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# Streamwater-Groundwater Interaction: The River Murray at Hattah-Kulkyne Park, Victoria: Summary of Results

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## Summary

In lowland rivers, water is continuously exchanged between the river and its floodplain through groundwater. In semiarid regions, modifications to this exchange (through dams, weirs, etc) have been implicated in river and floodplain salinisation and loss of riparian vegetation. However, the impact of modifying the groundwater – surface water regime on the nutrient cycles of rivers and floodplains is not as well known. This is of concern because the interface between groundwater and surface water is known to be an active site for biogeochemical transformations. Thus, modifications to the groundwater – surface water regime may have important implications for the load and the form in which nutrients are delivered to rivers through groundwater. Changes in the load, the form, and the timing of nutrient delivery could affect water quality in rivers, such as by promoting an increased occurrence of noxious algal blooms.

The nature of groundwater – surface water interaction and the spatial patterns in nutrient concentration at the interface between surface and groundwater were investigated for two years at a riparian site on the River Murray in Hattah-Kulkyne Park (Victoria). One transect of piezometer nests was established for each of the two types of riparian settings occurring in this area (gently sloping sand banks and steeper clay banks). Groundwater – surface water interactions were investigated using a combination of piezometric surface monitoring and environmental tracers ( $\text{Cl}^-$ ,  $\delta^2\text{H}$  and  $\delta^{18}\text{O}$ ). Nutrient cycles in riparian groundwater were investigated using a combination of water samples from piezometers and unsaturated zone pore water profiles.

While the study area is a regional groundwater discharge zone, piezometric surface monitoring and environmental tracers both indicated that groundwater discharge occurred in the floodplain but not in the river. Thus, the River Murray is primarily a losing stream at Hattah. Gradients towards the river were found in the riparian zone following a moderate flood in late 2000, indicating that the river was temporarily gaining through bank discharge during this period.

Recharge of the alluvial aquifer by surface water occurred by bank infiltration at baseflow and by diffuse recharge during floods. A significant increase in  $\text{Cl}^-$  concentration of shallow groundwater was noted following the late 2000 flood. This suggested that some of the solutes that had accumulated in the unsaturated zone during the preceding inter-flood periods had been displaced to the groundwater during the flood. Spatial distribution in environmental tracers suggests that the origin of groundwater in the Hattah floodplain is variable. Groundwater near the river appears primarily derived from surface water recharge while at distance it is mostly diffuse recharge from rainfall.

Groundwater in the riparian zone was generally enriched in nutrients relative to nearby surface waters. With the exception of an oxygenated fringe near the water table and the river, groundwater was usually anoxic at both transects. This resulted in relatively elevated concentrations of filterable reactive phosphorous, ammonium, and dissolved iron, and low concentrations of  $\text{NO}_3^-$  in groundwater. However, with the exception of hyporheic exchange, groundwater will not contribute to the nutrient budget of the River Murray at baseflow because it is a losing stream at this site. Bank discharge following floods will contribute to the nutrient load to the river but this has not been quantified.

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Complex nutrient cycles at the groundwater – unsaturated zone interface occur in semiarid floodplains. Unsaturated zone profiles demonstrated that both  $\text{Cl}^-$  and  $\text{NO}_3^-$  accumulated in the unsaturated zone during inter-flood periods. Nitrate accumulation was greatest near the soil surface, where warmer and well-oxygenated conditions should favour nitrification. However, while groundwater  $\text{Cl}^-$  concentration increased following the late 2000 flood,  $\text{NO}_3^-$  concentration did not. This suggests that  $\text{NO}_3^-$  produced in the unsaturated zone during the inter-flood period was rapidly consumed (probably denitrified) once displaced to the groundwater following the flood. While the significance of unsaturated zone processes are well recognized for the water and salt balance of floodplains, their role in nutrient cycles is poorly understood at present.

Results from this study provide a good basis into understanding mechanisms for groundwater – surface water interaction in floodplain environments. Despite the success achieved, we suggest that standard piezometer networks, as used at the Hattah site, may not be the most appropriate method for future investigations. Several of the problems encountered, such as chemical integrity of groundwater samples, small hydraulic gradients at the scale of the riparian zone, and loss of equipment during floods, are discussed along with technical suggestions for future work.

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## Introduction

The importance of floodplains for the ecology of riverine systems is increasingly recognized (MDBC 2001). A popular model of how lowland rivers interact with their floodplains is the “flood-pulse” concept (Junk *et al.* 1989). Following this model, floods are the key to the exchange of organic matter and nutrients between the two systems because they establish a “lateral connectivity” between them. The tacit assumption is that this connection is severed once floods recede. However, it is also increasingly recognised that rivers and floodplains can be permanently connected through groundwater. While floods would be the main determinant that shapes the nature of interactions between groundwater and surface water, this connection will not necessarily be completely severed between flooding events.

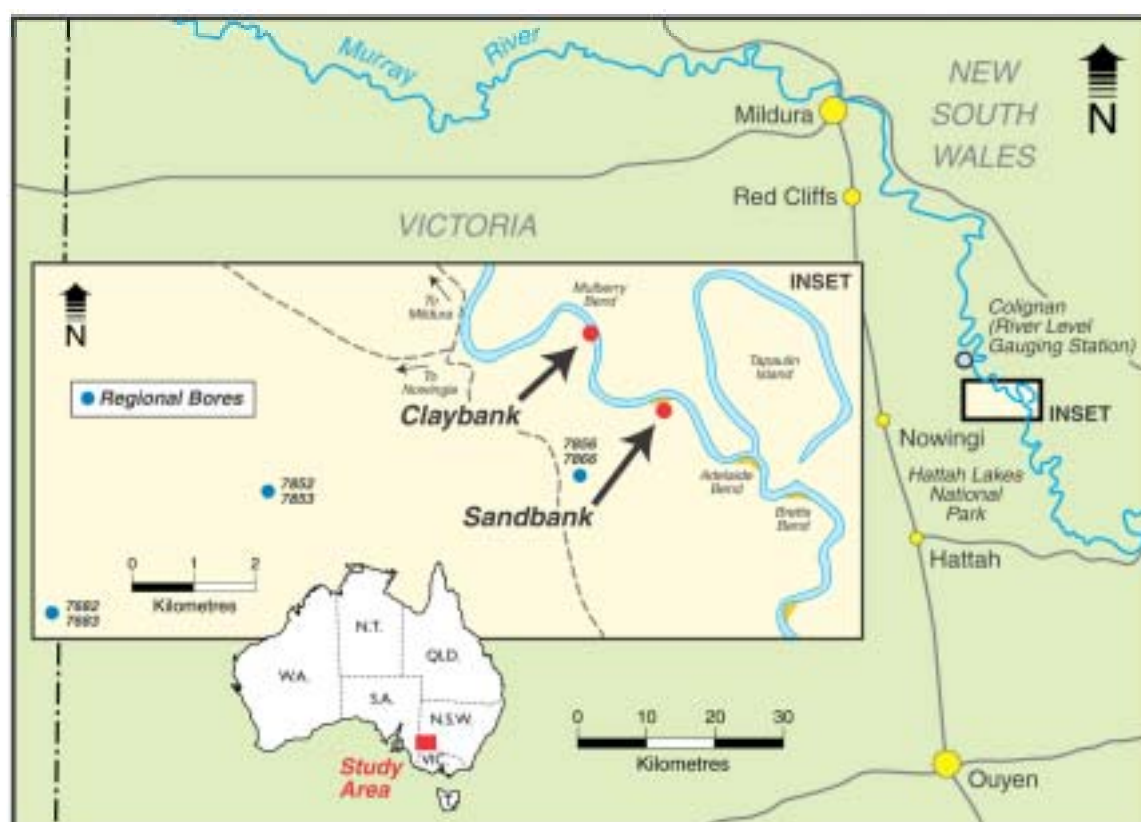
For the River Murray, the significance of groundwater in delivering salt loads to the river has received much attention in the last decades (MDBMC 1999). While natural sources of saline groundwater to the Murray have existed for centuries, irrigation in and near floodplains, raised water levels by dams and weirs, and dryland agriculture have exacerbated this problem during the last century. Groundwater – surface water interactions in floodplains are complex and may be altered by land-use change. Because floodplains are quite flat, small changes in river or groundwater level can completely modify the direction and magnitude of groundwater flow within the floodplain – river system. Changes in the hydrogeology of floodplains can significantly impact their water balance and salt cycle (Jolly 1996).

In contrast to salt, the role of groundwater for nutrient cycles in lowland rivers has received comparatively little attention. In addition to salinisation, increased incidence of algal blooms has contributed to degradation of water quality in many Australian rivers during the last century (Davis 1997). Algal blooms are controlled, in part, by the increased availability of nutrients such as nitrogen (N) and phosphorus (P). Most of the research on nutrient in rivers has been on in-stream cycling or in estimating gross sediment or catchment scale fluxes. It is not clear at the present whether groundwater is a significant source for these nutrients in lowland rivers, or how changes to hydrological processes in near stream environments has altered the nutrient cycles. It is also not known how significant groundwater – surface water interaction processes are to nutrient cycles within the riverine floodplain system. Unlike salt, which is transported conservatively throughout the landscape, many nutrients have complex cycles, especially in environments such as floodplains where the habitat is heterogeneous and biological productivity is high.

This report details a groundwater – surface water interaction study on the River Murray within Hattah-Kulkyne Park (Fig. 1). This site was chosen because it represents a relatively pristine riparian environment for the lower River Murray and it is a site that is being intensively monitored as part of a CRC for Freshwater Ecology project on carbon in lowland rivers. A similar study at the Wollombi Brook, NSW, will be presented in a companion report. The first aim of this report is to characterize the origin and the movement of groundwater in the floodplain – river system at Hattah. The second aim is to examine the patterns in nutrient concentration in groundwater at different periods of a flood cycle and to evaluate the significance of groundwater for the nutrient budget of the river and the floodplain.

## Site location and description

Two study sites were selected, both on the Victorian bank of the River Murray in Hattah-Kulkyne Park, ~90 km upstream of Mildura (Fig. 1). The Hattah-Kulkyne Park is on an open woodland floodplain dominated by mature river red gum (*Eucalyptus camaldulensis*). As is the case for much of the River Murray, the river in this area is regulated, primarily to guarantee sufficient water diversion in summer periods for irrigators and legislated minimum flow to South Australia. As a result, river levels rarely fall below a base value (~36 m AHD) with water releases upstream usually resulting in the river level rising by less than two metres.



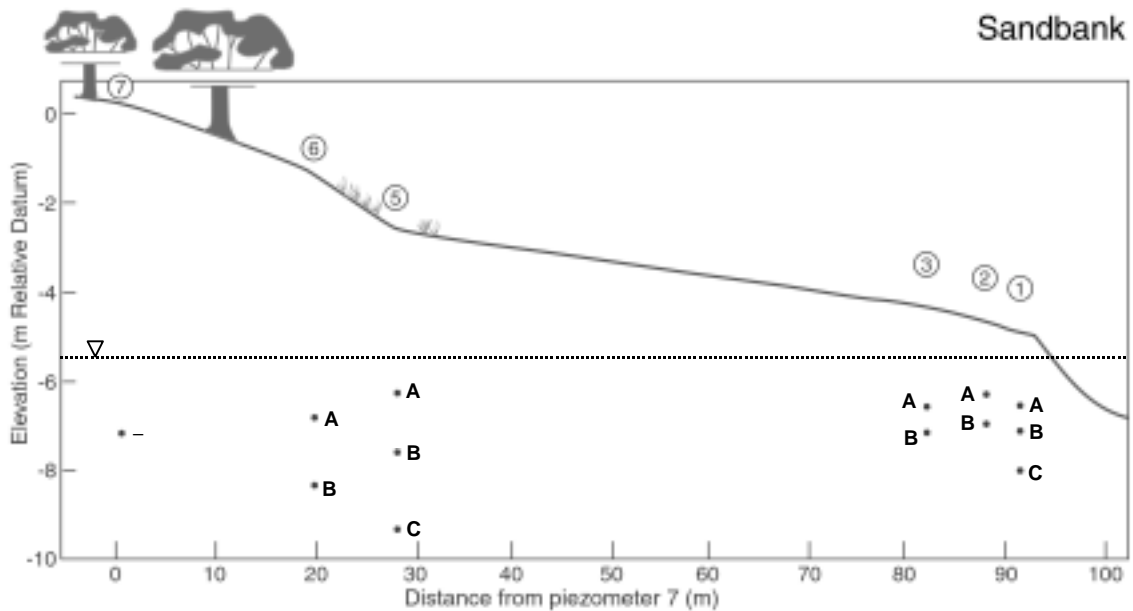
**Figure 1** Location of the Hattah-Kulkyne field site, the Sandbar and the Claybank piezometer transects, and the regional bores used during the study.

The first site (Sandbank or SB), was located on a sandy bank on a bend of the river. The sand bank itself is reasonably flat rising by ~2 m over a lateral distance of 60 m (Fig. 2a). Once the sand bank reaches the floodplain the incline is ~3 times greater, rising by ~3m over a lateral distance of 30m (Fig. 2a). The second site (Claybank or CB) is approximately one kilometre further downstream in an area with steep clay banks. The incline close to the river is ~2 m over a lateral distance of 5 m with the

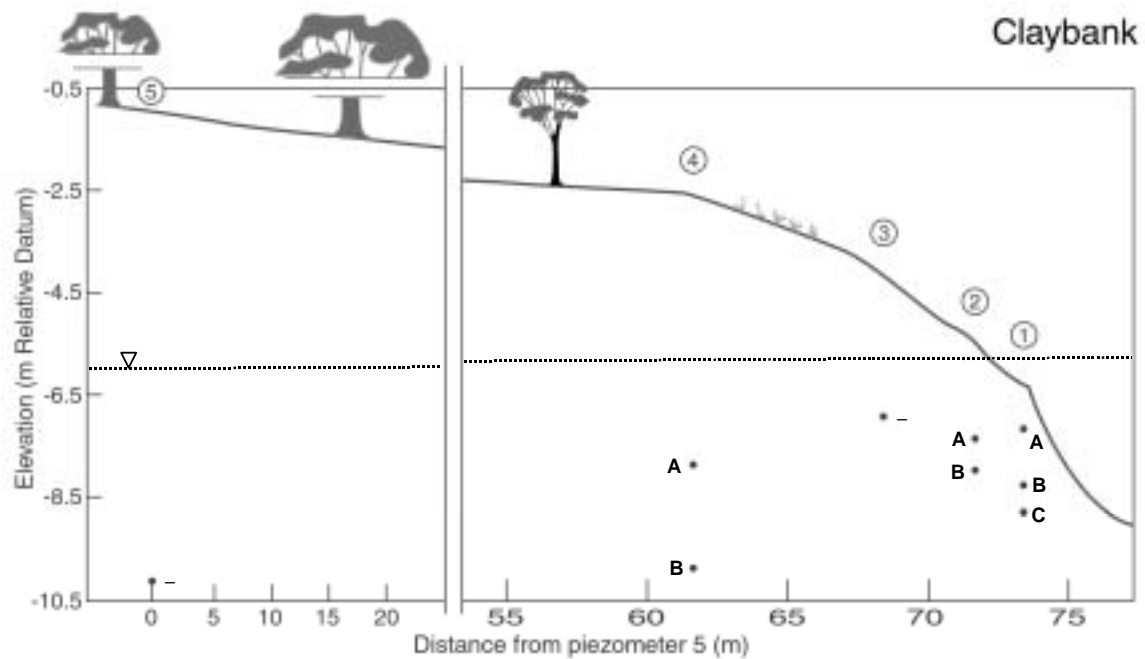
incline away from the river similar to that at the sandbank site (1 m over a lateral distance of 30 m; Fig. 2b).

### **Sampling and monitoring**

At each site, piezometers were installed during late March 1999 using a rotary drill-rig where access permitted, or by hand-augering for piezometers closer to the river. The piezometers were ~1-10 m deep and up to 90 m from the river (Table 1 and Fig. 2 a & b). Each piezometer was constructed of PVC, was approximately 50 mm in internal diameter, and was screened over the bottom 0.5 m of the piezometer with an outer lining of cloth material to minimize the entry of sand and silt during water sampling. Most of the piezometers were installed in nests, with screens at 2 or 3 different depths below the watertable. The piezometer installed ~50 m from the river at the Sandbank site (SB4) was removed by vandals soon after installation.

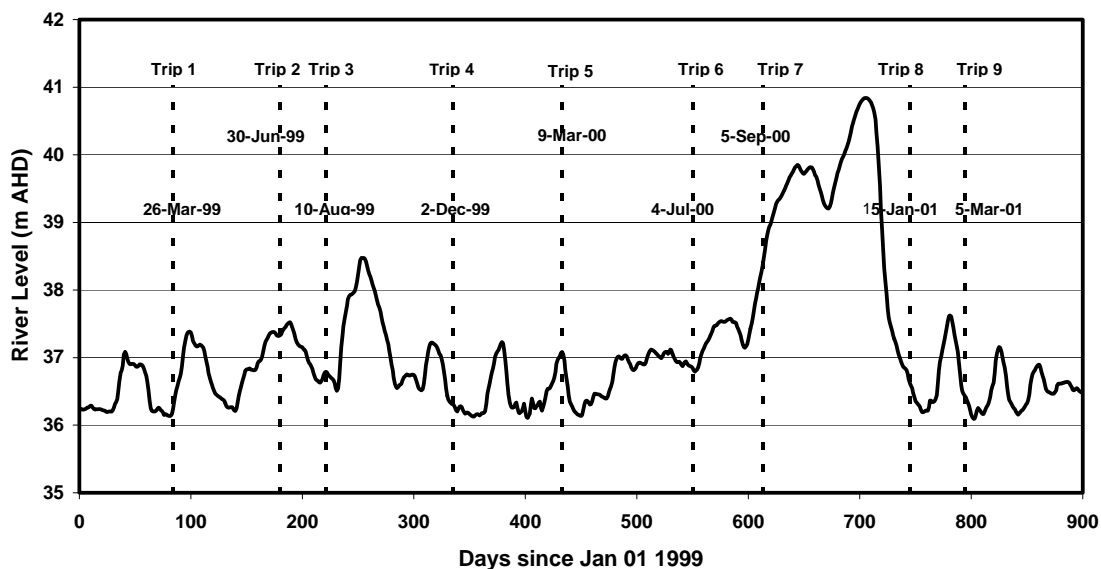


**Figure 2a** Cross-section of the Hattah Sandbank transect showing the position and depths of the monitoring piezometers.



**Figure 2b** Cross-section of the Hattah Claybank site showing the position and depths of the monitoring piezometers.

During the two years following the installation of the piezometers, water level was monitored in several of the piezometers using capacitance probes (GPSE model 32000). However, as a result of technical difficulties, the log was not continuous. In addition, there were 9 sampling trips to the study sites at 1-4 month intervals. The dates of the field trips are shown in relation to river stage height measured at Colignan, ~10 km downstream from the field sites (Fig. 3). Six of the nine field trips were at times of low flow. Two of the remaining field trips were during periods when the river flow had peaked. Trip 7 (6<sup>th</sup> September, 2001) was during the rising stage for the largest flood experienced during the study (~50 000 ML d<sup>-1</sup>). Road access was not possible once the river level reached ~39 m AHD.



**Figure 3 River level fluctuations at Colignan, with the dates of the sampling trips indicated**

On these trips, the following measurements were made and samples collected.

- Manual water level measurements.
- Field measurements for EC, pH, dissolved oxygen and temperature for groundwater and river water using a WTW Multiline P4 Universal meter.
- Samples for  $\delta^2\text{H}$  and  $\delta^{18}\text{O}$ , chloride ( $\text{Cl}^-$ ), and, for selected trips, major ions, field alkalinity, and nutrients.

On the first day of each sampling trip, the standard sampling procedure consisted of first measuring the standing water level and then pumping all of the piezometers either to dryness or for a minimum of three piezometer standing water volumes. On the following day, where possible, a further three volumes were pumped from each piezometer. Water quality samples were collected, ensuring minimal exposure to the atmosphere by inserting the end of the pump line into the bottom of a large erlenmeyer flask and allowing it to overflow. A sub-sample was collected from the bottom half of the flask using a large syringe equipped with a three-way valve. Samples were immediately filtered on-line with a  $0.45\ \mu\text{m}$  Supor<sup>®</sup> membrane filter (Pall). A small filtered subsample was collected for  $\text{Cl}^-$  analysis in a scintillation vial. Another subsample was collected in a 25-mL McCartney bottle for the analysis of the stable isotope of water. The remaining filtered water was collected in a 125-mL bottle and acidified to  $\text{pH} < 2$  using analytical grade HCl. Prior to each sampling trip, all sampling bottles and filtering equipment was washed in P-free detergent and in a mild acid bath before thorough rinsing with distilled deionised water. For piezometers that were slow to recover, it was not possible to remove three piezometer volumes prior to sample collection and collection commenced soon after pumping on the second day. Analytical methods are outline in Appendix 1.

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On the final field trip, groundwater was collected from 6 shallow regional boreholes that had been originally installed by the Victorian Rural Water Commission and are currently monitored by the Department for Natural Resources and Environment, Victoria. The bores were located in pairs at ~1.5 to 8 km from the river (Fig. 1). The watertable and screens were ~6 and ~12-14 m respectively below ground level for each pair of boreholes. River level data was available from the Colignan gauging station (Fig. 1). Also on the final trip, two soil cores (Profiles 1&2) were taken between SB5 and SB6 at the sandbank site by hand-augering to the maximum depth possible before the hole started collapsing. Samples were collected at 0.25 m intervals, placed in 500 ml sealed glass jars and placed in an ice-filled esky to keep the samples chilled. At the laboratory, the samples were analysed for gravimetric water content and chloride using standard procedures (Appendix 1). Within a few hours of collection, samples for soil exchangeable  $\text{NH}_4^+$  and  $\text{NO}_3^-$  concentrations were extracted in a field laboratory using a 2M KCl solution (10:1 brine volume: wet soil mass ratio). Ammonium and  $\text{NO}_3^-$  extracts were kept at 4°C and analysed at the Adelaide analytical unit facility using standard procedures (Appendix 1).

