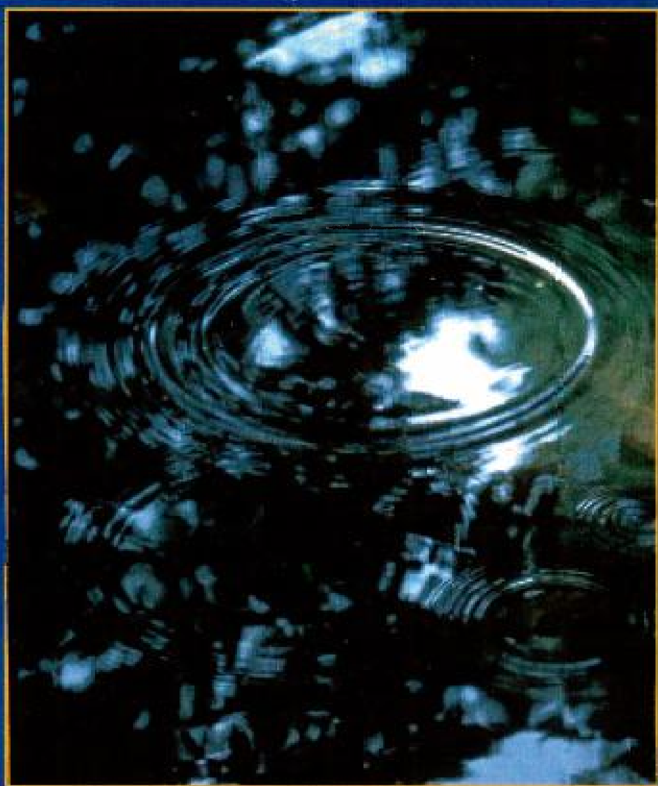
Murray Darling Basin Groundwater Workshop 1999

"Integrated perspectives"



Conference Proceedings

14 — 16th September 1999, Griffith, NSW

The Murray Darling Basin Groundwater Workshop 1999 is proudly sponsored by:

MURRAY-DARLING BASIN COMMISSION

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1999 Murray Darling Basin Groundwater Workshop

PROGRAM: TUESDAY 14TH SEPTEMBER, 1999

9.00AM	Introduction: **Scott Keyworth** Murray Darling Basin Commission Bureau of Rural Sciences
9.10AM	Ray Evans : Chair MDB Ew[illegible] group **OPENING: John Searson,** Regional Director, NSW Agricullture

KEYNOTE SPEAKERS:

9.25AM	Policy implications for COAG water reforms on groundwater **Mike Smith**, *Executive Officer, High Level Steering Group on Water*
9.50AM	Perspectives on sustainable development of water resources: a USA Great Plains outlook **Marios Sophocleous,** *Senior Scientist, Kansas Geological Survey, University of Kansas*
10.30AM	MORNING TEA

PLANNING AND POLICY:

11.00AM	Sustainable groundwater management in NSW, Australia **George Gates**, *DLWC* given by Vanessa O'Keefe
11.20AM	Prioritising management directions, determining sustainable yields and implementing sustainable management practice of groundwater resources in NSW, Australia **Joseph Ross,** *DLWC*
11.40AM	Transferable groundwater entitlements **Vanessa O'Keefe**, *DLWC*
12.00PM	The Lower Murrumbidgee Groundwater Management Plan **Scott Lawson**, *DLWC*
12.20PM	Environmental provisions in determining sustainable yield for groundwater management plans in the Lower Namoi Valley, NSW, **Phillip Kalaitzis**, *DLWC*
12.40PM	LUNCH

PROGRAM: TUESDAY 14 SEPTEMBER 1999

GROUNDWATER AND ECOSYSTEMS

1.40PM Protection of groundwater-dependent ecosystems in NSW – policy and practice, **Jan Gill**, *DLWC*

2.00PM An identification methodology for groundwater-dependent ecosystems, **John Ross**, *PPK Environment & Infrastructure*

2.20PM "One step ahead" – Groundwater-dependent ecosystem management in NSW , **Jenny Guice**, *Nature Conservation Council, NSW*

2.40PM Karstic groundwater ecosystems in the Murray Darling and Otway groundwater basins, **Andy Spate,** *NSW National Parks & Wildlife*

3.00PM Can we predict trends in floodplain tree health in the Lower Murray?, **Kate Nicholls**, *Centre for Groundwater Studies, Flinders University of South Australia*

3.20PM **AFTERNOON TEA**
Book launch: Desktop Methodology to identify Groundwater Dependent Ecosystems
Presenter: Tim Fisher, *Australia Conservation Foundation*

CONCURRENT SESSIONS

DRYLAND SALINITY

3.50PM Towards a predictive framework for land use impacts on recharge: a review of recharge studies in Australia
Cuan Petheram, *CRC for Catchment Hydrology*

4.10PM Catchment categorisation: what is it and how does it work? Unavailable today Swapped to Thurs
Mirko Stauffacher, *CSIRO Land & Water*

4.30PM Eastern Murray (NSW) Groundwater Investigation & Monitoring
Nimal Kulatunga & Stuart Lucas, *DLWC*

4.50PM The Liverpool Plains ICSM Project: lessons to be learned
W.R. Dawes, *CSIRO Land & Water*

6.00PM **WINE TASTING – McWilliam's Wines**

PROGRAM: TUESDAY 14 SEPTEMBER, 1999

CONCURRENT SESSIONS: AQUIFER INVESTIGATIONS & PLANNING

3.50PM	Groundwater management developments in central west NSW, **Greg Brereton**, *DLWC*
4.10PM	A regional sub-artesian groundwater investigation in a cainozoic sedimentary aquifer system in SW Queensland **Doug McAlister**, *QLD Dept of Natural Resources*
4.30PM	Hydrology and sustainable yield of the mid Murrumbidgee alluvial aquifer, **Liz Webb**, *DLWC*
4.50PM	Groundwater vulnerability mapping of the Upper Condamine River Catchment using the DRASTIC methodology **Allan Hansen**, *QLD Dept. of Natural Resources*
5.10PM	Groundwater quality in an aquifer aquitard system subjected to large volume abstraction for irrigation in the Lower Murrumbidgee **Wendy Timms**, *Groundwater Centre, UNSW*
6.00PM	**WINE TASTING – McWilliam's Wines**

PROGRAM: WEDNESDAY 15 SEPTEMBER 1999

IRRIGATION

9.00AM	Reducing subsurface drainage salt loads: development of drainage design and management guidelines **Evan Christen**, *CSIRO Land & Water*
9.20AM	Assessment of deep bore water quality for irrigation in the Murray irrigation districts, **Harnam Gill**, *NSW Agriculture*
9.40AM	Spatial interpolation of groundwater data and its implications for environmental management, **Tim Harrington,** Kinhill P/L
10.00AM	Review of regional groundwater model of Coleambally Irrigation area, **Shahbaz Khan**, *CSIRO Land & Water*
10.20AM	Leakage between shallow and deep aquifers in the South Eastern Murray Basin, **Scott Lawson**, *DLWC*
10.40AM	**MORNING TEA**

PROGRAM: WEDNESDAY 15 SEPTEMBER 1999

SALT/EFFLUENT DISPOSAL

11.10AM Development of guidelines for on-farm and community scale salt disposal basins on the Riverine Plain: underlying principles
Ian Jolly, *CSIRO Land & Water*

11.30AM Characterisation and ranking of saline disposal basins in the Murray Basin of Australia
Craig Simmons, *School of Earth Sciences,Flinders University of South Australia*

11.50AM Regional planning for disposal basins on the Riverine Plains: Testing a GIS-based suitability approach for environmental Sustainability,
Trevor Dowling, *CSIRO Land & Water*

12.10PM Factors affecting the financial viability of subsurface drainage with an on-farm disposal basin in the Murrumbidgee Irrigation Area
Jai Singh, *Visiting Resource Economist, CSIRO Land & Water*

12.30PM Soil hydraulic properties and pollutant removal in a pilot 'filter' system used for treating Griffith sewage
Tapas Biswas, *CSIRO Land & Water*

12.50 **LUNCH**

2.00PM **FIELD TOUR – BUSES DEPART FRONT GEMINI MOTEL**

5.10PM **FIELD TOUR RETURNS GEMINI MOTEL**

7.00PM **CONFERENCE DINNER**

Murray Darling Basin Groundwater Workshop 1999

Field Trip – Day 2 (15th September)

Buses will **depart at 2.00 pm** from the Workshop venue
Return by 5.30 pm. Afternoon tea will be provided.

The tour will incorporate aspects of:

- Serial Biological Concentration
- Evaporation basins and shallow drainage
- Town supply, river leakage and recharge
- Groundwater supply for irrigation

1. Serial Biological Concentration (Griffith, MIA)

Serial Biological Concentration offers an opportunity to improve overall water use efficiency whilst managing salt drainage from an irrigation area. The first cell of the current trial has been operating for five years the next two cells for two years. Theoretical performance is being achieved in large scale, practical demonstration, offering an exciting salt management alternative.

2. Evaporation basins and shallow drainage (Griffith, MIA)

In the MIA there are 15 on-farm evaporation basins which are used to store saline drainage water from subsurface "tile" drainage systems. We will visit an area near Hanwood where there are several evaporation basins associated with large vineyards. This will provide the opportunity to discuss the design and operation of shallow drainage systems and evaporation basins.

3. Town supply, river leakage and recharge (Darlington Pt)

The Darlington Pt township utilises groundwater for it's reticulated supply. Groundwater quality in the deep aquifers of the Lower Murrumbidgee is generally very good, with EC as low as 170 uS/cm. However, dissolved Fe and H_2S can be a problem for town supplies. This is also an opportunity to discuss river leakage, recharge and linkages between aquifers.

4. Groundwater supply for irrigation (Darlington Pt)

Visit to "Ringwoood", an irrigation property owned and run by the Toscan family, and dependent upon groundwater. Gerard Toscan is President of the Murrumbidgee Groundwater Pumpers Association, and a groundwater user representative on the Murrumbidgee Groundwater Management Committee.

PROGRAM: THURSDAY 16 SEPTEMBER 1999

SALINITY MANAGEMENT

Kate Chair

9.00AM	Mining groundwater for sustainability in the Mallee **Steve Barnett & Glennis McKee,** *Primary Industries & Resources, SA*
9.20AM	Prioritising salinity management in the Goulburn-Broken dryland **Bruce Gill**, *Sinclair Knight Merz*
9.40AM	Community salinity planning (Central West NSW) **Elita Humphries**, *DLWC*
10.00AM	Hydrogeochemical processes associated with the occurrence of Dryland salinity in the Longneck Creek Catchment, near Windsor, NSW, **Wendy McLean,** *Groundwater Centre, UNSW*
10.20AM	Catchment categorisation: what is it and how does it work ~~Regional scale dryland salinity risk prediction – a review~~ ~~**Glen Walker**~~, *CSIRO Land & Water* Warwick Dawes on behalf of Mirko Stüffüsher

SALINITY – URBAN AND INTERCEPTION

11.10AM	Development in an urban salinity affected catchment – A case study in Troy Gully, Dubbo, NSW **Allan Nicholson**, *DLWC*
11.30AM	The Wagga Wagga urban salinity bore field **Steve Nash,** *Westernport Water*
11.50AM	Urban salinity in Wagga Wagga (NSW): sources of recharge and impact of pumping the fractured rock groundwater system **Peter Cook**, *CSIRO Land & Water*
12.10PM	Buronga salt interception scheme – effiency review **Noel Merrick**, *National Centre for Groundwater Management, University of Technology*
12.30PM	Woolpunda and Waikerie salt interception schemes **Andrew Telfer**, *Australian Water Environments*
12.50PM	**LUNCH: presentation -GROUNDWATER TRAINING** **Trevor Pillar**, Centre for Groundwater Studies, Flinders University

CONCURRENT SESSIONS

WATERTABLE IMPACTS

1.50PM Trial results for a new electro-kinetic geophysical technique for remote measurement of sub-surface hydraulic conductivity (Author: **Chris Waring**) Presenter: **Richard Lowson** *Australian Nuclear Science and Technology Organisation*

2.10PM Vertososls do "leak"! – water and solute movement below irrigated cotton, **James Moss**, *QLD Dept. of Natural Resources*

2.30PM Modelling channel seepage interception by trees on a prior stream levee, **Greg Holland**, *Goulburn Murray Water*

2.50PM An integrated system for groundwater vulnerability assessment in the Liverpool Plains of NSW, **Fei Zhou**, *CSIRO Land & Water*

3.10PM **AFTERNOON TEA**

CONCURRENT SESSIONS

REGIONAL INVESTIGATIONS

1.50PM Mallee Region groundwater modelling, **Wei Yan & Steve Barnett** *Primary Industries & Resources, SA*

x **2.10PM** Using GIS for catchment water balances, **Steve Barnett** *Primary Industries & Resources, SA*

2.30PM Hydrogeology of the Jemalong and Wyldes Plains Irrigation District and Lake Cowal aquifer systems, Lachlan Catchment, NSW, **Ruben Lampayan**, *Centre for Resource and Environmental Studies, ANU*

2.50PM Ecologically sustainable opportunities for the Lower Murray Darling area, **Mazib Rahman**, *Murray Darling Water Management*

3.10PM **AFTERNOON TEA**

3.40PM PLENARY SESSION: **Summary : Ray Evans & Scott Keyworth**

4.00PM DISCUSSION

4.30PM FINISH

CSIRO LAND and WATER

A case study of improving irrigation and tile drainage management in a vineyard

Evan Christen and Dominic Skehan

A collaboration with the CRC for Catchment Hydrology,
partly funded by the Murray Darling Basin Commission

Benefits of improved irrigation and tile drainage management

To the farmer:

- Improved wine grape quality
- Water, fertiliser and labour savings
- Less waterlogging - deeper water tables, lower soil salinity = improved vine health
- Reduced size of evaporation basin (if required)

Benefits of improved irrigation and tile drainage management

To the community:

- Reduced on-farm effects - land values, production
- Reduced off-farm effects
  - less surface drainage water (nutrients, salt, pesticides)
  - less tile drainage (salt, nutrients, pesticides)
  - less recharge to water tables = less waterlogging and salinity in the area

The case study vineyard

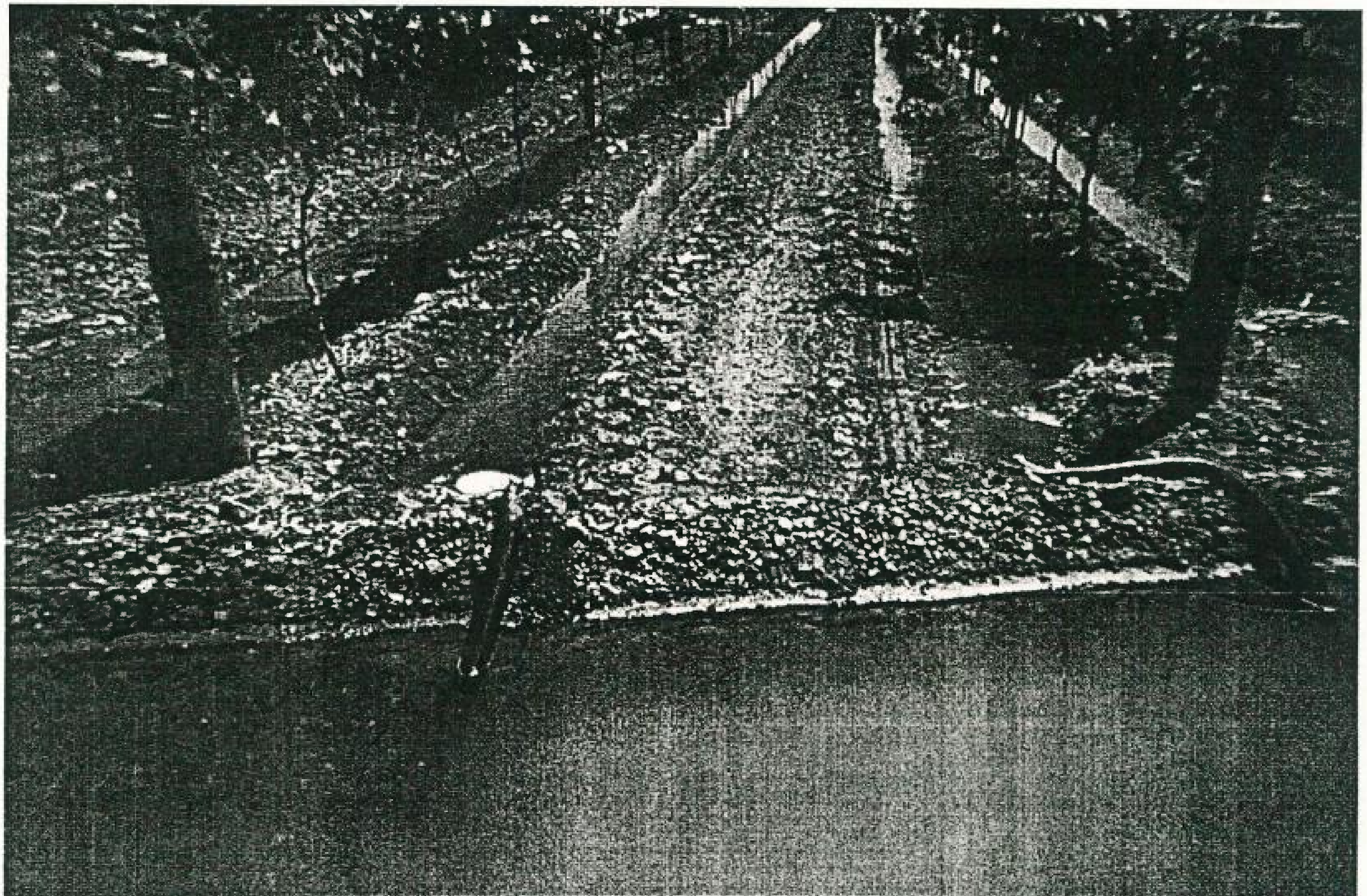
- 50 ha on clay soils
- 400 m rows with broad-based furrows
- Tile drainage with evaporation ponds
- Irrigation by "gut feeling"
- Irrigations slow to reach end of row and wetted whole area between vine rows
- Tile drainage left on all the time

What changes were made ?

To the irrigation system:

- Replaced broad-based furrows with Riverina Twin Furrow
- Used more siphons per furrow

Flood irrigation - Riverina Twin Furrow



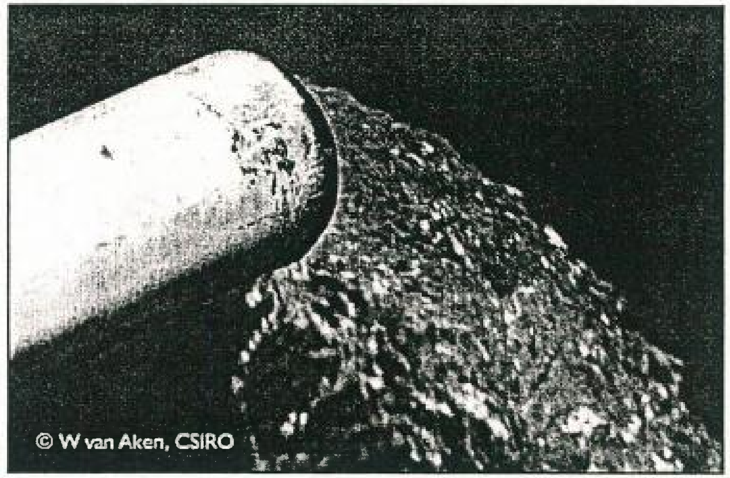
What changes were made ?

To the irrigation management:

- Tensiometers used for scheduling
  - 30cm, 60cm, 90cm
  - one group per variety

- Followed guidelines by Hardie and Martin
  - flowering keep root zone around 10kPa
  - fruit set to harvest let root zone dry to 80kPa

What changes were made ?



To the tile drainage management:

- Turn off pump during irrigation (bath plug)
- Turn off pump when water table more than 1.2m deep
  - used depth to water table around the farm to start with
  - later used water level in pump sump

Results of the changes

To the irrigation system:

- Only about 1/3rd the vineyard soil wetted
  - about 0.6m on each side of the vine row
  - 2.4m in the middle kept dry

  = less soil volume wetted, less waterlogging, rainfall easily stored

- Water reached the end of rows in 1/3rd to 1/2 the time

  = more even watering between top and bottom

Results of the changes

Effect of furrow size and siphons on time for water to reach end of 400m row

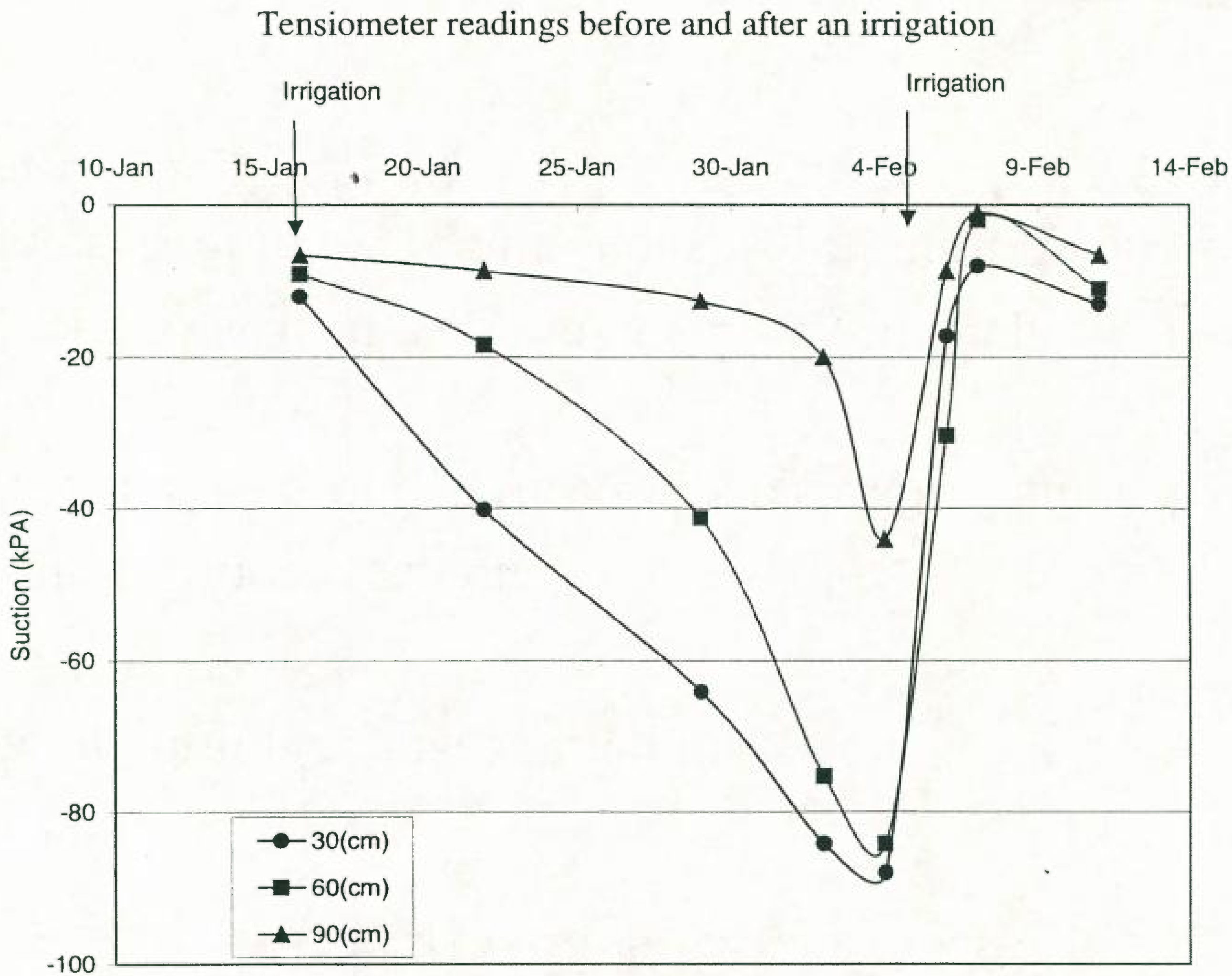
Furrow used	No. of siphons 25mm (1")	Time to end of row Hours
Broad base	2	15-18
Riverina Twin Furrow	2	8-10
Riverina Twin Furrow	4	6

Results of the changes

To the irrigation management:

- Increased irrigation interval from about 11 days to 17 days
- Dried out top 60cm to 80kPa
- Roots started using water from 90cm depth
- Matched crop water requirement with irrigation AND rainfall

Results of the changes



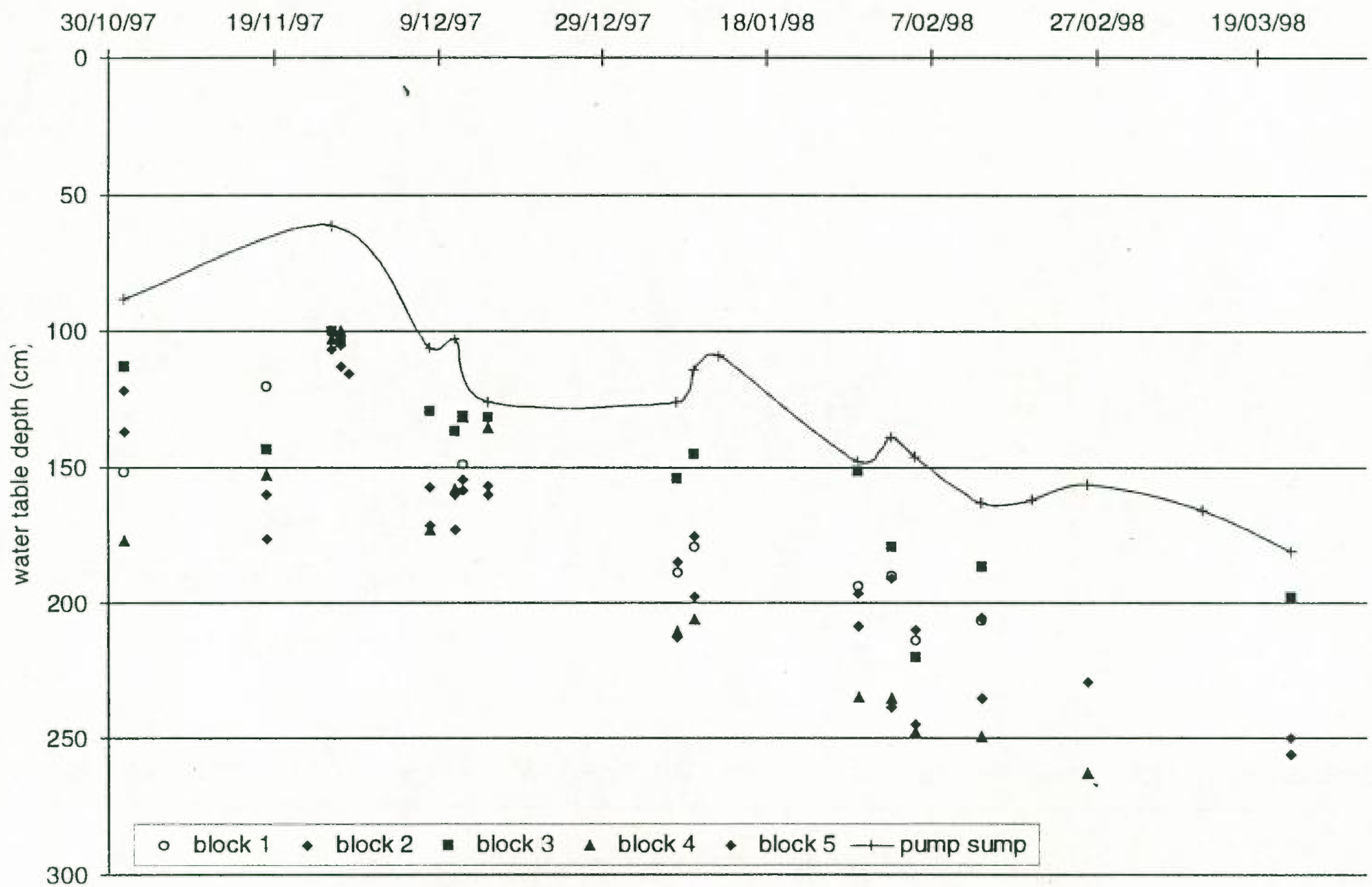
Results of the changes

To the tile drainage management:

- Pump off more than on
- Water tables dropped through the season (rather than built up)
- By end of season water tables 2 - 2.5m deep
- Water level in evaporation basins fell until by end of season they were dry

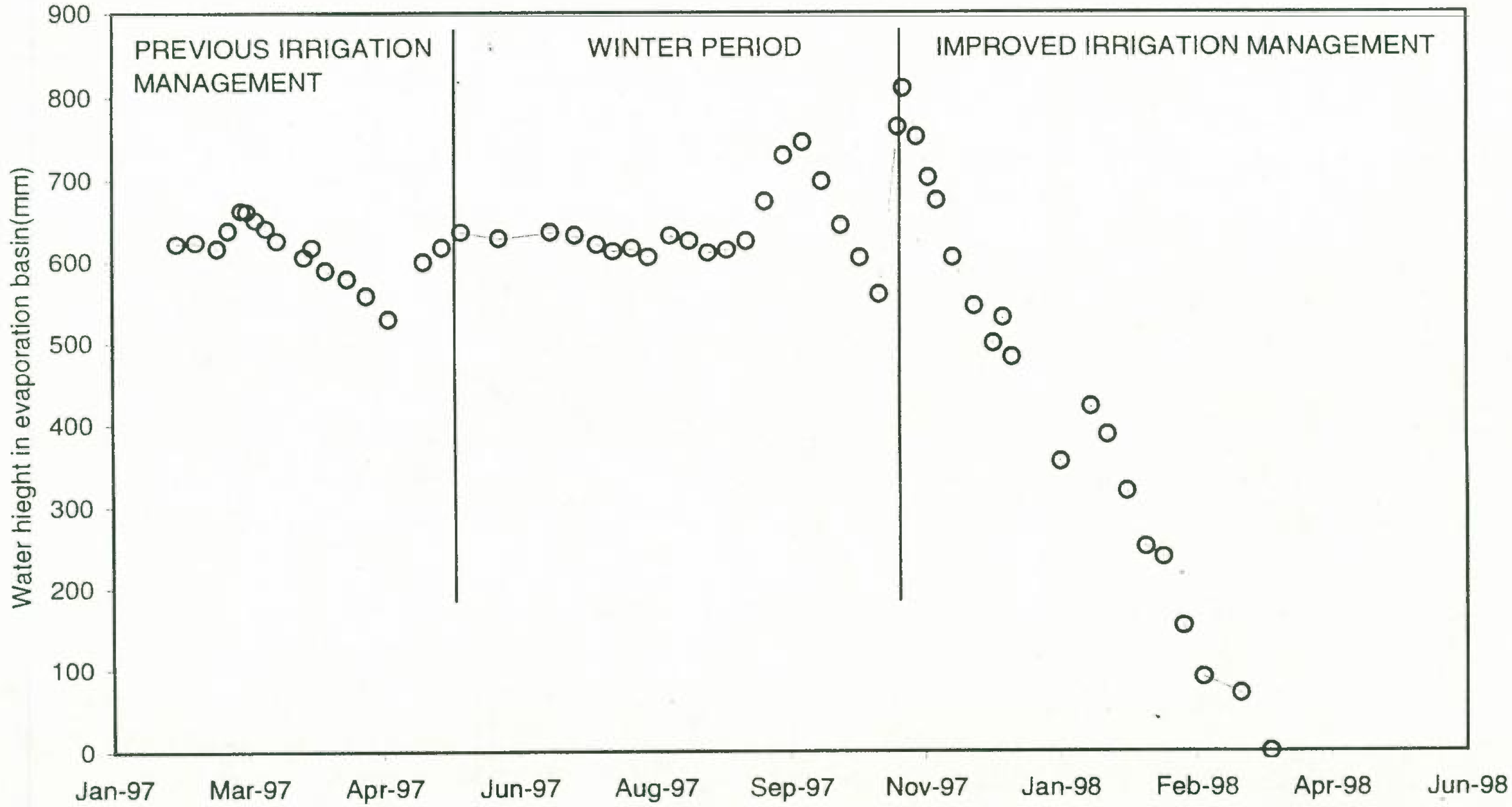
Results of the changes

Water tables with improved irrigation management



Results of the changes

Water level in evaporation ponds



CSIRO LAND and WATER

Comparison of the two seasons

SEASONAL Factors	Season (Jan –May)		Change
	1997	1998	
Monitoring - January to May (days)	150	150	0
Potential Evapotranspiration (mm)	951	887	Down 7%
Rainfall (mm)	93	132	Up 40% (39mm)
Crop water use (mm)	481	443	Down 8%

Comparison of the two seasons

IRRIGATION Factors	Management		Change
	Previous	Improved	
Irrigation applied total (mm)	590	354	Down 33 %
Amount per irrigation (mm)	56	66	Up 18 %
Irrigation interval (days)	11	17	Up 54 %
Irrigation compared to requirement (irrig. Applied / crop requirement)	23 % too much	20% too little	Down 43 %
Water use efficiency (irrig. + rainfall / crop requirement)	42 % too much	10 % too much	Up 32 %

Comparison of the two seasons

TILE DRAINAGE Factors	Management		Change
	Previous	Improved	
Water table depth (m)	1 - 1.8	1.5 - 2.3	Down 30 %
Tile Drainage (mm)	73	9	Down 88 %
Salt removed by tile drainage (kg/ha)	6655	711	Down 89 %
Salt applied by irrigation (kg/ha)	453	272	Down 40%
Salt removed by tile drainage compared to salt brought in	15 times more	3 times more	Down 80%

CONCLUSIONS

By changing the irrigation system and scheduling with tensiometers we:

- Reduced irrigation by 2 Ml/ha = 100 Ml total
- Matched irrigation (plus rainfall) to crop water use (+10%)
- Dropped water tables to below 2m

By managing the tile drainage:

- Reduced pumping by 0.64 Ml/ha = 32 Ml total
- Better matched salt applied to salt pumped out (15x to 3x)
- Dried out the evaporation ponds

Please Return to Kate Boyd

Sequential Biological Concentration (SBC) for SALINITY MANAGEMENT and improved WATER USE EFFICIENCY in Irrigation Areas

J. Blackwell, T.K. Biswas, N.S. Jayawardane[1], and J.T. Townsend

CSIRO Land and Water, PMB No 3, Griffith, NSW 2680. [1]CSIRO Land and Water, GPO Box 1666, Canberra, ACT 2601,

The problem ...

Salinity has bedevilled agriculture for centuries and irrigation speeds up the rate of salinisation.

Short of another "Green Revolution", perhaps inspired by advances in Biotechnology, irrigation will be critical in feeding the global population.

Projected population increases coupled with increases in consumption for better nutrition resulting from rising incomes will substantially increase the demand for food in developing countries. This increased demand has to be met when land and water resources, available for agriculture, are decreasing. With competing use by the other sectors, water is increasingly seen as an economic good and conflicts over water will grow in number and severity. The major challenge as we enter the twenty first century will be how to increase and sustain the productivity of irrigated agriculture while reducing the sectors water consumption.
To have any hope of a sustainable future we must face and resolve the population issue.
In the meantime it is our contention that Irrigated Agriculture will become increasingly important in meeting the food demands of the global population.

The Present Position

Currently irrigation supplies close to 30% of the world's food crops and 50% of the two major staples, wheat and rice. It achieves this from one sixth of the available arable land that is irrigated.
Total food production **from irrigation** will need to increase by 12% by 2010.

Is Irrigation up to The Task?

In Australia's harsh, dry climate, evaporation is high and irrigation is commonly used to enable us to grow the wide variety of food crops which we enjoy, but irrigation is not without problems which require careful management if we are to avoid them.
There is salt in irrigation water and even rain, and when water is applied to the ground, plants take up pure water as they grow and concentrate the salt in the soil. Unless this salt is leached below the rootzone, the land will become salty and be unfit for crop production.

Drainage of the landscape is necessary, whether through a natural system (river, creek or stream) or a man made system. Drainage MUST be an initial part of any well designed irrigation system. Surface drainage may not be enough to drain the soil adequately. Sub-surface drainage may also be needed to lower the water table and remove the salt.

Traditional methods of dealing with salt export, accessions to deep aquifers or discharge to a watercourse, may no longer be viable.
In the first case many of these deep aquifers now present high saline water tables which threaten the root-zone. In the second, public pressure for environmental responsibility, and consideration of downstream users, preclude its use.

To recycle drainage water back onto the land drained is a short sighted, short term "solution".
To manage LOW salinity water by evaporation in ponds, without, where possible, using the water for production is to waste a resource.

One option for managing inevitable drainage, which overcomes many of these shortcomings, is to sequentially use and re-use the drainage water to grow crops whilst concentrating the drainage to a manageable mass, perhaps 0.5 – 1.0% of the original.
This process known as "Sequential Biological Concentration also enables us to improve our overall water use efficiency. (figure 1.)

When re-using drainage or waste water salinity again becomes an issue, as waste waters (such as sewage effluent and drainage) usually have a higher salt content than irrigation water.

A Proposed Solution
A novel system for land based treatment of secondary treated effluent, which is both environmentally and economically sustainable, the "FILTER" system (figure 2), has been tested for four years as a cooperation between Griffith City Council and CSIRO Land and Water. A full-scale Pilot system of 16 hectares has been operating for two seasons.
"FILTER" has proved capable of overcoming the inherent problems with existing land based treatment systems.
The system removes nutrients and other pollutants from the waste stream, meeting Environment Protection Authority discharge requirements, and at the same time concentrating salts whilst producing an economic crop.

The sequential use of the FILTER modules will enable us to convert the drainage waste water containing a cocktail of pollutants, which threaten our environment and our productive base, into an economically viable and environmentally acceptable production system.

A System To Manage Saline Drainage Water From an Irrigation Area
The entire drainage from the Murrumbidgee Irrigation Area (MIA) is collected via an extensive net work of surface drains, which converge at a natural 3,200 hectare depression called Barren Box Swamp.
The water from Barren Box is later used for extensive irrigation downstream in the Wah Wah Irrigation District.
The present salinity of Barren Box ranges from 500-700 EC
(drinking water = 100 EC, seawater = 50,000 EC).
Downstream farmers in Wah Wah demand water at 450 EC.

Winter drainage is saltier than summer as it contains a higher proportion of tile drainage and surface runoff from non irrigated areas.
If we remove the winter drainage flow (9,000-35,000 ML at 1,200EC) from the total flow entering Barren Box, the swamp water quality will improve which will assist managers in achieving downstream water user requirements.

This winter flow will be stored en-route to the Sequential Biological Concentration system (SBC)
Storage of this winter flow will enable subsequent re-use through the different cells of the SBC system. (figure 3)

1,200 EC is 10 times saltier than the original irrigation water and 3 times saltier than downstream users will accept.
However, in practice if we can maintain a 30% leaching fraction through the soil profile the rootzone will come into equilibrium with the salinity of the applied water (see figure 4 and Result 1 and 2). This will allow the production of more salt sensitive crops at any given applied water salinity than would be the case with a lower leaching fraction. (see Result 3)

The drainage from cell one at 30% leaching fraction will have a salinity three times the applied water, i.e. 3,600 EC.

This sequential process is repeated four times (see figure 3), with each of the four production cells having the potential of being economic in its own right. The fourth cell now approaching sea water salinity must also drain otherwise it will tend to hyper-salinity and be useless for managed aquaculture.

The resulting drainage will flow through a series of sealed evaporation basins the first of which will be converted into a Salt Gradient Solar Pond which should produce enough electricity to power the entire site.

As the drainage concentrates through the sequential evaporation basins it should be possible to recover different salts, some of which should be saleable.

Figure 1: Showing the theory of Sequential Biological Concentration (Effective water use efficiency)

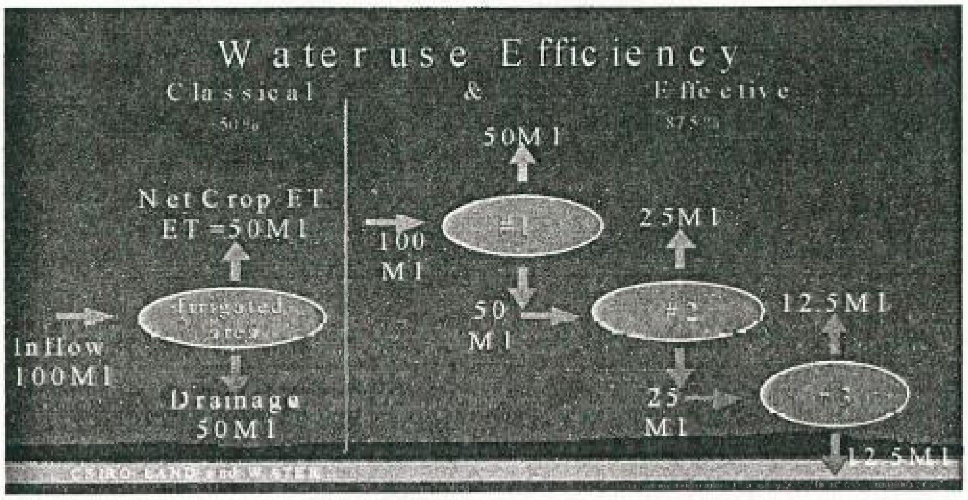


Figure 2 : Schematic representation of a typical filter bay

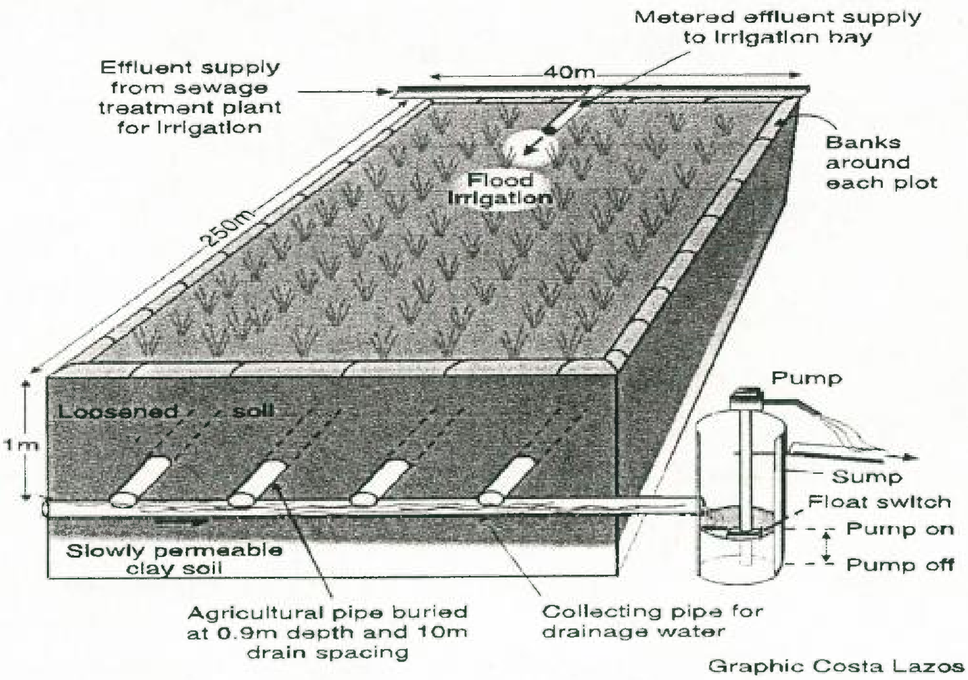
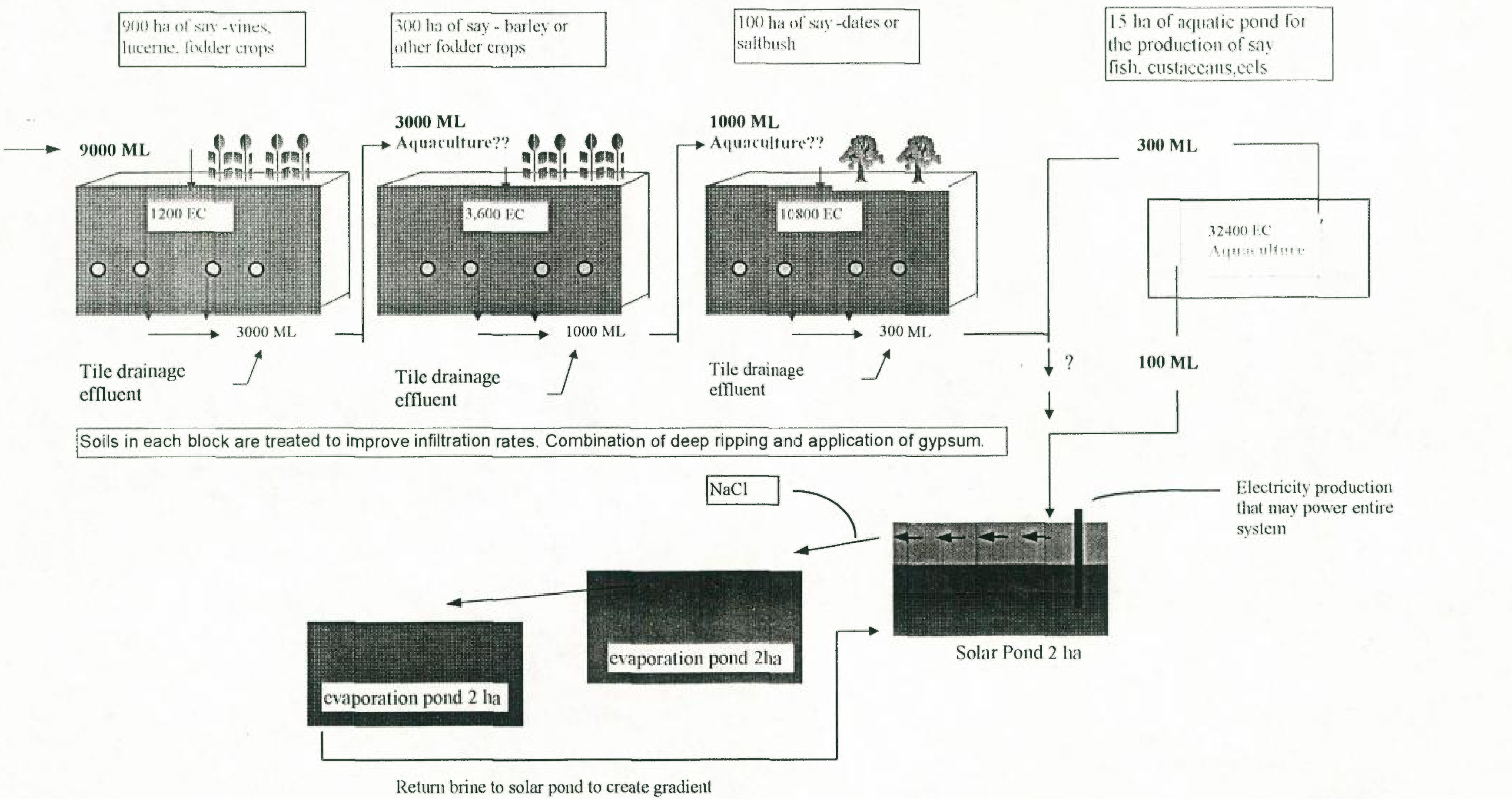
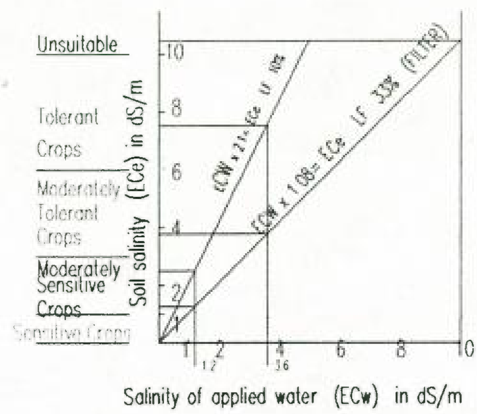


Figure 3. Schematic Representation of Possible Layout, Flows and Concentrations of SBC System

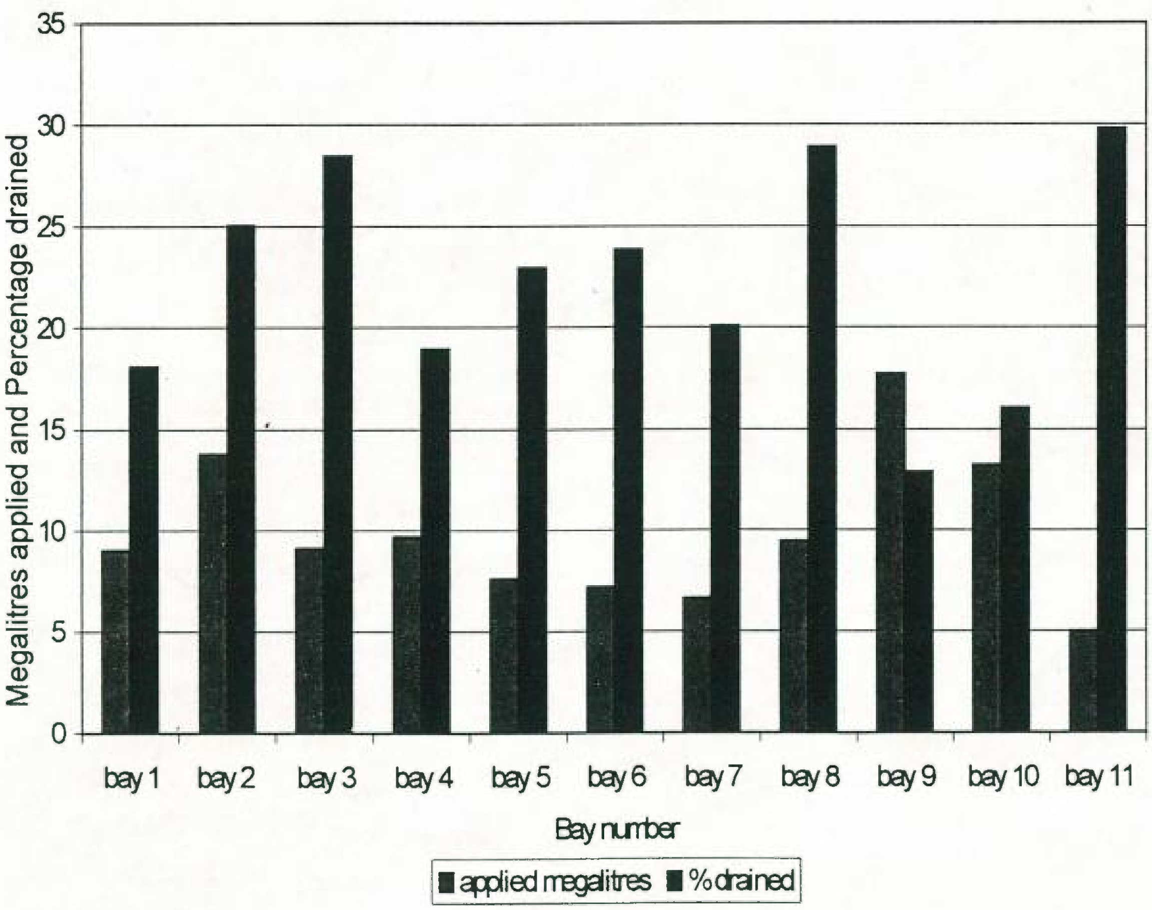


Relationship between salinity of applied water and soil salinity at different leaching fractions

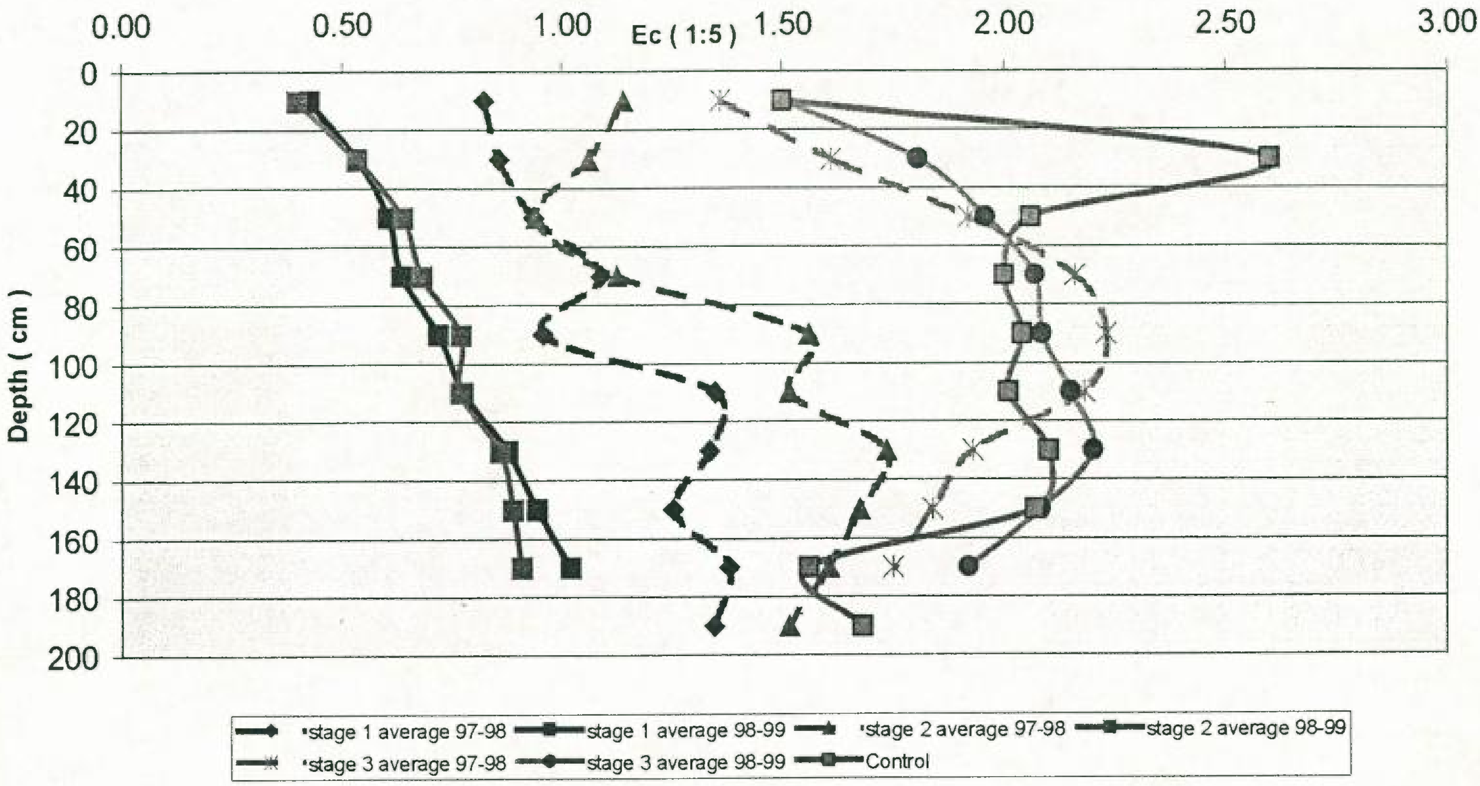
Figure 4 :



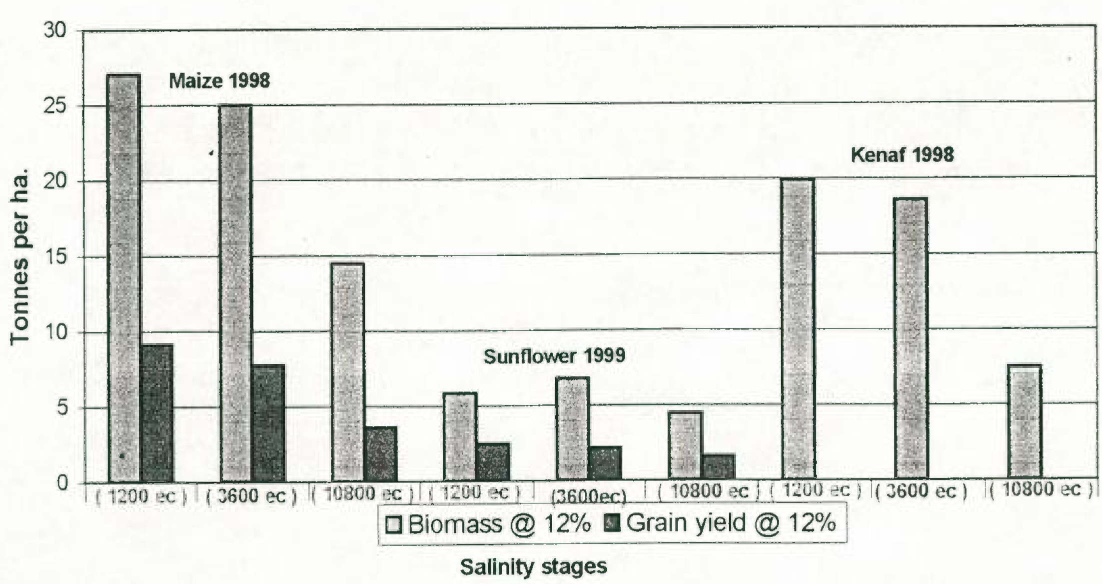
Result 1 : Leaching fractions obtained over 2 seasons averaged



Result 2 : Changes in soil salinity over time at the 3 applied salinities (Ec 1:5 * 4 = Ece)



Result 3 : Crop yields at the salinity levels in the first three stages



Eastern Murray (NSW) Groundwater Investigation and Monitoring

Nimal Kulatunga and Stuart Lucas
Department of Land and Water Conservation (DLWC)
P.O. Box 829, Albury NSW 2640.

Dryland salinity has been identified as an issue in several eastern Murray sub-catchments. Previous studies undertaken in West Hume and Holbrook Landcare areas (1989, 1991) revealed that groundwater pressure levels have been risen by 30-60 cm/year. The objective of the current study is to raise awareness of rising groundwater levels and related problems among all 11 Landcare groups in the eastern Murray dryland catchments. This has involved up to 130 landholders monitoring bores and piezometers regularly and DLWC preparing depth to watertable and pressure level maps, trend assessment and also groundwater salinity and recharge assessment maps.

INTRODUCTION

Dryland salinity in the eastern Murray catchments has been identified as a major issue with increase areas of waterlogging and soil salinisation. Landholders lower in the catchments have been affected with significant losses to the agricultural productivity in some areas. These production losses and the adverse effect on the environment have made the community more aware of such a situation and seek remedial measures to combat them by action plans for the catchments.

Previous studies undertaken in West Hume and Holbrook Landcare areas (1989, 1991) revealed that groundwater pressure levels have been risen by 30-60 cm/year. Subsequently, EM31 surveys were done in selected sub-catchments to establish baseline soil salinity levels. These two landcare areas are ahead of other landcare groups in the eastern Murray catchments in this regard and have embarked to develop Land and Water Management Plans. In the West Hume study, extensive waterlogging and land salinisation were predicted with up to 230 km^2 affected by 2020 in the 'do nothing' scenario. The West Hume landcare group in particular has recognised rising groundwater and salinity as the number one issue in their plan. The Department of Land and Water Conservation successfully obtained Salt Action funding for a groundwater project to raise awareness of groundwater related salinity issues among all landcare groups in the eastern Murray catchments and to provide groundwater levels, salinity hazards and recharge information to their catchment planning process. The current study covers an area of about 9000 km^2 in West Hume, Culcairn, Doodle Coomer, Mullengandra, Bowna, Bungowannah, Fowlers Wagra, Holbrook, Jingellic, Tooma and Tumbarumba landcare areas.

Dryland salinity occurs where groundwater outcrops (or near surface) at the land surface and due to the processes of evaporation and transpiration salts are concentrated. This occurs when groundwater levels rise and mobilise salts previously stored at depth in soil and weathered rock regolith. Of principal concerns to Murray Darling Basin Commission is the effect of increased baseflow and salt load to surface streams.

OBJECTIVES OF THE STUDY

- Raise awareness of groundwater related salinity issues in dryland catchments, and establish a partnership between DLWC and the community for a bore monitoring program

- Provide technical information in relation to watertable levels, salinity hazard and recharge potential to assist landcare groups in their catchment plans

- Establish a groundwater monitoring network consisting of bores and piezometers for continuous monitoring

PROJECT OUTCOMES

- Depth to watertable maps
- Groundwater pressure level maps
- Groundwater levels and pressure level trends
- Groundwater salinity maps
- Salinity hazard maps
- Recharge potential maps
- A monitoring network and continuous water level monitoring by community groups

OVERVIEW OF GEOLOGY AND HYDROGEOLOGY

There are two broad groups of geological formations where groundwater occurs.

1. Alluvial and other superficial deposits of clay, silt, sand, gravel and hill wash sediments: These material occupy the valley floors and are concentrated in the lower parts of the catchments along or close to the present creek systems. Thickness of these sediments is fairly low (few meters) in the project area except in the Billabong Creek where the thickness can be as deep as 60m. In general, permeability is relatively high and transmit water easily in sand and gravel layers but flow is retarded in clay and silt.

2. Rock Formations: These are hard rocks mainly granites, quartzites, slates, phyllites, schists and igneous rocks. They occur throughout the area, with granites and slates being more extensive but are commonly obscured by soil or unconsolidated material. Granite is a massive rock with less fissures and fractures and is often heavily weathered and overlain by a thick regolith cover. In contrast, slates consist of more fractures and fissures and have a potential for high recharge. Also a relatively thin regolith cover overlies fractured slates. In hard rocks, water can move through an interconnecting network of fractures towards low areas. In general hard rock permeability (fractured) is less than that of the alluvium cover.

The water levels generally follow the surface topography. As a consequence of the difference in permeability of the rock aquifers and unconsolidated deposits, and the relative position they occupy in the landscape, the latter act as a drainage system for water in the area. Groundwater drains from the rock aquifers in the upper part of the catchment into unconsolidated material in the lower part of the catchment which provides a path out of a local system. However, when the recharge in the upper area is higher than the volume that the flow system can handle (e.g. prolong wet periods, lack of deep rooted vegetation), obviously the groundwater system is imbalanced and therefore rising watertables and waterlogging can occur.

GROUNDWATER MONITORING

The approach

The study is based primarily on monitoring undertaken by landholders measuring their own private bores/wells. As there are very few government bores in the Eastern Murray, the project sought the support of landcare network and interested landholders to obtain regular readings across the landscape. The project reports to a Community Reference Group twice a year. The Group consisting of DLWC Project staff, a community representative from each landcare area and LWMP area, the Regional Landcare Co ordinator (Chair), and all landcare co ordinator as observers.

Monitoring Network

The monitoring network consists of :

- Any sites landholders have been willing to measure,
- A relatively smaller number of piezometers installed during the project, also measured by landholders.

Landcare groups were contacted and supplied with maps of their areas showing registered bore/well sites. Groups were asked to encourage landholders to participate in monitoring (registered or unregistered sites) and to confirm the availability of sites for the network, and some site details including bore depth. When this information had been collated, monitoring sites were plotted to determine where gaps needed to be filled. Sites were classified as deep (measuring pressure) or shallow (measuring watertables).

Piezometers were located in catchments with the aim of enhancing the private monitoring network, and then extrapolating this site data across the landscape with the preparation "depth to watertable" maps. Piezometers were sited with the assistance of geology maps, SCS land capability mapping (pseudo terrain/slope), existing private sites and easy access. Piezometer locations were then fine tuned by again approaching landcare groups determining the availability of interested and supportive landholders. Ninety-one (91) piezometers were installed on this basis building a shallow bore site intensity of 1 site/3-5,000 ha.

Monitoring

Site monitoring recommendations were quarterly for all sites, and monthly for wells and piezometers between April and October, for the 1998 calender year. Spreadsheets were provided to each landcare/subcatchment area collating site information and landholder contact details. A calendar highlighting recommended monitoring times, readings collation dates and the feedback periods were provided to each participating landholder. Measuring tapes/ploppers were made available to all landholders at a subsidsed cost.

Data Collation

Data collation has been left primarily the responsibility of landcare members. During consultation with landcare groups in setting up the monitoring network, landcare co-ordinators, and some areas, subcatchment landholder contacts, became very committed to the project. The benefits of this local ownership has been recognised and encouraged by implementing a reward system for groups achieving set target of number of readings. Appropriate background information and monitoring readings for depth and salinity (EC) have been entered into a custom designed Microsoft ACCESS database.

Extrapolation of Site Data

The main physical project outcome is the "depth to watertable" map for each sub-catchment. Previous reconnaissance bore studies have been criticised for being inaccurate and over estimating the future risk of high watertables and salinity as they did not take account of topography .

In the Eastern Murray study, topography and geology have been an integral part of using site data to prepare maps. To facilitate this, site data for watertable depth and EC has been printed on plastic. The hydrogeologist and the land resource officer then laid these prints over appropriate mapping of terrain/slope, geology and existing salinity, and together draw in by hand particular contour lines for depth to watertable and EC., for each sub-catchment.

Feedback to the Community

One of the foundations of obtaining community support for the project has been our commitment to provide regular and timely feedback to the community and individuals involved. This has involved:

- maintaining close links and providing support for the landcare co-ordinators and subcatchment contacts
- regular six monthly meetings of the Community Reference group
- nominating feedback periods on the monitoring calendar, and delivering on time
- providing individual hydrographs of sites to landholders
- presentations to each group of trends and "depth to watertable" maps.

Current State of the Study

The project remains in the data collection phase. Although the project set out to monitor and report on just the 1998 season, most of the Eastern Murray received less than 50% of the annual average rainfall during our monitoring year. Consequently DLWC and the community, decided to continue the monitoring over the 1999 season and assess those results.

However several statistics and trends evident from the 1998 (drought year) study are worth noting here.

- in October 1998, up to 130 landholders measured 200 sites for water depths in the Eastern Murray (NSW)

- deep bore pressure trends appear stable in some catchment but continue to rise in others at up to 0.6 m/year (1989-98),
- shallow watertable depths and groundwater salinity appears to vary significantly between and within sub-catchments (eg Doodle Cooma- no watertable within 10 m, but Major's Creek sub-catchment - watertable is shallower than 2 m for around 15% of the total area. Salinity (EC) of groundwater measured in piezometers (no surface expression of salinity) varies from 0.1 to 14.9 ds/m).

Ecologically Sustainble Opportunities for the Lower Murray Darling Area

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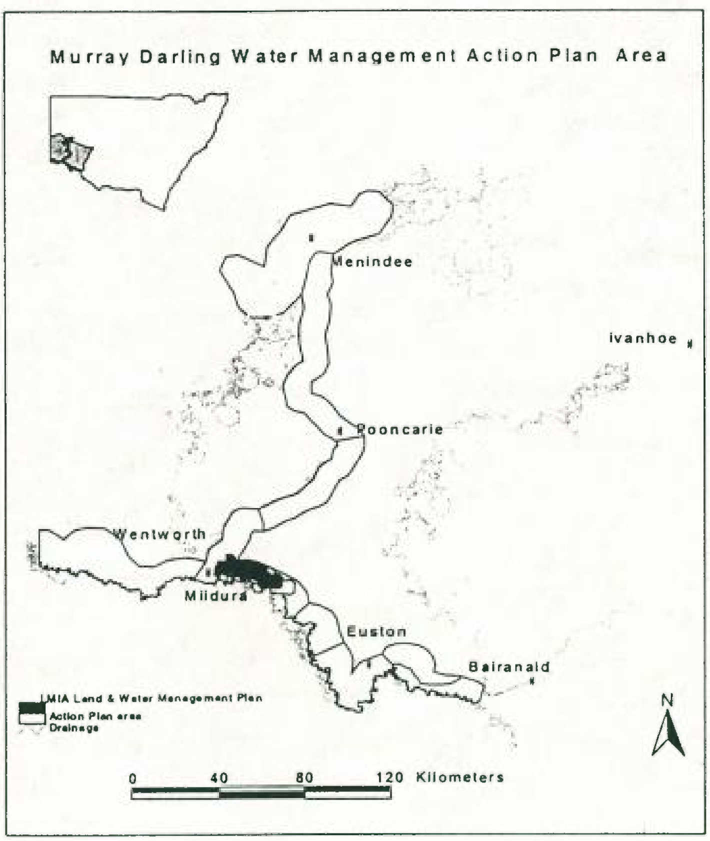
1. Introduction

The community of the Lower Murray Darling area of South Western NSW has been working together with the State and Federal government agencies in developing the Murray Darling Water management Action Plan (MDWMAP). The aim of the Plan is to achieve a sustainable water use industry for the Lower Murray Darling area. This paper describes an integrated research process that identified significant sustainability enhancement and development opportunities.



2. The Area of the Plan and Land Use

The Plan includes the parts of the Lower Murray Darling area that are currently under irrigation or have the potential to be irrigated in the future. It does not include the Western Murray Irrigation area or the Lake Victoria area as separate plans are being developed for these areas. The Action Plan area is about 885,600 ha, of which 91% is under native vegetation, parks and rangeland. 76,800 ha is cleared and used for dryland cultivation. A total of 28,000 ha of land are irrigated, of which 14,000 ha are intensively irrigated. Crops of the intensive irrigation areas are citrus, vines, stone fruits, vegetables, cereal and cotton. Cotton (5,000ha) is grown on one property at Menindee.

3. Studies of the Plan

A number of studies have been conducted to determine the extent and importance of issues identified by all interested parties. These studies are:

- Flora and Fauna survey and assessment of the conservation status of species in the Plan area
- Agricultural Resources
- Water Resources and scope for water savings
- Drainage impacts and options for effective management
- Hydrogeological investigations and opportunities for the future
- Best management practices
- Identification of Sites of cultural heritage and their protection
- Economic Profile of the area and the scope for improvements, and
- Regional land use study

Most of the above studies are interdependent. Hydrogeology, drainage, flora and fauna studies and the agricultural resources study formed the basis of he Plan. This paper addresses the findings of the Hydrogeology and Land use studies. All study outcomes are stored in a GIS format for the implementation of Plan options.

4. Hydrogeology Study (Extracted from reports prepared by SKM, 1999)

The groundwater of the Lower Murray Darling area is generally saline. The salinity varies from about 300 EC to more than 50,000 EC. Generally the groundwater salinity is low near the River. Mobilisation of this saline water has the potential to cause salinity impacts to the River, crops and vegetation. To assess the current groundwater condition and its flow and impacts, a groundwater modelling study was undertaken in two stages. -

<u>4.1 Stage 1</u>

Stage I determined the future impacts over periods of 30, 50 and 100 years under a no intervention scenario (ie if the current management and land use continue). The Stage II predicted the impacts over periods of 30 and 50 years for various scenarios of irrigation development. The 100-year time frame was not considered useful for the Stage II assessment due to the lack of data required for such a long term prediction.
Groundwater models were developed for each of eight sections of the Plan area. The groundwater flow modelling package MODFLOW was used for the development of the models. The results were reported in terms of increase in the Murray River salinity in EC

units at Morgan in South Australia (table 1 below) and rise in watertables in the model area.

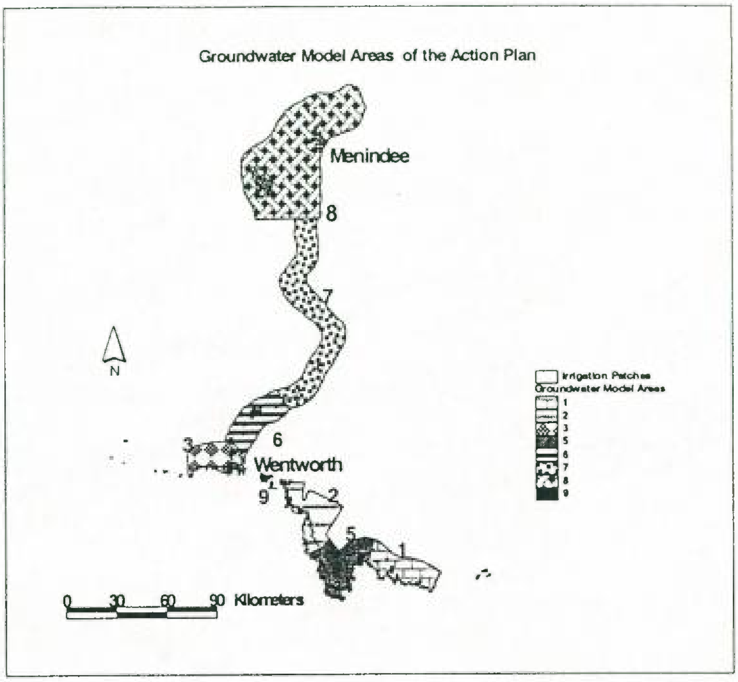


Table 1. Estimated Salinity Impacts to the River Murray (No Intervention Scenario)

Model No.	Salinity Impact at the end of calibration period (EC units)	Increase in Salinity impact in 30 years, (EC units)	Increase in Salinity impact in 50 years, (EC units)	Increase in Salinity impact in 100 years, (EC units)
1	0.53	0.01	0.02	0.02
2	3.8	3.6	4.0	4.2
3	2.05	0.01	0.03	0.06
5	7.3	0	0	0.1
6	<0.1	-	-	-
7	<0.1	-	-	-
8	<0.1	-	-	-
9	2.01	0	0	0

Note: the sign '–' has been used instead of any value. This is due to the fact that the estimated impact was less than 0.1 and the difference from the no intervention scenario (which was reported as <0.1) could not be calculated.

Model areas 1 to 5 and 9 are along the River Murray, and model areas 6 to 8 are along the Darling River.

The rise in watertables varies from model to model and is between 0.5 and 3m in most of the model areas after 100 years. At present, watertables are between 5 and 20 metres below the surface, so a rise of 2 to 3 metres is not considered to be of threat to agricultural crops and vegetation. However, this rise can have impacts in the model 9 area's lagoon system.

4.2 River Regulation

Groundwater flow in most of the model areas has been found to be away from the river and is attributable to weir pools created by the River regulation. This can be easily proven as bores at some distance from the River show fluctuations with the flood levels and there is a gradient of observation bore water levels away from the river. The rise of watertables in areas near weir pools has been predicted to affect vegetation in the floodplain of some model areas.

4.3 Calibration of models

The developed groundwater models were calibrated using the trend of water level changes in observation bores and matching the predicted water levels with the observed data.

The Action Plan Steering Committee wanted to use some form of standard calibration technique. This was to make the developed models comparable to others already developed and used in various catchments of the Murray River. This is important as salt enters the River from the groundwater system. Unfortunately, such a standard was not available. Therefore, the Steering Committee set the calibration criteria in consultation with the hydrogeologists and modellers of the Department of Land and Water Conservation (DLWC) and the consultant who undertook the study. The need for a formal peer review process and modelling standard documentation became evident through the consultancy.

DLWC is now working with experts of different States and the Murray Darling Basin Commission to develop calibration standards for groundwater modelling and a peer review process. This can help the industry and community to better understand and manage the groundwater and surface water resources.

4.4 Stage II

The outcomes of the Stage I Hydrogeology Study and the Flora and Fauna studies were used in Stage II to predict the future environmental impacts for different management scenarios.

Irrigation development scenarios of 1,000 ha, 5000ha and 10,000 ha were tested for a number of model areas. In addition, a scenario of reduction in depth of weir pools was considered. The weir pool scenario assumed a level of a metre lower than the normal operation levels during four months in the winter (May to August) when the irrigation demand is low and all other recreational activities are also limited.

The above-mentioned scenarios were tested for model areas where land is already under dryland cultivation. The salinity impacts to the Murray River after a period of 50 years are presented in Table 2 as the difference between the impacts for a development scenario and a "no intervention scenario" for the same time frame.

Table 2. River Salinity Impacts Under Various Irrigation Scenarios Over 50 years

Model No.	EC impacts, no intervention scenario	Change in EC impacts, 1000 ha irrigation development scenario	Change in EC impacts, 5000 ha irrigation development scenario	Change in EC impacts, 10,000 ha irrigation development scenario	Change in EC impacts for a metre change in weir pool level
1	0.5	0	0.1	0.3	0.2
2	7.8	0	0	0	6.0
5	7.3	0	0	0.1	-1.1
6	<0.1	0	0	0	0
7	<0.1	0	0	0	0
9	2.0	-	-	-	1.1

With up to 5,000 ha of new irrigation development in all areas except model 1, the rise in watertables has been predicted to remain around 0.5m in a 50 year time frame. In model 1 a rise of up to a metre would occur in 50 years with 5000 ha development and 0.3 m with 1,000 ha development. All models have been calibrated to the accuracy of +/- 0.3m of water level predictions. Therefore, a rise in watertable by 0.5 m can be considered manageable with the implementation of Best Management Practices. On the other hand there will not be enough water to undertake up to 5,000 ha of development in every model area and the community believes that the practical development scenario is between 1,000 ha and 2,000 ha for each model area.

A soil and land suitability study for the Lower Murray Darling area has been undertaken for potential development areas as indicated by the Hydrogeology Stage II study.

5. Land Capability

Soil suitability assessment is essential to maximise production and minimise environmental impacts. This paper describes a method used in the determination of soil suitability and land capability for perennial horticulture in the MDWMAP area. Best management practice in land use has lead to the determination of soils that are suitable for different crop types. Knowledge of soil physical and chemical characteristics used in suitability assessment is useful in deciding management practices to maintain its productivity and sustainability.

5.1 Method of suitability determination

There are many methods of determining soil suitability. Due to the vastness of the Action Plan area a non-intensive method was required to be used. The use of electro-magnetic or remote sensing methods needs to be supported by good ground-truthing results. In some cases, remote sensing methods do not provide accurate outcomes where assessments are needed for soils of greater depths in the profile. To select a method acceptable to all parties, the MDWMAP Steering Committee conducted a workshop with CSIRO, local experts from NSW and Victorian State agencies, and the Community. A rule-based method was adopted in the workshop. A method of study was then determined and the criteria for soil suitability assessment were decided.

The method used for the Murray Darling Water Management Acton Plan was a two-stage process. In Stage I, Anderson et al (1998) conducted a desktop soil assessment using the Soil and Landform Association of Eldridge (1985) and Walker (1991). In the Aeolian landscapes of South Western NSW, the presence and form of carbonate layers limit the choice of horticultural crops, Wetherby (1992). The depth of soil between the surface and the carbonate layer, determines the type of crop that can be grown. This is shown in table 3 below for some perennial horticultural crops.

Table 3. Topsoil depth suitability for horticultural crops

Crop	Depth of Top Soil
Almonds	40-60 cm
Avocado	80 cm
Citrus	50 cm
Grape vines[1]	30 cm
Stone fruit[1]	30-40 cm
Olives[1]	30 cm

The other criterion used for this assessment is the presence of clay. Clay topsoils restrict infiltration and drainage and were not considered suitable for development. Also the presence of medium and heavy clay within a metre from the soil surface represent drainage hazard.

[1] Lime tolerant crops

An analysis of soil suitability and land capability is given for the map units in table 4.

Table 4 A typical Land Capability and Soil Suitability assessment

Map unit	Land Unit	Text	% Ass	Principal Soil Profiles	Unit Cap	Soil characteristics	Soil Suitability	Comments
A1	Parallel Dune	I	40	Uc1.13	Y	2%CO3 at 0.38m, loamy-coarse sand	y	all crops
				Uc1.23		no CO3	y	all crops
				Uc1.11		light soils, mild CO3	y	all crops
				Uc5.12		light soils, 2% CO3	y	all crops
				Uc5.13		no CO3	y	all crops
A1	Swale	I	55	Gn2.13	Y	light clay 0.55m - drainage concern	n	
				Db4.53		heavy soils	n	
				Db1.53		heavy soils	n	
A1	Depression	I	5	Gn2.13	N	light clay 0.55m - drainage concern	n	
				Gc1.12		high CO3 above 0.3m	y	lime tol. possibly
D	Flood plain	V	90	Uf1.33	N	heavy CO3	n	
				Uf6		heavy clays	n	
D	Plain	IV	10	Gn2.13	Y	light clay 0.55m - drainage concern	n	

Findings of the desktop analysis were groundtruthed in the stage II (NRE, 1999) of the study. Other land suitability factors were taken into account in Stage II. These are areas of low conservation value and low cultural heritage significance. Survey sites were selected at an approximate rate of a site for every 50 to 100 ha. Samples were also collected for lab analysis, and the following assessment was conducted to determine the land suitability for different crops.

i. pH, salinity and boron content.
ii. Soil texture and structure
iii. Carbonate class of the soil and reaction to acid (1N HCL)
iv. Assessment of relative recharge risk (hydrogeological assessment)

The above soil assessment was conducted to facilitate regional planning. More intensive soil surveys will be required for soil suitability determination for crop selection and irrigation system design at the development stage.

6. Conclusion

The above studies and the planning process have created enormous interest in the community for sustainable management of natural resources and to invest in environmental management. This community enthusiasm will need to be utilised for environmental and river health benefits by allowing an efficient process that will allow

- i transfer of water from low value enterprises in high impact areas to high value enterprises in low impact areas.
- ii commitment to use best management practices as a part of licensing agreement
- iii development taking place in areas away from the floodplain and other high impact areas
- iv wool producers that are suffering a cost-price squeeze, to diversify
- v economy of scale for community infrastructure (supply system and drainage reuse facilities) to minimise environmental impacts.

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1999 Murray Darling Basin Groundwater Workshop
Conference Proceedings

CONTENTS

KEYNOTE PAPERS:

KEYNOTE PAPERS

Murray-Darling Basin Groundwater Workshop 1999

ABSTRACT

Policy Implications for COAG Water Reforms on Groundwater

Mike Smith
Executive Officer, High Level Steering Group on Water

The COAG Strategic Water Reform Framework, agreed by all jurisdictions in 1994, applies equally to groundwater as it does to surface water. In addition, a suite of 12 recommendations specifically dealing with groundwater issues were formally incorporated into the extended COAG Strategic Water Reform Framework in 1996.

The paper outlines the origins of the initiative, summarises the 1994 COAG water reforms and discusses the 12 specific groundwater reforms agreed in 1996.

THE AUSTRALIAN WATER REFORM AGENDA

National Inter-governmental Coordination

- **Council of Australian Governments (COAG)**
  - **Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ)**
    - Standing Committee on Agriculture and Resource Management (SCARM)
      - Supporting Committees and Working Groups
  - **Australian and New Zealand Environment and Conservation Council (ANZECC)**
    - Standing Committee on Environment Protection (SCEP)
      - Supporting Committees and Working Groups
  - **Murray-Darling Basin Ministerial Council (MDBMC)**
    - Murray-Darling Basin Commission (MDBC)
      - Supporting Committees and Working Groups

So What is "COAG" ?

- Council of Australian Governments
- Prime Minister, Premiers & Senior Ministers from all States and Territories
- Highest forum in Australia
- Broad agenda but WATER is major component

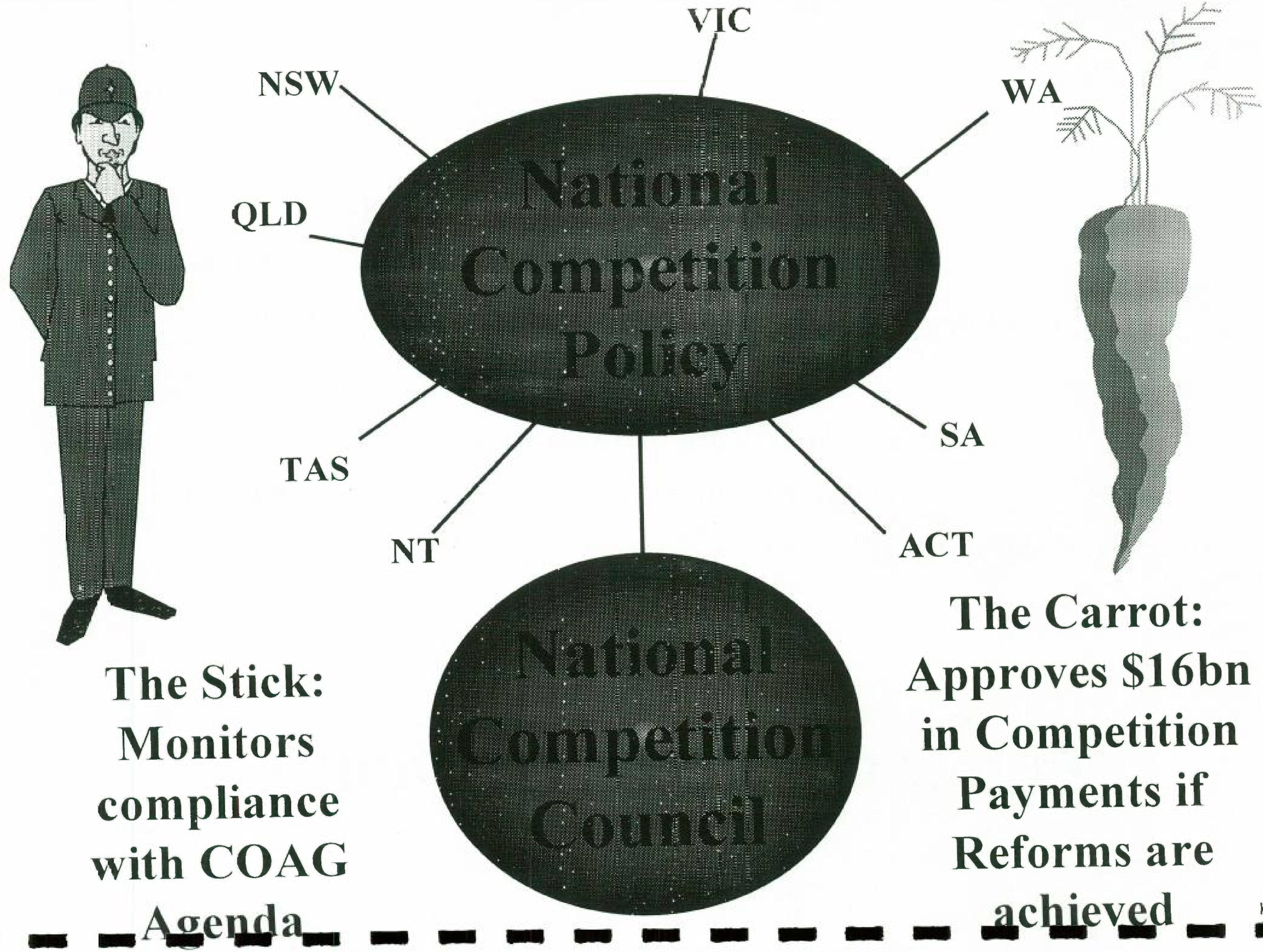
COAG Water Reform Agenda

- Comprised of two parts:
  - COAG Strategic Framework - defines specific water industry reforms *(Report of the Working Group on Water Resource Policy, 1994)*
  - National Competition Policy - defines non industry specific reforms *(Hilmer Report)*

National Competition Policy Reforms

- Extending competitive conduct rules to all business activity in Australia (including Government businesses)
- Third party access to significant infrastructure
- Introduction of competitive neutrality
  - no unfair advantage for government businesses
- Extending prices surveillance of monopolies
- Restructuring of public sector monopolies

6


VIC
NSW
WA
QLD
National Competition Policy
TAS
SA
NT
ACT
National Competition Council
The Stick: Monitors compliance with COAG Agenda
The Carrot: Approves $16bn in Competition Payments if Reforms are achieved

Objective of COAG Reforms

By 2001, achieve a water industry that is economically efficient and ecologically sustainable, and which delivers better environmental outcomes

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The COAG Strategic Framework for Water Industry Reform

Five Elements:

Element 1:
Cost Recovery and Pricing

- consumption based pricing
- full cost recovery
- cross subsidies removed or made transparent
- identifying Community Service Obligations (CSO's)
- new/extended irrigation schemes required to be both economically viable and environmentally sustainable
- consistent pricing and asset valuation to facilitate trading

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Element 2:
Institutional Reform

- **Institutional Separation**
  - **Separate roles/functions of service providers from regulators, natural resource managers and standard setters**
- **Performance Monitoring and Best Practice**
- **Commercial Focus for Water Services**
- **Devolve Irrigation Management**

Element 3:
Water Allocation and Trading

- **Allocations and Entitlements**
  - water allocations separated from land title
  - environment is a legitimate user or water
- **Trading**
  - water transferred to highest value end-uses to ensure maximum benefit
- **Groundwater**

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Element 4:
Environment and Water Quality

- Integrated Resource Management
- National Water Quality Management
- Wastewater/Stormwater Management

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Element 5:
Public Consultation and Education

- **Public Consultation**
  - agencies and service deliverers to consult with the public when change is proposed
- **Public Education**
  - jurisdictions to develop programmes in relation to water usage and benefits of reform
  - water agencies to work with schools
  - water agencies to explain the total water system
  - water agencies to provide level of service that provides best value for money to community

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Implementation

- State and Territory Governments are responsible for implementing the COAG Water Reform Agenda
- Progress has been made by all jurisdictions in implementing the key elements of the reforms

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That ubiquitous “someone” once said:

“We are continually faced with OPPORTUNITIES brilliantly disguised as insoluble PROBLEMS”

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Will the Reforms Produce Results ?

- YES !!
- WHY ??
- States demonstrating will to achieve reforms

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For Example:

- Institutional arrangements (separation, corporatisation, privatisation)
- Trade (intra and inter state)
- Better allocation process (WAMP, MDBC Cap)
- Clearer definitions of Water Property Rights
- Education & Community Involvement
- Pricing Reforms (IPART, MDBC Water Business)
- Third Party Access
- Benchmarking
- etc etc etc

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1994 COAG FRAMEWORK:

- Clause 3 (e) Groundwater:
  - (i) that management arrangements relating to groundwater be considered by ARMCANZ ... and advice from such considerations be provided to individual jurisdictions and the report be provided to COAG.

ALLOCATION AND USE OF GROUNDWATER

A National Framework for Improved Groundwater Management in Australia

12 Key Reforms:

1. **SUSTAINABILITY**
   – … employ the principles of ecologically sustainable development and should be directed at achieving sustainable use of the resource.

2. **DRILLING**
   – … adopt the National Drillers Licensing system for water production wells … and should seek to expand the system to all drilling.

3. **GROUNDWATER/SURFACE WATER MANAGEMENT**
   – Groundwater and surface water resource management should be better integrated, including approaches to pricing …, water allocations and trading …

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12 Key Reforms:(cont)

6. SUSTAINABLE YIELD/ALLOCATION/USE LINKAGE

- In developing groundwater management plans, ... those plans include identification of the sustainable yield and the levels of allocation and use of the aquifers. These plans should also include an identification of environmental water provisions .... Where allocations exceed the sustainable yield, ... develop strategies to reduce abstractions to sustainable levels within timeframes that minimise permanent damage to the resource.

4. INEFFICIENT WELL DESIGN

- In preparing groundwater management plans, policies and strategies, States should ensure that the effective utilisation of groundwater resources is not compromised by protection of existing users with inefficiently designed or constructed wells. This particularly applies to domestic and stock wells.

12 Key Reforms:(cont)

5. WATER MARKETS
   - … groundwater plans (should be) based on a sound understanding of the resource. These plans should be the primary support for the development of groundwater allocation and property right systrems to support intra-aquifer trading both within the States and across State borders.
7. AVAILABILITY OF WELL CONSTRUCTION DATA
   - … provision … of well construction data for all wells should be a mandatory requirement …
8. INFORMATION FROM HIGH YIELDING WELLS
   - … high yielding wells … should be monitored to … ensure adequate information is available to manage the resource sustainably…

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12 Key Reforms:(cont)

9. PRICING

- … full cost of groundwater management should be identified by the States. … cost of direct management … should be recovered from the users … apportionment of indirect costs be given consideration. … remaining subsidies should be transparent. Public communication on these matters will be important.

10. FEDERAL GOVERNMENT EXPENDITURE

- The Federal Government should identify its full costs of involvement in groundwater activities to assist the negotiation of priorities for Commonwealth funding of groundwater management activities.

11. INSTITUTIONAL ARRANGEMENTS

- State and Federal agencies should … eliminate conflict of interest situations in groundwater assessment and management.

12. EDUCATION

- SCARM should assess the opportunities for increasing public awareness of the value of groundwater, its vulnerability … and the need for groundwater management … and to develop appropriate awareness programs.

1996 COAG FRAMEWORK:

- Clause 3 (e) groundwater -
  - management arrangements for groundwater relating to sustainability of the resource, licensing of drillers, better integration of groundwater and surface water management, inefficient well design, environmental allocation, data availability and pricing are to be implemented by jurisdictions taking guidance from the recommendations at Annex A

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PERSPECTIVES ON SUSTAINABLE DEVELOPMENT OF WATER RESOURCES: A USA GREAT PLAINS OUTLOOK

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Abstract

This paper concentrates on the hydrologic fundamentals of the sustainability concept of safe yield and points out its shortcomings. It also indicates how these hydrologic underpinnings can be used for developing a sound water-use planning policy for stream-aquifer systems. The paper also addresses the more general concept of sustainability from the systems perspective and outlines our still-evolving ideas on environmental sustainability. The Kansas water resources management experience and its evolution towards sustainability are then outlined. Misguided "safe yield" rules, such as pumping the natural recharge, will not lead to a desirable economic or stable level of ground-water development. Because of the interdependence of surface water and ground water, operations on any part of the system have consequences for the other parts. Therefore the importance of integrated resource planning and management is stressed. The paper concludes by stressing the need for long-time education supported by research and technical assistance, as well as improved communications, so that people become more conscious of the complexities and constraints involved in water-resources management.

Introduction

The time has passed when abundant supplies of water were readily available for development at low economic, social, and environmental cost. According to Hufschmidt (1993), now we are entering the period of a "maturing water economy," with increasing competition for access to fixed supplies, a growing risk of water pollution, and sharply higher economic, social, and environmental costs of development. It should be well understood that most of the developable water in the western U.S. states, including Kansas, has been developed, and future water management is going to be heavily dependent on obtaining more mileage out of existing supplies. The great challenge facing the world today is how to cope with the impact of economic growth on environmental processes. The World Commission on Environment and Development (1987), better known as the Brundtland Commission, defined sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs."

Water is not only essential to sustain life, but it also plays an integral role in ecosystem support, economic development, community well-being, and cultural values. How all these values, which sometimes conflict, are to be prioritized, which are to be sustained, and in what fashion, are still unresolved questions (Gleick et al., 1995). The concept of sustainable development is intended to provide a framework within which the environment can be properly managed to support economic development while providing adequate resources for the future. This has lent weight to arguments for proactive rather than reactive environmental policies. However, although progress has been made in defining the goals of sustainable development, the mechanisms to bring about these changes are still a matter of debate. The challenge of our times is to turn the principles of sustainable development into achievable policies that lead to concrete change. Science can assist by exploring the implications of different interpretations of sustainability. Although science cannot say that one particular interpretation is the "correct" one for society, sustaining solutions will have to be based on fundamentally sound hydrologic analyses and related technology.

Hydrologic fundamentals underlying safe yield and ground-water depletion

To protect ground-water supplies from overexploitation, state and local agencies in Kansas and other states enacted regulations and laws based on the sustainability concept of "safe yield" (Sophocleous, 1998b). Safe yield is commonly defined as the attainment and maintenance of a long-term balance between the amount of ground water withdrawn annually and the annual amount of recharge.

Therefore, safe yield allows water users to pump only the amount of ground water that is replenished naturally through precipitation and surface water seepage, generally known as natural recharge. But over the long term under natural or equilibrium conditions, recharge to an aquifer results in an equal amount of water being discharged out of the aquifer into a stream, spring, or seep. Consequently, if pumping equals recharge, the streams, marshes, and springs eventually dry up (Sophocleous and Sawin, 1997). Continued pumping in excess of recharge also will eventually deplete the aquifer. This is what happened in several locations across the Great Plains of the USA (Sophocleous, 1998c). Probably the best-known example is the Ogallala or High Plains aquifer, where declines of more than 30 m are common in parts of Texas, New Mexico and Kansas (Gutentag et al., 1984). Maps comparing the perennial streams in Kansas in the 1960's to those of the 1990's show a marked decrease in kilometers of streamflow in the western third of the state (Sophocleous, 1998a).

To better understand this depletion, a thorough knowledge of the hydrologic principles (concisely stated by C.V. Theis in 1940) is required. Under natural conditions, prior to development by wells, aquifers are in a state of approximate dynamic equilibrium: over hundreds of years, wet years in which recharge exceeds discharge offset dry years when discharge exceeds recharge. Discharge from wells upsets this equilibrium by producing a loss from aquifer storage; a new state of dynamic equilibrium is reached when there is no further loss from storage. This can only be accomplished by an increase in recharge, a decrease in natural discharge, or a combination of the two.

Consider a stream-aquifer system such as an alluvial aquifer discharging into a stream. (Please note that I use the term "stream" in the broadest sense of the word; the issues, approach, and results also apply to rivers, lakes, ponds, and wetlands.) A new well drilled at some distance from the stream and pumping the alluvial aquifer will cause a cone of depression to form. The cone will grow as water is taken from storage in the aquifer. Eventually, however, the periphery of the cone will arrive at the stream. Then a difference will be produced between the head of the water in the stream and the head just inside the edge of the cone of depression. Water will either start to flow from the stream into the aquifer or cease to flow from the aquifer into the stream. The cone will continue to expand with continued pumping of the well until an equilibrium is reached in which induced recharge from the stream balances the pumping. Because the stream is the source of recharge, the cone will expand until its periphery along the stream is long enough, and the head gradient is sufficient, to cause flow from the stream into the cone that is equal to the rate of pumping from the well.

The length of time, t, before an equilibrium is reached depends upon the aquifer diffusivity, which is a measure of how fast a transient change in head will be transmitted throughout the aquifer system (and is expressed as the ratio of aquifer transmissivity to storativity, T/S), and upon the distance from the well to the stream, x.

Once the well's cone has reached an equilibrium size and shape, all of the pumping is balanced by flow diverted from the stream. Eventually there is no difference between a water right to withdraw ground water from the well, as described, and a water right to divert from the stream at the same rate. A crucial point, however, is that before equilibrium is reached (that is, before all water is coming directly from the stream), the two rights are not the same (DuMars et al., 1986). Until the perimeter of the cone reaches the stream, the volume of the cone represents a volume of water that has been taken from storage in the aquifer, over and above the subsequent diversions from the river. It is this volume that may be called *ground-water depletion*.

Thus, ground-water sources include ground-water storage and induced recharge of surface water. In contrast to *natural recharge*, which refers to water that moves through the ground water system under the boundary conditions imposed by natural topography, geology, and climate, *induced recharge* is surface water added to the natural ground-water system in response to such artificial boundary conditions as those imposed at well-fields, drains, recharge basins, or reservoirs. The timing of the change from storage depletion (or mining) to induced recharge from surface water bodies is key to developing water-use policy (Balleau, 1988).

The shape of the transition curve from storage depletion to induced recharge for a two-dimensional, homogeneous and isotropic system is shown in fig. 1 in nondimensional form based on Glover's (1974) analytical solution and tabulation, where the percent of ground-water withdrawal derived from ground-water storage is plotted against dimensionless time. This general shape of the transition or growth curve is retained in systems with apparently different boundaries and parametric values (Balleau, 1988). The rate at which dependence on ground-water storage (as shown at the left portion of the graph) converts to dependence on surface water depletion (as shown on the right portion of the graph) is highly variable and is particular to each case.

The initial and final phases of the growth curve (fig. 1), representing mining on the left and induced recharge on the right, are separated in time by a factor of nearly 10,000. This means that full reliance on indirect recharge takes an extremely long time. The distinct category of ground-water mining depends entirely upon the time frame. All ground-water developments initially mine water and ultimately do not (Balleau, 1988).

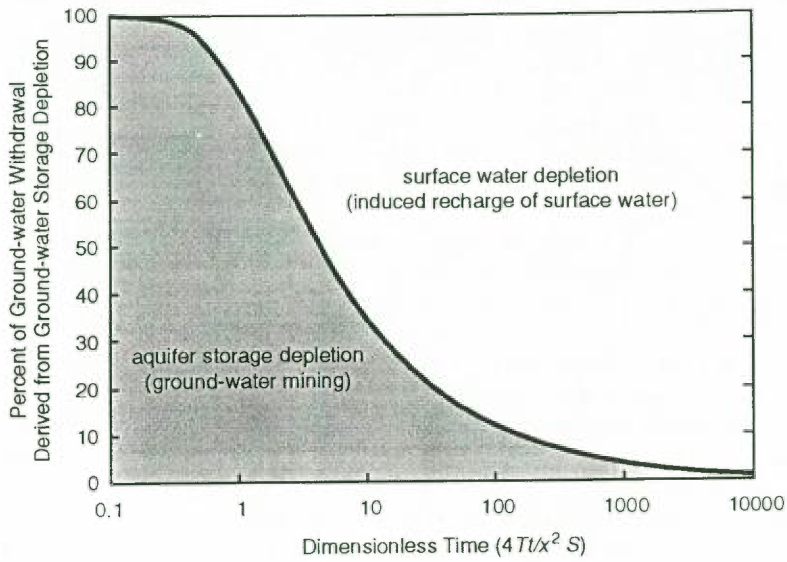

Figure 1. Transition of sources of water to wells from reliance upon ground-water storage to induced recharge of surface water. See text for notation used.

Limitations of safe yield

The concepts of safe or sustainable yield are often associated with a single-product exploitation goal—the number of trees cut, the number of fish caught, the volume of water pumped from the ground or river, year after year, without destroying the resource base. But experience has shown, over and over, that a single-product goal is too narrow a definition of the resource, because other resources inevitably depend on or interact with or flow from the exploited product. We can maximize our so-called sustainable yield of water by drying up our streams—but when we do, we find that the streams were much more than just containers of usable water.

The conventional safe yield approach is ambiguous, limited, and restrictive (Sophocleous, 1997). Any change in conditions, such as changes in vegetation, land use, urbanization, location of pumping wells, or incorporation of new water supplies would require calculation of a new yield. Clearly, no unique and constant value can be attached to safe yield. The term is also limited because of its failure to address impacts of ground-water exploitation on related surface water and on the aquifer's natural discharge areas, which might be outside the area of development.

The failures and unintended consequences of conventional and safe-yield approaches to water management provide some of the strongest incentives for retiring the concept. Such failures can have consequences ranging from local to global (Sophocleous et al., 1998).

For example, ground-water pumping has dried up or threatened numerous reaches of baseflow-dependent streams, wetlands, and subirrigated land—with many examples found in Kansas along the fringes of the High Plains aquifer, and in other states.

Irrigation has contaminated the land in many areas. Increases in consumptive water use leave behind the salts dissolved in the water. For example, irrigation drainage water contaminated the ponds at Kesterson National Wildlife Refuge in California with toxic levels of selenium (NRC, 1989). Saline water from irrigation return flow into the Upper Arkansas River basin now threatens the ground-water resources of the alluvial and Ogallala aquifers in Kansas. The Kansas Geological Survey (Whittemore, 1995) is now embarked on a multi-year study to analyze the impact of Arkansas River salinity on the underlying alluvial and Ogallala aquifers in western Kansas resulting from irrigation return flow in Colorado.

Part of water's role in the natural cycle is the transport of sediment, which it does best during floods. As streamflows are reduced and controlled, the sediment loads are not flushed; they gradually fill reservoirs, seal off stream channels from the alluvial aquifers—and starve the downstream deltas.

Whole regional ecosystems change and disappear with large-scale water development—the Gulf of California has changed from an estuary to a marine lagoon as the Colorado River has been dried up. Nutrient runoff from the central U.S. has changed the ecology of the area surrounding the mouth of the Mississippi and led to the hypoxia problem in the Gulf of Mexico we witness today (Rabalais et al., 1991).

Hydrologic basis for ground-water planning policy

The effects of ground-water development that concern policy-makers are primarily aquifer drawdown and surface-water depletion. Both are fundamentally related to pumping rate, aquifer diffusivity, location, and time of pumpage. The natural recharge rate is unrelated to any of these parameters. Nonetheless, natural recharge is often used by policy-makers to balance ground-water use, based ostensibly on a steady state. As Balleau (1988) pointed out, public purposes are not served by adopting an attractive fallacy that the natural recharge rate represents a safe rate of yield.

A suitable hydrologic basis for a ground-water planning policy aimed at determining the magnitude of possible development would be a curve similar to the transition curve we saw earlier, coupled with a projected pattern of drawdown for the system under consideration (Balleau, 1988). The level of ground-water development is calculated using specified withdrawal rates, well-field locations, drawdown limits, and a defined planning horizon.

Since the 1980's, three-dimensional numerical models of the complete stream-aquifer hydrogeologic system have been used for water-rights purposes. These models provide a predictive tool explaining the connection between well-field withdrawal and surface-water depletion at particular sites. Ground-water models are capable of generating the transition curve for any case by simulating the management or policy alternatives in those terms. Specified withdrawal rates, well distribution, and drawdown of water levels to an economic or physical limit are used in the model to project the sources of water from ground-water storage and from surface-water depletion throughout the area of response. The area of response is not known in advance of such a projection. A planning horizon must be defined to assess which phase of the transition curve will apply during the period of the plan. The withdrawal rate selected in this way relies first on aquifer storage and secondly on the potential for induced recharge. The plan can contain explicit physical and economic limits on drawdown and induced recharge rates, but the analysis is unrelated to the initial natural recharge.

Evolving Sustainability Concepts

Over the past several decades, the concepts underlying management of water resources have gradually led to a recognition that nature is characterized by chance and randomness, that natural systems are inherently variable, patchy, and often require disturbance to persist (Meyer, 1993). The management implication of these realizations is that we must manage for change and for complexity. This approach dictates management in the context of the ecosystem

So, the next level of sustainable yield addresses the sustainability of the system — not just the fish, but the marine food chain; not just the trees, but the whole forest; not just the ground water, but the running streams, wetlands, and all of the plants and animals that depend on them (Sophocleous et al., 1998). Such an approach, although worthwhile, is fraught with difficulty. We cannot use a natural system without altering it, and the more intensive and efficient the use, the greater the alteration. How much is too much? What are the central characteristics that must be preserved or sustained? And is there any way to answer these questions before it is too late? This is the crux of the sustainability problem—even if we care about the next generation, do we permit things that cannot be proven dangerous or forbid what cannot be proven safe?

Science will never know all there is to know. Science is a process, not an end point. Rather than allowing the unknown or uncertain to paralyze us, we must apply the best of what we know today, while providing sufficient management flexibility to allow for change and for what we don't yet know.

The need to manage ground-water resources as an integral part of overall available water resources and in recognition of their place in the equilibrium of the natural environment is better understood today by scientific experts and water-resource managers. *Integrated resource planning* has recently emerged as a tool for total water management, "assuring that water resources are managed for the greatest good of people and the environment and that all segments of society have a voice in the process" (AWWA, 1994). This concept of enlightened management, which in the past has eluded those directly or indirectly involved in the day-to-day management operations of ground water, is now taking hold in Kansas and other states.

Re-evaluation of safe yield policies in Kansas

The "safe yield" water management program in Kansas attempts to balance ground-water appropriations in a predefined area around a proposed point of diversion, such as an irrigation well, with an estimate of recharge for that area, which is delimited as a circular area of 3.2-km radius around the point of diversion. If the total quantity of permitted and proposed pumpage within the 3.2-km circle does not exceed the average annual recharge estimated for that circle, and if some well-spacing requirements are satisfied, then a water appropriation permit is granted. Otherwise it is rejected. However, such policies did not stop ground-water-level and streamflow declines.

As a result of the failure of the so-called "safe yield" policies of the Kansas ground-water management districts (GMD's) to prevent declines in ground-water levels and streamflow, the central Kansas GMDs have recently (early 1990's) reevaluated their "safe-yield" policies. They have moved toward conjunctive stream-aquifer management by amending their "safe yield" regulations to include baseflow (that is, the natural ground-water discharge to a stream) as ground-water withdrawals along with regular water-permit appropriations when evaluating a ground-water permit application.

The concept is to prorate the baseflow to a series of phantom wells, known as "baseflow or stream nodes" located on the stream centerline at 0.4-km intervals (the GMD's well-spacing requirement), each having an annual quantity of water assigned to it equal to its prorata share of the estimated baseflow. If there are such nodes in a standard 3.2-km circle from a proposed point of diversion, such as an irrigation well, they are each treated as water rights for purposes of determining whether or not a new application should be approved.

The Division of Water Resources and the Kansas Water Office also are attempting to develop a comprehensive basinwide-management program in areas of Kansas with significant water problems. The approach taken is that of the watershed ecosystem, recognizing that streams are the products of their drainage basins and their associated aquifers (or ground-water basins), and that to understand and model such stream-aquifer interactions, it is necessary to understand the flow paths within the surface- and ground-water basins associated with the stream. The Kansas Geological Survey was also involved in this program with responsibilities in developing and applying integrated watershed and ground-water models (Sophocleous et al., 1999; Perkins and Sophocleous, 1999). Close consultation and cooperation with the local district, irrigators, and other interested parties are integral parts of this program.

Concluding Comments

Ground-water management cannot be conceived of separately from management of surface waters. Because of the interdependence of surface and ground water, operations on any part of the system have consequences for the other parts. The impact of ground-water development on streams is highly variable. The management category of minable water may be a reasonable one to apply to well-field areas that would not progress beyond the earliest stages of the transition from storage depletion to induced recharge within a reasonable planning horizon. Thus, wise management of water resources needs to be approached both from the viewpoint of focusing on the volume and quality of water resources available for sustainable use, and from the impact of ground-water exploitation on the natural environment, including ground water, surface water, and riparian ecosystems.

Our understanding of the basic principles of soil and water systems and processes is fairly good, but our ability to apply this knowledge to solve problems in complex local and cultural settings is relatively weak (NRC, 1991). Communication is vital. We need people who can transfer research findings to the field and who can also communicate water users' needs to the researchers. Our education system has mostly failed to stress the importance of sustainability in water-resources management. As Balleau (1988) commended, "Hydrology as a science has not been markedly successful in communicating its basic principles, such as mass-balance," especially in stream-aquifer systems. A water-policy study team (DuMars et al., 1986) advising the New Mexico Legislature concluded that "This concept and its ultimate impact on the environment . . . is little understood by hydrologists and lay people alike." A strong public-education program is needed to improve understanding of the nature, complexity, and diversity of ground-water resources, and to emphasize how this understanding must form the basis for operating conditions and constraints. Such long-time education program needs to be supported by research and technical assistance in order to resolve both local and regional problems. This is the only way to positively influence, for the long term, the attitudes of the various stakeholders involved.

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PLANNING & POLICY

SUSTAINABLE GROUNDWATER MANAGEMENT IN NSW, AUSTRALIA

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ABSTRACT

In accordance with the nationally adopted principles of ecologically sustainable development, the NSW Government is reforming the management of the State's groundwater resources. In the first instance, this has involved the development of a State Groundwater Policy. Policy principles relate to three core objectives: to manage the quantity of the resource sustainably; to protect its quality, and; to protect significant groundwater dependent ecosystems. The policy recognises the financial constraints on groundwater management activities, and as such adopts a risk management approach to prioritising management effort.

This paper outlines the process of policy development, as well as the specific principles contained within the policy documents. It explores the mechanisms for the practical implementation of the policy principles, with particular emphasis on the role of local, community based "Groundwater Management Committees". It also describes a number of important tools used to assist both groundwater managers and the community to manage their local resources, including vulnerability mapping, and monitoring.

1.0 INTRODUCTION

In NSW, there is ample evidence that Aboriginal people used groundwater before Captain Cook and his crew stepped ashore at Botany Bay in 1770 and dug a well in the sand for drinking water. From the earliest European settlement until the middle of this century, groundwater extraction has primarily been for stock and domestic purposes. From the 1950s until the mid 1990s the emphasis changed to locating and extracting groundwater for a wider range of purposes dominated by the need to fulfil irrigation requirements. Management effort, therefore, was focussed on managing groundwater quantity.

There are over 75,000 licensed bores in NSW, extracting some 1,000,000 ML of fresh water annually. Groundwater is an important resource for town water supplies (130 towns), irrigation, industry and stock and domestic purposes. The State's aquifers are coming under additional pressure now to meet increased development demands, as the capacity of surface water has reached its limit, in most parts of the State.

The history of groundwater development in the State reveals a century of resource development, with a history of ad hoc and reactive policy formulation and implementation. Until the last decade, government had focused on managing groundwater abstractions, with little attention paid to groundwater contamination, or the impacts of groundwater management on dependent ecosystems.

1.1 Legislation

The right to control groundwater use in NSW is vested in the State under the Water Administration Act 1986. This means that the groundwater underlying land is not owned by the landowner. Groundwater is a resource managed by the State, on behalf of the community.

The Minister's functions are largely exercised by the NSW Department of Land and Water Conservation (DLWC). The Department licenses private users and local councils to use groundwater under the Water Act 1912. The Department has the right under the Act to apply management charges to all groundwater extractors.

The protection of groundwater from contamination is primarily governed by the Protection of the Environment Operations Act 1997, and the Environmental Offences and Penalties Act 1989, which make it an offence to pollute waters, including groundwater. The Environment Protection Authority (EPA) administers these two Acts.

2.0 BACKGROUND

Many of the States aquifers have large volumes of water held in storage but only a relatively small amount of annual recharge. A policy decision was taken in 1983, after a decade or more of rapid groundwater development, to allow some limited mining of groundwater storage. One third of the volume of water held in storage was allowed to be progressively removed over a 30 year period. This would leave two thirds of the water held in storage in perpetuity. After the one third of storage was removed, groundwater extractions were to be reduced to the long-term average annual recharge of the system (called safe yield). This approach, although good for short-term development, has resulted in several major aquifers now being over allocated and requiring allocations be reduced to the sustainable yield (SY) of the system.

Groundwater systems vary in their level of development around the State. Most, but not all, of the high yielding, low salinity alluvial aquifers, such as occurs in the Namoi, lower Gwydir, lower Macquarie, lower Lachlan, lower Murrumbidgee and lower Murray valleys, are extensively developed and require a high level of management. At the other end of the spectrum there are still many aquifers that are only lightly developed and require little active management other than applying licensing rules to regulate extraction and use.

Figure 1 below shows that in management terms aquifers can be categorised into 4 types in relation to development and SY:

Type 1 has a very small amount of water allocated and usage is smaller again
Type 2 is under allocated and under used
Type 3 is over allocated and but under used
Type 4 is over allocated and over used

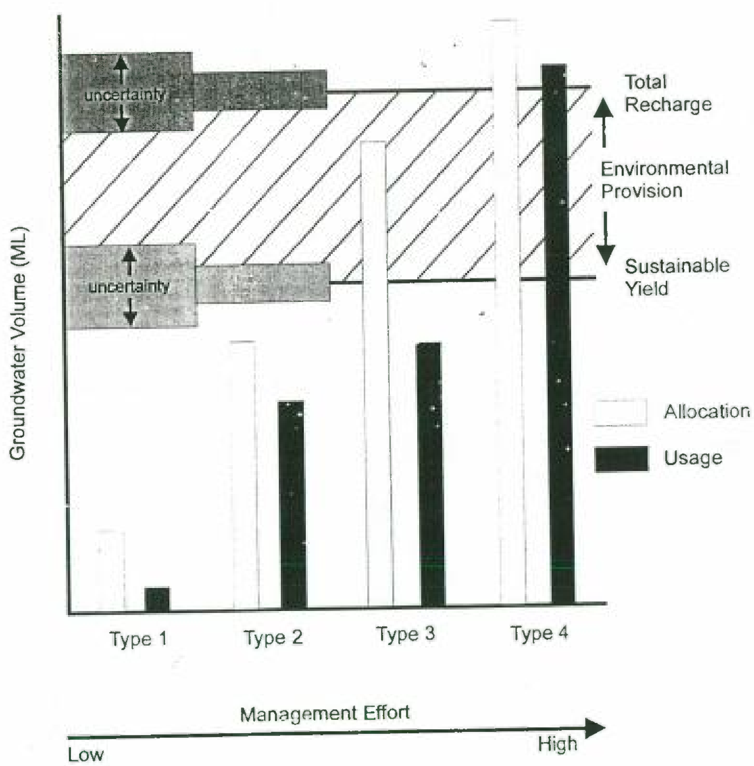


Figure 1 Relationship between Aquifer Management Categories and sustainable yield

3.0 POLICY CONTEXT

In the 1990's important changes have been made to the way water policy was made in Australia.

3.1 National Water Reforms

In 1994 the Council of Australian Governments (COAG) announced a Water Reform Framework Agreement for the management of the nation's water resources. The COAG reforms cover:

- Sustainable use of water resources;
- Provision of water for the environment;
- Pricing of water resources - to achieve full cost recovery by 2001;
- Establishment of water transfer markets; and
- Separation of operator and regulator functions in water resource management.

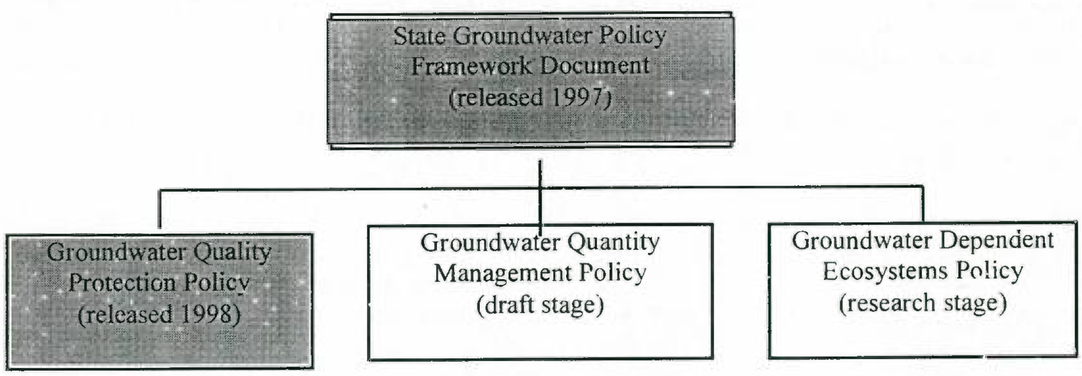
COAG did not deal with groundwater in detail in the 1994 Agreement although there is a general acceptance that it was inferred in the agreement. It asked the Agricultural and Resource Management Council of Australia and New Zealand (ARMCANZ) to further investigate and report on groundwater. Two documents were prepared: *Towards a National Groundwater Management Policy and Practice* (November 1995), and *Allocation and Use of Groundwater - A National Framework for Improved Groundwater Management in Australia* (December 1996). This latter document makes 12 recommendations for groundwater reforms, against which each State/Territory will be audited.

3.2 State Water Reforms

Water reform initiates in NSW, announced in 1995 and in 1997, have focused on delivering the COAG recommendations. Particular attention has been paid to pricing of water resources, environmental provisions, the development of transfer markets, and the establishment of community/government partnerships to guide water resource management decision making

Below is an outline of the groundwater initiatives foreshadowed in the reform announcement.

3.3 Groundwater policy



The NSW State Groundwater Policy Framework Document provides a clear NSW government policy direction on the ecological sustainable management of the State's groundwater resources. It guides the decision-making of State and local governments, as well as landholders in their management and use of groundwater. It sets out management objectives and principles which should be followed to achieve the State's goal for groundwater management:

> ***To manage the State's groundwater resources so that they can sustain environment, social and economic uses for the people of NSW.***

The quantity and dependent ecosystems policy are expected to be finalised by end 1999. The following draft principles for groundwater quantity management are still being negotiated. When finalised the Quantity management Policy will guide key groundwater licensing,entitlement and allocation management decisions.

1	Groundwater quantity should be managed in such a way that the availability of the resource is sustained for future generations, and that ecological processes remain viable.
2	Where necessary, in systems which are currently over allocated, entitlements will be adjusted to match total entitlements to the sustainable yield
3	All works which access groundwater, including bores and wells used for stock and domestic purposes, are required to be licensed.

4	In systems that are not fully allocated,within sustainability limits, groundwater entitlements will be granted once the water needs of the proponent have been fully accessed.
5	Groundwater entitlements, other than for low yielding stock and domestic purposes, will be specified an as annual volume available for extraction.
6	All groundwater extractions, other than those authorised under low yielding stock and domestic licenses, are required to be metered
7	No further conjunctive use licenses will be issued. Existing conjunctive use licenses will be split into separate groundwater and surface water licenses.
8	Sufficient monitoring of groundwater levels and quality will be undertaken to provide the basis for confirming or adjusting entitlement and usage management strategies.
9	Permanent and temporary transfer of groundwater will be permitted within sustainability constraints. No inter-aquifer transfers will be permitted.
10	Groundwater allocations and extractions should be managed in such a way that they do not cause unacceptable local impacts, but allow the resource to be utilised optimally. Unacceptable impacts will be defined for individual groundwater systems on advice from the relevant groundwater management committee.
11	Water quality changes resulting from groundwater pumping should not cause a change to the beneficial use of the groundwater system.
12	Significant groundwater dependent ecosystems should be protected, and the interrelationship between surface and groundwater managed to achieve this

3.4 Aquifer risk assessment

A risk assessment and classification of the State's major aquifers was completed in 1998. Each aquifer was classified as being at "low", "medium" or "high" risk of over extraction and/or contamination, based on a number of assessment criteria.

This process established a basis for setting management priorities, so that limited resources could be directed to systems where they can do the most good. The assessment identified 36 "high" risk aquifers - 32 at risk from over-allocation and 4 from contamination.

Aquifer classification will be re-evaluated every five years to give a State wide indication of the success and appropriateness of management strategies.

3.5 Management committees

Groundwater management committees have been established for the aquifers which are most stressed from over-extraction. These systems are mostly inland and associated with the major rivers. The committees are working in partnership with government to develop a groundwater management plan, which will be the basis for aquifer management over a 5 year period.

Where there is a high degree of connectivity between surface water and groundwater, joint river/groundwater management committees - 'water management committees' - are being established.

For aquifers which do not have committees, an upper limit to development will be determined and licensing, monitoring and other measures put in place to ensure that they do not creep into the high risk category.

4.0 SUSTAINABLE GROUNDWATER MANAGEMENT

In practical terms sustainable groundwater management means managing the groundwater resource at two scales, regional and local. Regional scale management requires that total water allocations do not exceed the SY of the aquifer. Local management operational rules are equally important and are required to limit groundwater pumping impacts to an acceptable level. Groundwater management committees have a large role to play in negotiating an adequate environmental provision and thus the proportion of recharge available for extraction from an aquifer and in also determining what is acceptable pumping impacts between the different user groups (including the environment).

4.1 Environmental provision

A policy decision was made as part of the 1997 Water Reforms to set a default SY at 70% of long term average annual recharge for all groundwater systems in NSW, thus making an environmental provision of 30% of recharge. A proviso was added that the size of the environmental provision could be modified by a groundwater committee. Likewise the government may vary from this default value in aquifers where a committee is not operating, when there are special environment circumstances.
Figure 1 shows the relationship between recharge, environmental provision and sustainable yield.

4.2 Conjunctive Use licenses

The original intent of conjunctive use licences was to provide drought security to water users in regulated river systems. When there is a reduction in surface water allocation, which is normally during dry conditions, conjunctive users make up the shortfall with groundwater. The make-up amount varies from system to system.

With further restrictions on surface water extractions in the Murray-Darling Basin, there has been an increasing use of conjunctive groundwater to make up the shortfall in surface flows - a problem in groundwater systems where usage equals or exceeds sustainable yield. It was untenable to allow a continual creep in access to groundwater, particularly in management areas where groundwater allocations and usage have to be reduced to achieve sustainability.

In 1997 it was decided that no new conjunctive licences would be issued and existing conjunctive use licences would be replaced with separate surface and groundwater licences.

A methodology is proposed for splitting existing conjunctive licences into separate surface and groundwater entitlements. It is intended that this methodology be applied statewide.

It is proposed that users will keep their full surface water allocation, and that the amount of groundwater they receive will be equivalent to their long- term average annual groundwater access under current levels of development and environmental rules.

4.3 Volumetric Conversions

High yielding groundwater licenses issued prior to 1972 were issued in perpetuity and were not restricted in the volume of water that could be extracted. From the period 1972 to 1983 renewable licenses were issued (generally 5 years) and irrigation licenses were restricted in the area that could be irrigated. After 1983 all high yielding licenses issued were renewable and had a maximum annual extraction volume written on the license.

Conversion of all old licenses to a volumetric basis coupled with metering of all water use, other than for stock and domestic purposes, is required so that the groundwater budget can be balanced.

4.4 Reductions in allocation

In groundwater systems that are over allocated, reductions in water entitlements will have to be made. Groundwater management committees will try to find a local solution to this issue through discussion and negotiation with all parties. If the issue is not resolved then government will decide how reductions are to be made.

The splitting of conjunctive use licenses and conversion of area based licenses to an annual volume should occur prior to any reductions so that all types of users are treated the same during the reduction process.

After lengthy discussions with groundwater pumpers there are essentially two methods for making reductions

1. Treat all license holders the same and apply across the board reductions or
2. Give positive weighting to those who have a large history of use.

Any reductions should ideally be phased to allow groundwater pumpers time to adjust to the changes and be accompanied with flexible carryover and borrowing rules in addition to trading rights.

4.5 Access Restrictions

There will be circumstances when additional restrictions over and above the annual amount of water that can be taken will is necessary. Situations may include but not be restricted to; severe droughts, pollution events, emergencies or the protection of existing groundwater users (including the environment). Responses will vary depending on the circumstances but may include restrictions on the amount of water that can be pumped daily if at all.

5.0 TOOLS FOR SUSTAINABLE GROUNDWATER MANAGEMENT

5.1 Licensing

All water supply bores and monitoring bores are required to be licensed under the Water Act. A volumetric allocation is assigned to each bore. All new licenses and water transfer applications are subjected to environmental assessment as defined in the Environmental Planning and Assessment Act 1979.

5.2 Embargoes

Once the sum of all allocation entitlements in a groundwater management area (GWMA) reaches the SY, an embargo on the issue of further entitlements is announced so as to prevent over allocation occurring. The embargo should cover the whole of the GWMA. Figure 2 shows the location of embargoed areas in NSW.

