ON-FARM AND COMMUNITY-SCALE SALT DISPOSAL BASINS ON THE RIVERINE PLAIN

BASIN LEAKAGE: SITE STUDIES AT GIRGARRE, VICTORIA AND GRIFFITH, NSW

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Foreword

There are pressures to minimise salt leaving irrigated catchments of the Murray-Darling Basin to limit salinity increases in the River Murray. Part of this strategy is to manage drainage disposal water in the irrigation areas using disposal basins. Unfortunately, there are no existing guidelines for siting, design and management of such disposal basins. The CRC for Catchment Hydrology and CSIRO Land and Water, with support from the Murray-Darling Basin Commission, have embarked on a project with the overall objective of producing such guidelines for the Riverine Plain of the Murray Basin.

This report is one of several being produced in this project to support the guidelines. It describes the results of intensive field investigations at a 30 ha community basin near Girgarre in the Shepparton Irrigation Region (SIR), and at a 2 ha on-farm basin near Griffith in the Murrumbidgee Irrigation Area (MIA). It also includes results from less intensive investigations at a further 13 on-farm basins in the MIA. The primary focus of these field investigations concerned leakage from the basins, believed to be a major factor in determining the overall viability of disposal basins as a repository for saline drainage in the Riverine Plain.

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Summary

Two evaporation basins have been intensively studied for two years to determine the rate and fate of leakage from them. The two basins: Girgarre (30 ha) and Nehme (2 ha) are very different. The Girgarre basin, commissioned in 1987, receives water from groundwater pumps in the area surrounding the basin and has unsaturated soil beneath the basin. As such, we describe leakage from the basin as *infiltration controlled* because the throttle to leakage is close to the bottom of the basin. Girgarre basin was established as a demonstration basin and the basin water, and groundwater salinity, has been monitored for most of the last 13 years. In this report, we summarise results from earlier studies on the basin and present new results from more recent field investigations.

The Nehme basin was established in 1997 as part of the field investigations for this project. The basin receives drainage water from a subsurface pipe 'tile' drain system. Initially there was a large unsaturated zone under the basin but, soon after commissioning, this became fully saturated. We describe leakage from this type of basin as *expansion limited* because the throttle to leakage is determined by how fast leakage can move laterally and vertically away from the basin.

As part of investigations of on-farm basins, a further 13 basins in the MIA were monitored on a monthly basis, in order to provide some general information on the functioning of other on-farm basins.

Leakage from the Girgarre basin was found to be quite stable at 0.5-1.5 mm/d over its life to date. Leakage from the Nehme basin was initially high (5-9 mm/d) and reduced as the unsaturated zone filled, stabilising at about 3 mm/d. The other MIA basins had leakage from 3.5 – 5.4 mm/d. Thus it would appear that the small on-farm basins have much higher leakage rates than the large Girgarre basin.

Investigations of flows in interception drains around the on-farm basins found that they accounted for 25 – 50% of the total leakage estimates. This would indicate that interception drains in small basins recycle a large amount of water.

Using natural tracers, it was found that the saline leakage plume from the Nehme basin had reached 5 m below the basin and up to 20 m outside the basin in the shallow groundwater. At the Girgarre basin, lateral leakage appeared to be the same order of magnitude, but there may be preferential vertical leakage paths that is moving the leakage plume to deeper shoestring sands.

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1. Introduction

The medium to long term viability of large irrigated areas in the Riverine Plain region of Victoria and New South Wales is closely linked with management of high water tables. Restrictions imposed by the Murray-Darling Basin Salinity and Drainage Strategy have led to reductions in the export of salt from the area from which it originated. These restrictions have resulted in an increase in the number of small-scale on-farm and community disposal basins in irrigation areas. The existing design and management of both types of basin vary widely as they have been developed under different administrative frameworks.

CSIRO Land and Water, in collaboration with the CRC for Catchment Hydrology, the Murray Darling Basin Commission (Strategic Investigation and Education Program, Project I7034 *Managing Disposal Basins for Salt Storage Within Irrigation Areas*) and other agencies have been investigating the siting, design and management conditions under which small-scale basins can be successfully used by individuals or groups of landowners. The biophysical and other technical information obtained in this project have been used to define a robust set of guiding "principles" and guidelines for responsible basin use.

This report

The disposal of saline irrigation drainage water is a problem that does not have an easy solution. Currently, disposal is mainly via drainage channels that empty eventually into existing regional basins or, more often, the drainage water eventually finishes up in streams or rivers. There is a growing expectation that this practice will be reduced in the near future and that the drainage water will be disposed at or near the irrigation site. If this is to happen, it is probable that there will be an increase in the use of on-farm and community basins in the Riverine Plain. These lead to potential risks such as groundwater salinisation and soil salinisation of nearby areas. The field investigations for this project were initiated in order to evaluate the functioning of a new basin and of an established basin in the Riverine Plain in order to add to the limited data base for basins in the Riverine Plain (Figure 1).

Leakage is one of the major factors that will determine whether or not disposal basins are a viable option for the safe disposal of saline water from irrigation areas. In this report, we concentrate on the process of leakage from disposal basins, both in terms of estimating the rate of leakage and with regard to the leakage plume (where the leakage water is going). We also measure factors that are likely to affect leakage from the disposal basins. Results from these field investigations have been combined with those from previous studies on disposal basins in the Riverine Plain to produce a companion report on the operation of disposal basins in this area (Leaney and Christen, 2000). This forms an important basis for the production of a set of guidelines for the use of on-farm and community-scale disposal basins in the Riverine Plain (Christen *et al.*, 2000).

Figure 1.   The Murray Basin showing the location of existing on-farm and community basins studied in the Riverine Plain. The Girgarre Community Basin is numbered 1 and 2; basins at Griffith (including Nehme basin) are numbered 7 (numerals refer to existing reports on the basins, see Leaney and Christen,2000).

This report presents the results of field investigations conducted for:

1. Girgarre basin, a 30 ha community basin located near Girgarre in the Shepparton Irrigation Region (SIR).

2. Nehme basin, a newly constructed 2 ha on-farm basin located near Griffith in the Murrumbidgee Irrigation Area (MIA).

3. Several other on-farm basins in the MIA.

2. Methodology

The key factor that governs the operation of a disposal basin is water loss by evaporation and leakage beneath the basin. In this report, we concentrate primarily on the leakage component of water from the disposal basins. More specifically, we evaluate different methods for estimating the rate of leakage from the basins, where the leakage is likely to go (i.e. the configuration of the leakage plume), and the soil physical and biological factors that may affect the rate and distribution of leakage from the basin.

In estimating the rate of leakage rate from the study basins we have used:

1. A water balance approach for the Nehme basin. Leakage from the Girgarre basin has also been estimated previously using water balance calculations (Goulburn-Murray Water and Sinclair Knight Merz, 1995).

2. A chloride balance for the Girgarre basin and salt balance at Nehme basin

3. Tracing the rate of movement of saline leakage water beneath the basin (using naturally occurring [Cl], deuterium, ^2H or oxygen-18, ^{18}O composition in the soilwater or groundwater).

4. Seepage meters.

2.1
Whole of Basin Estimates of Leakage

The most common method used to estimate leakage is by simple water balance; this method has been used for several evaporation basins in the Riverine Plain. The use of salt balance estimates is less common and, in fact, only applied to one other basin apart from those in this study (Ranfurly, see Leaney and Christen 2000). A salt balance can be done either with total salts or, preferably, a specific salt such as chloride.

2.1.1 Water balance

Estimation of leakage using the water balance of a basin involves estimating input to the basin by measuring the amount of water pumped into the basin and that entering as rainfall. This is balanced with output from the basin (evaporation and leakage) and with the change in basin volume.

Evaporation is estimated from pan evaporation, either measured on site or at the nearest suitably equipped meteorological station. Conversion of pan evaporation data to that for the basin involves calculating a pan factor. Several approaches are available to estimate the pan factor depending on the availability of data such as humidity, water temperature and wind speed. In general, the average pan factor reduces from near unity for very small basins (< 2 ha) to ~0.8 for large regional basins [Morton, 1986; see Leaney and Christen (2000) for summary].

Leakage is estimated using the water balance equation below. It assumes that any difference between water input to the basin and evaporation is leakage. This method is error prone, as inaccuracy in any of the measurements compounds into in the leakage value.

Leakage is determined for any time interval using the equation

$$L = (I + P) - (E + \quad V) \qquad\qquad (1)$$

where L is leakage, P is precipitation, I is input via drainage pumped into the basin, E is evaporation and V is change in storage.

Water balance calculations (as described above) have been used for estimating leakage from several evaporation basins in the Riverine Plain. A summary of these studies is presented in a companion report on the operation of basins in the Riverine Plain (Leaney and Christen, 2000).

The major difficulty in using the water balance as the only method of measuring leakage is that an error in any of the parameters will result in an error of equal magnitude in the leakage estimate. The inclusion of salinity or chloride data by means of a salt or chloride balance provides a semi-independent check on whether the data used in the water balance is likely to be correct. This is because it is possible to formulate the water and salt or chloride balances as two equations, and hence treat leakage and another water balance parameter as variables.

Chloride concentration [Cl] is normally used because it is conservative and non-reactive, and hence considered to move at the same rate as the leakage water. Providing water in the basin is not hyper-saline, [Cl] increases linearly as evaporation progresses (i.e. if half the water in the basin evaporates then [Cl] doubles). Historically, however, electrical conductivity measurements for water samples (EC) are more often measured than [Cl]. Salt balances from conductivity sampling are useful at low water salinities where salt precipitation is unlikely. This is the case for much of the historical samples from the Girgarre basin. Hence, we have converted salinity data to [Cl] using an empirical relationship developed for water samples collected from the Girgarre area. The outline below for a chloride mass balance applies equally for a salt mass balance.

When using a chloride mass balance to confirm leakage estimates, we assume that the chloride entering a disposal basin as pumping input or in rainwater must be either present in the basin water and surface sediment, or has been lost by leakage. This is generally true because there is negligible loss of [Cl] by aerial removal (deflation) especially if the basin remains filled. The solubility of most common chloride salts is very high (solubility of NaCl is ~357 g/L); hence, using a chloride mass balance rather than salt mass balance eliminates estimating the amount of chloride present as a precipitate under all but extremely high salt concentrations.

2.2
Salt or Chloride Mass Balance

In order to use this method, temporal [Cl] needs to be measured for the disposal basin, precipitation, and inflow to the basin (i.e. $[Cl]_{basin}$, $[Cl]_{prec}$ and $[Cl]_{input}$). For any time step, the equation relating to the water balance must hold true (Equation 1) as well as that relating to the [Cl] balance. The change in the amount of chloride in the basin ($V [Cl]_{basin}$) must equal that input from inflow and rain less that removed by leakage (Equation 2).

$$(V[Cl]_{basin}) = P[Cl]_{prec} + I[Cl]_{input} - L [Cl]_{basin} \qquad (2)$$

Note that the evaporation term disappears in Equation 2 because no chloride is removed during evaporation. Using Equations 1 and 2, it is possible to calculate the chloride concentration in the basin after a time interval, t, using the components for the water balance and their respective concentration during that time interval and hence, plot the temporal change in $[Cl]_{basin}$ for different leakage rates. By changing the leakage parameter, it is possible to generate a temporal plot for $[Cl]_{basin}$ that most closely approximates the observed data for a basin.

The description so far has been for the case of a single bay. An analogous approach can be used for the case of several bays in series, with the final bay being terminal (i.e. no outflow). This is the case for the basins at Girgarre, Nehme and numerous other basins in the Riverine Plain. A generic spreadsheet has been developed for this purpose. The required input for the model is temporal data for input to the basin (and $[Cl]_{basin}$), precipitation (and $[Cl]_{prec}$), evaporation and the amount of water in each bay (often related linearly with depth). The model requires that the basin does not empty.

2.3 Point Measurements of Leakage

The above methods determine leakage from the basin as a whole. Other methods have been used to determine the leakage rate at a particular location in the basin. These "point estimates" include tracer techniques to identify how far leakage has moved from the basin, and water loss from a small area in the basin using leakage meters (conventional seepage meters, and those similar to what is commonly referred to as the "Idaho leakage meter").

2.3.1 Natural tracers

The best natural tracers to use are those that move at the same speed as the water (i.e. are not retarded), are conservative, and easily identified from water in the surrounding soil or groundwater matrix. Because the chloride ion is conservative, and usually only present in low concentrations in soils (except as halite in discharge areas), it satisfies these requirements. For the same reasons, the stable isotopes of water (deuterium, ^2H and oxygen-18, ^{18}O) are also excellent tracers of water movement. When water is ponding, and thus evaporating (as in a disposal basin), the water which remains in the basin will be highly concentrated in most salts including chloride and enriched in ^2H and ^{18}O. The chloride, ^2H and ^{18}O concentration of the water leaking from the basin will be the same as the concentration of these elements in the basin.

One method to determine the chloride concentration in the leakage water is to sample soil beneath the basin, and determine the soilwater chloride concentration (or stable isotope composition) in the soilwater at discrete depths below the basin floor. The concentration of these elements extracted from the soil should be the same as those measured in the groundwater at the same depth, if the soil is saturated. From the profile obtained by plotting the soil water chloride concentration (or stable isotope composition) in the soil water as a function of depth, it is possible to recognise how far the water from the evaporation basin has travelled in the unsaturated zone or in the saturated zone.

If we assume piston flow occurs, the amount of water in the soil and groundwater to the depth where chloride concentrations change from high (representing leakage) to low (representing original water) is approximately the amount of water that has leaked since the basin was commissioned. The estimate may not be correct if water table fluctuations cause the soil-water to disperse or in situations when there is a throttle to vertical flow. Under these conditions, the components to the direction of flow (vertical or horizontal) will depend on the horizontal conductivity and gradient in each direction and may result in a large component of the leakage water moving laterally.

2.3.2 Seepage meters

There are numerous designs for seepage meters, but most consist of a solid tube tapped into the soil at the base of the basin. For the most basic meter, the tube extends out of the water and is almost sealed to minimise evaporation (except for a small airhole to equalise air pressure inside and outside of the tube). The water level inside the tube is maintained at the same level as that in the basin by routinely adding water to the tube, ensuring that the head of water in the basin and that in the tube remain as close as possible. The leakage rate is estimated by measuring the amount of water that is required to maintain the water level in the tube. This type of meter was used at Nehme basin.

If measurements are required over a longer timeframe, and/or routine measurement is difficult, it is possible to attach a bladder filled with water to the tube. This is similar to the system used by McBride and Pfannkuch (1975) except, in their field studies, they were attempting to measure groundwater inflow into the basin. The aim of this arrangement is to ensure that the water in the tube is kept at the same pressure as the water in the basin, to ensure there is no additional head of water increasing leakage from the tube beyond that of the basin. This type of system was used at the Girgarre basin. Evaporation from the tube is considered negligible for both types of leakage meter. The amount of water lost from the tube (at Nehme basin), or by the bladder (at the Girgarre basin), for any time period is equal to leakage over the area of the pipe.

The greatest difficulty with using leakage meters is ensuring that the tube seals well with the soil without compacting it. There are other technical problems such as blockages or puncturing of the bladders. Using point source estimates of leakage allows an evaluation of the spatial variability in leakage but there are rarely enough measurements to get an overall estimate of leakage from the basin.

2.4 Movement of Leakage From the Basin

Leakage of saline water from disposal basins has the potential to cause problems either to the land surrounding the basin, or to shallow or deep groundwater systems near to the basin. The only certain way that this potential problem can be overcome is by the use of an impervious barrier to prevent leakage, and to ensure that the integrity of the barrier is maintained for the life of the basin. Such barriers are not recommended for disposal basins (Leaney and Christen, 2000), primarily because of the loss in evaporative potential as the water in the basin becomes increasingly saline; therefore, much larger area of land that needs to be sacrificed for the basin.

Given that, in most situations, some leakage from basins is required (and is going to occur), it is essential that we develop some knowledge on where leakage from the basin is likely to go (i.e. on the process of leakage from the basin). For the basins studied in this report, we have looked for evidence of leakage from the basins:

1. by analysing samples of groundwater from piezometers around the basin for salinity, [Cl] and ^{2}H and ^{18}O composition (direct analysis).

2. by analysing samples of soilwater beneath the basin for salinity, [Cl] and ^{2}H and ^{18}O composition (direct analysis).

3. by analysing water in the interception drain at Nehme for salinity, [Cl] and ^{2}H and ^{18}O composition (direct analysis).

4. Measuring the apparent resistivity of groundwater using a downhole EM39 in piezometers near to the Girgarre basin (indirect measurement) and surface EM38 technique at Nehme and other MIA basins (indirect measurement).

All samples and resistivity measurements were made during sampling/monitoring from the time of commissioning the basin until the time of writing the report.

2.4.1 Direct measurements (salinity, [Cl], ^{2}H and ^{18}O)

Measurements of salinity, [Cl], ^{2}H and ^{18}O all provide an indicator of whether or not the leakage from the basin has reached the point at which the sample was collected. However, the concentration of the tracer in the leakage water must be significantly different from that in the groundwater or soilwater prior to the time when the leakage reached it and also different

from any other possible sources of saline water. This is not always the case in the situation where groundwater pumping is used as this may result in vertical or lateral movement of saline groundwater adjacent to the sampling point. Sampling can be undertaken from piezometers, soil samples or drain flows in interception drains:

- When piezometers are used, the sampling point is the depth of the screen at the site of the piezometer (i.e. a single point defined by depth and direction from the basin). Nested piezometers (or using open hole and pump-packer sampling if practical) will provide data for several depths. If samples are collected over a time interval, it is possible to determine when the leakage reaches that point. Collection of samples from the piezometer should be such that the piezometer is pumped for long enough to ensure the sample is representative at the sampling point, but not to induce excessive vertical or horizontal flow to the point of sampling.

- When soil is sampled, a range of samples may be collected over the depth interval sampled but only for a single point in time. Both soil sampling and sampling water in piezometers allow reasonable estimates of the rate of leakage to the points sampled. However, the major difficulty is determining how representative leakage to those points is for the whole of the basin.

- Sampling and analysing water from the interception drain provides a method for integrating leakage over the area in which the drain impacts. As the main aim of such drains is to minimise lateral seepage from the basin, this method provides information on shallow lateral movement from the basin. Interception drains were present at both the Girgarre basin and Nehme basin but only the drain at Nehme had sufficient flow to provide samples for analysis.

2.4.2 Indirect measurements

When using the direct methods described above, samples could be collected, analysed and the results interpreted to indicate where the leakage from the basin is moving. The disadvantage with these methods is that the results relate to where the samples were collected. Often, in an attempt to extend the spatial scale of measurement, researchers use indirect methods of measurement in which a more easily measured parameter is measured and related to the parameter of interest. In the case of leakage from a basin, geophysical techniques measure apparent conductivity over a large area and attempt to relate that measurement to the conductivity of the soilwater.

Geophysical techniques can, to a certain extent, be quantitative provided adequate calibration/ground truthing is available (Cook *et al.*, 1989). This, however, is not the case for the basins studied here. We use the geophysical data as qualitative indicators of spatial or temporal changes in groundwater

salinity and hence leakage (Goulburn-Murray Water and Sinclair Knight Merz, 1995). When using downhole EM39 data for bores near the Girgarre basin, we are able to look at temporal and spatial differences at different sites around the basin. When using the EM38 data from in and around the on-farm basins, the results were analysed to determine if there were large variations in apparent conductivity in the basin floor, possibly indicating varying leakage rates. Measurements outside the basins were compared to those inside by using transect sampling to assess how far saline water leaking from the basin may have moved laterally. Note that EM38 data only relates to the top 2 m of soil and thus would only reflect shallow lateral leakage.

By combining the results of direct and indirect measurements, we present summaries of the development of the salinity plumes at each of the sites studied. This data provides important background information on the process of leakage for basins in the Riverine Plain as presented in Leaney and Christen (2000) and Jolly *et al.* (2000).

2.5 Soil Physical and Biological Factors

The first two parts of the methodology provided ways of estimating leakage rates for disposal basins and for determining where the leakage was going. The final part of the methodology attempts to provide information on factors likely to determine both the rate of leakage and the leakage plume for disposal basins.

A wide spectrum of factors are likely to affect the rate of leakage from disposal basins. Leakage may be controlled by factors that impact at the base or sides of the basin, and throttle flow from the basin to the underlying unsaturated zone. The throttle-to-leakage may be at some depth below the basin, but still at an elevation above the regional groundwater table. The rate of leakage may be determined by the rate at which the groundwater mound under a disposal basin is able to dissipate. It is likely that, for any basin, different factors may control leakage at different stages of the basin life.

The following factors have been identified in the literature as potentially important in determining leakage from disposal basins:

1. Soil type and soil compaction (Heavier soils or compacted soils will, in general, have lower rates of leakage)

2. Hydraulic head of water (Deeper basins and/or lower regional water tables may enhance leakage).

3. Soil and water sodicity (Chemical interaction of water and soil may lead to dispersion or flocculation of the soil and changed infiltration rates).

4. Algal clogging (Polysaccharide production as a result of microbiological activity may reduce leakage from the basins).

5. Presence of preferential flow paths (Leakage may be enhanced if leakage from the basin proceeds via preferential rather than piston-type flow).

As part of the field investigations in this study, we report the results for the following parameters:

1. Clay content of soil beneath the basin.

2. Soilwater potential measurements (as an indicator of saturated or unsaturated conditions).

3. Soil sodicity.

4. Polysaccharide concentration in the soil.

5. EM39 (Girgarre) and EM38 (Nehme) profiles as indicators of spatial variation in leakage and areas of preferential flow.

Interpretation of these results for these, and other existing basins in the Riverine Plain, is included in a companion report (Leaney and Christen, 2000).

3. Girgarre Basin (Site and Study Description)

3.1 Hydrogeology

Irrigation, predominantly by flood irrigation, commenced in the Shepparton Irrigation Region (SIR) about 80 years ago providing water mainly for perennial pasture (65%) and annual pasture (25%). As is the case with most irrigation areas in the Riverine Plain, rising water tables and shallow groundwater salinity has been noted during the last 2-3 decades.

The Girgarre Evaporation Basin is located in the SIR, about 50 km west of Shepparton, Victoria. A comprehensive irrigation history of the area and the underlying reasons for the development of the basin have been presented in a report by Goulburn-Murray Water in association with Sinclair Knight Merz (Goulburn-Murray Water and Sinclair Knight Merz, 1995).

The Girgarre basin is underlain by fluviatile sediments of the Shepparton Formation which extend to a depth of about 70 m below the natural surface. The Formation is predominantly clayey but comprises several sandy units within the surficial 30 m (referred to as the shoestring sand aquifers in the Upper Shepparton Formation). The shoestring sand aquifers are divided into two aquifer groups. Group 1 aquifers are located in the top 12 m and may be up to 10 m thick with groundwater salinity typically 2 to 4 dS/m. The second aquifer group (designated Group 2), formed from prior streams, is found between 10 and 20 m from the surface; it has salinities ranging typically from 15 to 20 dS/m. It is present in 50% of the area and occasionally underlies the Group 1 system.

Between 6 and 9 wellpoints were installed at each of three sites (T101, 102 and 103) around the basin. The screens for all wellpoints are located in the shoestring sand aquifers at depths from 7-20 m below ground surface. Groundwater from T101 is the most saline (~15-19 dS/m; [Cl] ~5,800-7,000 mg/L) and is pumped into the basin. Groundwater from T102 (salinity ~3-5 dS/m) and T103 (salinity ~3.8-6 dS/m) are considerably fresher and are pumped into drainage channels. Numerous private pumps also pump groundwater into the drainage channels to lower groundwater on their farms.

3.2 Basin History and Construction

The disposal basin was constructed in 1987, occupying an area of 42 ha in one of the most badly salt affected parts of the irrigation area. The basin itself is divided into 3 bays totalling 30 ha and the remaining 12 ha used as a buffer zone from surrounding irrigation, Figure 2. Groundwater from T101 is pumped into Bay A (13 ha), flows by gravity feed into Bay B (13 ha) and

then into the terminal Bay C (4 ha). Flow to the basin is triggered when water levels fall by 0.1 m. Mean depths for Bays A and B have been maintained at ~0.3 m, occasionally rising to ~0.5 m and on one occasion (late 1993) being empty for 2-4 weeks. The base of Bay C is ~0.1 m below the other 2 bays resulting in it being, on average, 0.1 m deeper. Bay A had no treatment following construction while, in Bays B and C, the soil was compacted to see whether this would help reduce leakage.