## Results

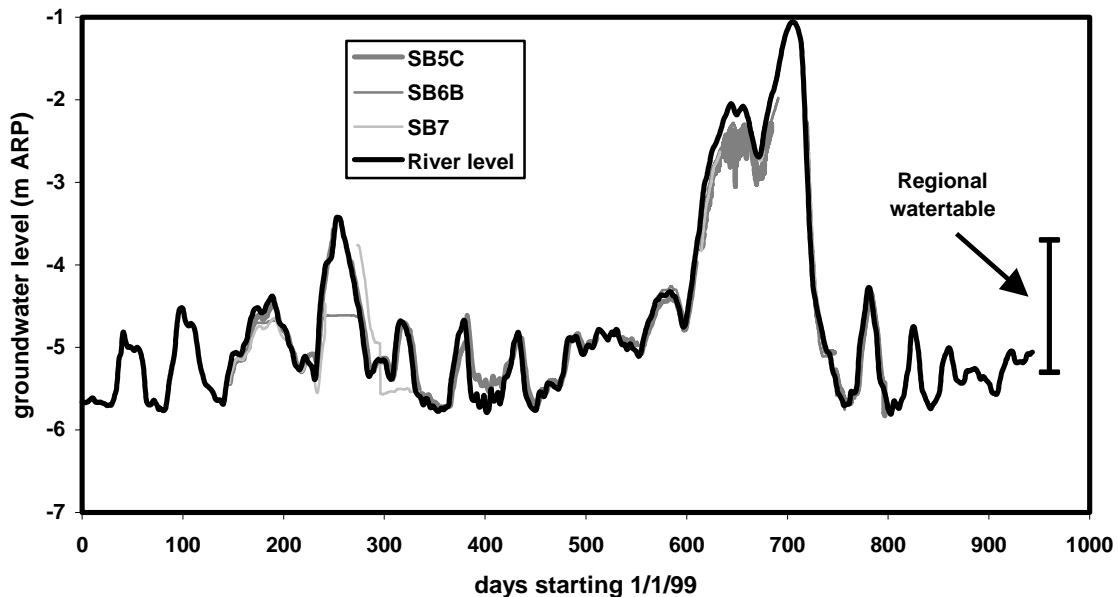
### *Water level variations*

Minor floods (10 000 – 20 000 ML d<sup>-1</sup>) during the study period resulted in river levels rising by 1-2 m from baseflow levels. The major flood for the study period (early September 2000 to mid January 2001) resulted in river levels increasing by almost 5 m with a maximum flow of ~50 000 ML d<sup>-1</sup> (Fig. 3). The minimum measurement error in water level would be that associated with surveying the height of the top of casing for the piezometers (estimated at ±0.03 m). Additional errors of measurement using the watertable recorders are mainly due to drift in calibration as a function of aging and changes in the height of the piezometers as a result of vandalism and floods. Manual measurements during sampling trips provided an opportunity for recalibration and also gave an indication of the long-term errors associated with automated recorders. Although this was variable between piezometers and measurement method, we have assumed an overall error of ±0.05 m for watertable measurements using either automated recorders or manual readings. Water level measurements are reported relative to an elevation reference point at the Sandbank site (ARP, Above Red Post). The height datum for ARP (~42 m Australian Height Datum, AHD) was determined by assuming that the river levels at times of low and stable flow corresponded to that measured at the Colignan river gauging station (ie no gradient during these times).

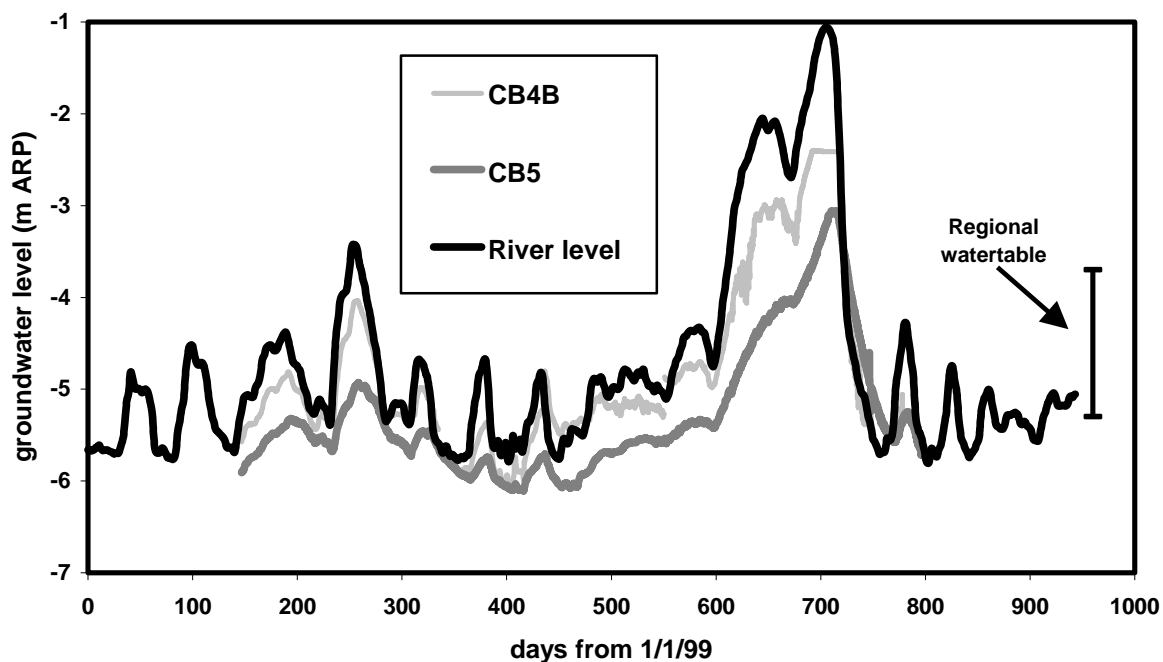
At the Sandbank site, water level records are only available for piezometers located at the junction of the sandbank and the floodplain and on the floodplain itself (SB5C, SB6B and SB7). The potentiometric level of the groundwater was close to that of the river for most of the study period, including the period of the major flood (Fig. 4a). There is, however, evidence for a period of up to 10 days for the changes in river level to be reflected at piezometers on the floodplain.

At the Claybank site, the potentiometric level of the groundwater was often lower than the river level, particularly during floods (Figure 4b). For example, the watertable at the piezometer farthest from the river was, on average ~1 m lower than the river and up to 2.5 m during the major flood.

The Victorian Department for Natural Resources and Environment has carried out quarterly monitoring of groundwater levels in the park since the early 1980s. For the three shallow bores closest to the field sites in this study, variations in the watertable were <0.3 m for the bore farthest from the river (7682) and up to 1.5 m for bores closer to the river (7853 and 7856). The locations of these bores are shown in Figure 1. For the period corresponding to this study (early 1999 to early 2002), the fluctuations at individual bores were <0.6 m with an overall range from 36.7 – 38.3 AHD, corresponding approximately to –3.7 to –5.3 ARP (as shown in Figs. 4a and 4b). In other words, except for periods of flood, the regional watertable was higher than that of the river and of the groundwater close to the river at both of the sites.



**Figure 4a** Potentiometric groundwater levels for bores installed with water level recorders at the Sandbank site.



**Figure 4b Potentiometric groundwater levels for bores installed with water level recorders at the Claybank site. Plateaus in groundwater level indicate that true water levels were above the piezometers.**

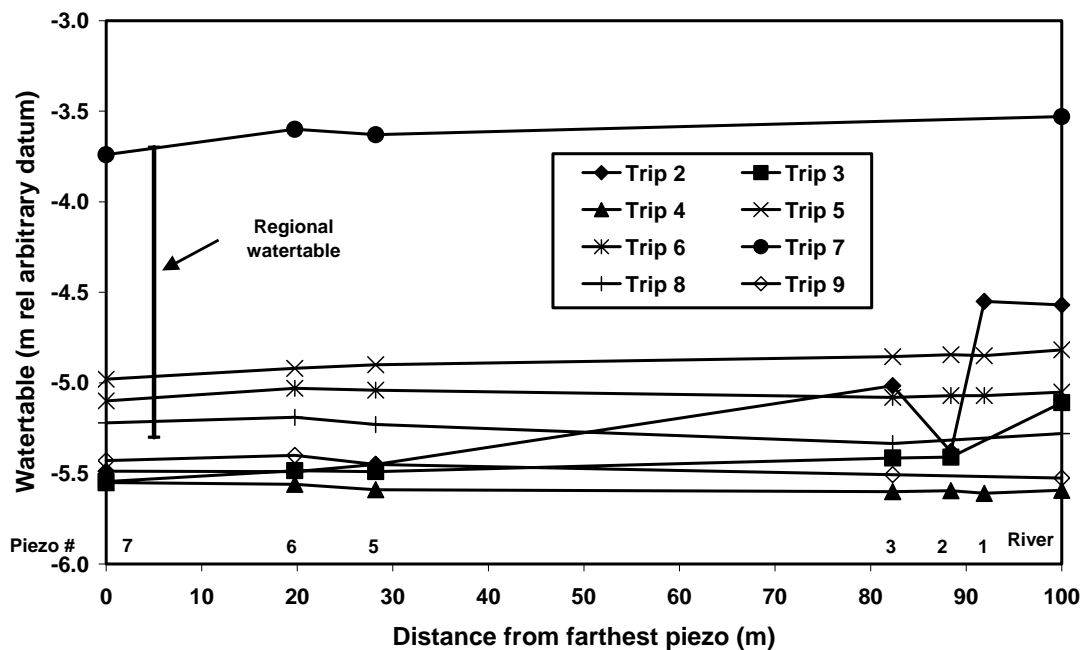
In addition to the temporal watertable data available from the continuous recorders, manual measurements were made at the piezometers during all sampling trips except trip 1. Assuming a  $\pm 0.05\text{m}$  error of measurement for each piezometer, the watertable at the Sandbank site appears relatively flat for all of the sampling trips for the piezometers within 30 m of the reference piezometer (Fig. 5a). All of these piezometers are located on or within a few metres of the floodplain. For the manual measurements, there is no significant difference in elevation in watertable height for piezometers close to the river (piezometers 80 & 90m from reference) compared to the floodplain (piezometers 0-30 m from reference) for all trips except trip 2 (Fig. 5a). The manual measurements are consistent with data from the automated water level recorders.

The manual measurements made during the second sampling trip suggests that the river level was  $\sim 0.8$  m higher than the water table in the riparian zone at the time of sampling. The sampling trip was also made at the height of a small (1.5 m) flood event and probably reflects a lag time associated with the potentiometric head at the river equilibrating with that on the floodplain. Note that this was not observed for a smaller flood during sampling trip 5 and, more surprisingly, for the rising limb of the

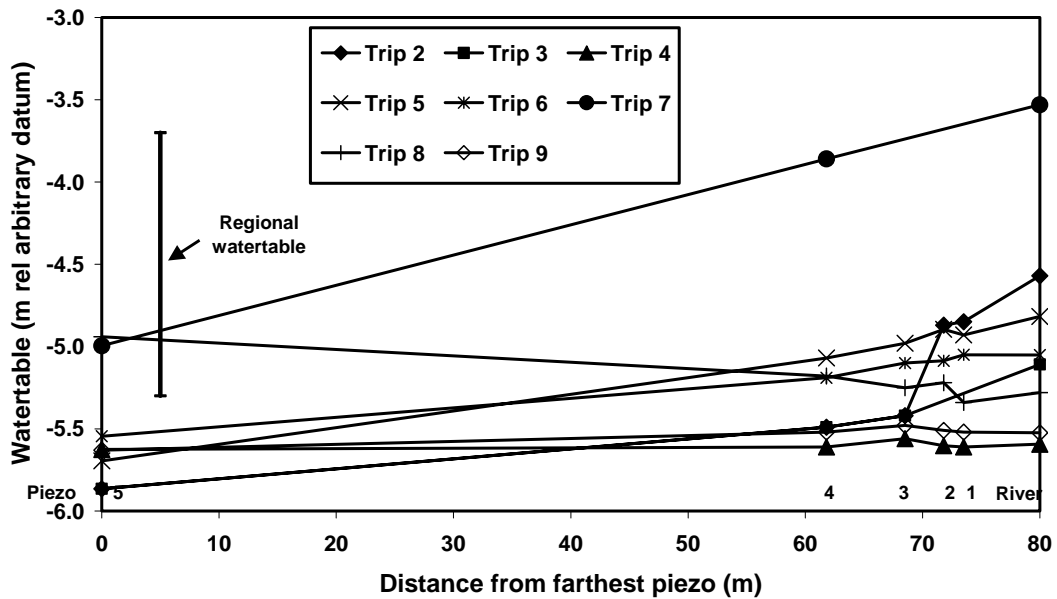
larger flood event (trip 7). This indicates that once the river covers most of the sand bank, there is greater hydraulic connection between the sand bank and the floodplain.

At the Claybank site, the watertable at the piezometer farthest from the river (CB5) is 0.1 – 1.5 m lower than that of the river and piezometers close to the river for all of the trips, excluding trips 8 and 9 (Fig. 5b). The greatest gradient was observed during trip 7, on the rising limb of the major flood. At the time of sampling trip 8, the major flood had receded and watertable at the river was lower than the surrounding groundwater. At sampling trip 9, the watertable was almost flat. This data indicates that the flow at the Claybank site is primarily from the river to the surrounding aquifer and that flow to the river only follows periods of high to very high flows. There is also a considerably longer lag time for the groundwater to respond to river level changes than that observed at the Sandbank site.

The mean potentiometric level of groundwater for bores 7682 and 7853 is 1 – 1.5 m higher than the river level during the period of study (excluding the period of the major flood) and 1.5 – 2 m higher than bores CB5 and SB7 in the floodplain adjacent to the river. The mean potentiometric level of groundwater for bore 7856 is close to the mean river level and slightly higher than CB5 and SB7 for this period. Bore 7856 is located in an area subject to inundation during large floods and, as discussed later, contains groundwater that is considerably fresher than the other bores sampled.



**Figure 5a** The water table and river level at the Sandbank site for sampling trips 2 to 9.



**Figure 5b** Watertable and river level at the Claybank site for sampling trips 2 to 9.

In summary, the watertable at the Sandbank site was flat for most of the study period (Table 1), probably due to the high hydraulic conductivity of the aquifer at that site. Only during sampling trip 2 was the river level and watertable close to the river substantially higher than that away from the river.

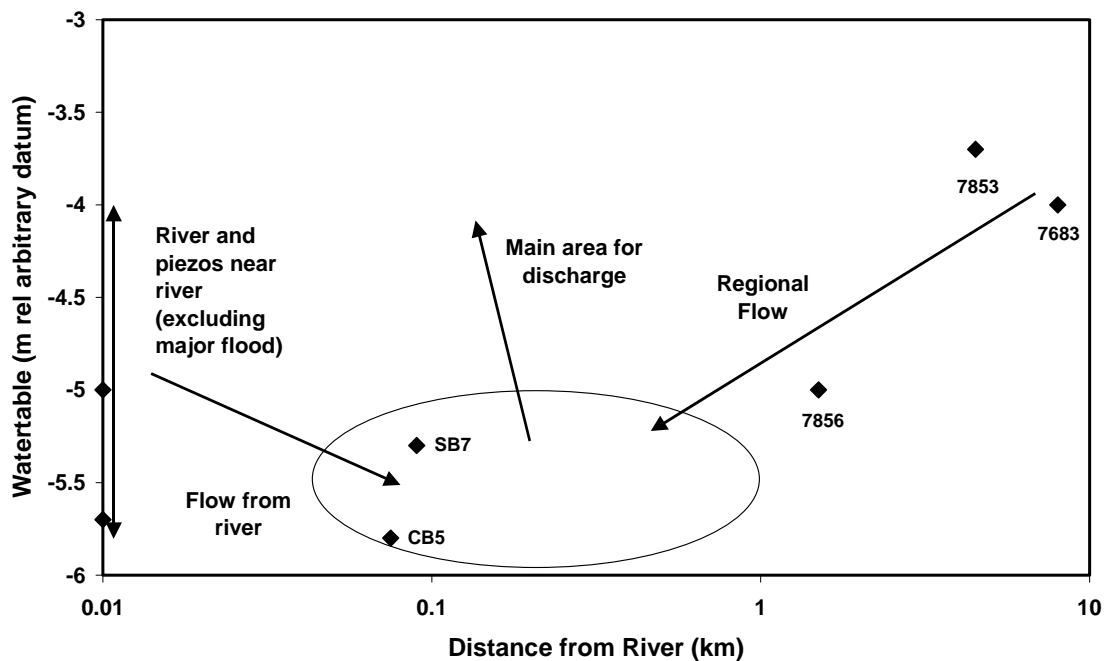
**Table 1** Hydraulic gradient at the Sandbank and Claybank sites at the time of sampling.

Trip #	*River level	Hydrograph position	Hydraulic gradient Away (a) or towards (t) river	
			Sandbank	Claybank
2	-4.57	Peak (1.5 m flood)	0.0 – 0.1 (a)	0.009 – 0.09 (a)
3	-5.11	Base after small flood	Flat	0.007 - 0.08 (a)
4	-5.59	Base after small flood	Flat	Flat
5	-4.82	Peak (0.8 m flood)	Flat	0.009 – 0.012 (a)
6	-5.05	Base level	Flat	0.007 – 0.010 (a)
7	-3.53	Rising arm of main flood	Flat	0.02 (a)
8	-5.28	Base after main flood	Flat	0.004 (t)
9	-5.53	Base after small flood	Flat	Flat

\* River level height relative to red post trig point (ARP) at sand bank site.

At the Claybank site, the observed hydraulic gradient suggests that flow was from the river to the groundwater for most of the study period. The gradient was flat for two of the sampling trips, coinciding with periods when the river level was at its lowest (-5.59 m and -5.53 m relative to ARP for trips 4 and 9 respectively). After the major flood event (trip 8), there was groundwater flow to the river (hydraulic gradient 0.004).

Overall, the River Murray in this area is probably a “losing” river for most of the time, with the exception of short periods following medium to large flood events. Regional groundwater is very saline with flow towards the river. It is probable that little if any of the regional flow discharges directly into the Murray. The preferred conceptual model is one where lateral flow of saline regional groundwater mixes with freshwater “leakage” from the river (Fig. 6). The mixed groundwater then discharges either via transpiration or evaporation in the floodplain (most probably within a kilometer or two of the River Murray).



**Figure 6 Mean potentiometric level for shallow groundwater from monitoring regional bores and piezometers during this study compared with river levels**

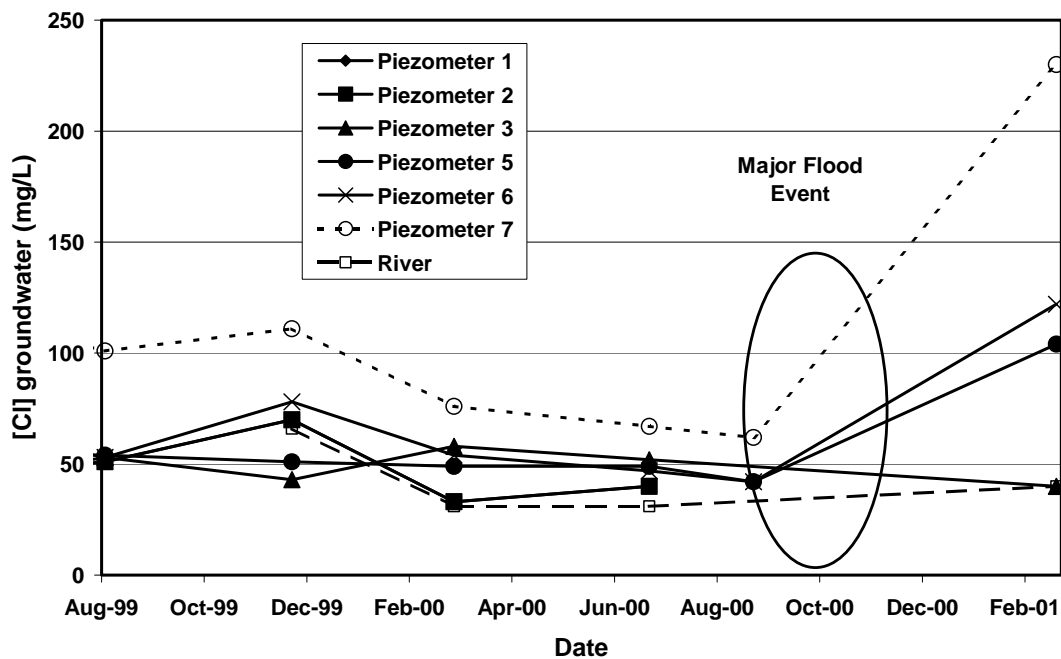
## ***Environmental tracers***

### **Chloride concentration of groundwater and soilwater**

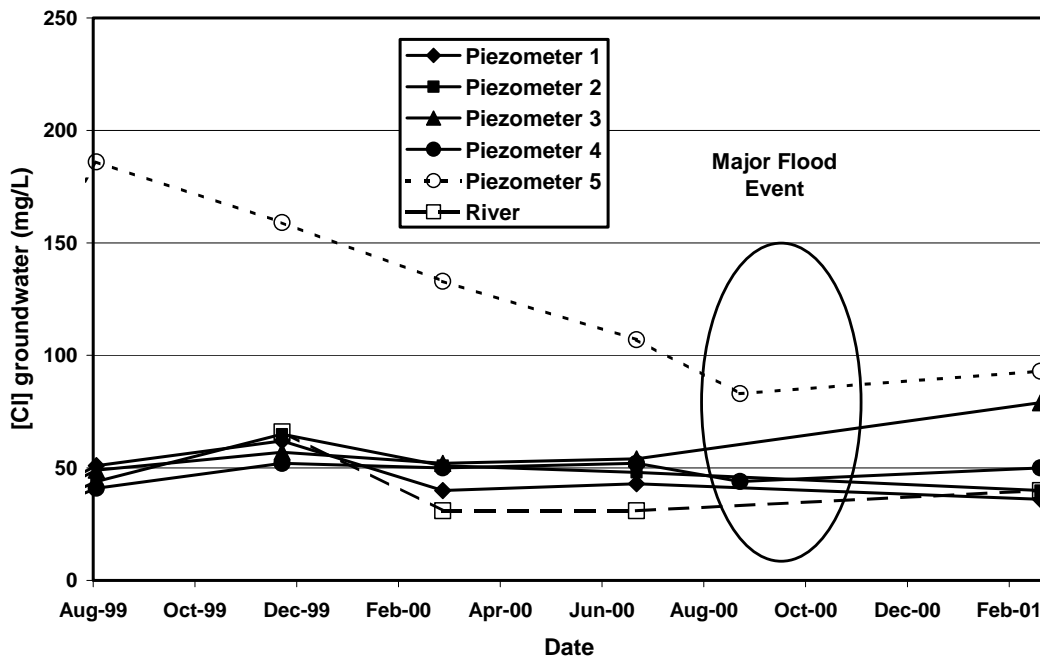
For ease of presentation, patterns in the environmental tracers, as discussed in this and following sections, will generally be presented as averages by piezometer nests or for all sampling trips. Details on a trip-by-trip basis can be found in Appendix 2 for each piezometer.

