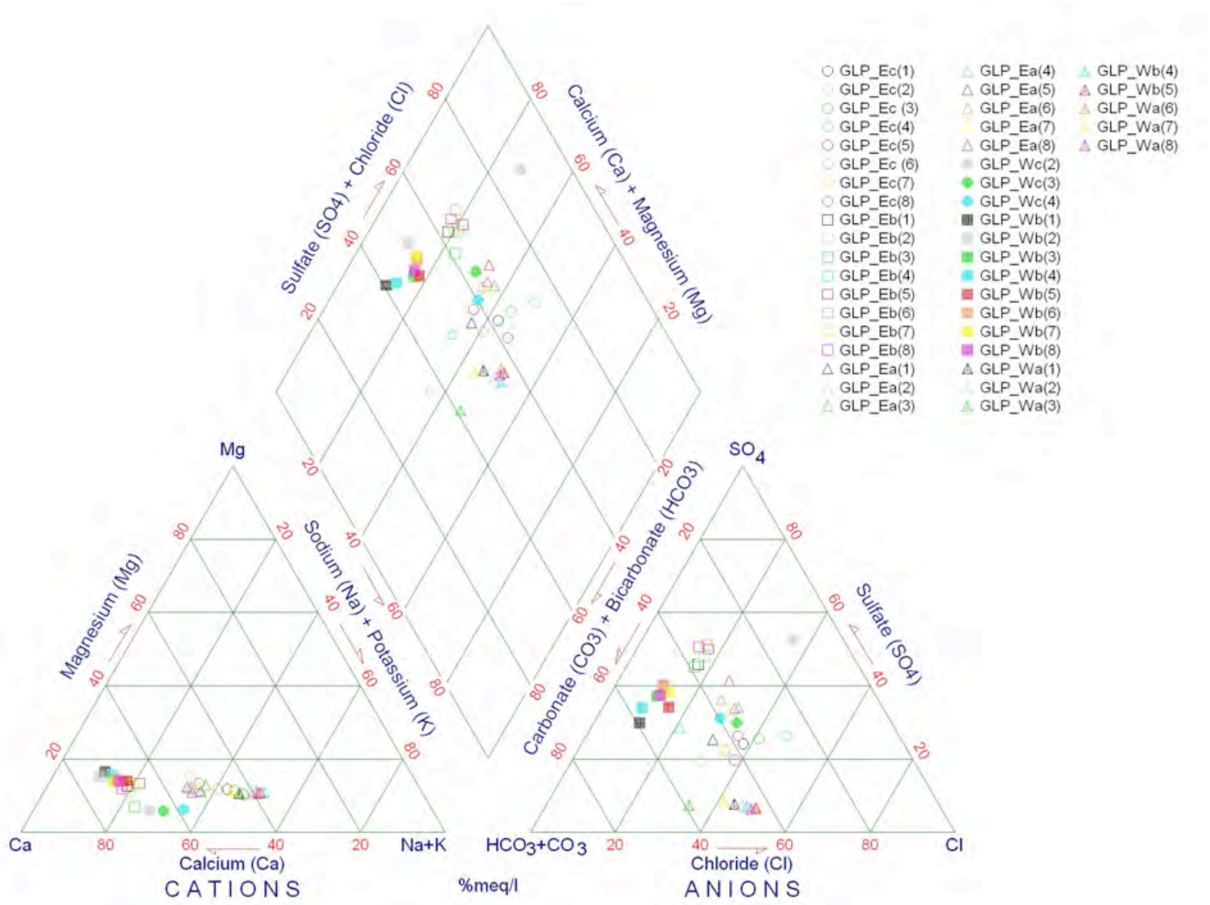


Figure 35 Piper diagram of cations and anions from SGS investigations, June 2009 to June 2010

Calcium

Calcium in groundwater of the Perth region is mostly derived from carbonates, gypsum and rainfall (Yesertener 2010). A calcium and bicarbonate dominated groundwater system is indicative of fresh groundwater within a limestone aquifer (Appelo & Postma 2005). The dominance of calcium in groundwater may be due to the dissolution of calcite. Calcite reacts with carbon dioxide, derived from respiration or the oxidation of organic matter. Carbon dioxide dissolved in water forms carbonic acid and the resulting hydrogen ions associate with carbonate to form bicarbonate (Appelo & Postma 2005), also increasing pH (Freeze & Cherry 1979).

Higher concentrations of calcium were identified flowing in to Lake Gwelup than flowing out of the lake. Higher concentrations of calcium were recorded in intermediate bores and the lowest concentrations of calcium were recorded near the watertable (Figure 36). Calcium ranged between 63 mg/L (GLP_Ec) and 290 mg/L (GLP_Eb), beneath the eastern edge of Lake Gwelup and between 36 mg/L (GLP_Wc) and 160 mg/L (GLP_Wb) beneath the western edge of the lake.

Figure 36 also shows the range in calcium near the base of the Superficial aquifer beneath the eastern edge of the lake (GLP_Ea). It is not clear what is causing this fluctuation, particularly as flow regimes do not appear to change on a seasonal basis at these depths.

Groundwater investigations up-gradient of Lake Gwelup recorded a greater range in calcium concentrations near the watertable (between 5 mg/L at GWA8 and 456 mg/L at MW4). High concentrations of calcium are probably due to acidification processes in groundwater causing calcite dissolution (Appendix D contains the full results).

Calcium concentrations at the eastern edge of Lake Gwelup appear to have increased since 1992. In 1992 calcium in groundwater around the Gwelup bore field ranged between 27 mg/L and 130 mg/L (Davidson 1995).

Surface water sampling in September 2010 (results in Appendix C), recorded similar concentrations of calcium with GLP_Wc, and lower than GLP_Ec. Calcium was recorded at 56 mg/L and 55 mg/L at the eastern (Lake_E) and western (Lake_W) edges of Lake Gwelup respectively. The Brushfield Wetland had calcium at 46 mg/L (Brushfield) (Figure 36).

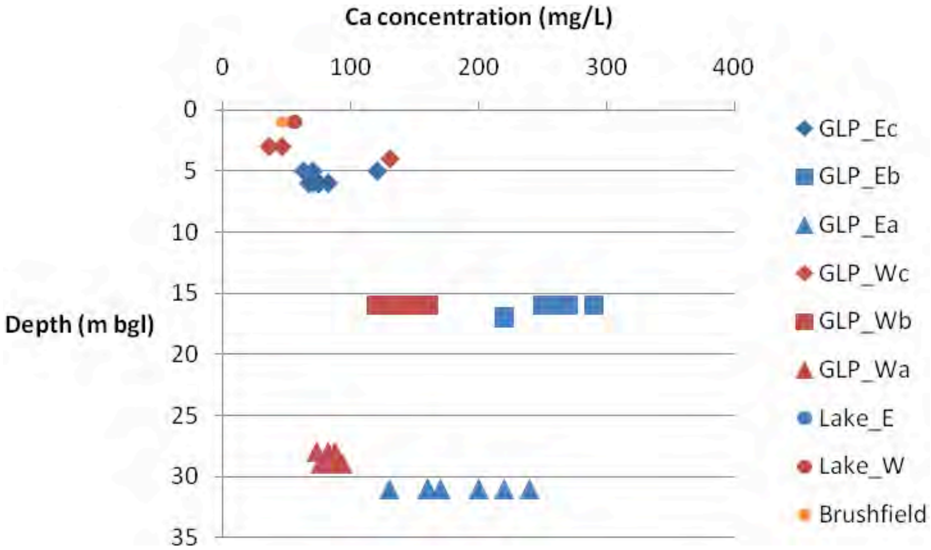


Figure 36 Calcium in groundwater and surface water, June 2009 to September 2010

Sodium

Sodium may be sourced as a result of ion exchange, clays, feldspars, evaporates and from rainfall (Hounslow 1995).

Higher concentrations of sodium were recorded in groundwater flowing into Lake Gwelup than in groundwater flowing out of the lake (Figure 37). Sodium ranged between 40 mg/L (GLP_Ec) and 160 mg/L (GLP_Ec, GLP_Ea) beneath the eastern edge of the lake and between 20 mg/L (GLP_Wc) and 110 mg/L (GLP_Wa) beneath the western edge of the lake. Highest concentrations of sodium were consistently recorded at the base of the Superficial aquifer (GLP_Ea and GLP_Wa).

Sodium concentrations at GLP_Ec quadrupled between July 2009 (40 mg/L) and November 2009 (160 mg/L) and similarly declined between November 2009 and June 2010 (50 mg/L). This may be due to recharge where sodium concentrations rise in accordance with rainfall recharging the groundwater system, or perhaps from leaching and perhaps at times of lower rainfall.

An increasing trend in sodium concentrations beneath the eastern edge of Lake Gwelup is indicated by comparing the results of this investigation with data collected from the Gwelup bore field in 1992. In 1992 sodium ranged between 47 mg/L and 66 mg/L (Davidson 1995). Sodium concentrations were highly variable up-gradient of Lake Gwelup in groundwater near the watertable similar to shallow groundwater at the eastern edge of Lake Gwelup ranging between 12 mg/L (GWA8) and 151 mg/L (GM13) (results Appendix D).

Surface water sampling at Lake Gwelup in September 2010 recorded lower concentrations of sodium than in groundwater beneath the lake. This is probably due to rainfall dilution and drainage inflow. Sodium was recorded at 26 mg/L and 21 mg/L at the eastern (Lake_E) and western (Lake_W) edges of Lake Gwelup respectively in September 2010.

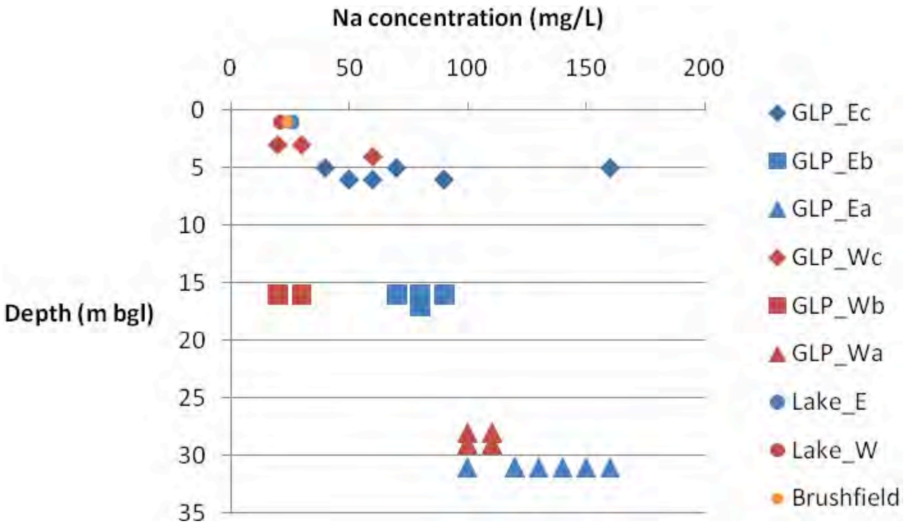


Figure 37 Sodium in groundwater and surface water, June 2009 to September 2010

Magnesium

Magnesium may be sourced from cation exchange with, and/or the weathering of, feldspars, mica and dolomites (Hounslow 1995).

Higher concentrations of magnesium were recorded flowing into Lake Gwelup than flowing out of the lake (Figure 38). Magnesium ranged between 10 mg/L (GLP_Ec) and 33 mg/L (GLP_Eb and GLP_Ea) beneath the eastern edge of Lake Gwelup, and between 2 mg/L (GLP_Wc) and 21 mg/L (GLP_Wb) beneath the western edge of the lake. Groundwater near the watertable (GLP_Ec and GLP_Wc) recorded the lowest concentrations of magnesium (Figure 38). Groundwater near the base of the Superficial aquifer beneath the eastern edge of the lake also reflects trends shown for calcium and sodium, where higher fluctuations in magnesium concentrations were recorded over the monitoring period. Magnesium concentrations at the base of the aquifer beneath the western edge of the lake remained stable (Figure 38).

Groundwater investigations in 1992 around the Gwelup bore field recorded magnesium between 7 mg/L and 16 mg/L (Davidson 1995). These concentrations appear fairly consistent with those recorded in this study near the lake.

Groundwater investigations up-gradient of Lake Gwelup recorded a greater range in magnesium concentrations than in 1992 and in groundwater near the lake in 2009-2010. Magnesium ranged between < 1 mg/L (GWA8) and 54 mg/L (MW1) (results in Appendix D). This may be a result of varied levels of alkalinity and acid generating processes in groundwater up-gradient of Lake Gwelup.

Surface water sampling of Lake Gwelup in September 2010 recorded low levels of magnesium, at 4 mg/L (Lake_W) and 5 mg/L at the western and eastern edges (Lake_E) of the lake (Figure 38).

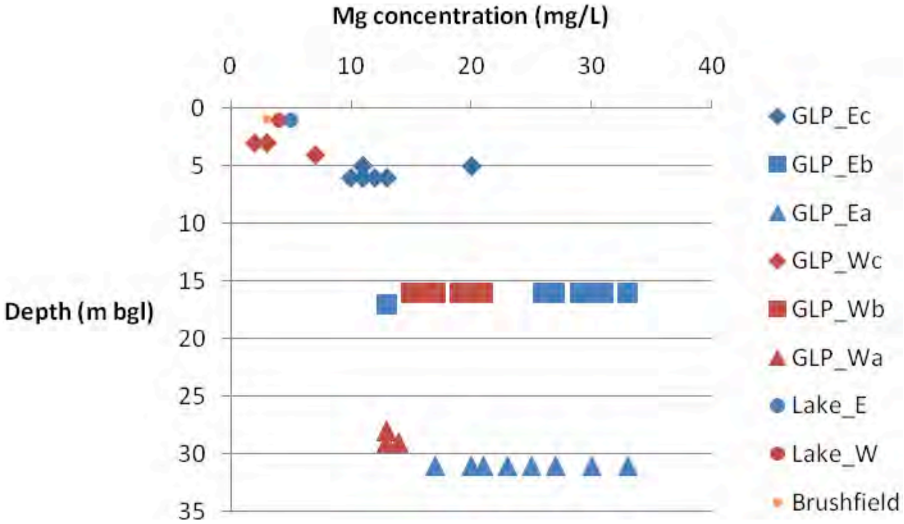


Figure 38 Magnesium in groundwater and surface water, June 2009 to September 2010

Potassium

Potassium may be sourced from feldspars and/or fertilisers (Hounslow 1995). There are many 'sinks' for potassium, including uptake by plants (Hounslow 1995).

Potassium was identified at low concentrations either side of Lake Gwelup. Potassium concentrations ranged between 6 mg/L (GLP_Eb) and 13 mg/L (GLP_Ec) beneath the eastern edge of Lake Gwelup and between < 1 mg/L (GLP_Wc) and 13 mg/L (GLP_Wb) beneath the western edge of the lake. Unlike other base cations measured throughout this monitoring period, concentrations of potassium remained stable at the base of the Superficial aquifer beneath the eastern edge of the lake (GLP_Ea) (Figure 39). The greatest range in potassium concentrations was recorded in the middle of the Superficial aquifer beneath the eastern edge of the lake (GLP_Eb). These potassium concentrations are similar to those identified from the Gwelup bore field in 1992, which were between 3 mg/L and 8 mg/L (Davidson 1995).

Potassium concentrations up-hydraulic gradient of Lake Gwelup range between 2 mg/L (GWA8, GWA10) and 29 mg/L (MW4), which is similar to potassium concentrations identified in groundwater beneath the lake.

Low concentrations of potassium (3 mg/L and 4 mg/L) were recorded in surface water at the eastern (Lake_E) and western (Lake_W) edges of Lake Gwelup respectively in September 2010.

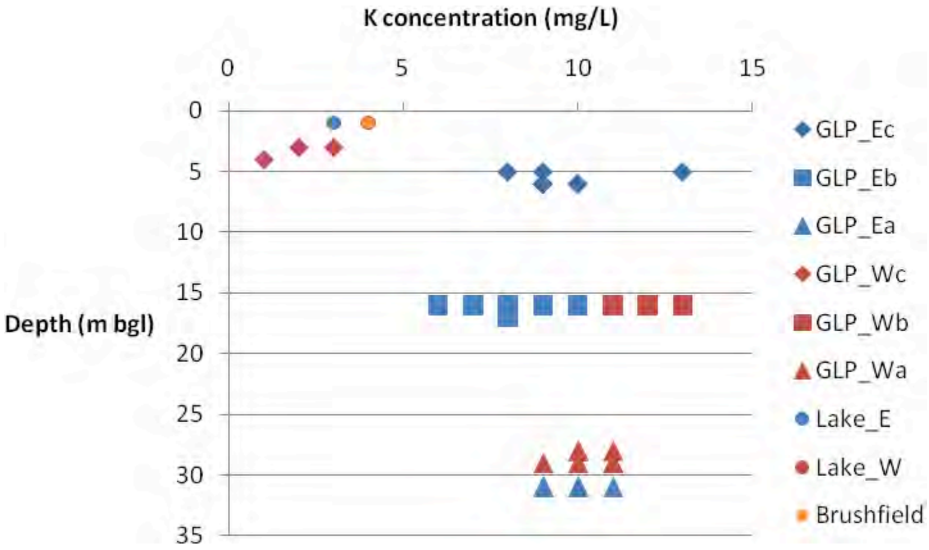


Figure 39 Potassium in groundwater and surface water, June 2009 - September 2010, Lake Gwelup.

Sulfate and chloride

Sulfate concentrations in the Perth region are generally less than 100 mg/L and higher concentrations may be due to sulfide oxidation and/or seawater intrusion (Yesertener 2010). High sulfate concentrations in the Gwelup area may also be due to sulfate based fertiliser input, given the extensive horticultural history of the area.

In 1992 sulfate concentrations in the Superficial aquifer around the Gwelup bore field ranged between 9 mg/L and 225 mg/L (Davidson 1995).

Higher concentrations of sulfate were identified flowing into Lake Gwelup than flowing out of the lake. Sulfate was recorded between 52 mg/L (GLP_Ec) and 480 mg/L (GLP_Eb) beneath the eastern edge of Lake Gwelup and between 29 mg/L (GLP_Wa) and 220 mg/L (GLP_Wc) beneath the western edge of the lake (Figure 40).

Surface water sampling at Lake Gwelup in September 2010 recorded higher levels of sulfate at the eastern edge (up-gradient) of Lake Gwelup (102 mg/L) than in the western edge (down-gradient) of the lake (61 mg/L). Higher sulfate concentrations in surface water at the eastern edge of the lake are probably due to the quality of water discharging into the eastern edge of the lake (near the sampling point) from the Brushfield Wetland. Sulfate concentrations in the Brushfield Wetland were recorded at 82 mg/L in September 2010 (Appendix C). Sulfate concentrations in Lake Gwelup are similar to sulfate concentrations in groundwater near the watertable, higher than in groundwater near the base of the aquifer beneath the western edge of the lake (GLP_Wa) and lower than intermediate bores and GLP_Ea.

All groundwater monitoring bores at Lake Gwelup recorded sulfate to chloride ratios exceeding 0.5, except GLP_Wa (Figure 40). This suggests an additional source of sulfate in groundwater beneath the lake, and that groundwater chemistry and/or groundwater flow regimes differ near GLP_Wa. Sulfate to chloride ratios were highest in intermediate bores, up to 5.0 at GLP_Wb, and up to 2.15 at the base of the Superficial aquifer beneath the eastern edge of the lake (GLP_Ea). This suggests deterioration of the entire vertical extent of the Superficial aquifer east of Lake Gwelup. This presents a long-term management issue for Lake Gwelup.

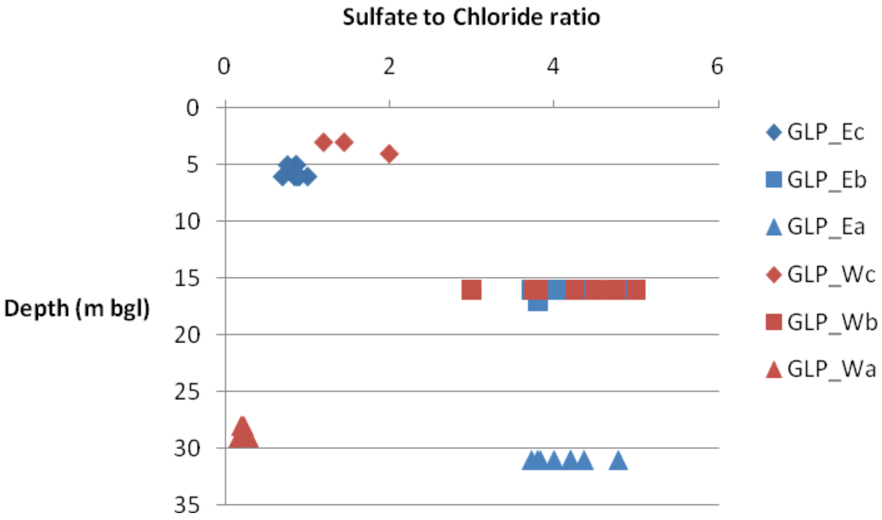


Figure 40 Sulfate to chloride ratio in groundwater, June 2009 - June 2010, Lake Gwelup

Higher sulfate concentrations were measured in the middle of the Superficial aquifer beneath Lake Gwelup. The downward hydraulic gradient induced by recharge from Lake Gwelup on the western edge of the lake may cause increased concentrations of sulfate to accumulate at these depths. The upward hydraulic gradient at the eastern edge of the lake, when full, is unlikely to cause the accumulation of sulfate in the middle of the Superficial aquifer. However, the dominant horizontal flow pattern when the lake is dry, (which has often been the case in the last decade) indicates an up-gradient source of sulfate. Since the lake has been drying out each year since 2001, concentrations of sulfate accumulating near the surface due to the upward hydraulic head on the eastern side of the lake has ceased. High sulfate levels in the intermediate and deep bores, particularly beneath the eastern edge of Lake Gwelup, indicate that there are groundwater quality problems up-gradient of the lake. These elevated sulfate concentrations extend to the base of the Superficial aquifer and probably reflect changes in groundwater chemistry in the Superficial aquifer up-gradient of the lake.

More significant variations in sulfate concentrations were recorded in groundwater near the watertable up-gradient of Lake Gwelup than at the edges of the lake. Sulfate ranged between 3 mg/L (GWA8 and GWA12) and 498 mg/L (MW3) (results in Appendix D). GWA7 has consistently recorded very high sulfate concentrations and acidic groundwater (pH 3.60 in September 2010) over the monitoring period. Sulfate concentrations at GWA7 have previously been recorded at 2,210 mg/L in October 2007, 1,220 mg/L in May 2008 and 821 mg/L in September 2010. The limited frequency of monitoring makes it difficult to provide firm conclusions regarding trends in sulfate concentrations in groundwater over time.

Sulfate concentrations in shallow groundwater up-gradient of Lake Gwelup vary independently of chloride concentrations, as shown by a greater variation in sulfate to chloride ratios. Sulfate to chloride ratios up-gradient of Lake Gwelup were consistently recorded greater than 0.5. GWA7 is an isolated area of acidic groundwater which has recorded sulfate to chloride ratios of 78.09 in October 2007 and 50.0 in May 2008. GWA10 (6.41), GWA13 (6.06), MW1 (10.00), MW2 (28.22), MW3 (9.22), MW4 (19.78) and MW6 (9.70) have also recorded very high sulfate to chloride ratios between 2006 and 2010, with near neutral pH values indicating an additional source of sulfate and degraded groundwater quality in the Lake Gwelup capture zone (results presented in Appendix D).

Salinity

Groundwater salinity beneath Lake Gwelup ranges between fresh and marginal. Lower concentrations of TDS were recorded near the watertable than at greater depths. Higher TDS was also recorded in groundwater flowing into Lake Gwelup than flowing out of the lake (Figure 41). This is also reflected in the Stiff diagram presented in Figure 42. Higher levels of TDS flowing into the lake than out of the lake is counter intuitive for a flow-through system and may be an indication of high volumes of drainage water flowing into the lake.

TDS ranged between 320 mg/L (GLP_Ec) and 1330 mg/L (GLP_Ea) beneath the eastern edge of Lake Gwelup, and between 170 mg/L (GLP_Wc) and 690 mg/L (GLP_Wc) beneath the western edge of the lake. Groundwater investigations in 1992 from around the Gwelup bore field recorded TDS between 281 mg/L and 684 mg/L (Davidson 1995).

TDS beneath the up-gradient (east) edge of Lake Gwelup contrast with information collected in the Superficial aquifer in the Perth region between 2003 and 2005 (Yesertener 2010) and the Gwelup bore field in 1992 (Davidson 1995). Yesertener (2010) suggested the salinity of groundwater as TDS in the Superficial aquifer of the Perth metropolitan area ranged between 75 mg/L and 3,600 mg/L, rarely exceeding 1,000 mg/L. TDS beneath the eastern edge of Lake Gwelup exceeded 1,000 mg/L on 11 occasions from a total of 24 samples. TDS exceeded 1,000 mg/L on 10 occasions out of 16 samples collected at GLP_Eb and GLP_Ea. The upward groundwater flow on the eastern side of Lake Gwelup, when full, may result in higher TDS at shallower depths due to groundwater being discharged towards the surface from greater depths. Regional TDS data may not be suitable for interpretation of water quality when vertical hydraulic gradients associated with flow-through wetlands need to be considered.

A significant fluctuation in TDS was identified at GLP_Eb between January 2010 (520 mg/L) and March 2010 (1,260 mg/L). However, this may be an analytical error (although the electrical balance for both January 2010 and March 2010 results were less than 5%, at −2.3% and 3.3% respectively). The corresponding EC value did not show a similar fluctuation. GLP_Eb recorded EC 1,800 mS/cm in January 2010 and 1790 mS/cm in March 2010. EC generally has a comparable relationship with TDS, as indicated by the following equation by Hounslow (1995):

Total dissolved solids (mg/L) = EC (µS/cm) x 0.66

Lower concentrations of TDS were recorded in surface water at Lake Gwelup than in groundwater. TDS recorded 270 mg/L at the western edge (Lake_W) of Lake Gwelup and 426 mg/L at the eastern edge (Lake_E) of the lake.

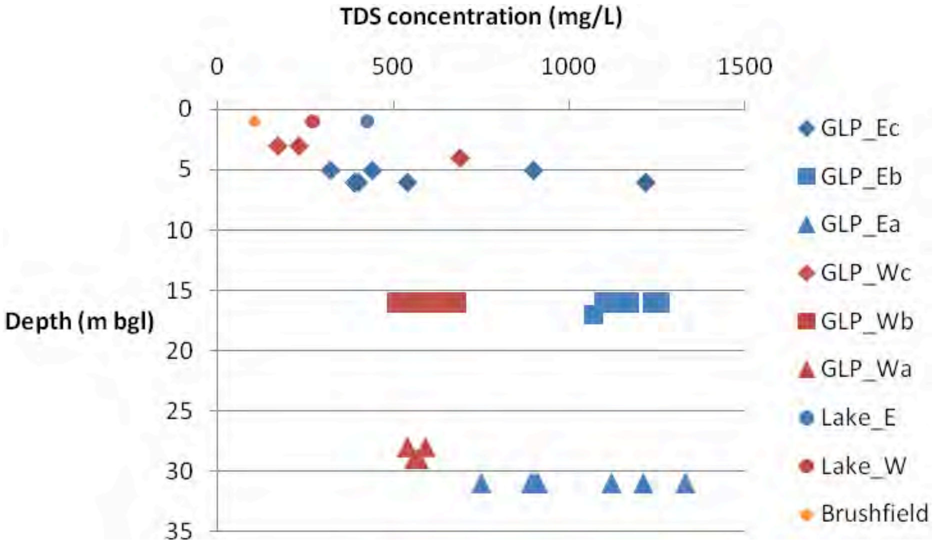


Figure 41 Total dissolved solids (TDS) in groundwater and surface water, June 2009 to September 2010.

The Stiff diagram (Figure 42), like the Piper, shows groundwater at Lake Gwelup to be of calcium bicarbonate type, reflecting its origins in the Tamala Limestone. It also illustrates higher levels of base cations and anions (TDS) flowing into the lake (or towards the lake) than flowing out of the lake. This is different to other groundwater-dependent flow-through wetlands on the Swan Coastal Plain such as Lake Mariginiup which recorded higher levels of TDS flowing out of the lake than flowing into the lake (Searle et al. 2010). This difference probably reflects the influence of drainage at Lake Gwelup compared to other wetlands on the Swan Coastal Plain.

