Implications of using Purified Recycled Water as an Adjunct to Groundwater Resources for Irrigation in the Lockyer Valley

Edited by Leif Wolf

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The Urban Water Security Research Alliance (UWSRA) is a $50 million partnership over five years between the Queensland Government, CSIRO's Water for a Healthy Country Flagship, Griffith University and The University of Queensland. The Alliance has been formed to address South East Queensland's emerging urban water issues with a focus on water security and recycling. The program will bring new research capacity to South East Queensland tailored to tackling existing and anticipated future issues to inform the implementation of the Water Strategy.

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Enquiries should be addressed to:

The Urban Water Security Research Alliance
PO Box 15087
CITY EAST QLD 4002

Ph: 07-3247 3005
Email: Sharon.Wakem@qwc.qld.gov.au

Project Leader – Leif Wolf
Karlsruhe Institute of Technology (KIT)
Karlsruhe, Germany

Email: leif.wolf@kit.edu
Email: leif.wolf@csiroalumni.org.au

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Authors (*alphabetical*): Ashley Bleakley[3], Brett Robinson[2], Catherine Moore[1], Craig Henderson[7], Darren Morrow[1], David Rassam[1], Don Begbie[1], Jenny Foley[2], Jerome Arunakumaren[5], Jun Du[8], Leif Wolf[1,6], Malcolm E. Cox[4], Malcolm Hodgen[1], Manuel Grimm[1], Maria Harris[2], Mark Silburn[2], Matthias Raiber[4], Mick Hartcher[1], Rai Kookana[8], Sebastian Most[1], Sreekanth Janardhanan[1], Shreevatsa Kodur[2], Thusitha Gunawardena[2], Tim Ellis[1].

1 CSIRO Land and Water, GPO Box 2583, Brisbane, 4001, QLD, Australia
2 Department of Natural Resources and Mines (DNMR), Resource Assessment and Information, South Region 203 Tor St, Toowoomba Q 4350
3 Department of Natural Resources and Mines (DNMR), Water Services South
4 Queensland University of Technology, School of Earth, Environmental and Biological Sciences, GPO Box 2434, Brisbane, 4001, QLD, Australia.
5 REN Consulting Pty Ltd, Brisbane
6 Karlsruhe Institute of Technology (KIT), Karlsruhe, Germany, leif.wolf@kit.edu; leif.wolf@csiroalumni.org.au
7 Department of Agriculture, Fisheries and Forestry, Horticulture and Forestry Science, Agri-Science Queensland
8 CSIRO Land and Water, Waite Campus, Glen Osmond, SA, Australia

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Deep soil coring in the Lockyer Valley; Photographer: Jenny Foley, DNRM

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FOREWORD

Water is fundamental to our quality of life, to economic growth and to the environment. With its booming economy and growing population, Australia's South East Queensland (SEQ) region faces increasing pressure on its water resources. These pressures are compounded by the impact of climate variability and accelerating climate change.

The Urban Water Security Research Alliance, through targeted, multidisciplinary research initiatives, has been formed to address the region's emerging urban water issues.

As the largest regionally focused urban water research program in Australia, the Alliance is focused on water security and recycling, but will align research where appropriate with other water research programs such as those of other SEQ water agencies, CSIRO's Water for a Healthy Country National Research Flagship, Water Quality Research Australia, eWater CRC and the Water Services Association of Australia (WSAA).

The Alliance is a partnership between the Queensland Government, CSIRO's Water for a Healthy Country National Research Flagship, The University of Queensland and Griffith University. It brings new research capacity to SEQ, tailored to tackling existing and anticipated future risks, assumptions and uncertainties facing water supply strategy. It is a $50 million partnership over five years.

Alliance research is examining fundamental issues necessary to deliver the region's water needs, including:

- ensuring the reliability and safety of recycled water systems.
- advising on infrastructure and technology for the recycling of wastewater and stormwater.
- building scientific knowledge into the management of health and safety risks in the water supply system.
- increasing community confidence in the future of water supply.

This report is part of a series summarising the output from the Urban Water Security Research Alliance. All reports and additional information about the Alliance can be found at http://www.urbanwateralliance.org.au/about.html.

Chris Davis
Chair, Urban Water Security Research Alliance

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EXECUTIVE SUMMARY

This report explores the feasibility of linking a large agricultural area and an extensive alluvial aquifer in the Lockyer Valley (Queensland, Australia) with the technically advanced reuse concept of the Brisbane metropolis where wastewater is treated to high standards for indirect potable reuse and branded as purified recycled water (PRW).

The supply of PRW, or local recycled water, into the Lockyer Valley also provides the opportunity to actively steer the currently overused ecosystem to any desired state. Thus, the Lockyer Valley can become an example of conjunctive use management with an urban area. The Alliance project developed a tiered assessment framework which is also transferable to other areas or other water sources than PRW.

The work focuses on environmental impacts and on understanding of the heavily used natural water resources in the Lockyer Valley under current and future management approaches. Lockyer Creek discharges into the Brisbane River and thus influences the water quality at the inlet of the Mt Crosby water treatment plant, which supplies drinking water to the inhabitants of Brisbane.

A key concern associated with additional irrigation in the Lockyer Valley is salt. Deep soil coring down to 20 m below ground, geophysical investigations, groundwater quality monitoring and numerical modelling confirmed that current salt accumulations in the deeper soil, produced as a result of irrigation with poor quality groundwater, will slowly move towards the groundwater table over the coming decades. Using low salinity PRW (with less than 25 mg/L chloride) for irrigation instead of the low quality groundwater in some parts of the Lockyer would actually reduce salinity issues and create long-term environmental benefit. Analysis of PRW chemistry and tests with soils from the Lockyer Valley showed only a minor potential for soil structural changes when irrigating with PRW, very similar to the risks already induced by rainfall or overhead irrigation with fresh surface water. These impacts are easily managed by the landholders.

The existing Lockyer Valley water resources were also tested for evidence of pharmaceuticals and other trace organic compounds to compare the quality of the existing water with the imported PRW. The screening showed influences of wastewater in Lockyer Creek and the three major local reservoirs, with traces of carbamazepine, acesulfame, metolachlor and caffeine. In this respect, the quality of PRW is significantly better than the existing water quality downstream of Gatton.

Since the supply of imported water to the Lockyer Valley comes with significant costs in terms of energy and financial resources, the demand for this additional water determines the feasibility and the success of such a scheme. The demand, however, depends to a high degree on the availability of the low-priced natural water resources. Thus, PRW demand is highly variable in time and relies on definition of environmental goals.

For this reason, the project undertook a reconnaissance of the hydrogeology of the Lockyer Valley and included the information in a 3D-visualisation system, now available free of charge from QUT, to increase public understanding of the resource. Furthermore, a chain of different models was employed and supported by analysis for remote sensing information.

Soil water balance was calculated with HowLeaky, and travel times of water and salt through the unsaturated zone to the groundwater were calculated using the numerical simulation code HYDRUS. Surface water levels and flows under different climate scenarios simulated with IQQM were provided by the Queensland Government Department of Environment and Resource Management, and the project team customised tools to use these data in the numerical groundwater model developed as part of this project. With the numerical groundwater modelling environment, it is possible to calculate dynamic demand profiles for any specified target water level. As an example, modelling suggested that 9–21 GL/a would be required in order to always maintain average groundwater levels given the historical pumping record. Also, the groundwater model enabled a linear predictive uncertainty analysis which, though approximate, allowed analysis of which additional investigations and monitoring network data would best enhance this predictive reliability. This method could be extended to optimise monitoring networks under the 'best return for investment' principle for different predictions.

Preliminary modelling of changing climate scenarios suggests that low groundwater levels typical of drought situations may occur twice as frequently in a median climate change scenario if pumping rates remain similar. It indicates that under these conditions, the Lockyer Valley system will require an input of water to sustain agricultural practices as they are now. Alternatively, the region will require higher efficiency in irrigation.

Finally, we conclude that while environmental and supply risks are manageable, the cost-effectiveness of various recycled water supplementation options needs to be examined further through hydro-economic modelling. Also, a multi-objective optimisation exercise could detail how PRW could be used most efficiently. This would include options to deliver PRW, like surface water, direct to the farm gate, where best to supplement existing water use, whether to inject directly into the aquifers, and at which times the import of PRW is actually cost-efficient. The respective assessment tools have been developed in this research project and are ready to be used in a collaborative approach with the appropriate stakeholder groups.

Finally the remarks and recommendations of a high level, interdisciplinary stakeholder group meeting held in November 2012 in Brisbane are documented.

1. INTRODUCTION (Leif Wolf and Don Begbie)

1.1 Target Audience

This report targets two audiences: stakeholders who need to have a concise overview of the project outcomes; and technical experts who require detailed information in order to apply the new knowledge for the Lockyer Valley, in South East Queensland (SEQ), or other regions with similar issues worldwide. The fact box below summarises the key information.

FACT BOX

Where: Lockyer Valley, 80km west of Brisbane, SEQ.

What the existing pressures are: Wells falling dry, increasing salinity, unused capacity of water purification plant.

Why we needed a research project: To assess potential risk associated with the supply of purified recycled water (PRW).

Who funded: Urban Water Security Research Alliance.

Who researched: CSIRO Land and Water, DERM, QUT, RPS Consultants.

Who should read this: a) affected landowners and stakeholders; b) decision makers and planning authorities; c) the scientific community; d) other regions with similar problems of importing a treated wastewater resource.

What was done: drilling; soil sampling; crop mapping; remote sensing; groundwater monitoring; hydrochemical analysis of surface water, groundwater, treated waste water effluent and purified recycled water; soil water balance modelling; irrigation demand modelling; salt transport modelling in the unsaturated zone; water balance studies; spatial demand change analysis; numerical groundwater modelling; climate change assessment; conceptual formalisation.

When: 2010–2012.

1.2 Project Background and Scope

The research work presented in this publication was carried out as part of the Urban Water Security Research Alliance (UWSRA) (Figure 1). The Alliance is focused on water security and recycling. For the Lockyer Valley research project, this brought together partners from the Queensland Government, CSIRO's Water for a Healthy Country National Research Flagship and the Queensland University of Technology.

Figure 1: Overview over the UWSRA projects in South East Queensland.

The Lockyer Valley project was first planned in 2008, with the key objective to review existing models and data, assess these for fit-for-purpose (modify if necessary) and use them to evaluate a series of 'what if?' scenarios regarding the use of purified recycled water (PRW) as a substitute and adjunct to groundwater (i.e. conjunctive use) for the irrigation of crops in the Lockyer and Warrill Valleys in SEQ. The project began in February 2010. By July 2010 it was evident that existing models and information were not suitable for an informed decision on the scheme, and a second project phase was granted. The project finished in 2012. In this report, we examine the challenges associated with the large-scale supply of recycled wastewater to a peri-urban agricultural area; demonstrate a suite of models, field and laboratory investigations to address those challenges; and summarise the findings in a tiered assessment framework.

1.3 Indirect Potable Reuse and the Urban–Rural Water Exchange

With growing water scarcity worldwide and decreasing costs for reverse osmosis water treatment, the use of treated wastewater is steadily increasing. The prolonged Australian drought that commenced in 2002 and the agreement between Australia's Commonwealth and State and Territory governments to progress water reform through the National Water Initiative have resulted in many new recycling projects in Australia's capital cities (Radcliffe, 2010).

The technically and socially most challenging concept is the use of highly treated wastewater for drinking purposes, either as direct or indirect potable reuse schemes. Indirect potable reuse (IPR) involves a natural system such as a major surface reservoir or groundwater aquifers as an additional safety barrier. The reviewed literature supports the practice of IPR as a reliable and safe addition to existing drinking water supplies, and it is anticipated that IPR will represent an essential element of sustainable urban water resources management in many more regions of the world in the future (Rodriguez *et al.*, 2009). Despite this, public acceptance of IPR is subject to large variations, depending on perceptions of acute water scarcity and the cost of alternative supplies (e.g. seawater desalination); others would object to IPR simply for aesthetic reasons (Hurlimann, 2010). As such, IPR greatly increases the supply security of large water grids, but remains a non-preferred source for the customer. We propose, therefore, to make maximum use of the treatment facilities, even in times of sufficient natural water availability, by recharging aquifers of surrounding areas to build up subsurface drought buffers. This can occur via direct recharge but also via substitution of current groundwater use. Some forms of indirect potable reuse are not obvious; for example, the release of treated wastewater to the environment or agriculture can add significantly to groundwater recharge, thus introducing fractions of treated wastewater to groundwater used for drinking water purposes, such as documented for the city of Berlin (Massmann *et al.*, 2004), Braunschweig (Ternes *et al.*, 2007) or occurring in Israel (Lahav *et al.*, 2010).

1.4 Purified Recycled Water in South East Queensland

The water supply security of SEQ has been increased recently by the Western Corridor Recycled Water (WCRW) scheme, providing advanced wastewater treatment of effluent from urban areas for potential indirect potable reuse of PRW. With a maximum combined production capacity of 232 million litres of PRW a day, it is the third largest recycled water scheme in the world and the largest in the southern hemisphere (Traves *et al.*, 2008). To ensure drinking water quality, a multiple barrier approach is used that involves source control, tertiary wastewater treatment, advanced water treatment by micro- or ultra-filtration (MF/UF), reverse osmosis (RO), and advanced oxidation, followed by storage in the natural environment and finally, drinking water treatment. Figure 2 depicts the seven safety barriers in a flow diagram in connection with upstream agriculture.

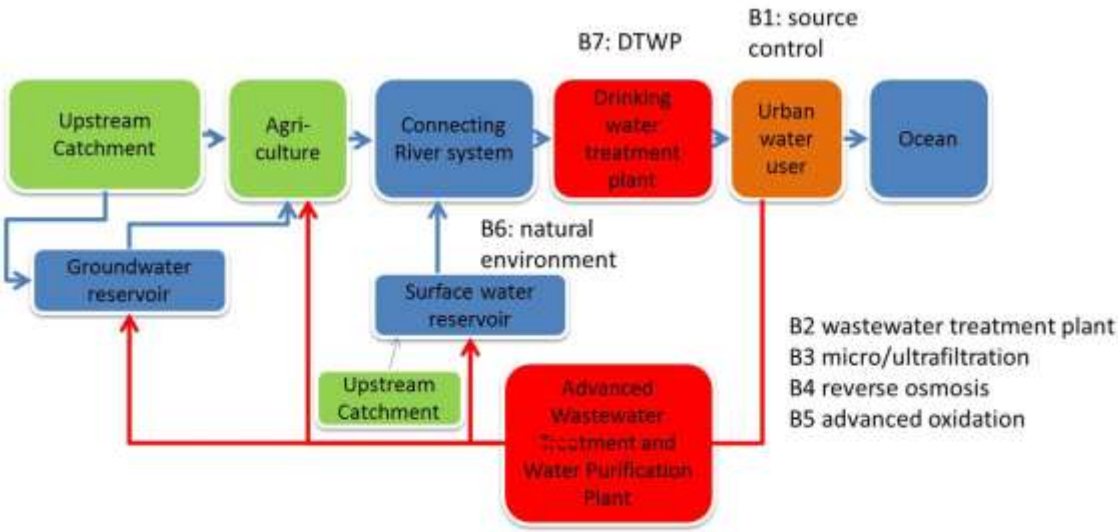

Figure 2: Urban–rural water exchange in the indirect potable reuse scheme for SEQ; abbreviations B1–B7 relate to safety barriers in the indirect potable reuse scheme.

In this scheme, wastewater from the Brisbane metropolis is treated to potable water standard (branded as PRW) with the intention of supplementing Wivenhoe Dam (approximately 1,200 GL capacity). Water from Wivenhoe Dam discharges into the Brisbane River and is tapped further downstream for the drinking water treatment plant at Mt Crosby (Figure 3). The current operating strategy for the SEQ Water Grid is to add PRW to Wivenhoe Dam only when the combined storage capacity of the major SEQ drinking water dams drops below 40%. However, simulations based on long-term historic flow records (1880–2010) indicate that reservoir levels would have dropped below the 40% trigger level only twice with existing and committed infrastructure projects (QWC, 2010). This infrequent requirement leaves significant volumes of PRW available to supplement non-potable uses such as agriculture. Against this background, we analyse the possibilities and implications of using PRW as documented in the following sections.

Figure 3: The Western Corridor Recycled Water (WCRW) scheme (red dotted line) in relation to the Lockyer Valley.

1.5 General Features of the Lockyer Valley

The Lockyer Valley (approximately 80 km west of Brisbane, see Figure 3) is an intensively used agricultural catchment, at times supplying approximately one-third of Queensland's irrigated vegetables (Cox and Wilson, 2005). Besides vegetables, grain and pasture crops are common.

The Lockyer Valley has undergone significant hydrological change over the past 100 years, from a situation with perennial streamflow at the beginning of the twentieth century to intermittent streamflow today. Irrigation bores have been drilled in increasing numbers since the early 1940s, with over 1,500 bores estimated to be currently servicing the Valley (Department of Primary Industries, 1994; KBR, 2003). In selected areas, groundwater table drawdown occurred rapidly and, as a result, 21 weirs were installed in the 1950s for groundwater recharge, storage of water for direct surface water access and management of releases from the dams. During the 1980s, average annual groundwater withdrawal in the Lockyer Valley was estimated to be 46,500 ML, while safe annual yield is estimated to be 27,000 ML (Department of Primary Industries, 1994); clearly, additional water is required to sustain current farming practice.

In the last two decades, including the millennium drought of 1997–2009, water use has reduced significantly, as will be documented later in the report. Still, it is estimated that at least one-third of productive land in the Lockyer Valley has already been withdrawn temporarily from cultivation due to lack of water. Over-extraction of groundwater, particularly in 2007, has exacerbated salinity in some aquifers, as either saline water seeps in from adjacent sandstone areas to replace the higher quality water taken from alluvial aquifers, or more salt is washed down from the unsaturated zone due to irrigation and dryland clearing. In short, the Lockyer Valley suffers from the difficulties of managing a common pool resource (compare with 'the tragedy of the commons' Sarker *et al.*, 2009). Recently, however, three consecutive wet years and the tragic flood events of January 2011 led to a major recharge pulse of the groundwater system. The current groundwater levels are close to long-term maximum and demonstrate the ability to recover in most areas.

1.6 Supply Scenarios for Purified Recycled Water

The discussion on the introduction of PRW comes in a period of changing water allocations and intense political debate. Despite the commitment to ecologically sustainable development (ESD), past norms of water allocation in Australia have created a legacy of over-allocation in many regions (National Water Commission, 2006). The return of all over-allocated or over-used systems to environmentally sustainable levels of extraction was, and is, a key requirement of the National Water Initiative (NWI) in 2004 (Grafton, 2010). The process of implementing a radical policy change to use ESD in water allocation in Australia is ongoing but has an active history of 20 or so years. There are over 190 water allocation plans in Australia, and most of these have reduced the amount of water allocated to growers in any year. This has been met by opposition from some groups, but the more subtle change is that there has been a change in the type of crops grown (McKay, 2011) and increased irrigation efficiency. Under these conditions, introduction of a new high-priced water resource, which will require revisiting the current allocations, faces additional opposition.

The stakeholder consultation in the Lockyer Valley opened up distinctively different positions on the delivery scenarios and the potential management of the PRW supply. Three major supply scenarios were discussed contemporary to the research project:

1. Delivery of PRW to individual farm gates;
2. Delivery of PRW to major reservoirs, with distribution via existing pipes to selected farmers and releases to the creek (see Figure 4a); and
3. Direct injection of PRW into the groundwater via wells.

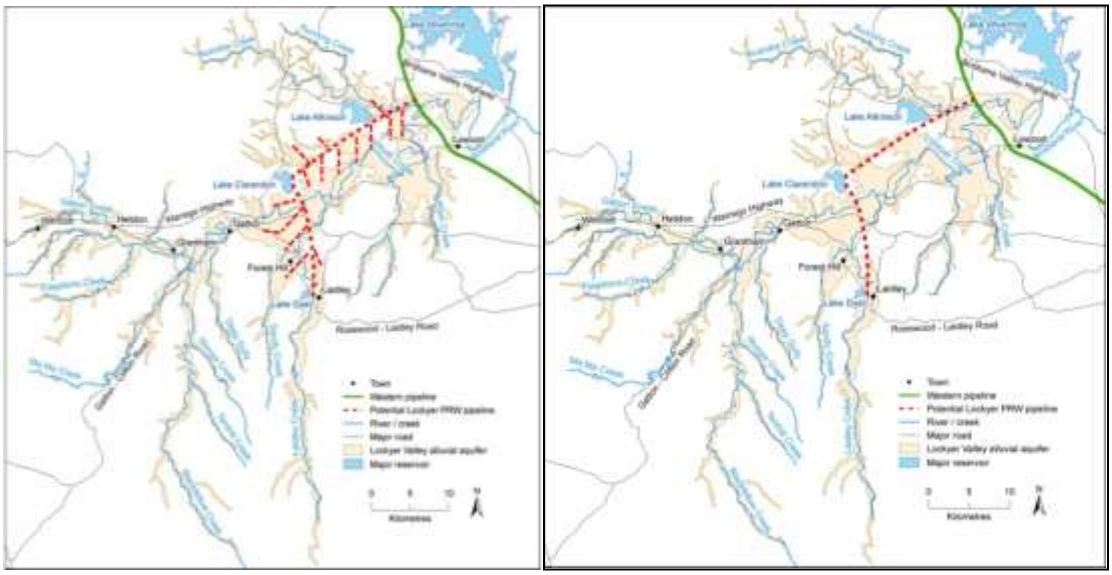

Figure 4: Schematic illustration of two different supply scenarios for PRW to the Lockyer Valley: a) supply scenario 1 - delivery to farm gates in the Lower and Central Lockyer; b) supply scenario 2 - delivery to the three major reservoirs and PRW release to creeks and existing distribution networks.

While supply scenario 2, with the lowest investment costs in distribution infrastructure, appeared as the preferred solution at the beginning of the research project in 2010, stakeholder opinion in 2012 tended towards supply scenario 1. This change is triggered by the realisation of the complexity of common pool resource management, which is a barrier to farmers' acceptance of the PRW supply. Supply scenario 1 provides farmers with increased control of their individual PRW use and, consequently, the costs associated with using the additional resource. Scenario 3, direct injection of PRW into the aquifer, appears as technically feasible (a preliminary feasibility screening according to the National Guidelines for Managed Aquifer Recharge was performed) but opens up the problem of cost recovery, that is, who needs to pay for this addition to a common pool resource?

All three scenarios share the requirement to decide on the appropriate or optimum mix between low-cost natural water resources and high-cost imported water resources. This comes back to the question of defining the sustainable yield for this aquifer system and the water volumes required to meet ecosystem targets. In recognition of the uncertainties associated with the estimation of the sustainable yield and the large climatic variability, we therefore propose to use defined target water levels as a trigger to decide on the necessity of PRW import (Figure 5).

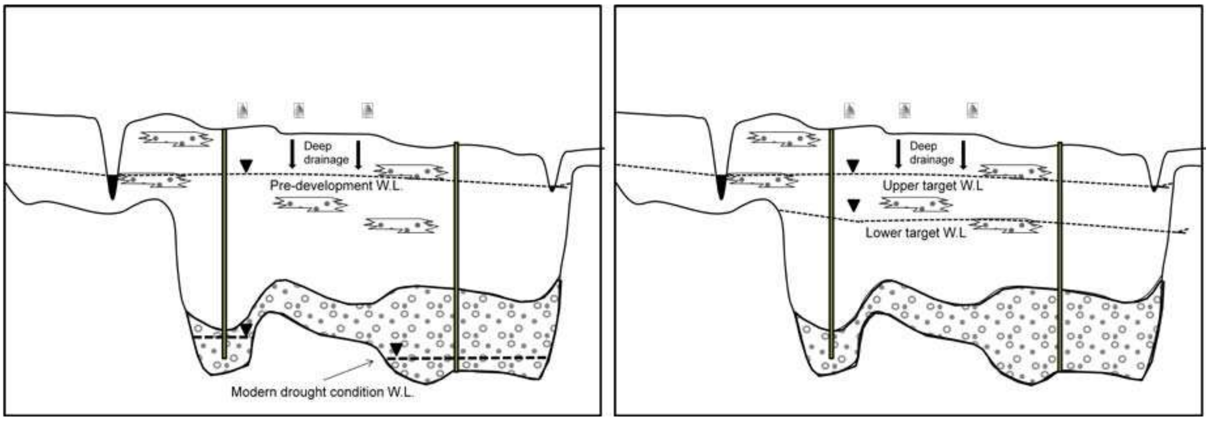

Figure 5: a) Schematic cross-section through the Lockyer Valley, including pre-development groundwater levels and modern drought conditions; b) Future management scenario with time-variant PRW supply to keep groundwater levels between upper and lower target level. Sand- and gravel-dominated aquifer sequences shown with dot texture.

In the unmanaged situation, groundwater levels vary widely between the high pre-development water levels and the low water levels during drought conditions (a). In the managed scenario (b), either through temporary reduction of groundwater pumping or through import of PRW, groundwater levels are kept between an upper and a lower target water level.

1.7 Structure of this Report

The structure of the different chapters of this report represents the build-up of the research program. Chapter 2 describes the general assessment framework that evolved during the project and which now serves to structure the research and ensure its transferability. In Chapter 3, the team led by Malcolm Cox from QUT provides the background knowledge on the hydrogeology of the Lockyer Valley, enriched with recent findings on the hydrochemical classification and evolution of the groundwaters. This chapter also describes the three-dimensional visualisation system that was developed for the Lockyer Valley.

As the first contact of the newly introduced PRW would be the soil, in Chapter 4 we summarise the tests performed with PRW and other local water sources on the soil structure, specifically, how they alter the transport behaviour of clay minerals. From there, we go one step deeper with the soil investigations to explore the salt and water content of the unsaturated zone down to 20 m below ground in Chapter 5. Chapter 5 also provides the primary data which is used later as an input to the soil water balance modelling and the salt transport modelling. As well as deep coring, geophysical techniques were used to map larger areas.

In Chapter 6 we now go even deeper with the field investigations to assess the groundwater. While information on groundwater monitoring was also provided in Chapter 3, we took the opportunity in this project to monitor the changes induced by the 2011 flood as a unique data set to constrain the further modelling. Winding up the field investigations in Chapter 7 is the initial screening for trace organics at selected groundwater, surface water and treated wastewater sampling points. From there we go up into space to analyse the changing crops and land use in the Lockyer Valley based on remote sensing satellite data (Chapter 8) and conclude our acquisition of primary data.

With this combined information at hand, we are ready to start with the modelling chapters. In Chapter 9, we begin with the soil water balance modelling to estimate irrigation demand and deep drainage at one-dimensional vertical sections performed in a large number of variations with HowLeaky. Then further down in the soil, water flow and solute transport down to 20 m below ground are numerically modelled in one dimension with HYDRUS in Chapter 10; here the reader can obtain an impression of how slow the salty water is actually moving downwards over several decades.

From the modelling in vertical columns we now move to larger areas and the Lockyer Valley as a whole. In Chapter 11, we firstly analyse the existing data on water use in the Lockyer Valley and employ basic groundwater table fluctuation methods to demonstrate the storage and buffer functionalities. Further on, the results from the one-dimensional soil water balance modelling are applied to the entire valley using the information from the crop mapping exercise to arrive at groundwater recharge maps. This information is now a critical input to the numerical groundwater flow model, described in Chapter 12. Chapter 12 also uses the numerical groundwater model to quantify the need to import water into the Lockyer Valley. Drawing on the same tools, Chapter 13 looks into the impact of the available climate change scenarios for SEQ on the water resources in the Lockyer Valley. For this we use both the numerical groundwater model as well as an innovative application of the so-called Eigenmodel method.

Finally, the relevant conclusions from the research are summarised and discussed in Chapter 14 under three different aspects: firstly, the implications and recommendations for PRW in the Lockyer Valley; secondly, how the methods can apply to other qualities of imported water in the Lockyer Valley; and thirdly, how the tools could be used in other areas where conjunctive use, connected surface–groundwater systems and large-scale common pool resources with injection or substitution schemes need to be planned.

2. A TRANSFERABLE ASSESSMENT FRAMEWORK (Leif Wolf)

2.1 Introduction

Despite thousands of water recycling projects and pilots carried out worldwide, there are few examples of integrated assessment of large-scale urban–rural water reuse schemes documented in the literature, especially not when a common pool groundwater resource is involved. Still, a lot of challenges are common between the different recycling projects. Looking at feedback from their members, the International Water Association (IWA) notes the following challenges (IWA Specialist Group Water Reuse, 2011):

1. Public health and regulatory concerns
    a. health issues related to pathogens or chemicals if waste water is not properly treated
    b. lack of adequate regulations and local incentives for reuse
2. Economical and financial barriers
    a. inadequate water pricing and problems for cost recovery
    b. need of high investment for distribution networks and funding for adequate monitoring
    c. uncertainties in supply contracts and pricing
    d. insufficient financial and economic incentives
    e. lack of appropriate models for cost–benefit analysis
3. Technical and environmental constraints
    a. need of high reliability of operation and water quality control
    b. land salination or soil toxicity
    c. technical competence and staff training.

In the early stages of the decision-making process, a general sustainability assessment should be carried out, irrespective of the difficulties of modified regulatory arrangements. For this, a number of sustainability assessment strategies have been applied to analyse individual aspects of sustainability. A recent review (Chen *et al.*, 2012) identified three major approaches:

- Material Flow Analysis (MFA) to calculate the systematic material flow of pollutants and nutrients in environmental sanitation systems over a given period of time;
- Life Cycle Analysis (LCA) to identify environment-related issues of different waste water treatment technologies or water resources on the ecosystem and natural resources life cycle; and
- Environmental Risk Assessment (ERA) studies to analyse the potential environmental risks (e.g. excessive pharmaceuticals and xenobiotic compounds on soil, surface water and groundwater) resulting from recycled water projects (Muñoz *et al.*, 2009).

Still, the assessment of the long-term viability and the feasibility of a large-scale reuse scheme is more complex and requires those approaches as components of a larger framework. As is evident in the example of the Lockyer Valley, the socio-economic boundary conditions and the institutional setting are key factors that might only allow a sustainable reuse scheme if there are financial incentives and if regulations are adapted to achieve a compromise with substantial benefits for all involved parties.

The idea of a large-scale use of recycled water in the Lockyer Valley is not new, and a substantial number of studies have been carried out in the last two decades (see literature reviews by Cresswell, 2008 and van Opstal, 2010). Together with the experiences gathered in the UWSRA project, we make the attempt to formalise this knowledge in a generic assessment framework. This should serve both the future application to the Lockyer Valley as well as other areas with similar settings.

In order to identify key issues upfront, we suggest a problem-mapping exercise shortly after the initial project idea is drafted. Based on the outcome of the problem mapping, the tiered assessment framework can be adapted.

2.2 Problem Mapping

Development of the assessment framework began with a problem-mapping phase based on stakeholder consultation. The stakeholder consultation involved the SEQ Water Grid Manager, Queensland Water Commission (QWC), a number of different units and offices of the Department of Environment and Resource Management (DERM), different members of the Lockyer Water Users Forum (LWUF), the Healthy Waterways Partnership, as well as local universities (UQ, QUT). The stakeholder consultation opened up distinctively different positions on the delivery scenarios and the potential management of the PRW supply.

To convey these different positions, a conceptual problem map provides a means to communicate system complexity and interconnectedness, in a tangible and easily accessible way, which acknowledges how different technical, economical, political and cultural factors affect the fulfilment of the key objectives (Tiberghien, 2011). For the Lockyer Valley, we applied this technique with slight modification, to identify the factors that lead to the fact that the PRW supply scheme to the Lockyer Valley is currently neither implemented nor in the detailed design phase (Figure 6). As will be explained in the section below, we suggest that the key factors hindering the implementation are: i) unknown financial returns for the scheme operator; ii) uncertain demand; and iii) opposition from irrigators who are concerned about high prices and changes in regulations upon PRW introduction.

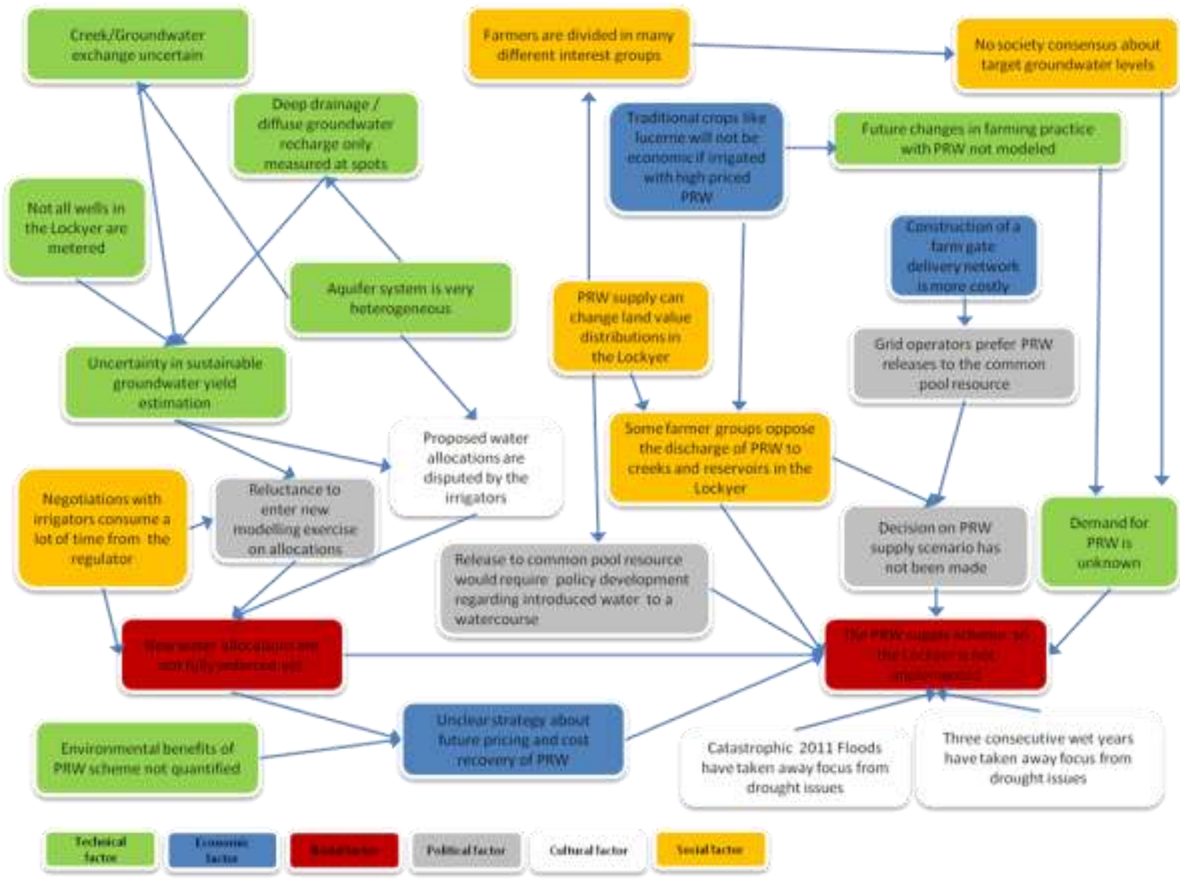

Figure 6: Conceptual map of technical, economic, political, cultural and social factors impacting on the supply of PRW to the Lockyer.

Purified recycled water comes at a relatively high production cost (> A$500 per megalitre), due to the extensive treatment required for indirect potable reuse. This cost can be compared to current water charges of a maximum of A$30 per megalitre in the Lockyer Valley. For this reason, irrigators will go a long way to avoid using PRW at all, as long as groundwater is available. A business case for irrigators to use PRW only exists where supply reliability is compromised due to reduced groundwater levels severely restricting the available pumping supplies from bores/wells, and/or because of the groundwater becoming more salty during low groundwater-level periods. In other years, such as the very wet 2010/2011 season with catastrophic flooding in the Lockyer Valley, the demand for PRW would be close to zero. Clearly, when defining management options to protect groundwater levels, the decision of where the target groundwater levels should be will differ between an environmental, regulators or water user's perspective, and will dictate future PRW demand.

Thus the key questions of the assessment, after the clarification of environmental impacts, are: what is the water demand and what are appropriate regulations?

2.3 Tiered Assessment Framework

Based on the experiences from the Lockyer Valley and a review of current literature, we propose a transferable assessment framework as a guide to future investigations either in the Lockyer Valley or elsewhere (as depicted in Figure 7). The framework applies not only to PRW, but also to importing waste water from alternate sources and of variable qualities, requiring a similar scheme.

In **tier 1** we suggest an 'issues' screening, starting with a simple demand estimate and a comparison of water quality parameters. When importing to a natural environment, the key concern is not the compliance with drinking water guidelines alone; additional key parameters to assess are potential soil structural impacts (using the Sodium Adsorption Ratio (SAR) as an indicator) and salt content in relation to the natural water resources of the area. If the supplied water is already of drinking water standard, such as is the case for PRW, the water quality impact assessment for downstream water supply systems can be limited in detail. Otherwise, minimum travel times on the path from irrigation to the next downstream drinking water plants should be investigated as part of a quantitative microbiological risk assessment. The low ion and nutrient content of PRW is not expected to have detrimental impact in this ion-rich setting. The issues screening in tier 1 can already identify available information on the likelihood and severity of the range of hazards, both of which are then detailed in the following tiers to arrive at a comprehensive environmental risk assessment.

The key objective of **tier 2** is to provide a sufficient understanding of the natural system and the identification of environmental risks. This starts with a review of past land use changes in response to different water availability and a mapping of areas that could be irrigated in future under additional availability of imported water. If metered use data is available, land use and irrigation water demand can be tested for correlation. Soil water balance models such as included in APSIM or HowLeaky (Robinson, 2009) can be used to model irrigation demand and deep drainage under different land use scenarios. In addition, tier 2 should involve investigations of the potential salt accumulation and salt mobilisation. This can be achieved in simplified form by catchment salt balances, the collection of existing soil salinity distribution information or the drilling of soil salinity profiles. Numerical salt transport modelling for the unsaturated zone can be used, as necessary, to estimate the speed of salt release to the groundwater in areas which previously did not receive intensive irrigation. If indicated as a potential risk in tier 1, tier 2 may also include more detail on the response of the aquatic ecology to the new water source.

The aim of **tier 3** is to quantify the demand for imported water from an ecosystem perspective, compare this demand with the reliability of supply perspective and identify potential consensus positions that recognise the trade-offs and costs involved. This requires a model of the groundwater system's response to climate, extraction, irrigation strategies and potential direct injection of imported water. Outputs from this model can inform stakeholder consultation, with the aims of eliciting preferences and allowing deliberative multi-criteria evaluation.

Finally, **tier 4** is designed to provide additional robustness to the exercise by allowing the risks to the future operability of the concept to be considered. At the highest level of complexity, coupled hydro-economic models estimate the impact of different water availability and prices on the future land use and allocation strategies and then subsequently use the new land use information to generate new estimates of water availability, thus allowing for the full feedback loop in this coupled system. This hydro-economic model may also incorporate more complex operational rules and be an instrument to test the performance of different adaptive management strategies. Where available, downscaled climate change scenarios should be considered for increased robustness. The availability of the additional knowledge from the hydro-economic coupling will likely trigger a new round of stakeholder consultation and multi-criteria evaluation.

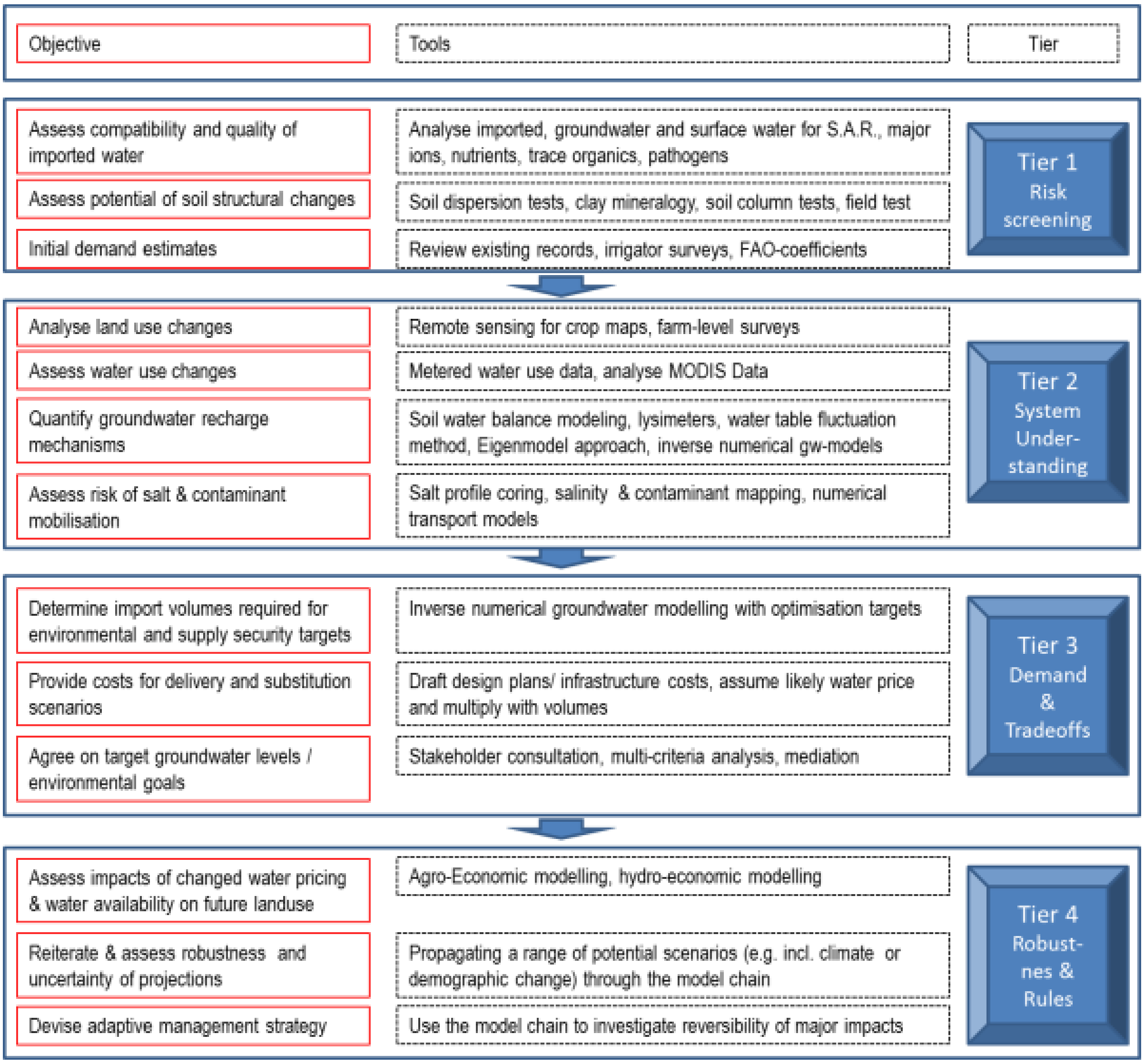

Figure 7: Tiered assessment framework for the import of recycled water to peri-urban agriculture.

3. HYDROGEOLOGICAL FEATURES OF THE LOCKYER VALLEY

(Malcolm E. Cox and Matthias Raiber)

3.1 Introduction

The Lockyer Valley is situated 80 km west of Brisbane and is bounded on the south and west by the Great Dividing Range. The valley is a major western sub-catchment of the larger Brisbane River drainage system and is drained by the Lockyer Creek. The Lockyer catchment forms approximately 20% of the total Brisbane River catchment and has an area of around 2900 km^2. The Lockyer Creek is an ephemeral drainage system, and the stream and associated alluvium are the main source for irrigation water supply in the Lockyer Valley. The catchment is comprised of a number of well-defined, elongate tributaries in the south, and others in the north, which are more meandering in nature.

3.2 Land Use and Setting

3.2.1 Geomorphology

The dominant factor in the development of the current landscape of the catchment is differential erosion with Lockyer Creek largely following the surface exposure of the erodible bedrock sandstone (Figure 8). Steep sided V-shaped valleys are formed in the headwaters of most streams where they incise the overlying and more resistant basalts, notably in the southern sub-catchments of the valley. The streams rapidly flatten out into broad alluvial plains flanked by gently undulating rises. Due to the relatively flat-lying nature of the sedimentary bedrock of the catchment, stepped hillsides are a noticeable feature of the landscape. Catchment boundaries typically have cliffs developed at the change from siliceous sandstones to softer feldspathic sandstones or at formation boundaries, such as between the more resistant basalts overlying the softer Walloon Coal Measures.

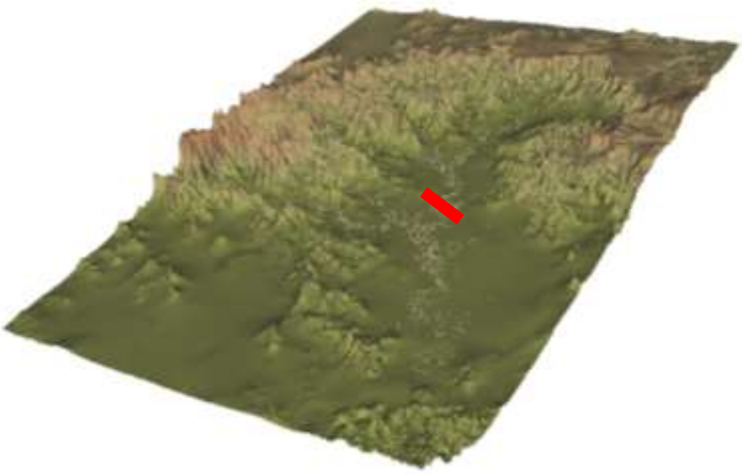

Figure 8: Topography of the Lockyer catchment looking to the south-west. In the foreground Lockyer Creek flows towards the Brisbane River. The central valley shows groundwater bores in alluvium, and elevated ranges to west and south. Vertical exaggeration around 7 x. The line shows the location of the profile in Figure 20.

Lockyer Creek flows for approximately 100 km in a generally eastern direction from its headwaters in the Great Dividing Range to the confluence with the Brisbane River just south of Wivenhoe Dam (Figure 10). The morphological history of the creek began in the Miocene as it cut narrow valleys into the basalt plateau of the Main Range Volcanics (Powell *et al.*, 2002). Further headward erosion, down-cutting, channel incision into bedrock and broadening of the valleys continued into the Pleistocene; by this time, the landscape is assumed to have reached its present form, with any further adjustments mainly within the valley floors (Beckmann and Stevens, 1978). The development of the alluvial plains filling the valley floors ceased under the influence of the more stable climate of the Holocene. The main creek currently flows within deeply incised channels in alluvium, with gentle levees and backswamp depressions toward the valley margins (Powell *et al.*, 2002). Within the lower sections of the valley, the meanders of Lockyer Creek become broader, traversing the entire width of the alluvial plain due to the lower gradients. In some areas in this lower section, paleochannels are obvious where the creek has avulsed across the floodplain despite its deep channels. Some of these areas develop swamps during wet periods.

3.2.2 Land Use

Most of the land area of the valley is forest or open woodland, and of this 85% is used for grazing and 10% for crops irrigated on alluvial plains. The Lockyer Valley is a major vegetable and grain producing area supported by groundwater irrigation, largely from alluvial aquifers (Figure 9 and Figure 10). Of the 20,000 ha that is farmed, approximately 15,000 ha is irrigated. The catchment, however, has a record of land and water problems resulting from past land use, limited valley-wide water management strategies and variable rainfall.

The main impacts on groundwater are related to factors such as the intensive use for irrigation, rising or falling water tables and a decrease in water quality. Both surface water and shallow groundwater in the catchment are highly susceptible to both quantity and quality issues during prolonged periods of low rainfall, during which groundwater extraction increases, as recharge to the total system becomes greatly reduced. For extended periods during the recent drought, any flow in the drainage system was in the headwaters and upper sections, and usually a result of moderate rainfall events, run off and limited periods of baseflow mostly in tributaries in the south and west.

The use of surface water has long been controlled by state government licensing; however, the use of groundwater is only regulated within a localised area of the Lockyer Creek catchment, central to the valley. The mean annual extraction of groundwater for the valley was previously estimated at 45,000 ML; safe annual withdrawal from alluvial aquifers has been estimated at 25,000 ML (QWRC, 1982 unpub). The implications of this over-extraction are wide ranging. During drought conditions, the lack of recharge from surface water leads to increased recirculation of irrigated water, and, combined with the practice of spray irrigation and the associated evaporative concentration, salination of shallow groundwater is occurring in some areas (e.g. Wilson, 2005, Cox and Wilson 2006). There is a tendency for increased soil salinity towards the edges of the floor of the main valley, as well as the sub-catchments draining from the south (Figure 16).

Figure 9: View of lower sub-catchment of Laidley Creek showing irrigation of crops from alluvial aquifer; creek channel is within the row of trees; hills at rear are sedimentary bedrock capped by basalt for the catchment boundary.

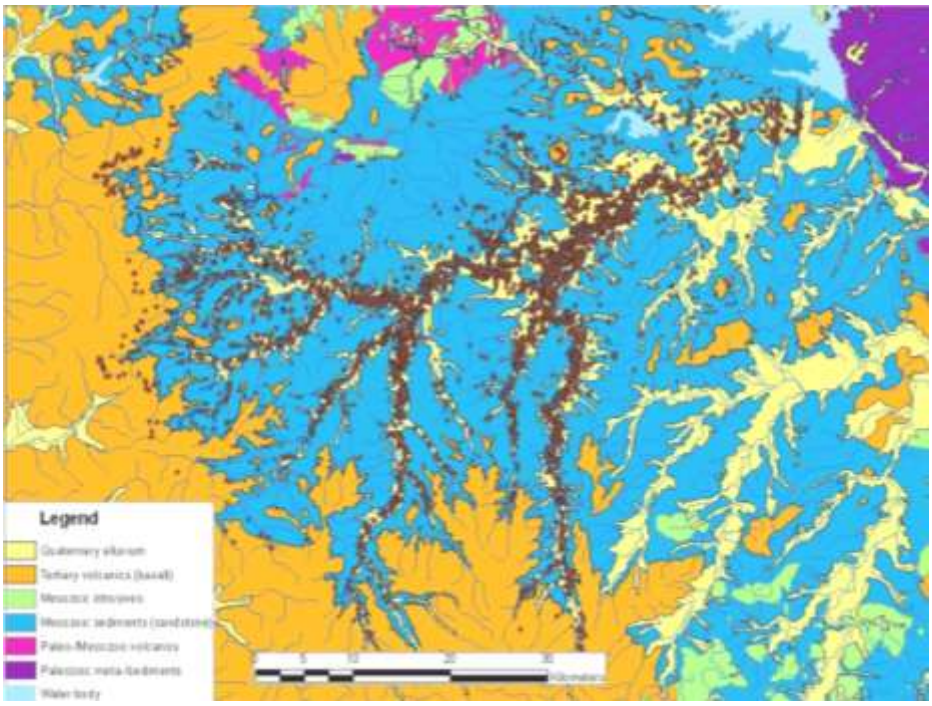

Figure 10: Geological map of the Lockyer Valley showing the concentration of groundwater bores in the alluvial aquifer. The relation between the headwaters of the streams in the sub-catchments and the Tertiary basalts is apparent, notably in the south and south-west.

3.3 Groundwater Investigations

Various studies over the last 40 years have confirmed that the groundwater of the Lockyer Valley indicates variable salinity which is increasing over time (e.g. Dixon, 1998 unpub.; Dixon and Chiswell, 1994; Wilson, 2005; Zahawi, 1975). The analyses of chemical data confirm that the geology of a sub-catchment can have a significant influence on the salinity and chemical composition of the surface water. High salinity levels in alluvial aquifers have resulted from both locally rising water tables and/or saturated soils, as well as lowered water tables during extended dry periods and increases in evaporation/evapotranspiration. McMahon and Cox (1996) examined the relationship between groundwater chemical type and Jurassic sedimentary formations within the Sandy Creek catchment, noting variations in composition related to formation. Macleod (1998) conducted research of the Ma Ma Creek catchment and showed that excessive irrigation from alluvial bores results in a decrease in the volume of alluvial groundwater and an increase of saline waters. Picarel (2004) and Cox and Picarel (2010) studied the lower section of the catchment under drought conditions and confirmed the alluvial water tables to be draw down to the lower 5 m of much of the alluvium, and noted the input of more saline bedrock groundwaters.

Combining the chemical and isotopic character of groundwater has also been a useful approach, and can, for example, demonstrate the difference between different groundwaters and hydrogeological units, as well as hydrological processes (e.g. Cox and Wilson, 2005). A MODFLOW numerical model developed for the alluvial aquifers at the confluence of Tenthill Creek and the Lockyer Creek (Wilson, 2005) confirmed the rapidly declining levels of groundwater and their local distribution at that time. A further study (Dvoracek, 2012, under review) examined the Laidley Creek catchment and produced a conceptual hydrogeological model which incorporated water chemistry, and related the variations in chemical type to source aquifer, as well as mixing. A groundwater model was developed for the Central Lockyer Valley management area by Durick and Bleakley (2000), an area of 140 km^2 with extraction of around 13,000 ML/a; this model incorporated bore monitoring data and is used to support water use management. A substantial report for the Brisbane City Council (KBR, 2002) included a compilation of current knowledge of the Lockyer Valley aquifer and produced conceptual models and numerical MODFLOW model as well as assessing variations in water quality.

Unsustainable use of groundwater in this catchment has resulted in historical water shortages. In response to aquifer depletion, the Queensland Government constructed a network of artificial recharge weirs throughout the Valley during the 1970s and 1980s, together with larger storages (Atkinson's Dam, Bill Gunn Dam and Lake Clarendon) to provide a combination of additional surface water irrigation capacity, and water for groundwater recharge. On Lockyer Creek and tributaries, there are 24 small weirs, four of which were originally constructed as supplementary irrigation storage. Throughout the valley, water for irrigation is sourced from both groundwater and surface water resources, with the vast majority of the groundwater resources being drawn from unconsolidated alluvial deposits. Atkinson Dam, in the north-east, constructed in 1970 over an existing lagoon, obtains water by diversion from a weir constructed on Buaraba Creek and from overflows from Seven Mile Lagoon. The dam is used to supply irrigation water via surface water releases back into Lockyer Creek, Buaraba Creek and associated channels and weirs.

3.4 Regional Geology

3.4.1 Regional Setting

The Lockyer Valley lies within the Laidley sub-basin, part of the Clarence-Moreton Basin (CMB) which is a large, Mesozoic age, intracratonic basin, set regionally within the New England Orogen (Wells *et al.*, 1990). Hydrogeologically, the CMB is a sub-basin of the Great Artesian Basin, which is a hydrogeological entity that includes a number of large geological basins. These basins are defined by structural and stratigraphic features that also shape their aquifer and aquitard geometry. The CMB covers around 40,000 km^2 in SEQ and north-eastern New South Wales. The basin extends from the New South Wales coast in the east to the Kumbarilla Ridge in the west, a broad basement high approximately 100 km west of Toowoomba (Wells and O'Brien, 1994). The location of this boundary, and therefore regional groundwater flow, is currently under reassessment. Most of the CMB fill consists of the sedimentary rocks of the Bundamba Group, which represent continuous non-marine deposition, which occurred from the Late Triassic to the Middle Jurassic (Wells and O'Brien, 1994). Changes in the depositional style and composition of stratigraphic units in the Clarence-Moreton Basin can be related to variations in tectonic uplift of the hinterland and basin subsidence. Paleocurrent data indicate that throughout the deposition of the Bundamba Group, rivers that traversed the basin were approximately north flowing and discharged from the basin via the northern margin (O'Brien and Wells, 1994).

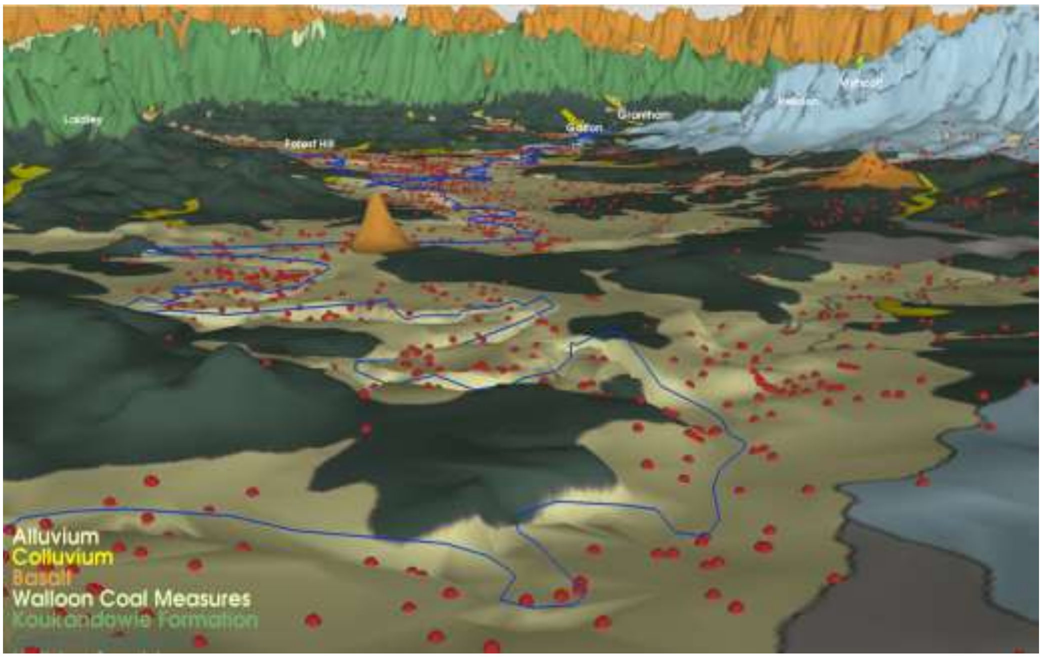

Figure 11: Oblique view to the west showing the course of Lockyer Creek and alluvium with groundwater bores. Geology is digital map from Geological Survey of Queensland draped over digital elevation. Image from GVS model (Hawke *et al.*, 2012). Vertical exaggeration around 20 x.

The CMB, Ipswich Basin and Esk Trough represent a genetically associated system of basins controlled by strike-slip faulting crustal extension. Trans-tension along the major fault, the West Ipswich Fault, and in the Laidley Sub-basin created a depo-centre bordered by the West Ipswich Fault and Gatton Arch. The Gatton Arch is a south-easterly plunging anticline at the northern margin of the CMB that remained as a relatively elevated area (Wells and O'Brien, 1993). The subdivision of stratigraphic units in the CMB is shown in the stratigraphic column in Figure 12 based on Wells and O'Brien (1994) and Raiber and colleagues (2012). The pre-CMB basement underlying the basin consists of a variety of stratigraphic units, including, for example, the Esk Formation and the Neranleigh-Fernvale Beds.

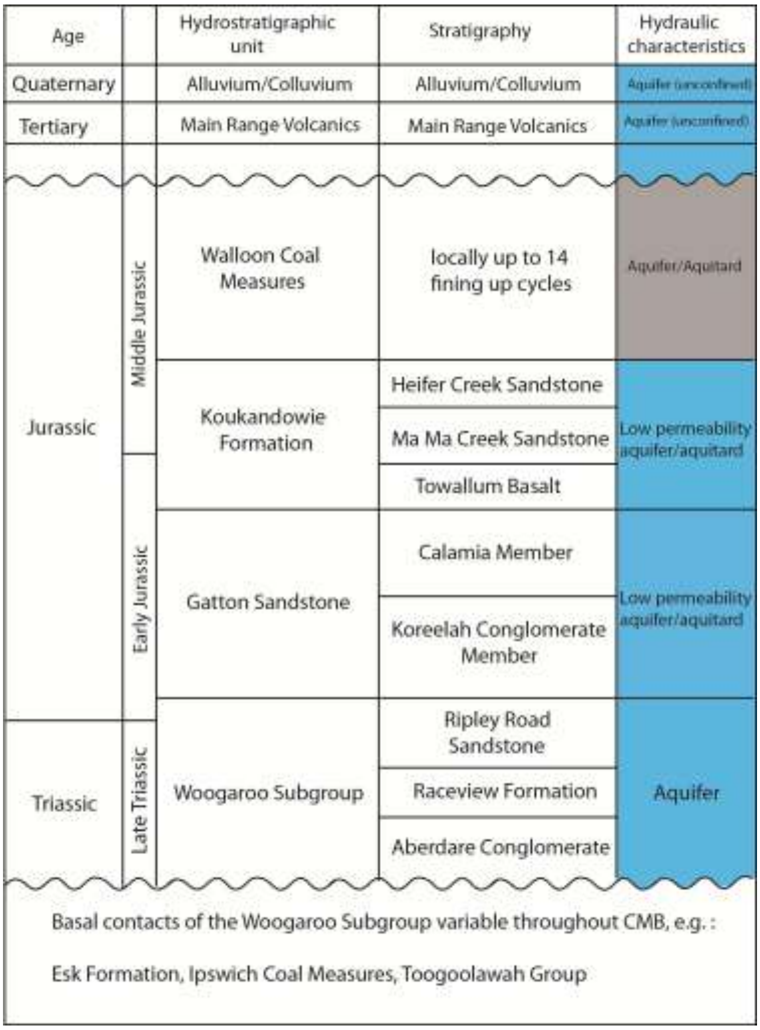

Figure 12: Stratigraphic column of major geological units in the Lockyer Valley. The valley has eroded through the Walloon Coal Measures and into the Woogaroo Subgroup. Depending on spatial variation of lithology (e.g. dominance of mudstones versus dominance of sandstones), a stratigraphic unit may be an aquifer in some areas and an aquitard in other areas.

3.4.2 Bedrock Formations

In this current stratigraphic classification, the units of the CMB that constitute the bedrock of the Lockyer Valley are summarised below, oldest (deepest) to youngest.

Woogaroo Subgroup (formerly known as Helidon Sandstone) is the lowest bedrock formation in the catchment and generally behaves as a useful aquifer; it is a strongly cross-bedded siliceous sandstone with accessory siltstones and persistent basal conglomerate (McTaggart, 1963). The unit is described by O'Brien and Wells (1994) as sub-litharenites to sub-feldsarenites to quartz-arenites, and interpreted as deposited by sandy braided streams in a humid setting. The Woogaroo Subgroup can be further subdivided into three units in the Queensland part of the CMB (Figure 12): the Aberdare Conglomerate, a basal polymictic conglomerate, the Raceview Formation, a mixed sand and shale sequence and the Ripley Road Sandstone, a massive to cross-bedded siliceous sandstone (Wells and O'Brien, 1994). The Woogaroo Subgroup is only exposed in areas along the northern part of the catchment and to the north of the course of Lockyer Creek.

Gatton Sandstone: The Gatton Sandstone consists primarily of thick bedded, relatively uniform, medium and coarse grained, quartz-lithic and feldspathic sandstone commonly with argillaceous matrix and calcareous cement. Pebble beds, carbonised wood fragments and large-scale planar and cross-bedding are characteristic of this formation (Wells and O'Brien, 1994). Discrete conglomerate beds, previously called the Winwill Conglomerate, were considered as a lithology that marks the top of the Gatton Sandstone, but is no longer considered to be a single mapable unit. However, these conglomerates and resistant sandstones in the upper Gatton Sandstone do appear to have hydrogeological significance (McMahon and Cox, 1996; Wilson, 2005; Zahawi, 1975). The Gatton Sandstone is more uniform in lithofacies than the overlying Koukandowie Formation (Wells and O'Brien, 1994). The unit is rich in sodium, calcium and magnesium carbonates (McTaggart, 1963), and rests conformably on the Ripley Road Sandstone. It is now described as having two members, the lower Koreelah Conglomerate Member, and the overlying and thicker Calamia Member. The Gatton Sandstone formation contains water, but overall it is of low permeability and the water of poor quality, and in places is saline. The formation is commonly exposed in the floor of the Lockyer Valley, where it underlies much of the alluvium.

Koukandowie Formation: The Koukandowie Formation also contains sheet-like, quartz-feldspathic-lithic sandstone units, but in contrast to the Gatton Sandstone locally has finer facies including minor coal. Within the Koukandowie Formation is a lower unit, the Towallum Basalt, of limited extent, but most of the formation is a mixed sandstone, siltstone, shale unit that includes two members: the Ma Ma Creek and the Heifer Creek Sandstone. In the lower parts, the Ma Ma Creek member comprises shale, siltstone and interbedded sandstone, consisting primarily of fine-grained facies. This member conformably overlies the Gatton Sandstone (McTaggart, 1963; Wells and O'Brien, 1994) and contains flaggy, lithic sandstones, shales and siltstones with minor fossil wood conglomerate bands. O'Brien and Wells (1994) interpreted the environment of deposition as low energy floodplains or lacustrine setting with an environment of raised salinity, suggesting a closed basin (McTaggart, 1963) or shallow marine incursion (O'Brien and Wells, 1994). Grimes (1968) described a sandstone sample from the Ma Ma Creek catchment as being feldspathic sub-labile sandstone that contained 5% volcanics, 6% plagioclase (weathered), 20% calcite and 20% ferruginous cement with the remaining 49% to be quartz.

The Heifer Creek Sandstone is the upper member of the Koukandowie Formation and is a resistant quartzose medium- to coarse-grained cross-bedded unit that commonly forms prominent topographic features with steep slopes with exposures in cuttings, cliffs and benches (McTaggart, 1963; Wells and O'Brien, 1994), with conglomeritic sandstone towards the top of the member. It is considered to have been deposited in braided, sandy, bed-load streams. Grimes (1968) suggested that the greater initial porosity of this upper member has allowed chemical cements to be deposited, which increases the resistance of the Sandstone Member to erosion. The formation is of highly variable permeability, but mostly acts as an aquifer.

Walloon Coal Measures: The Walloon Coal Measures (WCM) are composed primarily of mudstone, siltstone and thin layers of coal, and are the stratigraphically highest sedimentary formation in the Lockyer Valley. The WCM conformably overlie the Koukandowie Formation and consist of mixed lithologies, including carbonaceous siltstone, shale and mudstone, volcanolithic sandstone, coal and bentonitic siltstone (Wells and O'Brien, 1994). In the Lockyer Valley, the WCM remain only as thin residual layers, or preserved beneath Tertiary basalt flows, to a maximum thickness of approximately 60 m (McTaggart, 1963).

Coal seams in the WCM are commonly less than 1 m thick and usually include thin lenses of siderite and thin concordant veins of calcite (Wells and O'Brien, 1994). The unit also contains clays such as montmorillonite, which often results from the weathering of basic rocks, and is often cemented to ironstone concretions (Wells and O'Brien, 1994).

3.4.3 Younger Formations

There are several geologically younger units that overlie the bedrock within and peripheral to the Lockyer Valley. Within the centre of the valley, Lockyer Creek occupies a broad alluvial plain, up to 5 km wide, and connected to a network of tributaries. Capping the ranges surrounding the valley are basalt flows of Tertiary age. Powell and colleagues (2002) describe the basalts as being the primary source for the alluvium, due to their position at the top of the catchment and because they are the dominant rock type in the catchment, occupying more than 50% of the catchment area. Around the valley sides, the Koukandowie Formation is considered a significant contributor due to the tendency for landslips together with the high position in the landscape. The northern tributaries draining the uplands formed by rocks of the Woogaroo Subgroup are large suppliers of sandy sediment to the plains; alluvial deposits associated within the Buabara Creek catchment in the north-east tend to be more sandy as their provenance is almost solely Woogaroo Subgroup. Studies and anecdotal evidence have suggested that Lockyer Creek contributes a disproportionally large volume of sediment into the Brisbane River. Soils derived from the middle-level sedimentary formations were the main source of suspended sediment (Powell *et al.*, 2002).

Alluvial deposits: These deposits occur throughout the drainage system and are of primary importance within the catchment as sources of irrigation water, as well as providing fertile soils. Quaternary alluvial and minor colluvial deposits occur within the main stream and tributary valleys which tend to be elongate in distribution (Figure 13 and Figure 14). The alluvium is commonly several kilometres in width in the main valley and typically 20–35 m thick. The cross-sections of the alluvium in the main channel tend to be U-shaped, and more V-shaped in the tributaries. The base of the alluvium is marked by a contact with various bedrock formations, depending on location and elevation. The underlying bedrock is commonly weathered for several metres depth, and the surface morphology of the bedrock below the alluvium can be highly irregular locally. The alluvium is rarely in direct contact with the Main Range Volcanics anywhere in the catchment except in the headwaters of the creeks. The alluvial deposits consist of well-graded gravel, sand, silt, and clay, and are derived primarily from the weathering of basalt, sandstone and shale (Smith *et al.*, 1990). The alluvial soils in the upper several metres are commonly dark cracking clay that shows textural gradation both down the catchment and perpendicular to stream flow; the heaviest, most clay-enriched, soils are furthest from the source and the creek bank.

Tertiary Volcanics: Tertiary basalt flows of the Main Range Volcanics cap the sedimentary bedrock formations in the high relief and surrounding areas, and form the resistant ridges. These widespread, generally flat-lying basalt flows of alkali-olivine composition, with some vesicular beds, in places are interbedded with minor trachytes and rhyolites with jointing and highly vesicular bands (e.g. Zahawi, 1975). In several areas within the catchment there are small plugs of basalt, such as Mount Tarampa, in the eastern part, which has produced an offset of the Lockyer Creek.

3.5 3D Visualisation Model

As part of this overall project, a 3D visualisation model of the groundwater systems of the Lockyer Valley was produced using the GVS software (Cox *et al.*, 2010; Hawke *et al.*, 2010). The 3D model was used to display hydrogeological features of the whole catchment (Figure 11), as well as detail of the alluvial system as the main source of groundwater. Although the alluvial deposits are variable internally, they are flat-lying and can be mapped in 3D (Figure 13 and Figure 14) as in general having the following sequences:

1. upper layers: soil, loam and silt; this is related to flooding and overbank deposition, and forms the main soils for agriculture. In places these soils can locally perch irrigation waters and produce saturated zones.
2. intermediate layers: clay, silty clay, silty sand and sand; these layers are highly variable spatially, both laterally and vertically. Locally, they can produce some semi-confined groundwater conditions.
3. basal layers: coarse sands and gravels; this sequence forms the main alluvial aquifer, and is typically in contact with the underlying bedrock. The lateral extent can be highly variable and is not always continuous; in many locations these coarse sediments fill depressions within the bedrock.

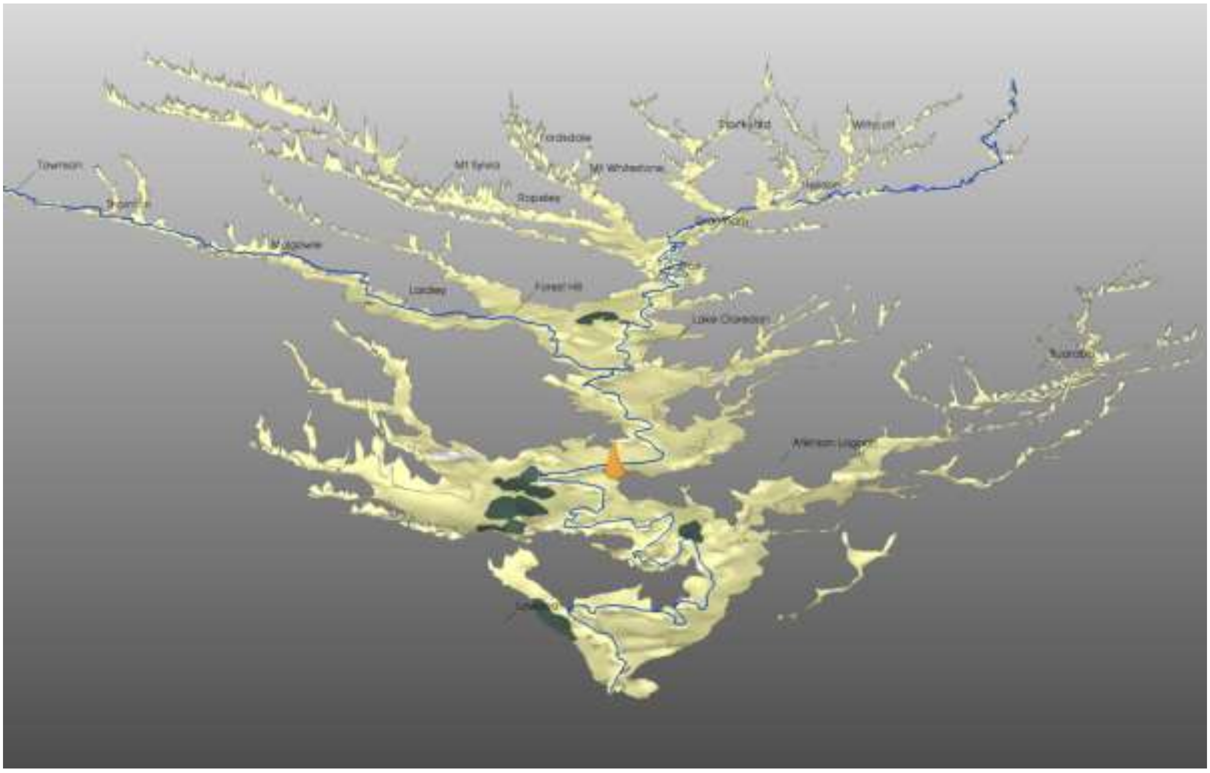

Figure 13: Oblique view of Lockyer catchment towards the west showing the extent of the alluvium and shaded contours for base of alluvium/surface of bedrock. Coloured zones are solid geology (green: bedrock; orange: basalt). Visual model produced in GVS (Hawke *et al.*, 2010).

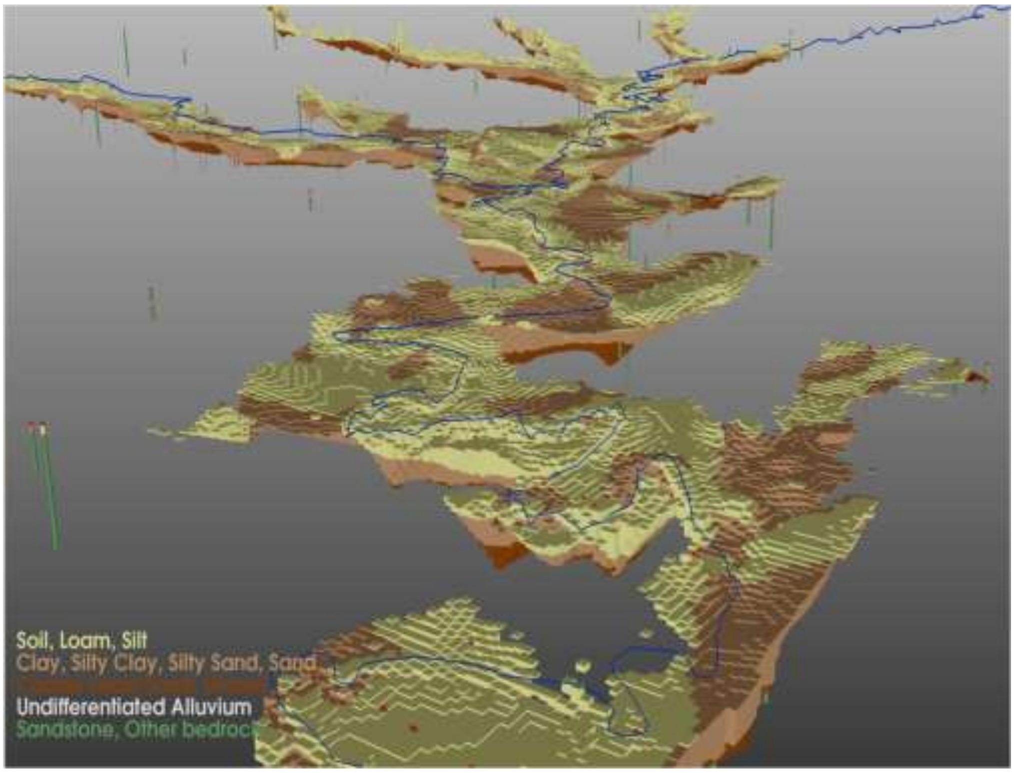

Figure 14: Solid geometry of the alluvial aquifer system looking upstream (to the west). The hydrogeological model was built using GoCAD software, and imported into GVS. The irregular nature of the alluvium deposits is displayed, and the three categories within the alluvium.

A vertical distribution of the three alluvial layers is shown in Figures 15 and 16. The detail of geologs (Figure 15) also displays the original drillers' geological logs from the state government records and used for these summaries. The relation of the basal coarser layer and underlying bedrock can be seen; typically groundwater bores are drilled into the bedrock (e.g. 1 m) to confirm the alluvial base.

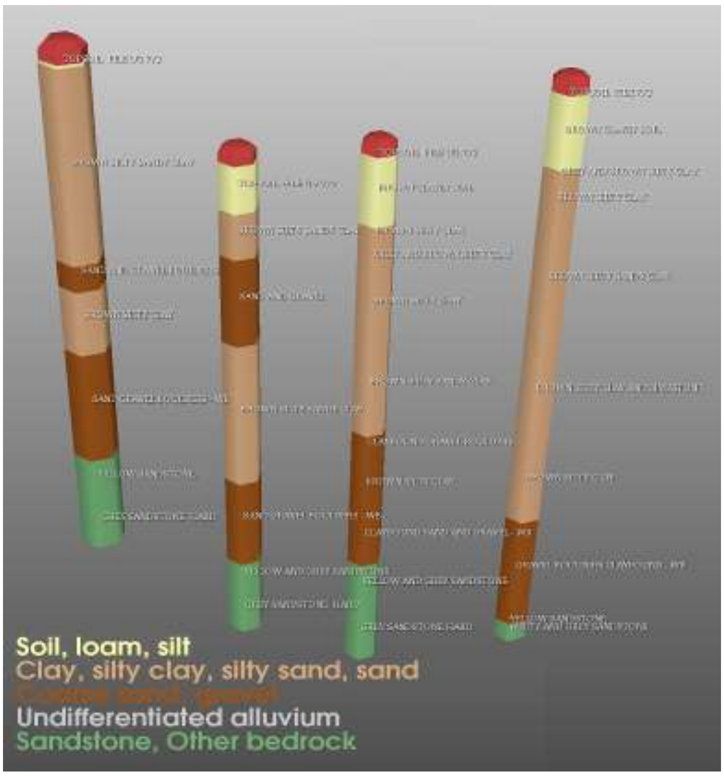

Figure 15: Examples of alluvial bore logs from the GVS software, showing (a) grouping into three categories, and (b) more detailed original geological logs.

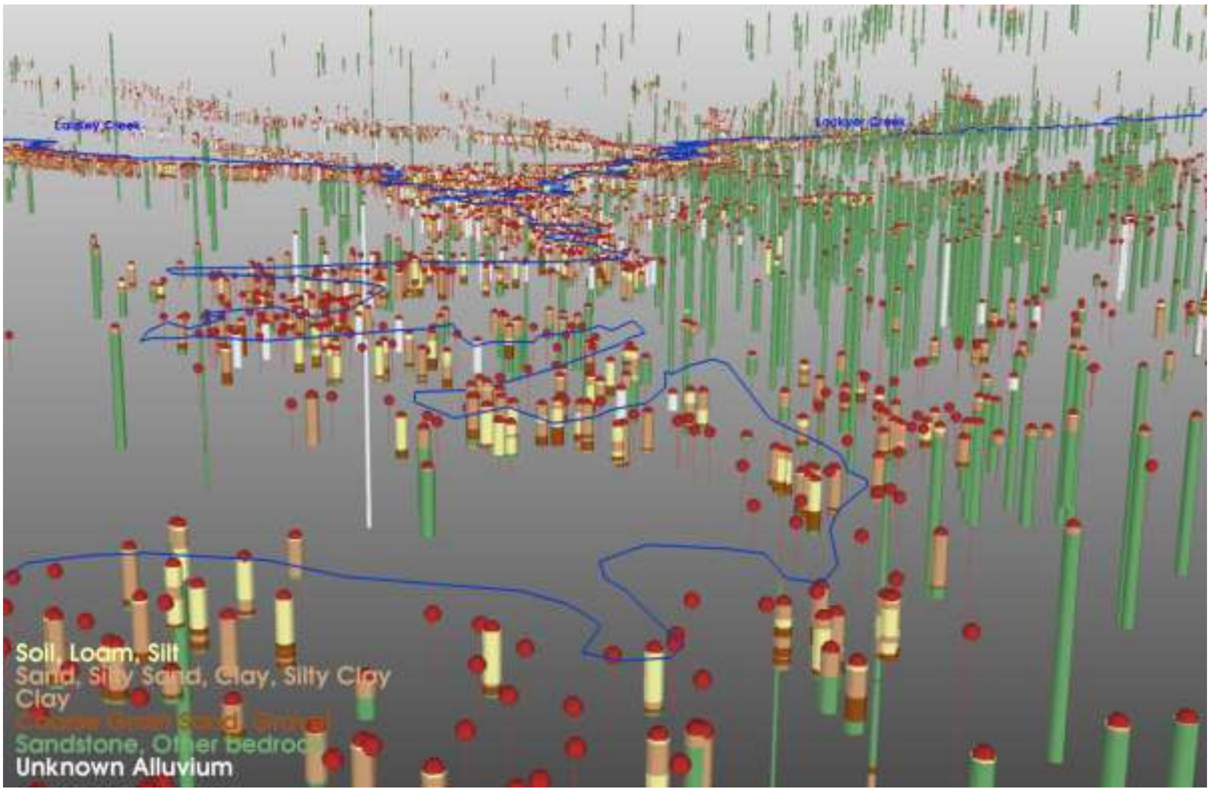

Figure 16: Screen image from the GVS visualisation model looking upstream from the lower Lockyer. Bore holes in alluvium show the different materials; bedrock sandstone is shown as green in the base of some alluvial holes, and in the peripheral drillholes which are within areas of exposed bedrock.

As can be seen in Figure 10 and Figure 16, there are a substantial number of drillholes within bedrock formations as well as the alluvium. These holes are typically deeper and have lower production than the alluvium. The groundwaters in Walloon Coal Measures, Koukandowie Formation and Gatton Sandstone typically have total dissolved ions commonly 3–5 times higher than that of alluvium, and are commonly used for cattle watering. Woogaroo Sandstone groundwaters are of much better quality and are commonly used for irrigation of pasture and orchards.

3.6 Hydrochemistry

3.6.1 Salinity

In very general terms, the salinity (as electrical conductivity [EC]) for much of the alluvial groundwater is in the range of 500–3000 µS/cm; however, there are areas where the EC is >5000 µS/cm. In addition to this, there has been a trend of increasing salinity due to overuse and prolonged dry periods (Figure 10). Related to this are strong effects of evaporation on the salinity of shallow ground/soil waters, notably in low-lying parts of the catchment, in particular those of intensive irrigation. A major impact on alluvial groundwater salinity occurs during prolonged dry periods due to a combination of: (a) limited or no recharge; (b) continued pumping; (c) enhanced seepage of bedrock groundwaters into the basal alluvial aquifers; and (d) enhanced movement of saline groundwater from shallow alluvial aquifers on the margins towards the main alluvium.

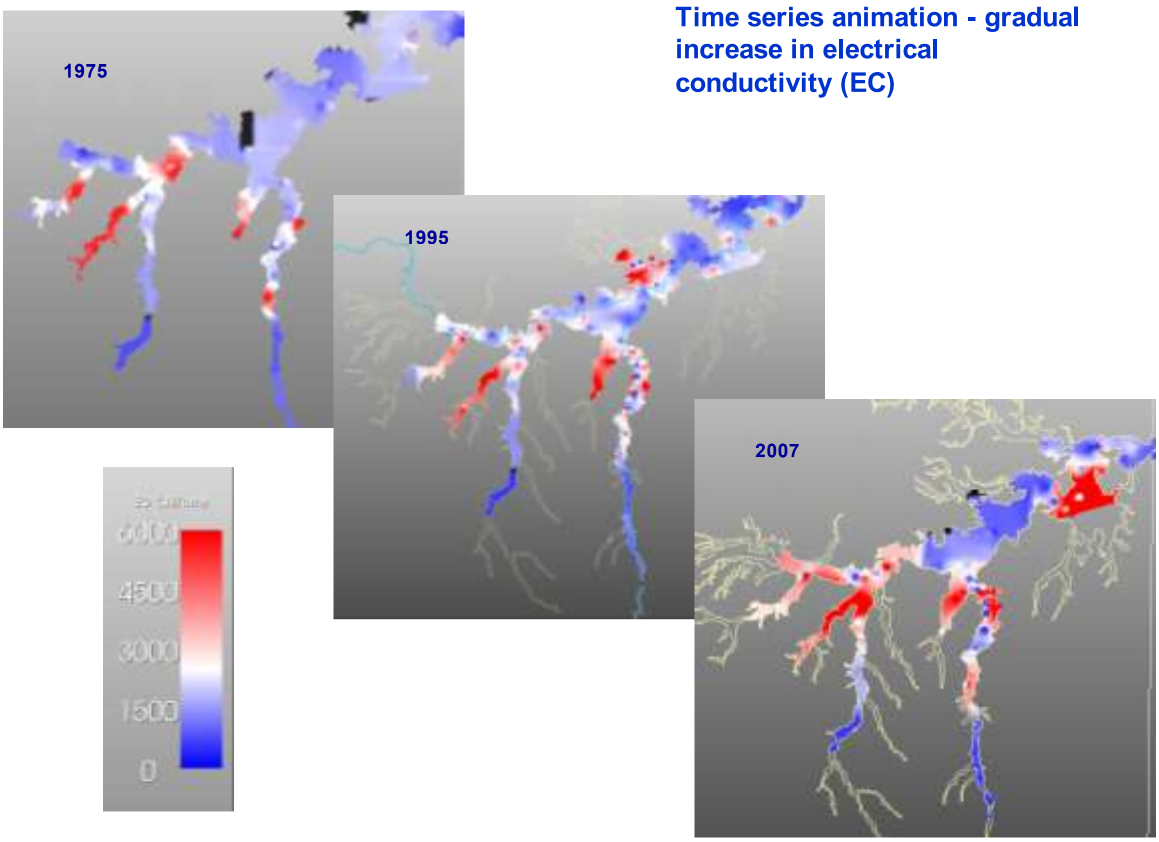

Figure 17: These images are plan views from the GVS model that show alluvial aquifer EC in 1975, 1995 and 2007; 1975 followed flooding in 1974, and 1995/2007 were within an extended drought period.

3.6.2 Types of Aquifers

In summary, there are three distinct aquifer systems in the Lockyer Valley. These are:

1. the fractured and vesicular basalts of the Main Range basaltic volcanic rocks;
2. the porous and fractured sedimentary bedrock formations of the CMB; and
3. the unconsolidated sand and gravel aquifers of the alluvium flanking Lockyer Creek and its tributaries.

Due to the extensive coverage of Tertiary basalt flows on the surrounding ranges and in the upper reaches of most tributaries where average rainfall is greatest, most groundwater is sourced originally from these aquifers (Dvoracek, 2012, under review; Dvoracek and Cox, 2008; Galletly, 2007; Raiber *et al.*, 2011) and flows down the valleys to the tributary streams and/or infiltrates into the formations of the CMB. As a result, most groundwaters have Mg/Ca ratios > 1.0, largely due to the presence of magnesium-rich olivine minerals in the basalts (Zahawi, 1975). Generally, the groundwater is Na dominant from weathering of feldspar minerals in both the basalts and the lower bedrock formations, such as the Gatton Sandstone into which the stream channels are cut. Many of the other formations of the CMB are relatively poor aquifers showing poor yields and variable salinity; some waters from both the upper and lower formations show very high salinity, in particular the Gatton Sandstone. Sodium and chloride are the major ions in most bedrock formations. The groundwaters of the lowest formation of the valley, the Woogaroo Subgroup, are mostly fresh with relatively high yields and typically have HCO_3 as the dominant anion. This formation is commonly used for irrigation of orchards in the northeastern section of the catchment. Fracture intensity and bedding can be important factors in groundwater yields. Variations in groundwater chemistry due to other processes are also evident; for example, a plot of log TDS (mg/L) versus δ^2H (Cox and Wilson, 2005) shows four 'end members':

(a) bedrock, (b) storm stream flow, (c) deeper artesian basin and (d) strongly evaporated alluvial groundwater. Various mixed waters fall between these end points.

3.6.3 Graphical Grouping of Groundwater Chemistry

Based on their major dissolved ions, groundwaters can be grouped using the tenary Piper plot (for example, Figure 18). A broad grouping and relation to aquifers shows the following: alluvium, bedrock, basalts, creeks and CO_3-rich waters. The latter waters are sourced from the Great Artesian Basin formations and sampled from bores in both alluvium and bedrock. The natural carbonated springs at Helidon, in the west of the catchment, are of note. Distribution of water chemical type indicates that different hydrogeological 'units' tend to produce characteristic hydrochemistry, previously observed by McMahon (1994) and McMahon and Cox (1996). Based on this, the depth and hydrogeological setting of a bore can often be indicated.

Below is a descriptive approach to chemical type, salinity range and hydrogeology; of importance is the indicated mixing between groundwaters of different sources, which is common, and displayed by their chemical character. The eight groups described below are also referenced to the Piper diagram (Figure 18).

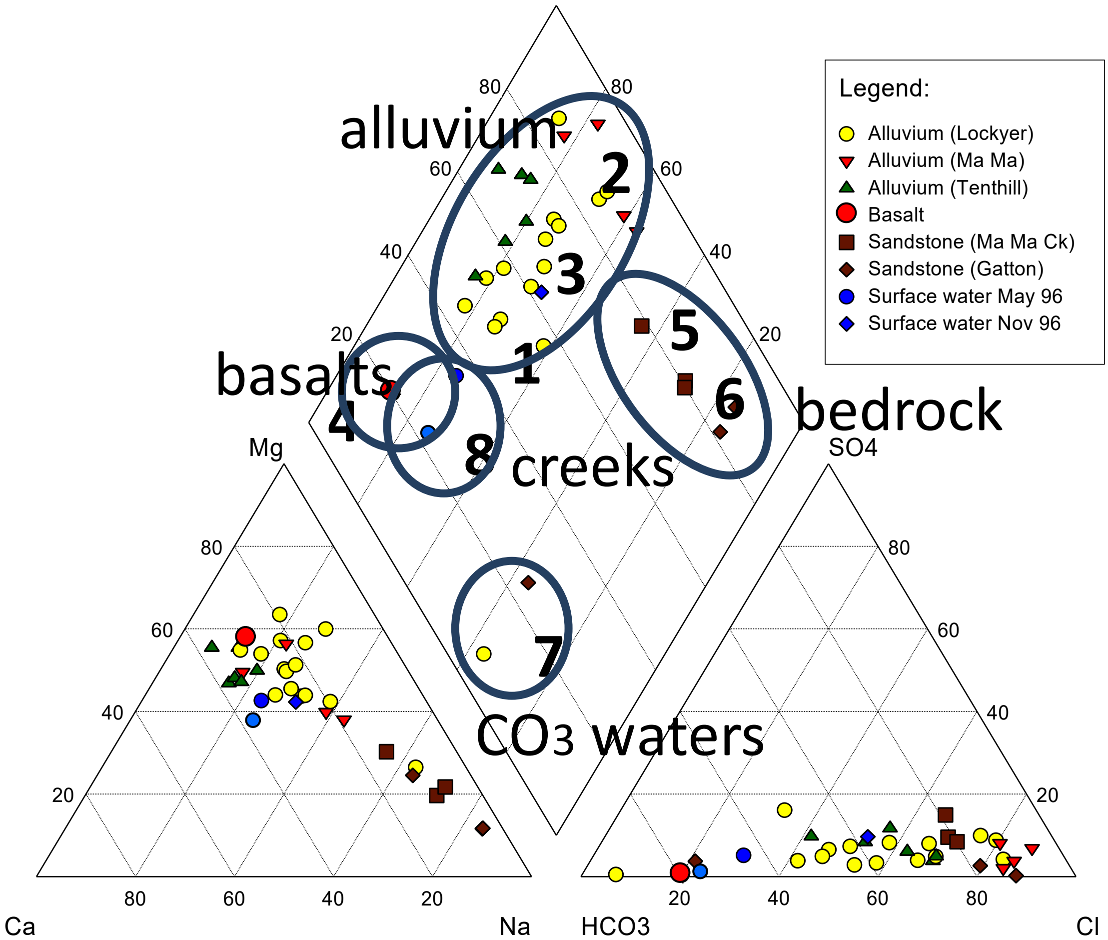

Figure 18: Piper plot of groundwater chemistry based on samples collected in 1996 and 2003 (Wilson, 2005) mostly in the central Lockyer valley. Generalised aquifer types are displayed, and the distribution based on chemical types shown.

Group 1: alluvial groundwater overlying the Koukandowie Formation is dominated by Mg, Na, Ca-HCO_3, Cl type water with EC from 700 to 2600 µS/cm. In the upper tracts of the alluvium, HCO_3^- is also the dominant anion. The low EC suggests that these groundwaters receive fresh HCO_3-dominated recharge water and only undergo small increases in EC after recharge.

Group 2: alluvial groundwater in aquifers overlying the Gatton Sandstone is more saline, from 5700–10,800 µS/cm and Mg,Na,Ca-Cl type. It is characterised by a decrease of Ca relative to Na. Conglomerate in the upper Gatton beds contains abundant calcite cement, and consequently groundwater from this unit has proportionally more Ca than groundwater from the Koukandowie Formation, but the effects of Na feldspar dissolution appear to be dominant.

Group 3: alluvium near the confluences of the tributary streams with Lockyer Creek in the central valley, overlying the Gatton Sandstone with EC 800–6100 µS/cm and Mg, Na, Ca-Cl, HCO_3 type. This alluvial groundwater is between Groups 1 and 2 on the Piper diagram, and separate from Gatton Sandstone groundwater (Group 6). These relationships indicate that Group 3 is a product of mixing of waters from upstream, with minimal contribution from the Gatton Sandstone. The lower Ca:Na ratios suggest the Gatton beds have the greater influence in these bores, while samples closest to creeks have a composition close to flood water, indicating recharge from creeks.

Group 4: groundwater from the basalts topped by laterite soil of the surrounding Main Range Volcanics is Mg, Ca-HCO_3 type with EC 1100 µS/cm.

Group 5: the Koukandowie Formation, is Na, Mg-Cl type water with EC ranging from 4000–19,000 µS/cm; some higher concentrations are seepage from bedrock springs. Not considered to be a good aquifer.

Group 6: the Gatton Sandstone is less saline, with EC 10,000 µS/cm and Na-Cl type water. Some of these groundwaters can have higher salinity. This formation underlies much of the alluvium in the main valley floor, but is not considered to be a good aquifer.

Group 7: Na-HCO_3, produced by CO_3-rich waters. The latter waters are sourced from the Great Artesian Basin and sampled from both alluvium and bedrock. They are likely to be related to bedrock faulting.

Group 8: stream recharge water from the May 1996 flood is notably fresh with EC 280 µS/cm and of Ca, Mg, Na-HCO_3,Cl type (data from 2011 flood is not included here).

The dominant cation in most alluvial groundwater, stream flow and basalt groundwater is Mg^{2+}, and the dominant anion in basalt groundwater and stream flood water is HCO_3^-. The abundance of these two ions is potentially from the dissolution of olivine, in either the basalts of the Main Range Volcanics, or in the basaltic gravel of the alluvium. The most saline waters of the catchment, however, are from bedrock and are of Na-Cl type. In many cases there has been strong evaporation of slow recharging waters.

3.6.4 Statistical Grouping of Groundwater Chemistry

Statistical approaches to assess groundwater chemistry have been applied to facilitate identification of different hydrogeological processes. Based on this, groundwater samples have been grouped by their aquifer type and geographical location, and the dominant influence over their hydrochemistry, such as (a) basalts, (b) GAB sedimentary formations or (c) mixing (e.g. Cox *et al.*, 2012 unpub.; Raiber *et al.*, 2012). Within the Lockyer catchment, seven clusters, or groups, have been identified based on Hierarchical Cluster Analysis (M. Raiber, unpub. data, 2012), summarised as follows in Table 1, and are shown in Figure 19.

Table 1: Chemical values for different clusters.

Cluster	pH	Ca	Mg	Na	K	Cl	HCO_3	SO_4	NO_3	EC (uS/cm)	HCO_3/Cl
1	7.6	230	305	505	4.9	1730	511.5	110	20.9	5610	0.3
2	7.9	94	110	118	2	390	441	32	15.9	1940	1.1
3	8.0	68	84	140	2.5	320	465	12	1.4	1670	1.5
4	8.1	48	40	54	1	85	305	9	3.1	790	3.6
5	7.5	20	14	46.5	2.9	82	115	5	2.4	525	1.4
6	7.9	83	148	599	6	1100	604	95	1.3	4465	0.5
7	7.9	22	13.5	635	24.2	253	1315	10	2	2700	5.2

Ionic concentrations as mg/L

Development of 3D hydrogeological models of the bedrock formations in the catchment (M. Raiber, unpub. data, 2012) have confirmed that the Gatton Sandstone is the most widespread formation of the valley floor, and most alluvial aquifers are deposited onto it. The sandstone, however, thins to the north, where the underlying Woogaroo Subgroup interfaces with the alluvium. The Woogaroo formation forms a good aquifer, due to higher permeability than the other bedrock formations and fresher groundwater.

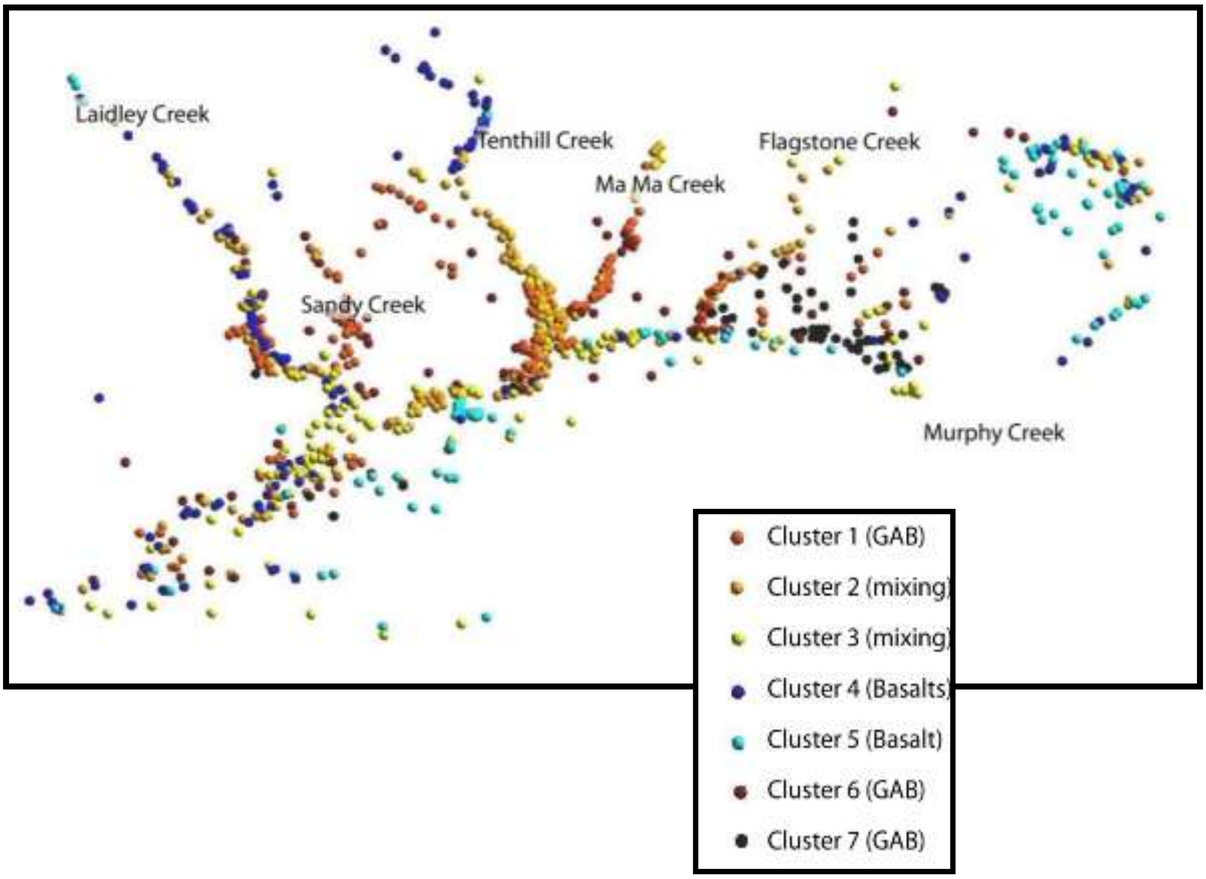

Figure 19: Hydrochemical Cluster Analysis (HCA) groupings for alluvial groundwater in the Lockyer catchment, with interpretation of hydrochemical origin shown in brackets. These groups relate to the chemical summary in Table 1.

