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Lake Alexandrina and Lake Albert: Analysis of Groundwater Measurements and Estimation of Acid Fluxes

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Cover Photograph: Taken by Gordon McLachlan CSIRO Land and Water, 2011 of Lake Albert with South Australian EPA boat used for sampling in foreground.

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EXECUTIVE SUMMARY

The drought in South-Eastern Australia in the previous decade resulted in record low levels of water inflow to the Lower Lakes. This resulted in exposure of sediments with pyritic minerals and the potential to generate substantial amounts of acidity in these sediments. The possible flux of this acidity to the Lower Lakes was a concern and has generate a number of previous studies including one (Earth Systems, 2010), which has provided much of the data for this report.

This report presents results of estimates of the acid flux to the lakes from the monitoring data available at, albeit only four sites. In order to be able to consider the flux of acidity to the lakes a conceptual model for possible processes that could contribute acidity to the lakes is present and then tested against the available evidence. The conceptual model considers four mechanisms for acid flux from the sediments to the lakes *viz*;

- wash off of acidic products from the sediment surface groundwater flux during rainfall or seiching events,
- flow of acidic groundwater to the lake,
- exfiltration of acid pore water during rainfall events or upon rewetting of near shore sediments,
- diffusion of and/or mixing of acidity in the sediments with the lake waters during seiching events or upon flooding following lake level rise.

This conceptual model is evaluated by analysing existing water table and water content monitoring data at four sites; two on Lake Alexandrina (Point Sturt and Currency Creek) and two on Lake Albert (Campbell Park and Windmill). These data were provided to CSIRO by South Australia Environmental Protection Agency and collected by Earth Systems (2010). From the data estimates of the capillary length scale result in values from 0.1 to >0.66 m from the three sandy sites where data was available. This indicated that near the shoreline sediments were unlikely to be unsaturated for maximum distances ranging from 30 to 750 m. Also the specific yield ranged from 0.01 to 0.19 and showed that the water level in the sediments would rise substantially when rainfall occurred.

The piezometer data was used to show that the horizontal gradients were low, on average about 10^{-3}, and in the near shore region from the lake towards land. This flow of water from the lake to the near shore sediments supports one of the hypothesis in the conceptual model. This along with the maximum likely travel distance towards the lake of 14 m suggests that during the monitoring period acid fluxes due to groundwater flow were insignificant.

The data used in this report is able to give estimates of the fluxes from all apart from the diffusion of acidity due to inundation of oxidised (acidic) sediments. The piezometric head data was used to calculate the horizontal hydraulic gradient and direction of the flow between the points in the piezometer transects. In such low relief landscapes the effect of density due to salinity and temperature could also affect the hydraulic gradient and this was calculated when suitable data was available. The effect of water density on the horizontal hydraulic gradient was found to be insignificant, even in this low relief landscape. The horizontal hydraulic gradient was calculated to be very small, of the order of 1 in 1000 and often from the lake toward the sediments. This result showed that at these transects acid flux to the lakes via groundwater was likely to be insignificant.

The results indicated that evaporation of water from saturated and near saturated sediments near the shoreline often resulted in hydraulic gradients away from the shoreline. They also indicated that the water flow was often towards the middle of the transect suggesting a possible accumulation by evaporative concentration of acidity in this region as was suggested in the conceptual model. However, calculations of the possible maximum distance of travel for a water particle horizontally suggested that the distance travelled would be minor, especially given strong upward vertical hydraulic gradients. Hence accumulation of acidity in the zone of convergence of the hydraulic gradients is not likely to be large.

The results suggested that the conditions for the generation of acidity was most likely at the furthest point from the shoreline on the transects. This area has also been suggested in other studies as being of high potential acid generation. Significant acid flux was more likely from runoff and exfiltration from these areas. Estimates of worst case acid fluxes via exfiltration and runoff were estimated and showed that during the time of monitoring acid fluxes were unlikely to have had a significant impact on the alkalinity of Lake Alexandrina at the water levels maintained during this time. The same analysis for Lake Albert showed that for the worst case scenario the alkalinity in the lake could be overwhelmed by the acid flux. This is because the amount of exposed sediments at Lake Albert are much greater than for Lake Alexandrina. The Windmill site on Lake Albert also indicated that net alkalinity could also be discharged to the lake from such sites.

The acid fluxes suggested here are one to two orders of magnitude less than acid flux estimates from Earth System (2010) for Lake Alexandrina and similar or greater than those for Lake Albert. However, some caution needs to be applied to the extrapolation of these and Earth systems results to whole of lake risk assessment as they are based on only one transect on Lake Alexandrina and two transects on Lake Albert.

Even so, given the acid fluxes estimated from runoff and exfiltration, localised acid hot spots are possible when the volume of receiving water, and hence alkalinity is small. This reduced receiving water volume to acid flux is likely in embayments and such small localised pockets of acidic water have been observed.

Some useful soil physical properties such as; the macroscopic capillary length scale; the specific yield; and moisture retention characteristics were able to be derived from the data. These were useful in estimating the distance from the shoreline that sediments would be saturated and will be used in subsequent modelling studies.

The results presented here highlight the need for careful analysis based on sound conceptual models when assessing environmental risk. The conceptual model described here and analytical methods used should be useful in similar studies.

1. INTRODUCTION

The Lower Lakes (Lakes Alexandrina and Albert) are located near the mouth of the Murray River in South Australia approximately 75 km south of Adelaide. The drought in the Murray Darling Basin has resulted in lower flows of water through the Murray River to Lake Alexandrina and Lake Albert. The Lakes were full in 2006 but by 2008 water levels had dropped to below mean sea level (figure 1). The lower water level in the Lower Lakes (Lake Alexandrina and Lake Albert) have exposed sediments with pyrite concentrations that can generate a potential net acidity of > 250 mols H^+ $tonne^{-1}$ in some of the sediments, with the majority of the sediments having acidity generating capacity of less than this, but some sediments especially in Lake Albert having high > 500 mols H^+ $tonne^{-1}$ in some sediments (Fitzpatrick et al., 2010). The Lakes are alkaline and have considerable buffering capacity but the acidity potential is much greater and even if a small percentage of this acidity was transported to the lakes it is possible that the lakes could become acidic (Hipsey and Salmon, 2008). However, for the acid sulfate hazard in the sediments to become a risk to the lake, the acidity must be transported from the sediments to the lake waters..

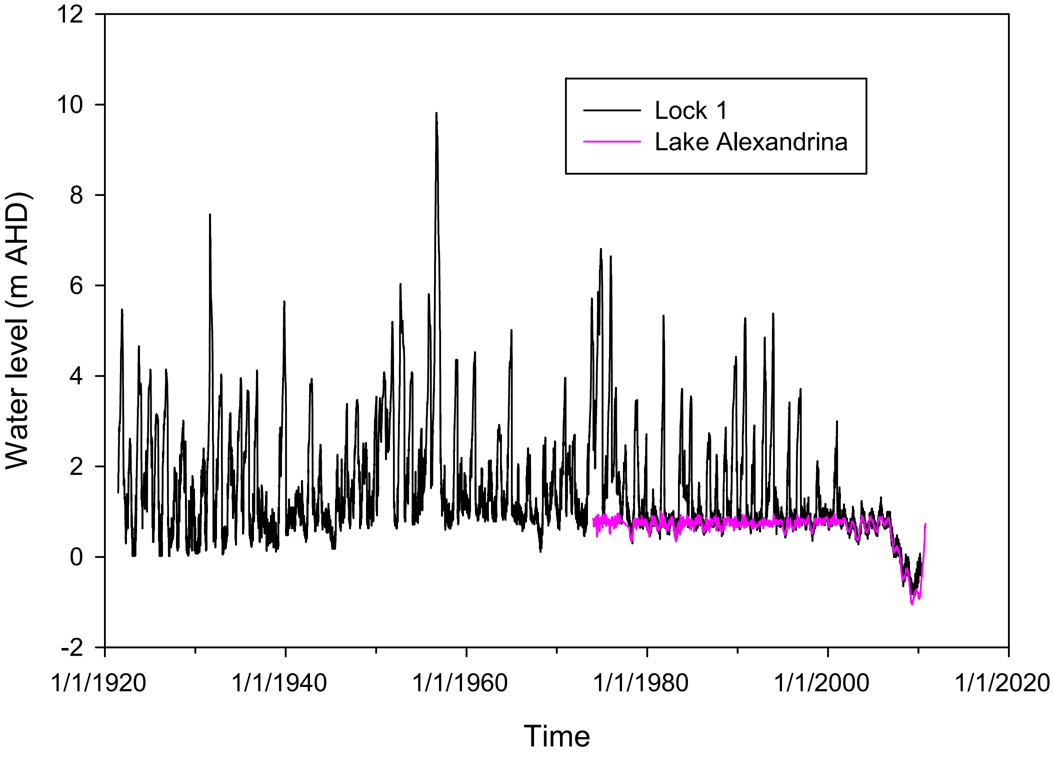


Figure 1. Historical water levels at lock 1 and Lake Alexandrina. Data supplied by Luke Mosley SA EPA.

Modelling has been done of the acid generation and flux (Hipsey and Salmon, 2008; Hipsey et al. 2010) and lake processes. This is based on laboratory estimates of the oxidation rate of the pyrite (Earth Systems, 2010) and modelled groundwater flux as the major mechanism for acid flux to the lakes. The oxidation rate studies carried out give the potential oxidation rate if atmospheric oxygen concentrations occurred within the sediments. Atmospheric oxygen conditions are unlikely to occur throughout the sediments due to consumption of oxygen by organic reactions and the consumption by the pyrite oxidation itself (Cook et al., 2004). Rigby et al. (2004) and Cook et al. (2004) showed that oxidation of organic matter competes for oxygen with pyrite oxidation and can reduce the amount of oxygen available for pyrite oxidation. The planting of vegetation and the natural revegetation will result in competition for the oxygen by plant roots (Cook and Knight, 2003). However, some plants have aerenchyma (canals that can transport air to roots) especially those that can survive in high iron and sulfate environments such as *Phragmites australis* (Armstrong and Armstrong, 1988). These plants can release oxygen to the soil around the roots and may cause an increase in the pyrite oxidation in these oxygenated sediments. Observations of iron formation in the vicinity of such roots suggest that these plants are likely to exacerbate acid generation.

The gradient for flow to the Lakes from the surrounding sediments may not be very large as evaporation in the sediments near to the water's edge may cause a gradient from the lake to the sediments during the falling stage of the lake level and during lake level rise again the gradient may be predominantly away from the lake. This flow of water from the lakes to the sediments is enhanced due to the shallow flat-bottomed nature of the lakes. In this report we will investigate these gradients and estimate acid fluxes based on measurements and previous data (Earth systems 2010, EPA 2011) obtained along transects orthogonal to the water's edge.

In order to better estimate the acid generation and fluxes there is a need to develop a conceptual model of these processes and then to measure and model the fluxes of acid to the lakes and determine under what conditions the various fluxes will become important. This conceptual model and related research will help to address the objectives of the South Australian Government to :

- Develop an understanding of the acidity generation, neutralisation and groundwater transport processes within the lake sediments of the Lower 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 test work programs.
- Provide recommendations for future management of the Lower Lakes.

Here we propose a conceptual model for the acid generation and fluxes, and analysis methogs to elucidate the processes proposed in this conceptual model.

1.1. Review of Existing Studies

The mapping studies by Fitzpatrick et al. (2008, 2010) have described the potential acidity generation and extent of acidity that could be produced if all the pyrite in the sediments was oxidised. However, the rate of generation and transport of acidity to the lake waters was not addressed in this study. A subsequent study by Simpson et al. (2009, 2010) allowed samples of the acid sulfate sediments to dried, to allow oxidation of the sulfidic materials. These dried oxidised samples were then rewetted by shaking in water, sourced from the Murray River and artificial rainwater. The acid released into the water gave an estimate of the maximum likely acidity release upon rewetting. However, the treatment of these samples would have exaggerated the transport from the sediment to the water. This study also showed that if the ratio of Murray River water (alkaline) to suspended acidic sediments was less than 100:1 the solution pH will drop to 6.5. Also the amount of precipitation of metals (Al and Fe precipitates) is close to 100% at this dilution, which will maximise the acid release. They also suggested "Flow pathways and interactions with sub-surface waters in the river, wetlands, ground waters (that usually have high alkalinity) and lakes are poorly defined, but may have a significant effect on the degree to which soil waters are neutralised. Detailed studies are recommended for those wetlands likely to be at risk from acidification by ASS.".

To estimate the likely risk to the lakes modelling of the acid flux to the Lakes, along with subsequent mixing and reaction within the lakes was performed by Hipsey and Salmon (2008). Their results suggested that the Lakes could become acid, if water levels fell below -1.11 to -2 m. Their interim report was criticised by Webster et al. (2008) In particular Webster et al. criticised the oxidation rates and acid flux transport mechanisms assumed in the interim Hipsey and Salmon (2008) report. This lead to a revision and the published report by Hipsey and Salmon (2008) included suggestions from Webster et al.. However despite the uncertainties noted in the earlier Hipsey and Salmon modelling successfully predicted Loveday Bay and Currency Creek acidification in winter 2009 assuming average oxidation rate scenarios (DENR 2010).

The oxidation rate of the sediments is important to be able to estimate the rate of acidity generation as well as the profile of acidity as a function of depth in the sediments. Earth Systems (2010) have made estimates of the oxidation rate as functions of water content and obtain a maximum rate similar to Rigby et al. (2004) for stirred suspended sediment. They did not assess the likely *in situ* rate when oxygen transport and consumption by organic matter occurs. Rigby et al. (2004) found even in their well stirred experiments, which maximise possible oxidation, that most of the oxygen was consumed by organic reactions. Thus the use of the oxidation rate from Earth Systems (2010) for the sediments is likely to overestimate oxidation and acid generation.

Experiments using mesocosms in which oxidised sediments with either river water or saltwater were carried out by Hicks, et al. (2009) and showed that much less acidity is transferred to the water column when river water was used than would be predicted by Simpson et al. (2009). They also showed increased acid release when saltwater was used, which suggested that flooding the sediments with saltwater could exacerbate the acid transfer to the lake waters. The experiments of Hicks et al. (2009) do not have the degree of wave action that will occur in the Lakes or the alternate wetting and drying caused by seiching and are likely to underestimate the flux upon rewetting. These experimental results (Hicks et al. 2009) provide estimates of the rate of reduction of sulfate compared to diffusive flux which is important in determining the time course of acid release upon rewetting.

Sulfide reduction studies (Sullivan et al. 2010) have shown that lack of carbon is likely to limit the sulfate reduction processes when the sediments are rewetted by lake level rise. Further *in situ* studies are required to determine the oxidation rate and penetration depth of drying fronts into the sediments, especially the clays. Hicks (unpublished report) has recently been able to model the reaction rates within the mesocosm experiments and this modelling will also add to the understanding of sulphide reduction processes.

The water table transects and water content profiles were monitored by Earth Systems (2010) and EPA (2011) for three transects and provide a very valuable data set, which are useful for determining the groundwater gradients and drying fronts for the sandy sites selected. More information can be derived from these samples, as will be shown in this report. Along with this piezometric head data, water quality was also measured on water samples extracted from the piezometers and provides data on the actually acidity of the the groundwater in the sediments.

On the basis of the further information on oxidation rates and groundwater transects Hipsey et al. (2010) subsequently modified their original model and provided revised estimates of acid fluxes to the lakes at different lake water levels. This modelling still does not consider some of the transport mechanisms that will be discussed in section 1.2 below and may still overestimate the acid fluxes to the lakes. Current management recommendations (DENR

2010) are that the risk of broad-scale lake acidification is reduced if water levels are stabilised at or above minus 1.5 m AHD in Lake Alexandrina and minus 0.5 m AHD in Lake Albert. The risk profile is understood to substantially increase past these water levels and/or with prolonged time near these levels

To assist with the development of a sound research program a conceptual model of the generation and fluxes of acidity is needed. Below we will develop such a conceptual model and examine this using data from the shallow water studies of Earth Systems (2010).

1.2. Conceptual Model

The conceptual model considers three scenarios *viz* initial lowering of the lake levels, refilling and subsequent lowering of the lakes.

1.2.1. Initial Lake level reduction

The initial lake level reduction will expose sediments and these will become unsaturated to some depth away from the lake (figure 2). These unsaturated sediments can then generate acidity by oxidation of the pyrite.

The sediments will remain saturated to some distance from the lake water edge, which can be approximated as $\Delta R_\lambda = \lambda / \tan\theta$, where λ is the macroscopic capillary length scale of the sediments (White and Sully, 1987), and θ is the angle of the soil surface away from horizontal. This is only approximate as evaporation will reduce this distance. It can be calculated more precisely if the evaporation rate and soil physical properties of the sediment are known. At distances greater that ΔR_λ the sediments will get progressively more unsaturated as the soil surface rises (depth to water table increases) and the potential for pyrite oxidation will increase with this increasing distance from the shoreline. Beyond a certain distance the evaporation rate will decrease as the transport of water to the soil surface limits the evaporation. This will then decrease the rate of increase in the depth of the drying front and limit the amount of sediment oxidised. This decrease in the drying depth will continue until some point where the upward flux of water due to evaporation becomes negligible.

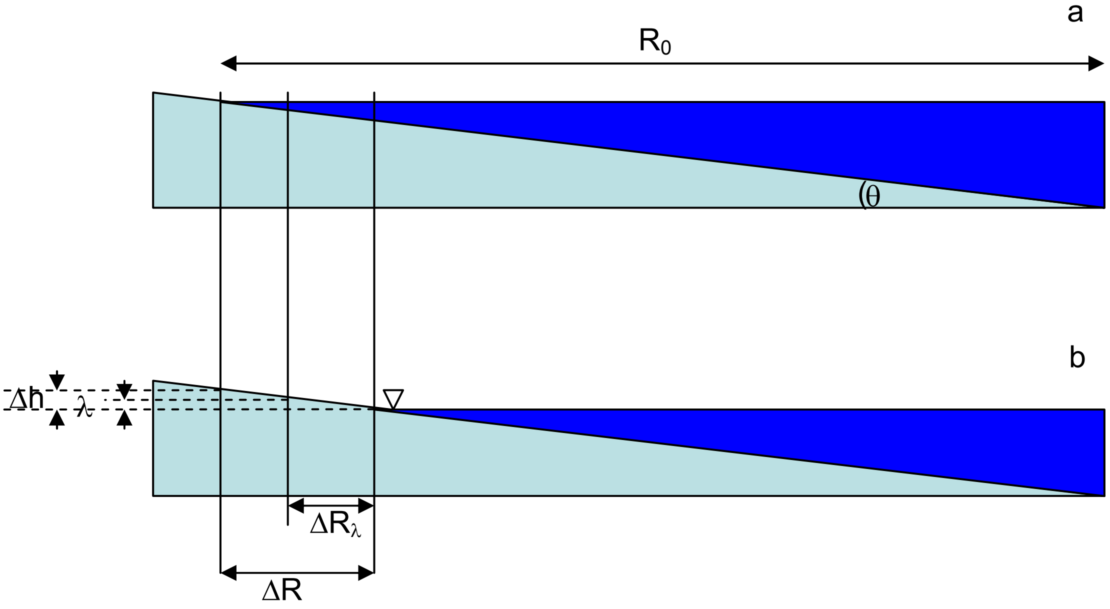


Figure 2. Schematic of the initial reduction of the lake water level a) initial water level and lake radius of R_0 and b) at a level Δh below the initial level.

Acid transport

The mechanisms for acid entering the lake waters are;

- washing of acid products from the surface of the oxidised sediments during seiching and rainfall,
- flow of acidic groundwater to the lake,
- exfiltration of acid pore water during rainfall events or upon rewetting of near shore sediments
- diffusion and/or mixing of the acidity in the sediments into the lake waters during seiching events or upon flooding following lake level rise.

In the near shore region beyond ΔR_λ, evaporation is likely to concentrate acidic salts on or near the soil surface. The amount of water required to saturate the sediments during a rainfall event or seiching will be small and the acidic salts can be washed into lake waters (figure 3). The amount of acidity generated in this region may be small and regular washing may also reduce the amount available in each event. This region is denoted by $\Delta R - \Delta R_\lambda$ in figure 2. As the water table depth increases the water content at the surface will decrease and the upward flux will be limited by the hydraulic conductivity of the sediments. The distance to where the upward flux becomes negligible, will dependent on physical properties of the sediment (macroscopic capillary length scale) and evaporation rate. The acidic salts will also be washed back down the soil profile in rainfall events which do not cause runoff. The amount of acidity in any runoff will depend on the exchange processes between the soil and runoff water. Tong et al. (2010) has developed a model for estimating such processes.

Runoff from soils can be caused by two mechanisms (Grayson *et al.* 1995), one is when the rate of rainfall falling on the soil exceeds the infiltration rate of the soil. The second mechanism is when the storage capacity of the soil becomes filled and no more water can enter the soil, so that any further rainfall, minus any water that can enter the soil, runs off. Three of the sites in this study Point Sturt, Campbell Park and Windmill have sandy soils where the infiltration rates will be high, so that the second mechanism is most likely at these sites. The Currency Creek site has a soil with much higher clay content so that the first mechanism is more likely at this site.

The potential for increased groundwater flux will occur during reduction in the lake water level which potentially induces a flux of water to the lake as the hydraulic gradient steepens. This process will not result in acid flux until the acidic products from the oxidation are washed down to the water table. Groundwater flux to the lake could also occur following rain when the water table on the land surrounding the lake rises (due to infiltration of rainfall) and results in a gradient towards the lake. There are a number of parts to this process. Firstly the acidic products need to be washed down to the groundwater through the unsaturated soil zone and/or the water table needs to rise into this zone and solubilise retained acidity. Depending on the amount of drainage some reduction in concentration will be possible if the flux through the saturated zone near the water table is slower than the rate of sulfate reduction or acidic groundwater passes through unoxidised sediments with carbonate content (figure 4a).

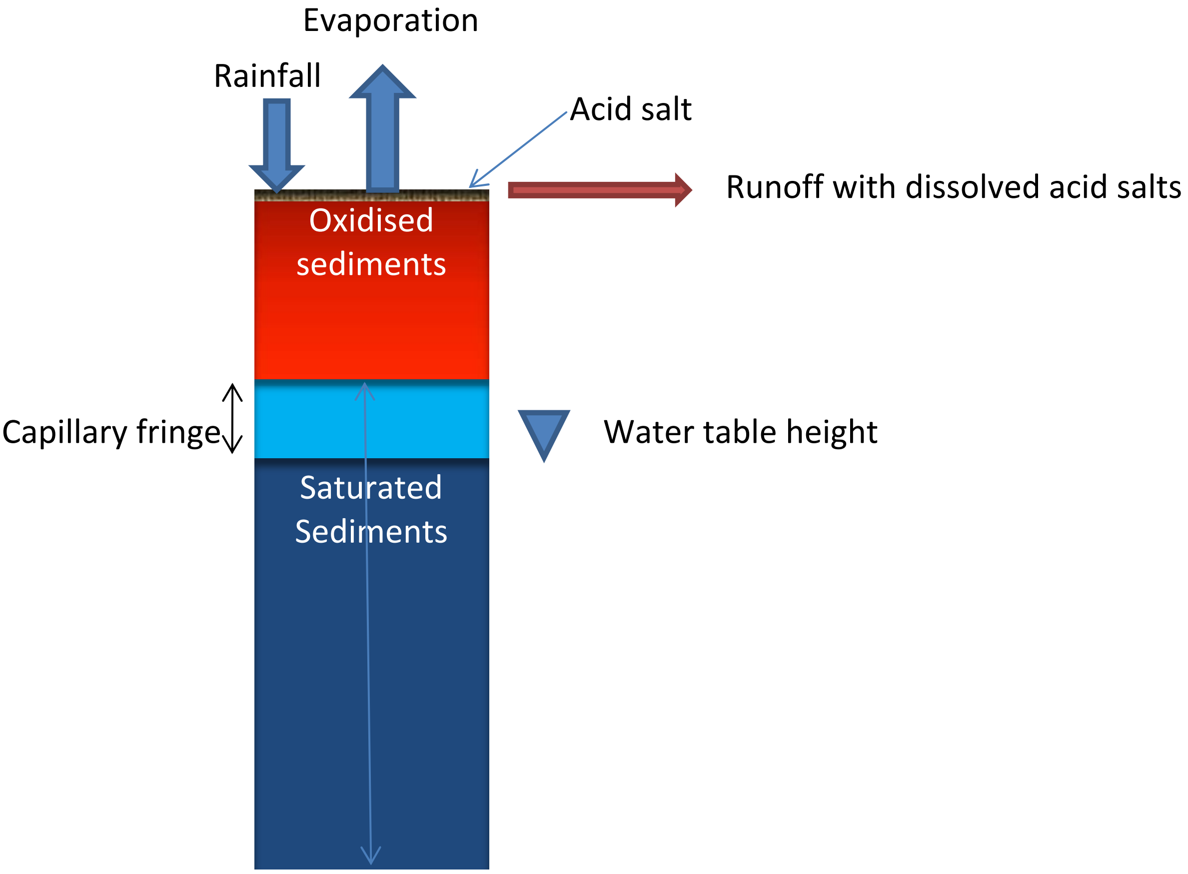


Figure 3. Schematic of formation of acid salts at the sediment surface and subsequent acid flux due to washing of acid products from near lake zone. The water table height will fluctuate in the sediments which can also result in ex-filtration causing acid salts to be washed from the surface.

Secondly, the acidic groundwater needs to be transported into the lake. The groundwater flux will also be reduced by evaporation in the near shore region lowering the water table in this region and resulting in water flux from the lakes to the sediment and/or reducing the overall hydraulic gradient. This has been observed in data collected by Earth Systems (2010) and EPA (2011). This will mean that the groundwater flux is only likely to occur when rainfall has caused a significant water table gradient towards the lake. However, the travel time for the flux of acidity from drier more acidic regions upslope may be considerable and estimation of the travel distances and times will be helpful in understanding the potential future acidic fluxes.

However, the groundwater pressure from upslope can also cause the water table near the lake to rise above the surface of the sediments and for exfiltration to occur. Exfiltration (Grayson *et al.* 1995) occurs when a water table rises to the soil surface usually as a result of water pressure from up-slope and water then exits from the soil and runs over the surface to some receiving water body. In the near shore regions of the lakes this is likely to occur during extensive rainfall events when flow from the surrounding land will cause the water level especially near the lake, where the soils are saturated, to rise and water to exfiltrate. It is also likely to occur in the near shore region when the lake level rises due to seiching or refilling of the lake as the water level will rise in the lake resulting in a rise in water pressure in the near shore sediments and exfiltration. However, this is only likely to result in acid flux during the refilling of the lake, as during a period when the lake level is falling the sediments near the lake will be saturated and unlikely to have generated acidity.

This exfiltration can cause the washing off of acidic salts (figure 3) and water rising up through the oxidised zone is likely to contain acidity and this mechanism will constitute a pathway for acid flux to the lakes (figure 4b). Exfiltration in the near shore area is also possible due to lake level rise during seiching events or lake refilling as the water pressure from the lake can cause water to exfiltrate from the sediments.

