Feasibility of a Pilot 600ML/yr Soil Aquifer Treatment Plant at the Arid Zone Research Institute

A Report to Power and Water Corporation

Anthony Knapton & Peter Jolly

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A report prepared by DIPE and CSIRO for the Power & Water Corporation

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EXECUTIVE SUMMARY

This report presents a conceptual design of a Soil Aquifer Treatment (SAT) system, operating at the Arid Zone Research Institute (AZRI) receiving the recycled water product of a “pre-treatment plant” treating effluent from the Alice Springs Water Stabilisation Ponds (WSP) at a rate comparable to an annual total of 600 ML.

This design, including the operational parameters, has been developed on the basis of available literature regarding SAT schemes in similar operational settings and approximately 12 months of site and process investigations undertaken at the AZRI site.

The sum of these site and international comparative assessments is such that from a ‘technical’ perspective, “there is a high degree of confidence that a scheme of this capacity, at this location, will operate without adverse operational or environmental outcomes. Present indications of the scheme capacity range from 600 to 1800 ML/annum with higher rates requiring some recovery for subsequent reuse”.

Operation of the scheme at the proposed scale of 600 ML/yr for several years is considered to be a minimum requirement to;

— inform the fine-tuning of both the “treatment” and “recovery” efficiency of the scheme, and
— establish the appropriate regulatory and monitoring tools to facilitate long term operational management.

An important operational aspect, that can only be determined through a significant scale operation over a relatively long period (12 to 24 months), is the optimal loadings and recovery regimes that maximise the long term water quality outcomes in relation to the existing “active” hydrogeological and hydrological setting at AZRI.

It is therefore recommended that full scale field trials of the SAT scheme are undertaken as soon as practicable to better define the long term operational treatment and/or disposal attributes at this location.
INTRODUCTION

Feasibility assessment of a Soil Aquifer Treatment (SAT) scheme was initiated in July 2003. This occurred within the ongoing context of the Alice Springs Urban Water Reuse Strategy in response to a regulatory timeframe being applied requiring cessation of overflow from the Alice Springs Waste Stabilisation Ponds (WSP) to Ilparpa Swamp by the end of 2005.

This assessment built on previous desktop assessments and comprised significant site investigations at the Arid Zone Research Institute (AZRI), located in the western portion of the “Outer Farm Area”. Works entailed a series of laboratory studies on clogging, site soil and hydrogeological characterisation, field infiltration trials, conceptual groundwater modelling of scheme operations and review of the literature on processes affecting quality of infiltrated water in the subsurface.

These works sought to inform the design of a SAT scheme operating at the 600 ML/yr scale, due to an assessment being made that such scale represents;

1. “practical” short-term relief to the WSP overflow to Ilparpa for average climate conditions,
2. a realistic scale of operation to “test” the longer term capacity at this location,
3. a scale realistically manageable in the context of present regulatory settings,
4. an opportunity to test and evaluate active management/monitoring capacity, and
5. a “staging” size with economy of scale benefits in relation to other civil construction options considered in relation to the regulatory requirement for cessation of overflows to Ilparpa Swamp.

This document outlines in summary form the conceptual design within a tested range of parameters and also proposes a framework for the management and optimisation of the scheme over time.

This account is provided in two sections to cater for different reading audiences. Section 1, comprising three summary technical appendices, provides a summary assessment of important technical aspects of this design. Section 2 provides a more complete technical statement of the works and assessments support the summary accounts of Section 1.
CONCEPTUAL DESIGN

A brief introduction to soil aquifer treatment (SAT)

Soil-Aquifer Treatment (SAT) is a proven technology for the storage, treatment and recovery of recycled water producing high quality product water, which, in some cases is used for direct potable use. Influent recycled water is intermittently ponded in shallow basins with wetting and drying cycles of days to several weeks duration depending on site characteristics. As the infiltrate moves to the watertable below the basins, the soil acts as a natural treatment process to reduce the physical, chemical and microbial constituents of the infiltrating water. Treatment also occurs while the water moves through the aquifer in the saturated zone. Treatment occurs through filtration, adsorption onto soil of inorganic and organic substances, and removal through in-situ reactions or degradation by in-situ soil microorganisms.

Internationally, SAT has proved very effective in treating recycled water to quite high quality standards such that direct human contact with the product waters is possible while the “waste” by-product, being the filtrate mats formed within the ponds, is often suitable for reuse as a biosolid.

Treated waters are then recovered for reuse from nearby bores or spear points which can be stored or immediately used without odour generation or re-contamination concerns.

AZRI SAT design parameters

The design parameters determined are tabulated in Table 1 with notation in relation to the method of determination, the confidence range for each, and any specific comment relevant to this application.

A significant operational parameter for this, indeed any, SAT scheme is the influent water quality particularly in relation to the propensity to create a “clogging” layer as a result of the suspended sediment levels of the influent water. At this stage, parameterisation of this aspect has been based on laboratory tests using a range of effluent qualities and field trials using potable reticulated supply water. Further works are required to optimise this parameter, however, it is clear from laboratory works that low suspended solids (SS) waters are of critical importance to both the overall “footprint” of the SAT ponds and the cycling duration between wet and dry phases, both of which form the critical hydraulic capacity determinants of the scheme. The product water quality parameterisation presented at this time is based on literature and theoretical assessment in the absence of direct field measurements.

Figure 1 summarises the results of the laboratory column studies by illustrating the high dependence of infiltration rate on the level of treatment given to the influent water, and hence SS concentrations. This clearly shows that the current level of WSP treatment has a negative impact on infiltration rates as compared with the higher quality waters. Due to the highly variable temperature effects on the WSP and its treatment process operation, it is not considered possible to produce a consistent or sufficiently low SS concentration in the absence of tertiary treatment systems. Significant nutrient stripping of the WSP overflow could be achieved through active wetland systems. However, unmanaged systems (such as Ilparpa Swamp) attract mosquito and adverse public health concern in populated arid regions.
Similarly the recovery system design is based on theoretical and comparative considerations in relation to the hydrogeological attributes of the nearby and adjoining groundwater systems, i.e. Town Basin.

Determination of the operational monitoring and regulation of this scheme requires consideration and refinement in concert with the appropriate regulatory agencies, however, in other jurisdictions this often take the form of a “demonstration” period used to define the longer term operational regulatory settings.

Figure 1: Average Infiltration Rate over 4 Wet/Dry Cycles in Relation to Level of Pretreatment and SS Concentration for “Sand” and “Loam” Columns
Table 1: AZRI Soil Aquifer Treatment Scheme Design Parameters

<table>
<thead>
<tr>
<th>Design Parameter</th>
<th>Measured Range</th>
<th>Design</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Site Capacity</td>
<td>NA</td>
<td>600 ML/year</td>
<td>600 ML/year selected as pilot scale. Range to 1200-1800 ML/year, requires confirmation</td>
</tr>
<tr>
<td>Average Day Infiltration Capacity</td>
<td>NA</td>
<td>1.8 ML/day or 20L/s over 24 hours</td>
<td>Current WSP inflows average 9 ML/day. 20% generally expressed as “overflow”</td>
</tr>
<tr>
<td>Storage Capacity</td>
<td>NA</td>
<td>1800 ML over 3 years</td>
<td>Based on Town Basin measurements between 2000-4000ML total storage capacity across AZRI site. Darcian Porosity range 0.1-0.2</td>
</tr>
<tr>
<td>Peak Day Infiltration Capacity</td>
<td>NA</td>
<td>2.2 ML/day or 820 ML/year</td>
<td>Based on proposed Pond configuration and infiltration design parameters.</td>
</tr>
</tbody>
</table>

Specific Parameters

<table>
<thead>
<tr>
<th>Design Parameter</th>
<th>Measured Range</th>
<th>Design</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average Day Infiltration Capacity</td>
<td>3.5 m/d$^t$ and 11 m/d$^+$</td>
<td>0.3 m/d</td>
<td>Design value accounts for clogging potential of recycled DAF product water of a quality defined below. Potable reticulated supply used in trial.</td>
</tr>
<tr>
<td>Influent Suspended Sediments</td>
<td>&lt;30mg/l</td>
<td>Consistent with SA Reclaimed Water Guidelines – Class B. Present studies suggest this target maybe consistent with Infiltration design target.</td>
<td></td>
</tr>
<tr>
<td>Influent BOD</td>
<td>&lt;20 mg/l</td>
<td>Consistent with SA Reclaimed Water Guidelines – Class B.</td>
<td></td>
</tr>
<tr>
<td>Individual Pond Area</td>
<td>0.55 ha per pond</td>
<td>For wet/dry cycle operation and flexibility in cycle duration to accommodate airborne pest potential</td>
<td></td>
</tr>
<tr>
<td>Number of Ponds</td>
<td>2</td>
<td></td>
<td>Minimum to expose suitable strata</td>
</tr>
<tr>
<td>Pond Depth</td>
<td>0.5 m</td>
<td>Maximum to limit surficial layer compaction and minimise residence time of water in pond</td>
<td></td>
</tr>
<tr>
<td>Water Depth in Ponds</td>
<td>0.3 m</td>
<td>7 days wet / 7 days dry</td>
<td>Can be adjusted to maximise recharge or if potential mosquito issue develops</td>
</tr>
<tr>
<td>Wet/dry cycle length</td>
<td>16 m</td>
<td>Average across site may exceed 20 m.</td>
<td></td>
</tr>
<tr>
<td>Depth to Groundwater</td>
<td>NA</td>
<td>Not required for indirect non-potable reuse. Only an issue if infiltrate migrates off-site.</td>
<td></td>
</tr>
<tr>
<td>Retention Time</td>
<td>Up to 11</td>
<td>Location to be determined</td>
<td></td>
</tr>
<tr>
<td>Recovery Wells</td>
<td>Up to 800ML/yr</td>
<td>Expected operational rate 5L/s per bore accessing existing plus recharged storage</td>
<td></td>
</tr>
<tr>
<td>Recovery Rate</td>
<td>300-1000 m/yr</td>
<td>Estimated aquifer transmissivity needs to be refined to reduce uncertainty</td>
<td></td>
</tr>
<tr>
<td>Site Flood Inundation</td>
<td>1 in 50 years</td>
<td>Flood out of Todd River. Operational interruption expected &lt; 2 months.</td>
<td></td>
</tr>
</tbody>
</table>

$^t$ Measured in 6m*6m silty sands basin using reticulated supply water, Roe Creek
$^+$ Measured in 6m*6m gravely basin using reticulated supply water, Roe Creek.
Physical Design

The collective application of these design parameters is expressed as a conceptual design physically illustrated as per Figure 2. Particular aspects of the design and layout are discussed in more detail later. This illustration serves to indicate the relative size and form flexibility of the scheme as distinct to any firm location, internal dimension or set aspect ratios. The footprint for the infiltration basins is approximately 2 ha and for the scheme including the recovery system is 10 ha in total. Layout can be further softened to accommodate site constraints either physically or aesthetically.

Figure 2: 600 ML/yr Design SAT Scheme Conceptual Design. (Note: precise location as yet undefined)
Operational Design considerations

The two prime drivers of this conceptual design are;

— the need for a low SS concentration of the influent recycled waters relative to the existing WSP overflow SS, and the expectation that a wetting/drying cycle length of 7 days is operable for the long term, and

— the permeable strata at this site are such that there is limited stratification of the infiltrate in relation to existing groundwater throughflow, exhibit an anisotropy that limits the lateral as opposed to the down-gradient infiltrate movement/dispersion and facilitate the practical design and operation of recovery scheme.

Based on the site investigations undertaken and the experience found from hydrogeologic investigations within the Town Basin, this SAT scheme operation can be well managed, from a quantity and quality perspective, through normal wet/dry cycling, active monitoring and tracking the quality of pond influent, infiltrate mound quality and dispersion and the operational recovery efficiency.

Infiltration Operation

It is possible to adjust the pond scale and wet/dry cycle length such that a wider range of influent SS loads could be accommodated, yet this involves negative tradeoffs in relation to the footprint of the scheme (bigger and more ponds), the hydraulic capacity of the treatment system (lower infiltration per unit area) and the quality of recovered waters.

For the adopted design parameters, it is expected that during basin wetting, the organic and other suspended solids in the influent water accumulate on the bottom of the basin, producing a layer that causes infiltration rates to decline over the wetting cycle. Once the infiltration efficiency declines to a set point, ponding ceases and the pond is dried, while the paired basin begins a wetting cycle. Drying of the basin causes this layer to dry, crack, and form curled-up flakes; the organic material also decomposes. These processes restore infiltration rates to close to the original levels. Design infiltration rates account for that expected as an average over wetting cycles allowing for these “restoration” processes.

Over a large number of cycles, the infiltration rate can be expected to decline to below the design target, due to the accumulation of fine materials. At this time mechanical methods such as disc ploughing or scraping should be utilised to remove the thin but persistent clogging layer. SAT operational experiences in the USA indicate that rejuvenation of the basin floor is generally required on an annual basis for matured schemes.

The maintenance of the basin floors will require the basins to be stopped and dried to allow access of the equipment, resulting in the loss of one wetting cycle in each of the basin sets per cleaning cycle. If soil removal is required, then subject to testing, this material offers a useful biosolid, which could be used as a soil conditioner to improve the soil structure and fertility of irrigated soils.
Infiltrate Mound Management (Extent, Quality and Recovery)

Other operational considerations such as the level of infiltrate mound growth and the potential for the SAT operation to mobilise additional stored salts within the soil profile require active monitoring in the field. Laboratory observations and limited field testings suggest a limited potential for these to present operational constraints to this SAT scheme.

Extent

The soils, geology and hydrogeology have been identified as being highly suitable for SAT from comparative assessment with other SAT schemes described in the literature and visited during international study tours undertaken by some of the authors.

The two dominant shallow soil types have been recorded on site: ‘silty sands’ and ‘gravelly sands’. The silty sands are the dominant soil in the areas investigated and the gravelly sands are found along ephemeral drainages. In the areas of silty sands there appears to be a good balance between the coarser textured soils needed for high infiltration rates and the finer textured soils needed for contaminant adsorption and removal.

The geology of the site consists of Quaternary aged silts, sands and gravel overlying Tertiary aged clays and sandy clays. An in-filled palaeochannel feature incised into the Tertiary clays, about 400 metre wide and at least several kilometres long has been located at between 17-30 metres below ground surface. The infill sediments of this feature show high permeability and storage characteristics making it the most prospective area for siting SAT, in relation to storage and containment for recovery and/or long term management.

A 0.5 metre recharge mound has been observed beneath the trial infiltration basins at the AZRI site after 18 ML of water was applied through paired trial basins over a 9 week period. An interpretive assessment has been made through mathematically modelling based on the existing hydrogeological knowledge and correlating these to the field infiltration tests. This indicates that the mounding below the basins will be small and should be less than 3 metres within the immediate vicinity of the SAT basins. The groundwater table is currently 16 metres below ground surface in this area.

Similarly, modelling based on the current level of knowledge, allowing significant safety factors for parameter uncertainty, indicates the infiltrate mound will be well contained within the AZRI site at the scale of operation proposed. Further, there are indications that the rate of “native” groundwater movement through this system is of similar order to that proposed to be applied through SAT scheme. The “native” groundwater has a higher salinity than the untreated infiltrate. The native groundwater appears to be derived from several sources, including the Town Basin outflow, Todd River floodouts, effluent irrigation at Blatherskite Park, Ilparpa Swamp flows and locally occurring runoff recharge. As a result of these multiple sources, and the high rates of aquifer throughflow, the native groundwater quality is variable over time, yet it is generally too saline to support potable use. The SAT operation is unlikely to produce discernible changes to the existing groundwater quality regime other than in relatively close proximity to the SAT site itself, where a quality enhancement is expected to take place.

Quality

The infiltrate mound is expected to be of a similar salinity to that of the infiltrate itself. The assessment of the soils through which the infiltrate moves indicate quite limited potential for remobilisation of salts stored within the soil profile. A factor expected to affect the quality of the infiltrate mound is that of its mixing or conversely stratification in relation to the existing groundwater.
Present indications are that the existing native groundwater is of similar constituent chemistry to the proposed infiltrate yet has an overall salinity level of 1500 mg/L. The influent waters will be of lower salinity, around 1000 mg/L, hence it is expected that any stratification will lead to preservation of the fresher infiltrate.

The mixing between the regional groundwater and infiltrate will become an important operational consideration in relation to the chemical quality of recovered water at this site. It is not possible to quantify this until such time as a “significant” mound of infiltrate is established in relation to the background throughflow occurring. Further works are also required to identify the origin(s) of the groundwater across the Outer Farm Area and provide information on the existing groundwater quality to establish the operational water quality baseline for the SAT scheme.

The microbiological and nutrient quality of the infiltrate mound, are expected to show significant quality enhancements over the recycled water from the wastewater treatment plant, based on comparative assessments from the literature on SAT performance. Marked reductions in nutrients and pathogens are expected to produce water comparable in quality to Class A of the South Australian Reclaimed Water Guidelines. These quality outcomes are expected to be delivered in a consistent and ongoing basis over the time scale of travel through the subsurface system. Further works are required to confirm and refine this assessment, yet a high degree of confidence exists for such an outcome at this site.

It is expected that the infiltrate at the proposed scale will not be detected on the basis of water quality attributes beyond the footprint occupied by the entire SAT scheme. Further, it is considered probable that at large scales of operation the infiltrate mound at such a distance will exhibit enhanced physical and microbiological signatures to that of the existing groundwater. It should be noted, that the existing groundwater cannot support a potable use due to their brackish TDS and are currently considered to be of “marginal” utility for irrigation use in significant quantities.

Recovery

Recovery of these enhanced waters for suitable use, such as for irrigation or export away from the site, is quite dependant on the infield confirmation and definition of these mound attributes and its co-location of suitable subsurface systems that produce a high degree of recovery efficiency.

Current field investigations and conceptual modelling suggest there is a high degree of confidence that recovery efficiencies will be sufficiently high to ensure capture of adequate quantities of water with a suitable quality for non-potable reuse. Aspects expected to affect to this efficiency include the degree of anisotropy in saturated permeability, the relative velocity of the mound progression down-gradient and the degree of infiltrate to native groundwater mixing or stratification. Based on comparative hydrogeological assessments within the Town Basin, it is not expected to see marked mixing of the waters prior to achieving significant storage to that of the natural throughflow and storage being in the order of 2000ML. Interception systems comprising appropriately located and designed bores will need to be constructed within close proximity to the infiltration scheme to achieve a high rate of infiltrate recovery. Potential exists to extract significantly more than the infiltrate volume from such a system given the rate of natural throughflow and storage in this paleo system, however, the quality in relation to reuse objectives will need to be a consideration.

Identification of the minimum residence time of the infiltrate in the aquifer will only result from trials producing pathogen attenuation data, however, residence times of between 3 and 12 months can be anticipated in the final design from experience elsewhere. Higher pre-treatment pathogen removal effectiveness can also shorten these requirements.
Regulatory Management
Prime considerations during the regulatory management of SAT schemes tend to focus on the external paths and/or opportunities for adverse impact to the environment and public. In this case the paths considered to be major concern relate to that of creating surface discharges due to water logging, long term salinisation due to deep flushing of soil salts to groundwater, potential for adverse human health outcomes due to direct contact, ingestion, of generation of mosquito and biting insect pests, and promotion of adverse odour and aesthetic outcomes.

Each of these are considered in more detail later, yet investigations to date indicate each of these to exhibit quite low potential for adverse outcomes as a consequence of one or or the combination of;

1. the physical, geomorphologic, hydrogeological attributes of the site,
2. the inherent flexibility of the SAT operation in relation to the cyclic operational regime and the physical attribute of the conceptual layout, (i.e. ability for progressive ponding within each basin),
3. the application of a “high” quality recycled water to the ponds and the recovery of even higher quality waters, (i.e. SA Reclaimed Water Class B in, with at least Class A recovered),
4. the location of the site in relation to any interacting landuse is relatively remote and outside of short to medium term SAT operational impacts, and
5. the same observations are required to determine operational effectiveness as would be required to detect potential for adverse near field and regional impacts, (accepting the resolution of these observations may differ).

CONCLUDING SUMMARY

In summary, it is considered that from a technical stance, the operation of a SAT scheme at the AZRI site, that is in keeping with an operational and regulatory reporting regime designed on the basis of knowledge gained to date, poses little risk of adverse outcomes, both locally and regionally.

It is not currently possible to determine the “optimal” operational regime for the SAT scheme without further investigation tracking a more significant scale of “infiltrate” than of the field trials undertaken to date, and several “baseline” data requirements exist to facilitate such investigation. Again, these requirements are considered complementary to that expected under a normal regulatory regime.