3.7 Groundwater Levels and Recharge

Recharge to the alluvial aquifers is largely from the surrounding basalt-capped ranges, with stream flow and seepage into the alluvium via loosing streams in wet periods. Locally, particularly in the main valley, there is also deep drainage from irrigation (in some locations this is ponded) and bedrock seepage into the base of the alluvium. This seepage is more dominant with continued alluvial groundwater pumping during extended dry periods. The long-term records of groundwater bore hydrographs clearly display the relation to rainfall, and to irrigation pumping. Long-term trends, for example 50 years, reflect overall decreasing water levels for the alluvial aquifers, commonly of more than 10 m. By the end of the drought in 2008, in many sections of the lower valley only around 5 m of water remained in the basal aquifer, for example Figure 20.

Current studies by Raiber and Cox (unpub. data) following the flooding of Lockyer Valley in early 2011 show that even under high stream flow events, the alluvial aquifer system does not immediately recharge to the full extent in some parts of the Lockyer Valley. This is shown in Figure 20, for a profile across the main Lockyer Creek alluvium, where post-flood levels are compared to pre-flood (drought) levels. In many such cases elevated alluvial groundwater levels are typically observed close to the creek channel, and recharge does not always extend distant from the creek.

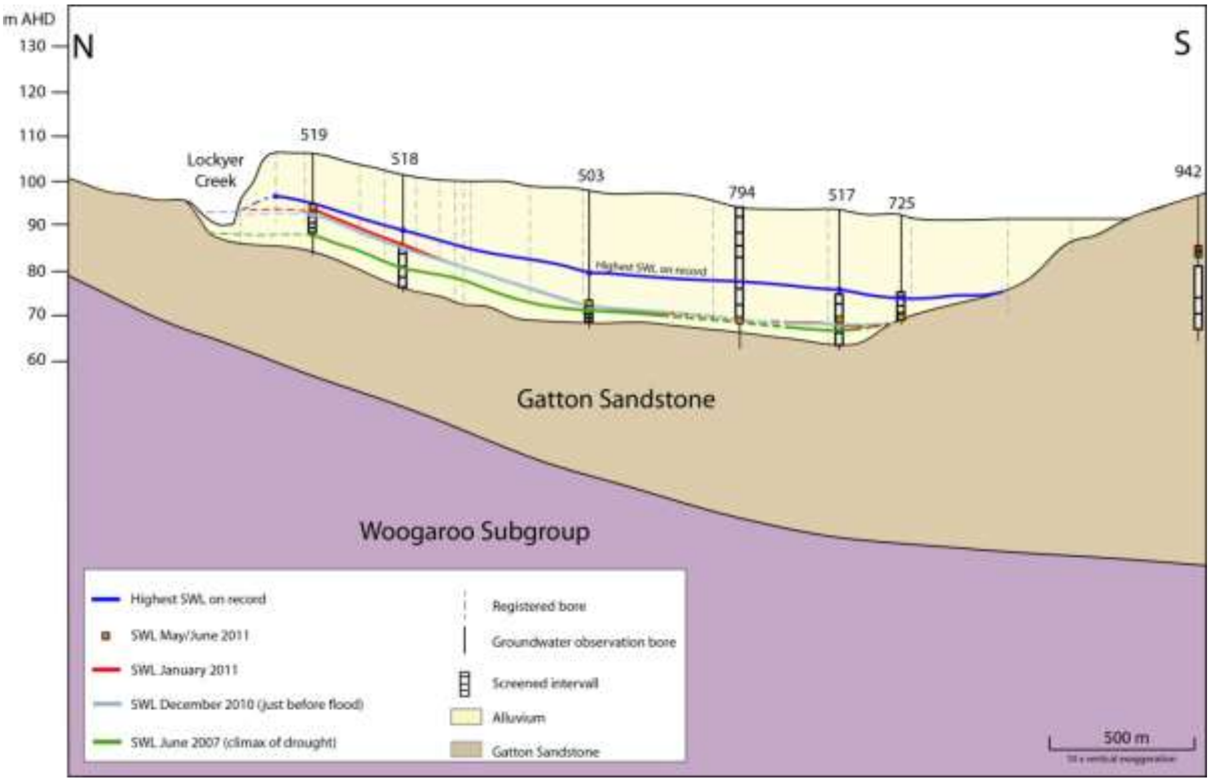

Figure 20: North–south cross-section of central section of main Lockyer Creek showing alluvial groundwater levels pre- and post- January 2011 flood. A series of bores are shown with numbers. The relationship of the Gatton Sandstone and alluvium is shown, as is the shallowing of the Woogaroo Subgroup northwards.

The lateral movement of groundwater recharged from creeks across the flood plain in this area is probably inhibited by bedrock highs and thinning of the aquifer. In other areas, it was observed that although water levels may rise in bores further from the creek, groundwater chemistry and isotopic signatures in some cases did not change. The latter fact suggests that in these areas, the water level rise is a pressure response from the high creek level rather than a mixing of creek water with the older alluvial groundwater. Figure 21 displays three bores from the cross-section. Bore #519 closest to the creek clearly shows recharge from rainfall related flow; #517 slows some response to the early 1990's after which abstraction dominates; and #725 bottomed on a bedrock high has been pumped dry from around 2001.

The morphology of the drainage system and the alluvial infill is obviously a factor. Alluvial aquifers in the major sub-catchments of Laidley Creek and Tenthill Creek recharge readily in their upper sections, as they extend closer to the headwaters and are typically composed of coarser material. The central, wider sections of the valley where the Lockyer Creek alluvium is deepest receive good recharge from high stream flow events, usually with an improvement in water quality. However, in many smaller catchments (e.g. Ma Ma, Flagstone, Plainhill and Blacksnake Creeks) flood-related recharge was limited and water quality remains poor.

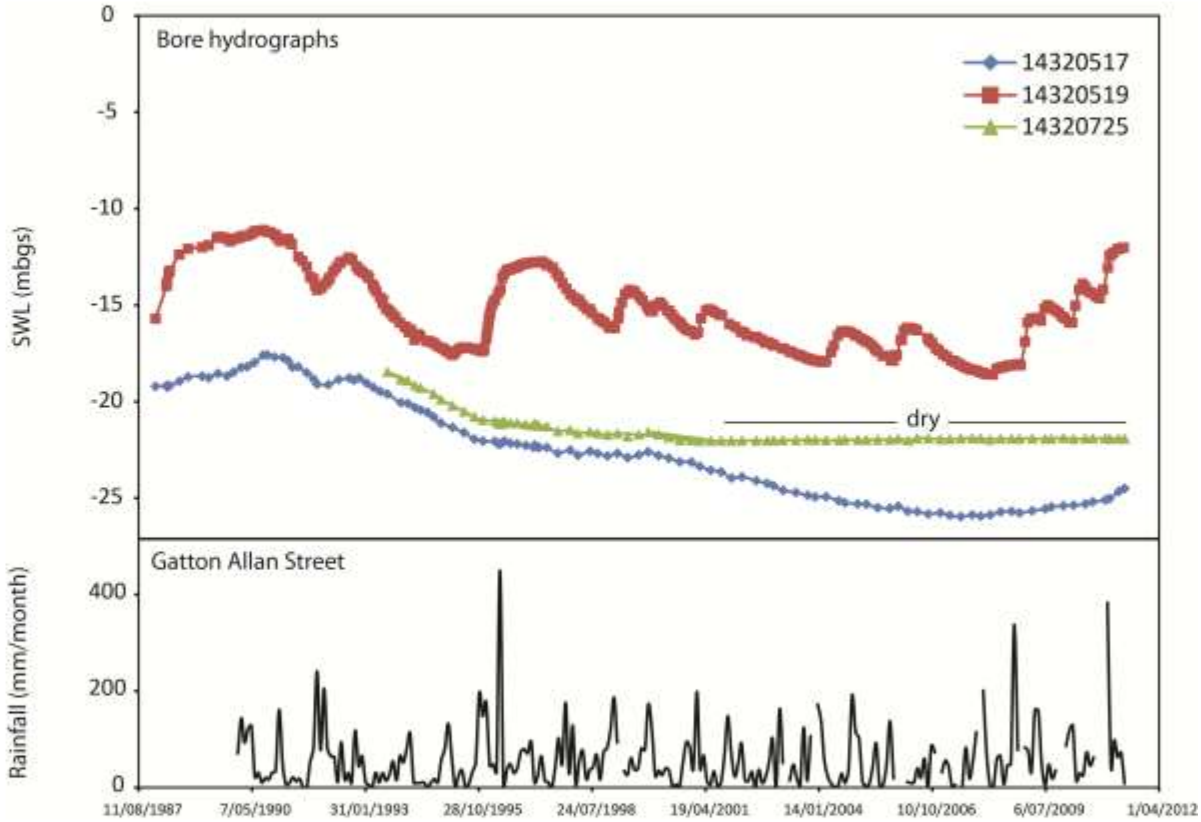

Figure 21: Bore hydrographs lower Lockyer Valley alluvium for period from 1987 to early 2012 which demonstrate the relation of distance from creek and groundwater level. Clear correlation with recharge and rainfall can commonly be seen for the creek proximal bore.

An overall conclusion from the studies is that the hydrological systems of the creeks, the alluvial aquifers and the bedrock aquifers are connected but have spatially and temporally variable interactions in different sub-catchments of the Lockyer Valley. Bedrock morphology is an important factor and these processes can be identified by monitoring of water levels and hydrochemistry. Mixing ratios identified from hydrochemical investigations can serve to constrain numerical groundwater modelling exercises.

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4. SOIL PHYSICAL ASSESSMENT OF PRW APPLICATION TO LOCAL SOILS AND LITERATURE REVIEW ON APPLICATION OF DESALINATED WASTEWATERS IN AGRICULTURE

(Tim Ellis and Leif Wolf)

This chapter addresses the possible effects on soil structure of the use purified recycled water (PRW) for irrigation and shows that chemistry governing these effects is complex and varied. The structural changes are dependent on the chemistry of the soil, the irrigation water, the irrigation history and method of application. As a consequence the exact effects are therefore difficult to predict. After applying a number of standard and non-standard measurements of likely clay dispersion, which are detailed in this chapter, the results remain somewhat inconclusive. However, we consider what might be the worst-case scenario (overhead irrigation following a history of saline groundwater application) and estimate what measures could be taken to ameliorate the effects. Given that the PRW is very pure, it is likely that any soil structural problems that occur will not differ significantly from those that are reported to occur following rainfall on slightly sodic soils that have previously been irrigated with saline water. The latter rainfall-induced problems have been managed successfully by farmers for some time.

4.1 Introduction

The productivity of clay soils can depend heavily on their macro structure, which affects water entry, water movement and the growth of plant roots. Some soils are predisposed to structural degradation when exposed to raindrop impact, tillage or the application of irrigation. These changes can range between slight (a lowering of productivity, requiring careful management) and severe (complete, irreversible collapse of structure and loss of productivity). It is important, therefore, to be aware of any potential problems associated with the application of water of varying quality to soils that might be predisposed to degradation. At the root of such problems is the combination of clay chemistry and the chemistry of the irrigation water.

The chemistry of clay and the attendant physical ramifications are complex. Most of the relevant chemistry pivots around the relative abundance of the sodium cation Na^+, and its ability to displace calcium and magnesium cations (Ca^{2+} and Mg^+) in the clay mineral interlayers. Other ions can be involved, but these are the main players. Because of the pivotal role of the sodium ion, the term 'sodicity' is used to describe the extent to which a soil is predisposed to chemical and physical changes. Further, because of the role of the sodium ion, the salinity of the soil (measured by the electrical conductivity [EC] as a proxy for salinity in a 1:5 soil:water extract) and the salinity of any applied water (direct measurement) are used to assess the mechanical behaviour of clay soils.

By structural degradation we generally mean a change in structure resulting in a lowering of pore volume and accompanying properties such as infiltration, hydraulic conductivity on wetting (thus changing local hydrology and water balances) and an increase in soil strength on drying (affecting tillage, seedbed preparation, seedling emergence and plant root growth). Structural degradation is often associated with a reduction in hydraulic conductivity. At high EC, the structural change is typically soil swelling; at low EC it is typically dispersion (Emerson and Bakker, 1973; Quirk and Schofield, 1955).

Slaking and dispersion are the main processes by which soils lose their structure on wetting. Slaking results from the rapid wetting of soil aggregates and the associated explosion of entrapped air and differential swelling of micro aggregates (Ruiz-Vera and Wu, 2006). This is a mechanical process rather than chemical, but pathways for water movement can be blocked by mobile particles or micro-aggregates – it is a predecessor to dispersion, but is reversible on drying and structure is typically maintained. Dispersion is the chemical dissociation of clay particles into suspension, the blocking of pores and the collapse of micro and macro structure. Dispersed soil typically sets hard on drying. This tends to be irreversible without chemical intervention such as the addition of ameliorants (e.g. gypsum: calcium sulphate). Clay dispersion can occur spontaneously (as the clay wets) or only as a result of mechanical disturbance such as raindrop impact, tillage, wheel traffic or trampling by livestock.

Clay soils, because of their fine particle size (<0.002 mm), plate-like shape and large surface area are chemically very active. Clay particles are principally phyllo-silicates. They have a residual negative charge and therefore hold positively charged ions (cations) between them. The space between clay particles is dependent on the distribution of electrostatic charges between the particles, called the 'diffuse double layer'. The full complement of forces acting at the clay particle scale are myriad, complexly related and cannot be fully described here, but we recommend the work of Quirk and Murray (1991) for a detailed account of these.

Because of the different mineralogies, some clays are more reactive than others. Laird (2006) gives a detailed description of the diffuse double layer in montmorillonite, one of the dominant minerals of the vertosols on the Lockyer Valley, but we provide here only a very brief description of the governing principles. The attachment of cations to clay particles is a result of the balancing forces of electrostatic attraction towards, and diffusion away from the particle surfaces. The former is related to the size (valence) of the charge of the cation and the latter is related to the osmotic pressure within the soil solution, that is, the tendency for ions to migrate away from the particles and into solution. A simplistic view of 'sticking' of particles together, therefore, occurs as two (or more) negatively charged clay particles 'sandwich' and 'share' a distribution of cations. The thickness of the double layer is inversely proportional to the salinity of the soil water solution (measured as electrical conductivity; EC dS/m) and inversely proportional to the square of the charge on the neutralising ion. If the dominant cations are Ca^{2+} and Mg^{2+}, the particles are held together much more strongly than if Na^{+} is dominant. This 'propensity' to dispersion is typically expressed as the Exchangeable Sodium Percentage (ESP) or the sodium adsorption ratio (SAR).

$$ESP = \frac{(100 \times exchangeable\ Na)}{\sum(exchangeable\ Ca + Mg + Na + K + Al)} \quad \textbf{(1)}$$

$$SAR = \frac{[Na^{+}]}{\sqrt{\frac{[Ca^{2+}] + [Mg^{2+}]}{2}}} \quad \textbf{(2)}$$

Where [ ] signifies concentrations in mmolc/L. ESP and SAR can be easily related under a wide range of conditions (Qadir and Schubert, 2002).

ESP is often used to categorise a soil's 'sodicity'. Helliwell and colleagues (2001) point to the work of Sumner (1993) for rough guidelines for labelling a soil as 'sodic': in Australia, soils with ESP ≥ 5; in USA, soils with ESP ≥ 15, because a higher EC tap water was used for the ESP determination. An alternative index is PAR (or KAR; similar to Equation 2), which can be used where the dominant monovalent cation is potassium K^{+} rather than Na^{+}. A more encompassing index, MCAR (monovalent cation adsorption ratio, Smiles and Smith, 2004); can be used, which is the summation of SAR and PAR. Jayawardane and colleagues (2011) describe the previously undocumented index of Rengasamy (unpub.), CROSS, which accounts for the relative flocculating powers (or inverse dispersive powers) of the major ions. They found this to be a more useful indicator for relative change in hydraulic conductivity with change in electrolyte concentration, but this is only half the story. It is these indices, in combination with the water SAR (in most cases SAR can be related to EC, as Na^{+} is typically the dominant ion), that determines whether or not soil structure degrades. The more saline the water, the less likely is dispersion as osmotic pressure is greater; hypersaline water is typically clear, as all sediment will flocculate and settle. In addition, because there is no clear threshold between coagulation and spontaneous dispersion, there exists a continuum ranging from non-dispersive, to tendency to disperse, through to spontaneously dispersive. Soils that do not spontaneously disperse can become dispersive when mechanically disturbed, for example, by raindrops – this process is called 'mechanical dispersion'.

Despite myriad studies and reviews on this topic, it is difficult, even for experts, to predict structural changes in clays for specified clay and water chemistries. Indeed, there are numerous indices for estimating the risk of structural degradation if water of a given salinity is applied to a given soil, and all are indicative only, for example Ayers and Westcot (1985) and Rengasamy (2002). As a starting point, Helliwell and colleagues (2001) provide a succinct background on the issues of concern to us here: the use of recycled water for irrigation. We are interested, in particular, in the likely effects of applying PRW to sodic soils in the Lockyer Valley. The PRW, compared to most recycled effluent, has a very low EC (0.2 dS/m, compared to local groundwater of up to about 6 dS/m) and low SAR (2.7, compared to local groundwater, up to 32). The soils of interest have SAR ranging between 0.02 and 0.33, with ESP between 0.4 and 4.5% and EC (1:5 water) between 0.4 and 4.5.

4.2 Water Quality and Soil Structural Decline

4.2.1 The Application of PRW to Soil

There is a very rich body of literature on the (often unwanted) effects of irrigation on sodic soils and several sub-topics, for example: sodic soils (Sumner, 1993); irrigation and sodicity (Rengasamy *et al.*, 1992); clay dispersion (Rengasamy, 2002); soil salinity and sodicity (Rengasamy, 2006). Jayawardane and colleagues (2011) look in greater detail at the effects of cation (positively charged ion) combinations on the hydraulic conductivity of soils. There are useful reports on the effects of effluent or recycled water to agricultural soils. Helliwell and colleagues (2001) review the effects of the main player, sodium, when irrigating with wastewater; Balks and colleagues (1998) document the effects of sodium accumulation following irrigation with effluent; Sepaskhah and Sokoot (2010) and Bhardwaj and colleagues (2008) specifically address the effects of dispersion on saturated hydraulic conductivity. Hulugalle and colleagues (2006) specifically deal with changes in an Australian Vertosol following effluent irrigation, and Stevens and colleagues (2003) discuss the potential dangers of irrigation with saline and high SAR water on the northern Adelaide Plains. As discussed in the Introduction, structural degradation can be exacerbated by mechanical disturbance, and this can form surface sealing and reduced infiltration (Ben-Hur *et al.*, 1998; Pilatti *et al.*, 2006; Shainberg *et al.*, 1992). For a comprehensive review of the interactions of clays, sodicity and salinity and their effects on soil structure and water movement, we recommend to readers Ezlit's (2009) thesis.

However, there is relatively little mention in the literature of the effects of the application of PRW for irrigation. Lahav and colleagues (2010) cite specific problems related to the use of desalinated seawater, particularly with high SAR in irrigation, can lead to soil dispersion and the associated structural and hydrologic problems. Yermiyahu and colleagues (2007), however, lament the nutrient deficiency of desalinated sea water – useful ions and micronutrients are removed in the demineralisation process just as effectively as the undesirable ones, and the authors urge that water purification guidelines be reassessed. Tarchitzky (2004) provides a brief review of the potential damage to soil and crops by irrigation with recycled waste water; included in these are sodicity and hydrophobicity. Leaching of clay fractions through the soil profile is another potentially damaging result of irrigation of sodic soils. In a coarse soil type, clay can be dispersed from upper layers (which then become depleted and infertile) and can lead to clogging of lower layers, restricting water movement (Warrington *et al.*, 2007).

Several studies have dealt with possible effects on soil chemistry and structure with the rate of change in the quality of irrigation water. The application of PRW following irrigation with more saline groundwater could cause structural breakdown in sodic clays. The studies most relevant for us here highlight importance of the irrigation history and the abruptness of changes in the chemistry of irrigation water. Although conducted in coarse-textured soils, Keren and Ben-Hur (2003) leached highly sodic soil columns with solutions gradually decreasing concentrations of calcium carbonate (from 500 mmol/L), down to deionised water (gradual leaching). In this case, hydraulic conductivity decreased gradually to a steady state value and there was no dispersed clay observed in the leachate. This was contrasted with another test with an abrupt change in calcium carbonate from 50 mmol/L directly to deionised water. In this case (abrupt leaching) resulted in a larger reduction in hydraulic conductivity and dispersed clay was observed in the leachate.

Bauder and colleagues (2008) undertook wetting and drying tests with saline and sodic water, followed by leaching with distilled water to simulate a rainfall event. The study reinforced the supposition that rainfall following irrigation with saline water can lead to dispersion in clay soils. Investigation of the behaviour of Na and Ca in the exchange complex showed lower hydraulic conductivity (and more dispersed clay in the leachate) in separate soil columns with progressively lower EC leaching water, and SAR of 15 (Dikinya *et al.*, 2007). Erratic changes in hydraulic conductivity were interpreted as the breakdown of macrostructure and ped swelling. Suarez and colleagues (2008) investigated the effects of irrigation waters with SAR 2, 4, 6, 8 and 10 $mmol^{1/2}/L^{1/2}$ and EC 1 and 2 dS/m, alternated with simulated rain (deionised water – 0.016 dS/m – from a sprinkler). The authors described in detail the effects of the increasing SAR on reducing soil hydraulic conductivity. Although alluding to the effects of subsequent 'rainfall' of low EC exacerbating dispersion, this issue is not explored as fully as we would hope.

In summary, there are abundant reports of the deleterious effects on the structure of sodic soils following the application of sewage, wastewater and treated wastewater. Some of these effects are attributed to mechanical blocking of pores by suspended solids. Most of the effects are related to combinations of soil ESP and the (low) EC and/or (high) SAR of the applied water. There are a small number of studies describing surprisingly poor results from applying desalinated seawater to agricultural soils. One study reports nutrient deficiencies in plants due to the highly demineralised water; another report warns of potential soil structural decline from irrigation with low EC but high SAR water. We found no reports of the use of PRW of the very high quality (potable) that we are dealing with in this study being applied to sodic soils. A small number of studies confirm that applying distilled water can disperse sodic clays which have previously experienced irrigation with a higher EC. The authors point to this as evidence that rain events could cause dispersion in these cases and, in fact, we have observed, and there is anecdotal evidence for, exactly this result in the Lockyer Valley. We note that the application of distilled water (probably with EC < 0.01 dS/m) might be an approximation of the effects of the application of PRW in these cases.

4.2.2 Methods for Measuring Soil Structural Degradation

Good soil structure implies the existence of peds, crumbs and pores and the associated interstices that allow the passage of water and nutrients and the growth of plant roots. As discussed above, at the heart of this structure is the chemical bonding of clay particles and, in particular, the exchange of sodium, calcium and magnesium ions between clay particles and the soil water solution. It follows, then, that structural degradation begins with the swelling of clays (as particles become less tightly bound to each other) and ends in the disintegration of groups of clay particles.

The most direct measurements of structural degeneration are therefore measurement of (changes in) hydraulic conductivity, surface hydrology (e.g. infiltration; overland flow) or both, and overland flow on degraded soils necessarily results in surface erosion. Experts in this field will generally agree that the 'best' measurements will be those made under the situations of interest (e.g. on undisturbed soils during a rainfall or irrigation event (Suarez *et al.*, 2008). While such studies are undertaken, the practical difficulties and the large resources required make them relatively rare. Some of the most extensive and famous uses of large-scale field rainfall simulation have formed the basis for the widely used USLE erosion index (Wischmeier and Smith, 1965; 1978). Although not the central objective, such studies often measure Hortonian overland flow (Horton, 1933), which can (although not necessarily) result from structural degradation at the surface. Ellis and colleagues (2006) describe the use of a 60 m^2 rainfall simulator to measure overland flow, and Leguédois and colleagues (2008) describe the effects on sediment transport, partly from structurally degraded zones. Rainfall simulators need not be so large and can provide useful answers when only ~m^2 are considered (e.g. Colloff *et al.*, 2010), and lysimeters can also provide opportunity for measurements of soil structural transitions (Warrington *et al.*, 2007). Rainfall simulators can be constructed in laboratories, but at this scale they are invariably applied to disturbed soil samples (e.g. Hairsine *et al.*, 1999). Wet sieving of aggregates – typically on disturbed samples – gives a measure of aggregate breakdown under the action of mechanical disturbance by water (e.g. Hignett *et al.*, 1995, applied by Ellis *et al.*, 2011). This can result from slaking or dispersion, and in the latter case the results are typically dramatic, leading to a massive structure on drying. Soil columns lend themselves much better to controlled laboratory experiments and can similarly be used to measure aggregate breakdown under rainfall (Legout *et al.*, 2005). In the context of structural degradation and clay dispersion, columns are typically used to measure the change in saturated hydraulic conductivity (K_{sat}) of disturbed samples (Dikinya *et al.*, 2007; Sepaskhah and Sokoot, 2010; Browning *et al.*, 2007; Bauder *et al.*, 2008), but occasionally undisturbed cores are used (e.g. Balks *et al.*, 1998). Detailed descriptions of methods for hydraulic conductivity measurement can be found in McKenzie and colleagues (2002) and Bond and colleagues (1998).

4.2.3 Scope for Predicting Possible Structural Degradation

Because the above field and laboratory measurements of the effects of structural degradation are laborious, specialised and costly, there have been various attempts to develop easily applied indices to quantify the propensity of a soil to clay dispersion. Emerson (2002) provides a contemporary account of his original dispersion indices devised for assessing the suitability of clay soils for building foundations. The Emerson tests fall into two categories: 1) the introduction of air-dry peds to distilled water to observe any spontaneous slaking or dispersion; and 2) the remoulding of wet peds and reintroduction to distilled water to test susceptibility to mechanically induced dispersion. Rengasamy and colleagues (1984) developed a less direct but more quantifiable method for measuring the amount of clay in suspension following agitation, compared to the amount of clay in suspension from complete chemical dispersion (see Wöbcke, 2009 for the methodology). Rengasamy (2002) described several categories of structural degradation and used this to 'map' zones in the EC – SAR (of the soils) space which can be used to approximate the risk of dispersion in clays. A similar exercise was previously undertaken by Ayers and Westcot (1985) to characterise water chemistry that is likely to cause structural problems on irrigated soils. Suarez and Šimůnek (1997), in their model HYDRUS 1D, use a model based on the clay swelling model of McNeal (1968) to predict changes in saturated hydraulic conductivity (K_{sat}), with changes in water chemistry for a given soil chemistry (SAR-EC-K_{sat} parameter space). Ezlit (2009) evaluated this model with soil column tests in the laboratory and showed it performed poorly for a Vertosol and a Sodosol (Isbell, 1996). In an attempt to better quantify all the variables, and to provide more specific guidelines for the use of saline mine effluent, Ezlit (2009) provided a modified version of the model but also found it performed rather badly, possibly because of the lack of representation of the role of pH. Similar investigations of the SAR-EC-K_{sat} parameter space and produced a 3-dimensional surface for each soil, from which they arbitrarily chose a contour in the SAR-EC plane, representing K_{sat} equal to a 20% reduction from that determined using a 'normalising' solution of calcium chloride (pers. comm. S. R. Raine, University of Southern Queensland, Australia). The 20% value was possibly adopted from Quirk and Schofield (1955), who defined a threshold electrolyte concentration at which a 20% reduction in hydraulic conductivity occurs, for a given ESP. The new relationships were then provided to the clients to guide decisions concerning the limits to SAR and EC combinations of the effluent water that could be applied to the soil without causing structural damage. This allowed estimates of the most appropriate 'shandies' of effluent water of different qualities for irrigation.

The predictor of the effects of combinations of specific water and soil chemistries is to apply the water that will be used to the soil in the field. What is consistent among the above studies and the opinions of other experts (e.g. R. S. Murray, University of Adelaide, Australia) is that indices of dispersive behaviour can only be described as 'rough guides' and that the exact behaviour of specific soils in the presence of specified water chemistry is very difficult to predict (Helliwell *et al.*, 2001). The Australian guidelines for the use of recycled water (EPHC, 2009) specify in general terms that the appropriate application of wastewater for irrigation must take into account any potential damage to soil structure. However, only general guides such as those of Ayers and Westcot (1985) are cited.

Quirk (2001) also observed there is a widely held misconception that the EC required for flocculation of dispersed clay is the same as the EC required to maintain structural integrity of a soil. However, the dispersion process involves the face-to-face swelling and separation of clay particles, whereas flocculation involves end-to-face attraction and coagulation of clay particles. These processes are quite different: they involve different forces and are not the inverse of each other.

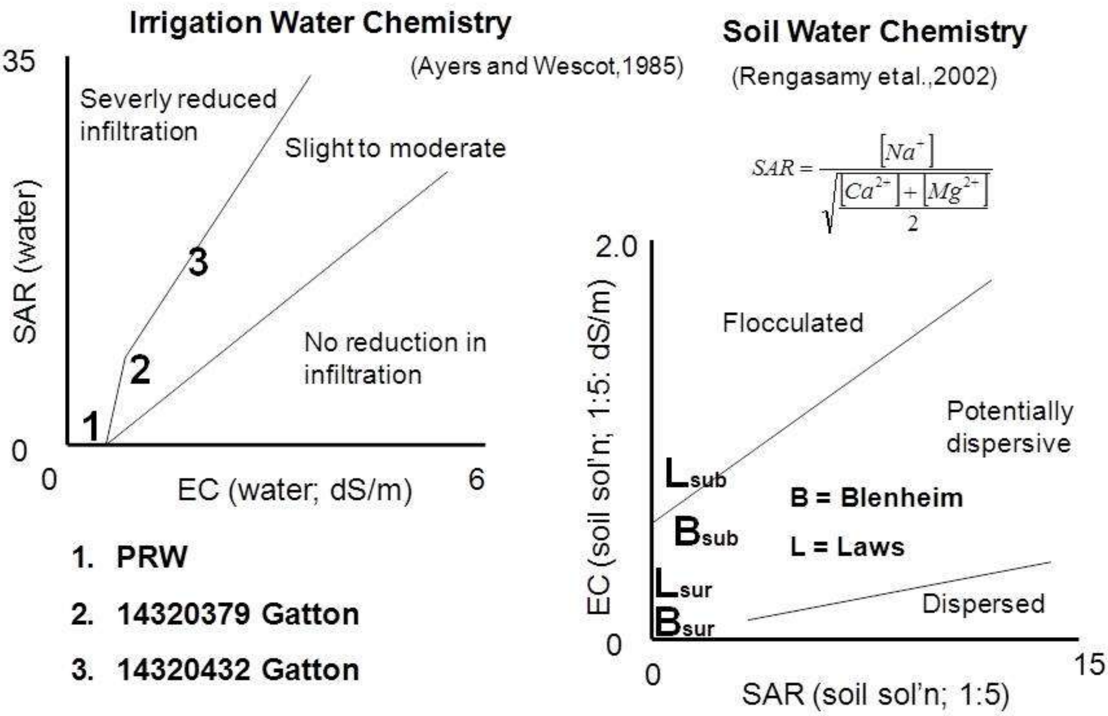


Figure 22: The application of rough guides for the suitability of irrigation water (Ayers and Westcot, 1985; left) and the susceptibility of soils to structural degradation (Rengasamy, 2002; right). Here we have plotted soil and water chemistries from the Lockyer Valley within the SAR and EC parameter spaces. L and B refer to Lawes and Blenheim soils respectively, and subscripts 'sur' and 'sub' indicate surface and subsoil respectively.

Figure 22 provides some guidance regarding: 1) the likely risks of applying local groundwater and PRW to soils in the Lockyer Valley (Ayers and Westcot, 1985); and 2) the propensity of some Lockyer soils to disperse (Rengasamy, 2002). (See also Threshold EC [Rengasamy *et al.*, 1984] and Dispersive Potential Index [Rengasamy and Olsson, 1991].) The approximate nature of these guides augers badly for any precision regarding predicting the effects of specific combinations of soil and water. We also describe in the second part of this study (Emerson, 2002) and modified Rengasamy and colleagues (1984) tests on Lockyer soils undertaken in an effort to predict the likely effects of applied PRW. There are other complicating factors surrounding the presence and type of organic matter, but this typically does not improve structural reactions at the clay particle scale. The best use for organic matter in preventing structural degradation is for protecting the soil surface from raindrop impact, although some improvements due to biological activity have been noted (Valzano *et al.*, 2001).

Overall, changes in soil structure and hydraulic properties are very difficult, if not, impossible, to predict with our current knowledge. The best available and practically applied information for our purposes include the guides produced by Rengasamy (2002) and Ayers and Westcot (1985), although there are others. Moreover, we have really only considered soil changes under saturated conditions. Under unsaturated conditions, results are even less predictable because of the changes in pore size distributions.

4.2.4 Managing Soil Structural Problems

The best management of potential soil structural problems is avoidance: that is, avoid potentially troublesome chemistries of irrigation water (i.e. SAR/EC ratios; Ayers and Westcot, 1985) or potentially problematic soil chemistries (i.e. EC/SAR ratios; Rengasamy, 2002). We note, again, that there are not, and not likely to be, any simple guidelines for predicting the effects of combining specific soils and waters.

The second most important strategies are to avoid mechanical disturbance on the soil in a moist or wet state, which, for susceptible soils, can lead to mechanical dispersion (see Introduction). The mechanical energy of raindrop impact significantly increases the risk of clay dispersion and structural breakdown in all soils, but is most pronounced in sodic soils. The only method available to farmers for avoiding raindrop impact is to ensure that soil surfaces are covered, as much as possible, by stubble or crop residue, to absorb the energy of raindrops. It follows that overhead irrigation similarly increases the risk of mechanical dispersion and that soil cover should be optimised. We note, however, that many of the horticultural husbandries in the Lockyer Valley require fine tilth and 'clean' seedbeds (i.e. free of crop residue) and are not conducive to maintaining soil cover. Similarly, soil disturbing actions (e.g. tillage; wheel traffic; animal traffic) should also be minimised and especially avoided on wet soil. Chemical methods such as the application of calcium sulphate (gypsum) prior to rainfall or irrigation (Brinck and Frost, 2009) – or in fact dissolved in the irrigation water – increase the opportunity for Ca^{2+} to occupy sites on clay particles, rather than Na^{+}.

Our principal concern in this study is the possible effect of the application of low EC, low SAR PRW to sodic soils previously irrigated with saline groundwater. The EC of the groundwater counteracts, to some extent, the diffusion (via osmotic forces) of ions away from clay surfaces and allows the electrostatic attractive forces to be more persuasive and bind the clay particles. If PRW were to be applied, there would be a risk of the reduced EC of the soil solution increasing osmotic forces, leading to clay swelling and dispersion. We have not applied PRW in field experiments in the Lockyer Valley; however, this is supported by observations of clay dispersion on sodic soils following rainfall (particularly summer storms; pers. comm. C. Henderson, Principal Horticulturist, Department of Agriculture, Fisheries and Forestry, Gatton, Australia.) and the application of low EC surface-harvested irrigation water. It is interesting, however, that at least one farmer in this situation uses the low EC surface water for irrigating early-growth stages of crops (particularly beetroot), before reverting to irrigation with higher EC groundwater, as beetroot plants are tolerant of this in later growth stages. This is counter to the expectation that the low EC water is more likely to cause crusting and seedling emergence problems – apparently in this particular situation it is not of concern.

There are anecdotal reports that some farmers manage the potentially deleterious effects of low EC water on their sodic soils by blending or 'shandying' surface water with groundwater (pers. comm, Ashley Bleakley, Principal Project Officer (Hydrology), Water Services South East). If this method is successful, it is likely that it could also be used to reduce any potential deleterious effects produced by PRW.

4.3 Local Soil and Water Characteristics

4.3.1 Soils of the Lockyer Valley

Powell and colleagues (2002) have provided a comprehensive description of landforms and assessment of the suitability of soils for irrigation on the Lockyer Valley alluvial plains (Quaternary alluvium). Of major interest agriculturally are the mainly black and brown cracking clays and gradational clay-loams. Of specific interest to this study, because of their location with respect to delivery points for PRW, are the soils of the 'major stream terraces and plains', particularly Blenheim and Lawes soil classes (see Powell *et al.*, 2002; Table 4.4):

> Blenheim: Dark self mulching, cracking medium to heavy clay with dark, brown or grey calcareous subsoil to 1.5 m deep or over medium to heavy clay palaeosol.
>
> Lawes: Dark self mulching, cracking medium clay with dark or brown calcareous subsoil to 0.1 to 1.4 m deep over brown friable lighter textured layers.

These soils are dominated by smectitic clays, which are responsible for their swelling and cracking nature. X-ray diffraction analysis undertaken during this study confirms this mineralogy. Table 2 gives the clay mineralogy of the four soils samples (a, b, c and d) of interest from the major stream terraces and plains.

Table 2: Clay mineralogy of some soils of the major stream terraces and plains. D, CD, SD, M and T denote dominant (> 60%), co-dominant (sum of components > 60%), sub-dominant (20–60%), minor (5–19%) and trace proportions (< 5%), respectively.

Soil	Quartz	Goethite	Hematite	Kaolin	Smectite	Augite	Anorthite/ Andesine	Magnesite
(a) DPI Gatton Lawes soil	CD		T	T	CD		CD	T
(b) Super Sat site Blenheim soil	SD		T	M	D	T	SD	T
(c) Forest Hill 'sodic' site Blenheim soil	SD		T	T	CD	M	CD	T
(d) Forest Hill 0805 0806 Blenheim soil	CD		T	T	CD	T	SD	T

Fluctuating areas of secondary dryland salinity principally result from historical land clearing in the uplands, although some also have resulted from irrigation with saline groundwater. Powell and colleagues (2002) make some cursory but inconclusive comments regarding the susceptibility of these soils to salination by irrigation. They quote Shaw (1995) (although it appears they mean Shaw and Thorburn 1985) as indicating that there would be significant deep drainage in these soils. They then cite Shaw and Dowling (1985), who attribute the deep drainage to the smectite-rich clays and other macro-structural characteristics found in well-structured clay soils; this observation is echoed by Churchman and colleagues (1993). It is unclear what the implications of these observations are with respect to the potential application of PRW to these soils. It is possible to infer that the saline irrigation water in some way helps maintain the structure of the clay, which is consistent with the theory. However, the very high calcium content somewhat refutes this (Table 3).

Table 3: Summary of analytical data for soils of the stream terraces and plains.

Horizon	pH (1:5)	EC (dS/m)	Silt %	Clay %	CEC (cmol(+)/kg)	Dominant cations	ESP	CEC/%clay
A	7.0-8.3	0.03–0.24	16–29	31–62	21–65	Ca^{2+}, Mg^{2+}	1.5–5.9	0.63–1.1
B	7.0-9.3	0.09–0.94	11–35	40–74	31–71	Ca^{2+}, Mg^{2+}	1.2–26	0.67–1.2
Substrate	7.4-8.5	0.03–0.62	9–30	16–43	19–51	Ca^{2+}, Mg^{2+}	1.8–10	>1.2

Source: Powell *et al.* (2002, Table 4.3, p. 26)

Powell and colleagues (2002) also draw attention to the possibility of the Lockyer Valley soils being irrigated with recycled wastewater, as this has been a topic of discussion for some time. They also cite a report by Heiner and colleagues (1999) that was commissioned to determine the sustainability of agricultural systems in the Lockyer Valley and Darling Downs if they were to use recycled water. The water considered in this report was to be Class A (quoted EC 0.6–1.2 dS/m; SAR 3.2–6.1) rather than the highly purified PRW that is currently in the offing (see Table 5). Despite this, Heiner and colleagues (1999) determine that irrigation with the recycled water is likely to be sustainable in the Lockyer Valley provided soil salinity, sodicity and groundwater were managed and monitored. It appears that their conclusions rest heavily on the dissertation of Shaw (1995) (although the authors cite this work as Shaw 1996). Heiner and colleagues (1999) appear to have taken Shaw's sketch of SAR-EC parameter space, indicating zones of likely effects on soil permeability (similar to the figure of Ayers and Westcot, 1985) and performed calculations to estimate the maximal permissible SAR irrigation water ranging from 100% effluent, to 50%:50% effluent:groundwater to 100% groundwater. While this is a reasonable approach, Powell and colleagues (2002, p. 48), in alluding to this work, provide the following cautionary note:

> With so many variables, reliance on predicted salinity and dispersion effects should be done cautiously. Further critical examination of these interactions is required to determine the impacts of long-term irrigation management using saline water.

4.3.2 Other Relevant Soil and Water Characteristics in the Lockyer Valley

Four soils were chosen from the group ‘major stream terraces and plains’ identified by Powell and colleagues (2002) using the following rationale:

- They were likely to be receiving PRW, should it be made available for irrigation use.
- They were known to be relatively sodic for Lockyer Valley soils.
- They had displayed dispersive behaviour.

The relevant chemical properties of these soils are shown in Table 4.

Table 4: Chemical properties of the four soils sampled for this study.

Site and Soil	pH	EC (dS/m)	[Ca^{2+}] (meq/100g)	[Mg^{2+}] (meq/100g)	[Na^{+}] (meq/100g)	[K^{+}] (meq/100g)	ESP (%)	CEC (meq/100g)
(a) DPI Gatton Lawes soil	7.3	0.08	19.1	13.2	0.13	1.18	0.4	36
(b) Super Sat site Blenheim soil	7	0.77	9.97	11.7	1.02	1.37	3.8	27
(c) Forest Hill ‘sodic’ site Blenheim soil	7	0.18	17	12.4	1.77	0.66	4.5	39
(d) Forest Hill 0805 0806 Blenheim soil	7.6	0.12	18.9	18.1	1.61	1.75	3.4	48

For this study, we were interested in the likely behaviour of the four soils if PRW were to be applied as irrigation water. It was important to compare these results with a ‘standard’, such as the Emerson test (Emerson, 2002) and the Rengasamy test (Rengasamy *et al.*, 1984), which used demineralised water. In addition, we also compared the effects of local irrigation waters: Logan Dam water (surface harvested water) and groundwater from two bores (Voigt and the University of Queensland). Figure 23 shows the relative colours of the five waters and the effect of filtering suspended sediment from the Logan Dam water. Table 5 gives some relevant chemical properties of these waters and others sampled from the Lockyer irrigation district.

Figure 23: The five water qualities tested: five 1 litre bottles on left – PRW, Voigt bore, University of Queensland bore, Logan Dam and Milli-Q deionised; and three vials on right – clarified Logan Dam water after passing through a 0.45µm filter, unclarified Logan Dam and Milli-Q deionised water.

Table 5: Chemical properties of PRW and additional six waters from the Lockyer irrigation district.

	Collection date	El. Con. @ 25 ºC µS/cm	SAR	Ca^{2+} mg/l	Mg^{2+} mg/l	Na^{+} mg/l	Cl^{-} mg/l	Nitrate nitrogen as N mg/l	Mn mg/l	Fe mg/l
Australian Drinking Water Guideline Standard (2006)							250	50	0,1	0,3
PRW Bundamba	12.07.2011	213	0,80	27	0,4	16	13	1,8	<0.01	<0.01
PRW Bundamba	9.12.2011	263	1,40	24	0,2	25	26	3,1	<0.01	<0.01
PRW Bundamba	9.12.2011	262	1,40	24	0,3	26	26	3,1	<0.01	<0.01
PRW Bundamba	10.12.2011	244	1,30	25	0,3	24	24	3,4	<0.01	<0.01
PRW Bundamba	10.12.2011	244	1,30	25	0,2	24	24	3,4	<0.01	<0.01
Irrigation GW at UQ[1] Gatton	9.06.2010	1060	2,74	74,4	83,7	104	182	1,29	<0.01	<0.04
Irrigation GW close to FH	10.06.2010	6590	13,59	142	304	807	1770	43,4	<0.01	<0.04
Logan Dam Surface Water	9.06.2010	208	3,50	6,3	5,3	37	33,5	0,55	<0.01	0,15
Lockyer Creek	11.05.2010	380	1,45	21,7	16	28	42,7	<0.50	<0.01	<0.04
Bore 14320379 Gatton	GW-DB	0,85	7,07	54,2	52,4	51,6	---	---	---	7,8
Bore 14320432 Gatton	GW-DB	1,83	14,8	87	107	146	---	---	---	8

[1] University of Queensland

4.4 Laboratory Tests

4.4.1 Emerson Test

To determine which of the four soils (Table2; Table 4) were susceptible to spontaneous dispersion, we conducted Emerson dispersion tests (Emerson, 2002) in deionised (Milli-Q) water, PRW, Logan's Dam water (a surface water storage for irrigation) and two groundwaters from Voigt and the University of Queensland – see Table 5 for chemical properties of the waters used. No spontaneous dispersion was observed by any of the four soils in the five waters.

Figure 24: Emerson dispersion test of four soils (rows) x five waters (columns): air dry small (0.55–0.60 g) peds (left); slaked, 2 minutes (centre); and 23 hours after immersion in distilled water. Slaking was observed but no dispersion. The colour associated with the Logan's Dam water existed prior to the test (see Figure 23).

4.4.2 Mechanical Dispersion

To investigate the effects of mechanical energy on the dispersion, the 4 soil x 5 water combinations were tested as follows:

- ~1 g of air dry soil introduced to a vial of 50 ml of water
- vial shaken manually (reciprocated end-to-end) 100 times (~30 seconds)
- allowed to stand
- settling observed visually.

Figure 25 shows the appearance of the agitated samples between 4 minutes and 22 hours after agitation. As a general observation, all soils in the 'fresh waters' (PRW, deionised and Logan's Dam) remained suspended for a longer period, from which we inferred that the clay particles remained dispersed. The 'saline' bore waters (Voigt and UQ) appeared to cause flocculation and the supernatant was less turbid and became clear more rapidly. The 'sodic' soil (C), second row from the front, remained dispersed in PRW more so than the other soils.

Figure 25: Samples of the four soils (rows) x five waters (columns) showing settling 4 minutes (top left), 10 minutes (top right), 1 hour (bottom left) and 22 hours (bottom right) after agitation.

4.4.3 R1 Dispersion Ratio

The R1 test has been designed to indicate the tendency for a soil to mechanically disperse and quantifies this as the ratio of suspended sediments (measured by hydrometer) from mechanical dispersion to that from (complete) chemical dispersion. The procedure for this is described in detail in Wöbke (2009), who provides three references: Puri and Keen (1925), Quirk (1950) and Richie (1963). The laboratory procedure is similar to that described by Rengasamy (2002) and involves the mechanical dispersion of a small soil sample (by agitation) and the complete dispersion of a small soil sample by agitation in the presence of a dispersive agent, sodium hexametaphosphate. In both the 'mechanical' and the 'complete' cases, deionised water is used. R1 is calculated as follows:

$$R1 = \{[R^1_5 + 1.0 + 0.00746(T^1_5)^2 - 2.9767].100 \times 30\} / \{[R_5 + 1.0 + 0.00746(T^1_5)^2 - 2.9767 - 2.4].100 \times 50\} \quad (3)$$

Where R1 = dispersion ratio and the numerator relates to 'mechanical dispersion' and denominator relates to 'complete' or 'chemical dispersion'; R^1_5 = Initial hydrometer reading in aqueous suspension; T^1_5 = Initial temperature reading in aqueous suspension; R_5 = hydrometer reading @ 6 min. (g L^{-1}) in chemical reagent suspension; and T_5 = temperature reading @ 6 min. (°C) in chemical reagent suspension.

However, there is no mention of the calculation of, nor typical values for, R1 in any of the above references. Approximate ranges of 0.6 < R1 < 0.8 and 0.8 < R1 < 1.0 reportedly (pers. comm. Dave Lyons, Manager Soil Chemistry Laboratory, Queensland Department of Environment and Resource Management) delineate 'potentially dispersive' and 'very dispersive', respectively. The laboratories of Environment and Resource Sciences, Department of Environment and Resource Management, 41 Boggo Road Dutton Park QLD 4102 (contact Dan Yusaf) undertook the tests for our project. Because we were interested in the behaviour of the four soils in the PRW, the bore waters and the dam water, we had the R1 tests evaluated for each soil, but using each of the five waters listed in Table 5. These tests were replicated four times and the results are shown in Figure 26 as 'Modified R1'; in each case the numerator of Equation (3) was determined using the test water, whereas the denominator was determined using the reference deionised water. It should be noted that the denominator (chemical dispersion) will be equal no matter what water is used.

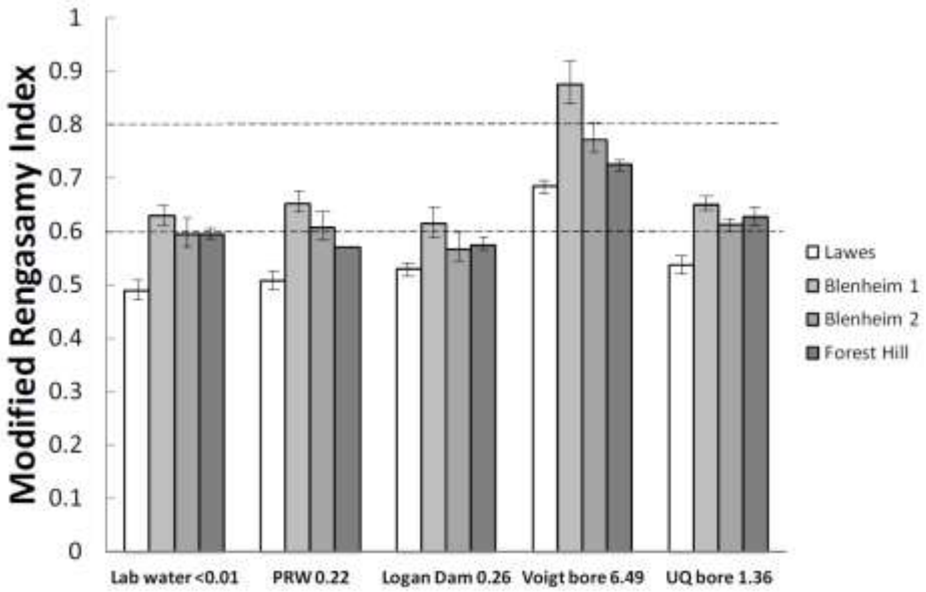


Figure 26: Modified R1 results for the four soils x five waters. Error bars show one standard deviation both sides of the mean of four replicates; numerical labels indicate the EC of each water.

4.4.4 Soil Column Tests

In addition to the tests described earlier in this chapter, five cores of 30 cm depth were taken from the major soil types of the Lockyer Valley and were used for soil column experiments where PRW was applied. Most of the sediments had quite a low permeability and just produced pore drainage water, therefore they were not suitable; however, the experimental set up was successful for the Lawes soil type and this column could be operated with flow rates of up to 450 ml per hour. The Lawes soil was operated with groundwater taken from a bore at Gatton (EC 1090 μS/cm) until steady-state conditions were reached. Then the feed water quality was changed to PRW, and within 8.5 hours the outflow was dominated by PRW at a flow rate of +/-80 ml/h. In the following days, the pressure to the system declined while the flow rate was kept constant. The decreasing pressure at a constant flow rate indicates an increase in hydraulic conductivity, which may result either from fine material leaving the column or rearrangement of clay particles in the column. Both processes are likely to be caused by clay dispersion (which can also support the transport of clay particles out of a sandy soil column).

Table 6 shows key water quality data for selected surface and groundwater water samples from the Lockyer Valley and the composition of drainage water from soil cores. Water quality data for PRW were shown in Table 5. In addition, PRW was tested for iron with less than 0.04 mg/l, lead with less than 0.05 μg/l and arsenic less than < 0.35 μg/l.

From the literature, it is apparent that the most likely combinations leading to soil dispersion are those with high SAR values and low electrical conductivity (EC). From Table 2, it is apparent that the surface water samples (Logan Dam and Lockyer Creek) taken have similar EC and higher SAR values than the PRW, thus comparing favourably for PRW. In summary, the water quality characteristics of surface water in the Lockyer Valley are similar to PRW. It is suggested that the current management strategies of farmers are sufficient to deal with the amount of soil dispersion expected from additional PRW.

Table 6: Water quality data for selected surface water and groundwater water samples from the Lockyer Valley and the composition of drainage water from soil cores.

	Sampling date	EC @ 25° C μS/cm	SAR	Ca2+ mg/l	Mg2+ mg/l	Na+ mg/l	Cl- mg/l	NO3-N mg/l	Mn mg/l	Fe mg/l	Pb μg/l	As μg/l
Australian Drinking Water Guideline Standard (2006)							250	50	0,1	0,3	10	7
Irrigation GW at UQ Gatton	9.06.2010	1060	2,74	74,4	83,7	104	182	1,29	<0.01	<0.04	0,15	0,81
Irrigation GW close to FH	10.06.2010	6590	13,59	142	304	807	1770	43,4	<0.01	<0.04	0,62	2,28
Logan Dam Surface Water	9.06.2010	208	3,50	6,3	5,3	37	33,5	0,55	<0.01	0,15	0,17	1,19
Lockyer Creek	11.05.2010	380	1,45	21,7	16	28	42,7	<0.50	<0.01	<0.04		
Alluvial Soil Core First flush	28.06.2010	1090	10,06	81,8	65,2	381	424	105	0,01	0,05	0,25	2,49
Blenheim Soil Core First flush	28.06.2010	4580	9,10	208	285	596	1410	29,5	0,05	<0.04	1,37	5,01
Creek Bed Sediment 1 pore water	22.04.2010	726	0,42	12,6	7,1	6	38,6	<0.50	3,78	0,28	0,12	1,97
Creek Bed Sediment 2 pore water	4.06.2010	310	0,94	58	23,9	28	47,6	<0.50	<0.01	<0.04	0,1	0,55
Lawes Soil Core flushed with PRW	20.08.2010	221	1,28	14,6	9,4	20	18,1	<0.50	<0.01	<0.04	<0.05	<0.35

In 32 samples analysed, arsenic concentrations were at or below 5 µg/l, compared to the Australian Drinking Water Guideline value of 7 µg/l. Cadmium guideline values were exceeded for drainage from creek bed sediment 1, with values up to 4.72 µg/l compared to the guideline limit of 2 µg/l. Lead was below 1 µg/l in 20 samples; one sample with fresh drainage water from a creek bed sediment showed 3.24 µg/l, still significantly below the drinking water guideline of 10 µg/l. Manganese concentrations range between 2 mg/l and 5 mg/l for seepage water from the creek bed sediment, thus exceeding the guideline value for drinking water of 0.1 mg/l. A second sediment core from the creek, however, did only exhibit manganese values up to 0.5 mg/l. Percolate from the alluvial soil showed manganese concentrations up to 0.16 mg/l.

4.5 Discussion and Conclusions

The use of high quality PRW for irrigation of crops carries the risk that it could cause structural degradation, via clay dispersion, of the soil, leading to lower porosity, hydraulic conductivity, water entry, seedling emergence and crop growth. This risk would be associated with sodic soils (high ESP). In the presence of the low EC PRW, the reduced osmotic pressures, together with the relatively wider diffuse double layer between clay particles, could lead to the spontaneous dispersion of clay. If conditions were not sufficiently severe for spontaneous dispersion, mechanical dispersion would be more probable. These reactions are more significant in clay soils dominated by smectitic clays, such as montmorillonite, due to their greater surface area and chemical reactivity.

The productive soils of the Lockyer Valley are largely Vertosols, rich in montmorillonite, and are irrigated mostly with groundwater of low to moderate EC. Some soils of the Lockyer Valley are considered to be slightly sodic, and minor dispersive structural degradation is regularly observed following summer storms (mechanical dispersion) on unprotected soil (no cover). This is consistent with clay chemistry theory; however, the significant body of literature covering this topic overwhelmingly reinforces that it is difficult, if not impossible, to accurately predict the effects of the application of water to sodic soils – except in extreme cases. There is a complex interplay between the clay chemistry, the water chemistry and the electrical conductivity of the water, particularly in relation to the presence and exchangeability of sodium ions. We can confidently predict, however, that low EC, high SAR water applied to sodic soils will lead to significant and irreversible structural decline. On the other hand, high EC but low SAR water will likely not lead to much structural change, but would reduce plant production for other reasons. If, however, low EC and low SAR water (e.g. PRW) were applied to moderately sodic soil, the response is difficult to predict. There are numerous guidelines that help determine the likely risk of soil-water-chemistry combinations, but they are of limited use for the case we are considering here. In addition, there is very little reported in the literature of the use of PRW for irrigation at all, although deleterious effects of desalinated, high SAR water are reported for a few cases.

We therefore undertook some laboratory tests using deionised water, PRW, local bore water and local surface water on four representative Lockyer Valley soils. These showed there was no risk of spontaneous dispersion with any of the soil–water combinations using the Emerson test (Emerson, 2002). However, mechanical dispersion was a risk in all cases. Two mechanical dispersion tests: 1) agitation and qualitative turbidity observation; and 2) a similar, quantified dispersion test (Wöbcke, 2009) showed some discrepancies. Results from (1) aligned with the theory that predicted a greater tendency for dispersion of slightly sodic soils, with decreasing EC but (2) showed the highest risk for dispersion was related to the combination of the highest EC (bore) water, for all soils. The latter contradicts clay chemistry theory. Both of these tests were repeated with almost identical results and we do not, at this stage, have a suitable explanation for these differences other than possible differences stemming from the two agitation methods. Quirk (1950) shows that the degree of disaggregation of clays is quite dependent on the duration of agitation (shaking) used. Thorburn and Shaw (1987) describe discrepancies between measurements made using a stirring agitation (R1) and a reciprocating shaker method, suggesting that this may be the cause of our results.

Despite this discrepancy, and from a broader perspective, it is unlikely that the application of PRW, even to the mildly sodic soils in the Lockyer Valley, will cause structural degradation worse than is already reported from rainfall during summer storms (C. Henderson, Principal Horticulturist, Department of Agriculture, Fisheries and Forestry, Gatton, Australia). Table 5 shows PRW to have EC between 213 and 263 $\mu S\ cm^{-1}$ and SAR between 0.8 and 1.4. This is a slightly higher salinity than rainwater which was reported with 38 +- 23 $\mu S\ cm^{-1}$ reported in Gatton (Biggs, 2006) but very similar to surface water of the Lockyer Creek (SAR 1.45, see Table 5). It is therefore safe to assume that PRW would have a similar effect to fresh surface water on sodic soil. It is clear from our tests that there is a risk due to mechanical dispersion from rainfall or overhead irrigation with ‘fresh’ water, and it is also clear that the farmers of the Lockyer Valley manage these effects satisfactorily.

Typically, the best management of the risk of mechanical dispersion is to avoid it, by protecting the soil surface with crop residue. However, this is not always possible, especially in an intensive horticultural zone where ‘clean’ (free of crop residue) soil condition is often required for appropriate tillage, seeding and planting operations. Gypsum (calcium sulphate) is occasionally applied in the Lockyer Valley to ameliorate structure for better aeration and root growth, but the costs typically outweigh the benefits. Matthew and colleagues (1989) report some deep tillage and gypsum trials to ameliorate sodic duplex soils in the Lockyer Valley, with a view to increasing crop production, but they did not record any. Neither did they report the changes in soil structure.

In summary, there is likely to be only a small risk of soil structural degradation from the use of PRW for irrigation in the Lockyer Valley. Any such effects would be small, and not worse than those already induced by rainfall or overhead irrigation with ‘fresh’ surface water. Farmers are accustomed to managing these effects and there is no reason to believe they would not be able to manage any effects caused by the application of PRW.

5. PHYSICAL ASSESSMENT OF REGOLITH SALT AND WATER BALANCE IN THE LOCKYER VALLEY UNDER IRRIGATION

(Jenny Foley and Mark Silburn)

5.1 Introduction

Shallow alluvial aquifers underlie most of the central region in the Lockyer Valley (Figure 27), providing a valuable and heavily utilised source of water for irrigating crops. To ensure long-term security of this (periodically depleted) groundwater resource, augmentation with 18–25 GL/a of purified recycled water (PRW) is proposed. The desired outcome of this proposal is the return to sustainable groundwater use and to increase agricultural productivity.

This will affect the landscape as more water will be entering the system. Aquifers are likely to recover as PRW will partially replace groundwater use and therefore replenish aquifer levels indirectly through less pumping. Water moving through the unsaturated zone is also likely to increase as irrigation supply becomes more plentiful, increasing diffuse recharge via deep drainage and mobilising stored salts. Assessing possible salt movement into groundwater with the introduction of PRW requires an understanding of current salt stores in the regolith and of mobilisation mechanisms with a likely increase in deep drainage. In alluvial landscapes such as the Lockyer Valley, this is a time-dependent process. Diffuse recharge from deep drainage is considered a secondary recharge mechanism (to creek recharge), dependent on land-use history and the presence of residual water deficit 'buffers' in the vadose zone.

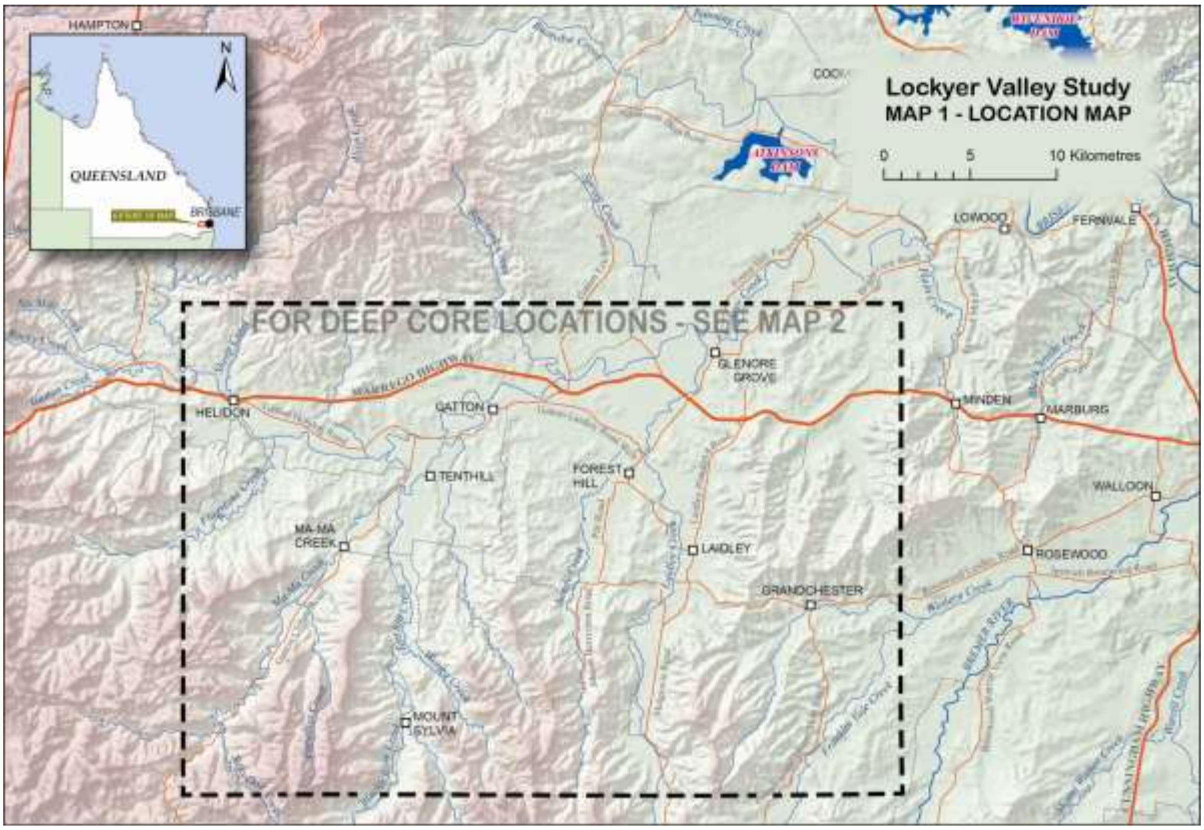


Figure 27: Location map of the Lockyer Valley.

A physical assessment was undertaken to gain a deeper understanding of the processes of water and salt accumulation and movement within the Lockyer Valley landscape. This study also looked at how climate and interim land management have affected the water (and salt) balance in the unsaturated zone over the last decade, and the future capacity to store additional salts and deep drainage (from irrigated cropping); that is, is the 'buffer' full, and if so, what quantity of deep drainage and dissolved salts are contributing to diffuse recharge to groundwater?

Three approaches were used:

1. Deep coring over time to look at longer timeframe changes in regolith salt stores and to provide information on the unsaturated zone hydrology;
2. Geophysical surveying using electrical resistivity tomography (ERT) to monitor soil water and salts within the root zone and deeper vadose zone across landscape transects and assess their spatial variability; and
3. Calculation of decadal changes to chloride levels in the vadose zone, estimation of rates of deep drainage, and quantification of the levels of salts entering receiving groundwater, using the SODICS model.

The Lockyer Valley is characterised by infrequent major flood events (20–30 year periods) caused by heavy rain within a short time. These events are a major source of creek-based groundwater recharge and landscape flow–based recharge. They are also the episodic events that cause large volumes of deep drainage, which can flush considerable quantities of salts through the unsaturated zone and into the groundwater. During the course of this study, an extreme wet season (2010–11) occurred; the impact of high levels of deep drainage on salt flushing is also reported in this chapter.

5.2 Decadal Changes in Regolith Salt Loads – Deep Coring

During 1996–98 the Queensland Department of Natural Resources and Mines (DNRM) conducted an investigation into changing groundwater quality in heavily utilised alluvial aquifers in the Lockyer Valley (Hillier and Ellis, 1996 unpublished milestone report for project QPI 30; Ellis, 1999). Five sites representative of the hydrogeology and land use in the central Lockyer were the focus of this study. A series of bores were constructed into the alluvium and bedrock layers at these sites (referred to as the Ellis monitoring bores). A deep core to ~20 m was also taken in the paddock adjacent to each site and analysed for chloride, nutrients, stable isotopes and pesticides. Locations of the Ellis monitoring bores and deep cores are given in Figure 28.

In 2010, DNRM re-cored these sites (beside the original core locations) to look at decadal changes in salt loads within the unsaturated zone (surface to groundwater) since the time of the original coring. Each core provided detailed information on the water and salts stored in the regolith (whether in equilibrium or accumulating/leaching) for the different soil types and cropping systems. At two of the Ellis coring sites a series of shallow cores were taken in 2011 along ERT transects and this provided information on the extent of spatial variability in water content, salts and texture in the surface layers.

Two additional deep cores were taken in the Sandy Creek catchment just upstream from Forest Hill; one in a paddock where groundwater had been used for many years to irrigate crops, Blenheim (B) and the other in a nearby paddock where groundwater had also been used for many years, but with a recent switch to using surface water, Blenheim (A). This area has very salty groundwater (up to EC 6000–8000 µs/cm), which has traditionally been used to irrigate salt-tolerant crops such as beetroot. In recent years (last 5–10 years) irrigators have increasingly moved away from using groundwater, and, when available, are using good quality surface water from large off-stream storage dams. At the same time that growers have been using more surface water (and less groundwater) to irrigate, groundwater in this area has shown rising trends (pre 2010–11 wet period), unlike the rest of the central Lockyer at the time (pers. comm. Ashley Bleakley, Principal Project Officer (Hydrology), Water Services South East). Just downstream of Forest Hill where the groundwater quality is very good from the Laidley Creek alluvium, but recharge is slow, salty groundwater from upstream has been moving down valley due to the changed hydraulic gradients and flow direction, carrying with it a salt plume (KBR, 2002; Ashley Bleakley, pers. comm). These soil profiles provide a useful look at the effects of long-term irrigation with poor quality groundwater on soil health, and also the resulting effects of changing to good quality surface water after many years of poor quality groundwater application (analogous to applying PRW). They also allow some estimate of the salt loads in the unsaturated zone likely to continue expressing into the groundwater in the future.

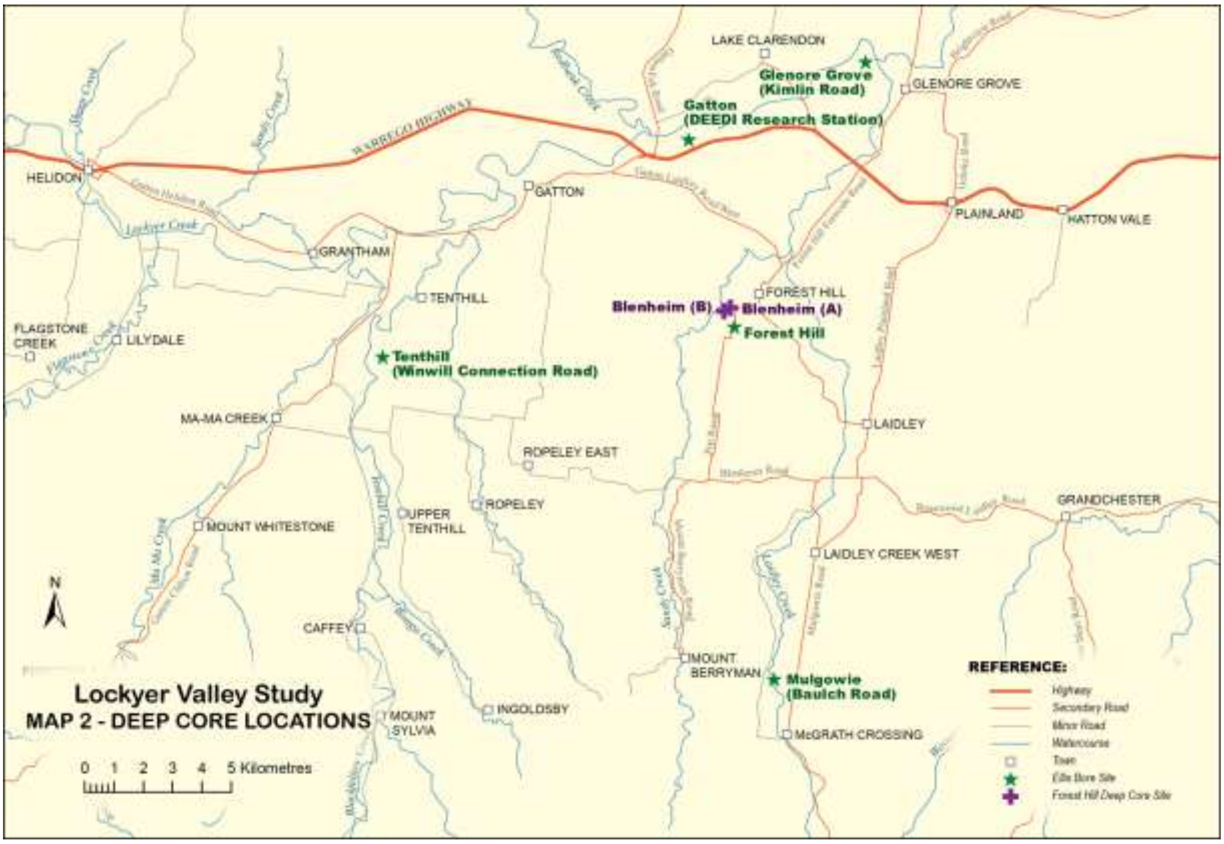


Figure 28: Deep core locations in the Lockyer Valley study area.

5.2.1 Deep Core Sampling Methods

5.2.1.1 Geoprobe Cores

Deep soil cores were taken with a direct push and percussion drill rig (Geoprobe™, Model 66DT, Kejr, Inc., Salina, KS, Geoprobe, 1998) (see Photo 1). The Geoprobe is a track-mounted machine that obtains a 38 mm soil core in 1.5 m long removable clear PVC liners (Photo 2). The rig is highly manoeuvrable; it is driven remotely into the paddock to minimise damage, and will sample all soil types from dry and wet sand to soft, sticky muds and also hard alluvial soils. Cores were taken 2–5 m from the original core location in the paddock to 15–20 m depth depending on groundwater levels. All sites were cored to at least 1 m into the alluvial groundwater, except at the Tenthill site, where groundwater levels were deeper, ~25 m.

Cores were kept cool in the field during sampling to prevent 'sweating' in the liners, and then stored in a constant temperature room (at 20 °C) until processed. During processing, the core sleeves were opened and each core described and then cut into depth increments for measurements of soil water content, a bulk density estimate at field water content, soil matric suction, particle size fractions and general chemistry (Photo 3). Detailed soil profile descriptions are given in Foley and colleagues (forthcoming).

Photo 1: **Geoprobe soil coring rig at the Tenthill site (operator Jeremy Manders).**

Photo 2: **Removable PVC liners inside coring tubes.**

Photo 3: **Describing and sampling the cores.**

5.2.1.2 Bulk Density, Water Content and Chemistry

Alternate sections of core (of similar texture) were used for either soil suction measurements and air dry chemical analysis, or for oven dry moisture content and bulk density. Core sub-samples were weighed and oven dried at 105°C for gravimetric water content and bulk density determination, or air dried at 40°C and then ground (< 2 mm) for chemical analysis. Particle sizes were determined using the Pipet method (Klute, 1986). The sand fraction is reported as the sum of all coarse and fine sand particles > 0.2 mm, and the clay fraction is all particles < 0.002 mm. Both are expressed as a fraction of 1. Samples taken for chemistry were dried and ground, then analysed for pH, EC, Cl, NO3-N (1:5 solution), cations, CEC, ESP and –15 bar moisture content. Chloride results are expressed both as $Cl_{1:5}$ mg/kg and also as Cl mg/L (per L of soil solution) using the measured soil gravimetric water contents to calculate chloride concentration in solution.

Soil bulk density was calculated from both the total core volume in each sleeve and from individual sample increments. The cutting tip diameter was used to calculate the area component of the volume calculation; however, determining the length component was more difficult. During field coring some core lengths compressed, particularly in wet clay layers, and some core lengths expanded once released from overburden pressure. The actual depth cored each time a sleeved sample was taken (usually 1–1.4 m at a time) was measured and recorded. These 'true' depths were then used to determine bulk density, using an expansion/compression ratio to cut core samples to their adjusted depth during the laboratory processing. Actual cut core lengths were also used in volume calculation.

5.2.1.3 Soil Water Suction

Soil water suction measurements were made on small core sub-samples at all sites except Forest Hill. For drier soil (> 150 kPa) a WP4-T dewpoint potentiameter (Decagon Devices, Inc., Pullman, WA) was used to measure soil suction (water potential). Measurements were taken on sub-samples (in a constant temperature room) every 4 minutes and logged through a computer interfaced logging program until values stabilised, usually within 20–30 minutes. The last 4 measures were averaged. For the wetter soil samples (< 80 kPa), mini tensiometers (UMS GmbH, München, Germany) were used to measure soil suction (Photo 4:, inset image). Core samples were wrapped and stabilised on a mini-frame. A micro-auger was used to cut a pilot hole into the centre of each core and the tensiometer tip inserted. Suction was recorded for varying durations, until readings stabilised.

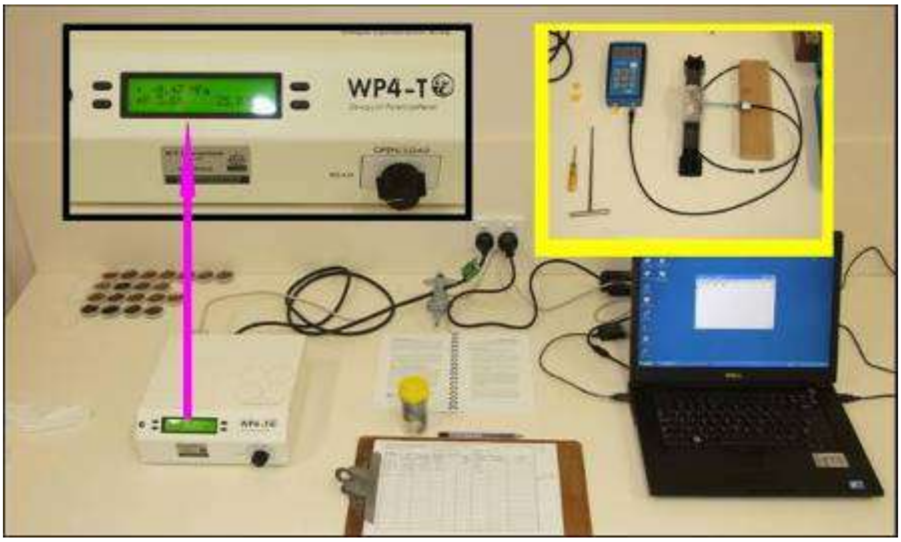


Photo 4: Measurement of soil water suction using computer interfaced WP4-T dewpoint potentiameter (main image and inset image top left) and mini tensiometers (inset image top right).

5.2.2 Deep Coring Results – Ellis Sites

5.2.2.1 Glenore Grove

The Glenore Grove site is located on alluvium alongside Lockyer Creek, upstream of the Laidley Creek and Lockyer Creek junction, on a Thornton Brown Vertosol. The Ellis bores are located ~300 m from the creek (Figure 28). Lucerne and vegetable crops are grown using good quality irrigation water of low salinity (EC 554 µs/cm, Cl 30 mg/L) from bores located beside the creek with good creek recharge.

This soil profile has a distinct sandy layer from 1–3.5 m as the texture changes from a heavy clay to a light silty clay loam. This layer is found in all the Ellis core profiles (at varying depths) and is associated with a fining-up process commonly seen in alluvial floodplains (Andrew Biggs, personal communication). These sequences reflect the last 10,000 years of climate, usually associated with high energy streams losing energy due to the climate becoming more arid. They may also be due to streams moving away in the landscape. In the Lockyer Valley, these sequences of coarser and finer sands and clay deposition are made more complex by myriad tributaries entering the major creeks and depositing layers of sands and clays in intersecting patterns. This sand layer has considerable influence on near surface layer (root zone or just below root zone) hydraulic behaviour and the resulting accumulation/distribution of water and salts in the deeper regolith.

Soil volumetric water content 'shadows' the clay fraction (Figure 29a) with the volumetric water content increasing with increasing clay content. This is seen in the other Ellis cores also.

At the time of coring, the water table was 14.6 m below ground level (b.g.l.). At this depth soil water content equals total porosity, that is, the total per cent of pores are saturated (Figure 29a). The capillary fringe (boundary condition separating the water table from the unsaturated zone) lies between 13.2 and 14.6 m where soil suction rapidly decreased from 180 kPa to zero (Figure 29c). Shallower in the profile (5 m) the soil was reasonably dry at 1140 kPa (close to defined lower limit of 1500 kPa) consistent with lucerne being grown. Unfortunately, soil suction was not measured above 5.5 m; however, the deeper curve suggests a sensible irrigation practice has been implemented by the grower, in terms of minimising deep drainage losses, with no indication of recent deep drainage accumulating below the root zone (water contents verify this).

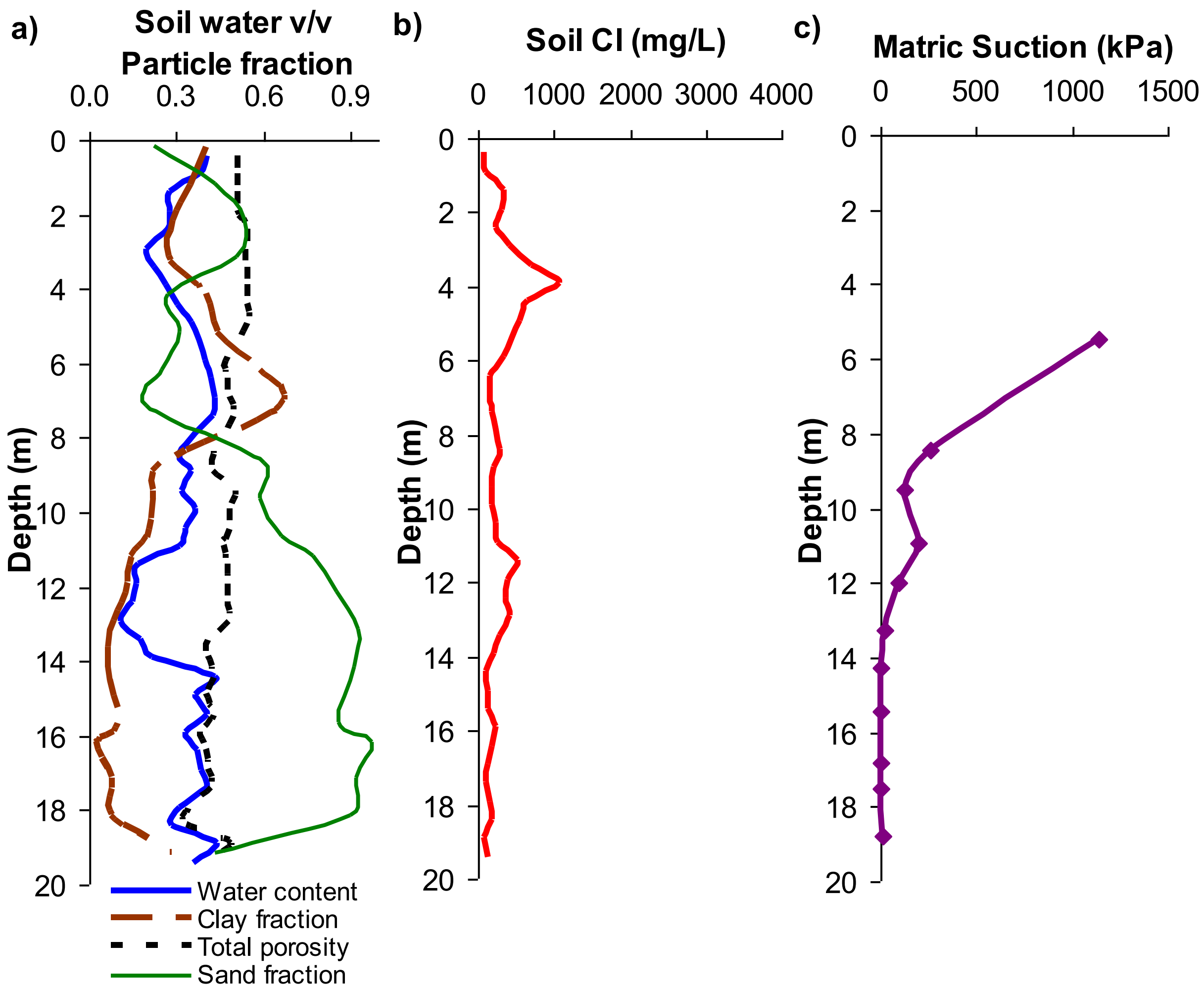


Figure 29: Glenore Grove Geoprobe core, a) soil volumetric water content and soil particle size fractions, b) soil chloride, and c) soil matric suction.

The high sand content in the profile from 9 m onwards suggest this is part of the alluvial groundwater storage zone. However, at the time of sampling it was dry as a consequence of the extended dry period, low creek levels and little creek recharge. When the alluvial Ellis bore (14320782) was first constructed in 1996, groundwater levels were at 21 m b.g.l. After the considerable rains and floods of 2010–2011, water levels rose rapidly, reaching 13.4 m b.g.l. by April 2011. Water levels have continued to rise since then, reaching 11.4 m by May 2012. The cored profile lithology suggests that the top of the conductive aquifer is at ~9 m.

Soil chemistry analysis of the cored profile indicates no salinity or sodicity problems, as would be expected given the good quality irrigation water used. Chloride levels are low; however, there is a slight accumulation at ~4 m (Figure 29b) where chloride concentration in soil solution peaks at 1060 mg/L. This accumulation occurs just below the sandy layer at the start of increasing clay content, that is, where movement of both water and salts are slowed due to a decrease in both porosity and hydraulic conductivity.

Magnesium levels are fairly high, both at this site and at several of the other sites cored (refer to Appendix B in Foley *et al.* [forthcoming] for soil chemistry). Higher magnesium in the surface can result in the soil behaving as a sodic soil (if Mg > 25% of CEC and Na < 12%), but as sodium (chloride) is very low throughout the profile, this soil can instead be considered magnesic. On the surface, this can potentially reduce infiltration. Management strategies for this magnesium-induced 'sodicity' include gypsum application, stubble retention and growing legume crops.

5.2.2.2 Tenthill

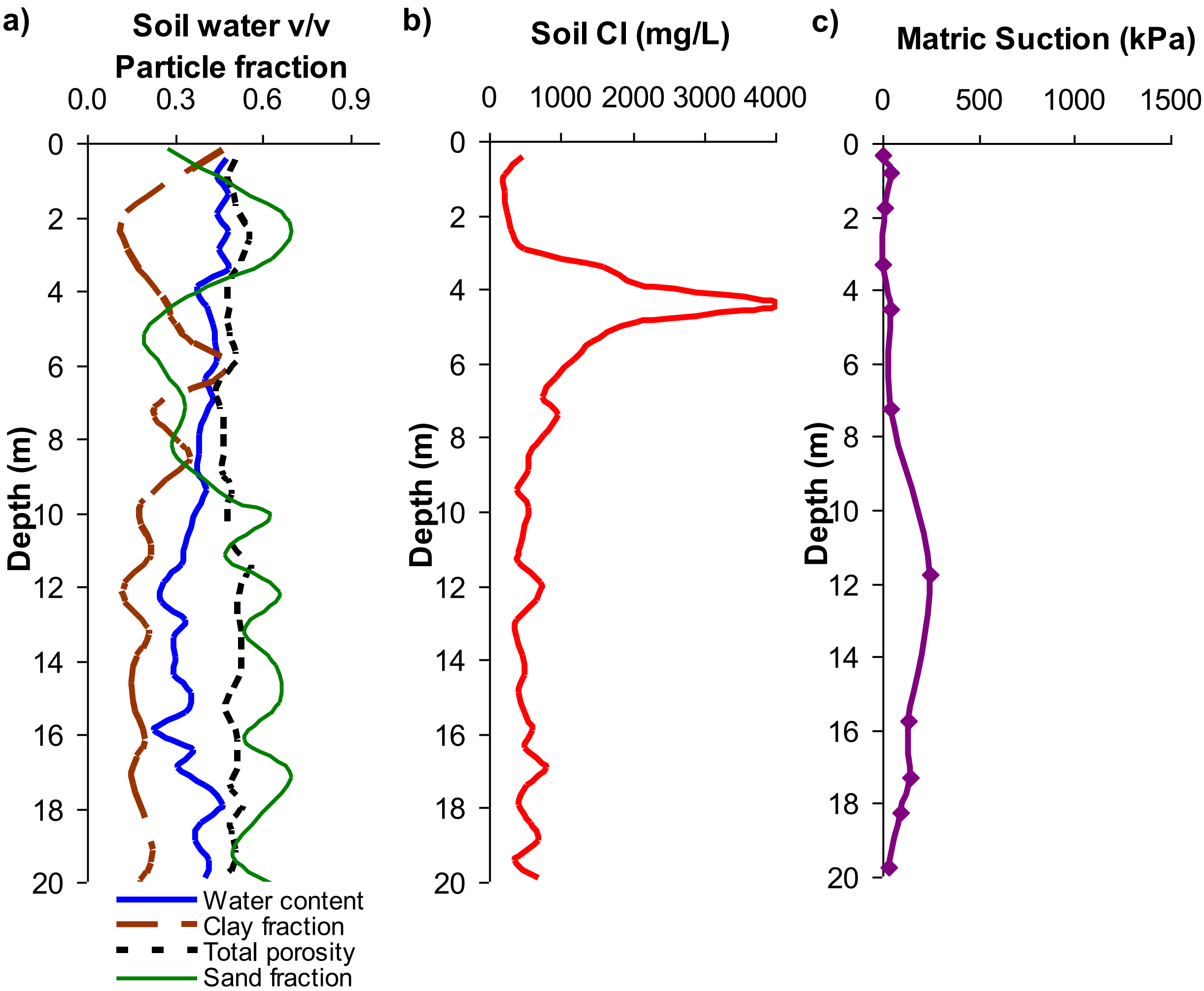


Figure 30: Tenthill Geoprobe core, a) soil volumetric water content and soil particle size fractions, b) soil chloride, and c) soil matric suction.

The Tenthill site is located on the alluvium ~250 m from Tenthill Creek on a (Tenthill) Black Vertosol (Figure 28). Historically, this site was planted to potatoes, onions and deeper-rooted crops, with irrigation used conservatively on the crops more sensitive to salt and water logging. However, for the past several years, with changing ownership, farm horticulture has shifted to leafy vegetable production requiring intense irrigation. Shallow-rooted vegetable crops (mainly lettuce and some melons) are currently grown in continuous rotation. Crops are watered every few days to keep surface soil moist and plants growing at optimal rates. Groundwater of medium salinity (EC 1600 µs/cm, Cl 260 mg/L) is used to irrigate these crops.

As with the other sites, this profile has a distinct sandy layer in the top 1.5–4 m, with the sand fraction peaking at 2–3 m depth (Figure 30a). A sharply defined clay throttle layer, restricting downwards flow of water and salts, occurs at 6 m. As a result of this restricting layer, deep drainage has accumulated above, developing into a saturated and highly saline layer containing high levels of chloride in the soil water, well above levels in irrigation (4100 mg/L peak, Figure 30b). Above 6 m the profile was very wet and at some depths fully saturated Figure 30c), indicating significant deep drainage has been occurring for some time. Hydraulic conductivity and associated gradient and porosity changes between textural layers would prevent saturation of the whole profile, with sandy layers normally draining while clay layers remained saturated. However, due to extensive deep drainage here, even deeper sand layers to 7 m were saturated. Lighter textured clays below 7 m were drier because the more slowly conducting heavy clay layer above would effectively reduce flux through all deeper layers; that is, deeper layers are 'starved' of water and flux becomes supply limited. Soil suction decreases towards zero again in the deeper, siltier layers, and by 19.5 m the soil is quite wet (30 kPa) despite the water table being 25.6 m b.g.l.

5.2.2.3 Forest Hill

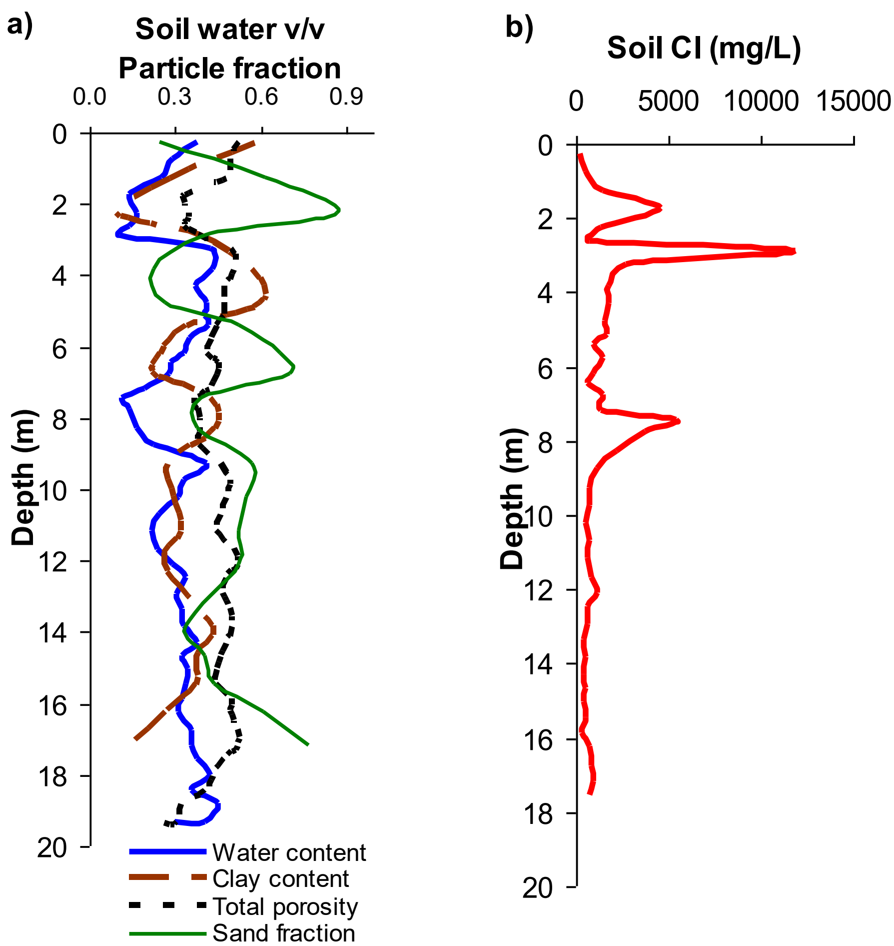


Figure 31: Forest Hill Geoprobe core, a) soil volumetric water content and soil particle size fractions, and b) soil chloride.

The Forest Hill site was originally chosen for study in 1998 (two years after the other sites) as it represented a known salty area where poor quality groundwater occurs and has been used for irrigation (Ellis, 1999). This site is located on a Lawes Black Vertosol (Figure 28) ~700 m from Sandy Creek. The grower currently plants one to two crops per year using a moderate amount of irrigation, 2.5 ML/ha per crop, and good quality surface water (EC 250 µs/cm, Cl 95 mg/L). Prior to 1996 salty groundwater had been used for irrigation (2000–8000 µs/cm). At the time of deep coring, the paddock was fallow after a corn crop. Groundwater was reached during the coring at 18.6 m depth in the profile.

The chemistry of the cored sample represents a historically salty profile that has been losing salts through leaching since the changeover to surface water 16 years ago. During the 12-year interval between the two deep coring samplings, 69 t Cl/ha was lost from the top 14 m of profile (refer to Section 5.3). This site was re-cored after the wet season of 2010–2011, and further significant chloride losses in the top 3 m had occurred (Section 5.2).

The sand layer within the shallow profile was consistent with the other core sites, with the sand fraction peaking at 2–3 m depth. Below this layer a thick clay layer was found (4–5 m depth), impeding downwards water and salt movement, as with the other sites. A high concentration of chloride (11,800 mg/L peak) was measured in this wet accumulation layer (Figure 31b). While soil suction was not measured on this profile, volumetric water content was close to the soil's total porosity, indicating the clay layer was at, or near, saturation (Figure 31a). Small chloride bulges were also measured at other depths in the profile where textural changes occurred, transitioning from either sand- or clay-dominated layers. A more detailed assessment of spatial variability of soil texture, water content and chloride across this paddock is given in Sections 5.3 and 5.4.

5.2.2.4 Mulgowie

The Mulgowie site is located in a gently sloping valley in the foothills of the upper Laidley Creek catchment. The soil is a Brown Dermosol, Lockyer soil type (Figure 28). The land has been managed as a family-owned enterprise for a number of years, using irrigation conservatively to grow lucerne on the cored paddock. Groundwater of moderately high salinity (EC 2200 µs/cm, Cl 430 mg/L) is used for irrigation.

The shallow sandy layer is sharper and slightly deeper at this site, peaking at 3.8 m with a layer of 90% sand (Figure 32a). Soil water content is lower and soil suction drier in the surface depths, suggesting efficient water use by the lucerne crop at the time of sampling (just after cutting). The lucerne crop appears to have dried out the upper profile to 5 m depth, with decreasing extraction efficiency between 3–5 m (Figure 32c). The 3–5 m layer is dry, salty and sandy, therefore less desirable for plant root growth. Despite the high levels of chloride in the irrigation water, chloride levels are low for most of the cored profile, with the exception of the salt accumulation at 2 m (2070 mg/L, Figure 32b).

The soil suction profile shows two wet layers (at drained upper limit) in the upper 8 m of soil, one at 5 m and the second between 7 m and 8 m. At 4–5 m water passes through the sandy layer back into clay. This profile clearly demonstrates the effect of the restrictive throttle; that is, the throttle layer is saturated and water is draining through it at maximum rate but surrounding layers are drier. A second saturated zone occurs at 7–8 m just above another dense clay throttle layer. Grey mottling was observed in the core sample at this depth, indicating long-term water logging.

The groundwater level at the time of coring was 14.9 m b.g.l and the capillary fringe extends upward to 13 m b.g.l. (Figure 32c).

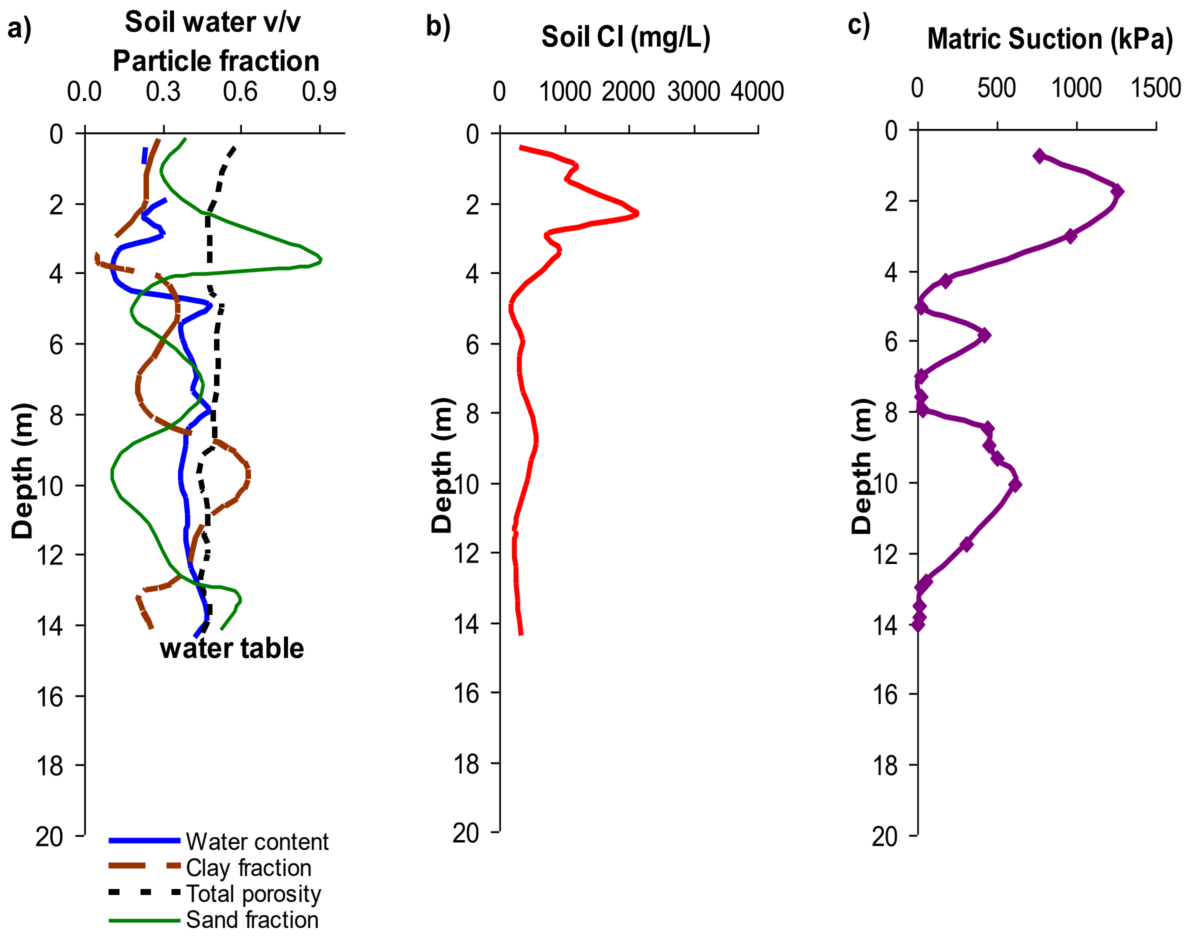


Figure 32: Mulgowie Geoprobe core, a) soil volumetric water content and soil particle size fractions, b) soil chloride, and c) soil matric suction.