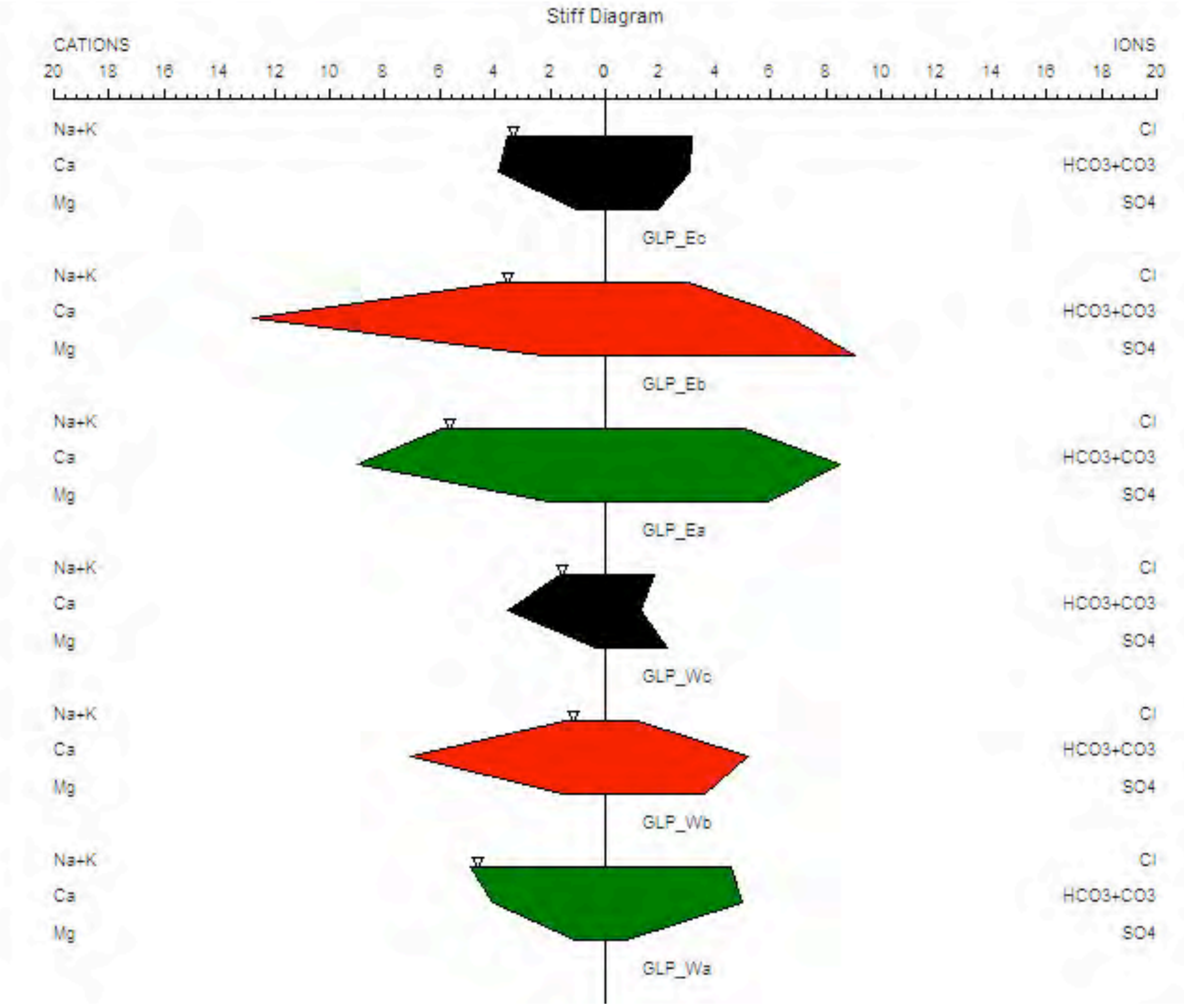


Figure 42 Stiff diagrams representing average concentrations of major ions collected throughout SGS monitoring

Acidity and alkalinity

The pH of groundwater beneath Lake Gwelup generally ranged between 6.5 and 7.5 (Figure 43). However, limited monitoring of GLP_Wc recorded pH between 6.0 and 6.5. Further monitoring at GLP_Wc is required, which probably means the installation of a second shallow bore in order to accommodate high seasonal groundwater level fluctuations. The pH at GLP_Wc was lower than the preferred pH of fresh water ecosystems, which is between 6.5 and 8.5 (DEC 2010). pH values outside this range may affect bioavailable concentrations of heavy metals and affect ecological values and biological diversity (ANZECC & ARMCANZ 2000).

The pH of the system plays a significant role in the speciation of dissolved inorganic carbon into bicarbonate and carbonic acid. At a pH of 6.3, the concentration of bicarbonate and carbonic acid are equal. When pH > 6.3, bicarbonate is the dominant species and when pH < 6.3 carbonic acid is the dominant species (Appelo & Postma 2005). This suggests that when the pH of a system is less than 6.3, which has been recorded at GLP_Wc, the risk of acidification is magnified due to the change in the speciation of dissolved inorganic carbon. Further monitoring of water quality should also be conducted in relation to ecological values. This report also highlights a lack of ecological data for Lake Gwelup to better quantify risks of ecological decline.

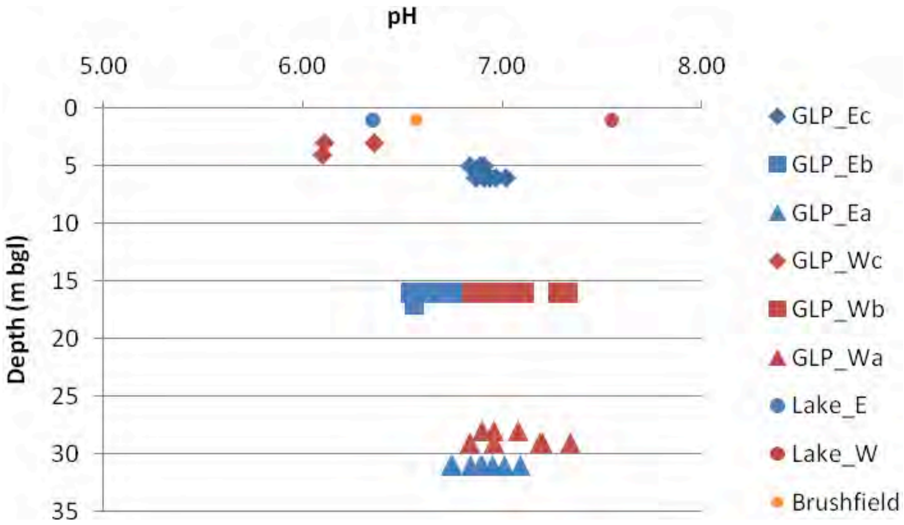


Figure 43 pH in groundwater and surface water, June 2009 - September 2010 at Lake Gwelup.

Groundwater investigations up-gradient of Lake Gwelup generally recorded pH between 6.5 and 7.0, however isolated areas of acidification were recorded. pH at GWA7 ranged between 2.66 and 3.60 between 2006 and 2010. pH at GWA1, GWA3 and GWA5 were only been measured once (March 2006), recording pH 5.99, 5.37 and 4.20 respectively. This suggests the risk of acidification is high, given extended periods of sulfide oxidation and subsequent acid input, particularly if the buffering capacity is limited.

Alkalinity is primarily present in the form bicarbonate, with a very small carbonate fraction at Lake Gwelup. Alkalinity speciation is dependent on pH. Bicarbonate is the predominant species for pH less than 10.3, and where pH is greater than 10.3 carbonate becomes the dominant species (Appelo & Postma 2005).

Higher levels of total alkalinity (as mg/L $CaCO_3$) were recorded flowing into Lake Gwelup than flowing out of the lake. This may reflect dilution by rainfall runoff into the lake. Drains are a major source of inflow to the lake. The oxidation of sulfide minerals may also be acting to deplete the available acid buffering capacity as evidenced from elevated sulfate to chloride ratios (> 0.5). Total alkalinity beneath the eastern edge of Lake Gwelup ranged between 140 mg/L (GLP_Ec) and 350 mg/L (GLP_Eb and GLP_Ea), with the exception of two results at GLP_Ea, recording 850 mg/L (24/7/2009) and 680 mg/L (9/11/2009). Total alkalinity beneath the western edge of Lake Gwelup ranged between 52 mg/L (GLP_Wc) and 370 mg/L (GLP_Wa).

Alkalinity and acidity varied at the base of the Superficial aquifer beneath the eastern edge of the lake (GLP_Ea) throughout SGS monitoring. This reflects similar variations identified for calcium, sodium, magnesium and sulfate. The reason for these variations is not clear, particularly given the base of the aquifer does not interact with the lake. The fluctuation in alkalinity corresponded with similar variations in acidity on one occasion when acidity increased from 26 mg/L in September 2009 to 170 mg/L in November 2009 and subsequently declined to 69 mg/L in January 2010.

Higher levels of alkalinity than acidity were recorded in surface water at Lake Gwelup in September 2010. Alkalinity at the eastern edge of Lake Gwelup (Lake_E) was 79 mg/L as $CaCO_3$ compared to 92 mg/L as $CaCO_3$ at the western edge of the lake (Lake_W) (Figure 44). Acidity was 28 mg/L and 5 mg/L as $CaCO_3$ at the eastern and western edges of the lake respectively (Figure 45). More significantly, a difference in pH of 1.2 units was recorded across the lake with pH 6.35 at the eastern edge of the lake and pH 7.55 at the western edge of the lake.

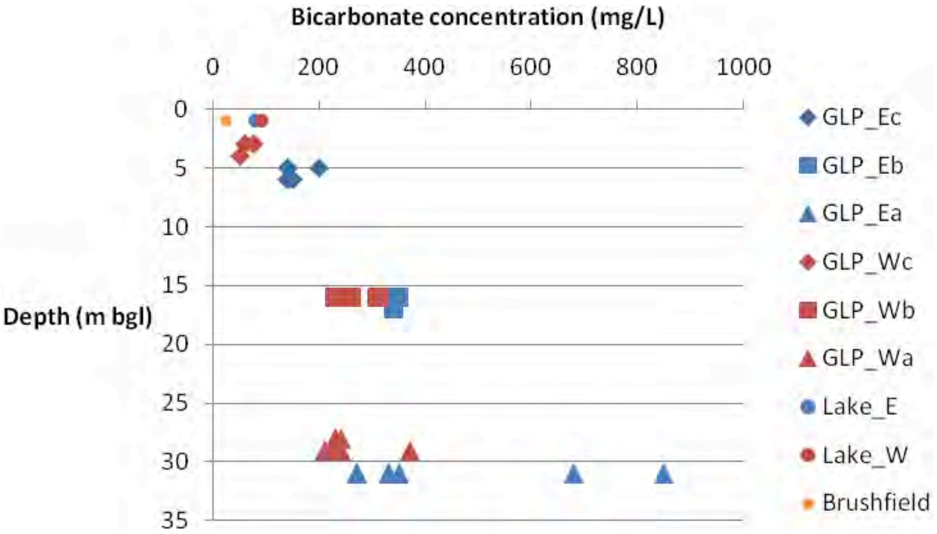


Figure 44 Bicarbonate concentrations in groundwater and surface water, June 2009 to September 2010, Lake Gwelup

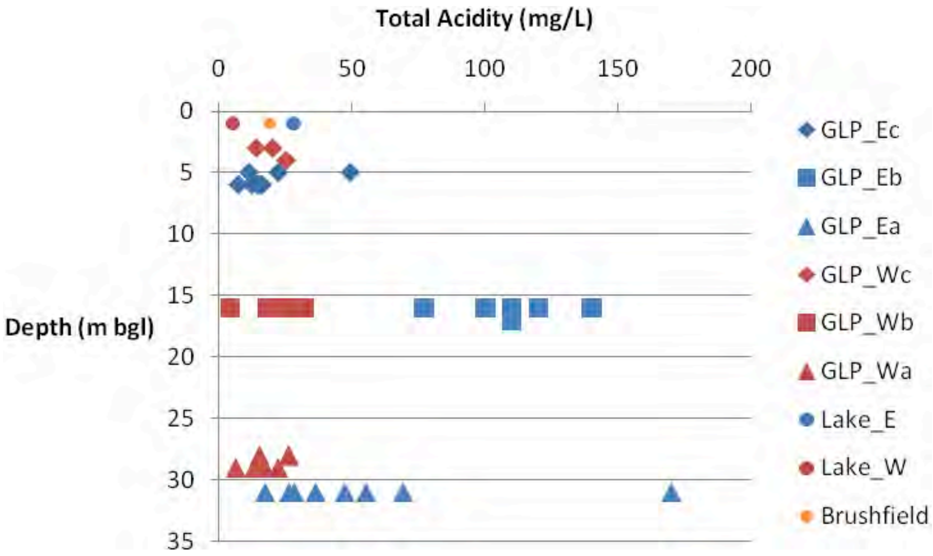


Figure 45 Total acidity in groundwater and surface water June 2009 - September 2010, Lake Gwelup.

Alkalinity was lower near the watertable at Lake Gwelup than at greater depths of the Superficial aquifer. Fluctuating groundwater levels act to oxidise exposed sediment depleting the available acid buffering capacity. Sediment investigations identified the presence of AASS and PASS at the western edge of Lake Gwelup (GLP_Wc), but not from GLP_Ec. Groundwater monitoring identified elevated sulfate to chloride ratios (> 0.5) in groundwater near the watertable at both GLP_Ec and GLP_Wc, indicating an additional source of sulfate, perhaps from sulfide oxidation. Organic acidification and nitrification are also other processes that may contribute to lower levels of alkalinity in groundwater near the watertable.

The Swedish Environmental Protection Agency (Swedish EPA 2002) provides guidance on total alkalinity values and the relationship with pH. For pH > 6.0 with alkalinity > 60 mg/L as $CaCO_3$ is expected to be sufficient to maintain stable and acceptable pH. All groundwater bores with the exception of GLP_Wc, GWA7, GWA8 and MW2 recorded total alkalinity levels greater than 100 mg/L. While elevated levels of alkalinity reduce the risk of acidification, the oxidation of pyrite may still release metals such as iron into the groundwater system.

If total alkalinity levels decline to less than 60 mg/L with pH less than 7.5, there is unlikely to be sufficient buffering capacity to maintain stable pH in areas vulnerable to acidification (Swedish EPA 2002). Limited monitoring of GLP_Wc recorded pH between 6.1 and 6.5 and alkalinity levels between 52 mg/L (July 2009) and 77 mg/L (September 2009).

Isolated areas of acidic groundwater were recorded near the watertable up-gradient of Lake Gwelup (some results presented in Table 13; all results detailed in Appendix D).

Table 13 Acidity and alkalinity of selected bores up-gradient of Lake Gwelup (results in full presented in Appendix D).

Sampling date	**Bore ID**	**pH**	**Total acidity mg/L as $CaCO_3$**	**Total alkalinity mg/L as $CaCO_3$**
10/3/2006	GWA1	5.99	35	NA
10/3/2006	GWA3	5.37	25	NA
10/3/2006	GWA5	4.2	620	NA
5/10/2007	GWA7	2.66	1260	< 1
2/10/2007	GWA8	6.84	< 1	1
8/5/2008	GWA7	3.09	700	< 1
7/5/2008	GWA8	6.71	7	34
13/1/2009	MW1	6.62	300	496
13/1/2009	MW2	5.21	504	44
13/1/2009	MW3	6.17	208	305
13/1/2009	MW4	6.67	567	538
13/1/2009	MW5	6.16	102	145
31/8/2010	GWA7	3.6	446	< 1
1/9/2010	GWA8	6.26	19	10
1/9/2010	MW3	6.24	274	126

Note: NA denotes 'not assessed'.

Redox conditions (Eh)

The oxidation reduction potential is measured in the field and is a qualitative assessment of redox conditions (Yesertener 2009). The oxidation reduction potential provides an important line of evidence in inferring chemical processes in an aquifer such as pyrite oxidation, nitrification and denitrification. To some extent it acts to regulate natural concentrations of oxygen, iron, sulfate, hydrogen sulfide and methane in groundwater (Appelo & Postma 2005). Furthermore, the presence of dissolved organic carbon (organic matter) typically acts as a reductant, while oxygen and nitrate act as oxidants (Appelo & Postma 2005).

Investigations conducted either side of Lake Gwelup recorded positive oxidation reduction potentials in groundwater near the watertable (up to +111 mV at GLP_Wc) indicating an oxidising environment. Negative oxidation reduction potentials were recorded in groundwater in the middle of the Superficial aquifer (down to −164 mV at GLP_Wb) and the base of the aquifer (down to −133 mV) indicating reducing conditions capable of achieving sulfate reduction.

Groundwater investigations up-hydraulic gradient of Lake Gwelup near the watertable commonly found positive oxidation reduction potentials in groundwater indicating an oxidising environment. Oxidation reduction potentials were highest at GWA7, up to +333 mV in 2007 (Appendix D).

Metals

Aluminium

Dissolved aluminium ranged between < 0.005 mg/L (GLP_Ec, GLP_Ea) and 1.2 mg/L (GLP_Ec) beneath the eastern edge of Lake Gwelup and between < 0.005 mg/L (GLP_Wa) and 1.4 mg/L (GLP_Wb) beneath the western edge of the lake (Figure 46).

Three samples detected aluminium in groundwater greater than 1 mg/L beneath Lake Gwelup. All three samples were collected in May 2010 at GLP_Ec (Figure 46). These results exceed aquatic fresh water guidelines of 0.055 mg/L (95% species protection level, ANZECC & ARMCANZ 2000). GLP_Wc was dry in May 2010 and not sampled. Dissolved aluminium in groundwater beneath Lake Gwelup was below irrigation guidelines of 5 mg/L and below the Department of Health (DoH) (2006) domestic non-potable groundwater use guideline of 2 mg/L (DEC 2010).

Total aluminium in groundwater significantly exceeded corresponding dissolved aluminium on two occasions. This may be due to the presence of suspended clays (colloids). In July 2009, GLP_Ea recorded total aluminium at 20 mg/L and dissolved aluminium at 0.025 mg/L. In November 2009 at GLP_Ea, total aluminium was 8.5 mg/L and dissolved aluminium was < 0.005 mg/L. These are significant differences, particularly at the base of the Superficial aquifer. The low dissolved oxygen readings (July 2009 5.2% or 0.5 mg/L, November 2009 5.3% or 0.49 mg/L) suggest sample methodology in the field did not caused disturbance and subsequent release of metals. High iron concentrations were also recorded.

Surface water sampling at Lake Gwelup in September 2010 did not detect the presence of dissolved aluminium (< 0.01 mg/L). Total aluminium was recorded at 0.06 mg/L and 0.01 mg/L at the western and eastern edges of the lake respectively. These values compare with other wetlands throughout the south-west of Western Australia (Kilminster & Cartwright 2011).

High concentrations of aluminium were recorded in the capture zone of Lake Gwelup exceeding irrigation and DoH (2006) domestic non-potable groundwater use guidelines (DEC 2010). High concentrations of aluminium were recorded in conjunction with acidic conditions. GWA7 recorded dissolved aluminium at 44.7 mg/L (October 2007) and 30.5 mg/L (May 2008). However, in September 2010, despite low pH of 3.60, aluminium levels declined to 1.49 mg/L which may have been due to precipitation of aluminium hydroxysulfate minerals in the aquifer (removing dissolved aluminium and sulfate). Significant changes in aluminium concentrations were recorded despite consistently low and acidic pH. This suggests the behaviour of aluminium is not determined solely by acidification and can only be determined by further investigation into the role organic matter and precipitation of aluminium hydroxysulfate minerals. This is also supported by the presence of elevated concentrations of aluminium in conjunction with near neutral pH values up-gradient of Lake Gwelup.

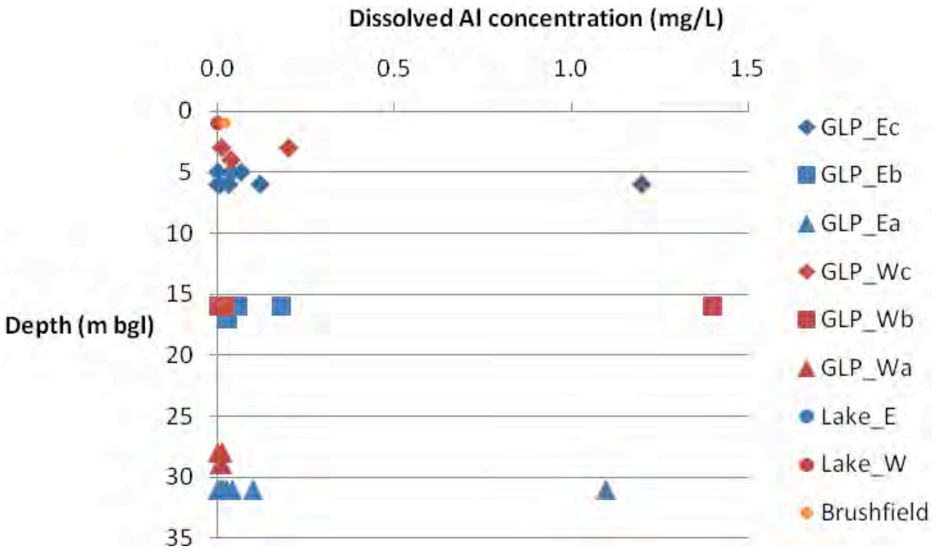


Figure 46 Dissolved aluminium in groundwater and surface water, June 2009 to September 2010, Lake Gwelup

Elevated concentrations of aluminium were detected at near neutral pH values in groundwater beneath the residential development, Gwelup Park. Dissolved aluminium was detected at GWA9 at 6.28 mg/L (pH 6.99) in September 2010 and at MW2 at 1.08 mg/L (pH 5.21) in January 2009. This suggests a second mechanism in addition to acidity, such as organic carbon, is triggering the release of aluminium into groundwater in the capture zone of Lake Gwelup. Other investigations have also identified aluminium mobilisation to be independent of pH, and rather than the dissolution of aluminium containing minerals, may be due to cation exchange (Adams et al. 1990; Astrom & Bjorklund 1995; Kilminster & Cartwright 2011).

Arsenic

Arsenic ranged between < 0.001 mg/L and 0.015 mg/L in groundwater beneath Lake Gwelup. The 'drinking water guide' (0.007 mg/L) was slightly exceeded on 13 occasions (Figure 47). As untreated groundwater is not used for drinking purposes in this area, there is no pathway posing a risk and therefore the assessment level is not relevant. The DoH (2006) domestic non-potable groundwater use guideline of 0.07 mg/L and irrigation guideline of 0.1 mg/L are more relevant as a human health risk criteria given that domestic groundwater bores are not used for drinking water. These guidelines were not exceeded. Fresh water guideline values are As(III) 0.013 mg/L and As (V) 0.024 mg/L (ANZECC & ARMCANZ 2000). These values were exceeded on two occasions at GLP_Eb, and this may have ecological implications if groundwater at these depths contributes to lake recharge.

Surface water sampling at Lake Gwelup in September 2010 detected low concentrations of arsenic at 0.002 mg/L and 0.003 mg/L at the eastern (Lake_E) and western (Lake_W) edges of the lake respectively (Figure 47).

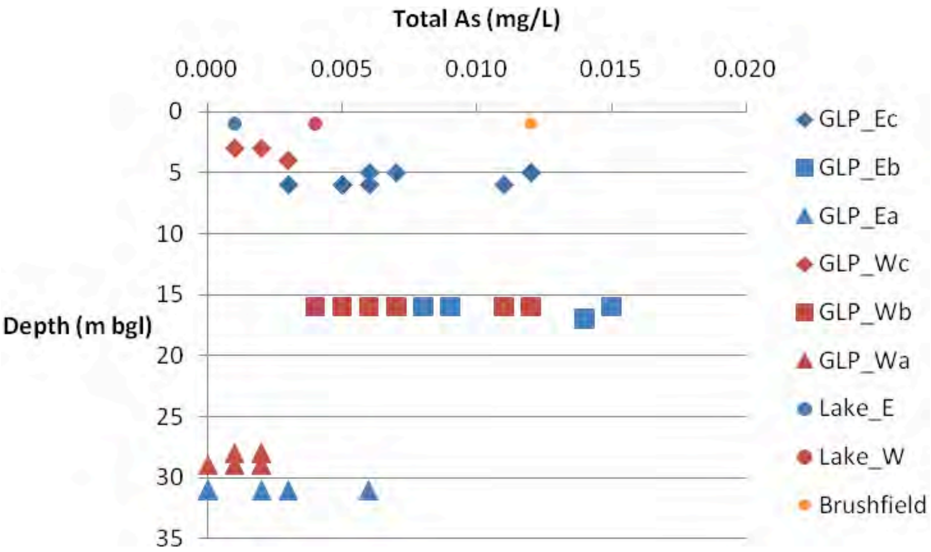


Figure 47 Total arsenic in groundwater and surface water, June 2009 - September 2010, Lake Gwelup

High levels of arsenic were recorded in groundwater in the capture zone of Lake Gwelup, beneath the Gwelup Park residential estate (Figure 53). Arsenic ranged between < 0.001 mg/L (several locations) and 3.59 mg/L (MW2) (all results in Appendix D). Groundwater monitoring between 2006 and 2010 recorded arsenic at GWA2 (0.290 mg/L), GWA6 (0.290 mg/L), GWA7 (1.89 mg/L), MW4 (0.206 mg/L) and MW5 (0.377 mg/L). These concentrations exceed the irrigation guideline (0.1 mg/L), DoH (2006) non-potable groundwater use guideline (0.07 mg/L) and fresh water aquatic ecosystem guidelines (ANZECC & ARMCANZ 2000). As these monitoring locations are in the lake's capture zone, the movement of arsenic in shallow groundwater over time should be monitored to determine risks to ecological values as well as risks posed to human health.

The Gwelup Park residential development is classified 'contaminated restricted use' under the *Contaminated Sites Act 2003*. The use of groundwater is restricted and not recommended for use without suitable analytical testing to determine if the quality of water is suitable for its intended use.

Iron

Groundwater flowing into Lake Gwelup contains higher concentrations of iron than groundwater flowing out of the lake. Dissolved iron ranged between < 0.005 mg/L (GLP_Ec) and 25 mg/L (GLP_Eb) beneath the eastern edge of the lake and between 0.046 mg/L (GLP_Wc) and 2.4 mg/L (GLP_Wa) beneath the western edge of the lake (figures 48 and 49).

High concentrations of iron were recorded in the middle of the Superficial aquifer beneath the eastern edge of Lake Gwelup (GLP_Eb) (Figure 48). Dissolved iron at GLP_Eb ranged between 16 mg/L and 25 mg/L, where the oxidation reduction potential ranged between −74 mV and −108 mV. Lower concentrations of iron (although still elevated) were measured at GLP_Ea, between 1.0 mg/L and 4.6 mg/L, where redox conditions were similarly reducing, recorded between −58 mV and

−119 mV. These reducing conditions and elevated levels of iron suggest conditions indicative of reductive iron dissolution. The reductive dissolution of arsenic containing iron oxyhydroxide minerals has previously been linked to the release of arsenic in the lower half of the aquifer in Gwelup (Appleyard et al. 2006).

High iron concentrations are a regional issue in the Gwelup area and are suspected to have been exacerbated by oxidation processes caused by intensive groundwater abstraction and residential development. In 1992 dissolved iron in groundwater east of Lake Gwelup, in the Gwelup bore field ranged between 1.18 mg/L and 12.80 mg/L (Davidson 1995). In a regional groundwater quality study of the Perth region, iron ranged between 0.05 mg/L and 19 mg/L in the upper half of the Superficial aquifer (Yesertener 2010).

Table 14 presents the highest concentrations of dissolved iron recorded in groundwater near the watertable in bores up-gradient of Lake Gwelup (Results in full Appendix D). Dissolved iron ranged between < 0.05 mg/L at several locations and 342 mg/L (GWA7; October 2007) between 2006 and 2010.

