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Alice Springs SAT Project
Hydrological and Water Quality Monitoring Report 2008-2009

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This is a report of the NWC Raising National Water Standards project:
Facilitating recycling of stormwater and reclaimed water via aquifers.

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Cover Photograph:

Description: Recharge Basin 1 in the Alice Springs SAT Trial site

Photographer: Karen Barry
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ACRONYMS

AZRI – Arid Zone Research Institute

DAF – Dissolved Air Flotation

DAFF – Dissolved Air Flotation and Filtration

DO – Dissolved oxygen

EC – Electric conductivity (Specific Conductance)

SAT – Soil Aquifer Treatment

SAR – Sodium Adsorption Ratio

SCADA – Supervisory Control and Data Acquisition – Fully automatic electronic monitoring and controlling system

RB – Recharge basin

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EXECUTIVE SUMMARY

The study site is located in Alice Springs (Northern Territory) in central part of Australia. Soil Aquifer Treatment (SAT) commenced here on 3rd June 2008 at the Arid Zone Research Institute using treated wastewater following dissolved air flotation (DAF) at Alice Springs sewage treatment plant. The project is led by Power and Water Corporation (PWC) and this report, prepared by CSIRO, and supported by PWC and National Water Commission, is to evaluate the performance of the project, in particular rates of infiltration and changes in groundwater quality to build a basis for understanding how to continue to improve the operation of the site.

In the period from June 2008 to December 2009, 317.4 ML of reclaimed wastewater was discharged to four recharge basins (total area of 7,640 m^2). When the evaporative loss and the gain from rainfall are taken into account about 308 ML of water infiltrated within 19 months in these basins with an average hydraulic loading of 40 m at an average rate of 70 mm/d over the total period of 575 days, or approximately 240 mm/d for the days that basins received flow. The maximum height of the groundwater mound in the regional water table at the SAT site was 1.5 m, which is 17.5 m below the floor of the basins, suggesting that the previously calculated height of 3 m for 600 ML/yr looks a reliable estimate, and the aquifer has adequate capacity to store and convey the recharged water. Salinity in the groundwater beneath the site and to a distance 200 m downgradient has been reduced as a result of recharge and nitrate concentrations have increased.

Infiltration rates varied with time in each recharge basin but insufficient data were available to allow correlation of changes with environmental and water quality parameters. Trend lines fitting infiltration rates measured infrequently during the drying of basins showed a tendency for typical recharge rates to be lower by 50% than their initial values in each basin (except RB3) over the 19 months. However against this trend an improvement in operating practices with more diligent attention to drying periods, and reduced use of RB4 gave an increase in average intake rates from 220 mm/d in 2008 to 260 mm/d in 2009. Basin RB2 performed best with a total hydraulic loading of 52.5m followed closely by RB1 (46.5m) and RB3 (40.9m) with RB4 with the least permeable soils coming in a distant last (18.3m).

As a result of low infiltration rates in basins RB3 and RB4 these were scraped in October 2009. Consequently RB3 infiltration capacity was restored to around its original value. However in RB4 scraping had no impact on infiltration rate. On 13 December 2009 RB5 the fifth and largest basin (2,629 m^2 area as constructed) commenced operation. Its infiltration rate measured on its first fill appears similar to other basins. The total current area of infiltration of five recharge basins is 10,269 m^2.

The infiltration rates in all basins are less than the design infiltration rates of 300 mm/d which were based on the results of field permeameter tests and two pit tests using Roe Creek water combined with laboratory column tests with two soil types and four water types including the quality of water anticipated to be produced by the DAF plant. On average, water from the DAF plant during 2008-2009 had substantially higher values of turbidity, membrane filtration index, total nitrogen, total organic carbon and bicarbonate than the water used in the column studies to represent it. The column studies also used raw waters with higher values of these constituents and found clogging was more severe than has been observed in the basins. There is no doubt that the quality of recycled water is reducing infiltration rates below their intended values.

The original design was for 600 ML/year recharge from 1.1 ha of basins and in 2009 the intake was 178 ML which aside from 3 ML in RB5 was recharged through

0.76 ha. If current infiltration rates could be maintained recharging 600ML/yr would require 2.5 ha of basins. There are alternatives including slotting the basins or extending the treatment by including granular media filtration as a post-treatment at the DAF plant (i.e. DAFF) or possibly both. Slotting the basins, e.g. starting with RB4 to intersect with coarse sand ~2m below surface and backfilling with similar sand would substantially increase infiltration rates, at least in the short term. This would reduce the unsaturated zone thickness by 10% but is not expected to materially substantially influence the current low level of treatment above the watertable. Before slotting or building new basins a geophysical exploration is warranted with the aim of identifying options and designs to maximise cost-effectiveness.

The operation of SAT has resulted in the freshening of the Quaternary aquifer and lowering of the sodium adsorption ratio in the plume downgradient. As a result of this salinity reduction groundwater is becoming more suitable for agricultural irrigation and can be recovered in the future. However, the intended reduction in nutrients in groundwater occurring beneath the basins is incomplete. While phosphorus removal is highly effective, the total nitrogen concentration in discharged water averaging 15 mgN/L is only attenuated about 20% by the time it reaches the regional watertable. Ammonium and organic nitrogen concentrations that are the dominant forms of N in source water are either adsorbed and/or converted to nitrate in the unsaturated zone. Almost all N reaching the watertable is in the form of nitrate.

The attenuation of nitrogen is low because the soil beneath the basin is aerobic and denitrification is inhibited. There are two contributory factors. Firstly, a sampling campaign in July 2009 revealed that the shallow soil immediately beneath basins RB2 and RB4 did not saturate, or was saturated for only a few hours after the filling commenced, as the clogging layer (schmutzdecke) established very quickly due to clogging agents in the source water. This inhibited perching of water on shallow silts and prevented saturation of the zone where denitrification is expected to occur fastest. Secondly, the organic carbon in the DAF recycled water seems to be quite recalcitrant. Biodegradation of the organic carbon could be enhanced by reducing the average residence time in lagoons of source water for the DAF plant.

Changing source water quality e.g. by lowering the concentration of nitrogen, and reducing other clogging agents (particulates, phosphorus, excess polymer and calcite saturation index) or increasing the fraction of labile organic carbon may be needed if the concentration of nitrogen in recovered groundwater is to be reduced. Inclusion of filtration after the DAF plant warrants consideration due to its potentially beneficial effects on infiltration rates which may also facilitate attenuation of nitrogen reaching the aquifer. Simple granular media column tests could be performed at the DAF plant to evaluate the likely change in water quality if filtration was included, and to also evaluate the impact on rates of clogging.

1. INTRODUCTION

1.1. Soil Aquifer Treatment

Managed aquifer recharge (MAR) of treated sewage effluent has a significant role to play in water storage and treatment for reuse. Soil aquifer treatment (SAT) is a particular method of MAR where treated effluent is intermittently placed in recharge basins, allowing for infiltration into the ground for the recharge of aquifers (Fig. 1). As the effluent moves through the soil and the aquifer, it can undergo significant quality improvements through physical, chemical and biological processes. The water is stored in the underlying unconfined aquifer for subsequent reuse, generally for irrigation water supplies. This is a valuable water resources management method in areas with high evaporation rates.

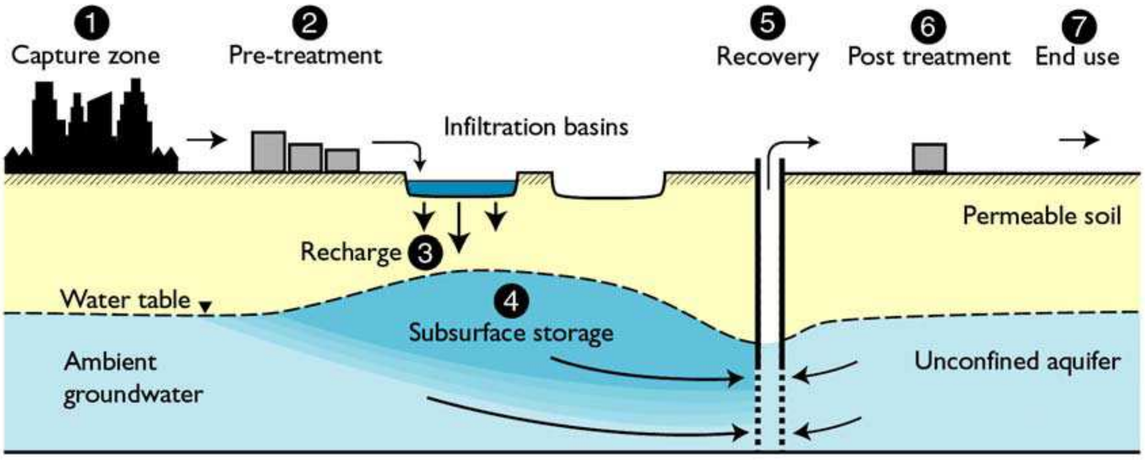


Figure 1 Soil Aquifer Treatment System infiltrating water to an unconfined aquifer (MAR Guidelines NRMMC-EPHC-NHMRC 2009)

SAT provides mechanical filtration of suspended particles. From a chemical point of view it is particularly focussed on producing intermittent aerobic and anaerobic conditions in the soil under the basin. This facilitates nitrification and denitrification and therefore partial removal of organic and inorganic nitrogen (N) and organic carbon (OrgC) from treated effluent. Other constituents in wastewater such as phosphorus (P), some non-aromatic organics (including polysaccharides and proteins), trace metals or pathogens are usually also effectively removed during SAT (Fox *et al.* 2001, Lin *et al.* 2004, Drewes *et al.* 2006).

The efficacy and sustainability of a SAT system is indicated by measurements of infiltration rates, as well as organic carbon, nitrogen and pathogen removal from the treated wastewater, as implied by groundwater sampling and analysis. The process can be controlled by an appropriate duration of drying and wetting cycles of individual basins with the objective of maximising and sustaining the hydraulic loading and producing water of acceptable quality for its intended uses. The operational should be dependent on quality of available water used for recharge.

When water enters a recharge basin it infiltrates into the underlying soil and at early time, increases the water content in the unsaturated zone. When the source of water is disconnected the basin drains and although surface infiltration stops, water continues to move downward due to both capillary and gravitational forces. Eventually, the water reaches the watertable and flows with the native groundwater.

In the soil percolation zone, oxygen is supplied continuously from the surrounding unsaturated zone. Nevertheless, this zone usually becomes anoxic during infiltration periods as a result of ammonium and organic carbon oxidation. Consequently other processes including anaerobic ammonium oxidation, denitrification or dissolution of metal oxides are favoured. When the infiltration is stopped, the upper soil layer is re-aerated resulting in cyclic aerobic-anoxic conditions in the unsaturated zone (Fox *et al.* 2005).

Operation of a SAT system results in the formation of a perched water table (mound) in the aquifer being recharged. The height of the water table beneath a recharge basin is a function of infiltration rate, infiltration time and transmissivity of the aquifer. A mound can also occur in an unsaturated zone when it is created just above a restricting layer (Todd and Mays 2005).

Infiltration rates from infiltration basins usually decline due to accumulation of biofilm, algae, suspended solids and chemical precipitates on the basin’s surface. A clogging layer in SAT system is defined as a “schmutzdecke”. The clogging layer development and reduced infiltration rates can adversely affect the hydraulic loading. Periodic drying of the basins is necessary to renew hydraulic conductivity. Occasionally the clogging layer has to be physically removed.

1.2. Alice Springs SAT operation

The Alice Springs SAT project was conceived in 2000 at a workshop to address overflows from the sewage treatment plant into Ilparpa Swamp. SAT had the benefit of increasing recharge to brackish groundwater resources to generate irrigation water supplies. There were suitable unconfined aquifers, ample land, treatment requirements were less onerous than for Aquifer Storage and Recovery (ASR), and storage costs were considerably lower than for tanks. Open water bodies would be subject to substantial evaporation losses with water quality deterioration and insect pests.

By 2004 field investigations were undertaken to select a site for the SAT operation and a feasibility study and environmental impact assessment were completed (Knapton *et al.* 2004). This was followed by a public consultation phase and finally construction of the Dissolved Air Flotation (DAF) plant, the pipeline, and four SAT basins were completed in early 2008. After commissioning of the DAF plant the basins were first operated on 3 June 2008. A fifth basin was commissioned in December 2009.

The study site is located at the Arid Zone Research Institute (AZRI) situated approximately 7 km south east of Alice Springs towards the airport (Fig. 2). The DAF plant is situated next to Ilparpa Swamp in Blatherskite Park. The Roe Creek Borefield is located approximately 7 km south west of the SAT site. Although this investigation has been confined to the AZRI property, the groundwater system below the AZRI site extends from Blatherskite Park in the north to south of the airport.

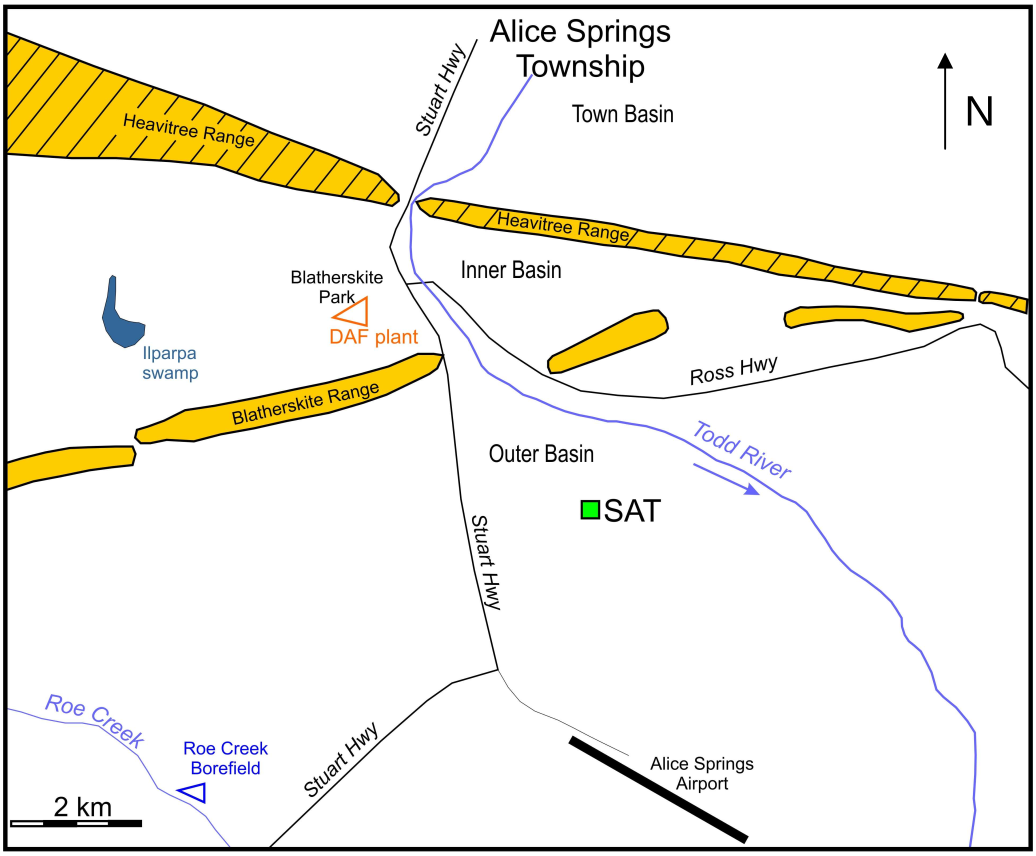


Figure 2 Location of SAT site in the area of Alice Springs

The objectives of the SAT project include:

- Ceasing dry weather overflows to Ilparpa Swamp:
  - return Ilparpa Swamp to its natural ephemeral state, i.e. stop dry weather overflows and minimise wet weather overflows to Ilparpa swamp,
  - eliminate excessive mosquito breeding associated with overflows,
  - reduce habitat available for exotic plant species,
  - reduce spread of weeds.
- Water conservation and reuse:
  - reduction in evaporative losses,
  - reduction in demand on Alice Springs potable water supply by creating a new water resource.

Alice Springs SAT scheme is poised to become a unique and exemplary water reuse project for Australia and an attractive site for SAT research. It offers a purposefully designed greenfields site with a capacity to monitor and evaluate impacts relative to the initial conditions with a degree of operational control.

2. GEOLOGICAL SETTING

2.1. General geology and hydrogeology

The area under examination is located on the margin of the Amadeus Basin. This large scale geological formation generally consists of sedimentary (sandstone, siltstone, conglomerate) and metamorphic rocks (shale, quartzite) age from Neoproterozoic to early Carbonifeous (Wells *et al.* 1970). Beneath the Quaternary aeolian, alluvial, and fluvial deposits are Neogene sediments of lacustrine origin. The term Neogene is used in this report since “Tertiary” is no longer recognised as a formal geological unit by the International Commission of Stratigraphy.

The Quaternary comprises silts, sands, and gravels (Fig. 3). A distinct buried valley called a “palaeochannel” has been recognized below the Inner Farm Basin and Outer Farm Basin. This feature of the fluvial origin is filled with sand and gravel. The interpreted boundaries of the palaeochannel are shown in the insert map in Fig. 3.

Quaternary aquifers are found in the Town Basin (in the area of Alice Springs town), Inner Farm Basin (between Heavitree and Blatherskite Ranges) and Outer Farm Basin (south of Blatherskite Range; Fig. 2). The highly permeable deposits are restricted to the extension of the buried valley, where coarse grained material is present. They exhibit hydraulic conductivity in the range 1-120 m/d and specific yield of 0.07 (Quinlan and Woolley 1969, Berry 1991). In addition to natural infiltration which is primarily by the Todd River (when it flows) and via infiltration of precipitation to a lesser extent, the Inner Farm aquifer has been recharged since 1986 through artificial irrigation via application of effluent in Blatherskite Park. Discharge from the Quaternary basins is mostly to the pumping wells. Lateral groundwater flow in the Outer and Inner Farm Aquifers is towards the south-east (Fig. 4).

The Outer Farm aquifer, within which the AZRI site is located, is unconfined and the water table occurs at a depth of 15-20 metres. Natural seasonal water level fluctuations in this Quaternary aquifer are normally in the range of 1-2 metres, although rises up to 5 metres are possible due to high rainfall events and flooding of the Todd River (Knapton *et al.* 2004).

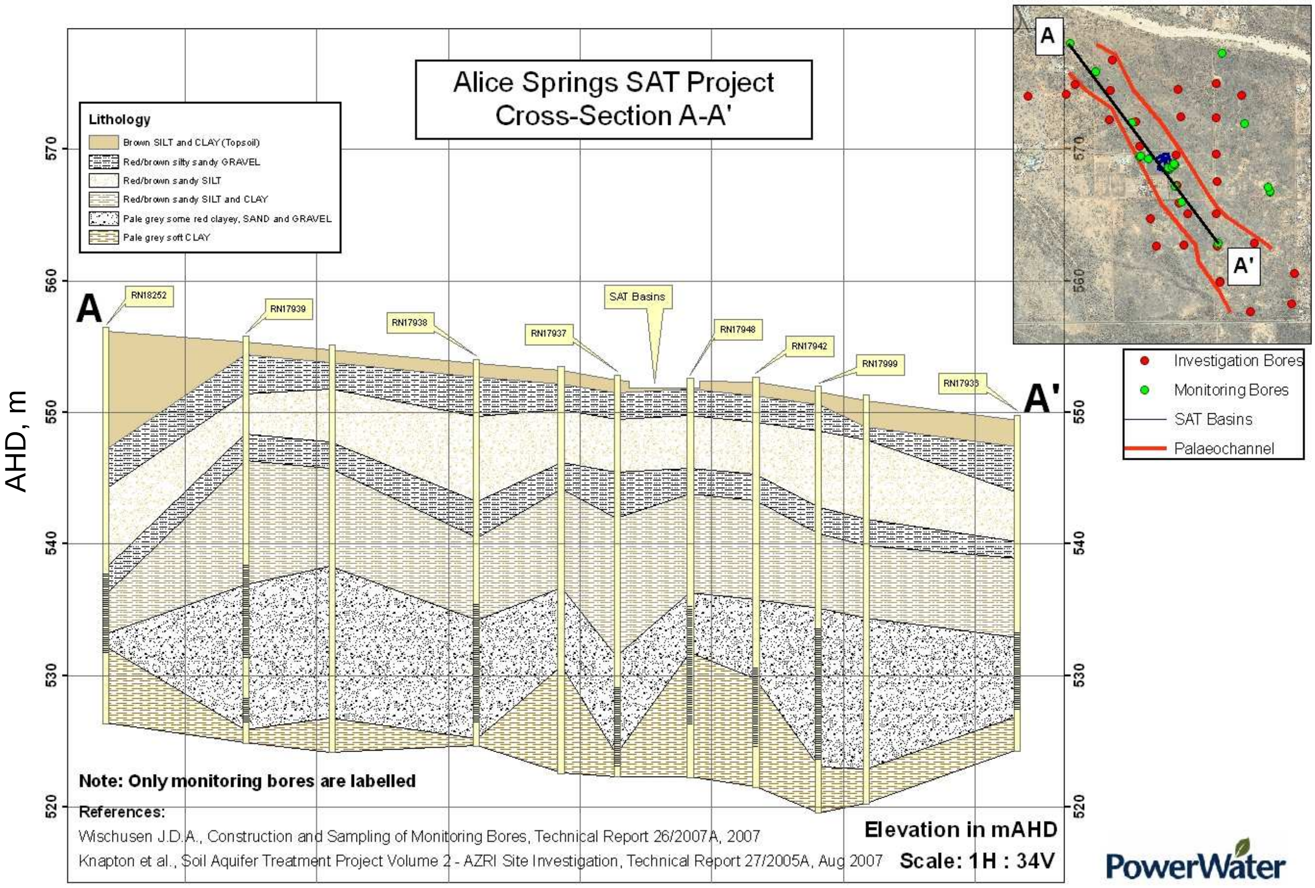


Figure 3 Geological cross section of Quaternary deposits at the AZRI site. The insert map shows the boundaries of the buried valley incised into Neogene clays (after Wischusen *et al*. 2007). Water table occurs at a depth ~18 m beneath the basins

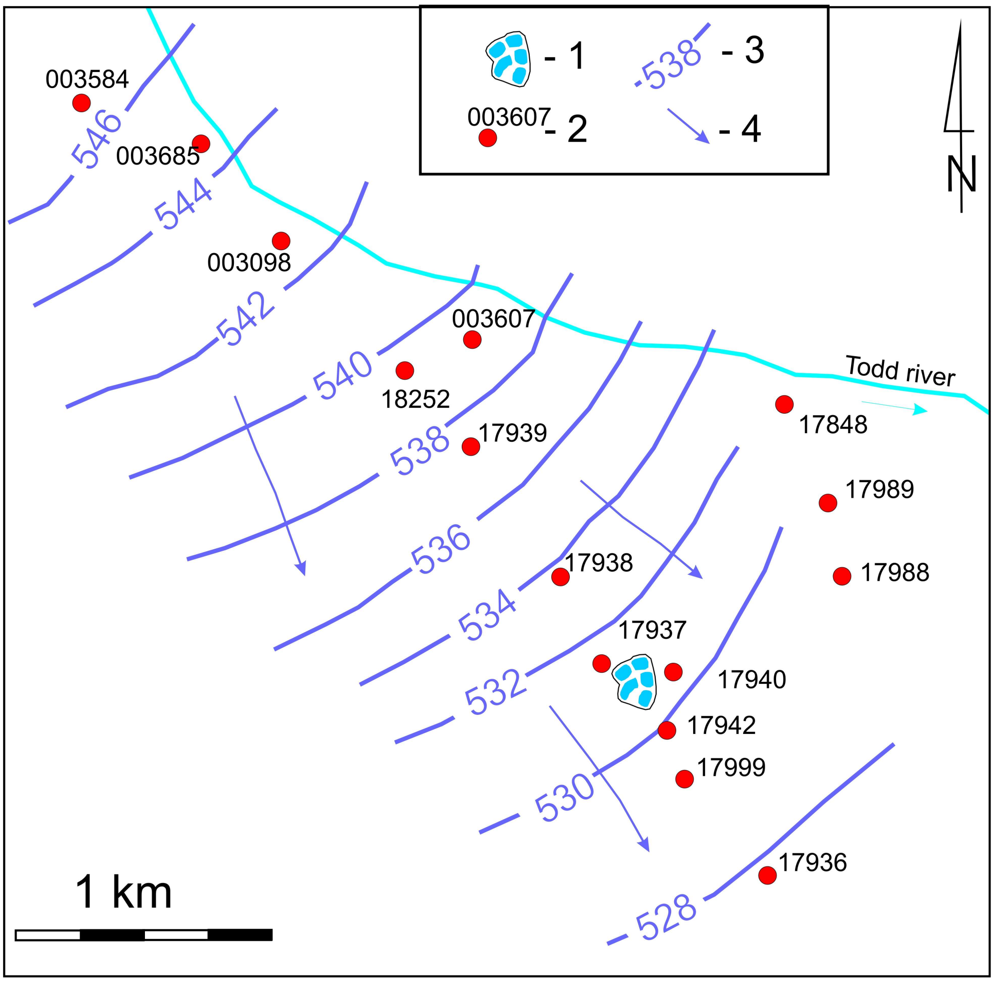


Figure 4 Hydrogeological sketch of the Outer Basin aquifer in the area of SAT site. Showing 1) infiltration basins, 2) monitoring wells screened in Quaternary deposits, 3) water level contours (m AHD, 2 metre interval) and 4) directions of groundwater flow. Water level contours are relevant for non-flooding conditions immediately prior to SAT operation.

The Neogene aquifer is of minor importance in the area of Alice Springs. Most of the pumping wells are located in the vicinity of the Alice Springs airport. The aquifer is believed to be well isolated due to existence of poorly permeable Neogene clays (Berry 1991, Knapton *et al.* 2004).

The Amadeus Basin contains a confined aquifer, the Mereenie Sandstone, which constitutes a major potable water supply for the town of Alice Springs. Due to excellent quality of groundwater, the aquifer is intensively extracted by pumping wells of the Roe Creek bore field (Fig. 2). Since the amount of water which recharges the groundwater system is much lower (500 ML/yr) than the extraction (10,000 ML/yr), pressures in the aquifer have continuously decreased since the 1960s (Knapton *et al.* 2004). In the area of the Roe Creek bore field the groundwater level has dropped to a depth of 150 m below surface which increases the cost of water extraction.

2.2. Local setting at the AZRI site

2.2.1. Geology and hydrogeology

The feasibility studies (Knapton *et al.* 2004) revealed that beneath the AZRI site the palaeochannel is approximately 300-400 metres wide, with a depth of 17 to 30 metres below the ground surface. The feature is filled with heterogeneous sediments of differing permeability (layered lensed series of gravels/cobbles, sands and clays). Typically the lowest part of the palaeochannel profile consists of gravel and cobble infill. Natural water level at the AZRI site occurs at a depth of 17-19 metres (Appendix A).

The upper 2 m of the geological profile consists of a continuous loamy and clayish layer (Appendix A, part c). Below this is a layer of coarse sand, 1-3 metres in thickness which has a variable distribution across the site, although is appears to coincide with the drainage depressions evident in the topographic maps. The dual ring infiltration tests implied distinct differences in infiltration rates across the site. Silty sandy soils are characterized by an infiltration rates of 3-10 m/d (4 m/d median), whereas gravelly sands have a much higher value, 2-90 m/day (30 m/d median). Tests of long-term (two months) infiltration rates in 6x6 meters pits revealed an average infiltration rate of 3 m/day (for the silty sands) and 10 m/day (for the gravelly sands). Soil permeameter tests could only be conducted on the silty sandy soils that exhibit average permeability of 4-5 m/day (Knapton *et al.* 2004).

2.2.2. Sediment geochemistry

The chemical composition and properties of sediments sampled at the AZRI (RN 17937-17940) were characterised in detail by Gates *et al.* (2009). Extraction experiments indicated substantial amount of iron oxides in the whole profile, which points to the large oxidation capacity of the sediments. Cation exchange capacity (CEC) of sediments sampled in the core 17939 ranged from 50 mmol/kg to 270 mmol/kg values indicating a large pool for cations (eg ammonium) present in recharge water during SAT operation. Carbonate content was generally low (<1-2 %), but was substantial (as high as 30 %) in lower sections of the core 17939 (Gates *et al.* 2009). The occurrence of natural organic matter was not reported by Gates *et al.* (2009).

2.2.3. Background groundwater chemistry

Native groundwater in the Quaternary aquifer is brackish (1340-3400uS/cm, 2650 uS/cm on average) with Ca-Na-Cl-SO_4 water type being dominant (Appendix B). According to guidelines for fresh and marine waters (ANZECC-ARMCANZ 2000), the native groundwater is unsuitable for irrigation of most crops due to high salinity values. The wells in the vicinity of the site prior to operation exhibited average Cl^- concentration of 530 mg/L, and EC 2730 µS/cm. Concentrations of nitrate in April 2008 varied from 2 to 6.5 mgN/L (5 mgN/L on average). The lowest salinity and concentration of most constituents was encountered in RN 17939 located around 1.2 km northwest of the site which is associated with flooding events of the Todd River. On 23/04/2008, groundwater in RN 17939 exhibited very low values of EC (1450 µS/cm), Cl^- (250 mg/L), NO_3 (2 mgN/L), Br^- (1550 µg/L), and B (80 µg/L).

The chemical composition of the ambient groundwater is primarily due to extensive evaporation and low recharge from precipitation as well as dilution of ambient brackish groundwater during infiltration from the Todd River. Most groundwater samples are oversaturated with respect to calcite, aragonite, and dolomite; and exhibit a 1:1 Ca/SO_4 ratio with subsaturation for gypsum (Jeuken 2004, Vanderzalm *et al.* 2010). This implies that the processes of equilibration with carbonate minerals in the saturated zone as well as precipitation/dissolution of gypsum in the unsaturated zone may be of importance.

2.2.4. Studies on clogging potential

Extensive laboratory experiments were performed with the aim of assessing the potential clogging of soil sampled from the AZRI site (Knapton *et al.* 2004, Mucha 2004). The effects of soil type (sand and loam), effluent pretreatment, hydraulic gradient (ponding depth), temperature and sunlight were evaluated using laboratory columns in an eight week experiment, with four wetting and four drying periods each of 1 week. In the final stage of the experiment with reclaimed water (4th wetting cycle), the hydraulic conductivity values varied from 0.8 to 2 m/day for sands and from 0.01 to 0.4 m/day for loams (compared with initial values of 23 m/d and 0.1 m/d respectively). The columns fed with potable water (Roe Creek borefield) exhibited a decline in hydraulic conductivity up to 1.5 m/day and 0.8 m/day for sands and loams, respectively. A 6-8 fold difference in hydraulic loadings observed among four source water quality types that had experienced different treatments demonstrated that the quality of recharge water is of prime importance in the development of clogging of both soil types tested. The reports made the recommendation that recharge basins should be placed on sandy soils and the quality of recharge water should not largely differ from that produced by the Bolivar DAF plant, South Australia.

3. METHODOLOGY FOR SITE INVESTIGATIONS

The site consists of five recharge basins (Fig. 5). RB1, RB2, RB3, and RB4 have been operated since June 2008. The total area of those four basins is 7640 m^2. The fifth basin (RB5) was incorporated in December 2009 which increased the total recharge area to 10269 m^2. The location of monitoring boreholes is shown in Fig. 4 and Fig. 5.

The monitoring program was developed to understand the hydraulic and water quality changes during a SAT operation with DAF treated wastewater. It consisted of three major field events; establishment of the site and installation of continuous monitoring equipment (March and May 2008); intensive field monitoring campaign 1 (September 2008); and the intensive field monitoring campaign 2 (July 2009).

RB4 was initially established as the main 'research' basin and was equipped with monitoring equipment (section 3.1). The location of sampling instruments downgradient of the site would allow to capture the recharge water plume. Subsequently RB2 was examined in more detail with additional instrumentation installed in 2009 as it appeared to be more representative of the system.