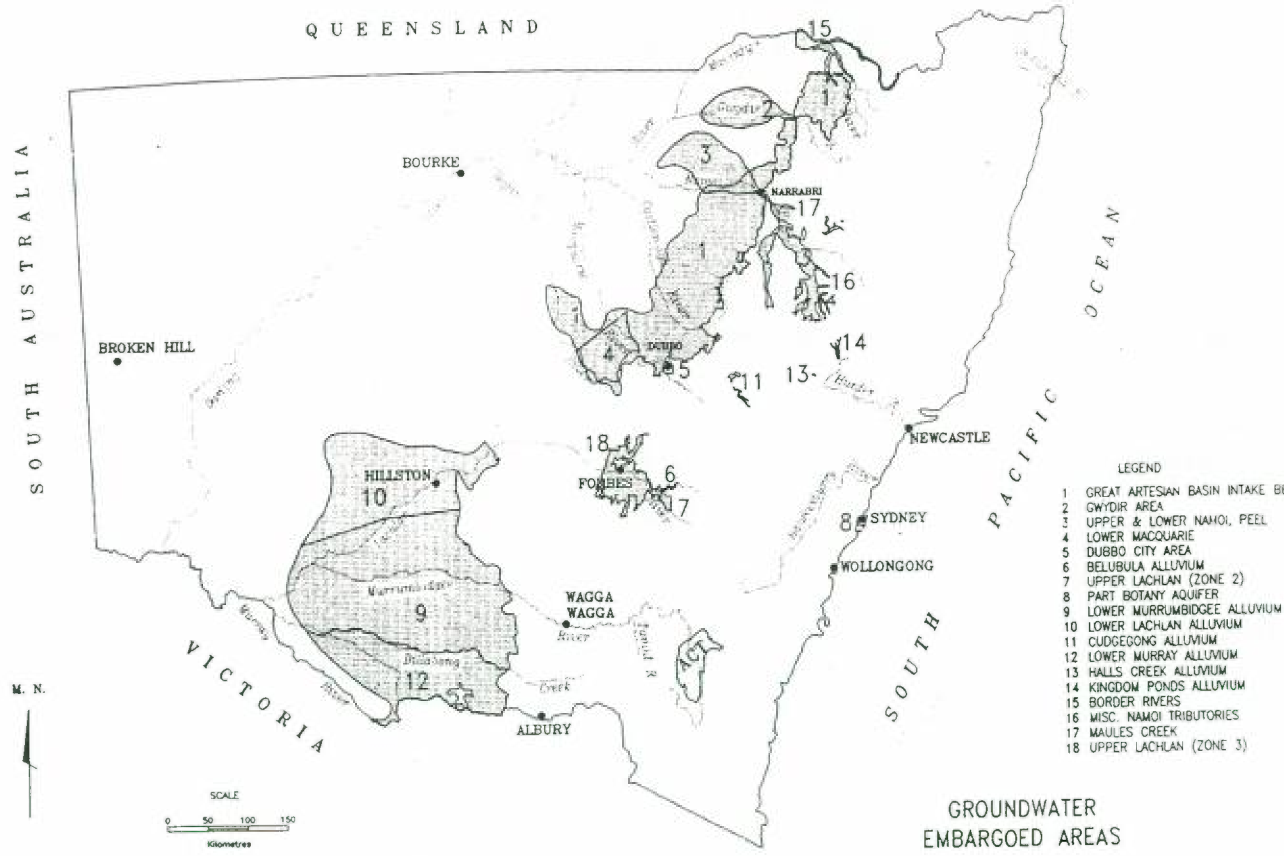


Figure 2 Embargoed areas in NSW

5.3 Trading rules

In groundwater systems, which are embargoed, new or expanding users will need to purchase entitlements from existing users if they wish to access groundwater.

A set of transfer rules have been developed and are currently under consideration. It should be emphasised that the proposed rules are interim. The proposed rules are not intended to provide a mature and efficient market system. It is envisaged that more definitive market rules will be established once a final decision is made on the form and content of any new water access and use rights system. In this regard, the interim rules need to be as simple as possible, so that they can be adapted if and when a new access and use rights system emerges, without establishing any fundamental inequities.

The interim permanent transfer rules will only apply until management committees decide upon and implement a preferred approach to the treatment of unused entitlements and develop more robust trading rules.

5.4 Water charges

Groundwater management charges have been in place since 1995/96. The level of charges is set by the Independent Pricing and Regulatory Tribunal (IPART) to reflect the level of management that occurs in an aquifer. It is the intention to have full cost recovery by 2001. The government exempts stock and domestic users from charges.

5.5 Groundwater vulnerability maps

Regional mapping of groundwater vulnerability to pollution is well advanced with 4 regional maps complete and full state coverage due for completion by 2003. These maps will provide a planning tool to be used by local authorities when making landuse planning decisions that have the potential to contaminate groundwaters.

5.6 Property plans

Property plans are an important tool in accessing water use efficiency and potential environmental impacts associated with new entitlements or water transfers.

6.0 CONCLUSIONS

There appears to be general agreement in the community that managing groundwater for both the present and future generations and providing some water for the environment is the basis for sustainable resource management. There is not however agreement on the fairest way to reduce allocations in those systems that are over allocated. The partnership approach between government and the local community to find solutions through groundwater management committees is a process where all stakeholders can input to the solution.

The technical validity of sustainable yield calculations is a concern to some stakeholders as it is derived in part from a relationship to long term recharge of the aquifer. It is not possible to accurately determine recharge without long term monitoring of groundwater behaviour, research into recharge processes and using predictive groundwater models. It is necessary therefor that all groundwater management plans are adaptive to new information and put in place a process that collects the necessary information to reduce knowledge gaps.

Prioritising Management Directions, Determining Sustainable Yields and Implementing Sustainable Management Practice of Groundwater Resources in New South Wales, Australia

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Abstract

Water resource management in NSW has undergone a rapid change in recent years. The State has moved beyond its development phase and has been set on a course toward sustainable use of all water resources, including groundwater, with explicit provision of water for the environment. The capacity of surface water to meet usage needs has reached its limit and increasing pressure is being placed on the State's aquifers. In the past, policy promoted groundwater development and allowed over-extraction in some systems. For example, in the Namoi, a policy decision was formulated with the water users to exceed the long term average recharge, but for a finite term.

Clearly, groundwater management responses in areas of over-extraction must include bringing use back to sustainable levels. In other areas, groundwater management has to adapt to working within the finite limits set by the goal of sustainability. Applying a comparatively rapid shift in management style to an area as large as New South Wales, where the use of groundwater is widespread, requires a careful and strategic approach. "How to prioritise?" and "how to establish acceptable entitlement and usage limits on aquifers in a relatively short period of time?" are two questions that most characterise the exercise.

The NSW Government has initiated the process of change with two linked projects. The first of these has involved classifying major aquifers (or groups of aquifers) within the State according to risk. This risk was assessed through the consideration of volume of entitlements and usage, water quality and threats, and the perceived presence of significant groundwater dependent ecosystems. This first project has then been used to prioritise management effort in making groundwater extraction sustainable. In the estimation of sustainable yield (the second project), those aquifers classified as "high risk" were given the highest priority and have been determined first.

The sustainable yield of a groundwater system has been calculated as a percentage of the long term average annual recharge. A default value of 70% of this recharge has been assigned as the sustainable yield figure where a more scientifically derived value is not available. This figure can be reassessed to reflect better information on environmental requirements, or changing community priorities, as expressed through Groundwater Management Committees.

In recognition that full scientific understanding of the State's groundwater resources may take years or decades to achieve, and that management needed to change relatively quickly, the approach taken has been to do a rapid analysis based on existing information. This has established an effective starting point and framework for the assessment and adjustment of resource allocation that supports the principle of sustainable management.

This paper outlines these rapid assessment methodologies, their benefits and limitations. It also proposes a pathway and actions to develop more robust estimates of sustainable yield over time, within the context of a risk assessment and risk management approach for the groundwater resources of NSW.

1. INTRODUCTION

The NSW Government followed its August 1997 Water Reforms announcement with a systematic assessment of the State's aquifers. This was a two stage process: stage one being categorising aquifers according to risk (DLWC, 1998); and stage two, the calculation of sustainable yields (DLWC, 1999). The risk classification prioritised groundwater systems according to the need for active management responses. The second stage directed management responses (in this case, sustainable yield determinations) in line with the prioritised list (ie. high risk aquifers first.).

As groundwater's contribution to the maintenance of particular ecosystems has become more apparent, the necessity to consider environmental requirements has become a reality. This substantially complicates groundwater management. Although an ecosystem's dependence upon groundwater might be able to be established, the dynamics of this dependence and its quantification is difficult to ascertain. Additionally, a dramatic increase in extractive use of the resource has occurred in recent history and this further increases the necessary level of management.

The style of management applied in an early "development" phase is not appropriate for when the level of resource usage (or commitment) approaches the limits of sustainability. As limits approach, a much more thorough knowledge of the resource is required for effective management. This comparatively quick shift in both the usage / allocation pattern of groundwater resources and management philosophy has required a corresponding rapid implementation of policy changes. Any delay in moving toward sustainability in the present will create future problems in the management of the State's groundwater resources.

This approach is seen to be the most timely and effective way of attaining sustainability in management. However, the increasing demand for groundwater and the accompanying change in the style of management necessitate close attention to the issue of equity, both within areas and between areas. Development of a consistent statewide approach to assigning priorities and establishing a single set of interim development limits is important in trying to achieve this equity. The periodic review of sustainable yield estimates and the prioritising of core Departmental activities (eg. monitoring and metering) ensure the statewide approach does not overlook reflect local needs.

Completing the risk assessment and initial sustainable yield investigations for high priority aquifers in a short period of time in an area as large as New South Wales presented many challenges. Whilst rapid assessment is adequate to offer immediate protection to the resource in 'high risk' areas, the approach is dynamic and iterative, with ongoing refinement built in to the process. As subsequent, more thorough investigations are carried out, more finely tuned management constraints are applied to each groundwater system. Establishing reasonable estimates for sustainability is imperative. Establishing the precise boundaries of that sustainability can come with time.

2. AQUIFERS AT RISK

As already mentioned, the aquifer risk assessment process classified groundwater systems according to their assessed level of risk from factors such as over-extraction and contamination, as well as the presence of groundwater dependent ecosystems. This classification guides both the management priorities and the application of policies, and recognises that not only environmental circumstances but also economic, social and equity factors vary significantly for each aquifer. It provides water users and the local community with clear information on the issues pertinent to each aquifer. A key outcome of the aquifer classification analysis is the prioritisation of aquifers for immediate management attention. This allows limited government resources to be directed to the areas where they will have the highest return.

2.1 Methodology

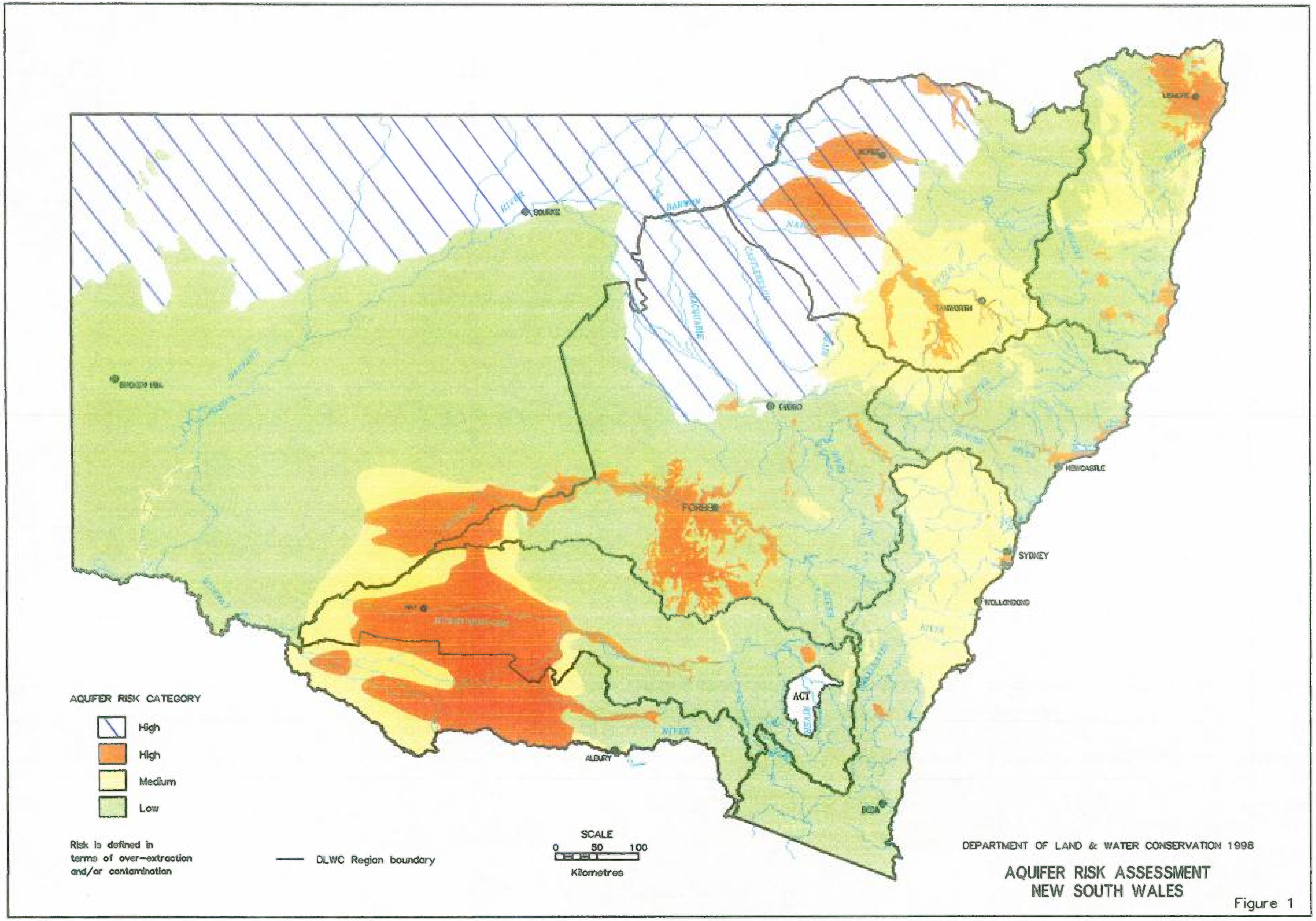
The methodology for the risk assessment was developed by the Department of Land and Water Conservation (DLWC) in close collaboration with the Environment Protection Authority (EPA). Advice was also sought from the NSW Groundwater Policy Working Group which includes farmer, environmental, local government and government agency representation .

2.1.1 Risk Assessment Criteria

A key part of the process involved the establishment of criteria to be used to evaluate the risk level of each aquifer. Eight criteria were used to define the total risk to an aquifer system - see Table 1. The criteria were given weightings to reflect the perceived relative importance of each.

2.1.2 Analysis Techniques

The central tool used in the analysis is multi-criteria analysis computer software which allows the quantitative assessment of qualitative data. This is critical as much of the information available to complete the classification is based on scientific and technical judgements rather than numerical analysis. This approach is considered appropriate for a desk-top analysis such as this. As more quantitative data becomes available, Groundwater Management Committees will be able to re-evaluate the classifications. Figure 1 gives an overview (by area) of the results of the NSW aquifer risk assessment.



AQUIFER RISK ASSESSMENT
NEW SOUTH WALES

Figure 1

Table 1: Risk Assessment Criteria

Criteria	Comments	Weighting (max 10)
1 Relationship between licensed water entitlements and the sustainable yield of the aquifer	This is an indicator of the risk of over-extraction of the groundwater. It is most appropriate for the major alluvial aquifers and confined systems, but not as good for upper catchment alluvial aquifers which are free-draining, nor fractured rock aquifers.	10
2 Local interference caused by pumping	This is an indicator of current stress of an aquifer caused by high levels of extraction.	8
3 Small or large flow systems	This will reflect the ability of the aquifer to cope with stress - large flow systems that occur over many kilometres have greater capability to assimilate stresses.	4
4 Vulnerability of the aquifer to pollution	This is an assessment of the physical characteristics of the aquifer and its susceptibility to land use changes.	4
5 Land use threats	This reflects the actual land use threats in an aquifer's catchment - from urban development, agriculture, ane industry.	10
6 Proximity of poor quality water that could be drawn in by over pumping	This reflects the potential for the aquifer to be polluted from adjacent aquifers and connections between aquifers.	6
7 Water level rise and salinity trends	This assesses the risk of an aquifer to rising watertable levels and salinity increases and is most applicable to surface aquifers.	6
8 Dependence of surface ecosystems on groundwater flows	This reflects both the potential for surface ecosystems to be contaminated by deteriorated groundwater quality and the potential for water losses from over-extraction	10

2.1.3 Multi Criteria Analysis

Multi-criteria analysis is a process to help rank a set number of options according to a defined set of criteria. It provides a structured, yet flexible approach to decision-making, using techniques that range from simple graphical methods to sophisticated mathematical programming. A significant outcome is that by systematically structuring the decision-making process, it makes the results clear and justifies the actions taken.

3. SUSTAINABLE YIELD DETERMINATIONS

One of the most basic pieces of data required for sustainable management of a resource is the quantity of input to a system or "recharge". In the past, the quantity of recharge to an aquifer was accepted as an amount equivalent to the "safe yield" or quantity of water that could be removed from an aquifer on a sustainable basis. We now understand that the "sustainable yield" of an aquifer is almost always a quantity that is considerably less than recharge to allow for adequate provision of water for the environment. Nevertheless, a sustainable yield figure is derived from a recharge determination and any sustainable yield study will usually involve the determination of recharge as a first necessary step.

Recharge calculations with "sustainability factors" applied to them act as interim sustainable yield figures. These "sustainability factors" are some proportion of long term annual average recharge. In line with the precautionary principle, sustainability factors are chosen according to the level of knowledge of an aquifer system, the level of resource use, the magnitude of perceived risk to that aquifer system and the environment, and the reliability of recharge to that system. "Sustainability factors" offer protection to the integrity of the groundwater system itself and ultimately all groundwater users including the environment and ensure that neither temporary nor permanent damage to the aquifer system results from overuse.

Initial sustainability estimates are intended to be conservative as it is easier to adjust values upward than downward. Sustainable yield values can - and indeed will - change over time: firstly, as our technical understanding of the dynamics of individual groundwater systems is enhanced as a result of more rigorous investigation; and secondly, in response to changing community attitudes in natural resource management and economic realities. In short, this is a commencement of a continuous process of periodic "review and adjustment" of sustainable yield estimates. It follows therefore that a set of sustainable yield figures will reflect a level of understanding that exists at a point in time, and that Groundwater management committees may choose to change the sustainable yield factor to suit local conditions.

3.1 Methodology

Where rigorous numerical models have been developed and have resulted in the generation of acceptable recharge figures for high risk aquifers, these values have been adopted as acceptable for use in sustainable yield determinations.

Most other high risk aquifers have not however been extensively modelled. In those systems, estimations of recharge have been developed from both rainfall and river sources. "Throughflow", "underflow" and "irrigation returns" have in most cases been omitted from calculations in the interest of both simplicity and conservatism. More rigorous assessments were not within the scope, resources or timeframe of the exercise.

3.1.1. Recharge and Sustainable Yield

Rainfall recharge was calculated simply according to assessed rainfall, area and proportion of rainfall accessing the aquifer. This represents the total recharge to the system where the only input is thought to be sourced from rainfall.

River recharge was assessed using a modified form of the "Darcy" equation that is used in the assessment of river recharge in the "Modflow" software package that models groundwater flow. The equation used in this calculation follows, where V_{r1} is the *"theoretical"* contribution of the river to overall recharge.:

$$V_{r1} = \frac{K * L * W}{M} * \Delta H$$

- K = Hydraulic Conductivity;
- L = River reach;
- W = River width;
- M = Bed thickness (impedance factor; in practice 1, 2, or 3);
- ΔH = Positive head driving flow.

An additional factor "P" is applied to this result and is an "adjustment" factor intended to reduce the theoretical river recharge and is set as *a) the fraction of the year and/or b). fraction of river reach - that is considered as a "losing stream"*. In this way an actual river recharge component is produced:

$$V_r = V_{r1} * P$$

- P = "the fraction of the river that is a losing stream";
- V_r is the volume taken as contribution of the river to aquifer recharge.

Total recharge to systems that have both rainfall and river components to recharge is given by:

Total recharge = rainfall recharge + river recharge

The approach taken is sometime referred to as the 'expert panel approach', and is appropriate where detailed natural resource knowledge is lacking. It is important to keep in mind that assumptions have been made, and there is an error band associated with the results. Subjectivity and the choice of arbitrary terms of assessment is not new and indeed is very much a part of computer groundwater modelling. However, it must be remembered that the precise results produced by the modelling can have a degree of inaccuracy built in through assumptions and the choice of these arbitrary terms of assessment.

Coastal Sands and Fractured Rock aquifers were assessed on rainfall input only. Most alluvial systems were assessed on both rainfall and river recharge components. The exceptions are those systems that have been modelled and where recharge figures derived from those models are considered acceptable.

All recharge figures have subsequently had "sustainability factors" applied to them in the derivation of corresponding sustainable yield determinations.

It is important to bear in mind the purpose of this exercise was not to produce figures that represent an absolute value for sustainable yield. It was rather, to offer initial reasonable estimates that provide protection to the resource without delay and form a good starting point from which to refine sustainable yield determinations over time.

4. BENEFITS AND LIMITATIONS OF APPROACH

As in any exercise of this nature, there are advantages and disadvantages to the approach applied. The NSW Government has adopted the principle of Ecologically Sustainable Development, and this requires that priorities be set and the determination of Sustainable Yield estimates be made so management of groundwater resources can reflect this change in policy. Additionally, a number of aquifers in the State are critical in terms of over allocation or potential to soon become over allocated. A fairly rapid response method was required so the process of managing to sustainable yield could commence as soon as possible. It is believed that the strengths of adopting this rapid assessment approach outweigh the weaknesses.

4. 1. Method Strengths

- Increases likelihood of statewide consistency;
- Relatively simple to carry out;
- Can be applied in areas of low data coverage;
- Offers an easily established reference point from which ongoing refinement of sustainable yield determinations can occur;
- Documented to facilitate future revision;
- Effective as a trigger mechanism for initiating a higher level of investigation into the understanding of an aquifer when allocation nears the sustainable yield.

4.2. Method Weaknesses

- Subjective and results have an unknown error band;
- In using a relatively simplistic approach there is a danger that non-technical interested parties might accept these results as final and "definitive" and not the "estimates" and evolutionary figures that they are intended to be.
- Results are open to technical and social debate, for which there are no correct answers, owing to gaps in the knowledge base.

5. IMPLEMENTATION AND FOLLOW UP

A number of comments can be made with regard to the sustainable management of groundwater resources and the steps that can be taken to ensure that the aims of sustainability are met:

1. The Sustainable Yield determinations once developed and agreed upon should be adopted as allocation ceilings for the associated aquifers and applied without delay within the licensing system to prevent the occurrence of further over allocation. Moratoria and embargoes should be applied as appropriate;
2. The tracking of entitlements is essential for sustainable groundwater management and should be kept up to date with data readily available through a licensing system data base;
3. All high yielding bores should be metered. Usage statistics should be collected and maintained on an appropriate licensing data base. Allocations should be strictly enforced;

4. Maximising the accuracy and the timeliness of both allocation and usage data should be a priority for effective sustainable groundwater management;
5. The amount of monitoring for groundwater behaviour should be proportional to the stresses (both quantity and quality) on the aquifer;
6. When aquifers move into a higher stress category , appropriate regional triggers should be in place to instigate higher level management responses. Such triggers might be for example, when allocation exceeds 80% of sustainable yield or usage exceeds 50% of sustainable yield. Alternatively, a trigger might simply be a trend of rapidly increasing development interest. These triggers will be the criteria for identifying systems that require more detailed and rigorous analyses. At that stage a refinement of sustainable yield determinations should be completed and will require investigation of greater detail than has previously been carried out;
7. Accepting that all aquifer systems will one day be highly valued for abstraction or environmental maintenance, management bodies should anticipate the information requirements (particularly historical data) of future investigations that will be initiated by stress triggers. Groundwater models, for example require good historical hydrogeological data for effectiveness.

It is envisaged that the sustainable management process will be one of continued monitoring, analysis, prioritisation, assessment and changed management. Such assessment will be applied to existing sustainable yield values which are intended to be regularly reviewed and adjusted as appropriate. The "evolution" of "sustainable yields" is intended to be built into the system.

6. SUMMARY

Responsible resource management requires numerous components that operate together to promote sustainable use. The effectiveness of the overall effort can only be as strong as the weakest link. This exercise has produced a rapid risk assessment for the State's groundwater resources, and what are believed to be an acceptable working set of allocation (sustainable yield) ceilings as a first pass in a continuing assessment of sustainable yield. This alone however cannot achieve the sustainable use of our groundwater resources. Adequate monitoring, good data management, and ongoing investigation in partnership with Sustainable Yield assessments are mutually dependent and supportive of the concept of sustainability.

Sustainable Yield determinations are concerned with 'macro' resource management. It is an exercise that sets the boundaries or limits of a system. It is not a substitute for the 'micro' management that is concerned with the specific and higher resolution features and issues within a system. The sensible zoning of a groundwater system and management within zones is the next step in the interest of sustainable resource management.

The sustainable yield determination is an essential tool for sustainable groundwater management. It is designed to be applied at a large scale and through a coherent licensing system that ensures groundwater resources are not over committed and are also maintained for the continued future benefit of all users. It is important to re-iterate that this is not intended to be final and definitive but a starting point from which ongoing refinement of these determinations will occur. They reflect an understanding of the resource as it exists at present.

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Title: **Transferable Groundwater Entitlements**

Author: **Vanessa O'Keefe**
Organisation: **NSW Department of Land and Water Conservation**

Conference Theme: **Policy Directions**

This decade has seen development of many groundwater systems in NSW reach , or exceed, its capacity. Embargoes on the issue of further entitlements have been declared. This is posing a significant problem for proposed new developments, or existing developments looking to expand, particularly in areas where there is little or no access to surface water supplies. A system of transferable groundwater entitlements would appear to be one mechanism for overcoming this problem. This paper outlines the national and state drivers for the introduction of a system of transferable groundwater entitlements. It explores the principles governing such a system, particularly in relation to the obvious constraints on any market that the physical nature of the resource imposes. Finally, some practical transfer management rules designed to ensure such a system does not adversely impact on other users, the aquifer or dependent ecosystems, are described.

THE FULL TEXT OF THIS PAPER WILL BE AVAILABLE AT THE CONFERENCE

THE LOWER MURRUMBIDGEE GROUNDWATER MANAGEMENT PLAN

T. Riley. Department of Land and Water Conservation. NSW
Murrumbidgee Groundwater Management Committee

Introduction

The Lower Murrumbidgee Groundwater Management Plan is one of several currently being developed in NSW to achieve an integrated approach to groundwater management. They will ensure consideration of interactions between groundwater quality, quantity and dependent ecosystems as well as possible impacts of groundwater use on soils, vegetation and surface water systems.

The goal of the Lower Murrumbidgee Groundwater Management Plan will be to manage the resource in a way which balances the sometimes competing demands of human uses and the environment.

To achieve this goal the plan will rely on the integration of four basic objectives:

- to slow and halt, or reverse any degradation of the groundwater resource;
- to provide for long term sustainability of the resource;
- to provide for equitable sharing of the resource; and
- to sustain economic efficiency in the use of the resource.

These objectives will be achieved through application of the following resource management principles:

- encouragement of optimal beneficial water use;
- management of the groundwater resource in such a way as to maintain the water quality and protect it from potential contamination;
- ensuring development is within environmentally acceptable guidelines;
- integration of the management of groundwater with other natural resources such as soil and vegetation;
- reduction of groundwater pumping to sustainable levels where lack of recharge is causing unacceptable long term water level declines;
- facilitation of equity of access to groundwater supplies;
- consideration of the social, cultural and economic impacts of management options.

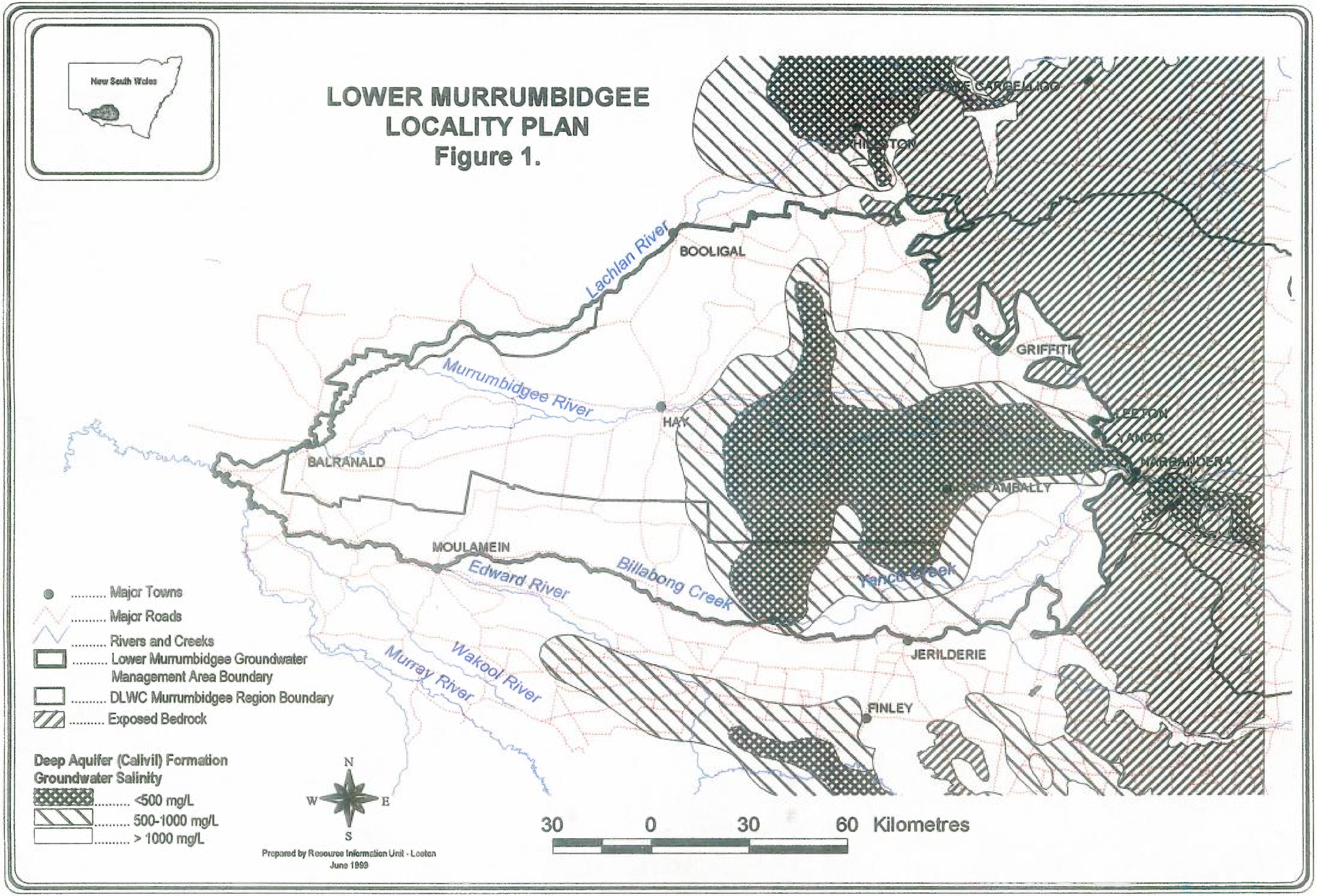
The location of the Lower Murrumbidgee Groundwater Management Area, and occurrence of low salinity groundwater in deep aquifers, is shown in Figure 1. Proposed Groundwater Management Zones, to assist in applying variations in management options such as announced allocations and conditions on transfers, are shown in Figure 2.

Policy and Water Reform Background

In 1994 the Council of Australian Governments developed a National Water Reform Agreement which provides a strategic national framework for water reform. As part of this strategy the States gave a commitment to implement management policies for natural resources based on Ecologically Sustainable Development principles. The NSW Government announced the first stage of its Water Reforms in 1995.

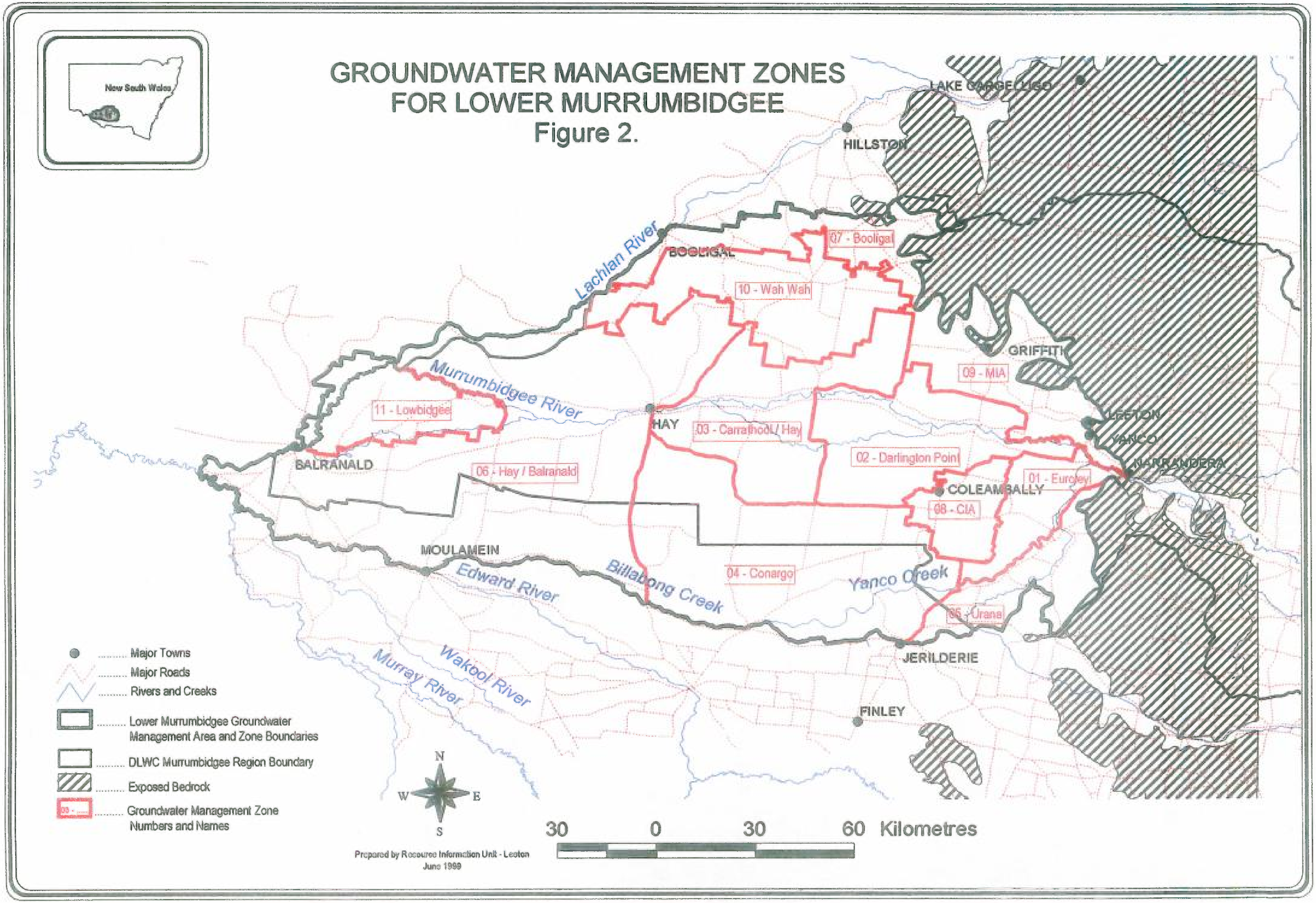
In August 1997, the second stage of Water Reforms was announced, including the establishment of groundwater management committees to advise the Minister and Government on resource management issues. These committees consist of representatives of groundwater users, local government, conservation groups and agency personnel.

The NSW State Groundwater Policy Framework Document was also released as part of the August 1997 Water Reforms. This Policy sets the scene for the management of groundwater across the State of NSW.



LOWER MURRUMBIDGEE
LOCALITY PLAN
Figure 1.

GROUNDWATER MANAGEMENT ZONES FOR LOWER MURRUMBIDGEE
Figure 2.



A Statewide Aquifer Risk Assessment program undertaken in April 1998 identified the Lower Murrumbidgee Groundwater Management Area as a "high risk" groundwater system, with the main risks identified being over-allocation, local drawdown/interference and invasion of aquifers by saline groundwater.

The Lower Murrumbidgee Groundwater Management Area

The Lower Murrumbidgee Groundwater Management Area has a total area of 32 000 km^2 (3 200 000 ha), and is situated within the eastern Murray Basin, mainly between the towns of Narrandera, Booligal, Balranald and Jerilderie. It consists of a layered sequence of unconsolidated clay, silt, sand and gravel deposits that generally extend to depths of 100 to 200 m in the east and deepen westwards to over 400 m near Balranald. Large reserves of low salinity groundwater occur in the eastern areas, where recharge occurs mainly as leakage from the Murrumbidgee River.

The first irrigation bores in the Lower Murrumbidgee were drilled in the late 1960's. Since then, groundwater entitlements and usage have continued to increase, especially throughout the 1990s. Total entitlements now stand at 494 000 ML, while usage in 1997/98 was 241 000 ML. The estimated annual recharge to the Lower Murrumbidgee aquifers is only about 250 000 ML, clearly indicating that the groundwater system is over-allocated. A groundwater moratorium was introduced in 1997 and prevents the issue of any additional groundwater entitlement for irrigation use.

Progress of Plan Development

The Murrumbidgee Groundwater Management Committee was formed in 1998 to develop a Groundwater Management Plan for the Lower Murrumbidgee. The Committee has met regularly since August 1998 in various locations throughout the Lower Murrumbidgee and have discussed the issues and their management options relevant to the area. Local community input has been sought and used in the development of the 20 issues outlined below. Issues 1 -4 relate to dependent ecosystems, 5 - 9 to groundwater quality, and 10 - 20 to groundwater quantity. Further details on these issues and possible management options are provided in a Discussion Paper released in July 1999. Submissions from the community are now being sought to assist the Committee in developing the Groundwater Management Plan.

Groundwater management issues

ISSUE 1: The need to maintain a proportion of the groundwater flow for dependent ecosystems

ISSUE 2: Minimise the effects of water balance changes on ecosystems

ISSUE 3: Provision of groundwater onto the landscape and its possible effect on biodiversity

ISSUE 4: Impacts of applied groundwater on soil structure and watertable depth

ISSUE 5: The need to maintain appropriate water quality for required uses

ISSUE 6: Ensuring effective bore construction and abandonment techniques to avoid water quality impacts

ISSUE 7: Avoiding contamination of aquifers

ISSUE 8: Reduce migration of shallow watertables, both laterally to adjacent areas and downwards into deeper aquifers

ISSUE 9: Ensuring compatibility of applied water with soil to avoid salinisation and soil structure decline

ISSUE 10: Establish the sustainable yield for the Lower Murrumbidgee Groundwater Management Area

ISSUE 11: Groundwater entitlements exceed estimated long term average recharge and thus are not sustainable

ISSUE 12: The need to establish the relationship between Lower Murrumbidgee River flows and groundwater recharge

ISSUE 13: To provide for flexible management of groundwater by utilising Groundwater Management Zones

ISSUE 14: Determining and measuring interference between groundwater users, including the environment

ISSUE 15: Setting guidelines to reduce interference between bores

ISSUE 16: Is compensation appropriate for interference?

ISSUE 17: Mechanisms for groundwater transfers

ISSUE 18: Encouraging efficient use of groundwater

ISSUE 19: Separation of groundwater and surface water entitlements

ISSUE 20: Dealing with anomalies created by the Water Reform Process

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Environmental Provisions in Determining Sustainable Yield for Groundwater Management Plans in the Lower Namoi Valley, NSW

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Abstract

The NSW approach to groundwater management is that groundwater quantity and quality should be managed in such a way that the current beneficial uses of the resource are sustained for future generations, and that ecological processes remain viable. Application of this principle requires that a Sustainable Yield be established for each groundwater system or management zone. This principle states that the long term average annual recharge minus a volume set aside for environmental purposes will determine the Sustainable Yield of the management zone. Effectively, this puts a ceiling on groundwater abstraction by extractive users (ie. the quantity of groundwater that can be abstracted from the aquifer on a sustainable basis.) For groundwater systems that are still in their development phase (that is, where allocations are significantly less than the estimated recharge), total allocations should be capped at 70% of the long term average annual recharge as a first approximation to prevent over allocation occurring. This will allow an investigation and determination to be made of the environmental provisions before setting the level of Sustainable Yield. For groundwater systems that are in a mature water economy (that is where allocations and usage are close to or exceed the recharge) environmental provisions should not be made in isolation from the consideration of economic and social provisions and will need to be phased in over time.

The Lower Namoi Valley is one such "mature water economy" where significant socio-economic considerations must be addressed. There is a need therefore to help reduce the severity of impact on extractive users that a large groundwater allocation reduction will cause. This paper puts forward special case arguments to re-allocate groundwater in the Lower Namoi Valley in a two staged, 10 year allocation reduction process. This will see the allocation reductions phased in over the next two management periods (years 1-5 and 6-10) for those zones that are over allocated. This involves a reduction from the current 202,000 ML/yr Base Allocation to 100% of the *annual average recharge* rate currently estimated at 76,200 ML/yr for these zones. At the same time new allocation will not be issued in the under-allocated zones. This will allow groundwater users to adjust to the re-allocation process, without further exacerbating the socio-economic impacts associated with the significant allocation reductions. During the first management period, an investigation into groundwater discharge processes and the degree of ecosystem dependence on groundwater within the valley will be undertaken and the Sustainable Yield set.

Following this investigation, a decision will be made to phase in further reductions (or increases) in allocation to the Sustainable Yield of the valley during the second management period which represents the second stage of the re-allocation process.

1 Introduction

The demand for irrigated agriculture in the Namoi Valley has expanded substantially over the last 30 years. The valley has entered a period of a 'maturing water economy' with increasing competition for access to fixed groundwater supplies, a growing risk of aquifer contamination, and higher economic, social and environmental costs of development. Until recently, these demands have been managed with little consideration given to the long term consequences of over-exploitation of groundwater resources, or the impacts of groundwater abstraction on dependent ecosystems. The groundwater management strategies of the early 1980's allowed access to groundwater in excess of recharge so that water users had time to recoup their large establishment costs. It was anticipated that the allocation ceiling would not be reached because wet years would recharge the aquifer system and replenish any decline in storage and water level decline would stabilise with time, albeit at a lower level.

It was not anticipated however, that the price of cotton would remain buoyant and therefore lead to increased groundwater development and that average use in some zones would surpass the average recharge on an annual basis. The drought years of 1992/93 - 1994/95 had a significant impact on water levels when abstractions increased to almost twice the recharge rate. Long term rates of decline increased from <1 m/yr prior to the drought to more than 4 m/yr in some areas during the drought. Fortunately, the return to non-drought climatic conditions and changes to groundwater management have returned this decline rate to pre-drought levels or better. The challenge today is to reduce groundwater allocation to a sustainable level whilst maintaining a viable economic and social framework within the limits of acceptable environmental changes.

2 The Policy Framework

It is not the purpose of this paper to discuss the definitions of groundwater sustainability, suffice to say that there is no universally accepted definition. However, since the development of the *National Strategy for Ecologically Sustainable Development* and its subsequent adoption by the Council of Australian Governments (COAG) in 1992, three features that are important in any definition, and pertinent to this paper, are identified:

- the need to consider the wider social, economic and environmental implications of decisions;
- the need to take a long-term rather than short-term view when making decisions; and
- the need for considerable community input in the judgement of what is sustainable.

Furthermore, for the purposes of sustainable groundwater management over the long term, equating the annual abstraction of groundwater with the annual recharge in any management zone, is widely considered to be no longer sustainable (Bredehoeft, 1997; Sophocleus, 1998; Sharp, 1998; Evans et al, 1998). In recognition of this principle, the Lower Namoi Valley is considered to be a special case.

2.1 The NSW Approach to Groundwater Sustainability

Application of the above principles require a Sustainable Yield (SY) to be established for each groundwater system or management zone. Initially, for the purposes of groundwater management in NSW, SY has been defined as follows:

> *"Sustainable Yield is that proportion of the long term average annual recharge which can be extracted each year without causing unacceptable impacts on groundwater users or the environment."*

This means that the average annual recharge (AAR) minus a volume set aside for environmental purposes will determine the SY of a groundwater system. Given the technical difficulty in assessing the recharge and environmental requirements, there may be considerable errors associated with these assessments. SY should not therefore be considered a fixed volume, but rather a target that will evolve over time. It may vary as social, economic and environmental values change and better knowledge is gained about the recharge processes and rates.

The size of the environmental provision will vary, according to the unique characteristics and dynamics of each system, the value of the groundwater-dependent ecosystems (GDE), and the socio-economic reliance of existing extractive users on the groundwater resource. On a State basis, where detailed information on environmental requirements is lacking, total allocations for a groundwater system (that is, the total volume that can be abstracted under all licences), should not exceed 70% of the long term AAR as a first approximation. In some areas it may be necessary to reduce this percentage, particularly where there are significant GDEs present. In other areas, this percentage can be increased depending to a large extent on the presence or significance of GDEs and the socio-economic framework. The intent of the default value is to ensure that as aquifers develop, consideration is given to any GDEs that may be present.

Once this default value is reached in groundwater systems that are still developing, a detailed investigation of GDEs should be carried out and a review of the locally appropriate SY value should be undertaken through a community/Government collaboration process. For groundwater systems that are in a mature water economy, such as the Lower Namoi Valley, where allocations and usage are close to or exceed the AAR, environmental provisions should not be made in isolation from the consideration of economic and social provisions.

3 Quantification of Average Annual Recharge

In order to set the environmental provisions, the AAR of the management zones must be quantified. The first estimates of recharge in the Lower Namoi Valley were carried out by Williams et al (1987) following hydrochemical and isotope studies by Calf (1978) who showed there was substantial pre-1954 groundwater recharge. Williams et al (1986) derived recharge estimates from model calibrations for the periods 1969-1981, 1981-1986, 1981-1994. In 1998, the model was recalibrated to undertake zone water balances for the period 1980 - 1994 (Merrick, 1999). This water balance was modified to account for recharge sources not included in the model, such as higher frequency flooding over the long term, losses from irrigation, on-farm storages and ephemeral creeks. This determined the AAR for each of the 7 management zones and reconfirmed the valley-wide 95,000 ML/yr AAR. More importantly, the 1998 modelling incorporated a cumulative probability distribution for total recharge in the calculations. Basically what this means is that there is an 80% confidence that the modelled AAR will fall between 65,000 and 85,000 ML/yr for the valley. With allowances for the extra recharge sources, there is an 80% confidence that the long term AAR will be between 86,000 ML/yr and 103,000 ML/yr, with a mean of 95,000 ML/yr.

Groundwater pumping occurs mostly in the eastern part of the aquifer between Narrabri and Burren Junction (Zones 1, 3, 4, and 5). There is little pumping in the part of the aquifer west of Burren Junction (Zones 2, 6, and 7). An average water balance for present conditions is shown in Figure 1 for the eastern and western zones of the aquifer. This covers the area east of Cryon and excludes the part of the aquifer between Cryon and Walgett that is not pumped for irrigation. The balance includes inflow from the Upper Namoi (1,500 ML/yr) and outflow to the west of Cryon (8,800 ML/yr), which is thought to contribute to natural groundwater discharge.

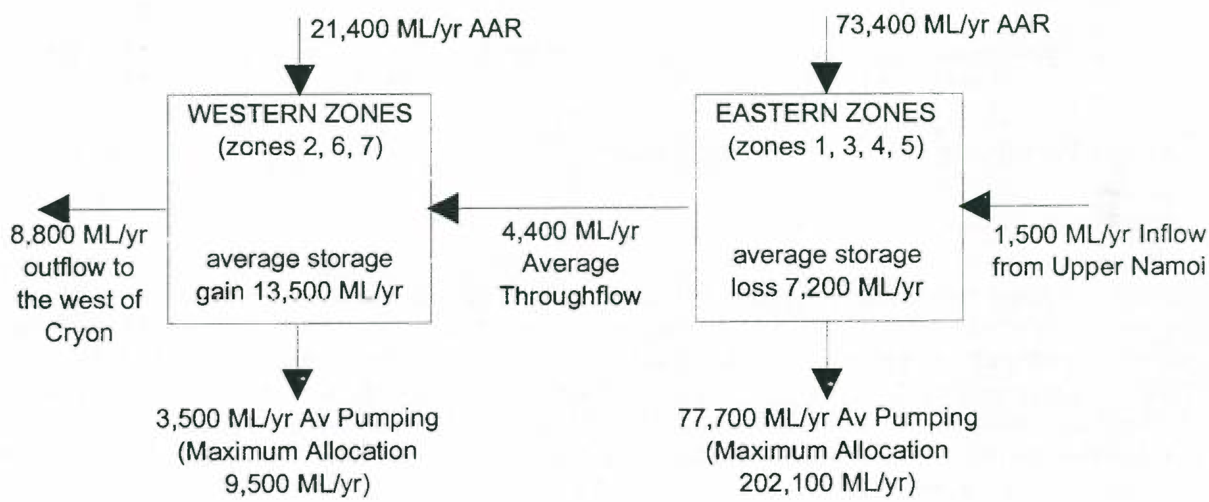

Figure 1: Average groundwater balance for present conditions in the Lower Namoi Valley.

The eastern zones experience a storage loss due to pumping in excess of recharge. This is offset by storage gains in the western zones. These gains are due to enhanced recharge from stream and weir leakage and above average rainfall. The eastern zones contain the majority of the allocation and usage.

It is possible to allow usage in the eastern zones and Zone 2 in the west, to reach 100% AAR and still maintain the outflow to the west. This would involve limiting allocations in Zones 6 and 7 to their present level to maintain the environmental outflow. The likely long term water balance resulting from such a strategy is shown in Figure 2. The average pumping for the whole aquifer is maintained at it's present level. Outflows to the west and groundwater discharge are also maintained while the eastern zones pump 100% of AAR. Total average pumping from the aquifer is decreased slightly to 80,000 ML/yr.

4 Quantification of Sustainable Yield

Quantification of SY is very difficult. As mentioned previously, AAR minus a volume set aside for environmental purposes, will determine the SY of a groundwater system. It is argued in this paper that environmental provisions cannot be assigned in isolation from the consideration of economic and social provisions in a mature water economy. This means that at one extreme, if we allocate a large proportion of the AAR to the environment, we can potentially eliminate the agricultural base of a region that would result in unacceptable social and economic impacts. At the other end of the spectrum, if we do not provide sufficient water to the environment, we can arguably maintain a socio-economic base for some time in the future, but eventually the environment (groundwater) will degrade to a point where the socio-economic framework is also destroyed. There is in reality only one path to take, and that is to achieve a realistic balance between the environmental requirements and the socio-economic expectations and to achieve long term sustainability within an acceptable timeframe.

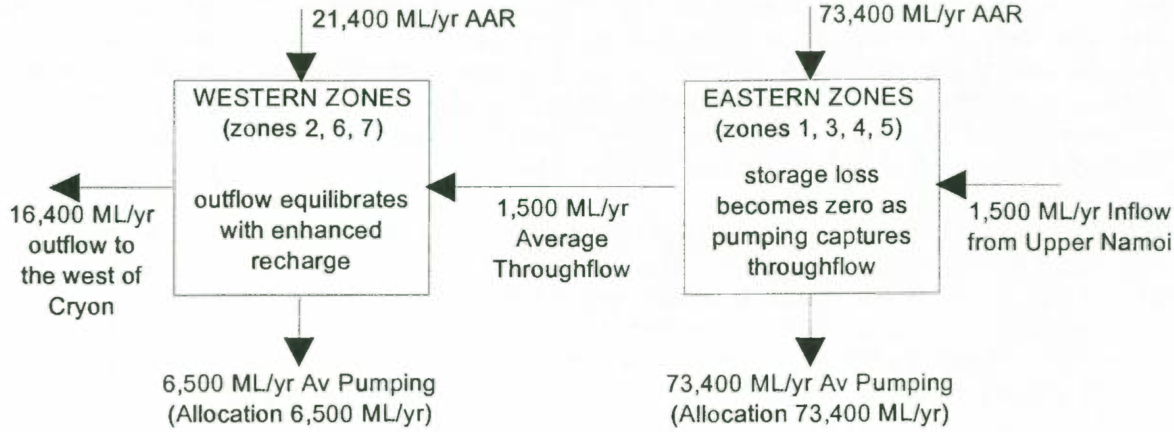

Figure 2: Likely long term average water balance for the Lower Namoi Valley given allocation equals 100% AAR in Zones 1, 2, 3, 4, and 5, and allocation in Zones 6 and 7 is kept at its present level. Groundwater discharge to the west is maintained.

5 Environmental considerations

- *groundwater throughflow*

Groundwater flows from east to west in the Lower Namoi Valley alluvium. This throughflow has been reduced by about 80% due to groundwater pumping for irrigation in the areas east of Cryon. Water balance calculations based on modelling of the aquifer suggest that the outflow from the irrigation area, whilst dramatically lower than it was before pumping commenced, is now relatively stable, and will not fall below the present level, provided long term annual average groundwater pumping is stabilised. If average pumping increases, this throughflow, and ultimately groundwater discharge, will be further reduced. The environmental impacts of such a scenario are presently not fully understood however current allocation reductions and other management initiatives to reduce consumptive use will prevent large development increases.

- *groundwater discharge*

The groundwater discharge processes for this aquifer are not fully understood. Natural discharge is thought not to occur in the area between Narrabri and Cryon. Most discharge is thought to be occurring west of Cryon, an area with little monitoring data. Possible GDEs include stream baseflow, wetlands, and terrestrial vegetation. The volume of groundwater discharge is estimated to be about 10,000 ML/yr. This could be partly contributing to the baseflow of part of the Barwon River and the extreme western part of the Namoi River, as well as terrestrial vegetation in this area.

- *stream losses*

Bore hydrographs show that groundwater does not provide baseflow for most of the Lower Namoi Valley. The surface water system that crosses this part of the flat alluvial plain is mostly a losing stream system. This means it is providing water to groundwater from river recharge. The rate of river recharge decreases in a westerly direction across the valley. Modelling has shown that river leakage to groundwater is not significantly affected by groundwater pumping. Further declines in groundwater levels are unlikely to further increase the rate of river leakage because this rate is thought to have reached its maximum limit. It is expected that average annual stream losses will now remain constant over the long term.

- *Wetlands*

According to Green and Dunkerley (1992), the Namoi Valley does not support any extensive wetland complexes. They define wetland as an area of land that is permanently or temporarily inundated by shallow water. Floodplain and riverine woodlands dominated by species dependent on infrequent inundation have been mapped as wetlands. A study on the physical and biological condition of the Namoi River system (Thoms, 1998) has shown that all the habitats recognised, including the small wetlands, are related to the geomorphic processes occurring within the stream. These studies however concentrated on geomorphological processes and did not investigate groundwater.

- *terrestrial vegetation*

The presence/absence of groundwater-dependent terrestrial native vegetation has not been identified in this valley and a trial is currently being investigated in the Upper Namoi Valley to investigate the potential of falling water tables to impact on tree health. Identification of this relationship may require an adjustment to local pumping scenarios but will not affect the value of AAR or SY on a zone basis.

- *Subsidence*

Large groundwater pumping has caused some minor subsidence (70mm - 210mm) in part of two management zones due to the dewatering of the shallow aquifer and aquitard above the main productive aquifer (Ross and Jeffery, 1991). Management strategies adopted to date have been aimed at restricting further subsidence in these zones. There is no direct evidence of subsidence, or the potential of subsidence in any other zone although this can only be substantiated by accurate ground surveys.

6 Initial reallocation to 100% AAR, final allocation to SY

Current Base Allocation (BA) in the Lower Namoi Valley is about 212,000 ML/yr with an Anncunced Allocation of 65% BA. AAR is 95,000 ML/yr and average use is 81,000 ML/yr. Whilst the valley-wide average use is less than the AAR, that is not the case on a management zone basis, with two zones being 514% and 617% over-allocated and requiring massive re-allocation to sustainable levels. This paper argues that the reallocation process should aim at reducing allocations in the over allocated zones (Zones 1, 2, 3, 4, and 5) to 100% AAR over the next two management planning periods (years 1-5 and 6-10). Allocations in the under allocated zones (Zones 6 and 7) should not be increased. During the first planning period, investigations will be carried out to ascertain and quantify the groundwater discharge processes and the extent of environmental dependence on this discharge. This will define the SY of the system. Further allocation reductions that may be necessary to achieve sustainability should occur over the second planning period (years 6-10). This process will allow the re-allocation process towards long term sustainability to occur without exacerbating, perhaps unnecessarily, the socio-economic impacts by allowing water users to adjust to the reallocation process within a reasonable time frame.

6.1 Socio-economic considerations of 100% AAR

Reallocation to 100% AAR over the first 5 years at least, followed by a review, is a management strategy with low environmental risk on a valley scale. Although maximum use in the early 1990's drought rose to almost 169,000 ML, average annual usage remains about 81,000 ML/yr with the last three years at about 50,000 ML/yr. The current re-allocation processes will limit future increases to such drought levels because of the following management strategies:

- amendments to the conjunctive access;
- reduced announced annual allocations;
- restrictions on transfers;
- embargoes on additional bores; and
- change to the Water Year
- continuing embargoes on issuing new allocation within the under allocated zones.

Average annual usage is expected to remain at similar levels under current climatic conditions which means that at the valley scale, 100% AAR should not be pumped on a continuous basis. This also means that throughflow and groundwater discharge west of the Valley will be maintained at about the present level (Fig 2). The re-allocation process will ensure that under expected drought conditions some time in the future, usage levels such as those experienced in 1994/95 will not prevail. It is expected however, that pumping in excess of recharge may occur during a drought period when water users activate their carry over. This is considered acceptable in systems such as the Lower Namoi Valley with a very large storage and an extensive bore monitoring network.

7 Conclusions

Sustainable groundwater management depends on the understanding of groundwater processes and its interaction with dependent ecosystems, consistent quality and quantity monitoring, matching allocation to the SY and the recognition of indicators of unsustainable practices. Groundwater usage must be sustainable on a long-term basis to provide water users with long term security and to ensure that any groundwater dependent ecosystems are not degraded to unacceptable levels.

Hydrogeological principles, and NSW Groundwater Policy, state that to achieve long term groundwater sustainability, the rate of long term average annual abstraction should be less than the long-term average annual rate of recharge to provide for the environment. This principle should be adopted early in developing groundwater systems to avoid social and economic disruption inherent in allocation reduction programs. In the Lower Namoi Valley, this principle is the main goal for groundwater management. However, adherence to this principle will require account to be taken of the significance of groundwater dependent ecosystems, the socio-economic framework, the planning period and community input to enable fair adjustments to allocations to be made within a reasonable time scale.

The Lower Namoi Valley is the most developed groundwater system in New South Wales and should be considered a special case. Over the next two 5-year planning periods, 100% AAR should be adopted as the reallocation target to allow water users to adjust to the process without exacerbating, perhaps unnecessarily, the severe socio-economic impacts inherent in the reallocation process. Proper identification of GDEs during the first planning period will require the SY to be set as the target for the second planning period.

Perhaps the most important strategy for sustainability, is the transfer of knowledge and understanding of groundwater systems to groundwater users so that consensus-driven, sustainable management plans can be achieved. In reality, the primary management tool for assessing groundwater sustainability is the knowledge gained from understanding and observing the past behaviour of the resource and its response to pumping stresses. The Lower Namoi Valley contains one of the most sophisticated and longest-serving groundwater monitoring networks in Australia from which predictions can be made on future resource behaviour with a high degree of confidence. This allows the DLWC to continuously monitor resource behaviour and adopt management strategies that will take the resource along the path to sustainability.

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GROUNDWATER AND ECOSYSTEMS

Protection of Groundwater-Dependent Ecosystems in NSW, Australia - Policy and Practice

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Abstract

National reform of water management has involved, amongst other things, the requirement to identify environmental needs and to make specific provisions for them in the allocation and management of water resources. Groundwater-dependent environments or ecosystems have received little attention in past management or research. Examples of groundwater-dependent ecosystems in NSW, Australia, include wetlands, streams, red gum forests, limestone cave systems, mound springs, and hanging valleys and swamps. In addition to these, there are also ecosystems within the aquifers themselves, about which very little is known. Groundwater-dependent ecosystems are technically difficult to characterise, vary dramatically from groundwater system to system, and may change within systems with variations in climate conditions. The NSW Government is developing a *"Groundwater Dependent Ecosystems Policy"* which attempts to address some of the issues and processes associated with dependent ecosystem protection in the relative absence of firm scientific information. This paper discusses that policy and some of the tools it will include for protecting and managing them.

Background

Groundwater can sustain important ecosystems. Wetlands, for instance, often have very close connections with a groundwater system. Some ecosystems, such as the Great Artesian Basin's mound springs, depend entirely on groundwater for their survival.

National water policy reform requires the states to identify environmental needs, and to make specific provisions for them in the allocation and management of their water resources.

NSW has been set on a course aimed at ensuring that groundwater is used sustainably and that water is specifically provided for its dependent ecosystems.

What are groundwater-dependent ecosystems?

Groundwater dependent ecosystems (GDEs) are ecosystems which have their species composition and their natural ecological processes determined by groundwater. They are found both on the coast and west of the Great Dividing Range in NSW. In their recent report on the dependence of ecosystems on groundwater in Australia, Evans and Hatton (1998) classified GDEs into four types, namely:

1. *Terrestrial vegetation* - shallow groundwater can support terrestrial vegetation, such as river red gums in the Murray-Darling Basin. The groundwater quality must be sufficiently high to sustain the vegetation.

2. *Wetlands* - groundwater plays a role in many of Australia's wetlands. The mound springs of the Great Artesian Basin, which have been extensively studied, are also included in this group.

3. *Stream base flows*- stream flow is often maintained partly by groundwater providing base flows long after a rainfall event. This base flow may be crucial for in-stream and near-stream ecosystems. Many streams originating in the Great Dividing Range are highly dependent on groundwater discharge.

4. *Aquifer and cave ecosystems* - Life exists in the tiny spaces between the grains within aquifers and in underground caves. The fauna living within aquifers are entirely dependent on groundwater. Micro-organisms are especially important because they consume organic matter, including contaminants, so can exert a direct influence on water quality. There are a number of cave systems in the Murray-Darling Basin. Limestone caves, for instance in the Lachlan Fold Belt, support fauna such as ancient crustaceans which have evolved within a unique, dark environment.

5. *Other ecosystems*: The Evans and Hatton classification system is largely vegetation-based. They also recognised that there is a fifth category of GDEs - fauna directly dependent on groundwater - which is more difficult to identify and characterise.

Scientific and community understanding of GDEs

Groundwater dependent ecosystems (GDEs) have, over time, adapted to the natural variation in groundwater levels and quality, including severe drought and flood conditions. The general level of scientific understanding of the role that groundwater plays in maintaining ecosystems in Australia is very low (Hatton & Evans 1998). Likewise, general community awareness of their values is fairly low. They have received little attention in past management or research. They are technically difficult to characterise, vary dramatically from groundwater system to system, and may change within systems with variations in climate conditions[1].

If we are to better manage GDEs, prioritised research is needed into groundwater and dependent ecosystem relationships.

Groundwater policy

For groundwater-dependent ecosystems in Australia the most important policy is the 'National Principles for Provision of Water for Ecosystems'. The twelve principles in this policy were adopted in 1996 by the Commonwealth Government and all state and territory governments. These principles are aimed at sustaining and, where necessary, restoring ecological processes and biodiversity of water dependent ecosystems'

NSW is now developing a policy specifically for the management of its groundwater-dependent ecosystems. Figure 1 shows how this policy fits into the overall policy framework for managing groundwater:

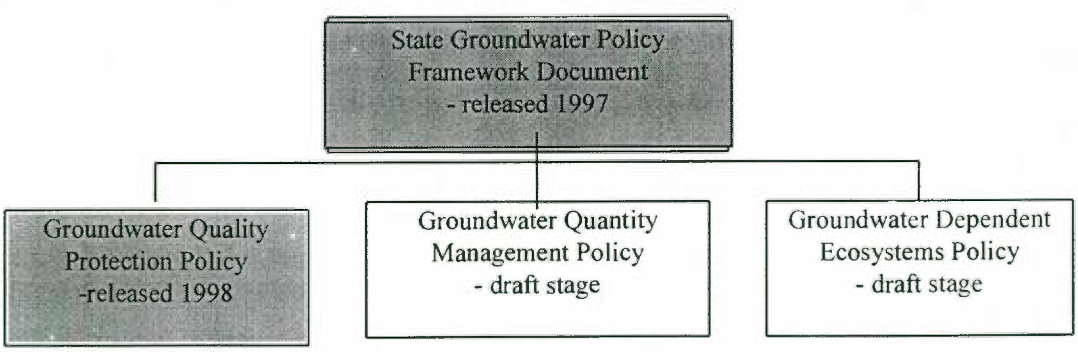

Figure 1: NSW Groundwater Policy Framework.

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1 Aquifers and caves, however, provide relatively stable environments. They are buffered from the effects of climate change. Some of the fauna that exist within aquifers and caves in NSW are quite ancient eg the Syncarida, a group of crustaceans, have existed since before dinosaurs inhabited the continent, at a time when south-eastern Australia was covered in lush, wet, rainforest. They have undergone very little evolutionary change in the last 250 million or more years. Like all fauna found in caves and in the tiny spaces in aquifers, they are entirely dependent on groundwater. Wellington Caves, in the Murray-Darling Basin, is one of the richest in NSW in fauna, in terms of both biodiversity and abundance of species (Serov, 1995; P. Serov, pers. comm. 1999; A. Spate, pers. comm. 1999).

The *'NSW State Groundwater Dependent Ecosystems Policy'* is due for release as a draft within the next month. The policy will discuss the different types of groundwater dependent ecosystems found in NSW. It will address issues and processes associated with dependent ecosystem protection in the absence of firm scientific information.

What's valuable about groundwater-dependent ecosystems?

GDEs have a number of values. For example:

- some are rare or unique eg the Great Artesian Basin's mound springs;
- the age of life in the aquifers themselves - these ecosystems may be among the oldest on earth, so have significant scientific value;
- there are water quality benefits - some fauna in groundwater assists in the 'clean up' of contaminants;
- they have biodiversity value;
- they add to the ecological diversity of a region;
- they are likely to be connected to other non-groundwater dependent ecosystems and thus integrated into the broader regional environment;
- they have social and economic values eg recreation and tourism;.
- they can be bio-indicators, ie indicators of biological health of an overall system;
- they can have cultural significance.

GDEs are under threat, especially on the coast where both the quality and the quantity of groundwater is highly vulnerable to both existing activities and further development.

Groundwater quantity issues

Pumping large amounts of groundwater for human use can cause local water table levels to fluctuate over a larger than normal range and, in some cases, water tables may be permanently reduced. This can create a significant threat to GDEs.

Ecosystems vary dramatically in the degree of their dependency on groundwater, from having no apparent dependence through to being entirely dependent on it. Some ecosystems, such as the Great Artesian Basin's mound springs, depend entirely on groundwater for their survival. Others are less dependent, but changes to the water availability or water quality still impacts on the ecosystem.

The amount of water assigned to the environment will vary according to the characteristics and dynamics of each system, the value of any groundwater-dependent ecosystems and the reliance of extractive users on the groundwater.

The starting point for assigning water to the environment, where detailed information on its requirements is lacking, is that total allocations to consumptive users from a groundwater system - ie the total volume that can be extracted under all licences - should not exceed a percentage of the (long term) average annual recharge. Many groundwater systems in NSW have an assigned sustainable yield of 70% of average annual recharge. For systems which are over-allocated to consumptive users, however, the process for bringing allocations back to sustainable yield levels will need to be negotiated and phased in over time.

Groundwater quality issues

Too much groundwater can also be a problem. Rising saline groundwater levels are causing many ecosystems to suffer. This has been brought about by excessive tree clearing over large parts of NSW, or intensive irrigation, especially in the south of the State.

Groundwater can also be contaminated by the activities that occur on the land above it. Groundwater moves and can carry contaminants to a location where it may affect the ecosystems that depend on it. On the coast, especially, increasing urban development is putting pressure on many of our groundwater-fed wetlands.

It can be impossible - or extremely difficult from a technical point of view and costly - to restore groundwater and any dependent ecosystems once degradation has occurred. Preventing damage to valuable GDEs is the approach taken to their management in NSW.

Tools for managing GDEs

There needs to be recognition by resource managers, including land users, of the importance of GDEs, both at a local and State-wide level.

There are a number of of tools which can be used to manage these ecosystems. These tools can be adapted to local conditions. They include:

- Identifying GDEs and developing objectives and strategies for managing valuable ones at the local level in management plans. For the Gnangara Mound, north of Perth, this has been achieved for terrestrial vegetation, mainly *Banksia* woodland, and some wetlands (Water & Rivers Commission, 1997).

- Establishing minimum distances, ie buffer zones, between bores and groundwater-connected wetlands or streams. These distances would vary according to the proposed pumping regime, the depth of the bore, local hydrogeological characteristics, the degree of groundwater dependency and the significance of the ecosystem. It may be necessary, for example, to exclude all groundwater extractions within, say 200m of a significant wetland or stream. For extractions of greater than 20ML/year this distance might be much greater. This can be implemented through a condition on a groundwater extraction licence. It will require water level and compliance monitoring. For Jandakot borefield, south of Perth, criteria have been developed for distances between bores and significant Banksia vegetation and seasonal wetlands (Kite, Lavery & Pound, 1992).

- Ensuring there are distances between contaminating industries in the groundwater flow path to a GDE.

- Specifying maximum limits to which water levels can be drawn down where there are dependent ecosystems. These limits might vary because of seasonal variation in ecosystem needs. This has been achieved in Western Australia; in 1988 statutory environmental water provisions were set for a number of wetlands around Perth (Kite, Lavery & Pound, 1992). In NSW, under current legislation, implementation can be achieved through a condition on all licences in the area which requires pumping to stop when a particular groundwater level is reached. This will require water level and compliance monitoring.

- Establishing education programs. These would be aimed at raising awareness in the general community of the importance of GDEs.

- Establishing and maintaining a register of groundwater dependent ecosystems.

- Establishing research, monitoring and modelling programs to improve our knowledge of these ecosystems. Priority should be given to the systems:
    - which are highly dependent on groundwater and
    - most at risk from development or other human activities; and
    - where there is a lack of information.

It is proposed that the policy will be finalised in 1999, then reviewed in five years' time when it is expected that our knowledge base will be improved.

Role of water management committees

Groundwater management committees have been established for the aquifers which are most stressed from over-extraction. These systems are mostly inland and associated with the major rivers.

Where there is a high degree of connectivity between surface water and groundwater, joint river/groundwater management committees - called 'water management committees' - are being established.

Where a groundwater management committee is preparing a management plan, suitable environmental provisions should be debated and negotiated.

Conclusion

Groundwater-dependent ecosystems have some important values. Ensuring that groundwater is provided to meet their needs is a complex task. The lack of both scientific knowledge and community awareness of GDEs presents a major challenge.

Only in recent years has the need to make specific provision for environmental requirements been adopted at a national level. NSW is the first state in Australia to develop a specific policy aimed at ensuring that this objective is met, although statutory environmental water provisions have been set in Western Australia.

A range of both regulatory (licence conditions) and non-regulatory strategies are proposed to implement the NSW policy. It is expected that the policy will be revised within five years of its completion as our knowledge base is improved.

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Title – An Identification Methodology for Groundwater Dependent Ecosystems

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Sustainability of all groundwater systems in Australia is a key objective under the COAG Water Reform agenda and management initiatives being implemented across Australia. Nowhere is the pressure on groundwater systems so intense as in some of the low salinity - high yield alluvial aquifers of the Murray Darling Basin. To achieve sustainability for all beneficial uses, it is important that environmental requirements are identified and this component is taken into account for the calculation of sustainable yields.

An understanding of environmental requirements for a Groundwater Management Area (GMA) is of fundamental importance in determining an allocation for the environment. The first component of this equation is to identify groundwater dependent ecosystems. The Nature Conservation Council of NSW has commissioned a study to provide a desktop methodology to focus attention on what ecosystems exist within a GMA, and to determine whether those ecosystems are groundwater dependent. PPK has developed an initial methodology that will assist hydrogeologists, environmentalists, groundwater management committees and other groundwater managers to identify groundwater dependent ecosystems for an area and then to assess the degree of dependency, vulnerability and uniqueness on an individual ecosystem basis.

The methodology is based on a classification system developed by Hatton and Evans in 1997 and is presented at this workshop for information and further development. The approach, if considered useful for groundwater managers, will grow and evolve into an effective tool.. It will be used, not only for the initial identification of systems, but as a tool to evaluate whether we are closing the gap with our knowledge and understanding of ecosystem groundwater dependence as areas are developed, the pressure on ecosystems becomes more intense, and GMAs are more tightly managed.

THE FULL TEXT OF THIS PAPER WILL BE AVAILABLE AT THE CONFERENCE

KARSTIC GROUNDWATER ECOSYSTEMS IN THE MURRAY DARLING AND OTWAY GROUNDWATER BASINS

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ABSTRACT

Small, impounded, karstic aquifers are found widely on the western fall of the Eastern Highlands. Many of the aquifers support highly significant, but little studied, invertebrate faunas. Little is known about ecosystems in the Murray Group limestones underlying the Late Miocene to Quaternary sediments in the lower parts of the Basin. Based on geological considerations, the Otway Groundwater Basin is part of the Murray Darling Groundwater Basin. In the Otway Basin, around Mount Gambier and to the southern coast, there are very many groundwater dependent ecosystems evident.

This paper discusses aquatic karst ecosystems within the impounded karsts of the NSW portion of the Murray Darling Basin and the major karst province of the Otway Basin. Some potential threats to these ecosystems are identified.

INTRODUCTION

Hatton and Evans (1998) have broadly surveyed the groundwater dependent ecosystems of Australia. However, they have significantly underplayed the presence of hyporheic and hypogean ecosystems (Spate and Thurgate 1998). This paper only notes the presence of hyporheic systems in passing and concentrates on those subterranean ecosystems associated with the solution of limestone – hypogean karst systems. Hyporheic ecosystems are those found within the interstial spaces between cobbles, gravels, sands and silts in the alluvium beneath and beside rivers.

The Murray Darling and Otway Basins contain large limestone deposits. In the Murray Darling the limestones are found as Palaeozoic rocks outcropping about 300 above sea level and as the high porosity, Tertiary, Murray Group limestones which underlie the alluvial sediments in the south of the Murray Darling Basin. The Murray Group limestones are inaccessible but drilling shows the presence of cavities , which will contain life forms. There is considerable surface and near-surface expression of these rocks in the western portion of the Otway Basin and well-developed karst systems are present (Grimes et al.1995). Aquatic flora and fauna dependent on karst waters are present in both surface and sub-surface environments, in the Mount Gambier region (Thurgate 1996, in press).

Why should we even mention the Otway Basin in a Murray Darling Basin Workshop? Smith (1988, p25) points out that the basins "are contiguous but the direction of groundwater flow, the geology and water quality are such that they are best considered as separate systems". The ecosystems present in the karsts of each basin are radically different in setting and biota present – yet there are zoogeographic and taxonomic affinities stretching back into Gondwanan times (Eberhard and Spate 1995, Serov and Eberhard 1995, Thurgate in press).

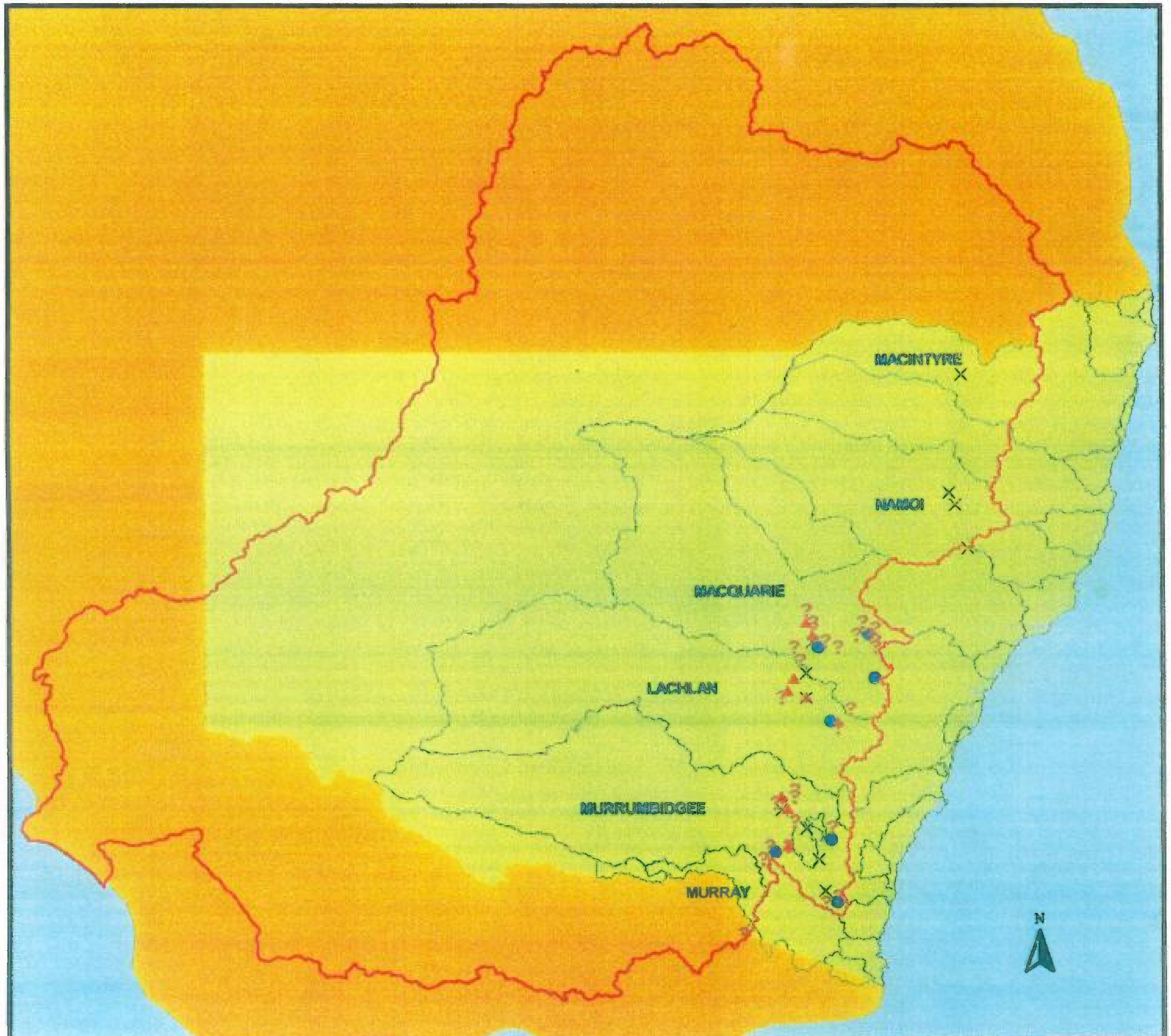
The Palaeozoic limestone outcrops are mostly quite small; about 60 separate areas with caves can be identified within the Basin (Eberhard and Spate 1995). Many of these caves reach down to the watertable and a surprisingly rich aquatic fauna has been identified in these areas. Again the fauna has Gondwanan origins and significance (Serov and Eberhard 1995, Wilson and Johnson in press).

The work of Andrew Boulton and colleagues in hyporheic environments in Australia (see, for example, Boulton and Suter 1986, Boulton 1999 and Boulton et al. 1998) is of particular importance. Such ecosystems will be found throughout the Murray Darling Basin and may be particularly important in maintaining the health of intermittent streams and of wetlands. Unfortunately hyporheic systems have been little studied. Some hyporheic systems may have close interactions with karstic groundwaters.

MURRAY DARLING BASIN AQUATIC KARST ECOSYSTEMS

The accompanying map and tables show the distribution and some statistics of the Palaeozoic limestone cavernous karst areas of the Basin. The survey referred to is that of Eberhard and Spate (1995). They examined at reconnaissance level the cave systems across the whole of NSW, revealing a previously unsuspected and diverse macroinvertebrate fauna with many new species and higher order taxa. Some areas were sampled intensively. About two-thirds of the State's cavernous karsts are found within the Basin and nearly half have caves, which carry streams, reach down to the watertable, or have perched pools – all of which may support stygobiont faunas. Those unsampled during the survey where thought to be of lesser interest from a biological viewpoint. However, some apparently "insignificant" sites have been demonstrated to have outstanding values – the precautionary principle needs to be applied.

It should also be noted that karstified limestones may not exhibit caves enterable by man but will possess solutionally enlarged spaces which can only be sampled by drilling or through spring discharge. Thus many of the areas "written-off" above may contain groundwater dependent taxa.



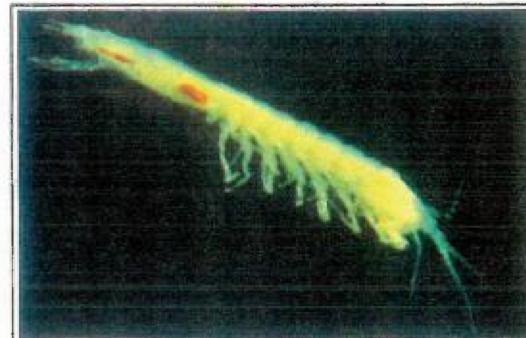
Cavernous karst areas in the NSW Murray Darling Basin

Karst areas

- Aquatic species present
- Stygobionts present
- Surveyed, none found
- Not surveyed

River Basins

Murray Darling Basin



Family: Psammaspididae,
Gen. nov. sp. nov.

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July 1999

CAVERNOUS KARST AREAS WITHIN THE NSW MURRAY DARLING BASIN

Area	River Basin	G/W access?[1]	Aquatic fauna[2]	Stygobionts present[3]	Alt.[4]	Area	River Basin	G/W access?[1]	Aquatic fauna[2]	Stygobionts present[3]	Alt.[4]
Abercrombie	Lachlan	yes	yes	yes	500	Limekilns	Macq	ephemeral	yes	probable	760
Apple Tree Flat	Macq	yes	yes	probable	600	Lobbs Hole	M'gee	?	n/s		650
Ashford	Macint	no	n/f		420	London Bridge	M'gee	n/a	yes		685
Bakers Swamp	Macq	yes	yes	yes	320	Lue	Macq	?	n/s		560
Boduldura	Macq	?	n/s		320	Macphersons Swamp Creek	M'gee	no	n/f		450
Borenore	Lachlan	yes	n/f	probable	640	Michelago	M'gee	no	n/f	possible	660
Bowan Park	Lachlan	yes	yes	yes	460	Molong	Macq	probable	n/s	probable	540
Burran Burran	Macq	?	n/s		300	Moore Creek	Namoi	no	n/f		350
Canomodine	Lachlan	yes	yes	yes	530	Mudgee	Macq	yes	n/s	probable	460
Canowindra	Lachlan	?	n/s		460	Numeralla	M'gee	possible	n/s		750
Capertee Valley	Macq	?	n/s		400	O'Hares Creek	M'gee	?	n/s		650
Cave Flat	M'gee	drowned	?		400	Paddys River	M'gee	no	n/f		500
Cliefden	Lachlan	yes	yes	yes	390	Portland	Macq	?	n/s		800
Cooinbil	M'gee	possible	n/s		1150	Queens Pinch	Macq	?	n/s		480
Cooleman Plain	M'gee	yes	yes	yes	950	Rock Flat Creek	M'gee	possible	n/s	possible	640
Copperhania	Lachlan	yes	yes	probable	680	Rockley	Macq	yes	n/s		910
Crawney Pass	Macq	?	n/f		320	Rosebrook	M'gee	no	n/f		760
Cudegong	Macq	?	n/s		640	Stuart Town	Macq	no	n/f		300
Cumnock	Macq	?	n/s		460	Sulcor	Namoi	yes	n/f	possible	500
Dripstone	Macq	?	n/s		360	Taemas	M'gee	yes	n/s	possible	400
Finches Caves	Macq	yes	yes	probable	560	Talmo	M'gee	yes	yes	yes	530
Geurie	Macq	?	n/s		320	Tuena	Lachlan	?	n/s		550
Goodradigbee	M'gee	possible	n/s		400	Walli	Lachlan	yes	n/f	possible	380
Ilford	Macq	?	n/s		780	Warroo	M'gee	yes	n/s	possible	500
Indi	Murray	yes	n/s	probable	850	Wee Jasper	M'gee	yes	yes	yes	400
Jeremiah Creek	M'gee	yes	n/s	possible	550	Wellington	Macq	yes	yes	yes	300
Jounama Creek	M'gee	yes	n/s	probable	800	White Rocks	M'gee	?	n/s		550
Kandos	Macq	yes	n/s		400	Windeyer	Macq	?	n/s		640
Kybean	M'gee	yes	yes	probable	1000	Yarrangobilly	M'gee	yes	yes	probable	960

KARSTS AND THE PRESENCE OF AQUATIC FAUNA BY RIVER BASIN

River Basin	No. of cavernous karst areas	No. with direct cave access to groundwater[1]	No. containing aquatic species	No. containing stygobionts[3]	Mean altitude of areas (m)[4]
Macquarie	23	7	5	2	500
Murrumbidgee	22	9	6	3	650
Lachlan	9	7	5	4	510
Namoi	2	1	no taxa seen	none	425
Macintyre	1	none	none	none	420
Murray	1	1	1 probable	1 possible	850
Totals	58	25	16+	9+	---

[1] Is groundwater accessible through caves? (Note: karstic groundwaters and dependent ecosystems may exist in cavities too small to be termed caves and may only be accessible through bores or *via* springs or through diffuse discharge into other aquifer systems – e.g. alluvial gravels).

[2] n/s = not surveyed; n/f = none found during survey (Eberhard and Spate 1995).

[3] Stygiobionts include troglobitic and troglophilic aquatic, subterranean fauna dependent on subsurface conditions.

[4] Very approximate altitude of groundwater.

The survey revealed faunal assemblages dominated by crustaceans including amphipods, copepods, ostracods, asellote and phreatocid isopods and syncarids. Non-crustaceans such as snails, bivalves and various worm groups were also collected (Eberhard and Spate 1995, Serov and Eberhard 1995, Bradbury and Williams 1997). Most appear to be cave-adapted – blind and unpigmented – and are grouped under the general term "stygobiont". "These taxa represent groups which no longer exist in the surrounding epigean (surface) environment and are true distributional relics. Most indications are that each karst has its own complement of distinct species" (Serov and Eberhard 1995, p3). Most of the new taxa remain undescribed.

AQUATIC KARST ECOSYSTEMS OF THE OTWAY BASIN

The groundwater dependence of aquatic communities in the western Otway Basin has been recognised for some time. Eardley (1943) demonstrated the dependence of the fens along Eight Mile Creek – a short, totally spring-fed stream rising from the Murray Group Limestones. Many other fen communities have now been described in this general area including such well-known sites as Piccanninie Ponds.

Recent investigations have revealed some highly interesting communities (Thurgate in press) including the presence of 21 distinct stromatolite types from cenotes (sheer-walled cave collapses exposing the regional watertable) and from karst waters in Blue Lake. Stromatolites are lithified, laminated, organo-sedimentary deposits, which are formed by complex ecological associations of algae, bacteria and other micro-organisms preferentially causing precipitation on the walls of the cenotes (Thurgate 1996).

As in NSW, the diversity of the obligate cave-dwelling macroinvertebrate fauna is now being revealed in the karstic groundwaters (Thurgate in press). Again, crustacea and molluscs are the most important elements with many cave–adapted forms (stygobionts) being present. Again they have phylogenetic and relictual distribution significance with an ancient history.

SIGNIFICANCE OF THE ECOSYSTEMS

The significance of the groundwater dependent biota in each of the Basins is quite different but with some remarkable affinities. Only two aspects will be discussed.

Bradbury and Williams (1997), Serov and Eberhard (1995) and Wilson and Johnson (in press) amongst others demonstrate how the area that is now the Murray Darling and Otway Basins (and some other areas of Australia) was covered by sea during the Middle Cretaceous (about 119-114 million years ago). The modern distribution of various stygobiont groups such as syncarids, amphipods and phreatoicidean isopods corresponds closely to the Middle Cretaceous sea. This suggests that the retreat of the sea forced many species into freshwater refugia. Stable conditions in groundwaters has allowed many groups to survive and to continue to evolve so that there has been differentiation between these "living fossils" (many of which were first described from the fossil record).

Thus, amongst the ancient syncarid group, for example, we now have the family Koonungidae restricted to southern Victoria, southeast South Australia and to Tasmania. The family Psammaspididae is restricted to NSW – with most records within the Murray Darling Basin. All except one representative of the family Psammaspididae come from karstic groundwaters – the other is a hyporheic species. Wellington Caves, on the Bell River, is a particularly important site owing to the diversity and abundance of the taxa in those groundwaters.

Grey et al. (1990) suggest that an area has high stromatolite diversity if four to six forms are present. Thurgate (1996, in press) points out that there are some 21 extant forms within the Otway Basin limestones. In an Australian context, the stromatolites from this area are the largest, deepest and most diverse forms in fresh groundwater. On a global scale the diversity is unequalled especially as seven branching forms are present – branching types are very rare in living stromatolites. Stromatolites have an even longer lineage than, say, the syncarids (recognisable during the Late Devonian – 400-380 million years ago) as they have their origins as far back as 3.5 billion years in the Archaean Era.

These two examples point to the significance of karstic groundwater systems. The significance lies in their extreme longevity with the consequent interest to science for the evidence they demonstrate of evolutionary process and of palaeoenvironmental conditions. The longevity is dependent on stable or slowly changing conditions, which allow for organisms to adapt to change. Rapid change is inimical to survival. Clearly there is also a high educational value and there may well be, perhaps unfortunately, a role for stygobionts in the monitoring of groundwater health.

THREATS TO THE AQUATIC KARST ECOSYSTEMS

Threats to aquatic karst ecosystems are similar in both groundwater basins. Threatening processes include (after Eberhard and Spate 1995):

- ◊ deforestation and afforestation with non-native species
- ◊ agriculture (including clearing, tilling and fertilising and biocide application)
- ◊ groundwater abstraction for irrigation and domestic water
- ◊ changes to water quantity regimes
- ◊ sedimentation
- ◊ input of toxins and nutrient enrichment
- ◊ rubbish dumping
- ◊ factory and intensive agricultural effluents (dairy, feedlot wastes)
- ◊ activities of scientists including over-collection
- ◊ disturbance by cave visitors.

Obviously these threats are not mutually exclusive. Many can arise from agricultural activities (Gillieson and Thurgate in press). All of these processes have operated on the aquatic groundwater communities of both the Murray Darling and Otway Basins. In the latter Basin the problem is well recognised and many important improvements in land management practices have been introduced. However, watertable levels continue to drop exposing some forms of stromatolites. In the Murray Darling Basin many of the Palaeozoic karst systems are still exposed to impacts from a variety of processes.

Of particular concern, whilst not a threatening process in itself, is the lack of knowledge of the fauna present and of ecosystem structure and function. The steadily decreasing taxonomic support available to the Australian community means that, even if taxa are collected, their affinities and significance may never be assessed.

A further issue in the Palaeozoic karsts is the relationship between the karstic groundwaters and those of surrounding aquifers. This is especially so in the case of the interactions between karst and alluvial systems where abstraction of the alluvial groundwater may exert an influence on the adjoining karst system. Superficially this would seem to be simple but the Bell River near Wellington demonstrates that considerable complexity can occur. This area has a highly significant karstic groundwater fauna and adjacent alluvial aquifers utilised for irrigation. But the apparently simple relationship between the two aquifers is quite unclear and probably complex (Houshold et al. 1990) and thus the lack of understanding prevents appropriate management consideration.

CONCLUSIONS

There is an interesting and diverse fauna inhabiting the karst aquifers of the Murray and Otway Groundwater Basins including many "living fossils" and taxonomic groups such as stromatolites and syncarids, which reach back toward the beginning of life on Earth. The fauna and their ecological relationships deserve further study. This includes support for taxonomy so that the relationships between various taxa can be better understood.

We know virtually nothing about the ecological requirements of the various known taxa – which are not formally described in most instances. This is a very difficult field of study but one that must be addressed. Without this knowledge the management of these faunas depends on "first principle" approaches – with the dangers consequent on uncertain knowledge readily apparent.

There is an urgent need for groundwater users and managers to recognise the existence of these faunas so that threatening processes can be avoided and impacts ameliorated. The "precautionary principle" must apply.

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CAN WE PREDICT TRENDS IN FLOODPLAIN TREE HEALTH IN THE LOWER RIVER MURRAY ?

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Andrew Telfer	Australian Water Environments

This paper reviews the physical processes leading to salt accumulation on the floodplain of the lower River Murray, with the aim of predicting trends and patterns in floodplain vegetation health. Salt accumulation has been shown to be the major cause of floodplain tree dieback caused by water stress. The understanding and spatial distribution of key physical processes can be used for broadscale vegetation health risk mapping. Key salt accumulation processes include regional groundwater discharge areas; natural or irrigation groundwater mounds; river regulation weirs; floodplain topography, soil types, groundwater depth and salinity. The tools are now available for assessing vegetation health, both instantaneously and incorporating the historical flooding and salinity regime.

1. INTRODUCTION

Currently, there are a number of initiatives to better assess the full range of impacts of salinity. These include the National Land and Water Resources Audit, the Murray Darling Basin Commission Salinity Audit and WA Salinity Action Plan. Environmental impacts of salinity are probably some of the most difficult to assess. It is only relatively recently that such impacts have been considered important and the tools developed to evaluate these and potential management options.

In the Murray Darling Basin, one of the largest environmental impacts is the decline of riparian vegetation in the lower River Murray (South Western NSW, North-West Victoria and SA). Salt accumulation has been identified as the major cause of floodplain tree dieback from water stress in the lower River Murray (Margules and Partners *et al.*, 1990). Recent estimates of vegetation affected by soil salinisation have suggested that up to 26,000 hectares (of a total 100,000 hectares) of floodplain along the lower River Murray in SA have been affected by salt (Nichols, 1998).

The intrinsic values of the riparian vegetation are what make the region unique. The wetland and floodplain vegetation of the lower River Murray is an important aspect for water-based activities, camping, bird watching and walking. The floodplains of the lower River Murray contain a high biodiversity of both flora and fauna; providing an important native fish nursery and breeding area for water birds and aquatic invertebrates (O'Malley and Sheldon, 1990; Cunningham *et al.*, 1981). The Chowilla floodplain has an UNESCO Ramsar Convention listing as a Wetland of International Importance and as a habitat for migratory birds (National Environmental Consultancy, 1988).

Sinclair Knight Merz (1999) evaluated the floodplain environment as part of the cost / benefit analysis of the 7,000 hectare of floodplain influenced by the proposed Chowilla Groundwater Control Scheme. The environmental values considered as part of the benefits of the scheme included:

- use-values; benefits to users of the environment, e.g. for fishing, hunting, sightseeing, recreation, house boats, wildlife viewing, etc of $187 per hectare
- non-use-values; conservation values based on the satisfaction people derive from knowing some aspect of the environment is to be preserved, even though they may never visit the area of $1390 per hectare.

Crudely, this gives a value of between $5 M and $35 M to the 26,000 hectares of salt affected land.

In order to better protect these floodplain environmental values in the lower River Murray, a good understanding of the processes causing the decline in vegetation health is required. This paper reviews a number of recent studies that deal with trends in floodplain vegetation health. Based on our present understanding, we discuss the likely success of various management options. The focus of this paper will be on issues directly related to floodplain tree health, particularly blackbox (*Eucalyptus largiflorens*) on the floodplains of the lower River Murray in South Australia.

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2. PROCESSES AND CHARACTERISTICS LEADING TO FLOODPLAIN SALINISATION

The floodplain environment has been affected by changes to river regulation in the lower River Murray, including a reduction in flooding frequency and duration; and changes to groundwater depth upstream of weirs. Intensive irrigated horticulture on the upland areas has raised groundwater levels, increasing flows of native saline groundwater towards the floodplain and river. Salt from the regional groundwater system naturally discharges into the river. Increased saline groundwater flows affect salt loads to the river (Allison *et al.*, 1990); floodplain salt accumulation and vegetation health (Bone and Davies, 1992, Jolly and Walker, 1995a); waterlogging; and saline seepage on the edges of the floodplain (Australian Water Environments, 1999; PPK Environment and Infrastructure, 1997, 1998).

The weirs and locks in the lower River Murray were constructed between 1920 and 1940, raising the river level by approximately 3 m upstream of each weir (Close, 1990) and form a series of stepped pools. These artificial weir pools, can extend for up to 70 km upstream of a lock (e.g. Chowilla, Jarwal *et al.*, 1996), permanently raising water levels, as shown in Figure 1. This drowns trees on the edges of the river; increases groundwater pressures beneath the floodplain, and raises the watertable by 2-3 m to 1-5 m below ground level (Slavich *et al.*, 1999a).

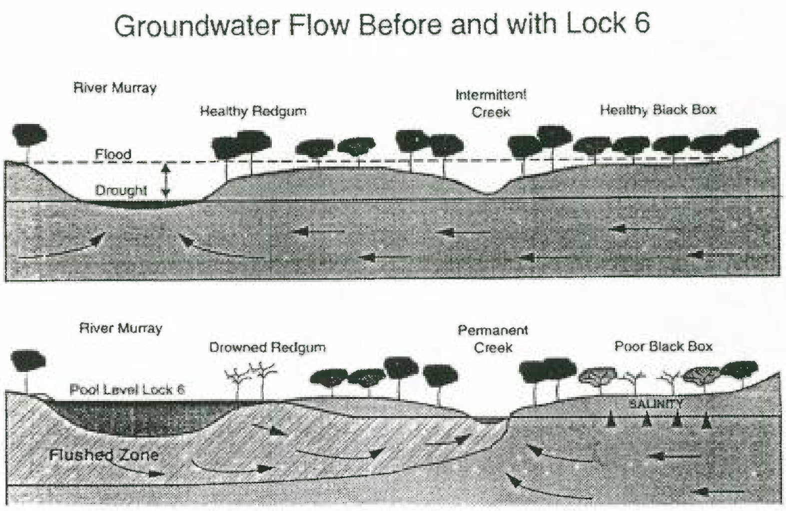

Figure 1. Transect of typical floodplain vegetation, pre- and post-construction of Lock 6. Red gums line the creeks and blackbox grows on the higher elevation areas. Lignum (not shown) tends to inhabit the low-lying, frequently flooded areas. Prior to the construction of the weir at Lock 6, groundwater flowed towards the river in times of low flow, however the creation of the permanent pool created a zone of low salinity, the 'flushed zone'. The regional groundwater now discharges to the anabranch creeks and into the soil (Adapted from Chowilla Resource Management Plan, 1995).

Major floods are very important for the long-term survival of blackbox overlying saline groundwater. Slavich *et al.* (1999b) demonstrated this in their modeling of the period 1970-1994, the major floods of 1974-1976 improved floodplain vegetation health for up to 12 years. The present reduction in flows is affecting vegetation health, mean flows have been reduced by 37% and median flows by 80% (Close, 1990). The frequency of medium sized floods has been reduced by a factor of 3 (Ohlmeyer, 1991); minor to medium floods (up to 1 in 7 year return period) have been eliminated (Caldwell Connell Engineers, 1981).

Irrigation mounds form due to increased recharge beneath irrigation areas. The mounds are several metres above river level, e.g. 18 m above river level at Loxton (PPK Environment and Infrastructure, 1997). The presence and extent of several of these irrigation mounds have been modelled, showing increased water table levels and flows to the river following irrigation development (e.g. Bookpurnong, Woodward Clyde, 1998a; Ral Ral Creek, Woodward Clyde, 1998b). The cross section in Figure 2 shows the relationship between the groundwater levels beneath the floodplain and in the irrigated uplands.

Soil type affects bank and floodplain recharge rates during floods, influencing the location of "green oases", where sandy soils increase localised recharge. Irrigation disposal basins on the floodplain raise local water levels and store large quantities of salts in the floodplain, affecting local vegetation. Stock grazing and trampling affect vegetation health, particularly natural regenerants survival. Mallee recharge is projected to increase groundwater discharge to the river substantially over the next 100 years, current work aims to quantify this increase.

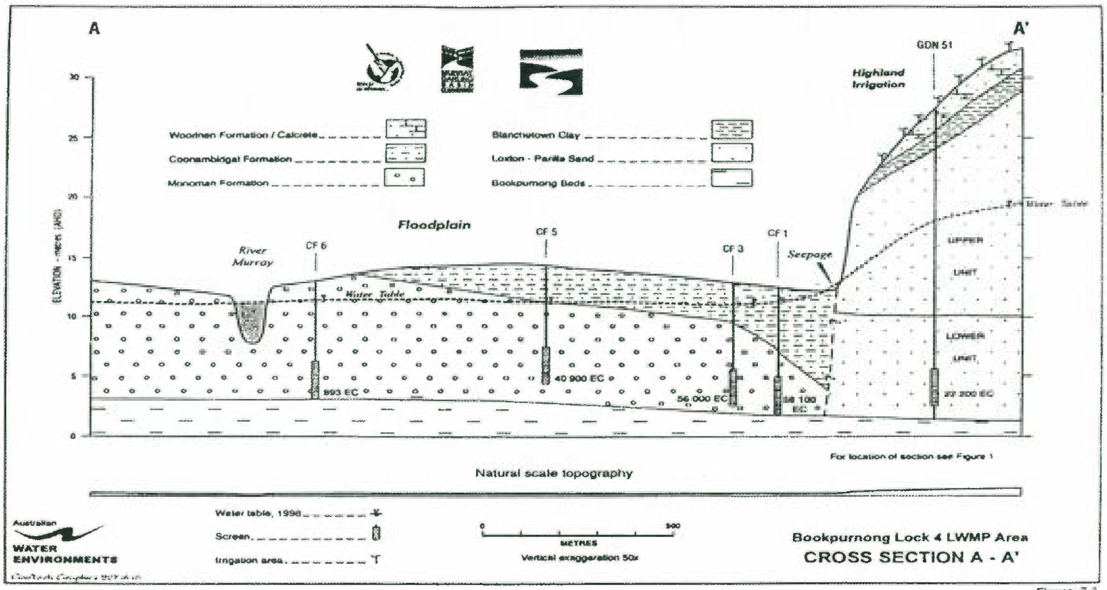

Figure 2. Illustration of impacts of irrigation development on water table levels. Cross section of groundwater table levels beneath Bookpurnong Irrigation area and Clarks floodplain (Australian Water Environments, 1999). Note seepage of high salinity groundwater (up to 50,000 EC) at base of cliffs, and mounding beneath irrigation area. The following vegetation associations occur around each piezometer:

- Piezometers CF1 and CF3 overlie a shallow saline water table, samphire is the dominant vegetation
- Piezometer CF5 is on an elevated bank of the river and is vegetated by blackbox and red gum mixed forest,
- Piezometer CF6 surrounded by red gum forest on a sandy peninsular underlain by fresh groundwater.

3. FLOODPLAIN VEGETATION OF THE LOWER RIVER MURRAY

The floodplains of the lower River Murray are typically vegetated by a mixture of river red gum (*Eucalyptus camaldulensis*), blackbox (*Eucalyptus largiflorens*) and lignum (*Muehlenbeckia florulenta* Meissner, formerly *M. cunninghamii*). River red gums tend to grow in less saline, more frequently flooded parts of the floodplain, typically adjacent to creekbeds. Blackbox is found at higher elevations away from the creeks, but with access to shallow groundwater (Slavich *et al.*, 1999a). Lignum grows in the lower, more frequently flooded areas of the floodplain (Craig *et al.*, 1991). Cooba (*Acacia stenifolia*) tends to co-occur with red gum. It is relatively salt and flooding tolerant, often replacing red gum in areas with shallow saline water tables.

Changes to the salinity and flooding regimes have altered the floodplain vegetation dynamics; the greatest impact is felt in the lower River Murray, due to naturally saline floodplain and groundwater conditions. Lignum grows in the more frequently flooded areas, almost independent of the shallow water table (Figure 3). Red gums have died out in areas remote from creeks with reduced flooding frequency and in some areas affected by groundwater discharge (Figure 3). Blackbox has moved to some of the more elevated areas formerly occupied by red gums. Samphire has replaced blackbox where health has declined due to shallow water tables and seepage.

Groundwater depth and salinity are key indicators of blackbox health due to their effect on soil salinisation rates. Blackbox tree health declines at infrequently flooded sites with high salinity groundwater (>40 dSm^{-1} EC) (Figure 3). Red gums show a similar trend, surviving at sites with high salinity groundwater when subject to frequent flooding. Lignum shows no relationship to groundwater salinity, preferring the more frequently flooded parts of the floodplain. The data presented in Figure 3 suggests a groundwater salinity of 40 dSm^{-1} EC is an important criterion for assessing blackbox health. Flooding frequency becomes important at higher groundwater salinities.

4. ASSESSING VEGETATION HEALTH STATUS AND TRENDS

Floodplain vegetation mapping using historical aerial photographs revealed trends in vegetation health decline closely correlated with the timing of irrigation development in the Loxton and Bookpurnong Irrigation Districts (PPK Environment and Infrastructure, 1997, 1998; Telfer and Overton, 1999). Vegetation health decline initially occurred at the base of the cliffs and later at the edge of the river adjacent the Loxton Irrigation District groundwater mound. The decline in vegetation health appeared to be independent of river regulation impacts. This indicates that the recent irrigation developments from traded water allocations, and uptake of sleeper water allocations may not affect floodplain vegetation health for 10 to 20 years.

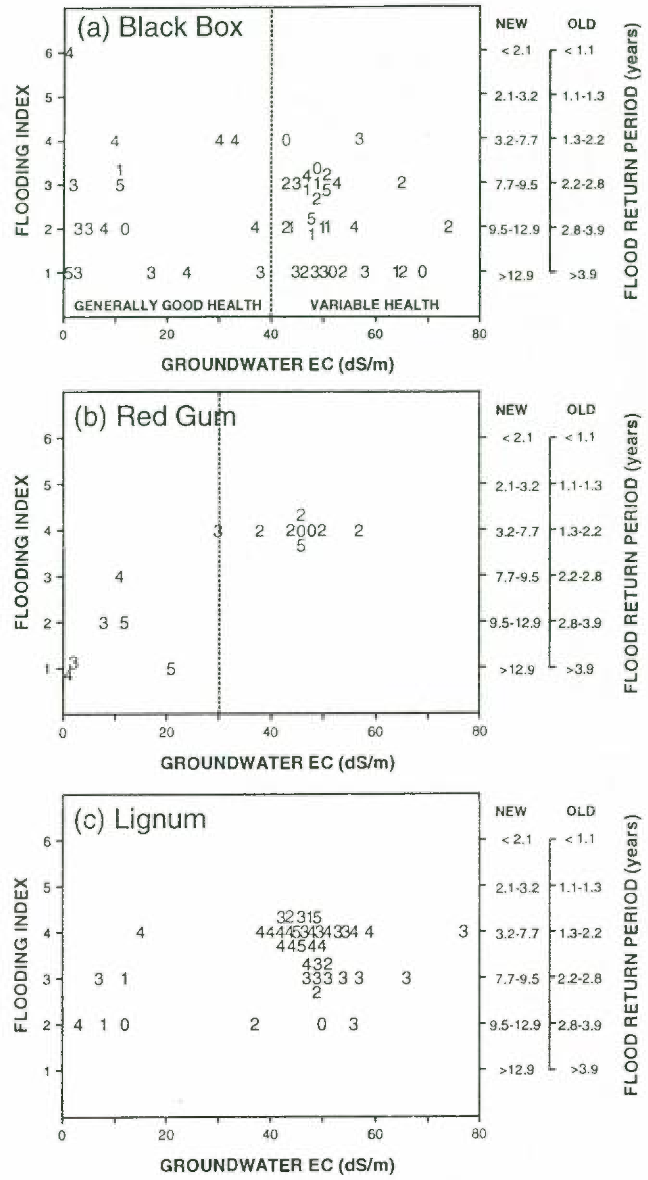

Figure 3. Relationship between flooding frequency, groundwater salinity and vegetation health (0 dead to 5 healthy). (a) The dashed line on this figure represents the division between blackbox vegetation generally in good health (< 40dSm^{-1}) and that in variable health. (b) Critical values for river red gum health appears to be associated with groundwater salinity (EC > 30 dSm^{-1}) and flooding frequency greater that 1 in 2.5 years. (c) There does not appear to be a relationship between lignum health and groundwater salinity, however flooding frequency appears to be the most determinant of lignum health.

Vegetation, soil and groundwater associations used in conjunction with other models can predict management outcomes and areas of floodplain vegetation likely to be at risk. A steady state soil salinisation model showed the effects of soil type on soil salinisation rates (Jolly *et al.*, 1993, Thorburn *et al.*, 1995). This model was extended using a dynamic soil, vegetation atmosphere transfer (SVAT) model to model the effects of various soil types, groundwater and flooding regimes on the vegetation (Slavich *et al.*, 1999a). Soil type and depth to groundwater strongly influenced soil salinisation rates between floods; the sandier soils reacted more strongly to floods than the clayey soils. Most floodplains are underlain by Coonambidgal Clay, a fine alluvial clay which is relatively impermeable. Little localised soil profile leaching was observed at Chowilla during early 1990s floods (Ackeroyd *et al.*, 1998; Jolly *et al.*, 1993, 1994; McEwan *et al.*, 1994).

Tree xylem water potential can be used to quantify vegetation health in relation to water stress, as it provides an instantaneous measure of how hard the tree has to work to extract water from the soil. The Grimes health index ranks tree health on a scale from 4 to 20, providing an objective measurement of tree health (Jolly *et al.*, 1996). A combination of measurements of soil and groundwater characteristics, modeling and historical aerial assessment of vegetation health, provide the necessary tools to assess and manage the floodplains of the lower River Murray.

5. Management options

Regional groundwater discharge must occur either into the stream, causing river salinisation, or into the floodplain, causing floodplain salinisation. Management can only shift the balance towards either stream or floodplain, or change the pattern of floodplain salinisation. Local variations in floodplain characteristics (aquifer, soils, flooding) determine the spatial pattern of discharge and hence floodplain salinity.

One of the ways to decrease salinity is to improve irrigation efficiency. However, there are limits, typical modeling scenarios use a maximum of 85% irrigation efficiency, allowing for the necessary flushing fraction. The 20-year time lags observed at Loxton, suggest that the extent of salt affected land is yet to be observed, and that improved irrigation efficiencies will not be observed for decades.

Revegetation is mostly ineffective in discharge areas, as the salinity of the groundwater in the lower River Murray in South Australia is often 50-60 dSm^{-1}. At these salinities, trees cannot remove sufficient quantities of water and even if they could, it would bring the salt to the surface more quickly. Trees adapted to saline areas transpire at a reduced rate. There is a limited niche for salt tolerant variants such as the blackbox green variant (Jolly and Walker, 1995b).

Slavich *et al.* (1996) showed that environmental flows maybe ineffective at controlling salinity in the higher parts of the floodplain inhabited by blackbox. Although floodplain soils are clayey and relatively impermeable, floods still play a part in providing both soil water and leaching, but for long-term health, large flood events are required. Given other restraints, releases of water from Menindee Lakes and Lake Victoria are restricted and would not be effective. On the other hand, large floods such as those of 1973 and 1974 provide sufficient leaching.

Groundwater interception schemes appear to be the most effective options available. The two operational schemes have shown they significantly reduce salt loads to the river, however vegetation benefits have not been quantified. There is a growing trend towards the incorporation of groundwater interception schemes into new irrigation development in order to minimise salt loads to the river. The implementation of these schemes has the potential to also benefit large areas of salt affected floodplain vegetation.

6. Conclusions

Soil salinisation and related decline of riparian vegetation have been identified one of the largest environmental impacts in the lower River Murray (South Western NSW, North-West Victoria and SA). In order to manage the floodplain vegetation, we need to have a clear understanding of the principal causes of the vegetation decline. We have developed the necessary tools to assess the causes of vegetation decline. To date, individual areas have been assessed with a range of methods, in a piecemeal manner.

This review has identified:
1. the impetus for investigating vegetation health decline, both economic and ecological reasons,
2. the processes causing a decline in floodplain vegetation health,
3. the methodologies or tools for assessing vegetation health, both instantaneously and incorporating the historical flooding and salinity regime.

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DRYLAND SALINITY

Towards a Predictive Framework for Landuse Impacts on Recharge: A Review of Recharge Studies in Australia

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Abstract

To develop sustainable salinity control strategies, a number of studies have produced estimates of recharge under specific soil and vegetation conditions. Yet there has never been an integrated review of these studies in a way that allows general predictions. By reviewing and collating many of the recharge studies from across Australia, this study seeks to develop generic relationships for assessing the impact of landuse changes on recharge.

Preliminary results indicate that for annual crops/pastures on sandy soils, Recharge = 0.20 * (Rainfall - 250). Large variation in recharge values for the loam and clay soils suggests that these two soil categories need to be considered more carefully to enable first order approximations of recharge to be made.

Introduction

Increased recharge is the cause of secondary dryland salinity. Quantifying this important term is necessary in order to assess management options, as well to provide input into groundwater models that assess the impacts on groundwater systems. While there have been several studies to estimate recharge for a specific area or groundwater system, there has never been an integrated review of these studies in a way that allows general predictions for sites elsewhere. This paper outlines preliminary efforts to develop generic relationships for assessing the impact of landuse change on recharge, by reviewing and collating information from many of the recharge studies from across Australia.

In order to develop any general relationships it is important to understand the key factors affecting recharge. Numerous studies (e.g. Kennett-Smith et al. 1994) have indicated that the three key factors controlling recharge are land use, soil type and climate, and that there are often interactions between these three factors. There is a general understanding that recharge is increased by more permeable soils, shallow-rooted annuals and by higher rainfall in a Mediterranean climate. However, there has been no attempt to develop this knowledge into empirical relationships. If generic relationships using these three factors could be developed, this would enable recharge/change in recharge to be estimated at any point, without the need for extensive field measurements. Where spatial data for these factors are available, it should be possible to develop these relationships to be used over large areas. Since, however, there is likely to be significant error in such relationships, we need to assess whether the error is too large for any particular purpose, and hence the extent to which fieldwork is required.

There are two main difficulties in completing such a task. Firstly is the problem that the wide spectrum of recharge estimation methods measure recharge at different depths and different scales, both temporal and spatial. For example, soil physical methods measure soil water drainage at a depth of less than one meter over the period of one year, while soil tracer methods measure vertical drainage over tens of meters over several years. Secondly, apart from the three primary factors controlling recharge, 'secondary factors' (e.g. aspect, slope, depth to watertable, preferential flow, impervious horizons and episodicity) can affect the amount of recharge, to varying degrees in different situations.

Looking at recent work by Zhang et al. (1999), the recharge values were put into context with other components of the water balance. Zhang et al developed a variation on the Holmes and Sinclair relationship, the rational function approach to evapotranspiration. By studying over 240 catchments from many parts of the world, the authors showed that for a given vegetation type, there is a good relationship between long-term average evapotranspiration and rainfall. The authors developed a simple two-parameter model that related the mean annual evapotranspiration to rainfall, potential evapotranspiration and plant available water capacity. This simple water balance model can be used to describe the effect of vegetation change on mean annual evapotranspiration.

The two parameters used in the above model are the plant available water coefficient (w) and the potential evapotranspiration (E_0). The soil water storage coefficient represents the ability of plants to store water in the root zone for transpiration. Figure 1 illustrates the sensitivity of the ratio of the mean annual evapotranspiration to rainfall with respect to the soil water storage coefficient, where low values of w have a low soil water storage coefficient and high value of w have a high storage coefficient.

Fitting the model to 240 catchments world wide, the best fit for forested catchments yielded E_0 of 1410 mm for a w value of 2, and for herbaceous plants E_0 was 1100mm for a w value of 0.5. Using the resulting relationship between annual evapotranspiration and annual rainfall for different vegetation types, the annual non-transpired water (i.e. recharge + runoff) can be easily calculated. The relationship between the annual non-transpired water or excess water and annual rainfall is illustrated in Figure 2 and will be referred to as the rational function approach to excess water.

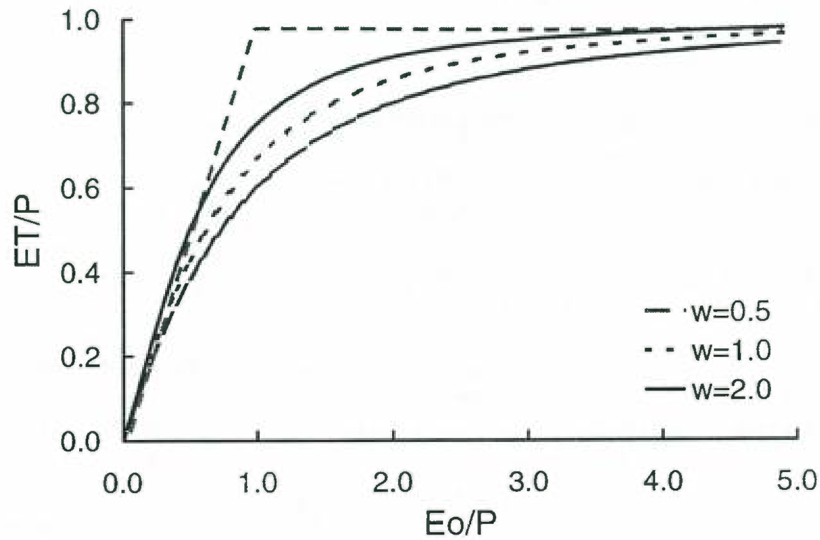

Fig 1. Ratio of mean annual evapotranspiration to rainfall as a function of the index of dryness (E_0/P) for different values of plant available water coefficient (w). (Source: Zhang et al. 1999)

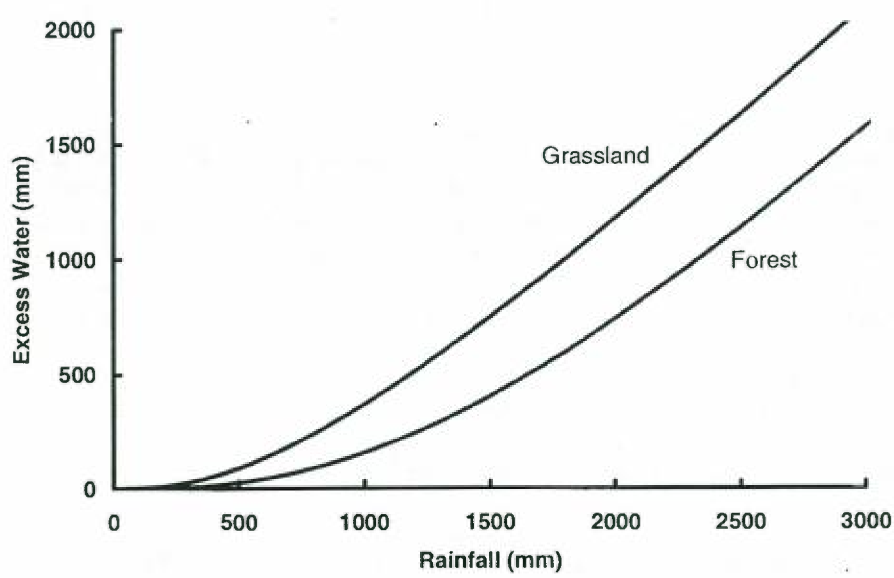

Figure 2. Relationship between water yield and rainfall for different vegetation types (Source: Zhang et al. 1999)

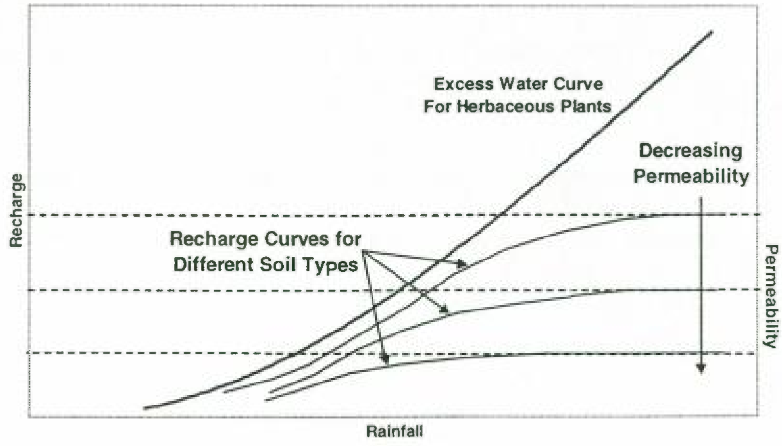

Figure 3. The anticipated effect of decreasing permeability on recharge.

In their present form, these curves act as an upper limit to the amount of water that can recharge underlying aquifers. To be useful in the estimation of recharge it will be necessary to partition the excess water into recharge and runoff. Runoff is generated when either or both the infiltration capacity of the soil or the soil water storage has been exceeded. One way to partition the recharge and runoff is to introduce two parameters, one based on the rooting depth of the vegetation and the second on soil texture.

It is proposed that for a given vegetation type, as the soil texture becomes finer and the infiltration capacity and the hydraulic conductivity of the soil decreases, the gradient of the partitioning curve will decline. It is thought that increasing rainfall will also cause the curve to continually decline. This is because at higher rainfalls there are likely to be more events which will exceed the infiltration capacity of the soil and secondly the soil water storage is likely to be at capacity for greater periods of time. For a given increase in rainfall, at high rainfalls a greater proportion of the increase will become runoff than at lower rainfalls. At high rainfalls it is expected that the partitioning curve will asymptote to a maximum recharge value based on the soil hydraulic conductivity (Figure 3).

In this study, data were reviewed and collated from across Australia. The data encompassed a wide range of landscapes, soil types, land uses and climatic zones. It is proposed that characteristic curves can be fitted through these data. These curves will enable estimates of recharge to be made and can be used to partition recharge and runoff in the rational function approach to excess water.

Method

This preliminary investigation is the summary of a review of over 30 recharge studies from across Australia. From these studies, the amount of recharge, method used, landuse, soil type and climate were all recorded. When presented, extra details (i.e. secondary factors) were added to the database. The inclusion of many different authors with a wide range of backgrounds meant that data were often presented in an inconsistent manner. This introduced an element of subjectivity when categorising or discarding data. To over come this difficulty a large emphasis was placed on maintaining consistentency in the decision making process on inclusion or rejection.

Data were discarded only when it was deemed that there were extraneous circumstances involved, i.e. 'where secondary' factors were found to be markedly influencing recharge, or where vegetation was in its first year of growth. Recharge was occasionally expressed as a range of values, and in such cases the mid value was taken. Where recharge values were expressed as being less than or greater than value '*x*', value '*x*' was taken to be the recharge.

Soils have been initially divided into three textural categories, sand, loam or clay. These broad groupings meant that soil type was the most subjective factor. Where soils were described in terms of clay content; clay content of under 10% was categorised as sand, clay content from 10-20% as loams and clay content over 20% as clay. Vegetation was also divided into three categories, annual crops/pastures, perennial pastures, and trees. Trees were categorised as annual crops/pastures until they reached an age of 5 years.

It should be mentioned that in this exercise existing studies were only reviewed where recharge had been physically measured. A separate, concurrent exercise is reviewing existing studies in which recharge values were synthesised using computer-modelling techniques. The spread of the recharge values from the two exercises will be analysed later.

Results

The data are presented according to soil type, i.e. sand (Figure 4), loam (Figure 5) or clay (Figure 6). A linear trend has been fitted through the data. In all three figures the rational function approach for excess water for herbaceous plants developed by Zhang et al. (1999) illustrates the excess water curve which serves as an upper bound for recharge (dotted line). In Figure 6, the rational function approach for excess water for trees is also illustrated.