The spatial and temporal changes in soil water and groundwater  $\text{Cl}^-$  concentration are often used to evaluate the movement of water in aquifers (e.g. recharge, discharge, leakage, etc.) because  $\text{Cl}^-$  is considered conservative and non-reactive. Chloride concentration in groundwater from riparian zone piezometers and in river water was relatively low, ranging between 30 – 200 and 30 – 70  $\text{mg L}^{-1}$ , respectively (Figs. 7 a&b). For comparison, chloride concentration in the regional groundwater bores ranged between 5700 – 18 000  $\text{mg L}^{-1}$ .

Chloride concentration in groundwater closest to the river (SB1 and CB1) shows small temporal changes at both sites and was usually similar to river water. At the piezometers farthest from the river (SB7 and CB5), the chloride concentration decreased throughout the monitoring period until the late 2000 major flood event (Figs. 7 a&b). After the flood, a small increase was measured at CB5 and a much larger increase at SB7. While the decline in groundwater  $\text{Cl}^-$  concentration during the inter-flood period was not as noticeable in the piezometers at an intermediate distance from the river, they also had a significant increase in  $\text{Cl}^-$  concentration following the late 2000 flood.



**Figure 7a** Temporal changes in the chloride concentration of the groundwater for the Sandbank piezometers and for the river.



**Figure 7b** Temporal changes in the chloride concentration of the groundwater for the Claybank piezometers and for the river.

Under base flow or near base flow, there is flow of fresh water from the river laterally into the sandy aquifer material at both the Sandbank and Claybank sites. This results in freshening the aquifer to a progressively greater distance from the river. Using a point dilution test, the rate of groundwater flow towards the floodplain was estimated at  $10 - 20 \text{ cm d}^{-1}$  at the Claybank site (Lamontagne *et al.* 2002). Thus, longer periods between floods will result in the expansion of a shallow fresh groundwater lens on either side of the river. The gradual mixing of infiltrating river water with groundwater would account for the decreasing trend in  $\text{Cl}^-$  concentration in riparian groundwater during the inter-flood period.

### Unsaturated zone profiles for $\text{Cl}^-$

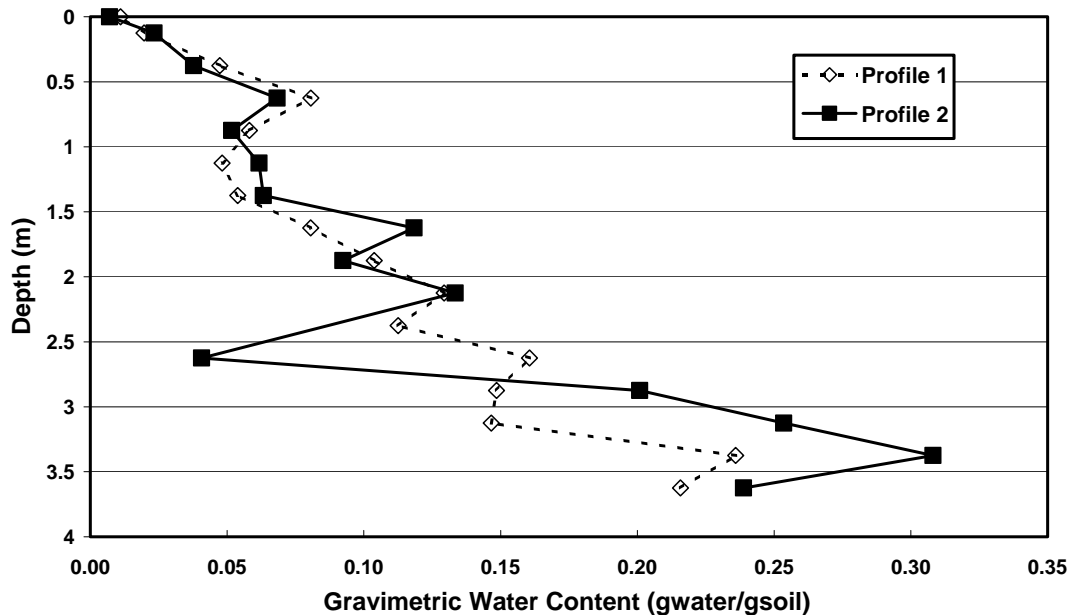
A significant increase in  $\text{Cl}^-$  concentration in shallow piezometers away from the river was observed following the late 2000 flood. We hypothesized that this  $\text{Cl}^-$  originated from flushing of salts that had accumulated in the unsaturated zone during the preceding baseflow period. To test the hypothesis that soilwater with a high chloride concentration accumulates in the riparian zone between flood events, two unsaturated zone profiles were taken between SB5 and SB6 in early March 2001, approximately 50 days after the flood had receded. These profiles were collected from the area where the exposed sand meets the riparian forest and would have been inundated by the flood.

The gravimetric water content ( $\theta_g$ ) profile was typical for an unsaturated zone undergoing evapoconcentration. Soil  $\theta_g$  ranged from 0.02 at the surface to near

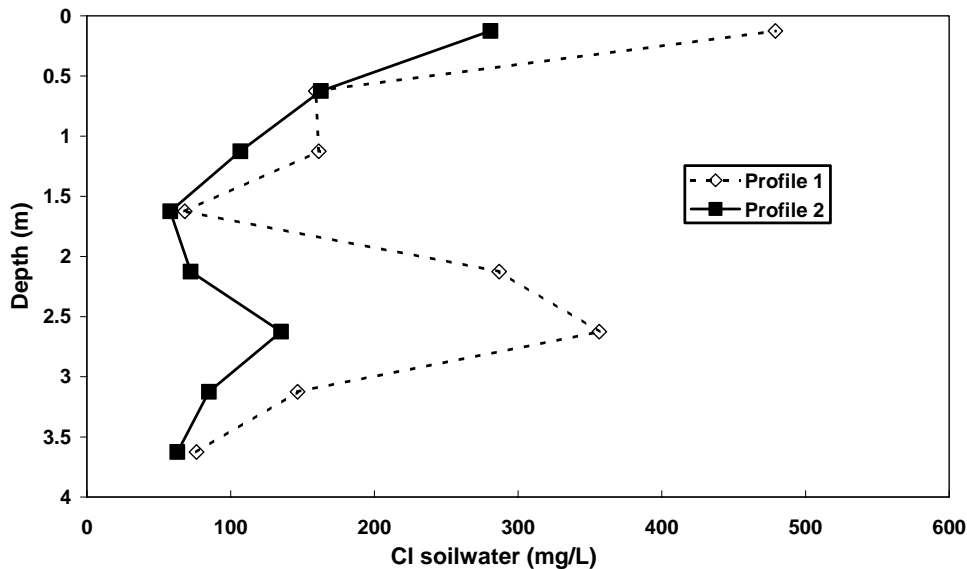
saturation ( $\theta_g = 0.25 - 0.3$ ) close to the watertable (Fig. 8). From a depth of  $\sim 0.5$  m to the surface, the soil profile becomes progressively drier presumably as a result of evapotranspiration. During sampling, it was observed that this depth range corresponds to that with the greatest intensity of roots at the site. Low water content at a depth of 2.5 m in Profile 1 was also probably caused by the presence of roots.

The patterns in soil water  $\text{Cl}^-$  were more complex than for  $\theta_g$ , with peaks in concentration near the surface and near the water table (Fig. 9). The peak in  $\text{Cl}^-$  concentration near the surface is consistent with an ongoing evapoconcentration of residual floodwater. However, the peak in  $\text{Cl}^-$  at a depth of  $\sim 3$  m may represent pre-flooding shallow soilwater displaced during the flood. At the second site (Profile 2), the 3 m  $\text{Cl}^-$  bulge is smaller, suggesting that most of the soil water was mixed with groundwater during the flood.

The factors that will determine the increase in groundwater salinity (as indicated by  $\text{Cl}^-$ ) following a flood are the amount of salt in the unsaturated zone and the potential for vertical recharge. The amount of salt in the unsaturated zone will be greater when the unsaturated zone is deeper and when there is infrequent flushing. The potential for vertical recharge is greater when the surface soil has a greater hydraulic conductivity (i.e., is sandier).



**Figure 8** Gravimetric water content of the unsaturated zone sampled between piezometers SB5 and SB6 (March 2001).



**Figure 9 Chloride concentration of soil water sampled between piezometers SB5 and SB6 (March 2001).**

This is the case at the start of the floodplain at the Sandbank (near SB5 – 7) where the unsaturated zones profiles were taken. At the Claybank site, the vertical hydraulic conductivity of the surface soil could be sufficiently low to restrict recharge following flood events and hence a smaller increase in groundwater salinity was observed at that site.

#### Unsaturated zone Cl<sup>-</sup> mass-balance

The mean chloride concentration of the soilwater was  $\sim 200 \text{ mg L}^{-1}$  at the time of sampling. This would probably be lower than the soil chloride concentration prior to the flood because the period of accumulation since the flood ( $\sim 50$  days) was much shorter than the previous inter-flood period ( $\sim 500$  days). Nevertheless, this value is used as a first approximation for the pre-flood soil water Cl<sup>-</sup> concentration. We also assume that soil water was primarily displaced to the water table by vertical piston flow during the flood. The amount of water present in the unsaturated zone was  $\sim 0.9 \text{ m}$  at the time of sampling (assuming a soil bulk density =  $1.5 \text{ g cm}^{-3}$  and an unsaturated zone depth of  $6 \text{ m}$ ). Thus, assuming a saturated porosity of  $\sim 0.37$ , the combined soil water pool could fill  $\sim 2.5 \text{ m}$  of the aquifer. In other words, a single flushing of salt from the unsaturated zone, would, via piston-flow, result in the chloride concentration equaling  $\sim 200 \text{ mg L}^{-1}$  for the top few metres of groundwater. This is consistent with post-flood values observed at sites SB5 – SB7 ( $100\text{--}225 \text{ mg L}^{-1}$ ). While several assumptions are involved, it is clear that displacement of soil water could easily account for the increase in groundwater Cl<sup>-</sup> observed after the late 2000 flood.

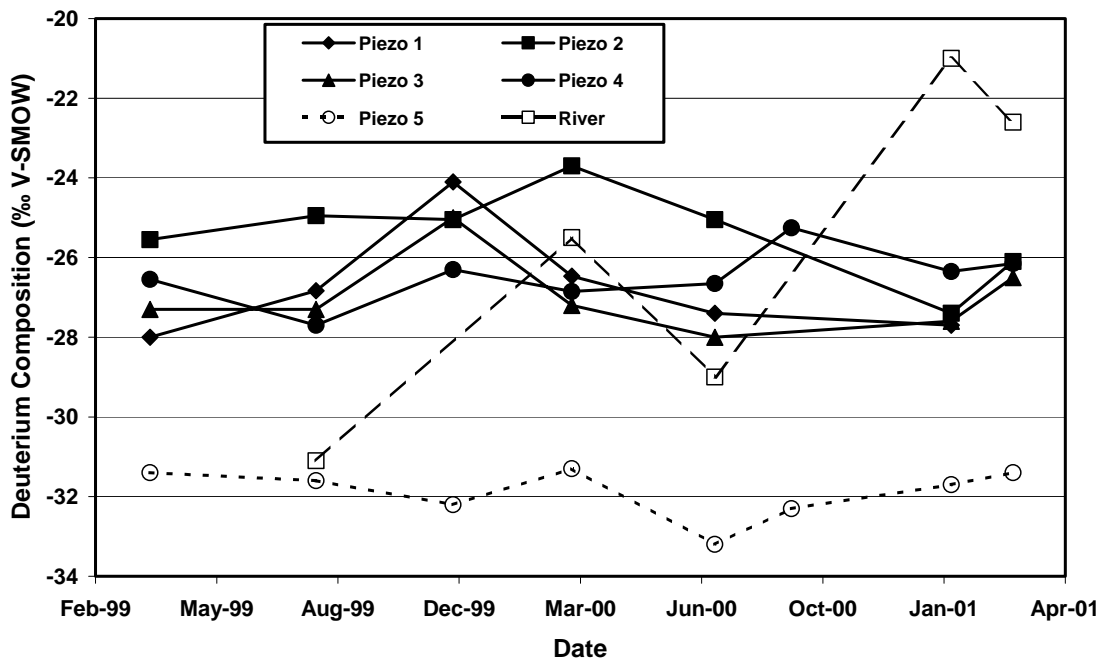
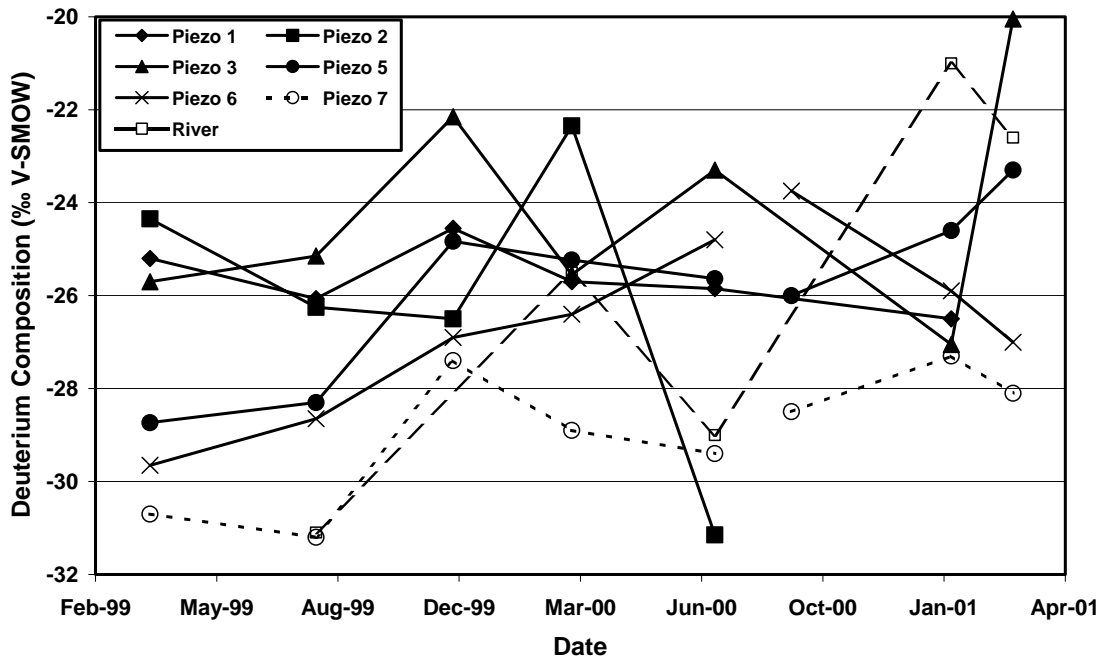
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## Stable isotope composition of groundwater

The  $\delta^2\text{H}$  composition of groundwater at the two sites ranges from  $-20$  to  $-31$  ‰V-SMOW (Fig. 10). There are greater temporal fluctuations at the Sandbank site than at the Claybank site. The  $\delta^{18}\text{O}$  composition of the groundwater ranges from  $-3.5$  to  $-5.5$  ‰ (data not plotted) with similar fluctuations observed to that for the deuterium data.

The  $\delta^2\text{H}$  and  $\delta^{18}\text{O}$  composition for the groundwater at the piezometers farthest from the river are different from the isotope composition of the groundwater at piezometers closer to the river at each site. In this case, the groundwater at the piezometers farthest from the river has a depleted isotopic signature (more negative) than the others. These patterns are consistent with infiltrating river water as the main source of water for the piezometers close to the river. Under low or average flow conditions, river water tends to be isotopically enriched relative to rainfall because of surface water evaporation (Simpson and Herczeg 1991). The more depleted signature for groundwater in the piezometers furthest from the river suggests that this water does not originate from infiltrating river water under baseflow conditions. However, recharge following large flood events could have such a depleted isotopic signature because heavy rain tends to have a relatively negative isotopic composition and there is relatively less evaporation in comparison to the volume of discharge during these periods.

Overall, the temporal changes in isotopic composition of the groundwater before and after the main flood are not as obvious as the ones seen for chloride concentration. However, because transpiration by plants has little effect on the isotopic composition of soil water (Walker *et al.* 2001), soil water will not necessarily have a strong contrast in isotopic signature relative to groundwater even if chloride does show such a contrast.



**Figure 10** Temporal changes in the deuterium composition of groundwater for piezometers at the Sandbank (top) and the Claybank sites (below).



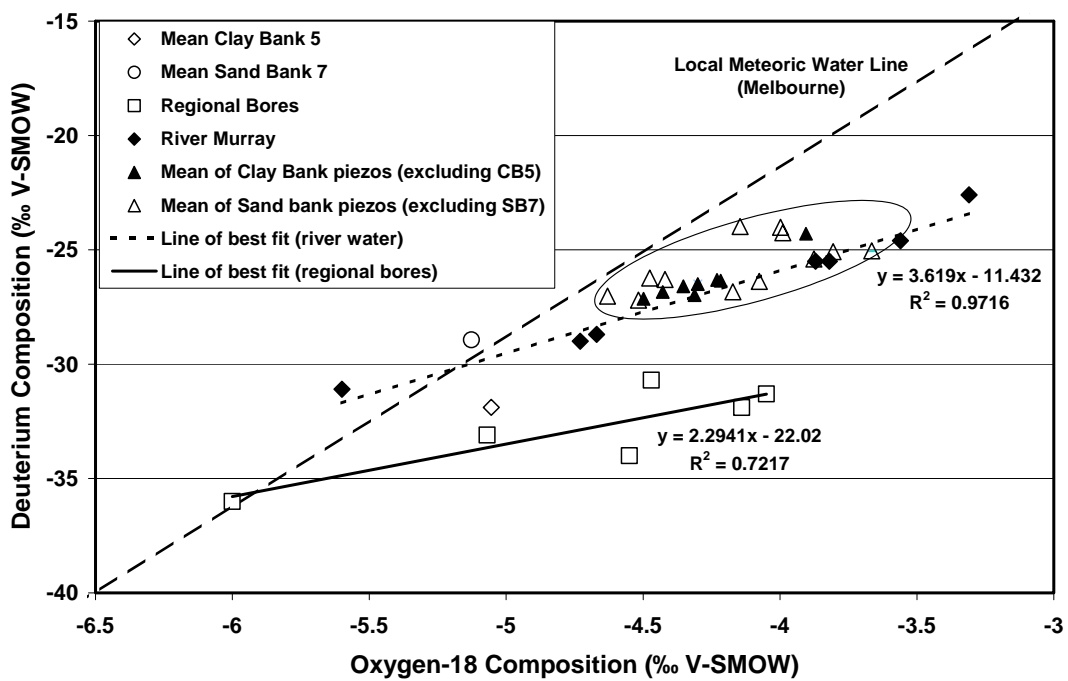
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mechanism of recharge to groundwater. The LMWL indicates the expected range in potential isotopic composition for rainfall in the study area. Regional groundwater typically has a  $\delta^2\text{H}$  and  $\delta^{18}\text{O}$  signature similar to mean winter rainfall. However, when a source of water undergoes evaporation, samples will tend to plot to the right of the LMWL (Fig. 12). The slope of this departure from LMWL provides some insight into the kind of evaporation process occurring (i.e., surface evaporation, water table evaporation, etc).

The  $\delta^2\text{H}$  and  $\delta^{18}\text{O}$  of groundwater and surface water from the Hattah floodplain area are distributed along two distinct evaporation lines (Fig. 12). Samples from all sites plot on a much lower slope than that of the Local Meteoric Water Line (LMWL), which is defined by the long-term rainfall record from Melbourne. Regional bores (including the piezometer furthest from the river at the Claybank site) plot on a lower slope (2.29) than that of the samples from the River Murray and the piezometers within 80 metres of the river (slope = 3.62). The offset in stable isotopic composition for the two data sets indicate that there is little mixing between the regional groundwater and the shallow groundwater adjacent to the river.

The trend followed by stable isotope compositions of the regional groundwater (Fig. 12) is typical of that of groundwater from the mallee area of the Murray Basin (Herczeg *et al.* 2001). That is, the trend to the right of the LMWL with a low slope indicates partial evaporation in the soil zone following predominantly winter derived rainfall source (indicated by a 'depleted' intercept on the LMWL). The shallow piezometers and River Murray surface waters all fall on the same general trend, although the shallow groundwater from the Sandbank site tend to have a greater scatter. The slope of 3.62 is lower than the overall trend of surface evaporation from the River Murray (5.1 – 5.9, Simpson and Herczeg 1991). And the intercept on the LMWL ( $-30\text{‰}$  and  $-5.2\text{‰}$  for  $\delta^2\text{H}$  and  $\delta^{18}\text{O}$ , respectively) is more negative than that of the long-term record of the River Murray from Merbein ( $-24\text{‰}$  and  $-4.4\text{‰}$  for  $\delta^2\text{H}$  and  $\delta^{18}\text{O}$ , respectively) (Herczeg, unpublished data).

The results reinforce the previous observations that recharge of the extremities of the Sandbank and Claybank occurs during large intermittent floods. The subsurface waters are then subjected to evaporation from the shallow water tables, and then these modified floodwaters are flushed back to the river in subsequent flood recession periods. Closer to the river there appears to be a greater influence of exchange between surface water and shallow groundwater. Thus there is a continuum of recharge to the groundwater with continual recharge exchange near to the river ranging to intermittent recharge/discharge for the extremities of the regularly flooded area.