An interception channel is located ~19 m from the northern and southern sides of the basin, and ~16 m from the eastern side; the Deacon drain is located ~40 m from the western side. The base of the interception drain is ~0.5 -1.0 m below the base of the basin. Deep (16-19 m) and shallow (4 m) monitoring bores were drilled around the basin (called perimeter bores) and between Bays A and B and Bays A and C (called internal bores) (Figure 3).
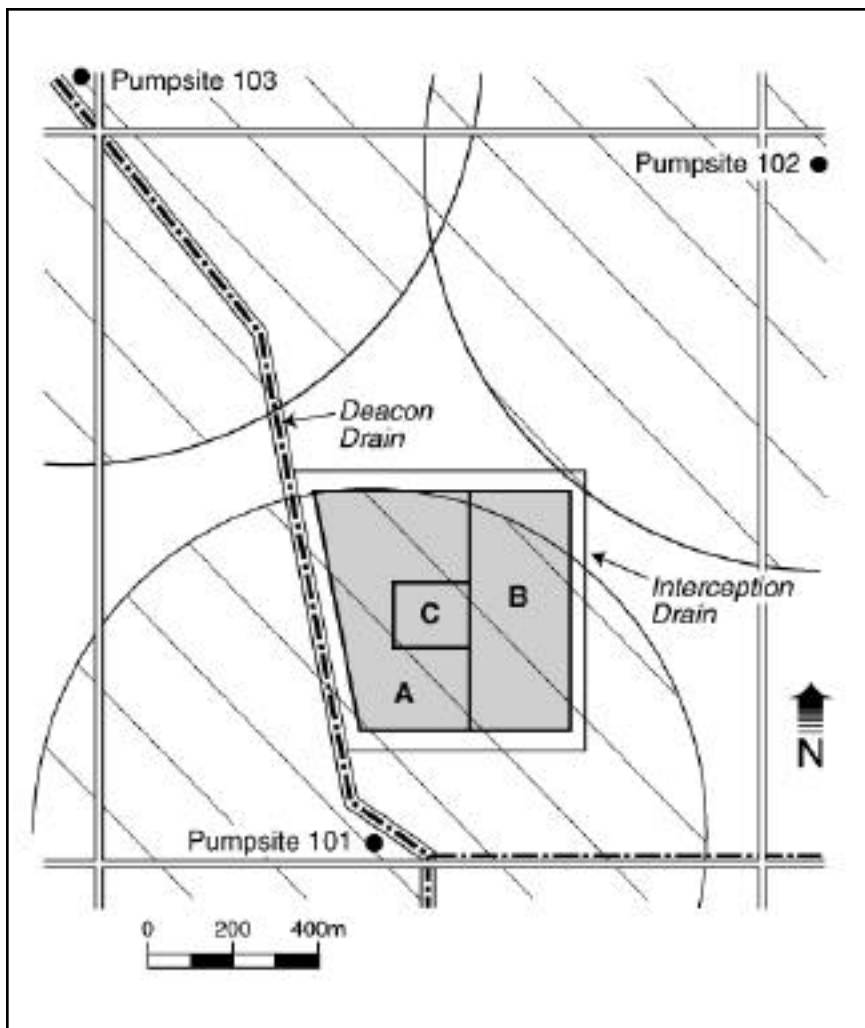
Figure 2.   Girgarre Basin site map and location of the surrounding groundwater pumps.

The basin is primarily in the zone of influence of Pump 101. Any leakage from the basin reaching the shoestring sands is likely to move S/SW towards Pump 101.

3.3
Sampling and Monitoring (Prior to 1997)

The following data is available from the field investigation and monitoring program by Goulburn-Murray Water and Sinclair Knight Merz (prior to 1997) and CSIRO Land and Water (1997 to 1999):

1. EC measurements for water in Bays A, B and C (1987-1999). Data converted to [Cl] concentrations using [Cl]:EC relationship for basins and used in leakage estimation using the [Cl] mass balance technique.

2. EC measurements from monitoring bores within and around the basin. Data is used to determine the development of the leakage plume around the basin.

3. EM39 profiles (downhole) for selected monitoring bores taken soon after commissioning the basin and every few years following. Data is used to determine leakage flow processes under the basin (e.g. piston vs preferential)

Also presented in the following section is a brief description of how the data is used in discussions later in this paper.
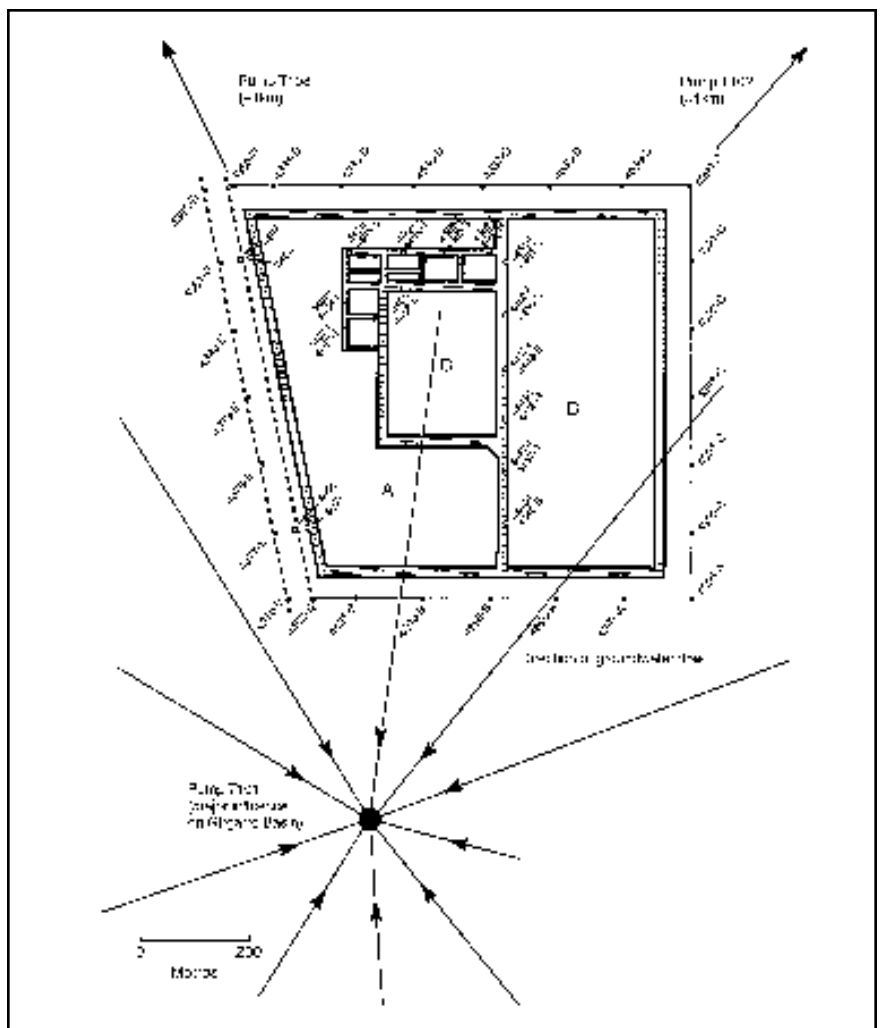
Figure 3   Bores for monitoring groundwater depth and salinity at the Girgarre Basin.

All perimeter bores have screen depths of ~15 m. Internal bores have both shallow (~3-4m) and deep (~15 m) screens.

3.4.1 Soil sampling and analyses

Soil cores were taken by hand-augering in Bays A and B at a distance of ~15 m from the edge. Two cores were also collected in Bay C at the edge and ~12 m from the edge (Figure 4). During coring, water from the basin was isolated from the auger hole using a 300 mm diameter steel tube partially inserted into the basin bed. Samples were taken at 100-300 mm intervals to depths of 3.6, 4.4, 4.3 and 3.6 m for Bays A, B, C and C (edge) respectively. Following augering, the holes were completed as piezometers with screens placed at the bottom 200 mm.

Soil samples were analysed for gravimetric water content and for chloride concentration in the soil-water. Some of the samples were analysed for stable isotopes ^2H, and ^{18}O after collecting the soil-water by azeotropic distillation. Particle size analyses, PSA, and soil-water potential measurements (SWP) were made on most cores. Short cores up to 1 m long were also collected from the basins and sampled at intervals (Figure 4). In addition to the above analyses, some of these samples were stored in an insulated container filled with ice and later analysed for polysaccharide concentration. Details of analytical methods are given in Appendix A.

3.4.2 Water sampling and analyses

Samples were collected from the basin water in each of the bays at similar locations to the historical sampling. In addition, during the course of this study, two sets of samples were collected from each of the three bays to determine the degree of mixing within each bay. The first was during mid winter when inflow from groundwater pumping was low, and the second during summer when inflow was near its maximum. The samples were collected at a distance of ~30 m from the edge of each bay at locations shown in Figure 4.

Groundwater was collected from beneath the bays from the piezometers drilled during soil sampling, and also from the monitoring piezometer 4447- S and 4447-D on most of the field trips (approximately every 3-4 months). The piezometers in the bays consist of 50 mm PVC tubing with slotted casing and a cloth sock at the bottom. They were drilled (as part of the soil sampling program) and the outside of the tube back-filled with Bentonite to prevent leakage down the outside of the annulus. Samples were also collected from many of the monitoring bores during March 1997, and as sub-samples from the monitoring bores collected during routine monitoring by Sinclair Knight Merz (SKM) in November, 1998. Samples were taken from the input to Bay A, when water was being pumped into the basin, and from the interception drain when it was flowing. These samples were analysed for ^2H, and ^{18}O, salinity and [Cl].

3.4
Sampling and Monitoring (This Study, 1997-1999)
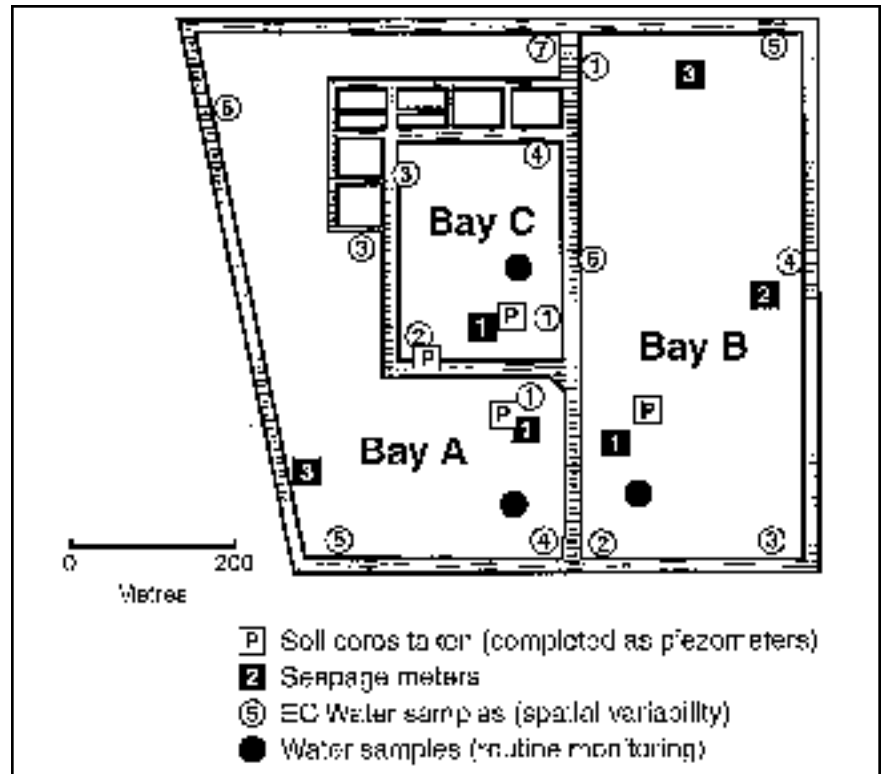
Figure 4.   Sites for seepage metres and for sampling basin water and soil at the Girgarre Basin.

3.4.3 Seepage Meters

A total of 11 seepage meters (5 in Bay A, 4 in Bay B and 2 in Bay C) were installed in the basin during 1997 (sites shown in Figure 4). These meters consisted of a 150 mm diameter PVC pipe capped at one end, and sharpened at the other. This pipe is attached, via 6 mm diameter tubing, to a 20 L plastic bladder. The meter is placed on the base of the basin, and tapped until a good seal is made with the clay at the bottom of the basin; the bladder is secured to the bottom of the basin nearby (Figure 5). This type of seepage meter allows the hydraulic head of water in the bag to be the same as that at the base of the basin. The amount of water lost by the bag for any time period is equal to leakage over the area of the pipe.

At four of the sites (two in Bay A and one in each of Bays B and C), paired seepage meters were placed within one metre of each other. In order to evaluate the effectiveness of the sludge as a retardant to leakage, one of the meters had the sludge removed prior to the pipe being inserted; the other did not. All other sites did not have sludge removed prior to installation of the seepage meters.

Initially, 20 L wine cask inserts were used as bladders. However, wave action resulted in these bladders being punctured; hence, they were replaced with containers used for water storage which had thicker PVC, but were still flexible enough to allow internal and external pressure of the water to remain equal. Initially, 15 L of water was accurately measured into each bladder and the remaining water measured on the subsequent field trip. Several of these bladders also punctured during the course of the study, but their success rate was much better than for the wine casks.
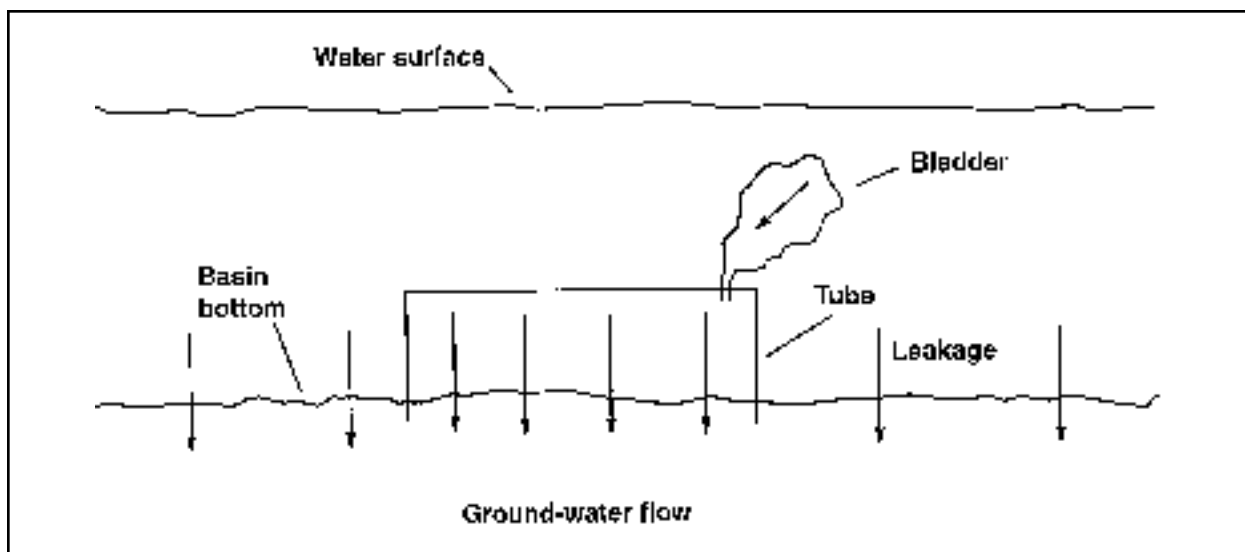
Figure 5. Schematic of seepage meter

4. On-farm basins in the MIA (Site and Study Description)

4.1
Nehme Basin

The Nehme basin was on a 50 ha vineyard in the MIA, 30 km north of Griffith. It was a newly constructed, triangular, 2 ha evaporation basin split into two bays. The vineyard was established in 1994 after previously being used to grow rice up until 1989, and vegetables until 1994. This site was chosen because the newly constructed evaporation basin enabled measurement of initial soil and hydraulic conditions under the basin, and initial rates of infiltration.

4.1.1 Farm description

Several red and white grape varieties were planted over a four year period from 1994 to 1997. The farm was flood irrigated using broad based furrows. Irrigation occurred around every 12 to 18 days and took 2 to 3 days to complete. Irrigation advance times were long, with water typically taking around 14 to 18 hours to reach the end of the 500 m long furrows. Farm drainage ditches were formed, but were usually too few; water was held at the bottom of the furrows for long periods. Water application across the farm was not evenly distributed as small sections of the farm were watered individually, depending on grape variety and age.

Subsurface pipe drainage using 100 mm corrugated pipe with gravel envelope was installed into 25 ha of the farm at the beginning of 1997. The remaining 25 ha of the farm was drained 9 months later, in September. The lateral tile drain lines were 1.8 m deep at 36 m spacing, with a sealed collector main running to the sump. All subsurface drainage was pumped from the sump into the evaporation basin.

When the basin was commissioned in January 1997, all subsurface drainage entering the sump was pumped into the evaporation basin. At the time the watertables in the farm were within 1.5 m of the surface. The tile drain pump was run continuously drawing down the watertable to the drain level. This proved to be poor management as the basin was rapidly filled within 3 weeks.

4.1.2 Basin design

The evaporation basin was an above ground construction, consisting of 2 bays of 1.07 and 1.04 ha; the gradient of the banks was 1:3. The maximum depth of the bays was 0.95m and 1.1m respectively, with a maximum capacity of 10.8 ML and 10.4 ML.

Drainage water was pumped into the first bay via a 150 mm PVC pipe from the pump, and overflowed into the second bay through a pipe between the bays when the water level was greater than 0.56 m. At a later date, the pipe between the bays was removed, and a channel dug between the basin at floor level; this enabled both bays to fill almost simultaneously. It also allowed the water to spread across the greatest possible area to increase evaporation. Water only left the basin by evaporation, vertical leakage and lateral leakage. There were no arrangements for overflow. Once the basin was full, pumping of drainage water was stopped.

An interception pipe drain to collect shallow lateral seepage was installed about 1.5 m below ground level around the perimeter of the evaporation basin, at a distance of about 10 m from the inside bank. This drain was connected to a subsurface drain line in the farm that returned any intercepted leakage to the main pump sump. Two inspection pits were inserted into this interception drain line, which enabled measurement of the quantity and quality of lateral leakage.

The basin area was 8 % of the drained farm area initially, when only 25 ha of the farm had subsurface drainage. When the remaining 25 ha of the farm was drained, the proportion of basin area to drained area was reduced to 4 %. This was well below the 10% basin area recommended by Murrumbidgee Irrigation and the Department of Land and Water Conservation, NSW at the time.

The basin was sited in a disused part of the farm, which was unsuitable for irrigation due to its elevation and triangular shape. This portion of land had never been irrigated; the water table was below 7 m when piezometers were inserted before the basin was filled. This basin was situated approximately 50 m from a main supply canal on one side, and 20 m from a drainage channel on the other.

4.1.3 Soil and geology

The site was on the edge of the Riverine Plain, against the foothills of the Palaeozoic massif of eastern Australia. On the Riverine Plain, there is generally a mixture lacustrine and fluviatile deposits. The plain is an alluvial fan with ancestral rivers and prior streams; these prior streams and rivers deposit coarse sediment layers in the dominant silt and clay. At this particular site there is no influence of ancestral rivers or prior streams. Nor are there any windblown sand drifts as found in many areas. In this area there is no shallow aquifer system, with up to 30 m of uninterrupted clay before brown coal and bedrock (van der Lely, 1974)

The soil at the site was a Griffith Clay Loam (Butler, 1979). This soil is described as typically having the top 0.3 m as a clay loam that becomes progressively heavier with depth down to about 0.9 m, and then continues as a medium clay. The deep subsoil ranges from a light to heavy clay with soft

and hard carbonate (Figure 6). Van der Lely (1974) documented hydraulic conductivity measurements of Griffith Clay Loam soils using the auger hole method. Measurements were documented from 96 sites with varying watertable conditions (Table 1).
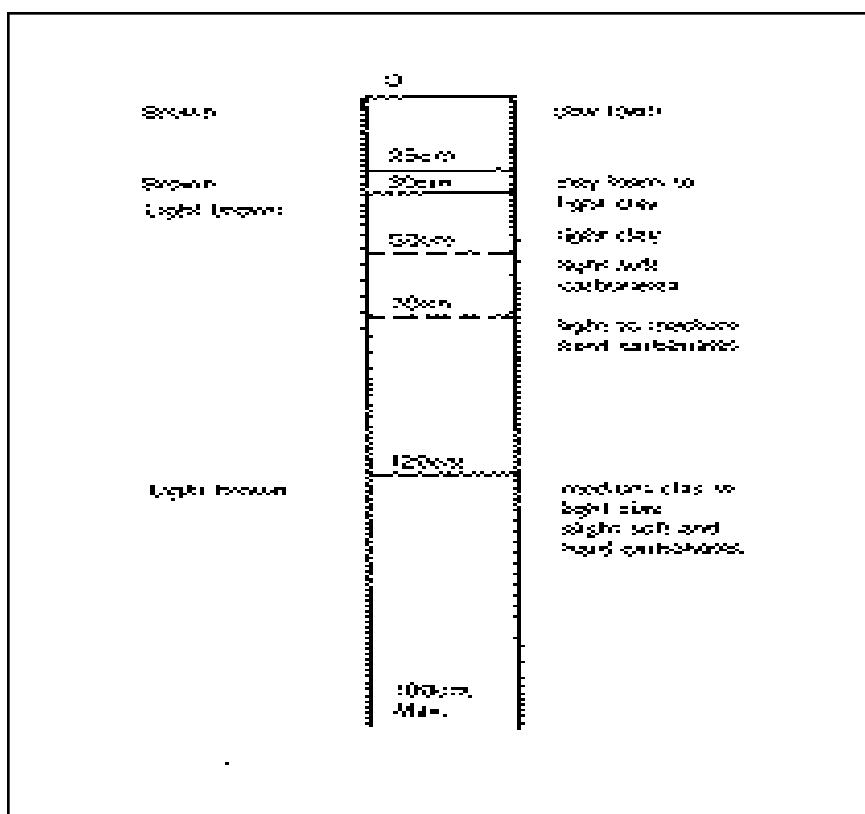
Figure 6.   Characteristic Griffith Clay Loam soil profile (after Butler, 1979)

Table 1.   Hydraulic conductivity of Griffith Clay Loam (after van der Lely 1974)

Depth range (m)	Average hydraulic conductivity (m/day)
0.5 – 0.9	0.55
0.9 – 1.4	0.12
1.4 – 1.9	0.06

4.1.4 Climatic conditions

The climate in the MIA is characteristic of semi-arid areas with hot dry summers, and mild winters and occasional frosts. The average highest and lowest temperatures in the summer months are 32^0 C and 14^0 C, and 16^0 C and 3^0 C in the winter months. Mean rainfall for the Griffith area is

418 mm, fairly evenly distributed throughout the year. Annual rainfall is highly variable ranging from 140 to 700 mm. Mean potential evapotranspiration is 1800 mm. Evapotranspiration greatly exceeds rainfall in the summer months and closely matches rainfall in the winter months between April to August (Table 2). The study period covered two years of fairly dry weather conditions (Figure 7).