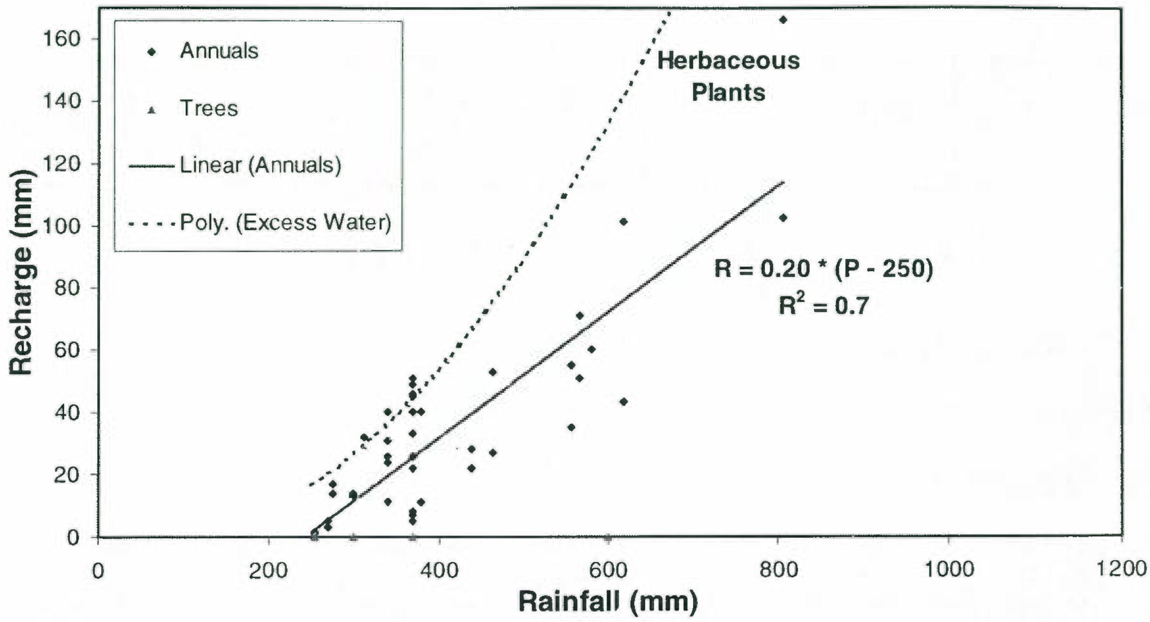

Figure 4. Recharge versus Annual Rainfall for Sand Soils

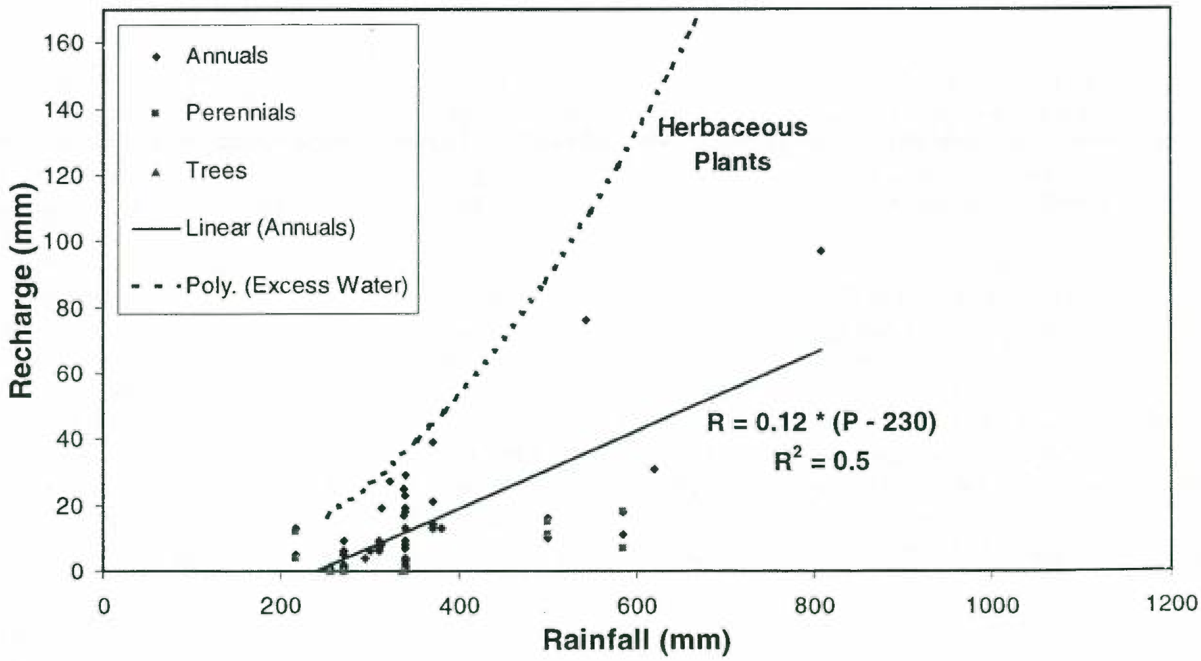

Figure 5. Recharge versus Annual Rainfall for Loam Soils

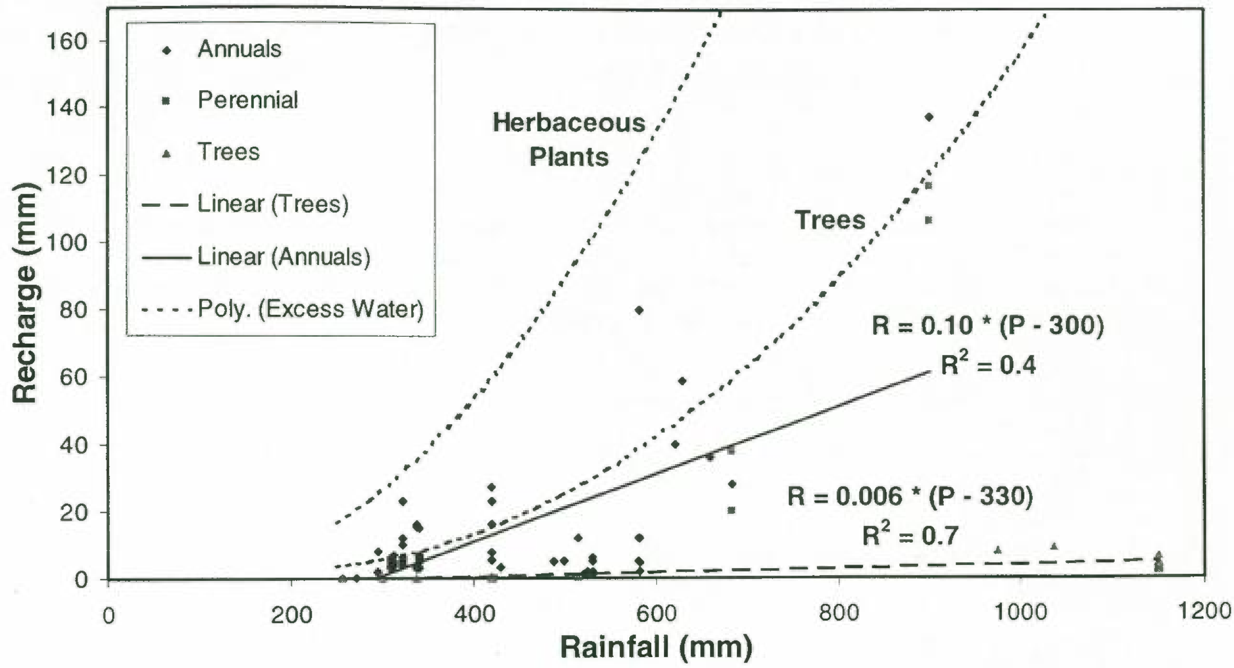

Figure 6. Recharge versus Annual Rainfall for Clay Soils.

Discussion

The large majority of the recharge studies reviewed to date have attempted to estimate recharge under annual crops/pastures in low to medium rainfall zones and under trees in low rainfall zones. Few studies have been reviewed in which recharge has been estimated under perennial pastures. There is also a deficiency of recharge data for trees and annual crops/pastures in medium and high rainfall zones. For these reasons, much of the discussion here will focus on annual crops/pastures.

The general trend of the data was that it moved away from the rational function approach to excess water curve as the soil texture became finer and as the rainfall increased. This suggests that higher rainfall and finer textured soils will have a greater runoff to recharge ratio. There was insufficient data to examine the effect of vegetation type, though initial results on clay soils (Figure 6) suggest that woody, deep rooted vegetation may also result in a greater runoff to recharge ratio. Despite having a greater runoff to recharge ratio, Figure 6 suggests that on clay soils the absolute runoff from landscapes covered in trees will be considerably less than the absolute runoff from landscapes planted with annuals.

In all three Figures (4, 5 and 6), the rational function approach to excess water which forms the upper bound on recharge lies above the vast majority of the recharge data. The few data values that did lie above the upper bound all had a sand soil type and a low annual rainfall. This is consistent with expectations, as annuals planted on sandy soil in a low rainfall zone would be expected to result in very little runoff, with most of the excess water becoming recharge. It is likely that some of the variation about the curve developed by Zhang et al. (1999) is due to different soil types. Vegetation on sand, which has a high permeability (i.e. much of the soil water drains below the root zone before it can be used by the plant), could be expected to transpire slightly less than the 'rational function approach to excess water' curve suggests. Thus some recharge values slightly above the excess water curve for sands are consistent with expectations.

Again consistent with expectations, the gradient of the linear trend fitted through the data decreased as the soil texture became finer, though the gradient of the trend through the clay data was only slightly less than the gradient through the loam data. Perhaps more importantly, the variation of values increased as the soil texture became finer. The distinction between the sand and the two finer textured categories was also more marked than the distinction between the clay and loam categories. This is thought to be attributed to the sand category being clearly defined, whereas the distinction between the loam and clay categories is less clear. At low clay contents it appears that the texture of the soil is important, whereas at higher clay contents (i.e. greater than 10%), there is little evidence that textural change is important. It is thought that the considerable variation within the clay category can be partly explained by the wide spectrum of clay varieties, which (although clearly falling into the clay category) may have quite different properties, e.g. data from both red duplex soils and grey cracking clays were grouped within the clay category even though these two soils have very different properties. Sub-dividing the clay category may reduce some of this variation, leading to a more robust relationship.

Kennett-Smith et al. (1994) found a strong relationship between the percentage clay content in the top 2m of the soil profile and potential recharge. However, many studies do not present information on clay content making this form of analysis impractical. Further more, this form of analysis does not allow the effect of textural change, or the depth at which textural change occurs to be assessed.

While difficulties in describing the soils may account for some of the variation in the results, the literature confirms that different techniques of measuring recharge and 'secondary factors' are also responsible. Such factors include aspect, slope, depth to watertable, preferential flow, impervious layers and climatic variations like summer versus winter dominant rainfall.

It is thought that trying to reduce variation in the data by incorporating secondary factors into the generic relationship would not be practical. This is because many studies do not present data on 'secondary factors', that there is insufficient spatial data for many secondary factors to enable their use in a generic relationship, secondary factors important at the point scale may not be important at a larger scale and finally, additional factors would add extra degrees of complexity to the task.

Conclusions

Preliminary study of the data reviewed indicates that, for each soil category, there are distinctly different trends in recharge between annual crops/pastures and trees. The data considered to date suggest that replacing annual crops/pastures with trees will bring about significant reductions in recharge. The confidence in these findings decreases as the soil texture becomes finer and as the rainfall becomes lower. The data show that landuse has by far the greatest influence on recharge.

Data on perennial pasture are too few to exhibit any trends. From the data presented in Figures 4, 5 and 6 it is unclear whether replacing annuals with perennials will reduce recharge. The recharge under perennials at a site may be less than the recharge under annuals, but unlike trees and annuals, there may not be enough difference in recharge to prevail over the variation between sites.

Preliminary results indicate that for annual crops/pastures on sandy soils, Recharge = 0.20 * (Rainfall – 250). Current data indicate that categorising finer textured soils needs to be done more carefully to enable first order approximations of recharge to be made for other soil types.

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Catchment Categorisation: what is it and how does it work?

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Abstract: The current annual cost of infrastructure, land and stream degradation by salinity is estimated at $350 million, not taking into account the invaluable loss of biodiversity. If large-scale land use changes are not undertaken immediately, the cost of salinity is estimated to increase at least five fold in the coming decades. These large-scale land use changes have to be underpinned by a good understanding of the biophysical processes driving salinisation and have to be implemented efficiently and rapidly, otherwise both funds and good will are wasted, with the long-term consequences mentioned previously.

Lack of data is often blamed for the inability to deliver reliable management guidelines for salinity control within a reasonable time frame in large or/and poorly documented catchments. But is data the problem or the lack of solutions for porting results and understanding from well to poorly documented catchments? The Catchment Categorisation proposes a methodology of information and technologies transfer from well understood catchments to poorly or unstudied catchments across Australia. The application of the Catchment Categorisation concept is currently under way in the dryland salinity theme of the National Land and Water Resources Audit and will be pursued in the MDBC funded Catchment Categorisation project.

1. Introduction

In the last 20 years, extensive research in salinity has been undertaken. In 1993, the National Dryland Salinity Program (NDSP) was set up to finance and coordinate the scientific effort and collate the research outcomes to ultimately provide solutions for dryland salinity. Whilst a big step forward was made under the NDSP, there is now a strong need for a framework within which scientific outcomes can be efficiently translated into on-ground action and policy development. A National Catchment Classification system is proposed as an aforementioned framework. It should provide conceptual models for the various hydrogeological systems which permit identification of catchments and landscape units that will respond in a similar way to land use change. It should also provide a good basis for transferring understanding and methods from well to poorly documented catchments.

2. The National Catchment Classification system

Replacing native vegetation by less water efficient ('leaky") agricultural practices caused a hydrologic and hydrogeologic disequilibrium resulting in dryland and stream salinisation in which groundwater processes play a key part (salt mobilisation and redistribution). The nature of aquifers is described by attributes that include geology, physiography and weathering style (Coram, 1998). These three interrelated characteristics are found in an almost unique variety of combinations in each catchment but, at a broad level, some regional similarities can be recognised (Coram, 1998). These similarities can be expected to provide a credible base for the transfer of understanding from well studied catchments to poorly documented ones of the same type, which will help to prioritise research work for remedial options.

In 1996, a group of hydrogeologists from all States proposed that the broad features of common occurrences of salinisation related to groundwater across Australia can be described by the fifteen hydrogeological models. The models are listed according to whether they represent local, intermediate or regional flow systems (Coram, 1998) primarily because these scales reflect the ease with which salinisation can be managed. However, in a number of instances, groundwater behaviour may be the result of several superimposed systems; for example, the specific area of discharge may be controlled by local hydrogeological features although the general region is controlled by regional processes.

The local hydrogeological systems described below have shallow circulation depths and have recharge and discharge areas close together. They tend to occur in areas of higher relief, and are usually unconfined. The typical scale of a local flow system is in the order of one to three kilometres.

The intermediate hydrogeological flow systems described below are intermediate in scale between the local systems and regional systems. They tend to occur in foothills and valleys, over a horizontal scale in the order of five to ten kilometres. Intermediate flow systems may be overlain by local flow systems.

The regional hydrogeological systems described below have deep circulation depths, and recharge and discharge areas separated by considerable distances. The aquifers are usually confined, and groundwater residence times are long. The typical scale of a regional flow system is in the order of fifty or more kilometres, and regional flow systems are likely to be overlain by local and intermediate flow systems.

The broad features of local, intermediate and regional systems are summarised below in Table 1.

	Local flow system	Intermediate flow system	Regional flow system
horizontal scale	• 1-3 kilometres	• 5-10 kilometres	• >50 kilometres
geomorphology	• subcatchments in higher relief areas on edges of plateaus and ranges	• alluvial and, occasionally, glacial valley fill in foothills and valleys	• broad riverine plains on depositional basins
geology	• fractured metamorphic and igneous rocks • colluvial sediments • aeolian sediments	• fractured metamorphic and igneous rocks • shallow (<50 metres deep) alluvial and colluvial sediments	• deep, interbedded marine, alluvial and aeolian sedimentary sequences (several hundreds of metres deep)
structural features influencing groundwater flow	• subsurface low hydraulic conductivity features such as bedrock highs and dykes • termination of aquifer at erosional surfaces	• reductions in hydraulic conductivity with distance from sedimentary source or aquifer weathering • reductions in hydraulic gradient or aquifer thickness with shallowing of the ground surface • subsurface low hydraulic conductivity features such as bedrock highs and dykes	• reductions in hydraulic conductivity with distance from sedimentary source or associated with structural deformation (faulting, folding) • reductions in hydraulic gradient or aquifer thickness with shallowing of the ground surface

Table 1: Summary of the broad features of local, intermediate and regional systems (from Coram, 1998)

3. Dryland Salinity Theme of the National Land and Water Resources Audit

The NLWRA aims to develop a basis for evaluating land use changes and land management options for salinity reduction in key Australian landscapes by:

- developing a consistent analytical framework for evaluating costs, benefits and the recharge and salinisation response at the catchment scale for a range of land use and land management options;
- testing this framework for selected representative case-study catchments;
- proposing methods for extrapolating the case-study results to develop Australia-wide land use change and land management implications, including the need for interventions and development of resource protection policies.

For each case study catchment, the following will be determined:

- areas of the catchment where changes in recharge will most affect catchment salinity
- the magnitude of recharge reduction required in those areas to produce a given level of salinity management (eg. % reduction in area of salt-affected land)
- land use and farming system options for producing recharge reductions of sufficient magnitude to achieve salinity management
- information on which to base an economic analysis of the costs, benefits and viability of the options for change
- constraints to achieving required changes

When coupled with the economic analysis performed in Capacity for Change theme of the Audit (Theme 6), the outcome for each catchment will be: (1) an analysis of the costs and benefits of effecting a given decrease in salinisation in that catchment; and (2) provide a framework to develop trade offs between salinity control and living with salt options.

A preliminary analysis of how the catchment-specific results may be extrapolated across key landscapes will be carried out. The aim is to extrapolate understanding, methodologies and results with a level of confidence adequate to provide (at a minimum) a first-cut assessment of the costs and benefits of land use change / land management options for reducing catchment salinisation and therefore a basis for prioritising works and management activities.

This analysis will account for the influence on catchment-specific outcomes of factors such as catchment topography, geology, climate, land use, agronomy, and local economics, as well as the nature and community cost of the impacts of salinisation (such as on roads, wetlands, towns, water resources and agriculture).

3.1 Australia-wide groundwater system mapping

At the time of writing, the first version of an Australia-wide map of groundwater types distribution[1] was produced by the Bureau of Rural Sciences in a GIS using the following information layers:

- Bedrock Geology Coverage, 1976 BMR/AGSO Geology of Australia, 1: 2 500 000
- Regolith Terrain Map of Australia, BMR/AGSO (Reclassed) Regolith Polygons: BMR record 1986/27, 1: 5 000 000
- Digital Elevation Model, AUSLIG, 1km cell size; derived from AUSLIG 250m 9 second DEM

It has to be noted that no generic catchment classification system will ever be right at all scales, all of the time, nor will it be any more accurate than the data sets upon which it is based. The challenge is here to develop a system that is accurate enough to be a useful strategic and operational management tool (Coram, 1998). The product will be reviewed by the contact hydrogeologists from each State and updated until general agreement.

Given that the spatial distribution of groundwater types is now known, the next step is to determine the range of processes leading to salinisation that occur within these types and develop or adapt biophysical modelling tools for each of these types. Within the time frame of this Audit theme (July 2000) it is not possible to cover focus catchments encompassing all 15 groundwater types. In the Audit, four focus catchments representative for four groundwater systems were selected and, in the future, more will be chosen for the MDBC Catchment Categorisation project. The Audit case-study catchments are:

- Wanilla, South Australia
- Lake Warden catchment, Western Australia
- Kamarooka, Victoria
- Upper Billabong, New South Wales

For each catchment, the aim is to develop a strong conceptual framework for the biophysical processes involved in salinisation and develop and apply optimised modelling tools based on the conceptual understanding to evaluate management options. Ultimately, the lessons learned on a given catchment/groundwater type should be transportable to other similar catchment/groundwater types. To date, the first case-study catchment is close to completion and the methodology and results are detailed in the following sections.

3.2 Wanilla catchment case study (South Australia)

3.2.1 Site description

Physiography: the Wanilla catchment is located in the southern part of the Eyre Peninsula, South Australia, 40 km north-west of the town of Port Lincoln. It covers an area of approximately 140 km^2. Two major drainage lines are present, flowing from the Lincoln Uplands through the middle of the plains down to Kellidie Bay. This bay is on the south western boundary of the catchment, near the town of Coffin Bay.

Climate: the annual mean rainfall is 550 mm, based on 100 years of daily rainfall recorded at Port Lincoln. Rainfall is highly seasonal, with the wettest period over the coolest months between May and September (mean monthly rainfall > 60 mm), and the driest period over the warmest months of January and February (mean monthly rainfall < 20 mm). The mean annual class 'A' pan evaporation at Tod River (15 km north-west of Wanilla) is 1475 mm, based on approximately 20 years of data. Mean monthly rainfall values exceed evaporation only during the winter months from May to September.

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[1] The map was not included in this article for obvious readability reasons.

Geology: the following geological description is largely taken from Parker *et al.* (1984), except where otherwise acknowledged. The Wanilla catchment can be divided into three geological provinces. The first is comprised of the Early Proterozoic basement rocks, which consist of Hutchinson Group gneiss and Lincoln Complex granites in the Wanilla catchment. The Early Proterozoic rocks outcrop predominantly in the eastern highlands of the catchment, with small outcrops also in the north east, and south east of the catchment. Overlying the Hutchinson Group, Tertiary and Quaternary sediments outcrop in the Wanilla catchment in the plains to the west of the Lincoln Uplands. These include the Tertiary Uley Formation, a unit consisting of fluviatile-regolithic quartz sand, sandy clay, clay and rare gravel with abundant laterite nodules. This has a maximum thickness of 40 m in the southern part of the Eyre Peninsula (Barnett 1978). Thin veneers of Quaternary sediments overly the earlier rocks. These include the extensive Pleistocene Bridgewater Formation, which consists of calcrete and carbonate-cemented aeolianite and exhibits large dune size cross beds outcropping in the southern part of the Wanilla catchment. The younger Late Pleistocene-Holocene Wiabuna Formation consists of clayey sand which forms the cores of sand dunes, and outcrops in the north west of the catchment where it constitutes a minor topographical feature. Recent alluvium can be found along the drainage lines in the centre and east of the catchment.

3.2.2 Conceptual model:

The conceptualisation of the Wanilla catchment involved both the identification of aquifer thicknesses, and the estimation of groundwater flow direction with a flow-net. Groundwater elevations from 38 surveyed observation bores at screened at depths of 3 to 38 m indicate that regional groundwater flow (Figure 1) is in a south-west direction from the Lincoln Uplands towards Salt Creek and Merintha Creek, towards Kellidie Bay (a direction coincident with that of surface drainage). Regional groundwater flow patterns suggest that a regional zone of groundwater discharge occurs to the north of the study area.

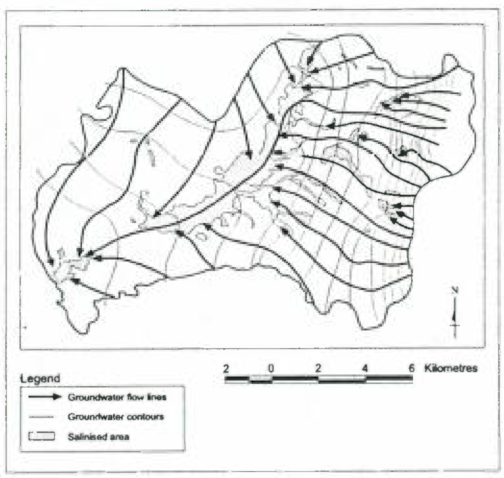

Figure 1. Groundwater flow paths (based on data collected in 1991 and 1992), and mapped salinity zones (from Henschke 1999)

The flow directions indicate that the Wanilla catchment groundwater system is independent from the regional system to the north (Cummins system). Most of the flow is occurring from the basement highs in the east and the north east of the catchment, and some flow comes from the Wiabuna Formation in the north-west of the catchment. Groundwater flows out of the catchment through the Bridgewater Formation into Kellidie Bay. Groundwater gradients are high (1.82×10^{-2}) in the east of catchment, as groundwater flows from the highlands of the relatively impermeable Proterozoic basement. It becomes lower (2.7×10^{-3}) in the Tertiary Sands, and lower again in the Bridgewater Formation (1.5×10^{-3}). Groundwater flows with a velocity of approximately 0.5 m/yr (assuming hydraulic conductivities of 0.1 m/day and 0.5 m/day in the Proterozoic basement and the Tertiary Sands respectively).

Groundwater discharge in the Wanilla catchment generally occurs along surface drainage lines and comparing the groundwater flow paths with the catchment geology and topography suggests that groundwater discharge is controlled by three main processes:

- convergence of groundwater flow lines from the uplands in the north east and the east of the catchment;
- local discharge of groundwater above bedrock highs in the north of the catchment, associated with the edge;

- changes in hydraulic gradient at the break of slope, as groundwater flows from the Lincoln Uplands to the plains.

3.2.3 Groundwater modelling:

3.2.3.1 Method

For this study, Flowtube model was used. Flowtube is a simple groundwater model based on conservation of mass with fluxes calculated using Darcy's Law (Dawes, 1999). All flow is one-dimensional along a tube defining the aquifer of interest, with recharge and discharge able to be distributed both spatially and temporally. The model can also simulate a network of tubes, if required.

The Wanilla catchment was modelled using two arms in the upper part, which merged to become a single lower arm for the lower part of the catchment (Figure 2). The main arm (Arm 1 & Lower Arm) was divided into 22 sections, coinciding with monitored bore locations and surface water and boundary features. In addition, a second arm was modelled (Arm 2) for part of the upper catchment (8 sections). The derived ground and groundwater surfaces were smoothed using a 4^{th} order polynomial and the surface was described with 25 strips each 1 km in length.

The width of the aquifer was taken from the surface catchment boundary; aquifer thickness was estimated from cross-sections based on bore information, hydraulic conductivity and porosity values were extended from those reported by Richardson *et al.* (1994a,b) in the Pope's Catchment.

To take the strong rainfall seasonality into account, the year was broken into two parts:

- a 3 month period when 20 mm of *recharge* (a rate of 0.21 mm/day) was distributed uniformly over the landscape, and
- a 9 month period when 20 mm of *discharge* (a rate of 0.07 mm/day) was distributed uniformly over the catchment.

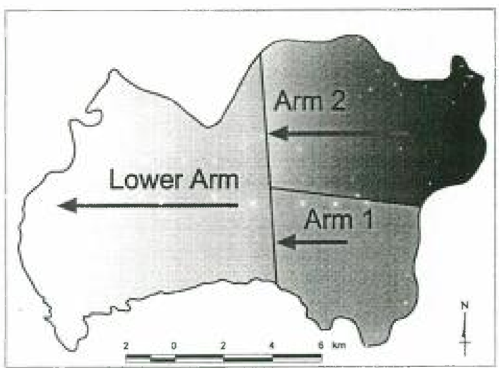

Figure 2: Simplified branch structure of the Flowtuble model set up - Wanilia catchment.

3.2.3.2 Results

The discharge is assumed to be evaporated water in excess of rainfall sourced from the shallow aquifer. The area at the top of the catchment, known as Pope's Catchment has a very steep basement (c. 3 to 5%) and requires 100 mm/year of recharge to be placed in the upper-end to maintain a water table. Given this detail, the very low conductivity, and the observed heads, it is unlikely that an aquifer existed upslope of the swamp (between 13-17 km upslope) under native vegetation. A good fit was obtained between smoothed observations and long term equilibrium (RMSE 3.6 m with n=24) with the largest differences of 5-7 m in the central swampy area where observation water tables were apparently 7-10 m below ground level. The predicted long-term heads are shown in Figure 3 for the scenario outlined.

These results confirm that the wet area at break of slope (discharge) is a permanent feature of the landscape. Discharge in the break of slope area is not only due to the change in topographic gradient, but also to a lateral groundwater flow constriction coinciding with the merging of the two sub-catchments of the upper catchment (figure2, arms 1 and 2). Under native vegetation, both the upper and the lower catchments were underlain by deeper watertables, the surface wetting up seasonally in the lower part of the catchment.

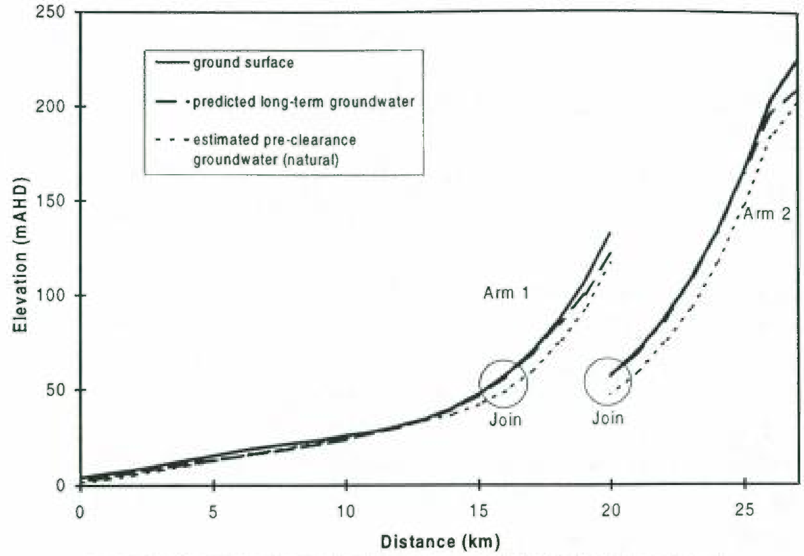

Figure 3. Observed and predicted steady-state heads for Wanilla catchment.

4. Conclusion

The Australian-wide groundwater system types map is a good first-cut of the spatial distribution of the different systems. Feedback provided by the State hydrogeologists involved in the project will improve the product with time.

The early case study results have shown each step of the work provides valuable feedback that can be turned into management recommendations. For instance, features like groundwater flow restrictions or flow convergence are common in a wide range of catchments and these areas are naturally prone to discharge and salinisation. At this stage, historical information can provide hints as far as the landscape evolution in response to land use change is concerned. If *historical* discharge features (waterlogging, salinity) due to flow restrictions can be identified, there is probably no biological option to "mop-up" the excess discharge water. These results inherited from the conceptual understanding of the groundwater processes can, if needed, be enhanced by groundwater balance modelling. In the case of the Wanilla catchment, a simple groundwater modelling approach (Flowtube) captured the main features related to salinisation like discharge in the middle part of the catchment and salinisation hinge line moving upslope with increased recharge. It demonstrated that under native vegetation, the upper part of the catchment was "dry", but that the lower part of the catchment was still seasonally affected by waterlogging. More fine-tuning of the model will give a range of acceptable recharge values for the upper part of the catchment to prevent the salinisation spread and that can be translated into management options. The next step, if needed, would the be to do some detailed water balance work for the upper part of the catchment to come up with the best socio-economic land use options for the area, but this might well be superfluous because it has already been achieved in the previous step.

The Wanilla example showed that a catchment study should be an iterative process focussing at each step on possible management options. Each step also involves an increase in data needs that translates into an increase in cost. The main purpose of the Catchment Categorisation framework is to prioritise the work and to help the professionals involved in a catchment study to ask the right questions and deliver answers at the level of complexity that is required to provide credible management options.

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Eastern Murray (NSW) Groundwater Investigation and Monitoring

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Dryland salinity has been identified as an issue in several eastern Murray sub-catchments. Previous studies undertaken in West Hume and Holbrook Landcare areas (1989, 1991) revealed that groundwater pressure levels have been risen by 30-60 cm/year. The objective of the current study is to raise awareness of rising groundwater levels and related problems among all 10 Landcare groups in the eastern Murray dryland catchments. This has involved up to 130 landholders monitoring bores and piezometers regularly and DLWC preparing depth to watertable and pressure level maps, trend assessment and also groundwater salinity and recharge assessment maps.

THE FULL TEXT OF THIS PAPER WILL BE AVAILABLE AT THE CONFERENCE

The Liverpool Plains ICSM Project : Lessons to be Learned

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Abstract. This is a personal view of the outcomes of a modelling project I worked on. The Integrated Catchment Scale Modelling Project (ICSM) determined the optimal land management system for dryland salinity affected catchments in the Liverpool Plains, NSW, in terms of economic return to farm enterprises and minimising recharge below the plant root zone. After defining the conceptual model of the groundwater systems and estimating the water balance for the main aquifer, it became clear that surface agricultural management would have little impact on the deep flow system. The role of modelling became one of narrowing the possible outcomes from a short list, rather than exploring the sensitivity of the groundwater system to changed inputs. A major conclusion was that a new approach is required for these problems, so the minimum appropriate level of effort is expended to get an outcome. Doing good science did not mean applying complex models and ticking off milestones, it meant gaining a clear understanding of the system so that management options could be evaluated easily and unambiguously.

1. Introduction

The discourse here is a personal reflection of what the author learned, or thought he learned, after being involved in this very ambitious project. At the time this project was being put together, there were not many results from studies that went through the entire spectrum of catchment investigation. From hydrogeology, biophysical modelling, socio-economics to policy on the ground, wasn't heard of. There were instances of several pieces being done at smaller scales, or single investigations at the catchment scale, but little or nothing that ran through the entire gamut. The available tools were limited, and often the province of a few groups, rather than being generally used or useable.

Within this context, it was unknown what would be gained from this project, or even if the project would achieve its objectives. There could be several threads. Would we get better tools to answer the land degradation questions? Would these be operational or research tools? Would the models be transferable? Would land management policy and outcomes be transferable? Would we get a clear understanding of the problems and solutions in taking a project from go to whoa? It is this last aspect that I will concentrate on.

Almost all the land degradation problems affecting catchments in Australia today have been blamed, at one time or another, on the consequences of clearing native vegetation and replacing it with European annual farming systems. The major consequence is increased recharge to the groundwater systems of this wide brown land. Increased recharge leads to rising water levels that bring with them salts and other chemicals stored either in the soil column or the groundwater itself. When this cocktail is near to, or at, the soil surface, a variety of land degradation problems result. Dryland salinity is a serious land degradation problem, and Lovering (1999) estimated that 25,000 km^2 of Australia has already been affected by it, and that up to 150,000 km^2, a six-fold increase, may ultimately be lost. There is an expectation in Australia that by modifying cropping practices and applying better land management that the ultimate extent of dryland salinity can be reduced. The use of biophysical models in developing Whole Catchment Plans has been advocated to put rigour into the process of determining these best practices. The ICSM project was designed to help address these issues. The following is a quote from the Executive Summary of Evans and Johnston (1998):

> *"The ICSM Project ... was funded as part of the National Dryland Salinity Program to provide a truly multi-disciplinary approach to the problem of catchment and farm scale planning to manage dryland salinity. The aim of the ICSM project was to assess the impact of a variety of land management options on water and salt balance at the whole-of-catchment scale, to better reflect the internalisation of all the costs (both on and off site) related to salinisation. ... The methodology specifically set out to test the use of existing knowledge and methods, with a minimum of additional data gathering or development of new methods or models. The project was undertaken in the Liverpool Plains (one of the five focus catchments of the NDSP) by a team drawn from AGSO, CSIRO, ABARE, NSW DLWC and LPLMC."*

This paper will detail the lessons that were learned by the multi-disciplinary team completing the objectives of the project. It was found that (1) preconceived "best bet" options would not work in this case, (2) an understanding of the sources, sinks and pathways for water and salt in the groundwater system was critical, (3) iterative development of the conceptual model with a simple groundwater model aided this understanding, and (4) different land management options could be evaluated at various levels of detail that suggested an appropriate level of effort and amount of data required. Good science is more that just applying models.

2. Possible Options and Outcomes

To labour an important point made in the Introduction, since we blame changed land management for most of our land degradation problems, specially dryland salinity, we expect that a change in land management will fix the problems. To this end there is a body of work related to farming systems that minimise recharge below the root zone to mimic native vegetation (Hatton and Nulsen, 1998). This is not the only option, and we can take a step back to look at remediation in several ways.

For any catchment affected by a land degradation problem, there are five categories of remediation work required:
1. The catchment doesn't have now, and will not develop, any problems so no work is required,
2. With minor biological or engineering action, future problems can be averted and current problems solved,
3. With minor work problems can be stabilised, and with major work they can be solved,
4. With major work catchment-scale problems can be stabilised, and locally the problem may be solved,
5. The problem is so bad that nothing can be realistically (or practically) done to solve it.

In socioeconomic terms, these categories relate to:
1. Farmers can try to maximise profit with little effect on the land,
2. Some careful management is required to prevent land degradation, but profits are not greatly affected, everyone is generally happy,
3. In many areas better management will be required, and in others engineering measures will be needed to avert land degradation, those suffering off-site effects and reduced income from their property will be most active,
4. Better and more radical land management including retirement of land for forests in concert with pumping of groundwater is required with a catchment-scale focus, profits will be affected in many cases, the whole catchment community becomes involved,
5. Any money the farmers can make is a bonus, large socioeconomic upheaval, the land is doomed for traditional farming practices.

In management terms, these categories will entail:
1. Doing nothing but making a good living,
2. Planting mixed annual and perennial pastures to prevent excessive drainage during fallowing, fence off small areas to promote local regeneration of native species,
3. Changing cropping rotations to include deep rooted summer active perennials such as lucerne to help dry the soil to a reasonable depth, maybe plant 10-25% of the catchment to trees or investigate agro-forestry, some land drainage may be required,
4. Plant 33-50% of the catchment to trees, investigate best placement for groundwater pumping bores and enhanced surface drainage,
5. Grow whatever you can, learn to love degraded landscapes, sell the farm if possible.

Much of the focus of work during the Landcare Decade has been in Categories 2 and 3, with the expectation that better crop management will halt the spread of salinity, or ultimately reverse the trends. More recent work by Hatton and Salama (1998) has pointed to many parts of Western Australia being in Category 4, and they suggest radical catchment-scale intervention to prevent the disastrous spread of salinity reported by Lovering (1999).

One of the conclusions from the ICSM Project was "a refinement of our research approaches." This entails gaining a detailed understanding of the system before starting a lot of work which may be unnecessary. So we might come into a catchment and look at the gross water balance terms for the groundwater system. If the best and worst case scenarios both indicate that the aquifer has sufficient flow and storage capacity to handle the recharge, then we are in Category 1 or 2 and may not need to do any detailed work. Similarly if both extreme cases show that land degradation is inevitable then we shift up to Category 4 or 5, and this suggests immediately what research work is required in the catchment. It is when the best case scenario is in Category 1 to 3 and the worst case is in Category 3 to 5, that detailed work to accurately determine the sensitivities of the system and the social and economic impacts of land management change is required.

3. In the Liverpool Plains

3.1 Importance of the Conceptual Model

The first step in the ICSM Project was to understand the surface and groundwater systems that had led to dryland salinity in the Liverpool Plains. This exercise is usually done at the same time as data collection and analysis for parameterising a large complex model, used later to test management scenarios. Stauffacher *et al.* (1997) presented a detailed conceptual model of the groundwater flow systems in the Liverpool Plains, the sources and sinks of water in the catchments, and simplifying assumptions that could be used for any modelling, but specially the simple numerical modelling of the deep confined aquifer.

The first revelation was that dryland salinity was not widespread in the catchment, and in fact was restricted to two sub-catchments with specific geological and surface characteristics. This immediately reduced the amount of work to be done looking at dryland salinity by reducing the number of sub-catchments to consider. It was expected at the outset that the five sub-catchments, or the entire 12000 km^2 area, would need to be modelled in some detail.

The second important detail was described by Zhang *et al.* (1997) who quantified the amount of water potentially entering the groundwater system from the various sources. The amount of runoff from the steep uplands that could potentially enter the groundwater system through a gravel contact zone, was an order of magnitude larger than the estimated vertical recharge below crops. It therefore appeared that the now standard approach of minimising crop rotation recharge, and the complex physical and economic modelling this usually requires, was not relevant to these catchments, with respect to the deep confined aquifer. Far more could be gained apparently by reducing the amount of runoff from the hills and ranges through reafforestation or engineering interception of runoff.

3.2 Iterative Refinement of Conceptual Model

An estimate of the partitioning of the vast amounts of runoff water produced in the Liverpool Ranges needed to be made to complete the conceptual model, and to resolve the relative importance of the water from the hills *versus* the plains. This process was an iterative refinement of the conceptual model through simple groundwater modelling.

Examination of the limited streamflow records for one of the salt affected sub-catchments, suggested that although the catchment floods periodically, very little water actually leaves the catchment; around 5% of the runoff from the hills, see Figure 1. This is a very important point, since the approach used by Zhang *et al.* (1997) would suggest that around 20% of annual rainfall should become runoff or recharge from the high rainfall year. Estimations of groundwater flow through the constricted lower parts of the sub-catchment with a very simple flow model suggest that around 5% of the hills runoff passes through the aquifer. From the few available bore hydrographs it appears that the deep aquifer is quite full but filling very slowly, so the amount going out must be very similar to the amount coming in. This leaves 90% of the runoff from the hills to disappear on the plains.

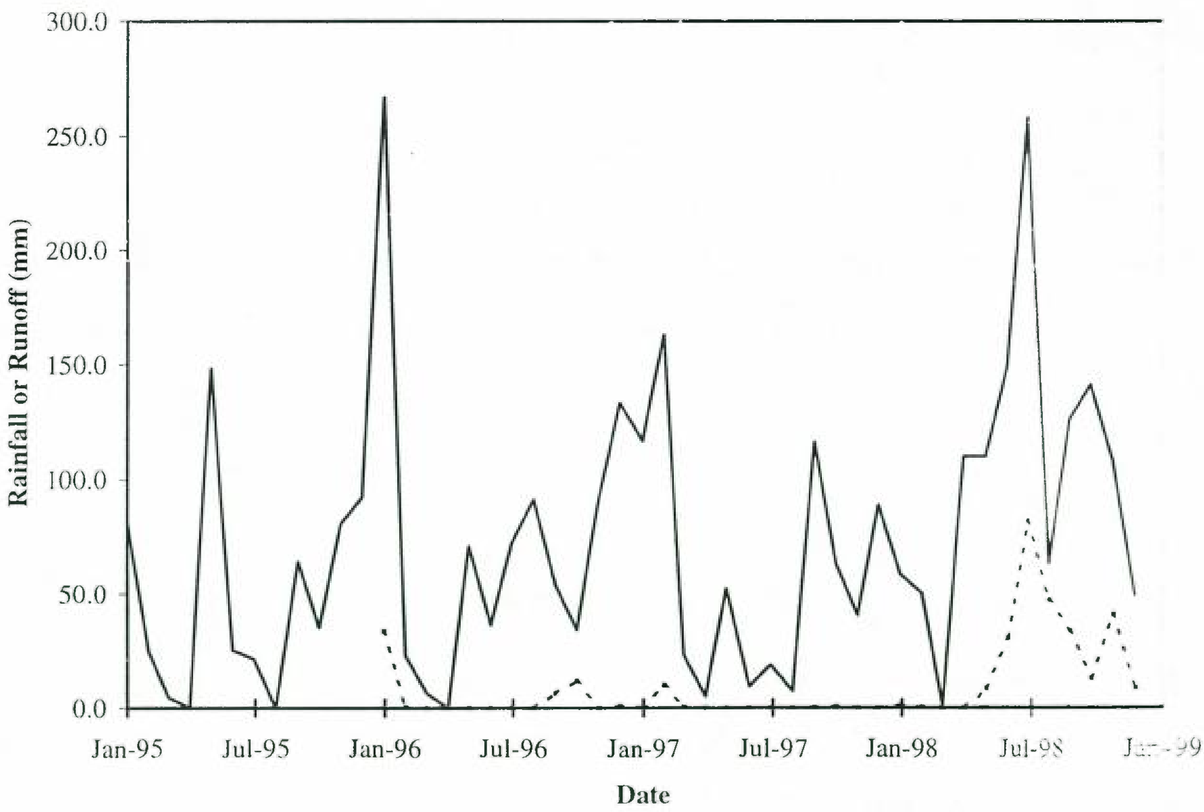

Figure 1 : Monthly rainfall and runoff from Big Jacks Creek, one of the larger sub-catchments in the Liverpool Plains suffering dryland salinity. Even in the wet year 1998, only 6% of rainfall was gauged as runoff.

The conceptual groundwater model states that there is limited hydraulic connection from the shallow saline aquifer to the deep gravel aquifer, and little transfer of water between them; see Figure 2. These two water levels are currently hovering between 1.5 and 2 m from the soil surface. During a flood event the shallow water level rapidly comes to the surface, and the amount of available storage in the shrink/swell clays is close to 90% of runoff from the hills. The shallow water level slowly recedes over three months while the deeper aquifer rises and falls on a time scale of several years.

This conclusion provides supporting evidence for the very low runoff measured from these sub-catchments, by creating a mechanism to account for the excess runoff water. When rainfall occurs on these very wet soils, anecdotal evidence suggests that flood up to 0.5 m deep can occur on the lower parts of the sub-catchments.

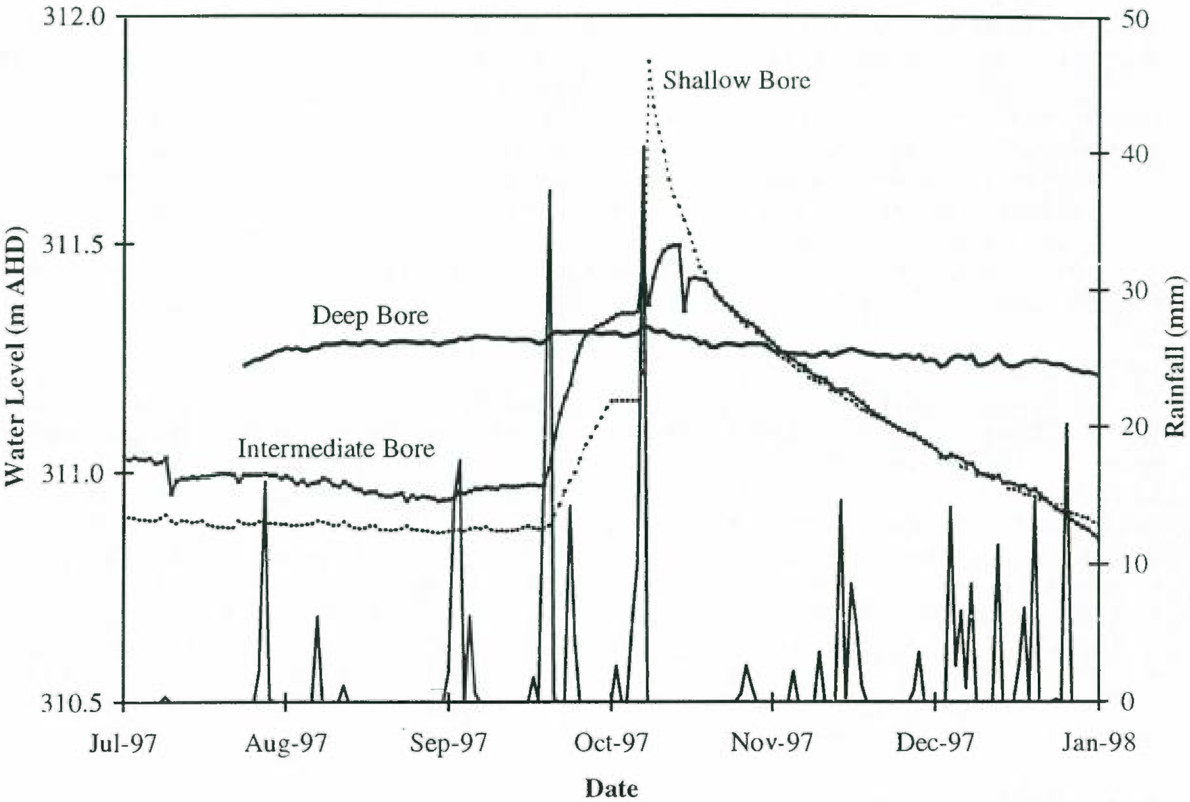

Figure 2 : Shallow and Deep Bore water level responses to a 1997 flood event, illustrating the limited hydraulic connection between the aquifers. Thick line is a deep bore in the gravel aquifer, thin line is within the clay layer, and the dashed line in a shallow bore in the perched aquifer.

With an estimate of groundwater flux and the assumption that negligible vertical recharge occurs, we can use our simple groundwater flow model to invert the current bore levels and gradients to generate the hydraulic conductivity distribution in each sub-catchment. Dawes *et al.* (1999) has performed these analyses and presented distributions of the state properties of the aquifers in the sub-catchments with dryland salinity.

3.3 Evaluating Land Management Options

At what stage could we evaluate the usual management options?

1. From examination of the conceptual model, with no computer modelling performed, it seemed clear that crop rotation manipulation would not be a feature of dryland salinity remediation in these catchments.
2. After some simple groundwater flow modelling, it was confirmed that the large gravel aquifer could only be greatly affected by reafforestation of the uplands parts of the catchment. However the magnitude of recharge component of the runoff from hills and ranges, and the likely change that could be made from total reafforestation from Zhang *et al.* (1997), makes this option not viable from either a hydrological or financial view point.
3. The option of groundwater pumping does warrant further investigation, once more drilling is done for more pump tests to evaluate sustainable water yield, possibly for export to catchments with dropping water levels due to pumping for irrigation. No complex modelling is required for this option.

4. Crop yield modelling associated with recharge estimation can be used in areas where water levels are stable, periodic flooding notwithstanding, to determine an optimised cropping rotation for maximum profit and minimum excess drainage to the shallow saline water table.
5. It is clear from bore hydrographs, stream flow records, and anecdotal evidence that there is a large amount of storage space in the root zone, if the vegetation can empty it, which can be filled and flushed. It would seem that drainage minimisation would be useful at a local level to slow any immediate water level rise, rather than at the catchment level for controlling dryland salinity.

3.4 Hierarchical Approach to Analyses

The whole experience applying simple through to complex models in the ICSM Project has led to the conclusion by Evans and Johnston (1998) that we have "a refinement of our research approaches." The necessary steps to perform investigations of this sort in the future are clear, with an approach along the lines of the TOOLS Project (Anon., 1995) must be adopted. The TOOLS Project advocated performing investigations in a structured hierarchy of effort that applies the simplest approach first to determine if more detailed investigation is required. This means that a research tool, such as a highly parameterised detailed physical model, is not applied synoptically and perceived as a panacea, but used where and when it is appropriate to do so given the analysis provided by simpler methods. Such an approach leads to a set of operational tools and procedures that can either (1) evaluate different land management options directly, or (2) locate where extra data collection or complex modelling is required to evaluate an option. The main problem with research tools and complex models in general, is the time and effort required to process the data to run them, actually using the model, and the interpretation and presentation of results. Where available data and a clear understanding of the whole system can be used to give unambiguous answers in a timely manner, this is always preferable.

4. Conclusions

The ICSM Project sough to use existing knowledge and technology to determine optimal outcomes for farmers in the Liverpool Plains, in terms of maximising profits and minimising adverse environmental effects. Complex water-balance and plant growth modelling was not required to determine the hydrological impacts of any revegetation or land management strategies. The conceptual model combined with desktop and simple groundwater flow modelling was enough to give unambiguous answers for the common land-use change scenarios. Best bet options would not work in these catchments. More complex modelling can be used to answer questions related to the profitability of cropping enterprises independently of the hydrology of the main aquifer. The project was a successful one, not because it ticked off all the project milestones on time, but because it was a solid scientific exercise that promoted understanding ahead of blanket solutions. As a result of this and other projects, and consistent with the direction of the National Dryland Salinity Program, our ability to assess and evaluate land management options has been elevated above a simple collection of models and users, purporting to address individual aspects of the greater land management issues affecting this country.

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AQUIFER INVESTIGATIONS AND PLANNING

GROUNDWATER MANAGEMENT DEVELOPMENTS IN CENTRAL WEST NEW SOUTH WALES.

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Introduction

The Central West Region of New South Wales encompasses the Lachlan, Bogan Macquarie and Castlereagh Rivers and their tributaries. The rivers start in the Great Dividing Range and flow to the West and North West towards the Murray Darling River System. The groundwater aquifers that underlie these valleys are the subject of this paper.

Since the mid 1990's, with access to surface water reaching its limits, water resource development has moved towards groundwater. In 1997 the New South Wales government initiated the water reform agenda which set the framework for groundwater management in New South Wales.

In Central New South Wales the water reforms committed the government to undertake an aquifer risk assessment and develop groundwater management plans in the aquifers at highest risk. This process outlined eight groundwater management areas where aquifers are rated as high risk.

Hydrogeology of Central West New South Wales.

The Central West Region of New South Wales comprises of four main groundwater systems;

1) The fractured rock aquifers of the Lachlan Fold Belt which includes high yielding limestone, basalt and granite aquifers;

2) The alluvial aquifers associated with the current river valleys and paleovalleys of the Cudgegong, Bell, Talbragar, Castlereagh, Macquarie and Lachlan River systems;

3) The Murray Basin associated with the Cainozoic alluvial aquifers downstream of Lake Cargelligo in the valley of the Lower Lachlan River;

3) The Great Artesian Basin including the intake beds within the Coonamble Embayment.

Lachlan Fold Belt

The Lachlan Fold Belt has fractured rock aquifers and supplies only small quantities of water for irrigation, although in some areas useful volumes of water for irrigation are obtained. Bore yields of 1 to 20 litres per second can be obtained in the limestone aquifers of Cudal, Molong and Mudgee and the basalt aquifers surrounding Orange. Within the granite aquifers surrounding Young, bore yields range from 1 to 5 litres per second.

Cainozoic Alluvial aquifers.

Macquarie Valley

The highest yielding irrigation bores are found in the alluvial aquifer west of Narromine where bores can produce 10 to 25 Megalitres/day(Ml/day). A large alluvial fan occurs in this region that fill in a valley incised through a sandstone shale sequence associated with the Great Artesian Basin. Upstream of Narromine the aquifer narrows but bores yielding up to 10 Ml/day can be found from Narromine to Geurie. Smaller alluvial aquifers with a close connection to the Bell, Talbragar and Cudgegong rivers have bores and wells producing 1 to 5 Ml/day.

Castlereagh Valley

The Castlereagh catchment main groundwater aquifer is the Pilliga sandstone of the Great Artesian Basin. In the upper reaches it encompasses part of the intake beds of the Great Artesian Basin. The lower reaches of the Castlereagh valley are in- filled by an unconsolidated sequence of sediments covering the Great Artesian Basin shales and sandstones. A deep alluvial aquifer system occurs downstream of Coonabarabran and extends to the North of Gilgandra. However bore yields are generally less then 10 litres per second and often associated with salinity and iron water quality problems.

Bogan Valley

The alluvial deposits in the catchment of the Bogan River have limited groundwater potential. The aquifers that are known are usually thin, saline and have low yields. In recent times a large aquifer downstream of Dandaloo has been located west of the Bogan River. The water quality of bores drilled to date is generally brackish to saline. The extent and the water quality of this aquifer requires further investigation.

Lachlan Valley

The Cainozoic alluvial aquifers of the Lachlan valley are divided into the Upper Lachlan and Lower Lachlan groundwater systems.

The Upper Lachlan System

Cainozoic alluvial sediments are found upstream of Cowra and fill a paleovalley that follows approximately the course of the present Lachlan River down to Lake Cargelligo where the Murray Basin commences. Williamson W. H (1986) identified two main formations within the paleovalley the Lachlan Formation and the Cowra Formation.

The Lachlan Formation occupies the base of the unconsolidated sediments , sequence and includes gravel and sand aquifers made almost entirely of quartz. The gravels aquifers can be up to 30 metres thick and are found at of 80 to 120 meters. Bores in the Lachlan Formation generally yield from 5 to 20 Ml/day.

The Cowra Formation disconformably overlies the Lachlan Formation. The gravels and sands contain parent material from rock outcrops with some quartz. Where the valley is wide the Cowra Formation can be spread across the whole valley in a braided stream form. Yields from bores in the Cowra formation range from 1 to 8 Ml/day.

Lower Lachlan Groundwater System

The Lower Lachlan aquifer occurs downstream of Lake Cargelligo and evolves into a large alluvial fan to the West of Booligal and Mossgiel, with the town of Hillston, located near the centre of the aquifer. The aquifer is part of the Murray Basin which began to develop about 60 million years ago with the basin's subsidence. The Murray Basin was slowly infilled by large river systems. The alluvial fan deposited by the Lachlan River comprises of three formations, the Renmark Formation deposited between 60 and 12 million years;, the Calivil Formation was deposited between 2 and 6 million years ago; and the Shepparton Formation deposited over the last 2 million years. The last formation consisting of a series of clay and shallow alluvial aquifers (Evans R 1990). The yields of irrigation bores screened into the Calivil and Renmark formations range from 5 to 20 Ml/day.

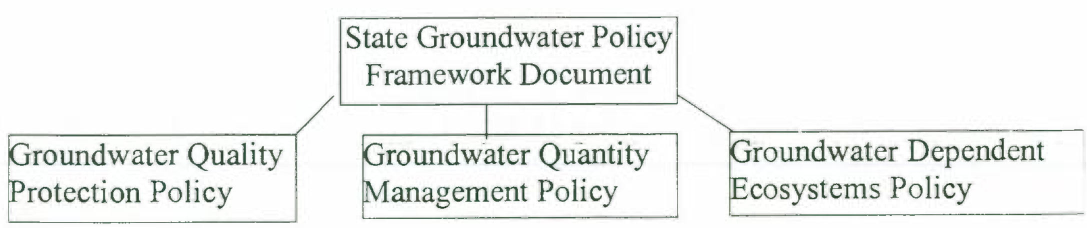
Great Artesian Basin

The Macquarie Valley downstream of Geurie cuts through rocks of the Great Artesian Basin. The Pilliga sandstone outcrops in the Dubbo region and is a major source of recharge to the Great Artesian Basin. The Macquarie, Castlereagh and Bogan Rivers all drain into the Coonamble Embayment of the Surat Basin. Many of the artesian bores south of Coonamble and Quambone have now become sub-artesian due to the drop of pressure levels since the Great Artesian Basin was drilled for agricultural and water supply purposes.

Water Reform Agenda

In 1997 the New South Wales Government released the water reform initiative for both surface and groundwater management. The State Groundwater policy framework document (1997) was released with the aim of setting a basis for the development of policies in the state of New South Wales.

Three sub component polices are proposed



The NSW Groundwater Quality Protection Policy was released in December 1998 and provides the framework of groundwater quality policies required for the development of groundwater management plans. The groundwater quantity management policy and Groundwater Dependent Ecosystems policy are currently being developed.

The Water Reform agenda has set the framework for groundwater management in Central Western New South Wales. This includes setting up two Groundwater Management Committees for the Macquarie and Lachlan respectively. These committees have the role of developing community based management plans for all high risk aquifers within each valley. The process for determining which aquifers are at high risk is outlined in the next section.

Aquifer at Risk Assessment

In 1998 an aquifer risk assessment was undertaken on a statewide basis DLWC (1998). The assessment was qualitative and based on 8 criteria. The central tool used in the assessment is a multi criteria analysis which allows the quantitative evaluation of qualitative data.

The assessment was based on eight criteria:

1. Relationship between licensed water entitlements and the sustainable yield of the aquifer;
2. Local interference caused by pumping;
3. Small or large flow systems;
4. Vulnerability of the aquifer to pollution;
5. Land use threats;
6. Proximity of poor quality water that could be drawn in by over pumping;
7. Water level rise and salinity trends;
8. Dependence of surface ecosystems on groundwater flows;

Each criteria was given a weighting and the Department's regional hydrogeologists assessed every aquifer system within the region. The assessment was undertaken on a state wide basis.

Table 1 illustrates the ranking's for Central West Region aquifers into High Medium and Low Priority. DLWC (1998).

Table 1 Groundwater aquifers level of risk.

High Risk Aquifer	Medium Risk Aquifer	Low Risk Aquifer
Upper Lachlan GWMA 011. Belubula River GWMA 012. Lower Macquarie GWMA 016. Cudgegong Valley GWMA 010 Molong Cudal Limestone. Young Granites. Dubbo (within GWMA 009). Lower Lachlan 012.	Bell River GWMA 020. Orange Basalts 801. GAB Main (601). Upper Macquarie 009. Talbragar-Coolaburragundy. GWMA 019	Casterleagh alluvium. Lachlan Fold Belt metasediments. Upper Tributaries alluvium. Castlereagh Basalts. GAB Shallow. Macquarie Lachlan Granites. Crookwell Basalts.

History of Groundwater Management Plans and Policies in Central West NSW

In 1984 groundwater management areas for aquifers with high use or potential high use where gazetted throughout the state of New South Wales. (J Ross 1990) In the Central West Region a groundwater policy for the alluvial aquifers of the Lachlan and Macquarie and their tributaries was released in 1984.

Subsequently individual groundwater policies were developed for the Cudgegong Valley alluvials, DWR (1987a); Bell Valley alluvials, DWR (1987b); Lower Macquarie, DWR (1992); Upper Lachlan, DLWC (1997); Belubula valley alluvials, DWR (1990); and Lower Lachlan, DLWC (1992). In addition, allocation rules were developed for the Young Granites, Orange Basalts, the Talbragar alluvials but no groundwater policy has been written for these groundwater management areas at this stage.

Development of Groundwater Management Plans

With the current resources the following groundwater management plans are being developed by the Groundwater Management committees.

Lower Macquarie

The Lower Macquarie Groundwater Management plan is being developed by the Macquarie Groundwater Management Committee. Keshwan and O'Shaughnessy (1998) have produced a status report for aquifers which divides the lower Macquarie area into four management zones.

The status report illustrates the current recharge estimates and usage for each of the four zones, as shown in Table 2.

Table 2 Summary of Recharge estimate, allocation and usage for Lower Macquarie aquifer zones.

Zone	Average Annual Recharge (Ml)	Base allocation	Current usage
8A	25,500	27,909	10,000
8B	26,600	35,153	11,000
8C	10,000	24,012	2,500
8D	7,000	18,789	6,500

To encourage community involvement in the Lower Macquarie Groundwater Management plan an Issues and Option paper has been released for community discussion. The main issues discussed by the paper are the over allocation of the aquifer and development of rules for new entrants. The Lower Macquarie Groundwater Management Plan is due to be completed in the year 2000.

Zone 8D Community Groundwater Management Plan.

Zone 8D is part of the Lower Macquarie Groundwater Management area where irrigation from GAB Sandstone aquifers have identified yield of up to 16 Ml/day in some bores. These high yields within a confined aquifer has caused significant interference effects of the bores up to 16 kilometres away. In the summer of 1997 a number of stock and domestic bores went dry.

A community groundwater management plan was developed by the local community with the assistance of the Department of Land and Water Conservation. The plan involved irrigators agreeing not to pump below predetermined water levels within agreed monitoring bores. Also a scheme to lower pumps and where required, deepen bores affected by pumping was developed.

The plan performed well in 1998/99 and is currently undergoing revision for implementation for the 1999/2000 irrigation season.

Cudgegong alluvial aquifer

The Cudgegong alluvial aquifer was embargoed in December 1998. A groundwater status report was released. Table 3 presents recharge estimates and current groundwater usage.

Table 3 Summary of Recharge yield estimate, allocation and usage for Cudgegong alluvial zones.

Zone	Average Annual Recharge (Ml)	Base allocation	Current average usage
1	7,000	11,546	3,000
2	800	2,619	300
3	7,800	438	na

The Cudgegong alluvial aquifer groundwater usage is about 20 to 25 % of its total allocation and about 40 % of estimated average annual recharge (Hamilton S 1998). The groundwater management plan will consider how to deal with over allocation and rules to allow incorporation of new entrants into the groundwater system.

Lower Lachlan Groundwater Management Plan

Due to the large number of applications for groundwater licenses an embargo of new licenses was gazetted in October 1998.

A groundwater status report is near completion for the Lower Lachlan aquifer. A management plan is due to be finished by the end of 2000 under the guidance of the Lachlan Valley Groundwater Management Committee.

The status report demonstrates that groundwater allocations have exceeded the sustainable yield.

Table 4 Summary of Recharge estimate, allocation and usage for Lower Lachlan aquifer.

Zone	Average Annual Recharge (Ml)	Base allocation	Current average usage
1	150,000	157,330	40,000

Groundwater management issues in Central West.

The rapid level of development in the Central West Region has led to a significant increase in areas which are being embargoed for receiving applications for a groundwater license. Despite the embargoes the demand for water is still high.

Issues which are being dealt with over the next 12 months are:

1) Developing interim transfer rules for some of the low usage groundwater management areas. The demand for transfers is increasing to allow new entrants into the groundwater system whilst ensuring the sustainable yield limit is not exceeded;
2) Converting conjunctive allocations to a base groundwater allocation. The conjunctive licenses are where surface water irrigators who have a bore and allowed to

make up shortfalls from the surface water supply during dry periods with groundwater. A key aim of converting the conjunctive license to base allocation is to ensure all groundwater license holders are treated equitably when managing the groundwater aquifer in the future.

3) Developing a system of carryover for groundwater. This is to encourage more efficient use of groundwater so that landholders will extract it when required;

4) Developing a groundwater policy for mining interests which is in harmony with agricultural interests. In the Central West Region there is a large potential for mining to expand and currently the groundwater policies do not cater for this expected demand for groundwater;

5) Improving the understanding of recharge and sustainable yield. Groundwater management plans are using recharge to determine sustainable yield. Currently our knowledge of recharge for most aquifers in the Central West is not well understood. Considerable expansion in groundwater monitoring is proposed to improve future estimates of recharge;

6) Improving the understanding of water quality within groundwater systems. Groundwater quality is becoming more of an issue as groundwater development from industry and groundwater irrigation expands. An expansion in groundwater quality monitoring is proposed, and;

7) Improving the understanding of groundwater dependent ecosystems. Groundwater Dependent ecosystems can be wetland, springs and creeks which are fed by groundwater systems.

The majority of these issues will be dealt with at the individual groundwater aquifer basis as the groundwater management plan is being developed. A few issues will be dealt with on a regional basis.

Conclusion

Groundwater management in the Central West of New South Wales has undertaken a rapid level of development in recent years. Currently three groundwater management plans are being developed the Lower Macquarie, Lower Lachlan and Cudgegong alluvial groundwater management plans. These plans are due to be completed in the year 2000.

Acknowledgement

Gabriel Salas and Martin O'Rouke provided valuable comments to the draft of this paper.

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A Regional Sub-artesian Groundwater Investigation in a Cainozoic Sedimentary Aquifer System in South West Queensland

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Introduction

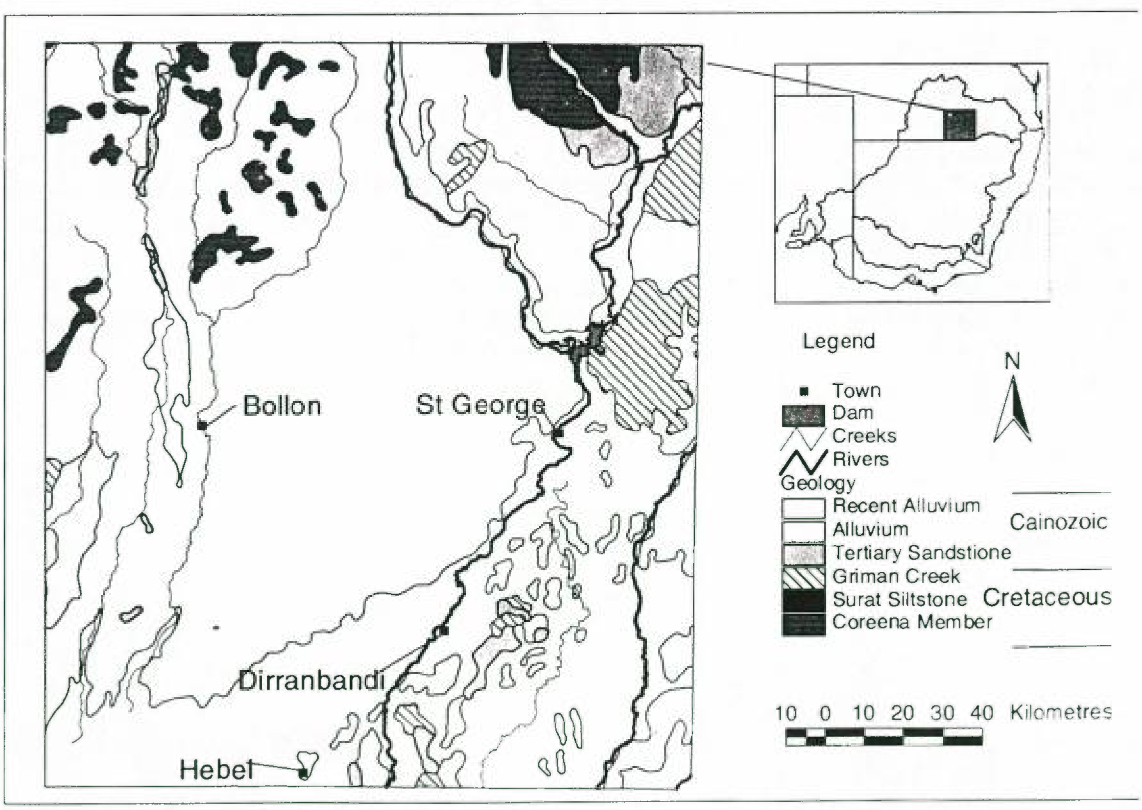
Demand for irrigation water in the St. George region, which forms part of the northern Murray-Darling Basin, has led to the exploration and development of a sub-artesian groundwater resource. Traditional demand for irrigation water in the region, has in the past been met by dam storage or river flood harvesting.

Increased development for irrigation of this sub-artesian groundwater resource in the region by landholders has necessitated a detailed groundwater investigation by the Department of Natural Resources (DNR) Queensland. This is a declared groundwater area, whereby exploration, development and extraction of groundwater require licensing by the DNR. Annual allocation levels for groundwater extraction are set by DNR to maintain sustainability in the system. Results from the hydrogeological study were used in the development of a groundwater flow model and form the basis of a subsequent groundwater management plan.

The study area is part of the Balonne-Condamine River Catchment and overlies the Jurassic to Cainozoic Surat Basin. The Balonne and Maranoa Rivers are the most prominent surface water systems in this region. The entire area drains into the Darling River system (Galloway *et al* 1974). One major dam (E.J. Beardmore Dam) and many weirs exist within the study area. Relief slopes from 340m (AHD) in the north and north-east to 130m (AHD) in the south-west. Climate of the study area is semi-arid, with median annual precipitation ranging from 550mm in the north-east to 350mm in the south-west. Annual pan evaporation ranges from 2000mm in the east to 2200mm in the west. Local town centres are St. George, Dirranbandi, Bollon and Hebel.

Cotton is the dominant broad scale crop produced in the area, with smaller areas of various cereal crops supplementing cotton production. A large cotton flood irrigation area exists to the south of St. George, which is supplied with water from the nearby Beardmore Dam. Local farmers outside this irrigation area grow dry land style cotton and cereal products. Cattle and sheep grazing are the major alternative industries for the farming community.

Figure 1. Location of the study area in the Murray-Darling basin (inset), regional surface geology, drainage, dams and town centres.



Background

Previous research in the region is related to sedimentology and petroleum exploration. Mond and Senior (1970), completed a shallow stratigraphic drilling investigation in the area. Senior (1970), expanded on this drilling information with water and petroleum exploration bore data to produce a basement contour map of Cainozoic sediments. This has proved useful in the planning and drilling phases of the operation. Other previous research is limited to reconnaissance seismic work, which focuses primarily on petroleum exploration and basement (underlying Permian to Triassic basins) definition. The use of reflection seismic data and interpretations has outlined faulting and down warping in underlying Great Artesian Basin sediments which has implications to the development of the trough containing Cainozoic sedimentary deposits in the study area.

There are 421 artesian and 224 sub-artesian bores within the study area registered on the DNR database. The stratigraphic drilling logs of 263 of these bores were used to supplement data produced from this groundwater investigation. Exploration and development during the previous eight years has produced a large number of drill holes, monitoring bores and production bores not registered on the DNR database. The data from 71 drill logs, 34 monitoring bores and 9 production bores has been incorporated into the groundwater conceptualisation and model. Production bores (ranging from 270mm – 400mm diameters) have recorded yields of up to 115 l/sec for sustained pumping periods.

Field Investigations

A drilling program was designed to provide stratigraphic lithological data, down-hole geophysical data and allow the construction of monitoring bores in relevant hydrogeological units. Drilling was completed with a Mahyew 1000 rotary drill rig. Drill site locations are plotted in Figure 2.

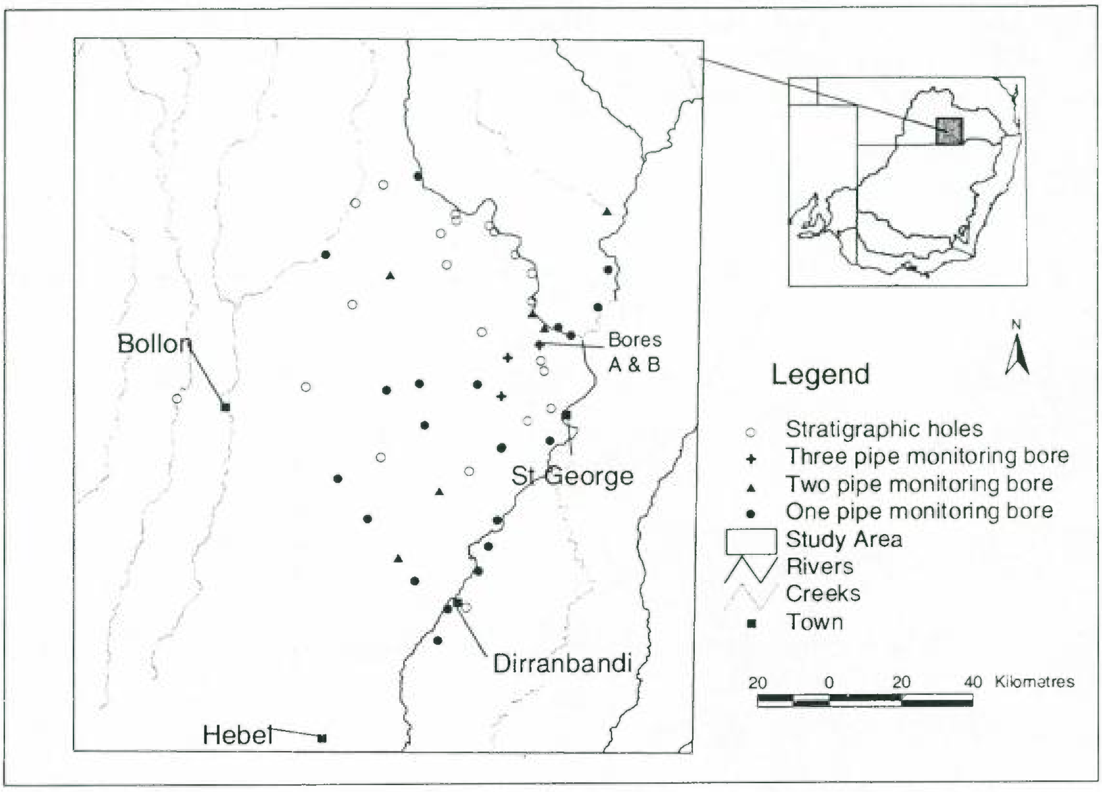
Down-hole geophysical gamma-ray, SP/resistivity and calliper logs were utilized in all open drill holes for stratigraphic interpretation and correlation. All monitoring bores were constructed in relevant water bearing layers with 50mm diameter PVC. Multiple bores were constructed in single locations to monitor the multi layer system. Water bearing layers were sealed with bentonite plugs to ensure accurate sampling. Bores were developed with compressed air and regular water level measurements were collected. The location of the constructed monitoring bore network can be seen in Figure 2. All bores and stratigraphic holes were surveyed with a single phase differential GPS for sub-meter accuracy (± 0.3m) in the x, y and z directions.

Table 1. A summary of drilling operations.

Table 1- Drilling and monitoring bore construction summary			
No of Stratigraphic Holes	68	No of 50mm Monitoring Bores Constructed	51
Total Depth of Stratigraphic Drilling (m)	4348	Total Depth of Casing Used (m)	2933
Average Hole Depth (m)	63.9	Average Monitoring Bore Depth (m)	57.5
Drill hole depth range (m)	4 – 166.8	Monitoring bore depth range (m)	2 – 130

Water samples were taken from all DNR monitoring bores, selected private monitoring bores and surface water locations. Major ion, soluble metals and insoluble metals sampling was carried out on 42 samples, δD and $\delta^{18}O$ sampling was carried out on 34 samples, $\delta^{13}C$ sampling was carried out on 31 samples and ^{14}C sampling was carried out on 13 samples.

Figure 2. Position of stratigraphic drilling locations and monitoring bores.



Geology

Geology of the study area comprise consolidated and unconsolidated Tertiary and Quaternary (Cainozoic) terrestrial sediments which overlie an effective basement of Cretaceous marine to terrestrial sediments. Cainozoic sediments were deposited in an erosional trough (termed the Dirranbandi Trough) developed from the faulting and down warping of underlying Mesozoic sediments. Reconnaissance seismic data supports this down warping and faulting theory (Davies and Lodwick, 1962). The Dirranbandi trough is a north north-east to south south-west trending system with a maximum depth of 220m below ground surface (structure contours of basement, Figure 3). Regional geology of the area can be seen in Figure 1. Sedimentology of the Dirranbandi Trough can be simplified into four major units (Unit A – D) outlined below from effective basement to the surface.

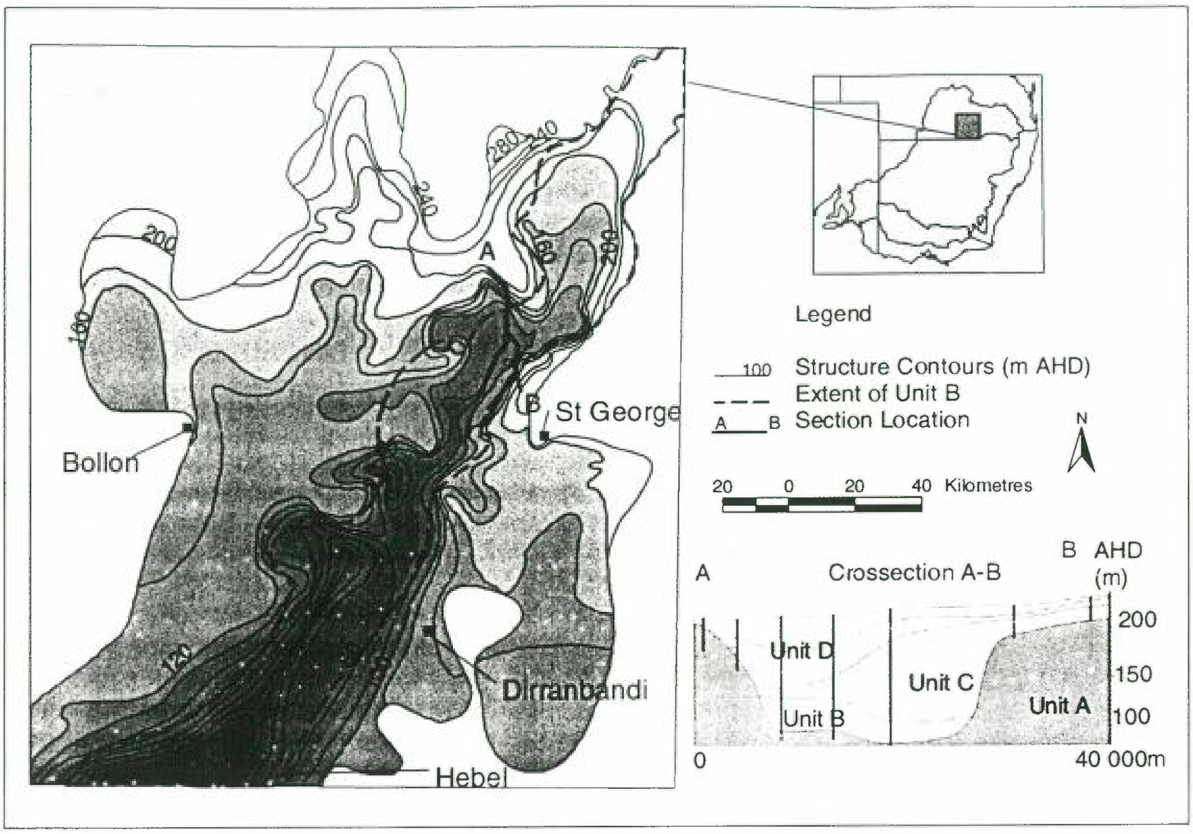
Unit A: The effective basement to the system (Unit A), composed of cretaceous marine to terrestrial sediments. Marine sediments consist of shale, siltstone and minor sandstone. These sediments have a blue grey or green grey colour related to the mineral Glauconite and is termed the Surat Siltstone (Exon, 1976). Exon (1976) suggests that a regression of the Cretaceous Sea, that deposited the Surat Siltstone, formed the overlying terrestrial sequence of grey mudstone, red siltstone, sandstone and conglomerate known as the Griman Creek. Griman Creek Formation sediments form the effective basement on the edges (0-60m) of the trough and have been eroded from the central region of the trough, leaving the Surat Siltstone as the effect basement. The Surat Silstone, Griman Creek and underlying Coreena Member are exposed around the edges of the basin (Figure 1).

Unit B: The first depositional unit (Unit B) in the deeper (60 – 220m) regions of the trough is a series of stacked fine to coarse sand beds with occasional gravel layers. Individual sand beds range in thickness from 1 to 10m, these are interbedded with grey clay and silty clay with occasional autochthonous and allochthonous coal layers to form a unit of up to 30m thick. Sand grains are angular and consist of quartz with minor lithic grains. Gravel layers generally have the same lithological distribution as the sand deposits but are rounded and sub-spheroidal. Geological and geophysical interpretation of the drill logs indicate a possible braided channel depositional environment. The extent of Unit B can be seen in Figure 3.

Unit C: Unit B is overlain by a sequence of grey clay, grey mudstone, red silty clay and red silstone (Unit C). This unit shows a reduction in the energy of the depositional environment from that of underlying Unit B.

Unit D: The remainder of the sedimentary sequence comprise thin (0.5 – 4m), fine to very coarse sand beds. Occasional gravel layers can be found at the base of a number of sand beds. These sand beds are interbedded with layers of silt, silty clay and clay with minor siltstone and mudstone. Geological and geophysical interpretation of the drill logs support a possible meandering river and flood plain depositional environment.

Figure 3. Structure contour map of basement of the Dirranbandi Trough and cross-section A-B.



Hydrogeology

The units described in the geology section have been broken into hydrogeological units for the development of a conceptual model of the aquifer system.

Unit A: The basement of sub-artesian water supplies for irrigation. Water supplies in this unit are attainable but usually have low yield and are of poor quality.

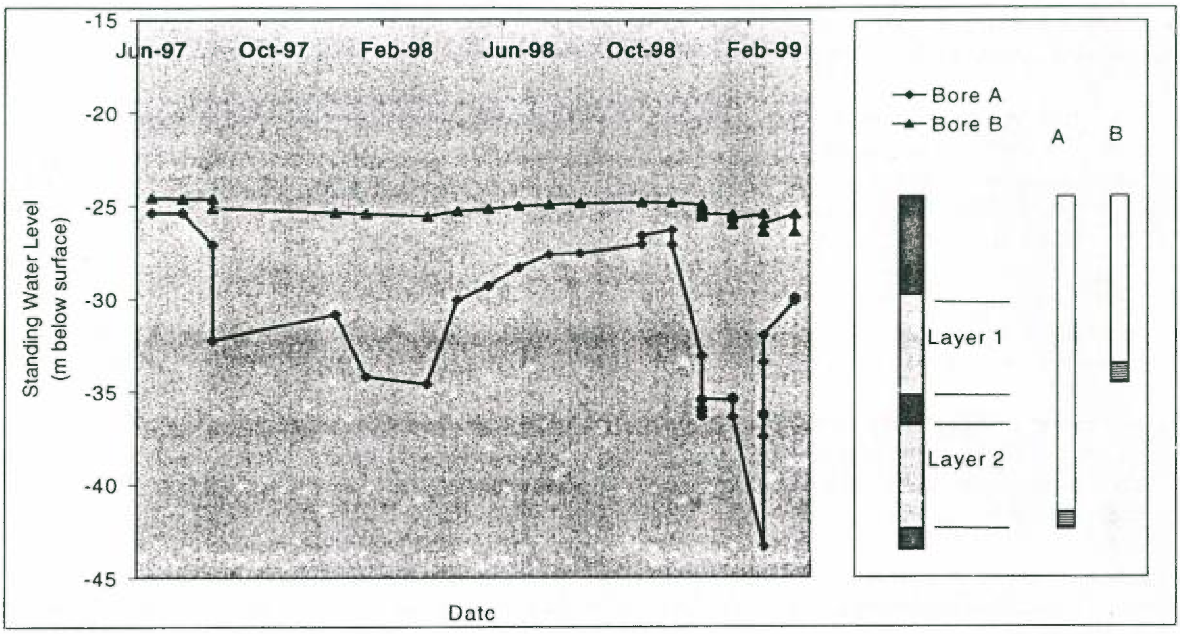
Unit B: Stacked sand sequences form a confined aquifer system, which has been termed Layer 2 in the hydrogeological conceptualisation. Several clay lenses within Layer 2 do not seem to affect the hydraulic connection between various sand units. Standing water levels drawdown with water extraction and recover after the cessation of extraction. Water level responses of a bore (A) screened in the lower sections of Layer 2 can be seen in Figure 4. Decreases in water levels are a result of two water extraction sessions (100 days each from Layer 2), followed by subsequent water level recovery (Figure 4). Standing water levels in the long term would be thought to diminish over time, with extraction being greater than recharge. High yield from this aquifer unit delineates this as the main target for irrigation water supply.

Unit C: This layer of silt and clay forms a leaky aquitard between Layer 2 and Layer 1.

Unit D: Sand beds form a series of unconfined and semi-confined aquifers. For the hydrogeological conceptualisation process these aquifers have been combined and termed Layer 1. Layer 1 has a lower yield than Layer 2 and groundwater extraction and exploration is generally for stock and domestic supply. Standing water level responses in Layer 1 show minor fluctuations through the year, possibly due to recharge. Layer 1 has a leaky connection to the underlying Layer 2. When water extraction occurs from Layer 2 standing water levels in Layer 1 drop about 10% of those in Layer 2. Bore B screened in Layer 1 (Figure 4) shows the water level response in Layer 1 at the same location as bore A over the same period when water extraction is occurring from Layer 2.

Recharge to the system is thought to be from downward percolation of surface precipitation and transmission from the base of rivers and creeks into Layer 1. Recharge to Layer 2 is a result of slow leakage from Layer 1 and direct infiltration from the north-eastern section of the Balonne River.

Figure 4. Standing water levels responses in bore A (Layer 2) and bore B (Layer 1). Location of bores A and B marked on Figure 2.



Groundwater Geochemistry

Groundwater quality demonstrates variability in the concentration and composition throughout the system. This variability is thought to be a combination of different recharge pathways, groundwater residence time and recharge from different sources and events. Total dissolved solids (TDS) in Layer 1 range from 390 – 28 900 mg/l with low (<1500 mg/l) TDS values found along the major rivers. Higher (>1500 mg/l) TDS values trend away from the major rivers and point locations where minimal recharge occurs. TDS in Layer 2 ranges from 420 – 13 000 mg/l but is generally below 5000 mg/l. The presence of low permeability zones in Layer 2 allows the partial separation of groundwater contributing to increases in TDS concentration with depth in this layer. The TDS concentration in Layer 2 is less dependent on lateral proximity to major rivers as Layer 1. The pH of both layers is between 6.1 and 7.5.

The dominant major ions in the system are Na and Cl and to lesser extent HCO_3 and SO_4. The water type of Layer 1 in the low TDS zones near the main rivers is Na-Cl-HCO_3 and Na-Mg-HCO_3, and Na-Cl in the central part of the basin. Two samples of Na-Ca-Mg-Cl-HCO_3 and Na-Ca-Cl-HCO_3 water type were also found close to the rivers. Layer 2 is classified as a Na-Cl type as a result of enrichment of Na and Cl possibly from the underlying marine sediments.

The radioisotope ^{14}C was selected to delineate an approximate age of groundwater, flow direction and identification of different aquifers. Analysis assumes that there is no interaction between soil CO_2 and the aquifer rock matrix. Groundwater ages in Layer 1 range from 860 years along the Maranoa River to 13 800 years in the central basin. The groundwater age in Layer 2 ranges from 3830 years along the Balonne River to over 30 000 years.

Interpretation from the groundwater geochemistry data supports the theory of a two layered system with salinity and age of groundwater increasing with depth from surface. The low TDS and relative age of groundwater in Layer 1 near the major rivers indicates this as a possible source of recharge.

Groundwater Model

A groundwater flow model of the system is being completed using MODFLOW. The PMWIN graphical interface package was used to construct the model and input relevant data. Model construction is complete, however steady state calibration is in progress. At this stage no results are available on aquifer parameters, storage or yield. The model will calculate total storage within the aquifer and allow for various simulated pumping scenarios to show the effect on standing water levels. Results from the model will be used to calculate a safe yield for the system. As this is only a newly developed and monitored system, there will be a number of limitations from the initial model results. As development continues over the next five years more data will become available on the system allowing recalibration of the initial model.

Model grid: A two-layer model was designed from the hydrogeological conceptualisation to simulate flow in the two distinct aquifer units (Layers 1 and 2). The thickness of the aquitard separating Layer 1 and 2 was specified and a leakage was incorporated into the model between the two layers. A variable cell size was used over the model grid, with a 1km^2 cells used where there was more stratigraphic and observation data, to a maximum cell size of 10km^2. There are 3346 active cells in Layer 1 and 1816 active cells in Layer 2.

Model Boundary: Fixed head boundaries were created at the northern and southern extremities of the model grid where flow in and out of the system occurs. An outcrop boundary is present around the remainder of the model grid. which was set as a no flow boundary.

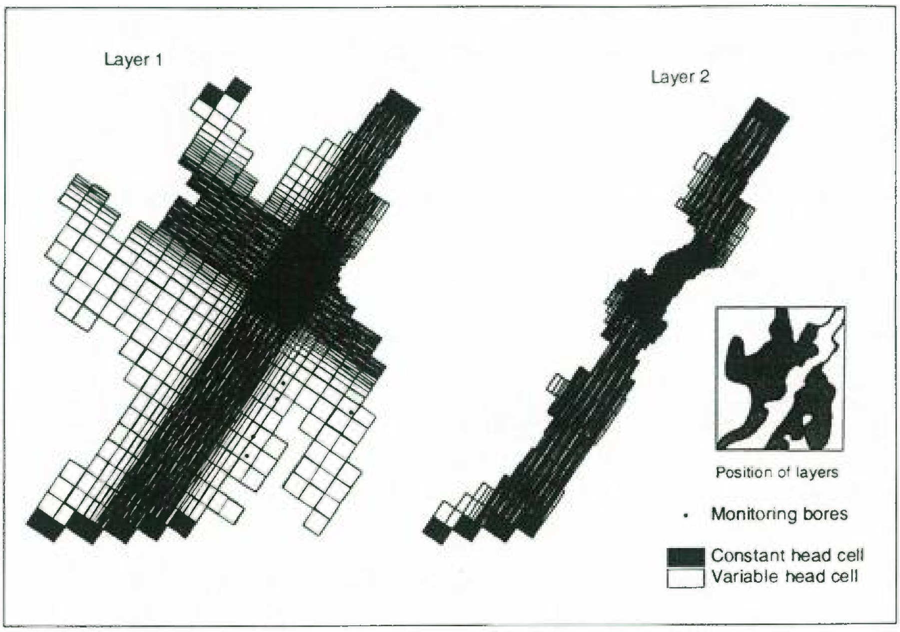
Observation and discharge bores: Water level information in the two layers was provided by 92 (46 in each layer) DNR and privately owned monitoring bores. Standing water level information has been gathered since 1997. Water extraction figures were supplied for nine irrigation bores. Discharge data is available for two irrigation seasons of 100 days each over two years.

Recharge: Recharge was calculated from a combination of direct rainfall infiltration and leakage from the base of rivers. The aerial zonation of different lithologies between the top of Layer 1, and the surface soil type was rated on its recharge potential. A percentage of rainfall over each recharge zone was entered into the model and is being calibrated. The next stage of recharge estimation, if needed, will involve the use of the SLASH package to estimate recharge potential of the nominated soil type, crop/vegetation type, rainfall and evaporation. The use of SPLASH combined with the RIVER package that estimates recharge from the base of the effluent river will give an enhanced estimation of recharge to the aquifer system. Recharge to the underlying Layer 2 is at present being calibrated incorporating leakage from layer 1.

Hydraulic Conductivity: Hydraulic conductivity was estimated for both horizontal and vertical components from layer lithology, pumping test analysis and local knowledge of potential aquifer yields. This was zoned into areas of varied hydraulic conductivity and is at present being calibrated.

Initial Heads: Initial starting heads in observation bores were taken from a period in 1997 when little development of the resource had taken place and therefore representative of a static natural system.

Figure 5. Model grid and observation bores in Layer 1 and 2



Conclusions

A multi layer sub-artesian aquifer system has been investigated and delineated within the northern Murray-Darling Basin to a depth of 220m below ground surface. Exploration and development of this resource for irrigation is continuing by landholders. Groundwater age ranges from 860 to >30 000 years. Groundwater quality is variable from 390 – 28 900 mg/l TDS. Recharge to the system is believed to be a combination of rainfall and river leakage. Proposed extraction levels would be thought to exceed recharge, therefore dropping standing water levels in the system will be expected in the future. A hydrogeological conceptualisation of the system has been completed and a groundwater flow model is in its final stages of calibration. The findings of the study and model will be implemented into an allocation policy for the aquifer system and will form the basis for a future groundwater management plan.

Acknowledgments

Financial support for the project was supplied by the Regional Infrastructure and Development section of DNR. Assistance from the staff of the Water Assessment and Planing work group of DNR. Mr Damir Horvat and the late Mr Phil Macnamarra DNR St George for field assistance and advice. Staff on the DNR rotary drill rig. Mr John Harris of Clearmagic Pty Ltd for the supply of drilling logs and advice.

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Hydrology and Sustainable Yield of the Mid Murrumbidgee Alluvial Aquifer

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Introduction

The Murrumbidgee Catchment covers an area of 81,400 km^2, and is located in southern NSW (figure1). The Mid Murrumbidgee Alluvial aquifer has an area of about 1,145 km^2, and extends from Gundagai in the east across to Narrandera in the west. The aquifer ranges in width from about 1.5 km wide at Gundagai, to about 20 km wide just upstream of Narrandera. This aquifer has been defined as the Mid Murrumbidgee Groundwater Management Area (GWMA 013).

Groundwater exploration was carried out in the 1960's, and boreholes drilled for this work have subsequently formed the monitoring network for this aquifer. There are about 100 monitoring bores in the Mid Murrumbidgee GWMA, and water levels from these bores are monitored on a regular basis. Preliminary studies of groundwater levels in the Mid Murrumbidgee indicate that some sections of the system are very closely linked with flows in the Murrumbidgee River.

Groundwater allocation in the Mid Murrumbidgee is based on sustainable yield estimates for this particular aquifer system. The estimated annual sustainable yield for the Mid Murrumbidgee Groundwater Management Area is 89,000 ML. Currently the total annual entitlement for this area is 50,799 ML.

Groundwater Hydrology

Geology

Extensive groundwater investigations for the Mid Murrumbidgee were carried out in the late 1960's, and this work was published as a Masters Thesis in 1972 by Mr. Don Wooley. This work incorporates results from the exploration drilling, seismic work, and groundwater chemistry sampling.

The sediments of the Mid Murrumbidgee can be divided into two major formations based on their age and depositional environment. The upper aquifer is referred to as the Cowra Formation, and consists of Quaternary aged sediments. The deeper sediments are Tertiary aged, and are referred to as the Lachlan Formation.

The Cowra Formation extends from ground surface down to varying depths. The formation is thinnest at Gundagai where it extends to a depth of about 15 metres, and thickest at Narrandera where it extends down to a depth of about 40 metres.

Water quality in the Cowra Formation is generally more saline than the Lachlan formation, with salinity also increasing with distance from the Murrumbidgee River. The salinity between Wagga Wagga and Gundagai is typically less than 1000 mg/L. However, between Wagga Wagga and Narrandera the salinity may be much higher (in excess of 10,000 mg/L), especially away from the river. The concentration of iron in this aquifer is also much higher than the Lachlan Formation.

Bores screened in the Cowra Formation are typically low yielding, and have a maximum yield of about 13 L/s at Wagga Wagga (Wooley 1972). Groundwater from the Cowra Formation is generally used for stock and domestic supplies due to the poor water quality, and low yields.

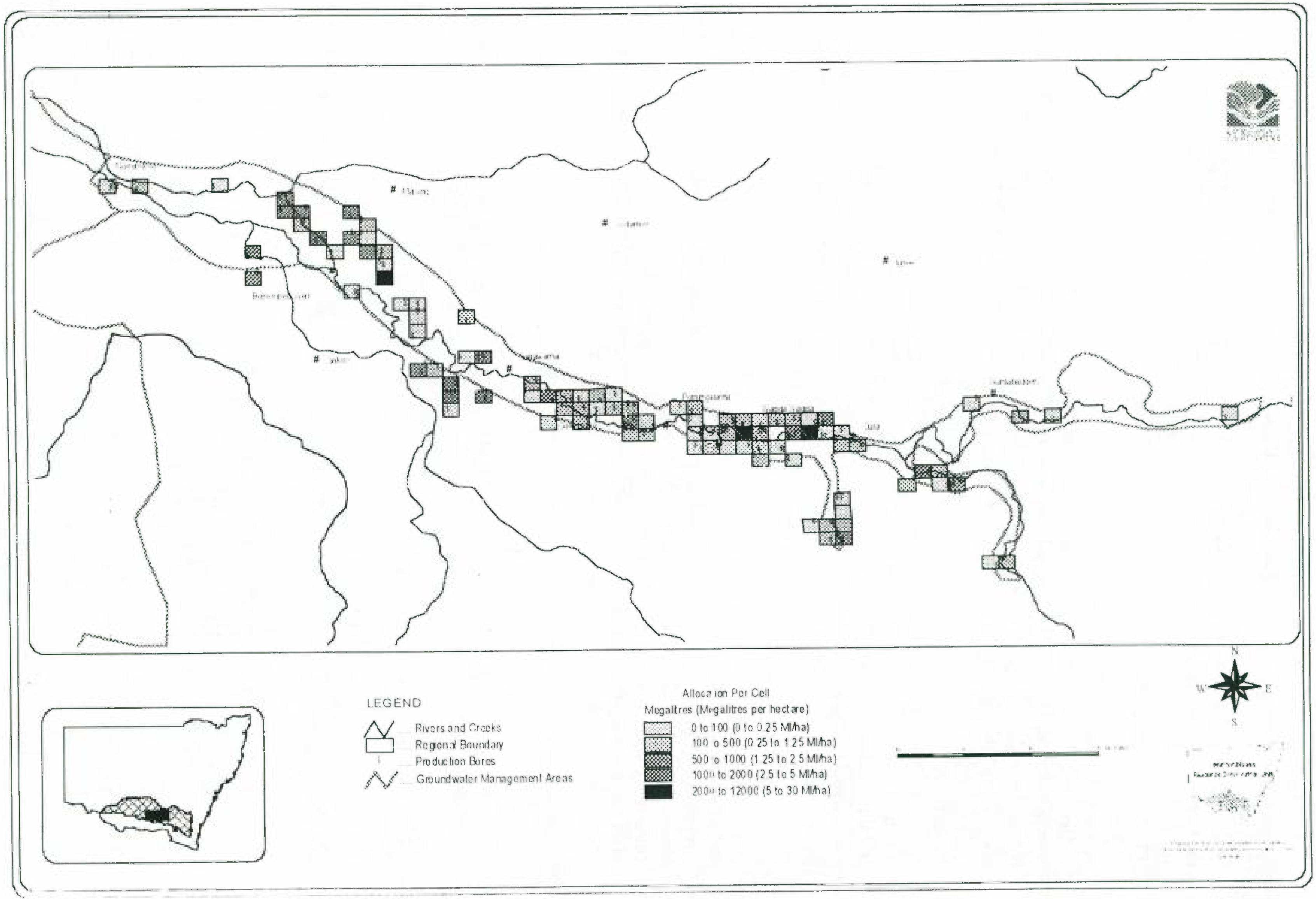

Figure 1. Allocation Distribution for the Mid Murrumbidgee Groundwater Management Area

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Sediments from the Lachlan Formation are commonly clean grey quartz sands and gravels, with intermittent layers of grey clay. The thickness of this layer varies from about 9 metres at Gundagai (15 to 24 metres below the surface), to about 125 metres at Narrandera (37 to 162 metres below the surface). The water quality of this formation is suitable for irrigation and town water supplies, with salinity ranging up to a maximum of about 500 mg/L. However, iron and manganese levels in this formation are quite variable, and values of up to 1.5 mg/L of iron have been measured.

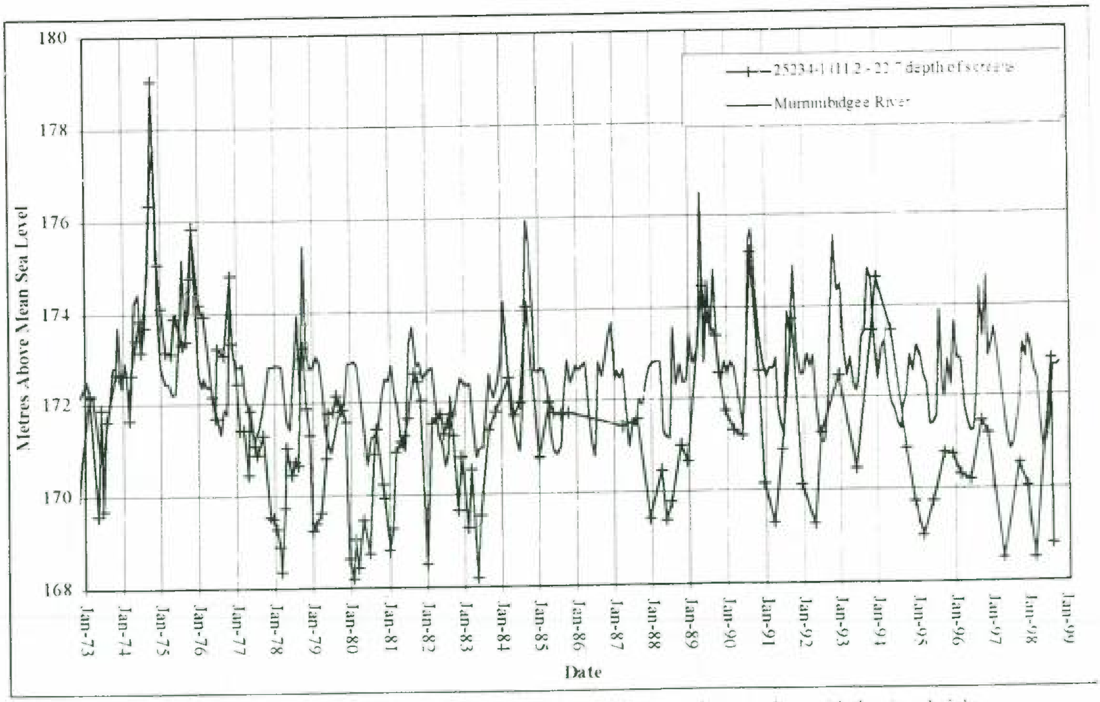
Yields from this aquifer vary depending upon the thickness and quality of sand and gravel layers intercepted. Yields range from about 5 L/s at Gundagai to about 150 L/s near Narrandera (Wooley 1972). The large yields seen in the downstream section of the aquifer are most likely a factor of the increasing thickness and transmissivity of Tertiary sediments.

Groundwater Level Trends

Groundwater levels for observation bores in the Mid Murrumbidgee have been monitored regularly for a period of about 30 years. Hydrographs of these bores show definite seasonal trends, however these trends vary depending upon the location of the bore in the system. The hydraulic connection between the Tertiary and Quaternary also varies depending upon the location of the bore in the system.

In the upper reaches of this aquifer system (upstream of Oura) the total depth of sediments is relatively shallow (40 to 70 metres), and the sediments are generally poorly sorted. Previous reports stated that rainfall was a major recharge source for the upper reaches of the main aquifer, and the tributary sections (Wooley 1972). Hydrograph trends agree with these initial observations for the unregulated tributaries as water level fluctuations seem to follow a pattern of highest groundwater levels during high stream flow, hence high rainfall months. In these upstream locations there also seems to be some connection between the Tertiary and Quaternary aquifers.

Much of the initial aquifer testing was conducted near Wagga Wagga, and bores at this location do not seem to have a close connection between the Tertiary and Quaternary systems. There is commonly a 5 to 10 metre layer of clay dividing the Tertiary and Quaternary aquifers, which retards downward leakage. However, hydrographs do indicate some minor hydraulic connection between these two layers. Hydrograph fluctuations of bores near Wagga Wagga have a close correlation with river heights (graph 1), and as the Murrumbidgee River is highly regulated aquifer recharge is most likely directly from the river, and not dependent upon rainfall.



Graph 1. Groundwater Hydrograph for an observation bore near Wagga Wagga, and comparison with the river height.

Hydrographs of bores at Narrandera indicate minimal, if any connection between the Tertiary and Quaternary aquifers. Water level fluctuations of bores screened in the Lachlan Formation do not correspond with water level fluctuations of bores screened in the Cowra Formation. Comparison of groundwater hydrographs with river heights at Narrandera indicates little correlation.

Sustainable Yield

The annual recharge to the Mid Murrumbidgee GWMA is estimated to be about 127,000 ML. Sustainable yield of the system is taken to be 70% of this total figure, and is therefore estimated to be 89,000 ML. This correlates to an average allocation of about 0.7 ML/hectare over the entire area of the system. (Ross 1998)

Groundwater Allocation Policy

The policy for allocating Groundwater in the Mid Murrumbidgee GWMA is based on total property area overlying the aquifer, and the relevant zone. The zones are based on groundwater availability, and are:

- Zone One — Upstream of Oura, and Tributaries
- Zone Two — Oura to Pomingalarna
- Zone Three — Pomingalarna to Narrandera

Total Property Allocation = 2 ML/ha x first 50 hectares	
Plus	
Zone one	0.5 ML/ha x remaining hectares
Zone two	1 ML/ha x next 400 hectares, plus 0.5 ML/ha x remaining hectares
Zone three	1.5 ML/ha x next 400 hectares, plus 0.5 ML/ha x remaining hectares

Current Status of Groundwater Allocation

Groundwater licenses for the Mid Murrumbidgee have been mapped and their entitlements have been cumulated on a 2 km grid (figure 1). Mapping entitlements in this manner gives an overall picture of entitlement hot spots within the system. This mapping technique was previously used to map entitlements in the Lower Murrumbidgee GWMA (Lawson 1994).

The allocated volume of groundwater from this system is currently 50,799 ML, and 60% of this total allocation is for irrigation purposes (Table 1). The dominant irrigation practise in this area is spray irrigation of lucerne or other pastures. Entitlement for town water supplies total 37% of the overall entitlement for the Mid Murrumbidgee. These town water supply bores are located at Gumly, Wagga, Collingullie, and Mateng.

At present there are no current figures for level of groundwater use in this groundwater management area, however usage figures will be collected from 1999 onwards. Based upon the number of constructed works in this system it is possible to estimate a maximum possible usage figure of 30,589 ML, which is 60% of the total entitlement. This figure assumes that constructed bores are using their total entitlement, therefore this figure is an overestimate.

Purpose	Number of Licenses	Number of Constructed Bores	Total Allocation Megalitres	Total Possible Usage (based on constructed bores)
Irrigation	119	45	30,539	10,677
Town Water Supplies	27	22	18,910	18,910
Industrial	10	6	745	712
Recreation	14	9	605	290
Total	**170**	**82**	**50,799**	**30,589**

Table 1. Groundwater Allocation for the Mid Murrumbidgee GWMA

Conclusion

The geology of the Mid Murrumbidgee alluvium consists of the Quaternary aged Cowra Formation. and the Tertiary aged Lachlan Formation. The Cowra Formation is typically low yielding. and high in salinity. and is generally used for stock and domestic supplies. The underlying Lachlan Formation has some very high yielding sand and gravel layers. and the water quality is generally very low in salinity.

There is an abundance of water level information for the various aquifers in the Mid Murrumbidgee GWMA. Preliminary studies of this information have revealed that the dynamics of the system (i.e. recharge. and leakage between aquifers) is not spatially uniform. As the system travels westward from Gundagai to Narrandera there seems to be less linkage of surface and groundwater systems. and there also seems to be less leakage from the Quaternary to the Tertiary aquifers.

Groundwater in the Mid Murrumbidgee GWMA is allocated with reference to the sustainable yield of the system. The sustainable yield of this system is estimated to be 89.000 ML. and the current entitlement volume is 50 799 ML.

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Groundwater Vulnerability Mapping of the Upper Condamine River Catchment Using the DRASTIC Methodology

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Abstract

Groundwater vulnerability mapping of the Upper Condamine River Catchment has been completed using the DRASTIC methodology. The Upper Condamine River Catchment is located within the headwaters of the Murray Darling Basin. The vulnerability mapping was designed to produce user friendly maps to educate decision makers and planning bodies alike, promoting increasing understanding of the fragility of the resource. Thereby, encouraging the most appropriate location to establish developments to minimise the potential risk of contamination of the underlying groundwater resources.

DRASTIC was modified to deal with the varied aquifer environments in the region. in particular the fractured basalts and possible rapid entry of contaminants by preferential flow along fractures. The map has been validated using available nitrate data.

Introduction

The Upper Condamine River Catchment forms part of the Murray Darling Basin, and is located in the south east corner of Queensland. Groundwater resource and aquifer pollution vulnerability mapping of the catchment has been jointly funded through the Queensland Department of Natural Resources and the Murray-Darling Basin Natural Resource Management Strategy. Location Map is shown in Figure 1.

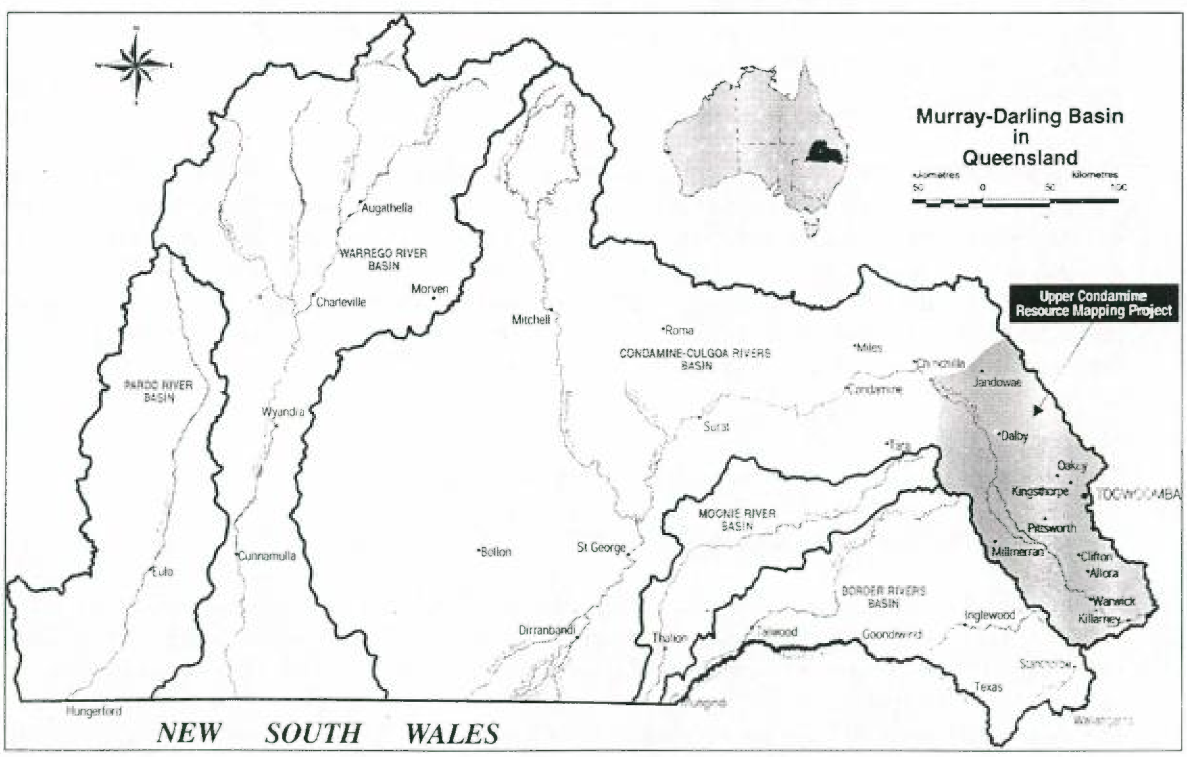

Figure 1. Location Map.

Methodology

Groundwater vulnerability mapping of the Upper Condamine River Catchment (UCRC) has been completed using the DRASTIC methodology.

Weights of Evidence was considered as an alternate methodology but not used because of the complexity of the UCRC groundwater systems, and the premise that electric conductivity (EC) is not a universal indicator of residence time. The statistical approach using EC as a response variable may be more suited to a single formation type assessment and is less relevant and more open to imbalance across a spread of formation types. In the UCRC there are locations where there is a direct hydraulic connection across geological boundaries, but a change in EC corresponding to geology eg. The Bongeen area where over pumping of the alluvial aquifers can draw in more saline water from the sandstone aquifers.

Although DRASTIC has been criticised for its subjectivity (Barber et al., 1995), and for the high intensity of data input (Sinclair et al., 1998), it was considered to be the best available methodology to deal with the complexity of the UCRC groundwater systems. A suggestion also exists that DRASTIC fails to properly address the problems associated with fractured formations. It tends to under-estimate the vulnerability of fractured, compared to unconsolidated, aquifers (Rosen, 1994).

For the UCRC project DRASTIC was modified in an attempt to deal with this. The most significant modification made is the reduction of the Depth to Water weighting from the US EPA recommended 5 to a weighting of 3. The modification increases the relative influence of Impact of the Vadose Zone which balance fractured rock vulnerability relative to the sediments. Water entry into fractured rock formations can be rapid due to preferential flow along fractures. Due to this rapid entry and preferential flow the Depth to Water parameter is considered to have less significance in fractured formations, as recharge water can travel to significant depths in a short time.

Determination of base map layers

Geological information has been obtained from the 1:250000 Geological Series published by the Bureau of Mineral Resources. The source of bore log information has been the Department of Natural Resources (DNR) groundwater database. 15000 bore logs have been sorted, cleaned and reduced to 8000 information sites.

Depth to water

The depth to water table data contained in the DNR database is spread over thirty years. To offset the effect of water level fluctuations and water level drops caused by various aquifer stresses, the number of DRASTIC ranges for depth to water have been reduced to four rather than the seven ranges recommended by the USEPA (Aller *et al*, 1987). This removes the uneven pattern caused by too many ranges and results in a more uniform final vulnerability map. The depth to water data is divided into the ranges 0-5m, 5-10m, 10-20m, and greater than 20m.

Recharge (Net)

The Net Recharge component is a measure of the soil deep drainage, where:

Deep drainage = Leaching fraction x rainfall

The leaching fraction has been calculated by the SALF Program which uses stratified soil profile data including the percentage clay content, cation exchange capacity [CEC] and exchangeable sodium percentage [ESP]. The program is a physically based empirical model relating key soil properties to salt concentration at the bottom of the root zone and thus deep drainage (Shaw , 1988). The soil data used to calculate deep drainage has been taken from Maher (1996a), Maher (1996b), Vandersee and Mullins (1977), Reeve *et al.*, (1960), Vandersee (1975). The rainfall isohyets were compiled using ANUCLIM (Hutchinson et al., 1998) based on data from Clewett et al., (1994).

Aquifer media

The aquifer media information has been drawn from bore log information on the groundwater database. Information from approximately 6000 drill logs were suitable for use. The aquifer media types are Walloon Coal Measures, Marburg Sandstones and Kumbarilla Beds, Basalts, Alluvium, Weathered Granites and Fractured Metamorphics.

Soil media

Soils polygonal data was sourced from Maher *et al.* (1998), Maher (1996), Vandersee and Mullins (1977), Vandersee (1975), Reeve et al., (1960), and Forster (1986). Data associated with these reports, and Biggs (pers comm) provided information necessary to rate soil types in terms of the DRASTIC parameters of, silt and clay content, clay type, soil thickness, quantity of organic matter, permeability and variations of permeability through the soil horizons and soil attenuation ability.

Topography

The AUSLIG 1:250000 topographic data set was used to produce a slope surface used for ranking and rating purposes.

(Impact of)Vadose zone

The vadose zone data has been drawn from bore log information on the groundwater database. The vadose zone types are Walloon Coal Measures, Marburg Sandstones and Kumbarilla Beds, basalt fractured, basalt and clay mix, silty alluvium, clayish alluvium, weathered granites and fractured metamorphics.

Hydraulic Conductivity

Hydraulic conductivity was determined using limited pump test data from the DNR database and a flow net type analysis. The principal used in the flow net analysis is that on a flow line normal to the potentiometric surface contour lines, the distance between contour lines is proportional to the hydraulic conductivity in each segment of the flow line. Areas of basalt that have contour lines close together have been mapped in the medium range and areas where the contour lines are further apart have been mapped in the high range of hydraulic conductivity. The hydraulic conductivity of the basalts has been divided into medium and high ranges. The low range has not been considered in the basalts because of the extensive fracturing of the basalt formation and the resulting preferential flow that can occur. Because the data is limited it is convenient to only use three ranges instead of the six ranges recommended by Aller et al. (1987).

Integration of the hydrogeologic settings into a vulnerability map.

The final vulnerability map was generated by combining all of the layers or mappable hydrogeological units mathematically, see Figure 2. This is done using GIS griding capabilities. Each boundary line on the resulting map is not a contour line, but rather a line depicting a setting boundary. A setting boundary line allows the user to evaluate two points but only from a relative and not gradational perspective (Aller *et al*, 1987).

Interpretation of the vulnerability map

The areas of high vulnerability are found primarily in the fractured basalts of the Main Range Volcanics. Aquifers within this class require a high level of protection from developments that have a capacity to cause contamination.

Areas of a moderately high vulnerability are also found within the fractured basalts. Relative vulnerability is reduced by a deeper depth to water table and also due to clay content in the vadose zone. In the south east corner where there is little bore data the determining factor is slope, where the lower the slope the greater the opportunity for contaminants to infiltrate.

The Palaeozoic formations have a higher vulnerability in the far south due to higher average annual rainfall. Most of the remaining granite country exhibits a moderate vulnerability due to the permeable vadose zone. The alluvial aquifers show a moderate and a low moderate vulnerability. The controlling influences here are depth to water table and either clay or sand content in the vadose zone. The consolidated sediments show a low to low-moderate vulnerability relative to the other formations. This is due mainly to the impact of the vadose zone caused by less permeable shales and siltstones. In addition soils tend to be of a texture contrast nature with impermeable horizons.

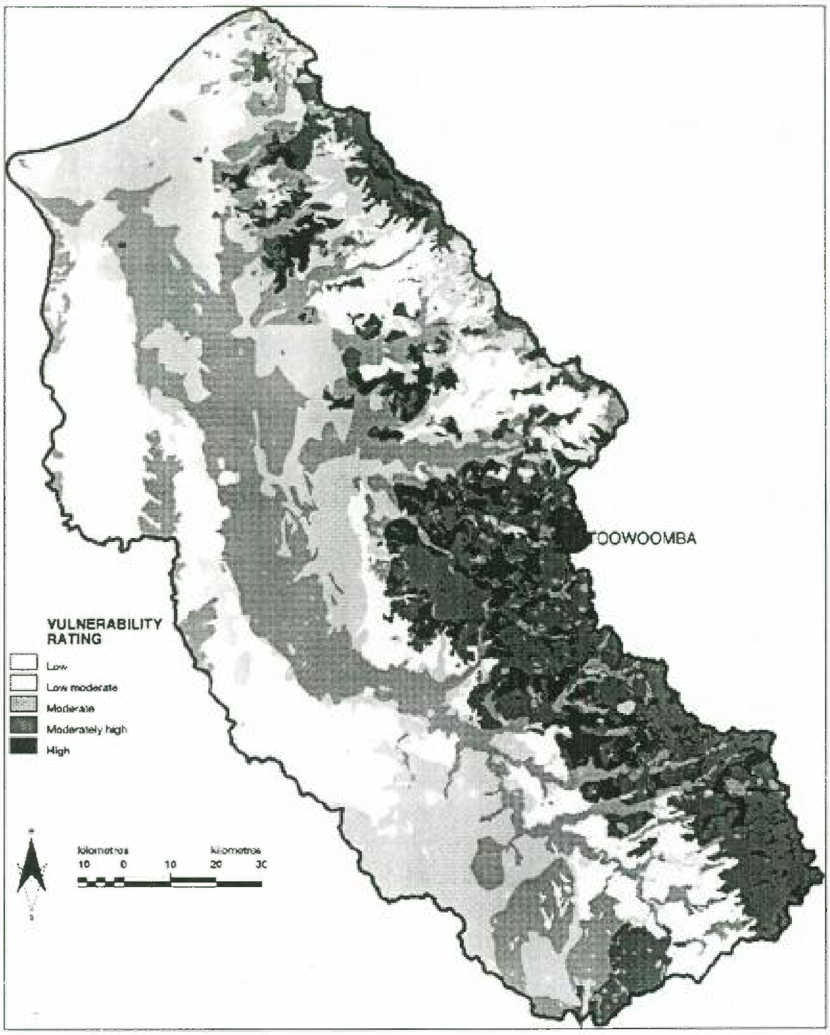

Figure 2. Groundwater Vulnerability Map of Upper Condamine River Catchment.

Validation of the vulnerability map

The final vulnerability map is validated by the statistically higher occurrence of elevated nitrate levels in the fractured basalts than in the other formations.

There are both natural and man made causes resulting in nitrates leaching through the soil profile. A natural cause is highly active nitrogen fixing legumes. Man made causes include septic tank discharge, over application of nitrogen fertilisers, and discharge from intensive animal industries to mention a few. Nitrogen not used by plants or returned to the atmosphere is converted to nitrate in the soil, and nitrate is soluble in water and can easily leach to the water table.

Using existing intensive animal industry location data that covers representative sections of both high to moderately high vulnerability area, and low to low moderate vulnerability area, plots show a correlation between elevated nitrate levels and intensive industries located on areas of high vulnerability. Existing data indicates that nitrates are not leaching from industries on the sandstone or low vulnerability areas to the same extent as they are leaching from industries on the fractured basalt or moderately high to highly vulnerable areas. The aim here is not to single out intensive animal industries as the only source of nitrates, as it is also known that septic tank discharge has caused problems in some areas, but to use them as one example that shows a correlation which validates the vulnerability map. This validation suggests that the vulnerability map of the Upper Condamine River Catchment has been successfully generated using the slightly modified DRASTIC methodology.

Conclusion

The vulnerability mapping was designed to produce user friendly maps to educate decision makers and planning bodies alike, promoting increasing understanding of the fragility of the resource to the potential risk of contamination. Thereby, encouraging the most appropriate location to establish developments to minimise the potential risk of contamination of the underlying groundwater resources. Map users will need to be made aware of the limitations of regional scale vulnerability maps.

Work is under way to produce a groundwater vulnerability coverage for the whole state of Queensland using the modified DRASTIC methodology developed through this project.

Acknowledgments

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Groundwater quality in a aquifer aquitard system subjected to large volume abstraction for irrigation in the Lower Murrumbidgee.

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Abstract

The sustainability of large volume groundwater abstraction for irrigation is currently under investigation at several sites in the Murray-Darling Basin. The Tubbo and Gundaline sites, on the Lower Murrumbidgee alluvial fan, are characterised by a leaky aquifer and aquitard system. The Na-HCO_3 waters EC range is 148 - 890 uS/cm between 120 and 15 metres below ground. Further west, saline groundwater may contaminate fresh groundwater at depth, with leakage primarily via corroded bore casings. Modern groundwater at depth at some sites, determined from ^{14}C and tritium, confirms the importance of the Murrumbidgee River as a recharge source over diffuse recharge.

Introduction

Achieving a sustainable balance between pumping of deep groundwater and maintenance of shallow watertables requires knowledge of groundwater dynamics and salt fluxes through interconnecting aquitards. While groundwater resource investigation and monitoring has historically focused on high yielding deep alluvial aquifers and the near surface has recieved recent attention due to soil salinity, little information is available on groundwater movement and geochemical processes in the intermediate zone. Yet low permeability aquitards play a role in controlling the rate of diffuse recharge and vertical leakage, act as a source and sink of leachable chemicals, and may constitute a significant source of exploitable water (Back, 1985).

The hydrodynamic and hydrogeochemical significance of aquitard sequences to sustainability of groundwater abstraction for irrigation is the subject of the current project. Intensive monitoring of several study sites in the Upper Namoi and Lower Murrumbidgee areas of the Murray-Darling Basin is in progress, the findings of which will contribute specifically to groundwater quality management. Indicators of the magnitude of recharge and leakage will also be of benefit to sustainable groundwater quantity management.

Spatial variation of groundwater salinity in the Lower Murrumbidgee between 110 and 1000 mg/l has been observed in the Calivil and Renmark Formations, with the lowest salinities in the east close to recharge sources. The Shepparton Formation is characterised by highly variable salinity up to 7,000 mg/l. Lawson and Webb (1998) note that long term groundwater quality trends due to over-extraction and inadequate bore construction and abandonment have not been addressed.

The Tubbo and Gundaline nested piezometer sites were chosen as representative of major groundwater abstraction zones outside of the proclaimed irrigation areas. The Tubbo site was located adjacent to a high volume irrigation pump close to the recharge area of the deep aquifer, while the Gundaline site to the west, was located in an area of large drawdown in relative proximity to the Murrumbidgee River. River water samples were taken at Euroley Bridge on the origin of groundwater flow lines to Tubbo.

Methods

Rotary rig drilling with an HQ attachment developed by UNSW Groundwater Centre enabled recovery of undisturbed continuous clay core in 1 metre clear PVC sections of 100mm diameter. Piezometers were completed and developed in June, 1998 (Table 1), with sampling of groundwater quality before, during and after the main 1998/99 irrigation pumping season, using a portable Grundfos monitoring pump. Unstable chemical parameters were analysed in a mobile field laboratory and completed at the UNSW Water Research Laboratory. ICP anion analysis was performed by UNSW School of Geology.

Environmental isotopes tritium, ^{13}C and ^{14}C were analysed by ultra-low level techniques and accelerator mass spectrometry (AMS) respectively at the Institute of Geological and Nuclear Sciences, New Zealand (IGNSNZ). The added analysis cost was justified in the case of tritium by a detection limit lowered from +/- 0.3 TU to +/- 0.016 TU, aiding interpretation of current low levels of tritium in the Australian environment. A significant advantage of AMS for carbon isotope analysis is the 1L sample size, permitting analysis of low yielding aquitard material.