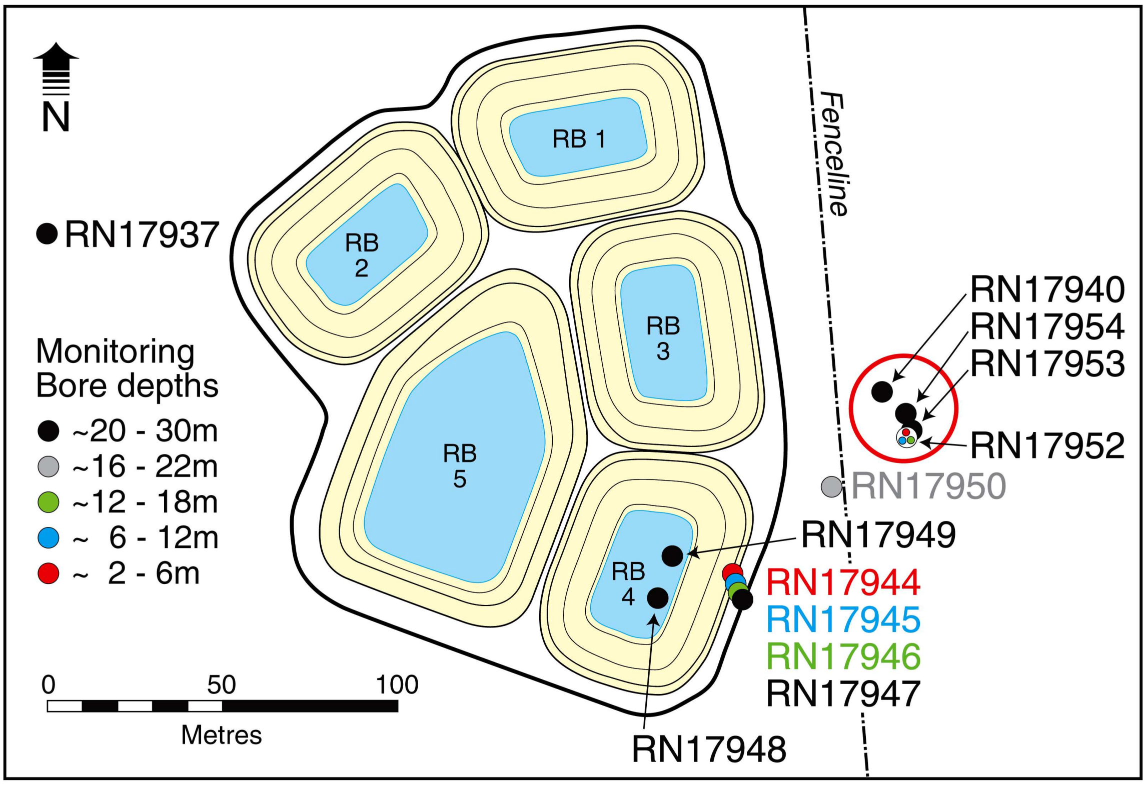


Figure 5 Location of the infiltration basins and the monitoring wells at the AZRI site. The final shape and size of RB5 was changed from that shown here to adapt to local requirements

3.1. Site instrumentation and monitoring of water quantity and quality

The quantitative effects of the site operation were examined by monitoring of continuous water inflow and water level in all basins. In RB4 soil moisture contents, soil suction, oxygen content and reduction-oxidation potential were also measured in the unsaturated zone as well as water level fluctuations in the saturated zone of the regional water table and in some intermittently perched layers above the watertable using piezometers.

Cumulative inflow to the recharge basins and water levels in the basins were recorded every 15 minutes at the end of the delivery line for each basin (RB1, RB2, RB3, RB4, and RB5).

Samples of soil sediments were collected at 0.1 m intervals to 2.4 m depth to allow water contents to be determined and also calibrate the continuous real-time soil moisture sensors (Table 1). Soil sampling was then included as part of the 3 monthly sampling program to capture gravimetric soil water content profiles at different stages of the wet/dry cycle. The first profile was also sub sampled for analysis for EC, pH, nutrients, carbon and metals.

Water content sensors (ECH$_2$O-EC10) were installed in March 2008 and 'Jet-Fill' Tensiometers (0-100 kPa) with pressure transducers (GT3-420) were installed in May 2008 beneath RB4, to record real time water content and suction in the unsaturated zone (Fig. 6). The base of the sensors were installed at the depths: 1, 1.8 and 2.4 metres and were influenced by operation of RB4. Porosity of 0.3 and bulk density of 1.8 g/cm^3 were assumed to calculate gravimetric water content, prior to calibration.

In March 2008 three shallow piezometers at 1.0 (P4 100), 1.8 (P4 180) and 2.4 (P4 240) m depths were installed with the intention to monitor water level and water quality beneath RB4 (Table 1(a)). Piezometers consisted of 40 mm PVC slotted over the bottom 30 cm interval, which was covered with gauze stocking and taped above the slots. For each depth a hole was augered 50 mm below the final target depth of 2.4 m to take into account the PVC end cap. The PVC pipe was installed then backfilled around the slots with 5-10 mm washed gravel. Bentonite was used at the top of the slots to block preferential flow and the hole was then re-filled with native soil. An attempt to sample water in P4 100, P4 180 and P4 240 was made during the first field campaign 17-24 September 2008, but the wells were dry. Subsequently six further piezometers were installed in July 2009, three beneath RB4 and three beneath RB2 at depths of 0.1, 0.2 and 0.5 m (Table 1(b)). The completion method was as above except that each piezometer casing was capped more than 0.5m below the slotted interval and the annulus backfilled with bentonite to near the base of the slots in order to hold the piezometers in position and to allow samples to be collected if the slotted interval became saturated. The sampling of the shallow wells screened at the depth <1 m beneath RB 2 and RB4 was undertaken during the second field monitoring campaign on 18-25 July 2009. The piezometers located at the depth 1.8 m and 2.4 m underneath RB4 i.e. P4 180 and P4 240 (Table 1(a); Fig. 7) were always dry.

Three pressure vacuum soil water samplers (model 1920F1) were installed during the initial site set up in May of 2008, at depths of 1, 1.8 and 2.4 m in RB4. Sampling of soil water was carried out during the intensive quarterly sampling campaigns during a wet/dry period. However, only samples from the 1 m depth sampler were able to be collected. During the field campaign in July 2009 and soil water sampler was also installed in RB2 to a depth of 1 m and samples collected.

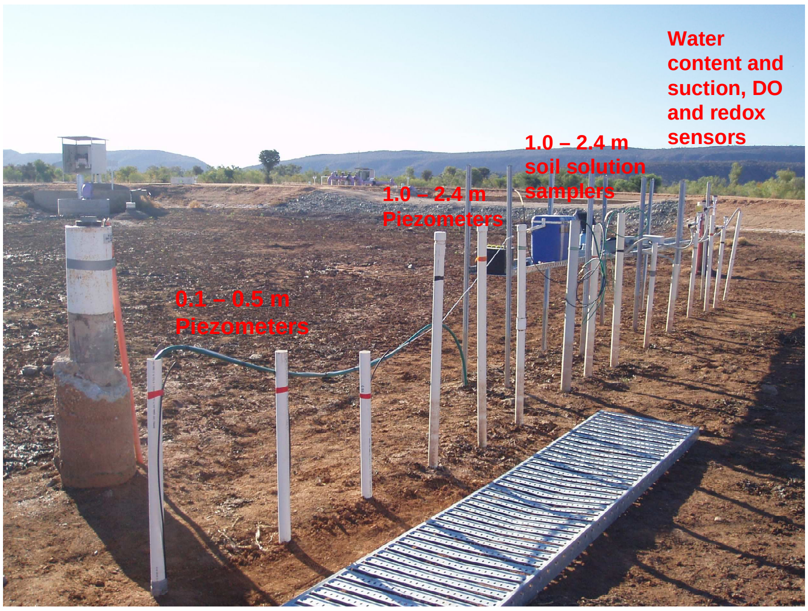


Figure 6 Observation well RN17948, shallow piezometers, suction cups, tensiometers, soil water content sensors, dissolved oxygen and Eh sensors located in RB4

Overall there were over 20 deep wells in which both water level and water quality could be monitored. Twelve of those wells were regularly sampled by Power and Water Corporation (Table 1(a)). Water level is monitored either continuously or once a week. Most of the wells were screened in the saturated zone at a depth of 20-30 m. Six wells were screened at a depth of less than 18 m (Fig. 5). All these were dry under natural conditions and some became wet while operation of the site was taking place.

Source water for quality monitoring was collected both at the DAF and AZRI sites. There were several sampling points of treated water at the DAF site (SAA045, SAA060, SAA080, SAB050 and inflow of the pipe connecting both sites - SAR080). Prior to the operation of the site, reclaimed water was sampled there. At the AZRI site, water was sampled at the outflow of the pipe (SAR100). Continuous real-time monitoring of electrical conductivity (specific conductance), turbidity, pH, and chlorine residual was measured in-line as well. Temperature, electrical conductivity, dissolved oxygen, redox potential and turbidity of source water discharged to RB4 were also measured. Sensors were in a shed located next to the inflow pipe, with water only pumped to the sensors from RB4 when triggered by a water level sensor installed in the pond.

In addition to regular sampling, dissolved oxygen (DO) and redox potential (Eh) were continuously monitored in RB4 (0.1 m above the basin base) and in the unsaturated zone at depths ranging from 0.7 - 1.0 m, 1.5 - 1.8 m and 2.1 - 2.4 m. (Table 1; Fig. 7). Dissolved oxygen sensors (US patent 5723769) were made up of a dissolved gas sensor encased within a silicone membrane diffusion cell (Patterson and Davis, 2008). Each sensor was 20 cm long and 3 cm in diameter, and when connected to the SCADA through a converter produced an output range of 0-12 mV. After an access hole was augured, sensors were buried (deepest first) at the specified depths and backfilled with the augured soil. Redox potential sensors consisted of pairs of single wire platinum electrodes connected to the SCADA via a converter to provide a voltage output. A 2.5 m hole was augured and 40 mm diameter slotted PVC was installed with an end cap at the base. The reference redox electrode (Ionode IJ14, AgAgCl) was installed in the base and backfilled with sand to ensure ‘wet/moist’ conditions were maintained to complete the ‘battery bridge’ and provide a record of comparative changes in voltage recorded by the platinum electrodes. Each pair of platinum sensors were then installed in order of depth 2.3 m (slotted 2.1 – 2.4 m), 1.7 m (slotted 1.5 – 1.8 m) then 0.9 m (0.7 – 1.0 m) and backfilled with fine sand over the slotted interval to ensure maximum connection with surrounding profile. Between each layer a 20 cm ‘plug’ of bentonite was added to minimise preferential flow between monitoring depths.

Table 1(a) Instrumentation established and sampling undertaken at the AZRI site in March and May 2008 and frequency of monitoring

Instrumentation/Sampling	No.	Well ID and depth (m) [a]	Frequency/ method of monitoring
Tensiometers (beneath RB4)	3	T100 (depth 1 m) T180 (depth 1.8 m) T240 (depth 2.4 m)	Continuous real-time monitoring Installed in May 2008
Dissolved Oxygen Sensors (in and beneath RB4)	4	DO +10 (above base) DO100 (depth 1 m) DO180 (depth 1.8 m) DO240 (depth 2.4 m)	Continuous real-time monitoring Installed in March 2008
Redox sensors (in and beneath RB4)	4	Eh +10 (above base) Eh100 (depth 1 m) Eh180 (depth 1.8 m) Eh240 (depth 2.4 m)	Continuous real-time monitoring Installed in May 2008
Soil solution samplers (beneath RB4)	3	SC4 - 100 (depth 1 m) SC4 - 180 (depth 1.8 m) SC4 - 240 (depth 2.4 m)	Water quality monitoring (quarterly and short term intensive sampling events) Installed in May 2008
Water content probes (beneath RB4)	3	10 cm long sensor WC100 (0.9-1 m) WC180 (1.7-1.8m) WC240 (2.3-2.4 m)	Continuous real-time monitoring Installed in March 2008
Shallow piezometers (beneath RB4)	3	P4 100 (0.7-1m) P4 180 (1.5-1.8m) P4 240 (2.1-2.4m)	100, 180 and 240cm piezometers installed in March 2008 Used during short term intensive monitoring and sampling campaigns
Piezometers located beneath and in the vicinity of RB4	6	2-6 m (#17944) 6-12 m (#17945) 12-18 m (#17946) 20-30 m (#17947, (#17948[b], #17949[b])	Continuous real-time monitoring (since 17 Dec 2008; before weekly manual measurement) Water quality sampling (quarterly) Water quality parameters being monitored are given in Appendix B. Existing piezometers
Other piezometers	6	18-22m (#17950) unsat/sat zone 20-30m (#17937, #17942, #17940, #17954 #18253)	Water levels (weekly) – manual measurement Water quality sampling (6 monthly) Water quality parameters being monitored are given in Appendix B. Existing piezometers
Soil sampling (RB4)	1	0-2.4 m RB4	Quarterly – water content Start of trial - chemistry

a – depth in metres below basin floor
b – sample for water quality is taken every 6 months

Table 2(b) Instrumentation established at the AZRI site in July 2009 and frequency of monitoring

Instrumentation	No.	Well ID and depth (m) [a]	Frequency/ method of monitoring
Soil solution samplers (beneath RB2)	1	SC2 – 100 (depth 1 m)	SC2 installed July 2009 Used during short term intensive monitoring and sampling campaign in July 2009
Shallow piezometers (beneath RB2)	6	P2 10 (0.05-0.15m) P2 20 (0.15-0.25m) P2 50 (0.45-0.55m) P4 10 (0.05-0.15m) P4 20 (0.15-0.25m) P4 50 (0.45-0.55m)	10, 20 and 50cm piezometers installed in July 2009 Used during short term intensive monitoring and sampling campaign in July 2009
Soil sampling (RB2, RB4 and RB5)		0 – 2.4 m depth RB2 and RB4 0 – 0.5 m depth RB2, BB4 and RB5	Full depth profiles (0 – 2.4 m) at start of July 09 field trip 0.5 m depth profiles at end of July 09 field trip

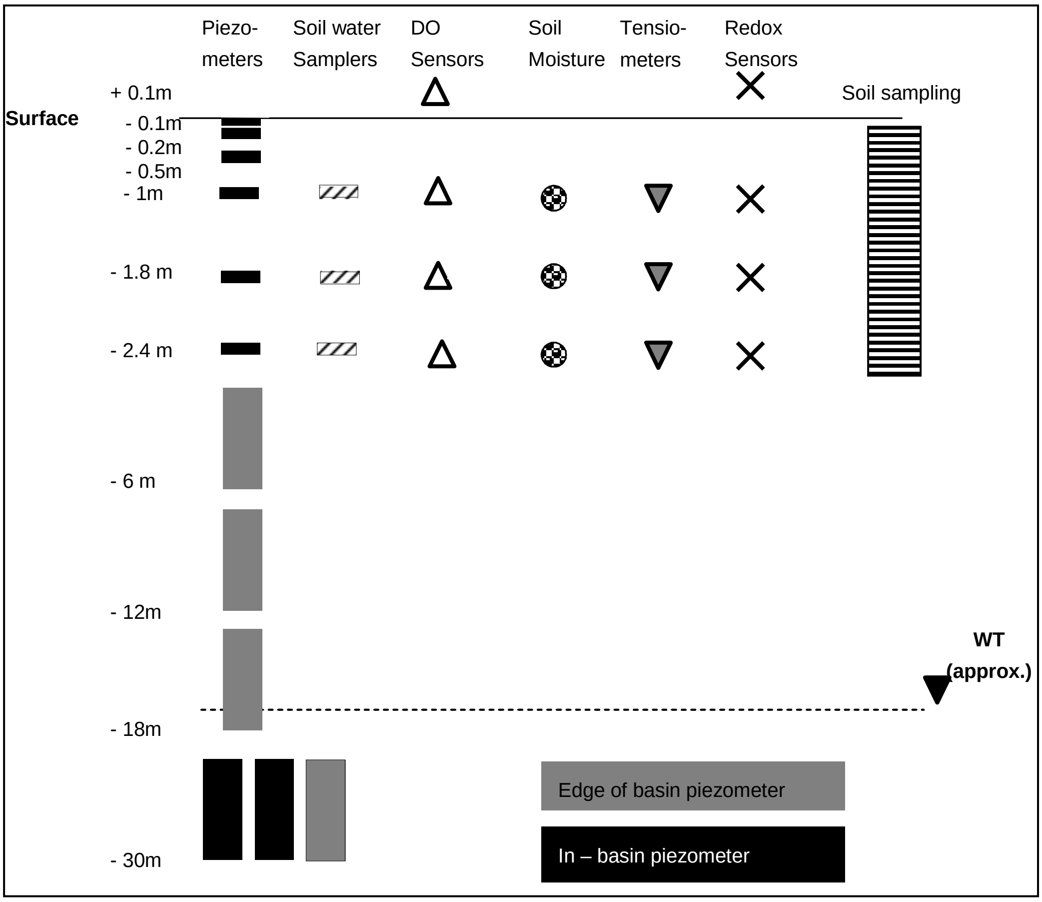


Figure 7 Schematic representation of water sampling locations at AZRI site in and adjacent RB4

Discrete water quality samples were collected at least quarterly from the deep wells located around the site (Tables 1(a) and (b)). Analyses of source water, perched water and groundwater samples were performed to determine concentrations of dissolved solids, total and dissolved inorganic constituents, and nutrients. Dissolved concentrations of major ions; calcium, magnesium, sodium, potassium, chloride, sulphate, and bicarbonate; trace metals; iron; and the organic compounds are discussed in this report to describe the effects of SAT operations. To date organic compounds were monitored only in source water. Faecal indicator bacteria were monitored in both recharge water and groundwater. The water quality parameters being monitored are summarised in Appendix C.

Sediments samples were analysed with a view to determining the behaviour of nutrients and metals in the soils during SAT operation. These studies were performed during the field campaign undertaken in July 2009. Prior to the wetting period in RB2 and RB4, sediment cores were collected to a depth of 240 cm in 10 cm increments. As each 10 cm sample was recovered, the soil was bulked and mixed in the field and sub-samples (10-20 g) were taken from each depth and kept in pre-weighed 50 ml of 2 M KCl container. In the lab, the sample was then shaken for a set period, re-weighed then left to settle prior to decanting the top 30 mL of extract for storage into 2x15 mL tubes. Samples were frozen for transport back to the analytical chemistry unit (CSIRO Land and Water, Urrbrae, SA) until ready for NO_3-N, NO_2-N, and NH_4^+-N analysis. Other subsamples were collected in chipettes for gravimetric water content. After the wetting phase a second set of 10 cm increment samples were collected to 50 cm depth from RB2 and RB4, plus a first set of samples from RB5 to 50 cm depth.

The citrate-dithionite method (Holmgren 1967, Rayment and Higginson 1992) was used to determine iron, aluminium and manganese content in the sediment samples. Most of reactive non-crystalline hydroxides and crystalline oxides should be dissolved with the use of this technique. The extraction itself and determination of Fe, Al and Mn in the extracts were performed in CSIRO Land and Water Laboratory in Urrbrae, SA.

The single ring permeameter (Perroux and White 1988) was used to determine vertical hydraulic conductivity values within RB2 and RB4 as well as the effect of development of the schmutzdecke on infiltration rates (Fig. 8). The measurements were performed under both wet and dry conditions during the extensive field campaign in July 2009. In total 24 measurements were carried out. Each measurement took 30-40 minutes.

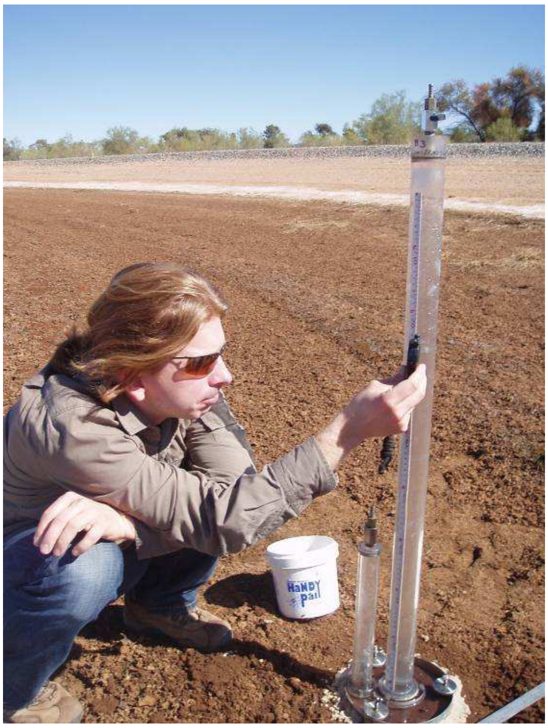


Figure 8 Measurement of hydraulic conductivity of RB2 using the single ring permeameter (Perroux and White 1988)

3.2. Attenuation of contaminants and saturation index calculations

Attenuation of contaminants can be deduced by calculating the expected composition based on conservative mixing of ambient groundwater and reclaimed water, and then comparing the result with the measured concentration in the water sample. The concentration of ion *i*, by conservative mixing is calculated as follows (Appelo and Postma 2005):

$$c_{i,mix} = f_{rec} * c_{i,rec} + (1 - f_{rec}) * c_{i,amb} \tag{1}$$

and

$$f_{rec} = \frac{c_{Cl,amb} - c_{Cl,sample}}{c_{Cl,amb} - c_{Cl,rec}} \tag{2}$$

Where ci is the concentration of i ion, frec the fraction of reclaimed water in the mixed water, calculated from the chloride concentrations (CCl) of mixed waters and end members, and subscripts mix, rec and amb indicate a conservative mixture, reclaimed water and ambient groundwater, respectively.

Any change in the concentration ci,react due to reactions then becomes simply:

$$c_{i,react} = c_{i,sample} - c_{i,mix} \tag{3}$$

Where ci,sample is the measured concentration in the sample. In this study chloride is assumed to be a conservative parameter during mixing. The percentage attenuation is calculated from:

$$attenuation\% = \frac{c_{i,react}}{c_{i,mix}} \times 100 \tag{4}$$

The saturation index is the log of the ion-activity product of a solution divided by the solubility product (Appelo and Postma 2005). Thus, the saturation index (SI) indicates the degree of disequilibrium of a solution with respect to a certain mineral; if a solution is at equilibrium with a given mineral, the mineral will not dissolve or precipitate. A negative saturation index implies that the solution is undersaturated for the mineral and has the potential to dissolve the mineral if it is present and kinetics allow. If the saturation index is positive, the solution is oversaturated with respect to a mineral and there is the potential for the mineral to precipitate. In this report SI was calculated for calcite ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$) using PHREEQC (Parkhurst and Appelo 1999).

4. INFILTRATION RATES

Precipitation and evaporation data (Fig. 9) were measured at Alice Springs airport weather station (No 015590; 23.80 °S, 133.89 °E) on a daily basis (9 AM). The station is located around 4 km south of the SAT site. Potential evaporation was measured with the use of an A-class evaporation pan. When calculating evaporation rate from the infiltration basins, an evaporation pan coefficient (K_p) of 0.8 was used.

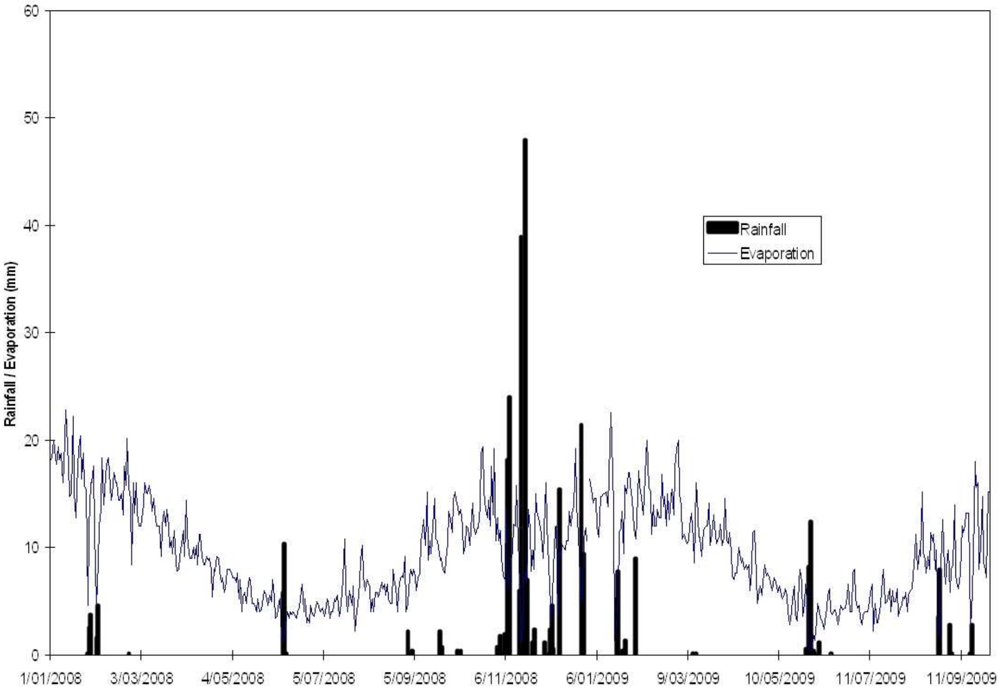


Figure 9 Daily rainfall (mm) and potential evaporation rates in Alice Springs Airport gauge station in 2008 and 2009

Most methods of infiltration calculation are based on the **mass balance equation** (Todd and Mays 2005):

$$I = Q + P - E - \partial S \tag{5}$$

Where I is the infiltration from the basin (m^3 or ML), Q is the inflow to the basin (m^3 or ML), P is the precipitation on the basin's surface (m^3 or ML), E is the evaporation from the basin's surface (m^3 or ML), and δS is the change in storage in a recharge basin. Some calculations using the mass balance equation are presented in Appendix D.

Due to the problems in estimation of the actual length of the wetting period, the calculation of infiltration was carried out using the **graphical regression** (Davis 2002). Real time continuous water quality data allow one to calculate infiltration rate during an individual wetting period. To do so, the graph of water level over time for each individual wetting period is required. Hydraulic head in a basin is measured as a function of time. Then, reading is corrected by using average values of rainfall and evaporation.

The advantages of this method are simplicity and high accuracy in determining infiltration rates during an individual wetting period. If a graph of water level over time is plotted, it is relatively straightforward to conclude time when the basin has dried out. The storage of the basin is not required to perform the calculation. The disadvantages include calculation of cumulative infiltration over time.

If total infiltration during an individual wetting period is calculated then it can be compared with total amount of discharge water:

$$\alpha = \frac{I_w}{V_w} \tag{6}$$

Where α is the infiltration coefficient (-), I_w is the total infiltration over the wetting period (m^3), V_w is the volume of water discharged to the basin during the wetting period.

4.1. Infiltration from the basins

In total 177.8 ML of treated water was discharged via 5 recharge basins in 2009. It constitutes only 30% of the target annual infiltration (600 ML), but the recharge area was much lower that that recommended by Knapton *et al.* (2004). It must be emphasised that RB5 started operation on 13/12/2009 and only two wetting cycles were performed in 2009. From the beginning of the operation (3/06/2008) until the end of 2009, 317.4 ML of reclaimed water was discharged to the recharge basins and about 308 ML of water could effectively recharge the aquifer (Table 2). The contribution of individual basins in the total recharge in 2009 was as follows: RB1 – 28.0%, RB2 – 34.4%, RB3 – 24.3%, RB4 – 11.7%, and RB5 – 1.6%. Calculation of efficiency takes into account evaporation and rainfall. Average efficiency for the site is 0.96, which suggests that evaporation losses do not play an important role in the overall water balance. Low efficiency values for RB4 are associated with long wetting periods in the basin due to lower permeability soils.

Table 3 Summary of data of the recharge basins

Recharge basin	**RB 1**		**RB 2**		**RB 3**		**RB 4**		**RB 5**	**TOTAL^{+} /MEAN***		
Area (m^2)	1928		1880		1964		1868		2629	10269		
Period of operation	3/06/08 31/12/08	1/01/09 31/12/09	7/06/08 31/12/08	1/01/09 31/12/09	3/06/08 31/12/08	1/01/09 31/12/09	11/7/08 31/12/08	1/01/09 31/12/09	13/12/09 31/12/09	2008	2009	2008 and 2009
Inflow (days)	96	110	106	116	84	80	55	49	5	85*	89*	174*
Inflow (ML)	43.2	49.7	41.4	61.1	39.1	43.3	18.9	20.8	2.9	142.6^{+}	177.8^{+}	320.4^{+}
Calculated infiltration (ML)	41.8	47.8	39.6	59.0	37.5	42.1	14.8	19.3	2.8	136.5^{+}	171^{+}	307.5^{+}
Hydraulic loading (m)	21.7	24.8	21.1	31.4	19.5	21.4	7.9	10.4	1.1	17.6*	22.0*	39.6*
Efficiency	0.97	0.96	0.96	0.97	0.96	0.97	0.78	0.92	0.97	0.96*	0.96*	0.96*
Inflow rate (ML/d)	0.46	0.45	0.39	0.53	0.47	0.54	0.34	0.42	0.58	0.41*	0.49*	0.45*
Inflow rate (m/d)	0.24	0.23	0.21	0.28	0.24	0.29	0.18	0.23	0.22	0.22*	0.26*	0.24*

$^{+}$ sum of all RBs

* mean values exclude RB5

The slopes of the cumulative inflow values (Fig. 10) also indicate the efficiency of basins in recharging reclaimed water. RB3, RB1, and RB2 received water at high rates in the first two months of operation and subsequently intake rates stabilised at similar values but in reverse ranking. However discharge to RB4 has remained relatively low and relatively constant over the entire period. Given the initial variation it is recognised that a longer period of record is required to comment on the performance of RB5.

Infiltration rate calculated using the regression method is presented together with cumulative inflow and water level for the recharge basins with respect to time (Fig. 10). The average infiltration rate inferred from the slope of the line representing the cumulative inflow appears to be relatively constant in each basin. However from the regression method calculated on occasions when the basins were draining, a slight decline in infiltration rate appears to occur in April 2009 following the increase in January-February 2009. The decline observed in April 2009 might be due to the change in source water quality or algal bloom, but evidently, in spite of subsequent regression infiltration rates being less than before April 2009, the average infiltration rate appears not to have deteriorated.