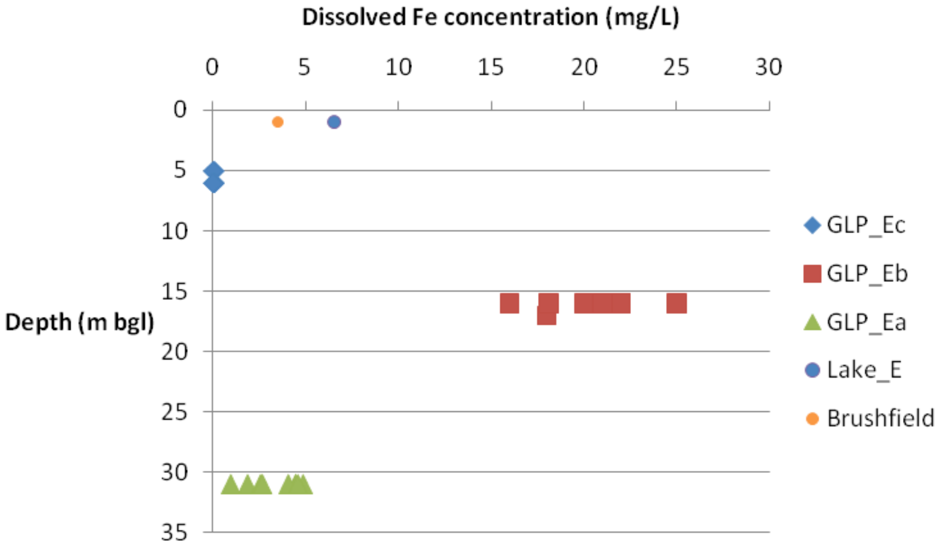


Figure 48 Dissolved iron in groundwater and surface water western edge of lake

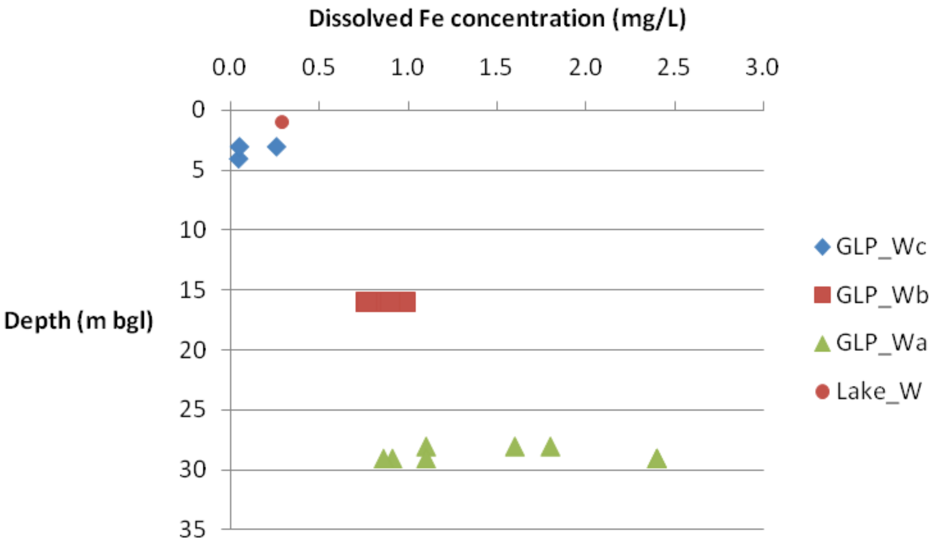


Figure 49 Dissolved iron in groundwater and surface water, eastern edge of lake

Table 14 Concentrations of dissolved iron at selected bores up-gradient of Lake Gwelup (results in full presented in Appendix D)

Sampling date	Bore ID	Dissolved iron mg/L
10/3/2006	GWA3	15
10/3/2006	GWA5	330
10/3/2006	GWA6	58
5/10/2007	GWA7	342
8/5/2008	GWA7	234
13/1/2009	MW2	128
13/1/2009	MW3	114
13/1/2009	MW4	9
13/1/2009	MW6	12
31/8/2010	GWA7	102
1/9/2010	MW3	124
1/9/2010	MW6	17

The Delawney Drain, Brushfield Wetland and Brushfield Drain are highly degraded, containing elevated levels of iron (figures 8 and 9). Ferrous iron (Fe^{2+}) is released at the eastern edge of Lake Gwelup and in the Brushfield Wetland, precipitating out as iron oxyhydroxide minerals such as ferrihydrite (Appelo & Postma 2005). The Brushfield Wetland discharges into the eastern edge of Lake Gwelup. A surface water sample collected from the Brushfield Wetland in September 2010, recorded dissolved iron at 3.53 mg/L. The surface water sample collected at the eastern edge of Lake Gwelup recorded dissolved iron at 6.53 mg/L (Lake_E) (Figure 48), and contrasts with 0.29 mg/L of dissolved iron detected at the western edge of the lake (Lake_W) (Figure 49).

The accumulation of monosulfidic black ooze (MBO; monosulfidic material) presents a management issue for the quality of water in Lake Gwelup and the Brushfield Wetland. Monosulfidic material forms in a reduced anaerobic environment with an adequate source of organic carbon, sulfate and iron. Sulfate reducing bacteria produces sulfide (and alkalinity) during the oxidation of organic matter which reacts with dissolved ferrous iron (Fe^{2+}) forming insoluble iron monosulfides such as mackinawite (Smith & Melville 2004). These can later transform to pyrite. Formation of monosulfidic materials consume acidity and metals, however these can be re-oxidised if anaerobic conditions cannot be maintained. This is common in areas with declining groundwater levels, or in the sediments of ephemeral wetlands. Oxidation of monosulfides (and pyrite) can release ferric iron (Fe^{3+}), other metals and sometimes acidity if alkalinity has been leached (Smith & Melville 2004). Ferric iron (Fe^{3+}) is only stable at low pH < 5.0, however ferric iron oxyhydroxide minerals may be present in surface water in pH > 5.0 (Smith & Melville 2004).

Manganese

Manganese was generally recorded as total concentrations around Lake Gwelup. However, sampling in September 2010 recorded dissolved manganese levels comparable with total manganese.

Higher concentrations of manganese were recorded in groundwater flowing into Lake Gwelup than in groundwater flowing out of the lake (Figure 50). Total manganese ranged between 0.021 mg/L (GLP_Ec) and 0.49 mg/L (GLP_Eb) beneath the eastern edge of Lake Gwelup and between 0.002 mg/L (GLP_Ec) and 0.067 mg/L (GLP_Ec) beneath the western edge of the lake.

GLP_Eb and GLP_Ea recorded manganese an order of magnitude greater than corresponding bores on the western edge of Lake Gwelup. GLP_Eb recorded the highest concentrations of manganese between 0.35 mg/L and 0.49 mg/L compared to GLP_Wb between 0.017 mg/L to 0.038 mg/L (Figure 50).

Groundwater investigations up-gradient of Lake Gwelup showed that manganese concentrations exceeded the irrigation guideline of 0.2 mg/L (DEC 2010). Elevated levels of manganese were recorded at GWA5 (0.68 mg/L), GWA6 (1.1 mg/L), GWA7 (0.886 mg/L), GWA11 (0.665 mg/L), GWA13 (0.269 mg/L), MW3 (0.238 mg/L), MW4 (0.234 mg/L) (Appendix D).

Surface water sampling in September 2010, recorded manganese at 0.094 mg/L at the eastern edge (Lake_E) of Lake Gwelup and at 0.006 mg/L at the western edge (Lake_W) of the lake (Figure 50).

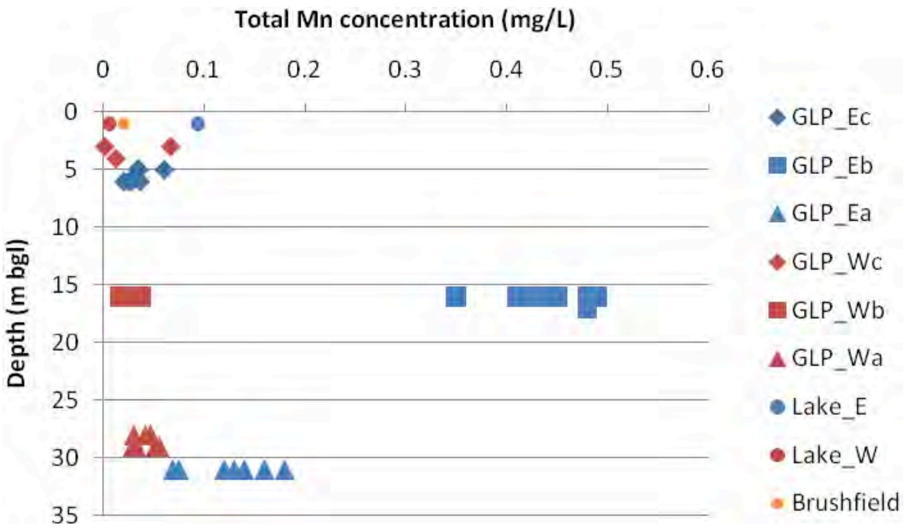


Figure 50 Total manganese in groundwater and surface water, June 2009 - September 2010, Lake Gwelup

Other metals

Groundwater investigations around Lake Gwelup and up-hydraulic gradient of Lake Gwelup also included assessments of boron, cadmium, chromium, mercury, nickel, selenium and zinc, which were all either recorded at very low concentrations or if present, were below laboratory detection limits.

Results of surface water sampling at Lake Gwelup and the Brushfield Wetland are presented in Appendix C and groundwater results up-hydraulic gradient of Lake Gwelup are in Appendix D.

Nutrients

Nitrogen compounds

The assessment of nitrogen considered:

- total nitrogen – organic nitrogen plus ammonia/ium and NO_X
- total oxidised nitrogen (TON) – a measure of nitrite (NO_2) and nitrate (NO_3) otherwise referred as NO_X
- ammoniacal-N – a collective measurement of ammonia (NH_3) and ammonium (NH_4).
- total Kjeldahl nitrogen (TKN) – total nitrogen minus NO_X.

Higher levels of nitrogen were recorded in groundwater flowing out of Lake Gwelup than flowing into the lake (Figures 51 and 52). This differs to other trends noted for cations, sulfate and alkalinity.

Total nitrogen in groundwater varied between 0.49 mg/L (GLP_Wc) and 11.0 mg/L (GLP_Wc) beneath the western edge of Lake Gwelup and between 0.45 mg/L (GLP_Ec) and 3.8 mg/L (GLP_Ec) beneath the eastern edge of the lake (Figure 51). Total nitrogen fluctuated near the watertable and was more stable at greater depths in the Superficial aquifer. This may be due to the influence of rainfall, recharge and dilution at certain times of the year.

Surface water sampling of Lake Gwelup in September 2010 recorded total nitrogen between 0.6 mg/L and 0.9 mg/L on the western (Lake_W) and eastern (Lake_E) edges of the lake respectively. The composition of nitrogen is dominated by TKN with negligible organic nitrogen and NO_X. The composition of nitrogen in groundwater near the watertable is different to the composition of nitrogen in surface water at Lake Gwelup, but concentrations of total nitrogen are similar. The Brushfield Wetland recorded total nitrogen at 0.6 mg/L. Groundwater discharging from the surface water body through the lake bed sediments may liberate sediment bound nitrogen and consequently down-gradient groundwater concentrations are elevated.

Sediments at the base of Lake Gwelup may have provided a sink for nutrients, particularly given the long horticultural history of the area. The drying of lake sediments may liberate previously sequestered nutrients, given the lake has been drying each year since 2001.

The highest concentrations of total nitrogen beneath Lake Gwelup were recorded in GLP_Wb, ranging between 5.1 mg/L and 6.7 mg/L. These levels exceed the fresh water aquatic guideline of 2.0 mg/L (ANZECC & ARMCANZ 2000). However, there are no immediate ecosystems down-gradient of this location that are likely to be affected. Nitrogen at GLP_Wb is dominated by ammoniacal-N, ranging between 4.7 mg/L and 6.5 mg/L. The fresh water guideline was not exceeded in either of the deep bores beneath Lake Gwelup, or GLP_Eb. However, GLP_Ec exceeded fresh water aquatic ecosystem guidelines on two occasions in June 2009 and November 2009. There is an increased risk of adverse biological effects where fresh water guidelines are exceeded and some ecosystem specific investigation may be required (ANZECC & ARMCANZ 2000).

The process of nitrification may be inferred in shallow groundwater, however elevated nitrate levels may also be a function of leaching from the unsaturated zone. The composition of nitrogen near the watertable is dominated by organic nitrogen, nitrite and/or nitrate (NO_X). Positive redox potentials (Eh) indicate an oxidising groundwater environment with low concentrations of ammonium.

In deeper groundwater nitrogen typically remains in a reduced form (Davidson 1995). Nitrogen transformations are dominated by microbial mineralisation of organic nitrogen (generating ammoniacal–N) and denitrification. Nitrate and nitrite (NO_X) were not detected (below limits of detection) at the base of the aquifer and at very low levels in intermediate bores. The composition of nitrogen is dominated by ammonia/ammonium.

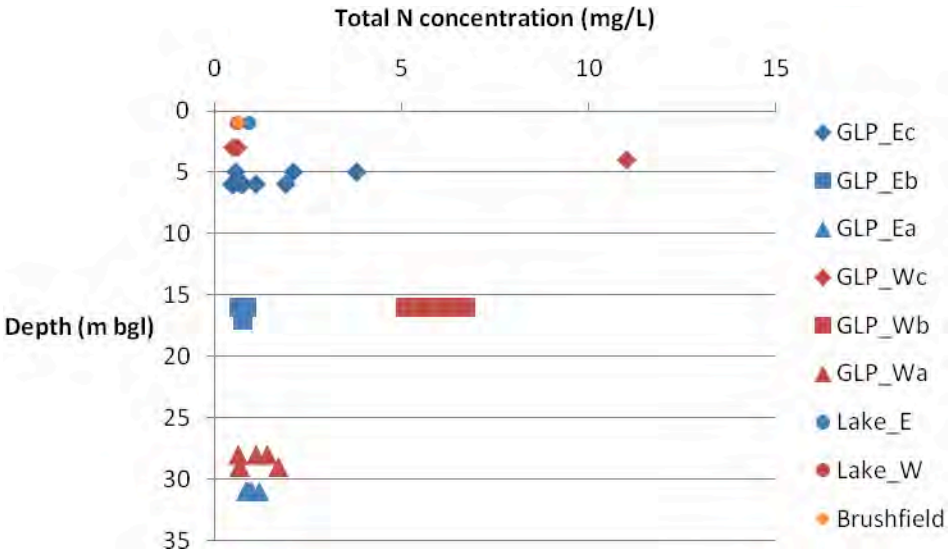


Figure 51 Total nitrogen in groundwater and surface water, June 2009 – September 2010, Lake Gwelup

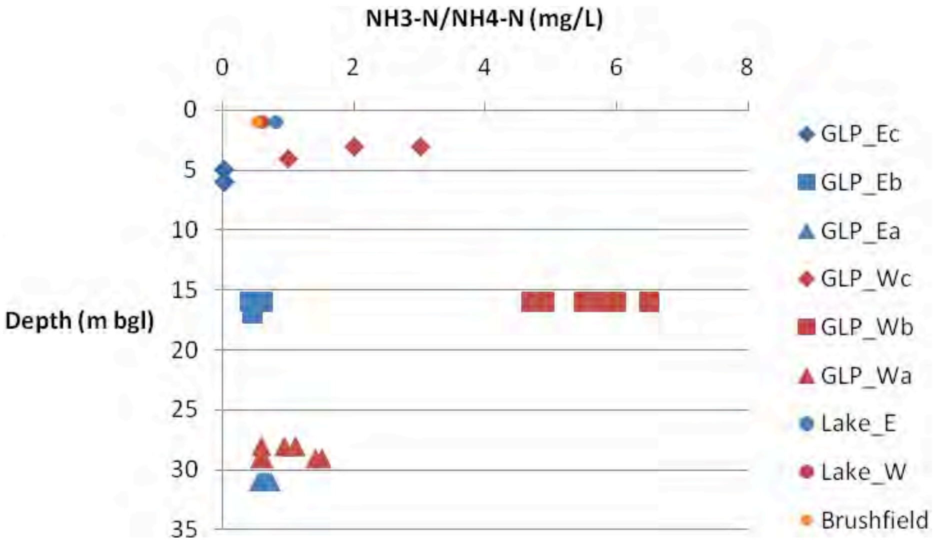


Figure 52 Ammonia/ammonium in groundwater and surface water, June 2009 – September 2010

Groundwater monitoring around the Gwelup bore field in the late 1980s and early 1990s indicated denitrification in the lower half of the Superficial aquifer. This was inferred from low nitrate concentrations in conjunction with elevated and/or rising levels of ammonia and total nitrogen (Barber et al. 1993). Production bores are screened in the lower half of the Superficial aquifer, therefore these findings represent aquifer conditions at these depths, consistent with the findings of this study.

Groundwater monitoring around the Gwelup bore field in 1992 measured nitrate between < 0.01 mg/L and 6.7 mg/L (Davidson 1995). NO_X was recorded up to 11.0 mg/L at GLP_Wc in July 2009, however limited interpretation is possible because of the limited frequency of monitoring. Subsequently, levels of NO_X were recorded at 0.013 mg/L in September and November 2009.

The composition of total nitrogen was dominated by nitrate in oxidising groundwater near the watertable up-gradient of Lake Gwelup. This may suggest nitrification in the capture zone of Lake Gwelup.

Table 15 shows the results of groundwater investigations for nitrogen and phosphorus in September 2010, up-gradient of Lake Gwelup (full list of results in Appendix D).

Table 15 Nutrients in groundwater for monitoring undertaken up-gradient of Lake Gwelup in September 2010

Sampling date	**Bore ID**	**Total N mg/L**	**TKN mg/L**	**NOx mg/L**	**Total P mg/L**
31/08/2010	GWA7	22.0	21.2	0.79	0.72
1/09/2010	GWA8	0.5	0.4	0.06	0.20
31/08/2010	GWA9	2.2	0.7	1.53	0.24
31/08/2010	GWA10	12.5	1.2	11.3	<0.01
2/09/2010	GWA11	20.7	1.2	19.5	<0.10
2/09/2010	GWA12	1.2	0.4	0.77	0.35
31/08/2010	GM13	1.1	0.5	0.61	<0.01
1/09/2010	GWA13	1.2	0.2	1.02	<0.10
1/09/2010	MW3	3.0	3.0	0.04	<0.01
1/09/2010	MW6	2.0	2.0	0.02	0.01

The composition of nitrogen at GWA7 (22.0 mg/L) is probably dominated by ammonia/ium given NO_X comprised only 0.79 mg/L (Table 15). GWA7 recorded an oxidation reduction potential of +263 mV (DO 4.07 mg/L). Further investigation into the chemical processes in groundwater at this location may be required as nitrogen present should be oxidised and be in organic form.

Groundwater monitoring at GWA7 between 2006 and 2010 previously recorded high ammonia concentrations. Ammonia concentrations up to 41 mg/L at GWA7 were recorded in October 2007. This exceeds the DoH (2006) domestic non-potable groundwater use guideline (5.0 mg/L).

Phosphorus

Total phosphorus levels were generally higher in groundwater flowing out of Lake Gwelup than flowing into the lake. This reflects the trend identified for total nitrogen and reinforces the likelihood that the lake is a source of nutrients to groundwater. The trigger value for south-west Australian wetlands is 0.06 mg/L (ANZECC & ARMCANZ 2000). This trigger is exceeded numerous times for groundwater at Lake Gwelup.

Total phosphorus varied between 0.027 mg/L (GLP_Ec) and 0.66 mg/L (GLP_Ea) beneath the eastern edge of Lake Gwelup (Lake_E) and between 0.036 mg/L (GLP_Wc) and 0.39 mg/L (GLP_Wb) beneath the western edge of the lake (Lake_W).

Pesticides

No pesticides or herbicides were detected in this study. There is no evidence to suggest pesticide or herbicide use associated with historical horticultural activities has affected the quality of groundwater beneath Lake Gwelup.

6 Discussion

6.1 Surface water–groundwater interactions at Lake Gwelup

This SGS study confirmed Lake Gwelup to be a flow-through wetland in hydraulic connection with up to half the depth of the Superficial aquifer when the lake is full (Figure 54). Groundwater flow patterns beneath the lake change depending on water levels in Lake Gwelup. This was highlighted at GLP_Wc, where groundwater levels declined by at least 2.0 m between December 2009 and February 2010 as lake levels declined, probably due to the presence of caverns in the Tamala Limestone. The extent and magnitude of these caverns is not known and may provide a basis for further investigation. The extent of the cavern influences the amount of water drained from the western edge of the lake as the lake dries, and the rate at which it drains. These conditions also appear to be a primary cause for groundwater levels falling below ecological water requirements (EWRs) recorded for *Melaleuca rhaphiophylla* and *Eucalyptus rudis* in Section 2.6 (Ecology). However groundwater levels at the eastern edge of the lake (GLP_Ec) remained above the minimum EWRs throughout the investigation.

Changing groundwater flow regimes also acts to change the distribution of chemical parameters in groundwater of the Superficial aquifer. Regional groundwater level decline has caused Lake Gwelup to dry each year since 2001 reducing the level of connectivity with the Superficial aquifer. This contrasts with the level of connectivity prior to 2001 where the lake was recorded as dry on only three occasions between 1921 and 2001.

Groundwater quality at Lake Gwelup is impacted by the quality of groundwater flowing into the lake from the capture zone, inflows from surrounding drains and the oxidation of lake sediments, when the lake is dry (figures 53 and 54). Poor groundwater quality was detected beneath the Gwelup Park residential estate, about 400 m up-gradient of Lake Gwelup, migrating towards the lake. Iron and sulfate enriched water is being discharged into Lake Gwelup from the Delawney Drain and Brushfield Wetland. The quality of groundwater and surface water inflow has affected the quality of water at Lake Gwelup. The quality of groundwater and surface water inflow to Lake Gwelup may have an adverse effect on ecological and social amenity values at this wetland and the surrounding Lake Gwelup Reserve.

Increased drying of the lake appears to be causing a deterioration in the quality of groundwater discharging from the lake. The increased drying frequency of lake sediments is expected to accelerate their oxidation causing a change in the chemical composition of groundwater down-hydraulic gradient of Lake Gwelup, and of the surface water in the lake. Given the increased hydraulic disconnection, the effects of sediment oxidation on water quality will be more limited to shallower depths. Lower concentrations of base cations (Ca, Na, Mg, K), lower alkalinity (reduced buffering capacity) and lower pH was measured in groundwater beneath the western edge of the lake, particularly near the watertable (Figure 54). These groundwater quality conditions may also be due to dilution from drain inflow in addition to the effects of increased oxidation of lake sediments (figures 53 and 54).

6.2 Influence of acid sulfate soils on water quality

Acid sulfate soil (ASS) was identified at the western edge of the lake and up-gradient of Lake Gwelup. The quality of groundwater shows signs of change due to the oxidation of acid sulfate soils (figures 53 and 54). This has been attributed to high groundwater level fluctuation at the western edge of the lake (GLP_Wc), with evidence of actual acid sulfate soil and elevated sulfate to chloride ratios in groundwater (> 0.5).

The acid generating potential of the Superficial formation (Tamala Limestone) is demonstrated by evidence of sulfide oxidation to the base of the Superficial aquifer beneath the eastern edge of the lake. This may have originated from up-gradient of the lake. High sulfate to chloride ratios, between 1.1 and 2.1, were recorded in groundwater at the base of the Superficial aquifer beneath the eastern edge of the lake (GLP_Ea), with sulfate concentrations between 180 mg/L and 430 mg/L. These sulfate concentrations suggest a widespread groundwater quality issue that this study has not delineated. Sulfate and calcium concentrations and sulfate to chloride ratios beneath the eastern edge of Lake Gwelup were generally greater than previously measured around the Gwelup bore field in 1992. This suggests an increase in sulfate and calcium in the lower half of the Superficial aquifer. These increased concentrations beneath the eastern edge of Lake Gwelup may be a result of groundwater quality degradation up-gradient of the lake, affected by leaching from historical horticultural activities and oxidation and dissolution processes from groundwater abstraction for public supply.

Sulfate to chloride ratios at the base of the Superficial aquifer beneath the western edge of the lake (GLP_Wa) significantly contrast with groundwater at the base of the Superficial aquifer beneath the eastern edge of the lake (GLP_Ea). GLP_Wa recorded sulfate to chloride ratios less than 0.5 indicating no additional source of sulfate or sulfide oxidation (Figure 54). This was the only monitoring point to not indicate an additional source of sulfate. The sulfate to chloride ratio at GLP_Wa may be more indicative of 'background' groundwater quality conditions as the sulfate to chloride ratio of rainfall in the area is about 0.3 (Bawden 1991).

The presence of ASS has affected the quality of groundwater in the capture zone of Lake Gwelup. Sulfide oxidation has contributed to high concentrations of sulfate (sulfate to chloride ratio > 0.5), iron, aluminium, arsenic, manganese and acidity in groundwater near the watertable up-gradient of Lake Gwelup. This poses a risk to the long-term ecological integrity of the lake in addition to the quality of water abstracted from domestic groundwater bores in the immediate area. The presence of actual acid sulfate soil (AASS; $pH_F < 4.0$) appears to be linked to dewatering and soil excavation associated with residential development triggering the oxidation of otherwise stable iron sulfide minerals such as pyrite, producing sulfuric acid.

Despite near neutral pH for most groundwater monitoring locations up-gradient of Lake Gwelup, there is evidence of acidification causing increased calcium concentrations in groundwater. This is due to carbonate dissolution, where acid solutions liberate calcium from naturally present carbonates in groundwater, characteristic of the Tamala Limestone. Calcium was recorded up to 456 mg/L and acidity up to 567 mg/L up-gradient of Lake Gwelup. These conditions are derived from sulfide (pyrite) oxidation given very high sulfate to chloride ratios up to 28.22 and dissolved iron up to 100 mg/L. These groundwater conditions suggest the need for continued monitoring to assess the quality of groundwater flowing towards Lake Gwelup, and the likely consequence of pyrite oxidation in this area for the future integrity of the lake ecosystem.

The accumulation of iron and sulfate at the eastern edge of Lake Gwelup in conjunction with the presence of abundant organic matter (mainly *Typha orientalis*) in waterlogged conditions provide a good habitat for the accumulation of monosulfidic material (monosulfidic black ooze or MBO). This process may become more limited due to the increased drying of the lake, but may also pose a risk of periodic acidification pulses from oxidation events. MBO forms under reducing conditions where sulfate reducing bacteria use sulfate instead of oxygen for respiration and convert sulfate to sulfide producing alkalinity (bicarbonate) as a bi-product. The dissolved sulfide reacts with metals, especially iron, to form iron sulfide minerals such as pyrite (a disulfide – FeS_2) and mackinawite (a monosulfide – FeS) (EPHC & NRMMC 2011). These processes increase the store of acidity while generating soluble alkalinity and can trap dissolved metals such as cadmium, copper, zinc, lead and iron. The accumulation of monosulfidic material may compound the risk to the environment if lake sediments oxidise and release sequestered metals. This highlights a further need to investigate the composition of lake sediments at Lake Gwelup as well as sediments of the adjoining Brushfield Wetland, which also show evidence of MBO accumulation.

6.3 Dissolved metals in Gwelup groundwater

Elevated levels of aluminium were recorded in groundwater up-gradient of Lake Gwelup and may pose a future risk to the ecological integrity of the lake (figures 53 and 54). The mobility of aluminium is largely dependent on pH and dissolved organic carbon. Interestingly, dissolved aluminium at GWA7 decreased from 44.7 mg/L in 2006 to 1.49 mg/L in 2010, however the most recent data (June 2011) shows dissolved aluminium to have again increased to 25.3 mg/L. This increase in aluminium also corresponded with a decline in pH. pH at GWA7 has been acidic since monitoring began in 2006. pH at GWA7 increased between 2006 and 2010 from 2.6 to 3.6, however in June 2011, pH at GWA7 was recorded at 2.48. The nature of these results reinforces the need for consistent monitoring over an extended period of time in order to gain a more adequate understanding of trends in groundwater quality.

The solubility of aluminium is significantly increased when pH < 4.5 (Appelo & Postma 2005). Aluminium may also remain soluble in near neutral pH conditions, where the solubility of aluminium is controlled by complexation of solid organic matter (Jansen et al. 2004). The toxicity of aluminium is reduced at or near neutral pH and high water hardness (high calcium and magnesium concentrations) (ANZECC & ARMCANZ 2000). Aluminium at near neutral pH is not likely to be toxic to aquatic organisms because it is probably not monomeric (Al^{3+}) (ANZECC & ARMCANZ 2000). Elevated dissolved aluminium concentrations were identified in groundwater at MW2 (1.08 mg/L) and GWA9 (6.28 mg/L) with pH 5.21 and 6.99 respectively. Aluminium may be regulated by adsorption of solid phase organic matter, and previous studies show that iron preferentially adsorbs to organic matter over aluminium. This may account for elevated levels of dissolved aluminium in iron rich environments, reflecting a greater fluctuation in iron concentrations and more stable aluminium concentrations (Jansen et al. 2004). In a study by Jansen et al. (2004), a lower metal to carbon ratio correlated with a higher percentage of mobile dissolved organo-aluminium and iron complexes (Jansen et al. 2004). Further groundwater monitoring should include dissolved organic carbon as it is likely to be a significant factor in the mechanisms associated with aluminium solubility and eco-toxicity.

The release of arsenic into the Superficial aquifer near the watertable in some cases is due to the oxidation of arsenopyrite, producing arsenate (an As(V) species), ferric iron (Fe^{3+}), sulfate and acidity (H^+) (Appleyard et al. 2006). This process is likely to have caused the release of arsenic at GWA7 (pH 3.60), which has consistently recorded elevated concentrations of dissolved arsenic up to 1.89 mg/L.

High concentrations of arsenic were also recorded in groundwater with pH between 5.0 and 7.0 near the watertable up-gradient of Lake Gwelup (beneath the Gwelup Park residential estate; Figure 53). This was recorded at MW2 (3.59 mg As/L), MW4 (0.206 mg As/L) and MW5 (0.377 mg As/L). Comparatively high concentrations of arsenic have not been shown to have migrated into Lake Gwelup and arsenic does not currently pose a risk to the ecology of the ecosystem. MW2 recorded the highest concentration of dissolved arsenic throughout the investigation period (2006–10). The mechanism for arsenic release near the watertable is likely due to sulfide oxidation, given high sulfate to chloride ratios at these locations (between 2.2 at MW5 and 28.2 at MW2). The oxidation reduction potentials did not provide conclusive evidence of aquifer conditions. Measurements ranged from +82 mV at MW2 (suggesting oxidising environment) to −37 mV at MW4 (suggesting reducing conditions). High concentrations of alkalinity in conjunction with a possible reducing environment at MW4 may indicate the reductive dissolution of iron oxyhydroxide minerals at MW4. High dissolved iron concentrations were also recorded at the three monitoring points (MW2, MW4, MW5) ranging from 5.43 mg/L at MW5 to 100 mg/L at MW2. Further investigation should include dissolved organic carbon to provide more clarity in the possible mechanisms for arsenic release.