Table 2.   Mean monthly climate data for Griffith (CSIRO Land and Water)

Month (m)	Rainfall (mm)	ET (mm)	Max Temp. (^{0}C)	Min Temp. (^{0}C)
January	36	275	31.8	16.1
February	27	228	31.0	15.9
March	33	187	28.1	13.5
April	36	112	23.1	9.2
May	41	65	18.4	6.1
June	34	43	15.0	3.8
July	37	49	14.2	2.9
August	37	74	16.2	3.9
September	36	118	19.5	5.6
October	43	172	23.2	8.7
November	28	224	26.9	11.7
December	30	268	29.9	14.3
Total	**418**	**1808**		
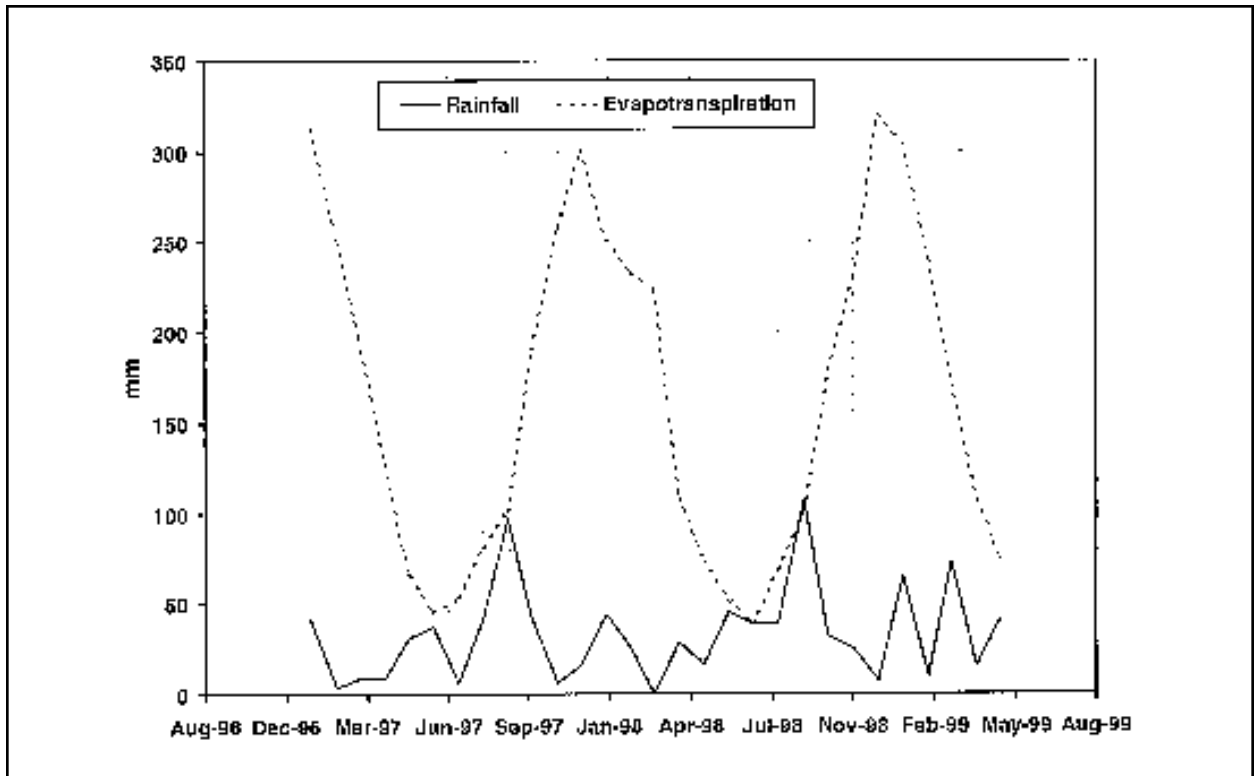
Figure 7.   Rainfall and reference evaporation for the study period

4.1.5 Measurements

A number of measurements were taken which are outlined below. Figure 8 shows the placement of various instruments and the sampling sites.
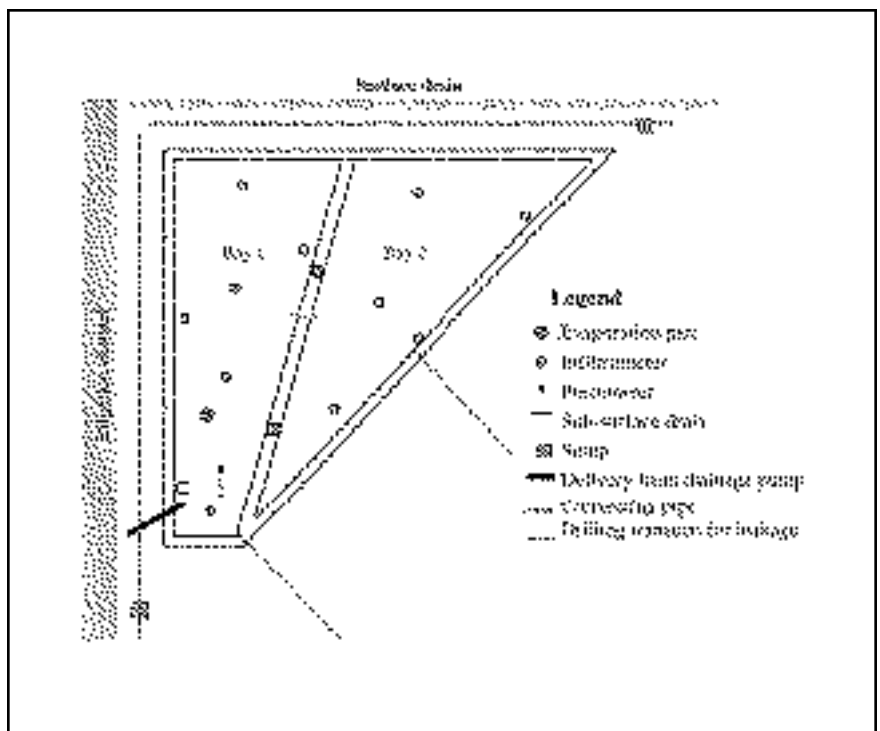
Figure 8   Position of instruments and sampling sites at Nehme Basin.

- *Basin inputs*

  Subsurface drainage water pumping into the basin was measured by an electronic paddlewheel flow meter. The salinity of this water was measured by collecting a 250 ml sample for laboratory analysis. Rainfall was measured by a rain gauge on the basin bank.

- *Basin water volume*

  The evaporation basin was surveyed at the beginning of the experiment before filling, and basin area and volume determined. The water level in each of the two bays was measured using a stilling well made of 50 mm PVC pipe slotted at the bottom inserted into the basin floor.

- *Basin water salinity*

  Basin water salinity was measured by collecting a sample in a 250 ml plastic bottle from each bay and analysing it for electrical conductivity and chloride in the laboratory. The sample was taken at the same point each time, from 50 to 100 mm below the water surface. Samples were taken coinciding with input drainage samples.

- *Piezometric pressures*

  The groundwater levels were measured using piezometers made of 90 mm PVC pipe at 2, 3, 4, and 7 m depths, with the bottom 300 mm slotted and sealed with bentonite. These were positioned in a group in the centre of Bay 1 before the basin was filled (Figure 8). When the piezometers were being inserted the soil conditions were very dry and no water was found at 7 m depth.

- *Basin leakage - Vertical*

  Vertical leakage was measured indirectly using a water and salt balance for the basin, and directly using seepage meters and piezometers installed in the basin before filling.

- *Water balance method*

  The vertical leakage was estimated using the water balance method. Estimates of evaporation were based on the Griffith laboratory weather station and assuming a pan factor of 1.

- *Seepage meters*

  Once the basins were filled with water, 11 seepage meters were installed in the basin, 6 in the first bay and 5 in the second bay. The seepage meters were made from 150 mm diameter UPVC pipe. These pipes were hammered into the floor of the basin to a depth of 250 mm. Once installed, lids were placed on the pipes to prevent water evaporation. The water level in the seepage meters was maintained close to basin water level. Changes in water level inside these seepage meters were measured weekly using Vernier callipers.

- *Piezometers*

  The vertical leakage was also estimated at the initial filling by using piezometers installed in the basin before filling at depths of 2, 3, 4 and 7 m. Whilst installing these piezometers the soil was found to be extremely dry (approximately that of the lower drainage limit), even at a depth of 7 m. This area had never been irrigated and supported native vegetation before basin construction.

- *Soil cores*

  Soil cores were taken from Bay 1 and were analysed for soil water potential and clay content whilst the bay was full. When the basin dried out after 17 months soil cores were taken at 100 mm intervals to 6 m. These samples were analysed for chloride and electrical conductivity.

- *Electromagnetic survey*

  An EM38 survey of the basin was carried out to assess the spatial distribution of salt and hence some assessment of the spatial variation in leakage. EM38 readings were taken in a 5 m grid across the basin. Five soil cores to 1 m were taken in the basin floor to calibrate the readings.

- *Lateral leakage*

  Lateral leakage was measured as the flow in the interception drain around the basin. Two sumps were inserted into the perimeter interception drain line (Figure 8), in May 1997, five months after filling. Flow rates were measured manually with a stop watch and measuring container. Water samples were collected at the same time in 250 ml plastic bottles, and analysed for electrical conductivity and chloride in the laboratory. Duplicate water samples were analysed for ^{2}H and ^{18}O.

  Water samples were also collected from the open drain running along the northern side of the basin to determine if saline water from the basin was seeping into the drain. This drain was about 2 m below basin bed level. Samples were taken upstream and downstream of the basin using 250 ml plastic bottles. Samples coincided with basin water samples.

  Two transects of holes were augered to 2 m deep. These started at the outside toe of the basin bank and went 100 m into the vineyard (Figure 8). The groundwater depth in these holes was surveyed to establish if there was a groundwater mound from the basin. Water samples were also collected for ^{2}H and ^{18}O analysis to determine if a leakage plume was developing from the basin.

- *Evaporation*

  There is a large variability in the literature values of evaporation coefficients for saline water bodies because of the many variable site specific factors that influence evaporation. At Nehme Basin, we attempted to measure evaporation. Four class A evaporation pans were

installed in the basin, two into Bay 1 and two on the bank between the bays (Figure 8). The pans were filled to within 50 mm from the rim and read using rulers glued to the insides of the pans. The water level on the outside of the pans was maintained within 50 – 100 mm of the rim by moving the pans up or down in the basin water. Readings were taken every 1 – 3 days, and the pan water was changed with basin water each time. For general calculations, evaporation was initially estimated using daily reference evapotranspiration data from CSIRO Griffith. These were then corrected to a pan factor of 1 from the field measurements over March to May.

4.1.6 Sequence of major events

Date	Event
13/12/96	Started pumping into ponds
6/1/97	Commenced intensive sampling
23/1/97	Bay 1 full
5/97	Sumps installed in perimeter line for flow measurement
5/8/97	Lateral seepage investigation using groundwater transects
12/9/97	Tile drains inserted in remaining 25 ha of farm, basin size decreased from 8% of drained area to 4% of drained area.
26/2/98	Basin Bay 2 dry, Bay 1 nearly dry
3/98	Soil sampling and EM38 survey of bays
10/98	Bays refilling

4.2 Other Basins in the MIA

Further to the intensive monitoring of the Nehme basin a further 13 on-farm basins in the MIA were monitored on a monthly basis for:

• input water - based on pump electricity readings calibrated for flow.

• salinity – monthly sample of input water salinity and basin water salinity.

• interception drain flows – occasional measurements.

• basin geometry.

These basins were sited on similar soils under similar conditions to the Nehme basin. All the basins were associated with subsurface pipe drainage in the farm and had subsurface pipe interception drains around their perimeter. Evaporation and rainfall was determined from the Griffith weather station and assumed to represent each site.

At some of these basins, EM38 surveys were undertaken as transects across the basin. Readings were taken every 5 m in an attempt to detect any shallow saline leakage plume movement. At the basins where there were open drains nearby, samples of the drain water were taken upstream and downstream of the basin to assess whether the basin leakage was entering the drain.

5. Results from Girgarre Basin

5.1

Spatial and temporal changes in basin water salinity

Historical data for salinity was converted to [Cl] using empirical data collected from the basin and piezometers during the period 1997-1999 (Figure 9). By using this EC vs. [Cl] relationship, the maximum [Cl] observed for the three bays at the Girgarre basin are 10, 25 and 100 g/L for bays A-C respectively (i.e. 10,000 – 100,000 mg/L).
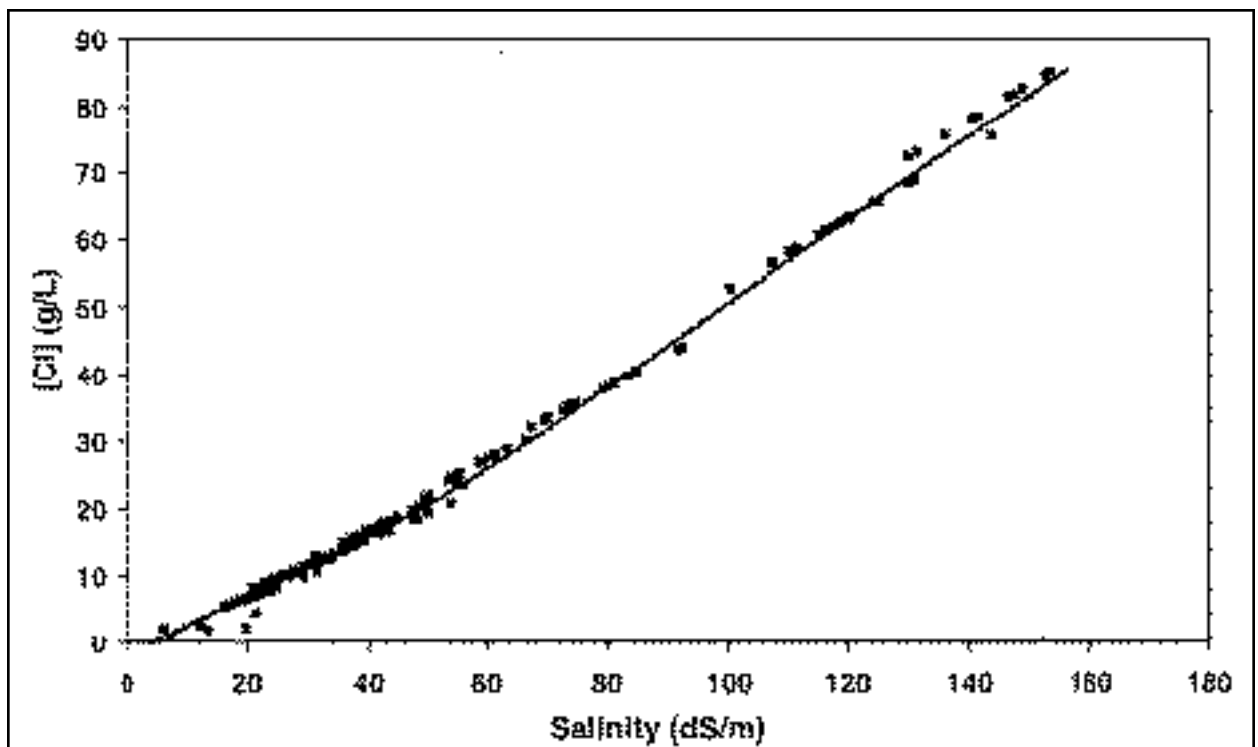
Figure 9.   Chloride concentration vs electrical conductivity relationship for groundwater near the Girgarre Basin.

Figure 10 shows that all three basins initially had the same increase in salinity. Then, after half a year, the water in bay A reached a [Cl] of approximately 10 g/L while that in the other two bays continued to rise. Bay B reached a steady state [Cl] of 10 – 20 g/L within a year or two, while that in Bay C reached 60 - 100 g/L by 1997. A few years of data is missing for the interval between the initial sampling ceasing and the sampling for this study commencing.

Results from the spatial sampling of the salinity for the bays suggest that the bays are relatively well mixed, despite being relatively large and shallow (see Figure 4 for sampling locations). The percentage standard deviation for salinity analyses ranges from 0.14 (for Bay C) to 3.3% (for Bay A) for the September sampling period (spring) and two to three times greater for the February sampling period (summer) (Table 3).

This is due to the shorter residence time experienced in each bay during summer, when the evaporative potential is higher, and rainfall usually lower, than in winter. This results in more groundwater pumping to the basin during summer months. The opportunity for water within the bay to mix throughout the bay is greater for higher residence times, compared to periods of lower residence times. Clearly, the size, geometry of the bays and the location of the water input and output also impact on the degree of mixing.
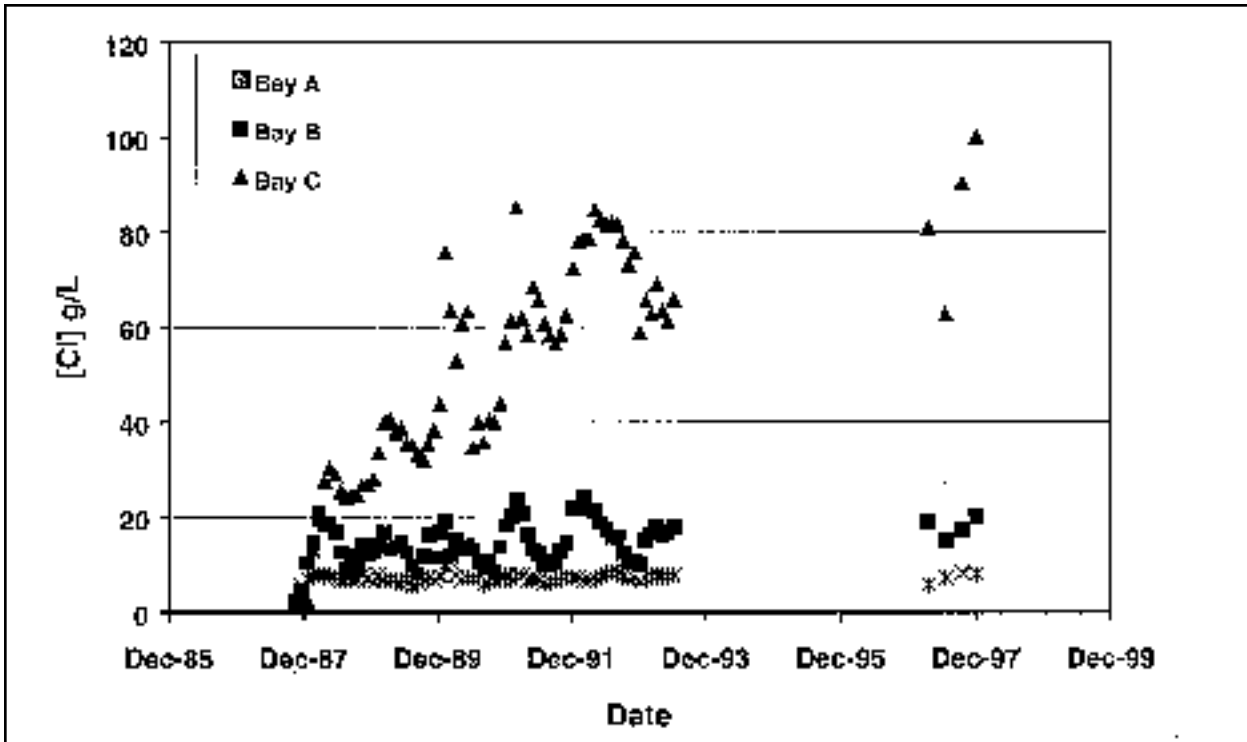
Figure 10   Change in basin water chloride concentration over time for Bays A,B and C at the Girgarre Basin.

The [Cl] (and salinity) of water in Bays A and B reach a steady state concentration soon after commissioning while that for the terminal bay (Bay C) continues to rise for several years. Seasonal changes in salinity resulting from dilution via winter rains and concentration via high evaporation in summer are observed for Bays B and C.

Table 3   EC measurements (dS/m) from varied sampling points across each bay at the Girgarre Basin.

Site	A1	A2	A3	A4	A5	A6	A7	Mean ± 1 S.D.
Sep 97	24.2	23.7	24.0	23.4	23.4	23.9	25.7	24.0 ± 0.8
Feb 99	23,2	24.4	24.7	20.4	23.5	25.4	26.2	24.0 ± 1.9
Site	B1	B2	B3	B4	B5			Mean ± 1 S.D.
Sep 97	45.5	47.5	46.6	45.3	45.2	47.2		46.2 ± 1.0
Feb 99	62.7	52.8	57.6	61.6	63.5	60.8		59.8 ± 4.0
Site	C1	C2	C3	C4				Mean ± 1 S.D.
Sep 97	138.6	139.0	139.0	139.0				138.9 ± 0.2
Feb 99	187.0	186.0	185.0	185.0				185.7 ± 1.0

Samples collected by Goulburn-Murray Water and CSIRO staff were collected at approximately the same place on each sampling occasion. Clearly, the conductivity of water collected from each of the bays is likely to be closer to the mean of the bay during winter than during summer. As the sites where water samples were taken is nearer the input than the output for Bay A, it is likely that the mean salinity for Bay A is greater than the measured values. Conversely, for Bay B, the site where the samples were taken is closer to the output and the mean concentration of the bay is likely to be less than that measured. However, the difference is relatively small and has little effect on the estimation of leakage rate (see Section 5.8).

The location of monitoring bores at Girgarre was given in Chapter 3 (Figure 3). In order to evaluate temporal changes in the salinity of the monitoring bores, we have grouped the deep and shallow bores according to their location with respect to the basin and the groundwater pumps (Table 4).

5.2 Groundwater

Table 4 .   Grouping of groundwater monitoring bores and trends in groundwater salinity for the Girgarre Basin.

Group #	Bore #	Location	Comments/Salinity Trend
Perimeter Deep - Group 1	4395-4398	SE corner of basin	GW salinity decreasing
Perimeter Deep - Group 2	4399-4402 & 4376	SW corner of basin	Closest to pumpT101 GW salinity increasing after 1992
Perimeter Deep - Group 3	4389-4394	NE corner of basin	GW salinity constant
Perimeter Deep - Group 4	4381,4383,4385-4387	NW corner of basin	GW salinity increasing after 1992
Perimeter Deep - Group 5	4377-4380	West of basin	Slight increase in GW salinity after 1992
Internal Deep - Deep	4446-4457	Internal W of bay B	GW salinity increasing after 1992
Internal Shallow - Wide Bank	4446S-4451S	Internal W of bay B	GW salinity increasing from 1988
Internal Shallow - Narrow Bank	4452S-4457S	Between bay C and A	GW salinity increasing from 1988

5.2.1 Groundwater Salinity

All deep bores showed very little change in salinity for the first 6 years of pump operation suggesting that leakage from the basin did not reach the shoestring sands during this time (Figure 11). However, changes have been observed in the conductivity of most of the groundwater collected from the monitoring bores since sampling recommenced in 1996/7. Some bores have had increases in conductivity while others show no difference or have decreased.

The EC for groundwater in bores from groups 2,4,5 and internal, I, has increased in salinity during the monitoring period; these bores are located between the bays or in the western half of the basin. The groundwater in bores from Group 3, where no significant change in salinity has been seen, are located to the north-east of the basin. Bores from Group 1, where a decrease in salinity is observed in the groundwater, is located to the south-east.

Salinity data for the shallow bores between Bays A and B (Wide Bank) and those between Bays A and the smaller trial bays (Narrow Bank) both show similar overall increases in salinity of ~20–25 dS/m from 1987 to 1998 (Figure 11). However, the bores sited on narrow banks display an earlier and

more rapid rise in salinity compared to those on wider banks. Both groups of bores have screens approximately 3-4 m from the ground surface (1-2 m below the water table). The bores on the wider banks are located approximately 4 m from the edge of Bay B while the bores on the narrower banks are located within a metre of the water in Bay A.
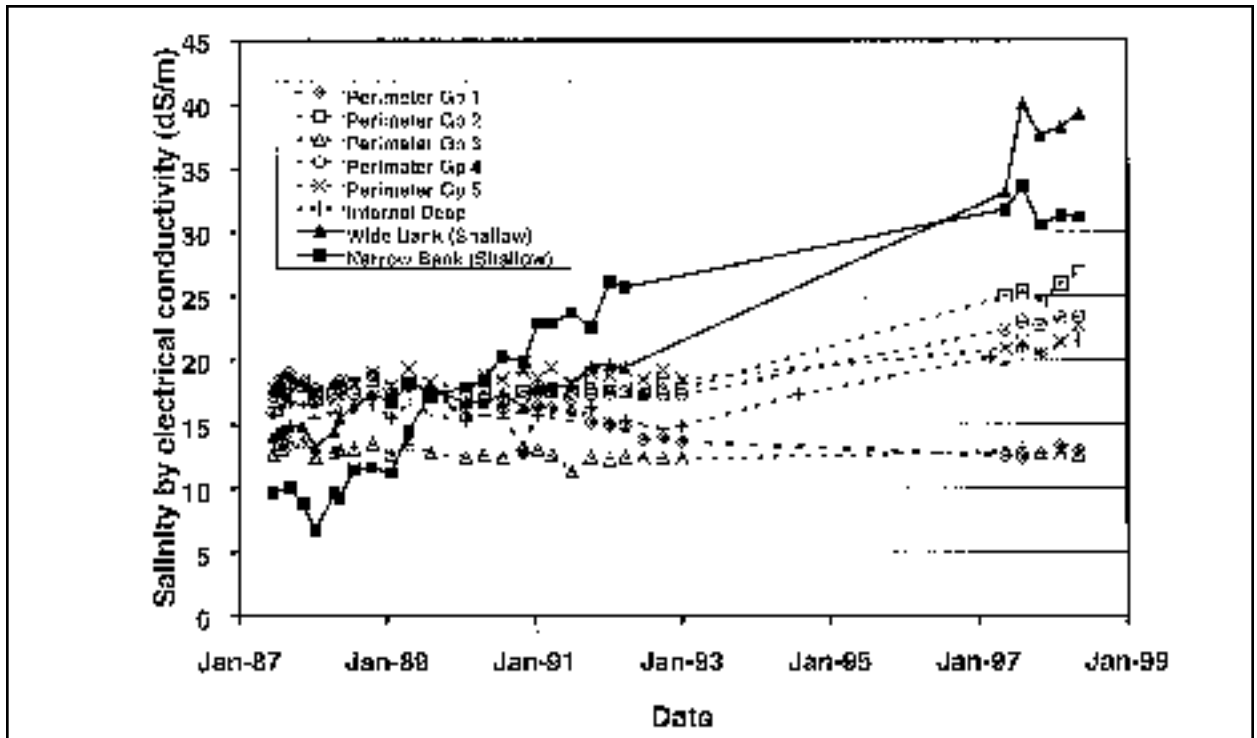
Figure 11.   Changes in salinity for monitoring bore groups in the Girgarre Basin

The change in salinity of monitoring bores surrounding (perimeter) and within (internal) the basin suggests that leakage from the basin reached a depth of 3-4 m within a year or two after the basin filled (solid lines). Leakage from the basin did not reach the shoestring sand aquifer until 6-9 years after filling and is mainly observed in areas where the monitoring bores are between the basin and Pump T101 (dotted lines). Groundwater pumping is causing lateral flow, in the shoestring sands beneath the basin. The increase in salinity of soilwater(~20 dS/m) and groundwater (<10 dS/m) is reasonably small compared to the salinity of the water in the basin (10–70 dS/m).