5.2.2.5 Gatton

This site is located along the Lockyer Creek alluvium on a Brown Vertosol at Gatton Research Station (Figure 28). Cores were taken within an area used for small plot research trials. The plot cored was planted with irrigated grass (oats) throughout 2010. For a number of years the research station has sourced groundwater for irrigation purposes. However, prior to the dry period, and since June 2011, creek water has been used. The groundwater has an EC of 2340 µs/cm and Cl level of 410 mg/L, and the creek water has a similar EC of 1720 µs/cm and Cl of 330 mg/L. Both irrigation waters have moderately high salt levels.

This profile is hydraulically dominated by the sandy layer in the top 6 m (Figure 33a). However, this profile differs slightly in that the sand content increases rapidly from the surface downwards, with no initial uniform clay rich profile. The sand layer is also more extensive (> 3 m deep). A sharp textural boundary occurs between 5 m and 6 m when moving into very heavy clay (dark grey, massive, with slickensides and manganese present) with visible water logging above this level (seen during core processing). High levels of salts were also present in this water-logged layer (Figure 33b, Cl up to 2470 mg/L). Soil water content is strongly linked to clay content in the deeper regolith.

Soil suction was only measured from 3 m onwards (Figure 33c). The first few samples taken in the deeper rooting zone indicate a wet profile at close to field saturation (3–4.5 m), suggesting considerable deep drainage has occurred, with leaching of irrigation chloride inputs. The profile dries out somewhat below the clay throttle until the capillary fringe zone above the water table at around 20 m.

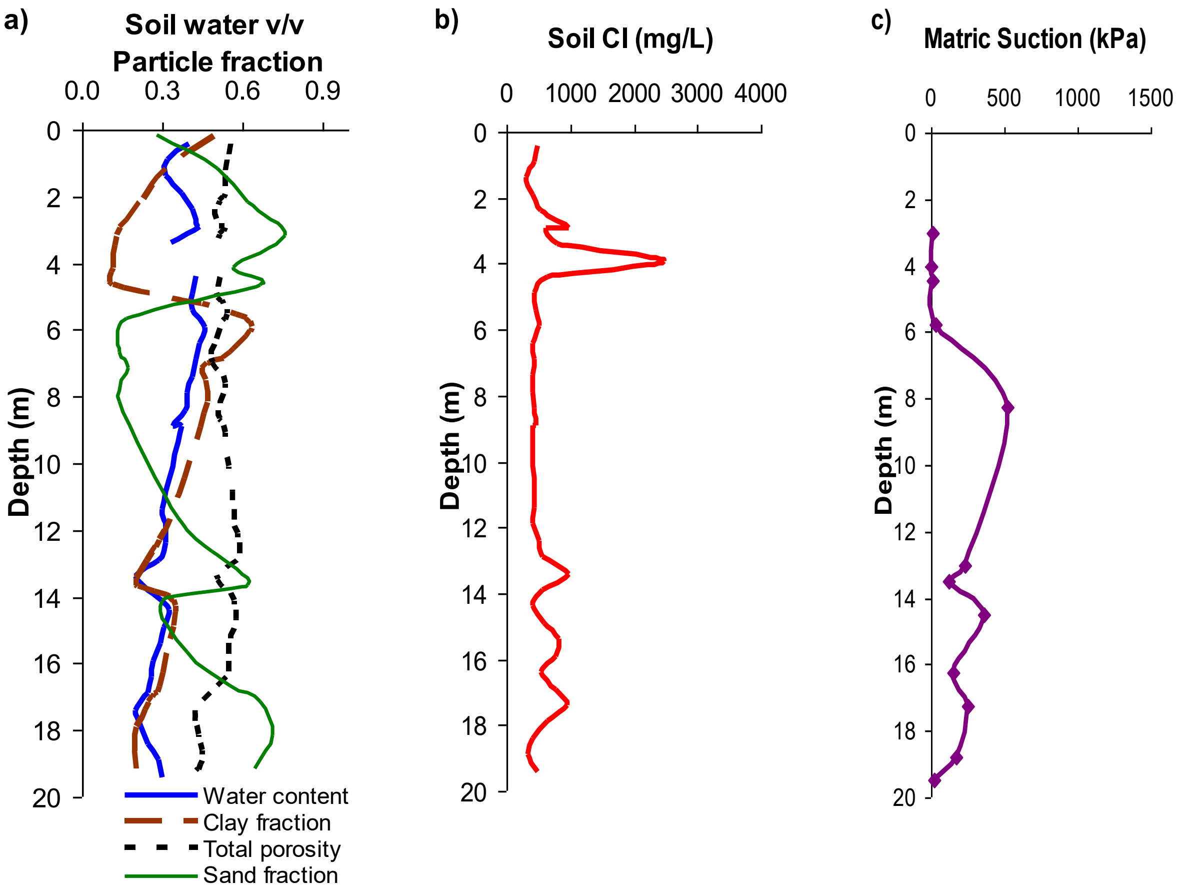


Figure 33: Gatton Geoprobe core, a) soil volumetric water content and soil particle size fractions, b) soil chloride, and c) soil matric suction.

5.2.3 Salty soil profiles in the Forest Hill area near Sandy Creek

Both deep cores were sampled in the area south of Forest Hill, on Blenheim Black Vertosols (Figure 28). One core, Blenheim (A), was taken close to Sandy Creek (~80 m from creek) and the second, Blenheim (B) was located 300 m from the first core (and further from the creek). Both sites have a history of irrigated crop production using salty groundwater. While this is still used on the Blenheim (B) paddock, good quality surface water has replaced groundwater for irrigating the Blenheim (A) paddock. Soil profile chemistry for both sites is given in Figure 34.

Photo 5: Liquefied 'super-saturated' layer of soil between 2.5–3 m in the Blenheim (B) soil profile.

5.2.3.1 Blenheim (B)

During coring of Blenheim (B), a layer of liquefied soil was found at 2.5–4 m. The coring rods passed through this mud-like slurry, with some collecting in the sleeves (Photo 5). It is possible this layer had sufficiently high sand content (Figure 34a) and degree of saturation to lose dilatancy and liquefy from the percussion action of the Geoprobe. It may also have already been in a liquefied state prior to coring. It is of significant concern that a layer this shallow is so excessively waterlogged, alkaline (Figure 34c) and structureless.

This liquefied layer overlay massive, compacted, and saturated brown clay soil. The last few metres of cored profile were grey and anoxic, a condition usually only seen in coastal muds (Andrew Biggs, personal communication). Chloride concentrations were high for most of the cored profile and pH very alkaline, reaching 10 at the surface and in the liquefied layer at 3 m (Figure 34b,c).

At depths of 8.8–10.4 m, no sample was collected in the coring sleeves, and the cause is uncertain. It is possible that more liquefaction had occurred in this deeper layer resulting in no sample being collected in the coring sleeves.

This profile represents a case where long-term irrigation with salty water and excessive deep drainage have resulted in serious soil degradation.

5.2.3.2 Blenheim (A)

This is the only deep profile cored in the Lockyer Valley that did not have the characteristic sandy layer in the upper 3 m of the profile (Figure 34a). Historically, this site was a waterlogged depression. Over the years growers have built up and levelled the natural contour of the land. This has produced an anthropogenic soil profile in the top 3–5 m, and the loss of the marked sandy layer so distinctive in the other soil profiles.

Chloride concentrations are high (similar to peak levels at Tenthill) across a wide depth range, 2–5 m, rather than accumulating in a narrow zone above a throttle (Figure 34b) as a result of uniform clay content in the upper 6 m. Some leaching of salts in the surface layers by recent application of good quality irrigation water (after historic use of saline groundwater) would be contributing to this accumulation. Lower concentrations with depth below the peak could indicate diffusion of chloride to a water table. The clay per cent increases with depth to > 80%, which may also be affecting the shape of the chloride profile.

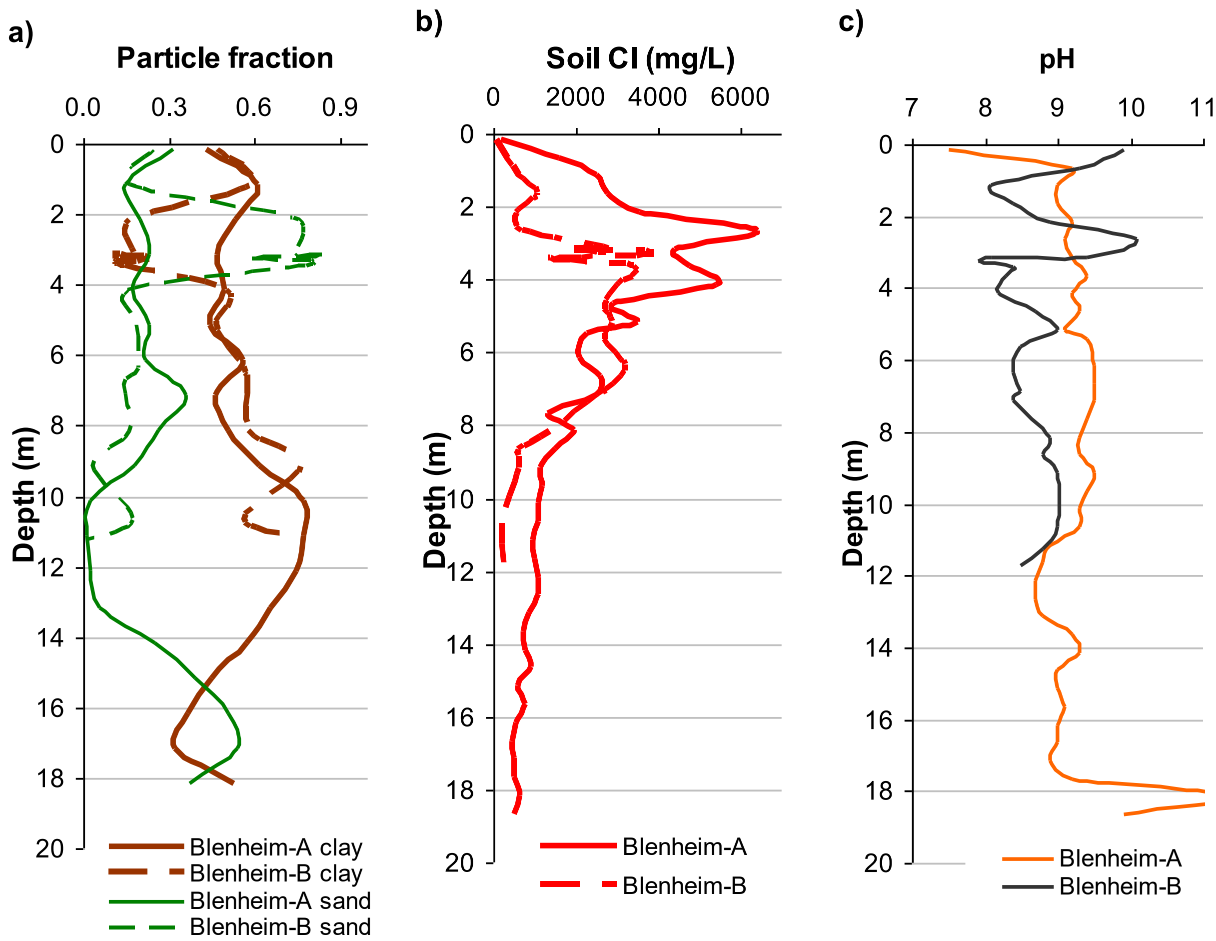


Figure 34: Additional deep cores sites Blenheim (A) and Blenheim (B), a) soil volumetric water content and soil particle size fractions, b) soil chloride, and c) soil pH.

Sodium adsorption ratios (SAR) and exchangeable sodium percentages (ESP) for the two profiles are given in Figure 35. The ESP is the proportion of sodium adsorbed onto the clay mineral surfaces as a proportion of the total cation exchange capacity (CEC) and is a measure of soil sodicity (SalCon 2011). The SAR is the relative concentration of sodium to calcium and magnesium in the soil solution or irrigation water, and is a measure of soil solution or water sodicity. In Australia, soil with an ESP greater than 6% is considered to be sodic. The Blenheim (A) profile is very strongly sodic, containing > 25% sodium relative to the other exchangeable cations (i.e. calcium, magnesium and potassium). The Blenheim (B) profile is also weakly sodic, increasing to strongly sodic with depth.

The presence of high levels of exchangeable sodium causes soil aggregates to disperse (deflocculate). This results in poor infiltration and drainage, leading to reduced leaching and salt accumulation over time. Sodic soils become dense and cloddy, as can be seen on the surface of the Blenheim (A) soil (Photo 6). On drying, these surfaces then become hard-set and crusted leading to reduced seedling emergence, plant growth and yield. There is also an increased susceptibility to soil erosion. At the Blenheim (A) site, surface dispersion has been exacerbated by the use of low-salt irrigation water in recent years, which acts in a similar way to rainfall by washing out salt in the surface layers, leading to dispersion and crusting. The surface application of PRW, without sufficient amendments, in areas currently using groundwater with moderate salt levels (most of the Lockyer Valley) could similarly affect surface stability.

The highly alkaline soil profiles for both sites (Figure 34c) are typical of soils with high levels of exchangeable sodium. Gypsum (calcium sulphate) can be applied as an ameliorant for dispersive soils. Gypsum initially provides an electrolyte effect to the soil, increasing its salt concentration. This prevents clay particles from swelling and therefore reduces dispersion. A longer-term effect of gypsum is to replace exchangeable sodium attached to clay particles with exchangeable calcium, which makes the soil less sodic and reduces dispersion.

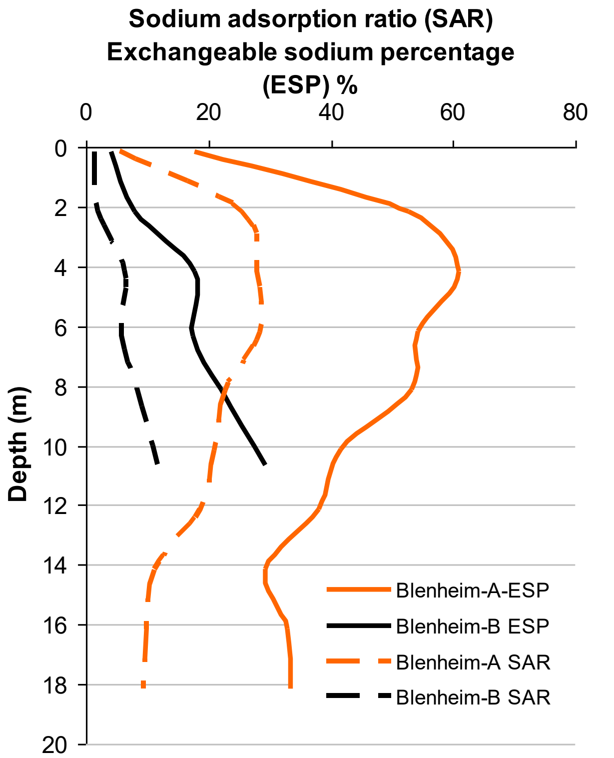


Figure 35: SAR and ESP profiles for Blenheim (A) and Blenheim (B) soil cores.

Photo 6: Clay dispersion after irrigating with good quality surface water – colour image taken at Blenheim (A) site next to core.

5.3 Decadal Changes in Chloride Profiles and Estimation of Deep Drainage using SODICS

Soil chemistry from 1996–98 (Ellis, 1999) and 2010 deep coring at the Ellis sites were used to assess decadal changes to chloride levels in the vadose zone and to estimate deep drainage, using the SODICS model.

Two transient approaches can be used to estimate deep drainage from chloride profile changes: 1) the transient solute mass balance model (SODICS software, Rose *et al.*, 1979; Thorburn *et al.*, 1990) which is based on the mass balance of chloride in the root zone, and 2) the chloride front displacement method (Allison and Hughes, 1983), which assumes piston flow and involves the observation of movement of a particular chloride pattern and pressure front with depth. The second method is suited to situations where deep drainage has increased in a profile that was historically drier – for example, after clearing native vegetation – and was therefore not used at these sites, given their long history of irrigated cropping.

Care must be taken in interpreting the highly heterogeneous alluvial profiles of the Lockyer Valley, as i) texture boundaries (diffuse or sharp) can result in throttles that can greatly alter suction without the direction of flow (gradient) necessarily altering, and ii) many of the observed chloride bulges in these profiles are simply due to chloride accumulations in sand layers above dense impeding clay layers and not due to a time-dependent salt pulse moving through the vadose zone.

The transient solute mass balance model, SODICS, uses the changes in mass balance of chloride in the root zone to estimate drainage fluxes of water below the root zone. It makes the assumption that the chloride concentration of the drainage water is proportional to the mean concentration in the root zone (Walker *et al.*, 1991), although this assumption is difficult to test because of the amount of spatial and temporal data required (Thorburn *et al.*, 1990). It also assumes complete mixing of the chloride in the soil, no anion exclusion and that bypass flow is not occurring.

A solution to the transient solute mass balance model proposed by Rose and colleagues (1979) is given by Thorburn and colleagues (1990), for two consecutive soil chloride profile samplings (at times t1 and t2) using $P = C_Z / C_{BAR}$, which is assumed to be constant over time, giving:

$$\overline{C}_{t2} = \overline{C}_{t1} + \left[\left(\frac{I\,C_{IW}}{LP}\right) - \overline{C}_{t1}\right]\left[1 - \exp\left(\frac{-t\,P\,L}{\overline{\theta}_S\,z}\right)\right] \quad (4)$$

Where:

Z is soil depth

θS_{bar} is depth-weighted mean water content at field saturation (DUL v/v) above Z

C_{bar} is depth-weighted mean soil water chloride concentration at DUL above Z

t is time

I is the average infiltration rate of water (rainfall + irrigation – runoff) (e.g. mm/yr)

C_{IW} is chloride concentration of infiltrating water

L is the average leaching or drainage rate (e.g. mm/yr)

C_Z is chloride concentration of soil water at depth Z, at field DUL water content.

This equation can be solved for drainage rate (L) using numerical iteration (here SOLVER in Microsoft Excel). It can also be used to calculate the time course of soil solute concentration (C_{BAR}) and therefore the leachate concentration $C_Z = P.C_{BAR}$, provided I, CI_W, L and P are known. The final steady-state mean soil concentration is $C_{BARf} = (I\ C_{IW})/(LP)$.

This modelling approach was used to estimate total salt and chloride loads in the vadose zone and to estimate deep drainage and solute movement to groundwater over the 12–14 years between coring. Deep drainage was also estimated between 2010 and 2011 at the Mulgowie and Forest Hill sites from the shallow (3–6 m) cores taken along electrical resistivity tomography (ERT) transects.

5.3.1 SODICS Parameterisation

Measured volumetric soil water, bulk density, air dry water content and soil chloride for discrete sample depths (2010 cores) were used in SODICS. Only chloride data was available for the 1996–98 cores, so other soil parameters were approximated from the 2010 data. Coring dates were used to derive the time interval modelled. Rainfall inputs were obtained from meteorological station data at Forest Hill and Gatton (1998–2010) and an annual average rainfall for this period of 747 mm was used in the model. For the 2010–2011 intense wet period, the actual rain that fell between coring dates was used, 1760 mm at Forest Hill and 1070 mm at Mulgowie. A value of 5 ppm was used for chloride concentration in rain. SODICS was run without an irrigation input for the 2010–2011 period modelled, given the intense rain and flooding that occurred during this time.

Irrigation history and temporal fluctuations in irrigation water quality were not accounted for as the volume of irrigation water applied was unknown and only recently (2011) measured chloride levels in irrigation waters were used in the model (Table 8). Either 200 mm/yr or 400 mm/yr of irrigation was assumed for SODICS to provide some assessment of the sensitivity of each site to irrigation application rate. At selected sites a wider range of scenarios, 100–600 mm, were modelled. Current cropping practice and some cropping/site history for each site are given in Section 5.2.2.

5.3.2 Depths Modelled

SODICS was built for root zone applications. Here the model was used to estimate deep drainage and potential recharge rates (to groundwater) within the lower regolith for depths starting *below* the root zone. While this is outside the range of normal application, the basic calculations for estimating chloride concentration in leachate and deep drainage hold for any depth. The model assumptions for use (root zone applications) are likely to be associated with the original requirements at the time of model development. During that era of research, soil sampling was predominantly carried out using a soil coring rig to sample the top 1–1.5 m of soil. Many rigs were not designed to take deeper cores. Hence, the SODICS model and the assumptions for use have been designed for data available at the time (shallow profile datasets). Some assessment is given here regarding the model's suitability for this (depth) extended application.

The first depth modelled was from below the root zone to ~6 m deep (except Gatton, which was only modelled to 4.7 m as the next 1996 depth with chemistry was 7.4 m, with no intermediate depth). The depth of the root zone at each site was taken as the first chloride value in the 1996–98 datasets (between 1.3–1.9 m) rather than on the actual rooting depth of crops grown at the sites. More detailed surface data from the 1996–98 coring was not available. This first set of modelled profiles, spanning the top 6 m, assesses more recent salt accumulation/leaching in the profile. This depth also includes the hydraulically dominant sandy layer at each site most strongly affected by salt accumulation. The second depth modelled was from below the root zone to 13–15.5 m (variations in depths modelled are due to limited 1996–98 depths to match with the detailed 2010 data). This deeper whole-of-regolith modelling looks at chloride changes over a longer timeframe and provides some interesting insight into the model's capability.

5.3.3 Root Zone Salt Profiles

The shape of the salt (EC) profiles in the root zone reflects not only the soil's hydraulic properties but also the cropping, irrigation and leaching (deep drainage) interaction (Shaw and Dowling, 1985). Curve 1 (Glenore Grove) in Figure 36 shows a permeable and well-leached profile, naturally a little higher in salts being a Vertosol clay soil, and a small degree of salt accumulation from irrigation with depth. Curves 2 (cluster of solid lines Mulgowie-red, Blenheim (B)-black and Forest Hill-blue) are typical of profiles with a moderate amount of salt accumulation with peaks well within the root zone, and variations related to texture changes. Curve 3 (Tenthill) is typical of a highly leached upper profile with significant accumulation just below the root zone. Curve 4 (Blenheim (A)) indicates significant contribution of salt from irrigation water accumulating at a shallow depth of < 1 m and remaining high. Crop productivity would be strongly impacted by salt concentrations in the root zone of this profile.

Although these are classic interpretations, the shape of these EC profiles indicates a history of higher drainage rates, often also with large salt addition, as they have not developed a 'natural' shaped profile. They appear to have been continually flushed with some accumulation zones in the upper part of the dense clay layers. The low salt concentrations are indicative of high rates of drainage and/or irrigation with low salinity water. Low salt concentrations in the lower parts of the profile indicate high rates of drainage some decades ago. Deep drainage is often episodic with salts accumulating in dry years and significant flushing occurring in wet/flood years.

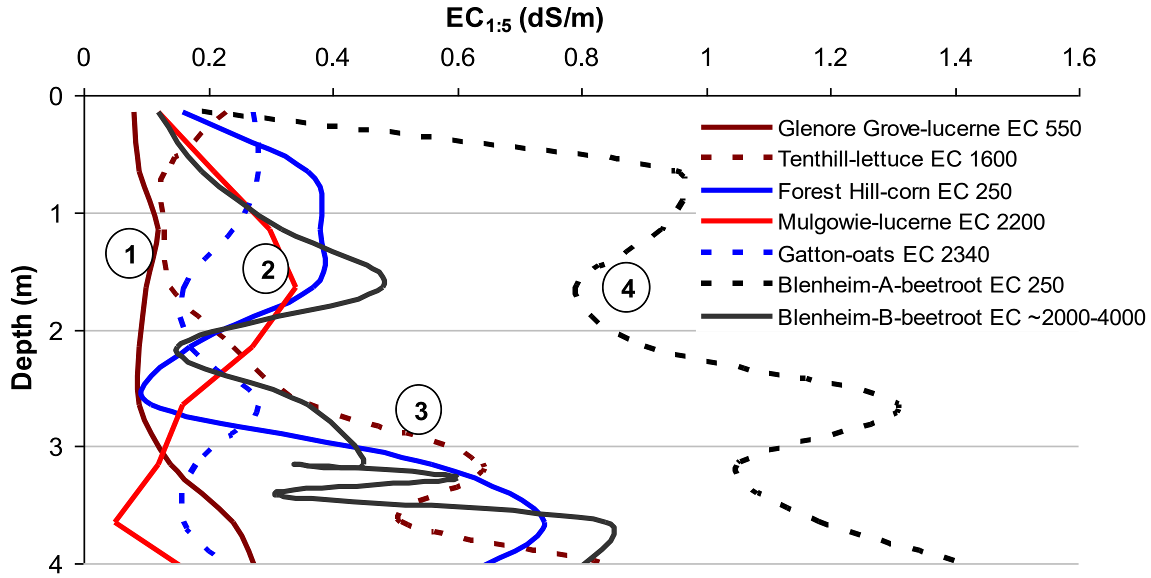


Figure 36: EC profiles within the root zone of deep coring sites in the Lockyer Valley, with legend showing crop at time of coring and EC (dS/cm) of applied irrigation water. Labelled numbers 1–4 are used as a reference in the text.

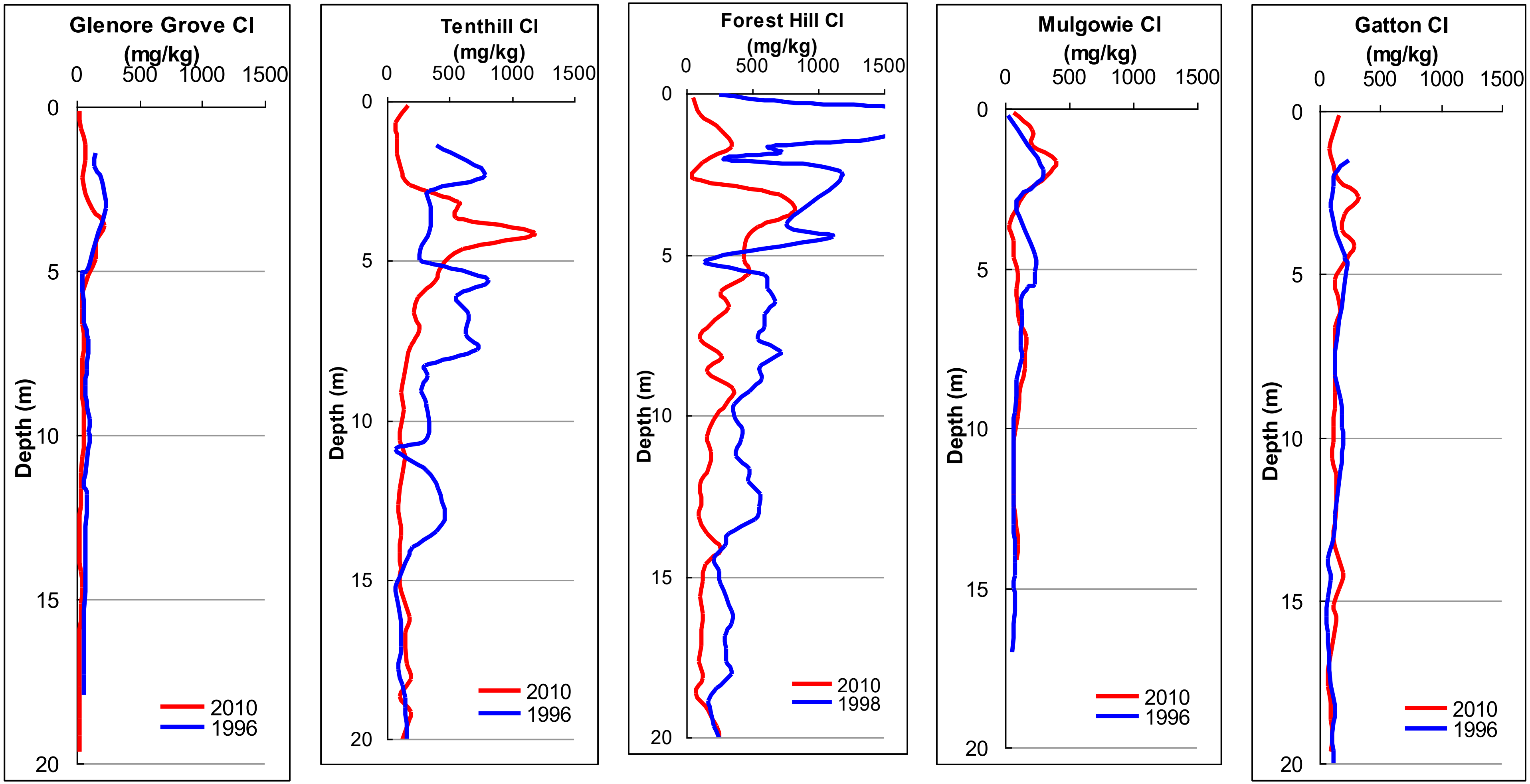


Figure 37: Decadal changes in chloride profiles for the five Lockyer Ellis sites.

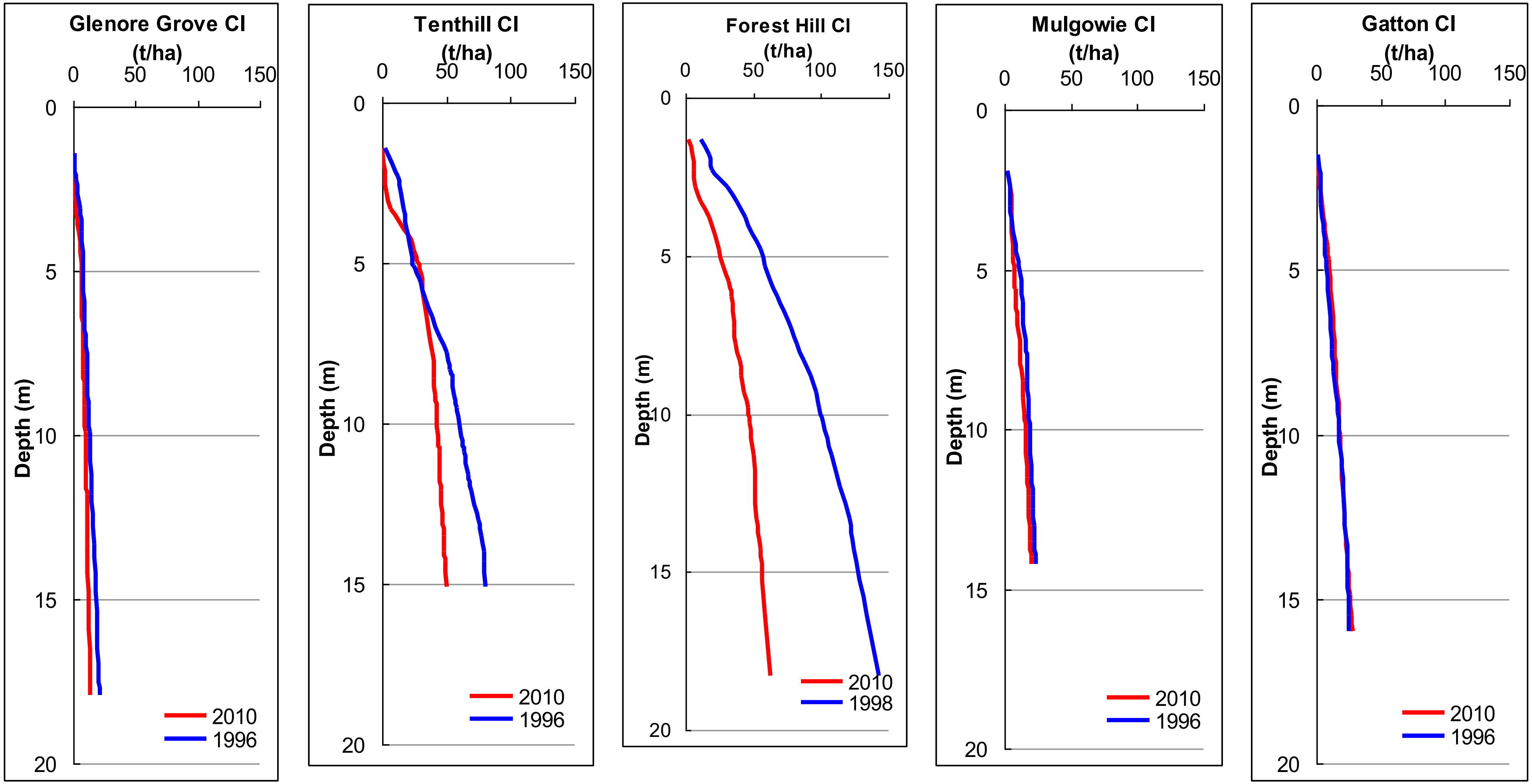


Figure 38: Cumulative decadal changes in chloride profiles for the five Lockyer Ellis sites.

5.3.4 Decadal Salt Accumulation/Leaching under a Range of Irrigation Scenarios

Of the five sites modelled, a clear grouping can be made between those sites where a significant change has occurred in total chloride mass in the unsaturated zone (Forest Hill and Tenthill) and those sites where little change has occurred as they are in (approximate) steady state (Glenore Grove, Mulgowie and Gatton) (Figure 37, Figure 38). The sites with significant change have lost between 30–70 t/ha of chloride from the profile over the 12–14 years modelled. The steady-state sites have lower amounts of stored chloride (11–26 t/ha to 13–15.5 m of modelled depth) and decadal change in storage is small (< 5 t/ha) (Table 7). Note that the lack of change in chloride at the steady-state sites does not mean that drainage has not occurred, as any chloride added in rainfall or irrigation has had to be removed via drainage to maintain steady state.

Table 7: Total chloride losses or gains and deep drainage (leachate) estimates below the root zone, for Ellis deep core sites during 1996–98 to 2010 (simulated irrigation application 200 or 400 mm/yr).

Site and Current Crop	Modelled Depth	Total Cl 1996–98	Total Cl 2010	Total Cl Lost	Irrigation	Deep Drainage	Av Cl conc. in Leachate**	Estimated av. EC in Leachate**
	(m)	(t/ha)	(t/ha)	(t/ha)	(mm/yr)	(mm/yr)	(mg/L)	(µs/cm)
Glenore Grove	1.4–6.2	9.0	7.1	1.9 (21%)	200	120	192	452
(lucerne)	1.4–13.3	16.8	11.5	5.2 (31%)	200	294	156	367
	1.4–6.2				400	152	192	451
	1.4–13.3				400	333	159	143
Tenthill	1.4–6.1	33.5	33.0	0.5 (1%)	200	79	750	1766
(lettuce)	1.4–14.3	79.7	48.9	30.7 (38%)	200	695	385	907
	1.4–6.1				400	148	750	1765
	1.4–14.3				400	834	383	902
Forest Hill	1.3–5.3	58.4	27.7	30.7 (53%)	200	110	2307	5432
(fallow/corn)	1.3–14.3	124.5	55.1	69.3 (56%)	200	538	1013	2385
	1.3–5.3				400	117	2183	5141
	1.3–14.3				400	552	987	2324
Mulgowie	1.9–6.5	14.4	9.4	5.0 (35%)	200	282	442	1040
(lucerne)	1.9–13.0	22.2	18.9	3.3 (15%)	200	275	408	960
	1.9–6.5				400	439	480	1129
	1.9–13.0				400	439	452	1065
Gatton	1.5–4.7	7.7	9.6	-1.9* (-25%)	200	120	587	1381
(oats trial)	1.5–14.4	24.4	25.7	-1.3* (-5%)	200	190	398	937
	1.5–4.7				400	255	597	1406
	1.5–14.4				400	395	399	939

* Profiles indicating decadal salt accumulation – Gatton
** Average concentration in leachate is the profile average (across the depth range) calculated from the volume of water draining (DD) and the total change in Cl or EC (plus inputs in irrigation and rain) in the profile over the 12–14 year time span

To keep chloride constant, SODICS estimates sufficiently high deep drainage rates to remove chloride to balance the chloride addition in rain and irrigation water. Estimated rates of deep drainage for the deeper profiles were 80–830 mm/yr, depending on assumptions made about irrigation water use (and chloride levels in irrigation waters) and the depth modelled. These are consistent with previous estimates in the Lockyer Valley (Thorburn *et al.*, 1990) where the range of minimum to average rates was consistent with the shallow depths modelled by Thorburn and colleagues (however, the deeper depths modelled estimated greater deep drainage fluxes at Forest Hill and Tenthill, and this is discussed in more detail in the following sections). These deep drainage fluxes are also consistent for inland furrow-irrigated Vertosols and Sodosols (Silburn and Montgomery, 2004) and beneath irrigated agriculture in the Amargosa Desert using chloride mass balance and chloride front displacement methods on deep soil profiles (Scanlon *et al*., 2009). Heiner and colleagues (1999) used the SaLF model of Shaw and Thorburn (1985) and predicted deep drainage losses through Lockyer Valley soils between 85–445 mm/yr under 300 mm of irrigation. Stable isotope studies and tritium injection methods (Dharmasiri, 1997; Dharmasiri *et al*., 1997) conducted at the Ellis sites during original coring in 1996–98 calculated a slightly lower average vertical infiltration rate for some of the soils of about 200 mm/yr.

5.3.5 Significant Change Sites: Forest Hill and Tenthill

Initial 1996–98 chloride mass was high in these profiles: 80 t/ha at Tenthill and 125 t/ha at Forest Hill (Figure 38, Table 7) with subsequent decadal losses of 31 t/ha and 69 t/ha, respectively. These losses were triggered by either a change in cropping and associated irrigations (amount of water applied) or a change in irrigation water quality (amount of chloride applied).

At Forest Hill, half the historic salt accumulated in the regolith was removed in leachate in just 12 years, after switching to low salinity surface water in 1996, following years of using salty groundwater for irrigation. SODICS estimated an average rate of 110–550 mm/yr of deep drainage, depending on the depth modelled. The 2010 chloride levels in the leachate are half what they were in 1998 (from 1026 to 654 mg/L). Leachate recharging the underlying aquifer has a *lower chloride* concentration than the receiving groundwater (Table 8).

At Tenthill, deep drainage increased with increasing irrigation frequency, as a consequence of the shift to more intensive shallow-rooted vegetable cropping several years ago. Estimated deep drainage ranged from 80–830 mm/yr depending on volume of irrigation and depth modelled (most sensitive to depth). At this site, chloride mass in the upper profile remained unchanged while considerable flushing occurred at greater depth. Over time, this would increase deep drainage through the deeper regolith, flushing old stored salts. SODICS modelling of only the deeper profile (6–14 m, with no irrigation input) still estimated that 400 mm/yr of deep drainage was required to flush 30 t/ha of chloride through the deeper regolith. Average chloride concentration in the leachate (300 mg/L) entering the underlying aquifer is *the same* as the concentration in the receiving groundwater (295 mg/L, Table 8) and only slightly higher than the levels of chloride in the irrigation water (260 mg/L, Table 8). As the chloride concentrations in the various waters/leachate are similar (indicative of steady state) it is probable that most of the decadal change in the deeper regolith occurred via an earlier flushing initiated by the land-use change several years ago, and the system is now in a new equilibrium. However, the interval between coring is too coarse to show this.

5.3.6 Steady-State Sites: Gatton, Mulgowie and Glenore Grove

At Gatton there was a small accumulation of chloride in the upper profile (1.9 t/ha) but no change in the deeper profile during the 14 years between sampling. When modelled, the SODICS equation collapsed back to the steady-state mass balance equation (USSL, 1954; Walker, 1998) to estimate deep drainage (indicative of steady state). Across a range of irrigation scenarios, 100–600 mm/yr, deep drainage through the deeper profile needed to be approximately equivalent to the irrigation volume to prevent chloride accumulation. Both measured and modelled chloride (and EC) levels in leachate are the same as the irrigation water, being *slightly higher* than the receiving groundwater (Table 8).

Irrigation water is not sourced from the underlying aquifer. It is sourced from a distant bore with higher salinity (Cl 410 mg/L, EC 2340 µs/cm) and for some years, due to poor creek flows, this has been used rather than creek water. The groundwater in the Ellis alluvial bore has risen ~6 m since soil coring, and continues to rise. Recent water quality analysis indicates salinity has also increased, from Cl 150 mg/L, EC 1280 µs/cm in 2011, to Cl 270 mg/L, EC 1630 µs/cm in 2012. This is now closer to the leachate concentration in the deeper soil profile recharging the aquifer (Cl 408 mg/L). While some of this increase may be due to the effects of diffuse recharge, it is most likely due to the large-scale lateral movement and mixing of groundwater after the 2010–11 wet season.

For the Mulgowie site there was a small amount of leaching from the upper to the lower profile; however, overall this site is close to steady state; that is, deep drainage is sufficient to remove chloride from irrigation and rain. Across 100–600 mm of simulated irrigation input, deep drainage through the deeper profile needed to be approximately equivalent to the applied irrigation to prevent chloride accumulation. Leachate had slightly *lower chloride and EC concentrations* than the applied irrigation water (Cl 330 µs/cm in leachate and Cl 430 µs/cm in irrigation, Table 8) and significantly lower levels than the receiving groundwater (Cl 1650 µs/cm, Table 8). Water used for irrigation is sourced from better quality groundwater at a distance from the cored paddock.

The Glenore Grove site had only small chloride losses over the 14 years between sampling. Long-term good quality irrigation water has been used (Cl 30 mg/L and EC 550 µs/cm) sourced at a distance from the cored paddock. Lower deep drainage rates are estimated with SODICS, 120–150 mm, (Table 7) as less chloride needs to be leached to maintain steady state. The leachate had *higher chloride and EC* concentrations than the receiving groundwater (Table 8).

Table 8: Chloride and EC of soil, irrigation water, leachate and receiving groundwater at the Ellis sites.

Site	Depth Modelled	Average Soil Cl in Lowest Depth 1996–98	Average Soil Cl in Lowest Depth 2010	Irrigation Cl	Irrigation EC	Leachate Cl conc. to Groundwater 1996–98*	Leachate Cl conc. to Groundwater 2010*	Alluvial Bore Cl 2011**	Alluvial Bore EC 2011**
	(m)	(mg/L)	(mg/L)	(mg/L)	(µs/cm)	(mg/L)	(mg/L)	(mg/L)	(µs/cm)
Glenore Grove	1.4–13.3	457	314	30	550	311	129	39	735
Tenthill	1.4–14.3	1243	764	260	1600	480	300	295	1945
Forest Hill	1.3–14.3	2475	1096	97	250	1026	654	1950	6330
Mulgowie	1.9–13.0	539	460	430	2190	278	329	1650	6335
Gatton	1.5–14.4	328	345	410	2340	210	408	150	1280

*concentration in leachate (soil water solution) at greatest depth modelled – at least 3 m above water table
** average of two pumped water quality samplings taken in alluvial bores during February and August 2011

5.3.7 Effects of Irrigation Volume and Depth Modelled on Estimated Deep Drainage

At sites where chloride levels in irrigation water are small (Forest Hill and Glenore Grove) deep drainage estimates are not affected by the irrigation amount input to the model; that is, similar deep drainage is estimated for a range of irrigation scenarios, because irrigation adds little to the overall chloride balance. However, as the water becomes more saline, the amount of irrigation increasingly affects the estimates of deep drainage, with Gatton most affected, then Mulgowie, then Tenthill.

At all sites except Mulgowie, the depth modelled strongly affected deep drainage estimates, with 2–8 times as much deep drainage needed to leach chloride (difference between 1996–98 and 2010 amounts stored in the profile plus annual inputs) below the modelled profile depths. To verify this depth sensitivity, a number of depths were modelled, including small increments throughout the profiles; for example, if 1 t/ha of chloride was lost between 2 m and 4 m over 12 years, the model would estimate a small deep drainage rate. However, if when examining the whole profile and the fate of this 1 t/ha of chloride, it was calculated that 15 t/ha over 12 m of soil depth was actually lost, not just 1 t/ha, then the deep drainage rate required to leach this larger mass of chloride would be significantly greater. The 1 t/ha initially modelled is likely to have leached below 12 m (not just below the 4 m initially modelled) along with the other 14 t/ha. If chloride change is modelled at too shallow a depth, it will not reflect the true mass, depth and travel time taken by the leaching salts and will give the most conservative estimate of deep drainage.

Forest Hill is a good example of the effect of depth modelled on deep drainage estimates, as the amount of irrigation water applied does not affect estimates, and a significant amount of chloride, 69 t/ha, was lost from 14 m of the regolith over 12 years. To leach the change in chloride mass from the shallower part of the profile only requires ~110 mm/yr of deep drainage. However, to move the full 69 t/ha though the whole profile (below 14 m) and not just beyond 5.3 m requires ~540 mm/yr. If we had only modelled the shallower depth we would have erroneously estimated a deep drainage rate of 110 mm/yr for this site. Ellis and Dharmasiri (1998) estimated an intermediary rate of deep drainage, 300 mm/yr, at the Forest Hill site from isotope and rainfall pattern interpretation.

Thorburn and colleagues (1990) and Thorburn and Rose (1990) also found that in three-quarters of the Lockyer Valley sites modelled using SODICS, rather than the expected behaviour of leaching flux (mm/yr) decreasing to a minimum with depth, they increased by up to three times the minimum with increasing depth (and the depths they modelled were still only root zone depths). This unanticipated increase was associated with a decrease in soil salinity and was largest for sites on the Lawes and Tenthill soils of their study; for example, they modelled a leaching flux > 700 mm/yr by 1.5 m depth on a Tenthill soil, where the minimum value is given as 219 mm/yr for shallower depths modelled. They attributed these large differences with increasing depth modelled to the effects of bypass flow (Thorburn and Rose, 1990).

5.3.8 Implications for Groundwater Recharge via Deep Drainage

In 1999, Ellis found from two numerical models used on the 1996–98 data (Ellis, 1999; Ellis and Bajracharya, 1999) that deep drainage from irrigation would not have long-term adverse effects on the water quality of the aquifers below the Ellis sites. Here we look at how this assessment has held up over the subsequent decade by comparing the average water and salt fluxes to groundwater (Table 8). In 2010 receiving groundwaters were more saline (greater chloride concentrations) than the recharging deep drainage, except for small increases at Glenore Grove and Gatton. Recent groundwater chloride levels at Gatton are closer to the leachate concentrations. At both Gatton and Glenore Grove very good quality groundwater underlies the coring sites, and irrigation waters are sourced from nearby bores with slightly poorer quality water.

Tenthill is the only site where the irrigation water is sourced from the aquifer underlying the core location, and chloride levels in the leachate and receiving groundwater are the same. While there has been extensive leaching of salts at this site and at Forest Hill, the leachate is still equal to, or more dilute than, the receiving groundwater (Table 8).

Groundwater quality in the Lockyer Valley at many places is the result of mixing between deep drainage waters and more saline water entering from the underlying sandstone. Diffuse deep drainage is actually a diluting agent. Thus, raising the concentration of chloride in the deep drainage can still worsen groundwater quality even if it is lower than the receiving water. Of the five sites modelled, chloride in leachate entering the groundwater is lower in 2010 than in 1996–98, with the exception of Mulgowie and Gatton (columns 7 and 8 in Table 8). These two sites had the highest chloride in irrigation water. T The irrigation water was sourced from a saltier part of the aquifer nearby and hence chloride levels in leachate and receiving waters cannot reach equilibrium.

The introduction of PRW is likely to continue to flush existing salts from the regolith, but given the very low chloride levels in the proposed PRW, there would be no accumulation of salts in the regolith and no recharge of salts to groundwater beyond current levels (instead they will decrease). For a more detailed exploration of this topic please refer to Chapter 10.

5.3.9 Mulgowie and Forest Hill 2010–2011: Changes over 12 months due to the Intense Wet Period

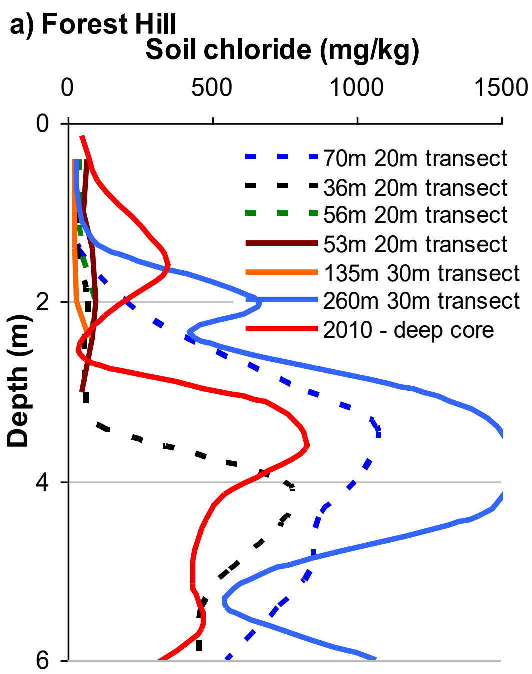
During extreme climatic events such as the 2010–11 wet season/flooding in the Lockyer Valley, considerable drainage fluxes can occur. SODICS was used to estimate deep drainage in the 11 months following the event using chloride measured on shallow cores taken along ERT transects (Section 5.4) at Forest Hill and Mulgowie.

Significant displacement of chloride occurred at both sites (Table 9, Figure 39) with estimates of drainage fluxes from 775 mm/yr to 870 mm/yr. Almost all chloride was fully leached from the upper 1.5 m of the profiles. At Forest Hill, what appears to be displacement of a chloride bulge from the 2010 position to the 2011 position (seen in 6 m core samples) is actually only spatial variability in clay content, volumetric water content, and correspondingly chloride concentrations (Figure 40). The high degree of spatial variability in the position and thickness of the upper sand and clay layers, even across a few metres, prevents interpreting bulges in chloride concentrations as time-dependent chloride movement. These results confirm the effectiveness of periodic wet seasons and flood events, such as the 2010–11 wet season, in flushing accumulated salts.

Table 9: Estimation of total chloride losses or gains and deep drainage (leachate) between June 2010 and May 2011 after 1760 mm rainfall at Forest Hill and Mulgowie sites.

Site	Modelled Depth	Total Cl 2010	Total Cl 2011	Total Cl Lost	Irrigation	Deep Drainage	Av Cl conc. in Leachate 2011*	Estimated av. EC conc. in Leachate 2011*
	(m)	(t/ha)	(t/ha)	(t/ha)	(mm/yr)	(mm/yr)	(mg/L)	(µs/cm)
Forest Hill	0.4–3.1	10.3	2.5	7.8	0	870	630	1483
Mulgowie	1.3–3.3	8.4	3.3	5.1	0	775	694	1633

* Average concentration in leachate is the profile average (across the depth range) calculated from the volume of water draining (DD) and the total change in Cl or EC (plus inputs in irrigation and rain) in the profile over the 12.5–14 year time span



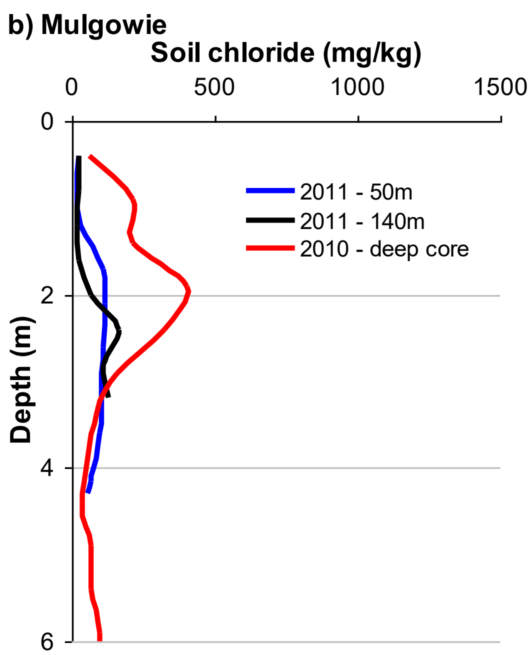


Figure 39: Changes in measured soil chloride between 2010 and 2011 from cores taken along ERT transects before and after the extreme wet season for a) Forest Hill and b) Mulgowie sites. Refer to images in Figure 41 and Figure 44 to locate positions along transect lines.

5.4 Electrical Resistivity Imaging – Assessment of Spatial Variability in Water, Salt and Soil Texture within the Landscape

Electrical resistivity tomography (ERT) is a rapid non-destructive way of determining soil electrical conductivity (EC) along paddock-scale transects. ERT measures absolute values for local resistivity/conductivity as a function of depth and provides dense datasets of true bulk electrical conductivity (EC) from image inversion (Loke and Barker, 1995). EC is predominantly a function of pore water, salinity and clay content and so provides information on soil water content, salt concentrations and textural variability. When one or two of the primary factors (water, salt or clay) are reasonably spatially constant then the image can be used to assess variability of the others. ERT can be used to monitor soil water within the root zone or deeper vadose zone. It can also be used for assessing long term patterns in water movement through the deeper vadose zone (Foley *et al.*, 2010; Timms *et al.*, 2011).

5.4.1 Transects

Transects were imaged in 2011 at the Glenore Grove, Forest Hill and Mulgowie deep coring sites. ERT images were taken using an ABEM SAS4000 Terrameter and LUND ES464, with Wenner alpha electrode configuration and an electrode spacing of 2.5 m or 5 m along transects, generating detailed resistivity profiles (> 460 measurements). Field-measured apparent resistivity was modelled using RES2DINV version 3.59 (www.geoelectrical.com) to obtain the modelled true bulk electrical resistivity.

At the Forest Hill site, two transects were imaged, a 68 m deep transect that ran 480 m along the length of the paddock and 30 m into the paddock from the roadway, and a second 12 m deep 'shallow' profile that ran for 80 m parallel to the deeper transect line but 10 m closer to the road and running over the top of the deep core location. The soil was in bare fallow at the time of imaging. The Glenore Grove and Mulgowie sites were each imaged to 24 m depth along 160 m transects running over the top of the deep core locations. Both sites were planted to lucerne, with images taken after cutting and baling. Gatton and Tenthill core sites were not imaged. Building infrastructure and a bitumen car park were too close to the core location at Gatton (10 m away). And the paddock was planted to lettuce (susceptible to trampling) with numerous above ground irrigation pipes, making imaging infeasible at Tenthill.

A soil-coring rig was used to strategically core along Forest Hill and Mulgowie transects to measure soil volumetric water, electrical conductivity, chloride and particle sizes in 0.3 m increments down profiles. Soil cores were not taken in the paddock at Glenore Grove to prevent damage to lucerne plants.

5.4.1.1 Spatial Variability of Soil Texture and Water Content along the Forest Hill Transects

Soil texture is highly variable in the top 6 m at Forest Hill (Figure 40) spatially and with depth. Each core has a unique signature, and averaging soil water content or other parameters would not be meaningful. All three cores shown in Figure 40 were sampled within 35 m of each other and within 30 m of the deep core, depicted by the black and white columns on the ERT image (Figure 41). The rapid textural changes are due to the way the fining-up layers were historically formed and laid down.

Water content is strongly related to clay content and the two curves shadow each other down each of the profiles. The clay-rich layers are at (or close to) saturation. This is confirmed by soil suction measurements (Figure 42c) and indicates that the clay layers are limiting flow in the unsaturated zone. The interspersed clay and sand layers seen in the Forest Hill ERT image drive water movement in the landscape. There would be considerable lateral flow and bypass flow, some water perching above clay throttles, and movement via established pathways between sand lenses. The overall average hydraulic conductivity would be greater than that of the clay but less than that of the sand. This kind of landscape is capable of moving great quantities of drainage water and salts (Figure 39a) over time, given sufficient inputs. This imaged landscape is an example of the catchment hydrogeological model results of Wilson (2005), who also confirms the important role of preferential flow paths, fractures and mass transfer between the more and less mobile fluid domains in accounting for mass transfer rates in the regolith/groundwater.

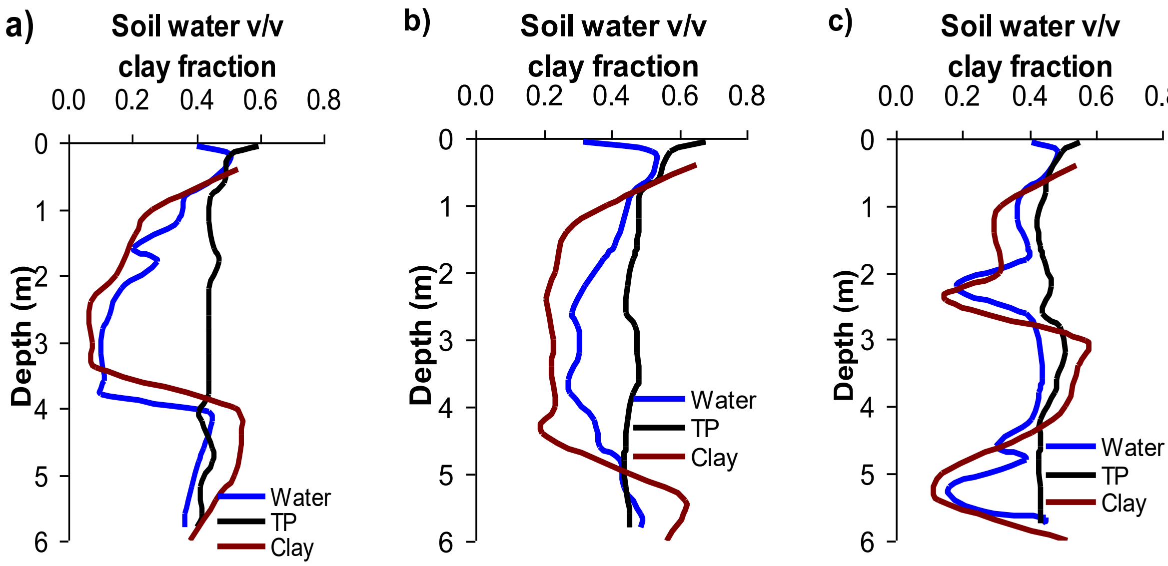


Figure 40: Shallow soil cores taken at Forest Hill along ERT transects for a) 36 m along 20 m transect, b) 70 m along 20 m transect, and c) 260 m along 30 m transect.

The Forest Hill ERT images (Figure 41) provide a snapshot of texture, water and salt variability, with a ten-fold variation in bulk electrical conductivity (20–260 mS/m) occurring in places within the space of only a few metres. Sand layers can be seen in the image as the drier (and low clay) red-yellow-orange colours. The deep core was located where there was a particularly dense and thick layer of sand (2.5–5 m) coloured purple in the shallow transect image (Figure 41b).

These images were taken after the 2010–11 wet season. An additional 1000 mm of rain (on top of the long-term average annual rainfall) fell between deep coring in June 2010 and ERT imaging in May 2011. The water table rose from 18.6 m b.g.l. when the deep core was taken to 13.5 m b.g.l. when imaged. The new position of the water table can be clearly seen in the deep transect image as the zone of widespread blue colour. Usually blues in the image would be associated with wet clay-rich layers, with groundwater alluvium as the yellow and tan areas lower in the image. However, the water table has risen 3 m beyond the alluvial storage zone (~16 m, Figure 31a) and into the clayed zone as the water table rose above the top of the aquifer, thus creating confined groundwater conditions.

A simple estimation of the storage capacity between 13.5 b.g.l. and 18.6 m b.g.l. (water table height from 2010 to 2011) can be calculated from the deep core data (Figure 31) by subtracting the 2010 volumetric water content from the total porosity. The original unfilled storage capacity in the 2010 profile, now filled, is equivalent to 620 mm of water. SODICS modelling for this 11-month period estimated 800 mm (870 mm/yr) drained below 3.1 m (Table 9). Both values are high. The amount stored in the regolith will be lower than the total amount drained because some deep drainage will be utilised to wet the unsaturated zone above the groundwater, and some entering the groundwater will have migrated (i.e. it is an unconfined system). Groundwater rises in the Lockyer area are typically dominated by creek recharge (Cox and Wilson, 2005); however, during this intense wet period there has been significant diffuse recharge from deep drainage in this specific area.

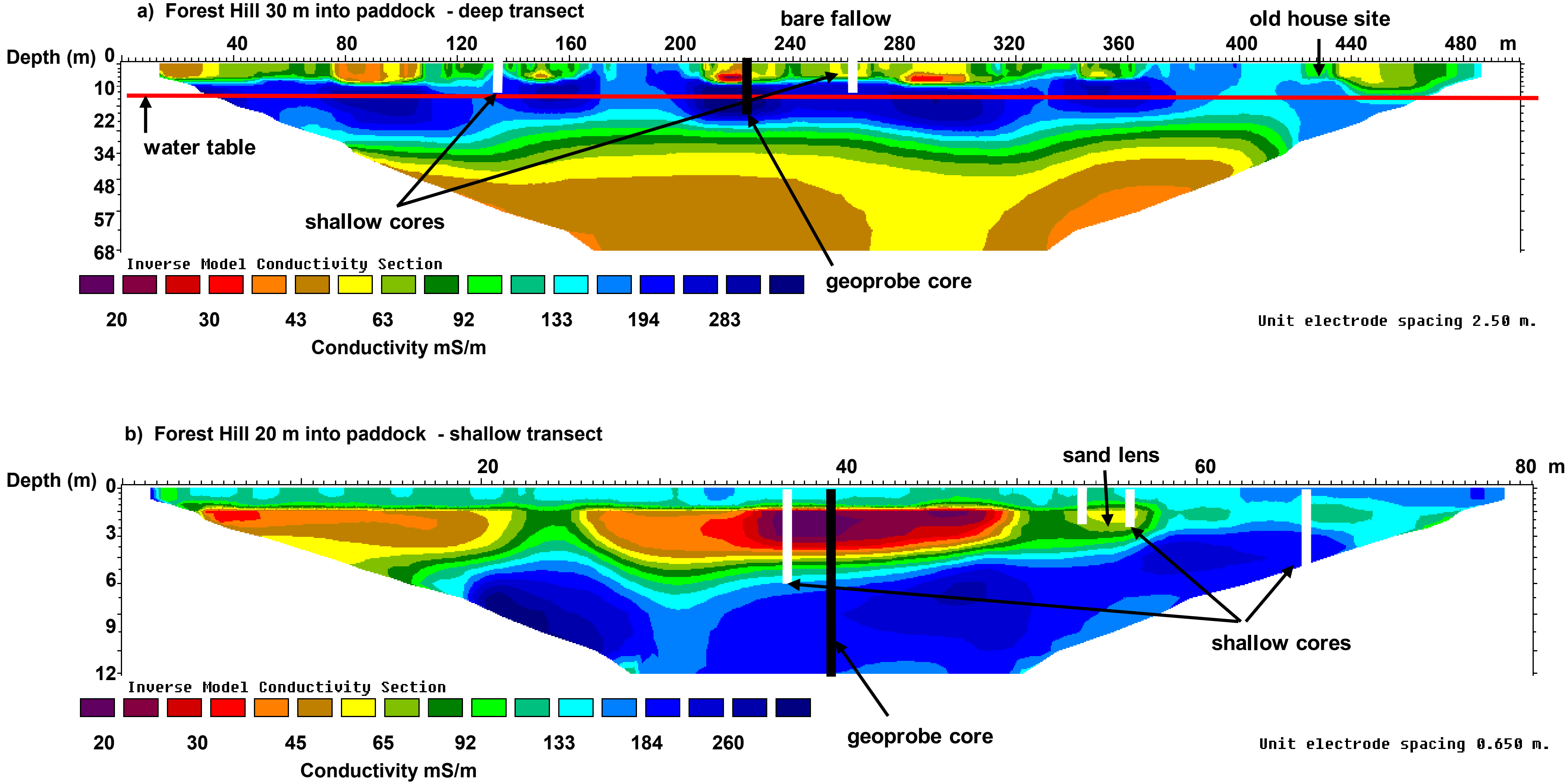


Figure 41: Electrical resistivity imaging along two transects at the Forest Hill Ellis bore site, taken in May 2011.

5.4.1.2 ERT Imaging at Glenore Grove and Mulgowie Deep Coring Sites

Both sites were cropped with lucerne. EC was reasonably uniform along both transects (Figure 43 and Figure 44). The wetter clay rich areas (blue and aqua coloured) had lower EC values than Forest Hill, being drier under lucerne.

An increase in EC, probably due to increasing water content, was imaged along the Glenore Grove transect when moving from bare soil into the region of irrigated lucerne and beyond that, irrigated silverbeet (possibly related to recent irrigation). The entire transect displayed fairly uniform and moderate EC in the top 9 m (Figure 43).

The alluvial aquifer storage zone starts at ~9 m (Figure 29) and this is confirmed by the transition to lower EC in the ERT image. Groundwater rose from 16 m to 13.2 m b.g.l. between coring in 2010 and imaging in 2011. The corresponding water needed to saturate this region of the regolith is 344 mm. Despite the site being under water during the floods of January 2011, 5 months before the image was taken, the upper profile does not appear to be as wet as the other transects.

At the time of imaging the Mulgowie site, a lateral move irrigator had recently passed over land to the left and middle of the image and the profile was quite wet (Figure 42a,b; Figure 44). The shallow band of lower EC to the right of the image had not been recently irrigated (note irrigator positioned to water this strip) and was drier. There is also a textural change to sandier loamy soil at this end of the paddock.

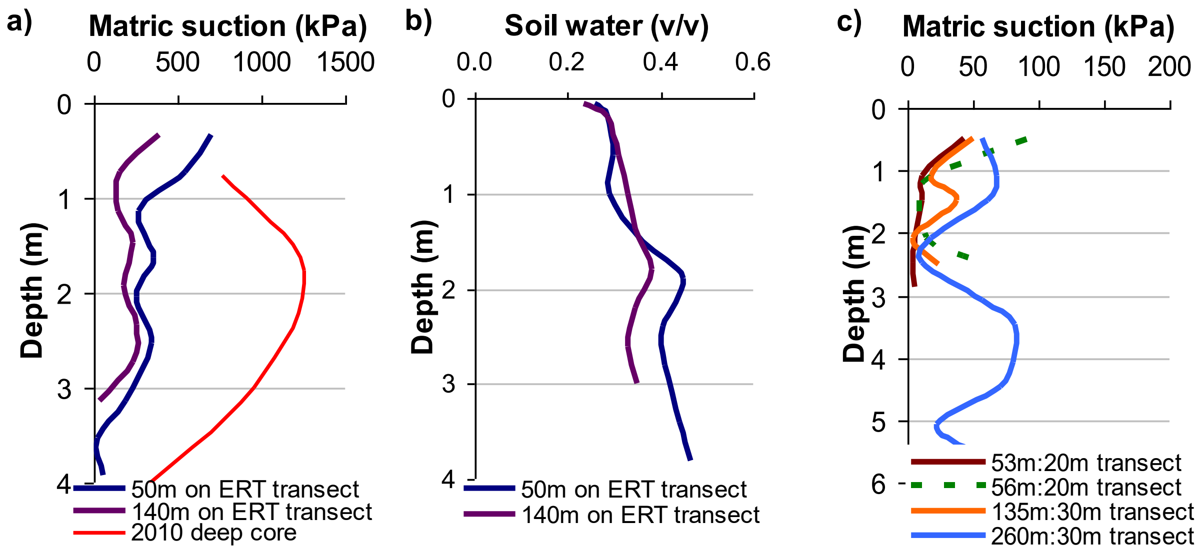


Figure 42: Shallow cores taken in 2011 along ERT transect showing a) matric suction, and b) volumetric soil water for Mulgowie site, and c) matric suction along Forest Hill transects.

At Mulgowie, in response to the extreme wet season, groundwater levels rose from 14.9 m when cored in 2010, to 6.8 m in May 2011 (falling to 8.4 m when imaged in October 2011). The volume of water needed to fill this initially unsaturated zone was 450 mm. SODICS modelling estimated 710 mm (775 mm/yr) drained below 3.1 m (Table 9). As with the Forest Hill site, there is a credible match between deep drainage estimated by SODICS and the volume of regolith saturated, given the limited cores modelled. Results from both this site and Forest Hill clearly show the significant scale of contribution of diffuse recharge (from deep drainage) to groundwater recharge during episodic wet periods.

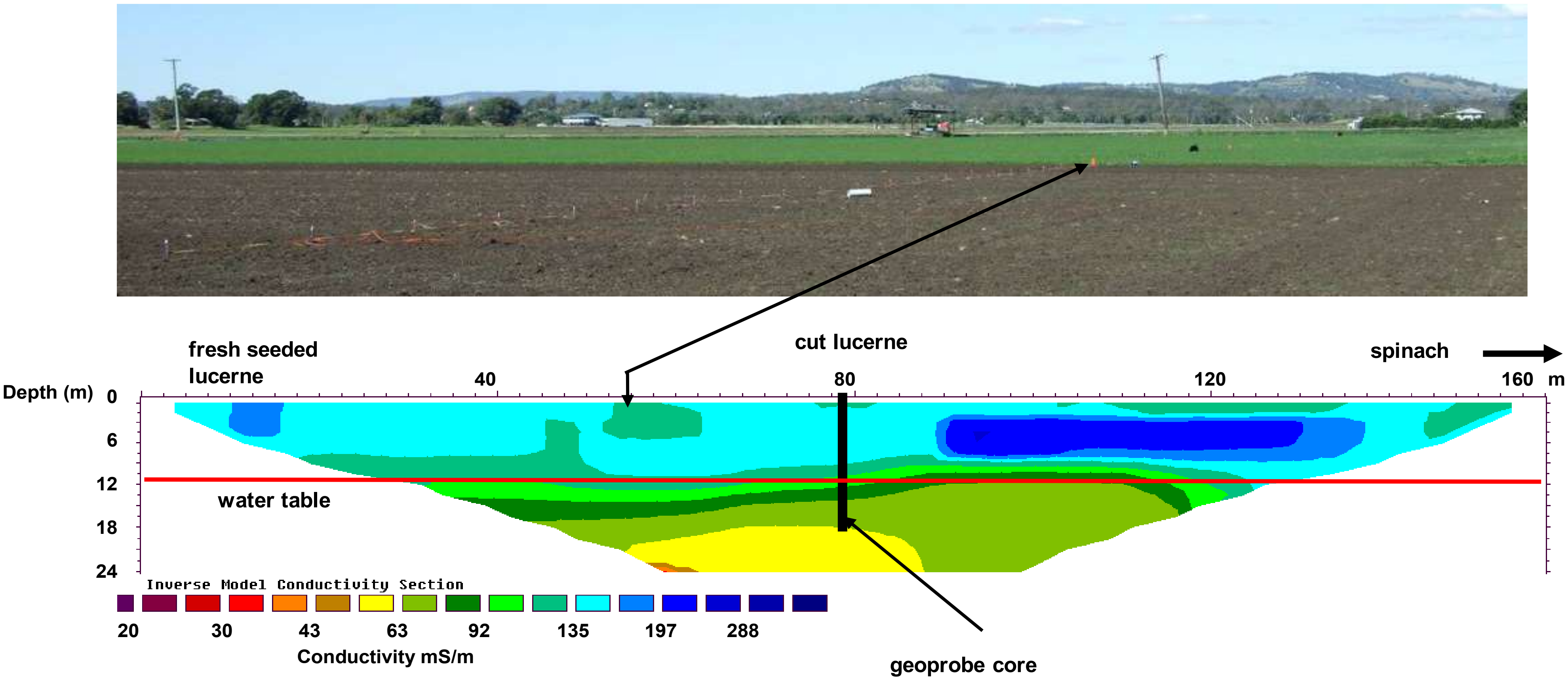


Figure 43: **ERT imaging at Glenore Grove Ellis bore site, taken in May 2011.**

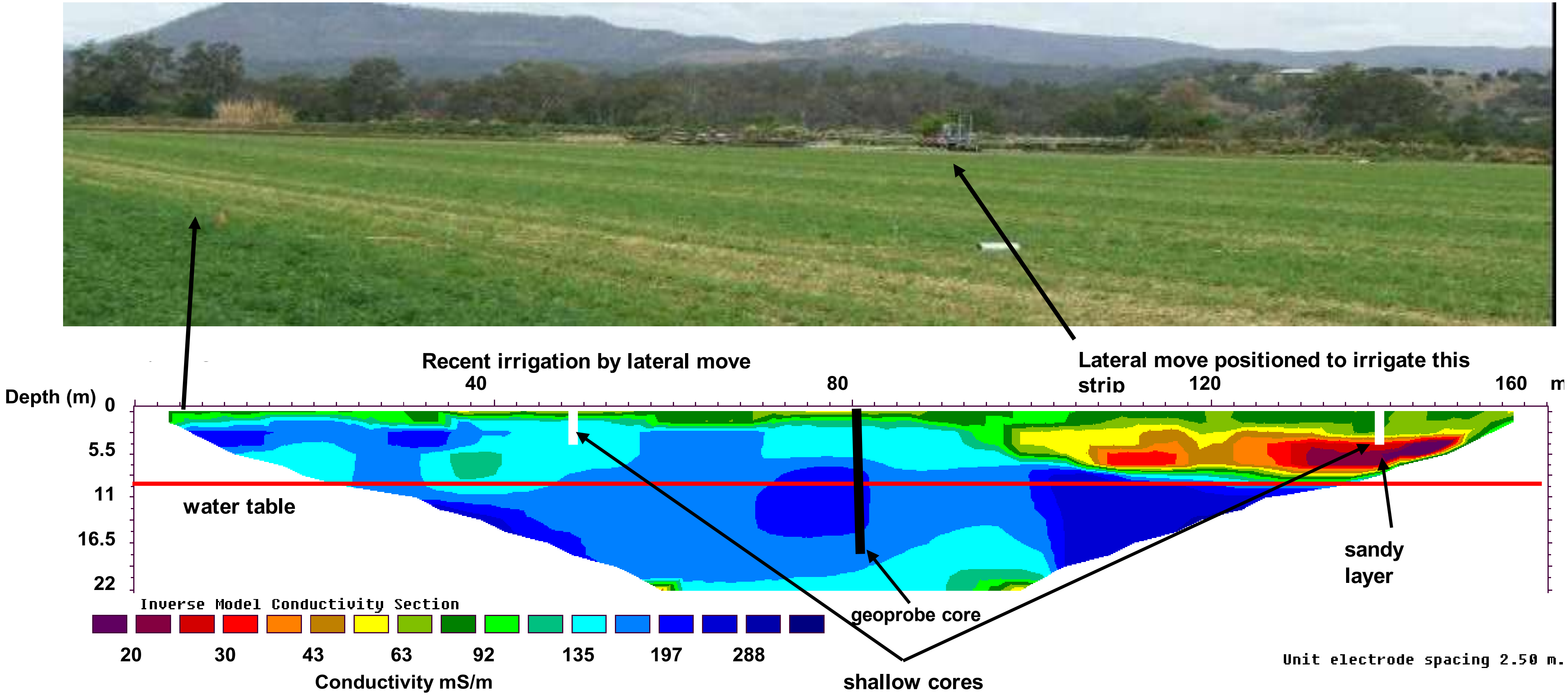


Figure 44: **ERT imaging at Mulgowie Ellis bore site, taken in October 2011.**

5.5 Conclusions

Large quantities of chloride and salt have been lost in the last decade from the regolith at some irrigated sites in the Lockyer Valley, most likely as a consequence of improved irrigation water quality and as a result of the high rainfall amounts since 2007. However, average salinity of the leachate (soil solution) is generally lower than, or similar to, the salinity of the groundwater sampled in close proximity to the coring sites. If less saline groundwater is used for irrigation – for example, PRW is used as a replacement or to augment existing supplies – the regolith and groundwater will progressively become less saline.

Deep drainage estimates from transient soil chloride mass balance and from changes in regolith water storage confirm previous estimates of large rates of deep drainage from some irrigated cropping systems and for the recent extreme wet seasons. It is now clear that both creek recharge and diffuse recharge should be considered in groundwater models of alluvial groundwater systems. Improvements in irrigation water use efficiency should be considered as a means of reducing groundwater use and preserving groundwater for drought periods.

6. DETAILED GROUNDWATER MONITORING AFTER THE 2011 FLOOD

(Thusitha Gunawardena, Jenny Foley, Maria Harris, Mark Silburn, Leif Wolf)

6.1 Introduction

Based on the initial screening of uncertainty in water balance modelling for the Lockyer Valley, it was evident that surface–groundwater interaction is a critical parameter in the estimation of sustainable aquifer yield. Groundwater recharge pulses from creeks may occur rapidly after major rain events, and their analysis requires more frequent monitoring than the monthly or quarterly measurements that are usually carried out. The research project had therefore considered detailed, high time-resolution surface- and groundwater-level monitoring at selected creek locations early on, especially since they provide a meaningful way to validate and constrain groundwater modelling.

By coincidence, the catastrophic floods of January 2011 occurred during the project period. The UWSRA research responded with the immediate deployment of monitoring equipment in the month after the flood receded. Groundwater levels and groundwater quality were monitored in more detail at selected wells close to the creeks in the aftermath of the floods. In a situation of already rising water levels from the wet period starting 2008, the extraordinary heavy rainfalls in December 2010 and January 2011 triggered a major groundwater recharge event which demonstrated the dynamic surface groundwater interaction. Apart from qualitative interpretations, these data provide a major contribution to the calibration of the numerical groundwater model (Chapter 12) and the Eigenmodel method (Chapter 13). The full details of the investigations carried out are documented in Foley and colleagues (forthcoming); only selected results are reported here.

6.2 Surface Water Levels

Gauging height data for Lockyer, Tenthill, Laidley and Sandy Creeks were extracted from the SPIN database to investigate creek heights during both peak and average flows. Locations of gauging stations are shown in Figure 45.

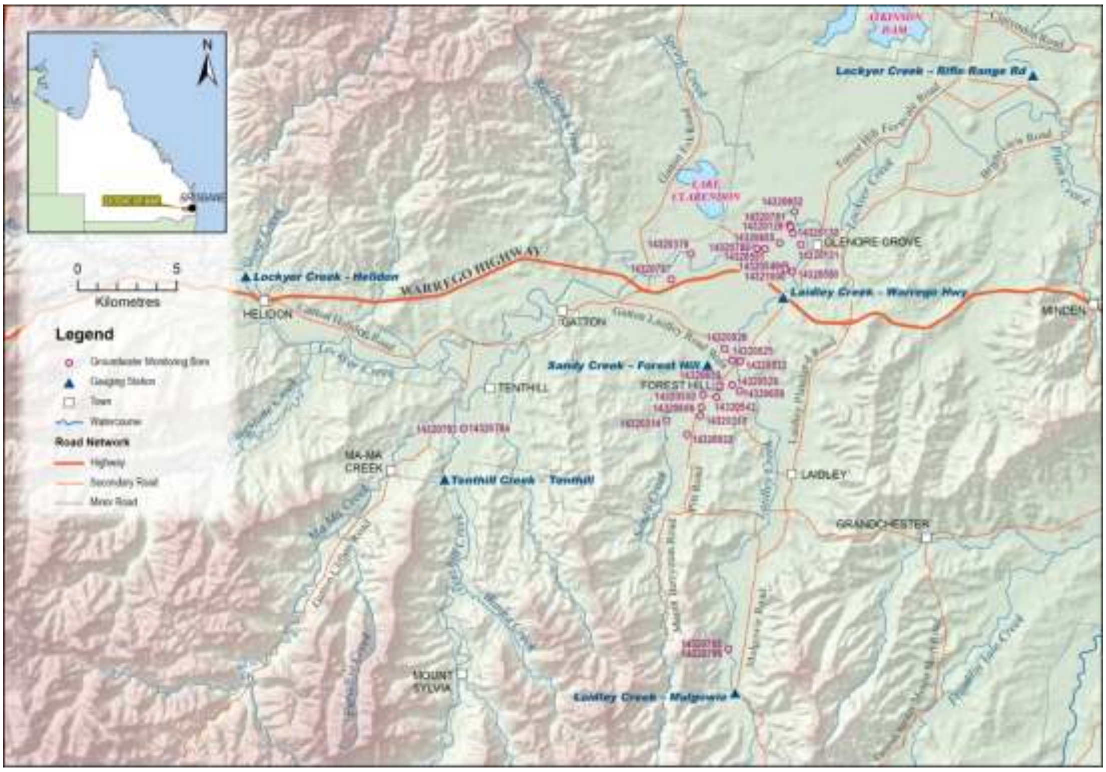
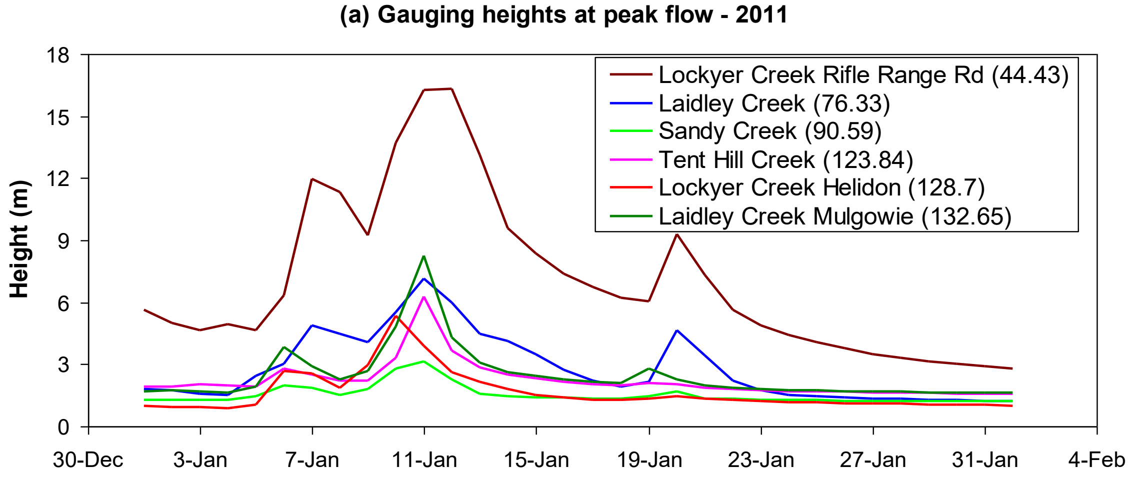


Figure 45: Locations of gauging stations and groundwater monitoring bores in Glenore Grove, Gatton, Forest Hill, Tenthill and Mulgowie transects.

Lockyer, Laidley, Tenthill and Sandy Creek heights peaked during 10–12 January 2011 floods, varying from 3.2 to 8.3 m except for the downstream gauging station on Lockyer Creek at Rifle Range Road (Figure 46a). Further upstream, creek heights (Australian Height Datum [AHD] of 132.65–76.33 m) never exceeded 8.3 m during peak flow. Lockyer Creek at Rifle Range Road peaked at 16.3 m on 11 January then fell before peaking again at 9.3 m on 20 January. During the nine-day period between peaks it remained at > 6 m at this downstream gauging point. By 23 January 2011 all creek heights had dropped to < 2 m except in Lockyer Creek at Rifle Range Road (Figure 47a). During January 2011, the number of days with stream height above 1.5 m were 10 (Lockyer Creek, Helidon), 31 (Laidley Creek, Mulgowie), 24 (Laidley Creek, Warrego Hwy), 31 (Tenthill Creek) and 10 (Sandy Creek).



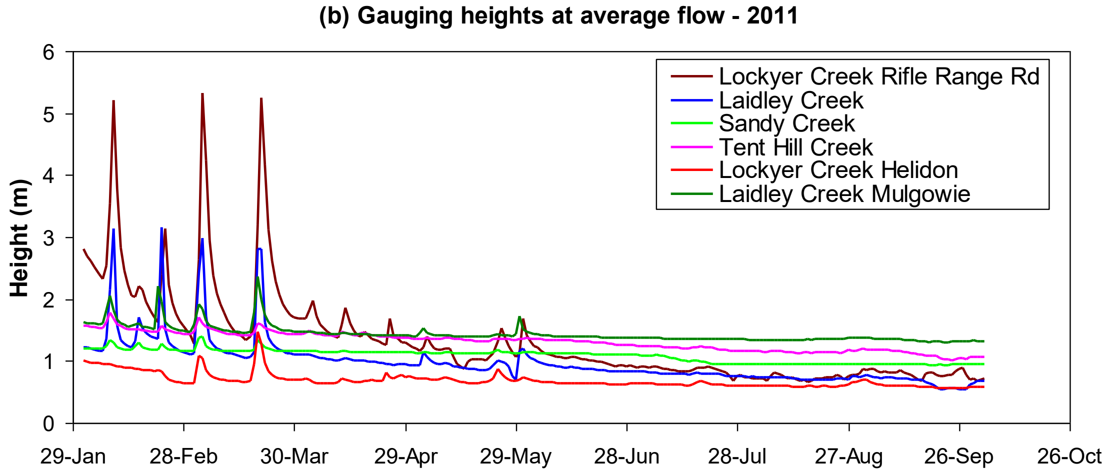


Figure 46: Gauging heights (a) during the peak flow in January 2011 and (b) during average flow (0.5–1.3 m). Numbers within brackets in legend of (a) show AHD. Gauge distance from stream mouth (or Adopted middle thread distance [AMTD]): Lockyer Creek 22.6 and 99.3 km; Laidley Creek Warrego Highway and Mulgowie at 5 km and 31 km; Sandy Creek at 2.5 km; Tenthill Creek at 14.6 km.

After late January 2011 there were a number of 3 m peaks in Laidley Creek stream heights, during the period 9 February – 21 March (Figure 46b) in response to small (20–30 mm) rainfall events. The whole system was still highly responsive to further runoff inputs. The rest of the gauging stations recorded less than 2.3 m heights for the same period, with the exception of Rifle Range gauging station. Since late April 2011 all creek heights have been < 1.5 m.

6.3 Groundwater Monitoring Methodology

Five hypothetical transects were selected for the investigation representing the major part of the catchment. These bores were in Glenore Grove, Gatton, Forest Hill, Tenthill and Mulgowie irrigations areas across Lockyer, Laidley and Sandy Creeks (Figure 45). Details of the eight logged and 22 manual dip bores are shown in Table 10.

Table 10: Details of the bores monitored in the Lockyer Valley.

Registered Number**	Easting	Northing	AHD (m)	Installation Date	Depth of Bore (m)	SWL (m) (2011)	Bore Line	Aquifer
14320129	439991	6955777	84.25	29/05/68	25.3	–12.54	GGL	Alluvium
14320130	440192	6955347	83.05	24/05/68	32.6	–16.47	GGL	Alluvium
14320131	440578	6954800	82.42	22/05/68	26.5	–19.51	GGL	Alluvium
14320313	435502	6946259	99.59	09/06/71	23.3	–10.67	BLE	Alluvium
14320314	433962	6945917	107.91	16/06/71	8.0	–3.94	BLE	Alluvium
14320379	435025	6954335	93.03	21/04/76	31.4	–11.77	LAC	Alluvium
14320525	437124	6948947	88.85	14/12/87	31.0	–7.66	BLE	Alluvium
14320526	436731	6949585	89.73	07/07/87	27.7	–21.15	BLE	Alluvium
14320528	437140	6947756	92.53	04/07/87	30.0	–9.30	BLE	Alluvium
14320532	437507	6948953	88.29	16/07/87	24.0	–2.21	BLE	Alluvium
14320542	436320	6947186	95.59	24/04/82	29.3	–16.44	BLE	Alluvium
14320549	439739	6953749	83.05	17/11/88	22.5	–15.44	BLE	Alluvium
14320550	440135	6953443	81.72	17/11/88	30.5	–18.35	BLE	Alluvium
14320551	438765	6954573	86.47	17/11/88	39.5	–25.33	BLE	Alluvium
14320555	435634	6947306	96.31	07/12/88	28.0	–8.17	BLE	Alluvium
14320652	440215	6956442	83.75	23/10/90	36.0	–10.99	LAC	Alluvium
14320655	439623	6955049	84.45	24/10/90	34.1	–22.63	BLE	Alluvium
14320658	437524	6947448	90.65	25/10/90	24.1	–6.50	BLE	Alluvium
14320659	436465	6947762	92.80	26/10/90	26.0	–11.63	BLE	Alluvium
14320781	440065	6955639	84.10	01/12/95	50.0	–12.90	BLE	BR
14320782	440057	6955654	84.08	01/12/95	34.0	–15.82	BLE	Alluvium
14320783	423502	6945645	128.30	02/02/96	40.0	–18.68	BLE	BR
14320784	423482	6945647	128.41	02/02/96	28.0	–18.14	BLE	Alluvium
14320785	436940	6934616	134.21	08/02/96	31.5	–7.70	BLE	BR
14320786	436927	6934606	134.19	08/02/96	41.0	–8.04	BLE	BR
14320787	434048	6953015	90.03	01/01/96	32.0	–15.32	BLE	Alluvium
14320788	438338	6954613	88.62	01/01/96	36.0	–20.37	BLE	Alluvium
14320805	435555	6946690	99.23	27/10/97	47.7	–16.86	BLE	BR
14320806	435555	6946659	99.37	28/10/97	25.5	–15.76	BLE	Alluvium
14320922	434822	6945307	104.13	03/03/05	24.7	–10.07	BLE	Alluvium

AHD - Australian Height Datum; SWL - standing water level; GGL - Glenore Grove Line; BLE - Blenheim Line; LCA - Lake Clarendon area; BR - bedrock.

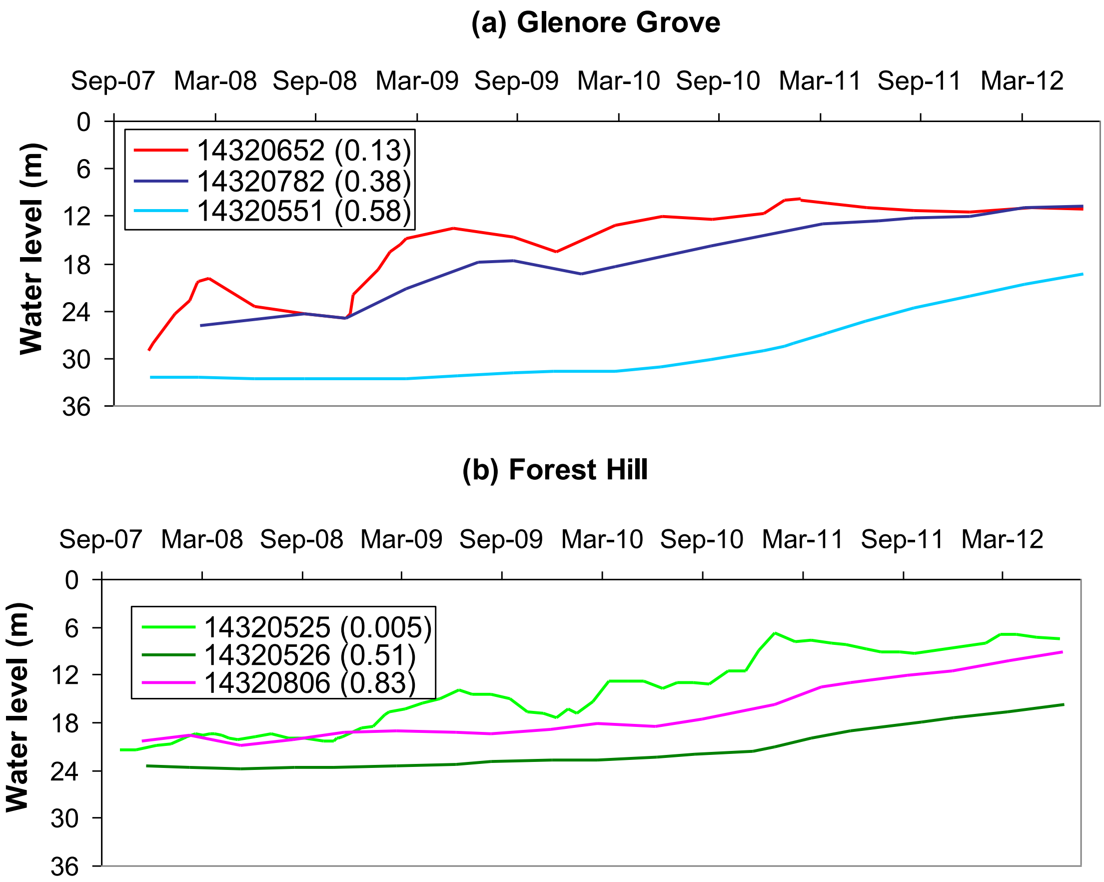
** First three digits (143) basin number; fourth digit (2) shire number; bore number is last four digits

Historical groundwater data (from manual dipping) were extracted from the Spatial Information Network version 4.1.0 (SPIN) from the original groundwater database to plot the last 4–5 years of groundwater levels (pre- and post-wet period) from bores along the three transect lines in Glenore Grove, Forest Hill and Gatton.

Groundwater recharge responses to the 2010/11 extreme wet season were monitored in detail along the three transect lines within the Glenore Grove, Forest Hill and Gatton areas, commencing in February and March 2011. Groundwater loggers were installed (CTD®, TD® divers and Solinst) (Schlumberger company ®) in eight of the monitoring bores, including the Ellis alluvial bores at Glenore Grove (RN 14320782; CTD), Forest Hill (RN 14320806; CTD) and Gatton (RN 14320787; TD). Additional bores were equipped with loggers in each area to create landscape transects (close to creek and distanced from creek) to provide detailed information on the travel time of the recharging waters through the regolith. Bores equipped included RN 14320551 and RN 14320652 at Glenore Grove, RN 14320525 and RN 14320526 at Forest Hill and RN 14320379 at Gatton. Water levels were logged every three hours and readings barometrically compensated from barometric data obtained from a Baro-Diver (Mini-Diver) installed centrally in Gatton bore RN 14320787 suspended at 2 m. Each water depth logger was suspended on monofilament fishing line in the range of 20–25 m depth based on the estimated rises and historic depths. Electrical conductivity (EC) was logged at three bores, Glenore Grove (RN 14320782; CTD diver) and Forest Hill (RN 14320525 and RN 14320806; CTD), to monitor salinity changes.

6.3.1 Medium-Term Groundwater Trends

The major trend observes since September 2007 is the almost continuous rise in groundwater levels up to May 2012 (Figure 47). There are a few minor exceptions provided by wells very close to the creeks (RN 14320652, RN 14320525 RN 1432379), which had minor drops in groundwater level around April 2011 after the floods receded. Still, all wells further than 300 m away from the creeks showed a constant rise in water levels. In summary, water levels in all but one of the transect bores are currently near or above their highest historic water levels, thus setting a new record. This may represent close to the maximum aquifer capacity.



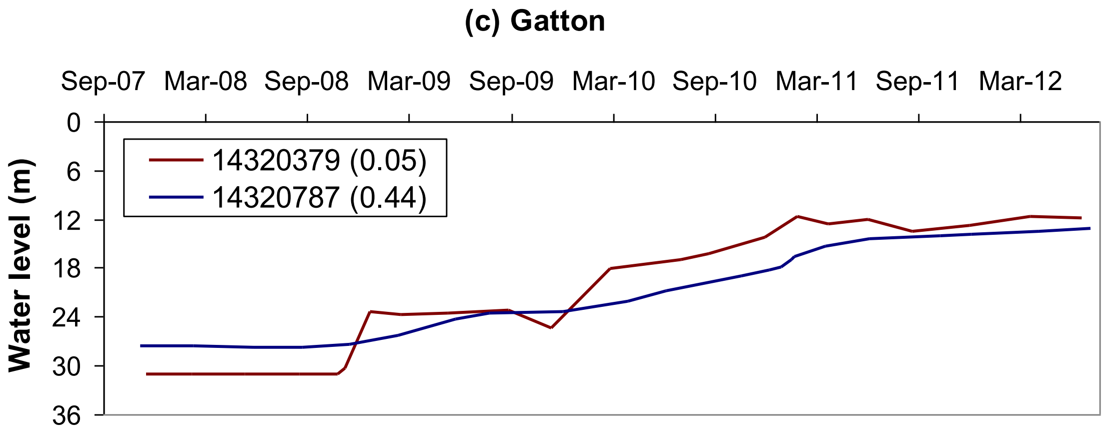
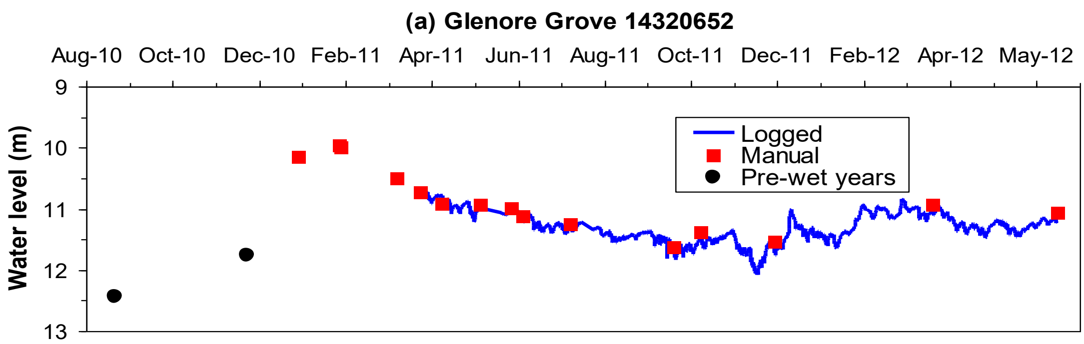
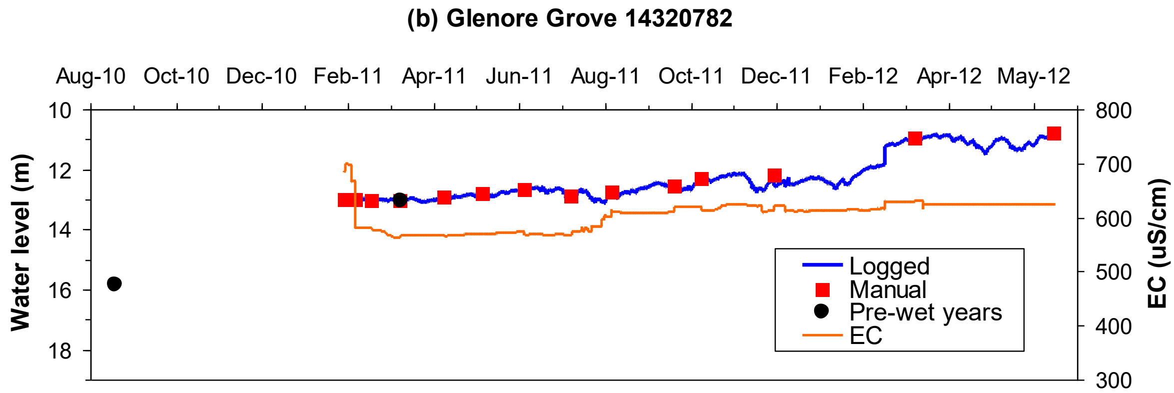


Figure 47: Recent historical pre-wet to wet years groundwater level from (a) Glenore Grove, (b) Forest Hill and (c) Gatton. Distance (km) from the creek is shown in brackets. Data are from SPIN database.