Arsenic has previously been identified in groundwater at greater depths of the Superficial aquifer in the Gwelup area in a reducing environment. Previous investigations have discussed the release of arsenic through the reduction of arseniferous iron oxyhydroxide minerals (FeOOH) such as goethite and ferrihydrite, Reductive dissolution of iron minerals by organic matter (represented as CH_2O) can be described by the following reaction:

$CH_2O + 4FeOOH + 7H^+ \rightarrow 4Fe^{2+} + HCO_3^- + 6H_2O$ (Appelo & Postma 2005)

This process has been suggested as a possible cause for the release of small concentrations of arsenic (5 µg/L to 15 µg/L) into groundwater in the lower half of the Superficial aquifer around the Gwelup bore field, which is dominated by a reducing environment (negative redox potentials) (Appleyard et al. 2006). Similar comparisons may be drawn with arsenic levels identified in the middle of the Superficial aquifer as part of the SGS investigation beneath the eastern edge of Lake Gwelup (GLP_Eb). GLP_Eb is characterised by a reducing environment and recorded arsenic up to 0.015 mg/L (15 µg/L) and consistently high concentrations of dissolved iron up to 25 mg/L (Figure 54).

Reductive iron dissolution may also explain high iron and sometimes high alkalinity in other bores at Lake Gwelup and up-gradient of the lake. Microbial decomposition of organic matter utilising available dissolved oxygen and nitrate can lead to reductive dissolution of iron oxyhydroxide minerals (Nickson et al. 2000). This process releases iron and bicarbonate, which are characteristic of the groundwater system at Gwelup and may explain the very high levels of alkalinity, up to 538 mg/L at MW4. Reducing conditions were identified near the watertable at MW3 and MW6 in September 2010, which also recorded elevated levels of arsenic (0.05 mg/L and 0.022 mg/L respectively) and alkalinity at 126 mg/L and 165 mg/L as $CaCO_3$ respectively.

Detailed investigations regarding the release of arsenic into the groundwater system in Bangladesh and West Bengal indicate the release of arsenic is due to the reductive dissolution of iron oxyhydroxide minerals (Appelo & Postma 2005). Investigations in Bangladesh and West Bengal in 1997 identified arsenic up to 0.332 mg/L with pH between 5.5 and 7.5, high concentrations of iron (up to 29.2 mg/L) and high levels of bicarbonate alkalinity (HCO_3^-) up to 697 mg/L) (Nickson et al. 2000). A significant difference between the quality of groundwater in Bangladesh and West Bengal compared with Gwelup is the high levels of sulfate in Gwelup. This indicates the liberation of arsenic in groundwater near the watertable in Gwelup may also be due to oxidation of arsenian pyrite.

6.4 Nutrients

The composition of total nitrogen is dominated by nitrate in groundwater near the watertable near Lake Gwelup and in its capture zone. This may be due to the leaching of nitrate through the unsaturated zone. It may also be due to nitrification, where ammonia is oxidised to nitrate (Figure 54). Nitrogen in groundwater at Gwelup is likely to be a legacy issue from historical horticultural activities and unsewered residential development. Nitrification is an acidification process releasing H^+ acting to reduce the available acid buffering capacity and reducing groundwater pH. The following reaction applies for the nitrification process:

$NH_4 + 2O_2 \rightarrow NO_3 + 2H^+ + H_2O$ (Appelo & Postma 2005)

In soils, this often results in acidification through organic mineralisation (above the watertable) with leaching of nitrate to groundwater. Ammonification and denitrification are likely to dominate nitrogen chemistry in the lower half of the Superficial aquifer beneath Lake Gwelup (Figure 54). These processes are stable under reducing redox conditions dominated by iron and sulfate reduction.

Nitrification and pyrite oxidation are the main factors influencing the quality of groundwater near Lake Gwelup and up-gradient of the lake. The deterioration of groundwater quality in the capture zone of Lake Gwelup represents a long-term management issue. This study suggests groundwater quality issues up-gradient of Lake Gwelup are currently affecting the quality of groundwater beneath the lake.

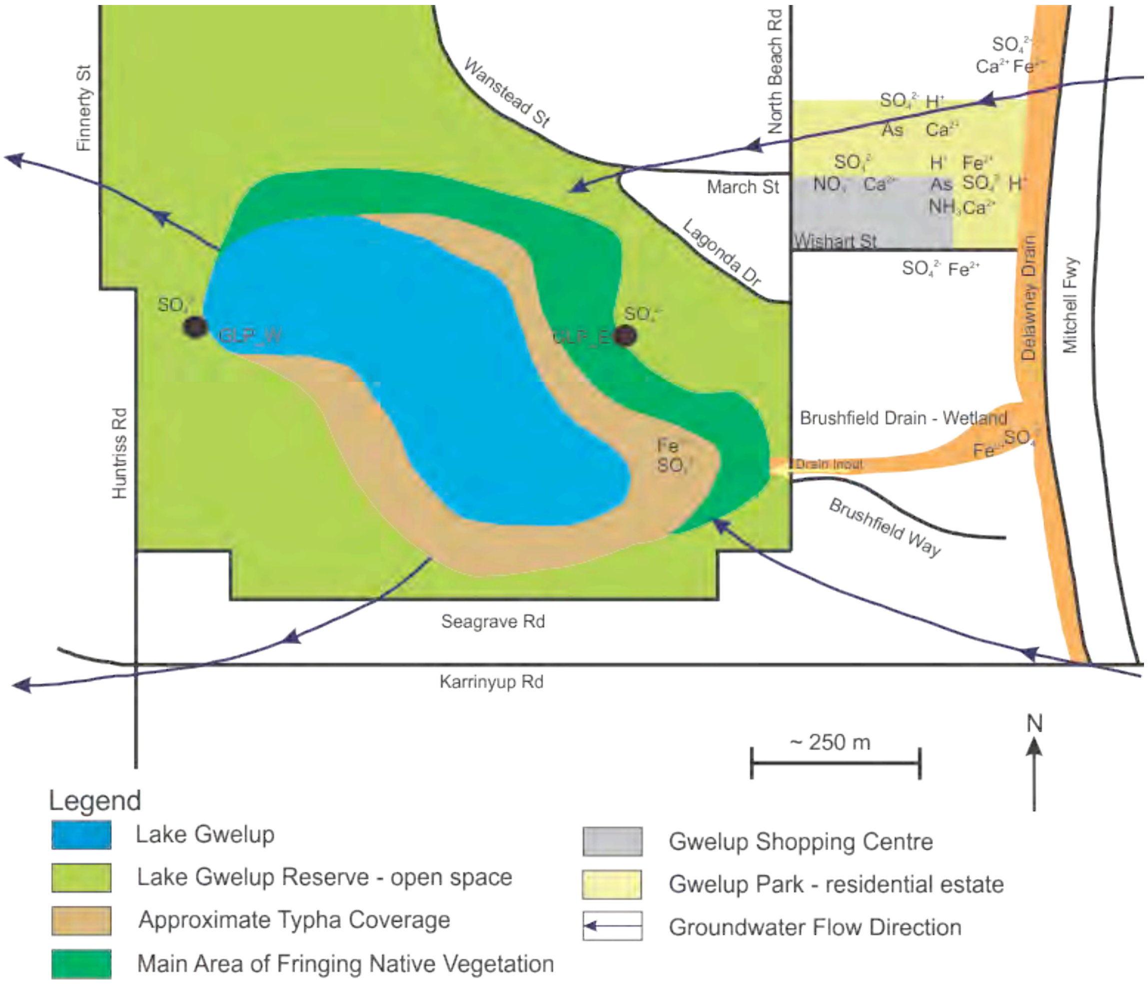


Figure 53 Conceptual diagram of relationships between hydrogeology, groundwater chemistry, drainage input and vegetation around Lake Gwelup

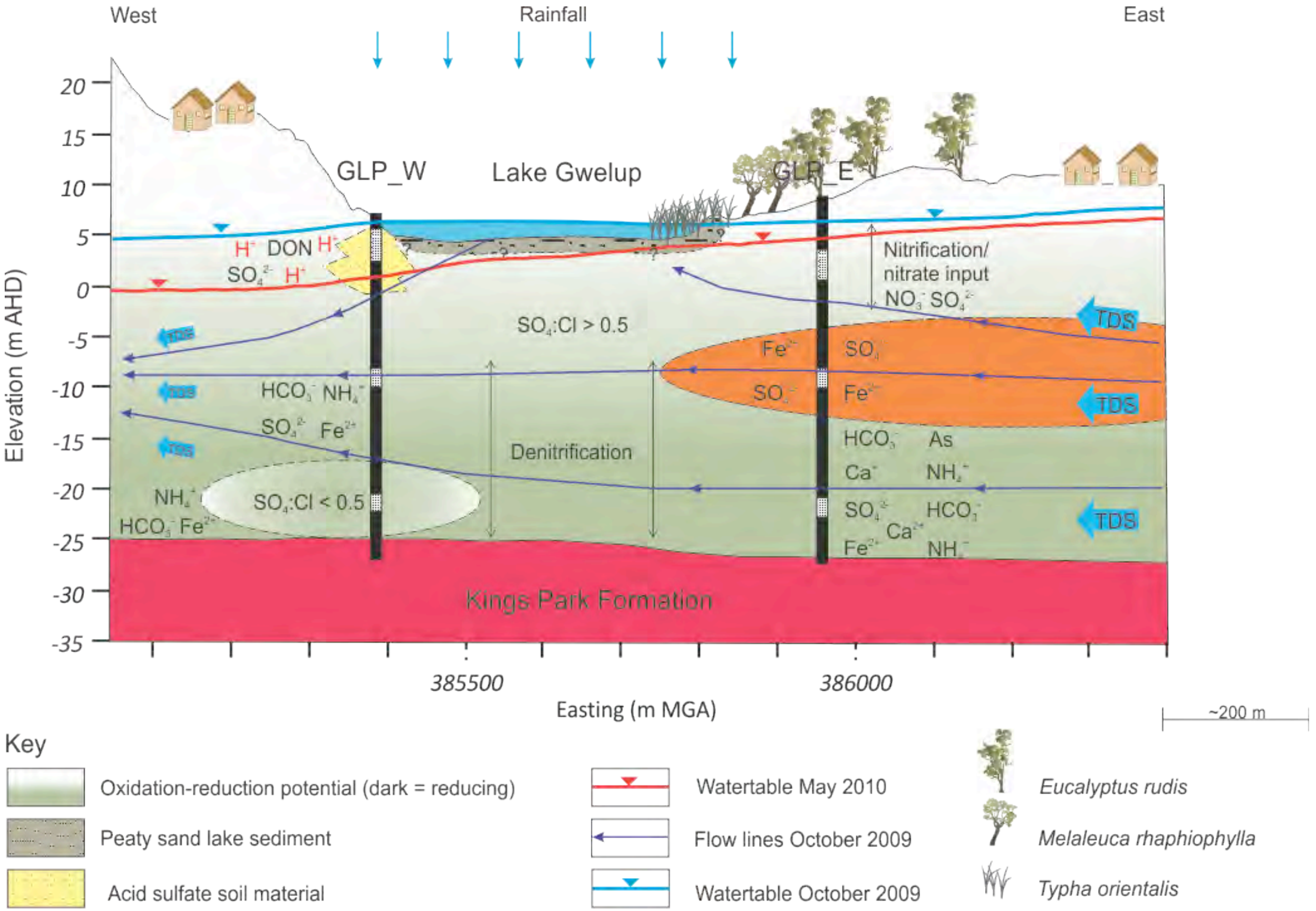


Figure 54 Conceptual diagram of relationships between hydrogeology, groundwater chemistry and ecosystem function around Lake Gwelup

6.5 Key findings

This study showed Lake Gwelup to be a flow-through groundwater-dependent wetland. Groundwater level decline has resulted in an increased disconnection between surface water and groundwater. This study showed that Lake Gwelup may interact with up to half the vertical extent of the Superficial aquifer when full. The increased drying frequency and lower connection between surface water and groundwater probably reflects an increased significance of surface water drainage inflow to the lake.

The quality of groundwater beneath Lake Gwelup and within its capture zone is mainly influenced by pyrite oxidation, elevated nitrate levels near the watertable and denitrification in the lower half of the aquifer. Groundwater near the watertable in the lake's capture zone contains high levels of sulfate, sulfate to chloride ratios (> 0.5) iron and aluminium. Isolated areas of groundwater acidification and high concentrations of arsenic were also identified.

Pyrite oxidation in groundwater beneath Lake Gwelup is indicated by high sulfate concentrations (up to 430 mg/L at GLP_Ea and 480 mg/L at GLP_Eb), sulfate to chloride ratios (> 0.5) and dissolved iron concentrations (up to 25 mg/L at GLP_Eb and 4.9 mg/L at GLP_Ea). These results suggest the oxidation of iron sulfide minerals (such as pyrite) to the base of the Superficial aquifer beneath the eastern edge of Lake Gwelup. However, groundwater quality at the base of the Superficial aquifer beneath the western edge of the lake (GLP_Wa) is different to other SGS bores investigated in this study. GLP_Wa does not indicate pyrite oxidation and recorded low levels of sulfate (< 50 mg/L), sulfate to chloride ratios (about 0.2) and lower concentrations of iron (up to 2.2 mg/L). This may indicate iron and sulfate reduction in groundwater at GLP_Wa.

Pyrite oxidation and elevated levels of nitrate have been shown to be significant factors influencing the quality of groundwater and surface water in other wetlands on the Swan Coastal Plain. These include Lake Mariginiup (which has acidified), Lexia Wetlands, Lake Nowergup and Egerton Seepage (Searle et al 2010; Department of Water 2011; Searle et al 2011; McHugh et al 2011). The susceptibility of groundwater and surface water to acidification from pyrite oxidation or nitrification processes is highly dependent on alkalinity and the ability to buffer against changes in pH (Swedish EPA 2002). This is well illustrated in Sweden which hosts approximately 20 000 acidified lakes in similar sandy unconfined aquifers of Quaternary age, due to acid generating processes consuming available acid buffering capacity (Knuttson 1995).

This Perth SGS investigation reinforces the need to be pragmatic and dynamic in implementing management strategies for shallow groundwater systems, responding to the drying Perth climate, changing land and water uses and an increasing population. The significance in preserving the ecological values at Lake Gwelup is increased in line with its high social value.

7 Recommendations

This report provides the basis for the following recommendations. The implementation of these recommendations are subject to Department of Water priorities and the availability of resources.

7.1 Management actions

- Develop a management strategy for the Delawney Drain, Brushfield Wetland and Brushfield Drain to improve the quality of water being discharged into Lake Gwelup.
- Review the Water Corporation's public water supply entitlements from the Gwelup groundwater area as part of the move to the reduced 120 GL baseline licence for the Integrated Water Supply Scheme.
- Review public water supply reserves in the Gwelup groundwater area.
- Review environmental sensitivity classifications of Superficial and Mirrabooka public water supply bores within the Gwelup groundwater area.
- The Superficial aquifer of the Gwelup groundwater area is not over-allocated (Department of Water 2009). However, the quality of groundwater through the entire vertical extent of the Superficial aquifer is degraded, with elevated concentrations of iron and sulfate from sulfide oxidation. Management of the Gwelup bore field should focus on geochemical characteristics of the Superficial aquifer in combination with aquifer hydraulics and groundwater levels.

7.2 Further monitoring

- Install an additional shallow groundwater bore adjacent to GLP_Wc, to assess the magnitude of watertable variation and allow consistent monitoring of groundwater quality near the watertable.
- Install a staff gauge at the lowest elevation of Lake Gwelup (about 4.6 m AHD) to record surface water elevations. The existing staff gauge should remain to allow for adequate comparisons of historical data.
- Re-install selected shallow groundwater bores, which were destroyed due to the development of the residential housing estate, Gwelup Park. MW1, MW2, MW4 and MW5 previously recorded high concentrations of arsenic, iron, sulfate and acidity.
- Monitor selected bores up-gradient of Lake Gwelup consistently, to assess the changing nature of the quality of groundwater migrating towards the lake. This program should be complemented by similar monitoring of SGS bores and surface water at Lake Gwelup.

7.3 Further investigation

- Delineate the extent and chemical composition of sediments at Lake Gwelup and the Brushfield Wetland.
- Determine the extent of the cavern(s) beneath the western edge of Lake Gwelup. This is suspected to be the main cause of significant watertable decline at GLP_Wc when the lake dries.
- Carry out a monitoring program to include chemical analysis of surface water quality at Lake Gwelup. One surface water sample was collected as part of this study in September 2010.
- Further investigation into groundwater quality and the chemical processes at the base of the Superficial aquifer beneath Lake Gwelup. Varied concentrations of alkalinity, base cations and sulfate were recorded at GLP_Ea. This was in contrast to stable concentrations detected at GLP_Wa. Evidence of sulfide oxidation was detected at GLP_Ea, while GLP_Wa was the only monitoring point not to detect an additional source of sulfate (sulfate to chloride ratio < 0.5).

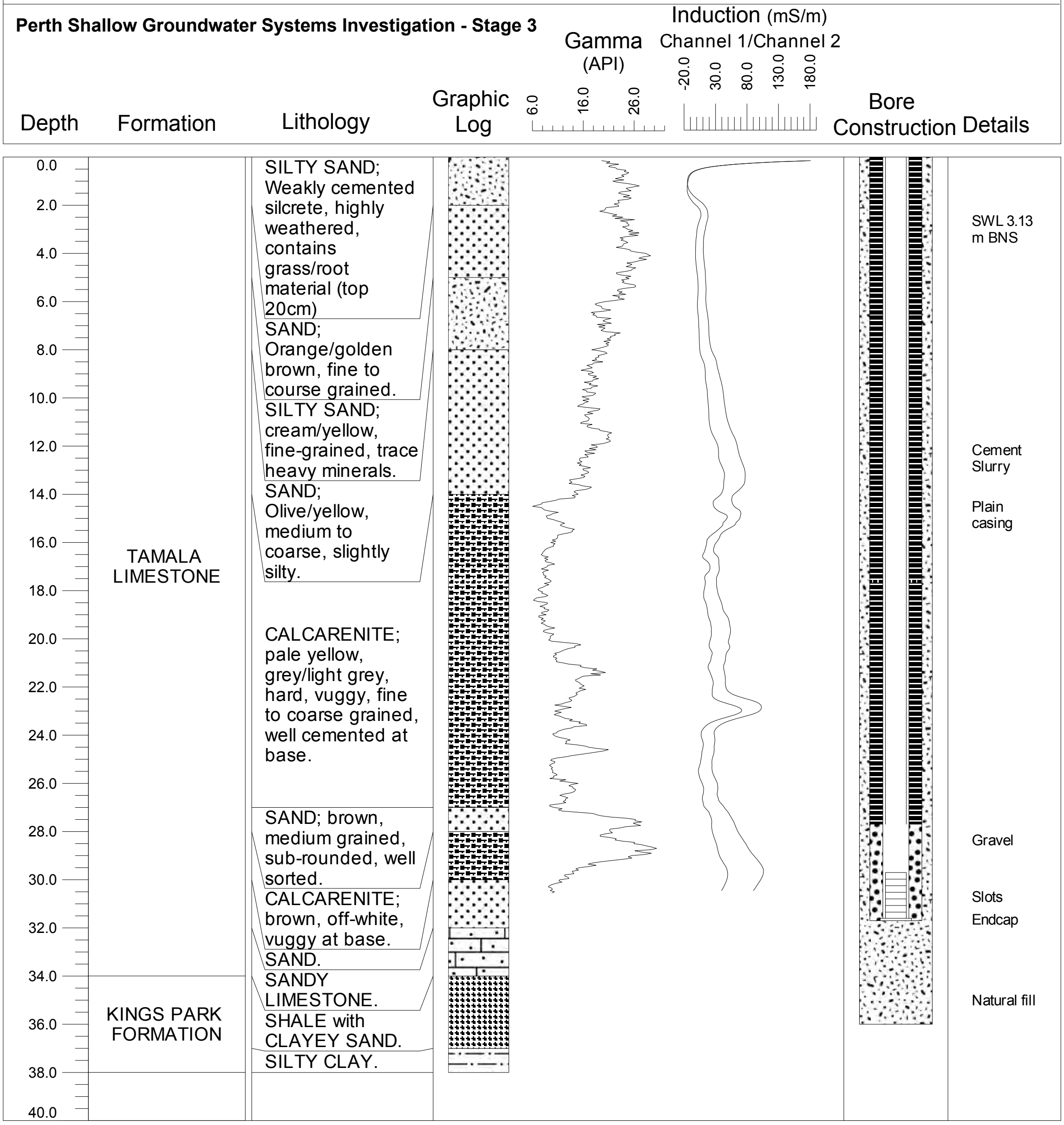
Appendices

Appendix A – Bore logs – SGS investigation

BORE COMPLETION DETAILS AWRC Name: **GLP_Ea**

AWRC Number: **61611867**

Perth Shallow Groundwater Systems Investigation - Stage 3



SILTY SAND; Weakly cemented silcrete, highly weathered, contains grass/root material (top 20cm)

SAND; Orange/golden brown, fine to course grained.

SILTY SAND; cream/yellow, fine-grained, trace heavy minerals.

SAND; Olive/yellow, medium to coarse, slightly silty.

CALCARENITE; pale yellow, grey/light grey, hard, vuggy, fine to coarse grained, well cemented at base.

SAND; brown, medium grained, sub-rounded, well sorted.

CALCARENITE; brown, off-white, vuggy at base.

SAND.

SANDY LIMESTONE.

SHALE with CLAYEY SAND.

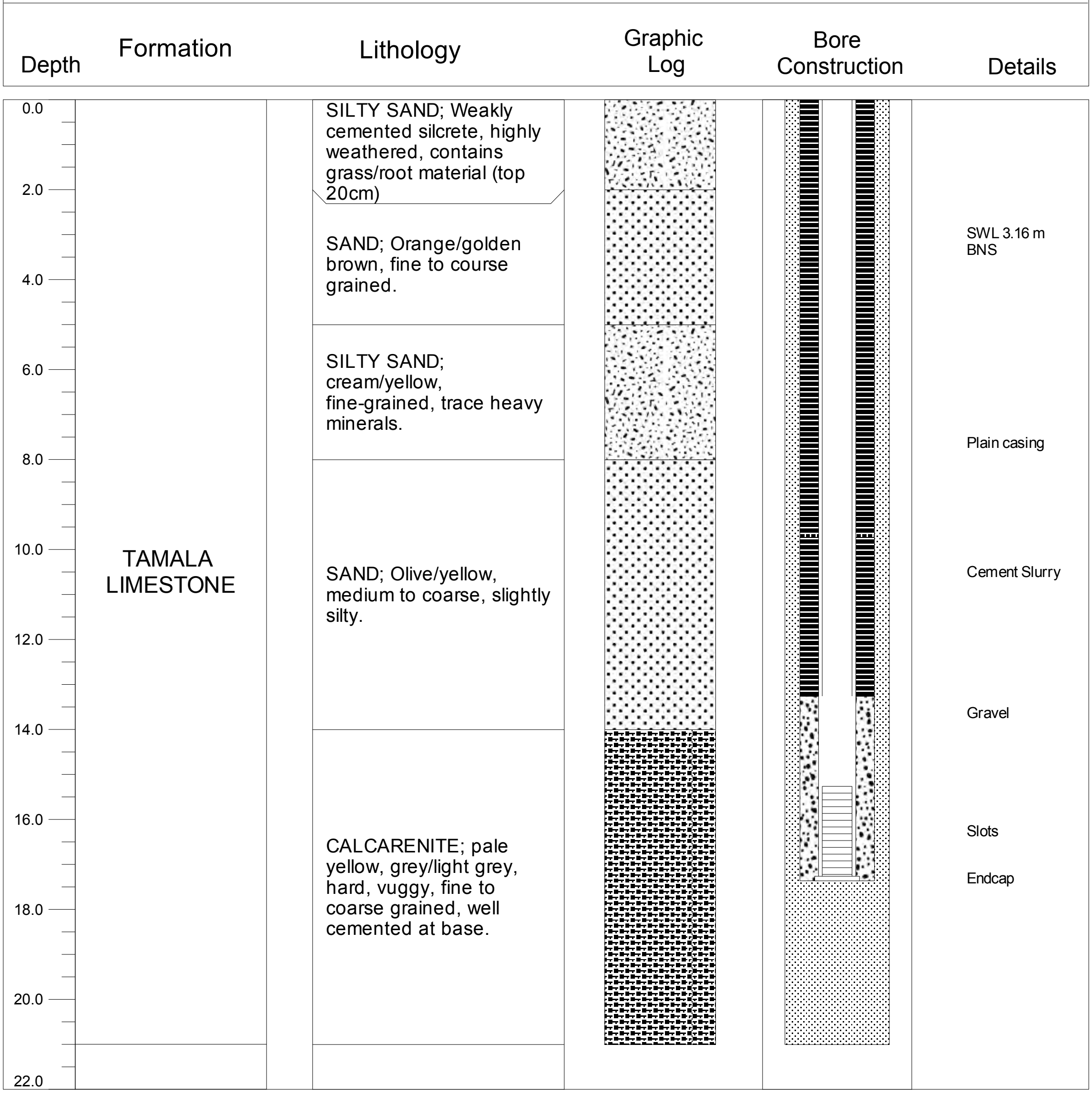
SILTY CLAY.

Drilled by:	Geo Sonic Drilling	Northing (Z50):	6472474	Screens (m BNS):	29.7 - 31.7
Date drilled:	3/31/09	Easting (Z50):	385957	SWL (m BNS):	3.13
Logged by:	Nathan Henderson	Surface (m AHD):	7.96	EC (mS/cm):	1.9
Aquifer:	Superficial	Drill depth (m BNS):	36	pH:	7.68

BORE COMPLETION DETAILS

AWRC Name:	**GLP_Eb**
AWRC Number:	**61611868**

Perth Shallow Groundwater Systems Investigation - Stage 3



SILTY SAND; Weakly cemented silcrete, highly weathered, contains grass/root material (top 20cm)

SAND; Orange/golden brown, fine to course grained.

SILTY SAND; cream/yellow, fine-grained, trace heavy minerals.

SAND; Olive/yellow, medium to coarse, slightly silty.

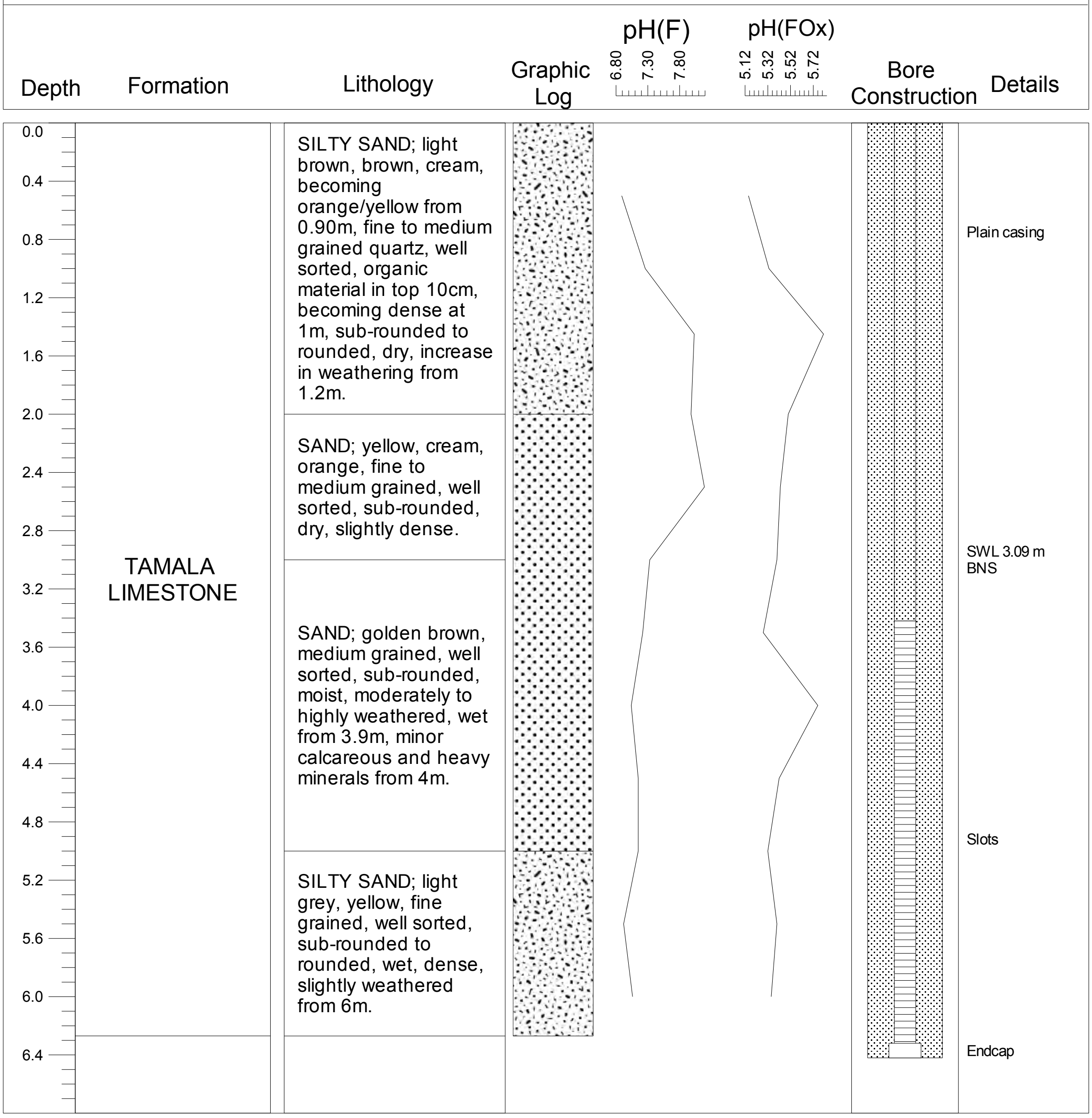
CALCARENITE; pale yellow, grey/light grey, hard, vuggy, fine to coarse grained, well cemented at base.