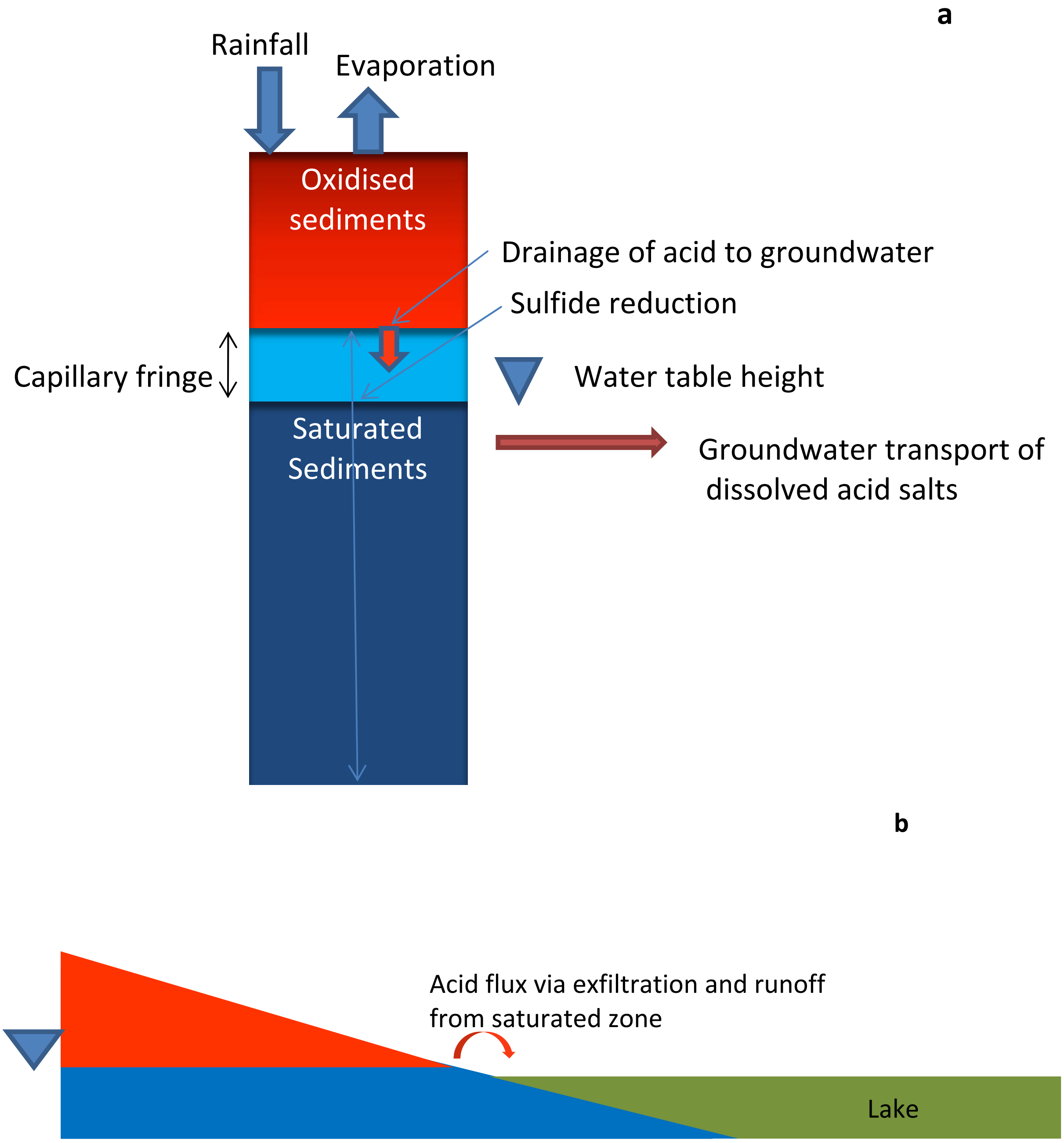


Figure 4, Flux of acid to the lake via groundwater a) fluxes in the profile b) exfiltration from a seepage face to the lake.

The third mechanism for acid flux into the lake is during seiching events when the acid can move from the sediments to the lake water via dissolution of acidic salts on the surface of the sediment, diffusion from the sediments, and/or convection induced by wave action. The diffusion process is likely to be very slow but dissolution and convection due to wave action and/or resuspension of the sediments will be a much faster process. The higher surface area due to cracks in the clay sediments will increase the surface area for diffusion and the flushing of the cracks by convection. For clays this process may be important especially after lake refilling. However, the clays are also highly sodic and dispersive, which means that the cracks are likely to fill through collapse of the peds upon lake refilling.

1.2.2. Refilling of lake after initial reduction

The refilling of the lake after the initial drop in lake level to some minimal value will result in a rising water table level in the near shore region. This is likely to suppress the groundwater flux to the lake and result in alkaline lake water flowing into acidic sediments. However, this may also cause a flux of acid water to be pushed up out of the soil (exfiltration) in the region ΔR_{λ} away from the new lake level (figure 4b). Sulfate reduction is also likely to be initiated in this region due to the saturated anoxic conditions. As the water level in the lake rises, the region where acidic products can be washed off by rainfall will extend into areas of increased acidity. The inundation of previously dry, acidic sediments will lead to an increase in the flux of acidity from diffusive and convective processes. The diffusive flux is likely to be low but the convective flux generated by wave action and fluid density differences could be significant.

1.2.3. Reduction in lake level following a refilling event

Sediments exposed prior to refilling will have oxidised and formed sulfates. Upon reinundation sulfate reduction is likely to occur, resulting in generally mono-sulfides being produced.

Subsequent reduction in the lake level will expose these mono-sulfides which can quickly oxidise to produce a source of acidity near to the lake water level. This acidity can be transported to the lake waters via seiching, wave action and wash off by rain in the near shore region. This could result in pulses of acidity into the lake waters when seiching or rain occurs.

1.3. Scope of this Report

This report will give a detailed analysis of the water content and piezometric level data collected by South Australian EPA (2011) and Earth systems (2010) at four sites in the Lake Albert, Lake Alexandrina near-shore area. The extent of this data varies for site to site and with site. The monitoring ceased when the lake water level overtopped the piezometers at a particular time. The site and piezometers information will be used to generate an estimate of the distance to the shoreline from the piezometer nearest to the shoreline, and subsequently the gradient of the piezometric head between this piezometer and the lake. This will be used to test the concept of near-shore evaporative flux limiting the flux of acid to the lake.

The water content data along with the piezometric level can be used to estimate the relationship between water content and matric potential of the sediments. The water content data can then be used to estimate the vertical potential gradient within the sediments. These estimates will in a later report (Cook et al. 2011) be compared with measured values of these relationships.

Estimates of the flux of acidity to the Lakes are made from the analysis of exfiltration and runoff events and horizontal flux of acidity to the lakes. These estimates give a first guess at the range of acid fluxes that would be expected. This can then be used as a check on modelling that will be done in a later report using HYDRUS2D/3D (Šimůnek and Šejna, 2007).

2. GROUNDWATER ANALYSIS

The groundwater was monitored at four sites *viz* Point Sturt, Campbell Park, Windmill and Currency Creek for varying lengths of time and the raw data has been published previously (Earth Systems, 2010; EPA, 2011) so only a summary will be given here. The length of record, position of the piezometer and surface elevation at the piezometer are given in Table 1.

The sites in relation to each other and their position on the lakes are shown in figure 5.

Table 1. Position, Length of record and land surface elevation at piezometer.

Site and Point	Position		Length of Record	Elevation (mAHD)
	Easting	Northing		
Point Sturt Point 1 shallow	321172	6070261	30/08/2009 – 09/09/2010	0.334
Point Sturt Point 1 deep	321169	6070263	15/09/2009 – 12/08/2010	0.286
Point Sturt Point 2 shallow	321203	6070329	30/08/2009 – 09/09/2010	-0.141
Point Sturt Point 2 deep	321200	6070331	15/09/2009 – 12/08/2010	-0.166
Point Sturt Point 3 shallow	321234	6070398	30/08/2009 – 25/08/2010	-0.398
Point Sturt Point 3 deep	321231	6070399	15/09/2009 – 25/08/2010	-0.375
Point Sturt Point 4 shallow	321265	6070466	30/08/2009 – 11/06/2010	-0.488
Point Sturt Point 4 deep	321263	6070468	15/09/2009 – 11/06/2010	-0.479
Campbell Park Point 1 shallow	341219	6056466	28/08/2009 – 27/09/2010	0.284
Campbell Park Point 1 deep	341216	6056466	16/09/2009 – 21/09/2010	0.279
Campbell Park Point 2 shallow	341215	6056515	28/08/2009 – 27/09/2010	-0.009
Campbell Park Point 2 deep	341212	6056515	16/09/2009 – 21/09/2010	-0.003
Campbell Park Point 3 shallow	341211	6056565	28/08/2009 – 27/09/2010	0.018
Campbell Park Point 3 deep	341208	6056564	16/09/2009 – 21/09/2010	-0.047
Campbell Park Point 4 shallow	341207	6056615	28/08/2009 – 27/09/2010	-0.03
Campbell Park Point 4 deep	341210	6056615	16/09/2009 – 08/09/2010	-0.164
Windmill Point 1 shallow	345597	6064184	29/08/2009 – 27/09/2010	0.008
Windmill Point 1 deep	345599	6064182	16/09/2009 – 08/09/2010	0.009
Windmill Point 2 shallow	345570	6064142	29/08/2009 – 22/09/2010	-0.044
Windmill Point 3 shallow	345543	6064100	29/08/2009 – 22/09/2010	-0.111
Windmill Point 4 shallow	345516	6064058	16/09/2009 – 22/09/2010	-0.131
Windmill Point 4 deep	345519	6064056	29/08/2009 – 08/09/2010	-0.168
Currency Creek Point 1 shallow	299269	6074127	12/05/2009 – 12/01/2011	0.465
Currency Creek Point 1 deep	301320	6072955	15/05/2009 – 12/01/2011	0.635
Currency Creek Point 2 shallow	299589	6073004	12/05/2009 – 12/01/2011	0.321

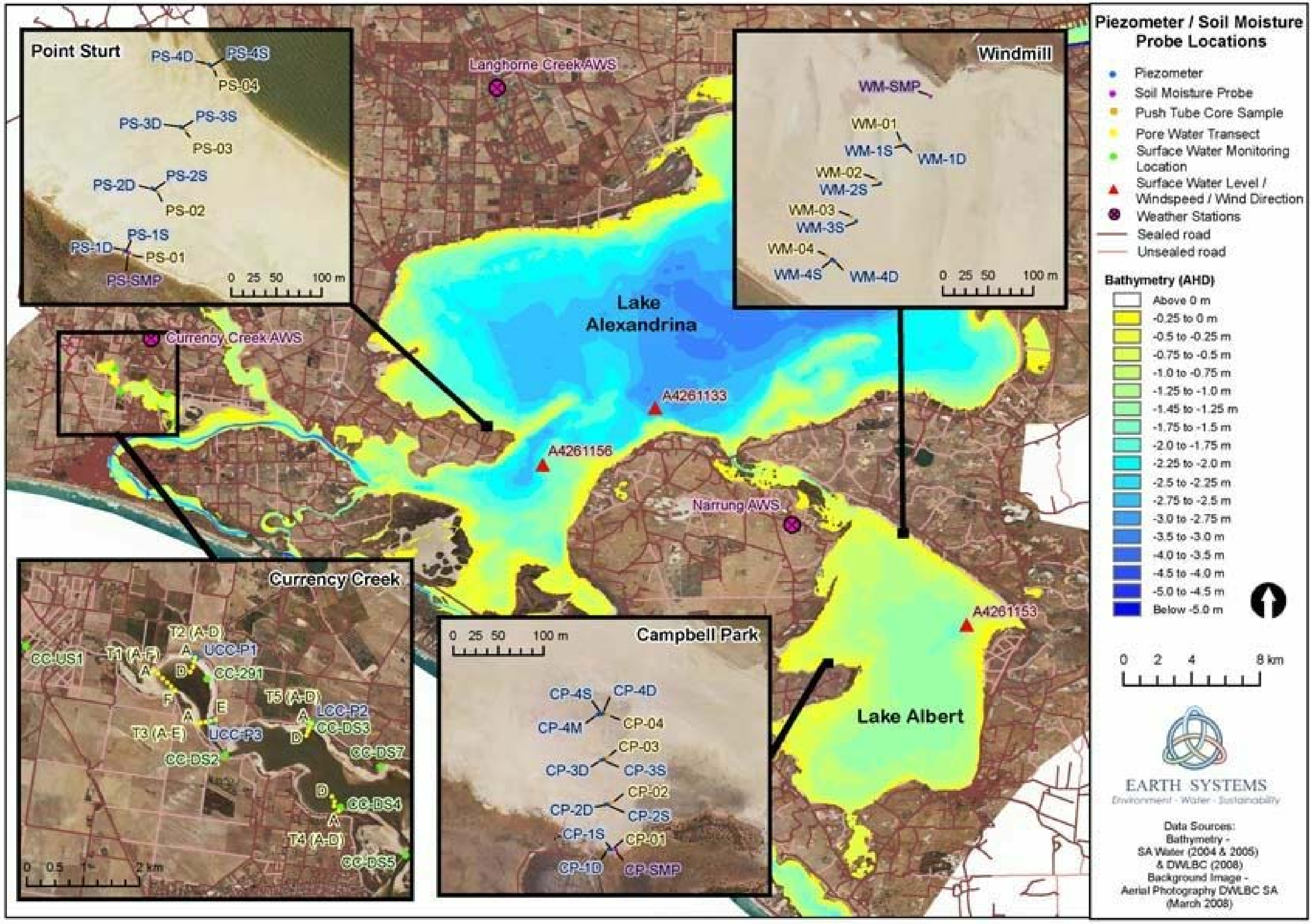


Figure 5. Location of sites with inserts showing transects. This figure was obtained from the report by Earth Systems (Earth systems, 2009).

2.1. Slope of Near-Shore Land Surface and estimation of Lake Water Level

The slope of the land surface in the near shore region can be determined from the position and elevation data in Table 1. Equations for the lake level as a function of distance from point 1 shallow for each site can be deduced (figure 6). These equations can then be used by extrapolation to determine the distance of the lake edge from the piezometers using the lake water level data (Table 2).

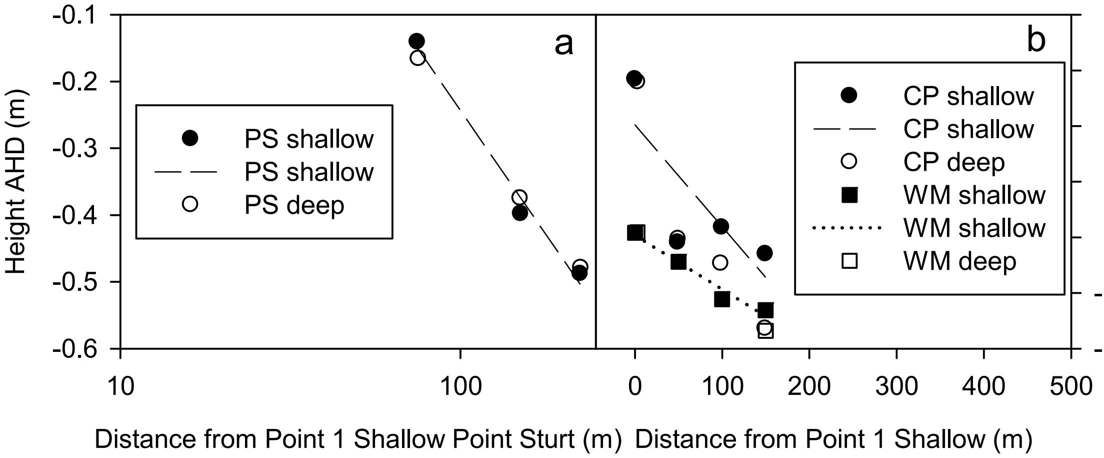


Figure 6. Height of piezometer measurement points (m AHD) with distance from point 1 a) Point Sturt site (semi-log plot), and b) Windmill and Campbell Park sites. Regression equation parameters are given in Table 2.

The relationship of height of the soil surface and distance is can be satisfactorily represented by a linear fit for three of the sites. For Currency Creek the points are not on a transect and so the slope cannot be determined using this approach. For the Point Sturt site a log linear relationship was found to be more appropriate as the slope decreased with distance from point 1.

The efficacy of these equations to represent the lake level was tested by comparing when these equations predicted the shoreline would reach each piezometer with the time lake water level was at the surface elevation of the sediment at each piezometer. The maximum difference in the time when the equations predicted the lake water level would reach each piezometer and that determined from the water level was 1 day. This indicates the equations are a good estimator of the distance of the lake water's edge from point 1 in the piezometer transect.

Table 2. Equations for relationship between height, Y, (m AHD) and distance from point 1, x, for each site along with the correlation coefficient (R2). This relationship is used to estimate lake water edge from point 1.

Site	Form of equation	A	b	R^2
Point Sturt	Y = a + b*log(x)	1.233	-0.738	0.986
Currency Creek* UCC-P1 LCC-P2 UCC-P3	Slope Y=1/350= 0.00286			
Campbell Park	Y = a + b*x	0.202	-1.83×10^{-3}	0.643
Windmill	Y = a + b*x	3.1×10^{-3}	-9.69×10^{-4}	0.970

*Obtained from Earth Systems (2010)

2.2. Horizontal Gradients between the Lake and Surrounding Land

The piezometric head data for each site was filtered to give the nearest value to midnight of each day, so comparisons could be made with the lake level and each piezometer. This is also a more severe test of the concept of water flow from the lake to the soil in the near shore region as any transient effect of daily evaporation will be reduced. This filtering caused little loss of resolution in the data (figure 7). This filtered data was time matched for all points and the lake level data.

The difference in the distance between point 4 in the piezometer transect and the lake water's edge along with the water level and the water level at point 4 was used to calculate the horizontal hydraulic gradient. The hydraulic gradient was also calculated between all points in the piezometer transect using:

$$dh / dx \approx (h_i - h_{i+1}) / (x_i - x_{i+1}) \tag{1}$$

where *h* is the piezometric head (m AHD) or lake water level, *x* is the distance from point 1 in the piezometer transect, and *i* is the point including the lake. When this gradient is negative water will flow from point *i* to point *i* + 1 and in the opposite direction when the gradient is positive.

One of the issues is that the piezometric head is not necessarily the same as the height of the water table because the piezometric head measures the pressure on the water at the depth where the piezometers is screened not the water table position. In sandy soils like those at three of the sites (Point Sturt, Campbell Point and Windmill) and where the slope and flow rate are low the piezometric head and water table heights are likely to be similar. Cook et al. (2009) discuss the latter point in relationship to modelling of ground water flow down sloping surfaces and showed that this was not related to soil type but to the flow rate and slope and is related to the capillarity of the soil. At a low slope (0.1) they showed that there was little difference between the piezometric head and the phreatic surface. The slopes found here are an order of magnitude less than 0.1, so a difference in piezometric head from water table height (phreatic surface) due to capillarity will be negligible. However, other effects such as low conductivity layers above the screen height and air entrapment could still result in a difference between measured piezometric head and water table height.

Another issue that arises with these data is due to the fact that there may also be a density difference in the water at points along the transects due to temperature and salinity differences in the water. This difference in density of the water can also cause the transport of water (Wooding et al., 1997; Simmons et al., 1999). To assess these potential effects in the Lower Lakes equation (1) was modified to consider the effect density on the water flow, as follows:

$$\frac{dh}{dx} \approx \frac{(h_i\rho_i - h_{i+1}\rho_{i+1})}{(x_i - x_{i+1})\overline{\rho}} \tag{2}$$

where ρ_i is the density of the water at point *i* and $\overline{\rho} = \left(\rho_i + \rho_{i+1}\right)/2$ can be considered the reference salinity (Wooding et al., 1997). In order to calculate the gradient with eqn (2) the density needs to be calculated. The density of water is related to both the salinity of the water and the temperature and can be calculated by (McCutcheon et al., 1993):

$$\begin{aligned}
&\rho_{S,T} = \rho_T + AS + BS^{3/2} + C^2\\
&\rho_T = 1000\left\{1 - \frac{\left(T + 288.9414\right)\left(T - 3.9863\right)^2}{508929.2\left(T + 68.12963\right)}\right\}\\
&A = 0.824493 - 4.0899\text{x}10^{-3}T + 7.6438\text{x}10^{-5}T^3 - 8.2467\text{x}10^{-7}T^3 + 5.3675\text{x}10^{-3}T^4 \quad (3)\\
&B = -5.724\text{x}10^{-3} + 1.0227\text{x}10^{-3}T - 1.6546\text{x}10^{-6}T^2\\
&C = 4.8314\text{x}10^{-4}
\end{aligned}$$

where $\rho_{S,T}$ is the density of water (kg m^{-3}) for, salinity (*S* (g kg^{-1})) and temperature (*T* (°C)) and ρ_T is the density of water in relation to *T*.

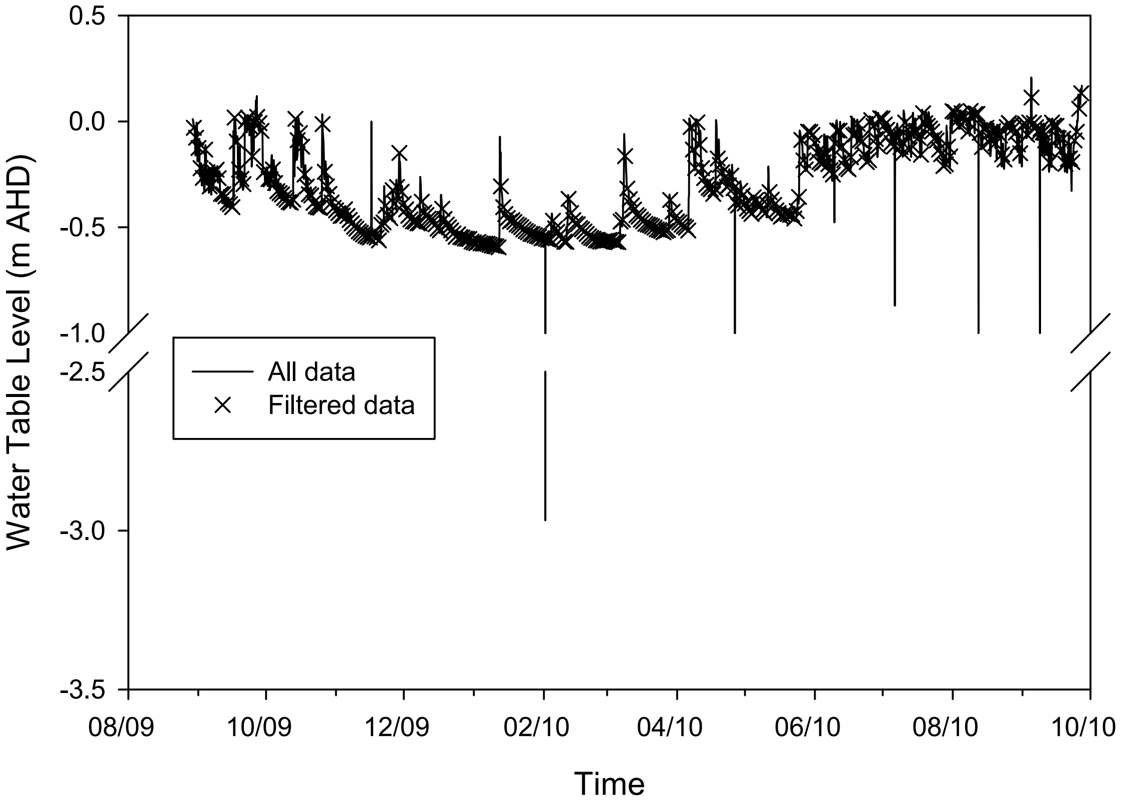


Figure 7. Comparison of raw water table height data and filtered data (midnight values) for point 1 at the Windmill site. Note break in y axis to show transient noise in raw data.

2.2.1. Water Table Gradients for Point Sturt site

The water table levels and lake level show that initially the lake level is above points 3 and 4 so we expect water to flow from the lake to the near shore region (figure 8). The water table level at point 2 is initially similar to the lake level and point 1 greater than the lake level. These initial water table levels and water table levels at selected dates are illustrated in figure 9.

The transect on the 15/10/2009 (figure 9) was following approximately 17 mm of rain which had raised the water table levels above the lake water level for all points. The transect on 17/01/2010 is at the extreme of drying out of the lake with the distance from point 1 now estimated to be over 1.2 km. The other two transects show that the lake has filled and the distance to the shoreline is now estimated to be less than that to point 4 (19/5/2010) and almost up to point 2 by 10/6/2010.

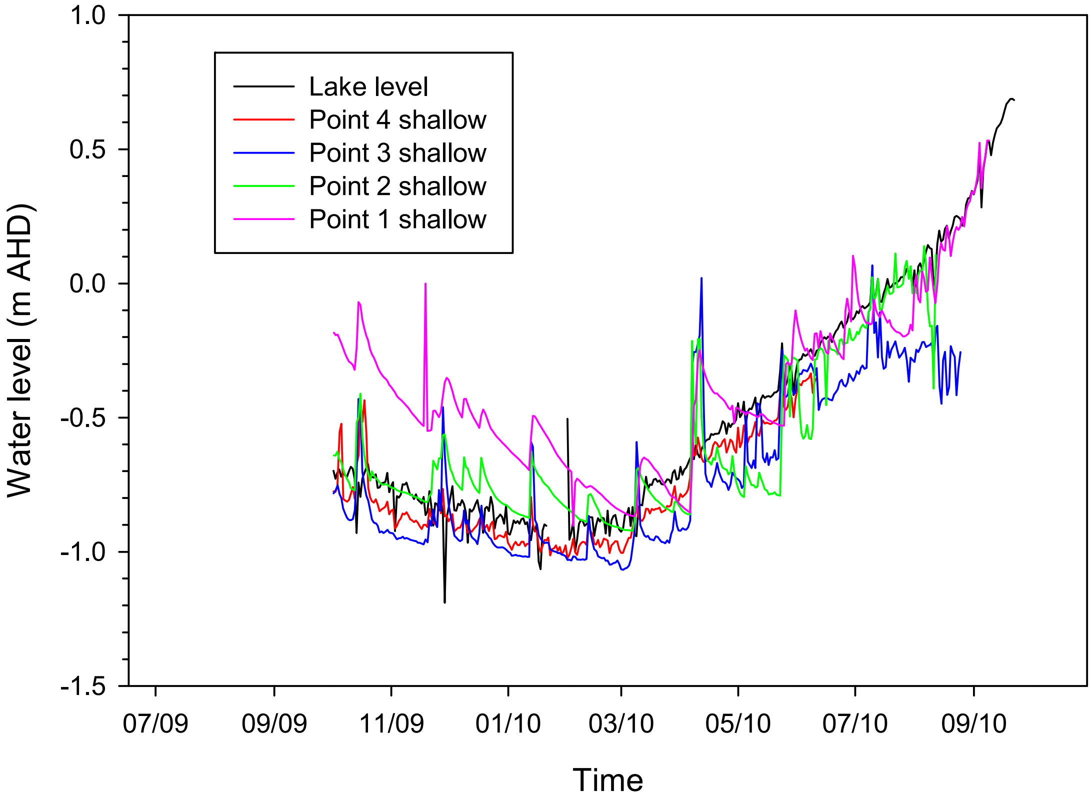


Figure 8. Water table and Lake levels for Point Sturt site.