Table 1. Piezometer installation details.

Site	Bore	Screen depth (m)	Easting	Northing	Lithology	Installation date	Diam mm	SWL mbcl* Feb99
Tubbo	B1	20	427410	6161865	sand	Jun-98	50	17.58
Tubbo	B2	32	427410	6161865	sand	Jun-98	50	21.55
Tubbo	B3	50	427410	6161865	clay (15mthick)	Jun-98	50	23.23
Tubbo	B4	65	427410	6161865	sand (8 m thick)	Jun-98	50	25.04
Tubbo	B5	74	427410	6161865	clay (14m thick)	Jun-98	50	25.09
Tubbo	B6	85	427410	6161865	sand	Jun-98	50	26.18
Tubbo	59113	86.8-120.7 (5 intervals)	427410	6161865	sand/gravel	Jan-84	406	
Tubbo	30282/1	100.2 - 117.2 (2 intervals)	429180	6161900	sand/gravel	May-72	117	24.73
Tubbo	30284	36.2 -39.5	429180	6161900	sand/gravel	May-72	203	21.25
Gundaline	A1	47.5	383100	6176150		Jun-98	50	32.88
Gundaline	A2	72	383100	6176150		Jun-98	50	35.45
Gundaline	30348	128 - 134	383100	6176150	sand	Dec-72	101	37.88
Gundaline	30349	15.2-18.2	383100	6176150	sand	Dec-72	101	16.42

* mbcl – metres below casing level

Results and Discussion

Significant heterogeneity of the aquitard-aquifer system is apparent in spatial and temporal groundwater quality variation. At Tubbo groundwater types ranged from Na-Mg-HCO_3-Cl type in the shallow aquifer, and Na-HCO_3 type in the mid-depth aquitard and aquifer to Na-Mg-HCO_3 type in the deep aquifer from which groundwater is pumped. Applied irrigation waters were Na-Cl-HCO_3 due to loss of bicarbonate at the surface.

Groundwater EC at the Tubbo site ranged from 148 to 890 uS/cm between 120 and 15 metres below ground. Murrumbidgee River waters, at the time of major field sampling in February, 1999, had an EC of 92.3 uS/cm and a TDS of 87 mg/L, considerably less than groundwater sampled. Further west, localised groundwater salinity over 4,000 uS/cm was recorded for shallow aquifers expected to be fresh. The presence of iron filings in pumped water from these observation bores suggested the possibility of corroded casings leaking saline groundwater.

Figure 1. Piper diagrams for Tubbo and Gundaline sites illustrating distinctive aquifer/aquitard chemistry.

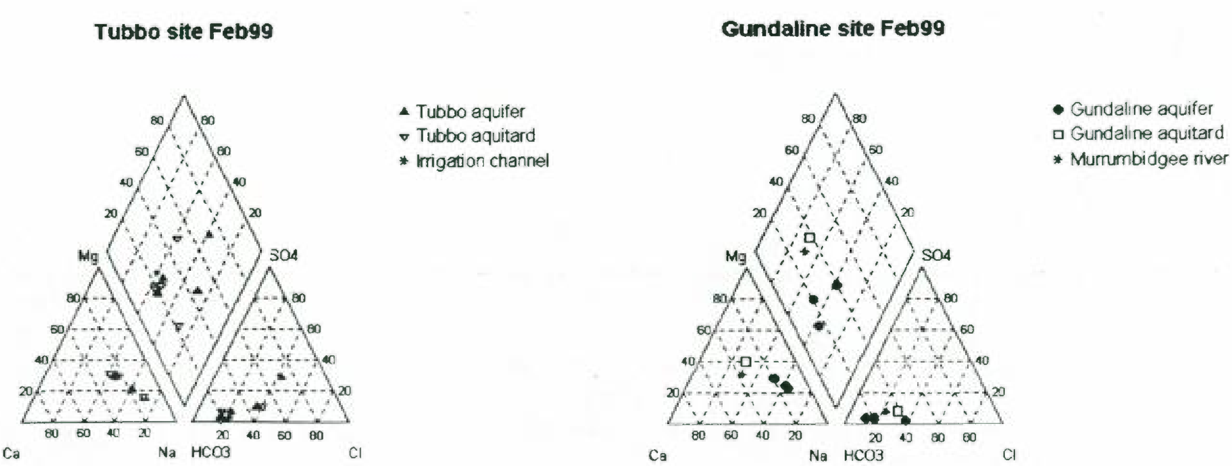
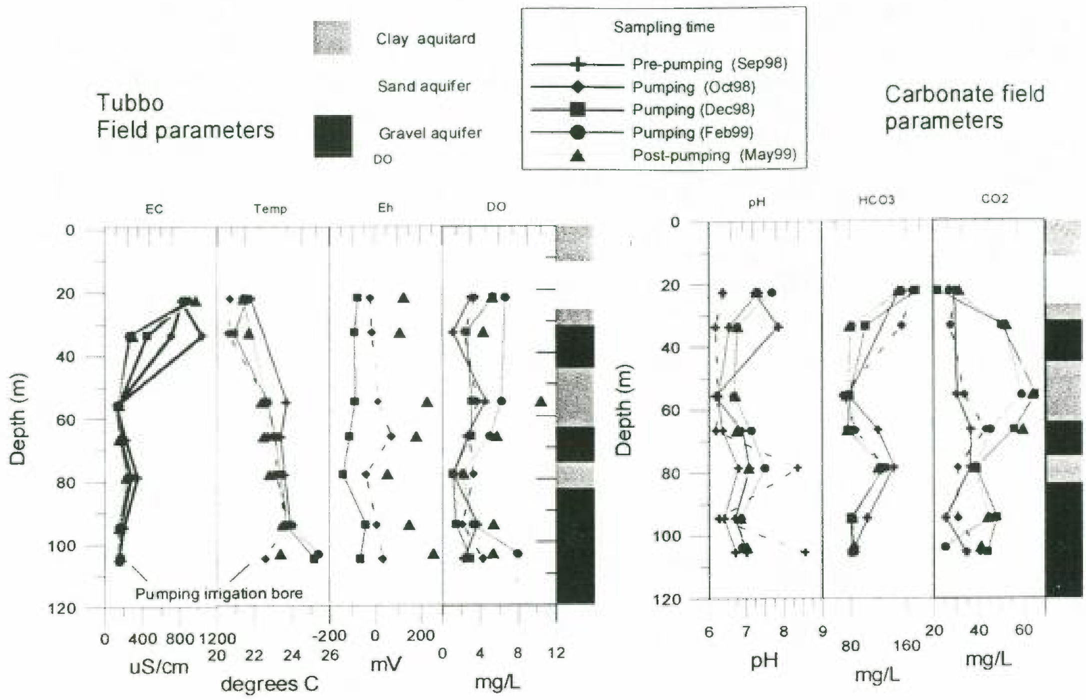


Figure 2 Vertical profiles of groundwater field chemistry for Tubbo site



Water Quality Criteria for Irrigation

The quality of applied irrigation water as defined by a number of salinity criteria, is of importance to sustainability. Total dissolved solids (TDS) is indicative of total salinity, while the sodium adsorption ratio (SAR), exchangeable sodium ration (ESR) and magnesium hazard (MH) indices reflect the hazard of specific ions to soil and plant health.

The SAR and ESR are both indicative of excessive sodium which may result in nutrient competition and ion sensitivity in plants.

$$SAR = Na/((Ca + Mg)/2)^{1/2}$$
$$ESR = Na/(Ca+Mg)$$

MH of irrigation water defines high magnesium content.

$$MH = Mg/(Ca + Mg) \times 100$$

The classification of water quality for irrigation according to these criteria is summarised in Table 2 below.

Table 2. Reference values for water quality criteria for irrigation.

TDS mg/l	SAR	MH
<250 Low salinity	<10 Low	>50 harmful effects
250 – 750 Medium salinity	10-18 Medium	
750 – 2250 High salinity	18-26 High	
2250 – 5000 Very high	>26 Very high	
Lloyd and Heathcote, 1985	Hounslow, 1995	

Groundwater quality for irrigation at the Tubbo and Gundaline sites was in general found to be very good (Appendix B). TDS ranged from 87.0 to 627.8 mg/l, SAR from 0.49 to 4.14 and ESR from 0.49 to 2.51, all classified as low, for both clay aquitard and gravel aquifer sequences. Water quality is excellent for irrigation use, classified as C1-S1 waters according to Hounslow, 1995. High magnesium concentrations were found to be of most concern, with groundwater samples above the recommended guideline. The MH ranged from 45.63 to 61.76. Surface waters from the Murrumbidgee River exhibited the lowest SAR, ESR, MH and TDS and are thus of better quality for irrigation usage than groundwater.

Groundwater age dating

Table 3 Groundwater dating results

Bore	Sample date	TR[1]	+/- TR	DIC mmol/l[2]	DIC mmol/kg[3]	δ13C	PMC	Drury et.al 1984 δ13C	PMC
30349	13.02.99	-0.005	0.017	5.73	2.935	-16.89	25.99		
A1	13.02.99			4.62	3.975	-15.71	55.20		
A2	13.02.99			5.14	3.891	-16.24	59.47		
30348	13.02.99	1.63	0.05	3.60	4.120	-16.78	105.81		
30284	14.02.99	0.137	0.016	2.39	2.856	-15.62	86.42	-18.1	94.4
30282	14.02.99			2.08	2.438	-14.18	87.60	-5.5	94.3
B1	14.02.99	0.022	0.017	2.87	2.662	-17.07	83.46		
B2	14.02.99			2.93	2.269	-17.33	92.62		
B3	14.02.99	0.020	0.016	3.12	2.343	-17.79	92.70		
B4	14.02.99			2.12	2.141	-16.70	85.39		
B5	14.02.99			2.63	2.330	-17.29	91.76		
B6	14.02.99	0.013	0.017	2.64	2.096	-17.17	83.24		

1 TR used by IGNSNZ is equivalent to TU
2 Calculated DIC by NETPATH (Plummer, Prestemon, Parkhurst, 1994)
3 DIC by gravimetric determination of water sample quantity and DIC recovered quantitatively as CO_2 (IGNSNZ)

Modern groundwater, as indicated by significant tritium levels were found for bores 30348 and 30284. The absence of signficant tritium levels for all depths at the Tubbo site and at 16.7 m at the Gundaline site, suggest that recharge from the surface is minimal. The detection of relatively high tritium at a depth of 130m at the Gundaline site is highly significant, reflecting direct recharge by the Murrumbidgee River. Stream bed recharge west of Darlington Point has not previously been considered important. Tritium in bore 30284 can be attributed to bore casing leakage, consistent with observed hydrograph peaks.

Interpretation of ^{14}C results is not straightforward due to the complex evolution of DIC in the groundwater system, including dilution of initial ^{14}C during carbonate dissolution. Modern water was found in bore 30348, with the apparent ages of all other waters ranging from 630 to 1100 years. There is little correlation between PMC and depth, or aquifer/aquitard material. Comparison with Drury (1984) suggests that older waters are now found at bores 30282 and 30284, possibly as a result of large volume groundwater pumping in the area. Younger waters in the overlying aquitard make it unlikely that vertical leakage is responsible for this trend.

Simple corrections based on Tamers (1975) alkalinity correction and Pearson (1965) $\delta^{13}C$ mixing model, reduced apparent age by a factor of between 0.5 and 0.95, with an anomalous result for bore 30282. The need for more detailed geochemical interpretation and process modeling to enhance the confidence of groundwater dates, define leakage and mixing, and better understand groundwater interaction with clay aquitard material, is apparent.

Conclusion

Groundwater quality sustainability of current groundwater based irrigation in the Tubbo and Gundaline areas is good, provided that there is no significant salt store in the unsaturated zone and groundwater levels do not rise. Of the water quality criteria SAR, TDS and MH, only magnesium concentrations exceed recommended limits. Undisturbed clay core analysis, in progress, including EC 1:5 extracts profiles from each of the major aquitard sequences at the Tubbo site, correlated with geophysical EM39 downhole logs will provide further insight into salt storage at depth. Further work aims to quantify vertical leakage through radial flow modeling, and hydrogeochemical modeling of mixing and aquitard matrix interaction.

The absence of salinity in these areas relative to the adjacent Coleambally Irrigation area may be attributed to a number of factors including the usage of groundwater instead of surface water, less intensive pasture irrigation, and a more highly permeable and leached aquitard/aquifer sequence. Comparative site investigations in the Coleambally Irrigation Area would be required to substantiate these factors and to better ascertain the sustainability and quality of the groundwater resource on a regional basis.

Acknowledgements

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APPENDIX A Selected groundwater quality results

Bore	Sample date	SWL mbcl	Field EC uS/cm	Temp °C	Field pH	Field Eh (May99)	Field DO mg/l	Field HCO3	Field CO2	NH3-N av mg/l	NO3 av mg/l	Phos av mg/l	S mg/l	Cl mg/l	Ca mg/l	K mg/l	Na mg/l	Mg mg/l
RIVER	11-Feb-99	HIGH	92.3	22.7	7.297*	261.7	6.2	46.46	3.33	0.08	0.50	0.13	4.02	9.07	6.70	1.28	6.33	3.41
30349	13-Feb-99	16.42	524	20.4	6.45		x	197.98	95.21	0.08	0.40	0.64	19.73	56.03	33.70	1.04	36.06	26.23
A1	13-Feb-99	32.88	435	23.5	7.22		0.3	250.46	21.97	0.02	0.33	0.51	8.64	21.14	13.50	1.04	70.09	13.23
A2	13-Feb-99	35.445	487	24.2	6.89		0.2	247.94	42.93	0.90	0.40	0.43	9.06	31.66	19.33	1.34	59.10	17.75
30348	13-Feb-99	37.88	422	24.8	6.66		2.11	151.11	36.81	0.41	0.80	0.33	3.99	55.53	11.60	8.99	57.74	12.11
30282	14-Feb-99	24.73	148.3	25.7	6.57		4.6	80.81	39.26	BDL	0.50	0.32	1.40	10.39	8.03	0.88	15.53	5.86
30284	14-Feb-99	21.25	365	25.5	7.11		4.5	125.54	35.92	0.12	0.60	1.03	16.99	48.87	9.18	1.20	56.80	6.34
B1	14-Feb-99	17.58	890	21.4	7.19	127.2	6.7	154.55	27.23	BDL	0.87	0.49	123.28	136.30	33.76	1.80	127.11	22.56
B2	14-Feb-99	21.55	267	x	6.27	110.4	x	82.08	51.98	x	0.30	0.39	12.61	38.61	14.55	0.94	25.16	9.93
B3	14-Feb-99	23.23	157.1	22.7	6.18	230	6.29	77.76	59.27	x	0.30	0.46	3.60	11.54	8.18	0.66	17.06	5.70
B4	14-Feb-99	25.04	176.8	22.7	6.64	183.4	5.06	86.17	45.45	0.02	0.30	0.46	5.77	14.94	9.31	0.76	19.06	6.47
B5	14-Feb-99	25.09	245	23.1	7	56.2	2.15	132.32	39.11	BDL	0.40	0.36	9.07	15.87	6.81	0.91	41.54	4.92
B6	14-Feb-99	26.18	156.7	24	6.37	151.2	3.25	83.98	45.09	x	0.30	0.24	2.05	10.33	7.99	0.81	17.26	5.79
CHAN	13-Feb-99	x	166.7	26.3	7.57		x	61.65	7.03	BDL	0.27	0.31	1.81	11.02	7.75	0.76	17.79	5.85
59113	13-Feb-99	x	168	25.4	6.39	260	8	85.93	25.12	0.00	0.27	0.36	2.20	10.78	7.90	1.00	18.00	6.01

APPENDIX B Water quality criteria for irrigation

Bore	Sample date	TDS[1] mg/l	SAR	ESR	MH
RIVER	11-Feb-99	87.0	0.49	0.49	45.63
A1	13-Feb-99	424.4	3.23	1.72	61.76
A2	13-Feb-99	397.5	2.32	1.06	60.21
30348	13-Feb-99	308.4	2.82	1.59	63.23
30349	13-Feb-99	379.1	1.13	0.41	56.18
59113	13-Feb-99	156.3	1.17	0.88	55.62
30282	14-Feb-99	180.1	1.01	0.76	54.61
30284	14-Feb-99	283.4	3.51	2.51	53.22
B1	14-Feb-99	627.8	4.14	1.56	52.4
B2	14-Feb-99	207.8	1.24	0.71	52.94
B3	14-Feb-99	154.3	1.12	0.84	53.45
B4	14-Feb-99	171.2	1.17	0.83	53.38
B5	14-Feb-99	233.4	2.95	2.42	54.35
B6	14-Feb-99	181.4	1.13	0.85	54.42
CHAN	13-Feb-99	122.2	1.17	0.89	55.43

1. TDS calculated on major cations, anions and silica by NETPATH (Plummer, Prestemon, Parkhurst, 1994)

IRRIGATION

Reducing subsurface drainage salt loads: development of drainage design and management guidelines

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Summary

A field investigation on a new vineyard in the MIA showed that improved subsurface drainage systems reduce the salt load in the drainage whilst continuing to provide waterlogging and salinity control. By only draining the rootzone the drainage volume and salinity were greatly reduced. Improved design and management options were tested against the current practice of deep pipe drainage. By managing existing drainage systems to only flow when the watertable was within 1.2 m of the surface, and not during irrigation, resulted in a 50 % reduction in the drainage salt load. Shallow closely spaced drains reduced the drainage salt load by 95 %. This improved design and management will significantly reduce the amount of salt that requires disposal. This work together with other field and modeling studies have been used to develop a set of guidelines for subsurface drainage with the aim of improving drainage water quality.

Introduction

The irrigation areas in southeastern Australia have developed shallow water tables to the extent that about 80 % of many irrigation areas experience water tables above 2 metres and the area with water tables above 1 metre is increasing. These water tables create serious problems of waterlogging and land salinisation. Waterlogging has in the past been controlled by the installation of subsurface drainage (tile drainage) which lowers water tables. This has been successful in horticultural farms of the Murrumbidgee Irrigation Area (MIA), Shepparton Irrigation Region and the Riverland along the Murray River. However, the nature of subsurface drainage is such that large amounts of salt are exported in the drainage effluent. At the time of installation the downstream consequences of salt export were not considered. Subsurface drainage schemes have been targeted as areas for salt export reduction, as the drainage is normally an order of magnitude more saline than surface drainage waters. In the MIA, the Land and Water Management Plan (L&WMP) identifies subsurface drainage as a major salt exporter from the area. About 30 % of the salt load leaving the area is from subsurface drainage, although only 7 % of the area has subsurface drainage installed. The MIA L&WMP sets out as a goal a 25 % reduction in the salt load from existing subsurface drainage. In the MIA new horticultural developments are to a large extent on the heavier soils that were previously used for annual crops such as rice and vegetables. These soils are quite different from those previously associated with horticulture which were freer draining lighter textured soils. Thus new drainage design and management is required to reduce salt loads and provide effective drainage in heavy clay soils.

This new subsurface drainage philosophy aims to reduce salt mobilised by drainage, whist providing effective drainage of the rootzone and prevention of rootzone salinisation. Since only the upper soil layers 'the rootzone' need to be drained, draining deeper soil layers which are usually more saline can be avoided. Previous research by Deverel and Fio (1991) and others investigating the hydraulic processes of deep and shallow drains have found that for shallow drains there was a 30 –50% reduction the contribution of saline flow from deep in the profile compared to deep drains. They also found that shallow drains flow less frequently resulting in reduced salt loads.

Methodology

Site Description

New subsurface drainage design and management strategies were tested in a replicated field trial located in a newly established vineyard in the MIA, situated 30 km north east of Griffith, NSW. The vineyard was established in 1995 after previously being used for growing rice. The soil was a Griffith Clay Loam, according to the soil description of Butler (1979). The top 0.3 m is 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. Irrigation was applied down narrow furrows on both sides of the vines. Water was siphoned into these furrows from a head ditch. Irrigations lasted for around 8 hours every 10 days or so. The irrigations were well managed with rapid advance times, about 4 hours to reach the bottom of the 400 m vine row, and only a small amount of run-off. The irrigation furrows were maintained in good condition throughout.

Drainage treatments

Drainage treatments installed in the experiment were:

- ***Deep Drains*** - Pipe drains at 1.8 m deep and 20 m apart allowed to flow continuously. This treatment replicated current traditional drainage design and management, commonly known as 'tile' drains.
- ***Managed Deep Drains*** - Pipe drains at 1.8 m deep and 20 m apart managed to flow only when the watertable was within 1.2 m of the surface, and not during irrigations.
- ***Shallow Drains*** - Shallow 'Mole' drains 3.6 m apart and 0.8 m deep.
- ***No Drainage*** - No subsurface drainage.

The objectives of the experiment were to assess the ability of shallow closely spaced drains to provide effective drainage of the rootzone and prevent salinisation, whilst minimising salt mobilisation into the drainage; to test the practice of managing existing drains to flow only when within 1.2 m of the surface and not during irrigations; and determine if deep drains with improved management are as effective as shallower drains in reducing salt mobilisation, whilst managing waterlogging and rootzone salinity.

Experiment layout

The experiment was laid out in two blocks of equal area with complete randomisation in each block, each treatment was 70 m long of varying width, see figure 1. Each individual drainage replicate had it's own sealed collector drain running to the pump sump where measurement s of drainage quantity and quality were made.

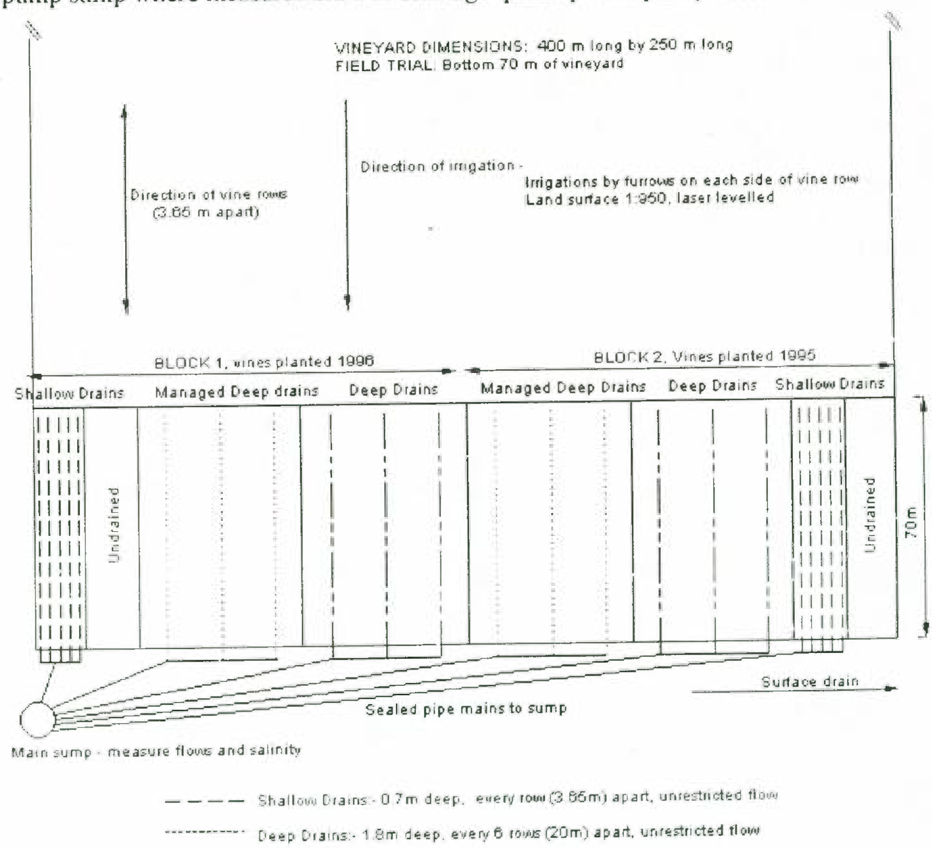

Figure 1. Field experiment layout

Experimental measurements

The experimental measurements aimed to quantify the drainage volume and quality from each treatment, and compare this with the effect of each treatment on water tables and soil salinity in the root zone. Crop measurements were also made to ascertain the overall effect of the drainage treatments on vine growth and yield.

Irrigation applied to the farm was measured at the Dethridge wheel located directly on the head channel to the vineyard. The wheel was calibrated using a Doppler ultrasonic flowmeter at the beginning of the experiment. Irrigation water salinity was sampled several times during each irrigation. Rainfall was measured at the sump. Run-off was estimated at 10% of water applied. Drainage discharge from individual treatments was measured manually at the pump sump. Measurements were taken at around half hourly intervals at times of peak flows after irrigation, and subsequently larger intervals as the flow rates declined. Drainage samples for electrical conductivity and chloride were taken in conjunction with flow rate measurements. Drainage water salinity was recorded for different stages of each irrigation event, post irrigation, and rainfall events. Perched watertables and piezometric levels in each drainage treatment were measured using 1 m test wells and 3 m piezometers. These were situated in the vine row between drains in the shallow drainage treatments, and above and between the drains in the deep drainage treatments. These were logged at half hourly intervals. Soil salinity was measured after each irrigation season by soil sampling to 2 m and EM38 survey. To determine the effects of the drainage treatments on the crop, leaf chloride and yield measurements were undertaken at the end of the experiment.

Results and discussion

The different drainage treatments resulted in markedly different drainage volumes and salinities, and hence salt loads. The differences in flow resulted from the drain position in the soil profile and the management of the drains. The Deep Drains flowed continuously for the irrigation seasons, a small saline flow being sustained between irrigations and a large flow during and just after irrigation. Figure 2. The Deep Drains continued to flow long after an irrigation had ceased because they were draining a larger soil volume, down to 1.6-1.8 m, and they were influenced by regional groundwater pressures. This was despite the area having no significant shallow aquifer systems and being in a fairly flat area so that hydraulic gradients from neighboring farms and channels were small. That there were some regional effects was demonstrated by the rise in piezometric levels at the beginning of the irrigation season in the experimental area before any irrigations had been applied. The Managed Deep Drains were less influenced by these regional effects and the Shallow Drains were completely isolated from them.

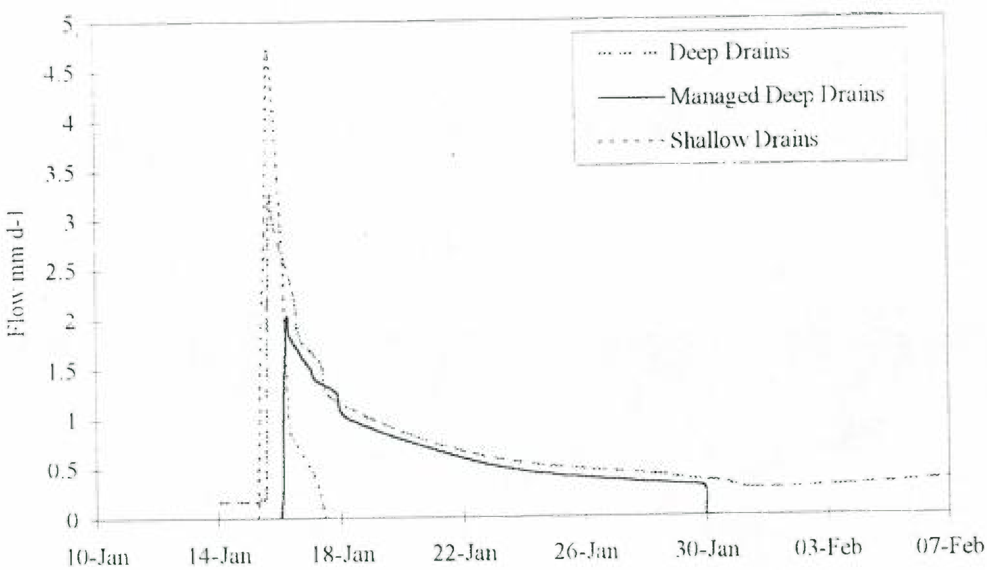

Figure 2. Drainage treatment hydrographs during and after an irrigation

The Deep Drains removed the most water at the highest salinity, about 11dS m^{-1}, and hence had the highest salt load. The Managed Deep Drains had 33 % less flow than the Deep Drains at a lower salinity, 7–8 dS m^{-1}, resulting in a 49 % reduction in salt load. Table 1. The Shallow Drains removed 78 % less water than the Deep Drains at a significantly lower salinity, about 2 dS m^{-1}, resulting in a 95 % reduction in salt load. The large amounts of water removed by the Deep Drains leads to reduced overall water use efficiency and increased farm costs in terms of increased pumping and nutrient loss. The extra salt removed by the Deep Drains compared to the Shallow Drains and even the Managed Deep Drains doesn't have a negative impact on the drained area but will adversely affect the receiving waters. If in the future farmers are charged for the amount of salt they export off farm then this extra salt export will have a negative effect upon farm income. Where farms are denied the option of off farm disposal of drainage water, then the use of shallower drains will be advantageous in reducing the overall volume requiring disposal and also the lower salinity of the drainage water will leave more options open for reuse.

Table 1. Drainage treatment salinity

Drainage Treatment	Drainage Volume (mm)	Average Drainage Salinity (dS/m)	Total Salt Load (kg/ha)
Deep Drains	70	11	5867
Managed Deep Drains	47	7 – 8	2978
Shallow Drains	15	2	319

Of the three drainage treatments tested only the Shallow Drains came close to a salt balance with the irrigation water, removing 0.7 of the salt applied in the irrigation water. This is actually a small accumulation of salt, but this was in absolute terms a very small amount 170 kg ha^{-1}, and was not accumulated in the root zone. The Deep Drains removed 11 times more salt than was applied, a large net leaching of salt. This leaching was not reflected in a large reduction in soil salinity in the top 2 m, thus this salt was from below drain depth. This was somewhat reduced by the managed treatment which exported 5 times more than the salt applied, still a large net export. This shows that drains placed deep in the soil profile will export large quantities of salt over and above that applied in the irrigation water. Assessment of the drainage water salinity with depth of water table confirms this, figure 3. When the water table was at one metre below the surface the drainage water salinity from the Deep Drains was around 8 dS m^{-1}; as the water table fell to 1.6 m below the surface the salinity increased to around 11 dS m^{-1}. This is consistent with the theory that deeper drainage intercepts deeper water flow paths that move through more saline portions of the soil profile.

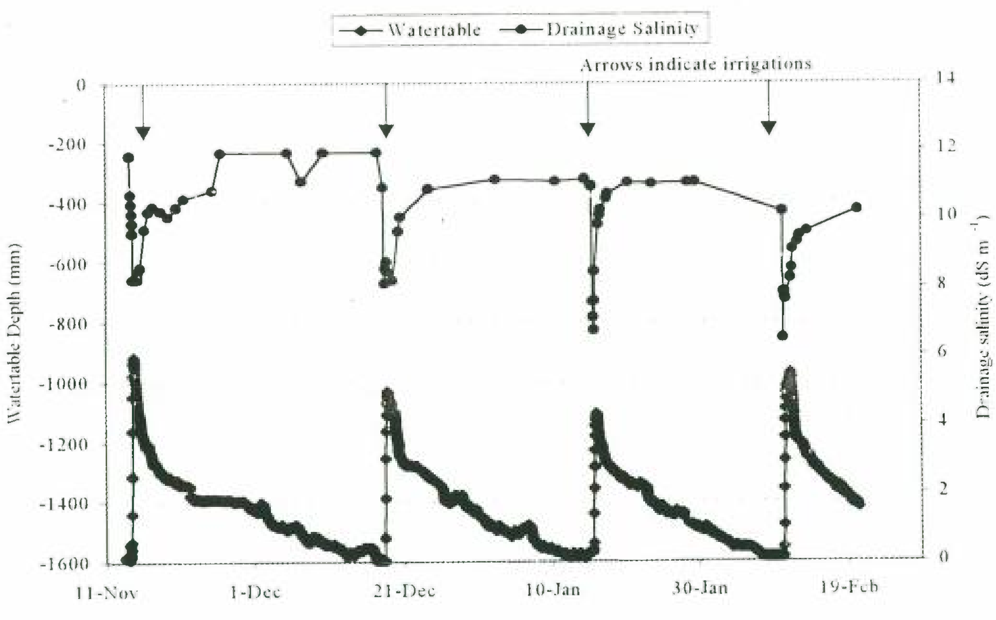

Figure 3 Drainage water salinity and watertable depth between drains

In terms of water table and waterlogging control the Deep Drains reduced the periods of high water tables and waterlogging to a negligible amount, Table 2. The management changes used to control water flow from deep drains had little effect on waterlogging. The Shallow Drains gave the best control of root zone waterlogging, the water table did build up beneath this treatment but was controlled at mole depth.

Table 2. Duration of water tables above specified depths (%)

	Water table depth (mm)		
Treatment	**300**	**500**	**1000**
Deep Drains	3	4	8
Managed Deep Drains	1	3	10
Shallow Drains	1	1	3
Undrained	3	8	42

These varying water table regimes resulted in some differences in the root zone soil salinity trends over the two seasons monitored, Figure 4. The Undrained treatment soil salinity remained static, whereas all the drained treatments showed a decrease in salinity after the first season. In both the Shallow Drains and Managed Deep Drains there was a rise after the second season resulting in no net change. For the average salinity down to two metres this picture was similar except the Deep Drains showed a fall in salinity after both the first and second seasons. These results are somewhat unclear in terms of the possible effects of the different treatments on long-term soil salinities especially since the undrained treatment did not show any change in salinity over the experimental period. However, there is an important outcome from this analysis in that, the drainage treatments had only small effects on the root zone salinity, no measurable effect on vine health over the experimental period, but still drained water and salt from the area. So over this particular time the water drained, salt removed, costs incurred and downstream impacts of drainage water resulted in little benefit to the farm. Under these circumstances of small benefit from a drainage system, which can occur due to site factors, dry climatic conditions and plants not highly susceptible to waterlogging, it is even more important that the drainage system incurs the least downstream impacts and least costs to the farmer.

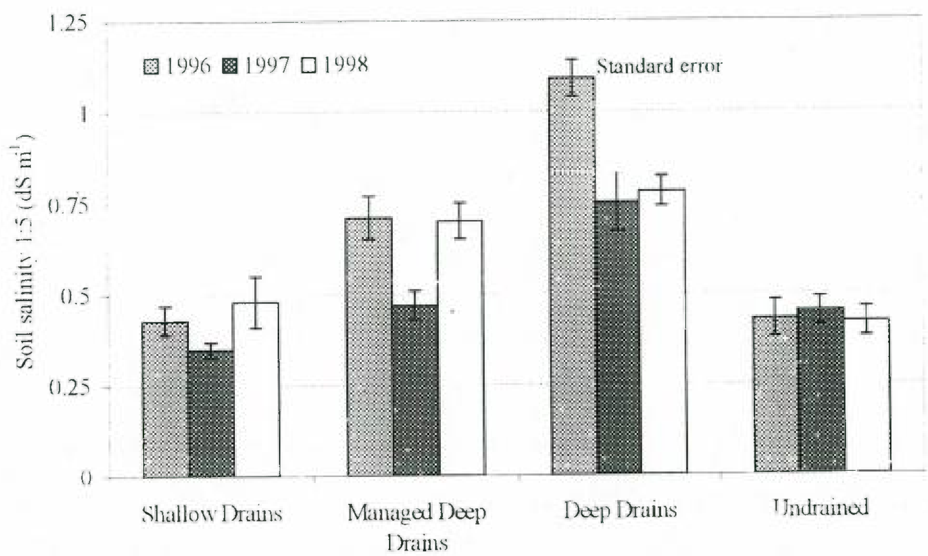

Figure 4. Change in salinity in the top 600 mm of soil

Discussion

This field trial was conducted in a dryer than average year, the relatively low drainage flows and static soil salinities reflect this. During wetter conditions it is likely that the drainage water reduction would be greater than measured here and that there would be more soil leaching due to rainfall, both of which are positive. However, there may be negative effects due to the design and management suggested, such as increased waterlogging. These negative effects are unlikely to be great and with good management could be monitored and controlled. For instance if the water table was remaining high for too long on the Managed Deep Drains then the drainage depth could be increased to provide a greater soil buffer to store rainfall. The main negative effect of a prolonged wet period on the Shallow Drains would be an increased rate of collapse in the mole drains. At this site, the soil was quite stable and as such it is unlikely that the moles would collapse to the point of being ineffective within a single season. Obviously if a shallow pipe system was installed this would not be a concern.

Predicting the effects of wetter periods on the results of drainage design and management tested here is possible. The likely drain flow under wet conditions can be considered by analysing single high input irrigation events. Figure 5 shows the proportion of water applied that drained through the different drainage treatments at a particular event. During wet periods when the soil has a small storage capacity due to antecedent rainfall then a lot of the water applied that does not run off will be drained out, similar to irrigations 3 and 4 in Figure 5. This gives an indication that under wet conditions it is likely that up to 25 % of water applied may be drained by a deep pipe drainage system, whether managed or not. The rate of drainage can be predicted by considering the drain hydrographs such as in Figure 1. The peak flow rates shown here are unlikely to be greatly exceeded, but the duration of the peak flows will be prolonged under wet conditions with high inputs of water. The effect of wetter conditions on drain water salinity can be considered using the drainage water salinity as a function of water table depth results. Since water tables are likely to be high during wet conditions the drain water salinity will be lower than dry periods when the water tables are deeper. The height and duration of water tables during wetter periods is harder to predict. Water table depth is a function of the time from the last recharge event, the drainage rate of the system and the combination of deep leakage and plant water use. If recharge events are larger and at shorter intervals then water tables will remain higher. Analysis of water table responses under a particular set of weather conditions can be undertaken using water balance models such as SWAGMAN Destiny (1996) and BASINMAN, Wu *et al.* (1999).

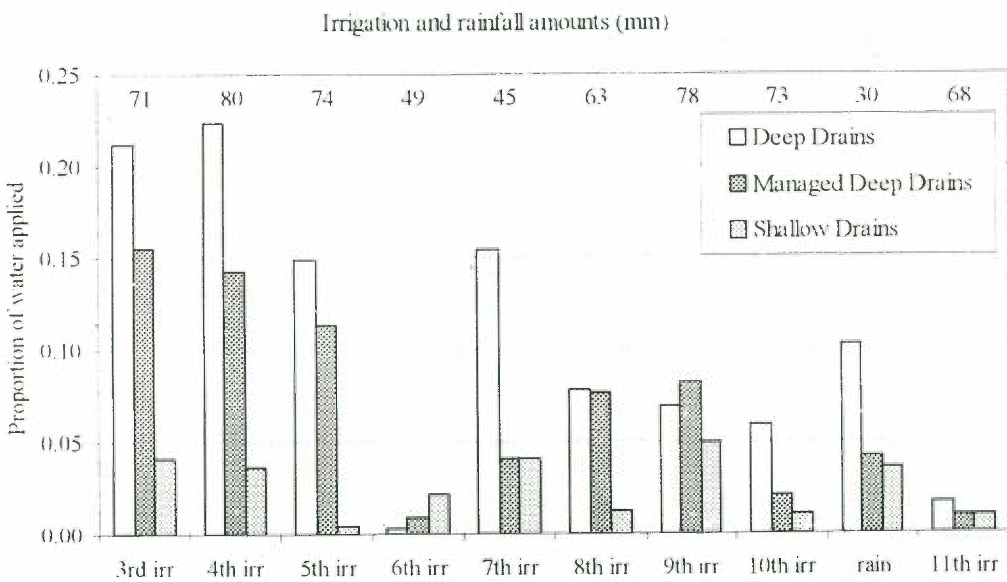

Figure 5 Drainage as a proportion of water applied at each event

In regard to minimising costs of a drainage system the Shallow Drain system of mole drains would cost about \$800 ha^{-1} for the collector mains and mole drain installation, whereas the pipe drains, installed 20 m apart to achieve an adequate degree of waterlogging control, would cost about \$2700 ha^{-1}. Thus a shallow drain system consisting of mole drains provides a much cheaper system of drainage as well as reducing the disposal problem of the drainage water. An indication of the impact of a drainage system on water use efficiency and hence total water costs is shown by the Deep Drains that drained 20 % or more of the water applied in 23 % of plot drainage events and drained 10 % or more in 65 % of plot drainage events. This is a considerable proportion of the water applied that was intended for use by the plant. Management of deep drains cut the proportion of plot drainage events draining more than 10 % of water applied to 37 % and events draining more than 20 % to 6 %. This is a significant improvement but does not match the Shallow Drains which drained less than 5 % of the water applied in 90 % of plot drainage events.

Conclusions

Drainage systems for irrigated areas on clay soils in south eastern Australia can be designed and managed better than the currently accepted practices, so that detrimental downstream environmental effects due to excessive salt export are reduced, without affecting the productivity of the farm.

Changing drainage design from deep widely spaced drains to shallow closely spaced drains:

- Shallow drains remove less irrigation water than deep drains, thus reducing irrigation losses.
- Shallow drains have low drainage water salinity and remove smaller drainage volumes, thus reduce salt load, up to 95 % reduction compared to deep drains in this trial.
- Shallow drains control waterlogging better than deep drains.
- The cost of a shallow mole drainage system can be about 75 % less than deep pipe drainage (to provide similar waterlogging control)
- Deep drains can take up to 32 years to leach all the salt from their capture zone and reach an equilibrium with irrigation water, whereas shallow drains can reach close to equilibrium within two irrigation seasons.

Managing deep drains by preventing discharge during irrigation and whenever the water table was below 1.2 m deep:

- Managed deep drains reduce irrigation water losses compared to unmanaged deep drains
- Managing drains reduces flow and drainage water salinity compared to unmanaged drainage, resulting in reduction in drainage salt load, 50 % in this trial.
- A more rapid decline in drainage water salinity can be achieved by managing deep drains.
- A deep pipe irrigation system, without major groundwater inflow from surrounding areas, only needs to be run for 2 to 7 days after an irrigation to control the water table below the root zone and then can be switched off.

This work together with other field studies and modelling of drainage by the authors in vineyards and orchards in the MIA has enabled the development of some general guidelines for the design and management of subsurface drainage in the Riverine plain. These guidelines are shown separately as the last page of this paper.

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Guidelines for subsurface drainage design and management for improved drain water quality

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Introduction

For the Riverine plain of southeastern Australia there is a need to consider drainage design and management with regard to water quality considerations. This requires that drainage designs include criteria that minimise salt mobilisation from outside the root zone and make management of the drainage system possible in order to manipulate the quantity and quality of drain discharge. This requires an approach that considers the design and management of the drainage system as an integral part of irrigation management. This approach needs to also always consider the potential for salt leaching and thus drainage disposal with any new drainage system.

Aim of the guidelines

To reduce subsurface drainage quantity and salinity from irrigated agriculture in the Riverine plain. to meet increasing disposal constraints.

Background

Research in the Riverine plain has shown that in general terms the following are true.

1. Deep drains (traditional tile or pipe systems about 2m deep and 20-40m apart):
   - Have high drainage volumes and salinity
   - Can unnecessarily extract large quantities of salt from the soil below the root zone
   - Can drain large volumes of water from sources outside the farm itself
2. Shallow closely spaced drains:
   - drain less water at lower salinity than deep widely spaced drains
   - have lower potential salt mobilisation than deep widely spaced drains
   - are less likely to be affected by water sources from outside the farm area
   - give the best waterlogging control in clay soils
3. By managing deep drains it is possible to:
   - reduce drainage volume and salinity
   - still control waterlogging and root zone salinity adequately

Guidelines

New drainage systems should consider the potential for salt mobilisation:

- Avoid sites where large volumes of drainage may occur from regional sources
- Install drains as shallow as possible
- Design drainage systems into management units aligned with irrigation units
- Install drainage control structures to manipulate water tables
- Main drains and sumps which are installed at depth should be sealed to prevent entry of saline water

Existing drainage systems:

- Should where necessary and possible be divided into management units. aligned with irrigation units
- Should have water table management control structures installed where necessary

Management of drainage systems:

- Drainage systems should be prevented from discharging during irrigation events
- Drainage systems should be controlled to maintain water tables safely below the root zone. not left to drain uncontrolled where water tables may fall much deeper than required.
- Drainage systems should normally be kept closed or turned off and then turned on as needed. rather than being left running at all times without consideration whether the drainage is necessary

ASSESSMENT OF DEEP BORE WATER QUALITY FOR IRRIGATION IN THE MURRAY IRRIGATION DISTRICTS

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ABSTRACT

Expanding demands for additional supplies of irrigation water has evoked keen interest among farmers to install deep bores in the Murray Irrigation Districts (MIDs) of southern NSW. Guidelines that are currently used for issuance of licenses are ambiguous considering sustainable use of ground water for irrigation. This investigation aimed at an assessment of water quality for sustained irrigation using deep bores installed in the MIDs. For this purpose, water samples from 85 deep bores (30 private or commercially used bores by farmers & 55 investigation bores maintained by DLWC) were collected and analysed for parameters pertinent to their evaluation for irrigation. These bores pump water from relatively deeper formations namely Calivil (70-140 m) or Upper Renmark (>140 m) or Onley aquifers and a few use combination of both.

Analytical and parametric results that describe water quality for irrigation indicated potential salinity and/or sodicity problems associated with most bores. This is due to their moderate to high salt content, sodium adsorption ratio (SAR), consistent presence of bicarbonates or residual alkalinity, and unfavourable Mg/Ca ratio or low calcium concentrations. The electrical conductivity (EC) of bore waters was found to vary between 0.13-51.5 dS/m. Mean (1.64 dS/m) and median (1.32 dS/m) EC of private bores was much less than the corresponding 6.39 and 2.86 dS/m for investigation bores respectively. Only about 9% bores had EC within the safe limit (<0.7 dS/m) of potential salinity problems whereas 56% were marginal and the rest considered unsafe for sustained irrigation usage. In addition, most bores pose potential soil sodicity or infiltration problems. High SAR waters require appropriate salinity level for their desired infiltration but only 5% of these bores had SAR less than 3 considered safe for irrigation. A significant majority (67%) of the bores had SAR between 3-12. Amount of dissolved calcium was mostly less than what (2.0 me/L) is regarded necessary for nutritional requirements of most crops. The Mg/Ca ratio of both private and investigation bores showed an imbalance between the two. Water quality in the west was comparatively lower than the eastern region and private bores were relatively better off than the investigation ones regardless their aquifer and geographic location. Considering rice based farming systems and relatively impermeable clay soils having low levels of soluble calcium, sustained irrigation with most deep bores will result in genesis of problems of soil sodicity. Research is thus needed to quantify their impact and development of strategies for sustainable use of deep bore waters for irrigation either alone or in conjunction with channel water.

INTRODUCTION

Adequate and secure supplies of good quality water are a must for sustainable irrigated agriculture. Recent droughts, MDBMC cap, and declining surface water allocations in the southern NSW have caused considerable increase in the groundwater uses. Hence, more pressure on groundwater resources. In the Murray Valley alone, there are more than 500 active irrigation, industrial, stock and domestic bores at present. Community concerns for sustainable environment, healthy rivers and other fresh water resources, increasing domestic and industrial requirements for surface water including their use for recreation have now created a tough competition for irrigated agriculture. Considering limited scope for a major boost in surface water supplies, use of groundwater resources is an option. Past experiences in many regions show that available surface waters when supplemented with groundwater supplies improve reliance on water availability. In Australia, groundwater accounts for only 12.6% of the total water supplies used for irrigated agriculture (AWRC, 1987). Availability of good quality groundwater and its irrigation use in combination with surface water minimise risks associated with soil salinisation and waterlogging.

Groundwater normally results from the seepage of surface or rainwater. But in Australia less than 0.2% of the annual rainfall (420 mm) recharges aquifers while 87% is lost through evaporation and/or transpiration, and 12.8% goes as run-off to various water bodies (AWRC, 1987). Limited recharge pose limitations related to quality and sustainable use of various aquifers. Assured water supplies are no good if their quality is unfit for sustainable irrigated agriculture. Quality accounts for several water characteristics that influence its suitability for a specific use (Ayers and Westcot, 1985). Most researches indicate salinity and sodicity the most important constraints related to irrigation water quality. In addition, presence of several important trace elements or specific ions for their toxicity to crops, soils and animals are also recognised. Use of low quality irrigation water for sustainable agricultural production depends largely on reactivity of chemical constituents present in irrigation water, chemical and physical properties of soils, climatic conditions, irrigation management practices, crops to be grown, depth of watertable and its quality, impact of irrigation on drainage quality and quantity, concentration of specific ions in crops or produce for their toxicity, if any, to human beings and animals, and economic issues such as acceptable limits of water quality-induced reduction in crop productivity or produce quality.

Expanding demands for additional supplies of irrigation water has evoked keen interest among farmers to install deep bores in the Murray Irrigation districts of southern NSW. Guidelines that are currently used for issuance of licenses are ambiguous considering sustainable use of groundwater for irrigation. In addition, installation of deep bores involves considerable investments. This investigation was, thus, conducted for assessment of water quality for sustained irrigation using deep bores installed in the Murray Irrigation districts. This paper reports technical information on important water quality parameters of deep bores installed by farmers and the DLWC.

Materials and Methods

Certain physical, chemical and biological properties of the water generally determine water quality. Quality of bore or groundwater is evaluated by considering chemical parameters identified through experience and investigations on "cause and the consequence" relationships of soil and plant problems. Use of groundwater for irrigation results in transport of soluble constituents to the soil surface. Long term impact of various constituents is primarily estimated from the analytical information. This usually requires laboratory analysis in knowing concentrations of important constituents. Based on the nature and extent of potential problems caused by different quality irrigation waters, quality parameters are broadly grouped (Ayers and Westcot, 1985) into (i) salinity problems, (ii) sodium or infiltration problems, (iii) specific ion toxicities, and (iv) miscellaneous effects. Differences in soils, crops, irrigation practices, groundwater features, management and local climate play an important role in the efficient and sustainable use of different quality water supplies. Details related to hydrogeology, sampling sites, collection and analysis of deep bore waters for this study are presented below.

Regional Hydrogeology of Murray Irrigation Districts

Murray Irrigation Districts are located in the eastern and central eastern part of the Murray Geological Basin which was filled by a sequence of sediments started in early Tertiary geological period. There are 3 main hydrogeological units in the study area. They are; the Olney Formation (part of the Renmark Group), the Calivil Formation, and the Shepparton Formation.

The Olney Formation overlies the basement rock at a depth of about 140 to 350m below the ground surface. It consists of sand and gravel layers up to 40 m thick and inter-bedded with clay and silt layers that are carbonaceous. There can be 3 layers of aquifers in this formation known as Upper, Middle and Lower Renmark aquifers. In the Berriquin and Denimein Irrigation Districts, irrigators pump water from Upper and Middle aquifers to a depth of about 300m. Groundwater salinities can be up to 3000 μS/m. In the western Irrigation Districts of Wakool and Deniboota, the use of Olney (Renmark) aquifer are less intensive due to the high salinity levels.

The Calivil formation (Pliocene sands) consists of sand and gravel layers inter-bedded with clay layers, overlies the Olney Formation and occupies about 70 to 140 m from the surface. Thickness of the formation is reduced towards the east. In the Berriquin Irrigation District, groundwater salinities in this aquifer is about 2500 μS/m but towards the western Irrigation Districts the salinities can be as high as 5000 μS/m. Majority of the irrigation bores tap the Calivil aquifers because of its shallow depths relative to Olney aquifers. In the Denimein Irrigation District, the available bores suggest that the Calivil aquifers are not productive but in all other Irrigation Districts they are productive to varying degrees. In some cases irrigators obtain water from both the Renmark and Olney and Calivil aquifers (mixed water) to increase the yield of bore. The Shepparton Formation represents the most recent major phase of fluvial sedimentation. The Shepparton Formation overlies the Calivil Formation (up to a depth of about 70m) consists of clay and silts inter-bedded with minor sand layers. These aquifers are not productive as far as irrigation bores are concerned except where shallow prior stream aquifers (depth < 20m) occur.

Sampling Sites

For this investigation, bores obtaining water from Calivil and Onley or Renmark Formations were considered as deep bores. Few stock and domestic deep bores were also included in the sampling network where deep irrigation bores are sparsely distributed. In addition to private irrigation bores, the DLWC deep investigation bores within Irrigation Districts were also sampled. All deep bores were categorised into 3 groups based on aquifer used for pumping water, (i) Calivil bores, (ii) Renmark bores, and (iii) the combination of Calivil and Renmark aquifers or mixed water. Although 111 sampling sites were selected initially for the study, only 85 sites could be sampled. Greater numbers of sampling sites were located in Berriquin Irrigation District basically because of the existence of more number of bores in that district. The choice of sampling sites was purely based on geographic location. From the bore data available with the DLWC, the stratigraphy and the aquifer details were determined to select respective bores. Figure 1 shows the geographic location of the private and investigation bores sampled for this investigation. Of the 85 deep bores whose water quality was evaluated in this study, 53 are the investigation bores maintained the DLWC and 32 the private ones farmers use for commercial irrigation.

Collection of Water Samples

About half litre of water was sampled from each of the selected bores after pumping out at least three bore volumes of groundwater. Sampling was conducted during late June and early July 1996. Airlift pumping was done on all DLWC investigation bores. Water samples were collected in 500 ml capacity plastic bottles and transported to the Chemistry Laboratory, Institute of Sustainable Irrigation Agriculture, Tatura (Victoria) for analysis. The samples were frozen on the day of sampling and the analysis was accomplished within 2 weeks of their collection.

Analysis of Water Samples

Analysis of water samples was performed for pH, electrical conductivity (EC_w), soluble sodium (Na^+), magnesium (Mg^{++}), calcium (Ca^{++}), potassium (K^+), bicarbonates (HCO^-_3), sulphates ($SO^=_4$), nitrates ($NO^=_3$), boron (B) and total hardness (TH). Estimation of chloride (Cl^-) was made indirectly by subtracting sum of given anions from the total of these four cations assuming electrically neutral state. The analytical data was further processed to derive other parameters commonly used for water quality evaluation for irrigation purposes. Important ones are Selected parameters and ratios based on interactions of dissolved cations and anions are calculated using their concentrations in irrigation water. These are sodium adsorption ratio (SAR), adjusted SAR, residual sodium carbonate (RSC) or alkalinity, soluble sodium percentage [SSP = soluble Na/sum of all the soluble cations x 100, total cations, sum of divalent cations, sum of monovalent cations, and some ratios such as $Cl^- : SO^=_4$, $Ca^{++} : Mg^{++}$ or $Mg^{++} : Ca^{++}$, cations : anions etc.

Climatic Considerations

Climate of a region is important in managing salinity problems of soils and irrigation water for sustainable agriculture. Principal meteorological parameters as recorded at Deniliquin for the last 30 years are presented in Fig 2. This indicates unimodal pattern of mean monthly rainfall. Average annual rainfall makes it a representative of most semi-arid regions on the earth and comparable to such regions elsewhere for best use of the research experiences related to the problems associated with irrigation water quality and soil salinity. Such regions in the world and Australia are reported (Greenland, 1977) to offer considerable promise for development as centres of major food producing industries. Mean monthly rainfall is always less than the loss through evaporation. Soil moisture deficit prevails during whole of the year prompting movement of salts into the root zone with upward movement of soil water due to the capillary action. Magnitude of soil water deficit during the winter season (May-August) is relatively less. Climatic considerations of the MID indicate that amount of rainfall is low to cause significant leaching. Thus, development of shallow watertable may cause salinisation of the root zone. Hence, special management and amelioration due to rainfall not enough for leaching salts. Thus, salinity and sodicity problems, depending upon the nature and concentration of soluble salts either in soils or irrigation water, may prevail due to insufficient annual rainfall and limited leaching from root zone of most crops.

RESULTS AND DISCUSSION

Use of deep bore water supplies for irrigation either alone or in combination with good quality channel water must be sustainable considering various crop and soil management options including irrigation practices. This largely depends on their water quality and suitability for irrigation within guidelines appropriate for local soils and agricultural systems. Guidelines are primarily quantitative arrangement of various water quality parameters in relation to their potential impact on productivity and soil health. Several guidelines related to irrigated agriculture are documented in the literature (US Salinity Laboratory Staff 1954; Doneen, 1954; Kanwar, 1961; Bhumbla et al. 1971; University of California Committee of Consultants, 1971; FAO/UNESCO, 1973; Ayers & Westcot, 1985; ANZECC, 1992). But none of them is universal. This is due to a wide variability in the complex relationships between climate, soils, plants, irrigation practices, water quality, groundwater features and the management levels. Analytical results on important constituents and relevant parametric factors were considered for assessment of deep bore water quality or potential problems on soils and crops associated with their long term irrigation use. Assessment involved whole range of groundwater qualities regardless ownership, aquifer, and geographic locations of the deep bores.

Potential Salinity Problems

The electrical conductivity (EC) of deep bore waters ranged between 0.13 and 51.5 dS/m. Only [illegible]% of bore waters had EC less than 0.7 dS/m considered good whereas 56% were marginal (0.7-3.0 dS/m) and the rest poor (>3.0 dS/m) due to their potential salinity problems if used for sustained irrigation. Total concentration of cations (Ca, Mg, Na, & K) or anions was highly related to EC of water samples (Fig. 3). Median of numerical ratio of EC (µS/m) to sum of cations (me/L) was relatively low (70.6) indicating presence of bicarbonate or sulphate. Median of the ratios of soluble salts (mg/L) to EC (µS/m) was 0.64 but the median of the ratios of EC (µS/m) to total cations (me/L) was 136.1. These values validate the chemical analyses of bore waters for their chemical composition similar to waters observed elsewhere. Coefficient for conversion of EC (dS/m) to cations or anions in me/L for these water was 9.4 against 10 (US Salinity Laboratory Staff, 1954) for water with EC range of 0.1-5.0 dS/m. This fluctuation was partly due to high EC of many bore waters.

Differences in EC of investigation and private bores were significant. Private bores had considerably lower EC than the investigation bores. Mean and median EC of private bores were 1.64 and 1.32 dS/m (Table 1) against 6.39 and 2.86 dS/m for the investigation ones respectively. Range of EC variation in investigation bores was very large, from as low as 0.13 dS/m (surface or fresh water EC) to 51.5 dS/m (seawater EC). There was no consistent trend in EC differences in bores that pump water from either Calivil and Renmark formations or their combination. Median of private bores pumping water from Calivil, Renmark, and their combination was 1.15, 1.40, and 1.32 dS/m respectively (Table 2). Comparative values for investigation bores mining water from the Calivil and Renmark aquifer were 3.10 and 2.88 dS/m. Most bore waters (66%) regardless their ownership and aquifer had EC to vary between 0.13 and 3.0 dS/m. There was no trend in EC variation and geographic location of bores. Relatively lower water quality of more bores in the western district was notable. Because of no specific difference in EC and SAR of water pumped from either Calivil or Renmark and their combination, all the water samples were treated regardless the aquifer and geographic differences for assessment of irrigation quality.

In addition to potentially marginal or high saline problems, other parameters of water quality namely SAR and adjusted SAR showed saline-sodic nature of most deep bore waters. Most of these waters were alkaline in reaction and pH of about 93% bore waters ranged between 7.5-8.5, considered conducive for precipitation of soluble calcium in the presence of bicarbonate ions. Regardless the EC, only 11% bores pump water with adjusted SAR less than 3 whereas 46% were found having values between 3-6 and the remainder more than 6. The SAR of bores exploiting Calivil aquifer was relatively lower than that of the Renmark (Fig. 4). The relationship between EC and SAR of bore water (Fig. 5) indicates an interesting pattern. In this relationship, seven waters of EC more than 10 dS/m were excluded due to high EC viewed for their unsuitability for irrigation. The SAR of bore water tends to increase with an increase in EC. Thus, high EC of high SAR water could help in infiltration if used for irrigation. The predicability (54%) of SAR based on EC of a wide variety of water was reasonable with the following relationship;

$$SAR = 4.02 + 0.90\ EC_w\ (dS/m).\ R^2 = 0.54^{**}\ (N = 78)$$

Deep bore waters with SAR 0-5 had EC variation between 0.13 to little more than 3.0 dS/m and the waters with SAR of 5-10 had EC range of 1 to 8 dS/m. However, waters with SAR greater than 10 (including those not included in this relationship) had a large EC variation (about 3.0 to 51.5 dS/m). According to the classification proposed by US Salinity Laboratory Staff (1954), nine of the 85 bore water samples were tested medium (0.25-0.75 dS/m EC) and only one low (EC $\leq$ 0.25 dS/m) in potential salinity problems. About 44% of samples had high EC (0.75-2.25 dS/m) whereas 46% were very high (EC $\geq$ 2.25 dS/m) in salinity. A comparison of EC and its variation relative to important irrigation water quality classifications (US Salinity Laboratory Staff, 1954; Carter, 1968; Rhoades, 1972; Ayers and Tanji, 1981; Ayers and Westcot, 1985), most deep bore waters are marginal irrigation quality wise. These classification schemes provide details of the salinity problem for given waters based on the broad generalisations regarding growth and productivity of various crops, irrigation & drainage management, and climate.

Several of the proposed classifications are based only on water quality parameters. Because in real situations, irrigation water properties are modified greatly as a result of their interaction with other factors (soil, climate, irrigation practices, management, crops etc). Without these considerations, some of the findings may not be valid for their application to other areas notwithstanding their importance in broad understanding of salinity problems. It has been usually considered that evaluation of the irrigation water for their suitability must consider achievable leaching, leaching efficiency, relative crop tolerance to irrigation water or soil salinity, and the level of soil water salinity that will occur as a result of using a particular type of irrigation water.

Mostly the absolute salt content is an index of irrigation water salinity. It has also been observed that the nature and relative amounts of various ions also modify salinity problems, if any. In deep bore waters, chloride and bicarbonate of sodium predominate that also define their saline-sodic nature. Mean and median concentration of chloride was 32.2 and 9.8 me/L against 3.5 and 0.97 me/L for sulphate respectively indicating chloride about 10 times more than sulphate. Because of the differences in solubilities of chloride and sulphate salts, plant roots exposed to chloride rich irrigation water encounter high salinity hazard than sulphate rich water. Formation of ionic pairs and precipitation is more with sulphate rich water Yields obtained with the chloride salinity of 3.2 dS/m were comparable to 6.8 dS/m salinity of sulphate ions (Manchanda *et al.*, 1981).

Potential Sodicity or Infiltration Problems

Infiltration is simply the water entry into the soil and differs from permeability (percolation of water through the soil). Diagnosis of potential infiltration problems of deep bore waters needs integration of their EC and relative concentrations of important cations (Ca, Mg and Na). Concentration of bicarbonates or residual alkalinity is another pertinent parameter. Sodicity problems generally impair the rate of irrigation water entry into the soil, causing inadequate supply of water to restore soil water deficit or availability within the root zone. Reductions in water entry or infiltration problems result in due to only few cm of top-soil. Thus, crop water requirements are affected by reduced water supply to plants just similar to salinity problem but due to different reason (decrease in amount of water addition to the root zone). Infiltration or permeability problems associated with irrigation waters are modified by important soil properties (nature and amount of clay minerals, organic matter, relative composition of exchangeable cations, soil EC etc). Sodium problem of irrigation water is related to both salinity and sodium concentration relative to calcium and magnesium as expressed by the concept of SAR. The infiltration rate normally increases with an increase in salinity and decreases with either decreasing salinity or increasing SAR. Hence, considering both EC and SAR together for diagnosis of potential infiltration problems of deep bore waters.

Data (Table 3) on SAR of deep bore waters indicate comparatively greater variation in investigation than private bores. Relative SAR values of investigation bore waters were also higher than those of private ones. Mean and median SAR of private bores with Calivil aquifer was lower than the Renmark aquifer. However, SAR variation across all the bores was not consistent in terms of their aquifer or geographic location. Hence, the need to evaluate different bores independently. As mentioned earlier, increase in EC of high SAR waters helps in improving their infiltration. However, this depends upon an appropriate match between the EC and SAR values. Calculation of adjusted SAR considering Ca and Mg ions together for computing p(Ca), Ca_X component did not show great variation in the SAR. But presence of bicarbonate in all the bore waters indicates concern for precipitation of calcium and or magnesium carbonates in soils. Calcium and magnesium together accounted for nearly 26% of the total cations in bore waters but comparative concentration of dissolved calcium was much less than magnesium and sodium (Table 4). Concentration of calcium was less than one me/L in nearly half of the waters sampled for this study and 72% of bore waters were characterised with soluble calcium characterised by only one-tenth of the total soluble cations unlike SSP (> 60%). However, calcium and magnesium together made up 25-30% of the soluble cations and their concentrations in bore waters showed significant relationship with the sum of Ca, Mg, Na, and K ions (Fig. 6). Similar relationship was also observed for Na+K.

Presence of bicarbonates in all the bore waters also adds to their potential sodicity problems. Its quantitative estimation using RSC concept (Wilcox, 1948; US Salinity Laboratory Staff, 1954) is reported unsatisfactory (Suarez, 1981). But consistent occurrence of bicarbonate in deep bore water deteriorates their quality and poses potential sodic hazard. The other parameters relevant to the sodium problem waters are shown in Table 5. The Mg: Ca ratio of private bore waters with a mean and median values of 4.0 and 3.6 were relatively greater than the corresponding 3.7 and 3.4 for investigation bores. This means calcium deficiency or imbalance in the cationic composition. The concentration of total cations was also less in the private than investigation bores. Difference in Mg/Ca ratios probably owe to relatively greater electrolyte concentration in investigation than private bores.

Water quality parameters aimed at evaluation of deep bore waters for their potential infiltration problems indicate that most of these waters are marginal and their use for sustained irrigation will sooner or later sodicate the soils. Rate of soil sodification will be more in rice soils irrigated with these waters alone. Due to availability of limited research conducted in the Murray Irrigation districts, water quality related investigations should be conducted to develop predictive modelling. This will help in making decisions based on the experience gained under climate and soil conditions of the Murray Valley and can later be extrapolated to different situations.

Mg/Ca Ratio: All deep bore water have Mg/Ca ratio greater than 1.0. An increase in proportion of Mg relative to Ca enhances rate of sodification in soils and cause dispersion of clay particles (Paliwal and Gandhi, 1976; Yadav and Girdhar, 1981). Deleterious impact of irrigation water of high Mg/Ca ratios is not fully understood. This ratio becomes critical for optimum plant growth when it exceeds 3. This is because of its influence in inducing dispersion and resultant decrease in infiltration and permeability of soils. In addition, it may influence calcium nutrition of crops. Higher Mg/Ca ratio is expected to affect uptake and translocation of Ca due to its antagonistic effect or competition for absorption sites on plant roots. Water rich in Mg also induce greater uptake of phosphorus and may further reduce translocation of Ca in plants. Presence of Ca in appropriate amount in the root zone is generally known to ameliorate toxicities of other ions due to its higher utilisation. Limited research evidences suggest that the ratio of Ca^{++}/M^{n+} (sum total of all cations) of 0.1-0.15 or greater is needed for healthy growth of roots.

This ratio in majority of deep bore water was found less than 0.10 and it was in between 0.10 and 0.15 for some waters. These reflect the demand for application of calcium to soils from the external sources if the soils in question are deficient in calcium. The soils that do not contain $CaCO_3$ or other adequate sources of Ca-silicates, an ESP of 5 and irrigation water salinity of 0.3 dS/m becomes critical diagnostic limits of sodicity and salinity hazards. The insufficiency of Ca^{++} to total cationic ratio was reported (Carter and Webster, 1979) to reduce barley and sugar beet yields due to increased Mg/Ca ratio in the soil solution. But there is need for systematic investigations to evaluate critical Mg/Ca ratios in irrigation water especially the deep bore waters that contain Mg 2-4 times more than Ca. However, limited research information indicate that if Mg/Ca ratio exceeds 2.5-3.0, special attention must be given to this imbalance. Solution lies in applying gypsum if such waters are used for irrigating vertisols or clay soils. As exchangeable cations in the soils strongly influence their physical properties, role of Mg is conflicting. Being more hydrated than calcium (Norrish 1954), it produces thicker diffuse layer (Shainberg and Kemper, 1967). Combination of Na with Mg is thus expected to deteriorate the soil more than with Ca at similar SAR or ESP of soils.

Continuous use of most deep bore waters for irrigation without special management will sodicate the soils. The rate and extent of sodification will depend upon important quality characteristics of irrigation water, quantity of water used, soil types etc. An ESP of 6 is considered a critical limit for differentiating sodic from non-sodic soils (McIntyre, 1979) in Australia unlike 15 elsewhere. As mentioned earlier, sodium problems are indicative of high SAR water usage. Critical value of SAR thus depends on EC of irrigation water. About 9% of deep bore waters indicate safe for potential salinity problem but there were only 5% bores with SAR less than 3 considered safe. Most (79%) bores showed SAR values ranged between 6-12. Integration of the relationship between SAR and EC of deep bore waters (ignoring 7 bores with EC > 12 dS/m) is shown in Fig.7 within the context of most widely used water quality guidelines (Ayers and Westcot, 1985). This indicates that considerable number of the deep bores have water that can cause slight to moderate reduction in the infiltration rate or pose sodium problem. Similarly, many of them pose no reduction in the infiltration rate but their EC values are high enough to cause salinisation of soils by sustained irrigation.

The management practices which rely on chemical approach require changing the chemistry of either soil or irrigation water for favourable infiltration rates. This is usually done by addition of an appropriate chemical amendment (industrial or mined gypsum) either to the soil or water. In some cases, this can also be achieved by mixing good and poor quality water supplies to lessen the potential problems. The physical approach involves cultural practices that help improve or maintain favourable infiltration rates during the irrigation or rainfall events. Choice of the suitable management options also depends on the local conditions such as soil types, available water supplies and their qualities, drainage etc. For example, the comparatively low infiltration rates of the heavy textured clayey soils must be distinguished from the water quality reduced infiltration. The pertinent management options that can help improve quality of deep bore water for supplemental or sustained irrigational uses are described below.

Problems of Specific Ion toxicities

These differ from salinity problem in limiting water availability to plants. Specific ion toxicities occur due to accumulation of certain ions to toxic levels within plants. Damage to plants in terms of growth and productivity depends upon other factors such as relative crop sensitivity and water use, growth stage or time, accumulated concentration and the plant component in which such ions accumulate. Most commonly recognised toxic ions of irrigation water are chloride, sodium and boron. They can cause toxicity either individually or in combination. In general, field crops are less sensitive to the usual concentrations of these specific ions in irrigation water. But tree crops or woody plants are usually more sensitive.

Chloride: This causes most commonly observed toxicity by its presence in irrigation water. It is highly mobile in the soil-plant system. Being an anion, chloride ions are not adsorbed readily in soils. Its high solubility makes it favourable for absorption by plants along with water and its accumulations in foliage occur as a result of transpiration stream. Leaf burning or drying up of leave tissues are common toxicity symptoms which first occur at the leaf tips. Severe toxicity results in leaf drop (defoliation) or necrosis. Its toxicity symptoms in sensitive crops occur at concentration of 3-10 g/kg on dry basis. Tree crops are generally more sensitive. Its accumulation is generally diagnosed through chemical analysis of either leaf blades or petioles. Chloride toxicity is associated not only with water quality but also with the soil chloride levels and ability of plants to exclude chloride. Absorption of chloride can also occur through leaves during overhead sprinkler irrigation. Tolerance of crops to chloride is not well investigated as to the irrigation water or soil salinity. Chloride concentration of deep bore water was not determined analytically. It has been rather estimated indirectly. Unlike Na, majority (79%) of bore water had Cl^- concentration between 6-12 me/L.

Sodium: Unlike chloride, Na toxicity is difficult to diagnose. This occurs due to the use of relatively high concentrations of Na or SAR in irrigation water. Morphological symptoms of its toxic accumulation are characterised by leaf burn, scorching and necrosis along outside edges of leaves as compared to initial start at the extreme steps in case of chloride. Symptoms of toxicity first appear on outer edges of older leaves and extend inward between veins toward the leaf centre. Sodium concentration of more than 2.5-5.0 g/kg (dry basis) in leaf tissue of most tree crops is normally an indicative of its toxicity. Most field crops can tolerate relatively much higher concentration of Na in their foliage. Leaf analysis of damaged and undamaged leaf tissues along with additional information on soils and plants analyses help diagnose Na toxicity correctly. Sodium toxicity is modified by the availability of calcium in soils. In some annual crops, Na toxicity is associated with calcium deficiency. Several field crops show Na toxicity. The SAR is a common irrigation water characteristic used for diagnosis of potential Na toxicity hazard, if any. Sodium toxicity could occur due to high Na or SAR of irrigation water and may either occur or modified by poor water infiltration of such irrigation water. Only sensitive crops such as few trees and some perennial crops show toxic Na accumulation in the absence of any infiltration problem. Only one-fifth of the sampled bores had Na less than 5 me/L whereas 11% of bores exceed 50 me/L. The respective Na concentration ranged 5-15 and 15-50 me/L in 49% and 20% of the remainder bores. Considerations of Na relative to Ca and Mg leave only 5% of these bores with SAR less than 3 indicating low concentrations of the divalent cations. Sodium toxicity has been reported (Bernstein 1962) to occur in sensitive fruit crops (avocadoes, stone fruits and citrus) at SAR as low as 5.5.

Boron: Groundwater in arid and semi-arid areas sometimes contains boron in toxic amounts. Boron, unlike sodium, is an essential element for plant growth. Boron concentration in surface water is rarely toxic. The concentration of B less than 0.7 mg/L is not toxic. But its concentration of 3.0 mg/L or more severely restrict the use of irrigation water. Its excessive accumulation in plant causes leaf necrosis. Being immobile in plants, its toxicity symptoms (yellowing, spotting, and leaf drying at tips and edges) first appear on older leave. Boron toxicity may also shorten leaves, thus, reduce leaf area and photosynthesis. Boron displacement in soils is not an easy option. Boron needs 3-4 times more water for leaching to reduce equivalent levels of salinity and boron concentrations. Because of its high immobility in soils, boron continues to accumulate even if its concentration in irrigation water is low. Its toxicity in legumes was reduced by application of fertiliser phosphorus. Toxicity symptoms in most crops occur when its concentrations in leaf blades exceed 250-300 mg/kg (dry basis). But leaf blades of many boron-sensitive crops do not accumulate B in their leaf blade. Boron concentration in deep bore waters were less than 0.1 mg/L except two investigation bores with 0.5 and 0.9 mg/L B concentration.

Miscellaneous Problems

Nitrate Nitrogen: Nitrate and ammonium are the two predominant forms of nitrogen. Nitrate concentration in most surface and groundwater is less than 5 mg/L. Some of the abnormal groundwater contains about 10 times more nitrates. Drainage water of applied irrigation also contains nitrates due to deep leaching of fertiliser and soil nitrogen. Waste water of domestic and food processing sources are high in nitrogen varying between 10-50 mg/L. Application of 1000 m^3 water with of one mg/L nitrate nitrogen contribute one kg N. Groundwater in some semi-arid regions may contain excess nitrates. Growth and yield of crops due to excessive nitrate nitrogen over-stimulates growth, delay maturity or damage quality of produce.

Most crops do not respond unless concentration exceeds 30 mg/L and nitrate nitrogen less than 5 mg/L causes no problem. But this concentration may stimulate unwanted growth of aquatic plants and algae. Unwanted growth of algae can occur when temperature, sunlight and other nutrients especially phosphorus is available. This results in damages such as plugged valves, pipelines and sprinklers requiring either mechanical control (screens, filters etc) or chemical materials (copper sulphate). All bore waters contained less than 0.5 mg/L of nitrate nitrogen except one with 2.67 mg/L in Berriquin district.

pH: Generally, crops and soils can tolerate irrigation water pH range of 4.5-9.0. But irrigation water with extreme pH values can cause potential indirect problems. Dissolution of aluminium, manganese or heavy metals can occur at pH below 4.8. This may produce toxic concentrations if acid soils are irrigated with such water. Water with pH greater than 8.4 indicate presence of sodium, carbonate and bicarbonate. These ions can cause sodium problems in soils if irrigation waters of given pH used continuously. The pH values of deep bore waters were within a pH range of 6.4-8.5. About 7% of samples have pH less than 7.0 whereas most (78%) bores were alkaline (pH 7.0-8.0) in reaction. For 15% of bores, pH varied between 8.0 and 8.5. Estimation of the saturation index was also calculated for deep bore water using their actual and calculated pH. About one quarter of bores showed tendency for precipitation of $CaCO_3$.

Total Hardness: The concentration of all the metallic cations, except alkali metals, present in water is called hardness of water. It is a measure of the calcium and magnesium ions in water and is frequently expressed as mg/L CaCO3 equivalent. Total hardness is made up of permanent and temporary hardness. Hardness is mainly a problem in ground water (NHMRC, 1987). Such water can choke household pipes by depositing scales and increase soap consumption for efficient washing. There is no evidence of adverse health effects of high levels of calcium and magnesium (WHO 1984). The salts responsible for hardness are not harmful to human being. But excessive presence of magnesium may impair organoleptic properties of water. Total, temporary, permanent, carbonate and non-carbonate are the various types of water hardness. For irrigation purposes, it is not very important. Just 7% bore waters had TH less than 50 mg/L (Table 6) while 52% were greater than 200 mg/L and it was 50-200 mg/L for the rest.

SUMMARY

Water quality assessment involved analytical results of waters sampled from 85 deep bores within the Murray Irrigation districts. Of 85 deep bores, 30 are private and used for commercial irrigation whereas remainder 55 are the investigation bores installed by the DLWC. These bores use either Calivil or Upper Renmark (Onley) or both aquifers. Variation in different water quality parameters due to aquifers was not significant. There was no definite relationship between water quality of deep bores and their geographic positions. Quality of private bores was better than investigation bores. Except a few bores, quality of most for irrigation was either marginal or severe for both salinity and sodicity (water infiltration problems). This is due to their moderate to high salt content, SAR, residual alkalinity or consistent presence of bicarbonates, and unfavourable Mg/Ca ratio or low calcium content.

Salinity problem: The electrical conductivity of deep bore water ranged between 0.13 and 51.5 dS/m. The median value of the ratio of soluble salts measured in mg/L or ppm to EC (μS/m) was 0.64. These values validate the chemical analyses of bore water for their important constituents. The coefficient for conversion of EC (dS/m) to cations or anions in me/L for this water was found to be 9.4 against 10 (US Salinity Laboratory Staff 1954) for water with EC range of 0.1-5.0 dS/m. The private bores had considerably lower EC than the investigation bores. The EC trend due to different aquifers was not remarkable. The mean and median values of EC of private bores were 1.64 and 1.32 dS/m respectively. The average salinity of the investigation bores was 6.39 dS/m against the median value of 2.86. Only about 13% bores has EC within safe limit of salinity problem whereas 44% were marginal. Conjunctive use of deep bore and channel waters could be the possible solution in lessening potential salinity hazards. This depends upon SAR and other factors affecting feasibility of conjunctive use. Most of past researches favour alternate mode of conjunctive use than mixing poor and good quality waters.

Sodium or Infiltration Problems: Considering guidelines and different parameters, most deep bore waters pose potential sodicity problems. The SAR trend across all the bores was not consistent in terms of aquifer type. The high SAR of irrigation waters require appropriate salinity level for their optimum infiltration. Nearly 13% of deep bore waters pose no salinity hazard whereas only 5% of the given bores have SAR less than 3 considered safe. A significant majority (79%) of them has SAR values ranged between 6-12. This again varies with the soil types. The consistent presence of bicarbonates in all the bore waters indicates concern for precipitation of calcium and or magnesium in soils. Because calcium and magnesium together accounted for nearly 26% of the total cations in bore waters. The comparative concentration of dissolved Ca was not only lower than Mg but it was less than 2.0 me/L considered appropriate for nutritional requirements of crops.

Presence of bicarbonates in all of the bore waters makes it saline-sodic. Its quantitative estimation using RSC concept was not fully satisfactory. But consistent occurrence of bicarbonates in deep bore waters lowers their quality and poses potential sodic hazard. The Mg : Ca ratio of private bore waters with a mean and median values of 4.0 and 3.6 were relatively greater than the corresponding 3.7 and 3.4 for the investigation bores respectively. This again implies calcium deficiency or imbalance in the cationic composition of these waters. This difference in Mg : Ca ratio between private and investigation bores possibly owes to relatively greater electrolyte concentration of the latter. These parameters necessitate evaluation of deep bore water for their potential problems related to infiltration of irrigation water or sodification of the soils used for either rice based or upland crops in the Murray Irrigation districts. Properties of soils to be irrigated with such water are very important. Due to availability of limited research information from investigations conducted in the Murray districts, water quality related investigations should be conducted. This will help in making decisions based on the experience gained under climate and soil conditions of the Murray Valley. These can later be extrapolated to different locations and climatic conditions.

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Table 1. Mean, median, SD, and range of EC (dS/m) variation of private and investigation bores.

Parameter	Private Bores		Investigation Bores	
	EC, dS/m	SAR	EC, dS/m	SAR
Mean	1.64±0.69	5.1±1.6	6.39±9.86	9.3±5.6
Median	1.32	4.7	2.86	8.4
Range of variation	0.26-8.24	2.6-8.7	0.13-51.5	0-25.2
Number of bores	32	32	53	53

Table 2. Mean, median, SD, and range of EC (dS/m) variation in private and investigation bores.

Aquifer	Private Bores				Investigation Bores			
	Mean	Median	Range of variation	N	Mean	Median	Range of variation	N
Calivil	1.60	1.15	0.26-4.79	10	6.70	3.10	0.13-51.50	28
Renmark	1.56	1.40	0.81-2.69	12	6.22	2.88	0.68-45.80	26
Calivil & Renmark	1.92	1.32	0.51-8.24	8	1.96	-	-	1

Table 3. Mean, median, SD, and range of SAR variation in private and investigation bores.

Aquifer	Private Bores				Investigation Bores			
	Mean±SD	Median	Range of variation	N	Mean±SD	Median	Range of variation	N
Calivil	4.4±1.2	4.4	2.6-6.4	10	8.9±6.1	6.7	1.1-22.2	28
Renmark	5.7±1.7	5.6	3.7-8.7	12	10.2±5.2	9.3	0-25.2	26
Calivil & Renmark	5.4±1.7	4.8	3.8-8.0	8	5.5	-	-	1

Table 4. Relative proportion (%) of dissolved sodium, magnesium, calcium and potassium in private and investigation bores.

Bore ownership	Range	Mean	Median	SD
		Private bores		
Sodium	47.1-82.4	66.7	67.5	10.0
Magnesium	14.0-39.5	25.0	23.9	6.9
Calcium	1.1-16.2	7.6	7.5	3.8
Potassium	0-1.9	0.7	0.6	0.6
		Investigation bores		
Sodium	0.1-84.9	62.6	61.7	13.6
Magnesium	9.3-40.0	28.5	28.9	12.7
Calcium	2.0-16.2	7.9	7.8	4.0
Potassium	0-2.4	1.0	0.7	0.8

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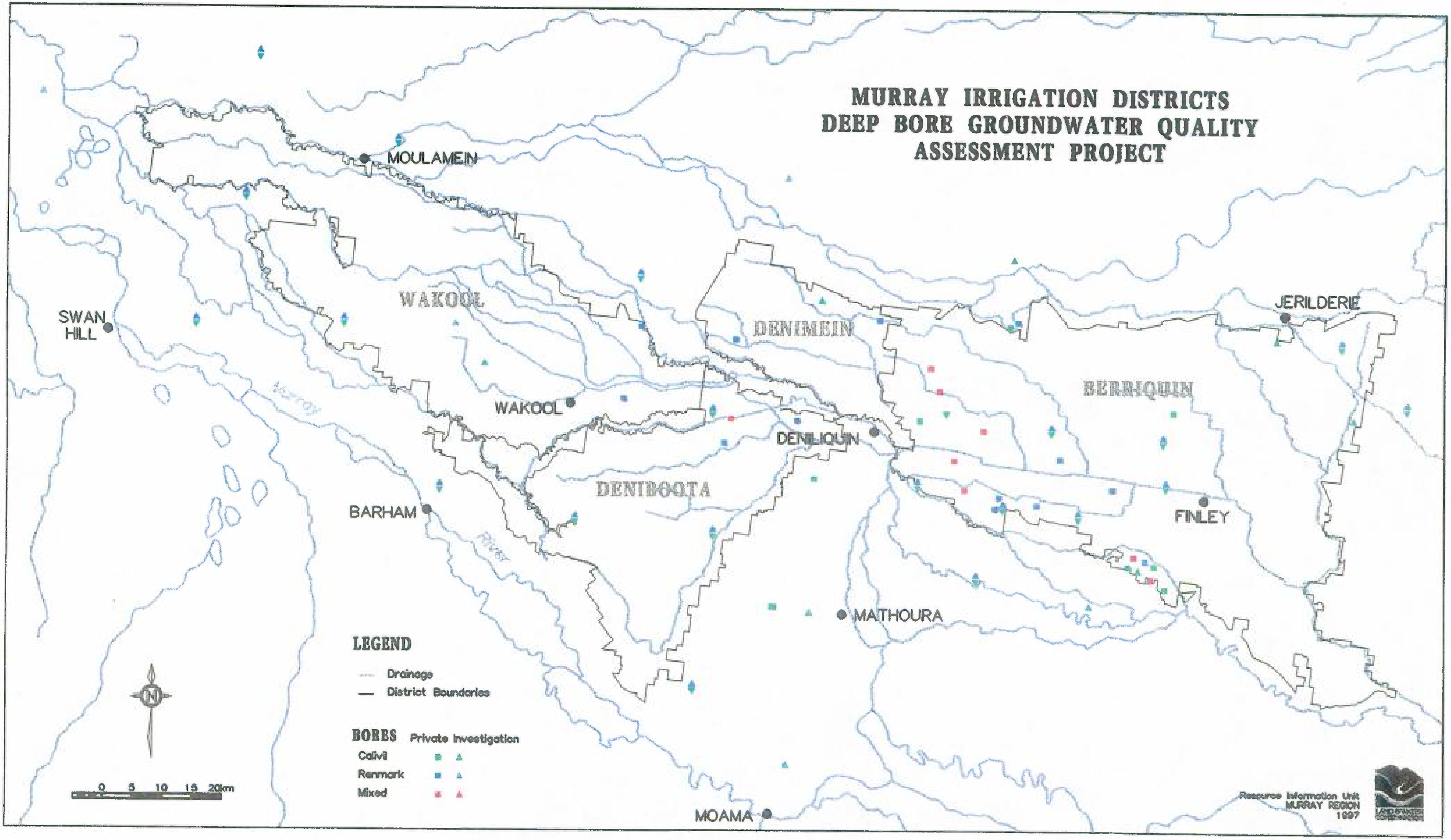
Table 5. Important statistical parameters of characteristics related to sodium hazard and their variation in private and investigation bores.

Bore ownership	Range	Mean	Median
Private			
Mg : Ca ratio	1.9-11.0	4.0	3.6
HCO3 (me/L)	1.0-3.5	2.5	2.5
Total cations (me/L)	1.7-69.2	11.3	8.5
Investigation			
Mg : Ca ratio	1.4-7.7	3.7	3.4
HCO3 (me/L)	0.5-7.3	3.7	3.3
Total cations (me/L)	0.02-508.5	54.1	22.1

Table 6. Frequency distribution (%) of deep bores with different classes of their important specific ions, pH and total hardness regardless of their ownership and aquifer type.

Parameter	Range Values of different classes and corresponding frequency			
Sodium, me/L	<5	5-15	15-50	>50
Frequency	20%	49%	20%	11%
SAR	<3	3-6	6-12	>12
Frequency	5%	39%	28%	8%
Chloride, me/L	<3	3-6	6-12	>12
Frequency	5%	6%	79%	15%
NO3-N, mg/L	<0.1	0.1-1.0	1.0-5.0	>5.0
Frequency	45%	35%	16%	5%
HCO3 (me/L)	<2.5	2.5-3.5	3.5-4.5	>4.5
Frequency	33%	33%	14%	20%
Boron, mg/L	<3	3-5	5-7	>7
Frequency	38%	47%	13%	2%
Hardness, mg/L	<50	50-100	100-200	>200
Frequency	7%	19%	22%	52%
pH	>6.5	6.5-7.5	7.0-8.0	8.0-8.5
Frequency	1%	51%	33%	15%

List of Captions for Figures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DEEP BORE GROUNDWATER QUALITY
ASSESSMENT PROJECT
MOULAMEIN
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DENIMEIN
BERRIQUIN
JERILDERIE
SWAN HILL
WAKOOL
DENILIQUIN
DENIBOOTA
FINLEY
BARHAM
MATHOURA
MOAMA
Murray
River
LEGEND
Drainage
District Boundaries
BORES Private Investigation
Calivil
Renmark
Mixed
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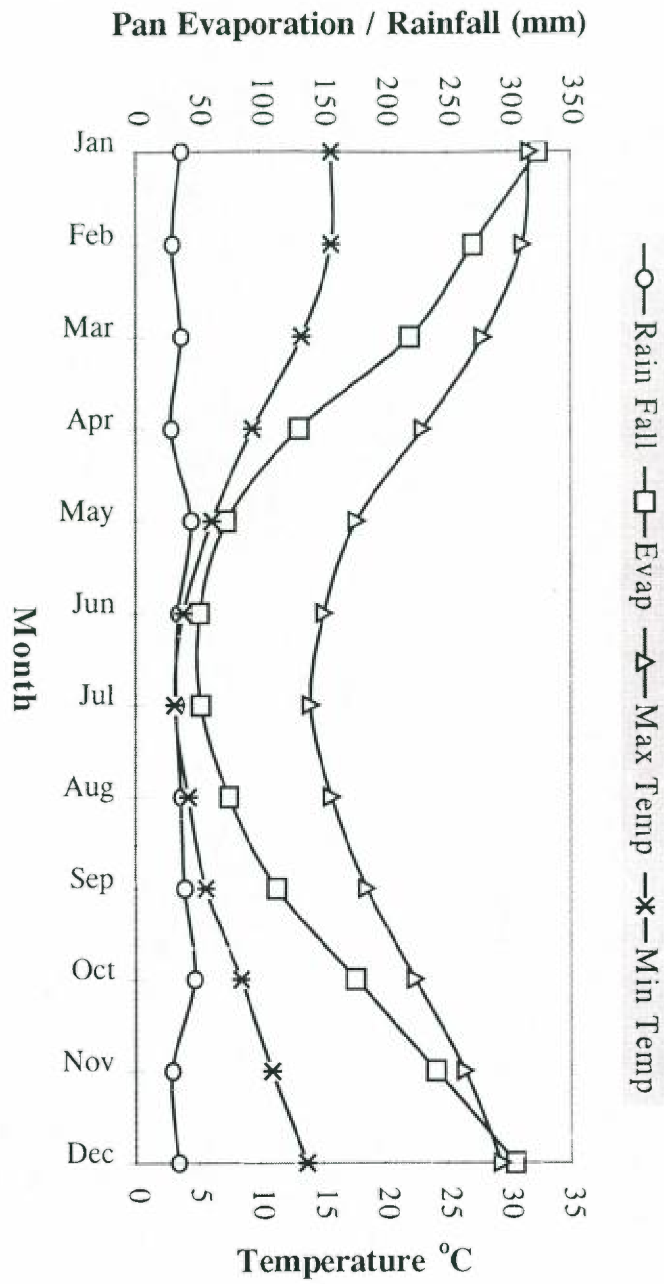
Resource Information Unit
MURRAY REGION
1997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50
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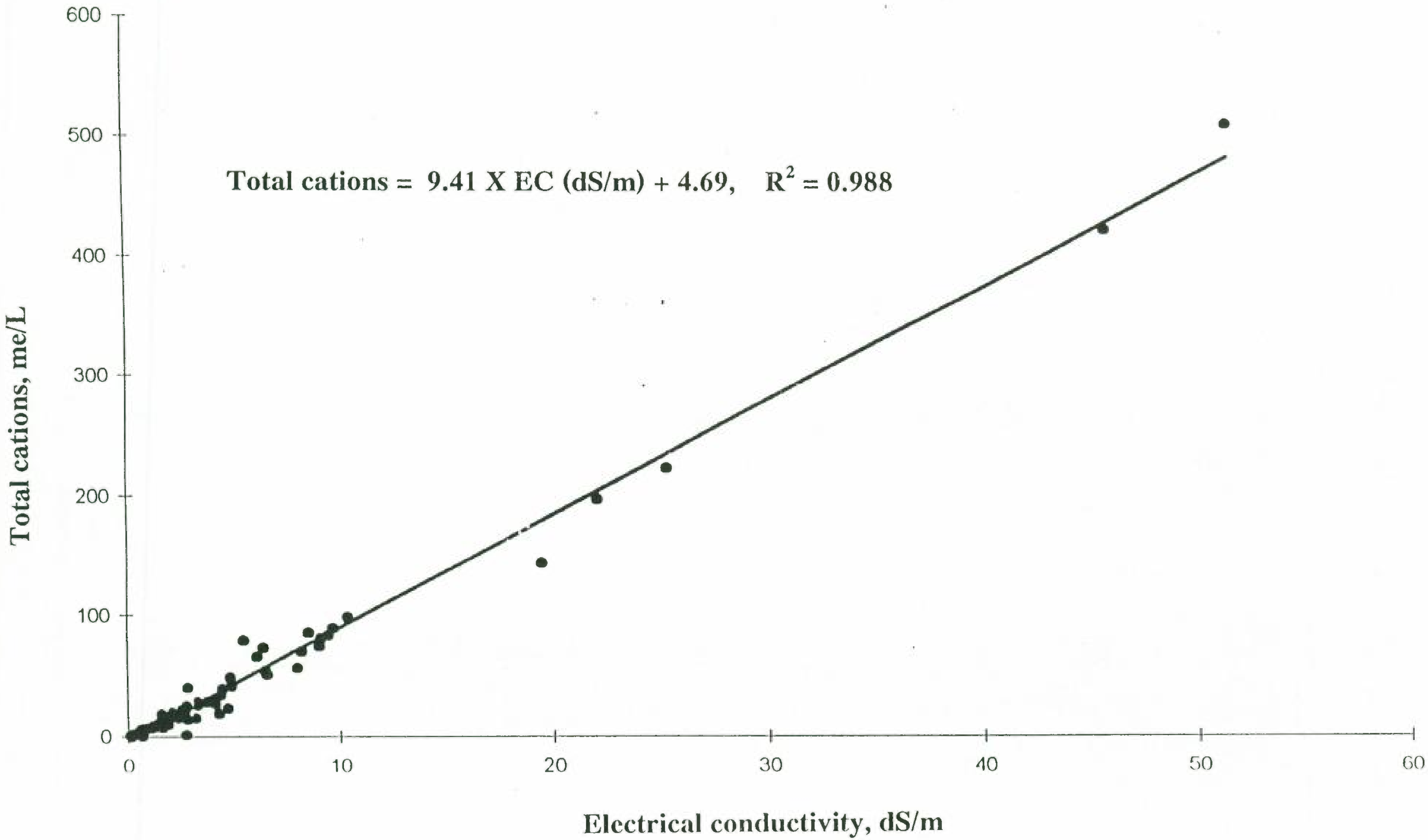
150
200
250
300
350
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Month
Rain Fall
Evap
Max Temp
Min Temp
0
5
10
15
20
25
30
35
Temperature °C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400
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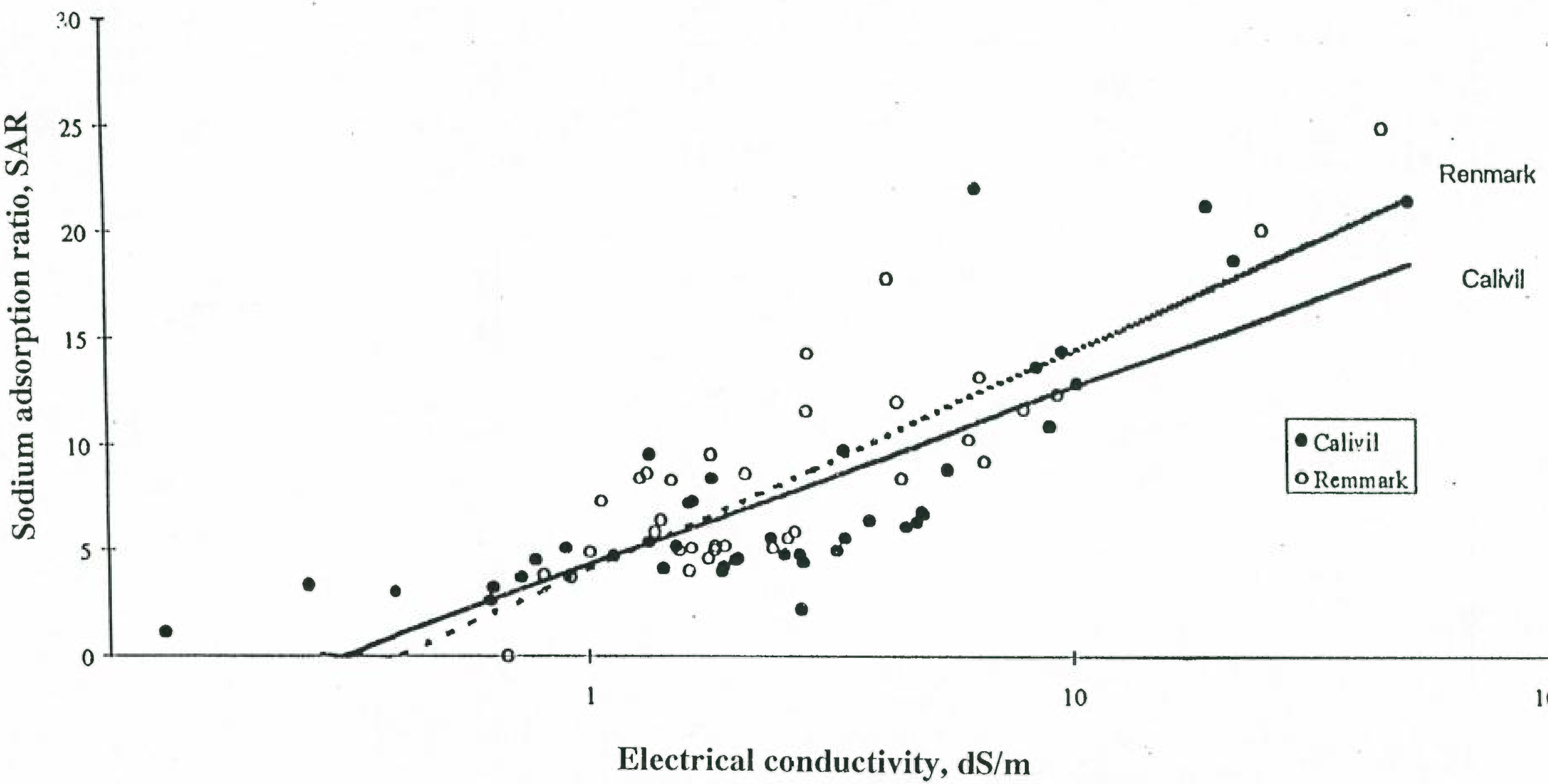
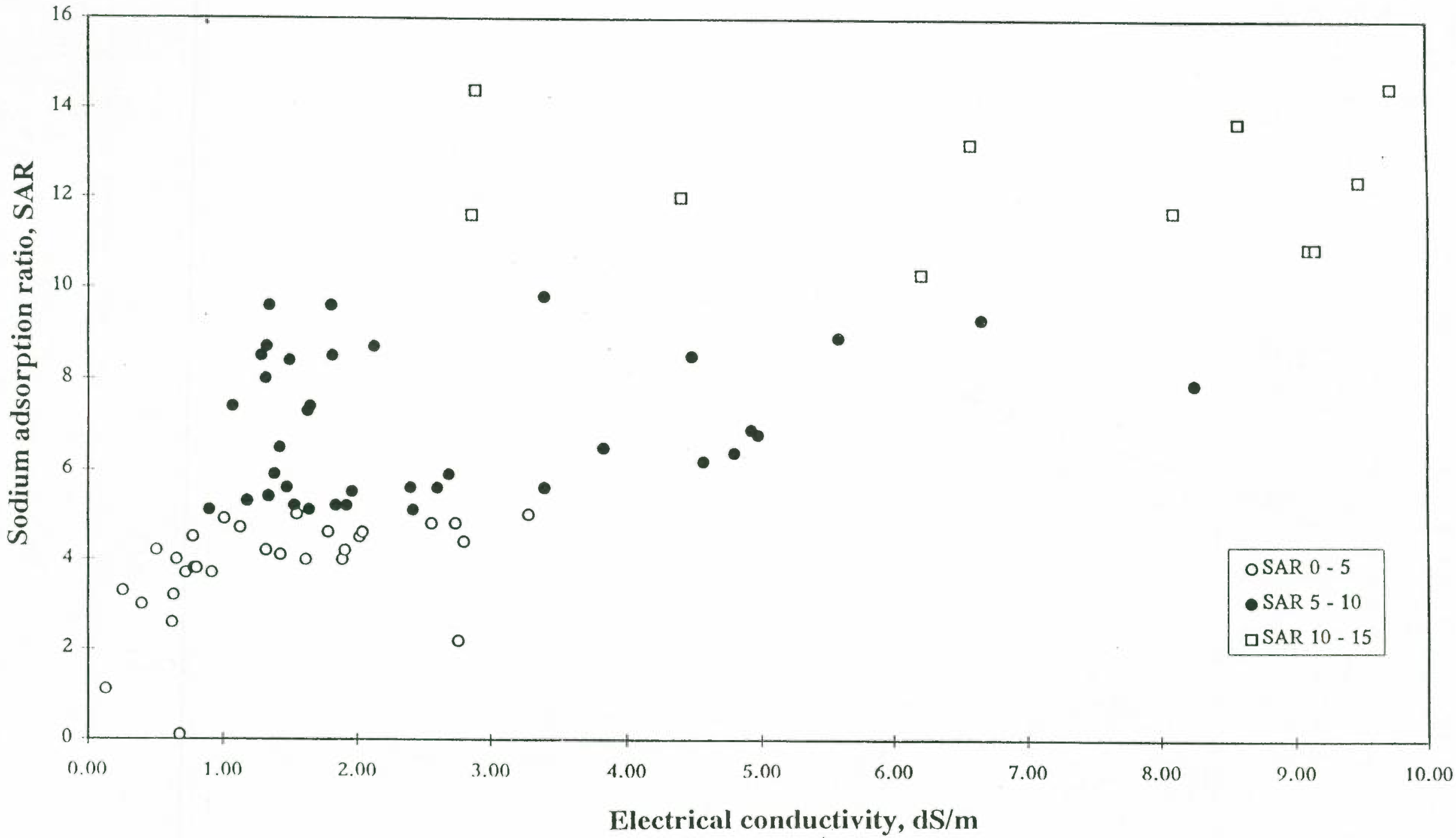
0
0
10
20
30
40
50
60
Electrical conductivity, dS/m


Sodium adsorption ratio, SAR
30
25
20
15
10
5
0
1
10
100
Electrical conductivity, dS/m
Renmark
Calivil
Calivil
Remmark


Sodium adsorption ratio, SAR
Electrical conductivity, dS/m
SAR 0 - 5
SAR 5 - 10
SAR 10 - 15
0
2
4
6
8
10
12
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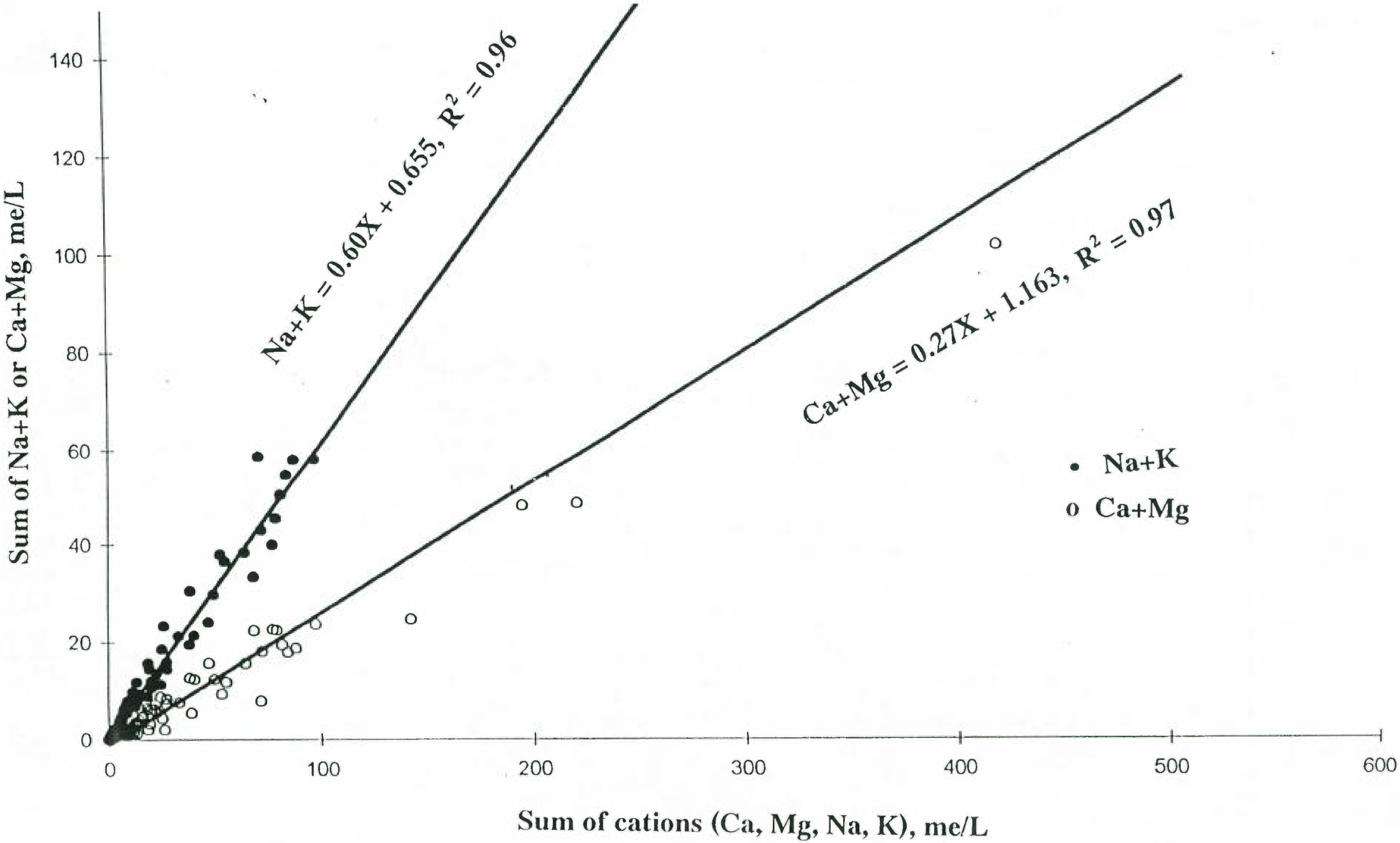
8.00
9.00
10.00


Na+K = 0.60X + 0.655, R² = 0.96
Ca+Mg = 0.27X + 1.163, R² = 0.97
• Na+K
o Ca+Mg
Sum of Na+K or Ca+Mg, me/L
Sum of cations (Ca, Mg, Na, K), me/L
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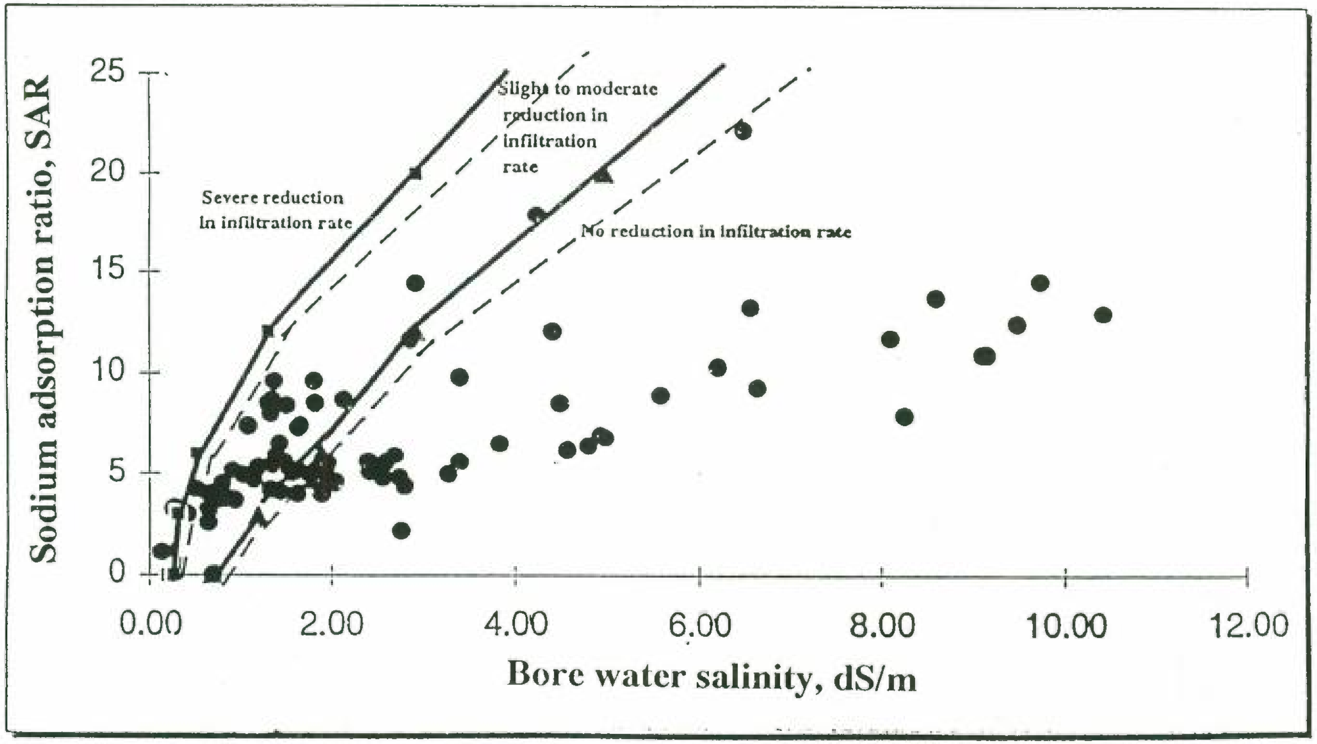
20
40
60
80
100
120
140
0
100
200
300
400
500
600


Sodium adsorption ratio, SAR
25
20
15
10
5
0
0.00
2.00
4.00
6.00
8.00
10.00
12.00
Bore water salinity, dS/m
Severe reduction in infiltration rate
Slight to moderate reduction in infiltration rate
No reduction in infiltration rate

Spatial interpolation of groundwater data and its implications for environmental management

T. Harrington[†] and A. Tiwari[‡]

Abstract

Management of the watertable is one of the most important issues facing irrigators. Effective watertable management is achieved by first understanding the factors that lead to watertable rise and fall and secondly by putting in place programmes that both monitor changes in the watertable level and actively bring about beneficial changes in that level. This paper briefly reviews the mechanisms that contribute to changes in watertable level and then considers the tools available for describing the watertable in four dimensions: three spatial dimensions and time. Orthodox methods of surface interpolation are reconsidered in light of experiences in the Coleambally Irrigation Area in New South Wales. The advantages and disadvantages of available methods are briefly summarised. The conclusion reached is that, in the absence of quantified parameters such as lateral and vertical transmissivity, no one model is necessarily preferable to another, however, the consistent use of one model will illuminate annual and seasonal changes in watertable that are, we contend, more important to the management of the watertable than a precise knowledge of the actual level.

Introduction

Environmental and therefore, economic sustainability of irrigated agriculture depends largely on effective management of the level of groundwater underlying the irrigated area. Programmes implemented to produce beneficial change in groundwater level will only be effective when reliable and timely information is available on the level of the groundwater. The Draft Land and Water Management Plan (DLWMP, 1995) developed for the Coleambally Irrigation area, contains objectives and targets that include reductions in the level of the groundwater surface and the total area affected by salination or waterlogging. Management programmes have been designed to alter agricultural practices and monitoring programmes have been implemented to measure the effect of the management programmes in terms of the objectives and targets. These programmes will be incorporated into an Environmental Management System (Kinhill, 1999) for accreditation by a third-party organisation to the ISO 14 001.

In this paper we discuss the methods used in the determination of groundwater level and suggest alternative methodologies that are more closely aligned to the objectives and targets of the DLWMP.

Determination of groundwater surface

The level of the groundwater surface depends largely on the quantity of water reaching and remaining in the shallow groundwater system. Water reaches the shallow groundwater system by two pathways: recharge from the root zone and seepage from supply channels, and leaves the shallow groundwater system by one of three pathways: capillary rise, leakage to the deep aquifer and use by trees. The residual quantity of water (i.e. inflow minus outflow) in the shallow groundwater system is referred to as 'net recharge'. Net recharge can be determined by either subtracting the total quantity of water that flows out of the root zone from the total quantity of water that flows in, or it can be indirectly estimated from changes in the areal extent of groundwater contours constructed from water level data obtained from bores. In neither case can all the parameters that influence recharge be measured. The two methods are discussed below.

Water balance method

Van der Lely (Van der Lely. 1994) presents a diagrammatic representation of compartments that comprise the water balance model, together with flow paths into and out of the compartments.

The model was reconstructed in a Microsoft Excel workbook together with those data that were available between the years 1976–1998.

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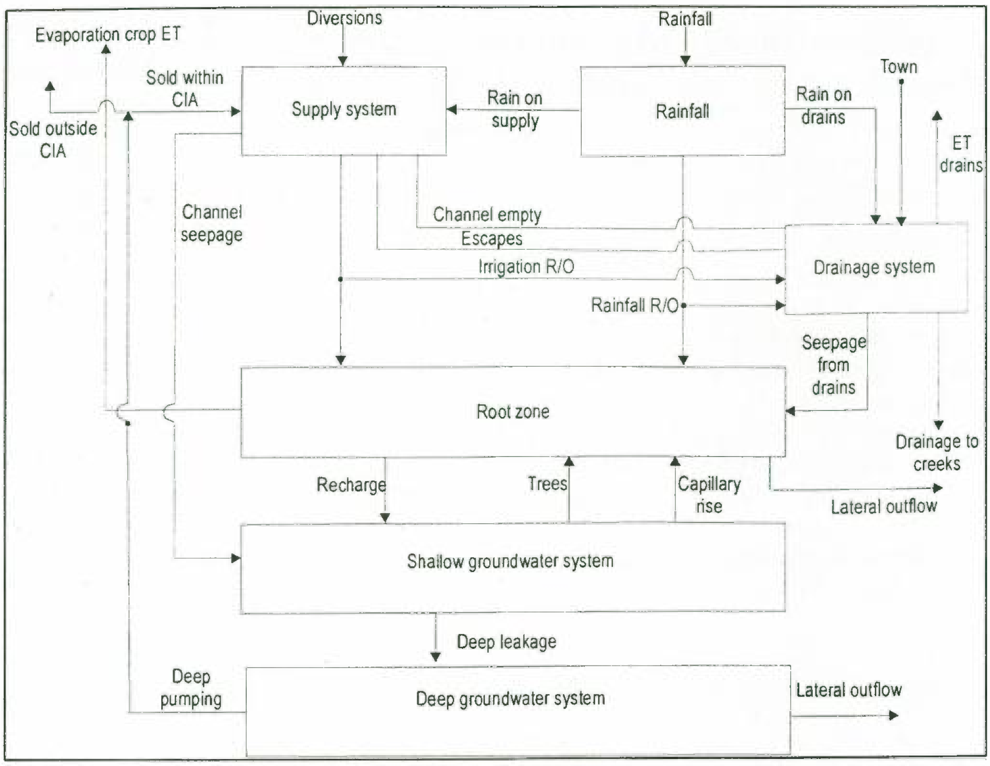

Figure 1 Water balance (after Van der Lely)

Results of water balance model

Figure 2 shows the output of the model for two major parameters: net water delivered to crops (GL/a) and net recharge (GL/a). It can be seen that prior to the year 1990/91 recharge was positive, with a general downward trend. Negative values for recharge occur in some years from 1990/91 to 1997/98. Recharge can be zero or positive, but it cannot be negative. Some obvious reasons for the illogical model output in some years present themselves. These include the fact that the area of each crop type sown in each year was not available, thus the same area and the same crop mix was used for all years. Lack of crop information has a direct effect on the total evapotranspiration, which is a major pathway leading from the root zone. While it is possible to remedy the lack of crop information in future years, it is not possible to quantify other parameters in the model. Of the twenty three inflow and outflow paths in the model, nine have had to be estimated and do not lend themselves either to direct measurement or annual re-estimation. The capillary rise component, for example, is assumed to be 15 GL/a and therefore represents a significant flow pathway. Thus there will always be a significant but unknown bias in the model.

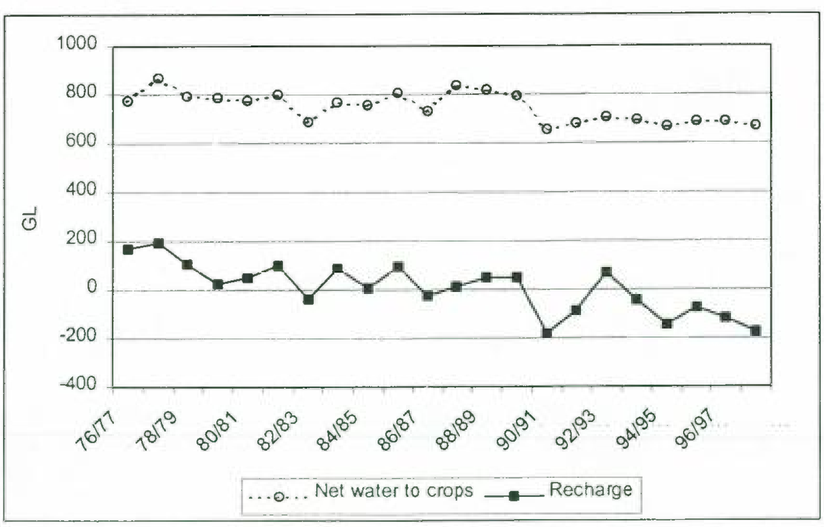

Figure 2 Output of the water balance model

Groundwater contour method

Approximately 667 piezometers are maintained and monitored within the Coleambally Irrigation area each year. A further 120 piezometers are located close to, but outside the irrigation area. The water level in all piezometers is measured twice each year, in March and September. Thirty piezometers within the irrigation area are measured once each month. Piezometers are either 'shallow' (drilled to depths between 5 and 12 metres) or 'deep' (drilled to depths between 12 and 35 metres. Data recovery is now close to 100%, however, in early years, recovery was as low as 40%.

Data from the years 1970 to 1998 were organised into a relational database and attached to a spatial information and analysis system (ArcView*). Several mathematical methods are available for converting spot water level (or pressure) data into contour maps that show regional water level. The methods can be grouped into those that are based on a regular, rectangular grid of nodes at which interpolated water level is computed, and those that use the actual water level reading at each piezometer location and compute contours that run between data points. The choice of method requires careful thought, since each method produces a different result. Some common methods are discussed below.

Kriging

Kriging is a procedure that generates an estimated surface from a set of scattered points with z values. It is in the group of methods that uses a regular rectangular grid of nodes to compute a surface. Unlike other interpolation methods, kriging requires the user to undertake an interactive investigation of the spatial behaviour of the phenomenon represented by the z values before choosing the optimum parameters for generating the output surface. In the case of groundwater level, a knowledge of the transmissivity, in three dimensions, of sub-surface features is required, since transmissivity will determine the spatial relationship between the water level reading in pairs of piezometers at a defined distance from each other. The kriging method assumes that this spatial relationship remains the same in all areas of the field of interest, i.e. the data are homogeneous. The kriging method is not appropriate for data sets that have pits or spikes and are thus inhomogeneous.

In the case of groundwater levels in the Coleambally Irrigation area, we contend that homogeneity is unlikely since the transmissivity of the strata that underlie the area is known to vary and recharge from the surface application of irrigation water varies from crop to crop, farm to farm and region to region. Localised spikes and pits in the groundwater surface are therefore likely to occur and should not necessarily be discounted by choosing an interpolation method that smoothes them out.

Inverse distance method

The inverse distance method first generates a regular, rectangular grid of nodes, then computes an interpolated value at each node by using a linearly weighted combination of a set of sample data points. The weight that is assigned is a function of the inverse of the distance. This method is most appropriate for locationally dependent variables. The significance of data points on interpolated points at nodes on the grid can be varied by choosing the power to which the distance between the points is raised. The most common power value used is the square of the distance. The greater the power value used, the smaller will be the influence of points at greater distances from the node. Lower power values produce smooth surfaces and higher power values produce surface with greater detail. The characteristics of the surface can be further refined by limiting the number of data points sampled to those that lie within a specified radius of the point of interest. The inverse distance method is most appropriately applied to data points that are densely spaced with regard to the local variation in z values (Watson and Philip, 1985).

The inverse distance method makes no assumption of homogeneity and can be adjusted to accommodate spikes and pits in data when used with care. We contend that this method is more appropriate to the interpolation of groundwater surface in the Coleambally Irrigation area than the kriging method.

Triangulation method

The triangulation method is an 'exact' interpolator in that it preserves the original data points. In this method, triangles are drawn between each data point in such a manner as to produce a patchwork of triangles over the entire extent of the field of interest. Each triangle defines a plane over the grid nodes lying within the triangle, with the tilt and elevation of the triangle determined by the three original data points that define the triangle. Because the original data define the triangle, the interpolated surface honours these data very closely.

Triangulation is most appropriately used for data that are evenly distributed over the entire area. In areas where data are sparse, triangulation produces distinct triangular facets in the contour plot. In the case of the Coleambally Irrigation data, the piezometers are unevenly spaced in some areas and are sparse in the region that surrounds the irrigation area. We therefore decided not to apply the triangulation method to data sets that included both those piezometers within the irrigation area and those that surround it.

* ArcView® GIS Ver. 3.1. Copyright Environmental Systems Research Institute, 1992-1998

Results of surface interpolation method

We applied four interpolation methods to data from the Coleambally Irrigation area and varied the parameters in each method that control the degree of smoothing of the interpolated surface. Table 1 shows a summary of our results, expressed in terms of the percentage of the total irrigation area under which the groundwater level is less than 2 metres. It can be seen that the computed area lies in the range 12–37 % for the same data set. No one method produces a result that is necessarily 'better' than another, unless the method and parameters are chosen as a result of better knowledge of the characteristics of the data set, i.e. a better knowledge of the physical processes that affect the water table.

Table 1 Effect of various interpolation methods on groundwater surface area

Interpolation method	Area (%)	Interpolation method	Area (%)
Kriging		Inverse distance method	
Linear interpolation, no drift	25	Quadrant search, power 2	18
Linear interpolation, linear drift	37	Quadrant search, power 3	23
Exponential interpolation, no drift	25	Octant search, power 2	19
Exponential interpolation, linear drift	37	Octant search, power 3	23
Quadratic interpolation, no drift	25	Shepard's method	
Quadratic interpolation, linear drift	37	Quadrant search, 6 samples	13
		Simple	12

The interpolation methods shown in Table 1 were used to construct groundwater contours. Figure 3 shows these contour maps constructed by four interpolation methods using identical data for each plot. The shaded area in each plot is the area enclosed by the 2 metre contour line, except in the triangulation method, where shading occurs in regions of no data, and in the plot of Shepard's method where shading appears as a result of the inappropriate use of this method with these data. The boundary of the Coleambally Irrigation area is shown, together with the location of piezometers.

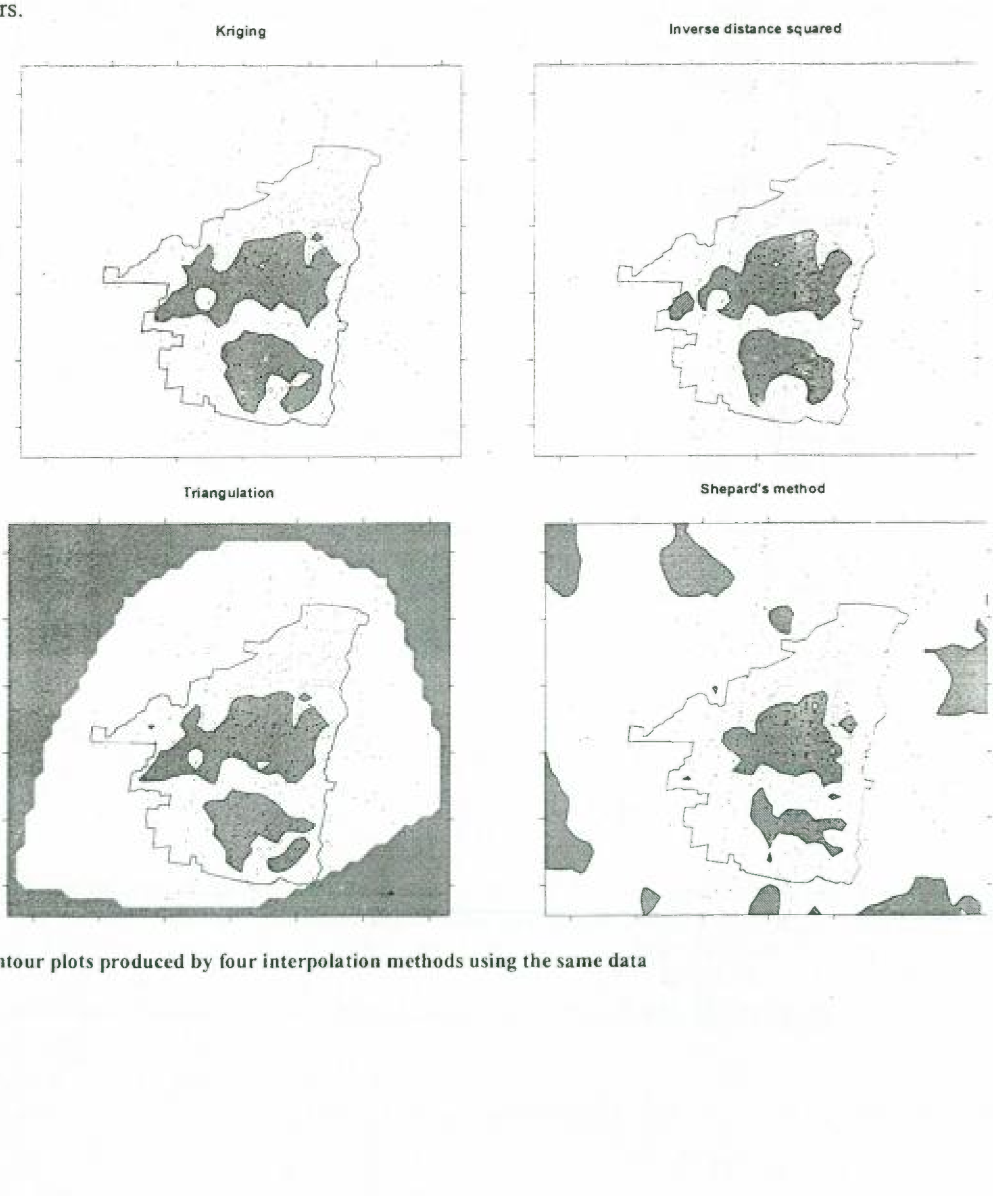

Figure 3 Contour plots produced by four interpolation methods using the same data

For reasons outlined above, we chose the inverse distance method, with a power of 2, to interpolate the surface of the groundwater and to contour the results. Finally, we calculated the area within the 2 metre contour line, in each year from 1989/90 to 1996/97. Prior to 1987, no groundwater levels were within 2 metres of the ground surface. Figure 4 is a graph of the areas calculated for each year.