5.3
EM39 Profiles

At several of the monitoring bores, down-hole EM39 profiles were measured, commencing approximately one year after the commissioning of the basin, with the most recent measurements in September 1996. EM39 measurements have been used to estimate absolute values for groundwater salinity when soil physical parameters are known. We have presented the results of EM39 profiles taken in 1988 and 1996 for three sites. The changes seen at these sites cover the range of changes seen for the basin at the monitored sites over the 8 year period. This data is interpreted as relative changes in the groundwater salinity at different depths below the basin for this period (Figure 12).

All of the EM39 profiles for the basin have higher values near the surface and at the bottom of the bores (in the shoestring sands). If the soil had similar texture throughout the profile, this would suggest higher salinity levels at these depths. The higher salinity levels near the surface correspond to the high salinity values observed from soil cores, for depths of up to a few metres below the basin; these clearly represent leakage via matrix flow from the basin.
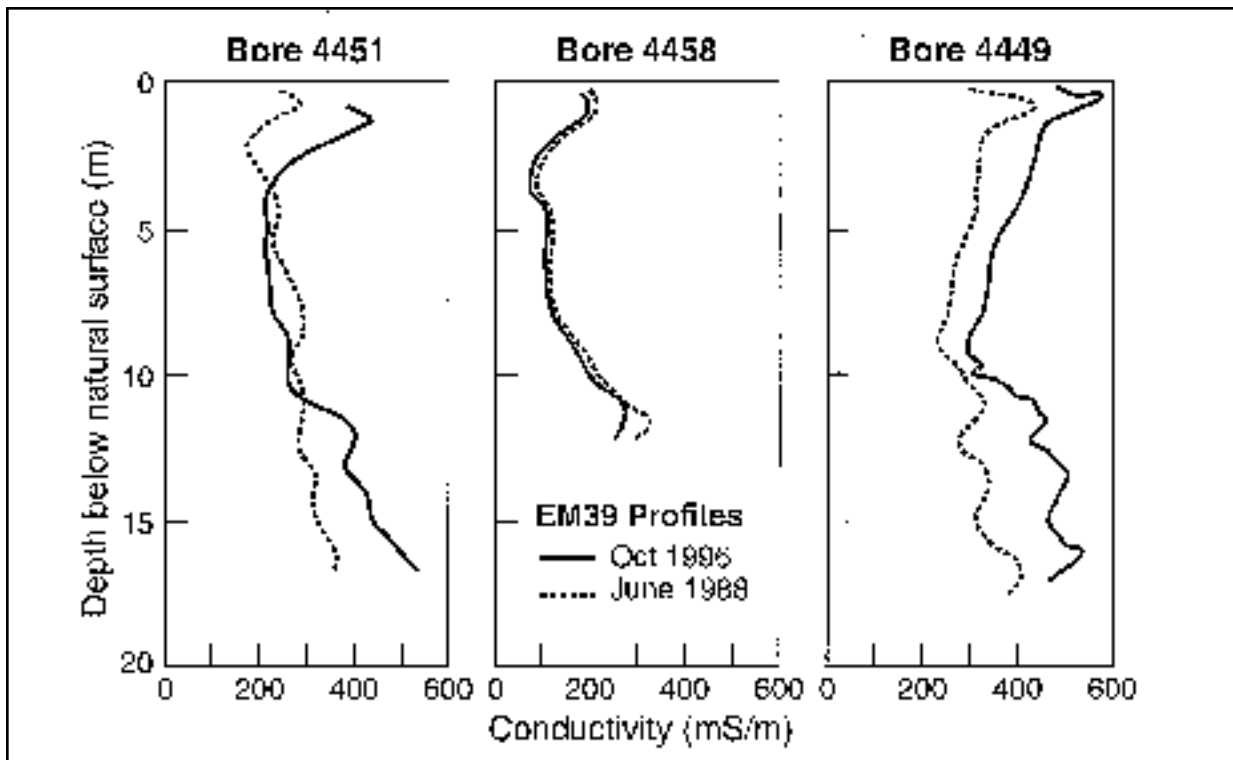
Figure 12.   Apparent conductivity of soil below basin using EM39 (representative profiles shown).

The salinity of soilwater and groundwater beneath the basin indicates that leakage from the basin is spatially variable. The most common profile, as shown for Bore 4451, suggests that leakage from the basin affects the salinity of the soil and groundwater immediately beneath the basin and that in the shoestring sands. Soil and groundwater at intermediate depths are less affected. Less commonly, there is evidence of leakage affecting the whole of the soil matrix (4449) or, alternatively, none of the soil matrix (4448).

For most of the bores, there is a marked increase in the salinity of the groundwater at the level of the shoestring sands. At intermediate depths, most profiles show little change in salinity (as shown in profile 4451) although, occasionally, salinity increases are seen throughout the profile (e.g. 4449) or there is no change in the groundwater at all (e.g. 4458).

5.4
Soil Sampling

Soil-physical measurements of water content (g), particle size and soil-water potential (SWP) were undertaken to assist in determining factors that may be controlling leakage.

A layer of black silty material was found at the base of each of the bays, the depth of which ranges from ~100 mm to ~400 mm. The layer varied in depth within each bay and from bay to bay. The layer of silty material appears to be less in Bay C than the other two bays. All of the cores appear saturated throughout, and for a distance of ~30 mm below the silty material. Below that depth, the samples appear unsaturated to the water table.

The soil in Bay A was categorised as Goulburn loam (with colour 7.5YR3/4) and that in Bays B and C as Congupna clay loam (10YR3/3). The gravimetric water content for soil beneath the basin ranges from ~0.2 to ~0.35 with a mean ~0.25 (Figure 13). Clay content for the soil ranges from ~20 to ~70% with a heavy clay layer ~0.5 m beneath the basin floor (Figure 14). The range in the water content for the soil is, in part due to the different clay content of the soil.
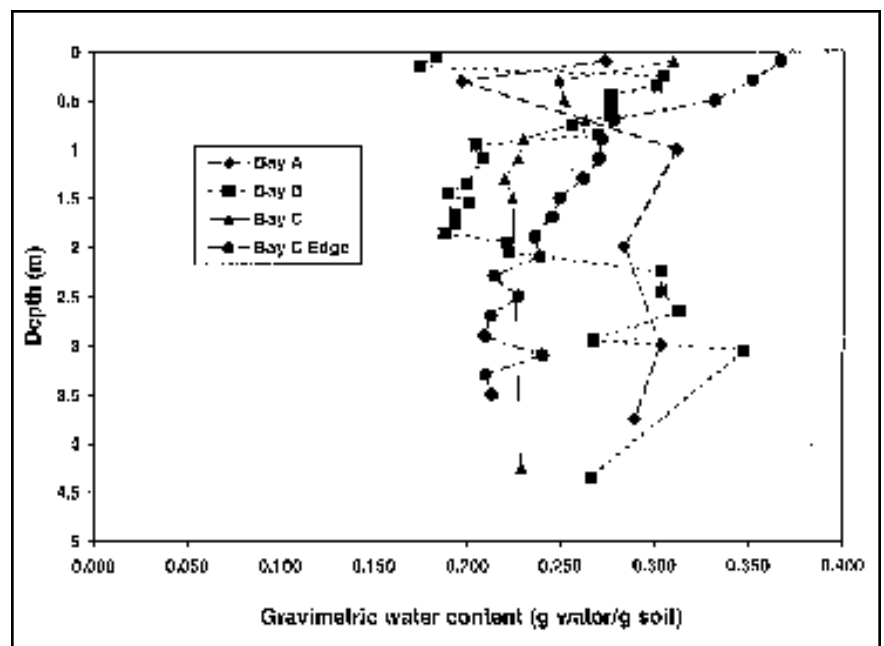
Figure 13.   Gravimetric water content for soil beneath the Girgarre Basin

SWP measurements were made on core samples collected from beneath Bay A (January 1997) and from short cores from each of the bays in February 1998. The water table was ~1.5 m below the basin bottom when both sets of cores were collected. Most samples are very close to saturated but, as suggested from visual observation, the samples are not saturated between the silty material at the base of the bays and the water table (Figure 15).
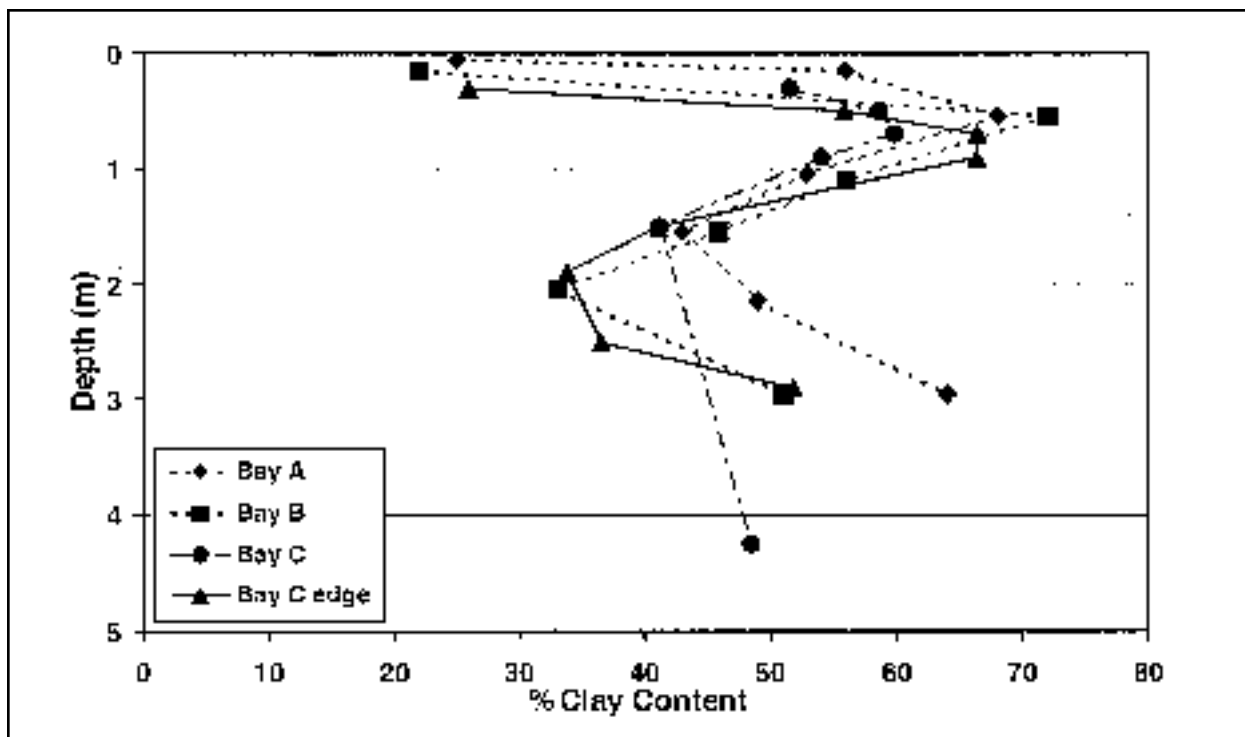
Figure 14.   Clay content for soil beneath the Girgarre Basin.

Clay contents in the soil are more than 30 % for most of the soil profile but highest at 0.5 – 1.0 m beneath the base of the bays.

There appears to be considerable variation in the SWP for the different cores, regardless of whether they are in the same bay or in different bays. The variation is most probably due to spatial variation in leakage rates between bays and within each bay. However, leakage from basin water while sampling the cores may have resulted in erroneously low values for SWP for some of the cores (i.e. the soil may be less saturated than indicated from the analyses).
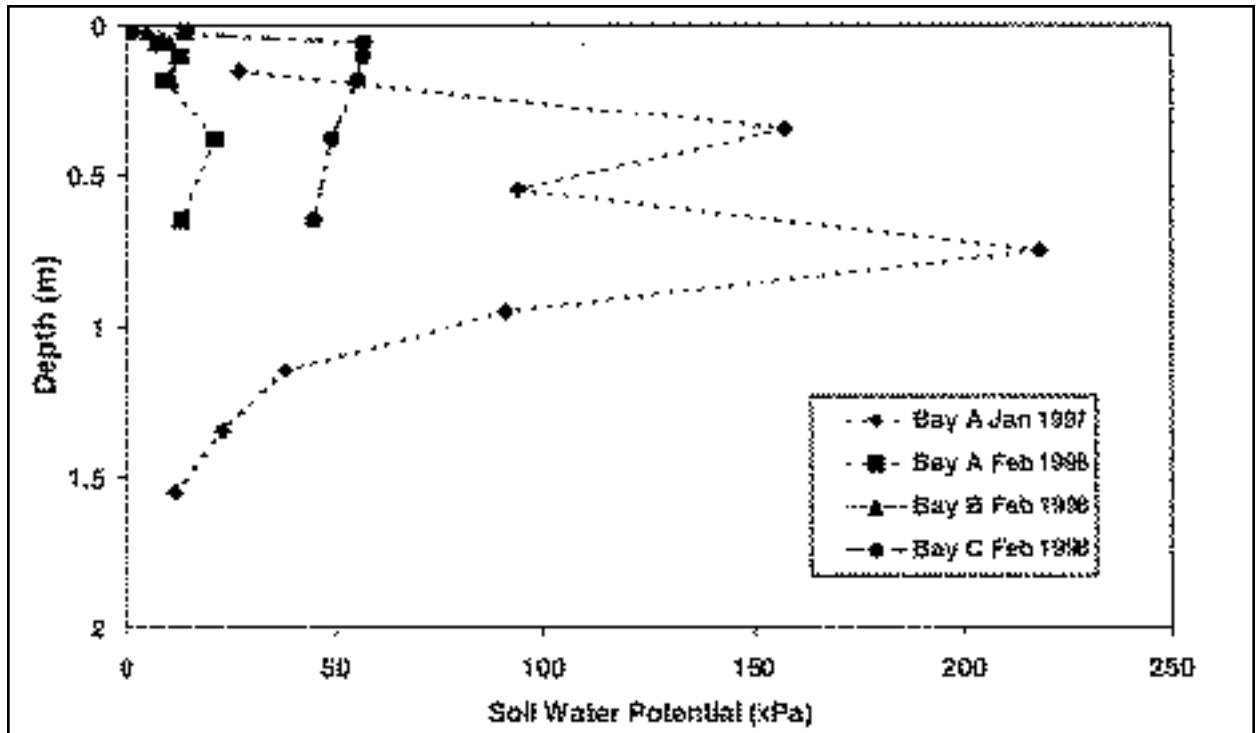
Figure 15   Soil water potential (SWP) for soil beneath the Girgarre Basin

The throttle for leakage at the Girgarre Basin is close to the soil/basin water interface near to the base of the basin as evidenced by the permanent unsaturated zone beneath the basin. The unsaturated zone ranges from ~0.5 m to > 1.5 m thick. Water table fluctuations beneath the basin are similar to those around the basin for areas within influence of the groundwater pumps.

5.5 Polysaccharide Analysis

We measured the polysaccharide concentration of sediment in the soil at the base of the bays at the Girgarre and Nehme (Griffith) basins (shown as solid points) and from soil at defunct basins at Pyramid Hill (shown as open points). For basins that are still operational, the polysaccharide concentrations from the base of the basin to a depth of ~0.1 m. ranged from ~1 to 20 mg/g dry sediment. The concentration decreased to <2 mg/g dry sediment at a depth of 0.7 m. For comparison, we also show the results for sediment in irrigation channels and for a laboratory column experiment by Ragusa *et al.* 1994, in which they seeded the soil with algal material. The polysaccharide concentrations are considered very high, reaching values ~4 to 10 times than that found in irrigation channels and, in many cases, are greater than the concentrations measured in the laboratory (seeded) experiments (Figure 16).

For the defunct basins, no samples were collected at depths less than 0.1 m. At depths greater than ~0.2 m, the polysaccharide concentrations were usually less than those measured in the laboratory experiments and from the sediment in the irrigation channels.
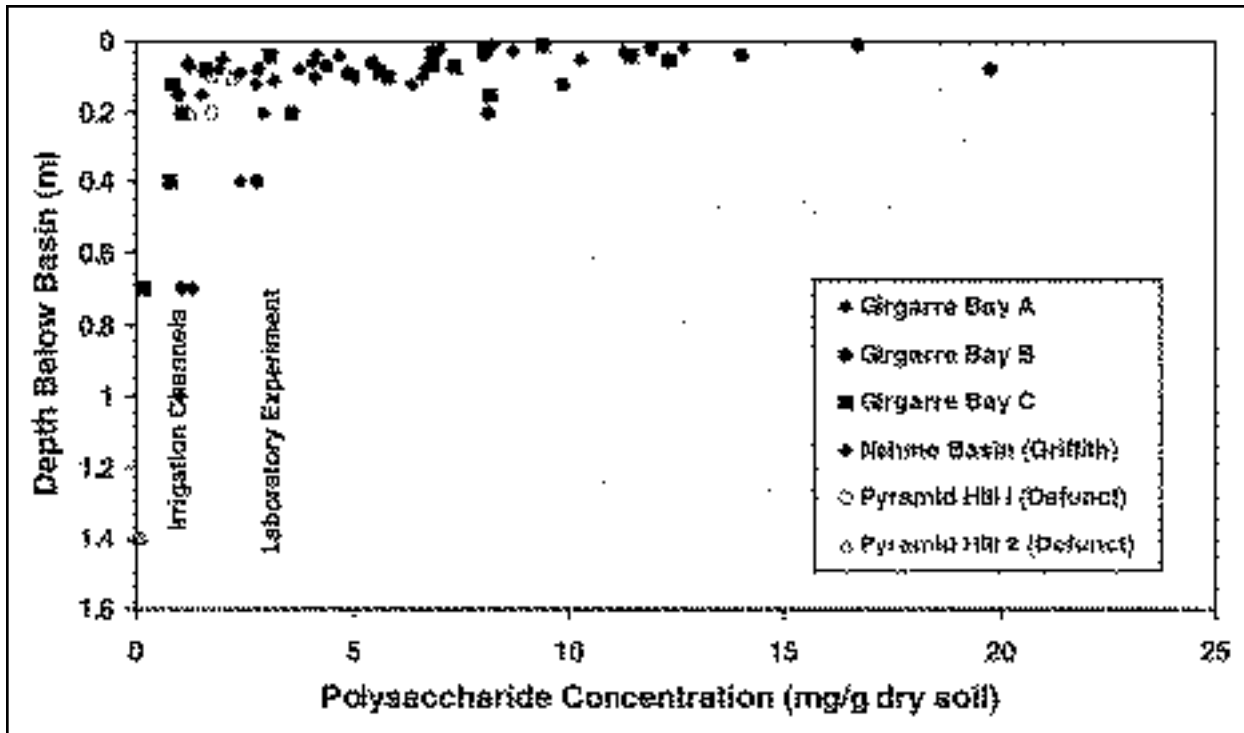
Figure 16   Polysaccharide analysis of basin soils in the Riverine Plain

Polysaccharide concentration is indicative of the amount of algal activity in a water body. High concentrations of polysaccharides may significantly reduce leakage by clogging pore spaces in the soil. Disposal basins provide an ideal environment for algal activity because they provide clear, warm, nutrient rich water for algal growth. In the soil at the base of basins that are still operational, polysaccharide concentrations are much higher than those observed in irrigation channels (~1 mg/g) and those in laboratory experiments (~3 mg/g), where conditions were designed to encourage polysaccharide production.

5.6
Seepage Meters

Leakage estimates were made using seepage meters in which the sludge at the base of the basin had either been left intact or removed. In removing the sludge, the first few centimetres of soil was also usually removed. Problems resulting from damaged bladders limited the results to 7 measurements at 3 sites for the sites with sludge removed, and 10 measurements from 5 sites for those with sludge intact. Mean leakage estimates for a period of about a year following the installation of the meters were 0.5 and 3.8 mm/d for leakage with sludge and without sludge respectively (Figure 17). Some of the leakage measurements (no sludge) made soon after installation were >12 mm/d (Table 5). Leakage for the meters with sludge removed tended to decrease with time after installation.
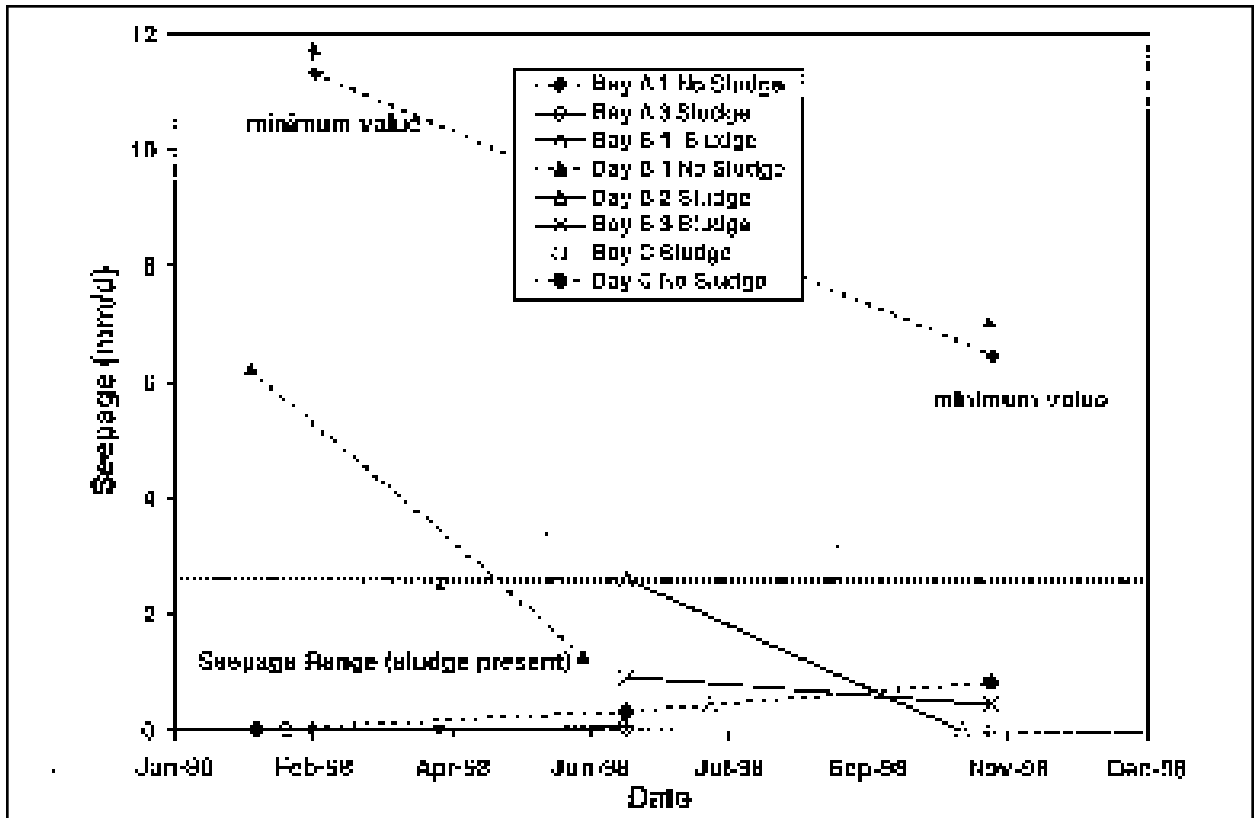
Figure 17   Seepage estimates using seepage meters at the Girgarre Basin.