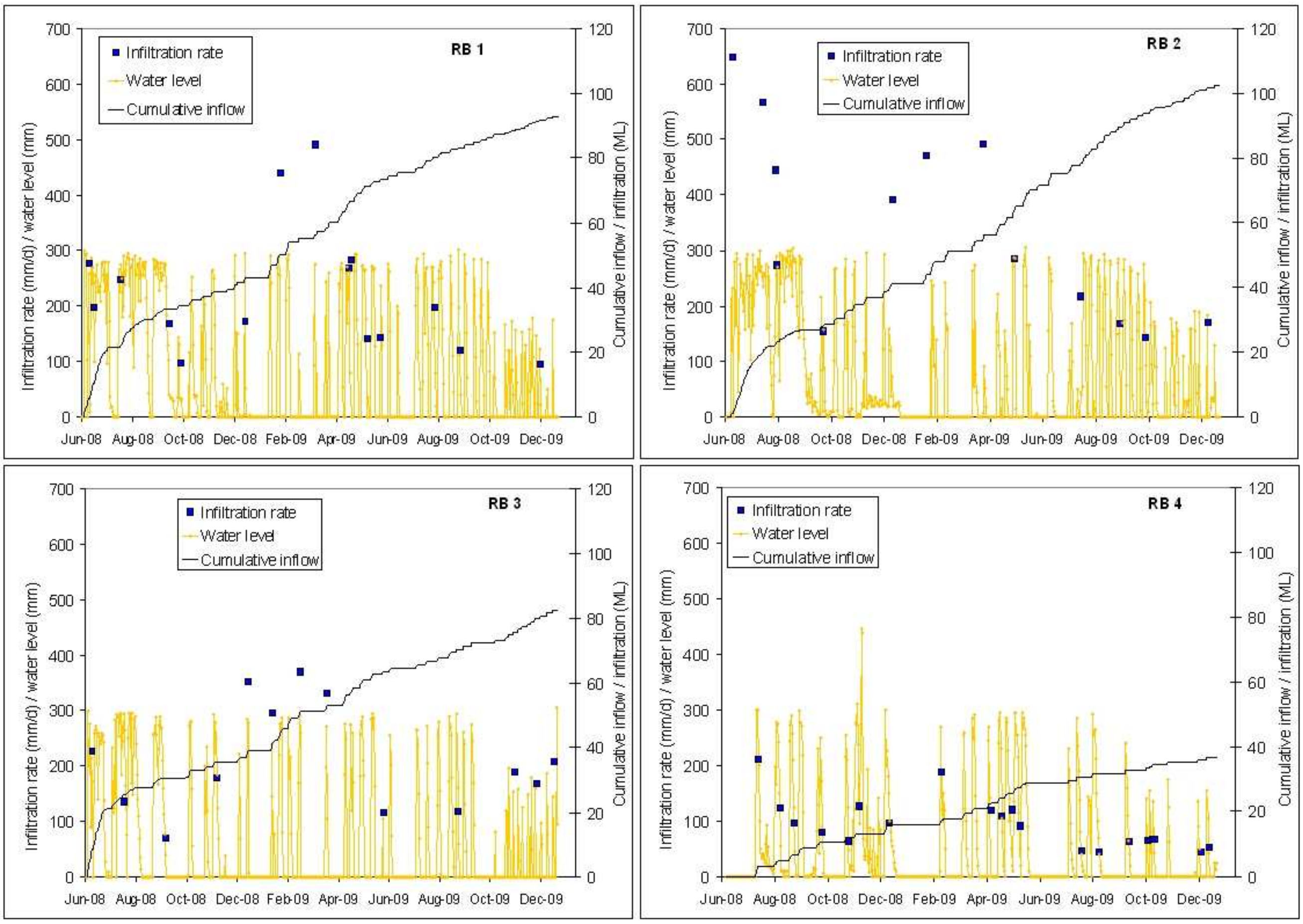


Figure 10 Cumulative inflow, calculated infiltration, and water level in RBs in the period June 2008-December 2009

In RB1 infiltration rates varied from 95 to 500 mm/d. Surprisingly low infiltration rates which decline from 275 to 105 mm/d were encountered at the beginning of the operation, which reflects very long lengths of the wetting period at that time (> 20 d). Much higher infiltration rates are documented for the beginning of 2009 after drying and a long period of desiccation of the basin bottom. From April 2009, however, the regression-calculated infiltration rates declined during sustained intermittent recharge. At the beginning of December 2009, these “instantaneous” infiltration rates had dropped to ~100 mm/d.

RB2 exhibits the highest permeability among the infiltration basins, which is evident by the largest amount of reclaimed water discharged to this basin. Nevertheless, continuous operation in the second half of 2008 resulted in sharp declines in infiltration rates. Recovery of infiltration rates was observed in the first quarter of 2009, but further declines are apparent from the middle of April. However, minimal regression-calculated infiltration rates do not drop below 150 mm/d.

RB3 is characterised by lower infiltration rates compared with RB 1 and RB 2 and in December 2009 it exhibited its highest infiltration rate. The lower initial infiltration rate is due to existence of fine grained deposits beneath the basin. The continuous low permeability layer (silty sand, sandy silt) was encountered in the southern part of the site. The different hydraulic properties between RB2 and RB3 are probably due to local heterogeneity including higher contribution of clay fraction beneath RB 3. In October 2009 the bottom of RB 3 was cleaned with the aim of enhancing infiltration rates. The cleaning included removal of 100 mm of material by using a front loader followed by ploughing of the basin to a depth of approximately 200 mm. This operation resulted in the increase in infiltration rate up to 200 mm/d the highest of all basins at that time.

RB 4 reveals much lower infiltration rates than all other basins. At the beginning of the operation the rates were up to 215 mm/d exhibiting exponential decline to about 70 mm/d. The lack of discharge to the basin in the period from 15/12/2008 to 10/02/2009 resulted in the partial recovery of hydraulic properties. Since February 2009 the infiltration rates have declined again and at the end of 2009 reached as low as 50 mm/d. The cleaning of this basin performed in October 2009 did not recover hydraulic properties of the basin. This is an indication that the high proportion amount of fine grained superficial sediment beneath RB4 limits infiltration from this basin and allows rapid development of the schmutzdecke. The values of hydraulic conductivity (calculated from infiltration rates with the assumption of a unit hydraulic gradient) coincide with hydraulic conductivity values of clogged sandy silt (loam) obtained during column tests (Knapton *et al.* 2004, Mucha 2004).

The SAT was primarily constructed with a view to preventing discharge to Ilparpa Swamp of excess treated wastewater in winter, but the results indicate that most infiltration occurred in the hotter early months of 2009. There was substantially reduced recharge in June and July 2009 due to very long drying periods in overcast cool conditions and low evaporation rates (<5 mm/d) compared with summer (>10 mm/d). Hence, operating recharge basins well in advance of the cooler winter months will be necessary to draw down storage within the treatment plant to accommodate the winter excess.

A number of ring permeameter tests in RB2 in July 2009 revealed large spatial differences in measured hydraulic conductivity. While infiltration rates were variable, it was evident that permeability increased when the schmutzdecke was dry and cracked and that this was more important than the presence or absence of schmutzdecke when the surface was still moist. Interestingly the top of a furrow had the highest infiltration rate in the nearly dry basin, even though schmutzdecke was present suggesting that the additional drying available to ridges may facilitate higher infiltration rates on rewetting. Detailed data are presented in Appendix E.

Inflow rate to the basin over time is a close approximation to the infiltration rate. Since it takes around 0.2 day for inflow to fill the storage of the basin inflow rates within this time period are not indicative of infiltration rate (Appendix F). Hence values of inflow rate in excess of 2 ML/d are regarded as not representative of infiltration rate. The values of inflow rates from RB1 and RB4 are shown in Appendix F with respect to the duration of the wetting cycle, and also with respect to the duration of drying immediately prior to wetting and in relation to turbidity of source water. These data indicate that infiltration rates declined marginally but not consistently with wetting duration for periods of 1 to 5 days. It also showed that for inflow rates less than 2 ML/d the duration of drying before the wetting cycle appeared not to have a significant affect on infiltration rates. Finally they also show a trend towards lower infiltration rates as turbidity increases. Interestingly Appendix F also shows that the highest turbidity (up to 120 NTU) occurs during the first fill of basin RB4, possibly due to suspended solids settling in the pipeline during rest periods and becoming mobilised when flow recommences and persisting for up to 0.2 day or more. Reducing the turbidity of source water would help overcome this issue. As a first strategy, maintenance of stable flows in the pipeline to the basins could assist in avoiding such slugs at the start of a wetting period.

4.2. The effect of infiltration on the unsaturated zone

In occasional cores of the vertical profile beneath RB4, the values of gravimetric water content increased with depth to 1.7 m bgs and decline from 1.7 m to 2.4 m (Fig. 11). This effect is associated with the soil texture at the location of the moisture meter, which reveals the occurrence of fine sands containing silt to a depth 1.7 m, and much coarser material below this depth (Appendix A).

The electronic sensor located at a depth of 1.8 m stopped working during the field trip of July 2009 (Fig. 12a). An apparent increase in gravimetric water content from 30% to 55% between August 2008 and July 2009 is not supported by the laboratory gravimetric soil moisture contents at this depth of between 5 and 27%. Similarly measurements at 2.4 m that are more responsive to wetting and drying cycle range between 21% and 33% which are considerably higher than the laboratory gravimetric values at this depth of between 2% and 8%. That is the numerical values of the logged moisture content are inaccurate for the 1.8 m deep sensor and values for 2.8 m need rescaling based on data determined in the laboratory.

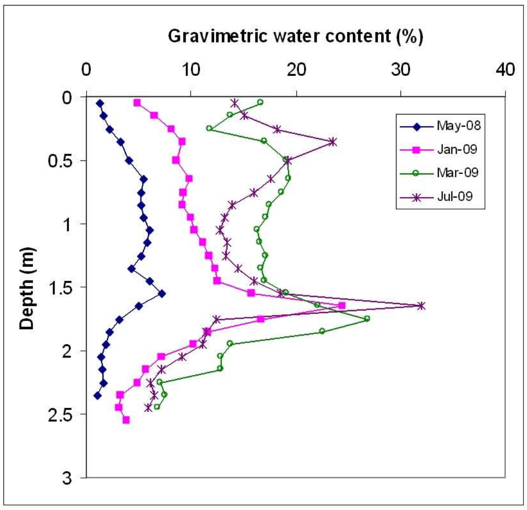


Figure 11 Laboratory measured gravimetric water content beneath RB4

As a soil dries out more negative values in pore pressure (suction) are typically recorded at 1 and 1.8 m in the first year. On wetting the pore pressure becomes less negative. The lowest range of the tensiometers being used is -100 kPa. The readings indicate that the prolonged wetting results in the decrease in suction from -100 to -60 kPa, although full saturation (0 kPa) has never been recorded by transducers at depths of 1.0, 1.8, and 2.4 m (Fig. 12b). The sensor located at the depth 1.8 m appears to indicate the deterioration of drying rate over time which potentially could be attributed to clogging development. The sensor situated at the depth of 2.4 m continuously indicates very dry conditions. There was no response of the tensiometers to the prolonged wetting events in April-May 2009. The tensiometers reading are inconsistent with laboratory-determined and inferred gravimetric water contents and suggest that the tensiometers were not providing reliable results and certainly this is the case beyond April 2009.

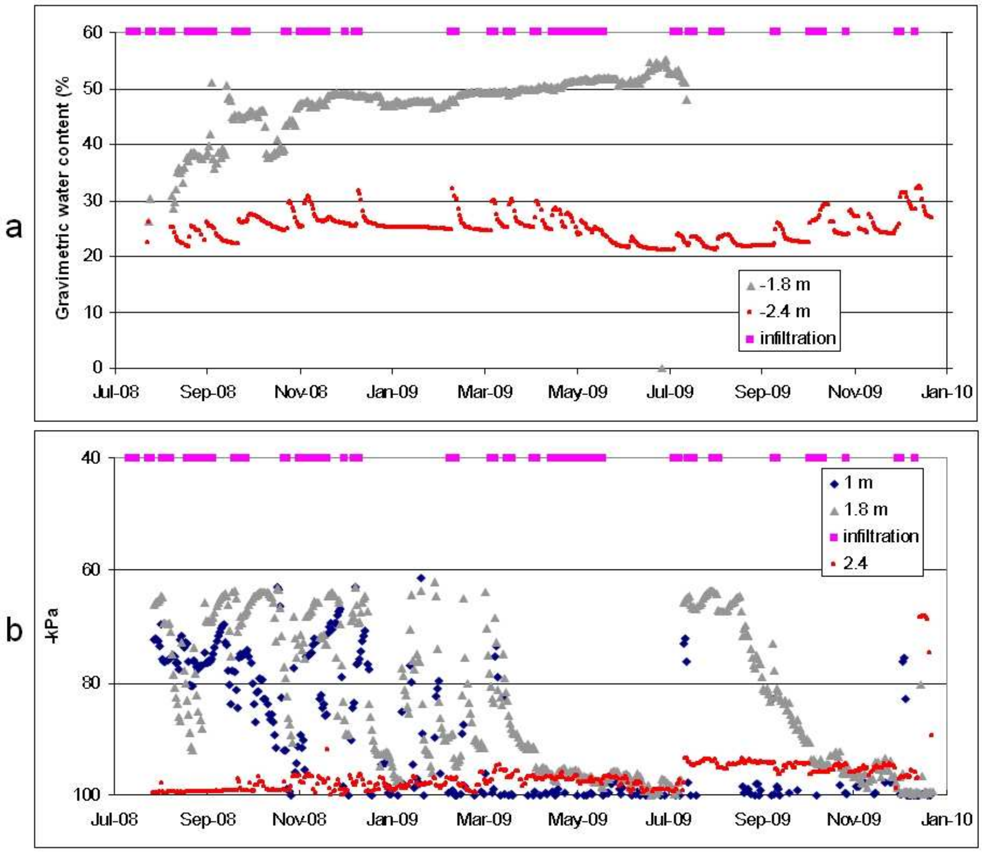


Figure 12 Inferred gravimetric water contents (a) and pore pressure (b) beneath RB4 at different depths in the period of July 2008 – December 2009. Pink dots pertain to the wetting period in RB4

4.3. The effect of infiltration on the saturated zone

Recharge basins were located at the elevation of 553 m AHD and the unsaturated zone had an initial depth of 18-20 m. A quick response to infiltration events in RB4 was observed in water levels of three of the deep wells (RN 17947, RN 17948 and RN 17949; Fig. 13) affected by operation of RB4. In 2009 the biggest loading to RB4 (4.8 m; 8.96 ML) took place in the period 15/03-23/05 which resulted in the upward movement of water level by 0.9 m. Following a wetting event the water level declined to its initial value in a similar timeframe to the rise, at about 20 mm/d. Water levels in deep wells further from the basins did not show any affect the SAT operation.

The most conspicuous increase in water level was observed in RN 17950 which is screened at the depth 16-22 m (531.1-537.1 m AHD, Fig. 14), slightly shallower compared to most wells. This well is located further away from RB4, and as such the influence of other basins may be large. This well was observed during the test infiltration trials in 2003-2004 to respond quickly to wetting of basins A and B (each 6x6 m) and water table rose up to 3 m (Knapton *et al.* 2004). During the operation of the SAT site the rise was up to 6.7 m compared to the initial condition which could be related either with the existence of perched water table which would confirm large heterogeneity of the site or with improper construction of the well (with perched water in shallow sands percolating down the outside of the casing). The latter seems to be likely since there is no other well around exhibiting similar behaviour. A decline in water level prior to site operation may suggest a datum error. For this reason all well data need checking.

The highest recorded elevation in RB 17950 was 542.1 m which means that if perched water table occurs at this spot, the depth to water has continually exceeded 10.9 m. RN 17945 responses very quickly to infiltration in RB4, which is reflected by the absence of water in the well unless RB4 is wet (Fig. 14). The highest water level recorded in RN 17945 was 546 m so the shallowest depth to a perched water table indicated from this well was 8 m. Well RN 17944 completed with a monitoring interval from 2-6 m below the basin floor remained dry on all occasions when manual measurements were taken.

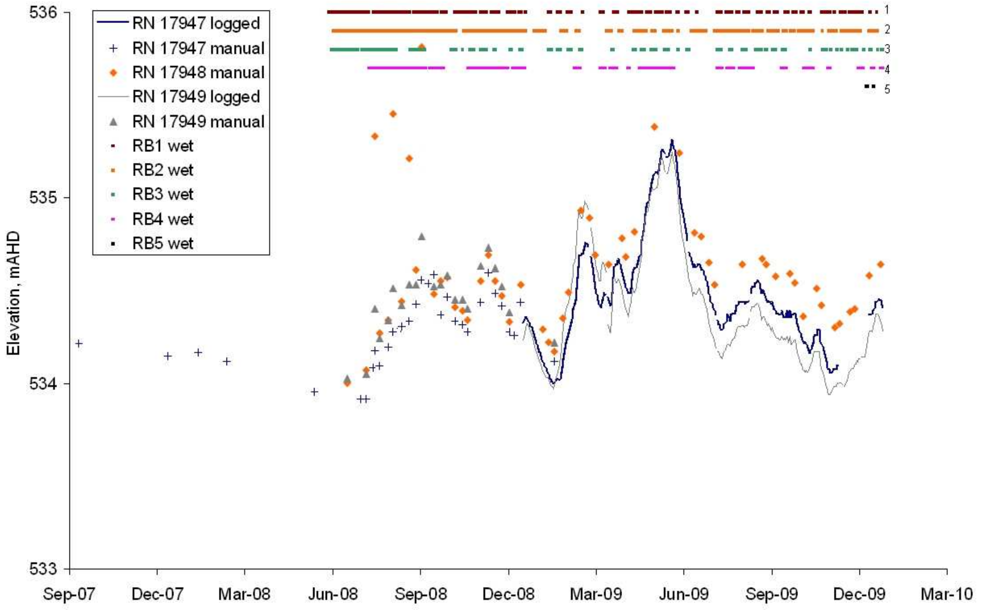


Figure 13 Water level elevation in the observation wells RN 17947, RN17948 (screening interval 522.9-532.9 m), and RN 17949 (523-533 m) that are situated in the vicinity of RB4. Elevation of recharge basin is 553 m AHD. The bars represent wetting in RBs

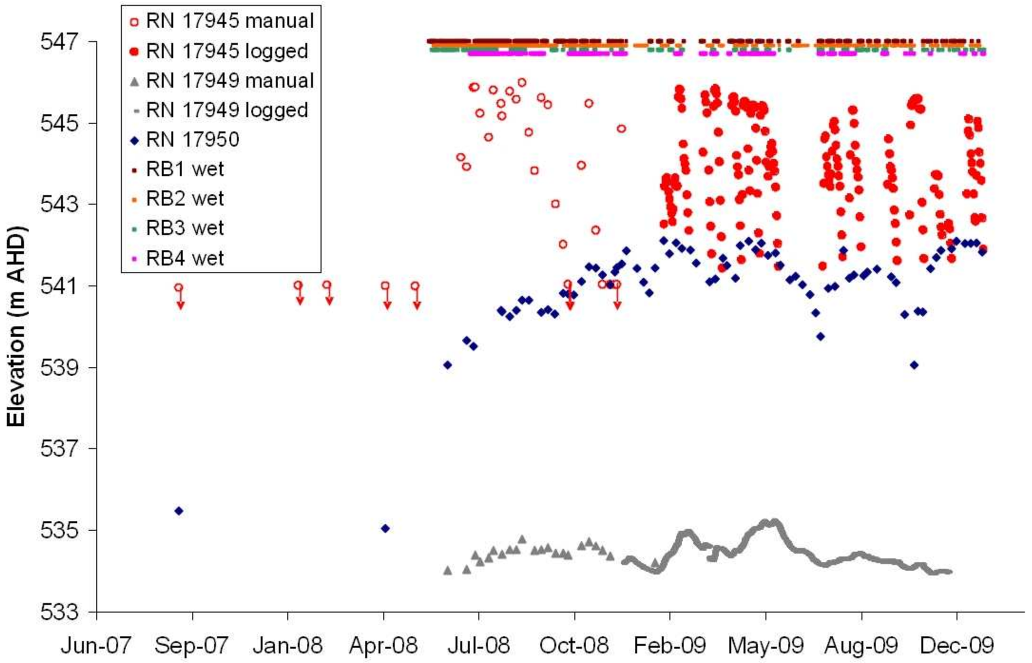


Figure 14 Water level elevation in the observation wells RN 17945 (screening interval 540.9-546.9 m) and RN 17950 (screening interval 531.1-537.1 m) situated on the eastern edge of the AZRI site. For comparison RN 17949 is shown. Elevation of recharge basins is 553 m AHD. Arrows indicate dry well. The bars represent wetting in RBs (note the bars eclipse the drying period when they are small)

The schematic map of regional monitoring wells is presented in Fig. 15. Water level in RN 17937 on the western side of RB2 looks very similar to water level in 17940 which is located on the eastern side of RB3 (Fig. 16). Since RB1 and RB2 have similar hydraulic loading histories and water table response is delayed it is not possible to attribute responses of wells to specific basins. The similarity between the wells implies that either the aquifer in the northern part of the site is hydraulically connected or the spreading from RB1 and RB2 is relatively uniform. In RN 17938, an upgradient well not influenced by site operation, the regional decline in water table was 0.5 m during the observation period of 18 months. Well 17939 located further upgradient is most affected by the Todd river and does not exhibit behaviour typical of the Outer Basin aquifer. The slight increase in water level observed in RN17942 in June 2009 might suggest a response to site operation (Fig. 16).

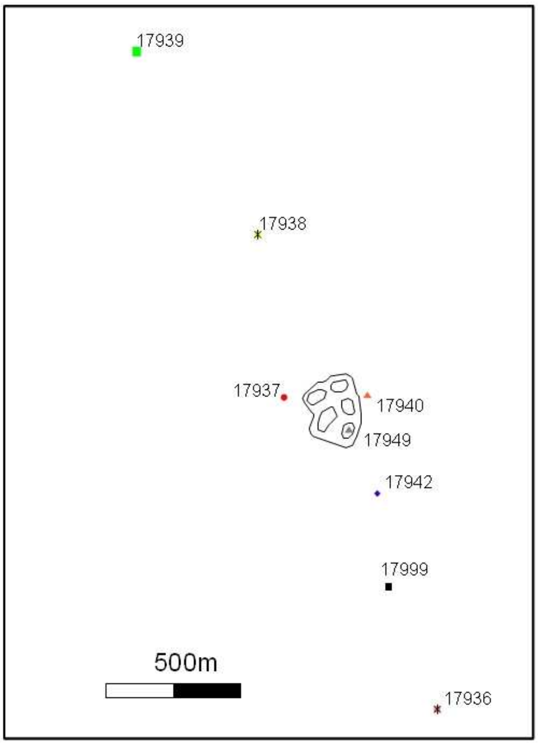


Figure 15 Location of the monitoring boreholes located in the area of AZRI

Very little increase in water levels have been observed to the end of 2009 in the well located downstream more than 1 km from the site (RN 17936; Fig. 16). The decline of 0.2 m in both RN 17999 at 500 m and RN 17936 at 1000 m from RB4 demonstrates that the recharge from SAT basins has not been sufficient to offset a regional decline in water table.

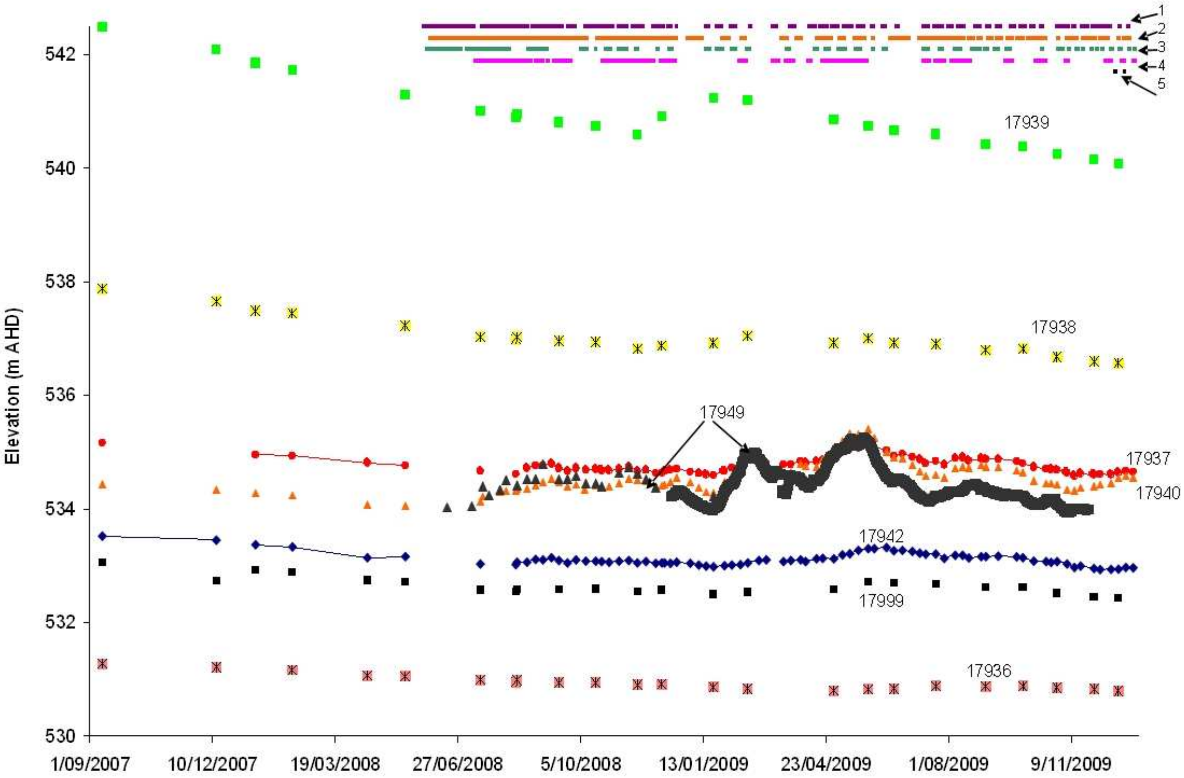


Figure 16 Water level elevation in the observation wells on local and regional scale. Elevation of recharge basins is 553 m AHD. The bars represent wetting in RBs

The spatial fluctuations of water level are shown in Fig. 17. Apart from RN 17950, the recharge mound is very low. The increases in water level up to 1 metre indicate that the watertable depth in the aquifer has continually exceeded 17.5 m. Nevertheless, perched water table beneath RB4 measured between 6-12 m depth at RN 17945 is situated 8m of base of RB4.

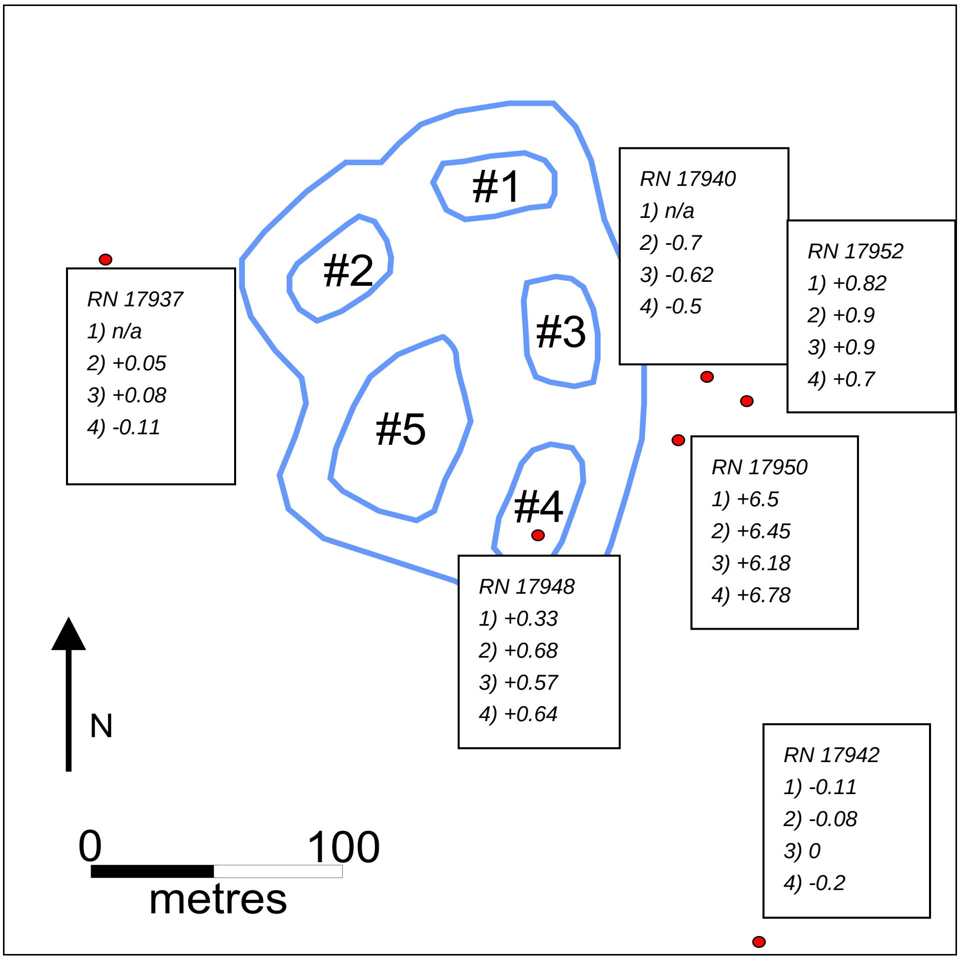


Figure 17 Differences between water level elevations in the deep wells with respect to initial condition (May 2008). Measurement taken in: 1) Dec 2008, 2) Apr 2009, 3) Sep 2009, and 4) Dec 2009

4.4. The effects of infiltration under RB4 and RB2 (July 2009)

The effects of infiltration from recharge basins on soil moisture content and suction cup and on water levels, are analysed on the basis of data gathered during the field campaign performed on 17-24 July 2009. Cumulative inflow, water level, and calculated infiltration rates in RB2 and RB4 are presented in Fig. 18 and 19. The infiltration rate of RB2 and RB4 were calculated from the rate of decline in basin level after the inflow valve was shut. These values were measured at 216 and 295 mm/d in RB2 and 45 mm/d in RB4 which overlies less permeable soil.

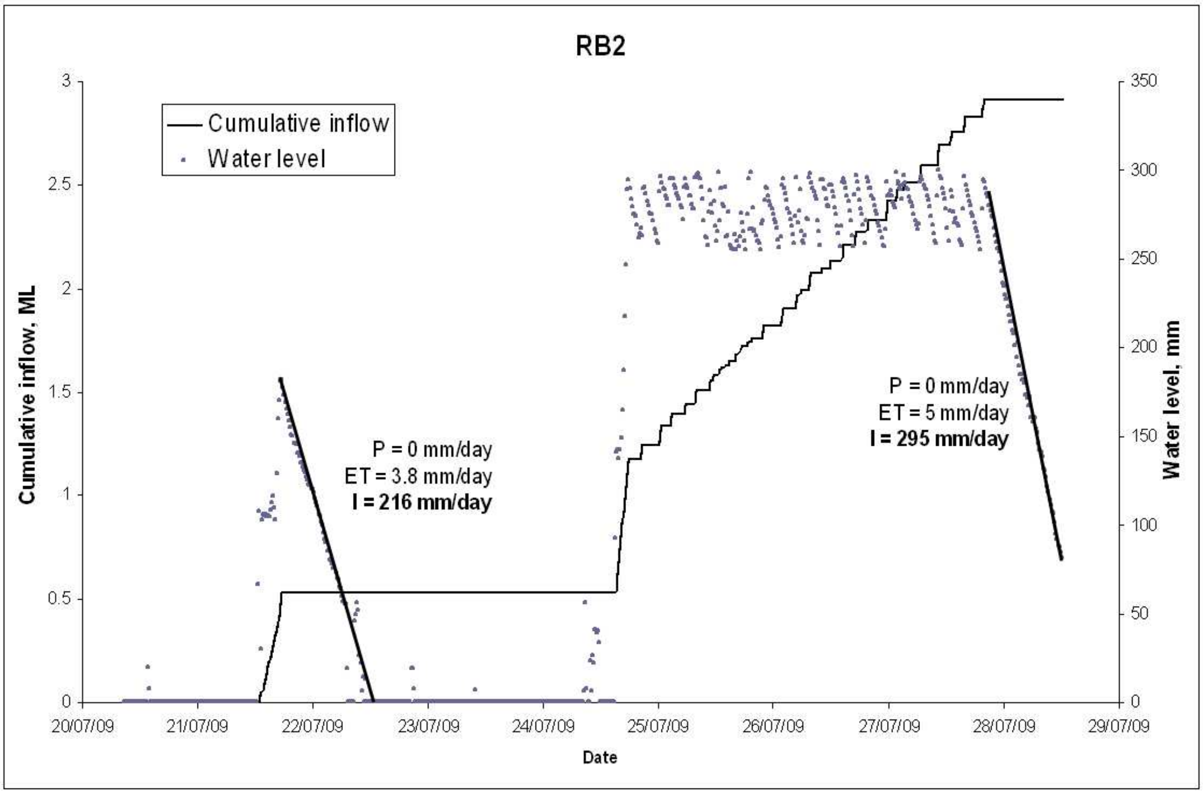


Figure 18 Calculation of infiltration rate in the RB 2 for the period 21-22/07/2009 using the regression (falling head) approach

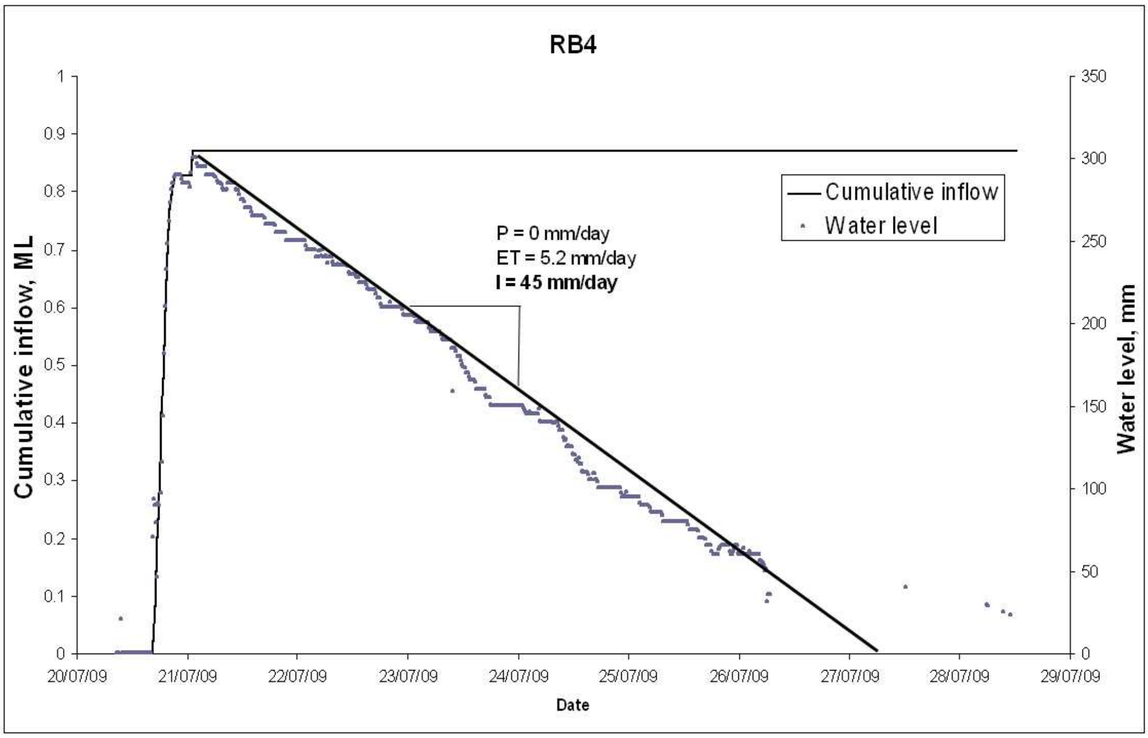


Figure 19 Calculation of infiltration rate in the RB 4 for the period 21-28/07/2009 using the regression (falling head) approach

The infiltration from RB 4 taking place on 21-27 July 2009 resulted in a small increase in water content in the unsaturated zone at 2.4 m from 22 to 23.5%. Apparent water content measured at the depth of 2.4 m changed 1.8 days after recharge commenced. It also started to decrease again about 3.8 days from the start of recharge (Fig. 20). On 22-24/07/2009 water level increased ~0.8 m/d. Subsequently (24-29/07/2009), a gradual decline in water level (~0.15 m/d) was observed. The rate of decline increased (up to 0.5 m/d) once the basin dried out completely on 29/07/2009. The response in apparent water content is not accompanied with suction measurements at this depth (Fig. 21). The variation is small and the very high suction is incompatible with moisture content suggesting that these probes are not operating correctly. Hence soil suction data is not discussed further in the report.