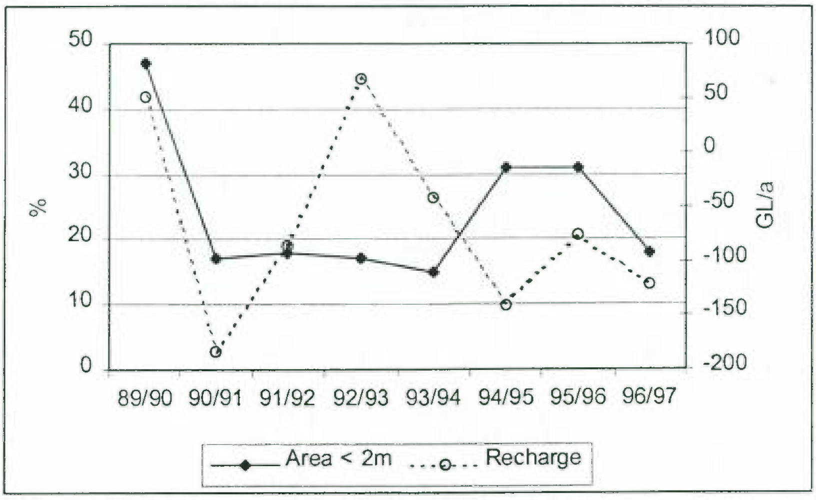

Figure 4 Relationship between 2 metre contour and annual recharge

The data shown in Figure 4 indicate a superficial correlation in some years, however, taken together the coefficient of correlation for all pairs of points is 0.32, indicating only a weak relationship between calculated net recharge and the area enclosed by the 2 metre contour line. Given that net recharge has been calculated using the water balance model. which is known to produce illogical results using the data available to us, it may be that a closer association between net recharge and area within the 2 metre contour line is being obscured by this effect, possibly compounded by the relatively few pairs of data points.

Discussion

We have presented the results of two methods of describing the net recharge of water to the root zone in the Coleambally Irrigation area. Neither method is without its problems and both warrant further development in order to produce unambiguous answers the management questions: 'Are the Land and Water Management Plan initiatives producing a beneficial change in groundwater level? If so, is the rate of change sufficient to meet objectives and targets?'.

In the case of the water balance method, improved data collection methods have been instituted by Coleambally Irrigation and it is possible that this will remove some of the uncertainty surrounding the calculation of net recharge. however, it will not be possible to quantify all important parameters without the expenditure of a great deal of research funds. The surface interpolation method is, as stated, an indirect method of assessing net recharge but has the advantage that no additional data or research are needed in order to calculate the area within the 2 metre contour line. In the absence of detailed information of sub-surface conditions, no one interpolation method is necessarily 'better' than another, provided that both produce contours that do not intersect, produce 'islands', or major discontinuities. The choice of method and the choice of smoothing parameters (i.e. 'tension' factors) depend on a proper understanding of the limitations and strengths of each method and on a judgement of the spatial relationship of the data set.

Alternative method

Figure 5 is a plot of the geometric mean of piezometric readings (points) taken from deep and shallow bores in March and September. The figure also shows trends (lines) computed using a power function.

It can be seen that the geometric mean of the piezometric readings in each year indicates a slowing in the rate of groundwater rise in the Coleambally Irrigation area. While the correlation between the geometric mean of piezometer readings in a data set and net recharge is logical, it has not been possible to date to extract pairs of data points from the historical records that quantify the coefficient of correlation. Nevertheless, Figure 5 provides information that is readily understandable and moreover, very simple data analyses techniques have been used. The geometric mean value was chosen over the arithmetic mean value because data in each set are highly skewed (log-normally distributed).

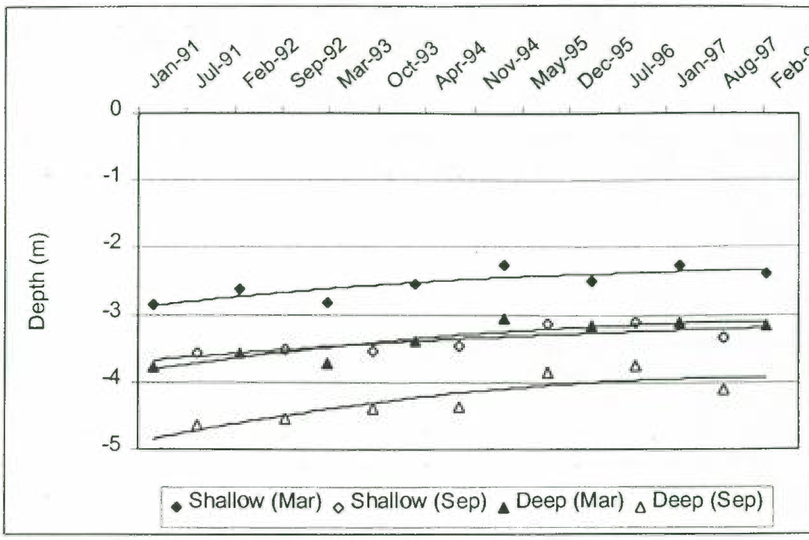

Figure 5 Geometric mean groundwater level in bores of two depth ranges

The figure indicates that the rate of rise of the groundwater level has decreased in the years 1991–1996 and may have been reversed in the years 1996–1998, i.e. the groundwater level may be falling. While the relationship between the geometric mean of all piezometer readings and the quantum of recharge in each year is tenuous, the Figure 5 is of more use to management than Figure 3 and 4.

Conclusions

The water balance method is complicated and includes several unknowns that it would be expensive or impossible to quantify accurately. The surface interpolation and contouring method is simple and relies on existing data, however, without a knowledge of the spatial variation in, for example, local recharge, appropriate surface tension factors cannot be determined. For management purposes, a simple calculation of the geometric mean of groundwater data sets provides a clear picture of the success or otherwise of initiatives designed to address the problem of the historical rise in water table.

The objectives and targets contained in the Land and Water Management Plan and in the Environmental Management System need to be couched in terms that monitoring and measurements are able to illuminate, without the expenditure of limited research funds.

We suggest that the geometric mean method answers the questions:

- Are the Land and Water Management Plan initiatives producing a beneficial change in groundwater level?
- If so, is the rate of change sufficient to meet objectives and targets?'.

Future directions

Coleambally Irrigation is currently participating in cooperative research programmes designed to provide information on sub-surface conditions. The results of the research may provide insights into the dynamics of groundwater movement and level that can be used to refine the water balance model and the contouring of piezometric readings.

In the interim, Coleambally Irrigation will continue to use the tools it has available to it in order to measure the effectiveness of its management initiatives. It may be that the ultimate choice of method is less important than the consistent application of a single method that can clearly show the magnitude and rate of change in environmental parameters.

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- Van der Lely, A. Coleambally Land and Water Management Plan: Regional Options. Technical Services Division, Department of Water Resources NSW. 1994.
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Review of Regional Groundwater Model of Coleambally Irrigation Area

S. Khan[1], N. P. Merrick[2], E. Christen[1] and A. Tiwari[3]

[1]CSIRO Land and Water, PMB 3 Griffith, NSW, 2680, [2]National Centre for Groundwater Management, University of Technology, Sydney, PO Box 123 Broadway NSW, 2007, [3]Coleambally Irrigation Corporation, Brolga Place PO Box 103, Coleambally NSW, 2707.

Abstract

This paper reviews a regional groundwater model developed by CSIRO Land and Water, Griffith Laboratory for Coleambally Irrigation Area (CIA) (Enever, 1999). The CIA consists of 79,000 hectares of the Riverine Plain in NSW. The main crop type of the area is rice, which is grown under flooded conditions. Historically the watertables in the CIA have risen due to recharge from irrigation. The piezometric hydrographs in the area show 5 to 15 m rises during the 1970-90 period, however, the rate of rise has reduced considerably in recent years (1990-1998). The rising watertables have resulted in shallow watertable conditions in many parts of the area and hence waterlogging and soil salinisation. Waterlogging and soil salinisation are major environmental concerns for farmers and pose serious threat to agricultural productivity and sustainability.

To develop an understanding of the hydrodynamics of the groundwater system in the CIA a four layer groundwater model consisting of upper Shepparton, lower Shepparton, Calivil and Renmark formations was developed using USGS, MODFLOW. This model uses a 1.25 km square mesh consisting of four layers, 60 columns and 66 rows covering an area of 618,750 ha. The model was calibrated for the 1985-1995 period. A number of pumping and recharge scenarios have been considered in order to investigate the future irrigation and groundwater management options for the CIA.

The present model shows that pumping from deep aquifers has a relatively small impact on groundwater conditions in the shallow aquifers. It is also observed that similar pumping from shallow aquifers achieves only localised spatial drawdowns in shallow layers. The model calibration has helped understand the effect of different recharge scenarios on watertable rise, regional flow characteristics and groundwater interaction between different geological formations underlying the CIA.

1. Introduction

This paper reviews the development of a groundwater simulation model of the Coleambally Irrigation Area (CIA) in the Riverine Plain of NSW (Enever, 1999). The future sustainability of irrigation in CIA is threatened by the development of a shallow watertable mound underlying the irrigation area. The piezometric hydrographs in the area show a 5 to 15 m rise during the 1970-90 period however, the rate of rise has reduced considerably in the recent years (1990-1998). The rising watertables have resulted in shallow watertable conditions in many parts of the area and hence waterlogging and soil salinisation. Waterlogging and soil salinisation are major environmental concerns for farmers and pose serious threat to agricultural productivity and sustainablity. In general, control of watertables involves reducing groundwater accessions, and/or removing water from the groundwater system. The problem facing resource managers in the area is how to achieve an adequate reduction in groundwater accessions and, if necessary, design engineering options for the removal of groundwater, whilst maintaining the economic viability of the area as a whole.

The Coleambally Irrigation Corporation and CSIRO developed a set of mathematical models for net recharge management in the CIA (Prathapar, 1994). The mathematical models included a farm scale recharge management model SWAGMAN Farm, policy models and a regional groundwater model. The CIA regional groundwater model was developed as a tool to aid water resource managers in the CIA to understand the groundwater system in the area, the sensitivity of groundwater pressures to system parameters, and to assess the effect of different management options on the system. The groundwater model will be used in association with SWAGMAN Farm and policy models to understand dynamics of water, crop and soil that affect the subsurface water and salt balance and to devise farm level policies to reduce net recharge. These models will be used for the entire Coleambally area along with intensive monitoring.

2. The Coleambally Irrigation Area (CIA)

General

The CIA comprises an area of 79,000 ha south of the Murrumbidgee River in the Lower Murrumbidgee region of NSW. The area was originally settled in the 1840's and was predominantly used as pastoral land (Coleambally Land and Water Management Plan 1996). The groundwater pressures in the CIA remained relatively unchanged until the advent of irrigation in the 1960's when 333 farms were allocated 79,000 ha to form the CIA. The main irrigated enterprises in the area are rice, sheep/annual pastures, winter crops, soybeans and some horticulture. By far the most prevalent land use is rice. Rice is grown under ponded conditions, and for this reason tends to use much larger volumes of water per hectare (~14 ML/ha) than other crop types (Wheat 3.6 ML/ha, Soybeans 9.5 ML/ha).

Average total annual rainfall in the Lower Murrumbidgee region decreases slightly from east to west and lies in the range of 400-450mm/yr. The annual evaporation varies between 1500-2000mm.

The water balance estimates for CIA (Coleambally Land and Water Management Plan 1996) suggest that the total amount of water entering the shallow groundwater system is about 54 GL/yr (average over 1985-1995). Of this, 14 GL/yr moves away from the area laterally, and 26 GL/yr flows downwards into the deep groundwater system. The remaining 14 GL/yr is the volume assumed to cause rising groundwater levels. It is important to note that downwards flow to deeper aquifers may result in higher groundwater potentials in deeper aquifers (in the absence of significant pumping) and therefore reduced potential gradients between shallow and deeper aquifer systems. The reduced groundwater gradients between the shallow and deep aquifers may result in decreased downward flows in future and therefore a greater rate of watertable rise for similar recharge. Therefore future hydrodynamics of the groundwater system need to be investigated using the groundwater model presented in this paper.

Geology of the CIA

Details of regional geology of CIA can be found in Prathapar et al. (1997). The Murrumbidgee alluvial fan forms a large part of the Lower Murrumbidgee region. The apex of the Murrumbidgee alluvial fan is situated near Narrandera. The sediments of this fan increase in thickness in a westerly direction from the eastern flank of the basin. Brown and Stephenson (1991) categorised these sediments based on age and type of deposition into three distinct units, these being Renmark, Calivil and Shepparton formations.

The Renmark formation is the oldest stratigraphic unit and directly overlies the pre-Cainozoic bedrock. It was deposited during the Palaeocene to Middle Miocene ages. The base of the Renmark formation consists of light brown quartz sand (Warina Sand), the upper sequence consists of more argillaceous and carbonaceous sediments (Olney Formation). It is estimated that 30 to 50% of the Renmark formation is comprised of sand. The horizontal hydraulic conductivity of the Renmark formation averages between 10 and 30 m day^{-1} but it can be as high as 100 m day^{-1} (Prathapar et al. 1997).

The Calivil formation was deposited during the late Miocene to Pliocene ages and consists predominantly of pale grey coarse to granular quartz sand, with lenses of kaolin and carbonaceous clay. It is estimated that this formation consists of 50 - 70% sand and gravel. The average hydraulic conductivity of the the Calivil formation estimated at Darlington point is approximately 130 m day^{-1} (Prathapar et al. 1997).

The Shepparton formation was deposited from the Pliocene age until present day. It consists of a matrix of clay, silt and silty clay, with lenses of fine to coarse sand and gravel. The clay is silty, variegated, mottled and red brown, yellow or white in colour. The proportion of sand in this formation is highly variable, but typically in the range of 10 - 30%. The geology of the Shepparton formation is complicated due to prior stream deposition which resulted in localised concentrations of coarse grade sands and connections to deeper aquifers. Due to the discontinuous nature of Shepparton sand lenses the average hydraulic conductivity is around 2 to 3 m day^{-1} but it may be 25 to 100 m day^{-1} in the more sandy parts (Prathapar et al. 1997).

Previous Groundwater Models in the CIA

Punthakey et al. (1994) reported a groundwater model of the Lower Murrumbidgee River Basin (LMRB). This model was constructed as an initiative of the Murray Darling Basin Commission Groundwater Working Group. This initiative involved the development of five regional groundwater models covering the entire basin, which were to be integrated into a Murray Darling Basin model. The LMRB model is a three-layer MODFLOW based model. This model used a 7.5 km square mesh. The mesh size of this model provided a relatively coarse representation of the groundwater system and therefore the development of a more detailed sub-model of the CIA was necessary.

3. CIA Regional Groundwater Model

Discretisation

The CIA Regional groundwater model has been developed using USGS MODFLOW (McDonald and Harbaugh, 1988) under PMWIN pre- and post processing environment (Chiang and Kinzelbach, 1996). Figure-1 shows the layout of the finite difference grid used for the CIA model. The grid consists of 60 columns and 66 rows (1 25 km square mesh) and encompasses an area of 6,187.5 km^2. Row 1, Column 1 in the CIA model grid equates to row 19 column 30 in the LMRB model. The eastern edge of the grid was set parallel to the bedrock. The southern edge was positioned to include Yanco Creek. The northern edge of the grid was set to include the Murrumbidgee river and was positioned far enough north of the Murrumbidgee to include the areas around Darlington Point to incorporate groundwater pumping over the last ten years. The western edge of the grid was set several kilometres outside the western limit of the CIA to minimise effects of boundary conditions on the groundwater regime in the CIA.

The four layers represented in the CIA model were defined by the stratigraphic breakdown shown in Figure-2. The Renmark and Calivil layers were defined by interpolation of the layer thicknesses for the equivalent formations in the LMRB model (Punthakey et al. 1994). The Shepparton formation was further divided into two separate model layers in order to improve the model's treatment of the vertical flow processes in the shallow aquifers. Kriging was used in the interpolation of all spatial data sets in the CIA model using *Surfer* (Golden Software, 1994).

The CIA model simulation period was March 1985 to March 1995 using monthly stress periods and 10 day time steps.

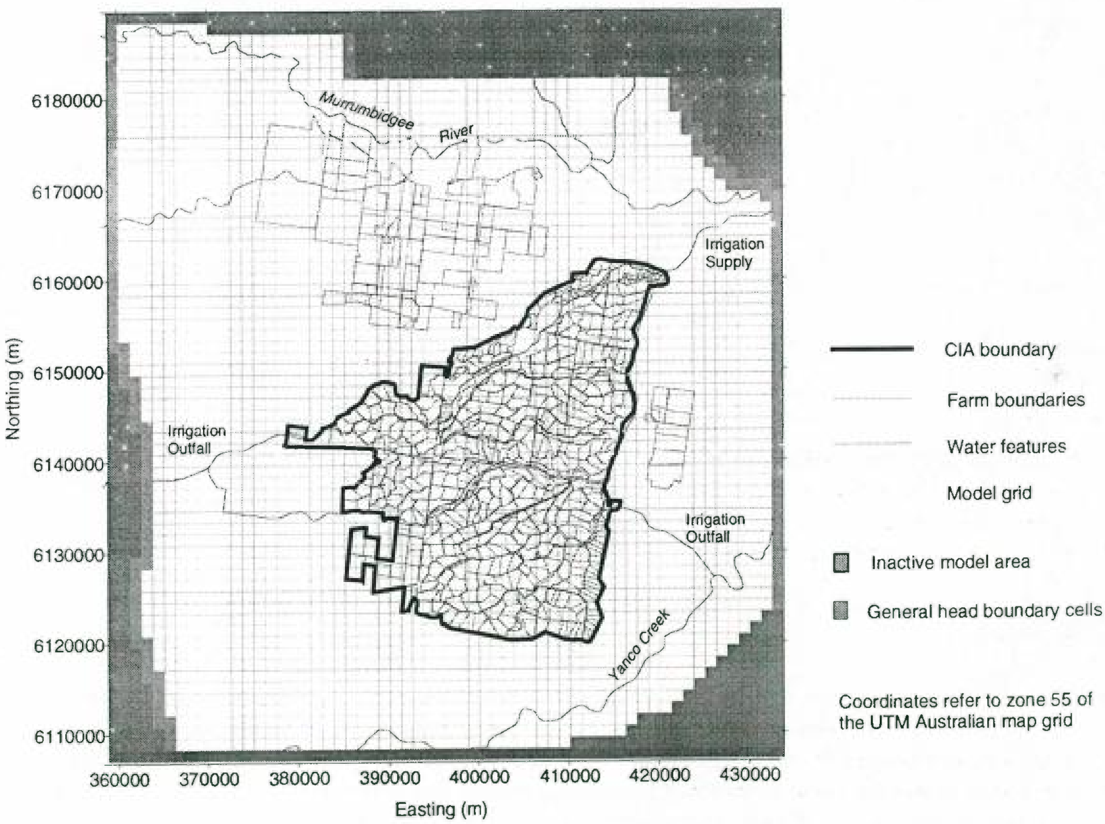

Figure-1 CIA model grid (Enever, 1999)

Hydraulic Properties

The spatial variation of hydraulic conductivities, specific yield and specific storage in different model layers was defined by using Punthakey et al. (1994) procedure of harmonic means based on average soil properties and bore log data. The number of bore logs used to derive the hydraulic properties of individual layers were 442, 405, 177 and 113 for the Upper and Lower Shepparton, Calivil and Renmark formations respectively.

The interactions between model layers were specified using vertical leakance terms of MODFLOW.

Vertical/horizontal anisotropy ratios of 1:20 to 1:100 were used to calculate vertical hydraulic conductivities. The vertical leakance for each cell was calculated from vertical hydraulic conductivities and layer thicknesses as described in McDonald and Harbaugh (1988).

Initial and Boundary Conditions

Using the Department of Land and Water Conservation bore records the initial piezometric levels in different layers were defined for March 1985. In total 616 observations were available for the upper Shepparton, 221 for the lower Shepparton, 27 for the Calivil and 25 for the Renmark formations. Specified head boundaries for the CIA model were included using the general head boundary package (GHB1).

Aquifer Recharge and Discharge Parameters

DLWC data on monthly pumping rates and bore locations was used to specify pumping rates in the model.

In the CIA model recharge for each cell was initially set at one constant rate for 1985 to 1990 and another constant rate for the period 1990 to 1995. The recharge was calibrated during model simulation by trial and error procedures. While the present procedure to determine recharge gives realistic aggregate volumes of net recharge over a 5 year period, it does not provide any insight to the recharge mechanism and does not allow tracking of recharge dynamics. This aspect will be improved by incorporating temporal and spatial distribution of recharge due to rainfall and irrigation using improved surface-groundwater interaction models.

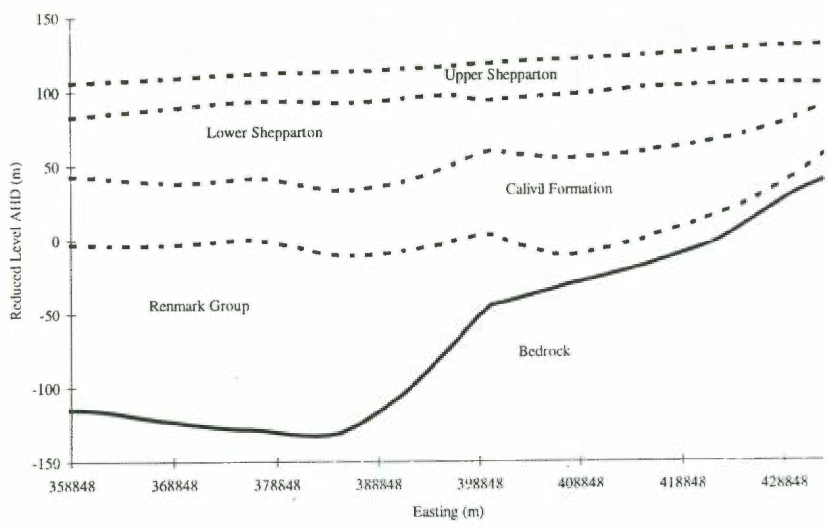

Figure-2 East to west transect through Northing 6149562 (m) (Enever, 1999)

Calibration

A trial and error procedure was adopted to calibrate the hydraulic parameters and recharge to match piezometeric water level hydrographs. Forty-three hydrographs were used to calibrate the CIA model, 14 in the Upper Shepparton, 12 in the Lower Shepparton, 8 in the Calivil and 9 in the Renmark. Figure-3 shows comparison of computed and observed drawdowns (negative drawdown is rise in groundwater potential) for 1985 to 1995 period at one of the piezometers in the upper Shepparton aquifer. It can be noticed that model results follow the overall trend of the observed piezometric levels but do not properly represent the seasonal variation of piezometric levels. This can be explained from the fact that the present model is based on average calibrated recharge values and can not therefore show seasonal fluctuations due to changes in irrigation and rainfall patterns. Similarly reasonable match between computed and observed hydrographs was achieved at other locations, however, the present model lacks spatial agreement with the piezometric contours for different periods. The greatest difference in spatial agreement occurs in the Renmark model layer after 10 years of simulation. However, lack of data to interpolate piezometric contours for deeper aquifers is a major constraint in developing reliable initial conditions and contour maps for other periods to compare with computed data. Calibration of the vertical leakance terms resulted in higher estimates under the irrigation and therefore needs improvement and fine-tuning. The present calibrated version of the CIA model includes very high estimates of specific storage at some locations for lower Shepparton, Calivil and Renmark formations (some values >0.002 m^{-1}). The storage properties of Calivil and Renmark formations have a major impact on the water levels computed in these layers and therefore need to be revisited along with new estimates for vertical leakance and recharge.

4. Model Runs

Simulation of 1985-1995 Period

For the historic simulation period 1985-1995 water balance components were computed for different time steps to understand the changes in groundwater dynamics of the CIA. Figure-4 shows the water balance for a 10 day period in summer 1995. The various components of flow balance have been converted to GL/yr. It can be observed that calibrated recharge to groundwater is 38.6 GL/yr, deep drainage is 19.8 GL/yr. The net rise in groundwater storage in the upper Shepparton aquifer is 17.4 GL/yr (average watertable rise 0.22 m/yr assuming an average specific yield of 0.1). The model results show a net decrease of 4 GL/yr in groundwater storage in Calivil and Renmark formation due to a 9 GL/yr pumping from Calivil formation. The model results show a net regional outflow of 13.7 GL/ yr from the CIA. The present pumping from Calivil formation is helping promote downward gradient from upper aquifers by reducing aquifer pressures in Calivil formation and is therefore an important factor in reducing the rate of watertable rise.

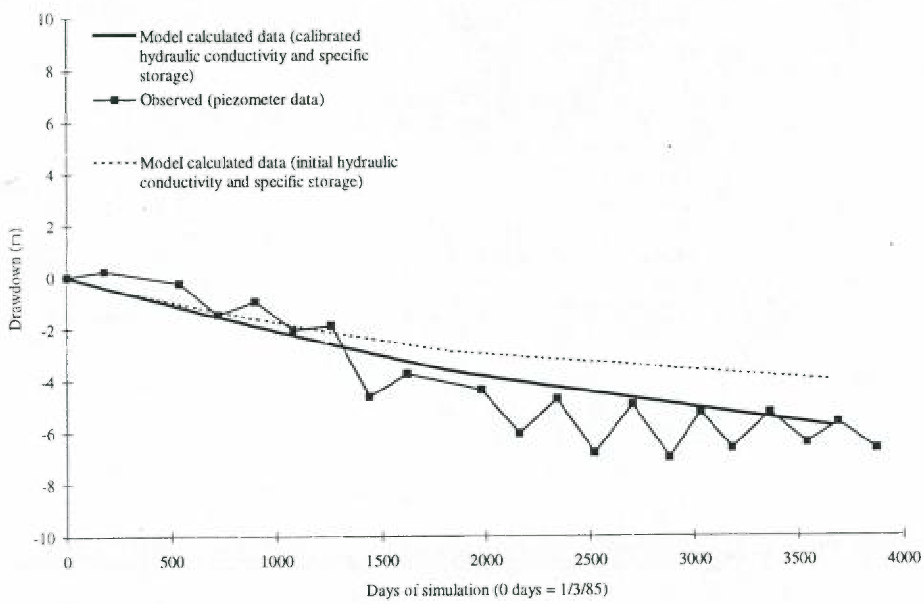

Figure-3 Observed and calculated hydrographs: Piezometer 12379 Model Layer = Upper Shepparton Piezometer Location = Easting 385956(m) Northing 6143058(m) (Enever, 1999)

Scenario Runs

A number of pumping and recharge scenarios were run to explore groundwater management options in the CIA. The details of these scenarios can be found in Enever (1999). A summary of results of these scenarios is presented below:

- Pumping from the Calivil layer from three locations at the rate of 6ML/yr from 1990 to 1995 resulted in large drawdowns in the Calivil and Renmark layers with only minimal impacts on the upper Shepparton and lower Shepparton layers.
- Pumping from the lower Shepparton layer from 21 pumps at the rate of 1ML/yr resulted in large drawdowns in the lower Shepparton and upper Shepparton layers, with minimal impact on the Calivil layer and no impact on the Renmark layer.
- The radius of influence of pumping in the lower Shepparton and upper Shepparton layers is very small compared to the Calivil and Renmark layers.

- If pumping is carried out from the lower Shepparton or upper Shepparton layers in order to control the watertable then at least 20 pumps pumping at 0.5 - 1ML/yr would be required in order to achieve widespread draw downs in the piezometric level in the upper Shepparton layer.
- The estimated recharge rate between 1985 and 1990 is 62.9GL/yr.
- The estimated recharge rate between 1990 and 1995 is 38.6GL/yr.
- The recharge rate required to maintain piezometric levels at the 1995 level until 2005 is 22.4GL/yr.

The above conclusions should be interpreted cautiously since the present version of the model requires hydrological and mathematical improvements.

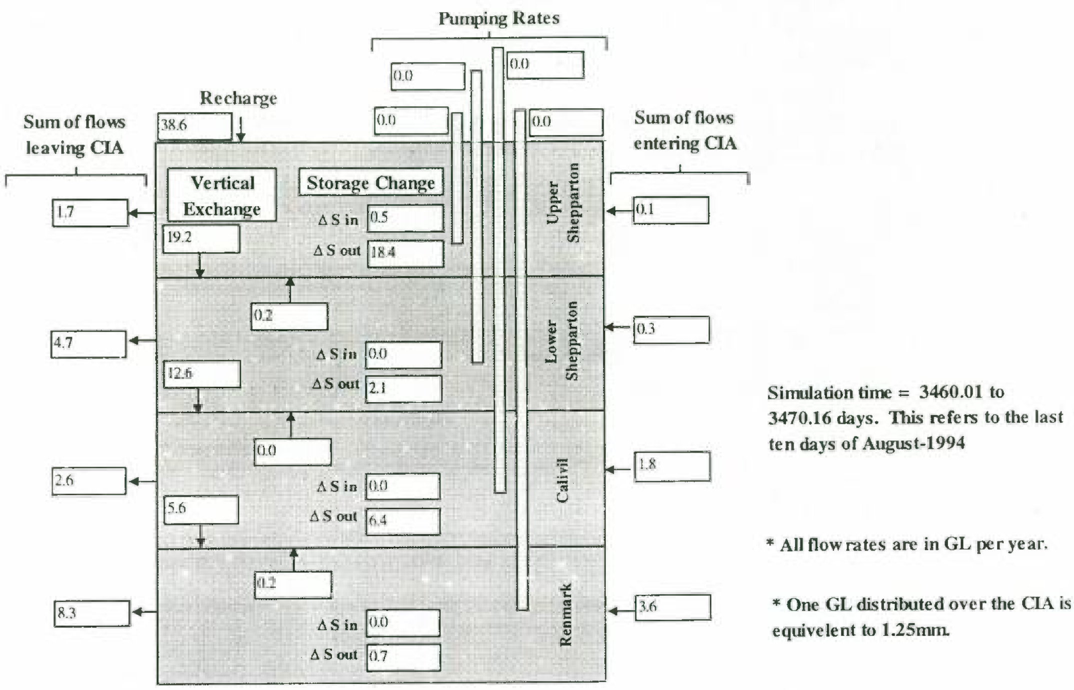

Figure-4 Water Balance February 1995 (Enever, 1999)

5. Conclusions, Future Model Improvements and Research Links

The Coleambally regional groundwater model has enhanced the understanding of hydrodynamics of regional groundwater flow and provided vital input into farm scale models such as SWAGMAN Farm. The groundwater model results have clearly demonstrated the significance of detailed regional models in managing net recharge in shallow watertables areas. The calibration of the present Coleambally groundwater model is to be refined and solute transport simulation added on the basis of better recharge estimates from the water balance models being developed under the following programs.

a) Conjunctive Water Management for Sustainable Irrigated Agriculture in Asia Project

The Coleambally Irrigation Corporation (CIC), Australian Centre for International Agricultural Research (ACIAR) and International Water Management Institute (IWMI) will be jointly working on this project. The Australian component of this project will focus on conjunctive water use in CIA. The CIA groundwater model will be further improved by incorporation of recharge estimation routines and used as a tool to investigate environmental sustainability.

b) Paddock Water Use Monitoring

The CSIRO, CIC, Murray Irrigation Limited (MIL) and Land and Water Resources Research Development Corporation funded Paddock Monitoring project (Coleambally Irrigation Corporation, 1998) will collect farm level water balance data in Coleambally and Murray Irrigation Areas. This data will be used to obtain better benchmarks for crop water use, irrigation efficiency and recharge for summer crops and rice. This data will be used to develop water balance models (e.g. SWAGMAN Farm) to determine irrigation efficiency and recharge at the paddock scale. The output of this project will be utilised to develop recharge estimation tools for the groundwater models and therefore help improve the CIA groundwater model.

c) CRC for Sustainable Rice Production

Under sub-program 1.4 hydrological models will be developed to quantify climatic and irrigation impacts on groundwater and soil salinity at farm and regional scale. The hydrological models developed will be linked with GIS database and groundwater models. These will help identify the effect of improved irrigation management practices under Land and Water Management Plans on the watertable accessions in the Coleambally and Murrumbidgee Irrigation areas.

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LEAKAGE BETWEEN SHALLOW AND DEEP AQUIFERS IN THE SOUTHEASTERN MURRAY BASIN

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G.F. Holland. Goulburn-Murray Water. Vic

INTRODUCTION

The southeastern Murray Geological Basin contains unconsolidated to semi-consolidated Tertiary sediments with total thicknesses ranging up to about 250 metres. Irrigation development from surface water sources occurs throughout large areas of both the NSW and Victorian parts of this region. resulting in widespread shallow watertables and associated salinity. Watertable rise has occurred largely within the Shepparton Formation. which is the uppermost unit. with varying levels of leakage to the deeper aquifers of the Calivil Formation and Renmark Group. Low salinity. high yielding aquifers occur in these deeper formations. especially where major rivers such as the Murrumbidgee. Murray. Goulburn and Campaspe enter the Basin.

Various options have been evaluated for watertable control within the irrigation areas. including pumping from deep aquifers. This paper summaries the results from pumping tests at a number of locations. which were carried out specifically to evaluate deep pumping as a watertable control option. The sites. indicated in Figure 1. range in location from Coleambally (NSW) in the northeast to Rochester (Vic) in the southwest. Even though common stratigraphic nomenclature is used between sites. hydrogeological conditions may vary significantly.

The evaluation of leakage between shallow and deep aquifers is valuable not only for assessing watertable control options. but also for:

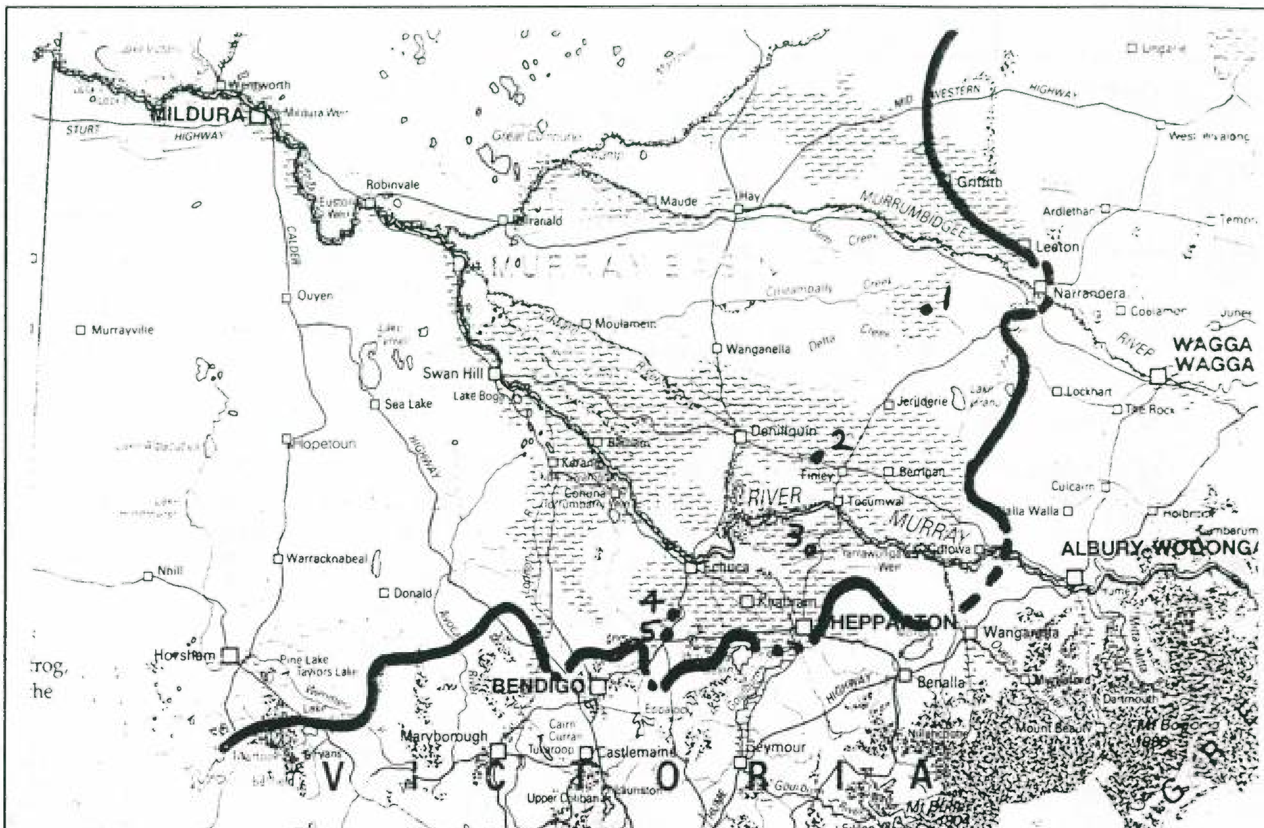
- Assessing interference between deep irrigation bores and shallower stock and domestic bores:
- Assessing recharge. and
- Input to groundwater models.

INVESTIGATION DETAILS AND RESULTS

Investigation details are summarised in Table 1. Key results are summarised in Table 2.

Table 1. Details of pumping tests

Location	Start of Test	Duration of Test (days)	Depth of Screens (m)	Test Pumping Rate (ML/day)
Coleambally. NSW	Nov 1990	467	74 - 134	28.5
Finley. NSW	Oct 1990	168	114 - 161	6.45
Murray Valley. Vic	Sep 1996	30	124 - 136	3.28
Campaspe West (Rochester). Vic	Aug 1996	32	67.7 - 72.5	3.38
Campaspe (Elmore). Vic	Nov 1989	29	62.5 - 97.5	13.8



N

Murray Geological Basin Boundary

Pumping Test Sites for Vertical Leakage Analysis

1. Coleambally Irrigation Area, NSW
2. Finley, NSW
3. Murray Valley, Vic
4. Campaspe West (Rochester), Vic
5. Campaspe (Elmore), Vic

Figure 1. Locality Plan

Table 2. Investigation results

LOCATION	K Horizontal (m/day in Calivil)	K Vertical (m/day in lower. Shep)	Ratio $K_h:K_v$	Salinity shallow aquifer (EC)	Salinity deep aquifer (EC)
Coleambally. NSW	50	0.0001	500.000:1	2000 – 10.000	510
Finley. NSW				200 – 25.000	1600
Murray Valley. Vic	5	0.00006 to 0.00042	83.000:1 to 11.900:1		2.200
Campaspe West (Rochester). Vic	40	0.001 to 0.01	40.000:1 to 4.000:1	1900 – 5.090 in L. Shepparton	1.200
Campaspe (Elmore). Vic	15 to 65	.004	3.750:1 to 16.250:1	500 - 20.000	370 - 520

The results generally indicate poor connection between watertable and Calivil Formation. or "deep lead". aquifers

In addition to the details given above:

- Lawson (1992) compared watertable observations with results from leakage analysis for the area within 2 km of the pumping site. The observed watertable decline. corrected against control areas. was about 0.5 m. The calculated watertable decline from leakage analysis was 0.2 m. The salinity of the pumped groundwater increased from 510 to 670 EC during the test.

- Bogoda and Lea (1991) concluded that "*Some relationship was noticed between the Calivil and lower section of the Shepparton Formation at some places. However, it would be necessary to pump at higher rates for a longer period to achieve a significant response from the upper section of the Shepparton Formation.*"

- Sinclair Knight Merz. for the Campaspe West site. concluded that "*The lack of drawdown in the shallowest bores indicates that most of the groundwater flow into the Calivil Renmark Group aquifer comes from the deepest part of the Shepparton Formation aquifer. However a vertical groundwater flux of 9 mm/yr from the upper part of the Shepparton Formation has been calculated from the vertical hydraulic conductivities...*". For the Murray Valley site. a vertical groundwater flux from the upper part of the Shepparton Formation of less than 1.8 mm/yr was calculated.

  For both sites they concluded "*pumping from the Calivil/Renmark aquifer promotes increased downward groundwater flux*". "*groundwater which flows downward into the Calivil/Renmark aquifer is drawn largely from the lower parts of the Shepparton Formation*". "*hydrographs show that pumping groundwater from the Calivil/Renmark aquifer has a small but measurable impact on watertable levels on a season to season basis*" and that "*pumping from the Calivil/Renmark aquifer may not be an effective management option for controlling high watertables.*"

- Brinkley. Reid and Mavrakis (1991) concluded that "*watertable response...... due to Deep Lead pumping was undetectable at most sites*" and that "*correlations between aquifer and river levels .... proved inconclusive*"

The University of NSW and NSW Department of Land and Water Conservation are currently investigating vertical leakage and aquitard properties at additional sites in the Lower Murrumbidgee. This work will utilise geochemical methods and groundwater level responses around existing production bores. and will complement previous investigations in the Coleambally Irrigation Area in the early 1990's. Geotechnical study of aquitard materials is also proposed to help separate upper aquifer hydraulic loading from actual leakage in evaluating past rising groundwater pressure trends in deep aquifers.

COMPARISON WITH OTHER AREAS

In evaluating the benefits of deep pumping for salinity control within much of the riverine plain, it is worth considering how the results from these pump tests compare with other areas where deep pumping has been recognised as a viable option.

The plains areas adjacent to the foothills

In the upper reaches of the ancestral river valleys, confining units overlying the Calivil formation are thinner and more permeable. These areas tend to be important sources of recharge to the deep groundwater system. Examples include the area south of Nagambie in the Goulburn Valley, and upstream of Narrandera in the Murrumbidgee Valley. While no known pump tests have been carried out to specifically quantify leakance within these upper areas of the plains, Figure 2 demonstrates the close relationship between the deep (70m) and shallow groundwater system at Tabilk south of Nagambie. In such areas, deep groundwater pumping can have a role in salinity control within overlying aquifers.

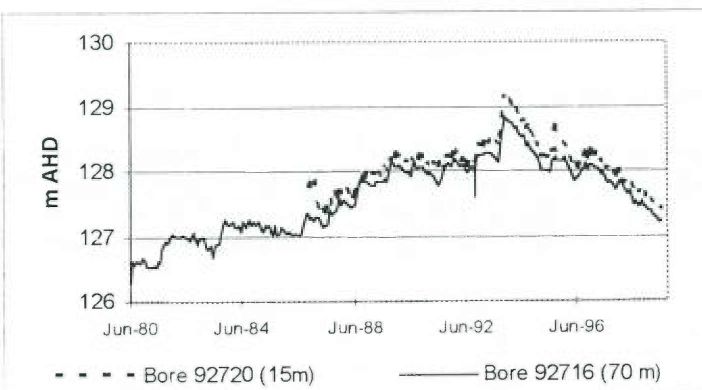


Figure 2. Comparison of shallow and deep groundwater levels at Tabilk, Victoria

The Indus Basin, Pakistan

The Indus Basin in Pakistan, one of the world's largest contiguous blocks of irrigated land, provides an example whereby groundwater pumping from deep aquifers has been utilised on a large scale for watertable control. The total irrigated area in 1959 was more than 51 million ha, with over 12 million ha affected by waterlogging and or salinity.

The Indus basin alluvial deposits are generally about 180 m thick, and thousands of bores have been constructed to depths of 60 to 120 m to pump groundwater for watertable control. Lateral hydraulic conductivities, at about 5 m/day, are comparable with those of the eastern Murray Basin. However, vertical hydraulic conductivities are much higher, in the range of 1 to 10 m/day. This results in horizontal to vertical ratios ranging from 50:1 to 5:1 - significantly lower than the ratios which range from 4,000:1 to 500,000:1 in the southeastern Murray Basin. The Indus Basin is clearly an area with good vertical hydraulic connection between aquifers, allowing deep pumping to function as an effective watertable control option.

Groundwater quality also differs in the Indus Basin. Low salinity groundwater generally exists throughout the entire alluvial profile, with the better quality groundwater mainly in the shallower aquifers. This indicates good hydraulic connection between aquifers, and is in contrast with the conditions in the eastern Murray Basin where the combination of high salinity shallow groundwater and low salinity in the deep aquifers indicates poor vertical connection.

IMPLEMENTATION OF RESULTS

Lower Murrumbidgee Groundwater Management Plan

For groundwater management planning in the Lower Murrumbidgee, the Murrumbidgee Groundwater Management Committee has taken a precautionary approach to deep pumping for shallow watertable control. It considers that, under the limitations of current knowledge, deep pumping is not an effective watertable control option. This is in response to both the lack of evidence to suggest sufficient hydraulic connection between aquifers, and also to concerns for long term groundwater quality decline in deep aquifers.

Shepparton Irrigation Region Groundwater Management Plan

Within the Shepparton Groundwater Supply Protection Area and Campaspe West Salinity Management Plan area, it has been concluded that deep pumping does not provide watertable control although it does have an important role in stabilising deep groundwater pressures. Increased pumping in recent years has lead to concerns relating to the potential impact on water quality from the overlying Shepparton formation and from poorer quality areas of the deep aquifer.

General Groundwater Resource Management

While the pump tests summarised above have provided a means of quantifying the contribution of vertical leakage to estimated sustainable yield of deep aquifers, annual extraction inducing small rates of leakage may result in long term changes in water quality that will lead to necessary adjustments to the estimate of sustainable yield. Groundwater salinity monitoring is therefore an essential requirement of groundwater management for deep aquifers.

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Rice Hydraulic Loading Perspectives

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Abstract

Rice hydraulic loading is a policy concept used in the northern parts of the Riverine Plain to restrict groundwater accessions to sustainable levels. Sustainability implies that productivity can be maintained indefinitely and the resource base is not degraded. Rice hydraulic loading is conjunction with land classification and rice water use targets is used to reduce accessions and the risk of salt accumulation in non-rice areas. Rice hydraulic loading rules involve the growing of not more than 30% of the landscape to rice. After its introduction there has been no re-consideration of this policy, even though different hydro-geological conditions are recognized. Rice areas on many farms actually are growing more than the original guideline. Some districts have adopted alternative policies, restricting irrigation intensity generally rather than the proportion of land under rice.

The effect of rice growing on groundwater levels may be examined from detailed data analysis. Soil salinity is the main performance monitoring parameter for sustainability. Trends may be evaluated using models supported by surveys. Approaches used to date are reviewed in the paper. These have been capable of providing only indicative interpretations, the spatial variation of several key factors being too great for adequate capture by the models.

Rice hydraulic loading may vary on a farm, between farms and within a district. The main areas at risk of becoming salt affected are depressed areas and areas less intensively irrigated. The process of salt transport from rice areas to the areas at risk may be strongly influenced by irrigation management and rice hydraulic loading rules, but it cannot be halted or reversed without ground water pumping strategies. In the eastern part of the Riverine Plain deep leakage provides a natural process helping towards sustainability.

Further research is needed to address sustainability questions for irrigation districts, and how rice hydraulic loading rules may assist. This would include ground water modeling supported by research on recharge and discharge processes. The paper provides a discussion of the key factors to be examined and explores the methodologies that may be employed in future research.

1. Introduction

Rice Hydraulic Loading may be defined as the application of a weight of water to a proportion of the landscape during the rice growing season. The effect of rice hydraulic loading is percolation and groundwater flow by gravity to areas or aquifers with deeper pressure levels. These may be the shallow water table areas adjacent to rice. Soil salinity may increase due to capillary rise. The correlation between possible soil salinity increases and variations in rice hydraulic loading is a question for sustainable management in Land and Water Management Plans.

The objective of sustainable natural resource management is continued productivity of the land resource without off-site impacts. The extent and degree of soil salinity in a landscape is the main performance indicator of the sustainability objective. High water table levels contribute to water logging of crops, and are a secondary performance indicator for the salinity risk. Ultimately, accessions affect water levels, and hence the salinity risk. A reduction in accessions is a key method to reduce the risk.

Rice growing policies have been used in all rice growing districts to control the volumes of accessions. Rice hydraulic loading (RHL) rules may be part of these restrictions and provide an additional tool to adjust the total volume of accessions to acceptable values. The acceptable volume often is not well defined and depends on many factors. It continues to be the subject of research in which ground water models are prominent.

Where RHL rules have been adopted they have been generic without consideration of local conditions. The consequences of various alternatives are nor clear. Models capable of evaluating RHL at the farm level are still in the development stage. More work is needed to further define the extent of sustainability that is being achieved with current policies. This paper provides a review of the factors involved and identifies the questions to be explored further.

2. Background

Rice growing in the Riverine Plain of South East Australia is subject to restrictions based on policy to prevent excessive accessions to the water table. Four approaches are generally used:

1. Land classification to restrict rice to the least permeable part of the landscape;
2. Restrictions on the maximum permissible rice water use per hectare;
3. Restrictions on the maximum proportion of land used for rice growing;
4. Restrictions on the maximum volumes of water used for irrigation, including rice.

The second approach would not be necessary if the first always was fully effective. Obviously it has not been, because irrigation company records show that rice water use for a proportion of farms is well above the crop water requirement. Estimates of soil permeability for land classification have not been perfect. Where excessive rice water use has occurred land may be re-classified as unsuitable. It is found however that its adoption often is difficult since most landholders depend on rice as their main source of income and re-classification of land causes potential hardship.

Before 1989 rice areas per farm were fixed and industry controlled. After deregulation in 1989 these controls were lifted, and environmental controls only could be applied. Accessions to groundwater are dissipated in the non-rice areas. It was feared that increasing proportions of land under rice would raise water levels in the non-rice areas excessively, making the cultivation of other crops in those areas impossible. Non-rice land and lower intensity irrigation farms would be subject to an increased salinity risk. It was clear that the combination of land classification and rice water use policies was unlikely to produce the desired outcome in terms of a sustainable industry. Based on these considerations and research carried out by van der Lely (1981) the rice hydraulic loading (RHL) policy was introduced in 1990. This allowed the growing of rice of up to 30% of the rice approved area (RAA) on any farm. The RAA is based on land classification.

The effect of the RHL policy was that larger landholders could grow an increasing area of rice but farms less than about 250 hectares of RAA would be forced to grow less than previously. To accommodate the change, a phasing down element was added to the RHL policy. The latter aspect however was abandoned after 3 years, and many landholders in the MIA and Coleambally at present are still allowed to grow more than 30% of their land to rice ([1]).

Van der Lely (1993) prepared a discussion paper on Rice Environmental Guidelines which was adopted by the Rice Environmental Policy Advisory Group, a committee of government agencies, irrigation stakeholders and the rice industry. The RHL concept however was not accepted in the Murray Region, mainly because the historic rice areas per farm hardly ever exceeded the 30% criterion. That area adopted restrictions on the total water availability for each farm in their LWMP. In many respects such type of restrictions would have a similar outcome.

Currently each LWMP includes a statement regarding rice policy or maximum water use in their area. CSIRO Land and Water is conducting research under its Net Recharge Management Program for the Coleambally and Murray regions, which may affect rice policy in the future.

[1] In areas without surface drainage being available the maximum allowed areas are restricted to 25%. This applies to for instance Wah Wah, the Murrumbidgee River rice growers and Bore pumping areas.

3. Land Salinity Trends

If the key performance indicator for land sustainability is the soil salinity status then this should be monitored. It is interesting to note that many areas do not have regular surveys to obtain spatial distributions of soil salinity, or a statistical representation providing areas exceeding a specific level. This is despite all LWMP areas having made estimates as to the future distribution for the No Plan scenario and the With Plan scenarios and relying on these predictions for estimating the economic merit of LWMP options which may reduce accessions.

The most suitable methods for carrying out inexpensive regional scale surveys still has not been agreed upon. EM surveys are being carried out for other purposes such as whole farm planning and rice land suitability. This may provide an opportunity to gather salinity information as well. Generally however no salinity survey based on the EM technique has been completed for any NSW district. On the other hand, there have been several surveys of the soil salinity status in individual sub-districts of the MIA and Coleambally based on random sampling.

In the MIA results of four sampling dates 1991, 1992, 1994 and 1998 are now available, representing years with initially high groundwater levels to years with dry conditions and lowered groundwater levels. For each date and eight sub-districts ([2]) at least 50 sites were visited and samples taken at 0.1m and 0.3m depth. Assuming that each sampling location represents an equal area, the proportions of land above a certain salinity level were assessed, as shown at Table 1.

Table 1: % of land affected by salinity in various sub-districts of the MIA (*1)

	% of Land >2dS/m		**%Land >4dS/m**		**%Land >8dS/m**	
Sub-District	Ave 91-94	98 survey	Ave 91-94	98 survey	Ave 91-94	98 survey
Gogeldrie	8.6	7.4	0.0	0.0	0.0	0.0
Benerembah South	17.3	32.8	3.6	4.2	1.3	2.2
Benerembah North	30.0	44.4	11.6	9.6	6.3	2.5
Hanwood	33.8	31.1	12.7	18.9	2.9	3.2
Kooba	4.8	10.0	2.2	1.5	0.9	0.0
Yenda	35.3	39.1	16.6	12.2	6.3	0.0
Koonadan	11.0	27.6	8.9	4.0	5.3	2.1
Calorofield	22.2	27.0	11.9	6.0	2.9	1.0
MIA	***22.0***	***27.6***	***8.8***	***8.1***	***2.9***	***1.5***
Wah Wah	53.5	70.1	11.1	12.5	5.1	5.1

(*1) from van der Lely, 1999, values converted from $EC_{1:2}$ data

The median values of salinity for 1998 had increased compared to previous surveys, but the areas with high salinity had decreased. Most of the change was attributed to changed water table conditions, these being at a 10 year low for the MIA. It was concluded that performance modeling of soil salinity in sub-districts requires close analysis to separate out the effects of rainfall and irrigation management factors.

Salinity data such as these have been used for prediction purposes (van der Lely, 1998). The twelve sub-districts referred to have varying histories and duration of high water table (HWT) conditions. The current salinity is the integrated effect the high groundwater conditions in the past. The assessment consisted of two parts. For the first part regressions of soil salinity over time for each water table category to 1.5 metres depth were calculated to make predictions for the No Plan scenario. The second part consisted of a lump sum ground water balance model to describe the relationship between the depth to the water table and recharge and discharge factors. This model provided estimates of changed water table conditions for various LWMP options on water table levels and its aerial distribution. The soil salinity regression of the first part were then applied to these changed conditions to calculate the effect of the plan.

[2] Twelve if the Coleambally sub-districts are also included.

A number of assumptions underpin the above analysis, which forces questions regarding the validity of the approach. The models were reviewed by van der Lely (1999), and two alternative techniques were developed. A statistical optimization model of all data with hydro-geological factors provided the most credible assessment of salinity trends, however it was not suitable for LWMP option evaluation, because it could not be linked to the recharge/discharge groundwater model. The other approach was a process model for the sub-district integrating the groundwater balance and salt accumulation processes. The latter model gave suitable results after rigorous calibration/optimization, but there appeared to be high sensitivity to small changes in key factors.

The above work provided much insight into key factors of lump sum parameter models for sub-districts. A suitable integrated representation of factors such as rice recharge, channel seepage and capillary rise was achieved. Interactions were found to be very complex. Modeling of soil salinity without the back up of regular survey data for verification is unlikely to produce meaningful and credible estimates.

4. Research on Rice Hydraulic Loading

The amount of research carried out to define optimum levels of RHL in a specific landscape situation is limited. It comprises work on water table behaviour in rice areas, and the use of two-dimensional analytical models. Ground water modeling tools potentially useful to analyze RHL are mostly still in a stage of development. Below follows a brief description of the various research efforts and findings.

Effect of Rice on Water Table Distribution

Van der Lely (1981) investigated the relationship between the proportion of land grown to rice and the depth to the water table in the non-rice area. March and September groundwater levels were compiled for 16 years of observations from 40 piezometers in a 2,400 ha area near Griffith. From this the September average depth to groundwater could be estimated from the proportion of land under rice in the previous season and winter rainfall. It was found for the area in question that the water table increased from September until March up to a distance of about 400 metres from rice fields. Beyond this the water table dropped. Evidently, at such distances the supply of rice seepage was less than groundwater discharge by capillary rise. The relative areas of non-rice land at various distances from rice fields were also assessed for three selected years with different rice hydraulic loading. This allowed the compilation of water table distribution as shown at Table 2.

Table 2: Water table distribution for three years with different rice hydraulic loading.

% of land with Water Table	36% Rice	24% Rice	17% Rice
Within 1 metre	34	15.6	6.4
Within 0.8 metres	22	7.2	2.8
Within 0.6 metres	12	3.2	1.1

Whilst questions regarding the methods used may be raised, the work identified the effect of increasing areas of rice on ground water levels for an area with specific hydro-geological properties. However, the work did not result in a quantification of the volumes of seepage involved, or salinity transport.

Two Dimensional Models for Groundwater Flow to Adjacent Area

Following on from the above, van der Lely (1981) used two-dimensional models describing flow from rice fields to adjacent areas. The flow in the surface clay layers to and from the aquifer is assumed vertical. The flow rate in the aquifer of transmissivity T is horizontal, declining with distance from the boundary by an exponential function. Several other assumptions are necessary to enable use of the model, and not all of these could be fully satisfied for the rice situation ([3]). The assumptions are critical for accurate assessments, but it is believed by comparing different solutions for different assumptions reasonable estimates of the seepage rate from rice may be made. For instance, the above two-dimensional model was modified and applied to the Wakool evaporation areas, where the results of seepage rates from the basins were found to be very consistent with results using conventional numerical groundwater modeling techniques (van der Lely, 1988).

Three key input factors exist to calculate the seepage rate per unit length of rice perimeter, the factors T and C described, and the depth to the water table H in the non-rice field at relatively large distance. The seepage rate may be assessed for different assumptions and by different approaches ([4]). The seepage rate may be multiplied by the total perimeter length values for the area in question to obtain total volumes involved in the flow. This procedure was applied to the 2,400 ha area near Griffith referred to above for three years with different rice hydraulic loading. Table 3 shows the outcome of this type evaluation.

Table 3: Volumes of rice seepage for three different levels of rice hydraulic loading (*1)

Volumes of Rice Seepage	17% Rice	24% Rice	36% Rice
Total seepage volume (ML) (*1)	650	740	830
ML/ha/year over whole area	0.27	0.31	0.35
ML/ha/year for Rice Area	1.5	1.3	1.0
ML/ha/year for non-rice area	0.33	0.41	0.54

(*1) This is an average of five different estimation methods over the selected 2,400 hectares

From this it appears that as the rice area is doubled the total volume of seepage increases by less than 30%. The water loss from rice on a per hectare basis is reduced by about one third, and the volume per unit area to be dissipated in the non-rice area per hectare increases by about two thirds.

Numerical Groundwater Models

CSIRO Land and Water developed the SWAGSIM model (Prathapar, 1994). This model considered the use of crops other than rice to "soak up" the rice accessions. A two-layer model including the Shepparton and Calivil Formations was developed and applied to a 3,750 ha area near Griffith. This area has The Shepparton aquifer was assumed to be unconfined. A significant deep leakage component was assumed. Data on crop areas and location over a lengthy period were collated, together with piezometer data and irrigation applications. Simulation of water table behaviour was the basis for calibration and a reasonable but by no means perfect match was found ([5]). The model produced estimates of recharge and discharge values for each model cell, and was capable of simulating the effect of larger or lesser areas of rice in combination with other crops. Required ground water pumping rates to achieve specific water table targets were calculated.

Recharge and discharge rates for the area varied with annual rainfall conditions. For the period modeled the proportion of land with water tables less than one metre varied from 46 to 90%. It was concluded for the area in question that at an RHL of 23% there would be a balance between recharge and discharge. This level of RHL coincidentally was the current situation in the field. The deep leakage component in the model was about half of the recharge volume, which is very significant. The SWAGSIM model did not include a salt transport function, which reduces its value as a sustainability assessment tool.

Other numerical models have been used for land and water management plans, e.g. the SHE model for the Berriquin and Wakool areas, but to date none has been used to evaluate the effect of more or less RHL in a landscape. The main restrictions to the models appear to be related to the difficulty in estimating recharge and discharge values for different crops for the range of soils and management regimes that exist, the large

[3] The assumptions are that the stratigraphy is homogeneous to infinity, the rice field is either infinity, or a circular field, or a long rectangle, and the water level (not pressure level) in the adjacent area is horizontal.
[4] One other method was based on capillary rise estimates for clay soils, and another based on gradients derived from mapped groundwater contours multiplied by transmissivity values.
[5] The Root mean square error in water table predictions was about 0.5 metres.

cell size, and the use of averages within cells. When the salt transport function needs to be added as well it becomes very difficult to get credible answers.

Coleambally Irrigation Area Groundwater Models

The approach to groundwater modeling in this area is discussed at another paper to this conference (Shahbaz et al. 1999). Several models are involved, a regional scale groundwater model based on MODFLOW, and a farm scale model, called SWAGMAN FARM. The groundwater model is the context which describes the spatial distribution of acceptable levels of recharge for sustainability. SWAGMAN FARM is a lump sum parameter model and estimates the excess water application from water delivery data and the total crop water requirements on the farm. The excess is compared to the acceptable recharge volume from the regional model. An optimization of crop areas and other inputs is then possible to find the desirable cropping rotation which reduces recharge and still produces financial returns to the landholder.

The Coleambally approach is attractive since it does not appear to require the same degree of data rigor for key recharge and discharge factors in the main model, allowing larger cell size.

5. Discussion

Salts to the root zone are introduced by irrigation water and capillary rise and exported by surface runoff and leaching. In a homogeneous landscape without gradients used by a perfect rotation of crops the capillary rise effect during non-rice may be compensated by leaching during rice, leaving irrigation water salts as the only problem in the long run. This is the optimistic scenario, in which the degree of rice hydraulic loading would not matter, except that very high water tables in non-rice areas cause water logging ([6]). The sustainability question is how this situation alters when topographical factors are superimposed, or variations in soil permeability and transmissivity occur. EM31 surveys of rice farms have shown significant variation in soil permeability. Does the same still apply, with a perfect rotation the salt accumulation effects are basically compensated by leaching?

The answer is likely to be negative. In depressions salt accumulation will be higher than the potential leaching under lesser gradients. Narrowing of aquifers will force groundwater to the surface in some locations.

Further complications are that some areas in the high water table zone are classified as unsuitable for rice, and some farmers grow a larger proportion of land to rice than others. These factors are likely to result in a net transfer of salts to less irrigated areas. In some districts some areas are not irrigated at all, and the inducement of high water tables puts these areas at risk.

On the other hand, leakage to deep aquifers provides a net leaching over the whole of the landscape in the eastern part of the Riverine Plain, which makes the irrigation areas more sustainable ([7]). The cost to areas where the groundwater moves has not yet been evaluated. Very long time scales are involved.

A non-reversible process of salt transfer between fields appears to occur and may be inevitable. In that context sustainability become a relative concept. How long for? Should we speak in terms of sustainability for the whole area, or just the irrigated part? Management of crop rotations, rice hydraulic loading, and best management practices will delay the process of land degradation of a proportion of land, but it won't be stopped or reversed. All that we are doing is buying time.

For most irrigation areas or districts only ground water pumping solutions can provide sustainability of all the land. Of course, in many LWMP areas these can not be adopted because of costs ([8]).

[6] The net leaching and capillary rise at each location may be the same over the period of a rotation, but the salt transport may still be different, dependent on how water moved up and down through soil peds in saturated and unsaturated states.
[7] Or fully sustainable in some parts, Northern Coleambally may be an example.

To prolong the sustainability of irrigation areas it appears necessary have an objective of keeping groundwater levels in fields at risk of salting at such levels that salting rates are minimized. The fields in question are dry areas, fields irrigated less intensively (with rice or other crops), and depressed areas. What is a suitable water table depth for these areas? In heavy clay soils capillary rise is small with water table depths deeper than about 1.5 metres, however, there is evidence that over time (decades) the soil will still becomes salt affected unless there is leaching. In areas with lighter textured soils the problem is worse and water tables need to be kept deeper, however compensating leaching may occur if the soil is irrigated for crops other than rice.

It is generally believed that the sustainability of irrigation areas is a function of the overall irrigation intensity, or rice hydraulic loading, but this relationship is not clear at all. Is it linear? This is doubtful. Or perhaps, the desirable level of these factors is affected by internal variation in land use. The greater the internal variation, the lower the RHL that should be allowed. In that context areas such as Coleambally or the MIA could be more sustainable than for instance the Berriquin District.

Answers are needed regarding the time scales involved, and how RHL rules affect these time scales. A combination of research efforts appear necessary:

- Development of regional scale groundwater models linked to GIS, to obtain performance monitoring tools to interpret general groundwater trends as a function of climatic and management factors. Such tools however will take time to develop and won't be effective for some time as an evaluation tool for RHL. They would also not be effective without complementary research into recharge and discharge.
- Development of groundwater models of smaller areas including several fields or farms, to answer questions regarding spatial variation at that scale.
- Collection of more knowledge regarding variation in capillary rise and percolation, the effectiveness by which the two processes compensate each other and how this can be influenced by irrigation, crop and land management.

Much knowledge is already available from published research. How can this be used effectively? It is probable that perfect models for land and water management are a myth. Decisions will always be based on less than perfect answers. One of the most difficult issues is when to decide that additional research is not justified

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[x] The costs include downstream costs or engineering works, and may include evaporation areas, or pipe lines where groundwater is too saline for reuse. The benefits relate to the productivity of salt affected land. The latter is often used for less intensive enterprises anyway.

SALT/EFFLUENT DISPOSAL

Development of Guidelines for On-Farm and Community Scale Salt Disposal Basins on the Riverine Plain: Underlying Principles

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Abstract

We describe the background and development of a set of **principles** which is a first step towards guidelines for the effective and environmentally safe use of small scale basins for saline drainage disposal in the Riverine Plains region of the Murray-Darling Basin. At present, there is no framework in place which guides the siting, design and management of on-farm and small community basins. While it is not possible to account for every combination of the complex array of soils, geology and land use in areas where disposal basins may be used, it is certainly possible to define a set of guiding **principles** for responsible basin design and management in the Riverine Plains region of the Murray-Darling Basin. **They should not be considered as regulation or law as they have not received any endorsment from any of the jursidictions they encompass.** In this paper we: (i) identify issues which need to be considered in disposal basin systems; and (ii) define a set of disposal basin **principles** that set out a responsible philosophy towards development and use of small scale (<100 ha) on-farm and community basins.

Introduction

Background

The Murray-Darling Basin is one of Australia's most important water and land resources. Approximately 73% of all water used in Australia is harvested from the Basin (Fleming, 1982) and approximately 80% of land irrigated in Australia (1.8 million hectares) is located within it's boundaries. Approximately 90% of cereal, 80% of pasture, 65% of fruit and 25% of vegetable production in Australia is derived from irrigated agriculture within the Basin (Murray-Darling Basin Ministerial Council, 1987). Meyer (1992) estimated that the annual value of irrigated agriculture in Australia exceeded $4.6 billion (including more than $2.7 billion in export income), the majority of which is from the Murray-Darling Basin. The majority of irrigation occurs in the south-central part of the Basin known widely as the Riverine Plain (Figure 1).

The Basin, in its pre-European state, contained vast amounts of salt which were stored in the soils and groundwater. The use of irrigation, the leakage of water from the associated network of water distribution and drainage channels, and the clearance of deep-rooted perennial plants and their replacement with shallow-rooted annual crops has altered the water balance causing water tables to rise throughout the Basin. This has resulted in mobilisation of the stored salt and, when the water table comes close to the soil surface, soil salinisation and waterlogging result, with detrimental effects on agricultural production. In addition, raised water table levels can increase hydraulic gradients between the groundwater and surface water resources, leading to increased movement of salt to drains, streams and rivers.

To maintain productivity in irrigation areas with shallow groundwater, water table reduction and control is carried out using measures such as sub-surface tile and deep open drains, and groundwater pumping from bores. This, however, creates the problem of disposing of large volumes of saline drainage water.

Disposal Options

While a large range of saline water disposal options for the Murray-Darling Basin were identified some time ago (Evans, 1989), a lot of emotion and misinformation still surrounds the disposal issue. There has always been debate over the environmental, social and economic implications of any disposal option. One thing that is unanimously agreed upon is that adequate and comprehensive saline water disposal is necessary to ensure the continued viability of irrigated agriculture, and to protect the quality of the surface water resources of the Basin. Disposal is therefore one of the most important components of Land and Water Management Plans for irrigation areas.

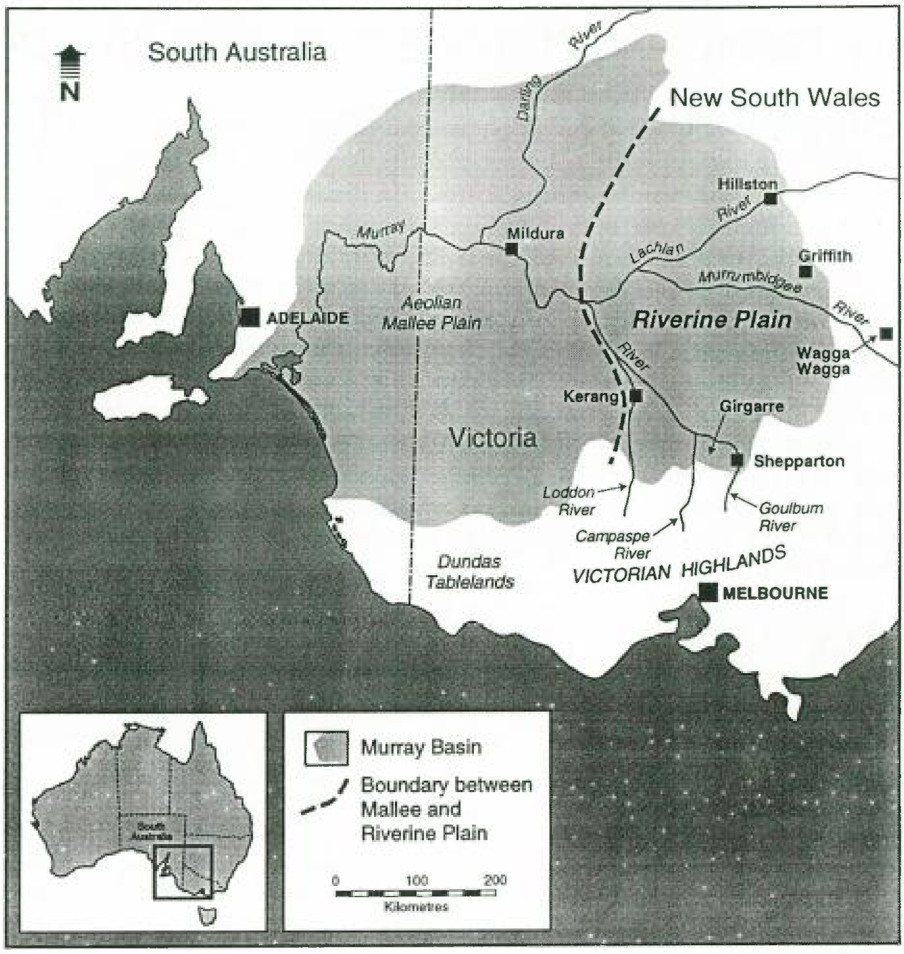

Figure 1: Map showing location of the Riverine Plain of the Murray Basin

There are three main disposal options: disposal to streams and rivers on an opportunistic basis, disposal using a pipeline to the sea and evaporative disposal to land. Some saline water is currently disposed of into river systems in periods of high flows but the salinity of pumped groundwater and drainage effluent is such that unmanaged disposal to rivers and streams on a continuous basis may result in unacceptable downstream impacts. There appears to be a trend in political and community attitudes towards less disposal to the river system. Since the river system has limited capacity to export salt from the irrigation catchment, the remaining options are export using a pipeline to the sea and land disposal to natural or engineered surface storages (saline disposal basins). Previous studies (State Rivers and Water Supply Commission, 1978; Earl, 1982; Gutteridge Haskins and Davey, 1990) have indicated that the pipeline option, compared with other disposal options available at the time, was uneconomic. This leaves evaporative land disposal as the most likely option available, at least in the short to medium term (50 years). As was shown by Evans (1989), saline disposal basins are the lowest cost option for high salinity drainage water.

Overseas experience with land disposal using basins is fairly limited. They have been used in the San Joaquin Valley in California (Tanji et al, 1993; Grismer et al, 1993), along the Indus River in Pakistan (Trewhella and Badruddin, 1991) and have been proposed for the Shapur and Dalaki River Basin in southern Iran (Shiati, 1991). The use of dryland evaporative "sinks" (low points in the landscape which are not ponded with water but have high rates of groundwater discharge) has also been proposed for the Indus Basin in Pakistan (Gowing and Ashghar, 1996).

The Murray-Darling Basin Ministerial Council recognised over a decade ago that disposal basins had a role to play in storing salt if they are properly designed and maintained (Murray-Darling Basin Ministerial Council, 1986). At present there are in excess of 180 saline disposal basins in the Murray-Darling Basin (Hostetler and Radke, 1995).

Saline Disposal Basins - Issues of Size

In the past, use of large regional scale (>100 ha) basins has been the most common approach. These generally accept drainage water from multiple farms and irrigation districts which in some cases may be located many kilometres away (hence salt is exported from the area in which it is produced). These basins most commonly use natural depressions in the landscape (e.g. Lake Tutchewop near Swan Hill), however they can be engineered storages (e.g. Wakool Basin near Deniliquin). Many have occurred by default or have been developed on an ad-hoc basis.

Regional basins were generally developed on the most convenient sites from an engineering standpoint, and often ignored environmental, socioeconomic, aesthetic impacts and any other community concerns. In addition, various unforeseen side-effects (leakage to adjacent farmland, insect and bird problems, odour) experienced by a number of regional scale basins has led in many cases to poor community perception of disposal basins. Moreover, under the Murray Darling Basin Salinity and Drainage Strategy (Murray-Darling Basin Ministerial Council, 1988) severe constraints have been imposed on salt export from a given area. This policy was designed to ensure that the beneficiaries of irrigation are responsible for their own drainage management on the assumption that this would help minimise other environmental effects. While disposal to regional basins will continue in the future, there is a view in environmental protection and other resource agencies that there is a need to depart from the existing "export the problem" mentality. It is becoming mandatory that the option to manage drainage effluent at the source be closely examined before resorting to export.

The above concerns have led to the use of much smaller scale (<100 ha) basins. These can take the form of on-farm basins, which occupy individual properties (such as those being used for new horticultural developments in the Murrumbidgee Irrigation Area) or community basins, which are shared by a small group of properties (such as the Girgarre Basin near Shepparton). The design and management of both types of basin varies widely and currently there are no set guidelines for their use. **It is small-scale on-farm and community basins which are the subject of this paper.**

There are a number of issues related to the choice between on-farm and community basins. With on-farm basins the costs can be more easily passed back to the farmer, the ownership of the basin remains with the beneficiary and the impact is localised. Thus, distribution of costs, especially in terms of lost land, is easier. However, on-farm basins are more difficult to supervise and monitoring systems need to be applied to ensure impacts on non-beneficiaries and the environment are within acceptable limits.

The distribution of costs of community basins presents problems and the siting of such basins may lead to land equity and other legal disputes. Community basins present better opportunities for optimisation of sites and thus can be more easily managed than numerous on-farm basins. The greater technical complexity of drainage transport and basin operation, combined with the complex ownership and management issues, means that community basins require a long term commitment on the part of the participating landowners. Factors such as long-term maintenance, change of land use and ownership, and decommissioning need to be considered before community basins are considered. On the other hand, if other disposal options become available in the future, decommissioning and reclamation of a small number of community basins may be more viable than for numerous on-farm basins.

Integration of Sub-Surface Drainage and Disposal Basin Systems

The aim of sub-surface drainage is to lower water tables to levels which maintain crop productivity (by controlling waterlogging and land salinisation) and minimise salt movement to drains, streams and rivers. Sub-surface drainage is generally carried out with buried horizontal (tile) drains, deep open drains or by groundwater pumping from bores (also known as tubewells) or spear points. The use of tile drains and tubewells for water table control is shown in Figure 2.

Also shown in Figure 2 is surface runoff, which often occurs as a result of irrigation. This is generally collected in shallow surface drains. In general, water quality in these drains is relatively good (EC<1000 µS/cm) and volumes are usually higher than tile drains, particularly following rainfall.

In this paper we use the term "drainage system" to mean any form of sub-surface drainage for water table control, such as tile drains, deep open drains, tubewells or spear points. A number of disposal options for sub-surface drainage are shown in Figure 2. In this paper we are concerned with disposal to on-farm or community evaporation basins. Common to both type of basins is the possible interaction between the basin and the sub-surface drainage system via leakage of water, and hence salt, from the basin. The role that the sub-surface drainage system has in either inducing or controlling leakage is such that it is best to consider both systems together when siting, designing and managing small scale disposal basins.

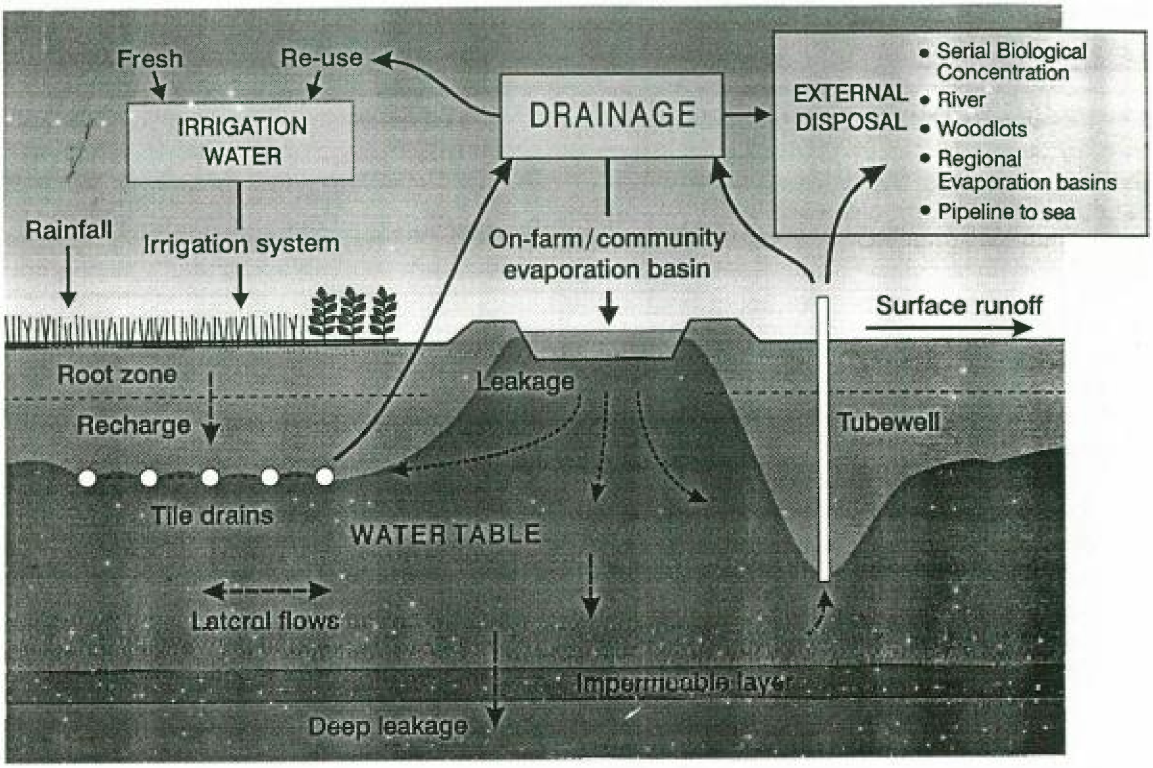

Figure 2: Schematic of sub-surface drainage and disposal basin systems.

Disposal Basins as a Source of Pollution

The primary purpose of a disposal basin is to store and evaporate drainage effluent. Continuous disposal and evaporation will increase the salinity of the stored water. Salt accumulation in the disposal basin is controlled by both the rates of evaporation and leakage of concentrated saline water into the sub-soil and groundwater system beneath the storage.

Since the input drainage water to a basin is derived from the surrounding groundwater system and the disposal basin concentrates the salt, there is merely a change from salt in a diffuse state to a point source. However, disposal basins can also be a potential source of environmental pollution caused by: rising saline water tables leading to waterlogging and salinisation of adjoining lands; greater flow of saline groundwater to the surface drainage systems due to increased local hydraulic gradients; and concentration of agricultural chemicals together with salt.

Where disposal water contains toxic materials, the issue of creating a point source of pollution needs to be carefully considered in terms of danger to wildlife and the surrounding environment (see the recommended national water quality guidelines (National Water Quality Management Strategy, 1992) and guidelines for groundwater protection (National Water Quality Management Strategy, 1995). It should be recognised that, in this instance, the pollutant is a concentrated form of salt, herbicides and pesticides that were already present in the shallow groundwater prior to the development of the basin. Under most circumstances, if the **principles** from this paper are adhered to, chemicals external to the drainage system are not added to disposal basins. The concentration process within the basins may, however, lead to toxicity problems with certain chemicals, especially herbicides and pesticides. However, the potential for herbicides and pesticides toxicity is reduced if surface drainage is excluded from the basin. Nevertheless, a comprehensive chemical screening program of water pumped into a prospective basin is required prior to implementation. This should ensure that no unforeseen toxicity problem occurs as was found with selenium in basins in California (Tanji et al., 1993).

The extent of the above effects is highly variable depending on site conditions and basin design and management. In view of these potential undesirable environmental effects, any use of disposal basins as a useful salt management tool must be carefully considered. The success of any disposal basin depends very much on the local environmental, physical, legislative, economic and social constraints of the specific site. It is important that disposal basins be efficiently designed, properly sited and effectively managed under controlled conditions. If this is achieved, a basin may in fact be seen as an environmental asset (Roberts, 1995).

Principles

The intention of the following **principles** is to define desirable objectives for disposal basin siting, design and management. They should apply unless there are compelling reasons to move away from them. However, it is recognised that in some instances they may contradict each other and so consideration will need to be made as to which is the overriding **principle** in such situations. **It is important to note that they should not be considered as regulation or law as they have not received any endorsment from any of the jursidictions they encompass.**

It is recognised that current legislation permits disposal of some saline water to the river systems under the EC credit scheme of the Salinity and Drainage Strategy and the future use of large regional basins will continue where appropriate. However, **further development of disposal basins may not have the option of river disposal and so the following principles have been developed under the assumption that no export of saline drainage water is allowed from the area of extraction.**

It is also implicitly assumed that best management engineering practices for above ground storages will be employed for siting and design (e.g. Queensland Water Resources Commission, 1984; New South Wales Agriculture, 1999), and that the principles should be considered within the framework of appropriate existing Land and Water Management Plans, State and Federal legislation, and Local Government regulations (viz. EPA Vic, 1994; EPA NSW, 1997), where appropriate.

The geographic area for which these principles have been developed is the Riverine Plains region of the Murrumbidgee and Murray Rivers.

Principle 1

Evaporation basins should be used for the disposal of highly saline drainage effluent, after all potential productive uses have occurred or the water is shown to be economically and environmentally unsuitable for use.
(in order to minimise the volume of disposal, thereby minimising the basin size and cost; and to ensure the best use of water as a resource)

Disposal basins are expensive to construct, will generally occupy land which would otherwise be used for agricultural production or other economic purposes, and usually will not generate any revenue. To minimise the volume of disposal water, and hence the basin size, the basin should only be used as a terminal point for saline drainage water that cannot be used elsewhere. This will also maximise the economic benefits to be gained from irrigation water.

Opportunities for economic re-use of both surface run-off and sub-surface drainage water should be carefully considered, especially if the water is not too saline. Re-use of this water by mixing with groundwater (Bethune et al., 1997) or surface water supplies should be considered. For example, in the Shepparton Irrigation Region there is already in excess of 2650 re-use systems in operation (Irrigation Committee, 1996). For greater levels of salt concentration prior to basin disposal, serial biological concentration schemes should be considered (Heath et. al., 1993).

Principle 2

Salts remaining in a basin due to evaporation should be stored not only in the ponded water but also in the soil and aquifer system below the basin.
(in order to maximise and maintain disposal capacity)

Disposal basins can be used to store salt in a confined area above ground in the basin and in the soil and aquifer system immediately below (in an environmentally responsible manner, see Principle 4). If storage is to be confined only to the basin then leakage must be prevented by lining the basin with impermeable materials such as plastic membranes. Widespread experience has shown that it is very difficult to maintain the impermeability of any type of lining after several years of operation.

There are two problems associated with basins in which leakage is prevented. Firstly, the basin water will continually increase in salinity causing an associated decline in evaporation rate (and hence disposal capacity), therefore necessitating a much larger basin for a given disposal volume. Secondly, precipitation of salts will occur in the basin, which will need to be removed and disposed of. The only situations where this would be appropriate is if salt production of commercial quality is desired, or when export to the sea or to a location where any adverse impacts would be acceptable, and is technically and economically feasible.

A basin in which **limited leakage** is allowed will concentrate salt in the basin water to a much lesser extent and so there will be minimal loss in evaporative potential, and hence disposal capacity. There will not be precipitation of salt in the basin under normal conditions.

It is important to note that leakage from a basin creates a localised concentration of salts in the underlying soils and aquifer system which is simply a redistribution of the mass of salt already stored within the soil and groundwater system. Provided leakage is contained to within the area of the drainage system it does not represent an external pollutant. However, it should be noted that the basin will still have a finite life, but instead of it being only a few years or decades (as in the case of a basin which is not allowed to leak) it may be of the order of hundreds of years.

Principle 3

Salt stored below the basin should remain in the area of influence of the drainage system, or within a specific salt containment area around a basin located outside the limits of a drainage system.
(in order to ensure that salt is not exported to adjoining land outside the drainage area benefiting from the drainage water disposal)

The principal consideration in disposal basin design is the containment of salts accumulating in a basin. The salt leaking from the basin needs to be kept local so that it does not salinise adjacent land and to ensure that there is no export of salt from the area. Thus, it is preferable that a basin be located within the area of influence of the drainage system. The option to locate a basin within the area of drainage influence should always be considered first. This will result in the salt remaining in the area from which it was extracted and ensures that ownership of the disposal, and its control, remains with the beneficiaries of the drainage system.

Under exceptional circumstances where no suitable disposal site exists within the drainage system, the basin may be located outside the area of influence of the drainage system. However, this effectively exports the salt from the area of extraction and so there must be more stringent restrictions on design, location and management to ensure that the salt stored in and below the basin will be guaranteed to be contained in close proximity to the basin. Increased buffer zones (referred to as impact or attenuation zones in environmental protection agencies) and effective interception drains or tubewells will be required in these cases.

If well managed, a fully operational drainage/basin system can contain salt within the drainage system. However, if the drainage system or basin is no longer required or should other salt disposal options become available in future, salt containment may be compromised. Therefore, basins require a decommissioning plan, which should be developed before implementation of the basin. These sites should be registered and managed under existing EPA guidelines (EPA NSW, 1997; EPA Vic., 1994), if appropriate, or other specifically formulated guidelines.

Principle 4

Leakage from a basin should not pollute groundwater with existing or potential beneficial use.
(in order not to pollute usable groundwater resources)

Leakage to the groundwater below the basin is only acceptable if the leakage will not pollute a resource beyond a defined impact (attenuation) zone, as measured by change in usability (usefulness) of that resource. To minimise the risk of polluting good quality groundwater, disposal basins should, if at all possible, not be sited in such areas.

Broad indications of groundwater quality can be obtained from various maps covering the Riverine Plain (Woolley, 1991; Woolley et al, 1992; Williams and Woolley, 1992; Woolley and Williams, 1994 and O'Rorke et al, 1992). While there is no agreement between the States regarding the exact definition of beneficial use, these maps broadly indicate water uses which are suitable for the range of salinity classes presented. However, it should be noted that these maps are on a very broad spatial scale. Hence, more detailed local information should be acquired as part of site investigations for any prospective basin to assess any site specific changes and to determine the potential impact of these changes on the resources and bio-diversity of the area.

Principle 5

Water stored in disposal basins should not be released to surface drainage systems.
(in order not to pollute fresh surface water resources)

For on-farm and community basins located on or close to the areas where the drainage is generated, there is no real advantage in allowing periodic emptying of such basins to surface drains or streams. The historical practice of periodic emptying of regional basins located near rivers during high river flows is not appropriate for these types of basins.

Emptying of basins into surface drainage systems introduces high levels of salt to flows that would otherwise be replenishing surface water bodies with fresh water. Also, periodic release sends an inappropriate signal to basin users that the use of their basin and drainage management system will be periodically relaxed and thus a high level of irrigation and drainage management is not required. Moreover, this may also create opportunities for other toxic pollutants to enter the surface drainage systems.

If a basin becomes full then drainage pumping must cease, unless there is serious waterlogging in the farmland. In this situation the possibility of pumping directly to the surface drainage could be considered - this is better than releasing the highly saline basin water to the surface drainage system. For well designed and managed basins there should be no need for emptying. The effect of extreme rainfall periods may result in slight waterlogging but is generally a second order effect in comparison to poor irrigation or drainage system management.

In extreme floods, it may not be possible to guarantee that basin water will not enter the surface water system, or that river or local runoff water will not enter the basin. The basin should be designed to ensure that flooding does not affect the basin except in extreme events.

Principle 6

Basins should be sited, designed and managed to minimise environmental, socioeconomic and aesthetic impacts
(in order to ensure that detrimental side effects of basin use are minimised)

Various side effects, such as local salinisation of land around the basin, insect and bird problems, poor aesthetics and unpalatable odours, have led to a poor image of disposal basins with the general public. This community dislike is exacerbated where the basin is not within the property boundaries of those who benefit from it primarily by using it for disposal of their drainage water.

Basins can be designed and managed to be a community and ecological resource (e.g. Roberts, 1995). To gain greater community acceptance, disposal basins should be sited and constructed in a manner that enhances the local environment by providing wildlife habitat (e.g. bird nesting areas) and reducing the visual impacts by planting trees in buffer zones around the perimeter and installing flatter batters to minimise erosion and exposure of bare soil. On the other hand, adverse environmental impacts on the community surrounding a disposal basin may arise from breeding of insects (e.g. mosquitoes), increased bird population feeding on farmers' crops and airborne salt blown from dried out areas of the basin. These impacts should be minimised in all cases. This will be assisted by not allowing the siting of basins near residential and business areas, community facilities such as schools, hospitals and sporting grounds, and other sensitive land uses.

As discussed in the Introduction there are a number of other socioeconomic issues related to the choice between on-farm and community basins. These include: distribution of drainage costs back to the beneficiaries, quality of basin management, land equity and other legal aspects, the need for long-term commitment to basin operation, and the ease of decommissioning.

As with most engineering structures, there are environmental risks associated with the use of saline disposal basins. The movement of salt (and possibly toxic materials) outside the drainage or salt containment area poses the greatest threat. As described above, this can be minimised by siting basins within the drainage system or within a well buffered containment area (both employing interception drains or tubewells). The risk that salt export poses to water quality, stream health and bio-diversity of surface waters can be further reduced by not allowing siting of basins close to streams, rivers, water supply channels, and flood-prone areas.

All basins should have a management plan which clearly identifies the off-site risks posed by the basin and the monitoring which will be undertaken to ensure basin integrity. Furthermore, all basins should have a decommissioning plan which details the likely off-site impacts of no longer using the basin and/or the associated drainage system.

Concluding Remarks

These principles are by nature generalised and non-specific, however by adhering to the six principles above, evaporation basins should provide a safe and effective tool for the disposal of saline water in the Riverine Plain. The use of saline disposal basins needs to be within the framework of Land and Water Management Plans and conform with existing environmental guidelines and legislation. The above principles set out the most desirable set of conditions for the design, siting and management of evaporation basins, however there may be exceptional and unforeseen circumstances or issues not identified or covered by these principles. As such, the use of saline disposal basins should be accompanied by policies of risk assessment and management.

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Characterisation and ranking of saline disposal basins in the Murray Basin of Australia

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Introduction

One of the most critical parts of all land and water management strategies in the Murray-Darling Basin is the disposal of large volumes of saline effluent from groundwater interception schemes and drainage effluent from irrigation regions. Saline disposal basins have been used since 1917 for disposal of such water on the land surface in topographic depressions in the landscape, salt lakes and freshwater lakes. The number of disposal basins in the MDB is currently estimated to be about 190 (Evans, 1989) but this number is on the rise. Concerns have been raised about the possibility that saline disposal basins can increase salt leakage into aquifers and ultimately into the river system (Chambers et al., 1995). The ever growing number of disposal basins, increasing demands for disposal capacity and potential environmental impacts disposal basins may have means that we need to be able to assess and compare a large number of disposal basins across the entire Murray Basin in a consistent, efficient and timely manner. This will allow for effective management decisions to be made within reasonable time frames.

A small number of disposal basins have been studied in detail using hydrochemical and modelling techniques, e.g., Lake Ranfurly (Narayan and Armstrong, 1995), Lake Tutchewop (Simmons and Narayan, 1998), Noora Basin (Watkins et al., 1991) and Wakool (DWR, 1988). These studies have provided insight into the key hydrogeologic processes that affect basin leakage, subsequent impact on underlying aquifers as well as surrounding land and streams. In addition, the factors that affect disposal capacity and storage potential are well understood. The development of generic criteria that make use of knowledge gained from extensive and detailed site-specific investigations provides a framework for rapid characterisation, ranking and inter-basin comparisons. In this way, we can readily identify "safe" and "at-risk" sites within a large group of disposal basins in relation to some particular objective e.g., minimum leakage. This would allow us to quickly identify sites for potential further storage, compare performance of basins in a relative sense, identify disposal sites which require additional information and also those at which we might look at a little more closely in terms of negative impacts.

Hoxley (1993) developed criteria for ranking a small number of disposal basins within Victoria. However, a basin wide comparison of disposal systems has not been undertaken. In this study, we compare on a standard basis major disposal basins throughout the Murray-Darling Basin in New South Wales, Victoria and South Australia. Many of the more notable basins detailed in Evans (1989) were especially selected for the study. The criteria used within this study are an extension of those developed by Hoxley (1993). In addition, new characterisation criteria are developed to reflect the potential leakage caused by density-driven convection (Simmons and Narayan, 1997) as well as to include time scales for return flows to the river system. This study also employs a series of separate characterisation and ranking objectives. In this way, several disposal basin ranking lists for each defined ranking objective are produced and a basin's overall performance score can be linked directly to specific attributes of the system. This is more informative than a single ranked list produced using a "lumped sum" approach that makes it difficult to assess what aspect of a disposal basin system makes it either a "good" or "bad" site. In principle, the methods developed in this paper can be easily applied to all disposal basin sites within the Murray Basin using an automated framework that is updated continually as more data becomes available for the disposal basins.

Site Selection and Ranking Methods

It was necessary to firstly select an appropriate number of disposal basins, "characterise" them and then rank them. Two criteria were employed for basin selection. The first was that a reasonable number of basins should be selected from each of New South Wales, Victoria and South Australia. Secondly, given the inherent hydrogeological differences between the Mallee and Riverine sections of the Murray Basin, an attempt was made to balance the number of sites between these two regions. An examination of the inventory of saline disposal basins (Hostetler and Radke, 1995) reveals an obvious bias towards disposal basins in the Mallee region, making it more difficult to choose suitable disposal basin sites in the Riverine Plain region.

Saline Diposal Basin Selection

28 sites were selected from the three major states that encompass the Murray-Darling Basin and also host a number of disposal basin sites. They were selected from the inventory of saline disposal basins (Hostetler and Radke, 1995). Several of these sites, as starred(*) below, were previously identified by Evans (1989) as the more "notable" basins within each state. They are:

South Australia:	Noora*, Katarapko Island*, Disher Creek*, Berri*, Bulyong Island*, Loveday*, Woolpunda, K Country.
Victoria:	Tutchewop*, William, Hawthorn*, Ranfurly*, Wargan*, Lamberts Mer-3, Karadoc Swamp, Yatpool, Woorinen (Murrawee), Woorinen (Holloways), Tresco (Golf Club), Tresco (Round), Girgarre.
New South Wales:	Wakool*, Rufus River*, Mourquong*, Holland Lake, Fletchers Lake, Farm 1061, Farm 1068.

Ranking Objectives

Given a set of disposal basins, a multitude of rankings may be possible depending upon what is being assessed and therefore the criteria the ranking is based upon. It was therefore necessary to define clear "ranking objectives". Five ranking lists were produced based upon each of the following ranking objectives (Simmons et al., 1999)

1. Minimum leakage and salt loss from basin
2. Maximum capacity for storage of water
3. Minimum leakage and salt loss & maximum capacity (combining 1 and 2)
4. Minimum impact on River Murray, neighbouring streams and other environmentally sensitive land
5. Minimum loss of water and salt, maximum capacity and minimum impact on River Murray (combining 1, 2 & 4)

It is clear that leakage and salt loss are prime issues that must be addressed in the potential impact of any disposal basin. For many sites, leakage inadvertently becomes a major part of the disposal mechanism. However, where possible, leakage should be minimised especially to protect groundwater quality in regions where underlying groundwater is not overly saline already. Another critical issue that must be addressed is determining sites that could be used for potential further storage. Potential impacts on neighbouring streams, environmentally sensitive land and the River Murray system were also considered. Furthermore, hybrid combinations of the above ranking objectives were examined e.g., a ranking which considered minimum leakage of salt as well as maximum capacity to store water. A "lump sum" total score was also produced that included all ranking objectives.

Characterisation Parameters and Data

Several key variables were identified in order to develop a quantitative method of characterising the selected disposal basin sites before ranking them. The following parameters were used as characterisation variables.

- Potential leakage (head difference between basin and groundwater) *(m)*
- Vertical hydraulic conductivity *(m/day)*
- Salinity difference between basin and groundwater *(TDS g/L)*
- Ratio of inflow volume to evaporation volume
- Capacity of basin *(ML)*
- Distance between basin and streams/River Murray in the Murray trench *(m)*
- Horizontal hydraulic conductivity *(m/day)*
- Viability (hydraulic pulse transmission time to River) *(years)*
- Size of basin *(ha)*

All data used in this ranking exercise was collected from the inventory of saline disposal basins (Hostetler and Radke, 1995). This inventory documents all known disposal basins in the Murray Basin. Where information was not available in the inventory, an attempt was made to find that information from individual reports on the disposal basin under consideration. However, in several cases, the information could not be found. This then created the problem of an incomplete data set and how to most appropriately deal with such cases within the characterisation and ranking framework. Information that could not be determined from the inventory or reports is inserted as "unknown".

Ranking Methods

To assist in the ranking process, an Excel spreadhseet was developed. Saline disposal basin data could then be converted into a score on a relative scale. The basic methodology incorporated in the study was that for each criteria, a point scale was assigned from 0 – 10, with 0 (or lowest score) representing a "best" site in terms of that criteria and 10 (or highest score) representing the "worst" site. Any particular ranking objective could consist of more than one criterion or field. For example, minimum leakage and salt loss from the disposal basin would be a function of several parameters including the difference in head between the basin and groundwater, the vertical hydraulic conductivity and salinity difference between the basin and underlying groundwater. A summary of the score tables is given in Table 1. A brief description of the variables is provided below. Further detail may be found in Simmons et al., (1999). It is worth pointing out that any basin for which information is unknown, a points value of 10 is assigned. This places the basin in the "worst" or poorest performing category. This may or may not be the case but in the absence of knowledge or data for the criterion, it is better to assume that the basin is in the poor performance category by assigning highest points value to it rather than to assume that the converse is true. This could potentially prompt further data collection to revise the scores in an effort to reduce the score assigned to the basin.

Points	*Potential Leakage $h_{ba} - h_{gw}$ (m)*	*Vertical hydraulic conductivity (Kv) (m/day)*	*Salinity difference between basin and groundwater (g/L)*	*Ratio of inflow volume to evaporation capacity*	*Basin capacity (ML)*	*Distance times trench (0 or 1)* (m)*	*Hydraulic Pulse transmission time (years)*
0	<(-5)	-	<0	-	-	0	$t=\infty^{+}$
1	(-5) ~ (-2)	<0.00001	0~1	<0.2	>10000	Dis.>25000	time >1000
2	(-2) ~ (-1)	0.00001~0.0001	1~5	0.2~0.5	5000~10000	20000~25000	500~1000
3	(-1) ~ (0)	0.0001~0.001	5~10	0.5~0.8	1000~5000	15000~20000	300~500
4	0 ~ (1)	0.001~0.01	10~20	0.8~0.9	500~1000	10000~15000	200~300
5	1~2	0.01~0.05	20~30	0.9~0.95	100~500	5000~10000	200~100
6	2~3	0.05~0.1	30~40	0.95~1.0	50~100	2000~5000	50~100
7	3~5	0.1~0.5	40~50	1.0~1.05	10~50	1000~2000	25~50
8	5~10	0.5~1	50~100	1.05~1.1	5~10	500~1000	10~25
9	> 10	>1	>100	>1.1	<5	<500	<10
10	Unknown	Unknown	Unknown	Unknown	Unknown	Unknown	Unknown

* If the basin is in the Murray trench, the effective distance is distance × 1. If the basin is out of the Murray trench, then 0 is given.

\+ Indicates that flow is away from the river. Effective transmission time is infinite.

Table 1. Scores assigned in the basin characterisation process.

Potential leakage

The potential for leakage can be assessed by considering the hydraulic head difference between the basin and the surrounding groundwater system. The critical parameter is $h_{ba} - h_{gw}$, where h_{ba} is the hydraulic head in the disposal basin and h_{gw} is the hydraulic head in the groundwater system. If $h_{ba} - h_{gw} < 0$, this indicates that flow is likely to be into the disposal basin (i.e., the disposal basin acts as a discharge feature). Conversely, if $h_{ba} - h_{gw} > 0$, this indicates that groundwater flow is likely to be directed out of the basin (i.e., the basin acts as a recharge feature).

Vertical hydraulic conductivity and salinity difference between disposal basin waters and groundwater

Whilst horizontal leakage is a function of hydraulic head drop between the basin and the hydraulic head in the aquifer at some point away from the disposal basin, for many basins a significant proportion of the leakage can occur through the basin floor. It is intuitively obvious that vertical hydraulic conductivity of underlying aquifer material or basin sediments must play a role in the resultant vertical leakage processes. In addition, Simmons and Narayan (1997) showed that vertical leakage of salt could be enhanced through a process called free convection, which is a density driven process. The key driving forces for this vertical leakage are the vertical permeability of the aquifer material (or hydraulic conductivity) and the salinity difference between the basin of higher salinity and the groundwater system of lower salinity. In any case, increasing the vertical hydraulic conductivity will enhance salt leakage and the potential for density-driven convection and increasing the salinity difference will enhance diffusion (as per Fick's Law of diffusion) and also promote the onset of even more rapid leakage through convection. A lower value of salinity difference between basin and groundwater also indicates better "compatibility" between basin waters and the groundwater in the region, a situation that is considered more favourable from environmental risk and impact view points.

Ratio of inflow volume to discharge volume

The ratio of inflow volume to outflow or discharge volume is a primary measure of a basins capacity to store inflow water without overtopping. Inflow is primarily the drainage disposal water. Evaporative outflow volumes are calculated using the surface area of the lake and the evaporation rate.

Basin capacity

In addition to the ratio of inflow volume to discharge volume, another indicator that should be considered is basin storage capacity. A low ratio of inflow to discharge volume is considered good, but might also indicate that the lake doesn't have significant storage potential in the first place and as such inflows are minimised. Even a very small disposal basin can have a low inflow to outflow ratio. The area of the basin could be used as an indicator, but has already been used in the evaporative volume calculations for basin discharge. Another useful indicator is basin volume or holding capacity.

Distance between basin and streams, rivers or environmentally sensitive land

A basin located in close proximity to the River Murray or other streams and environmentally sensitive land is considered to be a poorer site in terms of its relative environmental risk. In addition, whether the basin is located inside the Murray trench or outside of it is also considered. If the basin is located within the Murray trench, flow is generally directed towards the Murray River. The distance between the basin and the river is used to allocate a point score to that basin. Conversely, if the basin is located outside the Murray trench, the basin is assigned a 0 point score, since flow would not be directed towards the River Murray.

Hydraulic pulse transmission time

As discussed in Evans (1989) over time, a perched groundwater mound may develop beneath a disposal basin resulting in leakage to deeper aquifers and then the subsequent transmission of a hydraulic pulse and solute pulse through the aquifer to a lower hydraulic potential discharge location. In most cases in the Murray-Darling Basin, the discharge location is the Murray River. The rate of movement of the hydraulic pulse can be calculated using Darcy's Law. The approximate time taken before first arrival at river of actual water disposed in basin can be found using the simple distance, velocity and time calculation:

$$\textit{Time (years)} = (\textit{Distance to river})/(K_h \times \Delta h_{b/gw}/\textit{Distance to river})/365 \qquad (1)$$

where K_h is the horizontal hydraulic conductivity of the aquifer (m/day), $\Delta h_{b/gw}$ is the horizontal hydraulic head drop between basin and river (m) and the distance to the river is measured in metres. It should be noted that this time is significantly higher than the approximate time before displacement of existing regional groundwater occurs.

Results

For each of the ranking objectives, a total score was calculated by forming a weighted average of the criteria that are used to form that objective. For example, to calculate a total score for the objective of *minimum leakage and salt loss* from the basin, three criteria, namely, potential leakage, vertical hydraulic conductivity and salinity difference between basin and groundwater are used. For each basin, a score out of 10 was assigned for each of these three criteria. An average was formed to provide a total score also out of 10 for the basin. The basins were then ranked based upon that total score. Table 2 provides the list of ranked disposal basins for all five ranking objectives. Whilst there is an obvious range of total scores for each objective, it should be noted that there are often several bands observed where a group of basins all received the same total score. This makes it hard to distinguish between basins in that band. What is useful, however, is to be able to then look back at the data which was used to provide the total score and to identify at a glance what attribute lead to the score the basin ultimately received.

Discussion and Conclusions

It is useful to firstly provide some general comments on the ranking system. When the ranking system was designed, a concerted effort was made to make data fields independent from one another. It is inevitable, however, that certain fields will be dependent upon one another. For example, in the category of inflow to outflow volume ratios, the area of the basin was used to compute the total discharge flux as a result of evaporation. The total holding

Increasing score indicates a poorer ranking

Scenario 1		Scenario 2		Scenario 3		Scenario 4		Scenario 5	
Fletchers Lake	2.7	Woorinen murr.	1	Fletchers Lake	2.6	Woolpunda	0	Fletchers Lake	1.9
Hawthorn	2.7	Wargan	2	Katarapko	2.8	Noora basin	0	Tresco Golf	2.0
Holland Lake	3.0	Lake Tutchewop	2	Tresco Golf	2.8	Rufus	0	Woorinen murr.	2.3
Tresco Golf	3.0	Katarapko	2	Tresco (Round)	2.8	Fletchers Lake	0	Noora basin	3.0
Tresco (Round)	3.0	Bulyong Island	2	Woorinen murr.	3.2	Holland Lake	0	Katarapko	3.1
Yatpool	3.0	Karadoc Swamp	2	Lake Tutchewop	3.8	Lamberts Mer-3	0	Holland Lake	3.3
Katarapko	3.3	Woorinen Holl.	2	Hawthorn	4	Wargan	0	Wargan	3.4
Noora basin	3.3	Tresco Golf	2.5	Woorinen Holl.	4	Wakool	0	Lake Tutchewop	3.4
Girgarre	4.7	Tresco (Round)	2.5	Noora basin	4.2	K Country	0	Lamberts Mer-3	3.6
Berri	4.7	Fletchers Lake	2.5	Lake William	4.2	Woorinen murr.	0	Hawthon	3.7
Rufus	4.7	Loveday	3	Bulyong Island	4.4	Tresco Golf	0	Lake Mouquong	4.3
Woorinen murr.	4.7	Lake William	3	Holland Lake	4.6	Lake Mouquong	3	Tresco (Round)	4.3
Farm 1061	4.7	Lake Mouquong	3.5	Karadoc Swamp	4.6	Lake Tutchewop	3.5	Woolpuda	4.6
Lake William	5.0	Lamberts Mer-3	4.5	Lake Mouquong	4.8	Berri	4	Rufus	4.6
Lake Tutchewop	5.0	Noora basin	5.5	Wargan	4.8	Loveday	4.5	Karadoc Swamp	4.7
Disher	5.0	Berri	6	Lamberts Mer-3	5	Disher	4.5	Loveday	5.1
Woorinen Holl.	5.3	Hawthorn	6	Berri	5.2	Farm 1068	5	Berri	5.1
Lamberts Mer-3	5.3	Woolpunda	6.5	Loveday	5.4	Girgarre	6	Woorinen Holl.	5.1
Lake Mouquong	5.7	Disher	6.5	Disher	5.6	Yatpool	7.5	Disher	5.3
Bulyong Island	6.0	Holland Lake	7	Yatpool	5.8	Katarapko	8	Lake William	5.3
Farm 1068	6.0	Lake Ranfurly	7	Girgarre	5.8	Hawthorn	8	Bulyong Island	5.9
Lake Ranfurly	6.0	Girgarre	7.5	Woolpunda	6.4	Karadoc Swamp	8	Yatpool	6.1
Woolpunda	6.3	Rufus	9	Rufus	6.4	Lake Ranfurly	8.5	Girgarre	6.4
Karadoc Swamp	6.3	Wakool	9.5	Lake Ranfurly	6.4	Bulyong Island	9	Farm 1068	6.6
K Country	6.7	K Country	9.5	Farm 1061	6.8	Tresco (Round)	9	Lake Ranfurly	7.0
Wargan	6.7	Farm 1061	10	Farm 1068	7.6	Woorinen Holl.	9	Wakool	7.0
Wakool	6.7	Farm 1068	10	Wakool	7.8	Farm 1061	9	K Country	7.0
Loveday	7.0	Yatpool	10	K Country	7.8	Lake William	9	Farm 1061	7.1

Table 2. Disposal basin ranking lists for ranking objectives 1-5. *Scenario 1*: Minimum leakage and salt loss from basin, *Scenario 2*: Maximum capacity for storage of water, *Scenario 3*: Minimum leakage and salt loss & maximum capacity (combining 1 and 2), *Scenario 4*: Minimum impact on River Murray, neighbouring streams and other environmentally sensitive land, and *Scenario 5*: Minimum loss of water and salt, maximum capacity and minimum impact on River Murray (combining 1, 2 & 4). Weighted average scores for each basin are given out of 10. Low scores indicate a good ranking and higher scores a poorer ranking.

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capacity criterion is also a function of area as well as basin depth. The interdependencies cannot be avoided but do mean that some "doubling up" effects might be encountered resulting in higher scores in some cases.

The number of "unknown" fields is still an issue that biases results in the ranking lists. The resources required to fill all data fields completely and accurately would be enormous. Where data was not available, it was felt assigning a higher score was more appropriate than assigning a lower score. This means that a basin might receive an unfair high score in a ranking but that further data collection or measurement is necessary to move that basin to a more favourable position within a ranking list. Given the likely errors in some of the collected data and the missing data in some fields, it can also be difficult to determine which basins suffer from "real" physical problems and those which we simply do not know enough about. Nevertheless, the basin scores presented in this report are considered to be representative of the potential risk to the environment presented by a disposal basin under each ranking objective. For all cases, the higher the score, the greater the risk. Whether a high score is due to a lack of data or potential physical problems may not be apparent until a detailed investigation of the data used to form the ranking list is undertaken. The ranking lists produced here help to identify the sites that we might choose to look at more closely to make such distinctions. A review of the objectives for a ranking exercise must be undertaken before any further data surveys are conducted specifically to upgrade a ranking list.

There are still various factors that must be considered in sustainable management of a disposal site, not all of which are simply physical factors. For example, current and future use of the disposal basin, the surrounding environment and its environmental value, social and political attitudes towards the use of the basin not to mention the economics and costs involved in any decision. These factors will also be a critical part of any decision making process and in this light, the rankings provided in this report only provide a first order feel for the different disposal basins and how they compare to each other. However, the characterisation and ranking methodology developed is a relatively quick and simple method for comparing basin performance for a large number of basins. It helps to rapidly identify basins which we might look at in further detail or those that more data might be collected for. The developed methodology could be applied to many more disposal basin sites within the Murray Basin preferably within an automated framework and be updated continually as more data becomes available.

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Regional planning for disposal basins on the Riverine Plains: Testing a GIS-based suitability approach for environmental sustainability

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Background

The majority of irrigated agriculture in the Murray Darling Basin (MDB) occurs in the Riverine Plain of NSW and Victoria. The viability of irrigation in this region is closely linked with the management of high water tables and disposal of the resultant saline drainage effluent. Approximately 180 evaporation basins have been located in the MDB in natural depressions or engineered in convenient locations (with exceptions) on an ad hoc basis. There is increasing pressure on landholders to store salt at the source where the benefits of drainage are being derived rather than export the salt elsewhere into the river system or to regional disposal basins. This has led to the increasing use of small-scale on-farm (single landholder) and community (small number of landholders) disposal basins.

Different irrigation areas have different policies towards the choice of disposal strategy and the division between export outside the irrigation area, and the storage in regional, community and on-farm disposal basins. These different policies have not been uniformly developed nor have they been based on uniform guidelines for the responsible siting and use of on-farm and community disposal basins. Such guidelines are currently being developed and only recently has a set of principles been produced to underpin these (Jolly *et al.*, 1999). Even if they had been developed uniformly, there are likely to be differences in disposal strategies due to the differences in physical characteristics of the different irrigation areas.

The key planning issue for saline disposal basins is their siting. Whilst there are obviously siting issues with respect to odours, aesthetics, insects etc, there are also issues with respect to environmental sustainability. It is this latter issue that forms the basis of this paper. If a site is not suitable for a disposal basin, it is difficult and expensive to engineer and manage the basin sustainably. However, such sites do not occur uniformly and planning a suitable disposal strategy needs to account for the variation in availability of suitable sites.

The key issue in siting is leakage, both rate and impact. There is an expectation that basins will leak and it is acknowledged that a small amount of leakage up to 1mm/day is beneficial (Muirhead *et al*, 1997). However, excessive leakage can have adverse impacts on ground and surface water resources. The environmental impacts include degrading a potential groundwater resource, increased salt discharge in the stream network and land salinisation in adjacent areas. Not only is it important to ensure that the leakage rate is low, but also that the adverse impacts of any leakage (and associated contamination) is minimised by not siting over good quality groundwater nor allowing any contamination plume to move outside the vicinity of the disposal basin.

Thus, different policies for disposal may be needed between, and even within, irrigation areas. While export of salt may be philosophically undesirable, it may be necessary if there are insufficient sites available to store the amount of disposal water. Similarly, if there is a strong likelihood that most land holdings have no suitable land for disposal basins, any policy towards on-farm basins does not make sense. An objective method for the decision process is useful to show why different decisions have been made, while maintaining a responsible policy towards environmental sustainability. Thus, while siting of disposal basins requires specific on-site investigations, overall planning may benefit from a broad scale suitability analysis.

This paper addresses the question of whether such a broad-scale analysis, while useful for planning, is actually feasible. To enable us to do this, we have in this study assumed some likely suitability criteria, suggested a method for combining suitability criteria and applied this to some test areas. However, the overall feasibility is likely to be dependent on whether appropriate data is available, and how closely this data is directly translatable into suitability criteria and less on the choice of criteria. In trying to develop an objective methodology that can be applied across the Riverine Plains, it is necessary to use datasets available across the Riverine Plains at a regional scale. It may be possible to use finer resolution data within irrigation areas to provide a better analysis.

Thus, the specific aims of this study were to:
(i) Investigate the suitability of datasets available for the whole Riverine Plains for planning of disposal basin siting

(ii) Investigate whether these datasets when used for a suitability analysis can aid decision-making on disposal strategies
(iii) Assess whether finer resolution data available within irrigation areas would improve the analysis.

Methods

The Murrumbidgee Irrigation Area (MIA) and Shepparton Irrigation Region (SIR) were chosen to develop and test a suitability analysis at a regional scale (~1:250,000 or ~250m grid resolution). The software used for the analysis was the ArcInfo (ESRI, 1996) Geographic Information System (GIS). All GIS coverages were converted to grids in Universal Transverse Mercator projection, and classes were then amalgamated into a single suitability map using a relational overlay process in the Grid module of ArcInfo. Interpolated surfaces were generated using the Topogrid module.

Suitability Criteria

Much has been done in the area of land suitability evaluation, particularly by the FAO (1976) in standardising terminology and procedures. This analysis was adapted from methods described fully in Dent and Young (19[illegible]). In essence, suitability ranges were defined separately for each of the inputs, which were then combined according to relative importance (priorities defined by the user) to derive a manageable number of overall suitability classes from optimal to not-suitable. Without recognised guidelines on what criteria, or acceptable ranges, are needed for siting evaporation basins, those used here are considered as a first approximation that are flexible and may be refined as new data become available.

Data were identified based on availability and critical factors that reduce leakage and associated risks to the surrounding environment. The criteria for siting evaporation basins were:

1. **Low hydraulic conductivity** (heavy impermeable soils) is most important and has the greatest control on leakage. A rate of 1mm/day found to be beneficial to basin function (Muirhead *et al.*, 1997) was taken as the optimum with increasing rates being more detrimental. Other classes were arbitrarily defined for the suitability analysis as: 1-3mm/day acceptable and > 3mm/day not suitable.
2. **Shallow watertable depth** was taken as 0-2m being optimum, 2-4m was acceptable and beyond that was not acceptable. This criterion was used to avoid significant gradients away form the basin being developed and hence the migration of plumes away from the basin.
3. **Poor groundwater quality** (existing salinity or degradation) is preferable to minimise the risk of contaminating fresh water resources. Ranges were defined as >14,000 mg/l TDS being optimal, 3,000 – 14,000 acceptable, 1,500 – 3,000 marginal and <1,500 unsuitable, as it is too potable to be put at risk.

Others, such as low slope or existing depressions (minimising engineering costs and disturbance to soil profiles) and safe buffers to streams, irrigation channels and towns (reasonable distance dependent on soil hydraulic conductivities and other factors) could have been used in a GIS analysis but were not considered here.

Available Data

To explore what is achievable at a regional scale (1:250,000) it was necessary to know what consistent data were available covering the entire Riverine Plain. There are few data sets available that are; a) relevant to siting disposal basins, b) seamless coverages across the Riverine plain, c) appropriate scale or resolution, d) currently obtainable and affordable and e) digital. Those found were:

a) **Soil-landforms of the MDB** (Bui *et al.*, 1998) is a coverage of the MDB which predicted soil landforms and dominant Principal Profile Forms (PPFs) using land systems and other soil surveys and included geomorphic elements of the MIA. It was linked via PPFs to a soil attribute look-up table (McKenzie and Hook, 1992; McKenzie *et al.*, 1999) to soil properties, including hydraulic conductivity. However, closer inspection showed that these estimates were not particularly useful (for reasons discussed later), except to provide a broad classification of low to high permeability soils.
b) **GIS of the Murray Hydrogeological Basin** (BRS, 1999) provided interpreted map data for watertable depth and salinity. The watertable depth data generally had contours no better than 5 m and the ranges of the above watertable classes were changed to <5m and >5m.
c) **Geodata 9 second Digital Elevation Model** (DEM), (AUSLIG, 1996). Elevation data was used to provide terrain attributes of slope and depressions to further discriminate between classes. Artifacts present in the DEM suggested highly variable data quality. To avoid propagating errors in derived grid layers the decision was taken to exclude the DEM from this analysis.

Finer resolution data was sought primarily for validation of the coarse analysis, but was found to be only available for relatively small areas. Data found included:

a) **Geomorphic elements of the MIA** (Butler *et al.*, 1973) contained soils information under a different classification scheme to the Soil Landforms data.
b) **A review of hydraulic conductivities of soils of the MIA** (Hornbuckle and Christen, 1999) – it was difficult to relate soils attributes to the other coverages.
c) **Soils and land use of the northern Victorian irrigation region** (Skene, 1963; Skene, and Freedman, 1944; Skene and Harford, 1964; Skene and Poutsma, 1962; Butler, 1942; Johnston, 1952), 1:25,000 surveys in Technical Bulletins of the Victorian Dept. Of Agriculture. These surveys, produced between 1942 and 1965, have a local soil name and a rating for irrigated agricultural suitability. These groups only partly correspond to expected hydraulic conductivities because those assessments were based on a range of criteria and not just soil physical properties.
d) **Piezometer databases for the MIA and SIR.** For the MIA, a database for 1996 from the Department of Land and Water Conservation (DLWC). Bores screened below 15m were removed before interpolating a grid. For the SIR, a database was available for 1996 from Sinclair, Knight Merz (SKM) from which bores deeper than 15m could not be removed. A subset of the 1996 MIA piezometer data set from DLWC containing salinity information was also interpolated.
e) **Shallow aquifer potential salinity map of the SIR.** Ife (1987) mapped shoestring aquifers with associated potential salinities at a scale of 1:250,000.
f) **Water use data for the Murray Irrigation Ltd (MIL) area for 1995-6**. This dataset was collected by Murray Irrigation Ltd and contained point information on water use and areas used for rice growing. The presumption is that areas of low rice water use have low conductivity soils.

Test at 250,000 scale

Since there is no appropriate permeability data, landforms from the Bui *et al.* (1998) datasets were linked through a lookup table of McKenzie *et al.* (1999) to derive hydraulic conductivity classes. These classes were distributed on a log scale centred around a median value (see Fig. 1 for details). Classes were then grouped in order to have reasonable map coverage of at least two of the ranges, which led to the inclusion of some extremely high values. These were viewed as representing relative conductivities given the error and reliability estimates associated with the data. Ranges for the BRS (1999) depth to groundwater and salinity data were amended similarly. Overall suitability classes for the combined criteria were defined and are shown in Fig. 1.

Test of finer resolution data

No appropriate permeability data was available at high resolution either. Instead, irrigated agricultural suitability groups (not based on rice) were amalgamated on the basis of clay content as best they could. With no conductivity estimates attached to soil groups the thresholds could only be defined qualitatively. The resolution of the water table and salinity data meant that the original defined ranges of suitable criteria could be used. The thresholds and combinations of criteria are shown for the SIR in Fig. 2.

Validation test

To our knowledge, there is very little data available for testing the analysis. Obviously, a comparison with finer resolution data gives some level of confidence, but hardly constitutes a validation, particularly when one of the datasets (landforms) is being used as a surrogate for permeability. Even matching soil maps at different scales is difficult due to the difference in mapped attributes. Correlating results to existing evaporation basin locations was undesirable because they were not sited using the criteria defined in this paper. Maps of rice area provide some indication (generally being on slowly permeable soils) although rice is grown at sites which may not be suitable for evaporation basins, for example, in close proximity to irrigation channels. Leakage estimates (water use less crop adjusted evapotranspiration) in the Murray Irrigation Ltd area, are being used to provide a validation, although at time of submission, only some preliminary results are available. The use of these relies on wider availability of water use, detailed soil maps, potential rice growing maps or irrigation leakage data, some of which are commercially and privately sensitive – and hence not widely available.