A SAT scheme operating at 600 ML/yr will need to operate for a duration of at least 12-24 months to inform the longer term operational considerations in relation to the recovered water qualities and the down gradient water quality enhancements, these will in turn inform long term regulatory considerations in regard to community, environmental and social expectations.
SECTION 1

SUMMARY TECHNICAL APPENDICES
Site Suitability

The soils, geology and hydrogeology have been identified as being highly suitable for SAT.

The two dominant shallow soil types have been recorded on site: “silty sands” and “gravelly sands”. The silty sands are the dominant soil in the areas investigated and the “gravelly sands” are found along ephemeral drainages. In the “silty sands” there appears to be a good balance between the coarser textured soils needed for high infiltration rates and the finer textured soils needed for contaminant adsorption and removal.

The geology of the site consists of Quaternary aged silts, sands, clays and gravel overlying Tertiary aged clays and sandy clays. An in-filled palaeochannel feature incised into the Tertiary clays, 400 metres wide and at least several kilometres long has been located at between 17-30 metres below ground surface. The infill sediments of this feature show high permeability and storage characteristics making it the most prospective area for siting SAT.

Therefore the best site for the SAT system is in areas where there are soils with suitable infiltration rates and where an aquifer with permeability and storage characteristics suitable for the storage and efficient extraction of the recycled water. The location of the SAT scheme has been broadly identified (Figure 2), however the precise location will require a localised soil survey.

Sacred sites are a consideration in the development of the system, which will require an area of approximately 2 hectares. Several land features and three tree species on the AZRI site have been identified by the Aboriginal Areas Protection Authority (AAPA) as being of cultural significance. Although disturbances can be kept to a minimum, negotiations with traditional owners will be needed to remove trees if required. Given the proposed location for the scheme is predominantly low scrub with few trees this is not expected to limit the ability to site this scheme and its associated infrastructure, ie pipelines, ponds, control structures, etc.

Infiltration Rates and Basin Clogging

Infiltration trials in two preliminary 6 x 6 m basins using mains water indicate infiltration rates of 3.5 m/day for the silty sand basin and 11 m/day for the gravelly sand basin in the four cycles of 7 days wet / 7 days dry. No apparent decline in the infiltration rate has been observed to date, however, laboratory tests on clogging effects suggest that an order of magnitude reduction in the infiltration rate can be expected, depending on water quality. The minimum infiltration rate with recycled water, depending on its level of treatment, is estimated to be 0.3 m/day. This rate is based on providing an allowance for suspended sediments in the recycled waters, which as yet is untested, and enables a higher degree of scheme operational flexibility than may be possible if a higher rate was adopted at this time. Based on the infiltration rates achieved elsewhere with similar soils this figure could be a conservative underestimate.

The infiltration rates of the soils at the site selected for SAT should be measured to determine the initial rates for operational fine tuning of the scheme. This will assist the determination of the wetting drying cycle period together with that of the timeframe for more active pond base surface rejuvenation works. Similarly this work will inform the internal basin area selection within each of the paired ponds.

Preliminary Groundwater Mounding Response

Preliminary modelling and the on site infiltration tests both indicate that the mounding below the basins should be small. It is expected that within the immediate vicinity of the SAT ponds the
groundwater table, which is currently 16 metres below ground surface, will rise by no more than 3 metres.

Groundwater monitoring of the bores 300 metres up-gradient and 170 metres down-gradient of the trial basins have shown no increase in the groundwater level after infiltrating 18 ML over 9 weeks of operation.
TECHNICAL APPENDIX 2- SAT DESIGN PARAMETERS

Geometry
The basins have been designed to infiltrate 600 ML/yr operated over the entire 12 months of the year. At an average infiltration rate of 0.3 m/day this equates to an area of 5500 m² (0.55 Ha) per basin, or 1.1 Ha in total (since infiltration alternates between paired basins). The basins will be designed to accommodate a supply of greater than the 600 ML/yr, the design presented can sustain a peak daily input of 2200 KL (820 ML/yr). Their layout will be more integrated in the landscape and aesthetically pleasing than rectangular ponds, as well as being functional, by for example, utilising the natural topographic gradient across the AZRI site. A possible design scenario is presented in Figure 2, encompassing the following design features.

Basin Depth
The sub-basins have been designed with a minimum depth of 0.5 metres below the present ground surface. This is to accommodate a ponded depth of water of 0.3 metres.

Basin Operation

Wetting and Drying Cycles
A seven day wetting - seven day drying operation is envisaged at present. This is the operational cycle most commonly applied in SAT systems elsewhere in the world. As operational experience is gained some adjustment to this schedule may be beneficial to maximise the volume of water recharged and to refine the treatment provided by the unsaturated zone beneath the base of the basins. The wetting/drying cycle is also expected to mitigate mosquito outbreaks, since the mosquito lifecycle is of the order of 7-10 days.

Ponding Depth
The effects of ponding depth with respect to infiltration rates have been investigated during the laboratory column studies. They showed that although infiltration rates increased with ponding depth the relationship was not linear ie a percentage increase in head did not result in an equal percentage increase in the infiltration rate. This has been attributed to the compaction of the algal mat on the surface of the soil column. The operational ponding depth should be approximately 0.3 metres to minimise compaction and to allow rapid draining of ponds on commencement of each drying cycle.

Input Water Quality
Laboratory column studies have shown that some degree of clogging is to be expected. Water with higher particulate and nutrient levels, is expected to increase the rate of clogging, as observed in the laboratory tests. Additional treatment of the recycled water will be required to ensure satisfactory rates of infiltration. Table 1 indicates that a suspended solids target in the range of <30 mg/l should be adopted to achieve the design loading and operational efficiencies for the soils at this site.

SAT Performance Monitoring
The spread of infiltrated water in the aquifer and its’ travel time to pumping bores are important measures of SAT performance. The basins will be instrumented with lysimeters in the unsaturated zone and piezometers in the saturated zone to assess changes in water quality and the fate of the infiltrated water. Monitoring would serve to determine that the SAT system is operating in the
desired manner, and that the level of pre-treatment and operational management is adequate. This monitoring will also inform the determination of operational hydraulic loading to the pond basins.

**Recovery System**

To harvest the infiltrated recycled water for subsequent reuse, it is envisaged that a minimum of 4 to 5 recovery bores capable of extracting up to 5 l/s each (630-790 ML/yr) could be located within the AZRI site to intercept the infiltrated water. It should be noted that the peak demand associated with horticultural activities will, however, require up to 50% of the annual demand to be extracted during the summer months, therefore, approximately 10 to 11 bores will be necessary to meet these peak demand periods.

Present drilling and pumping results suggest that further work is required to determine the optimum location and design parameters for these extraction bores. Present indications are that these bores could be located within 200 to 300 metres of the SAT pond system.
Recycled Water Contact
As previously stated, additional treatment of the Alice Springs recycled water will be needed for direct delivery to irrigators and for SAT in relation to minimizing risk to public health. Within this report the pre-treated effluent is referred to as ‘recycled water’.

An important operational consideration for SAT is that of clogging both at the soil surface due to accumulation of suspended materials such as algae and within the soil substrate due to chemical and physical clogging. To this end the levels of nutrients and particulates in the recycled water influent to SAT must be managed to maintain viable rates of infiltration in relation to the wetting cycles, the scheme footprint and ongoing operational considerations including that of the aesthetics and/or socio-environmental interests.

After due consideration of these aspects, the levels of pretreatment occurring in similar schemes elsewhere internationally and the potential for other uses made of the recycled waters prior to application to SAT, a water quality targets for suspended solids concentration and BOD, as defined by the 1999 South Australian Reclaimed Water Guidelines for Class B water, has been adopted for this project as a minimum, although laboratory column testing suggests a lower SS target.

Environmental Water Quality Issues
An evaluation of the fate of contaminants in the sub-surface is essential to ensure that SAT is sustainable over the long term, that recovered waters meet targets for beneficial use, and that groundwater quality is protected in the long-term.

Water quality changes in the unsaturated and saturated zones can include water quality improvements (eg. N, P, pathogens), or water quality deteriorations (eg. leaching of salts, Fe, Mn).

Currently the groundwater quality in the proposed SAT area does not meet the National Guidelines for Drinking Water with TDS values of 1500 mg/l. Recycled water will have a salinity of approximately 1000 mg/l.

There is a negligible overall risk for the environmental values of the Quaternary aquifer to be adversely impacted, and water quality improvements are anticipated.

Plume Migration and Containment
The phreatic water level contours and the preliminary groundwater modelling results indicate that the flow of the infiltrated recycled water will move down-gradient to the south east of the AZRI site. With no extraction the infiltrated recycled water will travel preferentially along the more permeable palaeochannel feature to the south east at a velocity of between 300 and 1000 metres per year.

A high level of containment of the plume downstream of the SAT basins can be attained using recovery bores with combined yearly extraction rates equivalent to or greater than the volumes infiltrated.

Effects on Regional Groundwater Systems
The phreatic water level contours and the groundwater modelling results indicate that the flow of the infiltrated recycled water will move down gradient to the south east of the AZRI site. However, as stated above if the total volume of water extracted is equal to or greater than the total volume infiltrated, little of the recycled water is expected to move off-site. In the event that there is no
immediate extraction the recycled water will preferentially move along the palaeochannel to the south east and potentially impact on downstream users.

The closest bores downstream of the preferred SAT site are located to the east of AZRI in the rural residential area, the closest being approximately 1.5-2 kilometres (Figure 2). These bores are unlikely to extract any SAT infiltrated water due to its expected mounding confinement to the palaeochannel feature. The distance between the SAT basins and the rural bores means that the travel time is likely to be of adequate duration (in the order of years) to minimise any risks even if recycled water does eventually reach these bores. Should the SAT operation lead to greater occurrence of perched waters in the shallow groundwaters near these rural residential bores, management through recovery bore operation is proposed. Further works are required to determine the likelihood of this need which is current considered to be remote.

Modelled groundwater velocities indicate that even for a scenario with no extraction, the infiltrated recycled water will not reach the potable supply aquifers (Amadeus Group) in the area to the south east of the Airport for at least a decade or more. The preferential flow along the palaeochannel feature and the 10 year residence time will result in the constituents of the recycled water being further degraded and removed posing little to no risk down gradient. Any bores in zones containing groundwater sufficiently fresh to be used as drinking water supplies will continue to meet all drinking water requirements. Monitoring data from the observation bores will be used to assess the impacts on the aquifer and allow models to be refined.

Groundwater Levels and Groundwater Quality Monitoring

There are currently 10 monitoring sites across the AZRI site with 7 located in the target palaeochannel feature. The south easterly most monitoring bore is located approximately 900 metres down-gradient of the basins. An extended monitoring network to the southeast of the SAT site is suggested, to monitor both water level variations and more importantly the water quality variations due to the migration of the recycled water down gradient.

It should be noted that the salinity of the brackish groundwater at the SAT trial site is similar to that of the recycled water that will be infiltrated. Routinely used tracers to identify the presence and proportion of recycled water in groundwater such as EC and chloride are unlikely to be successful. Further work is needed to characterise the composition of the native groundwater and recycled water to seek distinguishing stable constituents that can be used to identify the infiltrated SAT water. In the event that these waters remain difficult to distinguish from native groundwater, an artificial tracer may be applied to offer certainty.

Surface Discharge/Water Logging

Infiltration trials and preliminary groundwater modelling results indicate that there is a very low probability of surface discharge or water logging issues arising from the SAT operation.

Salinisation

Low to moderate salt concentrations in soil cores suggest that flushing of salt into the groundwater is not likely to be a persistent issue. This is supported by results of groundwater monitoring adjacent to the trial infiltration basins, which showed that the resultant groundwater conductivity below the basins was very similar to that of the mains water during the infiltration trials.

Mosquitoes/Biting Pests

The operational cycle of the basins will initially be 7 days wetting and 7 days drying alternating between the two sets of basins. It is expected that this cycle will be short enough to disrupt the
breeding cycle of mosquitoes. If this is found to be inadequate, a shorter wetting and drying cycle can be utilised to prevent breeding of mosquitoes. A basin design consisting of several sub-basins affords a high degree of flexibility in this regard.

**Odour**

Anecdotal evidence of similar systems operating in the USA, show no strong evidence of odour issues. Odour will, however, be a consideration of the monitoring program. However, the pre-treatment targets are themselves expected to produce a recycled water product to be applied to the SAT that has limited potential to generate odour issues.

**Flooding Events**

Flood events greater than a 1 in 50 year event will probably inundate the SAT basins, during these times the flow of recycled water into the basins will stop. The water in the infiltration basins will infiltrate relatively quickly (<1 day) preventing recycled water mixing with the flood water. The flood water will also recharge the shallow aquifer system, and it is likely that the flood water will improve the quality of the water already in storage, since the salinity of the river water (~100 mg/L) is much lower than the recycled water (~1000 mg/L).

In general, the interruption to the SAT operation as a function of locally significant flooding of this scale is not expected to be for more than a few weeks to a month at a time.
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1 INTRODUCTION

1.1 Background and Scope

The Alice Springs Water Reuse Scheme (ASWRS) is a joint initiative of Power Water Corporation, the Department of Business, Industry and Resource Development and the Department of Infrastructure, Planning and Environment (DIPE), with technical support from CSIRO Land and Water (CLW). The project aims to develop a water recycling scheme for the township of Alice Springs by making productive use of 1200-1800 ML (million litres) of wastewater per year.

Stage 1 of ASWRS proposes to:

- provide additional treatment of the effluent from the Alice Springs Waste Stabilisation Ponds to a standard suitable for irrigation use
- pipe this recycled water approximately 8 kilometres to the Arid Zone Research Institute (AZRI) site
- utilise Soil Aquifer Treatment (SAT) to polish and recharge approximately 600 ML/yr of the recycled water into the underlying shallow aquifer
- extract the SAT water from nearby bores to supplement irrigation supplies

Since July 2003 DIPE and CLW have jointly investigated the technical feasibility of SAT at the AZRI site. This report documents the progress made to date. Specifically it details:

- hydrogeological characterisation of the AZRI site, including results of drilling, soil investigations, aquifer pump testing, and soil permeametry
- evaluation of the first results from trials using small-scale basins with mains water on infiltration rates and groundwater levels
- groundwater modelling to assess impacts of SAT on groundwater levels and movement of the infiltrated water
- laboratory studies on soil clogging that assesses the impacts of soil type, water quality, ponding depth and climate on infiltration rates
- description of water quality issues, including laboratory data on water quality changes through soil columns
- design principles for pilot SAT scheme, largely through the integration of the knowledge generated from the aspects listed above

1.2 Soil Aquifer Treatment and Water Banking

Soil-Aquifer Treatment (SAT), or geopurification, occurs during infiltration of water from intermittently filled basins through the soil to the watertable of an aquifer below. The soil acts as a natural filter to reduce concentrations of pollutants in the infiltrating water through physical, chemical and microbial processes. Thus suspended solids can be filtered out, transport of inorganic and organic pollutants can be retarded through adsorption on soil and organic pollutants degraded and decomposed by soil organisms. SAT is very effective at removing microorganisms, if any remain in the recycled water supplying the basins. Residence of the infiltrated water in the aquifer also provides effective treatment but at a slower rate than the unsaturated zone above the water table.

The operation of the SAT system with wet and dry cycles is a common operating strategy. The primary purpose of the wet/dry operation cycle is to control the development of clogging layers and
maintain high infiltration rates, and in some cases to disrupt insect life cycles (Fox, 2002). As a clogging layer develops during the wetting cycle the infiltration rate can decrease to levels impractical for operation. The desiccation of the clogging layer during the drying cycle can restore infiltration rates (Fox, 2002).

Water Banking is a generic name for a range of methods to enhance groundwater recharge in order to store water in an aquifer to be recovered later by pumping from bores. SAT is therefore both a water treatment process and a means of water banking. Other recharge methods such as bore injection and continuous pond infiltration were considered but SAT was selected because of its superior capability to improve water quality.

Water banking is no more expensive than other storage alternatives, such as above ground tanks or reservoirs. In fact, in Alice Springs, it will be much less expensive than most other options because so little land is required and relatively little hardware or construction is necessary. The scheme can be expanded so that larger amounts of water can be stored underground at a minimal cost compared to other storage methods. The AZRI SAT project is thought to be capable of storing up to 1800 ML of water underground per annum. This equates to a surface tank of approximately 4 metres height and 5 times the area of Blatherskite Park. Storing below ground also avoids evaporation losses and potential mosquito and algae problems.

Figure 1.1 presents the components in the SAT water banking concept, infiltration basins, aquifer storage and water extraction/recovery.

![Figure 1.1 Schematic depiction of soil aquifer treatment SAT](image)

1.3 Brief Review of the Literature on SAT

Soil-Aquifer Treatment has been widely applied, particularly in parts of the USA and the Middle East since the 1970s, and has been shown in a number of cases to be an effective technique for storing, treating and recovering a diverse range of effluent qualities. Comprehensive reviews of the experience gained from these operations in recent decades are available in works by National Research Council, (1994) and Bouwer, (2002). SAT projects such as the Flushing Meadows Project near Phoenix, Arizona and the Dan Region Project in Israel have been operating for 25 years. These and other projects have been documented as having soils and watertable depths comparable to those at the AZRI site.

In Australia the Alice Springs Water Reuse project is likely to be the first known study to proceed with SAT. Previous experience with the disposal of effluent in basins has occurred in Western
Australia. Ho et al, (1992) studied the fate of contaminants in infiltrating effluent where sandy soils were amended with bauxitic red mud along with unamended calcareous sand. Most recently, Toze et al, (2002) reported on the results of a field study at Halls Head where secondary treated effluent is disposed of in infiltration basins which recharge the shallow groundwater. The two-year study by Toze et al, (2002) considered the efficacy of newly-installed recovery bores to provide irrigation quality supplies by considering the fate of chemical and microbial contaminants. It demonstrated that the recovered water quality, with relatively short retention times (24 days), was suitable for irrigation with negligible associated health or environmental risks.
2 SITE CHARACTERISATION

2.1 Introduction

The following section presents a summary of the data from previous investigations and historical monitoring in the study area, providing a contextual background for the detailed site investigations conducted by DIPE from July 2003 to March 2004. These investigations conducted at AZRI include drilling investigations, soil investigations, test pumping and soil infiltration/permeameter measurements. The series of reports pertaining to these detailed studies are listed below and will be available mid 2004:

<table>
<thead>
<tr>
<th>Report Volume</th>
<th>Title</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Site Characterisation</td>
<td>Presents available data in the study area.</td>
</tr>
<tr>
<td>2</td>
<td>Site Investigations</td>
<td>Presents drilling, soil investigations and geophysics work conducted at AZRI.</td>
</tr>
<tr>
<td>3</td>
<td>Infiltration Trials</td>
<td>Presents the data from the infiltration trials at AZRI.</td>
</tr>
<tr>
<td>4</td>
<td>Groundwater Modelling</td>
<td>Groundwater modelling results of the AZRI site with respect to the infiltration tests.</td>
</tr>
</tbody>
</table>

2.2 Study Area Location

The field investigations for the SAT feasibility study were conducted on the Arid Zone Research Institute (NT Por 800 and NT Por 427) located approximately 5 kilometres south east of Alice Springs in the Northern Territory (Figure 2.1). Although the investigation work has been confined to the AZRI property, the groundwater systems below the AZRI site extend well beyond this quite localised area. For the purposes of this report the project area will therefore encompass the regions referred to historically as the Inner Farm Basin and the northern Outer Farm Basin (Figure 2.1).
2.3 Climate

The study area is in an Arid to Semi-arid climate with an average annual rainfall of 303 mm and a potential free evaporation rate of 3064 mm refer Table 2.1.

