

Acid Sulfate Soils Research Program

Quantification of Acidity Flux Rates
to the Lower Murray Lakes

Report 2 | Part 1 of 3: Main Report | April 2010



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Cover image

Currency Creek, June 2009 (DENR 2009)

Quantification of Acidity Flux Rates to the Lower Murray Lakes

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EARTH SYSTEMS
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Executive Summary

The Department of Environment and Natural Resources (DENR), South Australia, commissioned Earth Systems Consulting Pty Ltd (Earth Systems) to quantify acidity flux rates from acid sulfate soils (ASS) into the surface waters of Lake Alexandrina and Lake Albert (Lower Murray Lakes).

The majority of soils/sediments around the Lower Murray Lakes either contain sulfuric acid (sulfuric material) and/or have the potential to form sulfuric acid upon exposure of sulfidic material to atmospheric oxygen. The lowering of lake water levels increases the volume of sulfidic material that is exposed to atmospheric oxygen. As this material is exposed to oxygen it generates acid and metalliferous drainage (AMD) which has the potential to result in ecological, human health and water quality issues.

Key objectives of this study are to: (i) develop an improved understanding of acidity generation, neutralisation and groundwater transport processes within the lake sediments, (ii) quantify acidity flux rates during wetting events by assessing the hydrogeology and hydrogeochemistry of lake sediments, and (iii) provide recommendations for future management of the Lower Murray Lakes.

The study included design, establishment and implementation of a laboratory testwork program to measure sulfide oxidation rates, and a field monitoring program to collect geological, geophysical, hydrogeological and hydrogeochemical data at selected high risk locations. Laboratory and field data were integrated within a customised finite element acidity generation model. The model was verified with existing data from Currency Creek, and used to predict future acidity flux rates associated with forecast water levels in Lake Albert and Lake Alexandrina from 1 September 2009 to 30 June 2011. This report is based on the results of the laboratory testwork program, field monitoring results collected over a 3 month period and associated modelling outputs.

Geological information gathered from drilling, shallow pitting and transient electromagnetic (TEM) surveys identified a broadly consistent regional near-surface stratigraphy across both lakes. The majority of the areas examined to date contain a thin veneer of lake sediments (1-3 m thick) overlying a calcrete/silcrete-capped Bridgewater Formation limestone. The lake sediments generally comprise an uppermost 0.7-1.3 m thick layer of quartz-rich sand, which overlies a 0.2-1.0 m thick, occasionally calcareous clay layer. An additional quartz-rich sand layer may be found beneath the clay layer and immediately above the Bridgewater Formation.

The sandy (sulfide-bearing) sediments around the periphery of both lakes are considered to represent the most significant short-term ASS risk facing the lakes. This is primarily based on the understanding that sandy sediments will represent the majority of ASS exposed in both lakes over the next 1-2 years. While the clays typically contain a higher proportion of sulfides and are more extensive than the sands, they do not pose an immediate risk to lake water quality while they remain saturated. Desaturated upper sand horizons can be expected to retard the desaturation and oxidation of underlying clay layers. Hence, clay layers that are not capped by sand are only likely to become an AMD risk when falls in surface water levels expose them. Furthermore, the tendency for clays to maintain significant moisture, even when partially desiccated, is predicted to limit their potential for acidity generation in the short-term.

At this stage, emphasis is therefore placed on quantifying the risk associated with desaturation and oxidation of the sulfidic sandy sediments fringing both lakes. This is supported by observations at Currency Creek, where drying of the creek bed in 2008-09 resulted in the sandy sediments on the creek margin being exposed for longer and desaturating much more rapidly (when exposed) than the clays. Thus, the majority of acidity flux to Currency Creek in May 2009 was attributed to acidity generation within the sandy sediments. Clay-rich ASS in Lake Albert and Lake Alexandrina could, however, become an increasing concern if lake water levels dropped sufficiently to expose significant volumes of these sediments to oxidising conditions over extended periods of time.

Hydraulic conductivity values range from 0.09 to >30 m/day for lake sediment sands and 0.5 to >30 m/day for the Bridgewater Formation calcareous sands. Groundwater levels rise rapidly in response to significant rainfall events (temporary rises of 30 cm are typical in response to 10-15 mm rainfall events), but fall very rapidly towards pre-event levels over subsequent days. The significant rises in groundwater level relative to actual rainfall depth are believed to be due to the addition of infiltrating water to near-saturated sediments. The subsequent rapid drop in groundwater levels after rainfall is believed to be the result of evapotranspiration, due to a lack of evidence for lateral groundwater migration during the monitoring period (see below). In addition to rainfall and evapotranspiration, there has been some evidence of seiche affecting groundwater levels at the Windmill location (only) over the last 3 months. Overall, however, the hydraulic gradients have been relatively small at all monitoring locations, ranging from 0.001-0.002 in mid-November 2009. Thus, despite some of the high hydraulic conductivity values measured, there has been limited potential for groundwater migration towards the lake water. Clearer trends in groundwater levels are expected to emerge over the next few months as rainfall becomes less significant and lake water levels progressively recede. The rate of groundwater discharge from exposed shoreline sediments to the lake water has the potential to increase significantly as surface water levels decline and hydraulic gradients increase during rainfall events.

A good correlation exists between groundwater levels and moisture content profiles in the lake sediments. Sediment moisture data shows that a 0.3 m thick zone of essentially saturated sandy sediment, with a moisture content of around 40-50 vol% H₂O, exists above the piezometric surface at all sites examined. Above this zone, moisture contents progressively decrease with proximity to the ground surface, over a depth interval of around 0.3 m. The near-surface sediments are characterised by moisture contents of 2-6 vol% H₂O. From August to November 2009, groundwater levels have decreased by 0.5-0.9 m at all monitoring sites nearest the lake shores. The sediments have remained largely saturated at depths below 0.3 m during this time. However, it is expected that further lowering of groundwater levels over the next few months will result in a progressive desaturation of these sediments. Further monitoring data will assist in understanding the rate of sediment desaturation in response to falling groundwater levels.

Where the rainfall intensity or duration is sufficient to affect sediment moisture content, peak moisture values in the upper sediments are achieved within several hours. Vertical migration of infiltrating rainwater through the upper 40-50 cm occurs within approximately 1 day of the onset of a significant rainfall event.

Sulfide oxidation rates have been measured for both sandy and clay-rich material as a function of moisture content. The peak sulfide oxidation rate was around 1.2 wt% pyrite per day (1.2 wt% of available pyrite that is exposed to atmospheric oxygen will be converted to sulfuric acid pyrite per day) in sandy sediments (at 15 wt% water content) and 0.8 wt% pyrite per day in clays (at 23 wt% water content). At these rates, the majority of available pyrite would oxidise to produce H₂SO₄ acidity within approximately 3-4 months. Overall, the laboratory measured sulfide oxidation rates for sands and clays of the Lower Murray Lakes compare well with oxidation rates for ASS reported in the literature (and are consistent with field observations).

Despite water levels decreasing to -0.45 m AHD in Lake Albert and -0.75 m AHD in Lake Alexandrina in late November 2009 (based on DLWBC data), there has been no obvious impact on surface water quality (pH or alkalinity). This indicates that either little acidity has been generated, little acidity has been released from exposed sediments prior to November 2009, or any acidity that has been released has been neutralised within the lake sediments or surface water.

During the field monitoring program, groundwater level data indicated that the lake levels decreased from 0.1 m AHD in August 2009 to around -0.65 m AHD in November 2009 in Lake Albert, and remained around -0.9 m AHD in Lake Alexandrina. Thus, the lake levels have generally remained higher than the previously recorded minimum levels -0.59 m AHD in Lake Albert (March 2009) and -1.0 m AHD in Lake Alexandrina (April 2009).

The field monitoring data collected from August to November 2009 indicates that:

- Prior to the commencement of monitoring, some localised acidity generation had occurred within the upper profile lake sediments, as indicated by acidic groundwater observed in some piezometers (3 sites at Campbell Park and 2 sites at Point Sturt).
- There has been no evidence of significant additional acidity generation within the lake sediments during the monitoring program.
- Acidity generated within the upper lake sediments has migrated downward from sandy layers in the unsaturated zone to the groundwater via rainwater infiltration. However, there has been only limited vertical mixing/diffusion within the sediment profile below the water table.
- There has been no significant lateral migration of acidity from the sediments towards the lake water, based on relatively consistent water quality over time (at each site) despite significant chemical variations relative to other sites on the same transect. This is attributed to relatively low hydraulic gradients over the last 3 months, as well as the significant near-surface evapotranspiration water losses.
- Groundwater chemistry data shows that some degree of in-situ carbonate dissolution (ANC consumption) has occurred at all sites. At Campbell Park this has clearly been related to acidity generation. However, ANC consumption has been insufficient to counter the acidity in groundwater at Campbell Park. This is despite indications that sandy lake sediments generally have negative Net Acid Producing Potential (NAPP) values.
- There is evidence of sulfide precipitation (associated with bacterial sulfate reduction) within the upper sediments affected by acidity generation at Campbell Park (although not Point Sturt) based on the progressive increases in pH and Cl:SO₄ ratios observed over the last 3 months.

The minimal change in groundwater quality at any one site over time, combined with the variation in water quality along each monitoring transect, indicates there has been no significant lateral migration of groundwater, and therefore no significant acidity flux to the lakes over the 3 month monitoring period. This would appear to be due to relatively small changes in lake levels (relative to previously recorded minimum levels) and substantial rainfall inputs over the last 3 months.

The risk of acidity generation within the lake sediments could increase significantly as lake levels are forecast to decrease to unprecedented levels (around -1.0 m AHD in Lake Albert and -2.0 m AHD in Lake Alexandrina) over the next 1-2 years. The increased risk would result from:

- Exposure of larger volumes of ASS.
- Increased sulfide-sulfur content with depth in exposed sediments.
- Increased rate of groundwater flow through sandy sediments due to greater hydraulic gradients to lake water.

Key risk factors for acidity flux through the sandy sediments and subsequent impacts on surface water quality in the lakes will be rainfall and the magnitude of the hydraulic gradient towards the lake water. Further decreases in groundwater levels are predicted over the next 3 months, but without sufficient rainfall to recharge the sediments, relatively little acidity flux to the lakes is anticipated during this time. A significant increase in the hydraulic gradient is most likely to occur associated with an autumnal flush event, when lake levels are substantially lower than at present. The onset of autumnal rains is expected to be the highest risk period for the acidity fluxes to the lakes from the sandy sediments that have been unsaturated over the preceding summer. The degree of impact of acidity fluxes on lake water quality will depend on the magnitude of each flux (tonnes H₂SO₄ acidity) relative to the available buffering capacity in each lake. Models have been developed in this report to quantify the rate of acidity generation within the lake sediments and the likely timing and duration (and therefore magnitude) of acidity flux events.

Preliminary modelling results indicate that, without careful management intervention, significant acidity flux to the lakes is likely to commence during the autumnal flush of 2010. Key modelling outputs are summarised below. Note that the lack of evidence for significant acidity flux to the lakes to date is consistent with the modelling outputs described below.

In Lake Albert, the total potential acidity generation over 22 months (from September 1st, 2009) is estimated at ~50,000 tonnes H₂SO₄ assuming no ANC consumption, or ~38,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. The acidity generation rate is expected to drop significantly after 9 months due to pyrite depletion in the unsaturated zone.

In Lake Alexandrina, total potential acidity generation over 22 months (from September 1st, 2009) is estimated at ~180,000 tonnes H₂SO₄ assuming no ANC consumption, or ~115,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. Prior to the first autumnal flush, the total acidity generation is estimated at ~40,000 tonnes H₂SO₄ assuming no ANC consumption, or ~28,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~20-25% of the acidity load generated after 22 months. While there is likely to be sufficient soluble alkalinity in the lake water to neutralise the acidity from this initial flush, less alkalinity will remain for future buffering. The rate of acidity generation could increase substantially after 15 months due to water levels decreasing to -2.0 m AHD.

The acidity generation model is particularly sensitive to hydraulic gradients within the sands, the distribution and concentration of sulfides in the upper sandy sediments, and the extent of ANC consumption within the sandy sediments.

The likely rate and duration of acidity release (flux) events in Lake Albert and Lake Alexandrina has been estimated for a range of lake water level, hydraulic conductivity and acidity concentration scenarios, based on hydrogeological modelling conducted by Coletti and Hipsey (2010).

In Lake Albert, the acidity flux event associated with the first autumnal flush could have a duration ranging from 2-3 months (77 days) to several years (10,000 days), while the subsequent acidity flux event is estimated to occur in approximately half this time, due to the correspondingly lower tonnage of acidity generation.

In Lake Alexandrina, the acidity flux event associated with the first autumnal flush could have a duration ranging from 1-2 months (36 days) to several years (4,700 days), while the subsequent acidity flux event is estimated to occur over approximately three times this duration, due to the correspondingly higher tonnage of acidity generation.

Based on the observed acidity flux event at Currency Creek in 2009, the duration of future acidity flux events are likely to be at the lower end of the scale indicated here, ie. closer to 2-3 months for the first acidity flux event and 1-2 months for the second event in Lake Albert, and closer to 1-2 months for the first acidity flux event and 3-4 months for the second event in Lake Alexandrina. These estimates correspond to the lower minimum lake water level (-1.0 m AHD), hydraulic conductivity of 10 m/day and acidity values of 10,000 mg/L CaCO₃.

In Lake Albert, some of the acidity flux associated with the first autumnal flush will be neutralised by the available soluble alkalinity in the water column.

In Lake Alexandrina, while there is likely to be sufficient soluble alkalinity in the lake water to neutralise the acidity associated with the first autumnal flush event, less alkalinity would remain for future buffering. It may

be possible to cost-effectively add large tonnages of limestone at Wellington from barges to replenish the alkalinity lost during the second autumnal flush within Lake Alexandrina.

ASS management options for the lakes may comprise one or more of the following: (i) controlling lake water levels to maintain sulfidic sediments in a saturated state; (ii) subsurface barrier installation within the uppermost sandy sediments around the unsaturated margins of the lakes; (iii) shallow terraces constructed from ultra-fine grained limestone on clay horizons and application of surface water above the terraces (potentially from the Tertiary Limestone aquifer); and/or (iv) passive or active treatment of lake water and exposed sediment banks with limestone.

Further modelling of acidity generation and flux from clay-rich sediments will be required once lake water levels expose clay-rich sulfidic sediments for extended periods of time. This will require an assessment of the desaturation rates of clay rich sediments in order to clarify the risk posed by the oxidation of sulfides in these materials, and to focus mitigation efforts.

Recommendations

1. All efforts should be directed at keeping sulfidic lake sediments saturated to prevent acidity generation.
2. It is recommended that the acidity generation models developed as part of this study are used to investigate a range of alternative lake water level scenarios, for example:
 - Lake Albert:
 - Ephemeral lake (no pumping from Lake Alexandrina).
 - Maintenance of surface water level at -0.8 m AHD or -0.9 m AHD.
 - Implementation of ASS management options (eg. sub-surface barriers, terraces, passive / active addition of alkaline groundwater, treatment) combined with each water level scenario.
 - Lake Alexandrina:
 - No pumping to Lake Albert (additional 35 GL/year in Lake Alexandrina).
 - Purchase 100 GL/year and continue pumping 35 GL/year or more to maintain Lake Albert at -1.0 m AHD or higher level (additional 100 GL/year in Lake Alexandrina).
 - No pumping to Lake Albert **and** purchase 100 GL/year (additional 135 GL/year in Lake Alexandrina).
 - Maintenance of surface water levels around -2.5 m AHD (worst case scenario where inflows are balanced by evaporation).
 - Maintenance of surface water levels above -1.5 m AHD (via additional water purchase).
 - Implementation of ASS management options (e.g. sub-surface barriers, terraces, passive / active addition of alkaline groundwater, treatment) combined with each water level scenario.
3. To improve the accuracy of the existing acidity generation models and improve predictions for Lake Albert and Lake Alexandrina, the following investigations would assist:
 - The model is highly sensitive to the selection of a groundwater gradient. Hence, modelling of hydraulic conductivity vs. hydraulic gradient in lake sediments would improve the accuracy of acidity generation estimates.
 - Field measurement of hydraulic conductivity vs. hydraulic gradient (e.g. Poltalloch; south-west shore of Lake Albert) via additional piezometer installation and/or field based surveys (e.g. differential GPS).
 - Repeat ANC analyses at previous CSIRO sample sites in Currency Creek to quantify actual ANC consumption rates in acidified sediments. This work should measure the available ANC as well as the total ANC remaining.
 - Analyses of total sulfur (as an indicator of sulfide-sulfur) and ANC at depths of 0.5-2.0 m in the upper sandy sediments of both lakes.
 - Quantify the rate of bacterial sulfate reduction at the base of the water column in lake sediments as a function of pH. Quantify the rate of bacterial sulfate reduction in sandy and clay-rich sediments as a function of key parameters such as carbon and iron content, pH and moisture content.
 - Identify alternative methods to quantify current acidity flux rates to the lakes that are independent of the methods detailed in this report and independent of evaporative concentration and other water quality processes (e.g. sulfur isotope analysis or geochemical ratio assessment).
4. It is recommended that the existing ASS management plan for Lake Albert is reviewed on the basis of recent modelling results.
5. An ASS management plan should also be developed for Lake Alexandrina, including a detailed assessment of the most likely water level scenarios. Specific ASS management options applicable to each scenario should be assessed on the basis of:
 - Ease of implementation;
 - Expected performance in minimising/controlling acidity generation;

- Timeframe for implementation and achievement of management plan objective;
 - Cost-effectiveness;
 - Risks of implementation;
 - Potential to minimise impacts associated with wind erosion;
 - Potential for community involvement.
6. Conduct further investigation to resolve the discrepancy between lake water levels and piezometric levels.
 7. Develop an accurate geological map of the distribution of the uppermost sand and clay in both lakes to facilitate detailed barrier design and installation (if required). This is likely to include the use of one or more methods (e.g. interrogation of CSIRO data, push tube coring, lithological logging via manual coring/pitting of sediments, TEM methods).
 8. Conduct a trial using sonic remote mapping equipment (i.e. CHIRP; 10 cm resolution) to assess the viability of this method for geological mapping of the upper 5-10 m of lake sediments.
 9. Quantify the de-saturation/desiccation rate of sands and clays in order to determine the relative risk of acidity release from these materials during rainfall events. This information will also assist with quantifying groundwater requirements for maintaining saturation of clays via the use of terraces.
 10. Quantify the availability of groundwater from the Tertiary limestone aquifer for maintaining clay saturation in the event that shallow terraces are employed to retard sulfide oxidation in clay-rich sediments.
 11. Develop cost estimates for the management options outlined for lakes Albert and Alexandrina.

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1 Introduction

1.1 Background

The Department for Environment and Natural Resources (DENR), South Australia, commissioned Earth Systems Consulting Pty. Ltd. (Earth Systems) to quantify acidity flux rates from acid sulfate soils into the surface waters of Lake Alexandrina and Lake Albert (Lower Murray Lakes). The Lower Murray Lakes are located at the mouth of the Murray River, approximately 75 km south-east of Adelaide. A location map including lake bathymetry is provided in Figure 1. This report provides an assessment of acidity flux rates for the Lower Murray Lakes based on the results of a laboratory testwork program and preliminary field monitoring results collected over a 3 month period.

The majority of soils/sediments around the Lower Murray Lakes either contain sulfuric acid (which typically forms sulfuric materials) and/or have the potential to generate sulfuric acid upon exposure of sulfidic material to atmospheric oxygen. Sulfuric soils/sediments are defined as materials that have a pH of less than 4 when mixed in a 1:1 ratio with water. Sulfidic soils/sediments, on the other hand, generate a pH greater than 4 upon mixing with water (1:1 ratio) but have the potential to produce acidic drainage (pH < 4) following oxidation of sulfides. In the Lower Murray Lakes, sulfides are generally present in the form of pyrite (FeS₂) and iron monosulfide (FeS). The latter commonly occurs as a "monosulfidic black ooze" (MBO).

Water levels in the Lower Murray Lakes are declining as a result of the unprecedented drought currently affecting the catchment area (Fitzpatrick et al., 2008), and over allocation of river flows. The lowering of lake water levels increases the volume of sulfidic material that is exposed to atmospheric oxygen. As this material is exposed to oxygen it generates acid and metalliferous drainage (AMD) which has the potential to result in ecological, human health and water quality issues. Generation of AMD due to the oxidation of sulfidic material represents a significant environmental risk for the Lower Murray Lakes.

The widespread occurrence of carbonate-bearing lithologies is responsible for elevated alkalinity in both river and lake waters, and some groundwaters feeding the lakes. In the northern portion of the lakes, and marginal to the lower reaches of the Murray River, metamorphic basement is commonly overlain by Tertiary limestone. Further south, the basement is draped by Pleistocene aeolian calcareous sand and calcrete of the Bridgewater Formation. The Bridgewater Formation outcrops widely in the southern half of the Lower Lakes, and is overlain by more recent lake sediment accumulations. The lake sediments include fine to medium grained quartz-rich sands, organic-rich muds and thin ligneous horizons. A more detailed description of the lake sediments is provided by Barnett (1993).

For simplicity, the term 'ASS' is used in this report to describe both sulfuric and sulfidic lake materials. While ASS technically encompass both soils and sediments, the term 'sediments' is used hereafter (rather than soils) when describing the lake shoreline and bed materials, given the depositional history of the lake system.

The stratigraphy or 'layering' of lake sediments has important implications for characterising ASS risk and estimating acidity flux rates to the lakes, allowing extrapolation of field observations at representative sites to the broader lake system.

Pumping of water from Lake Alexandrina to Lake Albert commenced on 2 May 2008 to maintain comparable water levels in the two lakes, and thus prevent the drying of Lake Albert, which otherwise receives minimal surface water inflows. Since mid-2009, however, pumping operations were terminated to facilitate protection of Lake Alexandrina. The current management plan is to maintain a minimum water level in Lake Albert that facilitates permanent inundation of the relatively high risk ASS in the centre of the lake.

In Lake Alexandrina, the lower region comprising the Goolwa Channel and its tributaries, Currency Creek and Finniss River, was identified in early 2009 as being at particularly high risk of acidification (Fitzpatrick et al., 2009). Several other high risk areas around the perimeter of the Lower Murray Lakes were identified by Fitzpatrick et al., (2008).

A range of management options for ASS in the Lower Murray Lakes was investigated by Earth Systems (2008a and 2008b). ASS management approaches for the lakes were broadly categorised as follows:

1. Prevent AMD by managing lake water levels to ensure that ASS are permanently submerged and sulfide oxidation is therefore minimised.
2. Control AMD in-situ via neutralisation (addition of alkaline amendment to ASS) and/or reduction (addition of organic matter to ASS).
3. Treat AMD within the lake water bodies, either passively or actively, via neutralisation (alkalinity addition) and/or reduction (organic matter addition).

A detailed ASS management strategy for Currency Creek, Finniss River and Goolwa Channel was prepared in March 2009 (Earth Systems, 2009a). This comprised a six-stage approach, with the last 4 stages being subject

to pre-determined water quality triggers. To date, implementation of the strategy has included pre-emptive limestone addition trials as well as emergency response limestone dosing into acidic surface waters and adjacent acidic pore waters. Flow regulators have been installed at Clayton and Currency Creek, and water pumping from Lake Alexandrina to the Goolwa Channel commenced in August 2009 to re-establish saturated conditions in high risk ASS areas.

A preliminary ASS management plan has been developed for Lake Albert (Earth Systems, 2009b).

The assessment of acidity flux rates from exposed ASS (this study) will provide key information required for the development of detailed management strategies for the Lower Murray Lakes.

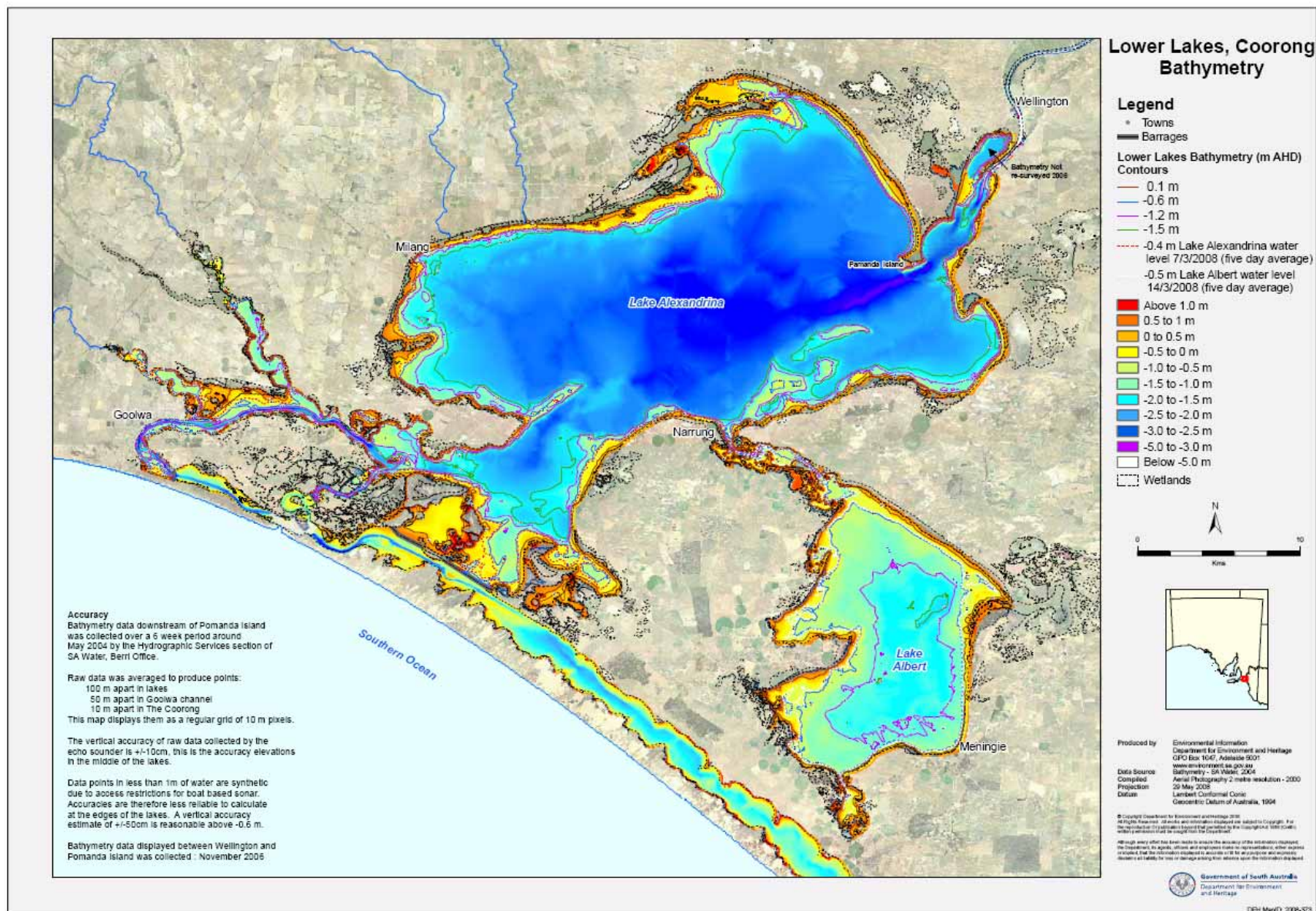


Figure 1. Location map and bathymetry of the Lower Murray Lakes (DEH, 2008).

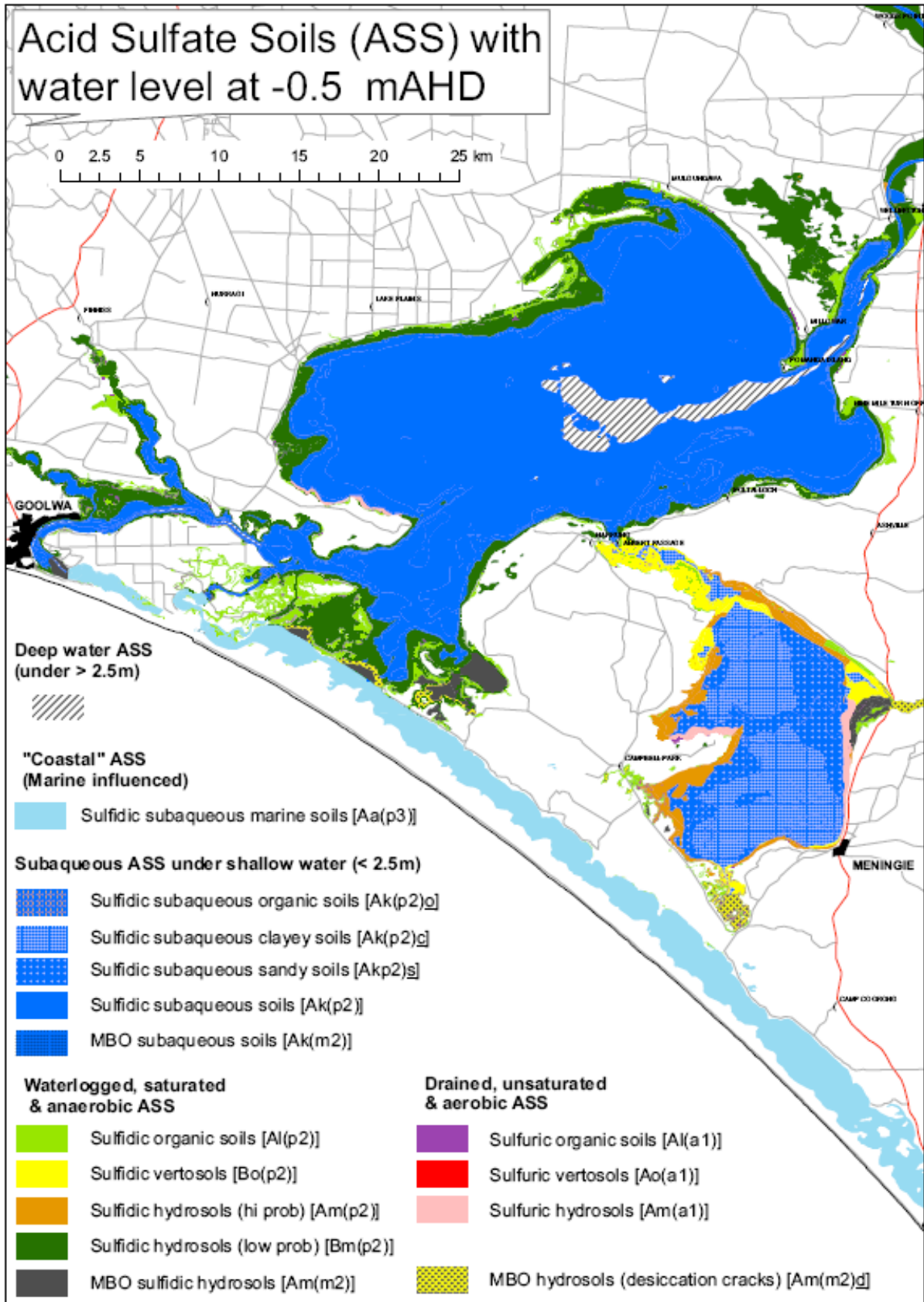


Figure 2. Predicted distribution of ASS of the Lower Murray Lakes at drought water levels (-0.5 m AHD). Taken from Fitzpatrick et al (2008). See Fitzpatrick et al (2008) for description of ASS categories.

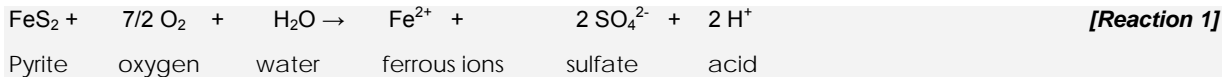
1.2 Acidity generation and flux in the Lower Murray Lakes

ASS have the potential to adversely affect water quality in the Lower Murray Lakes. When sulfidic material is exposed to oxidising conditions, sulfides begin to oxidise and water may subsequently transport reaction products including acidity, sulfate, iron and other metals into surface water and groundwater.

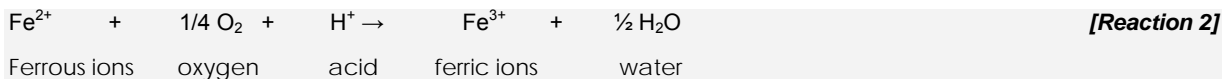
1.2.1 Acidity generation processes

Acid and metal production associated with pyrite oxidation is shown in Reactions 1 to 4.

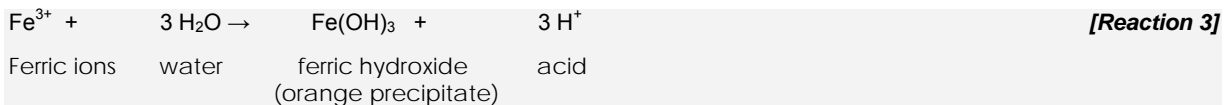
An initial oxidation reaction involves the oxidation of pyrite to produce ferrous ions (Fe^{2+}), sulfate and acid, as shown in Reaction 1.



The ferrous ions (Fe^{2+}) released by pyrite oxidation may be further oxidised to ferric ions (Fe^{3+}) consuming some acid (Reaction 2). Notice that this reaction does not involve pyrite.

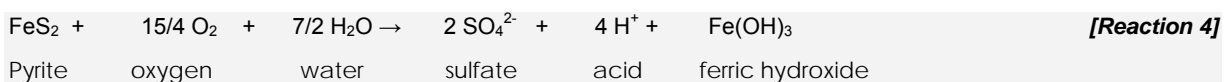


The ferric ions then reacts with water to form ferric hydroxide ($\text{Fe}(\text{OH})_3$), which may precipitate out of solution, producing additional acid (Reaction 3).

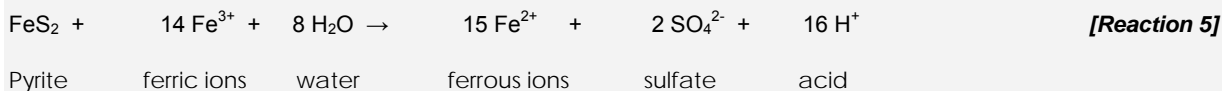


As shown in Reaction 3, the precipitation of ferric hydroxide is a key acid producing stage. Once sulfide minerals have oxidised and released Fe^{2+} , it is extremely difficult to prevent ferrous ions oxidising to ferric ions with concomitant iron hydroxide precipitation and further acid generation.

A summary reaction of the complete oxidation of pyrite (by oxygen) in sulfidic materials may be expressed as follows (Reactions 1-3 combined):

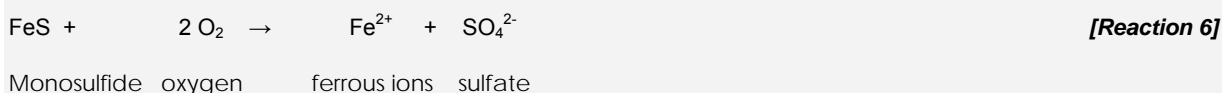


Furthermore, the presence of soluble ferric ions (Fe^{3+}) can accelerate the oxidation of pyrite, generating additional sulfate and acid, as shown in Reaction 5.



Note that in Reaction 5, 16 moles of acid are produced per mole of pyrite oxidised, as compared with 4 moles of acid generated when pyrite is oxidised by molecular oxygen (Reaction 4). This indicates that sulfide oxidation can proceed in the absence of oxygen if ferric ions are present. Whether pyrite oxidation proceeds through Reaction 4 or 5 depends on the chemical conditions in solution at the pyrite surface. Reaction 5 indicates that iron plays a significant role in promoting sulfide oxidising reactions that result in AMD.

Similar oxidation reactions occur for MBO. MBO oxidation is shown in Reaction 6:



The oxidation of MBO is not initially acid generating, but is *acidity* generating. Ferrous ions (Fe^{2+}) produced in Reaction 6 may oxidise to ferric ions, as shown in Reaction 2 and eventually precipitate as ferric hydroxide as in Reaction 3.

Two distinct processes, both promoted by oxidation of sulfide minerals, are responsible for decreasing the pH of an aqueous solution:

1. Acid (H^+) is directly generated by the oxidation of sulfur (Reaction 1).
2. Acid (H^+) is generated by the precipitation of metal hydroxides (eg. $\text{Fe}(\text{OH})_3$, $\text{Mn}(\text{OH})_4$; Reaction 3) during oxidation/neutralisation/dilution reactions.

While process 1 is controlled only by the availability of oxygen and water, process 2 depends on the solubility of the metal aqueous species, which in turn is controlled by factors such as pH of the solution and oxidation state of the metal. In other words, the generation of acid through process 1 is limited by the sulfide oxidation rate, while the generation of acid through process 2 can be delayed until metals can precipitate from solution (thus the term "latent acidity" or "mineral acidity").

The term "acid" quantifies only the actual amount of H^+ present in solution and is generally expressed as pH. The term "acidity", on the other hand, accounts for both the actual H^+ concentration of the aqueous solution and the potential for acid generation due to mineral or latent acidity (ie. H^+ produced by process 2).

1.2.2 Neutralisation, sulfide re-precipitation and acidity flux processes

The fate of acidity (i.e. acid and metals) generated within the lake sediments can be affected, to varying degrees, by the following processes:

- Neutralisation of acidity by carbonate minerals contained within the lake sediments;
- Neutralisation of acidity by aluminium silicate minerals (e.g. clays), contained within the lake sediments;
- Natural microbial precipitation of sulfides (e.g. pyrite or iron monosulfide) within the saturated zone of the lake sediments, in the presence of organic carbon and available iron. This is likely to occur relatively rapidly where the pH remains above 4.5.

Furthermore, any acidity that is transported to surface waters has the potential to be remediated by:

- Neutralisation by soluble bicarbonate alkalinity within the lake waters;
- Natural microbial precipitation of sulfides (i.e. pyrite or iron monosulfide) in the presence of some alkalinity, organic carbon and available iron.

A series of schematics in Attachment A documents likely localised sulfur cycling processes within exposed lake sediments, including examples of reactions that affect the fate of acidity generated within the lake sediments.

Attachment B provides a more extensive list of acidity generation, neutralisation, storage and release reactions likely to be occurring in the lake sediments, based on the range of minerals observed and documented by Fitzpatrick et al. (2008). For each reaction, the proportion of acidity generated from pyrite oxidation that is:

1. Neutralised in-situ,
2. Stored in acid salts, and
3. Released into water,

has also been estimated. This data has been summarised in Attachment B.

Attachments A and B highlight the significant role of sulfur cycling in partial neutralisation of the total potential acidity of the lake sediments. The data emphasise the importance of identifying the nature and distribution of secondary salts as a rapid means of semi-quantitatively determining the broader distribution of:

- Subsurface sand and clay;
- Iron sulfides;
- Carbonate minerals.

2 Objectives

The objectives of this study are to:

- Develop an understanding of the acidity generation, neutralisation and groundwater transport processes within the lake sediments of the Lower Murray Lakes.
- Quantify acidity flux rates to proximal water bodies during wetting events, by assessing the hydrogeology and hydrogeochemistry of lake sediments via a combination of laboratory and field testwork programs.
- Provide recommendations for future management of the Lower Murray Lakes.

3 Scope of Works

The scope of this study includes:

1. Design, establishment and implementation of a laboratory testwork program to measure sulfide oxidation rates of Lower Murray Lakes ASS as a function of sediment moisture content.
2. Design and establishment of a field monitoring program to collect geological, geophysical, hydrogeological and hydrogeochemical data at selected high risk locations in the Lower Murray Lakes including:
 - Currency Creek (tributary of Lake Alexandrina).
 - Point Sturt (Lake Alexandrina).
 - Campbell Park (Lake Albert).
 - “Windmill” location (Lake Albert, north-eastern shoreline).
3. Implementation of a field monitoring program at the four sites listed above over a period of 3 months.
4. Laboratory and field data analysis, including modelling, to estimate acidity flux rates to the Lower Murray Lakes based on available data.
5. Preparation of a final report (this document) incorporating laboratory and field monitoring results, modelling outputs, acidity flux rate estimates and management recommendations.

4 Methodology

The methodology for quantifying acidity flux rates to the Lower Murray Lakes included the following key steps:

- Design, establish and implement a laboratory testwork program (Section 4.1).
- Design, establish and implement a field monitoring program (Section 4.2).
- Conduct an assessment of laboratory and field data in order to develop acidity generation and acidity flux models for the lakes (Section 4.3).
- Determine the management implications of acidity flux modelling for the Lower Murray Lakes (Section 4.4).

4.1 Laboratory testwork program

4.1.1 Introduction

A laboratory testwork program was developed to measure sulfide oxidation rates of Lower Murray Lakes ASS as a function of:

- Sediment texture (sands vs. clays).
- Moisture content.

Acidity or pollutant generation rates are directly proportional to sulfide mineral oxidation rates as described in Section 1.2.1. While acidity generation rates are influenced by the concentration of sulfide minerals, sulfide oxidation rates are independent of sulfide mineral concentrations. They are, however, strongly dependent on the rate of supply of oxygen from the atmosphere to mineral reaction sites (by advection and/or diffusion). In relatively fine textured materials such as most ASS, the supply of oxygen to mineral surfaces is largely controlled by moisture content. Hence, an understanding of the relationship between moisture content and sulfide oxidation rate is vital to quantifying acidity flux rates from ASS.

Other factors that can influence sulfide oxidation rates include (DITR, 2007):

- Chemical composition of pore water in contact with the reaction sites, including pH and the ferrous/ferric ion ratio.
- Temperature at the reaction sites.
- Water content at the reaction sites.
- Microbial ecology of mineral surfaces.

There is only limited data available in the literature on the oxidation rates of sulfidic materials, particularly from ASS¹. Furthermore, there is no available information on the relationship between moisture content and oxidation rates in ASS.

The available literature indicates that pyrite oxidation rates can vary by more than 5 orders of magnitude, from less than 0.0001 wt% FeS₂ / day to 10 wt% FeS₂ / day (ie. 1.3 x 10⁻¹² kg O₂ / kg / s to 1.3 x 10⁻⁷ kg O₂ / kg / s) (Ward et al, 2004a; Ward et al, 2004b; Borma et al, 2003; Rigby et al, 2006; Di Nanno et al, 2007; Morse, 1991; McIlwaine, 1998; Earth Systems, 2009c; Earth Systems, 2009d; Earth Systems, 2009e; Anderson et al, 1999; Eidsa et al, 1997; Hollings et al, 2000).

As shown in Figure 3, the higher oxidation rates (0.1 to 10 wt% FeS₂ / day) are generally associated with naturally unconsolidated sediments such as ASS (Ward et al, 2004a; Ward et al, 2004b; Borma et al, 2003; Rigby et al, 2006; Di Nanno et al, 2007; Morse, 1991), while the lower rates (0.0001 to 0.01 wt% FeS₂ / day) are more typical of mine environments (McIlwaine, 1998; Earth Systems, 2009c; Earth Systems, 2009d; Earth Systems, 2009e; Anderson et al, 1999; Eidsa et al, 1997; Hollings et al, 2000).

Pyrite oxidation rates in ASS are known to be significantly faster than other forms of sulfidic material (rocks), due to the framboidal nature (i.e. ultra-high surface area) of sulfide minerals in ASS. This is consistent with

¹ Acidity generation data (eg. mass H₂SO₄ / area / unit time) are available from other published sources but are not directly comparable with other sites as they do not relate acidity estimates to sulfide content, groundwater levels, moisture content, etc.

visual observation of rapid generation of acidic salts (e.g. ferric hydroxide or ferrihydrite precipitate formation) within hours of exposure of sulfidic ASS to oxidising conditions.

Current approaches for estimating pyrite oxidation rates, and the benefits and limitations of these approaches, are shown in Table 1.

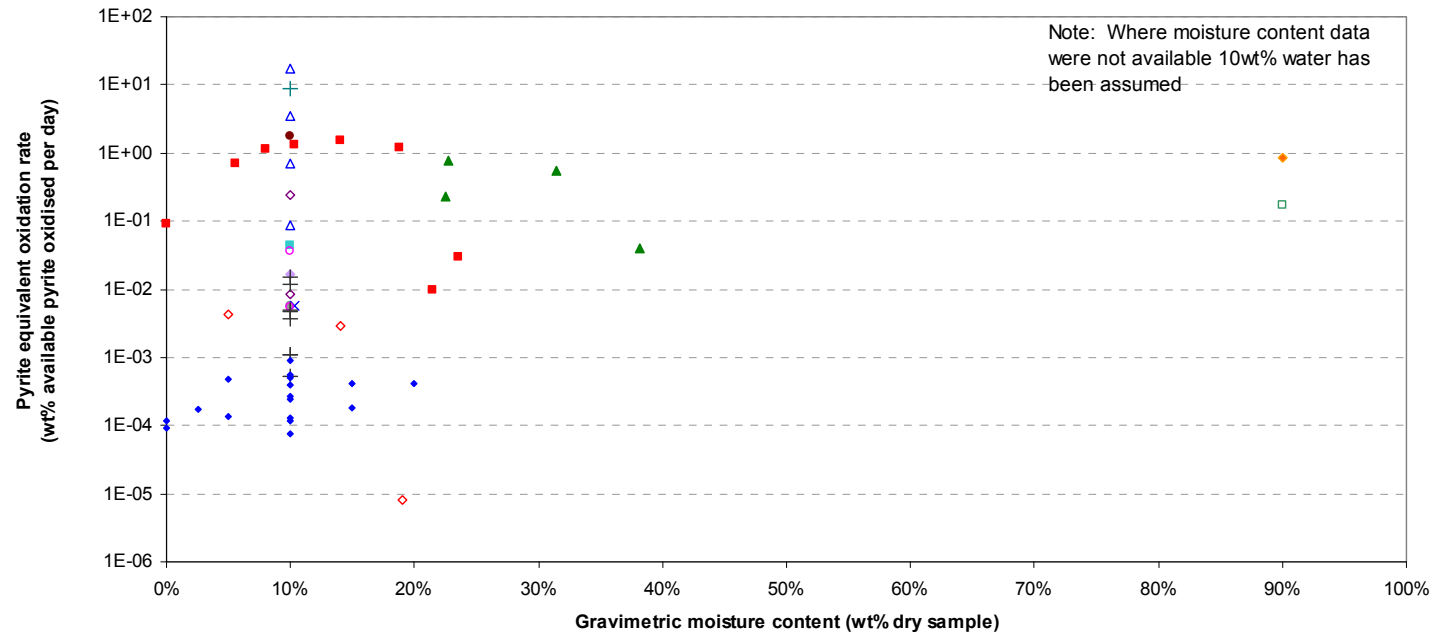
Due to the limitations of conventional methods for estimating pyrite oxidation rates as a function of moisture content, an alternative approach, the OxCon (Oxygen Consumption) Method, was developed by Earth Systems. This method is based significantly on the conventional laboratory-based oxygen consumption approaches utilised by ANSTO. The OxCon Method can be used to accurately determine sulfide oxidation rates by directly measuring oxygen consumption as a function of time (see Section 4.1.2). The OxCon method has been utilised successfully in both laboratory and field scale applications for the direct measurement of pyrite oxidation rates (Earth Systems, 2009c; Earth Systems, 2009d; Earth Systems, 2009e) and offers some advantages over alternative methods, as identified in Table 1 and summarised below:

- Low-cost compared with other methods and fast determination of sulfide mineral oxidation rates (weeks-months).
- Organic carbon oxidation rates can be simultaneously but independently determined.
- Determine sulfide mineral oxidation rates as a function of oxygen concentration and/or sample moisture content.
- Measures oxidation rates directly (via oxygen consumption) rather than inferences based on leachate sulfate concentrations.
- Can be used on a wide range of materials including unconsolidated sediments (e.g. ASS) and rock samples.
- Tests can be conducted on or off site with small sample sizes (<10 kg), with minimal on site space requirements.
- Many tests can be conducted simultaneously, permitting rapid assessment of potential pollutant loads from different sulfidic materials.
- Apparatus can be re-set within minutes to allow tests to be re-run to validate data.
- Oxygen concentrations are not affected by CO₂ generation from carbonate minerals.

Table 1. Comparison of available methods for determining sulfide oxidation rates.

Method	Description	Test duration	Direct measurement of oxidation rate	Accounts for organic carbon oxidation	Allows measurement of oxidation rate vs. moisture content	Allows measurement of oxidation rate vs. oxygen concentration	Applicable to various materials (eg. sand, clay, rock)	Accurate	Cost-effective
Column leach test	A column (eg. PVC) is filled with sulfidic material and open to atmospheric conditions (incident rainfall) or irrigated artificially to maximise leachate recovery; water quality of leachate draining from base of column (sulfate, etc.) provides an indication of sulfide oxidation rate	Months-years	x	x	x	x	x	?	x
Humidity cell (e.g. ASTM, 1996)	Similar to above but sample is subjected to artificial cycles of dry air permeation, humid air permeation, and irrigation, to simulate accelerated weathering	Weeks-months	x	x	x	x	✓	✓	✓
Field measurement of oxygen concentration	Field measurement of oxidation rate based on change in gaseous oxygen concentration over time (assumes O ₂ consumption assumed to be entirely attributable to sulfide oxidation)	Months-years	✓	x	x	✓	✓	✓	x
Oxygen Consumption Method - IOR Meter™, developed by ANSTO (Bennett et al., 2005)	Direct laboratory measurement of oxidation rate based on change in gaseous O ₂ concentration over time (O ₂ consumption assumed to be entirely attributable to sulfide oxidation)	Days-weeks	✓	x	✓	✓	✓	✓	✓
Oxygen Consumption Method - oxygen measurement via external Micro-Oxymax Respirometer setup (Eidsa et al., 1997)	Direct laboratory measurement of oxidation rate; computer controlled low concentration (O ₂ and CO ₂) gas measurement system external to test chamber; sulfide and carbon oxidation processes can be distinguished	Days-weeks	✓	✓	✓	✓	✓	✓	x
Oxygen Consumption Method - oxygen measurement via internal Galvanic electrochemical cell O ₂ sensor (Anderson et al, 1999; Hollings et al., 2001)	Direct laboratory measurement of oxidation rate via data-logged O ₂ sensor fitted inside test chamber (O ₂ consumption assumed to be entirely attributable to sulfide oxidation)	Days-weeks	✓	x	✓	✓	✓	✓	x
Oxygen Consumption Method - OxCon Method (Earth Systems, 2009a-c; Section 4.1.2)	Direct laboratory measurement of oxidation rate based on air displacement in reaction vessel; sulfide and carbon oxidation processes can be distinguished	Days-weeks	✓	✓	✓	✓	✓	✓	✓

**Comparison of pyrite equivalent oxidation rates
for various sulfidic materials**



- | | |
|--|--|
| ■ Lower Lakes ASS - Sand | ▲ Lower lakes ASS - Clay |
| ◆ Ward et al. (2004a) ASS, NSW | □ Ward et al. (2004b) ASS, NSW |
| ◇ Borma et al. (2003) Sediments | ● Rigby et al. (2006) ASS, Qld |
| + Di Nanno et al. (2007) Sediments with microbial and nutrient inoculation | △ Morse (1991) Marine sediments |
| ● McIlwaine (1998) Overburden material, coal mine, Vic | ◆ Earth Systems (2009c) Overburden material, coal mine, Vic |
| ■ Earth Systems (2009d) Gold mine tailings, Vic | × Earth Systems (2009e) Pyrite mine waste rock (pyrrhotite), SA |
| ◇ Earth Systems (2009e) Pyrite mine waste rock and tailings mixture (pyrrhotite), SA | ○ Anderson et. Al (1999) Nickel mine waste rock (pyrrhotite), Newfoundland, Canada |
| + Eidsa et. Al. (1997) Base metal mine ore, Norway | ◆ Hollings et. al. 2000; Uranium mine waste rock, Canada |

Figure 3. Comparison of reported pyrite oxidation rates for a variety of sulfidic materials including hard rock mine waste rock, tailings, coal mine overburden material and ASS. Pyrite oxidation rate shown on log scale due to significant range in oxidation rates, from 0.0001-0.01 wt% FeS₂ / day (typical of mine environments) to 0.1-10 wt% FeS₂ / day (naturally unconsolidated sediments). Scale of the y-axis ranges from 1.E-06 (10⁻⁶ or 0.000001wt% FeS₂ / day up to 1.E+02 (10² or 100 wt% FeS₂ / day).

4.1.2 OxCon method

The following section summarises the mode of operation of the OxCon apparatus that was used to quantify the oxidation rates of Lower Lakes ASS material as a function of moisture content. The OxCon apparatus, shown schematically in Figure 4, consists of three chambers:

1. Sample chamber;
2. Displacement chamber;
3. Fluid reservoir.

A sample of sulfidic material is placed in the sample chamber. The moisture content, sulfide content and total mass of the sample needs to be quantified. Fluid and a thin oxygen impermeable layer (over the fluid) are placed in the fluid reservoir. While the system remains barometrically open to the atmosphere, it also remains a closed system with respect to its gaseous oxygen concentration.

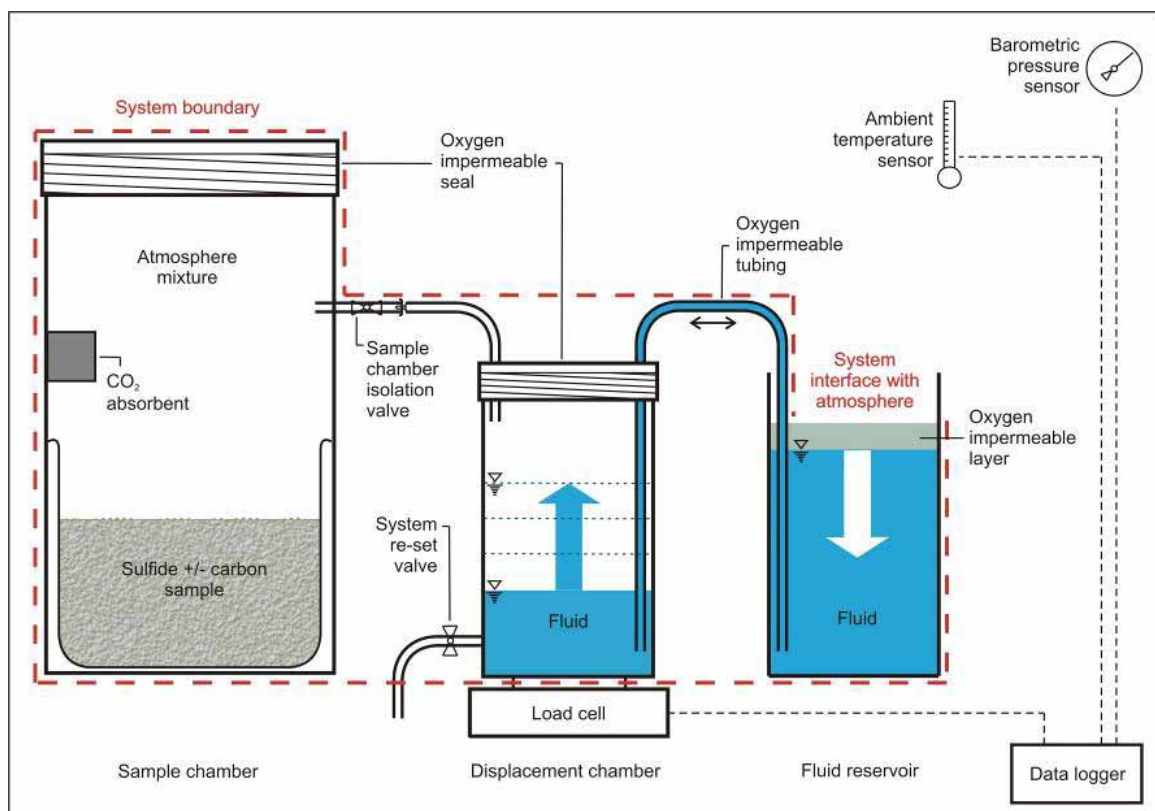


Figure 4. Schematic of OxCon apparatus.

Once the sample has been isolated from the atmosphere, and the testwork instigated, oxygen consumption proceeds via sulfide oxidation. Gaseous oxygen is converted to sulfate phases in this process. Oxygen consumption within the sample chamber and adjacent displacement chamber causes the internal pressure of the system to decrease. In order to maintain equilibrium with the atmosphere, fluid is drawn from the reservoir into the displacement chamber. The mass of fluid transferred into the displacement chamber is proportional to the volume of oxygen consumed in the sulfide oxidation process. These two parameters are related via Boyle's Law, and with continuous measurement (30 minute data logging) of ambient temperature (T), barometric pressure (P_{atm}) and displacement bottle mass changes over time, it is possible to calculate an oxygen consumption rate (moles per day) attributed to oxidation of pyrite and oxidation of organic carbon.

Oxygen consumption due to the bacterial oxidation of organic carbon is measured at the completion of testing by measuring the mass of carbon dioxide removed by an absorbent contained inside the main chamber during testing. The mass of absorbed carbon is converted into a volume of oxygen consumed assuming a respiratory quotient (RQ , molar volume ratio of bacterial O_2 consumption and CO_2 generation) of 1. Oxidation of organic carbon or CO_2 released from carbonate dissolution reactions is assumed to be

linear, and based on the mass of CO₂ adsorbed and the time over which the CO₂ is adsorbed, an oxygen consumption rate (moles per day) attributed to the oxidation of organic carbon or release of CO₂ from carbonate dissolution can be calculated.

The calculations assume that all sulfur is in the form of pyrite (FeS₂) and that the oxidation reaction proceeds to completion (ie. complete oxidation of both the sulfur and iron components of pyrite occurs, as per Reaction 4). The oxygen consumption rate attributed to pyrite oxidation (moles per day) is calculated by subtracting the oxygen consumption rate (moles per day) attributed to oxidation of organic carbon from the bulk oxygen consumption rate (moles per day, attributed to both pyrite oxidation and oxidation of organic carbon).

$$\begin{aligned}
 & \text{Total oxygen consumption rate (measured by OxCon apparatus)} \\
 & - \text{Oxygen consumption rate attributed to oxidation of organic carbon (measured from CO}_2 \text{ adsorbent)} \\
 & = \text{Oxygen consumption rate attributed to FeS}_2 \text{ oxidation}
 \end{aligned}$$

The pyrite oxidation rate (wt % available pyrite per year) can be calculated from the oxygen consumption rate (attributed to pyrite oxidation) and the known mass of pyrite in the sample using the stoichiometry in Reaction 4.

$$\begin{aligned}
 & \text{Oxygen consumption rate attributed to FeS}_2 \text{ oxidation (mol/day)} \\
 & \times 3.75 \text{ moles O}_2 \text{ consumed per mole FeS}_2 \text{ oxidised (see Reaction 4)}^* \\
 & \times 1/\text{mass of FeS}_2 \text{ in sample (mol)} \\
 & \times 100 \\
 & = \text{FeS}_2 \text{ oxidation rate (wt\% available pyrite oxidised per year)}
 \end{aligned}$$

Figure 5 shows a typical graphical output from a single OxCon test, with cumulative oxygen consumption vs. time plotted in blue. A linear “line of best fit” is fitted to the cumulative oxygen consumption data (bold blue line). The slope of this line is equal to the linear total oxygen consumption rate (mol/day) (ie. attributed to oxidation of organic carbon and pyrite). The calculated linear rate of oxygen consumption attributed to oxidation of organic carbon (mol/day) is subtracted from the total oxygen consumption rate (mol/day) to yield the oxygen consumption attributed to pyrite oxidation (dashed green line). Data plotted in red indicates the estimated residual volumetric oxygen concentration vs. time. The pyrite oxidation rate is calculated from the oxygen consumption rate (attributed to pyrite oxidation only), stoichiometric ratios in Reaction 4 and knowledge of the pyrite content of the sample.

Once the pyrite oxidation rate is known, it can be applied to field sulfide-sulfur data to calculate acidity generation rates. For example, if we had material with a pyrite oxidation rate of 10 wt% per year, and if we know that we have 1.0 million tonnes of material with a pyrite content of 0.05 wt% FeS₂, then we can calculate that this material is expected to generate approximately 83 tonnes of H₂SO₄ acidity per year. That is:

$$\begin{aligned}
 & 1.0 \text{ million tonnes of sediment} \times \\
 & 0.05 \text{ wt\% FeS}_2 \times \\
 & 10 \text{ wt\% FeS}_2 / \text{year (will oxidise to completion, ie. produce 2 moles H}_2\text{SO}_4 \text{ per mole of FeS}_2) \\
 & \approx 83 \text{ tonnes H}_2\text{SO}_4 \text{ acidity released per year} \\
 & \text{(example)}
 \end{aligned}$$

Hence we can convert a pyrite oxidation rate to a field-based acidity generation rate with knowledge of the pyrite content of the field material.

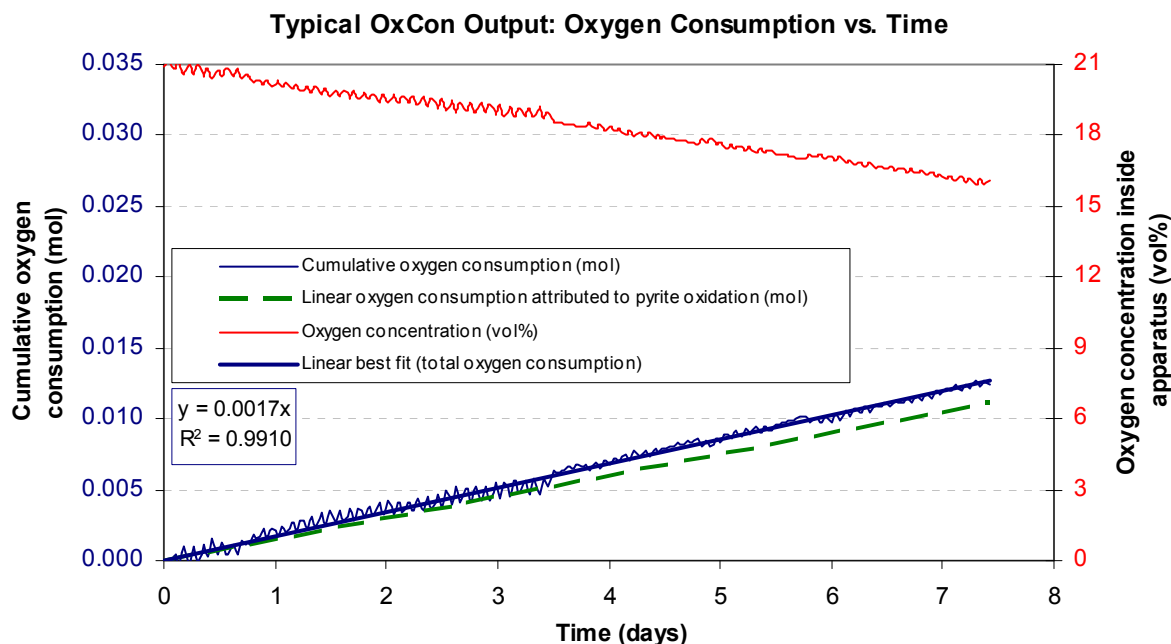


Figure 5. Typical outputs from an OxCon test. Cumulative oxygen consumption (attributed to oxidation of organic carbon and pyrite) vs. time for oxygen concentrations inside the apparatus ranging from ~16-21 vol%. Data shown in blue has been processed from raw data relating to the volumetric consumption of oxygen vs. time and room temperature and pressure considerations (refer to Section 4.1.2). A "line of best fit" was fitted to the cumulative oxygen consumption data (bold blue line). Oxygen consumption attributed to pyrite oxidation (linear best fit total oxygen consumption minus linear best fit oxygen consumption attributed to organic carbon oxidation) is also plotted (green dashed line). Data shown in red indicates the estimated volumetric oxygen concentration vs. time.

4.1.3 Design and fabrication of OxCon apparatus for Lower Murray Lakes ASS

A total of fourteen OxCon apparatuses were custom-designed and fabricated by Earth Systems for the measurement of sulfide oxidation rates of Lower Murray Lakes ASS as a function of sediment texture (sand/clay) and moisture content.

OxCon apparatus (depicted in Plates 1 and 2) were constructed from PVC water pressure fittings. Thick walled silicone tube was used to convey the test fluid. Tubing was connected to PVC fittings using stainless steel threaded hose barbs and hose clamps. All barbed joints were fitted with Teflon tape and reinforced internally with araldite glue. All glued PVC pipe fittings and o-ring fittings were double sealed with an external bead of sealant.

Load cells were mounted on plywood and MDF mounting plates. Load cells were connected to load cell controllers in sets of 2. Load cell controllers (7 in total) were connected to a data logger. A temperature sensor was mounted internally in one OxCon apparatus (depicted in Plate 1) to measure temperature changes inside the OxCon system. An external pressure sensor (depicted in Plate 2) was used to measure ambient room pressure changes. The temperature and pressure sensors were connected directly to the data logger.

Data were accessed via a PC interface software package.

Each OxCon apparatus was tested for leaks by applying and sustaining a vacuum inside the systems for 5 days prior to the commencement of testwork. The vacuum applied (approximately -0.15 bar) during leak testing far exceeds pressure differences during normal operating conditions as the OxCon system operates under atmospheric pressure only.

4.1.4 Sample collection and preparation for kinetic testwork

Separate bulk samples of sulfidic sand (341164 m E, 6056556 m N; 0.5-0.7 m below ground level) and clay (341221 m E, 6056568 m N; 1.0-1.4 m below ground level) were collected from the Campbell Park monitoring site near monitoring point CP-03 (see Figure 6). This site was chosen based on published sulfide content data (Fitzpatrick et al., 2008). Samples were collected on the day prior to the commencement of laboratory testwork and were kept saturated prior to use.

Eight sand samples and 5 clay samples were prepared for separate OxCon apparatus. The moisture content of sand and clay samples were adjusted to desired relative moisture contents (% saturation) by gravimetric measurement and via vacuum desiccation (vacuum exceeding -100 kPa). A temperature controlled water bath (50-60°C) was used during vacuum desiccation to increase the rate of water removal. Sulfide oxidation was assumed to be negligible during vacuum desiccation. Table 2 provides details of the samples prepared for testing, each in a separate OxCon apparatus.



Plate 1. OxCon apparatus at CSIRO Land and Water laboratories, Urrbrae, SA. Shown mounted on load cell with adjacent power supply, data logger and temperature sensor.



Plate 2. OxCon apparatus at CSIRO Land and Water laboratories, Urrbrae, SA. 12 of 14 apparatus shown with adjacent data logger, temperature and pressure sensors (top right of picture) and PC interface.

Table 2. Description of samples used to determine sulfide oxidation rates.