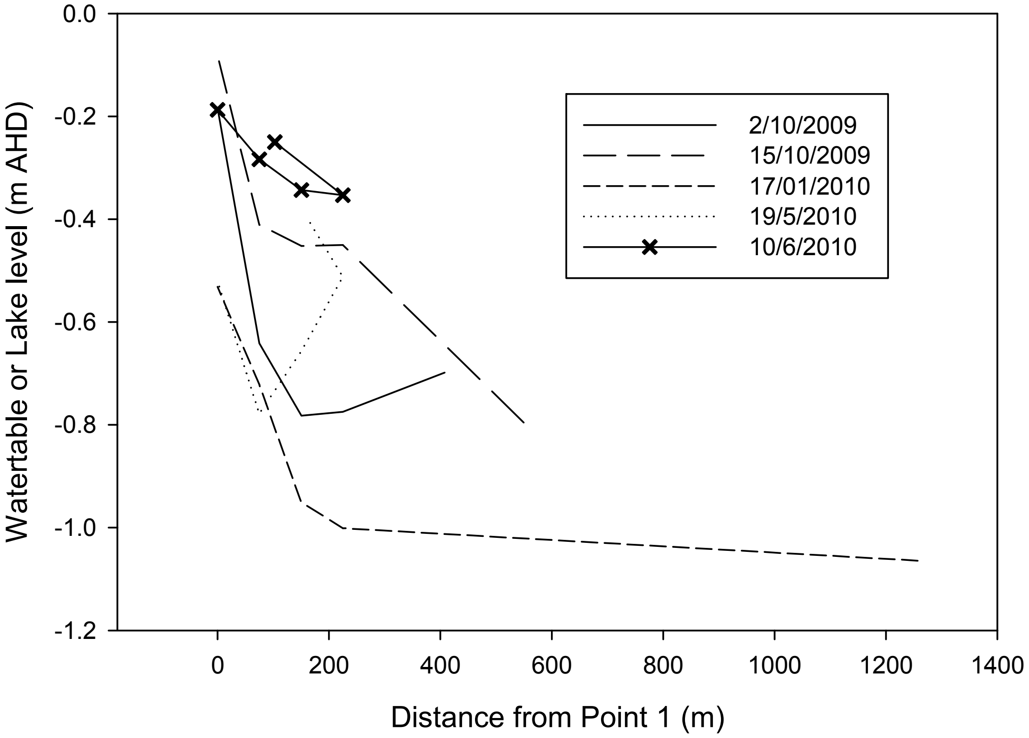


Figure 9. Water table heights and lake level with distance from point 1 at selected dates for Point Sturt site.

The horizontal hydraulic gradients were calculated from the piezometric height data and on days when the salinity and temperature had been measured the gradients were calculated with the density effect taken into consideration. The temperature data for the piezometers showed that all temperatures move in concert with time for all three points but point 1 which is further away from the lake showed a higher temperature from January 2010 until the end of monitoring (figure 10a). Point 1 generally had the lowest salinity and this did not vary much during the monitoring period. The salinity of the water was measured as electrical conductivity (mS cm^{-1}) and in order to use eqn (3) needs to be converted to salinity in units of g kg^{-1}. This conversion was achieved using $S = 0.64EC$ (www.dpi.nsw.gov.au/agriculture/resources/soils/salinity/general/measuring). The coefficient of 0.640 can vary from 0.5 to 0.7 depending on the composition of the solution, but as the composition at all the piezometers is similar the use of a constant value of 0.64 should not greatly affect the results. The density was then calculated with eqn (3) and the hydraulic gradients were calculated using eqns (1) and (2).

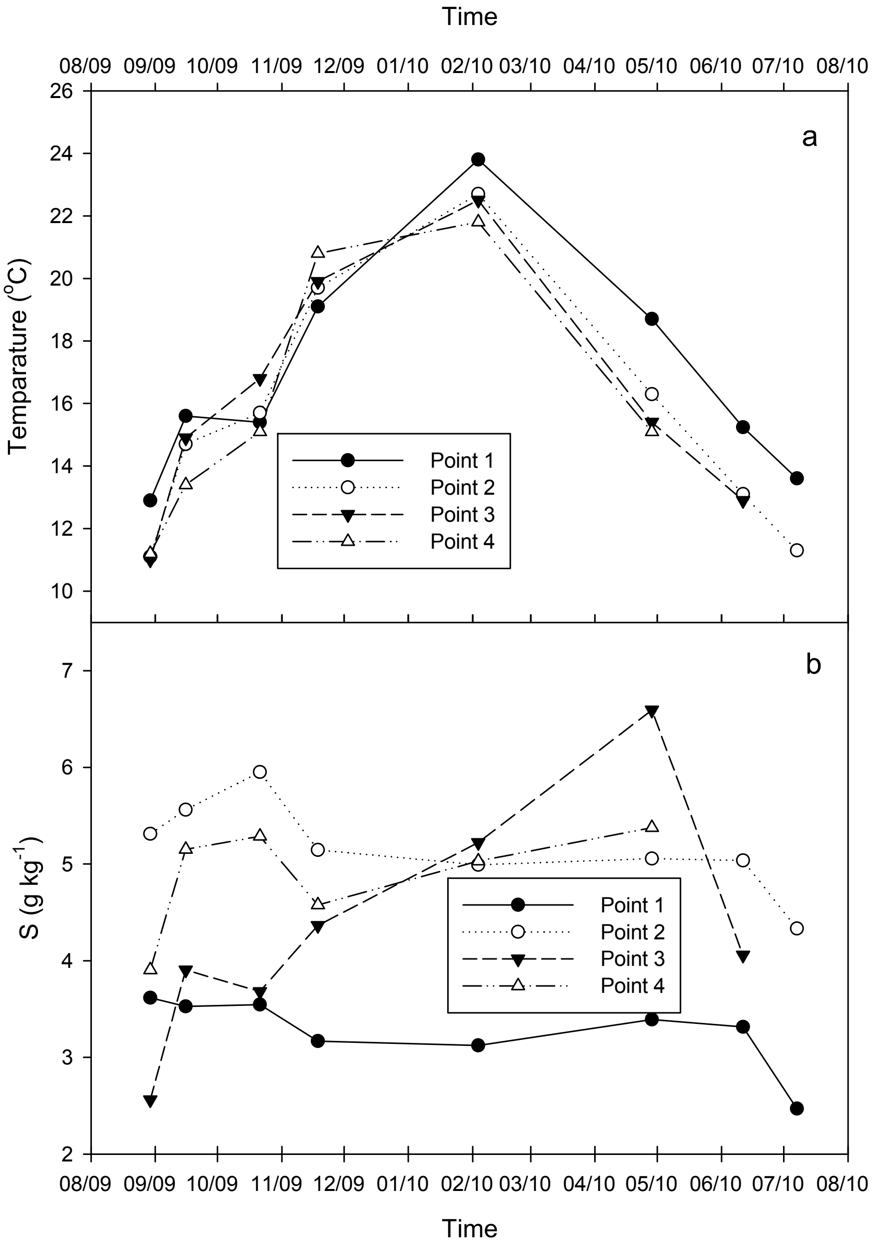


Figure 10. Temperature of water (a) and salinity (b) for the points in the transect at Point Sturt site with time.

The density effect on the hydraulic gradient was insignificant for the gradient between all points (figure 11). The horizontal gradient is hence due predominantly to the piezometric head.

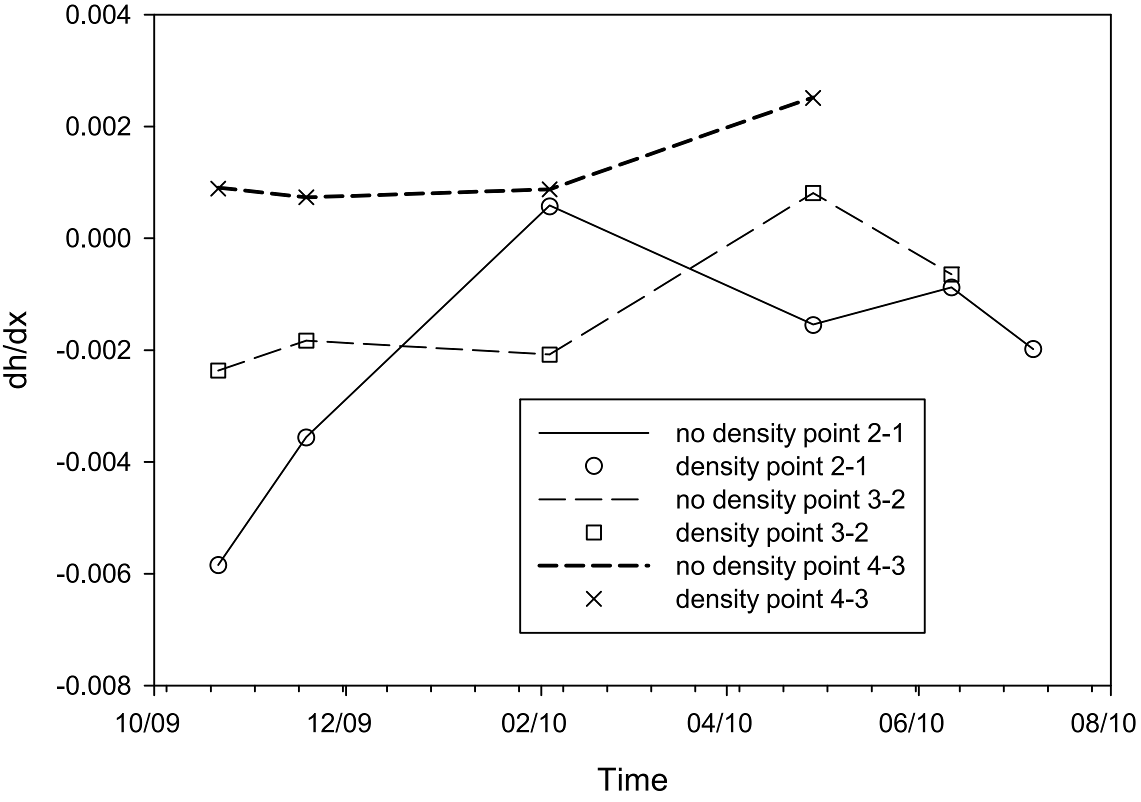


Figure 11. Comparison of the horizontal potential gradient with and without considering density effects for points along the transect at Point Sturt site.

The horizontal hydraulic gradient between the lake and point 4 is for most of the time positive indicating the flow is from the lake to the near shore sediments (figure 12a,b). This is probably due to the evaporation of water from this wetted region of the shoreline.

The capillary rise was estimated at 0.21 m (see section 3) which for the slope in the region of point 3 and beyond would represent a length of approximately 150 m of the shoreline. Somewhere less than this distance will be acting as an evaporating surface for the lake water. This distance will reduce to approximately 30 m for the slope between points 1 and 2 when the lake level rises into this region..

The horizontal gradients show that the water flow is mainly from point 2 to point 3 and point 1 to point 2 (figure 12c,d). This indicates that acidity in the water from these regions is likely to flow into the region from point 2 to point 4 and accumulate by evaporation possibly at the surface as was suggested in the conceptual model (section 1.2.1). These gradients are all low which means that the fluxes of acid to the lake are also likely to be low as there is little gradient to drive the flow. The exception to this was when the lake level during refill approached point 4 and the distance between the waters edge tended towards zero which caused a spike in the gradient away from the lake (figure 12a).

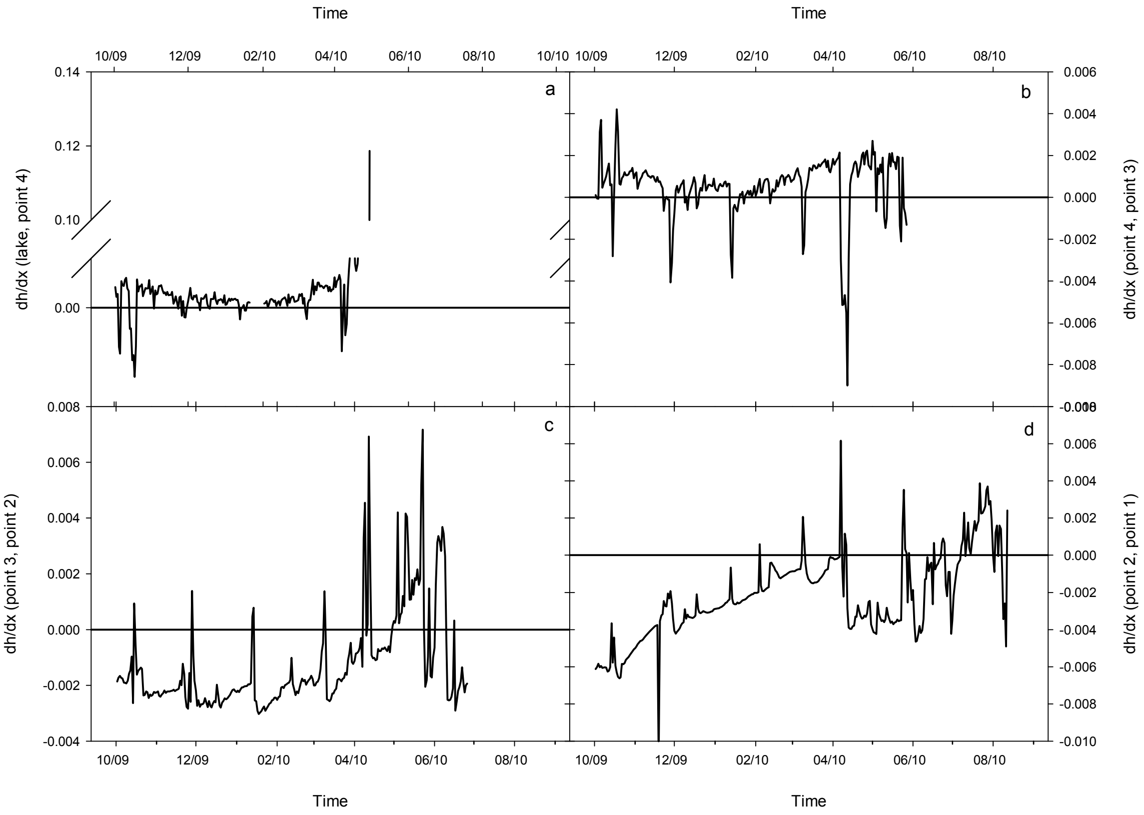


Figure 12. Horizontal piezometric head gradients for Point Sturt site for : a) Lake to Point 4, b) Point 4 to Point 3, c) Point 3 to Point 2 and d) Point 2 to Point 1. Note scale break on y axis of panel a.

2.2.2. Water Table Gradients for Campbell Park site

The water table height and lake level data for the Campbell Park site (figure 13) showed similar responses to the Point Sturt site. The Lake level shows a rapid rise in late September 2010 which meant the piezometers were inundated and no data from them is recorded after this date. At points 3 and 4 the water table level is generally lower than the lake level until about June 2010. While at points 1 and 2 the water level is generally greater than the lake most of the time.

The water table height transect at selected times shows that at the start of monitoring the flow was from the lake towards point 4 (29/8/2009) and also from points 1, 2, 3 towards point 4 (figure 14). The last date where all piezometers were measured (9/8/2010) shows that the levels at points 3 and 4 have risen by approximately 1 m since the start of monitoring and that there is a gradient towards the lake from point 4. The transect on 15/10/2009 which also shows a consistent gradient from all points towards the lake occurred after 9 mm of rainfall. The lake reached its lowest level in late January 2009 and the transect on 29/1/2009 shows that the estimation of waters edge is now greater than 1 km from point 1. The transect is very flat on 17/6/2010 as the lake level was rising.

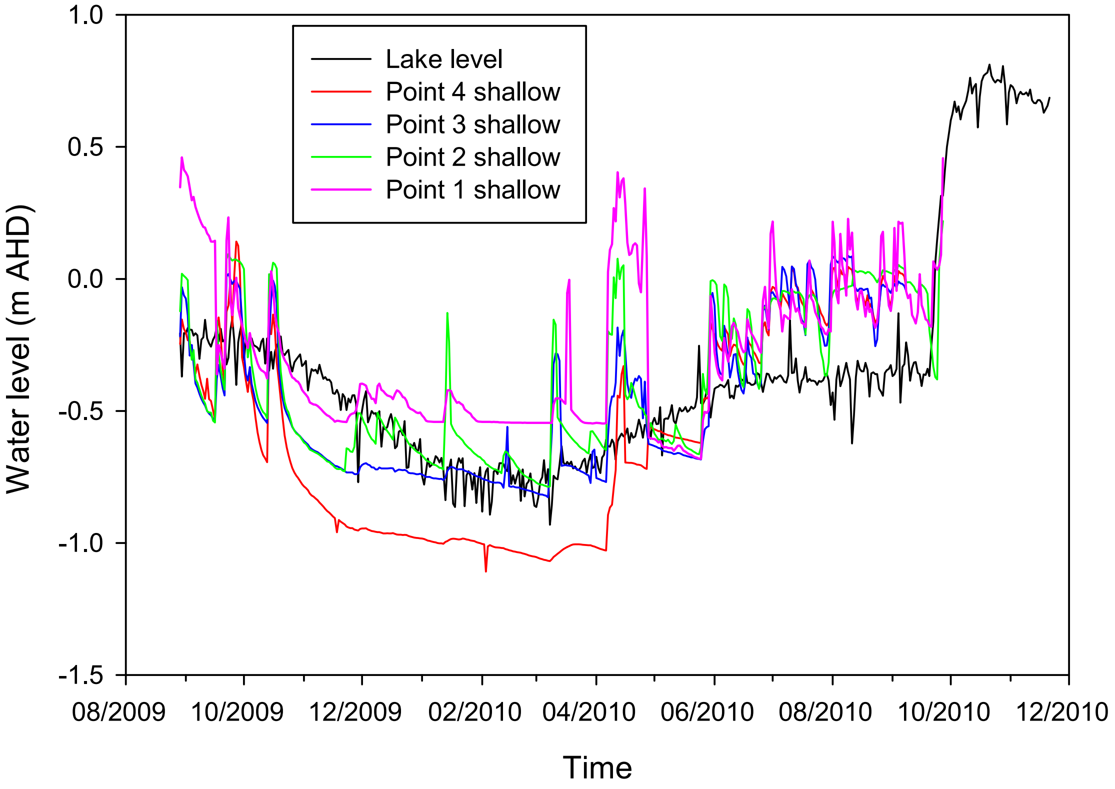


Figure 13. Water table and Lake level with time for Campbell Park site.

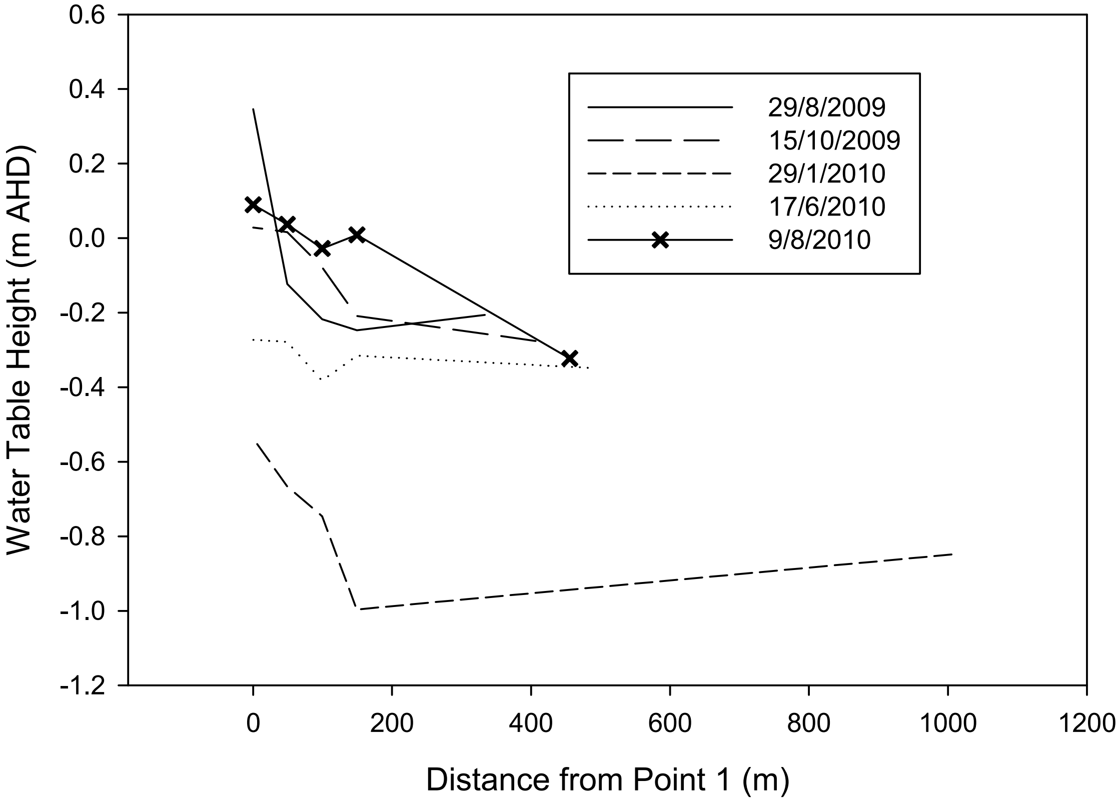


Figure 14. Water table height with distance from point 1 at selected times for Campbell Park site.

The temperature and electrical conductivity of the water was measured during the monitoring period (figure 15). The water temperatures showed a similar range to those found at the Point Sturt site but there was a greater separation of the temperatures between the sites during the summer of 2009-10 with the temperatures becoming similar again in autumn (figure 15a). The salinities at Campbell Park site are generally greater than those found at Point Sturt site and show a decrease from March 2010 until the end of monitoring with the exception of one point at point 2 and the last point at point 1 (figure 15b).

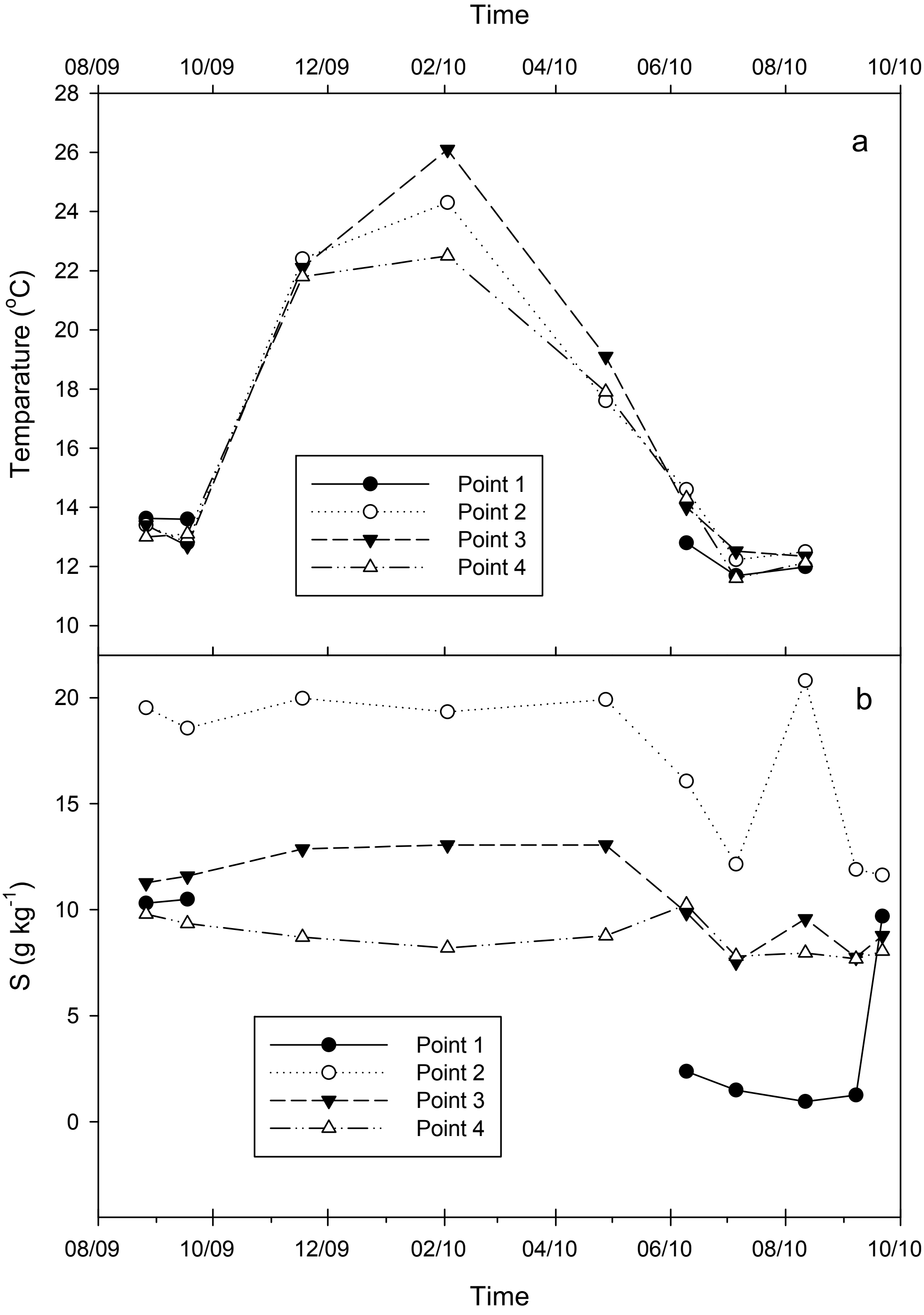


Figure 15. Temperature of water (a) and salinity (b) for the points in the transect at Campbell Park site with time.

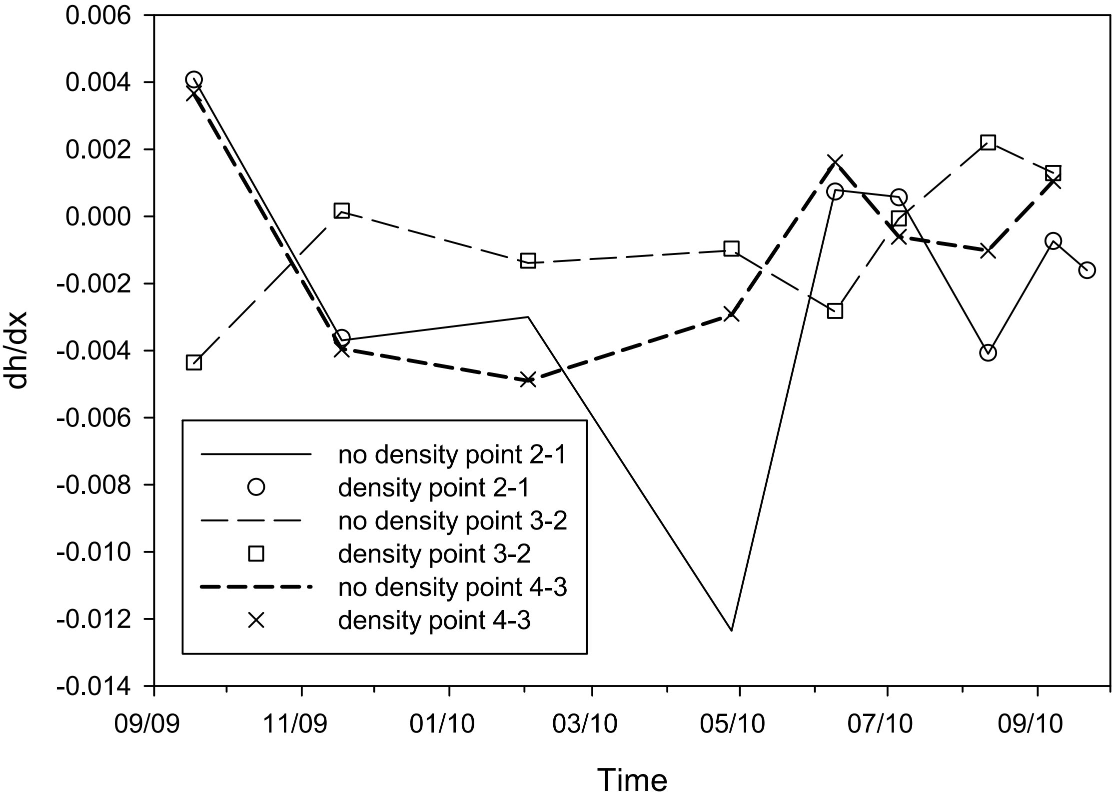


Figure 16. Comparison of the horizontal potential gradient with and without considering density effects for points along the transect at Campbell Park site.