Table 2.2 Average monthly climatic data from the Alice Springs Airport

<table>
<thead>
<tr>
<th>Month</th>
<th>Jan</th>
<th>Feb</th>
<th>Mar</th>
<th>Apr</th>
<th>May</th>
<th>Jun</th>
<th>Jul</th>
<th>Aug</th>
<th>Sep</th>
<th>Oct</th>
<th>Nov</th>
<th>Dec</th>
</tr>
</thead>
<tbody>
<tr>
<td>Max Temp</td>
<td>36.2</td>
<td>34.9</td>
<td>32.6</td>
<td>28</td>
<td>23</td>
<td>19.8</td>
<td>19.6</td>
<td>22.5</td>
<td>27</td>
<td>30.8</td>
<td>33.6</td>
<td>35.3</td>
</tr>
<tr>
<td>Min Temp</td>
<td>21.3</td>
<td>20.7</td>
<td>17.3</td>
<td>12.5</td>
<td>8.3</td>
<td>5.1</td>
<td>4</td>
<td>6</td>
<td>10.2</td>
<td>14.7</td>
<td>17.8</td>
<td>20.1</td>
</tr>
<tr>
<td>Daily Evap [mm]</td>
<td>12.9</td>
<td>11.4</td>
<td>10</td>
<td>7.3</td>
<td>4.8</td>
<td>3.6</td>
<td>3.9</td>
<td>5.5</td>
<td>7.8</td>
<td>10</td>
<td>11.5</td>
<td>12.2</td>
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<tr>
<td>Monthly Evap [mm]</td>
<td>399.9</td>
<td>319.2</td>
<td>310</td>
<td>219</td>
<td>148.8</td>
<td>108</td>
<td>120.9</td>
<td>170.5</td>
<td>234</td>
<td>310</td>
<td>345</td>
<td>378.2</td>
</tr>
<tr>
<td>Rainfall [mm]</td>
<td>38.1</td>
<td>43.8</td>
<td>32.5</td>
<td>17.6</td>
<td>19</td>
<td>14.6</td>
<td>14</td>
<td>9.8</td>
<td>8.6</td>
<td>21.7</td>
<td>27.6</td>
<td>38.3</td>
</tr>
</tbody>
</table>

Reference: Bureau of Meteorology

The temperature in Alice Springs typically exhibits a 16°C variation each day. In the summer months the daily temperature will typically fluctuate between 20°C and 36°C. During the winter months the daily temperature will typically fluctuate between 4°C and 20°C. This decrease during winter results in a three to four fold decrease in evaporative losses in the waste water treatment.
processes, consequently the highest production of sewage overflows from the evaporation ponds at the end of the treatment process occur at this time.

2.4 Geomorphology/Land Unit Mapping
Quinlan and Woolley (1969) originally described the sedimentation processes evident in the Inner and Outer Farm areas as similar to an alluvial fan, characterised by constantly shifting, low sinuosity, and braided channel systems.

Two aeolian dune deposits have been mapped, to the north east and north west of the Horticultural Block. These have been identified as having cultural significance to the local aboriginal people.

Lennartz (2001) has mapped the land units of the Alice Springs region and northern Outer Farm Basin at 1:25000 scale.

2.5 Hydrogeological Setting

2.5.1 Regional Geology
The geology of the Alice Springs region and in particular the area south of Blatherskite Range is referred to as the Alice Springs Outer Farm Basin (refer Figure 2.1).

The geology of the area has been described in detail by Quinlan and Woolley (1969). The Outer Farm area is typified by a deep sequence (up to 300 m) of relatively recent sediments (predominantly of Tertiary age with a veneer of Quaternary aged sediments) overlying the steeply dipping sedimentary formations of the Amadeus Basin Sequence, which, overlie the Pre-Cambrian rocks of the Arunta Block (refer Figure 2.2).

The more recent sediments (Quaternary aged) comprise silts, sands and gravel and are derived from alluvial and fluvial processes relating to sedimentation along meandering river channels and overbank flood events. A distinct palaeochannel (old river bed) of the Todd River has been located in the Inner Farm Basin (Woolley and Quinlan, 1969 and Berry, 1991) and at AZRI. The Quaternary aged sediments are underlain by a thick sequence of Tertiary aged clays. These sediments were deposited in a low energy, lacustrine (lake) environment. The Tertiary sediments have sandier horizons in the upper portions of the formation, which are utilised by local, mainly private bores. These units, however, are in poor hydraulic connection with the Quaternary sediments, due to the intervening clayey horizons, and exhibit semi-confined hydraulic conditions.

The Amadeus Basin comprises sandstone, shale and carbonate rocks. The Roe Creek borefield to the south west of the airport, which provides potable water supplies to Alice Springs, is located in these deeper formations of the Amadeus Basin and include the Mereenie Sandstone, Pacoota Sandstone, Goyder Sandstone and Shannon Formation. See Figure 2.2 with reference to Figure 2.1.

The sediments of the Amadeus Basin overlie the very old basement rocks of the Arunta Block Complex. The Arunta Block Complex is dominated by granite and gneiss with other minor igneous and metamorphic rocks.
2.5.2 Quaternary Aquifer Hydraulic Parameters

Quinlan and Woolley (1969) and Berry (1991) reviewed the hydraulic parameters of the surficial Quaternary aquifers in the Town and Inner Farm Basins. Quinlan and Woolley, (1969) concluded that a reasonable estimate of the hydraulic conductivity was 45 m/day (T=310 m²/day) and specific yield of 0.07. Berry’s work on the Inner Farm Basin indicated that the hydraulic conductivity of the Quaternary sediments ranged from 1 to 117 m/day with values of 40, 42 and 117 m/day observed in the deeper aquifer material (Berry, 1991).

2.5.3 Groundwater Level Hydrographs in the Quaternary Sediments

Hydrographs of bores intersecting the Quaternary aquifer have been collected for several sites to the north of the AZRI investigation site. On average the data has been collected quarterly, with some of the older bores having data as far back as 1957 an example is RN3098 (see Figure 2.4).

Groundwater hydrograph data is sparse within the Outer Farm Basin, at least within the Quaternary sediments (refer Figure 2.3), however, the data available does show very similar response characteristics. The data presented for the Inner Farm Basin are RN5193 and RN3584. The data presented for the Outer Farm Basin are for RN3098 and RN3607.
Figure 2.3  Hydrograph locations in the Inner Farm and northern portion of the Outer Farm Areas.

Selected hydrographs for both the Inner and Outer Farm areas are presented in Figure 2.4. The annual precipitation with a 3 year moving average fit is also presented for comparison.

The obvious variation between the hydrographs is their relative level, which decreases downstream to the south east. The bores located in the Inner Farm Basin show a relatively subdued response compared to those in the Outer Farm Basin. RN3584 shows a much smaller range compared with RN3098. This has been attributed to the reduction in the outflow from the Inner Farm Basin due to the restriction at Blatherskite Gap.
Figure 2.4  Annual rainfall data (DR015590) and groundwater level hydrograph data for selected bores constructed in Quaternary sediments.

Prior to the 1970’s the hydrographs show relatively high recession rates, corresponding to a period when high extraction rates were occurring in the Town, Inner and northern Outer Farm Basins. During this time water levels were at their lowest and at times water supplies were unreliable, this and the advent of mains reticulation resulted in a shift away from using bore water. The reduced extraction and the extremely long period (approximately 260 days) where the Todd flowed in 1973/74, resulted in the water table recovering to levels similar to pre-European settlement.

Analysis of the hydrographs after the mid 1970’s has shown that the application of waste water at Blatherskite Park has altered the groundwater level response to recharge events, indicating that a proportion of the throughflow is due to the application of waste water.

Prior to 1983, the first reported application of effluent to Blatherskite Park (Berry, 1991), the recession of the groundwater level was approximately 0.01 m/d, after 1983 the recession is reduced to 0.007 m/d. This effect is seen in hydrographs of both the Inner and Outer Farm Basins (Figure
2.4). The TDS plot versus time for RN5193 (Figure 2.5) also shows the advent of waste water application at Blatherskite Park.

2.5.4 Hydrochemistry

The groundwater in the study area has several sources, including the throughflow from the Town Basin, diffuse recharge from rainfall, and recharge from flood events of the Todd River and recharge from the application of waste water onto Blatherskite Park and overflows into Ilparpa Swamp and St Mary’s Creek. Groundwater quality data collected in the Inner Farm area (RN5193 in Figure 2.5) show that the advent of irrigation at Blatherskite Park has resulted in the increase in the observed TDS.

![Graph showing variations in TDS](image)

**Figure 2.5** Variations in groundwater quality in the Inner Farm Basin at RN5193.

2.5.5 Alice Springs Water Supply

The Alice Springs water supply, as stated above, is obtained from the Roe Creek Borefield, located 15 km south southwest of the town (Figure 2.1). A detailed description of the hydrogeology of the Roe Creek borefield is provided by Jolly *et al.*, (1994).

The Roe Creek borefield was commissioned in 1964 when the existing supply from the alluvial sediments beneath the town became inadequate. The borefield currently extracts approximately 30 ML/day from four discrete aquifers of the Amadeus Basin; the Mereenie Sandstone, Pacoota Sandstone, Goyder Sandstone and Shannon Formation, which for the lifetime of the borefield are considered as discrete and not interconnected (Jolly *et al.* 1994). Approximately 80% of the town supply comes from the Mereenie Sandstone (Jolly *et al.*, 1994).

Groundwater modelling of the Mereenie Sandstone aquifer has determined that the aquifers in the area of the Roe Creek borefield perform as a relatively closed system acting as a “tank” (Jolly *et al.*, 1994). Average inputs to the system from recharge, up dip flow and along strike flow from the Rocky Hill area are estimated at 500 ML/yr (Jolly *et al.*, 1994). This is relatively small compared to the annual volume extracted (10,000 ML) approximately 5% or the estimated volume stored in the aquifer to an economic depth (2x10⁶ ML) or 0.03% (Jolly *et al.*, 1994).

2.5.6 Regional Hydrographs in the Tertiary and Amadeus Basin Aquifers

The groundwater level hydrographs of monitoring bores in the vicinity of the airport (Figure 2.7) show the response in the Tertiary and Amadeus Basin aquifers to pumping in the Roe Creek borefield and recharge events from infiltration of flooding events of the Todd River. The hydrographs of RN4458, RN4461 and RN4481 are some of the few monitored bores in the Tertiary aquifer. The locations of these bores with respect to the Roe Creek borefield and the Todd River are presented in Figure 2.6.
The most obvious trends evident in the hydrographs (Figure 2.7) are the increase of the Tertiary aquifer water level after 1976 and the continuous decline in the Amadeus Basin aquifer water levels from 1964 although RN4462, the bore with the greatest distance to the Roe Creek borefield, shows no response to pumping until approximately 1985.

![Map of hydrograph locations in the vicinity of the Alice Springs Airport.

The Quaternary groundwater level hydrographs (Figure 2.4) show a major recharge event coinciding with the rain year of 1973/74, the hydrographs of the Tertiary aquifers, however, show appreciable responses to this event after approximately 3 years, in 1976. The Amadeus basin aquifer hydrographs, however, show no noticeable response to the recharge to date.

Conversely, the Tertiary hydrographs show no response to the pumping at Roe Creek, even though RN4481 is relatively close to RN4462 with approximately 1 km separation. The independent responses of the water levels in the Tertiary and Amadeus Basin aquifers indicate that a very poor hydraulic connection exists between the two aquifer systems due to the clay aquitard referred to previously. These clays can be seen to act as a buffer for groundwater in the Quaternary aquifer as it overlies the Tertiary sediments.

2.5.7 Travel Times of Recharge Water to Amadeus Basin Aquifers

Radioisotope studies provide age estimates of approximately 10,000-12,000 years for the groundwater of the Roe Creek borefield (Hostetler, 2002). It was concluded that the large depth to the water table acts to average the different recharge events so that recharge is essentially occurring
constantly at approximately 1-2 mm/yr and that the age estimation is related to the travel time through the overlying Quaternary and Tertiary sediments, rather than being due to a discrete recharge event (Hostetler, 2002).

![Annual Rainfall and Groundwater Levels](image)

**Figure 2.7** Annual rainfall data (DR015590) and the comparison of groundwater level hydrographs collected in Tertiary and Amadeus Basin aquifers.
2.6 Arid Zone Research Institute Site Investigations

The Department of Infrastructure, Planning and Environment has been conducting site investigations at the Arid Zone Research Institute (AZRI) from June 2003 to November 2003. These investigations were to determine the suitability of the soils and aquifers to infiltration and storage of recycled water using SAT. The site investigations are detailed in reports being prepared by DIPE (section 2.1).

2.6.1 Location of the Arid Zone Research Institute

AZRI is located 5 kilometres south of Alice Springs in a region referred to historically as the Outer Farm Basin (refer Figure 2.1).

The AZRI property comprised of NT Por. 800 and NT Por 427 and has an area of 772 hectares. The property is bounded to the north by the Todd River, to the south by Colonel Rose Drive, the Stuart Highway to the west and rural residential blocks to the east (see Figure 2.8).

The AZRI site shows little topographic relief (<10m) with elevations grading from 558 mAHD in the northwest to 546 mAHD in the southeast.

The site is dissected by two ephemeral drainage systems, St Mary’s Creek to the west and an overflow channel of the Todd River through the centre of the property.

![AZRI locality plan](image-url)
2.6.2 Soil Investigations
Soil investigations have been conducted across the site to characterise the upper 3 metres of the soil profile. The investigations have demonstrated that similar soils occur over the AZRI site, however, more locally the soils have been shown to have quite high variability. Generally the site shows a relatively consistent pattern of silty sands to 1-2 metres below ground surface, below this is a variable thickness layer of between 1-3 metres of coarse gravelly sand overlying silty sands. Areas of shallow coarse, gravelly sand have been located, which have variable distribution across the site, although they appear to coincide with the drainage depressions evident in the topographic maps. One of the trial infiltration basins (Basin B) has been located along the flank of such a depression. The depression coincides with the overflow channel referred to in section 2.4 and Figure 2.8.

2.6.3 Soil Infiltration Properties
The soil infiltration has been assessed using dual infiltration rings and soil permeameter measurements. Dual ring infiltration tests were conducted at the two prospective SAT sites with five tests conducted at each of the current basin sites and another 10 tests have been completed at the secondary SAT location. The duration of each of the tests was 3 hrs. The tests showed a distinct difference between the two soil types evident on site. The silty sandy soils showed infiltration rates of between 3 – 10 m/d averaging 4 m/d, in contrast to the gravelly sands, which demonstrate infiltration rates of between 25 and 90 m/d averaging 30 m/d.

Soil permeameter tests could only be conducted on the silty sandy soils, as the gravelly sands allowed the permeameter to dewater within a minute or so, making measurements impossible. The silty soils provided an average Ksat of 4-5 m/day.

Longer term infiltration tests, started at the end of January, have demonstrated the reliability of the infiltration tests with continuous infiltration rates of 3 m/d for the silty sands and 10 m/d for the gravelly sands over the duration of the trials. Details of these tests are presented in section 3 of this report.

2.6.4 Hydrogeology
A total of 52 bores have been drilled during the investigations. Of these 21 were completed as monitoring bores and 2 were completed as test production bores. The location of the investigation bores and the production bores are presented in Figure 2.9. The drilling has delineated a region of higher permeability interpreted to be a palaeochannel incised into the Tertiary clays. The palaeochannel feature is inferred to be a continuation of the system in the Inner Farm Area described by Quinlan and Woolley (1967) and Berry (1991). The feature is manifest as a broad channel approximately 300-400 metres wide, which, on the AZRI site occurs between 17 to 30 metres below the ground surface, in filled with a layered (lensed) series of gravels/cobbles, sands and clays. Typically the basal metre or so of the channel consists of gravel and cobble infill. The location of the feature is depicted in Figure 2.9 and a west to east cross-section is depicted in Figure 2.10.

Initially the feature was interpreted to consist of a single channel, however, detailed drilling for piezometer installation in the vicinity of the infiltration basins has shown that the channel is quite variable. Airlift yields have shown that the aquifer is also variable in its hydraulic characteristics, a feature, which has been noted in previous drilling to the north of the site in the Inner Farm and Town Basins.
Figure 2.9  Drilling investigation results
Figure 2.10  West - east geological cross-section across the site the location is in Figure 2.9.

2.6.5 Groundwater Levels

The groundwater level contours have been produced from the monitoring bores installed along the axis of the palaeochannel and the observations bores completed during previous investigations. The contours are presented in Figure 2.11.
2.6.6 Test Pumping

Two production bores (RN17951 and RN17943) were installed to determine the hydraulic characteristics of the palaeochannel sediments and likely pumping rates achievable. The bores were found to perform poorly. The bore construction techniques coupled with the variability of the channel sediments are thought to be the cause of these poor results.

RN17951 was seen to airlift approximately 6 l/s although RN17941 (only 10 metres to the northwest) airlifted approximately 10 l/s. RN17951 the production bore installed to the northwest, near RN17939 (Figure 2.9) produced a constant rate of approximately 5 l/s. Analysis of the drawdown curves from both bores provided hydraulic conductivities of 10 m/d over a saturated aquifer thickness of 8-10 metres. Analysis of the adjacent observation bores indicate hydraulic conductivities of between 43 and 63 m/day over the same saturated thickness. As stated in section 2.5.2 the equivalent palaeochannel sediments encountered in the Town and Inner Farm Basins and described by Quinlan and Woolley (1969) and Berry (1991) have been shown to have hydraulic conductivities of approximately 40 to 50 m/day, comparable to the results observed in the pump tests.

2.6.7 Hydrochemistry

Water samples collected during the drilling investigations provided standard water chemistry data for October 2003. The bore locations of these samples are depicted in Figure 2.9 and the results are
presented in Table 2.3. RN17939 is the most north westerly monitoring bore and RN17936 is the most south easterly monitoring bore. An increasing trend downstream in all parameters is observed except for the nitrate values, which, show a decrease in concentration with distance downstream.

Table 2.3 Hydrochemical data from drilling at AZRI

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>RN17939</th>
<th>RN17938</th>
<th>RN17937</th>
<th>RN17940</th>
<th>RN17943</th>
<th>RN17936</th>
</tr>
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<tbody>
<tr>
<td>pH</td>
<td></td>
<td>7.6</td>
<td>7.6</td>
<td>7.5</td>
<td>7.3</td>
<td>7.6</td>
<td>7.1</td>
</tr>
<tr>
<td>EC</td>
<td>µS/cm</td>
<td>1343*</td>
<td>2770*</td>
<td>2187*</td>
<td>2232*</td>
<td>3120*</td>
<td>3400*</td>
</tr>
<tr>
<td>TDS</td>
<td>mg/l</td>
<td>841*</td>
<td>1856*</td>
<td>1449*</td>
<td>1491*</td>
<td>2134*</td>
<td>2352*</td>
</tr>
<tr>
<td>Sodium</td>
<td>mg/l</td>
<td>135</td>
<td>310</td>
<td>227</td>
<td>214</td>
<td>300</td>
<td>299</td>
</tr>
<tr>
<td>Potassium</td>
<td>mg/l</td>
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<td>10</td>
<td>9</td>
<td>9</td>
<td>9</td>
<td>8</td>
</tr>
<tr>
<td>Calcium</td>
<td>mg/l</td>
<td>99</td>
<td>199</td>
<td>171</td>
<td>184</td>
<td>268</td>
<td>291</td>
</tr>
<tr>
<td>Magnesium</td>
<td>mg/l</td>
<td>24</td>
<td>47</td>
<td>36</td>
<td>44</td>
<td>69</td>
<td>100</td>
</tr>
<tr>
<td>Iron</td>
<td>mg/l</td>
<td>u/s</td>
<td>u/s</td>
<td>u/s</td>
<td>3.3</td>
<td>u/s</td>
<td>u/s</td>
</tr>
<tr>
<td>Tot Hard (calc)</td>
<td>mg/l</td>
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<td>690</td>
<td>575</td>
<td>640</td>
<td>953</td>
<td>1138</td>
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<tr>
<td>Tot Alk.</td>
<td>mg/l</td>
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<td>294</td>
<td>288</td>
<td>285</td>
<td>300</td>
<td>319</td>
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<tr>
<td>Silica</td>
<td>mg/l</td>
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<td>43</td>
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<td>45</td>
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<td>50</td>
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<tr>
<td>Chloride</td>
<td>mg/l</td>
<td>189</td>
<td>470</td>
<td>339</td>
<td>363</td>
<td>543</td>
<td>605</td>
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<tr>
<td>Sulphate</td>
<td>mg/l</td>
<td>132</td>
<td>538*</td>
<td>295</td>
<td>393</td>
<td>655*</td>
<td>739*</td>
</tr>
<tr>
<td>Nitrate</td>
<td>mg/l</td>
<td>9</td>
<td>12</td>
<td>4</td>
<td>6</td>
<td>3</td>
<td>1</td>
</tr>
<tr>
<td>Bicarbonate</td>
<td>mg/l</td>
<td>0.6</td>
<td>0.5</td>
<td>0.5</td>
<td>0.5</td>
<td>0.5</td>
<td>0.7</td>
</tr>
<tr>
<td>Fluoride</td>
<td>mg/l</td>
<td>0.6</td>
<td>0.5</td>
<td>0.5</td>
<td>0.5</td>
<td>0.5</td>
<td>0.7</td>
</tr>
</tbody>
</table>

* values exceed the Australian Drinking Water Guideline values
u/s – unsuitable for analysis

The data for TDS and nitrate concentrations are presented in Figure 2.12 with respect to increasing distance downstream from RN17939 along the axis of the palaeochannel. The location of this section is shown in Figure 2.9. The TDS values are seen to increase downstream with values of approximately 1500 mg/l observed at RN17940 near the trial infiltration basins (section 3).