OxCon Apparatus ID*	Sample type [#]	Water content	Pyrite content [^]
		wt%	wt% Pyrite
A	Sand	8.0%	0.30%
B		10.3%	0.30%
C		5.6%	0.30%
D	Clay	47.9%	1.97%
E	Sand	14.0%	0.30%
F		18.8%	0.30%
G		23.5%	0.30%
H		21.5%	0.30%
J	Clay	22.5%	1.97%
K		31.4%	1.97%
L		22.7%	1.97%
M	Sand	0.0%	0.30%
N	Clay	38.2%	1.97%

* Apparatus "I" was used to provide background conditions and hence contained no sample.

[#] Refer to Section 5.1.1 for sediment texture laboratory results.

[^] Based on the assumption that all chromium reducible sulfur measured in the sample is present as available reactive pyrite. Refer to Section 5.1.1 for laboratory results.

4.1.5 Static acid-base accounting testwork for samples used in OxCon apparatus

Sub-samples of saturated bulk sand and clay material were collected and analysed for:

- Total Sulfur (Total S);
- Acid Neutralising Capacity (ANC);
- Net Acid Generation (pH 4.5 and 7.0) and pH after oxidation (NAG pH);
- Total Organic Carbon (TOC);
- Particle size distribution (sand sample only).

The initial gravimetric moisture content (GMC) of both sample types was estimated by oven drying sub-samples at 105°C for 24 hours. Porosity for the sand sample was determined by gravimetric analysis, by saturating then oven drying a sub-sample of known volume at 105°C for 24 hours. Relative saturation of the samples was calculated based on the gravimetric moisture content and porosity.

4.1.6 Kinetic testwork data collection from OxCon apparatus

OxCon equipment was installed by Earth Systems at CSIRO Land and Water laboratories in Urrbrae, South Australia, on 17th August 2009. Equipment was installed in a temperature controlled (21°C±1°C) laboratory. Sample preparation and test commissioning was carried out from 24 August to 3 September 2009.

Data relating to the mass of the displacement chamber, temperature inside the OxCon system and ambient room pressure were collected in 30 minute intervals via a datalogger, and downloaded by CSIRO personnel on a weekly basis. Data was forwarded by CSIRO personnel to Earth Systems for conversion to oxidation rate data.

4.2 *Field Monitoring Program*

4.2.1 Site selection

Selection of sites for implementation of the field monitoring program was primarily based on the perceived ASS risk, in accordance with available data on the concentration and distribution of sulfidic and sulfuric materials as identified by Fitzpatrick et al. (2008 and 2009). Other factors affecting the identification of suitable field monitoring sites included access, security (e.g. vandalism, livestock) and budget considerations.

Field monitoring was initially prioritised in the Currency Creek / Finniss River / Goolwa Channel region at the lower end of Lake Alexandrina, in response to the identification of particularly high risk ASS in early 2009 (Fitzpatrick et al., 2009).

The field monitoring program was subsequently expanded to incorporate three additional locations, each consisting of multiple sites along a transect, perpendicular to the shoreline. One location was selected in Lake Alexandrina and two in Lake Albert, given the more significant short term ASS risks associated with receding water levels in Lake Albert.

In Lake Alexandrina, a location at Point Sturt was selected for the field monitoring program as this region was identified as a key ASS risk by Fitzpatrick et al. (2008).

In Lake Albert, three regions were considered for the field monitoring program – Campbell Park, Meningie and the north-eastern shoreline – based on results presented in Fitzpatrick et al. (2008) and subsequent liaison with Rob Fitzpatrick (CSIRO) regarding the presence of sulfuric materials and the ASS risk of specific locations within each region. Geophysical surveys and ground-truthing, as described in Section 4.2.2, were subsequently used to confirm two preferred locations for Lake Albert, Campbell Park and the north-eastern shoreline (hereafter referred to as the “Windmill” location). The Meningie region was considered to be a relatively low ASS risk due to extensive outcrops of calcrete/silcrete and calcareous sands of the Bridgewater Formation (limestone).

Each field monitoring location comprises 6-9 piezometers positioned along a four-point transect for groundwater monitoring, in addition to sediment moisture monitoring equipment, as described in Sections 4.2.3, 4.2.4 and 4.2.5, and displayed in Figure 6.

4.2.2 Geology

A key aspect of the field program was to investigate the detailed geology of the lake sediments and identify broad geological trends across both lakes that would enable extrapolation of data collected from specific sites (Section 4.2.1) to other areas of the lakes. An understanding of the trends in lake sediment distribution was required to develop realistic assumptions and input parameters for acidity generation rate modelling (Section 7). The geological component of the field program involved a combination of:

- Geological logging and sample collection via manual augering and drilling methods.
- Geophysical surveys and ground-truthing for large scale, rapid geological mapping.

These methods are described in detail below.

Geological logging and sample collection

Currency Creek

Geological logging and collection of sediment samples at Currency Creek was conducted by Earth Systems personnel between 30 April and 2 May 2009, at the locations shown in Figure 6.

Geological logs were recorded for three sites along the northern margin of Currency Creek, where sediments were collected by manual auguring to depths of up to 3 m.

Representative sediment samples (1-2 kg) were collected at several depth intervals (based on geology) for analysis of Net Acid Producing Potential (NAPP), Total Sulfur (Total S), Chromium Reducible Sulfur (S_{Cr}), Acid Neutralising Capacity (ANC) and Total Organic Carbon (TOC). Laboratory analyses of sediment samples were conducted by Australian Laboratory Services (ALS) in Brisbane (NATA registered laboratory).

In addition, test pits were dug to a depth of around 1 m along 5 transects of Currency Creek, to investigate trends in sediment profiles perpendicular to the shoreline (i.e. distribution of sands vs. clays, extent of oxidation in sands vs. clays, etc.).

Where possible, general water quality parameters were also monitored at each site, including pH, electrical conductivity (EC, mS/cm) temperature and redox potential (ORP, mV; platinum reference electrode) in groundwater/pore water, ponded surface water and crack water (in areas containing desiccated sediments). This information assisted in estimating the acidity load generated within the exposed creek sediments (Section 7).

Point Sturt, Campbell Park and Windmill locations

A rubber track mounted Geoprobe 7720DT drill rig was used to collect samples for detailed logging and analysis at the Point Sturt, Campbell Park and Windmill locations, as shown in Figure 6. The drill rig was operated by Boart Longyear, under the supervision of Earth Systems personnel, from 20-25 August 2009. The rig was also used for piezometer installation at the Point Sturt, Campbell Park and Windmill locations, as described further in Section 4.2.3.

Key specifications of the Geoprobe 7720DT system are summarised briefly below:

- Light weight (3.110 tonnes) and mounted on twin rubber tracks, providing a low density footprint (approximately 4 psi) which enables access to soft and boggy locations.
- Designed for vertical drilling up to a depth of 18 m using 1.524 m length probe rods (also capable of angle drilling up to 30 degrees from vertical).
- Includes dual tube system, push-tube system (32 mm diameter core) for accurate sample recovery.
- Fitted with a GH62 hammer, which facilitates push-tube sampling and drilling through unconsolidated and partially consolidated sediments.
- Fitted with a drive head for operating both solid auger rods (100 mm diameter) and hollow auger rods (200 mm diameter; facilitating piezometer installation).
- Hydraulically operated, powered by 54 HP Kubota turbo diesel engine.

The Geoprobe system was sufficiently lightweight to operate effectively on the saturated lake sediments, enabling push-tube sediment sampling to within 50 m of the waters edge (Plate 3).

The Geoprobe system was used to collect push-tube core samples for geological logging to a depth of up to 5 m. In some instances, samples could not be retrieved effectively using the dual tube system, either as a result of highly saturated sediments, or the requirement for augers to penetrate the cemented upper layer of the Bridgewater Formation. Where push-tube core samples could not be obtained, samples were collected from the flight of the hollow augers to complete the geological logs.

The sediment profiles were assessed at the time of sampling (Plate 4) to provide a basis for confirming site suitability for piezometer installation and developing detailed piezometer designs, as described in Section 4.2.3.

Representative sediment samples (1-2 kg) were collected at several depth intervals (sediment horizons) for analysis of NAPP, Total S, ANC, NAG_{pH} , $NAG_{4.5}$ and $NAG_{7.0}$, TOC and particle size distribution.

The NAG tests involved rapid and complete oxidation of sulfides by reaction of each sample with hydrogen peroxide (H_2O_2). After reaction with H_2O_2 , leachate was collected from each sample for measurement of pH (referred to as NAG_{pH} above) and subsequently titrated with NaOH to pH 4.5 then 7.0 to determine $NAG_{4.5}$ and $NAG_{7.0}$ in kg H_2SO_4 / tonne. Prior to titration, sub-samples of leachate were collected from 3 samples (one from each of the three key locations) for analysis of dissolved Al, As, Cd, Cu, Fe, Mn, Ni, Pb, Se and Zn. The rationale for conducting NAG leachate water quality analyses was to identify the range of metals that could potentially be released from sulfidic sediments under fully oxidising conditions.

Laboratory analyses of sediment samples were conducted by Australian Laboratory Services (ALS) in Brisbane (NATA accredited laboratory).

Plate 3. Double-tube push tube sampling conducted using a 7720DT Geoprobe drill rig.



Plate 4. Logging of push-tube samples to determine the local geology and assist with ground-truthing of geophysical survey data, piezometer site confirmation and detailed piezometer design (e.g. screened zone location and extent).

Geophysical surveys

A series of electrical geophysical surveys was conducted at the following locations, as shown in Figure 7:

- Lake Alexandrina – Point Sturt.
- Lake Albert – Campbell Park.
- Lake Albert – Meningie.
- Lake Albert – Windmill location (north-eastern shoreline).

A total of 12 TEM (transient electromagnetic) geophysical traverses, ranging in length from 200 to 420 m, were conducted from 3-5 June and 6-9 July 2009, to investigate the ability of this method to assist with geological mapping of sediments in the lakes. The survey transects are shown in Figure 7. The survey equipment is displayed in Plates 5-6.

Electrical measurements were collected using a 5 m x 5 m coincident-loop TEM configuration, established to provide an average depth resolution of approximately 10 m. Conductivity soundings were collected at 5 m intervals along the transect lines to ensure good lateral resolution of the electrical geophysical data. The horizontal location of each survey point was recorded using a GPS unit.

The survey results were processed and transformed into “pseudo-sections” showing (bulk) soil/water conductivity variations with depth along a cross-section. The pseudo-sections require careful interpretation as they are not based on absolute measurements of depth below ground level. Conductivity measurements represent a combination of water content and the salinity of the contained water. Hence, interpretations require an understanding of both hydraulic conductivity and groundwater chemistry. Identifying and assigning actual conductivity values/patterns to specific geological units generally requires ground-truthing through drilling or other means.

To assist in the interpretation of the geophysical survey data, hand auger holes and test pits were installed to a depth of up to 2 m at selected locations along each transect. This was followed up with push tube sediment sample recovery using the Geoprobe drill rig double-tube core sampling system.

The pseudo-sections of sub-surface conductivity were used to:

- Develop an understanding of the three-dimensional variability of sub-surface geology and groundwater content/salinity within each location and between locations.
- Assist in confirming the suitability of each location for piezometer installation and field monitoring (refer to Section 4.2.3).



Plates 5-6. Geophysical surveys were conducted across the exposed shoreline sediments of Lake Alexandrina (Point Sturt) and Lake Albert (Campbell Park, Meningie and Windmill locations). Survey apparatus at Point Sturt and Windmill locations pictured above.

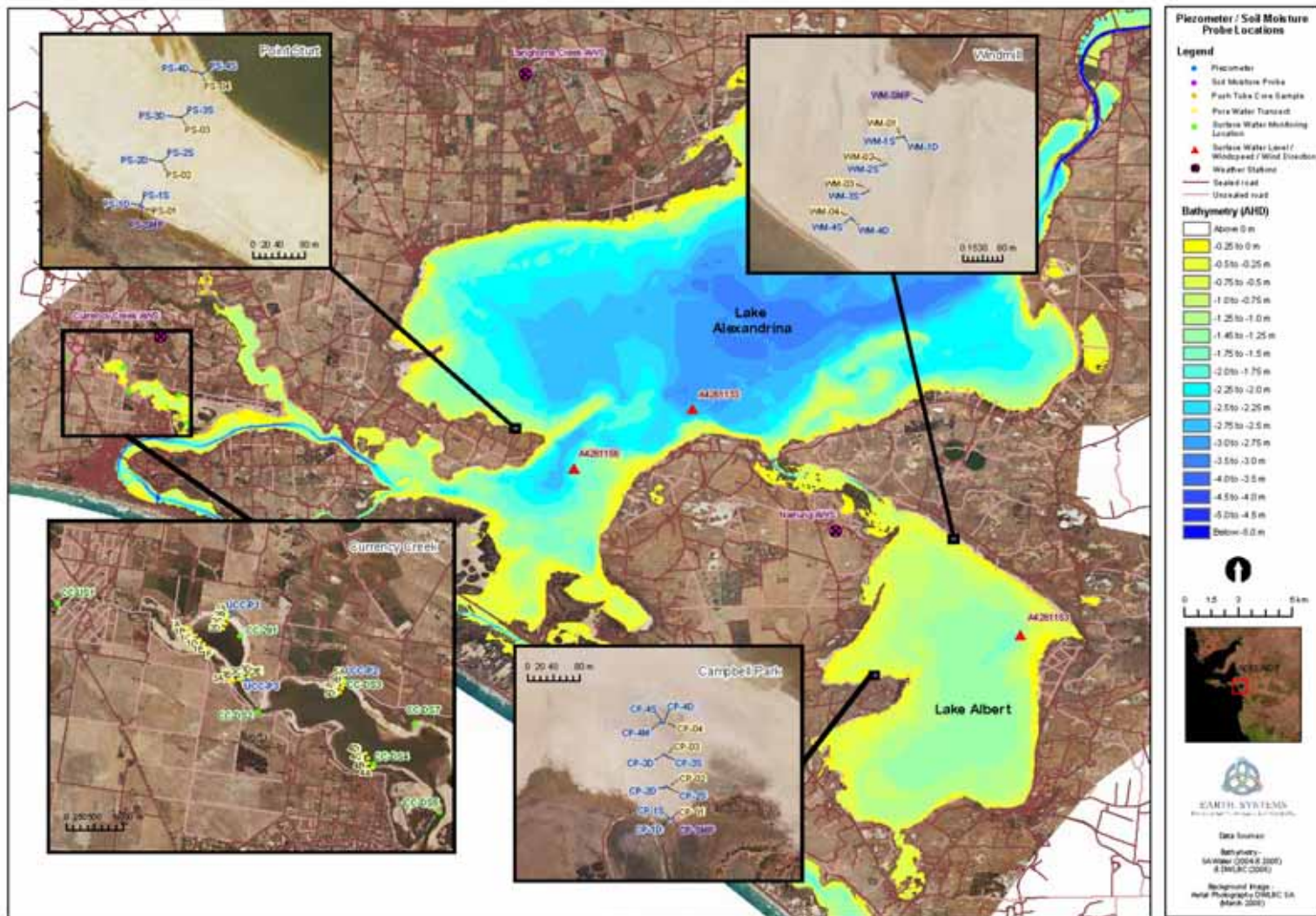


Figure 6. Location of piezometers and moisture sensors installed in the Lower Murray Lakes.

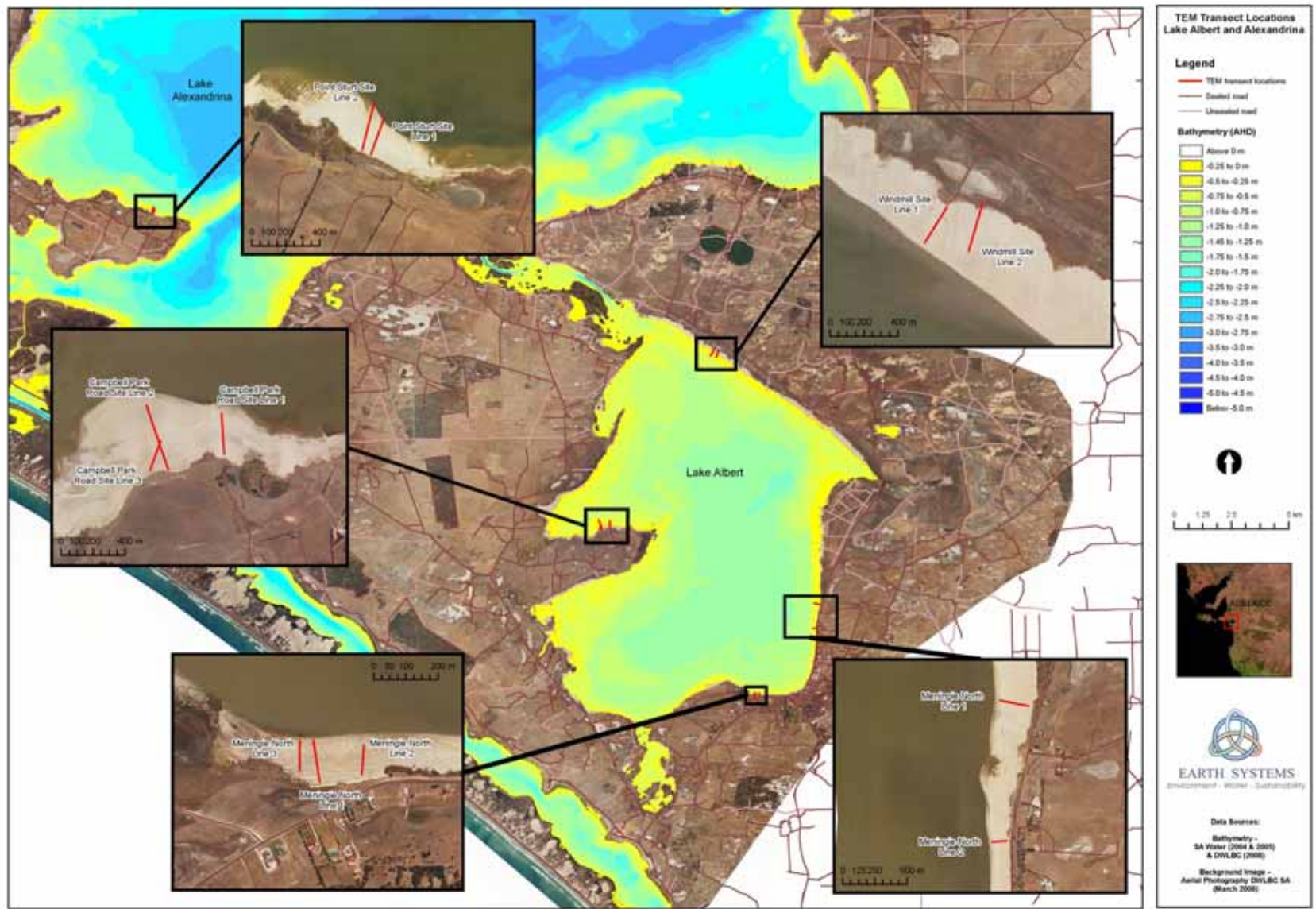


Figure 7. Location of geophysical survey transects.

4.2.3 Piezometer installation

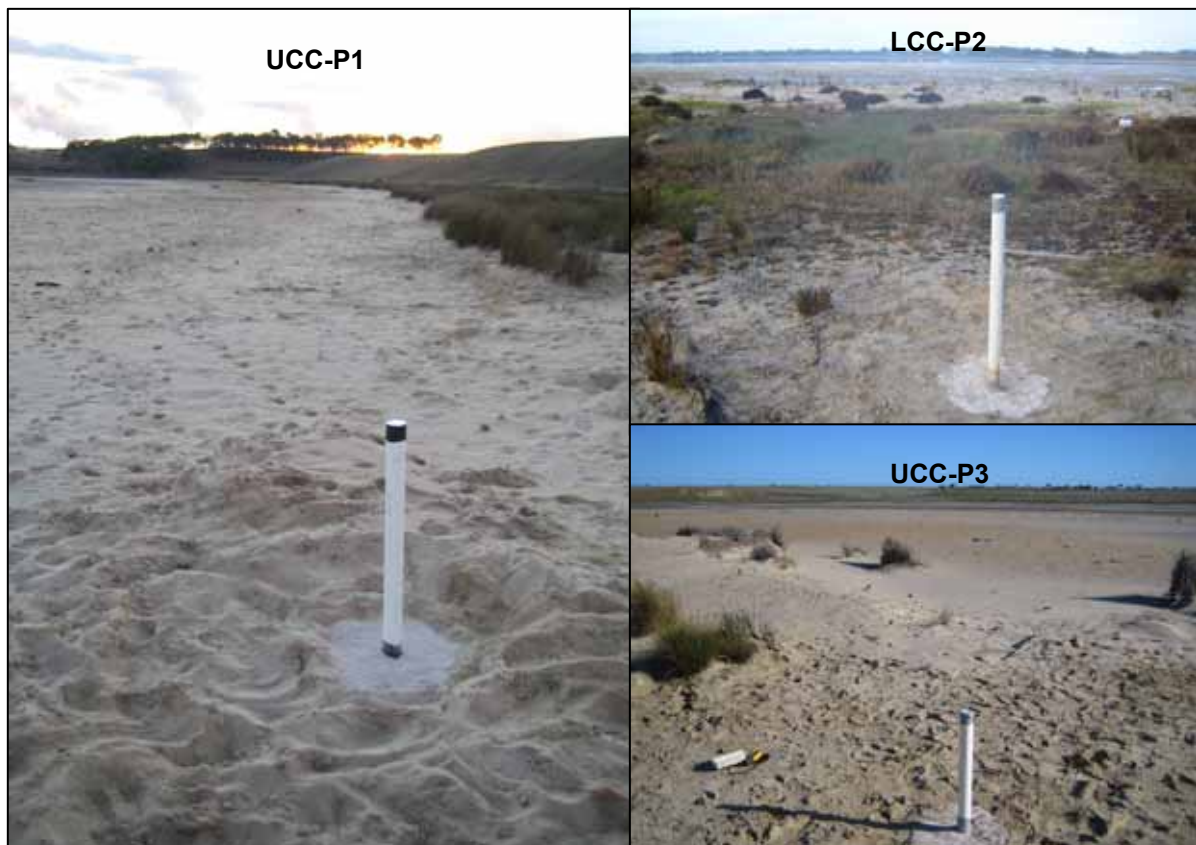
Currency Creek

Three piezometers were installed by Earth Systems personnel between 30 April and 2 May 2009 to enable long-term groundwater level and water quality monitoring along the northern margin of Currency Creek (Figure 6; Plates 7-9). The piezometer locations were selected on the basis of field data collected along 5 transects of Currency Creek from 30 April to 2 May 2009, as described in Section 4.2.2.

Boreholes for the piezometers were manually augered to a depth of up to 3 m. The piezometers were constructed of 50 mm (ID) Class 18 PVC pipe, (machine) screened from the base to within 0.1 m of ground level, and wrapped in 120-150 μm filter sock. The casing extended to around 0.5 m above the ground surface. After inserting each piezometer, boreholes were backfilled with clean quartz sand², with a 0.1 m thick bentonite seal³ at the surface to limit local surface and groundwater interaction.

At the time of piezometer installation, the borehole geology and piezometer construction details were recorded. Representative sediment samples (1-2 kg) were collected at several depth intervals (based on geology) for laboratory analyses, as described in Section 4.2.2. Baseline and ongoing monitoring of groundwater levels and water quality in the piezometers were conducted as described in Sections 4.2.6 and 4.2.7.

The horizontal coordinates and elevations of each piezometer site were surveyed to an accuracy of ± 5 mm by Daishat Geodetic Surveyors from 15-17 September 2009⁴. This enabled groundwater level measurements to be referenced to a common datum (m AHD).



Plates 7-9. Piezometers installed in upper Currency Creek (UCC-P1 and UCC-P3) and lower Currency Creek (LCC-P2).

² Washed, dried and graded sand, with particle size range 1-2.5 mm (Unimin 8/16 silica).

³ Dry sodium bentonite powder (Unimin product code BCN268-4MI).

⁴ Surveys were conducted using a Real Time Kinematic (RTK) Global Positioning System (GPS). A Leica survey level was also used to ensure a high level of security in the results. Piezometers were connected to local benchmarks and datums via GPS.

Point Sturt, Campbell Park and Windmill locations

A total of 23 piezometers were installed at the Point Sturt, Campbell Park and Windmill locations, as shown in Figure 6. The piezometers were installed from 20-25 August 2009 by Earth Systems personnel, to enable long-term groundwater level and water quality monitoring in the lake sediments.

The Geoprobe 7720DT drill rig, as described in Section 4.2.2, was required for the installation of piezometers at the Point Sturt, Campbell Park and Windmill locations (Plates 10-13). The piezometers were installed adjacent to the push-tube core sample sites as shown in Figure 6. The Geoprobe system, utilising hollow auger rods, enabled the installation of piezometers in the saturated lake sediments to depths of up to 5.2 m.

At each location, the piezometers were positioned along a transect perpendicular to the shoreline. Each transect consisted of 4 sites, spaced at 75 m intervals at Point Sturt (transect length 225 m) and 50 m at the Campbell Park and Windmill locations (transect length 150 m). The transect lengths were maximised according to the length of exposed shoreline sediments at the time of piezometer installation.

At each site, up to three piezometers were installed, targeting one or more depth intervals of interest, with a horizontal spacing of 3 m (parallel to the shoreline) between each piezometer. This was intended to minimise any potential for localised interferences between adjacent piezometers along the dominant groundwater flow direction (perpendicular to the shoreline, towards the lake). Furthermore, the sediment profile was less likely to vary along the contour (parallel to the shoreline) than perpendicular to the contour. The 3 m spacing represented a compromise between minimising any interference from adjacent piezometers on groundwater levels and water quality, while ensuring the push-tube core sample sites were representative of the piezometer installation sites.

The depth interval(s) of interest for each piezometer site were determined according to observations made during logging of the corresponding push-tube core. The prime objective was to target the surficial layer of sandy sediments which generally corresponded to the upper 1-2 m of the sediment profile. These sediments were considered to represent the greatest short term ASS risk as lake water levels (and hence groundwater levels in shoreline sediments) continue to decline over the next 1-2 years. Furthermore, as the upper sandy layers will be the first to desaturate as lake levels decrease, monitoring of the groundwater quality in these layers will provide an early warning of any acidity generation and potential flux to the lakes. A low permeability (clay) layer was observed at most sites (although less evident at the Windmill location), and at these sites the piezometer depth intervals of interest corresponded to the sandy sediments (aquifers) above and below these clay layers, to enable investigation of the degree of connectivity (if any) between the upper and lower aquifers, by comparison of piezometric levels⁵ and groundwater quality. The clay layers were not screened due to their inherently low permeability, which would preclude collection of groundwater samples⁶.

Of the 23 piezometers installed, 20 were screened in lake sediments while the remaining three were screened in the underlying sands of the Bridgewater Formation, as shown in Table 3. The Bridgewater Formation represents a source of alkaline groundwater with the potential to neutralise acidity generated from sulfides in the lake sediments. Piezometer installations in the Bridgewater Formation were therefore intended to enable investigation of the hydraulic connectivity of this aquifer with overlying lake sediments, and assessment of the potential for this source of groundwater to decrease acidity fluxes to the lakes, either naturally or via passive/active management intervention.

During any piezometer installation process in sediments containing clay horizons, there is potential for some clay particles to be transferred or 'smeared' through the sediment profile as the auger is raised to the ground surface. Smearing of clays along the surface of the augered hole can subsequently create localised artificial effects on hydraulic conductivity and groundwater quality. This issue was identified during piezometer installation via manual augering (solid auger) at Currency Creek. The Geoprobe augering method (hollow auger) enabled more controlled piezometer installation and auger removal, and was therefore less disruptive to the surrounding sediments, thus minimising the risk of clay smearing and the potential for localised effects on hydrogeology and water quality.

The piezometers were constructed of 50 mm (ID) Class 18 PVC pipe, with screened sections corresponding to lake sediments or the underlying Bridgewater Formation, depending on the aquifer of interest (refer to geological logs in Attachment F). The screened sections were wrapped in a 120-150 µm filter sock and the casing extended to around 0.15 m above the ground surface. After inserting each piezometer, boreholes

⁵ It is important to distinguish between "groundwater levels" and "piezometric levels". Groundwater level simply refers to the natural elevation of the groundwater surface or "water table", whereas piezometer level refers to the water surface elevation measured in a piezometer, which may be higher than the surrounding natural groundwater level. In an unconfined aquifer (such as the upper lake sediments), the measured piezometric levels provide a good indication of actual groundwater levels. However, in a confined aquifer (eg. sands overlain by an extensive low permeability clay layer) the piezometric level may be greater than the actual groundwater level.

⁶ The permeability of clays is around 4-6 orders of magnitude lower than that of fine sands such as those typical of the lake sediments. While the "bulk" permeability of clays may increase significantly if desiccation cracks are present, the potential for clay desiccation of subsurface clays is considered low, given the presence of at least 1 m of overlying sand in the sediments exposed at the time of piezometer installation (ie. desiccation of subsurface clay layers will be retarded by moisture retention in the overlying sands).

were backfilled with clean, coarse, quartz sand⁷, with a bentonite seal at the surface to limit surface and groundwater interaction. Silica coated sodium bentonite pellets were used to backfill saturated sediment layers, to minimise the risk of rapid hydration (clogging) prior to auger removal; standard ¼" sodium bentonite chips were otherwise used. Bentonite was sourced from Australian Mud Company.

Detailed piezometer design including the number of sites at each location, the depth of each piezometer, and the length of screened and unscreened zones, was based on the core profiles obtained at 4 points along each transect (Section 4.2.2). Key piezometer installation details are summarised in Table 3.

At the time of piezometer installation, the borehole geology and piezometer construction details were recorded. Representative sediment samples (1-2 kg) were collected at several depth intervals (sediment horizons) for laboratory analyses as described in Section 4.2.2. Baseline and ongoing monitoring of groundwater levels and water quality in the piezometers were conducted as described in Sections 4.2.6 and 4.2.7.

The horizontal coordinates and elevations of each piezometer site were surveyed to an accuracy of ±5 mm by Daishat Geodetic Surveyors from 15-17 September 2009. This enabled groundwater level measurements to be referenced to a common datum (m AHD).

⁷ Washed, dried and graded sand, with particle size of approximately 2.5 mm (Unimin 8/16 silica).



Plates 10-11. Geoprobe 7720DT drill rig used for sediment sampling and piezometer installation at Point Sturt (pictured), Campbell Park and Windmill locations.



Plate 12. Piezometer installation at Campbell Park, Lake Albert.



Plate 13. Hollow auger system used for piezometer installation. After inserting the piezometer to the required depth, the auger hole was backfilled with clean sand around the screened section and then capped with bentonite, whilst progressively removing the hollow auger rods.

Table 3. Piezometer installation details (see Attachment F for detailed geological and piezometer construction logs).

Site ID*	Easting#	Northing#	Distance along Transect (m)	Elevation (m AHD)					Screen Length (m)	Screened geology (see detailed logs in Attachment F)
				Top of Casing	Ground Surface	Base of piezometer	Top of screen	Base of screen		
<i>Currency Creek</i>										
UCC-P1	299269	6074127	n/a	1.215	0.465	-2.635	0.265	-2.635	2.9	Lake sediments
LCC-P2	301320	6072955	n/a	1.272	0.321	-2.279	0.221	-2.179	2.4	Lake sediments
UCC-P3	299589	6073004	n/a	1.205	0.635	-1.766	0.535	-1.666	2.2	Lake sediments
<i>Point Sturt</i>										
PS-1S	321172	6070261	0	0.514	0.334	-1.046	0.174	-1.026	1.2	Lake sediments
PS-1D	321169	6070263	0	0.411	0.286	-2.195	-1.575	-2.175	0.6	Lake sediments
PS-2S	321203	6070329	75	-0.011	-0.141	-1.241	-0.321	-1.221	0.9	Lake sediments
PS-2D	321200	6070331	75	-0.001	-0.166	-2.526	-1.466	-2.506	1.0	Lake sediments
PS-3S	321234	6070398	150	-0.308	-0.398	-1.593	-0.673	-1.573	0.9	Lake sediments
PS-3D	321231	6070399	150	-0.190	-0.375	-2.385	-1.465	-2.365	0.9	Lake sediments
PS-4S	321265	6070466	225	-0.358	-0.488	-1.608	-0.688	-1.588	0.9	Lake sediments
PS-4D	321263	6070468	225	-0.289	-0.479	-2.229	-1.479	-2.209	0.7	Lake sediments
<i>Campbell Park</i>										
CP-1S	341219	6056466	0	0.464	0.284	-0.636	0.134	-0.616	0.8	Lake sediments
CP-1D	341216	6056466	0	0.439	0.279	-3.341	-1.421	-3.321	1.9	Lake sediments
CP-2S	341215	6056515	50	0.176	-0.009	-1.229	-0.159	-1.209	1.1	Lake sediments
CP-2D	341212	6056515	50	0.137	-0.003	-3.123	-2.253	-3.103	0.9	Lake sediments
CP-3S	341211	6056565	100	0.183	0.018	-1.202	-0.132	-1.182	1.1	Lake sediments
CP-3D	341208	6056564	100	0.083	-0.052	-3.372	-2.552	-3.352	0.8	Lake sediments
CP-4S	341207	6056615	150	0.006	-0.159	-1.379	-0.409	-1.359	1.0	Lake sediments
CP-4M	341204	6056615	150	-0.014	-0.149	-2.319	-1.949	-2.299	0.4	Lake sediments
CP-4D	341210	6056615	150	0.025	-0.145	-4.565	-2.545	-4.545	2.0	Bridgewater Formation

Site ID*	Easting#	Northing#	Distance along Transect (m)	Elevation (m AHD)					Screen Length (m)	Screened geology (see detailed logs in Attachment F)
				Top of Casing	Ground Surface	Base of piezometer	Top of screen	Base of screen		
<i>Windmill</i>										
WM-1S	345597	6064184	0	0.146	0.008	-3.312	-0.192	-3.292	3.1	Lake sediments
WM-1D	345599	6064182	0	0.149	0.009	-5.211	-3.891	-5.191	1.3	Bridgewater Formation
WM-2S	345570	6064142	50	0.120	-0.044	-2.865	-0.244	-2.845	2.6	Lake sediments
WM-3S	345543	6064100	100	0.049	-0.111	-2.331	-0.261	-2.311	2.1	Lake sediments
WM-4S	345516	6064058	150	0.031	-0.119	-1.939	-0.269	-1.919	1.7	Lake sediments
WM-4D	345519	6064056	150	-0.006	-0.168	-3.188	-2.168	-3.168	1.0	Bridgewater Formation

* PS = Point Sturt, CP = Campbell Park, WM = Windmill. Numbers 1-4 represent position along the transect, from the shore (1) towards the lake water (4).

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4.2.4 Groundwater (piezometric) level sensor installation, rising head tests and calculation of hydraulic conductivity

Currency Creek

On 14-15 May 2009, rising head tests were conducted at piezometers UCC-P1 and LCC-P2 to enable the calculation of bulk hydraulic conductivity values at these sites. The rising head tests involved manual purging of each piezometer and subsequent monitoring of groundwater rebound rates over time. *In-Situ LevelTroll 500* level loggers⁸ were installed in the base of each piezometer and configured to monitor groundwater (piezometric) levels at 15 minute intervals throughout the rising head tests. The sensors remained installed in UCC-P1 and LCC-P2 for long term monitoring of piezometric levels.

LevelTroll equipment at Currency Creek, as well as Lake Albert and Lake Alexandrina (see below), was used for long term logging of the piezometric level at 15 minute intervals to allow comparison of piezometric levels with the 15 minute rainfall data available for Currency Creek, Langhorne Creek (for the Point Sturt sites) and Narrung (for the Campbell Park and Windmill sites). This frequency of logging was necessary to understand the key factors influencing groundwater levels, such as rainfall, evapotranspiration, creek/lake levels, and wind/seiche events, etc. The *In-Situ LevelTroll 500* has a battery life of up to five years and a memory capacity that is sufficient to store 15 minute interval piezometric level data collected over several months.

Piezometric level data were downloaded from the instruments at UCC-P1 and LCC-P2 on five occasions – 31 May, 10 July, 19 August, 14 September and 18 November 2009. On each occasion, manual measurements of piezometric level were obtained with a groundwater level dipper to confirm the accuracy of logged data. Raw piezometric level data are measured as a depth of water (mm) above the LevelTroll sensor plate, which is positioned approximately 20 mm from the base of the instrument. After downloading the data, the raw piezometric level measurements were subsequently converted to a common datum (m AHD) using accurate survey data collected by Daishat Geodetic Surveyors.

Point Sturt, Campbell Park and Windmill locations

Rising head tests for the Point Sturt, Campbell Park and Windmill piezometers were conducted from 26-30 August 2009. The piezometers were manually purged and *In-Situ LevelTroll 500* pressure sensors were used to monitor groundwater rebound rates over time, as described above (Plates 14-15). However, rising head test data were logged more frequently (1-5 second intervals) to account for the relatively rapid groundwater recovery in several piezometers relative to those in Currency Creek⁹.

Following collection of the rising head test data, the sensors were re-configured for long term monitoring of piezometric levels at 15 minute intervals. The sensors were installed in the uppermost piezometers along each transect, thus four sensors were installed at each location. Priority was given to the uppermost sediments as these represent the most significant short term acidity generation risk as lake levels recede. Furthermore, the piezometric levels in the uppermost sediments provide a more accurate representation of actual groundwater levels than the deeper sandy horizons, at least where a confining clay layer is present between the upper and lower sand horizons. (Nevertheless, the level sensors may be transferred to the deeper piezometers as groundwater levels recede over time and the upper piezometers progressively dry out.)

Piezometric level data were downloaded from the instruments onto a laptop computer in the field at Point Sturt, Campbell Park and Windmill from 15-17 September, 19-21 October and 16-18 November 2009 (Plate 16). Manual measurements of piezometric level were obtained with a groundwater level dipper to confirm the accuracy of logged data (Plates 17-18). Piezometric level measurements were subsequently converted to a common datum (m AHD) using accurate survey data collected by Daishat Geodetic Surveyors.

⁸ Level loggers are used in situations where continuous piezometric level data are required to be logged at set time intervals. In-Situ LevelTroll (LevelTroll) instruments measure the hydrostatic pressure above a pressure sensing plate, which is directly related to the depth of water above the plate. Based on the depth of the sensor within a piezometer, the piezometric level can be determined. However, the hydrostatic pressure level is subject to atmospheric pressure, which distorts the reading and this distortion must be taken into account when measuring the piezometric level. A LevelTroll with vented cable allows atmospheric pressure to be applied to the back of the pressure sensor, cancelling out the effect of atmospheric pressure on the piezometric level and thus gives the true piezometric level. Sensors that are not vented require barometric readings to adjust the piezometric level for atmospheric pressure affects. The In-Situ LevelTroll 500 instruments were fitted with vented cables so do not require barometric pressure readings to accurately measure piezometric levels. The In-Situ LevelTroll 500 has the highest level of accuracy of any marketed water level logger in Australia of +/- 0.05 % at 15° C and is constructed from materials that enable it to withstand acidic groundwater.

⁹ The faster groundwater recovery rates in the Point Sturt, Campbell Park and Windmill piezometers may have been partly related to the hollow auger installation technique which minimised the potential for clay smearing during piezometer installation (see Section 4.2.3). The manual augering technique used at Currency Creek may have inadvertently resulted in localised reductions in groundwater recovery rates.



Plates 14-15. Installation of groundwater (piezometric) levels sensors.



Plate 16. Data download from groundwater (piezometric) levels sensors.

Calculation of hydraulic conductivity

Hydraulic conductivity software (AQTESOLV) was used to calculate the hydraulic conductivity of the saturated sediments surrounding the screened section of each piezometer. The downloaded rising head test data (time vs. displacement), piezometer construction details (casing radius, piezometer radius, screen length, gravel pack porosity) and assumed aquifer thickness was entered into the software. The hydraulic conductivity was calculated using the Bouwer-Rice Method for unconfined aquifers.

A maximum of 25 data points from the rising head test data were used, evenly distributed between the start of the rising head test and the point of full groundwater level rebound. The following assumptions were made when calculating hydraulic conductivity:

- The point of rebound was assumed to be the time at which the water level was no longer continuously rising.
- The initial displacement was assumed to be the difference between the recorded static groundwater level and the first reading from the rising head test.
- The porosity of the quartz sand (2.5 mm grain size) placed around the screened length of each piezometer was assumed to be 45%.
- The thickness of the Bridgewater Formation was assumed to be 15 m¹⁰.



Plates 17-18. Verification of logged groundwater (piezometric) levels via manual dipping of piezometers.

4.2.5 Moisture sensor installation

To characterise the sediment moisture profile (variation in sediment moisture content with depth) between the ground surface and the water table level and investigate trends in sediment moisture content over time, Sentek EnviroSCAN moisture monitoring systems were installed at each of the Point Sturt, Campbell Park and Windmill locations, as shown in Figure 6. This information was required to understand the relationship between sediment moisture and key factors influencing moisture content, including groundwater levels, rainfall and evapotranspiration. Moisture content profiles represented a key input to the acidity generation modelling process (refer to Section 7) as sulfide oxidation rates in sediments above the groundwater table have the potential to vary significantly with moisture content.

¹⁰ Local hydrogeological surveys have been undertaken by CSIRO in the Angas Bremer Plains area to the west of Lake Alexandrina (Cresswell and Herczeg, 2004), and the Cooke Plains area to the north east of Lake Albert (Pavelic et al, 1997). The survey undertaken in the Angas Bremer Plains area estimated the unconfined limestone aquifer thickness to be 10-35 m, becoming thicker with distance from Lake Alexandrina, while the Cooke Plains survey indicated that an aquifer thickness of 20 m was common. An average aquifer thickness for the Bridgewater Formation of 15 m was used for calculating hydraulic conductivity where the screened zone of the piezometer was in this formation.

Sentek moisture monitoring systems are used extensively in commercial environmental, agricultural, research, mining, landfill and forestry applications worldwide. Sentek moisture systems have gained international credibility, through published peer review, among world leading research agencies and commercial enterprises.

Each system comprises multiple moisture sensors connected to a data logging facility, which is positioned within a readily serviceable PVC access tube inserted into lake sediments (Plates 19-20). The data logging facility enables flexible logging intervals ranging from 1 minute to over 16 hours. Each moisture probe is connected by a 2 m long cable to an EnviroSCAN head unit with a download port. The EnviroSCAN head unit is powered by four AA alkaline batteries (heavy duty) and is stored adjacent to the moisture sensors in weather-proof casing. Data are downloaded to a laptop computer in the field via download cable from the EnviroSCAN head unit to a laptop (Plate 21). Battery life is primarily dependent on the number of sensors installed and the frequency of measurement.

The EnviroSCAN sensor technology utilises Frequency Domain Reflectometry (FDR) to measure moisture content. Multiple sensors can be attached to each system with flexible depth placement at 10 cm intervals. The moisture content of the sediments is recorded as an average volumetric sediment water content (vol% H₂O) (+/- 0.003% vol% H₂O) measured from sediment lying 5 cm above and 5 cm below a 10 cm perpendicular (radial) distance from each sensor. For example, a sensor located at a depth of 10 cm below ground measures the average volumetric sediment water content from within the sediment depth interval 5-15 cm that is within a 10 cm distance (radially) from the access tube.

Moisture monitoring systems were calibrated prior to installation by HydroTerra Pty Ltd according to the methodology outlined in the 'Calibration of Sentek Pty Ltd Soil Moisture Sensors' manual. Calibration of the soil moisture sensors was made by comparing Scaled Frequency readings from an access tube installed in a container in a laboratory with values of volumetric water content (0 and 100 vol % H₂O) determined gravimetrically from immediately adjacent to the tube. Calibration is not required in the field.

Moisture monitoring systems were installed according to the Sentek Access Tube Installation Guide using the standard manual installation method and Sentek precision installation tools. In this method the access tube hole is hand augered into the sediment and the access tube (56.5 mm diameter) is inserted into the sediment. The moisture probe (diameter 50.5 mm) is subsequently placed within the access tube. This method prevents the formation of air pockets along the length of the access tube and causes minimum disturbance to the surrounding sediment profile.

Moisture monitoring systems were installed at Point Sturt, Campbell Park and Windmill. Characterisation of sediment moisture profiles at one site per location and comparison between each site was deemed sufficient for acidity generation modelling process. Moisture monitoring systems were installed closest to the original shoreline where groundwater levels were most subdued. This was to ensure maximum data recovery with depth and to minimise the risk of electrical damage to the sensors from groundwater. At the Windmill site, installation of the moisture monitoring system was delayed until 20 October 2009 due to elevated groundwater levels. Additional sensors can be installed in deeper sediments when groundwater levels subside. Table 4 shows the dates when sensors were installed at each site.

The sensors were configured for long term monitoring of sediment moisture content at 1 hour intervals. This monitoring interval was considered sufficient to identify key trends in moisture content over time, and represented a compromise between maximising the resolution of data and maintaining sufficient data storage capacity and battery life for operation of the sensors and logging facility between download events. New data can overwrite previously logged data if the memory capacity is exceeded prior to a download event. The number of days before overwriting is dependant on the sample interval. A worst case scenario of 10 minute sampling gives approximately 14 days before download is required.

Moisture content data were downloaded from the instruments onto a laptop computer in the field 15-17 September, 19-21 October and 16-18 November 2009.

The horizontal coordinates and elevations of each moisture system were surveyed to an accuracy of ±5 mm by Daishsat Geodetic Surveyors from 15-17 September 2009.

Table 4. Sediment moisture monitoring details (moisture content data logged hourly).

Moisture sensor position (m below ground surface)	Point Sturt	Campbell Park	Windmill
10 cm (5-15 cm)	28 August 2009 – present	28 August 2009 – present	20 October 2009 – present
20 cm (15-25 cm)	28 August 2009 – present	28 August 2009 – present	20 October 2009 – present
30 cm (25-35 cm)	15 September 2009 – present	28 August 2009 – present	20 October 2009 – present
40 cm (35-45 cm)	15 September 2009 – present	16 September 2009 – present	20 October 2009 – present
50 cm (45-55 cm)	n/a*	16 September 2009 – present	n/a*
60 cm (55-65 cm)	n/a*	n/a*	n/a*

* Sensor installation postponed due to elevated groundwater levels.



Plate 19. Installation of moisture probe into PVC access tube at Campbell Park.



Plate 20. Moisture probe comprising data logging facility and 3 moisture sensors spaced at 10 cm vertical intervals in preparation for installation at Campbell Park (August 2009). Two additional sensors were installed in September 2009.



Plate 21. EnviroScan head unit with download port and power supply for moisture sensors and logging facility. Sediment moisture data is collected at 1 hour intervals in 10 cm depth intervals at the Point Sturt, Campbell Park and Windmill locations (see Figure 6).

4.2.6 Baseline water quality monitoring

Currency Creek

Baseline water quality monitoring at Currency Creek consisted of:

- Field measurement of general water quality parameters including pH, EC, temperature and ORP, in groundwater/pore water, ponded surface water and crack water (as described in Section 4.2.2) from 30 April to 2 May 2009.
- Field measurement of general water quality parameters (pH, EC, temperature and ORP) at multiple depths in piezometers UCC-P1, LCC-P2 and UCC-P3, to establish baseline groundwater quality profiles on 2 May 2009 and 31 May 2009.
- Purging of piezometers UCC-P1, LCC-P2 and UCC-P3 to enable collection of fresh groundwater for monitoring of general water quality parameters (pH, EC, temperature and ORP) on 2 May 2009.
- Collection of filtered and unfiltered samples on 2 May 2009 (after purging) for laboratory analysis of:
 - General water quality parameters (pH, EC); unfiltered.
 - Alkalinity/acidity; unfiltered.
 - Major ions (Na, K, Mg, Ca, Cl and SO₄); unfiltered.
 - Dissolved metals (Al, As, Cu, Fe, Mn, Pb, Zn); field filtered.
 - Nutrients (total N, NO₃, total P); unfiltered.

Laboratory analyses of water samples were conducted by Australian Laboratory Services (ALS) in Melbourne (NATA registered laboratory).

Point Sturt, Campbell Park and Windmill locations

Baseline water quality monitoring at the Point Sturt, Campbell Park and Windmill locations was conducted from 26-30 August 2009 and consisted of:

- Purging and field measurement of general water quality parameters (pH, EC, temperature and ORP) at multiple depths in all piezometers to establish baseline groundwater quality profiles.
- Purging of all piezometers to enable collection of fresh groundwater for monitoring of general water quality parameters (pH, EC, temperature and ORP).
- Collection of filtered and unfiltered samples (after purging) for laboratory analysis of:
 - General water quality parameters (pH, EC); unfiltered.
 - Alkalinity/acidity; unfiltered.
 - Major ions (Na, K, Mg, Ca, Cl and SO₄); unfiltered.
 - Dissolved metals (Al, As, Cd, Cu, Fe, Mn, Ni, Pb, Se, Zn); field filtered.
 - Nutrients (total N, total P); unfiltered.

Laboratory analyses of water samples were conducted by Australian Laboratory Services (ALS) in Melbourne (NATA registered laboratory).

4.2.7 Ongoing monitoring

The ongoing field monitoring program comprised the following:

- Field measurement of groundwater quality profiles (Plates 22-23) in the Currency Creek piezometers (19 August, 14 September, 21 October and 18 November 2009) and Point Sturt, Campbell Park and Windmill piezometers (15-17 September, 19-21 October and 16-18 November 2009).
- Purging to enable collection of fresh groundwater for monitoring of general water quality parameters (pH, EC, temperature and ORP) and field acidity/alkalinity in the Currency Creek piezometers (19 August, 14 September, 21 October and 18 November 2009) and Point Sturt, Campbell Park and Windmill piezometers (15-17 September, 19-21 October and 16-18 November 2009). See Plates 12-13.

- Collection of filtered and unfiltered samples after purging the equivalent of at least 3-5 bore volumes, where possible (Plates 24-25) from the Currency Creek piezometers (19 August, 14 September, 21 October and 18 November 2009) and Point Sturt, Campbell Park and Windmill piezometers (15-17 September, 19-21 October and 16-18 November 2009) for laboratory analysis of:
 - General water quality parameters (pH, EC); unfiltered.
 - Alkalinity/acidity; unfiltered.
 - Major ions (Na, K, Mg, Ca, Cl and SO₄); unfiltered.
 - Dissolved metals (Al, As, Cd, Cu, Fe, Mn, Ni, Pb, Se, Zn); field filtered.
 - Nutrients (total N, total P); unfiltered.
- Download of 15-minute interval groundwater (piezometric) level data from sensors installed at the Currency Creek, Point Sturt, Campbell Park and Windmill locations (15-17 September, 19-21 October and 16-18 November 2009).
- Download of 15-minute interval moisture data from sensors installed at Point Sturt, Campbell Park and Windmill locations (15-17 September, 19-21 October and 16-18 November 2009).



Plates 22-23. Field measurement of general water quality parameters at multiple depths in the sediment profile.





Plates 24-25. Collection of bulk groundwater samples for laboratory analysis.



Plates 26-28. Field measurement of general water quality parameters (left) and acidity/alkalinity (below).



Groundwater sampling from discrete depth intervals in piezometers containing acidic groundwater was undertaken on 20-21 October 2009 to determine whether the technique could be employed to gather further information regarding stratification within the acidified upper sand horizons. Discrete interval sampling was undertaken at:

- Point Sturt (PS-2S) on 21 October 2009 at 0.79 m below ground level;
- Lower Currency Creek (LCC-P2) on 22 October 2009 at 0.2 m intervals from 0.25-1.05 m below ground level (inclusive).

A low-flow micro-purge sampling system (comprising a QED pneumatic bladder pump driven by compressed CO₂ and a QED MP10 digital controller unit with internal 12V power supply) was used for the discrete interval sampling (Plates 29-30). The principle of low-flow sampling is to purge water at a flow rate that is close to or less than the natural movement of water through the piezometer, permitting sampling from the surrounding formation rather than from stagnant water within the piezometer. Low-flow micro-purging is a standard methodology for sampling groundwater from discrete intervals of interest in the sediment profile, and is a cost-effective technique that is used widely for commercial environmental applications. Continuous monitoring of drawdown and general water quality parameters was undertaken during purging to ensure that samples were representative of the surrounding sediments. Stable drawdown during pumping indicates that water is being purged directly from the depth of interest at a rate equal to the recharge rate of the surrounding sediments corresponding to the level of drawdown. Drawdown was measured manually during purging and recorded. A TPS 90FLV multi parameter water quality meter was used to measure pH, temperature, EC and ORP during purging. Once water quality had stabilised, water samples were collected.



Plates 29-30. Above: All components of low-flow groundwater sampling kit: (1) Compressed CO₂ bottle; (2) QED MP10 controller; (3) Flow through cell for TPS water quality probe; (4) QED bladder pump; and (5) Steel cable for bladder pump and groundwater level sensor (for measurement of drawdown). Right: Low-flow sampling.



Baseline and ongoing results from the field monitoring program were supplemented by the following data:

- Rainfall, wind speed and wind direction data collected at 15 minute intervals at Langhorne Creek and Narrung, from 9:00 a.m. 17 August 2009 to 9:00 a.m. 23 November 2009, provided by SAMDBNRM (2009) or downloaded from:
<http://www.samdbnrm.sa.gov.au/Portals/7/AWMN/awsview.php>
- Rainfall data collected at 15 minute intervals at Currency Creek from 12:00 p.m. 30 April 2009 to 9:00 a.m. 25 November 2009, provided by SAMDBNRM (2009) or downloaded from:
<http://www.samdbnrm.sa.gov.au/Portals/7/AWMN/awsview.php>
- Hourly surface water level, wind speed and wind direction data collected at Lake Alexandrina at Beacon 97 (Site A4261133; 6:00 a.m. 19 September 2008 to 25 September 2009), and hourly wind speed and wind direction data collected at Lake Albert near Waltowa Swamp (Site A4261153; 10:00 a.m. 15 May 2008 to 25 September 2009), provided by the Department for Water (DFW) South Australia (DWLBC 2009) or downloaded from:
<http://e-nrims.dwlbc.sa.gov.au/swa/>
- Hourly surface water level data collected at Point McLeay (Site A4261156; 12:00 p.m. 22 September 2009 to 9:00 a.m. 24 November 2009) in Lake Alexandrina and near Waltowa Swamp (Site A4261153; 10:00 a.m. 15 May 2008 to 9:00 a.m. 24 November 2009) and Warringee (Site A4261155; 6:00 p.m. 16 December 2008 to 5:00 p.m. 24 November 2009) in Lake Albert, provided by DFW (DWLBC 2009).
- Hourly surface water level data collected at Currency Creek (Site A4261203; 5:00 p.m. 11 September 2009 to 5:00 a.m. 26 November 2009) downloaded from:
<http://e-nrims.dwlbc.sa.gov.au/swa/>

4.3 *Data analysis and acidity generation and flux rate modelling*

4.3.1 Laboratory testwork program data analysis

Results from the laboratory testwork program were analysed using proprietary software in order to establish a relationship between moisture content and sulfide oxidation rate for sands and clays.

4.3.2 Field monitoring program data analysis

Available field monitoring data were analysed in order to investigate:

- The relationship between rainfall and the moisture profile at 10 cm depth increments (in sandy sediments) on an hourly basis.
- The relationship between rainfall and groundwater (piezometric) levels at 15 minute intervals.
- The hydraulic gradient of groundwater along each transect at 15 minute intervals and hydraulic conductivity of lake sediments / Bridgewater Formation at each site.
- The hydraulic connectivity of lake sediments (upper and lower layers) and the Bridgewater Formation at each site and across each transect.
- The variation in water quality with depth, within lake sediments and the Bridgewater Formation at each site and across each transect.
- The temporal variation in water quality, within lake sediments and the Bridgewater Formation at each site and across each transect.

4.3.3 Acidity generation rate modelling

Currency Creek

An indicative estimate of the annual acidity generation rate from ASS in Currency Creek (tonnes H₂SO₄ equivalent / year) was obtained using three different methods, as outlined below:

- Field acidity measurements and estimates of water volume (surface water and pore water) in Currency Creek in May 2009 were converted to a total acidity flux which was assumed to correspond to 6 months of acidity generation from all creek sediments (the creek dried during the 2008-2009 summer).
- The mass of limestone added to Currency Creek during the 2009 wet season (to account for the acidity load flux over 6 months) was converted to an equivalent annual acidity flux.
- A three-dimensional finite element model was developed to estimate acidity generation rates using geological profiles, Acid-Base Accounting data, groundwater level data, surface water level predictions, moisture profile information and sulfide oxidation rate vs. sediment moisture curves. The model was used to represent the drying of Currency Creek by lowering the surface water from a depth of 1 m to 0 m and subsequent re-filling within a 1 year period. Key assumptions and inputs used in developing the model are described in Section 7.

The results obtained using each method were compared to confirm the reliability of the estimates.

Lake Albert and Lake Alexandrina

Laboratory testwork results and available field monitoring data from the Point Sturt, Campbell Park and Windmill transects, were integrated in a three-dimensional finite element model to estimate daily acidity generation rates per 100 m length of shoreline for Lake Alexandrina and Lake Albert (tonnes H₂SO₄ / 100 m / unit time) over a 22 month period.

Key assumptions and inputs used in developing the models are described in Section 7.

The models were used to investigate acidity generation rates and acidity flux rates for the following scenarios:

- Partial drying of Lake Albert from -0.163 m AHD over 5-6 months and subsequent maintenance of the water level at -1.0 m AHD.
- Partial drying of Lake Alexandrina from -0.776 m AHD to -1.400 m AHD over 6 months (remaining constant for 1 month), followed by increasing water levels to -1.299 m AHD over 4 months and subsequent drying to -2.043 m AHD over 8 months.

These scenarios are consistent with forecast lake water levels provided by DFW (DWLBC 2009). As the scenarios are subject to change over time, the models have been developed such that input parameters are flexible and can be readily updated to enable:

- Investigation of alternative water level forecasts.
- Update of existing models as more accurate data become available.
- Investigation of the potential suitability of different ASS management options.

4.3.4 Acidity flux rate modelling

The University of Western Australia (UWA) was engaged to conduct the hydrogeological modelling using HYDRUS-2D to model two-dimensional water movement through variably saturated sediments. The model is based on a finite volume solution of the two-dimensional Richard's equation, as described further by Coletti and Hipsey (2010).

Results of the hydrogeological modelling were subsequently used to convert the outputs of the finite element acidity generation rate model (Section 4.3.3) into acidity flux rate estimates for Lake Albert and Lake Alexandrina.

4.4 *Implications for ASS management in the Lower Murray Lakes*

The implications of results from the laboratory testwork program, available field monitoring data and preliminary acidity generation and flux modelling results were considered in terms of short, medium and long term management of ASS in the Lower Murray Lakes. The implications for ASS management are discussed in Section 8.

5 Results

5.1 Laboratory testwork program

5.1.1 Static acid-base accounting testwork data for samples used in OxCon apparatus

Static acid-base accounting testwork data for samples used in the OxCon apparatus for the measurement of sulfide oxidation rates are presented in Attachment C and summarised in Table 5. Specifications and raw output data for the OxCon tests are provided in Attachment D.

Table 5. Static acid-base accounting testwork data for samples used in OxCon apparatus.

Parameter	Unit	Sand	Clay
Acid-Base Accounting data			
Net Acid Producing Potential (NAPP)	kg H ₂ SO ₄	1.1	28.1
pH after oxidation (NAG pH)	-	5.4	2.4
Net Acid Generation at pH 4.5 (NAG _{pH4.5})	kg H ₂ SO ₄	<0.1	20.7
Net Acid Generation at pH 7.0 (NAG _{pH7.0})	kg H ₂ SO ₄	2.0	26.4
Acid Neutralising Capacity (ANC)	kg H ₂ SO ₄	3.8	4.4
Sulfide Sulfur (S) *	wt% S	0.16	1.06
Total Organic Carbon (TOC)	wt% C	0.11	0.64
Particle size distribution			
Clay (<2 µm)	wt%	4	n/a
Silt (2-60 µm)	wt%	2	n/a
Sand (0.06-2.00 mm)	wt%	94	n/a

*Reported by laboratory as "Total Sulfur" but assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

5.1.2 Kinetic testwork data derived from OxCon apparatus

Table 6 provides estimated pyrite equivalent oxidation rates and acidity generation rates for each OxCon test. Figure 8 summarises the pyrite equivalent oxidation rates as a function of moisture content.

Key results are summarised below:

- Pyrite oxidation rates for Lower Lakes ASS ranged from 0-1.24 wt% available pyrite per day. This means that up to 1.24 wt% of all available pyrite exposed to atmospheric oxygen is converted to sulfuric acid per day. These results compare well with oxidation rates for ASS reported in or calculated from the literature (e.g. Ward et al., 2004a; Ward et al., 2004b; Rigby et al., 2006; Borba et al., 2003).
- Bulk oxygen consumption for fully saturated sands and clays was assumed to be wholly attributed to bacterial oxidation of organic carbon (i.e. pyrite oxidation rates for saturated sands and clays equals zero).
- Oxidation rates for the sands were highest for moisture contents ranging from 10.3 wt% water to 18.8 wt% water. These rates correspond to acidity generation rates of 0.05-0.06 kg H₂SO₄ per tonne of unsaturated sandy sediments per day.
- Oxidation rates for the sands were lowest for highly saturated sands (0.02 wt% available pyrite per day at a moisture content of 21.5 wt% water) and for dry sands (0.09 and 0.61 wt% available pyrite per day at moisture contents of 0 and 5.6 wt% water, respectively).

- Oxidation rates for clays were a linear inverse function of moisture content, for the range of moisture contents assessed (23-48 wt% water). (The clays could not be desaturated to moisture contents below 23 wt% water with the vacuum desiccation technique employed in the laboratory.)

Table 6. Pyrite equivalent oxidation rates and acidity generation rates for Lower Murray Lakes ASS.

Sample ID	Gravimetric moisture content (GMC)	Volumetric moisture content	Degree of saturation	Pyrite oxidation rate (linear)	Acidity generation rate (linear)	Time to oxidise available pyrite (linear)
	wt% water	vol% water in moist sample	% of pore volume occupied by water	wt% available pyrite / day	kg H ₂ SO ₄ / tonne / day	days
Sand (GMC 0.0%)	0.0%	0%	0%	0.09%	0.00	1167
Sand (GMC 5.6%)	5.6%	8%	24%	0.61%	0.03	165
Sand (GMC 8.0%)	8.0%	11%	34%	0.90%	0.04	112
Sand (GMC 10.3%)	10.3%	15%	44%	1.04%	0.05	96
Sand (GMC 14%)	14.0%	20%	60%	1.24%	0.06	80
Sand (GMC 18.8%)	18.8%	27%	80%	1.18%	0.06	85
Sand (GMC 21.5%)	21.5%	31%	91%	0.01%	0.00	6702
Sand (GMC 23.5%)	23.5%	33%	100%	0.03%	0.00	3160
Clay (GMC 22.5%)	22.5%	n/a	n/a	0.23%	0.08	429
Clay (GMC 22.7%)	22.7%	n/a	n/a	0.78%	0.25	128
Clay (GMC 31.4%)	31.4%	n/a	n/a	0.54%	0.17	186
Clay (GMC 38.2%)	38.2%	n/a	n/a	0.04%	0.01	2252
Clay (GMC 47.9%)	47.9%	n/a	n/a	0.00%	0.00	n/a

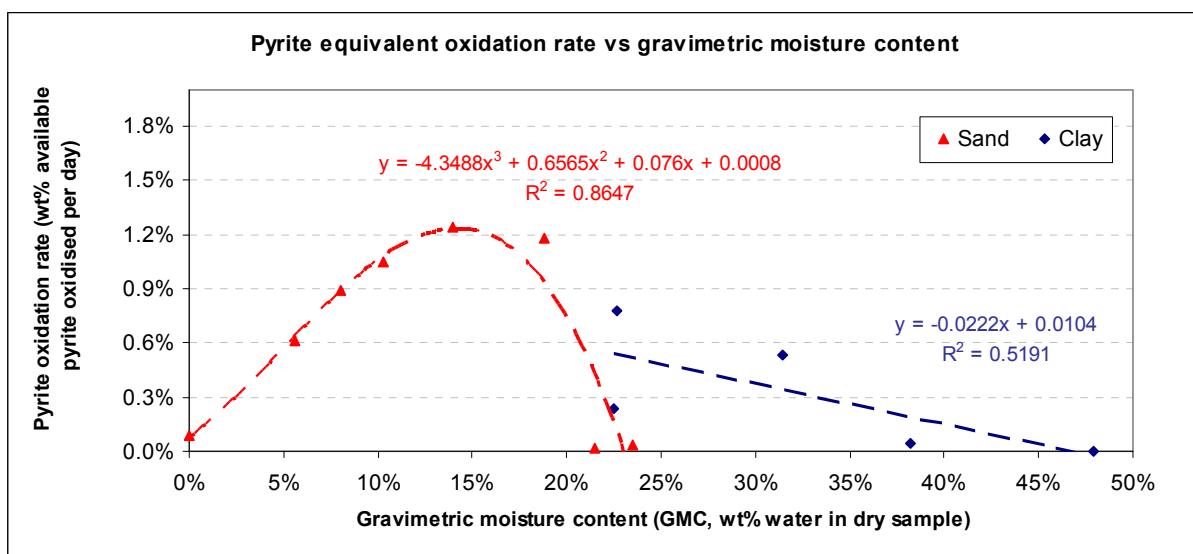


Figure 8. Pyrite equivalent oxidation rates vs. moisture content for Lower Murray Lakes ASS.

5.2 *Field monitoring program*

5.2.1 **Geology**

Geophysical survey results and interpretation are provided in Attachment E.

Geological and piezometer construction logs for the Currency Creek, Point Sturt, Campbell Park and Windmill locations are provided in Attachment F. Survey data are provided in Attachment G.

Geological cross-sections for the Point Sturt, Campbell Park and Windmill transects are provided in Figures 9 to 11, respectively.

Geological information gathered from drilling, shallow pitting and transient electromagnetic surveys (TEM) identified a coherent regional near-surface stratigraphy across both lakes. This stratigraphy is based on information collected within 500 m of the original shoreline of the lakes. The majority of the areas examined to date contain a thin veneer of unconsolidated lake sediments (1-3 m thick) overlying a calcrete/silcrete-capped Bridgewater Formation limestone. The lake sediments are generally composed of an uppermost 0.7-1.3 m thick layer of quartz-rich sand, which overlies a 0.2-1.0 m thick, occasionally calcareous clay layer. An additional quartz-rich sand layer may be found beneath the clay layer and immediately above the Bridgewater Formation. The Bridgewater Formation is an unconsolidated to poorly consolidated calcareous, quartz-rich sand that contains 5-30% carbonate. It is expected to be in excess of 10 m thick, and local onshore exposures indicate that it is likely to be up to 30 m thick at some locations.