Seepage is considerably higher if the sludge is removed from the base of the basin. In most cases, particularly when the sludge was removed, seepage rate reduced a few months after installation.

Table 5   Leakage measurements using seepage meter (3 monthly intervals between Feb-Oct 1998).

Site #	Comments	Leakage 1 (mm/d)	Leakage 2 (mm/d)	Leakage 3 (mm/d)	Average (mm/d)
A1	No Sludge	11.3		6.5	8.9
A3	Sludge	0	0	0	0
B1	No Sludge	6.2		1.3	3.7
B1	Sludge		2.6	0	1.3
B2	Sludge			0.5	0.5
B3	Sludge		0.9	0.5	0.7
C1	No Sludge	0	0.3	0.8	0.4
C1	Sludge	0		0.1	< 0.1
Average	No Sludge				3.8
Average	Sludge				0.5

For all of the bays, there is a decrease in the soil-water chloride concentration from the base of the basin (depth = 0 m) to a depth of ~2 m (Figure 18). At that depth, the soil-water chloride concentration is steady at ~3 g/L for Bay A, at ~4.7 g/L for Bay B, and at ~6.2 g/L for Bay C. Results for [Cl] profiles for the shorter cores are similar to those for longer cores in the same bay. The water level fluctuates by a metre or more beneath the basin but has not been recorded to drop below ~2 m below the basin base (i.e. the soil below 2 m is permanently saturated).

Most of the salt build-up beneath the basin, at least for these sites, is within the top few metres of the base of the basin. Given that the water ranges from ~0.5 to ~1.5 m table beneath the basin, this means that most of the increase in salinity of the groundwater is restricted to the top metre of the shallow groundwater, and to the unsaturated zone above it.

Chloride data was also measured on soilwater for the deep cores in Bay C from a site only covered with water during times when climatic conditions (high rainfall and low evaporation) result in the basin being >0.1 m deeper than its design depth. At this site, the maximum soilwater chloride measured near the base of the basin is lower than that measured at the sites in Bay C with permanent water cover. This is consistent with the leakage occuring during periods of high rainfall when the basin salinity has been diluted. However, the chloride concentration of the soilwater tends to be higher throughout most of the top 2 m of soil at the site with periodic wetting. This suggests maybe leakage is greater under these circumstances.

5.7
Chloride and Deuterium Profiles Beneath The Basin
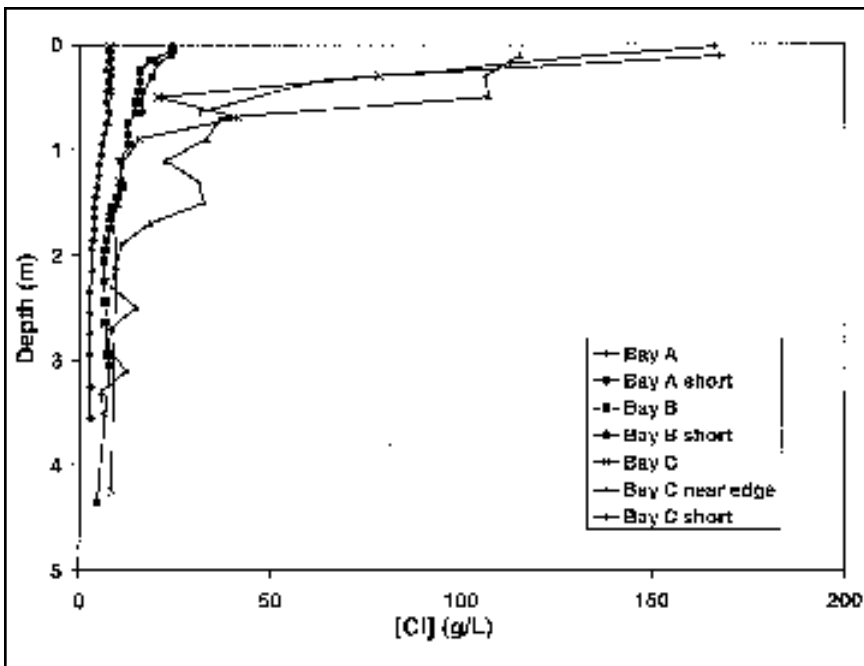
Figure 18   Chloride concentration of soilwater beneath the Girgarre Basin

At sites in the basin where there is permanent water coverage, most of the saline seepage is evident in the top metre of soil beneath the basin. At a site where water

cover is restricted to periods of high rainfall, saline seepage has progressed to two metres beneath the basin. Soilwater salinity for soil at depths greater than two metres are similar for all bays despite the large salinity range for the water in the bays overlying the soil.

For Bays A and C, the depth profiles of the ^{2}H composition of soil-water as a function of depth are similar to [Cl] profiles of soil-water (Figure 19). The soil-water is enriched in ^{2}H at the base of the basin, and decreases to a constant value at a depth of about 2 m. For Bay B, the ^{2}H composition of soil-water at depths below 2 m is enriched by at least 10%, compared to that for the soil-water in the other bays at the same depth. This may or may not have resulted from contamination during sampling. It would require ~12% contamination of the soilwater by basin water to give the results for Bay B and a lesser amount (~4%) for the other bays. However, contamination of this magnitude is unlikely and hence, the results probably indicate a greater rate of leakage at site B compared to that at the other sites. This conclusion is supported by data from groundwater sampling under the basin (Figure 21).

Under saturated conditions, the chloride concentration of the soil-water should be the same as the chloride concentration of the groundwater at the same depth, providing sufficient time has elapsed to allow equilibration between the smaller pores in the interstitial clay layers and the larger pores of the soil. Similarly, the ^{2}H composition of the soil-water should be the same as that in the groundwater. For the sampling period from August 1997 to February 1999, the [Cl] for groundwater from bore samples beneath bays A and C is relatively consistent (Figure 20).
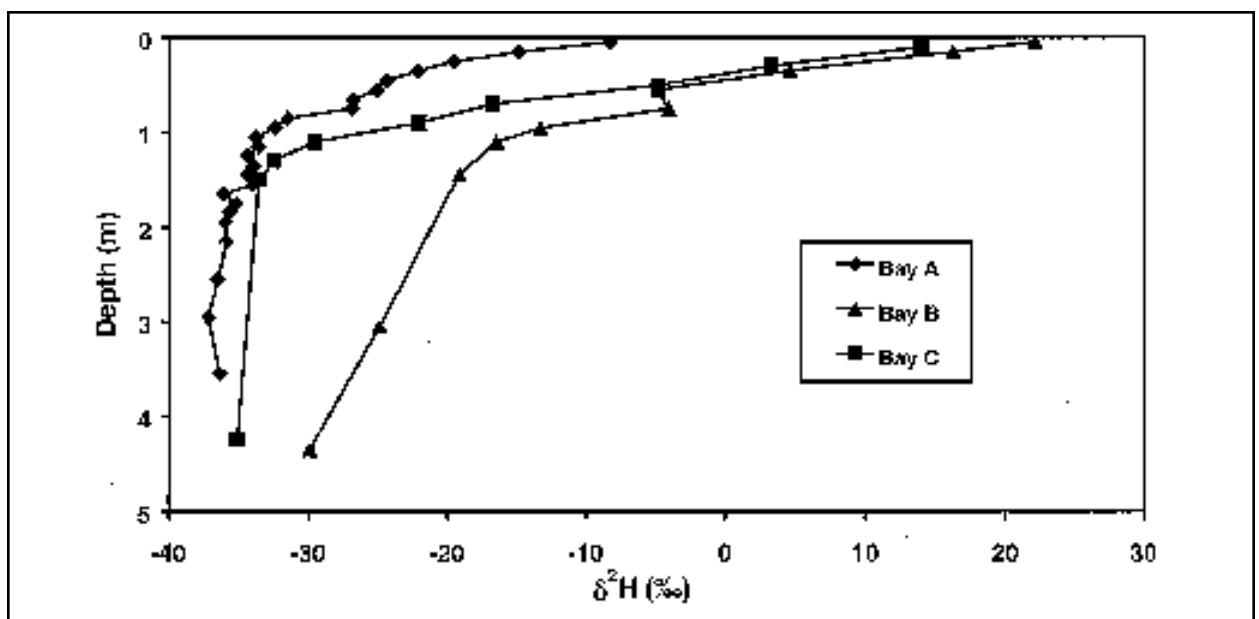
Figure 19   Depth profiles for ^{2}H composition of soil-water beneath the bays in the Girgarre Basin.

Soilwater in the top metre of soil beneath the basin is enriched in ^{2}H (indicating a component of seepage from the basin) compared to that at greater depths. For the soil under Bay B, the enriched ^{2}H signature extends to at least a depth of 4 m indicating seepage to greater depths at that site.
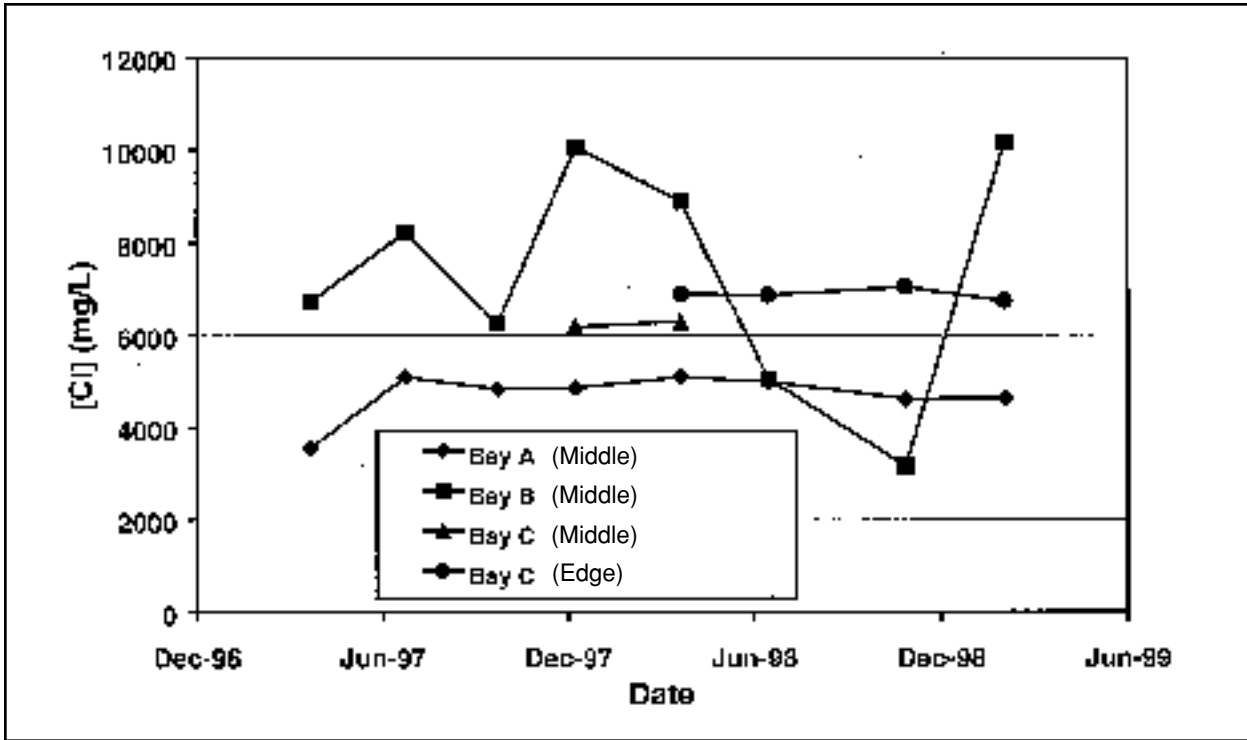
Figure 20   Temporal changes in groundwater chloride concentration beneath the bays at the Girgarre Basin.

The chloride concentration of groundwater beneath Bays A and C (~4 m deep) is temporally constant for the study period while that for bay B fluctuates by more than 6,000 mg/L.   This suggests more movement of groundwater at that site than at the other sites. The movement may be associated with the interaction of leakage water with that present in rising and falling water tables.

The [Cl] concentration from bores beneath Bay C is only slightly greater than that beneath Bay A, while that for Bay B ranges from ~4,000 mg/L to ~ 10,000 mg/L (i.e. from significantly less than to significantly greater than the other sites). This suggests that [Cl] in the groundwater under Bay B is not as homogenous as that for the other two bays, and that seasonal fluctuations in the groundwater is causing groundwater of different [Cl] to move. Overall, a higher rate of leakage at the Bay B site ([Cl]$_{Bay B}$ = 10,000-20,000 mg/L) is probably responsible for the groundwater not being homogeneous.

The ^{2}H composition of groundwater shows similar fluctuations. In this case, groundwater beneath Bay B is enriched by ~10% compared to the other sites, this suggests that a component of leakage from the basin has reached the piezometer screen depth in Bay B (Figure 21).
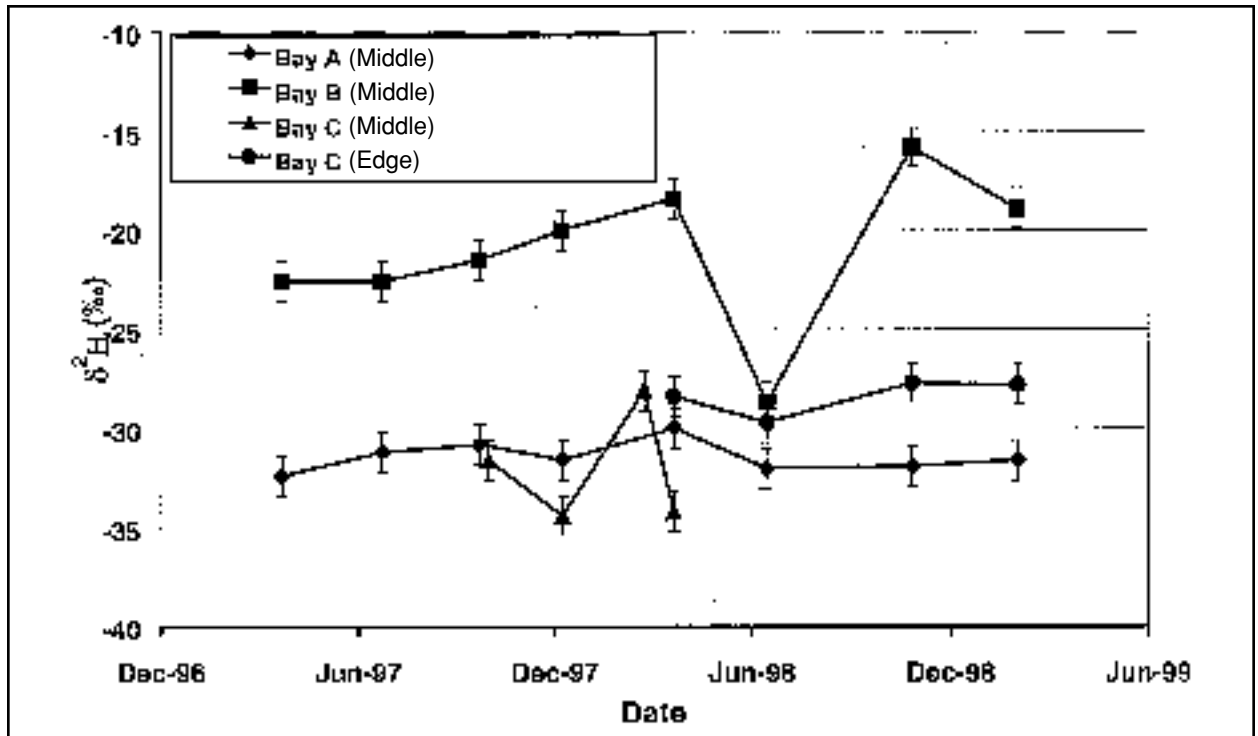
Figure 21   Changes in ^2H composition of groundwater beneath the bays at the Girgarre Basin the basin.

The temporal changes for ^2H are similar to those observed for the groundwater chloride concentration. The enriched composition under Bay B is consistent with higher leakage rates at that site.

5.8 Estimation of Leakage for the Girgarre Basin

Several approaches have been used to estimate leakage from the Girgarre Basin. "Whole of bay" estimates have been made using the chloride mass balance approach while tracer techniques and seepage meters have been used for "point estimates" of leakage. Results from these studies are as follows.

5.8.1 Whole of basin estimates

Chloride concentration was modelled for the three bays in the Girgarre Basin for the period August 1987 to June 1998 using the spreadsheet model described earlier. Data for the model consisted of pan evaporation and rainfall measurements (from monitoring at Girgarre and at Shepparton) and chloride data for the input water to the basin. Input to the basin was calculated from the water balance of the basin and leakage was chosen as a variable in the modelling exercise.

The theoretical chloride values were compared to the measured data (either calculated from EC data, pre 1995 or measured directly, post 1997) for different combinations of leakage rate for the basins. The best-fit estimate for leakage was a constant leakage rate of 0.7 mm/d for Bay C and an initial leakage rate for Bays A and B of 1.5 mm/d from 1987 to early 1993 reducing to 0.7 mm/d from then to present (Figure 22).
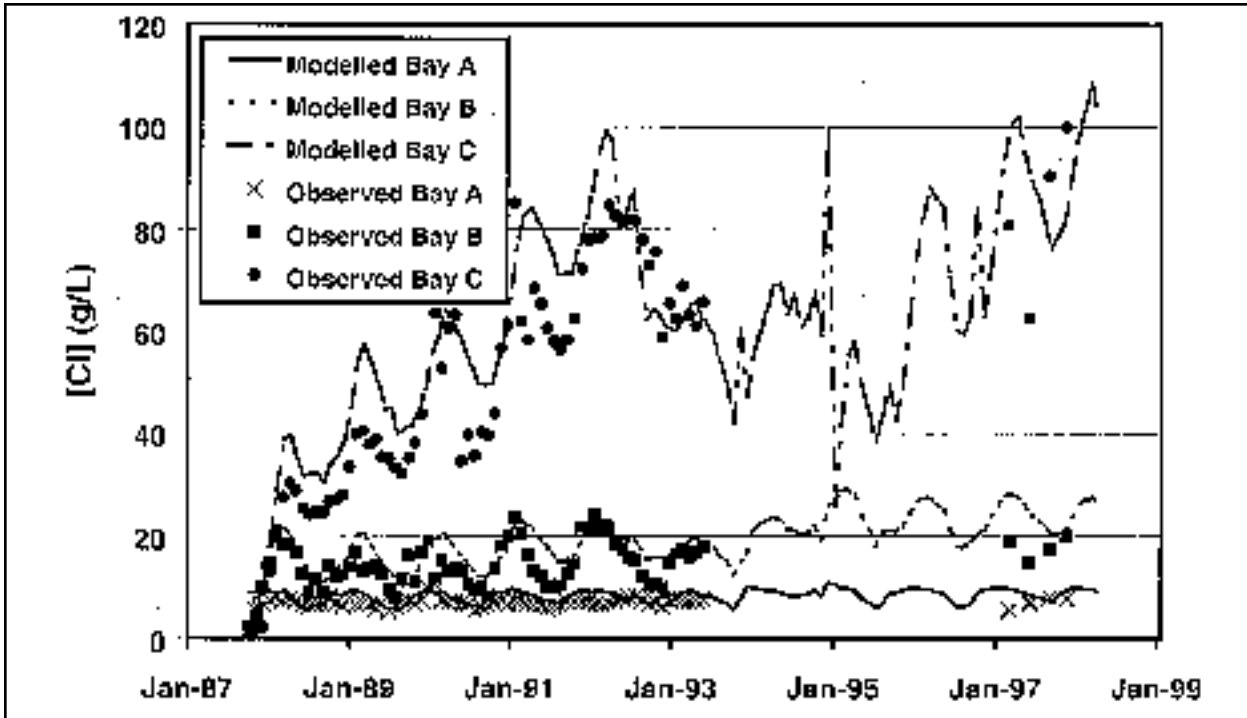
Figure 22.   Measured and modelled changes in the chloride concentrations for water in bays A-C at the Girgarre Basin.

Using a chloride mass balance approach allows accurate estimation of leakage for the three bays in the Girgarre Basin. Leakage for Bay C (0.7 mm/d) was half that estimated for the other two bays (1.5 mm/d) until early 1993. Since then, all basins have had similar leakage rates of ~0.7 mm/d.

The mean leakage rate for the basin for the period 1987-1993 is 1.4 mm/d and reduces by 30% for the period 1993-1998 (i.e. ~1 mm/d over the life of the basin). The leakage rate for the period 1987-1993 was estimated using water balance calculations to be 1.8 mm/d (Goulburn-Murray Water and Sinclair Knight Merz, 1995), ~30 % higher than that estimated in this study for the same period.

Modelled cumulative pumped input to the basin was determined for the life of the basin, assuming the best-fit leakage scenario (above) and assuming constant leakage rate throughout the life of the basin (Figure 23). Clearly, there has been a large reduction in leakage for the two larger bays of the basin since early 1993. The reason for this is not clear although, at that time, a problem with monitoring of the basin resulted in the bays approaching dryness. Why (or if) this should lead to a reduction in leakage for the next 5 years to values close to that in Bay C is not known.

From 1987 to present, ~23,500 t of chloride (~ 41,200 t of salt) has been pumped into the basin. In this case, the difference in leakage estimate using a water balance approach, and that using changes in bay salinity, is about 25%. The accuracy in the water balance measurement reflects the care and effort taken during the study. However, it is possible to foresee situations

where the reliability of data used in the water balance may not be as accurate and larger errors may arise. The likelihood of large errors is increased for basins with low leakage. Using a combined chloride/water balance approach will uncover any such discrepancies.
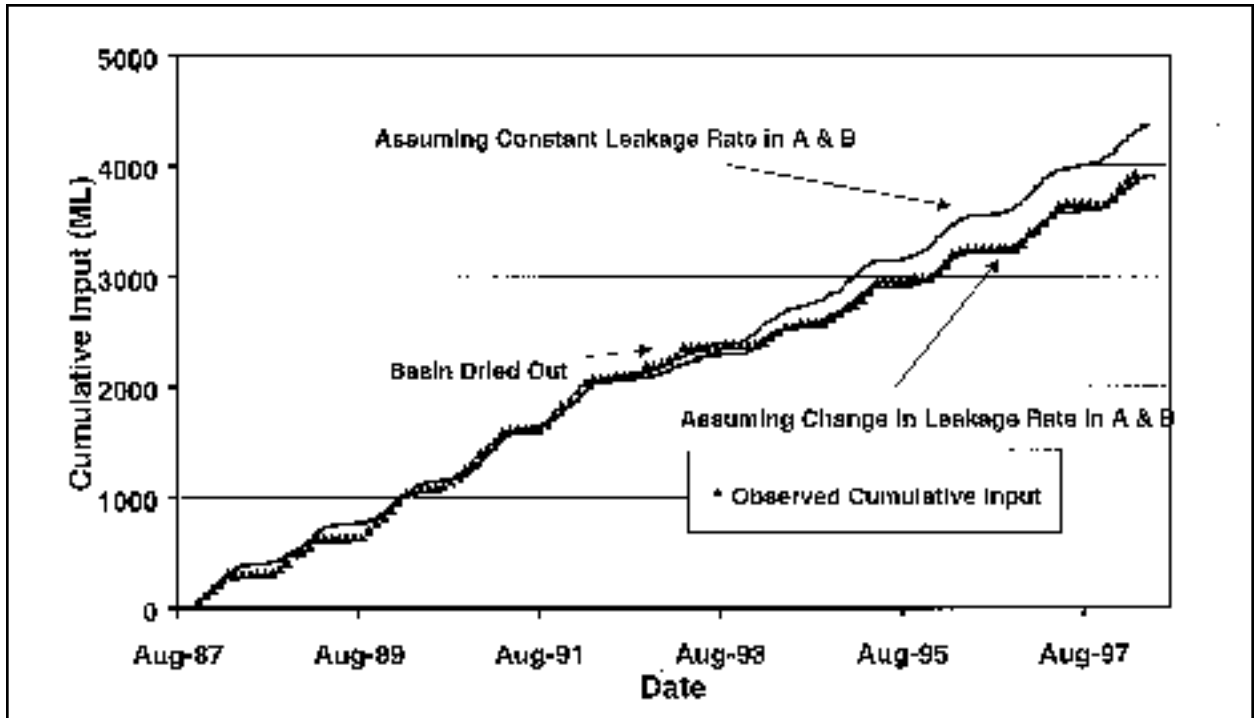
Figure 23.   Observed and modelled cumulative water input to the Girgarre Basin.