Figure 2.13 a) Distribution of TDS and b) NO₃ – N downstream along the axis of the palaeochannel.

2.7 Conclusions

Soil pits excavated across the AZRI site and centred on the palaeochannel have delineated soils suitable for SAT infiltration basins. The site shows some variability, however, areas with similar soils are expected in the direction of flood flow.
Drilling results have defined a palaeochannel beneath the AZRI site trending NW to SE. The palaeochannel is in-filled with sands, gravels and cobbles over the interval of 17 to 30 metres below ground level. Although data is sparse, the channel is inferred to run to the north of the airport and continue towards the Rocky Hill area some 15 km from the proposed site.

Detailed drilling has determined that the palaeochannel is composed of intercalated clays, silts, sands and gravels, which show considerable variability laterally. The locating of high yielding production bores has had limited success, and will require further work to determine a reliable siting and construction methodology.

The quality of the ambient groundwater makes it suitable for non-potable uses.

The Quaternary aquifer to be utilised for the SAT is in poor connection with the underlying Tertiary and Proterozoic aquifers. Travel times to the tertiary aquifer in the immediate vicinity of areas of recharge are expected to be in the order of 3-5 years, whilst travel times to the underlying Amadeus Basin aquifers is considered to be of the order of decades if at all.

2.8 Recommendations

Silty sandy soils have been identified as the preferred soil type for the location Pilot SAT basins. Soil investigations to determine the location of these soil types and their infiltration characteristics will be required in the areas deemed suitable for the Pilot Sat basins.

The palaeochannel is relatively well defined within the AZRI property, however, the section between Blatherskite Gap and RN17939 the channel is poorly defined, due to the lack of drilling information. This area to the northwest is important to define water quality variations due to the various sources of water identified.

At the present no known bores licensed for groundwater extraction, to the southeast of the AZRI site, have been completed in the Quaternary aquifer. To provide confidence in the location of the infiltrated recycled water plume, the delineation of the palaeochannel to the southeast of AZRI is recommended. Investigation work to further this would involve approximately 20 investigation bores, these would provide, geological, water level, water quality and long term monitoring data and would assist with defining the boundary conditions for the groundwater modelling. Surface electromagnetics surveying was trialled to provide initial estimates of the channel location, however, the similarity of the strata (in terms of electrical contrast), and the presence of brackish waters have resulted in data which is largely undiagnostic.

Investigation works will be required to provide a methodology for the siting of production bores capable of producing 5 l/s. It is expected that approximately 5 investigation bores will be required to determine a suitable site, this will lead to the construction of 1 large diameter test production bore.
3 FIELD TRIAL USING POTABLE WATER

3.1 Introduction

The proof of concept for SAT is being approached through staged trials. Initially two infiltration basins were constructed and instrumented at the at the Arid Zone Research Institute (AZRI) site between December 2003 and January 2004. The basins allowed for infiltration trials to take place, using mains water (town water supply – Mereenie sandstone). The objective of the infiltration tests are to determine rates of infiltration and the impacts of this infiltration on the groundwater levels. This section describes the initial results of these trials.

3.2 Representativeness of AZRI Site and Rational for Basin Design

Permeameter data, dual infiltration ring data and soil information indicate that the soils at the trial site are representative of the soils found in the prospective SAT sites. Drilling data indicate that the strata, although locally heterogeneous are relatively uniform across the AZRI site.

Permeable soils with low organic content and CEC provide the most favourable conditions for maximising infiltration rates and reducing clogging potential although retention and degradation of contaminants such as microbial pathogens are likely to be lower in the more permeable soils. Less permeable soils with higher organic carbon and CEC offer less favourable hydraulic performance but better treatment performance.

Recognising this, the two trial basins were chosen in locations with contrasting surface material: basin A at a site with silty sandy topsoil, and basin B was situated where coarse sands and gravels occupy the top 1.5 m of the profile. As section 5 will show, this mirrored the design of the column studies. The two basins are situated 70 m apart. This site could provide an excellent opportunity to examine in detail the trade-offs that occur between maximising recharge rates and geopurification in the subsurface, if it is decided that trials using recycled water should be conducted.

The location of the basins was decided on the basis of a number of drill holes that identified a major palaeochannel system at depth, as well as permeameter, dual infiltration ring, and soil excavation data. The site was also logistically attractive due to the proximity to the existing mains water supply.

3.3 Basin Design

The basins are located to the northeast of the Horticultural Block (see section 2). The basins have been constructed with 6x6 metre floors, with a side wall slope of approximately 1:2. The bottom of each basin is approximately 0.5 m below the level of the adjacent natural surface. Basin A is located on the western side of the fence line divide, whilst basin B is located approximately 70 m away on the eastern side (Figure 3.1). Appendix A offers a collection of annotated photographs of the trial site. The size of the basins was restricted by the capacity of the pipeline to deliver water to them.

3.4 Water Supply

The water supply to the site is via a 300 metre, 150 mm PVC pipeline, which taps the mains water supply to AZRI. Although initial testing indicated flow rate of approximately 6-7 L/s the actual available supply to the infiltration basins was approximately 4.5 L/s due to the requirements of the Horticultural Block. The pipeline to the basins is equipped with an electromagnetic flow meter and a level controlled actuated flow valve. The basins have ultrasonic water level sensors fitted to provide control for the actuated flow valve. The basin water level/inflow control system is computer controlled. The installation is powered by 24V solar/battery.
3.5 Basin Instrumentation and Monitoring

A summary of the instrumentation installed at the trial site is given in Table 3.1. Around each basin 7 piezometers were installed; 3 piezometers to total depth (30 m), 3 piezometers at 6m, 12m and 18m below ground level and 1 piezometer at 2.7m below ground level.

At the centre of each basin a piezometer, soil moisture content probe and tensiometer were placed at three different depths (0.1, 0.5 and 1.0 m below the basin surface). The six soil moisture probes (‘ECHO’ probe, Decagon Devices Inc.) were calibrated in the laboratory prior to field installation using soil recovered from the target depth during installation of the adjacent piezometers. Each tensiometer consists of a ceramic cup approximately 6 cm long and 1.5 cm diameter joined to a clear acrylic tube, and sealed at the top with a rubber septum. The soil water content and matric suctions were measured using a digital meters. The piezometers to 1.0 m were made from 50 mm ID PVC casing with a discreet slotted length of 10 cm.

The flow rate was monitored using an electromagnetic flow meter and the water level in the basin was recorded continuously using ultrasonic level sensors. The supply flow rate and the water level in the basins were automatically recorded every 10 minutes. An actuated valve is used to maintain water levels within the basin to as close to a constant depth of 0.3 m as possible. Groundwater levels in adjacent piezometers were monitored manually on a frequent basis, generally once per day except on basin transition days where measurements were taken as often as every 5 minutes.
### 3.6 Infiltration Schedule

Initially two short <1 day infiltration tests (T1 & T2) were conducted in each basin prior to the automation of the water delivery system. Between the 19/01/2004 to 31/03/2004 four entire 7 day (approx) wetting / 7 day drying cycles were conducted in each of the trial basins. Water was fed to each basin, such that at any given time one basin was wet whilst the other was drying. The choice of the cycling schedule is typical of that in use at operational SAT sites in the USA, and was replicated also in the laboratory column study. The timing and water consumption for these cycles are given in Table 3.2.

#### Table 3.2 Trial infiltration schedule to 31 March 2004

<table>
<thead>
<tr>
<th>Cycle</th>
<th>Basin</th>
<th>Period</th>
<th>Duration</th>
<th>Volume Infiltrated (m³)</th>
<th>Average Infiltration Rate (m/d)</th>
<th>Total Hydraulic Loading (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>T1</td>
<td>B</td>
<td>19/01/2004</td>
<td>6.5 hrs</td>
<td>105.0</td>
<td>10.5</td>
<td>2.9</td>
</tr>
<tr>
<td></td>
<td>A</td>
<td>20/01/2004</td>
<td>5.0 hrs</td>
<td>36.0</td>
<td>3.2</td>
<td>0.7</td>
</tr>
<tr>
<td>T2</td>
<td>B</td>
<td>21/01/2004</td>
<td>7.0 hrs</td>
<td>109.0</td>
<td>10.1</td>
<td>3.0</td>
</tr>
<tr>
<td></td>
<td>A</td>
<td>22/01/2004</td>
<td>3.3 hrs</td>
<td>24.0</td>
<td>3.2</td>
<td>0.5</td>
</tr>
<tr>
<td>1</td>
<td>A</td>
<td>28/01 - 04/02/2004</td>
<td>6d 16hrs</td>
<td>974.5</td>
<td>3.7</td>
<td>22.6</td>
</tr>
<tr>
<td></td>
<td>B</td>
<td>04/02 - 11/02/2004</td>
<td>6d 22hrs</td>
<td>2585.6</td>
<td>10.6</td>
<td>64.7</td>
</tr>
<tr>
<td>2</td>
<td>A</td>
<td>11/02 - 17/02/2004</td>
<td>5d 19hrs</td>
<td>927.7</td>
<td>3.7</td>
<td>18.8</td>
</tr>
<tr>
<td></td>
<td>B</td>
<td>17/02 - 24/02/2004</td>
<td>6d 18hrs</td>
<td>2567.6</td>
<td>10.6</td>
<td>64.3</td>
</tr>
<tr>
<td>3</td>
<td>A</td>
<td>24/02 – 02/03/2004</td>
<td>6d 20hrs</td>
<td>973.8</td>
<td>3.7</td>
<td>22.6</td>
</tr>
<tr>
<td></td>
<td>B</td>
<td>02/03 – 10/03/2004</td>
<td>7d 20.5hrs</td>
<td>2941.2</td>
<td>10.6</td>
<td>73.6</td>
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<tr>
<td>4</td>
<td>A</td>
<td>10/03 – 17/03/2004</td>
<td>6d 20.5hrs</td>
<td>960.6</td>
<td>3.7</td>
<td>22.3</td>
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<tr>
<td></td>
<td>B</td>
<td>17/03 - 31/03/2004</td>
<td>13d 21hrs</td>
<td>6103.4</td>
<td>11.2</td>
<td>152.7</td>
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### Table 3.1 Instrumentation of infiltration basins

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<tr>
<th>Instrumentation</th>
<th>No. Per Basin</th>
<th>Comments</th>
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<tbody>
<tr>
<td>Basin periphery piezometers</td>
<td>7</td>
<td>3 piezometers to total depth, 3 piezometers at 6m, 12m and 18m below ground level, 1 piezometer at 2.7m below ground level</td>
</tr>
<tr>
<td>Basin centre piezometers</td>
<td>3</td>
<td>0.1 m slotted section centred at 0.1, 0.5 and 1.0 metres below basin floor</td>
</tr>
<tr>
<td>Water content probes</td>
<td>3</td>
<td>Measuring point centred at 0.1, 0.5 and 1.0 metres below basin floor</td>
</tr>
<tr>
<td>Tensiometers</td>
<td>3</td>
<td>Measuring point centred at 0.1, 0.5 and 1.0 metres below basin floor</td>
</tr>
</tbody>
</table>
3.7 Infiltration Rates and Volumes

The total volume of water infiltrated up to 31 March was 18.3 ML, of which 3896.6 m$^3$ (3.9 ML) was supplied to basin A and 14411.8 m$^3$ (14.4 ML) was supplied to basin B. Water levels within basin A reach the desired depth of 25 cm at supply rate of approximately 1.7 – 1.9 L/s. At the maximum available flow rate of 4.5 L/s water levels in basin B only reached 10 cm due to the higher infiltration rate.

The water levels and supply rate were collected automatically at 10 minute intervals and the data recorded via telemetry at PWC’s Sadadeen Offices. Average rates of infiltration, based on the SCADA data (Figure 3.2) ranged from 3.5 m/day in basin A and 10.5 m/day in basin B. Infiltration rates take into account the effect of water level fluctuations on the active area of the basin. The empirical relationship between the area of each basin ($A$) and the height of the water in the basin is: $A = 63.003 \, h + 33.446$, where $h$ is the height of the water in the basin.

Infiltration rates take into account the effect of water level fluctuations on the active area of the basin. The empirical relationship between the area of each basin ($A$) and the height of the water in the basin is: $A = 63.003 \, h + 33.446$, where $h$ is the height of the water in the basin.

Instantaneous measures of infiltration, based on direct measurement of water level declines within the basin during temporary shutdown of the inflow give good agreement with values calculated from inflow data (refer Figure 3.2). Evaporative losses are small (<1% for both basins) and have not been taken into account. The effect of rainfall on the water balance was small over the trial period (<1% contribution for both basins), however, rainfall events will be assessed with respect to impacts of recharge on piezometric levels.

Figure 3.2 shows no significant decline in infiltration rate in any of the cycles tested. During the course of the trials the temperature of the recharge water has been seen to vary from 26 to 41°C, although the effect on infiltration rate does not appear to have been significant.
Figure 3.2   AZRI infiltration trials SCADA data relating instantaneous flow, basin water level and infiltration rates. Manual measurements of infiltration rate are shown as points.
3.8 Soil Water Dynamics in Shallowest Metre Beneath Basins

The comparative soil water content, matric potential and piezometric head data for the upper metre of the soil profile beneath each basin for two of the cycles are presented in Figures 3.3 and 3.4.

Because of the short duration of the T1 cycle, measurements were collected during the full wetting period as well as the drying period thereafter. The initially dry soil profile provided excellent initial conditions to track the advancement of the wetting front down the soil profile. During cycle 1 the focus of monitoring was the period following the end of the wetting phase.

Data from the finer texture basin A for the T1 event clearly shows wetting to 0.1 m within minutes of initial startup and to 1.0 m after a further two hours. Volumetric water contents after wetting varied from 30-35% with depth. Subsequent to the arrival of the wetting front saturated conditions developed and a groundwater levels rose in each of the three piezometers. The piezometric increase during the 6.5 hour T1 event was 0.3 m for the shallowest piezometer and up to 0.5 m for the piezometer at 0.5 m depth. A vertical hydraulic gradient was retained within the soil profile, with heads rising to above the basin floor in only the shallowest piezometer. Piezometric levels fell quite rapidly at the conclusion of wetting although moisture contents remained high during this period. Water contents declined by 5-10% within the first 24 hours of drying, and cycle 1 data shows that there is very little reduction in water content over 7 days. Matric suction data during cycle 1 gives further support for the dewatering of the soil profile, although the available period of measurement was limited, suctions reached air entry values quickly. Results from cycle 1 indicate that vertical gradients were maintained even over the 7 day drying period.

Data from the coarser textured basin B during T1 shows the advancement of the wetting front was more rapid than for basin A, with the top metre of the soil profile saturated within 20 minutes of startup. The response in piezometer levels to both wetting and drying is rapid due to the high permeability of the soil. Negligible vertical hydraulic gradients were observed for either wetting or drying. Water content data from cycle 1 shows the soils drained more extensively than basin A over the drying period.

At the piezometers situated at 2.7 m below the natural ground surface saturated conditions developed at basin B during infiltration, although data are limited. At basin A no response was observed. At the 6 and 12m depths groundwater rises were observed at both sites, however the data has not been examined in detail.

The maximum storage capacity of the unsaturated zone, given a depth to watertable of 15 m and porosity of 0.3 is estimated to be 5 m. Assuming 1D flow beneath the basin, and that edge effects are negligible, the minimum number of times the unsaturated zone has been flushed up to 24 February is estimated from Table 3.2 to be 8 for basin A and 27 for basin B. A response in the aquifer would clearly be anticipated from very early on (ie. early in cycle 1 for basin A and in T2 for basin B).
Figure 3.3  Soil water content, matric suction and hydraulic head data for basin A. Wetting phase ended for T1 ended at 300mins and basin drained at 410mins. (BoS= bottom of screen)

Figure 3.4  Soil water content, matric suction and hydraulic head data for basin B. Wetting phase ended for T1 ended at 360mins and basin drained at 365mins. (BoS=bottom of screen)
3.9 Effect on groundwater levels and hydrochemistry

3.9.1 Piezometric levels

Groundwater levels within the vicinity of the basins show a consistent falling trend in the order of 5 cm per month over the period from December 2003 to February 2004. This is consistent with the historic trend of a decline over the summer months (refer to section 2).

The hydrographs for two sets of piezometers are presented in Figure 3.5. Note that the piezometers are located within 9 m of the downstream (southeastern) edge of each basin with screened depths of 18 m and 30 m below the ground surface. The hydrographs show that groundwater mounding can be observed during the larger infiltration events. Mounding is most pronounced at basin B (up to 50 cm), although small increases can also be inferred at basin A.

Since an accurate survey of the piezometers has yet to take place, all hydrographs are presented with respect to a local datum (ie. top of casing). The apparent displacement for the hydrographs of the basin B piezometers RN 17952 and RN 17953 is unlikely to be real, since for basin A piezometers RN 17946 and RN 17947 where the casing heights are closely matched and no vertical gradient occurs. Furthermore, the similarity of the response of each pair suggests excellent hydraulic connection between deep and shallow piezometers. Surveying needs to occur as swiftly as possible.

The piezometric signatures arising from infiltration to date have been small, making them difficult to partition from the variations that appear to occur from diurnal changes in atmospheric pressure. Evidence of a barometric influence is based on measured fluctuations in groundwater levels of several centimetres over daily time scales during periods of no infiltration. When the data is filtered by only including measurements taken early in the day (around 08:00 hrs), much of the apparent ‘noise’ is removed. Some targeted monitoring to better understand this phenomenon is warranted, as identified later.

Anomalous behaviour was observed at piezometer RN 17950, situated midway between the two basins, where a rise of up to 3 m occurred from the commencement of cycle 1 in basin B (note that no significant change occurred in the previous corresponding cycle in basin A). What appears to be occurring is that infiltrating water migrates rapidly through coarse deposits until it intersect a less permeable layer and spreads laterally, then intersecting this piezometer, which provides a local conduit for migration of water to greater depths due to absence of bentonite seal. The response is therefore an artefact caused by stratigraphic controls on water movement and inadequate piezometer construction techniques. More careful attention to piezometer construction techniques is warranted in future, particularly with regard to placement of bentonite seal around the casing.
3.9.2 Hydrochemistry

Field measurements of EC, DO and Temp were taken from the 18 and 30 metre depth piezometers after two cycles had been completed (Table 3.3). No significant change was observed at basin A. However, at basin B the reduction in EC, enrichment of DO and increase in temperature of the groundwater of the shallower piezometer give clearly demonstrate that the infiltrated water has reached the watertable. Assuming that EC to be relatively conservative, then the groundwater in this piezometer is likely to be almost entirely composed of the mains water. No change in groundwater chemistry occurred in the deeper piezometer. Flushing of the salt storage in the unsaturated zone appears not to have increased the groundwater salinity. The significance of this issue will be assessed from the geochemical characterisation of soil cores described in Appendix B.

A future issue for trials with recycled water is the challenge presented by the similarity of salinity between the recycled water and the ambient groundwater. Routine tracers such as EC and chloride are unlikely to be as definitive due to the lack of contrast between the two waters. Further work will be required to identify useful tracers for quantitatively tracing the migration of the infiltrating water in the aquifer system.