At several locations, the Bridgewater Formation is exposed in the base of the lakes with essentially no significant capping of other sediments. At these sites, the Bridgewater Formation has been subjected to the same sulfate reducing bacterial processes as other lake sediments, and therefore contains elevated concentrations of diagenetic pyrite.

A combination of the near-shore geology collected in this study and the sediment surveys conducted by CSIRO indicate that the uppermost lake sands form a concentric wedge of sediment that thins toward the centre of the lake. In most instances, the sands progress from 1-2 m thick at the original shoreline, to 0 m thick approximately 1.5 km toward the centre of the lake.

It is expected that the thin layer of clay identified beneath the uppermost sand around the margins of the lakes thickens toward the centre of the lakes. This layer is believed to form a significant aquitard between the lake sediments and the Bridgewater Formation due to the presence of a small hydraulic gradient between these sand layers, across the clay layer.

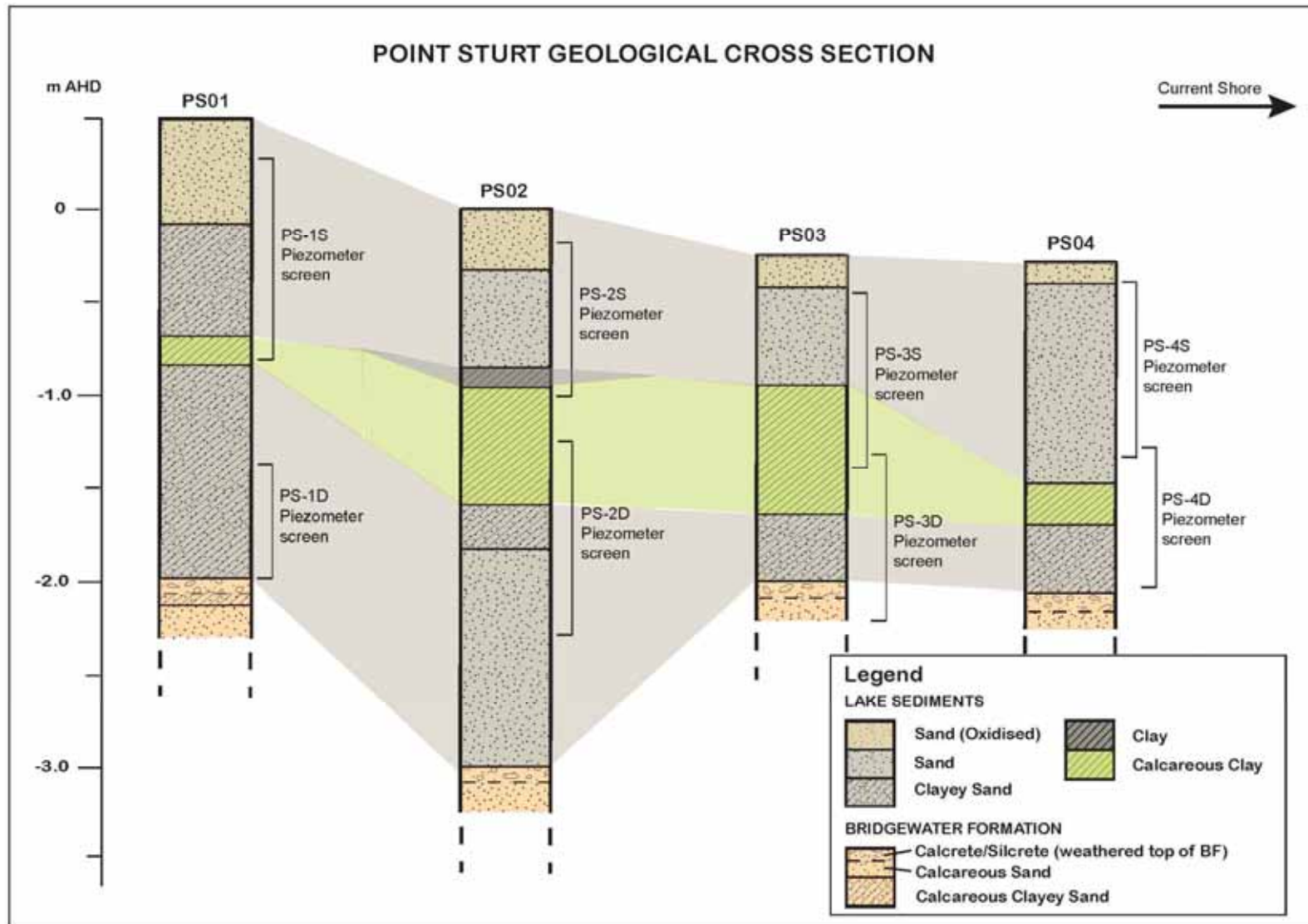


Figure 9. Point Sturt transect geology.

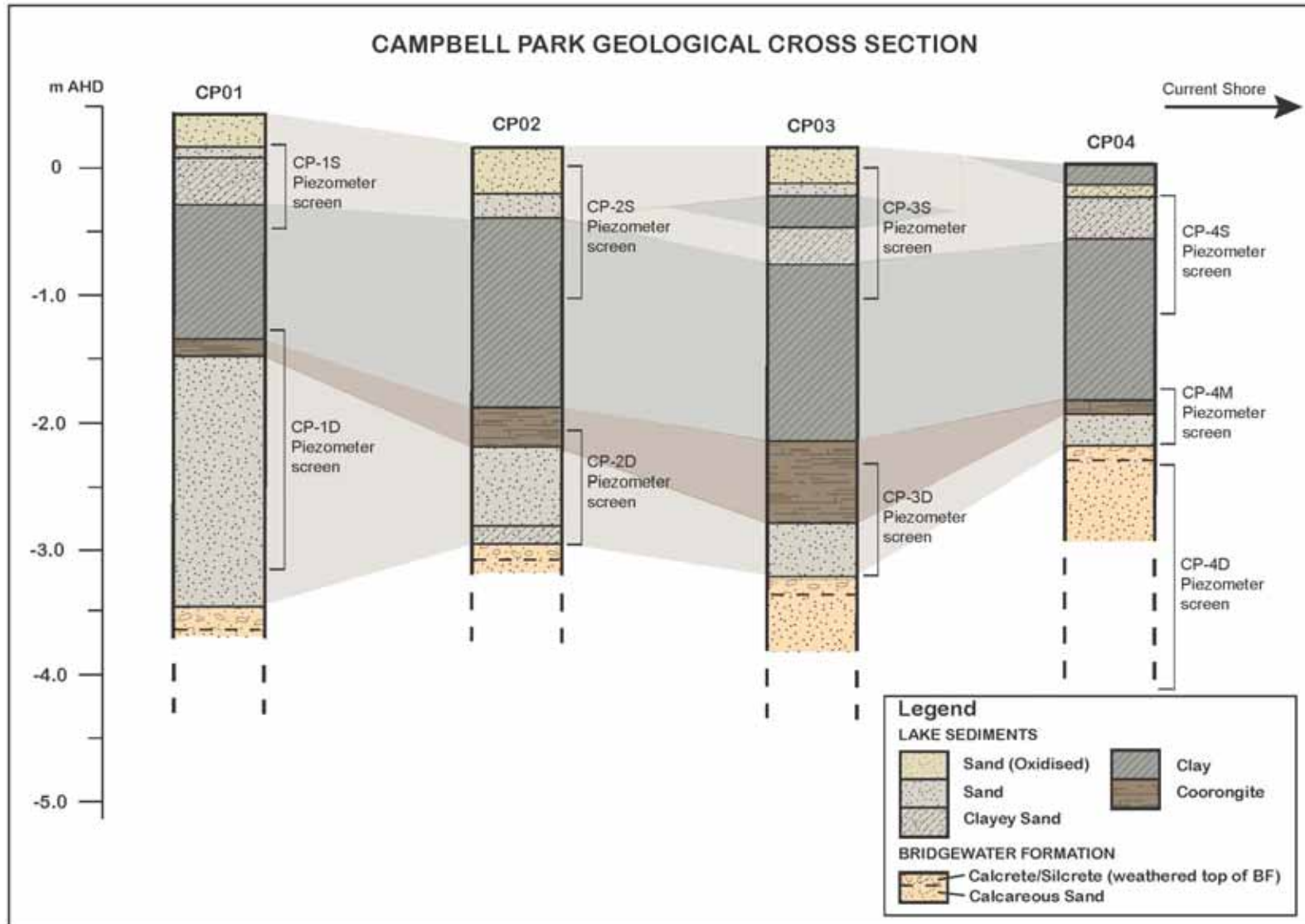


Figure 10. Campbell Park transect geology.

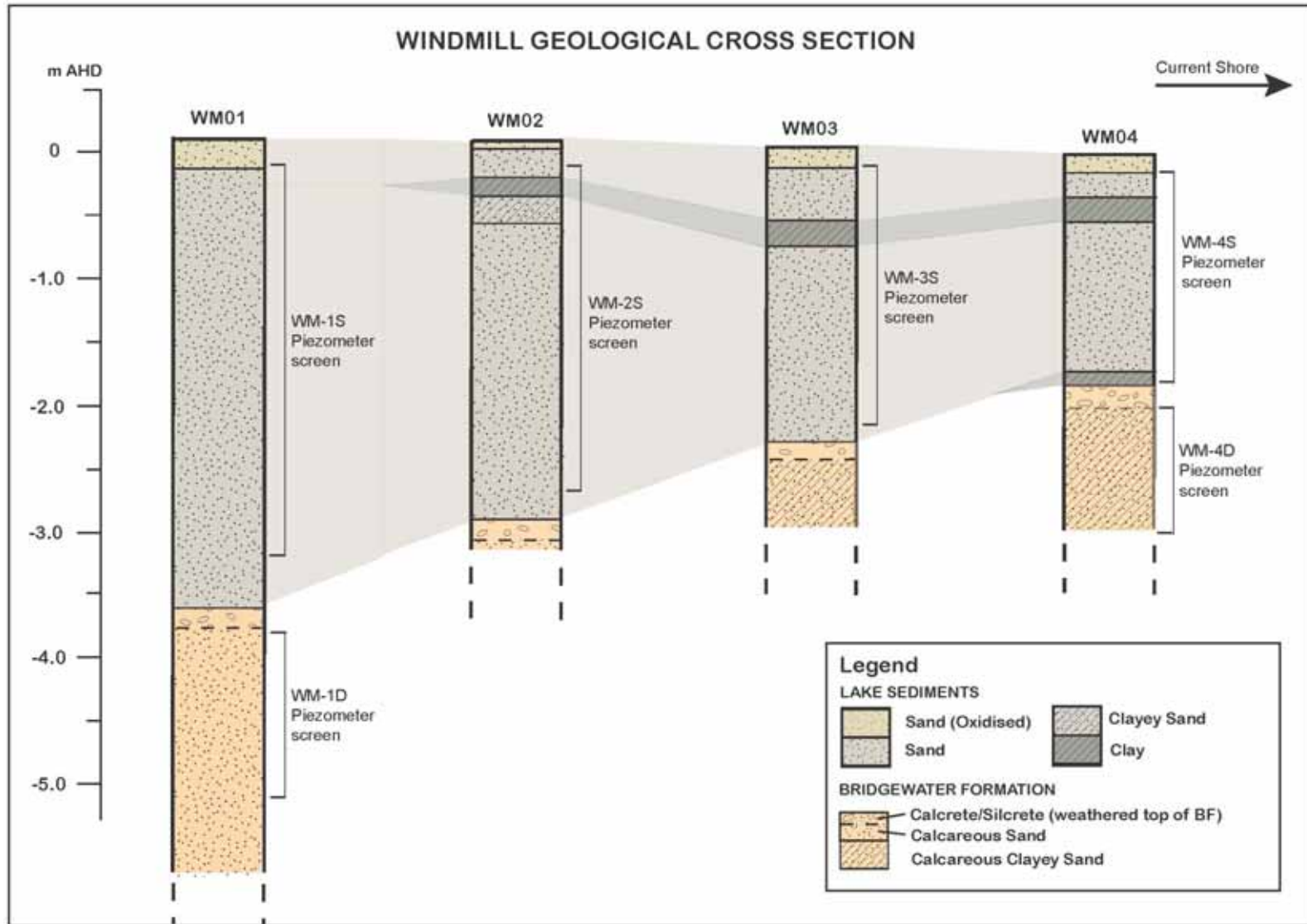


Figure 11. Windmill transect geology.

Currency Creek

Static acid-base accounting testwork data for selected sediment samples collected from the Currency Creek piezometer sites in May 2009 are presented in Attachment C and summarised in Table 7.

The key results are described below:

- Total sulfur ranged from <0.01 to 0.46 wt% S in UCC-P1, 0.02-0.19 wt% S in LCC-P2 and 0.04-0.50 wt% S in UCC-P3. Total sulfur values generally increased with depth below ground at all sites, with the highest concentrations associated with clay-rich sediments.
- Chromium reducible sulfur values typically represented between 50-100% of total sulfur values. The remaining 0-50% sulfur is assumed to be in the form of sulfate minerals, associated with sulfide oxidation.
- ANC values ranged from 1.9-6.4 kg H₂SO₄ / tonne in UCC-P1, <0.5-3.5 kg H₂SO₄ / tonne in LCC-P2 and <0.5-2.9 kg H₂SO₄ / tonne in UCC-P3. ANC values tended to increase with depth below ground at all sites, presumably influenced by the underlying Bridgewater Formation.
- NAPP values were generally in the range -2 to + 3 kg H₂SO₄ / tonne, with the exception of elevated NAPP values in UCC-P1 (7.6 kg H₂SO₄ / tonne; 0.8-3.0 m depth interval), LCC-P2 (5.8 kg H₂SO₄ / tonne; 1.0-2.2 m depth interval) and UCC-P3 (15.3 kg H₂SO₄ / tonne; 0.7-1.2 m depth interval).
- The lowest NAPP values were generally associated with upper layers of sediments where oxidation has taken place.
- Total organic carbon ranged from 0.04 to 0.46 wt% S in UCC-P1, 0.05-0.28 wt% S in LCC-P2 and 0.05-0.55 wt% S in UCC-P3. There was no consistent trend in carbon content with depth below ground.

Table 7. Static acid-base accounting testwork data for Currency Creek sediments (May 2009).

Sample ID	Geology (see Attachment F for detail)	Depth interval (m below ground)	Parameter				
			NAPP (kg H ₂ SO ₄ / tonne)	Total S (wt%S)	S _{cr} (wt%S)	ANC (kg H ₂ SO ₄ / tonne)	TOC (wt%C)
UCC-P1							
1	Fine sand, beige, dry (oxidised)	0-0.3	-1.9	<0.01	<0.02	1.9	0.04
2	Sand, light grey, moist (partly oxidised)	0.3-0.5	-1.9	0.03	<0.02	2.8	0.07
3	Sandy clay, dark grey, wet (reduced)	0.5-0.8	2.1	0.14	0.08	2.2	0.19
4	Clay, dark grey, wet (strongly reduced)	0.8-3.0	7.6	0.46	0.46	6.4	0.46
LCC-P2							
1	Medium sand, moist (partly oxidised)	0-0.1	-0.8	0.02	<0.02	1.4	0.07
2	Medium sand, grey, moist (reduced)	0.1-0.4	1.2	0.04	<0.02	<0.5	0.28
3	Clayey sand, grey, wet (reduced) overlying sandy clay, orange	1.0-2.2	5.8	0.19	0.10	<0.5	0.13
4	Clay, grey, dense (reduced)	2.2-2.5	<0.5	0.12	0.07	3.5	0.05
UCC-P3							
1	Medium sand, moist (oxidised)	0-0.1	2.3	0.13	<0.02	1.6	0.55
2	Sand, brown / grey / pink-orange, some clay, moist (partly oxidised)	0.1-0.7	<0.5	0.05	<0.02	1.9	0.11
3	Clayey sand, grey, moist (reduced)	0.7-1.2	15.3	0.50	0.45	<0.5	0.10
4	Sandy clay, brown-grey, wet, dense (reduced)	1.2-2.4	-1.7	0.04	0.03	2.9	0.05

To place the above results into context, Tables 8-10 provide a summary of key static testwork parameters based on all available sediment analyses for Currency Creek conducted by CSIRO (2009) and Earth Systems.

Representative sediment sampling performed by CSIRO and Earth Systems consisted of samples collected at various elevation intervals (m AHD) ranging from +0.6 m AHD to -2.5 m AHD in Currency Creek.

The sulfur and ANC data were classified into elevation intervals of +0.6 m AHD -1.4 m AHD. In the case of sulfur, the S_{cr} (CSIRO) and total sulfur (Earth Systems) results were merged into a single dataset, referred to as 'sulfide-sulfur' in Tables 8-10. Within each depth interval, the average 'sulfide-sulfur' content and ANC value was calculated based on all available data. Where no data were available for a particular depth interval, the data gaps were filled via linear interpolation (for modelling purposes only).

The key results are summarised below:

- The average sulfide-sulfur content, and therefore Acid Producing Potential (APP), of the sandy sediments increases with depth in the upper 1.2 m, from 0.02-0.09 wt% S (APP 0.5-2.7 kg H_2SO_4 / tonne) at elevations of +0.1 to +0.6 m AHD, to 0.3-0.9 wt% S (APP 9-27 kg H_2SO_4 / tonne) at elevations of -0.6 m to +0.1 m AHD. Lower sulfide-sulfur values were measured at greater depth (0.1-0.2 wt% S; APP 4-6 kg H_2SO_4 / tonne).
- Average ANC values increase with depth in the sandy sediments, from 1 kg H_2SO_4 / tonne in the upper 0.3 m (+0.3 to +0.6 m AHD) to 25 kg H_2SO_4 / tonne at -0.7 to -0.9 m AHD.
- Overall, the static test results indicate that the upper sandy sediments are slightly acid consuming (NAPP -7 to +2 kg H_2SO_4 / tonne; +0.1 to +0.6 m AHD) while there is increased potential for acid generation from the underlying sediments (e.g. NAPP +5 to +18 kg H_2SO_4 / tonne; -0.6 to +0.1 m AHD).
- The average sulfide-sulfur content profile of the loamy sediments was very similar to that of the sands, although no loams were identified in the upper +0.2 to +0.6 m AHD of the sediment profile. Average ANC values in the loamy sediments also increased with depth, with even higher values measured at -0.6 to +0.1 m AHD (up to 28 kg H_2SO_4 / tonne) and -1.4 to -0.6 m AHD (40-80 kg H_2SO_4 / tonne). Thus, average NAPP values were generally negative, ranging from -72 to +19 kg H_2SO_4 / tonne.
- The average sulfide-sulfur content profile of the clayey sediments followed a similar trend to the sands and loams, increasing from 0.2-0.4 wt% S (-0.2 to +0.1 m AHD) to 0.8-1.2 wt% S (-0.6 to -0.2 m AHD), then decreasing to 0.2-0.6 wt% S (below -0.6 m AHD). No clays were identified in the upper +0.1 to +0.6 m AHD of the sediment profile. Average ANC values in the clays were less than 10 kg H_2SO_4 / tonne, thus all depth intervals were characterised by positive average NAPP values, in the range +5 to +31 kg H_2SO_4 / tonne.

Table 8. Average sulfide-sulfur, ANC, APP and NAPP values of sandy sediments (classified by CSIRO as "coarse") at various elevation intervals (m AHD) in Currency Creek.

Sediment Depth (m AHD)	Average Sulfide-Sulfur*		Average ANC (wt% CaCO ₃)		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
+0.6 to +0.5	0.090	2	0.200	2	1.0	2.8	1.8
+0.5 to +0.4	0.025	2	0.200	2	1.0	0.8	-0.2
+0.4 to +0.3	0.020	4	0.151	4	0.8	0.6	-0.1
+0.3 to +0.2	0.024	9	1.079	9	5.4	0.7	-4.7
+0.2 to +0.1	0.031	6	1.534	6	7.7	0.9	-6.7
+0.1 to 0.0	0.338	10	1.018	10	5.1	10.4	5.3
0.0 to -0.1	0.504	8	1.183	8	5.9	15.4	9.5
-0.1 to -0.2	0.438	9	1.044	9	5.2	13.4	8.2
-0.2 to -0.3	0.493	8	1.357	8	6.8	15.1	8.3
-0.3 to -0.4	0.757	7	1.636	7	8.2	23.2	15.0
-0.4 to -0.5	0.672	8	1.475	8	7.4	20.6	13.2
-0.5 to -0.6	0.882	6	1.864	6	9.3	27.0	17.7
-0.6 to -0.7	0.138	4	2.541	4	12.7	4.2	-8.5
-0.7 to -0.8	0.199	2	5.033	2	25.2	6.1	-19.1
-0.8 to -0.9	0.199	2	5.033	2	25.2	6.1	-19.1
-0.9 to -1.0	0.190	1	0.000	1	0.0	5.8	5.8
-1.0 to -1.1	0.190	1	0.000	1	0.0	5.8	5.8
-1.1 to -1.2	0.190	1	0.000	1	0.0	5.8	5.8
-1.2 to -1.3	0.190	1	0.000	1	0.0	5.8	5.8
-1.3 to -1.4	0.190	1	0.000	1	0.0	5.8	5.8

n = number of samples

* Based on S_{cr} analyses for samples collected by CSIRO (2009) and Total Sulfur analyses for samples collected by Earth Systems. Total Sulfur analyses were assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

Table 9. Average sulfide-sulfur, ANC, APP and NAPP values of loamy sediments (classified by CSIRO as "medium") at various elevation intervals (m AHD) in Currency Creek.

Sediment Depth (m AHD)	Average Sulfide-Sulfur*		Average ANC (wt% CaCO ₃)		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
+0.6 to +0.5	n/a	0	n/a	0	n/a	n/a	n/a
+0.5 to +0.4	n/a	0	n/a	0	n/a	n/a	n/a
+0.4 to +0.3	n/a	0	n/a	0	n/a	n/a	n/a
+0.3 to +0.2	n/a	0	n/a	0	n/a	n/a	n/a
+0.2 to +0.1	0.031	2	0.272	2	1.4	0.9	-0.4
+0.1 to 0.0	0.005	1	0.000	1	0.0	0.2	0.2
0.0 to -0.1	0.016	4	4.291	4	21.5	0.5	-21.0
-0.1 to -0.2	0.075	5	5.224	5	26.1	2.3	-23.8
-0.2 to -0.3	0.400	5	0.075	5	0.4	12.2	11.9
-0.3 to -0.4	0.636	3	0.091	3	0.5	19.5	19.0
-0.4 to -0.5	0.904	4	3.984	4	19.9	27.7	7.8

Sediment Depth (m AHD)	Average Sulfide-Sulfur*		Average ANC (wt% CaCO ₃)		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
-0.5 to -0.6	0.379	3	5.635	3	28.2	11.6	-16.6
-0.6 to -0.7	0.128	2	8.309	2	41.5	3.9	-37.6
-0.7 to -0.8	0.191	1	15.499	1	77.5	5.8	-71.7
-0.8 to -0.9	0.197	2	12.088	2	60.4	6.0	-54.4
-0.9 to -1.0	0.197	2	12.088	2	60.4	6.0	-54.4
-1.0 to -1.1	0.197	2	12.088	2	60.4	6.0	-54.4
-1.1 to -1.2	0.203	1	8.677	1	43.4	6.2	-37.2
-1.2 to -1.3	0.203	1	8.677	1	43.4	6.2	-37.2
-1.3 to -1.4	n/a	0	n/a	0	n/a	n/a	n/a

n = number of samples

* Based on S_{cr} analyses for samples collected by CSIRO (2009) and Total Sulfur analyses for samples collected by Earth Systems. Total Sulfur analyses were assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

Table 10. Average sulfide-sulfur, ANC, APP and NAPP values of clayey sediments (classified by CSIRO as "fine") at various elevation intervals (m AHD) in Currency Creek.

Sediment Depth (m AHD)	Average Sulfide-Sulfur*		Average ANC (wt% CaCO ₃)		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
+0.6 to +0.5	n/a	0	n/a	0	n/a	n/a	n/a
+0.5 to +0.4	n/a	0	n/a	0	n/a	n/a	n/a
+0.4 to +0.3	n/a	0	n/a	0	n/a	n/a	n/a
+0.3 to +0.2	n/a	0	n/a	0	n/a	n/a	n/a
+0.2 to +0.1	n/a	0	n/a	0	n/a	n/a	n/a
+0.1 to 0.0	0.218	1	0.238	1	1.2	6.7	5.5
0.0 to -0.1	0.352	2	0.373	2	1.9	10.8	8.9
-0.1 to -0.2	0.415	4	0.338	4	1.7	12.7	11.0
-0.2 to -0.3	0.850	4	0.168	4	0.8	26.0	25.2
-0.3 to -0.4	0.802	7	1.920	7	9.6	24.5	14.9
-0.4 to -0.5	1.158	4	0.805	4	4.0	35.4	31.4
-0.5 to -0.6	0.803	6	0.831	6	4.2	24.6	20.4
-0.6 to -0.7	0.391	5	0.807	5	4.0	12.0	7.9
-0.7 to -0.8	0.591	3	0.757	3	3.8	18.1	14.3
-0.8 to -0.9	0.250	2	0.450	2	2.3	7.7	5.4
-0.9 to -1.0	0.250	2	0.450	2	2.3	7.7	5.4
-1.0 to -1.1	0.250	2	0.450	2	2.3	7.7	5.4
-1.1 to -1.2	0.250	2	0.450	2	2.3	7.7	5.4
-1.2 to -1.3	0.250	2	0.450	2	2.3	7.7	5.4
-1.3 to -1.4	0.250	2	0.450	2	2.3	7.7	5.4

n = number of samples

* Based on S_{cr} analyses for samples collected by CSIRO (2009) and Total Sulfur analyses for samples collected by Earth Systems. Total Sulfur analyses were assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

Geologic cross sections for survey transects carried out from 30 April to 1 May 2009 are provided in Figure 12. Key results from the surveys include:

- Upper Currency Creek was completely dry on 30 April 2009 with the exception of a small pond at Currency Creek Hill (Plate 31).
- As depicted in Figure 12, the sandy sediments of Currency Creek are thickest at the outer shoreline and decrease in thickness towards the centre of the creek. The sandy horizon thickness ranged from 0.5-2 m at the original shoreline (Sites 1A, 2A, 3E and 5A). The thickness of the oxidised sand layer ranged from 5-20 cm (Plate 32).
- Unconsolidated dark grey clay with a strong hydrogen sulfide (H₂S) odour was observed at the surface or immediately beneath sandy horizons at all locations surveyed. This clay-rich material covering the majority of the central Currency Creek area significantly impaired infiltration and enabled rapid ponding of water in the creek bed immediately after a significant rainfall event in early May 2009.
- A thin (1-2 cm) layer of white (when dry) fibrous organic matter covered the majority of the central areas of the creek bed (see Plate 33).
- The clay-rich sediments retained significant moisture, even at the ground surface, despite the evaporation of surface water and limited rainfall during the preceding months. Limited areas of surficial clay sediments had commenced desiccation on the lower margins of the creek, although the bulk of the desiccated material was internally saturated (with no clear evidence of sulfide oxidation).
- Despite substantial evaporation in near-surface sandy sediments, relatively high moisture contents were observed in the near-surface clay sediments. This was attributed to their fine grained nature and a protective covering of fibrous organic matter.



Plate 31. Site 2A. Upper Currency Creek dry creek bed on 30 April 2009 shown in background.



Plate 32. Test pit at Site 2A depicting oxidised and reduced sand horizons.



Plate 33. White organic matter overlying saturated dark grey clay at Site 2D.

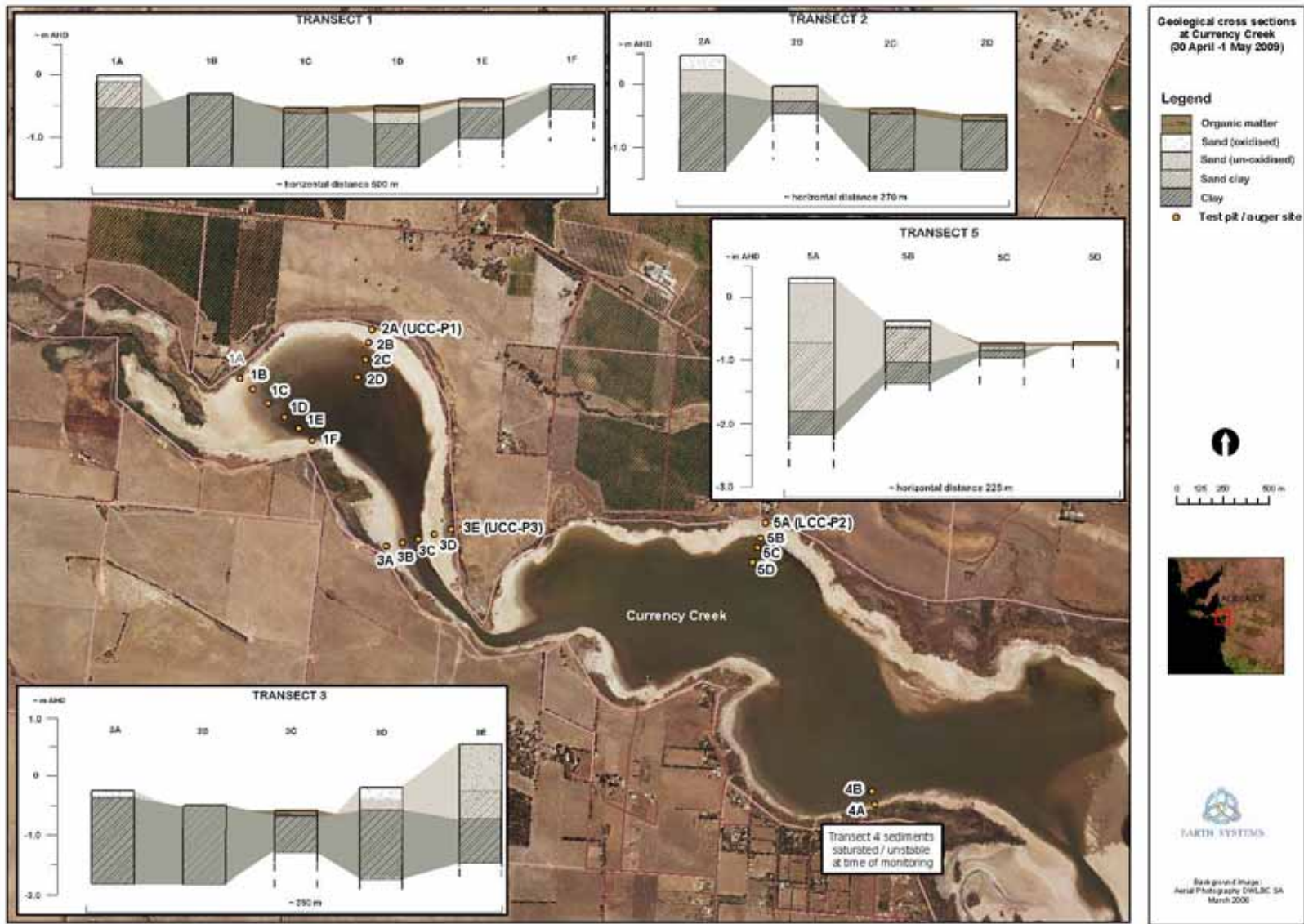


Figure 12. Geological cross sections at Currency Creek (30 April - 1 May 2009).

Point Sturt, Campbell Park and Windmill locations

Static acid-base accounting testwork data for selected core samples adjacent to the piezometer installation sites at the Point Sturt, Campbell Park and Windmill locations in August 2009 are presented in Attachment C and summarised in Table 11.

The key results are described below:

- Total sulfur ranged from 0.01-0.38 wt% S at Point Sturt, 0.07-2.03 wt% S at Campbell Park and 0.02-0.09 wt% S at the Windmill location. Median total sulfur content was highest for sediments at Campbell Park (0.61 wt% S), followed by Windmill (0.20 wt% S) and Point Sturt (0.09 wt% S).
- Total sulfur was generally associated with sediment texture, with elevated total sulfur related to clay content. Total sulfur contents were particularly high in green clay horizons (~1-2.5 m below ground level) at Campbell Park (1.43-2.03 wt% S).
- In general, total sulfur was lower at the surface and also at depth, with higher concentrations in between. This can be broadly attributed to oxidation and transport of sulfur from surficial horizons to lower down in the sequence.
- ANC values ranged from 2-267 kg H₂SO₄ equivalent / tonne at Point Sturt, 1.5-302 kg H₂SO₄ equivalent / tonne at Campbell Park and 0.6-290 kg H₂SO₄ equivalent / tonne at the Windmill location. Median ANC values were highest at Point Sturt (125.5 kg H₂SO₄ / tonne), followed by Campbell Park (12.1 kg H₂SO₄ / tonne) and Windmill (3.95 kg H₂SO₄ / tonne).
- ANC values were generally low in surficial sandy horizons where oxidation and neutralisation have previously occurred. High ANC values were measured in calcareous horizons associated with the Bridgewater Formation. The ANC values for sediments samples of the Bridgewater Formation ranged from 49.2-290 kg H₂SO₄ equivalent / tonne.
- NAPP values at Point Sturt ranged from -265 to +5.9 kg H₂SO₄ / tonne, associated with relatively low total sulfur contents and high ANC values. Most samples at Point Sturt had a pH after oxidation greater than 11. These unnaturally high pH values are an artefact of the NAG test conducted on sulfidic samples with significant neutralising capacity.
- NAPP values at Campbell Park ranged from -284 to +50 kg H₂SO₄ / tonne. NAPP values at the Windmill location ranged from -276 to +21.7 kg H₂SO₄ / tonne. NAPP values at these locations generally decreased with depth, due to increasing ANC.
- Median NAPP values were lowest at Point Sturt -120.5 kg H₂SO₄ / tonne, followed by Windmill (+4.15 kg H₂SO₄ / tonne) and Campbell Park (8.12 kg H₂SO₄ / tonne).
- TOC ranged from 0.06-0.18 wt% C at Point Sturt, 0.06-6.08 wt% C at Campbell Park and 0.02-1.55 wt% C at the Windmill location. Median TOC values were highest at Campbell Park (0.40 wt%), followed by Windmill and Point Sturt (both 0.1 wt%).
- The highest TOC values were associated with composite clay and coorongite and green clay horizons at Campbell Park (Sites 3 and 4, ~1-3 m below ground level).

NAG leachate water chemistry results are provided in Table 12. A comparison of water quality results with ANZECC/ARMCANZ (2000) trigger levels indicates that:

- NAG pH values after oxidation were below trigger levels for 80% protection of freshwater ecosystems at Campbell Park (Site 4, pH 3.6) and Windmill (Site 4, pH 5.6).
- The unnaturally high pH after oxidation at Point Sturt (Site 2, 11.6) is an artefact of the NAG test when conducted on sulfidic material with a significant excess of carbonate (7 kg H₂SO₄ / tonne). Leachate dissolved metal concentrations for Point Sturt (Site 2) will not be representative at this pH.
- Dissolved Al at Point Sturt (Site 2; 0.16 mg/L) and Campbell Park (Site 4; 1.03 mg/L) exceeded trigger values for 95% and 80% protection of freshwater ecosystems (0.055 and 0.150 mg/L, respectively).
- Dissolved Zn at Point Sturt (Site 2; 0.014 mg/L) and Campbell Park (Site 4; 0.011 mg/L) and dissolved Ni at Campbell Park (Site 4; 0.013 mg/L) exceeded trigger values for 95% protection of freshwater ecosystems (0.008 mg/L Zn; 0.011 mg/L Ni); however were below trigger values for 80% protection of freshwater ecosystems (0.031 mg/L Zn; 0.017 mg/L Ni).
- Dissolved Cu at Campbell Park (Site 4; 0.018 mg/L) exceeded trigger values for 95% protection of freshwater ecosystems (0.0014 mg/L).
- Dissolved As, Cd, Mn, Pb, Se, Cu and Ni at Point Sturt (Site 2) and Windmill (Site 4) and dissolved Zn at Windmill (Site 4) were below trigger values for 95% protection of freshwater ecosystems.

To place the above results into context, Tables 13-18 provide a summary of key static testwork parameters based on all available sediment analyses in Lake Albert and Lake Alexandrina conducted by CSIRO (2009) and Earth Systems.

Representative sediment sampling performed by CSIRO and Earth Systems consisted of samples collected at various elevation intervals (m AHD) ranging from +0.3 m AHD to -3.5 m AHD in Lake Alexandrina and +0.4 m AHD to -4.0 m AHD in Lake Albert.

For each lake, the sulfur and ANC data were classified into elevation intervals (m AHD) ranging from +0.2 m AHD to -1.8 m AHD for Lake Albert and 0.0 m AHD to -2.0 m AHD for Lake Alexandrina. In the case of sulfur, the S_{cr} (CSIRO) and total sulfur (Earth Systems) results were merged into a single dataset, referred to as 'sulfide-sulfur' in Tables 13-18. Within each depth interval, the average 'sulfide-sulfur' content and ANC value was calculated based on all available data. Where no data were available for a particular depth interval, the data gaps were filled via linear interpolation (for modelling purposes only).

The key results are summarised below:

- The average sulfide-sulfur content, and therefore Acid Producing Potential (APP), of the sandy sediments in Lake Albert increases with depth, from 0.01-0.1 wt% S (APP 0-3 kg H₂SO₄ / tonne) at elevations of -0.6 to +0.1 m AHD, to 0.1-0.2 wt% S (APP 3-7 kg H₂SO₄ / tonne) at elevations of -1.8 m to -0.6 m AHD. Similarly, in Lake Alexandrina, the average sulfide-sulfur content increased from 0.01 to 0.2 wt% S over the same elevation range.
- Average ANC values increase with depth in the sandy sediments of Lake Albert, from 2-8 kg H₂SO₄ equivalent / tonne at elevations of -0.6 to +0.1 m AHD to 8-58 kg H₂SO₄ equivalent / tonne at -1.8 to -0.6 m AHD. Similarly, in Lake Alexandrina, the average ANC increased from 0 to 42 kg H₂SO₄ equivalent / tonne over the same elevation range.
- Overall, the static test results indicate that the upper sandy sediments in Lake Albert are slightly acid consuming (NAPP -5 to 0 kg H₂SO₄ / tonne; -0.6 to +0.1 m AHD) with even greater acid consumption potential in the deeper sediments (NAPP -54 to -4 kg H₂SO₄ / tonne; -1.8 to -0.6 m AHD). Similarly, NAPP values range from -79 to 0 kg H₂SO₄ / tonne in Lake Alexandrina.
- The average sulfide-sulfur content of the loamy sediments in both lakes ranged from 0.1 to 0.5 wt% S, corresponding to APP values of around 1-16 kg H₂SO₄ / tonne. The majority of loam samples were identified between -1.2 and -0.3 m AHD. Average ANC values in the loamy sediments ranged up to 110 kg H₂SO₄ equivalent / tonne, while NAPP values ranged from -94 to +16 kg H₂SO₄ / tonne.
- The average sulfide-sulfur content of the clayey sediments in both lakes ranged from 0.1 to 1.7 wt% S, corresponding to APP values of around 4-50 kg H₂SO₄ / tonne. The majority of clay samples were identified between -2.0 and -0.7 m AHD. Average ANC values in the clayey sediments ranged up to 61 kg H₂SO₄ equivalent / tonne, while NAPP values ranged from -37 to +45 kg H₂SO₄ / tonne.

Table 11. Static acid-base accounting testwork data for Point Sturt, Campbell Park and Windmill sediments.

Sample ID	Geology^ (see Attachment F for detail)	Depth interval (m below ground)	Parameter										
			Geochemical composition							Particle size distribution (wt%)			
			NAPP (kg H ₂ SO ₄ / tonne)	Total S ** (wt%S)	NAG pH	NAG pH4.5 (kg H ₂ SO ₄ / tonne)	NAG pH7.0 (kg H ₂ SO ₄ / tonne)	ANC (kg H ₂ SO ₄ / tonne)	TOC (wt%C)	Clay (<2µm)	Silt (2- 60µm)	Sand (0.06- 2.00 mm)	Gravel (>2mm)
Point Sturt – Site 1													
<i>Upper piezometer</i>													
8	Grey medium clayey sand	0.58-0.89	5.9	0.26	5.4	<0.1	1	2	0.18	19	3	78	<1
9	Green calcareous clay	1.19-1.29	-193	0.19	11.1#	<0.1	<0.1	199	0.11				
<i>Lower piezometer</i>													
10	Grey clayey sand	2.00-2.47	-5.6	0.01	8.5	<0.1	<0.1	5.8	<0.02				
<i>Other</i>													
11	Grey clayey sand with pebbles of silcrete/calcrete	2.47-2.58	-123	0.03	11.4#	<0.1	<0.1	124	0.07	n/a	n/a	n/a	n/a
12	Light grey sand with pebbles of silcrete/calcrete	2.58-2.74	-221	0.03	11.4#	<0.1	<0.1	222	0.14	n/a	n/a	n/a	n/a
Point Sturt – Site 2													
<i>Upper piezometer</i>													
13*	Composite medium sand and clay	0.34-0.92	-4.6	0.08	11.6#	<0.1	<0.1	7	0.08	13	5	82	<1
14	Green clay	1.00-1.18	-43.4	0.38	11.4#	<0.1	<0.1	55	0.1				
<i>Lower piezometer</i>													
15	Green calcareous clay	1.18-1.59	-253	0.30	11.2#	<0.1	<0.1	262	0.1	n/a	n/a	n/a	n/a
16 & 17	Composite clayey sand and quartz sand	1.59-3.00	-38.2	0.02	11.7#	<0.1	<0.1	38.8	0.06	n/a	n/a	n/a	n/a

Sample ID	Geology^ (see Attachment F for detail)	Depth interval (m below ground)	Parameter														
			Geochemical composition							Particle size distribution (wt%)							
			NAPP (kg H ₂ SO ₄ / tonne)	Total S ** (wt%S)	NAG pH	NAG pH4.5 (kg H ₂ SO ₄ / tonne)	NAG pH7.0 (kg H ₂ SO ₄ / tonne)	ANC (kg H ₂ SO ₄ / tonne)	TOC (wt%C)	Clay (<2µm)	Silt (2-60µm)	Sand (0.06-2.00 mm)	Gravel (>2mm)				
<i>Other</i>																	
18	Grey sand with pebbles of silcrete/calcrete	3.00-3.16	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a				
Point Sturt – Site 3																	
<i>Upper piezometer</i>																	
1	Grey medium quartz sand	0.20-0.72	-12.4	0.15	11.6#	<0.1	<0.1	17	0.08	n/a	n/a	n/a	n/a				
2	Green calcareous clay	1.00-1.32	-118	0.31	10.6#	<0.1	<0.1	127	0.1								
<i>Lower piezometer</i>																	
3	Light green clayey sand	1.45-1.75	-146	0.19	11.3#	<0.1	<0.1	152	0.07								
Point Sturt – Site 4																	
<i>Upper piezometer</i>																	
4	Medium quartz sand	0.11-0.80	-19.8	0.04	11.6#	<0.1	<0.1	21.1	0.1	n/a	n/a	n/a	n/a				
<i>Lower Piezometer</i>																	
5	Light grey medium to fine sand	1.00-1.16	-265	0.09	11.2#	<0.1	<0.1	267	0.06								
6	Light green sandy clay	1.16-1.27	-240	0.02	11.4#	<0.1	<0.1	241	0.06								
7	Light grey fine sand	1.43-1.69	-209	0.02	11.4#	<0.1	<0.1	210	0.11								
Campbell Park – Site 1																	
<i>Upper Piezometer</i>																	
19	Composite medium quartz sand and clay	0.35-0.72	8.2	0.4	3.0	6.2	11	4	0.36	27	6	67	<1				
20	Green to grey clay	0.72-0.91	14.1	1.36	2.7	10.2	15	27.5	1.12								
<i>Other</i>																	
21	Grey Clay	0.91-0.99	n/a	n/a	n/a	n/a	n/a	n/a	n/a								
22	Composite green clay and silt	1.00-1.78	36.5	1.47	2.8	15.9	28	8.5	2.61								

Sample ID	Geology^ (see Attachment F for detail)	Depth interval (m below ground)	Parameter														
			Geochemical composition							Particle size distribution (wt%)							
			NAPP (kg H ₂ SO ₄ / tonne)	Total S ** (wt%S)	NAG pH	NAG pH4.5 (kg H ₂ SO ₄ / tonne)	NAG pH7.0 (kg H ₂ SO ₄ / tonne)	ANC (kg H ₂ SO ₄ / tonne)	TOC (wt%C)	Clay (<2µm)	Silt (2-60µm)	Sand (0.06-2.00 mm)	Gravel (>2mm)				
<i>Lower Piezometer</i>																	
23	Composite Coorongite and fine sand	1.78-1.98	n/a	n/a	n/a	n/a	n/a	n/a	n/a								
24 & 25	Composite quartz sand and clay	2.00-3.69	-3.9	0.16	8.6	<0.1	<0.1	8.8	0.06								
<i>Other</i>																	
26	Light grey to light green medium to fine sand	3.69-4.00	-284	0.61	11.0#	<0.1	<0.1	302	0.11	n/a	n/a	n/a	n/a				
Campbell Park – Site 2																	
<i>Upper piezometer</i>																	
27	Composite sand and clay	0.38-0.98	-32	0.41	4.6	<0.1	4.3	44.5	0.25	13	1	86	<1				
<i>Other</i>																	
28	Green clay	1.00-2.00	49.1	1.94	2.5	42	48	10.3	3.2								
29	Brown to green clay	2.00-2.08	n/a	n/a	n/a	n/a	n/a	n/a	n/a								
<i>Lower piezometer</i>																	
30	Composite Coorongite and silt	2.08-2.40	n/a	n/a	n/a	n/a	n/a	n/a	n/a								
31	Composite quartz sand and clay	2.40-3.00	-15.6	0.18	10.6#	<0.1	<0.1	21.1	0.08								
32	Green clayey sand	3.00-3.17	n/a	n/a	n/a	n/a	n/a	n/a	n/a								
<i>Other</i>																	
33	Grey sand with pebbles of silcrete/calcrete	3.17-3.21	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a				
Campbell Park – Site 3																	
<i>Upper piezometer</i>																	
34	Composite clay and fine sand	0.50-0.99	14.4	0.52	2.6	8.4	14	1.5	0.4	n/a	n/a	n/a	n/a				

Sample ID	Geology^ (see Attachment F for detail)	Depth interval (m below ground)	Parameter										
			Geochemical composition							Particle size distribution (wt%)			
			NAPP (kg H ₂ SO ₄ / tonne)	Total S ** (wt%S)	NAG pH	NAG pH4.5 (kg H ₂ SO ₄ / tonne)	NAG pH7.0 (kg H ₂ SO ₄ / tonne)	ANC (kg H ₂ SO ₄ / tonne)	TOC (wt%C)	Clay (<2µm)	Silt (2-60µm)	Sand (0.06-2.00 mm)	Gravel (>2mm)
<i>Other</i>													
35	Green clay	1.00-2.00	46.6	1.8	2.5	27.9	38	8.5	2.51				
36	Grey to green clay	2.00-2.35	50	2.03	2.4	41	54	12.1	3.72				
<i>Lower piezometer</i>													
37	Composite Coorongite and silt	2.35-2.81	35.3	1.53	2.8	23.8	30	11.5	6.08				
55	Green silt	2.81-3.00	-11.3	1.43	8.4	<0.1	<0.1	55	1.1				
38	Grey medium fine quartz sand	3.00-3.28	-13.1	0.07	10.2#	<0.1	<0.1	15.2	0.07				
<i>Other</i>													
39	Composite quartz sand with pebbles of silcrete/calcrete	3.28-4.00	n/a	n/a	n/a	n/a	n/a	n/a	n/a				
Campbell Park – Site 4													
<i>Upper piezometer</i>													
40*	Clayey sand	0.45-0.60	4.6	0.15	3.6	1.2	5.7	<0.5	0.32				
41	Grey clay	0.60-0.98	24.5	0.8	2.5	16.4	19	<0.5	0.65				
<i>Other</i>													
42	Grey to green clay	1.00-1.87	44.3	1.68	2.6	30.7	41	7.1	3.33	n/a	n/a	n/a	n/a
<i>Mid piezometer</i>													
43	Dark brown Coorongite	1.87-1.98	n/a	n/a	n/a	n/a	n/a	n/a	n/a				
44	Composite sand & clay	2.00-2.25	-36.8	0.1	10.4#	<0.1	<0.1	39.8	0.1				
<i>Lower piezometer</i>													
45	Medium sand	2.30-2.98	-34	0.5	11.3#	<0.1	<0.1	49.2	0.09	n/a	n/a	n/a	n/a
Windmill – Site 1													
<i>Upper piezometer</i>													
56	Grey medium quartz sand	0.16-0.44	5.9	0.3	2.9	5.4	8.2	3.3	0.13	n/a	n/a	n/a	n/a

Sample ID	Geology^ (see Attachment F for detail)	Depth interval (m below ground)	Parameter										
			Geochemical composition							Particle size distribution (wt%)			
			NAPP (kg H ₂ SO ₄ / tonne)	Total S ** (wt%S)	NAG pH	NAG pH4.5 (kg H ₂ SO ₄ / tonne)	NAG pH7.0 (kg H ₂ SO ₄ / tonne)	ANC (kg H ₂ SO ₄ / tonne)	TOC (wt%C)	Clay (<2µm)	Silt (2-60µm)	Sand (0.06-2.00 mm)	Gravel (>2mm)
57	Grey medium fine quartz sand	0.44-0.99	5.5	0.2	2.8	5.1	8.4	0.6	0.03				
58	Grey medium fine quartz sand	1.00-1.99	4.6	0.21	2.9	3.4	6.8	1.8	0.03				
Windmill – Site 2													
<i>Upper piezometer</i>													
59	Grey medium quartz sand	0.03-0.300	1.1	0.07	6.0	<0.1	0.7	1.1	0.06	n/a	n/a	n/a	n/a
60	Composite clay and sand	0.30-0.52	21.7	0.9	2.6	12.1	18	5.8	1.2				
61	Grey clayey sand	0.52-0.72	15.8	0.54	2.5	12.4	16	0.7	0.15				
62	Grey medium quartz sand	1.00-1.72	-30.8	0.16	6.8	<0.1	0.2	35.7	<0.02				
63	Grey medium fine quartz sand	1.72-1.98	n/a	n/a	n/a	n/a	n/a	n/a	n/a				
64	Light brown medium quartz sand	2.00-2.98	-22.5	0.03	10.3#	<0.1	<0.1	23.4	0.14				
Windmill – Site 3													
<i>Upper piezometer</i>													
65	Grey medium quartz sand	0.14-0.64	<0.5	0.09	5.3	<0.1	2	2.7	0.24	n/a	n/a	n/a	n/a
66-68	Composite quartz sand and clay	0.60-0.80	12	0.52	3.0	5.7	11	3.9	0.71				
77C	Green grey medium quartz sand	0.80-2.36	5.2	0.17	3.1	4.6	8.3	<0.5	0.05				
<i>Other</i>													
73 & 74	Green clayey sand with pebbles of silcrete/calcrete	2.36-2.67	-202	0.44	11.3#	<0.1	<0.1	216	0.12				

Sample ID	Geology [^] (see Attachment F for detail)	Depth interval (m below ground)	Parameter										
			Geochemical composition							Particle size distribution (wt%)			
			NAPP (kg H ₂ SO ₄ / tonne)	Total S ^{**} (wt%S)	NAG pH	NAG _{pH4.5} (kg H ₂ SO ₄ / tonne)	NAG _{pH7.0} (kg H ₂ SO ₄ / tonne)	ANC (kg H ₂ SO ₄ / tonne)	TOC (wt%C)	Clay (<2µm)	Silt (2-60µm)	Sand (0.06-2.00 mm)	Gravel (>2mm)
75 & 76	Green clayey sand with pebbles of silcrete/calcrete	2.67-2.98	-276	0.46	11.0 [#]	<0.1	<0.1	290	0.06				
Windmill – Site 4													
<i>Upper Piezometer</i>													
46*	Grey medium quartz sand	0.15-0.36	0.6	0.02	5.9	<0.1	1.7	<0.5	0.02	n/a	n/a	n/a	n/a
47	Grey clay	0.36-0.54	8.7	0.37	4.5	<0.1	3.1	2.6	1.55				
78C	Grey medium quartz sand	0.54-1.44	3.7	0.12	3.3	3.3	4.7	<0.5	0.05				
50	Brown green silt	1.44-1.51	n/a	n/a	n/a	n/a	n/a	n/a	n/a				
51	Grey fine quartz sand	1.51-1.78	-249	0.11	10.2 [#]	<0.1	<0.1	252	0.12				
52	Green clay	1.78-1.90	15	0.62	2.8	11.3	17	4	0.08				
<i>Lower Piezometer</i>													
53 & 54	Clayey sand with pebbles of silcrete/calcrete	1.90-3.00	-181	0.15	11.2 [#]	<0.1	<0.1	185	0.06				

* Refer to NAG leachate water quality data in Table 12.

High NAG pH values are an artefact of the NAG test method associated with samples with high ANC values.

[^] Blue = Lake Sediments. Yellow = Bridgewater Formation. Pink = Composite Lake Sediments and Bridgewater Formation. All samples were moist and reduced unless specified.

** Total Sulfur analyses assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

Table 12. NAG leachate water quality data for Point Sturt, Campbell Park and Windmill piezometer sediments.

Sample ID	Geology (see Attachment F for detail)	Depth interval (m below ground)	Parameter										
			NAG pH	Dissolved metal concentration in NAG leachate* (mg/L)									
				Al	As	Cd	Cu	Fe	Mn	Ni	Pb	Se	Zn
<i>ANZECC/ARMCANZ (2000) trigger values for 80% protection of freshwater ecosystems</i>													
n/a	n/a	n/a	6.5-9.0	0.150 (pH>6.5)	0.140 (AsV)	0.0008	0.0025	n/a	3.6	0.017	0.0094	0.034	0.031
<i>ANZECC/ARMCANZ (2000) trigger values for 95% protection of freshwater ecosystems</i>													
n/a	n/a	n/a	6.5-9.0	0.055 (pH>6.5)	0.013 (AsV)	0.0002	0.0014	n/a	1.9	0.011	0.0034	0.011	0.008
<i>Point Sturt – Site 2</i>													
13	Composite medium sand and clay	0.34-0.92	11.6 [#]	0.16	0.002	<0.0001	<0.001	<0.05	0.068	<0.001	<0.001	<0.01	0.014
<i>Campbell Park – Site 4</i>													
40	Clayey sand	0.45-0.60	3.6	1.03	<0.001	<0.0001	0.018	0.84	0.14	0.013	<0.001	<0.01	0.011
<i>Windmill – Site 4</i>													
46	Grey medium quartz sand	0.15-0.36	5.9	0.01	0.003	<0.0001	0.001	<0.05	0.195	0.001	<0.001	<0.01	<0.005

* Values exceeding ANZECC/ARMCANZ (2000) trigger values for **95%** protection of freshwater ecosystems are shaded in orange. Values exceeding trigger values for **95%** and **80%** protection of freshwater ecosystems are shaded in red.

[#] High NAG pH value is an artefact of the NAG test method associated with samples with high ANC values.

[^] Blue = Lake Sediments.

Table 13. Average sulfide-sulfur, ANC, APP and NAPP values of sandy sediments (classified by CSIRO as "coarse") at various elevation intervals (m AHD) in Lake Albert.

Sediment Depth (m AHD)	Average Sulfide Sulfur*		Average ANC		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
+0.2 to +0.1	n/a	0	n/a	0	n/a	n/a	n/a
+0.1 to 0.0	0.007	2	0.215	2	2.1	0.2	-1.9
0.0 to -0.1	0.098	7	0.318	6	3.1	3.0	-0.1
-0.1 to -0.2	0.077	14	0.254	12	2.5	2.4	-0.1
-0.2 to -0.3	0.079	22	0.257	20	2.5	2.4	-0.1
-0.3 to -0.4	0.074	31	0.504	28	4.9	2.3	-2.7
-0.4 to -0.5	0.089	40	0.706	34	6.9	2.7	-4.2
-0.5 to -0.6	0.096	40	0.761	36	7.5	2.9	-4.5
-0.6 to -0.7	0.122	44	1.262	40	12.4	3.7	-8.6
-0.7 to -0.8	0.154	29	1.696	26	16.6	4.7	-11.9
-0.8 to -0.9	0.143	23	1.968	21	19.3	4.4	-14.9
-0.9 to -1.0	0.143	19	1.648	18	16.1	4.4	-11.8
-1.0 to -1.1	0.189	10	2.763	10	27.1	5.8	-21.3
-1.1 to -1.2	0.165	4	0.957	4	9.4	5.0	-4.3
-1.2 to -1.3	0.165	4	0.957	4	9.4	5.0	-4.3
-1.3 to -1.4	0.233	5	1.717	5	16.8	7.1	-9.7
-1.4 to -1.5	0.149	5	0.783	5	7.7	4.5	-3.1
-1.5 to -1.6	0.149	5	0.783	5	7.7	4.5	-3.1
-1.6 to -1.7	0.147	5	5.926	5	58.1	4.5	-53.6
-1.7 to -1.8	0.135	6	5.073	6	49.7	4.1	-45.6

n = number of samples

* Based on S_{cr} analyses for samples collected by CSIRO (2009) and Total Sulfur analyses for samples collected by Earth Systems. Total Sulfur analyses were assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

Table 14. Average sulfide-sulfur, ANC, APP and NAPP values of loamy sediments (classified by CSIRO as "medium") at various elevation intervals (m AHD) in Lake Albert.

Sediment Depth (m AHD)	Average Sulfide Sulfur*		Average ANC		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
+0.2 to +0.1	n/a	0	n/a	0	n/a	n/a	n/a
+0.1 to 0.0	0.518	1	11.210	1	109.9	15.8	-94.0
0.0 to -0.1	0.264	2	5.720	2	56.1	8.1	-48.0
-0.1 to -0.2	0.202	4	3.910	3	38.3	6.2	-32.2
-0.2 to -0.3	0.213	5	3.803	3	37.3	6.5	-30.8
-0.3 to -0.4	0.188	6	3.070	4	30.1	5.7	-24.3
-0.4 to -0.5	0.217	7	0.088	3	0.9	6.6	5.8
-0.5 to -0.6	0.205	7	0.223	4	2.2	6.3	4.1
-0.6 to -0.7	0.218	10	0.311	6	3.0	6.7	3.6
-0.7 to -0.8	0.302	14	0.379	9	3.7	9.3	5.5
-0.8 to -0.9	0.368	10	0.960	7	9.4	11.3	1.8
-0.9 to -1.0	0.390	9	1.059	7	10.4	11.9	1.6
-1.0 to -1.1	0.374	8	1.210	6	11.9	11.4	-0.4
-1.1 to -1.2	0.406	5	0.780	4	7.6	12.4	4.8

Sediment Depth (m AHD)	Average Sulfide Sulfur*		Average ANC		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
-1.2 to -1.3	0.648	1	0.360	1	3.5	19.8	16.3
-1.3 to -1.4	0.089	1	0.000	1	0.0	2.7	2.7
-1.4 to -1.5	0.148	3	0.030	3	0.3	4.5	4.2
-1.5 to -1.6	0.119	3	0.030	3	0.3	3.6	3.3
-1.6 to -1.7	0.000	1	0.000	1	0.0	0.0	0.0
-1.7 to -1.8	n/a	0	n/a	0	n/a	n/a	n/a

n = number of samples

* Based on S_{cr} analyses for samples collected by CSIRO (2009) and Total Sulfur analyses for samples collected by Earth Systems. Total Sulfur analyses were assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

Table 15. Average sulfide-sulfur, ANC, APP and NAPP values of clayey sediments (classified by CSIRO as "fine") at various elevation intervals (m AHD) in Lake Albert.

Sediment Depth (m AHD)	Average Sulfide Sulfur*		Average ANC		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
+0.2 to +0.1	0.119	1	1.200	1	11.8	3.6	-8.1
+0.1 to 0.0	0.119	1	1.200	1	11.8	3.6	-8.1
0.0 to -0.1	0.288**	0	1.812**	0	17.8	8.8	-8.9
-0.1 to -0.2	0.458**	0	2.424**	0	23.8	14.0	-9.7
-0.2 to -0.3	0.627**	0	3.035**	0	29.7	19.2	-10.6
-0.3 to -0.4	0.796	4	3.647	3	35.7	24.4	-11.4
-0.4 to -0.5	1.148	6	1.506	4	14.8	35.1	20.4
-0.5 to -0.6	0.932	8	1.772	6	17.4	28.5	11.2
-0.6 to -0.7	0.886	7	1.720	6	16.9	27.1	10.3
-0.7 to -0.8	0.927	7	1.511	6	14.8	28.4	13.6
-0.8 to -0.9	1.234	12	1.410	11	13.8	37.8	24.0
-0.9 to -1.0	1.620	13	1.013	13	9.9	49.6	39.6
-1.0 to -1.1	1.687	16	0.978	16	9.6	51.6	42.0
-1.1 to -1.2	1.578	18	0.871	18	8.5	48.3	39.7
-1.2 to -1.3	1.664	14	1.021	14	10.0	50.9	40.9
-1.3 to -1.4	1.727	18	1.259	18	12.3	52.8	40.5
-1.4 to -1.5	1.499	33	1.122	33	11.0	45.9	34.9
-1.5 to -1.6	1.565	28	0.858	28	8.4	47.9	39.5
-1.6 to -1.7	1.699	12	0.767	12	7.5	52.0	44.5
-1.7 to -1.8	1.608	8	0.769	8	7.5	49.2	41.7

n = number of samples

* Based on S_{cr} analyses for samples collected by CSIRO (2009) and Total Sulfur analyses for samples collected by Earth Systems. Total Sulfur analyses were assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

** Interpolated from values at depths above and below (for modelling purposes only).

Table 16. Average sulfide-sulfur, ANC, APP and NAPP values of sandy sediments (classified by CSIRO as "coarse") at various elevation intervals (m AHD) in Lake Alexandrina.

Sediment Depth (m AHD)	Average Sulfide-Sulfur*		Average ANC (wt% CaCO ₃)		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
0.0 to -0.1	0.007	2	0.015	2	0.1	0.2	0.1
-0.1 to -0.2	0.001	4	0.185	3	1.8	0.0	-1.8
-0.2 to -0.3	0.023	14	0.089	8	0.9	0.7	-0.2
-0.3 to -0.4	0.020	29	0.293	18	2.9	0.6	-2.3
-0.4 to -0.5	0.024	38	0.313	31	3.0	0.7	-2.3
-0.5 to -0.6	0.021	59	0.578	52	5.7	0.6	-5.0
-0.6 to -0.7	0.042	60	0.625	51	6.1	1.3	-4.8
-0.7 to -0.8	0.055	50	0.784	48	7.7	1.7	-6.0
-0.8 to -0.9	0.051	42	1.423	40	13.9	1.5	-12.4
-0.9 to -1.0	0.050	33	0.983	32	9.6	1.5	-8.1
-1.0 to -1.1	0.047	17	0.514	17	5.0	1.4	-3.6
-1.1 to -1.2	0.066	14	0.639	14	6.3	2.0	-4.2
-1.2 to -1.3	0.057	11	0.564	11	5.5	1.8	-3.8
-1.3 to -1.4	0.055	8	0.223	8	2.2	1.7	-0.5
-1.4 to -1.5	0.027	9	3.749	9	36.7	0.8	-35.9
-1.5 to -1.6	0.081	15	3.481	15	34.1	2.5	-31.6
-1.6 to -1.7	0.142	10	4.294	10	42.1	4.4	-37.7
-1.7 to -1.8	0.190	4	1.129	4	11.1	5.8	-5.3
-1.8 to -1.9	0.093	3	6.508	3	63.8	2.8	-60.9
-1.9 to -2.0	0.061	5	8.199	5	80.4	1.9	-78.5

n = number of samples

* Based on S_{cr} analyses for samples collected by CSIRO (2009) and Total Sulfur analyses for samples collected by Earth Systems. Total Sulfur analyses were assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

Table 17. Average sulfide-sulfur, ANC, APP and NAPP values of loamy sediments (classified by CSIRO as "medium") at various elevation intervals (m AHD) in Lake Alexandrina.

Sediment Depth (m AHD)	Average Sulfide-Sulfur*		Average ANC (wt% CaCO ₃)		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
0.0 to -0.1	0.005	1	n/a	0	n/a	0.2	0.2
-0.1 to -0.2	0.042	2	n/a	0	n/a	1.3	1.3
-0.2 to -0.3	0.039	2	0.104	1	1.0	1.2	0.2
-0.3 to -0.4	0.278	3	0.780	3	7.6	8.5	0.9
-0.4 to -0.5	0.332	19	0.703	9	6.9	10.2	3.3
-0.5 to -0.6	0.407	21	0.242	12	2.4	12.5	10.1
-0.6 to -0.7	0.284	22	1.011	16	9.9	8.7	-1.2
-0.7 to -0.8	0.185	25	0.997	19	9.8	5.6	-4.1
-0.8 to -0.9	0.175	21	1.089	17	10.7	5.4	-5.3
-0.9 to -1.0	0.253	14	1.433	14	14.0	7.7	-6.3
-1.0 to -1.1	0.346	7	0.705	7	6.9	10.6	3.7
-1.1 to -1.2	0.152	3	0.566	3	5.5	4.7	-0.9
-1.2 to -1.3	0.263	4	0.449	4	4.4	8.0	3.6
-1.3 to -1.4	0.436	4	1.247	4	12.2	13.4	1.1

Sediment Depth (m AHD)	Average Sulfide-Sulfur*		Average ANC (wt% CaCO ₃)		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
-1.4 to -1.5	0.298	7	1.727	7	16.9	9.1	-7.8
-1.5 to -1.6	0.392	6	0.995	6	9.7	12.0	2.2
-1.6 to -1.7	0.292	7	4.103	7	40.2	8.9	-31.3
-1.7 to -1.8	0.127	3	8.370	3	82.0	3.9	-78.2
-1.8 to -1.9	0.274**	0	4.225**	0	41.4	8.4	-33.0
-1.9 to -2.0	0.421	1	0.080	1	0.8	12.9	12.1

n = number of samples

* Based on S_{cr} analyses for samples collected by CSIRO (2009) and Total Sulfur analyses for samples collected by Earth Systems. Total Sulfur analyses were assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

** Interpolated from values at depths above and below (for modelling purposes only).

Table 18. Average sulfide-sulfur, ANC, APP and NAPP values of clayey sediments (classified by CSIRO as "fine") at various elevation intervals (m AHD) in Lake Alexandrina.