The observed input to the Girgarre Basin decreases slightly post 1993 as a result of the reduction in leakage from Bays A and B. It is not clear why this occurs although the timing coincides with a period when the basins were nearly dry and basin salinity increased markedly.

5.8.2 Point estimates for leakage rate

Point estimates for leakage were determined, for the period of operation of the basin until the time of core sampling (1987-1997), by using the enriched [Cl] and ^{2}H composition as a tracer of the movement of leakage, and for intervals of a few months during 1998 using seepage meters.

The methodology for estimating leakage from a basin using natural abundance tracers such as chloride or ^{2}H was described earlier in this report. It assumes that:

1. Leakage water has a different tracer concentration than the water into which the leakage is moving.

2. The depth of leakage is the same as the depth of tracer movement and that depth is easily identified.

3. There is no contamination during sampling

4. Leakage is via piston-flow.

5. There is no lateral movement of leakage.

For the studies in the Girgarre Basin, Assumptions 1 and 2 are justified, particularly for Bays B and C, but there are potential problems with the remaining three assumptions. There is considerable potential for contamination of soil samples as a result of seepage from the basin water when coring through the base of the basin. This would lead to an over-estimation in leakage rate. Leakage may take place via preferential pathways or may move laterally upon entering the saturated zone. This would lead to under-estimation of leakage rate.

The soil water chloride concentration and ^{2}H composition profiles for each of the basins are highest immediately below the soil surface. Except for the ^{2}H composition profile in Bay B, the [Cl] concentration and ^{2}H composition of the soil-water (and groundwater) are near pre-basin levels at a depth of 1.0 to 2 m below the basin (Figure 18 and Figure 19). The shapes of the profiles suggest that either diffusion or mixing is occurring between the leakage from the basin and the existing groundwater. The seasonal changes in water table as a result of groundwater pumping would tend to enhance the mixing process.

Assuming that leakage has reached a depth half way along the zone of diffusion/mixing, then ~400 to 800 mm of water (the cumulative amount of water in the soil profile to a depth of 1 and 2 m respectively) has leaked from the basin since it was commissioned. Hence, using this approach, the leakage for each of these bays since mid 1988 is estimated to be about 400 - 800 mm in 9 years. This equates to about 44-90 mm/yr (0.12 to 0.24 mm/d) assuming a constant leakage rate during that time. If we consider the results from the ^{2}H composition profile in Bay B as valid (and not the result of contamination during sampling), the depth of leakage at that site may be double that seen at the other sites (i.e. ~0.5 mm/d).

The mean rate of leakage for the bays during this period, as determined from the whole of basin measurements, is ~3-10 times the point estimates. This may be because the spatial heterogeneity in leakage for the basin is not reflected in the limited number of sites sampled. Alternatively, one or more of the assumptions is not valid.

Seepage measurements for sites with the sludge present ranged from 0.0 to 2.6 mm/d (average 0.5 mm/d). The mean value for the seepage measurements (0.5 mm/d) agrees with the estimate for whole of basin leakage, using the chloride mass balance approach for the corresponding time period (0.7 mm/d). However, given the limited number of measurements, this is more likely the result of good fortune than a statistically sound observation. Nevertheless, the mean estimate for leakage using seepage measurements is 2–4 times the mean estimate using tracer techniques. In addition, there is a considerably larger range when using the seepage measurement. It is possible, therefore, that the assumptions that result in under-estimation of leakage rate using tracer techniques are not valid in this situation, and that preferential flow and/or lateral flow from the Girgarre basin are significant and can not be ignored.

5.9
Preferential Flow from the Basin

Results presented in this chapter clearly show that leakage from the Girgarre basin does not proceed via a piston flow process in a well-defined plume beneath the basin. Seepage through the base of the basin, as measured via the seepage meters, ranged from 0 to close to 3 mm/d. Beneath the basin, the evidence for piston type flow extends to a depth of 0.5 to 1.0 m for Bays A and B and less for Bay C. At these depths, the soilwater salinity is similar to that of the water in the basin. In fact, this is for areas with permanent water cover. In areas of periodic water coverage, the differences are even greater (Figure 19).

Once the leakage water reaches the zone at which the water table fluctuates (0.5 – 2 m beneath basin base), diffusion and lateral movement of the leakage results in large temporal and spatial differences in water salinity. The extent of this is such that leakage measurements determined by tracer techniques only account for 25 – 50% of leakage from the basin (as determined by whole of basin studies).

The remaining leakage clearly must diffuse laterally or leak vertically. Results from EM studies clearly show that much of the soil matrix is bypassed by leakage from the basin. Similar conclusions for secondary porosity were made by Ife (1984) who found that vertical conductivities were extremely variable and, for some sites, much higher than expected given the soil classification. The amount of soil bypassed is difficult to determine; from an evaluation of the EM data, it is likely to represent 50-80% of the total soil matrix for the soil matrix from 1-15 m beneath the basin. This is based on the overall fraction of EM data where there is an observed increase during the period of monitoring. We assume piston flow for the first metre of soil immediately beneath the basin in the following calculations.

The amount of pore space involved in the leakage process is therefore equal to the volumetric water content of the soil to a depth of ~15 m, multiplied by the fraction of the soil involved in the leakage process (unity for 0-1 m and 0.2-0.5 for 1-15 m). (estimated fraction of preferential flow). Using an estimate of 0.4 for volumetric water content, we suggest that approximately 1.5-3 m of water has leaked vertically from the basin from the time filling commenced until the time when the salinity of groundwater was first noticed (6-10 years). Approximately 0.4 m of this leaked water is still within a metre of the base of the basin. In the first 6-10 years after filling commenced, approximately 3-4 m of water has leaked from the basin. In other words, depending on the values used in these calculations, it is possible that at little as 30% or, at most, 100% of the leakage could be deep vertical leakage.

When all of these factors are considered, it makes definition of the plume difficult. What is clear is that, due to preferential lateral and vertical flow, there will be a component of leakage extending well beyond the basin and into the groundwater much sooner than would be expected if flow were predominantly piston flow. This will result in a gradual increase in the salinity of groundwater beside, and in the shoestring sand, beneath the basin.

In summary, we suggest that the leakage for the Girgarre Basin:

- Has an average rate of ~1 mm/d over the 10+ year life of the basin.

- Has halved in rate for the second half of its operation.

- Has a spatial variability of several mm/d within bays and from bay to bay (at least spanning the range from 0 to 3 mm/d).

- Is "infiltration limited". There is an unsaturated zone beneath the basin for most of the year.

- Moves by piston flow through the unsaturated zone and into the top metre or two of the groundwater beneath the basin.

- Has mostly remained within a few metres of the water table and spreads laterally within this zone possibly enhanced by water table fluctuations.

- Has a vertical component as little as 30 % or, in fact, most of the total leakage. Vertical leakage by-passes most of the soil matrix beneath the basin and then progresses in the direction of the groundwater pumps.

- The shallow open interception drain does not/can not intercept the lateral leakage occuring at greater than 2 m depth.

5.10
Summary of Leakage Rate and Process for the Girgarre Basin

6. Result from On-farm Basins in the MIA

6.1
Nehme Basin

6.1.1 Initial filling

The initial vertical leakage of water from the basin was extremely rapid; water entering the 7 m deep piezometer only 18 days after basin filling commenced. This can be equated to a mean leakage rate over that period of about 77 mm/day, if the volumetric soil water content moved from field capacity (~0.22), to saturation (~0.42). It is probable, however, that this is an upper estimate, because preferential flow will result in part of the soil matrix being bypassed during the initial stage of leakage from the basin. Nevertheless, the rate of leakage is likely to be very high rate and be attributed to the dryness and well developed structural properties of the soil before basin filling.

6.1.2 Leakage by water and salt balance

Water balance estimates were started a month after basin filling, monthly estimates are given in Figure 24. These values vary quite markedly from month to month due to errors in the water balance method, to give a clearer indication the results can be lumped into 4 time periods.
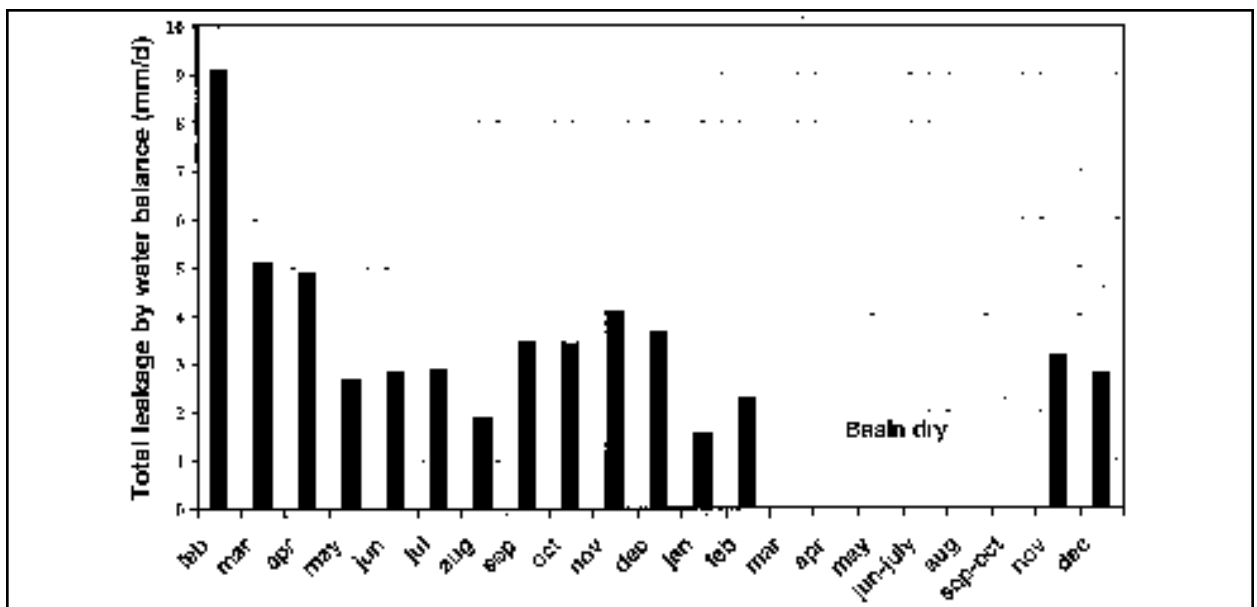

Figure 24.   Monthly leakage estimates for Nehme basin (1997-98) using the water balance method

Leakage from the Nehme Basin was high for the first month following filling but quickly reached a steady-state leakage rate of ~3 mm/d after a few months use. After a few months when the basin was dry, the basin was refilled and continued to leak at ~3 mm/d.

For the first 3 months February-April the mean leakage rate was about 6.1 mm/d, for the subsequent 6 months (May-October) the mean leakage rate was 2.9 mm/d. In the following 4 months (November-February) the mean leakage rate was 3.0 mm/d, at which point the ponds dried out for 6 months. In the 2 months (November-December) after subsequent refilling, the leakage rate did not appear to have changed much, at 2.6 mm/d.

These results can be compared with those obtained by salt balances over the same periods (Figure 25). For the first two periods (when the basins are full) they give reasonably good agreement; however, in the drying phases and filling phases, there is a greater difference between the measurements. In the drying phase this is probably due to an inappropriate assessment of the infiltrating water salinity; when drying occurs, the basin water salinity increases rapidly, and also the assessment of the salt stored in or on the topsoil, when the basin dries out. In the filling phase, there is difficulty in assessing the initial salt stored at the soil surface, which can either have accumulated over the dry period or, if it has rained, been leached.
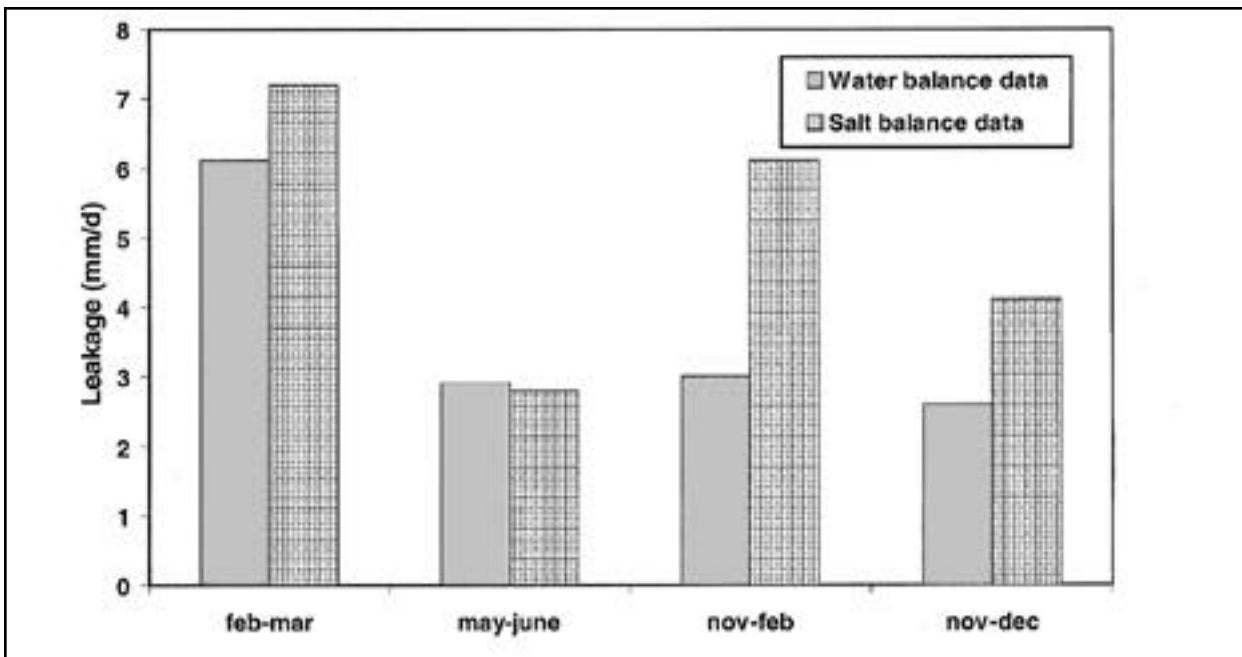
Figure 25.   Comparison between water balance and salt balance calculations for leakage estimation during 1997 at Nehme Basin.

Overall, these results point to initially very high leakage rates which have then stabilised around 3 mm/day. These rates compare with infiltration tests on similar soils by McIntyre *et al* (1982) who measured 14-31 mm/day in early ponding to 0.15 - 0.5 mm/day 70 days later. Infiltration rates under rice in the MIA have also been measured by van der Lely and Talsma (1977) at 0.3 and 1.7 mm/day, for transitional red brown earths and clay soils respectively. Both these workers, however, used water of low salinity (about 0.2 dS/m), whereas the water entering the basin was about 11 dS/m.

When conducting water balances for a basin to assess leakage it is important that measurements are as accurate as possible. In order to compute the water balance, 5 separate measurements are required. If each of these measurements is in 10% error, without any compensation between errors, the derived leakage rate can easily be in error by 20-80%.

A key difficulty arises from assessing the change in volume of ponds. To minimise errors resulting from this, it is important to derive a volume: depth relationship for the basin. When allowing for evaporation it is important to know the water area contributing; thus a depth: area relationship is also required. In estimating evaporation, an appropriate pan factor is required. At Nehme basin, the two pans in the water were found to be very close to Class A pan evaporation, 1.04 and 1.06, whereas the two pans on the bank were considerably higher, 1.22 and 1.32 (Figure 26). These results agree with the assessment of Morton (1986) that small water bodies should evaporate at about the rate of a Class A pan, although they may actually evaporate at a higher rate if the edge effects, as measured by the pans on the bank, are significant.
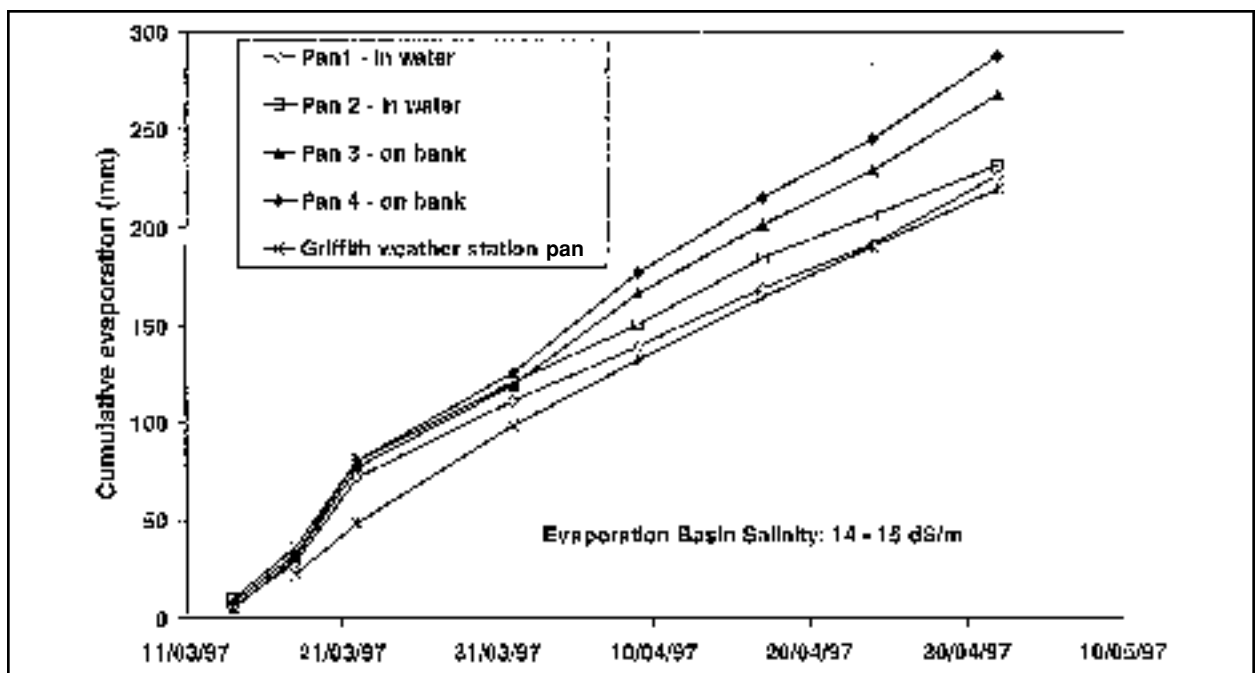
Figure 26. Comparison of evaporation from pans in and around Nehme basin with meteorological measurement of pan evaporation at Griffith.

• **Concentration factors**

Tracking basin water salinity, compared with the drainage water salinity, provides another gross measure of leakage. This works best over long periods, when there is little change in storage. For the Nehme basin, a period of 389 days has been used when the ponds did not dry out. The salinity of the pumped and basin waters have been regularly monitored over that time (Figure 27).

The average inlet salinity for that period was 12.9 dS/m, and for Bays 1 and 2 was 15.3 and 21.9 dS/m respectively. Dividing the bay salinity by the inlet salinity gives concentration factors of 1.2 and 1.7 for Bays 1 and 2 respectively. The pan evaporation from the Griffith weather station for this period was 1953 mm, and rainfall 332 mm, giving a net evaporation of 1621 mm. Thus, dividing the net evaporation by the number of days gives an average net daily evaporation of 4.2 mm/d. Dividing this by the concentration factor gives the leakage rate over that time; 4.2 / 1.2 = 3.5 mm/d for Bay 1. For Bay 2 the concentration factor is 1.7 resulting in a leakage rate of 2.5 mm/d, an average of both bays giving 3 mm/d. This very simple procedure can give a relatively accurate assessment of leakage rates. In overall terms for basins in the Riverine Plain, this procedure can be used as a check of probable leakage rates, and is a fundamental method in tracking long term changes in basin behaviour. Also importantly, as discussed earlier with salinity data, basin leakage can be modelled.
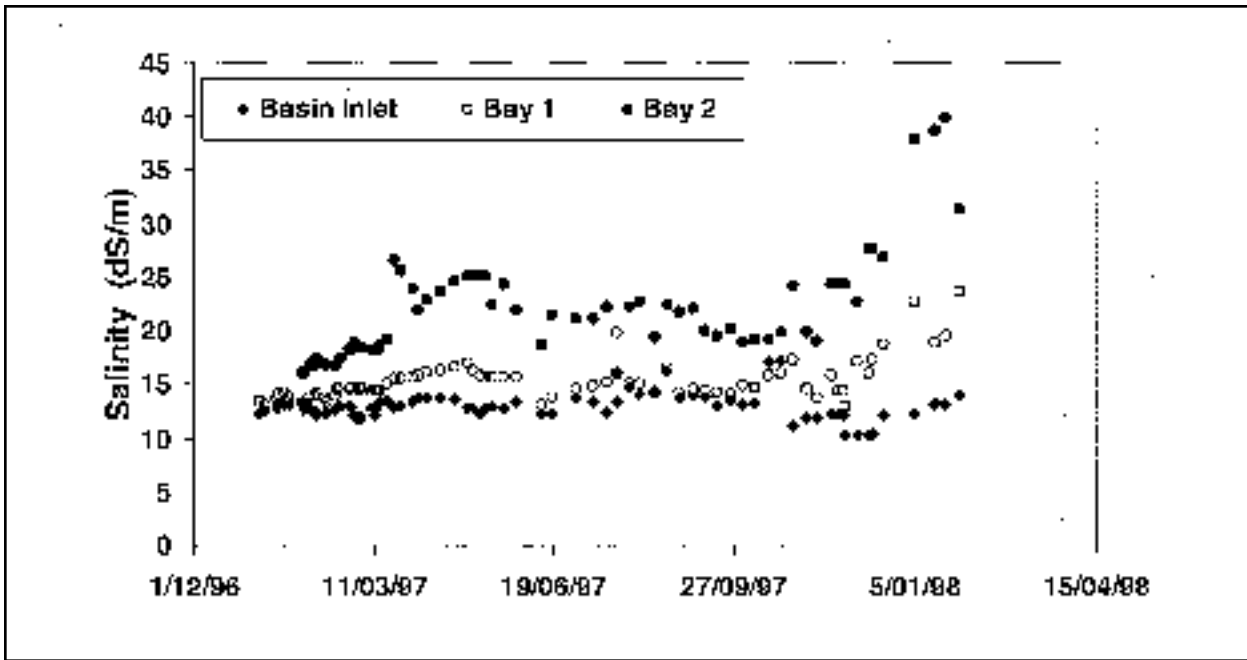
Figure 27.   Inlet water salinity and basin water salinity for Nehme basin.

6.1.3 Spatial distribution of leakage

• **Seepage meters**

Together with the water balance studies, eleven seepage meters were used for point measurement of leakage during March and April, the 3rd and 4th months after filling. During this period the water balance estimate for total leakage was ~5 mm/d. The seepage meters gave readings in the order of 0.1-1.2 mm/day with many negative values also. These readings were conducted on a weekly basis; as such the time interval was inadequate and

associated errors large. A further trial with the seepage meters was conducted over a 140 day period, starting at the end of October. All the seepage meters were filled and left for the whole period. This yielded results from 0.4-5.2 mm/d (average 1.3 mm/d, standard deviation 1.6 mm/d). During the same period, the water balance estimate for leakage was 3.0 mm/d. Soil particle size analysis at various positions indicated that the high leakage seepage meter had about 10% less clay, 40% compared to 50% at the other positions, and that this was accompanied by a 10% increase in sand. Whether this small reduction in clay content would be responsible for an order of magnitude increase in leakage is uncertain.

The seepage meter results indicate that point measurements tend to underestimate leakage from a basin; there is a wide variability in leakage across a basin, with tenfold differences possible. This shows that it is important to understand heterogeneity of soils when siting basins in order to avoid localised areas of very high leakage. This is best done by an electromagnetic survey of the site e.g. using the Geonics EM31 or EM38, to assess the variability of the site.