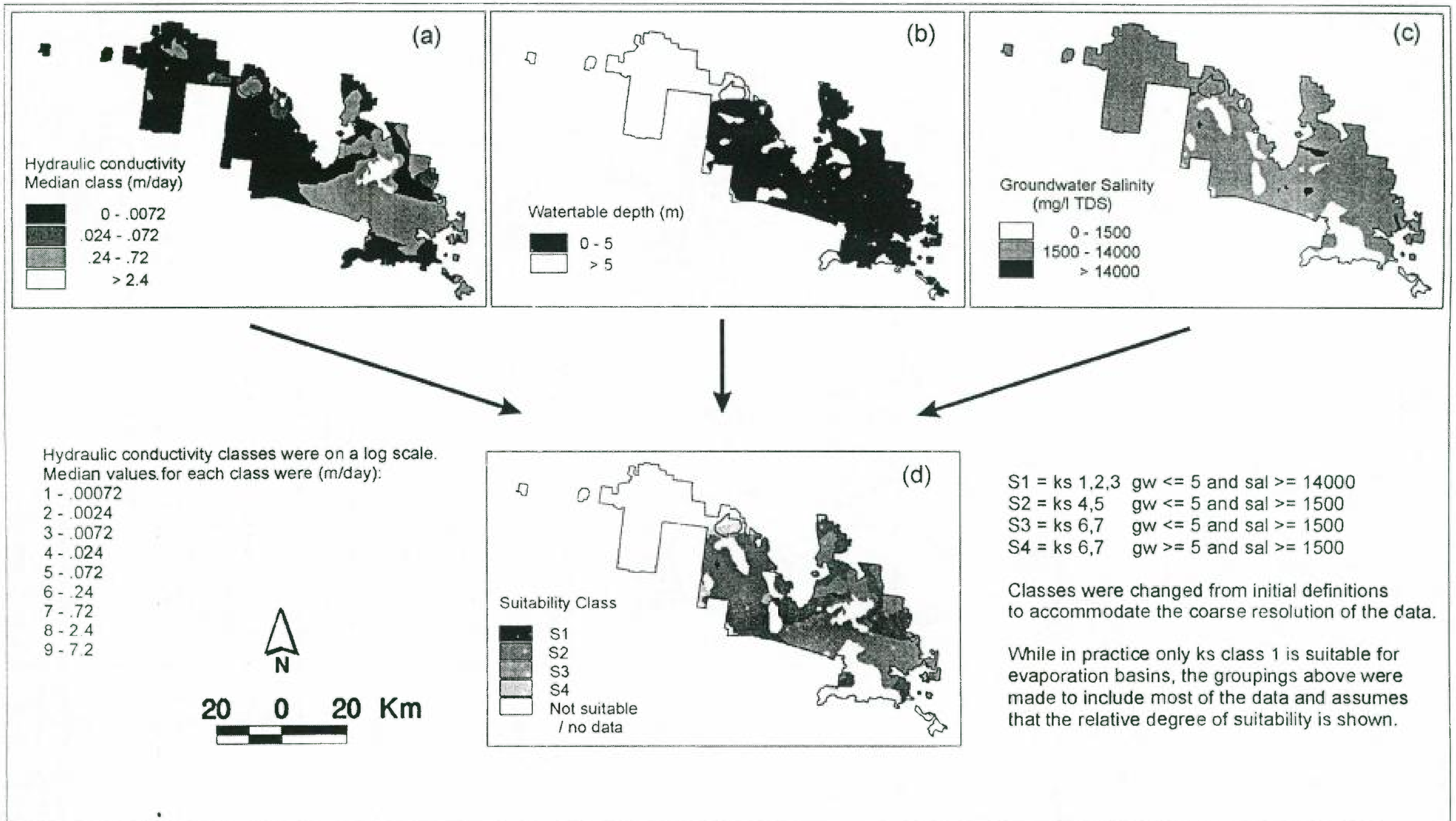

Figure 1. MIA coverages used in the coarse resolution suitability analysis. a) Hydraulic Conductivity, b) Water table depth, c) Salinity and d) Overall suitability. S1-4 refer to suitability based on the rules in the key above, where ks is hydraulic conductivity class, gw is depth to groundwater table and sal is groundwater salinity. Black in each map indicates the best ranges for the theme while lighter shades are decreasingly suitable.

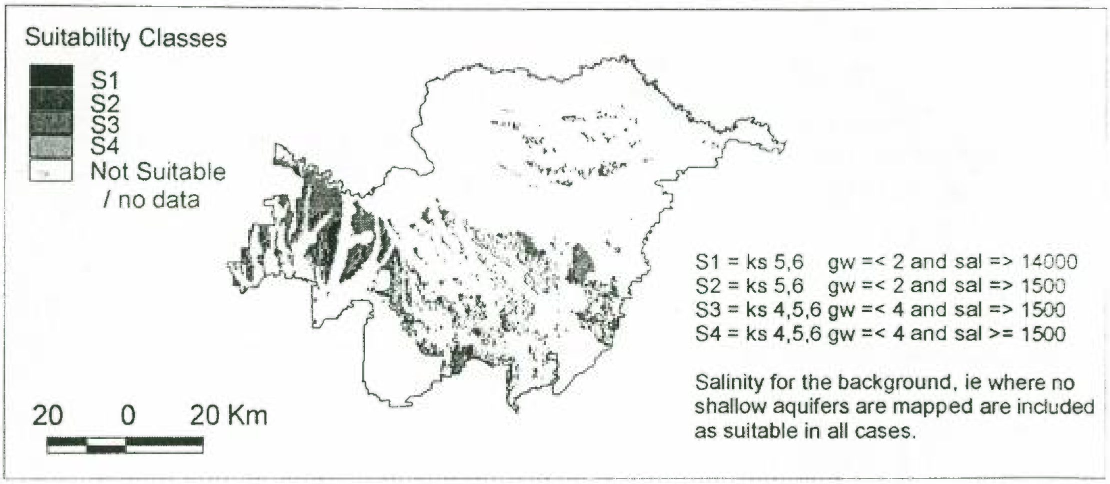

Figure 2. Suitability classes calculated for the SIR using irrigated crop suitability groups (Victorian Department of Agriculture, 1:25000), water table depths from piezometric data (SKM) and saline aquifers (Ife, 1987). Texture of the underlying shoestring aquifers is evident.

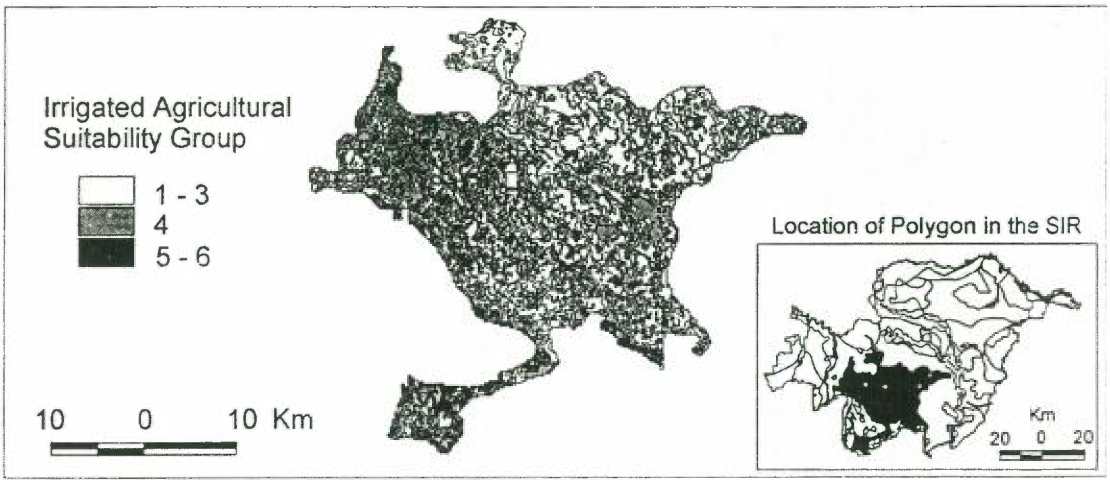

Figure 3. Soil groups from Victorian Department of Agriculture (1:25000) overlayed on a single Soil Landforms class polygon. Groups indicate irrigated agricultural suitability based on 1940-60 technology and management practices. There are no links between attributes in the two data sets.

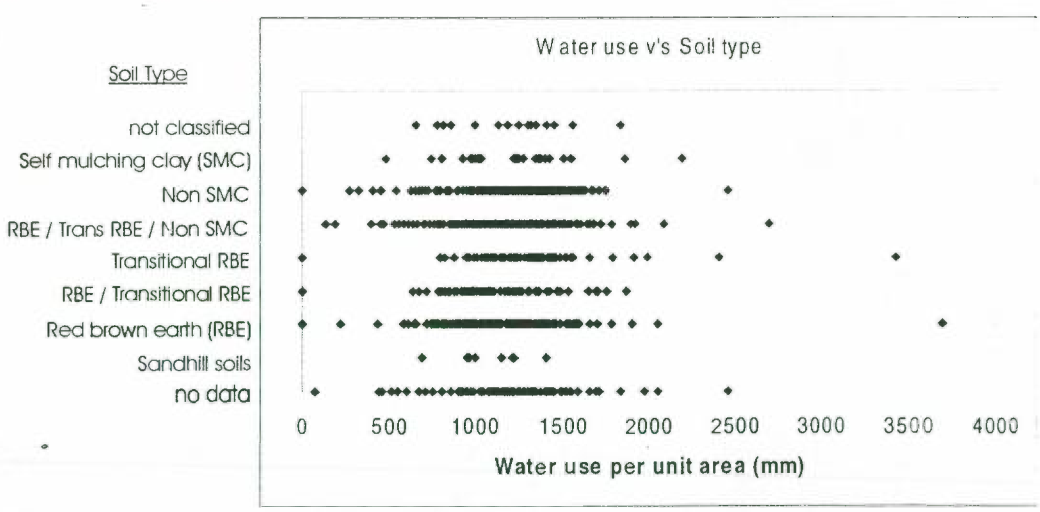

Figure 4. Correlation of high resolution soil types with water use/unit area for the Murray Irrigation Limited (Deniliquin) area. Water use (mm) was calculated by dividing rice water use by area under rice. The graph illustrates the difficulty relating permeabilty to soil types. There are many reasons for variations in water use besides hydraulic conductivity that limit the usefulness of this data.

Results

Overall suitability maps were produced for the MIA and SIR at both resolutions. Only a selection of the results can be addressed here to show; 1) what the method could produce if the appropriate data could be obtained, 2) what differences the finer resolution analysis provides and 3) how much we can relate soils to hydraulic conductivity.

Results for the MIA coarse resolution are shown in Fig. 1, which summarises the process and shows the overall suitability map using the available data and thresholds we used. Anomalies that contradict local knowledge and the finer resolution datasets are evident in all input coverages (Figs. 1a-c). While the overall suitability map (Fig. 1d) shows virtually nothing predicted for the most suitable class (S1) it does imply a ranking of the probability of finding suitable sites within broad areas and those areas most likely to have Not Suitable (NS) areas. The areas are generally large contiguous regions that, if true, indicate a higher likelihood of suitability for on-farm basins.

Figure 2 shows the suitability map derived for the SIR and the greater fragmentation obtained from using the fine resolution data. While it is difficult to assess the accuracy, except by local expert opinion, the map shows areas of relative suitability that could be investigated in more detail. For the MIA 52% of the region was predicted as the same suitability (or NS) class and for the SIR 39%. Correspondence in any class S1-S4 were 62% for the MIA and 12% for the SIR pointing to differences in the underlying high resolution datasets.

Figure 3 highlights some of the problems in validating data sets and therefore the overall suitability maps. There is a huge amount of detail represented in one soil-landform polygon where both datasets have different attributes that are not linked but are both surrogates for conductivity. Proportions of soils are; 66% permeable (groups 1,2,3), 13% least permeable (groups 5,6) and 20% mixed (group 4). Group 4 was included with 5,6 as suitable soils for the overall suitability classes. Figure 4 shows one attempt to bridge this gap between datasets by correlating the soil associations used in the MIL area with rice water use / unit area. There are many reasons for the poor relationship (and no attempt to account for them) including the inadequacies of the soils map, variable depth to water table, variable rainfall and channel leakage (accidental or otherwise).

Discussion

The suitability approach used allows for any number of spatial datasets of any attribute to be incorporated simply by defining acceptable ranges in the data and overall suitability classes in the GIS. If you can believe the input data then the overall suitability maps indicate, in relative terms, the likelihood of finding land suitable for evaporation basins from which to begin more detailed investigation. The coarse resolution analysis is relevant at a very broad level and the fine resolution can be used more locally but in both cases requires careful consideration of the accuracy and relevancy of the inputs for the particular use.

Soil attributes are the only data available to derive adequate surrogates for hydraulic conductivity. Methods to extract more from the soil properties look-up table need to be investigated, although the look-up table was intended only to give a generalised view of soil properties. Better estimates may be obtainable, at considerable cost, by cross-referencing fine resolution data sets to the Soil Land Forms of Bui *et al.* (1998) through one common attribute such as Principle Profile Form. Figure 3 shows the variability of 1:25,000 Victorian soil maps occurring in a single 1:250,000 Soil-landform polygon. Apart from the differences in detail, the irrigated agriculture groups in the 1:25,000 data are ambiguous with respect to hydraulic conductivity, and none of the attributes, including soil classes and groups, have any commonality. Information, including hydraulic conductivity and soil salinity, exists in the associated Technical Bulletins, however extracting that information is a significant task. Given the very high spatial variability of hydraulic conductivity it is questionable whether the effort to make use of this data is warranted.

Validation of water table depth and salinity was easier because there is a substantial coverage of monitored piezometers and bores in both areas. Salinity in both resolutions was reasonably similar for the MIA. Discrepancies in water levels in the SIR may be indicative of the age of the data given the rapid changes in water tables over the last decade. It is not surprising that the BRS (1999) hydrogeological data are inconsistent between regions given the number of authors who collated and interpreted the data and the variability in its quality and resolution. Salinity maps highlighted problems with data from different sources. While comparisons in the MIA using interpolated data showed broad agreement, the SIR was dissimilar between resolution because of the nature of the mapped shallow shoestring aquifers having substantial areas of no data.

Conclusions

Suitable data was found to be the limiting factor in this analysis, not the availability or development of GIS methods. We have attempted to make the best use of the available data, using three different regions (MIA, SIR and MIL) as test areas. Spatial resolution appears to be less a problem than the accuracy of measured attributes and inconsistencies between data sets. Surrogates for essential criteria were necessary in the absence of measured attributes. Effectively there is no choice but to use soil type as a surrogate for leakage though it still needs validation through the limited datasets available. The suitability maps are easy to produce and manipulate as new knowledge and data become available (or possibly scenario testing). Determining how much confidence we can have in the suitability maps at this stage requires local expert opinion.

Suitability maps show broad trends and areas in which to focus more detailed investigations. Methods and interpretations need to consider temporally changing attributes such as rising water tables and groundwater salinities, and differing soil classification schemes. Ultimately it would be useful if resources were put into updating these. Inevitably, detailed site investigations are needed to validate the predictions. With further work, suitability approaches like this could be used to help make planning decisions.

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Factors affecting the financial viability of subsurface drainage with an on-farm disposal basin in the MIA

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1. Abstract

This analysis studies the economic viability of subsurface drainage systems considering the evaporation basin as part of the whole drainage system rather than as a separate entity. The results indicate that in the MIA subsurface drainage with an evaporation basin is an effective strategy in controlling waterlogging and therefore improving crop yields, especially in more waterlogging sensitive crops. The financial performance of a subsurface drainage scheme is more attractive in larger farms because of economies of scale associated with farm size. The use of evaporation basins is better suited to waterlogging sensitive crops with high yield and price. Efficient irrigation management is a key factor affecting the financial attractiveness of a drainage system.

2. Introduction

The use of on-farm evaporation basins is often part of Land and Water Management Plans (L&WMPs) to reduce the salt load leaving irrigation areas. Thus, when installing tile drainage, an evaporation basin would have to be installed to hold drainage water, than discharging into the surface drainage system. This reflects the reality that in many present situations off site disposal is no longer possible. Therefore, with the constraints on salt export from the irrigation areas under the Murray Darling Basin Salinity and Drainage Strategy and the large cost involved in the construction and operation of a subsurface drainage system with evaporation basin, it is important to determine the financial desirability of the drainage system and identify and quantify the impact of key factors for successful use of subsurface drainage basins.

3. Methodology

The yield impacts due to waterlogging for grapes and citrus in the MIA, which have different sensitivities to waterlogging, were analysed using the BASINMAN model (Wu *et al.* 1999). The yield data generated were used in a cash flow budget for financial analysis of the subsurface drainage basin system. The budget was constructed for a 25 year period and total yearly benefits and costs were discounted at a rate of 8%. Details of the data sources, analysis and methods are in Singh and Christen (1999). Three scenarios were evaluated: (1) no subsurface drainage (ND), (2) subsurface drainage without an evaporation basin (DNB) and (3) subsurface drainage with an evaporation basin (DWB). Sensitivity analysis of the subsurface drainage/disposal systems was carried out using @ RISK©, a program that allows variation in variables, in this case farm size, market price and yield to be taken into account.

4. Results and Discussion

4.1 Financial viability of subsurface drainage with an evaporation basin

Comparison of the total costs and net income per year (average) were conducted for DNB compared to ND, DWB compared to ND and DWB compared to DNB, (Table 1).

Table 1. Income and cost comparison for a new vineyard under different drainage /disposal scenarios

Drainage scenarios comparison	Basin area (ha)	Yield (tonnes/ha)	Cost ($/ha)	Net income ($/ha)
ND to DNB	0 (0.0)	+2.01 (15.1)	+445 (10.1)	+962 (19.5)
ND to DWB	-3 (-7.5)	+1.75 (13.1)	+1068 (24.3)	+157 (3.2)
DNB to DWB	-3 (-7.5)	-0.26 (-1.7)	+623 (12.9)	-805 (-13.6)

Figures in parentheses are the percentage change

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Subsurface drainage with no basin (DNB) was most attractive in terms of income. The income and cost differences between No drainage (ND) and subsurface drainage with basin (DWB) indicated that the subsurface drainage with an evaporation basin has about 3 % higher returns than No Drainage. This marginal difference of returns between ND and DWB however, is not an indication of the long term financial attractiveness of a subsurface drainage scheme with basin (DWB) over the ND situation which may vary depending upon the cost of installing subsurface drainage and siting, design and construction of evaporation basin and the land value associated with the basin. The long term financial viability of any system is expressed by the performance of financial indicators as shown in Table 2.

Table 2. Financial performance of drainage /disposal schemes for a new vineyard

Evaluation criteria	**Subsurface drainage/ disposal schemes**		
	ND	**DNB**	**DWB**
Net Present Value ($ 000)	1039	1338	899
Benefit Cost Ratio (%)	1.47	1.54	1.35
Break Even Time (years)	8	8	8
Net Cash Flow ($/ha)	4946	5908	5103

The financial analysis of DWB for a vineyard indicated that in the long term ND is more financially attractive than DWB due to the large cost differentials between the two schemes and that vines are relatively less sensitive to waterlogging. The analysis, however, is purely based on waterlogging losses. In the long term salinity effects may be greater and hence make DWB more attractive than ND. Another factor to be considered for long term financial viability is the stability of annual returns in terms of Net Cash Flow (NCF) Figure 1. The fluctuations in NCF resulting from fluctuations in water table due to variation in rainfall were highest with ND, more stable in DWB and were most stable with DNB. In these terms DWB is desirable to give a better/ more stable cash flow.

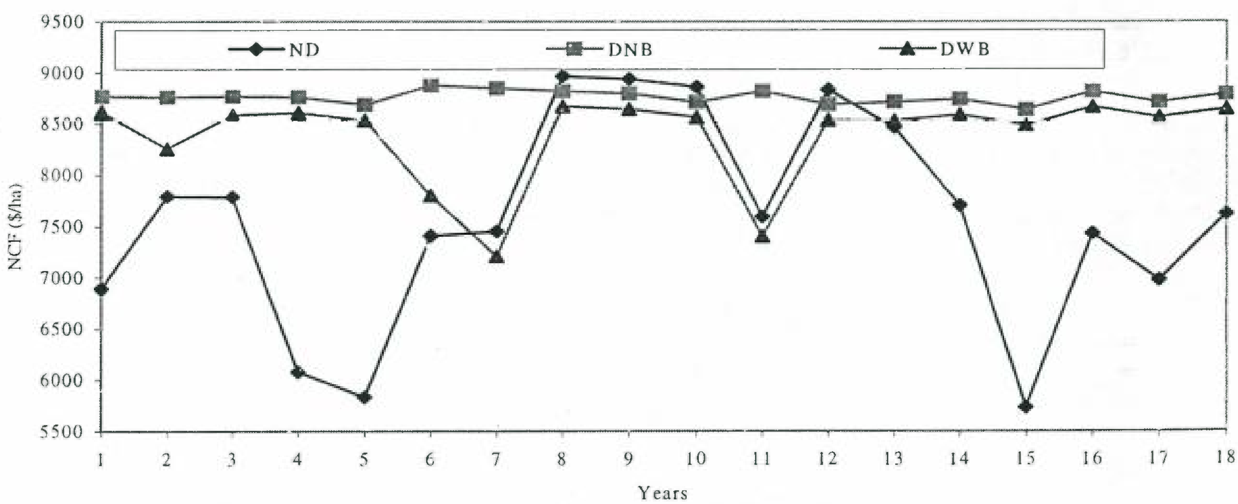

Figure 1. Annual Net cash Flow for new vines with different drainage/ disposal schemes

4.2 Factors affecting the financial viability of subsurface drainage basin schemes

The financial performance of subsurface drainage schemes with a basin (DWB) under various physical and economic factors were examined to identify important factors affecting financial desirability.

4.2.1 Farm size

Farm size has a considerable impact on the financial attractiveness of a drainage scheme for a new development Figure 2. The analysis showed that financial attractiveness of all the subsurface drainage schemes improved considerably with increasing farm size, however the rate of increase declines as farm size increased. The main reason behind such a trend is that with increasing farm size, all the capital assets are fully utilised due to greater economies of scale. It can be seen that drainage schemes with or without a basin became financially unattractive at farm sizes below 20 ha. Every unit increase in farm size between 20 and 40 ha increased profitability of different subsurface drainage schemes.

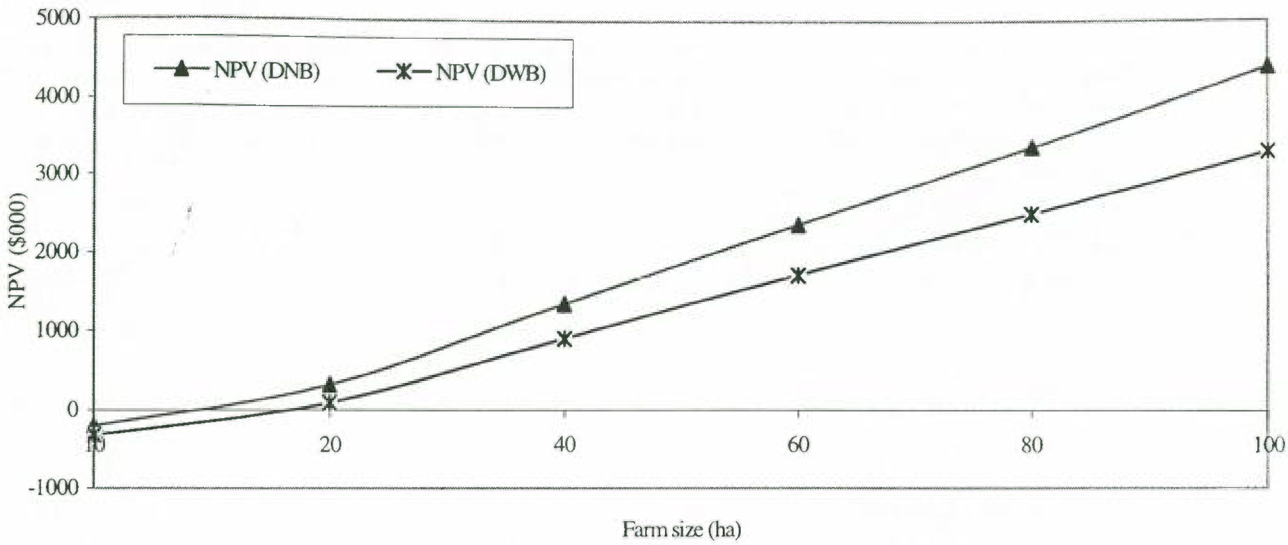

Figure 6. Effect of farm size on the financial attractiveness of drainage /disposal schemes in new vines

4.2.2 Crop price

Table 3 shows that crop price has a considerable impact on the financial attractiveness of a subsurface drainage system. The profitability of DNB changes at a rate of 1.6% per unit change in price, this rate of change for DWB was about 1.4 %. The DNB scenario becomes unattractive when about 40% decline in crop price occurs, whereas the DNB scenario becomes marginal with only a 20% decrease in crop price. However, the use of a basin makes the enterprise more sensitive to price fluctuations

Table 3. Effect of crop price on drainage schemes' financial attractiveness (farm size and yield constant)

Financial criteria	Drainage scheme	% change in price (base price $700/tonne)				
		-40	-20	0	+20	+40
BC ratio	DNB	0.92	1.23	1.54	1.86	2.17
	DWB	0.81	1.08	1.35	1.62	1.90
NPV ($000)	DNB	-1931	574	1338	2103	2866
	DWB	-500	202	899	1595	2291

4.2.3 Crop yield

The subsurface drainage schemes were less sensitive to change in yield than change in price, though a unit change in yield has more impact on the profitability of a subsurface drainage scheme than a unit increase in price Table 4. This is due to the fact that price change has an impact on gross income only whereas a yield change affects both income and costs. These results show that DNB can sustain a yield loss of up to 40% whereas DWB becomes financially unattractive when the yield declines by about 20 %. Lower yield also results in longer Break Even Times.

Table 4. Financial attractiveness of drainage / disposal scheme under changed crop yield of new vines

Financial criteria	Drainage system	% change in average yield				
		-40	-20	0	+20	+40
BC ratio	DNB	1.00	1.28	1.54	1.79	2.03
	DWB	0.86	1.12	1.35	1.58	1.79
NPV ($000)	DNB	-8	673	1338	2013	2687
	DWB	-331	293	899	1513	2127

4.2.4 Irrigation efficiency

The viability of a subsurface drainage/ disposal scheme is largely determined by the amount of water a system handles. Under inefficient irrigation management (applying more water than needed, here assumed 120% of evapotranspiration) there are several consequences; increased cost of water, more recharge to ground water, crop yield decline and larger basin area to handle the increased drainage. In this analysis the effect of inefficient irrigation is considered DNB and DWB with a 7.5% basin area, Table 5. Poor irrigation and hence greater

Table 5. Effect of irrigation efficiency on the financial attractiveness of drainage/ disposal Scheme in new vines

Criteria	Subsurface drainage /disposal scheme			
	DNB		DWB	
	Efficient (110% of ET)	*Inefficient (120% of ET)*	*Efficient (110% of ET)*	*Inefficient (120% of ET)*
Average yield (t/ha)	15.35	15.35	15.09	14.70
BC Ratio	1.54	1.54	1.35	1.32
NPV ($000)	1338	1333	899	820
BET (years)	8	8	8	9
Annual NCF ($/ha)	5908	5899	5103	4853

Waterlogging had no effect in DNB as the drainage system removed excess water. However, for DWB this resulted in about a 3 % decline in average yield and about 5% decline in the net cash flow, the BCR value declined by about 2%, and the Net Present Value by about 9%. Therefore, efficient irrigation management is a key factor in deciding the financial attractiveness of a subsurface drainage/disposal system.

4.2.5 Crop development - new or existing vineyards

The results in Table 6 show that the financial attractiveness of DWB is greater for existing vineyards. The income for existing vineyards increased by more than 20% whereas total cost was about 13% less than for a new vineyard development. The profitability of drainage disposal was more than doubled for existing vineyards, which strengthens the scope for adopting subsurface drainage with a basin in existing vines, assuming these do not have a drainage scheme already.

Table 6 Financial attractiveness of drainage/ disposal scheme for an existing vineyard

Criteria	Vineyard development	
	New	Existing
Average yield (tonnes/ha)	15.09	18.13
BC Ratio	1.35	2.54
NPV ($000)	899	3038
BET (years)	8	0
Annual NCF ($/ha)	5103	7888

4.2.6 Crop choice

Analysis of ND, DNB and DWB was undertaken for citrus, which is more sensitive to waterlogging than grapevines (Table 7). These results show that subsurface drainage is essential for a waterlogging sensitive crop such as citrus. Unlike grapevines the DWB outperforms ND for citrus.

Table 7. Income and cost comparison for a new citrus orchard under different drainage /disposal scenarios

Drainage scenarios comparison	Basin area (ha)	Yield (t/ha)	Cost ($/ha)	Net income ($/ha)
ND to DNB	0 (0.0)	+33.37 (1269.0)	+1376 (26.9)	+8335 (202.5)
ND to DWB	-3 (-7.5%)	+29.37 (1117.0)	+1628 (31.8)	+7083 (167.7)
DNB to DWB	-3 (-7.5%)	-4.0 (11.1)	+252 (3.9)	-1452 (33.68)

Figures in parentheses are the percentage change.

The income and cost difference between ND and DWB indicated that subsurface drainage with an evaporation basin has about 168% higher returns than no drainage situation, which are much higher than for vines. These results indicate that DWB has greater potential for citrus that are highly sensitive to waterlogging.

5. Conclusions

1. Subsurface drainage with an evaporation basin is an effective strategy in controlling waterlogging and therefore improving crop yields, especially in crops that are more sensitive to waterlogging.
2. The financial performance of a subsurface drainage scheme is better with larger farms because of economies of scale.
3. Subsurface drainage systems with an evaporation basin are financially more attractive for crops with high yields and prices that are more sensitive to waterlogging.
4. Irrigation management is a key factor in deciding the financial attractiveness of a drainage system.
5. Subsurface drainage with an evaporation basin has greater economic viability for existing planting than for new developments.

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SOIL HYDRAULIC PROPERTIES AND POLLUTANT REMOVAL IN A PILOT 'FILTER' SYSTEM USED FOR TREATING GRIFFITH SEWAGE EFFLUENT

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Abstract

Land treatment of wastewater using cropping and forestry are often considered less economical than engineering solutions due to cost of winter and wet weather storage and adverse impacts on reuse site. In order to overcome problems associated with traditional wastewater disposal schemes, the FILTER (Filtration and Irrigated Cropping for Land Treatment and Effluent Reuse) technique was developed at CSIRO, Griffith. FILTER combines the use of nutrient-rich wastewater for intensive cropping with filtration through the soil to a subsurface drainage system. This technique thus has the capacity to handle high volumes of wastewater during the periods of low cropping activity or periods of high rainfall. In order to produce minimum-pollutant drainage water which meets general environmental criteria for discharge to surface water bodies, the wastewater application and subsurface drainage in the FILTER system needs to be managed to ensure adequate removal of pollutants, while maintaining required drainage flow rates.

In this paper we describe the field evaluation of a pilot FILTER trial at the Griffith City Council sewage works site. The trial was designed and operated as one of eight irrigation blocks of a proposed 120 ha commercial FILTER system, required for around-the-year treatment of all Griffith's sewage wastewater. The field data from the winter cropping season of 1998 including hydraulic flows and removal of pollutants such as nitrogen, phosphorus, BOD, suspended solids, oil and grease, chlorophyll-a, and *E.coli* are discussed. Results indicate that a well managed FILTER technique can reduce pollutant levels in drainage waters below NSW EPA limits, while maintaining adequate hydraulic flow, crop yields and nutrient removal to potentially make it a sustainable system.

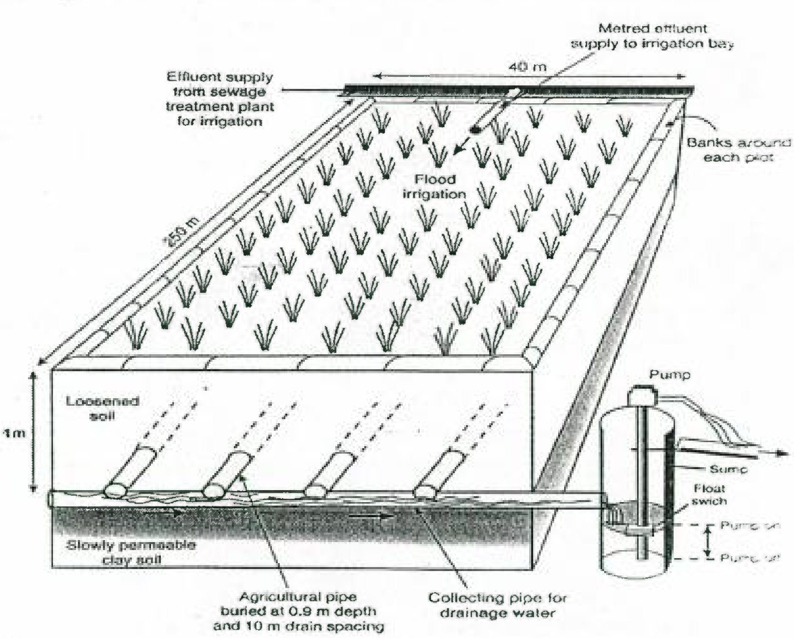
Keywords

wastewater treatment, pollutant removal, hydraulic properties, controlled subsurface drainage.

Introduction

The FILTER (Filtration and Irrigated cropping for Land Treatment and Effluent Reuse) technique was developed as a new approach for sustainable and around the year effluent treatment from Griffith Sewage Works (Jayawardane *et al.*, 1997a). The main objective being to reduce phosphorus and nitrogen content in the discharge water below Environmental Protection Authority (EPA) limits. In this technique, the use of the nutrient-rich effluent to grow crops is combined with filtration through the soil to an intensive subsurface drainage system as shown in Fig. 1. The FILTER system is thus potentially capable of handling high volumes of wastewater during periods of low cropping

Fig 1. Schematic diagram of FILTER plots



activity and/or high rainfall. In this system wastewater application and subsurface drainage is controlled in order to achieve adequate nutrient removal by the soil bio-chemical processes and crops, thereby producing low-nutrient drainage. This drainage water needs to meet EPA criteria for discharge to surface water bodies or EPA criteria for non-potable reuse. This system involves manipulation of the watertable depth above the subsurface drains by controlled pumping from the drainage system. The drains are located at about 1.2 m deep at a spacing of 8-10 m. The system needs to be managed to provide optimum conditions for crop growth and pollutant removal, while maintaining the required drainage flow rates.

Each fortnightly effluent application cycle or filter event consists of four consecutive stages. These four stages are effluent application (irrigation), followed by a post-irrigation equilibration period and by a pumping period (until drainage outflow approximately matches the net inflow) and finally a no-pumping equilibration period (leading to flattening of the watertable). The manipulation of these four-stage effluent application and drainage operations could be used to maximise the removal of nutrients, and increase the uniformity in nutrient distribution and retention in the soil across the FILTER plots.

Preliminary testing of the FILTER technique on one-hectare plots showed that the FILTER system met its objectives of reducing nutrient in drainage waters below EPA limits, while maintaining adequate drainage rates (Jayawardane *et al.,* 1997a, b). In addition, significant crop yields were obtained, which could be used to offset costs in a commercial system. The other beneficial effects were reduced suspended solids, increased N:P ratio and the potential to use the technique to ameliorate saline soils as well as handling saline effluent. Initially, the concentrations and loads of salt were increased in the drainage waters compared to incoming effluent load and this was mainly due to leaching of accumulated salts which had built up in the soil profile through previous effluent application without subsurface drainage. However, after a certain period an equilibrium will be reached and there will be no additional leaching of salt from the soil profile. Salt in the drainage water will then be due to the incoming effluent.

After the successful and encouraging results of the preliminary FILTER trials, a commercial FILTER system was planned on Griffith City Council's land which is located close to its sewage works site. The commercial FILTER system is planned to be easily operated and managed by council staff. This system will consist of eight irrigation blocks of which six will be located on a 100 ha allotment at the experimental site. Each irrigation block is approximately 15 ha. It is proposed that each of the eight irrigation blocks will be irrigated with sewage effluent for two days on a sixteen-day rotation.. The irrigation rotation period will be shortened if one or more blocks are out of rotation for agronomic management practices, such as planting or harvesting. This will allow for continuous land treatment of the Griffith's sewage effluent throughout the year. If the pilot trial indicates that a larger land area is required to ensure sustainability, additional irrigation blocks could be added and the rotation system adjusted accordingly.

Presently we are researching the performance of one (15 ha) of the proposed 8 irrigation blocks, when it is managed according to the plans for running a commercial FILTER system at the site. This paper presents the field results on the hydraulic flow, pollutant removal and crop growth measurements during the first winter cropping season.

Materials and Methods

To establish the pilot FILTER trial, the site was laser levelled to provide an irrigation slope of 1:4000. Four irrigation bays (430 m long by 82, 80, 86 and 102 m wide) with 0.4 m banks were constructed to provide good control of irrigation. A subsurface drainage system was installed within the pilot trial area, which was connected through the collector drains and the main drain to the main sump, fitted with an electric pump and flow meter. The subsurface drains were spaced 8 m apart at a depth of 1.2 m. The irrigation channels and associated structures for controlling and monitoring irrigation were installed. A dethridge wheel, MACE flow and current meters were used to measure irrigation and drainage volume.

A 200 m long ag-pipe was installed lengthwise (in the south-north direction) in the bottom half of each of the bays. at 1.3 m depth. These ag-pipes were located approximately at the mid-width of the plots, and placed exactly at the mid-distance between the two previously installed central ag-pipes of the sub-surface drainage system. All these ag-pipes were connected to a vertical PVC pipe, located just outside the bottom end bank of each bay. The watertable height inside this vertical PVC pipe indicates the watertable height in each of the four bays.

In autumn 1998, two of the bays were sown with Coolibah Oats (90kg/ha) and 150kg of diammonium phosphate (18:20) fertiliser was drilled in with the seed. The other two bays were planted with ryegrass pasture mix; 17 kg multimix, 6 kg demeter fescue, 8 kg Victorian rye and 5 kg guard rye per ha. Eight irrigation/FILTER events. of 2 weeks duration each, were carried out during that winter cropping season. Each FILTER event consisted of four stages, namely a two-day effluent application period, a one-day post-irrigation equilibrium period, a 8-10 day pumping period and finally a no-pumping period to allow a flattening of the water table.

Measurement of drainage rates and soil hydraulic properties

An important factor in optimising the design and operation of the FILTER system is the maintenance of adequate flow rates to the drains during the pump down period. The drainage flow rate depends on the hydraulic conductivity at different soil depths, depth to impermeable soil layers, depth of the drains and the spacing between drains. Hence the creation of soil macroporosity during FILTER installation to increase the soil water storage and soil hydraulic conductivity, and the maintenance of the macropores during subsequent FILTER operations are important considerations.

Accurate measurements of the soil hydraulic characteristics that influence flow rates to the sub-surface drains are necessary to optimise the drain spacing and depths for a given site. These measurements need to be representative of the whole FILTER plots. A method to evaluate such hydraulic characteristics of the entire field plots, which uses the drainage flow rates and watertable depths measured in the field is described by Youngs (1985). For subsurface drains running normally without back pressure Youngs (1985) found the Hooghoudt's equation could be simplified to:

$$\frac{H_m}{D} = \left(\frac{q_t}{K}\right)^{\frac{1}{a}} \qquad (1)$$

where q_t is the drainage flow rate (m s^{-1}); H_m is the water table height (m) at the midway point between the drains; K is the saturated hydraulic conductivity of the soil (m s^{-1}); and 2D is the spacing (m) between drains and a is a coefficient given by:

$$a = 2\left(\frac{d}{D}\right)^{\frac{d}{D}} \qquad (2)$$

where d (m) is the depth below the drain to the impermeable layer and $0 \leq d/D \leq 0.35$. For the situation where $d/D \rightarrow \infty$, the value of a was found by Youngs (1985) to be approximately equal to 1.36. Where $d/D = 0$, the value of a is equal to 2.0.

Rearranging (2) and substituting the value of the watertable height at a given time (H_t) for H_m, we obtain

$$q_t = K\left[\frac{H_t}{D}\right]^a \qquad (3)$$

$$\log q_t = a \log H_t + [\log K - a \log D] \qquad (4)$$

From Eq. (4) it is seen that q_t equals K, when H_t equals D. Thus in field experimental test plots in which the maximum measured $H_t << D$, the estimated value of K is highly dependent on the accuracy of the measurement of slope a, which is used in extrapolating the data from the measured H_t range to $H_t = D$.

Measurement of pollutant removal

Continuous irrigation and drainage water samples were collected using a GAMET auto sampler and a sample bleeding tube arrangement, respectively. Samples were stored at 4°C before analysis for pH, electrical conductivity (EC), biochemical oxygen demand (BOD_5), total suspended solids (TSS), ammonium, oxides of nitrogen (NO_x), total kjeldahl nitrogen (TKN), total phosphorus (TP), chlorophyll$_a$, total faecal coliforms, and oil and grease. Analyses were carried out following APHA, AWWA and WEF methods at the Analytical Services Laboratory of NSW Department of Public Works and Services, Lidcombe, Sydney 2141.

The soil profile was sampled up to a depth of 2 m and the core divided into intervals of 20 cm. The samples were analysed following the methods of Rayment and Higginson (1992) for 2M KCl extractable ammonium and NO_x (method 7C2), Colwell extractable P (method 9B2), 1:5 soil:water pH and EC (methods 4A1and 3A1) to assess nutrients and salt in the profile, at the beginning and end of the cropping season. The pasture and oats were cut for hay in November 1998, dry matter yields were recorded. The crop was analysed for total N using Etheridge *et al.*

(1998) method whereas total P and micronutrients were determined using Zarcinas *et al.* (1987) method in order to estimate nutrient removal.

Results and Discussion

Drainage rates and soil hydraulic properties

The relationship between drainage flow rate (q_t) and the height of the watertable measured at the mid-point between sub-surface drains (H_t or H_m) in the pilot FILTER plots during the pumping period of filter events 3 to 8 is given in Fig. 2. Analysis of this data using equation 4, yields values of K and *a* of 7.26×10^{-7} m/s and 1.21, respectively.

The value of *a* indicates that substantial flows occur below the depth of the drains to the sub-surface drains. The q_t on H_t relationship for different filter events appears to overlap and does not show any trends during successive filter events. This indicates that the soil hydraulic properties have not changed substantially during the winter cropping season.

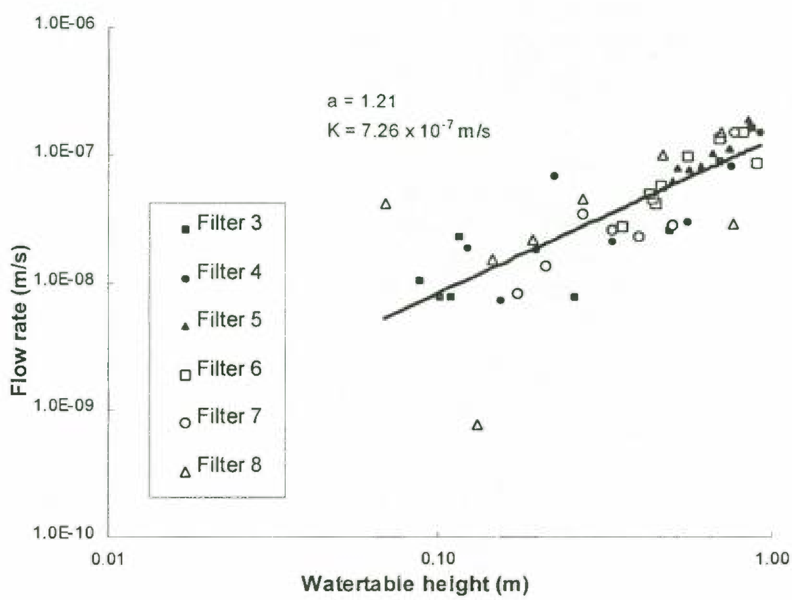


Figure 2. Relationship between flow rate and watertable height in the pilot FILTER plots

A comparison of the predicted relationship between q_t and H_t worked out from previous FILTER trials, with those observed for the pilot plots is given in Fig 3. The predicted line is based on the whole plot soil hydraulic properties measured on the preliminary one hectare plots and adjusted for the change in drain spacing of the present pilot trial in equation 1. The predicted line shows a good fit with the measured data indicating similar soil hydraulic properties, irrespective of the size of the FILTER bays and the drainage spacing.

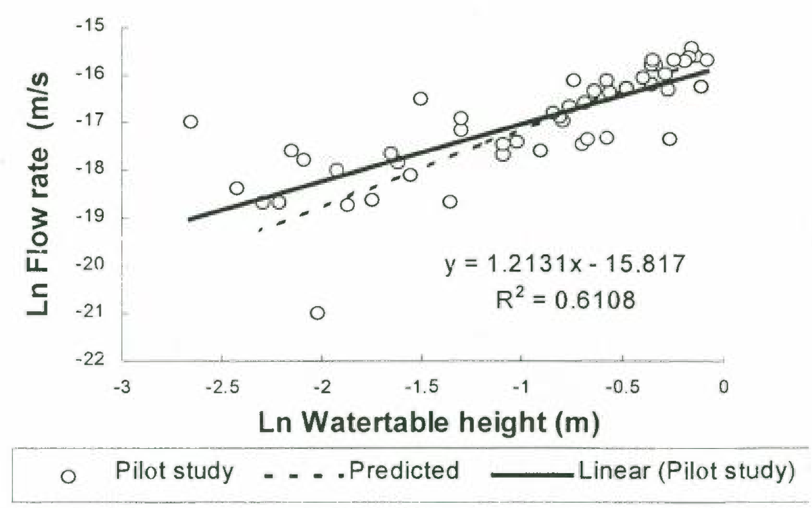


Figure 3. Comparison of measured relationship between flow rate and watertable heights in the pilot FILTER trial with predicted values based on the preliminary experiments

Pollutant removal

In the pilot FILTER experiment, the total-phosphorus (TP) concentrations were reduced from a mean value of 6.1 mg/L in the applied effluent to a mean value of 0.39 mg/L in drainage waters, which was far below the EPA limit of 1 mg/L as shown in Table 1. In contrast, total nitrogen (TN) concentrations were initially high due to leaching of pre-FILTER soil accumulated nitrogen, which then fell below 10 mg/L after FILTER event 4. However, the overall TN concentration in drainage water throughout the season was 15 mg/L and matched the EPA limit for freshwater discharge. Moreover, during the following summer cropping season, the drainage water always had TN below 10 mg/L. With respect to BOD_5, total suspended solids, chlorophyll$_a$, oil and grease, and *E.coli* levels in drainage water, these were all well below the EPA limits.

The incoming and outgoing load of pollutants at the FILTER site as well as the pollution load reductions for TP, TN, BOD_5, $chlorophyll_a$, *E.coli* and oil and grease are given in Table 1. During the period of eight effluent irrigation and filtration cycles, pollutant load reductions for TN, TP, and BOD were 57, 96 and 95% respectively. In addition to its nutrient removal capacity, FILTER completely removed all of incoming effluent's pathogen (*E.coli*), $chlorophyll_a$, oil and grease loads.

Table 1. Pollutant concentration and pollutant load reduction during winter Filtration 1998

Pollutant from Griffith sewage effluent	Pollutant concentration (mg/L)*		Pollutant loads (kg/ha)		%Load removal
	Incoming effluent	FILTER drainage	Effluent	Drainage	
Total phosphorus	6.1	0.4	46.7	1.7	96
Total nitrogen	19.0	15.0	131.0	56.0	58
Organic nitrogen	6.3	1.5	46.3	6.0	87
Ammonium-N	12.5	0.2	84.0	6.1	99
Nitrate-N	0.4	13.3	1.7	49.0	increase
BOD_5	10.3	0.6	81.5	3.9	97
$Chlorophyll_a$	0.1	0	1.3	0	100
Oil and grease	1.8	0	15.9	0	100
Total suspended solids	70.8	16.9	573.0	88.8	85
E.coli (CFU/100 mL)	9	0	-	-	100**

**E.coli* is expressed as colony forming unit (CFU) per 100 mL **Based on *E.coli* number in effluent and drainage.

Salt concentrations and loads in the drainage water were higher than in the effluent largely due to leaching of the pre-FILTER soil profile salt. As a result, the salinity of the soil layers above the drains in the pilot trial were reduced as shown in Fig. 4. This gives the FILTER system an advantage over traditional land applications, in that the FILTER system can not only operate without building up salt in the soil profile, but can also ameliorate previously salinised sites.

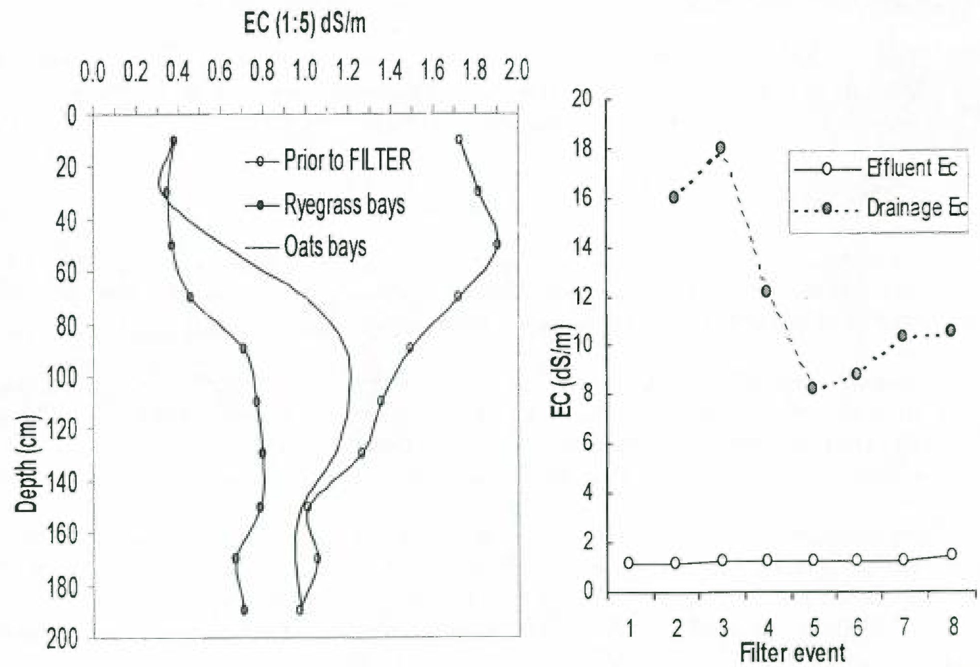


Figure 4. Effluent and drainage water salinity and salt in FILTER soil profile during winter 1998

Crop growth and nutrient removal by crops

Good dry matter yields of oats (8.9 t/ha) and pasture (14.3 t/ha) were obtained with substantial removal of N, P, K, Ca, Mg, Na, boron and heavy metals including copper, manganese, and zinc (Table 2). During the same timeframe net addition (effluent minus drainage) of N and P was 76 and 45 kg/ha respectively. This explains the measured depletion of the soil nitrogen storage and an increase in the soil phosphorus storage, during the winter cropping season. This data needs to be combined with nutrient addition and removal during summer 1998 cropping season in season. This data needs to be combined with nutrient addition and removal during summer 1998 cropping season in order to calculate the land area required to maintain annual nutrient balance, and to develop short-term and long-term options for nutrient management.

Table 2. Removal of nutrient and heavy metals by crops in pilot FILTER trial, and addition of N and P in the effluent

Crops	DM yield (t/ha)	Nutrient and heavy metal removal (kg/ha)									
		N	P	K	Ca	Mg	Na	Cu	Mn	Zn	B
Pasture	14.3	182	21.4	142	19.8	12.3	47.8	0.03	0.56	0.21	0.08
Oat	8.9	76	19.9	131	12.1	8.82	69.2	0.03	0.63	0.34	0.08
Supplied by effluent		76	45								

However, these results emphasise not only the use of nutrients from wastewater for cropping, but also the economic benefits through crop production. Further, through the combination of FILTER design and cropping management it will be possible to avoid build up of nutrients, salts and heavy metals at the reuse site.

Conclusion

The results from the pilot FILTER trial during the winter 1998 cropping season showed that a well managed system can maintain adequate hydraulic flow rates to the subsurface drains and reduce pollutant levels in drainage water below NSW EPA limits. The FILTER provides economic benefit when crops are grown commercially. Through the combination of an appropriate FILTER design and irrigation/drainage/cropping management, it should be possible to develop a sustainable system, which avoids build up of nutrient, heavy metals and excessive salt at the reuse site.

Acknowledgements

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SALINITY MANAGEMENT

MINING GROUNDWATER FOR SUSTAINABILITY IN THE MALLEE REGION

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GLENNIS McKEE, Murrayville Groundwater Taskforce

Introduction

The Mallee Region of South Australia and Victoria is a traditional wheat and sheep grazing area which has been undergoing serious rural decline (declining population, loss of services etc). The sole water supply for farms, towns, and irrigation, is the extensive Murray Group limestone aquifer which lies about 50 m below the surface. The increasing development of irrigation (mainly for potatoes using centre-pivots) has injected over $10 million into the region annually, but has raised two important issues – the sustainability of the development of the resource, and local drawdowns around areas of concentrated pumping.

Sustainability

In recent years, there has been a growing awareness of the imperative to achieve ecologically sustainable development of our resources. With this awareness has come a widespread recognition of the need to consider the long term effects of groundwater development on surface water availability, on natural ecosystems, and on the availability of water resources for future generations.

Most estimates of sustainable yield in Australian groundwater management rely on an extraction volume based on the average annual recharge rate. It is now widely recognised that this concept of safe yield is no longer valid for the purposes of sustainable management of groundwater systems over the long term, particularly in the context of ecologically sustainable development (Evans et al, 1998).

Sophocleous (1997) previously observed that under natural conditions, recharge is balanced in the long term by discharge to springs or streams. Consequently, if pumping subsequently increases to equal recharge, the discharge will decrease and probably cause adverse environmental impacts.

Evans et al proposed that sustainable yield in relation to Australian aquifers be defined as **the level of extraction that should not be exceeded in order to protect the higher value uses associated with the aquifer over a specified planning timeframe.** These may be agriculture, ecosystems, infrastructure, industry or other activities which are to some extent dependent on groundwater, and which the community reasonably expects will be maintained or developed for a defined period. The task of determining and ranking the value of potential uses/demands for any aquifer is likely to be a subjective process that will require a combination of community input and expert opinion. Achieving sustainable management will take considerable time, money, effort and ingenuity.

The Mallee Situation

Circumstances in the Mallee are unique. In the main aquifer developed for irrigation,town water supplies and stock and domestic purposes, the Murray Group Limestone, groundwater moves slowly from the margins of the Murray Basin in southwest Victoria toward the River Murray where it discharges (Fig.1). Current recharge rates in the semi-arid climate are low, less than 1 mm/yr. In fact, the large areas of good quality groundwater (Fig. 2) were probably recharged about 20 000 years ago during wetter regimes (Leaney and Herczeg, 1999) in areas of deep sand, such as the Sunset and Big Deserts. The effects of clearing have not yet reached the deep watertable, but when they do, not only will water levels rise, but salt previously stored in the unsaturated zone will be flushed into the aquifer and potentially raise its salinity by up to 3000 mg/L where it is unconfined in the west of the region.

The main attribute of the Murray Group Limestone aquifer in this region is its thickness, which averages over 100 m. with a maximum of about 140 m on the SA/Vic border to the north of Pinnaroo (Fig. 3). The volume in storage is estimated at 100 million megalitres, with a rate of groundwater movement of only 0.5 m/yr. Fully penetrating bores (about 180 m deep) yield up to 60 L/sec from open hole completions over the bottom 100 m.

There are no ecosystems dependent on groundwater, and it could be argued that it would be beneficial if saline groundwater discharges to the River Murray were to be reduced in the very long term by extractions in the Mallee. Also, if the current rate of rural decline were to continue, there would be virtually no future generations left in the Mallee to benefit from the use of the groundwater resource in years to come.

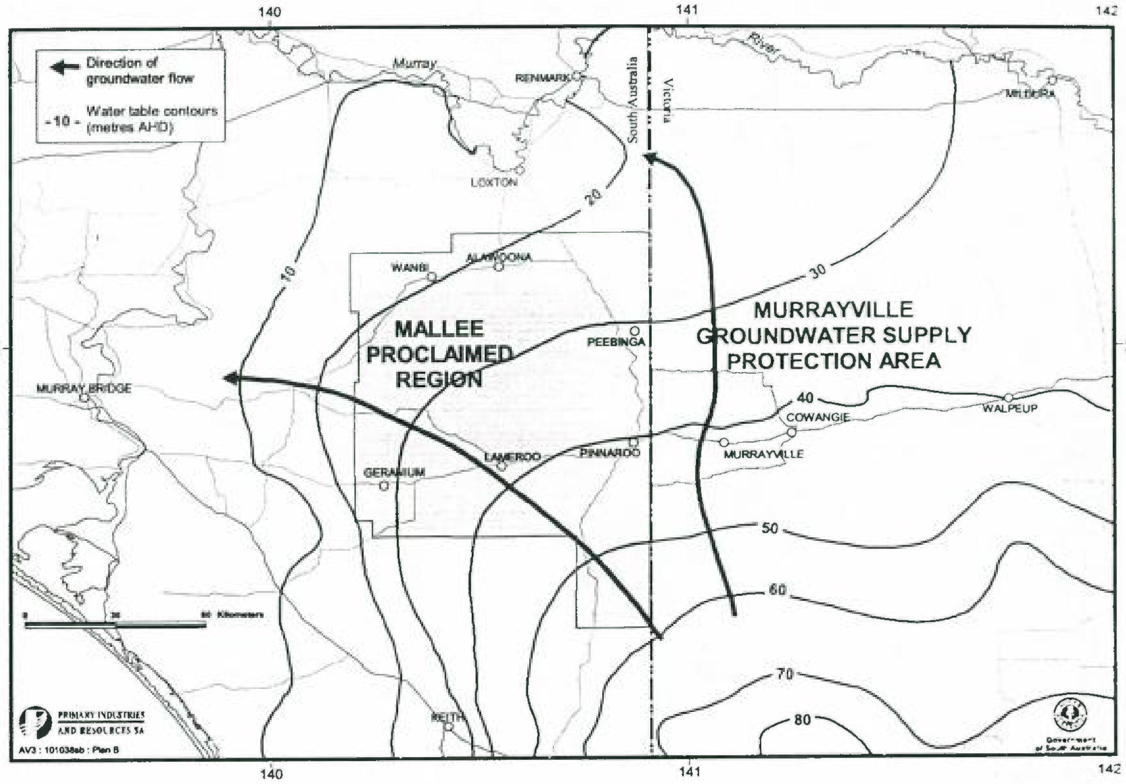

Figure 1 Potentiometric surface contours for the Murray Group Limestone aquifer

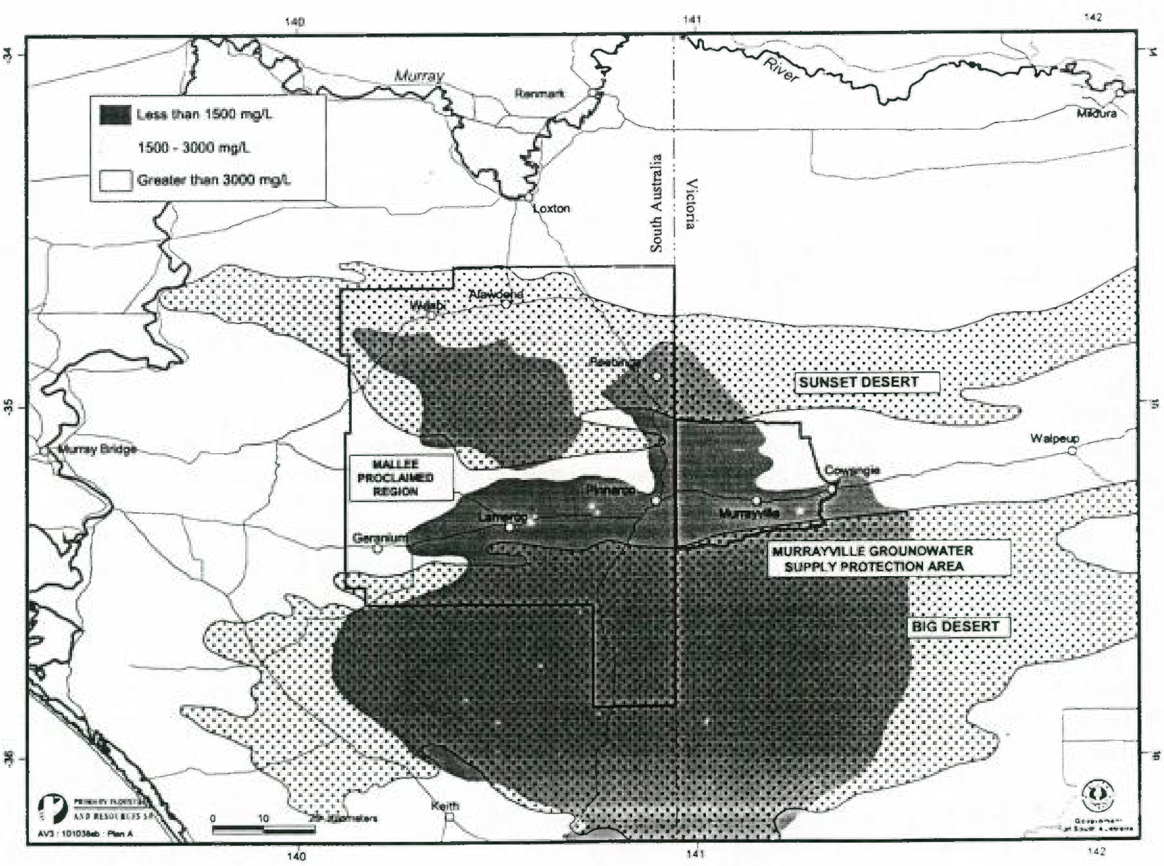

Figure 2 Salinity zones for the Murray Group Limestone aquifer

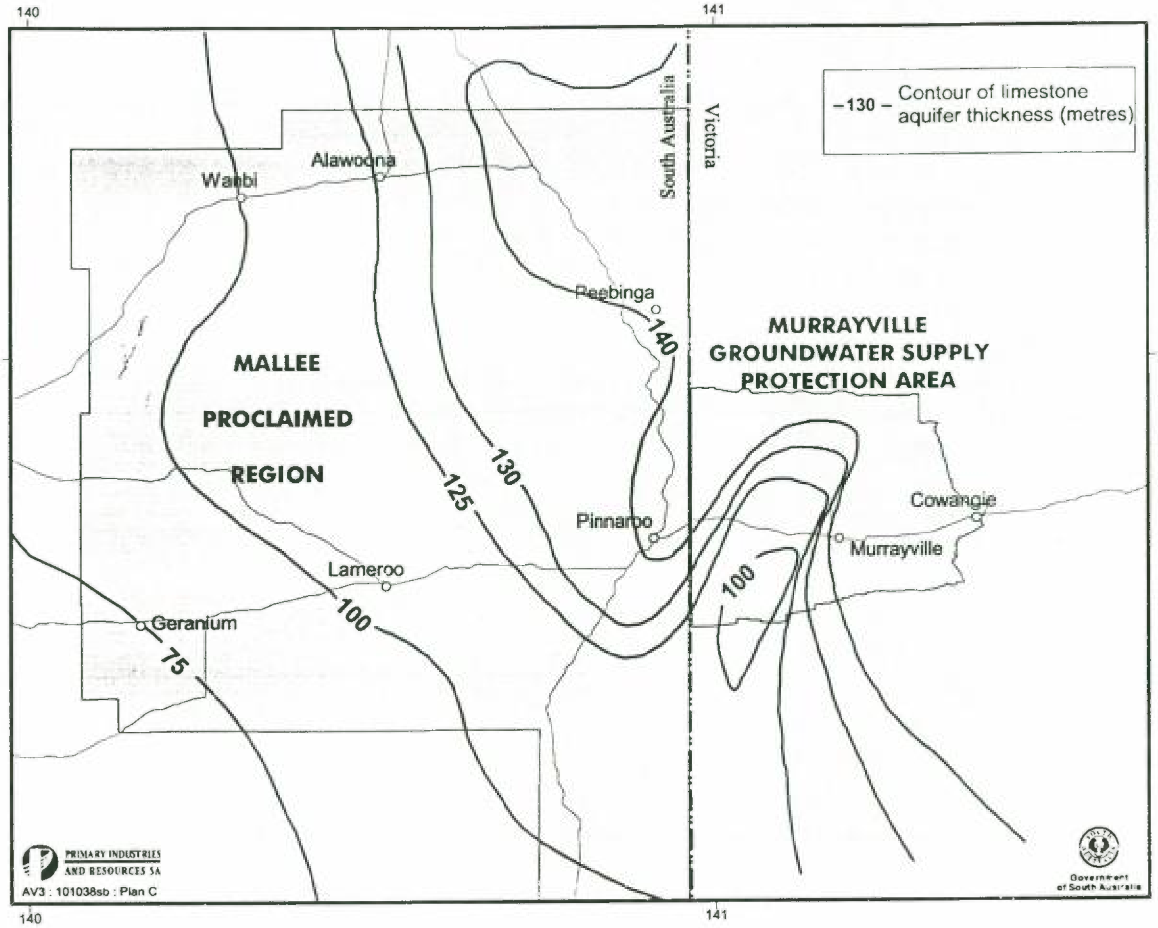

Figure 3 Murray Group Limestone aquifer thickness

Previous Management Strategies

The current management strategy in South Australia, which is currently under review, allows for controlled depletion or mining of the huge reserves in storage. The Permissible Annual Volume (PAV) is based on components of recharge, lateral throughflow and mining of storage which would result in a drawdown of no more than 5 cm/yr (Harris and Barnett, 1986). In Victoria, the current policy does not allow mining, and because the limestone aquifer is confined (no vertical recharge), only lateral throughflow is considered in determining the PAV which is consequently more conservative than SA.

Current Investigations

Evaluation of recent drilling information has shown that the Murray Group limestone aquifer is also confined in South Australia where most of the extractions are taking place. This has several consequences. Firstly, the extractions are from elastic storage and the current drawdowns (up to 14 m) are declines in the potentiometric surface and do not as yet, represent dewatering of the aquifer. Secondly, there is no contribution to the aquifer from vertical recharge, which is a long term blessing because the aquifer will be protected from the salt being flushed down through the unsaturated zone.

Examination of the potentiometric surface for the underlying confined Renmark Group aquifer and construction of a five layer MODFLOW groundwater model of the area (Yan and Barnett, this volume), have confirmed that upward leakage from the Renmark Group makes significant contributions to the water budget for the Murray Group limestone aquifer that has not been previously considered in management plans.

After taking the derived inflows, outflows and inter-aquifer leakage volumes from the model (Fig. 4), and assuming that current extractions were to increase by over three times to reach the maximum PAV, the mining of the resource would lead to a **depletion of only 15% after 300 years,** mainly due to the huge amount of groundwater in storage (100 million ML). Obviously, the drawdowns and salinity levels will have to be carefully monitored and modelled for adverse impacts from irrigation and the management strategies fine tuned as more information becomes available.

Local drawdowns

A major issue for the dryland farmers is the drawdown associated with irrigation bores and the effect on stock and domestic bores which extend only a few metres into the top of the aquifer and are often over 40 years old. In SA, the expansion of irrigation has been gradual, and irrigators have generally 'done the right thing' and paid for the deepening of pumps and bores, even though there was no legal requirement to do so under the legislation.

In Victoria however, the relatively sudden and concentrated development of irrigation in a small area close to the SA border caused a regional drawdown of 14 m by March 1998, which affected 32 stock and domestic bores. This understandably raised strong community concerns about the sustainability of the resource and culminated in the formation of the Murrayville Groundwater Task Force, a committee of 12 community representatives. A public meeting attracted 230 people.

Negotiations with the managing authority, Wimmera Mallee Water, ensured compensation for the costs of those affected (which was paid by the irrigators), and established cost sharing arrangements for future development. These include the installation of low flow cutoff switches at the landowner's expense, the lowering of pumps at the irrigator's expense, and a sliding scale of landowner / irrigator contribution for bore replacement depending on the age and casing type.

Management Options

Management plans are being formulated for the Mallee area in both States. In SA, a Water Allocation Plan is being prepared for the Mallee Proclaimed Wells Area, whilst in Victoria, preparation of a Management Plan is underway for the newly designated Murrayville Groundwater Supply Protection Area. The current cross-border cooperation in technical matters should continue in the policy area to ensure that sustainable development of the common aquifer is not hindered by superficial State boundaries and political interference. Extensive community consultation and education is also an essential ingredient in this process.

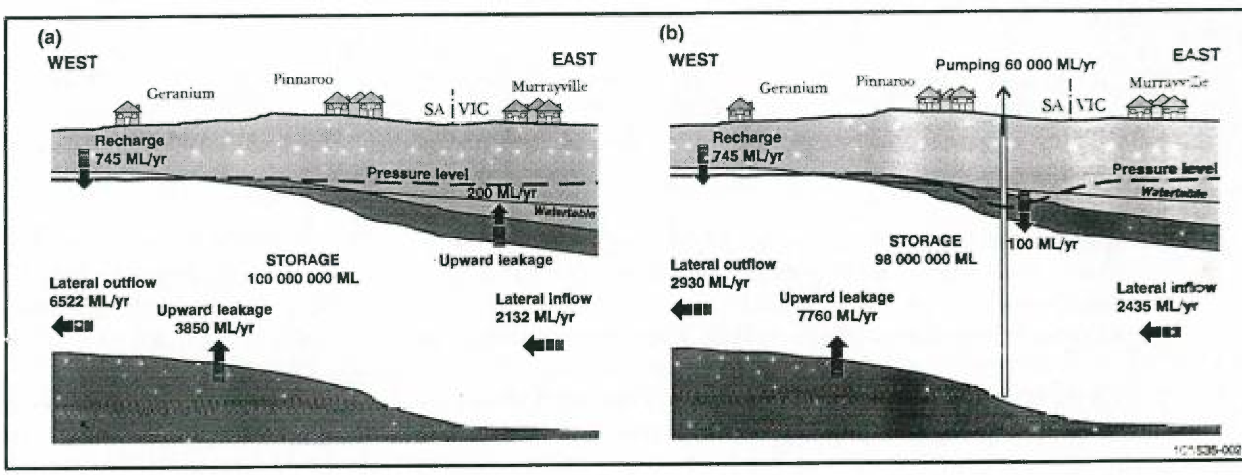

Figure 4 Water budget for the Mallee Region (a) before irrigation (b) after 30 years of maximum PAV irrigation

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MURRAY DARLING BASIN GROUNDWATER WORKSHOP September 1999, GRIFFITH, NSW.

Prioritising Salinity Management in the Goulburn – Broken Dryland

Bruce Gill[1], Greg Holland[2] ,Bill Trewhella[2], Xiang Cheng[3], and Mark Reid[3]

ABSTRACT: This paper describes the process developed to prioritise the salinity status of 69 dryland sub-catchments within the Goulburn – Broken Catchment. Using the available data throughout the catchment, scores were given for 6 key salinity parameters, namely discharge to land, discharge to streams, high water table, rising watertable, groundwater salinity and land clearance. A scoring system was developed which took into account the reliability and availability of data. A process to check the validity of the scores and overall rankings using expertise from throughout the catchment was also included in the study. The resultant ranking table allowed 16 high priority sub-catchments to be defined. This prioritisation exercise provides an objective system by which salinity monitoring and management expenditure decisions can be supported.

Background

In the Goulburn – Broken Catchment in North Central Victoria, significant on-ground works have been carried out during the last 10 years.to combat dryland salinity . Despite these works it is uncertain as to whether the current monitoring and implementation programs are targeted to the most appropriate areas.

With the need to weigh up the merits of salinity management projects in over 69 sub-catchments, a less subjective decision support system was required. It is with this aim in mind that the following project was undertaken.

The project utilised the greatly enhanced GIS database and mapping capabilities that are now available to make assessments of 69 sub-catchments (or parts of sub-catchments) in the Goulburn – Broken Catchment. These sub-catchments represent the more significant tributaries, but it is recognised that more detailed sub-divisions may be made in the future.

The key task was to systematically rank all the Goulburn – Broken Catchment Dryland sub-catchments in terms of the severity of existing salinity problems and future salinity risk. The catchment ranking produced will be used by the Goulburn – Broken Catchment Management Authority to prioritise works, identify areas where planning controls may be necessary, and expose areas where monitoring data was inadequate.

However it is recognised that in identifying sub-catchments in need of salinity mitigation works, a separate decision making process is necessary to determine the type of works that are appropriate. For example, some priority sub-catchments may have little prospect of effective groundwater pumping due to the absence of suitable aquifers, hence incentives for groundwater pumping would not be applicable. These catchments may therefore be more suitable for agronomic or saline drainage management options instead.

Method

Measures of Catchment Salinity Status

The sub-catchment boundaries of 62 defined sub-catchments were adopted in this exercise. However, due to significant differences in topography and salinity issues in sub-catchments containing both lowland hills and plains. seven were further sub-divided into plains and uplands sections. Of the 69 resulting sub-catchments, CLPR carried out assessment on the 53 upland sub-catchments (Cheng 1999) and SKM assessed the 16 plains sub-catchments (Goulburn-Murray Water 1999). G-MW was responsible for coordinating the whole exercise and in developing consistency across all the sub-catchments.

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The broad causes and symptoms of dryland salinity in the Goulburn – Broken Catchment are reasonably well understood. Based on this broad understanding, a group of key parameters was developed against which each sub-catchment could be scored.

The following listing is the initial group of potential salinity related parameters that were developed to assess the physical attributes of catchment salinity status together with an indication of local community readiness to recognise and confront salinity issues. The key parameters ultimately used for the ranking process are the 6 highlighted.

- ☐ ***Groundwater discharge to land(ie known salt problems)***
- ☐ ***Groundwater discharge to streams (ie elevated stream salinities)***
- ☐ ***Watertable depth (ie risk of salinity problems)***
- ☐ ***Rapidly rising watertables (ie risk of salinity problems)***
- ☐ ***High groundwater salinity(ie severity of land or stream salinisation)***
- ☐ ***Land clearance (major causal factor)***
- ☐ Land use (impact of salinity and management options)
- ☐ The catchment geology (landscape predisposition and expression of salinity)
- ☐ The level of community support (likely success of management programs)
- ☐ The current level of monitoring (adequate or inadequate)
- ☐ Recognition of previous priority areas

After some initial attempts to prioritise some trial catchments using the above list of parameters, it was realised that the main outcome desired was to rank the sub-catchments purely on the current salinity indicator conditions. Parameters such as the current level of monitoring, level of community support or geology were excluded from the ranking excercise for the following reasons:

- The level of community support is a management or implementation issue rather than a salinity risk factor. It should therefore be considered at a later stage when options for salinity control are considered.

- Geology and land use have a significant role in groundwater hydrology, however their affects are indirectly considered within other groundwater related parameters. Where the geology is considered to be a significant, salinity risk factor, it has been noted, but not included as one of the scored parameters.

- The current level of monitoring was initially to be used as a major 'skewing' factor to raise the ranking of catchments with little or no data. Catchments without bores for example would have received the maximum score for watertable depth. After trialing this approach, it was considered more appropriate to identify those catchments with inadequate monitoring. In this way, sub-catchments where there was little available data would be ranked relative to the available information but the need for investment in monitoring would be clearly identified.

Data sources

The results of the assessment for each sub-catchment is dependent on the data available. Data relevant to key parameters was collected from available maps, reports, files and public or community monitoring databases. The main activities related to the collection and interpretation of the existing information are described below:

- ☐ Discharge to Land – Known groundwater outcropping and salinised soil areas. A GIS discharge layer mapped by Allan (1994) was used to calculate the area of discharge to land. Any additional information from government or community groups was also considered.
- ☐ Discharge to Streams – Information on long term stream flow and salinity data was used to infer salinity levels due to either groundwater baseflow or salt wash-off from discharge areas. Some spot stream salinity data was also used. (Cook 1995 and Cook 1996)
- ☐ High Watertables – Recent standing waterlevels in shallow bores were used in conjunction with discharge area and topography to determine if a high watertable occurred, and if so, an estimate of the area.
- ☐ Rising Watertables – Where adequate time series data was available for observation bores, hydrographs were assessed for rising or falling trends.

- Groundwater Salinity – Observation bore salinity records were used to characterise the sub-catchment groundwater salinity.
- Land Clearance – The percentage of forest cover in each sub-catchment as determined from the 1:100,000-scale tree cover mapping in the GIS was used.

Scoring Procedure

It was agreed that existing saline discharge to land and streams are the most obvious symptoms of the economic and environmental damage arising from salinity. High watertables, rising watertables and land clearance were considered to be the main underlying factors of equal importance. Therefore, using the available data all sub-catchments were given a score out of 10 for each of the above parameters with the exception of groundwater salinity.

Although at higher salinities, groundwater has a greater impact on stream salinity, the impact on land salinisation is less critical. The long term presence of a shallow watertable can lead to waterlogging and significant land salinisation irrespective of the groundwater salinity. Groundwater salinity was therefore given only half the weighting of the other parameters (i.e. a score out of 5)..

Many sub-catchments had only limited stream salinity data or no stream data at all. In many cases, the data available was drawn from a range of lower reliability sources rather than formal monitoring programs. In many cases it was impossible to allocate an informed score.

There is likely to be some interaction between the colluvium/weathered bedrock mantle and the fringe riverine plain sediments at the fringes of the uplands. This potential interaction is relevant for development of remedial actions and monitoring strategies at the individual sub-catchment scale. Where this interaction is considered likely, comments highlighting likely interface issues were added to the ranking table.

Ranking Process

The average of the available parameter scores for each sub-catchment was computed and used for an initial priority listing. The results were used at a workshop attended by various people familiar with local characteristics of each sub-catchment. Recognising that scoring each parameter contains an unavoidable degree of subjectivity, extension staff and community representatives were able to discuss parameter scores and make adjustments if appropriate. For some sub-catchments, perceived similarities with other areas within the catchment justified modifications to a score. Through this process, care was taken to check for consistency across the Goulburn – Broken Catchment and the local input provided greater confidence to the scores given to each sub-catchment.

Based upon final ranking results, the 53 upland and 16 plains sub-catchments were divided into three priority groups, namely: high, moderate and low. However, the division of ranking into these categories is somewhat arbitrary. The considerable data variability and subjectivity applied in deriving some of the scores limits the ability to put definitive order to the rankings. In other words, the final ranking must be seen as a working document rather than as 'set in concrete order'. More detailed comparison of closely ranked sub-catchments, or subsequent consideration of other salinity management factors highlights the use of the ranking as no more than an aid to decision making.

Results

Ranking Summary Table

Summary tables produced for the Goulburn-Broken Catchment list all the sub-catchments with their scores and comments. They have been separated into plains and uplands sub-catchment groups because the salinity impacts and management options between the two physiographic divisions are likely to be substantially different.

Of the uplands catchments, 21 of the 53 have discharge sites mapped. The majority of these sites are found at break of slope locations or along stream incisions. Discharge sites are largely unknown in the plains sub-catchments, with only one having known discharge distant from the break of slope.

Stream gauging stations are present in only 27 of the 69 sub-catchments. Trend analysis results available for some of the gauged creeks providing improved confidence in the score. For many of the ungauged sub-catchments, spot salinity readings have been collected over two different years providing a score of moderate confidence. For the remaining catchments, determining the lack of data was a key outcome of the process.

There is a large variation in the quality and quantity of groundwater monitoring data in both the upland and the plains catchments. Of the 69 sub-catchments, 46 are considered to have inadequate groundwater monitoring.

The majority of catchments have more than 50% vegetation clearance, although the range is from less than 1% clearance in the mountainous south to nearly 100% clearance in many of the plains sub-catchments to the north. Many catchments therefore score quite highly for the land-use parameter.

Discussion of Results

Based on the ranking order and the priority divisions used, in the uplands there are 12 high, 19 moderate and 22 low priority sub-catchments. Within the plains, there are 4 high priority, 7 moderate and 5 low priority sub-catchments.

It is notable and not unexpected that the highest ranking catchments generally have a very high percentage of the land cleared. It has long been recognised that land clearing is a major causative factor in dryland salinity.

Sub-catchments with inadequate monitoring of groundwater and stream salinity have been clearly indicated. It should be noted that the scores only reflect an 'average' priority for the sub-catchment. For example, sub areas within a moderate priority sub-catchment may have significant salinity problems and a high priority at that smaller scale. This particularly applies in a large sub-catchment such as the Broken River/Creek sub-catchment.

It is also necessary to clearly highlight the limitations of the prioritisation process. Although one of the main aims of the project was to develop a more objective method of identifying areas most in need of resources to combat salinity, the variability in data quality and coverage has meant that many scores are inevitably based on subjective assessment.. Other scores may have been derived by extrapolating limited or unevenly distributed point data, for example observation bore readings targeted within saline discharge areas.

Nevertheless, the ranking tables do provide a starting point from which to begin addressing known dryland salinity problems. As additional data is collected and improvement to our understanding of the salinity condition of each sub-catchment evolves, so too would the sub-catchment rankings change over time.

Conclusions

- 69 dryland sub-catchments in the Goulburn – Broken Catchment were assessed for salinity status and risk. The ranking process produced 16 high priority, 26 medium priority and 27 low priority sub-catchments. High priority sub-catchments are predominantly cleared of their indigenous vegetation..

- The majority of presently salinised land occurs at the fringes of the uplands (break of slope) and along drainage lines.

- The method developed could be readily transferred to other catchments in the Murray-Darling Basin.

- Due to the variability in data quality and availability, the results of the ranking process should not be considered as a final product, but rather a working document which may be modified as a greater understanding of the sub-catchment is gathered.

- The ranking tables produced will provide useful input to the Goulburn-Broken Catchment Management Authority Salinity Management Planning process to support management and monitoring funding decisions.

- The results of the assessment should be regularly reviewed as more and better data become available. The format of the formal scoring process provides a ready reference for inclusion of new data as it becomes available and supplementary comments can be re-assessed at any time. It is expected that individual sub-catchment priorities will be reviewed on a needs basis while the overall catchment priorities can be reviewed as part of the Catchment's Salinity Management Plan five yearly review process.

Acknowledgement

The authors would like to acknowledge the funding of the Natural Heritage Trust for this project.

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Community Salinity Planning (Central West NSW)

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ABSTRACT

One of the greatest environmental threats facing the Central West of NSW is Salinity. The Central West Catchment Management Committee (CWCMC) has recognised that dryland and irrigation salinity requires immediate attention. A Strategic Plan is being developed to ensure that it is addressed appropriately and effectively. This plan offers a management approach which reflects a catchment perspective, identifying high hazard areas and then suggesting possible tools or techniques that will assist in achieving change. The process is based on the subjective evaluation of five criteria (geology, soils, landuse, slope and hydraulic loading) and exploring their individual and combined roles in salinity development. This project aims to promote community awareness and ownership of salinity within the Central West.

1. INTRODUCTION

The Central West Catchment includes the Macquarie, Castlereagh and Bogan Rivers, seen in Figure 1 and 2. It is located to the West of the Great Dividing Range, but is Central to the State of NSW. The Catchment is 92,000 km^2, approximately 10 percent of the Murray Darling Basin. The main drainage divisions of the Macquarie and Castlereagh Rivers head north west and join just before they meet the Darling River in the North of the Catchment. The Bogan River also heads north west and joins the Darling River downstream of the Macquarie.

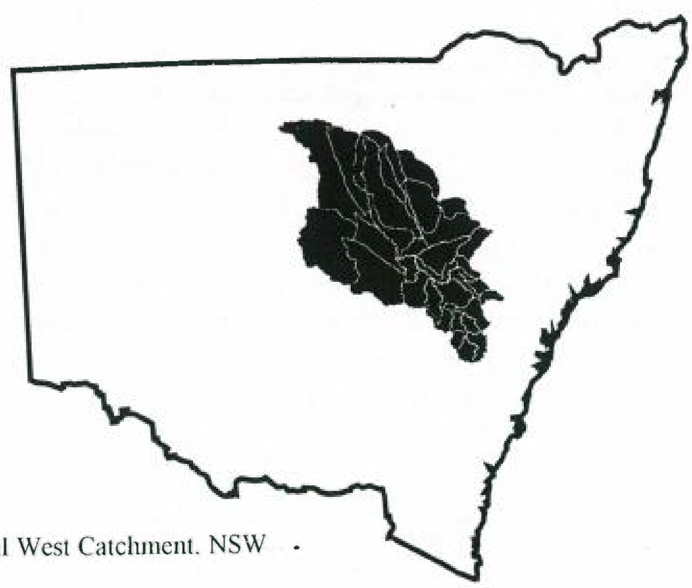

Figure 1. Central West Catchment, NSW

Salinity is regarded as one of the greatest environmental threats facing the Central West of NSW. It has been estimated that by the year 2050, the amount of salinised land in New South Wales will rise from current estimates of 120,000 hectares to 7.5 million hectares (PMSEIC, 1999). The Central West Catchment Management Committee (CWCMC) have recognised the importance of salinity within the catchment, but they do not have a comprehensive understanding of its extent, severity and distribution.

Dryland, irrigation and urban salinity are growing issues within the catchment community, but to date, no management approach has been developed that identifies focus areas, providing positive direction to the community. The CWCMC have identified that salinity requires immediate attention.

A strategic plan was needed to ensure that salinity was addressed and managed in the most effective way. This meant that the areas throughout the catchment that were most at risk from a current or potential salinity hazard could be recognised and given the necessary attention. More importantly, it would provide a catchment approach that identified the interrelationships that exist between biophysical features and land management occurring within the catchment. It would provide focus to areas that needed to be managed differently and in doing so, create benefits for the total catchment.

The overall objective of the project is:

> *"To provide the Central West Catchment Management Committee and the community, with a clear framework whereby salinity related activities are managed effectively and appropriately by all groups involved in their management"*

More specifically, the plan will provide the opportunity:

1. *To focus our efforts and concentrate on high priority salinity areas.*
2. *To provide effective methods of dealing with salinity throughout the catchment.*
3. *To provide a bridge between current activities and future objectives.*
4. *To improve the way we work together, striving for the common goal of improved catchment management.*
5. *To provide insight to individuals living in the catchment and improving their awareness of the extent, distribution and severity of salinity.*
6. *To provide an opportunity to equate activities at a local scale to those that are occurring throughout the catchment.*

2. BACKGROUND

Project Initiative

Recognition of the extent and scale of the problem, has until now, been the limiting factor in developing a strategic approach to salinity management. Natural Resource Government Agencies (Department of Land and Water Conservation, NSW Agriculture, Rural Lands Protection Board etc) have successfully recognised and addressed salinity issues at a local scale for a number of years. The majority of this work has occurred in response to community identification and concern and in most cases this has coincided within highly active Landcare Networks.

The strategic planning process was initiated by Allan Nicholson, Salinity Investigations Officer from the NSW Department of Land and Water Conservation (DLWC). He saw the need to improve the way salinity related activities were approached within the catchment. He recognised the need to move away from salinity related activities being determined, in the majority of cases, by public outcry. While remediation activities in response to public demand were justified, the question arose as to whether enough attention was being given to other areas within the catchment. Mr Nicholson was confident that his activities and those of the Salt Team (Salinity staff working in the Central West) were aimed at the right areas. He also recognised the need for a more strategic approach to facilitate better planning of activities at a catchment scale. The DLWC, in conjunction with the CWCMC applied for National Heritage Trust (NHT) funding to support this planning process.

The eighteen month project was approved and began in June of 1998. The first stage to this process was to determine what was required as project outcomes. The CWCMC outlined the following project objectives.

1. Improved direction and activity occurring within high risk catchments designated by the plan

2. Improved integration of works occurring by managing agencies within high risk catchments

3. Greater community awareness of the extent and scale of salinity within the Central West

4. A management approach that reflected a catchment perspective, ie. activities in one sub- catchment affecting the total catchment and work plans that consider these links

These objectives formed the foundation of the planning process.

Establishing an Audience

The diversity of 'stakeholders' and 'costs' involved in salinity, made it very difficult to plan for salinity in an integrated catchment manner. The only way to ensure planning success was to promote the involvement of key stakeholder groups throughout the project.

In the early stages of the project, time was spent identifying and interviewing various groups within the catchment. These people included Landcare groups and coordinators, Local Government Officers, Rural Lands Protection Board representatives, NSW Agriculture staff, DLWC staff, University Employees, Murray Darling Basin Commission etc. Each of these groups identified the importance of encouraging community participation; this would ensure that the planning outcomes were owned by the relevant stakeholder groups. All stages of the project were presented for comment to various community groups. The project's Steering Committee (CWCMC representatives) provided ongoing support for this process and continual community input.

Developing a Community Plan

To develop a community plan it was essential that the community had ownership over the project. The plan needed to incorporate the major concerns of stakeholders. These concerns were determined early on in the process and they highlighted the basic needs of stakeholders in improving their response to salinity. The plan needed to ensure that these issues were given consideration and addressed, while providing direction for future management.

The major concerns of stakeholders are:

- *What is the current salinity situation in the Central West?*
- *What will the future situation be?*
- *Who is responsible for salinity management?*
- *What does 'salinity management' incorporate?*
- *How can it be addressed in an integrated way with a catchment perspective?*
- *Do Federal, State and Local Agencies acknowledge the problem and do they have the skills tools to address it?*

3. THE PROCESS

Identifying Influential Criteria

Salinity is a response to a system out of balance, it is a symptom of change in our environment. Rate of groundwater rise and the amount of salt stored in the landscape were identified as the two defining factors in salinity development. Poor data coverage for these two criteria meant that other data would need to be identified and utilised to reach a similar outcome. The data listed below were nominated as preferable for the process.

- Salt Producing Potential of Lithology
- Geological Complexity Categorisation
- Change in Slope Categorisation
- Landuse Hazard Categorisation
- Soil Salt Store Categorisation
- Soil Permeability Categorisation
- Irrigation Loading Categorisation

Workshops were conducted for each of these criteria, where estimations were given for the percentage area of each category type. These areas were verified using spatial data in ArcView GIS. The percentage areas were multiplied by a subjective factor that indicated the hazard potential for that category. An example of the landuse factors used in this process are given in Table 1. These values represent a relative 'water use efficiency' factor for each land use type.

Cropping	10
Annual Pasture	9
Perennial Pasture (low water use)	6
Perennial Pasture (high water use)	2
Horticulture	8
Timber	1
Infrastructure (urban, water etc)	10

Table 1. Landuse Hazard Factors

These values were added together to obtain an overall value for each criteria, for each of the 37 sub-catchments. This value represented a 'hazard potential' for that criteria. Each of the criteria were then given a relative weighting against each other (1-10), to be used in a multi-criteria analysis process. This information aimed to highlight the significance of certain criteria in the overall development of salinity throughout the catchment. There were mixed opinions on the most appropriate way of weighting these criteria, however two schools of thought were evident. The first gave greater importance to the 'modifiers' of salinity such as Landuse and Irrigation loading. The second regarded the salt producing potential of the landscape to be the defining factor. Using either of the weighting options resulted in very similar outcomes, however the CWCMC regarded the 'modifiers' as being more important in the future management of salinity and hence the weighting's indicated this, seen in Table 2.

Salt Producing Potential of Lithology	4
Geological Complexity Categorisation	6
Soil Salt Store Categorisation	8
Soil Permeability Categorisation	5
Landuse Hazard Categorisation	10
Change in Slope Categorisation	4
Irrigation Loading Categorisation	10

Table 2. Final Weighting's for Bio-Physical Criteria

The final aim of this process was to rate sub-catchments or management areas according to their salinity hazard potential. A high 'hazard potential' could be generated as a result of an increased potential for salinity development within that catchment. However, it might also indicate the importance of changed land management in that catchment for the benefit of the total catchment. Each management area was given a rating (Very Low, Low, Medium, High, Very High), indicating a combined 'Bio-Physical' salinity hazard.

Acknowledging the Role of Socio-Economics

While each catchment had a 'bio-physical' risk, they also had a range of socio-economic inhibitors. The project investigated what social and economic issues were being faced by people living within the Central West Catchment. One question needed to be asked; What constraints are experienced by people living in a salinity affected catchment and how do these constraints limit the implementation of changed management techniques?

It was identified that there were a great range of issues that restricted individuals from adopting changed practices throughout the catchment. The Prime Ministers' Science, Engineering and Innovation Council, 1999, identified four issues that contribute to the lack of change occurring within a catchment.

- Failures in communication result in the poor adoption of changed management styles.
- The economic inability for farmers to act plays a significant role.

- Stimulating innovation to try something new conflicts with issues of historical and conventional farming trends, reliable markets, risk etc.
- Access to technical information is often a significant constraint in rural areas.

This plan allows for these issues to be recognised and considered. Four aspects have be incorporated into the plan, which explore the social and economic situations experienced in the Central West, the associated constraints, the factors that limit change and some recommendations on possible tools that would assist in achieving change.

4. CONCLUSION

This Salinity Strategic Plan for the Central West Catchment focuses on the recognition, ownership and responsibility of salinity. It does this by providing a framework whereby the extent and severity of salinity throughout the catchment can be prioritised, giving focus and direction to future management. It explores the factors which tip the balance in certain areas, resulting in further risk and it investigates the social and economic issues that influence the ability of a landowner to implement changed practices. By bringing this information together, a valuable tool has been provided to assist in promoting higher levels of awareness within the catchment. It also illustrates the connectivity of land resources and management over the catchment, clearly identifying the need for a integrated approach to salinity management in the Central West

5. REFERENCES

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Hydrogeochemical Processes Associated With the Occurrence of Dryland Salinity in the Longneck Creek Catchment, near Windsor, New South Wales.

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Abstract
The development of dryland salinity in the Longneck Creek catchment, near Windsor, New South Wales, is related to vegetation loss and soil structure decline caused by the discharge of saline waters with up to 25000 mg/l TDS. The source of salinity is attributed to connate salts in the Wianamatta Group Shales. Saline groundwaters from the deep confined fractured shale aquifers mix with lower salinity groundwaters in the Hawkesbury Sandstone and Rickabys Creek Gravel. The discharge of mixed groundwaters under artesian pressure with high sodium and chloride concentrations and hydrogeochemical reactions including ion exchange and the conversion of kaolinite to montmorillonite, coupled with reduced vegetation cover, results in the dispersion and erosion of soil.

1. INTRODUCTION
Dryland salinity is a land degradation issue of increasing environmental and economic importance. The consequences can be severe; declining agricultural productivity, reduction in surface water and groundwater quality, soil erosion and biodiversity loss; and subsequently dryland salinity has been the focus of many investigations. Dryland salinity can be expressed in two main morphological forms; seepages or scalds. The fundamental mechanisms involved in saline seepage development are the removal of native vegetation or change in land use, resulting in alteration of natural hydrological regimes and an increase in recharge and the mobilisation and redistribution of salts.

Scalding is the development of a hard impermeable surface on saline or sodic soils as a result of erosion of the surface horizon by wind or water or by the redistribution of salts in the absence of a groundwater system. The term is usually applied to the development of dryland salinity not associated with groundwater discharge. The term seepage scald, however, has been applied to bare saline patches characteristic of dryland seepage salinity in a classification by Williams and Bullocks (Taylor, 1993). The development of hard impermeable saline surfaces through deposition of salts and clays following evaporation of groundwater discharging under artesian pressure at Longneck Lagoon could therefore be called seepage scalds according to this classification. Aerial photograph studies have indicated the occurrence of dryland salinity at Longneck Lagoon prior to 1948. By 1982 five seepage scalds had developed around the perimeter of the lagoon.

Groundwater and surface water investigations at Longneck Lagoon have been conducted by Brodie (1991), Dames and Moore (1991), Kinhill Engineers (1992) and Barnes (1996). The aim of this study is to gain an understanding of the hydrogeochemical processes and water-rock interactions involved in the development of dryland salinity in the catchment from chemical and isotopic analysis. This study emphasises the importance of water-rock interactions and numerous hydrogeochemical processes in the development of dryland salinity identified in other studies (Jankowski et al., 1994). The majority of dryland salinity occurrences in New South Wales were found to be in areas with an average annual rainfall of 600 – 700mm, Ordivician metasediments as the geology and texture contrast soils (Wagner, 1987). The study at Longneck Lagoon is significant because it represents a different environmental setting for the development of dryland salinity, with a sandstone/shale geology and average annual rainfall significantly greater than 700mm.

2. ENVIRONMENTAL SETTING
Longneck Creek catchment is a semi-rural subcatchment of the larger Hawkesbury-Nepean catchment, located 40km northwest of Sydney (Figure 1). The catchment covers an area of $14km^2$ and is drained by two intermittent streams, Longneck Creek and Llewellyn Creek, which feed into Longneck Lagoon at the northern end of the catchment. The catchment has an average annual rainfall of 785mm pa and is often subject to flooding from rainfall runoff and backwater flooding from the Hawkesbury-Nepean River. The topography of the lower catchment is comprised of alluvial terraces associated with past and present courses of the Hawkesbury River and the upper catchment consists of low undulating hills lying on top of the Wianamatta Group Shales.

Longneck Creek catchment is located on the Cumberland Plain, a topographic depression at the centre of the Sydney Basin. The Wianamatta Group Shale, predominantly Ashfield Shale, is the dominant geological unit and underlies the majority of the catchment. The Ashfield Shale consists of shallow marine sediments and represents the base unit in a sequence of shales and minor lithic sandstones deposited during a single regressive episode in the Tertiary

(Gobert, 1976; Bembrick et al., 1991). The medium to coarse grained quartz sandstones of the Hawkesbury Sandstone outcrop to the east of Longneck Lagoon. To the west of the lagoon, Tertiary alluvial sediments known as the Rickabys Creek Gravels, associated with former courses of the Hawkesbury River over 6 million years ago, cover the sandstone and shale units. A relatively impermeable clay unit, known as the Londonderry Clay, deposited during the Tertiary as overbank deposits, overlies the gravels to the west of the Lagoon (Gobert, 1976). A surficial layer of Quarternary sand deposited by recent depositional events on the Hawkesbury River floodplain overlies the Londonderry Clay adjacent to the lagoon (Dames and Moore, 1991).

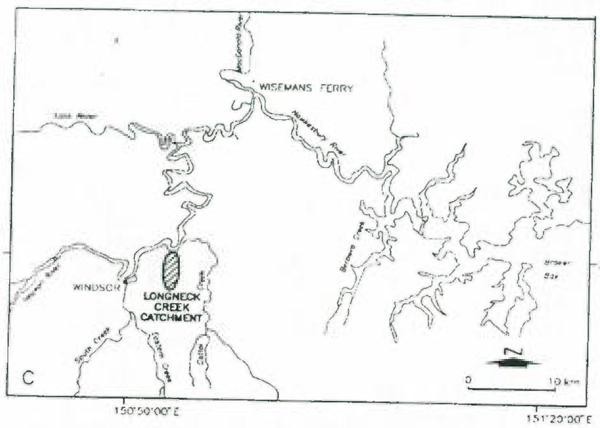

Figure 1a. Location map for Longneck Creek catchment.

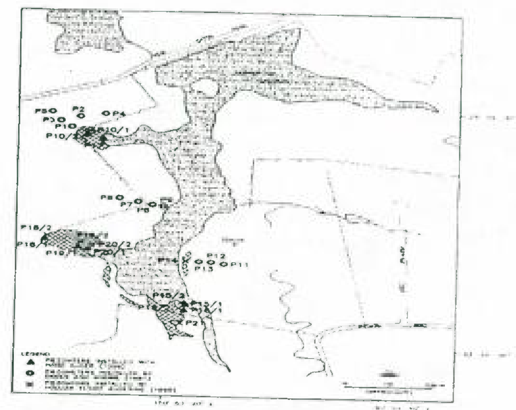

Figure 1b. Piezometer location at Longneck Lagoon.

3. HYDROGEOLOGY

The groundwater regime of the Longneck Creek catchment is characterised by a number of shallow and deep aquifer and aquitard systems. Shallow aquifers are located in recent and Tertiary alluvium along the Hawkesbury-Nepean Valley and deep confined aquifers are associated with Wianamatta Group Shales and Hawkesbury Sandstone. The majority of groundwater in the Hawkesbury Sandstone is located in fracture zones at depths less than 30m and bedding plane partings at greater depths. The groundwater table is controlled by the topography and varies from 10 – 30m across the catchment.

Longneck Lagoon forms a discharge zone for local and regional groundwater systems due to its position in a topographically low area. The Wianamatta Group Shale hills in the upper catchment form the recharge zone and have been cleared extensively for agriculture. Precipitation recharges the water table through fractures in the shale and the Hawkesbury Sandstone is supplied either by precipitation in outcrop areas or via saline accessions from Wianamatta Group Shales.

Transects of nested piezometers were installed on salt affected areas surrounding the lagoon by Dames and Moore (1991) and Barnes (1996) (Figure 1b). Deep piezometers have depths greater than 3m and lie in the Rickabys Creek Gravels which are confined by the overlying Londonderry Clay. Studies by Barnes (1996) of nested piezometers indicate that groundwater discharges under artesian pressure from the Rickabys Creek Gravels to the surface at the scalded areas. Prior to this investigation above average rainfall was experienced in the catchment and flooding and expansion of the lagoon resulted. Groundwater levels were much higher than previous studies, ranging from the surface to 0.24m. In wetter conditions the water level in the deeper piezometers was found to be lower than shallow ones at some piezometer nests (Figure 2), indicating rapid response to rainfall and recharge via rainfall and lagoon waters.

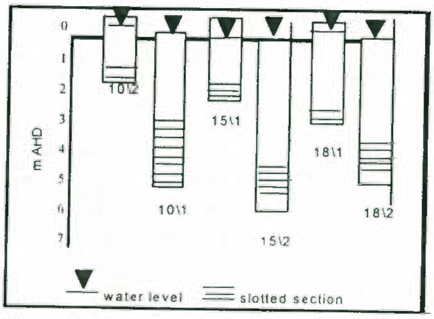

Figure 2. Water levels in nested piezometers identifying discharge and recharge zones.

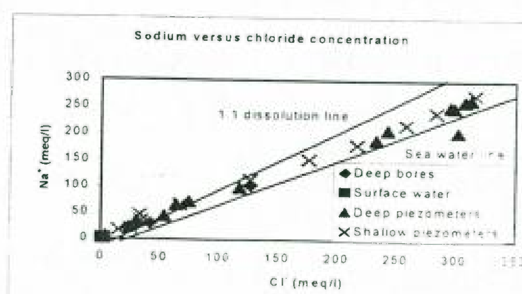

Figure 3. Relationship of sodium to chloride.

4. GROUNDWATER CHEMISTRY

The groundwater chemistry of nested piezometers and deep bores is shown in Table 1. All groundwaters are of similar type, being either Cl-Na-Mg or Cl-Na. Electrical conductivity (EC) generally increases along the flow path from approximately 3000μS/cm in recharge zones to 32750μS/cm in the discharge zone at Longneck Lagoon. Salinity generally increases with depth in salinised areas whilst the dissolved oxygen level and the Eh decrease. The pH values range from 3.9 to 7.0. Acidic groundwaters occur in the discharge zone through the oxidation of pyrite present in lagoon soils according to the reaction:

$$FeS_2 + 15/4O_2 + 7/2H_2O \rightarrow Fe(OH)_3 + 2SO_4^{2-} + 4H^+$$

The sodium and chloride in the system are derived from the simple dissolution of marine salts deposited in the Wianamatta Group Shales and concentrations of both ions increase along the flowpath. Mg is mainly derived by the weathering of chlorite in the aquifer matrix according to the reaction:

$$Mg_5Al_2Si_3O_{10}(OH)_{8\,(s)} + H_2CO_3 \rightarrow 5Mg^{2+} + HCO_3^- + H^+ + H_4SiO_4 + Al_2Si_2O_5(OH)_4$$

Some groundwaters in the discharge zone have a deficit of sodium to chloride (Figure 3) as these are located below the 1:1 dissolution/evaporation line, indicating the importance of reverse ion exchange and clay mineral formation in the discharge zone. If it is assumed that Cl^- is a conservative ion derived only from the dissolution of NaCl evaporites or mixing with connate salt waters, bivariate plots against Cl^- can be used to identify hydrogeochemical processes other than dissolution. In reverse ion exchange Na^+ exchanges for Ca^{2+} and Mg^{2+} resulting in an increase in the latter to the system according to the equation:

$$2Na^+ + Ca(Mg)\text{-clay} \rightarrow Na_2\text{-clay} + Ca^{2+}(Mg^{2+})$$

In this study chemical data was plotted on stability diagrams with respect to the aluminosilicate minerals and saline waters were found to be in equilibrium with kaolinite indicating that under wetter climatic conditions montmorillonite formation is not occurring. Barnes (1996) found the majority of waters were in equilibrium with Mg-montmorillonite and Na-montmorillonite under normal climatic conditions. Na^+ is therefore reacting with kaolinite in the discharge zone to form Na-montmorillonite according to the reaction:

$$3.5Al_2Si_2O_5(OH_4)_5 + 4H_4SiO_4 + Na^+ \rightarrow 3Na_{0.33}Al_{2.33}Si_{3.67}O_{10}(OH)_2 + H^+ + 11.5H_2O$$

Both ion exchange and reverse ion exchange processes are occurring in the discharge zone. Figure 4 shows the relationship between Ca + Mg versus SO_4 + HCO_3. The relationship will be 1:1 if the dominant processes in the system are the dissolution of gypsum, calcite and dolomite. Groundwaters undergoing ion exchange plot below the 1:1 dissolution line, with Ca and Mg depleted with respect to SO_4 + HCO_3 and those involved in reverse ion exchange plot above the line, due to an excess of Ca and Mg. The majority of groundwaters in the discharge zone plot above the 1:1 dissolution line indicating that they are undergoing reverse ion exchange. Ion exchange is not a significant process in deep bores but is in the discharge zone where there are readily available exchange sites on kaolinite and montmorillonite clays and abundant Na^+ which increases along the flow path due to the interaction between groundwater and the aquifer matrix.

Figure 5 shows the relationship between Ca vs Na-Cl, Mg vs Na-Cl and Ca+Mg vs Na-Cl. This figure can be used to discriminate between controlling ions (Ca or Mg) in ion exchange processes. Linear regression on the data set reveals the strongest correlation between Mg and Na-Cl (r^2 = 0.42). Therefore ion exchange processes (predominantly reverse ion exchange) are controlled by the exchange of Mg for Na.

Bicarbonate levels are extremely low or negligible in groundwaters with low pH due to the oxidation of pyrite. Corresponding SO_4^{2-} concentrations are high for these groundwaters. Acid sulphate soils lie 1 – 3m below the surface around the lagoon and seasonal fluctuations in surface water and groundwater levels expose pyritic sediments to oxygen, resulting in their oxidation and the generation of SO_4^{2-}. Although pyrite is common in shale it does not contribute SO_4^{2-} to deep groundwaters in the recharge zone because pyrite is not oxidised in the reducing environments that occur at this depth. Pyrite oxidation contributes significant amounts of Fe^{2+} (up to 48mg/l) to the groundwater in the discharge zone.

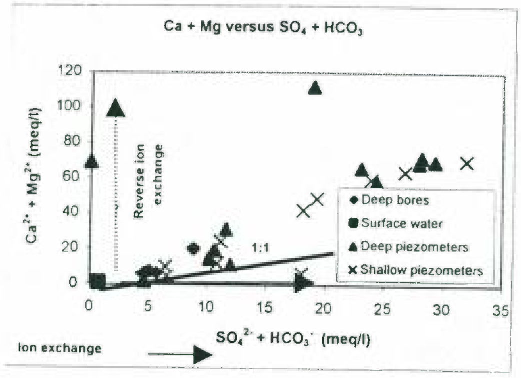

Figure 4. Relationship between Ca + Mg and SO_4

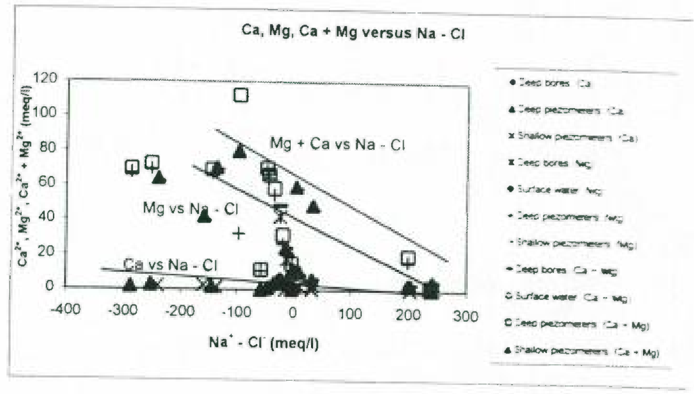

Figure 5. Relationships between Ca, Mg, and Ca + Mg vs Na – Cl

Oxygen-18 and deuterium isotopic data indicates that shallow groundwaters are rapidly recharged by rainfall but are effected by evaporative concentration during drier climatic conditions (Figure 6). Shallow groundwaters lying to the right of the local meteoric water line (LWML) indicate enrichment in heavier isotopes. This suggests that evaporation has occurred since recharge and is supported by a negative deuterium excess of –3.91 °/₀₀. Evaporation is likely to be occurring from the unsaturated zone or surface on discharge of saline groundwaters under artesian pressure. Shallow groundwaters that lie to the left of the LMWL and are located in non scalded areas are depleted in the heavier isotopes and are likely to have been recharged by heavy rainfall or infiltration from deep aquifers.

All groundwaters plot on a straight line indicating mixing of shallow and deep groundwaters in this system. Saline groundwaters from the Wianamatta Group Shales are mixing with fresher groundwaters from the Hawkesbury Sandstone and shallow groundwaters in the Tertiary alluvial sediments along the flow path. Mixed saline waters discharge near the lagoon. The scattered relationship between oxygen-18 and Cl (Figure 7) indicates that mixing is not complete and values that are depleted in heavy isotopes are likely to be caused by rapid recharge and incomplete mixing of these groundwaters with pre-existing groundwaters in the catchment. A linear relationship on this diagram would indicate evaporation. Evaporative concentration is only occurring in shallow groundwaters in seepage scald areas from the surface discharge of saline groundwater.

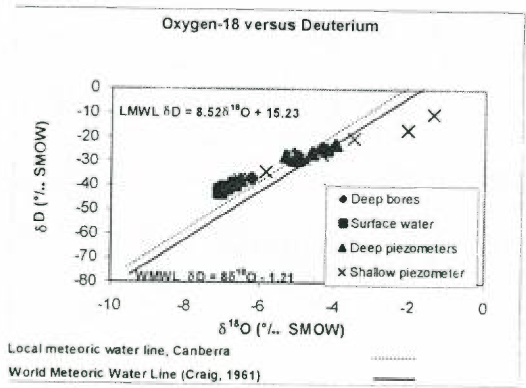

Figure 6. Plot of deuterium versus oxygen-18.

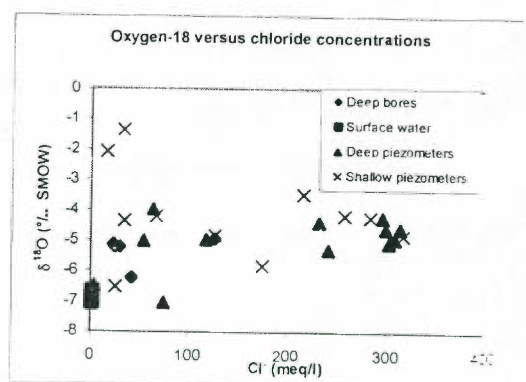

Figure 7. Relationship between oxygen-18 and chloride

5. DEVELOPMENT OF SALINISATION PROCESSES

The Wianamatta Group Shales are the main source of salinity in the Longneck Creek catchment. Values up to 31750mg/l TDS have been recorded in groundwaters from the shale (Woolley, 1991). The high salinity of the groundwater arises from connate seawater trapped during sediment accumulation in the Tertiary. Groundwaters from the Hawkesbury Sandstone are generally of reasonable quality, with TDS values ranging from 200 – 1200mg/l (Woolley, 1991). In the Longneck Creek catchment, isotopic studies indicate that saline groundwater accessions and mixing with groundwaters from the Wianamatta Group Shales, has increased the salinity of groundwaters from the Hawkesbury Sandstone. The discharge of mixed saline groundwaters from the deep confined fractured aquifers through the Rickabys Creek Gravels under artesian pressure has led to the development of seepage scalds near the lagoon. Groundwater movement through the system is slow due to low permeabilities and hydraulic conductivities of the sandstones and shales, resulting in increased water-rock interaction and accumulation of salts. Clearing of the catchment for agriculture since the early 19th century has increased groundwater movement through the system and enhanced the mobilisation of salts in shale and clay.

The discharge of saline waters with up to 25000mg/l TDS has resulted in the loss of vegetation surrounding the lagoon and a decline in soil structure with subsequent development of seepage scalds. Groundwaters are dominated by sodium (140 – 6310mg/l) and chloride (130 – 11225mg/l) and also contain appreciable amounts of magnesium (35 – 840mg/l). The ion exchange of Mg^{2+} and Na^{+} for Ca^{2+} has resulted in a loss in soil structure and subsequent dispersion of soils. Dispersion occurs because Na^{+} ions have a greater hydration radius than Ca^{2+} and Mg^{2+} ions and their incorporation into exchange sites in clays prevents contact between clay particles.

Salts are concentrated by evaporation, as indicated by isotopic data and ionic ratios, on seepage scalds during the summer months and a white salt crust often forms. In winter, heavier rainfall dissolves concentrated salts and increases the salinity of the soil pore water, resulting in the dispersion of soils surrounding the lagoon. The cyclical dissolution and precipitation of salts by evaporation and rainfall dilution has the effect of increasing salt concentration in the top of the soil profile. Run-off containing salt laden sediment has deteriorated the quality of lagoon waters, increasing the salinity and turbidity of Longneck Lagoon.

Swelling of clays reduces the permeability of the soil leading to a decline in drainage efficiency and increasing waterlogging. Waterlogging is prevalent in low-lying areas surrounding the lagoon. It is not necessarily related to salinity, but rather is the continuous saturated condition maintained by water tables located at or near the surface. The dispersion and erosion of soils on scalds has lowered the landsurface and effectively brought the water table closer to the surface. Waterlogging induces an oxygen deficiency in the root zone, placing vegetation under stress. The anaerobic conditions created by waterlogging allow reduction processes, such as the reduction of sulphate to HS^- and H_2S and nitrate to N_2, and fermentation processes, producing CO_2 and CH_4, to dominate. A combination of waterlogging and soil nutrient deficiencies induced by this process, in addition to discharge of saline groundwaters and insect attack has resulted in the dieback of mature eucalypts around Longneck Lagoon. Eucalypts began dying in 1975 and by 1978 were completely dead.

Indicators of water quality for irrigation include the sodium adsorption ratio (SAR) and the magnesium hazard (MH). The SAR relates the concentrations of sodium to magnesium and calcium ions in the soil by the following equation and indicates excessive sodium:

$$SAR = Na/\sqrt{(Ca + Mg)/2}$$

All groundwaters have a very high salinity hazard and a SAR above 8, which is the threshold for good soil structure and plant growth (Figure 8). Deep groundwaters have a lower SAR than shallow groundwaters near the lagoon because Na concentrations have been enhanced in shallow groundwaters by ion exchange processes in clays in the discharge zone. The magnesium hazard (MH) for irrigation waters is calculated by:

$$MH = Mg/(Ca + Mg) \times 100$$

All groundwaters in the Longneck Creek catchment had a MH which exceed the threshold of 50, indicating a very high magnesium hazard (Figure 9). High magnesium concentrations are derived from the weathering of chlorite and reverse ion exchange.

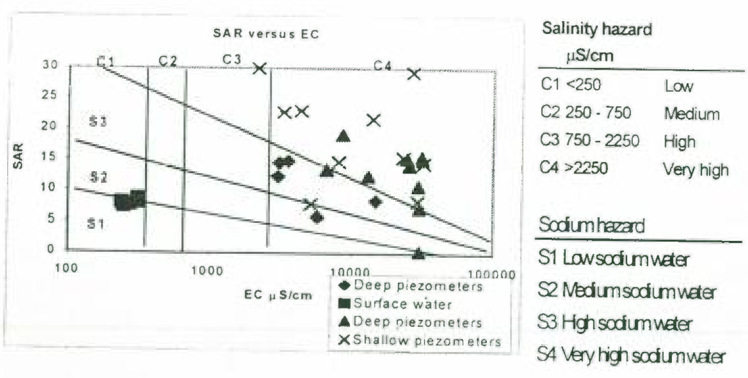

Figure 8. Sodium Adsorption Ratio versus EC

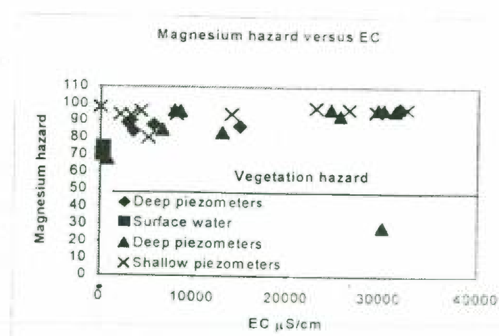

Figure 9. Magnesium hazard versus EC

Table 1. General parameters and major ions in the surface water and groundwater.

Sample Number		Temp °C	EC mS/cm	TDS mg/l	pH	Eh mV	DO mg/l	CO_2 mg/l	Na	K	Ca^{2+}	Mg^{2+} mg/l	HCO_3^-	SO_4^{2-}	Cl	SiO_2
B1		19.7	3085	1835.7	5.83	-67	0.6	176.0	542.8	6.62	13.76	70.0	192.2	119.4	870.1	[illegible]
B2		20.2	14885	8784.9	6.14	-108	0.2	307.2	2375.0	39.00	126.0	511.0	657.8	550.0	4506.7	[illegible]
B3	1	19.7	3575	2109.4	5.75	-87	0.5	162.0	627.9	7.78	23.60	75.00	162.0	114.2	1057.0	[illegible]
B4		19.6	3030	1683.1	6.01	-81	0.4	151.4	478.0	11.84	14.98	72.00	158.0	98.80	828.3	[illegible]
B5		20.2	5770	2949.8	6.07	-88	0.9	158.4	687.5	16.46	50.60	216.0	263.6	216.0	1485.1	[illegible]
S2		16.0	242	147.6	6.28	-67	2.2	12.32	33.27	3.74	4.30	6	28.31	11.97	54.08	[illegible]
S3		16.0	321	179.9	6.34	-68	2.7	11.44	41.52	4.05	5.24	7.51	30.51	9.74	74.91	[illegible]
S7	2	18.1	316	174.3	7.07	-4	3.9	10.56	41.52	3.74	4.19	7.43	26.73	10.95	75.50	[illegible]
S9		15.5	248	144.5	6.48	2	3.1	15.84	30.29	3.84	4.29	5.90	29.29	9.54	55.13	[illegible]
S10		14.7	277	161.1	6.41	-94	1.6	14.96	34.57	3.80	4.48	6.48	31.73	12.08	62.02	[illegible]
P1		16.7	13035	7495.4	6.41	-109	0.3	97.70	2261.0	11.80	104.0	318.2	147.1	443.0	4179.7	[illegible]
P2		17.3	31950	19632	3.97	-57	1.1	274.6	6118.0	31.00	23.60	835.1	0.00	1407.8	11120.4	[illegible]
P3		16.3	25650	15416	6.34	-172	0.0	220.1	4768.0	14.70	76.00	663.1	341.1	901.8	6603.5	[illegible]
P5		17.2	30300	18442	6.22	-257	0.0	144.4	4712.0	22.40	1601.4	390.1	162.9	782.0	11737.0	[illegible]
P7		16.7	8610	5070.3	6.72	-222	0.0	88.02	1644.7	10.52	12.82	173.0	325.2	234.0	2552.9	[illegible]
P8	3	15.2	24650	14607	6.52	-116	0.5	42.25	4351.0	7.30	37.00	779.7	222.1	933.0	9258.4	[illegible]
P10-1		16.8	30350	18603	5.48	-117	0.7	213.0	5814.0	17.82	39.60	814.1	94.58	1269.8	11501.7	[illegible]
P12		15.2	829	590.3	7.07	-140	0.2	29.93	137.4	3.20	12.42	15.83	254.4	20.77	129.7	[illegible]
P15-2		15.2	6665	3799.8	6.49	-115	0.5	88.02	1019.5	11.04	58.00	201.0	350.9	236.0	1906.9	[illegible]
P16-2		16.1	7940	4588.8	6.40	-94	0.8	132.0	1507.3	8.60	8.44	133.0	328.3	319.0	2263.1	[illegible]
P16-2		17.2	31550	19433	5.03	-128	0.1	264.1	6042.0	16.52	40.20	816.0	44.91	1467.2	10932.4	[illegible]
P19-2		17.7	29700	18732	6.11	-170	0.0	184.8	5757.0	20.80	61.00	840.8	235.5	1150.8	10814.9	[illegible]
P9		14.1	19270	11320	6.93	-89	0.3	34.33	3496.0	4.68	56.00	477.4	257.5	778.0	6238.2	[illegible]
P10-2		15.2	29500	17886	5.79	-45	0.6	264.1	5548.0	23.40	44.20	752.4	120.2	1248.4	10064.2	[illegible]
P13		14.0	3325	1922.2	6.66	-208	0.0	85.36	479.0	5.63	40.76	101.1	283.1	86.56	889.6	[illegible]
P14		13.8	5190	3336.8	6.99	-111	0.3	38.73	1075.0	0.30	7.82	71.00	623.8	376	1167.8	[illegible]
P15-1		13.7	2175	1258.4	6.48	-92	0.7	80.98	400.0	2.88	2.56	35.00	224.5	36.60	543.8	[illegible]
P16-1	4	15.6	4425	2447.6	6.27	-96	0.6	66.01	818.4	3.32	2.82	57.00	138.9	206.4	1204.1	[illegible]
P17		17.4	26650	16189	4.71	-46	0.8	434.8	5016.0	6.28	30.60	705.2	14.03	1136.0	9154.3	[illegible]
P18-1		16.4	32750	13470	3.87	-1	0.9	294.0	6308.0	13.72	31.60	837.0	0.00	1533.0	11225.0	[illegible]
P19-1		16.3	23050	20065	4.79	-122	0.1	169.0	4142.0	5.90	56.00	557.3	10.37	918.6	7690.2	[illegible]
P20-1		16.0	8145	4640.4	4.92	-86	0.2	126.8	1511.1	12.22	14.38	138.0	13.42	512.0	2381.7	[illegible]
P21		17.3	14015	7988.8	6.03	-69	0.5	55.45	2603.0	13.04	9.10	291.0	97.63	459.0	4500.5	[illegible]

1 Deep bores 2 Surface water 3 Deep piezometers 4 Shallow piezometers

6. CONCLUSIONS

The discharge of saline groundwater with up to 25000mg/l TDS under artesian pressure has led to the development of dryland salinity at Longneck Lagoon. Salts are derived from the simple dissolution of connate salts in the marine sediments of the Wianamatta Group Shales. Groundwaters from the deeper confined fractured aquifers are mixing with shallow groundwaters in the Tertiary and recent alluvial deposits and are discharging under artesian pressure near the lagoon. Many hydrogeochemical processes including ion exchange and the reaction between cations and kaolinite to form montmorillonite, have altered ion concentrations of discharging groundwaters. Discharge of groundwater with high TDS, SAR and toxic concentrations of Na, Cl, Mg, reduced elements and gases has resulted in the loss of vegetation, dispersion of soils and subsequent development of salt scalds. In addition. the oxidation of pyritic sediments near the lagoon has resulted in acidic groundwaters with low pH and bicarbonate concentrations and high sulphate and Fe^{2+} concentrations. The discharge of acid, saline groundwaters into Longneck Lagoon could have serious consequences for aquatic life and vegetation.

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Regional scale dryland salinity risk prediction – a review

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Abstract: A 'regional-scale' approach to dryland salinity prediction can provide a more consistent appraisal of salinity risk, and also allow a comparison of local problems with the wider region. This paper reviews a range of methods and approaches that have been used by various researchers and State Agencies to assess dryland salinity risk for a range of areas at different spatial scales. Salinity assessment methods are described in the first part of the paper. The second part of the paper then considers the use of historical trend data as part of a predictive approach to a consistent regional scale dryland salinity risk assessment.

1. Introduction

Researchers and State Agencies have used a variety of methods and approaches to assess changes in dryland salinity, depending on the spatial scale required and the data availability. A range of these predictive approaches are discussed in this paper. This review highlights that there is still no readily accepted and adopted method to predict dryland salinity in a consistent manner at a regional scale. Confidence in the results of predictive approaches is needed, to allow the extrapolation of other areas as part of an assessment of 'regional scale' or Basin-wide dryland salinity risk. This can allow managers to assess local problems in context with the wider region, and help them to focus their attention on key areas. Confidence in the outcomes of comparable predictive approaches at a regional scale can result from an understanding of the physical processes which are driving the expansion of dryland salinity.

Given that dryland salinity is a problem that needs to be examined consistently at a regional scale, this paper discusses methods which have been used to predict the spatial extent, and the temporal changes of dryland salinity extent into the future. The first part of the paper presents methods which have been used to assess dryland salinity risk, describing some approaches which have used *spatial* data to assess dryland salinity risk. The second part of this paper then suggests an approach which incorporates *temporal* trends in groundwater level to aid the predictive capability of dryland salinity risk prediction.

2. Salinity risk assessment

A number of previous studies have been conducted to predict the area of potentially salinised land, within several locations throughout Australia. A range of methods and assumptions have been used in this work which creates limitations when the results of these approaches are extrapolated to a regional or Basin-wide scale. This section first describes the links between dryland salinity processes, spatial data sets, and risk factors. It then discusses three categories of methods which have been used to weight the relative importance of these individual risk factors. Note that many of these approaches have aimed to predict the *current* rather than a *future* extent of salinity.

2.1 Processes leading to dryland salinity

Landscape features, surface biophysical processes and climatic drivers are three key catchment processes that contribute to dryland salinity, and which need to be considered when assessing salinity expansion in Australia. Although these processes and attributes are generally well understood, scientists and managers are not so clear on their spatial distribution at a regional scale or their precise contribution to salinity risk. This spatial distribution of salinity risk factors has been attempted in a number of previous studies, although the choice of data sets relevant to these factors has been inconsistent.

The clearance of native vegetation has led to higher groundwater recharge, which in turn has led to increased discharge of saline groundwater (Allison & Hughes 1983). As such, knowledge of both recharge processes and groundwater movement processes in a particular catchment can be used to predict the future extent of salt-affected land (or salt impact on streams), and also to make an assessment of various available options. However, without an extensive and costly groundwater investigations program needed to determine groundwater characteristics and other parameters across large areas, it is difficult to extrapolate these predictions to a regional scale.

The difficulty of predicting future dryland salinity extent is further exacerbated by the variation between catchments that can lead to different catchment behaviour. This variation can be the result of differing spatial scale, since groundwater systems can be regional or local in nature, or by impediments to groundwater flow which can cause discharge of saline groundwater (Figure 1). As a result of these variations, the extrapolation of salinity risk prediction across functionally different catchments remains problematic. A framework is required which can provide confidence in this extrapolation of results from study areas to provide larger scale assessments.