Drilled by:	Geo Sonic Drilling	Northing (Z50):	6472476	Screens (m BNS):	15.26 - 17.26
Date drilled:	4/1/09	Easting (Z50):	385961	SWL (m BNS):	3.19
Logged by:	Nathan Henderson	Surface (m AHD):	8	EC (mS/m):	1.31
Aquifer:	Superficial	Drill depth (m BNS):	21	pH:	7.4

BORE COMPLETION DETAILS

AWRC Name: **GLP_Ec**
AWRC Number: **61611876**

Perth Shallow Groundwater Systems Investigation - Stage 3



SILTY SAND; light brown, brown, cream, becoming orange/yellow from 0.90m, fine to medium grained quartz, well sorted, organic material in top 10cm, becoming dense at 1m, sub-rounded to rounded, dry, increase in weathering from 1.2m.

SAND; yellow, cream, orange, fine to medium grained, well sorted, sub-rounded, dry, slightly dense.

SAND; golden brown, medium grained, well sorted, sub-rounded, moist, moderately to highly weathered, wet from 3.9m, minor calcareous and heavy minerals from 4m.

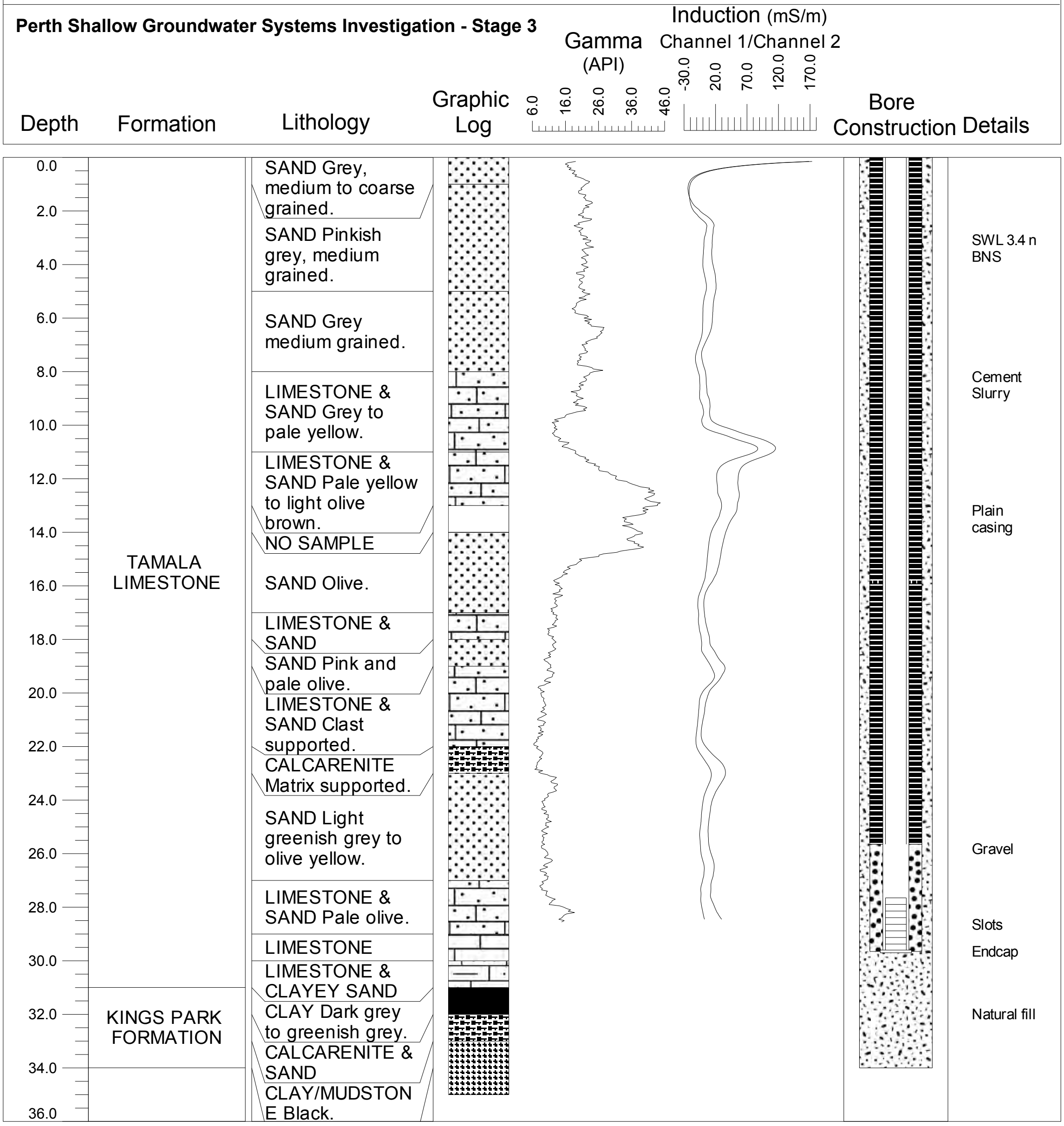
SILTY SAND; light grey, yellow, fine grained, well sorted, sub-rounded to rounded, wet, dense, slightly weathered from 6m.

Drilled by:	Curtin University	Northing (Z50):		Screens (m BNS):	3.42 - 6.42
Date drilled:		Easting (Z50):		SWL (m BNS):	3.09
Logged by:	Nathan Henderson	Surface (m AHD):		EC (uS/m):	0.912
Aquifer:	Superficial	Drill depth (m BNS):	6.27	pH:	8.01

BORE COMPLETION DETAILS AWRC Name: **GLP_Wa**

AWRC Number: **61611865**

Perth Shallow Groundwater Systems Investigation - Stage 3

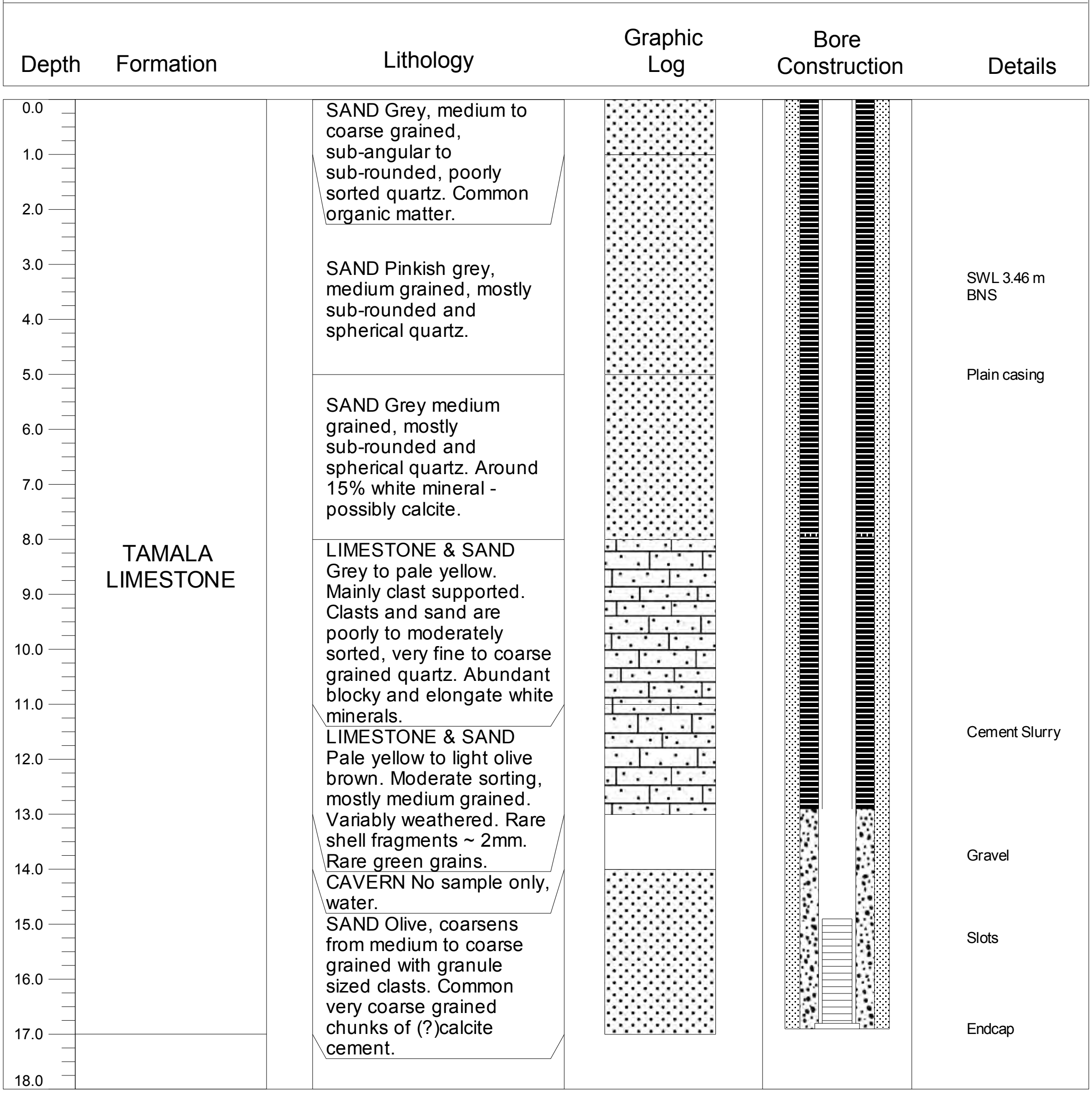


Drilled by:	Geo Sonic Drilling	Northing (Z50):	6472516	Screens (m BNS):	27.65 - 29.65
Date drilled:	4/7/09	Easting (Z50):	385386	SWL (m BNS):	3.415
Logged by:	Josephine Searle	Surface (m AHD):	7.1	EC (mS/cm):	0.994
Aquifer:	Superficial	Drill depth (m BNS):	34	pH:	7.39

BORE COMPLETION DETAILS

AWRC Name:	**GLP_Wb**
AWRC Number:	**61611866**

Perth Shallow Groundwater Systems Investigation - Stage 3



SAND Grey, medium to coarse grained, sub-angular to sub-rounded, poorly sorted quartz. Common organic matter.

SAND Pinkish grey, medium grained, mostly sub-rounded and spherical quartz.

SAND Grey medium grained, mostly sub-rounded and spherical quartz. Around 15% white mineral - possibly calcite.

LIMESTONE & SAND Grey to pale yellow. Mainly clast supported. Clasts and sand are poorly to moderately sorted, very fine to coarse grained quartz. Abundant blocky and elongate white minerals.

LIMESTONE & SAND Pale yellow to light olive brown. Moderate sorting, mostly medium grained. Variably weathered. Rare shell fragments ~ 2mm. Rare green grains.

CAVERN No sample only, water.

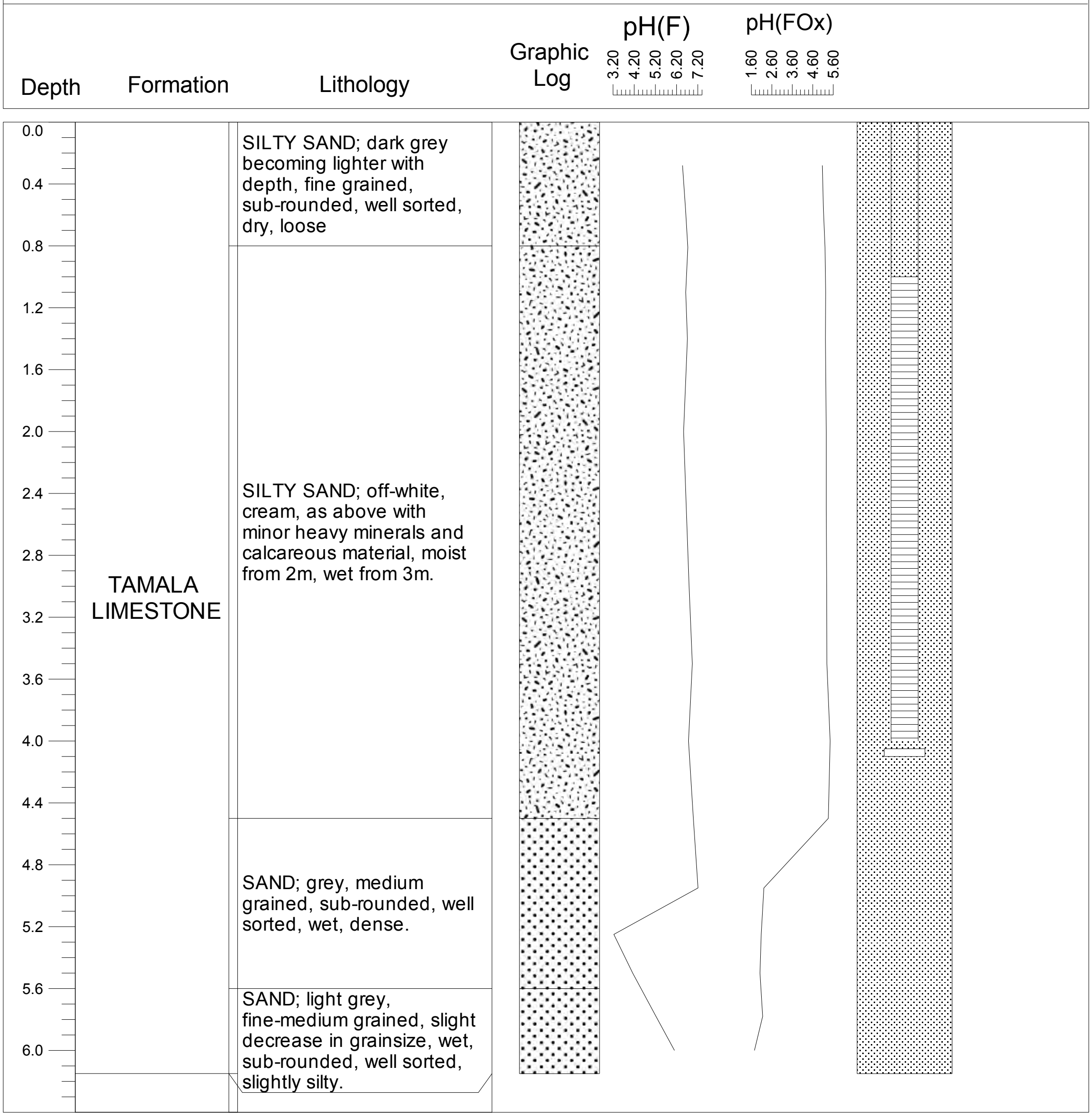
SAND Olive, coarsens from medium to coarse grained with granule sized clasts. Common very coarse grained chunks of (?)calcite cement.

Drilled by:	Geo Sonic Drilling	Northing (Z50):	6472517	Screens (m BNS):	14.9 - 16.9
Date drilled:	4/9/09	Easting (Z50):	385385	SWL (m BNS):	3.46
Logged by:	Josephine Searle	Surface (m AHD):	7.08	EC (mS/m):	0.931
Aquifer:	Superficial	Drill depth (m BNS):	17	pH:	7.68

BORE COMPLETION DETAILS

AWRC Name: **GLP_Wc**
AWRC Number: **61611877**

Perth Shallow Groundwater Systems Investigation - Stage 3



SILTY SAND; dark grey becoming lighter with depth, fine grained, sub-rounded, well sorted, dry, loose

SILTY SAND; off-white, cream, as above with minor heavy minerals and calcareous material, moist from 2m, wet from 3m.

SAND; grey, medium grained, sub-rounded, well sorted, wet, dense.

SAND; light grey, fine-medium grained, slight decrease in grainsize, wet, sub-rounded, well sorted, slightly silty.

Drilled by:	Curtin University	Northing (Z50):	6472524	Screens (m BNS):	
Date drilled:		Easting (Z50):	385386	SWL (m BNS):	2.93
Logged by:	Nathan Henderson	Surface (m AHD):		EC (uS/m):	
Aquifer:	Superficial	Drill depth (m BNS):	6.15	pH:	

Appendix B – Bore logs and construction details (DEC)

Location of groundwater bores installed by the Department of Environment and Conservation (DEC) in 2006

Bore ID	Coordinates		Coordinates		Date
	Latitude	Longitude	Easting	Northing	installed
GWA1	-31.878381	115.801565	386643.00	6472925.00	28/02/2006
GWA2	-31.874952	115.799954	386492.00	6472797.00	1/03/2006
GWA3	-31.875498	115.801374	386627.00	6472738.00	28/02/2006
GWA4	-31.876567	115.798062	386315.00	6472616.00	28/02/2006
GWA5	-31.877135	115.799905	386490.00	6472555.00	1/03/2006
GWA6	-31.878660	115.801904	386681.00	6472388.00	28/02/2006
GWA7	-31.875800	115.799100	386413.41	6472706.47	-
GWA8	-31.876140	115.798770	386379.07	6472660.21	-
GWA9a	-31.876560	115.798700	386375.31	6472617.47	-
GWA10	-31.875590	115.798550	386354.25	6472728.07	-
GWA11	-31.875650	115.797700	386289.16	6472726.60	-
GWA12	-31.875800	115.797550	386263.38	6472694.34	-
GWA13	-31.876340	115.797260	386237.73	6472640.51	-
GM13	-31.876542	115.799225	386425.00	6472620.00	Water Corp

Construction details of groundwater bores installed by DEC in 2006

Bore ID	Casing mm	Screen Interval mbgl	Total depth mbgl
GWA1	50	3.0–5.0	5.0
GWA2	50	0.5–3.0	3.0
GWA3	50	4.0–6.0	6.0
GWA4	20	3.0–5.0	5.0
GWA5	50	0.7–2.2	2.16
GWA6	20	1.0–3.0	3.0
GWA7	50	0.5–6.5	6.50
GWA8	50	-	-
GWA9a	50		
GWA10	50	1.5–5.5	5.50
GWA11	50	3.0–6.0	6.00

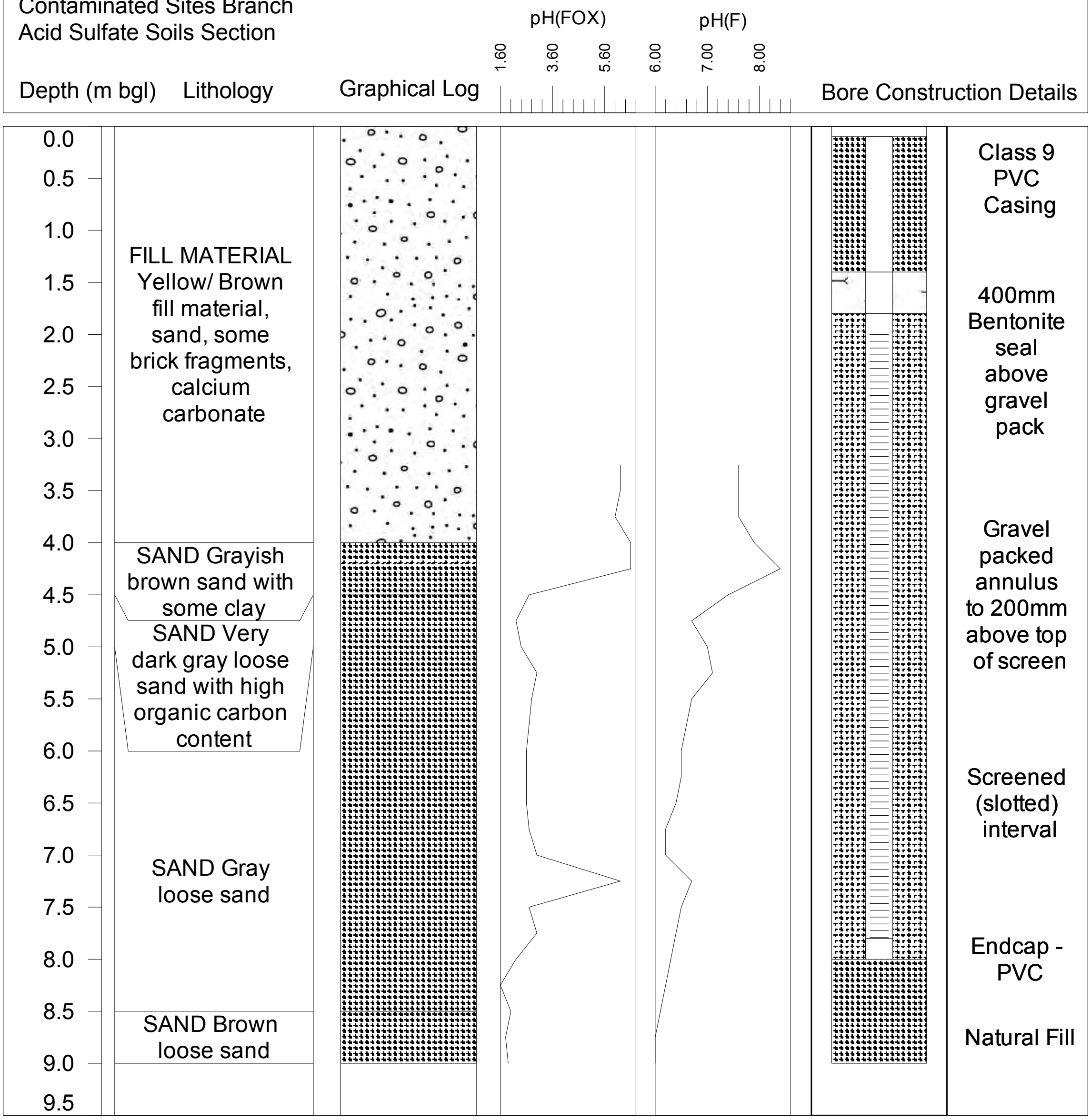
Bore ID	Casing mm	Screen Interval mbgl	Total depth mbgl
GWA12	50	2.0–5.0	5.00
GWA13	50	1.5–6.5	6.50
GM13	80	3.3–15.3	15.28

Groundwater bores installed by the Department of Environment and Conservation November 2008

Gwelup Soil and Groundwater Investigation
Bore Completion Details

Department of Environment and Conservation
Contaminated Sites Branch
Acid Sulfate Soils Section

Bore ID: MW1

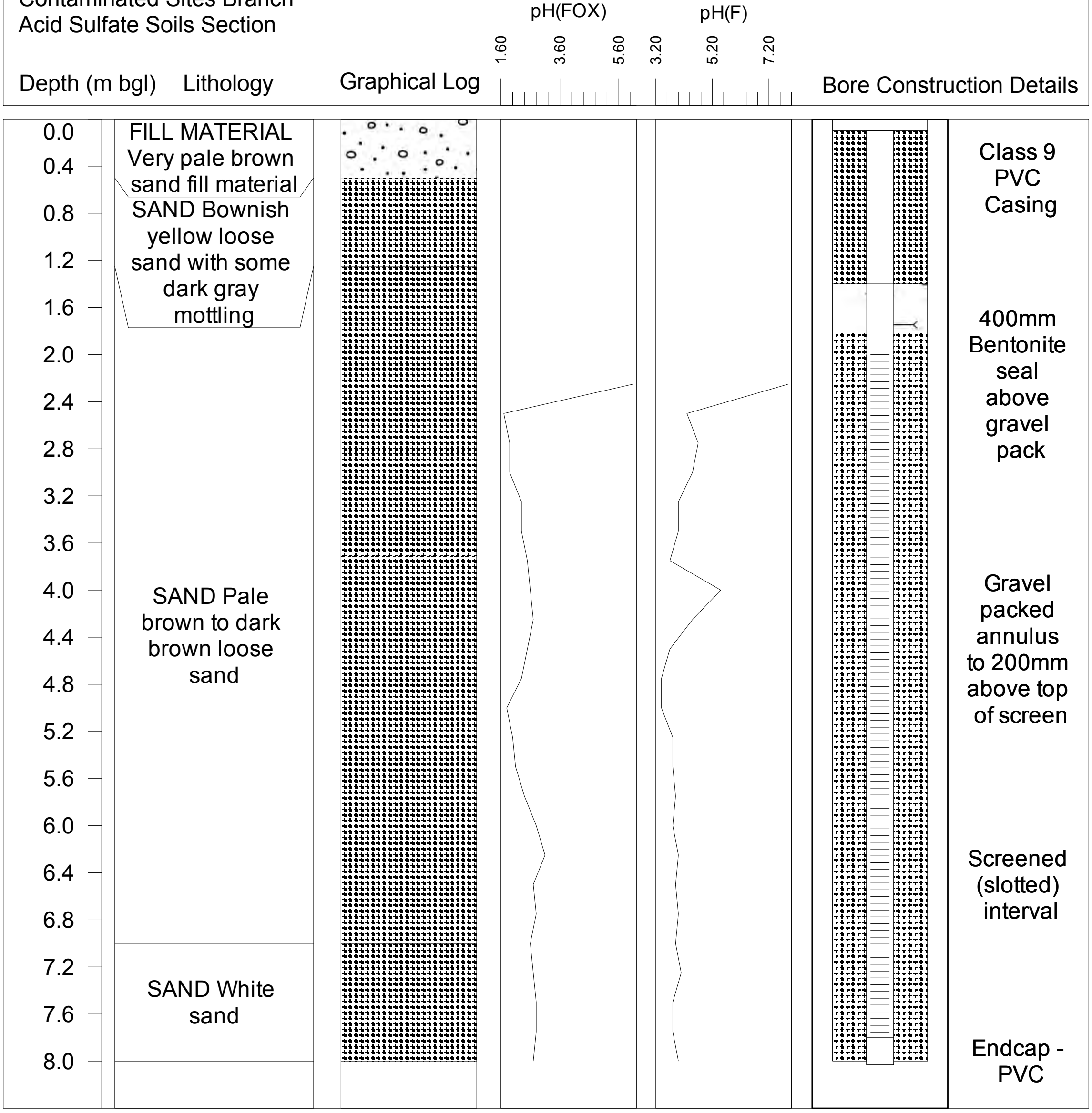


Drilled by:	Strataprobe	Northing:	6472824.068
Date Drilled:	11/18/08	Easting:	386410.866
Logged by:	Yash Pal	Soil Core Depth (m bgl):	9.0
Aquifer:	Superficial	Groundwater Bore Depth (m bgl):	8.0
SWL (m bgl):	3.0	Screen (m bgl):	2.0 - 8.0

Gwelup Soil and Groundwater Investigation
Bore Completion Details

Department of Environment and Conservation
Contaminated Sites Branch
Acid Sulfate Soils Section