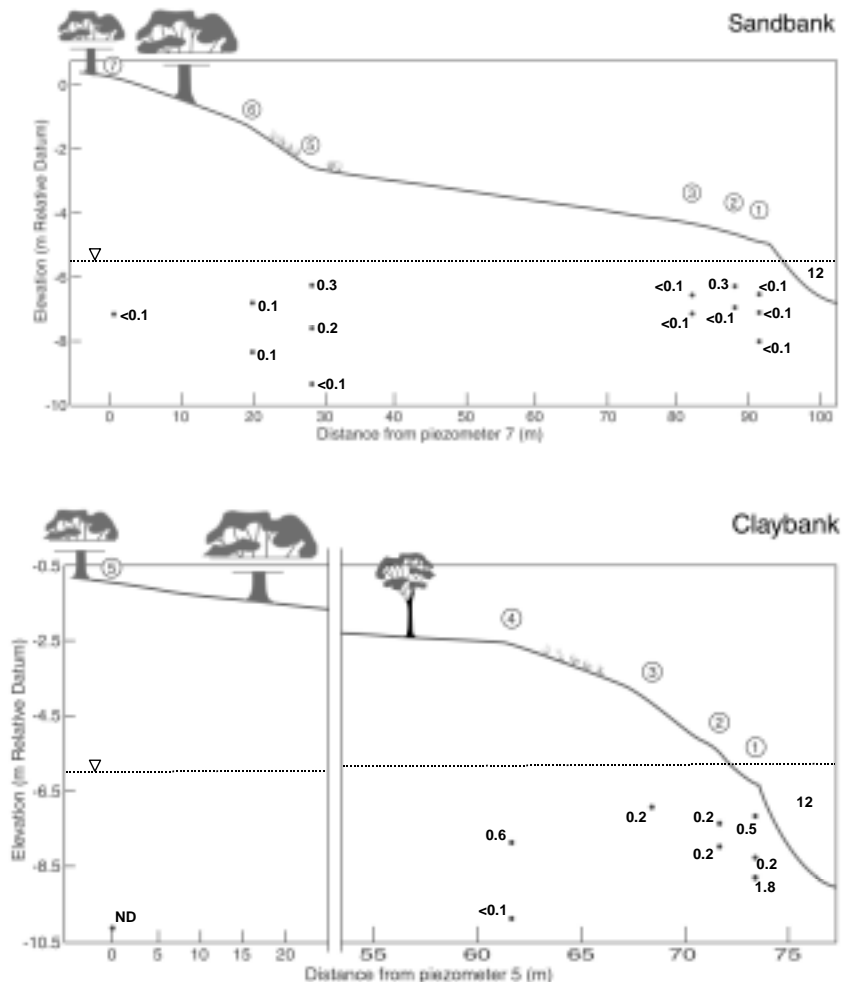
**Figure 12** Deuterium and oxygen-18 plot for water samples across the Hattah floodplain.

## Groundwater chemistry and nutrients

In this section, a brief overview of the patterns in water chemistry and nutrient concentration in riparian groundwater will be presented. Details for each sampling trip are also available in Appendix 2.

### Dissolved oxygen and pH

In general, groundwater in the riparian zone at both transects was suboxic to anoxic (i.e.,  $<1 \text{ mg O}_2 \text{ L}^{-1}$ ), as revealed by low DO measurements (shown for sampling trip 3, Fig. 13), high dissolved Fe concentrations, and copious amounts of Fe oxides precipitates in unflushed piezometers. Shallow piezometers  $>20\text{m}$  from the river occasionally had measurable but low oxygen concentrations. While field pH was generally circumneutral, groundwater tended to be slightly more acidic than river water (Table 2).



**Figure 13** Oxygen concentration ( $\text{mg L}^{-1}$ ) on 10 August 1999 at the Sandbank and the Claybank sites.

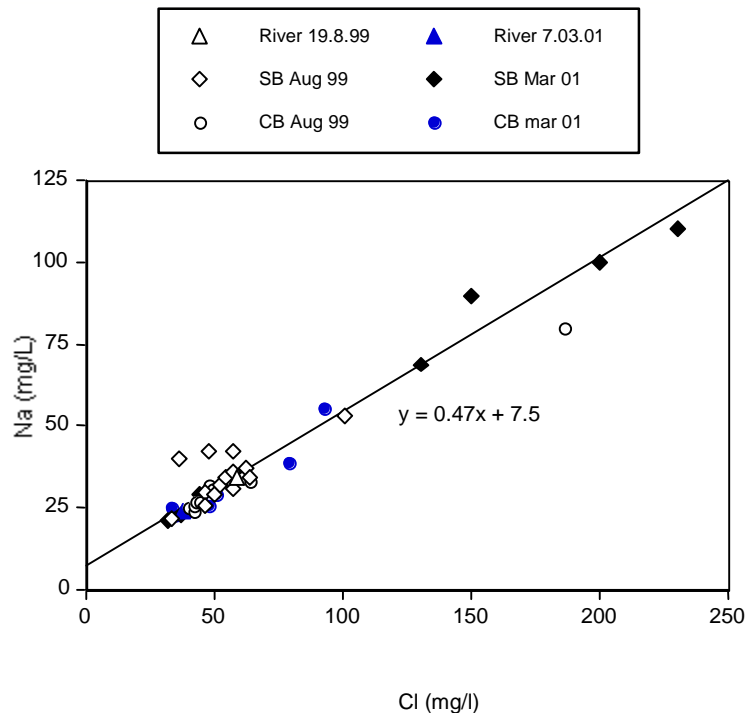
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## Major ion chemistry

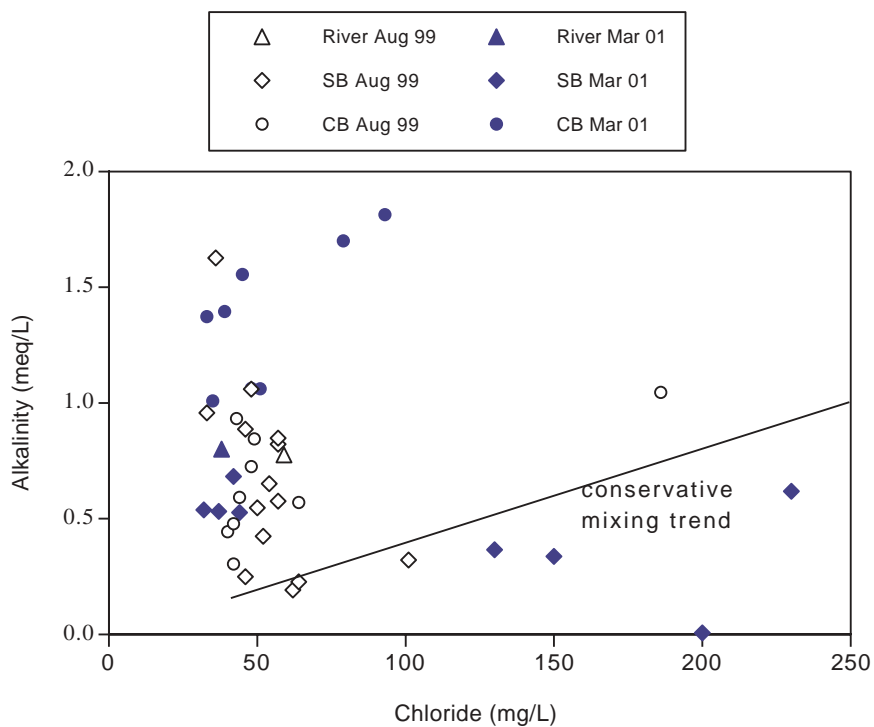
The behaviour of the major ions throughout the river water – riparian groundwater – regional groundwater continuum reflects a combination of mixing, evaporation and geochemical reactions. Groundwater in the riparian zone for both transects showed increases in concentration of chloride ion with distance away from the river which has been discussed in the section on environmental tracers. Because  $\text{Cl}^-$  concentration reflects primarily mixing and evaporation, the relationships between other major ions and chloride are often used to evaluate the extent of geochemical processes affecting the other non-conservative major element species.

Two examples are shown to highlight behaviour of a conservative element ( $\text{Na}^+$ –Fig. 14 a) and non-conservative species (alkalinity which is predominantly  $\text{HCO}_3^-$ –Fig. 14 b). Sodium,  $\text{Mg}^{2+}$  and  $\text{K}^+$  tend to exhibit linear relationships with  $\text{Cl}^-$  and thus are primarily controlled by the same evaporation and mixing processes that control  $\text{Cl}^-$  at both sites. Those groundwater samples that plot to the upper right of Fig. 14 a represent samples furthest from the river at the Sandbank and Claybank sites.

The behaviour of alkalinity (Fig. 14 b) is considered to represent the opposite end of the spectrum in the sense that it reflects geochemical processes occurring in groundwater in the near stream environment. There appears to be quasi-conservative behaviour at  $\text{Cl}^-$  concentrations  $>60 \text{ mg L}^{-1}$  which represent groundwater  $>20\text{m}$  from the surface water. There is considerable scatter of alkalinity concentrations, which vary over a factor of 5, in the  $30 - 60 \text{ mg L}^{-1}$  chloride concentration range. Alkalinity in the near stream zone is likely to be affected by 1) redox processes and 2) carbonate – mineral solution reactions. Alkalinity will be generated by the reduction of chemical species such as  $\text{Fe}^{3+}$  and  $\text{SO}_4^{2-}$ , which are common redox processes in anaerobic groundwater. The production of  $\text{CO}_2$  during organic matter decomposition and the dissolution of carbonate and silicate minerals will also affect alkalinity of riparian groundwater.

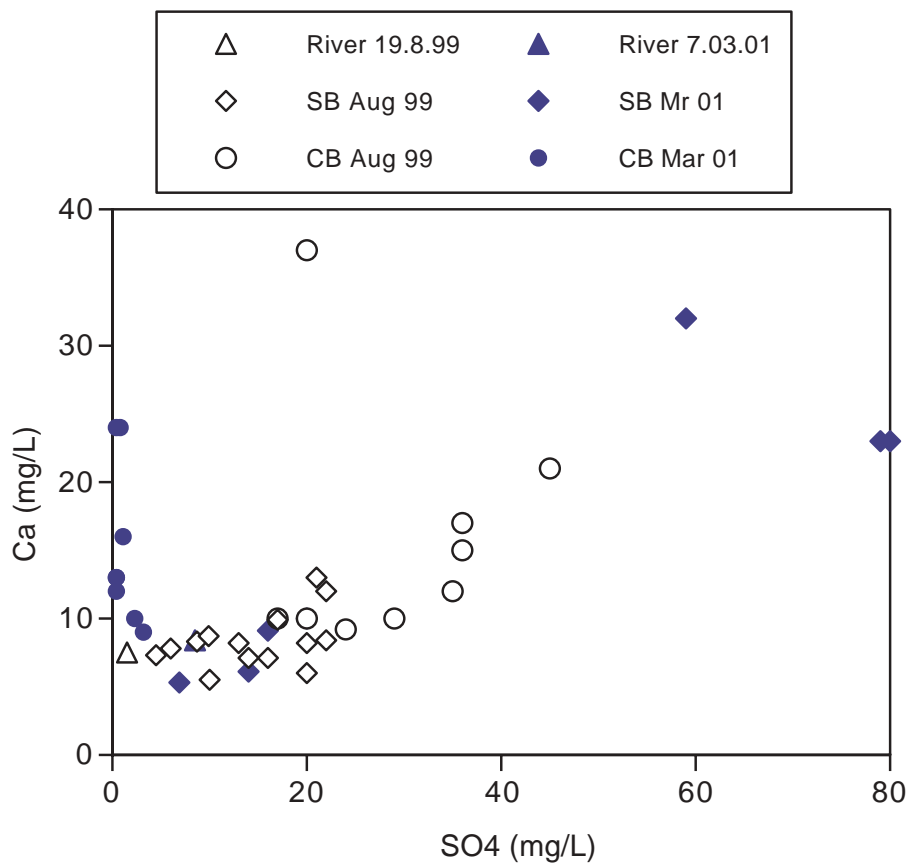


**Figure 14a**  $\text{Na}^+$  versus  $\text{Cl}^-$  concentrations for river waters and groundwaters at August 1999 (baseflow) and March 2001 (during flood recession).

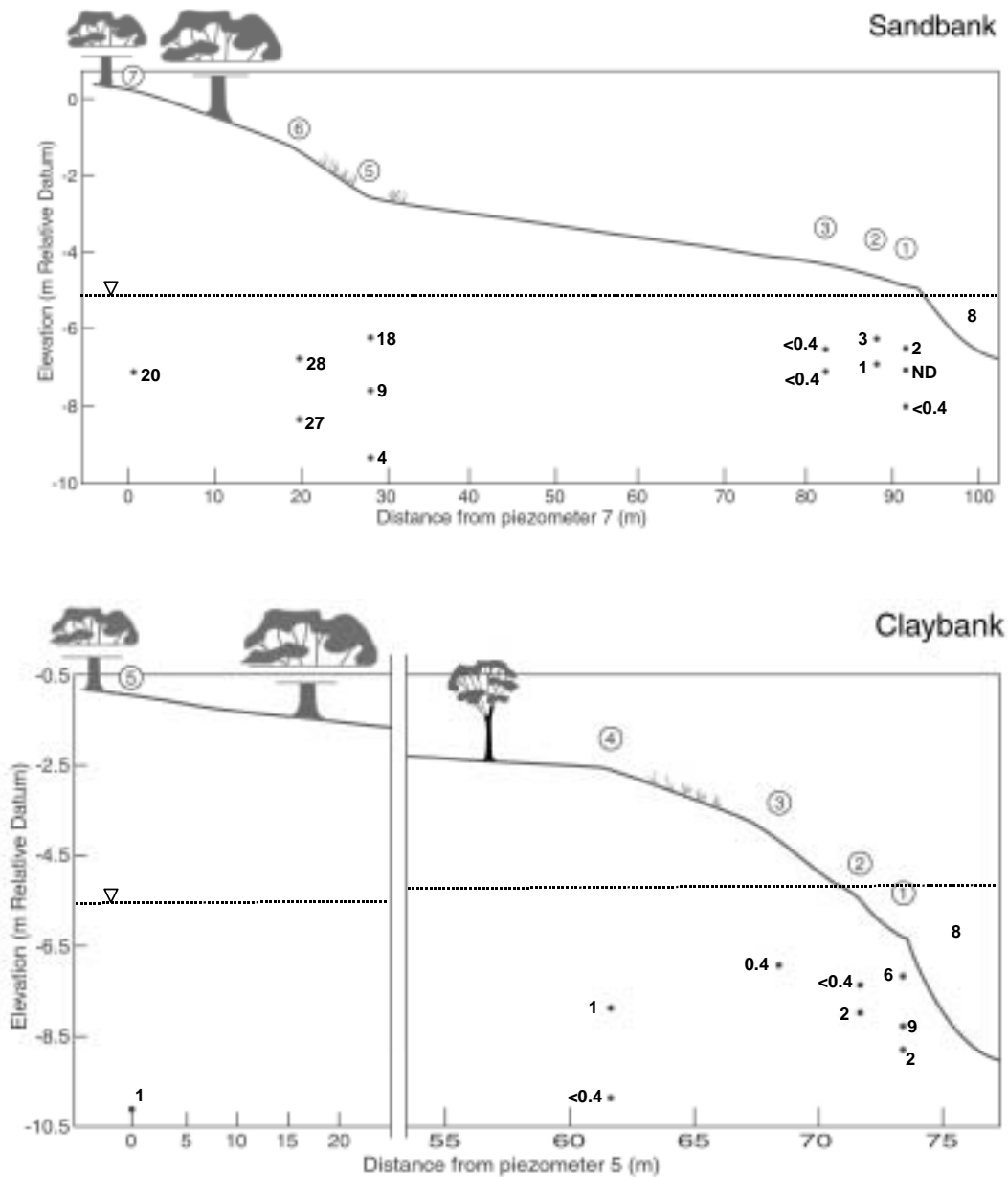


**Figure 14b** Alkalinity (as  $\text{HCO}_3^-$ ) versus  $\text{Cl}^-$  concentrations for river water and groundwater at August 1999 (baseflow) and March 2001 (during flow recession).

An interesting observation is the relationship between  $\text{SO}_4^{2-}$  and  $\text{Ca}^{2+}$  ion (Fig. 15) which shows roughly linear relationship above a  $\text{SO}_4^{2-}$  concentration of  $\sim 10 \text{ mg L}^{-1}$  with a slope very close to the molar 1:1 ratio (0.42 in mass units). This suggests dissolution of gypsum (following flooding) that had formed in the soil zone during the preceding dry (inter-flood) period. The relationship between  $\text{Ca}^{2+}$  and  $\text{SO}_4^{2-}$  in the Claybank site in March 2001 (following a flood) is completely different, displaying much lower  $\text{SO}_4^{2-}$  concentrations in comparison to  $\text{Ca}^{2+}$  concentrations. This is thought to be caused by sulfate reduction after the organic rich river waters had infiltrated into this zone. The same phenomena are not observed at the Sandbank site, presumably because of shorter residence time of water in the soil zone at this site (Fig. 16).



**Figure 15**  $\text{Ca}^{2+}$  versus  $\text{SO}_4^{2-}$  concentrations for river waters and groundwaters at August 1999 (baseflow) and March 2001 (during flood recession).



**Figure 16**  $\text{SO}_4^{2-}$  concentrations (in  $\text{mg L}^{-1}$ ) at the Claybank and Sandbank sites, 4 July 2000 (trip 6).

## Nitrogen

Nitrogen is the element that most often limits biological productivity in terrestrial and aquatic ecosystems. There was a clear pattern in the distribution of groundwater  $\text{NH}_4^+$  and  $\text{NO}_3^-$  across the riparian zone at Hattah. In general, groundwater had relatively high  $\text{NH}_4^+$  concentration ( $0.5 - 3.7 \text{ mg N L}^{-1}$ ), especially in the piezometers nearest to the river (Fig. 17). Nitrate concentration was usually low to non-detectable, with the exception of an occasional peak in shallower piezometers  $>5\text{m}$  from the river (Figs. 18 and 19). A relatively high  $\text{NH}_4^+$  concentration and low  $\text{NO}_3^-$  concentration is consistent with the generally suboxic to anoxic condition of the groundwater in the riparian zone.

It is increasingly recognised that dissolved organic nitrogen (DON) can be an important source of dissolved N in aquatic ecosystems (Harris 2001). The concentration of DON in riparian groundwater was not measured at Hattah. Based on DOC measurements, and assuming a C:N ratio of  $\sim 15$  for dissolved organic matter, riparian groundwater DON concentration would range between  $0.2$  to  $0.6 \text{ mg N L}^{-1}$  at Hattah. Thus, DON may be a significant component of the N pool in riparian groundwater. A more detailed comparison of DON and mineral N concentration in riparian groundwater will be presented in the Wollombi Brook companion report

**Table 2** Range in concentration for selected chemical constituents in river water, riparian groundwater, and regional groundwater at and near Hattah-Kulkyne Park. All units as  $\text{mg L}^{-1}$ .