Sediment Depth (m AHD)	Average Sulfide-Sulfur*		Average ANC (wt% CaCO ₃)		Average ANC	Average APP	Average NAPP
	wt% S	n	wt% CaCO ₃	n	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t	kg H ₂ SO ₄ / t
0.0 to -0.1	n/a	0	n/a	0	n/a	n/a	n/a
-0.1 to -0.2	n/a	0	n/a	0	n/a	n/a	n/a
-0.2 to -0.3	n/a	0	n/a	0	n/a	n/a	n/a
-0.3 to -0.4	0.924	1	n/a	0	n/a	28.3	28.3
-0.4 to -0.5	0.846	3	0.597	1	5.9	25.9	20.0
-0.5 to -0.6	0.685	7	1.079	2	10.6	21.0	10.4
-0.6 to -0.7	1.001	9	1.203	3	11.8	30.6	18.8
-0.7 to -0.8	1.305	10	0.993	3	9.7	39.9	30.2
-0.8 to -0.9	1.317	7	0.930	1	9.1	40.3	31.2
-0.9 to -1.0	0.993	3	1.161**	0	11.4	30.4	19.0
-1.0 to -1.1	1.491	3	1.393	3	13.6	45.6	32.0
-1.1 to -1.2	0.869	6	2.019	6	19.8	26.6	6.8
-1.2 to -1.3	0.243	3	2.341	3	22.9	7.4	-15.5
-1.3 to -1.4	0.764	9	6.187	9	60.6	23.4	-37.3
-1.4 to -1.5	0.975	13	4.458	13	43.7	29.8	-13.8
-1.5 to -1.6	1.017	12	4.184	11	41.0	31.1	-9.9
-1.6 to -1.7	1.068	12	4.074	11	39.9	32.7	-7.2
-1.7 to -1.8	1.165	12	3.962	12	38.8	35.6	-3.2
-1.8 to -1.9	1.307	10	0.896	10	8.8	40.0	31.2
-1.9 to -2.0	1.372	7	1.037	7	10.2	42.0	31.8

n = number of samples

* Based on S_{cr} analyses for samples collected by CSIRO (2009) and Total Sulfur analyses for samples collected by Earth Systems. Total Sulfur analyses were assumed to be representative of Sulfide Sulfur, due to reducing conditions in lake sediments at the time of sampling.

** Interpolated from values at depths above and below (for modelling purposes only).

5.2.2 Rising head tests and calculation of hydraulic conductivity

Calculated hydraulic conductivity values based on the rising head test data collected from the piezometers at Currency Creek, Point Sturt, Campbell Park and the Windmill location are provided in Attachment H and summarised in Table 19. The key results are described below:

- The bulk hydraulic conductivity of Currency Creek sediments was relatively low (0.02-0.06 m/day) in comparison with all other sites. This was expected given the Currency Creek piezometers were screened through clayey as well sandy horizons. Clay smearing during manual augering of boreholes may also have contributed to relatively low hydraulic conductivity values in Currency Creek.
- Hydraulic conductivity values in lake sediments (excluding Currency Creek sites) generally ranged from 0.09 to 5.56 m/day, but exceeded 30 m/day at three locations (CP-4D, WM-1S and WM-2S).
- The hydraulic conductivity of the Bridgewater Formation at the Windmill location (0.48-0.59 m/day) was lower than the overlying lake sediments (1.83->30 m/day). Conversely, the only piezometer at Campbell Park to be screened into the Bridgewater Formation (CP-4D) indicated a significantly higher hydraulic conductivity (31.3 m/day) than the other sites (lake sediments) at Campbell Park.

Table 19. Hydraulic conductivity at Currency Creek, Point Sturt, Campbell Park and Windmill sites.

Piezometer	Hydraulic Conductivity, K (m/day)	Piezometer depth (metres below ground surface)	Lake sediments / Bridgewater Formation
<i>Currency Creek</i>			
UCC-P1	0.02	3.00	LS
LCC-P2	0.06	2.60	LS
<i>Point Sturt</i>			
PS-1S	0.09	1.38	LS
PS-1D	4.41	2.48	LS
PS-2S	0.62	1.10	LS
PS-2D	4.05	2.36	LS
PS-3S	5.56	1.20	LS
PS-3D	1.87	2.01	LS
PS-4S	4.92	1.12	LS
PS-4D	3.94	1.75	LS
<i>Campbell Park</i>			
CP-1S	0.22	0.92	LS
CP-1D	1.45	3.62	LS
CP-2S	0.81	1.22	LS
CP-2D	3.87	3.12	LS
CP-3S	0.59	1.22	LS
CP-4S	2.52	1.22	LS
CP-4M	1.85	2.17	LS
CP-4D	31.30	4.42	BF
<i>Windmill</i>			
WM-1S	>30*	3.30	LS
WM-1D	0.48	5.22	BF
WM-2S	>30*	2.80	LS
WM-3S	4.91	2.22	LS
WM-4S	1.83	1.82	LS
WM-4D	0.59	3.02	BF

* Groundwater rebound in piezometers WM-1S and WM-2S was very rapid and reliable rising head test data were unable to be collected. However, the hydraulic conductivity in these two piezometers is considered likely to be greater than the highest measured value (CP-4D; 31.30 m/day). Thus, values of >30 m/day were assigned to WM-1S and WM-2S.

5.2.3 Rainfall, wind speed/direction and surface water levels

Rainfall and evapotranspiration¹¹ data collected at daily intervals at Currency Creek, Langhorne Creek and Narrung, from 1 January 2007 to 1 October 2009, are provided in Attachment I and summarised in Table 20. All available long term data are graphed in Figures 13-15. The key results are described below:

- In 2007 and 2008, annual rainfall ranged from 315 mm/year to 446 mm/year. Evapotranspiration rates were considerably higher, ranging from 1111 mm/year to 1436 mm/year.
- There was considerable variation in annual rainfall between the three sites in 2007-2008, with the highest rainfall at Currency Creek (386-446 mm/year) and lowest rainfall at Langhorne Creek (315-324 mm/year).
- Evapotranspiration of around 1300-1400 mm/year at Narrung was considerably higher than at Currency Creek and Langhorne Creek (1100-1200 mm/year) in 2007-2008, although this is inconsistent with available data for 2009.

Table 20. Long term rainfall and evapotranspiration at Currency Creek, Langhorne Creek and Narrung.

Time interval	Currency Creek		Langhorne Creek		Narrung	
	Rainfall (mm)	ET (mm)	Rainfall (mm)	ET (mm)	Rainfall (mm)	ET (mm)
1 Jan 2007 – 31 Dec 2007	446	1111	315	1197	371	1436
1 Jan 2008 – 31 Dec 2008	386	1160	324	1157	341	1279
1 Jan 2009 – 1 Oct 2009	375	757	286	752	359	737

Long Term Rainfall and Evapotranspiration - Currency Creek

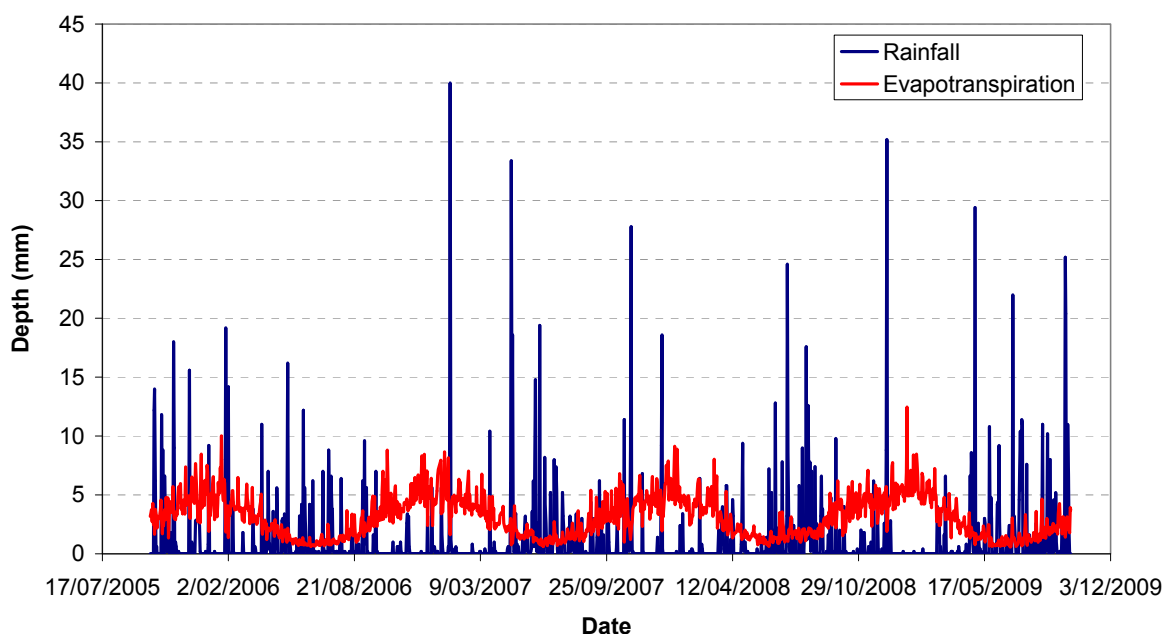


Figure 13. Daily rainfall and evapotranspiration at Currency Creek from 1 October 2005 to 1 October 2009.

¹¹ Evapotranspiration figures downloaded from SAMDNRM website refer to daily Reference Crop Evapotranspiration (mm) and are calculated from climatic data using the Penman-Monteith equation. Evapotranspiration is different to evaporation data derived from Class A pans in that it also accounts for transpiration of water vapour from plant foliage (SAMDNRM, 2009).

Long Term Rainfall and Evapotranspiration - Langhorne Creek

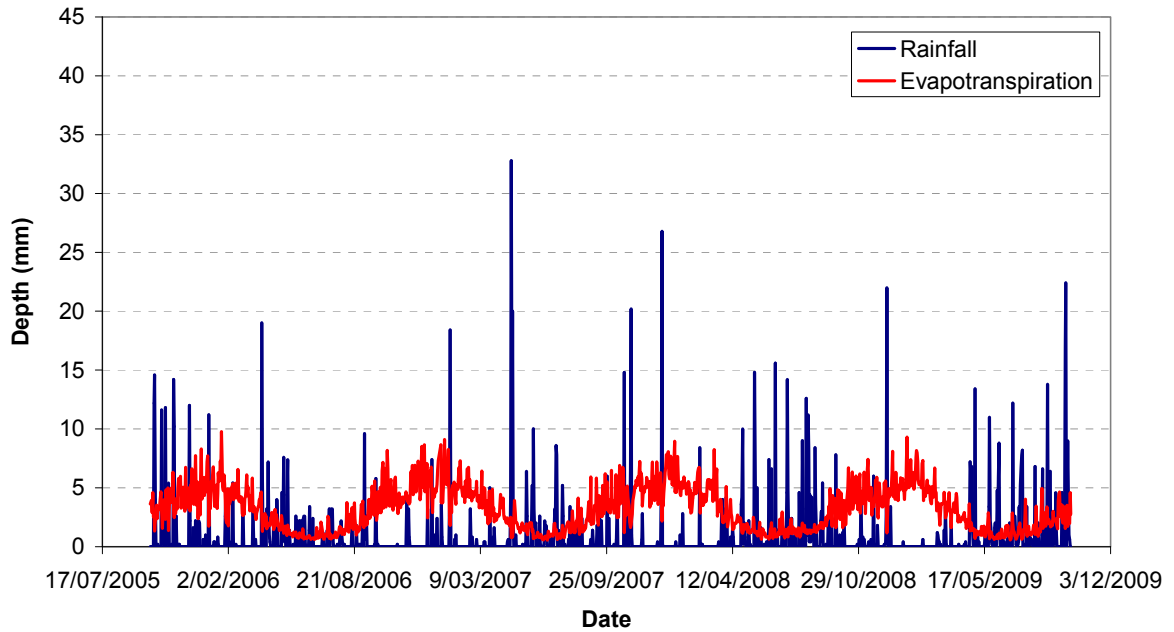


Figure 14. Daily rainfall and evapotranspiration at Langhorne Creek from 1 October 2005 to 1 October 2009.

Long Term Rainfall and Evapotranspiration - Narrung

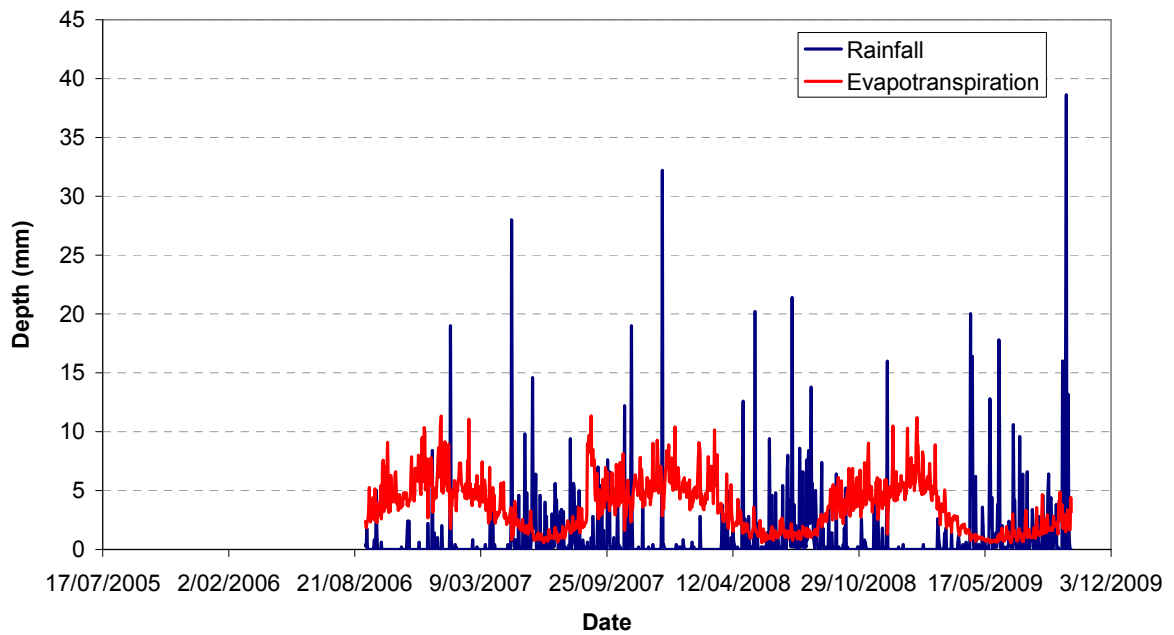


Figure 15. Daily rainfall and evapotranspiration at Narrung from 7 September 2006 to 1 October 2009.

Rainfall, wind speed and wind direction data collected at 15 minute intervals at Currency Creek, Langhorne Creek and Narrung, from 9:00 a.m. 17 August 2009 to 9:00 a.m. 17 September 2009, are provided in Attachment I.

The total rainfall depths during the month from 17 August to 17 September 2009 were 47.2 mm (Currency Creek), 42.0 mm (Langhorne Creek) and 50.6 mm (Narrung).

The 15 minute interval rainfall data and hourly surface water level data are graphed alongside piezometric levels and sediment moisture data in Sections 5.2.7 and 5.2.8.

Wind rose statistics at 1 hour intervals at Lake Albert near Waltowa Swamp (Site A4261153) over a 1 year period from 25/09/08 to 25/09/09 are presented in Figure 16. This figure displays the frequency of occurrence of winds in each of the specified wind direction sectors and wind speed classes. Daily variation in wind speed and monthly averages from 25/09/08 to 25/09/09 are plotted in Figures 17 and 18, respectively. Key observations at Lake Albert near Waltowa Swamp (over 1 year) are summarised below:

- The dominant winds are from the south (9-12% frequency), south-south-west (9-12% frequency) and north (9-12% frequency), with wind speeds commonly in the range 5.7-11.1 m/s (20-40 km/hr), as shown in Figure 16.
- Average monthly wind velocity ranged from 4.0 m/s (14.5 km/hr; May 2009) to 7.0 m/s (25.4 km/hr; September 2009). July to February had relatively high average wind velocities, whereas March to June was the calmest period.
- Wind speeds in excess of 50 m/s (150 km/hr) were measured 5 times from 25 September 2008 to 25 September 2009. Strong winds tend to be from the south to south-west, and to a lesser extent from the north.

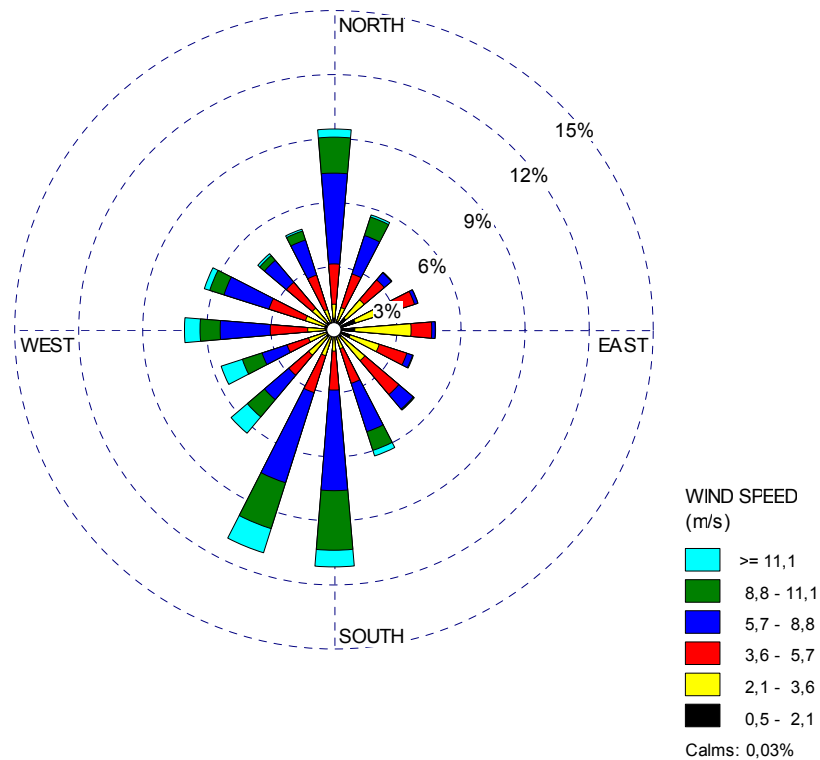


Figure 16. Wind rose depicting the frequency of occurrence of winds at Lake Albert (near Waltowa Swamp) from 25 September 2008 to 25 September 2009 (DWLBC, 2009).

Wind velocity at Lake Albert (near Waltowa Swamp)

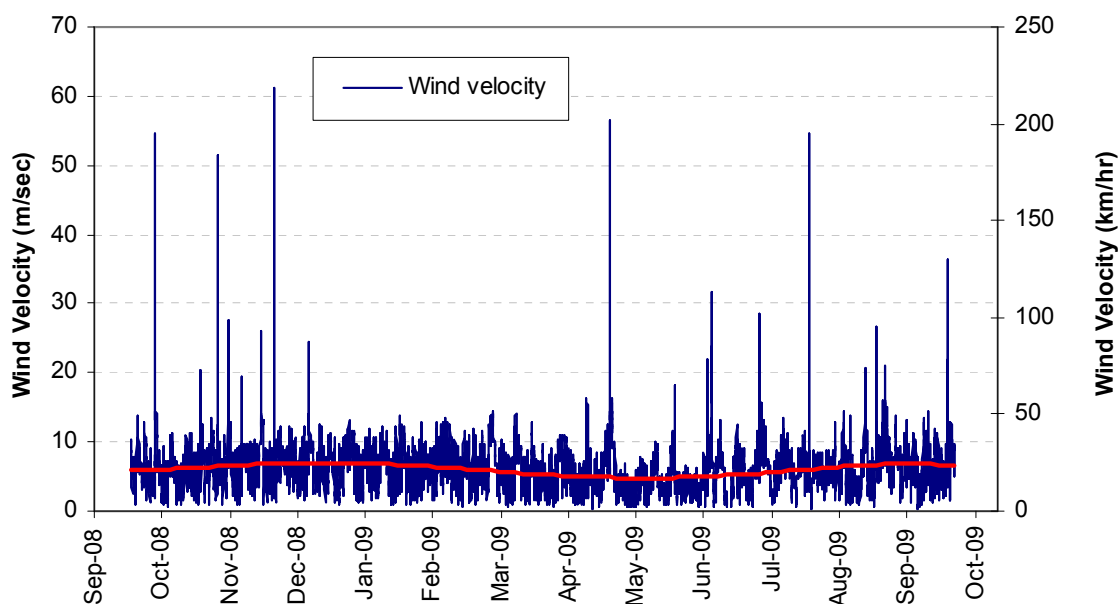


Figure 17. Daily wind velocity at Lake Albert (near Waltowa Swamp) from 25 September 2008 to 25 September 2009.

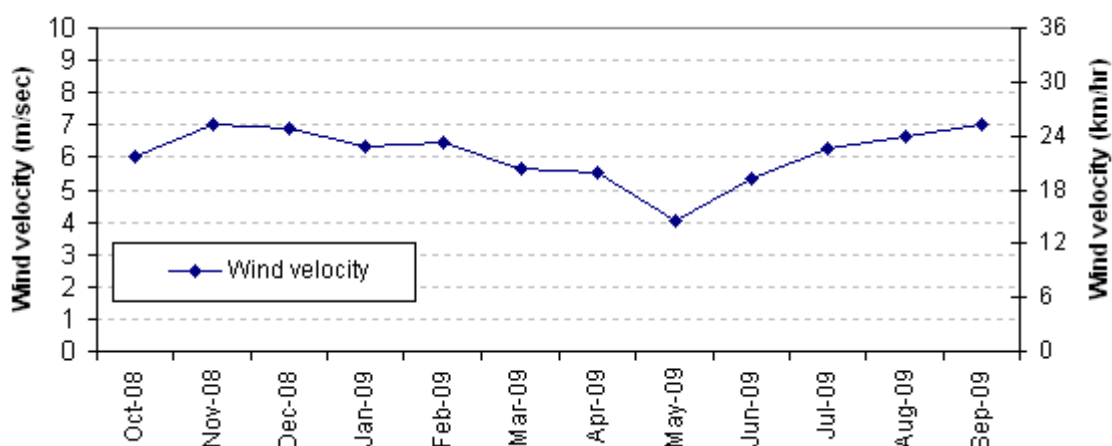


Figure 18. Average monthly wind velocity at Lake Albert (near Waltowa Swamp) from October 2008 to September 2009.

Wind rose statistics at Currency Creek (15 minute intervals), from 17 August to 17 September 2009 are presented in Figure 19.

Wind rose statistics at Langhorne Creek (15 minute intervals), Narrung (15 minute intervals) and Lake Albert near Waltowa Swamp (daily intervals) from 28 August to 16 September 2009 are presented in Figures 20-22. This time interval selected corresponds to the availability of groundwater (piezometric) level data, moisture data and groundwater quality data for the Point Sturt, Campbell Park and Windmill locations.

Key results are summarised below:

- The dominant winds at Currency Creek were from the west-north-west (20% frequency), north-west (15% frequency) and west (14% frequency), with wind speeds commonly in the range 0.5-8.8 m/s (2-32 km/hr), as shown in Figure 19.
- The dominant winds at Langhorne Creek were from the west (14% frequency), west-north-west (11% frequency), north-north-east (10% frequency) and north-east (9% frequency), with wind speeds commonly in the range 0.5-5.7 m/s (2-20 km/hr), as shown in Figure 20.

- The dominant winds at Narrung were from the north (13% frequency), north-north-west (13% frequency), north-north-east (9% frequency) and east-north-east (9% frequency), with wind speeds commonly in the range 0.5-8.8 m/s (2-32 km/hr), as shown in Figure 21.
- The dominant winds at Lake Albert near Waltowa Swamp were from the west (14% frequency), west-north-west (10% frequency) and north (12% frequency), with wind speeds commonly in the range 3.6-11.1 m/s (13-40 km/hr), as shown in Figure 22. Wind direction statistics from late August to mid-September 2009 differ significantly from the annual wind direction data shown in Figure 16.
- Between 28 August to 16 September 2009, significant variation in wind speed and direction was observed between different sites (Langhorne Creek, Narrung and Lake Albert near Waltowa Swamp), as shown in Figures 20-22.

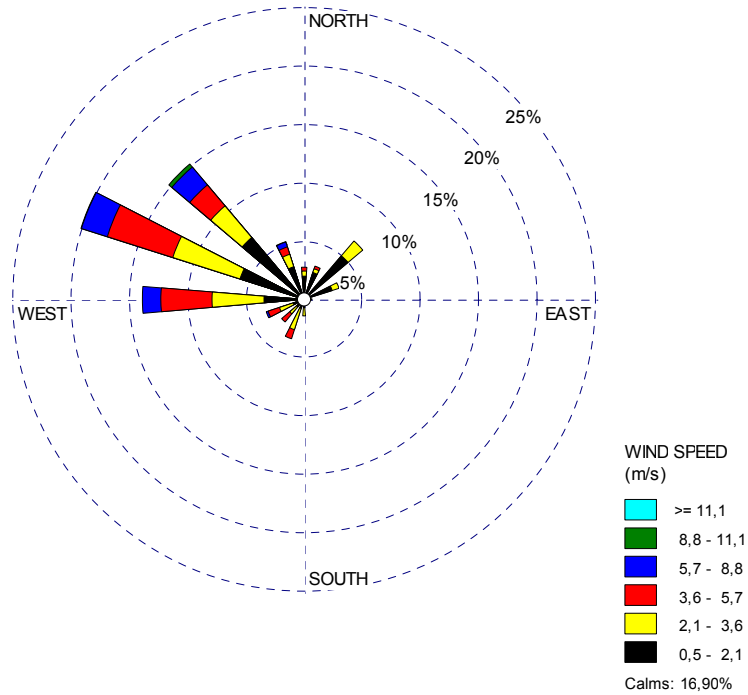


Figure 19. Wind rose depicting the frequency of occurrence of winds at Currency Creek, 17 August – 17 September 2009 (SAMDBNRM, 2009).

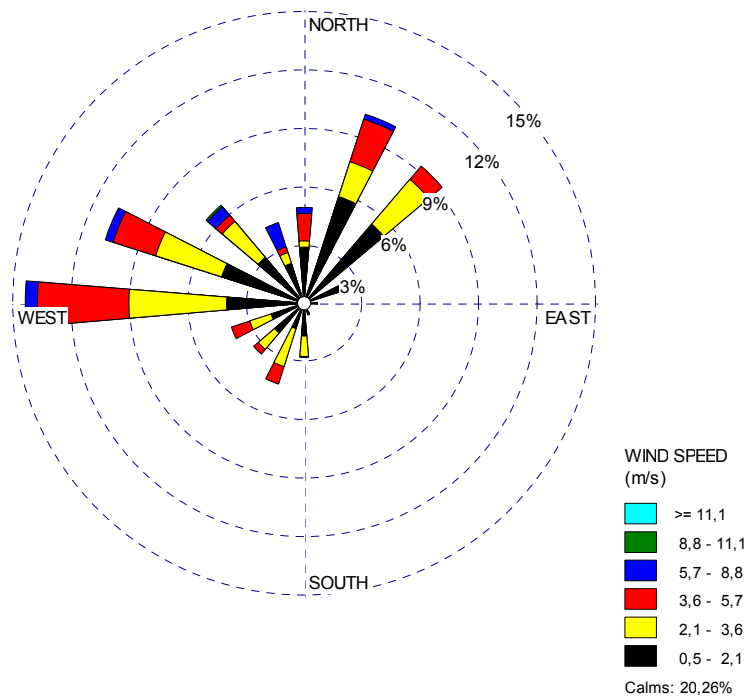


Figure 20. Wind rose depicting the frequency of occurrence of winds at Langhorne Creek, 28 August – 16 September 2009 (SAMDBNRM, 2009).

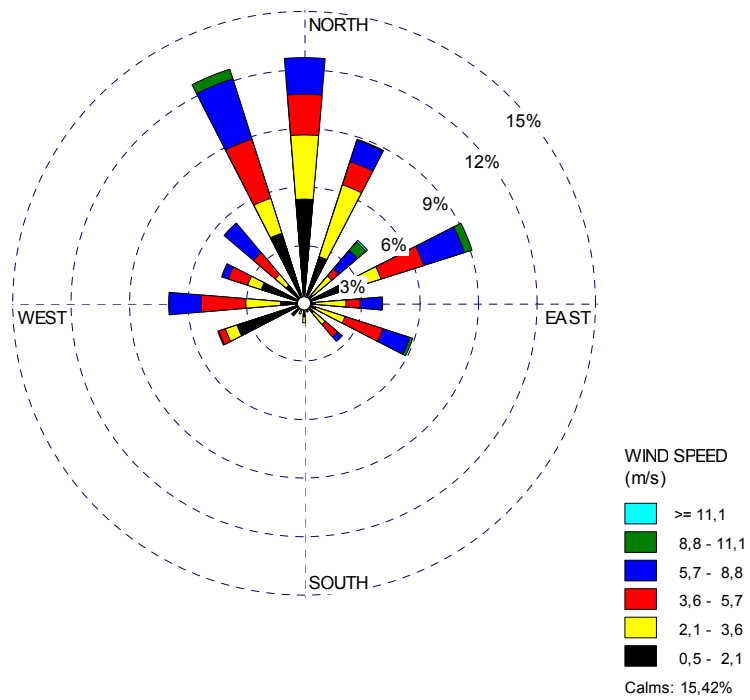


Figure 21. Wind rose depicting the frequency of occurrence of winds at Narrung, 28 August – 16 September 2009 (SAMDBNRM, 2009).

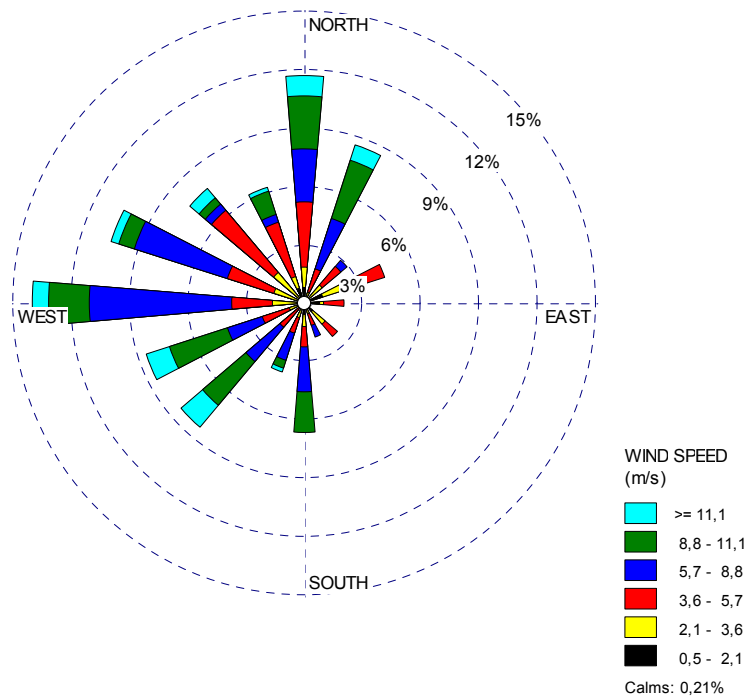


Figure 22. Wind rose depicting the frequency of occurrence of winds at Lake Albert (near Waltowa Swamp), 28 August – 16 September 2009 (DWLBC, 2009).

5.2.4 Surface water quality

Recent trends in surface water quality (pH, alkalinity and chloride to sulfate concentration ratio) from April 2008 to September 2009 are provided in Figures 23-25 for Lake Albert and 26-28 for Lake Alexandrina. Raw data are provided in Attachment J. The time period presented in Figures 23-28 was chosen based on the data available. Key findings from the water quality data are:

- pH in Lake Albert from April 2008 to September 2009 ranged from 7.8 (16 March 2009, South West Site – 2 km North of Warringe Point) to 8.9 (19 March 2009, Lake Albert Entrance, near the Narrows).
- pH at all locations in Lake Albert has remained in the vicinity of 8.5 from April 2008 to September 2009.
- pH in Lake Alexandrina from April 2008 to September 2009 ranged from 7.4 (4 August 2009, Clayton) to 9.4 (1 September 2008, Milang).
- pH in Lake Alexandrina has remained in the vicinity of 8.5 from April 2008 to September 2009, with the exception of the Clayton monitoring site, where the pH dropped below 7.5 in August 2009.
- Alkalinity at all sites analysed (except for Clayton and Murray River Opening, at the Murray River Inlet to Lake Alexandrina) at Lake Alexandrina ranged from 161-223 mg CaCO₃/L, and generally there was no significant trend in alkalinity from April 2008 to September 2009.
- Alkalinity at the Murray River Inlet (Opening) was generally lower than the rest of Lake Alexandrina, ranging from 104-150 mg CaCO₃/L, presumably influenced by inflowing water from the River Murray. Alkalinity at Clayton ranged from 65 mg CaCO₃/L (4 August 2009) to 266 mg CaCO₃/L (19 February 2009).
- Inflows from Finnis River (and Currency Creek) and/or flushing of acid water from local exposed sediments may have influenced water quality at Clayton during the period July-August 2009. This is indicated by the factor of 2 decrease in chloride to sulfate ratio (indicating addition of sulfate to the water) and corresponding decreases in alkalinity (to 65 mg CaCO₃/L) and pH (to 7.4).
- Water chemistry at Poltalloch Plains also indicated an addition of sulfate to Lake Alexandrina from April to July 2009, as indicated by a decrease in chloride to sulfate ratio (from 6.0-6.5 to 5.0-5.5) and corresponding decrease in alkalinity (from around 180 to 160 mg/L CaCO₃).
- Alkalinity in Lake Albert ranged from 166 mg CaCO₃/L (18 August 2008 at Lake Albert Entrance, near the Narrows) to 290 mg CaCO₃/L (20 April 2009 at Meningie). Trends in alkalinity at Meningie, Lake Albert Entrance and South West Lake Albert were similar. The variation in alkalinity at Lake Albert Entrance is thought to be due to dilution with lower alkalinity water from Lake Alexandrina. This seems to be consistent with the cessation of pumping on 30 June 2009, from which the trend re-aligns with that from the other Lake Albert monitoring sites.

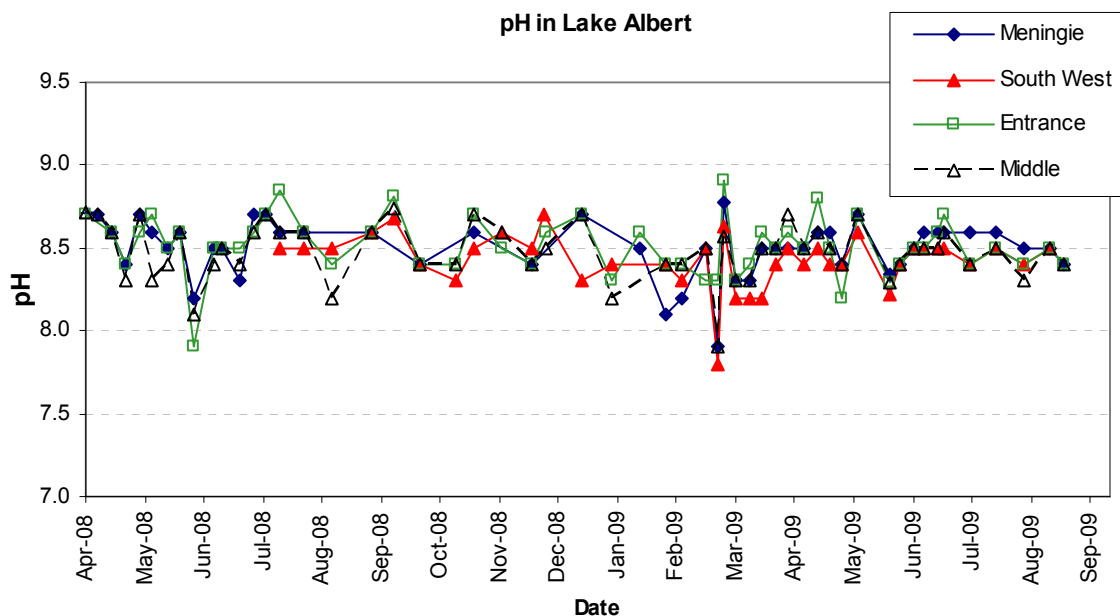


Figure 23. Recent trends in pH in Lake Albert (DWLBC, 2009).

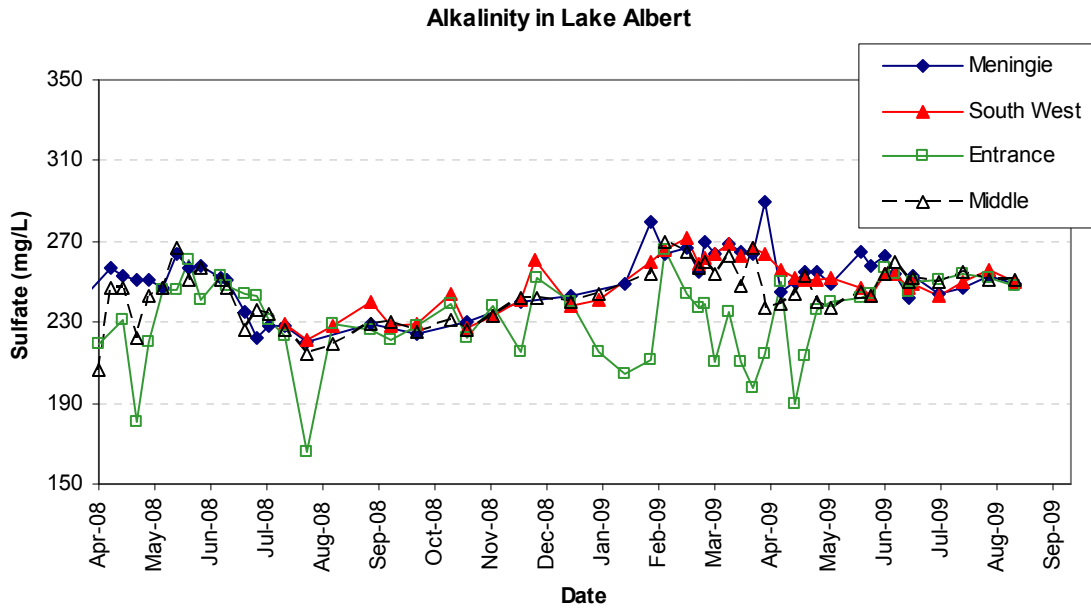


Figure 24. Recent trends in alkalinity in Lake Albert (DWLBC, 2009).

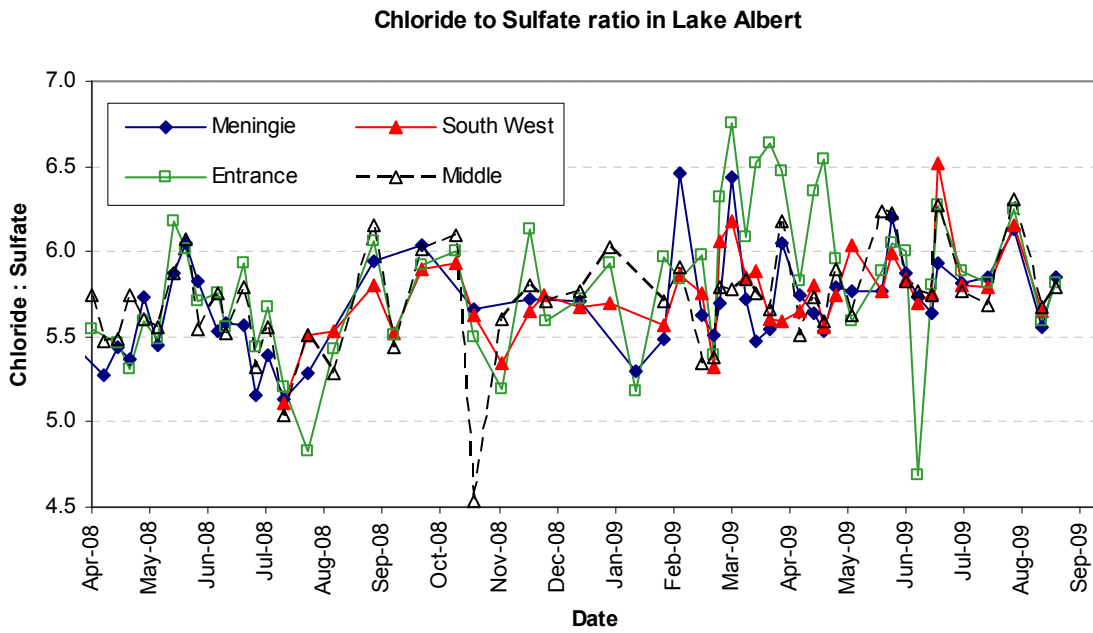


Figure 25. Recent trends in chloride to sulfate concentration ratios in Lake Albert (DWLBC, 2009).

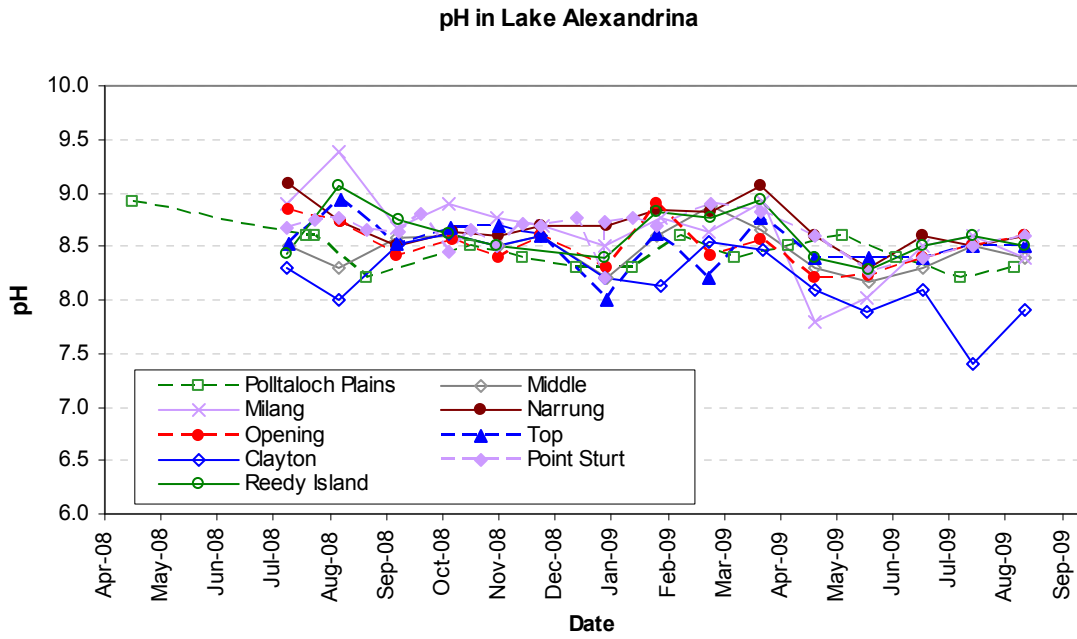


Figure 26. Recent trends in pH in Lake Alexandrina (DWLBC, 2009).

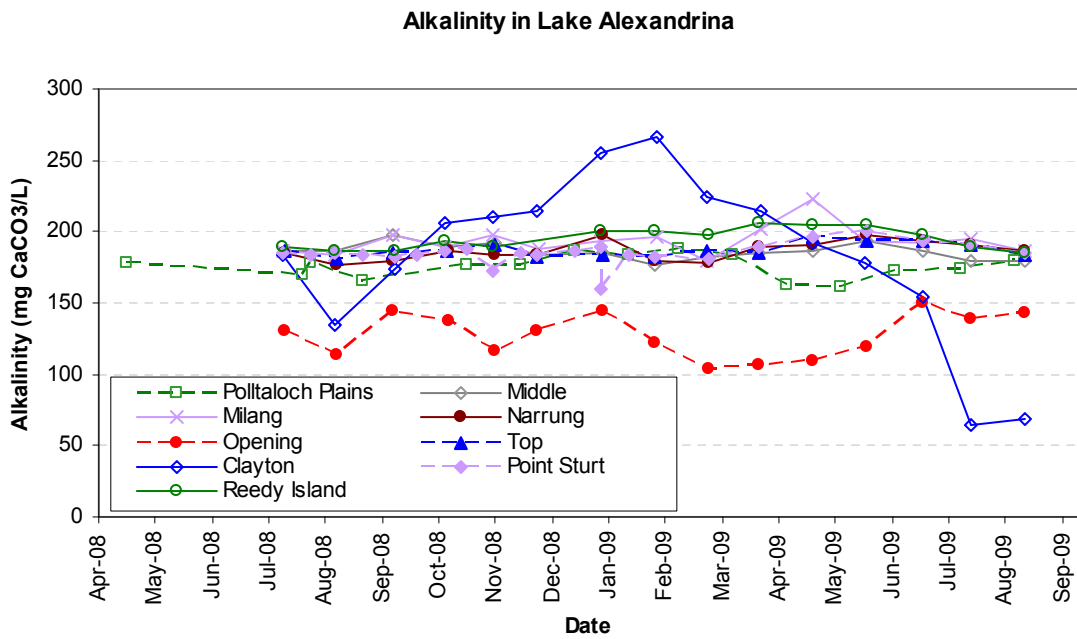


Figure 27. Recent trends in alkalinity in Lake Alexandrina (DWLBC, 2009).

Chloride to sulfate ratio in Lake Alexandrina

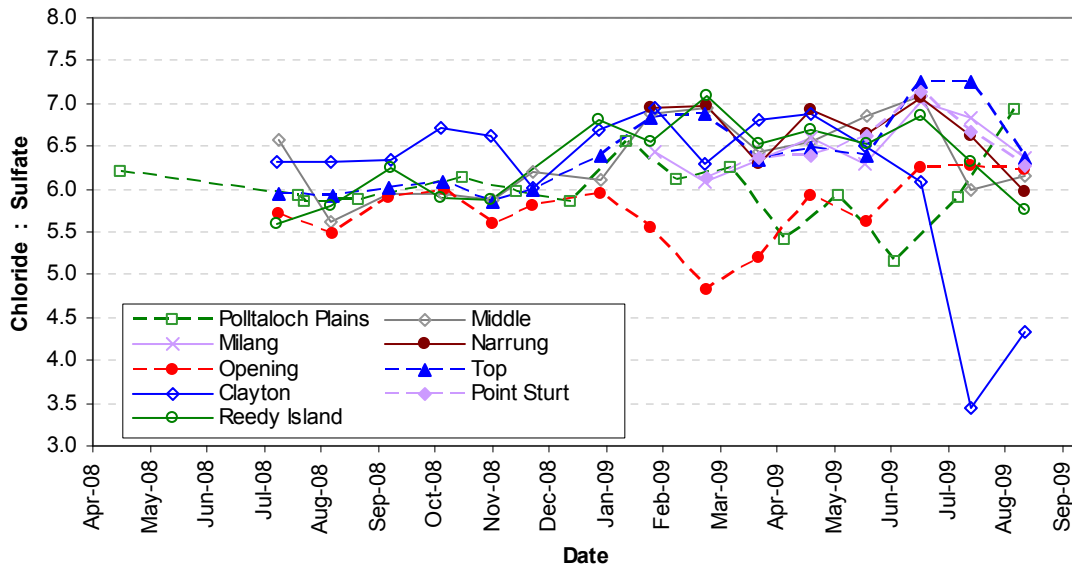


Figure 28. Recent trends in chloride to sulfate concentration ratios in Lake Alexandrina (DWLBC, 2009).

Recent trends in surface water quality (pH, EC, alkalinity and acidity) and rainfall in Currency Creek from May to October 2009 are provided in Figures 29-32. Raw data are provided in Attachment J (Currency Creek water quality) and Attachment I (15 minute rainfall depth at Currency Creek). The time period presented in Figures 29-32 was chosen based on the data available and includes the period of emergency response limestone addition from early June to August 2009 and when pumping from Lake Alexandrina to the Goolwa Channel / Currency Creek / Finniss River region commenced following the installation of a flow regulator at Clayton in late August 2009. Key findings from the water quality data are:

- Water upstream in Currency Creek (CC-US1) had pH ranging from 7.12 to 8.36, and an EC of 0.93 to 8.42 mS/cm. EC was generally higher during drier months. Average alkalinity in background water was 156 mg/L CaCO₃. Alkalinity dropped to between 65 and 85 mg/L CaCO₃ from 13 July to 28 September 2009.
- pH was lowest at Lower Currency Creek (CC-DS4) ranging from 2.81 on 22 May 2009 to 7.89 on 4 September 2009 (after commencement of pumping into Currency Creek). pH in Upper Currency Creek (CC-291) was in the acidic range (3.73 to 5.37) from early May to early July 2009 then remained in the near neutral range until 10 August 2009. pH at Downstream Currency Creek was initially measured at 5.26 (CC-DS5 on 20 July 2009), however has since remained in the neutral to alkaline range at CC-DS5 and CC-DS7.
- Water quality in Currency Creek was largely improved by mid-late July 2009 as a result of limestone addition to the creek and exposed acidic sediments. This is evident from trends in increasing pH and decreasing acidity at Upper Currency Creek (CC-291) and Currency Creek Hill (CC-DS2) from May to July 2009 and near neutral to alkaline water (alkalinity ranging from 45 to 145 mg/L CaCO₃) discharging from Currency Creek at site CC-DS5.
- Elevated acidity and low pH at CC-DS4 are considered to be related to localised discharge of acidic pore water during seiche events which create a localised increase in hydraulic gradient from the sediments towards the creek. Continued release of acidity from groundwater in the sediments at CC-DS4 (after limestone addition) affected surface water quality on the creek margin over several months via this mechanism. However, the acidity load from this area has not had significant long-term impacts on the quality of water discharging from Currency Creek further downstream (CC-DS5).
- Alkalinity at all sites increased following the commencement of pumping from Lake Alexandrina into the Goolwa Channel across the Clayton regulator in late August 2009. Alkalinity at Upper Currency Creek remained lower than sites at Lower Currency Creek.
- The sharp increase in pH at Lower Currency Creek (CC-DS4) from 4.44 on 24 August 2009 to 6.37 on 28 August 2009 can be attributed to rapid addition of alkaline water into Currency Creek.
- EC was highest at the monitoring site near Currency Creek Hill (56 mS/cm on 25 May 2009). EC at all sites in Currency Creek tended to peak from 20 to 22 May 2009, several days prior to the commencement of limestone addition. EC at all sites tended to decrease to below 10 mS/cm by mid-July 2009 due to significant fresh water dilution and acidity neutralisation.

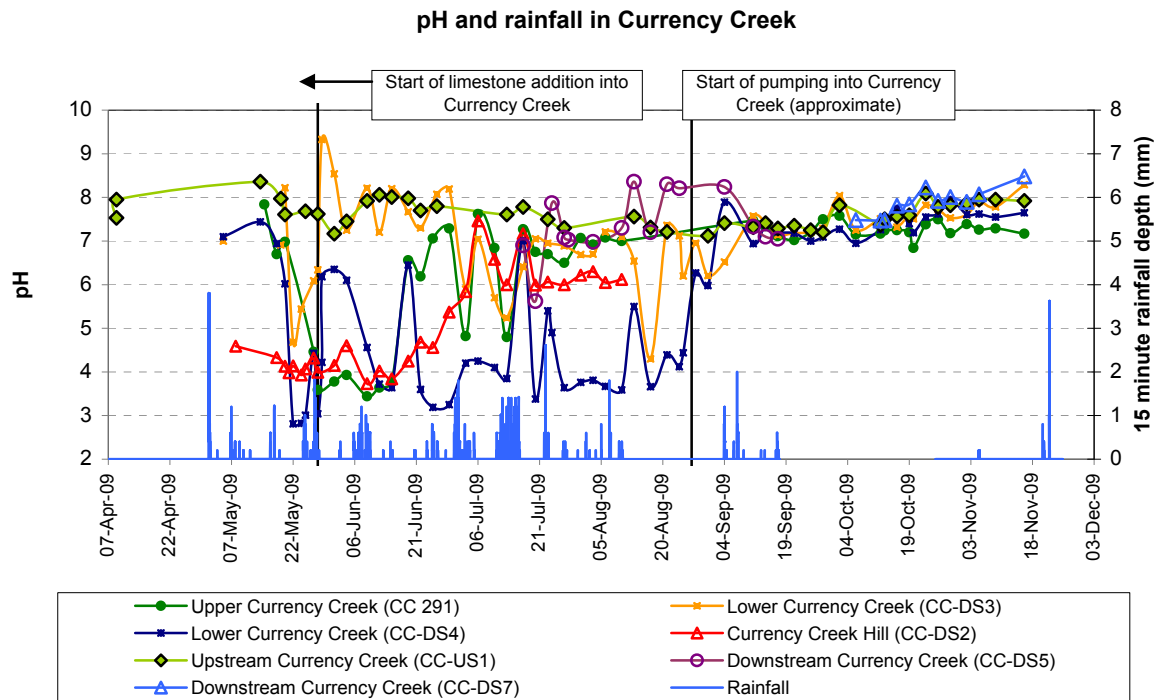


Figure 29. pH in Currency Creek from May to November 2009.

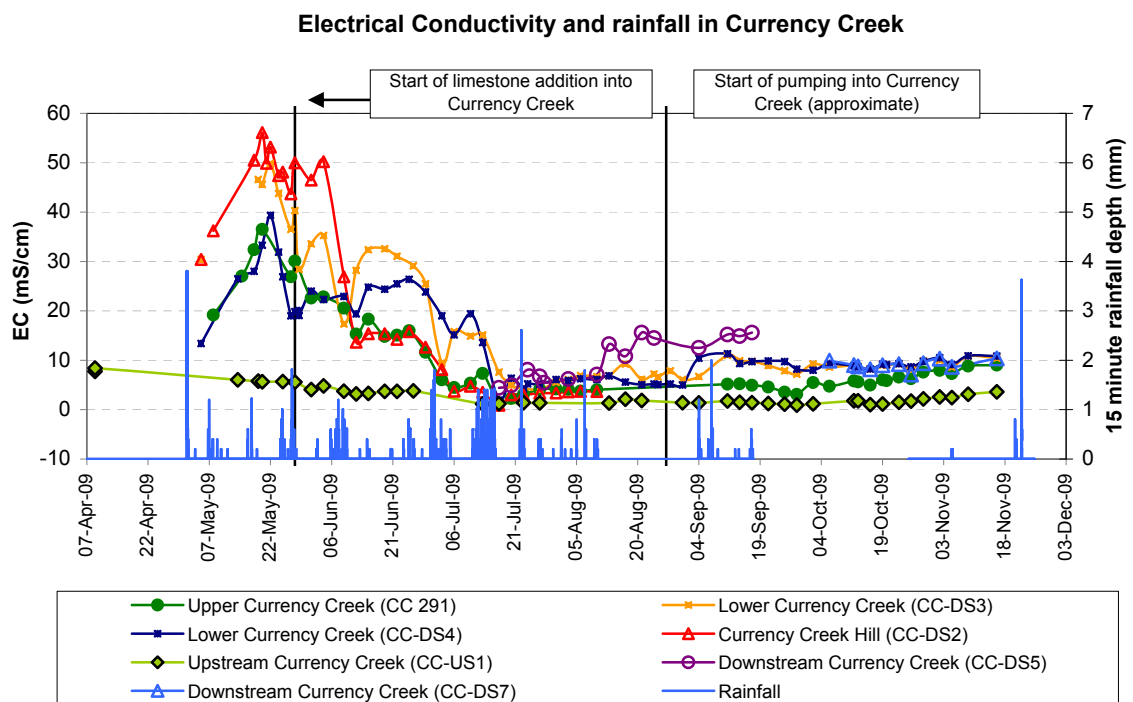


Figure 30. Electrical Conductivity (EC) in Currency Creek from May to November 2009.

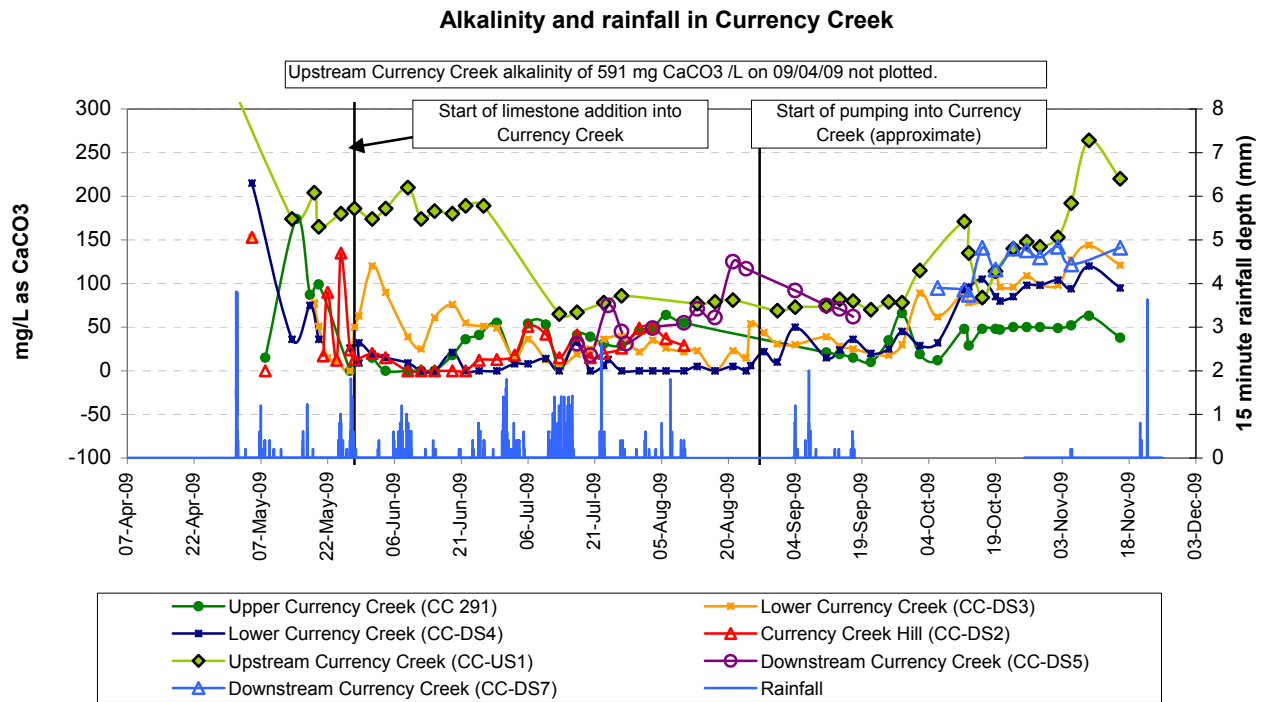


Figure 31. Alkalinity in Currency Creek from May to November 2009.

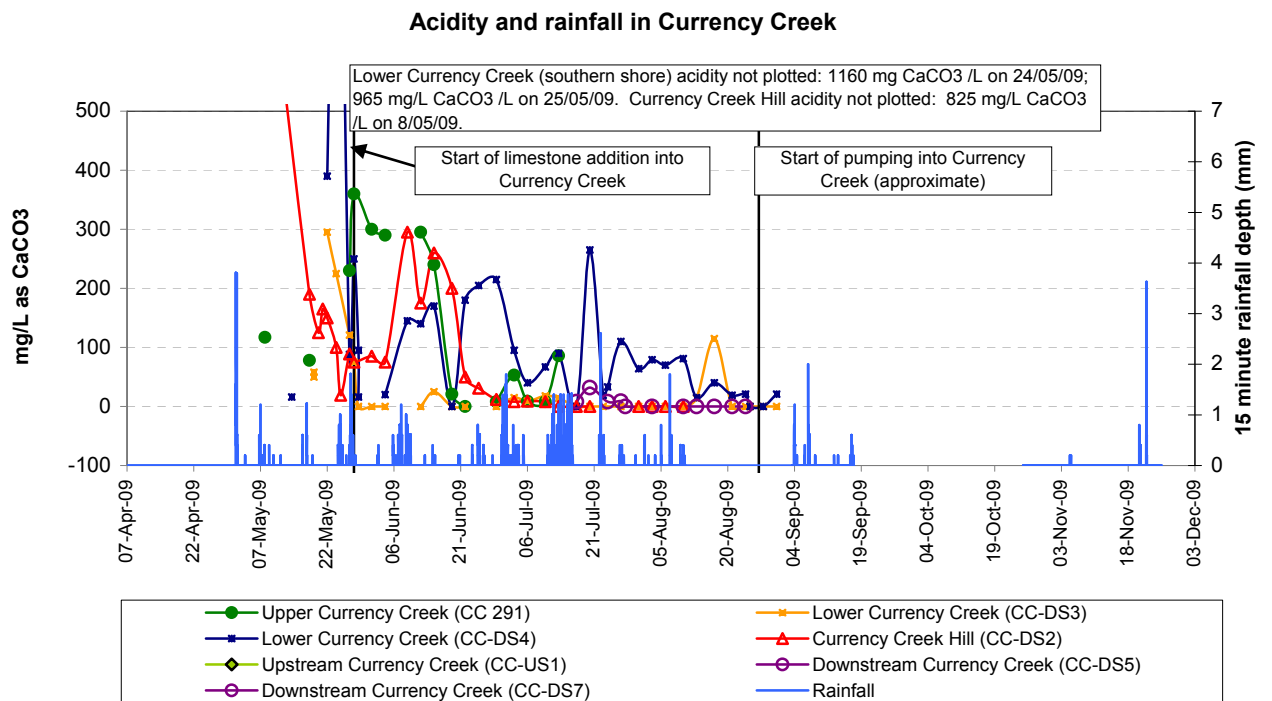


Figure 32. Acidity in Currency Creek from May to November 2009.

5.2.5 Sediment moisture

Raw data collected from the moisture sensors installed at Point Sturt, Campbell Park and the Windmill location are provided in Attachment K. The hourly moisture content data are graphed alongside 15 minute rainfall data in Figures 33, 34 and 35, respectively. Figure 36 shows all moisture content data alongside water levels for the adjacent piezometers at each site. In addition, Figures 37-39 provide a summary of average, minimum and maximum moisture profile data for the Point Sturt, Campbell Park and Windmill locations.

The key results are summarised below:

- At all three locations, the moisture content progressively increases with depth below ground, approaching saturation at a depth of 30-40 cm.
- The data across all three locations are generally consistent (given local lithological variations) suggesting that the results are broadly representative of moisture conditions throughout the sandy sediments around the periphery of both lakes.
- Saturated conditions are indicated by moisture contents of around 40-50 vol% H₂O, assuming a sediment porosity of 40-50 vol%. This is consistent with the relatively constant moisture content of 46-47 vol% measured at a depth of 40 cm at Point Sturt and 30 cm at Campbell Park.
- In general, there has been a progressive decrease in moisture content over time at all three sites, from late August to mid-November 2009. This trend is most evident at shallower depths in the sediment profile (10-20 cm below ground).
- Hourly moisture content data are responsive to some but not all rainfall events. This applies to all depths monitored from late August to mid-November 2009 (10-30 cm at Point Sturt and Campbell Park).
- The effect of rainfall on moisture content is most evident at shallower depths in the sediment profile and during higher intensity and/or longer duration rainfall events.
- Where the rainfall intensity or duration is sufficient to affect sediment moisture content, peak moisture values generally occur within a few hours of the onset of rainfall. Moisture contents can increase by up to 30 vol% following a significant rainfall event, particularly in the upper 10-20 cm of the sediment profile. After significant rainfall events, moisture contents can take several days to recover to pre-event values.
- Where the rainfall intensity or duration is sufficient to affect sediment moisture content, peak moisture values in the upper sediments are achieved more rapidly at the Windmill location (e.g. within 5 hours) than the Point Sturt (e.g. 10 hours) and Campbell Park (e.g. 15 hours) locations. This trend is consistent with the relatively high hydraulic conductivity values (coarser sediments) at Windmill and low hydraulic conductivity values (finer sediments) at Campbell Park. A delay of around 20-30 hours was generally observed between the initial rise in moisture content in the upper 10 cm layer, and the peak moisture content at a depth of 40-50 cm below ground, at all locations. This indicates that vertical migration of infiltrating rainwater through the upper 40-50 cm occurs within approximately 1 day of the onset of a significant rainfall event.
- A strong correlation exists between moisture content and piezometric levels at all three sites, as shown in Figure 36. This is particularly evident in the moisture content data for the upper (unsaturated) sediment layers.
- A moisture content of 46 vol% was measured 40 cm below ground at Point Sturt on 15 September 2009. In the adjacent piezometer (PS-1S) the piezometric level was approximately 70 cm below ground. Similarly, a moisture content of 47 vol% was measured 30 cm below ground at Campbell Park on 16 September 2009. In the adjacent piezometer (CP-1S) the piezometric level was approximately 75 cm below ground. These measurements indicate that sediments remain near-saturated within 30 cm of the piezometric level.
- Low magnitude diurnal oscillations in moisture content at Windmill and Campbell Park are interpreted to be associated with the effects of Earth tides¹². This observation is also consistent with fluctuations in piezometric levels. The magnitude of diurnal oscillations in moisture content is around 1-2 vol% at the Windmill location and less than 0.5 vol% at Campbell Park. The significantly higher hydraulic conductivity of Windmill sediments ($K > 30$ m/day at Site 1) may be associated with the greater response to Earth tides (c.f. $K = 0.22$ m/day at Campbell Park, Site 1). In comparison, Earth tides were

¹² Earth tides (distinct from ocean tides) refer to the sub-metre motion of the Earth associated with gravitational forces of the moon and, to a lesser extent, the sun. Earth tides comprise diurnal constituents, with a typical cycle length of around 24 hours (one high tide or 'bulge' and one low tide or 'depression' per day) and semi-diurnal constituents, with a typical cycle length of around 12 hours, among other longer term constituents associated with the Earth's axial tilt, etc. Earth tides encompass the entire body of the Earth, including the outer crustal layers, surficial sediments / rocks, groundwater, etc.

barely evident in the Point Sturt moisture data, corresponding to the lowest K value of the three sites ($K = 0.09$ m/day).