Even though the salinity was higher for the water in the Campbell Park piezometers the difference between these values at all points is not sufficient to cause a significant change in the hydraulic gradients from those calculated with the piezometric head alone (figure 16).

The horizontal gradient shows that the gradient between the lake and point 4 is positive for most of the time with flow from the lake to point 4, but changes to negative in late May 2010 (figure 17a). This is due to the rise in the lake level and also an increase in the water table on the land surrounding the lake due to increased rainfall. The gradient between points 3 and 4 is mainly negative until late May 2010 when the gradient oscillates around zero (figure 17b). This means that the flow is mainly towards point 4 from the landward side as the gradients for point 3 to point 2 and point 2 to point 1 are also mainly negative. These gradients are however all small which means that fluxes of water are likely to be small horizontally.

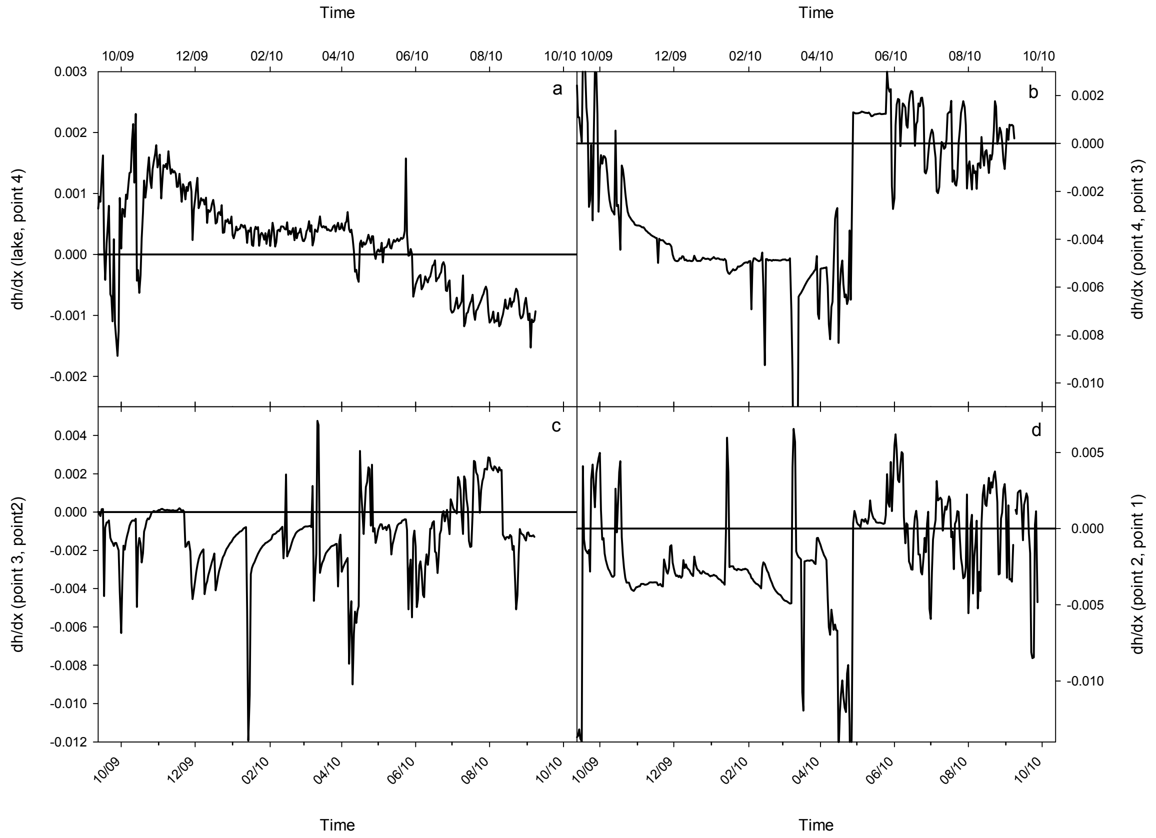


Figure 17. Horizontal gradients of piezometric head for Campbell Park site for: a) lake to point 4, b) point 4 to point 3, c) point 3 to point 2, and d) point 2 to point 1.

An estimate of the capillary rise for the Campbell Park site (see section 3) is 0.72 m which is large. This would suggest that given the slope at the Campbell park site that the distance that soil could be saturated away from the shoreline could be up to 750 m. In reality it is likely to be less than this as the hydraulic conductivity of the soil and the evaporation rate will reduce this distance. However, it does suggest that the soil is likely to be saturated for a considerable distance from the water's edge and hence the acid formation in this region is likely to be restricted.

2.2.3. Water Table Gradients for Windmill site

At the Windmill site the lake level range is not as great as at the Point Sturt and Campbell Park sites (figure 18). The piezometer water table levels closely follow the lake water level. Points 4 and 3 are generally less than the lake water until early 2010 and then all four piezometers are generally greater than the water level for the rest of the time. Seiching was suggested to have occurred at this site around the 25 October 2009 (Earth Systems, 2010).

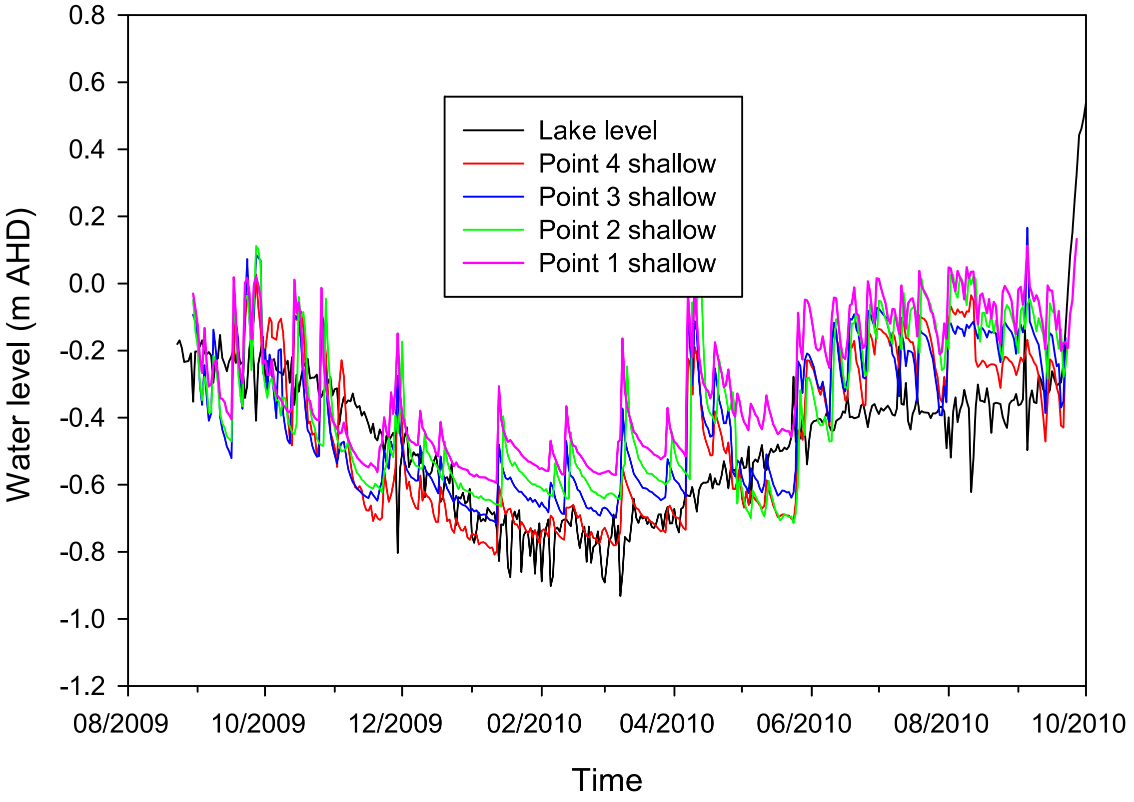


Figure 18. Water table and lake level with time for Windmill site.

The water table height transect at selected times shows more clearly these trends in water table compared with the lake water level (figure 19). At the start of recordings (16/9/2009) there is a gradient in water height from the lake towards points 3 and 4 and a gradient in water height from points 1 and 2 toward point 3. On the 27/9/2009 the gradient is more towards the lake as a result of 32.5 mm of rainfall immediately prior to this time. When the lake was at its lowest level there was still a gradient towards point 4 even though the lake water edge was estimated to be greater than 600 m away from this point. As the lake filled (20/5/2010) the gradient in water height was still in the direction from the lake to point 4. By the end of the monitoring period (21/9/2010) the lake water level was greater than the piezometric head in all four piezometers and the shoreline was estimated to be beyond point 4.

The temperature and electrical conductivity of the water was measured during the monitoring period (figure 20). The water temperatures showed a similar range to those found at Point Sturt and Campbell Park. The data for point 1 is discontinuous and shows a higher value prior to and lower value after this discontinuity (figure 20a). The salinities at Windmill are generally greater than those found at Point Sturt site and similar to Campbell Park site. They show a decrease from April 2010 until the end of monitoring and there is a greater spread in values between the points in the transect at this site than the other sites (figure 20b).

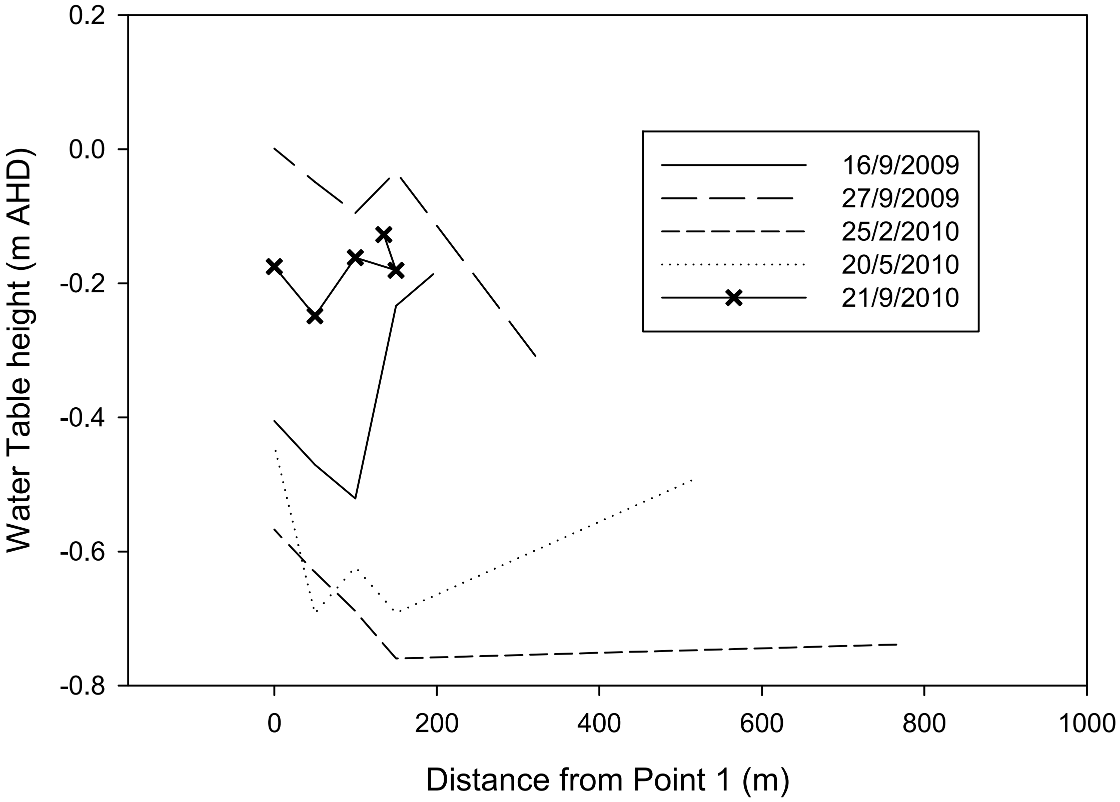


Figure 19. Water table height versus distance from point 1 for Windmill site at selected times.

Even with the difference in salinity between points 2 and 3 (figure 20) the effect of water density on the horizontal hydraulic gradient is not significantly (figure 21).

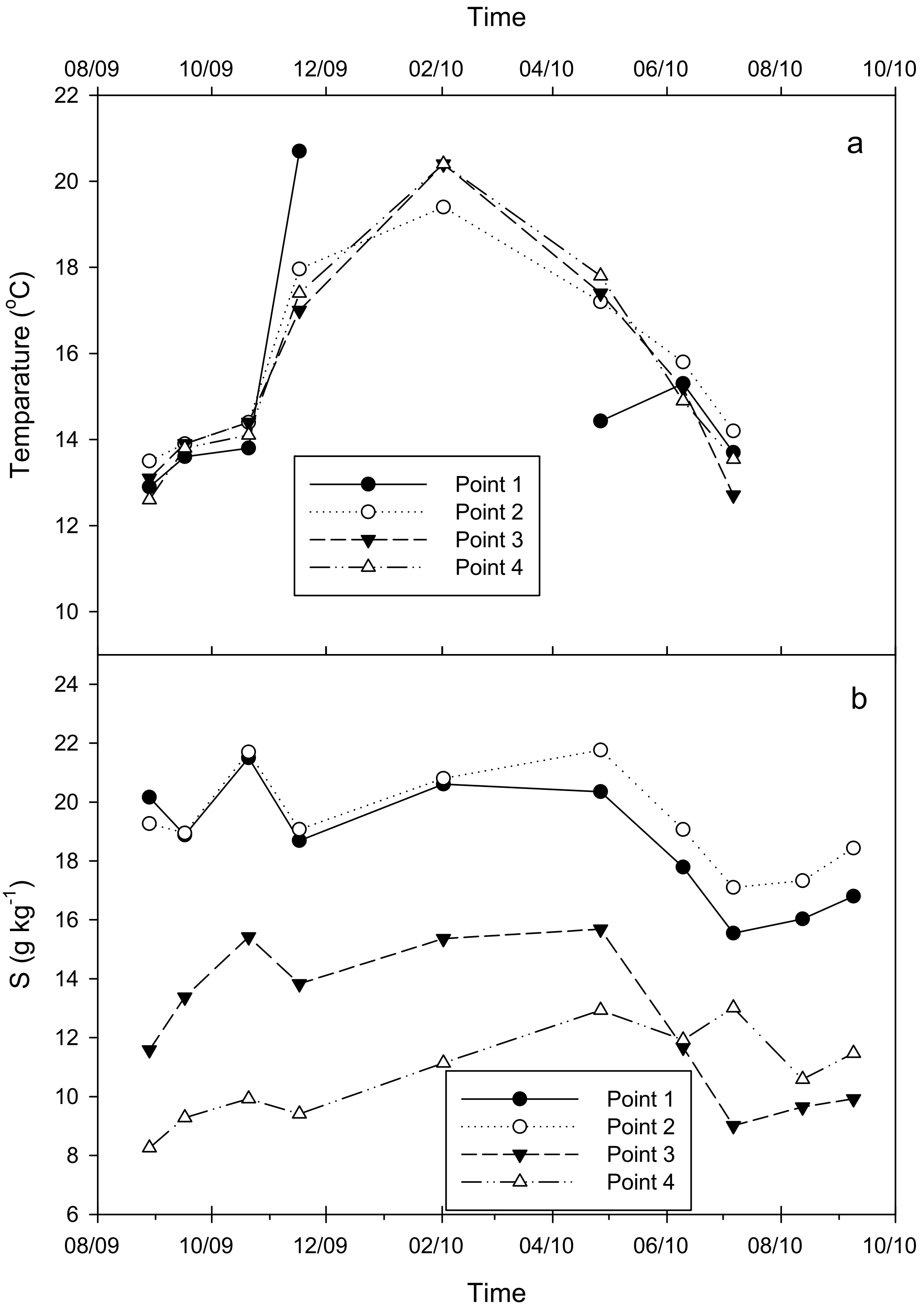


Figure 20. Temperature of water (a) and salinity (b) for the points in the transect at Windmill site with time.

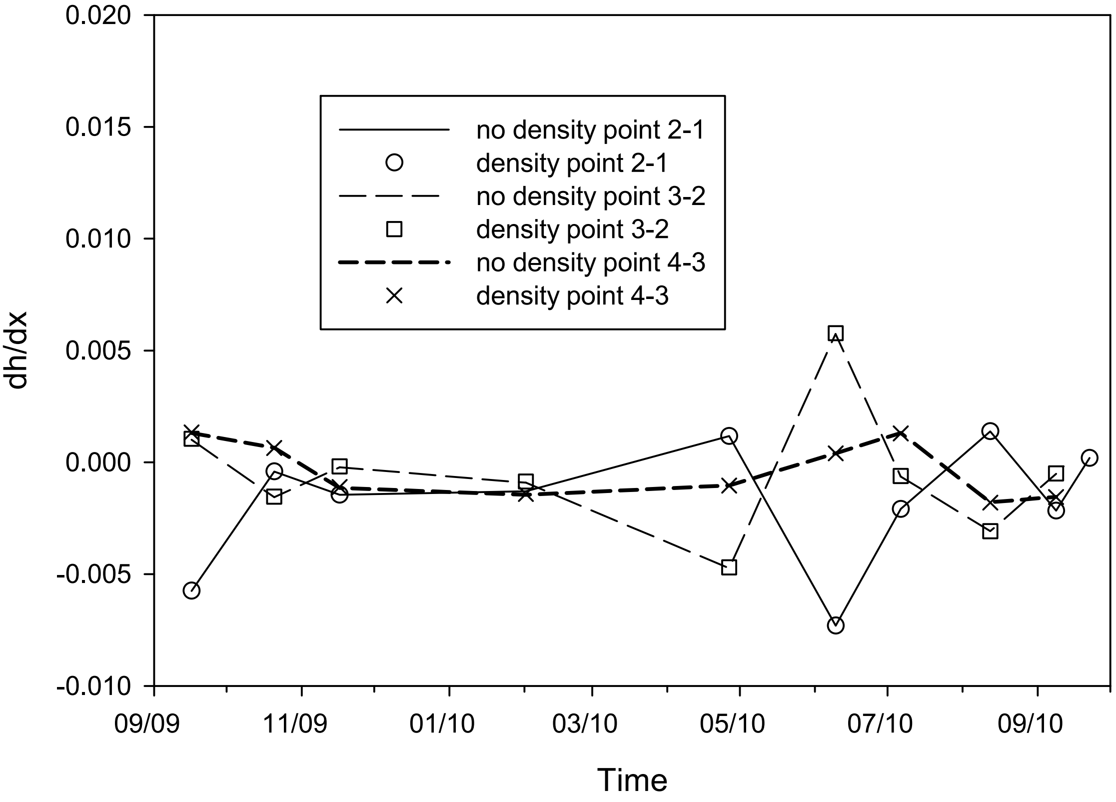


Figure 21. Comparison of the horizontal potential gradient with and without considering density effects for points along the transect at Windmill site.

Horizontal gradients between the lake and point 4 with time show large fluctuations in the gradient initially but generally positive gradients until early January 2010 (figure 22a). After this time the gradient is negligible, but even prior to early January 2010 the gradient is small. This would suggest that there was little flux of water from point 4 to the lake at this site. Again the initial gradients suggest evaporative losses in the region from the lake to point 4 and as far as point 3 (figure 22b). This proposition is supported by the increase in salinity seen at point 4 in figure 20. The gradients between the points in the transect for all the points at this site are small suggesting little lateral movement of water.

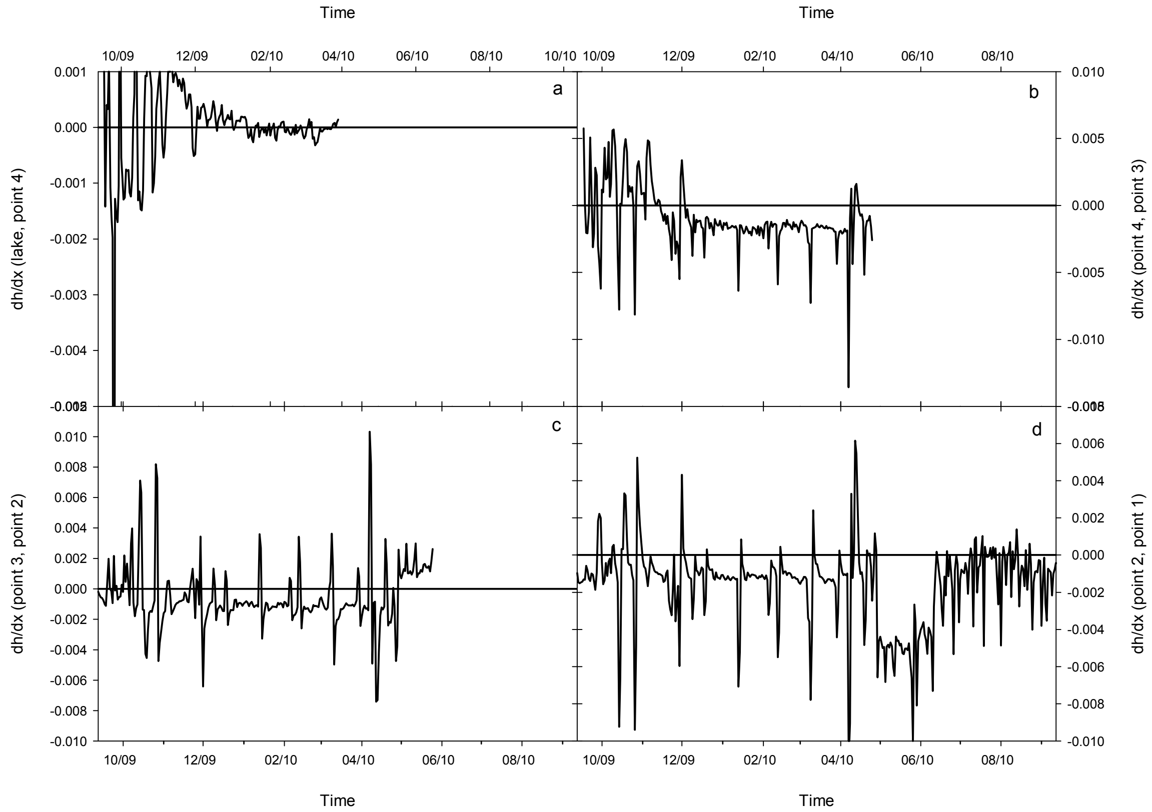


Figure 22. Horizontal gradients of piezometric head for Windmill site for: a) lake to point 4, b) point 4 to point 3, c) point 3 to point 2, and d) point 2 to point 1.

The estimated capillary rise for the Windmill site was 0.12 m (see section 3.3.1) and that along with the slope of near shore region suggests that the distance that could be saturated away from the waters edge could be 124 m. Water is likely to evaporate at close to potential rates in this region and the oxidation of the pyritic sediments will be substantially reduced due the saturation of the sediments.

2.2.4. Water Table Gradients for Currency Creek site

At Currency Creek there were only two piezometers installed and continuously monitored. The lake level data recording starts after the recording of the water table level of the piezometers (figure 23). These piezometers were not in a transect and so no slope data can be obtained from their height and position data. However, Earth Systems (2010) did estimate the slope to be 1/350 and this can be used to access the water level in Currency Creek from each site but not the gradient as this will just be the assumed slope of 1/350. This slope does give an estimate of the gradient of -2.86×10^{-3}, which again suggests that there is only a small gradient to transmit water flux to Currency Creek.

Initially the water table in the piezometers is greater than the water level in Currency Creek, so we would expect a flow of water from the land to the lake. At approximately 6/10/2009 with a raising of the water level in Currency Creek the levels become the same and at the peak in the water level in Currency Creek, at about mid November 2009, the water level in the Creek is greater than in the piezometers.

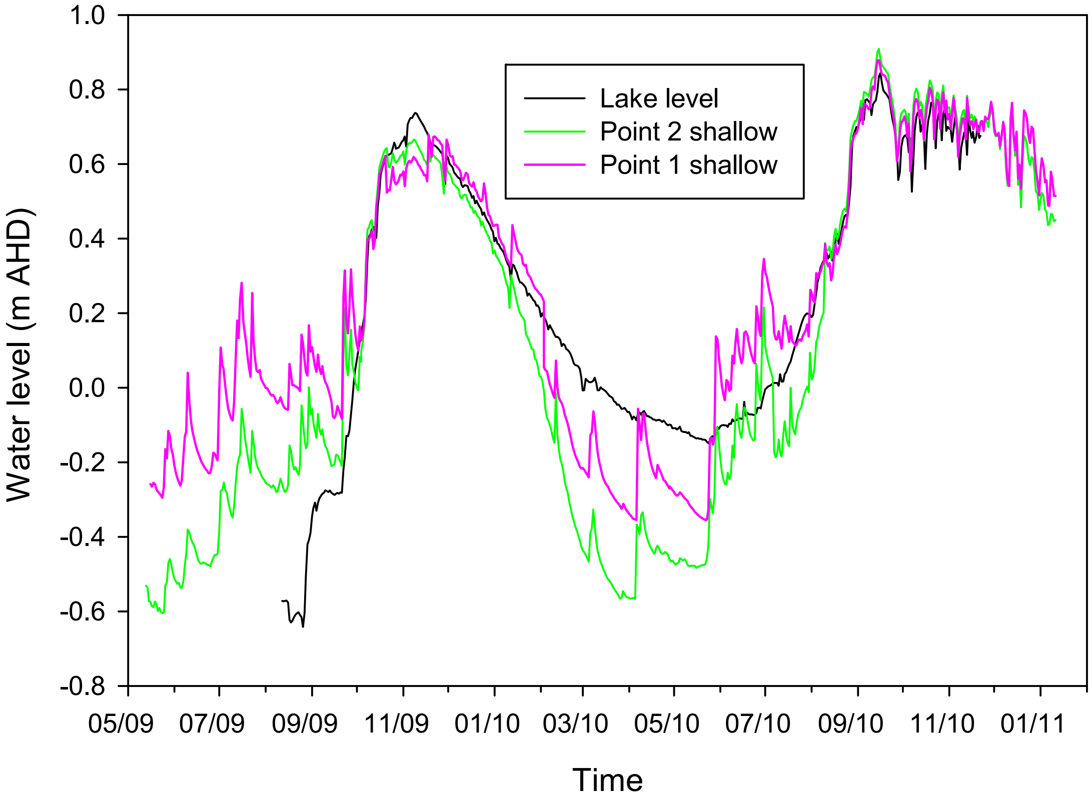


Figure 23. Water table for piezometers and water levels in the Creek at Currency Creek site.