Figure 3.5  Groundwater hydrographs for nested piezometers adjacent to basins A and B up to the end of June, 2004
Table 3.3  Spatial water quality variations at the infiltration basins

<table>
<thead>
<tr>
<th>Basin</th>
<th>Piezometer</th>
<th>Sample Time</th>
<th>EC (uS/cm)</th>
<th>DO (mg/L)</th>
<th>Temp (°C)</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>RN17946 (18 m)</td>
<td>Start Cycle 3</td>
<td>2020</td>
<td>1.1</td>
<td>26.2</td>
</tr>
<tr>
<td>A</td>
<td>RN17947 (30 m)</td>
<td>Start Cycle 3</td>
<td>2240</td>
<td>0.7</td>
<td>26.3</td>
</tr>
<tr>
<td>B</td>
<td>RN17952 (18 m)</td>
<td>End Cycle 2</td>
<td>812</td>
<td>5.1</td>
<td>28.1</td>
</tr>
<tr>
<td>B</td>
<td>RN17953 (30 m)</td>
<td>End Cycle 2</td>
<td>2410</td>
<td>0.5</td>
<td>26.3</td>
</tr>
<tr>
<td></td>
<td>Average recharge water</td>
<td></td>
<td>768</td>
<td>3.5</td>
<td>31.2</td>
</tr>
</tbody>
</table>

3.9.3 Re-evaluation of analytical modelling

Prior to the commencement of the trial Dillon, (2003) modelled the extent of groundwater mounding likely to be anticipated during the trial as well as for a range of other scenarios. In general, the model predictions suggested much more mounding than has actually been evidenced, ie. metres as compared to centimetres. This implies that the aquifer diffusivity is much higher than previously expected (ie. unconfined conditions).

The analytical modelling approach of Dillon, (2003), based on the Hantush equation for the 6x6m dimensions of the basins was re-evaluated using operational data from the first two cycles of infiltration. Two scenarios were tested, one where the aquifer hydraulic conductivity is 10 m/day; the other with a higher conductivity of 100 m/day. Figure 3.6 presents the results with respect to the hydraulic head build up beneath the basin after 14 days of continuous infiltration for various values of aquifer storativity. They demonstrate the relative independence of storage term, and that values of hydraulic conductivity of the order of 100 m/day would be necessary retain the mound to dimensions observed in the field.

Consequently, the numerical modelling work presented in section 4, which initially used a low value of hydraulic conductivity (4.3 m/day), will also include scenarios with higher values of hydraulic conductivity (50 m/day).
3.9.4 Other data

The potential for increased mosquito or other pest populations is being investigated. Mosquitoes in particular are a potential issue as time-frames for hatching larvae are understood to be less than the current duration of wetting. This has the potential for attracting mosquitoes or increasing populations, and direct discussions will occur with the Entomology Section of Health and Community Services to minimize potential for adverse impacts. To evaluate this potential issue the Low Ecological Services have installed mosquito traps at the site and at a control site distant from the basins and sampled periodically to enable an assessment of the any impacts of the basins on mosquito numbers. Results to date reveal a negligible number of mosquitoes in the traps, and no larvae have been observed in the basins.

3.10 Conclusions

Results from the first two cycles of infiltration tests in two small-scale basins with mains water offer encouragement in terms of the rates of infiltration in both the silty sand (3 m/day) and sand/gravels (10 m/day). No clogging has been observed to date. Water migrates rapidly through the soil profile and deep drainage occurs with minimal mounding (<0.1 m). Some preliminary groundwater sampling confirms the presence of recharge water in the shallowest layers of the aquifer beneath the coarser textured basin B. Although the trial is presently at a very preliminary stage, mains water testing is defining upper limits on infiltration rates and ultimately the capacity of the aquifer to store the volumes required in SAT operations.

3.11 Recommendations

A number of additional piezometers should be installed as soon as possible since the current network is inadequate to characterise the 3D nature of the flow systems beneath the basins due to heterogeneities in the unsaturated zone. Additional soil monitoring equipment including tensiometers and suction cups are required at depths in excess of 1 m, and water quality probes, which will be of most value at the pilot stage in which recycled water will be used should be installed to ensure confidence that the equipment is working well. Some verification that the suctions measured from the tensiometers are reliable and accurate is required. Four pressure
transducers are required to assess fluctuations in groundwater levels over the short term due to barometric effects.

3.12 Acknowledgements

Nadia Crous of DIPE and Jim Gibbons, Kelly Mashford, Frank Boehm, Dirk Van Den Driesen and Steve Sawyer at PWC are gratefully acknowledged for their ideas and efforts in these investigations.
4 MODELLING OF HYDRAULIC HEAD BUILD-UP AND PLUME MIGRATION

4.1 Introduction

Groundwater modelling has been utilised to provide information on the viability of infiltrating 600 ML/yr using SAT, with respect to the resulting groundwater mounding below the infiltration sites and the likely plume migration of the infiltrate under various extraction regimes.

The model has been developed using data from the site characterisation (section 2) and the infiltration tests (section 3) at AZRI to assess the short term effects of infiltrating 600 ML/yr (over the duration of the infiltration trials). Later a more regional model will be developed to provide information on long term effects.

4.2 Model Conceptualisation

The sparse nature of the available data, both temporally and spatially, has resulted in a staged approach to the modelling. The initial model considered here, is conceptually simple and consists of a single homogeneous, unconfined, saturated system (ie. no modelling of the vadose zone). This ‘simple’ model was used to eliminate unknowns about the area.

These include:

- The contribution to recharge from the Todd River and rainfall on the model area.
- Perching effects and unsaturated flow conditions due to variations in the lateral distribution and hydraulic parameters of the strata observed in the unsaturated zone.
- The throughflow from the Inner Farm Basin, Waste Stabilisation Ponds and the irrigation at Blatherskite Park.

Re-evaluation of the model to incorporate new data and to increase the area encompassed will be the next stage.

4.2.1 Model Specifications

The preliminary model domain is 9.2 km² in area and covers most of the AZRI site, spanning from the 557 mAHD contour at the upstream to the 547 mAHD contour at the downstream, based on the 1 metre topographic data. The extents of the model boundaries are seen in Figure 4.1.

A single layer of 33 metres thickness was used for this preliminary model. The AUSLIG 9 second digital elevation model was used to define the upper slice of the model.

The average saturated hydraulic conductivity determined from the permeameter tests of 4-5 m/day (section 2) was initially used for the entire layer.

A porosity of 0.25 was used as the water content of the soils in the area show porosities of between 0.2 and 0.3 as observed in section 3.8.

4.2.2 Boundary Conditions

The groundwater levels decrease to the SE along the axis of the palaeochannel and exhibit a gradient of approximately 0.006 (section 2.6.5). Although the groundwater level data is sparse, the groundwater contours are inferred to mirror the topography of the site.

The north west and south east constant head boundaries have been located along the topographic contours 557 and 546 mAHD respectively. The south west and north east no-flow boundaries, which assume no recharge from flow in the Todd River, run perpendicular to the topographic
contours. The boundary attributes of the model are defined below in Table 4.1, the constant head values provide a groundwater gradient of 0.006 under steady state conditions:

<table>
<thead>
<tr>
<th>Model Time Domain</th>
<th>Boundary</th>
<th>Boundary Condition</th>
<th>Range (mAHD)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Steady State</td>
<td>NW</td>
<td>Constant Head</td>
<td>547.5</td>
</tr>
<tr>
<td></td>
<td>SW</td>
<td>No Flow</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>NE</td>
<td>No Flow (assuming no flow in Todd River)</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>SE</td>
<td>Constant Head</td>
<td>520</td>
</tr>
</tbody>
</table>

Table 4.1 Model boundary conditions

![Figure 4.1 Model domain with respect to the AZRI site.](image)

4.2.3 Model Limitations

The current model has some limitations in that the model does not explicitly take into account infiltration reduction due to clogging processes or effects due to perching of infiltrated water, nor does the model take into account variations in density that would be expected, especially in the infiltration trials with potable water.

The initial model presented here did not take into account the palaeochannel feature, however, following the results of the infiltration tests it has been found that the local effects of the
palaeochannel feature were needed to be incorporated to provide a better representation of the system. The model has been modified and uses a strip of high hydraulic conductivity along the palaeochannel feature to simulate the palaeochannel.

4.3 Simulations

4.3.1 Groundwater Mounding Response to 600 ML/yr SAT Operation

The steady state mounding response using K values of 1, 3, 4.3, 30 and 100 have been determined to provide an estimate of the limit of K, below which, the infiltration of 600 ML/yr would produce excessive mounding. The modelled recharge basin has an area of 41 x 41 metres (1644 m²) with an infiltration rate of 1 m/d.

![Figure 4.2](image)

**Figure 4.2** a) North west – south east cross-section of site showing mounding response for varying model hydraulic conductivity for 600 ML/yr; b) relationship between depth to recharge mound and hydraulic conductivity.

It can be seen in **Figure 4.2** that a K of less than 3 m/d will result in the mound intercepting the basin floor, and values in excess of 4 m/day are required to avoid excessive groundwater level mounding.

4.3.2 Comparison with Response seen in Infiltration Tests

The modelling response observed has been seen to deviate considerably from the first 2 cycles of the infiltration tests (totalling 14 days of infiltration). Mounding at Basin A has been <0.1m, the exact response has proved to be difficult to measure due to barometric variations refer to section 3 for more details. The transient model response using the lowest K=4.3 m/d and a porosity of 0.25 shows a mound of approximately 0.68 metres directly below the basin and 0.50 metres at the observation bore 9 metres down gradient of the basin. This is approximately 10 times greater than the response observed.

4.3.3 Model Re-evaluation

In light of the field data from the infiltration tests, the numerical model has been seen to overestimate the resulting groundwater mound. Analytical modelling results (refer section 3) indicate that a hydraulic conductivity approaching 100 m/d is required to dissipate the infiltrated water and produce the small mound observed. This value was seen to be too high from the small response seen in the steady state simulation of 100 m/day and a value of 50 m/d has been used instead. Transient simulations of the infiltration tests have been conducted using both high and low hydraulic conductivity values of 4.3, 20 and 50 m/day and a specific yield of 0.25 to compare with the field observations. The simulation uses a 6x6 metre basin and assumes an infiltration rate of 3 m/d over 14 days. **Table 4.2** presents the head rise directly beneath as well as approximately 9 metres down gradient from the centre of the basin to provide a comparison with the piezometer response (RN17946). The mounding response to the two scenarios is seen in **Figure 4.3**.
The residualised response, ie. the regional trend has been removed, agrees with the analytical model and suggests that the lower mounding seen in the field is due to a much higher hydraulic conductivity beneath the basins.

Table 4.2 Numerical model mounding response at observation site approximating site geometry.

<table>
<thead>
<tr>
<th>Model Response</th>
<th>RN17946</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hydraulic Conductivity</td>
<td>Head Rise at Basin Centre</td>
</tr>
<tr>
<td>[m/d]</td>
<td>[m]</td>
</tr>
<tr>
<td>4.3</td>
<td>0.68</td>
</tr>
<tr>
<td>20</td>
<td>0.18</td>
</tr>
<tr>
<td>50</td>
<td>0.08</td>
</tr>
</tbody>
</table>

4.3.4 Particle Tracking to Simulate Plume Migration

The steady state simulation of the 600 ML/yr infiltration plume has been modelled using a K of 4.3 m/d and a porosity of 0.25 and a K of 50 m/d and a porosity of 0.25. As stated earlier a porosity of 0.25 was used, as the water content meters of the soils in the basins show porosities of between 0.2 and 0.3 as observed in section 3.8 and the texture of the soil samples show a fairly uniform distribution with depth (see Appendix B).

Figure 4.3 Numerical model calculation of mounding as a result of using K = 4.3, 20 and 50 m/d.

Figure 4.4 a) infiltration simulation with a K of 4.3 m/d and porosity of 0.25 and b) infiltration simulation with a K of 50 m/d and porosity of 0.25.

The simulations have been presented using particle tracking show the migration of the infiltrate with time. The first scenario presented in Figure 4.4 a) is considered an extreme case in terms of the mounding response and lateral containment of the plume. Having said this, it must be noted...
again that these models have not accounted for the palaeochannel feature or the containment effects that it will provide, which will considerably reduce the lateral spread of the plume. The second scenario presented in Figure 4.4 b) is for a K of 50 m/day and a porosity of 0.25. The contrast with the first scenario is considerable with the lateral migration of the infiltrated water being much smaller. The particles tracked over 90 days as presented in Figure 4.4 a) have migrated downstream approximately 150 metres compared with Figure 4.4 b) where the infiltrated water has migrated approximately 250 metres within the 90 days.

The palaeochannel feature has been simulated using a strip of higher hydraulic conductivity (K=50 m/day) along the interpreted location of the palaeochannel flanked by material that has a hydraulic conductivity of 4.3 m/day (Figure 4.5). Note that the simulation shows an almost identical response to Figure 4.4b, with only a slight decrease in the lateral distribution of the infiltrate when compared to the homogeneous layer simulation (Figure 4.4b). This indicates that the lateral distribution of the infiltrated water in both simulations is controlled primarily by the groundwater flow path to the southeast.

![Infiltration Isochrons for Channel of K = 50, Porosity = 0.25](image)

**Figure 4.5** infiltration simulation with the palaeochannel feature

### 4.3.5 Particle Tracking to Simulate Plume Migration/Interception

Three scenarios have been considered in terms of extraction/plume interception simulations. Each scenario infiltrates 600 ML/yr into a 1644m² basin at 1 m/d with a pumping bore situated 250 metres down stream of the basin (the distance was determined from the 90 day travel time for a K of 50 m/d in section 4.3.4) using a K of 4.3 m/d and a porosity of 0.25 and a K of 50 m/d and a porosity of 0.25. The first two extraction scenarios do not take into account the palaeochannel feature to provide the extreme situation, where there are no hydraulic barriers to reduce the lateral spread of the infiltrate. The scenarios are presented below in Table 4.3:
Table 4.3  Infiltration and extraction regimes considered for plume migration and interception

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Infiltration Rate [m³/d]</th>
<th>Extraction Rate [m³/d]</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>1600</td>
<td>0</td>
</tr>
<tr>
<td>2</td>
<td>1600</td>
<td>822</td>
</tr>
<tr>
<td>3</td>
<td>1600</td>
<td>1644</td>
</tr>
</tbody>
</table>

The results of the three pumping scenarios for K values of 4.3 and 50 are presented in Figure 4.6 a) and b) respectively. The steady state model and particle tracking over a time frame of >10 years has been used to demonstrate the long term effectiveness of the extraction bore. The bore has been located using the distance for 90 day travel time determined from the infiltration scenarios in section 4.3.4. The lower K value simulation shows that much of the infiltrated water by-passes the extraction bore and moves off site with much of the water being sourced from throughflow. It does, however, show that the lateral migration of the plume may be controlled using pumping. The higher K simulation shows that for a pumping rate comparable to the infiltration rate that most of the infiltrated water can be intercepted.

Figure 4.6 a) plume migration for K of 4.3 and porosity of 0.25 under the 3 pumping regimes; b) plume migration for K of 50 and porosity of 0.25 under the 3 pumping regimes
Figure 4.7 plume migration for the palaeochannel feature under the 3 pumping regimes.

The palaeochannel model described above in section 4.3.4 was used to simulate the three extraction scenarios. The results are presented in Figure 4.7. Note that the simulation shows an almost identical response to Figure 4.6b, with slightly less lateral distribution of the infiltrate in the palaeochannel simulation.

4.4 Conclusions

The results of the initial modelling indicate that the 600 ML/yr Pilot SAT scheme is feasible in terms of available storage and mound response, although the areal extent of the plume could be extensive. However, these results are expected to be conservative estimates and comparisons between the observed and the predicted mounding response show an over estimate of approximately 5 times those seen. Analytical and numerical modelling indicate that hydraulic conductivities much greater than those derived from the initial model are required to simulate the mounding observed during the field trial.

Estimates of mounding and plume migration using the higher hydraulic conductivity indicate that the lateral migration would be minimal, however, the migration of the plume down gradient could be quite considerable if no water is extracted for reuse with velocities ranging between 300 to 1000 metres per year indicating that the infiltrated water could move beyond the AZRI site after approximately 2 years.

Lateral spread of the infiltration plume is dominated by the groundwater velocity.

As the system is dominated by the flow down gradient, the capacity of the system to store water is a function of how far the water is allowed to move down gradient before being recovered, as opposed to storage due to the build-up of water directly below the infiltration basins.

Under a regime where extraction volumes are comparable or greater than the infiltrated volume, little recycled water is likely to move beyond the extraction bores, which can be placed to maximise capture.
4.5 Recommendations

The modelling has highlighted some shortfalls in the present understanding of the area. The next steps are to:

- Modify the localised model to fit the infiltration data as it becomes available.
- Separation of the saturated layers of differing hydraulic characteristics will be required to account for the higher hydraulic conductivity of the palaeochannel feature.
- Develop the unsaturated zone of the model to better represent the conditions observed on site.
- Develop a regional model that incorporates conservative mass solute transport to determine the extent of plume migration and degree of interception by pumping bores over the long term.
- As part of the regional model determine the relative contributions to recharge from the various sources stated in section 4.2.
- Develop and maintain a monitoring program of the AZRI site and infiltration sites to provide further validation of the model beyond the initial investigation program.
5 LABORATORY STUDIES ON CLOGGING

5.1 Introduction

Laboratory studies are a robust and effective method of assessing a range of SAT design and operational issues under a controlled set of conditions, particularly for a study of this complexity where field studies would be prohibitively costly and logistically challenging.

This section summarises the results to date from a major column study that aims to assist in selection of a SAT pretreatment process by assessing the effects of water quality characteristics, soil type, operational management and climatic influences on infiltration rates. At any location infiltration rates can decline over time due to clogging of the soil.

Clogging can occur due to suspended solids including clays, silts, algae, bacteria, and flocs. Additionally, the formation of biofilms, which consist of bacterial polymer mats develop on the soil surface, and whilst these help filter out particulates they reduce hydraulic conductivities and rates of infiltration. Cyclic drying of infiltration basins has been shown to restore infiltration rates to some extent, by breaking up and mineralising biofilms. Precipitation reactions (eg. of carbonates, phosphates or iron/manganese oxides) and formation of gases (either during microbial degradation of nutrients or when cold water increases in temperature) also can reduce hydraulic conductivities. Also dispersal of clays due to high ratios of monovalent to divalent cations in recharge water compared with native soil water or groundwater can also reduce hydraulic conductivities and mobilise fines.

5.2 Experimental approach

The study was conducted at the Adelaide laboratory of CSIRO Land & Water. The experimental apparatus involved running thirty two 2 cm diameter 10 cm long columns and two 70 cm long columns in parallel over four complete SAT cycles of 7 days of infiltration followed by 7 days of drying over the period from 10 October to 5 December 2003.

The experiment was designed to compare infiltration dynamics of two soil types using four types of water quality (three effluent with varying degrees of treatment as well as potable groundwater). The effect of different ponding depths was also tested. Most columns (28) were established in a controlled temperature (CT) environment in the absence of light, whilst the remainder (6) were replicates of one water type in a nearby glasshouse where temperature and lighting conditions more closely reflected field conditions (Figure 5.1).

Measurements of infiltration rates and basic measures of water quality due to soil passage were made daily. More detailed chemical analyses were undertaken less-frequently. At the end of the experiment the soil within the columns were sectioned and various chemical and microbial analyses performed. A more detailed account of the experimental methods will be provided in a final report in July 2004.
Figure 5.1  Schematic illustration of the experimental design showing a close up of the A2 column (lower middle)

5.3  Characteristics of soil and waters

5.3.1  Characterisation of the Soils

The physico-chemical properties of the two soil types are presented in Table 5.1. Both soils were collected from trench number 27 at the AZRI site (Volume 2, Site Investigations). One of the soils is a coarse textured palaeochannel deposit at a depth of 2.0-2.3 metres below ground surface, which for convenience will be referred to as the 'sand', whilst the other silty sandy soil, we shall call the 'loam' is a finer textured floodplain deposit from a shallower depth (0.6-0.85 m bgs). The sand is two orders of magnitude more permeable than the loam, has very low organic carbon, CEC and carbonate content. The loam has higher organic content and CEC but low carbonate, as per the sand. Both soils have low sodicities, with ESP’s of <2.