- The average moisture profile data in Figures 37-39 demonstrate the progressive increase in moisture content with depth below ground surface at all sites. Much of the variation in moisture content with depth occurs over an interval of around 0.3 m at all sites. The lowest moisture contents observed in the upper 0.1 m were around 2 vol%, 4 vol% and 6 vol% at Point Sturt, Campbell Park and Windmill, respectively. Minimum and maximum moisture contents for each depth interval, as shown in Figures 37-39, demonstrate that the upper sediment layers (upper 0.1-0.2 m) have experienced significant variation over 3 months. This is attributed to the increased response to incident rainfall as well as greater evapotranspiration within the upper layers.
- The moisture profile data for Point Sturt (Figure 37) indicate that, while the piezometric level in the adjacent piezometer reached a minimum of 0.87 m below ground between 28 August and 18 November 2009, the sediments remained saturated (average moisture content 46-47 vol%) at a depth of 0.4 m throughout this period. Thus, approximately 0.4-0.5 m of sediments were effectively saturated above the minimum piezometric level.
- The moisture profile data for Campbell Park (Figure 38) indicate that, while the piezometric level in the adjacent piezometer reached a minimum of 0.84 m below ground between 27 August and 17 November 2009, the sediments remained saturated (average moisture content 39-46 vol%) at a depth of 0.3-0.5 m throughout this period. Thus, approximately 0.5 m of sediments were effectively saturated above the minimum piezometric level.

The apparent anomaly (decrease) in moisture content observed at a depth of 0.4 m (Figure 38) is attributed to lower sediment porosity in this layer, which limits the moisture holding capacity of the sediment.

- The moisture profile data for the Windmill location (Figure 39) indicate that, while the piezometric level in the adjacent piezometer reached a minimum of 0.56 m below ground between 20 October and 16 November 2009, the sediments remained saturated (average moisture content around 50 vol%) at a depth of 0.3 m throughout this period. Thus, approximately 0.2-0.3 m of sediments were effectively saturated above the minimum piezometric level.

The relatively high moisture contents measured at the Windmill location may be attributed to higher porosity in these sediments. Furthermore, the effects of seiching at the Windmill location could contribute to the higher moisture contents. This is apparent from a rapid rise in piezometric level around 25 October 2009 at the Windmill location (Site 1) and subsequent rise in moisture content at multiple depths, despite the lack of rainfall prior to this event (Figure 35). Further evidence of seiching is discussed in Section 5.2.6.

The relatively minor variation in moisture contents at all depths at the Windmill location (Figure 39) is due to the considerably shorter monitoring duration at this site.

- The zone of effectively saturated sediments observed at all sites may reduce in thickness over time, although longer term monitoring will be required to confirm this.

Point Sturt moisture contents

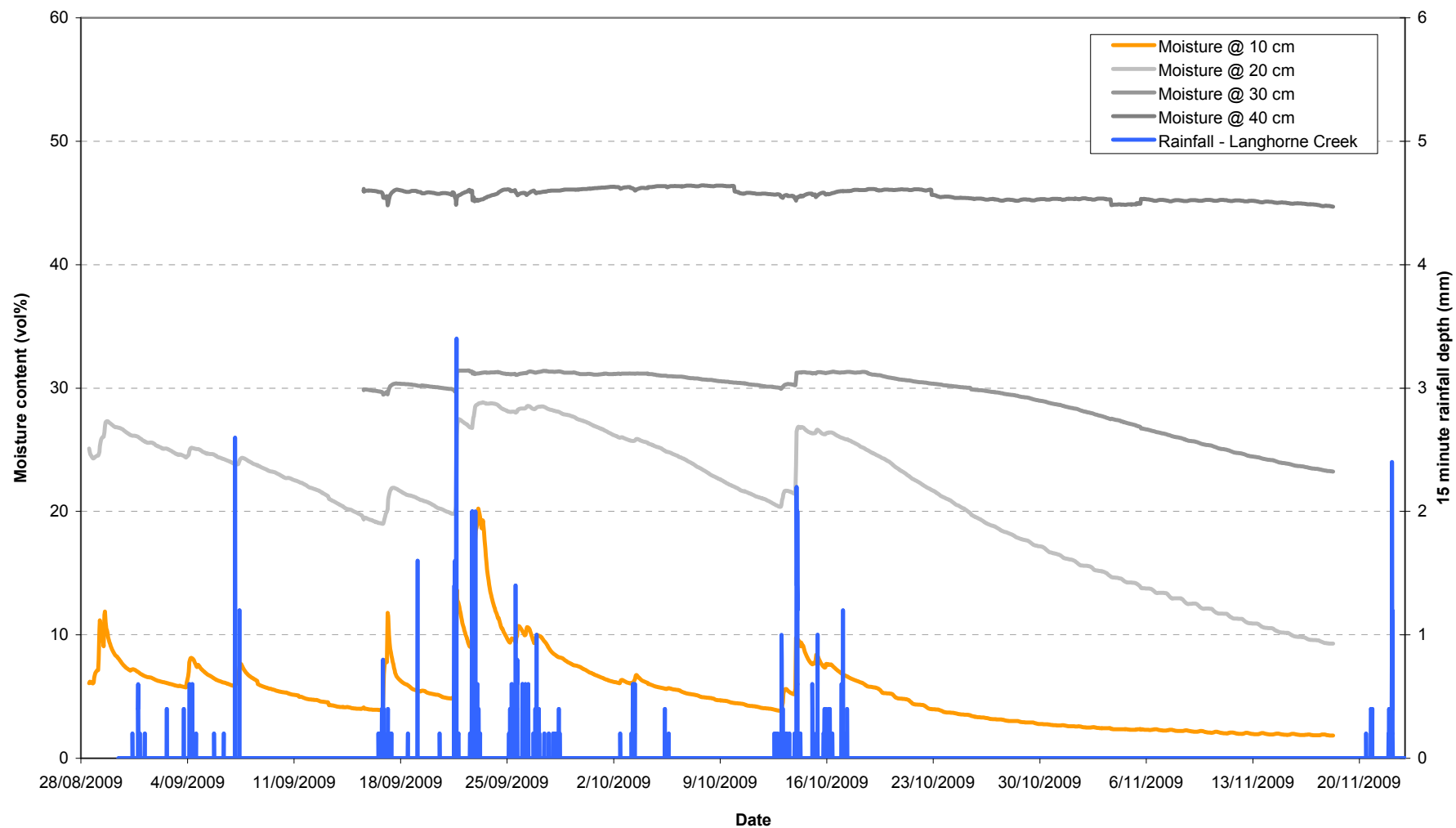


Figure 33. Temporal variation in sediment moisture contents at Point Sturt from 28 August to 18 November 2009.

Campbell Park moisture contents

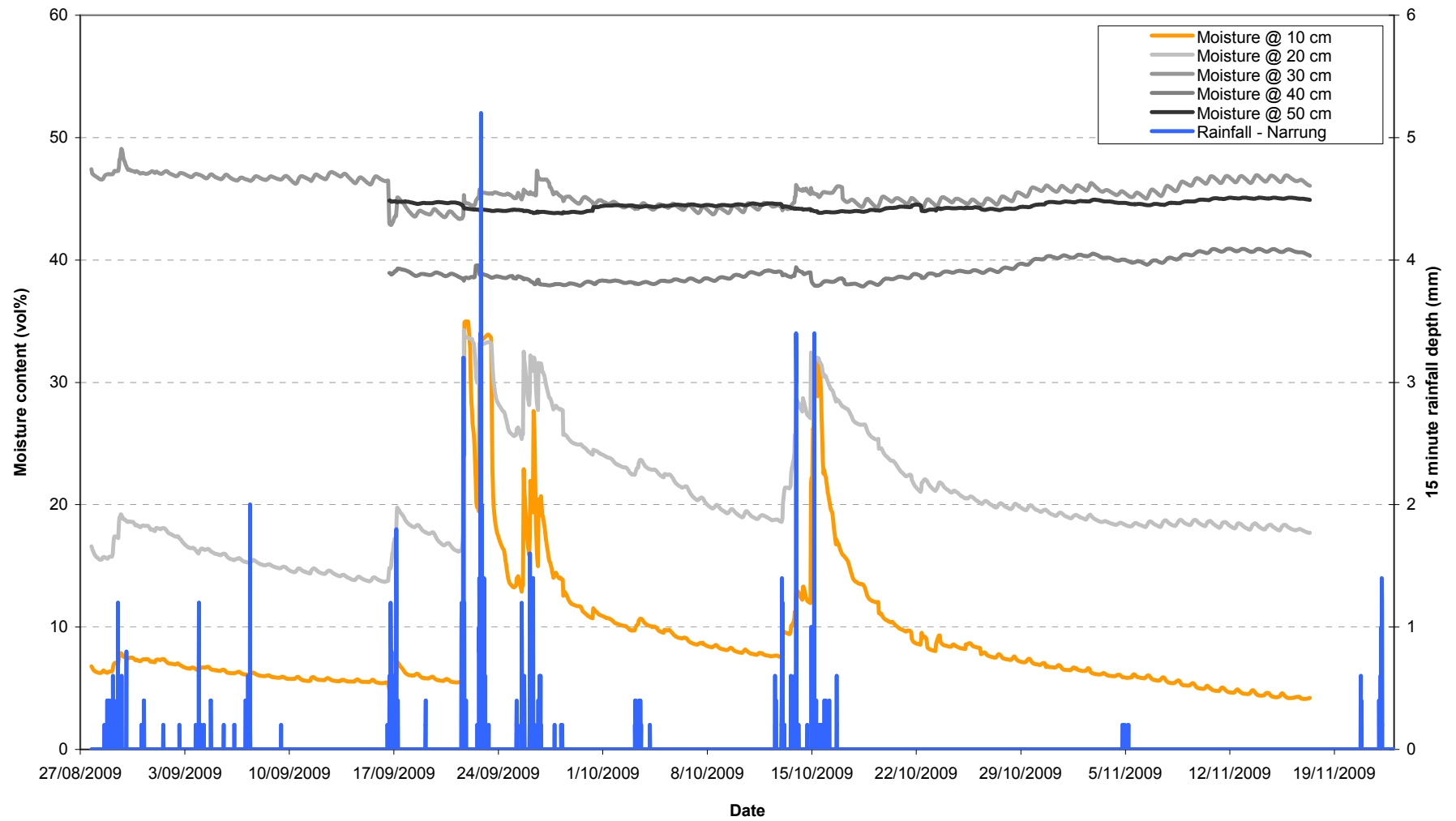


Figure 34. Temporal variation in sediment moisture contents at Campbell Park from 27 August to 17 November 2009.

Windmill moisture contents

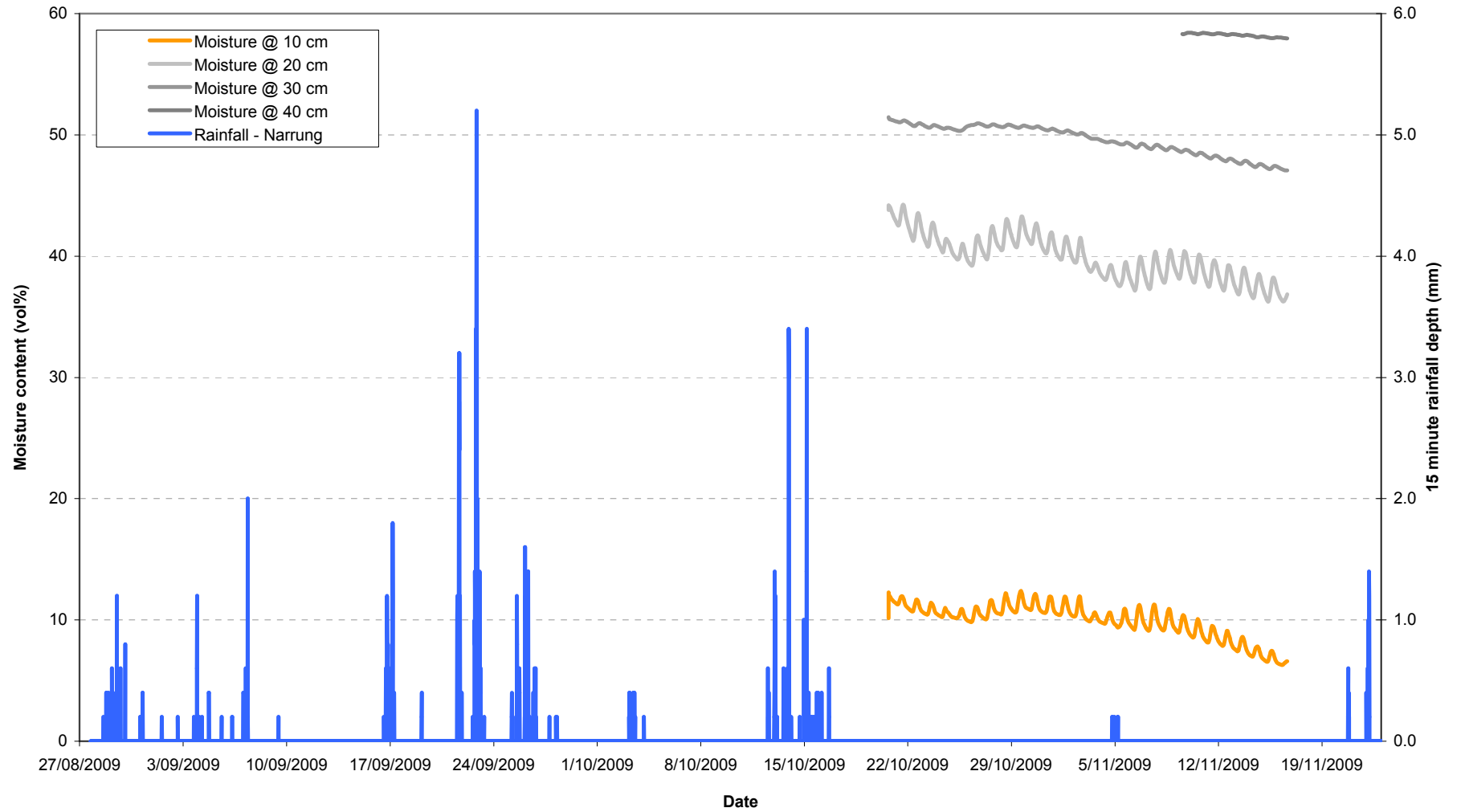


Figure 35. Temporal variation in sediment moisture contents at the Windmill location from 20 October to 17 November 2009.

Comparison of moisture contents and piezometric levels

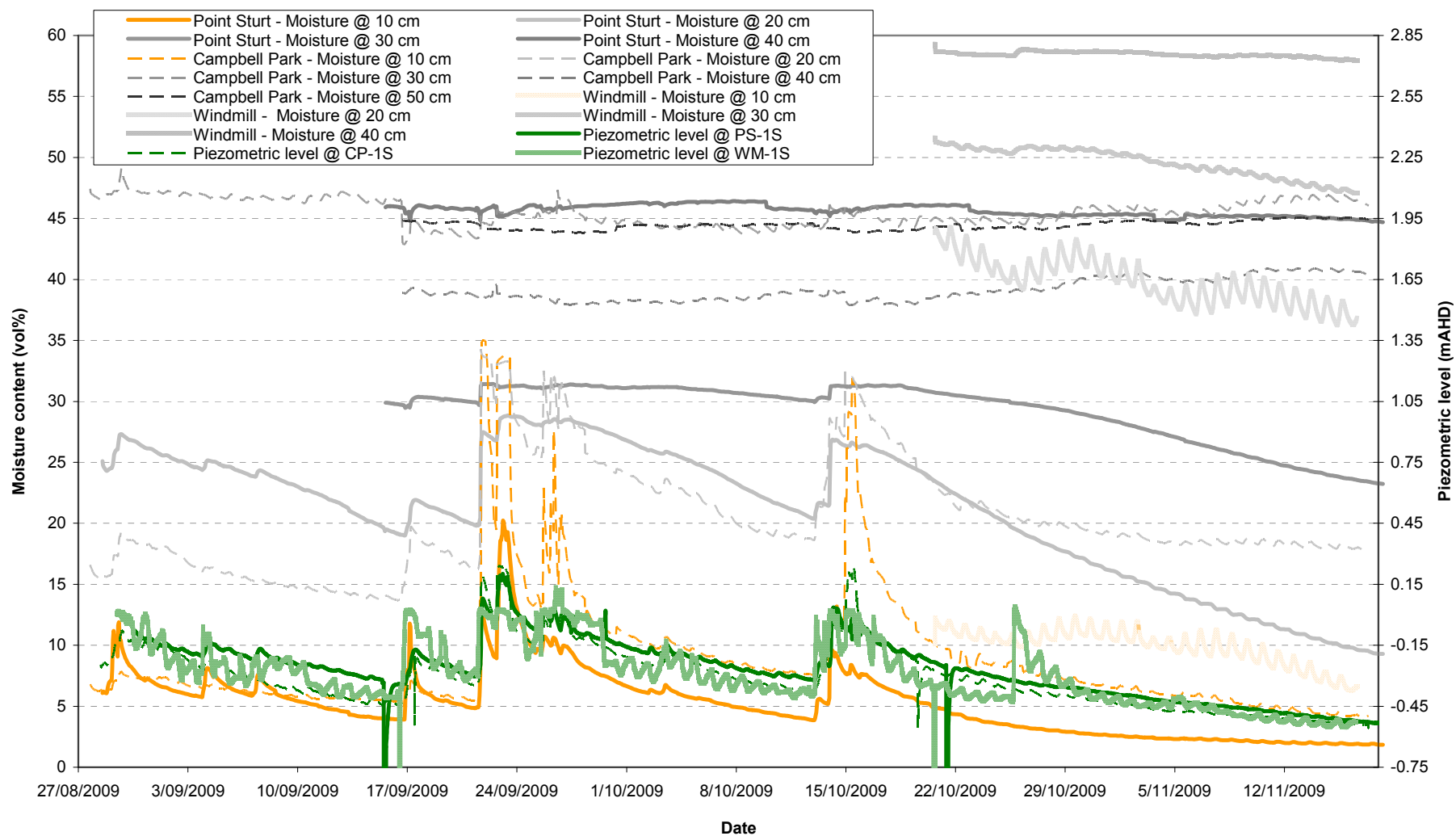


Figure 36. Comparison of moisture contents and piezometric levels at Point Sturt, Campbell Park and Windmill locations.

Point Sturt moisture profile data (28/08/2009 - 18/11/2009)

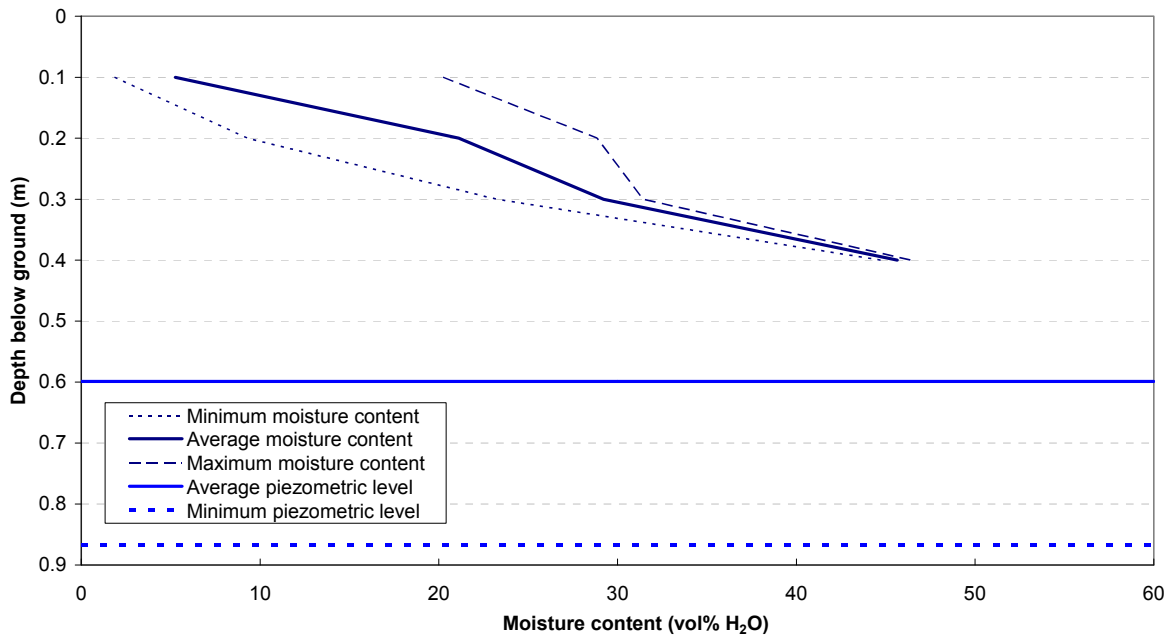


Figure 37. Average, minimum and maximum moisture content profiles at the Point Sturt location, from 28 August to 18 November 2009. Average and minimum piezometric levels at Piezometer Site 1 (nearest the moisture sensors) are shown for comparison.

Campbell Park moisture profile data (27/08/2009 - 17/11/2009)

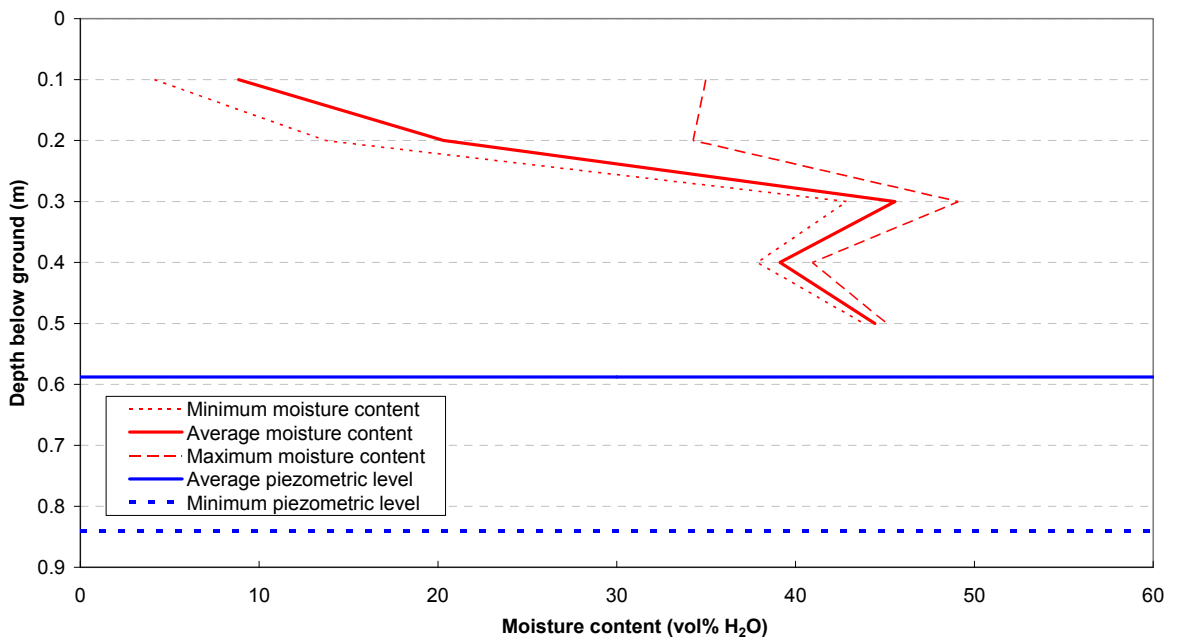


Figure 38. Average, minimum and maximum moisture content profiles at the Campbell Park location, from 27 August to 17 November 2009. Average and minimum piezometric levels at Piezometer Site 1 (nearest the moisture sensors) are shown for comparison.

Windmill moisture profile data (20/10/2009 - 16/11/2009)

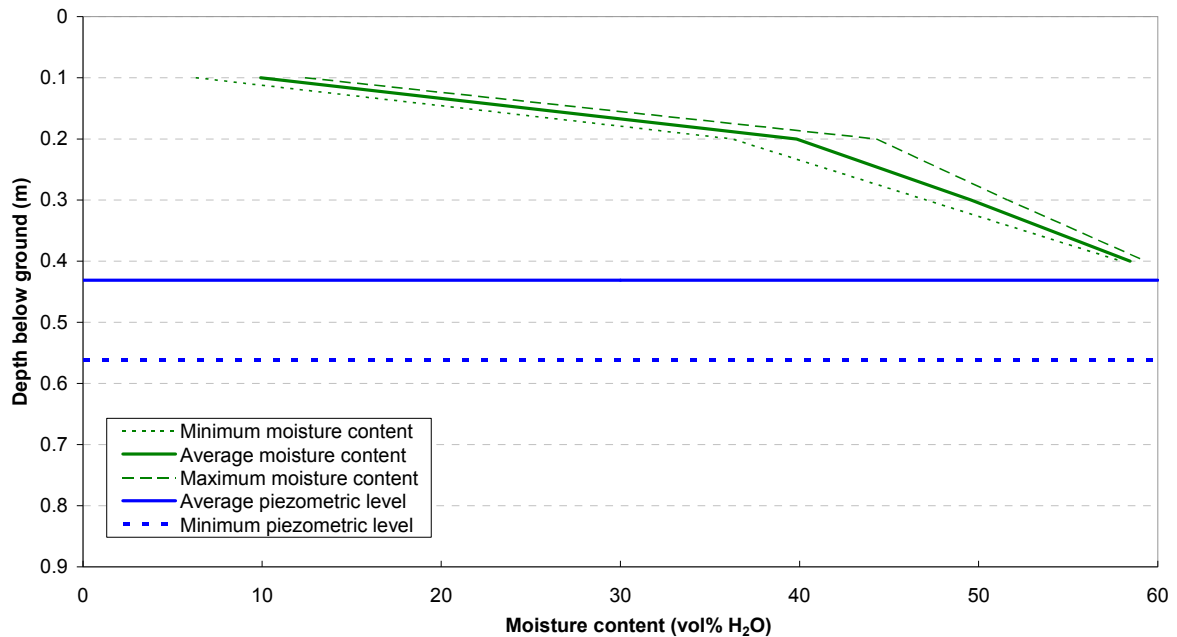


Figure 39. Average, minimum and maximum moisture content profiles at the Windmill location, from 20 October to 16 November 2009. Average and minimum piezometric levels at Piezometer Site 1 (nearest the moisture sensors) are shown for comparison.

5.2.6 Groundwater (piezometric) levels

Currency Creek

Piezometric level results are graphed alongside rainfall data and surface water level data in Figure 40. The key results are summarised below:

- Piezometric levels decrease with proximity to the Goolwa Channel, from UCC-P1 to UCC-P3 to LCC-P2 (as expected). From mid-May to mid-September 2009, levels in UCC-P3 were approximately 0.2 m lower than in UCC-P1, while the levels at LCC-P2 were approximately 0.2-0.3 m below UCC-P1.
- Piezometric levels increased by around 0.5-0.6 m from mid-May to mid-July 2009 at UCC-P1 and LCC-P2. In mid-July, levels reached 0.294 m AHD and -0.050 m AHD in UCC-P1 and LCC-P2, respectively. UCC-P1 levels subsequently declined to -0.045 m AHD by mid-September 2009, while LCC-P2 levels peaked at -0.005 m AHD in late August before decreasing to -0.212 m AHD in mid-September.
- Piezometric levels at UCC-P1 and LCC-P2 show a clear response to some but not all rainfall events. The effect of rainfall on piezometric levels is most evident during higher intensity and/or longer duration rainfall events.
- If the rainfall intensity or duration is sufficient to affect piezometric levels, there is a lag of around 1 day between the onset of rainfall and peak water levels. Piezometric levels generally take several days to recover to pre-event values.
- The rapid rise in surface water levels commencing in late August corresponded to the pumping of water from Lake Alexandrina into the Goolwa Channel / Currency Creek / Finniss River region. Surface water levels had risen by around 1.3 m by 7 November 2009. Throughout October and early November 2009, there was a close correlation between surface and piezometric levels. The surface water level was at or above the ground level at each piezometer site for much of this period.
- The above results demonstrate that rainfall was the key influence on changing piezometric levels in Currency Creek from May to September 2009. After October 2009, as the creek was refilling, surface water levels became the key influence on piezometric levels. Thus, the rewetting of previously exposed ASS in Currency Creek was initially dominated by rainfall, until the point at which surface water levels began to recover via pumping from Lake Alexandrina.

- The sharp rise in piezometric levels at both UCC-P1 and LCC-P2 on 21 September 2009 may be attributed to the effects of seiching, although it is unclear why a comparable rise in surface water levels was not observed on this date.
- Low magnitude daily oscillations in piezometric levels are interpreted to be associated with the effects of Earth tides. This observation is also consistent with fluctuations in moisture content, as described in Section 5.2.7.

Point Sturt, Campbell Park and Windmill locations

Piezometric level results are graphed alongside rainfall and surface water level data in Figures 41, 42 and 43, respectively. Variations in hydraulic gradient with time at each site are graphed in Attachment L. The key results are summarised below:

- Piezometric levels at Point Sturt generally decrease with proximity to the lake surface water, indicating the potential for groundwater to flow towards the lake (as expected). In some cases, however, piezometric levels at Site 4 temporarily exceed those at Site 3, indicating the potential for flow in the reverse direction. Such events are interpreted to be related to periods of seiching.
- From late August to mid-November 2009, piezometric levels decreased in all Point Sturt piezometers, with the magnitude of the change decreasing with proximity to the lake water (from Site 1 to 4). For example, levels decreased by 0.46 m at Site 1, 0.40 m at Site 2, 0.35 m at Site 3 and 0.11 m at Site 4.
- In the lower layer of sandy sediments at Point Sturt, piezometric levels were generally slightly higher (by around 0.05-0.10 m) than those measured in the upper sand layer. The only exception was at Site 1 (nearest the shore) where the piezometric levels in the deeper sediments were 0.15-0.25 m lower than in the upper sediments. Piezometric levels in the deeper sand layer at Site 1 were nevertheless only 0.6-0.7 m below ground. The discrepancies in piezometric levels at all sites indicate locally disconnected aquifers throughout the Point Sturt transect.
- At Point Sturt, the hydraulic gradient from Site 2 to 4 varied considerably during the wetter months of August to October 2009 but tended to stabilise in the range 0.0004-0.0006 by mid-November 2009, as graphed in Attachment L. In comparison, the ground surface gradient from Site 2 to 4 was 0.0023¹³. Thus, the hydraulic gradient in mid-November 2009 represented approximately 15-25% of the beach slope.
- Piezometric levels at the Point Sturt nearest the lake water (Site 4) generally exceeded surface water levels measured in Lake Alexandrina at Beacon 97 by less than 0.1-0.2 m in August and September 2009 (as expected). However, the reverse was apparent in October and November, with surface water levels at Pt McLeay exceeding piezometric levels at Site 4 by 0.1-0.2 m. As there is no clear evidence of seiching at Site 4 during this period, the surface water level data for Pt McLeay (from DLWBC) is assumed to overestimate actual surface water levels at the Point Sturt transect.
- Overall, piezometric levels at Point Sturt have generally remained below the ground surface, with the exception of Site 4 (nearest the lake water) in late September and mid-October 2009. These occasions coincided with significant rainfall events, indicating that rainwater ponding and infiltration through the sediments, rather than lake water seiching, was responsible for the elevated piezometric levels.
- Piezometric levels at Campbell Park (Sites 1-4) have decreased by around 0.5-0.9 m from late August to mid-November 2009.
- As with Point Sturt, piezometric levels at Campbell Park generally decrease with proximity to the lake surface water, indicating the potential for groundwater to flow towards the lake (as expected).
- At Campbell Park, the hydraulic gradient from Site 2 to 4 varied considerably during the wetter months of August to October 2009 but tended to stabilise around 0.0019 by mid-November 2009, as graphed in Attachment L. In comparison, the ground surface gradient from Site 2 to 4 was 0.0015. Thus, the hydraulic gradient in mid-November 2009 was approximately parallel with the beach slope.
- The variability in hydraulic gradients at Campbell Park was higher than observed at Point Sturt, particularly during the wetter months of August to October. Much of the variability occurred during

¹³ Hydraulic gradients from Site 1-4, Site 2-4 and Site 3-4 were calculated based on all available piezometric level data. The results are graphed alongside rainfall data in Attachment N. The estimates obtained from the Site 2-4 data were considered most indicative of average conditions in the lake sediments along the transect. The Site 1-4 data were likely to overestimate hydraulic gradient calculations due to the increased bank slope at the shore (Site 1). The Site 3-4 data were strongly affected by seiching throughout the monitoring period and therefore likely to either underestimate hydraulic gradients or result in reverse hydraulic gradients that are not representative of the entire transect. The Site 2-3 data were comparable to the Site 2-4 data but likely to be less accurate due to the shorter distance over which gradients were calculated (75 m between Sites 2 and 3, compared with 150 m between Sites 2 and 4). Thus, only the hydraulic gradients calculated between Sites 2 and 4 are discussed further in this report (this applies to Point Sturt, Campbell Park and Windmill locations).

rainfall events, while the clearest trends in piezometric levels at Campbell Park were observed during low/no rainfall periods such as 5-13 October and 20 October-17 November 2009.

- In the lower layer of sediments at Campbell Park, piezometric levels were similar to those measured in the upper layer (within 0.05 m) in late August. However, by mid-September, piezometric levels at Sites 2-4 were all approximately 0.2 m higher than those in the upper sediments. This suggests that the upper and lower aquifers are poorly connected, which is consistent with the thick clay layer observed between the two sandy horizons (Figure 10).
- Overall, piezometric levels at Campbell Park have generally remained below the ground surface, with the exception of Sites 2 and 4 in late September and mid-October 2009. These occasions coincided with significant rainfall events, indicating that rainwater ponding and infiltration through the sediments, rather than lake water seiching, was responsible for the elevated piezometric levels. This is consistent with observations in the Point Sturt piezometric level data.
- Piezometric levels at Campbell Park (all sites) were up to 0.3-0.4 m lower than surface water levels measured in Lake Albert near Waltowa Swamp and Warringee Point in mid-November 2009. The cause of this discrepancy is currently under investigation, but is believed to be associated with survey errors in DLWBC lake level monitoring sites.
- Recent data on piezometric levels in the upper sand layer at Campbell Park indicates that this layer is no longer saturated. Water levels in all upper sand piezometers are currently below the base of the upper sand layer.
- Piezometric levels at the Windmill location (Sites 1-3) have decreased by around 0.5-0.6 m from late August to mid-November 2009.
- As with Point Sturt and Campbell Park, piezometric levels at the Windmill location generally decrease with proximity to the lake surface water, indicating the potential for groundwater to flow towards the lake (as expected).
- At the Windmill location, the hydraulic gradient from Site 2 to 4 varied considerably throughout the monitoring period, approaching 0.0006 by mid-November 2009, as graphed in Attachment L. In comparison, the ground surface gradient from Site 2 to 4 was 0.0009. Thus, the hydraulic gradient in mid-November 2009 was approximately two-thirds that of the beach slope.
- In the lower layer of sandy sediments at the Windmill location, piezometric levels were similar to those measured in the upper layer at Site 1 (within 0.02-0.03 m). At Site 4, however, piezometric levels in the lower sand layer were 0.1 m higher than in the upper sand, indicating locally disconnected aquifers.
- As observed at Campbell Park, piezometric levels at the Windmill location (all sites) were around 0.3 m lower than surface water levels measured in Lake Albert near Waltowa Swamp in mid-November 2009. The cause of this discrepancy is currently under investigation, but is believed to be associated with survey errors in DLWBC lake level monitoring sites.
- Piezometric levels at the Windmill location have generally remained below the ground surface, however, there appears to be evidence of one significant seiching event (affecting two or more piezometers) around 25 October 2009, as indicated in Figure 43. During this event, piezometric levels increased rapidly by around 0.5-0.6 m at all sites, despite no corresponding rainfall event. Other occasions where piezometric levels exceeded the ground surface elevation (e.g. late September and mid-October 2009) coincided with significant rainfall events, indicating that rainwater ponding and infiltration through the sediments, rather than lake water seiching, was more likely to be responsible for the elevated piezometric levels. This is consistent with observations in the Point Sturt and Campbell Park piezometric level data.
- At all locations, piezometric levels show a clear response to some but not all rainfall events. The effect of rainfall on piezometric levels is most evident during higher intensity and/or longer duration rainfall events.
- If the rainfall intensity or duration is sufficient to affect piezometric levels, there is a lag of around 1-2 hours between the onset of rainfall and peak water levels. Piezometric levels generally take several hours to recover to pre-event values. This was more rapid than the response observed at Currency Creek, and is considered to be more realistic due to the method of piezometer installation. The rapid recovery of piezometric levels could be attributed to discharge of groundwater through the sediments to the lake water and/or increased evapotranspiration rates near the ground surface. The latter is considered more likely on the basis of groundwater quality results, as discussed in Section 5.2.7.
- Daily oscillations in piezometric levels are interpreted to be associated with the effects of Earth tides. This observation is also consistent with fluctuations in piezometric levels at Currency Creek and moisture content values, as described in Section 5.2.7.

Currency Creek rainfall and piezometric levels

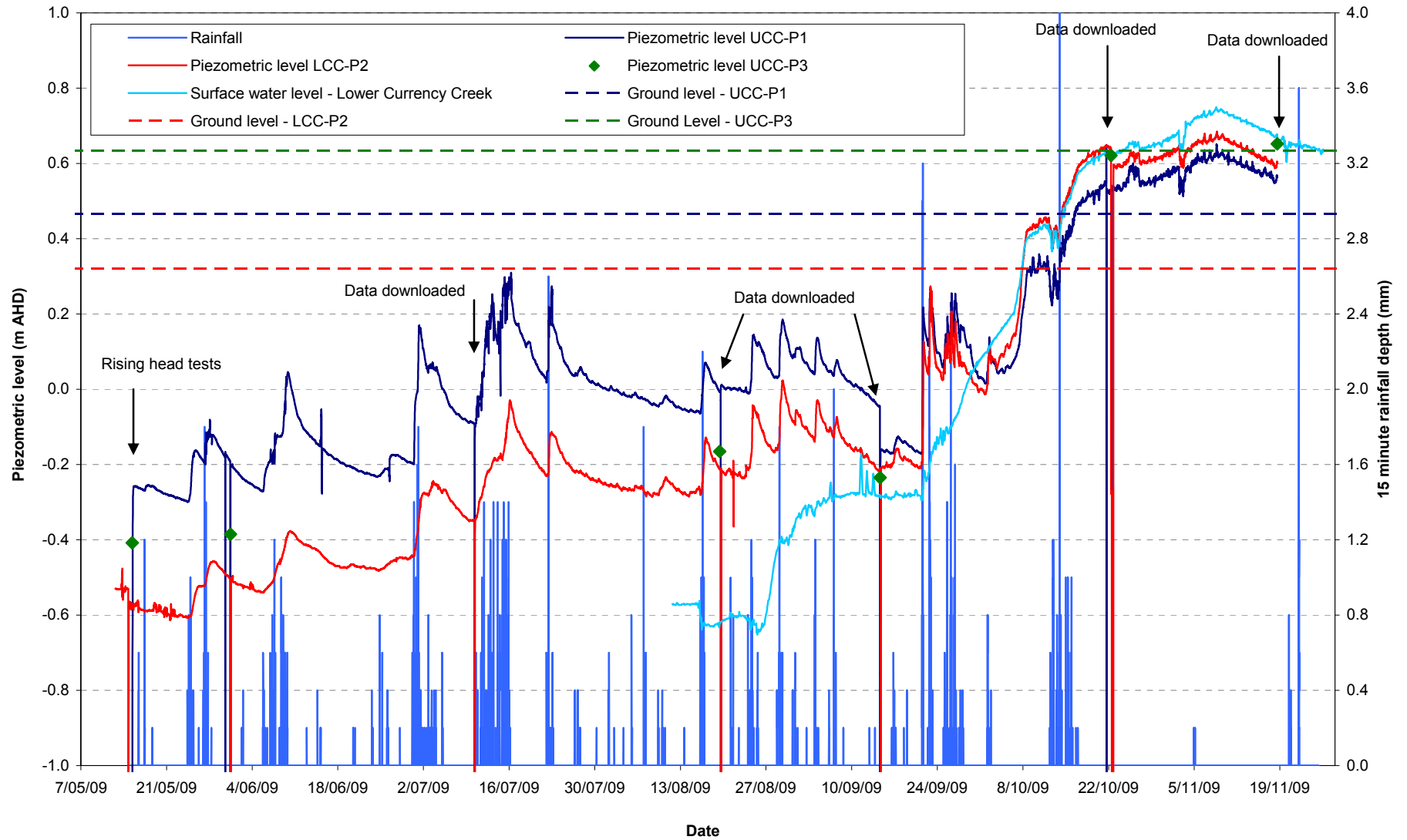


Figure 40. Temporal variation in piezometric levels at Currency Creek from 15 May to 18 November 2009.

Point Sturt rainfall and piezometric levels

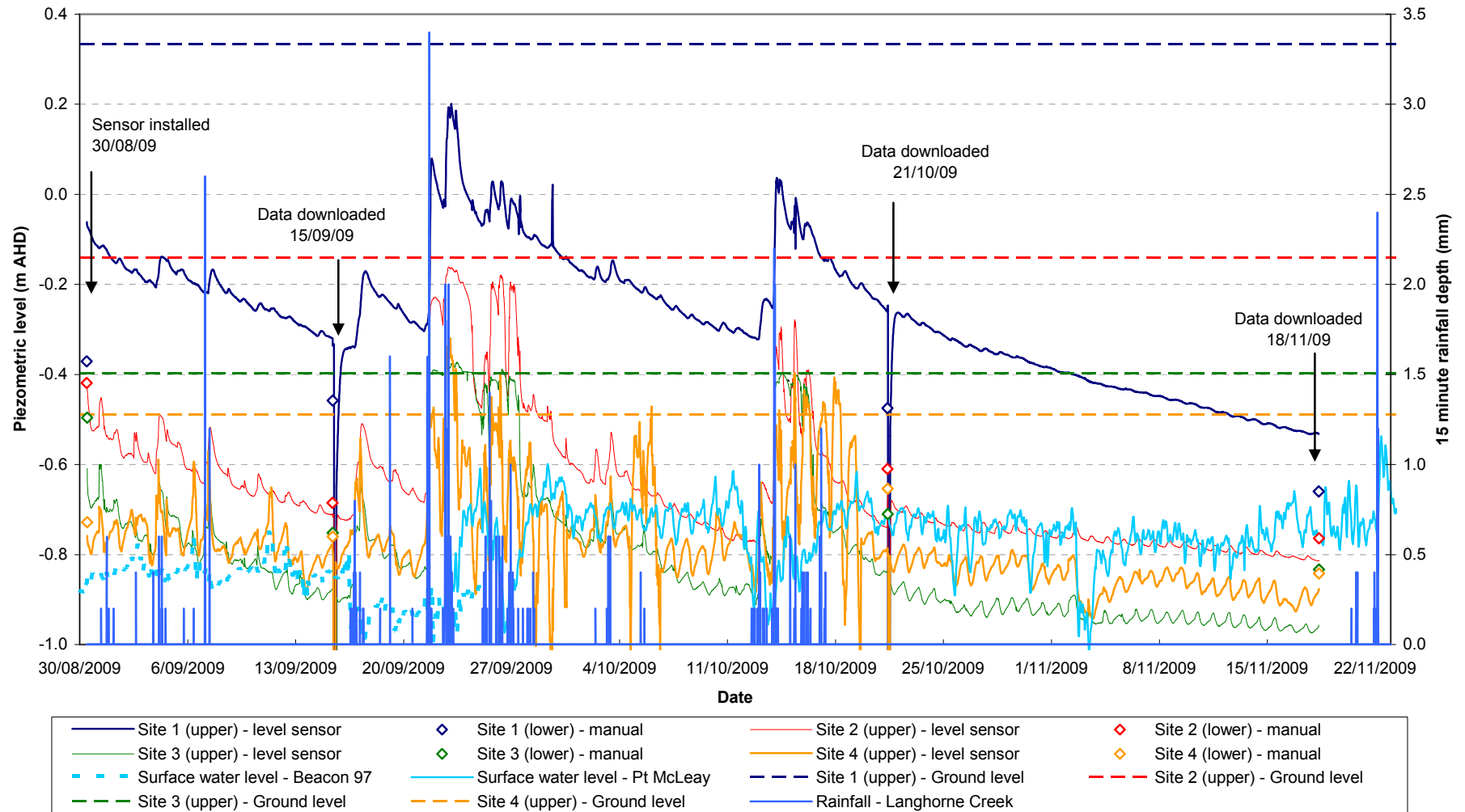


Figure 41. Temporal variation in piezometric levels at Point Sturt from 30 August to 18 November 2009.

Campbell Park rainfall and piezometric levels

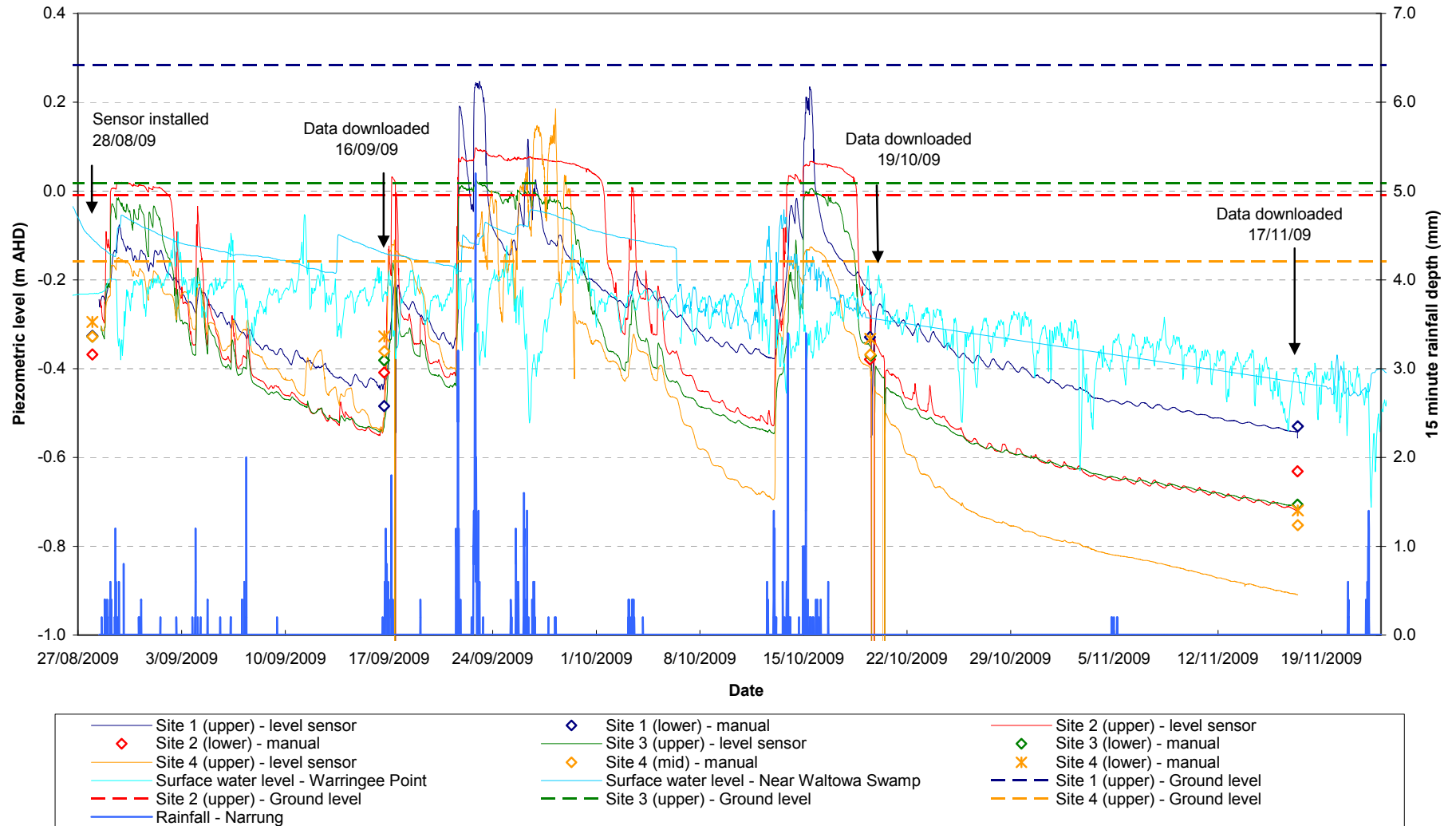


Figure 42. Temporal variation in piezometric levels at Campbell Park from 28 August to 17 November 2009.

Windmill rainfall and piezometric levels

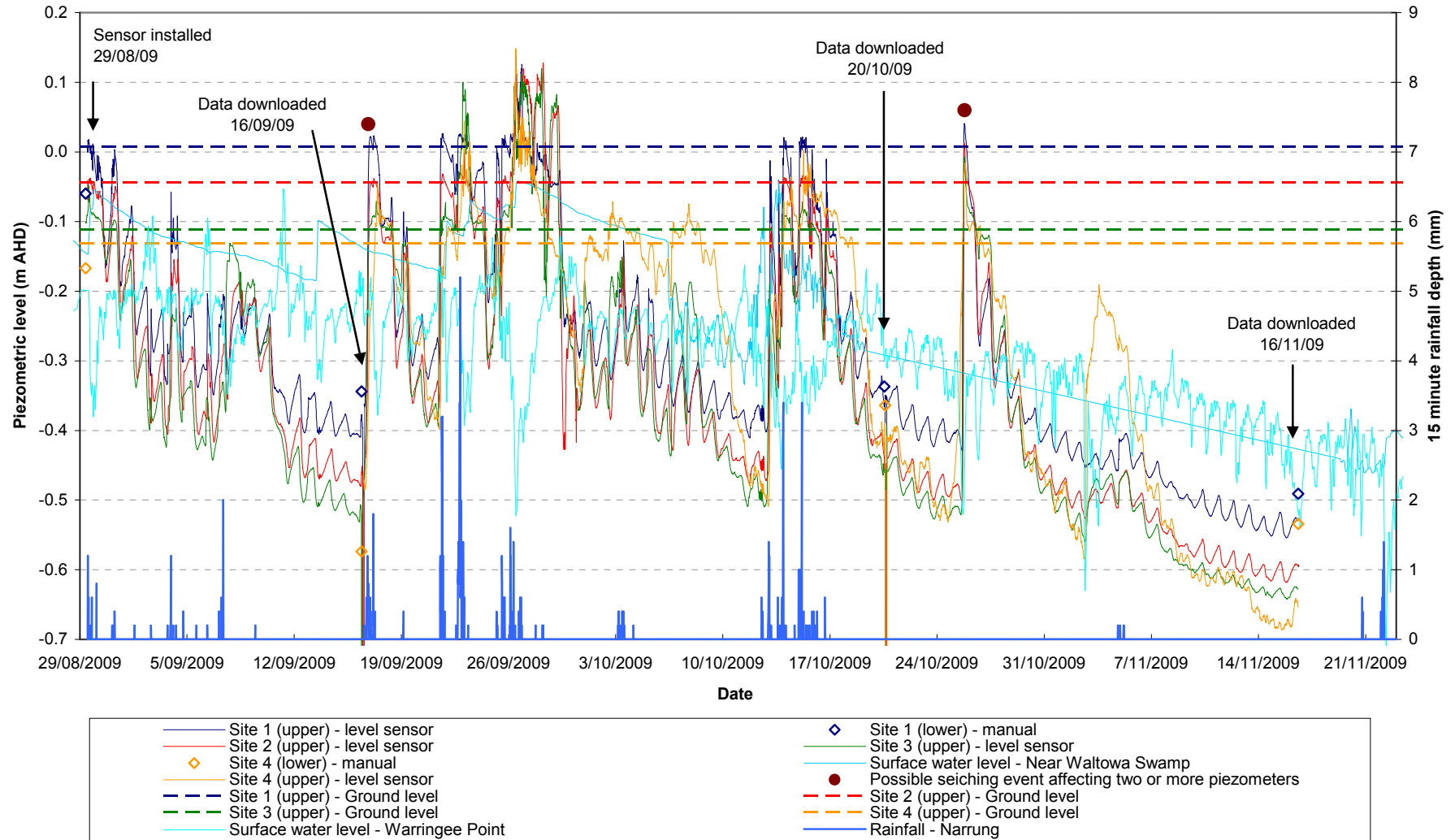


Figure 43. Temporal variation in piezometric levels at Windmill location from 29 August to 16 November 2009.

5.2.7 Groundwater quality

Currency Creek

Bulk water quality results for the Currency Creek piezometers are summarised in Table 21 and temporal trends in pH, EC and ORP, are shown in Figures 44-46. Refer to Attachment M for laboratory data. Graphs showing variations in pH, EC and ORP with depth at each location are provided in Attachment N. The key results are summarised below:

- Groundwater quality in UCC-P1 is characterised by near-neutral to slightly acidic pH, decreasing from pH 6.9 to 6.1 (marginally below the trigger value of 6.5) from early May to mid-November 2009. EC peaked at 15 mS/cm in May but dropped back to 7 mS/cm by mid-November 2009. ORP values indicate the groundwater was moderately to strongly reducing throughout the monitoring period. However, the rise in ORP from around -500 mV in June to around -50 mV in November suggests that progressive oxidation has been occurring, which is consistent with the observed pH decline. Significant alkalinity was present in UCC-P1 (only), although field/laboratory data indicate a decreasing trend in alkalinity, from 420 mg/L CaCO₃ to less than 50 mg/L CaCO₃ from mid-July to mid-November 2009. This alkalinity trend was consistent with the declining pH observed at this site.
- The pH in LCC-P2 decreased from 5.3 in early May 2009 to 2.5-3.5 (well below the trigger value of 6.5) between mid-May and mid-November 2009. The lowest pH value of 2.5 corresponded to an acidity of 1800 mg/L CaCO₃ equivalent. EC was in the range 5-10 mS/cm. ORP values were generally around 300-400 mV, indicative of oxidising conditions throughout the monitoring period, and consistent with the low pH values observed. The groundwater in LCC-P2 had an acidity of around 1300 mg/L CaCO₃ equivalent from May to mid-September 2009, decreasing to around 750 mg/L CaCO₃ equivalent by mid-November 2009. A significant chemical gradient clearly remains between groundwater at LCC-P2 and overlying surface water at CC-DS3, which had an alkalinity of 100-150 mg/L CaCO₃ equivalent in mid-November 2009. This indicates that acidic groundwater is unlikely to rapidly affect the creek water via diffusional exchange.
- The groundwater at UCC-P3 (in between UCC-P1 and LCC-P2) was slightly acidic with a pH around 5-6 from mid-May to mid-September 2009 (below the trigger value of 6.5), falling to around 4.5 in October-November 2009. The lowest pH value of 4.3 corresponded to an acidity of 165 mg/L CaCO₃ equivalent. Relatively high EC values (up to 23 mS/cm) were observed in this piezometer. ORP values indicate increasingly oxidised conditions over time, consistent with the observed decrease in pH.
- The key contributors to high salinities in all piezometers, in order of significance, are chloride (800-6300 mg/L), sulfate (750-5700 mg/L), sodium (700-4000 mg/L), magnesium (120-1100 mg/L), calcium (140-660 mg/L) and potassium (50-200 mg/L).
- Dissolved concentrations of Al, As, Cd, Cu, Mn, Ni, Pb and Zn exceeded trigger values for 95% protection of aquatic ecosystems on one or more occasions in all piezometers.
- The highest dissolved metal concentrations were observed in LCC-P2, consistent with low pH and high acidity values. Dissolved Al, Cd, Cu, Mn, Ni, Pb and Zn, exceeded trigger values for both 80% and 95% protection of aquatic ecosystems on one or more occasions in LCC-P2 and UCC-P3. Dissolved Fe and Al were elevated in LCC-P2 (up to 300 mg/L Fe and 110 mg/L Al) and UCC-P3 (up to 120 mg/L Fe and 14 mg/L Al).
- Dissolved Al, Cu, Ni, Pb and Zn exceeded trigger values for both 80% and 95% protection of aquatic ecosystems on one or more occasions in UCC-P1 (dissolved As and Cd also exceeded the 95% trigger value).
- The elevated Ca, Mg and Mn concentrations observed in LCC-P2 (average 315 mg/L Ca, 240 mg/L Mg and 3.7 mg/L Mn) and UCC-P3 (average 525 mg/L Ca, 765 mg/L Mg and 6.1 mg/L Mn) indicate that some degree of in-situ carbonate dissolution (ANC consumption) has occurred in response to acidity generation at these sites. The higher Ca, Mg and Mn concentrations at UCC-P3 relative to LCC-P2 are consistent with relatively high ANC values at UCC-P3.
- Nutrient concentrations (total N, NO_x and total P) exceeded trigger values in all piezometers on one or more occasions.

Plots showing the variation in groundwater quality with depth at UCC-P1, LCC-P2 and UCC-P3 (prior to purging) for each monitoring event are provided in Attachment N. These plots indicate that:

- Overall, few variations in groundwater quality with depth were observed at all Currency Creek piezometers on all monitoring events.
- Significant pH variations with depth were observed on 22 October 2009 at UCC-P1 (ranging from 3.5 at 3.05 m below ground level to 6.8 at 1.90 m below ground level); at LCC-P2 (ranging from 2.2 at 0.30 m

below ground level to 3.6 at 1.80 m below ground level); and at UCC-P3 (ranging from 3.7 at 2.13 m below ground level to 1.9 at 1.63 m below ground level).

- pH was generally lower and ORP generally higher when measured in piezometers prior to purging and bulk sampling. This is due to (localised) oxidising conditions within the piezometer. As such the groundwater quality measured in this manner should only be considered indicative of the surrounding formation.
- At all piezometers and for all monitoring events, EC generally increased with depth. This was most noticeable at UCC-P1 for all monitoring events with EC ranging from 3.79 mS/cm at 0.5 m below ground level to 16.60 mS/cm at 3.05 m below ground level on 2 May 2009 and 5.05 mS/cm at 0.55 m below ground level to 14.10 mS/cm at 2.25 m below ground level on 19 August 2009.

However, the above results need to be considered in the context that groundwater quality profiles were obtained prior to purging of the piezometers. Thus, the plots are not necessarily representative of actual trends in groundwater quality with depth in the surrounding sediments.

Preliminary low-flow discrete interval sampling was carried out at LCC-P2 on 22 October 2009 to assess the reliability of the pH, EC and ORP data obtained without purging, and the bulk sample pH, EC and ORP data obtained after purging of the piezometer.

Results from low-flow discrete interval sampling at LCC-P2 are provided in Figure 47. Stabilised water quality parameters (pH, EC and ORP) are plotted at 0.2 m depth intervals from 0.25 to 1.05 m below ground level (sandy horizon, see LCC-P2 construction log in Attachment F). Water quality parameters (pH, EC and ORP) representative of the bulk sample collected after purging and piezometric level recovery are also plotted for comparison.

The field data obtained within the upper sandy horizon prior to purging LCC-P2 on 22 October 2009 indicated pH values of 2.2-2.3, EC of 9.1 mS/cm and highly oxidised water (ORP +470 mV). In comparison, the stabilised low-flow samples had significantly higher pH (ranging from 2.8 to 3.0), lower EC (ranging from 7.84-8.43 mS/cm) and were less oxidised (+350-400 mV).

The bulk sample pH (2.7) and EC (7.21 mS/cm) were more consistent with stabilised low-flow sample pH (ranging from 2.8 to 3.0) and EC (ranging from 7.84-8.43 mS/cm). Similarly, ORP values for the bulk sample and stabilised low-flow samples were comparable. This suggests that bulk sample water chemistry is largely representative of the water in the upper sandy horizon (as expected).

From Figure 47, it is evident that little stratification exists within the upper sandy horizon, despite alkaline surface water being present at the time of sampling. This suggests that little diffusional exchange at the sediment-water interface is occurring at a slow rate.

Table 21. Groundwater quality data for Currency Creek piezometers.

Parameter	Unit	Date	Trigger values*		UCC-P1	LCC-P2	UCC-P3
			95%	80%			
General parameters							
pH (field)	-	2-May	6.5-9.0	6.5-9.0	6.93	5.33	n/a
		19-Aug			6.48	2.50	5.15
		14-Sep			6.39	3.06	5.26
		22-Oct			6.62	2.70	4.26
		18-Nov			6.13	2.72	4.57
pH (lab)	-	2-May	6.5-9.0	6.5-9.0	6.61	3.07	7.05
		19-Aug			6.7	2.91	5.60
		14-Sep			6.88	2.99	3.18
		22-Oct			6.35	2.95	3.76
		18-Nov			5.86	3.14	3.47
EC (field)	mS/cm	2-May	0.3-1	0.3-1	3.79	7.03	n/a
		19-Aug			10.48	9.49	21.02
		14-Sep			10.82	8.73	22.51
		22-Oct			6.10	7.21	15.71
		18-Nov			7.07	7.00	17.95
EC (lab)	mS/cm	2-May	0.3-1	0.3-1	11.1	8.84	23.3
		19-Aug			10.8	9.50	21.9
		14-Sep			11.7	23.9	9.00
		22-Oct			6.50	7.50	17.80
		18-Nov			6.26	6.68	18.70
ORP (field)	mV	2-May	n/a	n/a	74	109	n/a
		19-Aug			-40	359	145
		14-Sep			-19	172	332
		22-Oct			-150	382	300
		18-Nov			-52	318	139
DO (field)	mg/L	2-May	n/a	n/a	n/a	n/a	n/a
		19-Aug			3.34	5.22	6.02
		14-Sep			4.33	10.27	7.13
		22-Oct			3.65	3.16	4.78
		18-Nov			2.87	2.42	5.53
Alkalinity / acidity							
Alkalinity (field)	mg/L CaCO ₃	2-May	n/a	n/a	n/a	n/a	n/a
		19-Aug			n/a	n/a	n/a
		14-Sep			200	n/a	n/a
		22-Oct			51	n/a	n/a
		18-Nov			246	n/a	n/a
Alkalinity (lab)	mg/L CaCO ₃	2-May	n/a	n/a	n/a	n/a	n/a
		19-Aug			165	n/a	n/a
		14-Sep			297	<1	<1
		22-Oct			48	<1	<1
		18-Nov			52	<1	<1
Acidity (field)	mg/L CaCO ₃	2-May	n/a	n/a	n/a	n/a	n/a
		19-Aug			65	1790	275

Parameter	Unit	Date	Trigger values*		UCC-P1	LCC-P2	UCC-P3
			95%	80%			
		14-Sep			n/a	1380	395
		22-Oct			30	800	200
		18-Nov			n/a	1220	500
Acidity (lab)	mg/L CaCO ₃	2-May	n/a	n/a	50	1470	55
		19-Aug			10	1800	176
		14-Sep			n/a	1160	n/a
		22-Oct			27	912	165
		18-Nov			81	746	366
Acidity (calculated)	mg/L CaCO ₃	2-May	n/a	n/a	54	1341	3
		19-Aug			12	1315	227
		14-Sep			19	1225	451
		22-Oct			67	708	161
		18-Nov			102	819	400
Major ions							
Ca	mg/L	2-May	n/a	n/a	292	374	263
		19-Aug			265	412	606
		14-Sep			252	415	664
		22-Oct			136	213	567
		18-Nov			146	163	528
Mg	mg/L	2-May	n/a	n/a	248	271	503
		19-Aug			236	302	928
		14-Sep			276	332	1100
		22-Oct			122	154	519
		18-Nov			161	137	772
Na	mg/L	2-May	n/a	n/a	1830	740	3780
		19-Aug			1600	774	3600
		14-Sep			2180	809	3970
		22-Oct			909	784	2530
		18-Nov			1100	981	3460
K	mg/L	2-May	n/a	n/a	99	73	151
		19-Aug			111	48	162
		14-Sep			118	51	205
		22-Oct			46	30	117
		18-Nov			50	38	150
Cl	mg/L	2-May	n/a	n/a	2490	1380	5290
		19-Aug			2010	805	5130
		14-Sep			3410	1300	6320
		22-Oct			1270	851	3360
		18-Nov			1740	1560	5070
SO ₄	mg/L	2-May	n/a	n/a	1840	3240	2500
		19-Aug			1900	3760	5570
		14-Sep			1830	3660	5670
		22-Oct			750	1910	3550
		18-Nov			765	1350	4110
Cl:SO ₄ ratio	-	2-May	n/a	n/a	1.4	0.4	2.1
		19-Aug			1.1	0.2	0.9

Parameter	Unit	Date	Trigger values*		UCC-P1	LCC-P2	UCC-P3
			95%	80%			
		14-Sep			1.9	0.4	1.1
		22-Oct			1.7	0.4	0.9
		18-Nov			2.3	1.2	1.2
<i>Dissolved metals</i>							
Al	mg/L	2-May	0.055	0.15	1.87	111	0.01
		19-Aug			0.37	84.1	1.26
		14-Sep			1.45	69	14.4
		22-Oct			7.34	41.4	11.3
		18-Nov			0.2	15.8	11.8
As	mg/L	2-May	0.013 (AsV)	0.140 (AsV)	0.007	0.024	0.003
		19-Aug			0.007	0.019	0.007
		14-Sep			0.005	0.02	0.009
		22-Oct			0.004	0.013	0.004
		18-Nov			0.023	0.028	0.014
Cd	mg/L	2-May	0.0002	0.0008	n/a	n/a	n/a
		19-Aug			0.0003	0.0027	0.0006
		14-Sep			<0.0001	0.0016	0.0009
		22-Oct			0.0004	0.0012	0.0026
		18-Nov			0.0005	0.0016	0.0020
Cu	mg/L	2-May	0.0014	0.0025	0.011	0.101	0.008
		19-Aug			0.005	0.075	0.001
		14-Sep			0.005	0.066	0.016
		22-Oct			0.013	0.095	0.016
		18-Nov			0.003	0.033	0.013
Fe	mg/L	2-May	n/a	n/a	15.8	249	0.1
		19-Aug			3.35	288	77.6
		14-Sep			3.87	290	120
		22-Oct			9.51	139	30.9
		18-Nov			36.8	235	118
Mn	mg/L	2-May	1.9	3.6	0.33	5.58	1.48
		19-Aug			0.26	5.08	5.92
		14-Sep			0.25	4.77	8.42
		22-Oct			0.51	1.69	6.48
		18-Nov			0.90	1.29	8.20
Ni	mg/L	2-May	0.011	0.017	n/a	n/a	n/a
		19-Aug			0.016	0.494	0.033
		14-Sep			0.012	0.368	0.071
		22-Oct			0.021	0.23	0.117
		18-Nov			0.011	0.134	0.136
Pb	mg/L	2-May	0.0034	0.0094	0.177	0.142	0.001
		19-Aug			0.003	0.011	0.005
		14-Sep			0.003	0.023	0.021
		22-Oct			0.015	0.015	0.015
		18-Nov			0.001	0.009	0.007
Se	mg/L	2-May	0.011	0.034	n/a	n/a	n/a
		19-Aug			<0.01	0.01	<0.01

Parameter	Unit	Date	Trigger values*		UCC-P1	LCC-P2	UCC-P3
			95%	80%			
		14-Sep			<0.01	0.01	<0.01
		22-Oct			<0.01	<0.01	<0.01
		18-Nov			<0.01	<0.01	<0.01
Zn	mg/L	2-May	0.008	0.031	0.078	0.857	0.005
		19-Aug			0.037	0.562	0.045
		14-Sep			0.01	0.541	0.038
		22-Oct			0.039	0.337	0.127
		18-Nov			0.115	0.507	0.144
Nutrients							
NO ₂ + NO ₃	mg/L N	2-May	0.1	0.1	1.87	0.07	0.19
		19-Aug			2.82	0.33	0.15
		14-Sep			1.46	0.17	0.35
		22-Oct			1.85	0.23	3.54
		18-Nov			0.79	0.10	0.31
TKN	mg/L N	2-May	n/a	n/a	4.2	0.8	8.3
		19-Aug			0.8	3	7.2
		14-Sep			1.4	3.9	12.1
		22-Oct			2	7.1	26.8
		18-Nov			7.3	7.8	23.6
Total N	mg/L N	2-May	1	1	6.1	0.9	8.5
		19-Aug			3.7	3.3	7.4
		14-Sep			2.8	4	12.5
		22-Oct			3.8	7.3	30.4
		18-Nov			8.1	7.9	23.9
Total P	mg/L P	2-May	0.025	0.025	0.48	1.34	0.64
		19-Aug			0.19	1.42	0.05
		14-Sep			0.38	0.90	0.22
		22-Oct			0.09	0.37	<0.01
		18-Nov			3.54	1.17	0.18

* ANZECC/ARMCANZ (2000) trigger values for 80% and 95% protection of freshwater ecosystems. Values exceeding ANZECC/ARMCANZ (2000) trigger values for **95%** protection of freshwater ecosystems are shaded in orange. Values exceeding trigger values for **95%** and **80%** protection of freshwater ecosystems are shaded in red.