- **Electromagnetic survey**

An EM38 survey was conducted on the basin during a dry period after about 17 months of ponded water. The results of the survey are plotted as apparent conductivity contours in Figure 28.
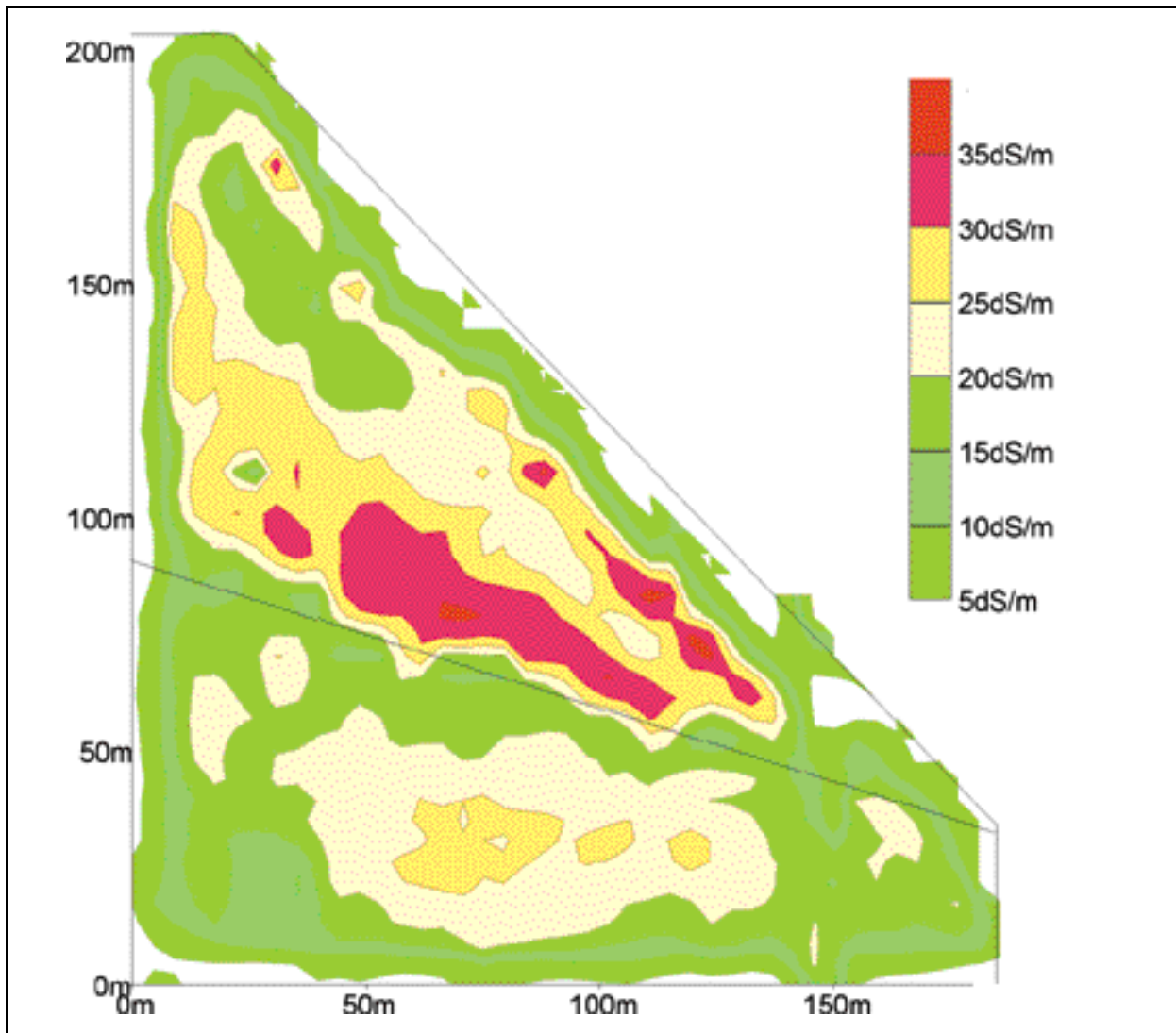
Figure 28.   Apparent conductivity (salinity) of soil below Nehme basin by EM38 survey

This EM38 survey found large variation in apparent conductivity across the basin, up to a sixfold difference from highest to lowest conductivity areas. This could be reflecting variation in leakage across the basin, with lower apparent conductivity indicating higher leakage. The lower conductivity areas are around the periphery of the basin, and may be due to greater leakage in these areas, or merely lower salinity in the bank and margins due to less ponding of water. However, there is large variation even in the base of the basin indicating that leakage is non-uniform. It would have been useful to undertake EM surveys before basin construction to assess the variability in soils.

- **Soil sampling**

Leakage can also be assessed by the change in soil water quality below the basin. After some period the soil water below the basin will take on the characteristics of the infiltrating basin water. The rate at which this occurs depends upon the rate of salt exchange between the pores and the soil solution. However, the general movement of water can be assessed by consideration of the differential between the upper and lower soil water quality. Figure 29 shows soil water quality below the basin after 377 days of ponding.
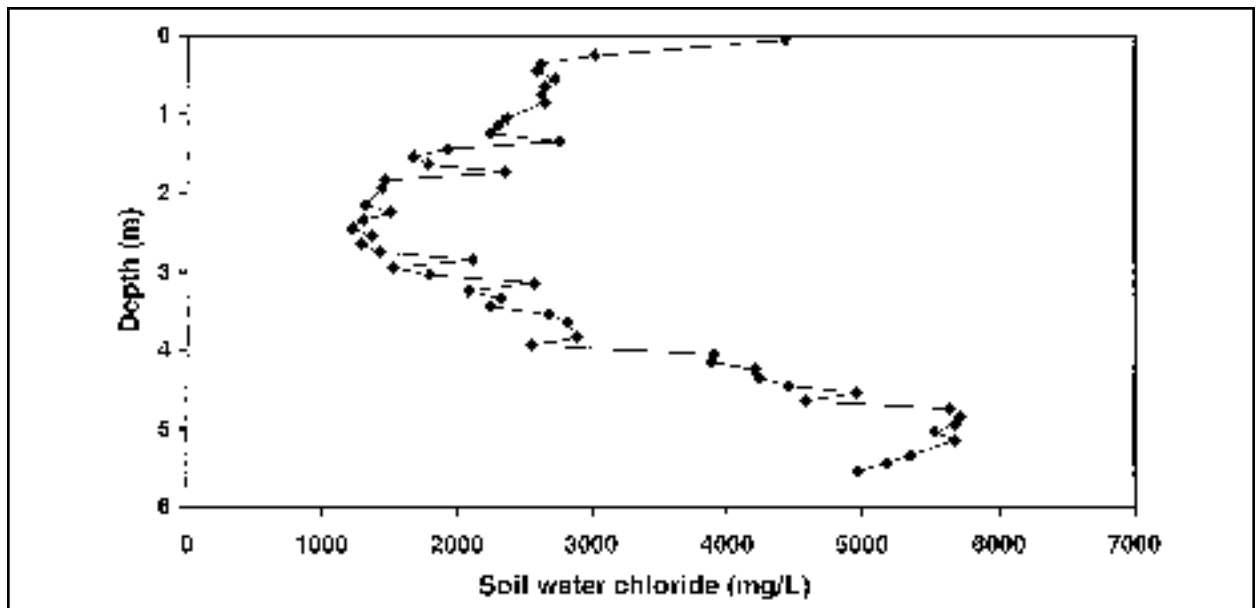
Figure 29.   Soil water chloride below Bay 2,Nehme basin,in March 98

The general soilwater chloride profile under the bay indicates high soilwater chloride at the base of the basin, with much lower values between depths of ~0.3 and ~4 m. At depths greater than ~4 m, the soilwater chloride concentration increases again. The mean salinity of the water in Bay 2 up until sampling was ~19 dS/m ([Cl] @ 4,400 mg/L). However, much of the leakage (particularly vertical leakage) occurred during the first few weeks, after filling the basin when the soil in the unsaturated zone became saturated. During this time, the mean salinity of water in Bay 2 was ~10 dS/m m ([Cl] @ 2,000 mg/L). This is approximately the salinity of soilwater for the depth range 1.5 – 4 m beneath the basin floor.

Soilwater deeper than ~4 m is more saline than basin water. Hence, it appears likely that, in this case, the leakage water from the basin ([Cl]soilwater < 4,400 mg/L) is less saline than the soilwater immediately beneath the basin (soilwater chloride > 5,000 mg/L), and that leakage from the basin is freshening the shallow soilwater. If this is the case, then it is probable that, at least for this site, most of the deep vertical leakage resulted from leakage during the filling of the basin when there was an unsaturated zone beneath the basin. Whether or not this is the case for the rest of the basin is not known.

6.1.4 Lateral leakage

During the water balance periods, the flows from the interception drains around the basin were also monitored. Figure 30 shows the total leakage estimate split into that intercepted in the drains (termed lateral leakage) and the remainder (termed vertical leakage). However the interception drains will not intercept all the lateral leakage; thus, this will be an underestimate of lateral seepage, and hence an overestimate of vertical leakage.
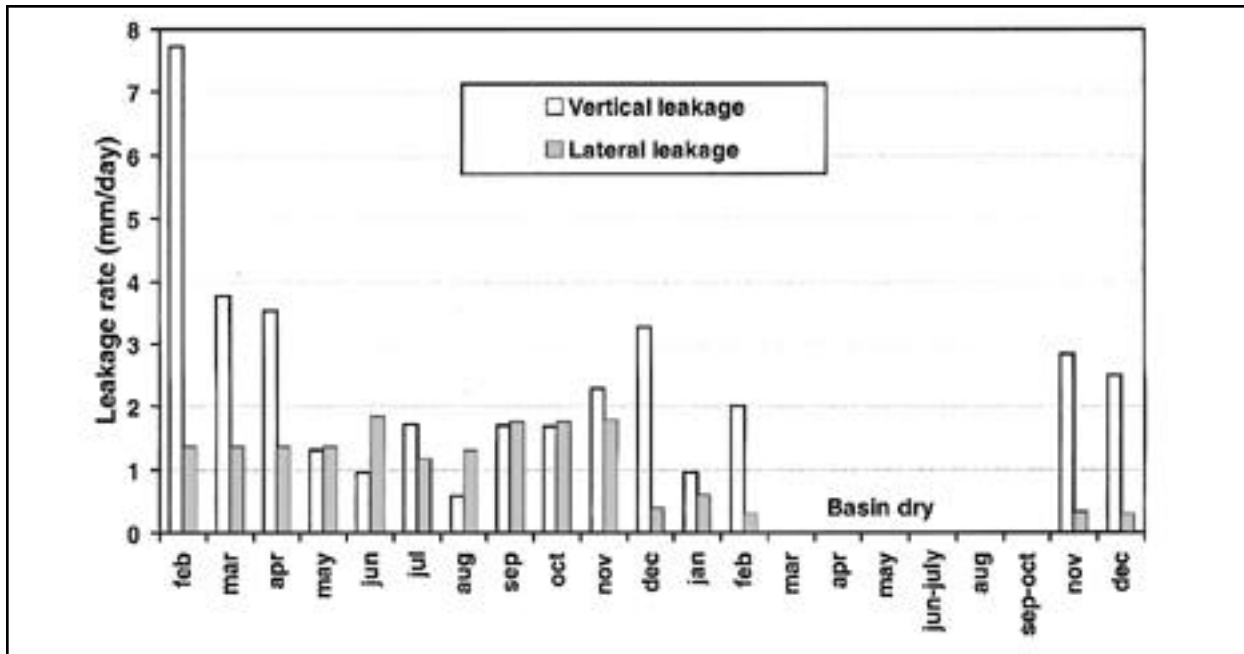

Figure 30.   Monthly estimates of vertical and lateral leakage for Nehme basin

In the initial 3 months, the component of total leakage that was collected in the interception drains was about 23%; this was the period when the groundwater mound was developing below the basin (Figure 31), when vertical leakage was dominating. In the subsequent 6 month period (May-October), the total leakage decreased but lateral leakage remained at similar levels; this accounted for 54% of the total leakage. In the drying down phase of the subsequent 4 months (November-February), the total leakage remained similar; that intercepted in the drains was reduced to about 24 % of the total. This can be attributed to the reduction in basin water level and hence reduced heads for lateral flow (Figures 32 and 33).

These results indicate that a significant proportion (~25%), of leakage is recycled in the interception drains, and that the subsurface pipe drainage system in the farm would also have the potential to recycle a significant proportion of the basin leakage. In the refilling phase of November-December, the total leakage was similar to the drying phase; however, the lateral leakage has dropped to only 12% of the total. This may be attributed to greater vertical leakage again, as in the first filling as the unsaturated zone becomes saturated and the groundwater mound redevelops.
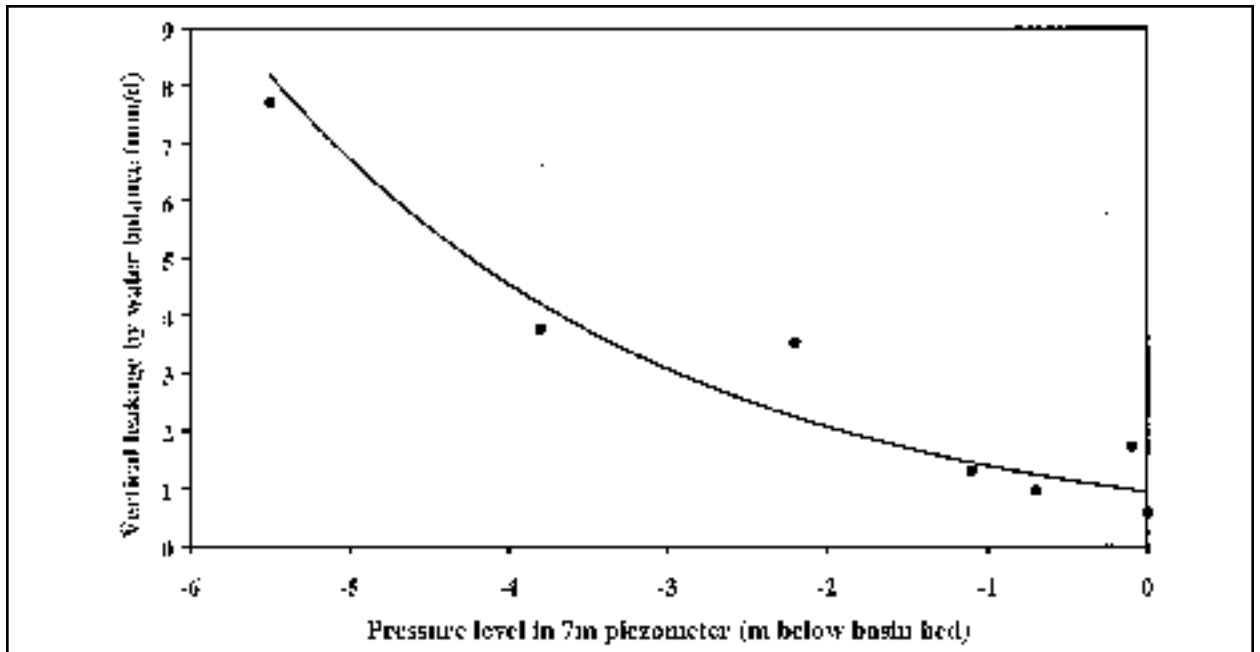
Figure 31   Relationship between effective leakage (total leakage less that measured in interception drains) and piezometer levels below the basin.

As the unsaturated zone beneath the basin approaches saturation and the water table rises, vertical leakage from the basin decreases.
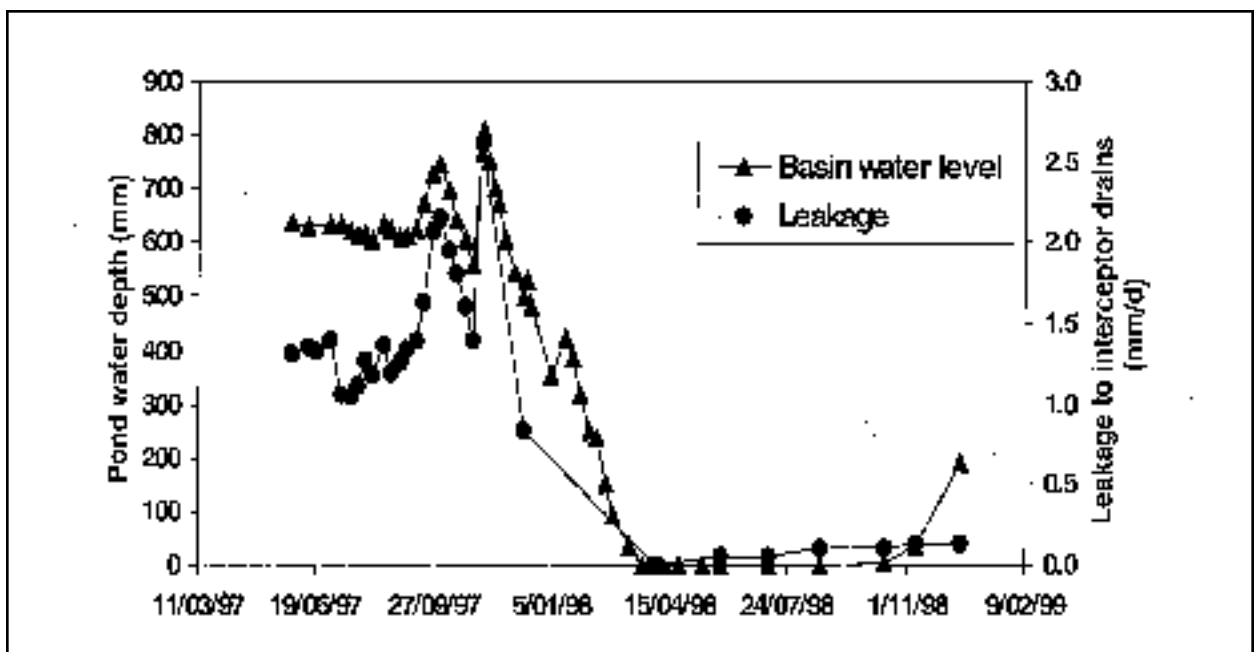
Figure 32   Temporal variation in the flow in interception drains for Nehme Basin

The interception drain ceases flowing when the basin is empty. This suggests that the drain is intercepting leakage primarily from the basin and not groundwater.
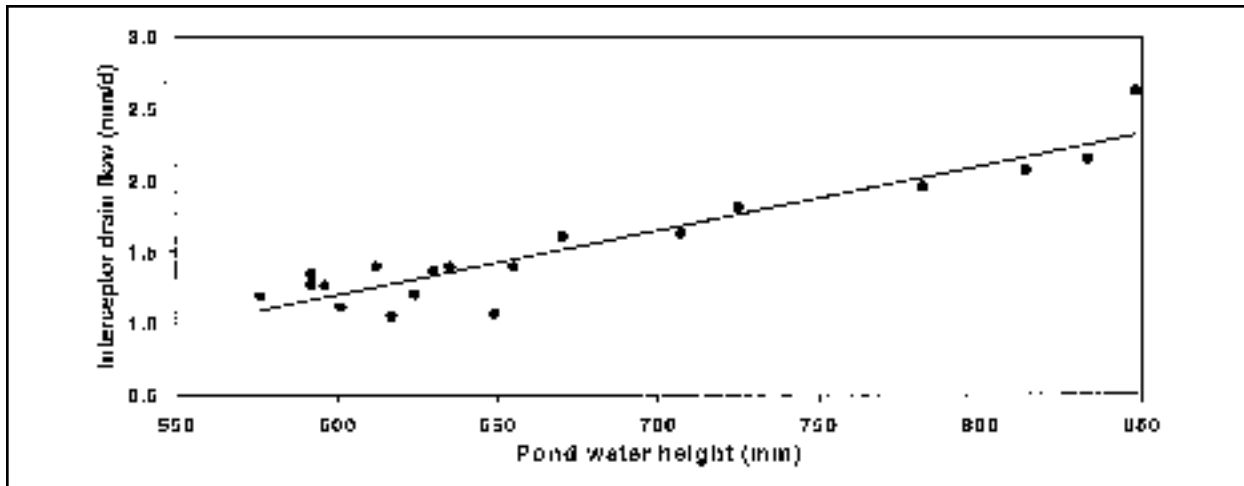
Figure 33.   Measurements of the flow in interception drains and water level in Bay 2 at Nehme Basin.

Flow to the interception drain is primarily determined by the height of water in the basin.

- **Isotope analysis for determining leakage plume**

The leakage plume from the basin has been monitored since the soil below the basin became saturated. This was done by measuring changes in the salinity and isotopic signature of water in piezometers sited beneath the basin, in the interception drain around the basin, and for shallow groundwater in a transect away from the basin. The isotopic signature of the water in the basin is considerably enriched compared to that of the groundwater, while there is less of a difference in the salinity of the basin and the groundwater. In fact, for much of the time that the basin has been monitored, the basin water (and hence leakage water) has been less saline than the groundwater beneath the basin. For this reason, we have predominantly used the results of isotopic analyses in the following interpretation.

Prior to the filling of the basin, the soil beneath the basin was unsaturated to a depth of at least 7 m. Immediately after filling commenced, piezometers beneath the basin indicated that the unsaturated zone beneath the basin the basin was becoming saturated. The process did not involve wetting from the bottom up or from the top down but rather a change in saturation status throughout the profile. After a period of two months, the zone beneath the basin was saturated and a permanent groundwater table established.

After the soil became saturated beneath the basin, the leakage rate from the basin decreased significantly to ~3 mm/d. During that time, the deuterium composition of the water increased from –15 to +25 and from 0 to +60% V-SMOW in the first and second bays respectively. For the corresponding period, the deuterium composition of the water increased from –22 to –1 and from –14 to –1% V-SMOW for the interception drains along side the first and second bay respectively (Figure 34). Clearly, this demonstrates that a component of leakage is collected by the interception drain, and that there is a lag time for the leakage to reach the drain.
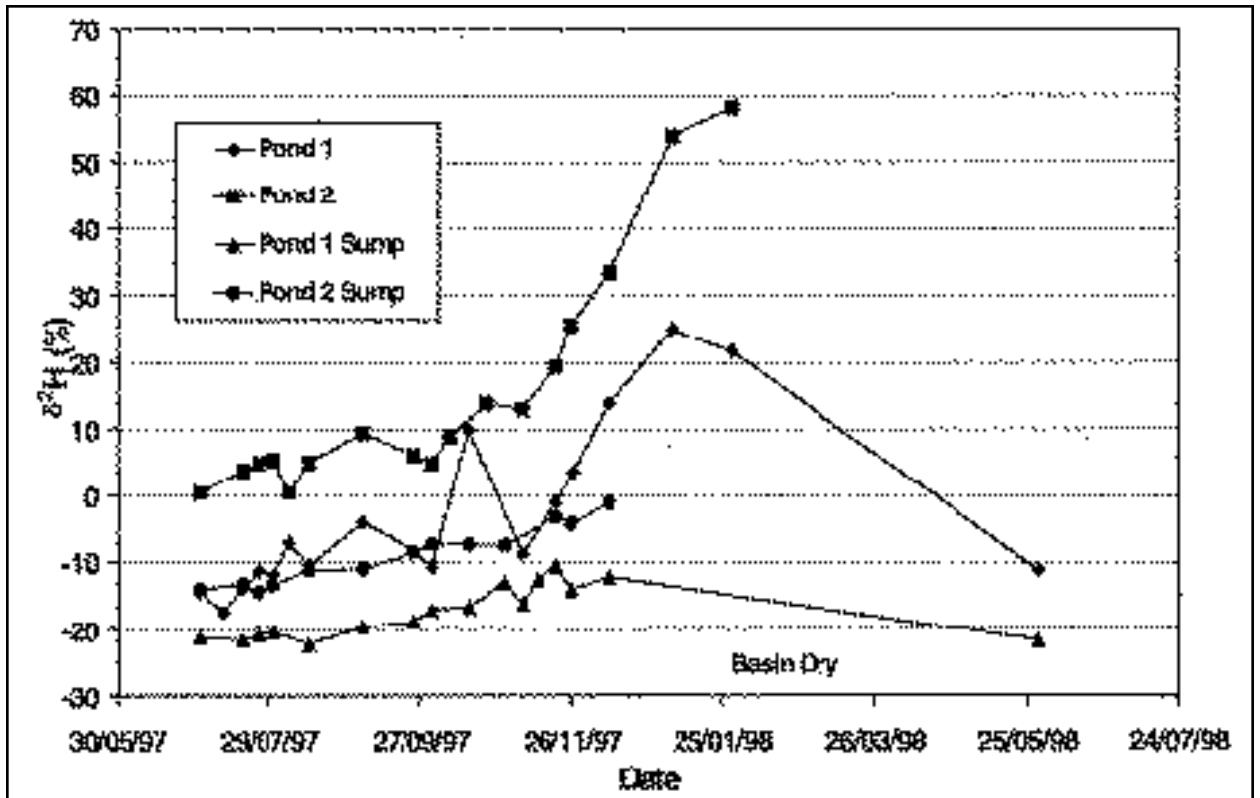
Figure 34.   Changes in deuterium composition of interception drain water at the Nehme basin.