Bore ID: MW2



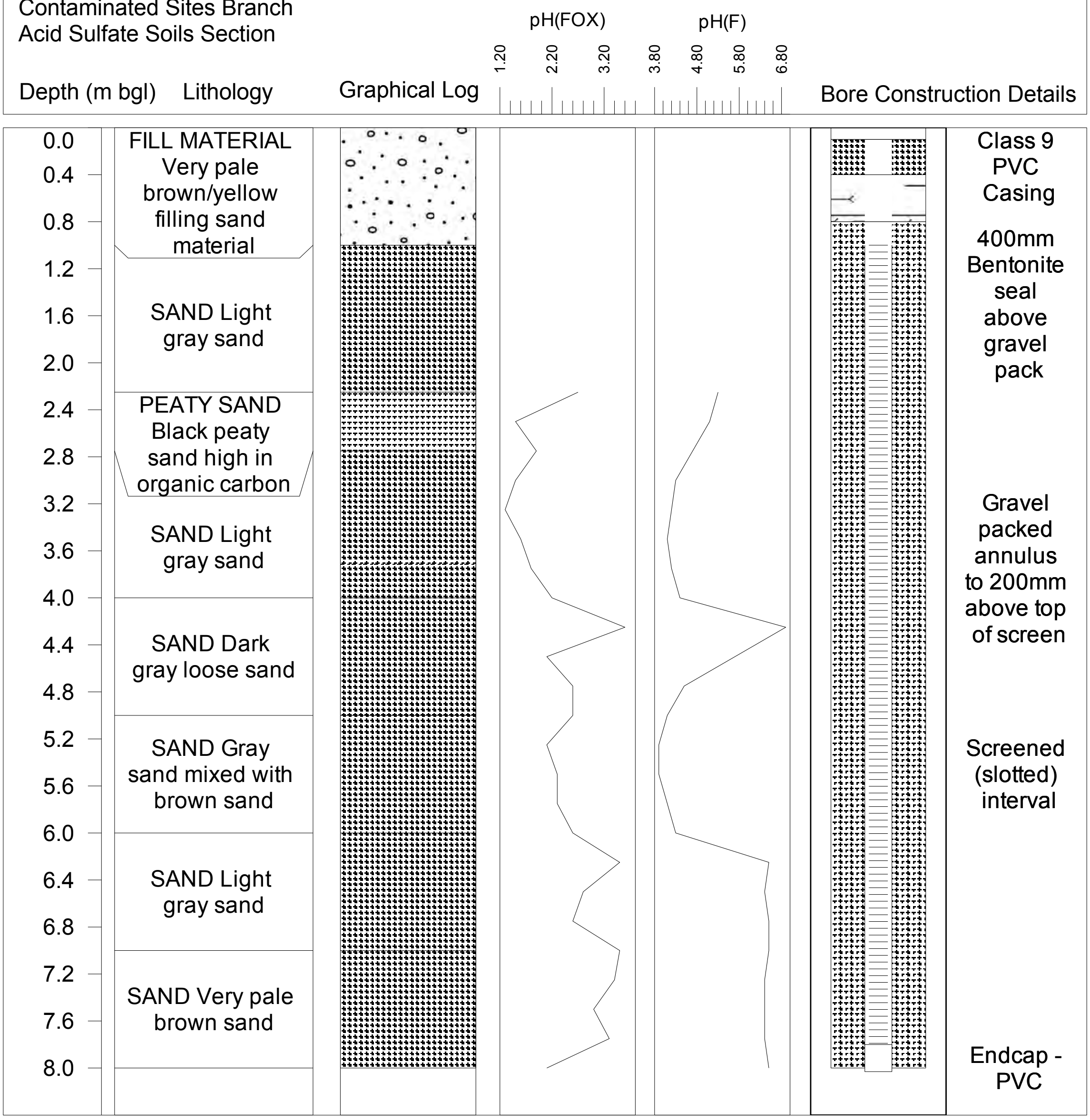
Drilled by:	Strataprobe	Northing:	6472823.143
Date Drilled:	11/18/08	Easting:	386327.309
Logged by:	Yash Pal	Soil Core Depth (m bgl):	8.0
Aquifer:	Superficial	Groundwater Bore Depth (m bgl):	8.0
SWL (m bgl):	3.0	Screen (m bgl):	2.0 - 8.0

Gwelup Soil and Groundwater Investigation

Bore Completion Details

Department of Environment and Conservation
Contaminated Sites Branch
Acid Sulfate Soils Section

Bore ID: MW3

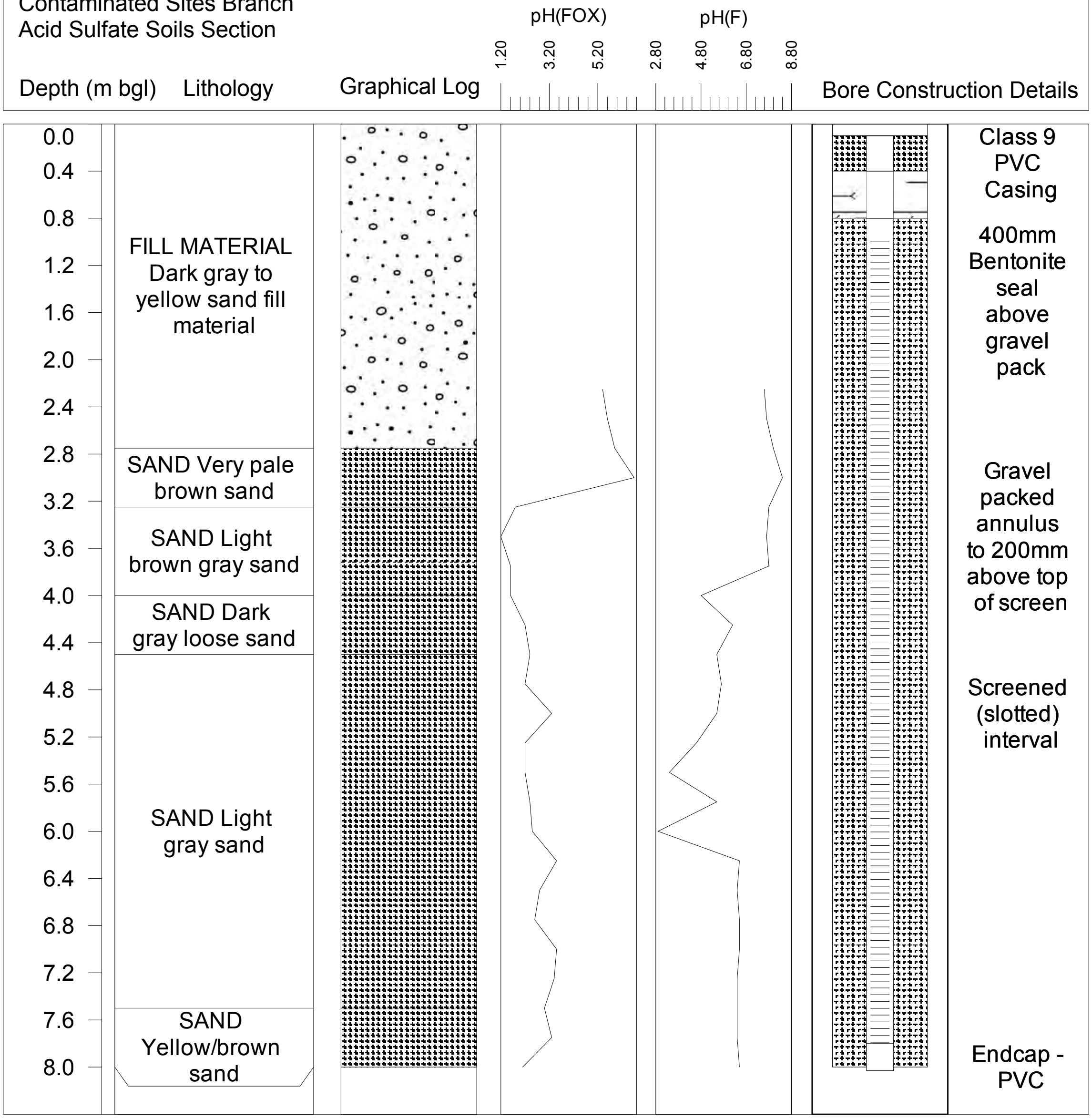


Drilled by:	Strataprobe	Northing:	6472780.479
Date Drilled:	11/19/08	Easting:	386477.587
Logged by:	Yash Pal	Soil Core Depth (m bgl):	8.0
Aquifer:	Superficial	Groundwater Bore Depth (m bgl):	8.0
SWL (m bgl):	1.5	Screen (m bgl):	1.0 - 8.0

Gwelup Soil and Groundwater Investigation
Bore Completion Details

Department of Environment and Conservation
Contaminated Sites Branch
Acid Sulfate Soils Section

Bore ID: MW4

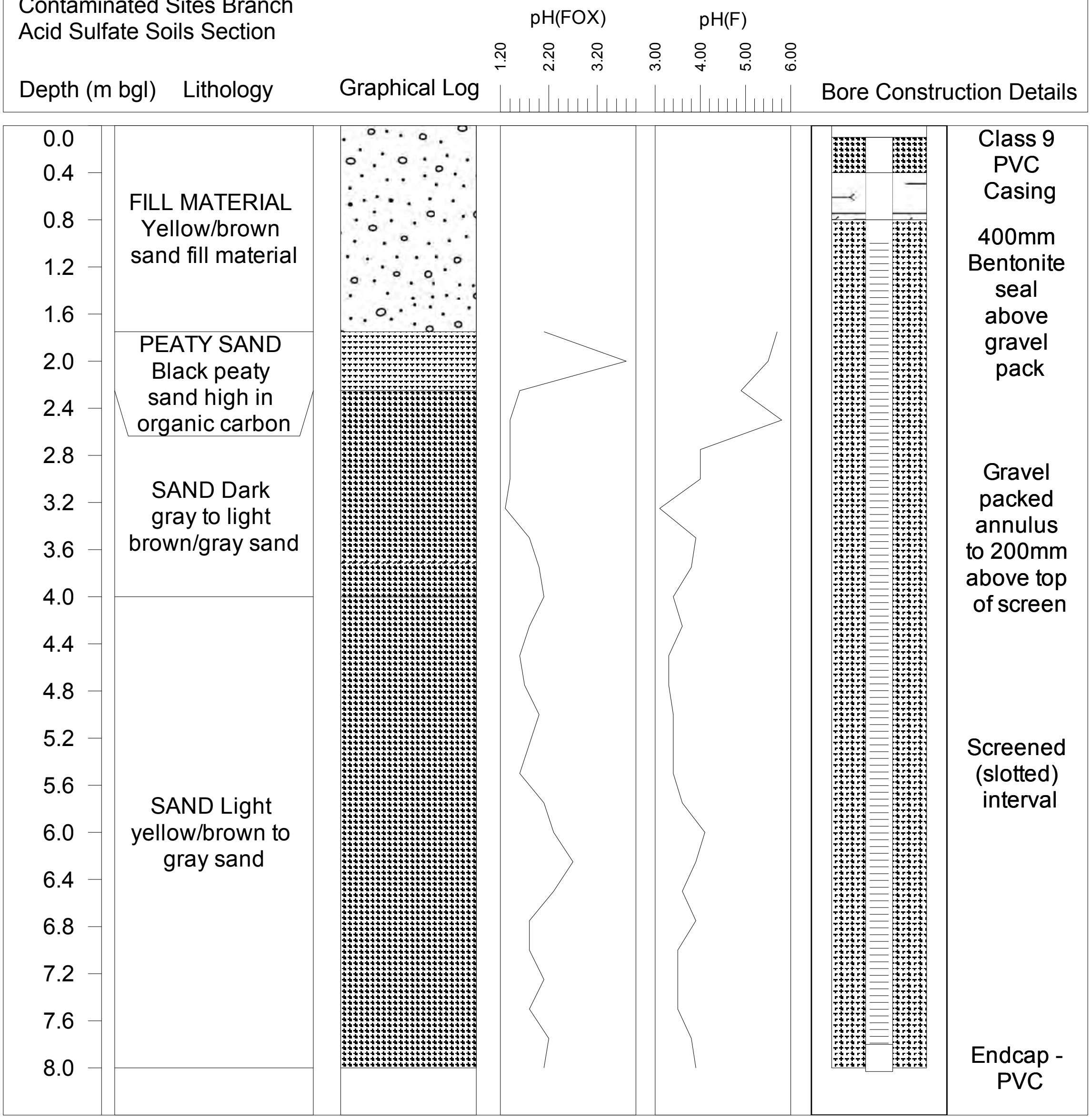


Drilled by:	Strataprobe	Northing:	6472702.551
Date Drilled:	11/19/08	Easting:	386450.052
Logged by:	Yash Pal	Soil Core Depth (m bgl):	8.0
Aquifer:	Superficial	Groundwater Bore Depth (m bgl):	8.0
SWL (m bgl):	2.0	Screen (m bgl):	1.0 - 8.0

Gwelup Soil and Groundwater Investigation
Bore Completion Details

Department of Environment and Conservation
Contaminated Sites Branch
Acid Sulfate Soils Section

Bore ID: MW5



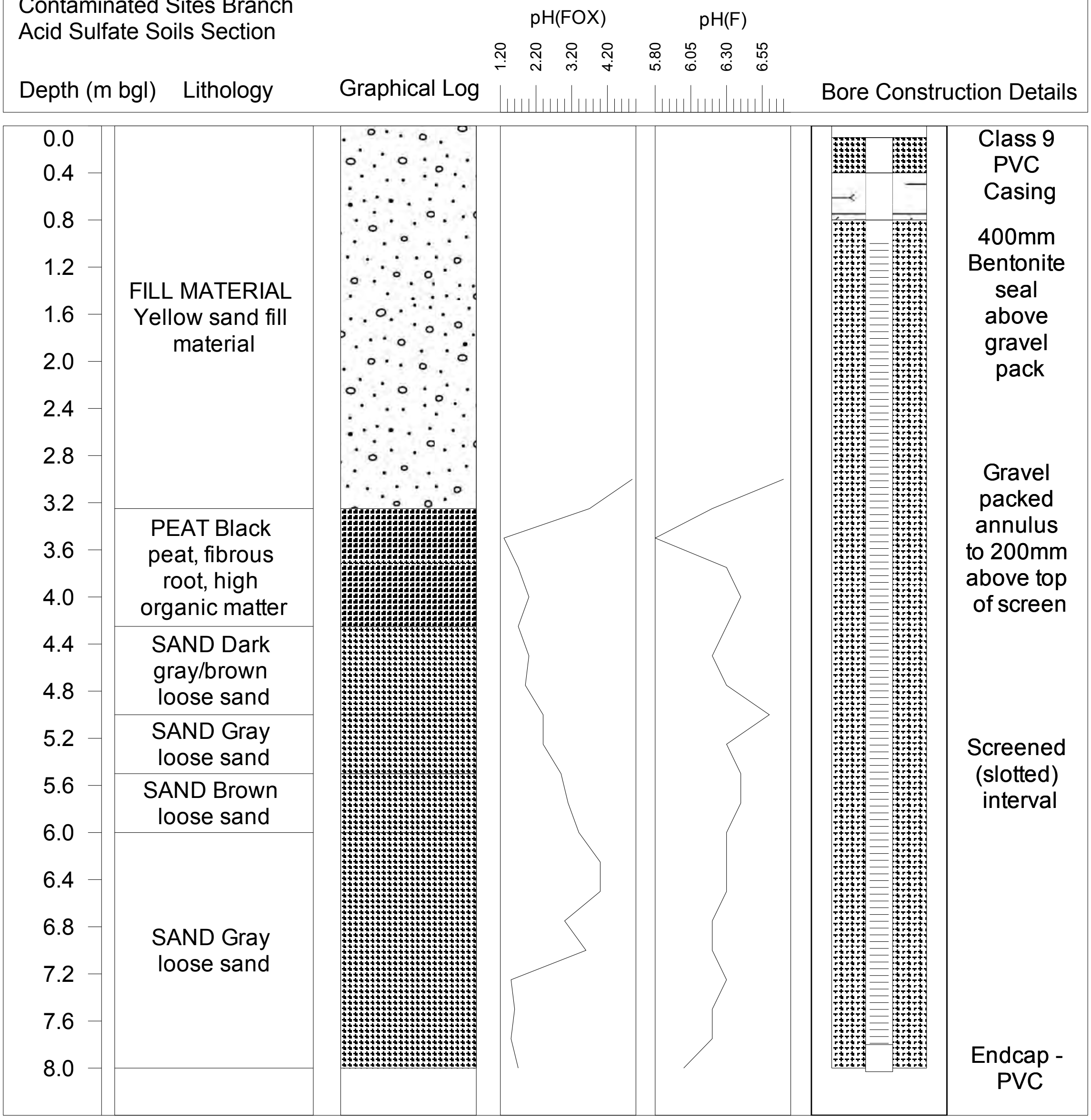
Drilled by:	Strataprobe	Northing:	6472636.354
Date Drilled:	11/19/08	Easting:	386479.182
Logged by:	Yash Pal	Soil Core Depth (m bgl):	8.0
Aquifer:	Superficial	Groundwater Bore Depth (m bgl):	8.0
SWL (m bgl):	1.7	Screen (m bgl):	1.0 - 8.0

Gwelup Soil and Groundwater Investigation

Bore Completion Details

Department of Environment and Conservation
Contaminated Sites Branch
Acid Sulfate Soils Section

Bore ID: MW6

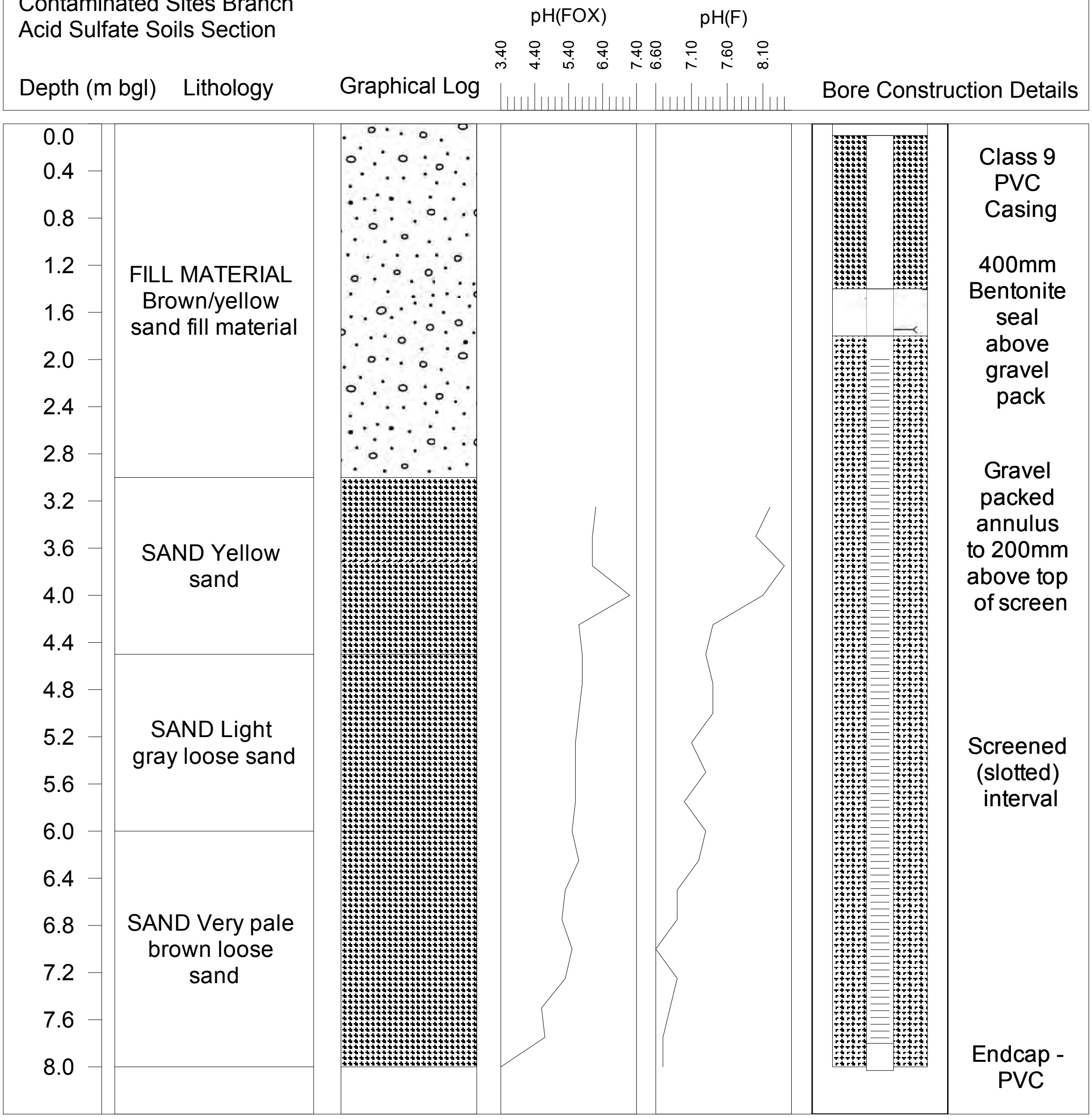


Drilled by:	Strataprobe	Northing:	6472902.520
Date Drilled:	11/19/08	Easting:	386485.682
Logged by:	Yash Pal	Soil Core Depth (m bgl):	8.0
Aquifer:	Superficial	Groundwater Bore Depth (m bgl):	8.0
SWL (m bgl):	1.5	Screen (m bgl):	1.0 - 8.0

Gwelup Soil and Groundwater Investigation
Bore Completion Details

Department of Environment and Conservation
Contaminated Sites Branch
Acid Sulfate Soils Section

Bore ID: GWA9



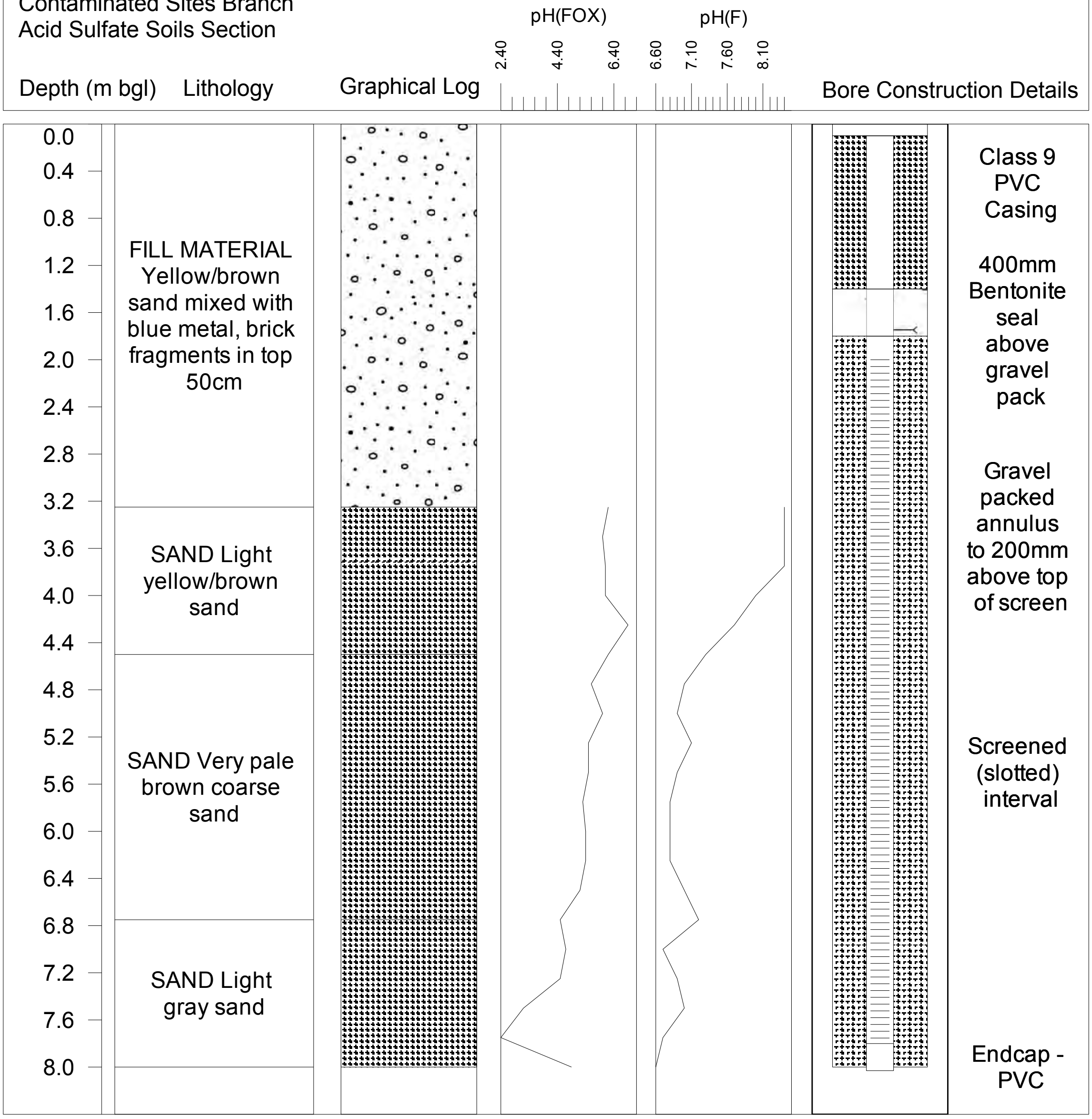
Drilled by: Strataprobe
Date Drilled: 11/20/08
Logged by: Yash Pal
Aquifer: Superficial
SWL (m bgl): 4.0

Northing: 6472612.422
Easting: 386318.622
Soil Core Depth (m bgl): 8.0
Groundwater Bore Depth (m bgl): 8.0
Screen (m bgl): 2.0 - 8.0

Gwelup Soil and Groundwater Investigation
Bore Completion Details

Department of Environment and Conservation
Contaminated Sites Branch
Acid Sulfate Soils Section

Bore ID: GM13



Drilled by:	Strataprobe	Northing:	6472616.000
Date Drilled:	11/20/08	Easting:	386424.000
Logged by:	Yash Pal	Soil Core Depth (m bgl):	8.0
Aquifer:	Superficial	Groundwater Bore Depth (m bgl):	8.0
SWL (m bgl):	3.1	Screen (m bgl):	2.0 - 8.0

Appendix C – Surface water sample results

Location and collection date of surface water samples – Lake Gwelup and Brushfield Wetland

Sample ID	Sample location	Coordinates		Collection date
		Latitude	Longitude	
Staff gauge	Lake Gwelup – West	-31.87635	115.78840	3/09/2010
Surface 2	Brushfield Wetland	-31.87930	115.79933	1/09/2010
Surface 3	Lake Gwelup – East	-31.87936	115.79630	2/09/2010

Results from surface water sampling undertaken in September 2010

Sample location		Lake Gwelup – West	Lake Gwelup – East	Brushfield Wetland
Lab sample ID		**200809616**	**200809614**	**200809610**
Analyte	**Units**			
Electrical conductivity	µS/cm	438	766	183
Total dissolved solids	mg/L	270	426	104
Total suspended solids	mg/L	< 5	11	128
Total alkalinity as CaCO3	mg/L	92	79	24
Total acidity as CaCO3	mg/L	5	28	19
Sulfate	mg/L	61	102	82
Chloride	mg/L	NA	NA	NA
Base cations				
Calcium	mg/L	55	56	46
Magnesium	mg/L	4	5	3
Sodium	mg/L	21	26	24
Potassium	mg/L	4	3	4
Dissolved metals				
Aluminium	mg/L	< 0.01	< 0.01	0.02
Arsenic	mg/L	0.003	0.002	0.004
Cadmium	mg/L	< 0.0001	< 0.0001	< 0.0001
Chromium	mg/L	< 0.001	< 0.001	< 0.001
Copper	mg/L	< 0.001	< 0.001	0.002

Sample location		**Lake Gwelup – West**	**Lake Gwelup – East**	**Brushfield Wetland**
Lab sample ID		**200809616**	**200809614**	**200809610**
Analyte	**Units**			
Nickel	mg/L	< 0.001	< 0.001	< 0.001
Lead	mg/L	< 0.001	< 0.001	< 0.001
Zinc	mg/L	< 0.005	0.024	0.046
Manganese	mg/L	0.006	0.094	0.02
Selenium	mg/L	< 0.01	< 0.01	< 0.01
Boron	mg/L	< 0.05	< 0.05	< 0.05
Iron	mg/L	0.29	6.53	3.53
Mercury	mg/L	< 0.0001	< 0.0001	< 0.0001
Total metals				
Aluminium	mg/L	0.06	0.01	0.24
Arsenic	mg/L	0.004	0.001	0.012
Iron	mg/L	0.86	4.65	7.78
Nutrients				
Nitrite	mg/L	< 0.01	< 0.01	0.01
Nitrite + nitrate (NOx)	mg/L	< 0.01	0.06	0.1
Total Kjeldahl nitrogen	mg/L	0.6	0.8	0.5
Total nitrogen	mg/L	0.6	0.9	0.6
Total phosphorus	mg/L	0.02	< 0.01	< 0.01

Note: NA denotes ‘not assessed’.

Appendix D – Groundwater quality data (DEC), 2006–10

To provide some context in the interpretation of the following results, comparisons have been drawn with guidelines published in, *Assessment levels for soil sediment and water* (DEC 2010), which includes the . As groundwater is generally used for irrigation purposes in Gwelup, the most relevant criteria are considered the 'Long-term irrigation criteria' and the Department of Health based non-potable groundwater use guideline (DEC 2010). Although the Gwelup bore field abstracts water for public potable supply, this water is treated and fit for use prior to public consumption.

Breaching the following criteria should be used as a basis for further investigation and risk assessment. These guidelines are not intended to be used as the sole basis on which 'remediation' is instigated or management plans are formed.

Note: The colour coding shown in the assessment criteria table is used in the results tables to indicate breaches of particular criteria.