6.3.2 High Time Resolution Monitoring of Groundwater Quantity and Quality

The comparison of groundwater hydrographs in different distances to the creeks along hypothetical transects demonstrates the non-uniform response of the aquifer to the flood (Figure 48). Wells close to the creek showed declining groundwater levels already three months after the flood (Figure 48a), whereas wells in greater distance continued with a long-term steady rise for more than 12 months after the flood (Figure 48b and Figure 48c). At the end of the research project in June 2012, groundwater levels reached historic high levels in many wells.





(c) Glenore Grove 14320551

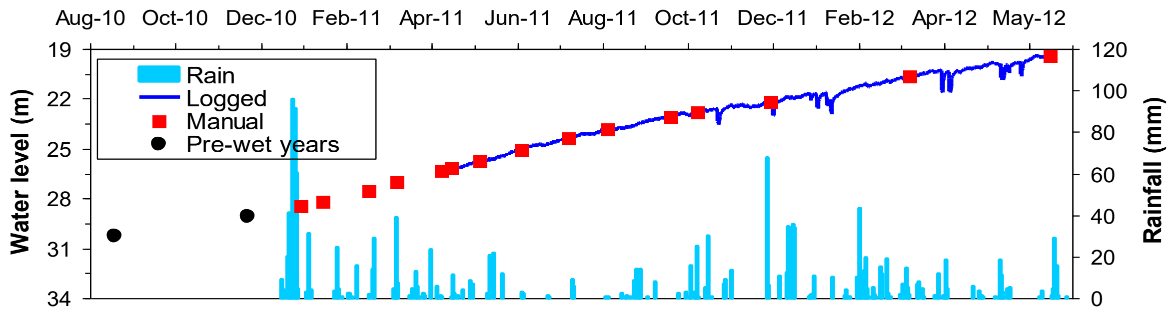
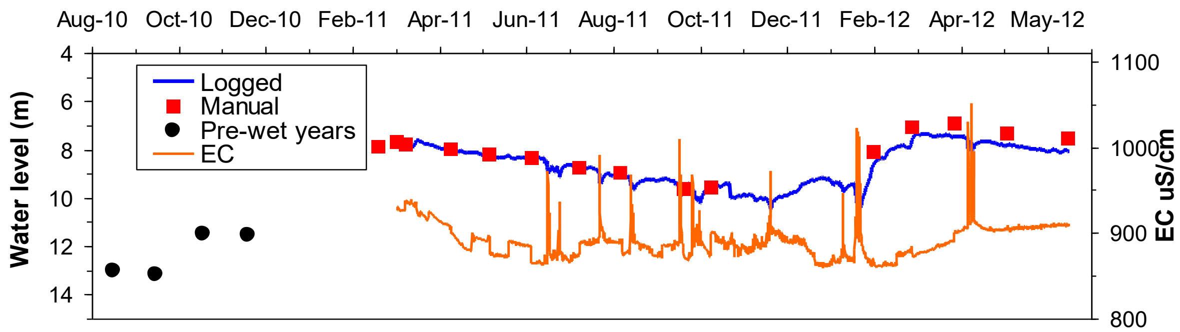


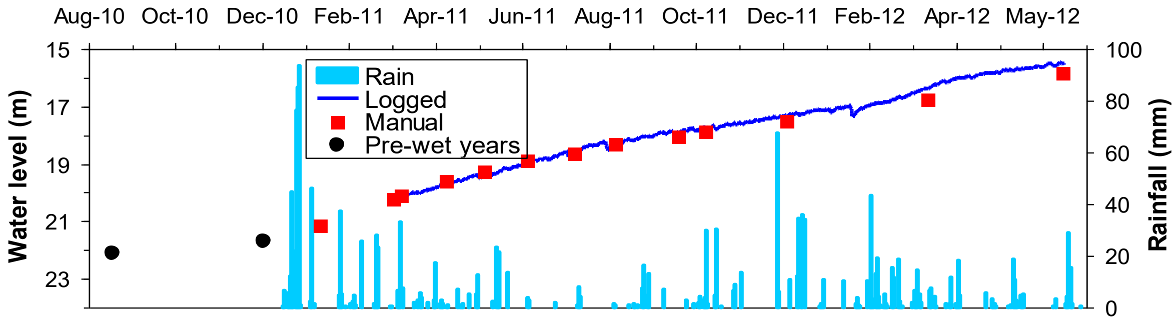
Figure 48: Post-wet years groundwater levels in three bores of the Glenore Grove transect (a) 14320652 (0.13 km from creek), (b) 14320782 (0.38 km from creek) and (c) 14320551 (0.58 km from creek). Rainfall for same period is also shown. Groundwater EC was logged in bore 14320782. (1000 uS/cm = 1 dS/m).

The immediate impact of pumping on groundwater quality could also be observed by simultaneous high frequency monitoring of groundwater level and electrical conductivity (Figure 49). Well RN14320525 and RN 14320806 in Forest Hill (Figure 49a and c) demonstrated increased salinity (up to approximately 10% to 20% increase) correlated with short-term water level declines. A likely hypothesis for this behaviour is that the monitoring well is close to an interface between fresh and more saline water. This interface is moving in accordance to the pumping of a nearby irrigation well.

(a) Forest Hill 14320525



(b) Forest Hill 14320526



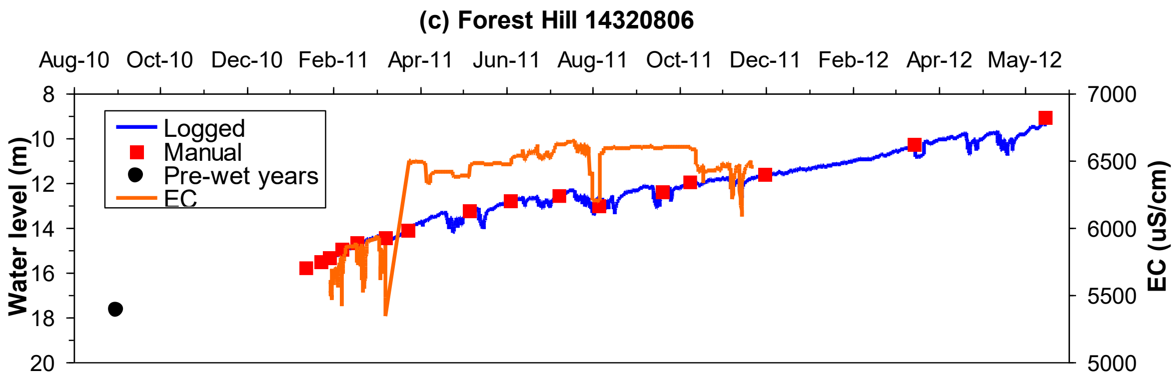
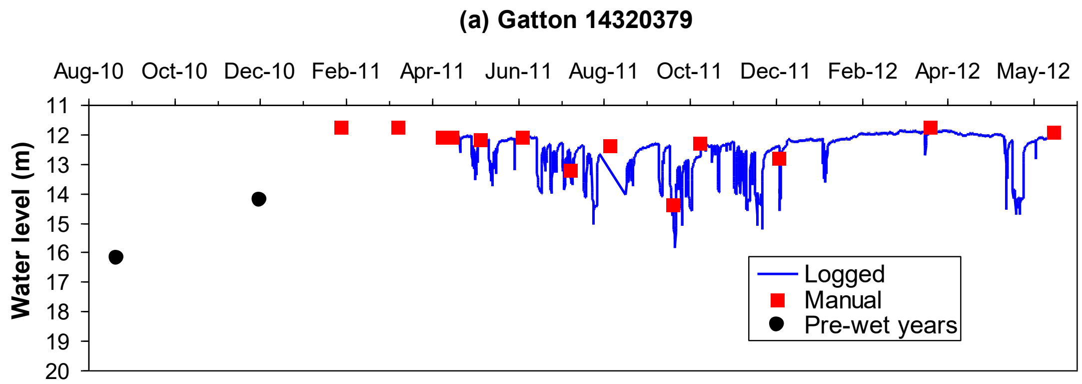


Figure 49: Post-flood groundwater recharge discharges in three bores of the Forest Hill transect (a) 14320525 (0.005 km from creek), (b) 14320526 (0.51 km from creek) and (c) 14320806 (0.83 km from creek). Rainfall for same period is also shown. Groundwater EC was logged in bores 14320525 and 14320806 (1000 uS/cm = 1 dS/m).

Groundwater level was more responsive to creek flow in the aquifer near Gatton (Figure 50a). Due to proximity to Lockyer Creek, groundwater level was frequently changing, with a maximum fluctuation of 4 m as noted for some other aquifers near Glenore Grove. Responsiveness was driven by pumping of the irrigation bore and changes in creek flow. The smoothness of the logged water level in a deeper aquifer (Figure 50b) shows delayed long-term recharge.



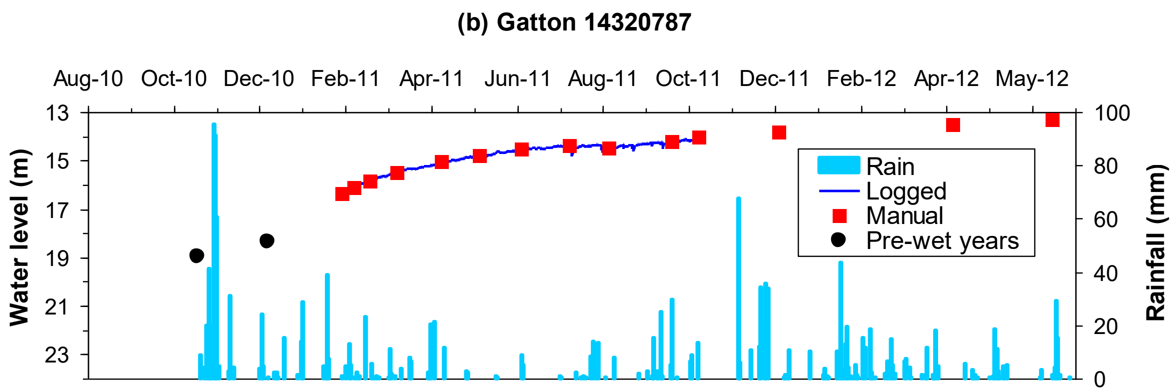


Figure 50: Post-flood groundwater recharge discharges in selected two bores of the Gatton transect (a) 14320379 (0.05 km from creek) and (b) 14320787 (0.44 km from creek). Rainfall for the same period is also shown.

6.3.3 Groundwater Quality Monitoring

A small set of paired bores with screens in the alluvial aquifer and the underlying bedrock were screened for hydrochemical parameters (Table 11). One of the main reasons for this campaign was to provide a baseline on the difference between the water quality in the bedrock and the alluvium. It is suggested to resample those wells in the coming years in order to monitor the differential impact which the wet years post-2008 will have on the two aquifers, that is, if a freshening of bedrock aquifers indicates ingress of alluvial water into the bedrock.

On average, groundwater alkalinity values were in the range 326 mg/L $CaCO_3$ (Glenore Grove) to 577 mg/L $CaCO_3$ (Tenthill) (Table 11). There is a slight to moderate hazard associated with irrigation water with Sodium Adsorption Ratio (SAR) of 3–9 when surface applied, but this is mitigated when the EC is > 1200 uS/cm. Nitrate (NO_3 - N) levels varied from < 0.5 to 25 mg/L; concentration above 100 mg/L is not recommended for livestock (Sundaram *et al.*, 2009). Salinity (EC) varied from 760 to 6600 uS/cm and the SAR ranged from 1.3 to 24. An EC of 650–1300 and 1300–2900 uS/cm is considered suitable for irrigation of moderately sensitive and moderately tolerant crops, respectively (SalCon, 1997).

Historical groundwater EC were extracted from the SPIN version 4.1.0 from the original groundwater database to plot average groundwater EC in Glenore Grove, Forest Hill, Gatton, Tenthill and Mulgowie (Figure 51). EC varied from 1033 to 6839 uS/cm.

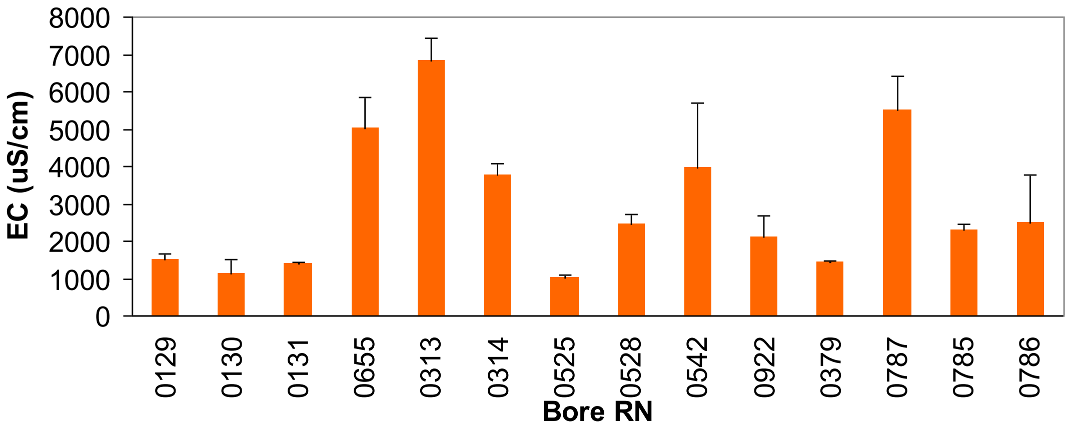


Figure 51: Average electrical conductivity (EC) for Glenore Grove (0129, 0130, 0131, 0655), Forest Hill (0313, 0314, 0525, 0528, 0542, 0922), Gatton (0379, 0787) and Mulgowie (0785, 0786). Vertical bars indicate (±) SD. Averages are from minimum of 4 and maximum of 16 measurements.

6.4 Conclusions

Groundwater levels rose continuously since 2008 and reached historic high marks in many wells by mid-2012. Individual wells showed rapid increases in salinity related to suspected pumping activities. In the three months following the flood, groundwater levels close to the creek (e.g. closer than 300 m) were falling while those further away kept rising. Average Sodium Adsorption Ratio in groundwater was 5.8 but may reach SAR 24.

In summary, the water level monitoring demonstrates the need for a transient groundwater model, calibrated also with high time resolution data of creek water levels and which acknowledges the small-scale spatial heterogeneity of the aquifer to arrive at the required predictive capabilities which are required for dynamic aquifer management.

Table 11: Water chemistry from pumped water samples taken during early February 2011 (post-wet) from Ellis bores.

Chemical Property	Unit	Glenore Grove		Forest Hill*		Gatton		Mulgowie		Tenthill	
		14320781	14320782	14320805	14320806	14320775*	14320787	14320785	14320786	14320783	14320784
Total Hardness (as $CaCO_3$)	(mg/L)	94	282	506	2430	574	388	1950	491	1180	1050
Temporary Hardness ($CaCO_3$)	(mg/L)	94	282	506	395	441	369	754	491	604	551
Alkalinity ($CaCO_3$)	(mg/L)	329	323	3000	395	441	369	754	594	604	551
Residual alkalinity	(meq/L)	4.7	0.8	50	0	0	0	0	2	0	0
Silica	(mg/L)	27	39	47	41	26	41	50	22	40	42
Total dissolved ions	(mg/L)	645	600	5220	3620	1990	851	4240	2520	1840	1610
Total dissolved solids	(mg/L)	474	441	3410	3410	1750	666	3820	2180	1510	1310
True Colour (Hazen)		<1	<1	5	<1	<1	2	<1	7	<1	1
Turbidity (NTU)		7	301	49	539	11	13	34	298	19	50
pH Sat (calc for $CaCO_3$)		7.8	7.4	6.3	6.4	7	7.2	6.1	6.7	6.5	6.6
Saturation Index		0.4	0.5	1.1	0.9	0.6	0.7	1.3	0.7	1.2	1.2
Mole Ratio		1.2	1.6	1.7	3.6	3	1.9	3.4	2.9	2.3	2.4
Figure of Merit Ratio		0.3	2.2	0.2	3.2	0.6	1.7	1.3	0.3	4.6	5.1
pH		8.14	7.86	7.36	7.25	7.61	7.83	7.39	7.43	7.69	7.74
HCO_3 -	(mg/L)	389	391	3650	480	535	444	918	720	729	667
CO_3 2-	(mg/L)	5.7	1.8	6	0.7	1.3	2.7	1.2	2.2	4.1	2.4
EC	(uS/cm)	810	760	5200	6200	3100	1200	6600	4000	2600	2300
Na^+	(mg/L)	150	59	1240	348	465	102	697	661	118	94
K^+	(mg/L)	1.6	1.1	9.5	2.8	3.9	2	4.4	7.8	4	2.8
Ca^{++}	(mg/L)	16	36	55	327	75	56	331	105	184	156
Mg^{++}	(mg/L)	14	47	90	393	94	60	273	56	175	160
SAR		6.7	1.5	24	3.1	8.4	2.3	6.9	13	1.5	1.3
Cl^-	(mg/L)	54	42	170	1900	810	130	1700	970	480	370
Fl^-	(mg/L)	0.25	0.13	0.19	0.13	0.65	0.13	0.17	0.57	0.13	0.08
NO_3^-	(mg/L)	3.3	4.9	<0.5	25	<2.5	1.4	<5.0	<5.0	19	11
SO_4^{-2}	(mg/L)	11.5	18	<1	92	<10	55	291	<20	126	137
Fe	(mg/L)	0.02	<0.01	0.03	<0.01	<0.01	<0.01	0.02	<0.01	<0.01	<0.01
Mn	(mg/L)	0.04	<0.01	0.19	<0.01	0.12	<0.01	2.1	0.13	0.13	<0.01
Zn	(mg/L)	<0.01	<0.01	0.13	0.04	<0.01	0.01	0.04	0.01	<0.01	<0.01
Al	(mg/L)	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
B	(mg/L)	0.07	0.05	0.28	0.11	0.11	0.05	0.88	0.2	0.04	0.03
Cu^{++}	(mg/L)	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03

Note: 14320781, 14320783, 14320785, 14320786 and 14320805 bores in bedrock aquifer. *This bore was not monitored for groundwater level (GWL) but water sample was taken from it, hence it does not appear in Table 10.

7. BASELINE SCREENING OF PHARMACEUTICALS AND TRACE ORGANICS IN SURFACE AND GROUNDWATER

(Leif Wolf, Darren Morrow, Jun Du, Maria Harris, Malcolm Hodgen, Rai Kookana)

7.1 Introduction

The waters of the Lockyer Creek discharge into the Brisbane River and constitute a fraction of the water which is used by Brisbane's Mt Crosby drinking water treatment plant for the potable supply of Brisbane. As such, any upstream use of treated wastewater needs to be subject to a risk assessment, where the worst case risks of the new technology are compared to the existing status quo.

A frequent concern for public perception is that purified recycled water (PRW) originates from wastewater that contains a significant number of persistent organic pollutants (POP). Extensive quality sampling performed by the operators reveals, however, that the produced PRW is virtually free of persistent organic compounds from wastewater, with just very few positive detects of organic compounds originating from disinfection agents (Seqwater, 2011). As an example, 62 pharmaceuticals and personal care products were tested for in the PRW in 44 samplings. Two of these compounds were detected in some of the samples taken during the period reported, namely:

- salicylic acid at 0.2 μg/L compared to a standard of 105 μg/L; and
- triclosan at 0.01 μg/L, compared to a standard of 0.35 μg/L (Seqwater, 2011).

In order to put this information into context, we wanted to check for the existing background of POPs, or more general trace organic compounds (TrOCs) in the Lockyer Valley before any PRW is imported into the valley.

A field sampling campaign to measure TrOCs has been completed in addition to the field investigations on surface and groundwater quality described in Chapter 4 and Chapter 6. A suite of TrOCs were screened using liquid chromatography tandem mass spectrometry (LC-MS/MS). These included artificial sweeteners, pharmaceuticals, perfluorochemicals (PFCs) and pesticides. Increasingly, selected TrOCs are being used as tracers of domestic waste water in studies that examine the potential influences and interactions with surface and groundwater systems.

The sampling of surface and groundwater in the Lockyer Valley was not designed to be a comprehensive study, rather an initial scoping study to inform future work in this area. No information relating to the profile of artificial sweeteners, PFCs or pharmaceuticals in the surface and groundwater of the Lockyer Valley in South East Queensland has previously been published. This chapter describes the methods and results for samples taken from three groundwater sites, six surface water sites, and a waste water treatment plant near Gatton, Queensland. Pesticides, on the contrary, were sampled in detail 12 years ago (Ellis and Bajracharya, 1999).

Artificial low-calorie sweeteners are consumed in considerable quantities with food and beverages. After ingestion, some sweeteners pass through the human metabolism largely unaffected, are quantitatively excreted via urine and faeces, and thus reach the environment associated with domestic waste water (Buerge *et al.*, 2009). A number of studies, particularly in Europe and the United States, have successfully used sweeteners and pharmaceuticals as tracers of human waste water in both surface waters (Mawhinney *et al.*, 2011; Veach and Bernot, 2011) and groundwater (Van Stempvoort *et al.*, 2011; Wolf *et al.*, 2012).

PFCs are a group of synthetic chemicals that are persistent and bioaccumulative in the natural environment (Sepulvado *et al.*, 2011). Perfluorinated compounds can be found in the effluent of waste water treatment plants, and it has also been reported that PFCs can be transported via atmospheric deposition to surface waters (Simcik and Dorweiler, 2005). Advanced water treatment facilities in Australia can reduce the levels of PFCs, but cannot eliminate them entirely (Thompson *et al.*, 2011).

Pesticides can reach surface waters through domestic waste water, spray drift, and runoff from agricultural land. As such, they may not be an ideal measure of domestic waste water but are an indicator for influence from agricultural activities. In this study we sampled pesticides to check if a) groundwater which would discharge to surface water bodies could introduce pesticides into the Lockyer Creek, and b) if pesticide levels in the surface water exceed levels in groundwater.

7.2 Methods

7.2.1 Sampling Sites

The groundwater sampling sites were selected to represent the worst-case conditions rather than a representative mix across the Lockyer Valley. Groundwater sampling was conducted in close proximity to the Lockyer Creek in distances between approximately 1 and 10 km downstream of the outlet of the Gatton waste water treatment plant. We selected the wells for sampling after checking hydrograph records for the connection the Lockyer Creek. Since the water level in the wells are highly responsive to water level variations in the creek, we assume that they receive a major amount of recharge from the creek.

The surface water sampling sites, on the other hand, are representative for the vast majority of surface irrigation water in the Lockyer Valley, since they were obtained from all of the major surface water reservoirs. Figure 52 below shows the spatial relationship of each of the sampling sites in the Lockyer Valley.

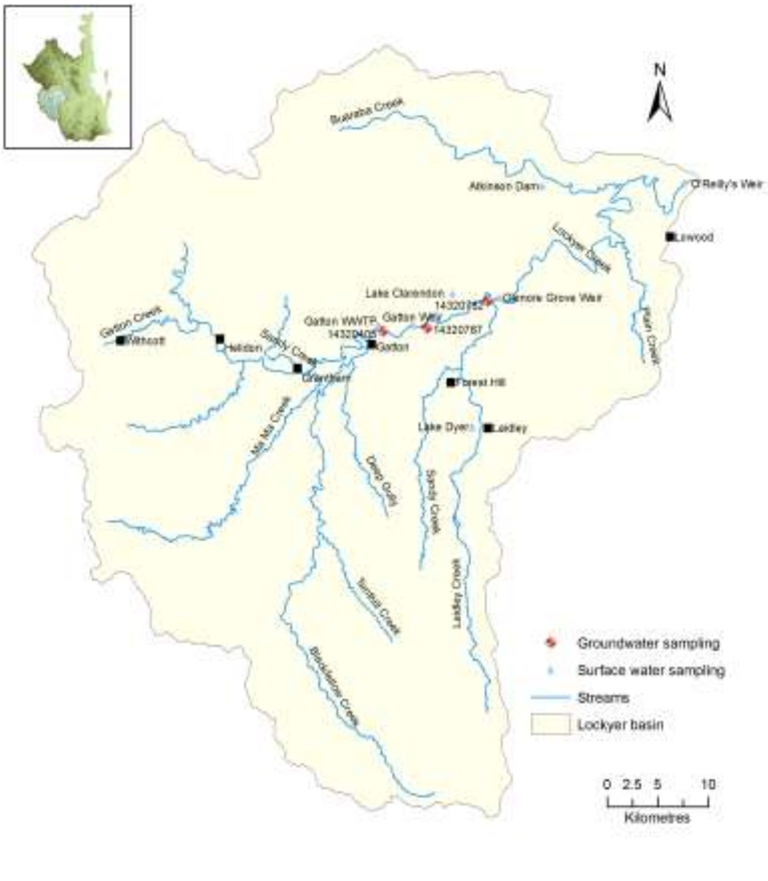


Figure 52: The Lockyer Creek catchment and the sampling locations.

Water samples were taken from the Gatton waste water treatment plant, which is upstream of the three weirs (Lockyer Creek) and three groundwater bores that were sampled. Atkinson Dam and Lake Dyer are used as irrigation reservoirs and accommodate recreational activities such as boating, camping and fishing. Lake Clarendon offers limited recreational use and is limited to day visitors on a small section of the south-western bank where the water samples were collected from. A pipeline connects Lake Clarendon to the Lockyer Creek for the purpose of supplementing irrigation water.

7.3 Field Sampling Procedure

7.3.1 Groundwater Collection

Groundwater samples were collected on the morning of 14 March 2012. Each of the three bores was purged three times (by known volume) before collecting sub-samples.

Photo 7: Water samples being taken at bore number 14320405.

7.3.2 Surface Water Collection

Surface water samples were collected on the morning of 21 March 2012. A scoop sampler was used to collect shallow surface waters (0.5 m composite) at least 2 m from the bank. This sampling gear was thoroughly rinsed between collection of each sample with deionised water, then with surface water from the respective sample site water.

7.3.3 Sample Glassware and Preservation

The sample containers used were 500 mL amber jars with Teflon-lined caps. All glassware was thoroughly cleaned, solvent-rinsed and baked at 400 °C prior to sampling. For stabilisation and preservation of the samples, 0.5 mL of 5M H_2SO_4 was added to the bottles prior to sampling.

7.3.4 Sampling Requirement

Three 500 mL samples (repetitions 1, 2 and 3) were collected from each sampling point. The glass sample bottles were filled to ensure minimal headspace.

7.3.5 Field Blanks

One field blank was prepared for each sample site. Before departing for the field, 500 mL of Milli-Q water (+ 0.5 mL 5 M H_2SO_4) was prepared and sealed. Each blank was placed next to the three sub-sample bottles with the lid removed for the entire sampling time at each site to assess the risk of cross-contamination or airborne particle interference on the entire sampling process.

7.3.6 Sample Transport

After being collected the samples were placed into cool boxes with ice bricks. The cool boxes were dispatched on the same afternoon the field samples were collected. An overnight courier was used to deliver the samples to the analytical laboratory within 24 hours.

7.4 Method for Analysis of Artificial Sweeteners, Pharmaceuticals, Perfluorinated Compounds and Pesticides using LC-MS/MS

7.4.1 Parameters included in the Screening

All samples were checked for 10 pesticides (desethylatrazine – DEA, simazine, atrazine, diuron, cyanazine, hexazinone, metribuzine, prometryn, metolachlor, terbutryn), 4 perfluorinated compounds (PFBA, PFHxS, PFOS, PFOA), 4 pharmaceuticals (carbamazepine, caffeine, diclofenac, crotamiton) 2 insect repellents (N,N-diethyl-m-toluamide – DEET, DCF) and 2 artificial sweeteners (acesulfame and saccharin).

7.4.2 Sample Preparation

Each 500 mL water sample was concentrated on to Bond Elute Plexa SPE cartridge, washed with 2 × 3 mL deionised water, then eluted with 3 mL of methanol, followed by 3 mL of 10% of ammonia in methanol. The elution was then evaporated to dryness under gentle nitrogen. The samples were reconstituted with 0.5 mL with 20% (methanol)/80% deionised water (Milli-Q water).

7.4.3 Liquid Chromatography

LC analysis was performed using a Thermo-FINNIGAN Surveyor Autosampler Plus and Thermo-FINNIGAN Surveyor MS Pump Plus. Restek Ultra IBD 3 µm column (100 mm × 2.1 mm, particle size 3.0 µm) was used. The column oven and the autosampler temperature were set at 30 °C and 10 °C, respectively. The mobile phase A was acetonitrile, B was a mixture of 0.1% formic acid and 10mM ammonium formate (see Table 12). The flow rate was at 0.25 ml/min. Ten µL of sample was injected onto the column.

Table 12: Mobile phase gradient.

Gradient	Mobile Phase	
Time (minute)	ACN	0.1% Formic Acid and 10mM Ammonium Formate
0	15	85
3	15	85
3,1	90	10
10	90	10
10,1	15	85
18	15	85

7.4.4 Mass Spectrometry

Mass spectrometry was performed using Thermo TSQ Quadrupole Mass Spectrometer (Thermo, USA). Electrospray ionisation (ESI) was used. Two MRM transitions (multiple reaction monitoring) were used for each analyte. One MRM transition for the labelled internal standard was monitored. Only the first transition was used for quantitation. The second MRM was used for confirmation. Relative retention times of the analyte were also monitored to ensure correct identification (Table 14). Mass spectrometry parameters are listed in Table 13. All transitions of analytes in ESI are presented in Table 14.

Table 13: Mass Spectrometry parameters.

Source	ESI
Ion polarity	Positive/negative
Spray voltage	4000(neg)/5000(pos)
Sheath/auxiliary gas	Nitrogen
Sheath gas pressure	40 arbitrary units
Auxiliary gas pressure	5 arbitrary units
Capillary temperature	350 °C
Scan type	SRM
Collision gas	Argon
Collision gas pressure (mTorr)	1.5

Table 14: Multiple reaction monitoring (MRM) transitions in electrospray ionisation mode.

Analytes	RT (min)	MRM transition (m/z)	Collision energy (V)	Tube Lens	Polarity
		293.12→261.17	12	54	
Caffeine	3.83	195.1→261.17	25	54	pos
		195.1→138.1	15	77	
DEA	7.58	188.05→104.05	25	77	pos
		188.05→146.05	25	77	
DEET	8.37	192.1→90.81	32	107	pos
		192.1→118.8	17	107	
Crotomition	8.53	204.14→134.02	21	105	pos
		204.14→136.05	23	105	
Carbamazepine	8.21	237.02→193	40	47	pos
		237.02→194	30	47	
Carbamazepine-d10	8.18	247→202	40	47	pos
Simazine	8.09	202.13→124.06	17	80	pos
		202.13→132.04	18	80	
Metribuzine	7.9	215.11→130.92	16	95	pos
		215.11→187.06	13	95	
Atrazine	8.31	216.2→104.15	28	72	pos
		216.2→174.3	16	72	
Diuron	8.69	233.06→46	15	79	pos
		233.06→72.08	17	79	
Cyanazine	7.99	241.10→104.04	27	93	pos
		241.10→214.22	13	93	
Prometryn	8.72	242.12→158.08	19	84	pos
		242.12→200.01	14	84	
Terbutryn	8.85	242.14→91.16	24	83	pos
		242.14→186.1	15	83	
Hexazinone	8.12	253.17→71.14	32	88	pos
		253.17→171.31	12	88	
Metolachlor	8.82	284.12→176.28	21	68	pos
		284.12→252.3	11	68	
Acesulfame	8.29	161.96→78.044	38	40	neg
		161.96→82.1	16	40	
Acesulfame-d4	8.26	166.07→78.08	40	40	neg
		182.04→106.02	21	65	

Analytes	RT (min)	MRM transition (m/z)	Collision energy (V)	Tube Lens	Polarity
PFBA	9.53	212.9→119	25	72	neg
		212.9→169	12	72	
PFBA-$^{13}C_4$	9.5	217.02→172.06	13	42	neg
Diclofenac	10.26	294.01→214	28	47	neg
		294.01→250	19	47	
Diclofenac-d4	10.22	298.1→217.02	28	47	neg
PFHxS	10.13	398.9→79.9	44	100	neg
		398.9→98.9	39	100	
PFHxS-18O2	10.13	403.1→102.92	40	47	neg
PFOA	11.49	412.9→169	20	100	neg
		412.9→369.1	13	100	
PFOA - ^{13}C8	11.49	375.99→172.06	43	84	neg
PFOS	11.02	498.8→80.17	56	105	neg
		498.8→98.73	35	105	
PFOS -^{13}C4	11.02	502.9→99	48	100	neg

7.5 Results

Irrespective of a potential future introduction of PRW, the results of the initial trace organics screening indicate an existing human footprint on the quality of the water resources of the Lockyer Valley, as it has been observed before also in other areas where waste water is entering the aquatic system. When interpreting the results of the present sampling campaign, the low number of samples and missing replication at the same sites need to be taken into account. The present screening is intended to provide early baseline information on TrOCs in the Lockyer Valley, thus enabling follow-up projects to investigate temporal patterns in TrOC distribution.

Nine TrOCs were not detected in any of the surface and groundwater samples taken and were also absent in the sample of the waste water treatment effluent. These are the pharmaceuticals diclofenac and crotomition, the perfluorochemical PFBA as well as the pesticides desethylatrazine, simazine, cyanazine, hexazinone, metribuzine and prometryn. Detailed results for all other substances are listed in Table 15.

Among the 10 pesticides screened (desethylatrazine, simazine, atrazine, diuron, cyanazine, hexazinone, metribuzine, prometryn, metolachlor and terbutryn), only atrazine and metolachlor were detected in surface waters of the Lockyer Valley (see Table 15 and Figure 53). No pesticides were found in the three groundwater monitoring sites. This is consistent with results from other pesticide screenings (Ellis and Bajracharya, 1999) in the Lockyer Valley.

Table 15: Substances with positive detects in the sampling campaign 2012.

Site Name	Water type	Acesulfame	Carbamazepine	N,N-diethyl-m-toluamide (DEET)	Caffeine	PFHxS	PFOS	PFOA	Atrazine	Diuron	Metolachlor	Terbutryn
		Art. Sweet.	Pharmaceuticals			PFCs			Pesticides			
Unit		ng/l	ng/l	ng/l	ng/l	ng/l	ng/l	ng/l	ng/l	ng/l	ng/l	ng/l
14320405	GW	366	<10	<5	<10	9	<10	<10	<5	<10	<5	<5
14320787	GW	41	4	<5	<10	<1	<1	<1	<5	<10	<5	<5
14320782	GW	15	3	<5	<10	<1	<1	<1	<5	<10	<5	<5
Atkinson Dam	SW	24	<1	21	78	<1	<1	<1	<5	<10	<5	<5
Glenore Grove Weir	SW	316	4	<5	<10	<1	<1	<1	7	<10	36	<5
Gatton Weir	SW	785	14	9	<10	<1	<1	<1	<5	<10	162	<5
Lake Clarendon	SW	163	3	<5	77	<1	<1	<1	10	<10	41	<5
Lake Dyer	SW	98	<1	25	58	<1	<1	<1	<5	<10	8	<5
O'Reilly's Weir	SW	46	<1	8	40	<1	<1	<1	37	<10	112	<5
Gatton WWTP	TWW	55421	1348	179	319	4	18	9	19	84	44	16
Detection limit		10	1	5	10	1	1	1	5	10	5	5

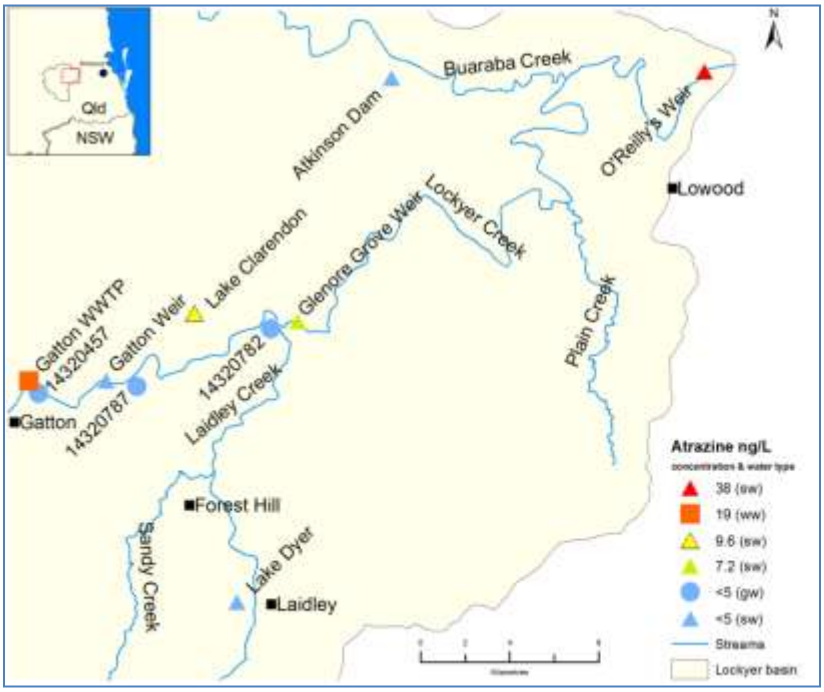


Figure 53: Distribution of atrazine in samples of the Lockyer Valley (ww = effluent from Gatton waste water treatment plant, gw = groundwater, sw = surface water).

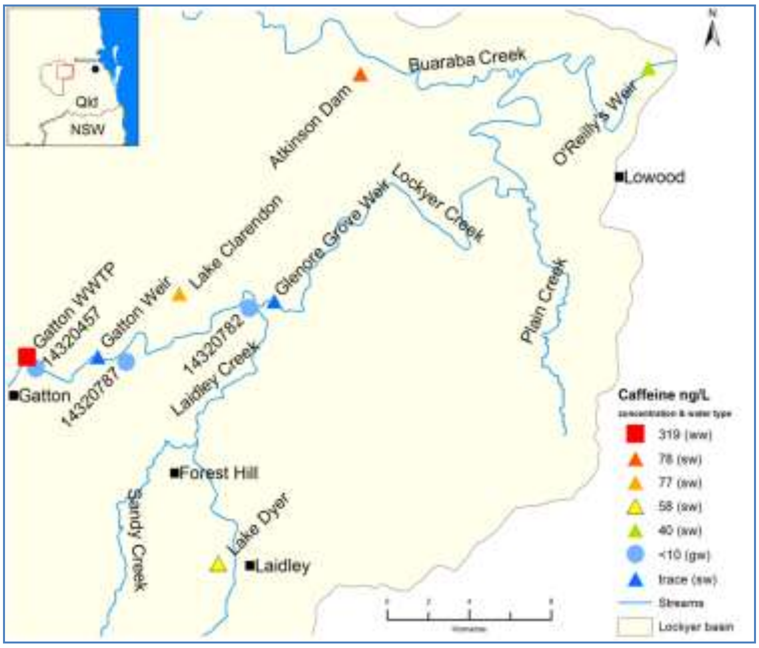


Figure 54: Distribution of caffeine in samples of the Lockyer Valley (ww = effluent from Gatton waste water treatment plant, gw = groundwater, sw = surface water).

Caffeine was found in all surface water reservoirs, as well as in the effluent of the Gatton waste water treatment plant (Figure 54). At the time of sampling, concentrations in the creek were below the detection limit, with the exception of O'Reilly's Weir at the downstream end of the Lockyer Valley.

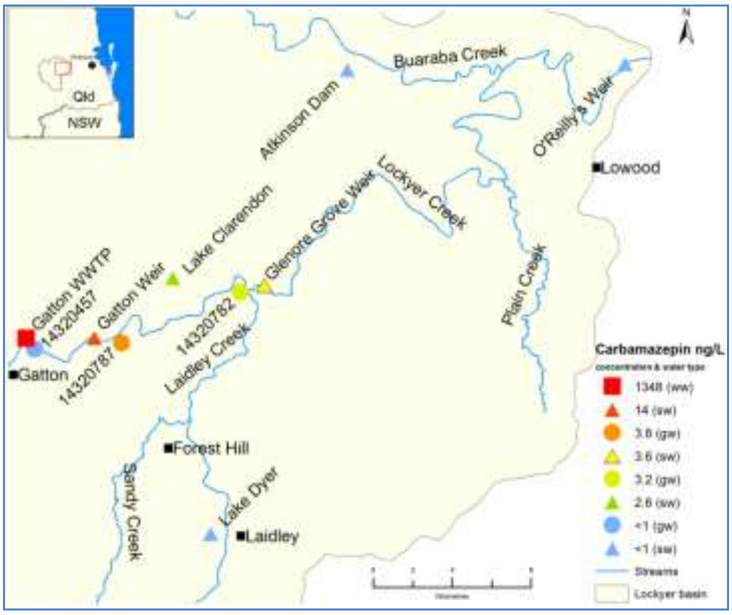


Figure 55: Distribution of carbamazepine in samples of the Lockyer Valley (ww = effluent from Gatton waste water treatment plant, gw = groundwater, sw = surface water).

Occurrence of the pharmaceutical carbamazepine, a well-known waste water tracer, seemed to be limited to the surroundings of Gatton as the main settlement in the Lockyer Valley (Figure 55). The insect repellent DEET, in contrast, is found also in the major surface water reservoirs Atkinson Dam and Lake Dyer (Figure 56).

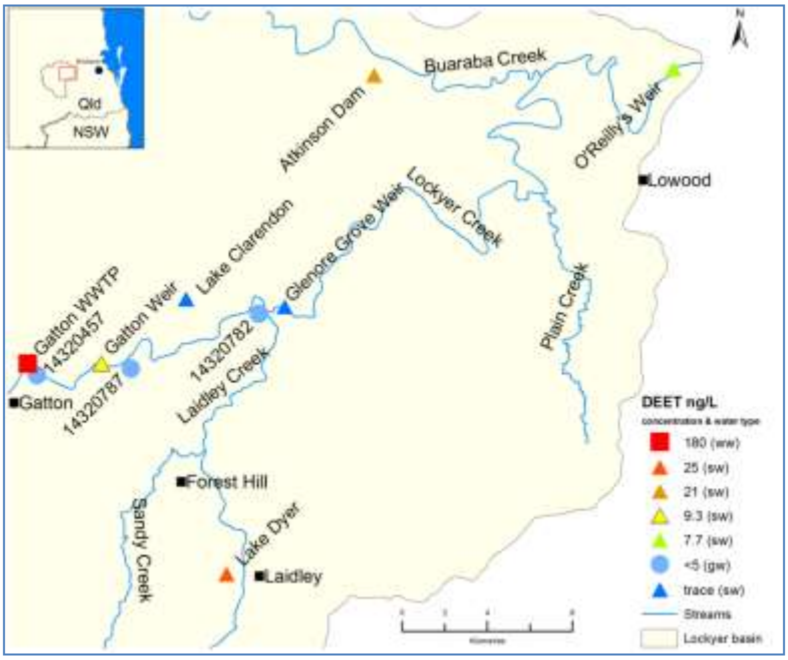


Figure 56: Distribution of DEET in samples of the Lockyer Valley (ww = effluent from Gatton waste water treatment plant, gw = groundwater, sw = surface water).

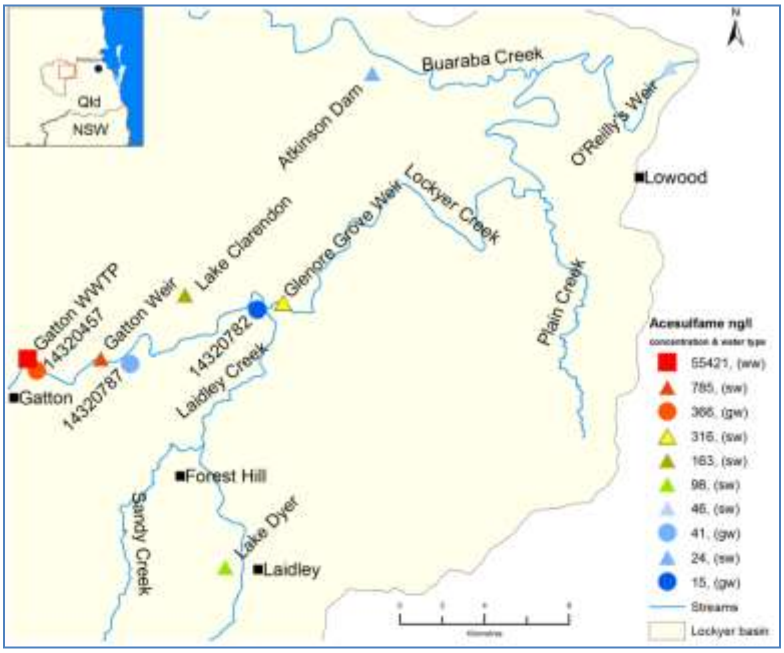


Figure 57: Distribution of the artificial sweetener acesulfame in samples of the Lockyer Valley (ww = effluent from Gatton waste water treatment plant, gw = groundwater, sw = surface water).

The artificial sweeteners acesulfame and saccharin were found at concentrations above the detection limit. Acesulfame was detected in all surface, groundwater and waste water samples in concentrations between 15 ng/l and 55421 ng/l. This demonstrates both the influence of waste water on the water resources of the Lockyer Valley as well as the excellent suitability of artificial sweeteners as anthropogenic tracers. Figure 57 suggests that the domestic waste water from Gatton is the input pathway for artificial sweeteners in the Lockyer Valley. The three groundwater monitoring bores show concentrations of 366 ng/l close to Gatton, 41 ng/l 4 km distance downstream and 15 ng/l approximately 8 km distance to the treatment plant. The reasons for this decline with distance is most likely not a result of attenuation or decay but a result of dilution and probably also different connectivity of the wells to the creek. The artificial sweetener saccharin (Figure 58) seems less popular than acesulfame in the Lockyer Valley, as indicated by the lower input signal of just 58 ng/l at the treatment plant. No detects above 10 ng/l could be found in the groundwater.

The findings in the Lockyer Valley correspond to the contemporary literature where artificial sweeteners are characterised by highest persistence in the water supply and were found frequently in other settings worldwide (Buerge *et al.*, 2009; Wolf *et al.*, 2012a).

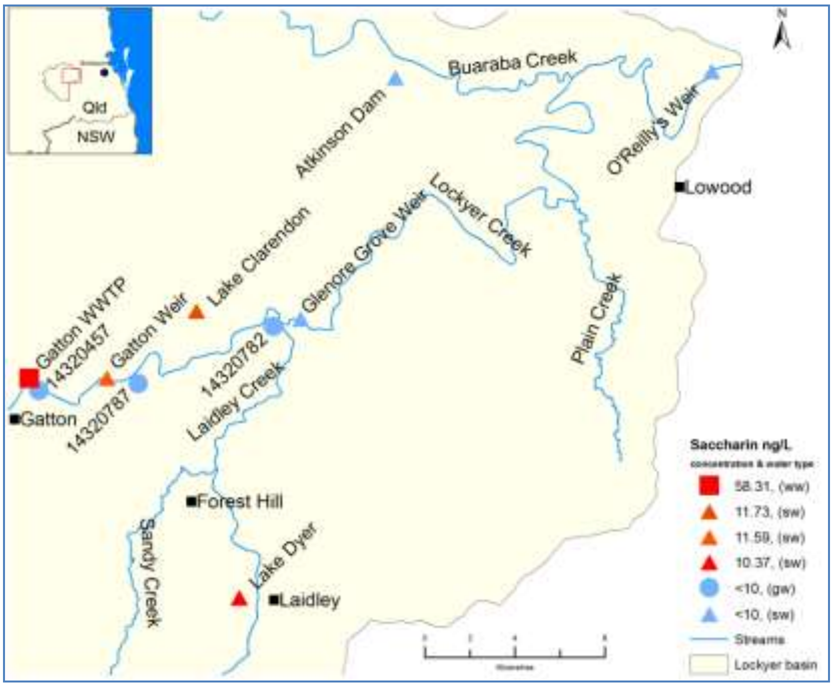


Figure 58: Distribution of the artificial sweetener saccharin in samples of the Lockyer Valley (ww = effluent from Gatton waste water treatment plant, gw = groundwater, sw = surface water).

7.6 Conclusions

The number of samples in this one-off initial screening exercise is too small to allow statistically relevant and solid conclusions to be drawn on mechanisms of TrOCs spreading in the Lockyer Valley. Still, the results published here are able to highlight the existing background concentrations of pharmaceuticals and TrOCs in water resources upstream of Brisbane. Among 2 artificial sweeteners, 5 pharmaceuticals, 4 perfluorinated compounds and 10 pesticides, only acesulfame, carbamazepine and perfluorohexane sulfonate (PFHxS) were detected in selected groundwater wells close to the Lockyer Creek and downstream of Gatton. Occurrence of TrOCs in surface waters was frequent, indicating a significant component of waste water already present in the Lockyer Valley water resources. Compared to other areas globally, concentrations of TrOCs in the Lockyer Valley are low to medium. A popular example for the sensitivity of the environment is the pharmaceutical diclofenac, which has been shown to impact kidney, liver and gills of selected fish species at concentrations as low as 500 ng/l (BMU-UBA, 2012). However, diclofenac was not detected even in the effluent of the Gatton treatment plant. On the other hand, the more conservative substance carbamazepine was detected in 2 out of 3 groundwater samples and 3 out of 6 surface water samples. The artificial sweetener acesulfame appears as an excellent tracer for waste water–surface water–groundwater interaction in the Lockyer Valley and could be used to constrain numerical models if screened on a more regular basis and in a larger set of monitoring wells. Still, the costs for such a measuring campaign need to be seen in relation to the actual uncertainty reduction which can be achieved in the model (Engelhardt *et al.*, 2011).

The potential risks associated with the introduction of PRW need to be seen against the existing background emissions. PRW, if produced along the set technical standards, does not contain any of the TrOCs in measurable quantities. Thus additional PRW supply to the Lockyer Valley would reduce the concentrations of TrOCs in the Lockyer Creek. In the current situation today, the drinking water supply of the downstream metropolis of Brisbane is not affected by an accidental risk associated with PRW but is already dealing with a continuous input of trace organics from upstream settlements. This situation is not unusual worldwide; many water supply systems in Europe (Massmann *et al.*, 2004) and the United States are characterised by existing trace organic loading in their source water before drinking water treatment.

8. MAPPING CROPS FROM SATELLITE IMAGERY

(Tim Ellis, Mick Hartcher, Manuel Grimm, Shreevatsa Kodur, Craig Henderson, Leif Wolf)

8.1 Summary

This study describes some methods used to estimate the proportions of crop types, crop rotation systems and to map crop type in time and space for the Lockyer Valley. This information was later used to inform water balance modelling and this, in turn, provided inputs to the groundwater flow model (see Chapters 11 and 12). This was not an exhaustive study but it provided some confidence that we were able to estimate crop areas which were otherwise unknown. Expert knowledge from district agronomists and crop scientists was used to approximate the relative proportions of irrigated crops and as a 'reality check' for the crop mapping component. Cloud-free Landsat 5 satellite imagery was collected for the study zone along with crop type/land use observations from ground surveys. The satellite imagery was then classified into crop types and crop type classes, depending on their seasonality, irrigation and rooting depth characteristics. The data set was composed from six ground surveys coordinated with images captured between August 2010 and July 2011 and the final broad categories comprised grazing, vegetables, cereals, lucerne and bare soil. The first classification of the satellite imagery was evaluated against the next image, for which ground-truthing (GT) data were collected. This produced 80% reliable prediction for the bare soil category (also comprising crop residue and seedlings); 65% for grazing; 53% for lucerne; 33% for winter vegetables; and 7% for winter cereals. Reliability improved as the data set increased with subsequent classifications, giving about 80–90% reliability for bare, grazed pasture and lucerne (*Medicago sativa*), but lower reliability for vegetables and cereals. We compared the results from the classification of satellite imagery with representative crop areas simulated using a small number of planting and land preparation guidelines developed from expert opinion. The seasonality of the crops (e.g. sweet corn, *Zea mays* convar. *Saccharata* var. *rugosa*, is a summer crop) made it difficult to perform thorough evaluation of the method on only six images spread over 12 months. Crop type maps were produced for the six contemporary images plus five 'historical' images: two from 'wet' periods in 1999 and three from drought conditions in 2006. These were evaluated using expert opinion from district agronomists. From the temporal distribution of crop types, three 'crop rotation systems' (i.e. 'wet', 'medium' and 'dry') were formulated and these were associated with specific areas within the study zone. For these rotation types, a water balance model (HowLeaky – see Chapter 9) was used to simulate irrigation demand and deep drainage using the assigned rotations. We found that this predicted much larger irrigation volumes compared to metered volumes within a defined zone of the Central Lockyer. We suspect this was due to the large differences between irrigation 'demand' and irrigation actually applied by growers. The model assumed no water restriction for irrigation and in turn, the model simulated adequate or 'theoretically probable' irrigation amount to meet the actual irrigation demand for the studied period (which received lower than long-term average rainfall). In addition, there were a large number of parameters in the water balance model which were not directly calibrated for the Lockyer Valley, although the model had been evaluated on similar soils. We then used trends in bare soil to moderate crop areas within drought periods and found better agreement with measured groundwater abstractions. The crop mapping methodology was successful, given the small number of images available to us, but it also showed that much more data would be required to suitably inform crop water balance models to confidently predict irrigation. In addition, we recommend that there be some feedback from groundwater supply to limit simulated crop areas, as would occur in reality.

8.2 Introduction

Crop irrigation with groundwater and deep drainage (which eventually becomes groundwater recharge) constitute the main vertical fluxes in and out of the aquifer, and quantifying them in time and space were critical inputs to the groundwater flow model. Both these fluxes are dependent on crop type and the irrigation methods used, both of which vary in time and space, due to crop rotation, soil type and different farm practices. Figure 59 shows how information on crop types was used as water balance model inputs to estimate irrigation that then became inputs to the unsaturated zone model (HYDRUS) and the groundwater flow model (MODFLOW). If the time lag between deep drainage (soil water travelling vertically below the crop root zone) and groundwater recharge (eventual addition of deep drainage to the groundwater) can either be neglected or quantified, in a different fashion, it is not obligatory to run the HYDRUS model. For the above reasons, and due to the dynamic nature of cropping systems in the irrigation area, we needed to map crop types in space and time, and to have some understanding of typical irrigation practices and how they might vary between rotation types. Farm-level knowledge of crop types and spatial and temporal distribution is held by farmers but they do not necessarily map it. In addition, it was not practical to interview farmers within the time available to the project and therefore we elected to adopt a remote sensing approach.

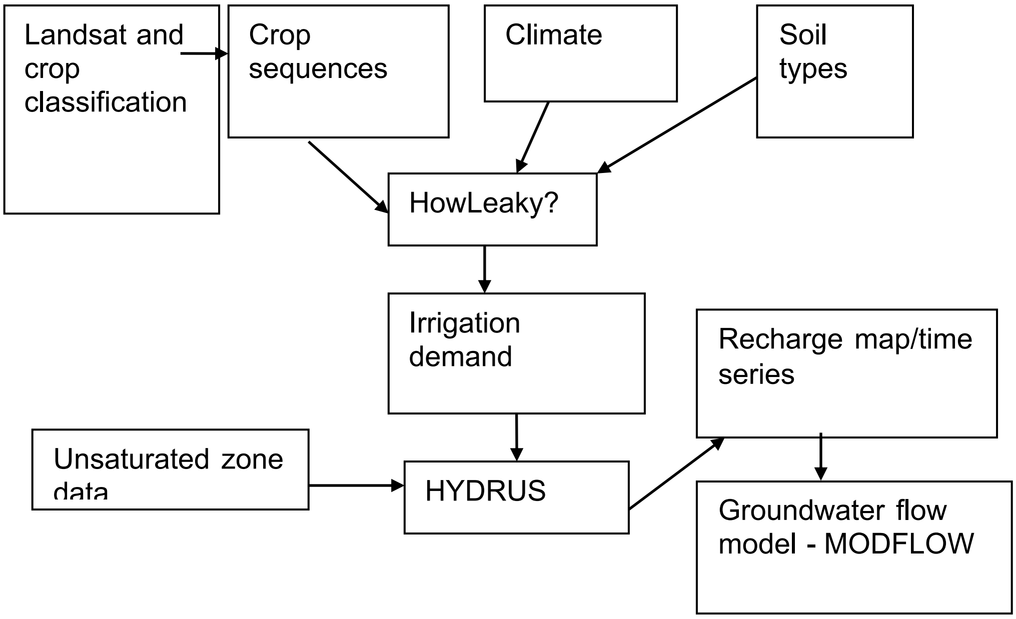


Figure 59: The hierarchy of information required to produce input to the groundwater flow model.

Satellite imagery can provide a broad spatial and temporal coverage of the landscape, which can be used to classify crop types. This information, used in conjunction with water balance modelling (see Chapter 9), can be used to approximate irrigation requirements and deep drainage fluxes (e.g. Castano *et al.*, 2010). Landsat Thematic Mapper (Landsat 5 TM supplied by NASA) imagery is purpose-built for mapping features on the earth's surface. It covers a wide area (swath width of 185 km) and has a repeat capture rate of once every 16 days. The Lockyer Valley region is situated within a single Landsat TM scene (see Figure 60), which means that no image mosaics were required. Apan and colleagues (2000) conducted a broad survey of land classes and structures within the Lockyer catchment using Landsat imagery but did not make distinction between individual crops, nor provide a time series. We decided that a Landsat crop classification approach could be applied and a pilot test GT exercise was implemented for an image capture in early September 2010. The results were encouraging and the method showed potential for developing current crop maps for 2010–2011 as well as some historical scenes from past significantly wet and dry years. Here we describe the methodology and the results from classifying a series of Landsat 5 images for the Lockyer Valley, which we used to estimate areas of crop types and crop rotation systems.

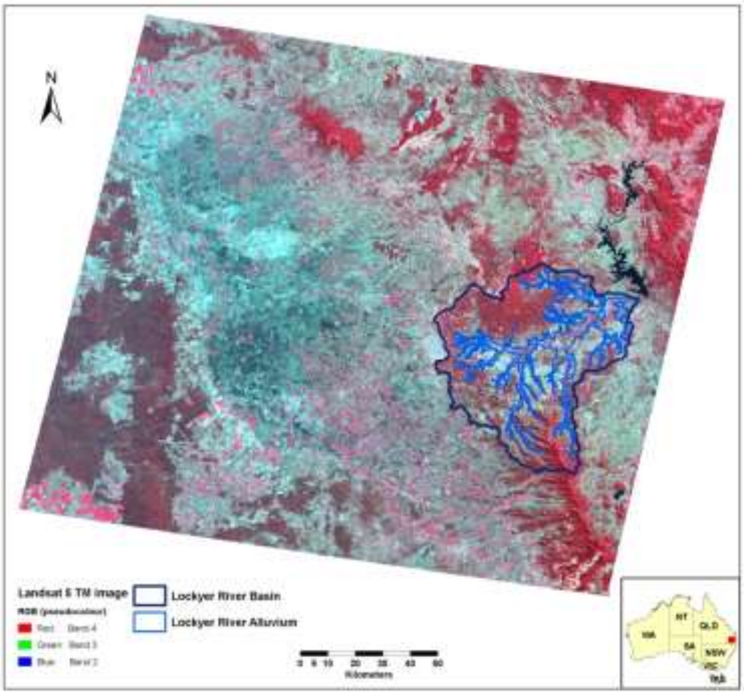


Figure 60: Location of Lockyer River Basin within Landsat TM scene.

8.3 Methodology

8.3.1 Estimation of Crop Areas from Expert Opinion

In the absence of quantitative information on the distribution of crops within the Lockyer Valley irrigation area, a meeting was held with district agronomist/crop scientists in June 2010, including authors of a 1997 survey (Harper *et al.*, 1997). A set of approximate 'guidelines' was devised to describe an 'average' rotation. These included:

- The main land uses were allocated as bare soil, lucerne (*M. sativa*), summer forage, summer vegetables, summer and winter grains and winter vegetables
- Total bare soil represented about 30% of the area over a 12-month period
- Lucerne (*M. sativa*) phases in a rotation lasted three years on average.

In an excel spreadsheet, 12 hypothetical 'fields' were represented at a monthly time step over a 12-year period, each including a lucerne phase and the other crops allocated within appropriate seasons. Growing periods for each crop in the simulation ranged between two months (summer vegetables), five months for grains and approximately three years for lucerne. At least one month for soil preparation (i.e. bare soil) was allocated between crops with at least two months for soil preparation following a lucerne phase. Initial crops for the time period were allocated randomly and the rotations developed by allocating crops using the above guidelines. This produced a 12 (fields) x 144 (month) matrix of land use, for which statistics were calculated for the average annual representative proportion of each, and the monthly representative proportion of each for an average year. These data were compared with representative proportions determined by the crop mapping from satellite imagery. Following on from this method, in an effort to simplify the application of the water balance modelling, 'rotation systems' were also specified using expert opinion and distributed across the irrigation area, informed by crop maps determined from the satellite imagery.

8.3.2 Satellite Image Spectral Signature Sampling

We used Landsat 5 TM imagery with atmospheric correction. Landsat images are available every 16 days but may be obscured partly or completely by clouds. On-ground crop identification, for both crop type and growth stage, was used as training data for supervised, parallelepiped, classification of images. There were 286 samples from five GT exercises and the classification algorithm was applied to five 'current' Landsat scenes from 2010–2011 and six 'historic' scenes in 1999 and 2006 (Table 16). The first stage of classification covered up to 41 different land use types, which were then reclassified into six major classes: Bare; Cereal/Cereal legume; Grazing/Forest; Lucerne/Forage; Vegetables; and Water bodies.

Table 16: Dates of Landsat images classified for crop areas. Field observations were gathered within days of each 'current' image except for 16 August 2010 and 13 April 2011. For the 'historical' images, no field observations were available.

Current	Historic
16 August 2010*	10 August 1999
2 September 2010	29 August 1999
3 October 2010	1 May 2006
13 April 2011*	20 July 2006
15 May 2011	24 October 2006
18 July 2011	-

*Classified images did not have accompanying field observations

A range of cover types were sampled from each image and used as training data in the classification. Each of the current images required crop sampling close to the acquisition time of each image, in order to calibrate signature sampling with the correct crop type and growth stage. On most occasions it was not possible to be in the field coincident with image capture and therefore in some cases a crop that appeared on an image had been altered by the time the GT sampling occurred (i.e. by harvest/tillage).

The crop type, growth stage, and any other crop characteristics (e.g. height, canopy structure, ground cover) that could support the classification were recorded in the field. In addition, oblique photographs were taken at ground level to help with the correct identification of signature samples and the bearing of each photograph was recorded. Two methods of locating GT samples were used: i) GPS location on the road adjacent to the observation; and ii) marking the field on a map made from the most recent satellite image. The latter was the most successful; however, it was time-consuming to identify the correct field and transfer these locations to the digital image, particularly due to the relatively coarse (30 m) spatial resolution of the imagery.

A major difficulty in the GT work was not having a current image to refer to when field data were collected. As the Lockyer Valley cropping area changes frequently, older images would not accurately depict where current crops were standing. After the GT data and the associated new image were acquired it was necessary to use a combination of data sources including GIS layers (roads, GT sites, recent imagery), Google Earth imagery (high spatial resolution but typically more than one year old), and GT photographs to assist with the correct location of signature sampling. Figure 61 shows the various data sources as well as the areas sampled (blue rectangular polygons) from a recent image.

Erdas IMAGINE (version 9.1; Leica Geosystems, Atlanta, Georgia, USA.) software was used to develop the spectral signature library, which was used as training data for the classification work. A signature sample was taken for at least a nine-pixel average (three x three pixels), but in most cases at least 20 pixels were sampled. Each sample was added to a signature file and named with the date, site number and crop type description. There were approximately 50 sites in the first GT exercise, which enabled 116 signature samples. Successive GT exercises provided more signatures to be added to the existing signature library. The final version of the library contained 286 signatures (see Table 17), which were used to classify all of the images.

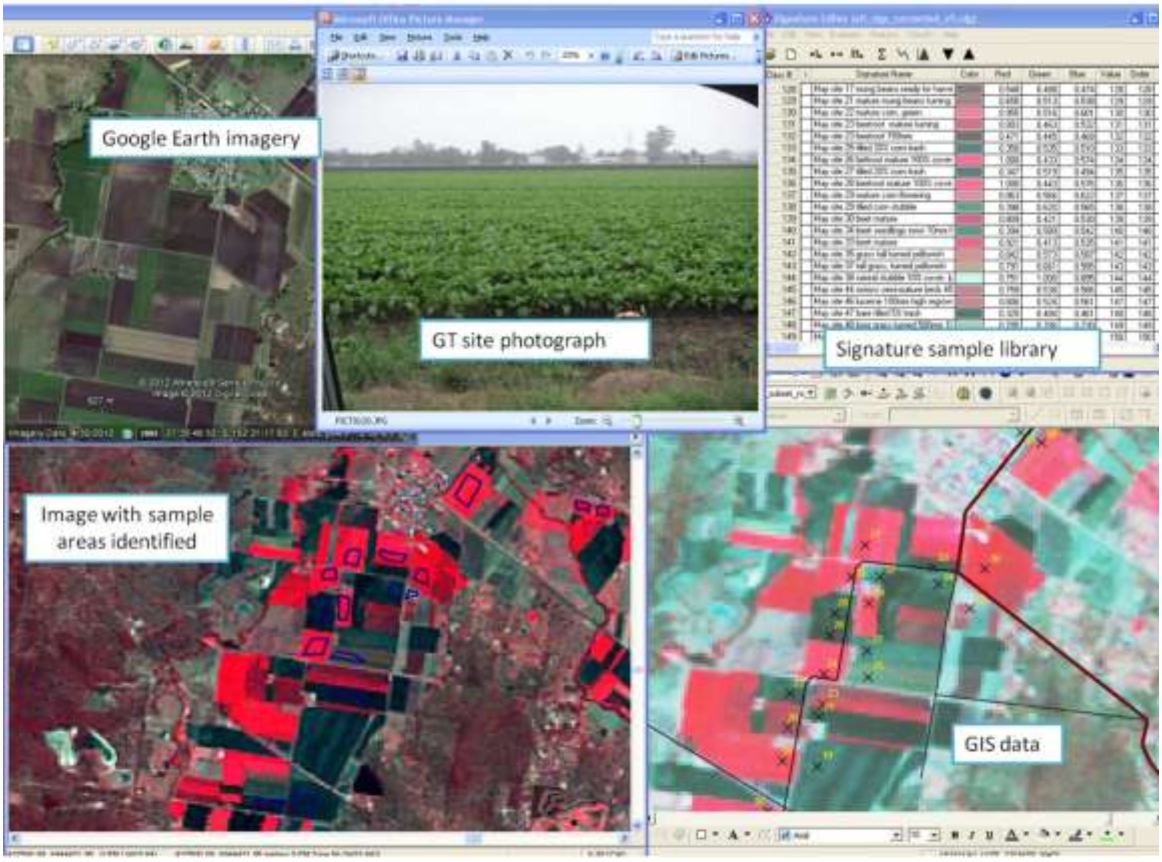


Figure 61: Data sources used to identify signature samples.

Table 17: Number of signatures for each major cover/crop type.

Cover* Type	Number of Signature Samples Used
Bare	82
Winter cereals	34
Grazing	30
Lucerne	25
Vegetables - all†	104
Other (clouds/forest/dams)	11
TOTAL	**286**

*Here, 'cover' refers to any type of ground surface
†All vegetable crops combined

8.3.3 Image Classification

Landsat Thematic Mapper (TM) images were acquired for five dates between September 2010 and July 2011 to characterise 'current' crop distributions and rotations. The classifications were run in iterations and results were checked to see for obvious errors, for example, a vegetable class in the middle of a field known to be cereal (from GT field excursions). The signatures causing the errors were then checked against field observations and in most cases were eliminated due to various factors, for example: a field being too small to sample, interference from different stages of the same crop or a poorly located sample.

The results from the first classification (September 2010) were assessed with regard to the level of homogeneity within crop fields and whether the correct crop type was achieved, based on comparison to known crop locations from GT work. The results were acceptable but also indicated that additional GT samples were required for various crop types, where few or no samples had been acquired (e.g. there were no GT samples of maize – *Zea mays*, or sweet corn – *Zea mays* convar. *Saccharata* var. *rugosa*, in September 2010). 'Reference' fields were sampled on each GT excursion and each of the classifications that followed provided feedback for any new crops or landscape positions that needed more information.

8.3.4 Reclassification

Each classified image comprised 286 crop/cover classes. The classified images were then reclassified using ArcGIS (version 9.3 Environmental Systems Research Institute, Redlands, California, USA) so that the like classes were grouped in output GRIDs. This resulted in 41 cover classes, which were then grouped again into a final set of seven major classes, including bare ground and water bodies. The major crop classes were chosen to represent the likely rooting depth and perenniality of the crop, that is, Cereals/Cereal legume; Grazing/Forest; Lucerne/Forage; and Vegetables. In hindsight, forage, because of its annual growth cycle, would probably be more appropriately grouped with cereals. However, the relatively small areas of forage observed meant any errors associated with the grouping we used would be small. Figure 62 shows the reclassified image and final grouping.

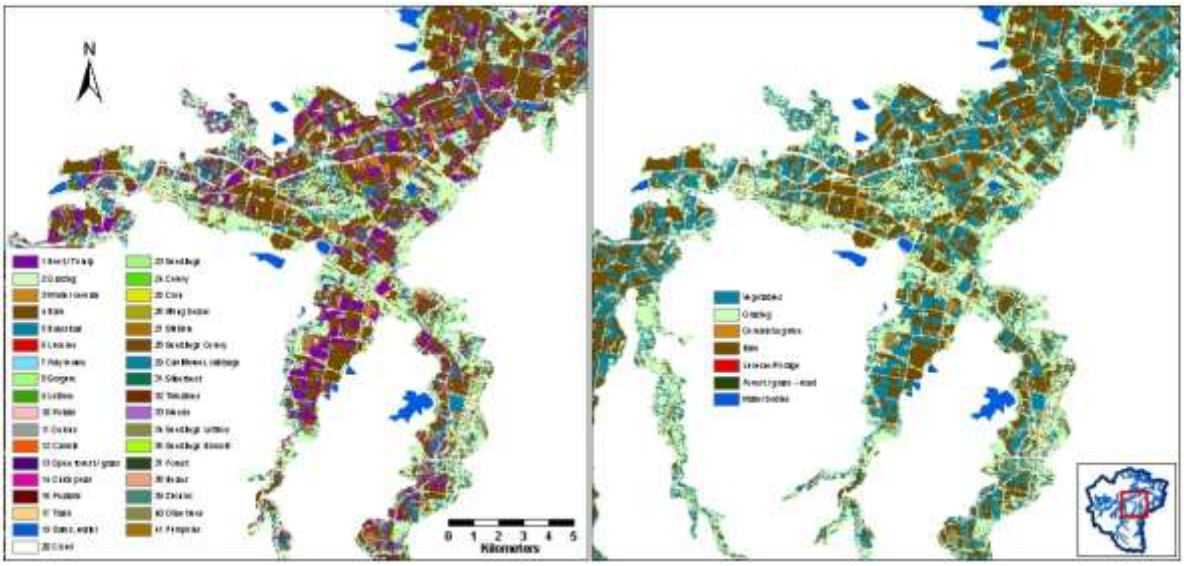


Figure 62: September 2010 image reclassification (41 classes) and final grouping (5 classes).

8.4 Major Errors and Analysis

The major errors included those resulting from: i) a mismatch between field sizes (or multiple crop stages in the same field – a common practice to spread the harvest period of vegetables) and the spatial resolution of the imagery; ii) multiple surface characteristics of bare ground (e.g. some with crop residue); poor vegetation signatures from young seedlings – that is, low crop cover; iii) weeds confounding vegetation signatures and causing spatial variation within fields; iv) cloud and cloud shadow in the satellite imagery; and v) the sample size (number of observations) possible for some crops due to:

- Timing of the satellite imagery (some short-duration crops could have come and gone between widely spaced GT excursions) the small area represented by that crop (e.g. pumpkins, *Curcubita spp.*)
- The low frequency (e.g. chick pea, *Cicer arietinum*, fields typically covered many hectares but were a relatively infrequent crop during the time of our observations)

- Unusual events such as the flood of January 2011, which removed or damaged large areas of crop and prevented planting of some summer crops like sweet corn (*Zea mays* convar. *Saccharata* var. *rugosa*).

The quantification of these errors was problematic, and we relied on our methods of image selection, signature verification and GT to minimise them. We determined the predictive power of the first image classification (2 September 2010) for the subsequent image from October 2010. This was done by giving a visual score to the accuracy of the class (what proportion of the field was correctly identified) for the classes relevant to the September image (Figure 63). This in itself was problematic, as some fields had changed their cover, their crop, or were tilled in the interim, and the crops themselves had changed colour with increased maturity. The significance of the predictive power of each class should be assessed in context with the representative proportions of each class (Figure 63), and the low predictive power of some classes (e.g. winter cereals) was mostly due to the small number of winter cereals in the October image; if they did exist, most cereal crops had turned colour by that stage. A single sorghum (*Sorghum bicolor*) crop in the September image was mature and uncharacteristic for that time of year – it appeared to have been standing in head for many months – and was still present in October. Bootstrapping methods were considered as an alternate method of error quantification but were not employed due to the small data set – see the small number of observations for some vegetable crops in Table 18. The basic methods we used gave a good indication of the source and magnitude of errors, and their ramifications for the production of an irrigation map. As indicated, other errors were likely to be larger, for example, the lack of supply limits in the water use model that did not represent the reduction in irrigated areas during the drought period (see later; Figure 76).

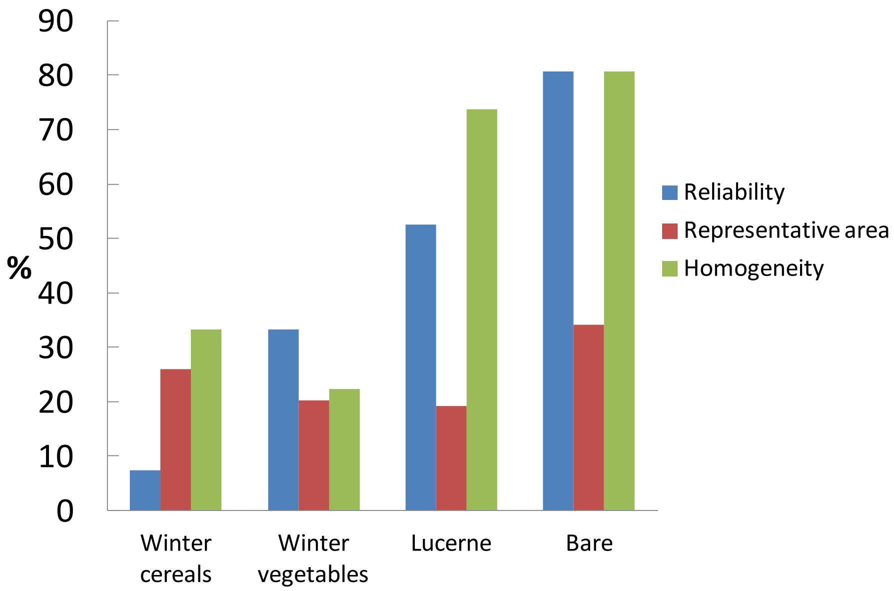


Figure 63: Scores of reliability and homogeneity of crop type predictions made from Landsat 5 imagery. The representative area of each provides an indication of the ramifications of uncertainty in each class. Estimates of winter vegetable reliability and homogeneity were conservative because they are an average of the multiple crops in that class – e.g. beetroot (*Beta vulgaris*) was 58% reliable and 90% homogeneous. Estimation of winter cereals remained problematic and this high uncertainty was not resolved.

8.4.1 Cell Resolution

The image resolution of 30 m x 30 m hindered the ability to differentiate long, thin strips of adjacent, yet different, crop types. In Figure 64 there are two stages of beans in the foreground and two stage of sorghum (*S. bicolor*) behind. Some crop types, such as lettuce (*Lactuca sativa*), have different varieties planted in long narrow strips, which cannot be differentiated within a pixel.

Figure 64: Different crops in close proximity: two stages of bean (*Phaseolus spp.*, foreground), sorghum (*S. bicolour*, middle) and senesced maize (*Zea mays*, far).

This was not a serious issue when the signatures of the different types are very similar; however, when there are significantly different varieties, such as the red and green lettuce shown in Figure 65, the signature can become ambiguous. Such areas cannot be used in signature sampling and may be classified incorrectly as well.

Figure 65: Different types of lettuce (*Lactuca sativa*) grown in narrow strips for salad leaf mix.

Fields of certain crops are planted over a time interval (e.g. four plantings, each two weeks apart) so that they can be harvested at intervals in order to have a controlled supply to the market. These are typically planted in long strips, which are less than a half or a quarter of an image pixel width, that is, 5–15 m strips. Figure 66 shows a number of stages and varieties of lettuce (*L. sativa*), which also have patches of weeds (Marshmallow weed – *Malva parviflora* – in the foreground). Each row in Figure 66 is only 1–1.5 m wide.

Figure 66: Different stages of lettuce (*Lacuta sativa*) in long, narrow strips, separated by strips of celery (*Apium graveolens* var. *dulce*, upper); Marshmallow weed left foreground (*Malva parviflora*).

Figure 67: Partially harvested cauliflower (*Brassica poleracea* L.) field. At the time of image capture, this field would have been homogenous.

In some cases, a field may be harvested partially (Figure 67), or completely, between the image capture date and the GT date, even if there is only a one-day delay. This can lead to problems in identifying areas for signature sampling.

8.4.2 Bare Ground

For the purposes of the water balance modelling, 'bare' fields were any that did not have an actively growing crop ('bare' for other studies typically means no [soil] cover). These included bare soil, crop residue and standing senesced crop. We also found the need to include seedlings in this category – see Seedling section – 8.4.3). Some 'bare' fields had a lot of weeds covering them, in some cases a homogenous cover of weeds. Trash also accompanied weeds on some bare fields (Figure 68). These conditions sometimes created false positives in the resulting classification where these fields were grouped in classes containing standing crops. For example, a very weedy field with annual rye grass could give a signal similar to cereals; and a bare field with crop residue could give a signal similar to recently harvested lucerne. Such situations, which became obvious as familiarity with the crops and the region increased, were resolved using combinations of photographic observations, local knowledge gained during the GT excursions and information from previous images and classifications which could shed light on the sequence crops or field operations.

Figure 68: Bare ground with weeds (10% cover) and cereal residue (30% cover).

8.4.3 Seedlings

The difficulty with early stage seedlings (Figure 69) is that there is so little vegetative cover and so much soil area that it is not possible to differentiate seedlings types or seedlings from weedy fields. The trade-off was to group the seedling fields in with the 'bare' class. This means that the 'bare' class does not simply represent the lack of a crop, or fallow, but also includes fields that have been planted and will have a standing crop within weeks. We assumed irrigation amounts at early crop stages were small and did not significantly affect deep drainage.

Figure 69: Young beetroot (*Beta vulgaris*) seedlings with < 5% cover.

8.4.4 Weeds

The presence of weed patches within and/or on the edges of fields has the potential to introduce errors in the classification results. In some cases weeds may infiltrate a crop field from neighbouring fields or easements (Figure 70) and can encroach on several rows of the crop. This will then change the reflectance for those rows, which results in them becoming false negatives, that is, appearing in a class other than the one in which they should appear. We note, however, that Figure 70 and Figure 71 show unusual amounts of weeds for typical horticultural enterprises in the Lockyer Valley.

Figure 70: Long strips with weeds – in this case, annual rye grass (*Lolium spp.*) and volunteer barley growing between rows of young broccoli (*Brassica oleracea* L.). This photo is used for example only, as the plot width in this case was too narrow to be properly represented by 30 x 30 m pixels.

Weeds may also appear within fields in patches, which results in a heterogeneous field when it should be one homogenous class. In Figure 71 a field of cabbages (*Brassica oleracea* L.) can be seen with large patches of weeds throughout, which creates a high degree of ambiguity in the resulting classification.

Figure 71: About 50% weed cover within about 50% cover from cabbage (*Brassica oleracae* L.).

8.4.5 Cloud and Shadow

Some of the satellite scenes had partial cloud cover over the area of interest (Figure 72). Although some of the clouds were classified correctly, some thin cloud and cloud shadow could not be distinguished and ended up in some of the crop classes – e.g. grazing and bare – which therefore added error to those classes and meant that whatever classes were actually below the cloud and shadow were not included in results. Due to the care observed when selecting images, the errors from clouds were infrequent and only affected a few percent of one image.

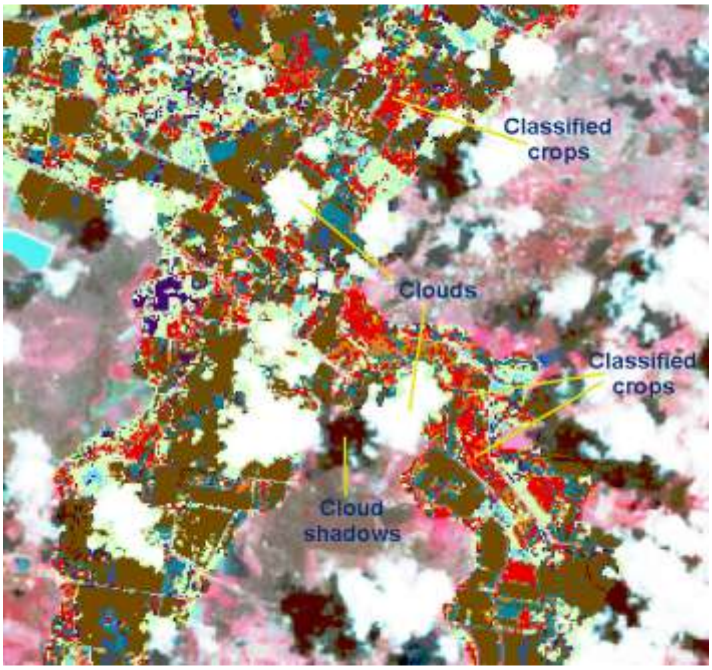


Figure 72: Clouds and cloud shadows obscuring actual crops.

8.4.6 Sample Size of Crop Types

Another potential cause of error was the limited number of samples for some classes, especially for vegetable crops. Table 18 shows the total number of samples for each vegetable class, which highlights the limitations in accurately classifying various classes, several of which had less than four signature samples.

Table 18: Number of samples for each vegetable crop class.

Vegetable Class	Number of Samples
Mung bean (*Vigna radiate*)	5
Bean (*Phaseolus spp.*)	9
Broccoli (*Brassica oleracea* L.)	16
Beetroot/turnip (*Beta vulgaris*)	21
Cabbage (*Brassica oleracea* L.)	2
Carrot (*Daucus carota*)	4
Cauliflower (*Brassica spp.*)	2
Celery (*Apium graveolens* var. *dulce*)	2
Chick pea (*Cicer arietinum*)	1
Corn (*Zea mays* convar. *Saccharata* var. *rugosa*) or Maize (*Zea mays*)	7
Lettuce (*Lacuta spp.*)	9
Onions/leek/garlic (*Allium spp.*)	10
Pumpkin (*Cucurbita spp.*)	1
Potato (*Solanum tuberosa*)	11
Silver beet (*Beta vulgaris* var. *cicla*)	1
Tomato (*Solanum lycopersicum*)	2
Zucchini (*Cucurbita pepo*)	1
TOTAL	**104**

8.5 Results

Figure 73 shows an excerpt of a simulation of land use over a 12-year period, on a monthly time step, for 12 hypothetical rotations assembled from a set of guidelines determined by expert opinion. This excerpt was used to estimate average annual representative proportional areas for forage (e.g. forage sorghum; *Sorghum spp.*), vegetables, cereal/cereal legume lucerne (*M. sativa*) and bare soil, which accounted for 11, 27, 14, 24 and 24% respectively. The % area of bare soil is similar to the long-term value of 30% estimated prior to the simulation from expert knowledge.

Average monthly representative proportional areas for the land use types (Figure 74, left) were calculated for comparison with four satellite imagery estimations made during 2010 and 2011 (Figure 74; right). Forage was included in the simulated areas but, being a summer crop, there were few field observations, given the timing of suitable satellite imagery; it was therefore omitted from the data (Figure 74, right).

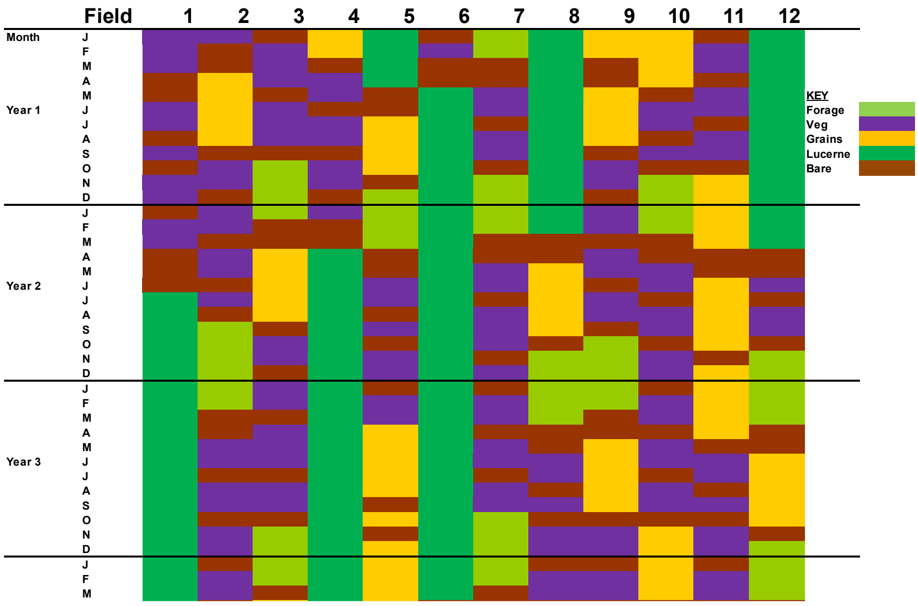


Figure 73: A three-year and three-month excerpt from a 12-year crop rotation calendar, for 12 hypothetical fields, assembled using the guidelines specified by experts.

When the two charts were compared (Figure 74), there was approximate agreement between simulated and observed representative proportional areas for the times of year the observations were made, and we used this as a very rough guide that observations were approximately correct.

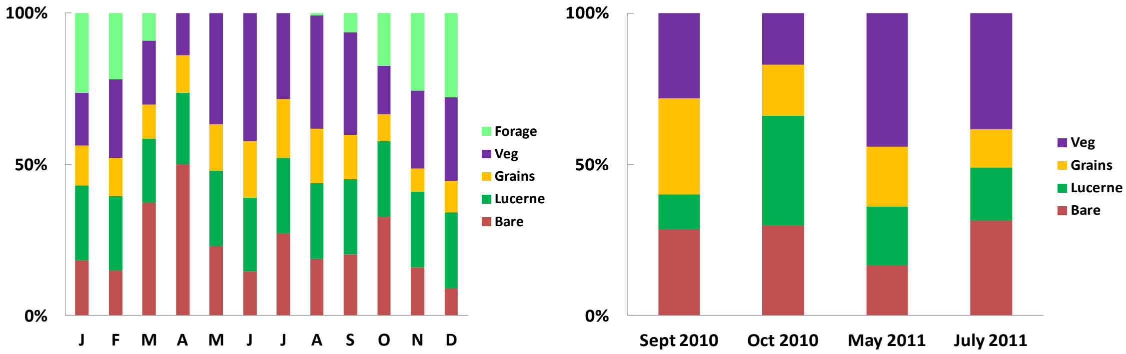


Figure 74: Representative irrigated land use types formulated from the expert panel meeting at DPI Gatton in 2010 (left; excluding lucerne) for a 12-month period. Note that this formulation includes forage, which had negligible (< 1%) representation in the surveys. The right hand chart shows the representative areas of land use types for the whole of the alluvium and (excluding grazing) for the four survey dates.

Two further cloud-free satellite images were obtained for August 2010 (just prior to field observations) and April 2011. The distributions of the major land uses on the Lockyer alluvium are shown in Figure 75 with three 2010 images in the left column and three 2011 images in the right. Severe floods occurred in the Lockyer Valley in January 2011 and this is partly responsible for the large proportion of bare soil in April 2011 as many fields were stripped and the planting of late summer crops was delayed or abandoned. The simulations (Figure 74, left) also suggested that the greatest proportion of bare soil would be observed around this time due to seasonal and crop rotation effects.

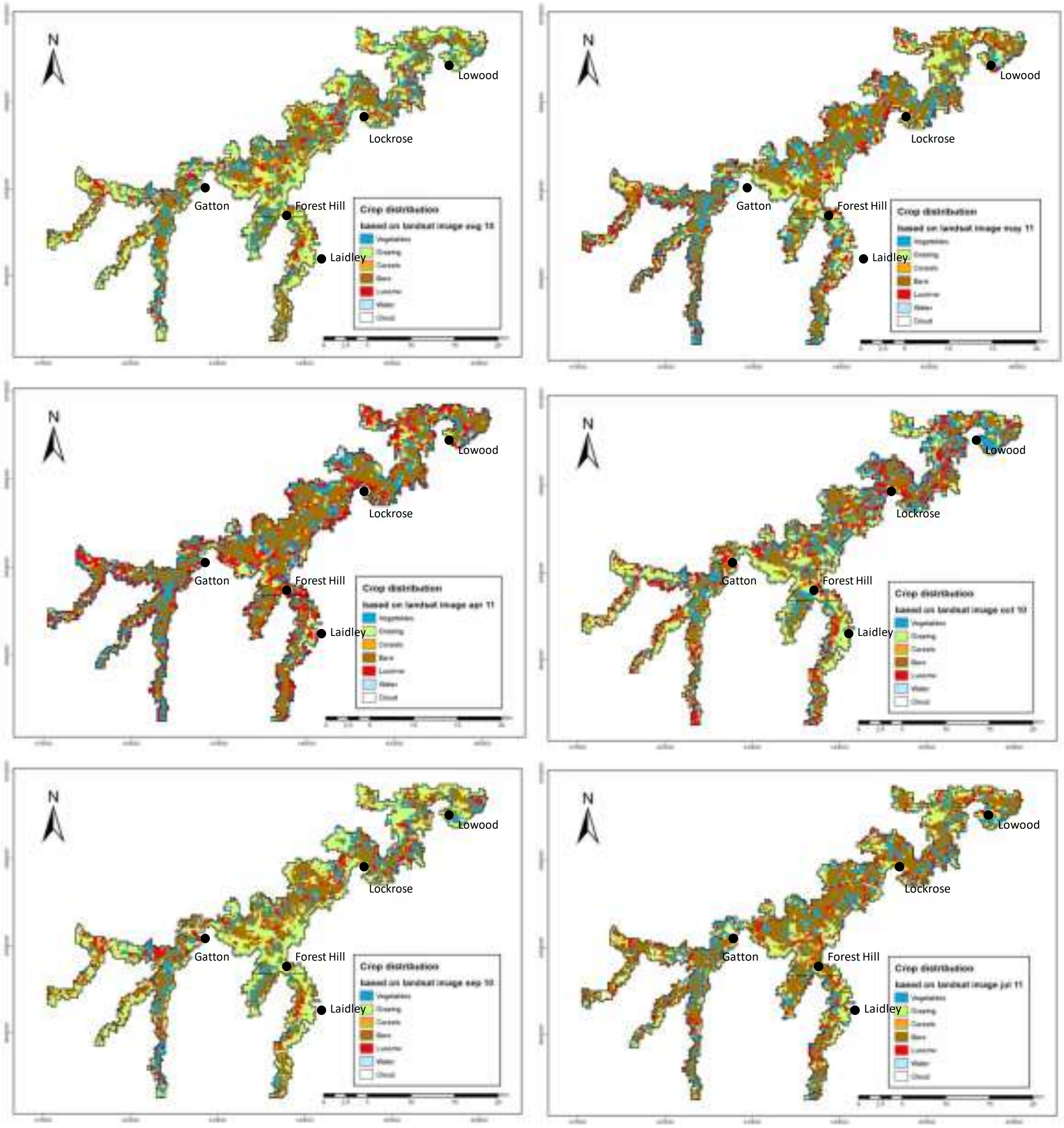


Figure 75: The six Landsat 5 images classified for the five crop types plus cloud and water bodies. The left column shows images for August, September and October 2010 and the right column from April, May and July 2011. The large representation of bare soil in Aril 2011 was in part due to the damage and disruption to normal cropping rotations from a severe flood in January 2011.

In an effort to represent some longer-term trends in land use in the irrigated areas, we applied the classification scheme developed from the four ground truthing surveys and the ‘current’ images (September and October 2010 and April and July 2011) to historical images chosen from past ‘wet’ (1999; two cloud-free images) and ‘dry’ (2006; three cloud-free images) periods. Figure 76 shows the trend in annual rainfall together with the trends in the major land uses that are also represented in Figure 73, Figure 74 and Figure 75. The most obvious trend is the increase in bare soil during the 'millennium drought' during the first decade of this century and the corresponding reduction in areas of crop.

The apparent large variation in the areas of lucerne (*M. sativa*) are likely caused by 'confusion' in the classification algorithm. When analysing the recent (2010 and 2011) images, lucerne was occasionally classified as carrots (*Daucus carota*) and grouped under vegetables. In addition, recently mown and baled lucerne resembled cereal stubble and was therefore included in the 'bare' category. We suggest that much larger data sets collected with greater frequency would be required to overcome these issues.

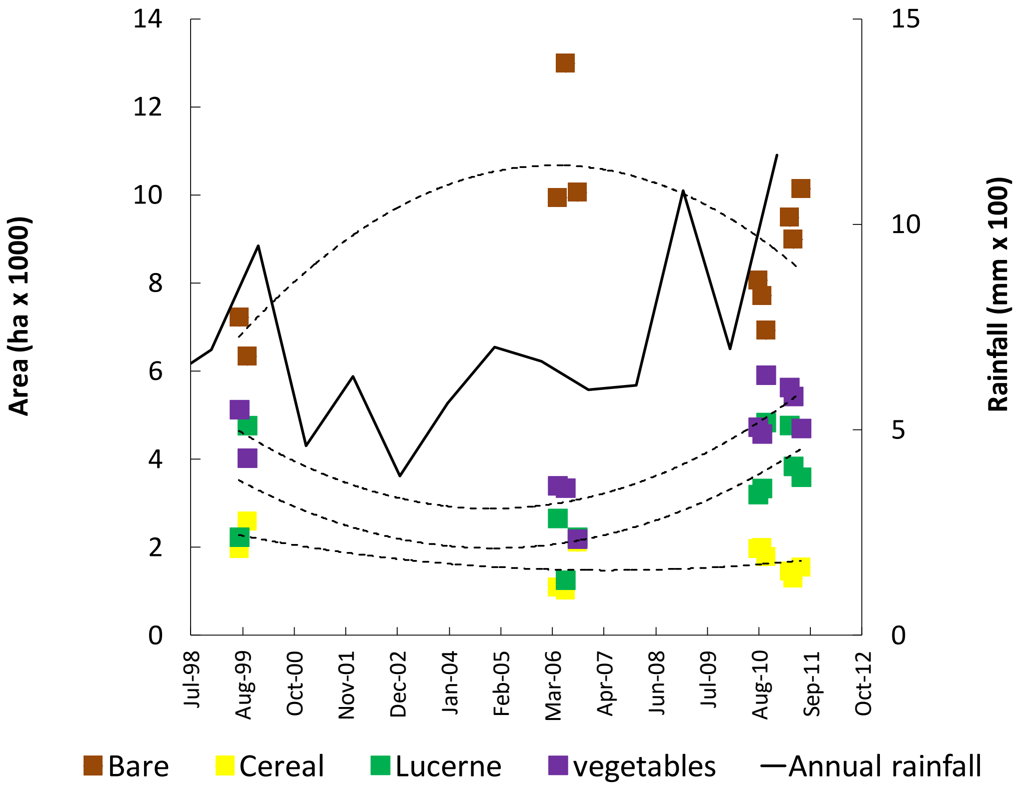


Figure 76: Changes in areas of the five major crop types, with time, composed from the six classified images from 2010 and 2011, as well as two and three historical images from 'wet' and 'dry' periods in 1999 and 2006, respectively.

To produce maps of irrigation requirement for the mapped crop types as shown in Figure 75 (for example) is a large task, given the spatial complexity, and it would be similarly difficult to infill crop rotations in time between the dates represented in Figure 76. We therefore chose to delineate areas within the irrigated area that could be represented by 'rotation systems', that is, that were characterised by recurring rotations of crop types and practices. Using expert knowledge, three rotation systems were chosen to represent 'wet', 'medium' and 'dry practices and these are specified in Table 19.

Table 19: Crop rotation systems representing 'wet', 'medium' and 'dry' practices, respectively. Irrigation triggers and target irrigation amounts are specified for the application of the water balance model. DUL = drained upper limit.

Crop Rotation System	Crops Represented and % Time of Crop and Bare Soil	Irrigation	
		Trigger, Based on Soil Water Deficit	Target Amount
'Dry' – Forage-grain-pasture – supplementary irrigation	Sorghum 12%, wheat 20%, lucerne 56%, bare 12%	Sorghum, wheat and lucerne = 140 mm	60 mm
'Medium' – Vegetable-forage-grain-grazed pasture – adequate irrigation	Sorghum 11%, broccoli 17%, lettuce 14%, wheat (*Triticum aestivum*) 14%, lucerne 21%, bare 23%	Forage, grain, lucerne = 100 mm, vegetables = 25–30 mm	DUL
'Wet' – Vegetables – adequate irrigation	Bean 17%, broccoli 16%, sweet corn 20%, lettuce 19%, bare 28%	Vegetables 25–50 mm	DUL

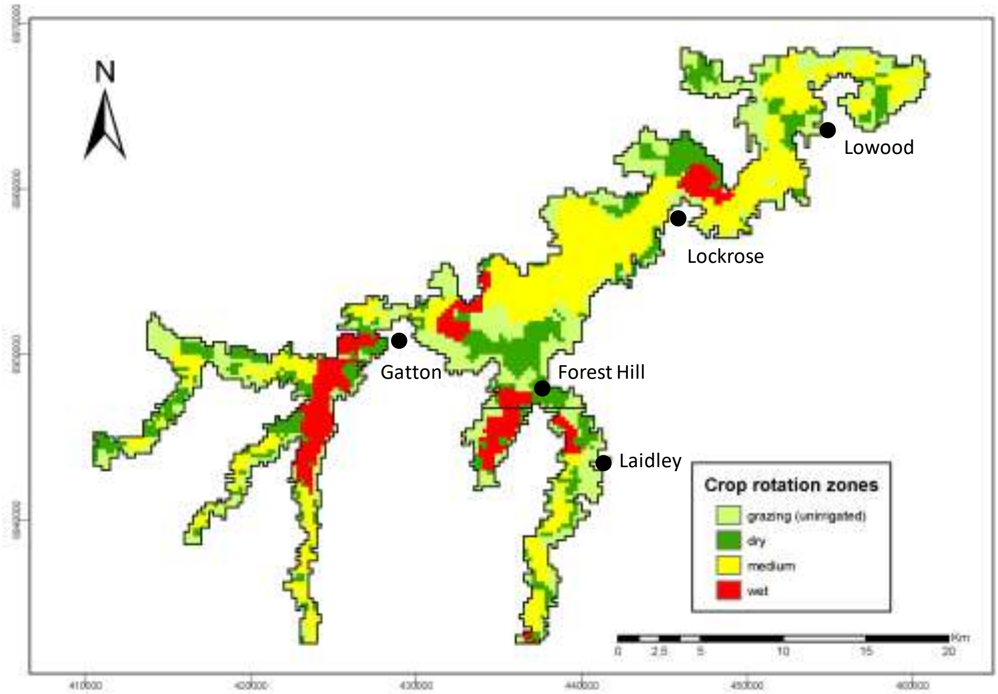


Figure 77: Crop rotation type zones distributed using a combination of expert knowledge and inspection of the six classified images from 2010 and 2011.