The surficial material beneath basin A is considered to be largely loam, whereas material beneath basin B is largely sand.
Table 5.1  Physico-chemical properties of the two soil types

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
<th>Sand</th>
<th>Loam</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sand</td>
<td>%</td>
<td>97</td>
<td>70</td>
</tr>
<tr>
<td>Silt</td>
<td>%</td>
<td>2</td>
<td>17</td>
</tr>
<tr>
<td>Clay</td>
<td>%</td>
<td>1</td>
<td>13</td>
</tr>
<tr>
<td>Porosity</td>
<td>%</td>
<td>54</td>
<td>44</td>
</tr>
<tr>
<td>$\rho_b$</td>
<td>g/cm$^3$</td>
<td>1.6</td>
<td>1.35</td>
</tr>
<tr>
<td>$K_{sat}$</td>
<td>m/day</td>
<td>23</td>
<td>0.09</td>
</tr>
<tr>
<td>OC</td>
<td>%</td>
<td>0.03</td>
<td>0.52</td>
</tr>
<tr>
<td>ESP</td>
<td>%</td>
<td>&lt;1.5</td>
<td>0.3</td>
</tr>
<tr>
<td>$\text{CEC}_{(\text{NH}_4)}$</td>
<td>cmol(+)/kg</td>
<td>3.2</td>
<td>12.1</td>
</tr>
<tr>
<td>$\text{CaCO}_3$</td>
<td>%</td>
<td>&lt;0.5</td>
<td>&lt;0.5</td>
</tr>
</tbody>
</table>

5.3.2  Characterisation of the Waters

The four types of water being tested include:

- Effluent from the Blatherskite Park Irrigation Pump Station
- “Rock-filtered” effluent from pilot plant trial
- Potable quality groundwater from the Roe Creek borefield
- DAF treated effluent from the Bolivar wastewater treatment plant

The three waters from Alice Springs were collected by PWC on September 30 and transported via rail to Adelaide. Bolivar DAF-treated water was collected from float tank number 12 immediately prior to the filtration step on October 3. All waters were stored at 4°C in the dark and subsamples returned to ambient temperature to provide column throughput requirements for up to 3 days. Chemical analyses were carried out periodically to gauge the effect of transport and storage on the stability of the water quality.

The water quality characteristics of the four water types, including the DAF water in both the CT and glasshouse environments, are shown in Table 5.2.
Table 5.2  Quality of water used in the column experiments, as determined from sampling on 27 November 2003

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>BLATH PARK</th>
<th>ROCK FILTERED</th>
<th>DAF-GLASS HOUSE</th>
<th>DAF-CONT. TEMP RM</th>
<th>ROE CREEK</th>
</tr>
</thead>
<tbody>
<tr>
<td>Electrical conductivity</td>
<td>µS/cm</td>
<td>1635</td>
<td>1583</td>
<td>2490</td>
<td>2190</td>
<td>776</td>
</tr>
<tr>
<td>pH</td>
<td>-</td>
<td>7.9</td>
<td>7.9</td>
<td>8.6</td>
<td>8.0</td>
<td>8.9</td>
</tr>
<tr>
<td>Turbidity</td>
<td>NTU</td>
<td>72.1</td>
<td>43.2</td>
<td>0.7</td>
<td>0.6</td>
<td>0.5</td>
</tr>
<tr>
<td>TSS mg/L</td>
<td></td>
<td>142</td>
<td>40</td>
<td>5</td>
<td>2</td>
<td>&lt;1</td>
</tr>
<tr>
<td>Total organic carbon</td>
<td>&quot;</td>
<td>46.0</td>
<td>37.2</td>
<td>12.7</td>
<td>11.7</td>
<td>1.1</td>
</tr>
<tr>
<td>Dissolved organic carbon</td>
<td>&quot;</td>
<td>21.8</td>
<td>17.8</td>
<td>12.1</td>
<td>11.1</td>
<td>1.2</td>
</tr>
<tr>
<td>Chloride</td>
<td>&quot;</td>
<td>211</td>
<td>231</td>
<td>504</td>
<td>464</td>
<td>82</td>
</tr>
<tr>
<td>Bicarbonate</td>
<td>&quot;</td>
<td>521</td>
<td>580</td>
<td>249</td>
<td>229</td>
<td>327</td>
</tr>
<tr>
<td>Sulphate</td>
<td>&quot;</td>
<td>135</td>
<td>104</td>
<td>184</td>
<td>164</td>
<td>53.7</td>
</tr>
<tr>
<td>Calcium</td>
<td>&quot;</td>
<td>59.4</td>
<td>54.1</td>
<td>45.8</td>
<td>39.7</td>
<td>47.7</td>
</tr>
<tr>
<td>Magnesium</td>
<td>&quot;</td>
<td>33.8</td>
<td>31.9</td>
<td>47.8</td>
<td>41.0</td>
<td>24.7</td>
</tr>
<tr>
<td>Potassium</td>
<td>&quot;</td>
<td>21.6</td>
<td>21.0</td>
<td>38.0</td>
<td>32.0</td>
<td>6.2</td>
</tr>
<tr>
<td>Sodium</td>
<td>&quot;</td>
<td>237</td>
<td>244</td>
<td>346</td>
<td>302</td>
<td>96.6</td>
</tr>
<tr>
<td>SAR</td>
<td></td>
<td>6.1</td>
<td>6.5</td>
<td>8.5</td>
<td>8.0</td>
<td>2.8</td>
</tr>
<tr>
<td>Ammonia - N</td>
<td>&quot;</td>
<td>4.3</td>
<td>10.0</td>
<td>0.70</td>
<td>1.1</td>
<td>&lt;0.005</td>
</tr>
<tr>
<td>Nitrate+nitrite-N</td>
<td>&quot;</td>
<td>&lt;0.005</td>
<td>0.68</td>
<td>6.3</td>
<td>6.3</td>
<td>1.2</td>
</tr>
<tr>
<td>TKN-N</td>
<td>&quot;</td>
<td>13.6</td>
<td>17.2</td>
<td>2.2</td>
<td>2.4</td>
<td>0.25</td>
</tr>
<tr>
<td>Filt react P</td>
<td>&quot;</td>
<td>7.2</td>
<td>4.2</td>
<td>4.5</td>
<td>4.2</td>
<td>0.009</td>
</tr>
<tr>
<td>Phosphorus - Total</td>
<td>&quot;</td>
<td>9.1</td>
<td>6.5</td>
<td>5.6</td>
<td>5.0</td>
<td>0.02</td>
</tr>
<tr>
<td>E.coli</td>
<td>/100mL</td>
<td>NA</td>
<td>NA</td>
<td>0</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Total Algae cells/mL</td>
<td></td>
<td>9.5E+06</td>
<td>1.0E+07</td>
<td>5.3E+02</td>
<td>2.7E+02</td>
<td>0.0E+00</td>
</tr>
<tr>
<td>Membrane Filtration Index (MFI)</td>
<td>s/L^2</td>
<td>9600</td>
<td>5900</td>
<td>36</td>
<td>44</td>
<td>22</td>
</tr>
</tbody>
</table>

NA = not analysed

The two effluents from Alice Springs (Blatherskite Park and Rockfiltered) have significantly elevated particulate concentrations (turbidity of 40-70 NTU) due to the high algal content of the water (~10^7 cells/mL), as compared to the more highly treated DAF water (<1 NTU), and naturally the Roe Creek water also (<1 NTU).

The effluents also have considerably more DOC than the mains water, and the Bolivar DAF recycled water only half. Nitrogen and phosphorus levels are typically a factor of 2 higher for the Alice Springs effluents than for the Bolivar DAF water, and 4-200 higher again for the Roe Creek water.

Sodium adsorption ratio’s (SAR) are highest for DAF waters (8-8.5), followed by Blatherskite Park and Rock Filtered water (6.1-6.5), and the Roe Creek (2.8). SAR values correlate positively with the salinity of the waters.

5.4 Results

5.4.1 Infiltration Rate Changes

Pronounced change occurred to infiltration rate in almost all cases over the course of the experiment. Several contrasting examples showing the cases of Blatherskite Park and Roe Creek waters through both sand and loam in the CT environment are given in Figure 5.2. The columns fed by the Blatherskite Park effluent show a rapid exponential decline in infiltration rate within the first cycle from >30 m/day to <5 m/day in sand and from >0.2 m/day to <0.1 m/day in loam. The first
The cases record a decline in the average K/K₀ as a result of the drying phase rather than an increase in capacity as a result of the weekly drying periods was not as significant as expected. Virtually all of the matter at the surface, although some capture still occurred over the soil volume.

The hydraulic conductivity of the soil, expressed as a ratio with respect to that of soil prior to SAT (K/K₀), ranged from 0.001 to 0.05 for sands and from <0.001 to 0.86 for loams. Contrary to the perception suggested from these ranges, the average degree of hydraulic conductivity reduction was lower for the sands than for the loams for reasons discussed below. The restoration of the hydraulic layer would have been most pronounced. This is attributed to the absence of any significant algal deposits on the surface were algal deposits observed over a small proportion of the soil surface over the four cycles. For the sand, the texture was too coarse to collect particulate matter at the surface, although some capture still occurred over the soil volume.

The level of clogging is inversely related to hydraulic loading, and was highest for Blatherskite Park water. The cumulative hydraulic loading (qₜ) through the columns with a 10 cm ponding depth over the four cycles varied from 65 to 565 m for the sands and from 0.3 to 11.7 m for the loams.

The hydraulic conductivity for Blatherskite Park and Roe Creek water through sand and loam (data for the two replicates shown)

<table>
<thead>
<tr>
<th>Sand + BPk</th>
<th>Loam + BPk</th>
</tr>
</thead>
<tbody>
<tr>
<td>G1</td>
<td>D1</td>
</tr>
<tr>
<td>C1</td>
<td>G2</td>
</tr>
<tr>
<td>10 Oct 17 Oct 24 Oct 31 Oct 07</td>
<td>0 0.2 0.4 0.6 0.8</td>
</tr>
<tr>
<td>Nov Nov Nov Nov Dec</td>
<td>Nov Nov Nov Nov Dec</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Sand + Roe Ck</th>
<th>Loam + Roe Ck</th>
</tr>
</thead>
<tbody>
<tr>
<td>G1</td>
<td>H1</td>
</tr>
<tr>
<td>C1</td>
<td>H2</td>
</tr>
<tr>
<td>10 Oct 17 Oct 24 Oct 31 Oct 07</td>
<td>0 0.2 0.4 0.6 0.8</td>
</tr>
<tr>
<td>Nov Nov Nov Nov Dec</td>
<td>Nov Nov Nov Nov Dec</td>
</tr>
</tbody>
</table>

Table 5.3 summarises the key hydraulic characteristics for the entire spectrum of treatments.

The cumulative hydraulic loading (qₜ) through the columns with a 10 cm ponding depth over the four cycles varied from 65 to 565 m for the sands and from 0.3 to 11.7 m for the loams.

The level of clogging is inversely related to hydraulic loading, and was highest for Blatherskite Park and Rockfiltered waters (which were both similar), were several fold greater than for the DAF water, which in turn, was greater than for the Roe Creek water.

The hydraulic conductivity of the soil, expressed as a ratio with respect to that of soil prior to SAT (K/K₀), ranged from 0.001 to 0.05 for sands and from <0.001 to 0.86 for loams. Contrary to the perception suggested from these ranges, the average degree of hydraulic conductivity reduction was lower for the sands than for the loams for reasons discussed below. The restoration of the hydraulic capacity as a result of the weekly drying periods was not as significant as expected. Virtually all of the cases record a decline in the average K/K₀ as a result of the drying phase rather than an increase apart for the two Alice Springs effluents due to the effect of the first drying as previously noted. Not even in the glasshouse did this occur, where the effect of drying and desiccation of any clogging layer would have been most pronounced. This is attributed to the absence of any significant algal deposits on the surface of the soil. Only in the loam soil, where the soil was sufficiently fine to retain some of the algae at the surface were algal deposits observed over a small proportion of the soil surface over the four cycles. For the sand, the texture was too coarse to collect particulate matter at the surface, although some capture still occurred over the soil volume.
Table 5.3  Summary of hydraulic property changes

<table>
<thead>
<tr>
<th></th>
<th>Water Type</th>
<th>$q_t$ (m)</th>
<th>$q_{avg}$ (m/day)</th>
<th>Final $K/K_0$</th>
<th>Avg change $K/K_0$ between cycles</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>SANDS</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Blath. Park</td>
<td>62.6 – 67.7</td>
<td>2.1 - 2.4</td>
<td>0.001</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td>Rock Filtered</td>
<td>55.6 – 56.3</td>
<td>1.9 - 2.0</td>
<td>0.0003–0.0004</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td>DAF-glass house</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>10 cm columns</td>
<td>190.3 – 210.2</td>
<td>7.1 - 7.8</td>
<td>0.013</td>
<td>-0.01</td>
</tr>
<tr>
<td></td>
<td>70 cm column</td>
<td>206.9</td>
<td>7.8</td>
<td>0.22</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>DAF- CT room</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>10 cm head</td>
<td>151.4 – 181.3</td>
<td>5.4 - 6.5</td>
<td>0.027 – 0.033</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td>30 cm head</td>
<td>299.5 – 328.5</td>
<td>10.4 - 11.4</td>
<td>0.011 – 0.018</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>50 cm head</td>
<td>265.1 – 350.4</td>
<td>9.4 - 12.5</td>
<td>0.002 – 0.004</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Roe Creek (mains)</td>
<td>328.2 – 460.7</td>
<td>11.8 - 16.5</td>
<td>0.029 – 0.045</td>
<td>-0.01</td>
</tr>
<tr>
<td><strong>LOAMS</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Blath. Park</td>
<td>1.7 - 2.0</td>
<td>0.06 - 0.07</td>
<td>0.052 – 0.083</td>
<td>0.037</td>
</tr>
<tr>
<td></td>
<td>Rock Filtered</td>
<td>0.3 – 2.1</td>
<td>0.01 – 0.07</td>
<td>0.002 – 0.071</td>
<td>0.014</td>
</tr>
<tr>
<td></td>
<td>DAF-glass house</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>10 cm columns</td>
<td>3.3 – 4.5</td>
<td>0.11 - 0.16</td>
<td>0.10 - 0.19</td>
<td>-0.14</td>
</tr>
<tr>
<td></td>
<td>70 cm column</td>
<td>2.7</td>
<td>0.09</td>
<td>0.096</td>
<td>-0.074</td>
</tr>
<tr>
<td></td>
<td>DAF- CT room</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>10 cm head</td>
<td>7.7 - 11.0</td>
<td>0.27 – 0.38</td>
<td>0.63 - 0.86</td>
<td>-0.14</td>
</tr>
<tr>
<td></td>
<td>30 cm head</td>
<td>13.4 - 17.1</td>
<td>0.46 – 0.59</td>
<td>0.85 - 1.18</td>
<td>-0.27</td>
</tr>
<tr>
<td></td>
<td>50 cm head</td>
<td>6.8 – 22.6</td>
<td>0.24 - 0.78</td>
<td>0.08 - 0.67</td>
<td>-0.075</td>
</tr>
<tr>
<td></td>
<td>Roe Creek (mains)</td>
<td>6.6 – 6.9</td>
<td>0.23 - 0.24</td>
<td>0.22 - 0.28</td>
<td>-0.27</td>
</tr>
</tbody>
</table>

5.4.2 Effect of Soil Type
On a per-unit time basis, the degree of clogging exhibited in the sands is proportionately higher than for the loam. This can be observed from the degree of flow rate reduction in Figure 5.2 and in terms of the relative change in the hydraulic conductivity ($K/K_0$) in Table 5.3. In filtration theory it is well known that physical clogging can be characterised on the basis of the particulate mass flux (Pérez-Paricio and Carrera, 1999) and hence the loam soil may clog as rapidly as the sandy soil when considered in terms of a per-unit mass flux (ie. considering the total hydraulic loading through the soil). Analysis of this issue is continuing.

5.4.3 Effect of Water Quality
The observed ranking for the various water types from highest to lowest clogging rates was:
Blatherskite Park ≅ Rock Filtered > Bolivar DAF > Roe Creek. This conclusion was consistent for both of the soil types tested. Thus, the overall level of clogging appears to be closely associated with the nutrient and particulate status of the infiltrating water.

The level of clogging of the Blatherskite Park and Rockfiltered waters were similar, with little improvement in infiltration as a result of the additional treatment through the pilot rockfilter beds. As can be seen from Table 5.2, the main changes in nutrient levels appear to be an increase in the level of ammonium and reduction in reactive phosphorus concentrations.
Interestingly, there appears to have been sufficient particulate material and/or nutrients present in the Roe Creek water to lead to clogging of the sands over the duration of the experiment. This has also been found in other studies with waters of a similar quality.

5.4.4 Effect of Ponding Depth

Table 5.3 shows that ponding depth had a substantial effect on infiltration rates as expected. As the ponding depth increased, the enhanced head gradient through the column failed to produce a corresponding increase in flow through the column. For instance a 3-fold increase in gradient from 10 cm to 50 cm produced only a 2-fold increase in flow through the column. Therefore greater ponding depth appears to have enhanced clogging. Whether this was due to an enhanced level of clogging agents, or due to greater penetration or compression of the clogging layer remains to be established by further analysis of the data.

5.4.5 Effect of Climatic Conditions

The controlled temperature environment experienced a temperature range of 19-21°C over the majority of the experiment, however, a mechanical failure during the fourth wetting cycle caused temperatures to rise as high as 26°C on occasions during the late stages of the experiment. Rates of pan evaporation varied from 0.05-1.5 mm/day during wetting periods and radiation levels were limited to fluorescent light at times of monitoring (and complete darkness at other times). In the glasshouse climatic conditions were of course much more variable. Temperatures varied diurnally, with ranges of 16-37°C recorded. Rates of pan evaporation were higher (1.5-3.5 mm/day) and solar radiation intensity was in the order of half that of outdoor conditions due to the glasshouse being covered with shade cloth material.

Monitoring of the volume changes of the outflow of one of the sand columns over several days was achieved with a load cell coupled to a PC-based data collection system. The quality of the water in the feed tank to the columns was also monitored on a frequent basis. Over a daily time-scale four-fold differences in infiltration rates have been measured in the glasshouse (Figure 5.3). Lowest flow rates have occurred in the late afternoon when water is warmest; highest rates in early morning when water is coolest. This is opposite to the behaviour, which would be induced by viscosity changes associated with the heating and cooling of the water. Close-up inspection of the column revealed gas bubbles on the soil surface during the warmest periods of the day. These were absent during cooler periods. Therefore infiltration rate variations appear to be linked to the liberation of gas bubbles which then presumably become entrained within the soil. Diurnal changes in the pH of the water, probably as a result of differences in algal respiration may also have an affect on infiltration rates.

Attempts to repeat the monitoring in a loam column during the subsequent cycle was unsuccessful due to the small volume change in outflow as compared to the evaporative losses. Observations made in the controlled temperature environment revealed no flow rate oscillations over the daily time-scale as expected.
Figure 5.3  Flow rate variations in one of the sand columns located in the glasshouse over 4 day period and associated changes in air temperature and water quality
5.4.6 Relationship between Clogging and Soil Microbiological Parameters

It has previously been shown that soil clogging can be described as the result of chemical, physical and biological processes. Unlike physical clogging, which happens in the top millimetres of the soil column, biological clogging is assumed to happen throughout the depth of the soil column. It is reasonable to assume that due to the interaction with physical and chemical processes as well as competition of the microbiological representatives the microbiological parameters will show gradients over depth. In order to assess microbial activity in relation to depth in the soil columns, they were sectioned and two key parameters were analysed. These are the biomass, at this preliminary stage presented as the ‘biomass index’ being the fatty acid concentration, and the polysaccharide concentration. The latter, being an indicator for biofilm formation and thus reduction of pore space is a main suspect for biological clogging. Figure 5.4 and 5.5 are meant to be representatives for the full range of combinations covering both sand and loam as well as polysaccharide and biomass index.

Please note that although the relative hydraulic conductivity $K/K_0$ is significantly higher for the loams the effective hydraulic conductivity of the loams is smaller by about two orders of magnitude. Since the water quality parameters are fairly different for the different waters and other effects are interfering, clear trends can not be expected here. However, it still allows comparisons to be drawn. The analysis of the data will focus on relating these microbiological parameters to changes in the water quality during soil passage, which will allow the identification of the critical water quality parameters responsible for biological clogging. Based on these findings soil-specific recommendations for a pre-treatment in order to minimize biological clogging can be derived.

![Relative Hydraulic Conductivity given as $K/K_0$ over Biomass concentration](image)

Figure 5.4 The relationship between the relative hydraulic conductivity at the end of the last wetting cycle and the average biomass index for the loam
Figure 5.5 The relationship between the relative hydraulic conductivity at the end of the last wetting cycle and the average polysaccharide concentration for the loam

5.5 Representativeness of the water used in this study

5.5.1 Blatherskite Park

Table 5.4 compares water quality data from the Blatherskite Park Irrigation Pump Station over the past 10 years in relation to that used in this study. We see that all but two of the analytes lie within the measured range. The exceptions are SAR (lower than previously measured), and bicarbonate (higher than previously measured). The low value of SAR of 6.1 for the water used in the column study is clearly the outcome of the relatively low sodium and high magnesium concentrations. Fortunately the SAR values of the Bolivar water were more typical of the Alice Springs effluent.