Assessment criteria	**Dissolved metals**									
	As	**Al**	**B**	**Cd**	**Cr**	**Cu**	**Pb**	**Mn**	**Ni**	**Zn**
	mg/L	**mg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**
FWG	0.024	0.055	0.37	0.0002	-	0.0014	0.0034	1.9	0.011	0.008
ADWG (2004)	0.007	0.2*	4.00	0.002	-	2.00	0.01	0.5	0.02	3*
DoH (2006)	0.07	2.00	40.00	0.020	-	20.00	0.10	5	0.20	30.00
LTIW	0.1	5.00	0.50	0.010	0.10	0.20	2.00	0.2	0.20	2.00

FWG: Fresh water guidelines

ADWG: Australian drinking water guidelines (NHMRC 2004)

DoH: Department of Health

LTIW: Long-term irrigation criteria

**: indicates an aesthetic guideline (eg. odour, staining, taste) rather than an indication of health implications*

Assessment criteria	Total metals								
	Fe	Se	NH_3	N	Total N	NO_3	NO_2	Total P	SO_4
(DEC 2010)	mg/L	(mg/L)	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
FWG	0.3	0.005	-	-	2	-	-	0.2	-
ADWG (2004)	0.30	0.01	-	-	-	50	3	-	500
DoH (2006)	3.00	0.1	5	-	-	500	30	-	5000
LTIW	0.20	0.02	-	5	-	-	-	-	-

FWG: Fresh water guidelines

ADWG: Australian drinking water guidelines (NHMRC 2004)

DoH: Department of Health

LTIW: Long-term irrigation criteria

Groundwater sampling – 10 March 2006

Field results

Bore ID	Collection date	Collection method	Standing water level mbgl	Field pH	Electrical conductivity mS/cm	Redox (Eh) (mV)
GWA1	10/03/2006	Hand bailer	4.37	5.99	1.995	−105
GWA2	10/03/2006	Hand bailer	1.28	6.72	1.342	−94
GWA3	10/03/2006	Hand bailer	4.45	5.37	0.900	−142
GWA4	10/03/2006	Hand bailer	3.97	7.30	1.070	−58
GWA5	10/03/2006	Hand bailer	0.96	4.20	1.927	148
GWA6	10/03/2006	Hand bailer	1.17	6.64	1.550	−174
GM13	10/03/2006	No sample	3.31	7.14	0.944	248

Laboratory results

Bore ID	Total acidity mg/L CaCO3	Dissolved metals							
		As mg/L	Al mg/L	Cd mg/L	Cr mg/L	Pb mg/L	Mn mg/L	Ni mg/L	Iron mg/L
GWA1	35	0.002	0.18	< 0.0002	0.005	0.001	0.072	0.004	1.0
GWA2	85	0.290	0.032	< 0.0002	0.002	< 0.001	0.31	0.008	0.3
GWA3	25	0.008	0.25	< 0.0002	0.003	< 0.001	0.37	0.009	15.0
GWA4	NA	0.008	0.23	< 0.0002	0.002	0.003	0.14	0.003	0.9
GWA5	620	0.021	6.70	< 0.0002	0.007	0.007	0.68	0.027	330.0
GWA6	140	0.290	0.14	< 0.0002	0.005	0.005	1.1	< 0.001	58.0
GM13	No sample submitted to the laboratory								

Note: NA denotes 'not assessed'.

Groundwater sampling – 27 September 2007 to 5 October 2007

Field results

Bore ID	Collection date	Collection method	Standing water level mbgl	Sample depth mbgl	Field pH	Electrical conductivity mS/cm	Redox (Eh) mV
GWA4	2/10/2007	Hand bailer	3.53	4.60	6.98	1.791	-
GWA7	5/10/2007	Peristaltic pump	2.85	4.00	2.66	3.71	333
GWA8	2/10/2007	Peristaltic pump	2.91	3.50	6.84	0.095	183
GWA9	2/10/2007	No sample	3.72	4.02	7.50	-	-

Bore ID	Collection date	Collection method	Standing water level mbgl	Sample depth mbgl	Field pH	Electrical conductivity mS/cm	Redox (Eh) mV
GWA10	4/10/2007	Peristaltic pump	3.21	4.00	6.47	0.886	147
GWA11	27/09/2007	Peristaltic pump	3.24	5.42	6.79	0.561	−54
GWA12	2/10/2007	Peristaltic Pump	3.98	4.90	6.97	0.561	152
GWA13	27/09/2007	Submersible pump	3.83	4.50	6.72	0.758	−27
GM13	2/10/2007	Peristaltic pump	3.05	5.00	6.57	0.635	188

Laboratory results

Bore ID	Total acidity mg/L $CaCO_3$	Total alkalinity mg/L $CaCO_3$	Sulfate mg/L	Chloride mg/L	SO_4:Cl	Ammonia mg/L
GWA4	15	269	100	358	0.28	0.080
GWA7	1260	< 1	2210	28.3	78.09	41.0
GWA8	< 1	1	3	3.5	0.86	< 0.01
GWA9	Insufficient water to collect a sample					
GWA10	21	233	84	13.1	6.41	0.011
GWA11	56	251	14	17.2	0.81	0.030
GWA12	6	269	11	29.4	0.37	0.100
GWA13	46	178	160	26.4	6.06	0.091
GM13	9	96	55	90	0.61	0.564

Bore ID	**Dissolved metals**									
	As mg/L	**Al mg/L**	**Cd mg/L**	**Cr mg/L**	**Cu mg/L**	**Pb mg/L**	**Mn mg/L**	**Ni mg/L**	**Zn mg/L**	**Fe mg/L**
GWA4	0.004	<0.01	0.001	<0.005	0.002	<0.001	0.005	0.001	<0.005	0.06
GWA7	0.269	44.7	0.0007	0.073	0.013	0.037	0.886	0.025	0.197	342
GWA8	< 0.001	0.01	<0.0001	<0.001	0.001	<0.001	<0.001	0.002	<0.005	0.11
GWA9	Insufficient water to collect a sample									
GWA10	0.002	0.02	<0.0001	<0.001	< 0.001	<0.001	0.008	0.004	<0.005	<0.05
GWA11	0.010	<0.01	<0.0001	<0.001	< 0.001	<0.001	0.665	0.004	<0.005	1.82
GWA12	0.006	<0.01	<0.0001	<0.005	0.003	<0.001	0.012	<0.001	<0.005	<0.05
GWA13	0.004	<0.01	<0.0001	<0.001	< 0.001	<0.001	0.269	0.001	0.008	1.17
GM13	0.002	<0.01	<0.0001	<0.005	< 0.001	<0.001	<0.001	<0.001	<0.005	<0.05

Groundwater sampling – 5 May 2008 to 8 May 2008

Field results

Bore ID	**Collection date**	**Collection method**	**Standing water level mbgl**	**Sample depth mbgl**	**Field pH**	**Electrical conductivity mS/cm**	**Redox (Eh) mV**
GWA4	5/05/2008	No sample	4.42	Insufficient water			
GWA7	8/05/2008	Peristaltic pump	3.80	4.50	3.09	1.886	217
GWA8	7/05/2008	Peristaltic pump	3.75	4.00	6.71	0.101	205
GWA9	8/05/2008	No sample	Dry				

Bore ID	Collection date	Collection method	Standing water level mbgl	Sample depth mbgl	Field pH	Electrical conductivity mS/cm	Redox (Eh) mV
GWA10	7/05/2008	Peristaltic pump	4.10	5.00	6.31	0.494	53
GWA11	6/05/2008	Hand bailer	4.14	5.50	6.74	0.781	172
GWA12	6/05/2008	Hand bailer	4.81	5.10	6.67	1.025	209
GWA13	5/05/2008	Peristaltic pump	4.74	5.10	6.41	0.623	48
GM13	7/05/2008	Submersible pump	3.85	5.00	6.57	0.415	−56

Laboratory results

Bore ID	Total acidity mg/L $CaCO_3$	Total alkalinity mg/L $CaCO_3$	Sulfate mg/L	Chloride mg/L	SO_4: Cl	Ammonia mg/L
GWA4	Dry					
GWA7	700	< 1	1220	24.4	50.00	5.040
GWA8	7	34	4	2.4	1.67	0.045
GWA9	Dry					
GWA10	34	147	49	62.0	0.79	0.062
GWA11	69	280	33	49.7	0.66	0.019
GWA12	53	260	44	127.0	0.35	0.038
GWA13	58	131	99	49.6	2.00	0.081
GM13	20	76	23	59.1	0.39	0.170

Bore ID	Dissolved metals									
	As mg/L	Al mg/L	Cd mg/L	Cr mg/L	Cu mg/L	Pb mg/L	Mn mg/L	Ni mg/L	Zn mg/L	Fe mg/L
GWA4	Insufficient water to collect sample									
GWA7	1.89	30.5	< 0.0001	0.013	0.002	0.012	0.183	0.006	0.035	234
GWA8	<0.001	<0.01	< 0.0001	< 0.001	<0.001	< 0.001	< 0.001	< 0.001	< 0.005	< 0.05
GWA9	Dry									
GWA10	0.011	0.01	< 0.0001	< 0.001	0.001	< 0.001	0.006	< 0.001	< 0.005	0.90
GWA11	0.001	0.01	< 0.0001	< 0.001	0.005	< 0.001	0.028	0.002	0.009	< 0.05
GWA12	0.002	0.01	< 0.0001	< 0.001	0.007	< 0.001	0.094	0.003	0.009	< 0.05
GWA13	0.012	< 0.01	< 0.0001	< 0.001	<0.001	0.001	0.002	0.002	0.033	0.22
GM13	0.001	< 0.01	< 0.0001	< 0.001	<0.001	< 0.001	0.006	< 0.001	<0 .005	< 0.05

Groundwater sampling – 13 January 2009, all samples collected with a low flow bladder pump

Field results

Bore ID	Collection date	Standing water level mbgl	Sample depth mbgl	Field pH	Electrical conductivity mS/cm	Redox (Eh) mV
MW1	13/01/2009	2.65	3.65	6.62	1.498	−30
MW2	13/01/2009	3.20	4.20	5.21	2.067	82
MW3	13/01/2009	2.35	3.35	6.17	1.592	−30
MW4	13/01/2009	2.45	3.45	6.67	2.210	−37
MW5	13/01/2009	2.05	3.05	6.16	0.401	9
MW6	13/01/2009	1.87	2.87	6.43	1.339	−53
GWA9	13/01/2009	3.82	4.82	7.17	0.928	86
GM13	13/01/2009	3.44	4.44	6.81	0.885	54

Laboratory results

Bore ID	**Total acidity mg/L $CaCO_3$**	**Total alkalinity mg/L $CaCO_3$**	**Ca mg/L**	**Mg mg/L**	**Na mg/L**	**K mg/L**	**SO_4 mg/L**	**Cl mg/L**	**SO_4: Cl**
MW1	300	496	242	54	55	16	430	43	10.00
MW2	504	44	317	37	51	8	1270	45	28.22
MW3	208	305	249	9	36	7	498	54	9.22
MW4	567	538	456	47	53	29	890	45	19.78
MW5	102	145	71	2	16	3	49	22	2.23
MW6	80	241	253	21	48	8	427	44	9.70
GWA9	7	155	52	4	135	8	47	166	0.28
GM13	20	196	90	7	151	11	95	202	0.47

Bore ID	Dissolved metals									Total metals	
	As mg/L	Al mg/L	Cd mg/L	Cr mg/L	Mn mg/L	Ni mg/L	Se mg/L	Zn mg/L	Fe mg/L	Al mg/L	Fe mg/L
MW1	0.006	< 0.01	<0.0001	<0.005	0.138	0.003	<0.01	0.010	1.70	28.10	28.90
MW2	3.590	1.08	0.0002	0.007	0.109	0.014	<0.01	0.170	100.00	6.89	128.00
MW3	0.069	0.02	<0.0001	<0.005	0.207	0.003	<0.01	0.058	88.00	1.54	114.00
MW4	0.206	< 0.01	<0.0001	<0.005	0.250	<0.01	<0.01	0.050	8.79	< 0.01	8.79
MW5	0.377	0.38	<0.0001	0.004	0.020	0.004	<0.01	0.046	5.43	0.38	5.43
MW6	0.022	0.02	0.0001	<0.005	0.975	0.003	<0.01	0.039	29.90	3.05	12.10
GWA9	< 0.001	0.03	<0.0001	<0.001	0.008	0.001	<0.01	<0.005	< 0.05	10.60	7.98
GM13	0.001	0.06	<0.0001	<0.001	0.023	0.003	<0.01	0.010	< 0.05	9.24	7.48

Bore ID	Ammonia mg/L	Total P mg/L	Reactive P mg/L
MW1	0.49	0.61	0.02
MW2	6.6	0.49	0.24
MW3	1.81	0.34	0.02
MW4	3.96	0.9	0.02
MW5	1.87	1.4	0.05
MW6	0.64	1.48	0.01
GWA9	< 0.01	1.36	<0.01
GM13	0.02	1.46	<0.01

Groundwater sampling – 31 August and 2 September 2010, all samples collected with a low flow bladder pump

Field results

Bore ID	Collection date	Standing water level mbgl	Sample depth mbgl	Field pH	Electrical conductivity mS/cm	Redox (Eh) mV	DO %	DO mg/L
GWA7	31/08/2010	2.28	3.5	3.60	1.798	263	45.3	4.07
GWA8	1/09/2010	2.35	3.5	6.26	0.505	4	7.7	0.75
GWA9	31/08/2010	3.22	4.5	6.99	0.964	180	44.9	3.98
GWA10	31/08/2010	2.50	3.5	6.77	0.608	35	43.9	3.96
GWA11	2/09/2010	2.75	4.0	6.88	0.848	135	53.2	4.75
GWA12	2/09/2010	3.21	4.5	6.85	0.337	146	71.1	6.47
GM13	31/08/2010	2.93	4.0	6.74	0.658	119	11.9	1.05
GWA13	1/09/2010	3.37	4.5	6.76	0.497	145	37.8	3.26
MW3	1/09/2010	1.64	3.0	6.24	1.394	-35	2.9	0.28
MW6	1/09/2010	1.20	3.0	6.83	0.739	-53	55.8	5.43

Laboratory results

Bore ID	**Total acidity mg/L $CaCO_3$**	**Total alkalinity mg/L $CaCO_3$**	**Ca mg/L**	**Mg mg/L**	**Na mg/L**	**K mg/L**	**Sulfate mg/L**
GWA7	446	<1	242	8	15	3	821
GWA8	19	10	5	<1	12	2	3
GWA9	13	221	67	8	92	7	35
GWA10	15	235	102	3	12	2	17
GWA11	33	233	114	9	23	5	93
GWA12	28	130	22	2	18	2	3
GM13	13	101	47	5	79	8	28
GWA13	24	146	73	3	25	6	50
MW3	274	126	144	7	25	4	404
MW6	33	165	105	8	35	4	125

Bore ID	Dissolved metals											
	As mg/L	Al mg/L	B mg/L	Cd mg/L	Cr mg/L	Cu mg/L	Mn mg/L	Ni mg/L	Pb mg/L	Se mg/L	Zn mg/L	Fe mg/L
GWA7	1.56	1.49	0.12	0.0013	0.005	0.012	0.096	0.004	0.002	<0.01	0.086	102
GWA8	<0.001	<0.01	<0.05	<0.0001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.01	<0.005	<0.05
GWA9	0.009	6.28	<0.05	<0.0009	0.035	0.005	0.029	0.011	0.009	<0.01	0.010	6.22
GWA10	0.012	0.02	<0.05	0.0001	<0.001	0.001	0.024	<0.001	<0.001	<0.01	<0.005	0.71
GWA11	0.003	0.07	<0.05	<0.0001	<0.001	<0.001	0.079	0.001	<0.001	<0.01	<0.005	0.30
GWA12	0.001	0.14	<0.05	<0.0001	<0.001	<0.001	0.010	<0.001	<0.001	<0.01	<0.005	<0.05
GM13	0.001	0.10	0.05	0.0008	<0.001	0.001	0.009	0.001	<0.001	<0.01	<0.005	0.18
GWA13	0.015	0.45	<0.05	<0.0001	0.002	<0.001	0.028	<0.001	0.001	<0.01	<0.005	1.97
MW3	0.050	0.06	<0.05	<0.0001	<0.001	<0.001	0.238	0.003	<0.001	<0.01	0.008	124
MW6	0.022	0.06	<0.05	<0.0001	<0.001	<0.001	0.234	<0.001	<0.001	<0.01	<0.005	17

Bore ID	Total metals		
	As mg/L	Al mg/L	Fe mg/L
GWA7	1.78	1.84	107
GWA8	<0.001	0.23	0.35
GWA9	0.008	7.00	6.73
GWA10	0.031	2.44	5.43
GWA11	0.022	1.70	9.70
GWA12	0.01	14.8	9.18
GM13	0.004	8.06	7.62
GWA13	0.015	17.40	17.40
MW3	0.088	0.01	127
MW6	0.048	5.80	30.80

Bore ID	TKN mg/L	Total N mg/L	NOx mg/L	Total P mg/L
GWA7	21.2	22	0.79	0.72
GWA8	0.4	0.5	0.06	0.2
GWA9	0.7	2.2	1.53	0.24
GWA10	1.2	12.5	11.3	<0.01
GWA11	1.2	20.7	19.5	<0.1
GWA12	0.4	1.2	0.77	0.35
GM13	0.5	1.1	0.61	<0.01
GWA13	0.2	1.2	1.02	<0.1
MW3	3	3	0.04	<0.01
MW6	2	2	0.02	0.01

Appendix E – Ion balance – SGS investigations

Electrical balance for SGS investigations (June 2009 and June 2010). Electrical balance greater than 5% highlighted in yellow.

Date	**Sample ID**	**pH**	**Cl mg/L**	**PO_4 mg/L**	**SO_4 mg/L**	**Na mg/L**	**Mg mg/L**	**K mg/L**	**Ca mg/L**	**Total alk mg/L**	**HCO_3 mg/L**	**CO_3 mg/L**	**TDS**	**Ion balance %**
17/06/2009	GLP_Ec(1)	6.91	100	<0.005	86	70	11	9	70	140	170.66	0.07	440	1.7
24/07/2009	GLP_Ec(2)	6.89	60	<0.005	52	40	11	8	63	140	170.67	0.07	320	3.6
22/09/2009	GLP_Ec(3)	6.91	130	<0.005	110	90	13	9	82	150	182.85	0.07	540	1.9
11/09/2009	GLP_Ec(4)	6.84	250	<0.005	190	160	20	13	120	200	243.83	0.08	900	−0.3
27/01/2010	GLP_Ec(5)	6.87	100	<0.005	100	90	11	10	75	150	182.86	0.07	1220	5.5
24/03/2010	GLP_Ec(6)	6.97	80	0.012	70	60	10	9	67	140	170.64	0.08	390	3.6
18/05/2010	GLP_Ec(7)	6.94	80	<0.005	72	50	12	10	74	140	170.65	0.07	400	4.1
21/06/2010	GLP_Ec(8)	7.02	90	0.007	63	50	11	9	70	140	170.62	0.09	390	1.1
17/06/2009	GLP_Eb(1)	6.54	110	<0.005	410	80	30	8	270	350	426.85	0.07	1170	2.6
24/07/2009	GLP_Eb(2)	6.55	100	<0.005	420	80	26	9	260	320	390.26	0.07	1110	2.3
22/09/2009	GLP_Eb(3)	6.56	100	<0.005	380	80	13	8	220	340	414.65	0.07	1070	−5.4
11/09/2009	GLP_Eb(4)	6.55	120	0.005	460	90	31	6	250	350	426.85	0.07	1240	−2.2
27/01/2010	GLP_Eb(5)	6.61	120	<0.005	480	90	31	6	250	330	402.44	0.08	520	−2.3
24/03/2010	GLP_Eb(6)	6.72	110	0.023	480	80	33	7	290	350	390.19	0.10	1260	3.3
18/05/2010	GLP_Eb(7)	6.70	100	0.025	420	80	29	10	250	320	390.20	0.10	1140	1.7
21/06/2010	GLP_Eb(8)	6.75	90	0.011	430	70	27	9	260	310	377.99	0.10	1100	2.2
17/06/2009	GLP_Ea(1)	6.90	160	<0.005	180	100	17	9	130	330	402.28	0.16	750	−8.8
24/07/2009	GLP_Ea(2)	6.89	180	0.008	230	120	20	9	160	850	1036.19	0.40	890	−28.1

Date	Sample ID	pH	Cl mg/L	PO$_4$ mg/L	SO$_4$ mg/L	Na mg/L	Mg mg/L	K mg/L	Ca mg/L	Total alk mg/L	HCO$_3$ mg/L	CO$_3$ mg/L	TDS	Ion balance %
22/09/2009	GLP_Ea(3)	6.84	180	<0.005	260	130	25	10	160	270	329.17	0.11	910	0.2
11/09/2009	GLP_Ea(4)	6.75	200	0.008	370	150	30	11	220	680	829.13	0.23	1210	−14.2
27/01/2010	GLP_Ea(5)	6.74	200	<0.005	430	160	33	10	240	350	426.76	0.12	1330	0.7
24/03/2010	GLP_Ea(6)	6.95	170	0.008	310	140	27	10	200	330	402.24	0.18	1120	1.9
18/05/2010	GLP_Ea(7)	7.01	160	<0.005	240	120	23	9	160	270	329.06	0.17	910	1.4
21/06/2010	GLP_Ea(8)	7.09	170	0.008	250	120	21	9	170	270	328.99	0.20	900	0.8
24/07/2009	GLP_Wc(2)	6.10	110	0.008	220	60	7	2	130	52	63.43	0.00	690	5.4
22/09/2009	GLP_Wc(3)	6.11	40	0.009	48	20	2	<1.	36	61	74.41	0.00	170	−8.4
11/09/2009	GLP_Wc(4)	6.36	40	0.023	58	30	3	3	46	77	93.92	0.01	230	0.6
17/06/2009	GLP_Wb(1)	7.09	40	<0.005	150	20	21	13	150	310	377.73	0.23	650	−0.2
24/07/2009	GLP_Wb(2)	6.98	40	0.032	200	20	20	12	160	250	304.71	0.14	680	2.4
22/09/2009	GLP_Wb(3)	6.91	40	0.017	180	30	16	11	130	260	316.94	0.13	550	−3.5
11/09/2009	GLP_Wb(4)	6.85	30	0.033	150	20	16	11	120	260	316.98	0.11	510	−4.0
27/01/2010	GLP_Wb(5)	6.99	50	0.016	150	30	15	11	120	230	280.33	0.14	540	−1.8
24/03/2010	GLP_Wb(6)	7.11	40	0.085	190	30	19	12	150	240	292.42	0.19	640	3.8
18/05/2010	GLP_Wb(7)	7.28	50	0.023	190	30	19	13	160	250	304.42	0.29	610	3.8
21/06/2010	GLP_Wb(8)	7.33	40	0.020	170	30	17	12	140	240	292.17	0.31	560	2.7
17/06/2009	GLP_Wa(1)	6.90	150	<0.005	35	100	13	10	87	230	280.38	0.11	540	2.3
24/07/2009	GLP_Wa(2)	6.96	170	0.005	36	110	13	10	82	240	292.53	0.13	590	−0.7
22/09/2009	GLP_Wa(3)	6.84	150	<0.005	44	100	13	9	86	370	451.09	0.15	570	−11.6
11/09/2009	GLP_Wb(4)	7.08	170	0.020	34	110	13	11	73	230	280.26	0.17	540	−1.6

Date	Sample ID	pH	Cl mg/L	PO_4 mg/L	SO_4 mg/L	Na mg/L	Mg mg/L	K mg/L	Ca mg/L	Total alk mg/L	HCO_3 mg/L	CO_3 mg/L	TDS	Ion balance %
27/01/2010	GLP_Wb(5)	6.96	170	0.012	31	110	13	11	76	210	255.97	0.12	570	1.5
24/03/2010	GLP_Wa(6)	7.20	170	0.100	29	110	14	11	79	210	255.79	0.20	560	2.8
18/05/2010	GLP_Wa(7)	7.19	140	0.015	39	100	14	10	93	240	292.35	0.22	560	4.2
21/06/2010	GLP_Wa(8)	7.34	170	0.093	30	110	13	11	78	220	267.81	0.29	570	1.1

Appendix F – SGS investigation – Groundwater elevations

SGS investigation (Department of Water) – June 2009 – September 2010

Bore ID	GLP_ Wa	GLP_ Wb	GLP_ Wc	Hydraulic gradient	Hydraulic gradient	GLP_ Ea	GLP_ Eb	GLP_ Ec	Hydraulic gradient	Hydraulic gradient	Surface water present	Date lake level measured	Lake level EWP 465
RL TOC m AHD	7.10	7.08	7.10			7.96	8.00	8.00					
Sample date	m AHD	m AHD	m AHD	Wc/Wb	Wb/Wa	m AHD	m AHD	m AHD	Ec/Eb	Eb/Ea			m AHD
17/06/09	3.62	3.37	4.12	1.22	0.93	4.84	4.79	4.90	1.02	0.99	No	12/06/10	5.00*
24/07/09	4.19	3.87	5.21	1.35	0.92	5.56	5.55	5.38	0.97	1.00	Yes	27/07/10	5.63
24/08/09	4.46	4.15	5.64	1.36	0.93	5.80	5.82	5.95	1.02	1.00	Yes	24/08/10	5.97
22/09/09	4.65	4.36	5.81	1.33	0.94	5.94	5.95	6.06	1.02	1.00	Yes	22/09/10	6.05
21/10/09	4.64	4.36	5.68	1.30	0.94	5.87	5.86	5.98	1.02	1.00	Yes	21/10/10	5.87
9/11/09	4.52	4.23	5.47	1.29	0.94	5.77	5.75	5.86	1.02	1.00	Yes	9/11/10	5.69
25/11/09			5.35					5.81			Yes	20/11/10	5.67
14/12/09	4.33	4.05	5.06		0.93	5.54	5.53	5.63	1.02	1.00	Yes	14/12/10	5.40
27/01/10	3.93	3.65			0.93	5.08	5.05	5.16	1.02	0.99	No	22/01/10	4.86*
17/02/10	3.71	3.46			0.93	4.90	4.87	4.97	1.02	0.99	No*	NM	-
24/03/10	3.62	3.33			0.92	4.95	4.85	4.93	1.02	0.98	No*	NM	-
29/04/10	3.56	3.29	3.86	1.17	0.92	4.91	4.86	4.97	1.02	0.99	No*	NM	-

Bore ID	GLP_ Wa	GLP_ Wb	GLP_ Wc	Hydraulic gradient	Hydraulic gradient	GLP_ Ea	GLP_ Eb	GLP_ Ec	Hydraulic gradient	Hydraulic gradient	Surface water present	Date lake level measured	Lake level EWP 465
RL TOC m AHD	7.10	7.08	7.10			7.96	8.00	8.00					
Sample date	**m AHD**	**m AHD**	**m AHD**	**Wc/Wb**	**Wb/Wa**	**m AHD**	**m AHD**	**m AHD**	**Ec/Eb**	**Eb/Ea**			**m AHD**
18/05/10	3.56	3.26			0.92	4.94	4.88	4.98	1.02	0.99	No	12/05/10	5.00*
21/06/10	3.89	3.56			0.92	5.31	5.26	5.36	1.02	0.99	No	9/06/10	5.00*
2/09/10 3/09/10	4.55	4.21	4.42	1.05	0.93	5.88	5.95	6.10	1.03	1.01	Yes	3/09/10	6.09

** No water level measured*

NM: Not measured

EWP 465 – Lake level – dry at ≤ 5.0 m AHD

RL: Reduced level

TOC: Top of casing

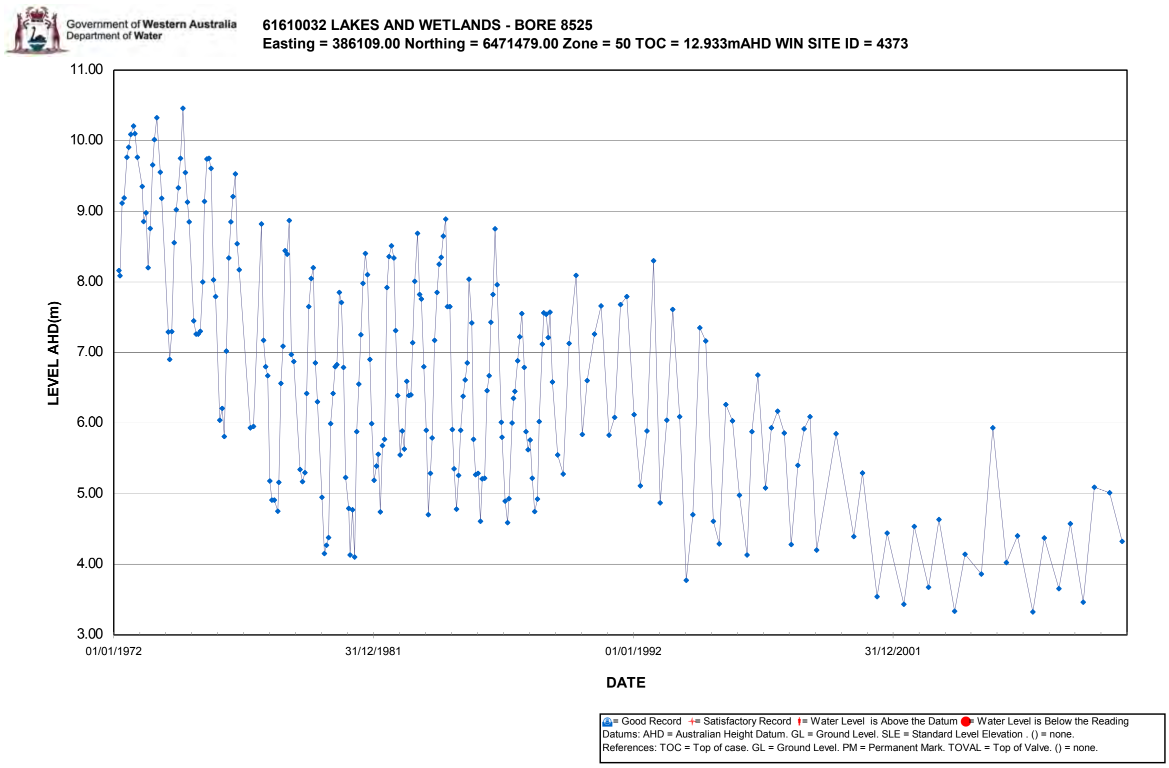
Groundwater elevations measured between 2006 and 2010, Gwelup soil and groundwater acidity investigation, (DEC)