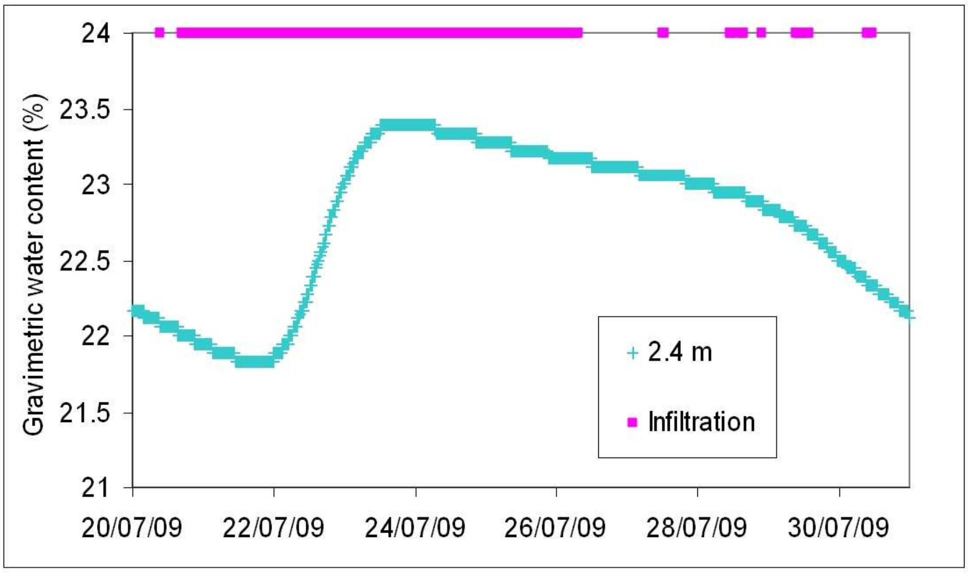


Figure 20 Inferred gravimetric water content beneath RB 4 on 20-31 July 2009 from the tensiometer at 2.4 m

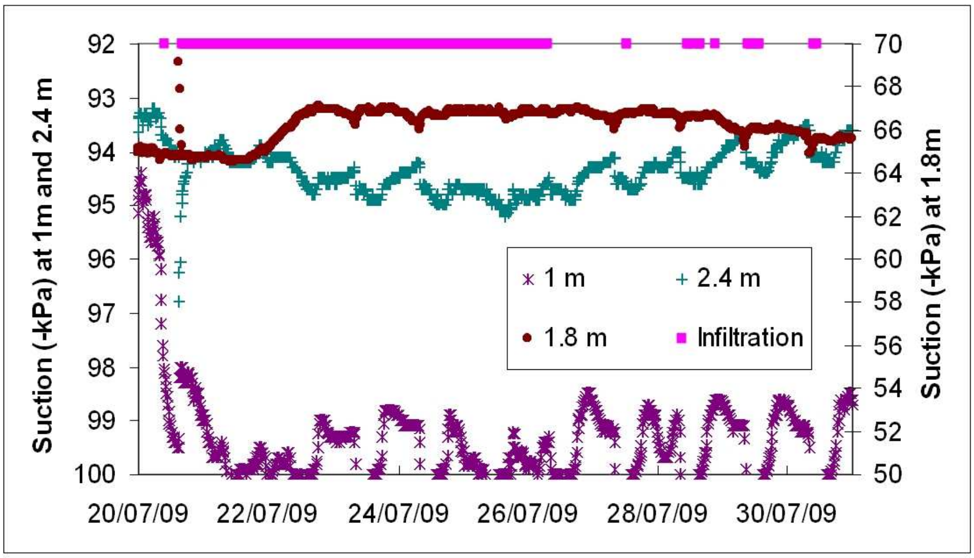


Figure 21 Suction beneath RB 4 on 20-31 July 2009

From among the shallow piezometers screened beneath RB2, only P2 50 (0.5 m depth) reacted to infiltration from the basin. The quick response of the piezometer combined with lack of response in shallower piezometers implies that there are preferential flow paths in the unsaturated zone and there is an infiltration rate-limiting layer beneath a depth of 0.5 (Appendix G), but that as the schmutzdecke formed rates of infiltration declined to less than the saturated hydraulic conductivity of this layer. A silty layer was encountered at a depth 0.7-1.1 m while augering the P2 50 on 17/07/2009.

There was a quick response to infiltration from RB4 in piezometers P4 10 (depth of 0.1 m) and P4 20 (depth of 0.2 m) while no perching was encountered in P4 50 (depth of 0.5 m). The profile 0.0-0.2 m was saturated only in the initial phase of wetting which indicate that saturation of this zone resulted in the increase in hydraulic conductivity and enhanced drainage of this zone or that schmutzdecke formation was very rapid (Appendix G).

RN17945 most conspicuously reacted to infiltration from RB 4 among the piezometers screened in the unsaturated zone (Fig. 22). It suggests the occurrence of a layer of low permeability somewhere in the depth interval of 9.5 to 12 m. The water level in RN 17945 started to increase about 3 days after infiltration of water in RB4 commenced. The water level returned to the initial position about 9 days thereafter. This well does not seem to be affected by the infiltration from the other basins. Fig. 23 presents water level elevation in the wells that are certainly influenced by the infiltration in the other basins.

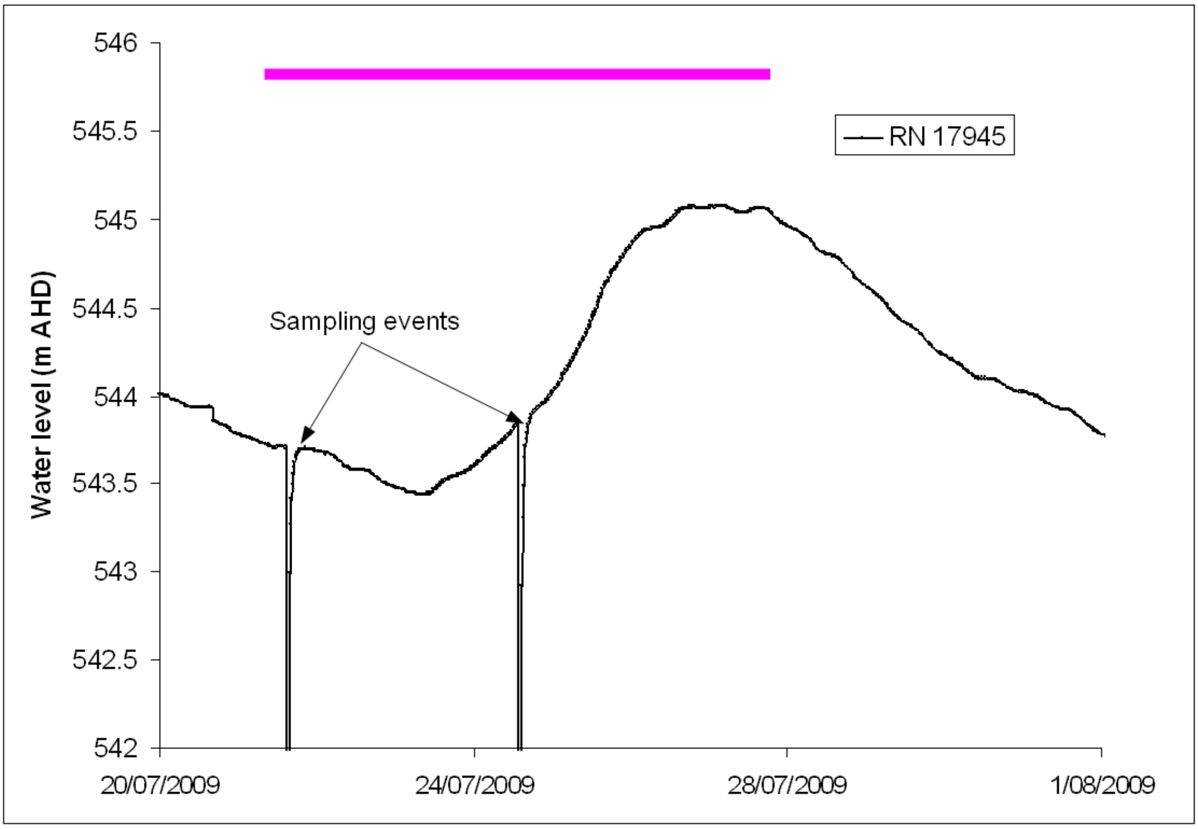


Figure 22 Water level over time in the well RN 17945 (screening interval 540.9-546.9 m). Elevation of recharge basins is 553 m AHD. Pink bar indicates infiltration from in RB4

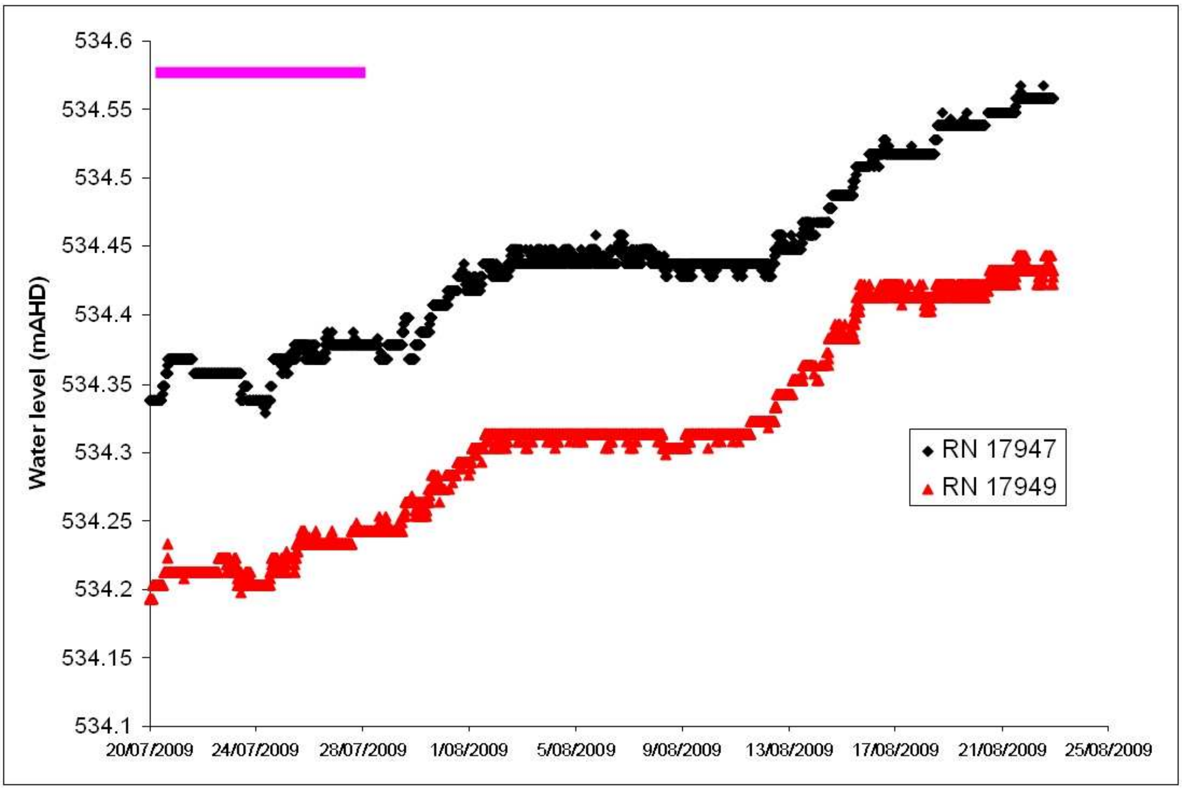


Figure 23 Water level in the wells RN 17947 (522.9-532.9 m) and RN 17949 (523-533 m) in the period of 20/07/2009-23/08/2009. Pink bar indicates infiltration from RB4

4.5. Quality of source water

Wastewater is pre-treated before it enters the recharge basins. Following primary treatment and retention in stabilisation lagoons pre-treatment involves coagulation using aluminium chlorohydrate combined with flocculation using cationic polymer. Next, pH is corrected using sulphuric acid to a target value of 7.5-8.0. Subsequently, the solids are removed with the use of dissolved air flotation. And finally, the reclaimed water is disinfected with chlorine.

Water quality parameters of reclaimed water are shown in Table 3. For comparison, reclaimed water from Bolivar wastewater plant used to feed the columns during the feasibility study (Knapton *et al*, 2004) is presented. Concentrations of TKN, ammonia, total organic carbon as well as turbidity and membrane filtration index (MFI) are much higher in DAF Alice Springs water compared to Bolivar DAF water used for the column study to assess clogging potential (Mucha, 2004). TOC was 60% higher, turbidity 400%, bicarbonate 90% higher, total N between 80% and 400% higher and membrane filtration index >100 % higher than water used in column studies. Exceptions to this trend among potential clogging agents were total phosphorus and sodium which were 20-25% and 67% respectively, of those used in Bolivar DAF water in the column studies. Relatively high concentrations of Ca^{2+}, Mg^{2+} and HCO_3^- in Alice Springs reclaimed water may be responsible for precipitation of carbonate minerals.

The source water produced by the Alice Springs DAF plant is characterised by the dominance of sodium among cations and bicarbonate among anions (Table 3, Appendix B). Average values for BOD and COD are 18 mg/L (n=18) and 44 mg/L (n=11). The low BOD:COD ratio indicates low degradability of organic matter. Temporal changes in selected parameters values as well as concentrations of selected compounds are shown in Fig. 24 and 25. Electrical conductivity of recharge water is in the range of 1500-1950 µS/cm (1624 µS/cm on average) and varies over time. Similar trends are observed with respect to Cl^- and B concentration and ammonium dominates the inorganic nitrogen species, although the concentrations are very variable. The concentrations are high compared with other (mostly American and Israeli) SAT sites. Concentrations of metals (not shown) in recharge water are very low mostly due to neutral pH of these waters. Oversaturation for calcite ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$) implies an affinity for precipitation of carbonates (Fig. 26). This process however may be strongly inhibited (Appelo and Postma 2005).

Table 4 Comparison of water qualities used for feasibility study and MAR activity in Alice Springs, NT and Adelaide, SA (mean or mean ± standard deviation)

Parameter (mg/L unless otherwise specified)	**COLUMN STUDY (Knapton *et al*, 2004, Mucha 2004)**					**Alice Springs**	**Bolivar**	
	Alice Springs			**Bolivar**				
	BLATH PK	**ROCK FILTERED**	**ROE CREEK**	**DAF-GLASS HOUSE**	**DAF-CONST TEMP RM**	**DAF** [a]	**DAFF Cycle 1** [c]	**DAFF Cycle 3** [d]
	Nov-03	**Nov-03**	**Nov-03**	**Nov-03**	**Nov-03**	**Jun-08 to Dec-09**	**Sept-99 to Dec-00**	**Dec-03 to May-04**
						avg n=12	avg n=17	avg n=15
Electrical Conductivity (uS/cm)	1635	1583	776	2490	2190	**1684 ± 144**	2352 ± 151	2000 ± 202
pH	7.88	7.90	8.86	8.62	8.04	**7.89 ± 0.11**	7.2 ± 0.3	7.9 ± 0.5
Turbidity (NTU)	72.1	43.2	0.45	0.65	0.60	**2.5 ± 8.6**	35.1 ± 55	1.8 ± 1.7
TSS	142	49	<1	5	2			
Total organic carbon	46.0	37.2	1.1	12.7	11.7	**19.2 ± 8.6** [b]	18.2 ± 2.4	12.6 ± 0.8
Dissolved organic carbon	21.8	17.8	1.2	12.1	11.1		16.7 ± 2.2	12.9 ± 1.1
Chloride	211	231	82	504	464	**234 ± 12.7**	444 ± 37.3	421 ± 47
Bicarbonate	521	580	327	249	229	**443 ± 45.6**	332 ± 47.8	162 ± 40
Sulphate	135	104	53.7	184	164	**140 ± 10**	203 ± 10.1	189 ± 29
Calcium	59.4	54.1	47.7	45.8	39.7	**48.9 ± 5.2**	44.8 ± 5.3	35.5 ± 3.5
Magnesium	33.8	31.9	24.7	47.8	41.0	**31 ± 2.5**	36.9 ± 2.4	33.8 ± 3.2
Potassium	21.6	21.0	6.2	38.0	32.0	**39.5 ± 2.5**	47.4 ± 4.0	39.5 ± 2.6
Sodium	237	244	96.6	346	302	**227 ± 3.5**	313 ± 20.4	290 ± 36
Ammonia - N	4.34	10.0	<0.005	0.699	1.09	**11.06 ± 5.9**	22.0 ± 7.6	0.15 ± 0.1
NO3+NO2-N	<0.005	0.683	<0.005	6.28	1.09	**1.76 ± 1.2**	1.5 ± 1.8	2.7 ± 3.5
Filt react P	7.22	4.22	0.009	4.51	4.23	**0.98 ± 0.56**	0.53 ± 0.51	1.3 ± 1.2
TKN-N	13.6	17.2	0.25	2.16	2.4	**13.5 ± 5.6**	25.2 ± 7.6	1.5 ± 0.2
N total	13.6	17.883	0.25	8.44	3.49	**15.26**	26.7	4.2
Phosphorus - Total	9.13	6.46	0.017	5.57	4.97	**1.25 ± 0.61**	1.01 ± 0.6	0.86 ± 0.9
Membrane Filtration Index (MFI)	9614	5881	21.67	35.77	43.7	**99.7 ± 41.9**	143.1 ± 87.5	na
MFI Coef. Of Variation %	16.6	13.1	8.9	4.9	10.1	**42.0**	61.7	na

[a] results from NT labs only; [b] analysis done at CSIRO laboratory only; [c] before activated sludge treatment connected; [d] after activated sludge treatment connected

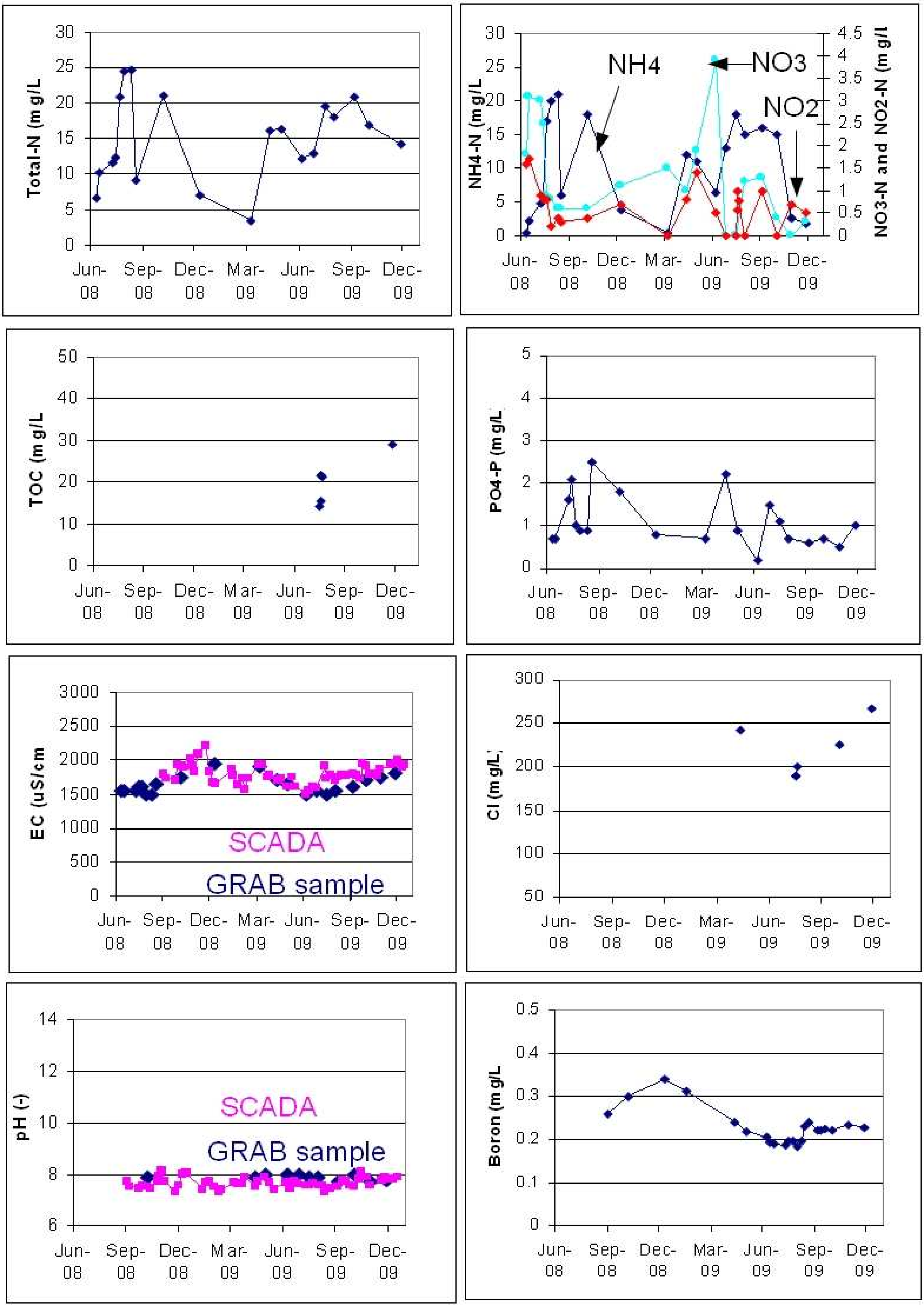


Figure 24 Temporal variations in concentration of selected compounds and pH in recharge water

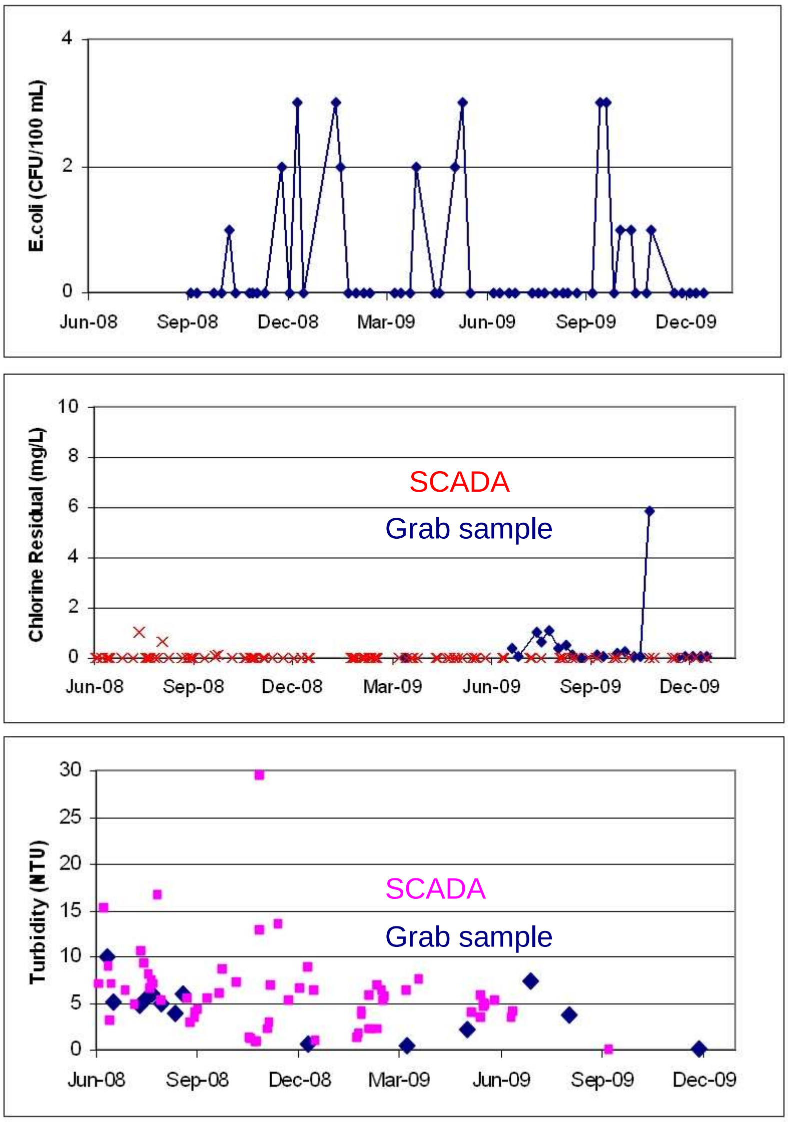


Figure 25 Temporal variations in selected quality parameters in recharge water

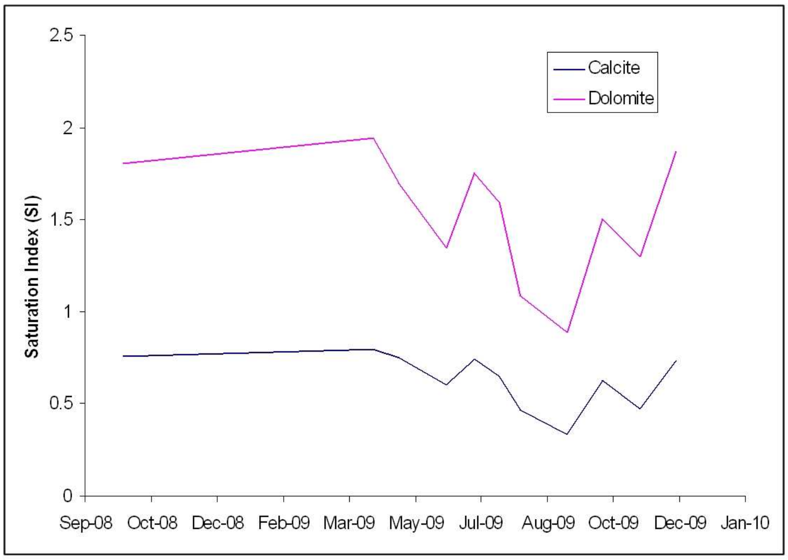


Figure 26 Saturation index (SI) of recharge water with respect to calcite and dolomite. Positive value indicates oversaturation for those minerals

4.6. Influence of water quality on infiltration rates

Average infiltration rates recorded in the SAT system are lower than anticipated during the feasibility study (Knapton *et al.* 2004). The analysis of the field trial with potable water indicated infiltration rates from 3 m/day (Basin A) to 10 m/day (Basin B) (Knapton *et al.* 2004). Basin A used for the feasibility study was located within the footprint of RB4 which has been found to have an average infiltration rate of 0.14 m/day. A geological cross-section showing the location of Basins A and B is included in Appendix A.

Although the scale of these basins differed (Basin A 36 m^2; RB4 1868 m^2), the quality of recharge water appears to have an important influence on infiltration rates from RB4. It must be emphasised that at the end of the fourth cycle of wetting of Basin A when 86.3 m of water was applied infiltration rate did not decrease at all (Appendix H), whereas the operation of RB4 decreased infiltration rate to 0.045 m/d with hydraulic loading of 18.3 m. Shallow piezometers (P4 10, P4 20, P4 50) drilled beneath the basin floor showed during July 2009 that at the depths of 10 and 20 cm the soil saturated very briefly at the start of wetting but the vertical hydraulic gradient was steep.

Table 4 summarises the hydraulic loading rates obtained during the laboratory column studies and their comparison with turbidity values. Initial hydraulic conductivity with potable water (Roe Creek) was ~25 m/d for sand and 0.2 m/d for silty sand (loam) for hydraulic gradient “i” = ~2.4 (Knapton *et al*, 2004, Mucha 2004).

These results suggest that the quality of source water may have an important influence on clogging development. Fig. 27 shows the comparison of hydraulic conductivity with quality parameters. All parameters were cross-correlated and so causative relationships are not clear but MFI, turbidity, Total N and TOC all were strongly negatively correlated with final hydraulic conductivity value in the laboratory studies.