	River	Riparian groundwater	Regional groundwater
DO <sup>a</sup>	7.4 – 11	<0.01 – 1.8	–
pH	6.8 – 8.1	5.6 – 7.5	–
Cl <sup>-</sup>	30 – 67	29 – 377	1500 – 20000
SO <sub>4</sub> <sup>2-</sup>	1.5 – 8.6	<0.4 – 82	120 – 4100
NH <sub>4</sub> <sup>+</sup>	<0.02 – 0.16	<0.02 – 3.7	0.35 – 2.8
NO <sub>3</sub> <sup>-</sup>	<0.02	<0.02 – 9.9	<0.005 – 0.49
FRP <sup>b</sup>	<0.005	<0.002 – 0.31	0.06 – 1
TDP <sup>c</sup>	<0.005	0.009 – 0.68	0.038 – 1.4
DOC <sup>d</sup>	5	2.4 – 9.2	1.5 – 6.5
TDFe <sup>e</sup>	<0.1 – 0.3	<0.1 – 20	<0.1 – 2.6
SrSi <sup>f</sup>	0.2 – 0.4	0.5 – 17	5.2 – 12

<sup>a</sup>Dissolved oxygen

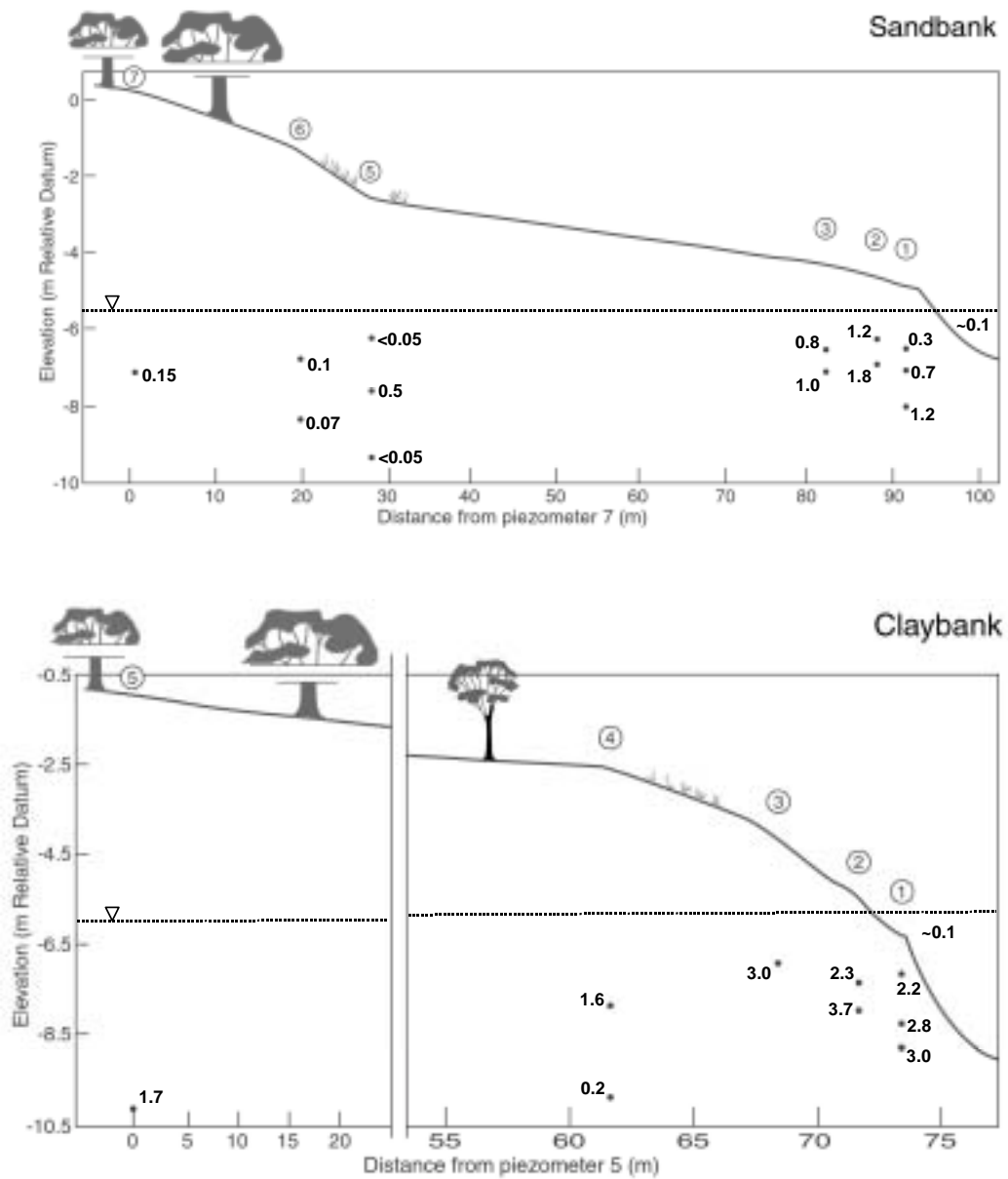
<sup>b</sup>Filterable reactive P

<sup>c</sup>Total dissolved P

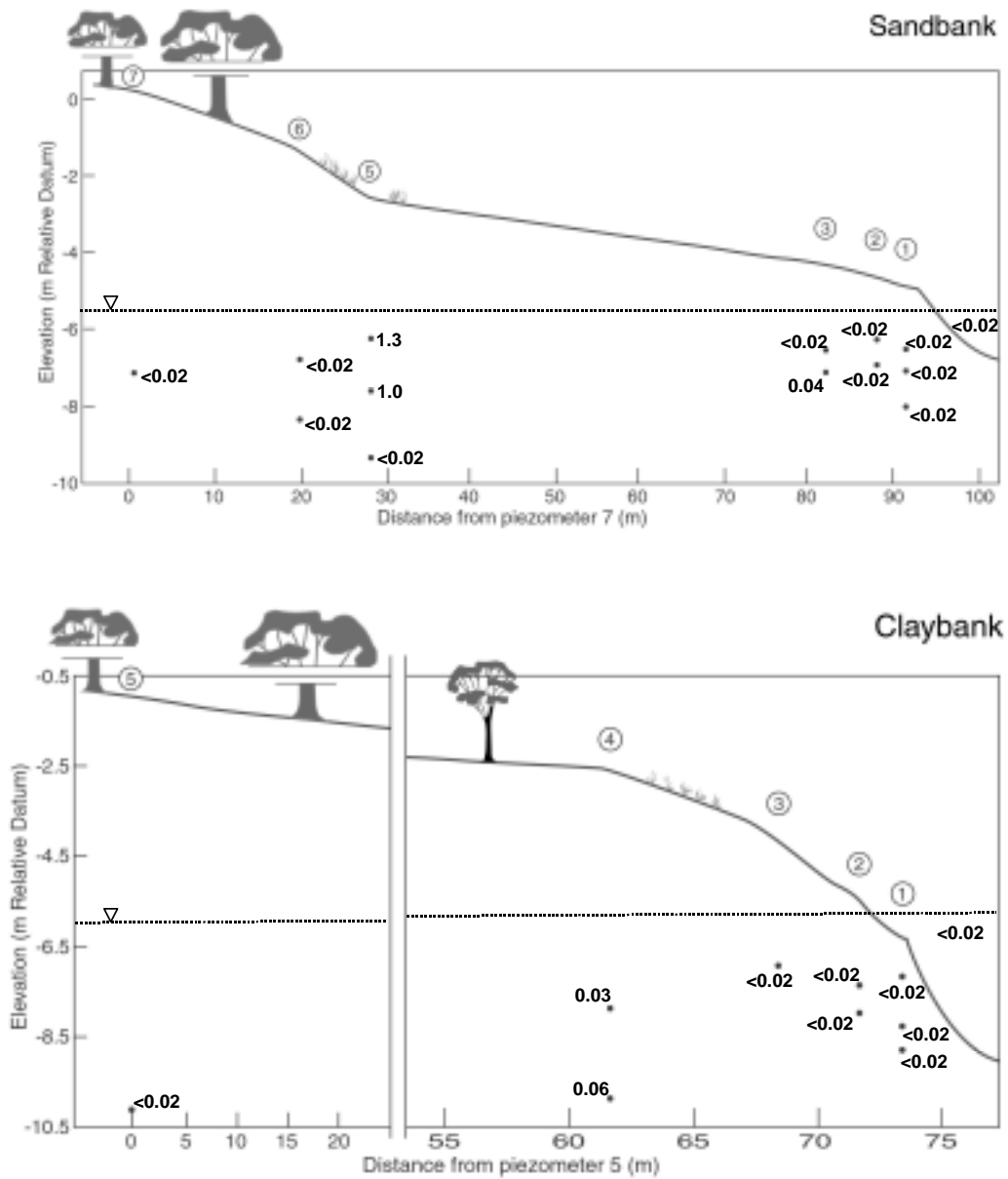
<sup>d</sup>Dissolved organic C

<sup>e</sup>Total dissolved Fe

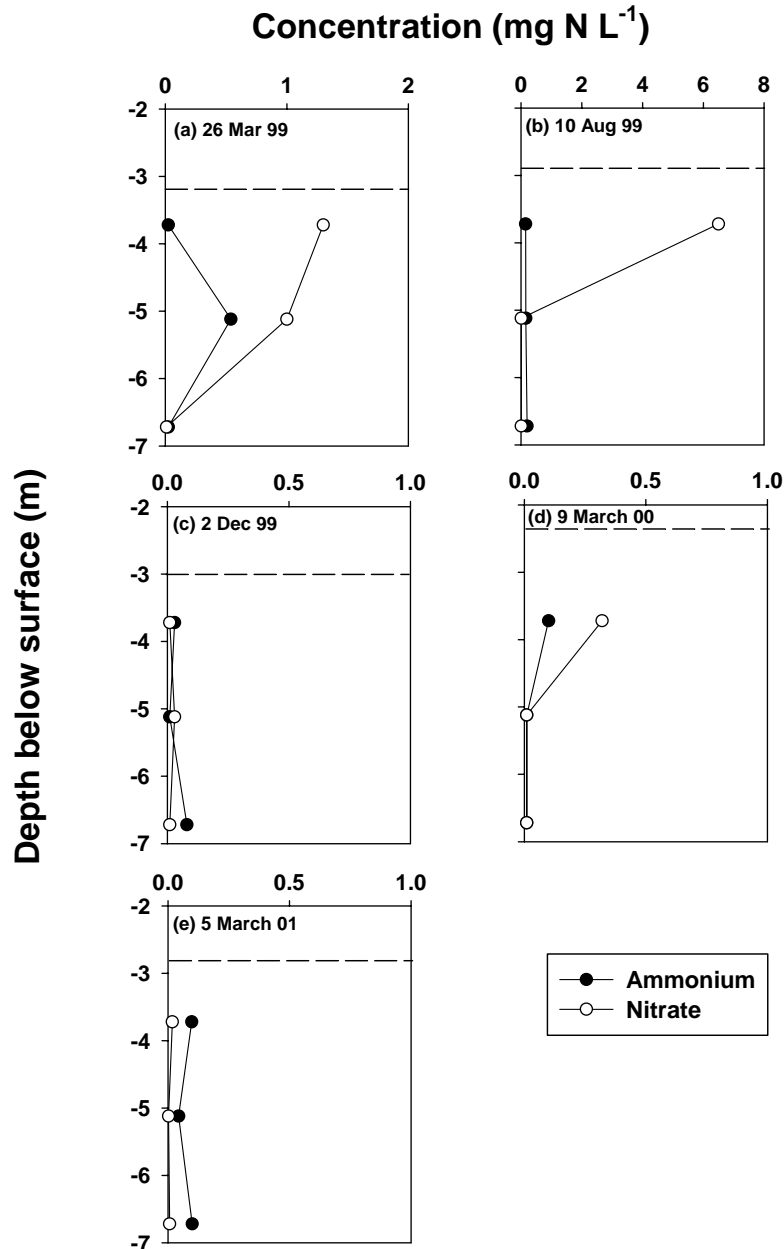
<sup>f</sup>Soluble reactive Si.



**Figure 17** NH<sub>4</sub><sup>+</sup> concentration (in mg N L<sup>-1</sup>) at the Sandbank and Claybank transects on 26 March 1999.



**Figure 18** NO<sub>3</sub><sup>-</sup> concentration (in mg N L<sup>-1</sup>) at the Sandbank and Claybank transects on 26 March 1999.



**Figure 19** Temporal trends in  $\text{NH}_4^+$  and  $\text{NO}_3^-$  concentration profiles at piezometer nest SB5. The dashed lines represent the water table.

There were some significant temporal trends in  $\text{NH}_4^+$  and  $\text{NO}_3^-$  concentration through time, especially in piezometer nests at distance from the river (Fig. 19). A peak in  $\text{NO}_3^-$  was occasionally found in shallow groundwater but never at depth (i.e., >3 m below the water table). This suggests that while most of the riparian groundwater is anoxic, an oxygenated fringe may occur at the interface with the unsaturated zone. Using detailed mini-piezometer profiles, we have identified oxygenated fringes at the water table spanning depth intervals ranging from centimeters to tens of centimeters in similar anoxic alluvial groundwater at Wollombi Brook (*see companion report*).

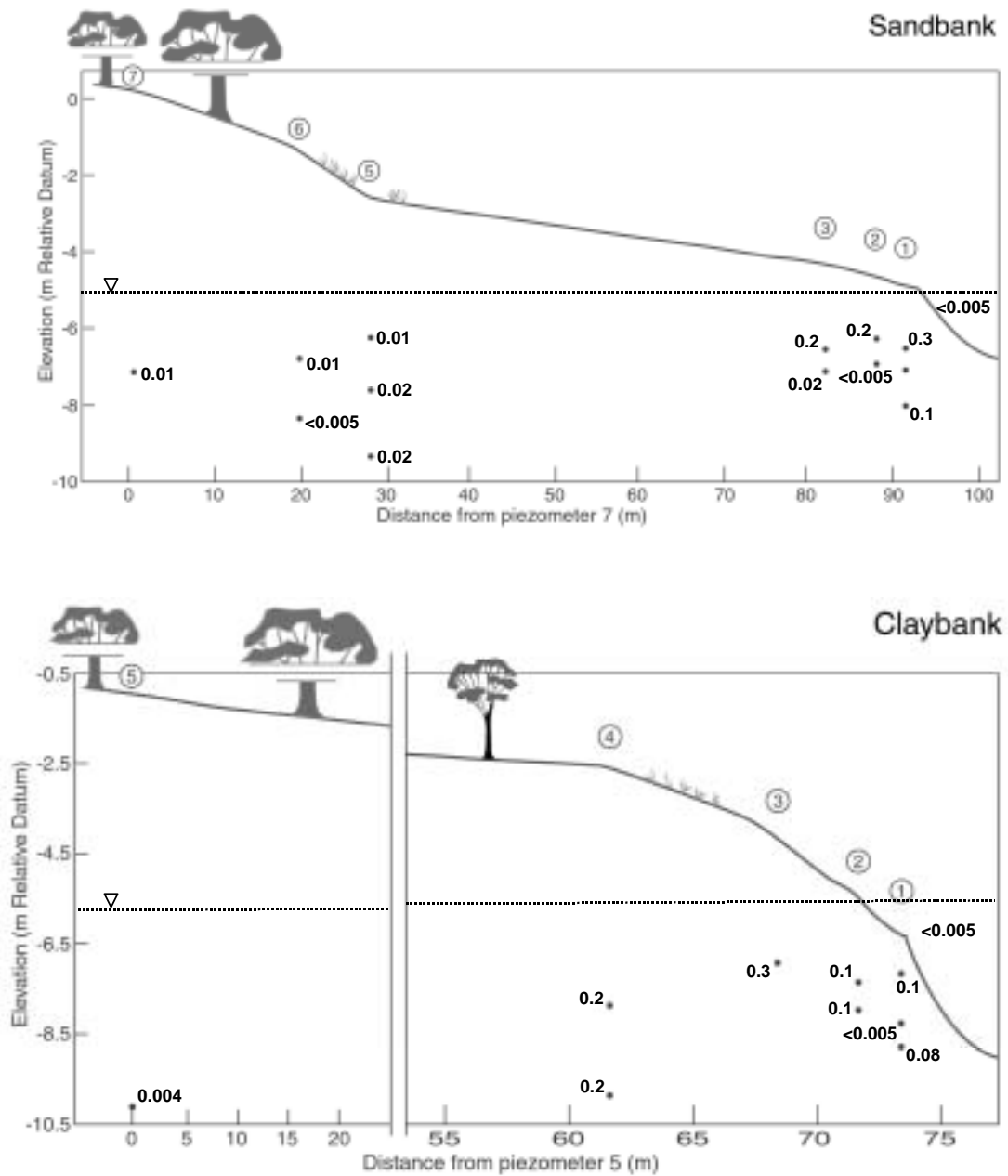
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Nitrate is usually rapidly consumed by micro-organisms once groundwater becomes suboxic. Both nitrification of groundwater  $\text{NH}_4^+$  along the oxygenated fringe and recharge of  $\text{NO}_3^-$ -rich soil water (*see below*) could be sources of  $\text{NO}_3^-$  for groundwater. The presence of sharp redox gradients, fluctuating water tables, and complex unsaturated – saturated zone processes suggest that the riparian environment has a very active N cycle.

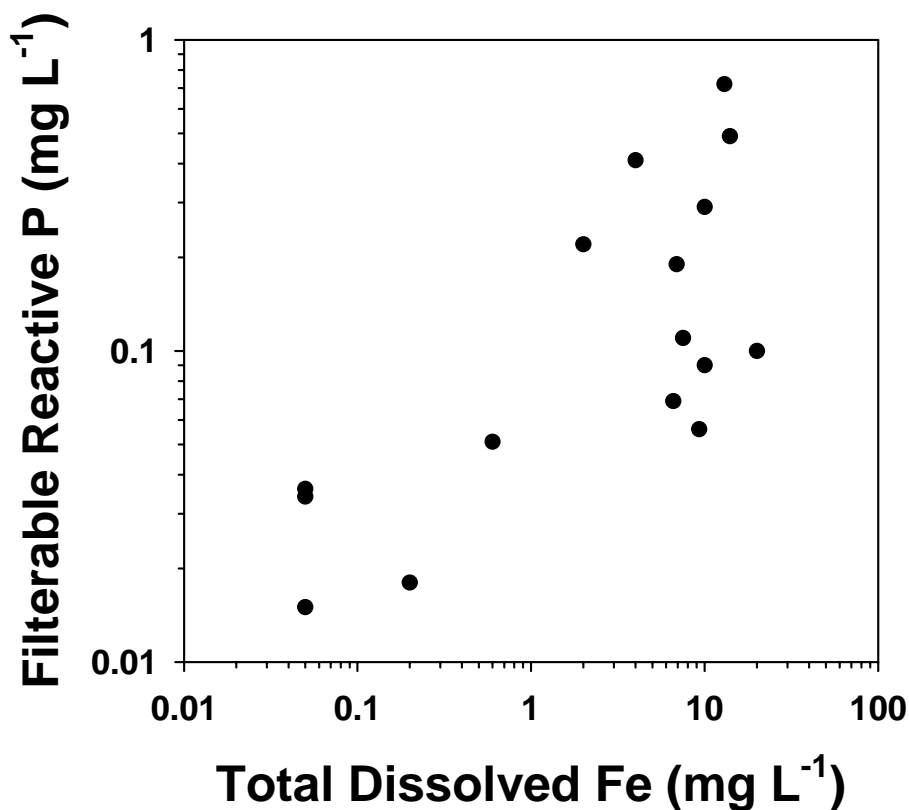
## Phosphorus

Phosphorus is an important limiting element for algal growth in freshwater. The total dissolved phosphorus (TDP) and filterable reactive phosphorus (FRP) concentration in riparian groundwater was variable but generally substantially higher than in river water (Table 2; Shown for trip 5 in Fig. 20). Most of the P in riparian groundwater appears to be in the FRP fraction (Table 2). Filterable reactive phosphorus measures the concentration of phosphate (the most biologically-available form of P) and also an unknown proportion of the colloidal-size organic P that passes through standard 0.45  $\mu\text{m}$  filters. Thus, an important fraction of the dissolved P found in riparian groundwater would be easily available to plants and micro-organisms.

There was a weak relationship between the total dissolved Fe concentration and FRP concentration in riparian groundwater (Fig. 21). This suggests that P concentration in riparian groundwater is partially controlled by sorption on Fe-oxides. Iron oxides and oxyhydroxides are thought to play an important role in the P cycle in freshwater sediments by adsorbing  $\text{PO}_4^{3-}$  on their surfaces (Baldwin and Mitchell 2000). However, under anoxia, Fe(III) oxides will be used by micro-organisms as an electron acceptor in lieu of oxygen (Roden and Edmunds 1997). The reduction of Fe(III) to Fe(II) will release the  $\text{PO}_4^{3-}$  adsorbed on the oxide surfaces. Part of the released  $\text{PO}_4^{3-}$  may form poorly soluble complex with  $\text{Fe}^{2+}$  (Baldwin and Mitchell 2000). However, if  $\text{SO}_4^{2-}$  reduction is also occurring, the formation of Fe-P complexes may be inhibited by the formation of insoluble Fe-S minerals (Caraco *et al.* 1989). High P concentration in riparian groundwater is consistent with the prevailing anoxic conditions in this environment.



**Figure 20** Filterable reactive P concentration (in  $\text{mg L}^{-1}$ ) at the Sandbank and Claybank transects on 8 March 2000.



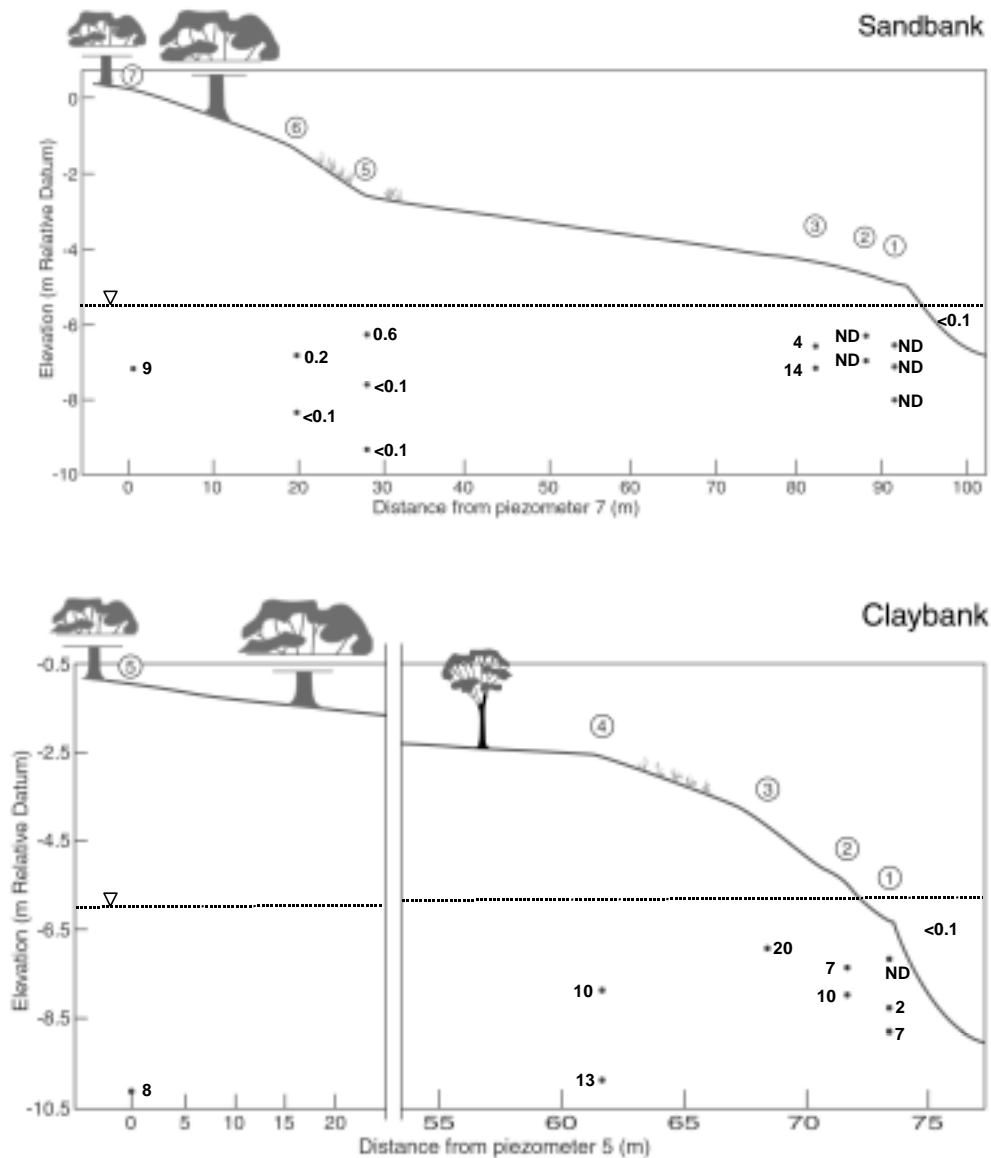
**Figure 21 Comparison of total dissolved Fe and filterable reactive P in riparian groundwater in March 2001.**

## DOC

Dissolved organic carbon (DOC) provides an important supply of energy for biogeochemical reactions in groundwater (Lamontagne *et al.* 2001). DOC concentration was measured on one occasion in riparian groundwater (7 March 2001; Appendix 2). Concentrations were generally similar to the ones found in river water (2.4 to 9.0 mg C L<sup>-1</sup>). No clear pattern emerged relative to distance from the river or depth below the water table. DOC measurements by themselves are of limited use to evaluate the availability of organic carbon for microbial processes. Organic C occurs as wide diversity of compounds in groundwater, each with different biological availability. Some sources of organic matter will be very labile while others will be refractory to microbial decomposition. More detailed spatial and temporal information will be required to further evaluate the cycle of DOC in riparian groundwater.