The water level heights continue to fall along with the Creek water and with the Creek water level greater than or at similar heights in the piezometers for the rest of the time except for a brief period from June 2010 to August 2010 when the water level at point 1 is greater than the Creek level. However, even during this time the water level at point 2 is mainly less than the Creek level, so evaporative accumulation of and water flow is likely to be occurring from the Creek waters edge to at least point 2 for all of this time. From August 2010 until the end of monitoring, water levels in the Creek and piezometers are at similar levels, so little if any flow is expected.

No water table gradients or horizontal gradients can be calculated for this site as no data exists. However, data on the temperature and electrical conductivity (EC) of the water in the piezometers is available and shows similar values of temperature for both piezometers (figure 24a). However, the salinity derived from the EC data is greater at point 2 than point 1 (figure 24b). The salinity values at both points are less than some of the values at Campbell Park and Windmill sites and greater than those at the Point Sturt site. The salinity at each site shows no particular trend with time (figure 24b).

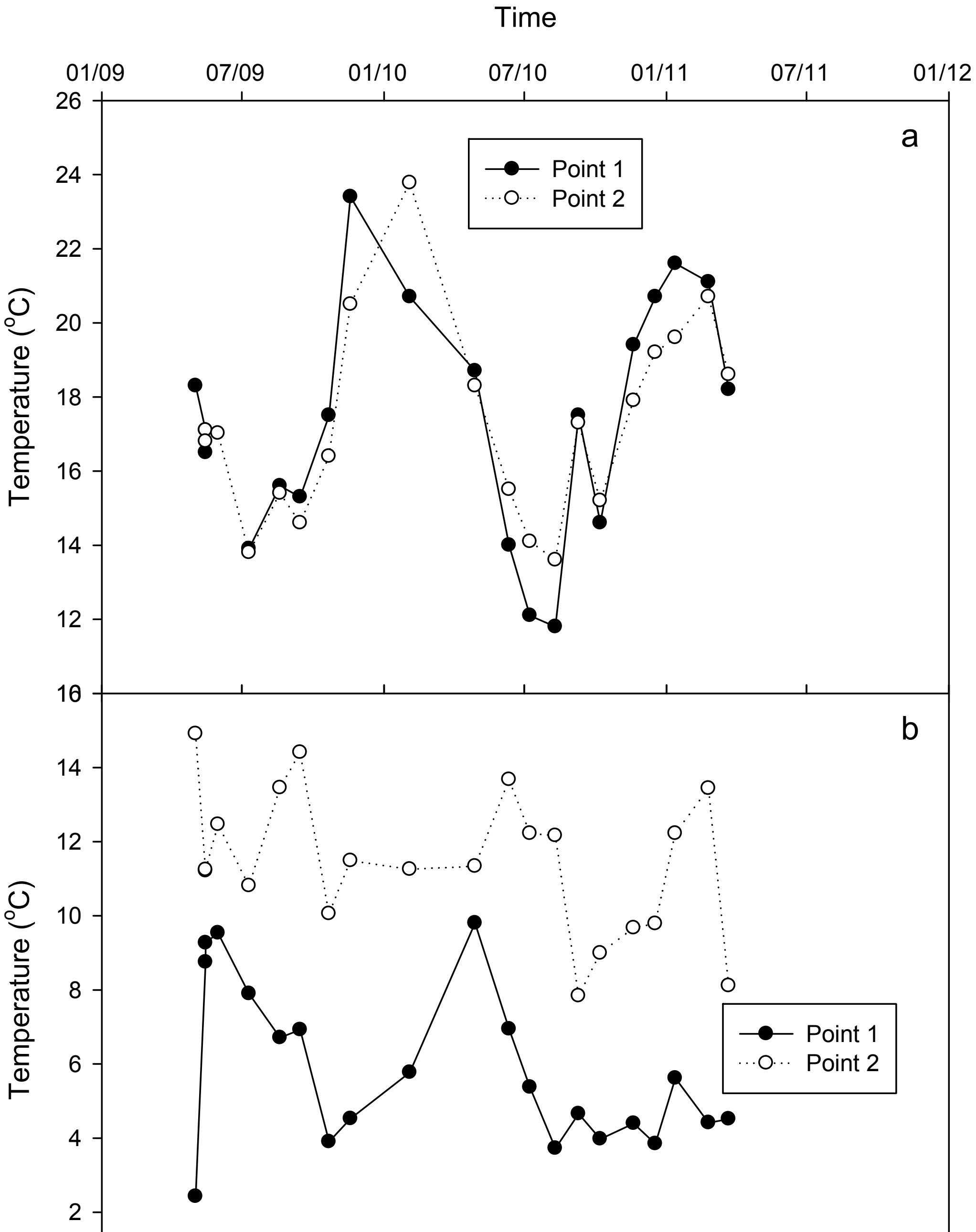


Figure 24. Temperature of water (a) and salinity (b) for the points in the transect at Currency Creek site with time.

Due to no water content measurements at this site no estimate of the capillary rise is possible.

2.3. Exfiltration and Runoff Events

The data provided was examined to see when the water table rose above the surface at each of the points where the water table height was measured. This assumes that the piezometric head is equal to the water table height. As mentioned earlier this is not always true, so events may be under or over-estimated by this assumption. However, given the sandy nature of these soils which can be seen in rapid response of the piezometric height to rainfall this assumption is likely to be valid. When the water table rises to or above the surface exfiltration (water flowing out of the soil) will occur and also any rainfall occurring at this time is likely to generate runoff. There may have also been a seiching event at the Windmill site that also caused exfiltration in October 2009. During these times any acidity that has accumulated at or near the soil surface due to evaporative concentration is likely to runoff into the lake.

These analyses rely on the water content data provided by Earth Systems (2010) which were measured with capacitance probes. These probes can be inaccurate in soils with high clay contents (Kellners et al., 2004) and can result in an offset bias in the measured water content. The soils where these sensors were installed are mainly sands especially, in the top 0.5 m, where these sensors are situated. Due to the frequency of the measurements these sensors are sensitive to salinity when combined with high clay contents (Kellners et al., 2005). They are also sensitive to installation as these sensors are strongly weighted to a small zone of soil close to the access tube (Kellners et al., 2004). There does not appear to have been any *in situ* calibration of these sensors, which means that the absolute values of water content may have a bias in them. However, for most of the measurements we are interested in it is the difference in the water content values, which are of interest and these should not be affected by this bias. This means that the magnitude of the water content changes recorded are likely to be valid.

2.3.1. Exfiltration and Runoff Events at Point Sturt site

From the elevation and piezometers head data the height of the water table above the soil surface can be estimated. This along with the rainfall allows estimation of likely runoff and exfiltration events. The rainfall was measured locally at each site and is relevant for that site. The amount and even the occurrence of rainfall varied between the sites.

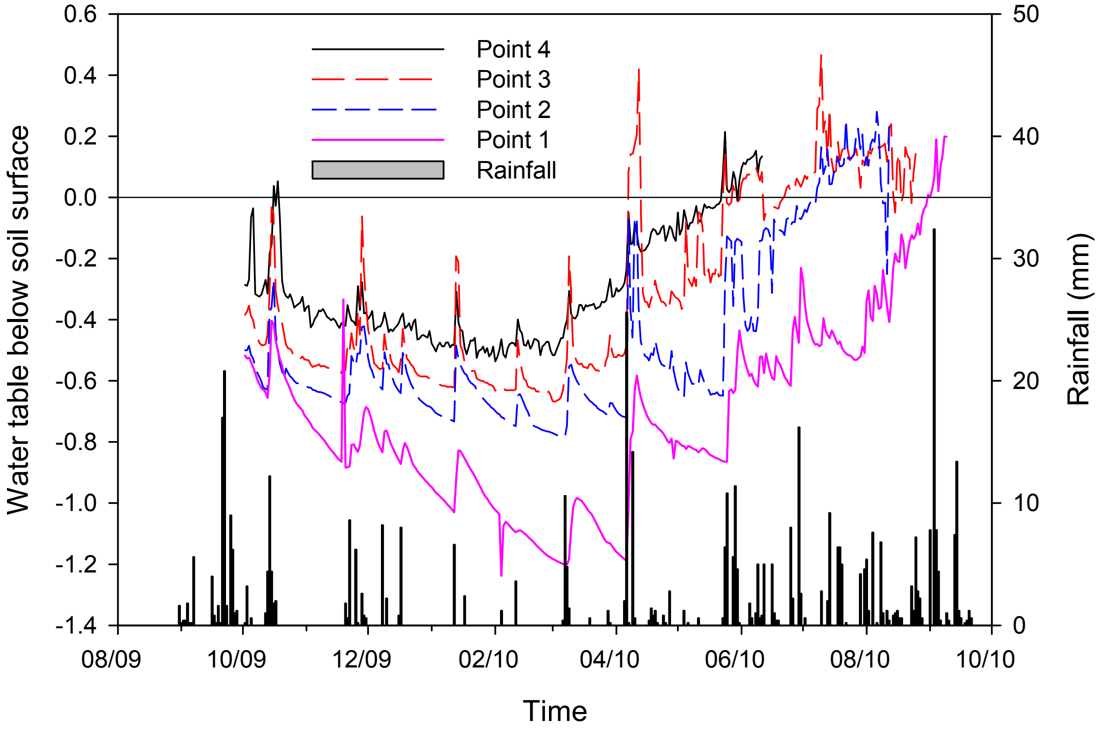


Figure 25. Water table heights relative to the soil surface and daily rainfall for the four measurement points at the Point Sturt site.

The highest daily rainfall rate is 32 mm day-1 this is unlikely to cause runoff at approximately this rate as this rainfall rate is greater than the infiltration rate at this site. However, a possible runoff event due to saturation excess at point 4 looks to have occurred around the 16th of October 2009 (figure 25) and at point 3 between the 6th and 13th of April 2010. This latter event at point 3 is not indicated to have occurred at point 4. Points 1, 2 and 4 show a rapid rise in water tables due to rainfall of 25.6 mm on the 6th and a further 14.2 on the 9th, but the water table is not estimated to reach the surface The response at point 3 could be due to change in slope of the sediments which is occurring at about point 3. The runoff/exfiltration indicated at point 3 may be due to transmission of water pressure from up-slope, but was insufficient due to the lower slope between points 3 and 4 to cause the same rise at point 4. We can see the progressive lake level rise and corresponding water level rise at each of the points with the water table reaching the surface progressively from point 4 to point 1 as the lake inundates these points. This progression starts with the water table level reaching the surface at point 4 on 21/5/2010, at point 3 on 20/6/2010, at point 2 on 6/7/2010 and lastly point 1 on 30/8/2010.

2.3.2. Exfiltration and Runoff Events at Campbell Park site

The water table in relation to the soil surface with time and rainfall shows that exfiltration possibly occurred on a number of occasions at the Campbell Park site (figure 26). The first of these is indicated not long after monitoring started on 27/8/2009 at point 1 and continued for a number of days until 8/9/2009.

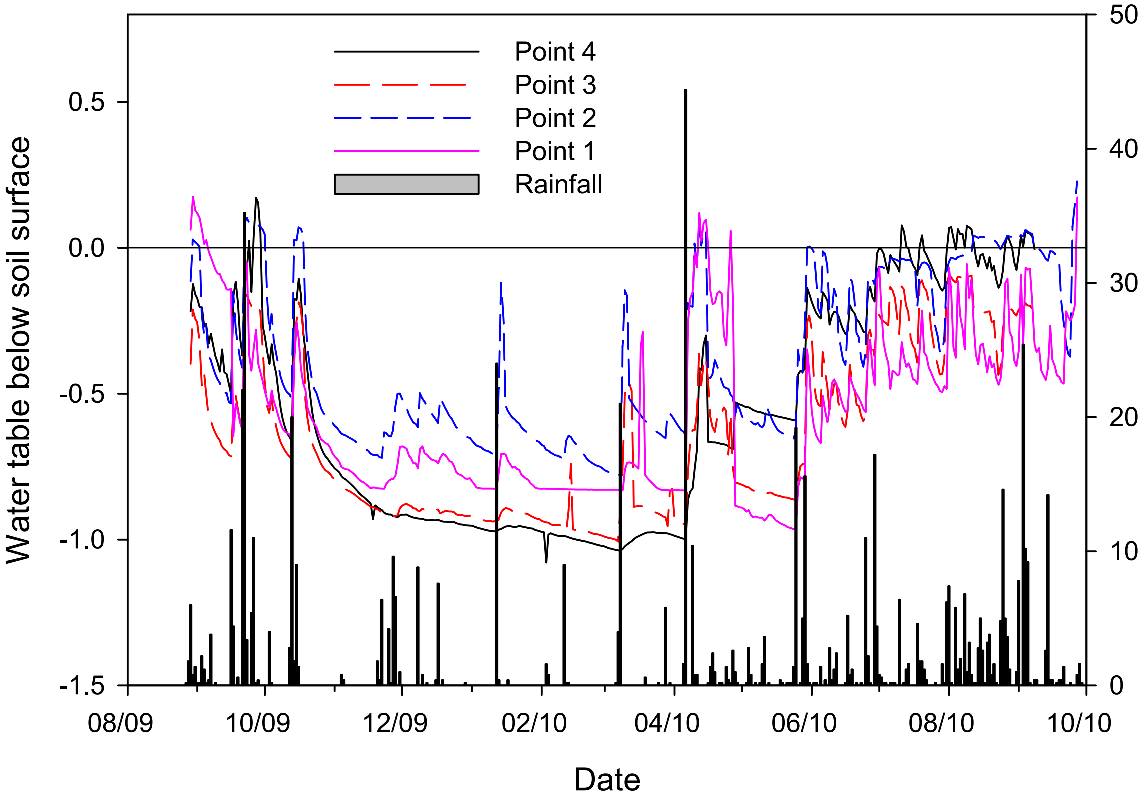


Figure 26. Water table heights in relation to soil surface and rainfall during monitoring for Campbell Park site.

Further exfiltration and possible runoff occurred at point 2 and point 4 following rainfall of 22 mm and 35.2 mm on 22nd and 23rd September 2009. At point 4 the data indicates that exfiltration occurred from 25/9/2009, while at point 2 this started earlier 22/9/2009 and finished later 1/10/2009.

Further exfiltration at point 2 and point 1 is indicated from 11/4/2010 for about 5 days following a large rainfall event of 40.4 mm on the 6/4/2010. Given that the water table took a couple of days to rise to the surface this is possibly caused by water pressure from upslope.

The water table at all sites rises towards the surface as from April 2010 as an increase in the frequency of rainfall occurs. Exfiltration events at points 4 and 2 are indicated during this time until late September 2010 when the lake level rises sharply and inundates these sites (figure 26).

2.3.3. Exfiltration and Runoff Events at Windmill site

At the Windmill site the water table is close to the surface or above for points 2 to 4 in the first two months of monitoring (September to November 2009) (figure 27). This is caused by 22 and 35.2 mm of rain on consecutive days on 21 and 22 September 2009 respectively. A further period of exfiltration and/or runoff occurs for points 2 to 4 after a 44.4 mm rainfall event on 6/4/2010.

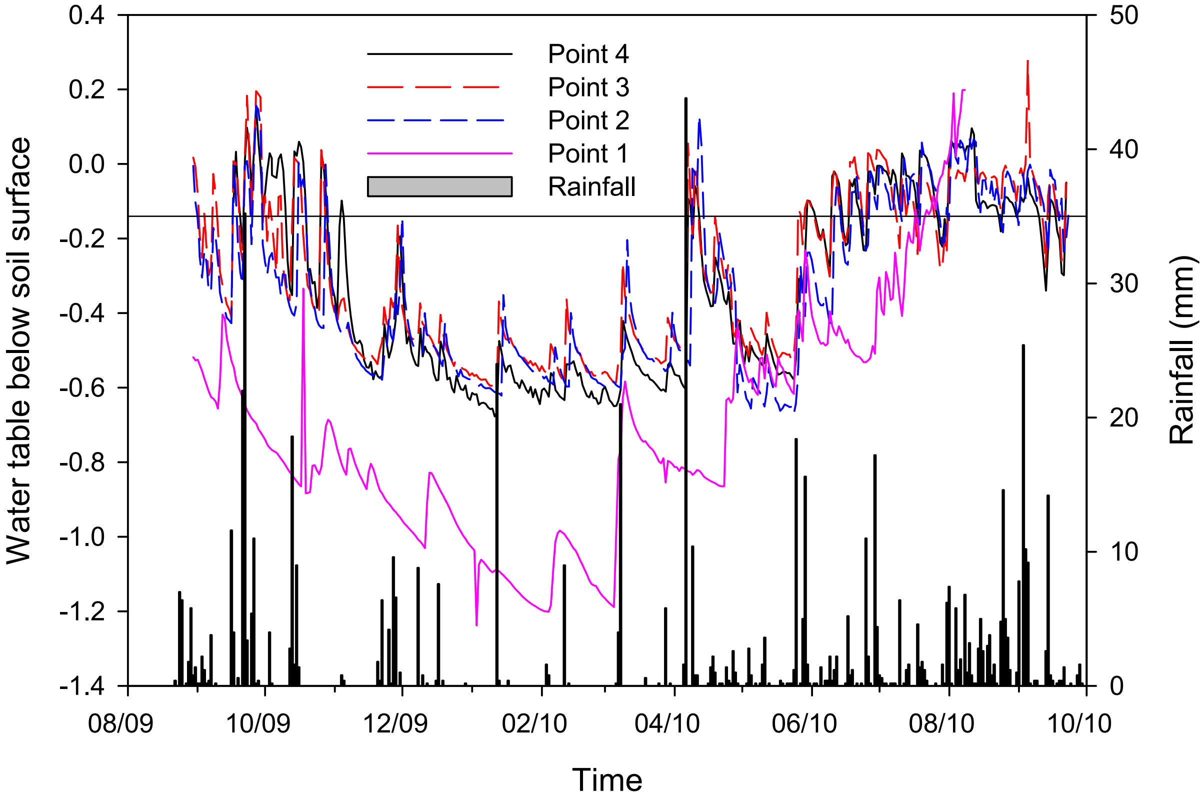


Figure 27. Water table heights in relation to the soil surface and daily rainfall for Windmill site.

During a sustained period of rainfall from April 2010 until the end of monitoring the water table at points 2 to 4 is at or above the soil surface for most of this time and runoff and exfiltration would have occurred for most of the points along the piezometers transect during this period. The water table at point 1 reaches the surface during this period in late July 2009.

2.3.4. Exfiltration and Runoff Events at Currency Creek site

There are two distinct periods of exfiltration and runoff possible for the Currency Creek site, the first from early October 2009 to early January 2010 and the second from early to mid August 2010 until the end of monitoring (figure 28). However, both these periods are associated with the water level in Currency Creek being at or above the water table at these points so only on a few days is runoff and exfiltration likely to occur. The first period is associated with the rise in water tables due to a 35.2 mm rainfall event on 22/9/2009 and followed by 18.6 and 8.1 on 13/10/2009 and 14/10/2009 respectively. The water level in Currency Creek tended to rise in concert with the rise in the piezometer water levels and so these sites would have been inundated at or close to the time when exfiltration would have occurred. Runoff from this site is more likely to be due to infiltration excess due to the low hydraulic conductivity measured at this site (Table 15) and is discussed further in section 4.

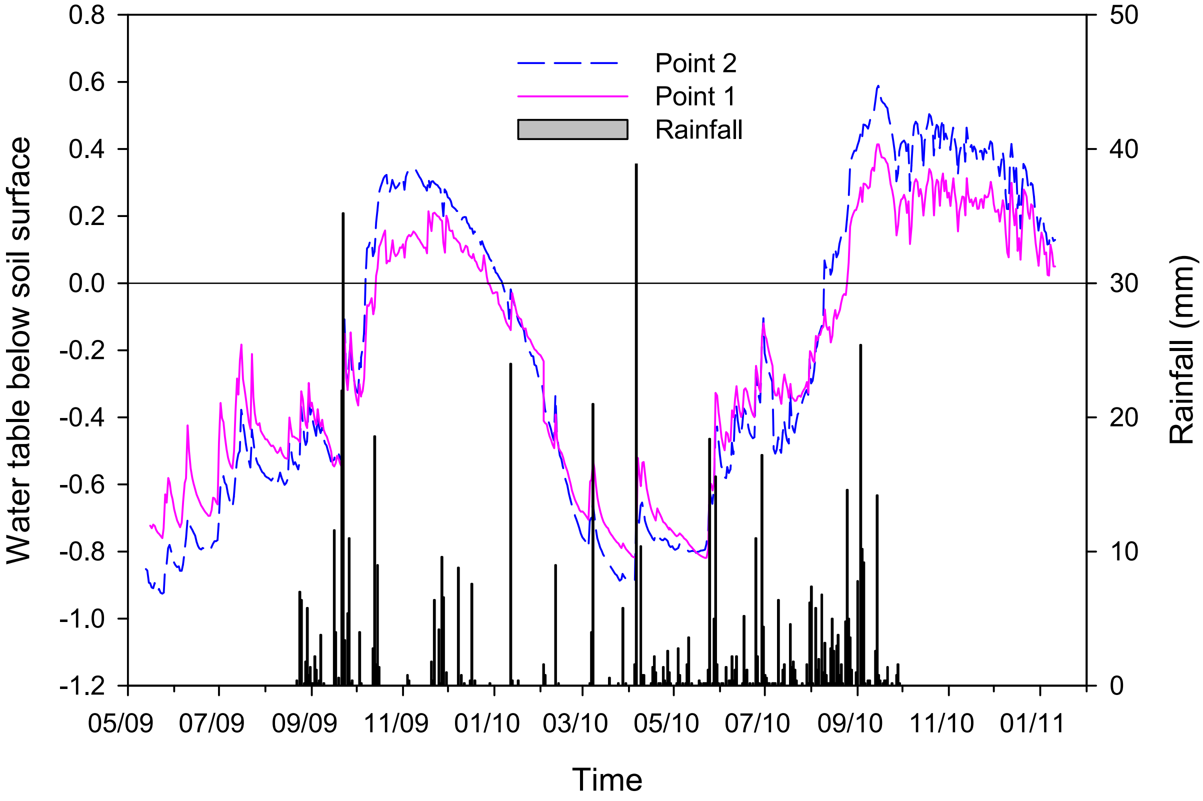


Figure 28. Water table heights in relation to the soil surface for piezometer points and rainfall at the Currency Creek site.

3. WATER CONTENT ANALYSIS

At three of the sites; Point Sturt, Campbell Park and Windmill, the water content at depths up to 0.5 m in 0.1 m increments were measured by Earth systems (2010) adjacent to point 1 in the piezometer transects. These measurements can be used for a number of purposes which will be outlined below. Firstly we can get some idea of the capillary fringe thickness and hence the macroscopic length scale (White and Sully 1987) from when the water content decreases from the saturated value during a period when the water table is declining.

The water table height is also the depth where the water potential is equal to zero and for heights above this point (z), when there is hydraulic equilibrium the potential (ψ) can be estimated as (Jury et al. 1991):

$$\psi = z - L \tag{4}$$

where *L* is the water table depth from the surface (m). This can then be used along with the water content data to get information on the relationship between water content and potential, which can then be used in the modelling. Not only does this data give the draining relationship between water content and potential but the wetting curve is also able to be obtained. This approach relies on the piezometer head being equal to the water table height, which as discussed earlier is a reasonable assumption in these sandy soils.

Combined with the rainfall and water table rise an estimate of the specific yield (water input to the water table/rise in water table height) of the sediments can be made from the estimation of the amount (depth) of water stored in the soil. These values are useful in assessing how much water input is required to raise the water table to a desired height.

Firstly the data was filtered similar to the filtering used for the piezometric data with the nearest measurement to midnight, on a particular day, selected. This is so the transients associated with evaporation during the day will have relaxed and the profile is likely to be at equilibrium. An illustration of the diurnal variation for the data for the Point Sturt site during drying and wetting sequences shows that the use of midnight as the point of comparison form water content and water potential is valid, as both values are close equilibrium at this time (figure 29).

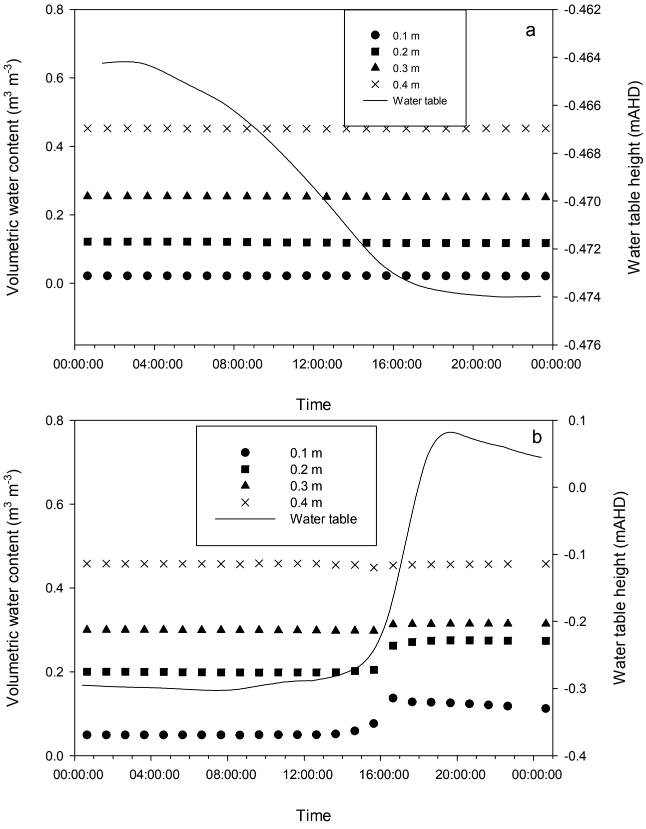


Figure 29. Examples of the diurnal variation in volumetric water content for the four depths and water table height for Point Sturt site; a) during a drying sequence on 11/10/2009 and b) a wetting sequence on 21/9/2009. The midnight values used would be the last points in each data set.

3.1. Water Content Analysis for Point Sturt site

3.1.1. Capillary Length Scale

The potential at equilibrium with the water table at each of the four depths was calculated from the piezometric data using eqn (4). This was plotted against the water content data (figure 30) and where possible the capillary length was estimated from the potential when the water content deviates from the saturated values. Even if the absolute value of the water content is not correct due to lack of calibration the value of the capillary length will be not be affected as it is the water content deviation from the 'saturated' value which determines the capillary length.