Table 5 Hydraulic loading and infiltration rate over four cycles (mean ± half-difference between replicates given) obtained during the column studies and infiltration trial and pilot-scale SAT

Water type		Laboratory columns					6x6 trials		Pilot-scale SAT			
		Blaster-skite Park	**Rock-bed filtered**	**Bolivar DAFF (Const Temp)**	**Bolivar DAFF (Glass House)**	**Roe Creek**	**Basin A**	**Basin B**	**RB 1**	**RB2**	**RB3**	**RB4**
Turbidity (NTU)		72.1	43.2	0.6	0.65	0.45	0.45	0.45	2.5 ± 8.6	2.5 ± 8.6	2.5 ± 8.6	2.5 ± 8.6
Ponding depth (cm)		10	10	10	10	10	10-30	10-20	0-30	0-30	0-30	0-30
Length of ponding (d)		28	28	28	28	28	28	35	206	222	164	104
Hydraulic loading (m)	Sand	65.2	56.0	168.3	200.3	394.9	86.3	355.3	46.5	52.5	40.9	18.3
	Silt (loam)	1.83	2.11	9.77	3.87	6.73						
Average inflow rate (m/d)*	Sand	2.33	2	6.01	11.22	10.99	3.1	10.15	0.45	0.46	0.50	0.38
	Silt (loam)	0.07	0.08	0.35	0.54	0.53						
Final infiltration rate (m/d)	Sand	0.048	0.024	1.05	1.3	2.5	~ 3	~10	0.12	0.17	0.12**	0.04***
	Silt	0.02	0.017	0.17	0.04	0.075						

* Average inflow rate (m/d) = Hydraulic loading (m)/length of wetting (d)

**scraping increased infiltration rates up to 0.21 m/d

***scraping increased infiltration rates up to 0.06 m/d

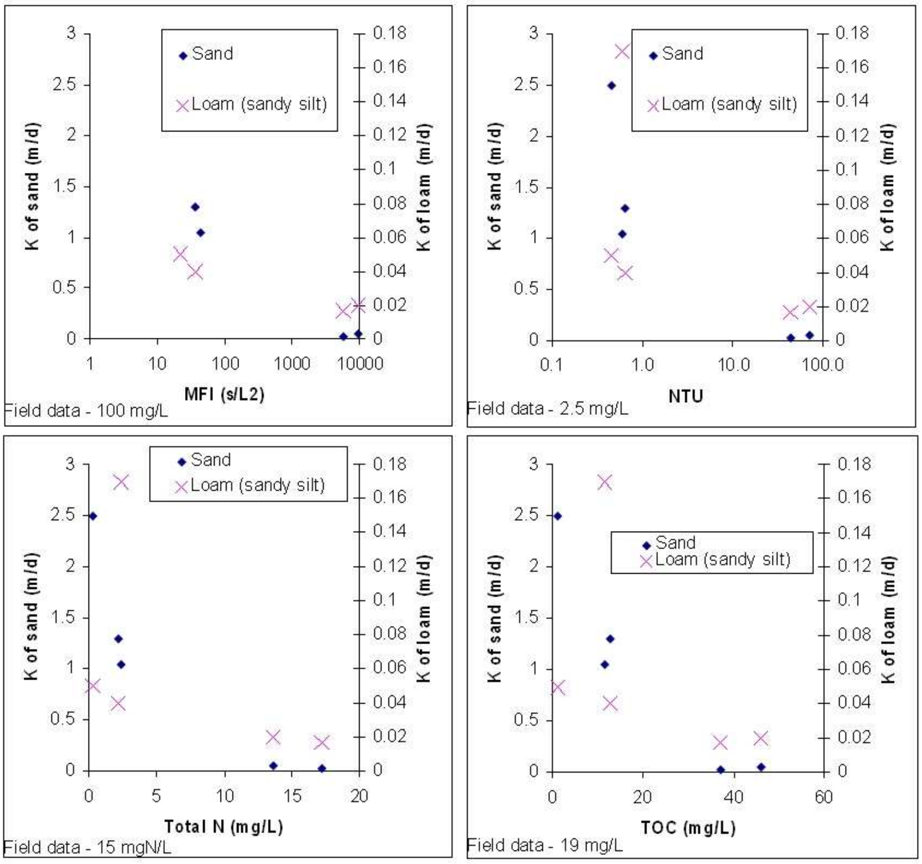


Figure 27 Comparison of hydraulic conductivity and water quality. Hydraulic conductivity is a final measurement taken after the fourth wetting cycle (1 wetting/drying cycle lasts for 1 week)

5. RESULTS CONCERNING FATE OF CONSTITUENTS

Water quality effects of the site operation are presented in this section. Reclaimed water chemistry which is used to recharge the aquifer is described in detail in Section 5.1. Statistical data on background groundwater, recharge water, and groundwater affected by mixing is presented in Appendix B.

The performance of SAT systems is assessed on the basis of fate of nutrients in reclaimed water. The other issues concerning SAT operation are as follows: changes in major cation composition in the ambient groundwater, leaching of metals, and behaviour of pathogens, viruses and trace organics.

5.1. Nitrogen, carbon and phosphorus fate

The quality of groundwater in observation well RN 17947 located on the eastern bank of RB 4 (screened at depth of 20-30 m) is strongly affected by the SAT operation as observed in the decrease in chloride concentration from ~500 mg/L to less than 250 mg/L. The variations in temporal concentrations of selected constituents and parameters are presented in Fig. 28. After December 2008, the groundwater in RN 17947 is >90% recharge water.

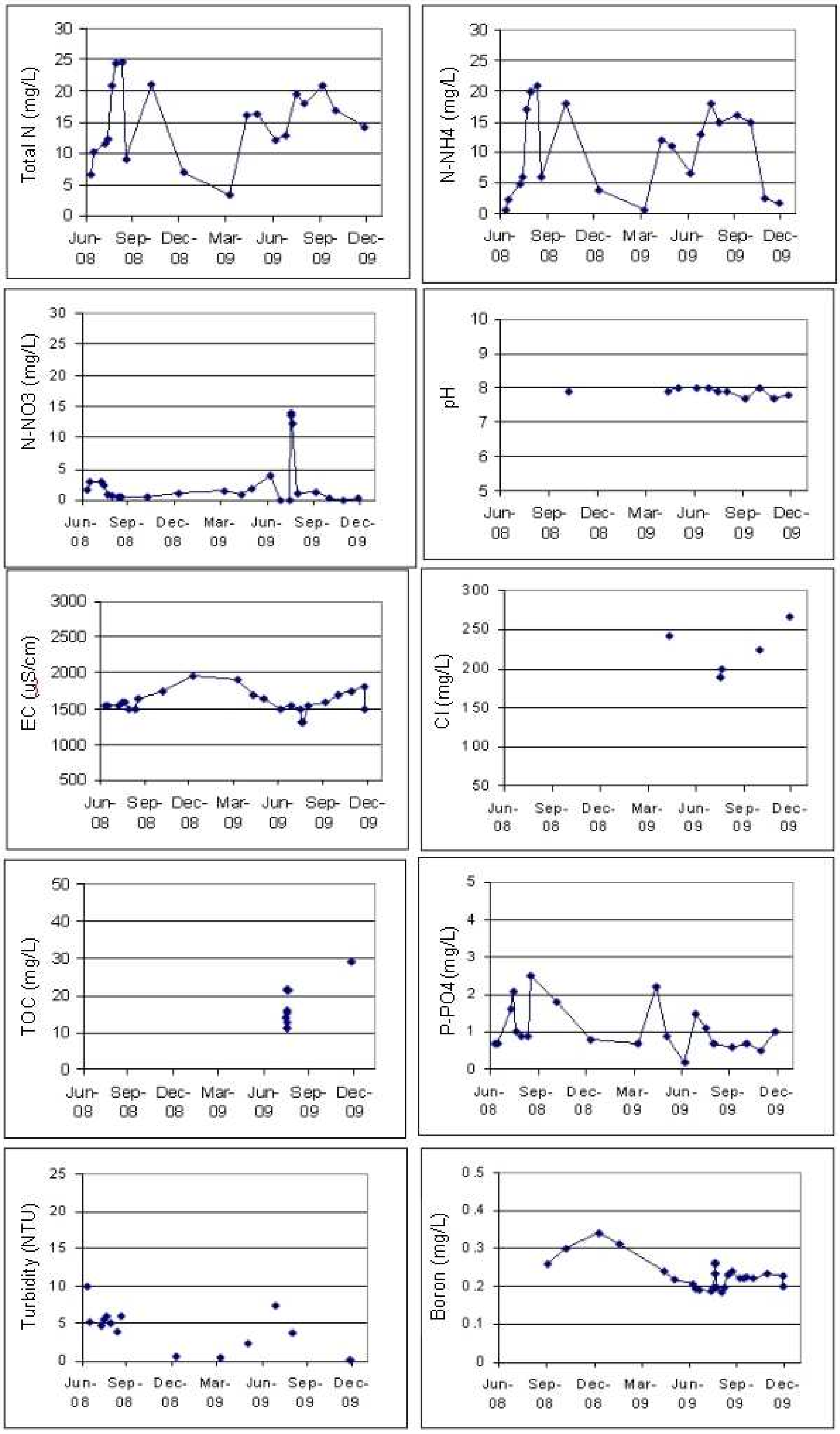


Figure 28 Temporal variations in selected parameters values and concentration of compounds in RN 17947. SAT commenced in June 2008

Attenuation of species estimated on the basis of conservative mixing of chloride is presented in Table 5. It must be noted, however, that there are large variations in both Cl^- and N concentration in the recharge water. The calculation for two samples indicates that around 18-42% of total N is being attenuated. Attenuation of TOC is ambiguous but small. Phosphorus seems to be effectively attenuated. For comparison, boron and bromide data are also given as they can be sometimes used as tracers.

Table 6 Attenuation of selected contaminants in reclaimed water during SAT operation

Constituent	**Ambient ground-water (mg/L)**	**Source water (average, mg/L)**	**24/07/2009**					**8/12/2009**				
			RN 17947 (mg/L)	**Mixing fraction f_{Cl}**	**Mixing alone**	**Reaction amounts**	**Percent attenu-ated**	**RN 17947 (mg/L)**	**Mixing fraction f_{Cl}**	**Mixing alone**	**Reaction amounts**	**Percent attenu-ated**
Cl^-	535	219	250	**0.90**				234	**0.95**			
N_{tot}	5	14.8	8		13.84	5.84	**42**	11.8		14.33	2.53	**18**
TOC	1.5	17	12		15.48	3.48	**22**	17.5		16.26	-1.24	**-8**
PO_4^{3-}	0.01	1	0.02		0.90	0.88	**98**	*n/a*		0.95	*n/a*	***n/a***
B	0.1	0.23	0.325		0.22	-0.11	**-50**	0.18		0.22	0.04	**20**
Br^-	3	0.69	0.5		0.92	0.42	**45**	0.6		0.80	0.20	**25**

Average concentration of organic nitrogen (N_{org}=TKN-NH_4) in recharge water is relatively low (~2.5 mg/L) as compared with average concentration of ammonium (11 mg/L on average). Both organic nitrogen and ammonium seem to be effectively attenuated since they are absent in groundwater (Fig. 29). The vertical profile of the concentration was plotted for water sampled on 21-24 July 2009. Ammonium seems to be converted to nitrate in the top 10 m of the profile. The decrease in nitrate concentration at a depth of 25 meters is probably due to dilution with ambient groundwater. The average concentrations of total organic carbon do not change in the profile, which points to low degradability of effluent organic matter. Fox *et al.* (2001) suggest that BOD:N ratio greater than 3 is necessary to sustain high nitrogen removal efficiency. They admit, however, that most of SAT systems have this ratio around 1, similar to Alice Springs site. The adsorption of phosphorus appears to take place in Alice Springs.

Nitrate was also detected in the wells located downgradient of the site (Fig. 30). On 15/12/2009 the contribution of recharge water in RN 17947 (~200 m from the site) was 93%, and attenuation of N was 53%. In the well RN 17999 (~ 500 m from the site) the increase in NO_3 and decrease in Cl^- concentration has been observed since September-December 2009. On 15/12/2009 the contribution of recharge water in RN 17999 was 28%, and attenuation of N was 23%.

Extraction with KCl revealed an accumulation of nitrogen compounds in the shallow unsaturated zone. The results imply that ammonium is strongly adsorbed in the subsurface, although some of it is being oxidised to nitrite and nitrate (Appendix I). There are conspicuous "spikes" in ammonium and nitrate concentrations due to a set of processes that control fate and transport of nitrogen in the subsurface. As indicated in Appendix I NH_4 contents were higher and more variable than expected and further coring sampling and analysis is warranted to help understand fate of nitrogen within the upper profile.

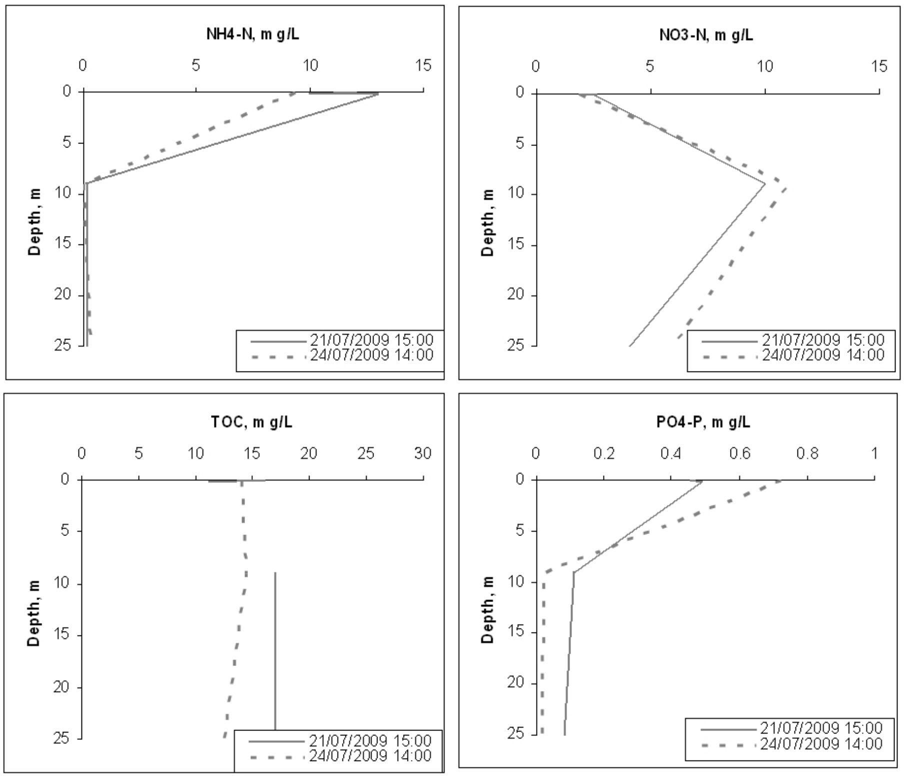


Figure 29 Concentrations of inorganic forms of N and P, and TOC in shallow piezometers (P4 10, P4 20 at 0.1 and 0.2 m) and observation wells (RN 17945, RN 17947 at 6-12 and 20-30 m) beneath RB4 (21 July 2009 – drying; 24 July 2009 – wetting).

5.2. Salinity and ability to recover recharge water

Ambient groundwater has a salinity of 2100 mg/L, which makes it unsuitable for drinking or for irrigating most crops. The infiltration of recharge water results in the decline in salinity of groundwater, which is particularly conspicuous in RN 17947 (Fig. 30).

Some observation wells located downstream of the site exhibited a decrease in Cl^- concentrations, which indicate prospects for recovery in the future. The well 17942 is located ~200 m from the RBs and the freshening effect was observed in September 2008 after recharge commenced. However freshening was not evident at RN 17999 500m downgradient. If chloride concentration in groundwater is maintained at this level, the recovered water will meet salinity requirements for irrigation (ANZECC-ARMCANZ 2000).

SAR (sodium adsorption ratio) in ambient groundwater is around 4.3, whereas in recharge water it varies from 6-6.5 (not shown). However, due to cation exchange in the subsurface SAR value in RN17947 decreased to 2.7. It is expected that SAR will increase in the future once Na^+ becomes dominant in exchange sites.

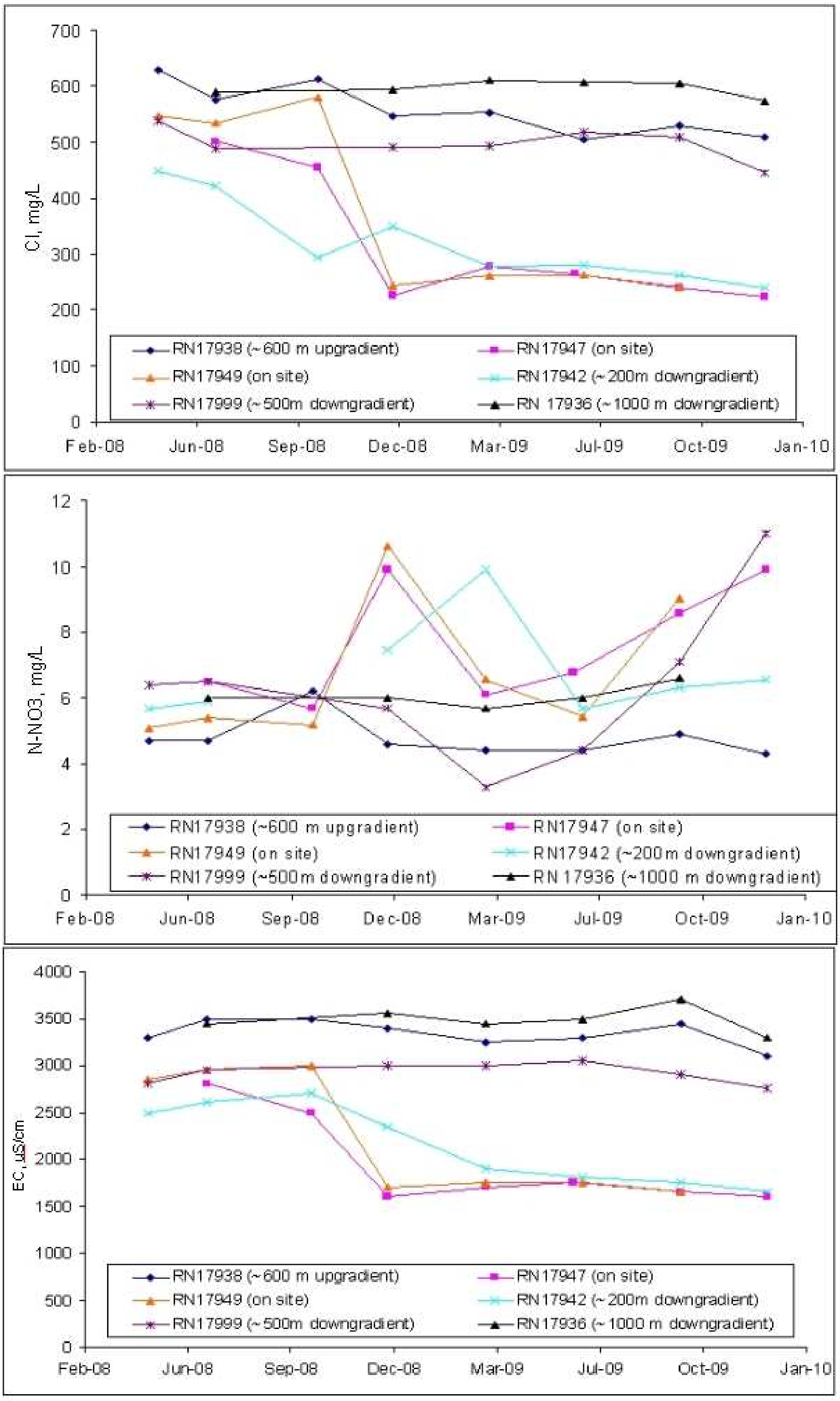


Figure 30 Concentrations of chloride and nitrate and EC value in selected wells. The distance from the site is also given. Average concentrations of Cl and Total N in source water are 219 and 14.8 mg/L, respectively, whereas EC value is 1578 µS/cm. SAT commenced in June 2008

5.3. Tracers of recharged water in the aquifer

As anticipated, chloride seems to be a good tracer of reclaimed water movement in the aquifer. Nevertheless, the chloride concentrations in reclaimed water exhibit annual variation (Fig. 24) and some correction may have to be applied.

In spite of adsorption to aquifer material, boron seems to be a good indicator of recharge water in Alice Springs. Boron concentration in source water ranged from 180-340 µg/L (240 µg/L on average, n=20) whereas background groundwater B concentration was 80-100 µg/L. In mixed water, B concentration is up to 260 µg/L following the elevated concentrations in unsaturated zone (340 µg/L). However, in this range of boron concentrations, especially in high Cl^- waters, the readings with the use of standard techniques may be not very precise (Gregoire, 1990). Depending on the crop, the acceptable B concentration for irrigation supplies are 0.5-5 mg/L.

Bromide in groundwater should behave similar to Cl^-. Nevertheless, in the low Br^- and high Cl^- concentrations the measurement using ion chromatography is usually not precise. Bromide concentration in source water ranged from 580 to 860 µg/L (685 µg/L on average, n=20), whereas native groundwater had Br^- concentrations of 2610 to 3790 µg/L (3260 µg/L on average, n=19). On 23/04/2008 ambient groundwater in RN 17939 exhibited anomalously low Br^- concentration of 1550 µg/L due to flooding events of Todd river (B: 80 µg/L, Cl^-: 250 mg/L, EC 1450 µS/cm).

5.4. Aeration of the profile and other field parameters

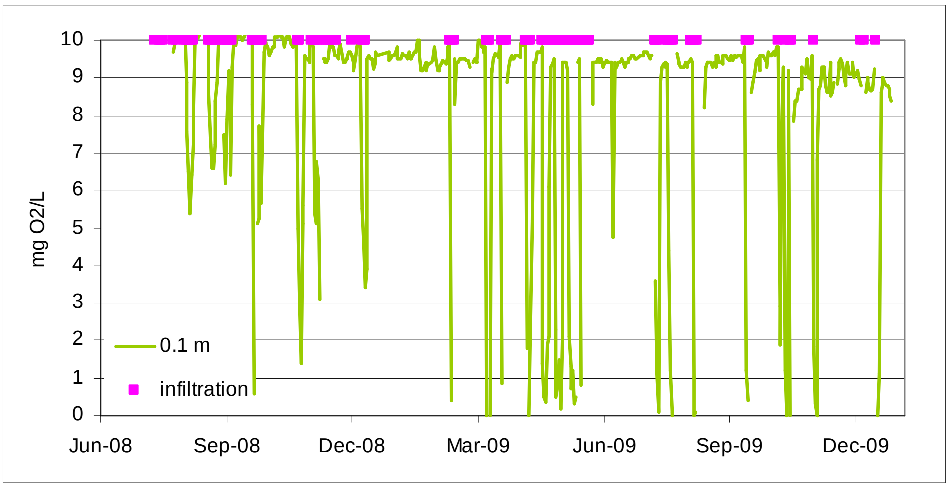
Infiltration events in RB4 saturate some soil pores which are normally unsaturated. Concentration of dissolved oxygen was measured in RB4 (0.1 m above bottom) and in the subsurface at depth of 1, 1.8 and, 2.4 m. The infiltration events from RB4 strongly coincide with the "spikes" of low oxygen concentration in RB4 (Fig. 31a). Nonetheless, the conditions in the basin hardly ever become anoxic. From among the sensors situated within the subsurface, the biggest variations in oxygen concentrations occur at the depth of 1 m, whereas the smallest changes occur at 2.4 m (Fig. 31b). This is a result of transport downwards of oxygen-depleted water, followed by reaeration during drying of the profile from the top downwards. The changes of oxygen concentration over time observed at different depth profiles seem to be ruled by advective transport of water in the unsaturated zone but also diffusion and dissolution of oxygen when the soil profile becomes drier.

On the basis of dissolved oxygen distribution in the vertical profile, the velocity of oxygen transport front can be calculated. The minimum value as the times of migration to each depth is used. In the period of 28/09/2008-10/10/2008, the oxygen front moved with velocity v_1=0.185 m/d between 1 m and 1.8 m and v_2=0.082 m/d between 1.8 m and 2.4 m. The decrease of velocity over depth is caused by the lack of infiltration in those days (infiltration ceased on 24 Sept). The average velocity in the depth interval 1-2.4 m amounts to 0.12 m/d. Another front is conspicuous in the period of 27/05/2009-30/05/2009. Since water was being infiltrated from the basin (Fig. 31b) in that period, the migration of the O_2-profile in the unsaturated zone was much faster and equalled v_3=1.0 m/day between 1-1.8 m and around v_4=0.2 m/d at a depth of 1.8-2.4 m. The latter value seems to be in the same order of magnitude as the rate of water level decline during drying phase (Fig. 20).

The lack of anoxic conditions in the unsaturated zone profile between 1.0 and 2.4 m suggests that attenuation of nitrogen by denitrification would be unlikely in this depth interval. Nevertheless, anoxic conditions can occur shallower in the profile which is confirmed by leaching of metals (section 5.5). Such conditions can only occur when the shallow subsurface becomes saturated (Appendix G).

Similar evaluation can be performed for the period of the field campaign in July 2009. Concentrations of oxygen are shown together with redox potential (Eh) (Fig. 32). Again, there is a conspicuous advance of a low-oxygen front after the wetting period. The profiles of both O_2 and Eh exhibit some similarities, which is chiefly apparent for data from the basin and at a depth of 1m. Very small change in oxygen concentration between 1 and 1.8 m was noticeable, which implies that the soil profile did not re-aerate in this depth interval.

a)



b)

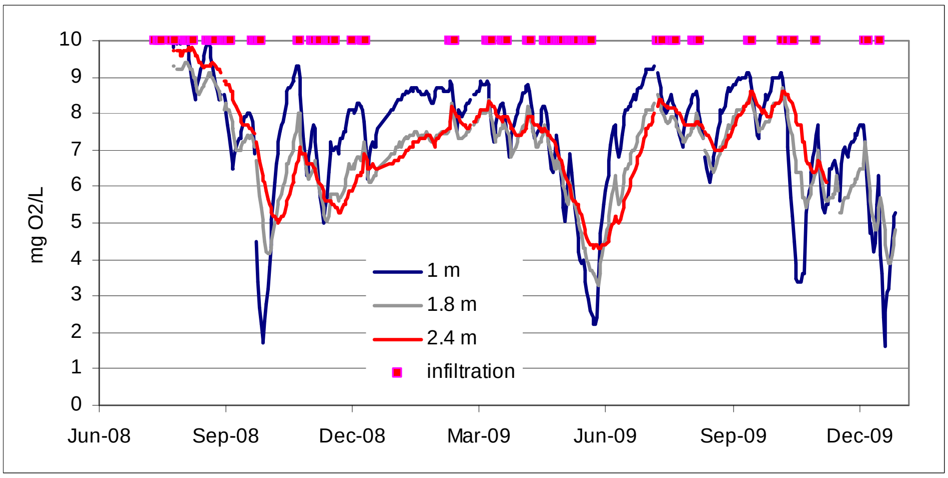


Figure 31 Concentration of dissolved oxygen (a) in RB4 (0.1 m) and in (b) the unsaturated zone (depths 1, 1.8, and 2.4 m) below RB4 for the period 23/07/2008-31/12/2009. Pink bars show the periods of infiltration in RB4

A slight retardation of O_2-profile with respect to Eh is apparent which may suggest that oxygen is not the only substrate taking place in redox reactions. The recovery of the system in terms of redox conditions (2.4 m depth) generally takes around three weeks after the application period finishes (not shown in the graph).

The field parameters in the profile that were observed during the field campaign in July 2009, did not exhibit very large variability. The samples often constituted perched water as piezometers were rarely saturated (Appendix G). Those parameters were measured with the use of field equipment rather that SCADA system and can not be compared directly with SCADA output. In RB2 the dissolved oxygen concentration dropped from about 8 mg/L in the RB to about 4 mg/L at the depth of 0.5 m (Fig. 33). There is an increase in EC value which is probably the effect of evaporation and mineralisation of organic nitrogen and carbon to some extent (the latter is confirmed by the increase in alkalinity). pH exhibits variations from 7.25 to 8.5 which is mostly related with algal growth and CO_2 saturation. Observation of those parameters deeper in a profile beneath RB4 implies that most of the parameters decrease over depth (Fig. 34).

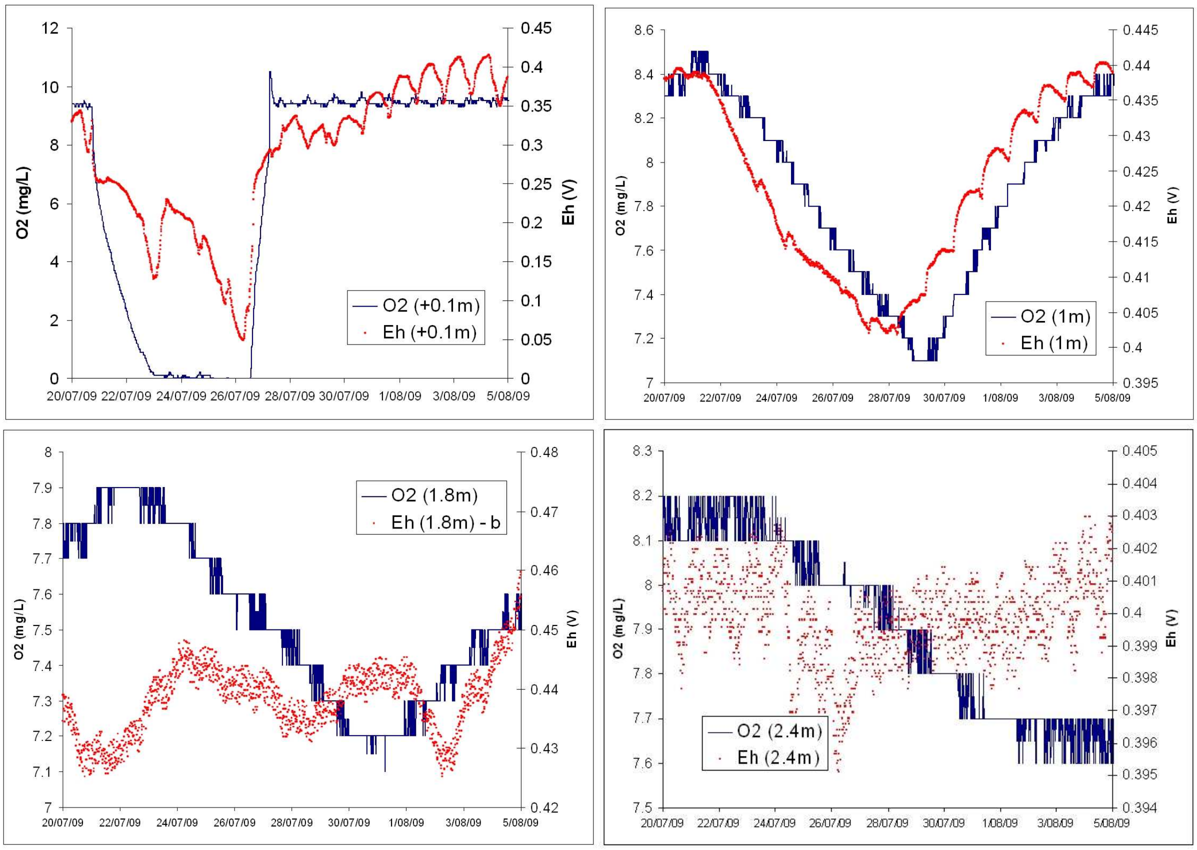


Figure 32 Concentration of dissolved oxygen (mg/L) and redox potential (V) in RB4 and in the unsaturated zone below RB4 for the period 20/07/2009-5/08/2009. Note the differences between the Y-axis scales

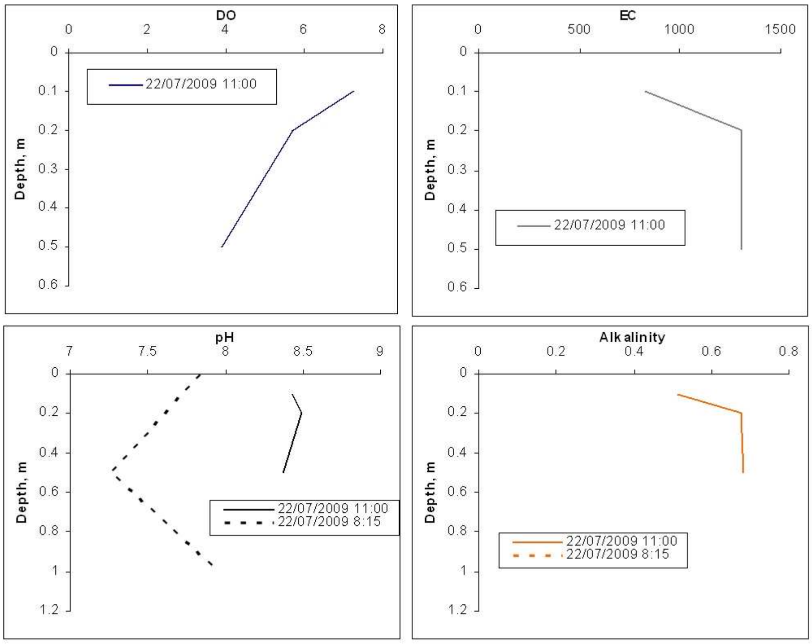


Figure 33 Dissolved oxygen (mg/L), electrical conductivity (µS/cm), pH, and alkalinity (meq/L) in the depth profile beneath RB2 (piezometers P2 10, P2 20, P2 50, and SC 100) measured during wetting cycle under the field campaign (July 2009)

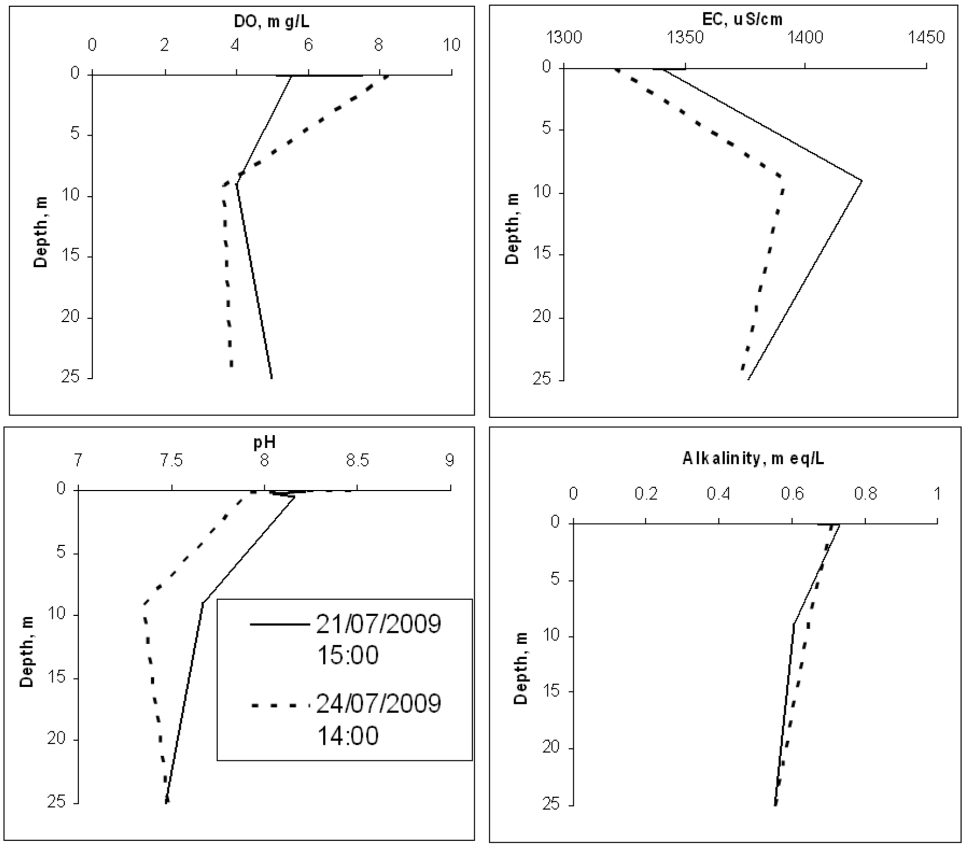


Figure 34 Dissolved oxygen, electrical conductivity, pH and alkalinity in the depth profile beneath RB4 (piezometers P4 10, P4 20, SC 100, and wells RN 17945, RN 17947) measured during the field campaign (July 2009)

5.5. Leaching of metal oxides from the profile

The sediments sampled beneath the RBs are characterised by high extractable Fe^{2+}, Mn^{2+}, and Al^{3+} contents. Lower concentrations of metals found beneath RB2 and RB4 as compared with RB5 may imply that the leaching of Fe^{2+}, Mn^{2+}, and Al^{3+} has taken place to a depth of 40 cm (Fig. 35). This would suggest that in the very shallow subsurface anoxic conditions intermittently occur during wetting of the basins.