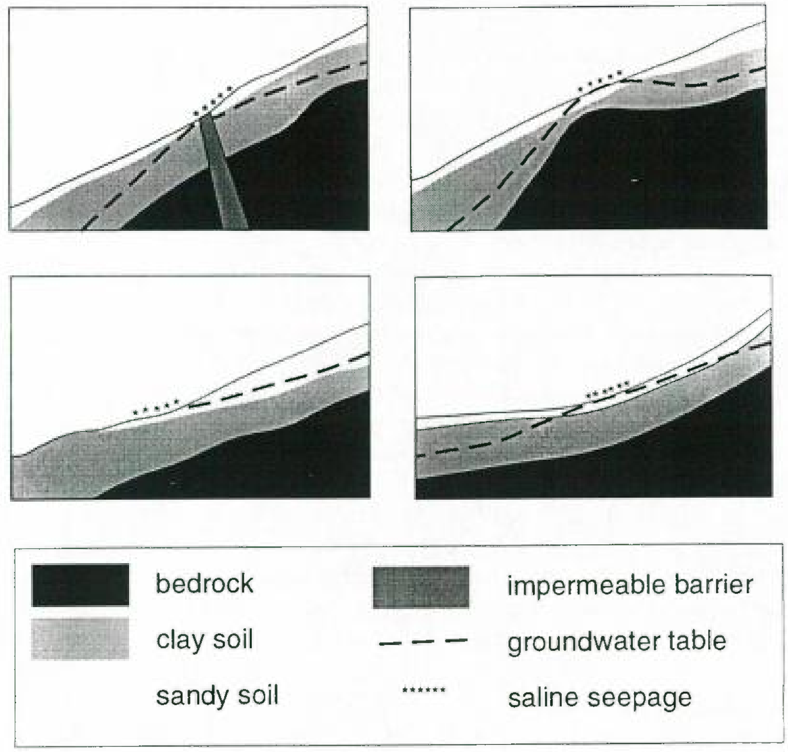

Figure 1 Examples of physical constrictions in groundwater flow leading to groundwater seepage (and the onset of dryland salinity)

2.2 Salinity Risk Factors

The catchment processes leading to dryland salinity include recharge and groundwater movement. These can be separated into a number of parameters or 'risk factors' which can then be used to assess dryland salinity risk:

- **Recharge** rainfall characteristics, vegetation cover / land-use, soil hydraulic conductivity
- **Groundwater** slope, aquifer width, aquifer conductivity, groundwater elevation

In order to provide an assessment of dryland salinity risk, these risk factors need to be linked to appropriate spatial data sets (Table 1). This is where much of the difficulty of prediction arises, due to the lack of spatial data sets at suitable scales. The paucity of catchment data often results in the need to use whatever data are actually available, as surrogates for actual 'risk factors'. For example, digital elevation data is one of the few data sets available across Australia which can be used in studies at a range of scales; it has increasingly been used to infer parameters for which no other data is available. Previous studies have also implicitly determined groundwater processes, from attributes such as land surface elevation (terrain attributes), since these groundwater processes are particularly difficult to determine spatially.

PROCESS CHARACTERISTIC	SPATIAL DATA SET
Recharge	
rainfall characteristics	interpolated rainfall
vegetation cover / land-use	land cover
soil storage / conductivity	soil type / geology
Groundwater	
slope	topography
aquifer width	
aquifer conductivity	geology
groundwater elevation	interpolated bore information

Table 1 Data sets used as parameters for dryland salinity process

In many of the small-scale study areas, it is not necessary to explicitly mention the variation in 'risk factors' such as rainfall seasonality. For example, it is common to use only median or mean annual rainfall, although regional factors such as winter dominance and episodicity of rainfall are known to be salinity 'risk factors'. The effect of these differences across Australia are shown clearly by the choice of annual rainfall belts for salinity risk (*e.g.* in northern Australia, the high risk rainfall belt has been 500 – 1500 mm, whereas in south western Australia it has generally been 300 – 1100 mm, and in south eastern Australia 400 – 800 mm). Another reason for the use of implicit factors is that potential problems can emerge due to the high degree of correlation between different 'risk factors'. causing difficulty with the statistical analysis used to assess risk (*e.g.* elevation can be highly correlated with rainfall. soils and land use).

2.3 Methods to Predict Salinity Risk

Salinity risk assessment is concerned essentially with two types of prediction – spatial and temporal. While some approaches are calibrated to incorporate change by tuning training sets to some estimated potential. most approaches aim only at predicting the current spatial extent of salinity.

Salinity risk assessment methods need to assign different 'risk factors' to a range of suitable process characteristic. depending upon their relative importance to salinity risk in the area of interest. In order to produce an overall prediction of salinity risk for a given catchment. the salinity risk factors need to be combined. There are several techniques of combining these risk factors. although most of the existing approaches use simple *composite index methods* which use authors' experience to weight the individual risk factors. *strongly inverse methods* that rely on statistical analysis. or *trend based methods* which incorporate temporal change.

As well as inconsistencies between these different weighting methods. there is also a range of approaches used to gauge their reliability. In some cases. the training sets encompass the whole study area. making it hard to distinguish whether the underlying assumptions are correct. or if the data is just being recycled. Ideally the prediction should be calibrated with a completely separate 'validation' area. which should be somewhat different in the combination of 'risk factors'. The use of a separate 'validation area' has been used occasionally. but rarely on an area distant from the training set. An example is the NSW study by Tassell (1995). which divided the Tout Park area into a northern part (training area) and southern part (test area).

2.3.1 Composite index methods

This type of method is one of the more simple ways of combining the salinity risk factors. and predicting a salinity risk from several layers of spatial data. The methods involve weighting a combination of risk factors based on the authors' experience and their understanding of the processes. Each of the data layers (*e.g.* soil type. land cover. topography *etc.*) are given a rank and weight. and then the data layers are summed them using a linear additive model (LAM). to provide a spatial representation of areas of dryland salinity risk (Figure 2).

Examples of approaches which have used this method include Tickell (1994) who used a LAM to apply an equal weighting to five salinity risk factors for the Northern Territory. Another approach was used by Searle & Baillie (1998) who ranked each risk factor using a 'rule based' weighting before using an index based LAM to predict landscape salinity hazard for an area of SE Queensland. Fuzzy logic approaches have also been applied. where a normalisation technique is used to weight each factor evenly. Rules are then used to combine various 'risk factors' (Dowling *et al.* 1997. NSW). A training set can then be used to define a threshold of this composite index related to salinity risk.

2.3.2 Strongly inverse methods

These methods are differ from the composite index methods. since the current salinised area is included as a spatial data set. These methods uses an approach which is able to objectively determine the relative weights of the risk factors based on the current salinised areas (Figure 3). This requires a large number of saline sites. with the actual number depending on the relative independence of the 'risk factors' for each process. The strongly inverse methods have relied on decision tree analysis' (Tassell 1995: NSW). or 'weights of evidence' (Bradd *et al.* 1997: NSW). as a method for weighting risk factors. 'Rule-based' and 'expert' methods are also being increasingly applied (Kirkby 1996: SA). However. one of the major limitations of these strongly inverse methods is that any aspects of future salinity which is unrepresented in the current training set will not be predicted. This is of concern where regional flow systems which have may contribute to the future onset of salinity but which presently have no indication of salinity can not be predicted as an area of future salinity risk.

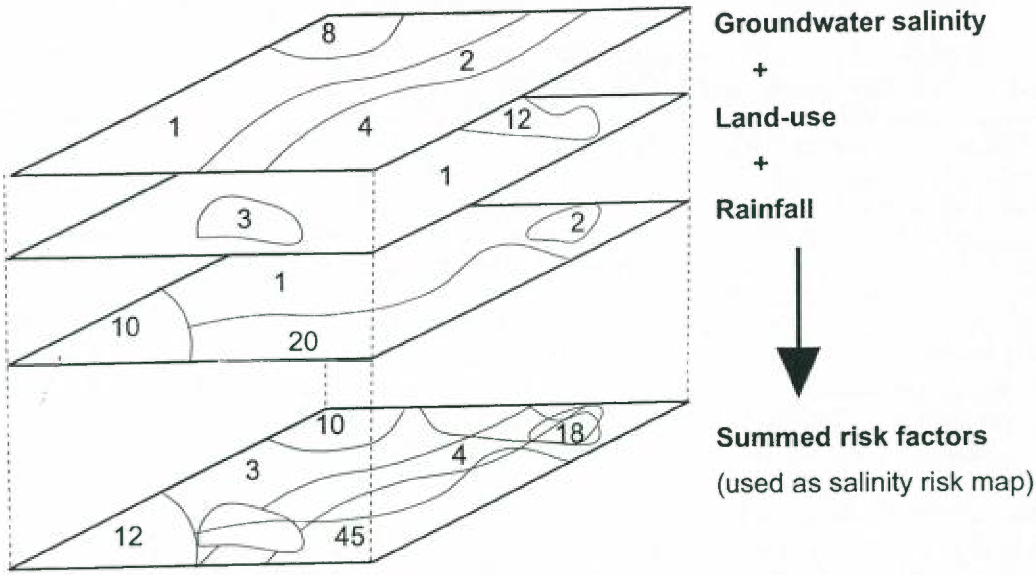

Figure 2 Composite index model.
The assumed risk factors for each data layer are summed to produce an overall salinity risk map. (*After* Tickell 1994).

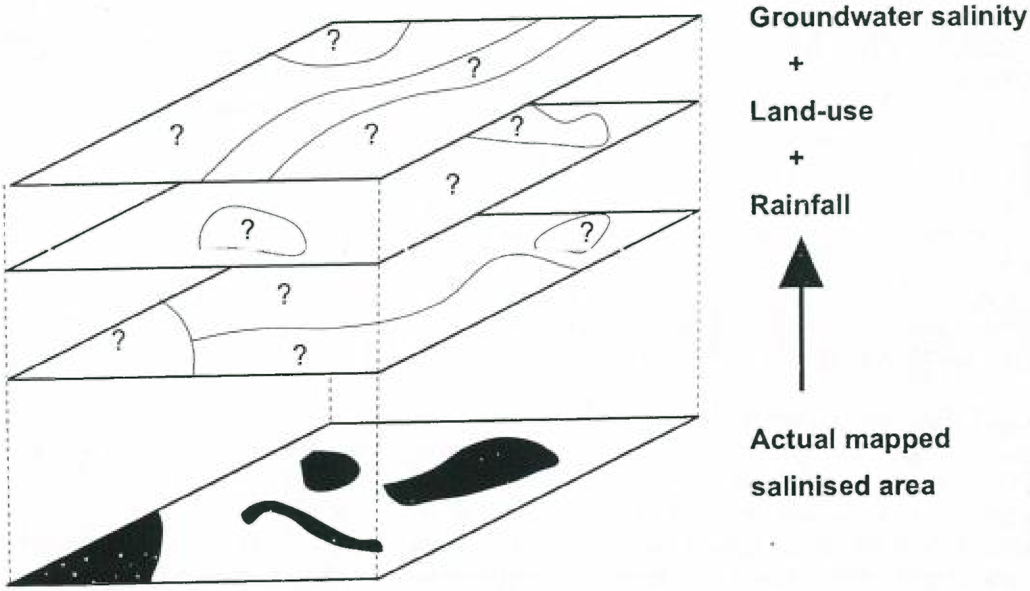

Figure 3 Strongly inverse model
The relative importance of risk factors in the data layers is predicted statistically, using a map of actual current salinity.

2.3.3 Trend based methods

Most approaches used to predict dryland salinity are concerned with the *spatial* extent of dryland salinity rather than attempting to provide a prediction of the *timing* of that salinity extent. Historical trend data provide a useful tool to link trends in salinity with changes in catchment processes. This link is the key to the accurate prediction of trends into the future. Although consistent trend data are not always available, especially at a regional scale, our knowledge of processes in a range of small-scale well-instrumented catchments can be used to extrapolate to the this larger scale The use of trends in past land salinisation, stream salinity, and groundwater elevation, are discussed in this section.

Land salinisation trend approach

To obtain an actual trend in land salinisation, a measure of change is needed. This can be achieved using topographic land surface data when extrapolating mapped trends of the expansion. By making an allowance for the land elevation, prediction of future salinity trends can be improved. This method has been used to obtain a prediction of land at risk for regional studies in New South Wales, Western Australia, and for the Victorian component of the Murray Basin (Ultimate Salt Loads Project, C. Clifton, *pers. comm*). This approach compares the changing area of shallow groundwater depth with surface salinity trends.

Historical changes in the salinised area over discrete time intervals can also be used to predict future trends in land salinisation. Sequences of remotely sensed maps have been used to look at temporal changes in land salinisation at a regional scale. For example Furby *et al.* (1995) used a series of Landsat data to determine the change in area affected by salinity in representative areas of the Western Australian wheatbelt. Kirkby (1996) also used Landsat data to determine vegetative cover for the Jamestown study area in South Australia. For both these studies the ground truthing for these images was gained from the analysis of visible salinity in historical aerial photographs, and through field work.

In most cases, the land salinisation trend method tells you the obvious - that existing salinised areas are becoming larger. As such it is most useful for shorter term predictions, since over longer periods salinisation may begin to occur in locations which currently show no signs of salinity, and these will not be included in any future salinity extrapolation. Also rates of expansion of salinised areas may change as a result of underlying catchment processes, or the effects of changes in the topography of the affected area.

Stream salinity trend approach

Stream quality as it exits a catchment represents one of the best integrators of all subcatchment processes and can be used to indicate of catchment salinity status (or other aspect of catchment health). Trends in stream salinity can provide a measure of catchment change, although this can be problematic with the extremely large natural fluctuations that occur in Australian stream flow and salinity. Methods have been devised to cope with these fluctuations, although finding sufficient and systematic records of stream salinity to determine stream salinity trends has been a major problem (Jolly *et al.* 1997). In a recent study within the Murray-Darling Basin, areas characterised by increasing stream salinity trends were likely to have high salt imbalances, increasing confidence in Jolly's interpretation (Walker *et al.* 1998). Greig & Devonshire (1981) have also developed a regression between an implicit trend in stream salinity and various 'risk factors' including rainfall, proportion of forest cover, and proportion of rock type for Victorian catchments. Unfortunately, such regressions are not necessarily transferable to relationships that exist outside the small-scale catchments from which they were derived, and so can not readily be used to predict the impacts of land use change on stream salinity in other catchments.

Long-term monitoring networks have been established to measure surface water discharge, although water quality parameters have not been measured in such a consistent pattern until recently. Even so, given that many catchments have long-term discharge records, stream data can still be used to compare the relative importance of various processes to salinity risk, for different catchments. However, the fact that stream salinity data is an integrator of catchment processes limits its potential for precise spatial prediction of salinity risk, since the trends are not specific enough to identify individual processes.

Groundwater trend approach

Groundwater trend data can be used as a first stage in predicting a land salinisation trend. They can provide a detailed temporal account of catchment changes, which can then be linked to other salinity risk processes. Since groundwater movement is a fundamental factor leading to dryland salinity, the use of groundwater trend data can improve our understanding of the processes directly. Comprehension of the processes that lead to changes in groundwater elevation at a study catchment scale will enable the relative contribution of physical processes and attributes to be quantified The ability to analyse the effect of historical catchment processes over time makes groundwater elevation data a powerful tool.

The assessment of groundwater elevation trends for an entire region or Basin can be simplified by dividing groundwater bores into separate categories such as groundwater type, 'bio-region', landscape element or screened aquifer. Bores that are located within a similar climatic zone, subject to the same land use, or in consistent groundwater type, could be expected to show similar trends. A recent study showed this to be the case in the Loddon Campaspe catchment in Victoria (Salama *et al.* 1996). If this result is more widely applicable, then the landscape could be disaggregated into individual groundwater catchment types, and data from fewer bores would be required in total.

From a management perspective, the effects of land use changes on dryland salinity are important information. Changes in land use which produce a change in groundwater recharge, could be expected to lead to groundwater elevation change and hence alter the salinity risk of an area. So the use of historical groundwater trends is a step towards assessing the effects of known land use changes with respect to changes in dryland salinity extent. However, increased recharge does not necessarily translate into a long-term groundwater rise. Drainage from the groundwater system, recharge of deeper fractured rock systems, or subsurface movement may all cause overestimation of the modelled prediction of groundwater rise and thus misprediction of aquifer behaviour. So, in the absence of detailed groundwater data, catchment modelling is perhaps the only option to help fill the gaps in current knowledge.

3. Predictive methods to predict dryland salinity risk

Most of our understanding of dryland salinity processes has come from the results of a number of well-studied catchments, which have been intensively mapped, instrumented, and in some cases subjected to changes in land use. The aim of such studies was to understand processes with the intent of applying this process understanding to other catchments. However, the cost of universally undertaking such detailed techniques is prohibitive, providing incentive for the development of less expensive extrapolation techniques which are able to expand this understanding more widely.

In order to develop 'risk factor' assessment monitoring and modelling to allow this extrapolation to areas other than the location of the study, it is necessary to have an approach based around a process understanding for a range of similar catchment types. Disaggregation of the landscape into sets of catchment categories will then allow the results of intensively modelled catchments to be extrapolated to other catchments with a much greater level of confidence than is currently possible.

A range of appropriate modelling tools can be developed and tested for each catchment type, such as the types described by the recently developed *National Classification of Catchments* (for land and river salinity control) (Coram 1988). This classification operates at a range of hydrogeological scales, linked to the scale of the groundwater processes (although it can then be aggregated up to the scale of interest is required).

3.1 Catchment modelling

Conventional groundwater models usually require estimation of the parameters recharge, specific yield, conductivity, and aquifer thickness at each cell. The confidence of the predictions is dependent on the quality of the conceptualisation and the quality of available data. However, even in an intensively studied catchment, such as the Liverpool Plains (NSW), there is scarcely enough data in the non-irrigated areas to justify the time and difficulty in interpreting results from complex modelling. The paucity of data means that a less time consuming approach could achieve similar degrees of confidence in output to that of complex modelling. However, in some of the more complex regional systems a simpler approach is not possible. In these special cases complex models will be required, since if appropriate data is available, any type of groundwater system can be modelled.

In catchments with sufficient data, it is possible to model groundwater processes, recharge and costs of impacts of salinity using conventional groundwater models such as MODFLOW and AQUIFEM-N (except in some fractured rock systems). Models such as TOPOG-IRM and MIKE-SHE are also able to simulate both surface water balances and groundwater. These models require not only the groundwater parameters for each cell, but parameters associated with soils, vegetation, climate and terrain. To model surface water processes accurately, cell size needs to be less than about 50 m, with an accurate digital elevation model (DEM). In practical terms, this leads to a maximum modelled area in the order of 10 km^2.

In some cases, the groundwater system is such that a simplified groundwater model can capture the key processes (*e.g.* FLOWTUBE model in the Liverpool Plains, or the groundwater component of HARSD approach). Other useful models may be those that specialise in the simulation of deep drainage leading to groundwater recharge (*e.g.* WAVES, APSIM, PERFECT and AgET). The advantage of these simpler models is their ease of calibration and interpretation, while a drawback is their lack of applicability to every groundwater system.

3.2 Salinity risk prediction

It is unlikely that there will ever be sufficient measured data to fully calibrate a complex, process model (such as MIKE-SHE) at a regional scale. A more realistic approach is to model a range of data-rich catchments as a basis for understanding the physical processes and estimating 'risk factors' for each catchment type. This will require great familiarity with geomorphic and groundwater processes within each region, a strong sense of the biophysical attributes influencing groundwater recharge, and an appreciation of their relationship with local land management practices.

An appropriate strategy would be to develop a tool-kit of various models, as part of a 'best approach' over the wide range of differing groundwater systems. An appropriate groundwater modelling framework can be applied to a data-rich example of each catchment type, in order to develop a set of simple rules relating to the relative contribution of salinity risk factors. Temporal changes in groundwater elevation can be linked to a set of 'risk factors' based on relevant physical salinity risk processes. After confirming the accuracy of the predicted salinity risk using mapped land and stream salinity data, results can then be extrapolated over the larger, data-sparse areas in catchments of the same type, using spatial interpretation through a geographic information system (GIS), together with the use of digital elevation models.

4. Conclusion .

To date, confidence in the results of a range of dryland salinity risk prediction studies has been generally restricted to the actual study catchments. This has prevented the consistent assessment of potentially salinised areas at either a regional or Basin-wide scale, particularly in data-sparse catchments. As well as an inconsistency between methods.

there has been inconsistency in the way in which the different risk factors have been weighted or combined to produce a salinity risk prediction.

A suitable approach is to use groundwater trends as a basis for modelling catchment behaviour. By modelling a particular catchment, a set of simple rules can be developed, which can be used to predict dryland salinity risk. Confidence in the transportability of these rules to other catchments of a similar type, is gained by the underlying comprehension of the processes in each of the catchment types.

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SALINITY – URBAN & INTERCEPTION

URBAN SALINITY IN WAGGA WAGGA (NSW): SOURCES OF RECHARGE AND IMPACT OF PUMPING THE FRACTURED ROCK GROUNDWATER SYSTEM

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1. Introduction

Damage caused by salinity to infrastructure is estimated to cost at least $100 million per annum. In addition to transport, telecommunications and electricity networks, townsites themselves are under direct threat. At least eighty rural towns Australia-wide are affected by salinity, and the number is growing. Urban salinisation is a result of rising groundwater tables and the recharge processes involved can be complex. In urban environments, recharge can either be local (leaky reticulation system, garden and park irrigation, residential rubble pits) or regional (agriculture in surrounding catchments). Understanding the partitioning of recharge between the different sources is critical for salinity control, and this understanding can not be complete without an in-depth knowledge of the groundwater system.

Management of urban salinity is more difficult in fractured rock catchments. Fractured rock aquifers underlie approximately 40% of Australia, including much of the southern and eastern highlands of Victoria, NSW and Queensland. Groundwater flow in these systems is governed by the length, spacing, aperture and orientation of the fractures. Well yields are usually highly variable, largely depending on the number and size of fractures intercepted by boreholes. Hydraulic conductivity is usually similarly variable, making determination of groundwater flowrates difficult. Also, in most cases, fractures have a preferred orientation, which may create a large anisotropy in hydraulic conductivity. This can mean that the groundwater flow direction will often not be in the same direction as the hydraulic gradient, and the drawdown cone under pumping may be elliptical, rather than circular.

Wagga Wagga was one of the first towns to recognise and attempt to address problems of urban salinity. Over the past six years, several studies have been undertaken to determine the magnitude of the problem, and investigate management options (e.g. Hamilton, 1995; Paul *et al.*, 1996). Preliminary estimates of the relative importance of recharge from the different sources were made, although these were based on numerous assumptions, casting some doubts on their reliability. Management options involving both recharge reduction and discharge enhancement were evaluated. Isotopic and hydrogeochemical analyses, and an in-depth study of the structural geology of the fractured rock aquifer beneath Wagga Wagga will improve the understanding of the groundwater processes involved in urban salinisation in Wagga Wagga, and help optimise management options.

2. Site Description

2.1 Location

Wagga Wagga is located 450 km southwest of Sydney, and has a population of approximately 60,000. Originally, the city was confined to the floodplain on the southern bank of the Murrumbidgee River. Currently, the urban development occupies 44 km^2, expanding towards the lower hillslopes south of the floodplain.

2.2 Geology

The Ordovician metasediments comprise the oldest formation locally, and are interpreted as a flysch deposit (Degeling, 1980). The sediments are composed of alternating shales and sub-greywackes. They were altered during the Silurian with the emplacement of the Wantabadgery granite, causing low-grade alteration to generally green schist facies. In the vicinity of the granitic intrusion, some may be altered up to granulite facies. Recent Cainozoic deposits of gravelly siltstone, and clay-rich alluvium have mantled the Ordovician metasediments and the Silurian granite.

2.3 Hydrogeology

The hydrogeology of the fractured metasedimentary aquifers is characterised by high variability in yield. This is due to the interactions of a low porosity, low permeability matrix with fractures that can provide substantial quantities of water. Individual production bore yields generally range from 0.1 to 2 L/sec although yields as high as 12 L/sec have been reported from more highly faulted unweathered sediments. These aquifers may have strongly anisotropic flow and a large range of permeabilities. Groundwater flow is from south to north, with discharge to the Murrumbidgee River. Within the past 100 years, however, aquifer recharge has increased, through a combination of land clearance and urbanisation. Urban recharge results from leakage from reticulated water supplies and sewerage lines, irrigation of domestic and public gardens, parks and playing fields, and from roof runoff, which in many areas is directed into rubble pits located on individual residential blocks. The increased recharge has caused water tables to rise, resulting in salinity problems in low-lying areas. Low permeability silts and clays associated with the river floodplain restrict drainage of the groundwater to the Murrumbidgee River, and thus contribute to the high water tables.

2.4 Salinity Management

Salinity management has involved schemes both to reduce recharge and to increase discharge. An ongoing education program aims to highlight the value of salt-tolerant vegetation, and to reduce garden overwatering by residents. Revegetation and rubble pit removal programs also aim to reduce local recharge. A borefield has been constructed to lower water tables in the worst-affected area, and should be operational from July 1999. Disposal of the saline groundwater abstracted will be to the Murrumbidgee River using a temporary permission to pollute permit from EPA NSW.

3. Methods and Results

3.1 Impact of Groundwater Pumping

The groundwater pumping scheme aims to reduce water levels by withdrawing water from fractured rock aquifers underlying the most saline areas. The location of pumping bores has been based on the common assumptions of isotropic, horizontally flowing aquifers. However, the wells are mostly screened in fracture shale, and the orientations of the major fractures are likely to affect the shape and extent of the drawdown cone under pumping. Mapping of fracture orientations at outcrops in the region has identified three primary fracture sets: bedding plane fractures and a set of shear plane fractures (Figure 1). The bedding is nearly vertical in the region, and the strike of the bedding is northwest/southeast. The preliminary hypothesis is that the bedding planes fractures are more permeable than the shear plane fractures. Thus hydraulic conductivity is expected to be highest in the northwest/southeast direction, with the drawdown cones elongated in the same direction. The absence of a regional set of horizontal fractures suggests that vertical permeability should be strong. This may result in a reduced lateral extent of drawdown, and necessitate a relatively dense bore network. This type of information will improve conceptual models of the aquifer and improve the effectiveness of the pumping scheme.

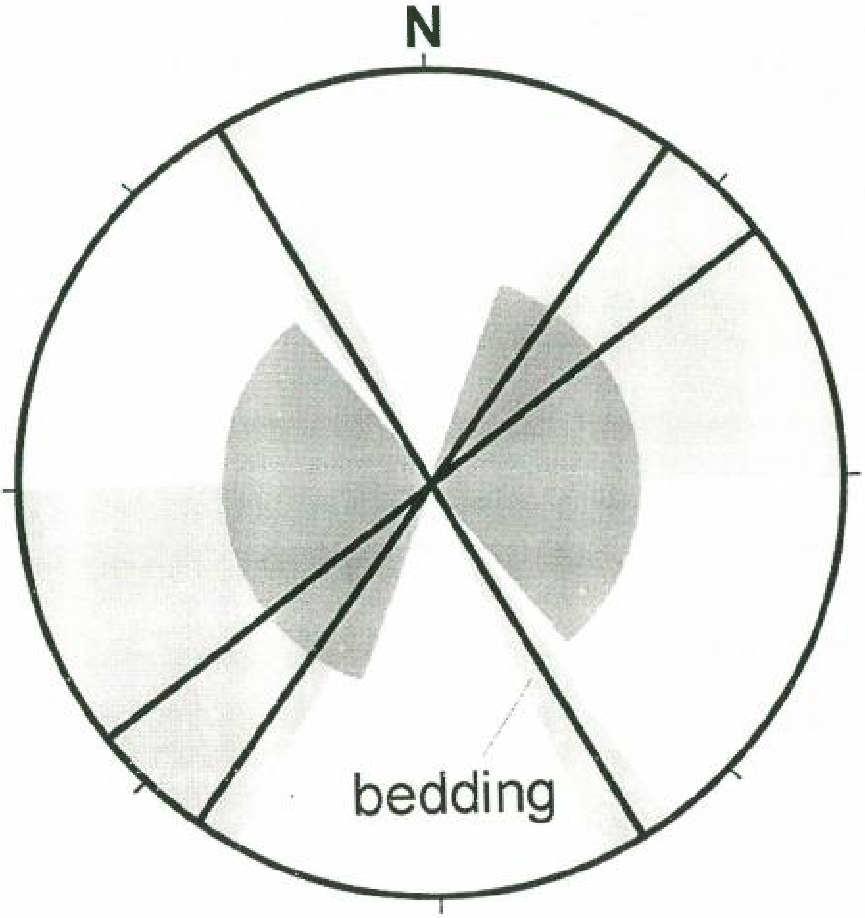

Figure 1. Rose diagram of fracture orientations observed in outcrops at the Southern Roadbase quarry, south of Wagga Wagga. Solid lines indicate average orientations of fracture planes. Shaded areas indicate the full range of variation in the data set. The bedding plane fracture are believed to be more permeable than shear fractures, suggesting higher hydraulic conductivities in the northwest-southeast direction.

3.2 Recharge Sources

Recharge reduction programs can be better focussed if the relative magnitudes of the different recharge sources can be properly determined. Hamilton (1995) used a water balance approach to obtain an initial estimate of the volumes of recharge over a 425 hectare area surrounding the most saline region. She suggested that pipe leakage amounted to approximately 55% of recharge, rubble pits 20%, and local recharge 25%. However, all of these estimates are based on a range of assumptions. Local recharge was not measured, but estimated as 2% of rain falling on unpaved areas, plus 3% of applied irrigation. (Hamilton considered a range of values for each recharge source; the above represent the mean values of her ranges.) Recharge from pipe leakage is similarly poorly known.

In order to improve these estimates, an attempt is being made to use water chemistry to distinguish between the various sources of recharge. The reticulated water supply for Wagga Wagga is derived from bores adjacent to the Murrumbidgee River, which draw groundwater recharged directly from the river. Preliminary analyses show that the reticulated water supply has a HCO_3/Cl molar ratio of approximately 3.5, whereas rainfall has a HCO_3/Cl ratio of approximately 0.5. Similarly, the Mg/Cl ratio of reticulated water is 0.75, and that of rainfall is approximately 0.1. Measurement of HCO_3/Cl and Mg/Cl ratios in saline groundwater beneath saline areas may thus allow determination of the relative proportions rainfall-derived recharge and that derived from the reticulated water supply. The latter would include both pipe leakage and irrigation recharge, and further tracers would be needed to distinguish these two sources. Stable isotopes of water (^2H and ^{18}O) may improve the discrimination.

4. Conclusions

While there has been considerable research into groundwater recharge beneath dryland and irrigated agriculture, recharge in urban environments has received relatively little attention. As urban salinity problems become more widespread, techniques for measuring urban recharge and discriminating between different sources of recharge will become more important.

Groundwater flow in fractured rocks is also poorly understood. Most research has focussed on unconsolidated, sedimentary aquifers, which can often be characterised by a single value for hydraulic conductivity. Prediction of groundwater flow in fractured rock aquifers is much more difficult, and the efficiency of groundwater pumping schemes may be difficult to predict in advance. Nevertheless, measurement of anisotropy in hydraulic conductivity, whether through fracture mapping in outcrops, or from specially designed pumping tests, should allow better bore network design and improved efficiency of these schemes.

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TITLE : DEVELOPMENT IN AN URBAN SALINITY AFFECTED CATCHMENT- A CASE STUDY IN TROY GULLY, DUBBO, NSW

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1.INTRODUCTION:-

Urban salinity is affecting most towns in the Central West of NSW. and is not widely accepted that it also affects most towns in the state as well. Considerable technical work has been carried out to define the nature of causes of salinity in Dubbo. This technical understanding has led to issues with regard to development in a salt affected urban catchment.

Causative functions of urban salinity relate to an increase in water accessing the watertable from diffuse sources such as drainage from rubble pits, leaky pipes (storm, sewer), overwatering and inefficient watering practices. as well as the process of urbanisation itself. Many towns were built on geologically high hazard sediment sequences, have catchment shapes that cause constriction, or town layouts that create drainage problems. In the case of Dubbo existing saline discharge areas were present prior to settlement, due to combination of geology and catchment shape.

The community is being impacted by urban salinity in the following ways :-
* Infrastructure damage to - roads, footpaths, guttering
  - parks, gardens, sporting fields
  - private and public buildings
  - public land and facilities
  - service delivery (gas,water, electricity, sewer)
* Heritage damage to buildings and other infrastructure (parks etc)
* Environmental degradation of soils, waterways and vegetation leading to loss in flora and fauna within urban areas and environs.
* Limiting options for future planning decisions
* Increasing water costs from repair of leaking reticulation systems.

In the Troy Gully Catchment impacts are evident in all the above scenarios. with major effects on private land and greenspace areas. Development within the area has been approached from a partnership of activity between Local Government, Urban Landcare group, Agency and Developers. The acceptance that a salinity problem exists within the Troy Gully area has had impacts on :-
1. Zoning of land
2. Building codes
3. Development consents and design
4. Strategic planning of city development
5. Stormwater design

2.CURRENT SITUATION:

2.1 EXISTING DEVELOPMENTS

Troy Gully catchment is located to the east of Dubbo, with an area of some 42 square kilometres. It contains industrial land to the west (bottom of catchment), and significant residential land in rest of catchment. There is increasing pressure to develop agricultural land for residential development.

Some current development has significant saline land within the development parcel. Changes in design of block layouts and major return of land for greenspace has occurred. This is due in part to the participation of the Troy Gully Working Group, which has representatives from Dubbo City Council, E.P.A., D.L.W.C., Developers and Urban Landcare Group; and from information derived from technical investigation and study. The Troy Gully Working

Group provides a forum for discussion of community issues and peer group review. Within this framework developers have voluntarily carried out the following on salt affected developments :-

* **Road network re-design** to allow roads to go perpendicular to the slope , and not across the slope so as to act as barriers for water movement.
* **Hazard demarcation within development**. Areas of high salinity hazard have been determined, and different construction methods are required to address this salinity. Other areas have been designated as low and moderate risk. In the high hazard areas, marine concrete and changes to damp course thickness have been recommended.
* **Return of severely affected areas to greenspace**. Areas that present too high a risk for development have been returned to council to be used as areas upon which trees can be planted for discharge control for community benefit.
* **Drainage reserves** have timber belts either side of waterway to increase the area of trees within the development.
* **Strip plantings across the slope** have been introduced to control local water movement. Plantings are at top, middle and bottom of development.
* **Increase in tree planting density** within the area as a recommendation of sale of block.
* **Contribution of funds for further investigation purposes** have been made by developers.

2.2 NEW DEVELOPMENT - INVESTIGATION TECHNIQUES

Currently, any new development requires a salinity investigation within the Troy Gully Catchment area. Techniques that enable the investigatory process are common to rural areas (peizometers, deep drilling, Electromagnetic Induction, geophysics, soil survey). The investigations are conducted by the DLWC on behalf of the developers. The techniques are outlined below:

* **Electromagnetic Induction mapping (EM31)** - In pre- development stage this technique is most useful. In some areas, the physical layout of the proposed development enables ground based EM mapping to be undertaken on a close grid scale. This information is also used to add to the pool of knowledge and understanding regarding the catchment.
* **Geological Mapping** - Structural mapping of geology is extremely important in understanding urban salinization processes. The level of accuracy of geological information is of prime importance in an urban environment. A desktop study is undertaken in all new developments within the Troy area.
* **Peizometer installation** - Peizometers in an urban environment commonly provide best data source, as well as provide good awareness focus points (eg. positive groundwater pressures in peizometers). Flow net models, 3D catchment models, recharge/discharge area definition, water level hydrographs and slug testing are all data sources in addition to simple water quality and water level determination made from installed networks of nested peizometers.
* **Geophysics** - Magnetic, seismic, resistivity , conductivity imaging, EM 39 and EM34 surveys have been conducted in urban area studies to understand geological structures and systems.
* **Deep bores** - In some towns a full understanding of the hydrogeological system can only be gained by a series of deep bores, and utilising information from groundwater data base.
* **Soil survey and infiltration testing** - Many new developments have soil surveys completed for urban capability and planning studies. catchments.
* **Saline Site Inspection & Description** - Mapping, cataloguing, and monitoring of sites & symptoms over time using standardised , structured methods which are incorporated into GIS data-base information. This data is used for assessment purposes.

3.DEVELOPMENT ISSUES

A number of issues arise from development and from the nature of the area, and impact on the future development of the Troy Gully area :-

* **High demand for land in Troy Gully catchment.** Land on the eastern side of Dubbo is preferred as a general rule. This puts increased pressure on both the developers and council to supply land. This high development pressure land has high recharge potential being higher in the catchment , and if not managed well, will put pressure on the downstream developments with increased salinity levels. Dubbo has a very high growth rate, and the larger block sizes in the Troy Area are a relatively short commodity.

* **Push for smaller subdivision of existing blocks.** A number of areas within the Troy Gully catchment have the perception that they should be allowed to develop their rural- residential blocks for closer subdivision to normal residential size.

* **Approved development plans have been around for many (10-20) years.** It is difficult to influence developments that were in some cases approved for development some 20 years ago. They are being released for sale with little consideration for the downstream salinity.

* **All new developments are in recharge areas** that have consequence for, and impacts on, downstream landholders. The Ballimore sediments, which comprise a significant proportion of the catchment, have geological sequences that have large local impacts on salinity occurrence.

* **Block yield** in developments is seen as a major factor, rather than environmental issues of integrated tree establishment and revegetation measures to control salinity.

4. CURRENT INFORMED DEVELOPMENT GUIDELINES & PRACTICE

Stakeholders are attempting to deal with some of the issues presented above, in trying to manage the urban salinity situation within Dubbo. This is a new and sometimes highly contentious issue. Existing guidelines and practice listed below indicate the framework within which we operate.

4.1 ZONING OF LAND

Industrial development has been limited to the bottom end of the catchment, where salinity is a minor issue. The industrial development does not entail any "wet" industry, so as to not impact adversely on groundwater.

High density residential areas comprise the mid section of the catchment. Saline discharge areas are ringed by development in this mid section of the catchment. Changes have been made in zoning land to greenspace that is severely salt affected; and salinity hazardous development parcels of land have instigated change. Greenspace for the community acts as an area where major tree planting can (and has) taken place, to lower local groundwater tables.

The upper reaches of the catchment are low density residential areas, with an urban buffer area on the higher, more vegetated areas of the catchment.

4.2 BUILDING CODES

The use of marine concrete and better damp coursing in the higher salinity hazard areas, has been a recommendation of the technical investigations being carried out. Limitations on area of lawn and provision of rainwater tanks has been discussed. The use of septic systems that do not add to water table are being investigated.

4.3 DEVELOPMENT CONSENT & DESIGN

As indicated in section 2.2 above, any new development requires salinity assessment, within the Troy Gully area . This usually involves an appraisal of lands with EM survey, geological investigation, soil survey and calibration, and peizometer installation. Developers have been very co-operative with making significant changes to design and layout, so as to accommodate activities targeted at salinity control.

4.4 STRATEGIC PLANNING OF CITY DEVELOPMENT

Salinity, both rural and urban has been taken into account as major factors with regard to Strategic Planning in the Dubbo City Council area. Significant discharge areas exist within the city limits and in surrounding rural areas. Salinity statements are incorporated into Local Environmental Planning guidelines.

Block size throughout the Troy Gully area has also been taken into account with the need for vegetation increase, and reduction in water use within the area. Larger blocks in Zone 1b (Rural Buffer) act as a buffer to rural holdings, but also act as areas where vegetation can have a larger impact. Owners of larger blocks are usually more inclined to plant trees, and these buffer blocks have a high proportion of existing tree cover.

The low density residential zone of the catchment is receiving most pressure to downsize blocks. Increased development in these areas may significantly increase recharge

A minimum percentage of tree cover for each zone of land has been suggested, particularly with any new development.

4.5 STORMWATER DESIGN

In November, 1998; a Stormwater Management Plan was developed for the Troy Gully Catchment area. As salinity is a major consideration within the catchment, stormwater design had to take into account this major environmental issue. The Troy Gully Urban Salinity Working Group became the primary focus for public consultation and problem solution within this framework. Groundwater quality is of significant consideration within the catchment due to high levels of salinity in groundwater that may enter the system. Removal of saline groundwater and/or the addition of water to groundwater can have adverse effects throughout the catchment. As runoff from stormwater systems act as mechanisms to flush salts through the system, they also have the potential to increase salinity if they recharge groundwater. The area also has significant surface water accumulation on saline discharge sites, that tend to concentrate salts as they evaporate.

As indicated above, the area has significant water quality, recharge, runoff and concentration of salts issues. A plan was developed for a number of different "reaches" of Troy Gully, to address these factors. Plans developed looked at a number of considerations:- low-flow drainage by giving existing gully areas some definition, depth of cut(60cm) limited so as to not access saline groundwater, making sure piped stormwater flows to creek and not just onto discharge area; and incorporating remediation tree plantings into design for ease of flood water flow.

5.PERCIEVED SOLUTIONS & ACTIONS

Solutions to urban salinity need a technical understanding of why, where and how the problem exists in the catchment and techniques to remedy the situation; as well as a planning framework to base technical investigatory information on. Technical understanding can be gained from :-

* **EM survey & hydrogeological study** over the remainder of the catchment to provide data to facilitate planning, as well as provide a basis for management actions.
* **Hazard mapping** process within the catchment, that incorporates information on salinity sites, geology, slope, soils, depth to groundwater, and additions to the water table including rainfall and irrigation(gardens). This will provide a salinity potential layer, which may assist in planning.

The planning framework may be assisted by the use of the following :-

* **149 certificates** issued with each property transaction, that discloses salinity as an issue. This will increase awareness within the local area.
* **Development Control Plan(DCP) for Salinity.** The detail that exists in a DCP may also highlight salinity as an issue, and bring it to the forefront of issues looked at in development.

6.MAJOR IMPEDIMENTS

There are two main issues that are thought to be major impediments to managing urban salinity in a residential area :-
* **EPA Clean Waters Act** severely limits the options for drainage of saline water within an urban environment. High value residential land may require engineering solutions to gain some relief for severely affected residential blocks.
* **Community awareness** is still a major factor. The majority of any urban centre do not realise that urban salinity is affecting their town. Community change in wateruse patterns on lawns would have a very significant affect on salinity status.

7.CONCLUSION

Urban Salinity is a recent emerging problem that has been un-recognised for many years. In the short time (3-4 years since communities have become aware, considerable headway has been carried out with practical solutions in a new and challenging working environment that urban salinity presents itself. Urban salinity is an emerging threat in the Central West (and NSW generally). The technical methods of understanding catchments are relatively straight forward. The challenge is to be innovative in providing awareness/extension information to all stakeholders, which include developers; and to provide a planning framework that is based on sound investigation that leads to informed. practical development decisions. The Troy Gully area has been an initial focus area due to it's obvious salinity problem , but many catchments in many towns and in many states all face similar problems. There is a need to share experiences, and pool ideas from across the nation.

THE WAGGA WAGGA URBAN SALINITY BORE FIELD

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ABSTRACT

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The City of Wagga Wagga is experiencing rising saline groundwater tables. It is causing damage to residential development and infrastructure. One of the proposed short to medium term solutions involves the construction of a series of deep bores designed to lower the groundwater level under the urban area, until more natural solutions can be implemented.

The paper would address issues relating to the investigations, development and operation of the Urban Salinity Bore Field, including an overview of urban salinity in Wagga Wagga, background and hydrogeology of bore field, aims and objectives, role in Urban Salinity Program, design of the bore field and discharge system, community education and consultation program, construction issues, monitoring program and results from the first month of operation.

OVERVIEW OR URBAN SALINITY IN WAGGA WAGGA

The city of Wagga Wagga is the regional centre of the Riverina with a population of 55,000. The Murrumbidgee River flows through the city centre with the topography ranging from river flats to hill side dwellings.

Salinity was noticed in the surrounding rural areas around 30 years ago with awareness in land degradation increasing during the 1970's. Salinity became a major interest to landcare groups in 1985.

Council was not interested in participating with major expenditure towards Salinity management in the rural area's. but this changed when salinity appeared within the heart of the City. In 1993 the Showground, located in the middle of the urban area, undertook redevelopment with the construction of a $2.5 million trotting track. The project was completed and the Show Society was trying to re-establish the grassed arena in the centre of the track. They sowed it 3 times and each time it died off. The Council and DLWC Staff where called in for advice. With the experience with rural salinity the DLWC quickly picked the symptoms. A backhoe was then used to dig 4 test holes, which filled with groundwater to within 200mm of the surface within 1 hour. The trotting track had been a cut and fill job with about 1.5m cut into the hill. This had intercepted the water table underlying the area. The salinity of the water was about one third of the level of seawater.

This lead to further assessment of the remainder of the city and it was quickly discovered that other evidence existed such as:
- Roads failing because of saturated pavements - previously blamed on springs.
- Houses with continuous damp under their floors - previously blamed on poor drainage of ventilation.
- Trees and lawns - previously blamed on insects.

Wagga Wagga City Council then undertook a program of installing piezometers across the city. An Urban Salinity Working Group was set-up comprising the Department of Land and Water Resources, Environmental protection Authority, Riverina Water, Charles Sturt University, an Engineering Consultant and Council. A three year action plan was developed which included investigation, monitoring, raising awareness, education and a program to implement change in a pilot area. A Hydrogeologist acted as the Urban Salinity Facilitator for the three years of the investigation phase and guided the working group through the first few stages.

BACKGROUND AND HYDROGEOLOGY OF THE BORE FIELD

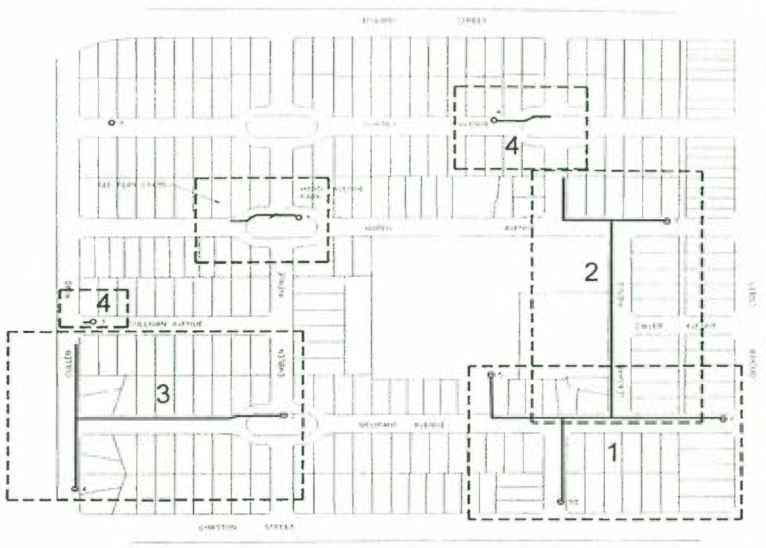
To determine the extent of the groundwater problem over the City a series of 50mm diameter uPVC piezometers were installed around the City. In the first year 25 were installed which has steadily increased to total over 100 presently. The regular monitoring of the piezometers and the production of annual reports indicated the depth to the ground water across the city and degree of seasonal fluctuations.

From the piezometer information a pilot area was selected for further study were the depth to the groundwater varied between 0-2 metres below the ground surface. A trial bore was constructed in 1995 within a small park in the pilot area to see if there was an interconnection between aquifers that would allow water to be lowered by this means. The test bore involved drilling the bore, soil logging, installation of piezometers, 7 day pump test and extensive monitoring.

Piezometers where installed in sets of three at depths of 15m, 30m and 60m around the bore to monitor the drawdown of the groundwater level and to gauge the performance of the bore. The bore was installed to a depth of 61 metres and pumping tests operated for 7 days at 0.5 L/s. The pump test indicated:
- that the unweathered slate has a transmissivity of 1.7 m2/day,
- a hydraulic connection exists between the deep and shallow groundwater zones,
- a safe production yield of 0.6 L/s,
- the shallow water levels can be lowered,
- vertical downwards drainage is induced to the pumped levels,
- water levels do not fully recover from pumping until the volume lost is replaced by local recharge events,
- the time taken to recharge the groundwater will assist in determining the schedule for operating the bore pumps.

The pump test results were used to develop a ground water bore model which incorporated a bore field to allow an estimation of the drawdown which could be achieved. It indicated that the most productive water bearing zone was found to be the unweathered fractured metasediments, which are predominantly comprised of phyllite and shale.

The trial urban salinity bore field was designed to radiate out from the trial bore. It was assumed that all bores would provide similar yields and draw downs as the trial bore which equated to bores being installed in a 1km square grid layout. he extent of the bore filed is presented in the diagram below:



The bores were drilled and pump test performed by the Contractor under supervision by the DLWC. The bores were drilled until satisfactory flows were detected and to a maximum depth of 120 metres. At one site, bore 9, no fractures were intercepted and insufficient flows were detected. It was converted into a deep piezometer. The flows and water quality at the other sites varied considerably with the presented below:

Bore	Depth	Standing Water Level		Bore Salinity		Pump Test	
No	m	Date	Level m	Date	EC (µS/cm)	7 Day (l/s)	Yield (l/s)
1	61	20/03/1996	0.83	20/03/1996	4400	0.6	0.19
2	73	09/07/1998	1.51	09/07/1998	4400	0.16	0.05
3	71	15/07/1998	0.32	15/07/1998	1410	1.6	0.53
4	73	17/07/1998	flowing	17/07/1998	1377	2.4	0.79
5	73	04/07/1998	3.01	04/07/1998	1620	2.2	0.72
6	72	06/07/1998	1.87	06/07/1998	2580	1.6	0.52
7	45	29/07/1998	1.2	29/07/1998	2200	1	0.33
8	42	23/07/1998	1.13	23/07/1998	2900	0.75	0.24
9		-	dry bore	-	dry bore	dry	-
10	73	03/07/1998	1.69	03/07/1998	2390	6	1.98
		Total				16.31	5.35

AIMS AND OBJECTIVES

It is anticipated that the trial bore field would need to operate for an extended period, probably at least five years, to allow a new water balance to be established and a variety of weather conditions to be experienced. The average EC of the bore water was estimated as being 2000 uS/cm with 864 kL pumped per day.

The trial bore field will provide data on the:

- Effectiveness of lowering shallow water tables by deep bore pumping.
- Long term changes to pumped groundwater EC.
- Pumping schedules and pumping rates required to effectively lower shallow water tables, and
- Costs involved in installing and operating a dewatering bore field within urban Wagga Wagga.

The aim is design a bore field that :

- dewaters the trial area satisfactorily to provide relief to the local area from high groundwater,
- provide a cost effective system that is efficient and has minimal impact on existing infrastructure,
- provides easy access for monitoring and educational visits,
- provides flexibility to explore,
- has a minimal aesthetic and noise pollution impact on the local residents and community.

A main objective throughout the design, drilling, construction and monitoring stages of the bores has been to keep the local community informed. The aims of this were to:

- raise awareness of the groundwater problem and efforts being taken to remedy the situation,
- ensure that the adjacent residents approved of the bore sites,
- provide the contact names and phone numbers for more information and complaints

ROLE IN THE URBAN SALINITY PROGRAM

The trial dewatering bore field is an integral part of the urban salinity program within the City of Wagga for purposes of providing immediate relief, education, funding and raising community awareness.

The aim of the dewatering bore field is to immediately pump ground water out of the aquifers and away from the local area, lowering the water table in the local area. This will enable the salt within the top soil layers to be leached or flushed deeper away from house foundations and tree roots by rainfall. The bore field will enable the vegetation to recover and provide relief to infrastructure.

The other programs, especially revegetation planting and removal of rubble pits, are not going to provide immediate relief, but are long term plans to lower the amount of water entering the groundwater. The aim is that in the long term the bore field will become redundant as the effectiveness of the other programs will be sufficient to prevent large quantities of water leaching into the soil in the catchment to cause the water table to rise.

The education value of the bore field is high as it clearly demonstrates the scale of the problem. The park within which the trial bore was installed has been developed into a salt tolerant park which is irrigated with the groundwater from the bore. The pump control board shows the volume of water pumped, the salinity level and the depth to the ground water at a piezometer located 20 meters from the bore. The figures combine with the sound of the water discharging into the tank to demonstrate the volume of water which is contained in the soil just below the surface.

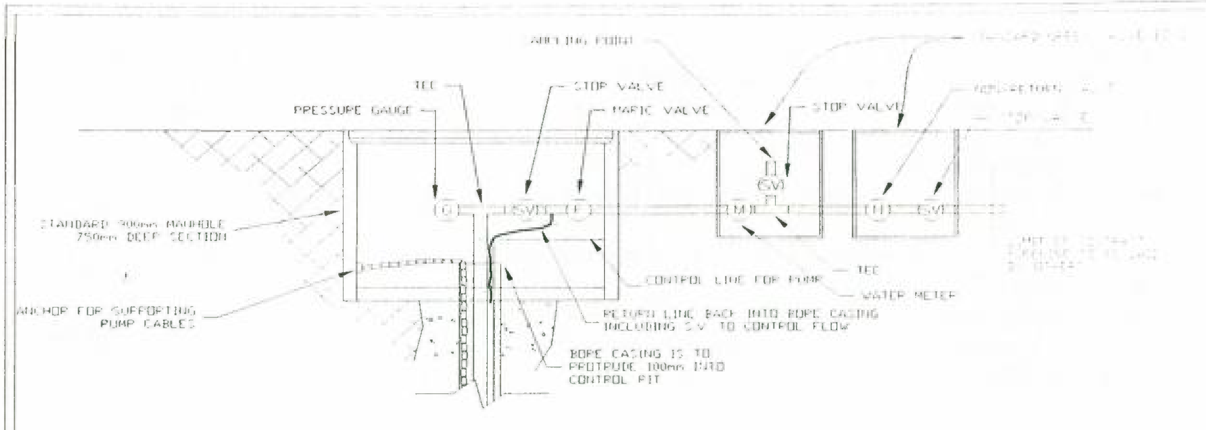
The City of Wagga Wagga is not alone in suffering from a high ground water table and urban salinity. The bore field will provide important information to determine if it is a viable option for further expansion in Wagga Wagga or establishment in other urban areas. The Bore Field was opened by the NSW State Agriculture Minister, Richard Amery, who also announced another $800,000 funding under the NHT in 1998/99.

The community awareness has been raised by the local media reporting of the performance of the bores, through school and community groups visiting the bore sites and pump stations and by seeing bores installed in the footpath out the front of their or friends houses.

DESIGN OF THE BOREFIELD AND DISCHARGE SYSTEM

The DLWC has provided a licence for the borefield to discharge into the Murrumbidgee River for a period of 5 years. With this is mind, the design of the borefield was kept cost efficient so that if the licence was revoked, the loss of capital would be minimised. The features of the borefield reflect this and include:

- the bores are installed below standard sewer manholes,
- each bore has a sampling point, flow meter, isolating valve and non-return valve,
- the flow rate from the bore is controlled by a maric flow valve and return line.
- the bore pipe fittings are standard uPVC fittings, water meters and valves.
- the control boxes are standard vandal proof electrical boxes painted heritage green,
- the upper and lower probes in the bores are manually adjustable,
- the bores
- discharge via PE pipelines into the existing stormwater system.



BORE DETAIL

N.T.S.

The maric flow valves were installed on the discharge to ensure that the design flowrate is not exceeded. A return line prior to the maric flow valve allows excess water to return into the bore. This ensures the pump operates at its duty point but the bore discharge does not lead to pump cavitation.

Utilising the stormwater system minimised the pipeline system that needed to be constructed and utilised an existing asset, which would be empty a majority of the time. Polyethylene pipelines were constructed from each bore to the nearest stormwater pipe.

A pump station was constructed downstream from the bore field to intercept the bore water from the stormwater pipe system and pump it into the Murrumbidgee River. This was necessary as the stormwater system discharges into a lagoon which was identified as being too sensitive to accept the slightly saline water. may have damaged the lagoon. The DLWC felt that the ecosystem within the lagoon may not be able to withstand an increase in salinity particularly during prolonged dry periods when the only inflow into the lagoon would be the borewater, which combined with evaporation and irrigation consumption. would increase the salinity.

The pump station is a 1200mm diameter RCP pipe, 2.4 metres deep, with a aluminium sump pump with a sacrificial anode. The water is directed into the well via a sump in the 1200mm drainage pipe. The pump station is to be connected to the sewer RADTEL SCADA system and maintained as per other sewer pump stations. The pump station is also to have a weather station so that it turns off during rain events and stays off for a suitable time. ^This will ensure that the pump is not pumping rainwater into the river. During this time the bore water will be diluted by the rain and will not impact on the lagoon water quality.

The volume and salinity of the water discharged from the pump station is to be monitored and recorded. It is also to be displayed on the pump station control box to inform visitors to the site. This measurements will quantify the quantity of salt pumped into the river from the borefield.

Further investigations are to be carried out to find uses for the water from the pump station, such as irrigation. or other disposal methods, such as evaporation basins, once the quality of the water is known.

COMMUNITY EDUCATION AND CONSULTATION PROGRAM

Education and community consultation have been important objectives of the trial dewatering bore field.

The local residents have been kept informed via letter drops and personal visits. Staff and contractors have also been instructed to answer all requests for information by residents during all stages of the project. The result has been that the local community has been very tolerant of the inconvenience created during the completion of the project and they have indicated appreciated the efforts being made to combat urban salinity. A drilling rig parked on the front footpath while the house is open for inspection prior to sale is inconvenient.

The consultation with the adjacent residents has continued after the bores were commissioned to ensure that issues such as noise, trench settlement, etc, were acceptable.

Efforts have also been made to make the borefield presentable as an education tool. The park around the trial bore has been transformed into a salt tolerant park. The control box displays the volume pumped, salinity level and depth to the ground water at an adjacent piezometer. Signs are to be erected describing the project and the park. The pump station site includes a display of discharge flows and salinity, with an open grate over the sump which allows visual assessment of the quantity and quality of the water.

The media have also been informed of major stages of the project. There has been TV, newspaper and radio coverage of the drilling and opening of the borefield. This has kept the wider community informed. The DOCENTS have also been regularly updated on the progress of the project.

CONSTRUCTION ISSUES

During the installation of the bores and the commissioning the following issues were detected which should have been included in the original design:

- the probes in the bores are difficult to adjust as they tend to tangle with the cables and pipe in the bore casing.
- the fittings above the bore should include flanges or universal couplings to enable the bore pump to be lifted easily.
- the design of the pump and discharge pipeline should have minimal safety factors to minimise the volume of water returned back into the bore and to maximise the run time of the pump.
- each bore should have an On button, which will start the pump before the water level reaches the top probe. This will enable a water sample to be taken, even if the pump is not running when visited.
- if the bore flow is much greater than expected, design for average flow, not maximum, to minimise pump starts and stops.
- the discharge pipe diameters should be uniform to minimise purchase and construction costs.

MONITORING PROGRAM

The monitoring program is to include all facets of the trial dewatering bore field which includes the bores, piezometers and pump station. This will ensure that rate of drawdown of the groundwater is controlled and that all components are working satisfactorily.

The pump station will be monitored automatically by the RADTEL system, so that it can be checked on a daily basis. It will also allow for alarms to be raised automatically. This could be for pump failure (therefor borewater bypassing straight into the lagoon), maximum daily salt discharge into the Murrumbidgee River exceeded, etc.

At each bore the settings for the probes are to be noted along with the pump hour counter, water meter and EC of the discharge water.

At the piezometers the depth to the groundwater is to be measured and recorded.

The frequency of the monitoring is to vary throughout the program. Prior to commissioning the piezometers were read monthly. Upon commissioning the bores and piezometers were monitored daily for the first fortnight. This was then increased to fortnightly and then monthly.

A quarterly report is to be prepared for the review by the DLWC on the operation of the trail dewatering bore field.

CONCLUSION

The bore field offers the opportunity to educate the community, study the groundwater hydrology and relieve a section of the City that is suffering from high saline water tables. The bore field has been developed through community assistance and consultation and will offer an important insight into the viability of such schemes, and the future costs of remediating communities if the early signs of Urban Salinity go unnoticed.

Buronga Salt Interception Scheme - Efficiency Review

Merrick N P[1], Middlemis H[2] Williams R M[3], Erny M[3] and Rozlapa K[2]

[1] National Centre for Groundwater Management, University of Technology, Sydney.
[2] Aquaterra Consulting Pty Ltd, South Perth.
[3] Department of Land and Water Conservation NSW

ABSTRACT

Buronga is on the NSW side of the Murray River directly opposite Mildura. As the area is the focus of regional groundwater outflow for the Murray Geological Basin, groundwater is often saltier than sea water. Although there is no record of natural salt inflow to the Murray River at Buronga, it is believed that construction of Lock 11 and a weir at Mildura in the 1930s created conditions suitable for salt inflow. The weir maintains a pool level of 34.4 m AHD compared to a downstream level of 30.8 m AHD. This has resulted in saline groundwater under the Buronga meander being discharged into the Murray River downstream of the structures, with an irrigation mound developed under the Buronga Irrigation Area having a secondary impact on saline inflows to the river.

In 1979 the Buronga Salt Interception Scheme was constructed to intercept the saline groundwater and to pump it via a pipeline for disposal in Mourquong Swamp which is remote from the river. In 1984, as part of a review of the Scheme, an analytical groundwater flow model was coupled with an optimisation model in order to improve interception effectiveness by nullifying the hydraulic gradients forcing saline water into the river. Recommendations were made for altering the pumping rates for each of the five existing bores, and for constructing a sixth bore This increased the installed capacity from 140 to 157 L/s. Subsequently, a numerical model was developed to better simulate the dynamics of the system and to incorporate factors which could not be included in the analytical model.

This study concludes that operation of the Buronga Salt Interception Scheme has reduced salt load significantly but not completely. It appears that, over the long term, about 40% of groundwater-borne salt destined for the river has been intercepted.

INTRODUCTION

Buronga, directly opposite Mildura on the Murray River, was declared a township in 1937 after settlement had commenced there in 1925 in consequence of a punt service (Figure 1). By 1930 a bridge had been constructed across the river (Moore, 1998). Around this time, in the 1930s, a series of weirs and locks was constructed along the length of the Murray River. Lock 11 is located at Mildura, where the upstream pool level is 34.4 m AHD compared with a downstream level of 30.8 m AHD. While stable river levels have improved the reliability of irrigation water sourced from the river, the steep drop in level has upset the natural groundwater gradients in an area of the Murray Geological Basin which is recognised as a groundwater discharge zone. In particular, upward vertical gradients from deep to shallow aquifers have been magnified in the vicinity of the river structures. Given that the natural groundwaters are saline, often saltier than sea water, the salt discharge to the Murray River has increased.

In an effort to intercept this saline groundwater discharge, the NSW Government installed a dewatering network of five bores (PS1 to PS5) to depths of about 30 metres (Figure 1). Effluent is piped to a large gypsum depression (Mourquong Swamp) 2-7 km to the north of PS5. The scheme, known as the Buronga Salt Interception Scheme, has been operating since February 1979. A companion groundwater interception scheme (Mildura-Merbein) has operated on the Victorian side of the river since 1980. This scheme comprises 17 bores which extend to 7 km west of Mildura Weir.

Two aquifers are recognised as interacting with the river: the Coonambidgal Formation (called the "shallow" aquifer in this study), silty sands interbedded with clay, deposited by modern and recent rivers; and the Parilla Sand (the "deep" aquifer), deposited during marine transgression and regression. Beneath the Parilla Sand are the Bookpurnong Beds of marine clay. The bores in the Buronga dewatering network are located on the floodplain of the Murray River. Since the commencement of the scheme, the maximum number of bores in the monitoring network has been 93 for the shallow aquifer and 31 for the deep aquifer (Figure 1).

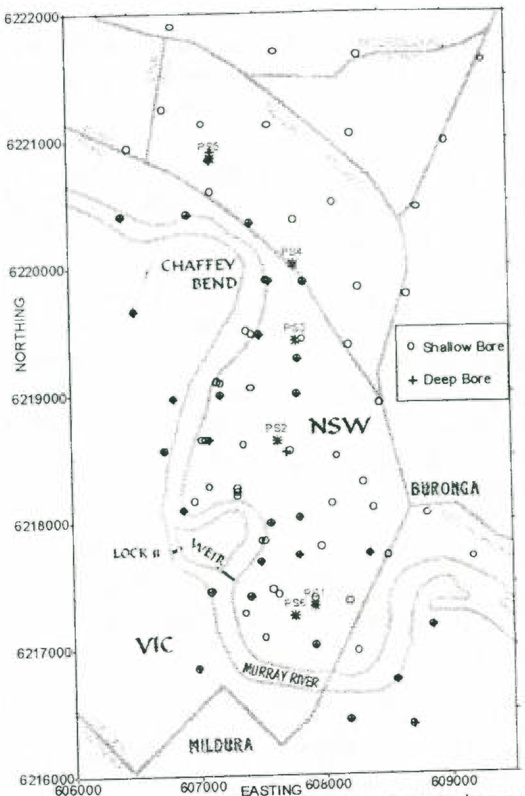


Figure 1. Location plan showing dewatering and monitoring bore networks

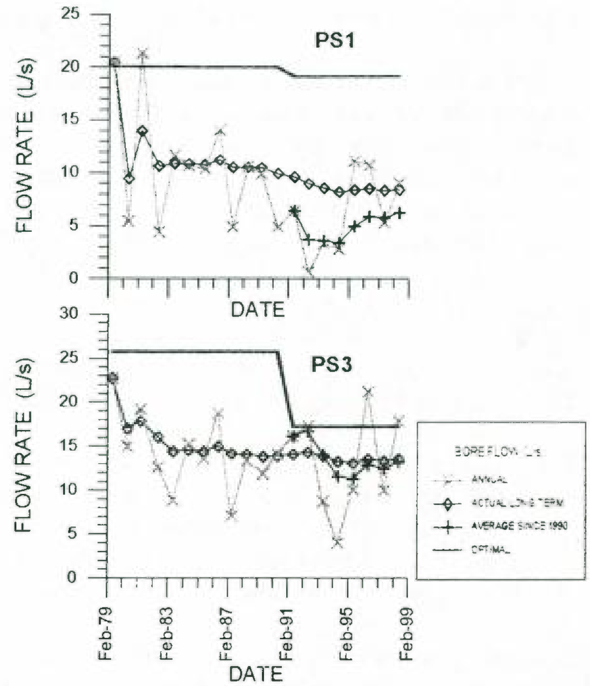


Figure 2. Pumping rates at bores PS1 and PS3

There have been several earlier studies which throw light on the effectiveness of the Buronga Salt Interception Scheme. A thorough review of the effectiveness of the scheme over the first few years was conducted by van der Lelij (1986). He reasoned that the effectiveness of the scheme was about 90% in salt inflow reduction when pumps are operated continuously. However, as pumps were active for only about 60% of the time, due to mechanical breakdowns and deliberate stoppages during high river flows, the overall effectiveness was lowered. Even with pumps active, he identified a reach of 400 metres below the weir where deep aquifer head was more than 1 m above river level. Merrick and Middlemis (1988, 1993) represented the aquifer system as a 2-layered analytical model, so that linear programming optimisation theory could be used to determine optimal pumping rates at each bore. The model was not able to account for rainfall and irrigation recharge, nor for any system dynamics. By constraining deep aquifer heads near the river to never exceed river levels, the approach was able to recommend optimal continuous pumping rates at each pump site. For the 5-bore network operating at that time, a total abstraction of 40% higher than had been achieved to date was needed to meet constraints, and hence to minimise salt upflow from the deep aquifer. Several possible sites for a sixth bore between PS1 and the weir and optimal rates for each bore were recommended. Middlemis (1990) incorporated spatial variability in aquifer properties and recharge stresses in a numerical model of the Buronga Interception Scheme. The finite element model, implemented with AQUIFEM-N software, included both NSW and Victorian interception schemes. As this software does not take density into account, it implicitly works with freshwater heads. Earlier, Ghassemi et al. (1986) had developed a finite-difference numerical model of the neighbouring Mildura-Merbein Interception Scheme.

DATA ASSESSMENT

Although river levels are controlled by weirs and locks, there are frequent flood events (six since 1990) which provide pulses of relatively fresh water to the aquifers, and suspend the opportunity for salt flows to the river. Current operating policy is to lower the weir dropboards as a flood approaches Mildura, and to remove the pumps from the bore site in order to minimise damage from flooding. As a result, the pumps could be out of action for several months at a time. In addition, mechanical breakdowns have also been responsible for the shutdown of pumps for extended periods. Pumping records indicate that the duty cycle for each pump is around 50-60%

Although the annual extraction from the borefield varies significantly from year to year, from about 35 L/s to 95 L/s, the long-term average production has been about 73 L/s which is very close to the optimal extraction from six bores recommended by Merrick and Middlemis (1988). Prior to 1990, however, the borefield consisted of five bores for which the corresponding optimal rate was about 97 L/s. During that time, the scheme output was about 30% below optimal. Since 1990, production has averaged about 68 L/s, which is about 10% below optimal.

Although total production is close to optimal, it does not follow that the borefield is operating in an optimal manner. Merrick and Middlemis (1988) recommended different optimal rates at each bore, and recommended preferred sites for the drilling of a sixth bore. In the event, their fourth choice for an extra bore was implemented, due to unfavourable aquifer conditions for the type of bore design proposed. The final site also was not ideal due to the fine and unstable nature of the sediments nearby. Figure 2 compares actual with optimal pumping rates at two pump sites. Long-term PS1 pumpage is about 55% below optimal, while short-term (since 1990) the deficit is about 70%. PS2 has operated about 10% below optimal over the long-term and about 20% below in the short-term. The operation of PS3 is about 20% below optimal. Merrick and Middlemis (1988) showed that PS4 should be shut down, but this bore continued to operate until 1996. Accordingly, its abstraction has been far from optimal. Similarly, it was recommended that PS5 be pumped at a nominal rate (about 6 L/s) but it has been operated at 2-3 times this rate. PS6 is operating at half the installed optimal rate of 15 L/s.

Pre-development measured groundwater levels show substantial agreement for the two aquifers. Groundwater flow is westerly towards the north-trending Murray River. Where the river turns west at Chaffey Bend, groundwater flow swings to the south. It is estimated that freshwater heads would be about 0.2 m to 0.8 m higher than observed heads in the deep aquifer (assuming 2:1 to 20:1 salinity contrast with the shallow aquifer). Shallow heads would normally differ little from freshwater heads (about 5 cm at 4:1 contrast).

A sampling round two weeks after commencement of pumping in February 1979 showed immediate impact in the deep aquifer, with subdued drawdown in the shallow aquifer. A clear change in vertical head differentials suggested stronger downward gradient over most of the area induced by pumping from the deep aquifer, with differentials of 2.5 m maximum. The region of potential upward flow, with maximum differential 1 m, had shrunk and become more localised. This general picture has persisted for the 20 years of the scheme's operation, punctuated by intermittent pumping schedules and occasional high river flows. Along much of the river bank downstream of the weir, the shallow water level has been commensurate with river stage (30.8 m). However, immediately downstream of the weir (for 2 km) there is a persistent zone of potential upward flow. The system is very dynamic and the potential for saline upflow to the river depends very much on the vigilance of operation of the dewatering scheme. While pumping must be maintained permanently to control hydraulic gradients, measured data suggest that the existing scheme cannot completely nullify salt discharge.

Recent electrical conductivity measurements at the six pump sites and at piezometers adjacent to the river range from 1,100 to 42,600 μS/cm in the shallow aquifer, and from 5,800 to 78,300 μS/cm in the deep aquifer. The river water conductivity at the Mildura Weir ranged from 180 to 310 μS/cm at this time. Half of the shallow readings are less than 4,000 μS/cm, while the other half are greater than 11,000 μS/cm. The high values occur between PS3 and the river. The highest deep values occur immediately downstream of the weir and the lock, and at Chaffey Bend. The lowest value occurs next to the river between PS2 and PS3. This is in the area where joint pumping of PS2 and PS3 encourages downward flow of groundwater, and probably induces fresh water recharge from the river. Pump site conductivities (deep aquifer) are generally in the range 50,000 to 60,000 μS/cm. Their behaviour with time exhibits extreme variations of up to 50% (30,000 μS/cm).

The salt load in any one year has varied from about 30,000 to 115,000 tonnes. The cumulative salt load has varied from 70,000 tonnes at PS6 to 355,000 tonnes at PS4. When PS6 came online in 1990, the production from PS1 was somewhat inhibited, due to interference between the two neighbouring bores. The total cumulative production from the borefield over 20 years has been about 42,000 ML of water and 1.5 million tonnes of salt.

NUMERICAL MODELLING

The numerical model developed by Middlemis (1990) has been applied in monthly stress periods from January 1990 to June 1998, to track the dynamic changes in aquifer heads and particularly the inferred fluxes along the reach of river downstream of Mildura Weir. The model was originally calibrated successfully for the time period from February 1979 to August 1983. Apart from simulating the response of the system to actual pumping, the model has been used to explore what might have happened if the scheme had never started, and also whether the scheme might have been more efficient if the optimal pumping rates recommended by Merrick and Middlemis

(1988) had been adopted. The latter scenario is really a verification of the analytical approach followed by Merrick and Middlemis (1988).

The conceptual model is a two-layered aquifer system corresponding to the Coonambidgal Formation and the Parilla Sand. Recharge occurs from rainfall, irrigation and the river. Discharge occurs to the river and from the aquifer by pumping. The boundary conditions are generally no-flow along flowlines, but prescribed flows are imposed on parts of the northern and southern boundaries. The river is modelled as a "first-type" boundary condition, that is fixed head in the shallow aquifer. The head is allowed to vary each month. This assumption is equivalent to full penetration of the river through the shallow aquifer. In the model, there is no direct connection between the river and the deep aquifer. Spatial variability is allowed for aquifer properties.

At the time of this study, only annual production figures for each pump site were on hand, so the model has applied an equivalent continuous rate at each site. The rates vary from year to year. This shortcoming in the dataset limits the validity of snapshot comparisons with observed head patterns, which are known to depend significantly on pumping fluctuations. Similarly, annual irrigation delivery volumes to the two irrigation areas and to private river pumpers formed the primary dataset. Temporal variability has been assumed to follow a seasonal pattern. Deep drainage of irrigation waters has been calibrated at 20-40%, with 50% at Mourquong where there is no drainage infrastructure. Rainfall recharge varies monthly at 1% of Mildura rainfall for agricultural land and 0.5% for urban land, subject to threshold antecedent conditions. Upstream and downstream river levels have been specified in the model as monthly averages. The model has the capacity to include Victorian stresses, but the original study showed that Victorian pumping had negligible impact on NSW groundwater levels.

Heads

As modelling assumes freshwater heads, one would expect simulated heads to be marginally higher (in general) than observed heads. Observed hydrographic data show sudden fluctuations in level but are not continuous enough to establish the detailed dynamics of the system. Simulated data replicate absolute levels quite well, and suggest heightened fluctuations in response to high river stages at times when no measurements were made. At times, the simulations show larger drawdowns than have been observed. This is due in part to reporting the response at the pump site, rather than a finite distance away.

For the end of the simulation period, the vertical head difference between the two aquifers (Figure 3) compares favourably with observed head differentials, but observed patterns are very sensitive to instantaneous pumping rates which are not yet incorporated in the model. The simulated head differentials are generally 0.5 m higher than observed, which is what one would expect when density is taken into account. There is very little difference in the zone of potential upward flux between recorded and optimal pumping, but the latter performs marginally better immediately downstream of the weir (Figure 4). Despite pumping for 20 years, there is still a persistent zone extending 2.2 km downstream of the weir where saline discharge to the river is expected to be occurring. If the scheme had not commenced, this zone would have been larger, would have extended along the entire reach of the river (past Chaffey Bend), and would have been more intense and therefore responsible for larger salt inflows. Numerical modelling shows that the optimal rates (due to some severe assumptions in the analytical model) do not achieve the target levels, but on the whole they would have performed better than actual operation.

Fluxes

Conceptually, the model provides salt discharge to the river by horizontal flow from the shallow aquifer and vertical flow from the deep aquifer (via the shallow aquifer). Modelling suggests that horizontal salt loads are negligible when compared with vertical loads (by two orders of magnitude).

An impression of the change in flux directions, and the variation in flux magnitudes along the river bank, is afforded by Figure 5 which looks at flux snapshots during high and normal river stages. When river stage is high, horizontal fluxes are away from the river and are strongest at the two westerly bends below the weir. Vertical fluxes are downwards and are strongest near PS3 and PS2. When river stage is normal, horizontal fluxes are uniformly weak and are towards the river except near the weir where flow is parallel to the bank. Vertical fluxes are upwards, except near PS3, and are strongest in the stretch of river between the weir and PS2. One would expect saline inflows to the river to be concentrated in this reach, in rough proportion to the flux magnitudes. Near PS3, due to pronounced pumping, the flux is almost always downwards. Fresher river water will recharge the deep aquifer and dilute the concentration of deep groundwater in this area, as observed in measured electrical conductivities.

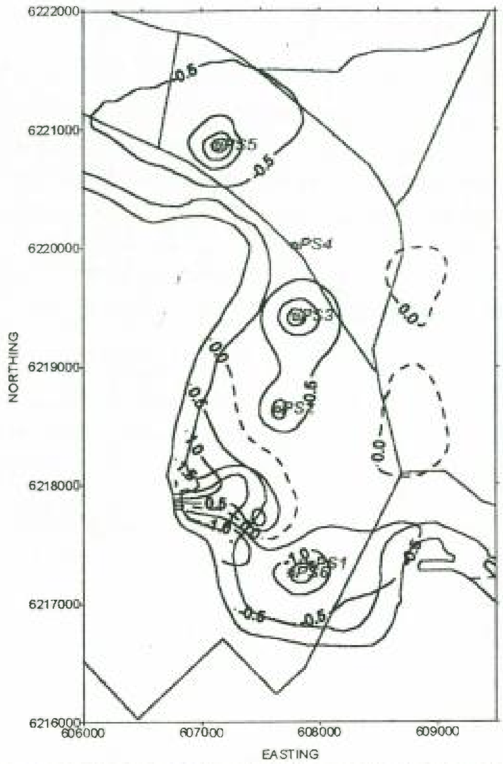


Figure 3. Predicted vertical head difference for Recorded Pumping, June 1998. (Positive means Deep Head > Shallow Level)

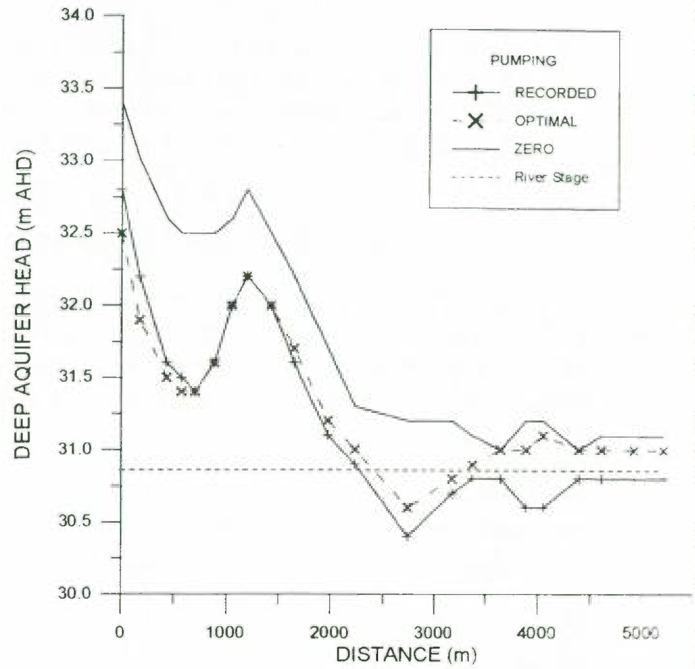


Figure 4. Simulated deep aquifer heads at nodes downstream of weir, June 1998 (m AHD)

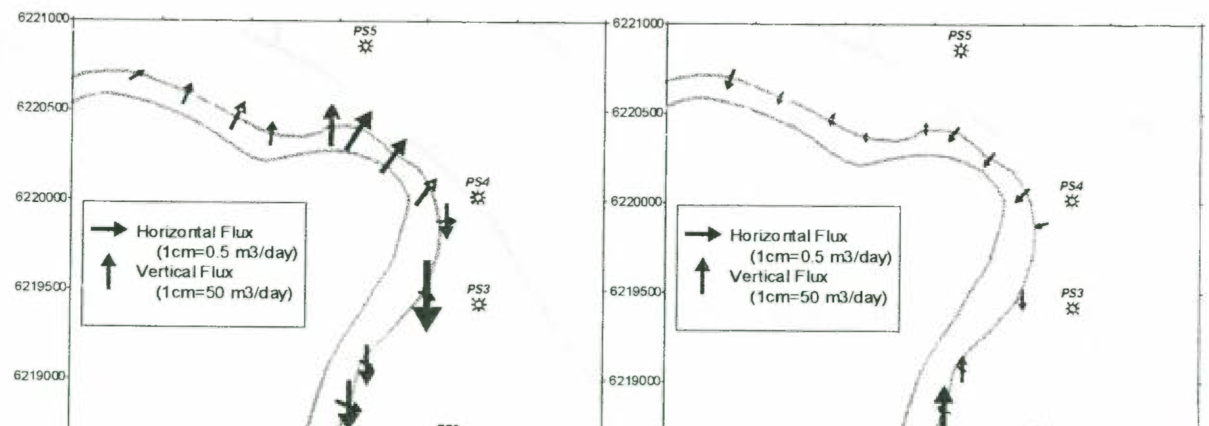


Figure 5. Flux patterns at (a) high river stage, and (b) normal river stage.

Salinities

Based on only the most recent electrical conductivity measurements, modelling suggests that the median salt discharge to the river from NSW (within 5 km downstream of the weir) would have reduced from about 710 tonnes/month under natural conditions to about 440 tonnes/month with recorded pumping. This is a reduction of 38%. The optimal pumping rates of Merrick and Middlemis (1988) would have achieved about 47% reduction. The absence of a full historical record of groundwater salinities will distort the estimates offered here, probably by underestimating salt load because the deep aquifer is being diluted by induced river recharge near PS3.

Over the period of operation of the scheme, modelling estimates that about 22,000 tonnes of salt destined for the river have been captured. To achieve this, the scheme had to extract 1.5 million tonnes of salt from groundwater, an efficiency of 1.5%. With optimal pumping rates, the mass captured would have been about 27,000 tonnes (efficiency 1.8%).

CONCLUSION

The actual operation of the Buronga Salt Interception Scheme has reduced salt load significantly but not completely. It appears that, over the long term, about 40% of groundwater-borne salt destined for the river has been intercepted. The total cumulative production from the borefield over 20 years has been about 42,000 ML of water and 1.5 million tonnes of salt.

The long-term average production of groundwater has been about 73 L/s which is very close to the optimal extraction from six bores recommended by Merrick and Middlemis (1988). However, there is no obvious correlation between actual pumping rate at an individual bore and the optimal rate suggested earlier. Although the optimal rates were derived on the basis of an analytical model with some limiting assumptions, numerical modelling suggests that the optimal rates would have performed better than recorded rates. The analytical optimal rates have not been able to nullify the salt upflow near the weir. It might not be possible to achieve this with the existing bore network.

The scheme is capable of performing better. This could be achieved by a higher duty cycle at pump sites, or more effectively by placement of an additional bore near a troublesome hot spot which has persisted below the weir. After high initial performance in the first year, the duty cycle dropped rapidly in subsequent years to settle at around 50-60% for each pump. For each tonne of salt prevented from discharging into the river, the scheme has had to pump about 60 tonnes of salt from the deep aquifer. This is a low efficiency factor.

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Woolpunda and Waikerie Salt Interception Schemes 10 years on (or thereabouts.....)

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The Woolpunda and Waikerie Salt Interception Schemes were designed and built in the late 80's and early 90's. The schemes, located adjacent the River Murray in South Australia, were designed to intercept natural (Woolpunda) and irrigation induced (Waikerie) salt loads. The salt loads derive from the inflow of saline (15,000 to 25,000mg/l) groundwater into the river.

The interception schemes, comprising 49 bores (Woolpunda) and 17 bores (Waikerie), pump into some 110 km of disposal main before discharging into Stockyard Plain Disposal Basin (a leaky evaporation basin).

Both schemes are working brilliantly. The bore flows are currently being optimised.

The paper will present the background and review the current performance of the schemes.

THE FULL TEXT OF THIS PAPER WILL BE AVAILABLE AT THE CONFERENCE

GROUNDWATER TRAINING

Groundwater Training

Trevor Pillar, Development Officer, Centre for Groundwater Studies (CGS)

Abstract:

Groundwater issues are at the heart of some of the most intractable problems in Australia's land and water resources. To begin reversing these problems two fundamental issues need to be addressed:

1. There is a low level of understanding about groundwater among industry, government and the community.
2. National land and water policies need to include the role of groundwater in the environment.

This paper will focus on the first issue above, discussing some of CGS' work in groundwater education and training. In addition it will highlight the importance of groundwater education if we are to achieve informed national land and water resource policies.

1. The Vision

The earth's environment is loaned to each generation to enjoy, and with it comes a solemn duty of care. The world's population is growing at a vast speed, creating enormous stress on food and water supplies, energy supplies and waste management. There are many bad news stories of famine, overlogging, forest fires and gross contamination. The good news is that we have the capacity to turn most of our humanly-spoilt physical environments into productive places of habitation. This vision could be put another way: *"To improve the long-term productive capacity, sustainable use, management and conservation of resources"*. (Mission Statement for Australian Land & Water Resources Research and Development Commission - LWRRDC).

2. The problems

There is a poor level of understanding of groundwater issues within Australia, within the water industry and particularly within the community. This is now becoming critical, given the national initiatives to combat salinity problems in most states, and the ever-increasing reliance on groundwater for domestic and irrigation supplies.
Centre for Groundwater Studies (CGS) identifies the need for communicating groundwater issues as follows:
"The increasing importance of environmental protection issues and a focus on restoration and remediation, has provided a need for much better understanding of human impacts on the environment at all levels. This is certainly true of the subsurface environment, where its vulnerability to pollution and over-exploitation has only been appreciated more recently."[1]

2.1. Undergraduates without hydrology education

Most students aiming for a career in land and water resources are not trained in hydrology at the undergraduate level. Geology students may complete an undergraduate degree with little or no understanding of hydrology and groundwater issues. Extension officers and field technicians may have an undergraduate degree in agriculture or environmental science but have no exposure to groundwater issues.

1. Barber, C. 1996. Interdata Handbook

2.2. Lack of in-house training

Due to major restructuring most government agencies no longer provide in-house training. It is critical that professional development education is available to a range of staff including engineers, agricultural extension officers, catchment management planners and environmental officers. The decline of in-house training in government agencies is having a flow down effect on consultants who traditionally come from the public sector. This reduces the level of groundwater understanding that private consultants can bring to land and water projects.

2.3. Public misconceptions

Misconceptions and ignorance about groundwater can lead to large investments of money, time and energy on projects that may not achieve their goals. Community awareness programs are needed for environment groups and the general public to provide basic understanding of groundwater's role in the environment.

2.4. Summary of the problem

The low level of groundwater education nationally among policy staff, planners and managers across a range of disciplines including environment science, catchment management, agriculture and mining, will inevitably lead to policies which are inadequate to cope with major issues like salinity, contamination and remediation.

"The lack of trained professionals in hydrology and hydrogeology particularly at graduate and post graduate levels has been recognised for some time, and new university courses offering such high level training are either in place now or being planned. Several universities in Australia now offer a range of courses and research opportunities. This high level training however neglects the increasing need for education of non-specialists who require a more general knowledge of subsurface and groundwater processes. Thus environmental regulators environmental engineers, conservationists, professional staff in EPA's and other agencies, and even the general public, have indicated a thirst for knowledge and understanding of processes beneath our feet [1]

3. The solutions

Wide-spread groundwater education is needed if we are to develop a national awareness of the role of groundwater in the environment. In Australia the following organisations work in the field of groundwater research and education:

- Centre for Groundwater Studies (CGS)
- UNSW Groundwater Centre
- UTS National Centre for Groundwater Management
- The Hydrogeology group formerly with the Australian Geological Survey Organisation, now with the Bureau of Rural Sciences (DAFF)
- CRC for Catchment Hydrology
- CRC for Fresh Water Ecology

The largest of these organisations, specialising in groundwater, is the CGS. The CGS is focussed on processes of groundwater recharge, discharge, contamination, remediation and management. The CGS is a non-profit, cooperative venture operating in SA and WA and involving 10 organisations:

- CSIRO Land and Water
- Flinders University of South Australia
- The University of Western Australia
- Department of Environment, Heritage and Aboriginal Affairs South Australia
- Primary Industries and Resources South Australia
- Water and Rivers Commission Western Australia
- Ministry for Planning Western Australia
- The Water Corporation Western Australia
- United Water International
- International Environmental Management Pty Ltd

"One of the aims of the CGS is to raise the profile of groundwater generally, through education and training courses in Australia." [1]

1. Barber, C. 1996. Interdata Handbook

The CGS undertakes a broad range of education initiatives including:

- **Postgraduate groundwater education.**
- **Groundwater training programs.**

3.1. Postgraduate groundwater education

The CGS actively encourages scholarship level postgraduate students to participate in its research activities, by attracting high achieving undergraduate students into postgraduate groundwater education. Currently there are 20 postgraduates studying with the Centre. Some of these students have undertaken summer scholarships or Honours degrees within the CGS, or are on secondment from employment in state government or consulting firms, working with the CGS in MSc or PhD programs. All have found employment in their field and some with multiple offers. The CGS' education program is a collaborative effort led by two of it's partners, Flinders University of South Australia and the University of Western Australia.

Scholarships, Honours prizes and stipend top-ups are awarded by the CGS to final year undergraduates as an incentive to take up groundwater studies at a postgraduate level. In addition the Centre supports students for periods of research abroad with outstanding researchers or institutions, and assists students with supplementary operating funds. Requests for bridging funds to help a student to complete their course after their scholarship runs out, or to allow time to write up further papers from their thesis could also be considered on their merits. CGS postgraduate scholars are given appropriate supervision, are treated as peers in each work environment, and are encouraged to contribute to seminar series and take an interest in other research within the Centre.

The CGS also contributes lecturers and Research Project supervision to the Joint Universities Masters Program in Hydrology and Water Resources. This is a venture of three South Australian Universities.

3.2. Groundwater training programs

The CGS' collaborative groundwater research and education programs provide the foundations for it's training courses attended by approximately 150 people per year. The Centre's training philosophy is to work with key individuals and organisations in industry and the community to develop relevant groundwater courses. In 1995 CGS undertoook a national survey of groundwater training needs. The survey revealed a need for continuing industry training in a range of groundwater issues and identified potential attendee numbers for course topics. CGS is now developing plans for groundwater awareness programs among community environment groups.

3.2.1. Aims of CGS training programs

The groundwater training programs have two key aims:

- To provide appropriate national and international groundwater training for a range of attendees.
- To break even financially.

3.2.2. Training categories

In transferring groundwater knowledge from research agencies into the broader community, the CGS has identified six "training" categories:

Conferences
International conferences provide the groundwater scientific community the opportunity to showcase their expertise and develop their industry networks. Currently the CGS runs one major international groundwater conference in Australia every second year.

Specialist Theme Workshops
These workshops are targeted at senior executives and public agency decision makers. In particular they will be concerned with Policy & Regulatory issues. Implications for Resource management. the impacts of these on Business Performance and Corporate Governance and other impacts that a changed water resource management environment will have on micro-economic development and the national business environment.

Specialist Courses
This segment provides for in-house industry training and will eventually lead to internet-based programs.

Professional & Academic Training
This category encompasses professional training at the academic level. Courses are required at both post graduate level and under-graduate level. CGS has affiliations with the University of Western Australia. Curtin University in Western Australia: Adelaide and Flinders University in South Australia. Monash University in Victoria. University of New South Wales and the University of Technology Sydney.

Training and the up-dating of skills are also required by Professional associations such as the Institute of Engineers. Typically such courses will be subject-focused e.g. Minesite Hydrology. Groundwater modeling. GIS. ASR. and there is now demand for including the "Groundwater Fundamentals" course. of the groundwater school. into existing University courses as an accredited module.

Technical & Occupational Training
This category is similar to the previous. however it is very much more focused at the practitioner level. A particular feature of this segment is the inclusion of Catchment planners and managers. Specifically it targets catchment coordinators. salinity coordinators and catchment extension officers The program requirements will be less technical in nature and focus more on 'issues management.'

Community Based Training
This is a broad group of people consisting of farmers. irrigators. graziers. small rural communities and industries associated with these communities. schools and organisations like Landcare. Waterwatch. salinity groups and the like. It is a very diverse market. but a significant one if there is to be an attitude change in the management of land and water resources.

3.2.3. Course attendees

People attending CGS training courses come from a diversity of natural resource backgrounds including: research: engineering: groundwater policy and management: environment consulting: mining: hydrogeology: water resources consulting: catchment management: agriculture: planning: landcare: environmental science: site contamination and remdiation: natural resources assessment: community environment groups: farming: education.

3.2.4. Course Leaders and Presenters

Lectures and tutorials for CGS courses are given by practising hydrogeologists and specialists who have hands-on experience with groundwater management. and by specialists in industry. universities and research agencies such as CSIRO.

3.2.5. Implementation

Led by recognised groundwater industry professionals, the Centre's training is focussed on industry needs. A mix of formal presentations, tutorials, informal discussion and field demonstrations ensures that the training offers many learning opportunities to participants.

3.2.6. Delivery

The Centre's training course program is offered as:
Generalist groundwater courses eg the annual Australian Groundwater School.

- specialist short courses presented in response to industry demand.
- in-house courses available to specific organisations and customised to suit clients' needs.

CGS has conducted over 30 short courses, since it commenced this training program in 1988. Course content and delivery is updated and improved using attendee evaluations.

3.2.7. Publicity

In 1995, CGS began developing it's data base of land and water people in Australia. This has now grown to 8000 people and is maintained by CGS to enable targetting of appropriate audiences and to assist with all aspects of running the CGS training course program.

3.2.8. Course examples

1. Groundwater fundamentals. This is the Australian Groundwater School, run by CGS under the guidance of National Groundwater Committee.
2. Managing groundwater issues. *(Groundwater for Managers)*
3. Groundwater aspects of planning decisions. *(Groundwater for Planners)*
4. Assessment of point & diffuse source contamination.
5. Investigation and remediation of soil and groundwater contaminated by organics.
6. Groundwater quality basics.
7. Groundwater flow modelling - Basic and Advanced.
8. Solute transport modelling in the unsaturated zone.
9. Groundwater protection and wastewater reuse
10. Artificial recharge of groundwater.
11. Water use of various crops and landcovers.
12. Riparian zone and wetland processes.
13. Beneficial use and vulnerability mapping with GIS
14. Salinity management
15. Groundwater microbiology
16. Groundwater database systems
17. Evapotranspiration measurements (groundwater discharge).
18. Isotopes and hydrochemistry for groundwater evaluation.
19. Use of natural tracers for recharge estimation and sourcing groundwater.
20. Minesite hydrology.
21. Bank filtration systems for managing blue-green algae
22. Tropical hydrology
23. Groundwater and coastal margins

4. CONCLUSION

Professionally-run groundwater education and training programs, that reach a wide spectrum of people, is critical if we are to achieve:

- Intelligent in-depth national debate on groundwater issues.
- Properly considered and well articulated policy decisions.
- Wide spread support for continuing groundwater research and education.
- A healthy national land and water resource to hand on to future generations.

Attached is the proposed CGS Training program for 2000.

Proposed CGS courses for year 2000

Title	When	Where	Course Leader
1. **Community Training** - program of 12 sessions in various locations throughout SA	March - June	SA	
2. **National Groundwater Modelling Workshop**	July	Sydney	Craig Simmons & Cliff Voss
3. **Joint, In-House – Fundamentals of Groundwater**	August	Perth	Water Corp / WRC etc
4. **Minesite Hydrology**	Sept	Kalgoorlie	Don Armstrong
5. **Surface water – Groundwater Interaction** In conjunction with Hydro 2000 (Hydrology & Water Resources Symposium)	Mon 20 Nov	Perth	Lloyd Townley
19th Australian Groundwater School	Dec.	Melbourne	Charles Lawrence
6. **Fundamentals**			
7. **Groundwater Contamination and Remediation**			Greg Davis

WATERTABLE IMPACTS

Trial Results from a New Electro-Kinetic Geophysical Technique for Remote Measurement of Sub-Surface Hydraulic Conductivity

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Abstract

Field trials of a new geophysical technique called an Electro-Kinetic Sounding (EKS) have been conducted in a range of geological settings, including upper Murray Basin sediments from sites near Kerang, Victoria, Australia. Successive regressions of the Pliocene coastline (<5Ma) have formed numerous beach strandline and dune sands, which are now buried from 0 – 35 m below the surface. In between the beach strandline and dune sands are finer and less permeable sediments. The different sedimentary facies incorporated in Loxton-Parilla Sands form important near-surface aquifers for the regional hydrological regime, and are host to over 20 titanium and zircon mineral sand deposits, some of which are currently under development. Successful development, operation and management of environmental impacts for irrigation, salinity control works and mines require a detailed understanding of the sub-surface hydrology.

The EKS geophysical technique has successfully detected and defined the relative depth and permeability of the sand systems from the Loxton-Parilla Sands where previously only drilling had been successful. Some of these sand units are recognisable as relict beach strandline sands, formed by regression of successive Pliocene strandlines. An example of a similar structure from the Wemen mineral sand deposit, also in the Murray Basin, is shown for comparison (Figure 4, after Mason 1999). The Wemen heavy mineral deposit is well defined by close spaced drilling and detailed mineral analysis. One difference between the two sites is the size. The Wemen beach strandline system is approximately 200m across, compared with 500m for the larger of the 3 beach strandline systems detected in the 1200m EKS cross section at Churchs Road, near Pyramid Hill.

Within the interpreted large beach strandline system there are zones of particularly high permeability. For a beach strandline geological setting, high permeability is likely to correspond with coarse sands and high concentrations of heavy minerals. If this inference proves to be correct, EKS surveys will be very useful for heavy mineral exploration. Heavy mineral sand mines also require hydrological models to be constructed to predict the environmental impact of sand mining. EKS data could be used for both purposes, significantly lowering the total cost and environmental impact of exploration, operation and environmental assessment to the operator. Similar strandline deposits are found throughout the Murray Basin.

Calibration of individual EKS data traces by comparison with geological drill logs and gamma logs, enables selection of a uniform seismic velocity model for processing all EKS data at any location. The survey technique involves collection of a sequence of EKS measurements along a profile line or in a grid pattern. Permeability values inferred from the processed signal may be interpolated to a 3D grid or mesh. These 3D block models are used to aid in the visualisation of the hydro-geology, or as direct input into finite difference or finite element hydrological flow models.

Application of the EKS in this geologic environment has significant cost and interpretive advantages compared with the use of drilling alone for hydro-geologic investigations.

Introduction

A geophysical investigation was undertaken to assess the utility of the Electro-Kinetic Sounding (EKS) technique for mapping shallow aquifer systems in seven areas near Kerang in Northern Victoria. This paper presents selected results of EKS data collected in April 1999, with a comparison between EKS data, gamma logs, and bore geological records provided by Goulburn-Murray Water. Full results are available in Hankin and Waring (1999).

EKS Principal and Equipment

The EKS equipment used in this survey is the commercially available GroundFlow-500 model. GF-500 is based on a relationship first described by Chandler (1981) where a sharp pressure pulse on a permeable rock core sample induces an electrokinetic signal or "streaming potential". The rise time of the electrical signal is inversely proportional to the rock core sample's permeability.

Groundwater contains dissolved ions, which are attracted to the walls of cracks and pores in the rock. When pore water moves in response to a seismic pulse, negative ions travel further and more easily than positive ions, so that an electric potential develops which is proportional to the applied pressure (Millar, 1997). This potential is typically 30-60 millivolts per atmosphere in a wide range of rock types. The amplitude of this effect is controlled by changes in the pore fluid resistivity. Also, the rise time of electrical response is a direct measure of how quickly flow is generated by a given pressure, or the permeability of the rock.

In practice, a 4 electrode array of 2 independent channels, records small electrical signals which are induced by the passage of a seismic pulse through water saturated porous permeable rock and unconsolidated sediments. Proprietary algorithms built into the EKS hardware and software interpret the electrical signal from each channel into a hydraulic conductivity vs depth trace (dependent on selected seismic velocities).

EKS Data Processing

The data was inspected on a site by site basis, and the best quality data was selected from each site for further processing. This requires inspection of each sounding to assess quality of data due to variability in seismic source, electrical properties at each electrode, possible recorder triggering delay and electromagnetic noise. Some soundings may be corrected or omitted at this stage, to prevent introducing noisy or unreliable data into the visualisation. Where suitable, an averaging process of multiple selected EKS data traces was applied to obtain improved signal to noise ratio.

Calibration of individual EKS data traces by comparison with geological drill logs and gamma logs, enables selection of a uniform seismic velocity model for processing all EKS data at any location. The data is then reprocessed using the new seismic velocities. A uniform depth to water table of 1.5 metres was selected for all areas and a seismic velocity of 0.5 m/ms used for the unsaturated zone. Depth corrections due to topographic effects were not necessary due to the flat topography of the survey areas. The selected data was then truncated to a depth of 100 metres below ground, and de-sampled by choosing every fifth record of permeability prior to griding.

Data Visualisation

The linear permeability scale processed data from each site was interpolated to produce a cross section of EKS calculated permeability. Due to the high amplitude response of the EKS signal in the unsaturated zone, the natural logarithm of EKS permeability is presented in the cross sections. The logarithmic scale representation helps to enhance visibility of lower amplitude structures below the water table. All vertical coordinates are presented as metres below ground. Note that varying degrees of vertical exaggeration are present in different cross sections and that varying colour scales have been used. All horizontal coordinates are presented on an arbitrary local grid coordinate system in metres, relative to the position of hydrological bores on site. Key bore positions are labelled on the cross sections. The position of EKS soundings is shown as a thin black line.

Figure 1 shows a comparison between a gamma log and drill log supplied by Goulburn-Murray Water and the EKS permeability. An EKS high is visible at approximately the same depth as a low in the gamma log. High relative EKS permeability values are present with significant low values in the gamma log, but without a corresponding sand unit in the drill log.

Direct correspondence between these data is not expected because they measure different sub-surface parameters. The particular lithological classification of the drill logs makes it difficult to represent continuously varying sedimentary units, and tends to favour representation as discrete narrow bands at subjective depth intervals. All sedimentary units logged as "Sand" are not identical and are defined on the subjective assessment of the person logging the drill hole. Gamma logs have particularly good depth resolution and are usually indirectly proportional to the clay fraction concentration. In this geological setting gamma logs may be ambiguous because some coarse sands with a low clay content contain elevated concentrations of radioactive minerals such as monazite in the heavy mineral fraction. EKS permeability data has the tendency to smear sharp geological boundaries and is unable to clearly resolve discrete narrow geological units, particularly if there is little contrast in permeability. EKS is a remote surface geophysical technique. Comparison with the down-hole equivalent of EKS, Electro-Kinetic Logging (EKL) is likely to greatly increase the depth and permeability resolution.

Geological Setting

Successive regressions of the Pliocene coastline (<5Ma) have formed numerous beach strands, which are now buried from 0 – 35 m below the surface forming the Loxton-Parilla Sands. In between the beach strandline sands are finer and less permeable sediments. Loxton-Parilla Sands form important near-surface aquifers for the regional hydrological regime.

Site Interpretation

Churchs Road

A seismic velocity of 2.0 m/ms was applied in this area. The survey area is a transect across a proposed evaporation basin for the ground-water interception scheme. A comparison diagram between individual drill logs and EKS permeability data was not constructed, as little detail is recorded on the drilling logs for this area. A cross section including zones of high EKS log permeability are displayed in cross section **Figure 2.** A more detailed view of the top 30 metres of the same cross section shown in **Figure 3**, reveals 3 diagonally aligned high permeability zones, bounded by relatively low permeability zones. Note the high level of vertical exaggeration of the section (30 metres vertical by 1200 metres horizontal), and the nugget-like appearance is a product of the interpolation process.

These diagonal features are recognisable as relict beach dunes (steeper angle) for the upper 15m portion of the cross section and beach strandlines at 15 – 30m depth, formed by regression of successive Pliocene shorelines. The sub-horizontal division between interpreted dunes and strandlines may represent the erosion surface between the lower Loxton and upper Parilla Sands. An example of a similar structure from the Wemen mineral sand deposit, also in the Murray Basin, is shown for comparison in **Figure 4** (after Mason 1999). The Wemen heavy mineral deposit is well defined by close spaced drilling and detailed mineral analysis. Contours shown in Figure 4 are heavy mineral grade outlines. One difference between the two sites is the size. The Wemen beach strandline system is approximately 200m across, compared with 500m for the larger of the 3 beach strandline systems detected in the 1200m EKS cross section at Churchs Road, near Pyramid Hill.

Within the interpreted large beach strandline system there are zones of particularly high permeability. For a beach strandline geological setting, high permeability is likely to correspond with coarse sands and high concentrations of heavy minerals. If this interpretation proves to be correct, EKS surveys will be very useful for heavy mineral exploration. Heavy mineral sand mines also require hydrological models to be constructed to predict the environmental impact of sand mining. EKS data could be used for both purposes, significantly lowering the total cost of exploration and environmental assessment to the operator. Similar strandline beach deposits are found throughout the Murray Basin.

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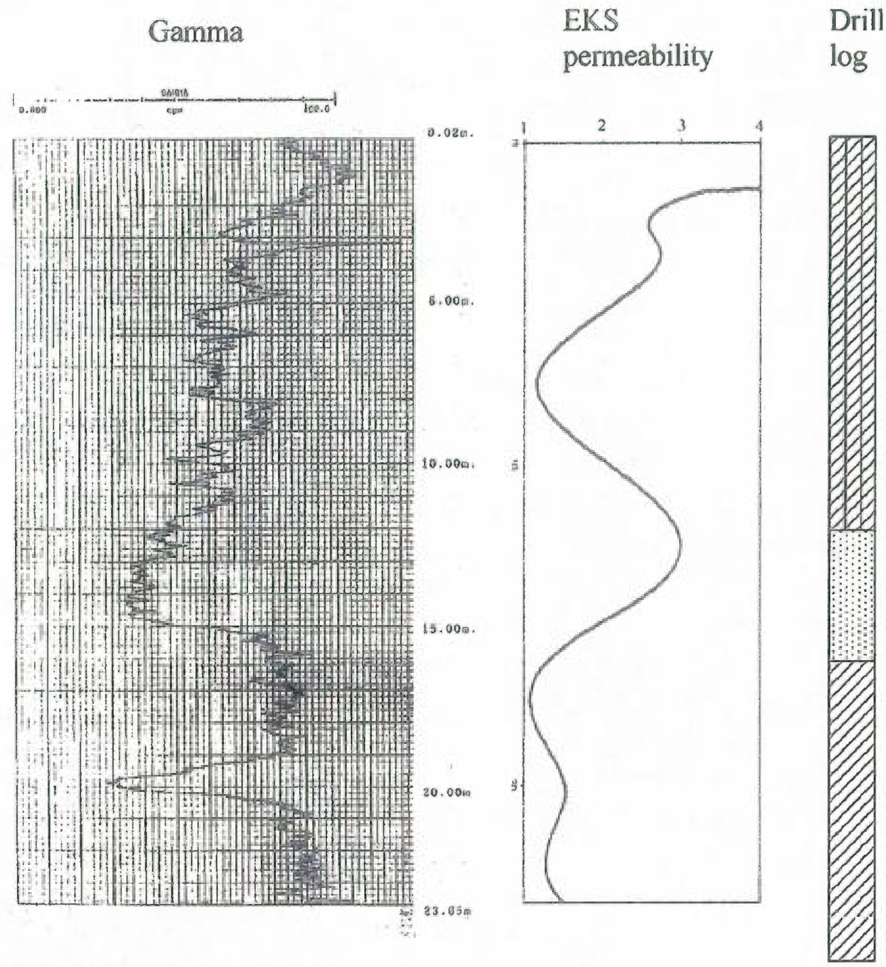

Figure 1 Kow Swamp - Bore KS1
Comparison of gamma log, drill log and EKS permeability (meters / day)

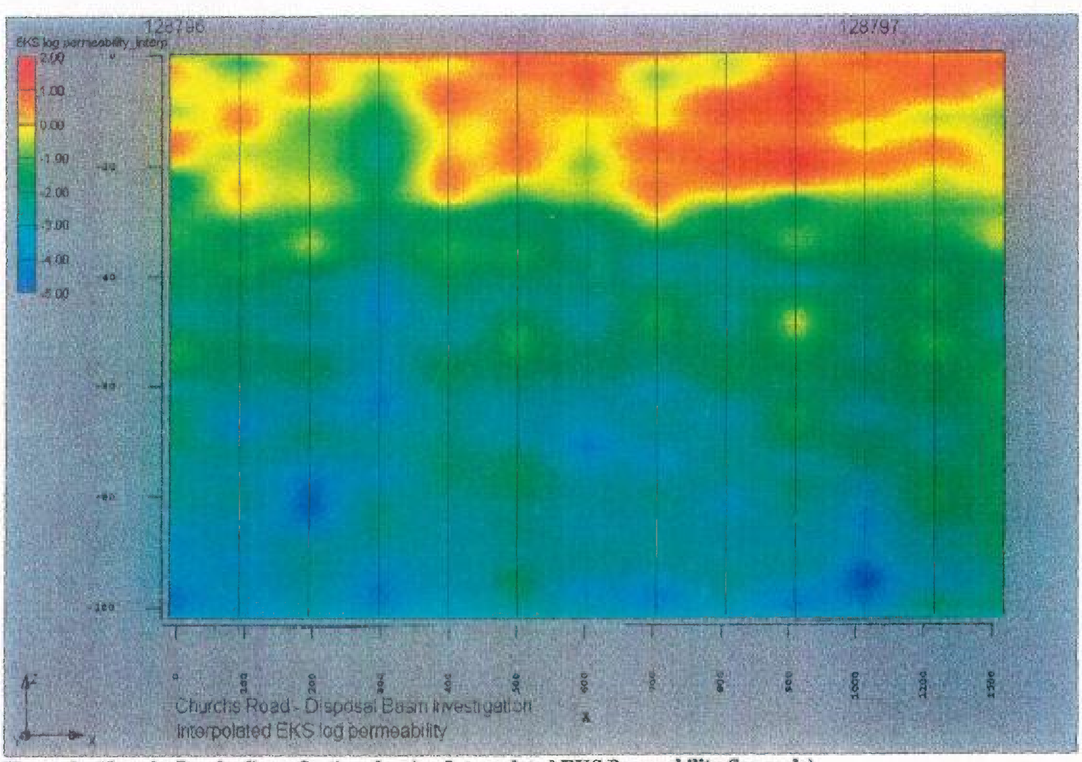

Figure 2. Churchs Road - Cross Section showing Interpolated EKS Permeability (log scale)

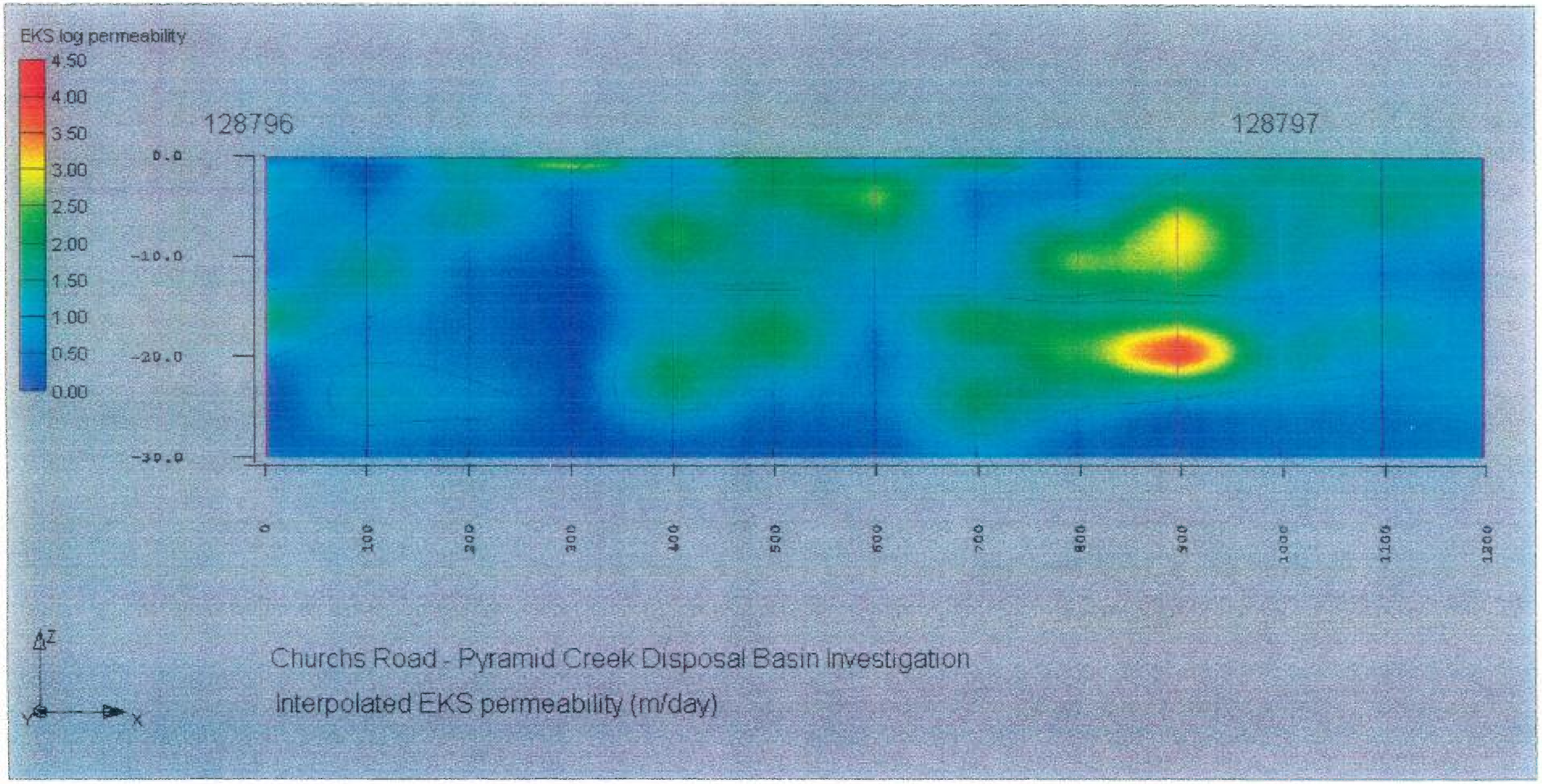

Figure 3. Churchs Road top 30 metres - Cross Section showing Interpolated EKS Permeability (log scale)
Note: Vertical axis is 10 times the horizontal axis. Apparent angle of diagonal structures is steeper than reality.
A different colour scale is used to highlight geological structure compared to Figure 2
Diagonal features bound 3 high permeability zones, interpreted as sets of beach dunes (steeper angle) for the upper <15m portion of the cross section and beach strandlines, at 15 – 30m depth. The sub-horizontal division between interpreted dunes and strandlines may represent the division between the lower Loxton and upper Parilla Sands.
Within the larger beach strandline system there is a very high permeability zone which may correspond with a concentration of heavy minerals.

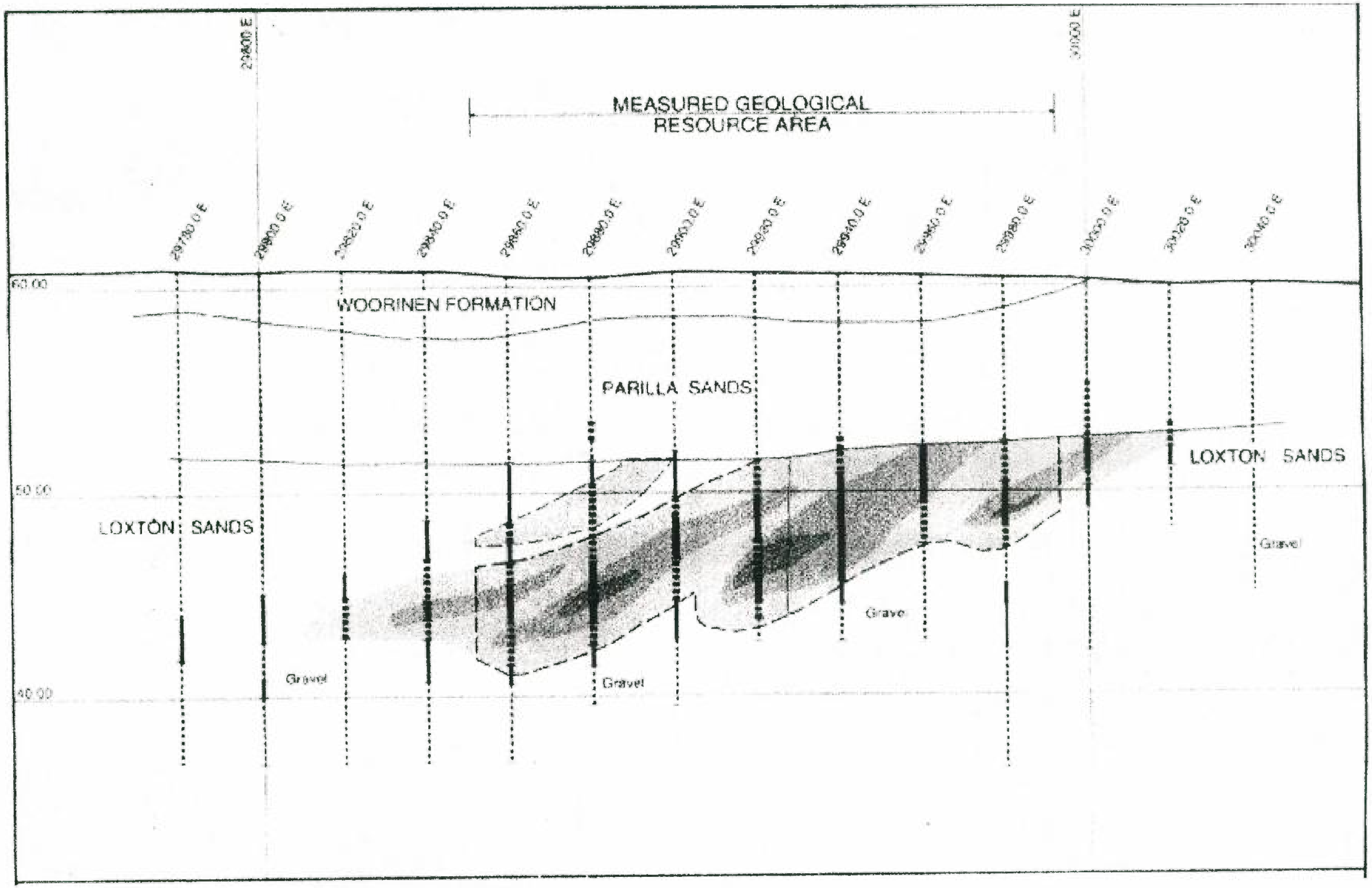

Figure 4. A cross section through the Wemen heavy mineral deposit showing diagonal structure from beach strandlines (After Mason 1999). Grey shading denotes mineral grade.

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Vertosols do "leak"! - Water and Solute movement below irrigated cotton

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Introduction

Off-site degradation of land and water resources from irrigation has been highlighted as an issue of national significance. Much of the research effort to date has focussed on the movement of water, sediment and chemicals from irrigation farms in surface run-off. However, the movement of water and chemical below the root zone of irrigated crops can have significant environmental consequences, particularly as a potential polluter of groundwater resources.

Cotton production is one of the major irrigation industries in the northern Murray-Darling Basin. It is important to gain an insight into water and chemical fluxes beneath irrigated cotton as even small changes in the rate of water movement can lead to serious impacts over the longer time period. There is currently little evidence of groundwater contamination or shallow water table problems within cotton-growing districts; however, the rapid expansion of the cotton industry and the time-frame for groundwater responses to irrigation is such that preventative management systems need to be identified in the near future.

The movement of excess chemicals below the root zone, especially nitrates (NO_3-N), provides a potential source of groundwater pollution. Although there are many sources of nitrogen (both natural and anthropogenic) that could potentially lead to the pollution of the groundwater with nitrates, the anthropogenic sources (septic tanks, fertilisers and agricultural processes) are amongst the largest sources. Nitrate is a problem as a contaminant of drinking water due to its harmful biological effects (Keating et al., 1996).

The project "Best management practices to minimise below root zone impacts of irrigated cotton" commenced in January 1996, with the support of Murray-Darling Basin Commission NRMS research funds. This paper summarises the results of the measurements of leaching fraction and chemical flux at two lysimeter sites selected to represent traditional and the "best bet" application techniques to minimise drainage below irrigated cotton.

Methods

To quantify the difference between furrow and sub-surface drip irrigation two large (3m x 1.5m x 2m) undisturbed suction lysimeters were constructed on irrigated grey Vertosols. These lysimeters allowed an assessment of the flow of nutrients and salts below the root zone of irrigated cotton. Two sites were chosen on neighbouring cotton growing properties near Macalister, 50km northwest of Dalby on the Darling Downs. The two sites were chosen to represent two irrigation systems, conventional furrow and sub-surface T-tape irrigation. A schematic diagram of the lysimeter and its components is provided in Figure 1.

The lysimeters enable capture of all water and chemical fluxes below a constrained block of undisturbed soil. Construction of the lysimeters took place between August and September 1995. The 1995-96 irrigation season was used to calibrate and test equipment and instrumentation. Work involved trenching to 2.5m around two blocks of soil and then filling the trenches with concrete to 40cm below the surface of the soil to allow for normal tillage practice over the lysimeter. One of the sides of this arrangement was then dug out to form the cellar and allow access the bottom of the undisturbed, lysimeter side. The soil beneath the walls was then dug out and stainless steel trays inserted below the soil column. These trays were then filled with silica flour, a substance that will allow water but not air to flow when under suction. Ceramic candles were inserted in the suction lines in the trays to collect the water from the silica flour and pipe it back to the collection tank.

Stainless steel tanks with a 250-litre capacity are located in the base of the cellar beside the lysimeter. A small tipping-bucket in the lid of the tank recorded flow rates and volume of leachate entering the tank. All materials used in the construction of the lysimeter were chosen to minimise interactions with pesticides and nutrients in the drainage water. The whole system of collection lines and trays were kept under suction by evacuating the tank to 30 centibars. A data logger that also recorded tip rates from the tank tipping bucket, rainfall, and soil moisture, controlled the vacuum system.

Water samples were taken from each tank pump out and submitted for nutrient, chloride and pesticides analysis. Samples of irrigation water were also taken during the season, bore water in the first season and from the ring tank

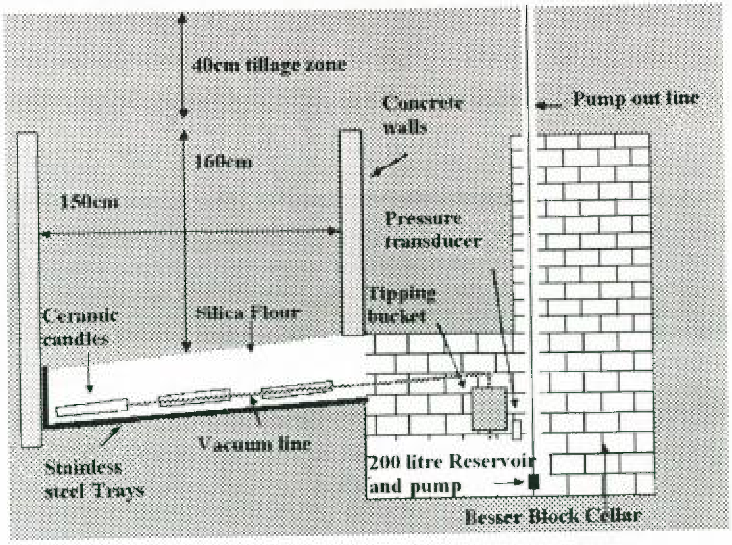

Figure 1. Schematic diagram of a lysimeter as installed under furrow and sub-surface drip irrigation

(surface water) in the second and third seasons. Annual soil sampling was undertaken for each of the lysimeter sites. Soil cores were taken down to 150cm and split up into 10cm increments for chemical analysis

Results

Seasonal data presented for the irrigation seasons represents the 12-month period June to June (except 1998-99 furrow due to equipment failer). Inputs of rainfall and irrigation were measured using rain gauges and water meter readings. Drainage is given as the total flux of water pumped out of the collection tank during the season. Fluxes of Nitrate (NO_3) and Chloride (Cl) were calculated by multiplying the average concentration for the season by the total drainage flux (Data provided in Table 1).

Water Flux

Total volumes of drainage water were collected by the lysimeters over three irrigation seasons. Data for the sub-surface drip plot is more variable than that for the conventional furrow irrigation plot. Leaching fraction (LF) varied from 12% to 50% under sub-surface drip but was relatively constant for the furrow irrigation site.

Table 1. Water, Nitrate and Chloride fluxes beneath furrow and sub-surface drip irrigation for three irrigation seasons

Sub-surface Drip	Irrigation (mm)	Rainfall (mm)	Total (mm)	Drainage (mm)	LF	NO_3 flux (kg/ha/yr)	Cl flux (kg/ha/yr)
96-97	150	478	628	305	0.49	691	4636
97-98	142	667	809	95	0.12	560	975
*98-99	0	739	739	857	1.16	1517	1567

furrow	Irrigation (mm)	Rainfall (mm)	Total (mm)	Drainage (mm)	LF	NO_3 flux (kg/ha/yr)	Cl flux (kg/ha/yr)
96-97	327	478	805	182	0.23	212	3509
97-98	343	667	1010	162	0.16	174	2865
#98-99	487	579	916	152	0.17	294	3441

* LF 1 due to water ponding over lysimeter; # Data through until Jan 1999 only.