## Iron

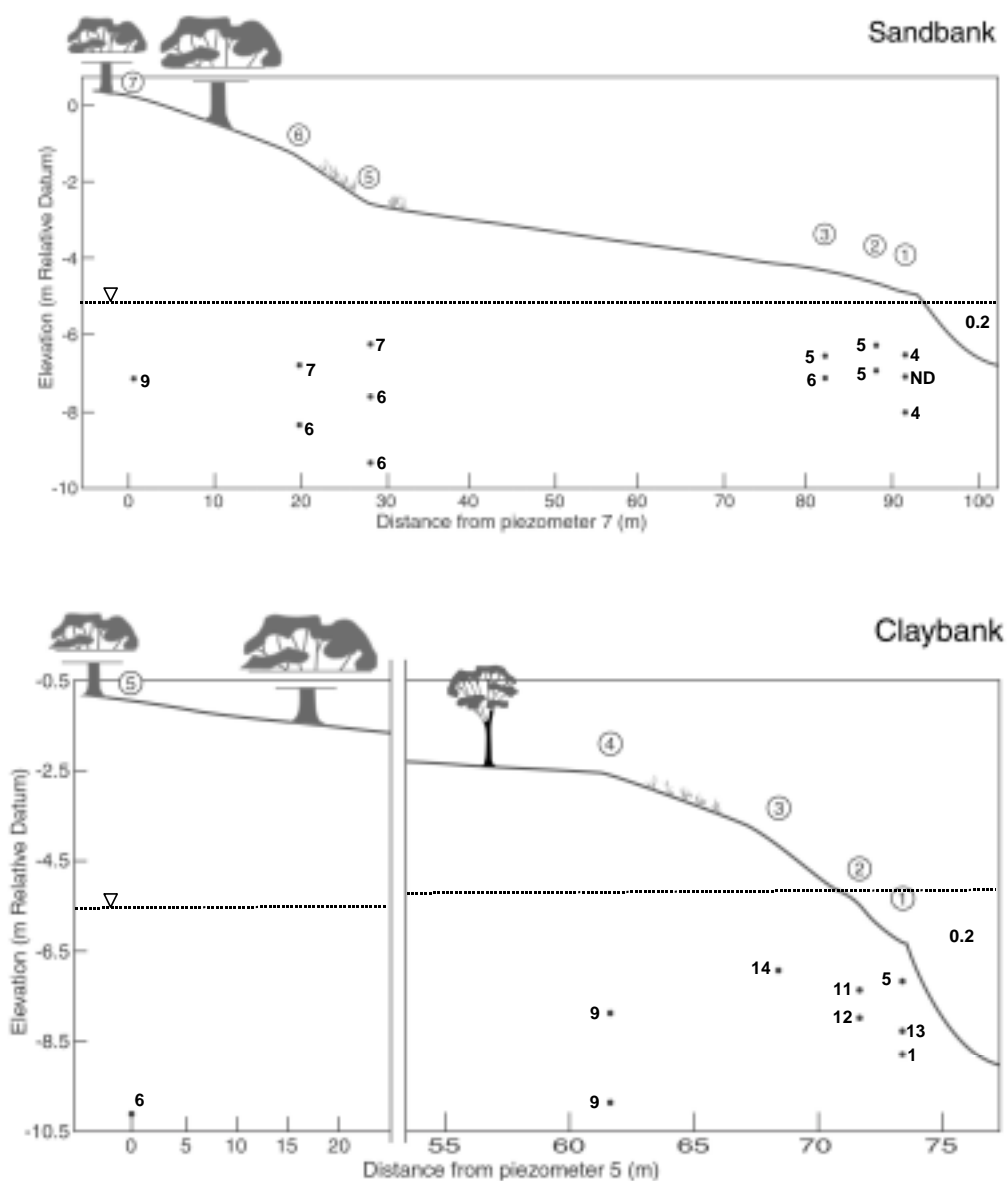
Iron is also an intrinsic component of the phosphorus and sulfur cycle in freshwater sediments (Baldwin and Mitchell 2000). Total dissolved Fe (TDFe) concentration in riparian groundwater at Hattah was variable but generally much higher than in river water (shown for trip 9; Fig. 22). Most of the TDFe in the groundwater was probably as  $\text{Fe}^{2+}$  originating from the reduction of iron oxides and oxyhydroxides. High TDFe concentrations in the riparian groundwater are consistent with the prevailing suboxic and anoxic conditions in this environment.



**Figure 22** Total dissolved Fe concentration (in  $\text{mg L}^{-1}$ ) at the Hattah transect on 7 March 2001.

## Silica

Silica is an important nutrient for algal species such as diatoms (Wetzel 1983). Riparian groundwater was systematically enriched in Si (as soluble reactive Si – SrSi) relative to river water (shown for trip 4-Fig. 23). While SrSi concentration in river water was  $\sim 0.2 \text{ mg L}^{-1}$ , concentration in riparian groundwater ranged from 0.3 to  $13 \text{ mg L}^{-1}$ . At the SB transect, there was a tendency towards a gradual enrichment in SrSi with distance from the river. Higher concentration of SrSi in groundwater probably originates from the slow dissolution of silicate minerals (Appelo and Postma 1993). As silicate mineral dissolution can be a kinetically slow process, increases in SrSi along a riparian transect may be indicative of increasing residence time in the subsurface.



**Figure 23** Soluble reactive silica concentration ( $\text{mg L}^{-1}$ ) in riparian groundwater at Hattah, 2 December 1999.

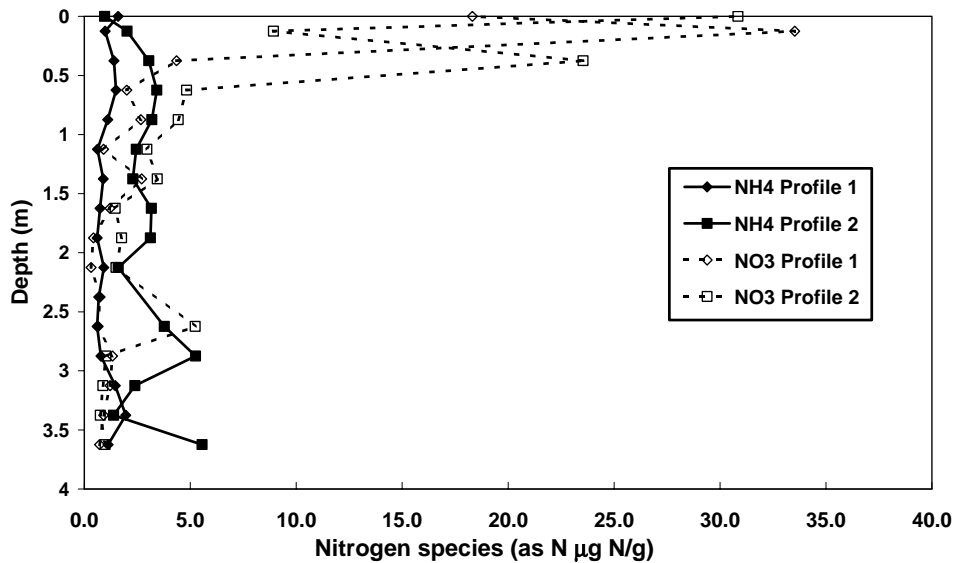
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## ***Nitrate and ammonium concentration in the unsaturated zone***

The patterns in the distribution of  $\text{NH}_4^+$  and  $\text{NO}_3^-$  in the unsaturated zones have some similarities and some differences relative to  $\text{Cl}^-$ . Nitrate concentration patterns in the unsaturated zone were similar to  $\text{Cl}^-$  but with a sharper peak near the soil surface. Nitrate concentration was less than  $1 \text{ mg N L}^{-1}$  near the water table but peaked between  $25$  and  $35 \text{ mg N L}^{-1}$  in the top  $50 \text{ cm}$  of the soil profile (Figs. 24 and 25). The second peak found at  $2.5\text{-m}$  depth for  $\text{Cl}^-$  was also found for  $\text{NO}_3^-$  in at least one of the profiles (Fig. 25). The concentration of soil exchangeable  $\text{NH}_4^+$  was more constant through the soil profile than for  $\text{NO}_3^-$ , ranging between  $1$  to  $5 \text{ } \mu\text{g N g}^{-1}$  soil and with no pronounced trend with depth (Fig. 24).

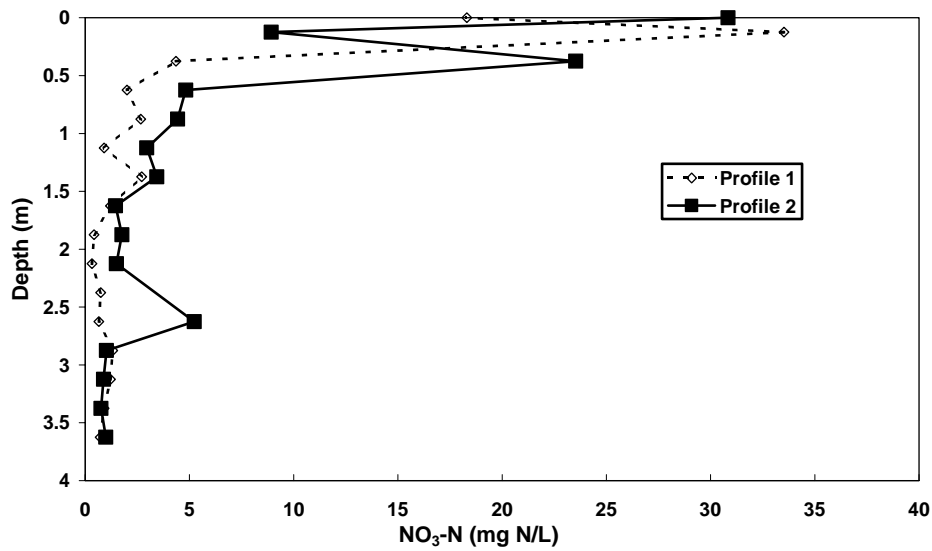
It is pointed out here that the salt extraction method will recover N species in both the pore water and on the exchange sites of the soil. Like  $\text{Cl}^-$ , almost all of the  $\text{NO}_3^-$  recovered with this method was probably in pore water. In contrast,  $\text{NH}_4^+$  was probably mostly associated with soil exchange sites. Thus, it should be expected that  $\text{NO}_3^-$  would be more mobile than  $\text{NH}_4^+$  through the unsaturated zone and behave more similarly to  $\text{Cl}^-$ .

The ratio of  $\text{NO}_3^-$  to  $\text{Cl}^-$  concentration ( $\text{NO}_3^-:\text{Cl}^-$ ) can be used to identify zones where  $\text{NO}_3^-$  could be produced or consumed independently (as a result of the changes in concentration caused by evapotranspiration). In general,  $\text{NO}_3^-:\text{Cl}^-$  was highest near the soil surface and tended to decline with depth. A high ratio near the surface is consistent with higher rates of nitrification in warmer, well-oxygenated parts of the soil profile. There was a strong decline in  $\text{NO}_3^-:\text{Cl}^-$  at  $2.0 - 2.5 \text{ m}$  depth in Profile 1 that did not occur in Profile 2 (Fig. 26, note that the  $\text{NO}_3^-:\text{Cl}^-$  data is presented on a log scale). A different origin of soil water at that depth between the two profiles or a greater consumption of  $\text{NO}_3^-$  in Profile 1 could account for this difference. Overall, these patterns suggest a net production of  $\text{NO}_3^-$  in the upper part of the soil profile and either a lower net production or a net consumption of  $\text{NO}_3^-$  closer to the water table.

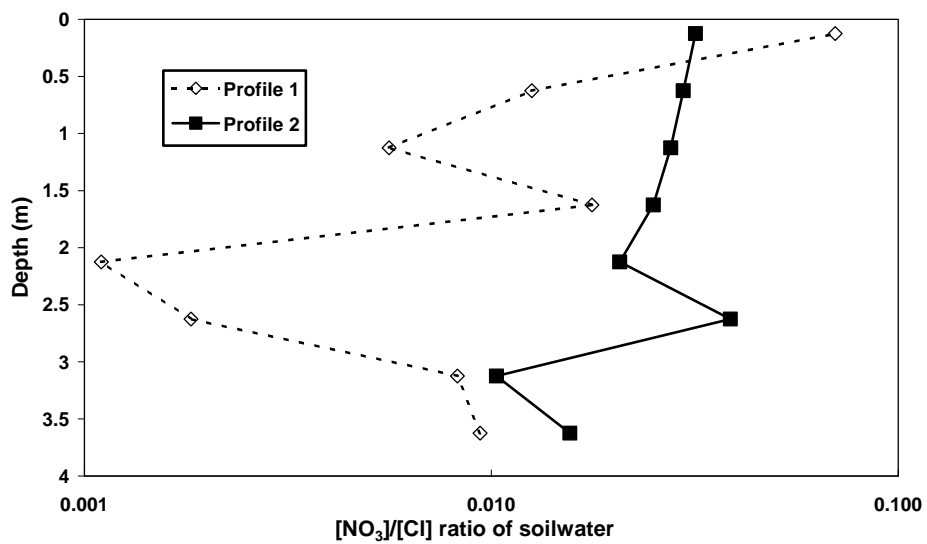


**Figure 24** Soil-exchangeable  $\text{NH}_4^+$  and  $\text{NO}_3^-$  concentration in the unsaturated zone between piezometer nest SB5 and SB7, March 2001.

A nitrate mass-balance for the unsaturated zone before and after the late 2000 flood can be attempted using similar assumptions as was used for the  $\text{Cl}^-$  budget. The total amount of chloride in the unsaturated zone following the flood (averaged for two profiles) is  $\sim 180 \text{ g m}^{-2}$  ( $0.9 \text{ m} \times 200 \text{ g m}^{-3}$ ). In comparison, the total amount of nitrate in the unsaturated zone is  $1.8 \text{ g m}^{-2}$  ( $0.9 \text{ m} \times 2 \text{ g m}^{-3}$ ). As discussed earlier for sites SB5 – SB7, the groundwater chloride concentration following the flood is consistent with soilwater from the unsaturated zone being flushed via piston flow to the depth of sampling of the shallow piezometers. If this were also true for nitrate, we would expect nitrate concentrations in the shallow groundwater at these sites to be  $\sim 2 \text{ mgL}^{-1}$ , approximately that in the soilwater. However, the post flood  $\text{NO}_3^-$  concentration measured at sites SB5 – SB7 is  $< 0.06 \text{ mg L}^{-1}$ . This suggests that most of the soil water  $\text{NO}_3^-$  displaced to the groundwater during the flood was rapidly consumed. Denitrification, dissimilatory reduction to  $\text{NH}_4^+$  and microbial uptake could all account for the rapid consumption of  $\text{NO}_3^-$  in the groundwater (Lamontagne *et al.* 2001).



**Figure 25** Soilwater NO<sub>3</sub><sup>-</sup> concentration (in mg per liter of soil water) for the unsaturated zone between piezometer nest SB5 and SB7, March 2001.



**Figure 26** Ratio of NO<sub>3</sub><sup>-</sup> to Cl<sup>-</sup> in the unsaturated zone between piezometer nests SB5 and SB7, March 2001.

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## Discussion

### Significance of regional groundwater flow to nutrient input to Australian semiarid rivers

The widespread  $\text{NO}_3^-$  contamination of groundwater in agricultural areas of North America and Europe has generated a strong interest in the potential for riparian zones to attenuate pollutants input from groundwater (Grimm and Fischer 1984; Hill 1996; Lamontagne *et al.* 2001). Riparian zones are located at the boundary between groundwater and surface water and provide active biogeochemical environments where contaminant attenuation may take place before groundwater discharges to surface water ecosystems. However, it is becoming clear that the findings from headwater catchments in temperate climates (where most of the research has taken place) may not be applicable to lowland rivers (Woessner 2000) and to different climates (Martí *et al.* 2000). The findings from two years of investigations at the Hattah-Kulkyne site suggest that the dynamics between riparian zones and rivers is markedly different in semiarid lowland rivers at several levels.

The River Murray is typical of semiarid rivers where most of the water supply is obtained from the wetter upstream section of the basin (Walker 1992). However, while only representing a few percent of the flow, regional groundwater still represents a significant input of solutes in the lowland sections of the river (Jolly 1996). Groundwater salinity in the western portion of the Murray Basin ranges from  $<7\,000$  to  $>35\,000\text{ mg L}^{-1}$ , at least two orders of magnitude greater than that in river water (Jolly 1996). However, while groundwater in semi-arid regions occasionally has high concentrations of nutrients such as  $\text{NO}_3^-$  (i.e.,  $>10\text{ mg N L}^{-1}$ ; Lawrence 1983), these levels of nutrient enrichment are still very small relative conservative solutes (Table 2). Thus, while regional groundwater can be a significant source of salts to semiarid lowland rivers, it is unlikely to be a significant source of nutrients.

### Does groundwater play any role in the biogeochemical cycles of lowland semiarid rivers?

While input from *regional* groundwater probably plays a minor role in the nutrient budget of semiarid rivers, other groundwater – surface water interaction processes may have a significant impact on nutrient cycles in the river – floodplain system. These would include bank recharge – discharge cycles during floods, density stratification, and hyporheic processes.

#### Bank recharge- discharge

The flood-pulse concept is often used as a conceptual model of organic matter and nutrient exchange in lowland rivers (Junk *et al.* 1989). Following this model, rivers tend to be disconnected from their floodplain under baseflow conditions, limiting exchanges between the two systems. The exchange of organic matter and nutrients would occur as “pulses” during the periods when the systems are connected by

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elevated water levels. This lateral connection between the river and the floodplain is severed once water levels recede.

Bank recharge-discharge processes could provide an intermediate link between the two stages of the flood-pulse cycle. River water recharged or infiltrated along stream banks during floods will tend to slowly seep back towards the stream once flows recede. This was observed at the Hattah site, where positive gradients towards the river only occurred for a few months following large floods. The significance of bank discharge for nutrient fluxes to rivers is not known at present. However, at the Chowilla site, the combination of bank discharge and point recharge in the floodplain was thought to increase groundwater and salt input to the River Murray for up to 18 months after a very large flood (Jolly and Walker 1995). Thus, a part of the nutrient pool present in riparian soil water and groundwater should be discharged to rivers during and following flooding events. The significance of bank recharge-discharge cycles on the nutrient input to surface water is not known at present but may contribute to the peak in biological productivity commonly observed following flood events in semiarid rivers.

### **Density stratification**

Groundwater may be a significant component of biogeochemical cycles in lowland rivers by modifying the physical environment of rivers at low flow. Saline groundwater intrusions commonly form density stratification in the deeper pools of semiarid rivers in Australia (Anderson and Morrison 1989; Bailey and James 1999). These saline pools are frequently anoxic and can be stable until dispersed by high flow events (Anderson and Morrison 1989). The development of density stratification by saline intrusions may cause a similar effect on riverine biogeochemical cycles as thermal stratification in weir pools (Bormans and Webster 1997; Sherman *et al.* 1998). In general, stratification should favour increased P and  $\text{NH}_4^+$  recycling from sediments but may promote denitrification. In turn, these conditions (high P availability with low N availability) may be favourable to the development of noxious N-fixing blue-green algal blooms. In the River Murray, saline groundwater intrusions tend to develop in wetlands and side channels in the floodplain rather than in the river channel (G. Walker, CSIRO Land & Water, *personal communication*). However, stable saline pools are a common feature of smaller semiarid zone rivers such as the Wimmera in Victoria (Anderson and Morrison 1989). It is likely that saline intrusions in rivers and streams will become more frequent as a result of dryland agriculture and irrigation agriculture in many parts of Australia. The impact of saline groundwater intrusions on riverine biogeochemical cycles has not been assessed in detail and requires further investigation (Bailey and James 1999).

### **Hyporheic processes**

Hyporheic processes include numerous small exchanges of water between streams and sediments that are induced by irregularities along the stream bottom. The extent of this exchange is a function of the type of substrate (cobbles, sand, mud, etc), the shape and density of irregularities on the stream bottom (bedforms, snags, etc), and flow velocity (Hutchinson and Webster 1998; Jones and Mullholland 2000). On the one hand, surface water infiltration adds particulate organic matter and oxygen to the subsurface (Huettel *et al.* 1996). On the other hand, hyporheic water exfiltration

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exports some of the recycled nutrients back to the surface (Duff and Triska 2000). Although small in scale (centimeters to meters), hyporheic processes are likely to be a significant groundwater-surface water process for the cycle of nutrients in lowland rivers. A more detailed discussion on the significance of hyporheic processes in Australian semiarid rivers will be presented in a companion report describing our work at the Wollombi site in NSW.

## **Biogeochemical cycles in semiarid floodplains**

Groundwater discharge through floodplains can result in large increases in salt concentration in both soil water and shallow groundwater (Jolly and Walker 1995). Yet, this increase in salt concentration is generally not matched by similar increases in nutrient concentration in groundwater. Clearly, processes other than evapotranspiration are occurring to modify nutrient concentration in floodplain groundwater. To review biogeochemical cycles at the groundwater – surface water interface in floodplains is beyond the scope of this report. In general, it is agreed that floodplains and wetlands are intensive sites for nutrient transformations because groundwater is exposed to a variety of redox environments at relatively small spatial and temporal scales (Belya 1996). A shallow and fluctuating water table is an example of a property of floodplains and wetlands conducive to extensive nutrient cycling and transformations (Baldwin and Mitchell 2000). In general, it is not clear at the present if and which nutrients limit primary production in Australian semi-arid floodplains. However, recent findings suggest that floodplains are self-sufficient for P and possibly limited for N (Ogden and Thoms 2001). These findings are consistent with an active recycling of nutrients because P tends to be more conservative than N during “spiralling” of organic matter throughout the river-floodplain systems. For example, while N can be lost permanently from the ecosystem through various gaseous forms ( $N_2$ ,  $NH_3$ ,  $N_2O$ , etc) generated during recycling, gaseous losses of P are usually negligible.