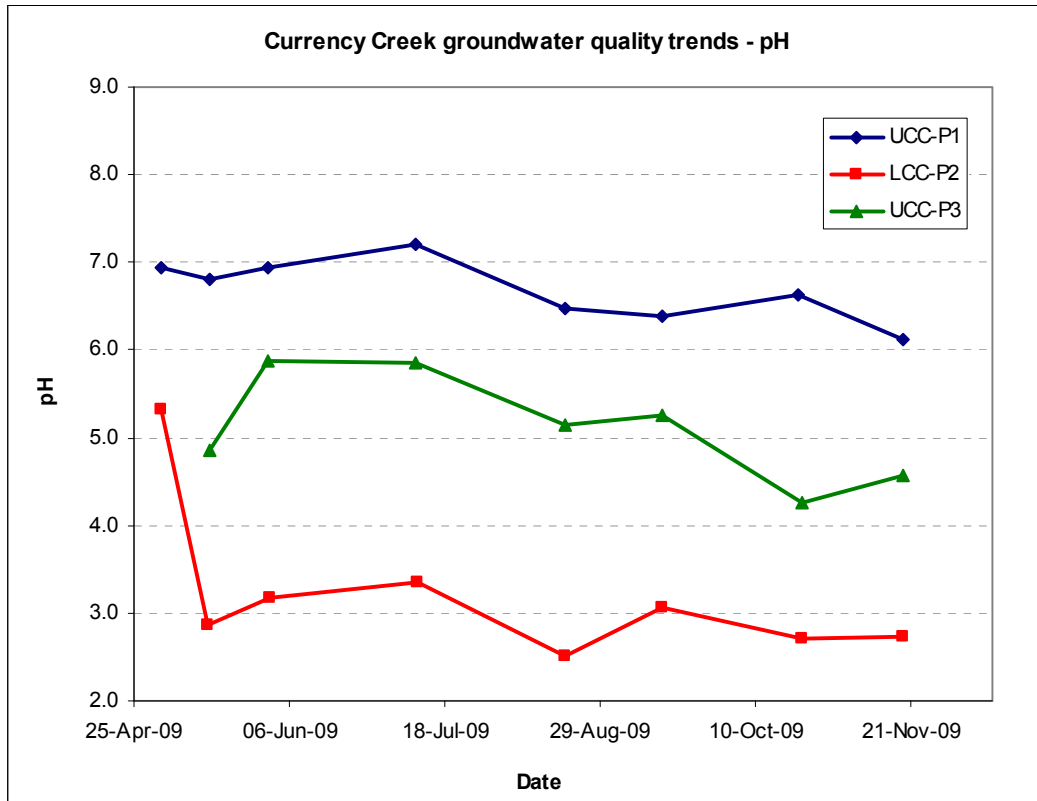


Figure 44. Groundwater pH at Currency Creek (after purging), 2 May - 18 November 2009.

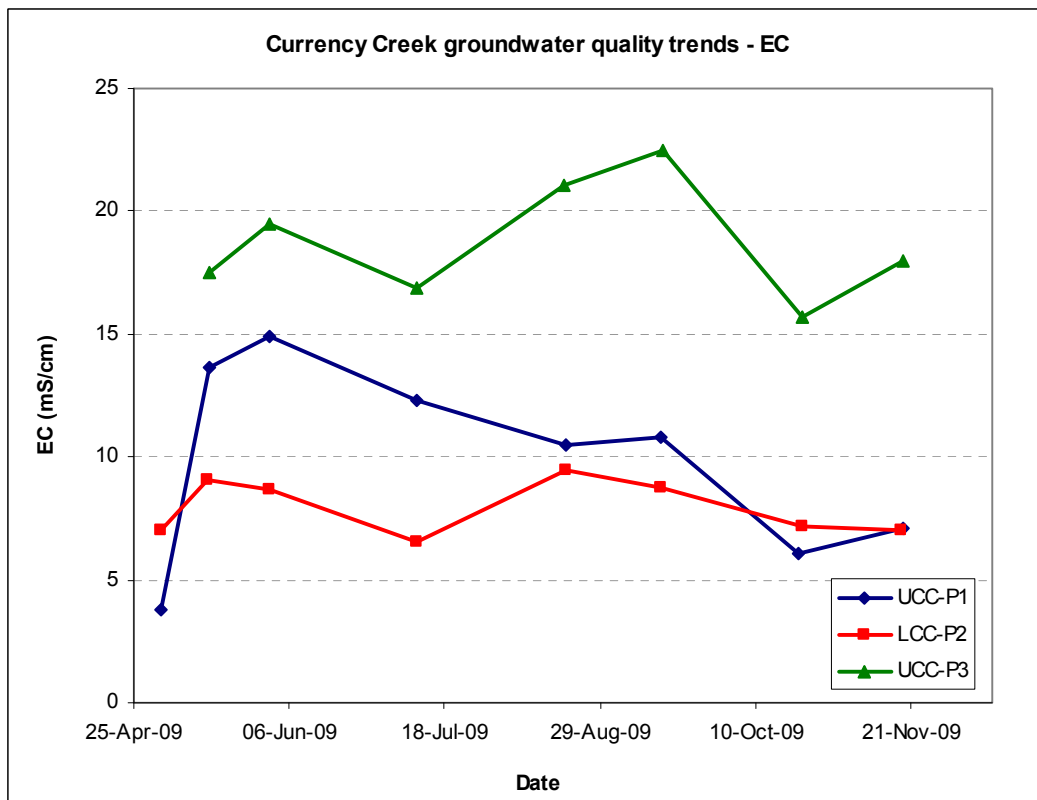


Figure 45. Groundwater EC at Currency Creek (after purging), 2 May - 18 November.

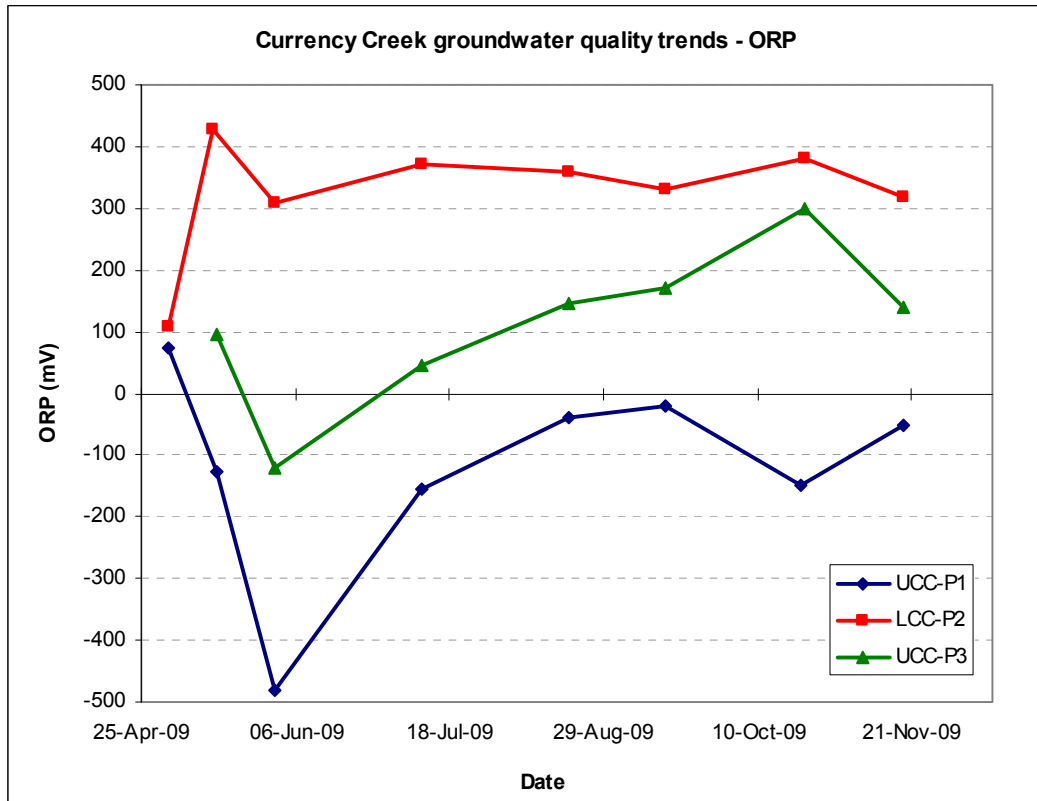


Figure 46. Groundwater ORP at Currency Creek (after purging), 2 May – 18 November 2009.

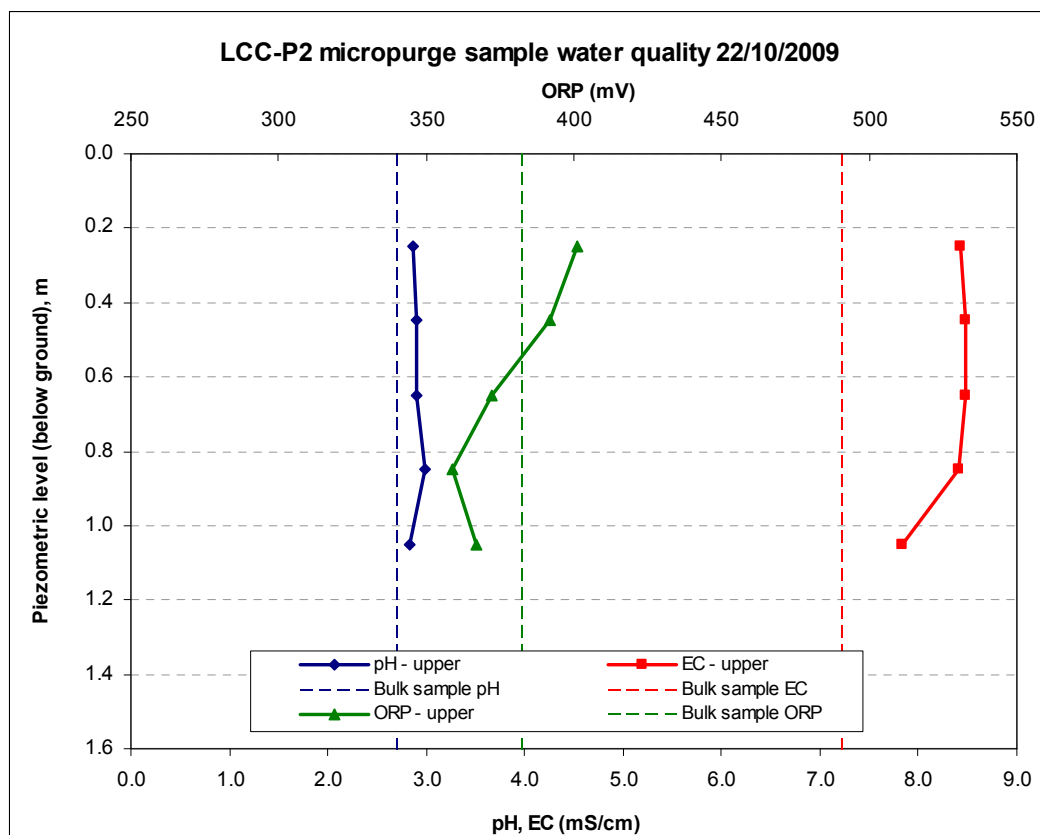


Figure 47. Water quality after low-flow sampling at Lower Currency Creek (LCC-P2) on 22 October 2009. Solid lines depict stabilised water quality parameters pH, EC and ORP for low-flow samples collected in 0.2 m depth intervals from 0.25 to 1.05 m below ground level. Dashed lines indicate the corresponding parameter value from the bulk sample collected after completion of low-flow sampling and after purging and recovery of the piezometer.

Point Sturt, Campbell Park and Windmill locations

Bulk water quality results for the Point Sturt, Campbell Park and Windmill piezometers are summarised in Tables 22-24. Trends in pH, EC, ORP, alkalinity/acidity and Cl:SO₄ ratio, with distance along each transect, are shown in Figures 48-67. Refer to Attachment M for laboratory data. Graphs showing variations in pH, EC and ORP with depth at each location are provided in Attachment N. Graphs showing temporal trends in these parameters at each location are also provided in Attachment O. The key results are summarised below:

- In the upper sediments at Point Sturt, groundwater was acidic at Site 2 (pH 4-5; acidity 300-700 mg/L CaCO₃ equivalent) from late August to mid-November 2009. Slightly acidic pH was measured at the original shore (Site 1) in late August 2009 (pH 6.0; acidity 50 mg/L CaCO₃ equivalent). All other sites at Point Sturt (upper and lower sediments) were characterised by near-neutral pH with alkalinities in the range 200-700 mg/L CaCO₃ equivalent. Relatively oxidised groundwater (ORP up to +250 mV) in the upper sediments at Point Sturt (Sites 1 and 2) corresponded to the lowest pH values. The pH did not vary significantly at any site on the Point Sturt transect over the 3 month monitoring period.
- EC values at Point Sturt ranged from 4.2 mS/cm to 17.7 mS/cm, with the lowest salinities associated with the upper sediments (<10 mS/cm). The higher salinities at depth are dominated by Na and Cl, rather than sulfate, indicating they are not related to acidity generation. No clear trends in EC were observed over the 3 month monitoring period, with the exception of the deeper sediments of Site 3, which increased from 12.83 mS/cm in late August to 17.67 mS/cm in mid-November 2009.
- The relatively minor variation in pH and EC observed in all Point Sturt piezometers over the 3 month monitoring period suggests that:
 - Some localised acidity generation occurred in the more exposed near shore sediments (Sites 1 and 2) prior to commencement of the monitoring program;
 - No significant additional acidity generation occurred at Sites 1 and 2 from late August to mid-November 2009;
 - Localised acidity generation in the near shore sediments (Sites 1 and 2) was limited to the upper sandy horizon;
 - The acidity has been transported vertically from sandy layers in the unsaturated zone to the groundwater via rainwater infiltration;
 - There has been no vertical transport of acidity from the upper sandy layer (aquifer) to the lower aquifer, indicating that the aquifers are hydraulically disconnected by the intervening clay layer and/or diffusional mixing of groundwater within the sediments is limited; and
 - Where acidity generation has occurred, groundwater flow has been insufficient to transport the acidity from one site to the next (75 metres). This is attributed to the relatively small hydraulic gradients measured at Point Sturt between late August and mid-November 2009.
- The key contributors to high salinities at Point Sturt, in order of significance, are chloride (250-5300 mg/L), sulfate (150-3800 mg/L), sodium (500-3000 mg/L), magnesium (50-490 mg/L), calcium (40-350 mg/L) and potassium (30-110 mg/L). Sulfate concentrations in the upper piezometers exceeded those in the underlying sediments at all sites. This was particularly evident at Site 2 (upper sediments 3200-3800 mg/L SO₄, lower sediments 300-600 mg/L SO₄) and Site 1 (upper sediments 2360-2650 mg/L SO₄, lower sediments 690-850 mg/L SO₄). The Cl:SO₄ ratio was also lowest in the upper sediments at Site 1 (0.1) and Site 2 (0.3).
- Dissolved concentrations of Al, As, Cd, Cu, Mn, Ni, Pb and Zn exceeded trigger values for 95% protection of aquatic ecosystems on one or more occasions in most piezometers at Point Sturt. These metals, excluding As and Cd, also exceeded the 80% trigger value.
- Dissolved Fe was generally elevated in all piezometers at Point Sturt, particularly in the upper sandy sediments (up to 107 mg/L at Site 2).
- Dissolved Mn was also considerably higher in the upper sandy sediments (up to 25 mg/L at Site 2) at Point Sturt.
- Unlike Fe and Mn, dissolved As was generally higher in the lower sandy sediments (up to 0.144 mg/L at Site 2) at Point Sturt.
- The elevated Ca, Mg and Mn concentrations observed in the upper sandy horizon at Point Sturt (Site 1 - average 207 mg/L Ca, 210 mg/L Mg and 6.1 mg/L Mn; Site 2 - average 296 mg/L Ca, 446 mg/L Mg and 25.4 mg/L Mn) indicate that some degree of in-situ carbonate dissolution (ANC consumption) has occurred in response to acidity generation at these sites. Much of this ANC consumption process appears to have occurred prior to monitoring, as indicated by the relatively stable Ca, Mg and Mn concentrations throughout the 3 month monitoring period.

- There is no clear evidence of sulfide precipitation (bacterial sulfate reduction) within the upper sandy sediments affected by acidity generation at Point Sturt (Site 1 and 2), based on the consistent Cl:SO₄ ratios observed throughout the 3 month monitoring period. At Site 2, the process of sulfate reduction is likely to have been inhibited by the low pH values observed.
- In the upper sandy sediments at Campbell Park, groundwater was acidic at Sites 2-4 (pH 2.4-5.4; acidity 300-1600 mg/L CaCO₃ equivalent) from late August to mid-November 2009. There was some improvement at these sites over the 3 month monitoring period, with pH increasing from 2.4 to 4.7 at Site 2, from 4.3 to 5.4 at Site 3, and from 3.0 to 4.2 at Site 4. Relatively oxidising groundwater (ORP up to 400 mV) in the upper sediments at Campbell Park (Sites 2-4) corresponded to the lowest pH values. The deeper sandy sediments at all Campbell Park sites (and upper sediments at Site 1) were characterised by near-neutral pH with alkalinities generally in the range 500-800 mg/L CaCO₃ equivalent.
- EC values at Campbell Park ranged from 8.9 mS/cm to 48.3 mS/cm. The highest salinities are associated with the deeper sediments (18.6-48.3 mS/cm), particularly those nearest the shore at Sites 1 and 2 (>40 mS/cm). The higher salinities at depth are dominated by Na and Cl, rather than sulfate, indicating they are not related to acidity generation. EC values in the upper and lower sandy layers at Campbell Park were highly variable (spatially) relative to the other locations and there were no consistent trends over time within each piezometer site.
- The trends in pH and EC observed at Campbell Park over the 3 month monitoring period suggest that:
 - Acidity generation occurred within the upper sandy horizon sediments (with the exception of Site 1, near the shore) prior to commencement of the monitoring program;
 - No significant additional acidity generation occurred in the upper sediments from late August to mid-November 2009;
 - The acidity has been transported vertically from sandy layers in the unsaturated zone to the groundwater table via rainwater infiltration;
 - There has been no vertical transport of acidity from the upper sandy layer (aquifer) to the lower sandy aquifer, indicating that the aquifers are effectively hydraulically disconnected by the intervening clay layer; and
 - Where acidity generation has occurred, groundwater flow has been insufficient to transport the acidity from one (piezometer) site to the next (50 metres), as indicated by the discrepancy in pH values between sites. This is attributed to the relatively small hydraulic gradients measured at Campbell Park between late August and mid-November 2009.
- The key contributors to high salinities at Campbell Park, in order of significance, are chloride (2300-18900 mg/L), sodium (1800-11000 mg/L), sulfate (500-6400 mg/L) and magnesium (300-1300 mg/L), calcium (230-880 mg/L) and potassium (70-370 mg/L). Sulfate concentrations in the upper piezometers at Sites 2-4 exceeded those in the underlying sediments at all sites. The Cl:SO₄ ratio was also lower in the upper sediments at Site 2-4 (0.5-1.7) than the lower sediments (4.5-10.6).
- Dissolved concentrations of Al, As, Cd, Cu, Mn, Ni, Pb and Zn exceeded trigger values for 95% protection of aquatic ecosystems on one or more occasions in most piezometers at Campbell Park.
- Dissolved Fe was generally elevated in all piezometers at Campbell Park, particularly in the upper sediments (up to 287 mg/L at Site 2). This is consistent with observations at Point Sturt.
- Dissolved Mn was considerably higher in the upper sandy sediments (up to 16 mg/L at Site 4) at Campbell Park. This is consistent with observations at Point Sturt.
- Other dissolved metals that were generally higher in the upper sandy sediments at Campbell Park included Cd (up to 0.006 mg/L), Cu (up to 0.172 mg/L), Ni (1.48 mg/L), Pb (up to 0.186 mg/L) and Zn (up to 0.678 mg/L).
- Unlike most other metals, dissolved As was generally higher in the deeper sandy sediments at Campbell Park. This is consistent with observations at Point Sturt, and is presumably related to the more reducing conditions in these sediments.
- The elevated Ca, Mg and Mn concentrations observed in the upper sandy horizon at Campbell Park nearest the lake water (Site 4 - average 560 mg/L Ca, 800 mg/L Mg and 13.4 mg/L Mn) indicates that some degree of localised in-situ carbonate dissolution (ANC consumption) has occurred in response to acidity generation at this site. Much of this ANC consumption process appears to have occurred prior to monitoring, as indicated by the relatively stable Ca, Mg and Mn concentrations throughout the 3 month monitoring period.
- There is some evidence of sulfide precipitation (bacterial sulfate reduction) within the upper sandy sediments affected by acidity generation at Campbell Park (Sites 2-4), based on the progressive increase in pH and Cl:SO₄ ratios observed throughout the 3 month monitoring period.

- In contrast to Point Sturt and Campbell Park, groundwater at the Windmill location was near-neutral to slightly alkaline in all piezometers (pH 6.2-7.9; alkalinity 390-1050 mg/L CaCO₃). Relatively reduced groundwater (ORP ranging from -55 to -160 mV) was observed in all Windmill piezometers, consistent with near-neutral to slightly alkaline pH values. The maintenance of near-neutral pH over time indicates that no significant acidity generation occurred at the Windmill location from late August to mid-November 2009, or prior to the commencement of monitoring.
- EC values at the Windmill location ranged from 13.1 mS/cm to 31.7 mS/cm, with the highest salinities (26.3-31.7 mS/cm) associated with the deeper sandy sediments at Sites 1 and 4, and the upper sandy sediments at Sites 1-2. The higher salinities are dominated by Na and Cl, rather than sulfate, indicating they are not related to acidity generation. The variations in EC with depth and distance along the transect indicate that:
 - There is some hydraulic connectivity between the upper and lower sediments nearest the original shore (Site 1), consistent with the lack of significant clay barrier between the sandy lake sediments and underlying sands of the Bridgewater Formation at this site;
 - In the upper sandy sediments, the EC ranged from around 30 mS/cm nearest the original shore (Sites 1 and 2) to around 15 mS/cm nearest the lake water (Site 4);
 - The significant contrast in EC in between the sandy lake sediments and underlying sands of the Bridgewater Formation nearest the lake water (Site 4) is attributed to hydraulic disconnection associated with a clay horizon;
 - While some dilution of groundwater by relatively lower salinity lake water may be responsible for the salinity contrast in the upper sand sediments between sites 1 and 4, the significant salinity gradient has been maintained throughout the 3 month monitoring period, indicating that groundwater flow has been insufficient to transport the saline water from one site to the next (50 metres). This is attributed to the relatively small hydraulic gradients measured at the Windmill location between late August and mid-November 2009.
- The key contributors to high salinities at the Windmill location, in order of significance, are chloride (3000-11900 mg/L), sodium (1450-6350 mg/L), sulfate (90-1850 mg/L), magnesium (440-1120 mg/L), calcium (310-800 mg/L) and potassium (50-175 mg/L). Sulfate concentrations in the upper piezometer exceeded those in the underlying sediments at Site 1, although the reverse was observed at Site 4.
- Dissolved concentrations of Al, As, Cd, Cu, Mn, Ni, Pb and Zn exceeded trigger values for 95% protection of aquatic ecosystems on one or more occasions in most piezometers at the Windmill location. These metals, excluding As and Cd, also exceeded the 80% trigger value.
- Dissolved Fe was generally elevated in all piezometers at the Windmill location. This is consistent with observations at Point Sturt.
- Dissolved Mn was considerably higher in the upper sandy sediments nearest the lake water (up to 6.3 mg/L at Site 4) at the Windmill location. This is consistent with observations at Point Sturt and Campbell Park.
- There is evidence that sulfide precipitation (bacterial sulfate reduction) has occurred within the upper sandy sediments nearest the lake water at the Windmill location (Sites 3-4) prior to the commencement of monitoring, based on the high Cl:SO₄ ratios observed in late August 2009. The Cl:SO₄ ratios at these sites have subsequently decreased over time throughout the 3 month monitoring period.
- Nutrient concentrations (total N and total P) exceeded trigger values in all Point Sturt, Campbell Park and Windmill piezometers on one or more occasions.

Plots of variation in groundwater quality with depth at piezometers at Point Sturt, Campbell Park and Windmill for each monitoring event are provided in Attachment N. These plots indicate that:

- Few variations in pH with depth were observed for all piezometers for all monitoring events, with the exception of CP-1S on 26 August 2009 (ranging from 7.3 at 0.320 m BGL to 3.58 at 1.07 m BGL) and PS-1S on 15 September 2009 (ranging from 7.47 at 0.75 m BGL to 4.36 at 1.30 m BGL).
- At all piezometers for all monitoring events, EC increased with depth. The highest variation in EC with depth occurred at CP-2S, CP-3S, WM-1S and WM-2S for all monitoring events.

However, the above results need to be considered in the context that groundwater quality profiles were obtained prior to purging of the piezometers. Thus, the plots are not necessarily representative of actual trends in groundwater quality with depth in the surrounding sediments.

Preliminary results for low-flow sampling undertaken at Point Sturt (PS-2S) at 0.8 m below ground level (~0.25 m below groundwater level) on 21 October 2009 are provided in Figure 68. The figure shows water quality

parameters (pH, temperature, EC and ORP) stabilising over volumetric intervals during purging. The dashed lines indicate the corresponding water quality parameter value for the bulk water sample collected after completion of low-flow sampling and after purging and recovery of the piezometric level.

From Figure 68, the stabilised pH of the low-flow sample at 0.8 m below ground level (4.1 at 16.7°C) was noticeably lower than the pH of the bulk sample after purging (4.8 at 15.7°C). Similarly, EC was slightly elevated in the low-flow sample (8.46 mS/cm) compared with the bulk sample after purging (8.03 mS/cm). This indicates that the water in the upper sand horizon is stratified. The thickness or extent of stratification is unknown. Deeper intervals were unable to be sampled due to the equipment dimensions (position of pump inlet relative to base of the pump).

When comparing the bulk sample water quality and the discrete interval sample water quality with the water quality profile measured prior to purging, pH is noticeably lower in the profile (3.6 to 3.7) than in the bulk sample (4.8). ORP is also noticeably higher in the profile (321 to 362 mV) than the bulk sample (248 mV). This is presumably explained by oxidation of the stagnant water in the piezometer.

Table 22. Groundwater quality data for Point Sturt piezometers.

Parameter	Trigger values*		Date	Site 1		Site 2		Site 3		Site 4	
	95%	80%		Upper	Lower	Upper	Lower	Upper	Lower	Upper	Lower
General parameters											
pH – field	6.5-9.0	6.5-9.0	29/08	5.97	7.44	4.42	7.40	7.32	7.35	7.16	7.31
			15/09	7.12	7.86	4.97	7.94	7.77	7.57	7.31	7.66
			21/10	7.07	7.60	4.78	7.48	7.27	7.26	6.56	7.45
			18/11	7.08	7.48	4.13	7.34	7.44	7.20	7.19	7.22
pH – laboratory	6.5-9.0	6.5-9.0	29/08	6.00	7.66	3.54	7.48	7.18	7.25	6.95	7.16
			15/09	6.80	7.64	3.18	7.50	7.42	7.37	6.98	7.31
			21/10	6.60	7.60	3.30	7.30	7.24	7.22	6.84	7.34
			18/11	6.95	7.56	3.20	7.40	7.30	7.20	6.85	7.27
EC – field (mS/cm)	0.3-1	0.3-1	29/08	5.32	11.01	7.74	7.18	4.22	12.83	6.05	5.74
			15/09	5.35	10.22	7.99	9.31	5.88	14.43	7.81	5.6
			21/10	5.3	10.6	8.03	11.15	5.186	14.69	7.77	5.67
			18/11	4.94	9.95	7.59	11.06	7.01	17.67	7.67	6.09
EC – laboratory (mS/cm)	0.3-1	0.3-1	29/08	5.65	11.4	8.3	7.76	4	13.7	6.1	6.06
			15/09	5.51	10.7	8.69	10.5	6.1	15.1	8.05	5.73
			21/10	5.54	11.80	9.30	12.60	5.75	16.20	8.26	6.00
			18/11	4.95	10.40	8.04	11.20	6.82	18.20	7.15	5.93
ORP – field (mV)	n/a	n/a	29/08	75.0	-16.0	2.18	-94.0	-118	-73.0	-94.0	-106
			15/09	19.0	-62.0	204	-82	-50.0	-70.0	-94.0	-106
			21/10	-17.0	-79.0	248	-79	-89.0	-39.0	-75.0	-100
			18/11	-40.0	-62.0	209	-83	-95.0	-72.0	-90.0	-101
DO – field (mg/L)	n/a	n/a	15/09	6.08	3.56	5.84	3.92	4.76	4.13	4.54	4.46
			21/10	5.23	3.57	4.98	3.23	-	-	5.78	5.36
			18/11	5.16	6.78	3.87	3.60	4.87	3.81	5.18	3.15
Alkalinity / acidity (units in mg/L CaCO₃)											
Alkalinity – field	n/a	n/a	29/08	-	-	-	-	-	1305	-	765
			15/09	120	630	-	1275	420	570	378	636
			21/10	165	756	-	1080	360	708	453	816
			18/11	258	708	-	2400	480	696	516	882
Alkalinity – laboratory	n/a	n/a	29/08	76	715	-	552	216	501	272	626
			15/09	126	696	<1	459	384	481	343	663
			21/10	161	714	<1	463	332	494	357	688
			18/11	224	727	<1	503	470	459	397	704
Acidity – field	n/a	n/a	29/08	-	-	290	-	-	-	-	-
			15/09	93	-	410	-	-	-	-	-
			21/10	-	-	761	-	-	-	-	-
			18/11	-	-	420	-	-	-	-	0
Acidity – laboratory	n/a	n/a	29/08	50	35	396	25	20	50	35	60
			15/09	33	-	404	-	-	-	-	-
			21/10	-	-	684	-	-	-	-	-
			18/11	-	-	309	-	-	-	-	-
Acidity – calculated	n/a	n/a	29/08	35	2	307	10	5	1	33	9
			15/09	77	105	513	211	109	255	187	367
			21/10	53	60	554	125	52	247	129	105
			18/11	9	12	340	4	3	6	57	8

Parameter	Trigger values*		Date	Site 1		Site 2		Site 3		Site 4	
	95%	80%		Upper	Lower	Upper	Lower	Upper	Lower	Upper	Lower
<i>Major ions (units in mg/L)</i>											
Ca	n/a	n/a	29/08	230	54	269	80	139	216	140	110
			15/09	235	53	305	123	125	254	166	105
			21/10	195	40	314	137	146	246	146	98
			18/11	168	47	294	158	132	347	143	114
Mg	n/a	n/a	29/08	238	55	391	101	86	262	100	114
			15/09	234	64	461	178	130	316	130	126
			21/10	184	57	446	194	114	293	116	117
			18/11	182	69	484	239	162	487	134	159
Na	n/a	n/a	29/08	713	2040	873	1240	497	2090	939	921
			15/09	788	2220	951	1810	930	2560	1260	887
			21/10	724	2220	891	2020	772	2510	1100	814
			18/11	871	2310	1040	2160	1230	2990	1040	1030
K	n/a	n/a	29/08	64	52	71	43	28	59	34	32
			15/09	65	54	82	56	34	54	32	33
			21/10	60	50	89	53	33	53	30	30
			18/11	62	52	110	58	42	65	33	36
Cl	n/a	n/a	29/08	268	2440	1080	2060	772	4220	988	877
			15/09	360	2780	1090	3050	1540	4550	2290	1440
			21/10	246	2450	850	3280	1200	4380	1770	1080
			18/11	285	2550	1080	3260	1790	5340	1750	1420
SO ₄	n/a	n/a	29/08	2600	775	3210	294	609	384	480	339
			15/09	2650	776	3660	423	486	465	400	242
			21/10	2360	693	3780	468	553	432	302	149
			18/11	2390	847	3810	603	372	628	321	214
Cl:SO ₄ ratio	n/a	n/a	29/08	0.1	3.1	0.3	7.0	1.3	11.0	2.1	2.6
			15/09	0.1	3.6	0.3	7.2	3.2	9.8	5.7	6.0
			21/10	0.1	3.5	0.2	7.0	2.2	10.1	5.9	7.2
			18/11	0.1	3.0	0.3	5.4	4.8	8.5	5.5	6.6
<i>Dissolved metals (units in mg/L)</i>											
Al	0.055	0.15	29/08	0.28	0.08	23.60	0.75	0.10	0.03	0.06	0.05
			15/09	6.10	13.70	32.00	22.20	12.50	32.30	10.80	42.50
			21/10	2.47	7.59	42.10	12.80	5.57	29.90	3.33	9.95
			18/11	0.04	1.39	16.90	0.19	0.13	0.09	0.10	0.12
As	0.013 (AsV)	0.140 (AsV)	29/08	0.001	0.046	0.004	0.020	0.004	0.007	0.009	0.030
			15/09	0.005	0.074	0.006	0.144	0.024	0.048	0.058	0.106
			21/10	0.002	0.072	0.003	0.107	0.011	0.052	0.045	0.071
			18/11	<0.001	0.067	0.003	0.042	0.006	0.029	0.040	0.044
Cd	0.0002	0.0008	29/08	0.0002	0.0004	0.0002	0.0017	<0.0001	0.0004	0.0001	0.0001
			15/09	0.0004	<0.0001	0.0003	0.0003	0.0002	0.0002	0.0002	0.0005
			21/10	0.0002	<0.0001	0.0001	0.0001	<0.0001	0.0002	<0.0001	<0.0001
			18/11	<0.0001	<0.0001	0.0004	<0.0001	0.0003	0.0001	0.0003	<0.0001
Cu	0.0014	0.0025	29/08	0.006	0.002	0.01	0.002	0.002	0.002	0.005	0.007
			15/09	0.011	0.027	0.019	0.024	0.110	0.036	0.012	0.016
			21/10	0.006	0.016	0.012	0.015	0.005	0.038	0.005	0.005
			18/11	0.006	0.006	0.008	0.001	0.001	0.002	0.001	<0.001

Parameter	Trigger values*		Date	Site 1		Site 2		Site 3		Site 4	
	95%	80%		Upper	Lower	Upper	Lower	Upper	Lower	Upper	Lower
Fe	n/a	n/a	29/08	6.17	0.13	50.20	1.39	0.32	<0.05	10.30	2.33
			15/09	10.90	10.60	107.00	31.80	13.40	27.50	43.80	47.50
			21/10	10.90	6.51	98.10	19.70	6.84	29.50	38.00	17.70
			18/11	1.30	1.27	72.70	0.90	0.19	1.69	18.20	2.03
Mn	1.9	3.6	29/08	9.03	0.287	20.9	0.838	2.10	0.458	2.76	1.29
			15/09	7.16	0.104	25.4	0.612	1.68	0.636	4.89	1.41
			21/10	5.32	0.085	29.9	0.467	1.520	0.617	4.65	1.06
			18/11	3.01	0.161	25.4	0.410	1.060	0.583	3.96	0.97
Ni	0.011	0.017	29/08	0.222	0.013	0.185	0.008	0.002	0.004	0.003	0.002
			15/09	0.136	0.019	0.165	0.031	0.010	0.042	0.011	0.048
			21/10	0.096	0.015	0.215	0.020	0.005	0.052	0.004	0.011
			18/11	0.030	0.010	0.092	0.004	0.001	0.002	0.001	0.002
Pb	0.0034	0.0094	29/08	0.002	<0.001	0.009	<0.001	<0.001	<0.001	<0.001	<0.001
			15/09	0.010	0.013	0.029	0.024	0.013	0.030	0.014	0.031
			21/10	0.005	0.008	0.015	0.015	0.010	0.032	0.005	0.013
			18/11	<0.001	0.002	0.009	<0.001	<0.001	<0.001	<0.001	<0.001
Se	0.011	0.034	29/08	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
			15/09	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
			21/10	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
			18/11	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0
Zn	0.008	0.031	29/08	0.116	0.008	0.287	0.010	<0.005	0.006	0.008	0.008
			15/09	0.084	0.033	0.219	0.035	0.034	0.05	0.035	0.049
			21/10	0.069	0.027	0.253	0.026	0.016	0.05	0.021	0.025
			18/11	0.018	0.017	0.218	0.015	0.020	0.02	0.024	0.020
Nutrients (units in mg/L)											
Nitrite + Nitrate as N	0.1	0.1	29/08	0.02	<0.01	0.04	<0.01	0.13	0.02	<0.01	<0.01
			15/09	0.02	0.02	0.05	0.02	1.04	0.03	0.29	0.02
			21/10	<0.01	0.02	0.05	0.17	1.14	0.04	0.28	0.01
			18/11	0.02	0.02	0.01	0.22	1.04	0.41	0.76	0.10
Total Kjeldahl Nitrogen as N	n/a	n/a	29/08	<0.1	1.0	2.4	2.2	4.6	7.1	17.7	14.5
			15/09	<0.1	<0.1	3.3	2.5	6.8	4.9	36.7	19.2
			21/10	3.4	0.7	2.9	2.2	4.7	4.6	26.0	14.0
			18/11	0.6	1.0	2.8	1.9	5.8	5.1	31.0	13.5
Total Nitrogen as N	1	1	29/08	<0.1	1.0	2.4	2.2	4.7	7.2	17.7	14.5
			15/09	<0.1	<0.1	3.4	2.5	7.9	4.9	36.9	19.3
			21/10	3.4	0.7	3.0	2.4	5.8	4.6	26.2	14.0
			18/11	0.6	1.0	2.8	2.1	6.8	5.5	31.7	13.6
Total Phosphorus as P	0.025	0.025	29/08	0.15	0.18	0.23	0.49	0.53	1.83	1.02	0.60
			15/09	0.08	0.11	0.12	0.22	0.67	0.33	0.64	0.43
			21/10	<0.01	<0.01	<0.01	0.11	0.42	0.24	0.42	0.04
			18/11	0.13	0.12	0.11	0.15	1.36	0.33	0.97	0.38

* ANZECC/ARMCANZ (2000) trigger values for 80% and 95% protection of freshwater ecosystems. Values exceeding ANZECC/ARMCANZ (2000) trigger values for 95% protection of freshwater ecosystems are shaded in orange. Values exceeding trigger values for 95% and 80% protection of freshwater ecosystems are shaded in red.

Table 23. Groundwater quality data for Campbell Park piezometers.

Parameter	Trigger values*		Date	Site 1		Site 2		Site 3		Site 4		
	95%	80%		Upper	Lower	Upper	Lower	Upper	Lower	Upper	Mid	Lower
General parameters												
pH – field	6.5-9.0	6.5-9.0	26/08	7.21	6.81	3.65	6.86	4.24	6.82	2.99	7.03	7.05
			17/09	6.75	6.34	2.43	6.38	4.30	6.32	3.01	6.63	6.87
			19/10	7.70	6.99	3.87	6.89	4.68	6.89	3.35	6.91	6.97
			17/11	-	7.12	4.69	7.16	5.44	7.04	4.18	7.60	7.65
pH – laboratory	6.5-9.0	6.5-9.0	26/08	6.41	6.89	3.29	6.93	3.26	7.06	3.14	7.07	6.94
			17/09	6.32	6.99	3.12	7.01	3.30	7.12	3.19	7.15	7.08
			19/10	6.70	7.03	3.33	7.00	3.34	7.03	3.30	7.05	7.10
			17/11	-	7.27	3.37	6.99	3.25	6.97	3.23	7.11	6.90
EC – field (mS/cm)	0.3-1	0.3-1	26/08	16.43	40.45	26.51	48.26	16.09	30.35	14.38	14.13	20.83
			17/09	15.95	41.50	28.80	42.20	11.07	18.61	8.92	14.83	20.50
			19/10	2.47	38.80	29.30	39.80	17.07	29.50	13.54	14.05	19.08
			17/11	-	39.7	29.7	45.6	19.74	30.2	12.76	14.34	18.89
EC – laboratory (mS/cm)	0.3-1	0.3-1	26/08	16.1	43.8	30.5	51.5	17.6	32.1	15.3	15.4	22.3
			17/09	16.4	43.4	29.0	48.5	18.1	31.9	14.6	15.0	20.6
			19/10	2.5	43.1	32.0	46.7	20.2	34.2	15.2	16.4	22.4
			17/11	-	40.7	31.2	46.2	20.1	30.4	13.6	14.6	19.1
ORP – field (mV)	n/a	n/a	26/08	-120	-236	32.3	-226	291	-264	397	-258	-179
			17/09	-27	-113	402	-205	158	-208	387	-244	-146
			19/10	1	-157	301	-176	177	-200	342	-208	-62
			17/11	-	-206	82	-244	-9	-225	327	-248	-176
DO – field (mg/L)	n/a	n/a	17/09	7.65	3.39	6.37	2.52	5.54	2.83	4.60	3.48	2.96
			19/10	5.13	2.32	-	1.46	3.03	3.64	2.99	3.87	3.78
			17/11	-	-	5.87	3.43	3.21	2.57	3.95	1.68	2.89
Alkalinity / acidity (units in mg/L CaCO₃)												
Alkalinity – field	n/a	n/a	26/08	-	-	-	-	-	-	-	690	735
			17/09	153	585	-	630	-	960	-	720	1800
			19/10	285	435	-	720	-	1095	-	840	1038
			17/11	-	630	-	642	-	1008	-	972	1002
Alkalinity – laboratory	n/a	n/a	26/08	118	515	-	495	-	-	-	746	674
			17/09	82	485	<1	488	<1	657	<1	626	627
			19/10	99	523	<1	670	<1	739	<1	691	693
			17/11	-	587	<1	625	<1	761	<1	792	239
Acidity – field	n/a	n/a	26/08	75	-	1650	-	375	-	625	-	-
			17/09	130	-	1950	-	575	-	1725	-	-
			19/10	-	-	1100	-	550	-	1250	-	-
			17/11	-	-	985	-	400	-	890	-	-
Acidity – laboratory	n/a	n/a	26/08	88	85	1580	85	502	120	1080	-	100
			17/09	57	-	1470	-	480	-	1360	-	-
			19/10	40	-	1260	-	519	-	1380	-	-
			17/11	-	-	708	-	295	-	698	-	-
Acidity – calculated	n/a	n/a	26/08	54	2	1311	42	413	15	510	3	13
			17/09	173	92	1521	146	443	171	1240	18	233
			19/10	1617	28	995	52	364	58	985	36	83
			17/11	-	4	741	8	293	4	688	11	11

Parameter	Trigger values*		Date	Site 1		Site 2		Site 3		Site 4		
	95%	80%		Upper	Lower	Upper	Lower	Upper	Lower	Upper	Mid	Lower
<i>Major ions (units in mg/L)</i>												
Ca	n/a	n/a	26/08	306	592	494	677	346	512	603	233	407
			17/09	333	654	491	686	366	546	596	289	398
			19/10	33	576	488	474	359	537	544	268	369
			17/11	-	704	549	875	423	591	494	272	384
Mg	n/a	n/a	26/08	409	992	920	821	598	790	839	296	549
			17/09	489	1110	913	1080	649	760	846	338	546
			19/10	27	944	920	507	614	686	751	299	490
			17/11	-	1280	1120	1260	803	871	775	349	546
Na	n/a	n/a	26/08	2570	8540	4970	11000	2670	6050	1840	2590	3660
			17/09	2940	9080	4990	9810	2930	6150	1760	2950	3660
			19/10	384	7510	5100	8700	2870	5690	1790	2700	3510
			17/11	-	8620	6330	10000	3590	5600	2090	2860	3810
K	n/a	n/a	26/08	84	214	186	209	133	189	145	70	76
			17/09	100	230	177	258	143	177	140	76	78
			19/10	20	195	187	145	141	161	121	69	74
			17/11	-	233	216	366	159	172	119	68	74
Cl	n/a	n/a	26/08	3520	10700	8440	17400	2970	10500	2580	4550	7140
			17/09	4670	16600	8120	18900	4750	10200	2360	4600	6950
			19/10	500	11900	8160	12100	4620	9000	2520	4250	6180
			17/11	-	13900	9780	16500	5140	10500	2770	4540	6930
SO ₄	n/a	n/a	26/08	1830	2810	5410	3650	3420	1700	5840	510	655
			17/09	2120	2930	5320	3200	3500	1850	6400	702	685
			19/10	202	2640	4660	2830	3280	1720	5500	613	599
			17/11	-	3340	5210	4150	3830	1800	5140	537	634
Cl:SO ₄ ratio	n/a	n/a	26/08	1.9	3.8	1.6	4.8	0.9	6.2	0.4	8.9	10.9
			17/09	2.2	5.7	1.5	5.9	1.4	5.5	0.4	6.6	10.1
			19/10	-	4.5	1.8	4.3	1.4	5.2	0.5	6.9	10.3
			17/11	-	4.2	1.9	4.0	1.3	5.8	0.5	8.5	10.9
<i>Dissolved metals (units in mg/L)</i>												
Al	0.055	0.15	26/08	0.37	0.07	90.00	2.36	23.70	1.40	52.90	0.20	0.09
			17/09	13.80	11.40	99.70	17.00	29.20	19.50	105.00	1.90	16.80
			19/10	185.00	3.52	53.20	6.28	20.00	6.97	75.60	4.09	5.52
			17/11	-	0.05	28.10	0.16	3.78	0.12	38.20	0.57	0.05
As	0.013 (AsV)	0.140 (AsV)	26/08	0.001	0.001	0.007	0.001	0.003	0.005	0.003	0.011	0.012
			17/09	0.014	0.017	0.012	0.035	0.007	0.030	0.011	0.009	0.134
			19/10	0.200	0.005	0.002	0.021	0.001	0.006	0.004	0.007	0.046
			17/11	-	0.013	0.008	0.008	0.005	0.008	0.007	0.004	0.016
Cd	0.0002	0.0008	26/08	0.0001	0.0002	0.0032	0.0002	0.0008	0.0002	0.0046	0.0002	0.0003
			17/09	0.0002	<0.0001	0.0022	0.0001	0.0004	0.0001	0.0060	0.0001	0.0002
			19/10	<0.0010	0.0001	0.0007	<0.0001	0.0004	0.0001	0.0022	<0.0001	<0.0001
			17/11	-	0.0002	0.0007	0.0004	0.0003	0.0002	0.0007	<0.0001	0.0003
Cu	0.0014	0.0025	26/08	0.005	0.007	0.059	0.008	0.013	0.006	0.104	0.002	0.003
			17/09	0.021	0.011	0.055	0.015	0.018	0.016	0.172	0.003	0.014
			19/10	0.143	0.008	0.024	0.009	0.013	0.009	0.065	0.006	0.007
			17/11	-	0.009	0.015	0.009	0.011	0.005	0.019	0.003	0.002

Parameter	Trigger values*		Date	Site 1		Site 2		Site 3		Site 4		
	95%	80%		Upper	Lower	Upper	Lower	Upper	Lower	Upper	Mid	Lower
Fe	n/a	n/a	26/08	15.8	<0.05	287	9.27	99.7	1.69	53.6	0.13	4.38
			17/09	31.4	10.1	279	18.0	99.4	21.8	214	2.44	50.7
			19/10	217.0	2.4	249	5.8	89.8	6.11	191	4.46	19.1
			17/11	-	0.7	209	1.7	97.3	0.47	166	2.40	3.7
Mn	1.9	3.6	26/08	5.17	0.779	13.0	1.84	4.77	1.53	10.2	0.721	0.176
			17/09	6.20	0.941	14.4	1.68	5.29	1.80	15.7	0.677	1.30
			19/10	1.45	0.74	11.70	0.99	5.03	1.25	14.40	0.72	0.45
			17/11	-	0.87	11.10	1.52	5.43	1.00	13.40	0.84	0.16
Ni	0.011	0.017	26/08	0.058	0.005	1.48	0.049	0.446	0.025	0.721	0.002	<0.001
			17/09	0.111	0.018	1.45	0.027	0.431	0.037	1.140	0.006	0.028
			19/10	0.133	0.003	0.831	0.008	0.328	0.009	0.902	0.008	0.008
			17/11	-	<0.001	0.444	0.001	0.208	0.002	0.500	0.007	0.001
Pb	0.0034	0.0094	26/08	<0.001	<0.001	0.038	<0.001	0.017	0.001	0.047	<0.001	<0.001
			17/09	0.024	0.009	0.043	0.017	0.020	0.019	0.039	0.004	0.029
			19/10	0.186	0.006	0.019	0.007	0.013	0.008	0.015	0.006	0.009
			17/11	-	<0.001	0.013	<0.001	0.005	<0.001	0.01	<0.001	<0.001
Se	0.011	0.034	26/08	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.01	<0.01	<0.01
			17/09	<0.01	<0.01	0.02	<0.01	<0.01	<0.01	0.02	<0.01	0.01
			19/10	<0.10	<0.01	<0.01	<0.01	<0.01	<0.01	0.01	<0.01	<0.01
			17/11	-	0.02	0.02	0.01	0.01	0.01	0.01	<0.01	<0.01
Zn	0.008	0.031	26/08	0.046	0.009	0.678	0.025	0.385	0.022	0.339	0.008	0.009
			17/09	0.110	0.023	0.638	0.050	0.366	0.091	0.634	0.014	0.036
			19/10	0.517	0.018	0.328	0.013	0.318	0.028	0.482	0.017	0.018
			17/11	-	0.014	0.243	0.020	0.266	0.014	0.307	0.022	0.009
Nutrients (units in mg/L)												
Nitrite + Nitrate as N	0.1	0.1	26/08	0.02	0.03	0.18	0.02	0.10	0.03	0.10	0.02	0.03
			17/09	0.03	0.02	0.16	0.02	0.06	0.03	0.21	0.02	0.02
			19/10	4.03	0.03	0.16	0.02	0.09	0.02	0.16	0.01	0.02
			17/11	-	0.01	0.09	<0.01	0.03	0.02	0.09	<0.01	<0.01
Total Kjeldahl Nitrogen as N	n/a	n/a	26/08	2.3	9.1	4.5	8.9	3.1	9.9	3.2	5.1	5.5
			17/09	3.0	10.4	6.0	13.2	2.3	11.0	3.9	4.0	7.7
			19/10	3.0	5.9	5.4	5.6	2.2	5.9	3.5	3.6	4.6
			17/11	-	7.0	4.6	8.3	3.9	6.5	3.6	4.2	4.2
Total Nitrogen as N	1	1	26/08	2.4	9.1	4.6	8.9	3.2	10.0	3.3	5.2	5.5
			17/09	3.1	10.4	6.1	13.3	2.4	11.0	4.1	4.0	7.7
			19/10	7.0	5.9	5.6	5.6	2.2	5.9	3.7	3.6	4.6
			17/11	-	7.0	4.7	8.3	4.0	6.5	3.7	4.2	4.2
Total Phosphorus as P	0.025	0.025	26/08	0.18	1.15	0.17	0.65	0.67	1.60	0.13	0.76	0.20
			17/09	0.32	0.67	0.18	0.77	0.09	1.15	0.12	0.44	0.52
			19/10	1.06	0.34	0.01	0.15	<0.01	0.46	<0.01	0.21	<0.01
			17/11	-	1.05	0.25	1.00	0.12	1.43	0.14	0.66	0.18

* ANZECC/ARMCANZ (2000) trigger values for 80% and 95% protection of freshwater ecosystems. Values exceeding ANZECC/ARMCANZ (2000) trigger values for 95% protection of freshwater ecosystems are shaded in orange. Values exceeding trigger values for 95% and 80% protection of freshwater ecosystems are shaded in red.

Table 24. Groundwater quality data for Windmill piezometers.

Parameter	Trigger values*		Date	Site 1		Site 2	Site 3	Site 4	
	95%	80%		Upper	Lower	Upper	Upper	Upper	Lower
General parameters									
pH – field	6.5-9.0	6.5-9.0	28/08	7.03	7.00	7.13	7.19	6.95	6.86
			16/09	7.91	7.65	7.70	7.59	7.35	7.29
			20/10	6.82	6.81	7.05	6.96	6.87	6.17
			16/11	7.20	6.86	6.98	6.89	6.69	6.55
pH – laboratory	6.5-9.0	6.5-9.0	28/08	7.08	6.97	7.02	7.33	6.98	6.65
			16/09	7.38	7.22	7.34	7.28	7.18	6.89
			20/10	7.20	7.10	7.13	7.13	7.07	6.72
			16/11	7.28	6.85	7.20	7.15	6.99	6.65
EC – field (mS/cm)	0.3-1	0.3-1	28/08	30.30	31.70	30.70	20.76	13.06	30.30
			16/09	27.70	26.60	28.10	19.38	13.57	26.30
			20/10	30.60	26.20	29.80	21.18	13.68	27.20
			16/11	29.00	26.10	29.60	21.74	14.63	27.30
EC – laboratory (mS/cm)	0.3-1	0.3-1	28/08	31.5	31.4	30.1	18.1	12.9	30.5
			16/09	29.5	27.9	29.6	20.9	14.5	28.2
			20/10	33.6	28.7	33.9	24.1	15.5	31.8
			16/11	29.2	25.9	29.8	21.6	14.7	27.3
ORP – field (mV)	n/a	n/a	28/08	-160	-55	-107	-115	-139	-81
			16/09	-96	-80	-107	-119	-82	-105
			20/10	-126	-42	-77	-96	-81	-80
			16/11	-77	-37	-69	-95	-102	-68
DO – field (mg/L)	n/a	n/a	16/09	9.40	8.39	7.71	7.52	12.15	8.80
			20/10	2.44	4.02	3.68	3.59	3.39	1.89
			16/11	3.69	5.04	3.32	3.30	4.41	3.66
Alkalinity / acidity (units in mg/L CaCO₃)									
Alkalinity – field	n/a	n/a	28/08	-	390	-	-	-	-
			16/09	510	450	660	870	660	651
			20/10	588	447	834	933	1059	648
			16/11	510	354	600	780	876	540
Alkalinity – laboratory	n/a	n/a	28/08	494	441	582	1050	-	392
			16/09	504	421	593	926	1040	449
			20/10	502	423	574	890	1020	436
			16/11	512	349	598	872	1030	443
Acidity – field	n/a	n/a	28/08	-	-	-	-	-	-
			16/09	-	-	-	-	-	-
			20/10	-	-	-	-	-	-
			16/11	-	-	-	-	-	-
Acidity – laboratory	n/a	n/a	28/08	55	55	60	95	130	100
			16/09	-	-	-	-	-	-
			20/10	-	-	-	-	-	-
			16/11	-	-	-	-	-	-
Acidity – calculated	n/a	n/a	28/08	6	16	4	10	13	50
			16/09	232	3	183	336	176	128
			20/10	140	39	247	211	78	158
			16/11	4	8	11	36	9	76

Parameter	Trigger values*		Date	Site 1		Site 2	Site 3	Site 4	
	95%	80%		Upper	Lower	Upper	Upper	Upper	Lower
<i>Major ions (units in mg/L)</i>									
Ca	n/a	n/a	28/08	390	420	413	417	311	756
			16/09	381	346	415	456	384	789
			20/10	403	342	433	499	384	823
			16/11	391	338	448	541	335	794
Mg	n/a	n/a	28/08	707	655	754	553	444	963
			16/09	709	564	762	653	542	940
			20/10	712	545	779	701	526	972
			16/11	764	576	854	822	545	1120
Na	n/a	n/a	28/08	5840	6200	5860	2710	1450	4400
			16/09	5800	5430	5770	3220	1800	4400
			20/10	5880	5310	5820	3530	1880	4250
			16/11	6350	5600	6200	3540	2080	4500
K	n/a	n/a	28/08	169	126	119	55	50	88
			16/09	167	118	116	67	53	83
			20/10	175	119	125	75	55	86
			16/11	174	114	122	77	57	83
Cl	n/a	n/a	28/08	9430	10400	9340	5850	3000	10000
			16/09	9440	9110	9600	6880	4360	9150
			20/10	9520	8580	9740	7210	3850	8750
			16/11	11900	8850	10100	7350	4020	10400
SO ₄	n/a	n/a	28/08	1750	1660	1600	216	90	1300
			16/09	1710	1240	1590	417	185	1420
			20/10	1710	1160	1680	536	202	1230
			16/11	1850	1200	1820	674	266	1250
Cl:SO ₄ ratio	n/a	n/a	28/08	5.4	6.3	5.8	27.1	33.3	7.7
			16/09	5.5	7.3	6.0	16.5	23.6	6.4
			20/10	5.6	7.4	5.8	13.5	19.1	7.1
			16/11	6.4	7.4	5.5	10.9	15.1	8.3
<i>Dissolved metals (units in mg/L)</i>									
Al	0.055	0.15	28/08	0.11	0.20	0.11	0.23	0.05	0.01
			16/09	23.2	<0.01	19.4	32.4	14.7	5.70
			20/10	14.2	3.29	25.0	16.4	7.13	8.70
			16/11	0.02	0.03	0.12	0.13	0.02	0.02
As	0.013 (AsV)	0.140 (AsV)	28/08	0.001	0.02	0.001	<0.001	0.002	0.012
			16/09	0.05	0.009	0.032	0.048	0.037	0.036
			20/10	0.02	0.027	0.034	0.015	0.011	0.037
			16/11	0.005	0.022	0.004	0.005	0.004	0.032
Cd	0.0002	0.0008	28/08	0.0003	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
			16/09	0.0001	<0.0001	0.0002	0.0002	<0.0001	<0.0001
			20/10	0.0003	0.0002	0.0005	0.0002	0.0006	0.0001
			16/11	0.0006	0.0001	0.0003	0.0002	0.0006	0.0003
Cu	0.0014	0.0025	28/08	0.005	0.005	0.004	0.002	0.002	0.004
			16/09	0.018	0.004	0.013	0.017	0.019	0.012
			20/10	0.013	0.006	0.021	0.011	0.010	0.018
			16/11	0.005	0.003	0.005	0.005	0.001	0.005

Parameter	Trigger values*		Date	Site 1		Site 2	Site 3	Site 4	
	95%	80%		Upper	Lower	Upper	Upper	Upper	Lower
Fe	n/a	n/a	28/08	1.58	4.26	0.54	2.62	0.41	17.1
			16/09	37.6	n/a	27.0	57.0	31.9	34.3
			20/10	21.9	6.55	38.9	43.6	11.3	39.0
			16/11	0.9	1.9	3.1	12.4	1.1	26.6
Mn	1.9	3.6	28/08	0.628	1.890	0.898	1.160	6.320	2.410
			16/09	0.958	1.590	1.150	1.250	4.500	2.360
			20/10	0.860	1.720	1.400	1.120	4.230	2.440
			16/11	0.511	1.660	1.110	1.170	3.240	2.530
Ni	0.011	0.017	28/08	0.004	0.007	0.004	0.001	0.006	0.002
			16/09	0.018	0.006	0.014	0.029	0.020	0.006
			20/10	0.012	0.009	0.020	0.013	0.009	0.012
			16/11	0.002	0.006	0.003	0.002	0.003	0.003
Pb	0.0034	0.0094	28/08	<0.001	0.001	<0.001	<0.001	<0.001	<0.001
			16/09	0.034	<0.001	0.026	0.036	0.025	0.007
			20/10	0.022	0.005	0.056	0.024	0.012	0.019
			16/11	<0.001	<0.001	<0.001	0.002	<0.001	<0.001
Se	0.011	0.034	28/08	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
			16/09	0.01	0.02	0.01	<0.01	<0.01	0.02
			20/10	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
			16/11	0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Zn	0.008	0.031	28/08	0.012	0.011	0.016	0.006	0.006	0.010
			16/09	0.045	n/a	0.028	0.040	0.030	0.017
			20/10	0.046	0.045	0.056	0.033	0.038	0.036
			16/11	0.012	0.018	0.017	0.054	<0.005	0.017
Nutrients (units in mg/L)									
Nitrite + Nitrate as N	0.1	0.1	28/08	0.02	0.02	0.02	0.02	0.02	0.01
			16/09	0.02	0.03	0.02	0.05	0.06	0.03
			20/10	0.01	0.01	0.19	0.26	0.04	0.01
			16/11	<0.01	0.04	0.56	1.60	0.99	<0.01
Total Kjeldahl Nitrogen as N	n/a	n/a	28/08	3.0	3.7	6.6	14.6	47.0	13.0
			16/09	3.0	4.7	8.9	17.9	49.9	16.7
			20/10	2.0	2.2	5.0	9.0	33.0	10.7
			16/11	1.80	2.10	5.90	11.70	32.10	12.40
Total Nitrogen as N	1	1	28/08	3.0	3.7	6.6	14.6	47.0	13.0
			16/09	3.0	4.7	8.9	17.9	50.0	16.7
			20/10	2.0	2.2	5.2	9.2	33.0	10.8
			16/11	1.80	2.20	6.40	13.30	33.10	12.40
Total Phosphorus as P	0.025	0.025	28/08	0.68	0.16	1.09	2.30	1.54	0.23
			16/09	0.64	0.38	1.0	1.52	1.66	0.07
			20/10	0.36	<0.01	0.94	2.09	1.26	<0.01
			16/11	0.74	0.06	1.29	1.13	1.62	0.03

* ANZECC/ARMCANZ (2000) trigger values for 80% and 95% protection of freshwater ecosystems. Values exceeding ANZECC/ARMCANZ (2000) trigger values for 95% protection of freshwater ecosystems are shaded in orange. Values exceeding trigger values for 95% and 80% protection of freshwater ecosystems are shaded in red.

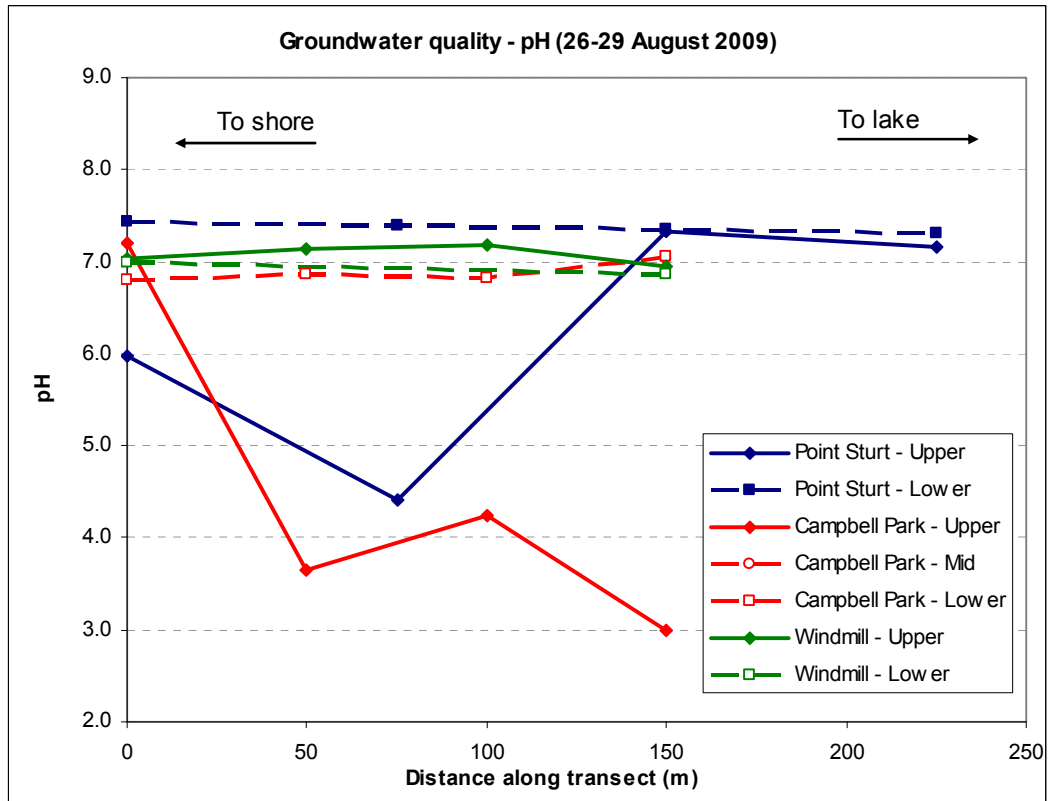


Figure 48. Groundwater pH at Point Sturt, Campbell Park and Windmill locations (after purging), 26-29 August 2009 (baseline).