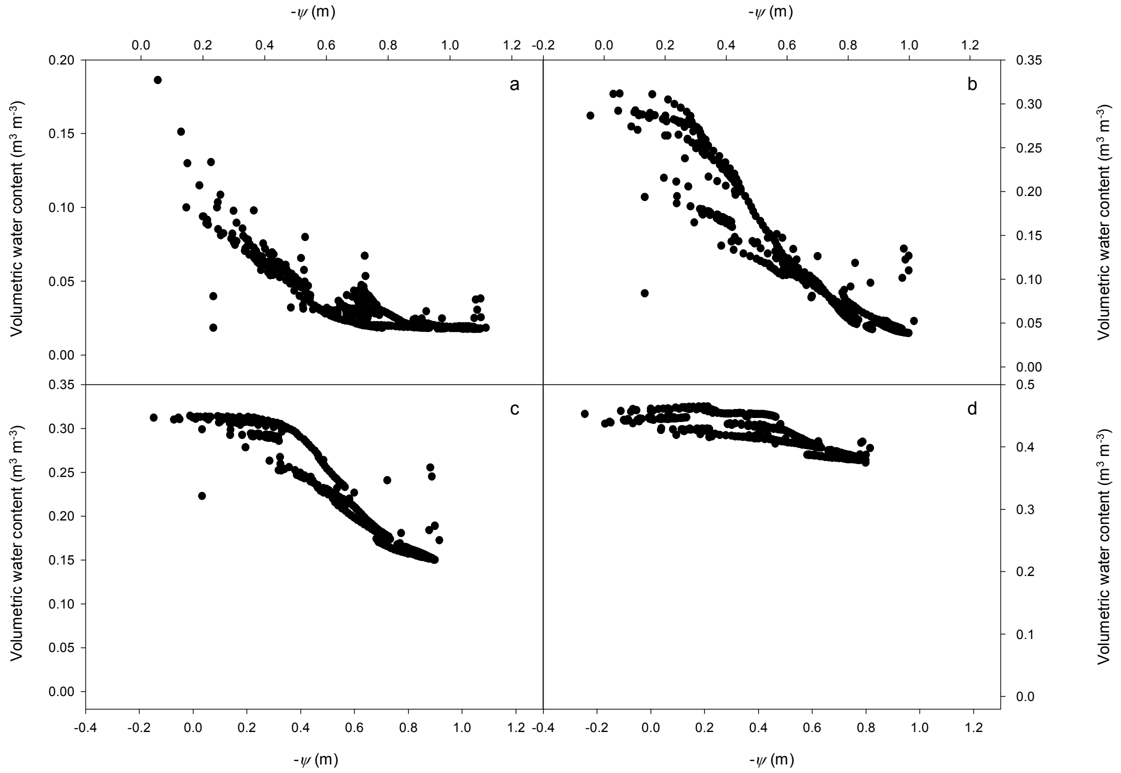


Figure 30. Potential estimated from water table height and eqn (4) with water content for point 1 at Point Sturt site for a) 0.1 m depth, b) 0.2 m depth, c) 0.3 m depth, and d) 0.4 m depth.

The estimates of the capillary length scale (λ) increase with the depth (Table 3) and may reflect an increase in finer textured materials in the sediment with depth. This suggestion of finer texture sediments with depth is supported by the data in (Earth systems 2010),. These results suggest that the water table has to be deeper than 0.86 m before the sediment at a depth of 0.4 m will become unsaturated. Fitzpatrick (2010) observed that mostly the oxidised zone was 0-0.4 m depth range except in well drained sands.

Table 3. Estimation of capillary length scale (λ) for soil at point 1 at Point Sturt site. The value of λ at 0.1 m depth could not be determined (nd)

Depth (m)	λ (m)
0.1	nd
0.2	0.21
0.3	0.30
0.4	0.46

The water table depth before unsaturated conditions occur (critical depth) will be 0.6 m for a depth of 0.3 m and 0.41 m for a depth of 0.2 m. Using these values the proportion of the time the water table was predicted to be below each of these depths and the sediment conditions were likely to be conducive to oxidising conditions, can be estimated during the time when the water table heights were monitored (Table 4). These results show that this time decreases with the proximity of each measurement point to the lake. This proportion of time when oxidising conditions could occur dropped from 82% at a depth of 0.2 m at point 1 to 0 at depths below 0.3 m at point 4. This is consistent with the conceptual model of acid generation discussed in section 1.

Table 4. Percentage of time the water table was below the critical depth at selected depths at each piezometer measurement point for the Point Sturt site.

Point	Depth (m)	% of time water table below critical depth	Total number of monitoring days
1	0.2	82	347
	0.3	57	
	0.4	24	
2	0.2	64	301
	0.3	32	
	0.4	0	
3	0.2	50	361
	0.3	13	
	0.4	0	
4	0.2	28	314
	0.3	0	
	0.4	0	

3.1.2. Moisture Characteristics

We can also clearly see the main draining and wetting relationships between the water content and potential as well as the scanning curves. The scanning curves result from intermediate changes between wetting and draining (Hillel, 1980, p153) as seen in figure 30. These data were sorted into primary wetting and draining curves and outliers removed. The van Genuchten moisture characteristics (van Genuchten, 1980) were then determined by fitting to these sorted data using RETC (Yates *et al.* 1992) (figure 31) and the results are presented in Table 5.

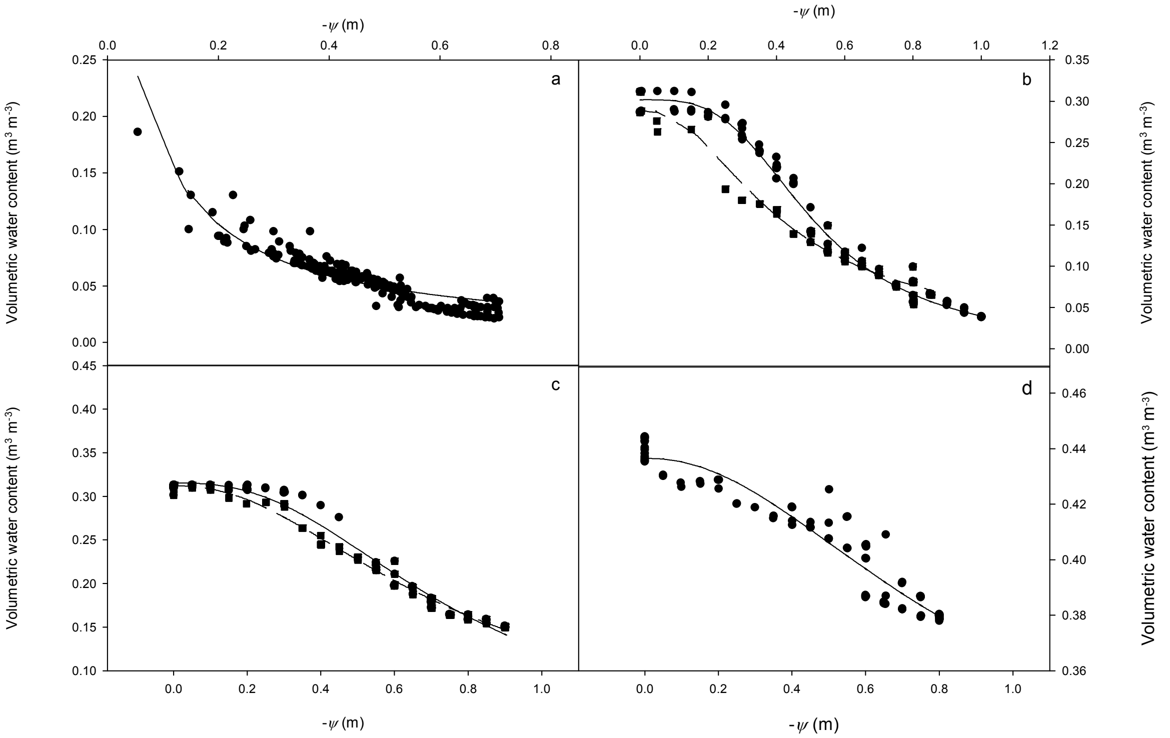


Figure 31. Moisture characteristic curves for Point Sturt site derived from water content and water table data; a) 0.1 m depth, b) 0.2 m depth, c) 0.3 m depth and d) 0.4 m depth. The solid line is the fit to the draining data (●) and the dashed line the fit to the wetting data (■).

The fitted parameters for the 0.1 m and 0.4 m depths will have a lot of uncertainty about them due to a limited range of data, water contents < 0.2 m^3 m^{-3} for the 0.1 m depth and >0.38 m^3 m^{-3} for the 0.4 m depth. For these two depths some of the parameters were fixed to prevent unrealistic results from being generated. The lack of calibration of the water content sensors will affect the van Genuchten parameters, in particular θ_s and θ_r which will be offset by any bias in the water content measurements.

Table 5. van Genuchten (1980) parameters fitted to the data in figure 31, and along with the coefficient of regression (R^2).

Depth	Limb	θ_s	θ_r	α (m^{-1})	*n*	R^2
0.1	Draining	0.310*	0*	16.13	1.88	0.84
0.2	Draining	0.302	0	2.22	3.50	0.987
0.2	Wetting	0.290	0	3.18	2.36	0.966
0.3	Draining	0.315	0	1.60	2.63	0.983
0.3	Wetting	0.312	0	1.79	2,24	0.992
0.4	Draining	0.436	0.29	1.5*	2.15	0.923

*values fixed and not fitted

Nevertheless these data can be used to estimate vertical potential gradients from the water content data using the van Genuchten equation:

$$\theta = \theta_r + \frac{\theta_s - \theta_r}{\left[1 + \left|\alpha\psi\right|^n\right]^m} \tag{5}$$

where θ, θ_s and θ_r are volumetric water content, saturated water content and residual water content respectively, α is a parameter related to the air entry value or bubbling pressure of the soil, n is the shape factor for the slope of the curve and $m = 1 - 1/n$.

3.1.3. Comparison of Vertical and Horizontal Potential Gradients at Point 1 for Point Sturt site.

The conceptual model presented in section 1 suggested that evaporation towards the surface was a more likely direction for water and hence solute transport rather than horizontally due to the low slopes of the near shore regions of the lakes. We can test this concept by determining the vertical potential gradients between the water content monitoring depths at point 1 in the piezometers transect. This was done using the parameters generated for the van Genuchten parameters (Table 5) and by rearranging eqn (5). The rearrangement of eqn (5) to obtain the potential results in:

$$\psi = \frac{1}{\alpha}\left[\left(\frac{\theta_s - \theta_r}{\theta - \theta_r}\right)^{-m} - 1\right]^{-n}. \qquad (6)$$

What we see is that the vertical potential gradient between 0.4 and 0.3 m, and 0.3 and 0.2 m is at least 2 orders of magnitude greater than the horizontal gradient (figure 32). Any bias in the water content measurements will be eliminated in estimating the potential due to water content differences being used in calculation of the potential.

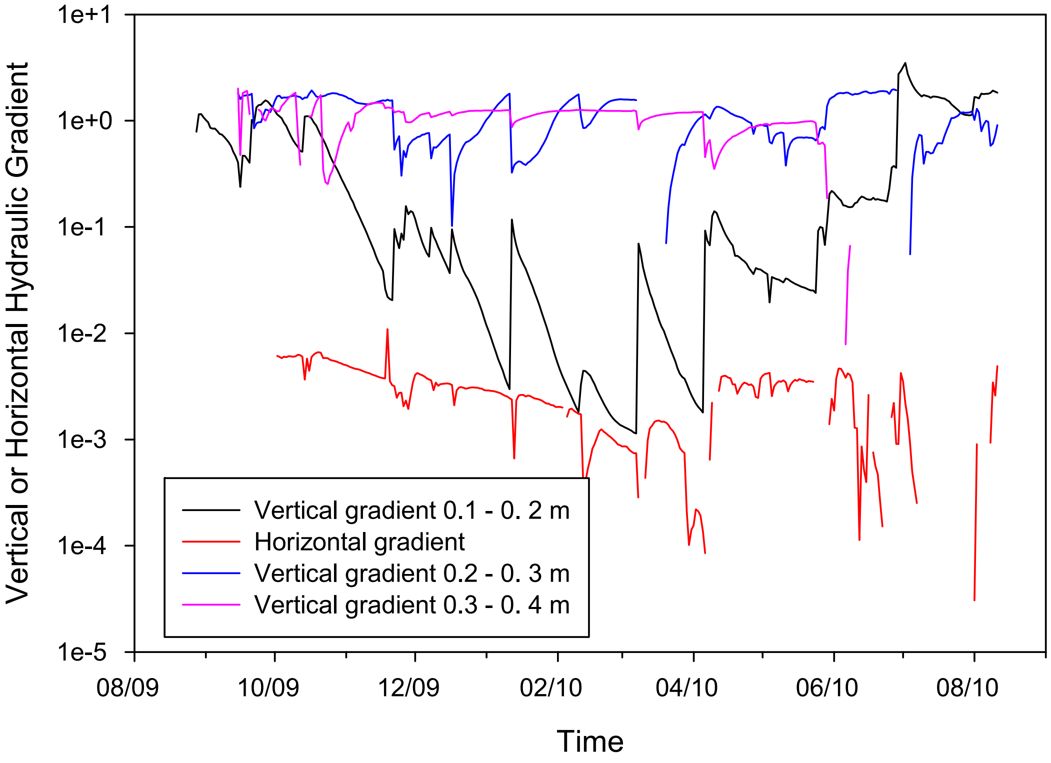


Figure 32. Comparison of vertical and horizontal potential gradients for point 1 at Point Sturt site.

3.1.4. Specific Yield for Point Sturt site

The specific yield of the soil at point 1 can be determined from measurements of water content, rainfall and water table response using:

$$S_e = \frac{\int_0^{T_s} P(t)dt - \int_0^{z_{wt}} \left[\theta_T(z) - \theta_0(z)\right] dz}{\left[h_T - h_0\right]} \tag{7}$$

where T_s is the length of the rainfall event, $P(t)$ is the rainfall rate, θ_T and θ_0 are the water contents at the start and end of the rainfall event, and h_T and h_0 are the water table heights at the start and end of the rainfall event. Again since we are using the water content difference any bias in the water content due to not being calibrated will not affect these results.

Events were selected where the rainfall event was short < 4 days and large enough to give a significant response in the water table height. For these selected events the specific yield was estimated using eqn (7). The results range from 0.014 to 0.080 which suggest that the water table response to 1 mm of water input to the water table could range from 71 mm to 19 mm (Table 6). Thus although rainfall amounts are relatively low in this region the water table response can be substantial.

Table 6. Estimated specific yield for Point Sturt site using data from Point 1.

Date	Event length (day)	Rainfall total (mm)	*h* range (m AHD)	S_e (m m^{-1})
8/3/2010	2.81	17.2	-0.811 to -0.663	0.043
7/4/2010	1.72	25.8	-0.851 to -0.411	0.014
9/4/2010	0.24	14.2	-0.411 to -0.290	0.053
25/5/2010	1.58	18.4	-0.529 to -0.288	0.021
28/5/2010	2.09	21.8	-0.315 to -0.099	0.040
8/6/2010	3.3	12.6	-0.286 to -0.190	0.080
mean				0.04
Standard error				0.01

3.2. Water Content Analysis for Campbell Park site

3.2.1. Capillary Length Scale

The potential at equilibrium with the water table at each of the four depths was calculated from the water table data using eqn (4). This was plotted against the water content data (figure 33) and where possible the capillary length was estimated from the potential when the water content deviates from the saturated values. Although at Campbell Park there was a water content sensor at 0.5 m, this data is not shown as the data is essentially the same as that at a depth of 0.4 m (figure 33d).

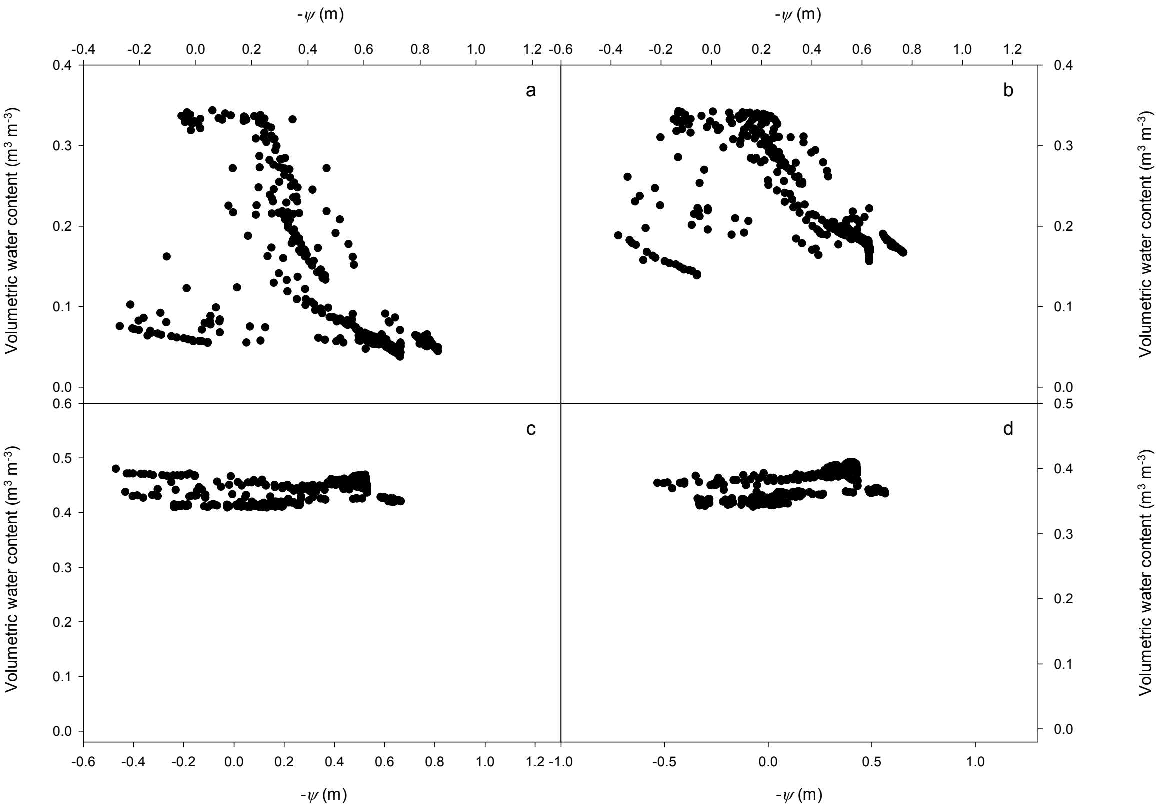


Figure 33. Potential estimated from water table height and eqn (2) with water content for point 1 at Campbell Park site for a) 0.1 m depth, b) 0.2 m depth, c) 0.3 m depth, and d) 0.4 m depth.

There is a lot more scatter in the water content data from Campbell Park site with some strange low water content values at low potentials for both the 0.1 and 0.2 m depths. This may be due to a problem with the sensors, as if it was due to a lack of equality between the piezometric head and the water table height the opposite response i.e. high water content and low estimated potential, would be expected. The capillary length scale at the 0.3, 0.4 and 0.5 m depths cannot be determined as the potential does not reach the point where desaturation occurs. However, these data indicate that the capillary length scale is substantial, > 0.6 m for the lower soil depths (Table 7).

Table 7. Estimation of capillary length scale for soil at point 1 at Campbell Park site.

Depth (m)	λ (m)
0.1	0.21
0.2	0.29
0.3	> 0.66
0.4	> 0.56
0.5	> 0.46

This will mean that for depths > 0.3 m the soil is never unsaturated during the time of monitoring, so negligible oxidation of pyrite is expected at these depths (Table 8). We can also calculate the time that unsaturated conditions may have occurred at these depths at the other points in the piezometers transect by extrapolation the values of λ in Table 8 to the other points along the transect. For the soil at depths of 0.1, 0.2 and 0.3 m the water table depth would need to be more 0.31, 0.49 and $\approx$ 1.0 m below the soil surface for the soil to be unsaturated at these depths. At all sites the soil was estimated to never be unsaturated at 0.3 m depth, while at 0.1 m depth the soil was estimated to be unsaturated for 82% of the monitoring time at point 1 and around 65% for the other three points.

Table 8. Percentage of time the water table was below the critical depth at selected depths at each piezometer measurement point for the Point Sturt site.

Point	Depth (m)	% of time water table below critical depth	Total number of monitoring days
1	0.1	82	347
	0.2	61	
	> 0.3	0	
2	0.1	61	301
	0.2	50	
	> 0.3	0	
3	0.1	68	361
	0.2	53	
	> 0.3	0	
4	0.1	63	314
	0.2	56	
	> 0.3	0	

3.2.2. Moisture Characteristics

It is easier to see draining relationship between the water content and potential than the wetting and scanning curves in figure 34 for the Campbell Park site compared to the Point Sturt site (figure 30). These data were sorted into primary wetting and draining curves and outliers removed (figure 34). Only the draining curve could be distinguished for the 0.2 m depth. The van Genuchten moisture characteristics (van Genuchten, 1980) were then determined by fitting to the data using RETC (Yates *et al.* 1992) and the results are presented in Table 9.

There is a similarity in the van Genuchten parameters for the Campbell Park and Point Sturt sites except for the 0.1 m depth data at Point Sturt site where the value of α is much greater than any other values. Given that both sites have sandy soils this similarity in the parameters is not surprising.

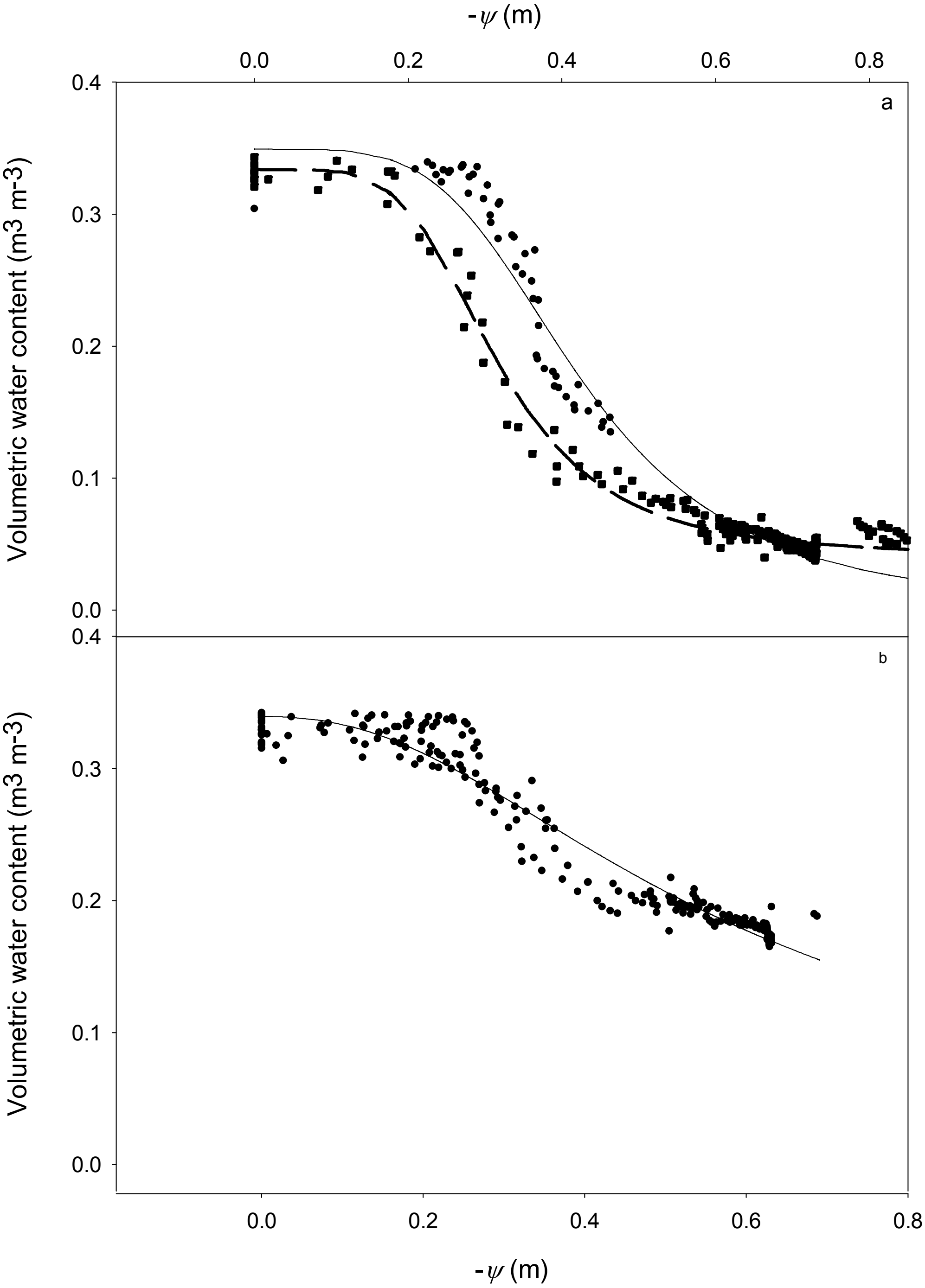


Figure 34. Moisture characteristic curves for Campbell Park site derived from water content and water table data; a) 0.1 m depth, b) 0.2 m depth. The solid line is the fit to the draining data (●) and the dashed line the fit to the wetting data (■).

Table 9. van Genuchten (1980) parameters fitted to the data in figure 34, and along with the coefficient of regression (R^2).

Depth	Limb	θ_s	θ_r	α (m^{-1})	n	R^2
0.1	Draining	0.349	0	2.56	4.41	0.971
0.1	Wetting	0.334	0.041	3.83	4.79	0.985
0.2	Draining	0.339	0	2.29	2.34	0.951

3.2.3. Comparison of Vertical and Horizontal Potential Gradients at Point 1 for Campbell Park site.

The vertical potential gradients were calculated using the data in Table 9 and the water table data at point 1 for depths of 0.1, 0.2 and 0.3 m (figure 35). The van Genuchten parameters for the 0.2 m depth were assumed to apply to the 0.2 to 0.3 m depth increment. The gradient between 0.2 and 0.3 m is approximately 3 orders of magnitude greater than the horizontal flux determined from the piezometers. This means that water is much more likely to move vertically to evaporate than travel towards the lake. Even when the soil is dry, the gradient between depths of 0.2 and 0.1 m is greater than the horizontal gradient.