The ^2H signature of water in the interception drains provides a good indication of the amount of basin water intercepted because of the significant differential between the ^2H composition of existing soilwater and water in the basin. A small component of the leakage water is evident in the intercepted water, clearly mixing with existing soilwater.

During the same period, there was very little change in the salinity of the water in the interception drains and, for most of the time, not much change in the salinity of the water in the bays in the basin (Figure 35). During the summer months (December 1997-January 1998), there was an increase in the salinity of the water in the second bay. This was not reflected in an increase in the salinity of the water in the interception drain. This suggests that, in this case, the isotopes are providing a more sensitive indicator of leakage to the interception drains. This is probably because the salinity of water leaking from the basin is similar to that already present in the soilwater, while there is significantly greater differentiation between the isotopes of the soilwater and the leakage.

In August 1998, samples of groundwater were collected from shallow groundwater (~0.5 m below the water table), in a transect perpendicular to the basin extending outward 100 m into the vineyard. Deuterium analyses on these samples were most enriched close to the basin and decreased with distance from the basin (Figure 36). At distances greater than ~20 m, the water samples from both transects had similar levels of deuterium.

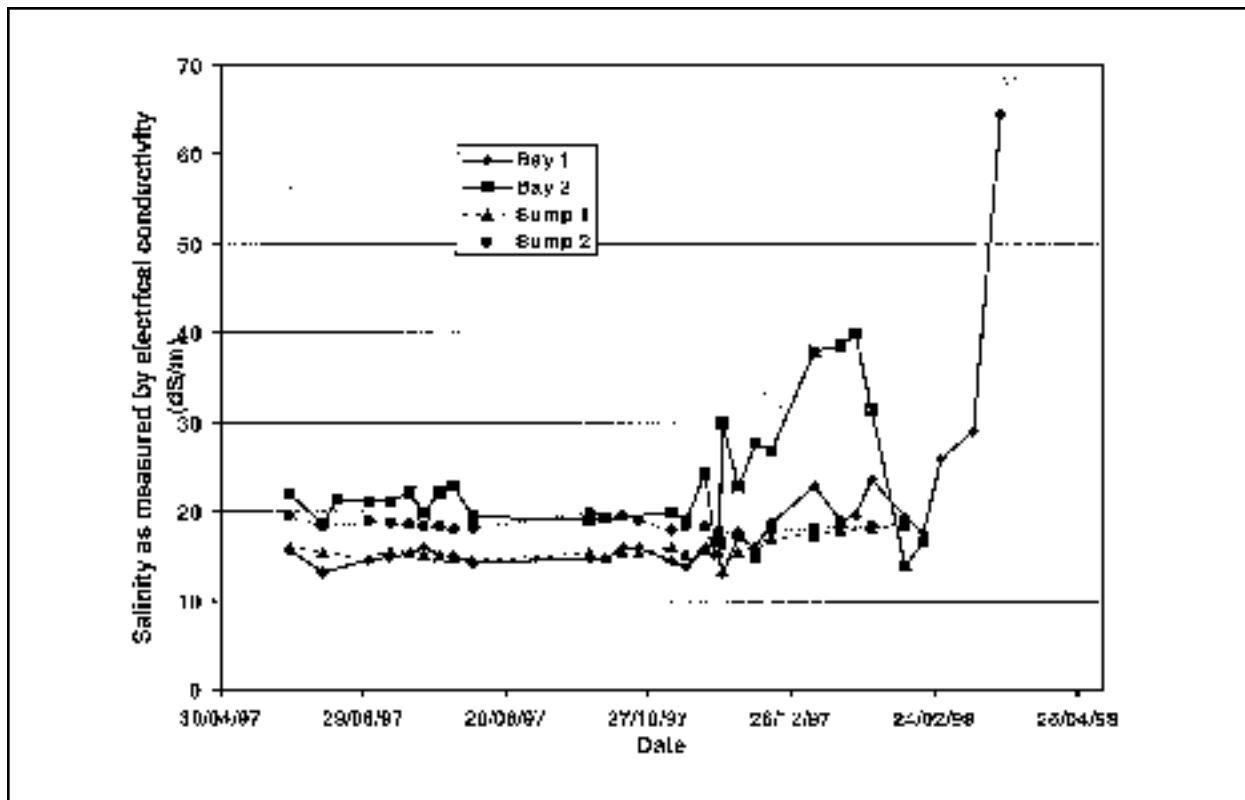Figure 35.   Changes in salinity of interception drain water and basin water at the Nehme basin.

Unlike ^2H, EC measurements do not provide a good indicator of whether or not basin water is being intercepted by the interception drain. The EC of basin water and that in the soilwater is similar for most of 1997
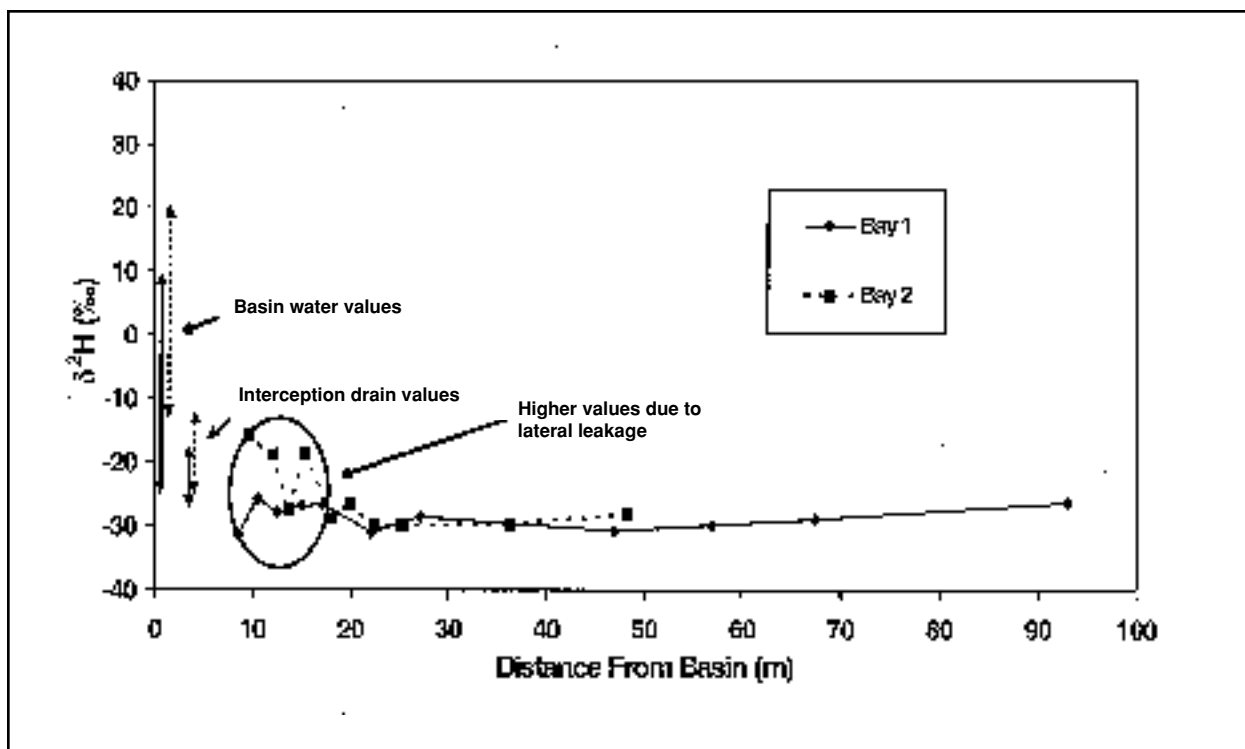

Figure 36.   Deuterium composition of shallow groundwater, transect from basin.

When this data is combined with the deuterium data from the bays in the basin and the interception drains near to the bays, it is clear that there is evidence of a component of lateral leakage extending ~20 m from the basin. The shallow layer of lateral leakage has taken place in the eight month period following the filling of the basin. However, flow was not via piston flow, because the isotopic signature of the groundwater was still considerably depleted when compared to that of the water in the basin.

6.1.5 Summary of leakage from Nehme basin

• **Total leakage**

A summary of all the leakage measurements from the basin over time and for different techniques is given in Table 6.

It is apparent that the initial leakage from the basin was very high for a period up to 18 days after the basin commenced filling. At that time, water reached and started to fill a 7 m deep piezometer. Due to the very deep unsaturated layer initially, the leakage rate over that period was estimated at up to 77 mm/day, assuming that there was no preferential flow and estimates of prefilling soil moisture content are valid. These assumptions are questionable, given the observed soilwater chloride profiles beneath the basin.

Table 6   Leakage summary for Nehme basin

Period	Technique	Leakage rate (mm/d)
Initial 18 days after filling	Filling of 7 m deep piezometer	up to 77 (probably considerably less)
Months 2 –4 after filling	Water and salt balance	6 - 7
Months 4-14	Water and salt balance	2.9 – 3.0
Refilling after 6 months dr y	Water and salt balance	2.6
Months 2 – 13	Concentration factor	3
Months 6 - 9	Seepage meter	0.4 – 5.2 (Av. 1.3, S.D. 1.6)

Subsequent salt and water balance estimates found that basin leakage stabilised at about 3 mm/day.

Point estimates by seepage meter were extremely variable, ranging from 0.4 – 5.2 mm/d. During the same period the water balance estimates were that leakage was 3.0 mm/d. This indicates that leakage is highly variable across the basin, a figure supported by a sixfold variation in apparent soil conductivity from EM38 survey.

The water balance method estimated leakage to average 3.7 mm/d. However, this does not account for the initial very high leakage period. The leakage by water and salt balance account for 1439 mm of leakage, not including the first 24 days of basin filling. The soil sampling suggests a total leakage of 1594 mm. The 155 mm difference may be due to initial leakage before intensive water balance analysis started; if so, the leakage over this 24 day period would equate to about 6.5 mm/d leakage. This is similar to the 5-9 mm/d measured by salt and water balance in the first three months. These results would indicate that the initial estimates of leakage by water reaching a piezometer are inaccurate. If this is so, then it is probable that not all of the profile became saturated during the early stages following filling; preferential flow has resulted in leakage reaching the deepest piezometer earlier than expected via piston flow.

The soil water chloride leakage analysis is clearly in the same order of magnitude as the water and salt balance estimates, and thus provides an independent check on leakage values derived by calculation. It is interesting that the results from the two soil cores are similar, despite having been taken near the highest and lowest leakage values measured by the seepage meters. This may indicate that leakage is highly variable in the surface layers, but this variation is 'smoothed out' with depth, due possibly to less variation in soil parameters and potentials in the deeper layers.

- **Lateral leakage**

Lateral leakage is taken as that measured in the interception drains. These drains, however, are unlikely to collect all the lateral leakage, as indicated by the stable isotope analysis of shallow groundwater outside the interception drains.

Initially, during the period when the groundwater mound was developing below the basin, the component of total leakage that was collected in the interception drains was about 23%. Subsequently, the total leakage decreased, but lateral leakage remained at similar levels; this accounted for about 50% of total leakage. The changes in flows in the interception drain are strongly related to the head of water in the basin, when the groundwater mound is fully developed under a basin.

Over the period of monitoring, the mean disposal rate to the basin was 5 mm/d. Of this, it would appear that 0.5-1 mm/d was water from the interception drains, representing 10-20% of the total input to the basin.

- **Leakage plume**

The volume of groundwater/soilwater that has been influenced by leakage from the basin appears to be about 5 m depth below the basin (by soil water chloride analysis), and about 20 m from the edge of the basin in the shallow groundwater (by deuterium analysis). However, it is possible that most of this volume may have been influenced during the short time frame following filling, when the soil beneath the basin became saturated. Thus, overall, it would appear that under the 2 ha of basin the groundwater has been affected to 5 m, and about an additional 1 – 2 ha has been affected around the basin in the shallow (<2 m depth interval) groundwater. For much of this area, there has only been a small component of leakage mixing with the existing water (i.e. the salinity and isotopic signature of the soilwater and groundwater is much closer to that for the pre-existing water than that for the leakage). The depth to which leakage has impacted on the groundwater around the basin is not known.

6.2
Other Basins in the MIA

In the MIA there are at present 14 on-farm evaporation basins. These vary markedly in shape, size, drainage water salinity and management (Table 7). However, they are all sited on clay soils and receive water from subsurface pipe drainage schemes; all have interception drains around their perimeter.

Table 7. Summary of on-farm evaporation basin characteristics in the MIA.

	Average	Minimum	Maximum
Basin area (ha)	4	0.6	14
Drained area (ha)	80	22	257
Percent of drained area	4	2	9
Percent of basin area utilised	37	0	100
Drainage water salinity (dS/m)	12	2	25
Basin water salinity (dS/m)	23	6	80

6.2.1 Leakage by water and salt balance

Measurements were only taken on a monthly basis, for a period of 18 months, so that of these 14 basins there were only four that provided sufficient data for estimates of leakage. The key difficulty was that many basins dried out intermittently which, when only monthly measurements are made, makes estimates of leakage impossible. Leakage for the four basins was estimated by water balance using electricity readings from the pumps and

also by salt balance on the basis of salt additions and change in salt stored. A further gross estimate of leakage was made using the concentration factor method, which took average drainage and basin salinities with the pan evaporation and rainfall measured at Griffith for 560 day period (Table 8).

Table 8. Summary leakage data from four MIA on-farm basins

Farm	Leakage by water balance (mm/d)	Leakage by salt balance (mm/d)	Concentration factor	Leakage by concentration factor	Interception flow	
					(mm/d)	(m³/d/m)
A	5.9	3.5	1.30	3.8	2.9	0.09
B	7.5	4.4	1.26	3.9	3.9	0.11
C	7.9	4.8	1.01	4.8	NA	NA
D	6.3	5.4	1.14	4.3	NA	NA
Average	6.9	4.5	1.2	4.2		
Nehme	3.7	3.0	1.45	3.0	1.5	0.05

The water balance estimates of leakage in most cases appear very high, and are considerably higher than the salt balance estimates. This can be attributed to problems in the estimation of evaporation due to variations in the basin area, errors in basin area estimates and also variation in rainfall. In the salt balance, only the pumping, drainage salinity, starting and finishing volume and average basin salt concentration over the period are required. This limits errors, although the starting and finishing volumes may be a problem, and the volume pumped into the basin can be a considerable problem. The estimates by concentration factor tend to agree closely with the salt balance estimates. The salt concentration factor method had a distinct advantage in such a survey in that knowledge of pumping or storage volumes was not required.

6.2.2 Lateral leakage

For two of the ponds, flow in the interception drains was measured occasionally and was found to represent a very high proportion (82-88%) of the total estimated leakage. Both of these ponds had relatively deep water, 800-1000 mm, and poorly constructed banks that allowed rapid lateral seepage. These lateral flows represent 26-32% of the total water additions to the ponds. As such, in these ponds, it would appear that a very high proportion of the leakage is being rapidly recycled. The high rate of recycling is not beneficial, as it increases cost of operation and reduces pump system capacity for drainage from the farm.

At the oldest on-farm basin in the MIA, an intensive EM31 survey was undertaken to try to track any leakage plume that may have developed under the basin. EM transects were run from the grapevines in the farm up to the edge of the surface drain that was only 10m from the other edge of the basin. The results show that there was only a small area of elevated salinity beyond the interception drains (Figure 37).
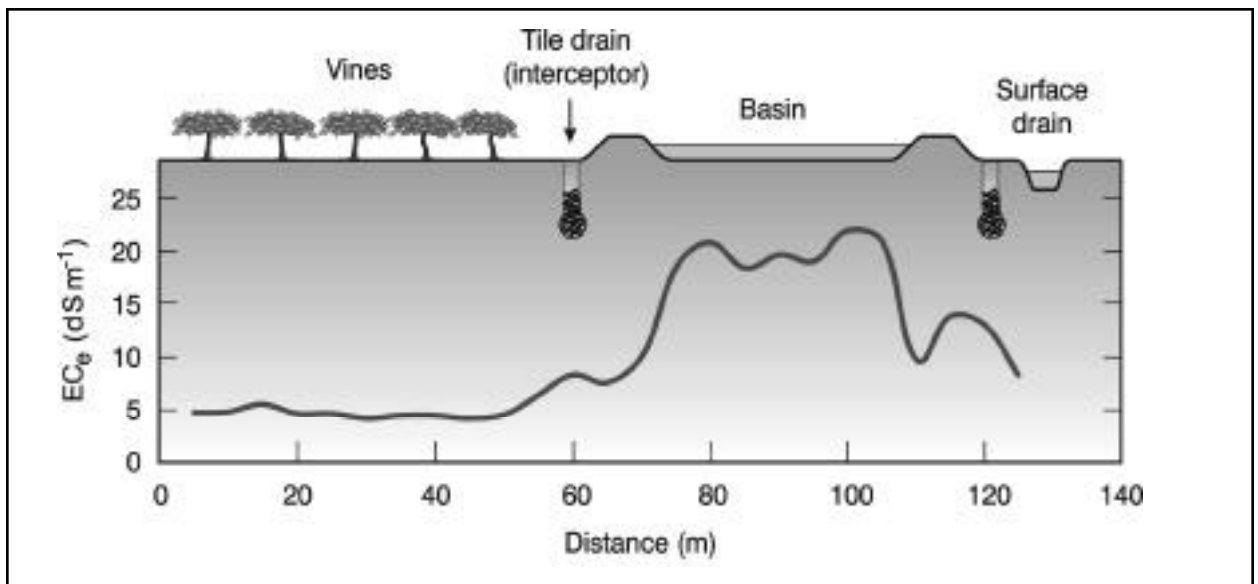

Figure 37.   Soil salinity transect across an 8 year old basin in the MIA using EM38 readings.

EC measurements suggest that leakage from the basin has influenced the soil up to ~20 m from the basin and ~10 m beyond the interception drain at this basin.

The elevated salinity in the direction of the grapes appeared to reduce immediately into the grapes, where there is a subsurface drainage system installed. This indicates that, on the grapes side, the leakage plume is being intercepted by the farm subsurface drainage system. This prevents the vines being affected by the leakage, and also presumably prevents the leakage plume migrating off the farm in that direction. However, on the opposite side which is bounded by an open drain, it would also appear that there are elevated salinity levels past the interception drain. The last measurement point is on the edge of the surface drain. The salinity moving past the interception drain is then likely to move off farm and possibly into the surface drain.

All the basins in the MIA are located near the farm boundary; eight of these are located close to surface drains, less than 10 m away in some cases, and the surface drain can be deep, up to 2 m in some situations. In order to assess whether there was leakage from these basins into these surface drains, the drain water was sampled upstream and downstream of the basin. These values were then compared to determine if an increase in salt or chloride could be detected in the drain.

Of the eight basins, an increase in the downstream water salinity was only detected for three. These three basins increased the surface drain water salinity by 1.3 to 2.6 times on average across the period of monitoring. However, the maximum increases measured at the basins ranged from 5.5 to 7.5 times. The worst two basins showed average increases in salinity from 2 dS/m upstream to 5 dS/m downstream, and 0.3 dS/m upstream to 1 dS/m downstream. The salt load discharging into these drains from the evaporation basins is not known. The basins that were discharging into the surface drains had several factors in common: edge of basin less than 15 m from the surface drain, surface drain 1 to 2m deep, basin kept very full, pump turned off for periods when basins very full and thus interception drains not functioning.

7. Conclusions

These field studies show that leakage from evaporation basins varies temporally and spatially. The Girgarre system has been functioning for 13 years and appears to have prolonged periods during which the leakage rate has been quite stable. For the first half of its life to date (6 or 7 years), the leakage rate for the basin was ~1.3 mm/d. For the second half of its life to date, the leakage rate decreased by approximately half.

The Nehme basin, by contrast, has displayed varied leakage rates over a short time interval. Leakage rates were very high at filling, then stabilised after some months to about 3 mm/d. The leakage rate remained stable for about 12 months until drying out. It is not known if this will be the long term leakage rate.

The leakage rate from small (< 5ha) evaporation basins in the MIA was in the range of 3.5 –5.4 mm/d, considerably higher than that for the much larger (30 ha) Girgarre basin (~1 mm/d).

Salt balance estimates are more reliable in estimating leakage than using a water balance. The method of comparing drainage water salinity to basin water salinity "concentration factor" is simple and useful for stable basin systems. The use of a salt and water balance model can be used to assess and track leakage rates accurately in the long term.

If the groundwater mound below a basin is not fully developed, it is likely that vertical leakage will dominate and lateral leakage will be much less. This is likely to be the case soon after a basin is filled, and may also be the case for areas where water control is via groundwater pumping. Under tile drain systems, once the groundwater mound is developed, vertical leakage is reduced and lateral leakage becomes at least 50% of the leakage for the small basins in the MIA. A significant proportion of leakage is recycled in the interception drains. It is likely therefore that the subsurface pipe drainage system in the farm will also have the potential to recycle a significant proportion of the basin leakage (however, this has not been demonstrated in this study).

Piezometers and natural tracers can be used to track the development of the saline leakage plume below and around a basin. It would appear that a component of leakage in the Nehme basin has reached 5 m below the basin and up to 20 m outside the basin. At Girgarre, the main part of the salinity plume has moved 2-4 m deep below the basin. However, there is strong evidence of leakage impacting in the shoestring sands, 10-15 m below the basin. This contamination must have occurred primarily by preferential flow.

In the shoestring sands, the saline leakage plume has moved a short distance outside the basin (20-50m), predominantly in the direction of the groundwater pump. Thus, this leakage should be eventually captured by the groundwater pump.

From the results of these studies, it is clear that evaporation basins should be sited a suitable distance away from surface drains. For tile drainage systems, the interception drains around the basin should be on a separate pump from the main subsurface drainage pump. A separate pump ensures that the interception drains will continue to function, even if the main subsurface drainage pump is turned off. Where a groundwater pump is used, then the basin should be positioned such that leakage will be in the capture zone of the pump.

These results also show that basins sited within the drainage system have the best possibility for containment of the saline leakage. Basins sited on the edge, or away from, the drainage system run the risk of not containing the saline leakage that may migrate off farm and contaminate groundwater or surface water features.

The results from these site investigations, when combined with those from previous studies on basins in the Riverine Plain, provide the basis for estimates of the disposal capacity for disposal basins in the Riverine Plain (as presented in the companion report, Leaney and Christen, 2000).

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Appendix A. Analytical Methods for Soil Analyses

Gravimetric water content

Calculated from the difference in weight of the soil sample and the sample dried at 105^0 C for 24 h divided by the dry weight of soil.

Soilwater chloride concentration (as per Taras et al., 1975)

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

Soilwater potential (as per Greacen et al., 1987)

Soilwater potential measurements were made by placing three 55 mm Whatman 42 filter papers at three levels in a 500 ml glass jar filled with soil compacted using a rubber plunger. The SWP of the soil equilibrates with the filter paper. The amount of water in the filter paper indicates the SWP of the filter paper and hence, the soil.

Particle size analysis (as per Lewis 1983)

The percentage of different particle size fractions was determined using the pipette method (Lewis, 1983) using air dried soil samples. Sand is considered to be coarser than 0.02 mm, silt between 0.02 and .002 mm and clay to be finer than 0.002 mm (2000 micron).

Analysis of stable isotopes

Water was extracted from ~100 g of soil by azeotropic distillation with kerosene (Revesz and Woods, 1990). Saline water samples were similarly treated to produce water suitable for isotopic analysis.

The deuterium composition of water (^{2}H) was measured by standard isotope ratio mass spectrometry. ^{2}H was measured after reduction of water over hot uranium (Dighton *et al.*, 1997).