Spatial trends in these crop rotation systems became obvious when inspecting maps of crop types developed from the satellite imagery and these were delineated on a map of the irrigation area. This map was reviewed by two experts in the local irrigation industry and adjusted slightly on their advice (a small area was reclassified from 'medium' to 'dry'). The resulting final map in Figure 77 shows the allocated distribution of the three crop rotation systems used to advise the water balance modelling and to distribute the irrigation requirements. As shown in Chapter 11, this method over-predicted irrigation probably because there was representation of supply limits that would normally exist, rather than applying the total atmospheric demand of the crop. This over-prediction was addressed by adjusting crop areas as the amount of bare soil varied as shown in Figure 76 and is described in Chapter 11.

8.6 Discussion and Conclusions

The brief of this work was to provide a method to estimate irrigation amount and its spatial and temporal distribution in the Lockyer Valley as an input to a groundwater flow model. Some groundwater extraction data existed for the Central Lockyer Valley and are described in Chapters 11 and 12 and used for evaluating the methods we have described in this chapter. The method proposed was to use a water balance model to predict irrigation requirements for the various crop types. Up until this work, however, there was just one land use classification from 2002 (pers. comm. Ashley Bleakley, Principal Project Officer (Hydrology), Water Services South East) and expert opinion on rough proportions of crops types (Harper *et al.*, 1997) represented within the Lockyer Valley. We were able to find only one other study of landscape categories in the Lockyer Valley and, while cropped areas were quantified, there was no attempt to distinguish between crop types (Apan *et al.*, 2000). We developed a refined method using satellite imagery, and tested it against ground observations, for estimating the areas of individual crops and crop types, determined by their water requirements and likely rooting depths. Although this was a modestly resourced activity, it provided new data on current and historical temporal and spatial distributions of major land uses within the Lockyer Valley. Further, this allowed the identification of crop rotation systems and their spatial distribution, which simplified the application of the water balance model for predicting irrigation.

To undertake this work rigorously with a more precise estimate of uncertainty would have required much more data – the resources to undertake such a task can be estimated by reading Castano and colleagues (2010). However, the results have been satisfactory for the purposes of the greater study, and Chapter 12 demonstrates that the largest uncertainty in predicting groundwater flow processes was related to estimations of aquifer hydraulic properties. Our preliminary error analysis (Figure 63; evaluation of September 2010 classification using October 2010 ground observations) shows that substantial errors existed in the identification of individual crops. However, identification of bare soil and lucerne (*M. sativa*) was quite reliable and that the remaining crops represented in this preliminary evaluation (winter vegetables and winter cereals) could be estimated by difference. That is, together, they comprised the remaining proportions. This is not ideal, seeing that the irrigation requirements for vegetables and cereals are quite different (see Table 19).

The general trends and relationships in the crop mapping agreed with those from other indications. For example, Figure 74 shows approximate agreement between expert opinion and the relative proportions of crop areas determined using the classification of satellite imagery. Secondly, the temporal trends shown in Figure 75 and Figure 76 agree with those observed following the severe flood of January 2011 and during the 'millennium drought' from about 2000 to 2009. As described in the following chapters, significant other sources of uncertainty existed in the process of estimating applied irrigation – the greatest possibly being caused by the water balance model predicting irrigation 'demand' and not irrigation applied, which would be limited by water supply and was not represented in the model. In Chapter 11 it is shown how accounting for fluctuations in bare soil areas was able to reduce this uncertainty.

8.7 Potential Improvements

In addition to the above, a number of potential improvements could be applied to future remote sensing crop classification work in order to reduce potential errors.

More ground-truthing data: The ground-truthing work focused only on crop type and growth stage and did not take into account additional factors such as crop disease, weeds scattered within crops, soil type and soil moisture. Such additional data may prove to refine the classification and improve the accuracy of results.

Increased sample size: The number of samples for various crop types needs to be increased. Some statistical analysis should be conducted to determine adequate sample size. The ground-truthing work should then be conducted over more than one day so that enough samples can be collected.

Spectral analysis of ambiguous classes: Some crop classes were difficult to differentiate due to their similarity in both colour and structure. In particular, lucerne was sometimes difficult to differentiate from carrots and from some cereals. Some additional work focusing on the spectral nature of these classes may yield significant improvements in accuracy.

Bareness index: A fractional ground cover index has been developed by Dr Michael Schmidt at the Queensland Department of Environment and Resource Management (DERM). The index is based on Landsat TM imagery and provides images that have different layers which contain bare (bare ground rock, disturbed), green vegetation, non-green vegetation, residual errors, and the sum of the fractions of bare, green and non-green (Figure 78). This index could be used to refine the classification results, especially where seedlings have been grouped with the bare class. The index did not become available until after the crop classification work was completed.

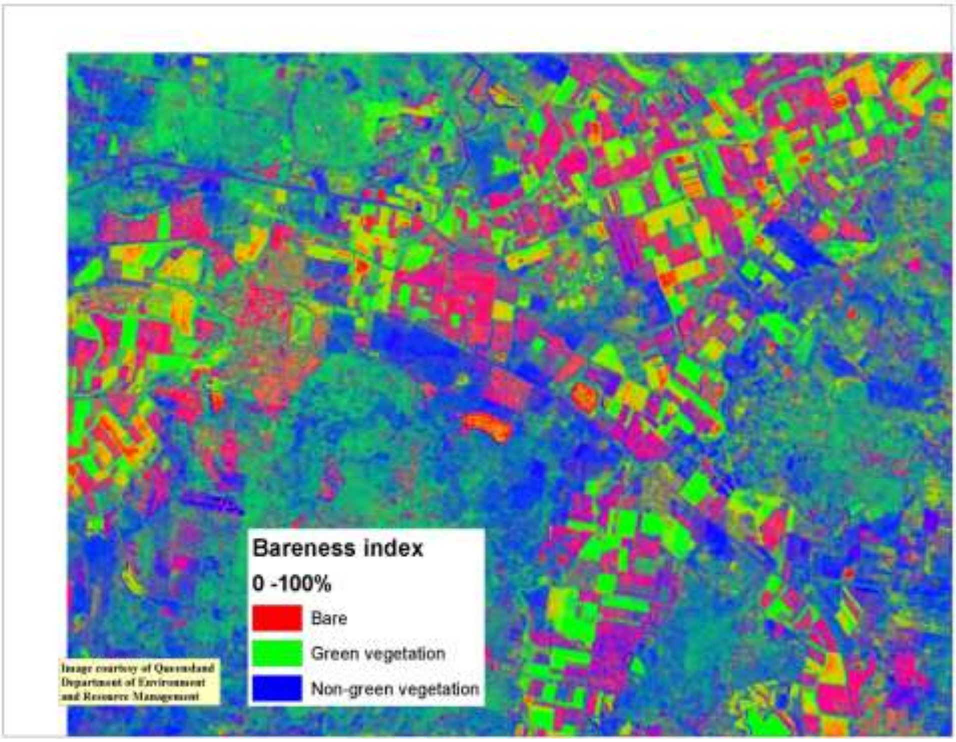


Figure 78: An image of the central Lockyer Valley irrigation area showing a bareness index which could be used to refine classifications that include bare soil – e.g. seedling stages.

8.7.1 GIS Modelling

Where there are data gaps (e.g. crops might go through most, or all, of their growth cycle between available cloud-free satellite images) a GIS model could be developed, with rules of logic to determine the most likely crop type to have been growing during in the intervening period. This would need approximate knowledge or simulation of crop growth stages.

9. ESTIMATION OF SOIL WATER BALANCE WITH EMPHASIS ON DEEP DRAINAGE FOR THE LOCKYER VALLEY

(Shreevatsa Kodur and Brett Robinson)

9.1 Introduction

The Lockyer Valley, in South East Queensland (SEQ), Australia, is an important area for high-value, year-round crop production, supplying approximately one-third of Queensland's irrigated vegetables (Cox and Wilson, 2005). However, intensive irrigated cropping and drought conditions prior to 2008 (Figure 79) had led to the depletion of groundwater supplies, affecting crop production. Recent wet spells (2008–2011) have eased the situation to some extent but the future of groundwater availability for continued crop production is not known. The use of purified recycled water (PRW) has been proposed for the Lockyer Valley (Wolf *et al.*, 2011). This may be a way of lessening pressure on the groundwater resource by acting as an adjunct to groundwater via surface supply or by recharging the aquifers. However, resulting changes to the soil water balance and subsequent effects on the underlying groundwater systems are not known.

Salinity is a major issue for the Lockyer Valley (White, 1980). Changes to the soil water balance, for example excessive deep drainage, can increase the salinity risks. Therefore, understanding soil water balance – deep drainage in particular – of various cropping and irrigation systems and their impacts on the underlying groundwater system is of prime importance.

Crop rotations in the Lockyer Valley are comprised of high-value vegetable, forage, grains and pasture. Vegetable cropping demands intensive management but is profitable. Grain and pasture generally require less intensive management and are also suitable for moisture-limited conditions. Deep drainage for irrigated cropping in the Lockyer Valley is known to be higher than dryland cropping (Thorburn *et al.*, 1990). However, there is limited information, including from field studies, on soil water balance for different cropping sequences (particularly under irrigated conditions) and factors that affect soil water balance in the Lockyer Valley.

To address this knowledge gap, soil water balance scenarios were developed for a range of cropping sequences that differ with respect to crop types, crop characteristics and irrigation strategy. Simulated cropping sequences may lead to varying rates of irrigation application due to differences in the water demand. This, in turn, will lead to varied rates of deep drainage.

Simulation models are useful tools for studying both the soil water balance and the factors that affect soil water balance for the Lockyer Valley. Modelling was chosen as an ideal method of analysis because of the: i) large number of variables involved; ii) need for assessing hypothetical systems; iii) long-term crop rotations comprised of various cropping sequences and crop types; iv) relative cost effectiveness compared with direct methods, such as lysimetry; and v) usefulness where limited field data are available, as was the case for the Lockyer Valley.

Given the importance of soil water balance information for the Lockyer Valley and the usefulness of the modelling approach, the aim of the current study was to estimate the soil water balance and the factors that affected the soil water balance under representative cropping sequences and irrigation strategies using soil water balance modelling. This knowledge in turn will contribute to i) understanding of the trends in soil water balance under various growing conditions, ii) development of deep drainage maps, iii) estimations of groundwater recharge and in turn, assessments of possible impacts of soil water balance on the underlying groundwater system, and iv) development of management strategies that are water-use efficient and economical, leading to sustainable land and water management.

9.2 Soil Water Balance for Cropping Sequences under 14 Years of Rotation Cropping

9.2.1 Methodology

9.2.1.1 Simulation

The HowLeaky model (Rattray *et al.*, 2004) was used to estimate soil water balance. HowLeaky simulates crop growth and the vertical components of the soil water balance on a daily basis. HowLeaky can also simulate cropping sequences and irrigation management. This model was selected as an appropriate tool for the current study because it: i) has been validated for similar growing environments, including soil, climate and vegetation; ii) is appropriate for long-term rotation analysis; and iii) has a wide range of options to modify various inputs, especially vegetation parameters and irrigation, and iv) has been successfully used to study soil water balance in both dryland and irrigated agriculture (e.g. Robinson *et al.*, 2007; Yee Yet and Silburn, 2003). The long-term water balance comprises rainfall and irrigation which lead to infiltration of water to the soil, runoff, soil evaporation, transpiration and deep drainage in the short-term; there may also be changes to the amount of water stored in the soil. Soil water balance was studied for 14 years from January 1997 to December 2010. This period was chosen to exploit the existing data on cropping sequence and crop management.

The study comprised simulations of cropping sequences, namely: i) forage-grain-pasture; ii) vegetable-forage-grain-pasture; and iii) vegetable. Forage-grain-pasture was deficit irrigated, whereas vegetable-forage-grain-pasture and vegetable sequences were ideally (adequately) irrigated (see Section 9.2.2 for detail). These cropping sequences respectively represented low, semi-intensive and intensive cropping conditions, and were devised based on expert opinion, communication with local farmers and available literature regarding the farming in the Lockyer Valley (e.g. Henderson, 2003). To understand the effect of cropping sequence alone on soil water balance, that is, to separate the effect of irrigation strategy which differed among sequences, we also simulated forage-grain-pasture sequence for an ideal irrigation strategy similar to the other two sequences, although this practice seemed to be less common in the Lockyer Valley.

9.2.1.2 Climate and Soil

The climate data were for the Department of Primary Industries, Gatton Research Station, Queensland (152°19' E, 27°32' S) and were obtained from the SILO Patched Point dataset (Jeffrey *et al.*, 2001). To establish the rainfall conditions of the study period in relation to the long-term (1900–2011) mean rainfall, we studied i) cumulative deviation in the rainfall from the long-term mean, ii) annual rainfall for the long-term, which includes the study period, and iii) the trends in the long-term mean rainfall, which are identified based on the maximum and minimum cumulative deviation from the long-term mean. The study period had (average) annual rainfall of 701 mm, annual pan evaporation of 1986 mm; daily radiation of 18.8 MJ/m^2 and daily maximum and minimum temperatures of 27.1 °C and 13.3 °C respectively.

Soil data were for a Black Vertosol (Isbell, 1996), which is one of the predominant soils for cropping in the Lockyer Valley (Powell *et al.*, 2002). Runoff was calculated using the modified USDA runoff method (Littleboy *et al.*, 1996). Curve number, which describes runoff potential for bare soil, was set according to Robinson and colleagues (2010) and data from Littleboy and colleagues (1996). Soil evaporation was simulated using the procedure described by Ritchie (1972). The values for stage 1 evaporation (U) and stage 2 evaporation (Cona) were derived from nearby field experiments on Black Vertosol at Greenmount (151°55' E, 27°46' S) (Silburn and Freebairn, 1992) and Kingsthorpe (151°47' E, 27°30' S), and were adjusted to the Lockyer Valley conditions based on the sensitivity analysis (data not shown). Soil physical properties were set for each soil layer up to 1.8 m depth (Table 20). The general procedure for HowLeaky parameterisation has been described in detail by Robinson and colleagues (2010) and Melland and colleagues (2010).

Table 20: Soil parameters for a Black Vertosol of the Lockyer Valley.

Detail	Unit	Value (for selected depth)			
Layer Depth	**m**	**0.2**	**0.3**	**1.2**	**1.8**
Air dry water content	% (v/v)	13	20	20	20
Saturated water content	% (v/v)	46	48	45	43
Field capacity water content	% (v/v)	41	43	40	38
Wilting point water content	% (v/v)	28	28	31	34
Bulk density	g/cm³	1.3	1.3	1.4	1.4
		Value (for all depths)			
Evaporation stage 1 (U)	mm	6.5			
Evaporation stage 2 (Cona)	mm/day0.5	3.8			
Runoff curve number for bare soil		73			
Curve number reduction at 100% cover		18			

9.2.1.3 Vegetation

Vegetation parameters for each crop were developed using either the 'crop cover' or 'dynamic' module within HowLeaky. The crop cover module has inputs of green and total cover (%), residue cover (%) and root depth (mm) for various times of the year. The dynamic module uses detailed management and biophysical parameters concerning phenology, growth and other plant processes. Vegetation parameters were set according to local knowledge of the average growth conditions and other cropping conditions (e.g. tillage patterns) of the region. Root growth was assumed to occur linearly at a rate of 15 mm/day to a maximum depth and was reset to zero at harvest for annual crops. For perennial crops such as lucerne, root depth was kept at maximum once crops were established. In both the crop cover and dynamic modules, fallow conditions (of varied duration, either between two crops or as a main fallow) were simulated as bare soil. Forage-grain-pasture sequence was represented respectively by forage sorghum (summer planting), wheat (winter) and lucerne (autumn or winter); whereas vegetable crops were represented by broccoli and lettuce (year-round planting under vegetable-forage-grain-pasture and autumn or winter under vegetable sequences), bean (spring) and sweet corn (summer). Approximate total cropping duration (depending on crop and climate) was 2–3 months for broccoli, lettuce and beans, 3 months for sweet corn, 4–5 months for sorghum, 6 months for wheat and 2–3 years for lucerne.

9.2.1.4 Irrigation

Irrigation triggers (time to start irrigation) for each crop in each cropping sequence were based on the critical soil water deficit (SWD) below drained upper limit (DUL). The SWD to trigger irrigation for each crop was set according to local knowledge on irrigation practice and based on available literature for management practices for the Lockyer Valley (e.g. Henderson, 2006).

Irrigation target amount for crops was either DUL or a deficit depth (not refilling the SWD to DUL). Deficit depth irrigation was used to mimic the farmer's practice of supplementary irrigation or irrigation at a reduced rate with some crops (e.g. sorghum). A target amount of DUL was simulated to represent the ideal (adequate) irrigation practice for the Lockyer Valley. That is, this practice fulfilled the SWD without causing deep drainage due to irrigation. Any deep drainage that occurred was then due to rainfall after an irrigation that exceeded the soil water-holding capacity. The crops under the forage-grain-pasture sequence were each irrigated to a fixed depth of 60 mm when the SWD reached 140 mm. All the crops in the vegetable-forage-grain-pasture and vegetable sequences were irrigated to DUL when the SWD reached 25 mm (e.g. lettuce) to 100 mm (e.g. sorghum) depending on the crop or crop characteristics.

Soil water content at the beginning of each simulation for each crop was set based on actual soil water content at the end of the previous crop or fallow (derived from HowLeaky's SWD output option) and based on local knowledge of the prevailing soil and cropping conditions.

9.2.1.5 Validation and Data Interpretation

The HowLeaky model or its predecessor PERFECT model (Littleboy *et al.*, 1989, 1992) was validated across various soils and climates of Queensland (e.g. Freebairn *et al.*, 1991; Littleboy *et al.*, 1992, 1996; Owens *et al.*, 2004; Robinson *et al.*, 2010; Silburn and Freebairn, 1992; Thornton *et al.*, 2007). This model combines the modified USDA runoff model and the linear cascading drainage technique originally from CREAMS (Knisel, 1980) and the Ritchie (1972) evapotranspiration model. This kind of water balance model and its derivatives have also been widely validated throughout the world.

In addition to using an already validated model, a potential validation of the current deep drainage estimates is through the comparison with SODICS (Rose *et al.*, 1979) results for the Lockyer Valley. SODICS is a computer program (an Excel spreadsheet with SOLVER function) that uses a transient solute mass balance equation (Thorburn *et al.*, 1987) to estimate the drainage rate from paired soil Cl data. As there were limited data on soil water balance for the cropping sequences studied here, we also simulated soil water balance for a continuous fallow and a wheat-fallow cropping for the Lockyer Valley for the same study period and compared the results with the similar data from literature.

9.2.2 Results: Effect of Cropping Sequences on Soil Water Balance

The long-term rainfall analysis for the Lockyer Valley is presented in Figure 79a,b. The cumulative deviation in monthly rainfall from the long-term (1900–2011) mean decreased from 1902 to reach a maximum negative deviation of ~2000 mm/month during the 1940s to 1950s. A continued wetter climate led to an increase in the deviation until the early 1990s (>700 mm). Cumulative deviation decreased again from 1993 till 2007 but the converse is true 2007 onwards.

As shown in Figure 79b, periods 1942–1966 and 1967–1992 respectively had 37 mm and 97 mm/a higher than long-term (1900–2011) mean rainfall of 775 mm/a, with 2008–2011 being the wettest rainfall period (243 mm/a above long-term mean). However, periods 1901–1913 and 1914–1941 had respectively 16 mm and 75 mm/a lower than long-term mean rainfall, with 1993–2007 being the lowest rainfall period (135 mm/a below long-term mean).

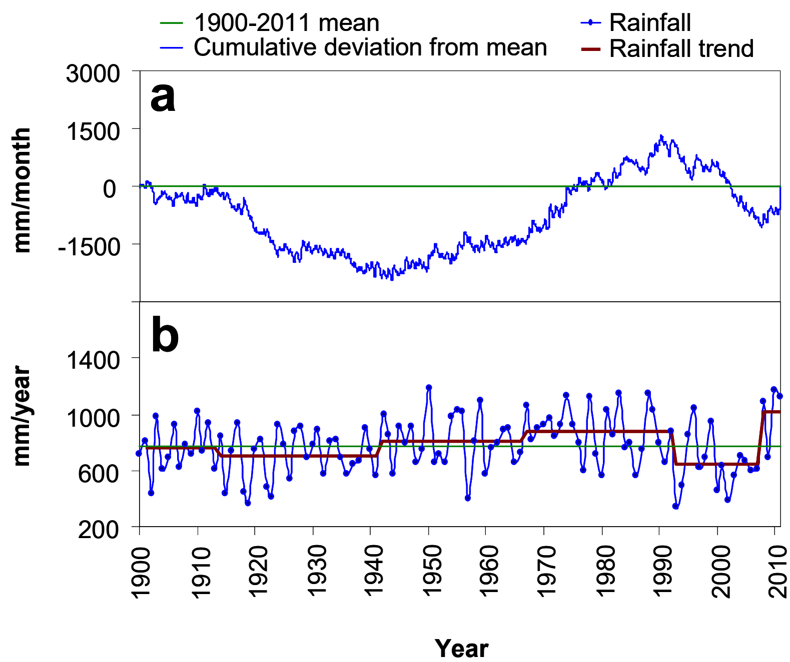


Figure 79: a) Cumulative deviation in the monthly rainfall from long-term (1900–2011) mean, and b) annual rainfall, and rainfall trend based on cumulative deviation from mean for the Lockyer Valley.

Results show 1.8-fold differences in irrigation and 11-fold differences in deep drainage among the cropping sequences (Table 21). Forage-grain-pasture sequence had the lowest irrigation, deep drainage, runoff and soil evaporation and highest transpiration compared with the vegetable-forage-grain-pasture and vegetable sequences (Table 21).

Table 21: Soil water balance components (rainfall, P; irrigation, I; deep drainage, DD; runoff, Q; soil evaporation, E and transpiration, T; mm/a) and DD and T each as a percentage of P + II, for representative cropping sequences from 1997 to 2010 for the Lockyer Valley. Average P= 701 mm/a.

Cropping sequence	I	DD	Q	E	T	DD (%)	T (%)
1. Forage-grain-pasture	326	11	37	373	703	1.1	69
2. Vegetable-forage-grain-pasture	575	88	74	414	681	6.9	53
3. Vegetable	598	121	67	383	602	9.3	46

DD (%) = DDx100/(I+P), T (%) = Tx100/(I+P).

Deep drainage as a percentage of rainfall + irrigation increased, whereas transpiration as a percentage of rainfall + irrigation decreased from forage-grain-pasture to vegetable sequence, that is, with increasing cropping intensity and irrigation amount. Deep drainage per unit of water (from rainfall + irrigation) was about 9 times lower but that of transpiration was 1.5 times higher under forage-grain-pasture than vegetable sequence (Table 21). The relationship between deep drainage and irrigation + rainfall shows a highly significant positive correlation (P = 0.01) under the vegetable sequence (Figure 80a), whereas the relationship between deep drainage and rainfall (alone) shows a significant positive correlation (P=0.05) under the forage-grain-pasture and highly significant positive correlations (P=0.01) under vegetable-forage-grain-pasture and vegetable sequences (Figure 80b).

Figure 80a shows that the minimum amount of rainfall + irrigation required to cause deep drainage was between 936 mm and 1065 mm, depending on the cropping sequence. This amount was met almost every year under sequences that included vegetables but not under the forage-grain-pasture sequence.

For a deep drainage event of above 100 mm, the amount of rainfall + irrigation was at least 1200 mm (with the exceptions of 2010 and 2006 respectively for forage-grain-pasture and vegetable sequences). This amount was met at least 9 out of the 14 years (i.e. at least 64% of the time) under sequences that included vegetables. In contrast, forage-grain-pasture sequence had only 2 out of the 14 years (i.e. 14% of the time) with rainfall + irrigation in excess of 1200 mm (Figure 80a).

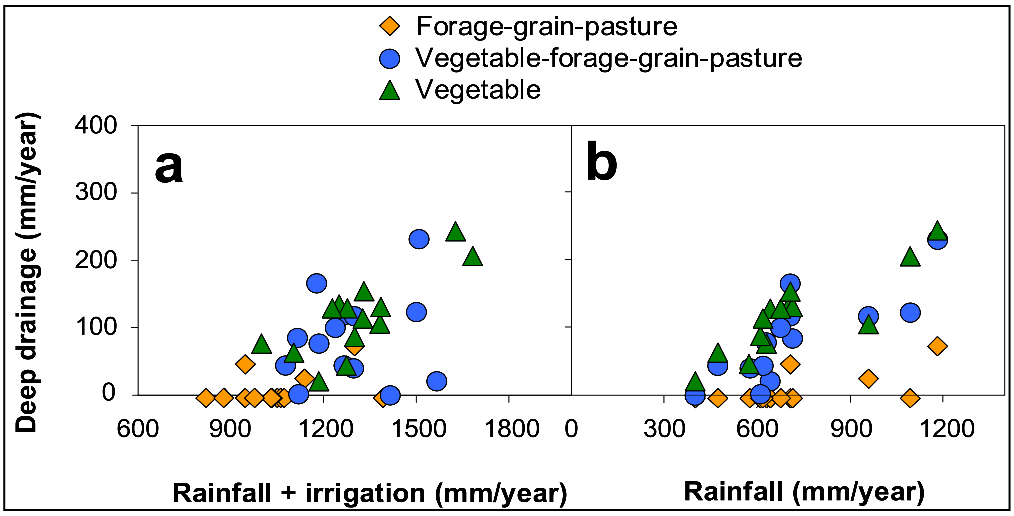


Figure 80: Relationship between deep drainage and a) rainfall + irrigation, and b) rainfall, for representative cropping sequences from 1997 to 2010 for the Lockyer Valley. Correlation under figure a) for forage-grain-pasture and vegetable-forage-grain-pasture, P = n.s., and vegetable sequences, r = 0.80, P = 0.01. Correlation under figure b) for forage-grain-pasture, r = 0.62, P = 0.05; vegetable-forage-grain-pasture, r = 0.80, P = 0.01; and vegetable sequences, r = 0.87, P = 0.01.

Deep drainage, runoff, soil evaporation and transpiration each as a percentage of water input (rainfall, irrigation and stored soil water) for fallow and crops under three cropping sequences are presented in Figure 81. Deep drainage, runoff and soil evaporation as a percentage of water balance input was higher for fallow than for crops under all cropping sequences. Under forage-grain-pasture sequence, wheat and sorghum had lower soil evaporation but higher transpiration than lucerne (Figure 81a), whereas under vegetable-forage-grain-pasture sequence, wheat, sorghum and lucerne had lower deep drainage and soil evaporation but higher transpiration as a percentage of soil water balance input than broccoli and lettuce (Figure 81b). When stored moisture was accounted for, transpiration as a percentage of water balance input for sorghum and wheat under forage-grain-pasture sequence exceeded 100% (data not shown). That is, these crops are using stored water. Among crops under the vegetable cropping sequence, lettuce had the highest soil evaporation but lowest transpiration as a percentage of soil water balance input, but a contrasting result was evident with sweet corn and bean (Figure 81c).

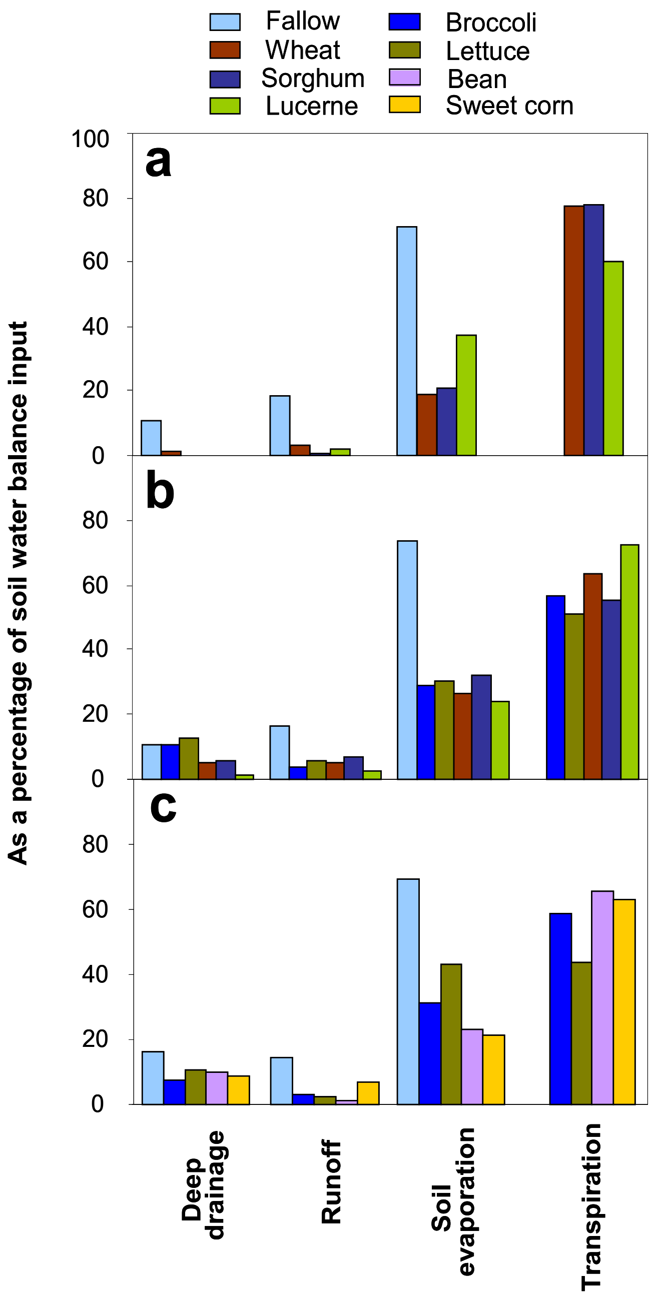


Figure 81: Deep drainage, runoff, soil evaporation and transpiration each as a percentage of water input (rainfall and/or irrigation) for a) forage-grain-pasture, b) vegetable-forage-grain-pasture, and c) vegetable sequences from 1997 to 2010 for the Lockyer Valley. (Note: Stored soil water not accounted for.)

To study the effect of cropping sequences alone on soil water balance (i.e. to separate the affect of irrigation strategy), soil water balance was also studied under an ideal irrigation practice (irrigation to DUL), similar to the other two cropping sequences. The resulting soil water balance for forage-grain-pasture sequence (annual average: irrigation = 589 mm, runoff = 49 mm, soil evaporation = 345 mm, transpiration = 903 mm and deep drainage = 36 mm, data not shown in Table 21) shows that irrigation and transpiration were increased by 263 mm/a and 200 mm/a respectively, whereas deep drainage and runoff were increased by 25 mm/a and 12 mm/a respectively, compared with the same sequence under deficit depth irrigation strategy.

Annual distribution of the components of soil water balance (Figure 82) showed that rainfall was in excess of 800 mm/a for the years 1999, 2008 and 2010, whereas rainfall was less than 600 mm/a for the years 2000, 2002, 2003 and 2006 (Figure 82a). Irrigation was inversely related to rainfall in each year and this was apparent for the very wet and very dry years under all cropping sequences. Deep drainage varied among cropping sequences and there were only three years (1999, 2009 and 2010) that had deep drainage under the forage-grain-pasture sequences (Figure 82b), whereas deep drainage occurred every year under the vegetable-forage-grain-pasture (Figure 82c) and the vegetable sequences (Figure 82d). Deep drainage patterns were similar to the rainfall pattern, especially under the vegetable-forage-grain-pasture (Figure 82c) and the vegetable cropping sequences (Figure 82d).

Average monthly distribution of the components of soil water balance (Figure 83) showed that rainfall was in excess of 50 mm from October to March. The highest rainfall (e.g. >100 mm) occurred in November and December, whereas the lowest rainfall (e.g. <25 mm) occurred in July and August (Figure 83a). Irrigation distribution was inconsistent among months in all the cropping sequences. Under the forage-grain-pasture sequence, monthly irrigation corresponded somewhat inversely to monthly rainfall trends, especially in very wet months (Figure 83b), whereas the relationship between irrigation and rainfall was somewhat unclear under the vegetable-grain-pasture (Figure 83c) and the vegetable sequences (Figure 83d). Deep drainage trends corresponded to the rainfall trend with all cropping sequences. Deep drainage was infrequent, that is, occurred only during November to February and in April under the forage-grain-pasture sequence (Figure 83b). In contrast, deep drainage occurred in all the months under vegetable-forage-grain-pasture (Figure 83c) and the vegetable sequences (Figure 83d).

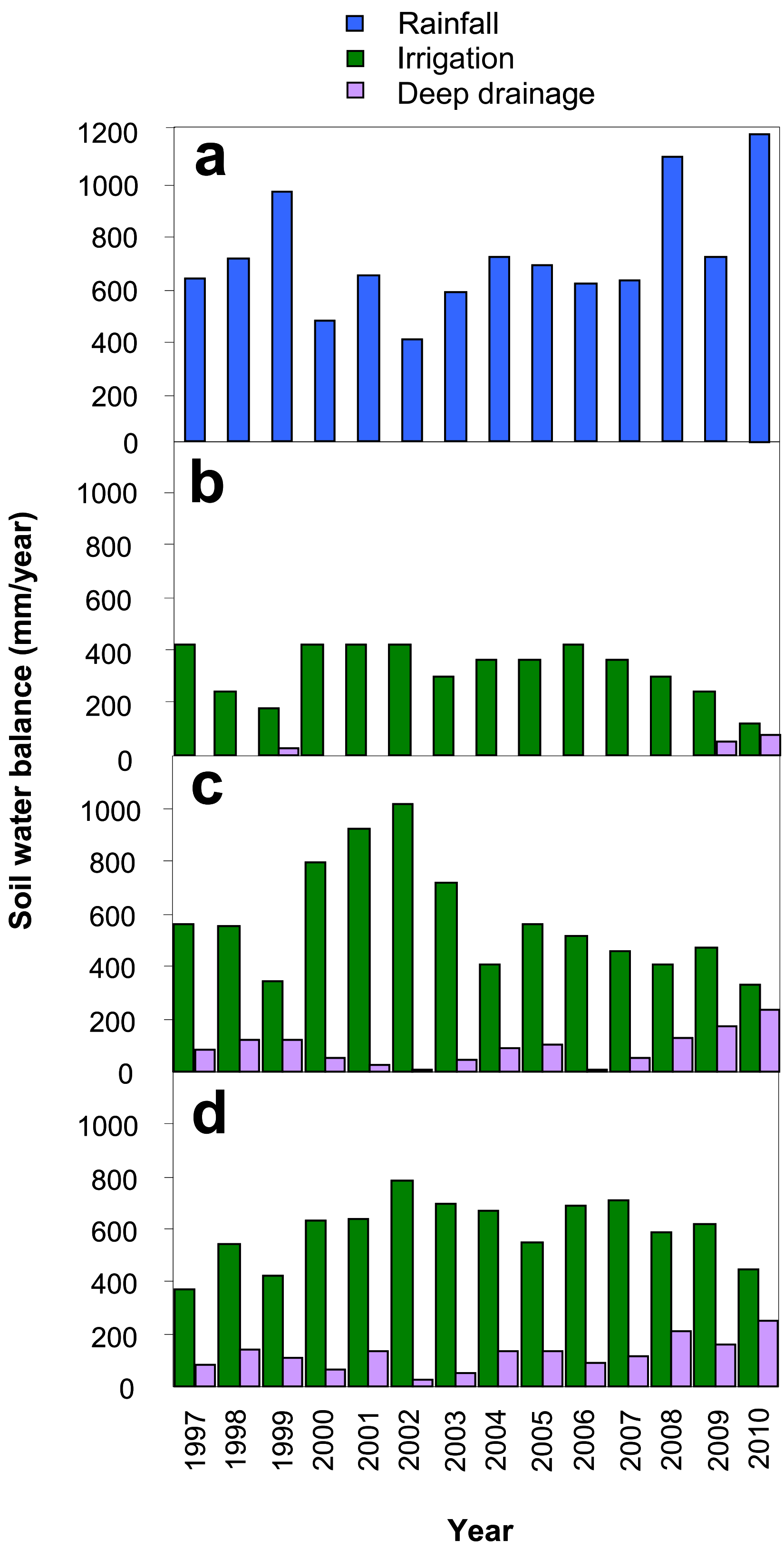


Figure 82: Annual distribution of rainfall (a) and irrigation and deep drainage for forage-grain-pasture (b) vegetable-forage-grain-pasture, and (c) vegetable sequences, from (d) 1997 to 2010 for the Lockyer Valley.

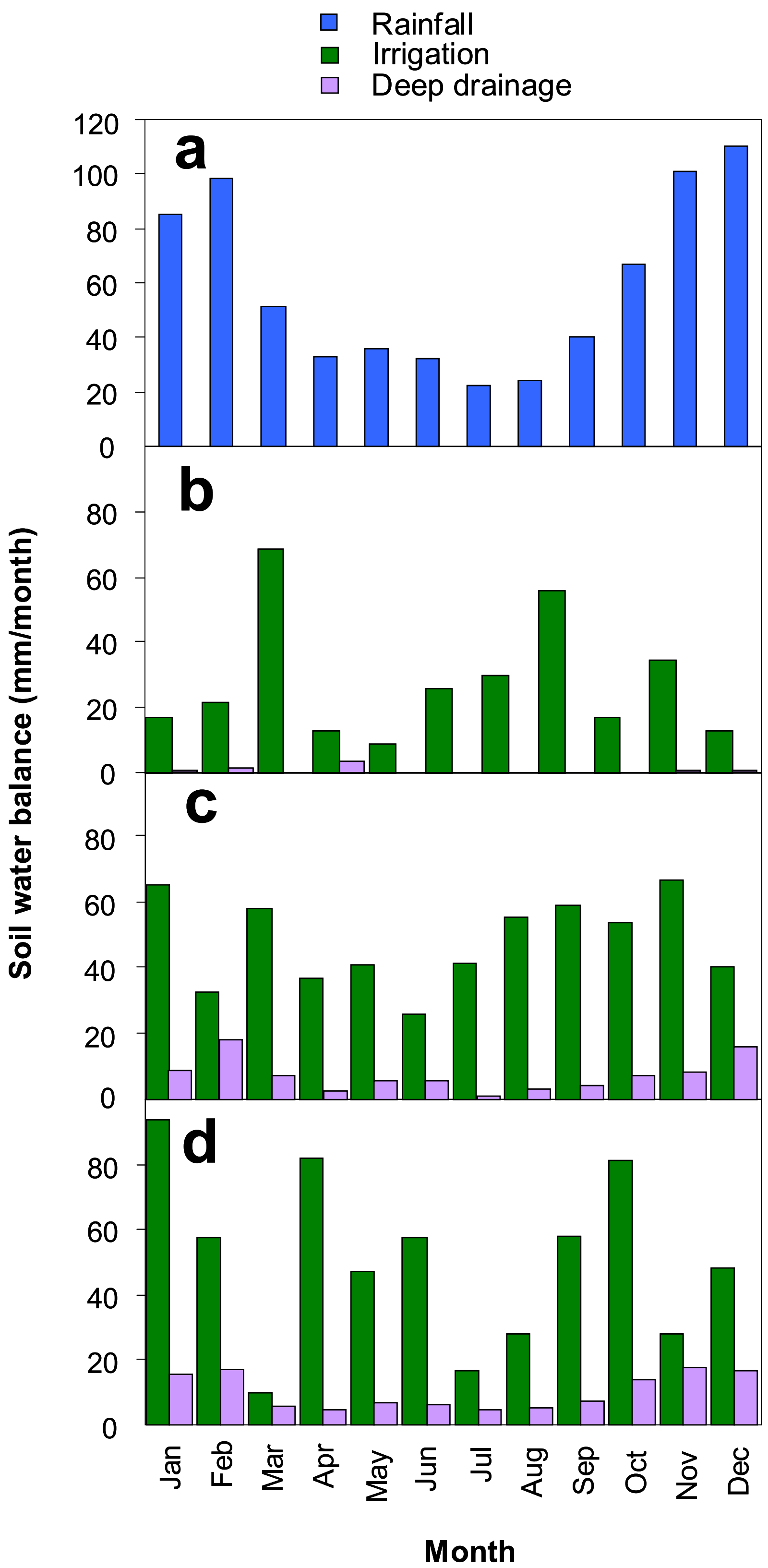


Figure 83: Average monthly distribution of rainfall and (a) irrigation and deep drainage for forage-grain-pasture, (b) vegetable-forage-grain-pasture, and (c) vegetable sequences from (d) 1997 to 2010 for the Lockyer Valley.

SODICS results show that deep drainage ranged from 0 mm/a to 518 mm/a with an average deep drainage of 148 mm/a for Black or Grey Vertosol of the Lockyer Valley, depending on the sampled site (Figure 84).

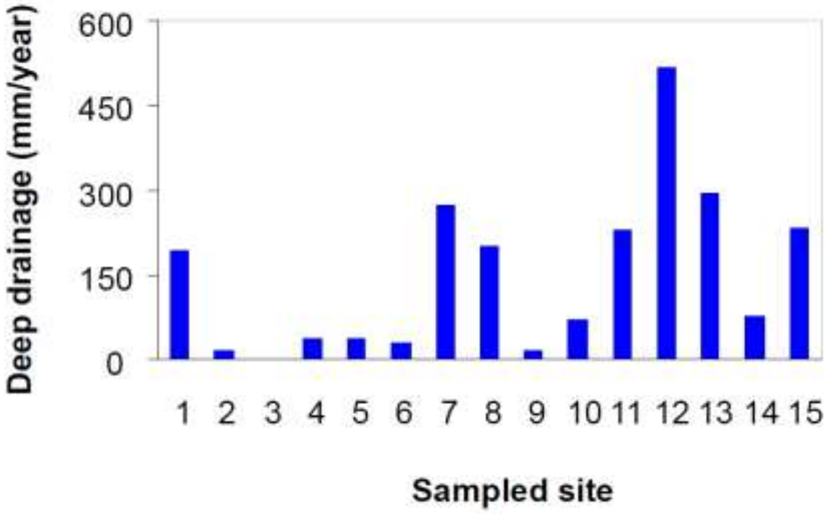


Figure 84: Deep drainage estimates from SODICS for a Black or Grey Vertosol of the Lockyer Valley (adopted and modified from Thorburn *et al.*, 1990).

For the continuous fallow simulation, runoff, soil evaporation and deep drainage comprised 9%, 65% and 26% of the total rainfall respectively (Figure 85). For wheat-fallow simulation, runoff, soil evaporation and deep drainage constituted 8%, 53% and 9% of the total rainfall respectively (Figure 85).

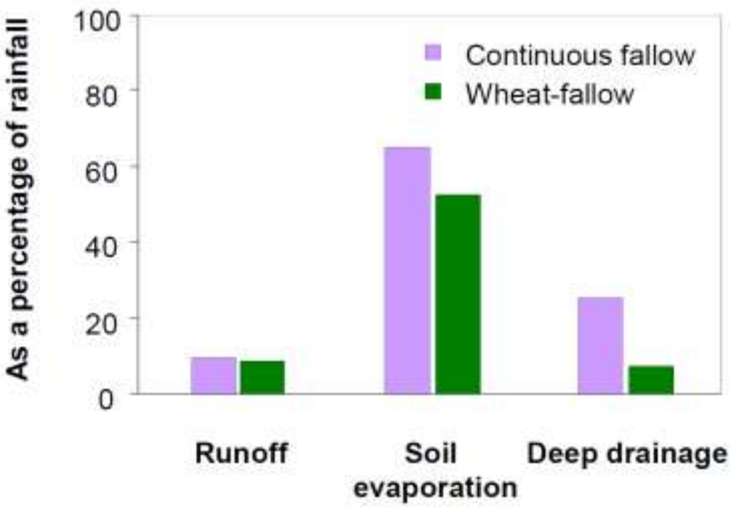


Figure 85: Runoff, soil evaporation and deep drainage each as a percentage of rainfall under a continuous fallow and a wheat-fallow from 1997 to 2010 for the Lockyer Valley.

9.2.3 Discussion

Cropping sequences is one of the major factors that affect soil water balance. Differences in the components of soil water balance among the cropping sequences (Table 21) suggest that soil water can be efficiently exploited by minimising deep drainage and maximising transpiration. Large differences in deep drainage indicate a high sensitivity of deep drainage to cropping sequences that also differed in crop characteristics and irrigation management. For example, in the forage-grain-pasture sequence, crops received deficit irrigation (i.e. root zone was partially refilled with water when water in the root zone was nearly empty), hence this sequence was less sensitive to deep drainage when rainfall occurred after irrigation. In the vegetable sequence, crops received adequate irrigation (i.e. root zone was completely refilled when water in the root zone was partly empty), hence this sequence was very sensitive to deep drainage. Since deep drainage is the driver of groundwater recharge, differences among the cropping sequences can have considerable impact on the groundwater recharge.

Significant areas of the Lockyer Valley are affected by groundwater salinity (Bajracharya and Ellis, 1999). Deep drainage estimates from SODICS (Figure 84) can equate to substantial quantities of Cl displacement below the root. When this Cl is added to already saline groundwater, groundwater salinity would further increase. The saline groundwater can reach the root zone within a short period when pumped as a source of irrigation water. Under poor irrigation practices, frequent irrigation with salty water under vegetable cropping may add up salts in the root zone, especially in saline areas with shallow groundwater tables. This may further increase the salinity risks.

High rates of deep drainage under vegetable cropping are expected even under ideal irrigation management. A positive correlation between deep drainage and rainfall + irrigation under the vegetable sequence (Figure 80) substantiates these findings. Irrigation is a routine practice for year-round, profitable crop production in the Lockyer Valley. To maintain high produce quality and to achieve high yield, the soil profile needs to be kept relatively moist (i.e. low SWD) for most of the cropping period. It is not uncommon in the Lockyer Valley for farmers to maintain soil moisture above the DUL for much of the growing season, causing continuous deep drainage (data not shown). Maintenance of a low SWD also means that there is little capacity for soil to hold rainfall following irrigation, and therefore rainfall tends to either runoff or drain, as seen in the current study (Table 21).

Differences in soil water balance among cropping sequences were related to differences in the crop characteristics. Forage-grain-pasture sequence represented crops such as lucerne and sorghum. Low deep drainage and high transpiration as a percentage of rainfall + irrigation with these crops (Figure 81a) show their efficiency in water use. Transpiration as a percentage of rainfall + irrigation was in excess of water input during some years with crops such as sorghum (data not shown). This is evidence of their ability to explore stored soil water. Deep root systems associated with these crops help to explore and dry the soil to greater depth. This in turn helps to increase the soil water storage capacity, which otherwise may result in deep drainage. Depending on the moisture input, moisture use pattern and stage of the crop growth, stored water may also be useful for subsequent crops or rotations, as in the case of lucerne where the water stored in the cropping year will be available for the perennial phase (Ridley *et al.*, 2001). Crop duration also plays an important role in soil water balance. Although not apparent with deep drainage, a lower net transpiration with shallow-rooted crops such as lettuce (Figure 81c) was due to their shorter growing periods. A similar effect is true with planting time where, for example, autumn- and winter-planted broccoli and lettuce under vegetable sequence had lower transpiration (broccoli in particular) and deep drainage as a percentage of water input than year-round planted broccoli and lettuce under vegetable-forage-grain-pasture sequence. A lower transpiration and deep drainage percentage with autumn- and winter-planted broccoli and lettuce corresponded to the cooler and/or lower rainfall periods shown in Figure 83.

Soil water balance is also affected by a crop's minimum irrigation requirements. Some deep-rooted crops (e.g. sorghum) were assumed to tolerate considerable moisture stress and hence suit limited irrigation supply. In contrast, most high value vegetables such as lettuce and broccoli have shallow root systems that cannot extract stored water from deeper soil layers and are also susceptible to water stress. Even the deeper-rooted vegetable crops such as sweet corn may not be suitable for moisture-limited conditions due to issues concerning the product quality under reduced moisture supply. Therefore, successful vegetable cropping requires reliable water supply.

Irrigation management plays an important role in determining the actual soil water balance. Forage-grain-pasture sequence was deficit irrigated, hence reduction in the deep drainage was partly due to reduction in irrigation. When forage-grain-pasture sequence was ideally irrigated (i.e. irrigated to

DUL) similar to other cropping sequences, irrigation and transpiration increased considerably, while deep drainage increased marginally and was considerably lower than sequences involving vegetables. Thus, our results further substantiate the suitability of deep-rooted crops in minimising deep drainage and maximising transpiration and also show the effect of irrigation strategies on the components of soil water balance. Although changes to the irrigation strategies had little effect on deep drainage under forage-grain-pasture, their effect may be large under vegetable cropping. For example, up to a 3-fold increase in the simulated deep drainage was shown when irrigation application level was increased from DUL to DUL+50% in the Lockyer Valley (see Section 9.3.2). Irrigation requirements, and in turn deep drainage, may further increase if the irrigation water and root zone already contain salts, as extra water is needed to flush out the salts from the root zone.

Annual and monthly irrigation and deep drainage were related to the corresponding rainfall for all the cropping sequences (Figure 82 and Figure 83). The inter-year and intra-year variability in the rainfall and corresponding changes to components of the soil water balance is similarly evident (Abbs and Littleboy, 1998; Nulsen, 1993). A significant positive correlation between deep drainage and rainfall for all cropping sequences (Figure 80b) further substantiates the effect of rainfall on deep drainage. Relationships between rainfall and deep drainage is widely reported (e.g. Keating *et al.*, 2002; Shaw and Thorburn, 1985; Foley *et al.*, 2013); including those at catchment scale (Zhang *et al.*, 2001). Higher deep drainage was associated with higher rainfall amount, longer rainfall duration and low evaporative periods such as winter (Keating *et al.*, 2002). The episodic nature of deep drainage is apparent in the Lockyer Valley (Figure 82 and Figure 83). Episodic nature of deep drainage was related to rainfall distribution (Figure 82a and Figure 83a), clay content (Lewis, 1997), fallow conditions, crop characteristics such as green cover, growth stage (data not shown) and rooting depth (Lewis, 1997). Though rainfall is one of the major factors that regulate soil water balance, differences in the annual and monthly irrigation and deep drainage among cropping sequences (Figure 82b,c,d and Figure 83b,c,d) further substantiate the effect of cropping sequence and related factors (e.g. crop characteristics) in the regulation of soil water balance.

Modelled results agree with other methods of deep drainage estimation. Although there are no direct measurements of deep drainage for the Lockyer Valley, current deep drainage estimates fall well within the range of SODICS results (Thorburn *et al.*, 1990, Figure 84) that were estimated for a range of growing conditions of the Lockyer Valley. Our deep drainage estimates were under an ideal irrigation practice where irrigation alone led to no deep drainage, whereas the SODICS estimate was under realistic irrigation practice where irrigation contributes to deep drainage. Thus, a good comparison of current results with the lower range of the SODICS is reasonable. We also acknowledge that the result comparison is affected by study period and/or spatial heterogeneity. The study period received 74 mm/a lower rainfall than long-term (1900–2011) mean (Figure 79b). Therefore, deep drainage and irrigation requirements for the study period should respectively be lower and higher than long-term or other periods with higher rainfall, under identical management practices.

Similar to deep drainage, results on other components of the water balance are substantial. There have been limited water balance studies for the Lockyer Valley; however, the results are comparable with other studies from similar growing environments. For example, long-term average soil evaporation as a percentage of fallow rainfall ranged from 65% to 73% for Greenmount (Freebairn, unpublished data), Wallumbilla catchment (Freebairn *et al.*, 2009) and Goondoola basin (Robinson *et al.*, 2010) studies, whereas runoff as a percentage of rainfall for wheat-fallow ranged from 7.7% to 11% for Greenmount (Silburn *et al.*, 2007), Wallumbilla catchment (Freebairn *et al.*, 2009), Brigalow Catchment (Thornton *et al.*, 2007) and Goondoola basin (Robinson *et al.*, 2010) studies. Our results (Figure 85) are in agreement with these studies.

The current study demonstrates the usefulness of modelling in exploring the trends in soil water balance and the factors that affect soil water balance, especially under long-term cropping. Although the model has been validated across various soils and climates and the current study results were justified with field and published research, wide differences in the management practices under realistic conditions and uncertainties in precise representation of the field conditions are the limitations of the current study. Studies that address the effect of irrigation strategies on soil water balance (e.g.

field studies in particular that address the sensitivity of deep drainage to irrigation management) and long-term modelling studies that account for climate variability (e.g. wet and dry spells) for a range of crops and management practices will further enhance our understanding in this direction.

To sum up, deep drainage in the current study ranged from 11 mm/a (for a forage-grain-pasture sequence) to 121 mm/a (for a vegetable sequence) when irrigated up to DUL for a Black Vertosol in the Lockyer Valley. Forage-grain-pasture sequence comprised of crops such as lucerne and sorghum utilised moisture efficiently and resulted in higher transpiration but lower deep drainage compared with crops such as broccoli and lettuce. The current study shows the effects of crop characteristics (e.g. crop duration, green cover, rooting depth and crop water requirements), irrigation strategies and rainfall trends on the soil water balance and on deep drainage in particular. Increases in the irrigation application (from deficit to DUL) in the forage-grain-pasture sequence increased the transpiration considerably but increased deep drainage only slightly. This further highlighted the usefulness of crops such as sorghum and lucerne in improving water-use efficiency. Deep drainage in the Lockyer Valley can be minimised by selective incorporation of deep-rooted drought-tolerant crops such as lucerne and sorghum. These crops can be a useful option, especially where salinity and uncertainty in the continued water availability are important issues for sustainable crop and land management.

9.3 Soil Water Balance for Irrigation Practices under 12 Years of Rotation Cropping

Similar to cropping sequences, irrigation practices in the Lockyer Valley vary and depend on factors such as crop choice and water availability. Irrigation practices by farmers can have a significant impact on crop production and soil water balance. For example, differences in the irrigation practices may lead to varying rates of irrigation application due to differences in the irrigation application amount. This, in turn, can lead to varied rates of deep drainage and transpiration. However, while we understand many of the effects of irrigation application levels on soil water balance in principle, the actual outcomes for the Lockyer Valley are poorly quantified.

To address this knowledge gap, irrigation scenarios were developed based on irrigation application levels which represent ideal and practical irrigation practices for farms in the Lockyer Valley. The objective was to estimate the soil water balance and deep drainage in particular, under various irrigation application levels that were applied with or without soil moisture monitoring. The soil water balance estimated from this study in turn will contribute to i) understanding of the trends in the soil water balance, ii) development of deep drainage maps and estimations of groundwater recharge under a range of irrigation practices, and iii) development of irrigation strategies that can minimise deep drainage and maximise transpiration.

9.3.1 Methodology

9.3.1.1 Simulation

Soil water balance for various irrigation application levels were simulated using the HowLeaky model (Rattray *et al.*, 2004). The simulation study comprised scenarios based on irrigation application levels, namely, refilling the soil water deficit i) up to drained upper limit (DUL), ii) up to DUL+25% (where 25% is the percentage of soil water between DUL and saturation), iii) up to DUL+50%, and iv) fixed amount. The first three scenarios represent irrigation with soil moisture monitoring (i.e. soil water deficit [SWD]) based irrigation practices, whereas the fixed amount scenario represents irrigation without soil moisture monitoring (i.e. non-SWD) based (see Section 9.2.2.1.4 for details). These irrigation scenarios represent ideal and practical irrigation target amounts and may also depict a range of irrigation efficiencies associated with local irrigation systems or practices for farms in the Lockyer Valley. Soil water balance simulations were studied from November 1997 to April 2010.

9.3.1.2 Climate and Soil

The study site was Forest Hill (152°20′ E, 27°36′ S). Climate data were for the Department of Primary Industries (DPI) Gatton station (152°19' E, 27°32' S) and were obtained from the SILO Patched Point dataset (Jeffrey *et al*., 2001). The study period (1997–2010) had (average) annual rainfall of 697 mm, annual pan evaporation of 2034 mm; daily radiation of 19 MJ/m² and daily maximum and minimum temperatures of 27.3 °C and 13.4 °C respectively. Soil data were for a Black Vertosol (Isbell, 1996). Soil physical properties were set for each soil layer up to 1.8 m depth. Soil moisture content and bulk density data were obtained from the soil samples cored from Forest Hill site using a soil corer. The soil parameters used in HowLeaky are presented in Table 22.

Table 22: Soil parameters for a Black Vertosol of the Forest Hill, central Lockyer Valley.

Detail	Unit	Value (for selective depth)			
Layer Depth	**M**	**0.2**	**0.3**	**1.2**	**1.8**
Air dry water content	% (v/v)	13	20	20	20
Saturated water content	% (v/v)	47	46	43	43
Field capacity water content	% (v/v)	44	43	41	41
Wilting point water content	% (v/v)	26	26	25	26
Bulk density	g/cm³	1.2	1.3	1.4	1.4
		Value (for all selective depths)			
Evaporation stage 1 (U)	mm	6.5			
Evaporation stage 2 (Cona)	mm/day$^{0.5}$	3.8			
Runoff curve number for bare soil		73			
Curve number reduction at 100% cover		18			

Note: Water content and bulk density data are from soil samples of Forest Hill, Queensland.

9.3.1.3 Vegetation

Vegetation parameters for each crop were developed using the 'crop cover' module within HowLeaky, wherein the user supplies temporal patterns of total and green cover. All irrigation applications had the same cropping sequence based on historic cropping data from a farmer's diary for the study period for Forest Hill. Cropping sequence comprised of cotton (spring/summer sown), wheat (autumn/winter), mung bean (summer), beetroot (year-round), broccoli (autumn/winter) and sweet corn (summer) rotated at varied intensity during the study period.

9.3.1.4 Irrigation

All crops except wheat (rainfed) were irrigated under all irrigation applications. Irrigation applications varied in their target amount and were estimated by: (i) an irrigation 'end-point', such as drained upper limit (DUL) or higher; or (ii) a fixed amount at each irrigation event. Ideal application was simulated by topping up the soil to its DUL, with no waste to tail water and causing no deep drainage due to irrigation. Any deep drainage that occurred was then due to rainfall after irrigation exceeding the soil water-holding capacity. Application in excess of DUL was simulated to account for: i) inefficiencies in irrigation; ii) additional water that may be required to meet the leaching requirement; and iii) generous irrigation practiced by some growers (to overcome the risk of under-irrigation and associated reduction in crop yield and quality). Under up to DUL application, all the crops were irrigated to DUL (except wheat), whereas under up to DUL+25% and up to DUL+50%, shallow-rooted crops (i.e. beetroot and broccoli) were irrigated to DUL+25% and DUL+50% respectively, whereas other crops (i.e. cotton, mung bean and sweet corn) were irrigated to DUL and DUL+25% respectively.

Irrigation trigger (time to start irrigation) under up to DUL, up to DUL+25% and up to DUL +50% applications was based on critical SWD (mm below DUL), and were 60 mm for cotton and sweet corn, 50 mm for mung bean and 25 mm for beetroot and broccoli. Irrigation trigger under fixed amount was based on fixed intervals (i.e. non-SWD, e.g. weekly, fortnightly). With and without SWD applications respectively mimicked irrigation practices with and without soil moisture monitoring. Soil water content at the beginning of each simulation for each crop was set based on actual soil water content at the end of the previous crop or fallow (derived from HowLeaky's SWD output option).

9.3.2 Results: Effect of Irrigation Application Levels on Soil Water Balance

As the irrigation increased from 530 mm/a (up to DUL) to 717 mm/a (up to DUL+50%), deep drainage increased from 84 mm/a to 262 mm/a, that is, by 3.1-fold (Table 23). However, runoff, soil evaporation and transpiration were similar across all SWD-based applications. Fixed amount had the highest runoff and evaporation but the lowest transpiration (Table 23).

Table 23: Average annual water balance (rainfall, P; irrigation, I; deep drainage, DD; runoff, Q; evaporation, E; and transpiration, T; mm/a) and DD and T each as a percentage of P+I for four irrigation application levels from 1997 to 2010 for the Lockyer Valley. Average P=697 mm/a.

Irrigation Application Level	I	DD	Q	E	T	DD (%)	T (%)
1. Up to DUL+50%	717	262	72	401	676	19	48
2. Up to DUL+25%	614	164	68	401	677	13	52
3. Up to DUL	530	84	67	403	673	7	55
4. Fixed amount	595	122	90	443	595	9	46

DD (%) = DDx100/(P+I), T (%) = Tx100/(P+I).

Deep drainage and transpiration each as a percentage of rainfall + irrigation is shown in Table 23. As the irrigation increased, deep drainage as a percentage increased, whereas transpiration as a percentage decreased with all SWD-based irrigation applications. Fixed amount irrigation application had the lowest transpiration as a percentage, although irrigation was lower compared with both up to DUL+25% and up to DUL+50% (Table 23).

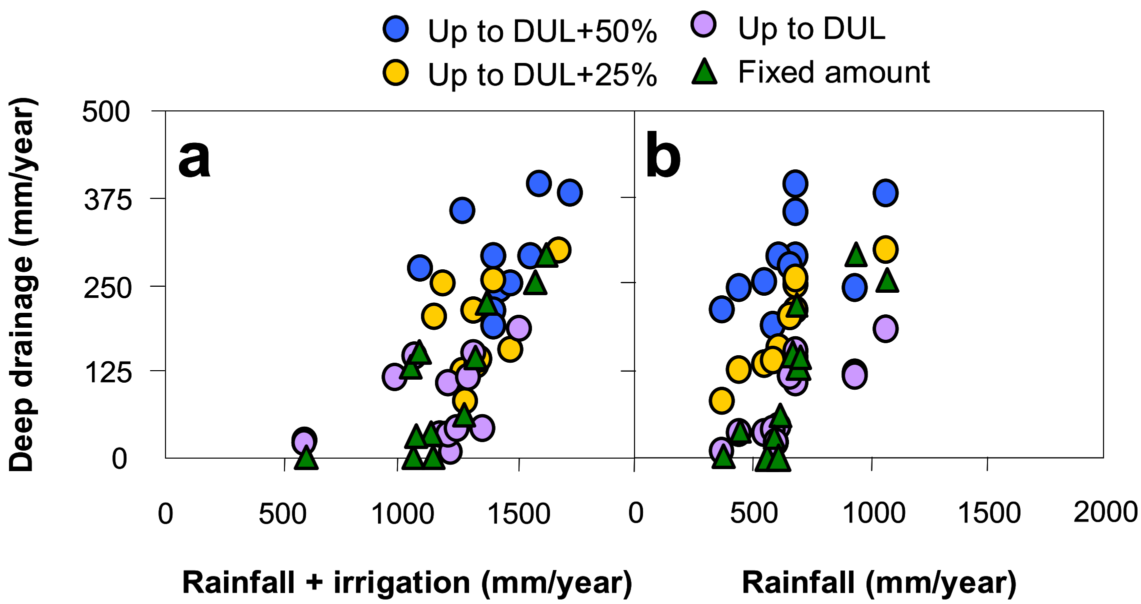


Figure 86: Correlations between deep drainage and a) rainfall+irrigation, and b) rainfall, for four irrigation application levels from 1998 to 2009 in the Lockyer Valley. Correlation under a) for Up to DUL+50%, r = 0.78, P= 0.01; Up to DUL+25%, r = 0.66, P =0.05, Up to DUL, P=n.s., and Fixed amount, P=n.s; Correlation under b) Up to DUL+50%, P=n.s., Up to DUL+25%, r =0.58, P = 0.05; Up to DUL, r=0.81, P=0.01, and Fixed amount, r = 0.86, P=0.01. Note: data for partially studied years (1997 and 2010) not included.

Significant positive correlations were found between deep drainage and a) irrigation + rainfall for i) up to DUL+25% and ii) up to DUL+50% (Figure 86a), b) rainfall, for all irrigation applications except up to DUL+50% (Figure 86b).

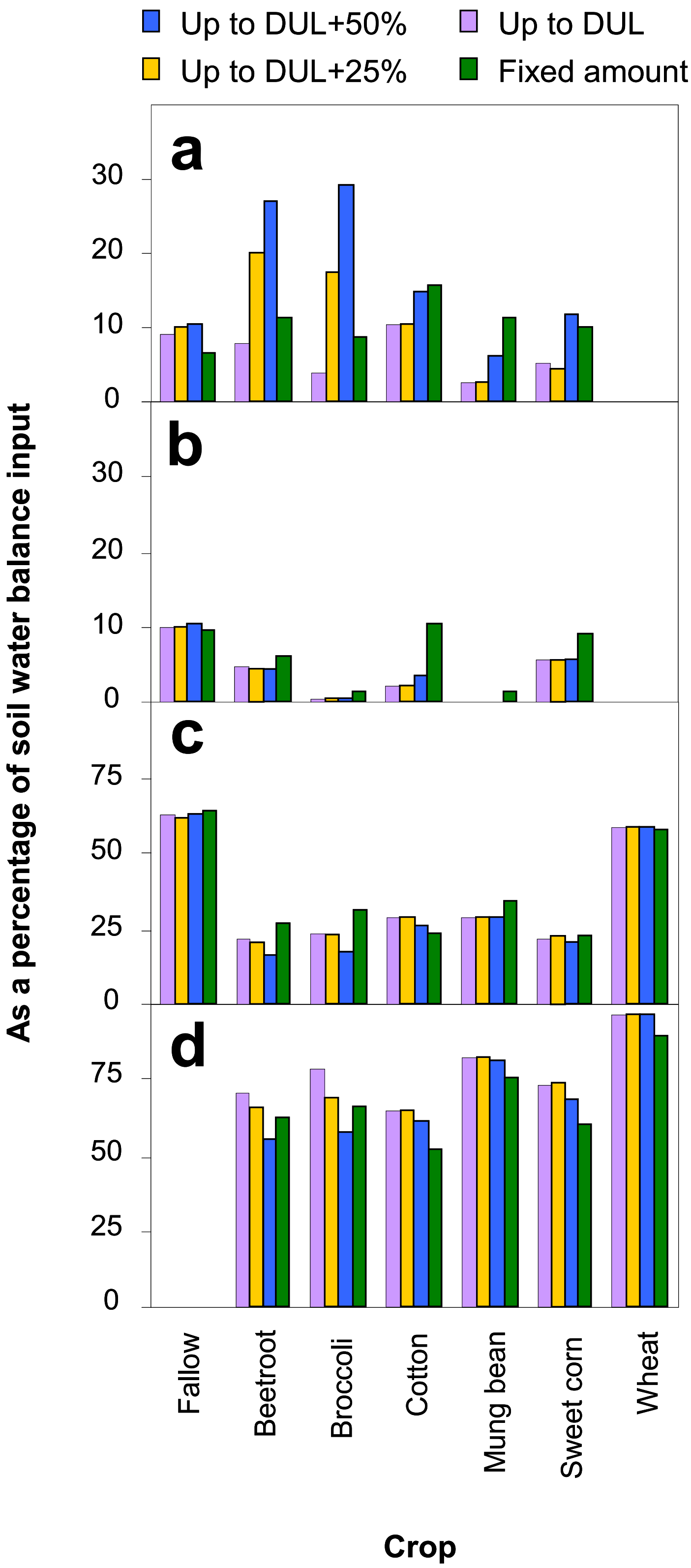


Figure 87: **a) Deep drainage, b) runoff, c) soil evaporation, and d) transpiration each as a percentage of soil water balance input (rainfall and/or irrigation) for crops (or fallow) for four irrigation application levels from 1997 to 2010 for the Lockyer Valley. (Note: stored soil water not accounted for.)**

Deep drainage, runoff, soil evaporation and transpiration each as a percentage of soil water balance input (rainfall and/or irrigation) for fallow and crops are presented in Figure 87. Deep drainage as a percentage was generally lower for fallow, beetroot and broccoli, but was higher for cotton and mung bean under Fixed amount irrigation application than most or all of the SWD-based scenarios (Figure 87a). Among SWD-based irrigation applications, deep drainage as a percentage increased with increase in irrigation (i.e. from Up to DUL to up to DUL+50%) for crops and fallow (a residual effect of irrigation during cropping phase). When irrigation increased from DUL to DUL+25%, the degree of change in the deep drainage as a percentage was higher for beetroot and broccoli than for other crops (degrees of changes were not estimated from DUL+25% to DUL+50% as not all the crops were irrigated to DUL+50%).

Runoff and soil evaporation as a percentage of soil water balance output (Figure 87b,c) were similar for fallow but were generally higher for all irrigated crops under fixed amount irrigation application than SWD-based. Among SWD-based scenarios, changes to the runoff and soil evaporation were marginal.

Transpiration as a percentage of soil water balance input (Figure 87d) was lower under Fixed amount irrigation applications than all SWD-based, except Fixed amount was higher than DUL+50% for beetroot and broccoli. Among SWD-based, transpiration (as a percentage) decreased with increase in irrigation above DUL. As the irrigation increased from DUL to DUL+25%, the magnitude of change in the transpiration as a percentage was higher for beetroot and broccoli.

Annual distribution of the components of soil water balance (Figure 88) showed that rainfall was in excess of 800 mm/a for the years 1999 and 2008 but less than 600 mm/a for the years 2000, 2002, 2003 and 2006 (Figure 88a). As soil water balance was studied for part of 1997 and 2010, they were omitted from comparison with other years. Irrigation amounts are inversely correlated with the rainfall and also related to the crops grown, especially under SWD-based irrigation applications (Figure 88b,c,d). Similarities between Fixed amount and SWD-based irrigation applications in the irrigation trends were apparent during very dry years (e.g. 2002, 2003) and in some wet years (e.g. 2009). Irrigation amounts under Fixed amount application were more even over the years than SWD-based application, which was apparent in the period 1998–2003. Similar to transpiration, deep drainage during the study period varied among irrigation applications (Figure 88b,c,d,e) and was related to rainfall trends. Deep drainage in each year increased with increase in irrigation application.

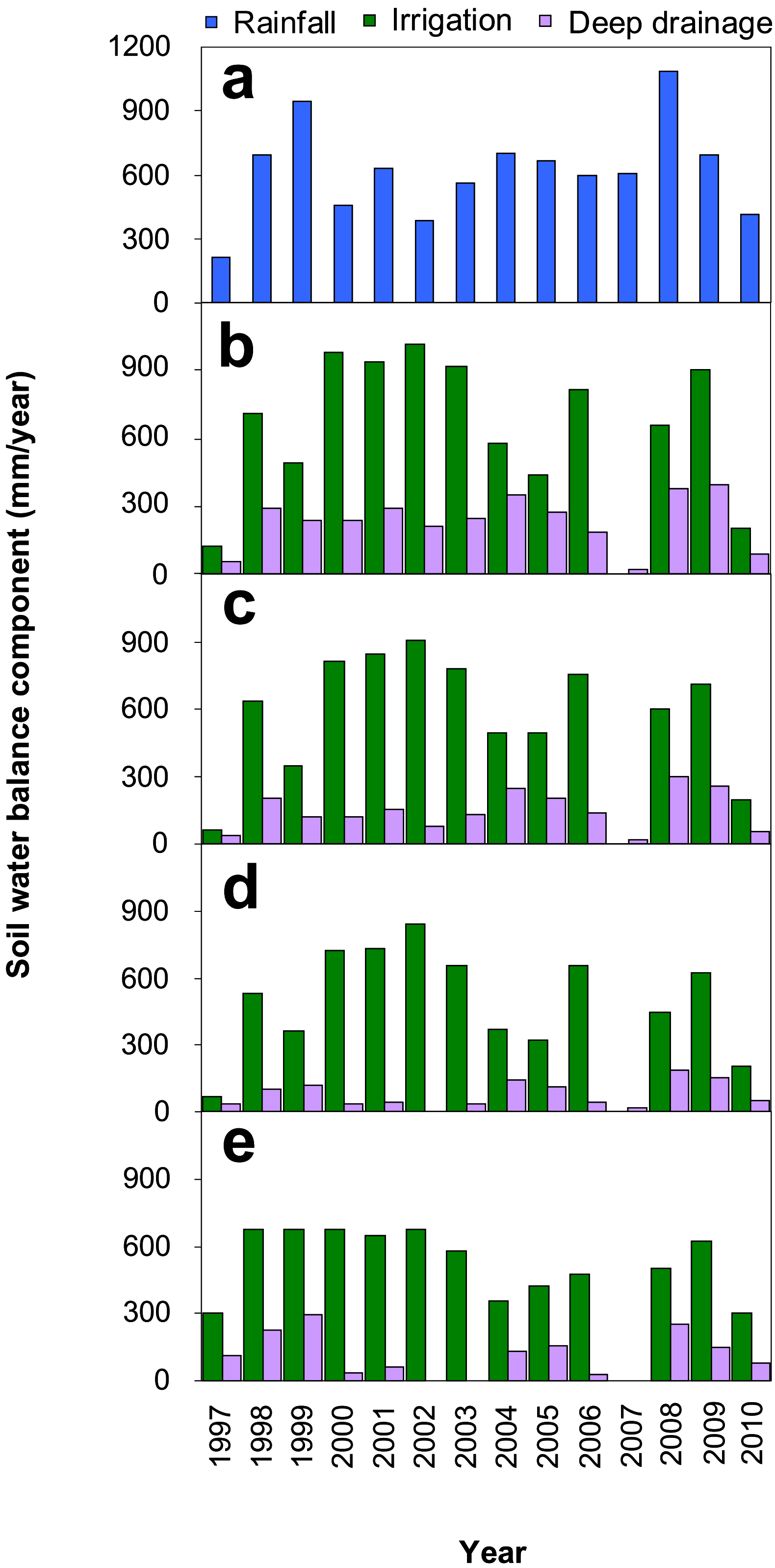


Figure 88: Annual distribution of a) rainfall, and irrigation and deep drainage for b) Up to DUL+50%, c) Up to DUL+25%, d) Up to DUL, and e) Fixed amount irrigation application levels, from 1997 to 2010 for the Lockyer Valley. (Note: 1997 and 2010, partially studied years).

9.3.3 Discussion

Soil water balance was affected by irrigation application levels (Table 23). Irrigation was the major soil water balance input component that varied among irrigation scenarios. Therefore, the differences in the output components, particularly deep drainage, were mainly due to differences in the irrigation application levels. These differences suggest that the soil water balance can be regulated by adopting a suitable irrigation strategy. The irrigation scenarios studied here can reflect irrigation application inefficiencies associated with irrigation systems under field conditions. For example, irrigation to DUL reflects 0% irrigation inefficiency, whereas irrigation to DUL+50% reflects 50% irrigation inefficiency. Irrigation application levels or irrigation systems like these have been studied and used in commercial practice under similar growing environments, for example, in the Emerald Irrigation Area in central Queensland (McHugh, 2003), where sub-surface drip irrigation (SDI) at 90% and 105% of crop water requirement (or crop evapotranspiration, ETc) are similar to irrigation Up to DUL, SDI around 120% is similar to irrigation Up to DUL+25% and furrow irrigation is somewhat similar to irrigation Up to DUL+25%, DUL+50% or Fixed amount of the current study.

Irrigation beyond DUL has limited benefit for cropping. There was a 3-fold difference in deep drainage but not in transpiration, in relation to a 1.4-fold difference in irrigation among the SWD-based irrigation applications (Table 23). That is, deep drainage was found to be sensitive to irrigation beyond DUL or with increase in irrigation inefficiency. Increases in irrigation increased deep drainage as a percentage of rainfall + irrigation, but decreased the transpiration as a percentage of rainfall + irrigation (Table 23). This further substantiates that the extra irrigation was not utilised by the crops for transpiration but contributed mainly to deep drainage. Similar results on deep drainage were apparent under field conditions. For example, McHugh (2003) found substantial deep drainage of 118 mm and 46 mm (16% and 6% of rainfall + irrigation), nearly a 3-fold difference between the furrow and wettest SDI, respectively. Reduction of the 120% SDI treatment to 105% and 90% of ETc reduced drainage to close to zero, whereas SDI with 75% or with 50% of ETc had no deep drainage. These field results and the current modelled results suggest that deep drainage decreases as the applied water in relation to the crop water demand decreases. Higher deep drainage associated with furrow irrigation systems of higher irrigation inefficiency has been reported for cotton (Gunawardena *et al.*, 2011; McHugh, 2003). A poor or negative response in transpiration to excessive irrigation is also similarly evident in the literature (Abu-Awwad, 1998; Amer, 2010; Faci and Fereres, 1980). A negative response in the transpiration to increased irrigation could be due to water logging (McHugh *et al.*, 2003).

Transpiration can be maximised and runoff and soil evaporation can be minimised by responding to results from soil moisture monitoring. Transpiration was lowest under fixed amount application (Table 23) and unlike the SWD-based applications, fixed amount represented no moisture monitoring practice where crops were irrigated at fixed intervals. A lower transpiration under fixed amount therefore can be attributed to the poor timing of irrigation in response to the SWD. When applied irrigation was less than the SWD, transpiration and deep drainage were lower, whereas when applied irrigation exceeded the SWD, runoff, soil evaporation and deep drainage were higher. Reduction in the crop yield with reduction in the irrigation amount was similarly evident under field conditions (McHugh, 2003). These results suggest the importance of precise soil water monitoring and irrigation according to crop water demand for higher transpiration and in turn, higher yield. In the current study, fixed amount application assumed irrigation at moderate amounts. Thus, a lower deep drainage but higher runoff and soil evaporation under fixed amount compared with some or all of the SWD-based application is reasonable.

Contribution of irrigation and/or rainfall to deep drainage varied with irrigation applications. As seen in this study, when irrigated at or in excess of DUL+25%, the applied water exceeded the SWD and hence both irrigation and rainfall contributed to deep drainage. A significant positive correlation between irrigation beyond DUL and irrigation + rainfall substantiates this relationship (Figure 86a). When irrigated closer to DUL, the applied water closely matched the SWD and hence rainfall mainly contributed to deep drainage. A significant positive correlation between rainfall and deep drainage under all irrigation applications except up to DUL+50% (Figure 86b) justifies this contribution.

Soil water balance was affected by cropping conditions and crop characteristics. Differences in the soil water balance output as a percentage of input were apparent between the SWD-based and fixed amount applications and also among the SWD-based irrigation applications (Figure 87). A higher deep drainage as a percentage of rainfall + irrigation under fixed amount application for crops such as cotton and mung bean (Figure 87a) corresponded with the wetter period of the year (i.e. October to March). Lower SWD but continued irrigation (due to non–soil-moisture monitoring) in turn led to deep drainage at few time points during the cropping period (data not shown). As the irrigation increased above DUL among the SWD-based applications, changes in the deep drainage as a percentage were apparent, particularly with shallow-rooted crops. That is, when irrigation increased, the degree of change in the deep drainage as a percentage was higher for shallow-rooted than for deep-rooted crops. These results substantiate the sensitivity of deep drainage to irrigation above DUL for various cropping conditions and also highlight the susceptibility of shallow-rooted crops to deep drainage under increased irrigation. The effect of crop characteristics such as planting time and root depth in the regulation of soil water balance has been described previously (see soil water balance for crop rotation sequences). A link between high deep drainage and early irrigation events has been evident from field studies where high deep drainage was related to early stages of crop growth that were characterised by low transpiration (Gunawardena *et al.*, 2011) and shallow root systems. Fallow condition was found to be more susceptible for both runoff (Figure 87b) and soil evaporation (Figure 87c) than cropping phase under various irrigation applications, signifying the role of crop cover in minimising the runoff and soil evaporation.

Annual variations in irrigation requirement and deep drainage were evident for various irrigation applications. Lower rainfall periods led to lower deep drainage when irrigated closer to DUL but when irrigation increased above DUL, irrigation demand and, in turn, deep drainage increased (Figure 88). That is, irrigation demand and deep drainage in each year were dependent on rainfall as well as irrigation application levels. An opposite but dependent trend between irrigation and rainfall under the SWD-based applications was due to irrigation based on soil moisture monitoring. Least or no deep drainage in 2007 under all irrigation application scenarios corresponded to a non-irrigation period (rainfed wheat cropping). This further highlights the role of irrigation as a major soil water balance input in the regulation of deep drainage.

To sum up, deep drainage in the current study was mainly related to irrigation application levels, and deep drainage ranged from 84 mm/a (when irrigated to a fixed amount) to 264 mm/a (up to DUL+50%) for a vegetable-dominated cropping sequence for a Black Vertosol in the Forest Hill, Lockyer Valley. Modelled deep drainage was sensitive to irrigation beyond DUL. Irrigation beyond DUL did not increase the transpiration but led mainly to deep drainage. Transpiration was lowest and runoff and soil evaporation were highest under fixed amount application where soil moisture was not monitored, suggesting the importance of moisture monitoring to improve transpiration. Soil water balance was also related to crop type or crop characteristics (e.g. rooting depth, growing season) for various irrigation applications. Inter-year variability in the irrigation demand and deep drainage was apparent and was related to rainfall trends especially for the SWD-based irrigation applications. Unlike rainfall, the irrigation component of the water balance can be managed. Improved irrigation practices – for example, not irrigating above DUL and judicious moisture monitoring – can improve both irrigation water-use efficiency and crop water-use efficiency by minimising deep drainage, runoff and soil evaporation and by maximising transpiration. This, in turn, leads to improved crop productivity.

9.4 Soil Water Balance for Crop Types and Land Use Patterns under 52 Years of Fixed, Repetitive Cropping

Previous studies estimated the soil water balance under various cropping sequences (Section 9.1) and irrigation practices (Section 9.3) for up to 14 years. The studied period had detailed information about actual cropping sequences and management practices and thereby represented the soil water balance for the most probable growing conditions for the Lockyer Valley. However, the interpretation of deep drainage results from these studies to large areas is difficult due to the lack of continuous crop type and land use information that covers the entire Lockyer Valley. Given this limitation and the sensitivity of modelling to the assumptions on irrigation strategy and cropping sequence, estimation of deep drainage to develop deep drainage maps for the entire Lockyer Valley required simplification and further modelling.

Therefore, the current study estimated deep drainage and irrigation requirements for various crop types and land use patterns under a wide range of climate conditions, ignoring modelling crop rotation in the one-dimensional soil profiles. Inter-year variability in the climate conditions was accounted for by following fixed and repetitive crop types and land use patterns using climate data from 1960 to 2011. Long-term average rates of deep drainage and irrigation demand for the entire Lockyer Valley are therefore referred to the assumption that the land use did not change dramatically during this period. These land use changes were analysed using remote sensing from 1990 to 2011 (as documented in Chapter 9).

Soil water balance modelling for fixed, repetitive crop types and land use patterns included average, above average and below average climate conditions. Therefore, the current study provided data with ease for deep drainage mapping and groundwater modelling and also improved the confidence in modelled output. The current study should, however, be considered as a standalone from the two previous studies due to the way the modelling was carried out.

9.4.1 Methodology

9.4.1.1 Simulation, Climate and Soil

The HowLeaky model (Rattray *et al.*, 2004) was used to estimate soil water balances. The study was for a period of 52 years from 1960 to 2011. Simulations on crop types and land use patterns were repeated every year to account for the effect of climate conditions. The climate data were for Gatton DPI. Average rainfall for the study period was 801 mm/a. The soil was a Black Vertosol (Isbell, 1996). The soil parameters are presented in Table 20.

The simulations on crop types and land use patterns (Table 24) draw remote sensing information. The simulations ignore crop rotation sequence that comprises more than three crops. Fallow period is assumed to be absent or minimum between two crops. The simulations also assume that the soil moisture in the top 1.8 m soil depth will adapt reasonably fast to the changes from one crop type to the other. On a larger spatial scale, the average deep drainage rates for the Lockyer Valley are the result of the sum of the deep drainage rate for each crop type or land use pattern multiplied by the percentage of the respective crop type or land use pattern on the remote sensing pictures (detailed in Chapter 9).

9.4.1.2 Vegetation and Fallow

The cropping duration and other vegetation parameters are similar to those studied earlier (Section 9.2.1.1). The soil water balance data for forage and cereal simulation were derived from summer-sown sorghum and winter-sown wheat, whereas the data for vegetable simulation were from summer-sown sweet corn, autumn-planted broccoli and winter/spring-sown bean. Both simulations had minimum fallow period (e.g. < a month) before or after the harvest of each crop in a year. This minimised the contribution of fallow phase on deep drainage (the soil water balance for fallow phase is separately simulated as year-round fallow). Year-round fallow simulation assumed land cultivation at the beginning of summer every year and the SWD was reset to a predefined value at this time point. Perennial pasture and grazing conditions were represented respectively by lucerne and a deep-rooted grass (or pasture) species (root length ~1.8 m) both grown year-round without fallow period.

9.4.1.3 Irrigation

Perennial pasture and forage and cereal were deficit irrigated to a fixed depth of 60 mm at 140 mm SWD (wherever irrigated). All vegetable crops were ideally irrigated to DUL at 25 mm SWD. Further information related to modelling parameterisation is given in Section 9.2.1.1.

9.4.2 Results: Effects of Crop Types, Land Use Patterns and Climate Conditions on Soil Water Balance

As shown in Table 24, deep drainage was lowest under perennial pasture and grazing and highest under year-round fallow and vegetables simulations. Results show that deep drainage was related to crop type and irrigation strategies as also seen previously (Section 9.1 and Section 9.3) and on fallow duration. As the simulations varied in irrigation strategy (which accounted for the differences among crop types in the factors such as crop's economic value and crop's tolerance to moisture deficit), irrigation requirement was also higher for vegetables (ideally irrigated) than perennial pasture, forage and cereal (deficit irrigated). The deep drainage for perennial pasture (unirrigated) and grazing (unirrigated) was simular due to similar vegetation characteristics (e.g. rooting depth, growing duration).

Table 24: Estimates on deep drainage and irrigation demand for various crop types and land use patterns from 1960 to 2011 for a Black Vertosol in the Lockyer Valley.

Crop/Land Use Pattern	Deep Drainage (mm/a)	Irrigation (mm/a)
Perennial pasture (no irrigation)	8	-
Grazing (no irrigation)	11	-
Perennial pasture (deficit irrigation)	11	293
Forage and cereal (no irrigation)	19	-
Forage and cereal (deficit irrigation)	34	216
Year-round fallow (no irrigation)	169	-
Vegetables (ideal irrigation)	225	418

The current study provides an indication of long-term average deep drainage and irrigation demand for individual crop type and land use pattern. Combining these data with the remote sensing land use maps will contribute to the integration and interpolation of the soil water balance data at different time points for larger areas of the Lockyer Valley (detailed in counterpart studies).

9.5 Summary

The Lockyer Valley in South East Queensland is a valuable food production area with vegetable, grain and pasture crops. However, intensive irrigated cropping and drought have led to the depletion of groundwater resources in the past and have impacted on sustainable crop production. A purified recycled water (PRW) scheme has been proposed to bring extra water into the Lockyer Valley. This could act as an adjunct to groundwater via surface supply or by recharging the aquifers. However, the potential mobilisation of solutes due to likely changes in the soil water balance and their impacts on the underlying groundwater system are not known. An understanding of changes in the soil water balance is therefore of prime importance. The current study estimates soil water balances, in particular deep drainage, which in turn helps to estimate groundwater recharge and mobilisation of salts. Deep drainage could be a significant risk with regard to soil salinity in this region.

Soil water balance was estimated using the HowLeaky model for a period up to 14 years for various cropping sequences and irrigation application levels and for 52 years for various crop types and land use patterns. Estimates were for a Black Vertosol. Studies on cropping sequences show a deep drainage of 11 mm/a and transpiration of 703 mm/a in relation to irrigation of 326 mm/a for a forage-grain-pasture cropping sequence. In contrast, a vegetable-based cropping sequence had a deep drainage of 121 mm/a and transpiration of 602 mm/a in relation to irrigation of 598 mm/a. Lower deep drainage and higher transpiration with the forage-grain-pasture cropping sequence was related to the inclusion of deep-rooted plants such as sorghum and lucerne. Results suggest that soil moisture must be effectively exploited to minimise deep drainage and maximise transpiration.

Irrigation application levels also had an impact on the soil water balance. An irrigation application level that refilled the soil up to drained upper limit (DUL) resulted in deep drainage of 84 mm/a and transpiration of 673 mm/a with irrigation of 530 mm/a. However, as the irrigation amount increased, for example, when irrigated up to 50% above DUL, irrigation and deep drainage increased to 717 mm/a and 262 mm/a respectively, but transpiration remained similar. That is, as the irrigation increased, the contribution of irrigation (or rainfall) to deep drainage increased (from 7% to 19% of the total water balance) but the contribution to transpiration decreased (from 55% to 48%). This suggests that irrigation beyond DUL leads mainly to deep drainage. Soil water balance studies for crop types and land use patterns show deep drainage of 8, 11, 169 and 225 mm/a respectively for lucerne (unirrigated), grazing, year-round fallow and vegetables (irrigated). Deep drainage was higher for shallow-rooted crops and under irrigated cropping.

To sum up, the current study shows the effects of cropping sequence, crop characteristics (e.g. rooting depth, green cover, tolerance to moisture-limited conditions), fallow duration and irrigation application levels on the soil water balance. Under best management practices and year-round cropping involving minimal fallow period, the order of deep drainage was found to be unirrigated perennial pasture = (unirrigated) grazing < deficit-irrigated perennial pasture < deficit-irrigated forage and cereal < year-round fallow < well-irrigated vegetables. The study also highlights that deep drainage can be minimised through the selective incorporation of deep-rooting, drought-tolerant crops such as sorghum and lucerne to the farming systems and by adopting best irrigation strategies (e.g. irrigating to DUL with judicious soil moisture monitoring).

10. SALT FLUX MODELLING IN THICK UNSATURATED SOIL COLUMNS UNDER IRRIGATION WITH PURIFIED RECYCLED WATER

(David Rassam, Jenny Foley, Leif Wolf)

Salinity poses a serious threat to agricultural land, with salts being introduced to the landscape via rainfall and irrigation. The latter may exacerbate the damage when the irrigation water used is highly saline. Salts that accumulate in the unsaturated zone eventually get mobilised and transported to the groundwater system, thus posing a further threat to this important resource. Silburn and colleagues (2008) stated that loss of chloride and an increase in deep drainage under cropping was found across the agricultural lands of Queensland. Deep drainage varies significantly from as low as 1–20 mm/yr for dryland cropping to values as large as 100–200 mm/yr for furrow-irrigated annual cropping (Silburn *et al.*, 2008; Tolmie *et al.*, 2011).

The major expansion in irrigation development in the Lockyer Valley started in 1940 with the groundwater as the main resource (White, 1980). Shaw (1979) identified 25 major outbreaks of salinity where vegetation growth had been retarded by salinity. The salinity was attributed to prolonged irrigation with saline groundwater (Gardner, 1985; Bruce, R, Section 5.2 in White, 1980). In some areas, the increased salinity has reduced the crop types to only those that are salt tolerant (Brough *et al.*, 2008).

10.1 Objectives

This section addresses the issue of salinity in the Lockyer Valley and aims to achieve the following:

- Estimate the travel time of the existing salt peak to the groundwater table
- Estimate the average annual salt flux to the groundwater table
- Assess temporal variations in the salt mass balance within the soil profile.

Two sites were chosen to test the effects of the following factors:

- Effect of vegetation type
- Effect of irrigation intensity
- Effect of irrigation water quality
- Effect of replacing current irrigation water with purified recycled water (PRW).

10.2 Methods

A numerical modelling approach was adopted to simulate salt transport in the unsaturated zone of two selected sites of the Lockyer Valley, namely, Forest Hill and Tent Hill. Those two sites were selected as they had significant amounts of salt stored in their unsaturated zone, which extends to about 20 m. HYDRUS-1D of Šimůnek and colleagues (2008) was used to model water flow, solute transport, and root water uptake in those sites. Details of the modelling methodology are discussed in the following sub-sections.

10.2.1 Input Data and Model Parameters

10.2.1.1 Rainfall and Irrigation Data

Rainfall and potential evapotranspiration data were obtained from the Queensland Government Department of Environment and Resource Management (DERM). Two irrigation scenarios were adopted: moderate and intensive irrigation. The total combined water input (rainfall + irrigation) annual average for the two scenarios were 1294 mm/yr and 1405 mm/yr, respectively. Figure 89 and Figure 90 show the daily data for a 5-year period. The irrigation demand was derived from the HowLeaky modelling conducted by DERM (detailed in Chapter 9). Based on samples tested by DERM, the input chloride concentration for the irrigation water for the Forest Hill and Tent Hill sites were 62 ppm and 167 ppm, respectively. Chloride concentration of the PRW was 25 ppm, and in rainwater chloride was 1.5 ppm.

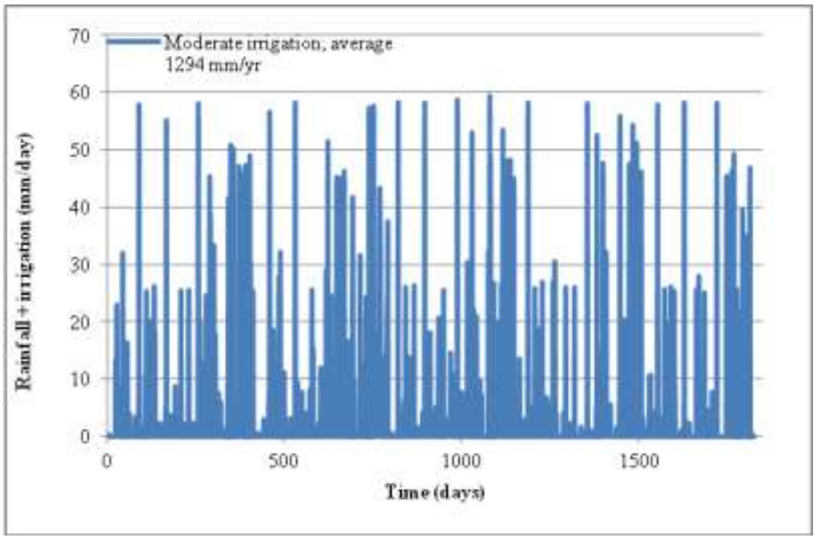


Figure 89: Daily combined water input (rainfall + moderate irrigation).

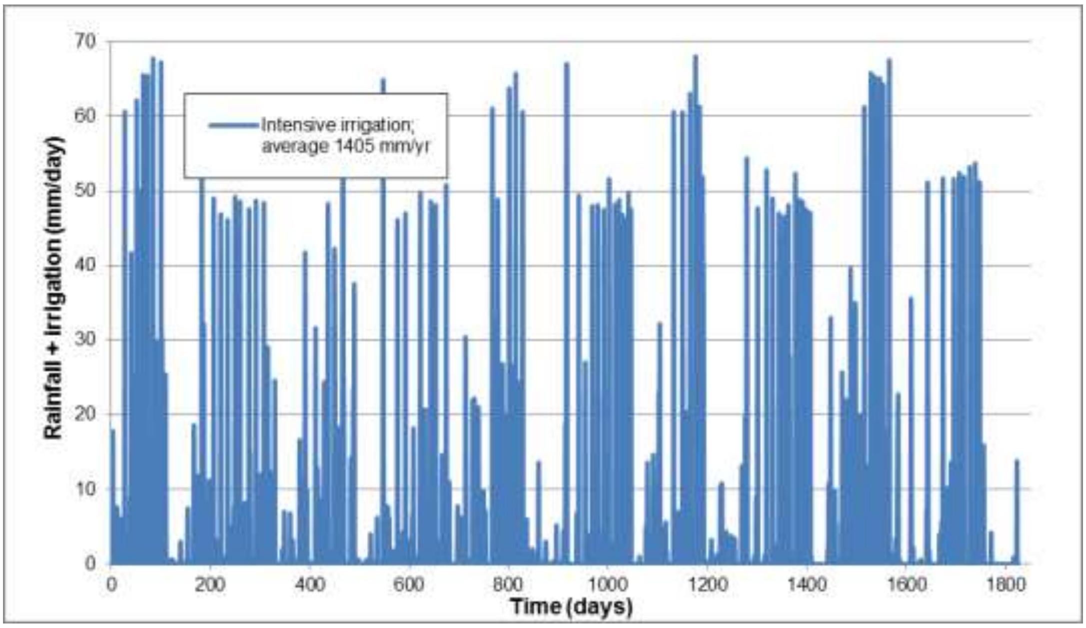


Figure 90: Daily combined water input (rainfall + intensive irrigation).

10.2.1.2 Soil Properties and Salt Profiles

Soil properties for the two sites were derived from the bore log data provided by DERM (refer to Chapter 5), which include particle size distribution (sand, silt, and clay fractions) and bulk density. After close examination of the bore log data, it was decided that five and four soil types would be satisfactory to describe the variability in Forest Hill and Tent Hill, respectively (refer to Tables 25 and 26, respectively; the soil type numbers are used by the HYDRUS-1D software to define heterogeneity along the profile). These data were incorporated into the Rosetta software (Schaap *et al.*, 2001) to derive the hydraulic parameters for the two soil profiles at Forest Hill and Tent Hill (see Tables 25 and 26), residual water content Qr, saturated water content Qs, the van Genuchten (1980) parameters Alpha and 'n' and saturated hydraulic conductivity Ks. A sample soil-type distribution for Forest Hill is shown in Figure 91A. Initial chloride concentrations in the soil profiles were derived from the 1998 bore log data provided by DERM (refer to Chapter 5); a sample salt distribution for Forest Hill is shown in Figure 91B.

Table 25: Soil hydraulic parameters for Forest Hill.

Soil Type Numbers	Qr	Qs	Alpha (cm^{-1})	n	Ks (cm/day)
1	0.0985	0.4702	0.0198	1.2613	13
2	0.0796	0.4465	0.0226	1.3331	31
3	0.0510	0.3103	0.0321	1.5772	31
4	0.0866	0.4503	0.0176	1.3408	14
5	0.0619	0.3782	0.0265	1.3228	18

Table 26: Soil hydraulic parameters for Tent Hill.