The floor of RB4 is enriched in carbonate, which indicates precipitation of calcite (or dolomite). The process is suggested by oversaturation with respect to carbonate minerals (Fig. 26). Organic carbon content is constant over depth beneath all active basins and the enrichment beneath RB5 which at this point had not contained water could not be confirmed as representative of other basins before SAT operations commenced (Fig. 35). The mass of carbon leached or oxidised per square metre of basin is 3 kg/m^2. Evidently if such organic carbon depletion had occurred entirely by leaching this would have resulted in an increase in organic carbon in leachate of 70 mg/L over an average hydraulic loading of say 40 m. Clearly if organic carbon has been depleted it has been through oxidation or has been adsorbed lower in the profile as there has been marginal net depletion of organic carbon reaching the water table beneath the basins (Table 5).

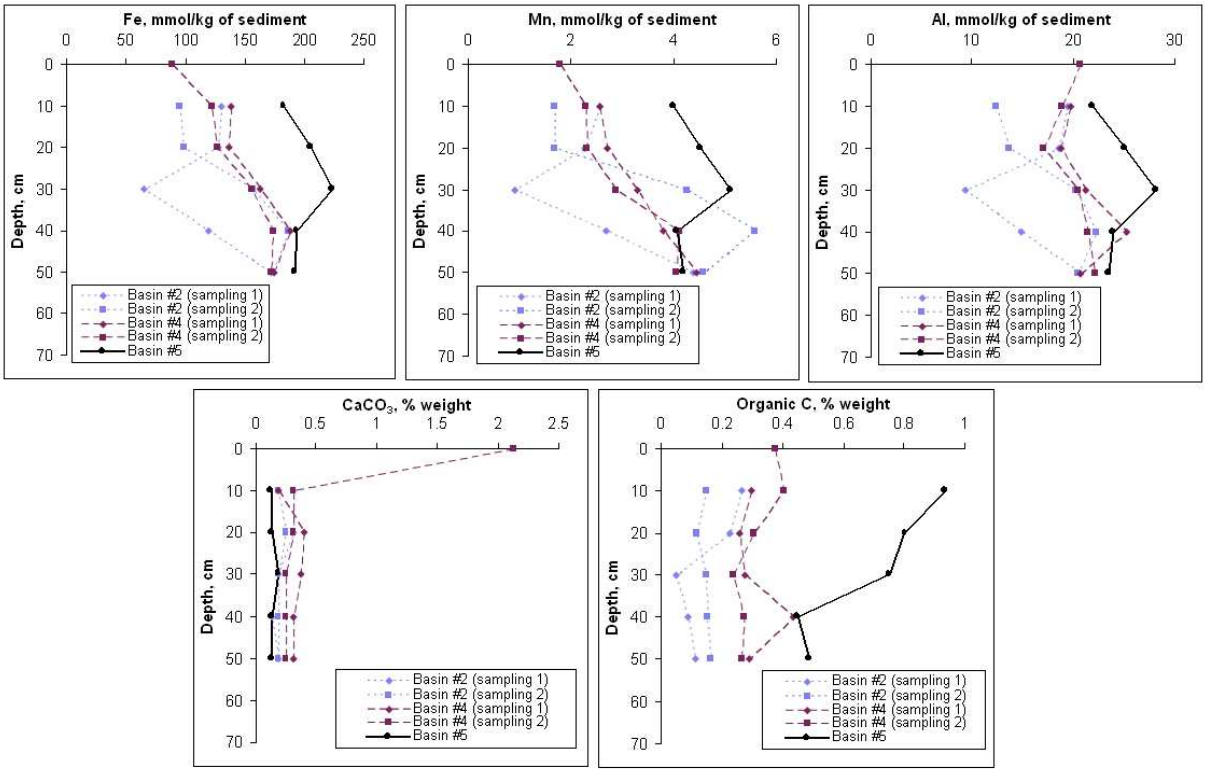


Figure 35 Content of metals and carbon in the shallow unsaturated zone measured in cores taken in RB2, RB4 and RB5 in July 2009

5.6. Fate of trace organics

The most abundant pharmaceutical compounds and herbicides present in the source water sampled from the pipeline to the AZRI site (SC100) were carmazepine, sulphamethoxazole, dicamba, MCPA, simazine and diuron (Appendix J). From seven separate samples of source water between July 2008 and December 2009 no compounds were found at concentrations that exceeded guidelines for the intended uses of the recovered water. Further evaluation with respect to augmentation of drinking water supplies is recommended in order to prepare for future questions on the potential for higher valued uses of this water resource, both in source water and water post SAT treatment.

5.7. Biogeochemical processes during SAT operation

The observation of water quality in the vicinity of the SAT site in Alice Springs indicates that several reactions are taking place during infiltration resulting in the improvement of reclaimed water quality as well as decreasing the salinity of local groundwater. These reactions are in addition to physical mixing of waters including mechanical dispersion and diffusion. Chemical reactions predominantly involve mineralisation of effluent organic matter, dissolution and precipitation of minerals, ion exchange and surface complexation. Most biogeochemical reactions in SAT systems are redox reactions. Many physical and chemical effects of the site operation are coupled with and determined by the formation of biomass in a subsurface due to discharge of high nutrient reclaimed water.

The attenuation of compounds like organic carbon and nitrogen are intended. This is particularly so for the nitrogen compounds since both organic nitrogen and ammonia entirely disappear in groundwater in Alice Springs. Organic nitrogen is mineralized to ammonia which can be exemplified by the hydrolysis of an amino compound (Jansson and Persson 1982, Sumner and Bradner 1996):

$$R-NH_2+HOH \rightarrow R-OH+NH_3 \tag{6}$$

Subsequently ionised ammonia (ammonium) is oxidised to some extent by the reaction of nitrification (Schmidt 1982; Sumner and Bradner 1996):

$$\begin{aligned} &2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 2H_2O + 4H^+ \\ &2NO_2^- + O_2 \rightarrow 2NO_3^- \end{aligned} \tag{7}$$

Oxidation of organic carbon by oxygen is favoured from a thermodynamic point of view, but strongly depends on reactivity of organic compounds (Postma and Jakobsen 1996):

$$CH_2O+O_2 \rightarrow CO_2+H_2O \tag{8}$$

The aforementioned processes are strongly dependent on oxygen availability in unsaturated zone. Oxygen concentration at depths up to 2.3 m suggests that relatively fast reaeration of the soil is apparent, which makes those reactions very favourable.

Alternatively if anaerobic microniches occur NH_4^+ can be oxidised anaerobically with reduction of nitrite to gaseous nitrogen via ANAMMOX process (Mulder *et al.* 1995, Fox 2002):

$$NH_4^+ + NO_2^- \rightarrow N_2 + 2H_2O \tag{9}$$

The existence of this process can not be confirmed on the site, although the accumulation of ammonium in the unsaturated zone may favour this mechanism to occur in the future. So far, the large concentration of oxygen beneath RB4 (at least below 0.4 m) does not favour the reduction of ammonium (or ammonia) by nitrite.

Since oxygen saturated water contains about 0.33 mM O_2, the maximum reduction of ammonium according to equation (7) is 0.22 mM NH_4^+. Since more ammonia is being oxidised, diffusive supply of oxygen must take place. This is in accordance with the process of profile reaeration even during wetting of the basin (Appendix G).

Denitrification is the enzymatic reduction of nitrate to nitrite and gaseous nitrogen compounds, which may escape to the atmosphere, removing nitrogen from soil water and groundwater. In some cases, however, nitrate reduction can result in conservation of nitrogen due to dissimilatory nitrate reduction to ammonium (DNRA) (Korom 1992, Sumner and Bradner 1996):

$$\overset{(5+)}{NO_3^-} \rightarrow \overset{(3+)}{NO_2^-} \rightarrow \overset{(2+)}{NO} \rightarrow \overset{(1+)}{N_2O} \rightarrow \overset{(0)}{N_2} \xrightarrow{DNRA} NH_4^+ \tag{10}$$

The process of denitrification coupled with organic carbon (heterotrophic denitrification) occures in systems contaminated with organic carbon and proceeds as follows (Korom 1992, Sumner and Bradner 1996):

$$5C+4NO_3^- + 2H_2O \rightarrow 2N_2 + 4HCO_3 + CO_2 \tag{11}$$

In many case studies where organic matter has a large degree of biodegradability, this process is responsible for the increase in alkalinity (Christensen *et al.* 2000). In Alice Springs, however, organic material seems to be poorly reactive. As a result, both nitrate and organic carbon are not being effectively attenuated. To evaluate methods to facilitate this process, a detailed examination of the forms of effluent organic matter is required (Drewes *et al.* 2006, Drewes 2009). Dissimilatory nitrate reduction to ammonium does not occur in Alice Springs due to aerobic conditions prevailing.

The nitrogen cycle in an anoxic environment can be also facilitated by reactions of ammonia and nitrate with Mn^{2+} and Mn-oxides (Luther *et al.* 1997):

$$2NH_3+3MnO_2+6H^+ \rightarrow 3Mn^{2+}+N_2+6H_2O \tag{12}$$

$$5Mn^{2+} + 2NO_3^- + 8OH^- \rightarrow 5MnO_2 + N_2 + 6H_2O \qquad (13)$$

Transport of ammonium in geological media is also strongly determined by sorption and ion exchange processes (Ceazan *et al.* 1989; Bohlke *et al.* 2006). The exchange reaction with Ca^{2+} is as follows:

$$2NH_4^+ + Ca - X_2 \rightarrow 2NH - X + Ca^{2+} \qquad (14)$$

The processes of sorption and ion-exchange largely contribute to nitrate "spiking" in soil water and are the prerequisites for the ANAMMOX process. High CEC values of sediments from the AZRI site (Gates *et al.* 2008) confirm that ANAMMOX is feasible.

Sorption also determines the fate of phosphate, which is taken up particularly into iron, aluminium, and manganese oxides (Stollenwerk 1996) which are abundant in sediments at this site (Gates *et al.* 2009). Phosphate is certainly transported in groundwater and could be found in the deep wells, but it usually requires much longer operation of the site (Parkhurst *et al.* 2003). For example, in groundwater next to the Shafdan site in Israel, one of the largest SAT systems in the world, concentration of phosphorus remained <0.05 mg/L after 25 years of operation (Lin *et al.* 2006).

Other transport mechanisms include colloid-facilitated transport (Grolimund *et al.* 1996) including straining. Colloid-facilitated transport is particularly favoured in high flow rate systems like SAT. As a result of this process, unoxidised forms of nutrients may be transported through an unsaturated zone. Straining is a process in which particles or flocculations of colloids are lodged within small diameter pores (Bradford *et al.* 2006). Particulate forms of nitrogen (non-dissolved organic nitrogen or particulate-adsorbed ammonium) are particularly susceptible to this process (Sumner and Bradner 1996).

Other mechanisms include dissolution and precipitation of minerals. Carbonate dissolution and precipitation are based on carbonate equilibrium reaction:

$$CO_{2(g)} + H_2O + CaCO_3 \leftrightarrow Ca^{2+} + 2HCO_3^- \qquad (15)$$

An increase in CO_2 results in dissolution of $CaCO_3$. Removal of CO_2 causes $CaCO_3$ to precipitate. Recharge water is oversaturated with respect to calcite and dolomite, which indicates that there is a potential for precipitation of these phases. If CO_2 is produced due to degradation of effluent organic matter then the dissolution of carbonates in soil should take place, whereas photosynthesis or extensive algal growth consumes CO_2 and will result in precipitation of $CaCO_3$ on the basin bottom. The latter process was observed during the field trip in July 2009.

The comparison of metal contents detected during extraction with citrate-dithionite and oxalate methods (Gates *et al.* 2009) suggests most of reactive iron occurs in highly crystalline forms, whereas most of both Al and Mn phases are poorly crystalline and amorphous. The main aim of the extraction experiment performed in July 2009 was to assess whether the dissolution of the oxides as a result of the SAT operation is taking place. For this reason the citrate-dithionite method which extracts all reactive oxides was applied.

Dissolution of metals is mostly related with the processes of dissolution of iron oxyhydroxides, manganese oxides, and aluminium oxyhydroxides. First two processes act as redox buffer in the subsurface. They can be exemplified as the reactions of organic matter with amorphous ferric oxyhydroxide and with pyrolusite:

$$CH_2O + 4Fe(OH)_3 + 8H^+ \rightarrow 4Fe^{2+} + 11H_2O + CO_2 \qquad (16)$$

$$CH_2O + 2MnO_2 + 4H^+ \rightarrow 2Mn^{2+} + 3H_2O + CO_2 \qquad (17)$$

Dissolution of those phases is suspected in Alice Springs, because lower concentrations of extractable metals are found in the upper part of the profile. Nevertheless, those measurements need to be repeated in due course to verify this hypothesis. The leaching of metals may potentially result in the decrease in sorption capacity and the increase in Fe^{2+}

and Mn^{2+} concentration in recovered water. In addition to a low degree of organic carbon reactivity, the degree of crystallinity of reactive iron minerals is high, which suggests that the process of dissolution must be sluggish and therefore difficult to perceive. In the case of Mn-oxides the process of oxidation is more probable due to higher energy yield (Stumm and Morgan 1996). The processes of Fe- and Mn-oxide dissolution were documented in the Shafdan SAT site and elicited the increase in Mn^{2+} concentration from 0 to about 2.2 mg/L in groundwater being extracted, although the increase in Fe^{2+} concentration has not been confirmed yet (Oren *et al.* 2007). On the one hand, the concentrations of Fe^{2+} and Mn^{2+} may start to increase in Alice Springs due to oxide dissolution. On the other hand, both Fe^{2+} and Mn^{2+} can re-precipitate in the profile, which was also observed in Shafdan (Oren *et al.* 2007).

Dissolution of aluminium oxides (gibbsite, lepidocrocite and amorphous Al-oxyhydroxide) acts as a pH buffer. The reaction proceeds as follows:

$$Al(OH)_3 + 3H^+ \rightarrow Al^{3+} + 3H_2O \qquad (18)$$

All those phases are soluble at low and high pH values. At pH=10 the solubility of amorphous $Al(OH)_3$ is 0.1 mmol/L (Appelo and Postma 2005), which gives an Al^{3+} concentration of 2.7 mg/L. pH of reclaimed water from Alice Springs is usually around 8.0 which does not favour extensive Al-oxide dissolution. Nevertheless, the mobilisation of Al^{3+} may become an issue when recharge water is too alkaline.

The application of fresh water to the brackish aquifer results in the ion-exchange process (Reaction 14). Since the cations with a different exchange affinity compete for the substantial capacity of the exchange sites, long-term application of fresh water may lead to the formation of a chromatographic pattern in the aquifer. This phenomenon was identified during injection of advanced treated municipal effluent into an aquifer (Valocchi *et al.* 1981) and is explained in detail by Appelo (1994). A chromatographic pattern results in a decrease in Sodium Adsorption Ratio (SAR) at the initial phase of the operation, which has been observed in Alice Springs. This has a positive effect in view of using this water for irrigation purposes. At some stage, however, SAR will start to increase again, because Na^+ is expected to be mobilised. Numerical modelling could be helpful in determining this issue.

6. CONCLUSIONS

The operation of Soil Aquifer Treatment system in Alice Springs commenced on 3rd June 2008. By the end of the year 2009, within 19 months of the site operation, 317.4 ML of reclaimed wastewater was discharged to the system by four recharge basins (total area of 7,640 m^2). When the evaporative loss and the gain from rainfall are taken into account about 308 ML of water infiltrated within 19 months on the area of 7,640m^2 i.e. 40 m hydraulic loading at an average rate of 70 mm/d over the total period of 575 days. The fifth basin with the largest area (2,629 m^2) has been recently added to the system. Its efficacy in terms of infiltration rate has not been quantitatively assessed yet. The total area of infiltration of the five recharge basins is 10,269 m^2. The aforementioned numbers imply that the infiltration of 600ML/year will require 2.5 ha if current infiltration rates continued.

The infiltration rates are lower than anticipated primarily due to the poorer quality of water in the basins than was used in the column studies during the design stage. There are two reasons for this (1) the water quality entering the basins was more nutrient-rich and contained more colloids, organic carbon, and bicarbonate than anticipated; and (2) during residence within the basins algal growth occurred, increasing the filamentous organic matter filtered out on the floor of the basins and increasing pH resulting in the possibility of calcite precipitation. However there is no clear evidence to enable target water quality criteria to be set in order to sustain infiltration rates by exclusion of source water at higher than critical concentrations.

Dominant causes of clogging are:

1. **Algal growth** expected to:
    a. increase with water residence time in a basin,
    b. increase with temperature and light intensity and duration,
    c. depend on the initial state of the basin and the extent to which resident nutrients and algal spores are present on rewetting of the basin,
    d. increase with nutrient concentration in source water (N and P in particular),
    e. increase pH of source water as carbon dioxide is consumed,
    f. produce oxygen as a by-product of photosynthesis.
2. **Physical clogging expected to:**
    a. increase with enhanced turbidity of source water,
    b. combine with biological clogging to enhance reduction in infiltration,
    c. be enhanced under windy conditions due to wave action mobilising fine sediments from basin sides.
3. **Chemical clogging expected to:**
    a. depend on calcite saturation index of water in a basin. This will depend on pH and inorganic carbon concentration and temperature of the water in the basin.
    b. depend on algal development as enhanced pH increases saturation index for calcite. This will help to bind colloidal and biological clogging layers.
4. **Biofilm** development expected to:
    a. enhance with algal growth as algae produce extracellular polysaccharides,
    b. be associated with any polymers added during DAF treatment to reduce turbidity. Those that are not removed in the sludge could contribute to “slime” or biofilm which is effective in clogging pores in porous media and also forming a gel with colloidal and algal material and precipitated calcite, and possibly protect calcite from redissolution on pond rewetting.

Due to large temporal changes in both composition of source water and infiltration rates, and in part due to the inherent instability of measurements of turbidity and some other parameters in recycled waters, it is not possible to definitively state the primary causes of infiltration reduction, nor the maximum values of water quality parameters that should enter the basins to achieve the required infiltration rates. However, further work is proposed to address this.

Infiltration of reclaimed water resulted in the formation of a very slight recharge mound beneath the site. The highest increase in water level of the main aquifer was 1.5 m, but the mound quickly disappears when the infiltration is reduced or ceased. It is anticipated that the increase of annual recharge up to 600 ML will not cause any environmental or operational issues related with the upward movement of water table.

The fate of nutrients occurring in treated recharge water is incomplete. While phosphorus removal is quite effective, the total nitrogen concentration in discharged water averaging 15mgN/L is only attenuated about 20%. The decrease in ammonium and organic nitrogen concentration was conspicuous, but those forms of nitrogen are mostly being converted into nitrate. Changing source water quality by having a lower concentration of nitrogen or higher concentration of labile organic carbon may be needed if the concentration of nitrogen in recovered groundwater is to be reduced. The attenuation of organic carbon is low due to poor degradability of organic compounds. This is another reason why the unsaturated zone remains aerobic and denitrification is not taking place.

The operation of SAT results in the freshening of the Quaternary aquifer and the lowering of Sodium Adsorption Ratio. As a result groundwater is getting more suitable for agricultural purposes in terms of salinity and can be recovered in the future.

7. RECOMMENDATIONS

1. It is recommended to improve the quality of recharge water since the reclaimed water being discharged to the recharge basins generates a plume of nitrate-rich water in the aquifer. The quality of reclaimed water has a vast influence on infiltration rates as well, which was confirmed by the column studies. Thus, the removal of nitrogen compounds at the stage of pre-treatment in the sewage plant should improve infiltration rates at the site. Further examination of type of effluent organic matter would help to resolve why so little of it is labile, and may provide clues as to how to enhance denitrification beneath the basins, or to remove nitrogen at the treatment plant. For example, sourcing water for the DAF plant from water with a shorter time in stabilization lagoons could be considered.

2. The impacts of water quality on infiltration rates warrant further exploration. A definitive evaluation of effects of water quality on clogging rates could be obtained by linking the water sampling at the DAF plant product water with running column studies with packed sand columns to measure the change in permeability over time. Such tests would need to be conducted under low pressure (0.3 m) so as to maintain similar shear stresses as would occur beneath the basins.

   Repeated measurements taken each time analyses are made of product water and say at least 12 samples over a year would be valuable in helping to isolate causes of clogging and assist in informing operators immediately through their own observations on the “cloggability” of product water.

   Measurements to support the evaluation should be extended beyond nutrients and major ions to include polysaccharide concentration, chlorophyll A concentration, calcite saturation index and total suspended solids. Such evaluations would enable evaluation of the benefits of a filtration treatment at the DAF plant, and could be a precursor or part of studies to facilitate design of a filtration treatment. Chemical analysis of schmutzdecke may need to be performed.

3. Recharge basin 4 (RB4) is not representative of the basins through which most of the infiltration takes place. The value of continuing to monitor Eh, DO, moisture, and suction in RB4 is limited from an operational perspective. It is suggested that DO and moisture content measurements and soil water sampling are undertaken in RB2 and pressure transducers installed in piezometers in RB2.

   Given the low cost of installing new shallow piezometers beneath RB2 in comparison with the cost of transducers the design could be improved to dispense with large storage volume beneath the slotted intervals to enable more reliable sampling (after checking that pressure transducer life is not reduced by sustained dry period). It is recommended that transducers are installed at the depths of 0.1, 0.2, 0.5, and 1.0 m with a view to defining the effect of very shallow perching layers and the onset of schmutzdecke clogging during the course of operations which results in desaturation of the profile and inhibition of denitrification. It is suggested that three pairs of soil moisture sensors & pressure transducers at different locations in the basin including near the centre of the basin could be installed at a depth of 0.1 m to demonstrate when saturation and desaturation of the shallow soil under the schmutzdecke occurs.

4. In keeping with the work of Breton (2009) to enable operators to build an appreciation of the ‘natural’ treatment system of SAT it is suggested that web cameras be set up to take photos of at least one of the areas being monitored by soil moisture transducers to enable operators to get a clearer perception of when to reintroduce water into that basin. If complemented with frequent site inspections using qualitative evaluation criteria as proposed by Breton (2009) it is expected that operator knowledge would grow and ultimately remote operation and use of cameras an all basins could enable inspections to be reduced.

5. Further work is warranted to characterise the organic carbon in source water and to determine whether benefits could be achieved of having shorter residence times in

stabilisation lagoons. Characterisation of the composition of schmutzdecke is also warranted in order to identify principal causes of clogging.

6. Further work is recommended to assess nitrogen removal processes in the subsurface, to evaluate metal leaching and ion exchange processes, including modelling to predict future SAR in recovered water. Continued occasional monitoring of trace organics in source water and in groundwater is recommended to provide information of value for future uses of this resource.
7. If higher capacity is required, further extension of the infiltration area will need to be considered. If current infiltration rates can be maintained the total basin area required is 2.5 ha. However slots cut to a depth of ~1.8 m in RB4 and other ponds to intersect a coarse sand layer would reduce and possibly alleviate the need for more basins. Such slots would likely be lined on the base with geotextile, be backfilled with clean sand which was spread out to a wide thin layer on the basin surface to maximise infiltration and minimise maintenance. Simple modelling should be performed to determine the effect of slots. The cost of slotting compared with the cost of excavation of new basins to this depth and to excavation of new basins can then be evaluated for economic efficiency.
8. Geophysical methods can be very helpful in a further investigation of the site, especially with a view to expanding a site in the future. Electromagnetic, sonic, and seismic methods can be useful in the determination of heterogeneity of the unsaturated zone, presence and depth of shallowest sand layers and palaeochannel. Additional infiltration basins should likely be positioned eastwards from the existing basins, where sandy layer outcrops at the location of the test basin B, which is already fitted with the observation wells. The delineation of the reclaimed wastewater plume may be performed by using the electrical resistivity technique on benchmark sections that can be resurveyed.
9. Sitting and operation of recovery wells should be based on geophysical surveys and groundwater modelling to ensure these give water of acceptable salinity. The water quality in recovery wells may differ from those in monitoring wells and a trial extraction, for example to test agricultural methods and crops, could be of benefit for calibrating a groundwater model for the design of a fully operational system.
10. Data from the SAT site should be evaluated on an annual basis at least for the next two years to synthesise learning from the operations and to adapt operation and plan future expansion of the site, to maximise the efficiency of the combined treatment and SAT basins and water recovery operations to achieve the intended quantity and quality at an acceptable capital and operating cost.