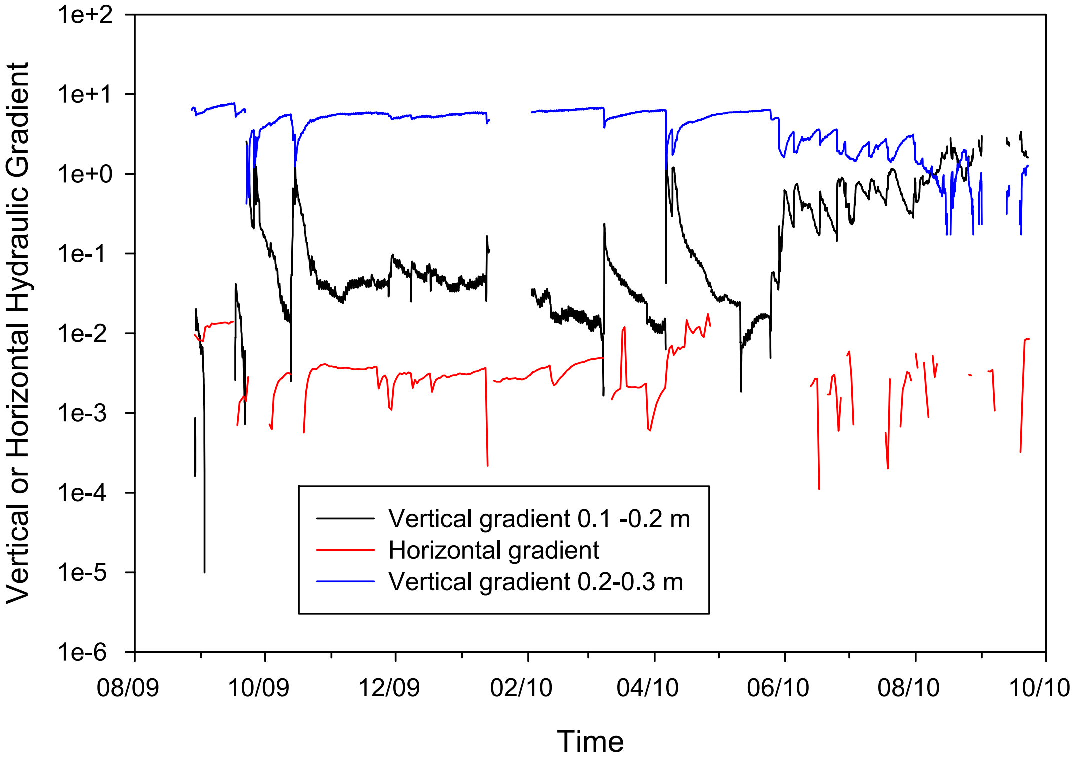


Figure 35. Comparison of vertical and horizontal potential gradients for point 1 at Campbell Park site.

3.2.4. Specific Yield for Campbell Park site

The specific yield was calculated using eqn (7). Again events were selected for analysis where the event was < 4 days and was large enough to give a measureable response. This resulted in a range of specific yield values from 0.193 to 0.017, a greater than 10 fold difference (Table 10).

The mean specific yield for the Campbell Park site is double that of the Point Sturt site but is not significantly different at the 5% level. The range of specific yields means that the water table response to 1 mm of water input to the water table could range from 5 mm to 59 mm. The lesser values of specific yield occur when the water table is at or near the soil surface, suggesting that when the water table nears the surface only small amounts of rainfall would raise the water table substantially.

Table 10. Estimated specific yield for Campbell Park site using data from Point 1.

Date	Event length (day)	Rainfall total (mm)	*h* range (m AHD)	S_e (m m^{-1})
22/11/2009	1.46	8.6	-0.355 to 0.191	0.017
8/12/2009	0.2	8.8	-0.473 to -0.404	0.095
12/1/2010	4.98	24.4	-0.542 to -0.423	0.163
8/3/2010	1.87	25.2	-0.546 to -0.453	0.193
6/4/2010	0.74	46.0	-0.547 to 0.054	0.039
8/6/2010	1.35	20.6	-0.679 to -0.502	0.062
Mean				0.09
Standard error				0.03

3.3. Water Content Analysis for Windmill site

3.3.1. Capillary Length Scale

The potential at equilibrium with the water table at each of the four depths was calculated from the water table data using eqn (4). This was plotted against the water table data (figure 36) and where possible the capillary length was estimated from the potential when the water content deviates from the saturated values. The data at the Windmill site shows that the capillary length at 0.1 m depth is small $\approx$ 0.1 m (figure 36a).) Although it is difficult to see in figure 36b the capillary length at 0.2 m can be determined as 0.22 m (Table 11).

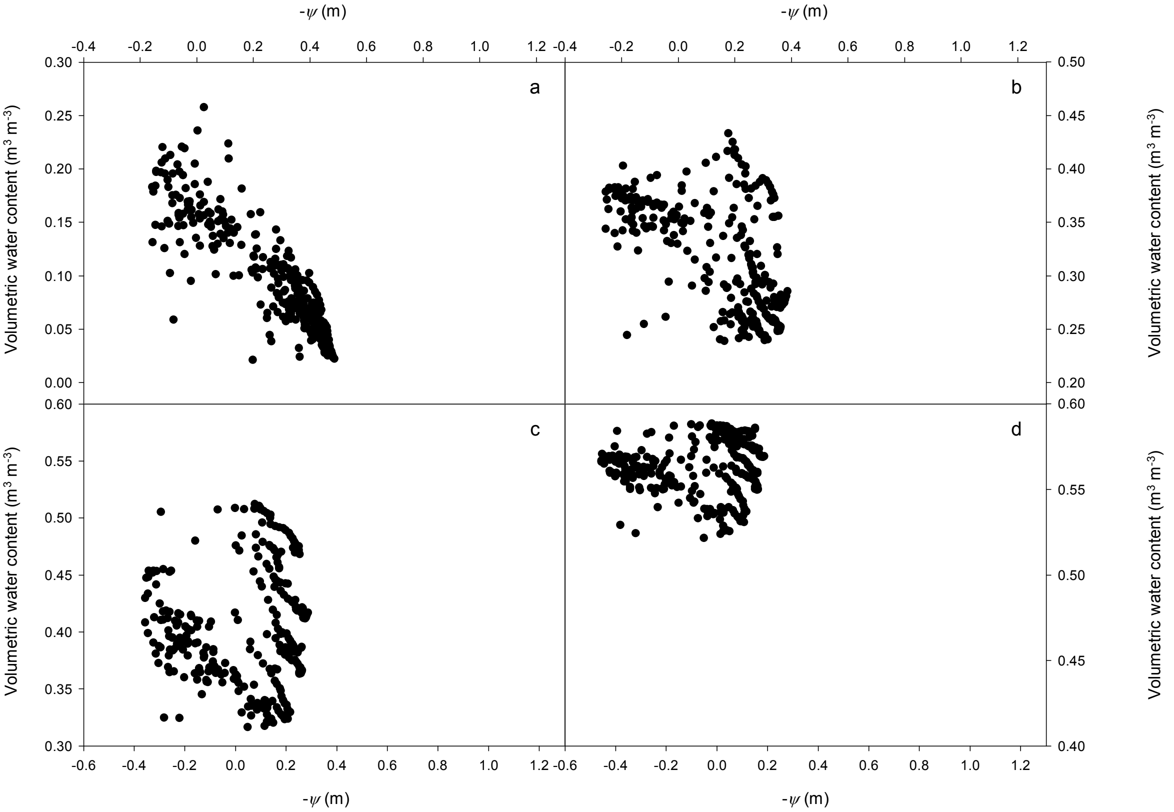


Figure 36. Potential estimated from water table height and eqn (2) with water content for point 1 at Windmill site for a) 0.1 m depth, b) 0.2 m depth, c) 0.3 m depth, and d) 0.4 m depth.

There is a lot of scatter in the data from depths 0.2, 0.3 and 0.4 m which is due to the wetting and draining sequences with the scanning curves (Hillel, 1980, p153) clearly visible. The values of the capillary length are < those found at the Point Sturt and Campbell Park sites especially in the lower depths. This means the sediments at this site are much more likely to desaturate when the water table drops exposing the pyrite to oxidation.

Table 11. Estimation of capillary length scale for soil at point 1 at Windmill site.

Depth (m)	λ (m)
0.1	≈ 0.1
0.2	0.22
0.3	≈ 0.13
0.4	≈ 0.15

The time that the various depths in the soil will have been exposed to unsaturated conditions during the monitoring is greatest at 0.1 m depth at point 2 (Table 12). The maximum depth to which unsaturated conditions may have occurred using the data in Table 11 and the piezometers data was found to be between 0.53 m (point 4) and 0.45 m (point 1). At a depth of 0.2 m the conditions would have been unsaturated approximately 50% of the time (Table 12).

Table 12 Percentage of time the water table was below the critical depth at selected depths at each piezometer measurement point for the Windmill site.

Point	Depth (m)	% of time water table below critical depth	Total number of monitoring days
1	0.1	67	388
	0.2	40	
	0.3	39	
	0.4	11	
2	0.1	72	388
	0.2	47	
	0.3	46	
	0.4	22	
3	0.1	65	388
	0.2	44	
	0.3	42	
	0.4	10	
4	0.1	61	371
	0.2	47	
	0.3	46	
	0.4	27	

3.3.2. Moisture Characteristics

The moisture release relationships shown for the Windmill site should be considered as only approximate as the data required a lot of subjective selection to obtain what was considered to be reasonable data sets (figure 37). Due to the restricted water content data range at depths of 0.3 and 0.4 m, eqn (5) was not fitted to data from these depths. The wetting data did not return to the same saturated water content as the draining data for the 0.1 m depth. This may be due to occluded pores or compaction upon draining.

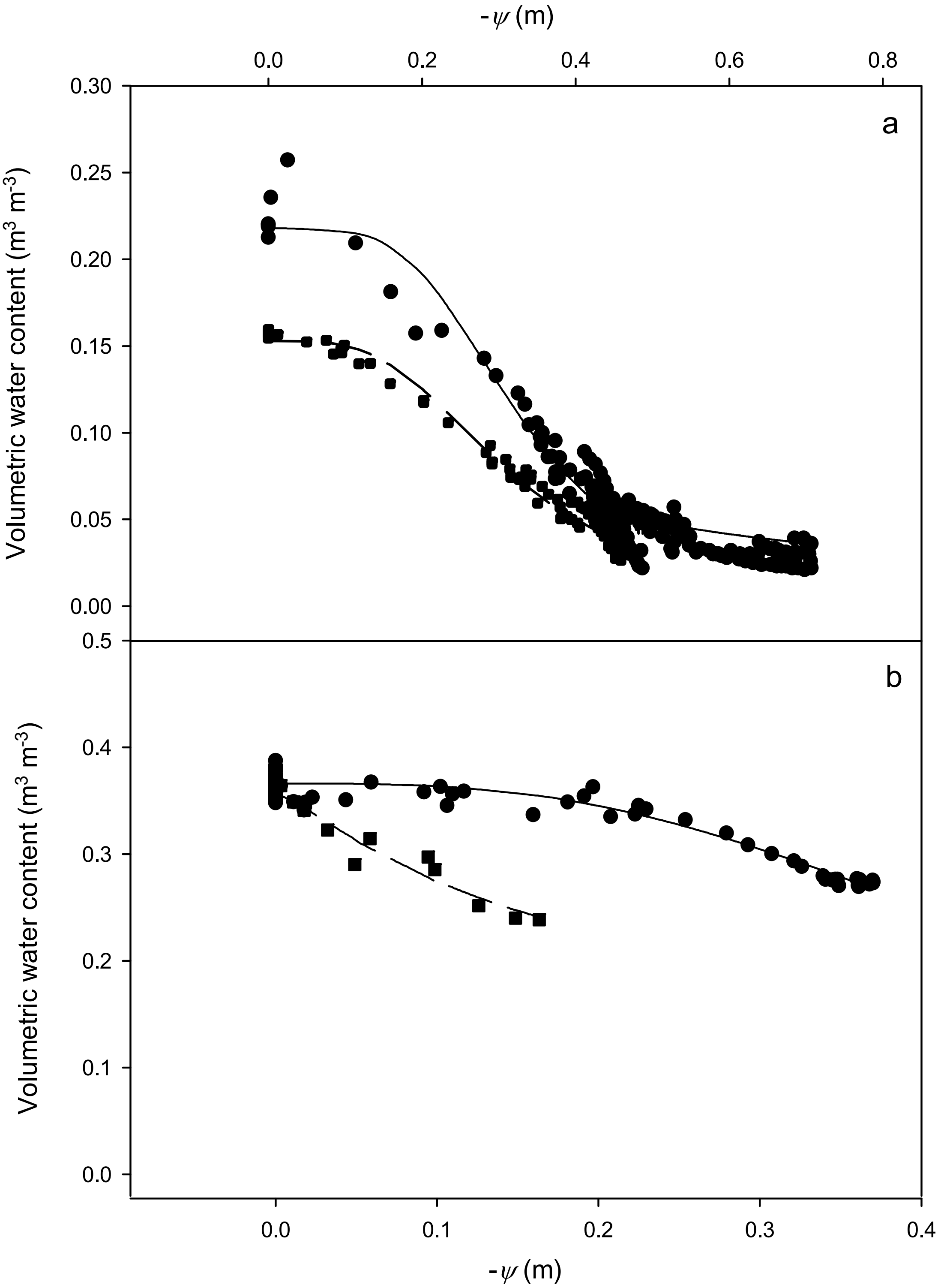


Figure 37. Moisture characteristic curves for Windmill site derived from water content and water table data; a) 0.1 m depth, b) 0.2 m depth. The solid line is the fit to the draining data (●) and the dashed line the fit to the wetting data (■).

The van Genuchten parameters for the fitted moisture characteristics indicate similar saturated water contents at 0.2 m depth to those at Campbell Park (Table 13). The large value of α at 0.2 m for the Windmill site (Table 13) is similar to the value for 0.1 m at the Point Sturt site (Table 5). These large values of α are usually associated with soils with small capillary lengths.

Table 13. van Genuchten (1980) parameters fitted to the data in figure 37, and along with the coefficient of regression (R2).

Depth	Limb	θ_s	θ_r	α (m^{-1})	N	R^2
0.1	Draining	0.218	0	3.35	4.19	0.950
0.1	Wetting	0.153	0	3.63	3.54	0.983
0.2	Draining	0.366	0	2.28	3.03	0.930
0.2	Wetting	0.356	0	13.6	1.39	0.957

3.3.3. Comparison of Vertical and Horizontal Potential Gradients at Point 1 for Windmill site.

The vertical potential gradients were calculated using the data in Table 11 and the water table data at point 1 for depths of 0.1, 0.2 and 0.3 m (figure 38). The van Genuchten parameters for the 0.2 m depth were assumed to apply to the 0.2 to 0.3 m depth increment. The gradients between 0.1 and 0.2 m, and between 0.2 and 0.3 m are approximately 3 orders of magnitude greater than the horizontal flux determined from the piezometers. This means that water is much more likely to move vertically to evaporate than travel towards the lake at point 1.

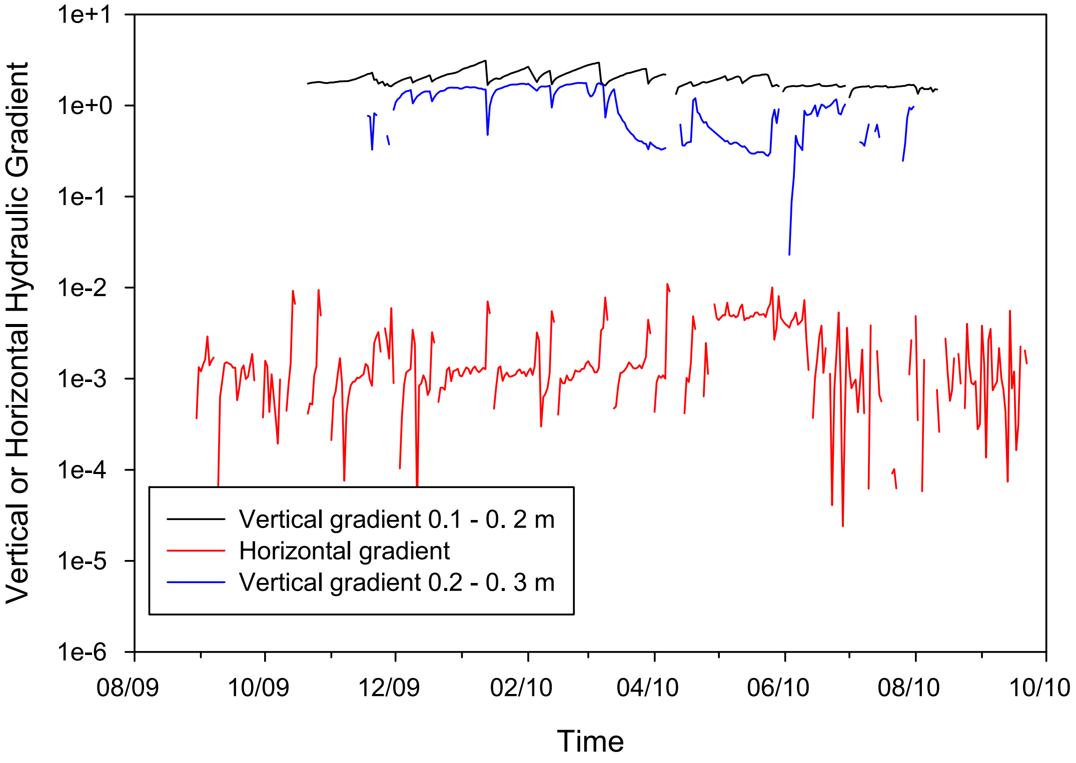


Figure 38. Comparison of vertical and horizontal potential gradients for point 1 at Windmill site.

3.3.4. Specific Yield for Windmill site

The specific yield was calculated using eqn (7). Again events were selected for analysis where the event was < 1 day and was large enough to give a measureable response. This resulted in a range of specific yield values from 0.013 to 0.075, a greater than 5 fold difference (Table 14).

Table 14. Estimated specific yield for Windmill site using data from Point 1.

Date	Event length (day)	Rainfall total (mm)	*h* range (m AHD)	S_e (m m^{-1})
17/9/2009	0.72	16	-0.407 to 153	0.013
12/1/2010	0.32	24	-0.592 to -0.072	0.030
4/2/2010	0.41	2.4	-0.554 to -0.507	0.075
12/2/2010	0.88	9.2	-0.569 to -0.433	0.069
7/3/2010	0.74	46.0	-0.516 to -0.133	0.075
Mean				0.05
Standard error				0.01

The mean specific yield for the Windmill site is between the values estimated for the Point Sturt and Campbell Park sites and not significantly different from these values. The range in specific yield values suggests that a 1 mm input of water to the water table could result in a rise in the water table of between 77 and 13 mm. Using the mean value of 20 mm suggests that a water table rise of 1 m would require on average 50 mm of water input to the water table with the range from 13 to 75 mm.

3.4. Water Content Analysis for Currency Creek site

There is no data for Currency Creek.

4. ACID FLUX ESTIMATION

The above analysis of the groundwater and water content data has given some understanding of what the likely fluxes are and possible mechanisms for water and associated acidity flux to the Lower Lakes. Estimates were made of the flux from exfiltration and runoff and also by groundwater flux to the lakes at each of the 3 sites where reliable slope data was available.

The hydraulic conductivity (k_S) used for the Point Sturt, Campbell Park and Windmill sites was obtained from the studies by Earth Systems (2010) (Table 15). Earth Systems k_S rates are much larger than the infiltration rates found by Hicks et al (2009) of 17 and < 1mm day^{-1} for the fresh water in mesocosm studies for sandy and a clayey sediments at Boggy Creek, respectively.

Table 15. Saturated Hydraulic conductivity (K_s) for soils at measurement sites (data from Earth Systems 2010). The name used in the Earth Systems report is shown in brackets.

Site	Point	k_s (m day^{-1})	Depth of measurement below surface(m)	Texture
Point Sturt	Point 1 (PS-1S)	0.09	1.38	clayey sand
	Point 2 (PS-2S)	0.62	1.10	sand/clay
	Point 3 (PS-3S)	5.56	1.20	calcareous clay
	Point 4 (PS-4S)	4.92	1.12	Sand
Campbell Park	Point 1 (CP-1S)	0.22	0.92	clayey sand/clay
	Point 2 (CP-2S)	0.81	1.22	sand/clay
	Point 3 (CP-3S)	0.59	1.22	sand/clay
	Point 4 (CP-4S)	2.52	1.22	sand/clay
Windmill	Point 1 (WM-1S)	>30	3.30	Sand
	Point 2 (WM-2S)	>30	2.80	sand/clay
	Point 3 (WM-3S)	4.91	2.22	sand/clay
	Point 4 (WM-4S)	1.83	1.82	sand/clay
Currency Creek	Point 1 (UCC-P1)	0.02	3.00	sand/clayey sand/clay
	Point 2 (LCC-P2)	0.06	2.60	sand/clayey sand/clay

The hydraulic conductivity was not measured at the Currency Creek site where the intensive water table monitoring occurred (UCC-P3), so the average for the other two sites (0.4 m day^{-1}) was used in the acid flux estimates calculated below. Also the sites at Currency Creek do not constitute a transect so the slope cannot be estimated from the height data. A slope of 1/350 = 0.286 the same as used by Earth Systems (2010).

Runoff (*RO*) was considered to have occurred if rainfall occurred on a day when the water table height was above the soil surface. This will tend to overestimate the runoff as evaporation on this day was not subtracted. However, there will also be days when the rainfall intensity is greater than the infiltration capacity of the soil and runoff will be generated due to this process. Here we are not accounting for this as a full water balance calculation is required to do this and this will be included in the subsequent report (Cook et al., 2011). However, given the uncertainties this approach will at least allow a 'ballpark' estimation of runoff.

The exfiltration flux was estimated to occur when the water table rose above the surface. The water flux is calculated from the slope (S_1) at that point and the saturated hydraulic conductivity. This may under or over estimate the exfiltration if the hydraulic gradient is greater than or less than the slope, but again is unlikely to greatly bias the results. The exfiltration was calculated by:

$$J_e = k_s . S_1 \tag{8}$$

The acid flux from was then estimated by:

$$J_A(t) = \left[\left(RO(t) + J_e(t) \right) \right] . C_1 + J_{GW} C_2 \tag{9}$$

where C_1 is concentration of either acidity or net acidity (acidity – alkalinity) of the surface water, C_2 is the concentration of the groundwater and J_{GW} is the flux of groundwater. The groundwater flux is considered to be insignificant during the time period of monitoring, so only the aci flux from runoff and exfiltyration will be considered below. The net acidity was determined by titration to a pH of 8.3 and the alkalinity by titration to a pH of 4.5. Kirby and Cravotta (2005) discuss the definition of net acidity and or alkalinity and suggested that the even though these methods have their flaws, these result in self compensating errors which results in an insignificant error in estimation of net acidity as defined above.

The values of C were taken as the maximum, mean and minimum values of acidity or net acidity measured in the laboratory, or if data was missing calculated from the chemical composition, for water samples taken from the piezometers during the monitoring period. These values were measured by SA EPA (2011) and are presented in Table 16. From these data it appears that Campbell Park and Currency Creek 2 are the most likely sites where acidity could discharge to the Lower Lakes.

Table 16. Maximum, mean, minimum acidities and net acidities measured in piezometers during monitoring period. Negative values indicate that alkalinity is > acidity.

Site and Point	Acidity (mgCaCO$_3$ L^{-1})			Net acidity (mgCaCO$_3$ L^{-1})		
	Maximum	Mean	Minimum	Maximum	Mean	Minimum
Point Sturt						
Point 1	176	101	33	90	-100	-288
Point 2	1450	659	309	1450	638	307
Point 3	109	29	3	-467	-325	-198
Point 4	187	92	33	-342	-266	-156
Campbell Park						
Point 1	88	63	34	-25	-50	-152
Point 2	1580	967	513	1580	967	513
Point 3	502	392	142	502	388	102
Point 4	1380	851	195	1380	851	195
Windmill						
Point 1	72	66	55	-272	-398	-508
Point 2	109	89	60	-328	-432	-527
Point 3	375	138	58	-42	-555	-955
Point 4	203	170	130	-501	-711	-1021
Currency Creek						
Point 1	64	42	10	-21	-125	-278
Point 2	728	283	55	728	277	38

4.1. Acid Flux Estimation for Point Sturt site

The flux of acidity through the sediments at the Point Sturt site is going to occur mainly from points 1 and 2 towards point 3. It is instructive to consider how far this acidity is likely to travel away from points 1 and 2. The average hydraulic gradient when the gradient was towards the lake (negative), along with the amount of time the gradient was negative was calculated between the sequential points on the transects. Using the hydraulic conductivity, the average negative gradient and the time when the gradient was negative, the distance that a particle of water will travel towards the lake was calculated. For points 1 and 2 the time (T_t) that the hydraulic gradient (*dh/dx*) is towards the Lake is 272 and 266 days respectively during the monitoring period. Assuming a porosity (θ_S) of 0.3 m^3 m^{-3} (Table 5) and taking the average hydraulic gradient of approximately -0.002 for flow from point 1 to point 2, and point 2 to point 3, the travel distance for a water particle can be calculated using:

$$L_T = -\frac{dh}{dx}\frac{\overline{k}_s}{\theta_s}T_t \tag{10}$$

where $\overline{k}_s$ is the average hydraulic conductivity of the two points. Using eqn (10) the resulting values of L_T are 0.65 and 6.0 m respectively for travel from point 1 to point 2 and point 2 to point 3. There are short periods when the hydraulic gradient between point 1 and 2 was < -0.01 and so the travel distance would have been increased on these days. The number of days was 14 and this would have allowed the solute to travel a distance of 0.013 m using the k_S of 0.09 m day^{-1}. Even using the k_S of 5.56 m day^{-1}, measured at point 4, only increases this travel distance to 0.78 m on these 14 days. What this suggests that the lateral groundwater discharge is unlike to have contributed acidity to Lake Alexandrina.

The discharge from exfiltration and runoff is summarised in Table 17 and shows that if the alkalinity did not react in time to prevent acidity discharging to the Lake then a maximum of 1450 $kgCaCO_3$ ha^{-1} of acidity could be discharged to Lake Alexandrina. The lake is approximately 14.3 km in radius (*R*). If we consider that the acid flux is generated from a band 100 m (ΔR) in width around the lake, then the area of this acid flux zone can be calculated by:

$$A_a = \pi\left[\left(R+\Delta R\right)^2 - R^2\right] = \pi\left[2R\Delta R + (\Delta R)^2\right] \tag{11}$$

This area (A_a) would be 901.6 ha. If the maximum value for the acidity (1450 $kgCaCO_3$ ha^{-1} Table 17) is multiplied by this area then the maximum acid flux that occurred would result in 1300 tonnes of acidity being discharged to the Lake Alexandrina. However, if the mean values of the net acidity are summed across all four points then the net acidity flux per hectare would be 38 $kgCaCO_3$ ha^{-1} and would represent an acid flux to Lake Alexandrina of 35 tonnes of acidity.

However, Lake Alexandrina got to low in water level of approximately -07 m AHD which would have exposed an area of approximately 400 m in annulus around the lake this would represent an area of 3562 ha. This then could represent a maximum acid flux of 5149 tonnes of acidity being discharged to the Lake Alexandrina. This is in comparison with the estimated 180,000 tonnes of alkalinity in Lake Alexandrina and 17,500 tonnes of alkalinity annually added from the Murray river (Earth Systems, 2008).

These values are one to two orders of magnitude less than the suggested by Earth Systems (2010). However, we are using the data measured in one transect to make this estimate of the acid discharge. This extrapolation is hence only speculative but what it does show is that acidity fluxes would have to be many orders of magnitude greater than this to significantly impact the water quality of all of Lake Alexandrina. Localised pockets where much higher acidity fluxes may occur and will impact locally on the water quality of the Lake, especially in areas such as an embayment where water exchange with the lake is low. Extrapolation of these results suggest that the annulus of area exposed would have to expand to approximately 1500 m for the maxumum acidity generated to equal the alkalinity from the Murray river and to more than 10,000 m if the net acidity value is used. However, such linear extrapolation of these data is unlikely to be correct.

Table 17. Estimated acid flux from exfiltration and runoff for Point Sturt site. The negative values indicated net alkalinity.