The column study water is slightly more neutral and marginally higher in nutrients and organic matter than average values, but is still within the range previously measured.
Table 5.4  Quality of effluent at Blatherskite Park Irrigation Pump Station between 1993-2003 in relation to water collected from this site for study collected on 30 September 2003

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>N</th>
<th>Max</th>
<th>Min</th>
<th>Mean</th>
<th>1997-2003 data</th>
<th>This Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>-</td>
<td>89</td>
<td>10</td>
<td>8</td>
<td>9</td>
<td>7.9</td>
<td>7.9</td>
</tr>
<tr>
<td>EC</td>
<td>µS/cm</td>
<td>92</td>
<td>2,920</td>
<td>1,041</td>
<td>1,716</td>
<td>1,635</td>
<td>1,635</td>
</tr>
<tr>
<td>TDS</td>
<td>mg/L</td>
<td>51</td>
<td>1,641</td>
<td>657</td>
<td>991</td>
<td>981</td>
<td>981</td>
</tr>
<tr>
<td>TSS</td>
<td>“</td>
<td>52</td>
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* median value
# estimated from EC

5.5.2 Bolivar DAF

The Bolivar DAF water used in the column study easily met its target requirement of Class A water for unrestricted reuse for irrigation, even at the point of collection prior to the filtration and chlorine dosing steps (Bosher et al, 1998). On the occasion sampled the water was marginally above average but still below the reuse standards in terms of salinity (1300 cf 1500 mg/L TDS), whilst particulate levels were well below average (<1 cf 10 NTU) and so too were E. coli counts (0 cf 10 cfu/100mL) which are strongly associated with turbidity levels. Nitrogen and phosphorus levels are considered typical of values since the commissioning of the activated sludge plant.

This shows that the waters sampled on 3 October 2003 were of a very good quality relative to typical plant performance. The quality of the water produced by the DAF/F plant is known to be strongly dependent on the algal population within the stabilisation lagoons, which vary seasonally of course, but can also vary significantly over shorter time-frames (Buisine and Oemcke, 2002). The high quality of water within the lagoons at the time of sampling appears to have been a key determining factor on the quality of water used in this study.

5.5.3 Roe Creek

Due to the scale, age and limited replenishment of the Mereenie Sandstone aquifer the quality of the Roe Creek groundwater is expected to be reasonably stable over time. This is supported, for
example, by the results of water quality analyses reported by Hancock et al, (1986), which are similar in terms of salinity, nutrient and organic carbon to the water used in this study.

5.6 Conclusions
A comprehensive column study involving the running of 34 columns in parallel was undertaken over an 8 week period to assess the effect of source water quality, soil type, ponding depth and climate. Infiltration characteristics are highly dependent on the quality of the source water for the two soil types tested – the higher the nutrient and particulate levels, the higher the clogging observed. Biological clogging and some physical clogging are suspected to be the main causes. The average infiltration rate through the sands over 4 wet/dry cycles ranged from 16 m/day for the mains water, followed by 6-7 m/day for the Bolivar DAF water, and 2 m/day for the two Alice Springs effluents (Blatherskite Park and Rock Filtered). Fluxes through the loam were naturally lower (<0.6 m/day), but the trend was unchanged. These results demonstrate that the current level of treatment of the effluent would lead to high rates of clogging and further treatment will be required to reduce clogging to acceptable levels.

5.7 Recommendations
Current work involves completing the interpretation of the data and producing a detailed CSIRO report by May 2004.

It is recommended that if a field trial with recycled water is delayed, that a columns study is undertaken in larger columns to assess water quality improvements in both media types and to determine the need for and effect on clogging rates of a vermiculite amendment for the sand columns.

5.8 Acknowledgements
The efforts of Toney Hynick (CSIRO), Kelly Mashford (PWC), Don Pidsley (PWC) and Danny Tintor (UWI) assisted greatly in making this study possible.
6 WATER QUALITY ISSUES

6.1 Description of processes

Soil Aquifer Treatment (SAT) involves the process of geopurification by infiltrating recycled water through the unsaturated zone using cyclic wetting and drying cycles. This is believed to provide conditions that are more conducive to both the treatment and recharge than could be achieved through the continuously ponding of water. Water quality deteriorations due to leaching of minerals such as iron, manganese in the unsaturated zone may also occur initially.

The constituents that may be present in recycled waters may include particulates, nutrients, metals, bulk and trace organics, as well as pathogenic microorganisms. Cycling wetting and drying produces recurring aerobic and anaerobic conditions immediately below infiltration basins and uses the soil as a natural filter to reduce concentrations of contaminants in infiltrating water through physical, chemical and microbiological processes. Thus, suspended solids can be filtered out, transport of inorganic and organic pollutants can be retarded through adsorption on the soil matrix, organic pollutants can be degraded and decomposed by soil microorganisms and pathogenic bacteria, viruses and protozoa are attenuated by adsorption, predation by indigenous bacteria and die-off (National Research Council, 1994). Comprehensive discussions on water quality aspects of recharge enhancement are given by the National Research Council, (1994) and Dillon, (2002).

A brief description of some of the key processes, which positively or negatively affect water quality as presented next, followed by a summary of the initial results of a lab study conducted as part of this study.

6.2 Water quality improvements

6.2.1 Microbial pathogens

The main health-related problem associated with the reuse of recycled water is microbial pathogens, even where the required use is irrigation. Therefore inactivation of pathogenic microorganisms including pathogenic bacteria, viruses, protozoan cysts and helminth eggs are of great importance with respect to safeguarding human health. Survival of pathogenic microorganisms in groundwater diminishes with residence time. This is because the introduced microorganisms are filtered, adsorbed, die-off or are degraded by native microorganisms to varying degrees in different hydrogeological settings (Pavelic et al., 1996; Toze and Hanna, 2002).

The survival of pathogens and their resistance to disinfection in some instances may be considerably higher than that of indicator bacteria (eg. E. coli and coliforms). Enteric protozoa are usually present in the form of cysts, and these have the potential to remain viable in groundwater for substantial periods of time. Typical inactivation rates, defined as the time for a one log removal (90% reduction), are in the order of 3 to 6 days for indicator bacteria, but may be higher for other bacteria, between 5 to 30 days for viruses, and unknown for protozoa.

The effectiveness of soils, aquifer media and groundwater to remove microbial pathogens from infiltrating water depends on a wide range of factors. The issues relating to microbial movement and survival are complex and poorly understood. Studies are currently underway that address this (Toze and Hanna, 2002, Gordon et al. 2002).

Safeguarding human health can be achieved by ensuing an adequate minimum residence time of the recharge water in groundwater. For example, in Europe it was common practice to allow 50 days residence time in groundwater (van Waegeningh, 1985) between points of recharge of undisinfect water and points of extraction for potable supplies.
In this case a minimum residence time of the recharge water in the groundwater system is needed to provide adequate safeguard for the protection of human health. From a risk management perspective, such a safeguard is necessary to deal with uncertainties in aquifer characteristics, most particularly due to preferential flow which causes faster-than-average movement of water and contaminants through part of the soil/aquifer volume to other groundwater users.

6.2.2 Nutrients

Nitrogen is ubiquitous to all but the most highly treated recycled waters. SAT can also transform and remove dissolved nitrogen through initial adsorption of ammonium on soil during wetting cycles, nitrification of dissolved and adsorbed ammonia during aerobic drying cycles, followed by denitrification during subsequent wetting and leaching of residual nitrate.

Nitrogen removal is of particular appeal in places where recycled water is to be used for potable purposes, however removal of nitrogen compounds is characteristically variable. A study of the Dan site in Israel show high removals (50 to >90%) but denitrification is reported to be associated with perching of infiltrate during passage to regional groundwater table. Crites (1984) reports nitrogen removal rates of 93% using primary effluent SAT. Field reports from the Flushing meadows site show 30% removal of total-N where loadings are high and 65% where low (Bouwer et al., 1980). Ho et al., (1992) reports no observable denitrification in pilot field studies using mixed secondary and primary effluent. Removal of total-N (a measure of the combined effectiveness of nitrification and denitrification) can be high if denitrification is optimised. Since organic carbon is characteristically low in most aquifers, denitrification usually takes place near the soil surface.

Phosphorus is removed by rapid sorption reactions followed by slower precipitation reactions, either under low or high pH conditions.

Organic carbon (OC) is degraded ultimately to carbon dioxide, water and turnover of biomass in nutrient transformation reactions. Microorganisms also participate in producing these end products or smaller molecular weight compounds.

6.3 Water quality deteriorations

6.3.1 Metals / Trace elements

Changes in the geochemical composition of the soil profile occur due to dissolution and mineral weathering that are enhanced by SAT process. The long-term capacity of the soil to remove heavy metals and other trace elements is presently unknown, however, concentrations of these constituents in samples of the water from the waste water treatment plant are currently either below detection limits or the values recommended in the drinking water guidelines.

A study to characterise the physical and geochemical nature of sediments at the AZRI site is being undertaken based on core samples from five locations. The initial results arising from this work are presented in Appendix B.

6.3.2 Salinity / Recovery efficiencies

Recovery efficiency (RE) relates to the quantity of water of a suitable quality which may be recovered and is particularly relevant where the ambient groundwater is of poorer quality than the source water, typically with respect to salinity. Much has been learnt about recovery efficiencies during aquifer storage and recovery operations (eg. Pavelic et al., 2002) and there are obvious similarities with SAT.

RE will be affected by hydrogeological variables (eg. transmissivity, regional groundwater flow rate), which cannot be controlled, and by management variables (eg. infiltration volumes, distance to recovery bores), which are easier to control. Because RE is controlled by so many variables, RE is
difficult to predict in advance. On the other hand, as section 4 demonstrated, groundwater modelling predictions can be indicative where aquifer hydraulic and transport properties are reasonably well known. These do not however preclude the need for field trials.

Recovery of SAT water is necessary for reuse to occur and assist in the containment of the infiltrating water. Most SAT schemes use recovery bores to hydraulically isolate the SAT area from other groundwater users. If storage / travel times to recovery bores are sufficiently long the aquifer can provide additional treatment and residual contaminants will become attenuated and diluted in the aquifer.

6.4 Preliminary Results from Laboratory Study

A secondary objective of the laboratory column study detailed in section 5 of this report is to determine the removal of organics, nutrients and pathogens monitoring some of the water quality transformations that occur during passage through the shallow soil profile. Changes in water quality as a result of soil passage at any point during a wetting phase were determined from simultaneous sampling of the inflow and outflow from the column.

Unfortunately at the time of writing the full suite of water quality data had not yet been received from the analytical laboratory. However sufficient data is available for the Bolivar DAF water to warrant some discussion. This shows that in the controlled temperature environment water quality changes due to passage of Bolivar DAF water through the sand columns appear to negligible over the 10cm scale (ie. most parameters behave conservatively during flow through the column. Nitrate, total phosphorus and dissolved organic carbon behave conservatively whilst a small amount of ammonia removal occurs (<5%) as well as TOC, which is consistent with the retention of some of the particulate matter in the infiltrating water.

In the glasshouse environment ammonia removal over short (10 cm) and long (70 cm) columns is again less than 5%. A small increase in nitrate levels occurs in response to this decline, presumably due to nitrification. Some removal of TOC occurs, particularly for the 70 cm column and removal of DOC of up to 10% occurs only over 70 cm. Negligible phosphorus was removed. Unfortunately an absence of *E. coli* in the recharge water prevented the possibility of observing removal of this indicator organism. Data for the loam soil are as yet unavailable.

6.5 Conclusions

An evaluation of the fate of contaminants in the subsurface is essential to ensure that SAT is sustainable over the long term, that recovered waters meet targets for beneficial use, and that groundwater quality is protected in the long-term. Water quality changes in the unsaturated and saturated zones can include water quality improvements (eg. N, P, pathogens), or water quality deteriorations (eg. leaching of salts, Fe, Mn). At the project site there is a negligible overall risk for the environmental values of the aquifer to be adversely impacted, and water quality improvements are anticipated. Risks associated with potential leaching of minerals in the unsaturated zone and the capacity of the soil to act as an adsorbing/purifying filter is currently being investigated.

6.6 Recommendations

*Identify sources of groundwater at AZRI site*

Information recently collected shows that the groundwater quality across the AZRI site varies, and there are at least four potential sources of water in the Outer Farm area. A study is required to identify and map the relative contributions of these potential sources using geochemical and isotopic techniques. This would lead to an improved understanding of the groundwater flow systems at the AZRI site as well as generate baseline information for the SAT investigations.
Assessment of tracers for monitoring the infiltrating plume

Characterisation of the plume migration and travel time to pumping bores is an important measure of the SAT performance. However the salinity of the brackish groundwater at the SAT trial site does not differ substantially to that of the recycled water that will be infiltrated. Routinely used tracers to identify the presence and proportion of recycled water in groundwater such as EC and chloride are unlikely to be successful. Further work is needed to characterise the composition of the two end-members and identify if naturally occurring tracers can be used. In the event that these cannot be found, the possibility of applying artificial tracers such as noble gases etc should be considered.

Extend knowledge on water quality changes through lab study

Trials with recycled water at the 600 ML/yr scale are unlikely to commence until early 2005 at the earliest. In order to progress the project and minimise ‘down-time’ studies on water quality changes should commence as soon as possible. The most feasible option would be to conduct a laboratory experiment at CLW in Adelaide to assess water quality changes through large soil columns. Details are outlined in section 5.
7 DESIGN PRINCIPLES FOR A 600 ML/YR PILOT SAT SCHEME

7.1 Introduction

The feasibility of the application of SAT technology at the Arid Zone Research Institute near Alice Springs is being verified through staged trials. Two small (6m x 6m) basins recently constructed and instrumented at AZRI have been recharged with the mains water supply from the Mereenie sandstone aquifer to define the upper limits on infiltration rates and to assess the storage potential of the shallow Quaternary aquifer. This site is expected to remain a test-bed for SAT investigations/research due to the high density of monitoring in the unsaturated zone and the groundwater. Subsequent testing should involve the use of recycled water.

Although the analysis of the data from the preliminary trial with mains water is as yet incomplete, sufficient knowledge now exists to allow design of the Pilot SAT recharge basins capable of infiltrating 600ML/year of recycled water. This section builds upon the CSIRO Land and Water document dated 1 October 2003 titled ‘Operational Plan for Preliminary Trial and Trial of SAT Basins’, along with knowledge gained over the past 5 months of this study.

7.2 Conceptual Design of the Pilot SAT System

7.2.1 Basin location

The location of the Pilot SAT basins should be selected on the following criteria:

- Silty sandy soils are preferred for the basin floors (similar to basin A)
- they overly the palaeochannel feature
- that they are located in areas with negligible tree cover (to prevent disrupting culturally sensitive trees species)
- proximity to preliminary basins to maximize the benefits of the existing monitoring infrastructure
- relatively close proximity to the delivery pipeline to reduce costs
- the present location is fully fenced, which will prevent livestock from entering the basins and causing erosion and possible basin floor compaction

The silty sandy soils in Basin A are preferred to the coarser gravelly sand soil of Basin B to maximise the retention of clogging agents near the surface and to minimise the opportunity for the lateral spread of infiltrated water in more permeable layers. Localised soil investigation will be needed to map the distribution of the silty sandy soils at the selected location (see below).

7.2.2 Principles of basin design

The design of the Pilot Basins should be as simple as possible and allow for:

- alternative filling between two sets of basins to provide intermittent wetting and drying (notionally 7 days wetting followed by 7 days drying as per preliminary basin trial and column study)
- either direct supply to each of the sub-basins with the flow being controlled by the basin water levels or by allowing gravity flow between the sub-basins controlled by spillways constructed to the operating depth of 0.3 m above the basin floor.
- expansion of the scheme in future as the technical viability is demonstrated and demand for SAT water increases
- targeting the silty sandy soils
7.2.3 Basin size

The anticipated size of basins necessary to infiltrate 600 ML over a 12 month period operated with equi-length wetting and drying cycles, is conservatively estimated to be two hectares (20,000 m²). This assumes a minimum infiltration rate of 0.3 m/day (and with allowances for surge flows, through say rainfall runoff inputs). The volumetric supply rate to the basins would therefore be 1600 m³/day or 1.6 ML/day (~20 L/s).

The infiltration rate estimate of 0.3 m/day is based upon the lowest of the rates of 3 m/day, determined during preliminary infiltration trials (section 3) and allowing for an order of magnitude decline due to permeability reductions associated with soil clogging (section 5), assuming the minimum target for the level of pretreatment of the recycled water defined previously is met.

7.2.4 Pilot SAT Conceptual Design

Figure 7.1 presents a possible design scenario for the SAT Pilot Plant using a direct controlled supply to each of three pairs of basins (six in total). The basins allow for 1600 kL/day infiltration at 0.3 m/day (600 ML/yr over a 12 month period). The basins have been designed to provide flexibility in the infiltration rate or the supply rate. As the infiltration rate decreases or the supply increases the other sub-basins can be employed to account for the short fall in surface area of the operating basins. For instance, only one pair of basins would be active for an initial infiltration rate of 1 m/day, whilst all three sub-basins would be needed as the infiltration rate reduces to 0.3 m/day, due to clogging. The “surge” basins provide extra flexibility in terms of the peak flow into the basins, providing another 530 kL/day infiltration capacity at an infiltration rate of 0.3 m/day.

The water supply to each of the basins in the presented scenario is to be controlled by the individual basin water levels with an actuated flow valve close to the outflow point (utilising a system with better control characteristics than the butterfly valve currently employed at the preliminary infiltration basins).

Each of the basins will require an energy dissipater at the point of inflow to prevent erosion of the basin floor, possibly including a rock or concrete apron type structure.

The basins will have a minimum depth of 0.5 metres below the surface, with the individual sub-basins having flat, horizontal floors (constructed using laser levelling). Excess soil material could either be mounded around the perimeter or transported off-site.

The slope of the basin walls should be around 1:2 to reduce the erosion of the walls and to provide an easy exit for small animals, which may enter the basins. Matting or waste rock material will also be required to stop erosion of the banks. The ponding depth of the basins is 0.3 m.

Whilst the proposed 7 day wet/dry cycles should allow for the restoration of short-term infiltration rate declines, some residual clogging will occur over the longer term which will require manual intervention. Solutions to the problem include disc ploughing to break up the clogging layer, or scraping and removal of the clogging layer. Access points should also be considered for the maintenance of the basin floor and sides (ie, possibly included in the energy dissipater construction). Regular drying also provides mosquito control, as described in section 7.6.
7.2.5 *Relationship between Pilot SAT plant and the trial basins*

The Pilot SAT basins presented in Figure 7.1 are located to the northwest of the trial infiltration basins. It is anticipated that the trial basins would be utilised for intensive studies on the hydraulic mounding changes, infiltration rate changes due to clogging, pathogen attenuation and geochemical studies in the future. As the location of the larger scale Pilot SAT basins would “swamp” the groundwater and hydrochemical response from these small basins, it is suggested that locating the basins approximately 200 metres to the southeast should also be considered if the surficial soil type is suitable.

7.3 *Monitoring Program for Pilot Basins*

Monitoring is necessary to provide assurance that the groundwater quality is protected, that the water recovered from pumping bores is fit for its intended use, and to provide early-warning if undue risk is occurring (these are identified below). Monitoring is also necessary to ensure that the SAT operation performs as intended, and that the recharge rates and volumes are known.

The objectives of monitoring are to:

- assess the hydraulic loading (ie. capacity per unit area of infiltration basin)
• assess the changes in groundwater storage
• assess the quality of the recovered infiltrate after passage through the subsurface
• confirm that there are no adverse environmental or health impacts

The flow rate of water into each of the basins should be monitored using a meter that records flow rate and cumulative volume. Transfers of water from one basin to another must also be taken into account. The water level in each basin should also be recorded.

Water levels in piezometers, both in the immediate area and across the Outer Farm area should be monitored to assess seasonal and longer-term changes in groundwater storage.

Samples of the recharge water should be collected at a frequency adequate to characterise the seasonal differences in water quality, approximately every 2 months. Groundwater samples in the upper layers of the aquifer immediately adjacent to the basins should be collected at a similar frequency. Groundwater sampling should also occur at one or more locations within close proximity of the basins (<20 m), intermediate to the basins and the extraction bores, and at the extraction bores. Water levels and water samples should also be collected regionally to assess the dimensions of the infiltrated plume.

The initial frequency of 2 months is higher than would be expected over the longer term in order to demonstrate that the primary objectives are met and that no hazard exists. The analytes to be monitored in the recharge water and in the groundwater should include field parameters, major ions, metals, nutrients and microbial indicators.