The comparison of  $Cl^-$  and  $NO_3^-$  concentration in soil and groundwater between flood cycles at Hattah provided some clues on how N may be lost during recycling in floodplains. During inter-flood periods,  $Cl^-$  and  $NO_3^-$  accumulate near the soil surface. The source of  $Cl^-$  is probably water table evaporation. Water table evaporation is a complex process that is a function of the depth to the water table and the type of soil present. The source for  $NO_3^-$  is not so clear. Nitrate could originate from the decomposition of organic matter deposited on the floodplain during floods and from the litter produced by floodplain vegetation. In a similar manner as for  $Cl^-$ , groundwater dissolved organic matter and  $NH_4^+$  brought to the surface by water table evaporation may also be sources of N for the unsaturated zone. Part of the soil  $NH_4^+$  pool could be lost by  $NH_3$  volatilisation, especially if soil pH is neutral to alkaline (Dawson 1977). Warm temperatures and an ample supply of oxygen would also be ideal conditions for nitrification to occur near the soil surface. An accumulation of  $NO_3^-$  in shallow soils during dry periods is a common feature of many ecosystems.

During floods, part of the  $Cl^-$  and  $NO_3^-$  that had accumulated within the unsaturated zone could be flushed in surface runoff and or recharged to groundwater. The increase in  $Cl^-$  concentration in shallow groundwater at Hattah is evidence that part of the unsaturated zone  $Cl^-$  was displaced to groundwater during the late 2000 flood.

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Soil  $\text{Cl}^-$  could be incorporated in groundwater by either downward recharge of flood water or from a rise in the water table. As for  $\text{Cl}^-$ , part of the soil  $\text{NO}_3^-$  pool will also be displaced to groundwater. However, because of generally suboxic to anoxic conditions within the alluvial aquifer, most of this  $\text{NO}_3^-$  will be reduced to other forms (either denitrified to  $\text{N}_2\text{O}$  and  $\text{N}_2$ , or reduced to  $\text{NH}_4^+$ ). It can be seen that if this process is repeated frequently (through floods or important rain events) that large amounts of N could potentially be lost from the system through processes such as denitrification. The key in this case is to successively expose organic matter to oxygenated conditions (to efficiently convert organic into  $\text{NH}_4^+$  and  $\text{NO}_3^-$ ) and anoxic conditions (to foster denitrification). The cycle of N at the saturated zone – unsaturated zone interface at Hattah is consistent with the tendency of semiarid floodplains to be N rather than P-limited.

The prevalence of anoxia in riparian groundwater suggests that significant sources of labile particulate or dissolved organic matter are present. While other reduced compounds can be used in biogeochemical reactions in groundwater (Lamontagne *et al.* 2001), organic carbon is usually the main driver for nutrient cycles in riparian zones. Several external sources of organic matter could contribute to the organic carbon input to riparian groundwater. Particulate organic matter can be deposited on the floodplain or buried in the subsurface during floods (Boulton and Foster 1998; Robertson *et al.* 1999). In addition, large amounts of litter are deposited on the forest floor by the riparian vegetation during inter-flood periods (Robertson *et al.* 1999). The decomposition of organic matter on the forest floor is an important source of DOC (Baldwin and Mitchell 2000). Part of the soil DOC pool could be recharged to groundwater during floods or following large rain events. Hyporheic processes (Huettel *et al.* 1996) and bank recharge would also result in the input of fine particulate organic matter and DOC to riparian groundwater. Fine root turnover is an *in-situ* source of organic matter to both the unsaturated and saturated zones of the floodplain. Below-ground production of organic matter is often greater than above-ground production in forests (Vogt *et al.* 1986) and fine root turnover could represent a large source of carbon to the subsurface.

Studies of biogeochemical cycles in river – floodplain systems have primarily focussed on processes occurring at the sediment-water interface of rivers and billabongs (Baldwin and Mitchell 2000). The studies at Hattah suggest that larger scale groundwater – surface water processes may also be significant for the cycle of nutrients and probably carbon. Because of the prevailing reducing conditions found in the subsurface, shallow groundwater could be a significant source for the production of greenhouse gases ( $\text{N}_2\text{O}$ ,  $\text{CO}_2$ ,  $\text{CH}_4$ , etc) in semiarid floodplains (Sorrel and Boon 1992). In a process similar to rapid destratification of reservoirs (Sherman *et al.* 2001), discharge of riparian groundwater during floods may be a significant source of greenhouse gases to the atmosphere.

## **Limitations of piezometer networks as research tools in riparian environments**

Several problems were encountered with the use of standard piezometer networks at the Hattah site. These included issues about the chemical integrity of groundwater samples, small hydraulic gradients at the scale of the riparian zone, and loss of

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equipment during floods. These are recounted here so that future studies in these environments can benefit from our experiences.

### **Integrity of the subsurface redox environment**

Several limitations should be kept in mind when interpreting nutrient concentration samples obtained from piezometers, especially in the reactive geochemical environment frequently encountered in riparian zones. In anoxic groundwater, piezometers may act as conduit to “pollute” the aquifer with atmospheric oxygen. This could drastically modify the geochemical environment in the vicinity of piezometers. This bias can be partially circumvented by flushing with several well volumes before collecting a sample. Unfortunately, this will also result in a sample that integrates over a larger aquifer volume. When a piezometer is located near a sharp redox interface, parcels of water from different redox environments and having contrasting water chemistries will be integrated. In addition, it is often not possible to flush several well volumes from piezometer that recover slowly following flushing.

Because of the sharp redox gradients common in riparian zones at a small spatial scale (centimeters to meters), several precautions should be taken to collect groundwater samples in this environment. Firstly, it may be preferable not to use the same design of piezometer for both water sample collection and water level monitoring. In general, piezometers for water level monitoring must be large enough to accommodate the installation of recording instruments. In addition, larger wells are less sensitive to changes in water levels created when instruments are added or removed from them. Artifacts in water level time series can also be generated when water samples are extracted from water level monitoring wells, especially when they recover slowly. In contrast to water level monitoring, piezometers for water chemistry should be designed to limit disturbance to the subsurface environment. In substrates with a high hydraulic conductivity (sands and gravels), nests of mini-piezometers made of nylon tubing allow the collection of concentration profiles at small spatial scales (0.1 – 1 m). Capping and the use of small diameter tubing limits the introduction of oxygen to the subsurface. Larger piezometers are still required in substrates with lower hydraulic conductivity because mini-piezometers cannot be pumped as freely as in the case of sandy substrates. To maintain their chemical integrity, larger piezometers should be “capped” with a dense inert gas such as argon between sampling events. The presence of a dense inert gas will limit the contamination of the subsurface with atmospheric oxygen. It must be noted that piezometers and wells intended for water level monitoring cannot be physically capped to prevent atmospheric oxygen contamination because equilibrium with changes in atmospheric pressure would be prevented.

The possible designs for groundwater monitoring in the riparian zone are infinite and depends on the particular goals of a given study. Nevertheless, care must be taken when chemical species with a strong redox cycle are investigated. A more detailed presentation of the use of different piezometer designs in alluvial aquifers will be presented in the companion report describing the work at the Wollombi site.

### **Hydraulic gradients**

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At the scale of riparian zones (i.e., 1 – 100 m), the use of piezometric surfaces to infer the direction of groundwater flow can be problematic because the differences in hydraulic head between piezometers can be small relative to measurement error. At small spatial scales (1 – 30 m), potentiomanometer readings should be used to complement hydraulic head measurements from piezometer networks. The rationale, design and operation of potentiomanometers has been presented elsewhere (Winter *et al.* 1988). Potentiomanometers measure the *difference* in hydraulic head between a piezometer and a reference water level (usually surface water). Potentiomanometers can detect differences in hydraulic head of a few millimeters, which is difficult to achieve with standard techniques. Whenever possible, potentiomanometer measurements should be used to complement the estimate of the hydraulic gradient obtained with piezometric surfaces.

### **Equipment loss**

The use of fixed piezometers may not be optimal in sandy riparian zones prone to frequent flooding. At the Hattah sandbank site, damage and loss of the piezometers closest to the river occurred following each flood event. One option to limit piezometer loss is to make them very short. However, this limits the potential to sample them at high water levels. Another option is to dispense with piezometers and rely on drive point systems. Drive points can be used to collect groundwater profiles up to a depth of several metres in sandy alluvial aquifers. Drive points also have the advantage that they do not leave permanent equipment on the ground and allow sampling intervals to be modified horizontally and vertically as a function of changes in water level. Details on the construction and use of drive points and mini-piezometers will be presented in the Wollombi companion report.

### **Conclusions**

The studies of groundwater interactions with the River Murray at Hattah-Kulkyne Park illustrate the complexity of groundwater surface – water interactions in lowland semiarid rivers. Our work highlights that current conceptual models of river – floodplain interactions, such as the “flood-pulse” concept, may be simplistic because they neglect hydrogeological processes. Rivers and floodplains remain linked through groundwater during baseflow conditions.

Our work has several important implications for river – floodplain interactions in the River Murray system. We have shown that regional groundwater discharge tends to occur in the floodplain rather than in the river itself at Hattah-Kulkyne. Raised river levels by locks and weirs and high rates of evapotranspiration in the riparian forest would enhance the tendency of regional groundwater discharge to occur in the floodplain rather than in the river. We have demonstrated that surface water recharge occurs by bank infiltration during baseflow and also by vertical diffuse recharge during floods. We have also shown that riparian groundwater tends to be enriched in nutrients that limit algal growth. Part of this groundwater nutrient pool could be exported to surface water by hyporheic processes at baseflow and by bank discharge following floods. Perhaps the most significant new finding is the potential for unsaturated – saturated zone processes to be significant in floodplain biogeochemical cycles. The role of unsaturated zone processes for the water balance and the cycle of

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salts in floodplains had been recognised in the past but their role in nutrient cycles had not.

Future research in groundwater – surface water interactions in semiarid floodplains should try to quantify the importance of bank discharge to nutrient loads to rivers and greenhouse gas emissions to the atmosphere. The biogeochemical processes occurring at the unsaturated – saturated zone interface also require further studies.

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## Appendix 1 Analytical Techniques

### Gravimetric Water Content

Calculated from the difference in weight of the soil sample and the sample dried at 105 °C for 24 h divided by the dry weight of soil.

### Soil-Water Chloride Concentration (as per Taras *et al.* 1975)

The amount of chloride in the soil (soil chloride ie. mg of Cl per kg of dry soil) was determined by extracting the chloride from soil samples using a 5:1 dilution with water. Soil-water chloride (mg of Cl per Litre of water) was calculated by dividing the soil chloride by the gravimetric water content.

### Analysis of Stable Isotopes

Stable isotopes of water ( $\delta^2\text{H}$  and  $\delta^{18}\text{O}$ ) were measured by standard isotope ratio mass spectrometry.  $\delta^2\text{H}$  was measured after reduction of water over hot uranium.  $\delta^{18}\text{O}$  was measured on  $\text{CO}_2$  gas equilibrated with 1 ml of water at 25 °C. Results are quoted relative to Vienna Standard Mean Ocean water (V-SMOW) with analytical uncertainties of  $\pm 1$  ‰ and  $\pm 0.15$  ‰ respectively.

### Analytical methods for nutrients and major ions

Analyte	Method	Reference
$\text{Ca}^{2+}$ , $\text{Mg}^{2+}$ , $\text{Na}^+$ , $\text{K}^+$	The samples were determined directly by ICP emission spectrometry. Any concentrations above the linear analytical range were diluted with 1% (v/v) $\text{HNO}_3$ before re-analysis.	Standard methods for the examination of water and wastewater (1999). 20 <sup>th</sup> edition American Public Health Assoc., American Water Works Assoc., Water Environment Federation. Method 3120.
$\text{Cl}^-$ , $\text{SO}_4^{2-}$	These anions were determined by ion chromatography (IC) using chemical suppression and electrical conductivity detection.	Standard methods for the examination of water and wastewater (1999). 20 <sup>th</sup> edition American Public Health Assoc., American Water Works Assoc., Water Environment Federation. Method 4110.
Soluble reactive Si	The filtered samples were analyzed by segmented flow analysis (SFA) using ammonium molybdate and oxalic acid then reduced with ascorbic acid and determined colorimetrically at	Standard methods for the examination of water and wastewater (1999). 20 <sup>th</sup> edition American Public Health Assoc., American Water Works Assoc., Water Environment Federation. Method 4500-SiO <sub>2</sub> Automated method for molybdate-reactive silica (modified). Perstorp Analytical Environmental

	815nm.	EnviroFlow 3500 method procedure document 000595 (pers.com.)
Total dissolved Fe (see Ca, Mg, Na & K)	The samples were determined directly by ICP emission spectrometry.	Standard methods for the examination of water and wastewater (1999). 20 <sup>th</sup> edition American Public Health Assoc., American Water Works Assoc., Water Environment Federation. Method 3120.
NH <sub>4</sub> <sup>+</sup>	Determined by segmented flow analysis (SFA) using the less hazardous sodium salicylate, sodium nitroferricyanide (nitroprusside) and DCIC in alkaline solution with citrate buffer.	Standard methods for the examination of water and wastewater (1999). 20 <sup>th</sup> edition American Public Health Assoc., American Water Works Assoc., Water Environment Federation. Method 4500-NH <sub>3</sub> G Automated phenate method (modified). Krom, M.D. (1980) Spectrophotometric determination of ammonia: a study of a modified Berthelot reaction using salicylate and dichloroisocyanurate. <i>The Analyst</i> , <u>105</u> , 305-316.
NO <sub>3</sub> <sup>-</sup>	Determined by segmented flow analysis (SFA). Nitrate reduced to nitrite by Cu/Cd. Total nitrite then determined colorimetrically after reaction with sulfanilamide and NEDD in acid solution.	Standard methods for the examination of water and wastewater (1999). 20 <sup>th</sup> edition American Public Health Assoc., American Water Works Assoc., Water Environment Federation. Method 4500-NO <sub>3</sub> F Automated cadmium reduction method.
Filterable reactive P	Determined by segmented flow analysis (SFA) using ammonium molybdate and potassium antimony tartrate in the presence of ascorbic acid at pH 1.0 to form a molybdenum blue colour.	Standard methods for the examination of water and wastewater (1999). 20 <sup>th</sup> edition American Public Health Assoc., American Water Works Assoc., Water Environment Federation. Method 4500-P F Automated ascorbic acid reduction method.
Total dissolved P	The samples were determined directly by ICP emission spectrometry. Lower detection limits were achieved using an ultrasonic nebulizer sample introduction system.	Standard methods for the examination of water and wastewater (1999). 20 <sup>th</sup> edition American Public Health Assoc., American Water Works Assoc., Water Environment Federation. Method 3120.

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Dissolved organic carbon	Thermal combustion of filtered solution to form CO <sub>2</sub> that is determined by IR detection. Inorganic carbon removed before analysis or determined separately.	Standard methods for the examination of water and wastewater (1999). 20 <sup>th</sup> edition American Public Health Assoc., American Water Works Assoc., Water Environment Federation. Method 5310 B High-temperature combustion method.
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## Appendix 2      *Tables of Results*

**Table A1.      Piezometer characteristics at the Hattah site.**

<b>Piezometer label</b>	<b>Distance from furthest piezometer (m)</b>	<b>Total depth below the surface (m)</b>	<b>Depth relative to arbitrary reference (m)</b>
SB site			
1A	91.9	1.50	-6.52
1B	91.9	2.04	-7.14
1C	91.9	2.90	-8.00
2A	88.4	2.00	-6.28
2B	88.4	2.64	-6.92
3A	82.3	2.34	-6.57
3B	82.3	2.93	-7.12
4	-	3.07	-
5A	28.2	3.72	-6.26
5B	28.2	5.12	-7.62
5C	28.2	6.72	-9.34
6A	19.8	5.52	-6.82
6B	19.8	7.18	-8.35
7	0	7.50	-7.17
CB site			
1A	73.500	1.280	-7.19
1B	73.500	2.410	-8.28
1C	73.500	2.950	-8.81
2A	71.800	2.070	-7.39
2B	71.800	2.720	-8.01
3	68.500	3.080	-6.94
4A	61.800	5.480	-7.87
4B	61.800	7.650	-10.0
5	0.000	9.298	-10.1

**Table A2.** Chemistry data for 26 March 1999 (Trip 1). Data in mg L<sup>-1</sup> unless otherwise noted.

Label	Field DO	Field pH	Field EC (µS/cm)	Alk. (mmol/L)	Ca <sup>2+</sup>	K <sup>+</sup>	Mg <sup>2+</sup>	Na <sup>+</sup>	TDFe	Cl <sup>-</sup>	SO <sub>4</sub> <sup>2-</sup>	TDP	FRP	NH <sub>4</sub> <sup>+</sup> -N	NO <sub>3</sub> <sup>-</sup> -N	DOC
<i>SB</i>																
River	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
1A	-	7.0	-	-	12	3.0	11	23	-	32	6.6	-	0.002	0.34	<0.02	-
1B	-	6.9	-	-	7.2	2.2	4.8	26	-	40	2.7	-	0.002	0.69	<0.02	-
1C	-	7.4	-	-	9.3	1.6	4.5	28	-	42	3.6	-	0.029	1.15	<0.02	-
2A	-	6.9	-	-	10	2.0	5.5	27	-	41	13	-	0.013	1.13	<0.02	-
2B	-	7.2	-	-	9.8	1.4	4.2	29	-	44	3.0	-	0.038	1.79	<0.02	-
3A	-	6.9	-	-	10	2.6	6.8	23	-	36	5.4	-	0.004	0.76	<0.02	-
3B	-	6.9	-	-	11	2.3	5.0	23	-	37	5.4	-	0.017	0.97	0.04	-
4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
5A	-	6.2	-	-	8.2	2.8	7.5	46	-	56	42	-	0.006	<0.05	1.32	-
5B	-	7.1	-	-	6.3	1.8	5.5	40	-	55	23	-	0.010	0.54	1.00	-
5C	-	6.4	-	-	7.4	2.1	6.6	40	-	60	18	-	0.006	<0.05	<0.02	-
6A	-	6.1	-	-	9.2	2.3	8.3	48	-	74	23	-	0.003	0.10	<0.02	-
6B	-	6.1	-	-	9.0	2.3	8.1	47	-	74	23	-	0.003	0.07	<0.02	-
7	-	6.1	-	-	17	2.7	12	64	-	125	30	-	0.002	0.15	<0.02	-
<i>CB</i>																
1A	-	6.8	-	-	13	2.3	9.8	27	-	32	1.5	-	0.007	2.16	<0.02	-
1B	-	6.7	-	-	13	1.9	6.8	22	-	43	0.9	-	0.004	2.83	<0.02	-
1C	-	6.9	-	-	15	2.0	6.9	27	-	46	1.5	-	0.013	3.01	<0.02	-
2A	-	6.7	-	-	14	1.8	7.5	28	-	43	0.9	-	0.026	2.33	<0.02	-
2B	-	6.8	-	-	18	2.0	8.2	29	-	47	0.9	-	0.006	3.73	<0.02	-
3	-	6.7	-	-	18	1.8	8.3	37	-	54	1.2	-	0.004	3.01	<0.02	-
4A	-	7.0	-	-	12	1.6	6.5	34	-	45	1.2	-	0.003	1.55	0.03	-
4B	-	6.9	-	-	9.2	1.7	5.5	29	-	43	1.2	-	<0.002	0.15	0.06	-
5	-	6.6	-	-	74	19	36	140	-	377	27	-	0.002	1.73	<0.02	-

**Table A3.** Chemistry data for 19 August 1999 (Trip 3). Data in mg L<sup>-1</sup> unless otherwise noted.

Label	Field DO	Field pH	Field EC (μS/cm)	Alk. (mmol/L)	Ca <sup>2+</sup>	K <sup>+</sup>	Mg <sup>2+</sup>	Na <sup>+</sup>	TDFe	Cl <sup>-</sup>	SO <sub>4</sub> <sup>2-</sup>	TDP	FRP	NH <sub>4</sub> <sup>+</sup> -N	NO <sub>3</sub> <sup>-</sup> -N	DOC
<i>SB</i>																
River	11.8	7.5	–	0.54	7.5	2.0	6.9	34	0.3	59	1.5	–	<0.005	0.16	<0.02	–
1A	<0.1	6.8	–	0.58	7.8	2.1	6.3	31	1.3	57	6	–	<0.005	0.36	<0.02	–
1B	<0.1	7.1	–	0.53	7.3	2.3	6.7	30	1.2	46	4.5	–	<0.005	0.41	<0.02	–
1C	<0.1	7.1	–	0.65	8.3	1.7	5.1	29	1.9	50	8.7	–	0.108	0.98	<0.02	–
2A	0.3	7.0	–	0.16	8.7	2.7	6.9	36	0.5	57	9.9	–	<0.005	0.19	<0.02	–
2B	<0.1	7.0	–	0.28	5.5	2.1	3.6	26	2.5	46	10	–	<0.005	0.39	<0.02	–
3A	<0.1	6.5	–	0.18	8.2	2.5	6.0	34	0.5	54	13	–	<0.005	0.17	3.30	–
3B	<0.1	6.7	–	0.92	7.1	2.6	4.5	32	3.8	52	14	–	<0.005	0.28	<0.02	–
4	0.4	6.6	–	0.90	13	3.4	7.7	22	0.3	33	21	–	<0.005	0.32	0.91	–
5A	0.3	5.9	–	0.61	9.9	2.5	8.5	40	<0.1	36	17	–	<0.005	0.15	6.5	–
5B	0.2	6.2	–	0.40	6.0	2.1	4.8	37	0.3	62	20	–	0.006	0.16	<0.02	–
5C	<0.1	6.8	–	0.37	7.1	2.2	5.8	34	<0.1	64	16	–	<0.005	0.20	<0.02	–
6A	0.1	5.9	–	0.61	8.4	2.3	6.9	42	0.2	48	22	–	<0.005	0.19	2.6	–
6B	0.1	5.9	–	0.56	8.2	2.4	7.0	42	<0.1	57	20	–	<0.005	0.16	0.61	–
7	<0.1	6.0	–	0.54	12	2.3	8.1	53	2.0	101	22	–	<0.005	0.30	<0.02	–
<i>CB</i>																
1A	0.5	6.8	–	0.92	10	2.2	9.0	33	<0.1	64	17	–	<0.005	ND	<0.02	–
1B	0.2	6.8	–	1.3	12	2.0	8.5	24	4.4	42	35	–	0.224	1.3	<0.02	–
1C	1.8	7.0	–	0.78	10	1.8	6.8	32	0.4	48	20	–	<0.005	0.59	<0.02	–
2A	0.2	6.7	–	1.1	15	1.7	7.5	27	1.3	44	36	–	<0.005	1.8	<0.02	–
2B	0.2	6.8	–	1.9	17	2.0	10	27	1.1	43	36	–	<0.005	2.7	<0.02	–
3	0.2	6.7	–	2.0	21	1.4	8.9	31	5.6	49	45	–	<0.005	2.8	<0.02	–
4A	0.6	6.8	–	0.90	10	1.4	5.2	26	0.4	42	29	–	<0.005	1.3	<0.02	–
4B	<0.1	7.0	–	1.1	9.2	1.9	5.8	25	3.1	40	24	–	0.028	1.8	<0.02	–
5	ND	6.5	–	1.7	37	2.6	16	80	9.1	186	20	–	<0.005	1.1	<0.02	–