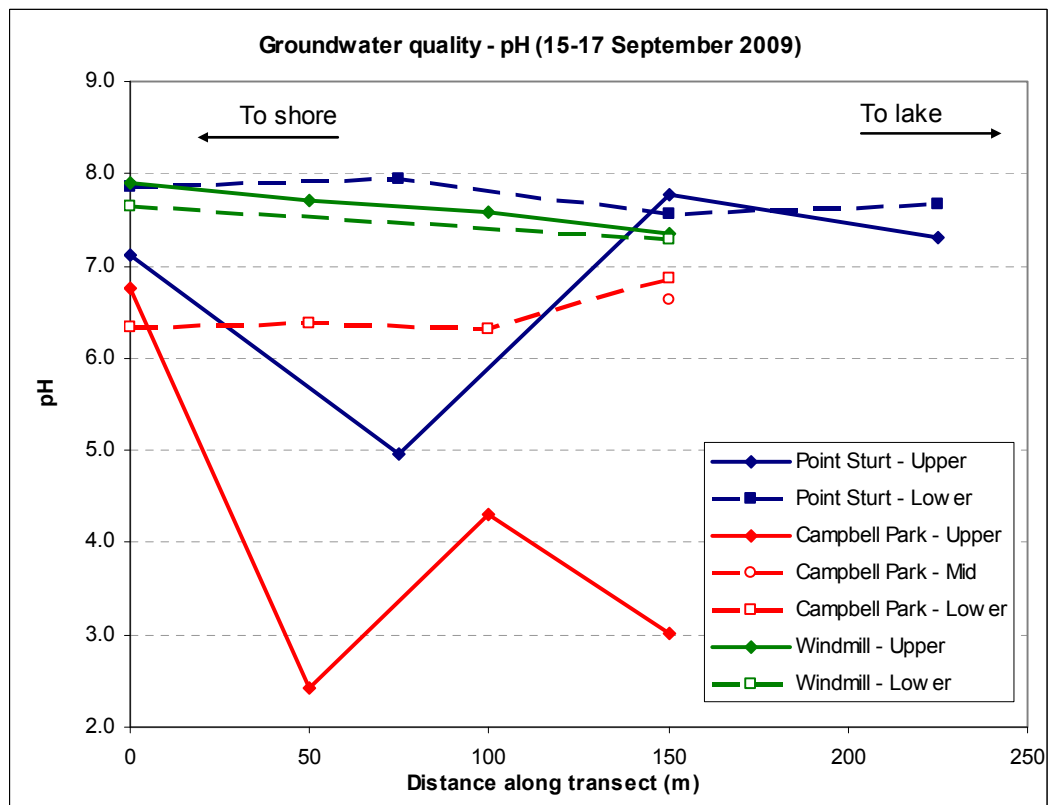


Figure 49. Groundwater pH at Point Sturt, Campbell Park and Windmill locations (after purging), 15-17 September 2009.

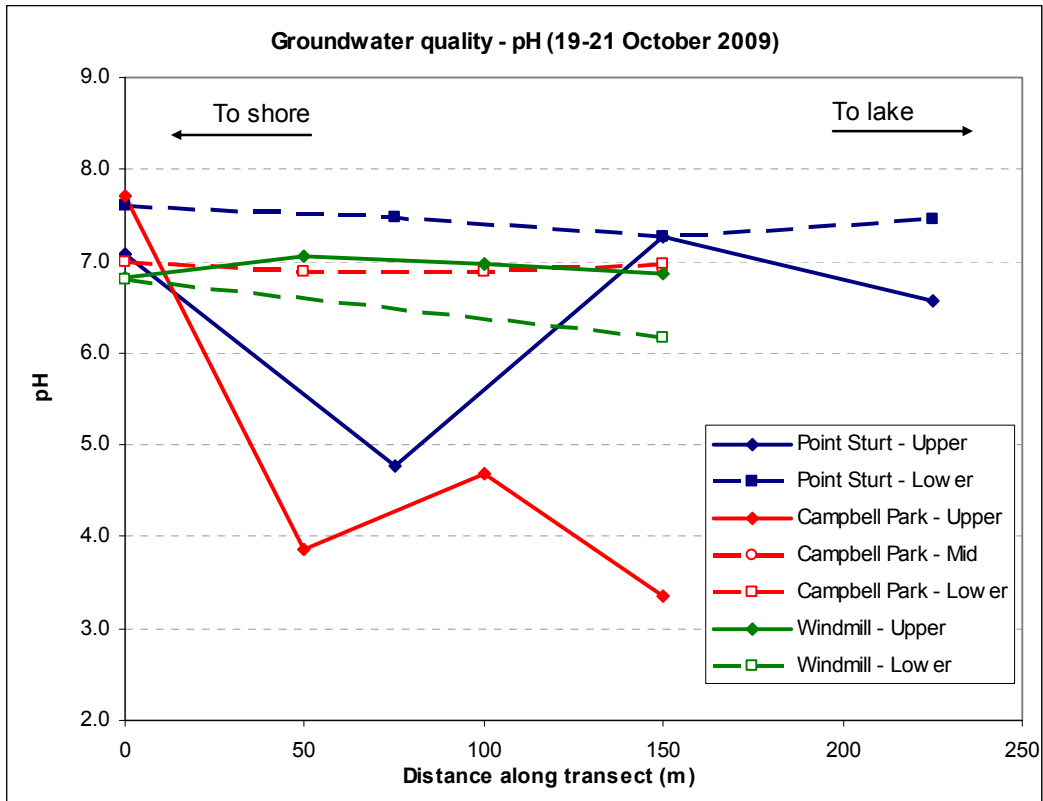


Figure 50. Groundwater pH at Point Sturt, Campbell Park and Windmill locations (after purging), 19-21 October 2009.

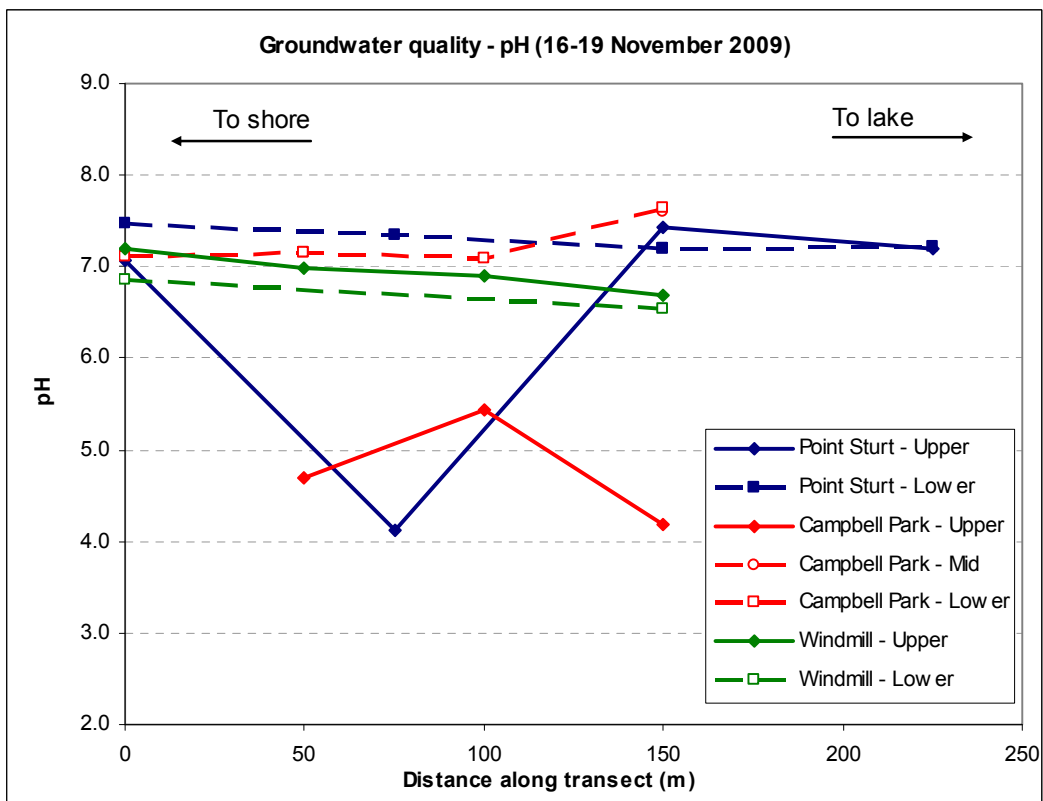


Figure 51. Groundwater pH at Point Sturt, Campbell Park and Windmill locations (after purging), 16-19 November 2009.

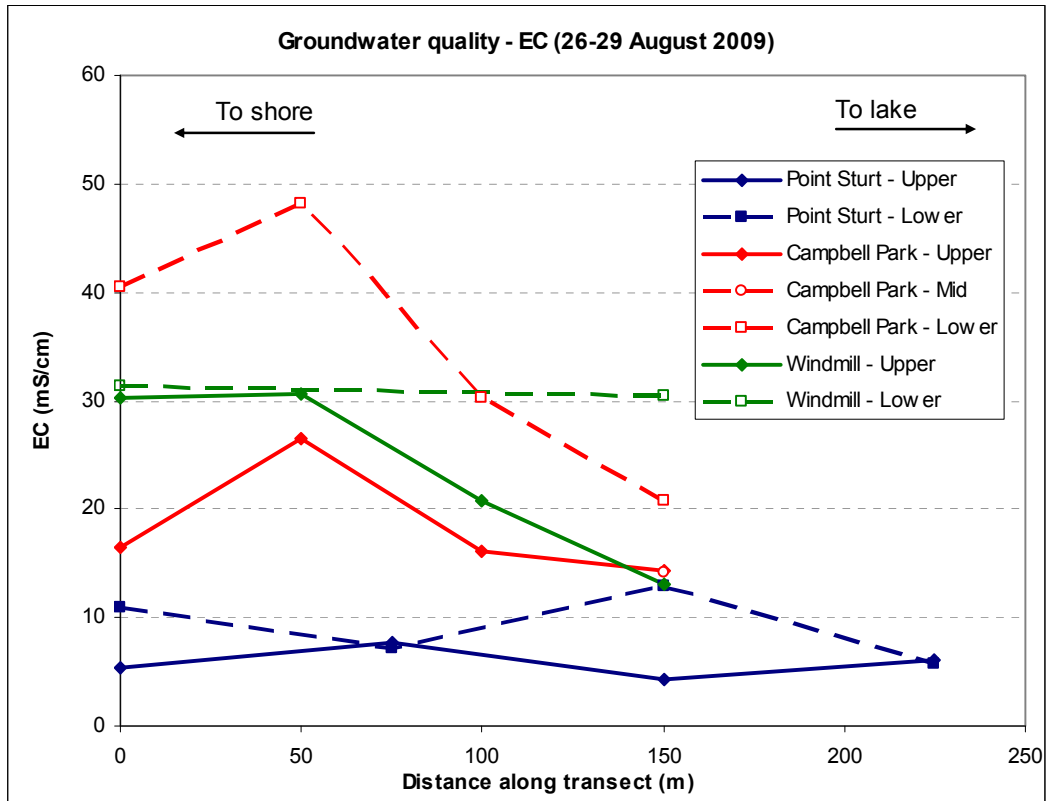


Figure 52. Groundwater EC at Point Sturt, Campbell Park and Windmill locations (after purging), 26-29 August 2009 (baseline).

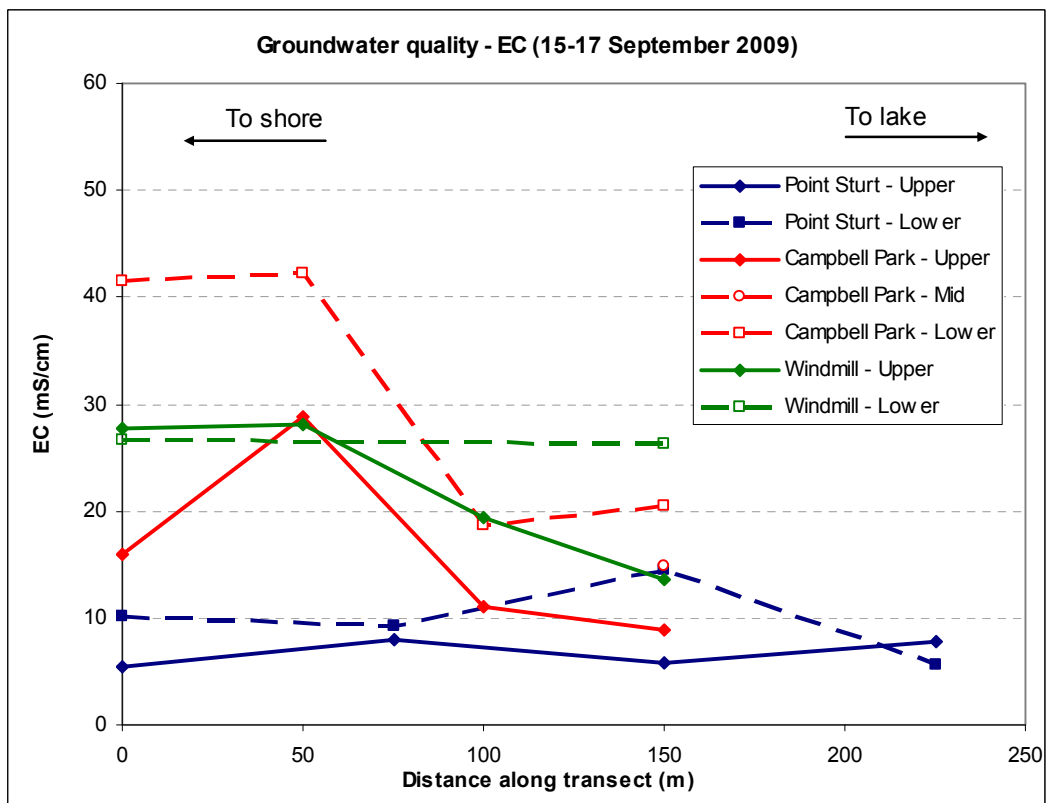


Figure 53. Groundwater EC at Point Sturt, Campbell Park and Windmill locations (after purging), 15-17 September 2009.

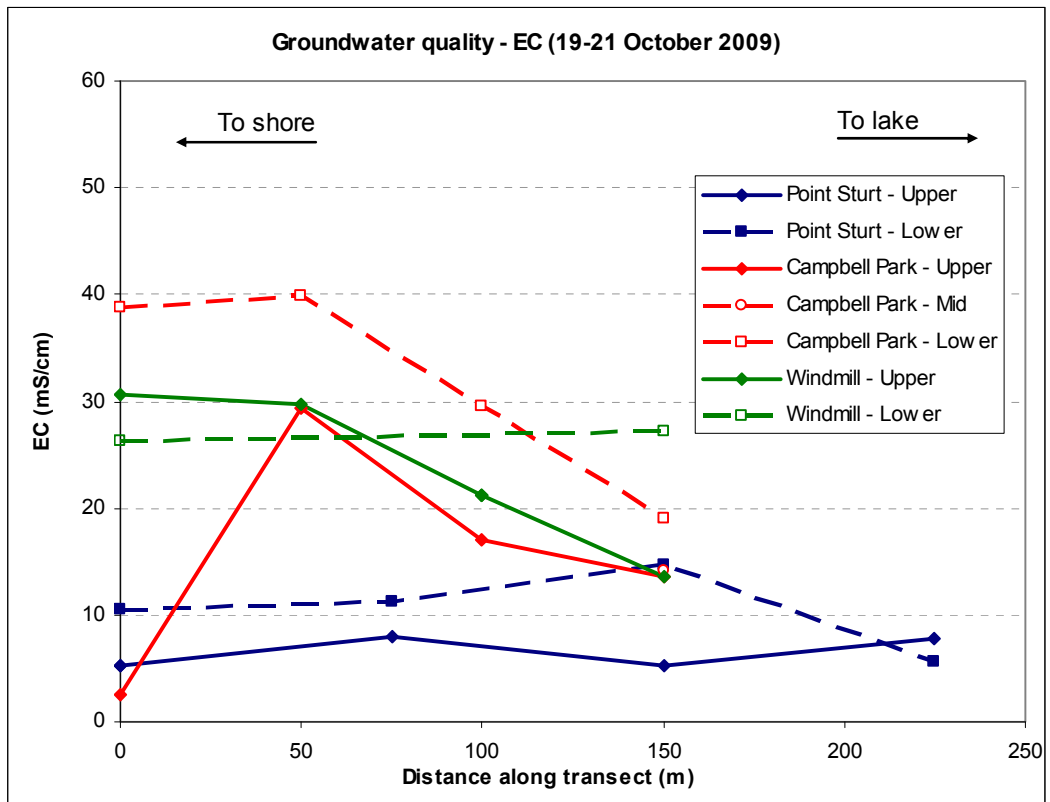


Figure 54. Groundwater EC at Point Sturt, Campbell Park and Windmill locations (after purging), 19-21 October 2009.

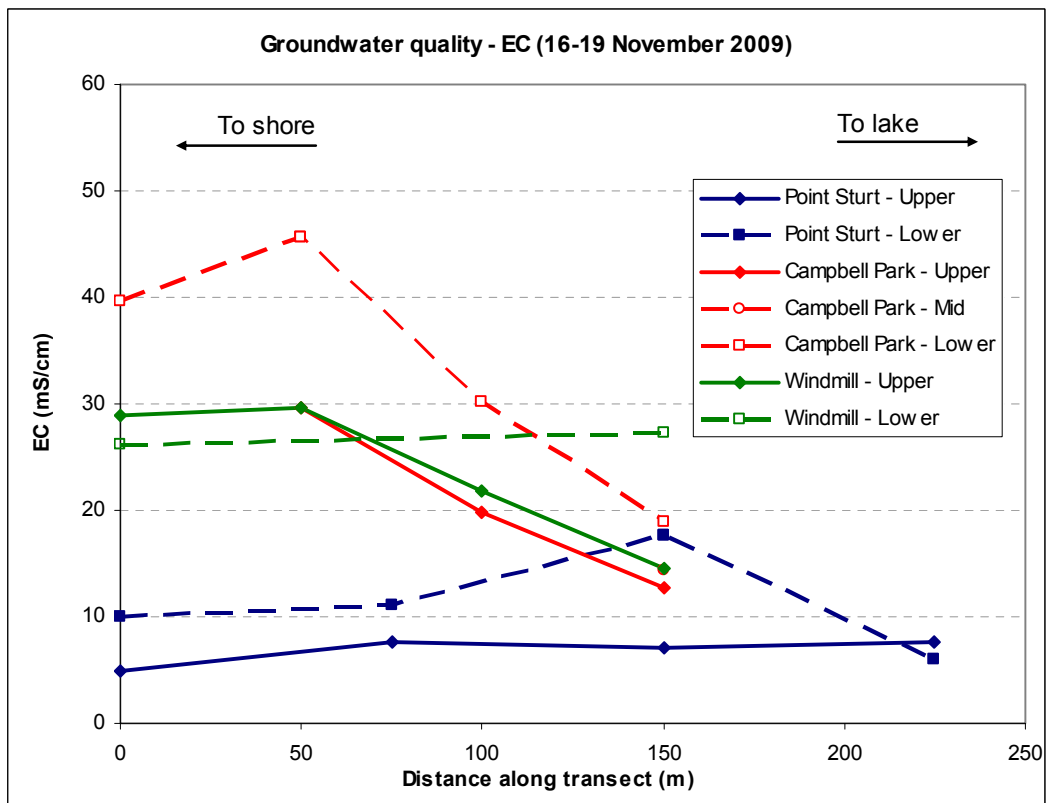


Figure 55. Groundwater EC at Point Sturt, Campbell Park and Windmill locations (after purging), 16-19 November 2009.

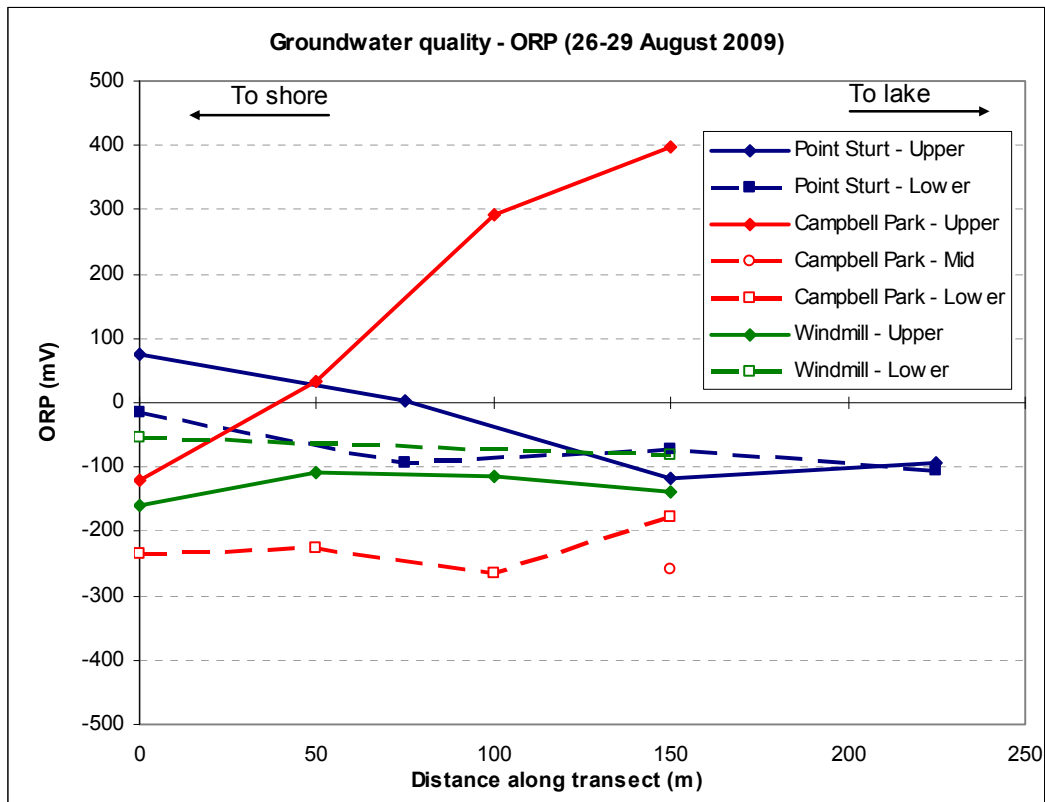


Figure 56. Groundwater ORP at Point Sturt, Campbell Park and Windmill locations (after purging), 26-29 August 2009 (baseline).

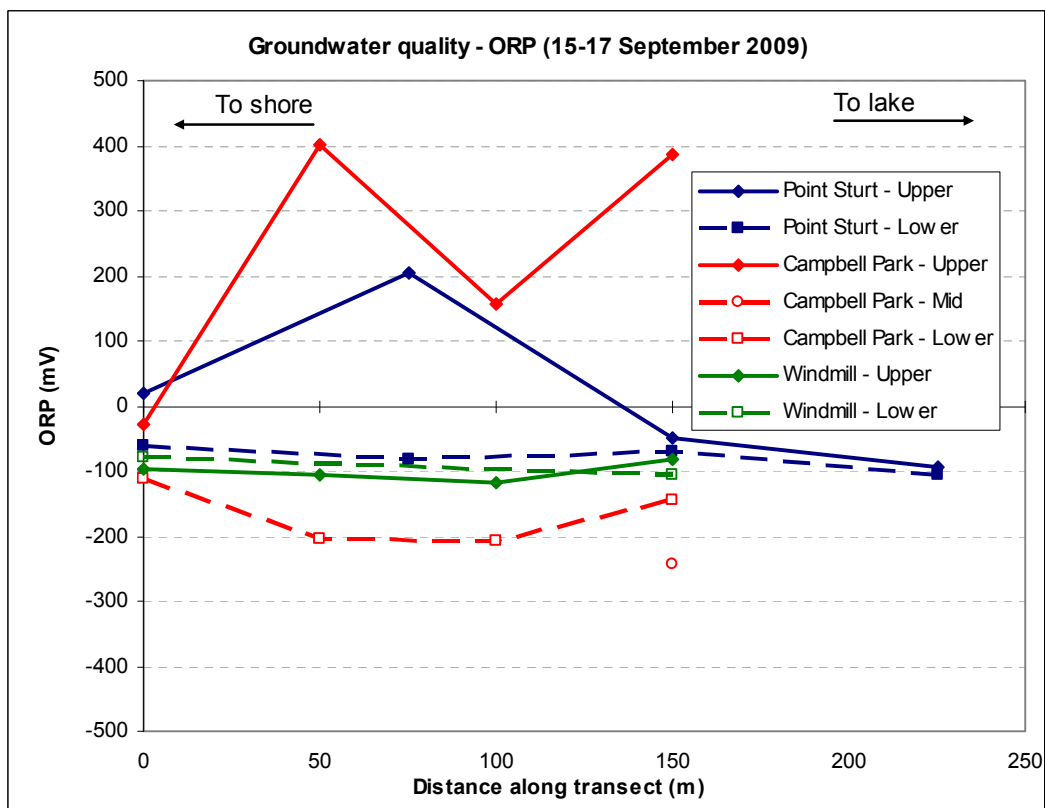


Figure 57. Groundwater ORP at Point Sturt, Campbell Park and Windmill locations (after purging), 15-17 September 2009.

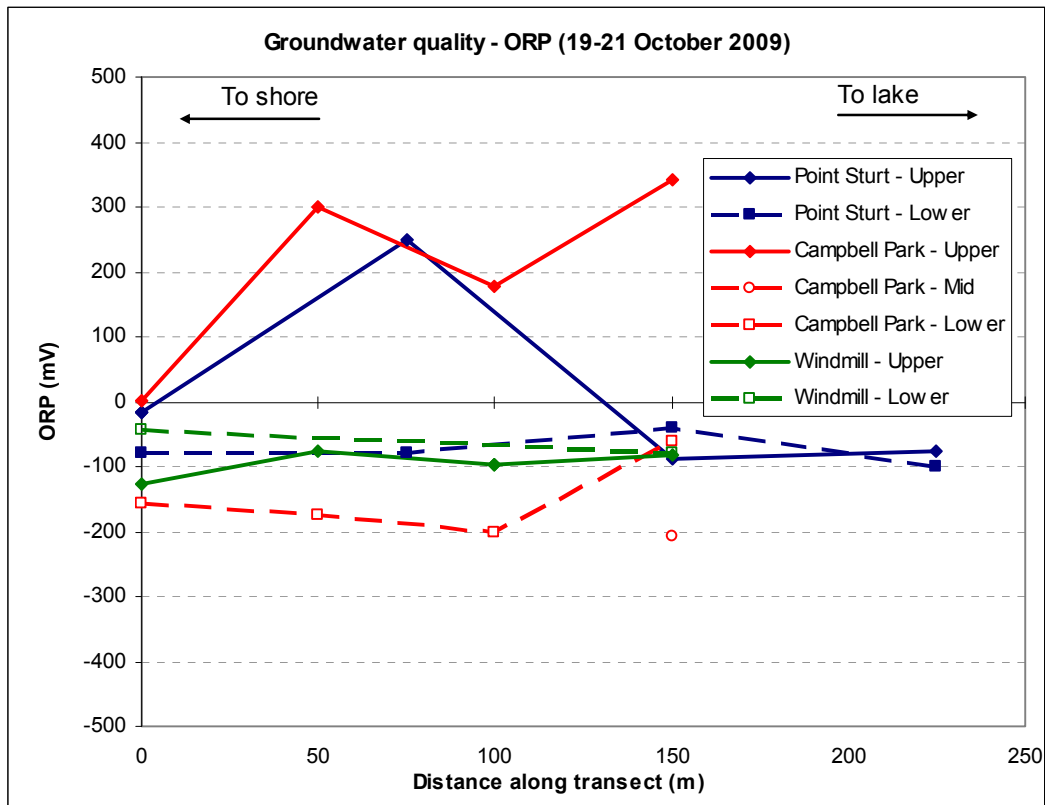


Figure 58. Groundwater ORP at Point Sturt, Campbell Park and Windmill locations (after purging), 19-21 October 2009.

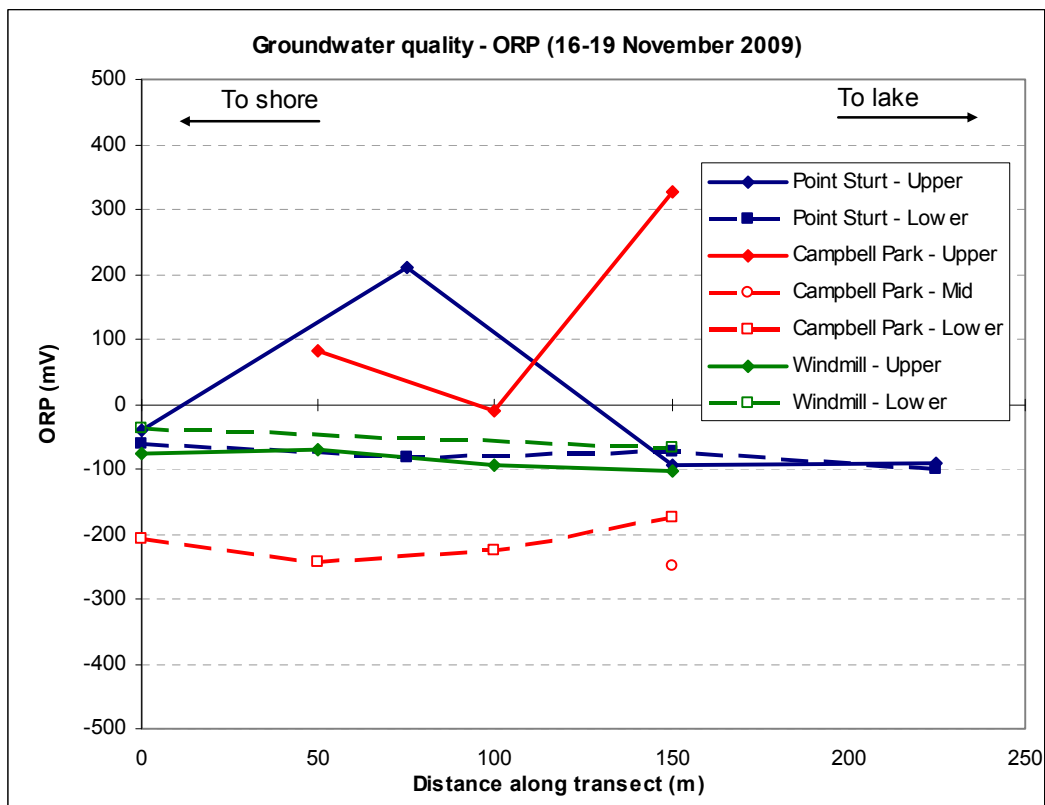


Figure 59. Groundwater ORP at Point Sturt, Campbell Park and Windmill locations (after purging), 16-19 November 2009.

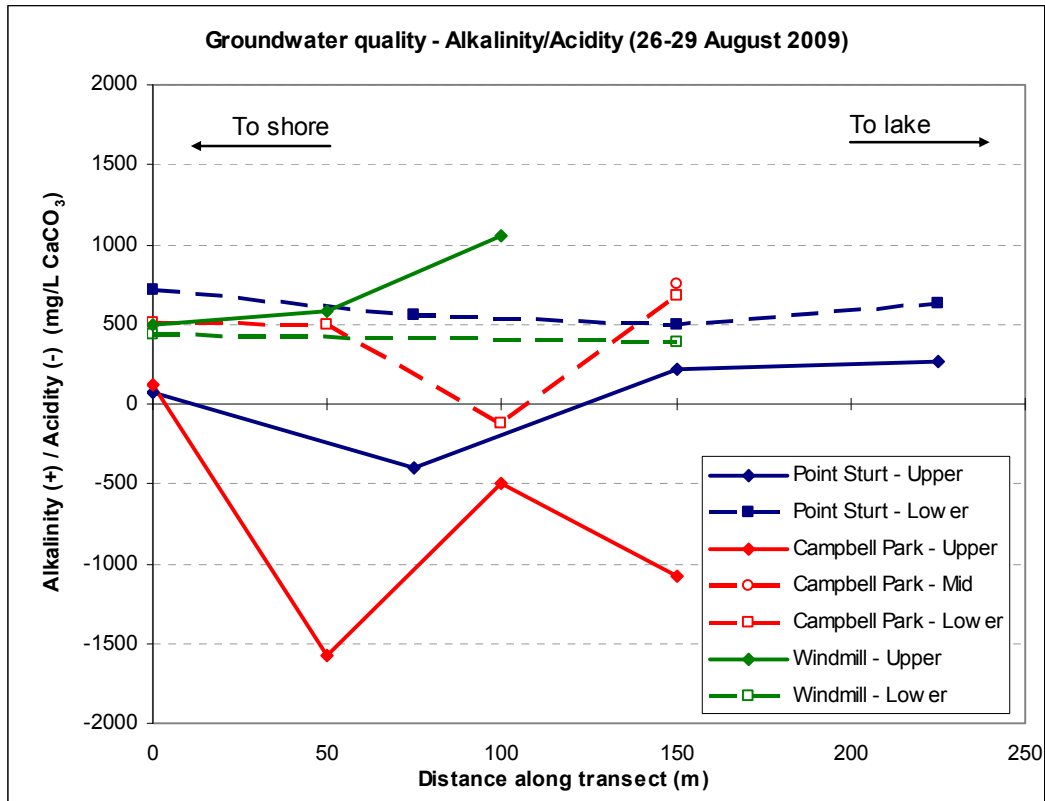


Figure 60. Groundwater alkalinity (positive values) and acidity (negative values) at Point Sturt, Campbell Park and Windmill locations (after purging), 26-29 August 2009 (baseline).

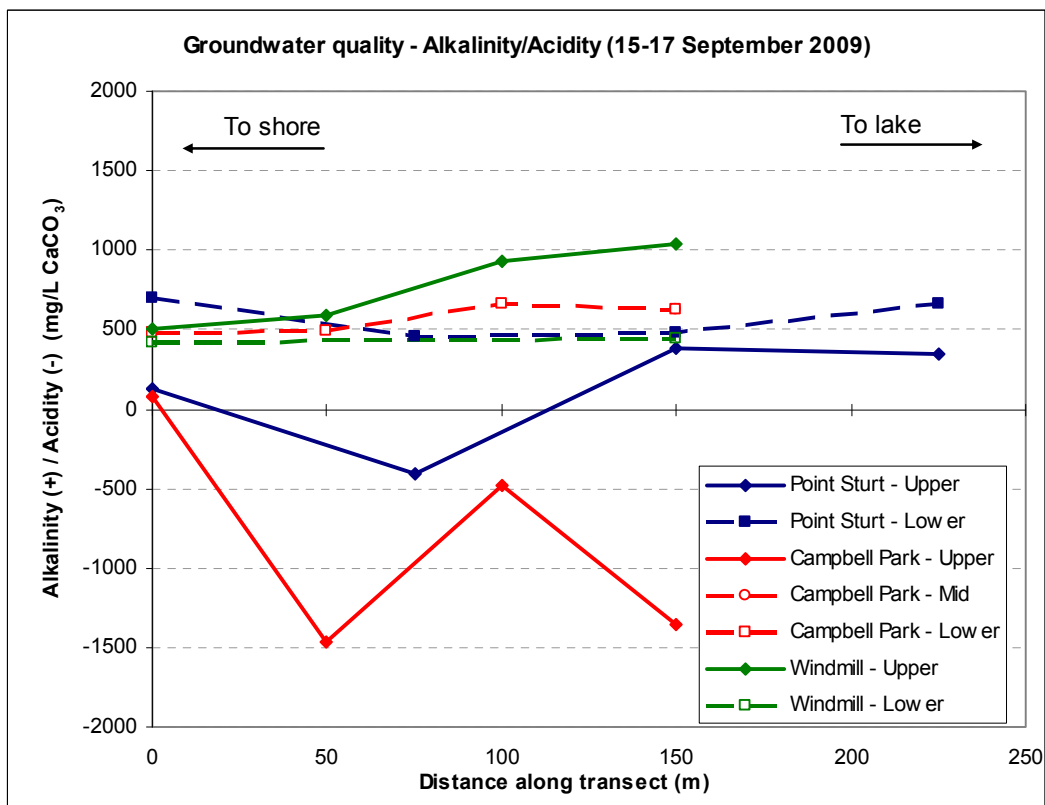


Figure 61. Groundwater alkalinity (positive values) and acidity (negative values) at Point Sturt, Campbell Park and Windmill locations (after purging), 15-17 September 2009.

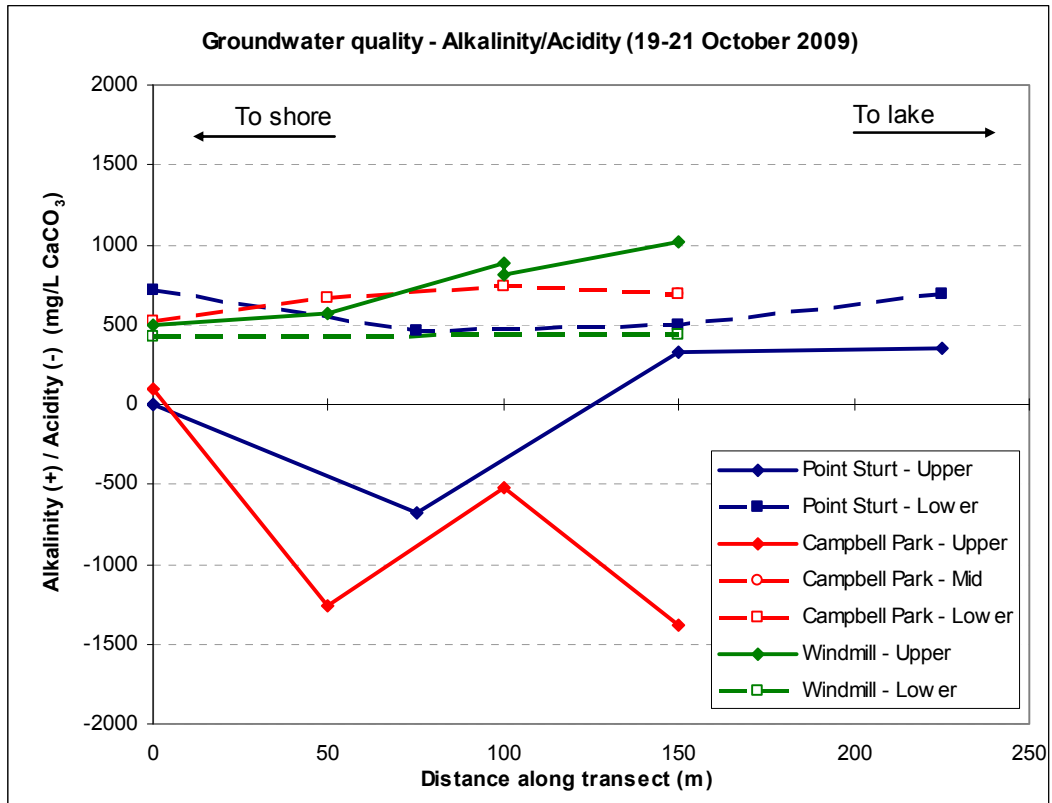


Figure 62. Groundwater alkalinity (positive values) and acidity (negative values) at Point Sturt, Campbell Park and Windmill locations (after purging), 19-21 October 2009.

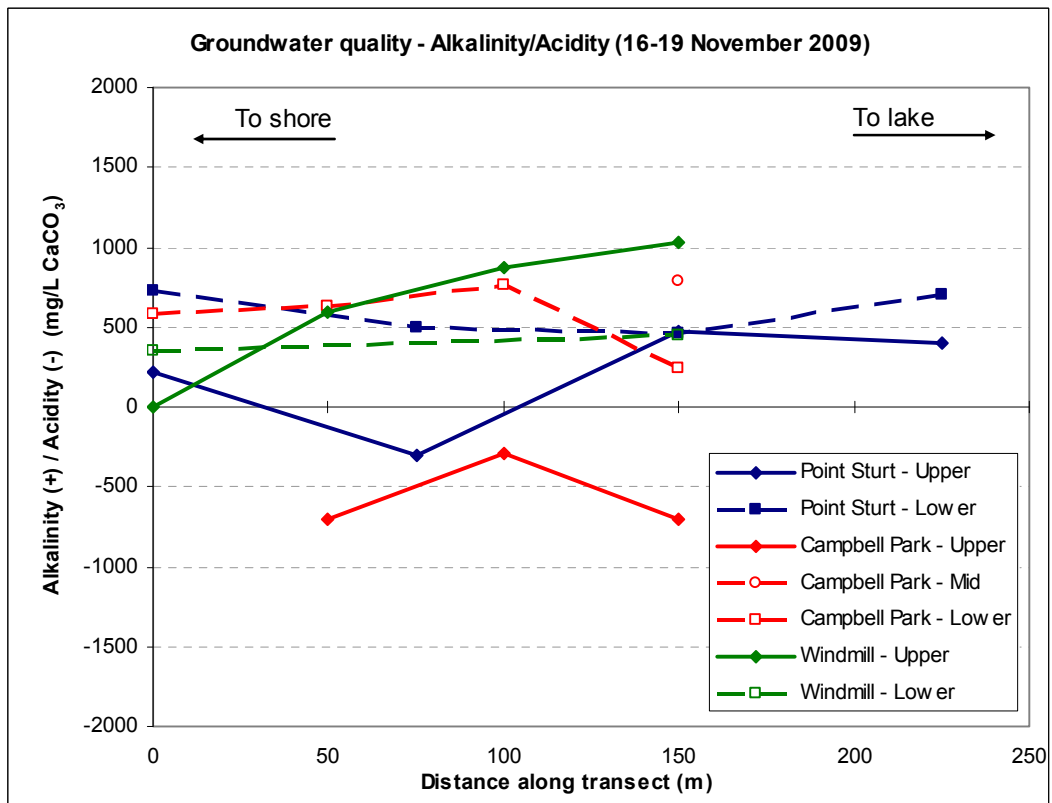


Figure 63. Groundwater alkalinity (positive values) and acidity (negative values) at Point Sturt, Campbell Park and Windmill locations (after purging), 16-19 November 2009.

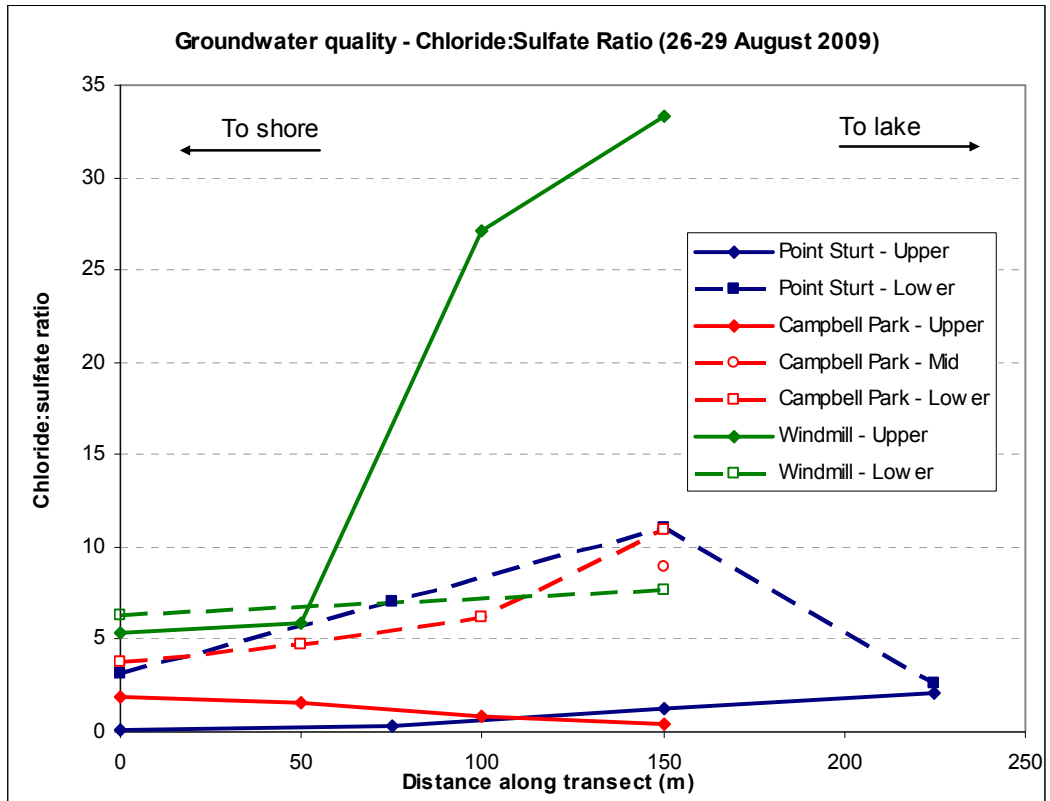


Figure 64. Groundwater chloride:sulfate ratio at Point Sturt, Campbell Park and Windmill locations (after purging), 26-29 August 2009 (baseline).

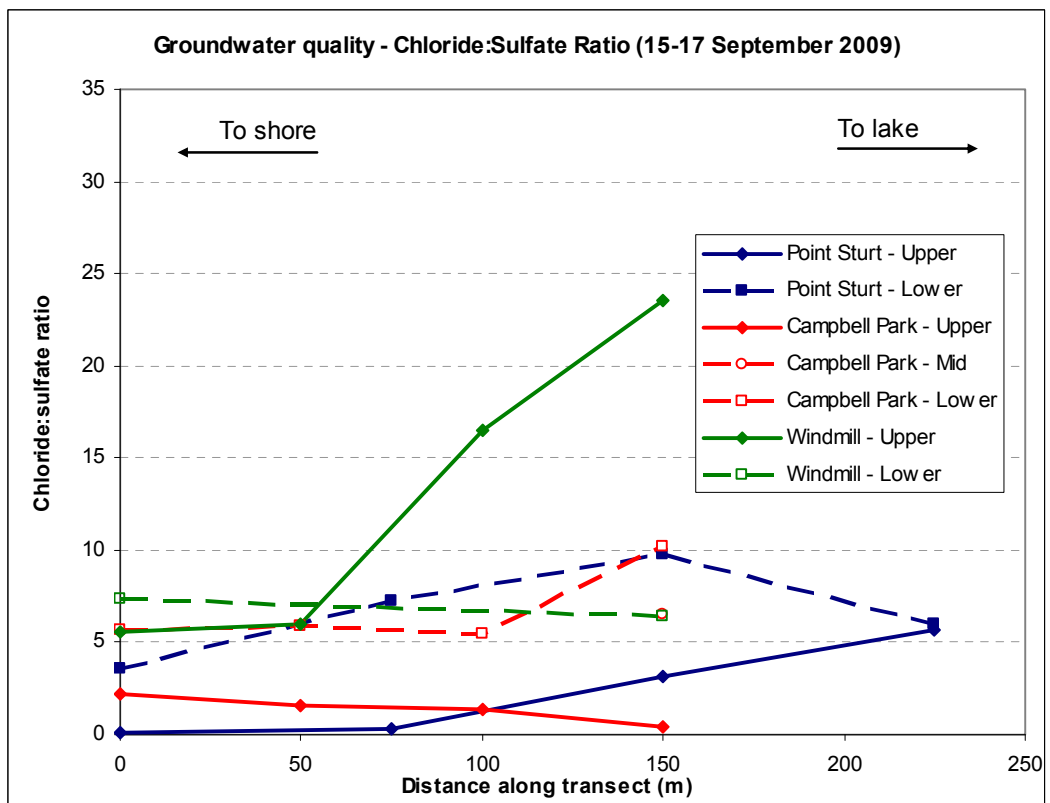


Figure 65. Groundwater chloride:sulfate ratio at Point Sturt, Campbell Park and Windmill locations (after purging), 15-17 September 2009.

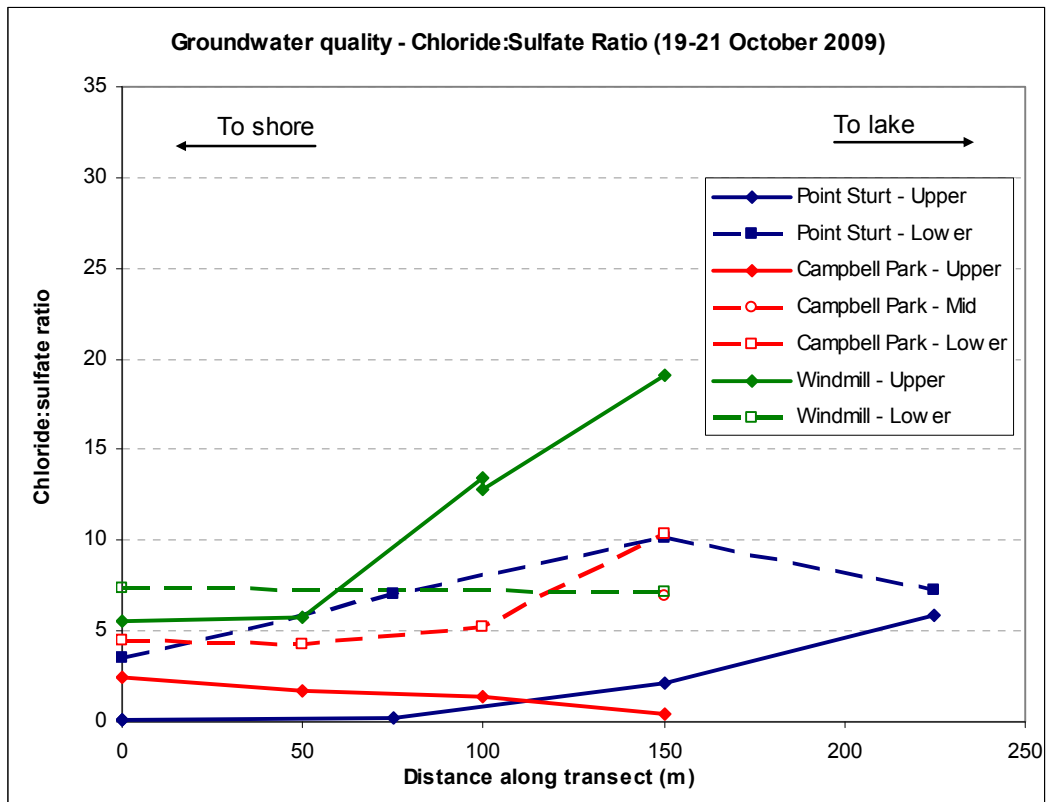


Figure 66. Groundwater chloride:sulfate ratio at Point Sturt, Campbell Park and Windmill locations (after purging), 19-21 October 2009.

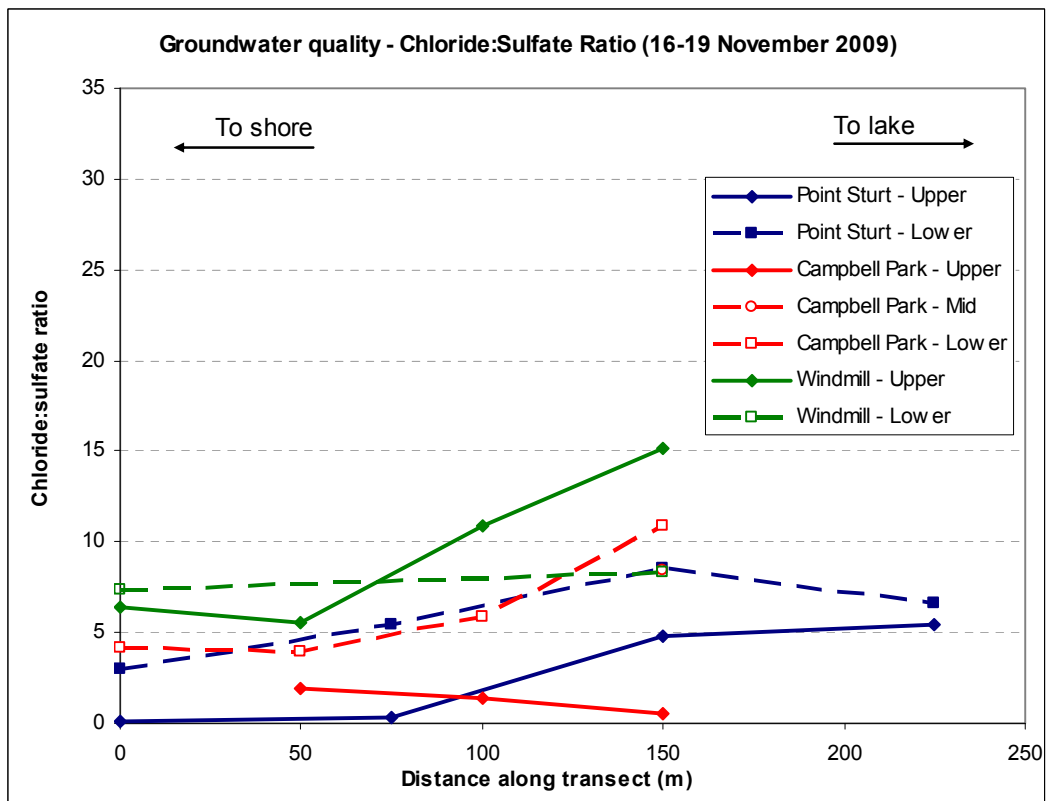
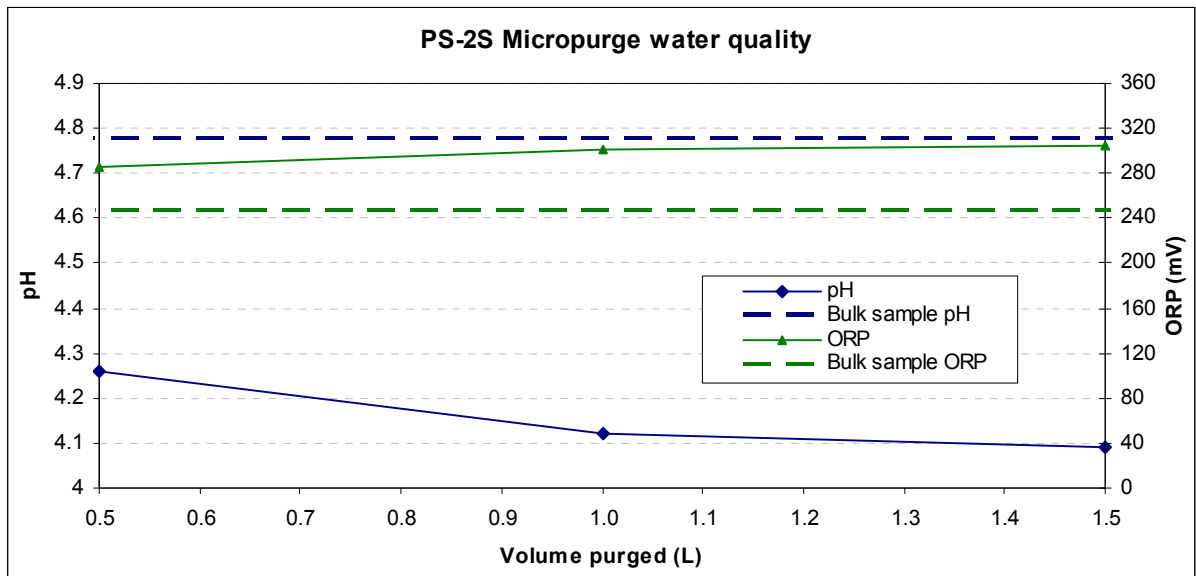
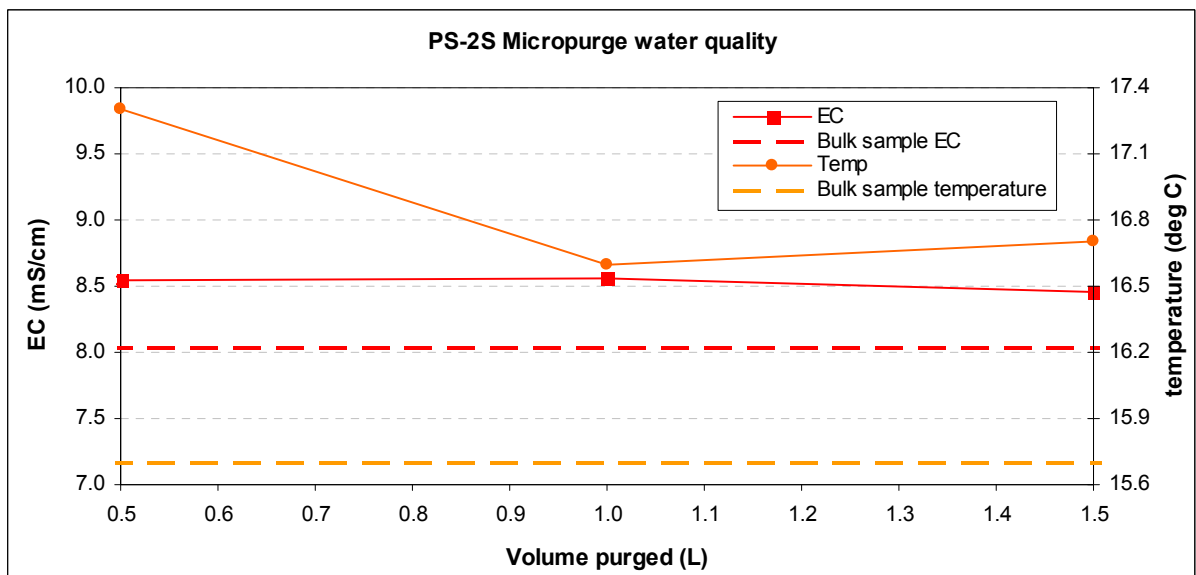


Figure 67. Groundwater chloride:sulfate ratio at Point Sturt, Campbell Park and Windmill locations (after purging), 16-19 November 2009.



(a)



(b)

Figure 68. Water quality during low-flow sampling at Point Sturt (Site 2, shallow) on 21 October 2009 at 0.8 m below ground level. Solid lines depict water quality parameters (a) pH and ORP and (b) EC and temperature stabilising over volumetric intervals. Dashed lines indicate the corresponding parameter value of the bulk sample collected after completion of low-flow sampling and after purging and recovery of the piezometric level.

6 Discussion

This section discusses the implications of laboratory results and field data obtained (to date) in terms of the potential for acidity generation and flux to Currency Creek, Lake Albert and Lake Alexandrina. The outcomes of this discussion form the basis for developing acidity generation and flux models for each region (Section 7).

6.1 *Distribution of sands vs. clays*

The distribution of sands and clays (both laterally and vertically) has important implications for the modelling of acidity generation processes in the lakes and the development of appropriate ASS management strategies in the short, medium and long term.

The peripheries of lakes Alexandrina and Albert are dominated by sandy sediments, in contrast to the inner lake sediments which are clay dominated. Sandy deposits form a concentric wedge around the lake margins, with a maximum typical thickness of 1-2 m in near the original shoreline, and progressively thinning to 0 m over an average estimated distance of 1500 m. Clay-rich sediments progressively increase in thickness towards the centre of each lake.

At present, the upper sandy sediments are becoming increasingly exposed as a result of recent declines in lake water levels. Current predictions indicate that lake levels will approach -2 m AHD in Lake Alexandrina and -1 m AHD in Lake Albert within the next 1-2 years (DWLBC, 2009). Thus, the first exposed sediments around the edges of both lakes will be dominated by sands. Any further reductions in lake water levels will result in progressive exposure of more clay-rich sediments, as the beach width around each lake increases beyond approximately 1500 m.

Both sands and clays represent a potential ASS risk for the Lower Murray Lakes. The magnitude of the risk associated with each material type is dependant on a combination of factors, in particular, the total volume of sediment exposed, the sulfide content of those sediments (and variability of sulfide content with depth in the sediment profile), and the rate of sulfide oxidation, which varies with moisture content. Moisture content, in turn, will be strongly affected by sediment texture, lake water/groundwater levels, rainfall/evapotranspiration patterns and other effects such as lake water seiching and wave action.

The sandy ASS around the periphery of both lakes are considered to represent the most significant short-term AMD risk facing the Lower Murray Lakes. This is primarily based on the understanding that sandy sediments are likely to represent the majority of ASS exposed in both lakes over the next 1-2 years. While the clays typically contain significantly higher sulfides concentrations, and extend over a larger area than the sands, they do not necessarily pose an immediate risk to lake water quality as long as they remain inundated, or at least fully saturated. Furthermore, the tendency for clays to retain significant moisture, even following extended drying, will retard the rate of acidity generation. Clay-rich ASS will, however, become an increasing concern if lake water levels drop sufficiently to expose significant volumes of these sediments to oxidising conditions, over extended periods of time.

6.2 *Composition of sands vs. clays*

There is considerable variation in sediment composition, and thus acidity generation and neutralisation potential, with sediment texture (sands vs. clays) and depth in the lake sediments.

In the case of sandy sediments in Currency Creek, Lake Albert and Lake Alexandrina, average sulfide-sulfur contents range from around 0.01-0.1 wt% S in the upper layers to 0.1-0.9 wt% S at a depth of 1-2 m. These values correspond to Acid Producing Potential (APP) values of around 1-30 kg H₂SO₄ / tonne of sediment. With ANC values averaging around 2-8 kg H₂SO₄ / tonne in the upper layers and 10-50 kg H₂SO₄ / tonne at depth, average Net Acid Producing Potential (NAPP) values are generally negative in the sandy sediments.

In comparison, the clay-rich sediments of Currency Creek, Lake Albert and Lake Alexandrina, are characterised by average sulfide-sulfur contents of around 0.1-1.7 wt% S, corresponding to Acid Producing Potential (APP) values of around 4-50 kg H₂SO₄ / tonne of sediment. Significant ANC is also present in the clay-rich sediments, although generally not sufficient to balance the APP. Based on these static test results, the clay-rich sediments have the potential to become a significant long-term ASS risk in both lakes, if lake water levels recede to the point where clays become exposed for an extended period of time.

While the negative NAPP values in the sandy sediments are suggestive of relatively low ASS potential, recent impacts on water quality in Currency Creek, localised acidification of groundwater at Point Sturt and Campbell Park, as well as visual observation of localised zones of acid salt formation on the sandy beaches of Lake Albert and Lake Alexandrina, indicate otherwise. This emphasises the importance of understanding not only the static test results, or the potential for acidity generation and neutralisation in the lake sediments, but also the kinetic processes that determine the rate and extent of acidity generation and neutralisation in the

sediments. Indications from the acidity generation model results for Currency Creek (Section 7.1) are that ANC consumption of around 10% per year is realistic. Factors that limit the effectiveness of ANC include:

- The low solubility of carbonate minerals;
- The slow dissolution rate of carbonate minerals;
- The potential for armouring (surface passivation) of carbonate minerals with neutralisation precipitates;
- The rapid rate of surface infiltration through the unsaturated, acidic-salt rich sandy sediment profile; and
- The lack of vertical mixing (and slow rate of diffusion) of acidic groundwater within the sediment profile (beneath the water table).

6.3 Sulfide oxidation rates in sands vs. clays and relationship with moisture content

Overall, the laboratory measured sulfide oxidation rates for sands and clays of the Lower Murray Lakes compare well with oxidation rates for ASS reported in or calculated from the literature (and are consistent with field observations).

Laboratory measured sulfide oxidation rates in sandy sediments show a complex relationship with moisture content. Sulfide oxidation rates peak in sand samples with 15 wt% water content, and fall (at different rates) for samples with both lower and higher water contents. This unusual relationship has been identified in mine site materials by other researchers (Hollings et al., 2001).

The relationship between sulfide oxidation rate and moisture content in clay-rich sediments is more linear, with progressive decreases in oxidation rate as a function of increasing moisture content.

The peak sulfide oxidation rate was around 1.2 wt% of the available pyrite per day in sandy sediments (at 15 wt% water content) and 0.8 wt% pyrite per day in clays (at 23 wt% water content). At these rates, the majority of pyrite that is exposed to atmospheric oxygen would oxidise to produce H₂SO₄ acidity within around 3-4 months.

6.4 Groundwater flows and hydraulic gradients

Hydraulic conductivity values for lake sediment sands range from 0.09 to >30 m/day. Hydraulic conductivity values for the Bridgewater Formation calcareous sands range from 0.5 to >30 m/day. These values indicate that discharge from the sandy lake sediments has the potential to be relatively rapid, depending on the hydraulic gradient.

Groundwater levels in the lake sediments are often slightly lower than those in the underlying Bridgewater Formation, indicating that these horizons are at least locally hydraulically isolated, and that the latter is confined by a layer of low permeability sediments. The widespread exposure of Bridgewater Formation in the base of both lakes indicates that this formation is also locally unconfined. The hydraulic conductivity values indicate that discharge from the Bridgewater Formation sandy sediments has the potential to be relatively rapid, depending on the gradient and degree of confinement.

Groundwater levels rise rapidly in response to significant rainfall events at all sites. Rises of 30 cm are typical in response to 10-15 mm rainfall events. The significant rises in groundwater level relative to actual rainfall depth are believed to be due to the addition of infiltrating water to near-saturated sediments. These rises fall very rapidly over subsequent days, presumably as a result of near surface evapotranspiration. Since recording began in August, the periodic rises in groundwater levels due to rainfall are overlain on a background of progressively falling groundwater levels, even though lake water levels have remained largely static.

Groundwater levels, and therefore hydraulic gradients, in the lake sediments were highly variable over time. Much of this variation was associated with rainfall events and surface water ponding, with some evidence of seiche events at the Windmill location, over the last 3 months.

Overall, the hydraulic gradients have been relatively small at all monitoring locations, ranging from 0.001-0.002 in mid-November 2009. Thus, despite some of the high hydraulic conductivity values measured, there has been limited potential for groundwater migration as a result of the relatively low hydraulic gradients from groundwater in the lake sediments towards the surface water bodies.

Surface topography is likely to be a key influence on long-term groundwater levels. The hydraulic gradients reported above range from around 20% to 100% of the corresponding beach slope. Until further data become available, it has been conservatively assumed for acidity generation modelling purposes (Section 7) that hydraulic gradients represent 20% of the beach slope (consistent with the lower end of this range).

Clearer trends in groundwater levels are expected to emerge over the next few months as rainfall becomes less significant and lake water levels progressively recede. The rate of groundwater flow into the lake water has the potential to increase significantly as surface water levels decline and hydraulic gradients increase following substantial rainfall events (e.g. in autumn).

6.5 Sediment moisture profiles

A good correlation exists between groundwater levels and moisture content profiles in the lake sediments.

Sediment moisture data shows that a 0.3 m thick zone of essentially saturated sandy sediment, with a moisture content of around 40-50 vol% H₂O, exists above the piezometric surface at all sites examined. Above this zone, moisture contents progressively decrease with proximity to the ground surface, over a depth interval of around 0.3 m. The uppermost sediments are characterised by moisture contents of 2-6 vol% H₂O.

Over the last 3 months, groundwater levels have decreased by 0.5-0.9 m at all monitoring sites nearest the lake shores (Site 1 at each location). The sediments have remained effectively saturated at depths below 0.3 m during this time. However, it is expected that further lowering of groundwater levels over the next few months will result in a progressive desaturation of these sediments.