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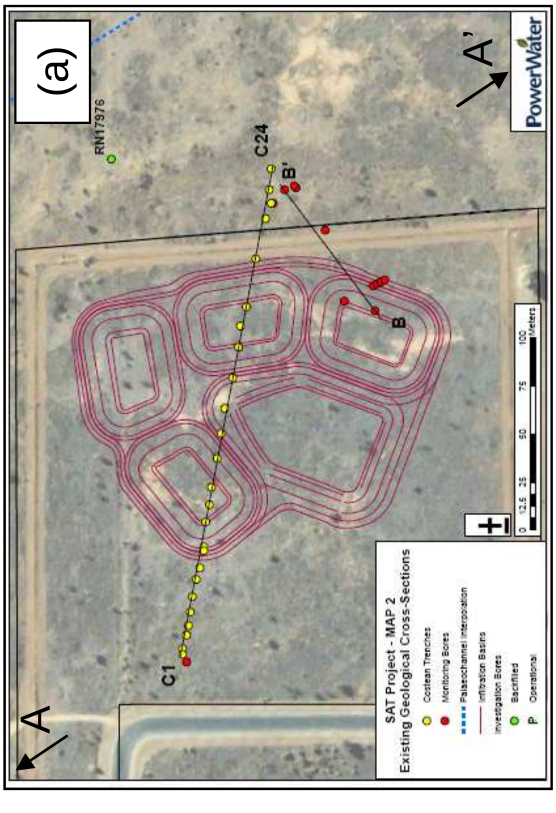
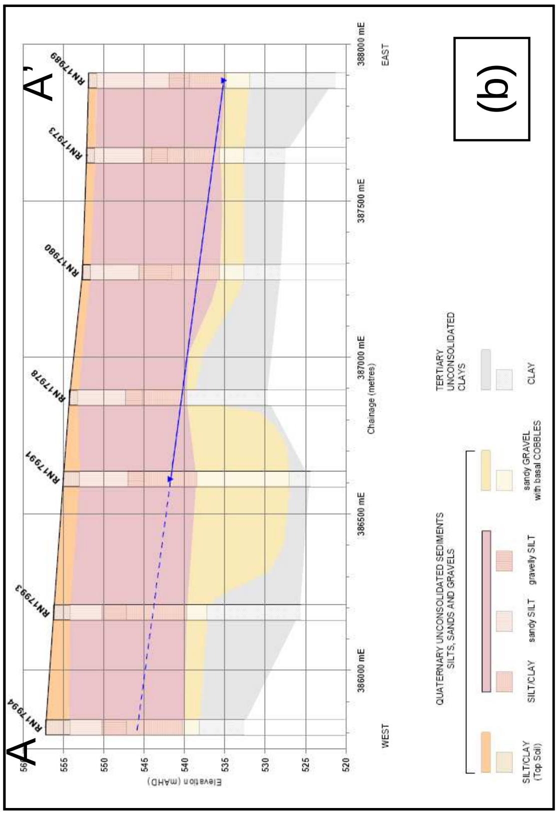
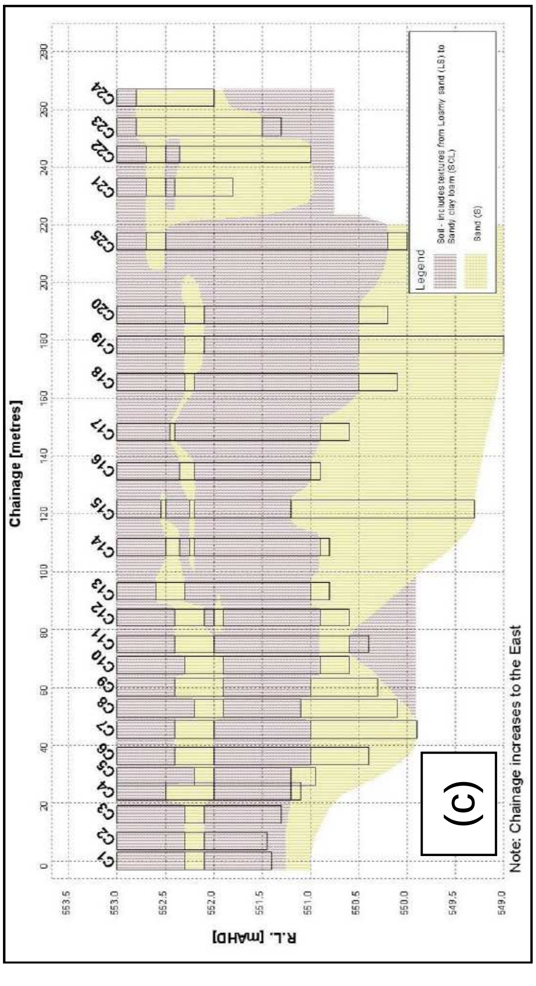
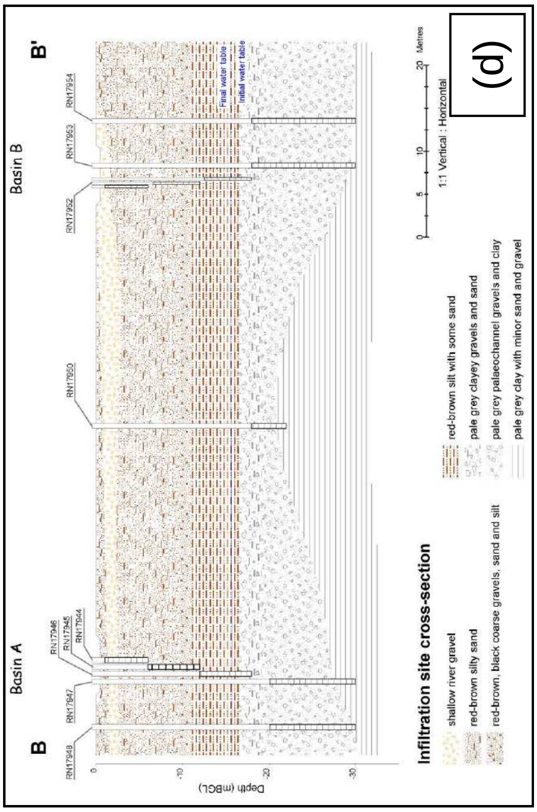
APPENDIX A: GEOLOGICAL CROSS-SECTIONS









APPENDIX B: STATISTICAL DATA ON RECHARGE WATER, AMBIENT GROUNDWATER, UNSATURATED ZONE WATER, AND SATURATED ZONE MIXED WATER

PARAMETER		RECHARGE WATER [a]	AMBIENT GROUNDWATER [b]	UNSATURATED ZONE [c]	SATURATED ZONE [d]
EC	Min	1303	1343	20	20
uS/cm	Max	1950	3400	2800	2800
	Ave	**1578**	**2559**	**1529**	**1499**
	n	23	21	24	20
pH	Min	7.7	6.5	7.2	6.95
	Max	8.97	7.7	8.2	7.6
	Ave	**8.0**	**7.2**	**7.8**	**7.4**
	n	10	21	20	18
Ca++	Min	43	99	83	102
mg/L	Max	57	310	260	265
	Ave	**51**	**217**	**128**	**160**
	n	13	21	24	20
Mg++	Min	27	24	8.3	25
mg/L	Max	36	105	64	53
	Ave	**31**	**56**	**23**	**36**
	n	13	21	24	20
Na+	Min	194	135	80	105
mg/L	Max	250	395	328	305
	Ave	**216**	**283**	**238**	**154**
	n	6	21	24	20
K+	Min	18.3	5.9	5.7	4.2
mg/L	Max	26	10	33.4	9
	Ave	**22**	**7.9**	**15**	**7**
	n	6	21	24	20
Cl-	Min	190	187	180	201
mg/L	Max	267	719	486	501
	Ave	**219**	**470**	**259**	**256**
	n	6	21	24	20
SO4--	Min	120	132	120	122
mg/L	Max	146	740	560	540
	Ave	**131**	**508**	**186**	**191**
	n	6	21	24	20
HCO3-	Min	400	255	311	335
mg/L	Max	517	416	430	400
	Ave	**442**	**340**	**385**	**373**
	n	9	21	4	7
Br-	Min	0.52	1.6	0.39	0.46
mg/L	Max	0.86	5.0	1.1	3.1
	Ave	**0.67**	**3.3**	0.65	0.8
	n	24	11	21	17
NO3-N	Min	0.4	0.27	2	4.1
mg/L	Max	14	12	11	12
	Ave	**3.3**	**4.2**	**5.5**	**9.1**
	n	20	21	6	12
NO2-N	Min	0.2	<0.005	<0.005	<0.005
mg/L	Max	1.7	<0.005	0.88	0.012
	Ave	**0.82**	<0.005	**0.19**	**0.01**
	n	17	0	17	13
NH4-N	Min	2.3	<0.005	<0.005	<0.005
mg/L	Max	21	0.02	0.16	0.32
	Ave	**12**	**0.02**	**0.09**	**0.245**
	n	21	1	15	10

PARAMETER		RECHARGE WATER [a]	AMBIENT GROUNDWATER [b]	UNSATURATED ZONE [c]	SATURATED ZONE [d]
N total	Min	3.5	**0.36**	1.2	8.0
mg/L	Max	24.7	**1.9**	30.8	15
	Ave	**14.8**	**1.1**	**7.7**	11
	n	23	**3**	20	14
P total	Min	0.4	<0.005	0.158	0.04
mg/L	Max	5.4	0.03	2.22	0.04
	Ave	**1.6**	**0.01**	**1.06**	**0.04**
	n	22	5	19	1
Fe total	Min	0.04	0.05	<0.03	0.05
mg/L	Max	1.06	3.3	5.9	1.5
	Ave	**0.17**	**0.66**	**0.93**	**0.31**
	n	13	12	24	20
Al+++	Min	0.00	0.02	0.03	0.08
mg/L	Max	0.99	3.7	1.9	1.3
	Ave	**0.13**	**0.61**	**0.55**	**0.51**
	n	23	12	24	20
B	Min	0.18	0.04	<0.05	0.08
mg/L	Max	0.34	0.16	0.42	0.35
	Ave	**0.23**	**0.10**	0.24	0.26
	n	24	15	24	19
TOC	Min	11	1.3	14	3.2
mg/L	Max	29	2	53	34
	Ave	**17**	**1.5**	**29**	**19**
	n	10	4	20	14
BOD	Min	<1	n/a	n/a	n/a
mg/L	Max	36	n/a	n/a	n/a
	Ave	7.5	n/a	n/a	n/a
	n	18	-	-	-
COD	Min	15	n/a	n/a	n/a
mg/L	Max	59	n/a	n/a	n/a
	Ave	43.6	n/a	n/a	n/a
	n	11	-	-	-

[a] includes results from location SAR80 prior to Sept 2008 and SAR100 Sept08 to Dec09

[b] includes data from 1 sample prior to recharge from well 17947 and earlier data from Knapton *et al* (2004) and Jeuken (2004)

[c] unsaturated zone – analyses from suction cup (depth 1 m) and RN 17945 (6-12 m)

[d] analyses from the well RN 17947

APPENDIX C: WATER QUALITY PARAMETERS MONITORED AT THE AZRI SITE*

Compound/ parameter	**NT LABS Recharge water (min monthly)**	**NT LABS 6 monthly GW**	**CSIRO ACU Intensive piezo. & soil solution samplers**	**Detection Method**	**Detection Limit**
General Compounds/Parameters					
TDS	√	√	√	(Derived from EC)	
EC	√	√	√	APHA 2520 B	1 mg/L
Turbidity	√	Not monitored	Not monitored	APHA 2130 B	0.1 NTU
Calcium	√	√	√	APHA 3120 B	0.1 mg/L
Magnesium	√	√	√	APHA 3120 B	0.3 mg/L
Potassium	√	√	√	APHA ICP 3120 B	1 mg/L
Sodium	√	√	√	APHA 3120 B	0.5 mg/L
Chloride	√	√	√	APHA 4500-Cl-E	4.0 mg/L
Bicarbonate	√	√	√	APHA 2320 B	1 mg/L
Sulphate	√	√	√	APHA ICP 3120 B	1.5 mg/L
Iodide	√	√		APHA ICP 3120 B	
Bromide	√	√		APHA ICP 3120 B	
pH	√	√	√	APHA 4500-H-B	0.1 pH units
Alkalinity	√	√	√	(Derived HCO_3)	
TOC/DOC	√	√	√	USEPA 9060A	0.3 mg/L
Fluoride		√		APHA 4500-F-C	0.1 mg/L
Boron	√	√		APHA 3120 B	0.04 mg/L
Silica	Not monitored	√	Not monitored		
Total Nitrogen/TKN	√	?	√	APHA 4500-N-ORG	0.05 mg/L
Ammonia	√	?	√	APHA 4500-NH_3H	0.005 mg/L
Nitrate	√	√	√	APHA 4500-NO_3F	0.005 mg/L
Nitrite	√	?	√	APHA 4500-NO_2B	0.005 mg/L
Total Phosphorous	√	?	√	APHA 4500-P-E	0.005 mg/L
Ortho-phosphate	√	?	√	APHA 4500-P-F	0.005 mg/L
BOD	√	Not monitored	Not monitored	?	
COD	√	Not monitored	Not monitored	?	
UV absorbance 254nm	√	Not monitored	Not monitored	?	
Metals					
Arsenic - total	√	√	√	USEPA 200.8	0.001 mg/L
Antimony	√	√	Not monitored	?	
Barium - total	√	√	√	USEPA 200.8	0.0005 mg/L
Berrylium - total	√	√	Not monitored	?	
Cadmium - total	√	√	Not monitored	USEPA 200.8	0.0005 mg/L
Chromium - total	√	√	Not monitored	?	
Copper - total	√	√	Not monitored	USEPA 200.8	0.001 mg/L
Iron - total	√	√ + dissolved	√	APHA 3120 B	0.03 mg/L
Lead - total	√	√	√	USEPA 200.8	0.0005 mg/L
Mercury - total	√	√	Not monitored	USEPA 200.8	0.0005 mg/L

Compound/ parameter	**NT LABS Recharge water (min monthly)**	**NT LABS 6 monthly GW**	**CSIRO ACU Intensive piezo. & soil solution samplers**	**Detection Method**	**Detection Limit**
Manganese – total	√	√ + dissolved	√	?	
Molybendum – total	√	√	Not monitored	?	
Nickel - total	√	√	√	?	
Silver - total	√	√	Not monitored	?	
Selenium	√	√	Not monitored	?	
Tin - total	√	√	Not monitored	?	
Uranium – total	√	√	Not monitored	?	
Zinc - total	√	√	Not monitored	?	
Pathogen Indicators					
E. coli	√	Not monitored		APHA 9225	/100ml
Total coliforms	√	Not monitored		APHA 9225	/100ml
Isotopes					
Deuterium & O18	√	√	CSIRO – Isotope laboratory Urrbrae, SA		
Organic chemicals					
THMs Triazine herbicides Organochlorine residues Organophosphate residues Volatile hydrocarbons Semi-volatile hydrocarbons Phenols (speciated) PAHs lower level Detergents (MBAS)	All (detailed list in appendix)	Not monitored	National Measurement Instiitute – Pymble, NSW		
Pharmaceuticals & NDMA	All (detailed list in appendix)		Qld Health Scientific Services		

* source water is also collected at the DAF site

APPENDIX D: HYDRAULIC LOADING, AVERAGE DAILY INFILTRATION AND INFILTRATION COEFFICIENT VALUES CALCULATED FOR INDIVIDUAL WETTING PERIODS USING THE MASS BALANCE (EQUATION 5)

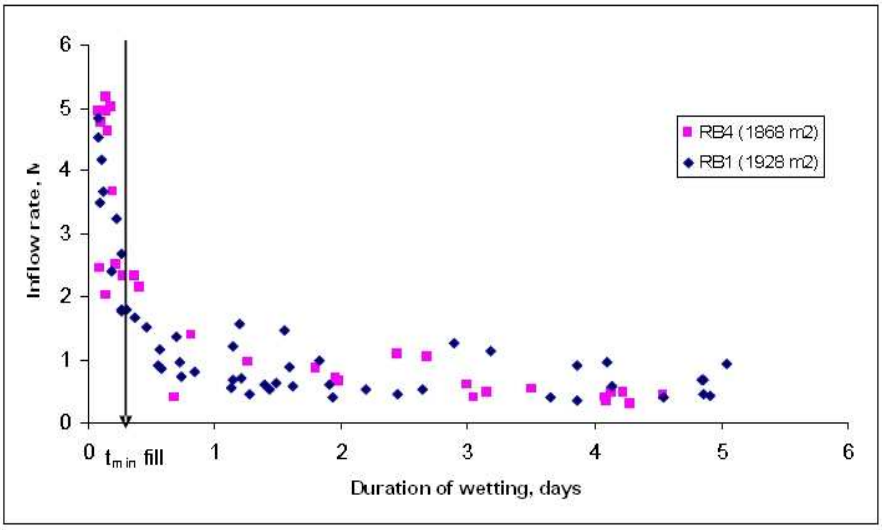
Basin	Date of wetting	Time of wetting* (days)	Amount of discharged water (m^3)	Hydraulic loading (mm)	Average infiltration (mm/day)	Infiltration coefficient
RB 1	12-16/06/2008	4.70	2030	1041	221	0.99
RB 1	16/6-5/07/2009	18.83	6160	3130	166	0.98
RB 1	26/8-14/09/2008	18.96	3352	1604	85	0.92
RB 1	24-30/09/2008	5.33	781	354	66	0.87
RB 1	23-27/01/2009	4.17	3325	1680	403	0.97
RB 1	23-25/03/2009	1.38	959	488	355	0.98
RB 1	29/5-1/06/2009	2.38	675	370	156	1.06
RB 1	30/7-4/08/2009	4.67	1085	542	116	0.96
RB 1	1-4/09/2009	2.79	694	344	123	0.96
RB 2	11/6-1/07/2008	20.0	9340	4898	245	0.99
RB 2	1-23/07/2008	21.9	5876	3034	139	0.97
RB 2	28/7-3/08/2008	5.50	1542	783	142	0.95
RB 2	3/8-3/09/2008	30.0	3800	1878	63	0.93
RB 2	24-25/09/2008	1.29	431	216	167	0.94
RB 2	3-7/05/2009	3.67	3396	1785	487	0.99
RB 2	14-18/06/2009	4.46	3733	1973	443	0.99
RB 2	24-28/07/2009	4.00	2384	1252	313	0.99
RB 2	31/7-4/08/2009	4.17	2024	1060	254	0.98
RB 2	2-6/09/2009	3.79	1569	809	213	0.97
RB 3	8-10/06/2008	2.04	900	453	222	0.99
RB 3	11-27/06/2008	15.7	4430	2203	140	0.98
RB 3	3-10/07/2008	7.00	2575	1287	184	0.98
RB 3	22/8-7/09/2008	16.1	3023	1453	90	0.94
RB 3	17-20/02/2009	2.88	2380	1177	409	0.97
RB 3	22-24/03/2009	2.25	1828	911	405	0.98
RB 3	3-9/05/2009	5.79	2650	1318	228	0.98
RB 3	23-27/07/2009	3.71	948	466	126	0.96
RB 3	29/8-1/09/2009	3.17	977	478	151	0.96
RB 4	10-14/07/2008	4.08	2058	1086	266	0.99
RB 4	1-8/08/2008	7.33	2082	1078	147	0.97
RB 4	18-25/08/2008	7.17	1948	1010	141	0.97
RB 4	29/8-5/09/2008	6.63	1998	1035	156	0.97
RB 4	23-27/09/2008	3.35	641	324	96	0.94
RB 4	11-13/02/2009	2.38	1394	726	306	0.97
RB 4	7-12/04/2009	4.38	1208	621	142	0.96
RB 4	19-24/04/2009	5.38	1518	777	145	0.96
RB 4	8-15/05/2009	6.88	1603	826	120	0.96
RB 4	20-26/07/2009	5.63	871	441	78	0.95

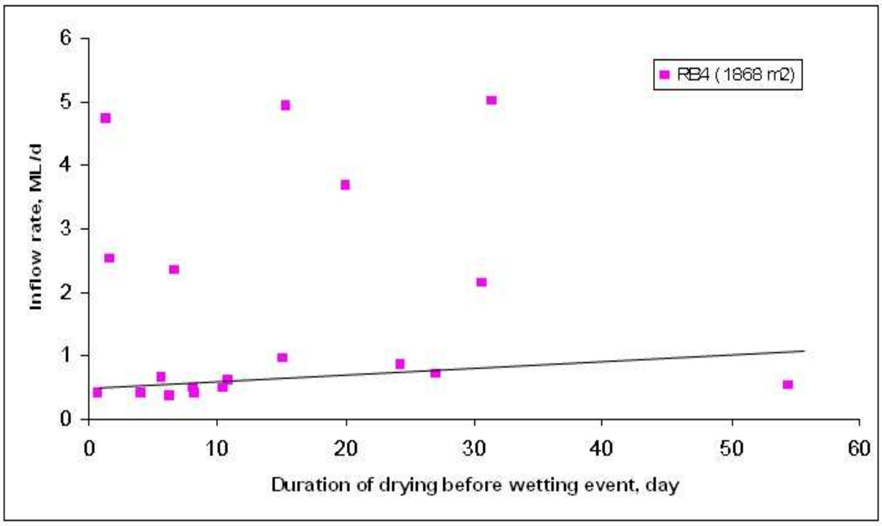
* until the basin drained

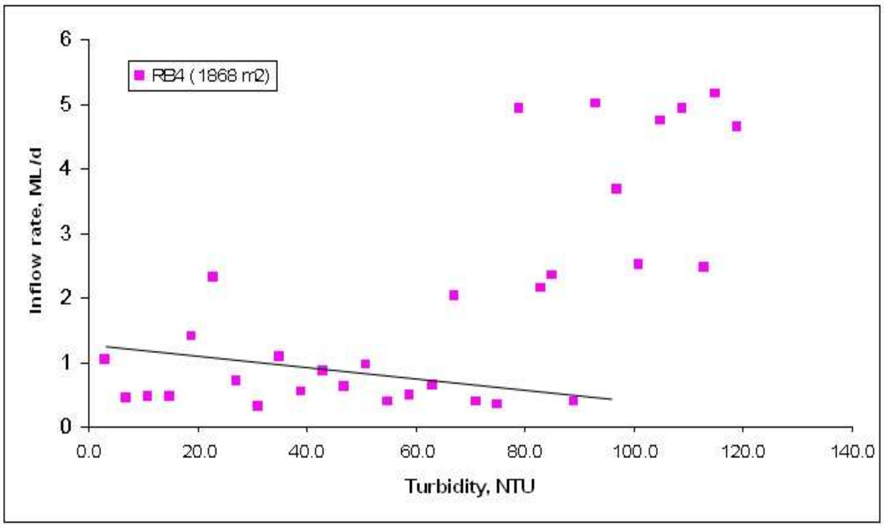
APPENDIX E: PERMEABILITY OF BASINS' BOTTOMS DETERMINED WITH DISC PERMEAMETER USING TAP WATER (ROE CREEK)

Time of measurement		K mm/hr	K m/d	K m/s	Existence of schmutzdecke	Comments
19/7/09 Basin 2 - Near dry						
Test 1	13:30	4.81	0.12	1.33E-06	Yes	NE side of basin
Test 2	14:00	68.51	1.64	1.90E-05	No	NE side of basin
Test 3	14:20	37.56	0.90	1.04E-05	Yes	NE side of basin
Test 4	15:00	51.83	1.24	1.41E-05	No	NE side of basin
Test 5	15:35	68.23	1.64	1.89E-05	Yes	centre of basin
Test 6	16:00	78.74	1.89	2.18E-05	No	centre of basin
Test 7	16:20	155.2	3.73	4.31E-05	Yes	centre of basin - top of furrow
Test 8	16:50	13.09	0.31	3.63E-06	No	centre of basin
Mean (1-8)			**1.45 m/d**	**Std dev 1.12**		
23/7/09 Basin 2 - draining						
Test 9	10:25	12.21	0.29	3.39E-06	Yes	
Test 10	10:55	17.93	0.43	4.98E-06	No	getting 'gusty'
Test 11	11:20	266.7	6.40	7.41E-05	Yes	cracked surface
Test 12	11:45	35.69	0.86	9.91E-06	No	
Test 13	12:15	87.10	2.09	2.42E-05	Yes (but dry)	
Test 14	12:30	106.2	2.55	2.95E-05	No	
Test 15	16:20	116.0	2.79	3.22E-05	No	from channel near pond inlet
Mean (9-15)			**2.2 m/d**	**Std dev 2.11**		
24/7/09 - Basin 2 drying						
Test 16	13:15	138.1	3.31	3.83E-05	Yes (but dry)	near inflowing channel
Test 17	13:35	206.1	4.95	5.72E-05	Yes (but dry and cracked)	
Test 18	13:45	287.9	6.91	7.99E-05	Yes (but dry and cracked)	
Mean (16-18)			**5.66 m/d**	**Std dev 1.8**		
Mean (1-18)			**2.34 m/d**	**Std dev 2.04**		
20/7/09 - Basin 4 still draining in one corner but drying other						
Test 1	8:40	10.69	0.26	2.97E-06	Yes	No bentonite seal at start
Test 2	9:10	23.16	0.56	6.43E-06	No	Windy conditions
Test 3	9:30	99.20	2.38	2.75E-05	Yes	
Test 4	9:55	60.33	1.45	1.67E-05	No	
Test 5	10:20	30.60	0.73	8.49E-06	Yes (tight)	
Test 6	10:45	61.59	1.48	1.71E-05	No	Very windy conditions
Mean (1-6)			**1.14 m/d**	**Std dev 0.78**		

APPENDIX F: INFLOW RATES OVER DURATION OF WETTING, DURATION OF DRYING, AND TURBIDITY

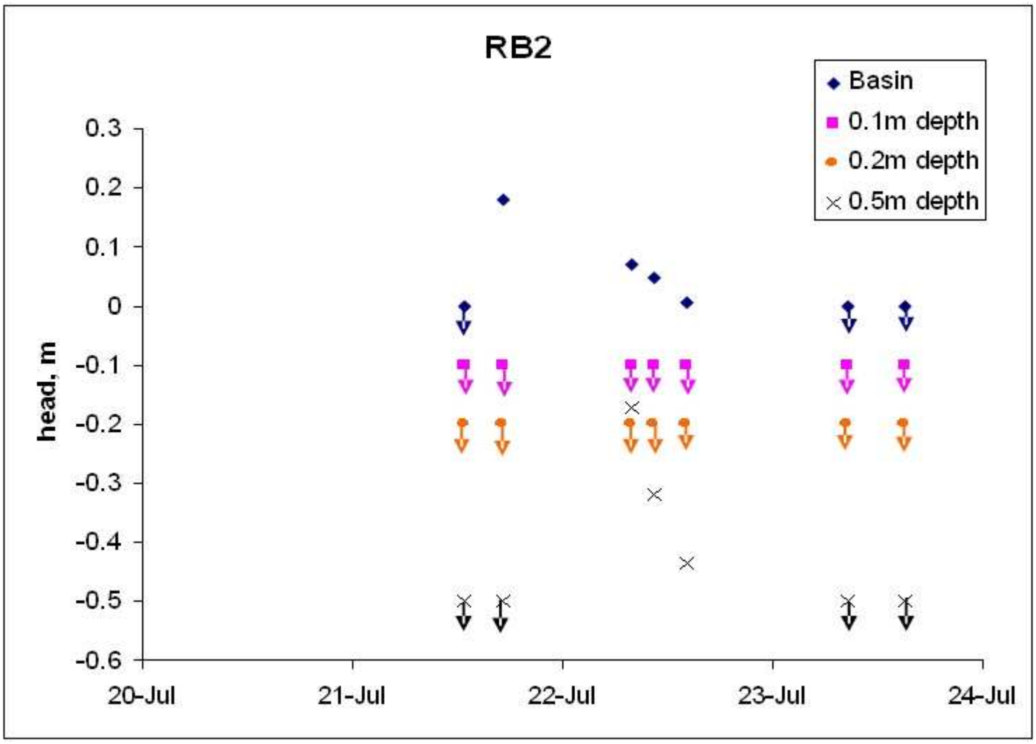


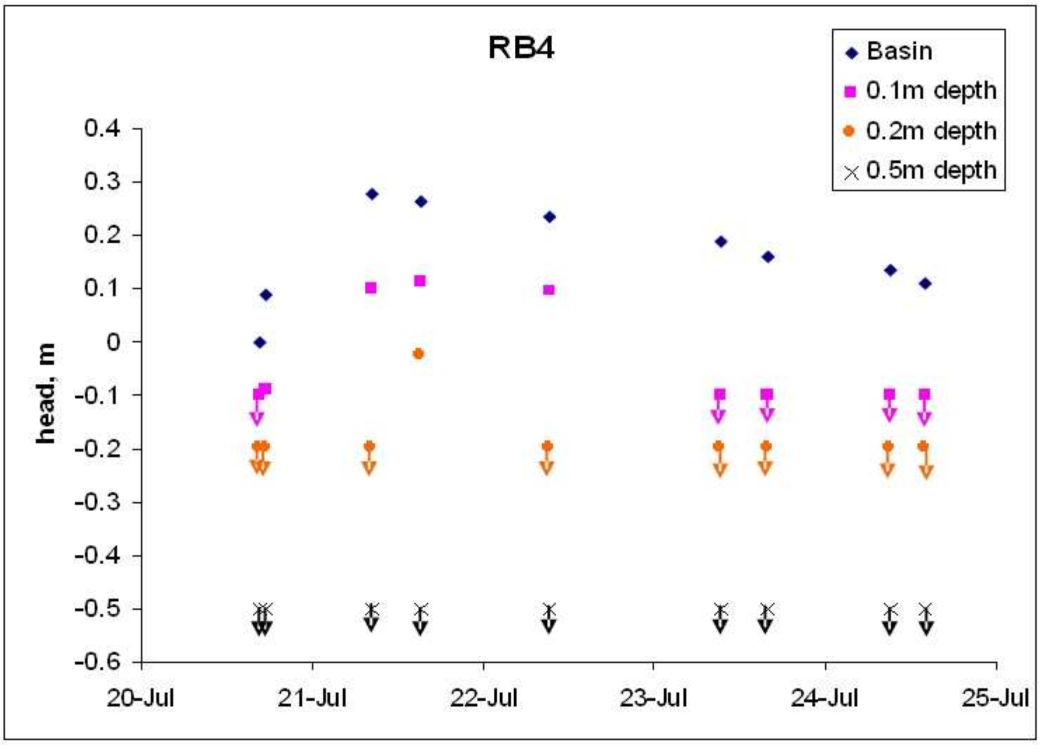




The values of inflow rate > 2 ML represent the inflow to basin's storage capacity. Areas of RB 1 and RB4 are 1928 and 1868 m^2, respectively.

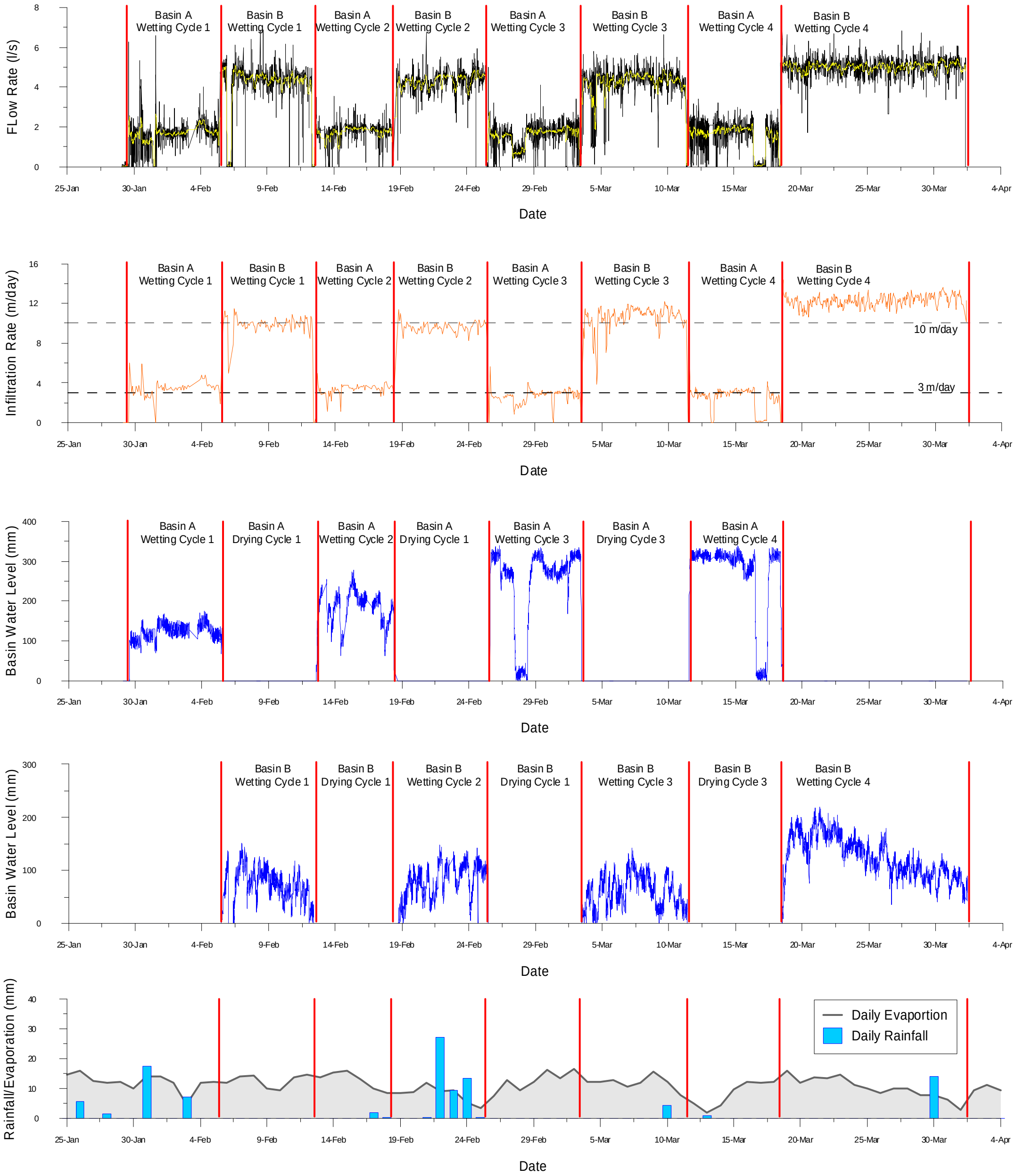
APPENDIX G: HYDRAULIC HEADS IN THE SHALLOW PIEZOMETERS BENEATH RB2 AND RB4 DURING THE FIELD CAMPAIGN IN JULY 2009





Head is measured relative to the floor of the basin. Arrow indicates piezometer is unsaturated.