**Table A4.** Chemistry data for 10 December 1999 (Trip 4). Data in mg L<sup>-1</sup> unless otherwise noted.

Label	Field DO	Field pH	Field EC (μS/cm)	Alk. (mmol/L)	Ca <sup>2+</sup>	K <sup>+</sup>	Mg <sup>2+</sup>	Na <sup>+</sup>	TDFe	Cl <sup>-</sup>	SO <sub>4</sub> <sup>2-</sup>	TDP	FRP	NH <sub>4</sub> <sup>+</sup> -N	NO <sub>3</sub> <sup>-</sup> -N	SrSi
<i>SB</i>																
River	9.4	8.05	–	0.7	–	–	–	–	–	67	–	–	<0.005	<0.02	<0.02	0.4
1A	<0.1	7.07	–	0.8	–	–	–	–	–	74	–	–	<0.005	0.40	1.0	4.4
1C	<0.1	6.94	–	0.9	–	–	–	–	–	65	–	–	<0.005	0.99	0.01	3.5
2A	<0.1	6.60	–	0.6	–	–	–	–	–	64	–	–	0.007	0.32	1.4	4.5
2B	<0.1	6.70	–	0.7	–	–	–	–	–	59	–	–	<0.005	1.0	<0.02	4.9
3A	<0.1	6.70	–	0.6	–	–	–	–	–	38	–	–	<0.005	0.18	9.9	4.7
3B	<0.1	6.60	–	1.0	–	–	–	–	–	47	–	–	<0.005	0.26	0.72	6.2
5A	0.8	6.01	–	0.7	–	–	–	–	–	47	–	–	0.007	0.03	<0.02	7.1
5B	<0.1	6.05	–	0.7	–	–	–	–	–	51	–	–	0.017	<0.02	0.03	5.8
5C	<0.1	6.10	–	0.8	–	–	–	–	–	56	–	–	0.007	0.08	<0.02	5.7
6A	<0.1	6.00	–	0.6	–	–	–	–	–	84	–	–	<0.005	0.04	<0.02	7.0
6B	<0.1	6.07	–	0.5	–	–	–	–	–	72	–	–	<0.005	0.04	<0.02	6.0
7	<0.1	6.04	–	0.7	–	–	–	–	–	111	–	–	<0.005	0.19	0.23	8.7
<i>CB</i>																
River			–	0.7	–	–	–	–	–	64	–	–	<0.005	0.05	<0.02	0.19
1A	<1	6.80	–	0.8	–	–	–	–	–	57	–	–	<0.005	0.24	0.10	4.8
1B	<1	6.60	–	1.1	–	–	–	–	–	69	–	–	0.068	1.6	0.03	13
1C	<1	6.90	–	0.7	–	–	–	–	–	59	–	–	0.007	0.14	0.11	1.4
2A	<1	6.60	–	1.4	–	–	–	–	–	66	–	–	0.009	1.2	0.03	11
2B	<1	6.60	–	1.3	–	–	–	–	–	64	–	–	0.006	2.5	0.02	12
3	<1	6.50	–	2.0	–	–	–	–	–	57	–	–	0.016	2.0	0.06	14
4A	<1	6.70	–	1.0	–	–	–	–	–	47	–	–	0.008	1.1	<0.02	8.6
4B	<1	6.70	–	1.2	–	–	–	–	–	56	–	–	<0.005	1.9	0.02	8.7
5	<1	6.80	–	1.9	–	–	–	–	–	159	–	–	<0.005	0.79	0.02	6.0

**Table A5.** Chemistry data for 8 March 00 (trip 5). Data in mg L<sup>-1</sup> unless otherwise noted.

Label	Field DO	Field pH	Field EC (μS/cm)	Alk. (mmol/L)	Ca <sup>2+</sup>	K <sup>+</sup>	Mg <sup>2+</sup>	Na <sup>+</sup>	TDFe	Cl <sup>-</sup>	SO <sub>4</sub> <sup>2-</sup>	TDP	FRP	NH <sub>4</sub> <sup>+</sup> -N	NO <sub>3</sub> <sup>-</sup> -N	DOC
<b>SB</b>																
River	7.4	6.8	–	0.6	–	–	–	–	–	30	–	–	<0.005	0.07	<0.02	–
1A	<0.1	6.7	–	0.8	–	–	–	–	–	29	–	–	0.253	1.1	<0.02	–
1C	<0.1	6.7	–	1.2	–	–	–	–	–	37	–	–	0.135	0.09	1.5	–
2A	<0.1	6.6	–	1.1	–	–	–	–	–	31	–	–	0.165	0.36	<0.02	–
2B	<0.1	6.6	–	1.0	–	–	–	–	–	33	–	–	<0.005	1.0	<0.02	–
3A	<0.1	6.4	–	1.1	–	–	–	–	–	56	–	–	0.195	0.88	<0.02	–
3B	<0.1	6.7	–	0.8	–	–	–	–	–	60	–	–	0.022	0.9	<0.02	–
5A	<0.1	6.0	–	0.9	–	–	–	–	–	45	–	–	0.008	0.1	0.32	–
5B	<0.1	5.9	–	0.7	–	–	–	–	–	47	–	–	0.019	<0.02	<0.02	–
5C	<0.1	6.0	–	0.8	–	–	–	–	–	55	–	–	0.024	<0.02	<0.02	–
6A	<0.1	5.8	–	0.7	–	–	–	–	–	55	–	–	0.013	<0.02	1.1	–
6B	<0.1	5.8	–	0.7	–	–	–	–	–	53	–	–	<0.005	<0.02	0.41	–
7	<0.1	6.1	–	0.7	–	–	–	–	–	76	–	–	0.008	0.19	<0.02	–
<b>CB</b>																
River	–	–	–	0.6	–	–	–	–	–	30	–	–	<0.005	0.02	<0.02	–
1A	2.7?	6.3	–	1.6	–	–	–	–	–	33	–	–	0.101	0.51	<0.02	–
1B	2.2?	6.5	–	0.6	–	–	–	–	–	31	–	–	<0.005	<0.02	<0.02	–
1C	<0.1	6.4	–	1.1	–	–	–	–	–	55	–	–	0.076	1.30	<0.02	–
2A	<0.1	6.4	–	1.6	–	–	–	–	–	49	–	–	0.111	1.5	0.04	–
2B	0.3	6.6	–	1.3	–	–	–	–	–	53	–	–	0.115	3.0	<0.02	–
3	0.8	6.6	–	1.3	–	–	–	–	–	52	–	–	0.308	2.9	<0.02	–
4A	<0.1	6.8	–	0.8	–	–	–	–	–	53	–	–	0.165	1.0	<0.02	–
4B	<0.1	6.8	–	1.2	–	–	–	–	–	47	–	–	0.244	2.2	<0.02	–
5	0.3	6.7	–	1.6	–	–	–	–	–	133	–	–	0.004	1.1	<0.02	–

**Table A6.** Chemistry data for 4 July 2000 (Trip 6). Data in mg L<sup>-1</sup> unless otherwise noted.

Label	Field DO	Field pH	Field EC (μS/cm)	Alk. (mmol/L)	Ca <sup>2+</sup>	K <sup>+</sup>	Mg <sup>2+</sup>	Na <sup>+</sup>	TDFe	Cl <sup>-</sup>	SO <sub>4</sub> <sup>2-</sup>	TDP	FRP	NH <sub>4</sub> <sup>+</sup> -N	NO <sub>3</sub> <sup>-</sup> -N	DOC
<i>SB</i>																
River	10.6	7.3	183	–	–	–	–	–	–	31	7.8	–	–	–	–	–
1A	1	6.9	238	–	–	–	–	–	–	39	1.8	–	–	–	–	–
1C	0.1	6.9	300	–	–	–	–	–	–	40	<0.4	–	–	–	–	–
2A	<0.1	6.7	205	–	–	–	–	–	–	27	3.2	–	–	–	–	–
2B	0.1	6.9	240	–	–	–	–	–	–	30	1.3	–	–	–	–	–
3A	0.2	6.5	308	–	–	–	–	–	–	52	<0.4	–	–	–	–	–
3B	<0.1	6.9	309	–	–	–	–	–	–	52	<0.4	–	–	–	–	–
5A	<0.1	5.9	257	–	–	–	–	–	–	39	18	–	–	–	–	–
5B	<0.1	6.0	258	–	–	–	–	–	–	50	9.2	–	–	–	–	–
5C	<0.1	6.1	264	–	–	–	–	–	–	52	4.3	–	–	–	–	–
6A	0.3?	6.0	307	–	–	–	–	–	–	45	28	–	–	–	–	–
6B	0.1	6.0	297	–	–	–	–	–	–	48	27	–	–	–	–	–
7	<0.1	6.0	338	–	–	–	–	–	–	67	20	–	–	–	–	–
<i>CB</i>																
River	–	–	184	–	–	–	–	–	–	32	8	–	–	–	–	–
1A	–	–	235	–	–	–	–	–	–	40	5.6	–	–	–	–	–
1B	–	–	219	–	–	–	–	–	–	35	9	–	–	–	–	–
1C	–	–	258	–	–	–	–	–	–	54	1.9	–	–	–	–	–
2A	–	–	345	–	–	–	–	–	–	43	<0.4	–	–	–	–	–
2B	–	–	322	–	–	–	–	–	–	53	1.9	–	–	–	–	–
3	–	–	305	–	–	–	–	–	–	54	0.4	–	–	–	–	–
4A	–	–	270	–	–	–	–	–	–	59	1.4	–	–	–	–	–
4B	–	–	292	–	–	–	–	–	–	44	<0.4	–	–	–	–	–
5	–	–	526	–	–	–	–	–	–	107	1.3	–	–	–	–	–

**Table A6.** Chemistry data for 5 September 2000 (Trip 7). Data in mg L<sup>-1</sup> unless otherwise noted.

Label	Field DO	Field pH	Field EC (μS/cm)	Alk. (mmol/L)	Ca <sup>2+</sup>	K <sup>+</sup>	Mg <sup>2+</sup>	Na <sup>+</sup>	TDFe	Cl <sup>-</sup>	SO <sub>4</sub> <sup>2-</sup>	TDP	FRP	NH <sub>4</sub> <sup>+</sup> -N	NO <sub>3</sub> <sup>-</sup> -N	DOC
<i>SB</i>																
River	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
1A	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
1C	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
2A	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
2B	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
3A	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
3B	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
5A	0.1	6.4	217	–	5.8	2.7	5.2	28	–	35	6.2	–	–	–	–	–
5B	<0.1	6.3	226	–	5.7	2.3	5.1	32	–	44	4.2	–	–	–	–	–
5C	<0.1	6.1	247	–	6.4	2.2	5.8	34	–	48	1.1	–	–	–	–	–
6A	<0.1	6.0	242	–	6.7	2.3	6.5	30	–	40	10	–	–	–	–	–
6B	<0.1	6.2	237	–	5.5	2.3	5.9	31	–	43	6.0	–	–	–	–	–
7	<0.1	6.2	337	–	8.1	2.1	6.2	45	–	62	21	–	–	–	–	–
<i>CB</i>																
River	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
1A	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
1B	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
1C	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
2A	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
2B	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
3	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>	<i>flood</i>
4A	<0.1	7.0	318	–	7.0	1.6	5.0	29	–	43	2.2	–	–	–	–	–
4B	<0.1	6.9	260	–	11	1.5	6.6	32	–	45	<0.5	–	–	–	–	–
5	<1	7.1	464	–	14	2.3	8.4	73	–	83	2.5	–	–	–	–	–

**Table A7.** Chemistry data for 7 March 2001 (Trip 9). Data in mg L<sup>-1</sup> unless otherwise noted.

Label	Field DO	Field pH	Field EC (µS/cm)	SrSi	Ca <sup>2+</sup>	K <sup>+</sup>	Mg <sup>2+</sup>	Na <sup>+</sup>	TDFe	Cl <sup>-</sup>	SO <sub>4</sub> <sup>2-</sup>	TDP	FRP	NH <sub>4</sub> <sup>+</sup> -N	NO <sub>3</sub> <sup>-</sup> -N	DOC
<i>SB</i>																
River	8.4	ND	ND	0.2	8.4	1.4	6.7	24	<0.1	38	8.5	<0.005	<0.005	0.05	<0.005	5.1
3A	<0.1	6.4	258	4.3	9.1	2.0	7.4	25	4.0	42	16	0.46	0.41	1.3	0.07	4.3
3B	<0.1	6.6	278	4.8	10	1.9	5.4	23	14	37	20	0.55	0.49	1.5	0.09	3.7
5A	<0.1	5.6	697	13	20	1.9	16	90	0.6	150	82	0.046	0.051	0.1	0.02	3.2
5B	<0.1	5.7	616	9.0	23	2.8	18	69	<0.1	130	80	0.030	0.034	0.04	<0.005	3.5
5C	0.1	6.0	184	5.2	5.3	0.8	4.7	21	<0.1	32	6.9	0.039	0.036	0.10	0.007	4.7
6A	0.1	5.8	819	11	23	2.8	21	100	0.2	200	79	0.028	0.018	0.07	0.07	3.4
6B	0.1	5.9	222	6.0	6.1	1.0	5.7	29	<0.1	44	14	0.009	0.015	0.06	0.005	3.5
7	<0.1	7.5	938	6.7	32	2.5	23	110	9.3	230	59	0.049	0.056	0.38	0.060	2.4
<i>CB</i>																
River	ND	7.5	226	0.2	8.5	1.6	6.6	24	<0.1	41	8.6	<0.005	<0.005	0.10	<0.005	5.3
1B	0.4	6.9	227	9.7	10	6.8	7.2	25	2.0	33	2.3	0.27	0.22	1.0	0.05	6.8
1C	ND	ND	ND	9.6	13	4.4	8.5	24	6.9	39	<0.4	0.21	0.19	1.5	0.1	9.0
2A	<0.1	7.1	244	0.3	9.0	3.1	6.5	23	6.6	35	3.2	0.12	0.069	1.4	0.09	ND
2B	0.6	6.8	345	13	16	2.2	8.9	29	10	45	1.1	0.090	0.090	3.2	0.15	6.9
3	0.3	6.8	505	17	24	2.2	12	39	20	79	<0.4	0.12	0.10	3.2	0.08	9.2
4A	0.2	7.0	320	10	13	1.2	6.9	29	10	51	<0.4	0.29	0.29	1.7	0.06	4.2
4B	0.2	7.0	318	9.1	12	1.5	8.0	26	13	48	<0.4	0.68	0.72	2.1	0.07	5.2
5	0.5	7.0	496	6.3	24	1.6	10	55	7.5	93	0.8	0.15	0.11	1.1	0.05	4.7
<i>Bores</i>																
		δD	δ <sup>18</sup> O													
7853	-	-31.9	-4.14	7.3	290	120	1100	14000	<0.1	18000	3800	0.99	0.060	1.9	<0.005	5.3
7682	-	-31.3	-4.05	5.2	340	120	1200	12000	<0.1	18000	3500	1.2	0.32	2.8	0.5	6.5
7866	-	-36	-6	6.2	16	13	51	990	1.1	1500	120	0.038	0.030	0.4	0.010	1.5
7856	-	-33.1	-5.07	12	140	10	340	4900	2.8	5700	2100	1.4	1.0	0.7	<0.005	3.6
7683	-	-34	-4.55	7.9	330	230	1300	14000	2.6	20000	4100	1.0	0.084	2.1	<0.005	6.7
7852	-	-30.7	-4.47	12	120	48	360	5200	<0.1	7300	1100	0.30	0.079	0.5	0.4	3.8

**Table A8.** Deuterium and  $\delta^{18}\text{O}$  (‰ V-SMOW) for rivers and piezometers (data for regional bores in Table A7).

	26 March 99		10 August 99		1 December 99		8 March 00		4 July 00		5 September 00		15 January 01		7 March 01	
	$\delta\text{D}$	$\delta^{18}\text{O}$	$\delta\text{D}$	$\delta^{18}\text{O}$	$\delta\text{D}$	$\delta^{18}\text{O}$	$\delta\text{D}$	$\delta^{18}\text{O}$	$\delta\text{D}$	$\delta^{18}\text{O}$	$\delta\text{D}$	$\delta^{18}\text{O}$	$\delta\text{D}$	$\delta^{18}\text{O}$	$\delta\text{D}$	$\delta^{18}\text{O}$
<i>River</i>	ND	ND	-31.1	-5.6	ND	ND	-25.5	-3.85	-28.9	-4.70	ND	ND	-21.4	ND	-23.6	-3.44
<i>SB</i>																
1A	-26	-3.62	-27.2	-4.4	-24.5	-4.53	-30	-4.29	-24.2	-3.54	<i>flood</i>	<i>flood</i>	<i>lost</i>	ND	<i>lost</i>	<i>lost</i>
1B	-24.2	-3.25	-25.9	-4.08	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>
1C	-25.4	-3.35	-25.1	-3.9	-24.6	-4.35	-21.4	-3.45	-27.5	-3.98	<i>flood</i>	<i>flood</i>	-26.5	ND	<i>lost</i>	<i>lost</i>
2A	-24.2	-3.39	-27.5	-4.35	-27.5	-4.99	-21.8	-3.58	-33.2	-4.55	<i>flood</i>	<i>flood</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>
2B	-24.5	-3.45	-25	-4.1	-25.5	-4.37	-22.9	-3.42	-29.1	-4.04	<i>flood</i>	<i>flood</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>
3A	-24.8	-4.47	-24.3	-4.02	-22	-3.65	-26.2	-4.48	-23.6	-3.67	<i>flood</i>	<i>flood</i>	-28.1	ND	-19.1	-3.7
3B	-26.6	-4.51	-26	-4.12	-22.3	-3.92	-24.9	-4.55	-23	-3.61	<i>flood</i>	<i>flood</i>	-26	ND	-21	-3.23
4	ND	ND	-22.6	-4.09	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>
5A	-27	-5.04	-25.9	-4.51	-23	-3.84	-23.6	-3.85	-23.4	-3.6	-26.6	-4.27	-20.5	ND	-21.9	-3.91
5B	-29.4	-5.12	-29.9	-5.33	-26	-4.47	-24.6	-4.28	-26.3	-4.13	-24.9	-3.82	-26.4	ND	-22.9	-3.79
5C	-29.8	-5.24	-29.1	-4.87	-25.5	-4.4	-27.5	-4.25	-27.2	-4.15	-26.5	-4	-26.9	ND	-25.1	-4.71
6A	-29.9	-5.3	-28.3	-4.74	-26	-4.99	-26.5	-4.39	-24.8	-3.81	-23.7	-3.66	-24.5	ND	-26.2	-4.44
6B	-29.4	-5.21	-29	-5.36	-27.8	-4.98	-26.3	-4.34	-24.8	-4.06	-23.8	-3.73	-27.3	ND	-27.8	-4.73
7	-30.7	-5.26	-31.2	-5.57	-27.4	-5.32	-28.9	-5.16	-29.4	-4.95	-28.5	-4.96	-27.3	ND	-28.1	-4.67
<i>CB</i>																
1A	-26.4	-4.15	-28.3	-4.6	-22.1	-4.38	-27.7	-3.98	-27.1	-4.04	<i>flood</i>	<i>flood</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>	<i>lost</i>
1B	-30.1	-5	-25.3	-3.99	-27.1	-4.51	-24.4	-4.24	-27.3	-4.22	<i>flood</i>	<i>flood</i>	-27.2	ND	-26.5	-4.61
1C	-27.5	-4.51	-26.9	-4.2	-23.1	-4.21	-27.3	-4.65	-27.8	-4.55	<i>flood</i>	<i>flood</i>	-28.2	ND	-29.2	-4.87
2A	-24.5	-3.85	-24.2	-3.96	-25.3	-4.44	-21.1	-3.53	-23.5	-3.27	<i>flood</i>	<i>flood</i>	-26.3	ND	-25.2	-4.38
2B	-26.6	-4.11	-25.7	-4.17	-24.8	-4.25	-26.3	-4.4	-26.6	-4.23	<i>flood</i>	<i>flood</i>	-28.5	ND	-27	-4.64
3	-27.3	-4.1	-27.3	-4.33	-25	-4.22	-27.2	-4.44	-28	-4.45	<i>flood</i>	<i>flood</i>	-27.6	ND	-26.5	-4.33
4A	-26.1	-3.97	-28.9	-4.83	-26.2	-4.33	-26.3	-4.37	-28.2	-4.71	-24.9	-4	-26.7	ND	-25.5	-4.26
4B	-27	-4.07	-26.5	-4.1	-26.4	-3.87	-27.4	-4.6	-25.1	-3.79	-25.6	-4.39	-26	ND	-26.8	-4.7
5	-31.4	-4.9	-31.6	-5.04	-32.2	-5.04	-31.3	-5.09	-33.2	-5.2	-32.3	-4.98	-31.7	ND	-31.4	-5.13