7.4 Minimum Pre-treatment of Recycled Water for SAT

Additional pre-treatment will be required if the recycled water is to meet the dual needs of the SAT system and the restricted non-potable direct reuse.

Restricted direct reuse is envisaged to require water quality targets similar to those provided in the 1999 South Australian Reclaimed Water Guidelines. Whilst we are not suggesting that these guidelines are appropriate or applicable to Alice Springs, they could serve as a useful reference point. These guidelines suggest values of suspended solids < 30 mg/l and BOD < 20 mg/l and pathogen reduction is required for irrigation with no direct contact of water with crops. It is clear from the column study that targets such as these would also have a beneficial effect on basin infiltration rates.

The column study results presented in section 5 demonstrated that considerably higher rates of infiltration are achieved with the more highly treated Bolivar DAF water as compared to Alice Springs effluent at the current level of treatment. In terms of their clogging potential, these effluents differ primarily in terms of their particulate and nutrient concentrations. The primary benefit of improving water quality for SAT will, therefore, be to enhance infiltration rates and hence reduce the area that must be dedicated to the infiltration basins.

The requirement for disinfection, whilst necessary for direct reuse to ensure risks to human health are minimized, is unnecessary for SAT since it will remove pathogens with provided there are adequate residence times. Disinfection of the recycled water will however, reduce the risk of polluting the aquifer through the introduction of pathogens to virtually nil. The risks associated with the introduction of potentially harmful disinfection by-products associated with chlorination such as trihalomethanes and haloacetic acids, should be considered, although they are thought to be a much lower risk to health than if the water for direct reuse were not sterilised.
7.5 Residence Time Requirements and the Recovery System

Survival of pathogenic microorganisms in groundwater diminishes with residence time due to filtration, adsorption, die-off and degradation by native microorganisms. A minimum period of time for the recycled water to reside in the groundwater system is required to provide adequate safeguard for the protection of human health in terms of the microbial quality where the groundwater is used for potable purposes. This is a consideration in order to protect groundwater users (there are no known potable users) situated well-away from the AZRI area, however, for recovery of the infiltrated water at the AZRI site for irrigation this residence time is a non-issue since the recycled water will already meet the required standards for reuse.

From a risk management perspective, such a safeguard is necessary to deal with uncertainties in aquifer characteristics, most particularly due to preferential flow, which causes faster-than-average-movement of water and contaminants through part of the soil/aquifer volume. Routine monitoring of groundwater quality focusing on constituents that are present in the recycled water that behave conservatively in the subsurface is fundamental to ensuring that the recycled water meets this criterion. An apparent lack of contrast between the salinity of the source water and the groundwater quality at the site provides additional challenges. Further work in this area is recommended.

Preliminary modelling suggests that a minimum of five bores will be required to intercept the infiltrate plume as it migrates downstream from the basins (yields of bores in the area range from 1-5 l/s). It is possible that further pumping bores could be utilised approximately 50 metres upstream of the Pilot SAT basins to reduce the volume of infiltrated water migrating downstream (see Figure 7.1). It should be noted that the peak demand associated with horticultural activities will, however, require up to 50% of the annual demand to be extracted during the summer months, therefore, approximately 10 to 11 bores pumping at 5 l/s will be necessary to meet these peak demand periods.

7.6 Environmental Risks and their Management

A list of potential risks was drawn up simply to ensure that they were accounted for in the site location, design, operation and monitoring. These are:

1. Potential for contamination of the underlying aquifer – The groundwater in the shallow aquifer system throughout this area has a salinity exceeding the bounds for drinking water supplies (section 2.6.5) and the applied recycled water from the treatment plant is expected to have salinities below the present groundwater salinity. Observation bores will be constructed on the down gradient side of the basins, and for most basins on other sides as well to enable assessment of changes in groundwater level and quality as a result of the recharge operation. A sampling and analysis program for a suite of analytes including major ions, metals and nutrients will commence before infiltration commences to measure ambient concentrations in the nearest bores. There is a small probability that mobilization of salts will occur as a first-flush effect and that if the peak concentrations exceed those of ambient groundwater they will soon decline. There is a negligible overall risk for the environmental values of the aquifer to be adversely impacted.

2. Potential for soil degradation – The primary interest here is the potential for changes in hydraulic conductivity due to sodicity. If this occurred, this would impact on infiltration rates and possibly require changes in operation.

3. Potential for adverse impacts on adjacent vegetation – traffic on site can disturb plants, and so general access to the site will be on existing roads. There is also the possibility that water may move laterally through shallow sand layers that could water some plants at a rate beyond their normal habitat. A basic vegetation survey in the immediate vicinity of the ponds, eg via photographic records before and after infiltration, and observations in shallow
piezometers would identify whether this is an issue. This issue is regarded as having a very low risk.

4. **Potential for increased mosquito or other pest populations** – mosquitoes require still water for about four days for hatching larvae and understood to require emergent vegetation. Ponds are expected to be filled for seven days before drying for a similar duration in a cycle that is repeated. This has potential for attracting mosquitoes or increasing populations, and direct discussions will occur with the Entomology Section of Health and Community Services to minimize potential for adverse impacts. To evaluate this mosquito traps will continue to be deployed before infiltration commences and during the wetting and drying cycles. If numbers become excessive it is possible to reduce the duration of the wet part of the cycle. If problems persist chemicals will not be used to controlling mosquitoes unless it can be demonstrated that the chemical is removed through soil aquifer treatment.

5. **Potential for visitors or native animals to slip into the ponds and drown** – The pond depth will be only approximately 0.3m, so it is unlikely that creatures or visitors could drown in the basin. Side slopes will be no steeper than 1:2 enabling relatively easy escape. The problem will be averted by fencing the ponds. For ponds on AZRI land there is the added security of a perimeter fence, locked gate and security guard.

6. **Potential for water table to rise to the surface causing salinity** – piezometers will record piezometric heads at a range of depths, typically 6, 12, 18 and 30 m and piezometers will be installed in shallow sand layers (at about 1m depth). Monitoring of these will allow early warning of a rising water table and if this occurs this suggests changes in design or location of ponds is required.
8 REFERENCES


APPENDIX A - INFILTRATION SITE PHOTOGRAPHS

SCADA telemetry, flow meter and actuated flow valve components. Measurements of flow and basin water level are recorded every 10 minutes.

Basin B water delivery and monitoring instrumentation.

Basin B Monitoring instrumentation prior to first wetting phase – cycle1.
Beginning of drying phase cycle 2 for Basin A. Note the PVC tube on the basin floor to reduce ripples on water in basin directly below the water level sensor.
APPENDIX B - STAGE 1 CHARACTERISATION OF PHYSICAL AND GEOCHEMICAL NATURE OF UNSATURATED SEDIMENT CORES

Issues
Some of the main issues that drive this study are:

1. the placement of monitoring equipment; piezometers, tensiometers, soil moisture measurement and soil solution sampling equipment (ie identify texture contrasts, air entry values)

2. the potential for changes in soil structure due to sodicity (geochemical clogging)

Soil core processing
The as-received sub-surface core sections were photographed prior to sub-sampling and crushing prior to physical and chemical analysis and for IR and XRD analysis. Most of the surface (0-6 m) samples were dried at 60 °C prior to sub-sampling. A total of 92 core sections were variously examined, and we still await results of XRD and have just begun to process the chemical/physical data. The IR data will be used to predict much of the chemical, physical and mineralogical data for possible future studies.

For chemical and physical analyses (CSIRO Land and Water Analytical Chemistry Unit), each sample was ground, separated to determine the >2 mm content and oven dried. The < 2 mm fraction was used for the appropriate tests.

Analysis
All sections of each core were examined for: electrical conductivity (EC), pH (1:5 soil:water as well as with 0.01 CaCl₂), organic C, CO₃ as free CaCO₃, textural and coarse fragments. Selected core sections were examined for: exchangeable cations (Ca, Mg, Na, K), cation exchange capacity (CEC) performed using NH₄, exchangeable Na percentage (ESP), sodium absorption ratio (SAR), soluble salts (Ca, Mg, Na, K, S). All analyses were conducted on Core 17939 to serve as a data analysis training set using IR.

For XRD, samples were pre-ground for 15 seconds in a tungsten carbide mechanical mill to pass through a 0.5mm sieve. Two 1g sub-samples were further ground for 10 minutes in a McCrone micronizing mill under ethanol. The resulting slurries were oven dried at 60°C then thoroughly mixed in an agate mortar and pestle before being lightly pressed into aluminium sample holders for X-ray diffraction analysis.

Results
Qualitatively IR and XRD analyses indicate significant carbonates at depth (~20 m). Substantial smectitic clays occurred between 2 to 22 m in Cores 17937, 17938 and 17939. We still await the results of quantitative XRD analysis so that correlations can be made with IR.

Coarse fragments (>2 mm) cores ranged from <1% (non gravelly) to as high as 73% (very gravelly). Textural analyses on <2 mm fractions gave average compositions (across cores and with depth) of coarse gravelly sandy loam: 70% sand, 9% silt and 21% clay, although considerable variability existed (e.g. range in clay was 1-58%). Textural analysis with depth for Core 17939 (Figure B-1) was very typical (71% sand, 9% silt, 20% clay). With the exception of the 20-9-21m section, clay contents were ≥20% between 4 to 24.6 m for Core 17939, with as much as 45% clay at 18-18.1 m depth.
In all core sections studied Ca$^{2+}$ dominated the exchange complex: for Core 17939, Ca$^{2+}$ occupied ~50-80% of the complex (Figure B-2a). In contrast, Na$^+$ dominated the soluble salt cation fraction (Figure B-2b). Interestingly, exchangeable Ca$^{2+}$ tended to be highest when soluble Na$^+$ and S were high. Sulfur tended to follow Na$^+$ trends throughout the profile of Core 17939 (Figure B-2b).

Generally the CEC measured by NH$_4^+$ exchange was equal to the sum of the exchange cations measured. The exceptions were the near-surface sections (0-6m) indicating that some weathering has occurred in the upper profile. It is possible that exchangeable Al$^{3+}$, oxyhydroxides of iron or kaolin minerals are contributing to this difference, although the pH values (see below) indicate that weathering is not severe. CEC values ranged from very low to moderately high, indicating a range in mineralogy consistent with the particle size estimations: CEC values were generally highest for those samples with less than ~55% sand. A CEC bulge is evident in Core 17939 (Figure B-2b) and is indicative of a weathered upper profile.

As expected, the sodium absorption ratio (SAR, 1:5 soil:water) (Figure B-2c) and exchangeable sodium percentage (ESP) (Figure B-2d) were generally elevated with soluble Na$^+$: all measures showed two concentration bulges between 5-6m and 14.7-14.8m and 14.7-14.8m and 20-20.15m depth in Core 17939. The electrical conductivity (1:5 soil:water) values (Figure B-2d) for Core 17939 indicate slight to moderate salinity. However, ESP values are high (>6 %) throughout the upper 24m of Core 17939 (Figure B-2d). Thus, in sections where clay content exceeds ~20% inherent sodicity may impact drainage characteristics.

Soil reaction (pH) was moderately invariable with depth, but in general pH values between 8.0 and 8.5 are consistent with a matrix dominated by calcium carbonate, pH values near neutral consistent with gypsum and pH values >8.5 with bicarbonate (Figure B-2c). In Core 17939, some sections yielded 1:5 soil:water reactions consistent with a matrix dominated by bicarbonate, but these values followed an opposing trend to soluble Na$^+$ (i.e., when soluble Na$^+$ was high, the 1:5 pH was closer to neutral). Available data indicates that S is relatively high when near neutral CaCl$_2$ reaction. This may suggest that could gypsum play an important role in buffering pH and alleviating the effects of sodicity in certain sections. Carbonate (as free CaCO$_3$) was low (generally < 1-2%), but was substantial (as high as 30%) in lower sections of core 17939 (Figure B-2e).

Oxalate and dithionite extractable Al$^{3+}$ (Figure B-2f) were relatively constant throughout (80-570 and 130-670 ppm, respectively in all sections studied). Very low levels (<100 ppm) of oxalate extractable Mn$^{2+}$ were observed to be generally consistent with high levels (> 100ppm) of soluble S. In core 17939, oxalate and dithionite extractable Mn$^{2+}$ were 2 to 4 times higher at 15.5-15.6m, but the available data do not indicate an association with any other chemical signature. Dithionite extractable Fe$^{3+}$ was high, ranging from 8-12% for much of core 17939, indicating a high level of iron oxyhydroxides in these sediments, which is consistent with deep weathering.
Figure B-1  Textural analysis of Core 17939.

Core 17939

1 = 1-2m   11 = 16-16.1m
2 = 2-3m   12 = 17-17.1m
3 = 3-4m   13 = 18-18.1m
4 = 4-5m   14 = 19-19.1m
5 = 5-6m   15 = 19.6-19.7m
6 = 10.5-10.6m  16 = 20-20.15m
7 = 11.9-12m  17 = 20.9-21m
8 = 13.3-13.4m  18 = 22-22.1m
9 = 14.7-14.8m  19 = 24.5-24.6m
10 = 15.5-15.6m  20 = 24.9-25m
Figure B-2  Physical and chemical data for Core 17939.  a) exchangeable cations and CEC; b) soluble cations and S; c) pH and SAR; d) CO3, organic and total C and ESP; e) EC; f) oxalate and dithionite extractable Al, Mn and Fe.
APPENDIX C – SAT PILOT SCHEME IMPLEMENTATION WORKS

Soil Aquifer Treatment Pilot Scheme

Tasks involved with the implementation of the Pilot SAT Scheme include:

- Basins construction.
- 2 x 200 metre pipeline capable of delivering approximately 70 l/s.
- Installation of monitoring equipment, 8 multi level piezometers, shallow piezometers, lysimeters, tensiometers and water content meters to 6 metres.
- Water quality analysis
- Groundwater modelling maintenance

Basin Construction and Pipeline

The construction of the SAT basins will involve the excavation of two 7,500 m² basins and the delivery pipeline to the basins. Delivery system should be capable of delivering a peak of 70 l/s.

Monitoring Program

Lysimeters, tensiometers, water content meters and shallow piezometers to monitor the unsaturated zone below the basins 10 sets of each instrument at four basins (2 upstream and 2 downstream) to a depth of 6 metres.

Shallow piezometers (<30m depth) around the basins to monitor lateral spreading, mounding and water quality:

- 4 multi-level piezometers to monitor the mounding response below each of the elongated basins.
- 4 multi-level piezometers approximately 50 – 100 metres up and down gradient of the basins to provide breakthrough times of applied recycled water.
- 10 days drilling approximately

Water Quality Analysis

Water quality in both the basins and the monitoring bores needs to be analysed frequently approximately once a fortnight, if we assume a yearly program then:

- The suite of physical, chemical and bacterial analyses given in Table D-2 should be taken for the 8 monitoring bores initially + infiltrate every 2 months.

Groundwater Modelling

The increased sampling data from the Pilot Plant will provide continuous verification/calibration of the groundwater model especially with respect to the solute transport component. The maintenance of the model should be completed every 6 months. Collation of data and preparation for comparison with modelling results will take approximately 2 weeks and modelling a further 3-4 weeks.

Staff Resources

Monitoring of the trial infiltration basins has involved approximately 60% of a level 2 professionals time, it is therefore suggested that the following staffing be available for a period of 12 months during the Pilot Plant implementation and commissioning of the additional treatment processes at the waste water treatment plant:
Professional Level 2 to implement basins, monitoring program, collate data, maintain groundwater model

Technical Assistant Level 2 to provide monitoring support for water levels, water quality samples and general collation of data.
APPENDIX D – STAGE II MODELLING AND COLUMN STUDIES & FURTHER SITE INVESTIGATION WORKS

Stage II SAT Pilot Scheme

Groundwater Modelling
The Stage 1 modelling component has identified that further work is required to provide a groundwater model suitable for solute transport simulations. It is expected that a further 6 weeks will be required to develop a model as per the recommendations outlined in section 4.

Study of water quality improvements during passage through soil columns
This work builds upon the previous laboratory work that focussed on clogging processes in the top 10 cm of the soil profile. It will provide data on water quality improvements due to passage through the top one metre of the soil profile with an emphasis on demonstrating the removal of microbial pathogens, nutrients and organic matter. The work will involve the following:

1. Experimental Design
   Up to four columns will be constructed and run simultaneously. One or perhaps two types of water quality could be tested. For instance, at the current level of treatment (ie. the water being used to irrigate Blatherskite Park), and/or treated to the level expected to be supplied to the SAT Pilot Plant. Because of the slow acclimation times for microbial growth the experiment should proceed for at least 6 months (approximately 13 weekly wetting/drying cycles).

2. Water quality analytes and sampling frequency
   The analytes to be sampled in the inflow and outflow of each column are listed in Table D-1. Samples will be submitted to AWQC for analysis twice a week during the beginning and end of each wetting phase. The total number of samples over a 6 month period is 156, in-situ measurements will be made on a more frequent basis.
Table D-1 Parameters routinely measured on the inflow and outflow from each soil column at CSIRO and the Australian Water Quality Centre (AWQC)

<table>
<thead>
<tr>
<th>Class</th>
<th>Analyte</th>
</tr>
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<tbody>
<tr>
<td>In-situ</td>
<td>EC</td>
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<tr>
<td></td>
<td>pH</td>
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<tr>
<td></td>
<td>Dissolved Oxygen</td>
</tr>
<tr>
<td></td>
<td>Redox Potential</td>
</tr>
<tr>
<td></td>
<td>Temperature</td>
</tr>
<tr>
<td>Anions &amp; cations</td>
<td>Calcium</td>
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<tr>
<td></td>
<td>Magnesium</td>
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<td></td>
<td>Potassium</td>
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<tr>
<td></td>
<td>Sodium</td>
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<tr>
<td></td>
<td>Sulphate</td>
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<td></td>
<td>Bicarbonate</td>
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<td></td>
<td>Chloride</td>
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<td></td>
<td>Bromide</td>
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<td></td>
<td>Boron</td>
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<tr>
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<tr>
<td></td>
<td>Manganese – Total</td>
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<tr>
<td>Nutrients</td>
<td>Nitrate + nitrite - N</td>
</tr>
<tr>
<td></td>
<td>Ammonia - N</td>
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<tr>
<td></td>
<td>TKN</td>
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<td></td>
<td>Soluble - P</td>
</tr>
<tr>
<td></td>
<td>Total - P</td>
</tr>
<tr>
<td>Microbiological</td>
<td>E. Coli</td>
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<tr>
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<td>Total Organic Carbon (TOC)</td>
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<tr>
<td></td>
<td>Dissolved Organic Carbon (DOC)</td>
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<td>Turbidity</td>
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<td>Derived Data</td>
<td>Total Dissolved Solids (TDS)</td>
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<tr>
<td></td>
<td>Alkalinity as CaCO₃</td>
</tr>
<tr>
<td></td>
<td>Sodium adsorption ratio (SAR)</td>
</tr>
</tbody>
</table>

3. Pathogen survival/transport study
Details to be completed when available.
Further Site Investigation Works

Geochemistry/isotope investigations across the Outer Farm area
Work is required to identify and map the relative contributions of the potential sources of groundwater across the Outer Farm area using geochemical and isotopic techniques given that there are at least four potential sources of water in the area: natural (diffuse) recharge, episodic Todd River recharge, seepage from effluent via St Mary’s Ck and Blatherskite Park as well as Bitter Springs and/or Inner Farm Basin inputs. This would lead to: 1) an improved understanding of the impacts of the long-term effluent disposal on groundwater quality, 2) assist to characterise the groundwater flow systems which will add value to the groundwater modelling, and 3) generate baseline information for the SAT investigations. This study compliments the investigations proposed around St Mary’s Creek (see below).

St Mary’s Creek water quality investigation:
It is proposed that 7 investigation bores be constructed within the Study Area. The objective of these holes is to obtain hydraulic and water quality parameters that will provide an understanding of the hydraulic processes that have occurred where water discharged from Ilparpa Swamp has infiltrated through the bed of Saint Mary’s Creek into the palaeochannel aquifer. The understanding gained from the work will provide a valuable input into understanding the processes that will occur underneath the proposed SAT ponds.

Production bore “proofing”
In section 2 it was identified that the siting of production bores capable of providing relatively high yields would require some investigation work and an improved methodology in production bore design.

Palaeochannel delineation Airport Block North
In section 2 it was determined that the delineation of the palaeochannel to the southeast needs to be improved. 10 investigation bores completed with 100 mm PVC to a depth of 40-50 metres have been proposed in the northern portion of the Airport Block to alleviate this data deficiency.