For acidity generation modelling purposes (Section 7) it has been conservatively assumed that moisture contents in the sandy sediments are directly related to the corresponding groundwater level on the same day. In other words, the delay in response of moisture content to a change in groundwater level has been assumed to be less than 24 hours.

Further monitoring data will assist in understanding the rate of sediment desaturation in response to falling groundwater levels.

6.6 Water quality in Currency Creek

Complete evaporation of surface water in Currency Creek over the 2008-09 summer period resulted in significant acidity generation within the unsaturated sandy creek sediments.

Multiple shallow excavations throughout Currency Creek indicated that the sediments are dominated by clay-rich material which is fringed by a thin wedge of sandy material toward the original shoreline. While the sandy fringe can be over 1 m deep on the northern margin of the Currency Creek tributary, clay-rich, sulfidic sediments are exposed at the surface for much of the central portion of the drainage line.

A significant rainfall event occurred over 2 days in late April – early May 2009, resulting in the ponding of acidic water in Currency Creek. As there were no inflows to the creek during this rainfall event, the ponded creek water was attributed to incident rainfall on the creek bed and surrounding sediments. The acidic nature of the ponded water could only be explained by significant and rapid flushing of acidic salts from within the sandy creek sediments.

The majority of the acidity load flushed into Currency Creek during this event was attributed to the peripheral wedge of sandy material, based on the following:

- Given the rapid and significant extent of ponding observed in the creek on 1 May 2009, infiltration must have occurred through relatively high hydraulic conductivity surface sediments (sands) and travelled laterally upon encountering an impermeable clay layer.
- While clays occupied a larger areal extent of the creek bed, these sediments remained largely saturated throughout the preceding months. Localised desiccation of clays had commenced on the lower margins of the creek, although the exposure of clays to oxidation (acidity generation) was limited to a thin veneer of material surrounding the clay peds in these areas.

The acidification of Currency Creek occurred despite the fact that most of the sandy sediments were characterised by negative NAPP values, particularly in the upper layers where desaturation would have been most significant. Groundwater quality data indicates that some degree of in-situ carbonate dissolution (ANC consumption) has occurred in response to acidity generation in the creek sediments. Nevertheless, the majority of the available ANC could not have been consumed. Reasons for the limited degree of ANC consumption were discussed above.

6.7 *Water quality in Lake Albert*

Based on surface water level data provided by DFW (DWLBC 2009), the water level in Lake Albert was below -0.4 m AHD for the first 4 months of 2009. The lowest level reached during this period was -0.59 m AHD (March 2009).

Despite the lake level falling to -0.59 m AHD in March 2009, there was no clear evidence of major acidity flux to the lake. This is supported by the following:

- There was no significant decrease in surface water pH, alkalinity or Cl:SO₄ ratio associated with decreasing lake water level.
- Groundwater levels in the sandy sediments would have been greater than -0.59 m AHD and sediments would not have been fully unsaturated above the groundwater table. For example, a beach width of around 400 m and hydraulic gradient of 0.001 would correspond to a groundwater level of -0.2 m AHD in sediments nearest the original shore. If sediments were effectively saturated over a 0.3 m thick zone above the groundwater level, then limited if any sulfide oxidation (acidity generation) would be expected.
- Any acidity generation would have been limited to the upper layers of unsaturated sediments which are characterised by relatively low sulfide-sulfur contents.
- The hydraulic gradient in the lake sediments would have been relatively flat (e.g. 0.001-0.002) based on data collected in November 2009 when surface water levels were around -0.5 m AHD. Therefore, there would have been limited potential for any acidity generated in the lake sediments to migrate laterally towards the lake water.
- Any acidity flux to the lake water was likely to be minor in comparison with the buffering capacity of the lake.

During the field monitoring program, the lake level decreased from -0.1 m AHD in August 2009 to around -0.65 m AHD in November 2009, based on groundwater level data. Thus, the lake level has generally remained higher than the minimum level of -0.59 m AHD recorded in March 2009.

The field monitoring data collected from August to November 2009 suggests that:

- Prior to the commencement of monitoring, some localised acidity generation had occurred within the upper lake sediments, as indicated by acidic groundwater observed in some piezometers (3 sites at Campbell Park).
- There has been no evidence of significant additional acidity generation within the lake sediments during the monitoring program.
- Acidity generated within the upper lake sediments has migrated vertically from sandy layers in the unsaturated zone to the groundwater via rainwater infiltration. However, there has been only limited vertical mixing / diffusion within the groundwater profile.
- There has been no significant lateral migration of acidity from the sediments towards the lake water, based on reasonably consistent water quality over time (at each site), despite significant chemical variations relative to other sites on the same transect. This is attributed to relatively low hydraulic gradients over the last 3 months, as well as the significant near-surface evapotranspiration water losses.
- Groundwater chemistry data shows that some degree of in-situ carbonate dissolution (ANC consumption) has occurred at both sites in Lake Albert. At Campbell Park this has clearly been related to acidity generation. However, ANC consumption has been insufficient to counter the acidity in groundwater at Campbell Park. This is despite indications that sandy lake sediments are generally NAPP negative.
- There is evidence of sulfide precipitation (bacterial sulfate reduction) within the upper sediments affected by acidity generation at Campbell Park (Sites 2-4) based on the progressive increases in pH and Cl:SO₄ ratios observed over the last 3 months.

The risk to surface water quality in Lake Albert could increase significantly as lake levels are forecast to decrease to unprecedented levels (-1.0 m AHD) over the next few months. The increased risk would result from:

- Exposure of larger volumes of ASS.
- Increased sulfide-sulfur content with depth in exposed sediments.

- Increased rate of groundwater flow through sandy sediments due to greater hydraulic gradients to lake water.

6.8 *Water quality in Lake Alexandrina*

Based on surface water level data provided by DFW (DWLBC 2009), the water level in Lake Alexandrina was below -0.9 m AHD over a 4 month period from March to June 2009. The lowest level reached during this period was -1.0 m AHD (April 2009).

Despite the lake level falling to -1.0 m AHD in April 2009, there was no clear evidence of major acidity flux to the lake. This is supported by the following:

- There was no significant decrease in surface water pH, alkalinity or Cl:SO₄ ratio associated with decreasing lake water level.
- Groundwater levels in the sediments would have been greater than -1.0 m AHD and sediments would not have been fully unsaturated above the groundwater table. For example, a beach width of around 400 m and hydraulic gradient of 0.001 would correspond to a groundwater level of -0.6 m AHD in sediments nearest the original shore. If sediments were effectively saturated over a 0.3 m thick zone above the groundwater level, then limited if any oxidation (acidity generation) would be expected.
- Any acidity generation would have been limited to the upper layers of unsaturated sediments which are characterised by relatively low sulfide-sulfur contents.
- The hydraulic gradient in the lake sediments would have been relatively flat (e.g. 0.001-0.002) based on data collected in November 2009 when surface water levels were around -0.9 m AHD. Therefore, there would have been limited potential for any acidity generated in the lake sediments to migrate laterally toward the lake water.
- Any acidity flux to the lake water was likely to be minor in comparison with the buffering capacity of the lake.

During the field monitoring program, the lake level remained around -0.9 m AHD. Thus, the lake level has remained higher than the minimum level of -1.0 m AHD recorded in April 2009.

The field monitoring data collected from August to November 2009 suggests that:

- Prior to the commencement of monitoring, some localised acidity generation had occurred within the upper lake sediments, as indicated by acidic groundwater observed in some piezometers (2 sites at Point Sturt).
- There has been no evidence of significant additional acidity generation within the lake sediments during the monitoring program.
- Acidity generated within the upper lake sediments has migrated vertically from sandy layers in the unsaturated zone to the groundwater via rainwater infiltration. However, there has been limited vertical mixing / diffusion within the groundwater profile.
- There has been no significant lateral migration of acidity from the sediments towards the lake water, based on reasonably consistent water quality over time (at each site), despite significant chemical variations relative to other sites on the same transect. This is attributed to relatively low hydraulic gradients over the last 3 months, as well as the significant near-surface evapotranspiration water losses.
- Groundwater chemistry data shows that some degree of in-situ carbonate dissolution (ANC consumption) has occurred at both sites in Lake Albert. At Campbell Park this has clearly been related to acidity generation. However, ANC consumption has been insufficient to counter the acidity in groundwater at Campbell Park. This is despite indications that sandy lake sediments are generally NAPP negative.
- There is evidence of sulfide precipitation (bacterial sulfate reduction) within the upper sediments affected by acidity generation at Point Sturt (Sites 1-2) based on the progressive increases in pH and Cl:SO₄ ratios observed over the last 3 months.

The risk to surface water quality could increase significantly as lake levels are forecast to decrease to unprecedented levels (below -2.0 m AHD) over the next 1-2 years. The increased risk would result from:

- Exposure of larger volumes of ASS.
- Increased sulfide-sulfur content with depth in exposed sediments.

- Increased rate of groundwater flow through sandy sediments due to greater hydraulic gradients to lake water.

7 Acidity Generation and Flux Rate Modelling

This section provides an assessment of the total potential acidity generation from Currency Creek, Lake Albert and Lake Alexandrina sediments and modelled acidity generation rates in each region over time. The modelled acidity generation rates incorporate key findings from the laboratory testwork program and field monitoring program, as detailed in Sections 5 and 6.

Two-dimensional hydrogeological model outputs have been used to convert modelled acidity generation rates (within the lake sediments) to acidity flux rates (from the lake sediments to the surface water bodies) for Lake Albert and Lake Alexandrina.

This assessment utilises available field data and observations at Currency Creek over a 1 year period (Section 7.1) as a basis for model verification. Application of the acidity generation and flux rate models to Lake Albert and Lake Alexandrina are discussed in Sections 7.2 and 7.3, respectively. Note that the modelling results discussed in this section are focussed on the upper 1-2 m of lake sediments (predominantly sands) that are likely to become exposed as lake levels recede over the next 1-2 years.

7.1 Currency Creek

7.1.1 Total Potential Acidity Generation

The total potential acidity generation from Currency Creek is estimated to be approximately 45,000 tonnes H_2SO_4 , based on the following assumptions:

- Average depth of exposed sediments = 0.5 m.
- Area of dry creek bed = 6.4 km² (640 ha).
- Average sulfide-sulfur content of sand = 0.27 wt% S. This estimate is based on available sulfur content data (excluding outliers) for all sand samples from -0.4 m AHD to +0.6 m AHD analysed by ALS (UCC-P1, LCC-P2, UCC-P3) and Fitzpatrick et al (2009; other sites in Currency Creek).
- Bulk density of exposed sediments = 1.7 t/m³.

The proportion of this total potential acidity generation (45,000 tonnes H_2SO_4) actually released to the creek over 1 year is discussed below. The discussion applies to typical Currency Creek flows to Goolwa Channel as occurred prior to the recent installation of flow regulators in the Goolwa Channel (Clayton) and in Currency Creek itself in August-September 2009. Acidity flux rates in future will remain very low as long as the regulators maintain water levels such that the creek sediments remain permanently saturated.

Field acidity measurements and estimates of water volume (surface water and pore water) in Currency Creek in May 2009 indicated a total acidity flux of 500-700 tonnes H_2SO_4 . This was assumed to correspond to the majority of 1 year of acidity flux from all creek sediments, given that the creek had dried during the 2008-2009 summer and additional acidity generation for the remainder of the year (after May 2009) was likely to be minor as the sediments remained largely saturated throughout the wet season.

Approximately 1000 tonnes CaCO_3 (equivalent to ~1000 tonnes H_2SO_4 acidity) was added to Currency Creek between April and June 2009 to account for acidity generation associated with the creek drying over the preceding summer. This was assumed to correspond to the majority of 1 year of acidity generation from all creek sediments (as described above) and is broadly consistent with the acidity load of 500-700 tonnes H_2SO_4 predicted in May 2009.

7.1.2 Acidity Generation and Flux Rates

To more accurately quantify the annual acidity load from Currency Creek, an acidity generation finite element model with daily time step was developed, utilising the following assumptions and input data (see Attachment N for further information):

- The creek water depth decreased from 1 m to 0 m over 3 months (November 2008 – January 2009) and remained dry for a further 3 months (February – April 2009) before returning to 1 m by the end of October 2009. The water level was assumed to decrease linearly from November 2008 to January 2009 at a rate of approximately 0.0109 m/day (1 m over 92 days) and increase linearly from May 2009 to October 2009 at a rate of approximately 0.0054 m/day (1 m over 184 days).
- Sediments exposed as a result of the 1 m decrease in creek water depth were divided into wedges, each with a 100 m shoreline length, 1 m depth and 350 m width (perpendicular to the shoreline). Each wedge was further sub-divided into 55 cells of 0.1 m height, 100 m length and 35 m width (cell volume 350 m³). Thus, each wedge corresponded to a volume of 19,250 m³.

- The slope of exposed sediments was assumed to be approximately 0.0029 (= 1 / 350).
- The total perimeter of exposed sediments was assumed to be approximately 12 km.
- Groundwater levels were assumed to follow the same trend as surface water levels. The hydraulic gradient was assumed to be constant and equal to 20% of the slope of the exposed sediments.
- Sediments exposed as a result of the 1 m decrease in creek water depth were assumed to comprise sands with average sulfide-sulfur content and ANC (wt% CaCO₃) varying with elevation as outlined in Table 8. These estimates are based on available sulfur content and ANC data (excluding outliers) for all sand samples analysed by ALS (UCC-P1, LCC-P2, UCC-P3) and Fitzpatrick et al (2009; other sites in Currency Creek).
- Sediments were assumed to be saturated to a depth of 0.3 m above the groundwater level, with moisture contents decreasing linearly from 46 vol% H₂O (representing fully saturated sands) to 4 vol% H₂O in the overlying 0.3 m of sediments, and remaining at 4 vol% H₂O in the upper layers. These assumptions are consistent with measured moisture contents at Point Sturt, Campbell Park and the Windmill location in mid-November 2009.
- Moisture contents for each cell were converted from volumetric units (ψ , vol% H₂O) to gravimetric units (u , wt% H₂O) according to the following equation based on data collected during the laboratory testwork program:
 - $u \text{ (wt\% H}_2\text{O)} = 0.5109 * \psi \text{ (vol\% H}_2\text{O)}$
- Pyrite oxidation rates for each cell were calculated using the following relationship established for sands during the laboratory testwork program:
 - Pyrite oxidation rate (g FeS₂ / day) = $-4.3488 u^3 + 0.6565 u^2 + 0.076 u + 0.0008$
- The mass of pyrite in each cell was determined from wt% S data. These values were combined with calculated pyrite oxidation rates to determine the mass of pyrite oxidised within each cell and the mass of pyrite remaining after oxidation, on a daily basis.
- Estimates of the mass of pyrite oxidised within each cell were converted into acidity generation estimates (kg H₂SO₄) according to the Reaction 4 (see Section 1.2).
- The mass of CaCO₃ in each cell was determined from ANC data. The ANC within each cell available for acidity neutralisation was assumed to be 10 wt% CaCO₃ per year¹⁴. ANC availability was calculated on a daily basis, according to the amount of ANC remaining at each time step, and converted from kg CaCO₃ to kg H₂SO₄ equivalent units. Within each cell, the actual ANC consumed (kg H₂SO₄ equivalent) was not allowed to exceed the acidity generation (kg H₂SO₄).
- The process of sulfide precipitation (bacterial sulfate reduction) within the creek sediments could not be accurately quantified but was assumed to be minor, on the basis of field and laboratory results, and therefore not incorporated into the model.
- Estimates of net acidity generation (kg H₂SO₄) were obtained by subtracting the mass of ANC consumed from the acidity generation within each cell, on a daily basis.

Preliminary acidity generation modelling results for Currency Creek between 1 November 2008 and 31 October 2009 are shown in Figure 69. Key observations from Figure 69 include:

- Total acidity generation over 1 year (1 November 2008 – 31 October 2009) is estimated at ~1400 tonnes H₂SO₄ assuming no ANC consumption, or ~1000 tonnes H₂SO₄ assuming 10% ANC consumption per year. Thus, ANC consumption of only 10 wt% CaCO₃ / year would have reduced the total acidity flux to Currency Creek by ~30%.
- No significant acidity generation occurred during the initial 2 months (1 November 2008 – 31 December 2008) despite the water depth decreasing from 1 m to ~0.35 m.
- By the end of January 2009 the creek had fully dried and total acidity generation is estimated at ~190 tonnes H₂SO₄ assuming no ANC consumption, or ~150 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~15% of the annual acidity load.
- Prior to the autumnal flush (1 May 2009), the total acidity generation is estimated at ~1200 tonnes H₂SO₄ assuming no ANC consumption, or ~900 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~85-90% of the annual acidity load. Most of this acidity was released into Currency Creek during the autumnal flush (1-2 days) in early May 2009.

¹⁴ ANC consumption rates of 5%, 20% and 30% per year were also modelled for comparison.

- The total acidity generation assuming 10% ANC consumption per year by 1 May 2009 (~900 tonnes H₂SO₄) was consistent with initial predictions of acidity flux rates based on (i) surface water and pore water volumes and acidities, and (ii) limestone addition requirements.
- The rate of acidity generation decreased significantly after May 2009 due to pyrite consumption in the unsaturated zone and rewetting of the creek bed via natural inflows. Acidity generation ceased altogether by the end of July 2009.

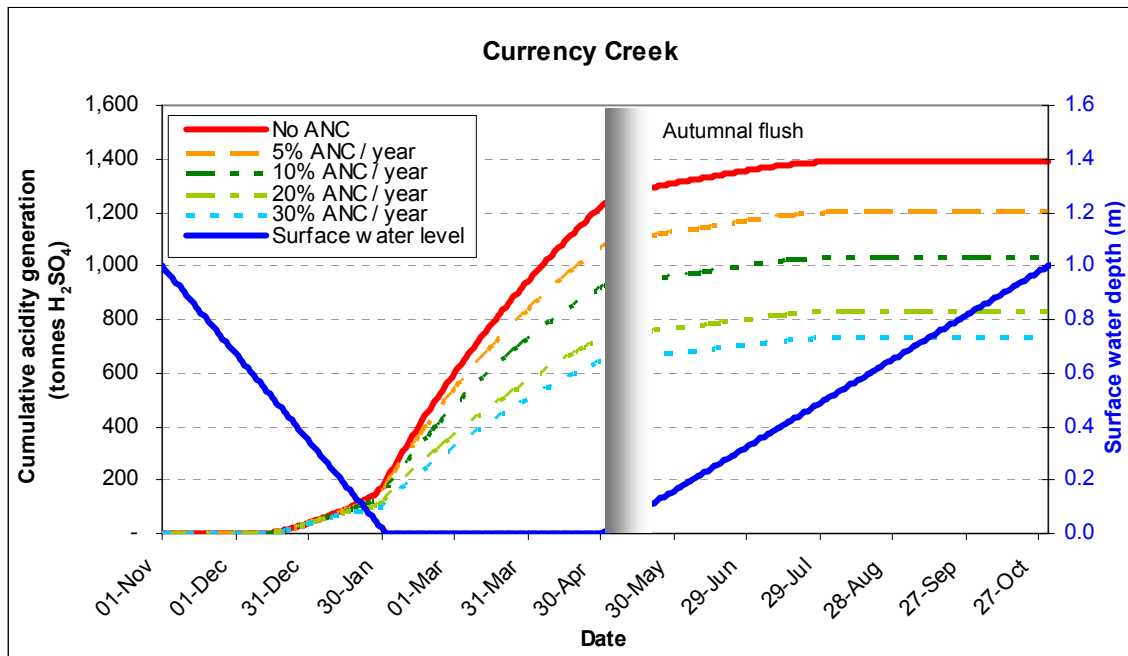


Figure 69. Preliminary estimates of cumulative acidity generation in Currency Creek from 1 November 2008 to 31 October 2009. When the water depth reached 0 m (31 January 2009) ~190 tonnes H₂SO₄ acidity had been generated assuming no ANC consumption (or ~150 tonnes H₂SO₄ if ANC consumption = 10 wt% CaCO₃ / year). By 1 May 2009, cumulative acidity generation increased to ~1200 tonnes H₂SO₄ (~85-90% of annual load) assuming no ANC consumption (or ~900 tonnes H₂SO₄ if ANC consumption = 10 wt% CaCO₃ / year).

7.2 Lake Albert

7.2.1 Total Potential Acidity Generation

The total potential acidity generation from Lake Albert is estimated to be approximately 210,000 tonnes H₂SO₄, based on the following assumptions:

- Average depth of exposed sediments = 0.550 m (total depth of exposed sediments = 1.100 m, associated with lake levels decreasing to -1.000 m AHD; ground level assumed to be +0.100 m AHD).
- Area of exposed sediments associated with lake levels decreasing to -1.000 m AHD = 75 km² (7459 ha).
- Average sulfide-sulfur content of sand = 0.10 wt% S. This estimate is based on available sulfur content data (excluding outliers) for all sand samples above -1.0 m AHD analysed by ALS (Campbell Park and Windmill locations) and Fitzpatrick et al (2009; other sites in Lake Albert).
- Bulk density of exposed sediments = 1.7 t/m³.

7.2.2 Acidity Generation Rate

Key assumptions and inputs used in modelling acidity generation rates from Lake Albert sediments are listed below (see Attachment N for further information):

- The lake water level was assumed to decrease from -0.163 m AHD to -1.000 m AHD over 5-6 months and remain at -1.000 m AHD thereafter (DWLBC, 2009).

- Lake sediments were divided into wedges, each with a 100 m shoreline length, 1.1 m depth and 1500 m width (perpendicular to the shoreline). Each wedge was further sub-divided into 66 cells of 0.1 m height, 100 m length and ~140 m width (cell volume ~1400 m³). Thus, each wedge corresponded to a volume of 90,000 m³.
- The slope of exposed sediments was assumed to be approximately 0.0007 (= 1.1 / 1500).
- The total perimeter of exposed sediments was estimated to be approximately 50 km.
- Groundwater levels were assumed to follow the same trend as surface water levels. The hydraulic gradient was assumed to be constant and equal to 20% of the slope of the exposed sediments.
- Sediments exposed as a result of decreasing lake water levels were assumed to comprise sands with an average sulfide-sulfur content and ANC (wt% CaCO₃) varying with elevation as outlined in Table 13, with the exception of the upper 0.3 m of exposed sediments that were assumed to contain no sulfur or ANC. These estimates are based on available sulfur content and ANC data (excluding outliers) for all sand samples analysed by ALS (Campbell Park and Windmill locations) and Fitzpatrick et al (2009; other sites in Lake Albert).
- Sediments were assumed to be saturated to a depth of 0.3 m above the groundwater level, with moisture contents decreasing linearly from 46 vol% H₂O (representing fully saturated sands) to 5 vol% H₂O in the overlying 0.3 m of sediments, and remaining at 5 vol% H₂O in the upper layers. These assumptions are consistent with measured moisture contents at Campbell Park and the Windmill location in mid-November 2009.
- Moisture contents for each cell were converted from volumetric units (ψ , vol% H₂O) to gravimetric units (u , wt% H₂O) according to the following equation based on data collected during the laboratory testwork program:
 - u (wt% H₂O) = 0.5109 * ψ (vol% H₂O)
- Pyrite oxidation rates for each cell were calculated using the following relationship established for sands during the laboratory testwork program:
 - Pyrite oxidation rate (g FeS₂ / day) = -4.3488 u^3 + 0.6565 u^2 + 0.076 u + 0.0008
- The mass of pyrite in each cell was determined from wt% S data. These values were combined with calculated pyrite oxidation rates to determine the mass of pyrite oxidised within each cell and the mass of pyrite remaining after oxidation, on a daily basis.
- Estimates of the mass of pyrite oxidised within each cell were converted into acidity generation estimates (kg H₂SO₄) according to the Reaction 4 (see Section 1.2).
- The mass of CaCO₃ in each cell was determined from ANC data. The ANC within each cell available for acidity neutralisation was assumed to be 10 wt% CaCO₃ per year. ANC availability was calculated on a daily basis, according to the amount of ANC remaining at each time step, and converted from kg CaCO₃ to kg H₂SO₄ equivalent units. Within each cell, the actual ANC consumed (kg H₂SO₄ equivalent) was not allowed to exceed the acidity generation (kg H₂SO₄).
- The process of sulfide precipitation (bacterial sulfate reduction) within the lake sediments could not be accurately quantified but was assumed to be minor, on the basis of field and laboratory results, and therefore not incorporated into the model.
- Estimates of net acidity generation (kg H₂SO₄) were obtained by subtracting the mass of ANC consumed from the acidity generation within each cell, on a daily basis.

Preliminary acidity generation modelling results for Lake Albert over the 22 month period modelled are shown in Figure 70. Key observations from Figure 70 include:

- Total potential acidity generation over 22 months (from September 1st, 2009) is estimated at ~50,000 tonnes H₂SO₄ assuming no ANC consumption, or ~38,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. Thus, ANC consumption of 10 wt% CaCO₃ / year could reduce the total acidity flux to Lake Albert by ~25%.
- No significant acidity generation is expected during the initial 4 months despite predictions that the water depth decreasing from -0.163 m AHD to -0.682 m AHD.
- After 5-6 months when the surface water level is forecast to reach -1.0 m AHD (DWLBC, 2009) total acidity generation is estimated at ~9,800 tonnes H₂SO₄ assuming no ANC consumption, or ~9,200 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~20-25% of the acidity load after 22 months.

- Prior to the first autumnal flush, the total acidity generation is estimated at ~28,000 tonnes H₂SO₄ assuming no ANC consumption, or ~25,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~60% of the acidity load after 22 months.
- The rate of acidity generation is expected to decrease significantly after 9 months due to pyrite exhaustion in the unsaturated zone.
- The preferred management strategy for preventing acidity generation (as described above), from a technical perspective, would be to control lake water levels to maintain sulfidic sediments in a saturated state.
- If lake water cannot be maintained at sufficient levels to keep sulfidic sediments saturated, sub-surface barriers have the potential to prevent the formation of the majority of acidity generated, if they are installed within the initial 5 months. Barrier installation could be delayed if water levels were maintained at -0.9 m AHD, for example, although this would require additional water pumping from Lake Alexandrina.
- Sub-surface barriers would facilitate in-situ neutralisation (ANC consumption) and sulfide precipitation to some degree by decreasing groundwater migration rates to the lake. ANC consumption could be further improved by active / passive mixing of deeper alkaline groundwater with acidic groundwater in the overlying sediments behind the barriers.
- Water treatment by limestone addition would also be a feasible option for addressing the acidity loads shown in Figure 70. Up to ~50,000 tonnes limestone addition would be required on a one-off basis, with water levels maintained at -1.0 m AHD.
- Another option that would significantly decrease acidity generation would be to maintain the surface water level at a higher elevation. A suitable level (e.g. around -0.8 m AHD) could be determined at which sub-surface barrier installation or limestone addition would not be required. This would require additional water pumping from Lake Alexandrina.

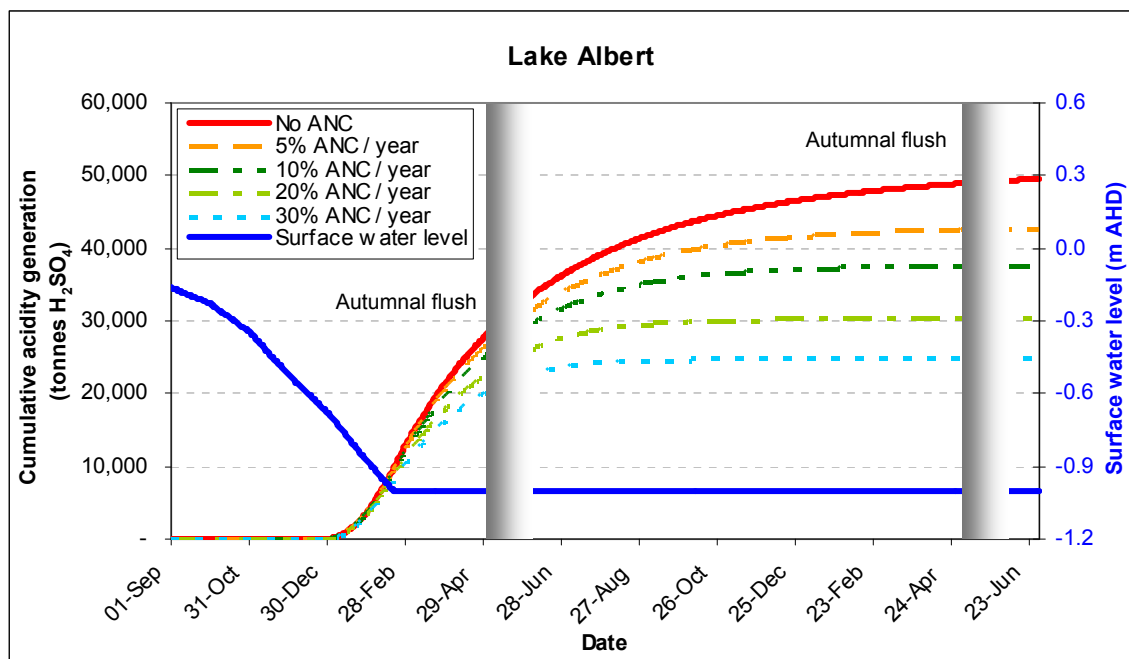


Figure 70. Preliminary estimates of cumulative acidity generation and flux from Lake Albert sediments over 22 months (from September 1st, 2009).

7.2.3 Acidity Flux Rate

As noted in Section 7.2.2, prior to the first autumnal flush, the total acidity generation is estimated at ~28,000 tonnes H₂SO₄ assuming no ANC consumption, or ~25,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~60% of the acidity load after 22 months. An additional ~22,000 tonnes H₂SO₄ acidity is estimated to be generated prior to the second autumnal flush (or ~13,000 tonnes H₂SO₄ acidity assuming no ANC consumption).

The likely rate and duration of this acidity release (flux) has been estimated for a range of lake water level, hydraulic conductivity and acidity concentration scenarios, based on hydrogeological modelling conducted by Coletti and Hipsey (2010). The scenario descriptions and estimated rate and duration of acidity release are presented in Table 25.

As shown in Table 25, the acidity flux event associated with the first autumnal flush could have a duration ranging from 2-3 months (77 days) to several years (10,000 days). The subsequent acidity flux event is estimated to occur in approximately half this time (40-5,200 days), due to the correspondingly lower tonnage of acidity generation. Based on the observed acidity flux event at Currency Creek in 2009, the duration of future acidity flux events in Lake Albert are likely to be at the lower end of the scale indicated here, i.e. closer to 2-3 months for the first acidity flux event and 1-2 months for the second event. These estimates correspond to the lower minimum lake water level (-1.0 m AHD), hydraulic conductivity of 10 m/day and acidity values of 10,000 mg/L CaCO₃.

Some of the acidity flux associated with the first autumnal flush in Lake Albert will be neutralised by the available soluble alkalinity in the water column.

Table 25. Estimated duration of acidity flux events associated with autumnal flushing – Lake Albert.

Minimum lake water level (m AHD)	Hydraulic conductivity (m/day)*	Seepage flow rate (L/day)		Acidity (mg/L CaCO ₃) [^]	Estimated acidity flux rate (tonnes H ₂ SO ₄ / day)	Estimated duration of acidity flux event (days)	
		m ³ /m/day	L/day [#]			First autumnal flush (25,000 t H ₂ SO ₄) ^{**}	Second autumnal flush (13,000 t H ₂ SO ₄) ^{**}
-0.1	1	0.05	2500000	1,000	3	10000	5200
				5,000	13	2000	1040
				10,000	25	1000	520
	5.5	0.273	13625000	1,000	14	1835	954
				5,000	68	367	191
				10,000	136	183	95
	10	0.500	25000000	1,000	25	1000	520
				5,000	125	200	104
				10,000	250	100	52
-1	1	0.065	3250000	1,000	3	7692	4000
				5,000	16	1538	800
				10,000	33	769	400
	5.5	0.354	17712500	1,000	18	1411	734
				5,000	89	282	147
				10,000	177	141	73
	10	0.650	32500000	1,000	33	769	400
				5,000	163	154	80
				10,000	325	77	40

* Lower and middle estimates based on low and average hydraulic conductivity data for sandy sediments in Lake Albert and Lake Alexandrina. Upper limit based on likely maximum hydraulic conductivity throughout the lakes.

Assumes lake perimeter of 50 km at the minimum lake water level.

[^] Lower limit based on observed groundwater quality in lake sediments (August-November 2009). Middle estimate based on observations during Currency Creek acidity flux event in 2009. Upper limit based on conservative estimate of likely maximum acidity values.

** Assumes 10% ANC consumption per year.

7.3 Lake Alexandrina

7.3.1 Total Potential Acidity Generation

The total potential acidity generation from Lake Alexandrina is estimated to be approximately 900,000 tonnes H₂SO₄, based on the following assumptions:

- Average depth of exposed sediments = 1 m (total depth of exposed sediments = 2 m, associated with lake levels decreasing from to -2.043 m AHD; ground level assumed to be 0 m AHD).
- Area of exposed sediments associated with lake levels decreasing to -2.043 m AHD = 300 km² (30,000 ha).
- Average sulfide-sulfur content of sand above -2.0 m AHD = 0.06 wt% S. This estimate is based on available sulfur content data (excluding outliers) for all sand samples analysed by ALS (Point Sturt) and Fitzpatrick et al (2009; other sites in Lake Alexandrina).
- Bulk density of exposed sediments = 1.7 t/m³.

7.1.2 Acidity Generation Rate

Key assumptions and inputs used in modelling acidity generation rates from Lake Alexandrina sediments are listed below (see Attachment N for further information):

- The lake water level is assumed to decrease from -0.776 m AHD to -1.400 m AHD over 6 months, remain at -1.400 m AHD for 1 month, increase to -1.299 m AHD over 4 months and then decrease to -2.043 m AHD over 8 months (DWLBC, 2009).
- Lake sediments were divided into wedges, each with a 100 m shoreline length, 2.0 m depth and 1500 m width (perpendicular to the shoreline). Each wedge was further sub-divided into 210 cells of 0.1 m height, 100 m length and 75 m width (cell volume ~750 m³). Thus, each wedge corresponded to a volume of 157,500 m³.
- The slope of exposed sediments was assumed to be approximately 0.0013 (= 2.0 / 1500).
- The total perimeter of exposed sediments was assumed to be approximately 120 km.
- Groundwater levels were assumed to follow the same trend as surface water levels. The hydraulic gradient was assumed to be constant and equal to 20% of the slope of the exposed sediments.
- Sediments exposed as a result of decreasing lake water levels were assumed to comprise sands with an average sulfide-sulfur content and ANC (wt% CaCO₃) varying with elevation as outlined in Table 10, with the exception of the upper 0.3 m of exposed sediments that were assumed to contain no sulfur or ANC. These estimates are based on available sulfur content and ANC data (excluding outliers) for all sand samples analysed by ALS (Point Sturt) and Fitzpatrick et al (2009; other sites in Lake Alexandrina), as detailed in Table 16.
- Sediments were assumed to be saturated to a depth of 0.3 m above the groundwater level, with moisture contents decreasing linearly from 46 vol% H₂O (representing fully saturated sands) to 2 vol% H₂O in the overlying 0.3 m of sediments, and remaining at 2 vol% H₂O in the upper layers. These assumptions are consistent with measured moisture contents at Point Sturt in mid-November 2009.
- Moisture contents for each cell were converted from volumetric units (ψ , vol% H₂O) to gravimetric units (u , wt% H₂O) according the following equation based on data collected during the laboratory testwork program:
 - u (wt% H₂O) = 0.5109 * ψ (vol% H₂O)
- Pyrite oxidation rates for each cell were calculated using the following relationship established for sands during the laboratory testwork program:
 - Pyrite oxidation rate (g FeS₂ / day) = -4.3488 u^3 + 0.6565 u^2 + 0.076 u + 0.0008
- The mass of pyrite in each cell was determined from wt% S data. These values were combined with calculated pyrite oxidation rates to determine the mass of pyrite oxidised within each cell and the mass of pyrite remaining after oxidation, on a daily basis.
- Estimates of the mass of pyrite oxidised within each cell were converted into acidity generation estimates (kg H₂SO₄) according to the Reaction 4 (see Section 1.2).
- The mass of CaCO₃ in each cell was determined from ANC data. The ANC within each cell available for acidity neutralisation was assumed to be 10 wt% CaCO₃ per year. ANC availability was calculated

on a daily basis, according to the amount of ANC remaining at each time step, and converted from kg CaCO₃ to kg H₂SO₄ equivalent units. Within each cell, the actual ANC consumed (kg H₂SO₄ equivalent) was not allowed to exceed the acidity generation (kg H₂SO₄).

- The process of sulfide precipitation (bacterial sulfate reduction) within the lake sediments could not be accurately quantified but was assumed to be minor, on the basis of field and laboratory results, and therefore not incorporated into the model.
- Estimates of net acidity generation (kg H₂SO₄) were obtained by subtracting the mass of ANC consumed from the acidity generation within each cell, on a daily basis.

Preliminary acidity generation modelling results for Lake Alexandrina over the 22 month period modelled are shown in Figure 71. Key observations from Figure 71 include:

- Total potential acidity generation over 22 months (from September 1st, 2009) is estimated at ~180,000 tonnes H₂SO₄ assuming no ANC consumption, or ~115,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. Thus, ANC consumption of 10 wt% CaCO₃ / year could reduce the total acidity flux to Lake Alexandrina by ~35%.
- As the water level in Lake Alexandrina decreases from -0.776 m AHD (1 September 2009) to -1.4 m AHD (2 March 2010), acidity generation is estimated at 24,000 tonnes H₂SO₄ assuming no ANC consumption, or ~18,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~10-15% of the acidity load after 22 months.
- Prior to the first autumnal flush, the total acidity generation is estimated at ~40,000 tonnes H₂SO₄ assuming no ANC consumption, or ~28,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~20-25% of the acidity load after 22 months.
- Relatively minor acidity generation is expected during the 6 month period following the initial autumnal flush, due to pyrite consumption in the unsaturated zone and relatively stable lake water levels over this period.
- The rate of acidity generation has the potential to increase substantially after 15 months due to water levels decreasing to -2.043 m AHD. Key factors that may limit the acidity generation and release into Lake Alexandrina are the hydraulic gradient and degree of ANC consumption within the lake sediments.
- The preferred management strategy for preventing acidity generation (as described above), from a technical perspective, would be to control lake water levels to maintain sulfidic sediments in a saturated state.
- If lake water cannot be maintained at sufficient levels to keep sulfidic sediments saturated, sub-surface barriers in sandy sediments may be feasible and could limit acidity generation and enhance ANC consumption. If ANC consumption rates can be raised to 20% per year, total acidity generation after 22 months could be lowered to ~85,000 tonnes H₂SO₄.

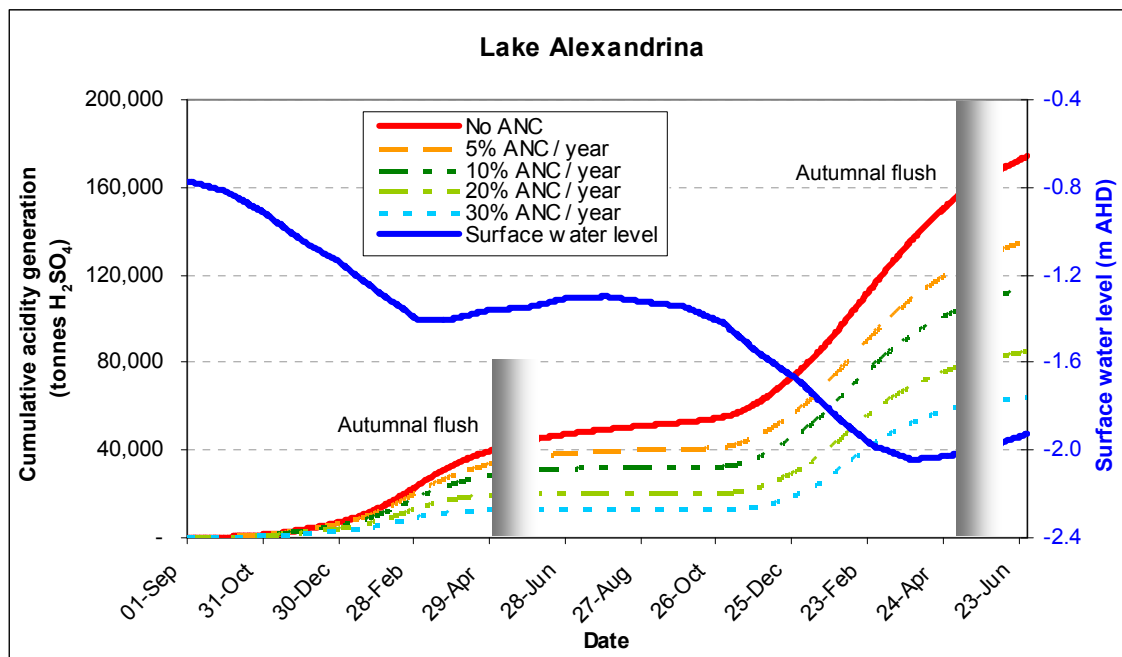


Figure 71. Preliminary estimates of cumulative acidity generation and flux from Lake Alexandrina sediments over 22 months (from September 1st, 2009).

7.3.3 Acidity Flux Rate

As noted in Section 7.3.2, prior to the first autumnal flush, the total acidity generation is estimated at ~40,000 tonnes H₂SO₄ assuming no ANC consumption, or ~28,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~20-25% of the acidity load after 22 months. An additional ~140,000 tonnes H₂SO₄ acidity is estimated to be generated prior to the second autumnal flush (or ~87,000 tonnes H₂SO₄ acidity assuming no ANC consumption).

The likely rate and duration of this acidity release (flux) has been estimated for a range of lake water level, hydraulic conductivity and acidity concentration scenarios, based on hydrogeological modelling conducted by Coletti and Hipsey (2010). The scenario descriptions and estimated rate and duration of acidity release are presented in Table 26.

As shown in Table 26, the acidity flux event associated with the first autumnal flush could have a duration ranging from 1-2 months (36 days) to several years (4,700 days). The subsequent acidity flux event is estimated to occur over approximately three times this duration (112-14,500 days), due to the correspondingly higher tonnage of acidity generation. Based on the observed acidity flux event at Currency Creek in 2009, the duration of future acidity flux events in Lake Alexandrina are likely to be at the lower end of the scale indicated here, ie. closer to 1-2 months for the first acidity flux event and 3-4 months for the second event. These estimates correspond to the lower minimum lake water level (-1.0 m AHD), hydraulic conductivity of 10 m/day and acidity values of 10,000 mg/L CaCO₃.

While there is likely to be sufficient soluble alkalinity in Lake Alexandrina to neutralise the acidity associated with the first autumnal flush event, less alkalinity would remain for future buffering.

It may be possible to cost-effectively add large tonnages of limestone at Wellington from barges to replenish the alkalinity lost during the second autumnal flush within Lake Alexandrina.

Table 26. Estimated duration of acidity flux events associated with autumnal flushing – Lake Alexandrina.

Minimum lake water level (m AHD)	Hydraulic conductivity (m/day)*	Seepage flow rate (L/day)		Acidity (mg/L CaCO ₃)^	Estimated acidity flux rate (tonnes H ₂ SO ₄ / day)	Estimated duration of acidity flux event (days)	
		m ³ /m/day	L/day#			First autumnal flush (28,000 t H ₂ SO ₄)**	Second autumnal flush (87,000 t H ₂ SO ₄)**
-0.1	1	0.05	6000000	1,000	6	4667	14500
				5,000	30	933	2900
				10,000	60	467	1450
	5.5	0.2725	32700000	1,000	33	856	2661
				5,000	164	171	532
				10,000	327	86	266
	10	0.5	60000000	1,000	60	467	1450
				5,000	300	93	290
				10,000	600	47	145
-1	1	0.065	7800000	1,000	8	3590	11154
				5,000	39	718	2231
				10,000	78	359	1115
	5.5	0.354	42510000	1,000	43	659	2047
				5,000	213	132	409
				10,000	425	66	205
	10	0.650	78000000	1,000	78	359	1115
				5,000	390	72	223
				10,000	780	36	112

* Lower and middle estimates based on low and average hydraulic conductivity data for sandy sediments in Lake Albert and Lake Alexandrina. Upper limit based on likely maximum hydraulic conductivity throughout the lakes.

Assumes lake perimeter of 120 km at the minimum lake water level.

^ Lower limit based on observed groundwater quality in lake sediments (August-November 2009). Middle estimate based on observations during Currency Creek acidity flux event in 2009. Upper limit based on conservative estimate of likely maximum acidity values.

** Assumes 10% ANC consumption per year.

8 Implications for ASS management in the Lower Murray Lakes

8.1 *Currency Creek*

Maintenance of permanently saturated sediments throughout the year will ensure that no significant acidity generation occurs in Currency Creek. This can be achieved by natural creek flows in the wet season, supplemented by pumping from Lake Alexandrina via the Clayton flow regulator.

This also applies to Finniss River and the Goolwa Channel.

8.2 *Lake Albert*

The preferred management strategy for preventing acidity generation, from a technical perspective, would be to control lake water levels to maintain sulfidic sediments in a saturated state. If lake water cannot be maintained at sufficient levels to keep sulfidic sediments saturated, additional management options may need to be considered.

A summary of management options for the scenario modelled in Section 7, and a range of alternative lake water level scenarios, is provided in Table 27.

8.3 *Lake Alexandrina*

The preferred management strategy for preventing acidity generation, from a technical perspective, would be to control lake water levels to maintain sulfidic sediments in a saturated state. If lake water cannot be maintained at sufficient levels to keep sulfidic sediments saturated, additional management options may need to be considered.

A summary of management options for the scenario modelled in Section 7, and a range of alternative lake water level scenarios, is provided in Table 28.

Table 27. ASS management options for Lake Albert.

Water level scenario	ASS management option	Information required
Maintenance of surface water level at -1.0 m AHD (as modelled in Section 7).	Pumping of ~35 GL/year from Lake Alexandrina required (DWLBC, 2009) and Sub-surface barrier installation in sandy sediments situated above -1.0 m AHD or One-off treatment of up to ~50,000 tonnes CaCO ₃ .	Refinement of existing acidity flux model is required to confirm suitability of management option. An accurate geological map of the distribution of the uppermost sand and clay is required for detailed barrier design and installation. Cost estimate and comparison between options.
Ephemeral lake (no pumping from Lake Alexandrina).	Sub-surface barrier installation in sandy sediments situated above -1.0 m AHD and Low permeability (eg. CaCO ₃) terrace installation on clay sediments situated below -1.0 m AHD; utilisation of Tertiary limestone aquifer to maintain clays in a saturated but not inundated state.	Acidity flux model development required to assess suitability of management option. Water balance model required to estimate extent and duration of water ponding. An accurate geological map of the distribution of the uppermost sand and clay is required for detailed barrier design and installation. Quantify the de-saturation / desiccation rate of sands and clays in order to determine the relative risk of acidity release from these materials. This information will also assist with quantifying groundwater requirements for maintaining saturation of clays. Quantify the availability of groundwater from the Tertiary limestone for assisting saturation. Cost estimate and comparison between options.
Maintenance of surface water level at -0.9 m AHD.	Additional water pumping from Lake Alexandrina required (>35 GL/year) and Sub-surface barrier installation in sandy sediments situated above -0.9 m AHD or One-off treatment of up to ~25,000 tonnes CaCO ₃ .	Refinement of existing acidity flux model required to confirm suitability of management option. An accurate geological map of the distribution of the uppermost sand and clay is required for detailed barrier design and installation. Quantify the annual volume of water required from Lake Alexandrina to achieve -0.9 m AHD. Cost estimate and comparison of options.
Maintenance of surface water level at -0.8 m AHD.	Additional water pumping from Lake Alexandrina required (>>35 GL/year) and One-off treatment of up to ~10,000 tonnes CaCO ₃ .	Refinement of existing acidity flux model required to confirm suitability of management option. Quantify the annual volume of water required from Lake Alexandrina to achieve -0.8 m AHD. Cost estimate and comparison of options.

Table 28. ASS management options for Lake Alexandrina.

Water level scenario	ASS management option	Information required
Surface water level as modelled to mid-2011 in Section 7 (continue pumping 35 GL/year or more to maintain Lake Albert at -1.0 m AHD or higher level); long-term surface water levels maintained around -2.5 m AHD (worst case scenario where inflows are balanced by evaporation).	Sub-surface barrier installation in sandy sediments situated above -2.0 m AHD and Low permeability (e.g. CaCO ₃) terrace installation on clay sediments situated between -2.0 m AHD and -2.5 m AHD; utilisation of the Tertiary limestone aquifer to maintain the clays in a saturated but not inundated state. or Some treatment with limestone may be appropriate.	Refinement and extension of existing acidity flux model required to confirm suitability of management option. Water balance model required to estimate minimum long-term surface water level. An accurate geological map of the distribution of the uppermost sand and clay is required for detailed barrier design and installation. Quantify the de-saturation / desiccation rate of sands and clays in order to determine the relative risk of acidity release from these materials. This information will also assist with quantifying groundwater requirements for maintaining saturation of clays. Quantify the availability of groundwater from the Tertiary limestone for assisting saturation. Cost estimate and comparison between options.
Similar to above, but no pumping to Lake Albert (additional 35 GL/year in Lake Alexandrina); long-term surface water levels maintained around -2.3 m AHD.	As above but slightly lower cost due to larger area of inundated clays with water levels above -2.3 m AHD.	As above.
Purchase 100 GL/year but continue pumping 35 GL/year to or more maintain Lake Albert at -1.0 m AHD or higher level (additional 100 GL/year in Lake Alexandrina).	As above but increased cost of water and decreased cost of terraces due to larger area of inundated clays.	As above.
No pumping to Lake Albert and purchase of 100 GL/year (additional 135 GL/year in Lake Alexandrina)	As above but decreased cost of terraces due to larger area of inundated clays.	As above.
Maintenance of surface water levels above -1.5 m AHD (via additional water purchase).	One-off treatment of up to ~100,000 tonnes CaCO ₃ .	Refinement and extension of existing acidity flux model required to confirm suitability of management option. Water balance model required to estimate volume of water required to achieve minimum level of -1.5 m AHD.

9 Conclusions

Geology and Geochemistry

- Geological information gathered from drilling, shallow pitting and transient electromagnetic surveys (TEM) identified a coherent regional near-surface stratigraphy across both lakes. This stratigraphy is based on information collected within 500 m of the original shoreline of the lakes. The majority of the areas examined to date contain a thin veneer of unconsolidated lake sediments (1-3 m thick) overlying a calcrete/silcrete-capped Bridgewater Formation limestone. The lake sediments are generally composed of an uppermost 0.7-1.3 m thick layer of quartz-rich sand, which overlies a 0.2-1.0 m thick, occasionally calcareous clay layer. An additional quartz-rich sand layer may be found beneath the clay layer and immediately above the Bridgewater Formation. The Bridgewater Formation is an unconsolidated to poorly consolidated calcareous, quartz-rich sand that contains 5-30% carbonate. It is expected to be in excess of 10 m thick, and local onshore exposures indicate that it is likely to be up to 30 m thick at some locations.
- At several locations, the Bridgewater Formation is exposed in the base of the lakes with essentially no significant capping of other sediments. At these sites, the Bridgewater Formation has been subjected to the same sulfate reducing bacterial processes as other lake sediments, and therefore contains elevated concentrations of diagenetic pyrite.
- A combination of the near-shore geology collected in this study and the sediment surveys conducted by CSIRO indicate that the uppermost lake sands form a concentric wedge of sediment that thins toward the centre of the lake. In most instances, the sands progress from 1-2 m thick at the original shoreline, to 0 m thick approximately 1.5 km toward the centre of the lake.
- It is expected that the thin layer of clay identified beneath the uppermost sand around the margins of the lakes thickens toward the centre of the lakes. This layer is believed to form a significant aquitard between the lake sediments and the Bridgewater Formation due to the presence of a small hydraulic gradient between these sand layers, across the clay layer.
- Including the CSIRO sediment database, the near-surface sands of the lake sediments display an average of approximately 0.1-0.2 wt% sulfide-sulfur and an average ANC of 10 kg H₂SO₄/t in Lake Albert, and 0.1-0.2 wt% sulfide-sulfur and an average ANC of 100 kg H₂SO₄/t in Lake Alexandrina. These numbers reveal an average NAPP (Net Acid Producing Potential) of approximately -5 kg H₂SO₄/t for Lake Albert and -95 kg H₂SO₄/t for Lake Alexandrina. The negative NAPP values indicate that the sands (on average) are theoretically not acid generating.
- Including the CSIRO sediment database, the near-surface clays of the lake sediments display an average of approximately 1-2 wt% sulfide-sulfur and an ANC of 10-20 kg H₂SO₄/t in Lake Albert, and 1-2 wt% sulfide-sulfur and an average ANC of approximately 20-30 kg H₂SO₄/t in Lake Alexandrina. These numbers reveal a NAPP (Net Acid Producing Potential) of -8 to +20 kg H₂SO₄/t for Lake Albert and 0 to +40 kg H₂SO₄/t for Lake Alexandrina. The positive NAPP values indicate that the clays (on average) are theoretically acid generating.
- The Bridgewater Formation has been recorded with sulfide-sulfur ranging from 0.02 to 0.61 wt% and ANC values ranging from 50 to 300 kg H₂SO₄/t. Hence, all NAPP values are negative indicating that this material is theoretically not acid generating.

Hydrogeology and Hydrogeochemistry

- Hydraulic conductivity values for lake sediment sands range from 0.09 to >30 m/day. Hydraulic conductivity values for the Bridgewater Formation calcareous sands range from 0.5 to >30 m/day. These values indicate that discharge from the sandy lake sediments has the potential to be relatively rapid, depending on the gradient.
- Groundwater levels in the lake sediments are often slightly lower than those in the underlying Bridgewater Formation, indicating that these horizons are at least locally hydraulically isolated, and that the latter is confined. The widespread exposure of Bridgewater Formation in the base of both lakes indicates that this formation is also locally unconfined. The hydraulic conductivity values indicate that discharge from the Bridgewater Formation sandy sediments has the potential to be relatively rapid, depending on the gradient and degree of confinement.
- Groundwater levels rise rapidly in response to rainfall events at all sites. Rises of 30 cm are typical in response to 10-15 mm rainfall events. These rises fall very rapidly over subsequent days. Since recording began in August, the periodic rises in groundwater levels due to rainfall are overlain on a background of progressively falling groundwater levels, even though lake water levels have remained largely static.

- Sediment moisture data shows that a 30 cm thick zone of essentially saturated sandy sediment exists above the piezometric surface at all sites examined.
- Where the rainfall intensity or duration is sufficient to affect sediment moisture content, peak moisture values in the upper sediments are achieved within several hours. Vertical migration of infiltrating rainwater through the upper 40-50 cm occurs within approximately 1 day of the onset of a significant rainfall event.
- Laboratory measured sulfide oxidation rates in sandy sediments show a complex relationship with moisture content. Sulfide oxidation rates peak in sand samples with 15 wt% water content, and fall (at different rates) for samples with both lower and higher water contents. This relationship has been identified in mine site materials by other researchers (Hollings et al., 2001).
- The relationship between sulfide oxidation rate and moisture content in clay-rich sediments is more linear, with progressive decreases in oxidation rate as a function of increasing moisture content.
- Groundwater quality in the lake sediments varied from site to site. Acidic and metalliferous groundwater was recorded in three of four piezometers at Campbell Park and in two piezometers at Point Sturt. No acid or metalliferous groundwater was identified at the Windmill site.
- Groundwater quality did not vary significantly over the four sampling events at each of the sites (piezometers) and locations. This strongly supports the conclusion that there has been little lateral groundwater migration during the monitoring period.
- Fluctuations in groundwater levels at all sites appear to be responding to either rainfall, evapotranspiration or seiche events. Little or no discharge (seepage) from the sediments to the lake waters is indicated during the monitoring period at any site.
- The salinity of groundwater in the Bridgewater Formation is relatively high (20-50 mS/cm) and varies across Lake Albert.
- The salinity of groundwater in lake sediments is significantly lower than the salinity in the underlying Bridgewater Formation, but only where a locally continuous aquitard (clay layer) separates the units.

Lake Levels

As of mid-November 2009, the surface water levels in lakes Albert and Alexandrina were around -0.45 m AHD and -0.75 m AHD, respectively. These figures are based on DLWBC lake level data. Lake levels based on piezometric data proximal to the lake at the Windmill and Point Sturt locations in mid-November are -0.65 m AHD and -0.9 m AHD for lakes Albert and Alexandrina respectively. The lower lake levels recorded in this study need to be assessed in more detail.

Acidity Generation

Despite water levels decreasing to -0.45 m AHD in Lake Albert and -0.75 m AHD in Lake Alexandrina in late November 2009 (based on DLWBC data), there has been no obvious impact on surface water quality (pH or alkalinity). This indicates that either little acidity has been generated, or little acidity has been released from exposed sediments prior to November 2009, or that any acidity that has been released has been neutralised within the lake sediments or surface water.

Lake Albert

- The total potential acidity generation from sands situated above -1.0 m AHD in Lake Albert is estimated to be approximately 210,000 tonnes H₂SO₄.
- Total potential acidity generation in Lake Albert over 22 months (from September 1st, 2009) is estimated at ~50,000 tonnes H₂SO₄ assuming no ANC consumption, or ~38,000 tonnes H₂SO₄ assuming 10% ANC consumption per year.
- No significant acidity generation is expected in Lake Albert during the initial 4 months despite predictions that the water depth will decrease from -0.163 m AHD to -0.682 m AHD.
- After 5-6 months when the surface water level in Lake Albert is forecast to reach -1.0 m AHD (DLWBC, 2009) total acidity generation is estimated at ~9,800 tonnes H₂SO₄ assuming no ANC consumption from sediments, or ~9,200 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~20-25% of the acidity load after 22 months. This acidity load will not be neutralised by soluble alkalinity in the lake water if it remains stored in the sediment until the autumnal flush.

- Prior to the first autumnal flush, the total acidity generation in Lake Albert is estimated at ~28,000 tonnes H₂SO₄ assuming no ANC consumption, or ~25,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~60% of the acidity load after 22 months.
- The rate of acidity generation in Lake Albert is expected to decrease significantly after 9 months due to pyrite exhaustion (ie. complete conversion to sulfuric acid) in the unsaturated zone.
- Key factors that can limit the acidity generation and release into Lake Albert are increases in the moisture content of the sands, and the extent of ANC consumption within the lake sediments.

Lake Alexandrina

- The total potential acidity generation from sands situated above -2.0 m AHD in Lake Alexandrina is estimated to be approximately 900,000 tonnes H₂SO₄.
- Total potential acidity generation in Lake Alexandrina over 22 months (from September 1st, 2009) is estimated at ~180,000 tonnes H₂SO₄ assuming no ANC consumption, or ~115,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. Thus, ANC consumption of 10 wt% CaCO₃ / year could reduce the total acidity flux to Lake Alexandrina by ~35%.
- As the water level in Lake Alexandrina decreases from -0.776 m AHD to -1.4 m AHD over the initial 6 months, acidity generation is estimated at 24,000 tonnes H₂SO₄ assuming no ANC consumption, or ~18,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~10-15% of the acidity load after 22 months.
- Prior to the first autumnal flush, the total acidity generation in Lake Alexandrina is estimated at ~40,000 tonnes H₂SO₄ assuming no ANC consumption, or ~28,000 tonnes H₂SO₄ assuming 10% ANC consumption per year. This represents ~20-25% of the acidity load after 22 months.
- Relatively minor acidity generation is expected in Lake Alexandrina during the 6 month period following the initial autumnal flush due to lake water level increases over this period, as well as substantial exhaustion of pyrite in the (new) unsaturated zone.
- The rate of acidity generation in Lake Alexandrina has the potential to increase substantially after 15 months due to water levels decreasing to -2.043 m AHD.
- Key factors that can limit the acidity generation and release into Lake Alexandrina are increases in the moisture content of the sands, and the extent of ANC consumption within the lake sediments.

Acidity Flux

The likely rate and duration of acidity release (flux) events in Lake Albert and Lake Alexandrina has been estimated for a range of lake water level, hydraulic conductivity and acidity concentration scenarios, based on hydrogeological modelling conducted by Coletti and Hipsey (2010).

In Lake Albert, the acidity flux event associated with the first autumnal flush could have a duration ranging from 2-3 months (77 days) to several years (10,000 days), while the subsequent acidity flux event is estimated to occur in approximately half this time, due to the correspondingly lower tonnage of acidity generation.

In Lake Alexandrina, the acidity flux event associated with the first autumnal flush could have a duration ranging from 1-2 months (36 days) to several years (4,700 days), while the subsequent acidity flux event is estimated to occur over approximately three times this duration, due to the correspondingly higher tonnage of acidity generation.

Based on the observed acidity flux event at Currency Creek in 2009, the duration of future acidity flux events are likely to be at the lower end of the scale indicated here, ie. closer to 2-3 months for the first acidity flux event and 1-2 months for the second event in Lake Albert, and closer to 1-2 months for the first acidity flux event and 3-4 months for the second event in Lake Alexandrina. These estimates correspond to the lower minimum lake water level (-1.0 m AHD), hydraulic conductivity of 10 m/day and acidity values of 10,000 mg/L CaCO₃.

In Lake Albert, some of the acidity flux associated with the first autumnal flush will be neutralised by the available soluble alkalinity in the water column.

In Lake Alexandrina, while there is likely to be sufficient soluble alkalinity in the lake water to neutralise the acidity associated with the first autumnal flush event, less alkalinity would remain for future buffering.

It may be possible to cost-effectively add large tonnages of limestone at Wellington from barges to replenish the alkalinity lost during the second autumnal flush within Lake Alexandrina.

Management Options

- All efforts should be directed at keeping sulfidic lake sediments saturated to prevent acidity generation.
- Subsurface barrier installation within the uppermost sandy sediments around the unsaturated margins of both lakes is expected to assist with retarding sulfide oxidation by maintaining the sulfidic material in a saturated, or largely saturated state. The barriers are expected to have the added benefit of enhancing ANC consumption.
- Shallow terraces constructed from ultra-fine grained limestone strategically installed along contours on top of large areas of exposed clays could be useful for maintaining saturation (not inundation) of exposed clay materials during dry periods. The water required for surface application above the terraces could potentially be obtained from the Tertiary Limestone aquifer.
- Treatment of lake water and exposed sediment banks could be conducted with limestone. This could involve either pre-emptive or post acidification limestone addition, and could potentially be done from the lake surface (e.g. barges), from the shoreline (e.g. mixing and dosing equipment), or from the air (e.g. air tractors).

10 Recommendations

1. All efforts should be directed at keeping sulfidic lake sediments saturated to prevent acidity generation.
2. It is recommended that the acidity generation models developed as part of this study are used to investigate a range of alternative lake water level scenarios, for example:
 - Lake Albert:
 - Ephemeral lake (no pumping from Lake Alexandrina).
 - Maintenance of surface water level at -0.8 m AHD or -0.9 m AHD.
 - Implementation of ASS management options (e.g. sub-surface barriers, terraces, passive / active addition of alkaline groundwater, treatment) combined with each water level scenario.
 - Lake Alexandrina:
 - No pumping to Lake Albert (additional 35 GL/year in Lake Alexandrina).
 - Purchase 100 GL/year and continue pumping 35 GL/year or more to maintain Lake Albert at -1.0 m AHD or higher level (additional 100 GL/year in Lake Alexandrina).
 - No pumping to Lake Albert **and** purchase 100 GL/year (additional 135 GL/year in Lake Alexandrina).
 - Maintenance of surface water levels around -2.5 m AHD (worst case scenario where inflows are balanced by evaporation).
 - Maintenance of surface water levels above -1.5 m AHD (via additional water purchase).
 - Implementation of ASS management options (e.g. sub-surface barriers, terraces, passive / active addition of alkaline groundwater, treatment) combined with each water level scenario.
3. To improve the accuracy of the existing acidity generation models and improve predictions for Lake Albert and Lake Alexandrina, the following investigations would assist:
 - The model is highly sensitive to the selection of a groundwater gradient. Hence, modelling of hydraulic conductivity vs hydraulic gradient in lake sediments would improve the accuracy of acidity generation estimates.
 - Field measurement of hydraulic conductivity vs hydraulic gradient (e.g. Poltalloch; south-west shore of Lake Albert) via additional piezometer installation and/or field based surveys (e.g. differential GPS).
 - Repeat ANC analyses at previous CSIRO sample sites in Currency Creek to quantify actual ANC consumption rates in acidified sediments. This work should measure the available ANC as well as the total ANC remaining.
 - Analysis of total sulfur and ANC at depths of 0.5-2.0 m in the upper sandy sediments of both lakes.
 - Quantify the rate of bacterial sulfate reduction at the base of the water column in lake sediments as a function of pH. Quantify the rate of bacterial sulfate reduction in sandy and clay-rich sediments as a function of key parameters such as carbon and iron content, pH and moisture content.
 - Identify alternative methods to quantify current acidity flux rates to the lakes that are independent of the methods detailed in this report (e.g. sulfur isotope analysis or geochemical ratio assessment).
4. It is recommended that the existing ASS management plan for Lake Albert is reviewed on the basis of recent modelling results.
5. An ASS management plan should also be developed for Lake Alexandrina including a detailed assessment of the most likely water level scenarios. Specific ASS management options applicable to each scenario should be assessed on the basis of:
 - Ease of implementation;
 - Expected performance in minimising / controlling acidity generation;
 - Timeframe for implementation and achievement of management plan objective;
 - Cost-effectiveness;

- Risks of implementation;
 - Potential to minimise impacts associated with wind erosion;
 - Potential for community involvement.
6. Conduct further investigation to resolve the discrepancy between lake water levels and piezometric levels.
 7. Develop an accurate geological map of the distribution of the uppermost sand and clay in both lakes to facilitate detailed barrier design and installation (if required). This is likely to include the use of one or more methods (e.g. interrogation of CSIRO data, push tube coring, lithological logging via manual coring/pitting of sediments, TEM methods).
 8. Conduct a trial using sonic remote mapping equipment (i.e. CHIRP; 10 cm resolution) to assess the viability of this method for geological mapping of the upper 5-10 m of lake sediments.
 9. Quantify the de-saturation / desiccation rate of sands and clays in order to determine the relative risk of acidity release from these materials during rainfall events. This information will also assist with quantifying groundwater requirements for maintaining saturation of clays via the use of terraces.
 10. Quantify the availability of groundwater from the Tertiary limestone aquifer for maintaining clay saturation in the event that shallow terraces are employed to retard sulfide oxidation in clay-rich sediments.
 11. Develop cost estimates for the management options outlined for lakes Albert and Alexandrina.

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