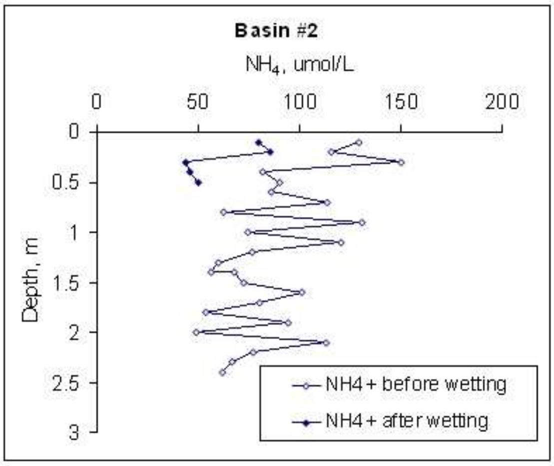
APPENDIX H: INFILTRATION TRIAL OF THE BASINS A AND B (6X6 M)

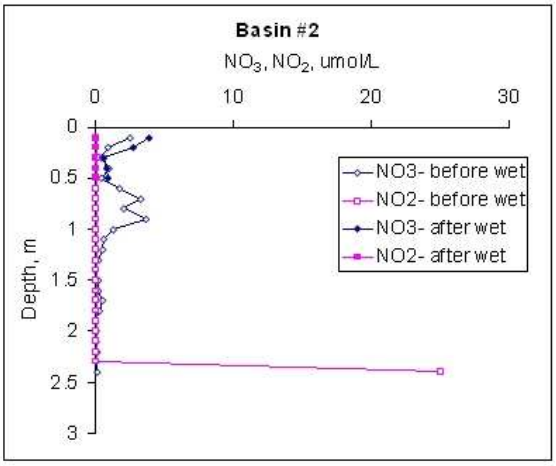


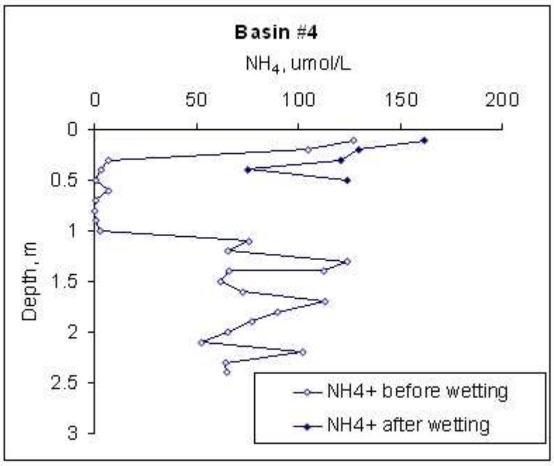
From Knapton *et al.* 2004

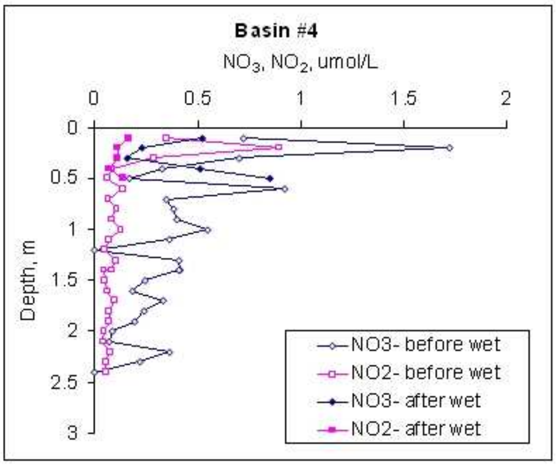
APPENDIX I: CONCENTRATION OF INORGANIC NITROGEN COMPOUNDS FROM SOIL CORES IN JULY 2009

Dates of sampling:
18/07/2009 – before wetting
24/07/2009 – Basin #2 after wetting
28/07/2009 – Basin #4 after wetting









NH_4 in soil cores were unable to be validated. Results are higher than expected. It is recommended that further sampling and analysis be undertaken.

APPENDIX J: PHARMACEUTICALS AND HERBICIDES IN RECHARGE WATER

(a) NDMA, Pharmaceutical and Herbicide results from samples submitted to the Queensland Health Scientific Services Laboratories (values detected at 5 times the detection limit are highlighted in bold)

Analyte	**Guideline Value (ADWG)**	**Health Value (ADWG)**	**Public Health Regulation Standard**	**LOR**	**SAR80 22/07/08**	**SAR80 (replicate sent via Alice Sp) 22/07/08**	**SAR80 (replicate sent via Adelaide) 22/07/09**	**SAR100 29/09/08**	**SAR100 17/03/09**	**SAR100 10/08/09**	**SAR100 26/10/09**	**SAR100 16/12/09**
Nitrosamines (ng/L)												
NDMA‡			10	5	<5	<5	<5	**11**	**7**	**46**	NR	NR
NDEA			10	10	<10	<10	<10	<10	<10	<10	NR	NR
Nitroso-pyrrolidine				10	<10	<10	<10	<10	NA*	<10	NR	NR
Nitroso-morpholine (NMOR)			1	10					<10	<10	NR	NR
Nitroso-piperidine				20	<20	<20	<20	<20	<20	<20	NR	NR
NDBA				20	<20	<20	<20	<20	<20	<20	NR	NR
%surrogate recovery								73	89	85	NR	NR
(Deuterated NDMA)										17	NR	NR
Pharmaceuticals (µg/L)												
Acetylsalicylic acid			29	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Atenolol			25	0.01	0.40	0.40	0.4	0.03	**0.08**	**0.23**	0.01	**0.05**
Atorvastatin			5	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Caffeine			0.35	0.01	**0.18**	**0.19**	**0.19**	**0.05**	**0.08**	**0.08**	**0.07**	**0.07**
Carbamazepine			100	0.01	**0.47**	**0.45**	**0.45**	**0.76**	**0.89**	**0.47**	**1.33**	**0.90**
Cephalexin			35	0.01	**0.07**	<0.01	0.01	**0.66**	<0.01	**0.09**	**0.16**	na
Chloramphenicol			175	0.1	<0.10	<0.10	<0.10	<0.10	<0.10	<0.1	<0.1	<0.1
Chlortetracycline			105	0.1	<0.10	<0.10	<0.10	<0.10	<0.10	<0.1	<0.1	<0.1
Ciprofloxacin			250	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Citalopram			4.0	0.01	0.02	0.02	0.02	0.03	0.02	**0.05**	**0.05**	0.03
Codeine			50	0.1	<0.10	<0.10	<0.10	<0.10	<0.10	<0.1	<0.1	<0.1
Cyclophosphamide			3.5	0.01				<0.01	<0.01	<0.01	<0.01	<0.01
Dapsone				0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
DEET			2500	0.01	0.03	0.03	0.03	0.02	**0.07**	0.03	0.04	0.04
Desmethyl Citalopram			4	0.01	0.02	0.02	0.02	0.01	0.02	0.01	0.01	0.03

Analyte	Guideline Value (ADWG)	Health Value (ADWG)	Public Health Regulation Standard	LOR	SAR80 22/07/08	SAR80 (replicate sent via Alice Sp) 22/07/08	SAR80 (replicate sent via Adelaide) 22/07/09	SAR100 29/09/08	SAR100 17/03/09	SAR100 10/08/09	SAR100 26/10/09	SAR100 16/12/09
Desmethyl Diazepam			3	0.01	<0.01	<0.01	<0.01	0.01	0.01	<0.01	0.01	<0.01
Diazepam			2.5	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Diclofenac			2	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Doxylamine				0.01	0.04	0.03	0.03	**0.12**	0.02	**0.07**	**0.15**	**0.09**
Enrofloxacin			22	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Erythromycin			17.5	0.01	0.02	0.02	0.01	0.01	<0.01	0.02	0.01	<0.01
Erythromycin anhydrate				0.01						0.03	0.03	<0.01
Fluoxetine			10	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Fluvastatin				0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Frusemide			10	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Gabapentin				0.01	**4.80**	**5.00**	**4.6**	**0.15**	**3.55**	**7.50**	**5.60**	**6.40**
Gemfibrozol			600	0.01	**0.86**	**0.81**	**0.8**	**0.61**	**0.26**	**0.32**	**0.42**	**0.36**
Hydrochlorthiazide			12.5	0.01	0.02	0.01	0.02	<0.01	<0.01	0.02	<0.01	<0.01
Ibuprofen			400	0.01	**0.09**	**0.09**	**0.09**	<0.01	<0.01	**0.13**	<0.01	<0.01
Ifosfamide			3.5	0.01				<0.01	<0.01	<0.01	<0.01	<0.01
Indomethacin			25	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.01	<0.01	<0.01
Iopromide			750	0.2	<0.20	<0.20	<0.20	<0.20	<0.20	<0.2	<0.2	<0.2
Lincomycin			3500	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Metoprolol			25	0.01	**0.05**	**0.06**	**0.05**	**0.05**	0.02	**0.13**	**0.07**	**0.05**
Naproxen			220	0.1	0.13	0.14	0.11	<0.10	0.23	0.28	<0.1	<0.1
Norfloxacin			400	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Oxazepam			15	0.01	**0.09**	**0.08**	**0.08**	**0.08**	**0.08**	**0.09**	**0.10**	**0.05**
Oxycodone			10	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Oxytetracycline			105	0.1	<0.10	<0.10	<0.10	<0.10	<0.10	<0.1	<0.1	<0.1
Paracetamol			175	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Phenytoin				0.01	**0.21**	**0.19**	**0.19**	**0.26**	**0.31**	**0.13**	**0.27**	**0.27**
Praziquantel			70	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Primidone				0.01						**0.10**	**0.19**	**0.24**
Propranolol			40	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Ranitidine			26	0.01	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Roxithromycin			150	0.01	**0.07**	**0.07**	**0.07**	0.01	<0.01	**0.08**	0.01	<0.01
Salicylic acid			105	0.01	0.01	0.01	0.01	0.01	0.01	<0.01	<0.01	<0.01

Analyte	**Guideline Value (ADWG)**	**Health Value (ADWG)**	**Public Health Regulation Standard**	**LOR**	**SAR80 22/07/08**	**SAR80 (replicate sent via Alice Sp) 22/07/08**	**SAR80 (replicate sent via Adelaide) 22/07/09**	**SAR100 29/09/08**	**SAR100 17/03/09**	**SAR100 10/08/09**	**SAR100 26/10/09**	**SAR100 16/12/09**
Sertraline				0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Simvastatin				0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Sulfsalazine			500	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Sulphadiazine			35	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Sulphamethoxazole			35	0.01	**0.08**	**0.09**	**0.09**	0.04	**0.05**	**0.08**	**0.05**	0.01
Sulphathiazole			35	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Temazepam			5	0.01	**0.11**	**0.10**	**0.1**	**0.11**	**0.14**	**0.11**	**0.14**	**0.08**
Tetracycline			105	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.02
Tramadol				0.01	**0.29**	**0.32**	**0.32**	**0.35**	**0.23**	**0.26**	**0.40**	**0.39**
Triclosan			0.35	0.01	0.01	0.01	0.01	<0.01	<0.01	<0.01	<0.01	0.02
Trimethoprim			70	0.01	**0.06**	**0.06**	**0.06**	0.02	0.02	0.04	0.01	0.02
Tylosin			1050	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Venlafaxine			75	0.01	**0.15**	**0.16**	**0.16**	**0.12**	**0.09**	**0.17**	**0.15**	**0.09**
Warfarin			1.5	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Phenoxyacid Herbicides (µg/l)												
Dicamba		100	100	0.01	**0.24**	**0.24**	**0.24**	**0.07**	<0.01	0.03	<0.01	<0.01
Mecoprop			10	0.01	0.01	0.01	0.01	0.02	0.02	<0.01	0.01	0.01
MCPA			2	0.01	**0.43**	**0.44**	**0.44**	**0.08**	<0.01	0.02	0.02	0.03
2,4-DP			100	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
2,4-D	0.1	30	30	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	**0.12**	0.01	<0.01
Triclopyr		10	10	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
MCPB				0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Fluroxypyr			700	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
2,4-DB				0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Picloram		300	300	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Dalapon		500	500	0.01	<0.01	<0.01	<0.01	0.04	<0.01	0.01	0.02	0.03
Herbicides (µg/L)												
Ametryn	5	50	50	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Atrazine	0.1	40	40	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Bromacil	10	300	300	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Carbaryl	5	30	30	0.01							<0.01	<0.02
Desethyl Atrazine			40	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01

Analyte	Guideline Value (ADWG)	Health Value (ADWG)	Public Health Regulation Standard	LOR	SAR80 22/07/08	SAR80 (replicate sent via Alice Sp) 22/07/08	SAR80 (replicate sent via Adelaide) 22/07/09	SAR100 29/09/08	SAR100 17/03/09	SAR100 10/08/09	SAR100 26/10/09	SAR100 16/12/09
Desisopropyl Atrazine			40	0.01	<0.01	<0.01	<0.01	<0.01	0.02	<0.01	0.03	<0.01
Diazinon	1	3	3	0.02							<0.02	<0.02
Diuron		30	30	0.01	**0.08**	**0.08**	**0.08**	**0.18**	**0.15**	**0.08**	**0.32**	**0.25**
3,4 Dichloroaniline			0.35	0.01						0.01	**0.05**	0.02
Fluometuron		50	50	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Hexazinone	2	300	300	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Metolachlor	2	300	300	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Prometryn			105	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Propoxur			70	0.01							<0.01	<0.01
Simazine	0.5	20	20	0.01	**0.06**	**0.06**	**0.05**	**0.05**	**0.14**	0.02	**0.12**	**0.12**
Tebuthiuron				0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Terbutryn	1	300	300	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Haloxyfop (acid)		5	1.05	0.01						<0.01	<0.01	<0.01
Haloxyfop-2-etotyl			1.05	0.01						<0.01	<0.01	<0.01
Haloxyfop-methyl			0.175	0.01						<0.01	<0.01	<0.01
Total Haloxyfop		5	1.05	0.03						<0.03	<0.03	<0.03

(b) Trace organic results from samples submitted to the National Measurement Institute Laboratories (values detected are highlighted in bold)

		SAR80	**SAR100**	**SAR100**	**SAR100**	**SAR100**
		24/07/2008	**29/09/2008**	**8/12/2008**	**26/10/2009**	**14/12/2009**
ALL DETECTS						
Miscellaneous (mg/L)						
Detergent as MBAS	NW_S6	**0.09**		**0.1**		**0.09**
Trihalomethanes (ug/L)						
Dibromochloromethane	NGCMS_1120	<1	<1	<1	<1	<1
Dichlorobromomethane	NGCMS_1120	<1	<1	<1	<1	<1
Tribromomethane	NGCMS_1120	<1	<1	<1	<1	<1
Trichloromethane	NGCMS_1120	<1	**3.2**	<1	<1	<1
Trihalomethanes NMI 1120 Screen (µg/L)						
Chloroform	NGCMS_1120	<1	**3.2**	**17**	<1	<1
Bromodichloromethane	NGCMS_1120	<1	<1	**7.9**	<1	<1
Dibromochloromethane	NGCMS_1120	<1	<1	**2**	<1	<1
Bromoform	NGCMS_1120	<1	<1	<1	<1	<1
ALL REMAINING – Non Detects						
Poly Aromatic Hydrocarbons (µg/L)						
Naphthalene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Acenaphthylene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Acenaphthene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Fluorene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Phenanthrene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Anthracene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Fluoranthene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Pyrene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Benz(a)anthracene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Chrysene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Benzo(b)&(k)fluoranthene	NGCMS_1111	<0.02	<0.02	<0.02	<0.02	<0.02
Benzo(a)pyrene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Indeno(1,2,3-cd)pyrene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Dibenz(ah)anthracene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Benzo(ghi)perylene	NGCMS_1111	<0.01	<0.01	<0.01	<0.01	<0.01
Organochlorine (OC) Pesticides (µg/L)						
HCB	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Heptachlor	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Heptachlor epoxide	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Aldrin	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
gamma-BHC (Lindane)	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
alpha-BHC	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
beta-BHC	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
delta-BHC	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
trans-Chlordane	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
cis-Chlordane	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Oxychlordane	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Dieldrin	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
p,p-DDE	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
p,p-DDD	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
p,p-DDT	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Endrin	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Endrin Aldehyde	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Endrin Ketone	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
alpha-Endosulfan	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
beta-Endosulfan	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Endosulfan Sulfate	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Methoxychlor	NR_19	<0.01	<0.01	<0.01	<0.01	<0.01
Phenols (µg/L)						

		SAR80 24/07/2008	SAR100 29/09/2008	SAR100 8/12/2008	SAR100 26/10/2009	SAR100 14/12/2009
Phenol	NGCMS_1111	<0.1	<0.1	<0.1	<0.1	<0.1
2-Chlorophenol	NGCMS_1111	<0.1	<0.1	<0.1	<0.1	<0.1
2-Methylphenol	NGCMS_1111	<0.1	<0.1	<0.1	<0.1	<0.1
2,4-Dichlorophenol	NGCMS_1111	<0.1	<0.1	<0.1	<0.1	<0.1
3-& 4-Methylphenols	NGCMS_1111	<0.2	<0.2	<0.2	<0.2	<0.2
2,4-Dimethyphenol	NGCMS_1111	<0.1	<0.1	<0.1	<0.1	<0.1
2,6-Dichlorophenol	NGCMS_1111	<0.1	<0.1	<0.1	<0.1	<0.1
2-Nitrophenol	NGCMS_1111	<0.1	<0.1	<0.1	<0.1	<0.1
4-Chloro-3-methylphenol	NGCMS_1111	<0.2	<0.2	<0.2	<0.2	<0.2
2,4,6-Trichlorophenol	NGCMS_1111	<0.2	<0.2	<0.2	<0.2	<0.2
4-Nitrophenol	NGCMS_1111	<0.1	<0.1	<0.1	<0.1	<0.1
2,4,5-Trichlorophenol	NGCMS_1111	<0.2	<0.2	<0.2	<0.2	<0.2
2,3,4,6-Tetrachlorophenol	NGCMS_1111	<0.2	<0.2	<0.2	<0.2	<0.2
Pentachlorophenol	NGCMS_1111	<0.2	<0.2	<0.2	<0.2	<0.2
Triazine Herbicides (µg/L)						
Atrazine	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Hexazinone	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Metribuzine	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Prometryne	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Simazine	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Organophosphate (OP) Pesticides (µg/L)						
Demeton-S-Methyl	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Dichlorvos	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Diazinon	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Dimethoate	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Chlorpyrifos	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Chlorpyrifos Methyl	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Malathion	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Fenthion	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Azinphos Ethyl	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Azinphos Methyl	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Chlorfenvinphos (E)	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Chlorfenvinphos (Z)	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Ethion	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Fenitrothion	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Parathion (Ethyl)	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Parathion Methyl	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Pirimiphos Ethyl	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Pirimiphos Methyl	NR_19	<0.1	<0.1	<0.1	<0.1	<0.1
Monocyclic Aromatic Hydrocarbons NMI 1120 Screen (µg/L)						
Benzene	NGCMS_1120	<1	<1	<1	<1	<1
Toluene	NGCMS_1120	<1	<1	<1	<1	<1
Ethylbenzene	NGCMS_1120	<1	<1	<1	<1	<1
m & p-Xylenes	NGCMS_1120	<2	<2	<2	<2	<2
o-Xylene	NGCMS_1120	<1	<1	<1	<1	<1
Styrene	NGCMS_1120	<1	<1	<1	<1	<1
Isopropylbenzene	NGCMS_1120	<1	<1	<1	<1	<1
n-Propylbenzene	NGCMS_1120	<1	<1	<1	<1	<1
1,3,5-Trimethylbenzene	NGCMS_1120	<1	<1	<1	<1	<1
tert-Butylbenzene	NGCMS_1120	<1	<1	<1	<1	<1
1,2,4-Trimethylbenzene	NGCMS_1120	<1	<1	<1	<1	<1
sec-Butylbenzene	NGCMS_1120	<1	<1	<1	<1	<1
4-Isopropyltoluene	NGCMS_1120	<1	<1	<1	<1	<1
n-Butylbenzene	NGCMS_1120	<1	<1	<1	<1	<1
Halogenated Aliphatics Hydrocarbons NMI 1120 Screen (µg/L)						
Chloromethane	NGCMS_1120	<1	<1	<1	<1	<1

		SAR80 24/07/2008	SAR100 29/09/2008	SAR100 8/12/2008	SAR100 26/10/2009	SAR100 14/12/2009
Vinyl chloride	NGCMS_1120	<1	<1	<1	<1	<1
Bromomethane	NGCMS_1120	<1	<1	<1	<1	<1
Chloroethane	NGCMS_1120	<1	<1	<1	<1	<1
Trichlorofluoromethane	NGCMS_1120	<1	<1	<1	<1	<1
1,1-Dichloroethane	NGCMS_1120	<1	<1	<1	<1	<1
Dichloromethane	NGCMS_1120	<1	<1	<1	<1	<1
trans-1,2-Dichloroethene	NGCMS_1120	<1	<1	<1	<1	<1
1,1-Dichloroethene	NGCMS_1120	<1	<1	<1	<1	<1
2,2-Dichloropropane	NGCMS_1120	<1	<1	<1	<1	<1
cis-1,2-Dichloroethene	NGCMS_1120	<1	<1	<1	<1	<1
Bromochloromethane	NGCMS_1120	<1	<1	<1	<1	<1
1,1,1-Trichloroethane	NGCMS_1120	<1	<1	<1	<1	<1
Carbon tetrachloride	NGCMS_1120	<1	<1	<1	<1	<1
1,1-Dichloropropene	NGCMS_1120	<1	<1	<1	<1	<1
1,2-Dichloroethane	NGCMS_1120	<1	<1	<1	<1	<1
Trichloroethene	NGCMS_1120	<1	<1	<1	<1	<1
1,2-Dichloropropane	NGCMS_1120	<1	<1	<1	<1	<1
Dibromomethane	NGCMS_1120	<1	<1	<1	<1	<1
cis-1,3-Dichloropropene	NGCMS_1120	<1	<1	<1	<1	<1
trans-1,3-Dichloropropene	NGCMS_1120	<1	<1	<1	<1	<1
1,1,2-Trichloroethane	NGCMS_1120	<1	<1	<1	<1	<1
Tetrachloroethene	NGCMS_1120	<1	<1	<1	<1	<1
1,3-Dichloropropane	NGCMS_1120	<1	<1	<1	<1	<1
1,2-Dibromoethane	NGCMS_1120	<1	<1	<1	<1	<1
1,1,1,2-Tetrachloroethane	NGCMS_1120	<1	<1	<1	<1	<1
1,1,2,2-Tetrachloroethane	NGCMS_1120	<1	<1	<1	<1	<1
1,2,3-Trichloropropane	NGCMS_1120	<1	<1	<1	<1	<1
1,2-Dibromo-3-chloropropane	NGCMS_1120	<1	<1	<1	<1	<1
Hexachlorobutadiene	NGCMS_1120	<1	<1	<1	<1	<1
Halogenated Aromatics Hydrocarbons NMI 1120 Screen (ug/L)						
Chlorobenzene	NGCMS_1120	<1	<1	<1	<1	<1
Bromobenzene	NGCMS_1120	<1	<1	<1	<1	<1
2-Chlorotoluene	NGCMS_1120	<1	<1	<1	<1	<1
4-Chlorotoluene	NGCMS_1120	<1	<1	<1	<1	<1
1,3-Dichlorobenzene	NGCMS_1120	<1	<1	<1	<1	<1
1,4-Dichlorobenzene	NGCMS_1120	<1	<1	<1	<1	<1
1,2-Dichlorobenzene	NGCMS_1120	<1	<1	<1	<1	<1
1,2,4-Trichlorobenzene	NGCMS_1120	<1	<1	<1	<1	<1
1,2,3-Trichlorobenzene	NGCMS_1120	<1	<1	<1	<1	<1
Polycyclic Aromatic Hydrocarbons(volatile) NMI 1120 Screen (µg/L)						
Naphthalene	NGCMS_1120	<1	<1	<1	<1	<1
Poly Aromatic Hydrocarbons NMI 1122 Screen (µg/L)						
Acenaphthylene	NGCMS_1122	<10	<10	<10	<10	<10
Naphthalene	NGCMS_1122	<10	<10	<10	<10	<10
Acenaphthene	NGCMS_1122	<10	<10	<10	<10	<10
Fluorene	NGCMS_1122	<10	<10	<10	<10	<10
Phenanthrene	NGCMS_1122	<10	<10	<10	<10	<10
Anthracene	NGCMS_1122	<10	<10	<10	<10	<10
Fluoranthene	NGCMS_1122	<10	<10	<10	<10	<10
Pyrene	NGCMS_1122	<10	<10	<10	<10	<10
Benz(a)anthracene	NGCMS_1122	<10	<10	<10	<10	<10
Chrysene	NGCMS_1122	<10	<10	<10	<10	<10
Benzo(b,k)fluoranthene	NGCMS_1122	<20	<20	<20	<20	<20
Benzo(a)pyrene	NGCMS_1122	<10	<10	<10	<10	<10
Indeno(1,2,3-cd)pyrene	NGCMS_1122	<10	<10	<10	<10	<10
Dibenz(a,h)anthracene	NGCMS_1122	<10	<10	<10	<10	<10

		SAR80	**SAR100**	**SAR100**	**SAR100**	**SAR100**
		24/07/2008	**29/09/2008**	**8/12/2008**	**26/10/2009**	**14/12/2009**
Benzo(g,h,i)perylene	NGCMS_1122	<10	<10	<10	<10	<10
Phenols NMI 1122 Screen (µg/L)						
Phenol	NGCMS_1122	<10	<10	<10	<10	<10
2-Chlorophenol	NGCMS_1122	<10	<10	<10	<10	<10
2-Methylphenol	NGCMS_1122	<10	<10	<10	<10	<10
3&4-Methylphenol	NGCMS_1122	<20	<20	<20	<20	<20
2-Nitrophenol	NGCMS_1122	<10	<10	<10	<10	<10
2,4-Dimethylphenol	NGCMS_1122	<10	<10	<10	<10	<10
2,4-Dichlorophenol	NGCMS_1122	<10	<10	<10	<10	<10
2,6-Dichlorophenol	NGCMS_1122	<10	<10	<10	<10	<10
4-Chloro-3-methylphenol	NGCMS_1122	<20	<20	<20	<20	<20
2,4,5-Trichlorophenol	NGCMS_1122	<20	<20	<20	<20	<20
2,4,6-Trichlorophenol	NGCMS_1122	<20	<20	<20	<20	<20
2,3,4,6-Tetrachlorophenol	NGCMS_1122	<20	<20	<20	<20	<20
Pentachlorophenol	NGCMS_1122	<20	<20	<20	<20	<20
Oxygenated Compounds NMI 1120 Screen (µg/L)						
Acetone	NGCMS_1120	<10	<10	<10	<10	<10
Vinylacetate	NGCMS_1120	<10	<10	<10	<10	<10
2-Butanone (MEK)	NGCMS_1120	<10	<10	<10	<10	<10
4-Methyl-2-pentanone (MIBK)	NGCMS_1120	<10	<10	<10	<10	<10
2-Hexanone (MBK)	NGCMS_1120	<10	<10	<10	<10	<10
Sulfonated Compounds NMI 1120 Screen (µg/L)						
Carbon disulfide	NGCMS_1120	<10	<10	<10	<10	<10
Phthalates NMI 1122 Screen (µg/L)						
Dimethyl phthalate	NGCMS_1122	<10	<10	<10	<10	<10
Diethyl phthalate	NGCMS_1122	<10	<10	<10	<10	<10
Di-n-butyl phthalate	NGCMS_1122	<10	<10	<10	<10	<10
Butyl benzyl phthalate	NGCMS_1122	<10	<10	<10	<10	<10
Bis(2-ethylhexyl) phthalate	NGCMS_1122	<20	<20	<20	<20	<20
Di-n-octyl phthalate	NGCMS_1122	<10	<10	<10	<10	<10
Chlorinated Hydrocarbons NMI 1122 Screen (µg/L)						
2-Chloronaphthalene	NGCMS_1122	<20	<20	<20	<20	<20
1,4-Dichlorobenzene	NGCMS_1122	<20	<20	<20	<20	<20
1,2-Dichlorobenzene	NGCMS_1122	<20	<20	<20	<20	<20
1,3-Dichlorobenzene	NGCMS_1122	<20	<20	<20	<20	<20
Hexachlorobenzene	NGCMS_1122	<20	<20	<20	<20	<20
1,2,4-Trichlorobenzene	NGCMS_1122	<20	<20	<20	<20	<20
Hexachloroethane	NGCMS_1122	<20	<20	<20	<20	<20
Hexachlorocyclopentadiene	NGCMS_1122	<20	<20	<20	<20	<20
Hexachloro-1,3-butadiene	NGCMS_1122	<20	<20	<20	<20	<20
Ethers NMI 1122 Screen (µg/L)						
4-Bromophenyl phenyl ether	NGCMS_1122	<20	<20	<20	<20	<20
4-Chlorophenyl phenyl ether	NGCMS_1122	<20	<20	<20	<20	<20
Bis(2-chloroethyl)ether	NGCMS_1122	<20	<20	<20	<20	<20
Bis(2-chloroethoxy)methane	NGCMS_1122	<20	<20	<20	<20	<20
Bis(2-chloroisopropyl)ether	NGCMS_1122	<20	<20	<20	<20	<20
Amines Nitroaromatics & Nitrosamines NMI 1122 Screen (µg/L)						
Azobenzene	NGCMS_1122	<20	<20	<20	<20	<20
2,4-Dinitrotoluene	NGCMS_1122	<20	<20	<20	<20	<20
2,6-Dinitrotoluene	NGCMS_1122	<20	<20	<20	<20	<20
Nitrobenzene	NGCMS_1122	<20	<20	<20	<20	<20
N-Nitrosodimethylamine	NGCMS_1122	<20	<20	<20	<20	<20
N-Nitrosodiphenylamine	NGCMS_1122	<20	<20	<20	<20	<20
N-Nitrosodi-n-propylamine	NGCMS_1122	<20	<20	<20	<20	<20
Aniline	NGCMS_1122	<20	<20	<20	<20	<20
4-Chloroaniline	NGCMS_1122	<20	<20	<20	<20	<20

		SAR80 24/07/2008	SAR100 29/09/2008	SAR100 8/12/2008	SAR100 26/10/2009	SAR100 14/12/2009
2-Nitroaniline	NGCMS_1122	<10	<10	<10	<10	<10
3-Nitroaniline	NGCMS_1122	<20	<20	<20	<20	<20
4-Nitroaniline	NGCMS_1122	<20	<20	<20	<20	<20
Organochlorine Pesticides NMI 1122 Screen (µg/L)						
Aldrin	NGCMS_1122	<20	<20	<20	<20	<20
a-BHC	NGCMS_1122	<20	<20	<20	<20	<20
b-BHC	NGCMS_1122	<20	<20	<20	<20	<20
g-BHC (Lindane)	NGCMS_1122	<20	<20	<20	<20	<20
d-BHC	NGCMS_1122	<20	<20	<20	<20	<20
4,4,'-DDD	NGCMS_1122	<20	<20	<20	<20	<20
4,4,'-DDE	NGCMS_1122	<20	<20	<20	<20	<20
4,4,'-DDT	NGCMS_1122	<20	<20	<20	<20	<20
Dieldrin	NGCMS_1122	<20	<20	<20	<20	<20
Endosulphan I	NGCMS_1122	<20	<20	<20	<20	<20
Endosulphan II	NGCMS_1122	<20	<20	<20	<20	<20
Endosulfan sulphate	NGCMS_1122	<20	<20	<20	<20	<20
Endrin	NGCMS_1122	<20	<20	<20	<20	<20
Endrin Aldehyde	NGCMS_1122	<20	<20	<20	<20	<20
Heptachlor	NGCMS_1122	<20	<20	<20	<20	<20
Heptachlorepoxide	NGCMS_1122	<20	<20	<20	<20	<20
Organophosphate Pesticides NMI 1122 Screen (µg/L)						
Dimethoate	NGCMS_1122	<20	<20	<20	<20	<20
Diazinon	NGCMS_1122	<20	<20	<20	<20	<20
Fenitrothion	NGCMS_1122	<20	<20	<20	<20	<20
Malathion	NGCMS_1122	<20	<20	<20	<20	<20
Chlorpyrifos	NGCMS_1122	<20	<20	<20	<20	<20
Ethion	NGCMS_1122	<20	<20	<20	<20	<20
Others (µg/L)						
Dichlorobenzidine	NGCMS_1122	<20	<20	<20	<20	<20
2-Methylnaphthalene	NGCMS_1122	<10	<10	<10	<10	<10
Isophorone	NGCMS_1122	<20	<20	<20	<20	<20
Benzyl alcohol	NGCMS_1122	<20	<20	<20	<20	<20
Carbazole	NGCMS_1122	<20	<20	<20	<20	<20
Dibenzofuran	NGCMS_1122	<20	<20	<20	<20	<20

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