Bore ID	RL TOC m AHD	SWL m AHD Mar-06	SWL m AHD Oct-07	SWL m AHD May-08	SWL m AHD Jan-09	SWL m AHD Sep-10
GWA1	11.532	7.17				
GWA2	8.010	6.73				
GWA3	11.490	7.04				
GWA4	10.825	6.86		6.41		
GWA5	7.924	6.96				
GWA6	8.832	7.67	5.30			
GWA7	10.201		7.35	6.40		7.92
GWA8	10.164		7.25	6.41		
GWA9a	11.024		7.30			
GWA9b	11.024				7.20	7.80
GWA10	10.531		7.32	6.43		8.03
GWA11	10.509		7.27	6.37		7.76
GWA12	10.944		6.96	6.13		7.73
GWA13	10.776		6.95	6.04		7.41
GM13(a)	10.159	6.85	7.11	6.31		
GM13b	10.627				7.19	7.70
MW1	9.977				7.33	
MW2	10.383				7.18	
MW3	9.490				7.14	7.85
MW4	9.653				7.20	
MW5	9.208				7.16	
MW6	9.183				7.31	7.98

Note: TOC – top of casing

SWL – standing water level

Appendix G – Hydrographs



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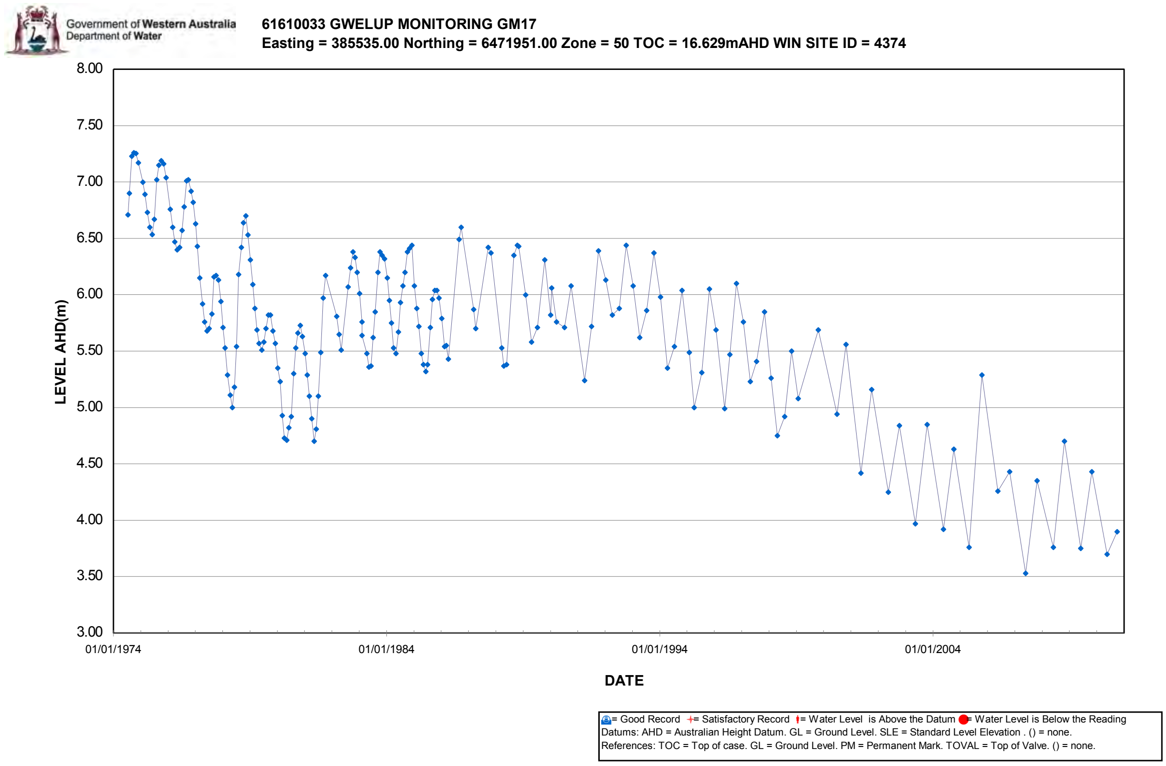

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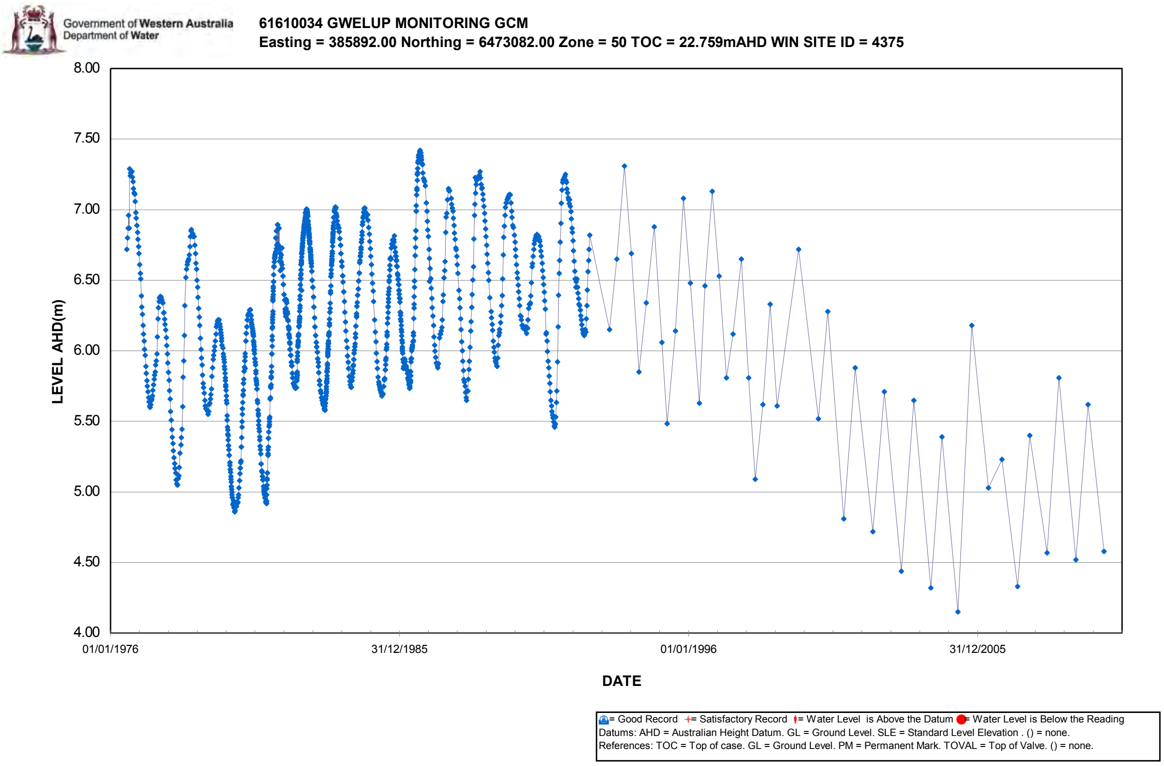

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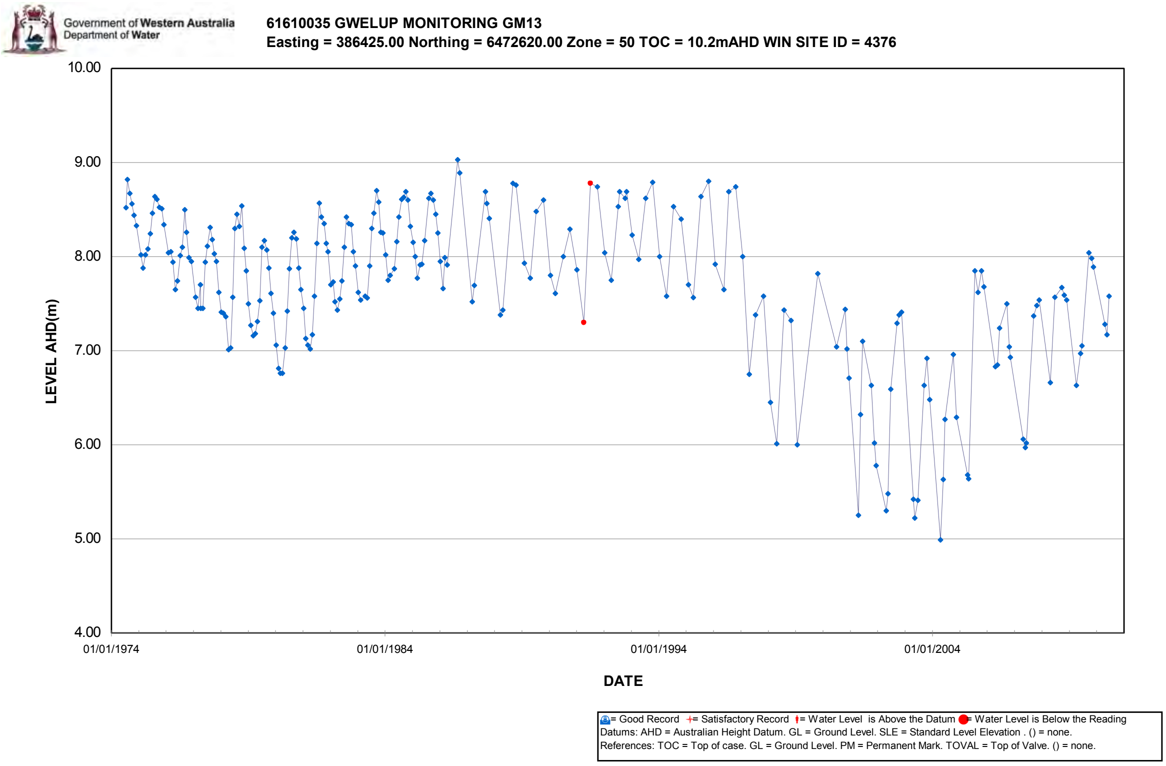
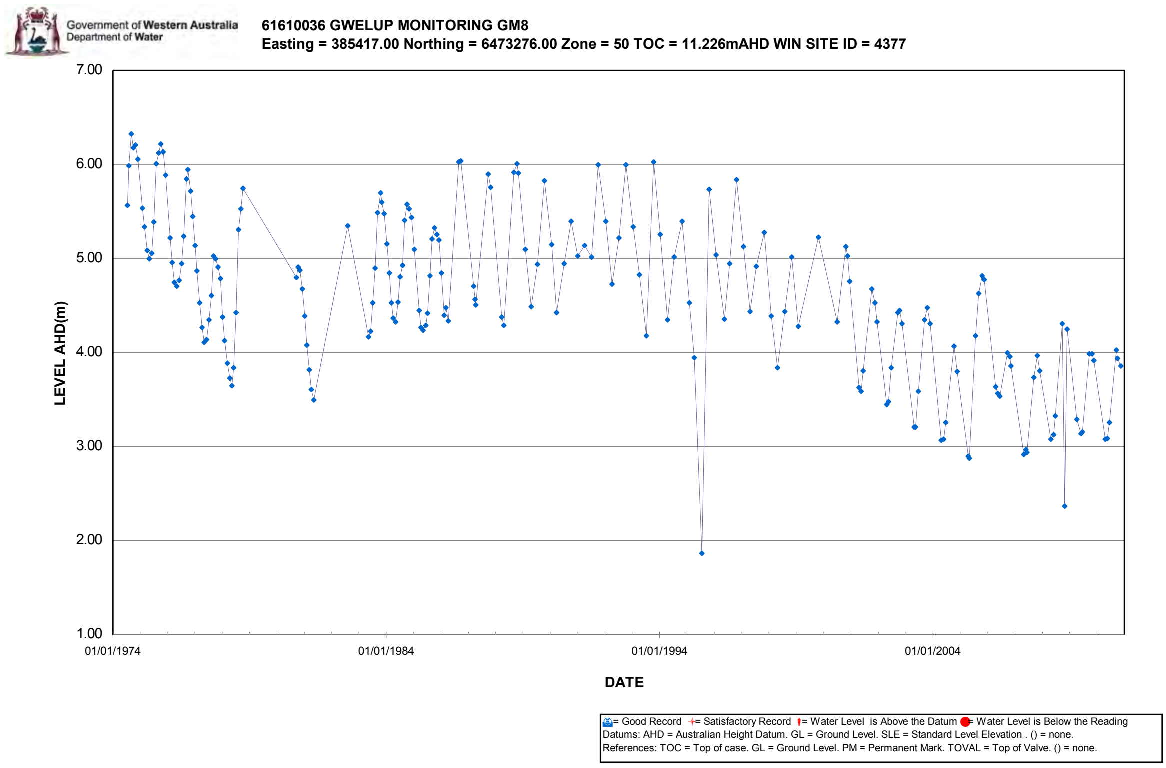
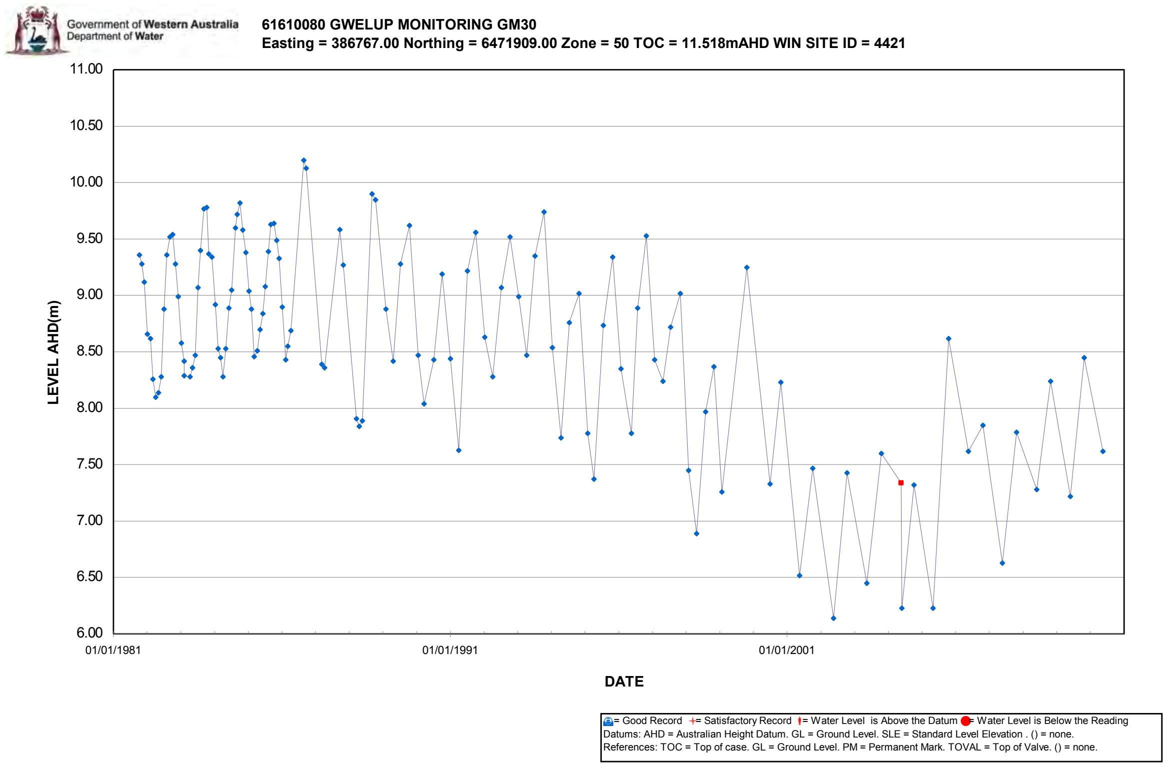


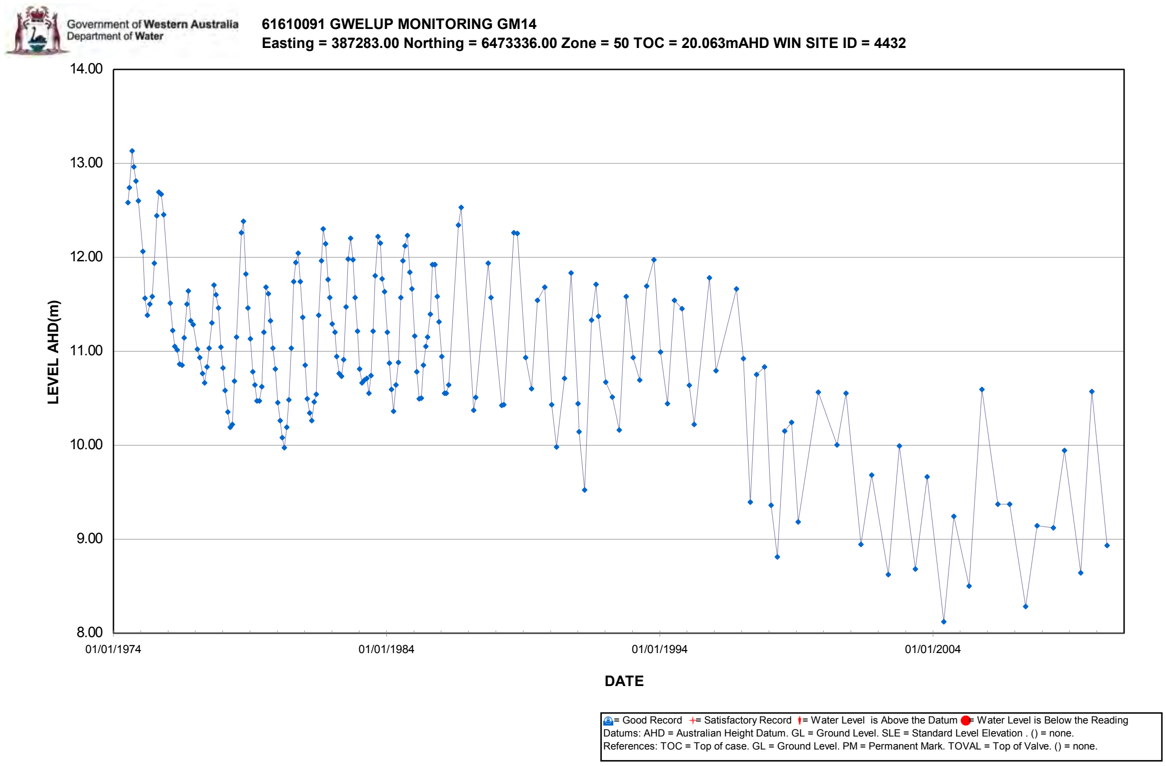
WIN ID 4376 (AWRC name: GM13)



WIN ID 4377 (AWRC name: GM8)



WIN ID 4421 (AWRC name: GM30)



WIN ID 4432 (AWRC name: GM14)

Shortened forms

AASS	Actual acid sulfate soils
AHD	Australian height datum
ALS	Laboratory service of the ALS Environmental company
ANZECC	Australia and New Zealand Environment and Conservation Council
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
ASS	acid sulfate soils
AWRC	Australian Water Resources Council
CRS	Chromium reducible sulfur suite of analyses
DEC	Department of Environment and Conservation
DO	Dissolved oxygen
DoH	Department of Health
DOC	Dissolved organic carbon
DON	Dissolved organic nitrogen
EC	Electrical conductivity
Eh	Redox potential
EIL	Ecological investigation level
EPA	Environmental protection agency
EPP	Environmental protection policy
EPHC	Environmental Protection and Heritage Council
ESC	Environmental sensitivity classification
EWR	Ecological water requirement
FRP	Filterable reactive phosphorus
FWG	Fresh water guideline

GDE	Groundwater-dependent ecosystem
GSS	Gnangara Sustainability Strategy
IOCI	Indian Ocean Climate Initiative
IWSS	Integrated Water Supply Scheme
LTIW	Long term irrigation water guideline
mbgl	Metres below ground level
NMI	National Measurement Institute
NO_X	A measure of nitrite (NO_2) and nitrate (NO_3)
NRMMC	Natural Resource Management Ministerial Council
ORP	Oxidation reduction potential
PASS	Potential acid sulfate soils
pH_F	Field pH
pH_{FOX}	Field peroxide pH
pH_{KCl}	Laboratory pH (with potassium chloride)
pH_{OX}	Laboratory peroxide pH
S_{Cr}	Chromium reducible sulfur measurement
SGS	Shallow groundwater systems
SPOCAS	Suspension peroxide oxidation combined acidity and sulfur suite of analyses
SRP	Soluble reactive phosphorus
TAA	Titratable actual acidity
TDS	Total dissolved solids
TKN	Total Kjeldahl nitrogen
TN	Total nitrogen

TON	Total oxidised nitrogen
TP	Total phosphorus
TPA	Titratable peroxide acidity
TSA	Titratable sulfidic acidity
TSS	Total suspended solids
WIN	Department of Water’s Water Information System
WRC	Water and Rivers Commission

Glossary

Abstraction	The permanent or temporary withdrawal of water from any source of supply, so that it is no longer part of the resources of the locality.
Acid buffering capacity	A measure of the resistance to changes in pH following the addition of an acid. Also called ‘pH buffering capacity’ or just ‘buffering capacity’.
Acid sulfate soils	Naturally occurring, these are soils that contain significant quantities of reduced sulfur (pyrite and other sulfides). When these soils are disturbed, the reduced sulfur is oxidised resulting in the release of acidity and often toxic metals.
Acidification	The process by which soil, or water becomes more acidic (decreasing pH and/or increasing acidity concentrations).
AHD	Australian height datum
Alkalinity	A measure of a solution’s ability to resist changes in pH due to the addition of an acid. In natural waters this usually relates to the amount of bicarbonate, carbonate and hydroxide compounds present in the water.
Allocation limit	Annual volume of water set aside for use from a water resource.
Aquifer	A geological formation or group of formations able to receive, store and/or transmit large amounts of water.
Biodiversity	Biological diversity or the variety of organisms, including species themselves, genetic diversity and the assemblages they form (communities and ecosystems). Sometimes biodiversity includes the variety of ecological processes within those communities and ecosystems.
Bore	A narrow, normally vertical hole drilled into a geological formation to monitor or withdraw groundwater from an aquifer (see also well).
Buffer	A solution which resists changes in pH when a small amount of strong acid or base is added.

Buffering capacity	*See* acid buffering capacity.
Confined aquifer	A permeable bed saturated with water and lying between an upper and a lower confining layer of low permeability, the hydraulic head being higher than the upper surface of the aquifer.
Confining layer	Sedimentary bed of very low hydraulic conductivity.
Conformably	Sediments deposited in a continuous sequence without a break.
Contaminants	A substance that is either present in an environment where it does not belong or is present at levels that might cause harmful effects to humans or the environment.
Correlation	Indicates the strength and direction of the linear relationship between two random variables.
Cretaceous	Final period of Mesozoic era, 65–144 million years ago.
Decline	The difference between the elevation of the initial watertable and its position after a decrease in recharge (i.e. rainfall).
Dewatering	Short-term abstraction of groundwater to lower the watertable and permit the excavation of 'dry' sediment.
Discharge	The water that moves from the groundwater to the ground surface or above, such as a spring. This includes water that seeps onto the ground surface, evaporation from unsaturated soil and water extracted from groundwater by plants (see evapotranspiration) or engineering works (see groundwater pumping).
Dissolved oxygen	The concentration of oxygen dissolved in water, normally measured in milligrams per litre (mg/L).
Drawdown	The difference between the elevation of the initial piezometric surface and its position after pumping or gravitational drainage.
Ecological values	The natural ecological processes that occur within water-dependent ecosystems and the biodiversity of these systems.

Ecological water requirement	The water regime needed to maintain the ecological values (including assets, functions and processes) of water-dependent ecosystems at a low level of risk.
Ecosystem	A community or assemblage of communities of organisms, interacting with one another, and the specific environment in which they live and with which they also interact, e.g. lake, to include all the biological, chemical and physical resources and the interrelationships and dependencies that occur between those resources.
Eutrophic	An excess of nutrients (nitrogen and phosphorus) in an ecosystem, often resulting in excessive primary production.
Evapotranspiration	The combined loss of water by evaporation and transpiration. Includes water evaporated from the soil surface and water transpired by plants (Water and Rivers Commission 2001a).
Fault	A fracture in rocks or sediments along which there has been an observable displacement.
Feldspar	Group of mineral silicates.
Formation	A group of rocks or sediments that have certain characteristics in common, were deposited about the same geological period, and that constitute a convenient unit for description.
Geomorphic	Pertaining or related to the form of the earth or its surface features.
Glauconitic	Of the mineral glauconite, which is an iron potassium phyllosilicate (mica group) mineral of characteristic green colour.
Groundwater	Water that occupies the pores within the rock or soil profile.
Groundwater area	Are the boundaries that are proclaimed under the *Rights in Water and Irrigation Act 1914* and used for water allocation planning and management.
Groundwater level	An imaginary surface representing the total head of groundwater. Defined by piezometer readings.

Groundwater mound	A mound-shape formation of the watertable that results from rainwater trickling down into the open space between particles in an elevated area of deep sand or other porous material. Groundwater will move slowly away from the central area to discharge into wetlands, rivers and oceans.
Groundwater recharge	The rate at which infiltration water reaches the watertable.
Groundwater-dependent ecosystem	An ecosystem that is dependent on groundwater for its existence and health.
Head	The height of the free surface of a body of water above a given sub-surface point.
Hydraulic	Pertaining to water motion.
Hydraulic gradient	The rate of change of total head per unit distance of flow at a given point and in a given direction.
Hydrogeology	The hydrological and geological science concerned with the occurrence, distribution, quality and movement of groundwater, especially relating to the distribution of aquifers, groundwater flow and groundwater quality.
Hydrograph	A graph that shows the height of a water surface above an established datum plane for level, flow, velocity or other property of water with respect to time.
Inter-dunal	Between dunes.
Ion	An atom which has lost or gained electrons and therefore carries an electrical charge.
Leach	Remove soluble matter by percolation of water.
Limit of reporting	The lower limit of reliability given by the laboratory responsible for carrying out the analysis. Greater than the limit of detection.
Mesozoic	An era of geological time between 250 to 65 million years ago. It included the Triassic, Jurassic and Cretaceous periods.
Neutralisation	The chemical reaction in which an acid and a base react to produce salt and water (H_2O).

Organism	Is a living system. In at least some form, all organisms are capable response to stimuli, reproduction, growth and development, and maintenance of homeostasis as a stable whole.
Oxidation	A process that results in the loss of electrons from a chemical species, accompanied by an increase in oxidation state. This process does not necessarily require the presence of oxygen.
pH	The negative logarithm of the concentration of hydrogen ions.
Precipitate	The solid formed when two ionic solutions are mixed together to produce an insoluble product.
Quaternary	Relating to the most recent period in the Cainozoic era, from two million years ago to present.
Recharge	Water that infiltrates into the soil to replenish an aquifer.
Redox	Chemical reactions leading to changes in the oxidation sate of atoms.
Redox potential	In aqueous solutions, the reduction potential is the tendency of the solution to either gain or lose electrons and is measured in volts (V), millivolts (mV), or Eh (1 Eh = 1 mV). Because the absolute potentials are difficult to accurately measure, reduction potentials are defined relative to the standard hydrogen electrode which is arbitrarily given potential of 0.00 V.
Reduction	A process resulting in the gain of electrons by a chemical species accompanied by a decrease in oxidation state.
Salinity	A measure of the concentration of total dissolved solids in water 0–500 mg/L; fresh 500–1500 mg/L; fresh-to-marginal 1500–3000 mg/L; brackish >3000 mg/L; saline.
Screen	A special form of bore liner used to stabilise the aquifer or gravel pack, while allowing the flow of water through the bore into the casing and permitting the development of the screened formation by an appropriate process.

Sulfate reduction	In the aquatic environment, the microbially catalysed process which converts sulfate to sulfide.
Surficial	Pertaining to the surface.
Tertiary	The first period of the Cainozoic era; 2 to 65 million years ago.
Toxicity	The degree to which a substance is able to damage an exposed organism.
Transpiration	The loss of water vapour from a plant, mainly through the leaves.
Unconfined aquifer	A permeable bed only partially filled with water and overlying a relatively impermeable layer. Its upper boundary is formed by a free watertable or phreatic level under atmospheric pressure.
Water quality guidelines	Values or ranges of acceptable or unacceptable levels of a chemical, beyond which management response may be triggered.
Watertable	The surface of a body of unconfined groundwater at which the pressure is equal to that of the atmosphere.

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