Soil Type Numbers	Qr	Qs	Alpha (cm^{-1})	n	Ks (cm/day)
1	0.0907	0.4667	0.0162	1.3600	16
2	0.0757	0.4471	0.0165	1.4106	22
3	0.0491	0.4246	0.0272	1.4736	74
4	0.0580	0.4316	0.0218	1.4489	50

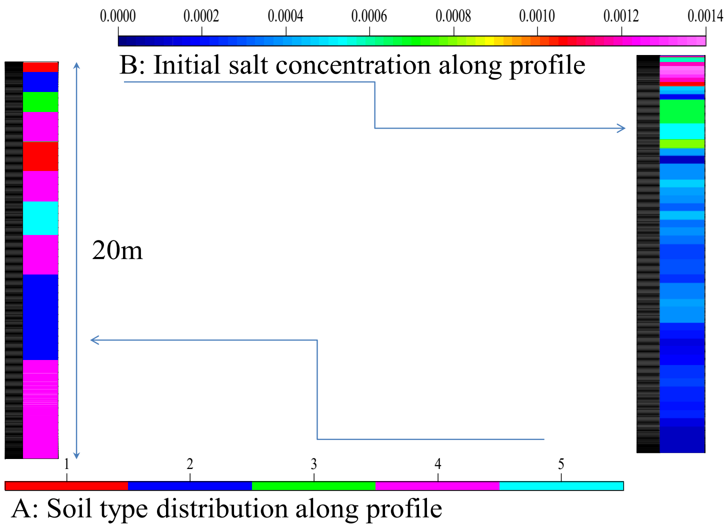


Figure 91: (A) Material distribution and (B) initial salt concentration gm/cm³ for Forest Hill; material numbers correspond to those listed in Table 25.

10.2.1.3 Vegetation Type and Root Water Uptake

HYDRUS-1D adopts the Feddes and colleagues (1978) method to model root water uptake in the unsaturated zone. Water uptake is assumed to be zero close to saturation (i.e. wetter than some arbitrary 'anaerobiosis point' P_0). Root water uptake is also zero for pressure heads less than the wilting point (P_3). Water uptake is considered optimal between pressure heads P_{opt} and P_2, whereas for pressure heads between P_2 and P_3 (or P_0 and P_{opt}) water uptake decreases (or increases) linearly with pressure head. P_{2H} and P_{2L} are values of the limiting pressure head, below which roots can no longer extract water at the maximum rate, (for upper and lower limits of potential transpiration rates, respectively).

The root water uptake parameters used were obtained from the built-in HYDRUS-1D library and are listed in Table 27. For the shallow-rooted and deep-rooted scenarios, the root depth used was 600 mm and 1500 mm, respectively.

Table 27: Root water uptake parameters.

	P_0 (cm)	P_{opt} (cm)	P_{2H} (cm)	P_{2L} (cm)	P_3 (cm)
Shallow roots (D-series)	-10	-25	-750	-2000	-8000
Deep roots (W-series)	-10	-25	-1500	-1500	-8000

10.2.2 Design of Modelling Exercise

The one-dimensional HYDUS-1D model was used to simulate water flow, solute transport, and root water uptake in two Lockyer Valley sites, namely, Forest Hill and Tent Hill. The 20 m deep unsaturated zone was discretised into 1000 elements with finer discretisations near the upper and lower ends of the flow domain. The surface is an atmospheric (variable flux) boundary that allows the infiltration of rainfall and irrigation water, whereas the lower end is a free drainage boundary that assumes unit-gradient flow. Simulation time was varied depending on the irrigation scenario, which was 100 years and 28 years for moderate and intensive irrigation scenarios, respectively. Those periods were long enough to allow the salt peak to arrive at the groundwater table for the shallow-rooted vegetation type.

Table 28 summarises the modelling experiment having a total of twelve simulations, six for each site. Two irrigation options were considered, the first using the currently available water at each of the sites, and the second using PRW. For each of those options (1 and 2), two vegetation types were considered: shallow-rooted (series W) and deep-rooted (series D). An additional scenario was considered for the shallow-rooted vegetation where intensive irrigation was adopted; this option represented a worst-case scenario for salt mobilisation and transport.

Table 28: Details of modelling exercise (FH=Forest Hill; TH=Tent Hill; series W is normally irrigated shallow-rooted vegetation; series D is normally irrigated deep-rooted vegetation; series WW is intensely irrigated shallow-rooted vegetation.

		Forest Hill (FH)	Tent Hill (TH)
Initial salt content based on 1998 profiles		High	Moderate
Current Irrigation water quality		Moderate salinity	High salinity
Irrigation with currently available water (Option 1)	Shallow-rooted Moderate irrigation 100-yr simulation (Series W)	FH-W1	TH-W1
Irrigation with currently available water (Option 1)	Deep-rooted Moderate irrigation 100-yr simulation (Series D)	FH -D1	TH-D1
Irrigation with currently available water (Option 1)	Shallow-rooted Intensive irrigation 28-yr simulation (Series WW)	FH -WW1	TH-WW1
Irrigation with PRW (Option 2)	Shallow-rooted Moderate irrigation 100-yr simulation (Series W)	FH -W2	TH-W2
Irrigation with PRW (Option 2)	Deep-rooted Moderate irrigation 100-yr simulation (Series D)	FH -D2	TH-D2
Irrigation with PRW (Option 2)	Shallow-rooted Intensive irrigation 28-yr simulation (Series WW)	FH -WW2	TH-WW2

10.3 Results

Figure 92 shows the salt profiles for Forest Hill at various times. The initial 1998 salt peak, which was located at a depth of about 2 m, disperses as it moves slowly down the profile. For Forest Hill (based on estimates from Figure 92), the salt peak travels at an average downward rate of 250 mm/yr; this rate is directly correlated to the recharge rate Table 29 shows that this correlation is valid for all the simulations. Hence, deep-rooted vegetation that results in lower deep drainage (and recharge) rates results in much slower advancement of the salt peak (series D). On the other hand, intensive irrigation results in higher recharge rates and a faster advance of the salt peak for both sites (Table 29, compare results of series WW with series W).

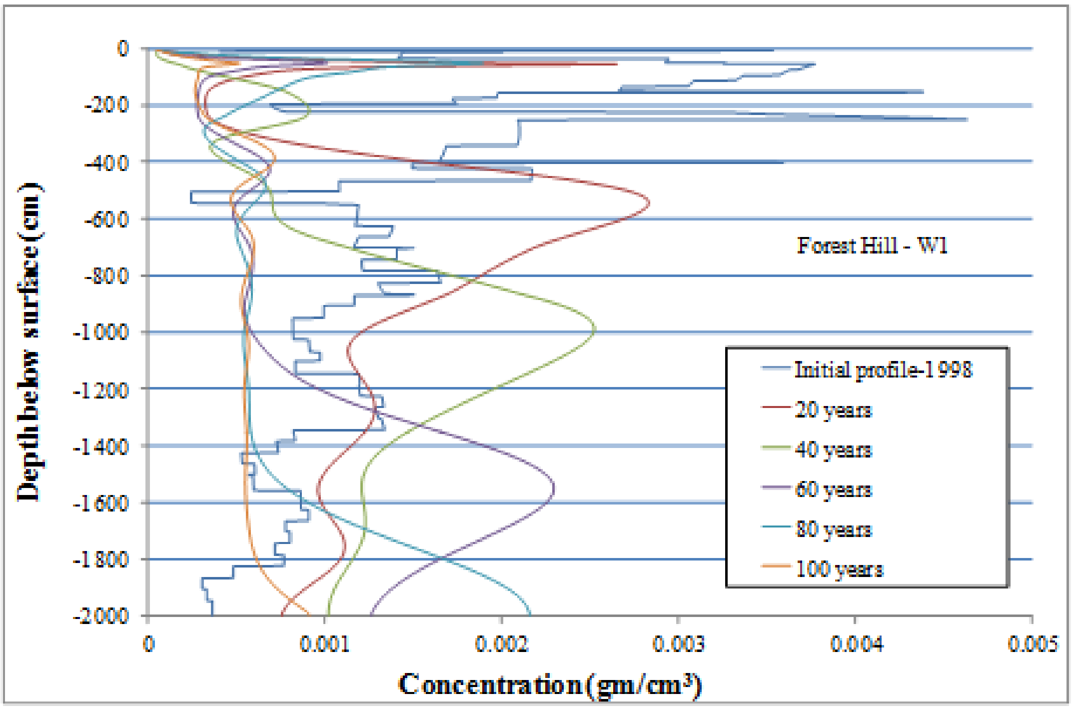


Figure 92: Chloride concentration profiles at various times; FH-W1.

The cumulative salt flux at the groundwater table (lower end of the flow domain) is shown for Forest Hill in Figure 93. The total salt flux for the shallow-rooted vegetation (FH-W1 and FH-WW1) is very similar; however, the rate is much higher for the intensive irrigation scenario (WW1). For the deep-rooted vegetation, the total salt flux is less than half that for the shallow-rooted case. Table 29 shows a comparison of the average annual flux for all the simulations; they range from 0.2–0.4 t/ha/yr for deep-rooted vegetation, 0.6–0.8 t/ha/yr for shallow-rooted vegetation, and 2–3 t/ha/yr for intensively irrigated shallow-rooted vegetation.

Table 29: Results of the modelling exercise (current = existing irrigation water; PRW = irrigation with purified recycled water).

			Average Recharge Rate (mm/yr)	Average Advance Rate of Salt Peak (mm/yr)	Average Salt Flux (t/ha/yr)	% Salt Mass Remaining in Profile
Forest Hill	FH-D1	Current	37	130	0.42	105 (100 yr)
	FH -D2	PRW			0.40	62 (100 yr)
	FH -W1	Current	68	250	0.81	43 (100 yr)
	FH -W2	PRW			0.78	19 (100 yr)
	FH -WW1	Current	270	910	2.92	13 (28 yr)
	FH -WW2	PRW			2.84	5 (28 yr)
Tent Hill	TH-D1	Current	24	80	0.20	236 (100 yr)
	TH-D2	PRW			0.20	66 (100 yr)
	TH-W1	Current	54	231	0.67	162 (100 yr)
	TH-W2	PRW			0.55	27 (100 yr)
	TH-WW1	Current	237	900	2.27	44 (28 yr)
	TH-WW2	PRW			2.00	7 (28 yr)

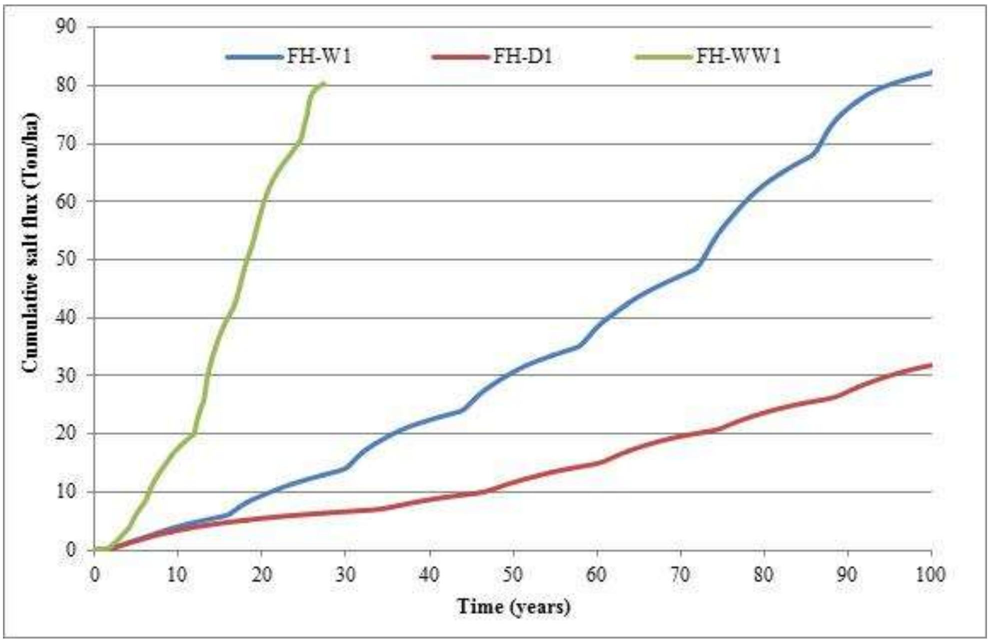


Figure 93: Cumulative salt fluxes (Ton/ha/yr) for Forest Hill.

Figure 94, Figure 95 and Figure 96 show the trends of salt-mass balance for Tent Hill. For shallow-rooted vegetation (series W1, Figure 94), the highly saline irrigation water results in increasing the salt mass stored over the entire simulation period; the figure illustrates the advantages of switching to PRW as an alternative for the currently available irrigation water. With PRW the salt mass in the profile continuously declines to about 27% of the initial mass.

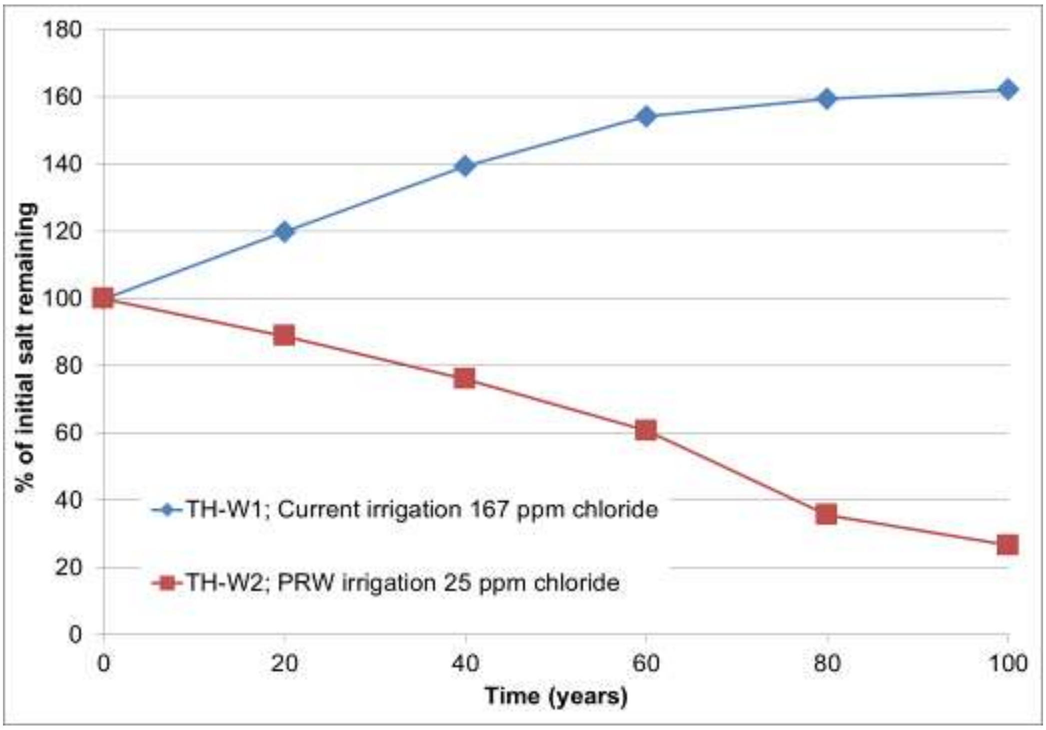


Figure 94: Salt mass balance for Tent Hill, shallow-rooted; effect of applying PRW.

Figure 95 shows that intense irrigation results in a quick flushing of the initial salts in the profile. However, for the case of the highly saline irrigation water, the salt mass seems to be approaching a quasi-equilibrium state of about 44% of the initial salt mass. Again, the advantages of using the PRW are obvious as the salt mass decreases to 7% of the initial mass.

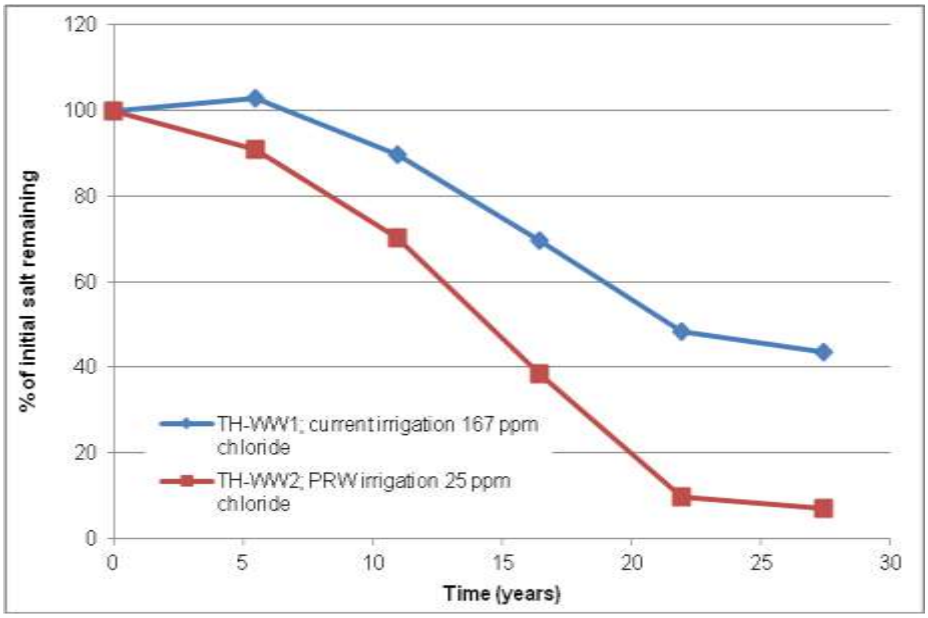


Figure 95: Salt mass balance for Tent Hill, shallow-rooted intensely irrigated; effect of applying PRW.

Figure 96 illustrates the slow change in salt mass balance due to the slow movements of the salts, which is a result of the low recharge rate (for the case of deep-rooted vegetation). Continuing to use the currently available water results in a huge build-up of salts within the profile (236% of initial); note that simulation TH-W1 (Figure 94) has a final salt mass balance of 162%; this is due to the higher water extraction capacity of deep-rooted vegetation (thus leaving more salt behind). The PRW is still advantageous but requires a much longer period to wash the salt out due to the low recharge rate. The salt mass balance results of all the simulations are listed in Table 29.

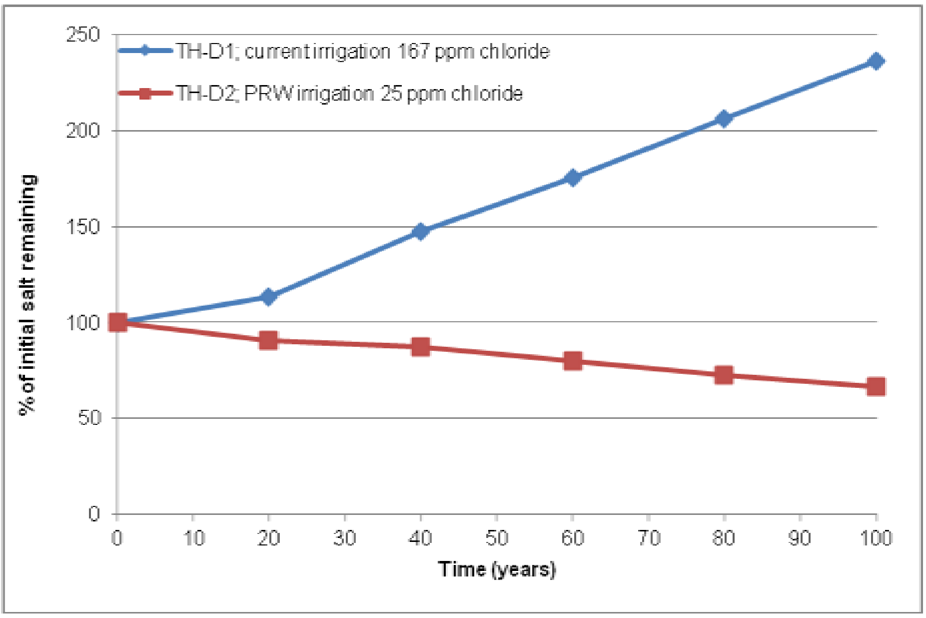


Figure 96: Salt mass balance for Tent Hill, deep-rooted; effect of applying PRW.

10.4 Conclusions

Based on the modelling results, the following can be concluded.

- The average advance rate of the salt peak ranges from 80–130 mm/yr for moderately irrigated deep-rooted vegetation, is 240 mm/yr for moderately irrigated shallow-rooted vegetation, and is 900 mm/yr for intensively irrigated shallow-rooted vegetation. This is an average velocity for matrix flow and does not address the possibility preferential flow phenomena, which may result in faster travel times.
- For a 20 m depth to the groundwater table, the timing of the salt-peak breakthrough varies from 22 years for intensively irrigated shallow-rooted crops to over 200 years for moderately irrigated deep-rooted crops.
- Under current irrigation practices (e.g. using chloride concentrations of 62 and 167 ppm), a long-term salt accumulation in the soil profile is predicted.
- Switching to PRW is mostly beneficial in areas where current irrigation water is highly saline.
- If irrigation with PRW is not administered at higher than current rates then PRW will not lead to an increase of salt fluxes to the groundwater table.
- The benefits of switching to PRW (flushing out existing salts) would be realised much earlier in intensively irrigated land. However, in such areas, this would mean an early breakthrough of the stored salt to the groundwater system.
- The average annual salt flux to the groundwater system ranges from 0.2–0.4 t/ha/yr for moderately irrigated deep-rooted vegetation, 0.6–0.8 t/ha/yr for moderately irrigated shallow-rooted vegetation, and 2–3 t/ha/yr for intensively irrigated shallow-rooted vegetation.

11. MAPPING LARGE SCALE WATER USE CHANGES AND GROUNDWATER RECHARGE IN THE LOCKYER VALLEY

(Manuel Grimm, Tim Ellis, Mick Hartcher, Malcolm Hodgen, Leif Wolf)

11.1 Introduction

In the past, high use of groundwater during dry years has resulted in water shortages in the Lockyer Valley. Due to aquifer depletion, the Queensland Government has constructed a network of artificial recharge works and some storages to provide surface water irrigation (i.e. Atkinson’s Dam, Bill Gun Dam and Lake Clarendon), but the majority of the irrigation water is still sourced from groundwater, indicated by metered water use data from the Queensland Department of Environment and Resource Management (DERM). The drought from 2002 to 2007 forced farmers into limited irrigation amounts. According to Douglas Partners Pty Ltd (2007), an additional supply of purified recycled water (PRW) to the Lockyer Valley could at least prevent some of the crop failure during dry periods.

The goal of the study was to produce a deep drainage map for the Lockyer Valley model zone (see Figure 100) using the soil water 1D model HowLeaky (Rattray *et al.*, 2004). HowLeaky simulated deep drainage and irrigation demand estimations for different land use scenarios over time. Those estimations were distributed using land use maps, derived from Landsat images for different points in time, evaluated against observations of the growing crops. In order to validate the modelled data, several methods such as recharge estimations using groundwater fluctuation methods and maps of metered water use have been applied. In addition, the study was designed to show what kind of land use change could be possible by providing a certain amount of PRW to the Lockyer Valley farmers (Wolf and Moore, 2011) to augment irrigation from groundwater.

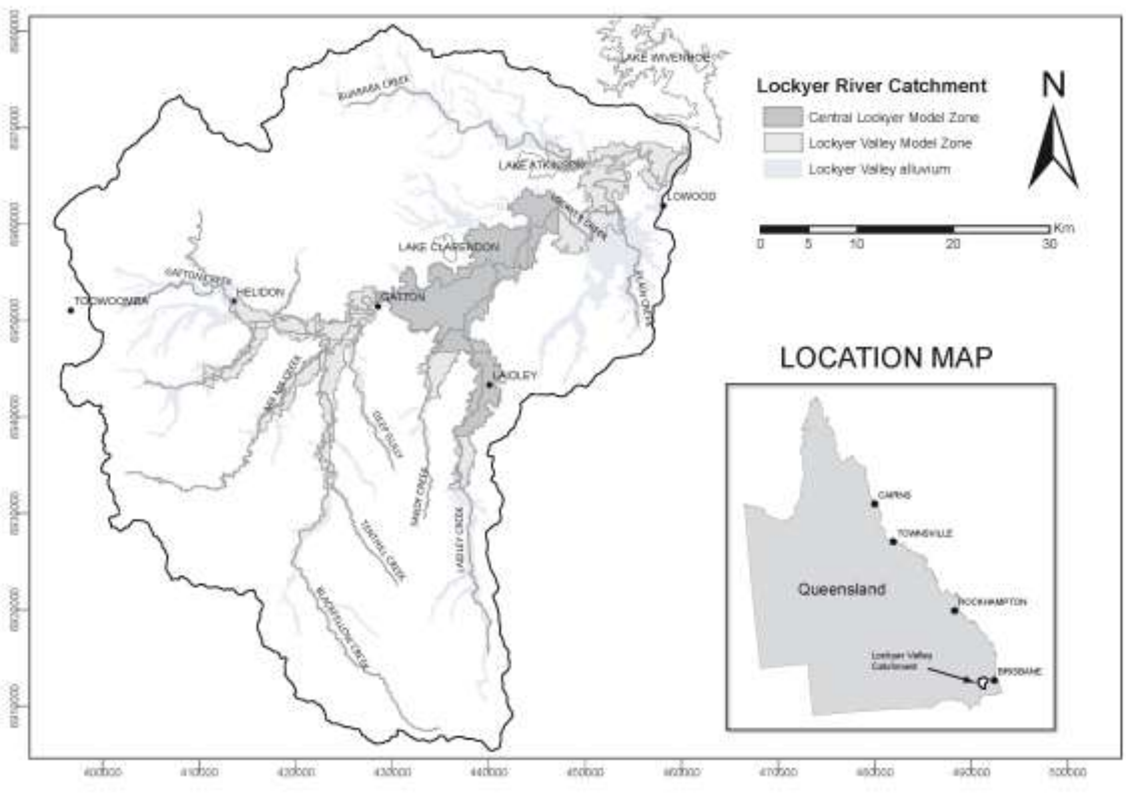


Figure 97: Lockyer Valley Catchment, with outlines of Alluvium, Lockyer Valley model zone and Central Lockyer.

Figure 97 shows the Lockyer River Catchment and the expansion of the alluvial plain. It also indicates areas of the Lockyer Valley model zone, and the Central Lockyer model zone, which define the study area.

The analysis of historic water use is a key component both for the calibration and validation of any numerical groundwater model and for the understanding the future water demand. Detailed data on water use were available for the central part of the Lockyer Valley since 1992.

As the Lockyer Valley area is only partially proclaimed as a sub-artesian water district, data on water use outside this proclaimed area are few. Within the proclaimed area, irrigation bores and surface water abstractions were metered. DERM studies also use terms 'benefited' and 'non-benefitted' to indicate areas which receive benefits from the operation of the major reservoirs and are, therefore, required to pay a fee of approximately $30/ML of water used. Water use data were available from DERM and, since 2000, also from the new bulk water provider SunWater. The proclaimed area represents almost exactly the Central Lockyer model zone as specified in Figure 97.

The quality of the water use data might vary over time. Bleakley (2011) stated that some inaccuracies developed with water use data in the Central Lockyer benefited area sometime after a private utility took over management of meters in June 2000. Spot field checks by DERM officers confirmed problems such as lack of routine maintenance and controls on meter set ups. This uncovered a number of dysfunctional meters and hence water use from these bores was generally under recorded. Bleakley (2011) identified from historical metered water use data that, prior to June 2000, the metered groundwater use in the non-benefited area was about 60% of the metered use in the benefited area. Given that DERM is responsible for maintaining water meters in an accurate state in the non-benefited area to the present day, we have more confidence in the accuracy of water meters in the non-benefited area (Bleakley, 2011) since that time. In order to provide a more accurate level of water use data, the metered water use in the benefited area was adjusted from 2002–2003 onwards based on the assumption that the average relationship between benefited and non-benefited area was the same as in the years before 2000.

Figure 98 shows the total adjusted water use and the rainfall in the Central Lockyer model zone and their inverse relationship to each other. In addition, there is a general trend of decreasing water use over the past 20 years, which probably can be explained by two causes: first, to higher efficiency irrigation methods that have been adopted; and second, to farmers having changed crop types (pers. comm. Craig Henderson, Principal Horticulturist, Department of Agriculture, Fisheries and Forestry, Gatton, Australia).

The information from the metering programs was used to generate maps of water use at different time steps (see Figure 101b) for validation of the soil water balance model HowLeakey as well as input to the numerical groundwater model.

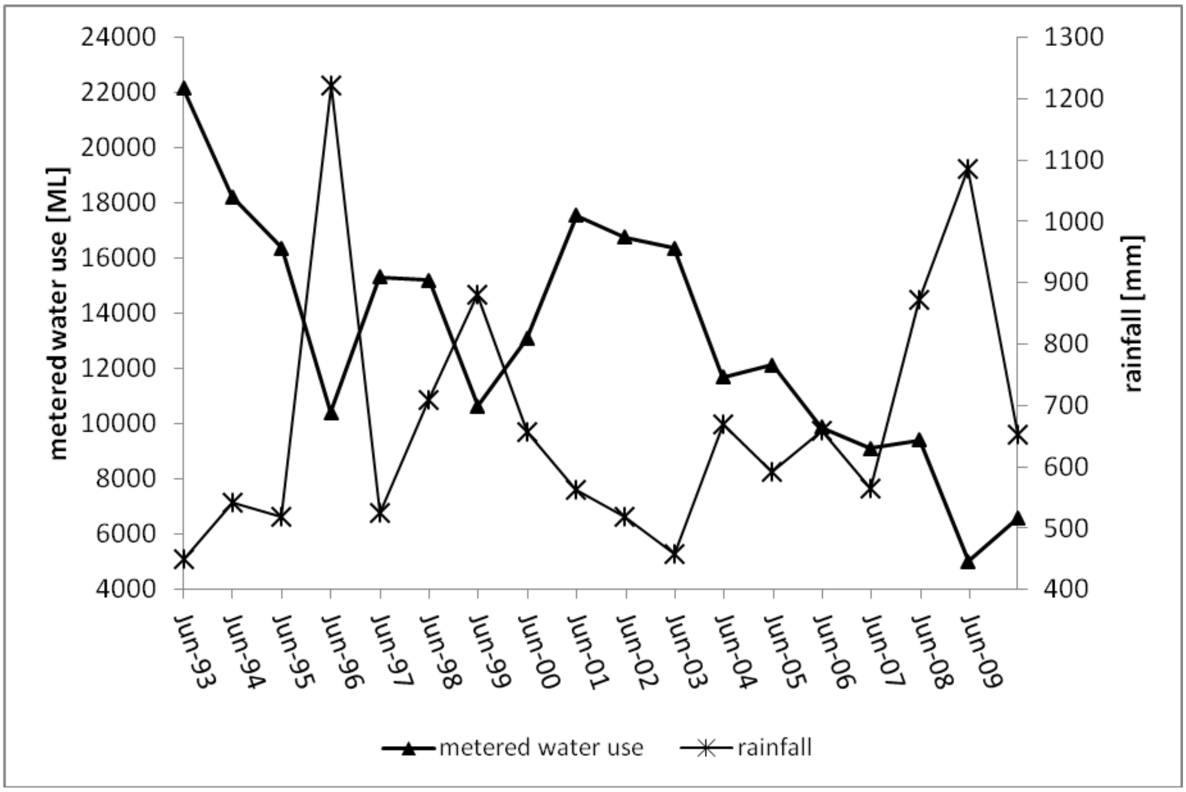


Figure 98: Total adjusted metered water use (groundwater use + surface water use) for the Central Lockyer model zone, data source: DERM.

11.2 Estimating Large Scale Groundwater Recharge using Water Table Fluctuation Measurements

Total groundwater recharge (diffuse + creek recharge) can be estimated with the groundwater fluctuation method. The method is based on volume calculations between the interpolated surface of groundwater levels and the base layer of the aquifer. This volume multiplied with the effective porosity of the aquifer material is the effective volume of water stored in the aquifer. The storage change is given by the difference of groundwater levels between year one and year two multiplied with an assumption on specific yield. From this storage change, we deduct the amount of groundwater pumping to arrive at the groundwater recharge.

For this method the aquifer had to be assumed as unconfined. Also lateral in- and out-flows could not be calculated in a transient way and were assumed as steady state. This exercise can be refined in future with the information of the newly calibrated groundwater model. Figure 99 shows a qualitative overview of possible groundwater flow in and out of Central Lockyer model zone.

The study of annually changing water levels resulted in groundwater recharge estimations between 1.7 and 25.5 GL/a for an effective porosity of 8%. The average groundwater recharge was around 10 GL/a (73 mm/a) for the Central Lockyer Model Zone for different effective porosities (see Table 30).

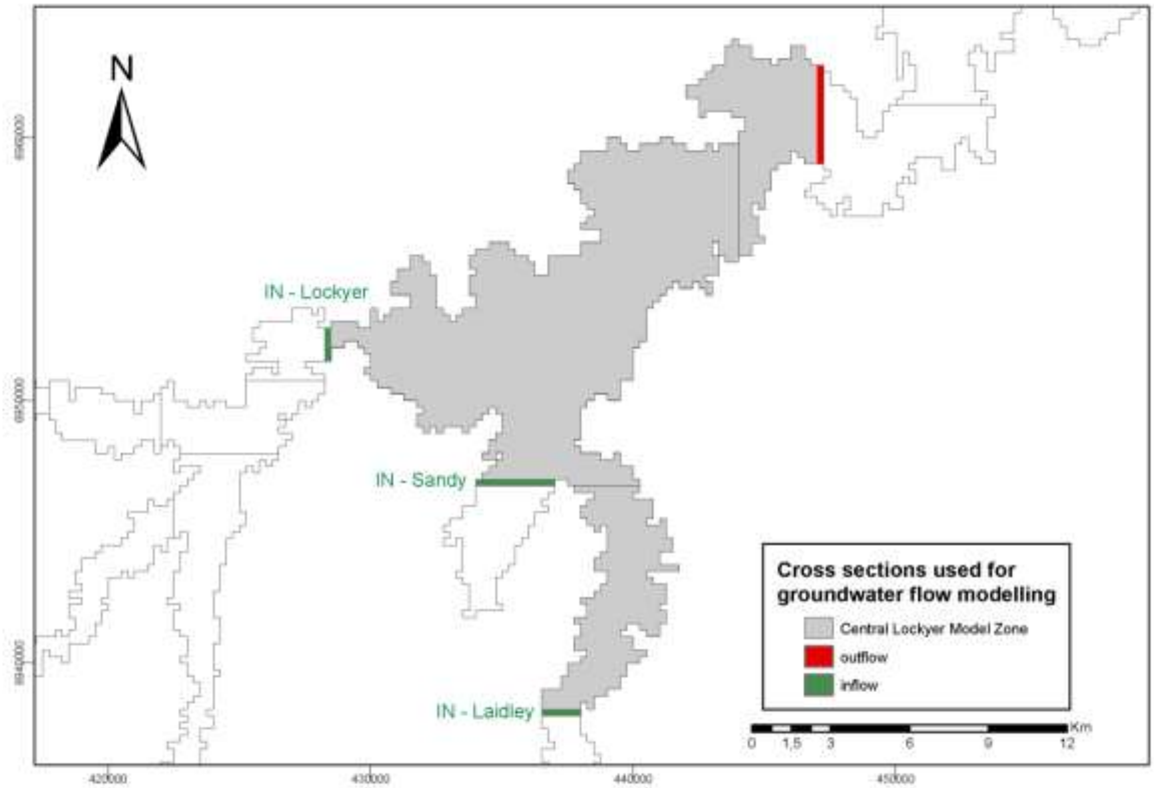


Figure 99: Cross sections where groundwater flow is expected.

Table 30: Storage change and estimated recharge for different specific yields calculated with the groundwater fluctuation method based on interpolated groundwater surfaces. Negative recharge estimations are adopted as zero for average calculation. All values are shown in ML.

		Specific Yield: 5 %		Specific Yield: 8 %		Specific Yield: 12 %		Specific Yield: 15 %	
Year	GW Use *	Stored Volume	Estimated Recharge	Stored Volume	Estimated Recharge	Stored Volume	Estimated Recharge	Stored Volume	Estimated Recharge
Jun 92		43347		69356		104033		130042	
Jun 93	19120	36561	12334	58497	8262	87746	2833	109682	0
Jun 94	18070	26358	7867	42172	1745	63258	0	79073	0
Jun 95	16274	19880	9797	31809	5910	47713	728	59641	0
Jun 96	8954	30177	19251	48283	25429	72425	33666	90531	39844
Jun 97	10817	32543	13183	52069	14602	78103	16495	97628	17915
Jun 98	13843	27856	9156	44570	6344	66855	2595	83569	0
Jun 99	9406	32015	13564	51223	16059	76835	19386	96044	21881
Jun 00	10513	32900	11398	52639	11929	78959	12637	98699	13168
Jun 01	16300	30502	13903	48804	12464	73206	10547	91507	9108
Jun 02	16316	22400	8213	35840	3351	53759	0	67199	0
Jun 03	16271	17236	11106	27577	8008	41365	3877	51707	778
Jun 04	11299	17276	11340	27642	11364	41463	11397	51829	11421
Jun 05	11717	15400	9842	24641	8716	36961	7216	46201	6090
Jun 06	8741	16040	9381	25664	9765	38496	10276	48120	10660
Jun 07	9048	13911	6920	22258	5643	33388	3940	41734	2663
Jun 08	5188	14745	6022	23592	6522	35388	7189	44235	7689
Jun 09	4999	21958	12212	35133	16540	52700	22310	65875	26638
Jun 10	6568	26677	11287	42683	14118	64025	17893	80031	20725
Average	**11858**		**10932**		**10376**		**10166**		**10477**

11.3 Mapping Deep Drainage and Irrigation Demand using Soil Water Balance Modelling

In recognition of the crop rotations which are described in Chapter 8 and which were considered in Chapter 9 for the soil water balance modelling, we employed a simplified approach to apply the 1-D model results to the entire Lockyer Valley alluvium. This approach is based on long-term static crop types (bare, grazing, lucerne, cereals and vegetables). The spatial distribution of deep drainage and irrigation demand is based on the existing land use maps. After comparing land use classifications from eight different Landsat scenes, the land use of September 2010 was selected as most representative of average or typical land use in the Lockyer Valley. Using the deep drainage rate calculations for five different land use types described in Chapter 10 and the land use classification from Chapter 9, a GIS map of deep drainage rates was constructed.

The average (1992–2011) deep drainage rate for the Lockyer Valley was calculated as 85 mm/a based on the land use classification for September 2010. In comparison, the application of other land use patterns, such as for dry conditions in October 2006 or wet conditions such as in October 2010, would both result in 91 mm/a deep drainage. This shows that differences in land use distribution have only a small influence on the average deep drainage. Still, we noted that the different assumptions on land use have a strong influence on spatial distribution of deep drainage. Therefore, long-term deep drainage maps must not be used to identify spot location patterns with high or low deep drainage rates. Figure 100 shows the spatial distribution of long-term average deep drainage for the Lockyer Valley for the land use classification September 2010 as an example.

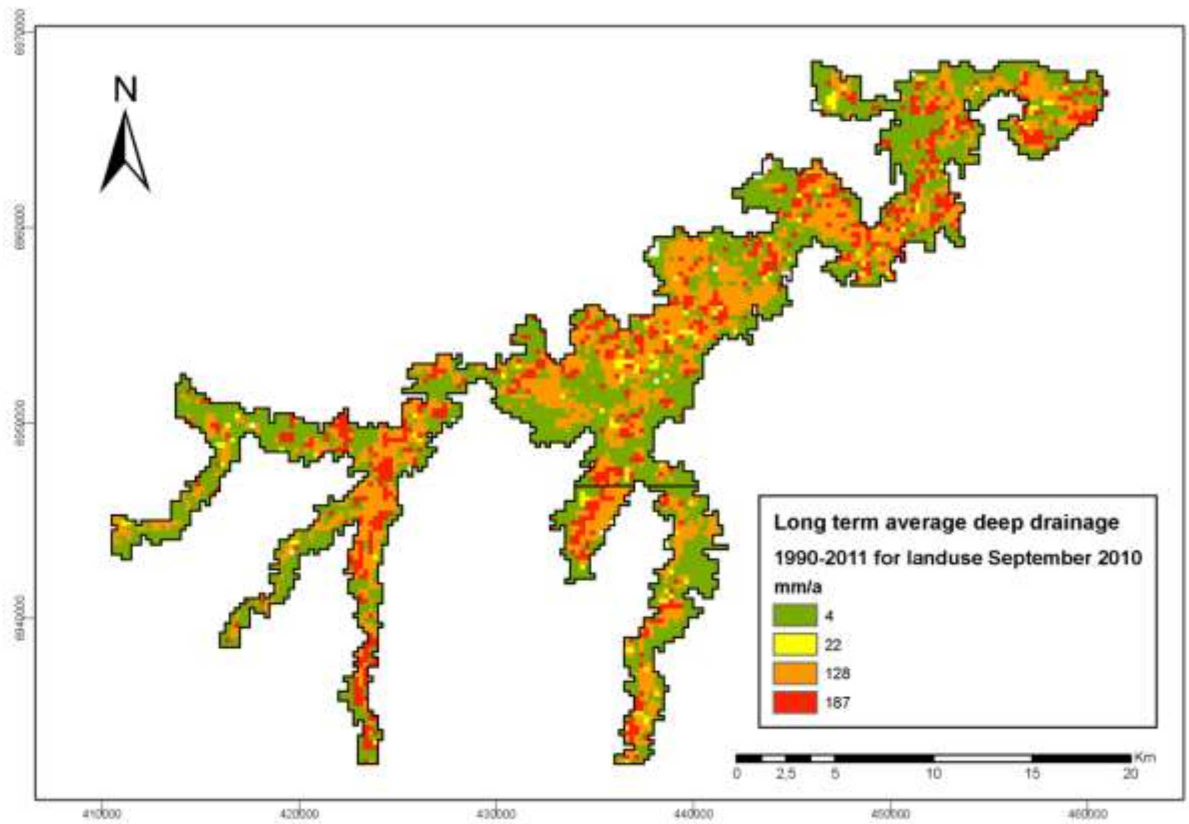


Figure 100: 52-year average deep drainage map based on the land use classification for September 2010.

In the same fashion as for the deep drainage, the irrigation demands calculated with HowLeaky were also applied to the area of the Lockyer Valley alluvium within a geographical information system.

Figure 101 shows a comparison of the spatial distribution of modelled long-term average irrigation demands and average unadjusted metered water use for Central Lockyer model zone. The spatial distribution of water use and the modelled irrigation demand show the expected similarities. The areas marked in red are irrigated with surface water from Lake Clarendon and Atkinson Dam but the abstraction rates were, unfortunately, not included in the DERM water use data. Highly irrigated areas occur mainly along the creeks, probably also indicating the good groundwater recharge and productive aquifer conditions.

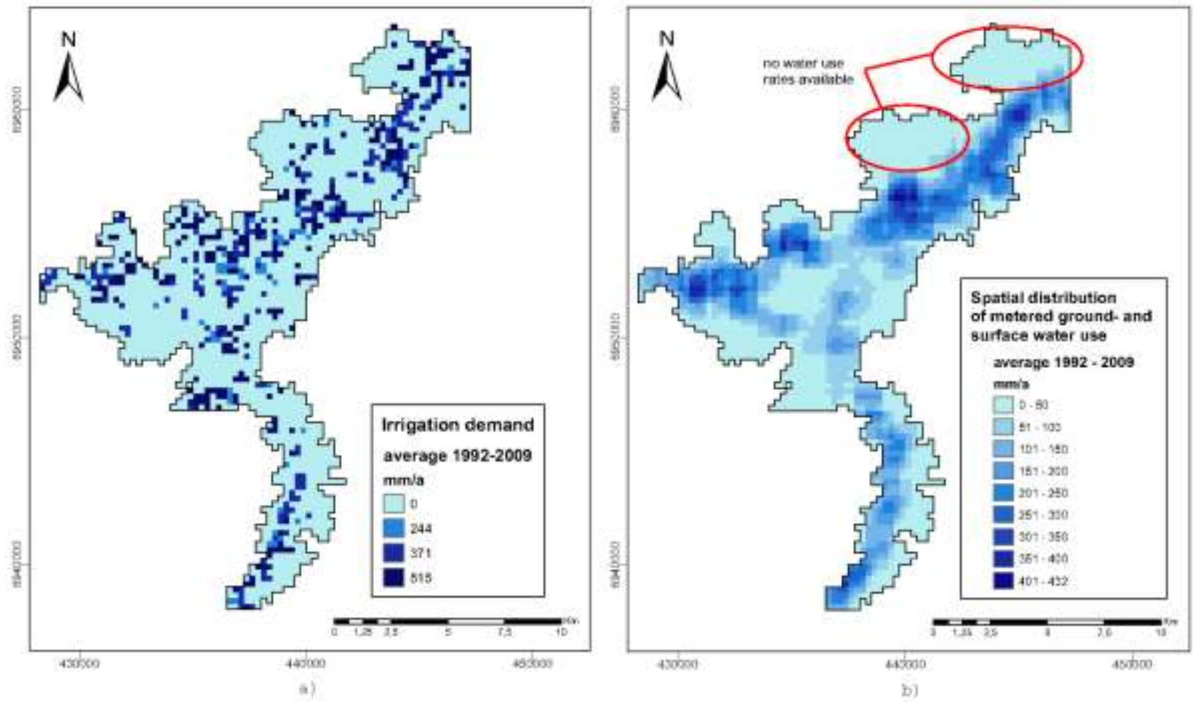


Figure 101: a) Spatial distribution of long-term average irrigation demands modelled with HowLeaky for the Central Lockyer model zone, b) Spatial distribution of average ground and surface water use for the Central Lockyer model zone.

Figure 102 shows a comparison of total adjusted metered water use and irrigation demand modelled with HowLeaky (based on land use September 2010) for the Central Lockyer model zone. The modelled irrigation demand is not following the decreasing trend of the actually metered water use. This could be on the one hand due to the fact that the model was not adjusting irrigation strategies over time and on the other hand due to not considering limited availability of irrigation water in the simulation (see Chapter 9). For example, during the drought from 2002 to 2007 the farmers were forced to irrigate their crops with water-saving irrigation strategies.

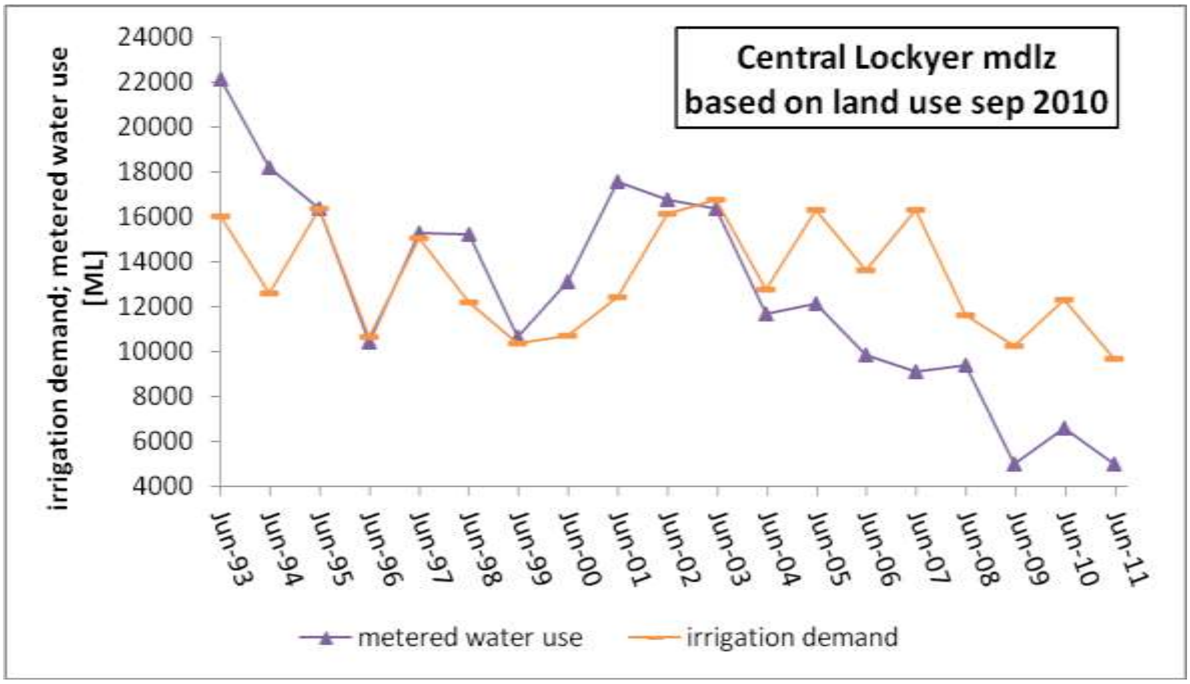


Figure 102: Adjusted metered water use and irrigation demand based on long-term static crop type modelling for the Central Lockyer model zone.

11.4 Conclusions

Estimations of groundwater recharge were calculated using the water table fluctuation method for the years 1992–2010 for the Central Lockyer model zone. The long-term average groundwater recharge of approximately 10 GL/a for the Central Lockyer is just barely influenced by the different specific yield assumptions, while the groundwater recharge calculations in individual years are very sensitive to the specific yield assumptions.

The groundwater fluctuation method has shown that for longer periods of dry conditions, such as from 2002 to 2008, the stored volume of water in the aquifer declines to approximately one-third (31 GL) compared to years with higher water levels (87 GL in 1992). As a consequence, at 50% of the bore sites in the Lockyer Valley and at almost all bore sites in the area of Ma Ma and Tenthill Creek, the saturated zone thickness contracted to less than 1 m. This means that further groundwater abstractions were almost impossible out of these bores. Land use maps show a reduction of irrigated areas by 56% in July 2006 compared to the average irrigated area in wet years (September 2010) (Chapter 8). Farmers were obliged during the drought to apply less water to their fields even after reducing the crop area (pers. comm. Ashley Bleakley, Principal Project Officer (Hydrology), Water Services South East).

Reduction in crop area and eventual crop failure can lead to substantial loss of income for farmers during dry periods. One mitigation option could be the supply of PRW to the Lockyer Valley as discussed in Chapter 1. A supply of 20 GL/a to the entire Lockyer Valley during dry periods would increase the total available water (rainfall + irrigation) for cultivated areas from 456 mm to 575 mm for the second half of 2006 (very dry). That is approximately as much as the available water in the second half of 1999 where the land use almost reached the average.

As detailed in Chapter 9, deep drainage and irrigation demand were calculated for selected soil profiles with the soil-water balance model HowLeaky. In order to utilise these spot information to construct maps for the entire valley, two different approaches were considered, the first termed 'crop rotation', the second 'static crop'. Within the first approach, the detail of crop rotations at a specific location was represented in the soil water balance modelling, thus providing a relatively accurate estimate for this specific spot (under the assumption that the actual crop history was well known). Then, based on the remote sensing land use information, the Lockyer Valley was classified into rotation schemes (e.g. wet, medium, dry) as explained in Chapter 8. The first deep drainage maps we developed for the Lockyer Valley were based on the results of a HowLeaky model for the crop rotation approach. However, a comparison with the actual metered irrigation water use and the modelled irrigation water demand uncovered major discrepancies and the need to re-parameterise the model. This, however, was not done due to resource constraints.

Instead, we followed a different approach: 'static crop'. In this, we run long time-series calculations for each crop type. Then we use those results and apply them to a detailed land use map from one point in time. Both methods have drawbacks; still, the latter has the advantage of generating a reasonably realistic time series with just a low spatial resolution.

The deep drainage rate, calculated for a representative land use distribution (September 2010), resulted in 85 mm/a. The application of other land use patterns, for example, for dry conditions in October 2006 or wet conditions such as in October 2010, only slightly influenced the long-term average deep drainage resulting in a deep drainage rate of 91 mm/a for both alternative land use classifications.

The average irrigation demand estimation (13,083 ML/a) matches the average metered water use (13,101 ML/a) for the twenty-year period, but appears to be systematically underestimated from 1992 to 2003 and generally overestimated from 2003 onwards (a mainly dry period). The average deep drainage estimation calculated with the static crop type approach comes up to 8573 ML/a (1992–2010), which appears too high in comparison to the total recharge of about 10,000 ML/a, calculated with the groundwater fluctuation method.

The assumption to use a representative land use distribution in order to generate a map of irrigation demands apparently leads to general overestimations during dry periods with a reduced cultivation of irrigated crops. In addition, HowLeaky does not consider improvements in irrigation strategies, which have reduced the water use over the past 20 years, or the actual availability of irrigation water.

In summary, it can be stated that the forward soil water balance modelling required validation and calibration data to constrain the modelling results. The availability of metered water use data is critical. Still, the model calculations consistently point towards a higher amount of deep drainage than previously assumed for the Lockyer Valley.

12. USING COUPLED MODEL SYSTEMS FOR THE ESTIMATION OF OPTIMUM WATER IMPORT VOLUMES FOR THE LOCKYER VALLEY

(Catherine Moore, Leif Wolf, Sreekanth Janardhanan, Ashley Bleakley, Jerome Arunakumaren)

12.1 Background

This chapter describes the development and calibration of a numerical groundwater flow simulation model for the Lockyer Valley alluvial aquifer. The calibrated model simulates the aquifer responses to a set of groundwater stresses including groundwater extraction, irrigation recharge, and flood recharge. The model is intended to serve as a predictive tool to assess management options for the groundwater resource, and specifically the option of importing Purified Recycled Water (PRW) into the Lockyer Valley. This has involved the estimation of the optimum volume of PRW to be imported into the Lockyer Valley to meet selected environmental targets, both under historical conditions, and over a series of climate change scenarios. The climate change scenarios considered are discussed in Chapter 13.

The Lockyer model is built on and is an extension of a pre-existing model developed by Kellog Brown and Root Pty Ltd to assess the sustainability of the water resource system in the Lockyer Valley (KBR, 2002). In this report, this model is henceforth referred to as the KBR model. The KBR model represents the Lockyer Valley groundwater system as a single-layer alluvial aquifer, and assumes that inflows to and discharge from the underlying sandstone aquifers of the valley are negligible when compared to the groundwater fluxes within the alluvial aquifer. The KBR model was calibrated using transient data for the time period 1991–2000, and hydraulic conductivity and specific yield were adjusted as calibration parameters. The authors of the KBR modelling report identified the key short-comings of the model as being caused by the relative inaccuracy of the digital elevation models (DEM) and the lack of reliable groundwater usage figures in some areas.

In the UWSRA work, the KBR model was revised and extended to address the identified shortcomings, and to make use of the additional aquifer monitoring undertaken since the previous modelling period which ended in 2000. In addition, the model was altered so that it was able to serve as a PRW predictive tool. Specifically, these alterations comprised:

- use of a new DEM with a finer resolution (Harris, 2012) and existing strata logs to redefine the the top and bottom elevations of the aquifer;
- inclusion of post-2000 groundwater level monitoring data, water use monitoring data, and river loss gauging data in the calibration data set;
- creation of a more detailed physical representation of model parameters in terms of hydraulic properties, river recharge and rainfall recharge; and
- inclusion of knowledge from recharge estimation studies described in Chapter 4.

In summary, the key objectives of the groundwater modelling work described in this chapter were to:

1. update and re-calibrate the original KBR model, including new pumping monitoring, recharge and river data, to enable the model to serve as a predictive tool to assess the groundwater resource and the impact of PRW import into the Lockyer Valley;
2. use the model to determine the optimum PRW import volumes required to meet a range of selected environmental targets; and
3. analyse the impact of specified climate change scenarios on the groundwater resource in the Lockyer Valley.

The chapter is organised as follows: Section 12.2 contains a summary of changes to the conceptual structure of the model. This section includes work completed by the consulting company RPS Ltd and led by Jerome Arunakamaren, who co-authors this chapter. The limitations of the conceptual structure of the model are also described. Section 12.3 describes the numerical model specifications and model calibration outputs, the model limitations and the identifiability of model parameters and contributions to parameter uncertainty given the current model and parameter disposition. The optimal PRW import volumes required to meet selected environmental targets are then described in Section 12.4. This includes an analysis of the uncertainty associated with these model-based estimates of PRW import volumes. Finally, Section 12.5 analyses the worth of the existing monitoring data set in terms of how well it informs the prediction of PRW import volumes. The chapter concludes with a brief summary of the major findings of this aspect of work.

12.2 Details of Further Conceptual Model Development

A detailed description of the topography, climate, geology and hydrogeology and other related information can be found in the KBR report (KBR, 2002) as well as the preceding chapters of this USWRA report and are not repeated here. Instead, we discuss the changes to topographic and aquifer elevation definitions, and changes in aquifer stress definition based on new data sources and alternative parameterisations.

12.2.1 Topography and Aquifer Top and Bottom Elevation

A key shortcoming identified by the authors of the KBR model was the limited accuracy of the representation of the surface topography. Errors in the range of -14 to +12 m were identified during a comparison of the DEM data with surveyed spot levels (KBR, 2002). In the present study a 20 m mesh DEM is used to define the surface topography. Strata logs were then reinterpreted with this new DEM by RPS to redefine top and bottom elevations of the aquifer. The contours of the top and bottom of the aquifer are shown in Figure 103 and Figure 104 respectively. The surface topography obtained from the 20 m DEM served as the aquifer top. The measured thickness of the alluvium was interpolated to obtain the thickness of the aquifer over the entire model grid. These values were then subtracted from the elevations of the top of the aquifer to obtain the aquifer bottom elevations.

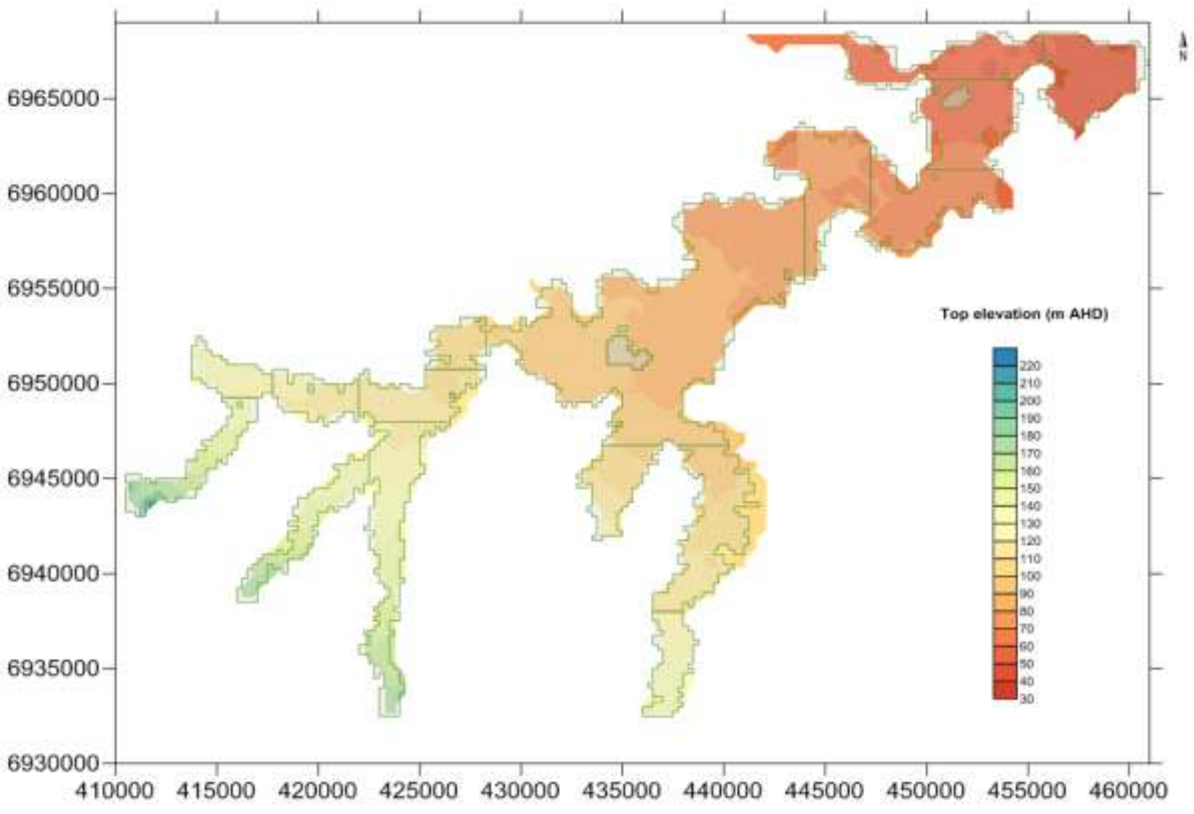


Figure 103: Contours of the aquifer top (m AHD).

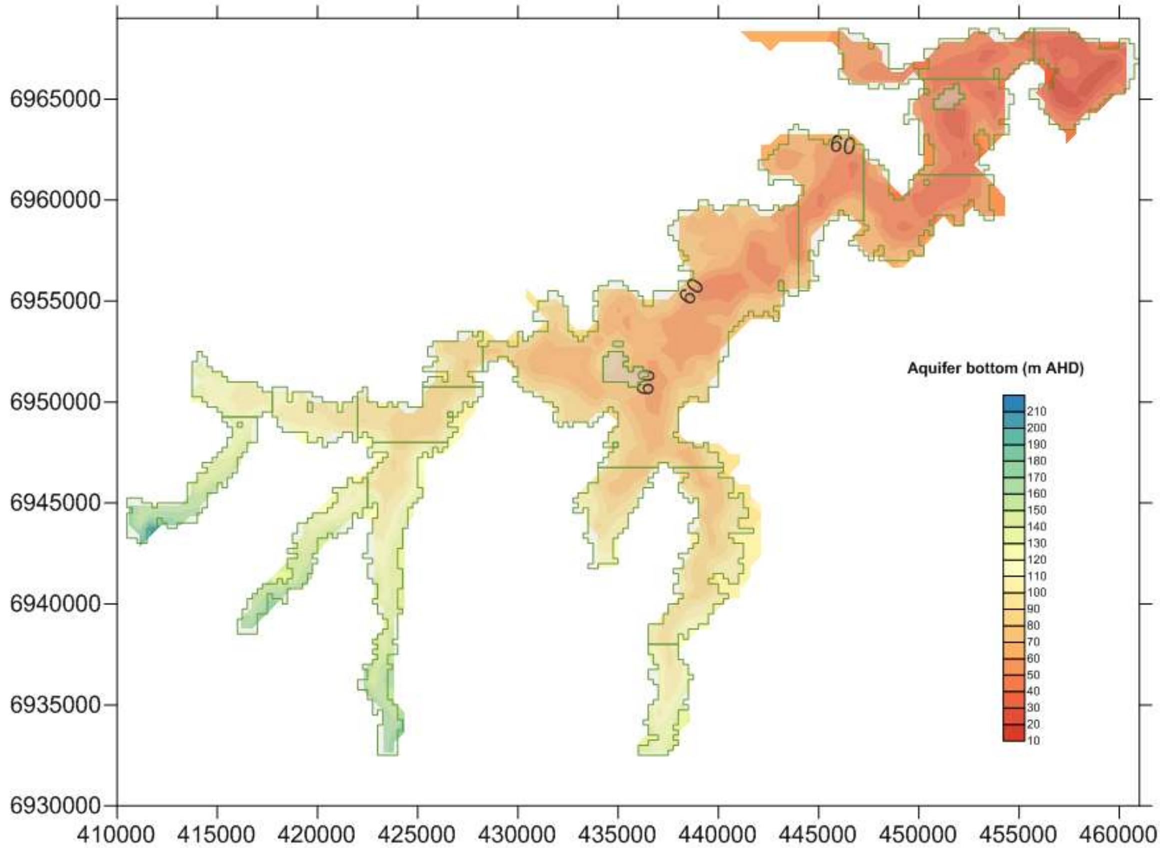


Figure 104: Contours of the aquifer bottom (m AHD).

12.2.2 Water Usage

The KBR model used estimates of groundwater pumping in different zones and distributed them uniformly among the pumping bores in these zones. For this study, metered water use in the central Lockyer region was also available (see Chapter 11). These data, wherever available, replaced the KBR estimates, while the KBR estimates were retained at all other pumping bores beyond the metered area.

The distribution of wells in the Lockyer Valley is shown in Figure 105. A total of 1166 production wells are considered. Metered pumping rates for the central Lockyer region were available from DERM for the modelled period 1991–2011. This data was compared with the KBR estimates used in the original model development. It is noted that the maintenance of the water meters within the valley is less regular than in the past, and therefore the accuracy of the water meter data may be less reliable towards the end of the monitoring period.

The water meter data were compared with the KBR estimates used in the development of the original model. The metered data volumes were very similar to the original KBR model estimates for the central Lockyer region, with the KBR estimates being slightly higher than the metered data. A comparison of the pumping data for the central Lockyer region obtained from DERM and KBR is shown in Figure 106.

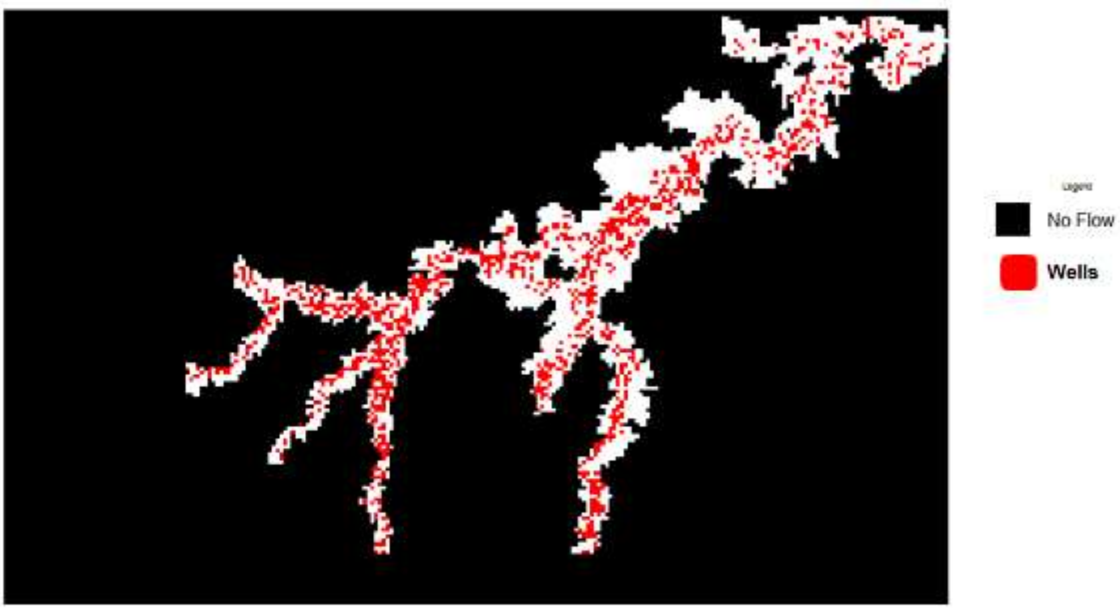


Figure 105: Distribution of the pumping wells in the Lockyer Valley.

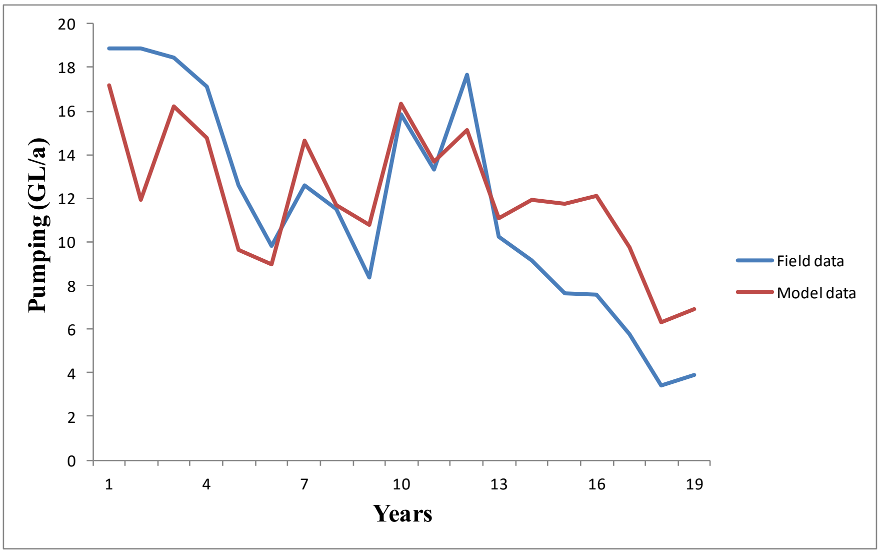


Figure 106: Comparison of the pumping data from the field and that previously used in the KBR model for the central Lockyer region (1991–2011).

A histogram of the errors (difference between the DERM data and KBR model values) in the pumping rates for those wells which were present in both the KBR model and the DERM database is shown in Figure 107. Since metered water use was only available for the central Lockyer region, and the KBR estimates are used for the rest of the region, this results in uncertainty in the pumping volumes which propagates into parameter and predictive uncertainties. This error distribution was used to determine the uncertainty of the pumping volumes, in the assessment of parameter and prediction uncertainty discussed later in this chapter.

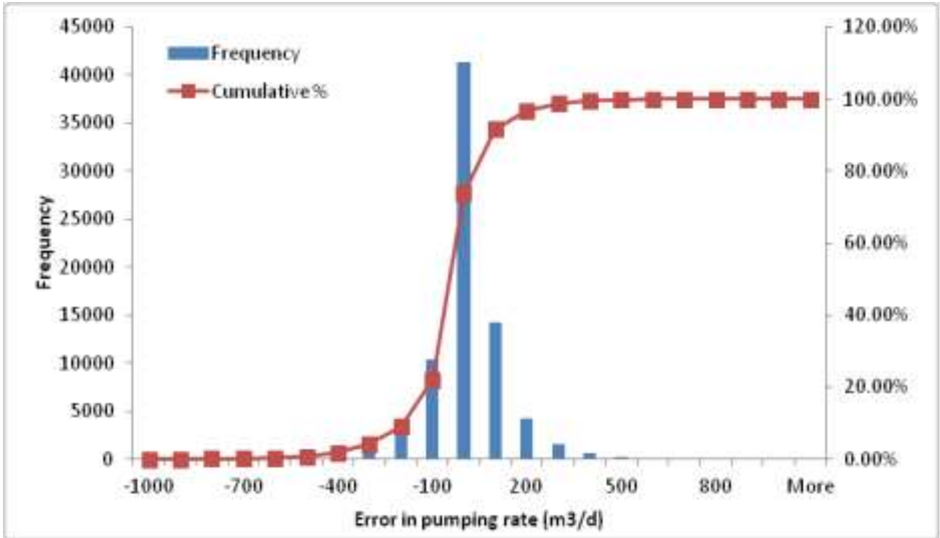


Figure 107: Histogram of the errors in the pumping rates and fitted normal distribution.

A comparison of the total pumping based on the DERM data and the KBR estimates is given in Table 31. The final estimate of groundwater abstraction rates was based on the metered data where available in the central Lockyer region, and beyond this region the estimates correspond to the KBR model.

Table 31: Total pumping estimates comparison.

	Total in Present Model(GL/a)	Total KBR model (GL/a)
Central Lockyer	11.72	12.15
Entire model	32.80	33.23

12.2.3 River Representation

In the KBR model the river was represented as a component of the distributed recharge module. In the UWSRA project, a physically explicit representation was desired so that the option of PRW import directly into the river system could be investigated if desired in the future. Consultants RPS Ltd were contracted to estimate the river bed and river stage heights as described below.

The river bed elevations were estimated by identifying the incised stream elevation in the updated DEM. The stage height was created by creating a “depth of water” layer to be draped over the riverbed elevation over each river cell. This was done in two parts. Firstly by obtaining a reference river height at each cell and secondly by superimposing the upstream attenuation of the weir water levels. The IQQM model (Department of Land and Water Conservation, 1995, 1996) simulates varying stage height data at a number of sites in the region (see Figure 108). These simulations are informed by a series of stream gauging sites, established by DERM, which are distributed throughout the valley. The difference between the gauge zero elevation and the stage height data was considered to be equivalent to the depth of water at that site. In some cases this approach resulted in the modelled river flowing when actual field observations were of a dry river bed. A manual check and adjustment of the river stages estimated by this approach resolved these inconsistencies. River conductances were assigned to river reach zones that were specified as coincident with IQQM calculations of river loss. The river conductance values were estimated as part of the model calibration process, which was implemented using a program that rewrites the model river input package each model iteration.

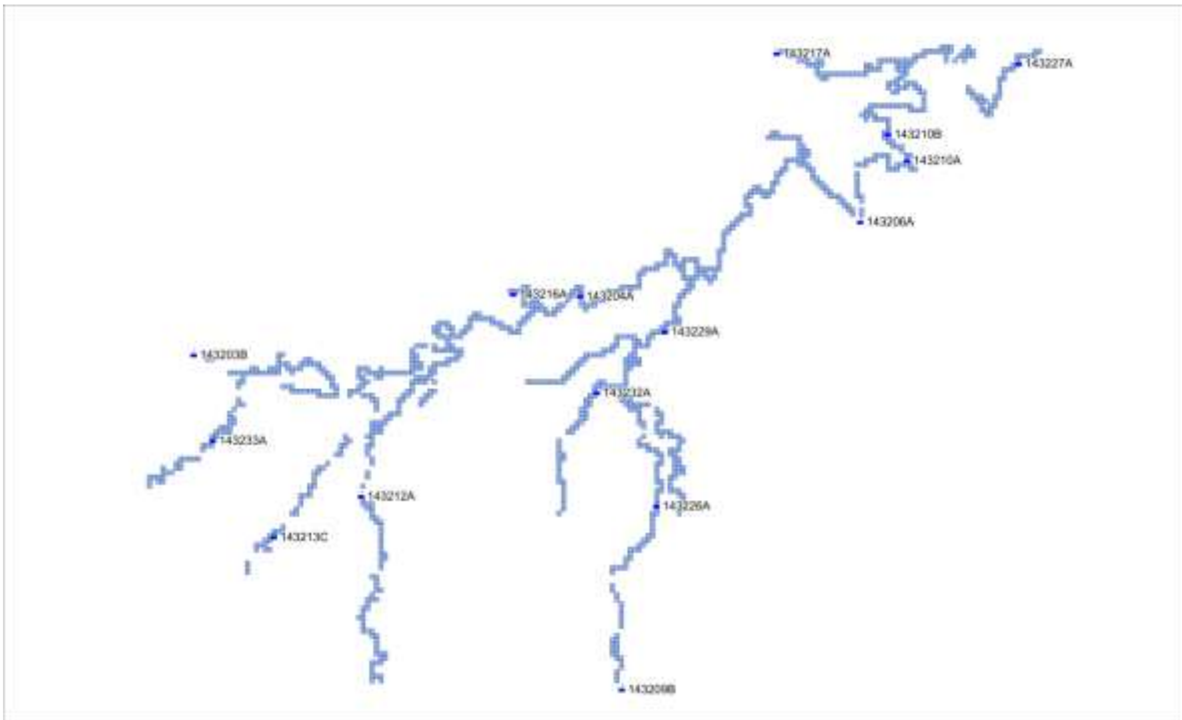


Figure 108: Gauging stations used to interpolate the stream data.

12.2.4 Recharge Representation

The diffuse recharge rates for the entire grid of the Lockyer groundwater model were computed based on the soil water balance data for various crop types obtained from DERM (Chapters 9 and 11). The recharge reaching the groundwater table is a function of the deep drainage. The deep drainage rates corresponding to different land use classes were obtained using the model outputs from the HowLeaky software simulations (as described in Chapter 9) and were distributed spatially using mapped cropping areas (as described in Chapters 9 and 11). A series of crop-based multipliers to convert the deep drainage into the diffuse recharge rate were included as parameters for calibration.

A computer program was written to include recharge multipliers as calibration parameters. The inputs required for this program include the array of 'deep drainage' values obtained from HowLeaky for all the time periods considered in the model calibration. The HowLeaky output has the deep drainage values for all the five different crop types. The second input in to the program is the array of crop types in the model grid for the Lockyer, based on the mapped cropping areas described above (Figure 109). The program reads the crop type for each model cell from these arrays, reads the value of 'drainage' for the corresponding crop and multiplies it with corresponding crop multiplier, and then writes the MODFLOW recharge file. The program is also able to apply a lag to the application of diffuse recharge by any number of days depending on the time taken by the diffuse recharge water to reach the water table, by means of lag parameters. Three lag parameters were added in parameter estimation based on three different unsaturated zone depths classified for the Lockyer Valley. Crop multipliers and lag parameters are read from a parameter file. However, the lag parameters were later fixed at zero-lag values as the predictions were not sensitive to these parameters. The use of this program is discussed further in the model calibration section.

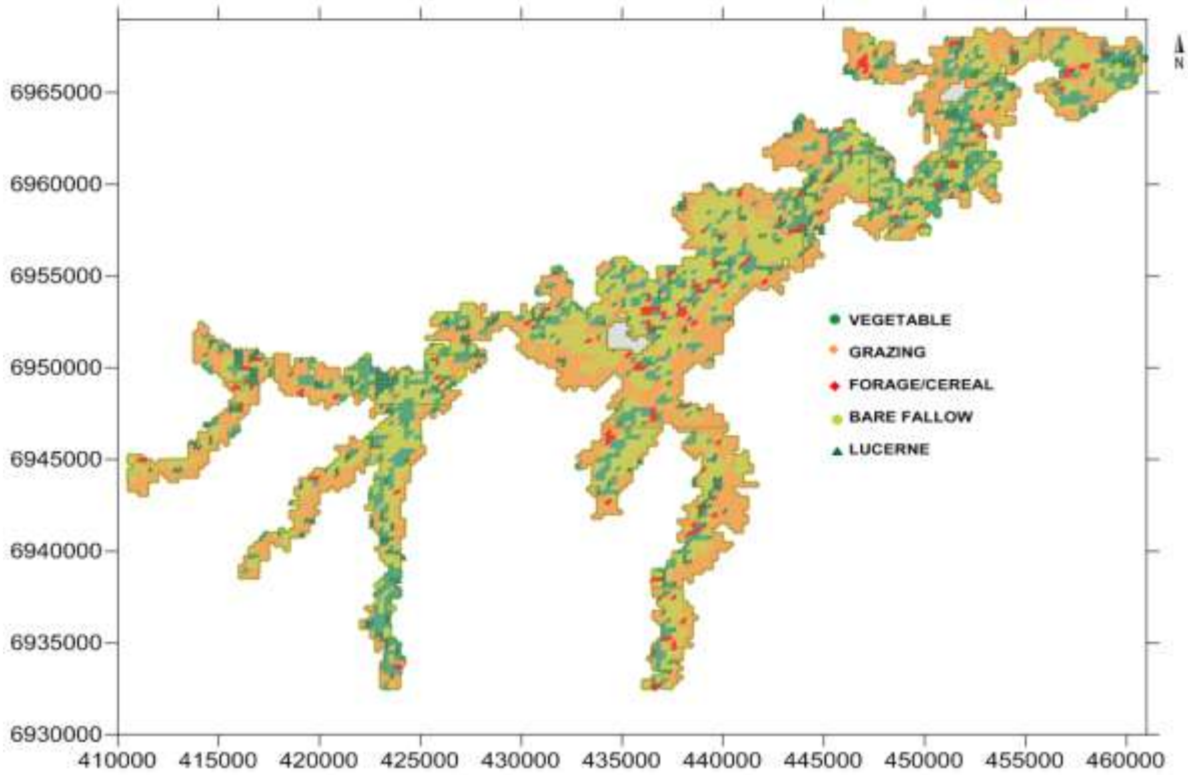


Figure 109: Map of cropping areas used to estimate the diffuse recharge inputs.

12.2.5 Limitations of the Current Conceptual Structure of the Model

The changes to the model conceptualisation described above addressed the major issues that were identified in the KBR report. However, the following omissions in the model conceptualisation must be acknowledged and accounted for in any future work.

The uncertainty of the simulated pumping rates depicted in Figure 106 and Figure 107 propagate through the entire model calibration and predictive simulation analysis. Future work to improve the accuracy of these pumping rate estimates would allow a significant improvement to simulation of groundwater in this groundwater system.

There is geochemical evidence of small amounts of seepage occurring from the underlying sandstone formations into the alluvial aquifer system, particularly during dry climatic periods where groundwater levels drop within the alluvial aquifer system. These fluxes were omitted in this model as the fluxes were small compared to the volumes of flow within the alluvial aquifer system. However, if in the future water quality simulations are undertaken, it would be important to represent these fluxes from the underlying strata.

The river bed conductances are specified as constant along long reaches of the river, and these reaches are coincident with those used for IQQM reported outputs. There is evidence that the river–aquifer interaction varies significantly within some of these reaches. Therefore the coarseness of this river bed conductance representation limits the degree of precision to which the system responses can be simulated. Any further work using this model would be improved by refining the detail of the parameters describing river–aquifer interaction.

River recharge is also complicated by the representation of the river using 250 m by 250 m model cells, whereas the creek is typically less than 50 m wide. The river bottoms derived from the DEM therefore must be represented as averages over the model cell area. This introduces a level of abstraction into the river representation in the model, and means that the river bottom cannot be represented as being as low as the DEM values may indicate. Various general adjustments to the river bottom definition were trialled, for example, river bottoms 1 m and then 5 m below the land surface elevation, with the 5 m adjustment allowing greater aquifer–river recharge. Further work assessing the impact of these river bottom representations is recommended.

The spatial disposition of diffuse recharge entering the alluvial aquifer system is defined by a cropping map. The hard-wiring of this single temporal snapshot of the spatial disposition of cropping and its contribution to diffuse recharge results in the model inputs being unable to reflect the more recent reduction in cropping that occurred within the valley as groundwater levels reduced in the aquifer. Once again, further work using this model would need to address this shortcoming.

12.3 Numerical Model Development

12.3.1 General

The numerical groundwater model was developed using the MODFLOW-NWT software package, in conjunction with parameter estimation software PEST (Doherty, 2012a) and its utility software. In addition, a series of custom programs were written to include additional parameters such as river conductance and recharge rates as calibration parameters, and to process output files. MODFLOW-NWT was selected as it is able to simulate the de-rating effect on pumping when the hydraulic head falls below a certain defined percentage of the total volume of the finite difference cell. This helps in the realistic simulation of reducing pumping rates as water levels are lowered.

The grid used in the KBR model is retained in this study. This grid covers an area of 30,000 ha with a uniform grid with cell size of 250 m x 250 m having a north-south/east-west orientation. The model is comprised of 4806 active grid cells, all of equal size. Figure 110 illustrates the model grid, active cells and river and GHB boundaries. The model has a single unconfined layer. The model is calibrated over 246 monthly stress periods starting from January 1991 to June 2011, thus including the 2011 flooding within the calibration period.

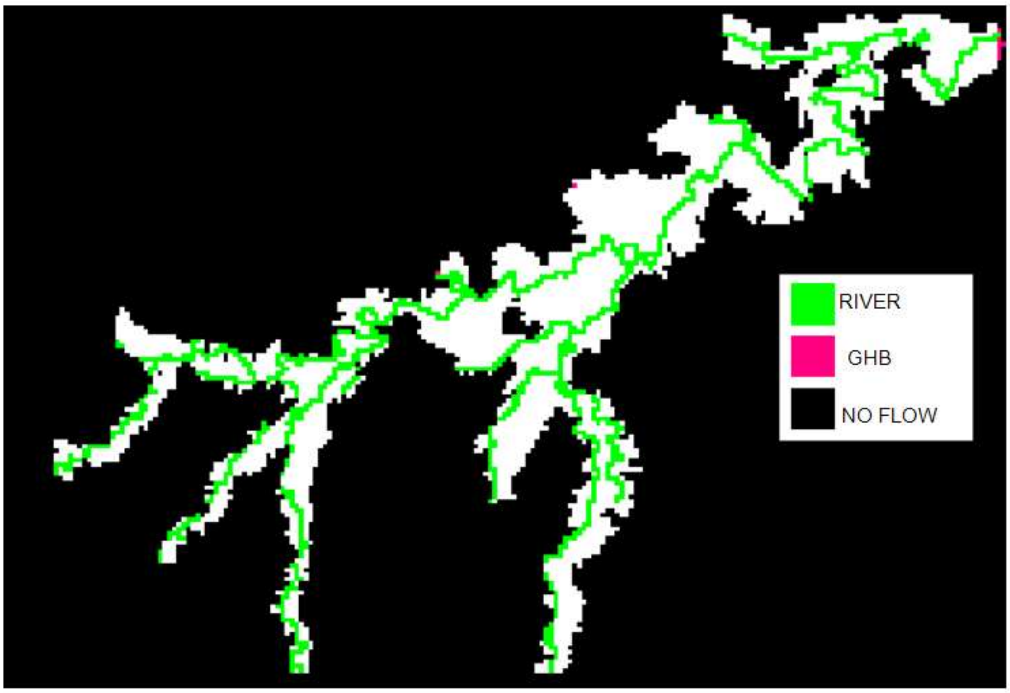


Figure 110: Lockyer grid indicating river, GHB and no-flow boundaries.

12.3.2 Parameterisation

Hydraulic properties in the model were represented using spatially distributed pilot points which are used together with geo-statistical interpolation. A total of 199 uniformly placed pilot points were used in the calibration of each hydraulic property type, including hydraulic conductivity (horizontal and vertical directions) and specific yield, which gave a total of 597 pilot point parameters. Initial estimates for pilot points were obtained from the calibrated KBR model, to help faster convergence of the PEST parameter optimisation algorithm. The locations of these pilot points in the grid are shown in Figure 111.

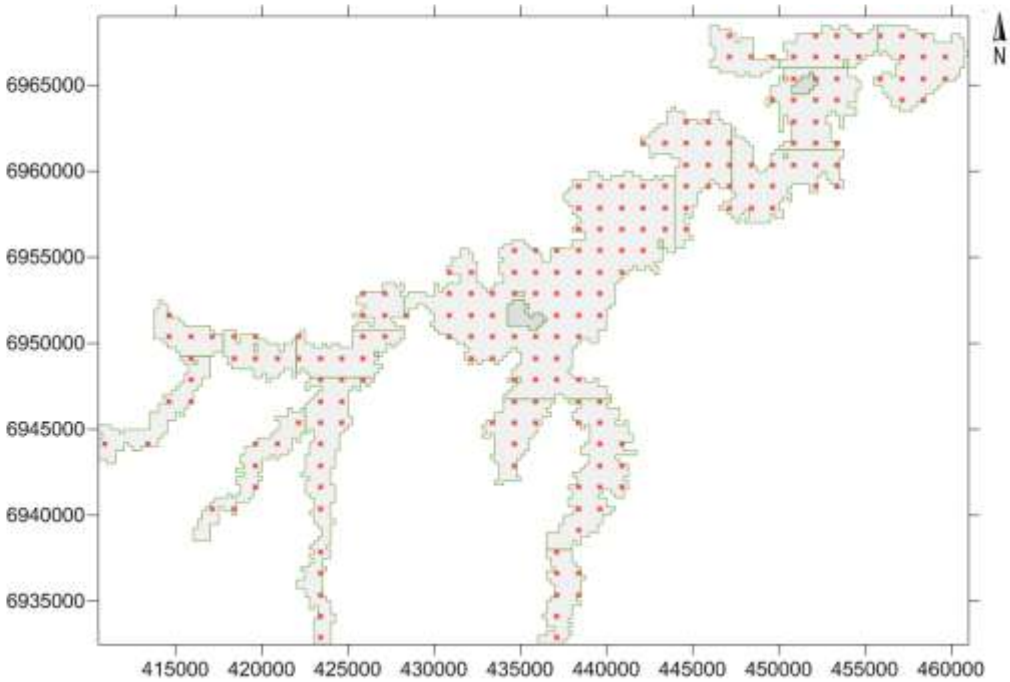


Figure 111: Pilot points for the hydraulic properties uniformly distributed in the model grid.

River stage and bed elevations were determined on the basis of the work undertaken by RPS as discussed above. River bed conductance was estimated in the model calibration process and was parameterised over 26 sections of the river system coincident with the IQQM output reporting discretisation.

Diffuse rainfall recharge and irrigation excess recharge was pre-estimated using the work discussed in Chapters 9 and 11. These pre-estimates were allowed to vary during the model calibration using multiplier parameters. These multiplier parameters were applied for each crop type. In addition, a recharge lag term was applied to the pre-estimated time series, to account for different thicknesses of the vadose zone above the aquifer. In total, five different crop multipliers were considered. Also, three different lag parameters were initially considered to account for the time lag for the rainfall and excess irrigation water to reach the water table. However, the lag parameters were later found to be insignificant and were discarded. The river sections for which a separate conductance parameter was assumed are shown in Figure 112.

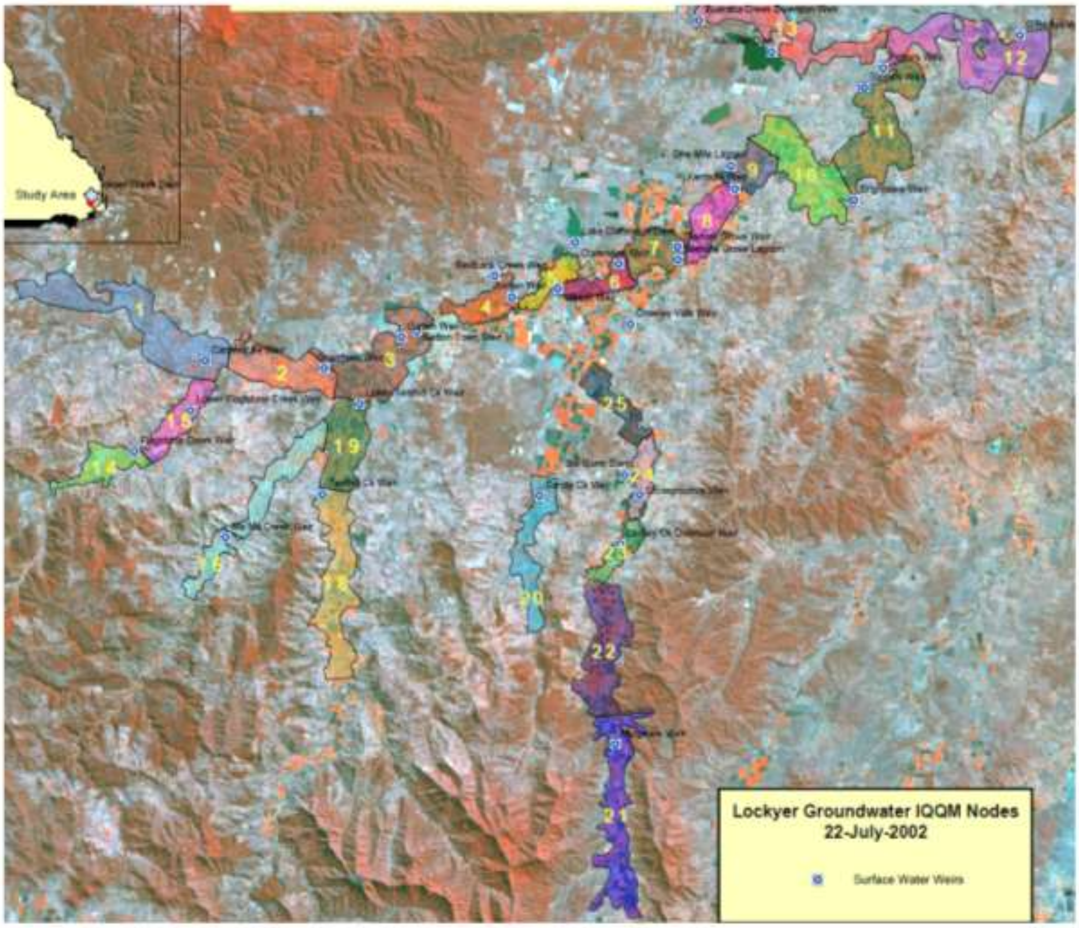


Figure 112: River reaches for which an independent conductance parameter were assumed (Seqwater, 2012).

12.3.3 Boundaries and Initial Conditions

The groundwater flow in a transient model is primarily governed by the initial and boundary conditions. Groundwater stresses, including diffuse recharge and pumping, are expressed as specified flow boundary conditions. Recharge and pumping are modelled by means of RECHARGE and WELL packages respectively in the MODFLOW model. The river is modelled using the MODFLOW river package. No flow boundaries represent regions with zero gradient resulting in no flow across such boundaries. General head boundaries (GHB package) were used to represent head-dependent flux boundaries. The locations of the river, GHB, no flow and river boundaries, as used in the developed model, are shown in Figure 110. Thus the boundary conditions used in the model development include:

- No flow – Delineation of no flow zones
- General head boundaries (head-dependent flux boundary)
- Rivers (specified flux boundary)
- Recharge (Specified flux boundary)
- Wells (Specified flux boundary)

The initial distribution of head was interpolated from the measured head at the monitoring locations corresponding to the start of the transient model simulation period. Measurements at 503 bores were used to estimate the initial heads. Unfortunately, for a variety of reasons, a number of these bores were later identified as being not representative of the aquifer, leading to some errors in initial head estimates as discussed in the model limitations section.

12.3.4 Model Calibration

The adjustment of parameters, to achieve a match between model outputs and measurements – that is, calibration, was implemented with the assistance of the model independent parameter estimation software PEST (Doherty, 2012a). The calibration targets comprised:

- Groundwater level monitoring data for the period January 1991 to June 2011; and
- Estimated monthly recharge volumes from the 26 stream sections for the period January 1991 to June 2011, as simulated by the IQQM model.

The previous KBR model calibration data set contained 18,318 temporally varying water table elevation measurements from 242 locations. The updated data set used in the current modelling work has 64,960 water table elevation measurements, from 355 locations, spanning the 20-year period from 1991 to 2011. The data for this period within the model domain were extracted from the DERM database. The bore locations corresponding to the old and updated data sets are shown in Figure 113. The continuous groundwater level measurements (monthly average) obtained from the groundwater loggers (Chapter 6) were also included in the updated database. It is important to note the greater spatial density of these calibration targets compared with the pilot point density depicted in Figure 111, river bed conductance zones depicted in Figure 112 and the cropping zones depicted in Figure 109. The relative spatial coarseness of these parameter zones compared to the observations places a limit on the level of model-to-measurement fit that can be achieved with this current model.

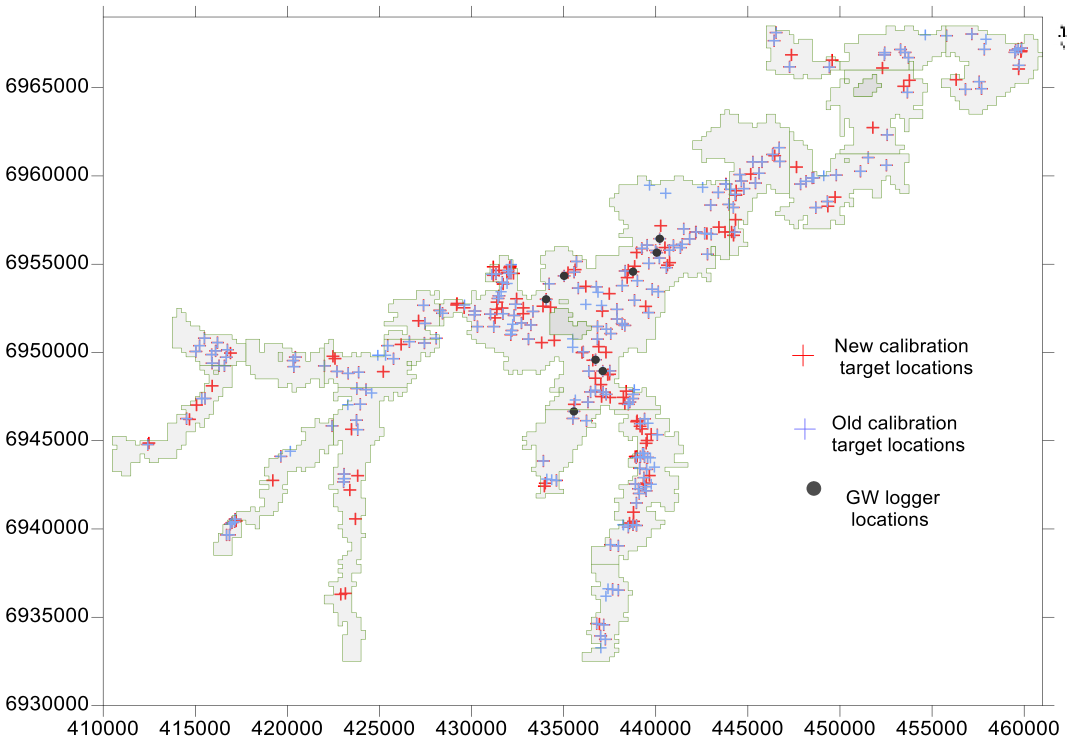


Figure 113: Observation bore locations for the old and new calibration data sets.

The conceptual framework of the calibration procedure adopted in this study is shown in Figure 114. PEST starts with the initial parameter vector for the model and computes the objective function, namely, the sum of the squares of the difference between the simulated values and the calibration targets. Then it computes the gradient of the objective function with respect to each of the parameters. Then, based on the steepest descent algorithm, it computes the new parameter vector. For the hydraulic parameters, a new set of values is generated for the pilot points. Based on these, the values at all other grid cells are computed by kriging with respect to the considered variogram. Recharge and river packages are updated based on the new set of parameter values corresponding to their defined parameter zones.

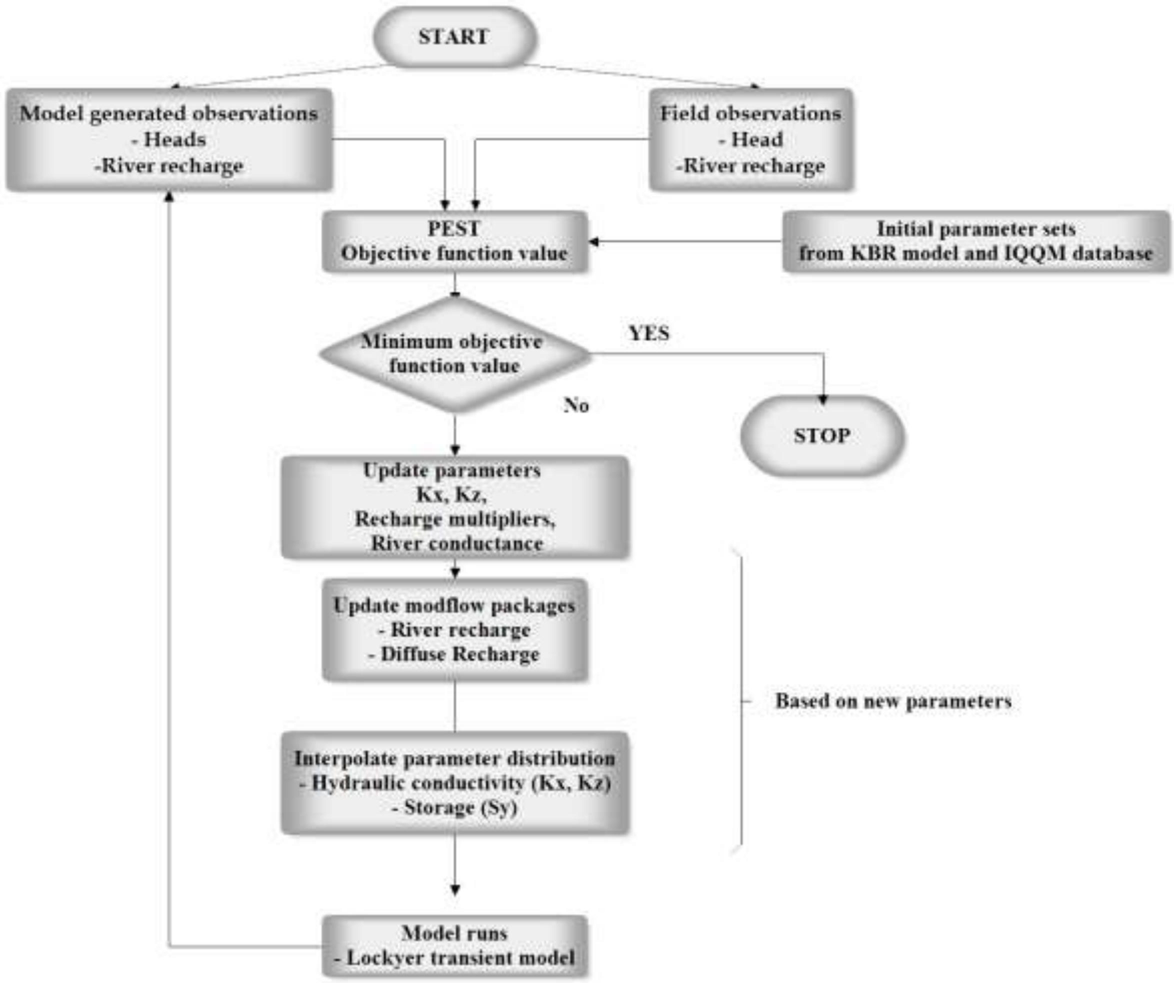


Figure 114: Calibration flow chart for the Lockyer model.