Point	Runoff (mm)	Exfiltration (mm)	Acidity ($kgCaCO_3\ ha^{-1}$)			Net Acidity ($kgCaCO_3\ ha^{-1}$)		
			Maximum	Mean	Minimum	Maximum	Mean	Minimum
1	58.2	7.4	115	66	22	59	-66	-189
2	32	66	1446	657	308	1446	636	306
3	15.6	93	118	32	3	-215	-353	-508
4	14	10	45	22	8	-37	-66	-82

4.2. Acid Flux Estimation for Campbell Park site

The maximum distance travelled by a particle of water in the direction of the lake from each point at the Campbell Park site was calculated using eqn (7) and the porosity used was 0.35 $m^3\ m^{-3}$ (Table 9). These calculations show that the maximum distance that a water particle is likely to travel in the time of the monitoring is < 4 m and so the rising of the lake level and discharge from runoff and exfiltration is much more likely to result in acid fluxes of any significance (Table 18) rather than flux from ground water.

The gradients of < -0.01 occurred for 21, 1, 4, 0 for points 1 to 2, 2 to 3, 3 to 4 and 4 to lake respectively. Travel distance if the maximum gradients are used along with the days where the gradient is < -0.01 would give travel distances of 0.19, 0.01, 0.09 and 0 for points 1 to 2, 2 to 3, 3 to 4 and 4 to lake respectively. Considering these largere gradients does not change the expectation of insignificant acid flux due to groundwater flux at this site.

Table 18. Estimate of travel distance for Campbell Park site.

Point	T (day)	Average dh/dx	$\bar{k}_s$ (m day^{-1})	L_T (m)
1 to 2	272	-0.003	0.52	1.20
2 to 3	294	-0.002	0.70	1.17
3 to 4	267	-0.003	1.56	3.56
4 to Lake	125	-0.0002	2.52	0.19

The results for Campbell Park indicate that a much greater amount of acidity is likely to have been added from this site into Lake Albert than was indicated from the Point Sturt site into Lake Alexandrina. This suggests that similar sites like this could be a potential acid generation threat to Lake Albert. The radius of Lake Albert is approximately 7.4 km and taking a radial disc of 100 m with the maximum acid discharge of 5371 kg$CaCO_3$ ha^{-1} (Table 19) and would result in a total acid discharge to Lake Albert during the monitoring period of 2500 tonnes of acidity. Averaging the mean net acidity values would give a net acidity of 1456 kg$CaCO_3$ ha^{-1} and represent an acid discharge to Lake Albert of 680 tonnes of acidity. These amounts of acidity are still much less than the alkalinity in Lake Albert but localised acidity is likely. However, the annulus of exposed sediments for Lake Albert was much greater than 100 m and at its lowest point > 5000 m. Using a radius of 5500 m the maximum acid discharge rises to 86,400 tonnes of acidity and the net acid discharge rises to 17,600 tonnes of acidity. The volume of Lake Albert in the Earth System (2008) report was estimated as 76.4 GL and using an alkalinity of 200 mg L^{-1} gives a total alkalinity of 15,300 tonnes kg$CaCO_3$. This suggests that if the areas surrounding Lake Albert were all similar to Campbell Park then the Lake was susceptible to becoming acidity during the monitoring period. This extrapolation of the acid flux is only speculative and relies on the accuracy of the data used to generate it and extrapolation from this one site to the rest of the lake.

Table 19. Estimated acid flux from exfiltration and runoff for Campbell Park site. The negative values indicated net alkalinity. No exfiltration or runoff events were estimated to have occurred at point 3.

Point	Runoff (mm)	Exfiltration (mm)	Acidity (kgCaCO3 ha^{-1})			Net Acidity (kgCaCO3 ha^{-1})		
			Maximum	Mean	Minimum	Maximum	Mean	Minimum
1	15.2	6.5	19	14	7	-5	-10	-33
2	252	88	5371	3287	1743	5371	3287	1743
3	0	0	0		0	0	0	0
4	127	1	1770	1092	250	1770	1092	250

4.3. Acid Flux Estimation for Windmill site

The Windmill site has the highest hydraulic conductivities and hence the longest travel distances for points 1 and 2 (Table 20). The porosity used was 0.4 which is a mean value for all depths (figure 36). However, the largest value of L_T is still < 15 m so in the time of monitoring any acid would have only moved 1/3 of the distance between point 1 and 2. For this site gradients < -0.01 occured for points 1 to 2, 2 to 3 and 4 to lake on 2, 1 and 1 day respectively. This would have resulted in the travel distances of 1.14, 0.05 and 0.02 m respectively. The increase in travel distance due to the larger negative gradient is only significant for travel between points 1 and 2 and would increase the total distance from 14.5 to 15.6 m. Even with this increase in travel distance it is unlikely that groundwater flux will have contributed acidity to Lake Albert during the monitoring period.

The exfiltration and runoff are much more likely to have generated acidity and/or alkalinity to Lake Albert. The net acidity data suggests that this site could have contributed alkalinity to the lake (Table 21). The high hydraulic conductivity values measured for this soil are the reason for the high exfiltration estimates at points 1 and 2. Again using a band of 100 m around the shoreline as the contributing area and a radius of 7.4 km for Lake Albert the maximum amount of acidity that is estimated to have entered the Lake is 500 tonnes of acidity. If the average of the mean net acidity value is used then the amount of alkalinity that is estimated to have entered the Lake is 800 tonnes. However, when an exposed radius of 5500 m is used the maximum acidity and net acidity values increase to 12,100 tonnes and 22,800 tonnes respectively. If Lake Albert was only surrounded by sediments like those at the Windmill and the maximum acidity alone was transported to the lake it the amount would be close to the alkalinity of the lake and could have a significant effect on water quality. However, if the net acidity was transported then Lake Alberts alkalinity could double.

Table 20. Estimate of travel distance for Windmill site.

Point	*T* (day)	Average *dh/dx*	$\overline{k}_s$ (m day^{-1})	L_T (m)
1 to 2	333	-0.001	30	14.5
2 to 3	273	-0.001	17.5	11.9
3 to 4	277	-0.001	3.37	2.4
4 to Lake	220	-0.0004	1.83	0.38

This suggests that not only is acidity able to reach the Lake via runoff and exfiltration, alkalinity also can. The Campbell Park site and Windmill site are both sited on Lake Albert then these results represent some of the range of acid fluxes likely to enter the lake. Given its shallow nature, lower volume and hence lower alkalinity buffer Lake Albert is more vulnerable to acidification than Lake Alexandrina as found in previous studies (Hipsey and Salmon 2008, Hipsey et al. 2010). It would be thus prudent to continuously monitor the water quality in Lake Albert to determine if any significant changes in water quality occur.

Table 21. Estimated acid flux from exfiltration and runoff for Windmill site. The negative values indicated net alkalinity. No exfiltration or runoff events were estimated to occur at point 3.

Point	Runoff (mm)	Exfiltration (mm)	Acidity (kgCaCO$_3$ ha^{-1})			Net Acidity (kgCaCO$_3$ ha^{-1})		
			Maximum	Mean	Minimum	Maximum	Mean	Minimum
1	23.2	251	1080	197	181	-745	-1090	-1391
2	99	612	775	633	427	-2333	-3073	-3748
3	181	123	758	279	117	-85	-1122	-1931
4	164	59	454	380	290	-1119	-1589	-2281

4.4. Acid Flux Estimation for Currency Creek site

The Currency Creek site by comparison with the Windmill site has the lowest measured hydraulic conductivities. These values were only measured at the point 1 (UCC-P1) and estimated for point 2 (see above). The gradient was taken as 1/350 = 0.00286 as it was not measured at this site. This shows that a water flow via the groundwater to the Creek is very unlikely and acid discharge to the creek via this mechanism is negligible.

Table 22. Estimate of travel distance for Currency Creek site.

Point	T (day)	Average dh/dx	$\bar{k}_s$ (m day^{-1})	L_T (m)
1 to creek	197	-0.00286	0.02	0.02
2 to creek	155	-0.00286	0.04	0.04

Exfiltration and runoff are far more likely mechanisms for acid discharge to Currency Creek. Unfortunately at this site due to the Creek water level and the water tables moving in tandem there are zero days when runoff can be calculated at this site. For this site runoff was also considered to occur when rainfall was > 17 mm day^{-1} the value found by Hicks et al. (2010) for long term infiltration. This approach results in 8 days when runoff is predicted and at point 1 the water level was already above the surface, so this was not considered to provide runoff with acid solutes. It is suggested in Earth Systems (2010) that seiching, also caused some exfiltration of acidic water to occur at site CC-DS4 (figure 5).

Table 23. Estimated acid flux from exfiltration and runoff for Currency Creek site. The negative values indicated net alkalinity.

Point	Runoff (mm)	Exfiltration (mm)	Acidity (kgCaCO3 ha^{-1})			Net Acidity (kgCaCO3 ha^{-1})		
			Maximum	Mean	Minimum	Maximum	Mean	Minimum
1	33.8	4.9	25	16	4	-8	-48	-107
2	99	612	775	633	427	364	187	62

The values estimated for the acidity flux from exfiltration and runoff suggest that point 1 will produce either a small amount of acidity or alkalinity and point 2 acidity (Table 23). Given the much smaller volume of water in Currency Creek and possible acid generated areas like point 2 acid flux from runoff and exfiltration at such sites water quality in Currency Creek may be impacted as shown in the period 2008-2009 when large areas acidified (Figure 39).

5. SUMMARY AND CONCLUSIONS

Following unprecedented water level decline in Lakes Alexandrina and Lake Albert (figure 1) concern was expressed about acid generation from oxidation of pyritic minerals in the Lake sediments and possible flux of acid to the lakes (Fitzpatrick et al., 2008). Monitoring studies of the water levels in piezometers in the near shore region, lake levels and water content of the soils/sediments were initiated and monitored until due to rising lake levels this monitoring ceased. Here we have analysed this data in order to determine the likely fluxes of acidity to the lakes and extract the valuable information that can be inferred from this data.

A conceptual model for acid generation and transport of this acidity to Lakes Albert and Alexandrina is presented. The conceptual model considers four mechanisms for acid flux from the sediments to the lakes *viz*;

- wash off of acidic products from the sediment surface groundwater flux during rainfall or seiching events,
- flow of acidic groundwater to the lake

- exfiltration of acid pore water during rainfall events or upon rewetting of near shore sediments.
- diffusion of and/or mixing of acidity in the sediments with the lake waters during seiching events or upon flooding following lake level rise

All apart from the diffusive flux, these mechanisms were investigated using the piezometer head and water content data collected by Earth Systems (2010), and data water quality data from SA EPA. The piezometric head and water quality data was collected at four sites but water content data from only three sites. At the three sites where the water content data was collected the piezometer points were in a transect orthogonal to the shoreline (Table 2) and hydraulic gradients could be determined. The water content measurements were made adjacent to the highest piezometer point in the transects.

This piezometer transect data allowed estimates of the horizontal gradient in piezometric head to be estimated (section 2.2). The gradient in slope is low in the landscape surrounding the lakes, so it was thought that effects of water density on horizontal hydraulic gradients could not be ignored. The results presented here show that even in this low relief landscape hydraulic gradients due to water density are insignificant and only the piezometeric head gradients need to be considered. These results suggest that the flux of acidity to the lakes via groundwater transport will be insignificant even in these sandy soils. These results are not unexpected given the low slope of the land in the near shore region. Further modelling to confirm this will be part of a second report (Cook et al., 2011).

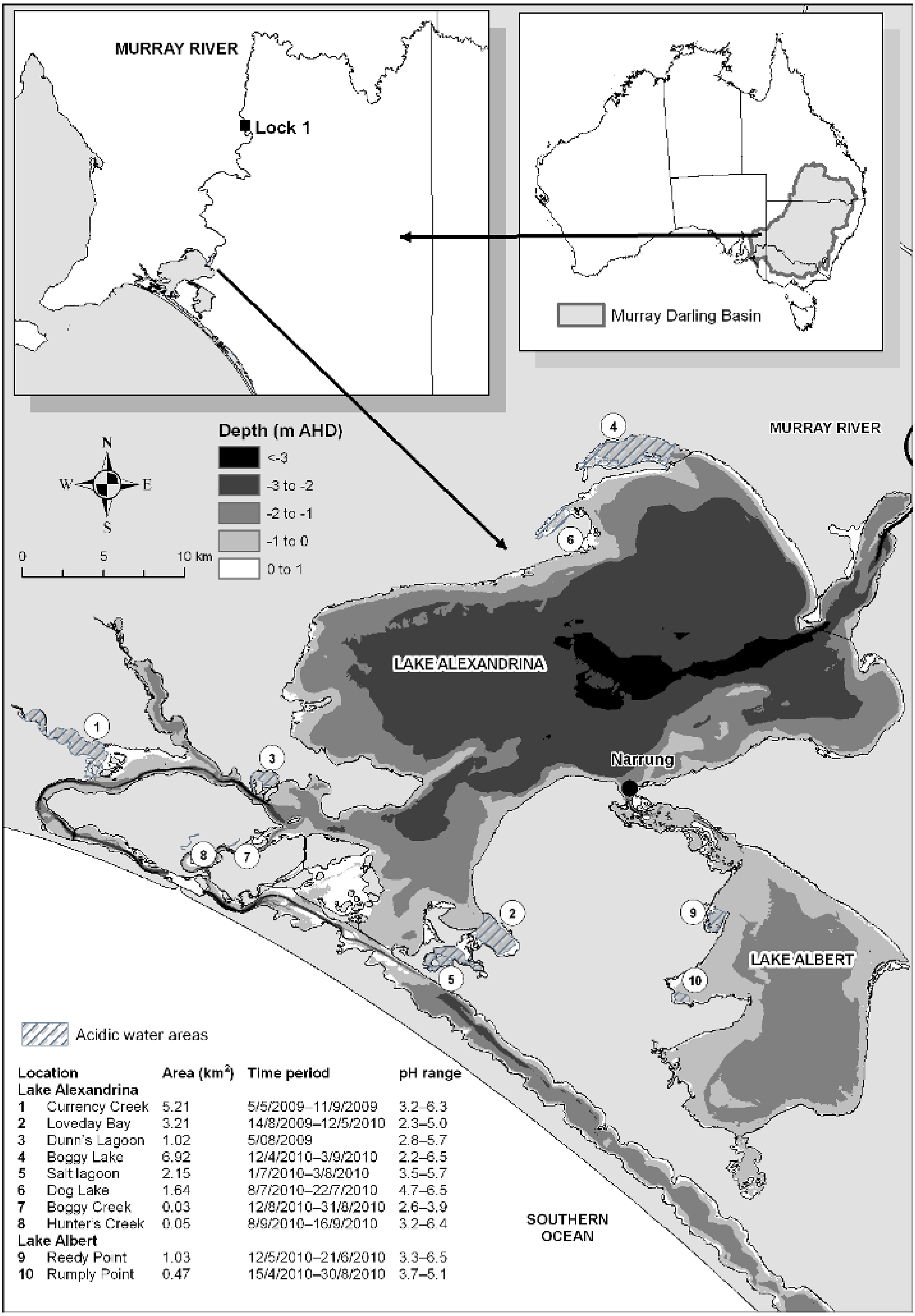
The hydraulic gradients were estimated at the three sites with transects data, Point Sturt (on Lake Alexandrina), and Campbell Park and Windmill (on Lake Albert). The gradients were generally low < 1/1000. Also in the near shore region the gradient was often away from the lake. This gradient away from the lake supported the process suggested in the conceptual model of evaporation in the near shore region reducing or reversing the flux of acidic water to the lakes. This is also consistent with Hipsey et al. (2010) who suggested that the main mechanisms for acid discharge to the Lakes would be through runoff and throughflow.

The hydraulic gradients suggested that evaporation generally caused flow from both the lake and land towards a region of the lake sediments 10s to 100s of metres from the shoreline. This would suggest that acidic products could collect beyond this region and then be washed off in rainfall events when runoff occurred or in seiching events. This would require acidity generated upslope in the oxidised sediments to be transported down to this evaporative accumulation zone. The estimated maximum distance that a water particle could travel during the monitoring period was calculated and found to be < 15 m. Also the upward vertical hydraulic gradient was shown to be orders of magnitude > the horizontal gradient, so it is more likely that the water particle will travel to the surface than move horizontally. This suggested that evaporative concentration of acidity from up slope would be a minor concern and any evaporative concentration was only likely from vertical flux of acidity.

The water content data was used to estimate the capillary length scale (section 3) of the soil/sediments and the distance that saturated soils/sediments could be expected away from the waters edge (section 2). This varied from 10s of metres to 100s metres and is likely to have significantly reduced the volume of pyritic sediments that would have otherwise oxidised. The capillary length scale was also used to estimate the percentage of time soil at set depths would be below saturation. For example at the Point Sturt site the sediment at a depth of 0.2 m would be below saturation for 82, 64, 50 and 28% of the time for points 1, 2, 3 and 4 respectively (Table 4). This illustrates that the likelihood of oxidation of the soil/sediments increases with the greater the distance from the water's edge. Although *in* *situ* calibration of these sensors was not carried out this is unlikely to have affected these estimations.

The other mechanisms suggested for acid flux in the conceptual model was exfiltration and runoff. The acid flux from these processes was estimated from the rainfall, piezometric head and water content data (sections 2 and 4). The analysis done here is only very approximate to illustrate that this is the most likely mechanism for acid flux to the lakes when compared to groundwater flux. The estimated values are also much less than estimates of acid fluxes suggested by (Hipsey and Salmon, 2008) and more recently by (Earth Systems, 2010). The estimated acid fluxes are orders of magnitude less than those suggested by Earth Systems (2010) and suggest that Lake Alexandrina in particular is not likely to become acidic. It however, must be noted that this is based on extrapolation of data from one transect at one site.

Lake Albert because of its smaller size (both diameter and depth) is more vulnerable to acidification. The buffering capacity of Lake Albert is large and given the results presented here and the suggested acidity potential of sediments in Lake Albert it was prudent that pumping of water into Lake Albert occurred to maintain the level. Again the fact that these results are based on two transects means that the results need to be treated with caution. The results for Lake Albert suggest that localised pockets of acidity are possible where the volume of the water and hence alkaline buffering capacity is small compared to the acid flux. This is likely to occur in embayments where the receiving water volume is small and circulation with the larger water body is low. Support for this suggestion is indicated by measurements of areas where acidity has been found in the lakes (figure 39).



Location		Area (km^2)	Time period	pH range
Lake Alexandrina				
1	Currency Creek	5.21	5/5/2009–11/9/2009	3.2–6.3
2	Loveday Bay	3.21	14/8/2009–12/5/2010	2.3–5.0
3	Dunn's Lagoon	1.02	5/08/2009	2.8–5.7
4	Boggy Lake	6.92	12/4/2010–3/9/2010	2.2–6.5
5	Salt lagoon	2.15	1/7/2010–3/8/2010	3.5–5.7
6	Dog Lake	1.64	8/7/2010–22/7/2010	4.7–6.5
7	Boggy Creek	0.03	12/8/2010–31/8/2010	2.6–3.9
8	Hunter's Creek	0.05	8/9/2010–16/9/2010	3.2–6.4
Lake Albert				
9	Reedy Point	1.03	12/5/2010–21/6/2010	3.3–6.5
10	Rumply Point	0.47	15/4/2010–30/8/2010	3.7–5.1

Figure 39. Map of acidic water areas for Lakes Alexandrina and Lake Albert. Map provided by SA EPA.

This report suggests that considerable further information was able to be extracted from the water table monitoring and water content data obtained by Earth Systems (2010). However, the number of transects was only three and caution is required in extrapolating the results presented in this report to inferred behaviour for the whole of the sediments surrounding the lakes.

The other mechanisms of acid discharge from diffusion of inundated sediments following lake level rise is not considered in this report. Modelling by Hipsey et al. (2010) based on the mecocosm experiments of Hicks et al. (2009) suggested that this could be a significant source of acidity into the lakes and this will be examined in more detail in a future report (Cook et al, 2011).

REFERENCES

Armstrong, J. and Armstrong, W. (1988). *Phragmites australis* – A preliminary study of soil-oxidising sites and internal gas transport pathways. New Phytologist **108(4)**: 373-382.

Cook, F.J., Dobos, S.K., Carlin G.D. and Miller, G.E. (2004). Oxidation rate of pyrite in acid sulfate soils: *in situ* measurements and modelling. Australian Journal of Soil Research **42**: 499-507.

Cook, F.J. (2001). Vertical and horizontal components of flow on a sloping bed with evaporation. Unpublished report.

Cook, F.J. and Knight, J.H. (2003). Oxygen transport to plant roots: Modeling for physical understanding of soil aeration. Soil Science Society of America Journal **67**: 20-31.

Cook, F.J., Knight, J.H. and Wooding, R.A. (2009). Steady groundwater flow to drains on a sloping bed: Comparison of solutions based on Bossinesq equation and Richards equation. Transport in Porous Media **77**, 357-372.

Cook ,F.J., McLachlan, G., Corkhill, E. and Mosley, L. (2011). Physical properties of soils/sediments of Lower Murray Lakes and modelling of acid fluxes. In Prep.

DENR (2010). Acid sulfate soils research program summary report. Prepared by the Lower Lakes Acid Sulfate Soils Research Committee for the SA Department of Environment and Natural Resources, Adelaide.

Earth Systems (2010). Quantification of acidity flux rates to the Lower Murray Lakes. Prepared by Earth Systems Pty Ltd.for the Department for SA Department of Environment and Natural Resources, Adelaide.

EPA (2011). Lower Lakes Groundwater Acidification Risk Monitoring Project Monitoring Report February 2011. Environment Protection Authority South Australia.

Fitzpatrick, R.W., Marvanek, S.P., Shand, P., Merry, R.H., Thomas, M. and Raven, M.D. (2008). Acid sulfate soil maps of the River Murray below Blanchetown (Lock 1) and Lakes Alexandrina and Albert when water levels were at pre- drought and current drought condition, CSIRO Land and Water Science Report 12/08. CSIRO, Adelaide, 10 pages.

Fitzpatrick, R.W., Grealish, G., Chappell, A., Marvanek, S. and Shand, P. (2010). Spatial variability of subaqueous and terrestrial acid sulfate soils and their properties, for the Lower Lakes South Australia. Prepared by the Commonwealth Scientific and Industrial Research Organisation (CSIRO) Land and Water for the SA Department of Environment and Natural Resources, Adelaide.

Hicks, W.S., Creeper, N., Hutson, J., Fitzpatrick, R.W., Grocke, S. and Shand, P. (2009). The potential for contaminant mobilisation following acid sulfate soil rewetting: field experiment, Prepared by The Commonwealth Scientific and Industrial Research Organisation (CSIRO) Land and Water, for the SA Department of Environment and Natural Resources.

Grayson, R.B., Bloschl, G. and Moore, I.D. (1995) Distributed parameter hydrologic modelling using vector elevation data: THALES and TAPES-C. In 'Computer models of watershed hydrology.' pp. 669-696. (Water Resources Publications: Colorado USA)

Hillel, D. (1980). Fundamentals of Soil Physics. Academic Press, 413p.

Hipsey, M.R. and Salmon, S.U. (2008). Numerical assessment of acid-sulfate soil impact on the River Murray Lower Lakes during water level decline, Final Report, Centre for Water Research, The University of Western Australia.

Hipsey, M.R., Bursch, B.D., Colleti, J. and Salmon, S.U. (2010). Lower lakes Hydro-geochemical model development and assessment of acidification risks. Prepared by University of Western Australia for SA Water.

Jury, W.A., Gardner, W.R., and Gardner, W.H. (1991). Soil Physics, 5th Edition. John Wiley & Sons, New York.

Kellners T.J., Soppe, R.W.O., Robinson, D.A., Schaap, M.G., Ayars, J.E. and Skaggs, T.H. (2004) Calibration of capacitance probe sensors using electric circuit theory. Soil Science Society of America Journal **68**, 430-439.

Kellners, T.J., Robinson, D.A., Schouse, P.J., Ayars, J.E. and Skaggs, T.H. (2005) Frequency dependence of the complex permittivity and its impact on the dielectric sensor calibration in soils. Soil Science Society of America Journal **69**: 67-76.

Kirby, C.S. and Cravotta III, C.A. (2005). Net alkalinity and net acidity 1: Theoretical considerations. Applied Geochemistry **20**: 1920-1940.

McCutcheon, S.C., Martin, J.L and Barnwell, T.O. Jr. (1993). Water Quality in Maidment, D.R. (Editor). Handbook of Hydrology, McGraw-Hill, New York, New York.

Rigby, P.A., Dobos, S.K., Cook, F.J. and Goonetilleke, A. (2006). Role of organic matter in framboidal pyrite oxidation. Science of the Total Environment **367**: 847-854.

Simmons, C.T., Narayan, K.A. and Wooding, R.A. (1999) On a test case for density-dependent groundwater flow and solute transport models: The salt lake problem. Water Resources Research **35**: 3607-3620.

Simpson, S., Jung, R., Jarolimek, C. and Hamilton, I. (2009), The potential for contaminant mobilisation following acid sulfate soil rewetting: lab experiment. Prepared by the Commonwealth Scientific and Industrial Research Organisation (CSIRO) Land and Water for the SA Department of Environment and Natural Resources, Adelaide.

Simpson S.L., Fitzpatrick RW, Shand P, Angel BM, Spadaro DA and Mosley LM (2010). Climate-driven mobilisation of acid and metals from acid sulfate soils, Marine and Freshwater Research 61, 129–138.

Šimůnek, J., and Šejna, M. (2007). The HYDRUS-2D/3D Software Package for Simulating Two- and Three- Dimensional Movement of Water, Heat, and Multiple Solutes in Variably Saturated Media, User Manual, Version 1.02. PC-Progress, Prague, Czech Republic. 203p.

Sullivan, L.A., Bush, R.T., Ward, N.J., Fyfe, D.M., Johnston, M., Burton, E.D., Cheeseman, P., Bush, M., Maher, C., Cheetham, M., Watling, K.M., Wong, V.N.L., Maher R. and Weber, E. (2010). Lower Lakes laboratory study of contaminant mobilisation under seawater and freshwater inundation. Prepared by Southern Cross GeoScience for the SA Department of Environment and Natural Resources, Adelaide.

Tong, J. X., Yang, J.Z., Hu, B.X and Bao R.C. (2010) Experimental study and mathematical modeling of soluble chemical transfer from unsaturated-saturated soil to surface runoff. Hydrological Processes, **24**(21): 3065-3073

Webster, I.T., Cook, F.J. and Hicks, W.S. (2008). Appendix C. Review of Hipsey and Salmon: Numerical Assessment of Acid-Sulfate Soil Impact on the River Murray Lower Lakes During Water Level Decline. In Peer Review of Acidification Thresholds for Lake Alexandrina and Lake Albert, Aquaterra (2008), 38-50.

White, I. and M. J. Sully (1987). Macroscopic and microscopic capillary length and time scales from field infiltration. Water Resources Research **23**(8): 1514-1522.

Wooding, R.A., Tyler, S.W. and White, I. (1997) Convection in groundwater below an evaporating salt lake .1. Onset of instability. Water Resources Research **33**, 1199-1217.

van Genucten M.T., 1980. A closed-form equation for predicting the hydraulic conductivity of unsaturated soils. Soil Sci. Soc. America J., **44**:892-898.

Yates S.R., Van Genuchten M.T, Warrick A.W., Leij F.J. (1992) Analysis of measured, predicted, and estimated hydraulic conductivity using the retc computer-program. Soil Science Society of America Journal **56**, 347-354

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