Irrigation in the furrow-irrigated paddock was approximately double that for the drip irrigated site for the first two seasons (1996-97, 1997-98). The annual leaching fraction for the furrow irrigated plot remains similar at approximately 0.2 reflecting similar input volumes and drainage for the three irrigation seasons. Furrow irrigations in response to plant water stress in season three (1998-99) were nearly always followed by significant rainfall, leading to significant leaching events (Figure 2).

Differences in irrigation between sites in the third season were due to management practices and irrigation scheduling. Sub-surface drip is often promoted with the suggestion that no paddock levelling is required, however during winter of 1998 and 1998-99 summer rainfall season the sub-surface drip paddock was too wet to traverse and displayed significant scouring and ponding of water. It was this scouring that led to the positive leaching fraction

(LF) in the sub-surface drip for the third season, as no irrigation was added to the sub-surface drip paddock during the third season (1998-99). After rainfall, water ponded directly over the lysimeter and consequently inflated the amount of water that infiltrated through to the collection trays. This does however highlight the fact that under saturated conditions there is significant drainage through these heavy clay soils.

Drainage below the root zone of irrigated cotton is a significant proportion of the water supplied during the season (Table 1). Leaching fractions up to 50% of the total inputs were recorded with the average being 30% for sub-surface drip and 20% for furrow irrigated cotton. Over a season this represents between 1ML/ha and 3ML/ha of water moving past the root zone of irrigated cotton on these heavy clay soils.

What is also surprising is the rate at which this drainage moves to depth in the soil. Figure 2 shows two selected drainage events from the data collected by the lysimeters. It shows that within a day of significant rainfall and/or irrigation, water is collected in the lysimeter trays at 2m below the surface. This would indicate far greater rates of leaching than suggested by the soil properties (McGarry, 1996).

Data from the lysimeters also indicates that drainage events from irrigation followed by rainfall are larger in volume than those generated by irrigation or rainfall alone. When the soil becomes saturated, water flows at the saturated conductivity rate, which is higher than the unsaturated rate. When rainfall follows irrigation there is more water available, the profile stays saturated longer, and more water moves down the profile in a given amount of time.

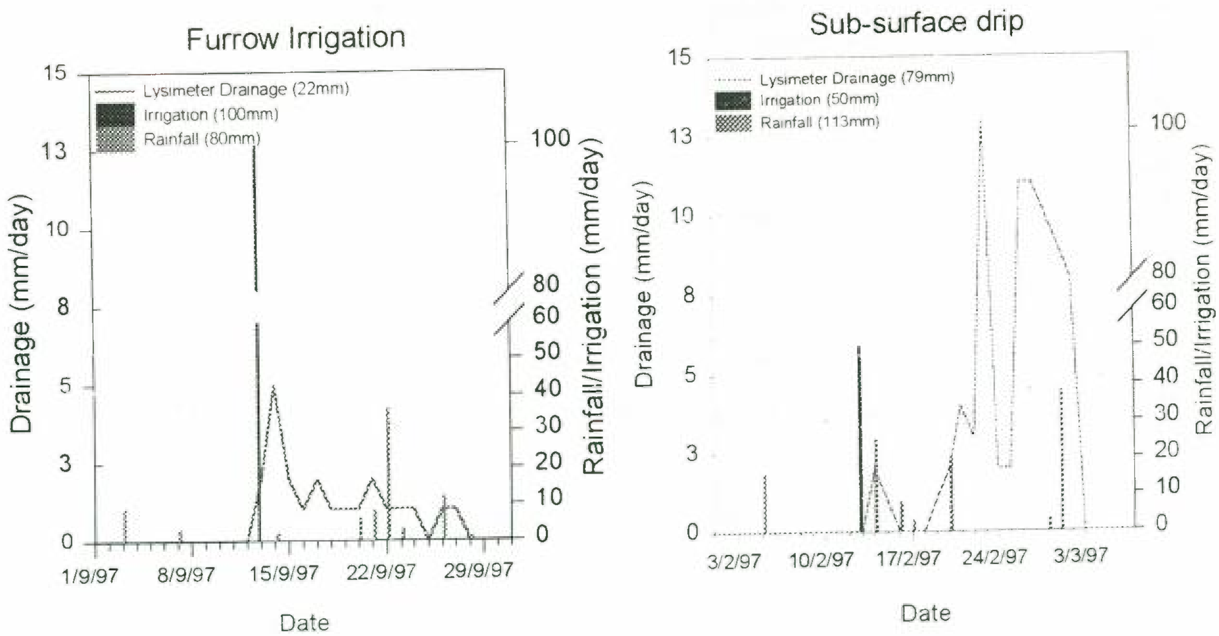

Figure 2. Two selected drainage events measured by the lysimeter under furrow and sub-surface drip irrigation. Totals for each parameter are given in brackets beside the legend

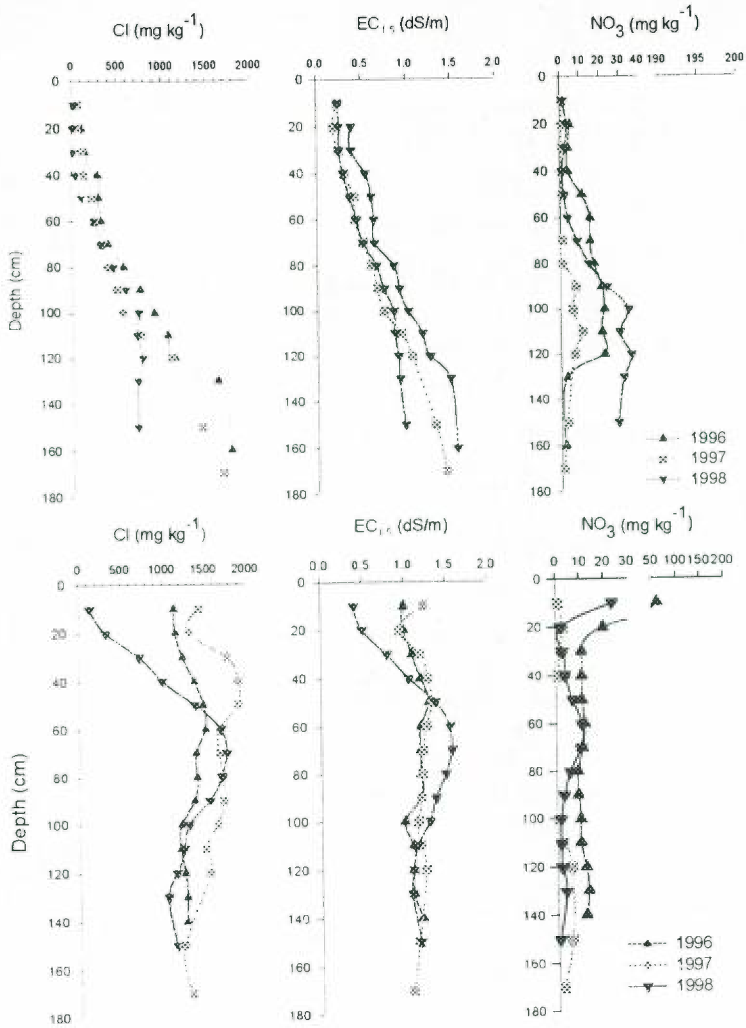

Figure 3. Measured soil properties, Chloride, Nitrate & Conductivity levels for furrow (bottom) and sub-surface drip (top) cotton sites.

Chemical Flux

Nitrate (NO_3) and Chloride (Cl) fluxes calculated from leachate captured by the lysimeters are presented in Table 1. Significant quantities of these two chemicals were measured in the drainage water and are consistent with the differences in drainage fluxes and levels of nitrate and chloride in the soil profile. Under furrow irrigation an average of 3123 kg/ha of chloride was leached below the root zone. This is higher than the 2393 kg/ha leached in the sub-surface drip site (Table 1).

Both sites were irrigated with bore water in the first season and then surface water in the following two seasons. Water quality is significantly different between the bore and surface waters. At the furrow irrigated site bore water irrigation of 3 ML added 16500 kg/ha in the first season compared to and an average of 8700 kg/ha added in 3 – 5ML the second and third seasons in the surface water irrigation. At the sub-surface drip site. 7400 kg/ha of chloride was added in 2 ML of irrigation from the bore during the first season and 1239 kg/ha in 1 ML from the surface water in the following two seasons.

The difference in drainage flux and chloride application between the two sites. are reflected in the soil chloride profiles (Figure 3). In the furrow irrigated site. an increase in chloride content of the soil between the first and second seasons can be seen at all depths with a significant bulge around 40 cm. In the third season (1998-99) where an extra 2 ML was applied in irrigation and rainfall. chloride has been mobilised and the bulge now appears around 70 cm with a drop in chloride content lower in the profile (Figure 3). This mobilisation lead to a slight increase in chloride leaching. measured by the lysimeter. from around 2800 kg/ha to 3000 kg/ha.

The relatively low chloride leaching at the sub-surface drip site is consistent with the lack of chloride in the soil profile and the higher rates of water flux. In the first season. irrigated with bore water added 7410 kg/ha to the profile and 4636 kg/ha leached. Irrigation with surface water in the second two seasons reduced chloride application to an average of 1240 kg/ha with 1270 kg/ha being leached. This reduction in chloride application is reflected in the subsequent lowering of the soil chloride content (Figure 4. 1998 sampling).

Total quantities of Nitrate (NO_3) collected from the two sites again reflects the differing application rates and drainage fluxes. An average of 215 kg/ha NO_3 leached each year under furrow irrigation compared to between 600 kg/ha – 1040 kg/ha under sub-surface drip (Table 1). Nitrate leaching of 215 kg/ha corresponds to about 50 kg/ha total Nitrogen (N). However approximately 200 kg/h N was applied each year to the furrow site. This equates to a nitrate flux of around 25%. which correlates well with the rates of water leaching. The rest of the N applied can be assumed to have been lost to crop uptake and or other parts of the nitrogen cycle. Low soil NO_3 profiles for the furrow site indicate no accumulation of NO_3 in the soil profile with values generally below 20 mg/kg (Figure 3).

Under sub-surface drip irrigation there is some accumulation of NO_3 in the soil (10 mg/kg – 30 mg/kg @ 100 cm) after three years with values mostly below 30 mg/kg of soil (Figure 4). There is however a large discrepancy between the amount of applied N and the equivalent N leached from the profile. Only 50 kg/ha N was applied to the sub-surface drip site compared to the 120 – 300 kg/ha seen in the leachate collected at 2 m which would seem to indicate a large N deficit in the soils of this site. An explanation of this can be found in the change from dryland cropping to irrigation on this site.

Fertiliser application to dryland cropping systems is far lower than that for irrigated lands. However there is a far smaller amount of water available to leach N from the soil profile. Modest N application rates combined with smaller leaching events and less crop uptake can combine to form a nitrate bulge in the soil in dryland cropping systems. If the paddock is converted to irrigation then the increased leaching fraction will mobilise this NO_3 and move it down the profile. It is this stored NO_3 that is being measured in the leachate from the lysimeters. Following the high rates of leaching in the 1998-99 season we would expect there to be little excess NO_3 to be leached in subsequent years. Further monitoring of these sites is required to prove this assumption.

Water samples collected from the lysimeter were submitted for analysis of pesticide content to determine if other chemical fluxes were occurring. The water samples were analysed for organochlorine. organophosphate. N-methyl carbamate and pyrethroid pesticides. triazine and phenoxyacid herbicides as well as the herbicides diuron. trifluralin. tebuthiuron and pendamethalin and the fungicides propiconizole. procymidone and dichlofluanid.

Both sites show significant levels of Prometryne (0.5 µg/L – 1.1 µg/L). This is applied to the crop each year and is not unexpected. What is unexpected are the levels of the herbicide Atrazine and its derivatives which have not been applied to the sites for over 5years. Up to 16.8 µg/L of Atrazine and between 0.1 µg/L – 0.8 µg/L of the derivatives have been detected in leachate under both sites. The pesticides are detected in almost all samples but the levels seem to be declining.

This would suggest release of bound material from the soil during drainage events. with the store being depleted through time. Although Atrazine is considered to have a short half life. low soil retention and moderate water solubility. these properties are affected by soil pH and CEC (Weber. 1994). Adsorption and persistence of the pesticides in soils is a function of both pesticide properties and the site microenvironment (Triegel and Guo. 1994). The soils of this site are highly active (CEC 60–90 meq/100g. ESP 20–30 meq/100g) and are not acidic (pH 7 – 8). properties that may have limited the mobility of Atrazine under the previous dryland farming system

Conclusions

Lysimeters have been used to study the movement of water and nutrients through soils for nearly 3 centuries (Kohnke. Dreibelbis. and Davidson. 1940). The objectives of this study were to utilise field lysimeters to quantify the volumes of leachate and movement of nutrients and chemicals through the soil profile under irrigated cotton. Irrigation management practises are the most significant factor influencing off site effects on the Murray-Darling Basin. Abbs and Littleboy (1998). suggest that there are large variations in leaching fraction between soil types and management systems for cropping in northern New South Wales. Previous experience within the cotton industry has indicated that the greatest management induced reduction in the leaching of water and chemicals is likely to be achieved by the implementation of sub-surface drip irrigation. The findings from this report would seem to be at odds with that notion: with up to two times the leaching measured below sub-surface drip irrigation.

Up to 50% of total input water was collected as drainage beneath sub-surface drip irrigated cotton. Even the lowest measured leaching fraction of 10% is still a significant fraction of the water applied. High leaching fraction gave rise to significant movement of Nitrate (NO_3-N) and Chloride below the root zone. The equivalent of between 600 kg/ha – 1500 kg/ha of NO_3 was detected in the leachate water beneath sub-surface drip irrigation. Chloride fluxes ranged between 1000 kg/ha and 4500 kg/ha depending on the season. Under furrow irrigation there was also significant leaching measured at the lysimeter site. Over three seasons an average of 20% of the water applied to the paddock in rainfall and irrigation was lost below the root zone of the crop. And with that leaching 200 kg/ha of Nitrate and 3000 kg/ha Chloride was flushed through the soil profile.

Yield data from these and most cotton growing areas does not indicate any decline due to management practises, despite the potential problems indicated by water and soil quality data. It is therefore even more pertinent to ask the question, what is happening to the large amounts of nitrate and chloride moving below these soils? The data from the lysimeters suggests that the only real avenue open to preventing these leaching losses is to change the management regime to minimise excess fluxes. Reduced nitrogen application and better irrigation scheduling are the keys to reducing the occurrence of leaching events, and minimising losses below the root zone.

ACKNOWLEDGEMENTS

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Modelling channel seepage interception by trees on a prior stream levee

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Introduction

Seepage from irrigation supply channels is a source of groundwater accession in many irrigation areas. Within northern Victoria, seepage reduction through channel lining or other engineering solutions is often not cost effective. Tree planting is then an alternative option. While tree plantations do not prevent seepage, they have the potential to intercept groundwater seepage flow and hence control local saline discharge.

Salinity Management Plans in northern Victoria have provided incentives to encourage tree planting under the assumption that all plantations will intercept seepage. These incentives are supported by a limited number of studies of tree planting alongside channels where groundwater drawdown responses have been demonstrated (Sonogan and Patto 1985; Tagirov and Dosahmetov 1970). At some sites however, plantations may not be appropriate for seepage interception. A recent study of groundwater/tree interactions in the Shepparton Irrigation Region in Northern Victoria has demonstrated salt accumulation within the rootzone in shallow watertable areas (Heuperman 1999). Salt accumulation would be expected to reach an upper limit that in some circumstances may allow plantations to survive but with restricted water use and growth (Morris and Collopy 1999).

Groundwater salinity is a critical factor in determining plantation water use and sustainability (Silberstein et al. 1999). Interception of relatively "fresh" groundwater in elevated positions of the landscape may contribute to a reduction in down slope discharge without the risk of salt accumulation. An example is 'break of slope" plantings in the Goulburn-Broken dryland catchment (Clifton and Miles 1998). Within irrigation areas, hydrogeological characteristics associated with some channels also offer potential for sustainable plantations. Factors determining sustainability and effectiveness of vegetation based interception strategies have been outlined within the context of dryland salinity in Western Australia (Hatton and Salama 1999). They include the position of trees within the landscape, groundwater salinity, proximity of seepage to the surface, hydraulic gradient, aquifer depth and transmissivity.

In the past, trees planted adjacent to channels in northern Victoria have been sited and designed with little consideration given to topography and hydrogeology. A study has therefore been initiated in the Pyramid Boort Irrigation Area to develop design criteria that will optimize seepage interception, and support sustainable tree growth.

Method

The area has an average annual rainfall and evaporation of ~400 mm and 1760 mm respectively. Irrigation within the area is not intense but supports some significant agricultural enterprises including dairying, mixed grazing, cropping, lucerne and tomatoes. The site selected for this study was a section of the Boort No. 2 channel located south of the Boort/Durham Ox Road. The channel was constructed during the 1960s into the levee of a prior stream. The soil associated with the site is a red brown earth (Mysia Loam). The prior stream is the dominant topographical feature in the area and is surrounded by clay floodplain soils. Historical records show that excavation during construction of the channel intercepted a shallow sand layer and that a complaint was made to the then State Rivers and Water Supply Commission in 1960 about seepage soon after construction. A more recent landholder commenced a staged tree planting program along the channel in 1991.

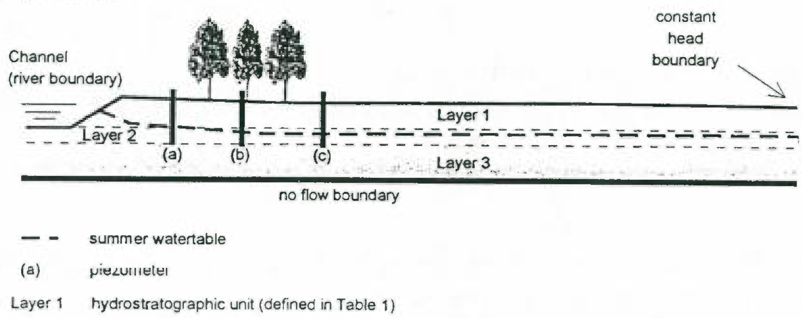
Approximately 200 m of trees, which were the subject of this study, were planted as an extension to the existing plantation in 1992 with assistance from Goulburn-Murray Water's predecessor, the Rural Water Commission. The plantation design included three rows of trees. The row closest to the channel was approximately 14 m from the waters edge. The distance between rows one and two was 2.5 m, with a 3.5 m spacing between the middle and outside rows. Trees within rows were planted 4.5 m apart. The outer two rows were planted with *Eucalyptus camaldulensis*, and the inner row, *Casuarina glauca*.

While canopy closure of the plantation was not achieved by June 1999, *E. camaldulensis* had developed a dense canopy. *C. glauca* had a relatively open canopy. This was possibly at least a partial consequence of the position of *C. glauca* within the plantation. Mean trunk diameter measured at 1.4 m above ground level for a sample of trees, increased from 0.13 m to 0.19 m and 0.06 m to 0.09 m for *E. camaldulensis* and *C. glauca* respectively between January 1998 and April 1999. Estimated height of the trees in January 1998 was 5.5 m and 4.5 m for *E. camaldulensis* and *C. glauca* respectively.

Drilling during the construction of a piezometer transect, confirmed the presence of a clayey fine to medium grained sand 1.5 –2.5 m below the surface and extending at least 65 m from the channel where it became progressively shallower (1.0 – 2.0 m) as it approached the bed of the prior stream. Regional groundwater salinity in the area is high (up to 67,000 EC). However sampling from the shallow aquifer indicated salinities between 350 and 1,020 EC below the plantation. Salinities increased to 1,340 - 2,040 EC at a distance of 65 m from the channel. Groundwater of less than 1,000 EC is unlikely to restrict tree growth (Silberstein et al. 1999).

The nested piezometer transect was installed at right angles to the channel through the tree plantation to a point approximately 65 m from the channel. Nests included 1 piezometer at 2 m into the clayey sand layer and one piezometer at 3 m into an underlying clay layer. Groundwater pressures were monitored from August 1997 to July 1999. A conceptual three-layered slice model (Figure 1) was developed based upon drilling information, gamma logs and groundwater salinity data. Given limited field data to describe vegetation and soil characteristics but a reasonable understanding of seepage processes, the saturated zone model MODFLOW (McDonald and Harbaugh 1988), was used to derive the water balance at the site, over a 23 month period.

Figure 1 Conceptual model



Boundary conditions

Investigations at the site have shown that the seepage process is dominated by saturated flow to the watertable. For channels with high seepage rates, a groundwater mound will develop forming a groundwater divide beneath the channel with some seepage losses dissipating to each side of the divide. For modelling purposes, the channel was therefore defined as a river boundary. It was then assumed that half of the channel width (i.e. 5.5 m) contributed to groundwater accessions to the model domain (Figure 1). Approximately 65 m down gradient from the channel, a constant head boundary was established corresponding to the location of the furthermost piezometer from the channel.

Assuming constant hydraulic and channel heads for 30 day periods, boundary levels were computed for each 30 day stress period from a two point running average of irregularly collected field measurements. Rainfall recharge to the groundwater system was assumed to be 10% of rainfall based on McMahon (1984). Cumulative rainfall for each 30 day period was obtained from an automatic weather station at Pyramid Hill.

Groundwater use by trees was simulated using MODFLOW extraction wells (McDonald and Harbaugh 1988). Trees were assumed to be capable of extracting to a distance of 4 m from the edge of the plantation. ET_{pot} was also estimated from the weather station at Pyramid Hill. Missing ET_{pot} data was infilled using a linear regression relationship between the Pyramid Hill and Kerang weather stations (r^2=0.94). Estimates of evapotranspiration from the watertable from vegetation other than trees (ET_{other}), was calculated from a linear decline in ET_{pot} from the surface to an extinction depth at 1 m below the surface. The linear decline in ET_{other} is a simplification of an inverse exponential relationship between upward capillary flow and watertable depth for a range of soils in Poulton (1985). The extinction depth used for this study originates from lysimeter studies by C. Lyle in Poulton (1985) for a Shepparton fine sandy loam. While some capillary rise would be expected at depths deeper than 1 m, rates are likely to be less than 0.1 mm/day and are therefore considered to be insignificant in terms of the total water balance.

While deep leakage may be significant from a salt balance perspective, it was not considered important in terms of the water balance. A no flow boundary was therefore assumed at 5 m below natural surface. Estimates for model parameters are shown in Table 1. Isotropic and homogeneous conditions were assumed within each layer. The model was calibrated by adjusting tree groundwater use until modelled hydraulic heads visually approximated measured heads. Modifications were also made to channel bed conductance (K_B/l_B) during the calibration process where K_B is the vertical hydraulic conductivity of the bed sediment and l_B is the thickness of the sediments.

Table 1. Parameter estimates for each model layer in Figure 1.

Parameter	Layers as shown in Figure 1		
	1	2	3
Layer description	Sandy silt	Clayey fine to medium grade sand	Light to medium clay
Depth (m)	0-1.5	1.5-2.5	2.5-5.0
Hydraulic Conductivity (m day^{-1})*	0.37	2.5	$1.5x10^{-2}$
Storage Coefficient	$1.0x10^{-4}$	$1.0x10^{-4}$	$3.0x10^{-4}$
Specific Yield	0.10	0.10	0.02
Channel bed conductance (day^{-1})	NA	0.03	NA

*Thorne (1974) measured hydraulic conductivities of levee soils in the area to a depth of 1.5 m. Values ranged between 0.37 and 4.27 m/day

Results

Channel bed conductance and hydraulic heads

Rapid rises in hydraulic heads in response to channel level fluctuations suggested that channel bed conductance did not unduly restrict seepage. The conductance of the channel bed was set at 0.03 /day as a result of the calibration process. This corresponds to a silt layer of 0.3 m thickness with a vertical hydraulic conductivity of 0.009 m/day.

Matching of modelled hydraulic heads with measured hydraulic heads is shown in Figures 2 and 3. As boundary heads were smoothed relative to measured heads, the model output would not be expected to completely replicate measured heads. However the calibration results were pleasing given the homogeneous and isotropic conditions assumed within each layer.

Figure 2. Calibration of hydraulic heads within the plantation

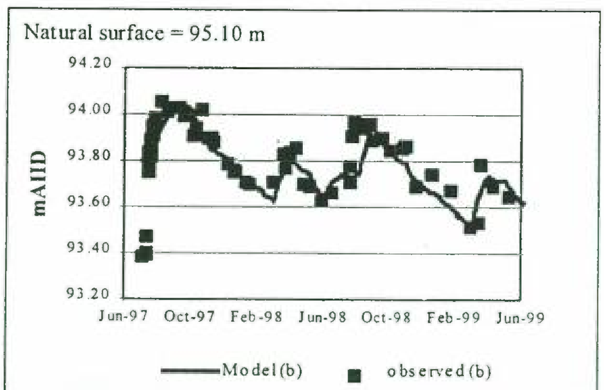
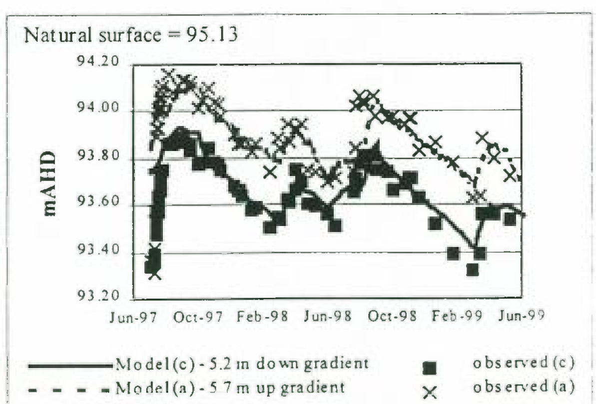


Figure 3 Calibration of hydraulic heads on each side of the plantation



Figures 2 and 3 show peak hydraulic heads during spring when the channel was full and there was low evaporative demand. During summer, levels were reduced by up to 0.5 m. Water levels recovered slightly towards the end to the irrigation season and then fell again during winter while the channel was empty.

Tree water use

The water balance approach used in this study does not incorporate rainfall stored within the unsaturated zone. During some periods of high evaporative demand such as late spring, trees may have access to significant soil water. Fortunately this study was conducted during a period of low to average annual rainfall. Groundwater extraction by trees therefore followed expected patterns.

Groundwater use derived from the calibration process is shown in Figure 4. Assuming plantation extraction up to 4 m from the trunks of the outer trees, groundwater use over the area of the plantation, varied from 0 mm/day during winter to 6.8 mm/day during peak summer periods (Figure 4). Lower groundwater use during winter reflects low evaporative demand as well as available rainfall storage within the unsaturated zone. High groundwater use occurred during periods of peak evaporative demand when there was depleted rainfall storage in the unsaturated zone, leading to greater tree reliance on groundwater. An increase in peak groundwater use from 5.1 to 6.8 mm/day and a corresponding increase in watertable drawdown were apparent from the 1997/98 to 1998/99 irrigation season reflecting tree growth during this period.

Estimates of peak groundwater use per tree of 70-90 L/tree/day is high compared with ~30 L/tree/day reported by Morris and Collopy (1999) at Girgarre. However the groundwater at this site is significantly fresher (<1,000 EC) and the supply almost unlimited.

Effectiveness of the plantation at intercepting seepage

Seepage rate estimates per unit area of channel water surface (Figure 5) were realistic when compared with pondage tests at the site that indicated an average rate of 10-12 mm/day (Hartley 1996; Thorne 1974). Figure 5 presents total seepage over the modelled period, and quantifies seepage not intercepted by the plantation. Seepage beyond the plantation was calculated as the difference between total seepage and tree water use for each 30 day stress period.

Figure 5 suggests that while the existing plantation intercepts most seepage during the period of peak evaporative demand (November – March), the plantation is relatively ineffective at intercepting seepage during periods of low evaporative demand when the channel is full (i.e. Sept - Oct and April - May). Figure 5 also shows that the amount of seepage not intercepted fluctuates significantly during late autumn to early spring as channel water level is the dominant factor influencing seepage during periods of low evaporative demand.

Figure 4. Tree groundwater use

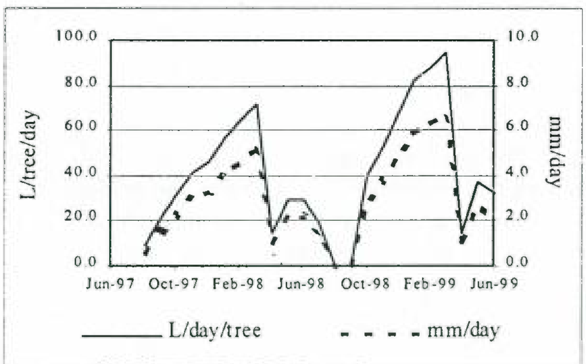
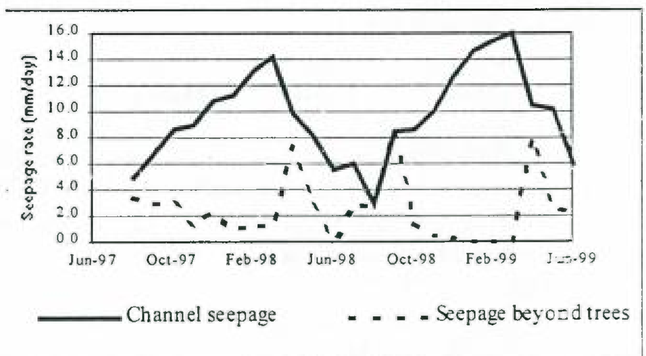


Figure 5. Effectiveness of trees at intercepting seepage



Additional Model runs

The modelling in this study is a simplification of a very complex process involving channel seepage, groundwater movement and water use by vegetation. In considering the results presented in the previous section it must be noted that the water balance approach to estimating groundwater use by vegetation leads to accumulation of errors from each component (Bouwer 1975). Modelling to test the impact of plantation design and site characteristics is therefore more useful than are the derived estimates of seepage rate and tree groundwater use.

The model was therefore re-run to test the effectiveness of the plantation design at intercepting seepage when the hydrogeological parameters of the highest conducting layer (layer 2) were varied. The results are summarised in Table 2 as average daily seepage and as average daily seepage beyond the plantation that "escaped" interception. In assessing the results of these model runs, it must be pointed out that tree groundwater use derived from the calibration process is assumed to remain constant under changing hydrogeological conditions whereas in reality, tree water use will vary in response to changes in water availability.

Run 2- removal of trees

Where trees have access to shallow lateral seepage, water extraction by trees would be expected to increase the difference in head between the channel and the watertable, inducing higher seepage rates. When trees were removed from the model, seepage rates declined significantly during the summer/autumn period. In fact the plantation appears to almost double the average seepage rate from 5.5 to 9.7 mm/day (Table 2). Seepage increases induced by the plantation were less during periods of low evaporative demand. Such a response is consistent with conclusions from other seepage studies that considered seepage response to seasonal watertable fluctuations (McLeod 1993) and groundwater use by vegetation (Bouwer 1975; Collis-George and Smiles 1963).

Table 2. Testing impact of varying hydrogeological parameters on seepage rates.

Run No.	Model run	Average channel seepage[1] (mm day^{-1})	Average seepage[1] beyond plantation (mm day^{-1})
1	Calibrated model (parameters in Table 1)	9.7	2.3
2	Calibrated model with trees removed	5.5	5.5
3	Channel bed conductance = 0.06 day^{-1}	11.1	3.6
4	Doubling of plantation distance from channel	8.9	1.4
5	Channel bed conductance = 0.06 day^{-1} Doubling plantation distance from channel	10.1	2.6
6	$K_{Layer\ 2}$= 4 m day^{-1}	11.9	4.5
7	$K_{Layer\ 2}$= 1.25 m day^{-1}	7.4	0[2]

[1]Seepage is expressed as depth of seepage over half of the channel surface width.

[2]Seepage flux extended beyond the channel on occasions where there were high channel levels and low evaporative demand. However for periods of high evaporative demand, groundwater use exceeded channel seepage.

Run 3 – channel bed conductance

Sediment in the bed of the channel provides an important impediment to seepage. McLeod (1993) used sensitivity analysis within a model to determine the impact of channel bed conductance on seepage. Repeated reduction of the bed conductance eventually eliminated seepage. Increases in conductance induced higher seepage until a level of conductance was reached whereupon the transmissivity of the underlying aquifer limited seepage rather than the permeable channel bed. Run 3 shows that a doubling of the channel bed conductance only increased seepage by 1-2 mm/day. Additional trees would be necessary to intercept increased seepage at higher values of channel conductance.

Runs 4 & 5 – plantation distance from the channel

If seepage from the channel is not unduly impeded by low channel bed conductance, Darcian flux (q) per unit area of channel will have a significant impact on seepage rate. Darcy's Law is given by $q=Ki$, where K is the hydraulic conductivity and i is the hydraulic gradient. As Darcian flux is proportional to the hydraulic gradient, an increase in distance between the plantation and the channel reduces the magnitude of the seepage increase, as there is a corresponding decline in hydraulic gradient from the channel to the point of maximum drawdown.

However channel bed conductance at this site is such that the reduction in the hydraulic gradient from moving the plantation (run 4) only reduced average seepage by 0.8 mm/day. The benefit in moving the plantation was that the reduced seepage had previously been escaping interception. A similar result was achieved when the distance between plantation and channel was increased for a channel with higher bed conductance (Run 5). As indicated previously, these results are based on the assumption that trees will not modify their water use in response to changes in water availability or salinity. This is unlikely to be the case.

Irrespective of this assumption, the results show that as a result of moving the plantation, the increase in seepage is not large. As groundwater salinity is generally found to increase along a flow path, planting trees at increasing distance may risk the sustainability of a plantation. A possible trade off exists then, between sustainability and minor reductions in seepage as a result of reduced groundwater drawdown. A further consideration may be the value of the land sacrificed between the channel and the plantation, and possible salinisation of this area depending upon depth to watertable.

Runs 6 & 7 – hydraulic conductivity

Runs 6 and 7 demonstrate that hydraulic conductivity of the shallow aquifer can have as significant impact on seepage as channel bed conductance. When the hydraulic conductivity was increased to 4 m/day (run 6), additional tree water use is required to adequately intercept seepage. Increased tree density or plantation width may therefore be necessary. For the case where the hydraulic conductivity was halved (run 7), trees intercepted all seepage. This situation has potential to threaten the sustainability of the plantation as the trees may begin to utilise water from the regional watertable. This would lead to salt accumulation if deep leakage were insufficient to maintain adequate leaching. Run 7 therefore demonstrates that the issue of sustainability may be of concern in areas of high regional groundwater salinity, if too many trees are incorporated into the plantation design.

Conclusion

Simple water balance groundwater modelling has been used to demonstrate the effectiveness of trees at intercepting relatively "fresh" channel seepage over a 23 month period. Modelling indicates that the plantation design at this site appears to be adequate to intercept most seepage. However for sites with higher aquifer hydraulic conductivity and/or channel bed conductance, a plantation may require additional trees.

The next stage of the study is to consider sites where seepage processes are more complex and groundwater more saline. This will involve the analysis of data at a site where deep seepage flux moves through preferential pathways to a deeper aquifer such that trees have only limited access to low salinity seepage while being exposed to saline regional groundwater.

Acknowledgments

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AN INTEGRATED SYSTEM FOR GROUNDWATER VULNERABILITY ASSESSMENT IN THE LIVERPOOL PLAINS OF NSW

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Summary

A system for evaluating the likelihood of groundwater contamination was developed using an attenuation factor model at the regional scale and solute transport models at the field scale. These models were integrated into a GIS system for efficient groundwater vulnerability assessment in the Liverpool Plains of NSW. The system allows the user to access the data. run the assessment models and evaluate the outcomes from the same interface in a GIS environment. The system could be extended for many applications in the Murray-Darling Basin. such as pollution control licensing and salinity hazard assessment.

1. Introduction

Widespread use of agrochemicals has become a major concern in the protection of water resources. Otto (1996) proposed a multi-level strategy to assess the leaching potential of pesticides and this was implemented through the development of an integrated groundwater vulnerability assessment system. This paper presents the underlying theory for groundwater vulnerability assessment and its modelling. and the integration and use of vulnerability assessment models in a GIS framework.

2. Groundwater Vulnerability Concept

The vulnerability of groundwater to contamination is defined as:

The tendency or likelihood for contaminants to reach a specified position in the groundwater system after introduction at some location above the uppermost aquifer.

Pesticides in the vadose zone experience phase-transfer. transformation and transport processes. In phase-transfer processes. pesticides may be sorbed. such as by clays or sediments in the vadose zone. or volatilised. In transformation processes. pesticides may degrade into metabolites and carbon dioxide abiotically or biotically. In transport processes. pesticides are carried into aquifer by downward water flow. although upward movement of pesticides may also happen due to evaporation or plant uptake. Transport of pesticides through downward water flow is the major mechanism for pesticides to leach into an aquifer system while sorption and degradation slow down or reduce the leaching. The rates of the interacting processes and the fate of pesticides depend on the chemical properties of pesticides. soil properties and conditions. weather and climatic conditions. microbiological conditions. plant types and growth stages. and soil-crop-pesticide management practice.

3. Assessment Methodology

Groundwater vulnerability due to non-point pollution sources may be evaluated through a multi-scale strategy:

- Identify the areas of potentially high risk of groundwater contamination from the application of a pesticide in a region.
- Examine the relative likelihood of pesticide contamination in different polygons within each area identified previously.
- Evaluate the fate of a pesticide within a specific soil profile.

The strategy is implemented by using the attenuation factor (AF) screening method (Rao et al. 1985) to assess the groundwater vulnerability at the regional and sub-regional scale and the process-based simulation methods at the field scale in a GIS environment.

3.1 Attenuation Factor (AF) Screening Model

The AF index, denoting the mass emission of a chemical substance from the unsaturated zone to groundwater, is defined by

$$AF = \frac{M_2}{M_1} = \exp\left(-\frac{0.693 R_f L_g}{\frac{Q_n}{\theta} T_{1/2}} \right)$$

where M_2 is the mass of the chemical exiting the vadose zone of depth L_g. M_1 is the mass of the chemical applied at the ground surface. Q_n is the net aquifer recharge. θ is the volumetric moisture content of soil at field capacity. $T_{1/2}$ is the soil biological half-life of the pesticide. The AF values range from 0 to 1, with larger values indicating higher contamination potential.

The retardation factor R_f, which reflects the delay of pesticide leaching due to sorption, is given by

$$R_f = 1 + \frac{\rho K_{oc} f_{oc}}{\theta}$$

where ρ is the soil bulk density, f_{oc} is the fraction of organic carbon in the vadose zone. K_{oc} is the organic carbon-normalised sorption coefficient of the pesticide. Soil of higher organic or clay content reduces the contamination potential of a pesticide in larger extent.

Although the AF model encapsulates the major influential factors in the pesticide leaching process, computationally, it is hard to represent preferential flow paths in many types of soils. Preferential flow refers to all forms of rapid downward movement bypassing major portion of soil matrix through high-conductivity pathways in vadose zone. Preferential flow occurs in continuous, non-capillary structural voids or in the area of capillary pores that have locally high hydraulic conductivities. The macropores take the form of cracks, root channels, worm holes, and the like. Preferential flow can also occur due to instability in wetting fronts under certain conditions. As a result of preferential flow, water and solute fronts penetrate greater depths, and much faster, than the prediction of leaching models. To consider the effect of preferential flow, a reduction factor is introduced for L_g, such as in fine-textured or sandy soil zone, where preferential flow can widely occur.

In contrast to preferential flow, thick shaly limestone or clay layers of low permeability in the vadose zone can effectively protect the underlying aquifer from the pesticide contamination. An amplification factor for L_g may be adopted to lower the AF value where this kind of geological setting is known or the area may be excluded from the AF modelling.

The inherent uncertainty of each factor in the AF model can deviate its prediction from the field observation. The uncertainty, however, is not considered in the model since the data required for uncertainty study is generally unavailable, not to mention at regional scale. Data scarcity, after all, has been driving the development or adoption of relatively simple vulnerability assessment models.

It is worth mentioning that relatively higher AF value in an area does not necessarily mean higher concentration of pesticides in the groundwater of that area. The actual occurrence of pesticides in groundwater heavily relies on agricultural practice, for example, the amount of pesticide applied, the pesticide formulation (eg. use of additives), the timing of the application relative to weather condition (eg. rainfall and temperature), the cultivation habits, the amount of irrigation water, and the method of application (eg. direct application to soil or spray). By understanding the process of groundwater pollution and using the pesticides wisely in practice, however, the risk of contamination can be controlled at an acceptable level.

3.2 Process-based Simulation Models

The AF model is supplemented by the models that simulate the chemical migration, retention, and degradation in unsaturated vadose zone. The simulation will answer the questions like:

- How long does it take for a dissolved pesticide to leach to the water table?
- What is the maximum concentration at the water table, and when does it happen?
- How far does the center of mass of the dissolved pesticide travel downward through the soil profile within a specified period?

Two representative simulation models are integrated into the GIS system:

- *Chemical Movement in Layered Soils* (CMLS), developed by Nofziger and Hornsby (1986).
- *Leaching Estimation and Chemistry Model* (LEACHM), developed by Wagenet and Hutson (1987). The sub-program LEACHP is integrated into the assessment system.

3.2.1 CMLS

CMLS is a relatively simple one-dimensional process-based simulation model and therefore reduces the demand for input data. It extends the AF model by taking the variables, such as time and soil layers, into account. It assumes that chemicals move only in the liquid phase, and all of the water in the soil is active in the movement. Water already in the soil profile is pushed ahead of the inflowing water in a piston-like manner. The chemical movement is retarded by the sorption process represented by a linear and reversible equilibrium model, and no chemical dispersion is considered. Up to 20 soil layers of different properties in root zone are allowed in the model. The depth that the chemical travels is predicted by

$$D_{t+\Delta t} - D_t = \frac{q}{R_f \theta}$$

where D_t is the depth of the chemical at time t, q is the amount of water passing the depth D_t during the time Δt (daily), θ is the volumetric water content of the soil at field capacity, and R_f is the retardation factor for the chemical in the particular layer of soil, calculated using the same R_f formula in the AF model (Section 3.1). The depth of the chemical in a soil layer increases if the infiltrating water is more than the water required to recharge the layer to the field capacity after deducting the evapotranspiration. The mass of the chemical is assumed to experience first-order degradation process and the residual chemical after Δt is estimated by

$$M_{t+\Delta t} = M_t 0.5^{\frac{\Delta t}{T_{1/2}}}$$

where $T_{1/2}$ is the degradation half life of the chemical in the soil layer. The soil water flow model of CMLS suggests that it is suitable for modelling chemical movement in sandy soil.

3.2.2 LEACHP

LEACHP is a complex one-dimensional solute transport model. Transient soil water flow in vertical direction is expressed by Richard's equation

$$C_\theta \frac{\partial h}{\partial t} = \frac{\partial}{\partial z}(K_\theta \frac{\partial H}{\partial z}) - U$$

where t is the time, z is the depth, H is hydraulic head, h is the soil water pressure head, U is the water loss due to transpiration, θ is the volumetric water content, K_θ is the hydraulic conductivity and C_θ is the differential water capacity. Water retention and conductivity relations are based on the Campbell's equation

$$h = a(\frac{\theta}{\theta_s})^{-b}$$

where θ_s is the volumetric water content at saturation, a and b are empirical constants. Water retention parameters may also be estimated using a regression model of soil particle size distribution, bulk density and organic matter content. Numerical method is essential for solving the Richard's equation and LEACHP adopts the finite difference method. The depth is thus represented by a number of horizontal segments (defined by nodes) and the time is divided into a number of intervals. The solution of the equation requires the boundary conditions to be specified. The upper boundary condition can be set to model ponded or non-ponded infiltration, evaporation or zero flux. The lower boundary condition can be set to represent fixed depth water table, or free drainage, or zero flux or specified flux. Crop growth model is used to estimate crop cover fraction for partitioning evapotranspiration and the depth of root distribution for determining the crop uptake of the water and the chemical. In addition, a capacity model and steady-state flow model are available for simulating soil water flow where the models are suitable.

Transient chemical movement is represented by the convection-diffusion equation

$$\frac{\partial C_T}{\partial t} = -\frac{\partial J_s}{\partial z} \pm \phi$$

where C_T is the total solute concentration in all phases, J_s is the total solute flux, ϕ is the sources or sinks of solute. C_T is partitioned between sorbed, solution and gas phases. The sorbed chemical is assumed to be proportional to the chemical concentration in solution and calculated using the partition coefficient K_d. The liquid-vapour partition is calculated similarly using a modified Henry's law constant. The diffusion flux density in both liquid and gas phase is determined by

$$J_D = -D_o \frac{dc}{dz}$$

where D_o is the appropriate molecular or ionic diffusion coefficient and c is the chemical concentration. The convective flux density of solute may be calculated by

$$J_L = -\theta D_M \frac{dc}{dz} + qc$$

where q is the macroscopic water flux and D_M is the mechanical dispersion coefficient that represents the mixing between large and small pores as the result of local variation in mean water flow velocity. It may be estimated by multiplying the pore water velocity and the dispersivity. The gas convection of volatile chemical in soil may be considered by enlarging the molecular diffusion coefficient.

Chemical transformation and degradation, including temperature and water content effect on the transformation and degradation rate, may be considered in the simulation. Together with crop uptake, they constitute the sink term of the convection-diffusion equation. During the simulation, the chemical is either directly applied or initially present in soil profile or dissolved in infiltrating water.

Obviously, LEACHP allows much more effects to be considered but the demand for input data is also high.

3.3 Regional Groundwater Level Prediction

The regional ground water level, required by the AF model application (Section 3.1), is predicted using Hydrogeomorphic Analysis of Regional Spatial Data (HARSD) approach (Salama *et al.* 1996). The HARSD approach was designed for hydrogeomorphic classification of catchment and construction of hydraulic head surface at regional scale in the absence of detailed hydrogeologic description.

According to HARSD approach, the landscape in the study area may be classified into hydrogeomorphic units which are expected to have similar aquifer properties and recharge as well as discharge behaviour. The topographic attributes, such as elevation, slope, break of slope, plan and profile curvature, are used solely or in combination as the classification rule. These topographic attributes are derived from digital elevation model of the landscape using the grid modelling extension of GIS software ArcView.

For instance, potential discharge areas are characterised by abrupt change in slope, or significant break of slope (BOS), and concave surfaces represented by negative profile curvature. Appropriate BOS critical values may be selected by referring to BOS attribute in known discharge areas. Potential waterlogged areas are relatively flat and of lower elevation. These areas are conceptually distinct from discharge areas in that they are in poorly drained areas. Waterlogged areas can be located by defining small slope and low elevation as the classification rules. Once the landscape is divided into hydrogeomorphic units, regression formula is derived from the bore data to estimate the hydraulic head surface (or groundwater table whichever appropriate) in each hydrogeomorphic unit.

4. Use of the Integrated System

The groundwater vulnerability assessment models were developed into the plug-in modules which run inside the GIS system. They are

- *Groundwater Vulnerability Assessment (gwva.avx)*, an ArcView extension, contains the functions for enhanced data inquiry and the attenuation factor (AF) mapping at regional scale.
- *HARSD (harsd.avx)*, an ArcView extension, contains the functions for hydrogeomorphic modeling and hydraulic head surface prediction.
- *CMLS (cmls.avx and cmls.dll)*, an ArcView extension and Windows dynamic link library, provides functions to run CMLS in the GIS environment.
- *LEACHP (leachp.avx and leachp.dll)*, an ArcView extension and Windows dynamic link library, provides functions to run LEACHP in the GIS environment.

The system is accompanied by two manuals (Zhou, 1998a and 1998b). The theory manual presents the theoretical background of groundwater vulnerability assessment and its computer modelling. The user manual shows the installation and use of the integrated system. Although statistical analysis has been considered in the design of the assessment system and part of theoretical work has been carried out by the author, the technique was not integrated because the data required to practically implement the technique is not available. A simplified sample database is provided for demonstration of the use of the system.

The integrated groundwater vulnerability assessment system is a user-friendly Windows application. It consists of GIS software, relational database management software and plug-in modules for predicting groundwater vulnerability and simulating chemical movement in soil. GIS software ArcView is used to visualise and process the spatial data and Microsoft Access is used to manage the attribute data. The spatial and attribute database are linked at run time to retrieve and synthesise the required data set. The user can access the data and run the models from the same interface. The user interface will respond dynamically and guide the user through the vulnerability assessment in GIS environment. It will prompt the user to enter only the data required by the particular type of assessment. For instance, a groundwater manager may use the AF model to evaluate the groundwater vulnerability in their management area. They may use the data browser to view the datasets in the database and use the inquiry tools to obtain more information. They can add the data layers into a view and use the AF calculation dialog to generate the AF maps. Then they can use AF ranking function to assign the likelihood class and use AF inquiry tool to further explore the AF maps.

The system was developed on the basis of the land and water database in the Liverpool Plains of NSW, supplied by DLWC of NSW and AGSO. Figure 1 shows a sample AF map inside the Liverpool Plains catchment area. Among frequently used pesticides, Atrazine was singled out due to its frequent detection in the groundwater sampling in the area. The AF study area was selected by overlaying the soil landscape map and the alluvial formation map where most of agricultural land were situated and thus the agrochemicals were most likely to be applied. The depth to groundwater table at the regional scale was predicted using the shallow bore data and the HARSD approach. It is noted that the depth thus predicted may actually be the depth to the hydraulic head surface in the presence of heavy clay soil and a confined aquifer. Alternatively, a designated depth may be used since the soil sampling depth is often limited to the range of 1.5 m. The soil properties were calculated from the soil survey data for each soil landscape and assigned to the soil polygons belonging to the soil landscape. In the sample AF map, the area of relatively higher leaching potential and thus deserving higher attention is highlighted by red color. Although the input data is rough, it still distinguishes the area around Lake Goran as an area of higher contamination likelihood. This area is known to have a shallow water table and soils of lower permeability.

The accuracy of the AF map will depend on the accuracy of the input data used in the AF calculation. Often, the challenge faced by the users of the system is the availability of quality data. With the aid of the system, the users may start to produce simplified AF maps under some assumptions, eg uniform vadose zone depth if the study area is not large and relatively flat. Since AF is a relative index, the users can go ahead with the calculation if they can estimate the relative ratios of a variable among geographic zones even without knowing the exact values of the variable. Judged by their local knowledge, they may use the AF maps as the road map in their management work. For instance, they may arrange to collect more data from the zones of higher predicted leaching likelihood. With the data, they will be able to improve the AF maps and gradually build up their confidence on the prediction in the iterative process.

6. Conclusion

In response to increasing environmental concern over widespread use of agrochemicals, an integrated system has been developed for groundwater vulnerability assessment. It allows rapid evaluation of the likelihood of groundwater contamination in a GIS environment and it is an efficient decision-making tool for groundwater management.

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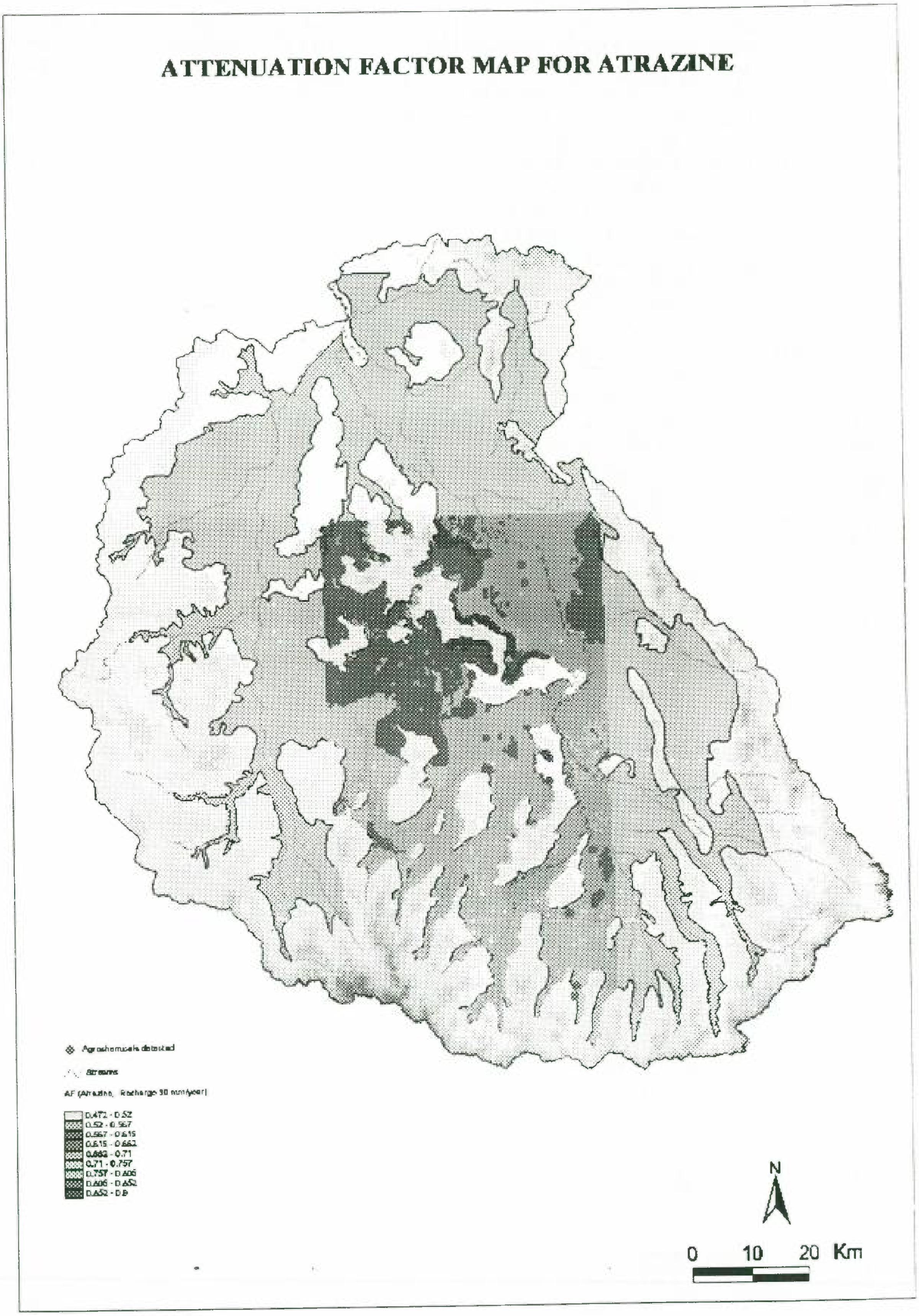

Figure 1: Attenuation factor map (sample output) in an area of Liverpool Plains of NSW

REGIONAL INVESTIGATIONS

MALLEE REGION GROUNDWATER MODELLING

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INTRODUCTION

The Mallee Region is predominantly a dryland farming area which covers areas of South Australia and Victoria in the Murray Basin (Figure1). Due to the large quantity of good quality groundwater in the underlying aquifer, more than 70 irrigation wells have been drilled over the last ten years or so. In areas of concentrated pumping, falling groundwater levels have affected neighbouring stock and domestic wells.

Management of the groundwater resources by licensing irrigation use, allocation and Permissible Annual Volumes (PAV's) of maximum extraction, occurs in the areas. In order to achieve sustainable development in all these areas, a detailed five layer MODFLOW groundwater flow model (McDonald and Harbagh, 1988) and MT3D solute transport model (Rumbaugh, 1993) were developed as a management tool to:

predict the changes in regional groundwater levels and any salinity changes due to various pumping scenarios;

estimate the maximum local drawdown at end of pumping seasons;

calculate the water balance and groundwater flows between aquifers.

HYDROGEOLOGY OF THE MODELLED AREA

The model area is located in the Mallee Region of the Murray Basin, where there are five main hydrogeological units (aquifers and confining layers) as shown in Figure 2. The units are, in order of increasing depth below the surface:

Layer 1: Pliocene Sands aquifer - generally an unconfined aquifer over 50 m in thickness. Salinity in the aquifer ranges between 1000 and 35 000 mg/L. Groundwater movement is generally from the south to north under fairly low gradients;

Layer 2: Bookpurnong Beds (confining layer) - this unit is absent in the western third of the modelled area. however it dips down gradually to the east and increases the thickness;

Layer 3: Murray Group Limestone (MGL) aquifer - generally occurs as an unconfined aquifer in the western third of the area and is confined over the remainder. This is the aquifer developed for irrigation, town water supplies and stock and domestic use. The thickness of the layer averages 100 m and groundwater movement is from the south to the northwest under fairly low gradients. Salinity of the groundwater ranges between 500 and 3000 mg/L;

Layer 4: Ettrick Formation (confining layer) - occurs between the Murray Group Limestone and the underlying aquifer. The layer is around 15 m in thickness;

Layer 5: Renmark Group aquifer - confined by the Ettrick Formation, it is about 150 m in thickness. Groundwater flow is from southeast to the west and northwest. The salinity of the groundwater ranges from 500 to 3000 mg/L.

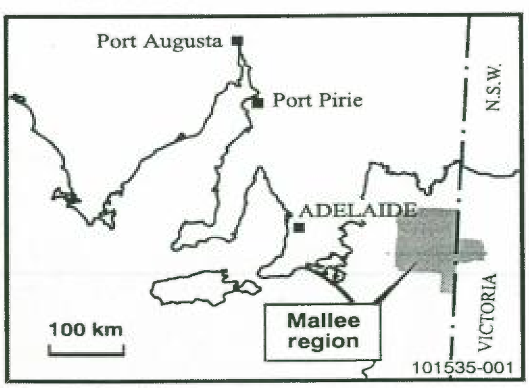


Figure 1 Locality map

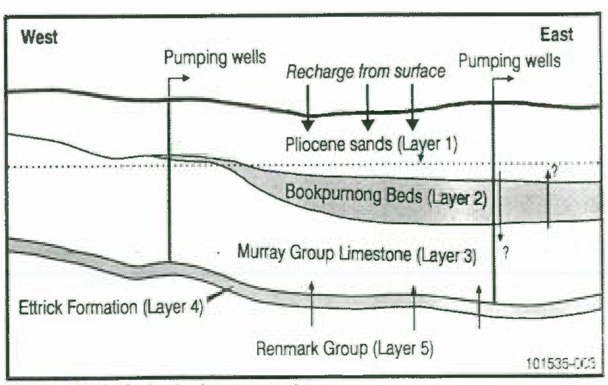


Figure 2 Hydrological cross section

MODEL GRID

The model area extends 115.75 km (east to west) by 103.5 km (north to south) covering the Mallee Proclaimed Wells Area, the Murrayville Groundwater Supply Protection Area, and Border Zone 10 and Part Zones 9 and 11. The minimum grid size is 500 by 500 m covering the main irrigation areas with the maximum 1000 by 1000 m along the boundaries. Details of the boundary conditions and hydraulic properties in the model are presented in Yan and Barnett, (1999).

STEADY STATE CALIBRATION

The first stage of the modelling exercise involved a steady state calibration undertaken over the entire model domain. As part of this process, the initial boundary conditions, hydraulic conductivities, recharge rates and upward leakage from the underlying aquifer were firstly applied on the basis of the basic conceptual model and values obtained from a previous model (Barnett 1990). These parameters were then adjusted until the calculated and observed water levels were matched very closely. The accuracy of the steady state calibration was evaluated by considering the regional groundwater flow directions and a comparison of the observed groundwater levels and modelled water levels (Figure 3). These steady state calibrated water levels were then used as initial groundwater levels (pre-pumping) for the following transient calibration exercise.

TRANSIENT CALIBRATION

The transient calibration was undertaken using the observed water levels and the corresponding pumping volumes during the 1997/98 irrigation season. The calibration was again performed by adjusting the specific storage, leakage (upward/downward) and hydraulic conductivity values within reasonable limits to achieve an acceptable match between calculated and observed water levels. The transient calibration was concentrated on Layer 3 (MGL aquifer) in areas where monthly pumping volumes and water level observation data were available. The Pliocene Sands and Renmark Group aquifers are the sources of the downward and upward leakage respectively into the MGL.

Comparison of the calculated and observed hydrographs was used to evaluate the accuracy of the transient calibrations (Figure 4). The calculated drawdown contours also show a good comparison with the observed contours.

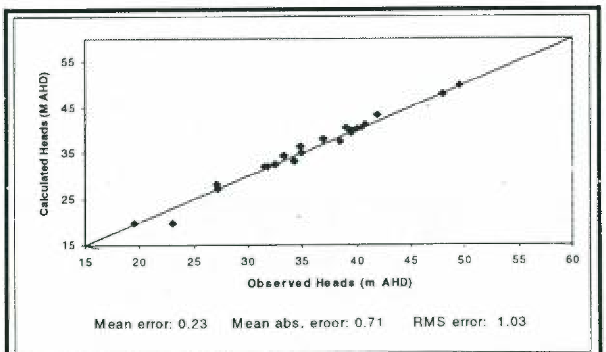


Figure 3 Steady state calibration results

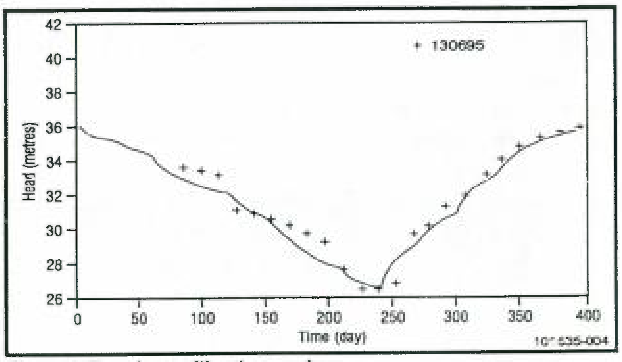
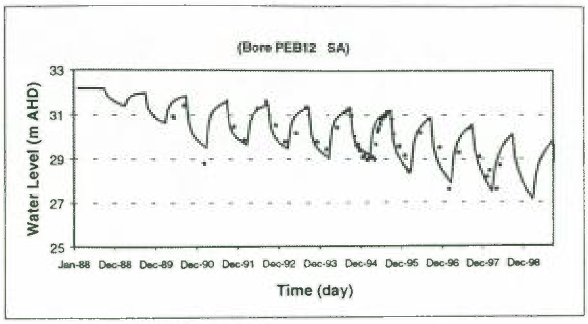


Figure 4 Transient calibration results

MODEL VALIDATION

Model validation was carried out by undertaking a 10 year transient run from 1988 to 1998 in an effort to match the observed drawdown levels due to the long term pumping. Because only a very small number of bores are metered, pumping volumes for each irrigation well during the 10 year period were estimated by interviewing each irrigator where possible to obtain the number of hours of pumping and the pumping rate of each bore.

The results of the validation can be seen in the comparison of the calculated and observed water level between 1988 and 1998 (Figure 5). The high accuracy of the validation is shown by the small difference between the observed and calculated water levels, and also the similar trend in water level decline.



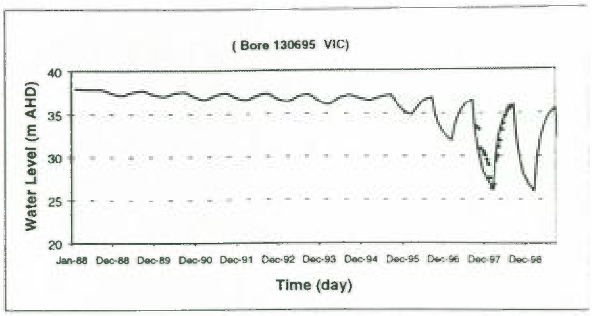


Figure 5 Two validation results

SCENARIOS

The validated model was used to predict future groundwater level declines due to irrigation for 35 years from 1988 to 2024 using the following scenarios:

Scenario 1: current usage for 1997-98 (16 000 ML)
Scenario 2: use of all allocated water (45 000 ML)
Scenario 3: use of full PAV (60 000 ML)
Scenario 4: testing long term drawdown response over 60 years (10%, 25% and 50% of full PAV)

The results of the predictions for each scenario has been assessed on the basis of the following:

- Water level decline (hydrographs) at the observation bores
- Drawdown contours at the end of the pumping season (maximum) and the end of recovery (minimum)
- Groundwater elevation contours and flow directions at end of pumping season in 2024
- Increased leakage from Pliocene Sands to MGL aquifer and the TDS concentration increase over time.

RESULTS

The prediction results gives the maximum drawdown contours at the end of the pumping season and the residual drawdown contours at the end of the recovery. The maximum drawdown at the end of pumping season in 2024 varies from 18 m (Scenario 1) to 28 m (Scenario 3). The results for Scenario 3 (use of full PAV) are shown in Figure 6. The residual drawdown similarly varies from 6 to 12 m.

After 25 years pumping at the full PAV, it was noticed that the hydrographs were still showing a steady decline of 15 cm/year. Scenario 4 was modelled to determine whether lower extractions could lead to a steady state drawdown. The results show that the drawdown only reached steady state after 60 years when the pumping has been reduced to only 10% of the maximum PAV. This is because the relatively low aquifer transmissivity does not allow inflows to the pumping areas to keep pace with the outflows, resulting in a gradual decline in storage. It also suggests that inter-aquifer leakage (upward and downward) is relatively low.

DEPRESSURISATION

One of the impacts of drawdowns in the confined MGL aquifer, is that the centres of the large cones of drawdown will become unconfined when the potentiometric surface drops below the top of the aquifer. In practice, the drawdown will stabilise in these areas, because it will cease to become an instantaneous pressure effect and will respond only to the gradual dewatering of the aquifer.

DOWNWARD LEAKAGE

For Scenarios 1 to 3, the increased downward leakage from the Pliocene Sands watertable aquifer down to the MGL aquifer as a result of the reduced heads from 35 years of pumping, has been presented in Figure 6. Assuming a salinity of 10 000 mg/L for the Pliocene Sands aquifer, the resultant salinity changes in the MGL aquifer are also shown. The results show virtually no salinity impact whatsoever. It is anticipated that increased monitoring and sampling of the Pliocene Sands aquifer will allow more accurate determinations of this leakage and its salinity impact.

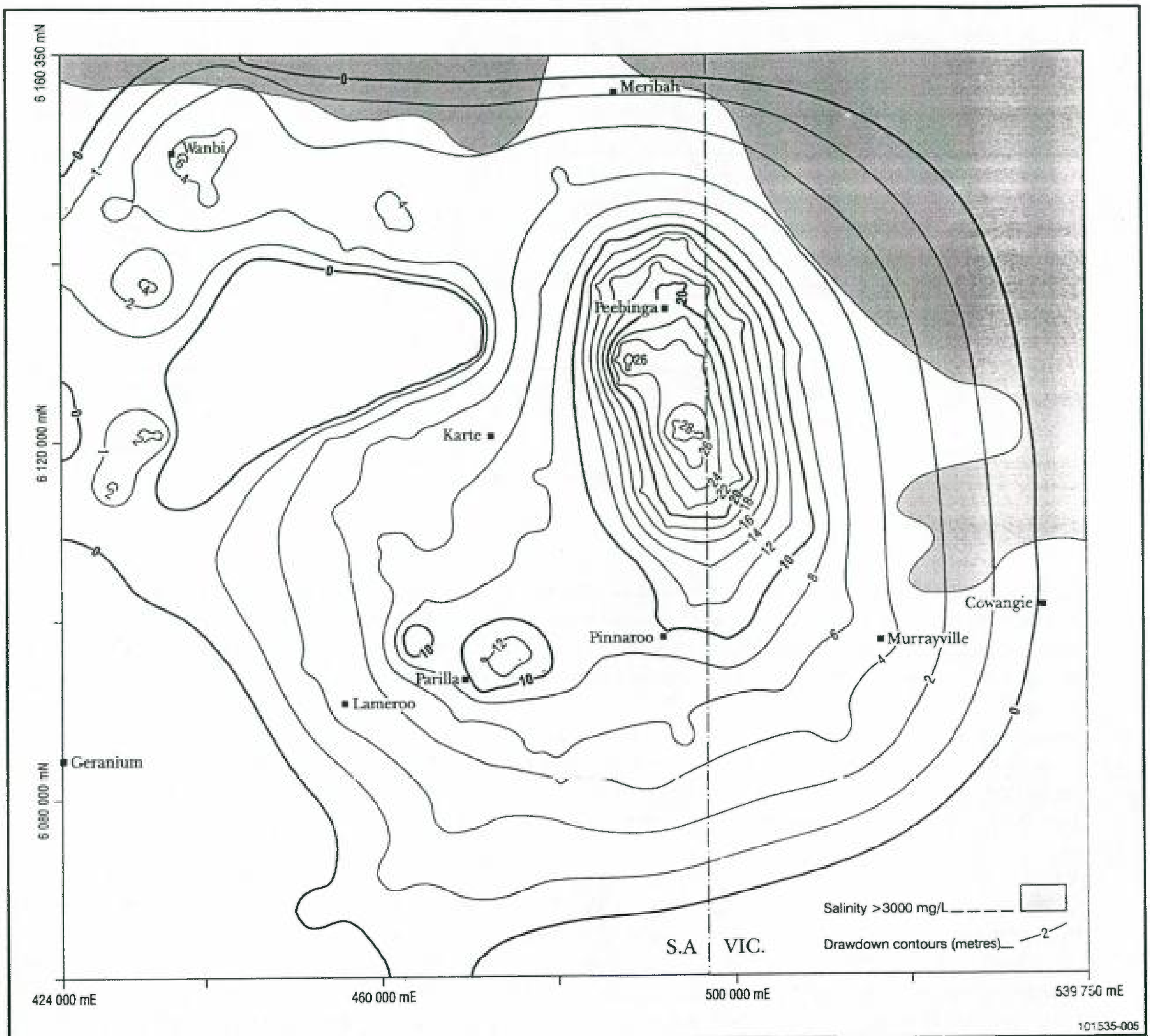


Figure 8 Maximum drawdown contours for Scenario 3 for the year 2024

UPWARD LEAKAGE

A significant contribution to the MGL aquifer by upward leakage from the Renmark Group was necessary to enable calibration of the model. This is consistent with the hydrogeological evidence such as head difference contours and the presence of sand aquifers up to 30 m thick within the upper Renmark Group (Olney Formation). This contribution from upward leakage has not been considered in determining the sustainable yield for the MGL aquifer in previous management strategies. Figure 7 shows the increased upward leakage from the Renmark Group confined aquifer to the MGL aquifer due to the greater head difference after 35 years of pumping from the MGL aquifer.

SALINITY CHANGES

Based on the prediction flow model of Scenario 3, the solute transport model (MT3D) was undertaken to predict the salinity changes. The initial concentrations were taken from the salinity contours in the Murray Basin Hydrogeological Map Series and were represented as different salinity zones. The horizontal and vertical dispersivity were calculated by considering the liner velocity of the water flow and the model grid size. The molecular diffusion coefficient was obtained from existing references.

The results show virtually no change in salinity in the MGL aquifer due to lateral inflows induced by irrigation drawdowns. This is due to the resultant low potentiometric surface gradients which change a northerly flow of

0.5 m/year to a westerly flow of 2 – 3 m/year toward the pumping centre. The 3000 mg/L contour is 10 km from the nearest irrigation bore.

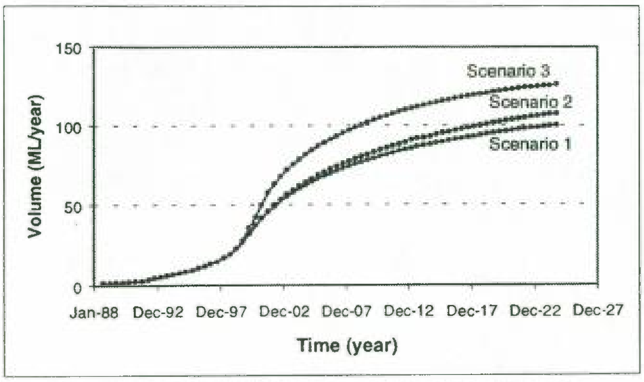


Figure 6 Predicted downward leakage from Pliocene Sands to MGL

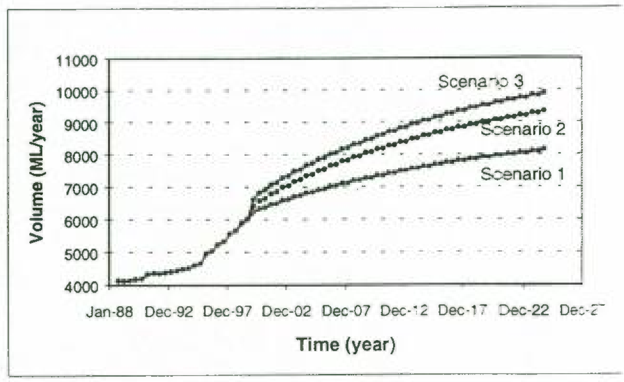


Figure 7 Predicted upward leakage from Renmark Group to MGL

SENSITIVITY ANALYSIS

As mentioned earlier, a steady-state solution was performed by adjusting hydraulic parameters (hydraulic conductivities, recharge to the top layer, inter-aquifer leakage) and boundary conditions. The water level in the Pliocene Sands aquifer was found to be very sensitive to the recharge rate, with the water level in MGL aquifer very sensitive to hydraulic conductivities and upward leakage from Renmark Group aquifer.

Sensitivity analyses were also undertaken during the transient calibration and model validation processes. The results show that the trend of water level decline and the maximum drawdown in each individual pumping season are mostly controlled by the specific yield and specific storage coefficient in the surrounding area. The water level recovery after the pumping season is dominated by upward leakage from the Renmark Group aquifer.

Additional sensitivity tests were carried out on the vertical hydraulic conductvities of both confining layers and the resultant effects on water level fluctuations in the MGL aquifer. The results shows that varying the K_v of the Bookpurnong Beds by a factor of 10 upwards or downwards makes no difference to the MGL response from downward leakage. This low sensitivity is probably due to the thickness of the confining layer and the small head difference with the Pliocene Sands aquifer which is driving the leakage. Similar variations in the K_v for the Ettrick Formation by a factor of five showed a moderate sensitivity to upward leakage, with the recovery in the MGL aquifer after pumping being most affected. An increase in K_v results in too much recovery by 1 – 2 m, and vice versa.

LIMITATIONS

The main limitation is probably the lack of accurate data on pumping volumes. These uncertainties may have led to a few calibration results not matching very well to observed water levels. The introduction of metering will overcome this problem. The water level and salinity data for the Pliocene Sands aquifer is sparse, making calibrations with recharge difficult, and calculations of downward leakage impacts uncertain. There are no suitable observation bores in the Renmark Group in the vicinity of the areas of maximum drawdown in the MGL aquifer to measure the effect of the upward leakage in the long term.

FURTHER WORK

The model will be re-calibrated when additional information becomes available. This will include the more widespread monitoring information of the impacts of the 1998/99 irrigation season on both the MGL and Pliocene Sands aquifer. In addition, salinity sampling and monitoring of the recently completed Pliocene Sands observation wells will enable more accurate prediction of any salinity increases due to downward leakage by providing actual salinity values and head differences.

Future modelling will also validate the salinity impacts of lateral groundwater movement within the MGL aquifer and upward leakage from the Renmark Group. Impacts of pumping from the Renmark Group confined aquifer will be tested for decreasing upward leakage and increasing drawdowns in the overlying MGL aquifer.

SUMMARY

A finite difference groundwater flow (MODFLOW) model was developed for irrigation areas in the Mallee Region with the assistance of the Visual MODFLOW package.

The model was calibrated for steady-state conditions over the entire model area, with a transient calibration undertaken in areas where the local pumping data and observation bore hydrographs were available. The model was also validated by using estimated pumping volumes and water level data over the past ten years. The high accuracy of the calibration and validation exercises can be evaluated by comparing the calculated and observed values for both the hydrographs and water level contours.

So far, this model has been used as a prediction model for various pumping scenarios and it will be updated at regular intervals when additional observation well data becomes available. It is a very useful management tool for ensuring the sustainable development of groundwater resources in the area, by predicting the impacts of different pumping scenarios (both location and pumping rate) on groundwater levels and salinities.

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Characteristics of the Pilliga Sandstone Aquifer in the Coonamble Embayment

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ABSTRACT

The 1995 cap on surface water diversions in the Murray-Darling Basin (MDB) has increased the interest in groundwater resources. In the Macquarie catchment, within the Coonamble Embayment, the recent two year drought ending in 1996 prompted questions about groundwater usage for irrigation purposes. The freshwater aquifers of the Coonamble Embayment, while being of interest locally are also managed in the context of the Great Artesian Basin (GAB) and so present complex issues in natural resource management.

The Pilliga Sandstone is the only freshwater artesian aquifer in the GAB. Its characteristics in the western Coonamble Embayment are yet to be fully described, despite its importance as a freshwater source. A geologic model constructed from well log and interpolated using the ANUDEM package reveals the complex geometry within which the aquifer exists. Hydrogeochemistry is used to identify flow patterns, possible recharge/discharge areas, and to better define the Pilliga Sandstone aquifer.

Existing groundwater chemistry data has been reviewed and a discrete database representative of the Pilliga Sandstone aquifer has been produced. ANUDEM was then used to model the spatial distribution of major ions for the lower Macquarie catchment region. In combination with the geologic model, the hydrogeochemistry has enhanced the understanding of flow characteristics of the aquifer. Possible discharge sites, close to the Macquarie Marshes appear to exist.

INTRODUCTION

The 1994 COAG Water Reforms and the 1995 cap on surface water diversions in the MDB have increased the importance of water management in the MDB in general and in the Macquarie River catchment in particular. Water allocation and use is a major issue particularly when a large wetland like the RAMSAR-listed Macquarie Marshes also receives its own surface water allocation. The area under irrigated cotton production within the Macquarie catchment is estimated to have increased by 400% from 1981 to 1992 and now dominates the farming region [Kingsford and Thomas, 1995]. The demands on both surface and groundwater resources in the Macquarie catchment continue to increase.

Groundwater resources in the catchment are contained in the unconsolidated sediments, consolidated sandstones, fractured rocks, and basalts of the GAB. Mesozoic-age Pilliga Sandstone, Keelindi beds, Drildool beds, and Rolling Downs Group contain groundwater of varying quality and quantity in the GAB. Of these, the Pilliga Sandstone is the only aquifer under pressure, comparably fresh, and with significant capacity. This research focuses on the Pilliga Sandstone aquifer. This aquifer is currently used mainly for town water supplies and stock watering.

Though the Pilliga Sandstone aquifer has been used as a water source since the late 1800's and is the only artesian aquifer in the western Coonamble Embayment, the aquifer has never been fully characterised in terms of its hydraulics and hydrogeochemistry. This research characterises the Pilliga Sandstone aquifer by defining its water budget and sustainable yield.

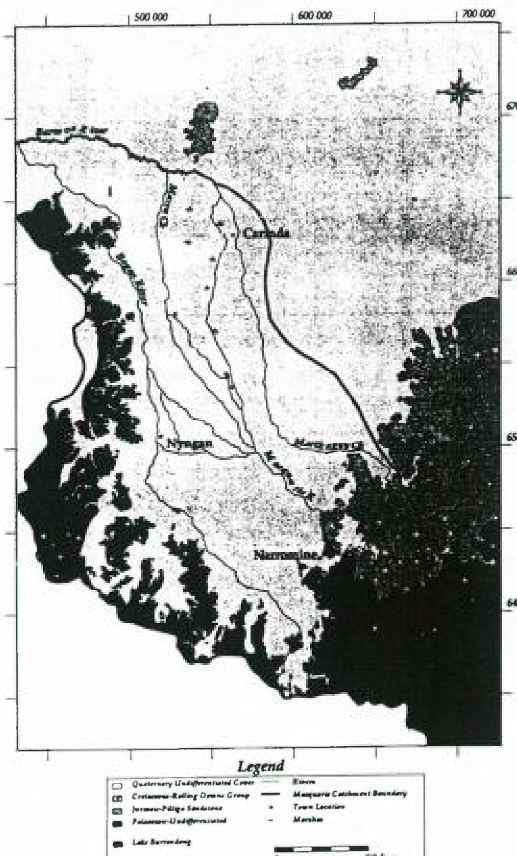
Geologic History

The Macquarie basin is bordered on the south and west by the Lachlan Fold Belt (LFB) [Watkins and Meakin, 1996] (see Figure 1). The deformed, Palaeozoic Lachlan Fold Belt includes the Hervey Group, Milmiland and Little Plains Granites, Mt. Foster Monzonite, Booralee Volcanics, and the Girilambone Group. The younger Mesozoic GAB sequence includes the Garrawilla Volcanics, Purlawaugh Formation, Pilliga Sandstone, Keelindi and Drildool beds, and Rolling Downs Group. The Lachlan Fold Belt is exposed in an arc around the GAB sequence in the lower catchment, with its western extent of outcrop passing along the western flank of the Warrumbungle Mountains, through Dubbo, and then curving north, following the western edge of the Bogan River. It also forms the basement to the GAB as it dips northward, unconformably underlying the GAB in the lower Macquarie catchment.

The Pilliga Sandstone has regions of greater thickness in 'potholes' scattered throughout the region. These may act as areas of stagnant groundwater. Along the Bogan River and south and west of Narromine, there is a comparatively shallow Palaeozoic basement and the absence of Pilliga Sandstone and it is expected that GAB groundwater is less in this region. In this part of the Coonamble Embayment, the Pilliga Sandstone may have been deposited in an alluvial fan environment. Pilliga Sandstone deposition is also influenced by the basement structure in the lower Macquarie catchment. The Pilliga Sandstone is thickest (300m), in the east of the Macquarie catchment and shows as the core of a fan-like projection splayed outwards from the Pilliga Sandstone outcrop near the Warrumbungle Mts. This is called here the Yungundie Fan.

The structural top of the Pilliga Sandstone is deepest in the northeast (-600m). It thins towards the east of the Macquarie catchment and towards the LFB. The elevations of the Pilliga Sandstone confirm that the Sandstone was deposited in the depressions of the Palaeozoic basement.

Figure 1: Geological Base Map



Geophysical Data

Analysis of temperature and neutron geophysical logs offers further information on the Pilliga Sandstone aquifer. Twenty temperature logs and 13 neutron logs were used spanning an area of approximately 50km in a broad NE-SW line in the middle-lower Macquarie catchment [Habermehl, et al., 1997].

To use temperature logs we assume that different aquifers have different temperature waters. Neutron logs reveal the presence of moisture. Finally, linking interpreted gamma ray logs with the neutron and temperature logs allows correlation of hydrogeological areas directly to the geological units.

Geophysical logs indicate that: (1) water is moving vertically in the system at a regional scale and can move through many geological layers at the borehole scale; and (2) deeper water is leaking upwards through all geological units and cooler water is leaking downwards through the Cainozoic layers, Rolling Downs Group, Drildool beds, and Keelindi beds. High moisture areas are present in all geologic units, particularly in the Pilliga Sandstone and along boundaries between the geologic units. Low moisture areas (possibly aquitards/aquifuges) are present in all units particularly in the Cainozoic-age strata and the Rolling Downs Group. With this information plus the assumptions in Table 1, a method was chosen to establish the Pilliga Sandstone geochemical database.

Table 1: Assumptions Used to Define the Pilliga Sandstone Geochemical Database

1. 1970's Bore Chemistry Data: This was the most comprehensive dataset available with good distribution around the catchment.
2. Sufficient Depth: Bore must reach at least to the structural top of the Pilliga Sandstone.
3. Screen Positions: Screen positions must not be unreasonably (< 150m) beyond the structural top of the Pilliga Sandstone.

Analysis of Groundwater Chemistry Data

Examination of the several hundred bore water chemistry samples in the Macquarie catchment in the Department of Land and Water Conservation (DLWC) database for 1890-1980, revealed significant variations. This could be due to several factors, such as data entry, laboratory, or field sampling error, updated laboratory procedures, field sample preparation, or water from different sources. Another significant problem with the chemical database was the identification of the aquifer from which the samples were taken. The DLWC database had changed format over time and this may have resulted in some data loss or alteration.

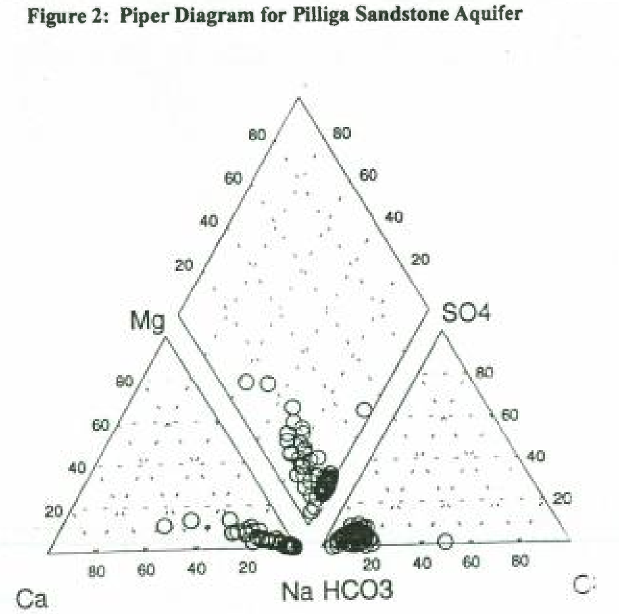
Checking of the quality of the chemistry data was a necessary first step. As a first cut only analyses which had a charge balance of ±5% (in meq/L) were accepted. The goal was to produce a discreet, good quality database of the Pilliga Sandstone aquifer. The basic assumption that, if the bore reached past the structural top of the Pilliga Sandstone then the water chemistry sampled from the bore would be characteristic of the Pilliga Sandstone was considered doubtful. The assumptions used in selecting characteristic Pilliga Sandstone bores are listed in Table 1.

The analysis began with several hundred artesian (flowing) bores from the DLWC database. The method chosen to produce the discreet database was to ensure a cation-anion charge balance and ensure that screen position fell within or close to the Pilliga Sandstone aquifer. All major ions in the bore chemistry of the artesian bores were checked for the cation-anion balance. Only charge balance results of ±5% meq/L of total dissolved ions were accepted. This is based on the fact that errors greater than 10% are likely to be caused by analytical error or an unanalysed ion [Lloyd and Heathcote, 1985]. In this analysis if an ion measurement was missing from the set of analyses for a particular borehole in the database it was not used.

A knowledge of the perforation geometry of a bore is essential in characterising samples (Mazor, 1997). Screen positions and details are included in the DLWC databases for most bores. From bore logs it was noticed that screening (perforations, slots, etc.) was often placed in a bore beyond the first or primary source of water described in the drillers' logs. This raises the possibility of other water entering the screen from overlying geological units. Approximately one hundred bores were checked, with only 78 passing the successive tests. The spatial distribution of the database is reasonable, with bore locations including areas of variability of the Pilliga Sandstone and cover an area progressing away from possible recharge locations in the east to the Pilliga Sandstone areal limit in the west.

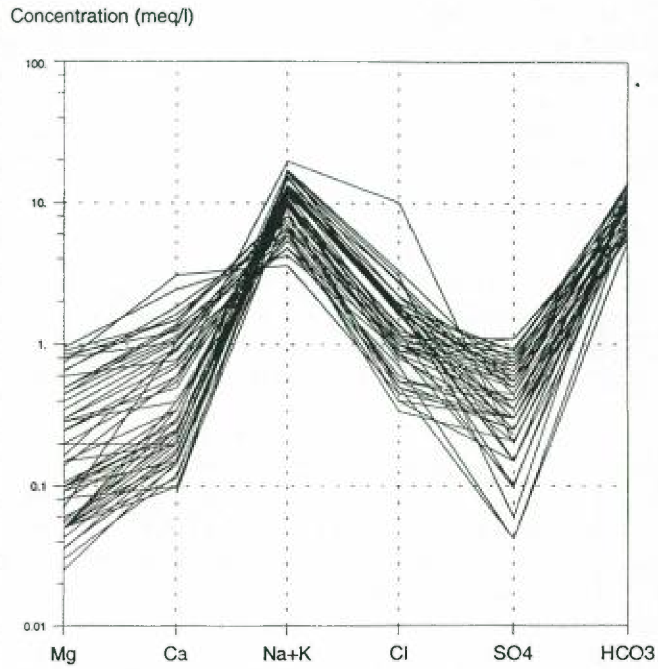
The geochemical analysis of the Pilliga Sandstone groundwater included: (1) Analysis of major ions by the aqueous geochemical plotting software, AquaChem [Calmbach and Waterloo Hydrologic, 1997]; and (2) Spatial analysis with ANUDEM [Hutchinson, 1997] of major ions and groundwater temperature to create distribution maps. AquaChem Version 3.6 [Calmbach and Waterloo Hydrologic, 1997] was used for plotting Schoeller and Piper diagrams and generating statistics for the geochemistry of the Pilliga Sandstone database.

Figure 2: Piper Diagram for Pilliga Sandstone Aquifer



Piper diagrams show possibilities of mixing of waters and evolutionary pathways [Piper, 1944]. The Piper diagram for the Pilliga Sandstone (Figure 2), shows a vector running outwards from the Na apex, and a group of bores (except for one) clustered in the HCO_3 apex. This is consistent with the observation that GAB waters are Na-HCO_3 waters [Habermehl, 1983]. The bores furthest from the Na apex are bores #4158 and #8213, with bore #4158 being the bore point furthest away. Bores #4158 and #8213 are in the area between the probable recharge area in the Warrumbungle Mts. and the eastern Macquarie catchment boundary. This appears to indicate rapid ion exchange between recharge water and clays. The bores further away from the recharge area (closer to the Piper diagram Na apex) have higher sodium. This may indicate upward or downward leakage of marine-based formation waters. One of the bores near the Na apex is #8063. This bore is located between the Macquarie and Marthaguy Rivers, further away from the recharge area than either #4158 and #8213.

Figure 3: Schoeller Diagram for Pilliga Sandstone Aquifer



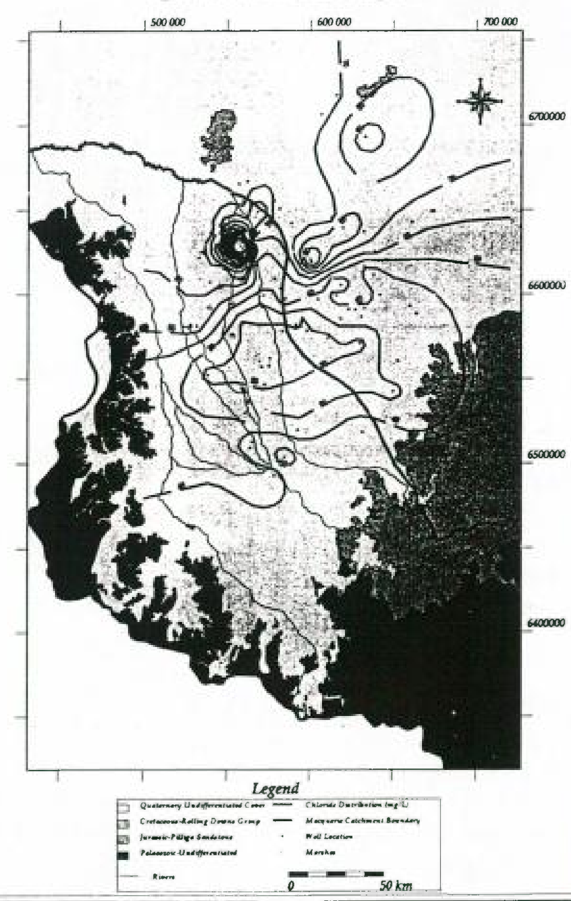
Bicarbonate may be generated either by the reaction of water with $CaCO_3$ or by microbial reduction processes. The current hypothesis is that most HCO_3 in the GAB is due to reduction processes. Bores have relatively low chloride and low sulphate. Further detailed examination of groundwater composition is underway.

The Schoeller Diagram, also called a fingerprint diagram, is another way to graphically represent the chemical data [Schoeller, 1954]. From the diagram for the Pilliga Sandstone geochemical database, (Figure 3), it is clear that the waters vary compositionally but that the waters are Na-HCO_3 dominant. The ions Ca, Mg, and SO_4 show considerable variation. The different patterns reflect the evolutionary stages of the Pilliga Sandstone database. This supports the vectors showing on the Piper diagram, which essentially shows different stages of ion exchange.

The analysis of how major ions and compounds are distributed in the aquifer help to interpret the system mechanics and reveals flow paths and potential recharge areas. Spatial analysis by ANUDEM [Hutchinson, 1997] of cations and anions has been completed and chloride (Cl) is shown here.

Chloride values are highest for the Pilliga Sandstone in the area of the northern Macquarie Marshes (Figure 4). They are lower (40mg/L) towards the area of recharge (Warrumbungle Mts.) and in the southern Macquarie Marshes (20mg/L). The most rapid change in concentration appears to be in the northern Macquarie Marshes. Chloride also increases from 20 to 40-50mg/L in the southern, lower catchment.

Figure 4: Chloride Distribution (mg/L)

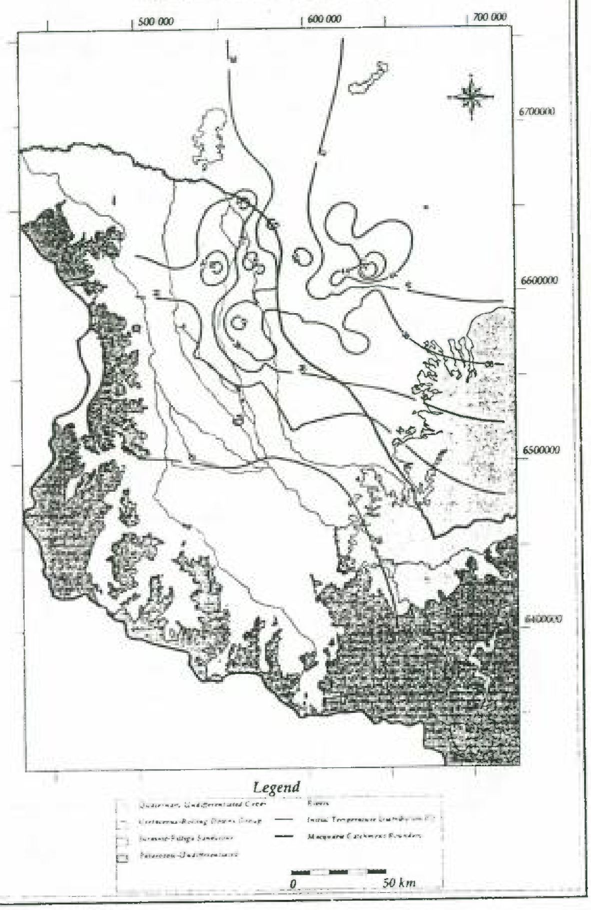


Chloride concentrations normally increase down the hydraulic gradient and with groundwater flow and residence and as such it is an excellent indicator of flow direction and preferential permeability conditions and determining possible mixing [Lloyd and Heathcote, 1985]. Flow lines, if drawn on Figure 4 would point mostly going towards the north or northwest away from the area of recharge, towards the Macquarie Marshes.

Groundwater temperature is important for obtaining the depth of groundwater circulation [Mazor, 1997]. Temperature information was extracted from DLWC microfiche records that contained data for selected artesian bores around the lower Macquarie catchment. The highest initial (first sample taken from the bore) temperature in the database is >58°C with the lowest initial temperature being <20°C. The average initial temperature is 35.6°C.

The highest temperature (>50°C) is located east of the Macquarie catchment boundary (Figure 5). The largest gradient in temperature over geographic distance is found in a north-south line, through the Macquarie Marshes. The lowest temperatures are towards the western and southern edges of the Pilliga Sandstone (<30°C). Lower temperatures ought to be found in a more shallow layer of Pilliga Sandstone. Higher temperatures would be expected in the area where Pilliga Sandstone is at its deepest. An anomaly (<40°C water in an area of <30°C water) is situated in the southern Macquarie Marshes. The Pilliga Sandstone temperature distribution maps have a characteristic pockmarked appearance. The structure in the basement rocks may be affecting temperature of the Pilliga Sandstone aquifer in particular areas. Four spots stand out; (1) east of the catchment boundary; (2) and (3) in the Macquarie Marshes near the Macquarie River; and (4) in the curved area of Marthaguy Creek. There appears to be a correlation with basement structure.

Figure 5: Temperature Distribution (C)



The average borehole geothermal gradient in the Macquarie catchment ranged from 13.59°C/km to 63.79°C/km with an average gradient of 37.12°C/km. On average, temperature increases at 25°C for each km of depth (Freeze and Cherry, 1979), but this may be affected by: (1) variations in thermal conductivity between formations; (2) volcanic intrusives; or (3) spatial redistribution of heat by flowing groundwater. In the GAB, the average gradient has been found to be 48°C/km, higher than the average of 33°C/km, probably due to higher radioactivity in the Precambrian shield [Polak and Horsfall, 1979].

CONCLUSION AND FUTURE WORK

This work has identified bores from the 1970's whose screen depths penetrate the Pilliga Sandstone aquifer. Using these bores a spatial database has been assembled for water quality in the aquifer. In keeping with findings elsewhere in the GAB the waters in the Macquarie catchment (Pilliga Sandstone) are essentially Na-HCO_3 dominant. A Piper diagram of the Pilliga Sandstone water reveals evolutionary trends in the data with Ca and Mg moving towards Na as sample locations move away from the recharge area.

The complex patterns of ion distribution are consistent in general terms with the structure deduced for the Pilliga Sandstone aquifer and with temperature logs. These reveal pools of relatively stagnant water and areas of flowing waters. Intriguing anomalies occur in the vicinity of the Macquarie Marshes and should be investigated further.

Future work for this research includes further defining the hydrodynamic parameters of the Pilliga Sandstone aquifer and a Pilliga Sandstone water budget. These will help lead to an application of the work by building a new management framework for the Pilliga Sandstone in the Coonamble Embayment.

Acknowledgments

We would like to acknowledge gratefully the contributions from DLWC Dubbo and Parramatta offices for contributing data and allowing access to databases. The Land and Water Division of BRS, particularly Dr. Jim Ferguson (ret.) and Dr. Bruce Radke were helpful in discussions of the broader hydrogeochemical model in the Great Artesian Basin.

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Hydrogeology of the Jemalong and Wyldes Plains Irrigation District and Lake Cowal Aquifer Systems, Lachlan Catchment, NSW

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INTRODUCTION

The Lachlan Catchment is the third largest catchment in the Murray-Darling Basin and is situated in the central west of New South Wales. In recent years, there has been an increasing incidence of salinity in the catchment. The Jemalong and Wyldes Plains Irrigation District (JWPID) which is located between Forbes and Condobolin is an example, where the district is threatened with rising watertable and salinisation. Several studies have been undertaken in the area with conflicting conclusions about the contribution of irrigation to groundwater recharge and salinisation processes. However, estimates show that irrigation water adds 49 td^{-1} of salts to the district. Moderate to severe salinisation in the district has already occurred on the central Warroo, Bogandillon Swamp and isolated sections along the irrigation channel system. The DLWC (1997) estimated that around 50% of the dryland area and 4% of the irrigated lands in the district had electrical conductivity (EC) levels greater than 2,000 $\mu S\ cm^{-1}$ in 1995. Management of this irrigation area and salinisation occurring in its western part requires that the contributions of all processes impacting on the groundwater dynamics and salinisation be clearly identified.

This paper describes the progress made so far in a PhD research project which aims to investigate the groundwater systems and salinisation processes in the JWPID, Lake Cowal and Bland Creek sub-catchment and to predict potential areas of salinisation. Specific objectives of the research are to: (1) investigate the hydrogeological features of the aquifer system in the study area; (2) investigate the interaction of the aquifer system with the Lachlan River, Lake Cowal, Bland Creek and the irrigation system; (3) investigate the dominant salinisation processes; (4) develop and calibrate a numerical model of the aquifer systems; (5) use the calibrated model to examine watertable rise, interactions between aquifers, surface water and the irrigation system; and (6) develop the best management strategies for conjunctive use of surface and groundwater resources in order to satisfy various demands in the study area and to minimise the risk of land salinisation.

THE STUDY AREA

The study area includes the JWPID, Lake Cowal and Bland Creek area (Figure 1). Established in 1944, the irrigation district has a total area of 93,123 ha and about 25% of this is capable of being irrigated. Water for the irrigation district is diverted from the Lachlan River at Jemalong Weir and fed by gravity through some 306 km of earthen supply and drainage channels. Lake Cowal, which supports grazing and minor cropping activities when dry, is an ephemeral lake (8 km x 15 km in area) and is primarily fed by flood water from the Lachlan River and drainage from the Bland Creek.

[Figure 1. Geological map of the study area, locations of geological cross-sections and rainfall isohytes.]

CLIMATE

Annual rainfall distribution in the study area decreases from east to west, ranging from 698 mm in Wombat to about 434 mm in Warroo (Figure 1). Monthly rainfall is almost uniformly distributed throughout the year, with relatively higher mean monthly rainfalls occurring during October and January. An approximately 50 years flip flop cyclic pattern of annual rainfall variation has recurred in the study area since the last century. The most widespread changes occurred around 1945-1946. This change is consistent with the changes in the catchment and over much of the south eastern Australia. The average maximum (January) and minimum (July) temperatures are about 32.1°C and 2°C, respectively. The average annual evapotranspiration from 1995 to 1997 at the irrigation district is about 1,387 mm, which is three times higher than the average annual rainfall.

HYDROLOGY

Surface runoff

In general, surface water runoff at JWPID drains in westerly or south westerly directions towards Lake Cowal, Nerang Cowal, Manna Creek and Bogandillon Creek. Flood runoff from the Lachlan River provides a major groundwater recharge for the district and surface water source for Lake Cowal. The flood water flows south-west along the floodway from the flood breakouts on the Lachlan River, through the JWPID. When Lake Cowal is full, floodwater begins to fill the adjoining Lake Nerang Cowal and will eventually overflow to the Lachlan River through the Manna and Bogandillon Creeks. Surface water runoff from Bland Creek and its tributaries also flows into Lake Cowal. Lake Cowal has a storage capacity of 194,000 ML when full (DWR, 1992) and frequency of its partial inundation is approximately 50% (31 out of 64 years from 1930 to 1994).

Streamflow characteristics

The average (1941 to 1995) annual streamflow of the Lachlan River near Jemalong Weir is about 1,204,000 ML. In general, streamflows are high at this station during winter months and low during summer. The average monthly streamflow in August is the highest (228,000 ML), whereas January has the lowest streamflow (53,000 ML). Streamflow at this station is continuous due to water release from the headwater reservoir. Diversion of streamflow for irrigation at the Jemalong Weir was about 70,000 ML yr^{-1} from 1986/1987 to 1996/1997.

The Manna Creek has an annual streamflow of about 76,000 ML since 1975. Monthly streamflow is high from July to September, but very little amount of water flows during summer months. In Bland Creek at Morangarell (Figure 1), the average annual streamflow recorded since 1977 is about 50,000 ML. Monthly streamflow distribution is quite similar to the other stations with high flows occurring in winter and low flows during summer. In Bogandillon Creek, streamflow is very low. Average annual streamflow in 1994 and 1995 at this station was 780 ML.

Surface water salinity

Between 1970 to 1981, the measured EC at the Jemalong Weir ranged from 210 to 640 $\mu S\ cm^{-1}$ with a median value of 350 $\mu S\ cm^{-1}$ (Kelly, 1988). Using the average annual diversion of the irrigation water of about 70,000 ML and assuming a mean daily EC of 400 $\mu S\ cm^{-1}$, and a conversion factor of 0.64 it is estimated that the average daily salt load into the irrigation district is about 49 $t\ d^{-1}$.

In Bogandillon Creek, an EC value of 39,000 $\mu S\ cm^{-1}$ has been recorded in October 1994 at Birrack Bridge. In fact, at this section, about 34% of the recorded data are highly saline (above 15,000 $\mu S\ cm^{-1}$) and 52% fall in brackish to saline category (1,500-15,000 $\mu S\ cm^{-1}$). The remaining 14% have an EC of less than 1,500 $\mu S\ cm^{-1}$. Meanwhile, water in Lake Cowal is fresh (less than 1,500 $\mu S\ cm^{-1}$). Its highest recorded EC is only about 6,000 $\mu S\ cm^{-1}$ and was measured in February 1988.

GEOLOGY

Figure 1 shows the geology of the study area and indicates that a large proportion of the area is covered by unconsolidated (Tertiary and Quaternary) sediments. The stratigraphic units in the study area include: Ordovician (tuff, phyllite, schist, sandstone, latite, siltstone, limestone, andesitic volcanics, basalt, and numerous igneous intrusions); Silurian (granite, sandstone, shale, slate, phyllite, conglomerate, acid lava and limestone); Devonian (massive sandstone, conglomerate, siltstone, granite, granodiorite, andesite, rhyolite, tuff, limestone and shale); Tertiary (basalt , laterite, conglomerate) and Quaternary (gravel, silt and clay).

Two distinct groups of unconsolidated sediments infill the paleochannels and floodplain of the study area, namely Lachlan and Cowra Formations (Williamson, 1986). The Lachlan Formation is the older and deeper unit consisting of clays, silts, sands and gravels in varying admixtures. This formation is confined in the paleochannels and is believed to be of Pliocene age. The Cowra Formation, deposited since the Pleistocene, overlies unconformably the Lachlan Formation and basement rocks. It consists of moderately well sorted sand and gravel with interbedded clays. To investigate the subsurface geology of the study area, 9 cross-sections have been prepared. Locations of these cross-sections (A-A' to I-I') are shown on Figure 1, while Figure 2 shows the profile of the selected F-F' cross-section. In general, the density of the sand and gravel lenses increases from the southern part of the Bland Creek subcatchment towards the Lachlan River.

[Figure 2. Geological cross-section F-F'.]]

HYDROGEOLOGY

Groundwater in the study area occurs in the unconsolidated sediments and in the fractured rocks. However, groundwater yields from fractured rocks are generally less than 1 litre per second. Unconsolidated sediments form the main aquifer consisting of Lachlan and Cowra Formations. In hydrogeological terms, these two formations act in combination as a watertable aquifer, however on a local scale individual sand and gravel lenses display their own characteristics and behaviour (as local confined or semi-confined aquifers).

Groundwater monitoring system

Groundwater level monitoring in the irrigation district started in 1944 at a limited number of sites. However, more monitoring wells were installed in 1968 to examine the watertable movements and groundwater quality of the entire irrigation district (Kelly, 1988). More than 100 single or nested observation wells have been installed in the irrigation district since 1968. Around Lake Cowal, about 20 observation wells have been installed since 1994 to monitor groundwater level and electrical conductivity. Observation wells were also installed by the North Mining Limited around the proposed Lake Cowal Gold Mine in 1994. Aside from the intensive groundwater level monitoring in the irrigation district, the Department of Land and Water Conservation has maintained regional observation bores at 75 sites in the study area.

Watertable rise

The watertable in the irrigation district has been rising since 1944. Table 1 shows that the watertable has risen by more than 8 m over the period of 1944 to 1992. A number of remarks regarding the estimates shown in Table 1 are as follows:

- The estimated watertable rise over the period of 1944-1968 is higher than the estimate (4 m) provided by Kelly (1988) over the same period. Unfortunately data were not available to verify these estimates.
- The low estimate of watertable rise over the period of 1969-1988 is due to a major gap (more than 7 years) in the data.
- The estimate of watertable rise over the period of 1969-1992 is consistent with our estimate over the same period.

Table 1. Estimated watertable rise in the JWPID from 1944 to 1992.

Period	No. of years	Watertable rise in:	
		m	mm yr^{-1}
1944-1968	24	6	250
1969-1983	14	0.07	5
1983-1988	5	0.8	160
1988-1992	4	1.5	375
Total	47	8.37	178

Source of data: Williams (1993).

Watertable aquifer

Lachlan River has clearly a dominant influence on the watertable aquifer in the irrigation district along its course in a northwesterly direction from the Jemalong Gap (Figure 3). The River is recharging the aquifer in the upstream areas as indicated by convex contour in the vicinity of the river course. Further downstream, shape of the contours indicates that groundwater discharges to the river. Within the irrigation district, the response of the aquifer to the river stage declines with the distance away from the river. The watertable near the river only responds well during a long duration high river stage. Also evident is a dominant groundwater mound which coincides with the position of the Warroo Channel and is generating a significant component of groundwater flow to the west and southwest. This confirms Anderson *et al.*'s (1993) suggestion that the mound has been developed by channel seepage over the years. The most significant feature of Figure 3 is the groundwater movement towards Lake Cowal. It is believed that in the past, groundwater from the Bland Creek sub-catchment had a natural gradient towards the Lachlan River. It appears that the direction of the groundwater flow has been reversed and that the Bland Creek sub-catchment has a much lower contribution to the groundwater system of the JWPID. Figure 4 shows the depth to watertable in January 1995 depicting a large area in the JWPID with watertable of 2 m below the ground surface. These areas with shallow watertable coincide with salt-affected areas.

[Figure 3. Watertable elevation map of the JWPID in January1995.]

[Figure 4. Watertable depth map of the JWPID in January 1995.]

Groundwater in sand and gravel lenses

Piezometric analysis of the groundwater system within the sand and gravel lenses is complex because of their structure and irregular distribution. Piezometric heads appear to be influenced by the thickness of the individual lenses, and the hydraulic properties of the clay layers that separate the lenses. Because the lenses are discontinuous, it is not appropriate to prepare a piezometric map using data from individual lenses.

Hydrodynamic parameters

Pumping tests carried out in the JWPID indicate that transmissivity values range from 60 to 300 $m^2\ d^{-1}$. Horizontal hydraulic conductivity values vary from 7.5 to 30 $m\ d^{-1}$. Storativity at the JWPID also varies from site to site, ranging from 0.0025 to 0.004. It should be noted that the low values of storativity (which are not representative of a watertable aquifer) are due to the fact that pumping tests were undertaken in the sand lenses.

Several DLWC investigation bores around the study area have been pump tested. The estimated transmissivity values range from 113 to 780 $m^2\ d^{-1}$ for the Cowra Formation and 587 to 1160 $m^2\ d^{-1}$ for the Lachlan Formation, respectively. The computed hydraulic conductivity of the respective formations are of the same order of magnitude, from 0.8×10^{-3} to 2.7×10^{-3} $m\ d^{-1}$ for the Cowra Formation and 1.7×10^{-3} to 4.5×10^{-3} $m\ d^{-1}$ for the Lachlan Formation.

Salinity of the watertable

About 120 shallow observation wells have EC records since 1968, with an additional 19 and 22 observation wells in 1983 and 1994, respectively. EC values near the Lachlan River are generally low (Figure 5). A large area of very high groundwater salinity stretches south-westwards towards Lake Nerang Cowal. In fact, some observation wells have EC values of over 40,000 $\mu S\ cm^{-1}$.

[Figure 5. Groundwater electrical conductivity map of the JWPID in January 1997.]

Salinity of the sand and gravel lenses

Electrical conductivity measurements at the regional groundwater bores were taken mostly during the time when groundwater investigation commenced in 1968. Although the historical data for these groundwater bores are very limited (only three measurements of EC have been taken since the start of the investigation), EC values clearly vary with depth. In general, EC is lower in the deeper lenses (>50 m below the ground surface) with a range of 190 to 48,700 $\mu S\ cm^{-1}$ and an average of about 3,700 $\mu S\ cm^{-1}$. In the shallower lenses (<50 m below the ground surface) EC ranges from 195 to 60,000 $\mu S\ cm^{-1}$ and averages 5,700 $\mu S\ cm^{-1}$. However, there are some sites where the EC of the deeper lenses is higher than that in the shallower lenses.

In the Lake Cowal area, EC is in the range of 32,400 to 72,000 $\mu S\ cm^{-1}$,which is significantly more saline than the Lake water itself. However, the Lake sits on a 7 to 10 m thick laterally continuous clay layer with very low vertical hydraulic conductivity (0.00077 to 0.000027 $m\ d^{-1}$) which protects the lake from the underlying aquifer.

MODEL DEVELOPMENT

A three-dimensional groundwater flow model covering the unconsolidated sediments of the JWPID and Lake Cowal area is being developed using MODFLOW. The modelled area extends from the Lachlan River in the north to Lake Cowal in the south. The primary objective of this model is to develop the best management strategies for conjunctive use of surface and groundwater resources in order to satisfy various demands in the study area and to minimise the risk of land salinisation.

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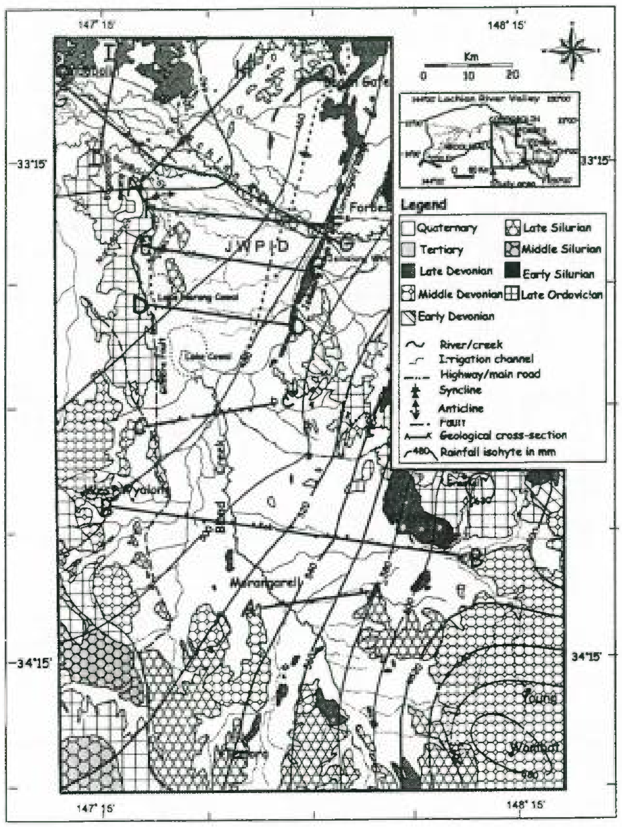
147° 15'
148° 15'
Km
0 10 20
Lochlan River Valley
-33°15'
33°15'
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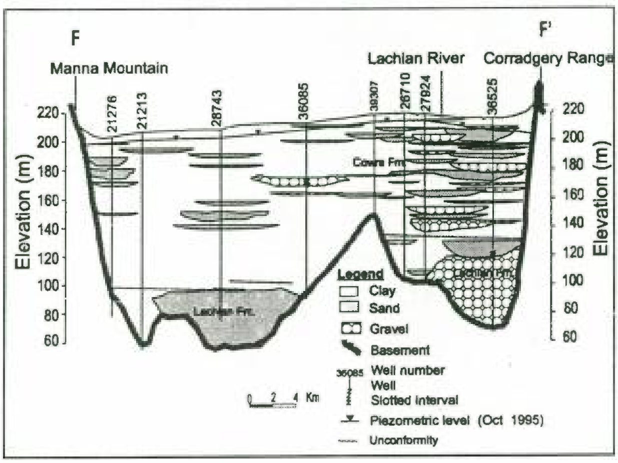
34°15'
Forbes
JWPID
West Wyalong
Marangarell
Young
Wombat
Legend
Quaternary
Tertiary
Late Devonian
Middle Devonian
Early Devonian
Late Silurian
Middle Silurian
Early Silurian
Late Ordovician
River/creek
Irrigation channel
Highway/main road
Syncline
Anticline
Fault
Geological cross-section
Rainfall isohyte in mm

Figure 1.


F
F'
Manna Mountain
Lachlan River
Corradgery Range
21276
21213
28743
36085
39307
26710
27824
36525
Elevation (m)
220
200
180
160
140
120
100
80
60
Cowra Fm
Lachlan Fm.
Legend
Clay
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36085 Well number
Well
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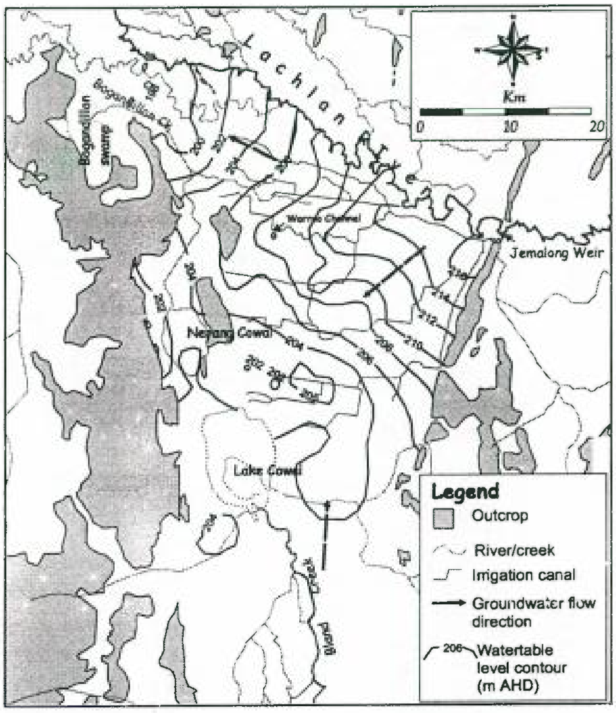
Piezometric level (Oct 1995)
Unconformity
0 2 4 Km

Figure 2.


Lachlan River
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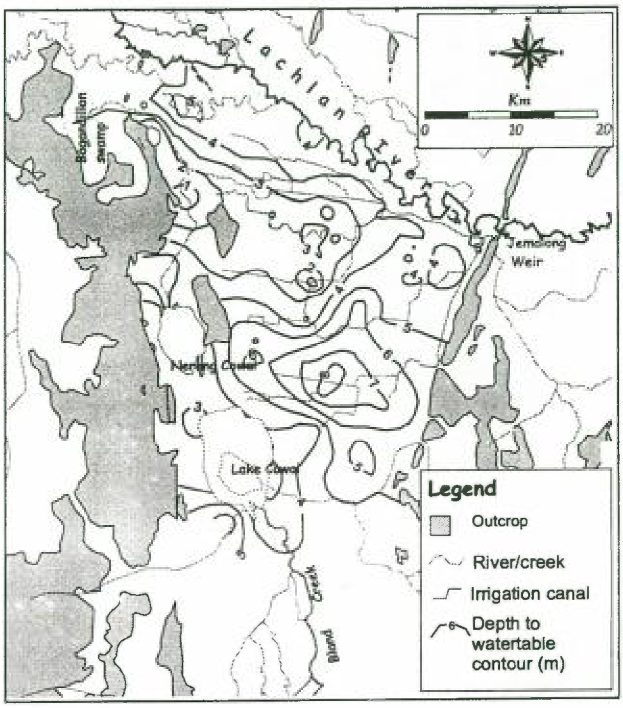
Bogandillan Ck
Jemalong Weir
Nerang Cowal
Lake Cowal
Bland Creek
Legend
Outcrop
River/creek
Irrigation canal
Groundwater flow direction
206 Watertable level contour (m AHD)

Figure 3.


Lachlan River
Bogandillan swamp
Jemalong Weir
Lake Cowal
Bland Creek
Km
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Legend
Outcrop
River/creek
Irrigation canal
Depth to watertable contour (m)

Figure 4.

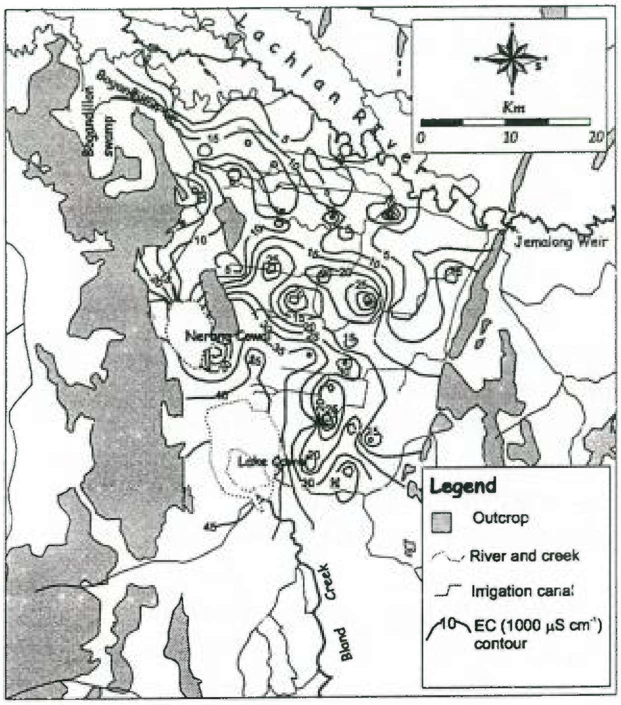
Lachlan River
Bogandillon swamp
Km
0
10
20
Jemalong Weir
Lake Cowal
Legend
Outcrop
River and creek
Irrigation canal
EC (1000 μS cm⁻¹) contour
Bland Creek

Figure 5

Ecologically Sustainable Opportunities for the Lower Murray Darling Area

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ABSTRACT

The Murray Darling Water Management Action Plan is being prepared by the community of the Lower Murray Darling area of South Western New South Wales. The aim is to protect and enhance the natural resources of the Region by implementing a Plan that will bring environmental, economic and social sustainability. Nine of fourteen studies have been completed. This paper focuses on the outcomes of the Groundwater and Land Use studies. These studies have stimulated community participation and identified unexpected potential. That being to enhance floodplain and river health, upgrade irrigation infrastructure and create a sustainable industry whilst expanding local horticultural development away from the river and floodplain. Best Management Practices developed will enhance water use efficiency, reduce groundwater recharge, increase productivity, and create river and floodplain benefits. A National standard for groundwater modelling and a peer review process are required. The Land Use Study identified significant irrigation development with minimum disturbance to current land use. Water will be transferred from low efficiency and high impact enterprises creating environmental benefits. Development will generate economies of scale allowing upgrade of existing infrastructure, further enhancing existing environment and long term sustainability.

THE FULL TEXT OF THIS PAPER WILL BE AVAILABLE AT THE CONFERENCE

ABSTRACTS: POSTERS

Murray-Darling Basin Soil Information Strategy

Elisabeth Bui[1] and Rob Kingham[2]

The Murray-Darling Basin Soil Information Strategy project, a collaborative project between CSIRO Land and Water, the Australian Geological Survey Organisation (AGSO) and the Bureau of Rural Sciences (BRS), funded by the Murray-Darling Basin Commission, has created a spatially referenced overview of the soil and geological resources within the Murray-Darling Basin (MDB).

Until now, the only complete basin-wide soil coverage available for the MDB was the *Atlas of Australian Soils* at a scale of 1:2 000 000. The MDB Soil Information Strategy project has now produced a new soil-landforms map of the MDB that can be used at a scale of 1:250 000.

To generate the soil distribution, all existing digital maps with any soil information were collated regardless of initial purpose for mapping, scale, and age. All soil types described were converted to Principal Profile Forms (Northcote, 1979) to refer to soils in a consistent manner in the new map. Several gaps in detailed soil coverage became apparent when the available digital soil data was mapped, especially in the irrigated areas of northern NSW and in the Murray-Murrumbidgee Riverine plain. Where soil maps were completely lacking, soil distribution had to be determined by extrapolation using computer-based methods rather than fieldwork (due to the project's limited budget for field work as well as time and personnel constraints).

Using existing soils maps, rules integrating the 9-second DEM of Australia, the new lithology map of the MDB (Kingham, 1998), and 4 bands of Landsat MSS were developed to extend the soil mapping to neighbouring unmapped areas. The assumptions underlying rule development are that soil distribution reflects the long-term interactions between terrain variables, geology, and vegetation in landscapes and that the existing soil maps have captured those interactions. These rules were created using the C5.0 data mining software.

Polygon boundaries for the soil-landforms map are derived from the intersection of a relief map of the MDB categorised into 5 classes and the new lithology map of the MDB (Kingham, 1998). In the alluvial plains of the Murray and Darling, the lithology was sub-divided further into sedimentary facies units after Butler et al. (1973) and Watkins and Meakin (1996). A total of 394 map units were created by the intersection of relief, lithology, and sedimentary facies but some unrealistic and small-in-extent combinations were merged with others to give a total of 319 classes. The 3 most probable dominant soil types in the soil-landforms map units are given.

As part of the collaborative MDB Soil Information Strategy project, the BRS Land and Water Sciences Division (formerly part of AGSO) has integrated the geology of the MDB at 1:250 000 scale to produce a new lithological classification. Since the data was compiled in the first instance as an input to identifying soils, a strategy was required to ensure that the database was able to reflect the influence of parent material as one of the dominant soil forming factors. Rather than attempt standardisation over the whole of the MDB, the basin was sub-divided into major tectonic units using the framework devised by Palfreyman (1984). Within these tectonic units, rock types were grouped according to dominant lithology in broad time periods (recognising that specific lithologies can be characteristic of particular stratigraphic periods). This provided a simple classification on a lithostratigraphic basis.

The mapsheets were compiled into a seamless dataset using the ArcInfo GIS. The database incorporates ninety-two 1:250 000 scale mapsheets and combined existing 1:250 000 scale geological, metallogenic and surficial geology mapping from the New South Wales Department of Mineral Resources, the Victorian Department of Natural Resources and Environment, the Queensland Geological Survey and AGSO, with more recent 1:100 000 scale mapping where available.

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Groundwater Availability & Vulnerability Mapping In NSW (Poster Paper)

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The development of groundwater vulnerability maps for the use by land management planners is a key product of the NSW Water Reforms. The maps are designed to assist with the appropriate siting of potentially polluting developments in the less sensitive areas as well as highlighting those areas where significant environmental sensitivity exists and engineered protection would be required to prove a project.

Maps for second order catchments covering the entire state are being developed. Several methodologies were trialled however the selection was biased by the scale and reliability of the least well known data sets. A modified DRASTIC approach was adopted over statistical methods.

The paper defines the methodologies trialled, demonstrates the maps produced to date and outlines the innovative spatial GIS representations which have meshed with the planners and community inherent understanding of the spatial environment that has facilitated the uptake of the complex concept of aquifer vulnerability.