Both preferred value and subspace regularisation constraints were used in the model calibration (refer to Doherty, 2012b for details). Parameter values obtained from the KBR model were used to frame regularisation constraints. The SVD-ASSIST methodology was then used to impose the subspace regularisation constraints. The large number of model runs was facilitated using PEST's BEOPEST option, which enables parallel processing of the MODFLOW runs and significantly reduces the computational burden by distributing the model runs on many computer processes.

12.3.5 Calibration Results and Model Limitations

Model calibration can be assessed in terms of how uniquely model parameters can be estimated, how credible the estimated parameters are when compared with other 'non-model' parameter knowledge and model conceptualisation, and how well model outputs match measured values. These three assessment types are now discussed in relation to the Lockyer model.

12.3.5.1 Parameter Identifiability

While trialling different model conceptualisations, it was noted that the overall model-to-measurement fit was often similar using different parameter combinations, and hence the model solution is non-unique to some degree without imposing geological knowledge on the solution, as we have done in this case based on the river recharge dynamics. One measure of this non-uniqueness is parameter identifiability, which indicates the relative extent to which parameters can be estimated uniquely, given the selected model conceptualisation and its parameterisation and the calibration data set (Doherty, 2012b). Parameter identifiability has a range from 0 to 1, with 0 indicating non-identifiability, and 1 indicating full identifiability. It is important to note, however, that parameters with an identifiability of 1.0 may still be in error, if the model conceptualisation and parameter discretisation is not a perfect representation of the real world, which is more often the case than not. Furthermore, other factors, such as uncertainty during observation, may generate additional errors. However, despite these limitations of the identifiability measure, it is a reasonably robust measure for identifying the relative reliability of parameter estimates for a given model conceptualisation and parameterisation.

Identifiability values for horizontal hydraulic conductivity and specific yield assumed values in the range 0 to 1. Figure 115 and Figure 116 indicate the distribution of identifiability for Kx and Sy respectively for the initial conceptualisation described above. These figures both depict a spatial distribution, which indicates that the lowest parameter identifiability occurs along the model boundaries. They also indicate that the hydraulic conductivity parameters were estimated more uniquely than the specific yield parameters.

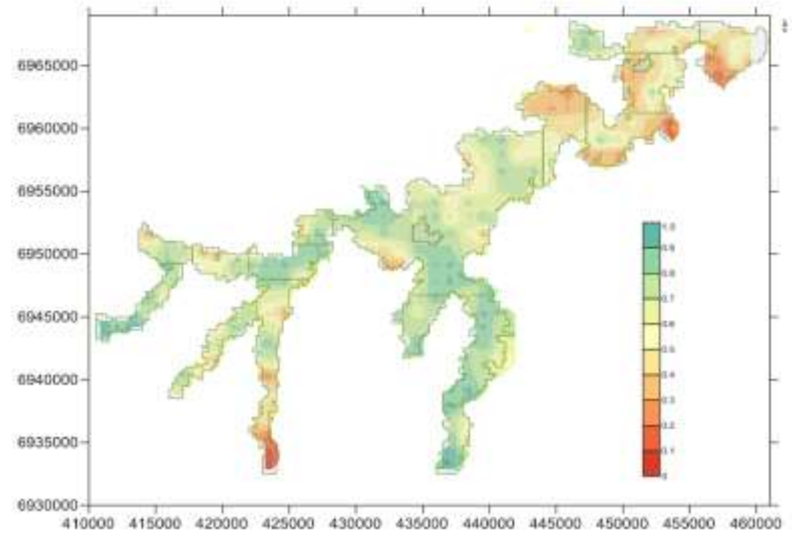


Figure 115: Contour of horizontal hydraulic conductivity identifiability.

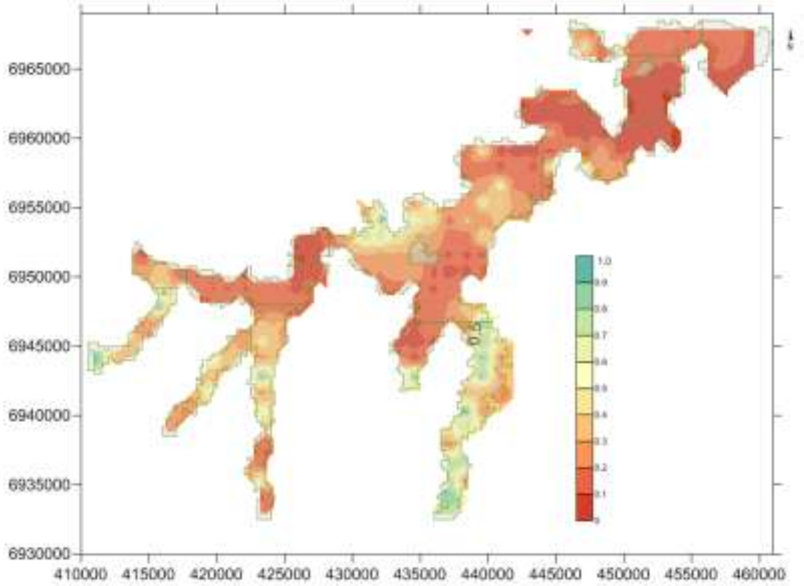


Figure 116: Contour plot of specific yield identifiability.

Parameter identifiability of the crop multiplier and river conductance parameters could also be depicted if a more detailed representation of these parameters was adopted in the model. However, because it is recognised that the current use of zones of constant parameter value is too simple a representation of this system, the identifiability of these parameters has not been included.

12.3.5.2 Parameter Credibility

A number of pump tests have been carried out in the Lockyer Valley by DERM. The histogram of hydraulic conductivity values obtained from these pump tests is given in Figure 117. The histogram indicates a range in tested hydraulic conductivity values between approximately 5 and 300 m/day. These hydraulic conductivity approximations are obtained by assuming a general aquifer thickness intercepted by the screened section in tested wells of 6 m (e.g. Bouwer, 1989). An alternative and more robust approach would be to assess the saturated thickness of the aquifer, the well construction and the degree of well penetration of the aquifer on the particular date of testing, and assess hydraulic conductivities in light of this information. Such analyses are recommended for further work, and it should be noted that the values in Figure 117 may tend to the higher side of what actually occur.

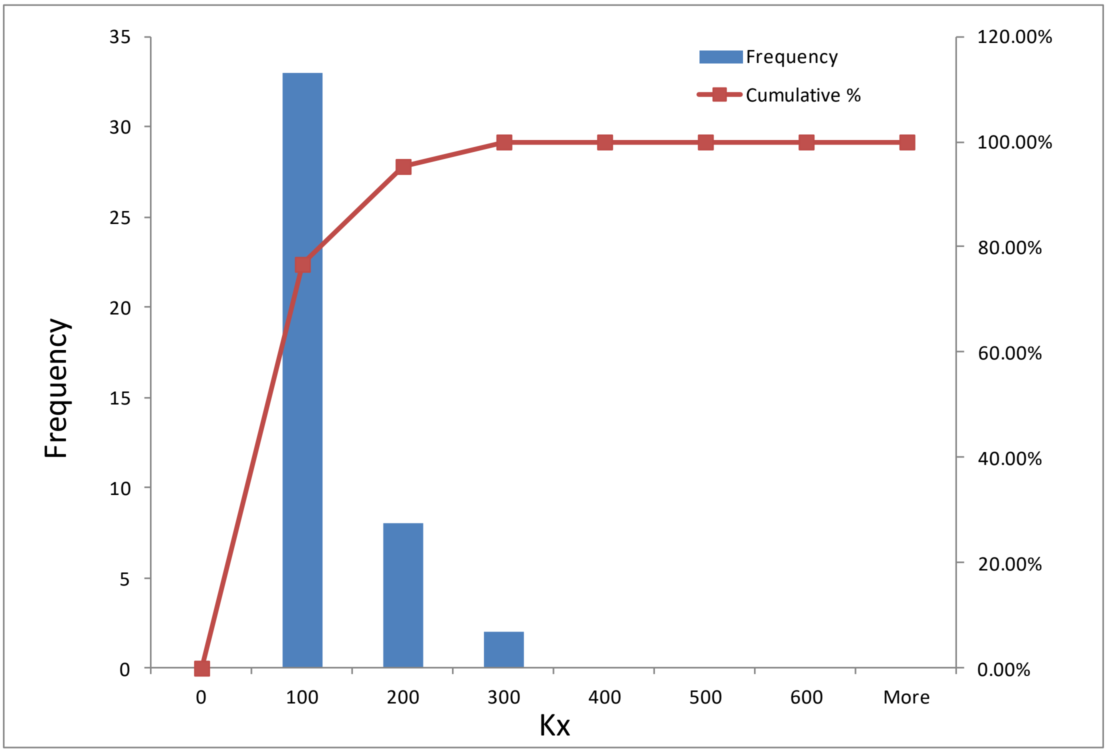


Figure 117: Histogram of hydraulic conductivity from aquifer pumping tests.

Figure 118 depicts the spatial distribution of results from the aquifer pumping tests. Hydraulic conductivity measured at 43 locations was analysed to obtain the histogram and spatial plots. The highest values are located within the central Lockyer Valley, and lower values tend to occur towards the edges of the aquifer system.

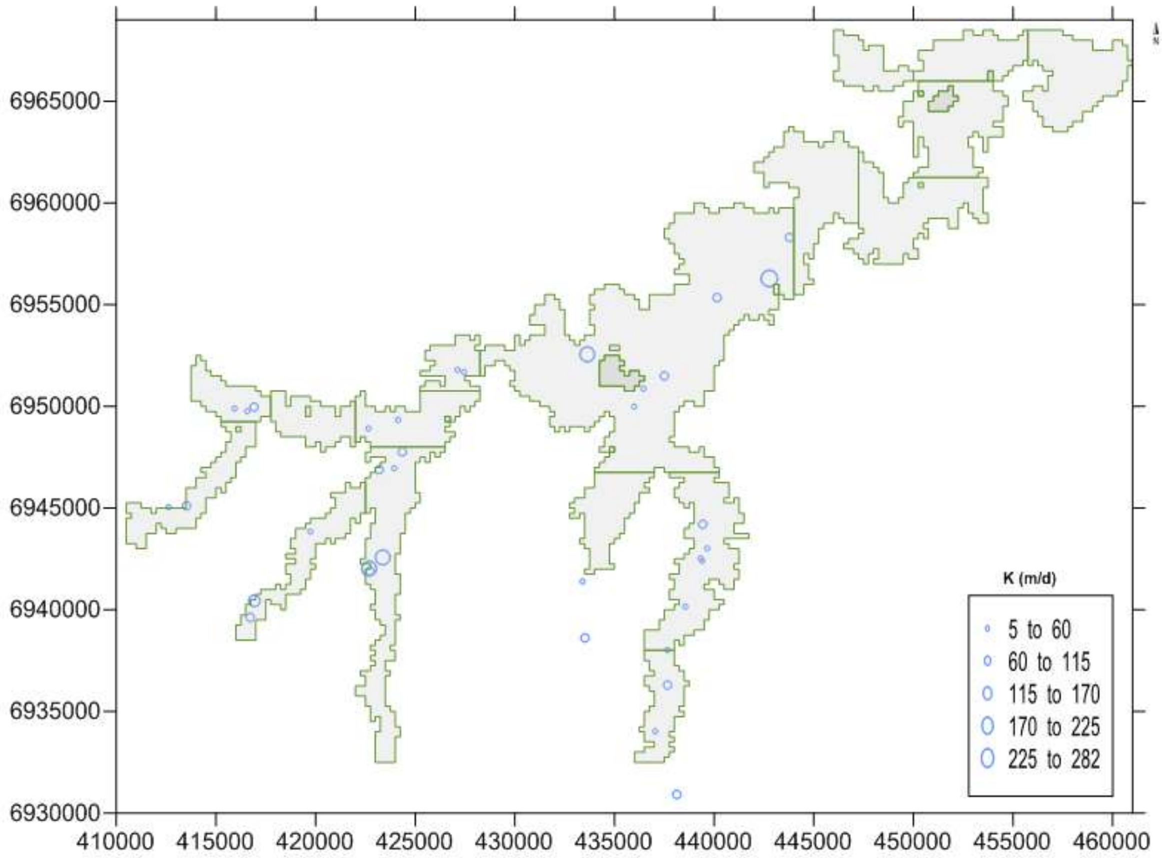


Figure 118: Spatial plot of hydraulic conductivity from pumping tests.

Specific yield values were also determined in a number of pumping tests. They ranged from 0.001 to 0.3. The distribution of the log of these values is shown in Figure 119.

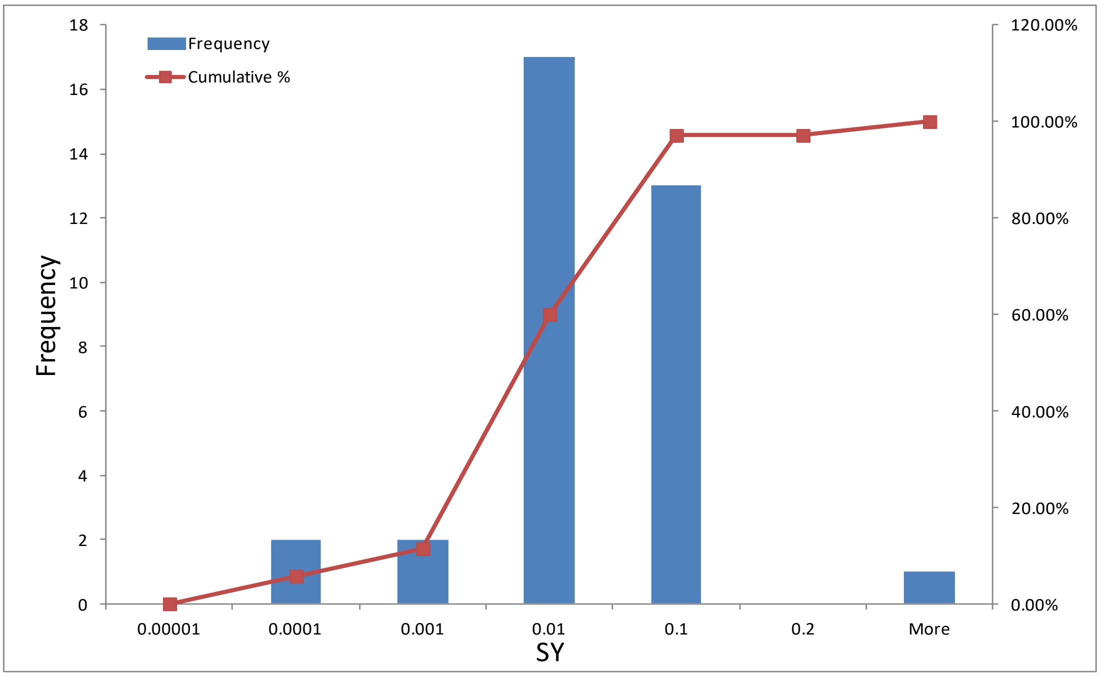


Figure 119: Histogram of Sy values from the pumping tests.

The spatial distribution of specific yield (SY) values derived from aquifer pumping tests is depicted in Figure 120.

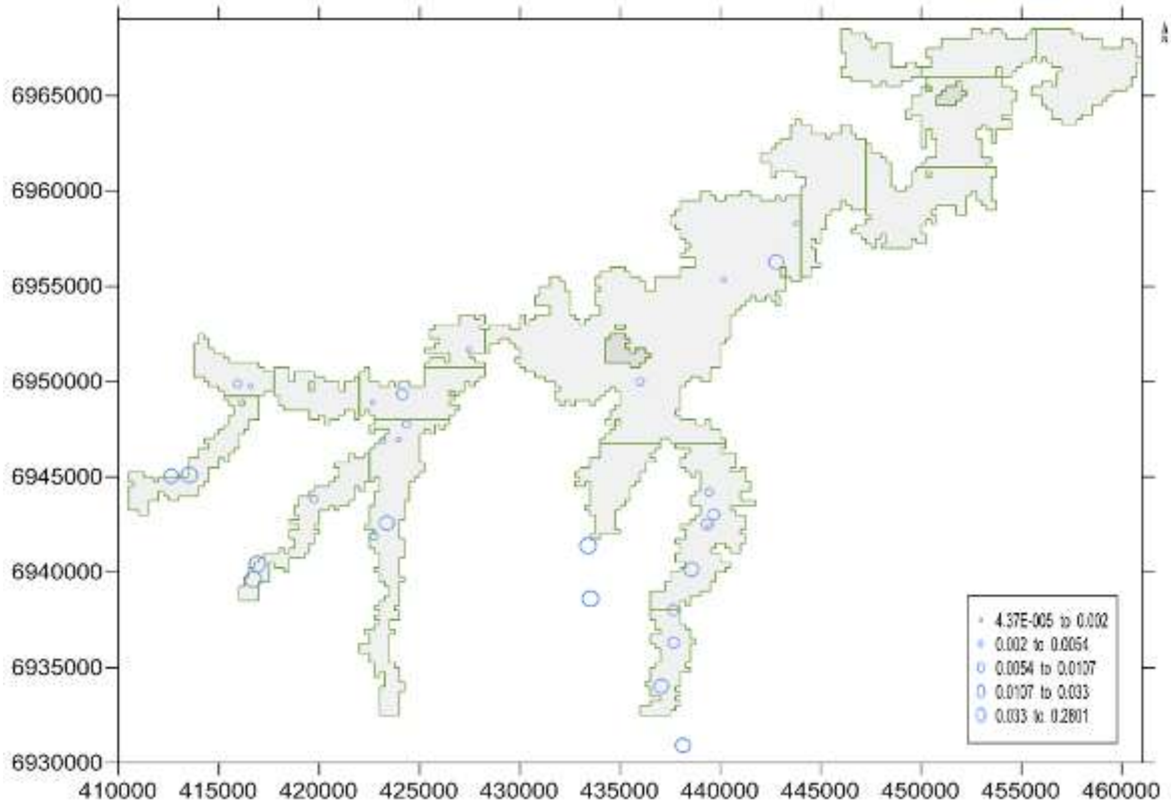


Figure 120: Spatial plot of SY from pumping tests.

These results were also analysed in the past (Bleakley and Durick, 2000; Durick and Bleakley, 2000; Lane, 1981). These earlier analyses identified that the aquifer test–derived specific yield values tended to err on the low end of what is expected in this region and indicated that the short duration of the tests may be the cause of some error in the derived results. Also, the spatial distribution of the parameters based on a relatively fewer number of pumping tests may not be representative of the entire range of their value.

The calibrated model hydraulic parameters can be compared with these test derived distributions. Recall from the discussion on parameter non-uniqueness that there are several model calibrations for different model conceptualisations, with *equivalent model fits*. We elected to select one, which we believed was the most credible in terms of parameter credibility, and now report on these results.

The model hydraulic conductivity values have a larger range than those of the reported pumping tests. The calibrated model has hydraulic conductivities from 0.001 to 1500 m/d. Figure 121 and Figure 122 provide the histogram and spatial distribution of these calibrated horizontal hydraulic conductivity values.

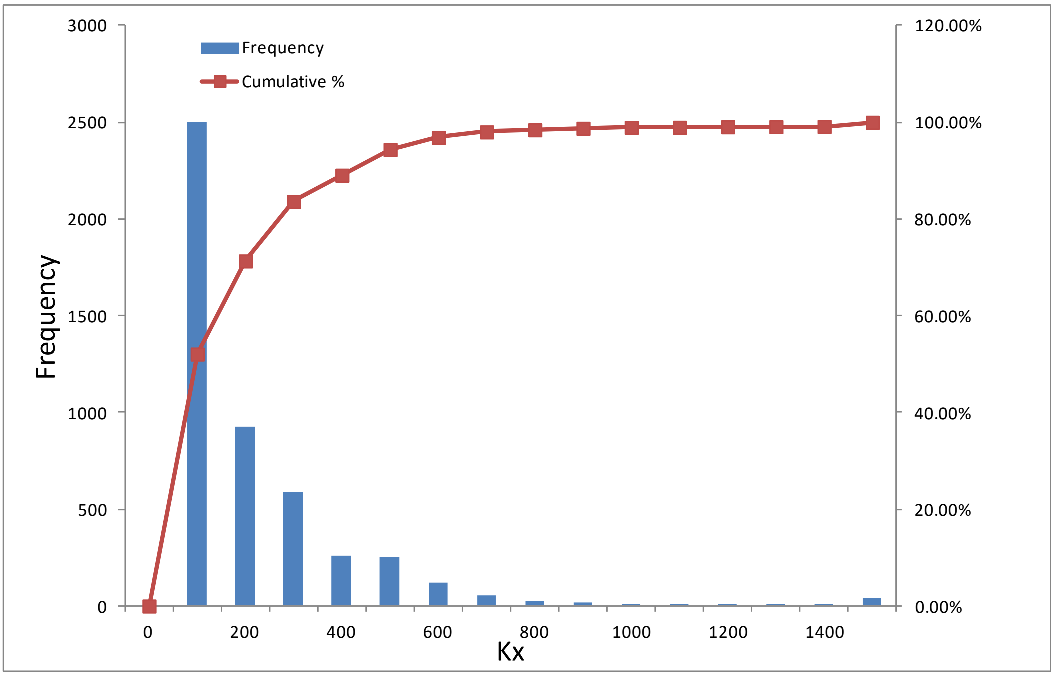


Figure 121: Histogram of the calibrated Kx values.

The higher values are generally centred in the central Lockyer region.

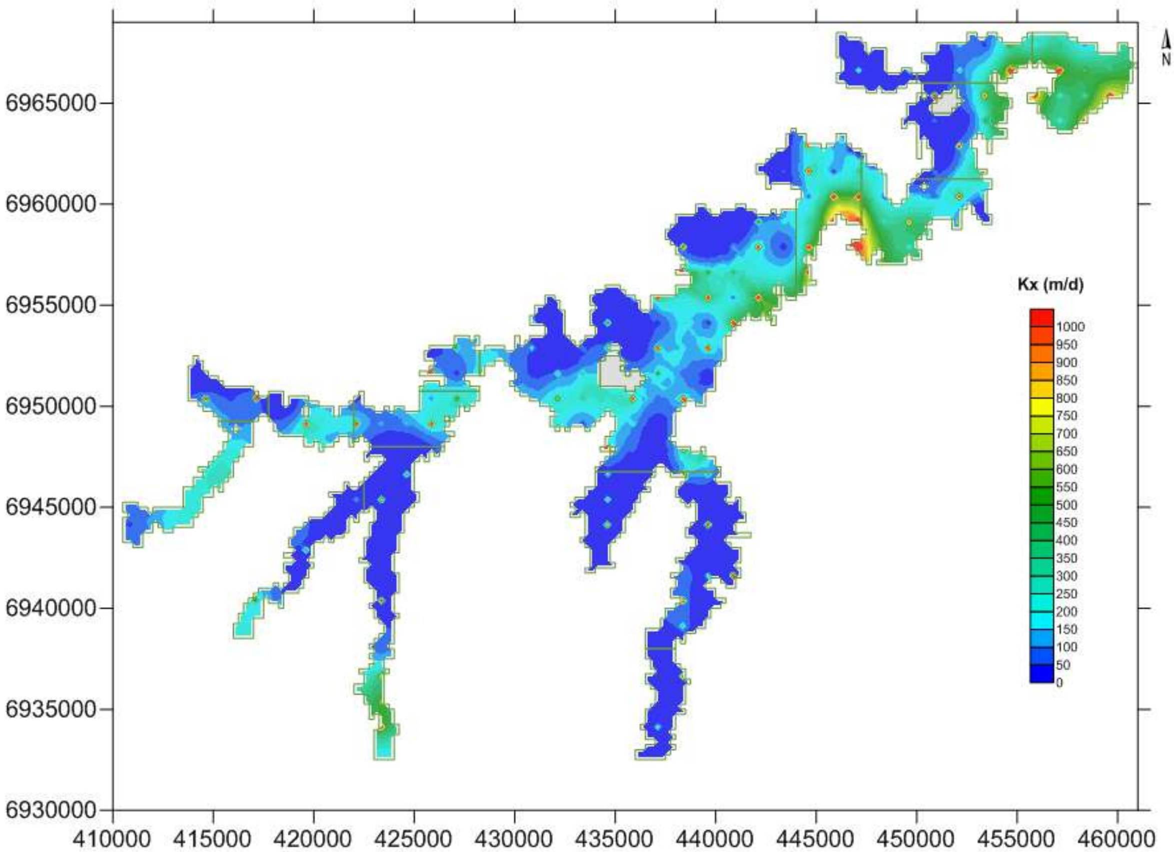


Figure 122: Spatial plot of the calibrated Kx values.

Specific yield was also estimated at the same 199 pilot point locations and prior estimates from the KBR model were included as regularisation constraints. Figure 123 and Figure 124 provide respectively the histogram plot and spatial distribution of calibrated specific yield value, indicating a majority of the specific yield values lying between 0.001 and 0.2.

Note that the calibrated parameter range is again wider than indicated in the pumping tests. Table 32 gives the value or range of values for different parameters as obtained from the calibration.

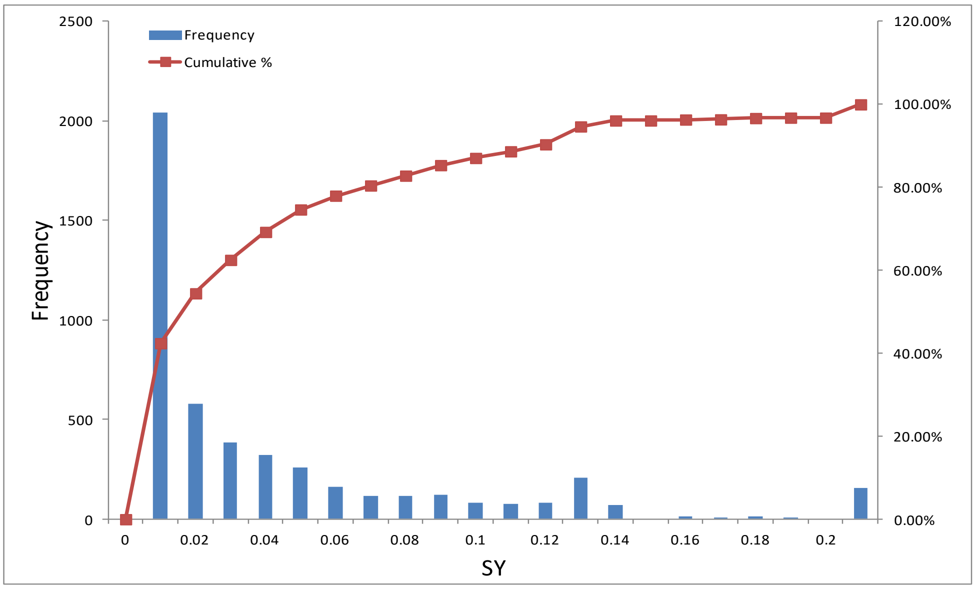


Figure 123: Histogram of the calibrated SY values.

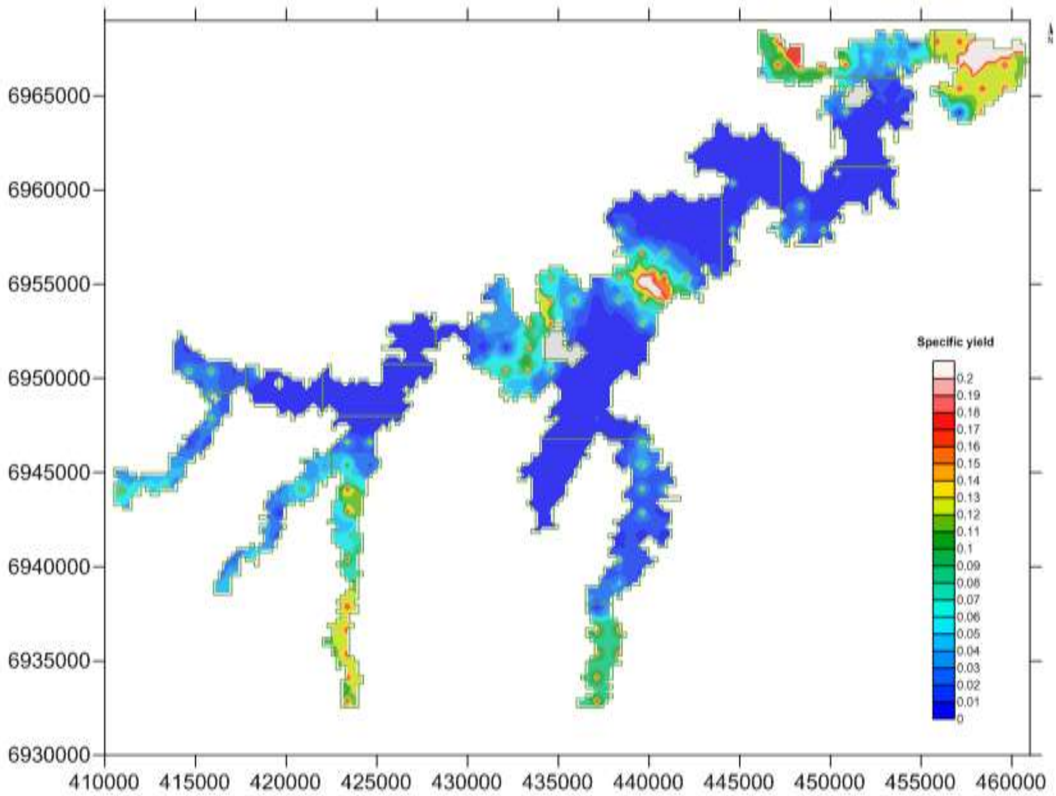


Figure 124: Spatial plot of calibrated SY (log domain).

A summary of the parameter ranges observed in the calibrated model is given in Table 32.

Table 32: Values/range of different parameter groups.

Parameter	Calibrated value/ range	Lower 95 %	Upper 95 %
Kx	0.001–1500	1.260	519.254
Kz	0.11–88.18	0.23	6.88
Sy	0.001–0.2	0.001	0.13
River conductance	1–11815.11	1	11815.11
Crop multiplier	0.01 - 0.8	0.01	0.8
Well abstraction multiplier	1	0.9	1.1

12.3.5.3 River Conductance and Recharge Parameters

A multiplication factor was estimated for the recharge flux simulated for each of the five crop type groupings in the valley. These estimated multiplication factors were in the range 0.01–0.8. The lag parameters were fixed at their zero-lag values, meaning that the diffuse recharge all over the Lockyer is assumed to reach the water table without any time delay.

The river conductance parameters were estimated for the 26 river stretches depicted in Figure 112 and were included as parameters for calibration. The lower and upper bounds for all these parameters were specified as 1.0 and 1.0E7 respectively. The river conductance values ranged between 1.0 and 11,815 m^2/day (Table 32).

12.3.6 Model-to-Measurement Match

12.3.6.1 Groundwater Level Measurements

Measured water levels from a total of 503 bores spread over the Lockyer were used as calibration targets. Out of these, 148 bores were removed for a variety of reasons, the majority because they were located along the valley margins and were causing numerical difficulties in the model when they dried out during periods of low rainfall. Also, some bores initially included in the data set were removed when identified later as sandstone bore or dry/perched bores.

12.3.6.1.1 Discussion on Model-to-Measurement Error

The overall fit of the remaining bores was reasonably good, with a regression coefficient of 0.99, as depicted in Figure 125. However, the mean absolute error (MAE) and root mean square error (RMSE) of the water level predictions were found to be -1.73 m and 4.54 m respectively, which is a reasonable proportion of the saturated aquifer thickness of between 10 m and 20 m. This degree of model-to-measurement misfit indicates a degree of model structural error and is accounted for in predictive uncertainty analysis. As already indicated, the relatively high density of calibration bores compared with the pilot point density, and the lumped river bed and diffuse recharge disposition, is in part responsible for this structural error and misfit. Future refinement of the model with a greater density of pilot points and greater parameter discretisation of the river and diffuse recharge parameters would be straight forward (though this would have a high computational burden) and may be able to reduce this model-to-measurement error significantly.

A selection of 11 sets of observed versus model simulated water levels is given in Figures 126 to 136. A plot of simulated vs. observed groundwater levels for all the bores is available on request through the authors and via the UWSRA website.

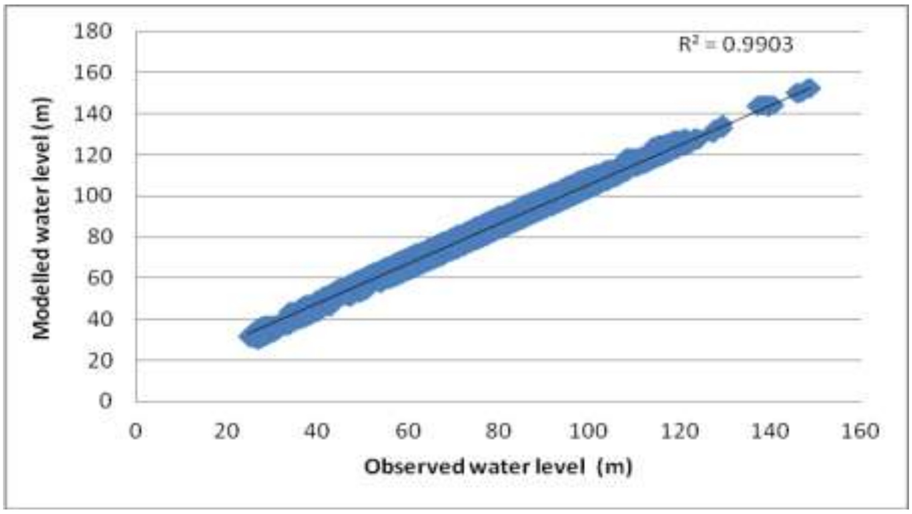


Figure 125: Predicted vs. observed water level for all the bores across all stress periods.

Further detailed discussion on the model-to-measurement errors is found in the model limitations section. The larger model-to-measurement errors generally occur where the model is too simple in its structure (e.g. the single layer, when in fact some interaction with the underlying sandstone aquifer is possible) and parameterisation (e.g. the pilot point density, and river and diffuse recharge parameter lumping), to fit all the measured data (e.g. causing model structural errors).

12.3.6.2 River–Groundwater Fluxes

River recharge values were also included in the model calibration dataset. These values comprised IQQM model estimates of the river recharge, rather than field measurements of river losses. The mean average difference and the root mean square difference in the river recharge estimates from the corresponding IQQM estimates was -1473.62 m^3/d and 9002.84 m^3/d respectively. A histogram of the differences in the calibrated river recharge is shown in Figure 137. This figure indicates that in some parts of the valley, the MODFLOW model is estimating more river recharge than the IQQM model. The river recharge estimates from the IQQM model were given less weight in the calibration exercise, as these targets were themselves estimated by a model rather than generated by field measurements, and so cannot be considered a point of truth, but rather an alternative model output.

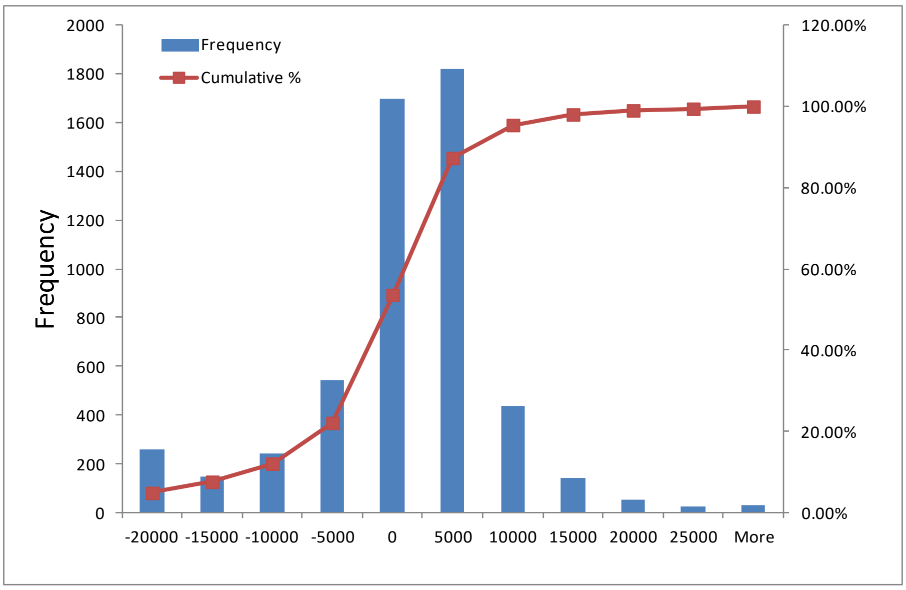


Figure 126: Histogram of errors (log domain) in calibrated river recharge.

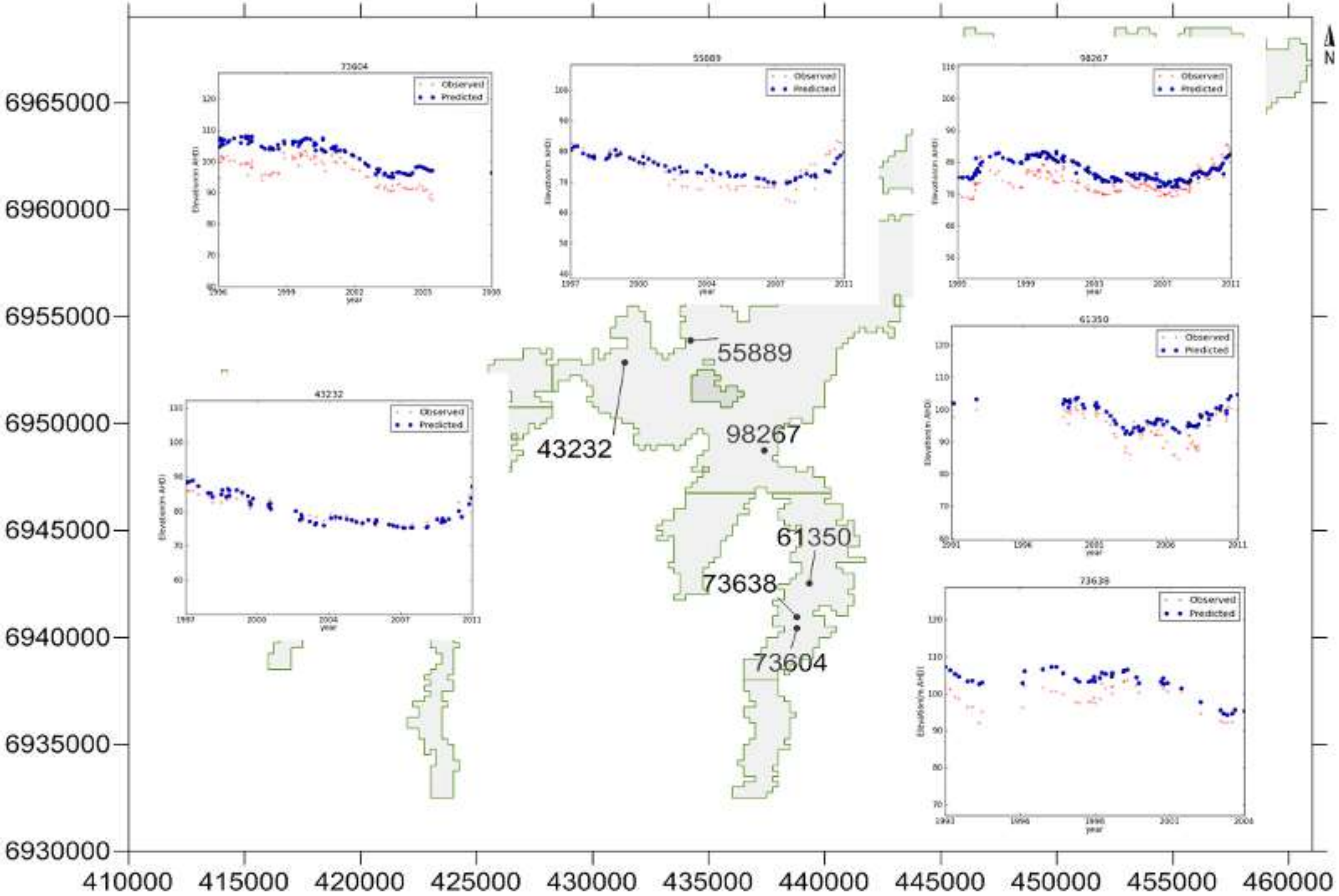


Figure 127: Selected time series plots for six bores and their corresponding location in Lockyer – set 1.

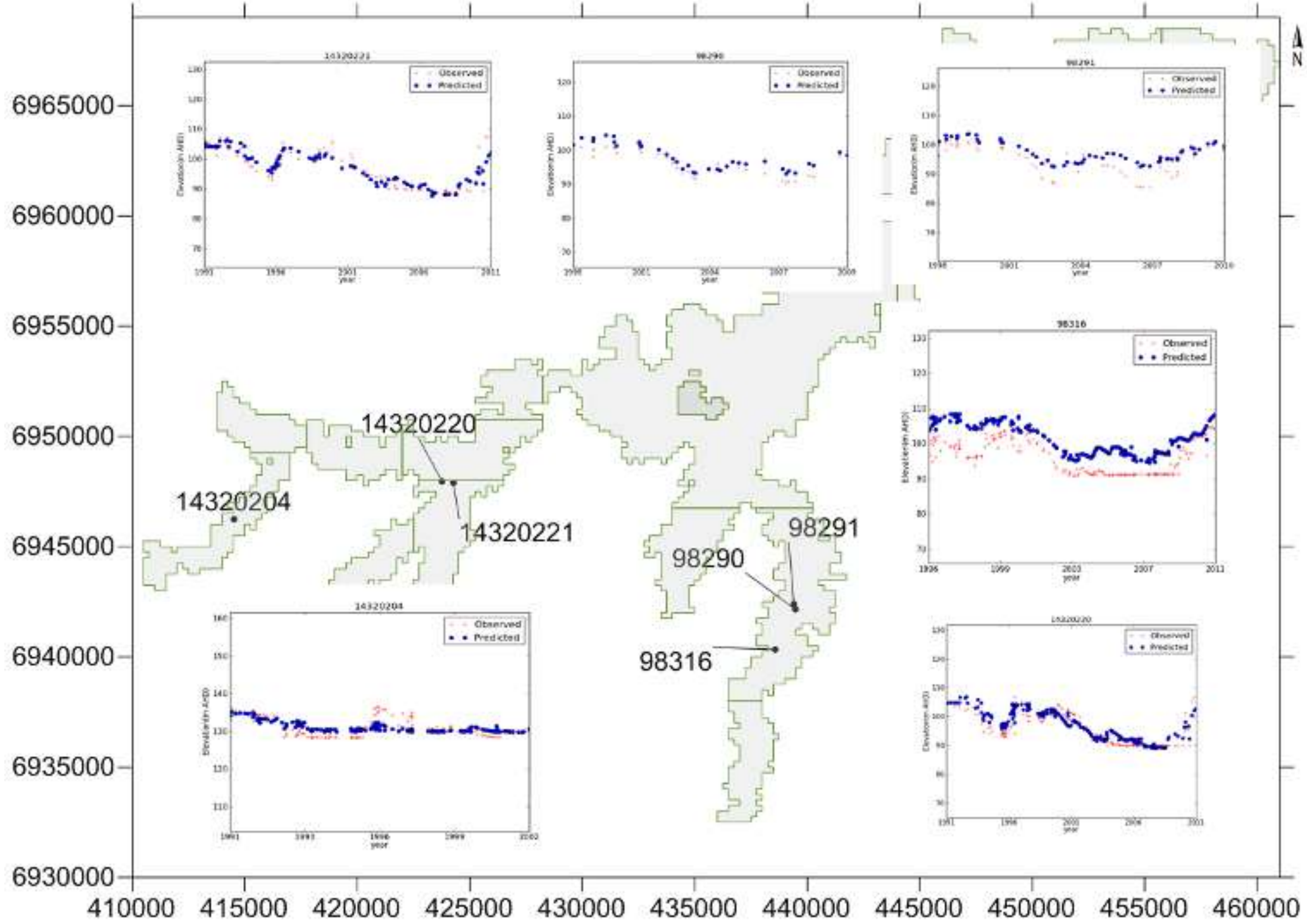


Figure 128: Selected time series plots for six bores and their corresponding location in Lockyer – set 2.

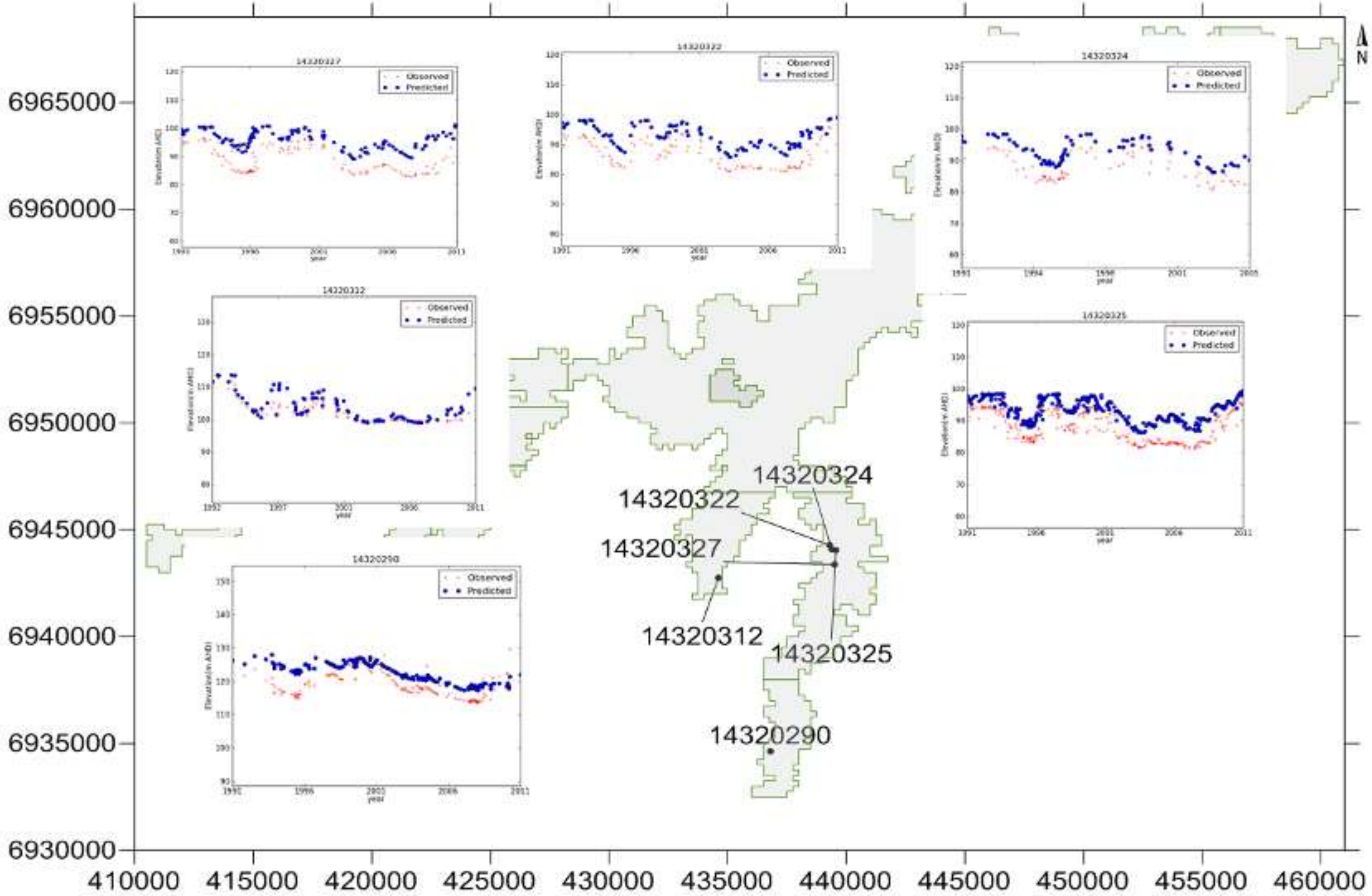


Figure 129: Selected time series plots for six bores and their corresponding location in Lockyer – set 3.

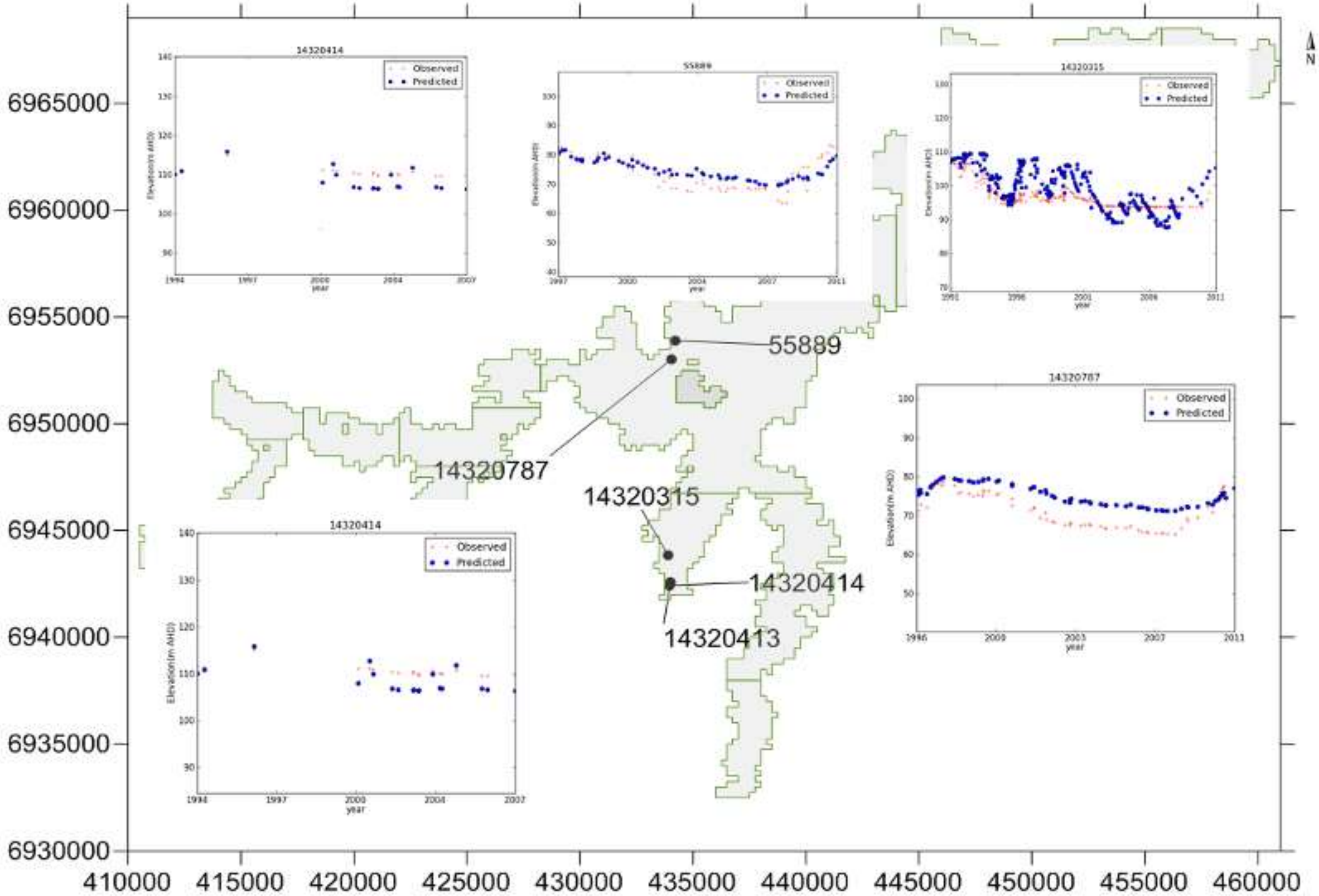


Figure 130: Selected time series plots for bores and their corresponding location in Lockyer – set 4.

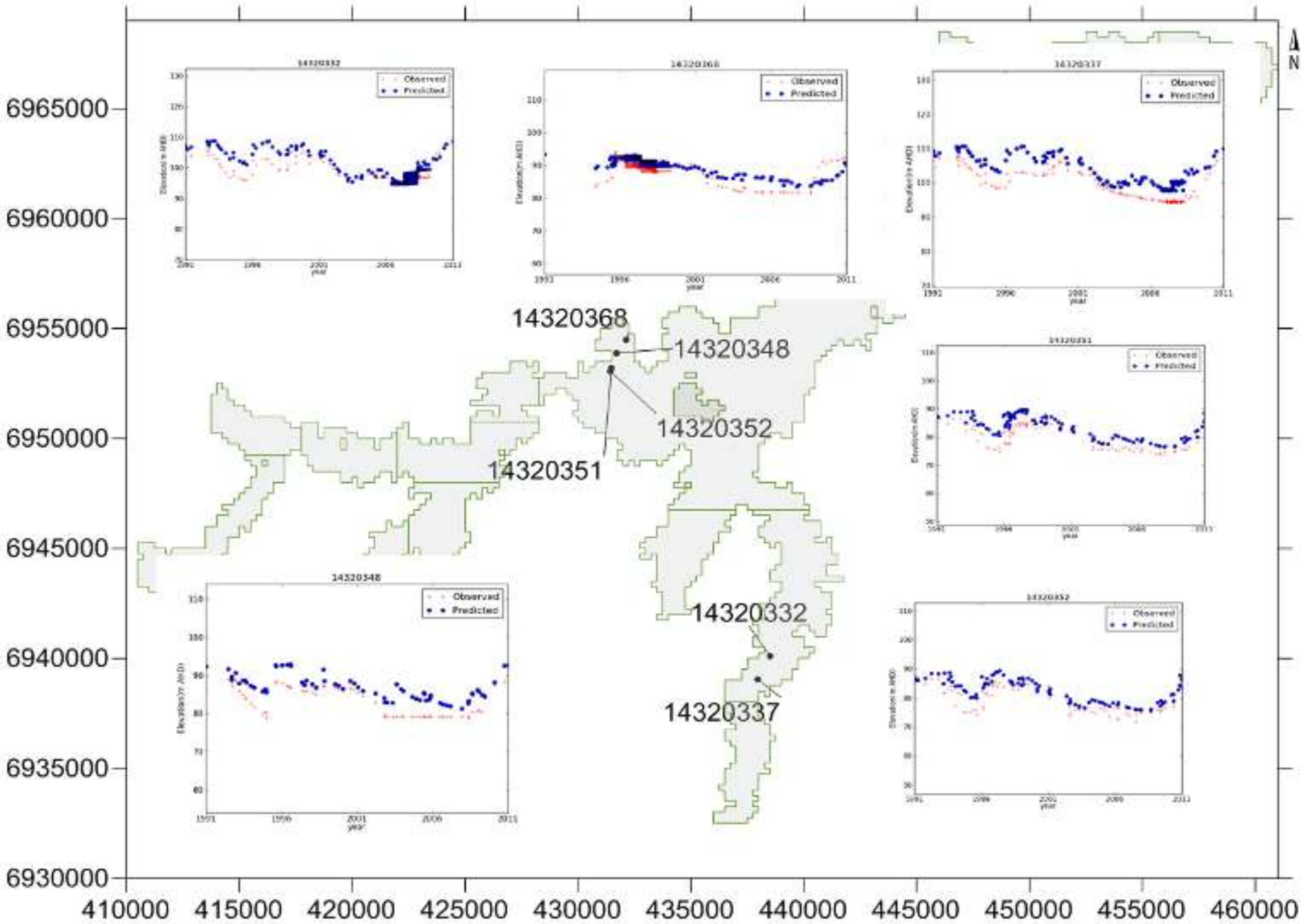


Figure 131: Selected time series plots for six bores and their corresponding location in Lockyer – set 5.

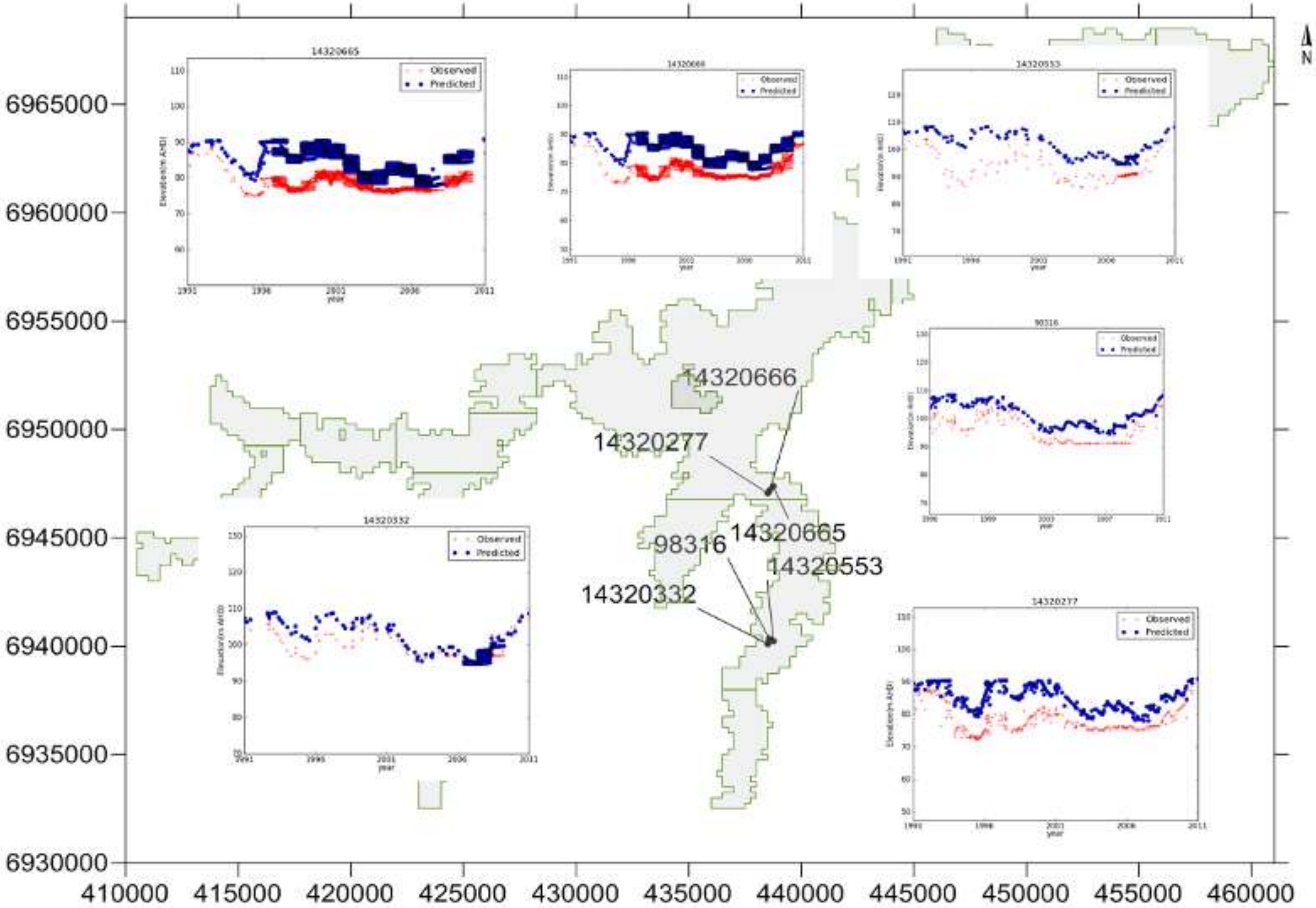


Figure 132: Selected time series plots for six bores and their corresponding location in Lockyer – set 6.

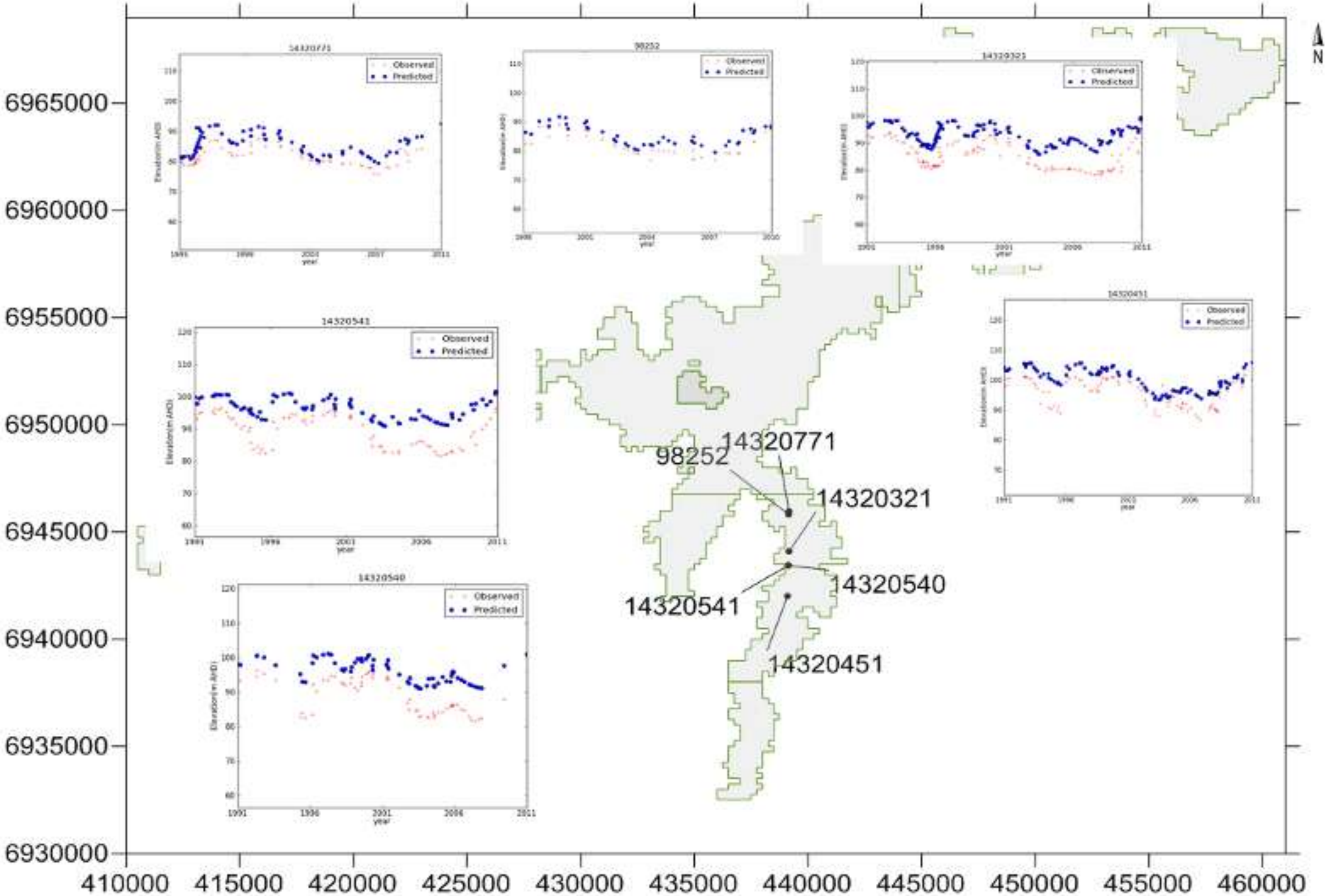


Figure 133: Selected time series plots for six bores and their corresponding location in Lockyer – set 7.

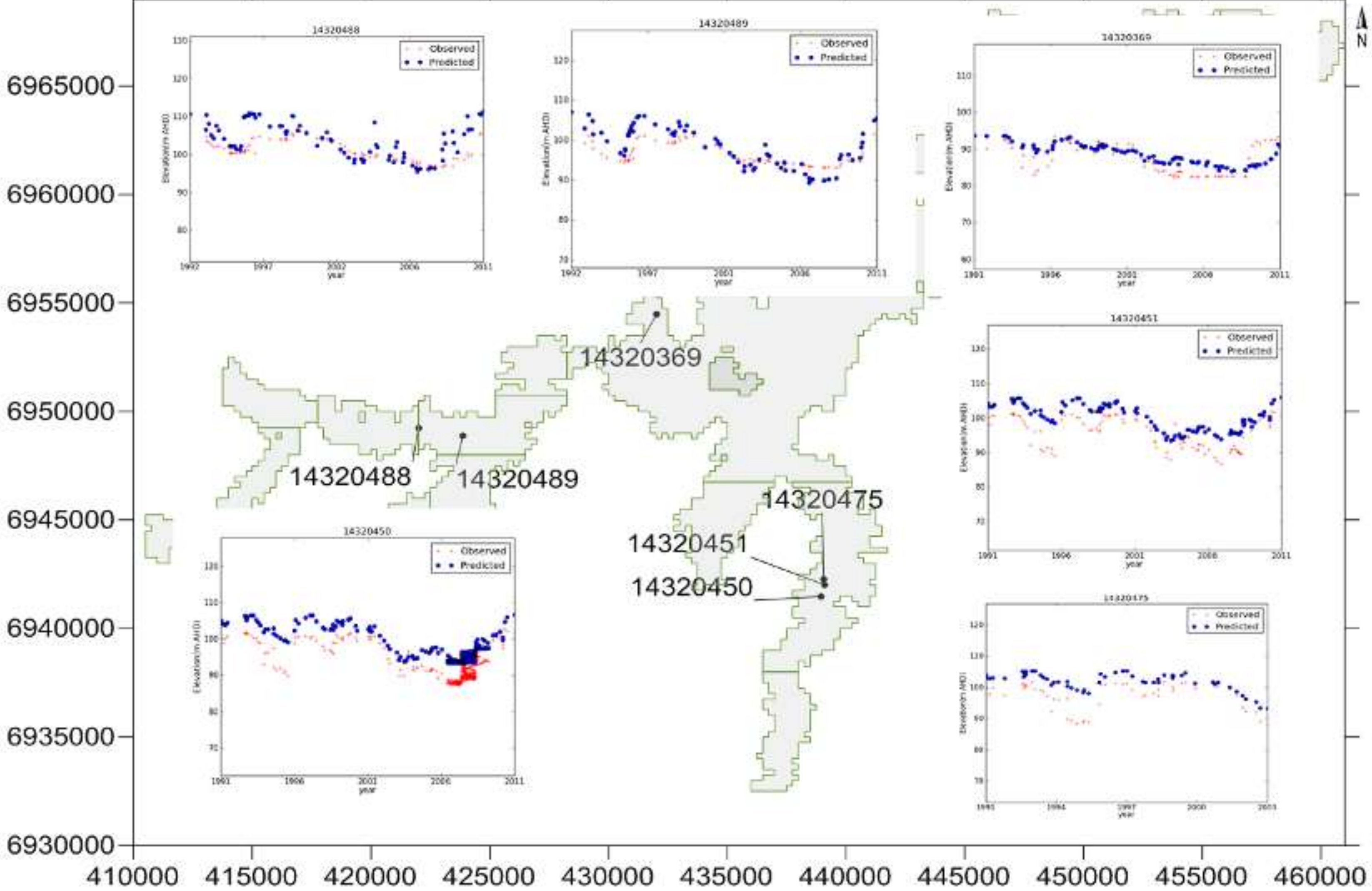


Figure 134: Selected time series plots for six bores and their corresponding location in Lockyer – set 8.

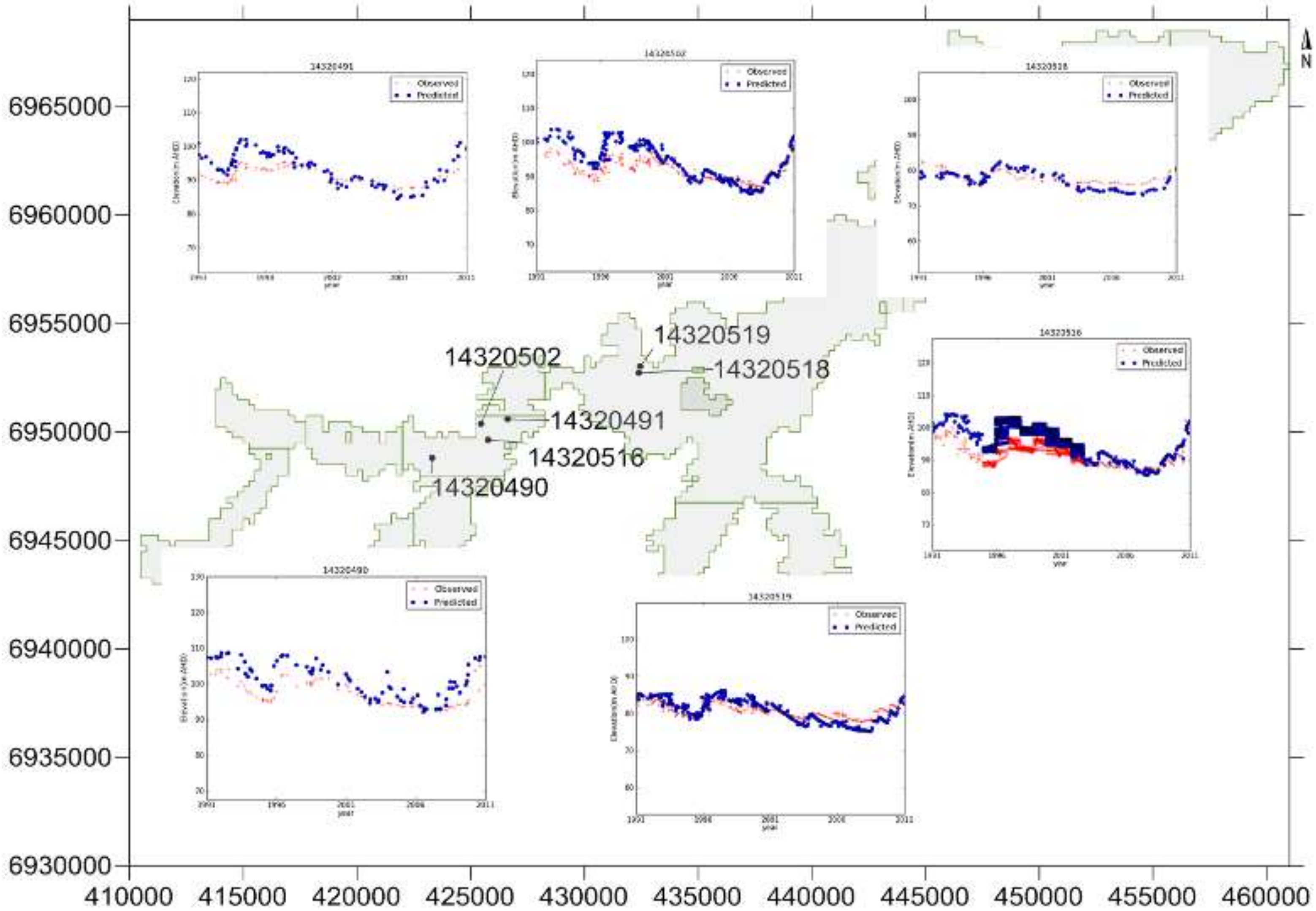


Figure 135: Selected time series plots for six bores and their corresponding location in Lockyer – set 9.

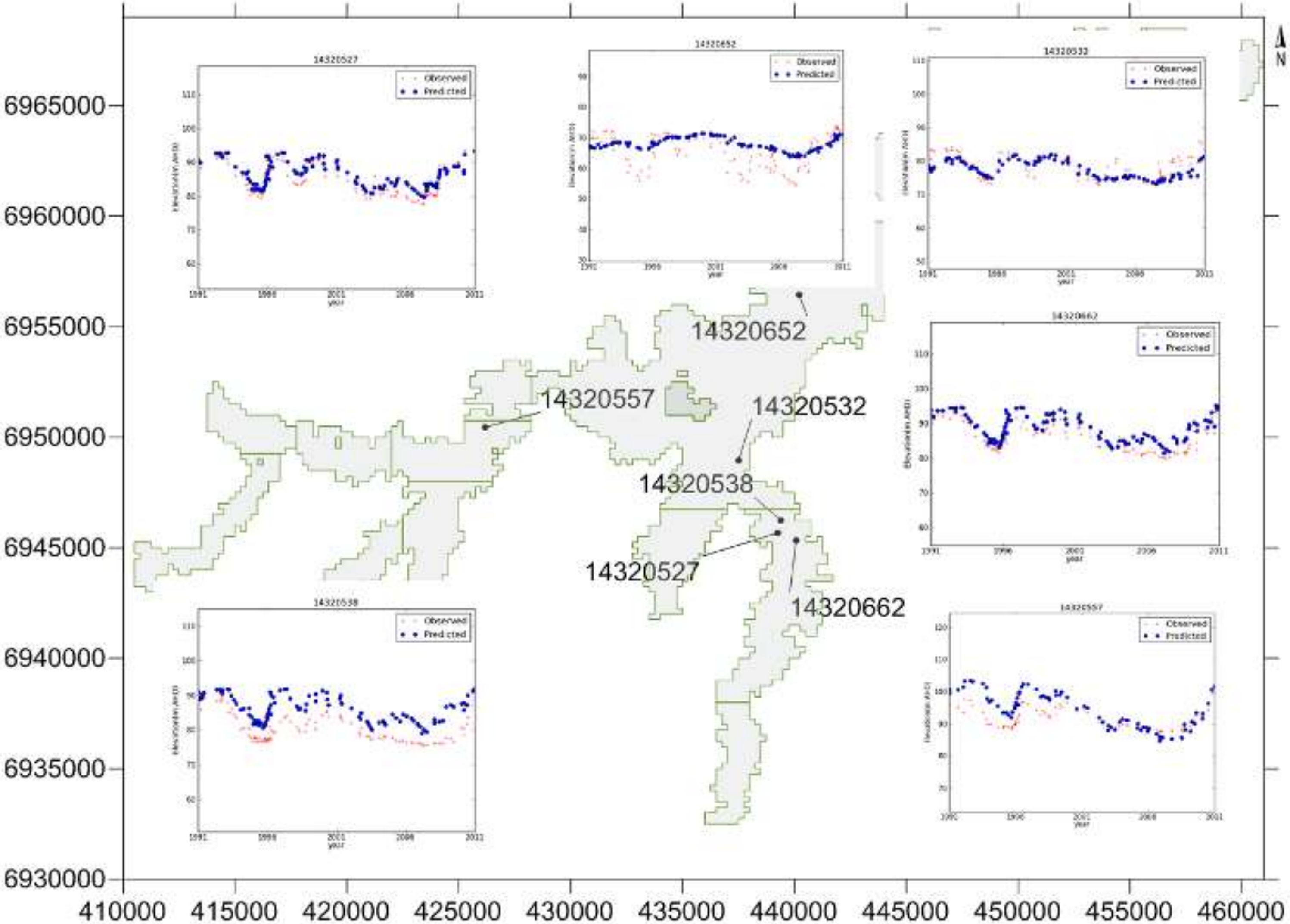


Figure 136: Selected time series plots for six bores and their corresponding location in Lockyer – set 10.

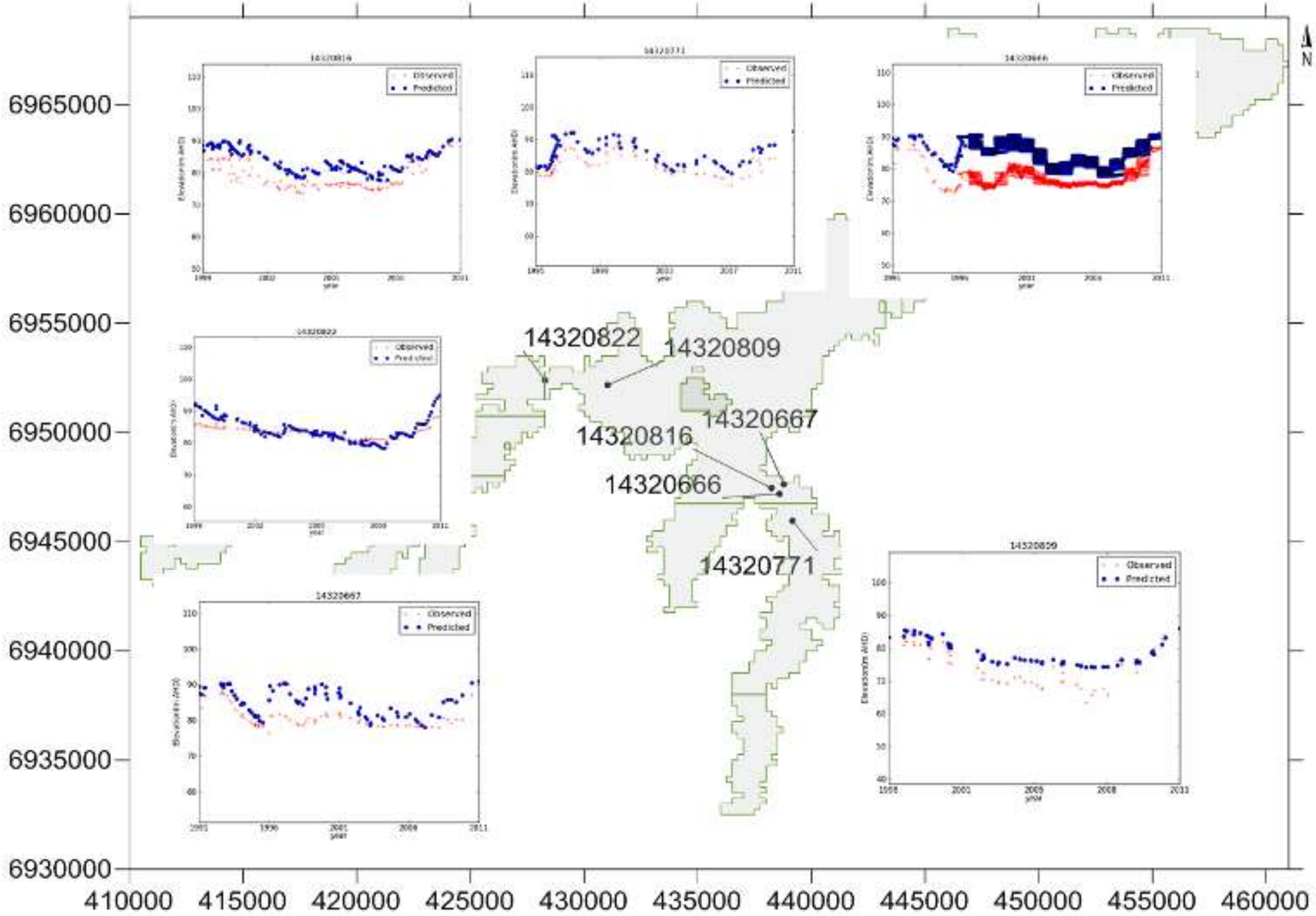


Figure 137: Selected time series plots for six bores and their corresponding location in Lockyer – set 11.

12.3.6.3 Water Budget

The numerical model budget terms indicate whether the model outputs represent a stable numerical solution. In each stress period, the model budget is scrutinised to check that these model budget terms are very small, ideally less than 1%. Table 33 gives the water budget at the end of the calibration run indicating such small errors, which were consistently observed throughout the model simulation.

Table 33: Volumetric budget for the entire model at the end of 5 time steps in stress period 246.

Cumulative Volumes (m^3)		Rates for Last Time Step (m^3/d)	
IN:		**IN:**	
STORAGE	3.96E+08	STORAGE	13072.12
CONSTANT HEAD	380738	CONSTANT HEAD	0.6833
WELLS	839432.7	WELLS	148.413
RIVER LEAKAGE	9.9E+08	RIVER LEAKAGE	136873.6
RECHARGE	7.10E+07	RECHARGE	1986.02
TOTAL IN	1.46E+09	**TOTAL IN**	63653.94
OUT:		**OUT:**	
STORAGE	4.94E+08	STORAGE	20058.81
CONSTANT	2.18E+08	CONSTANT	38157.98
WELLS	5.35E+08	WELLS	35087.43
RIVER	2.12E+08	RIVER	58776.65
RECHARGE	0	RECHARGE	0
TOTAL OUT	1.46E+09	**TOTAL OUT**	**152080.9**
IN-OUT	-512		-1.56E-02
PERCENT DISCREPANCY	**0**	**PERCENT DISCREPANCY**	**0**

12.3.6.4 Numerical Model Limitations

In addition to the model conceptualisation limitations already discussed, specific shortcomings of the current numerical model are now identified.

The detail of the river and diffuse recharge components of the model were varied in a number of ways, and a separate calibration exercise was conducted for each of these conceptualisations. The final conceptualisation was selected because it best reflects the observed dynamics of rapid river recharge of groundwater close to the river recharging zones, and the dampening of this river recharge with increasing distance from the river. The river bottom elevation was set to 5 m below the land surface elevation to allow some flow back from the aquifer into the river. In addition, the diffuse recharge flux was constrained to be no more than 0.2 of that calculated by the HowLeaky modelling.

In the areas where there is creek recharge occurring, the dynamics of the water levels are robust. However, away from these locations, the model is often predicting higher groundwater levels than actually observed, despite the reasonable good dynamic response. For example, in the Central Lockyer region observation values for bores 14320517 and 14320503 are too high. Possible reasons for these discrepancies relate to errors in the calculation of river stage, and possibly also the initial head distribution. The river stage estimates were based on simple linear combinations of the two nearest stream gauging point records. However, these points are sometimes in areas of quite distinctly different recharge mechanisms. In such areas, for example in the Upper Lockyer Creek area (where groundwater levels are too low) and the Central and Lower Lockyer (where groundwater levels can be too high), these river stage heights should be determined on the basis of which stream gauging point best represents the groundwater level fluctuations, rather than on a linear combination of stream gauging point data. In addition, there are issues in some upstream sections such as Upper Tenthill

Creek and Upper Flagstone Creek where it is clear from actual groundwater level response in monitoring bores to creek flow events that recharge from the creek is high. However, the model is unable to replicate this response and it appears tied to the fact that the area is located well upstream of a stream gauging point and how relative elevations of stream level and groundwater level are represented in these areas.

The initial heads distribution appears to cause a problem in many areas away from the creeks, where the initial heads assigned seem to be much higher than they really are. This appears to be a hangover from the use of a number of bores early on which were not typical of the main alluvial aquifer. While these bores were removed from the calibration process, the elevated groundwater level they supplied to the initial heads determination may well continue to cause problems. The left bank of Lockyer Creek between Glenore Grove and Kentville is one of a few examples. Perhaps in this area the initial heads at places near bores 14320084, 14320086, 14320087, 14320101, 14320480 are affected by contouring using the high elevations of those bores in the Morton Vale area (recall that a number of these bores were removed from the model) which are not connected to the alluvial gravel system. These incorrect initial heads tend to put too much water in the system at the start of the simulation period, and thereby compromise the calibration process.

Despite these remaining issues, this selected conceptualisation represents the best of those trialed, in that it reflects the dynamics of the river recharge better than the others. It is also important to recognise these outstanding model limitations when considering the estimates of top-up import volumes discussed in the following section.

12.4 Optimal Import Volumes

One of the two main applications of the calibrated model in this work was to demonstrate how optimal import volumes of PRW into the Lockyer Valley may be estimated. In this work we calculated the volumes that would be required to ensure that a selected groundwater level surface was maintained, where this surface could be considered to represent an environmental and/or water supply security target.

The selected target groundwater level surface discussed was one of many possible selections and is used to demonstrate a method that could be adopted if other water level targets were explored in future (Figure 138). This target water level corresponds to the modelled water level surface of January 2000, which has been chosen to represent a period of average groundwater levels. In work carried out by Bleakley (2011) it was estimated that in the period from June 1999 to June 2000 the aquifer in the Central Lockyer was in the order of 70% full.

The process used by DNMR to estimate aquifer storage condition is explained in Tenthill Volumes Determination (Bleakley and Boreel, 2012). A number of representative bores are used to compare the average depth of saturated aquifer at any point in time with the average maximum depth of saturated aquifer that occurred in those bores under highest water level on record conditions.

Using this process, Bleakley (2011) provided annual estimates of aquifer storage in the Central Lockyer. It was noted that in a period from 1987 to 2011 in the Central Lockyer the lowest storage level was in June 2007 when the aquifer was 31% full, and the highest storage level was in June 1990 when the aquifer was 91% full. In June 1999 and June 2000, the aquifer storage was described as 69% and 70% respectively. This period represented an average or marginally above average aquifer storage condition for the period reviewed (1987–2011).

Hence, the period of January 2000 has been adopted as an average groundwater level period for the Lockyer Valley, for the purposes of estimating top up volumes.

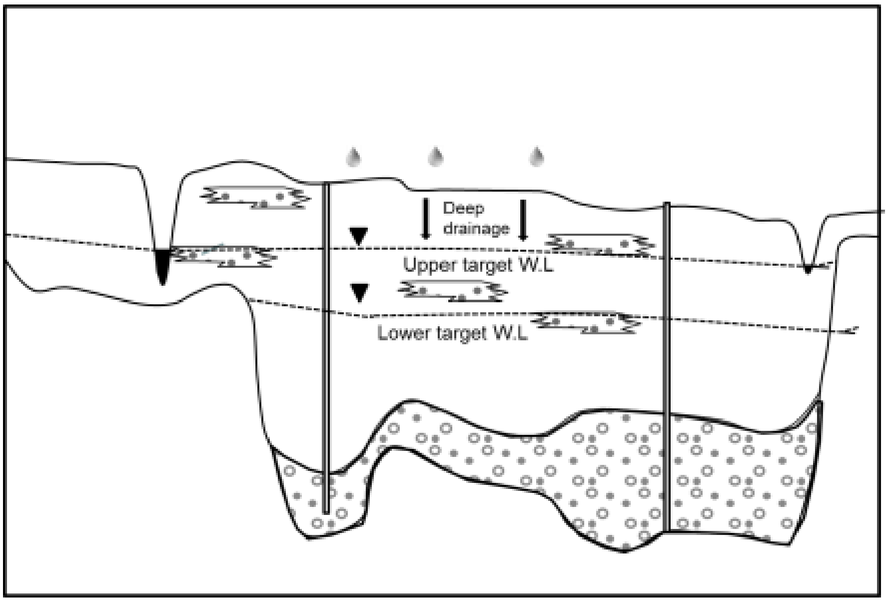


Figure 138: Illustration of the target levels to estimate import volumes.

12.4.1.1 Method

The import volume estimates required to maintain average water levels throughout the aquifer system were computed using a modified version of the MODFLOW GHB (Moore *et al.*, 2011). This modification ensures that the head-dependent boundary only allows flow into each cell of the model grid until the specified water level target is met, and the boundary is deactivated if water levels rise above this level, so there is no reverse flow. Therefore the volume of water required to top-up the storage is easily calculated.

Note that this approach provides the estimates of the minimum import volumes required to top-up the aquifer, assuming that top-up could occur in every model cell, and was undertaken to assess the general feasibility of PRW top-up. The approach does not consider the impact of different re-injection locations, well density and well design, and other considerations required for the PRW import, which would be undertaken within a formal design exercise. If the import volumes were distributed via reinjection wells, the reinjection rates would be dependent on the location of the reinjection wells. The volume required to top-up the aquifer to the target levels throughout the aquifer is lessened when the injection wells are closely spaced compared to when the wells are placed far apart, as indicated in Figure 139; however, this closer spacing would incur a higher well installation cost. The specific design of the re-injection well field and the import volumes are dependent on detailed design factors which are beyond the scope of this work. However, the results provided here give the top-up requirement to the two water level targets under the assumption of a well spacing with a minimum of 250 m distance, corresponding to the grid size of the numerical groundwater.

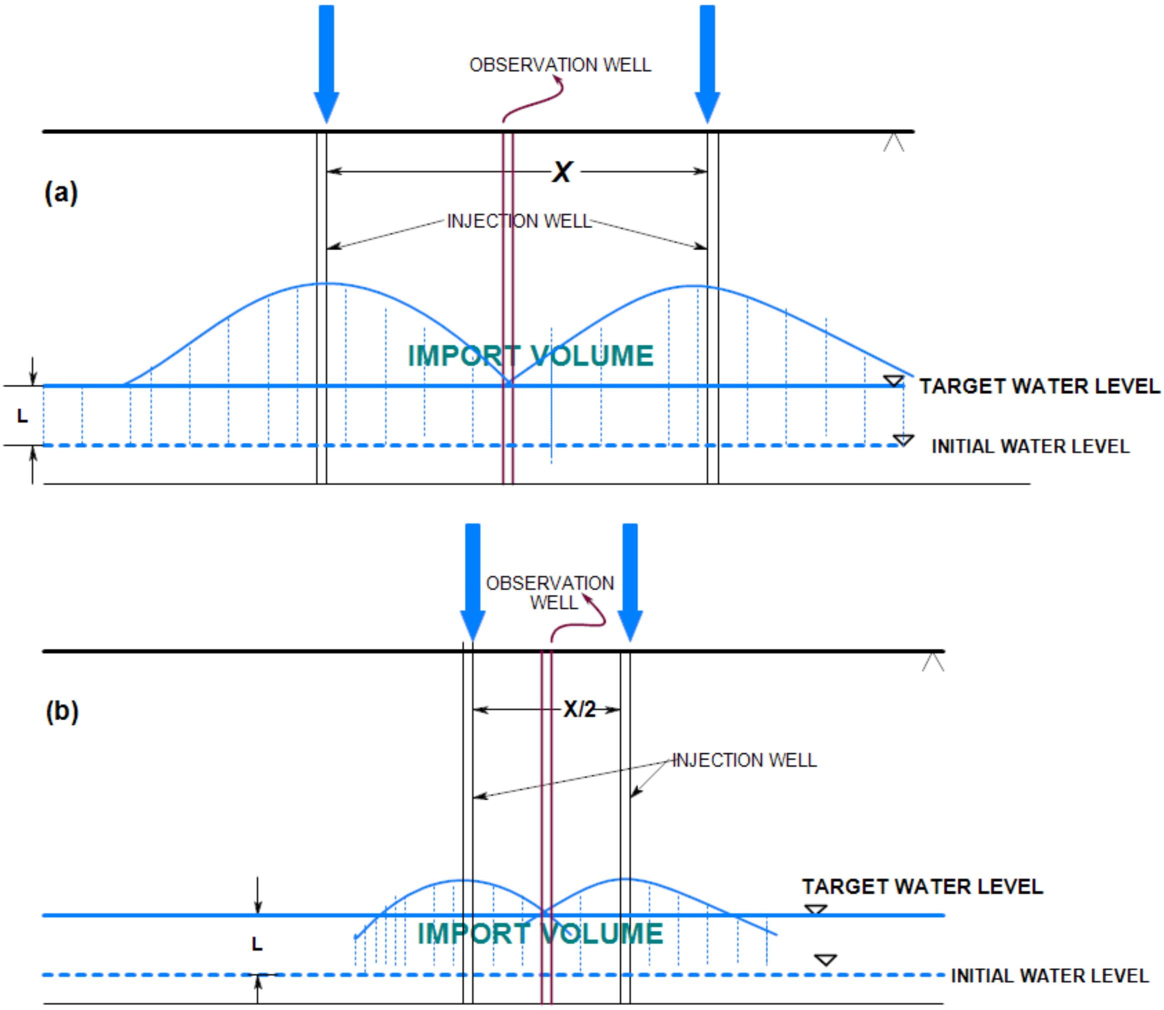


Figure 139: Illustration of the dependency of import volumes on the spatial distribution of wells.

12.4.2 Results

The time series of import volumes required to maintain the selected groundwater level surface was computed for the model simulation period. It was assumed that the pumping, diffuse recharge and river stage levels of the calibration model continued in this modelled import volume simulation. In each model stress period, the top-up volumes that would be required to meet this target water level were calculated. If differing pumping rates occur in the future, commensurate changes to the top-up volumes would be required.

One outcome of adding an import volume into the aquifer to maintain average groundwater levels is the reduction of the hydraulic gradients between river and groundwater levels. The reduced river-aquifer hydraulic gradients cause less river recharge to occur in the top-up simulation than occurred in the historical simulation. Therefore, this reduced river-to-aquifer recharge creates greater top-up volume requirements than if historical river recharge is maintained. To illustrate this issue, Figure 140 depicts both the calculated top-up volumes that would be required if historical river recharge were maintained, and the top-up volumes required when average ‘topped-up’ groundwater levels are maintained over the simulation period.

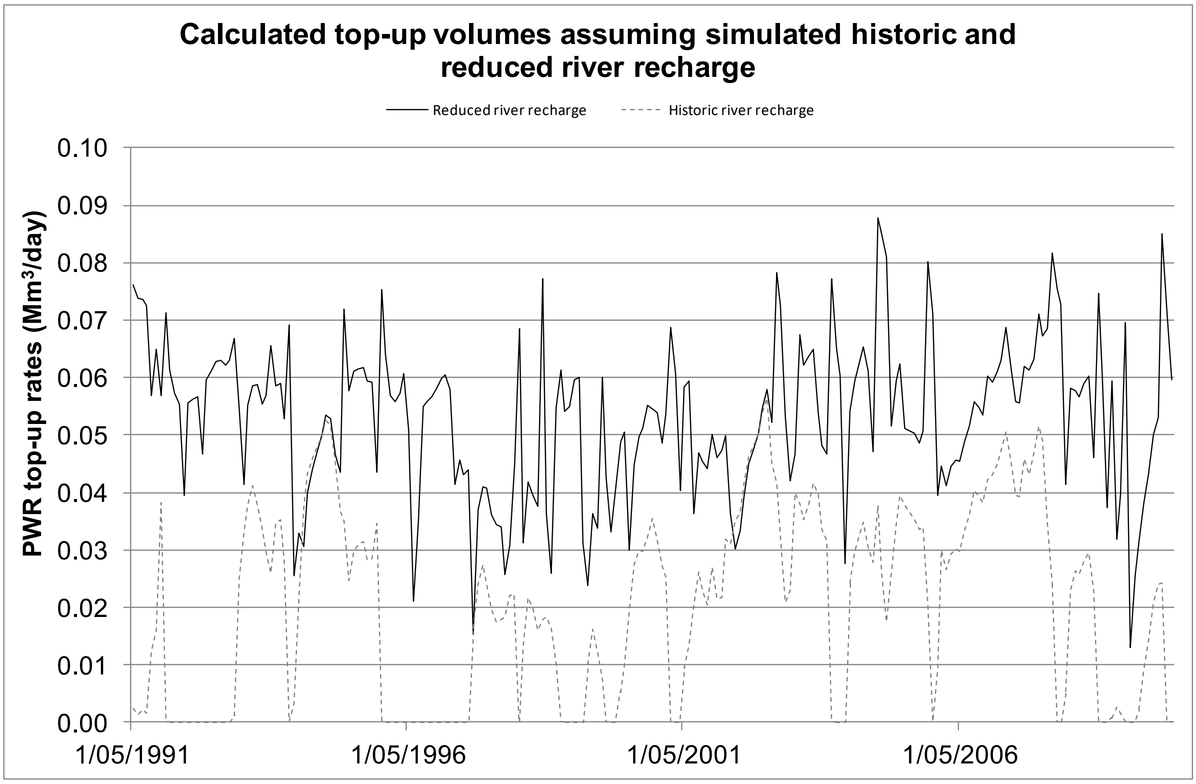


Figure 140: Time series plot of the import volumes required to top up to different target water levels.

The top-up volumes calculated with the altered river recharge depicted in Figure 140 were calculated in three steps:

a. Firstly, the top-up volumes simulated by the model were calculated (which resulted in a top-up volume of 29 GL/a), and then an adjustment was made.

b. The adjustment, which is essentially an imposition of expert knowledge, was used in the context where there was a simulated reversal of the river–aquifer gradients, and resulting high flows from groundwater into the river. The adjustment took the river fluxes, calculated above assuming a constant top-up volume, and replaced these with half of the simulated historical river recharge; the top-up volumes were then adjusted accordingly.

c. The third adjustment was where we simply adopted the full historical river recharge.

This adjustment was adopted to address concerns around the accuracy of the river stage heights used in the model inputs, and the concern that the resulting magnitude of groundwater to river flows was misleading, given that there was no historical evidence for such large groundwater to river flows (within the model area), even during periods of sustained high groundwater levels (pers. comm, Ashley Bleakley, Principal Project Officer (Hydrology), Water Services South East, December 2012).

The model calculated long-term top-up averages of 0.02 Mm^3/day (or 8.59 GL/a) and 0.056 Mm^3/day (or 20.66 GL/a), based on the two different river recharge assumptions depicted above, namely, with historical river recharge and with top-up altered river recharge respectively. This range of top-up volume values can be compared with the average groundwater pumping volumes over the same simulation period, which averaged 0.089 Mm^3/d (or 32.8 GL/a). While a reduced river recharge during periods of high groundwater levels would occur, it is unclear to what extent this would affect the top-up volumes. However, the minimum and maximum time series depicted in Figure 140 can be considered a lower and upper limit. This range reflects the outcome of model conceptualisation/structural errors. These are distinct from the model parameter errors (discussed in the following section), which instead relate to the uncertainty in the model parameters given a selected model conceptualisation or structure. Both error sources need to be considered together.

In terms of the optimum spatial distribution of top-up water, this would be best directed to the current pumping well locations. For example, Figure 141 indicates the pumping volumes for the model stress period relating to 05/05/1992 (a period with low recharge, high groundwater levels and high pumping rates), and the top-up volumes could target these locations of higher pumping rates.

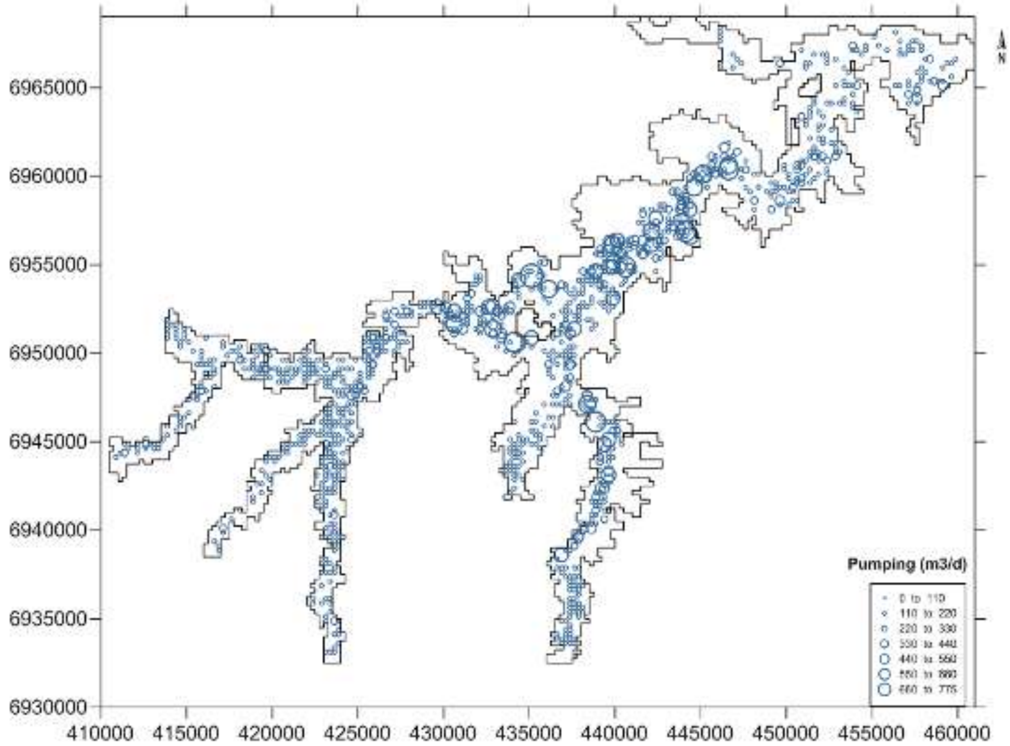


Figure 141: Spatial distribution of pumping in the model simulation period corresponding to 05/05/1995. Larger circles correspond to higher pumping rates, and smaller circles correspond to lower pumping rates.

12.4.3 Predictive Uncertainty and Data Worth Analysis

The predictive uncertainty of the annual potential PRW top-up volume caused by model parameter uncertainty was estimated using linear uncertainty analysis techniques. As noted above, these calculated uncertainty ranges need to be combined with the model conceptualisation/structural errors discussed in the preceding section. The requirements for the parameter predictive uncertainty analysis include:

1. a prediction of a groundwater response and a physically based model used to make that prediction, where the model contains sufficient 'real-world' hydraulic process detail for which the prediction is sensitive, for example, the total top-up volume over the model domain and simulation period;
2. parameter identifiability values (refer to discussion earlier in this chapter);
3. a description of the probability distribution of the error of measurements used in the calibration dataset is required and must include model structural error as evidenced by model-to-measurement misfit;
4. a description of the probability distribution of both fixed model inputs and estimated parameters prior to imposition of calibration constraints (e.g. *a priori* distributions) which are consistent with that listed in Table 32; and
5. calculated sensitivities of the prediction to the measurements, model inputs and model parameters. (refer to Predunc utilities in Doherty, 2012a, and Moore and Doherty, 2005 for more information about this methodology).

The predictive uncertainty was calculated for the total import volumes over the entire model domain for the 1991–2011 model simulation period. The post-calibration total uncertainty standard deviation is around 10% of the predicted top-up volumes. This error must be considered in addition to the structural errors already discussed.

Note that these linear uncertainty analysis techniques can only be considered approximate because of their linearity assumption; however, their advantage is that they can be performed rapidly, allowing comparative analyses of contributions to uncertainty, such as the worth of alternative data and information sources, to be undertaken. Two types of comparative analysis are now described.

The first analysis assesses the extent to which acquisition of better knowledge of different types of parameters and model inputs would reduce the uncertainty of the prediction of current interest. This analysis is applied to the model with and without imposition of calibration constraints, and therefore pertains to the pre- and post-calibrated model. (Recall that the analysis requires only sensitivities of model outputs with respect to parameters and not the values of these model outputs or of the parameters themselves.)

There is some reduction in prediction uncertainty achieved from the imposition of calibration constraints evident when comparing the pre- and post-calibration contributions to the predictive standard deviation. The normalised contribution of each parameter group to the uncertainty of this prediction is depicted in Figure 142. This plot indicates that the prediction is principally sensitive to hydraulic conductivity and river bed conductance terms. It also indicates the contribution made by uncertainty of water abstraction quantities (estimated at potentially 50% error in the pumping rates at the 95% confidence interval), and depicts correctly how this contribution is not improved through the calibration process, as there is no information in the calibration data set to constrain this quantity.

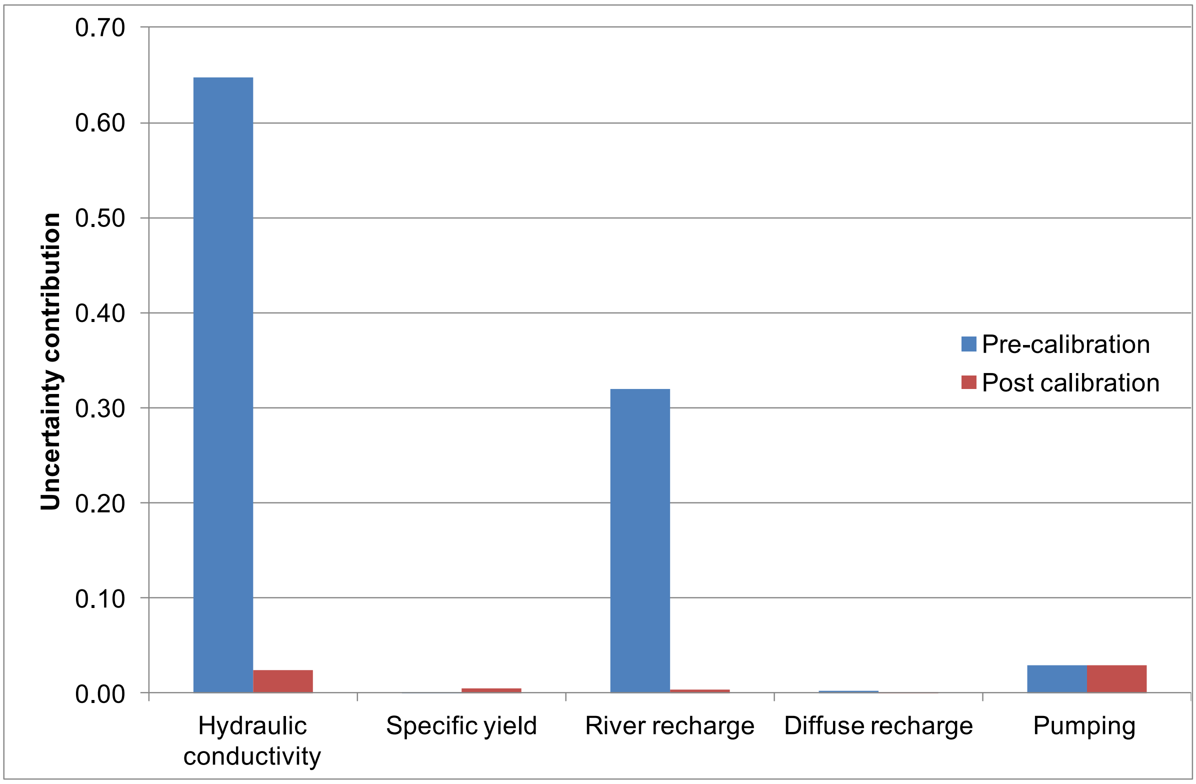


Figure 142: Pre- and post-calibration parameter group contributions to prediction uncertainty.

The second analysis involves assessing data worth in terms of i) the increase of the top-up prediction uncertainty that occurs if the observation is removed (i.e. starting conditions assume all observations are available), or ii) the reduction of the top-up prediction uncertainty that occurs by adding the observation to the data set (i.e. starting conditions assume no observations are available). In both of these cases the observations can be existing or nominal future observations, as the actual value of the observation is not used in the uncertainty calculation. In this example, we used the existing groundwater level monitoring network, which is reasonably extensive. The worth of the monitoring data is determined by its effect on the total model predictive uncertainty resulting from the inclusion or exclusion of the monitored information in the model. This effect can be measured in two different ways. One is by computing the total decrease in the predictive uncertainty variance resulting from the inclusion of a monitoring bore into the monitoring data set. The other is by computing the total increase in the variance of the predictive uncertainty resulting from the exclusion of a monitoring bore from the monitoring data set.

The spatial plots of the relative increase and decrease in predictive uncertainty variance resulting from inclusion and exclusion respectively of monitoring bores in the Lockyer are shown in Figure 143 and Figure 144. These plots are indicative of the relative worth of the data collection in different regions of Lockyer Valley in terms of improving the understanding of the system. This information can also be used to reduce the redundancy in the monitoring data collection whereby bores can be shut down at locations where the information obtained is redundant. In both of these figures the uncertainty reduction has been normalised by the number of observations at each bore, so that the analysis depicts data worth in a spatial sense only, rather than also reflecting the actual length of measurement records and the frequency of measurements.

In Figure 143, the uncertainty reduction with each new observation, assuming no other monitoring bores are in use, is depicted. This plot indicates that if there were no data, monitoring in the central Lockyer region and Flagstone Creek area would have the greatest impact in terms of reducing the uncertainty of the top-up prediction. This plot also indicates that if all monitoring data were removed, there is no single location that would significantly reduce the uncertainty of the top-up prediction on its own.

Figure 144 shows the increase in uncertainty that occurs with a one-by-one removal of monitoring locations, assuming that all other monitoring bores are still in use. It can be seen that removal of any single monitoring well would in general have little impact on the top-up prediction uncertainty, except in the lower Flagstone Creek area. This result indicates that in general there is currently sufficient groundwater level monitoring data in the region for the estimation of an average PRW top-up volume. Reasons why the uncertainty would particularly increase if groundwater level records from the lower Flagstone Creek data were removed might include the complicated nature of the junction of the Flagstone Creek alluvium with the Lockyer Creek alluvium and recharge that occurs from Carpendale weir on Lockyer Creek in this area. The analysis was not extended to other types of measurements; however, future analyses could include an assessment of the relative worth of other types of data, for example, river–groundwater measurements or water chemistry data and isotope analyses.

It is important to note that if alternative predictions were being assessed this result of monitoring sufficiency may not occur; as the analysis is relevant only to the specific prediction being explored. Furthermore, because the model prediction represents the sum of top-up volume over the entire model domain, and the model-to-measurement errors are randomly distributed spatially with a normal distribution, these errors cancel out for the aquifer-wide prediction. However, if predictions were specific to a point within the aquifer, we could expect much larger errors in our predictive simulations, and the modelling network may not be considered sufficient for such point predictions, depending on their location.

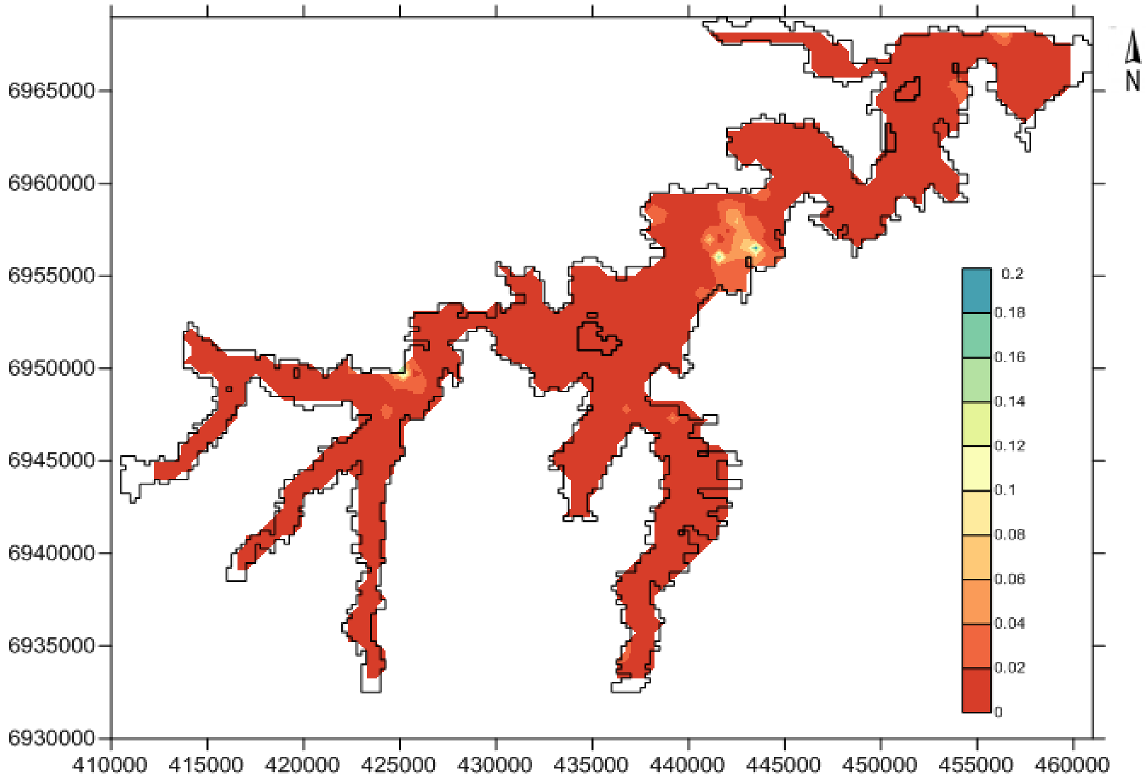


Figure 143: Spatial plot of predictive uncertainty variance decrease resulting from the addition of monitoring information.

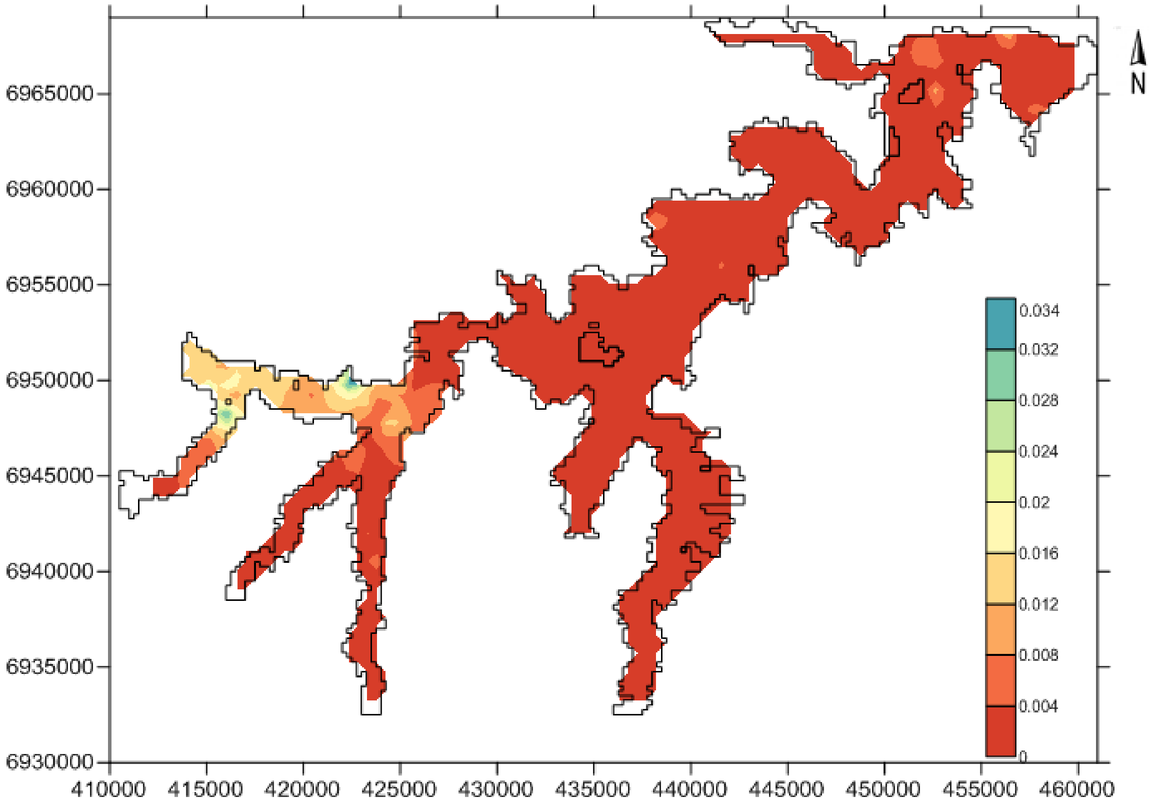


Figure 144: Spatial plot of predictive uncertainty variance increase resulting from the exclusion of monitoring information.

12.5 Conclusions

This chapter describes the extension of a groundwater model development, and its use for estimating top-up volumes required to meet environmental groundwater level targets. The model extension has lengthened the simulation period of the model to include the major aquifer recharge event associated with the January 2011 floods in the Lockyer Valley. The extension also comprised building the physical representation of river recharge into the model.

The single layer construction, its discretisation of using 250 m dimensioned cells, and the comparatively simple parameter discretisation (particularly with diffuse and river recharge parameters) compared to the observations result in a degree of model-to-measurement misfit which is consistent with earlier modelling efforts. This could be improved with a refinement in the model conceptualisation and parameterisation in future modelling efforts. However, the new model does have advantages over the previous model versions for this current exercise, because it provides a more physically realistic representation of the aquifer system (e.g. the rivers represented explicitly). This representation would help future scenario explorations of direct PRW input into the river. It also has the advantage that it allows the exploration of the relative contribution to uncertainty from the rivers versus the diffuse recharge. Furthermore, the model is fit for purpose in terms of demonstration of a methodology for the estimation of the range of PRW top-up volumes that may be required for selected water level environmental targets.

Exploration of other selected targets could be undertaken with the model in the future. Calculations of the top-up volumes required to keep the aquifer system from ever falling below average groundwater levels were undertaken. Using two different model river conceptualisations, simulation estimates of top-up volume ranged between 9 and 21 GL/a would be required to maintain average groundwater levels given the historical pumping record. Predictive estimation error resulting from inadequacies in the parameter estimation solution must also be taken into account. The predictive error resulting from parameter estimation error, was calculated to have a standard deviation of 10% of the estimated values of 9 and 21 GL/a. These calculated PRW top-up volumes can be compared with the 37 GL/a that is potentially available from Seqwater to augment the water resources in the Lockyer Valley and Bremer Valleys. Alternative simulations may be explored in the future to see what targets could be met given the available water. The methodology presented in this chapter provides one way for this process to be undertaken successfully.

The linear predictive uncertainty analysis, though approximate, allowed a number of ancillary analyses that enabled an exploration of which increased knowledge areas and monitoring network data enhance this predictive reliability the most. This method could be extended to optimise monitoring networks under the ‘best return for investment’ principle for different predictions.

13. IMPACT OF CLIMATE CHANGE ON GROUNDWATER IN THE LOCKYER VALLEY

(Sreekanth Janardhanan, Leif Wolf, Catherine Moore, Sebastian Most, Shreevatsa Kodur)

13.1 Introduction

Over the last few decades Australia has observed significant changes in climate, particularly in terms of significant warming and changes in rainfall patterns. These changes are consistent with worldwide climate change trends. The impact of the changing climate on water resources has been extensively studied, both nationally and internationally, with particular focus on surface water resources. However, potential impacts of climate change on groundwater resources can also be expected to be significant, particularly where aquifers are heavily allocated and/or are vulnerable to changes in recharge patterns. In a heavily allocated system, any reduction in the available water, resulting from a varied recharge due to climate change, can affect the groundwater use in terms of quantity and quality of the water entitlements. This impact is even larger for unconfined aquifer systems, where responses to the variation in recharge are more quickly reflected on the available water resources. The Lockyer Valley is both heavily allocated and taps a predominantly unconfined aquifer, and hence is vulnerable to any climate change impacts on aquifer recharge.

Climate change impacts on the groundwater recharge and groundwater resources across different aquifers in Australia have been assessed in a study commissioned by the National Water Initiative (Barron *et al.*, 2011). The median future climate at 2030 and 2050 projects a decrease in diffuse recharge across most of the west, centre and south-east of Australia, while increases in recharge are projected across northern Australia and a small area of eastern Australia. The study identified 14 aquifers of priority around Australia based on their vulnerability, sensitivity to climate change and regional significance. The Toowoomba Basalt near the Lockyer Valley was identified as a priority aquifer system. The study identified annual rainfall as the most important parameter for describing the potential change in recharge; however, temperature, solar radiation and vapour pressure deficit can influence recharge, particularly in low recharge arid zones.

The primary impact of climate change on groundwater resources caused by the change in the recharge volumes and patterns is assessed in conjunction with effects on pumping, evaporation, river discharge and head and flux boundary conditions. The impact of climate change on the groundwater resources is also dependent upon the hydrogeological setting of each aquifer. Hence each aquifer has to be studied using rigorous modelling studies to identify the impact of climate change on the groundwater resource.

In this chapter the potential impacts of climate change on the alluvial aquifer system of the Lockyer Valley are investigated in two ways: i) using a simple model, the 'Eigenmodel' approach; and ii) using a three-dimensional, MODFLOW-based numerical groundwater modelling approach. Previously, a study by Gooda and colleagues (2011) used the integrated quality quantity model (IQQM) to assess the impact of climate change projections from 11 global circulation models (GCM) and 12 dynamically downscaled GCMs, using the CSIRO Cubic Conformal Atmospheric Model (CCAM). That study showed that the majority of downscaled GCMs show significantly lower inflow than the original GCM projection and indicated that the water supply planning in SEQ may underestimate the impact of climate change on available water resources.

13.2 Methodology

Climate change scenarios are generated for each of the selected GCMs by scaling the historic climate sequences. In this manner, climate change counterparts for the historic rainfall and evaporation were generated. IQQM estimates of the river recharge and HowLeaky estimates of the diffuse recharge were used to estimate how these changes to rainfall and evaporation would change groundwater stresses. Two different approaches were used in this project to assess the climate change impacts on the

Lockyer groundwater system. One approach makes use of the groundwater flow model described in the previous chapter, which is used to simulate the groundwater impacts corresponding to specified climate change scenarios. The second approach makes use of a simple or 'reduced' model, which is based on an Eigenmodel tool, to predict the groundwater impacts.

13.2.1 Climate Change Scenarios

The availability of high-resolution dynamically downscaled climate models for SEQ has allowed an assessment of the sensitivity to the spatial resolution of climate change projections on water availability. The impact of climate change on the groundwater resource of the Lockyer Valley is studied using three scenarios derived from three global circulation models (from CSIRO's CCAM). The three dynamically downscaled GCMs are CCSM 3.0 (referred to as 'CCSM3'), MPI ECHAM 5 (referred to as 'ECHAM') and GFDL CM 2.1 (referred to as 'GF'). These three models correspond to the second wettest, median and second driest climate change scenarios analysed by the UWSRA project on climate change scenarios in SEQ.

Future climate sequences for rainfall and evaporation are produced for each of the three GCMs (Gooda *et al.*, 2011), by scaling the long-term historic daily climate sequences (1889–2000). For each model, the scaling factors are calculated on a monthly basis, by determining the per degree of global warming response for each climate variable, multiplied by the best estimate of the change in temperature for the year 2050 under emissions scenario A1FI (from Table 4.3 of the *Climate Change in Australia* Report, CSIRO, 2007). This is consistent with the methodology that has been adopted by DERM for climate change analysis throughout Queensland, and provides a platform for comparing projections from GCMs that have been run under differing emissions scenarios.

13.2.2 Eigenmodel Approach

Eigenmodels are mathematical tools that employ the concepts of eigenvalues and eigenvectors ('eigen components') from linear algebra. Bidwell and Morgan (2002) employed eigen components to simplify the mathematically complex system of equations solved in a numerical groundwater modelling problem to allow rapid assessments of system responses (e.g. in terms of groundwater level changes). The eigenmodelling results in a simpler representation of the aquifer system, with some loss of flexibility in defining aquifer boundaries. However, this simplified approach offers a rapid method to represent the integration of heterogeneous properties of an aquifer and to assess the relative impact of various changes to model stresses. In this example, an Eigenmodel was used on a single well-by-well basis, but it could equally have been used with multiple wells analysed simultaneously.

To allow for a rapid assessment of climate change impacts, an Eigenmodel tool (Bidwell and Burberry, 2011) was calibrated to the water level records of three bores in the Lockyer Valley, during a 21-year period (refer to further details in Most, 2012). The Eigenmodel tool predicts groundwater levels using climate and surface water levels as input parameters. The impact of downscaled climate changes scenarios on surface water levels in the Lockyer was estimated by DERM using IQQM (Gooda *et al.*, 2011), and used as an input to the Eigenmodel tool. The analysed groundwater level changes correspond to the three climate change scenarios CCM3, ECHAM and GF. Climate change impacts on groundwater level fluctuations in three bores, namely 14320297, 14320528 and 14320487, were analysed using the Eigenmodel approach. For brevity, only the results of analyses related to bore 14320528 are shown here, but the three bores did yield consistent behaviour in the climate change simulations.

Figure 145 depicts the overall average water level changes calculated with the Eigenmodel tool using the three climate change scenarios in bore 14320258.

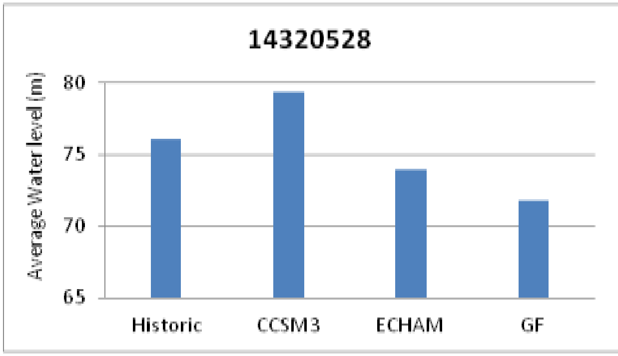


Figure 145: Eigenmodel predicted average water levels over 100 years from 1900 to 2000 for the bore 14320528.

The second scenario (ECHAM) represents an 'average' climate change scenario in regards to precipitation, evaporation and surface water conditions. Analysis of groundwater levels in bore 14320528 suggests about 600 months of low groundwater levels during the hundred-year simulation period. This means that the groundwater levels in the Lockyer Valley would have dropped below the 25% quantile of historic groundwater levels in approximately 50 of the 100 years (e.g. approximately twice the occurrence of historically low groundwater records). Similarly, groundwater levels below the 10% quantile of historic records would occur more than a quarter of the entire simulation time using the ECHAM climate scenario, as depicted in Figure 146.

The results using the GF scenario are more severe. This scenario led to more than 1000 months (83 years) of low groundwater levels. The same investigation was also conducted using the 10% quantile. The results vary more than before. The evaluation revealed 40 years below the 10% quantile for bore 14320297 and about 70 years for bores 14320487 and 14320528. Both scenarios revealed a distinct alteration in groundwater conditions.

These alterations would predominantly effect the irrigation behaviour of farmers. For instance, the 10% quantile of bore 14320528 amounted to a groundwater level of 72.2 m AHD. This water level is characteristic for the drought between 2002 and 2008. During this period, farmers were forced to reduce their water use until agriculture was severely restricted. If we assume that climate change generates a situation similar to the average scenario (ECHAM), or even a tendency to the dry GF scenario, farmers in the Lockyer Valley will probably be forced to add external water occasionally if they want to sustain their productivity. One solution to meeting water demand could be the use of PRW. It must be remembered that the majority of climate change scenarios lead to lower water levels in the Lockyer Valley. As such, farmers will need to reconsider their attitude towards using recycled water for irrigation purposes.

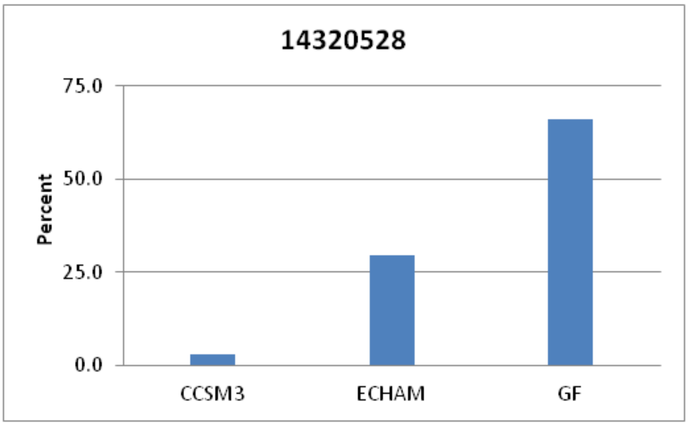


Figure 146: Percentage of months with water levels below 10% quantile of the historic as obtained from the Eigenmodel.

13.3 Numerical Simulation Model Based Climate Change Scenario Analysis

The Lockyer Valley groundwater flow model discussed in Chapter 10 was employed to simulate the groundwater level variations corresponding to the three climate change scenarios. These simulations were applied to the historical period from January 1991 to June 2000. The climate change–affected inputs into the model include both the river recharge resulting from the varied river flows and diffuse recharge resulting from the varied rainfall and other climate data.

River stages corresponding to the climate change scenarios were obtained from the IQQM model. These inputs were then used in the river recharge program detailed in Chapter 12 to obtain the RIVER package for MODFLOW. Similarly, the drainage values for different crop types corresponding to the climate change scenarios were obtained by running the HowLeaky model and then rewriting the MODFLOW recharge package.

The model was then used to simulate groundwater levels with these river and diffuse recharge model inputs. For this analysis, groundwater pumping in the region was assumed to remain the same as the historic usages; however, there is a very real possibility of variation in the water demand owing to climate change.

13.4 Soil Water Balance and Stream Flow Inputs for Climate Change Scenarios

The rainfall and evapotranspiration outputs from the climate modelling were used as inputs in the HowLeaky model (Rattray *et al.*, 2004) described in Chapter 9. The estimates of irrigation, transpiration and deep drainage for three climate change scenarios were then applied to the same land-use patterns used in the groundwater model described in Chapter 12.

The three climate change scenarios tested were i) second wettest climate (CCSM3), ii) median climate (ECHAM), and iii) second driest climate (GF). Each simulation covered the period from 1991 to 2000, using data from the DPI Gatton research station, Gatton (152°19' E, 27°32' S). For the purposes of this analysis it was assumed that rainfall and evaporation have the major impact on the climate change effects on the soil water balance compared with temperature and radiation.

13.5 Numerical Model Results for the Climate Change Analysis

Using the numerical groundwater model discussed in Chapter 12, groundwater level changes were simulated for the period January 1991 to June 2000. The groundwater level predictions at eight selected bores located throughout the valley, which calibrated reasonably well, were chosen to analyse the effects of climate change (refer to Figure 147 for bore locations). Water level predictions over a time period of 10.5 years from January 1991 to June 2000 at these bore locations were obtained using the calibrated model. The time series plots for these bores are shown in Figure 148 to Figure 156.

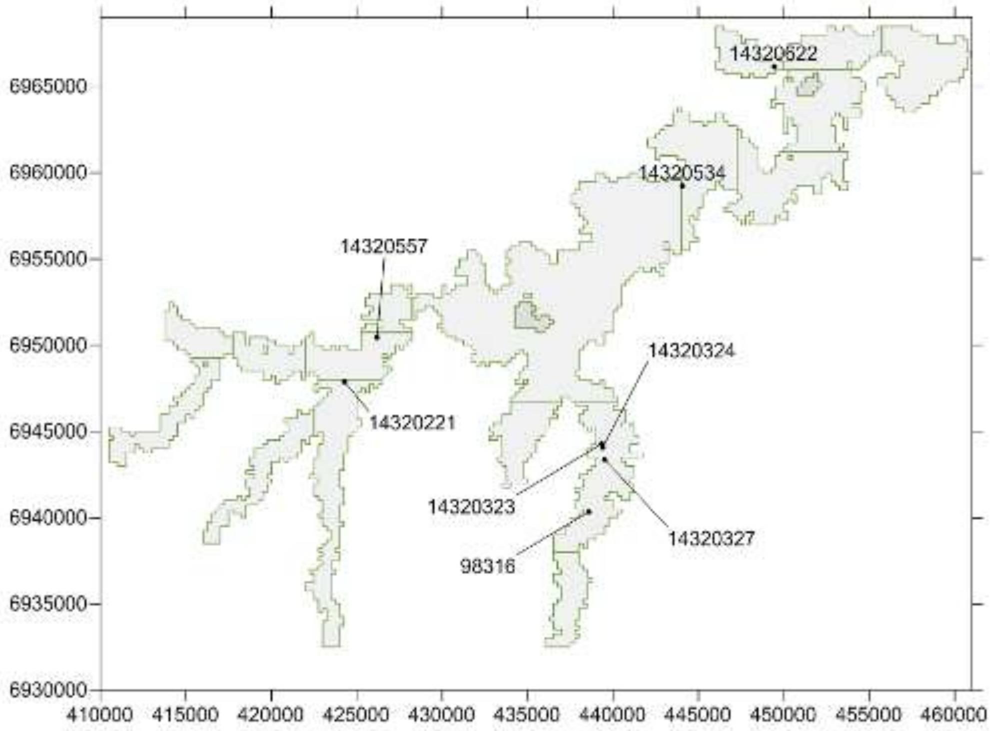


Figure 147: Location of the bores for which climate change water levels were analysed.

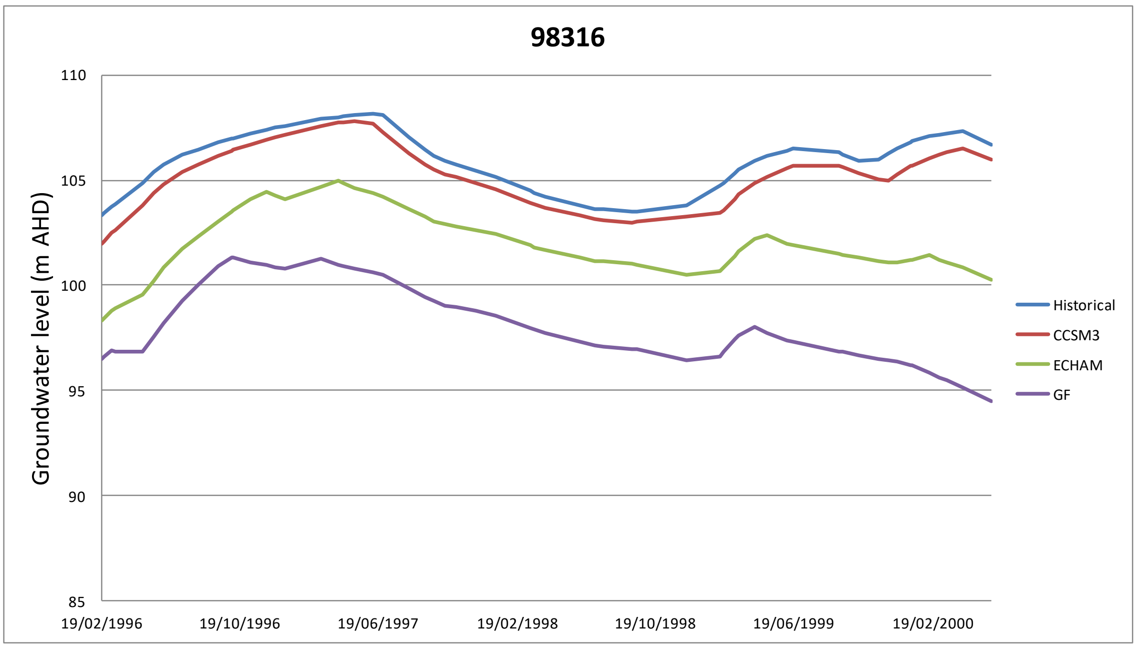


Figure 148: Water level plots for bore 98316.

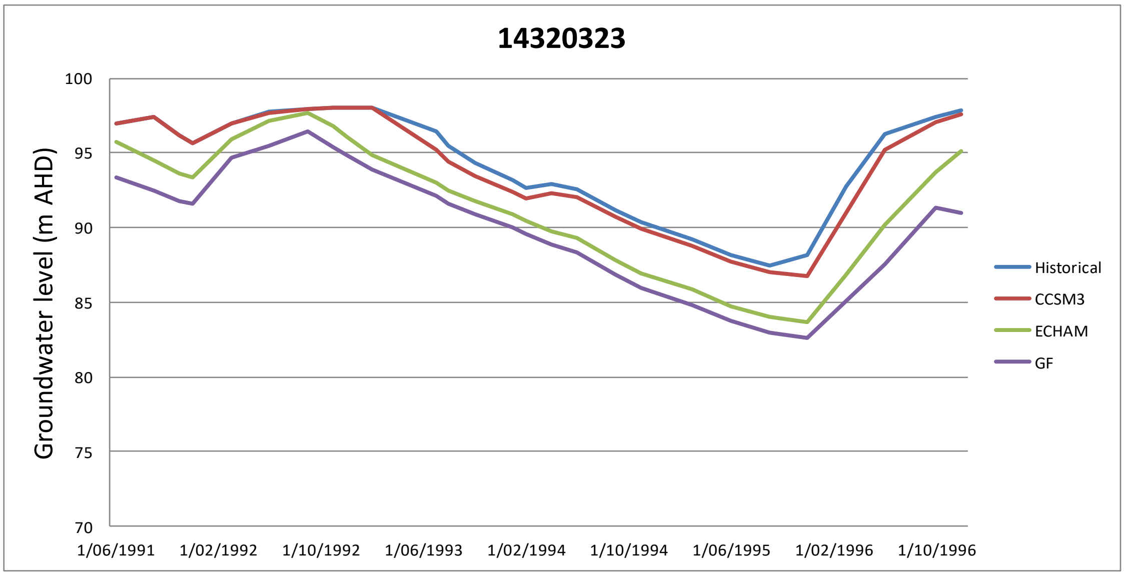


Figure 149: Water level plots for bore 14320323.

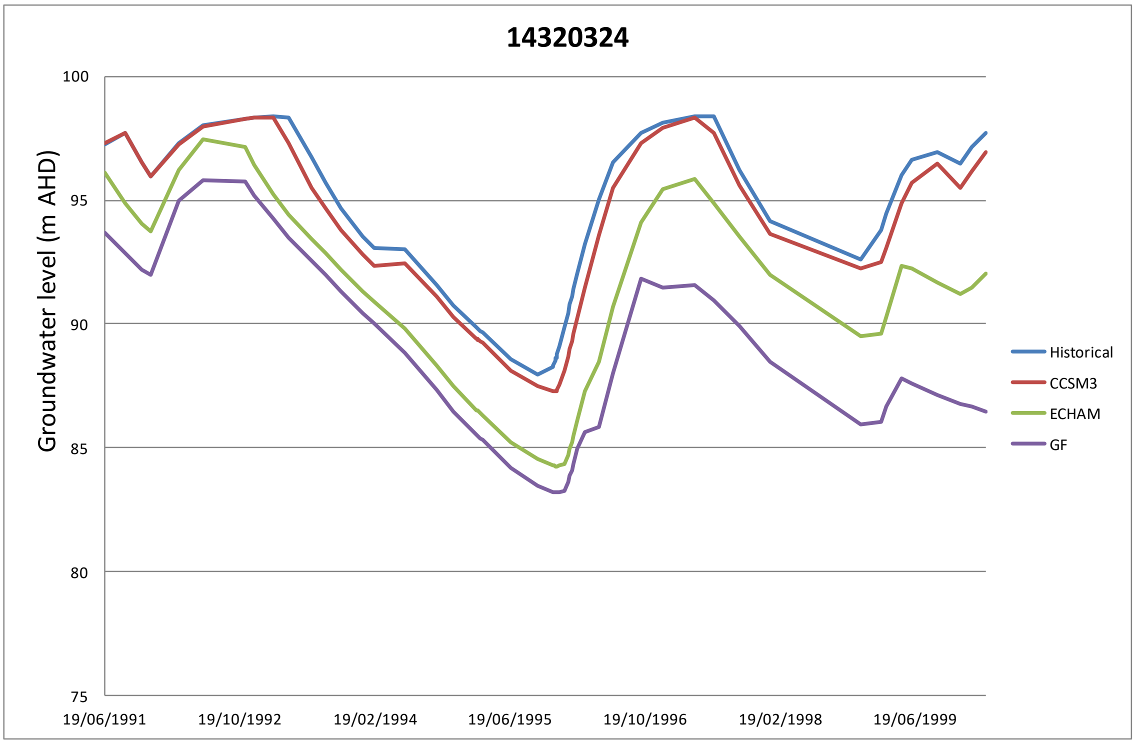


Figure 150: Water level plots for bore 14320324.

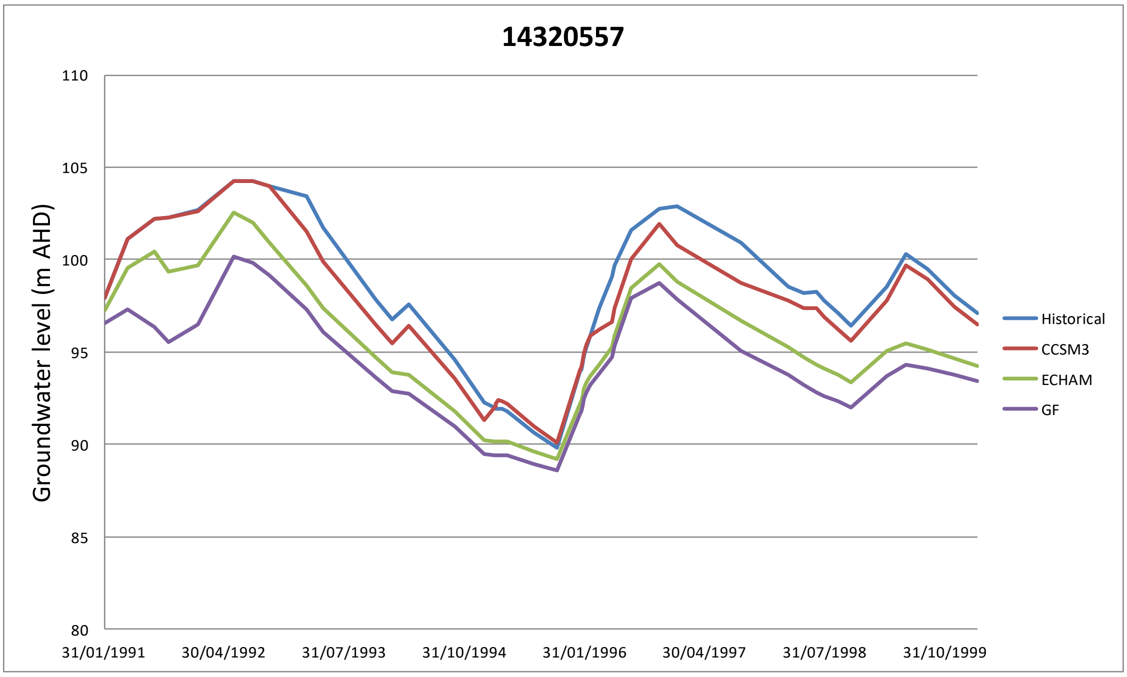


Figure 151: Water level plots for bore 14320557.

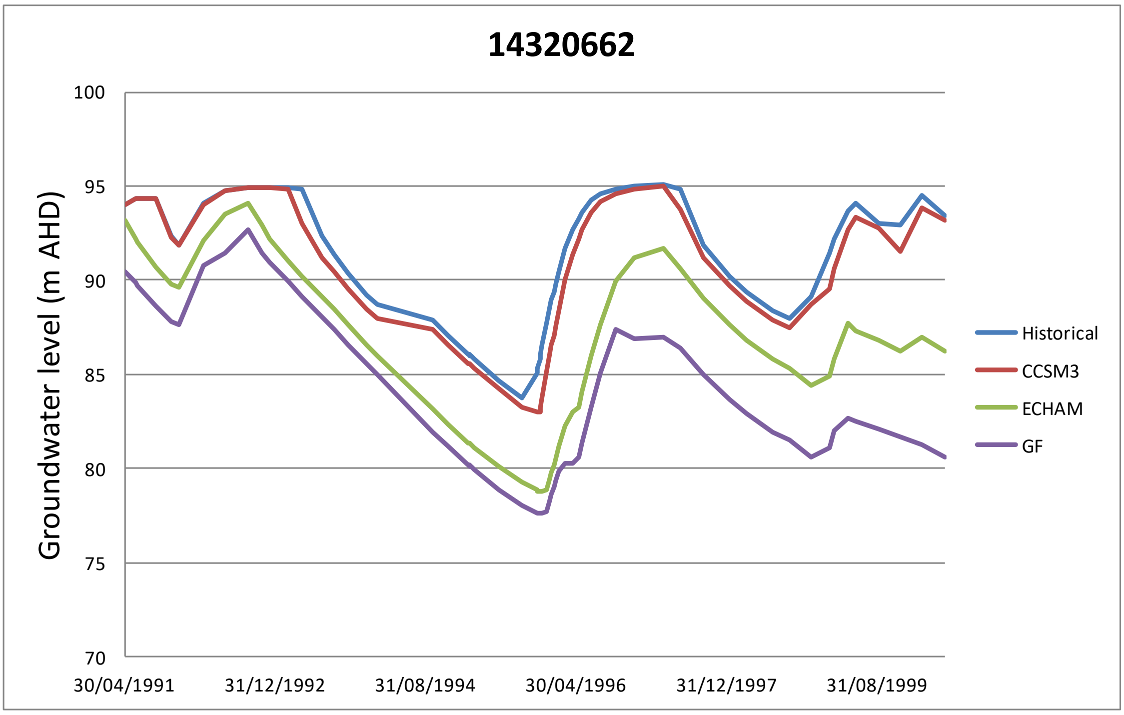


Figure 152: Water level plots for bore 14320622.

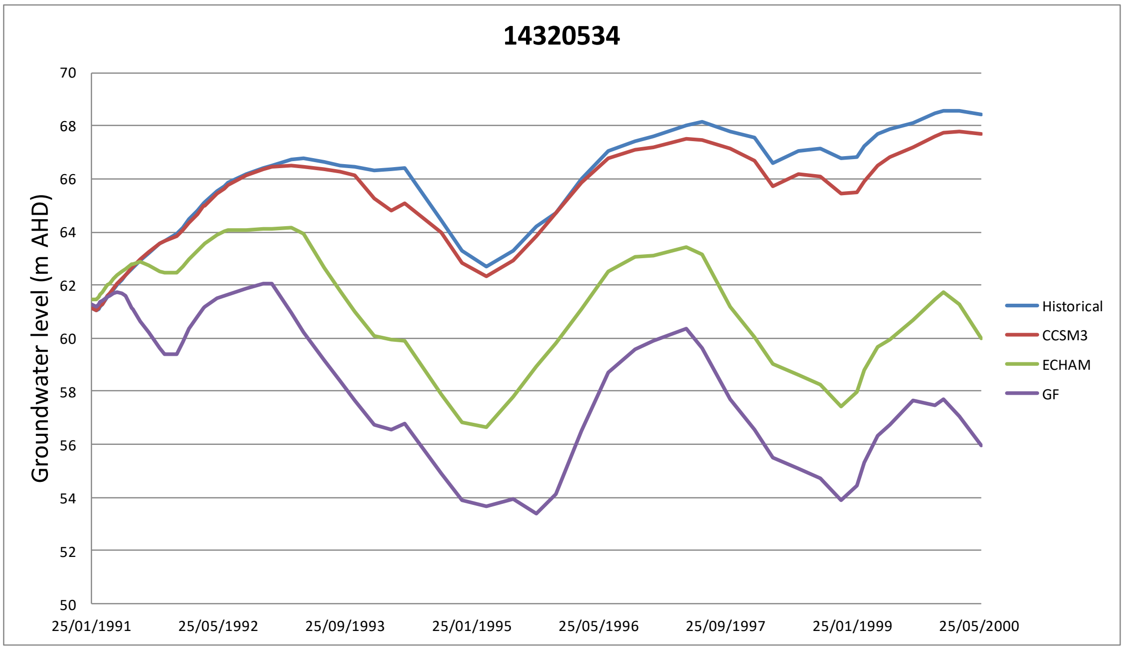


Figure 153: Water level plots for bore 14320534.

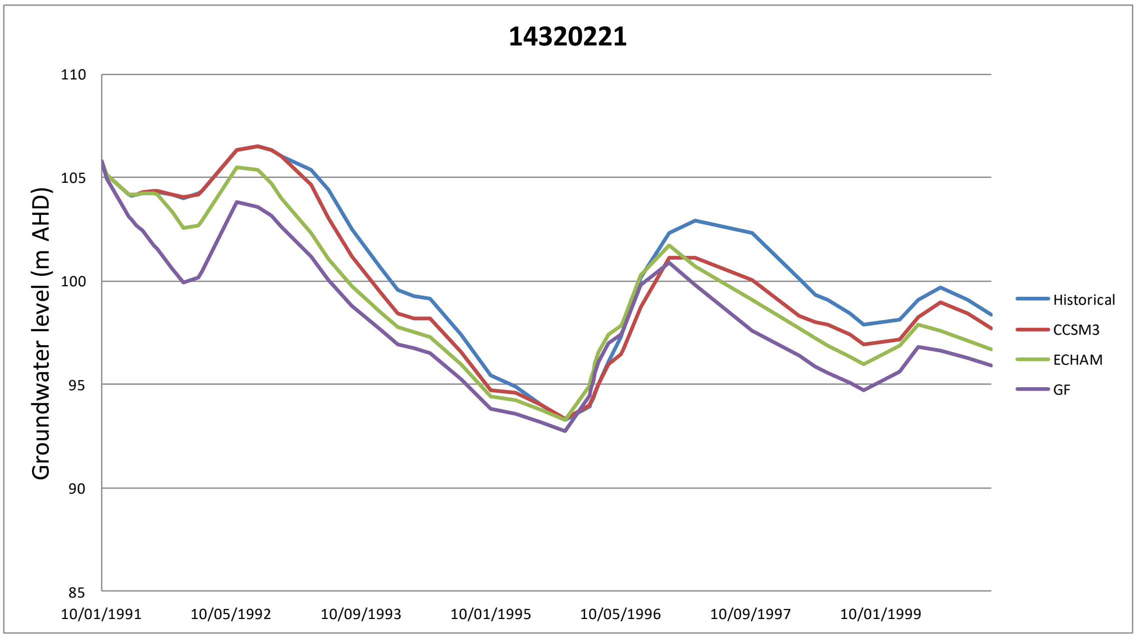


Figure 154: Water level plots for bore 14320221.

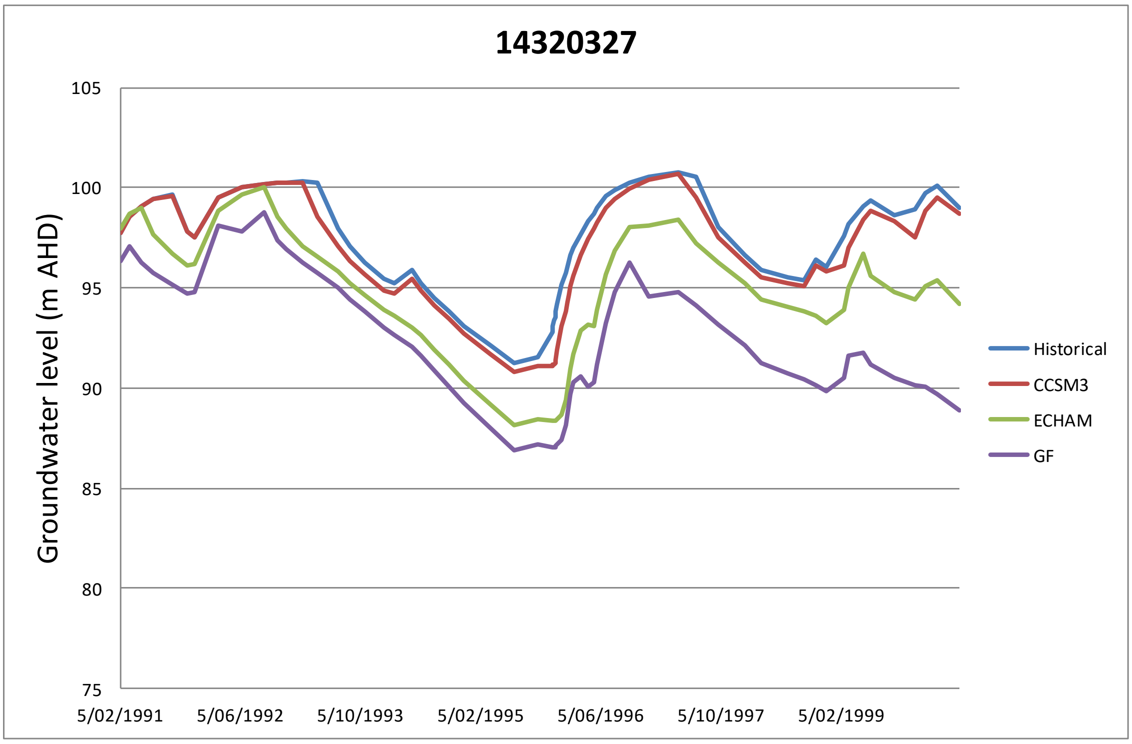


Figure 155: Water level plots for bore 14320327.

In all the bores it was found that for all three scenarios, the water levels are lower than the historic values predicted by the model although the CCSM3 scenario was very close to the historical. The difference in the water levels from the historical predictions increase towards the end of the simulation period. The median and dry climate scenarios, ECHAM and GF respectively, resulted in similar water levels, which were considerably less than the CCSM3 scenario. The median climate change scenario has predictions closer to the GF values as against the historic values.

Benchmark values of 10% quantile and 25% quantile of the historical water levels were adopted to analyse the variations in the water levels owing to climate change. CCSM3 scenario resulted in similar situations to the historical, where water levels were maintained at these levels for 10% and 25% of times respectively. Some bores indicated higher levels of water and some indicated lower levels of water for CCSM3 scenario. However, both ECHAM and GF scenarios indicated consistent decline in water levels. ECHAM scenario results in water levels falling below the historical 10% quantile between 20% and 65% of times. Also, ECHAM scenario may result in the water levels falling below the historical 25% quantile up to 85% of times in some bores. The realisation of the second driest scenario may result in the worst conditions of water levels in some bores dropping below the historical 10% quantile as much as 98% of times. A comparison of the water levels as obtained from Eigenmodel and the numerical groundwater model for the bore 14320528 for the three climate change scenarios is shown in Figure 156. It can be observed that climate change predictions by both the models indicate similar trends.

The estimates of the climate change water levels in all the bores near the creek are highly dependent on the representation of the influx from the creek in the Lockyer model. In the current model, this is represented approximately by means of a MODFLOW river package in which the recharge from the creek is proportional to the difference between the stage in the river and the water table and the conductance of the river bed. Hence the conceptual representation of the river in the groundwater model is highly influential on the groundwater level estimates corresponding to climate change. The results illustrate that reduction in flow in the creek resulting from climate change could result in huge decline in the groundwater levels, particularly near the creek.

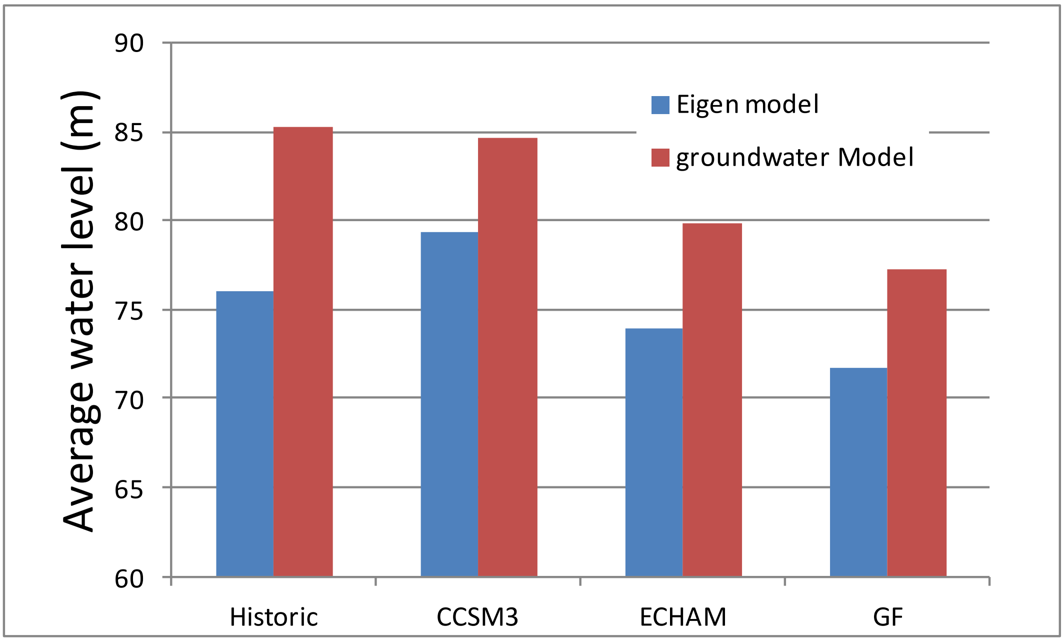


Figure 156: Comparison of the Eigen and numerical model prediction of water levels.

Figures 157 to 160 show the percentage of months with water levels predicted to be below the historic 10% quantile level at four representative bores. Collectively these simulations indicate that climate changes may result in considerable lowering of the water table. Occurrence of dry climate scenarios like the GF may result in water levels lower than the 10% quantile of the historic levels for as high as 90% of times.

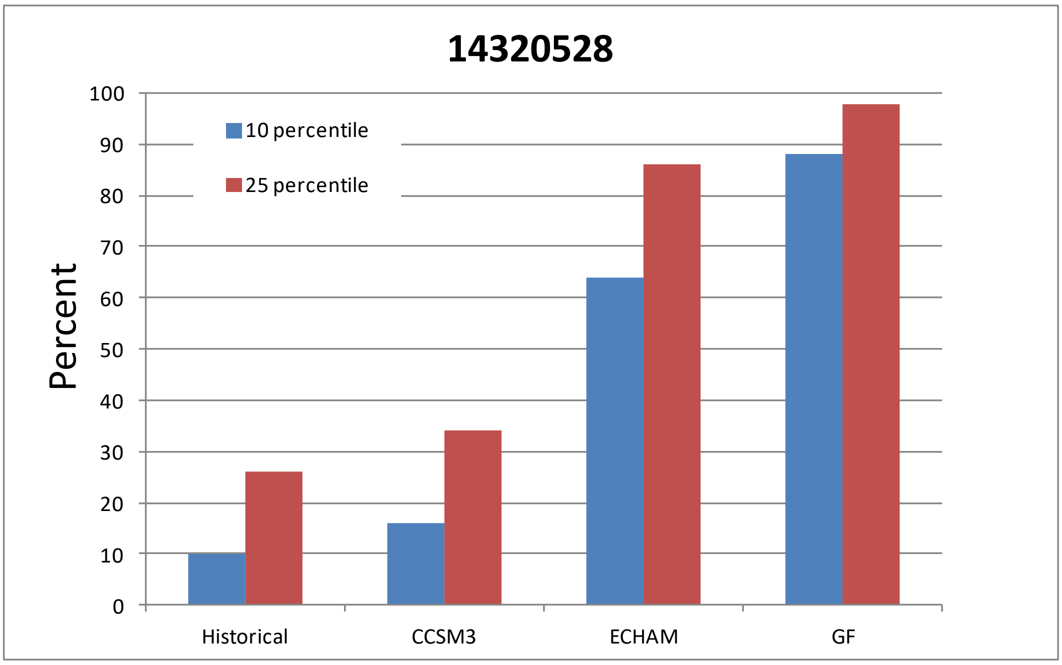


Figure 157: Percentage of months with water levels below 10 percentile of the historic at bore 14320528.

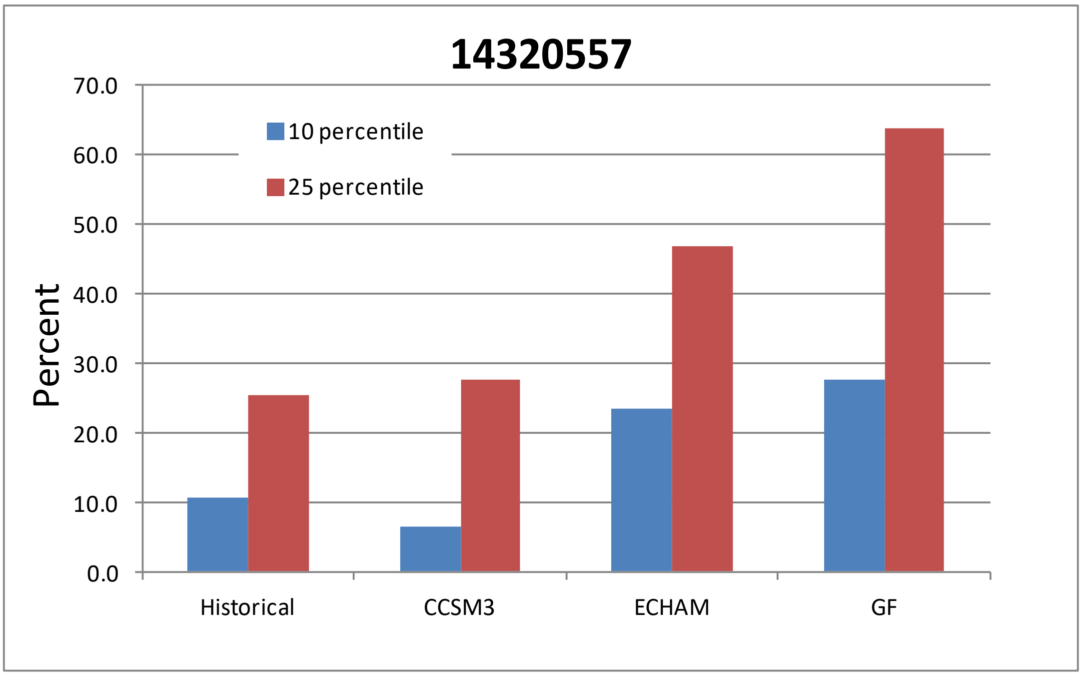


Figure 158: Percentage of months with water levels below 10 percentile of the historic at bore 14320557.

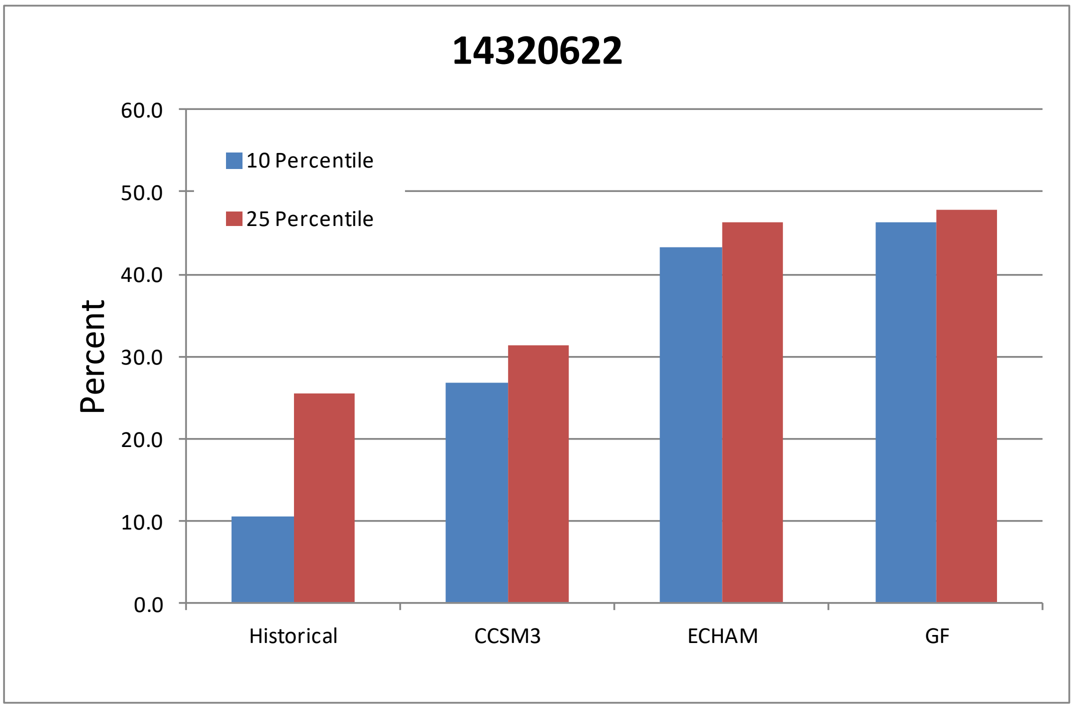


Figure 159: Percentage of months with water levels below 10 percentile of the historic at bore 14320622.

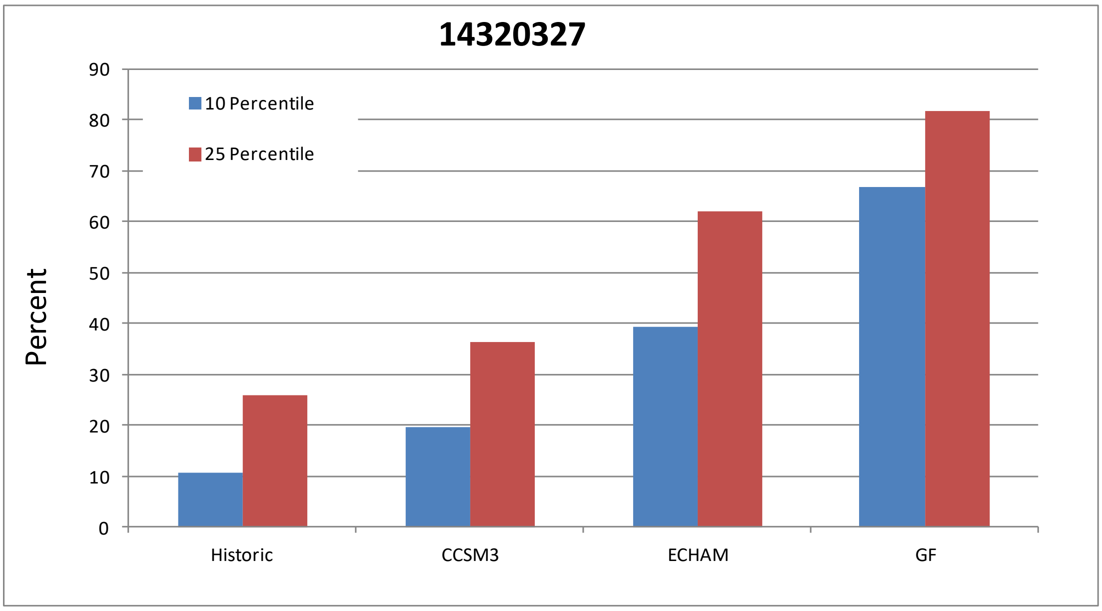


Figure 160: Percentage of months with water levels below 10 percentile of the historic, at bore 14320327.

The potential lowering of the water table can be estimated by applying climate change scenarios to a historically observed low water level. Within the considered time window for the climate change analysis, 05/01/1995 had one of the lowest recorded water levels. The extent to which the water table could be further lowered resulting from a climate scenario is estimated by applying the climate change scenario inputs to the groundwater model and then comparing the resulting water levels for the same stress period with corresponding historic values. The contour plots of depths by which the water table could be lowered further from the historically observed low water levels of 05/01/1995 for the three climate change scenarios is show in Figure 161, Figure 162 and Figure 163.

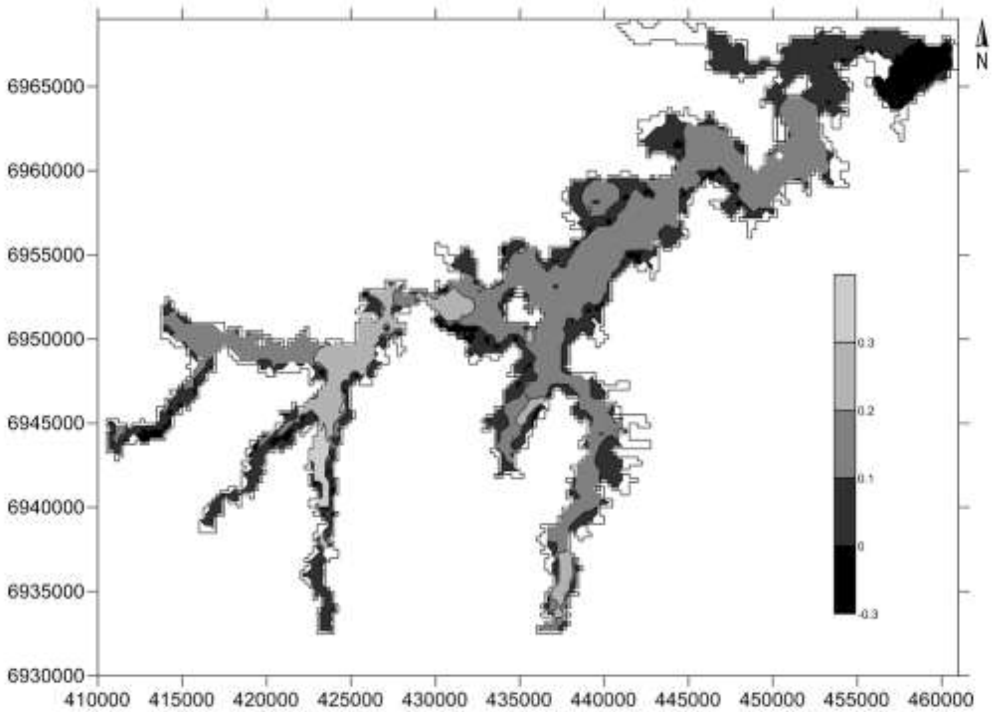


Figure 161: Fall in the water table below the historic level of 05/01/1995 for CCSM3 scenario.

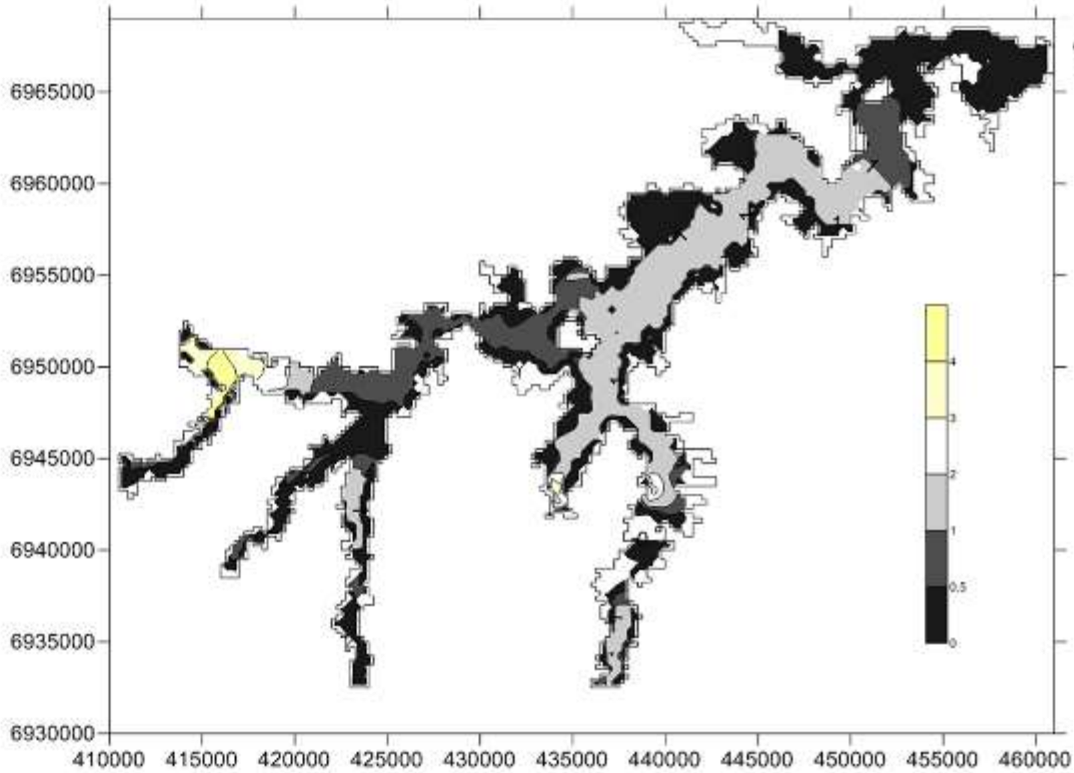


Figure 162: Fall in groundwater table below the historic level of 5/01/1995 for ECHAM scenario.

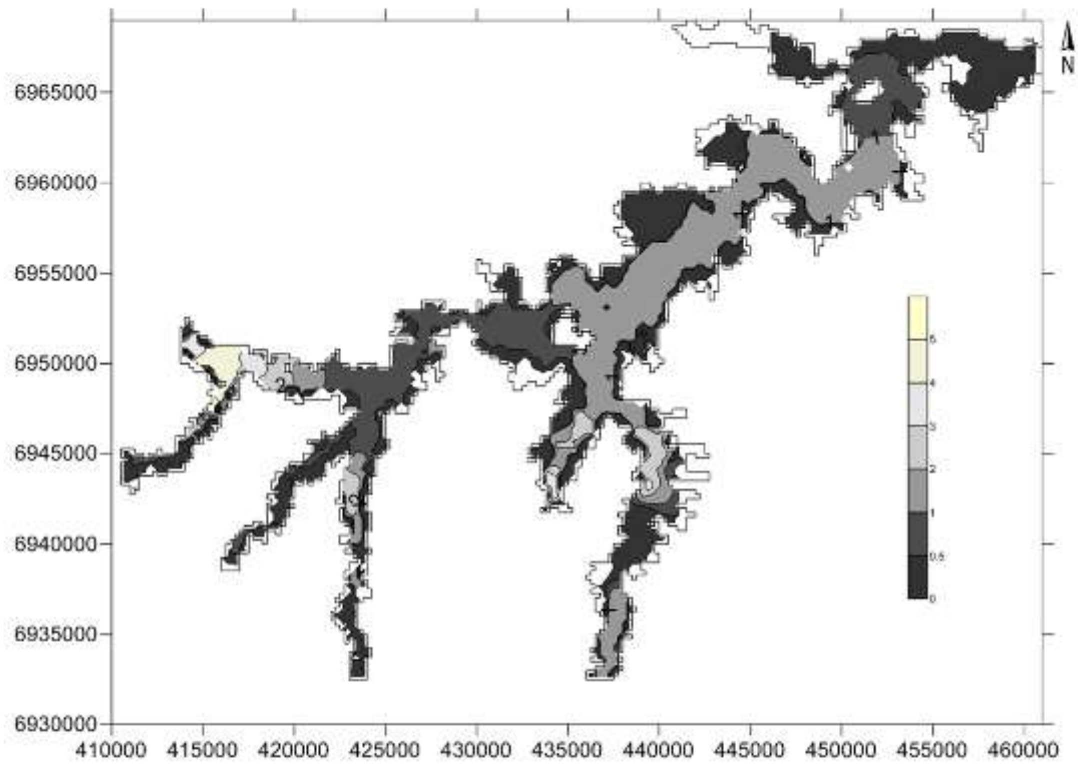


Figure 163: Fall in groundwater table below the historic level of 05/01/1995 for GF scenario.

13.6 PRW Import for Climate Change Scenarios

The potential of PRW import in relation to climate change scenarios was quantified by estimating the import volumes to top up the low water levels in the Lockyer Valley on 05/01/1995 to the average water levels described in Chapter 10 for all three climate change scenarios. The same procedure described in Chapter 10 was made use of to perform this analysis. It was found that for all three climate change scenarios, import requirement is higher than the import required under historic conditions.

The excess import volumes required to top up to the medium water level, over the analysis period, for the three climate scenarios is shown in Figure 164. It could be observed that, for brief period between 1996 and 1997, the import volumes are very less, indicating no significant requirement of excess import for that period. However, except for this period, the climate change simulations indicate higher import volumes would be required to meet the average target water levels, owing to decreased groundwater availability resulting from climate change. The average additional top-up volumes calculated were 2.04 GL/a, 4.39 GL/a and 9.04 GL/a for the high (CCSM3), medium (ECHAM) and low (GF) recharge models respectively.

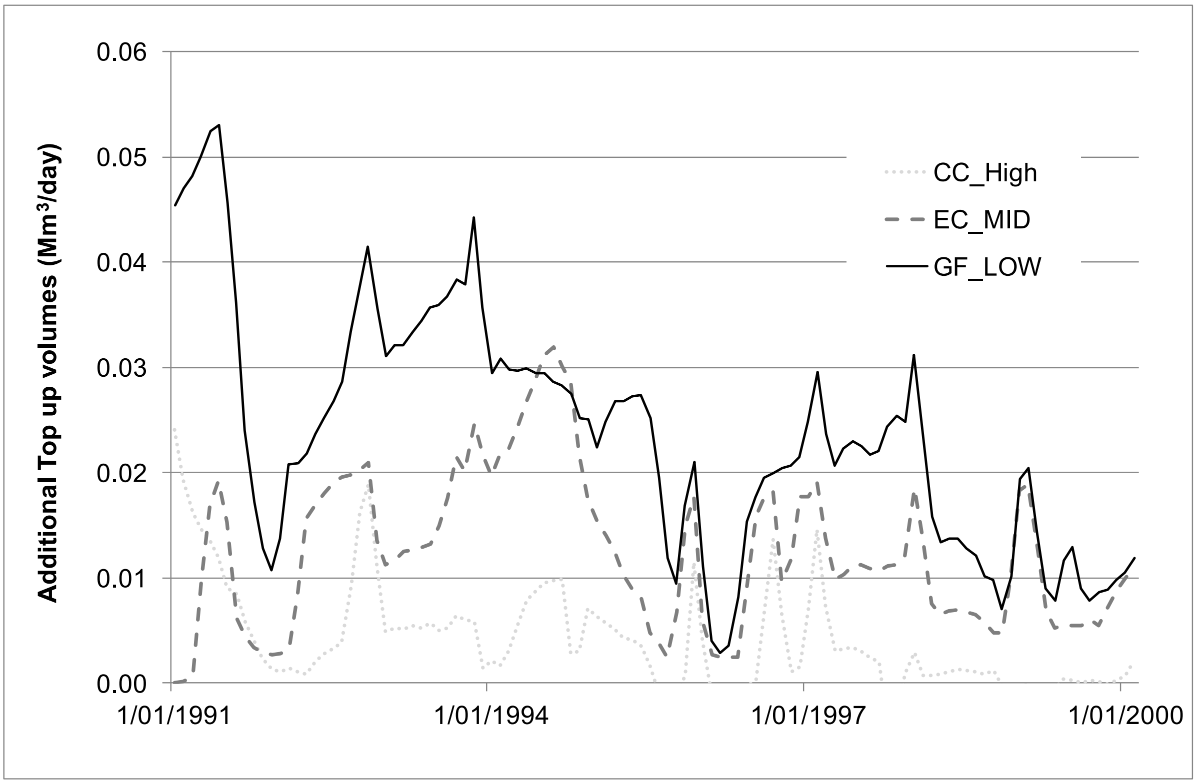


Figure 164: Excess PRW import volumes required to top up to the medium water level for the three climate change scenarios.

13.7 Conclusion

The limited number of analyses based on the Eigenmodel and the numerical modelling exercises indicate that there is a possibility of considerable lowering of the water levels in the Lockyer Valley under projected downscaled climate change scenarios such as CCSM3, ECHAM and GF (Gooda *et al.*, 2011). The modelling studies as described in the current and previous chapters indicate that the Lockyer Valley groundwater system is hugely dependent on the influx from the creek recharge and to a lesser extent on the diffuse recharge. Adverse climatic conditions will considerably reduce the recharge volumes. This effect, combined with the increase in the water demand owing to the adverse climatic conditions, would result in significant lowering of the groundwater levels, particularly near the zones of creek recharge. Further extension of the analyses may include the potential increase in pumping demand triggered by adverse climate scenarios. Import of PRW, either to augment the groundwater resource or as a substitute for it, could be a potential management option to strike a balance between the demand and availability. The results presented in this chapter are subject to a number of uncertainties, namely: i) the uncertainty in climate modelling, which has been considered by analysing a range of different scenarios; ii) the uncertainty introduced by the IQQM model, which was used to calculate the differing surface water flows for each climate scenario; and iii) the uncertainty introduced by the numerical groundwater modelling exercise or conceptual limitations of the Eigenmodel approach.

14. SUMMARY AND CONCLUSIONS

(Leif Wolf, Ashley Bleakley, Don Begbie)

14.1 Application of the Tiered Assessment Framework to the Lockyer Valley

A wide range of investigations was undertaken in the UWSRA research project 'PRW in the Lockyer Valley', which correspond to and which helped to develop the framework introduced in Chapter 2 of this report.

Table 34 summarises the applied methods and the key results. The methodologies developed and adopted for this project address the particular challenges in the Lockyer Valley and reflect the professional expertise of the researchers and partner institutions. If the tiered assessment framework is applied in other areas, some aspects might receive less detailed investigation. For example, salt mobilisation would not need a detailed assessment in a very humid climate, whereas the accumulation and transport of trace organic compounds or nitrate accumulation should receive more detailed investigations if Class A treated wastewater is used.

Hydro-economical modelling, as suggested in Tier 4, is not within the scope of the current research project.

Table 34: Elements of the assessment framework which were applied to the Lockyer Valley and selected results.

	Methodology Applied	**Results/Impacts**
Tier 1	Hydrochemical analysis including Sodium Adsorption Ratio of PRW, surface water and groundwater; mechanical and spontaneous dispersion test	SAR of PRW < 2, EC around 220 µS/cm, PRW water chemistry very close to current surface water chemistry, no spontaneous dispersion, low risk expected.
Tier 1	Trace organics screening in natural waters and the imported waters	The existing water resources in the Lockyer Creek and the major surface water reservoirs are already influenced by wastewater discharges; also, groundwater wells close to the Lockyer Creek exhibited traces of pharmaceuticals (carbamazepine) and artificial sweeteners (acesulfame), whereas PRW is confirmed to be of significantly higher purity according to the screening campaigns published by Seqwater.
Tier 2	Analysed historical land use change using multi-spectral Landsat Thematic Mapper (Landsat 5 TM) imagery with atmospheric correction	Bare land appears to change significantly according to water availability between 31% (wet period September 2010) and 56% (dry period July 2006) of the area (entire Lockyer Valley model zone).
Tier 2	Analysed water use changes	Metered surface and groundwater use strongly declined in the period 1992–2010, correlating to lower availability of water, and increased irrigation efficiency but probably also to problems with aging meter infrastructure. Metered groundwater use for the central Lockyer Valley declined from ca. 19,120 ML/a in 1992/1993 to ca. 4413 ML (2009/2010 metered) or 6568 ML/a (2009/2010 adjusted for suspected meter failures). Water demand estimated with soil water balance models ranged between 10–16 GL/a, with an average of 13.5 GL/a for the central Lockyer Valley and 36 GL/a for the entire Lockyer Valley if a land use such as in September 2010 is assumed. Especially for the last decade, modelled water demand is exceeding the actual recorded usage in the central Lockyer. This suggests that the significant variations in crop areas, caused by varying groundwater availability and increased irrigation efficiency, are not adequately being accounted for. Ageing meter infrastructure may also be a contributing factor.

	Methodology Applied	Results/Impacts
Tier 2	Groundwater recharge assessed: Groundwater table fluctuation method applied	Different groundwater table fluctuation methods suggest annual recharge volumes between 0 ML/a and 40 GL/a for the central Lockyer; long-term average groundwater recharge as a combination of creek recharge and diffuse recharge ranges at 10–11 GL/a. In the period 1987–2011, groundwater storage in the central Lockyer Valley varied between 31% and 91% of historic recorded full groundwater.
Tier 2	Groundwater recharge focusing on creeks using the Eigenmodel approach combined with a numerical groundwater model	Aquifer parameters were determined using an Eigenmodel approach for 11 wells located at various distances from Lockyer and Laidley Creeks. The fast response of the wells suggests that recharge from creeks is the dominant recharge mechanism in the Lockyer Valley. The final recharge assessment will only be available with the calibrated numerical groundwater model.
Tier 2	Diffuse groundwater recharge assessed: Forward modelling of deep drainage using soil water balance modelling	Deep drainage rates suggested to range between 85 and 91 mm/a as a long-term average over the entire valley, and deep drainage maps were produced based on the remote sensing of land use to feed the numerical groundwater model. On individual plots, different crop rotation sequences were modelled, ranging from a dry forage-grain-pasture cropping sequence with a deep drainage of 11 mm/a to a vegetable-based cropping sequence with a deep drainage of 121 mm/a. Soil water balance modelling proved to be very sensitive to assumption on irrigation strategy.
Tier 2	Groundwater recharge assessed: Numerical modelling of the unsaturated soil zone with Hydrus, sensitivity study	Richards-equation based modelling of sites at Forest Hill and Tent Hill suggests groundwater recharge rates between 24 mm/a and 270 mm/a, depending on the irrigation strategy.
Tier 2	Salt risk: Catchment salt balances, estimating the amount of salt added to the system	Annual volume of 25 GL/a PRW into the Lockyer Valley, with a typical chloride concentration of 19.3 mg/l, would equal an import of 482 t of chloride per year compared to chloride input via rainfall of approximately 1084 t/a. The additional salt loading could be offset by increased baseflow to the Lockyer Creek and increased lateral groundwater movement. PRW input of salt to soils is small compared to current use of salty groundwater, which contributes a first order estimate of 9800 t/a chloride to the soils of the Lockyer.
Tier 2	Salt risk: Analysis of salinity trends in groundwater	Rising salinities exist before PRW supply in most of the upstream side valleys; no major trends were observed in the central parts of the alluvium. High salinities occur outside and at the fringe of the alluvium. Suggest not to use additional water to irrigate high salinity areas outside the main alluvial aquifer. Following the 2011 floods, significant rises in groundwater levels occurred but, until 2012, with only minor impact on water quality in the focus monitoring wells.
Tier 2	Salt risk: Deep soil profiles taken down to 20 m	Major amounts of salt already washed out of the soil profile in areas where irrigators switched to surface water. Spatial heterogeneities prevent construction of salt flux maps. In selected profiles, significant amounts of salt are on their way towards the groundwater table.
Tier 2	Salt risk: Hydrus modelling of salt flux	For a 20 m depth to the groundwater table, the timing of the salt peak breakthrough varies from 22 years for intensely irrigated shallow-rooted crops, to over 200 years for moderately irrigated deep-rooted crops. This is salt originating from existing or previous irrigation with brackish water but not caused by PRW. Under current irrigation practices (e.g. using chloride concentrations of 62 and 167 ppm), a long-term salt accumulation in the soil profile is predicted. Switching to PRW is mostly beneficial in areas where current irrigation water is highly saline. If irrigation with PRW is not administered at higher than current rates, then PRW will not lead to an increase of salt fluxes to the groundwater table.
Tier 3	PRW demand to fulfil ecosystem service and grid buffer goals: Coupled numerical surface groundwater modelling exercise using customised Modflow and IQQM	Proof-of-concept modelling suggested that 9–21 GL/a would be required in order to always maintain average groundwater levels given the historical pumping record. A transient demand profile was calculated along with maps of import demand.

	Methodology Applied	Results/Impacts
Tier 4	Rules: Use the numerical groundwater model and the pumping information to determine if managed aquifer recharge is required or if substitution of existing use is sufficient	Map overlays constructed showing areas where substitution of existing use is insufficient to maintain groundwater levels.
Tier 4	Robustness: Limited survey of irrigators on potential land use changes under different water availability	All six irrigators interviewed would expect to expand their irrigated area under availability of PRW; two out of six would consider stopping using groundwater (van Opstal *et al.*, 2012).
Tier 4	Robustness: Climate change impacts. Downscaled climate change scenarios were processed with IQQM and subsequent information on changed surface water flows was fed into a novel Eigenmodel tool at representative bores	For the ECHAM scenario (as a median of the climate scenarios), the number of months which are below the 25% quantile of the historic groundwater levels in a period of 1200 months increases by 373, 330 and 266 months at three representative bores.
Tier 4	Robustness: Climate change impacts. Downscaled climate change scenarios were processed with IQQM and the numerical groundwater model	The average PRW import requirement under a median climate change scenario suggested an additional demand of 4.39 GL/a to the already estimated demand using historic data.

14.2 Implications for the Lockyer Valley

Besides the general advances in scientific methodology, the project noted a number of observations with relevance to the future planning as follows:

Salt and Water Quality

- Irrigation with salty water has created a legacy of salty pore water in soils in parts of the Lockyer Valley. These stocks of salt are currently slowly migrating to the groundwater.
- Switching from irrigation with salty groundwater to surface water resources has already improved soil salt conditions at three profiles in the Lockyer Valley.
- The supply of PRW as it is produced today, with less than 25 mg/L chloride, is expected to lead to lower salinity in soils and groundwater of the Lockyer Valley if irrigation volumes are not significantly increased.
- Risks from potential irrigation outside the currently used alluvial aquifer were not investigated in the research project. Due to the suspected risk of salt mobilisation and water logging, irrigation is not recommended on areas outside of the alluvium without detailed risk assessment.
- Lockyer Creek discharges into the Brisbane River and thus influences the water quality at the intake of the Mt Crosby water treatment plant, which supplies drinking water to the inhabitants of Brisbane. Releases of PRW to the creeks of the Lockyer Valley are therefore similar to release of PRW to the Wivenhoe Dam.
- Initial screening campaigns for trace organic compounds in the Lockyer Valley suggests that, in comparison with the PRW produced at the Bundamba advanced water treatment plant, the current system in the Lockyer Valley exhibits more pharmaceuticals and wastewater–derived trace organics than the PRW, which was tested to be virtually free of trace organics except disinfection by-products (Roux *et al.*, 2010).
- With regard to pharmaceuticals and trace organics, the quality of PRW appears to be significantly better than the existing water quality downstream of Gatton.

Soil

- Analysis of PRW chemistry and tests with soils from the Lockyer Valley showed only a minor potential for soil structural changes when irrigating with PRW, very similar to the risks already induced by rainfall or overhead irrigation with fresh surface water. These impacts are easily managed by the landholders.

Understanding the Natural Water Resources System

- The capacity of the Lockyer Valley alluvial aquifer to store water is expected to range between 230 and 350 GL.
- Numerical groundwater models need to account for a dynamic representation of creek recharge as well as the diffuse groundwater recharge underneath irrigated agricultural land.
- Diffuse groundwater recharge is higher than in previous modelling exercises.
- Preliminary modelling of changing climate scenarios suggests that drought condition water levels may occur twice as frequently in a median climate change scenario if pumping rates remain similar.
- The assumption of a negligible inflow of groundwater from the basement rock into the alluvium is only justified in terms of water quantity but not in terms of water quality.
- A 3D hydrogeological visualisation system is available free of charge from QUT and can be used to increase public understanding of the resource.
- We replace the traditional concept of estimating sustainable yield based on groundwater recharge calculations with the concept of target water levels and acceptance criteria for the violation of those targets under a variable climate.

Demand and Management

- Due to the energy involved in producing PRW and transporting it to the Lockyer Valley, the amount of water imports into the catchment should be minimised.
- In most places, the preference of irrigators with regard to their water source is surface water > groundwater > imported water (PRW).
- PRW demand relies on definition of environmental goals. Tools to calculate PRW requirement for different goals were developed by the UWSRA project and are available for further use.
- As an example, modelling suggested that 9–21 GL/a would be required in order to always maintain average groundwater levels given the historical pumping record. With the numerical groundwater modelling environment, it is possible to calculate dynamic demand profiles for any specified target water level.
- PRW demand is highly variable in time.
- According to the set target water level concept, PRW import is also not required everywhere but rather at specific locations in the Lockyer Valley, as outlined in maps produced by the UWSRA project.
- Groundwater in the Lockyer Valley is a common pool resource which is not yet fully considered and managed as an economic commodity. The valuation of the groundwater resource will impact on the future acceptance of a PRW scheme.
- The direct injection of PRW into the aquifer is technically possible but incurs major management challenges with regard to cost recovery of water introduced into a common pool resource.
- Using PRW to substitute existing groundwater use can avoid the costs for constructing injection schemes.
- In combination with an imported water source, the alluvial aquifer in the Lockyer Valley has the potential to be actively managed as a drought buffer.

Finally, we conclude that while environmental and supply risks are manageable, the cost-effectiveness of various recycled water supplementation options needs to be examined further through hydro-economic modelling. Also, a multi-objective optimisation exercise could detail how PRW could be used most efficiently. This would include options to deliver PRW direct to the farm gate (like surface water), where best to supplement existing water use, whether to inject directly into the groundwater, and at which times the import of PRW is actually cost-efficient. The respective assessment tools have been developed in the research project and are ready to use together with the appropriate stakeholder groups.

14.3 Transferable Aspects for Other Areas

The Lockyer Valley problem is an example for other areas worldwide that need to optimise the performance of a water grid including agricultural areas. The instruments developed here not only apply for PRW but for any imported water source. As a recent Australian example, the supply of treated coal seam gas water to the farmers in the Condamine region is a very similar problem, albeit with the exception that the treated CSG water comes at very low cost to the farmers. On the other hand, treated Class A wastewater can easily go through a similar assessment to PRW, just with a more stringent focus on the higher salinity loads and trace organic content.

While the idea of water reuse seems attractive at first, a number of implementation barriers need to be thoroughly addressed, especially in the areas of: (a) public health; (b) economical and financial barriers; and (c) technical and environmental concerns. As a transferable methodology, we propose to draw on the tiered assessment framework that is presented in Chapter 2 and which was used to summarise results in section 14.1. It is important to structure the assessment in terms of rising complexity, with some easy-to-carry-out reality checks in the initial stages of the feasibility assessment. These initial checks include the comparison of the quality of the imported water compared to the existing water resources in the catchment. Key parameters are salinity, sodium adsorption ratio and any pollutant loadings (including pathogens). Together with some general knowledge on water balances, this also allows for initial catchment salt balances to estimate the relevance of the additional salt input along with the water import. If these initial checks suggest a general feasibility, it is important to look into the possibility of salt accumulations at hot spots, for example, in areas where there would be intensive irrigation or in areas with potential water logging. Also, the mobilisation of pre-existing salt accumulations needs to be considered, especially in areas that were not irrigated before the import water became available.

As early as possible, estimates need to be made of the potential demand for imported water at various water prices. In many areas, the demand for imported water is governed by the price and availability of natural water resources in the catchment. Thus, a reappraisal of the existing water budget figures is recommended and, as time and resources permit, incorporating the water demand parameters into a numerical groundwater model.

Despite the considerable scientific work undertaken in the Lockyer Valley, there remains a considerable uncertainty in the estimates of deep drainage and groundwater recharge. Thus, it is advisable to focus effort on the quantification of that uncertainty and compare it with the uncertainty of the other elements of demand forecasting which result from inherently uncertain economic boundary conditions. As an example, even if we could accurately predict groundwater recharge under different future land use patterns, we would still be very uncertain if farmers will actually plant vegetables in 20 years time or how the market prices will develop. Under those circumstances, it is advisable not to spend too much effort on more precise quantification of sub-problems, but to aim at regulations and management strategies which are sufficiently robust and adaptive to match the remaining uncertainty. While simulation strategies for those coupled hydro-economic problems were documented in the last two years (Bulatewicz, 2010), most recent work also includes structured assessment of this uncertainty (Guillaume *et al.*, 2012). Certainly, we employed the most up-to-date techniques to quantitatively describe the groundwater model uncertainty and the sensitivity of the model output to the uncertainty in input variables. This exercise is recommended to inform the set-up of costly and labour intensive monitoring programs.

Constructing maps of demand for water imports can be an important tool, but it requires an interactive process with the stakeholders to achieve realistic results. Stakeholders need to be made aware of the consequences that a set target water level will have on the future allocation regulations.

Finally, we conclude that the starting point for an advanced management of groundwater as a common pool resource relies on the formal acknowledgement of groundwater as a finite economic good. Optimal groundwater management is defined also as the rate of extraction by location and time that maximises the present value of net benefits (Reinelt *et al.*, 2012). In the case of PRW or import of other water sources, we need to also add the rate and location of water import into this optimisation exercise. In summary, we conclude that sustainable planning with high public acceptance requires a holistic framework that recognises both **environmental** and the **economic benefit** from the new high value water resource **to the local community.**

14.4 Outlook

On 15 November 2012, a final stakeholder adoption meeting was held to ensure that the research products will be taken up by responsible planning bodies and the community in the Lockyer Valley. Meeting participants comprised representatives from the Queensland Water Commission, Regional Development Association Australia, Seqwater, Queensland Urban Utilities, Department of Natural Resources and Mines (DNRM), Department of Science, Information Technology, Innovation and the Arts (DSITIA) and CSIRO. The stakeholder adoption meeting acknowledged the progress and new information achieved by the research project. The UWSRA research delivered relevant input into a wider array of planning challenges that are currently being addressed or discussed by the institutions, for example:

- How to ensure the long-term sustainability of the Lockyer Valley as a food bowl?
- Can the use of water from multiple sources with different water qualities and multiple value concepts lead to more efficient water management?
- When will the Lockyer Valley move towards groundwater management according to standards and recommendations of the National Water Initiative. That is, when will volumetric entitlement be used to manage groundwater abstractions within sustainable bounds? This compares to the current practice where pumping continues until the well runs either dry or salty.
- Is there potential to supply water from the Brisbane River / Wivenhoe Dam to the Lockyer Valley for conjunctive use and for securing surface and groundwater supplies? This would rely on PRW water supplementation to Wivenhoe Dam in about 5% of years when the storage drops below 40%, and the intermittent PRW supplementation costs annualised along with infrastructure costs may be attractive to irrigators who could reliably produce more food and secure more lucrative markets.
- Given the very low demand for PRW under the current climate and pricing structure, and a possibility that the production of PRW might come to an end, is it possible to supply recycled water of a lower quality to the Lockyer Valley?
- Is there scope for increasing use of treated wastewater, for example from the local wastewater treatment plant at Gatton?
- How will any alternative supply options impact on the risks from persistent organic pollutants or pathogens compared to the current practice of treated wastewater discharges to the surface water bodies in the Lockyer Valley?
- How will alternative supply options besides PRW impact on soil and groundwater quality, especially with regard to salinity?
- Can the supply of imported water to the Lockyer Valley become a key contributor to improved quality of living, and is it possible for the community to see PRW positively (even as an achievement to be proud of)?

From these questions, the panellists derived a number of recommendations for future research directions:

- Model agro-economic aspects coupled with hydrologic models: This will provide information about the expected socio-economic benefits of different water management strategies and supply scenarios as well as the associated risk profiles (e.g. a strategy which results in a low risk of irrigation constraints and low risk of crop failure will justify a more complex distribution scheme with higher investment cost). The coupled model could take into account which crop choices are made based on the water resource status at the start of a stress period, calculate the changes in water demand as a result of the crop choice and use these changed demand figures to calculate the future water availability.

- Take into account the multiple supply sources, including their variable value over time.
- Quantify environmental benefits and introduce these data into multi-objective decision making.
- Consider the various options for managed aquifer recharge in the Lockyer Valley in more detail.
- Build on the existing UWSRA salinity research and modelling to predict impacts of other possible import water qualities.
- Move on from modelling salinity in soils and modelling of groundwater flow to modelling of groundwater salinity and quality.
- Hand over the numerical groundwater model which was set up during the UWSRA research project to the regulator for further review and validation as well as for the inclusion of operational rules to investigate optimum management strategies.
- Utilise newly generated data on groundwater isotope composition and age to constrain the models. Set up a new groundwater model that includes the interaction with the underlying bedrock (i.e. Gatton Sandstone) and that addresses the impact of salty bedrock water on water quality in times of drought.
- Provide land use maps and crop distribution analysis with high frequency to improve planning of crops in the Lockyer.
- Extend investigations on pharmaceuticals and wastewater tracers, estimate quality impact and risk from different management strategies.

From the above, we can conclude that there is an ongoing need for scientific advice to the Lockyer Valley stakeholders to optimise environmental benefits, economic returns and energy efficiency. As evidenced by the current development of the Western Corridor Recycled Water scheme, a very careful exploration of future water demand and public acceptance is required before the implementation of major wastewater reuse schemes.

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